Department of Built Environment

Modelling hydrology and sediment transport in a drained peatland forest

Focus on sediment load generation and control after ditch network maintenance

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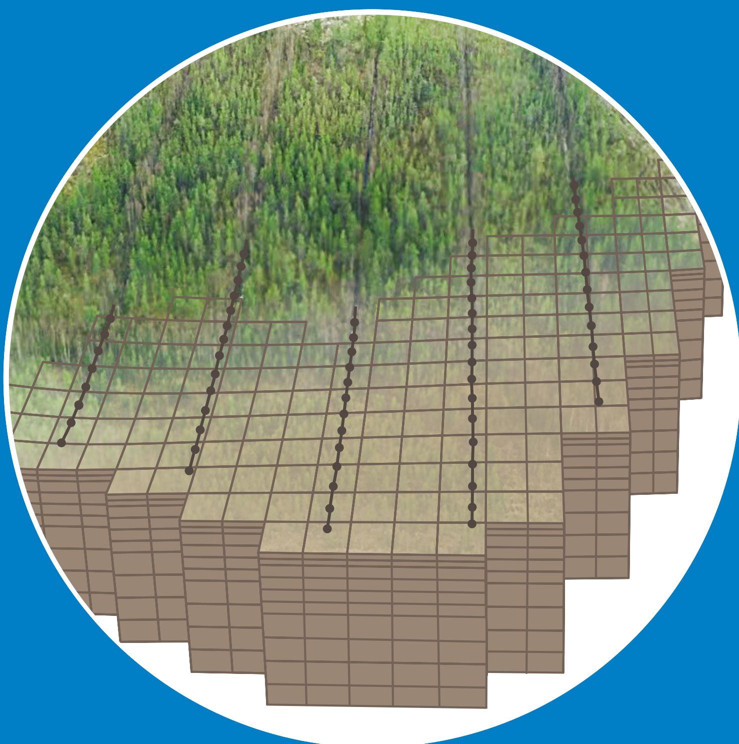
Kersti Haahti



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Modelling hydrology and sediment transport in a drained peatland forest

Focus on sediment load generation and control after ditch network maintenance

Kersti Haahti

A doctoral dissertation completed for the degree of Doctor of Science (Technology) to be defended, with the permission of the Aalto University School of Engineering, at a public examination held at the lecture hall F239a of the school on 6 April 2018 at 12.

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Abstract

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Abstract

Peatland drainage has been common practice in the boreal zone for centuries. In Fennoscandia, drainage has focused on improving forest production, and the maintenance of drainage conditions is an important component of operational forestry. Elevated sediment loads released after drainage have been widely recognized, but associated process understanding is still limited, especially when it comes to controlling the loads. In this thesis, a distributed process-based model is tailored and applied to describe hydrology and sediment transport in a drained peatland forest with aims to assess the generation and the control of sediment loads after ditch network maintenance (DNM). The chain of model applications revolves around a 5.2 ha forestry-drained peatland, where hydrology and sediment processes were intensively monitored after DNM.

The first model development step included coupled modeling of soil hydrology and ditch network flow. This enabled the spatiotemporal assessment of flow conditions in the ditch network, which is a necessity for sediment transport modeling. Distributed modeling of catchment hydrology further showed potential to disentangle the importance of spatial factors on local soil moisture conditions, which is relevant as drainage aims to control these conditions. Extending the model to describe sediment transport in the ditch network demonstrated its utility in identifying the role of different processes and in complementing experimental data. The model described bed erosion, rain-induced bank erosion, floc deposition, and consolidation of the bed. Simulation results suggested that the loose peat material, produced during excavation, contributed markedly to high concentrations immediately after DNM. Erosion during spring snowmelt was driven by ditch flow whereas during summer rainfalls, bank erosion by raindrop impact was an important process. Finally, the model was applied to investigate scenarios featuring sediment control structures implemented in operational peatland forestry. Results suggested that bed erosion can be efficiently prevented with breaks in ditch cleaning and structures ponding water. It was proven less efficient to trap already eroded material with sedimentation ponds and pits. Furthermore, coupled modeling of soil hydrology and ditch processes showed that structures raising ditch water level had a minor effect on water table in the strips between ditches, plausibly not impairing tree growth.

The process-based modeling presented in this thesis provided a yet unexplored approach to comprehensively evaluate spatiotemporal hydrological variables, sediment load generation and alternatives for sediment control after DNM in a forestry-drained peatland. Future prospects of such a modelling approach include extending the assessment to sites on shallow peat with a higher erosion risk, and the investigation of forestry management options, e.g., harvesting alternatives, on drained peatlands.

Keywords distributed process-based modeling, ditch cleaning, ditch erosion, forestry, hydrology, peatland drainage, sediment transport, water protection

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Tiivistelmä

Tekijä
Kersti Haahti

Väitöskirjan nimi
Hydrologian ja kiintoaineen kulkeutumisen mallintaminen ojitetussa suometsässä – Painopisteinä kiintoainekuorman muodostuminen ja hallinta kunnostusojituksen jälkeen

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Tiivistelmä
Boreaalisen vyöhykkeen soita on ojitettu laajasti viimeisten vuosisatojen aikana. Fennoskandiassa ojitus on keskittynyt suometsien kasvuolosuhteiden parantamiseen ja kuivatuksen ylläpito kunnostusojituksin on tärkeä osa suometsätaloutta. Ojitus tunnetaan haitallisena kiintoainekuorman lähteenä, mutta sen muodostumiseen ja etenkin hallintaan liittyvä prosessiymmärrys on puutteellista. Tässä väitöskirjassa kehitetään prosessipohjainen mallikokonaisuus kuvaamaan ojitetun suometsän hydrologiaa ja ojaverkoston kiintoaineprosesseja kunnostusojituksen jälkeisessä tilanteessa. Mallia sovelletaan väitöskirjan eri vaiheissa 5.2 hehtaarin suometsäalueelle, jossa kunnostusojituksen jälkeen seurattiin intensiivisesti alueen hydrologiaa ja kiintoaineprosesseja.

Mallikehityksen ensimmäisessä vaiheessa metsämaan hydrologian mallinnukseen yhdistettiin kuvaus virtauksesta ojaverkostossa. Tämä mahdollisti ojaverkoston virtausolosuhteiden ajallisen tarkastelun, mikä on olennaista kiintoaineen kulkeutumisen mallintamisen kannalta. Lisäksi metsämaan hydrologian hajautettu mallintaminen osoitti, että menetelmää voidaan hyödyntää paikalliseen kuivatustilaan vaikuttavien tekijöiden määrittämisessä. Mallin laajentaminen kiintoaineen kulkeutumisen kuvaamiseen puolestaan osoitti, että mallinnuksen avulla voidaan osoittaa eri prosessien rooli kiintoainekuorman muodostamisessa, sekä täydentää kokeellisesti hankittua tietoa. Kiintoainemalli kuvaa pohjaeroosion, sateen aiheuttaman seinämäeroosion, flokkien laskeutumisen ja laskeutuneiden hiukkasten stabiloitumisen ojan pohjalle. Tuloksista havaittiin, että kunnostuksen aikaansaama häiriintynyt kerros ojan pohjalla oli kuvattava mallissa, jotta heti toimenpiteen jälkeen esiintyvät korkeat pitoisuudet saatiin simuloitua. Kevätsulannan aikainen kuorma oli pääasiassa lähtöisin pohjaeroosiosta, kun taas kesäsateiden aikana seinämistä sadepisaroiden vaikutuksesta irtoavalla turpeella oli merkittävä vaikutus. Lopuksi mallia sovellettiin suometsätaloudessa käytettävien vesiensuojelumenetelmien tehon arvioimiseen. Tulokset antoivat viitteitä, että pohjaeroosio voidaan saada melko tehokkaasti kuriin perkauskatkoilla tai padottavilla rakenteilla. Sen sijaan jo liikkeelle lähteneen kiintoaineen pysäyttäminen laskeutusaltailla tai lietekuopilla osoittautui vähemmän tehokkaaksi. Mallilla pystyttiin myös tarkastelemaan ojavedenpintaa nostavien rakenteiden vaikutusta saran kuivatustilaan. Tulosten mukaan rakenteiden vaikutus oli vähäinen viitaten siihen, etteivät puuston kasvuolosuhteet heikentyneet.

Väitöskirjassa rakennettu mallikokonaisuus muodostaa uudenlaisen lähestymistavan suometsän spatiaalisen kuivatustilan ja kunnostusojituksen jälkeisen kiintoainekuorman muodostumisen ja hallinnan tarkasteluun. Mallin tarpeellisia jatkokehityskohteita olisivat mm. tarkastelun laajentaminen ohutturpeisille eroosioherkille alueille, sekä mallin hyödyntäminen vaihtoehtoisten hakkuumenetelmien arvioimiseen suometsätaloudessa.

Avainsanat hajautettu prosessipohjainen mallintaminen, hydrologia, kiintoaineen kulkeutuminen, kunnostusojitus, turvemaa, metsätalous, ojaeroosio, vesiensuojelu

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I wish to thank my advisors Prof. Bassam Younis, Dr. Lassi Warsta, and Dr. Teemu Kokkonen. The initial spark for my doctoral research can be traced back to the course on open channel hydraulics held by Prof. Younis on his visit in Aalto in 2013. I am thankful for Prof. Younis for subsequently accepting the role of instructing my master's thesis while hosting me at the University of California Davis, as well as for his contributions to the first two papers of my dissertation. Lassi and Teemu came into the picture as my doctoral research started and their support has been valuable. Special thanks to Lassi for his guidance in getting started with FLUSH, as well as his never ending optimism and cheerfulness. To Teemu I owe thanks for coming up with the difficult questions and thereby challenging my thinking, and of course for his perceptive commenting.

In addition to the strong supervisor/advisor team, I had the privilege to work with people experienced in peatland forestry and experimental work. I wish to thank my co-authors Prof. Leena Finér, Dr. Hannu Marttila, Dr. Mika Nieminen, Dr. Leena Stenberg and Mr. Antti Leinonen. Discussions and email exchange with co-authors had a vital role in the progression of my thesis research.

The success of any modelling effort largely depends on the underlying data. I was very fortunate to start my research work with such an extensive experimental data set available from the Koivupuro site. Especially monitoring sediment processes at Koivupuro has been a huge effort. I am most grateful for all the people from the University of Oulu, the Natural Resources Institute Finland, and Aalto University School of Engineering who took part in the instrumentation and monitoring of the site. I wish to acknowledge Metsähallitus for providing the site as well as the Academy of Finland (ModStream and ReFFECT projects) and Ministry of Agriculture and Forestry (MAHA project) for funding the fieldwork. Further thanks belong to Dr. Leena Stenberg and Mr. Tapio Tuukkanen who processed most of the data and patiently answered all my data-related questions.

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Finally, I want thank my (extended) family for the many addresses, Käpylä, Nuottaniemi, Myllykoski, Mankkaa, where you would be sure to be warmly welcome anytime. Thanks to Maija for the gym company, Joanna, Sophie and Dad for ambitious cooking and baking projects, Mum for after-work coffees – all providing much needed distractions from my thesis work. Warm and loving thanks belong to Lauri for his patience, support, and understanding through these years, as well as all the happy moments!

Espoo, 15 February 2018

Kersti Haahti

Contents

List of Abbreviations

1-D	One-dimensional
2-D	Two-dimensional
3-D	Three-dimensional
AFP	Air-filled porosity
Baseline	Scenario used as baseline for sediment control scenarios (see Figure 9)
Break	Sediment control scenario with a break in cleaning (see Figure 9)
Break-Pond-PRC	Sediment control scenario combining scenarios *Break* and *Pond-PRC* (see Figure 9)
Breaks	Sediment control scenario with 4 breaks in cleaning (see Figure 9)
D1–D4	Four main ditch lines in studied catchment (see Figure 4c)
D8	Single-direction topography based flow algorithm
DEM	Digital elevation model
DNM	Ditch network maintenance
DOC	Dissolved organic carbon
FMI	Finnish Meteorological Institute
LAI	Leaf area index
LOI	Loss on ignition
MSE	Mean squared error
NS	Nash–Sutcliffe model performance measure
NTU	Nephelometric turbidity unit
PEST	Model-independent parameter optimiser
Pit-Break	Sediment control scenario *Break* with a pit added before the structure
Pit-PRC	Sediment control scenario *PRC* with a pit added before the structure
Pits1	Sediment control scenario with 4 sedimentation pits (see Figure 9)
Pits2	Sediment control scenario with 15 sedimentation pits (see Figure 9)
Pits-Breaks	Sediment control scenario with 4 sedimentation pits followed by short breaks in cleaning (see Figure 9)

Pit-SWeir	Sediment control scenario *SWeir* with a pit added before the structure
Pond	Sediment control scenario with a sedimentation pond (see Figure 9)
Pond-PRC	Sediment control scenario with a sedimentation pond followed by a PRC structure (see Figure 9)
Pond-SWeir	Sediment control scenario with a sedimentation pond followed by a submerged weir (see Figure 9)
PRC	Peak runoff control
PRC	Sediment control scenario with a PRC structure (see Figure 9)
R1–R4	Model setups for sediment transport with different complexity levels in Paper **III**
$RMSE$	Root mean squared error
SS	Suspended sediment
SWE	Snow water equivalent
SWeir	Sediment control scenario with a submerged weir (see Figure 9)
V-notch	Scenario of prevailing conditions with the V-notch weir at catchment outlet

List of Symbols

A	(m^2)	cross-sectional flow area
B	(m)	water surface width
B_{bf}	(m)	bankfull channel width
B_{weir}	(m)	width of the weir surface
C	($m^3\,m^{-3}$)	sediment concentration (volumetric)
D	($m^3\,m^{-1}\,s^{-1}$)	rate of deposition (volumetric)
E	($m^3\,m^{-1}\,s^{-1}$)	rate of erosion (volumetric)
E_{bed}	($kg\,m^{-2}\,s^{-1}$)	erodibility coefficient of ditch bed
E_{rain}	($kg^{-1}\,m^{-2}\,s^2$)	rainfall erodibility coefficient
g	($m\,s^{-2}$)	gravitational acceleration
H_{cell}	(m)	hydraulic head in the cell
H_{ditch}	(m)	hydraulic head in the ditches within the cell
h	(m)	water depth
h_b	(m)	water depth at the junction
h_{over}	(m)	overland flow depth exceeding the depression storage
h_{weir}	(m)	height of weir structure from ditch bed
I	($mm\,h^{-1}$)	rainfall intensity
K_{sat}	($m\,s^{-1}$)	saturated horizontal hydraulic conductivity
k_ρ	(-)	parameter to adjust the dry bed density based on the consolidation time
k_τ	(-)	parameter relating dry bed density to critical shear stress for erosion
k_ω	(-)	parameter accounting for increasing settling velocity with increasing concentration
L	(m)	length of the break in cleaning
L_{ditch}	(m)	total length of the ditches within the cell
l	(m)	average flow path length (set to a quarter of the cell width)
M	($kg\,m^{-2}$)	bed mass per unit bed area
M_R	($kg^2\,s^{-3}$)	momentum squared for rain
n	($s\,m^{-1/3}$)	Manning's flow resistance
n_a, n_b	(-)	parameters accounting for increased flow resistance with decreasing flow rate

n_{bed}	(s m$^{-1/3}$)	Manning's roughness associated only with bed material
Q	(m^3 s^{-1})	discharge
q	(m^2 s^{-1})	lateral inflow per unit length
q_{over}	(m^3 s^{-1})	flux from an overland cell to the ditches within the cell
q_s	(m^3 m^{-1} s^{-1})	side discharge of sediment (volumetric)
q_{sub}	(m^3 s^{-1})	seepage flux from a subsurface cell to the ditches within the cell
R	(m)	hydraulic radius
R^2	(-)	coefficient of determination
S_0	(m m^{-1})	bottom slope
S_f	(m m^{-1})	friction slope
t	(s)	time
t_{con}	(h)	consolidation time
t_{stab}	(days)	time needed for deposited peat to stabilize
U	(m s^{-1})	flow velocity
w_i	(-)	weight assigned to observation type i in objective function Φ
x	(m)	longitudinal distance
z_{cell}	(m)	bottom elevation of the subsurface cell
α	(-)	factor to ensure numerical stability in Eq. (8)
γ, δ	(-)	empirical coefficients to compute momentum squared for rain from rainfall intensity
δ_{init}	(m)	initial thickness of loose bed layer in sediment transport model
Δt	(s)	time step
Δx	(m)	longitudinal distance step
Δz_{sat}	(m)	saturated thickness of the cell above ditch bottom
ρ	(kg m^{-3})	density of water
ρ_{dry}	(kg m^{-3})	dry bed density
ρ_{dry0}	(kg m^{-3})	dry bed density of freshly deposited sediments
$\rho_{dry\max}$	(kg m^{-3})	dry bed density of fully consolidated bed
ρ_s	(kg m^{-3})	density of the sediment particles
τ_b	(N m^{-2})	bed shear stress
τ_{bd}	(N m^{-2})	critical shear stress for deposition
τ_{ce}	(N m^{-2})	critical shear stress for erosion
τ_{ce0}	(N m^{-2})	critical shear stress for erosion of freshly deposited sediment
$\tau_{ce\max}$	(N m^{-2})	critical shear stress for erosion of fully consolidated bed
Φ	(-)	objective function in calibration with PEST
ω_s	(m s^{-1})	settling velocity
ω_{s0}	(m s^{-1})	settling velocity of dispersed sediment particles
$\omega_{s\max}$	(m s^{-1})	maximum floc settling velocity

List of Publications

This doctoral dissertation consists of a summary and of the following publications which are referred to in the text by their numerals

I. **Haahti, Kersti**, Younis, B.A., Stenberg, L. & Koivusalo, H. 2014. Unsteady flow simulation and erosion assessment in a ditch network of a drained peatland forest catchment in Eastern Finland. *Water Resources Managemen*t, 28(14): 5175–5197. ISSN: 1573-1650. DOI: 10.1007/s11269-014-0805-x.

II. **Haahti, Kersti**, Warsta, L., Kokkonen, T., Younis, B.A. & Koivusalo, H. 2016. Distributed hydrological modeling with channel network flow of a forestry drained peatland site. *Water Resources Research*, 52(1): 246–263. ISSN: 1944-7973. DOI: 10.1002/2015WR018038.

III. **Haahti, Kersti**, Marttila, H., Warsta, L., Kokkonen, T., Finér, L. & Koivusalo, H. 2016. Modeling sediment transport after ditch network maintenance of a forested peatland. *Water Resources Research*, 52(11): 9001–9019. ISSN: 1944-7973. DOI: 10.1002/2016WR019442.

IV. **Haahti, Kersti**, Nieminen, M., Finér, L., Marttila, H., Kokkonen, T., Leinonen, A. & Koivusalo, H. 2018. Model-based evaluation of sediment control in a drained peatland forest after ditch network maintenance. *Canadian Journal of Forest Research*, 48(2): 130–140. ISSN: 1208-6037. DOI: 10.1139/cjfr-2017-0269.

Author's Contribution

Paper I: The author was responsible for designing the research together with Prof. Younis and Prof. Koivusalo. The author was responsible for developing the model for unsteady flow in an open channel network with the help of Prof. Younis. The author was fully responsible for the ditch network flow routing simulations and mainly responsible for writing the article. Prof. Koivusalo assisted in the simulation of runoff from the forested land areas. All authors participated in writing the article.

Paper II: The author was responsible for designing the research together with Prof. Koivusalo, Dr. Kokkonen and Dr. Warsta. The author was responsible for integrating the ditch network flow routing description into the FLUSH model with the help of Dr. Warsta. The author was fully responsible for the model simulations and mainly responsible for writing the article. All authors participated in interpreting the simulation results and writing the article.

Paper III: The author was mainly responsible for designing the research as well as fully responsible for developing the model description for sediment transport and conducting the model simulations. Prof. Koivusalo and Dr. Kokkonen participated in designing the research. All authors participated in interpreting the simulation results and writing the article.

Paper IV: The author was mainly responsible for designing the research, writing the article, as well as fully responsible for the model simulations. Dr. Marttila and Mr. Leinonen assisted in selecting the dimensions for structures implemented in the sediment control scenarios. All authors participated in selecting the scenarios, interpreting the results, and writing the article.

1 Introduction

1.1 Background

1.1.1 Drainage of peatlands – Broad perspective

Peatland drainage dates back more than 2500 years, when agricultural practice was extended to peat soils south-east of Rome (Paavilainen and Päivänen, 1995). Since then agriculture, and later forestry, peat extraction (for fuel and horticulture), and flood alleviation have all benefited from peatland drainage (Holden et al., 2004). Systematic drainage on peatlands, however, started only a few centuries ago, and in many countries, e.g., Britain and Finland, the rate of drainage activities peaked in the 1960s–1970s (Holden et al., 2004; Paavilainen and Päivänen, 1995). In North America and Europe, drainage of pristine peatlands has practically ceased and the focus has switched to managing and restoring existing drained peatlands (Holden et al., 2004; Ramchunder et al., 2012). However, in the tropics, especially in Indonesia, drainage of intact peatland continues as large areas have recently been claimed for palm oil plantations (Comte et al., 2012).

Drainage causes fundamental changes in the hydrological behaviour of the peatland. Consequent to drainage water table lowers and restricts evaporation from the peat surface (Paavilainen and Päivänen, 1995). As a result, runoff typically increases and hydrographs become flashier due to enhanced connectivity (e.g., Holden et al., 2006; Lundin, 1994; Robinson, 1980; Seuna, 1981). The hydrological response to drainage, however, depends on the site characteristics, e.g., peat type, and therefore some studies have reported contradicting findings (see Holden et al., 2004). As time from drainage elapses, the land use practice affects the water balance further. For example, on forestry sites interception and transpiration by the growing tree stand increase evapotranspiration reducing the share of runoff in the water balance (e.g., Seuna, 1981).

In addition to endangering the mire habitat itself, peatland drainage has several other environmental impacts closely linked to the changes in the site hydrology. Enhanced connectivity between sediment sources (exposed ditch surfaces) and downstream watercourses have been recognized to markedly increase released suspended sediment (SS) loads (e.g., Holden et al., 2007; Prévost et al., 1999; Robinson and Blyth, 1982). This leads to a simultaneous increase in concentrations of sediment-bound nutrients, especially phosphorus (Päivänen and Hånell, 2012). Furthermore, increased air-filled porosity (AFP) in the surface peat layer affects microbial activity. The availability of oxygen enhances the mineralization of nutrients, which is vital for agricultural and forestry practice on peatlands, but may cause increased leaching (Nieminen et al., 2017b). Elevated concentrations of dissolved organic carbon (DOC) from drained peatlands are also widely acknowledged (e.g., Wallage et al., 2006). Such changes in water quality affect the ecological status of downstream watercourses causing siltation, water discolouration, eutrophication, and oxygen depletion (Paavilainen and Päivänen, 1995).

Water table drawdown further affects the soil carbon balance of peatlands as enhanced aerobic decomposition emits carbon dioxide (CO_2) to the atmosphere (e.g., Camporese et al., 2008). Especially in temperate and tropical regions peat oxidation can lead to significant land subsidence in areas claimed for agriculture (Gambolati et al., 2006; Wösten et al., 1997). Simultaneously, methane (CH_4) emissions from the peat surface decrease (von Arnold et al., 2005), except for the ditches which continue to emit CH_4 and may have a significant role in the site carbon balance (Minkkinen and Laine, 2006; Roulet and Moore, 1995). In natural state peatlands accumulate carbon but drainage may convert them to carbon sources depending on the land use and the climatic zone (e.g, Hirano et al., 2012). Forestry-drainage on boreal peatlands has, however, been estimated to result even in a net carbon gain, when the carbon stored in the tree stand is considered (Cannell et al., 1993; Minkkinen et al., 2002). Accounting for the full forest rotation period, the use of the harvested wood products, as well as the warming climate, on the other hand, has resulted in opposite conclusions (He et al., 2016). Peatland drainage also increases the risk of fire, e.g., in Southeast Asia, fires have had substantial impacts on global annual carbon emissions (Page et al., 2002).

1.1.2 Forestry on drained peatlands

In Finland, forestry is the dominant land use on peatlands covering one half of the 10 Mha of peatlands in the country (Finnish Forest Research Institute, 2014). Besides Finland, peatland drainage has been an important component of operational forestry in Sweden, Norway, Russia, the Baltic states, the United Kingdom, and some areas of the United States and Canada (Paavilainen and Päivänen, 1995). The objective of forestry-drainage is to increase the aeration of the root zone creating more favourable conditions for tree growth (Kozlowski, 1986). Design parameters for forestry drainage schemes, including ditch spacing and depth, vary largely between sites and countries (e.g., Amatya et al., 2000; Préfontaine and Jutras, 2017; Sarkkola et al., 2010). In Fennoscandia, ditches are typically spaced 35–50 m apart and dug 0.8–0.9 m deep (Päivänen and Hånell, 2012). Numerous studies have reported increased stand growth after drainage, commonly with more pronounced changes closer to the ditches (Heikurainen, 1980; Préfontaine and Jutras, 2017). The post-drainage stand volume increment depends on the site type and may be negligible on the nutrient poorest sites (Hånell, 1988).

As boreal pristine peatlands are no longer drained for forestry, the focus is on the management of existing forestry-drained peatlands. The traditional and still most common management option practiced on drained peatlands is even-aged management, where the stand rotation ends in harvesting by clear-cut (Nieminen et al., 2017c). In Fennoscandian conditions, such management often includes thinning, fertilization, and ditch network maintenance (DNM, Figure 1c) during the stand rotation (Päivänen and Hånell, 2012). Harmful water quality consequences of the operations are typically highest during the first years after the operations and are then assumed to return close to pre-drainage conditions within a decade (Finér et al., 2010). However, recent studies have reported elevated levels of nutrient concentrations also in old forestry-drained sites (Nieminen et al., 2017b) as well as increased SS concentrations even 20 years after ditching operations (Joensuu et al., 2012b).

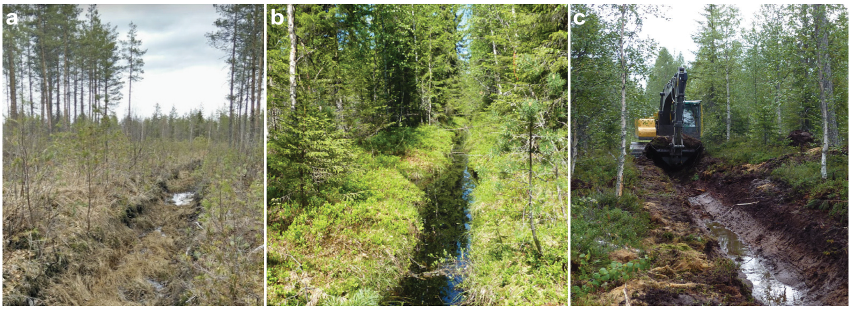

Figure 1. Forestry ditches in varying conditions (a–b) and ditch network maintenance (c).

Considering the prevalence and water quality effects of different peatland forestry operations, DNM is currently regarded as the most concerning forestry practice in Finland for receiving watercourses (Finér et al., 2010). Foremost, DNM increases erosion in the excavated ditches, which produces elevated SS loads (see Nieminen et al., 2017a). Especially in headwater catchments, where peatland forestry is dominant, SS loads by DNM can contribute markedly to the ecological status of receiving watercourses (Åström et al., 2001; Kauppila et al., 2016).

DNM is typically undertaken every 20–40 years (Hökkä et al., 2000) as the drainage capacity of the ditch network deteriorates (Figures 1a–b) due to, e.g., collapsing banks, vegetation, and peat subsidence (Silver and Joensuu, 2005). DNM has been reported to lower the growing season water table and increase tree growth (see Sikström and Hökkä, 2016). However, in highly stocked peatland stands, evapotranspiration may dominate the water balance during the growing season and DNM may be unnecessary (Lauhanen and Ahti, 2001; Sarkkola et al., 2012, 2013b). Lately attention has been given to avoid DNM on such sites (Sarkkola et al., 2013a) as well as recognizing sites where DNM would fail to improve tree growth due to the poor site characteristics (Ahtikoski et al., 2012). Nevertheless, the practice extent of DNM in Finland is expected to remain at the level of recent years (50 000–60 000 ha a^{-1}) as the shift to bio-based economy increases the demand for forest biomass (Nieminen et al., 2017a). Therefore, DNM remains an important research subject, especially regarding SS load generation.

1.1.3 Sediment load generation in drained peatlands

Ditch excavation produces large quantities of loose sediment in the ditch network, which is observed as a conspicuous increase in SS concentrations during the first post-treatment months (Hansen et al., 2013; Manninen, 1998; Stenberg et al., 2015b). A more constant source of SS arises from the erosion of the exposed surfaces of the treated ditch network (Robinson and Blyth, 1982). Marttila and Kløve

(2010a) underlined the erosive strength of high-intensity rainstorms, in addition to spring season snowmelt, which is traditionally viewed as the most critical period for SS load generation in high-latitude conditions. High-level and long-term SS export is generally related to sites where ditches extend to the fine-textured mineral subsoil beneath the peat layer (Joensuu et al., 2012b).

Besides studies concentrating on the quantification of released SS loads (e.g., Prévost et al., 1999; Robinson and Blyth, 1982), more recent studies have attempted to shed light on the sources and mechanisms of erosion within the drained sites. For example, studies assessing topographical changes of ditch cross-sections have recognized ditch banks as substantial sediment sources (Evans and Warburton, 2005; Stenberg et al., 2015a, 2015b). Furthermore, monitoring ditch cross-sections has revealed greater tendency towards widening of those ditches that extend to the mineral subsoil (Holden et al., 2007; Stenberg et al., 2015b; Tuukkanen et al., 2014), and more pronounced erosion in sections with a steep bed slope (Holden et al., 2007). Accumulation of deposited sediments on the ditch beds has also been reported, indicating that only a fraction of the eroded material is transported through to the network outlet (Stenberg et al., 2015b).

Erosion of an exposed peat surface is affected by numerous factors. For example, it has been established that an exposed peat surface requires a period of weathering (e.g., by frost or desiccation) to produce an easily mobilized surface layer (Evans and Warburton, 2005). Kløve (1998) emphasized the role of raindrop impact causing surface erosion on a peat extraction site. Rainfall intensity has also been recognized to regulate SS concentrations on the rising limb of hydrographs after DNM in a deep peat forestry (Tuukkanen et al., 2016). Laboratory flume experiments on deposited peat sediments have further revealed that fluff peat deposits erode readily and that they stabilize on a time scale of days (Marttila and Kløve, 2008). In contrast, an intact peat surface has been observed to be relatively resistant to erosion (Carling et al., 1997; Tuukkanen et al., 2014).

1.1.4 Controlling sediment loads from drained peatlands

Controlling sediment loads from peatlands drained for different purposes has been addressed, e.g., by peak runoff control (PRC) structures (Amatya et al., 2003; Marttila and Kløve, 2010b), ditch-blocking (Holden et al., 2007), wetland buffers (Clément et al., 2009; Nieminen et al., 2005), and sedimentation ponds (Es-Salhi et al., 2013; Joensuu et al., 1999). The principle of these structures relies on reducing flow velocities in order to decrease erosion or enhance deposition. Oftentimes when implementing such structures the aim is not only to control sediment loads but also to mitigate nutrient loads (e.g., O'Driscoll et al., 2014) or to regulate outflow (e.g., Amatya et al., 2000).

In countries such as Finland and Sweden, where drained peatland forestry covers large areas (Päivänen and Hånell, 2012), planning sediment control alongside DNM is encouraged by guidelines for forestry water protection (e.g., Ring et al., 2008; Vanhatalo et al., 2015). Recommended structures for controlling water quality include sedimentation ponds and pits, PRC, breaks in cleaning, submerged weirs, and wetland buffers (Figure 2). Guidelines with recommendations (Päivinen et al., 2011; Joensuu et al., 2012a; Vanhatalo et al., 2015) have led to the routine-like implementation of these structures in operational peatland forestry in Finland. However, scientific knowledge on their efficiency in controlling sediment loads is incomplete while only selected structures have been studied to date. These structures include sedimentation ponds (e.g., Joensuu et al., 1999), PRC (e.g., Marttila and Kløve, 2010b) and wetland buffers (e.g., Nieminen et al., 2005), which typically are applied in locations with a contributing area of up to 50 ha. In contrast, there is limited knowledge on the efficiency of structures, which are either scattered evenly within the ditch network (sedimentation pits) or implemented in ditch sections with severe erosion (breaks in cleaning and submerged weirs).

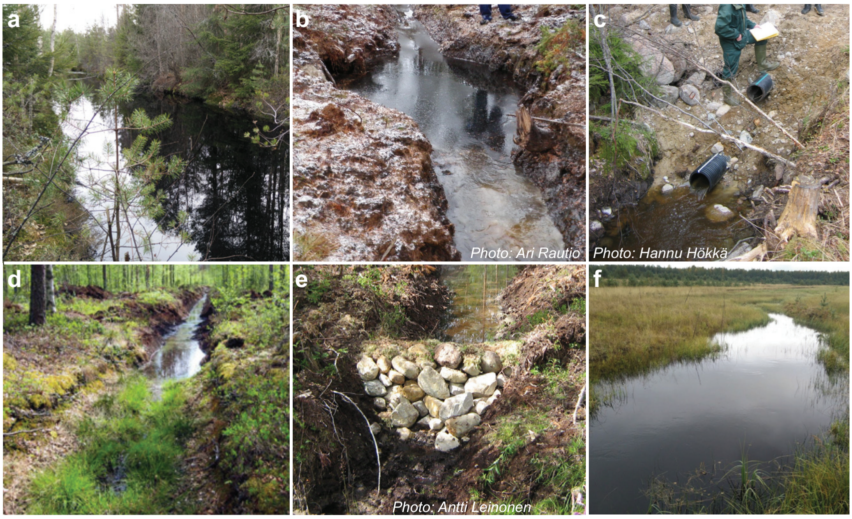

Figure 2. Structures applied in drained peatland forestry to control water quality: sedimentation pond (a), sedimentation pit (b), peak runoff control structure (c), break in cleaning (d), submerged weir (e), and wetland buffer (f).

Sedimentation ponds in forestry drained peatlands have been extensively studied by Joensuu et al. (1999), who reported an average SS concentration reduction of 18% during the first year after DNM based on observations from 37 catchments. The lowest or even negative reductions were associated with deep peat sites or sites where ditches extended to the fine-textured mineral subsoil (Joensuu et al., 1999). Poor efficiency of a pond on deep peat was also reported by Manninen (1998), who observed a reduction of 20% immediately after DNM, but no further retention later. Reductions by ponds in peat extraction sites have also been reported in a large range; 41–68% by Kløve (2000), -379–85% by Samson-Dô (2015), and -216–73% by Ihme et al. (1991).

PRC structures, typically equipped with a set of throughflow pipes (Figure 2c), are designed to retain runoff during peak flows. An orifice-weir studied in North Carolina by Amatya et al. (2003) led to annual SS load reductions of 0.3–89% over five years. In a similar setup in Finland, sediment exports were reduced by 81–90% (Marttila and Kløve, 2010b). Comparable high reductions have been reported from peat extraction sites (Kløve, 2000; Marttila and Kløve, 2009). As the efficiency of PRC relies on temporary storage of water within the ditch network, impaired drainage conditions due to PRC are a typical concern in forestry sites. Hökkä et al. (2011a), however, reported no significant water table level rise in the forestry strip next to a PRC structure. In contrast, Amatya et al. (2003) observed a water table level increase of 0.07 m due to the implemented orifice-weir.

Among structures aiming to retain SS, wetland buffers have been acknowledged as highly efficient reaching SS load reductions up to 80–100% when properly designed (Sallantaus et al., 1998; Nieminen et al., 2005). However, the implementation of wetland buffers is far less common than, e.g., sedimentation ponds as they require a land area of at least 0.5–1.0% of the upstream catchment area (Nieminen et al., 2005). Furthermore, the upstream catchment area should have a sufficient slope in order to enable high water table only in the buffer itself, without raising water table and disturbing tree growth in upstream areas.

1.1.5 Assessment of hydrology and water quality in drained forests

The impacts of forestry drainage and related practices on peatland hydrology and water quality have most commonly been studied with a paired catchment approach (e.g., Nieminen et al., 2010). Such a setup includes a control and a treatment catchment, which are first monitored for a pre-treatment period used for calibration and then post-treatment for impact assessment. Typically, in these studies discharge and water quality parameters are monitored at the catchment outlet, resulting in information about changes in runoff (e.g., Amatya et al., 2000; Hansen et al., 2013) and load magnitudes (e.g., Marttila and Kløve, 2010b; Nieminen et al., 2010). Additionally studies have addressed changes in water table depth monitored in selected locations (e.g., Amatya et al., 2003). The assessed treatments include, e.g., initial drainage (e.g., Prévost et al., 1999), DNM (e.g., Nieminen et al., 2010), and PRC (e.g., Marttila and Kløve, 2010b). Sedimentation ponds and wetland buffers, on the other hand, have been studied by comparing inlet and outlet concentrations (e.g., Joensuu et al., 1999; Nieminen et al., 2005).

Although paired catchment studies have had a significant contribution to peatland drainage research, such an approach has its limitations. For example, Koivusalo et al. (2008) identified it can be problematic to isolate parallel treatment and control catchments from each other. Laurén et al. (2009) underlined the risk for over-interpretation of results when the uncertainty in the pre-treatment dataset is neglected. Uncertainties arguably are even greater, when investigating the effects of a sediment control structure implemented alongside DNM. In this case, both treatment and control catchments are operated with DNM (e.g., Marttila and Kløve, 2010b) assuming that they react similarly to DNM and that the only difference is caused by the structure applied as a treatment. Because of the stochastic nature of erosion and the role of small differences in site characteristics, such an assumption may prove unrealistic. This, together with the expense of setting up numerous parallel sites, may explain why knowledge of the efficiency of sediment control structures and their combinations after DNM is limited (see Section 1.1.4).

Statistical models comprise another widely applied method to assess water quality and water table depths of drained peatland forests (e.g., Nieminen et al., 2017b; Sarkkola et al., 2010). In case of water quality, such studies aim to determine factors controlling differences in load magnitudes and identify risk sites (e.g., Joensuu et al., 1999; Tuukkanen et al., 2014). Analysis of water table depths has been motivated by the need to predict tree growth conditions in drained sites (e.g., Hökkä et al., 2008; Sarkkola et al., 2010). Regression models presented by these studies propose parameters, such as ditch spacing and depth, stand volume, precipitation, and northern latitude, to predict mean late summer water table depth. Other studies have assessed the variability of water table depth within a drained site and have recognized its high variability due to, e.g., topography and the vicinity of the ditch (e.g., Haahti et al., 2012; Holden et al., 2006).

Hydrological models have also been applied to study drained forest sites (e.g., Amatya et al., 1997; Nieminen et al., 2017a). Most commonly the applied models describe water balance either in a soil column midway between two parallel ditches (e.g., He et al., 2016; McCarthy et al., 1992) or in a 2-D soil profile representing the vertical cross-section between two parallel ditches (e.g., Koivusalo et al., 2008). Such models have provided understanding on the hydrological impacts of forestry practices including, e.g., harvesting, thinning and DNM (e.g., Koivusalo et al., 2008; Tian et al., 2012). However, because of their structure, they cannot address the spatial variability of water table depth due to, e.g., topography. Distributed hydrological modeling, on the other hand, provides this possibility. A few studies have explored this approach to assess effects of drainage and afforestation on water table drawdown and water balance components (e.g., Dunn and Mackay, 1996; Lewis et al., 2013). The spatial extent and focus of these studies however ruled out the investigation of water table depth on a between-ditch scale.

Sediment transport processes within ditch networks have been assessed based on visual observations (e.g., Marttila et al., 2010; Tuukkanen et al., 2012) and measured topographical changes (e.g., Stenberg et al., 2015b). Furthermore, Lappalainen et al. (2010) reported an application of a flow and sediment transport model in a single collector ditch of a forestry-drained site. Erosion in the model was, however, restricted to the ditch bed, which contradicts findings from recent experimental studies (see Section 1.1.3). Furthermore, the model was developed for drained thin-peated areas with a non-cohesive mineral subsoil. Peat, on the other hand, exhibits cohesive behaviour (Marttila and Kløve, 2008) requiring special considerations in the model descriptions (Lin and Wu, 2013).

1.1.6 Distributed process-based hydrological models

Distributed process-based models simulating overland and subsurface flow have become widely applied and their number is growing (Paniconi and Putti, 2015). Essentially, such models enable the description of catchment hydrological processes, which are driven by meteorological variables, spatial site characteristics (e.g., drainage configuration, topography, vegetation), and locally implemented structures (e.g., weirs). These factors are subject to changes due to, e.g., forestry operations and global warming, and therefore tools supporting their impact assessment are useful. In many applications, the interest is not only in modelling hydrological processes, but also in sediment or solute transport (e.g., Warsta et al., 2013b; Weill et al., 2011), which foremost are driven by hydrological processes.

Existing integrated hydrological models include, e.g., InHM (VanderKwaak and Loague, 2001), MODHMS (Panday and Huyakorn, 2004), MIKE SHE (Graham and Butts, 2005), Parflow (Kollet and Maxwell, 2006), PIHM (Qu and Duffy, 2007), GSFLOW (Markstrom et al., 2008), HydroGeoSphere (Therrien et al., 2010), CATHY (Camporese et al., 2010), tRIBS-OFM (Kim et al., 2012), and FLUSH (Warsta et al., 2013a). They apply adapted forms of Richards equation for 3-D subsurface flow, and typically 2-D descriptions of the St. Venant equations for overland flow. Furthermore, models may include special features, such as descriptions for macropore flow and soil cracking (e.g., Warsta et al., 2013a), or soil shrinkage and swelling (e.g., Camporese et al., 2010).

Channel flow is commonly routed either embedded in the 2-D overland flow domain (e.g., VanderKwaak and Loague, 2001) or using 1-D features (e.g., Graham and Butts, 2005). To accurately represent the geometry of drainage ditches, the first approach requires detailed topographical information and significant refining of the grid resolution. Then again, the second is more flexible as the ditches can be characterised with separately defined cross-sections allowing also the representation of structures within them (e.g., Panday and Huyakorn, 2004). Channel flow is solved using the full St. Venant equations (e.g., Graham and Butts, 2005) or their approximations such as the kinematic wave (e.g., Kollet and Maxwell, 2006) or the diffusive wave (e.g., Qu and Duffy, 2007). The limitations of the kinematic wave approximation include the inability to represent water pooling and backwater effects within the channel. Furthermore, some of the implemented flow routing algorithms run in an upstream-downstream sequence, which restricts the model to dendritic drainage networks (e.g., Markstrom et al., 2008).

In addition to overland and subsurface flow, there are other hydrological processes to be considered in such models. For example, year-round simulations in high-latitude conditions require description of snow accumulation and melt. Two main approaches applied for this purpose comprise the degree-day approach (e.g., Graham and Butts, 2005) and a more physically-based snow energy balance scheme (e.g., Markstrom et al., 2008). In relation to this, models may include also descriptions for soil heat flow and soil freezing (e.g., Turunen et al., 2015).

1.2 Research gap

Sections 1.1.1–1.1.5 point out that peatland drainage research, especially on Finnish forestry sites, has a strong foundation. A common feature for most studies is, however, that they rely heavily on experimental work (see Section 1.1.5). Despite the fact that distributed process-based hydrological models have gained a lot of attention (see Section 1.1.6), their implementation on peatland forests has been limited and no earlier study has reported applications where flow in the ditch network is considered in the simulations. The need for such modelling, including also description for sediment transport in ditches, is evident for several reasons.

Firstly, water table depth, which has a key role in controlling tree growth conditions as well as soil carbon balance (see Section 1.1.1), is highly variable in space and time. A limited number of point measurements or hydrological simulations restricted to conditions between two parallel ditches (see Section 1.1.5) cannot address the full extent of water table depth variation. In addition to spatial site characteristics, water table depth can be affected by structures implemented in ditches (e.g., Amatya et al., 2000; Hökkä et al., 2011a). Such impacts can only be predicted with distributed models including the routing of ditch network flow.

Secondly, sediment transport after ditch excavation has been studied with a wide range of experimental approaches (see Sections 1.1.3 and 1.1.5) but often the underlying mechanisms can only be superficially addressed as they easily become masked by each other. Modelling of erosion and sediment transport provides tools to disentangle the role of various processes, to assess their temporal and spatial scales, as well as to complement experimentally gained knowledge. Fundamental understanding of the mechanisms of sediment load generation is topical regarding, e.g., the continuous need for DNM in Finland (see Section 1.1.2).

Finally, sediment control structures are routinely implemented alongside DNM in operational peatland forestry, but scientific knowledge on their efficiency in controlling sediment loads is limited (see Section 1.1.4). This calls for a comprehensive evaluation of sediment control alternatives, including scenarios with, e.g., sedimentation pits and breaks in cleaning scattered within the ditch network. The feasibility to conduct such an evaluation experimentally is questionable due to challenges described in Section 1.1.5. Process-based modelling, on the other hand, enables the explicit representation of structures within the ditches and thereby provides a novel and transparent method to approach this problem.

Although a vast number of distributed process-based hydrological models are available (see Section 1.1.6), they typically have a certain focus that reflects the purpose for which they were developed. From this point of view, drained peatland forests are rather unique, especially immediately after DNM with exposed soil surface in the ditches. It is doubtful that existing models as such would meet the requirements to tackle the above outlined modelling needs in drained peatland forests. In this regard, openly available hydrological model source codes (e.g., Warsta et al., 2013a) provide a convenient platform as they can be extended and adjusted along the way.

1.3 Objectives and scope of the thesis

Based on the reasoning in Section 1.2, the main objective of this thesis is to tailor and apply a distributed process-based model to describe hydrology and sediment transport in a drained peatland forest. The four papers forming this thesis build upon each other aiming ultimately to assess sediment load generation and control after DNM (Figure 3). Papers **I**–**II** focus on developing a modelling framework for describing hydrological processes and calibrating–validating the resulting model against field data. A separate forest hydrological model FEMMA (Koivusalo et al., 2008) was applied to provide input to the ditch network flow model in Paper **I**, and to the 3-D soil hydrological model FLUSH (Warsta et al., 2013a) extended to describe ditch flow in Paper **II**. Paper **III** concentrates on describing sediment transport in the ditch network after DNM, and finally, Paper **IV** applies the model to investigate sediment control scenarios. The stepwise objectives of this thesis are to:

(i) model flow in a ditch network of a forested catchment (Paper **I**)
(ii) incorporate the description of ditch network flow in a process-based distributed hydrological model (Paper **II**)
(iii) assess simulated soil moisture patterns in a drained peatland (Paper **II**)

(iv) develop a sediment transport model suitable for describing conditions after DNM in a drained forest on deep peat (Paper **III**)

(v) identify the role of different processes in generating SS load during the first two years after DNM (Paper **III**)

(vi) assess sediment control practices carried out in operational peatland forestry based on model scenarios (Paper **IV**)

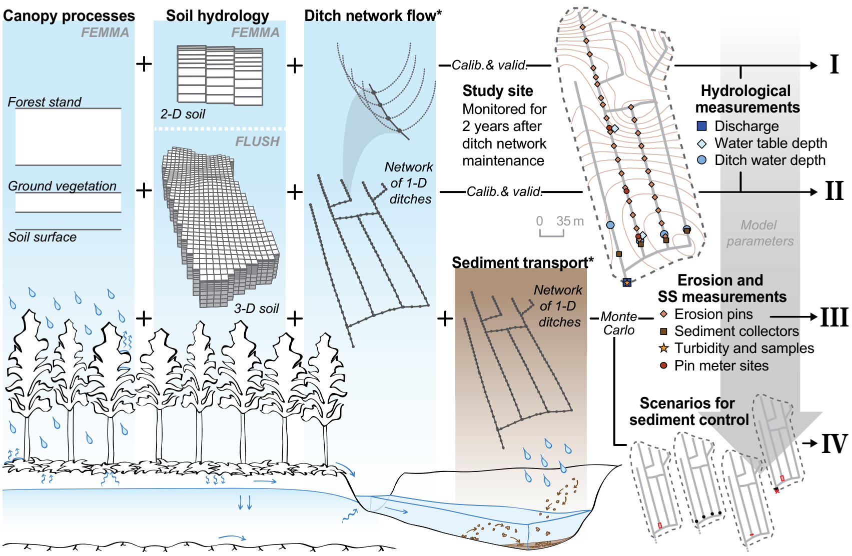

Figure 3. Graphical outline of the thesis focusing on modelling hydrology and sediment transport in a forestry-drained peatland. Each paper builds on a modelling framework, which consists of existing models (FEMMA, FLUSH) and submodels developed during this thesis (*). The modelling framework is evaluated against field data in Papers **I**–**III**, and applied to investigate sediment control scenarios in Paper **IV**.

The chain of modelling applications conducted in Papers **I**–**IV** (Figure 3) revolve around a 5.2 ha forestry-drained peatland site, where hydrological and sediment processes were intensively monitored for about two years after DNM. The experimental setup and data collection at the study site (see Section 2.1) were carried out before the research forming this thesis (see Stenberg, 2016). Therefore, the experimental design is out of the scope of this thesis. Most of the ditches of the study site were dug into deep peat, which limited the assessment to peat erosion and transport. Based on Section 1.1.3 such a site has not the highest erosion risk, but it was selected due to the unique dataset collected at the site after DNM.

The spatially distributed description of the modeling domain can be conducted with regard to various characteristics, e.g., by defining areas with differing soil profiles or by accounting for the distribution of the forest cover. As the focus of this thesis was foremost on sediment transport within the ditch network, the spatial aspects of the modeling domain were limited to the topography and the drainage network. Furthermore, modeling sediment transport was restricted to the ditch network, as the vegetation cover typically remains intact outside the ditches during DNM. Finally in Paper **IV**, the investigation of wetland buffers was left out as they require specific circumstances (see Section 1.1.4) and are still rarely implemented in practice.

2 Materials and Methods

2.1 Study site

2.1.1 Description

The site subject to the modelling applications in Papers **I**–**IV** (Figure 4c) is located in Eastern Finland (63°53'N, 28°40'E; Figure 4a). The site is characterized by a long-term (1981–2010) mean annual air temperature of 2.3°C and precipitation of 591 mm. In this region, snow typically covers the ground from late October to the end of April.

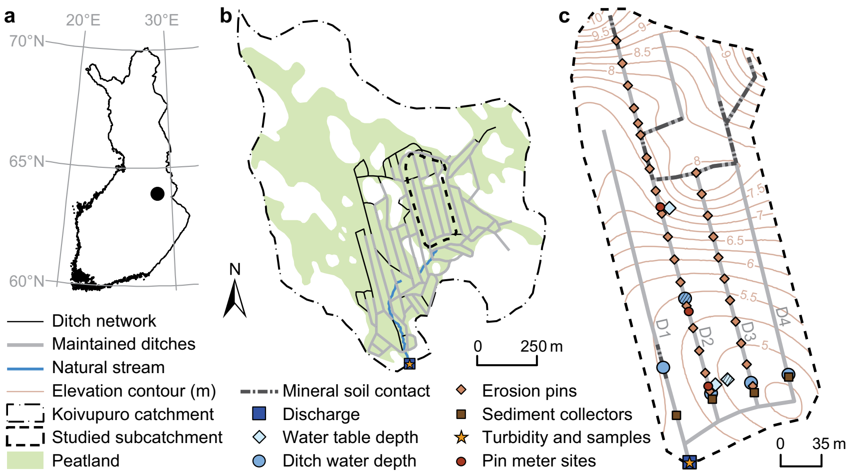

Figure 4. Location of the study site in Finland (a), the Koivupuro catchment (b), and the studied subcatchment (c). Loggers functioning only during the summer of 2012 are striped.

The 5.2 ha study site lies within a larger Koivupuro catchment (113 ha; Figure 4b), which was initially open-ditch-drained (32 ha) for forestry purposes in 1983. Monitoring of the Koivupuro catchment dates back to 1978 when the outlet stream was instrumented with a V-notch weir (Ahtiainen et al., 1988). The studied subcatchment (Figure 4c) was set up for intensive monitoring after DNM on 16–17 August 2011 by blocking the ditches at the edges of the subcatchment except for the outlet where a V-notch weir was installed (Stenberg et al., 2015b). The length of the cleaned ditches within the subcatchment totalled 1.5 km, of which about 20% extended through the peat layers into the mineral subsoil (Figure 4c). The ditches were dug to the depth of 1 m and had a width of approximately 2 m.

The peat soil in the subcatchment was woody *Sphagnum* peat and woody *Sphagnum-Carex* peat. The decomposition of peat varied from 3 in the surface peat layers to 8 in the deeper peat layers on the von Post scale (von Post, 1922). The mineral subsoil exposed in part of the ditches varied from silt to till. The stand (89 $m^3 ha^{-1}$ in 2012) was dominated by Scots pine (*Pinus sylvestris* L.) with a mixture of Norway spruce (*Picea abies* L.) and Birch (*Betula pendula* Roth).

2.1.2 Measurements

The data used in Papers **I**–**IV** as model input, for model setup and parametrization, and to calibrate and evaluate the model, covered a wide range of measurements from weather stations and monitored catchments (Table 1).

Table 1. Summary of measurements and their purpose in the modelling applications*

Measurement location	Model input	Model setup and parameterization	Model calibration and evaluation
Iso Kauhea	air temperature, precipitation, relative humidity, wind speed, global radiation	-	-
FMI stations (Sotkamo and Valtimo)	air temperature, precipitation, relative humidity, wind speed	-	snow depth
Koivupuro catchment	-	-	SWE, discharge, *turbidity and water samples*
Studied subcatchment	-	digital elevation model, longitudinal ditch bottom profiles, stand characteristics, *dry bulk density*, *critical shear stress*	discharge, water table depth, ditch water depth, *turbidity and water samples*, *sediment collectors*, *pin meter data*, *erosion pins*

*measurements in italic were used in the sediment transport model application (Papers **III–IV**)

Meteorological data, including air temperature, precipitation, relative humidity, wind speed, and global radiation, were available at a 20-min time interval from the weather station Iso-Kauhea (at 3 km distance) operated by the Natural Resources Institute Finland. Supplementary hourly meteorological data were available from two weather stations of the Finnish Meteorological Institute (FMI), Valtimo (26 km) and Sotkamo (30 km), to complete the data requirements. As longwave radiation was not recorded at any of the stations, it was estimated based on the Stefan-Boltzmann law, where emissivity was calculated based on air temperature, relative humidity and global radiation (see Turunen et al., 2015). Furthermore, hourly snow depth was recorded at the FMI stations. Manual snow water equivalent (SWE) measurements were available from the Koivupuro catchment covering three measurement points below the forest canopy with 4–5 observations each winter.

In September 2011, the longitudinal bottom profiles of the subcatchmen ditches were determined by levelling the ditch bottom elevation at 10-m intervals. Furthermore, a 10 × 10 m^2 digital elevation model (DEM) provided by the National Land Survey of Finland was available for the subcatchment.

The subcatchment stand characteristics (number of stems, diameter at breast height, and stand height) were determined using systematically selected experimental plots in 2013. Each characteristic was determined for Scots pine, Norway spruce, and Birch. The leaf area index (LAI) of the stand and canopy closure were estimated based on the aforementioned variables (see Haahti, 2014).

Discharge was measured at a 15-min interval using V-notch weirs and pressure sensors at the outlets of the Koivupuro catchment and the subcatchment (Figures 4b–c, 5a). Manual water depth measurements were taken regularly at both weirs to be compared against the pressure sensor recordings, which were then corrected if necessary. Discharge was calculated using the stage-discharge relationships for the weirs. Over the subcatchment weir, flow followed the relationship:

$$Q = 1.381(h - h_{weir})^{2.5} \qquad (1)$$

where Q is discharge ($m^3\ s^{-1}$), h is the upstream water depth (m), and h_{weir} is the distance between the ditch bed and the bottom of the V-notch (0.27 m).

The discharge measurements at the subcatchment covered most of the frost-free periods during August 2011–November 2013. The discharge recordings from the Koivupuro catchment were used as supplementary data. However, neither discharge time series covered the snowmelt period in 2013. Therefore the continuous daily runoff time series compiled by Stenberg et al. (2015b) for the Koivupuro catchment using data from the nearby Välipuro catchment to estimate missing values, was applied to evaluate the simulated year-round runoff in Paper **III**.

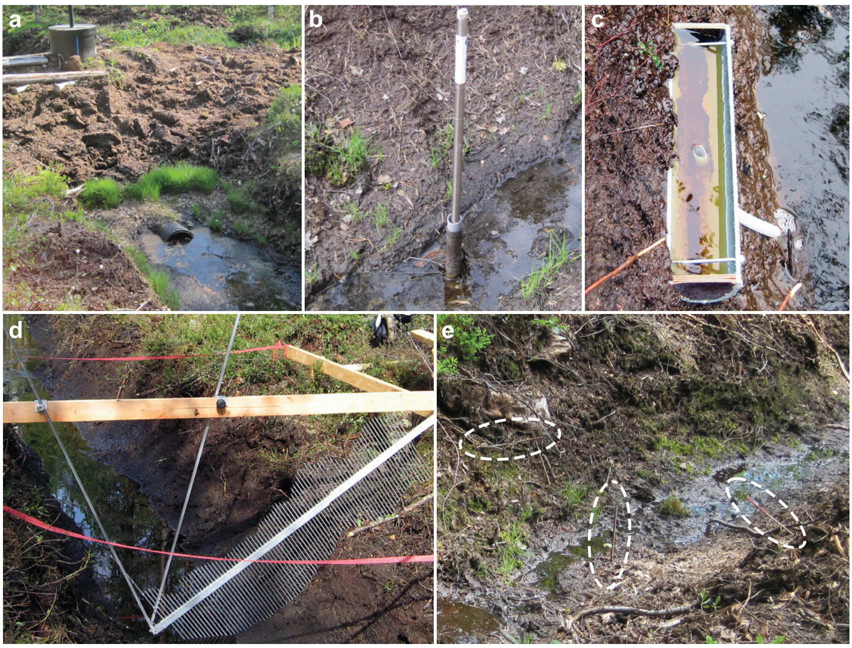

Figure 5. V-notch weir installation at the outlet of the subcatchment (a), logger for ditch water level (b), sediment collector installed in ditch bank (c), pin meter measurement setup (d), and erosion pin arrangement in ditch bed and banks (e).

In the subcatchment, water levels in the ditches were measured at 5 locations, and water table depth at 3 locations using automatic loggers (Figure 4c). The loggers were placed in perforated pipes installed in the bottom of the ditch (Figure 5b) or in the ground in August 2011. Measurements were taken every 15 min during the frost-free periods. Water level readings were converted to depths based on manual measurements of ditch water and water table depth in 2011 and 2013.

Turbidity sensors were installed in parallel with the pressure sensors at the catchment outlets (Figures 4b–c) to estimate SS concentrations in runoff waters (Stenberg et al., 2015b). For the calibration of the turbidity sensors, water samples to be analyzed for SS concentrations were collected at the outlets with automatic water samplers. Calibration curves between the SS concentrations of the water samples and the associated turbidity values were adopted from Stenberg et al. (2015b). Again, the data from the Koivupuro catchment were used as reference to assess the reliability of the data from the subcatchment. Loss on ignition (LOI) measured from the SS samples from the subcatchment indicated an organic matter content of 87.1±9.4% (Tuukkanen et al., 2016) implying that the upstream ditches with mineral soil contact (see Figure 4c) had a minor role.

Sediment collectors were installed in ditch banks in 4 locations (Figures 4c, 5c) to quantify bank erosion after DNM in the subcatchment (Stenberg et al., 2015b). The sediment collectors were emptied 8 times during October 2011–August 2012 providing an estimate of average bank erosion for the first year after DNM.

Topographic changes in the ditches after DNM were monitored using a pin meter (Stenberg et al., 2016, 2015b) and erosion pins (Tuukkanen et al., 2016). The pin meter measurements (Figure 5d) were conducted along three 4-m-long ditch sections (Figure 4c), while erosion pins were installed at about 20-m intervals in the beds and banks (Figure 5e) along ditches D2 and D3 (Figure 4c). Changes in peat surface level were measured with both methods in August 2011, October 2011, May 2012, September 2012, and June 2013 (only erosion pins).

Critical shear stress for bed erosion was determined along ditches D2 and D3 (Figure 4c) using a cohesive strength meter both *in situ* and in the laboratory (Khalid, 2014). Laboratory measurements were conducted from 21 undisturbed samples collected from the ditch beds. Khalid (2014) reports a difference of one order of magnitude between the median critical shear stress values measured *in situ* and in the laboratory, which were assumed to indicate the erodibility of the loose peat surface layer and the more stabilized ditch bed, respectively.

Finally, close to the pin meter sites (Figure 4c) the peat on the ditch banks was sampled at 0.1-m intervals from the soil surface to the water surface to determine the dry bulk density. The median dry bulk density measured close to the water surface ranged from 33.5 to 59.2 kg m^{-3} with an average of 47.8 kg m^{-3}.

2.2 Model descriptions

2.2.1 FEMMA model

FEMMA is a hydrological model designed to simulate impacts of forestry practices in boreal conditions (Koivusalo et al., 2008, 2006). The model employs hourly meteorological data above the forest canopy and computes hourly water balance components for a 2-D soil profile covered with two vegetation layers. The model simulates interception and transpiration, overland flow, subsurface flow and soil heat balance, as well as snow accumulation, melt, and ground freezing.

Mainly (in Papers **II**–**IV**), FEMMA was used for its canopy model to produce input time series for FLUSH (Figures 3, 6), which currently has no descriptions for vegetation layers. These inputs included potential transpiration, precipitation reaching the soil surface (or snowpack), as well as radiation and wind speed beneath the forest canopy. In Paper **I**, FEMMA was used to produce runoff into the ditch, which was then distributed to enter the ditch network based on the catchment topography according to the single-direction flow algorithm D8 (Jenson and Domingue, 1988; Figure 6).

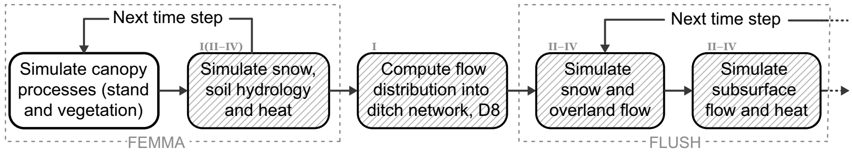

Figure 6. Flow chart to simulate hydrological processes from precipitation to runoff entering the ditch network (see Figure 7 for following steps). Grey striped steps are only included in the indicated papers while other steps are included in all Papers **I**–**IV**. Because of feedbacks in FEMMA between the soil and the canopy models, the FEMMA soil model was also run in Papers **II**–**IV**, although its outputs were not used (denoted by the numerals of these papers in parentheses).

The canopy model in FEMMA (Koivusalo et al., 2006) simulates interception of rain or snowfall by the forest stand, unloading of intercepted snow, and rainfall interception by ground vegetation. The remaining fraction of precipitation falls to the soil surface (or snowpack). Potential evapotranspiration is computed separately for the two vegetation layers based on the Penman–Monteith type combination equation (e.g., Wigmosta et al., 1994). Intercepted water evaporates at the potential rate until it is depleted. The remaining energy is available for transpiration, which is restricted by the stomatal resistance (see Nijssen et al., 1997). To produce the potential transpiration (FLUSH input), the impacts of meteorological variables and soil temperature on stomatal resistance were accounted for, but the soil moisture control was inactivated as it was subsequently considered in FLUSH (see Section 2.2.2).

Soil water movement and runoff generation is modelled in FEMMA along a vertical soil profile (with soil columns and layers, see Figure 3) extending from a water divide to a ditch. Rainfall to the soil surface or melt water from the snowpack either infiltrates or is transported downslope as 1-D overland flow (kinematic wave). Subsurface flow is solved with a quasi-2-D approach, i.e. vertical flow in the unsaturated zone based on Richards equation and horizontal flow in the saturated zone according to Darcy's law. Flow into the ditch occurs only when the water level in the adjacent soil column rises above the ditch water level.

The snow model in FEMMA is based on the energy balance scheme, where the snowpack is divided into two layers (Koivusalo et al., 2006, 2001). The winter process descriptions include snow accumulation and melt, heat conduction through the snow into the soil, liquid water retention in the snowpack, meltwater discharge, compaction of snow, and evaporation from the snowpack. Soil heat flow is solved following the method by Karvonen (1988), accounting for 1-D vertical heat conduction in the soil columns. Heat flow in the soil is affected by freezing and thawing processes, calculated based on a freezing curve (e.g., Koivusalo et al., 2001), but these processes do not affect the water flow in the soil. Hereby water is not stored as ice in the soil and the effect of decreased pore space or temperature on hydraulic conductivity is neglected in the simulations.

2.2.2 FLUSH model

The simulation of soil hydrology was upgraded in Paper **II** to describe spatially distributed hydrological processes in 3-D (Figure 3). The open-source hydrological model, FLUSH (Warsta et al., 2013a), was chosen for this purpose as the availability of the source code allowed the model to be extended with descriptions for flow and sediment transport in the ditch network (see Section 2.2.3). In addition to overland and subsurface flow, FLUSH describes subsurface heat flow, as well as snow and frost processes, which supports its use in high-latitude conditions (Turunen et al., 2015; Warsta et al., 2013a). As FLUSH was originally developed for clayey subsurface drained agricultural fields, it also has special descriptions for macropore flow, and soil shrinkage and swelling (Warsta et al., 2013a), which however were inactivated in the applications on forested peatland of this thesis.

In FLUSH, precipitation is initially stored on the soil surface where it can infiltrate into the soil. Excess water ponded on the soil surface is available for overland flow, which is generated when the depression storage is exceeded. Overland flow is routed in 2-D following the diffusive wave approximation of the St. Venant equations. Ditches function as sinks in the overland domain and the computation of the flux into them was slightly modified in Paper **II** to follow the equation for flow over a broad-crested weir (e.g., Panday and Huyakorn, 2004):

$$q_{over} = 2L_{ditch} \cdot 0.36\sqrt{2g}{h_{over}}^{3/2} \tag{2}$$

where q_{over} is the flux from an overland cell to the ditches within the cell ($m^3\,s^{-1}$), L_{ditch} is the length of the ditches within the cell (m), g is gravitational acceleration ($m\,s^{-2}$), and h_{over} is the overland flow depth exceeding the depression storage (m).

In the subsurface domain, 3-D water flow is computed in the unsaturated and saturated zones based on Richards equation, and using the Mualem–van Genuchten schemes to describe water retention and unsaturated hydraulic conductivities (van Genuchten, 1980). Water in the root zone can be removed by transpiration (sink-term), which is determined based on input time series of potential transpiration (see Section 2.2.1), the vertical root mass distribution (linear decrease from the soil surface to the maximum root depth), and the pressure head in the soil (Feddes et al., 1978). Ditches also function as sinks in the subsurface domain, removing water from the soil profile based on Darcy's law (Paper **II**):

$$q_{sub} = 2K_{Hsat}L_{ditch}\Delta z_{sat}\frac{H_{cell} - \max(H_{ditch}, z_{cell})}{l} \tag{3}$$

where q_{sub} is the seepage flux from a subsurface cell to the ditches within the cell ($m^3\,s^{-1}$), K_{sat} is the saturated horizontal hydraulic conductivity ($m\,s^{-1}$), Δz_{sat} is the saturated thickness of the cell above the ditch bottom (m), H_{cell} and H_{ditch} are the hydraulic heads in the cell and in the ditches (m), respectively, z_{cell} is the bottom elevation of the cell (m), and l is the average flow path length (m) set to a quarter of the cell width. Flow into the ditch is activated only when $H_{cell} > H_{ditch}$. In the model water from the ditches thus cannot reinfiltrate into the soil, which means that the coupling is incomplete in situations where the water table level in the surrounding soil is lower than the water level in the ditch.

The winter process descriptions in FLUSH (Turunen et al., 2015) correspond to those in FEMMA (see Section 2.2.1). In Paper **III**, the snow albedo formulation was changed to correspond to the method presented by Dutra et al. (2010). Computation of heat flow in FLUSH differs from FEMMA as it is solved in 3-D accounting for both convection and conduction (see Warsta et al., 2012).

2.2.3 Developed submodels for ditch network

Unsteady flow model

The model for computing 1-D unsteady flow in a network of open ditches (Figure 3) was first implemented with MATLAB (Paper **I**), and then with slight alterations the model was integrated to FLUSH using C++ (Paper **II**). The network flow model is based on an iterative algorithm (Figure 7), which consists of solving flow in each of the network ditches independently and correcting water depths at the junctions until conservation of mass is satisfied (Chen et al., 2012; Zhu et al., 2011). In FLUSH (Paper **II–IV**), the network flow model is explicitly coupled to the 3-D subsurface and the 2-D overland flow submodels, which feed water to the ditches as lateral inflow (Eqs. 2–3). The computed ditch water depths are in turn accounted for in the calculation of seepage flux from the soil to the ditch (Eq. 2).

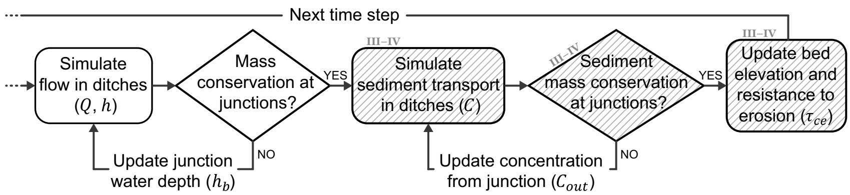

Figure 7. Flow chart of the solution algorithm used for modelling flow and sediment transport in a network of open ditches (see Figure 6 for preceding steps). Grey striped steps are only included in the indicated papers while other steps are included in all Papers **I–IV**.

Flow in the network channels is governed by the full St. Venant equations (Eqs. 4–5), which in the model are discretized using the Preissmann scheme and solved by the Newton–Raphson method (e.g., Cunge et al., 1980).

$$\frac{\partial A}{\partial t} + \frac{\partial Q}{\partial x} - q = 0 \tag{4}$$

$$\frac{\partial Q}{\partial t} + \frac{\partial}{\partial x}\left(\frac{Q^2}{A}\right) + gA\frac{\partial h}{\partial x} + gA(S_f - S_0) = 0 \tag{5}$$

where A is the cross-sectional area of flow (m^2), Q is discharge (m^3 s^{-1}), q is lateral inflow per unit length (m^2 s^{-1}), h is water depth (m), S_f is the friction slope (m m^{-1}), S_0 is the bottom slope (m m^{-1}), x is the longitudinal distance (m), and t is time (s). In Paper **II,** to avoid numerical oscillations during low flow, the inertia terms are suppressed and the solution of Eq. (5) is switched from central to fully upstream weighting when the derivative of Q with respect to the downstream water level becomes positive (e.g., Cunge et al., 1980). In Paper **I**, such numerical oscillations were avoided by using a small distance step (Δx).

The friction slope in Eq. (5) is estimated from Manning's equation:

$$S_f = \frac{n^2 Q|Q|}{A^2 R^{4/3}} \tag{6}$$

where R is the hydraulic radius (m), and n is Manning's flow resistance (s m$^{-1/3}$). The low flow rates in the ditches of the subcatchment (Figure 4c) increased the influence of roughness elements causing Manning's n to be strongly dependent on the flow conditions (Haahti, 2014). Hence, Manning's n was described in the model as a function of local discharge taking its final form in Paper **II**:

$$n(Q) = \min(0.02 + n_a|Q|^{-n_b}, 3) \tag{7}$$

where n_a and n_b are parameters accounting for increased flow resistance with decreasing flow rate.

After solving flow in the ditches, water depths at the junctions are corrected to satisfy conservation of mass (Chen et al., 2012):

$$h_{b_{k+1}} = h_{b_k} + \alpha \frac{\sum Q_{in} - \sum Q_{out}}{\sum A_{in}\sqrt{g/h_{in}} - \sum A_{out}\sqrt{g/h_{out}}} \tag{8}$$

where h_b is water depth at the junction (m), k is the iteration index, in and out refer to the incoming and outgoing flows, respectively, and α is a factor to ensure numerical stability.

Sediment transport model

The 1-D model for cohesive sediment transport in the network of ditches was integrated to FLUSH in Paper **III** (Figure 3). The developed model includes descriptions for bed erosion, rain-induced erosion from ditch banks, deposition with the effect of flocculation, and the consolidation of the ditch bed.

Along the ditches the transport of total bed-material load for fine sediment is expressed in the model as (e.g., Wu, 2007):

$$\frac{\partial(AC)}{\partial t} + \frac{\partial(AUC)}{\partial x} = E - D + q_s \tag{9}$$

where C is the volumetric sediment concentration (m^3 m^{-3}), U is flow velocity (m s^{-1}), E and D are the rates of erosion and deposition, respectively (m^3 m^{-1} s^{-1}), and q_s is the side discharge of sediment (m^3 m^{-1} s^{-1}). Eq. (9) is discretized using a fully-implicit upwind scheme and solved in the direction of flow in each of the ditches. Subsequently, the concentrations flowing out of the junctions (C_{out}) are updated using the incoming flow rates and concentrations (see Wu, 2007). These steps are repeated until sediment mass conservation is satisfied at all junctions (Figure 7).

The rates of erosion and deposition for cohesive sediment depend on the bed shear stress, τ_b (N m^{-2}), computed as:

$$\tau_b = \rho g \frac{{n_{bed}}^2 U^2}{R^{1/3}} \tag{10}$$

where ρ is the density of water (kg m^{-3}), and n_{bed} is Manning's roughness coefficient associated only with the bed material (s m$^{-1/3}$). Bed erosion occurs when τ_b exceeds a critical level (e.g., Partheniades, 1965):

$$E = B\frac{E_{bed}}{\rho_s}\left(\frac{\tau_b - \tau_{ce}}{\tau_{ce}}\right),\ \tau_b > \tau_{ce} \tag{11}$$

where B is the water surface width (m), E_{bed} is the erodibility coefficient (kg m^{-2} s^{-1}), ρ_s is the density of the sediment particles (kg m^{-3}), and τ_{ce} is the critical shear stress for erosion (N m^{-2}). Deposition on the other hand occurs, when τ_b decreases below a critical level (e.g., Krone, 1962):

$$D = B\omega_s\left(\frac{\tau_{bd} - \tau_b}{\tau_{bd}}\right)C,\ \tau_b < \tau_{bd} \tag{12}$$

where ω_s is the settling velocity of the sediment particles (m s^{-1}), and τ_{bd} is the critical shear stress for deposition (N m^{-2}).

Flocculation of cohesive sediments impacts their settling velocity, which can be accounted for by relating it to sediment concentration (e.g., Krone, 1962):

$$\omega_s = k_\omega \, C^{4/3}, \ \omega_{s0} < \omega_s < \omega_{s\max} \quad (13)$$

where k_ω is an empirical parameter, and ω_s (m s^{-1}) is limited by the settling velocity of dispersed sediment particles, ω_{s0} (m s^{-1}), and a maximum settling velocity, $\omega_{s\max}$ (m s^{-1}).

Side discharge of sediment into the ditch was described as erosion from the ditch banks by raindrop impact. Raindrop erosion was calculated with the same principle as in overland flow schemes (e.g., Wicks and Bathurst, 1996; Warsta et al., 2013b):

$$q_s = \left(B_{bf} - B\right) \frac{E_{rain} M_R}{\rho_s} \quad (14)$$

where B_{bf} is the bankfull channel width (m), E_{rain} is the rainfall erodibility coefficient (kg^{-1} m^{-2} s^{2}), M_R is the momentum squared for rain ((kg m s^{-1})^{2} m^{-2} s^{-1}) which is computed as $M_R = \gamma I^\delta$ where I is the rainfall intensity (mm h^{-1}), and γ and δ are empirical coefficients (see Wicks and Bathurst, 1996).

After solving sediment concentrations in the ditch network, the bed elevation and resistance to erosion (τ_{ce}) are updated (Figure 7) based on deposition and erosion rates, and bed consolidation. To compute consolidation over time, the bed is divided into a thin surface layer (i.e. mixing layer) and two underlying layers, which all have a specific mass, consolidation time (zero for freshly deposited material) and dry density (see Lin and Wu, 2013).

First, the mass exchange between the bed and the water column is computed by:

$$\frac{\partial M}{\partial t} = \frac{\rho_s}{B}(D - E) \quad (15)$$

where M is the bed mass per unit bed area (kg m^{-2}). Based on this, the mass and the consolidation time of each layer are updated so that the mass of the mixing layer remains constant. The consolidation times increase by one time step length (Δt) at each time step, and mass exchange between the layers, or the water column and the layers, is accounted for by mass-weighted averaging.

Second, the dry density of each layer, ρ_{dry} (kg m^{-3}), is calculated based on its consolidation time:

$$\rho_{dry} = \min\left(\rho_{dry0} + k_\rho t_{con}, \rho_{dry\max}\right) \quad (16)$$

where ρ_{dry0} and $\rho_{dry\max}$ are the dry bed densities of freshly deposited and fully consolidated sediments (kg m^{-3}), respectively, t_{con} is the consolidation time (h), and k_ρ is an empirical parameter, which was derived from the time needed for deposited material to stabilize (t_{stab}). At this point, the change in bed elevation can be calculated using the new dry densities and the mass of each layer.

Finally, the critical shear stress of the bed (to be applied at the next time step) is derived from the dry density of the mixing layer (Lin and Wu, 2013):

$$\tau_{ce} = \tau_{ce0} + k_\tau \left(\rho_{dry} - \rho_{dry0}\right)^{3/2} \quad (17)$$

where τ_{ce0} is the critical shear stress for erosion of freshly deposited sediment (N m^{-2}), and k_τ is an empirical parameter set based on the critical shear stress of the fully consolidated bed ($\tau_{ce\max}$).

2.3 Modelling approaches

2.3.1 Discretization and simulation periods

The discretization used to simulate canopy, soil, and ditch network processes in Papers **I**–**IV** are summarized in Table 2. The input and output time step used in all simulations was 1 h, but the applied computational time step varied (Table 2). Both FEMMA and FLUSH support the subdivision of time steps depending on the hydrological conditions.

Table 2. Discretization in space and time used in Papers **I–IV** for modelling domains

Modelling domain	Paper	Discretization	
		Space	Time (Δt)
Canopy	**I–IV**	2 vegetation layers	1.5–15 min
Soil	**I**	3 soil columns (5.8–5.9 m wide), soil layers: 10 × 0.1 m, 0.5 m, 2.5 m	1.5–15 min
	II–IV	5 × 5 m^2 soil columns, soil layers: 8 × 0.05 m, 6 × 0.1 m, 4 × 0.25 m	1–7.5 min
Ditch network	**I**	$\Delta x = 1$ m	1 h
	II	$\Delta x \leq 10$ m	1–7.5 min
	III–IV	$\Delta x \leq 10$ m	1 min–1 h

The 2-D soil profile in Paper **I** extended from the middle of the strip (no-flow boundary) to the ditch. In Papers **II**–**IV**, the modelling grid (surface topography interpolated from DEM) covered the 5.2 ha subcatchment area to a depth of 2 m. In both cases, an impermeable bottom boundary was present and the diches extended to the depth of 1 m. In Paper **I**, a constant ditch water depth of 0.05 m was specified as boundary condition to the FEMMA soil model, whereas in Papers **II**–**IV** the network flow model provided dynamic ditch water depth information for the 3-D FLUSH soil model. In the FLUSH applications, the borders of the catchment were treated as open boundaries, which allow horizontal water outflow in the saturated zone based on the soil surface slope (see Turunen et al., 2013).

The ditches were characterized by a cross-section determined based on the excavator shovel, which resembled a semicircle with a 0.35-m-wide flat bottom. Ditch bottom elevations were interpolated from the levelling measurements. The boundary conditions of the network flow model were specified as no-flow at the upstream ends and according to Eq. (1) at the catchment outlet. To avoid dry conditions, the sum of lateral inflow was limited by a minimum of 0.5 L s^{-1}. The sediment transport model only needed upstream boundary conditions for SS concentration, which were set to zero.

The simulations covered periods between the time of DNM on 17 August 2011, and mid-October 2013 (Figure 8). In Papers **II**–**IV** the model was initialized with a fully saturated soil profile, which was assumed to represent the conditions after snowmelt (Paper **II)** and DNM (Papers **III**–**IV**). In Paper **I** measured water table levels were used to determine initial conditions in the 3 soil columns. The sediment transport model (Papers **III**–**IV**) was run with an initial SS concentration of 50 mg L^{-1}, which was based on measurements immediately after DNM. Furthermore, a wash load of 2 mg L^{-1} was added to the simulated concentrations, as the measured concentrations remained at this level even during times of low flow.

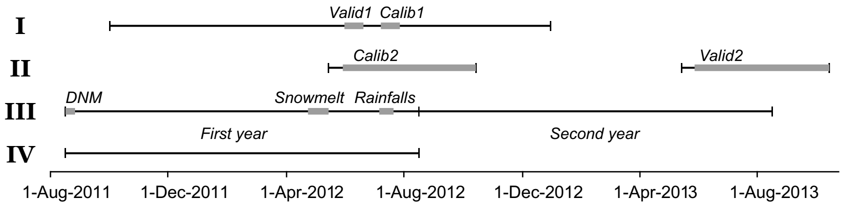

Figure 8. Simulation periods in Paper **I–IV** showing shorter interest periods in grey. *Calib* and *Valid* refer to calibration and validation periods, respectively.

2.3.2 Calibration and validation of hydrology

Papers **I**–**II** applied calibration–validation procedures to obtain parametrizations suitable for the subcatchment. The simulation results with the calibrated parameter sets were used for spatial assessments of erosion risk in the ditch network (Paper **I**), and of soil moisture conditions (Paper **II**). These papers represent important steps in the model development of this thesis (Figure 3), as they enabled the extension of model to sediment transport (Paper **III**), as well as the evaluation of effects of ponding structures on the surrounding soil hydrology (Paper **IV**).

In Paper **I**, 10-day periods during the summer of 2012 were chosen for calibration and validation (Figure 8). First, FEMMA (canopy + soil hydrology, Figure 3) was calibrated against measured runoff at the subcatchment outlet in the least squared error sense, and visually against water table depth observations (3 locations). This was achieved by manually adjusting selected parameters: saturated soil hydraulic conductivities (vertical and horizontal) and water retention parameters for the two topmost soil layers, and the scaling coefficient for potential evapotranspiration. Second, the ditch network flow model was calibrated against measured ditch water depth (5 locations) by adjusting the parameters in the description of Manning's n (see Eq. 10 in Paper **I**). The calibration was performed by minimizing the sum of squared errors with a built-in MATLAB optimization function based on the Nelder–Mead simplex algorithm (Lagarias et al., 1998).

In Paper **II**, the model (soil hydrology + ditch network flow, Figure 3) was calibrated and validated using 4.5-month-long periods in 2012 and 2013 (Figure 8). The calibration consisted of initial manual calibration and automatic calibration of selected parameters using PEST, a gradient-based parameter optimiser (WN Computing, 2004). The seven parameters to be calibrated with PEST were chosen based on a local sensitivity analysis, which provided information on parameter sensitivity and correlation (see Foglia et al., 2009). The calibration was performed against observations of discharge (1 location), water table depth (2 locations), and ditch water depth (4 locations). The mean squared error (MSE) values of each observation type were aggregated to an objective function (Φ) by assigning each observation type weights relative to their magnitudes:

$$\Phi = \sum_{i=1}^{3} {w_i}^2 MSE_i \tag{18}$$

where i refers to the observation type, and weights were set to $w_1 = 0.2$, $w_2 = 10$, and $w_3 = 50$, for discharge, water table depth, and ditch water depth, respectively. The parametrization resulting from the calibration was applied in Papers **III**–**IV** to compute year-round simulations (see Figure 8).

Model performance for the modelling approaches in Papers **I**–**II** were recalculated in Section 3.1.1 for the calibration and validation periods of both papers (Figure 8) to enable comparison. The calculated performance criteria included the Nash–Sutcliffe model efficiency measure (NS) and the root mean squared error ($RMSE$) for runoff at the network outlet, as well as $RMSE$ for water table depth from 2 locations, and for ditch water depth from 4 locations (loggers functioning in 2012 and 2013, see Figure 4c). Furthermore, the hydrological simulations in Paper **III** were extended until 15 October 2013 to visualize the hydrological conditions over the simulation periods of all papers in Section 3.1.2.

2.3.3 Parametrization and evaluation of sediment transport

In the application of the sediment transport model (Paper **III**) the focus was on recognizing the importance of each modelled process, as well as on assessing the diverse experimental dataset alongside the simulation results. Due to uncertainties inherent in the SS concentration data (e.g., due to sensor fouling), the sediment transport model was run using the Monte Carlo approach to produce ranges of possible model outcomes. The role of different modelled processes was assessed by increasing the model complexity stepwise. The first step (R1) included ditch bed erosion and deposition. To the next steps the effect of raindrop erosion was added first (R2), then the flocculation effect on settling velocity (R3), and finally the consolidation of the bed (R4). For each step (R1–R4), 100 parameter sets were sampled from uniform distributions covering the parameter ranges shown in Table 3. To decrease simulation times, only the submodels for ditch network processes (steps in Figure 7) were run repeatedly (4 × 100 times). The full hydrological simulations were only run once to produce lateral inflow to the network prior to the Monte Carlo simulations.

Table 3. Sediment transport model parameters in model setups R1–R4 (modified from Paper **III**)

Symbol	Range	Description	R1	R2	R3	R4
n_{bed}	0.025 s m$^{-1/3}$	Manning's roughness for peat bed	x	x	x	x
ρ_s	1100 kg m^{-3}	Sediment density	x	x	x	x
$\rho_{dry(\max)}$	55 kg m^{-3}	Dry density of (consolidated) bed	x	x	x	x
τ_{bd}	0.01–0.1 N m^{-2}	Critical bed shear stress for deposition	x	x	x	x
ω_s	0.5–1.09 m h^{-1}	Constant settling velocity	x	x		
$\tau_{ce(\max)}$	0.06–0.13 N m^{-2}	Critical bed shear stress for erosion (of consolidated bed)	x	x	x	x
E_{bed}	3.4–7.5 ·10^{-6} kg m^{-2} s^{-1}	Erodibility coefficient for ditch bed	x	x	x	x
E_{rain}	0.0–5.0 kg^{-1} m^{-2} s^{2}	Rainfall erodibility coefficient		x	x	x
k_ω	10.0–40.0	Empirical parameter for flocculation			x	x
ω_{s0}	0.048–0.072 m h^{-1}	Settling velocity for free settling			x	x
$\omega_{s\max}$	1.35–2.03 m h^{-1}	Maximum settling velocity			x	x
ρ_{dry0}	45 kg m^{-3}	Dry density of freshly deposited bed				x
t_{stab}	3–5 days	Time for deposited peat to stabilize				x
τ_{ce0}	0.006–0.018 N m^{-2}	Critical bed shear stress for erosion of freshly deposited peat				x
δ_{init}	0.0–0.05 m	Initial thickness of loose bed layer				x

The simulation results with R1–R4 were evaluated against the measured SS concentrations at the catchment outlet during three periods of special interest: days after DNM, spring snowmelt, and high-intensity summer rainfalls (Figure 8). NS

was applied to assess consistency between measured (turbidity-based) and modelled SS concentrations during these periods. Erroneous sections of data, likely caused by sensor fouling, were not considered in the calculation of NS. Furthermore, the simulated SS loads, bank erosion rates, and changes in ditch bed elevation during the two years after DNM were compared against estimates derived from the diverse measurements in the subcatchment (Table 1).

2.3.4 Setup and assessment of sediment control scenarios

In Paper **IV**, the modelling framework was further extended to investigate the efficiency of sediment control practices, including sedimentation ponds and pits, breaks in cleaning, submerged weirs and PRC. Guidelines for forestry water protection (Joensuu et al., 2012a; Päivinen et al., 2011; Vanhatalo et al., 2015) were used to select the investigated scenarios (Figures 9b–l) and the dimensions applied for the structures (Figures 10, 11b). In addition to the scenarios presented in Figures 9b–l, pits were combined to the structures in Figures 9d–f resulting in 3 additional scenarios: *Pit-Break*, *Pit-SWeir*, and *Pit-PRC*.

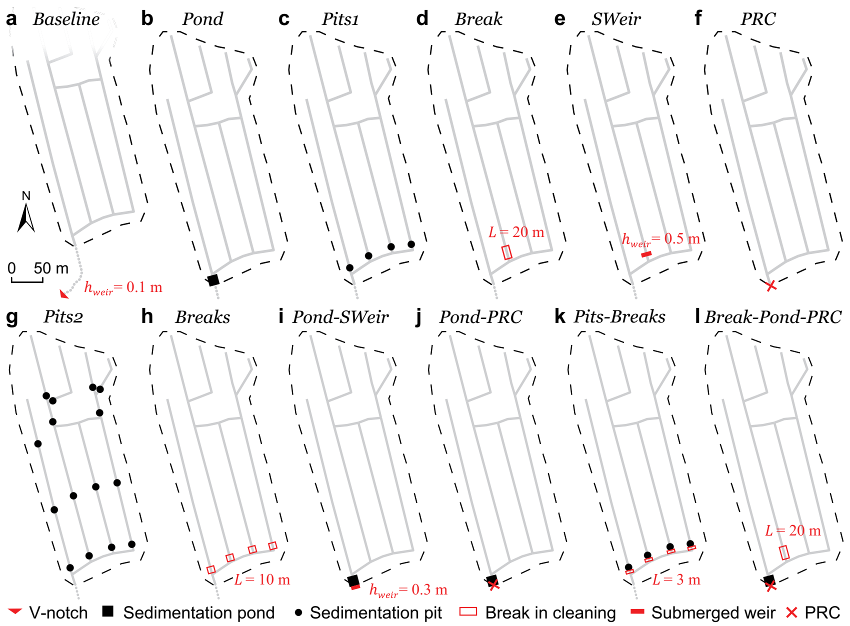

Figure 9. Structure locations within the ditch network of the subcatchment in the baseline (a) and sediment control scenarios (b–l). L is the length of the break in cleaning and h_{weir} is the height of the weir structure. (Modified from Paper **IV**)

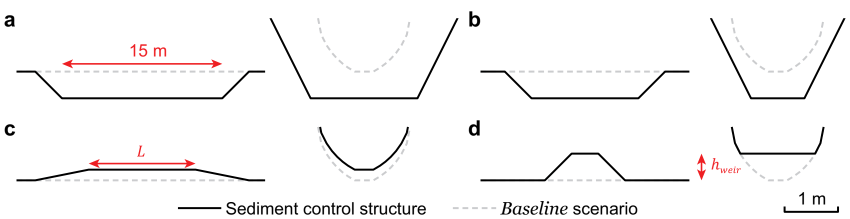

Figure 10. Longitudinal ditch bottom profile and cross-section for the sedimentation pond (a) and pit (b), the break in cleaning (c), and the submerged weir (d). All dimensions are in scale except for the indicated ones. L and h_{weir} varied depending on the scenario. (Modified from Paper **IV**)

To assess the efficiency of sediment control scenarios in reducing SS loads, a *Baseline* scenario needed to be setup. This was realized by moving the V-notch weir at the catchment outlet (Figure 4c) 90 m downstream and by lowering its bottom elevation to h_{weir} = 0.1 m (Eq. 1; Figure 9a). The prevailing setup with the V-notch weir at the catchment outlet did not represent a proper baseline as such, because the weir ponded water. However, this scenario (*V-notch*) was still computed among the other scenarios.

All structures were implemented in the model by altering the cross-section and bottom elevation of a selected ditch section (Figure 10), except for weir structures located at the catchment outlet (e.g., Figures 9f, i), which were described by altering the downstream boundary condition. For the submerged weir, the downstream boundary condition was expressed by the equation for flow over a broad-crested weir (Hamill, 2001):

$$Q = 0.36\sqrt{2g}B_{weir}(h - h_{weir})^{1.5} \tag{19}$$

where B_{weir} is the width of the weir surface (m), and h_{weir} is the height of the weir from the ditch bed (m). For the PRC setup consisting of two pipes (Figure 11a), the rating curve used as downstream boundary condition is shown in Figure 11b. Some structures also required adjustments to the flow or erosion related parameters. The break in cleaning was described in the model by a higher flow resistance (Eq. 7 restricted by a minimum of n = 0.5), non-eroding banks (E_{rain} = 0), and a fully consolidated bed at the initial state after DNM. Along the submerged weir, E_{bed} was set to zero, as the material used to construct the weir (wood or stones) should prevent erosion.

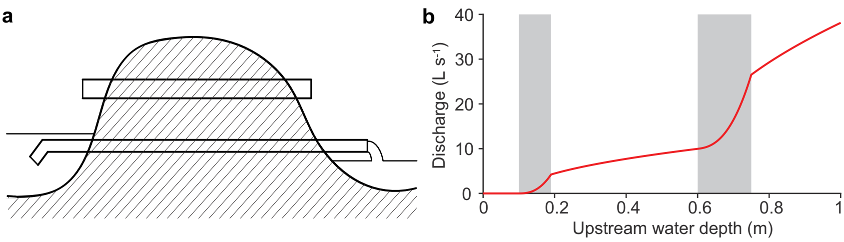

Figure 11. Schematic figure of a two-pipe PRC structure (a) and the rating curve of the selected pipe setup showing pipes as grey areas (b). (Modified from Paper **IV**)

All scenarios (*Baseline* + *V-notch* + 14 sediment control scenarios) were simulated for the first year after DNM (Figure 8) with all sediment transport model processes active (R4). The same procedure of first running the hydrological simulations followed by the 100 runs of only the ditch network processes, as described in Section 2.3.3, was adopted here for each scenario. The simulation results of the scenarios were compared against the results of the *Baseline* scenario. The changes caused to the flow conditions in the ditch network and their feedback to the water table levels in the strips between ditches, as well as the changes in sediment transport processes were investigated. Regarding the changes in water table levels it should be noted that structures ponding water in ditches cannot result in water reinfiltrating into the soil. In case that the ditch water level exceeds the surrounding water table level in the soil the simulation results (e.g. soil moisture) in the vicinity of the structure may be misleading.

3 Results

3.1 Hydrology

3.1.1 Calibration and validation

The NS efficiencies of the modelling approaches in Papers **I**–**II** for producing runoff during the periods in 2012 ranged from 0.68 to 0.80 (Table 4). The higher NS values for the approach in Paper **II** during both calibration periods (Table 4) were due to the models better ability to capture the peak flows during the intense rainfall events. The validation period in 2013 of Paper **II** produced a poorer NS value of 0.32 (Valid2, Table 4). However, $RMSE$ calculated for runoff during the dryer summer of 2013 was smaller (0.036 mm h^{-1}) than for any of the periods during the wetter summer of 2012 (0.068–0.137 mm h^{-1}). In both modelling approaches, simulated runoff was sensitive to the saturated horizontal soil hydraulic conductivity and the water retention parameters of the topmost peat layer (0.1 m). Calibration in both cases led to a horizontally highly conductive surface peat layer, especially in Paper **II**.

Table 4. Model performance for outputs of the hydrological modelling applied in Papers **I** and **II** during both calibration and validation periods

Period*	Paper	Runoff at network outlet NS (-)	Runoff at network outlet $RSME$ (mm h^{-1})	Water table depth $RSME$ (m)	Ditch water depth $RSME$ (m)
Calib1	**I**	0.67	0.137	0.048	0.023
	II	0.80	0.107	0.020	0.012
Valid1	**I**	0.76	0.080	0.048	0.021
	II	0.73	0.085	0.048	0.013
Calib2	**I**	0.68	0.076	0.058	0.017
	II	0.75	0.068	0.039	0.013
Valid2	**I**	-	-	-	-
	II	0.32	0.036	0.101	0.012

*see Figure 8

Calibration of water table depths, which was performed more systematically in Paper **II**, resulted in more accurate representation of the measured water table depths compared to Paper **I** (Table 4). The validation period of Paper **II** differed in performance as $RMSE$ between observed and simulated water table depths was 0.101 m (Table 4). During this period, the water table depths were systematically underestimated when they exceeded 0.2 m. Ditch water depth was produced with rather consistent accuracy throughout all periods, $RSME$ for the approaches in Papers **I** and **II** were 0.017–0.023 m and 0.012–0.013 m, respectively.

3.1.2 Year-round simulations

Simulation results (covering periods modelled in Papers **II–III**) and observations for the 26 months following DNM are presented in Figure 12. Precipitation during the first year was notably higher (939 mm) than the long-time average for the region (see Section 2.1.1) or during the second year (747 mm). Overall interception by the forest stand and ground vegetation was 19 % of precipitation. During the first summer (June–August) runoff dominated the water balance (66 % of precipitation) over evapotranspiration, while during the second summer it was reversed (evapotranspiration 63 % of precipitation). As noted in Paper **II**, water table depth remained shallow during the first summer while during the second summer water table level decreased down to 0.4 m from the soil surface, which was poorly reproduced by the model (Figure 12b; Table 4).

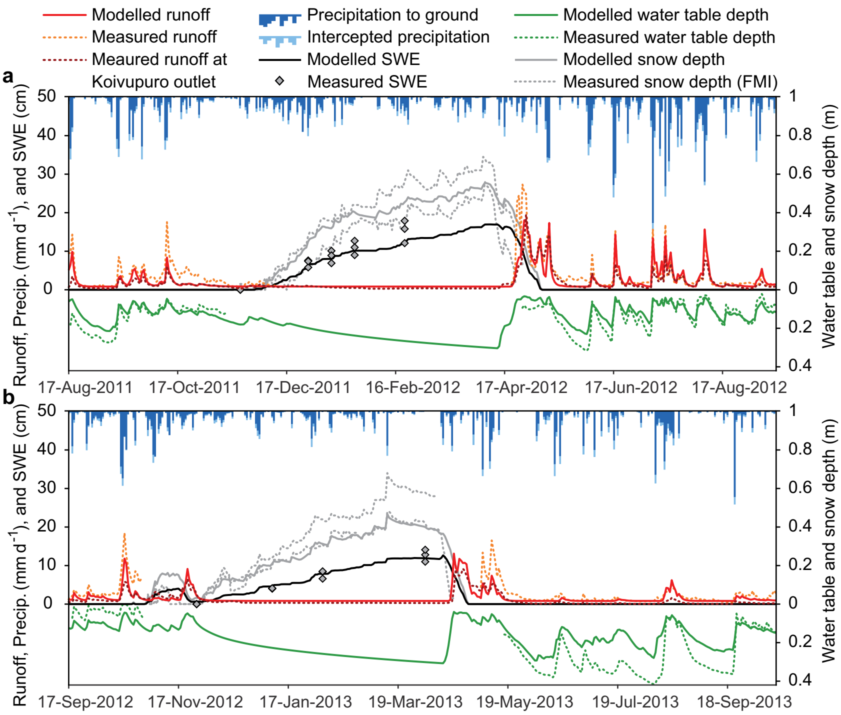

Figure 12. Simulated daily precipitation, runoff, snow water equivalent (SWE), as well as water table and snow depth for 17-Aug-2011 to 16-Sep-2012 (a) and 17-Sep-2012 to 15-Oct-2013 (b) compared to measurements. Runoff for Koivupuro outlet compiled by Stenberg et al. (2015b). Measured snow depth from FMI stations in Sotkamo and Valtimo (see Table 1).

Both winters featured an approximately 5-month-long continuous snow cover period with maximum SWE of 170 mm and 126 mm during the first and the second winter, respectively (Figure 12). Simulated snow depth and SWE followed the dynamics of observations, which in the case of SWE, however, were sparse and showed high variability. Observed runoff, especially during spring, showed some inconsistencies between the two observation locations. Overall the simulated daily subcatchment runoff of the two years after DNM (Paper **III**) correlated well with the continuous runoff time series compiled by Stenberg et al. (2015b) for the Koivupuro catchment (correlation: 0.92 for first year, 0.87 for second year).

3.1.3 Spatiality in simulation results

According to the hydrological simulations of Papers **II–III**, the flow conditions within the ditch network followed the spatial patterns presented in Figures 13a and c. The spatial distribution of ditch water depth (Figure 13a) revealed that the V-notch weir installed at the catchment outlet ponded water in the downstream parts of the ditch network. The high frequency of ditch water depths around 0.3 m in Figure 13b during both years was also caused by the ponding. Otherwise, the depths remained rather low, most frequently below 0.1 m.

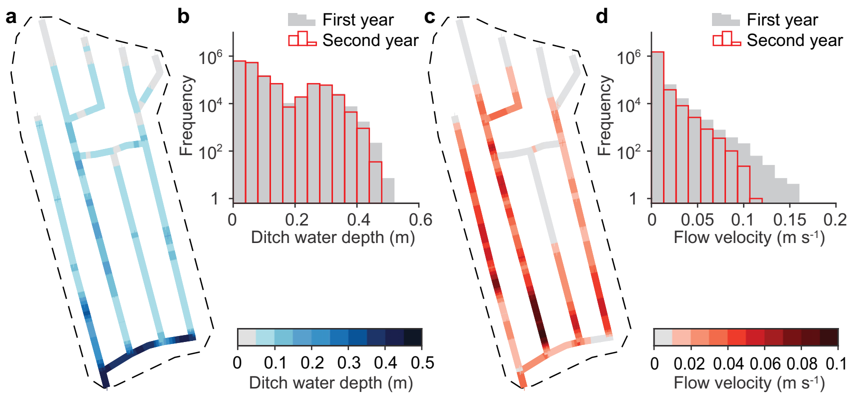

Figure 13. Simulated ditch water depth (a) and flow velocity (c) during a rainfall event on 7-Aug-2012, and the frequency distributions of ditch water depth (b) and flow velocity (d) over the two years after DNM in the entire ditch network.

The flow velocity distribution within the network during the flow event on 7 August 2012 was spatially highly variable (Figure 13c). As noted in Paper **I**, the highest risk for bed erosion according to the model would be in the steep southern end of ditch D2. In the upstream feeder ditches, the discharge and thereby the flow velocities remained low even during high flow events. Also in the downstream end of the network, where water was ponded by the V-notch weir, flow velocities remained low. Overall, values of velocities within the ditch network were relatively low, even more so during the second year after DNM (Figure 13d). The highest

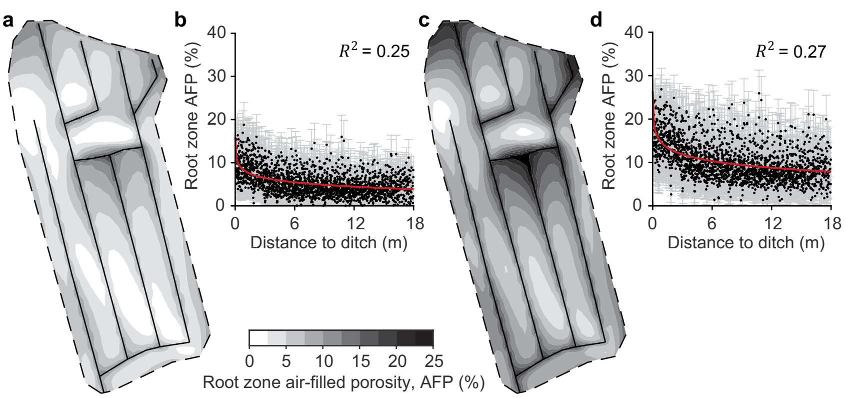

Figure 14. Simulated mean root zone air-filled porosity (AFP) during July–August spatially in subcatchment in 2012 (a) and 2013 (c), and against distance to nearest ditch in 2012 (b) and 2013 (d). Error bars show the standard deviation caused by temporal variation and R^2 is the coefficient of determination of the fitted logarithmic regression curves ($p < 0.05$). (Modified from Paper **II**)

flow velocity values reached in the ditches were about 0.16 m s^{-1} during the first year and 0.11 m s^{-1} during the second year (Figure 13d).

The simulated root zone air-filled porosity (AFP) in July–August studied in Paper **II** was notably different during the two years (Figure 14). In 2012, the mean root zone AFP was 5.5% and in 2013 10.2%. The spatial distributions presented in Figures 14a and c show that the vicinity of the ditch had a distinct effect on the root zone AFP. Figures 14b and d further show the relation between the mean root zone AFP and the distance to the nearest ditch. Logarithmic regression curves fitted to the data (significant at $p < 0.05$) suggested that 25% and 27% of the variation in the mean root zone AFP (July–August) was explained by the vicinity of the nearest ditch in 2012 and 2013, respectively. The temporal variability of root zone AFP, also shown in Figures 14b and d, revealed larger variation during the dryer year 2013 compared to the wetter year 2012.

3.2 Sediment transport

3.2.1 Outflow SS concentrations

Table 5 represents the model performance (NS) between modelled and measured SS concentrations obtained for the setups R1–R4 during the periods investigated in Paper **III**. Immediately after DNM, the model setups R1–R3, which featured a ditch bed with constant erodibility properties throughout the period, were not able to reproduce the observed elevated SS concentrations immediately after DNM resulting in low NS values (Table 5). The model performance notably improved when bed consolidation was accounted for (R4, Table 5). The loose, easily erodible surface layer present on the ditch bottom in R4 enabled reproduction of the high SS concentrations recorded during the first days after DNM (Figure 15a).

Table 5. Model performance during investigated periods with increasing model complexity*

Period**	NS between modelled and measured (turbidity-based) SS concentrations			
	R1	R2	R3	R4
DNM	-0.455	-0.450	-0.173	0.619
Snowmelt	0.035	0.034	0.029	0.031
Summer	0.151	0.546	0.615	0.630

*see Section 2.3.3 for descriptions of R1–R4, **see Figure 8

The snowmelt in spring 2012 spanned over two weeks and was followed by a rainfall event during 11–12 May. The measured SS concentrations at the outlets of both catchments followed the daily dynamics of spring runoff caused by daytime melting of the snow pack (Figure 15b). Although the model results of Paper **III** featured the same type of behaviour (Figure 15b), NS values remained low through all model setups (Table 5). As shown in Figure 12a, there were problems in the runoff recordings at the subcatchment in the beginning of the snowmelt period, which may also have affected the turbidity recordings. For example, the high turbidity spikes recorded at the subcatchment on 28 and 29 April differed substantially from the turbidity recorded at the Koivupuro catchment outlet (Figure 15b).

The studied two-week period in July 2012 (Figure 15c) was rainy with rainfall intensities up to 13.9 mm h^{-1} and frequent runoff events (Figure 12). Measured SS concentrations during this summer period peaked on the rising limb of the hydrographs, which was reproduced by the model in Paper **III** only after the bank erosion by raindrop impact was accounted for (R2). Bed erosion occurred only during the high runoff events on 9 and 16 July, but when bank erosion was accounted for the SS concentrations recorded during lower intensity events, were also captured (Figure 15c). This led to the notable increase in *NS* moving from R1 to R2 (Table 5). Further improvement, albeit smaller, was provided by including the rest of the model processes (Table 5).

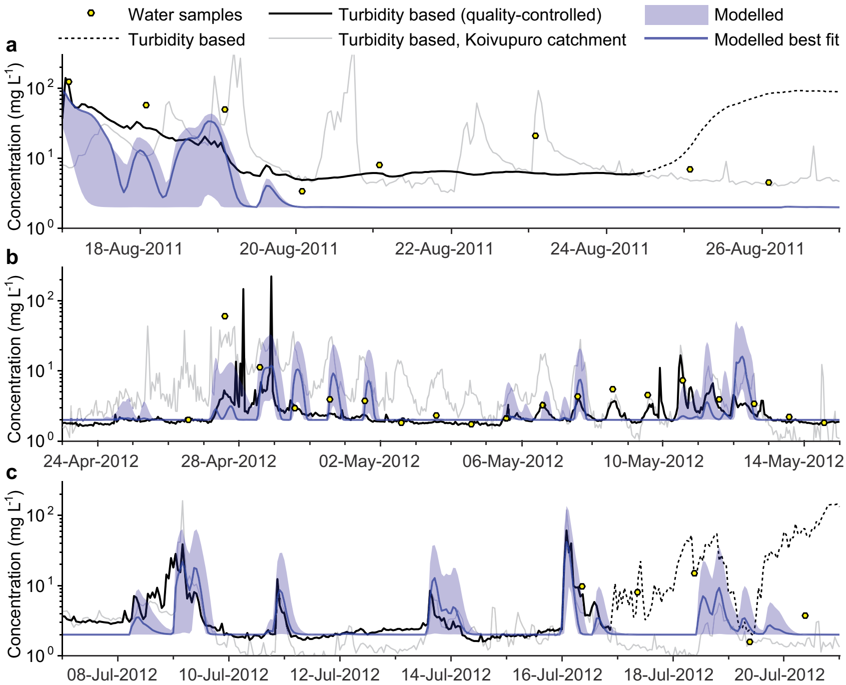

Figure 15. Measured and modelled SS concentrations with R4 (see Section 2.3.3) after DNM (a), during snowmelt (b), and during rainfalls (c). The modelled best fit corresponds to the highest *NS* calculated against the quality-controlled measurements (see Table 5). (Modified from Paper **III**)

3.2.2 Erosion, deposition and SS loads

Figure 16 presents the simulated changes in bed elevation in ditches D2 and D3 (see Figure 4c) from October 2011 to May 2012 computed in Paper **III**. This is one

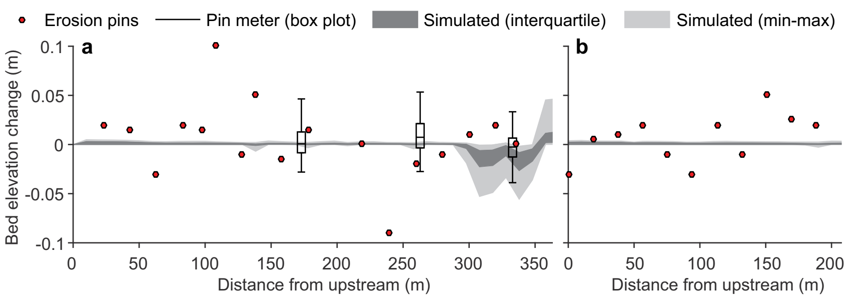

Figure 16. Simulated change in bed elevation computed with R1–R4 (see Section 2.3.3) compared to pin meter and erosion pin observations in ditches D2 (a) and D3 (b) from October 2011 to May 2012. (Modified from Paper **III**)

of the periods for which the changes were measured with erosion pins and the pin meter (see Section 2.1.2). The measured bed elevation changes showed high variability both during this period (Figure 16) and other periods (Figure 6 in Paper **III**). The erosion pin measurements ranged from -0.17 to 0.15 m, with rather inconsistent behaviour in space and time. The model suggested erosion mainly in the downstream section of D2 (Figure 16a), which was somewhat comparable with the pin meter observations, but differed markedly from the erosion pin measurements.

The annual SS loads simulated with R4 in Paper **III** for the two years after DNM (Figure 17) can be set against the estimates by Stenberg et al. (2015b), which were based on SS concentrations analysed from water samples and turbidity values measured at the subcatchment outlet (see Section 2.1.2). According to the measurements, the SS load for the first year was 39–75 kg ha^{-1} a^{-1}, and for the second year 3–12 kg ha^{-1} a^{-1}. The different meteorological conditions between the two years (see Section 3.1.2) were the main reason for the higher simulated SS load during the first year compared to the second. Adding the description of the loose bed surface layer at the initial state immediately after DNM (R4) had only a minor contribution as it increased the load of the first year on average by 3 %.

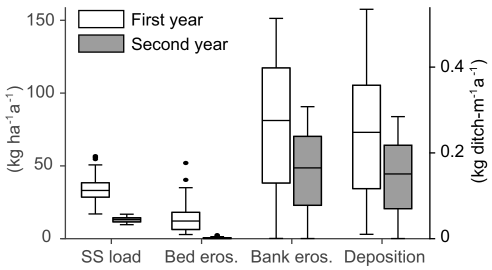

Figure 17. Simulated sediment masses per catchment area (left axis) and per ditch length (right axis) during the two years after DNM with model setup R4 (see Section 2.3.3). (Modified from Paper **III**)

Figure 17 (Paper **III**) further shows that according to the simulations only a small fraction of the eroded mass (from the ditch bed and the banks) was transported to the subcatchment outlet. This is supported by the measurement-based estimates of bank erosion within the subcatchment during the first year after DNM. The estimate derived from the sediment collectors was 0.35 kg m^{-1} a^{-1} (Stenberg et al., 2015b) while the estimates derived from measured topographic changes of the ditch banks with the erosion pins and the pin meter were considerably higher 3 kg m^{-1} a^{-1} and 1.54 kg m^{-1} a^{-1}, respectively (Stenberg et al., 2015b).

3.3 Sediment control scenarios

3.3.1 Ponding effects and changes in flow conditions

According to the simulations in Paper **IV**, adding structures in the ditches altered flow conditions in the ditch network (Figures 18a–b) compared to the *Baseline* scenario (Figure 9a). Figures 18a–b only show the changes during a single rainfall event, whereas in Paper **IV** conditions over the whole first year were analysed. Most wide spread effects during the rainfall event were caused by the *PRC*, which markedly raised the water level and decreased flow velocities. The *Pond* decreased flow velocities (Figure 18b) and the velocity of 0.03 m s^{-1} was never exceeded in the pond (Figure 5f in Paper **IV**). Similarly, within the pits velocities were lowered, but higher velocities occurred before the pits (*Pits1,* Figure 18b). The *Break* and the *SWeir* increased ditch water level along and upstream of the structure (Figure 18a). Both structures mostly lowered flow velocities (Figure 18b) except for the downstream end of the break in cleaning and the submerged weir location (short section not distinguishable in Figure 18b).

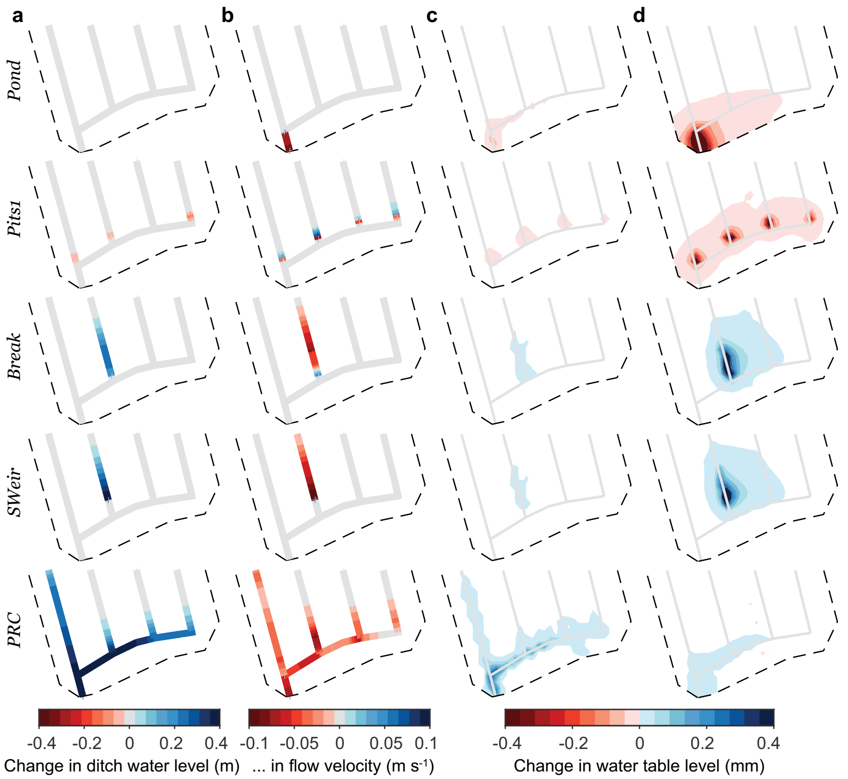

Figure 18. Spatial change caused by sediment control scenarios in Figures 9b–f during a rainfall event on 7-Aug-2012 in ditch water level (a), flow velocity (b), and water table level (c), as well as during 15-Jun-2012 to 15-Aug-2012 in water table level (d). Absolute water table level changes < 0.0025 mm are not shown. (Modified from Paper **IV**)

Although the ditch water levels rose, they never exceeded the surrounding water table level, which indicates that for this period reinfiltration (not handled by model) would not have affected the results. In case of the *SWeir*, the ditch water level rose closest to the surrounding water table level but even then it remained 0.11 m lower. Hereby the results can be used to assess the effects of the ponding structures on the water table level. According to the simulations, the raised ditch water levels caused by the *Break*, the *SWeir* and the *PRC* were reflected in the simulated water table level in Paper **IV** (Figures 18a, c–d). The changes in water table level were, however, clearly less pronounced than the changes in ditch water level. For all structures except for PRC, the changes in water table level were clearer in Figure 18d, illustrating the mean summertime conditions, than in Figure 18c, showing the conditions during the rainfall event. The *Pond* and the *Pits1* showed decreased water table level (Figures 18c–d), most likely resulting from the increased contact area between the ditch and the peat soil along the structure.

3.3.2 Effects on sediment transport

All sediment control scenarios simulated in Paper **IV** decreased SS load except for the scenarios featuring only pits (Figure 19a). The load increase by the *Pits1* and the *Pits2* was caused by the increased bed erosion (Figure 19c) resulting from the higher flow velocities before the pits (Figure 18b). The *V-notch* decreased SS load notably as the ponding effect (Figure 13) reduced bed erosion and enhanced deposition (Figures 19b, d). The *PRC* affected bed erosion and deposition similarly, but resulted in larger SS load reductions (15–40%). The effect of the *PRC* was, however, smaller during the first autumn (Figure 19a). The *Pond* decreased SS load marginally (2–15%) purely by enhancing deposition (Figures 19a, d). The *Break* and the *SWeir* decreased SS loads mainly by reducing bed erosion (Figures 19a–b). Among the single structure scenarios, the *SWeir* resulted in the highest reductions (max 46%).

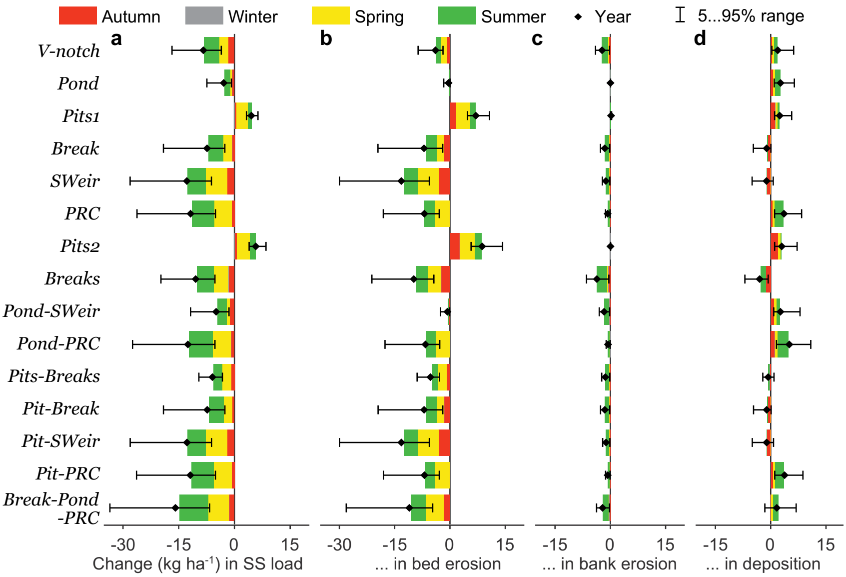

Figure 19. The effects of the V-notch and sediment control scenarios (see Section 2.3.4) on SS load (a), bed erosion (b), rain-induced bank erosion (c), and deposition (d) as a change compared to the *Baseline* scenario during the first year after DNM. (Modified from Paper **IV**)

Combining structures (*Break*, *SWeir*, *PRC*) with pits (*Pit-Break*, *Pit-SWeir*, and *Pit-PRC*) did not improve SS load reductions (Figure 19a). The short breaks in cleaning after the pits (*Pits-Breaks*) improved sediment control compared to the *Pits1* (Figure 19a) as they ensured that flow velocities would not increase before the pits (as in Figure 18b). Nevertheless, replacing the pit-break combinations with breaks only was more efficient (*Breaks*, Figure 19a). Combining a damming structure with the pond (*Pond-SWeir* and *Pond-PRC*) improved sediment control compared to the *Pond* (Figure 19a). However, the reduction by the *Pond-PRC* was only marginally superior compared to the scenario without a pond (*PRC*). The last combination scenario (*Break-Pond-PRC*) was the most efficient one, decreasing SS load by 19–52%.

4 Discussion

4.1 Parameterization approaches

Process-based hydrological models, such as the one applied in this thesis, aim to describe dominant processes based on physical principles (Fatichi et al., 2016). In theory, this means that model parameters have a physical meaning and are measurable. The differences between modeling and measurement scales (Refsgaard, 1997), as well as the fact that no model is purely based on physical principles (Beven and Young, 2013), however, complicates model parameterization. A typical empirically derived description included in such models is, e.g., Manning's equation (Eq. 6). In Paper **I**, it was recognized that no literature-based constant value for Manning's n was suitable for representing the flow conditions in the Koivupuro ditch network. Although the increase of flow resistance with decreasing flow rate (Eq. 7) has been reported in field studies (e.g., Hosia, 1980), it is rarely implemented in models. Tuukkanen et al. (2012) applied a constant Manning's n of 0.022 to compute steady-state flow conditions in a forestry ditch network. However, to assess erosion risk, they assumed that the erosion risk depended, in addition to the computed flow velocity, on the size of the contributing catchment area. The need for such an assumption may reflect the different flow resistance in the feeder versus the collector ditches.

The description of flow resistance thus included model parameters that required calibration. In addition, selected other parameters were calibrated in the model applications of Papers **I**–**II**. The automatic calibration (Doherty, 2004) used in Paper **II** to tune parameters is quite an uncommon approach in computationally intensive distributed hydrological modelling. Based on Paper **II**, it was judged that such an approach was transparent and easily documentable compared to manual calibration, which is the most widely used approach in complex hydrological modelling applications (e.g., Thompson et al., 2004; Turunen et al., 2013). Ideally, as many parameters as possible should be determined from field measurements (e.g., Refsgaard, 1997). Tuning parameters, even if restricted to only a few, can lead to some parameters masking the effect of others, especially with automatic approaches. For example, in Paper **II** the automatic adjustment of the horizontal conductivity of the topmost peat layer seemed to have an overpowering role. Then again, in Paper **I**, manual calibration allowed the user to adjust the soil hydrological properties with more consideration.

Calibration of model parameters rests upon the reliability of monitoring data. In the Koivupuro subcatchment monitoring was challenged by many factors

(Stenberg et al., 2015b; Tuukkanen et al., 2016). For example, runoff and turbidity recordings suffered from freezing episodes during the spring seasons, and the positions of the ditch water level loggers in reference to the surface of the ditch bed were difficult to keep track of. Turbidity was furthermore affected by sensor fouling, and the conversion from turbidity to SS concentration included uncertainties. Hereby, the parameterization of the sediment transport model was approached in a different way (Papers **III**–**IV**). Parameter values or ranges for them were derived from literature and measurements at Koivupuro (Table 3) in order to produce ranges of model outputs rather than a single output with calibrated model parameters as in Papers **I**–**II**. In Paper **III**, this proved to be useful when assessing time periods dominated by different processes. As shown Paper in **II,** calibrating parameters, identified as sensitive during the wet conditions in 2012, led to poor representation of water table depth during the dryer validation period in 2013 (Figure 12). Similarly, if in Paper **III** the model would have been calibrated during the period right after DNM (Figure 15a), performance during the summer rainfall events (Figure 15c), with bank erosion as an important component, would probably have been poor.

Finally, as models are simplified mathematical representations of physical reality (Refsgaard and Henriksen, 2004), they include only a limited number of process descriptions. For this reason, calibrating model parameters may sometimes compensate for missing process descriptions. For example, the high horizontal conductivity in the topmost peat layer may indicate the presence of preferential flow paths. The role of macropores in draining water from peatlands has been acknowledged in earlier studies (Heikurainen and Joensuu, 1981; Holden et al., 2006) as well as included in the modelling of, e.g., clayey agricultural fields (Warsta et al., 2013a). Sequential shrinkage–swelling behaviour in peat soil due to changes in water content is also known to affect the moisture dynamics, as it alters the pore structure and thereby the hydraulic properties (e.g., Price and Schlotzhauer, 1999). Camporese et al. (2006) proposed an approach to consider this phenomenon in a model developed for peat soils. Overly complicating a model should, however, be avoided (Fatichi et al., 2016). The approach in Paper **III** showed that adding process descriptions one by one can help to judge whether their inclusion is warranted.

4.2 Soil moisture patterns

The 3-D modelling of the soil domain implemented from Paper **II** onwards enabled the assessment of the spatiotemporal variability of soil moisture, the key factor controlling tree growth (Kozlowski, 1986) and therefore essential for forestry management on peatlands. It has been recognized that excess moisture is highly likely to suppress growth of Scots pine on peat when root zone AFP decreases below 10% (Paavilainen, 1967). A constant critical value for AFP is, however, deceptive as the restriction on growth originates from oxygen stress in the root zone, which varies depending on its demand (Bartholomeus et al., 2008). In Paper **II**, the assessment of the root zone AFP, during the period of highest oxygen demand

(July–August), proposed that aeration would be an issue especially during the wetter summer 2012 in the studied subcatchment (Figure 14a).

The spatial distributions of AFP, simulated in Paper **II**, indicated wet areas in the middle of the strips, but also other wet locations, which were not evident based on the drainage configuration (Figure 14a, c). Further inspection revealed that 25–27% of the variation of the late summer AFP was explained by the distance to the nearest ditch (Figure 14b, d). Site topography, which was the other spatially described factor in the model setup for Koivupuro, contributed to the remaining variation. The role of site topography on drained peatlands was also underlined by Holden et al. (2006) and Haahti et al. (2012). Earlier modeling studies on drained peatlands describing conditions between two parallel ditches (e.g., Koivusalo et al., 2008; McCarthy et al., 1992) disregard the effect of site topography. Furthermore, the horizontal grid resolution applied in Paper **II** enabled the assessment of soil moisture on a spatial scale finer than the ditch spacing, which makes it stand out from earlier distributed modeling studies on drained peatlands (e.g., Dunn and Mackay, 1996; Lewis et al., 2013). The variation in soil moisture within the strips between ditches on forestry sites is important to consider as it can have significant consequences on local tree growth conditions as shown, e.g., by Heikurainen (1980).

The integration of the network flow model to the 3-D soil hydrological model in Paper **II** was designed to account for the effect of the dynamic ditch water level on soil moisture in the strip between the ditches. This was motivated by the need to assess the effect of structures implemented in the diches on the surrounding soil moisture. The description for the dynamic connection between ditch water level and strip soil moisture was successfully included in the integrated model as demonstrated in Paper **II**, and later with the sediment control scenarios in Paper **IV**. A notable shortcoming in the coupling is, however, that reinfiltration from the ditches to the surrounding soil is not represented. Under dryer conditions, ponding structures may have more significant effects on surrounding soil moisture than suggested by Paper **IV**, which featured an exceptionally wet year. Regarding future model applications, proper representation of reinfiltration should be considered.

The simulated water table rise due to structures in the ditches (Paper **IV**) was, however, marginal (Figures 18c–d). Although the structures affected the ditch water level markedly (Figure 18a), the water level remained below the highly conductive surface peat layer, which explained the marginal effect on the strip water table level. Hökkä et al. (2011a) correspondingly reported no significant water table level rise due to PRC in a Finnish site. In contrast, an orifice-weir constructed in North Carolina was reported to increase the water table level by 0.07 m (Amatya et al., 2003).

4.3 Sediment load generation after DNM

Numerous studies have reported high SS concentrations immediately after drainage operations (e.g., Hansen et al., 2013; Manninen, 1998; Robinson and Blyth, 1982). This was also evident in the data from Koivupuro (Stenberg et al., 2015b), and successfully reproduced by the model developed in Paper **III** (Figure 15a). The disturbed conditions in the ditches after DNM were described in the model as a freshly deposited sediment layer, which consolidated over a few days. This process had a key role in reproducing the recorded high SS concentrations after DNM. The importance of stabilization was also recognized by Marttila and Kløve (2008), who measured the erodibility of deposited peat after different stabilization times in a laboratory flume. However, even after stabilization, they observed an easily erodible fluff surface layer, which according to Mehta and McAnally (2008) has a thickness of a few floc diameters and does not consolidate easily. Describing such phenomenon in the model would require knowledge on the time and the conditions needed for such a layer to develop.

During spring snowmelt, simulated SS load originated from ditch bed erosion, which was driven by flow (Paper **III**). Evaluating the sediment transport model performance for the snowmelt period was hampered by the monitoring problems at the subcatchment V-notch weir (Stenberg et al., 2015b). Compared to measurements that can suffer from freezing episodes, resulting in unrealistically high runoff (Figure 8), modelling can plausibly provide more reliable SS load estimates for such critical periods. On the other hand, freezing and thawing cycles, which have been reported to loosen the peat structure (Evans and Warburton, 2007), may have a role in generating SS loads complicating its modelling during spring. In addition, the simulation of runoff during winter and spring may benefit from a more comprehensive description of soil freezing. Neglecting the effect of freezing on water flow in the soil may explain the poor representation of runoff during 8–11 May 2012 (see Figure 4a in Paper **III**).

The third studied period in Paper **III** comprised intense summer rainfalls featuring the positive hysteresis behaviour commonly reported for SS concentrations released from drained peatlands (e.g., Holden et al., 2007; Marttila and Kløve, 2010a; Tuukkanen et al., 2016). The positive hysteresis effect is usually associated with limited sediment supply, which is likely the case in more stabilized ditches (e.g., Holden et al., 2007). Nevertheless, if ditches have been recently excavated exposing bank surfaces (Figure 1c) as in the Koivupuro case, raindrop impact may be the dominant cause for the positive hysteresis effect as suggested by the modelling results (Paper **III**). Describing erosion by raindrop impact is common in overland flow descriptions (e.g., Wicks and Bathurst, 1996; Warsta et al., 2013b) and its role has also been recognized in experimental studies in peat extraction and forestry sites (Kløve, 1998; Tuukkanen et al., 2016). Including rain-induced erosion from channel banks in models is less common, but Paper **III** demonstrated its key role in reproducing SS concentrations during the year after DNM.

Bank erosion by raindrop impact can be expected to diminish within a couple of years as vegetation starts to cover the peat banks protecting them from erosion. Visual inspection of the Koivupuro ditch network in June 2013 already showed revegetation of the lower banks, close to the water surface. Holden et al. (2007) also reported that peat ditches revegetate more readily than ditches extending to mineral subsoil. Bank revegetation in deep peat sites may be the dominant cause for SS load decrease as years from DNM elapse. Compiled data from Finnish peatlands suggest that the excess load is the highest during the first post-treatment year, drops to a third of this during the second, and is then assumed to linearly decreases to the pre-drainage level within ten years (Finér et al., 2010). In addition to the modelled processes, the first year load is notably affected by the load generated during the DNM operations (Tuukkanen et al., 2016). Also, the disturbed conditions after DNM described in the model could contribute to exported SS loads more significantly than in the Koivupuro case if DNM was followed by high runoff events.

In addition to recognizing the importance of various sediment transport processes during different times, modelling in Paper **III** enabled the inspection of the sediment budget within the catchment (Figure 17). Such inspection based solely on measurements can be difficult as they can only cover limited sections of the investigated system (Evans and Warburton, 2005). For example, Stenberg et al. (2015b) reported that the locally measured net erosion was 10–20 times larger than the released SS load, highlighting that local measurements can overestimate loads significantly. According to the simulations, bed erosion concentrated in distinct ditch sections, where flow velocity was highest (Figure 13c). This is analogous to the findings of Holden et al. (2007) who reported the bottom slope and the contributing catchment area regulated the amount of erosion. Bank erosion, markedly increasing sediment supply (Figure 17), was spread out evenly as there were no major differences in the exposed ditch bank area. Most of the sediment released by raindrop impact was, however, deposited on the ditch beds (Figure 17), which is in line with the reasoning by Stenberg et al. (2015b). An extended area in favour of deposition was identified in the downstream end of the catchment, where the V-notch weir ponded water (Figure 13a).

Setting the simulation results (Paper **III**) against the spatially measured bed elevation changes proved to be challenging (Figure 16). The spatial and temporal variability of the erosion pin measurements was large, indicating they may reflect local changes caused by small-scale pooling, rills and obstacles fallen into the ditch. The model was not designed to capture such effects as it employed a 10-m distance step. In addition, the bank erosion rates (in kg a^{-1}) derived from the pin meter and erosion pin measurements were an order of magnitude higher than indicated by the sediment collectors and the model results. Altogether, this raises concern about the deceptiveness of measuring erosion or deposition based on topographical changes of a fluff, mostly fully saturated peat surface. Firstly, it is difficult to define the peat surface especially below the ditch water level (Mehta and McAnally, 2008), secondly the surface level is affected, e.g., by the shrinkage–swelling behaviour of peat (e.g., Camporese et al., 2006; Päivänen, 1982), and finally, in case of erosion pins, frost episodes may cause the pins to heave (Evans and Warburton, 2005).

4.4 Sediment control after DNM

In Paper **IV**, the model tailored for sediment transport after DNM was applied to assess the efficiency of alternative sediment control practices derived from forestry guidelines (e.g., Vanhatalo et al., 2015). This showed the true and yet unexplored potential of modelling as it enabled the investigation of numerous alternatives under the same conditions. Such comprehensive comparison would be hard to carry out based on an experimental setup, which is the plausible reason for previous research focusing on individual clearly definable structures (e.g., Amatya et al., 2003; Joensuu et al., 1999). In addition to the novelty value related to the overall comparison of scenarios, modelling provided understanding of the underlying processes controlling the efficiency of different structures, which is a necessity for further development of sediment control.

The simulations revealed that on a deep peat site (ditches in peat), sedimentation ponds and pits are ineffective (Paper **IV**), which is concerning as they represent the most widely applied structures in operational peatland forestry in Finland. It is worth noting, however, that the 1-D flow representation in the modelled ditches may overly simplify the deposition processes in large sedimentation ponds (Mohammadighavam et al., 2015). Still, the obtained poor performance of sedimentation ponds was in line with earlier research from deep peat sites (Joensuu et al., 1999; Manninen, 1998). The efficiency of pits in peat ditches has not been addressed earlier, and if they can even increase erosion as proposed by the simulations, their routine implementation alongside DNM should be reconsidered.

A noteworthy observation made in Paper **IV** was that the efficiency of the V-notch was superior compared to that of PRC during the autumn of 2011. This was recognized to be caused by the sensitivity of the ditches to erosion immediately after DNM at lower flow rates, which were not efficiently regulated by PRC. Either temporarily sealing the lower pipe of the structure, or raising it to the same level as the V-notch (about 0.3 m), could enhance the efficiency of PRC during this critical period. Poor performance of PRC immediately after DNM has not been reported in earlier experimental studies. However, other common features with earlier studies were observed, including the prolonged discharge hydrographs (Amatya et al., 2003; Marttila et al., 2010), increased deposition (Marttila and Kløve, 2010b), and the highest reductions occurring during snowmelt (Kløve, 2000).

Overall, the simulated scenarios with structures protecting ditches from erosion (PRC, breaks in cleaning, submerged weir) reduced SS load more efficiently compared to scenarios with sedimentation ponds and pits (Paper **IV**). Regarding these efficiencies, it should be noted that the model assumed that the submerged weir is fully watertight and unaffected by erosion, and that water in the PRC pipe cannot freeze (e.g., Marttila et al., 2010). Nevertheless, the results clearly indicate that sediment control on deep peat sites should rather focus on protecting ditches from erosion than on structures that aim to trap already eroded material. Implementation of PRC, breaks in cleaning, and submerged weirs in practice is, however, affected by the concern of compromised drainage efficiency, the planning effort, as well as the unavailability of the needed pipes or stones on site. As discussed in Section 4.2, compromised drainage was no concern in the studied subcatchment based on the simulation results of Paper **IV**. Planning of PRC may be argued to include most uncertainties as the dimensioning is based on a theoretical rating curve, which does not account for, e.g., possible ponding downstream of the structure. Furthermore, the ponding extent of PRC should be assessed to identify whether it covers erosion sensitive ditch sections, which identification is also key in the implementation of breaks in cleaning and submerged weirs. To facilitate practical planning, erosion risk maps generated by RL-GIS (Lauri and Virtanen, 2002; Tuukkanen et al., 2012) are available for operational use. Altogether, breaks in cleaning could be recommended as easily applicable and efficient structures targeting erosion. However, their impact depends on the condition of the ditches before DNM, e.g., ditch depth and vegetation cover.

Compared to bed erosion occurring in distinct ditch sections, rain-induced erosion from banks, identified as an important process in Paper **III** taking place over the entire ditch network, is more difficult to control. Considering current DNM practices resulting in large exposed bank surfaces (Figure 1c), loads originating from bank erosion can only be managed by trapping eroded sediments. Sedimentation ponds and pits were, however, shown to be ineffective in Paper **IV**, thus leaving only the option of a wetland buffer. Wetland buffers are expected to function efficiently (e.g., Nieminen et al., 2005; Sallantaus et al., 1998), but they are unsuitable for application in many cases (see Section 1.1.4). Controlling SS loads originating from bank erosion hereby calls for approaches other than the ones recommended in current forestry guidelines (e.g., Vanhatalo et al., 2015).

4.5 Future perspectives

Peatland drainage has adverse environmental impacts including, e.g., deteriorated ecological status of receiving watercourses (e.g., Kauppila et al., 2016) and altered carbon balance (e.g., von Arnold et al., 2005). On the other hand, peatland drainage is an important component of, e.g., forestry, agriculture, and horticulture (Holden et al., 2004). The diverse harmful as well as beneficial consequences of drainage are essentially driven by the changes in the hydrological behaviour of the site. Distributed hydrological modelling therefore has a great potential in this context, especially as water table depth and its spatiotemporal variability often plays a key role (e.g., Hirano et al., 2012; Sarkkola et al., 2012). Paper **II** addressed the spatial patterns of simulated soil moisture in a drained peatland forest, but merely scratched the potential that distributed modelling possesses in disentangling the role of different factors. Foremost, this thesis demonstrated how such modelling, including process descriptions for ditch network flow and sediment transport, can provide useful information on sediment load generation and control after DNM on a forestry site (Papers **III**–**IV**).

In Paper **II**, the spatial aspects of the input data were limited to the topography and the configuration of the ditch network. The spatial distribution of other factors such as stand characteristics, soil properties, and microtopography were disregarded. Investigating the role of these factors in regulating local soil moisture over time could provide understanding on the need for DNM in different situations. Such an assessment would, however, require an experimental setup where the hydrological behaviour of the site is measured in space (e.g., Haahti et al., 2012), instead of monitoring, e.g., only at the catchment outlet. A model describing the tree stand distribution could further be applied to evaluate alternatives for the currently dominating harvest by clear-cut. For example, conditions for natural regeneration (e.g., Hökkä et al., 2011b), as well as the need for DNM, when harvesting only in small patches are questions that future modelling applications could address. The effect of management options affecting local water table depth could furthermore be evaluated from the point of view of greenhouse gas emissions. Related descriptions of carbon and nitrogen cycles have already been implemented and assessed in 1-D modelling of peatland forests (He et al., 2016).

Modelling experiments in this thesis concentrated on a 5.2 ha drained peatland forest site. The small catchment size and the depth of its peat layer kept the SS loads consequent to DNM lower than the average values reported from Finnish sites (Finér et al., 2010). Drainage areas, where the ditches extend to the fine-textured mineral subsoil, have been recognized to export significantly higher SS loads (e.g., Joensuu et al., 2012b). Due to peat subsidence, the increasing number of forthcoming DNM operations resulting in mineral subsoil contact in ditches has raised concern (Nieminen et al., 2017a). Hereby, an essential future need would be to extend the model descriptions to cover mineral subsoil erosion and transport, which would require the representation of multiple sediment classes in the model (e.g., Lin and Wu, 2013). The process description included in the model in Paper **III** may be applicable e.g. for clay, but as high organic matter content in SS exports has also been observed in sites with shallow peat (Marttila and Kløve, 2010a; Tuukkanen et al., 2016), considering clay alone would not be sufficient. In addition to describing SS load generation, accounting for its composition would be essential as organic matter has more adverse impacts on receiving watercourses compared to mineral particles (Paavilainen and Päivänen, 1995).

The model-based evaluation of sediment control practices in Paper **IV** produced useful information for operational forestry on deep peat. However, as sites with mineral subsoil contact pose a higher erosion risk (e.g., Joensuu et al., 2012b), extending the model descriptions and investigations to such sites would have broader implications to operational level practices. The simulations in Paper **IV** further identified the need for novel methods to control the SS load originating from ditch banks by raindrop impact. Supposedly this applies to shallow peat sites, too, as there banks have also been recognized to supply organic sediment to the system (Tuukkanen et al., 2016). One possibility would be to adopt practices from peat harvesting where ponds are equipped with barrier structures to increase settling (e.g., Kløve, 1997; Mohammadighavam et al., 2015). Also alternative methods for DNM, disturbing the banks as little as possible, should be further assessed (e.g., Hansen et al., 2013).

Finally, as shown in Paper **IV**, structures implemented in ditches alongside DNM can only limitedly reduce the SS load. Hereby, it is highly important to avoid DNM when it is unnecessary due to high stand evapotranspiration (Sarkkola et al., 2012, 2010) or unprofitable due to poor site productivity (Ahtikoski et al., 2012). Furthermore, it can be hypothesized that, due to peat subsidence and consequent decrease in peat hydraulic conductivity (e.g., Silins and Rothwell, 1998), improving drainage by DNM may become more difficult to achieve in old drainage areas. The tree growth response, as well as the water quality impacts (Nieminen et al., 2017a), of forthcoming DNM operations may thus be difficult to deduce from earlier experiences highlighting the necessity of modeling applications. This, together with the pressure for forest biomass production, underlines the need to critically assess future forestry management practices on drained peatlands.

5 Conclusions

The primary objective of draining peatlands in Fennoscandia has been to increase forest productivity. Since the extensive drainage of pristine peatlands in the 1960s–1970s focusing strongly on increasing forest productivity, it has been recognized that forestry practice on peatlands needs to find a balance between productivity and environmental values. In present-day operational forestry, considerable attention is given to the mitigation of harmful water quality impacts of forestry operations, especially regarding DNM. The management of Fennoscandian drained peatlands is, nevertheless, still driven by the fundamental objective of productive forestry practice, thereby underlining the importance of the link between drainage schemes and soil moisture conditions. This thesis is the first study to present integrated modelling of ditch network flow and spatial soil moisture conditions in a forestry-drained peatland, providing advanced tools for peatland forestry research. In this thesis, the presented model, describing the essential hydrological processes in a drained peatland forest, was further extended to describe post-DNM erosion and sediment transport. The resulting model provided a novel approach to address the topical issue of sediment load generation and control after DNM in forested peatlands, responding to research needs that purely experimental studies have not been able to address previously.

Model development and applications to a 5.2 ha drained forest on deep peat presented in this thesis led to the following major conclusions:

1) Tailoring a network flow routing model for a forestry ditch network required special attention regarding flow resistance as Manning's *n* could not be described as a constant literature-based value.
2) Integration of the ditch network flow model into a distributed soil hydrological model showed potential to disentangle the role of spatial site characteristics and structures (e.g., weirs) implemented in the ditches, on the local soil moisture conditions. Hereby, the integrated model enables the impact assessment of forestry management operations, which induce changes to these spatial features within the site.
3) Extending the model to describe sediment processes in the ditch network, including bed erosion, rain-induced bank erosion, floc deposition, and consolidation of the bed, provided a useful approach to identify the role of different processes, as well as to support and complement existing knowledge gained from experimental studies.

4) Modelling sediment processes after DNM highlighted the role of erosion of loose material immediately after the excavation, as well as the role of raindrop impact in eroding and supplying sediment from the ditch banks to be transported in the network during rainstorms.
5) Simulations of sediment control scenarios, featuring individual structures and their combinations, suggested that on deep peat forestry sites the focus should be on structures decreasing erosion (e.g., breaks in cleaning and PRC) rather than structures aiming to trap already eroded material (e.g., sedimentation ponds). Especially sedimentation pits were found ineffective or even harmful as they occasionally increased erosion.
6) Based on the simulations, sediment control structures raising ditch water level (e.g., PRC) had marginal effects on water table levels during a rainy year in the peatland forest strips with poorly conductive subsoil between ditches, implying that drainage conditions and tree growth would not be compromised.
7) While the modelling applications in this thesis concentrated on a forestry site on deep peat, a future need for similar investigations in sites with shallow peat was identified. Sites where ditches extend to the mineral subsoil have a higher risk of erosion, and due to peat subsidence their prevalence may increase if current DNM practices are not revised.

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