Assessing the effects of subsurface drainage on hydrology and nitrogen transport in Nordic fields

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Abstract

Subsurface drainage is the primary water management approach in field cultivation in Nordic areas. Installing and improving a subsurface drainage system change water flow dynamics and routes in the soil, which affect nitrogen (N) load from the field. The role of soil properties, drainage system improvements and surrounding areas in the formation of water flow routes is not fully understood. The objective of this study was to quantify the effects of subsurface drainage on water flow and N transport using field monitoring data, statistical analysis and mathematical modelling.

The performance of two drainage installation methods applied in the Sievi experimental field were investigated with statistical analysis. Differences in groundwater level occurred due to drainage installation method and soil type at the drain depth, but the absolute differences were small (0.1 m). A state-of-the-art process-based hydrological model was applied to investigate the effects of soil properties and drainage systems on water flow routes. Field subsurface drainage schemes were simulated with 3D, 2D and 1D model applications using data from a clayey field in southern Finland (Nummela). Model applications showed how field drainage can be described with models of different dimensions and scale (from drain spacing to field section scale). The 3D drain spacing simulations demonstrated the benefits of using detailed soil data in model parameterization as an alternative to model calibration. In the 2D long-term simulations, the 3D soil parameterization was up-scaled to field section scale. The short-term 3D model simulations showed the dominant nature of soil macropores over the drainage system description. Comparison of the long-term 2D model simulations revealed that the improved drainage installations changed the shares of all the water flow routes, including groundwater outflow. A generic solute transport component was developed and tailored to describe N cycle transport and processes in 1D model simulations. Autumn period simulations of a poorly and well drained field sections showed that nitrate N loading was mainly controlled by the initial soil N storages after harvest and the timing of the precipitation events, while the soil moisture content differences explained the magnitudes of gaseous N losses.

Long-term monitoring data series, statistical analysis and process-based modeling showed that the practical effects of subsurface drainage are site specific and comprehensive view on the local water and nutrient management is needed when controlling the environmental impacts of field cultivation. The 1D, 2D and 3D model applications could all be used to replicate the measured drain discharge data, even though the drainage system description differed between the cases due to the differences in water flow directions and the boundaries of the simulated domain. The finding of the research suggests that field water management moves the N load from one path to another rather than affecting to the total amount of the water volume or N loading.

Keywords process-based modeling, statistical analysis, macropores, drainage installation, drain trench, envelope material, agriculture, clay

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Kahden salajoustusmenetelmän toimintaa Sievin köyhepollolla tutkittiin tilastollisilla menetelmissä. Erot eri pohjavedenpinnan tasoissa johtuivat salajustusten tehostamista ja typpikuorman määrää.

Tutkimuksen tavoitteena oli selvittää kuivatustäjästejä merkitystä veden virtaukseen ja tyyppeihin kulkeutumiseen ja muiden hallintamenetelmien, tilastollisten menetelmien sekä matematiikan mallintamisen avulla.

When I started my studies in the Helsinki University of Technology in 2008, I had no idea that after 11 years I would be defending my Doctoral thesis. I am grateful for the opportunity to do my doctoral studies in Aalto University School of Engineering during 2014–2019. The thesis was founded by School of Engineering, Drainage Foundation sr, Maa- ja vesitekniikan tuki ry, and Sven Hallin research foundation sr. My doctoral research was part of the following projects: Salaojitustekniikat ja pellon vesitalouden optimointi (PVO2), Toimivat salaojitustenel-mät kasvintuotannossa (TOSKA), and Vesitalouden hallinta vesien suojelussa (VesiHave). These projects were led by the Finnish Field Drainage Association and founded by Maa- ja Vesitekniikan tuki ry, Drainage Foundation sr, and Ministry of Agriculture and Forestry. I am grateful for the International Commission on Irrigation and Drainage (ICID) for funding my participation in the Young Professionals training program. I also wish to acknowledge the CSC – IT Center for Science, Finland, for the allocation of computational resources.

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Espoo, September 2019

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Appendix A Maps of the study fields
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:


Papers I–IV are reprinted with permission and copyrighted as follows:

Paper I: © 2018 Elsevier B.V.  
(https://doi.org/10.1016/j.agwat.2018.12.010)

Paper II: © 2016 Elsevier B.V.  

Paper III: © 2019 Elsevier B.V.  
(https://doi.org/10.1016/j.agwat.2019.03.039)

Paper IV: © 2015 Taylor & Francis Group  
(https://doi.org/10.1080/09064710.2014.971861)
Author’s contribution

Paper I  The author was responsible for conducting data processing and analysis. The statistical analysis and tests were done in co-operation with LicPolSc, MSc Mellin. The author was mainly responsible of writing the manuscript. LicPolSc, MSc Mellin and Prof Koivusalo helped with the presenting and interpreting the results. MSc Sikkilä was mainly responsible for the experiment and gathering the data. All the authors participated in interpreting the study results and writing the manuscript.

Paper II  The author was mainly responsible for the study design, additional model development, conducting the model simulations, interpreting the simulation results and writing the manuscript. Dr. Warsta provided guidance on the development of additional components for FLUSH and ideas for the study. MSc Nurminen and MSc Myllys were responsible for gathering the data. All the authors participated in interpreting the study results and writing the manuscript.

Paper III  The author contributed in the study design and instructed the modeling work. Dr. Turunen participated in the study design and instruction work. In co-operation with MSc Häggblom, the author developed a method for using structured computational grids with irregular cell dimensions, which were built outside the model. MSc Häggblom was responsible of setting up the model and conducting the simulations. MSc Nurminen and MSc Myllys were responsible for gathering the data. All the authors participated in interpreting the study results and writing the manuscript.

Paper IV  The author was mainly responsible for the study design, solute transport model development, conducting model simulations, interpreting the simulation results and writing the manuscript. Dr. Warsta provided guidance with the development of the solute transport component into FLUSH. Dr. Turunen provided guidance on the model parameterization for the Nummela field. MSc Nurminen was responsible for gathering the data. All the authors participated in interpreting the study results and writing the manuscript.
**List of abbreviations**

<table>
<thead>
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<th>Abbreviation</th>
<th>Full Form</th>
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<tr>
<td>1D</td>
<td>One-dimensional</td>
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<tr>
<td>2D</td>
<td>Two-dimensional</td>
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<tr>
<td>3D</td>
<td>Three-dimensional</td>
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<tr>
<td>ADE</td>
<td>Advection–dispersion equation</td>
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<tr>
<td>DP</td>
<td>Dual-porosity</td>
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<tr>
<td>DUP</td>
<td>Dual-permeability</td>
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<tr>
<td>FFA</td>
<td>Finnish Food Authority</td>
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<tr>
<td>GSF</td>
<td>Geological Survey of Finland</td>
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<tr>
<td>GWD</td>
<td>Groundwater depth</td>
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<tr>
<td>GWL</td>
<td>Groundwater level</td>
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<tr>
<td>Lo</td>
<td>Loam</td>
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<tr>
<td>LS</td>
<td>Loamy sand</td>
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<td>LUKE</td>
<td>Natural Resources Institute Finland</td>
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<tr>
<td>MAD</td>
<td>Mean absolute deviation</td>
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<tr>
<td>MAE</td>
<td>Mean absolute error</td>
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<tr>
<td>N</td>
<td>Nitrogen</td>
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<tr>
<td>NH₄-N</td>
<td>Ammonium nitrogen</td>
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<tr>
<td>NO₃-N</td>
<td>Nitrate nitrogen</td>
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<tr>
<td>NS</td>
<td>Nash-Sutcliffe</td>
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<tr>
<td>Org. N</td>
<td>Organic nitrogen</td>
</tr>
<tr>
<td>PET</td>
<td>Potential evapotranspiration</td>
</tr>
<tr>
<td>SL</td>
<td>Sandy loam</td>
</tr>
<tr>
<td>SOR</td>
<td>Successive Over-Relaxation</td>
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<tr>
<td>SP</td>
<td>Single pore</td>
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<tr>
<td>T₀</td>
<td>Trenchless</td>
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<td>T₁</td>
<td>Trencher</td>
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<tr>
<td>WRC</td>
<td>Water retention curve</td>
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</table>
List of symbols

\(\alpha\) \[L^{-1}\] Van Genuchten WRC parameter
\(\Omega_S\) \[L\] Subsurface drainage entrance resistant
\(\mu\) \[T^{-1}\] Decay coefficient
\(n\) [-] Van Genuchten WRC parameter
\(\theta\) \[L^2 L^3\] Soil moisture content
\(\theta_s\) \[L^3 L^3\] Saturated soil moisture content / Soil porosity
\(\Psi_W\) \[L^{-1} T^{-1}\] Water exchange coefficient
\(B\) [-] Moisture restriction parameter
\(d_S\) \[L\] Subsurface drain diameter
\(h_{W,THR}\) \[L\] Overland flow threshold
\(K_{FS,MUL}\) \[L T^{-1}\] Lateral saturated hydraulic conductivity multiplier (macropore)
\(K_{sat}\) \[L T^{-1}\] Saturated hydraulic conductivity
\(K_{sat,F}\) \[L T^{-1}\] Saturated hydraulic conductivity in macropore system
\(K_{sat,M}\) \[L T^{-1}\] Saturated hydraulic conductivity in soil matrix system
\(t_{bas}\) \[°C\] Parameter of temperature restriction function
\(t_{q10}\) \[°C\] Parameter of temperature restriction function
\(T\) \[°C\] Soil temperature
\(w\) [-] Fraction of the soil macropores
1 Introduction

1.1 Agriculture in Finland

Challenges for agriculture in high latitudes

In Finland and elsewhere in the Nordic region, agriculture is challenged by short growing periods, occasional lack of water during growing seasons, wet conditions outside growing seasons, and sediment and nutrient loads into receiving waters. To avoid the problems associated with the short growing season, field cultivation is started as early as possible in the spring and harvest is done as late in the autumn as possible. In high latitudes, snowmelt and soil frost in the spring typically determines the beginning of the cultivation period, and rains in the autumn influence the time of the harvesting.

In Nordic conditions, precipitation is often not enough to fulfill the crop water demand (e.g. Peltonen-Sainio et al., 2016). Annual precipitation exceeds annual evapotranspiration (e.g. De Schepper et al., 2017; Koivusalo et al., 2017; Turunen et al., 2015b); but during growing periods precipitation volume is less than evapotranspiration (e.g. Wesström et al., 2014; Jin and Sands, 2003). On the other hand, the lack of crop water uptake during spring snowmelt and autumn rains results in high runoff volumes. Climate change is expected to increase weather variability between years (e.g. Ruosteenoja et al., 2016; Øygarden et al., 2014), and farmers have to adapt to warmer summers, milder winters, and more common occurrences of weather extremes. These phenomena have been observed more frequently during the past decade (e.g. Kivinen et al., 2017; Lehtonen et al., 2014) together with increasing winter runoff (e.g. Rankinen et al., 2016).

Most of the environmental load (including sediment and nutrient leaching) in high latitude areas occurs outside the growing season (e.g. Øygarden et al., 2014; Vagstad et al., 2004). Agricultural lands are the main sources of a diffuse nutrient load (e.g. Rankinen et al., 2014; Stålnacke et al., 2014; Vuorenmaa et al., 2002), and nutrient export from agricultural areas to surface waters is likely to continue in the form of background load (e.g. Finér et al., 2010), even when agricultural practices in that location are terminated. For example, Rankinen et al. (2016) found that large scale land use changes did not explain trends in N flux. One of the key drivers for nutrient load is climatic conditions (e.g. Rankinen et al., 2016; Huttunen et al., 2015; Stålnacke et al., 2014); and in the Nordic areas, seasonal snowmelt has the largest influence on solute transport (Frey et al., 2012a). There have been attempts to control the loads—for example with biofilters to treat the outflowing field waters (Bell et al., 2015; Brandstaka et al., 2010)—in order to move towards more
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sustainable food production. A comprehensive view on the leaching pathways is needed to assess the overall effect of such control measures. A juxtaposition exists between the need to decrease the detrimental effects of agricultural production and the need to increase the productivity of field cultivation to feed the growing population (e.g. Ayars and Evans, 2015; Huttunen et al., 2015).

Characteristics of clay soils

Finland has 2.5 million hectares (Mha) of agricultural fields (7% of the total area), and approximately 1.1 Mha (40% of the field area) are on clayey soils (FFA, 2016; GSF, 2016). Generally, clay soil matrix conducts water poorly, i.e. its hydraulic conductivity is low. However, the effective hydraulic conductivity of clay soils depends on the amount of larger soil pores, which can conduct water very efficiently (Mualem, 1976; Beven and German, 1982). In the previous studies, the diameter of the soil pore is often used to categorize them into micropores and macropores (e.g. Gerke, 2006). The threshold diameter for macropores is usually set to 30 μm or macropores are defined as pores that empty in a suction of 0.1 m (Jarvis, 2007). The soil medium that comprises the smaller pore sizes is called the soil matrix. Only a small fraction of the total porosity is composed of macropores (e.g. Guo and Lin, 2018; Alakukku et al., 2010), but this volume facilitates the phenomena called preferential water flow and solute transport. Water and solutes move in the macropore network, bypassing the soil matrix and enabling rapid flow and transport from the soil surface to deeper layers in the soil. This process has been shown to be accountable for a major part of water outflow and nutrient leaching in clayey soils (e.g. Frey et al., 2016).

Laboratory scale studies on preferential flow and transport (e.g. Akay et al., 2008; Jarvis et al., 2008) have demonstrated the theoretical effects of macropore networks on vertical water flow and solute transport in isolated soil columns and monoliths. However, generalization of the laboratory results to larger spatial areas has not been straightforward. Field experiments have mostly focused on the spatial occurrence of macropores in the field (e.g. Alakukku et al., 2010; Shipitalo et al., 2004), but the effects of spatial variability of macroporosity on water flow and solute transport are site specific (e.g. Frey et al., 2012c) and comprehensive understanding remains practically unsolved (e.g. Vereecken et al., 2016) and warrants more research (Frey and Rudolph, 2011). It has been shown that macropores are causing anisotropy to the hydraulic conductivity of the soil because the networks can be more vertically or horizontally oriented (Birisso et al., 2013). The occurrence of macropores varies spatially, but also vertically between the different soil layers. Limited evidence is available on the existence of disconnected or vertical macropores (e.g. Klaus and Zehe, 2010; van Schaik et al., 2010); but in general, models describing macropore flow (see Section 1.5) assume that the macropore network is fully connected in all directions (Turunen et al. 2017; Jarvis et al., 2016).

The variability of macropores in the vertical direction of the soil profile creates differences in the hydraulic and structural properties of the different soil layers (topsoil, plow pan, subsoil, and bottom soil). Macroporosity and consequently hydraulic conductivity are generally higher in the topsoil layer (from soil surface to a depth of 0.2–0.3 m) (e.g. Shipitalo et al., 2004). This is due to different aggregate
producing mechanisms such as soil drying and wetting, soil freezing (e.g. Rasa et al., 2009), and tillage operations (Turtola et al., 2007). Also, plant roots and earthworm burrows are most abundant in the topsoil layer (Alakukku et al., 2010). Beneath the top soil layer there is typically a compacted plow pan layer (depth of 0.3–0.4 m) that conducts water poorly and works as a barrier for vertical water flow (e.g. Shipitalo et al., 2004; Turtola and Paajanen, 1995). The hydraulic conductivity in the plow pan layer can be up to 10 or 100 times lower than in the topsoil layer (Alakukku et al., 2010; Vakkilainen et al., 2010), which causes horizontal preferential flow in the topsoil layer. Hydraulic conductivity in the subsoil (depth of 0.4–1.0 m) is usually lower than in the topsoil layer, but higher than in the plow pan. For the field scale agricultural studies, there is not much evidence on the structural or hydraulic properties of the deep bottom soils below 1 m depth (Yli-Halla et al., 2009).

Field experiments

The effects of agricultural practices on water flow and environmental loads have been studied in several field experiments (e.g. Frey et al., 2012a; van der Velde et al., 2010). The studied field can be divided into field sections that are then treated differently, for example in terms of the drainage (e.g. Wessström et al., 2014). The effects of the treatment are then assessed by comparing and analyzing the data collected from the field sections. The monitoring data includes weather conditions (air temperature and precipitation), point (groundwater depth or soil moisture), and aggregated spatial (discharge or runoff) output variables. Both short-term (1–5 years) and long-term (5–10 years) campaigns have been used to analyze the effects of the treatments, such as drainage installations (e.g. Mirjan and Kanwar, 1992), tillage (e.g. Turtola et al., 2007), or improvements (e.g. Turtola and Paajanen, 1995).

Long-term monitoring campaigns on experimental fields produce valuable data that can be used to separate the effects of field treatments (such as a drainage system installation or change in the crop type) from other factors such as hydrological conditions (e.g. Koivusalo et al., 2017; Puustinen et al., 2005). Many monitoring studies have reported that it was possible to discern the effects of the studied factors such as drainage installation method (e.g. Kanwar et al., 1986) or improved drainage (Turtola and Paajanen, 1995) on water flow and nutrient leaching only a couple of years after the start of the experiment. Also, to study the effects of several factors such as precipitation patterns and soil warming on nitrogen processes (see Section 1.4), longer monitoring periods are needed in order to identify the long-term effects of the field treatments from the cultivation conditions (e.g. Patil et al., 2010).

Field experiments have an important role in answering questions about the effects of agricultural practices in situ. Each research site has individual properties that should be taken into account when interpreting and applying results collected from a particular field (e.g. Gramlich et al., 2018). Intensive site monitoring should be maintained and extended to identify how variability in the data series is linked to field activities and processes in water flow and nutrient transport (Kyllmar et al., 2014). The problem with short-term field experiments is that the effect of the investigated treatment, such as drainage installation, can be mixed or hidden by the
operational work in the field, such as soil digging. It can take a few years for the field to recover from the disturbance caused by the initialization work of the field experiment (e.g. Äijö et al., 2014) and to gain a meaningful assessment of the treatment effects.

1.2 Field drainage

Field drainage history and need in high latitudes

Field drainage is the primary water management practice in Finland. At first, local field drainage was implemented with open channel drains, and subsurface pipe drains became more popular after the 1950s (Saavalainen, 2001). Initially research focused on assessing the optimal drain spacing and depth to gain an adequate field drainage capacity in different soil types (e.g. Oosterbaan and Wind, 1978). Soon after subsurface drainage installation increased, the research was needed to decipher how installation conditions and the applied drainage machine affected the drainage system performance (Vakkilainen and Suortti-Suominen, 1982; Kanwar et al., 1986). Later studies have tried to solve the relationship between different drainage materials and drainage efficiency (Grismer et al., 1988; Stuyt, 1992; Sikkilä, 2014) as well as the hydrological effects of improved drainage practices (Turtola and Paajanen, 1995; Äijö et al., 2014, 2017). Understanding the effects of local drainage on discharge water quantity and quality is necessary when assessing areal loads, as the local field drainage systems are connected to a main ditch network, which in turn discharges into surface waters.

In high latitudes, field drainage enables suitable conditions for crop cultivation, especially in the spring after snowmelt and during autumn rains. Draining the soil also decreases surface runoff and erosion, prevents soil compaction, and can improve soil structure (Chamen et al., 2003), further avoiding the release of greenhouse gas emissions from the compacted soil (e.g. Vereecken et al., 2016). Consequently, efficiently functioning drainage can increase workability and fertility of the soil (e.g. Alakukku et al., 2003). Clayey soils with low matrix permeability need to be efficiently drained, usually with local subsurface drains, to enable crop cultivation in such areas. There is a risk that the soil bearing capacity is exceeded when heavy agricultural machines enter the field, especially in wet soils (e.g. Alakukku et al., 2003).

Field drainage implementation and maintenance

Most of the fields in Finland are currently subsurface drained (Field Drainage Association, 2016). Also, subsurface drainage systems are being planned or being installed for most of the remaining fields with open ditches (Field Drainage Association, 2017). A large fraction of the subsurface drains were installed in the latter half of the 1900s (Saavalainen, 2001), and they are aging and losing efficiency (e.g. Nousiainen et al. 2015). This is causing a growing need for improved or renewed drainage installations (Puustinen et al., 1994). Drainage improvements are done by installing new drains next to old ones and connecting the lines to existing collector pipes. Vakkilainen et al. (2010) and Turtola and Paajanen (1995) showed that improved drainage clearly decreased surface runoff in a heavy clay
field in the autumn period. Wesström et al. (2015) showed that repairing the existing drainage system decreased the subsurface seepage, as the water flow was captured by more efficiently functioning subsurface drains. The change in water outflow components does not give a straightforward answer to how an improved subsurface drainage system increases the efficiency of the resulting drainage system comprised of old and new parts. For example Äijö et al. (2017) noticed that the measured total runoff coefficient increased after improved drainage, but was still lower compared to other similarly drained fields.

1.3 Subsurface drainage and water flow

Effect of subsurface drainage on water flow pathways

Subsurface drainage influences water flow pathways on the field surface and below the surface. A large number of studies have investigated the effects of subsurface drainage on the distribution of runoff volume between drain discharge and surface runoff (e.g. Äijö et al., 2017; Vakkilainen et al., 2010; Jin and Sands, 2003; Turtola and Paajanen, 1995). Those two pathways are usually thought to form the total drainage volume (e.g. Turtola et al., 2007). However, Seuna and Kauppi (1981) already noticed that after subsurface drainage installation, the amount of drain discharge was greater than the reduced amount of topsoil layer runoff. This indicated that increased drainage volume originated from another source.

Several studies demonstrate groundwater outflow to be an important water flow component (e.g. Hansen et al., 2019; Rozemeijer et al., 2010), but it is rarely taken into account in field scale water balance studies (e.g. van der Velde et al., 2010). Turunen et al. (2013) reported that especially topography of the field and its surrounding areas affected the groundwater outflow volume. Wesström et al. (2015) noticed that there were additional flow pathways when they investigated why simulated drain discharge was overestimated compared to the measured discharge.

Factors affecting the performance and efficiency of subsurface drainage

The performance of field subsurface drainage depends on several factors such as drain spacing and depth, which have been extensively studied in the past (e.g. Singh et al., 2007; Breve et al., 1998), but also on the drainage installation method, which has been investigated in far fewer studies. The drainage installation method includes the applied drainage machinery, which can be roughly divided into trenchless and trencher machines, and the envelope material (e.g. gravel) around the pipe and the backfilling (e.g. dried topsoil) above the drain pipe in the trench. The performance differences between drainage systems installed with the two machines are ambiguous (e.g. Vakkilainen et al., 2010; Mirjat and Kanwar, 1992; Kanwar et al., 1986).

The trenchless drainage installation method is usually faster and cheaper, as it does not require soil excavation: The soil is temporarily lifted up or pushed aside with a plough blade while the drain pipe is placed to the designated installation depth (e.g. Ritzema et al., 2006). However, the installation conditions are more flexible for the trencher installation method because the trench excavation and the drain pipe and envelope material insertion are done separately from backfilling
(e.g. Nijland et al., 2005). When backfilling is done with the soil from the trench, it can be piled and allowed to dry before the backfilling, which can increase the conductivity of the soil in the trench. In addition to the drainage installation method, different drainage machines can lead to varying drainage performance.

The functioning of subsurface drainage is not only dependent on the system design (spacing, depth, properties of drain pipe and envelope, and installation method), but also on the surrounding soil characteristics. Interestingly, the role of different spatial features in the soil (like preferential flow pathways and gravel envelopes in drain trenches) have rarely been studied in detail. Messing and Wesström (2007) noticed a large spatial variability in terms of macropore connectivity that was superior above the drain line compared to the midpoint between the drain lines. Akay et al. (2008) reported that there was a relationship between drain flow initiation time and the distance of an artificial macropore from the subsurface drain. Alakukku et al. (2010) found that vertical distance from the drain (rather than the horizontal distance) explained the occurrence of macropores.

Subsurface drains do not influence water flow routes beneath the drain depth, but flow routes beneath the drains have an effect on drain discharge volumes. Therefore, water flow pathways below the drainage depth, even several meters below the drains (Hansen et al., 2018), are important to consider to fully understand the effects of subsurface drainage on field drainage performance and water outflow components.

The reasons behind the changes in drainage performance can be difficult to confirm (Nousiainen et al., 2015), as the changes can be due to multiple factors such as soil compaction (e.g. Rimidis and Dierickx, 2003) or pipe clogging (e.g. Sikkilä, 2014) that vary both in space and time. In fine textured soils, like clays and silts, drain pipes are prone to clogging, and the drain trenches might collapse or become obstructed with soil particles from the field surface (e.g. Sikkilä, 2014; Grimser et al., 1988). These processes separately or together can lead to the deterioration of drain performance and decrease the life time of the system (Äijö et al., 2014). Messing and Wesström (2007) reported that the cause for malfunctioning of a drainage system can be detected more accurately with head loss observations near the trench segment compared to observations just above the drain lines.

1.4 Nitrogen transport and leaching

Nitrogen cycle and leaching

Plant available nitrogen is a prerequisite for crop growth and field cultivation. Most of the soil N is in the form of organic N, which has to be first transformed into inorganic N to be available for crop uptake. The decomposition of organic matter depends on the soil temperature, which has led to arguments that N leaching might increase due to increased air temperature in autumn (e.g. Rankinen et al., 2016; Patil et al., 2010) as a result of climate change.

Excess nitrogen in surface water bodies is harmful for the well-being of organisms, as it can increase the growth of aquatic plants and algae, which causes oxygen depletion and eutrophication (e.g. Mitikka and Ekholm, 2003; Cederwall and Elmgren, 1990). The N leaching is unwanted from the field cultivation viewpoint.
because the valuable inorganic N is washed away, decreasing the amount in the soil for plant use. In Finland, field cultivation is mainly located on the coast of the Baltic Sea in southern and western Finland, causing the main share of the N load to the sea (Rankinen et al., 2014). Fertilizers are commonly used in the beginning of the growing period to ensure adequate inorganic N storage and supply for crop growth. Before crop N uptake occurs, fertilizer N is stored in the soil near the soil surface and is prone to leaching in case of precipitation events. In agricultural areas, problems with an increased amount of inorganic nitrate N (NO$_3^-$-N) in groundwater sources have been noted (Saleem, 2018; Rozemeijer et al., 2010).

Water flow in the field is the key driver for nitrogen (N) transport and leaching (e.g. Patil et al., 2010), as most of the N load is in the form of soluble NO$_3^-$-N (e.g. Seuna and Kauppi, 1981). In the absence of plant N uptake, N leaching is controlled by mineralization of organic nitrogen to ammonium N (NH$_4^-$-N), which is further transformed into NO$_3^-$-N by nitrification and the transport of NH$_4^-$-N and NO$_3^-$-N from the field via runoff. In Nordic conditions, nearly half of the yearly precipitation is recorded in autumn at the time of low potential evapotranspiration (PET) (September to December) (e.g. Pirinen et al., 2012).

**Studying field-scale N transport**

In the past decades, numerous studies have investigated the controlling factors behind N load from subsurface drained cultivated fields (e.g. Valkama et al., 2016; Wesström et al., 2014; Skaggs et al., 2006; Breve et al., 1998; Seuna and Kauppi, 1981). However, a quantitative description of the N processes in subsurface drained fields is still incomplete. Many studies have reported that subsurface drainage increases the N load (e.g. Youssef et al., 2018; Sunohara et al., 2016) and that the load can be reduced with water management practices regulating drain discharge during the growing periods. Controlled drainage and subirrigation have been found to reduce N leaching via subsurface drains (e.g. Wesström et al., 2014). Carstensen et al. (2016) could not verify on a field scale monitoring study the fate of soil profile N that was not discharged from the field due to application of controlled drainage in the area. Also, controlled drainage does not seem applicable in fields with high clay content or sloping fields (Paasonen-Kivekäs et al., 1996). For the sloping fields, the problem is the steep hydraulic gradient upslope of the control point (i.e. the water level control has an effect only at the immediate surroundings of the control location). The issue with high clay content fields is the low hydraulic conductivity, which weakens the horizontal water flow towards the drain pipe, causing more waterlogging problems in clay fields than in fields with coarser soil textures (e.g. sands). Hence other means for reducing N load are also needed.

Catchment scale studies provide the means to assess spatially large scale N processes (e.g. Huttunen et al., 2016; Stälnacke et al., 2014), but field scale studies provide a more controlled study environment to investigate the role of different flow pathways and processes governing N leaching in cultivated soils (e.g. Sunohara et al., 2016; Gärdenäs et al., 2006). In catchment and field scales, N leaching has been estimated with N balance calculations (e.g. Valkama et al., 2016; Salo and Turtola, 2007). However, N balance calculations are black box type approaches and do not give insight into the processes happening in the soil that affect N leaching.
Salo and Turtola (2007) noticed that the N balance could be used as an indicator for N leaching in some fields, but not in general.

Mathematical models can be used to simulate the processes occurring in the investigated natural system even though they are simplifications of the real system. Simulation models provide a quantitative system to investigate the effects of N processes on N leaching in different circumstances (e.g. Youssef et al., 2018). Field experiments can reveal knowledge gaps on processes that should be further studied with modeling, for example the combined effect of soil warming and precipitation patterns on N leaching (e.g. Patil et al., 2010). Likewise modeling can increase understanding of the experimental design and provide clues as to what kind of data should be gathered from the field (e.g. Nousiainen et al., 2015).

1.5 Modeling tools and data analysis

Roles of modeling and statistical analysis

Statistical methods have long been used to study field drainage performance and how drainage system design parameters affect field hydrology (e.g. Vakkilainen and Suortti-Suominen, 1982; Kanwar et al., 1986; Sikkilä, 2014; Tuohy et al., 2016). Use of mathematical models in hydrological studies have become more common as the availability of computers and models have increased in the latter part of the 20th century. Karvonen (1988) modeled 1D and 2D unsaturated water flow and the effects of drainage on soil moisture and crop yields. Also, 2D models were required to study the effects of field slope on subsurface drainage performance (Fipps and Skaggs, 1989). Aura (1995) were among the first to develop a 2D drainage model including spatial features to simulate the effect of gravel envelope and gravel inlets. Nowadays, state-of-the-art process-based models can simulate water flow and solute transport with one-, two-, or multiple soil pore systems in 1D, 2D, or 3D model applications (e.g. De Schepper et al., 2017; Frey et al., 2016; Turunen et al., 2013 Gärdenäes et al., 2006; Haws et al., 2005).

Field data have been an important part of model development. Mathematical models can be used to combine measurements into one quantitative system. The resulting system can be used to study the effects of different field treatments (e.g. drainage installation and improvements) on key hydrological variables such as drain discharge and groundwater level. Lately, models combined with field data have been used to explain field water balance and the role of preferential flow in clay soils (e.g. Frey et al., 2016; De Schepper, 2015), and to investigate small scale details such as the effects of pressurized pipe flow on field drainage (Henine et al., 2010), as well as to analyze the suitability of different modeling approaches in such settings (e.g. Gerke et al., 2007). Statistical data analysis of hydrological data series can facilitate detection of features (such as climate conditions) to be considered in the simulations (e.g. Rankinen et al., 2016). The hypothesis about the explanatory factors in a data set, such as soil type or drainage design, can be tested with statistical methods (e.g. Sunohara et al., 2016; Tuohy et al., 2016) before applying mathematical models to the problem.

Regional and national scale estimates of nutrient loads from field cultivation are produced with relatively simple, process-based models (e.g. Huttunen et al., 2016).
Data collected from experimental fields in long-term campaigns and knowledge gathered from field-scale modeling studies are critically needed to validate the results of these large-scale models (Hansen et al., 2013; van der Velde et al., 2010). The trend in Finland and other Nordic countries is that less effort is directed to long-term monitoring campaigns and more emphasis is on modeling. Still, modeling should be closely linked to monitoring, which can reduce uncertainties in both simulation results and recorded data (Kyllmar et al., 2014). Regional and national scale models have to be able to simulate reference fields reasonably well in order to produce reliable results in larger scales (see Section 1.1). Hansen et al. (2013) reported that the inclusion of small-scale processes (local subsurface field drainage) resulted in more accurate catchment scale results even though the simulation results did not correspond well with the individual field scale measurements.

Data needs for modeling

Field-scale models benefit from high-resolution data on soil characteristics and drainage systems that are not always available (e.g. Vereecken et al., 2016). Filipović et al. (2014) conducted numerical experiments with HYDRUS 2D/3D to simulate subsurface water flow in cultivated soils with different drainage scenarios. This approach can give insight into the theoretical effects of drainage, but the actual effects can be different if the underlying assumptions in the model are not valid. For example, when the effects of macropore network or the drain trench on water flow are not correctly described in the simulations, the model can produce biased results. Therefore, the functioning of the simulated drainage treatment should also be studied with field experiments to gain understanding on how the treatment functions in real world conditions. Traditionally, computational models are calibrated and validated with data, and when the model can reproduce both data sets reasonably well (e.g. Reilly and Harbaugh, 2004), it can be used to study hypothetical drainage scenarios.

A more complex model requires more information about the simulated system, and simpler models should be used in the absence of data (e.g. Gårdenäs et al., 2006). Applications of dual-permeability models need more knowledge about the soil structural and hydraulic properties, which is rarely available (Vereecken et al., 2016; Klaus and Zehe, 2010). In many modeling applications, these unknown model parameters have to be defined by calibrating the model against measured aggregated variables such as drain discharge data recorded during a specific time period (Varvaris et al., 2018). Assessing the model performance by matching simulated and measured hydrographs for a single outlet may not contain enough information of the studied system (e.g. Haws et al., 2005). Direct measurements of flow routes would be more useful than aggregated discharge measurements when calibrating spatially varying model parameters (Rozemeijer et al., 2010).

Modeling water flow and solute transport in structured soils

A common approach to simulate the effects of preferential flow and transport in structured soils is to divide the soil domain into soil matrix and macropore systems (e.g. Šimůnek and van Genuchten, 2008). In the dual-permeability (DUP) approach, water flow and solute movement occur in both pore systems and between
the pore systems. In the dual-porosity (DP) approach, flow and transport are restricted to the macropore system and the matrix acts as a sink and a source via water and mass exchange between the pore systems. The third option is to use a single porosity (SP) approach where water flow and solute transport occur in one continuum, and macropores are explicitly parameterized into the computational domain (e.g. Klaus and Zehe, 2010). In some models, water and solutes are instantaneously transported to a certain depth to simulate preferential flow and transport in macropores (e.g. Jansson, 2012; Christiansen et al., 2004). There are many transport models available (COUP, CROPWATN, Daisy and DRAINMOD-NII in Table 1), but these rarely describe bypass flow properties of preferential flow and transport (e.g. Christiansen et al., 2004).

Even though the DUP approach has been reported to work well in simulating water flow in clay fields (e.g. Turunen et al, 2013, 2015a), varying results have been reported in terms of drain discharge when using the SP, DP, or DUP approaches. Gärdenäs et al. (2006) noticed that the DP model produced drain discharge peaks that were 10 and 100 times higher than the ones simulated with the DUP and SP models, respectively. Simulations of Haws et al. (2005) showed higher drain discharge peaks for the SP model compared to the DUP model. Both Gärdenäs et al. (2006) and Haws et al. (2005) used HYDRUS 2D/3D for the simulations. Van der Laan (2014) pointed out that even when using the same model, the outcome can be highly user- and parameterization-dependent. Also, the parameterization of the reactive solute transport as well as characterization and parameterization of transport processes in soil macropores remain a challenge (e.g. Frey et al., 2016; Jarvis, 2007).

The adaptability of models used for simulating water flow and solute transport for site-specific research studies can be limited if they are commercial models (Mike-SHE, Hydrus package, HydroGeoSphere in Table 1), not open source (e.g. MACRO) or not developed for Nordic conditions with snow accumulation and melting descriptions (e.g. ParFlow). Open source models like FLUSH (Warsta et al., 2013) enable custom model development and application in different experimental conditions. Flexibility of the model gives freedom to add features that are specific for the site in question. For example, simulation of N processes in clayey soils can benefit if soil moisture and temperature variables are used in the solute reaction descriptions in both soil matrix and macropore systems.

Combined simulations of water flow and solute transport have been conducted with varying model complexities, including 1D (e.g. Rankinen et al., 2007; Jarvis, 1995), 2D (e.g. Gärdenäs et al., 2006), and 3D models (e.g. Frey et al., 2016). A problem especially associated with 1D models is that they cannot adequately describe the drain trench and surrounding soils. In clayey soils, the drain trenches are supposedly primary water flow routes due to permeable envelope and backfill materials, while the surrounding clay soil conducts water far less efficiently. 2D models describing a soil transect orthogonal to the drain lines can simulate the spatial heterogeneity between the subsurface drains, but 3D models are needed when simulating drain capacity differences within the drainage system (pipe clogging or exceeding of the pipe flow capacity).
Table 1. Mathematical, process-based models used for simulating water flow and solute transport in subsurface drained fields. In the Drainage column, D refers to the Darcy’s law, DE to the drainage equation, P to the pipe flow component, and S to seepage.

<table>
<thead>
<tr>
<th>Model</th>
<th>Transport</th>
<th>Dimension</th>
<th>Drainage</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>COUP</td>
<td>N</td>
<td>1D</td>
<td>DE</td>
<td>Jansson (2012)</td>
</tr>
<tr>
<td>CROPWATN</td>
<td>N</td>
<td>1D</td>
<td>DE</td>
<td>Karvonen and Kleemola (1995)</td>
</tr>
<tr>
<td>Daisy</td>
<td>N</td>
<td>1D</td>
<td>DE</td>
<td>Abraham and Hansen (2000)</td>
</tr>
<tr>
<td>DRAINMOD-NII</td>
<td>N</td>
<td>1D</td>
<td>DE</td>
<td>Youssef et al. (2005)</td>
</tr>
<tr>
<td>HydroGeoSphere</td>
<td>Generic</td>
<td>3D</td>
<td>D, P</td>
<td>Brunner and Simmons (2012)</td>
</tr>
<tr>
<td>Hydrus</td>
<td>Generic</td>
<td>3D</td>
<td>D</td>
<td>Simunek and van Genuchten (2008)</td>
</tr>
<tr>
<td>MACRO</td>
<td>Generic</td>
<td>1D</td>
<td>S</td>
<td>Jarvis and Larsbo (2012)</td>
</tr>
<tr>
<td>Mike-SHE</td>
<td>Generic</td>
<td>3D</td>
<td>D</td>
<td>Refsgaard and Storm (1995)</td>
</tr>
<tr>
<td>RZWQM</td>
<td>N</td>
<td>1D</td>
<td>DE</td>
<td>Ma et al. (2012)</td>
</tr>
<tr>
<td>ParFlow</td>
<td>Generic</td>
<td>3D</td>
<td>D</td>
<td>Maxwell et al. (2015)</td>
</tr>
<tr>
<td>SWAP</td>
<td>Generic</td>
<td>1D</td>
<td>DE</td>
<td>Kroes and Van Dam (2003)</td>
</tr>
</tbody>
</table>

*) N transport models include NO3-N, NH4-N transport and the related N cycle processes.

1.6 Research gap

The literature review (Sections 1.1–1.5) highlights research gaps concerning soil water flow and the effects of drainage systems and their installation method on water balance and nitrogen (N) processes in clayey, subsurface drained fields in Nordic conditions.

Long-term monitoring campaigns

Long-term monitoring campaigns produce data for investigating treatment effects on flow processes under varying hydrological conditions (e.g. Kyllmar et al., 2014; van der Velde et al., 2010). Statistical methods applied to long-term data provide a useful tool for finding anomalies and trends in the data and providing clues for further investigation with mathematical models. Long-term modeling studies combined with field data are rare, but useful in quantifying hydrological processes that are difficult to measure directly and consequently closing the water balance of the investigated area. The effects of improved drainage practices on field drainage efficiency and water balance in poorly drained fields have rarely been studied using process-based simulation models together with long-term monitoring data series (e.g. Turunen et al., 2013) that provide means to assess the influence of weather variability on the investigated field system.

Preferential flow

Many studies have found that the soil macropores and the preferential flow in structured soils have an important influence on the dynamics of drain discharge (Varvaris et al., 2018; Gärdenäs et al., 2006; Haws et al., 2005). The hydrological implications of spatial variability in preferential flow pathways in subsurface drained clayey fields have been studied to some extent (Frey et al., 2012c), but have not been adequately addressed in previous studies in Nordic conditions. The role of preferential flow in drain trenches that connect the field surface to subsurface drains and the overall effect of the preferential flow route on water balance remain unclear. Several factors including drain envelope material, trench backfill-, and the surrounding soil influence the flow in the trench.
The effect of drainage installation method on drainage performance

The performance of a drainage system is expected to depend on the soil characteristics and the applied drainage installation method (see sections 1.2 and 1.3). Identifying the role of these factors in drain discharge and water balance requires long-term monitoring campaigns where each installation method is investigated individually. The data needs to be collected systematically so that the differences can be tested with statistical analysis. The hydrological effects of drainage and soil can be explained by studying the data set with a mathematical model. In order to quantify and characterize real world behavior, physical processes, and model uncertainty, the most promising methodology seems to be comprised of monitoring data, statistical methods, and mathematical models.

Application of directly measured soil hydraulic properties

With an intensively monitored research field, where data are already available, it is possible to test the suitability of directly measured soil hydraulic properties (e.g. Klaus and Zehe, 2010) to evaluate model performance, bypassing the time-consuming calibration–validation procedure. The effect of spatial differences in soil properties on water flow and solute transport has been studied to a certain extent (e.g. Filipović et al., 2014; Klaus and Zehe, 2010; Gärdénäs et al., 2006; Vogel et al., 2000), but model applications with detailed soil parameterizations based on measured soil properties are rare (e.g. Vereecken et al., 2016). If directly measured soil hydraulic properties can be used to parameterize a model in an adequate way, it might provide a faster and cheaper way to parameterize a distributed mathematical model when collection of long-term runoff data for parameter calibration is not viable. It is not clear, how the modeling dimension affects the model results, i.e. if the same parameterization can be used in 1D, 2D, and 3D simulations.

N processes

Combinations of mathematical models and intensively monitored experimental sites are efficient tools for identifying and quantifying factors affecting nitrogen processes. To study N transport processes, it is necessary to adequately describe soil water flow dynamics and solute leaching pathways. There is a lack of modeling studies that take into account the effects of preferential flow pathways such as drain trenches on N leaching. While specific N models have been developed and tested in field-scale applications (e.g. Youssef et al., 2005), simulating reactive solute transport using a generic solute transport model has only recently been documented (Frey et al., 2016). The advantage of using a generic flow and transport code to simulate N processes is the possibility to choose between different flow and transport descriptions and include features such as a preferential flow and transport processes and drainage approaches in the simulations when necessary. Other important phenomena, such as heat transport and the effect of soil temperature on reactive transport, can be included when needed. Also, with a generic transport model, the reaction chains and solute adsorption behavior are not fixed but can be altered to suit the application. Generic flow and transport models thus provide flexibility for the modeler not necessarily available in specific N models.
1.7 Objectives

Based on the research gaps presented in Section 1.6, the thesis focuses on two main objectives with specific aims:

**Understanding the effects of field drainage installation methods and supplementary drainage on water flow and nitrogen transport**

1. Investigate how narrower drain spacing of a subsurface drainage system affects water balance (papers II–IV) and what the effect of drainage installation method is on field drainage performance (Paper I).
2. Study the effects of site specific features (drainage installation method, soil type and surrounding areas) on field hydrology and nitrogen losses (papers I–IV).
3. Investigate if it is possible to manage N transport by focusing on drainage and what is the effect of soil initial N storages on N loads during autumn period (Paper IV).

**Selecting an appropriate methodology for studying field drainage and N transport at field scale**

1. Investigate the benefits of using mathematical modelling and statistical analysis for assessing the functionality of drainage solutions (Papers I–IV).
2. Test the performance of models with different structures by applying (1) measured soil hydraulic data directly in the model parameterization without calibration, and (2) different descriptions for field drainage systems in 1D, 2D and 3D model applications (Papers II–IV).
3. Develop a generic solute transport model for the FLUSH model that is tested to simulate N transport in clayey, subsurface drained soils during autumns. The resulting model is used to assess what processes and factors are important in the simulations (Paper IV).

These objectives together aim to promote sustainable agricultural land use. The aim is to decipher if there is a conflict between reducing N losses and improving field drainage. For this, it is necessary to understand the local flow and transport processes in order to apply functioning solutions to manage N transport at field scale.
The effect of subsurface drainage improvements on water flow, the effect of drainage installation method on drainage performance, and the role of subsurface drainage on N load were studied at two experimental sites in Finland (Fig. 1). The two sites enabled the analysis of a variety of subsurface drainage solutions in clayey (Nummela) and loam–loamy sand–sandy loam (Sievi) fields. Both fields were equipped with monitoring devices, and gathered field data was studied with statistical analysis (Sievi) and mathematical model applications (Nummela).

Figure 1. Illustration of the thesis work in (a) Sievi, and (b) Nummela fields. The drain pipes installed with the trencher and trenchless drainage method are marked as T1 and T0, respectively. Loam, sandy loam, and loamy sand soil types are labeled as Lo, SL, and LS, respectively.
2.1 Study sites

2.1.1 Sievi (Paper I)

An experimental field where two different drainage installation machines were tested was established in Sievi (63°55’98”N, 24°20’65”E), located in northern Ostrobothnia (Fig. 1). Äijö et al. (2017) initiated the field experiment in 2015 when subsurface drainage was installed in the field with trencher and trenchless methods. The field area was divided into eight field sections (Fig. 1), which were each drained with three plastic pipes installed at depth of 1.0 m and with 15 m drain spacing. Four sections were drained with the trencher drainage (T1) method and the other four with the trenchless drainage (T0) method. No artificial barriers (e.g. plastic sheet) were used between the field sections. Both drainage installation methods were done at the same time to maximize the similarities between sections drained with T0 or T1. Also, in both methods, gravel was used on top of the drain pipe as an envelope material, and dried topsoil material was applied as trench backfilling (Äijö et al., 2017, 2016).

The soil type of the field sections at the drain depth (Table 2) was loam (sections 1, 2, 7, and 8), sandy loam (3, 4 and 5) or loamy sand (6). Above the drain depth layer (0.5–0.8 m), the soil type was sandy loam except in field section 8 where the soil was loam.

Groundwater depth was monitored for two months (March to April, 2015) before the drainage installation took place. After the drainage installation, groundwater observation tubes were installed at seven locations of each field section to observe groundwater levels as a function of distance from the middle drain line of the three drains (see Fig. 1 in Paper I). Manual groundwater depth observations were gathered during a two-year observation period (4.6.2015–30.6.2017) twice a week, except only once a week during winter.

Table 2. Soil texture class (IUSS, 2014) at the soil layers (0.5–0.8 m and 0.8–1.0 m) for the trenchless (T0) and trencher (T1) sections.

<table>
<thead>
<tr>
<th>Layer depth</th>
<th>1 (T0)</th>
<th>2 (T1)</th>
<th>3 (T1)</th>
<th>4 (T0)</th>
<th>5 (T1)</th>
<th>6 (T0)</th>
<th>7 (T0)</th>
<th>8 (T1)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.5–0.8 m</td>
<td>SL</td>
<td>LS</td>
<td>LS</td>
<td>SL</td>
<td>LS</td>
<td>SL</td>
<td>SL</td>
<td>Lo</td>
</tr>
<tr>
<td>0.8–1.0 m</td>
<td>Lo</td>
<td>Lo</td>
<td>SL</td>
<td>SL</td>
<td>SL</td>
<td>LS</td>
<td>Lo</td>
<td>Lo</td>
</tr>
</tbody>
</table>

SL = Sandy loam, LS = Loamy sand, Lo = Loam

2.1.2 Nummela (papers II–IV)

The Nummela experimental field is located in Jokioinen in southern Finland (60°51’59”N, 23°25’50”E). The area was divided into monitored field sections with their own drainage systems (Vakkilainen et al., 2010). The heavy clay field was originally subsurface drained in the 1950s with a drain spacing of 16 (sections B and C) and 32 m (section D). In 2008, part of the 16 m drain spacing area was supplementary drained, resulting in 8 m drain spacing (section C). In 2014, the 32 m drain spacing area (section D) was supplementary drained, resulting in 10.7 m drain spacing. Part of the original 16 m drain spacing area remained as a control section (section B). The subsurface drains (Fig. 1) were installed with trencher
drainage machinery, and gravel was used in the supplementary drainage as an envelope material (approximately 0.3–0.5 m above the drain pipe). The original drainage system was composed of clay tile drains, and presumably only a small amount of gravel was used in the junctions of the tiles.

Soil samples were gathered from each field section at five different locations. Soil properties were determined for depths of 0–0.2, 0.2–0.4, and 0.4–0.6 m representing the topsoil, plow pan, and subsoil layers, respectively. Water retention curves were fitted to soil moisture observations with suctions of 0.1, 1.0, and 150 m (for C and D in Äijö et al., 2017 and for B in Turunen et al., 2013).

All the field sections were intensively monitored during 2007–2017. Drain discharge and topsoil layer runoff are collected from each field section and routed to a measurement hub where outgoing flows were measured automatically with a 15-minute time interval using Datawater WS Vertical helix meter (Maddalena, Povoletto, Italy). In autumn 2011, topsoil layer runoff collectors were repaired, which caused some disturbance to the soil. Groundwater depth and soil moisture were manually observed and measured at the middle of the drain lines approximately once a week or every other week. Also, precipitation was measured at the field site with a tipping bucket rain gauge (RAINEW 111, RainWise Inc., USA). Other hourly meteorological data (air temperature, relative humidity, wind speed, and global radiation) were available from the Finnish Meteorological Institute, which administers the Jokioinen Observatory (7 km from the field site).

Cultivation practices during the field experiment were the same for all the field sections and between the years. The cultivated crops were barley and oats. Mainly mineral fertilizer was applied in the beginning of growing periods. Manure was applied in 2008. A detailed description of the cultivation activities was reported by Äijö et al. (2017, 2014). The field is administrated by the Natural Recourses Institute Finland (LUKE).

2.2 Statistical analysis (Paper I)

Manually gathered groundwater level time series from Sievi (Fig. 1) were studied with graphical analysis, statistical tests, Sen’s slope, and cumulative sums to detect differences between field sections drained with the trencher (T1) and trenchless (T0) methods. Two types of time series groupings were used—one according to the drainage method (T1 and T0) and another according to the soil type (loam and sandy loam, loamy sand). The statistical tests, cumulative sums, and Sen’s slope were applied to the matched pairs between the different drainage method groups (T0-T1) and the same drainage method groups (T0-T0 and T1-T1) to detect whether or not the difference in the groundwater levels was caused by the drainage method. For the soil-type grouping, cumulative sums and Sen’s slope were used to determine if differences occurred between the same or different soil type groups.

Because groundwater level observations were collected at irregular time intervals (twice a week, and once a week during winter), the customary time series methodology requiring equal observation intervals could not be used (Newbold et al., 2013). As all the observations were time-dependent and the observations from
different measurement points were likely to be correlated, two sample tests—like $t$-test and Mann-Whitney U-test—could not be used (Newbold et al., 2013). In order to compare observations from different measurement points, the effects of correlation were removed by using matched pair tests, like the one sample $t$-test, Wilcoxon signed-rank test, and sign test (Newbold et al., 2013). Cumulative sums and Sen’s slope were used in order to minimize the effects of the time dependency of the observations (De Gooijer, 2017; Wilcox, 2011).

In addition to testing differences between time series where all the observation time points were included, depth-filtered time series were also tested. The depth-filtered time series were formed so that only the time points when all the groundwater level observations were above the drain depth were included. This approach was chosen because the aim was to detect if the differences occurred due to the drainage installation method or due to the soil properties above the drain depth.

2.3 Numerical modeling (papers II–IV)

2.3.1 Water flow in FLUSH

FLUSH (Warsta et al., 2013) is a spatially distributed model that simulates the main water flow processes in the 2D overland domain and 3D subsurface domain. FLUSH has been tested in several experimental fields with a preferential flow description in the subsurface domain (e.g. Koivusalo et al., 2017; Nousiainen et al., 2015; Turunen et al., 2013; Warsta et al., 2013).

The water flow model follows a dual-permeability (DUP) approach in which water is mobile in both soil matrix and macropore systems. In FLUSH, precipitation is first stored at the soil surface depression storages, and surface flow occurs according to the soil topography and water depth, if the storage capacity is exceeded. Water is lost from the surface domain via runoff to open ditches and infiltration from the overland storage into the soil matrix and macropore systems. Water is lost from the subsurface soil domain via subsurface drain discharge, groundwater outflow, seepage to open ditches, and evapotranspiration caused by crop water uptake. FLUSH includes a description for winter time processes such as snow accumulation and melting.

In FLUSH, the 2D water flow in the surface domain is described with the diffuse wave approximation of the Saint Venant equations. In the subsurface domain, 3D water flow is calculated with the Richards (1931) equation in both pore systems. Water exchange between the pore systems is calculated according to Gerke and van Genuchten (1993). Unsaturated hydraulic conductivity and soil moisture content is calculated using the van Genuchten (1980) approach. Discharge to open ditches and subsurface drains is based on Darcy’s law. A full mathematical model description is documented in Warsta (2011). Snow processes and soil temperatures are simulated with the energy balance submodel (Warsta et al., 2012) based on Koivusalo et al. (2001).

In the numerical solution, spatial components of the Richards equation are discretized using the implicit finite volume method and the temporal components with the backward difference method (Warsta, 2011). The computational grids are
regular. The grids are built inside the model (Papers II and IV) or imported to the model (Paper III). To find the numerical solution for the Richards equation, the model applies the pentadiagonal matrix algorithm (PDMA) (papers III and IV) or the Successive Over-Relaxation (SOR) solver that is a modification of the Gauss-Seidel method (Paper II). PDMA solves the hydraulic heads for each cell in one cell column in both pore systems at the same time, and the model iterates between the columns to find a global solution for the computational domain. In the case of SOR, water flow between the cells in all directions is solved through iteration.

2.3.2 **Solute transport in FLUSH (Paper IV)**

The solute transport component developed in Paper IV follows the dual-permeability concept of water flow in FLUSH. The model component is designed so that it is possible to describe multiple solutes with different transport properties and form reaction chains between the solutes. Solute transport is solved using the solute continuity equation (Eq. 1 in Paper IV), where solute transport is purely advective. Solute transport in the soil matrix and macropore systems of the subsurface domain is described with the advection–dispersion equation (ADE) in three dimensions (eqs. 2 and 3 in Paper IV). Retardation and first-order decay are included in the ADE. Solute flux between the pore systems is computed with mass exchange (Gerke and van Genuchten, 1996), which has both advection and dispersion components. Solute transport to sinks (ditches, drains, and groundwater outflow) is computed with advection. Solute transport from the surface to subsurface domain is computed with mass exchange between the surface cell and corresponding subsurface matrix or macropore cell.

In the numerical solution, the spatial components were discretized based on an implicit finite volume method. Solute transport is solved by iteration in the surface domain. In subsurface domain, transport is solved with PDMA in the vertical direction, and iteration is used for solving the transport between cell columns in the horizontal direction. Transport of solutes is solved one solute at a time.
2.3.3 Nitrogen model (Paper IV)

The solute transport component was tailored to describe the processes in the nitrogen cycle. The simulated N cycle included organic N, NH4-N, and NO3-N storages, and the reaction chain between the N fractions (Fig. 2). Organic N storage described the N in crop residues and in other easily degradable organic matter and was therefore set as an immobile solute storage, in which only decay occurred. NH4-N and NO3-N were mobile solutes that moved in the soil matrix and macropore systems. Transport of NH4-N was delayed by sorption.

The reactions in the N cycle are highly dependent on the surrounding conditions (Section 1.4), and therefore they were restricted by temperature and soil moisture coefficients (eqs. 1 and 2). Temperature restriction had the same effect on all the reactions (mineralization, nitrification, and denitrification), reducing the bacterial activity during low temperatures. High moisture content had a positive effect on denitrification, but a negative effect on nitrification and mineralization. Based on the literature (see Section 1.4), the restrictions were needed for simulating autumn periods (see Section 4.3).

\[
\mu_{nitrification} = \mu_{amm} [1 - f(\theta)] f(T) 
\]

(1)

\[
\mu_{denitrification} = \mu_{nit} f(\theta) f(T)
\]

(2)

where \(\mu_{amm} \text{ [h}^{-1}\text{]}\) is the nitrification rate, \(\mu_{nit} \text{ [h}^{-1}\text{]}\) is the denitrification rate, and

\[
f(\theta) = \left(\frac{\theta}{\theta_s}\right)^B
\]

(3)

Figure 2. Simulated N storages and processes. The conceptual model includes the main parts of the N cycle described in Section 1.4.
Methodology

\[ f(T) = t_{q10}^{(T-t_{bas})/10} \]  

where \( \theta \) [m³ m⁻³] is the soil moisture, \( \theta_s \) [m³ m⁻³] is the saturated soil moisture, \( B \) [-] is a calibrated parameter, \( t_{q10} \) and \( t_{bas} \) [°C] are calibrated parameters for the bacterial activity (Bunnel et al., 1977), and \( T \) [°C] is the soil temperature. Soil moisture restriction was adapted from Kroes and van Dam (2003). The limiter allowed nitrification reaction to occur if the water content was below saturation.

2.3.4 Model applications

Water flow simulations were done with 1D, 2D, and 3D grids for a variety of drainage solutions in the Nummela field (Fig. 3). Model simulations were carried out for four autumn periods (2007, 2008, 2011, and 2012) in addition to long-term simulations of nine years (2008–2017). Hourly input data series were formed from the meteorological observations (see Section 2.1.2). Data series of NO3-N and NH4-N air deposition were formed according to monthly measurements by the Finnish Environmental Institute. No fertilizer or manure applications were considered for the autumn period simulations. The measured N balance of the growing periods together with the findings of Pietola et al (1999) were used to estimate the initial soil N storages (Paper IV). Initial groundwater depths for the simulations were set according to observations in the fields.

![Figure 3. Studied drainage systems and model applications. DUP is the dual-permeability model structure, SP is the single porosity model structure and Trench refers that the drain trench is parameterized in the computational grid.](image-url)
The dimensions of the computational grids (length × width × height) used in drain spacing scale simulations were 8.0 m × 4.0 m × 1.5 m (Paper II), 16.0 m × 16.0 m × 2.6 m, and 32.0 m × 32.0 m × 2.6 m (Paper IV). The horizontal grid dimensions were selected according to the simulated drain spacing (8, 16, or 32 m). The dimensions of the computational grid used in the field section scale simulations were 300 m × 4.0 m × 6.0 m, and the grid extended through the simulated field section (Paper III). The depth of the field section scale computational grid was greater compared to the drain spacing scale grids due to elevation differences in section D including the bottom elevation of the steep slope at the field boundary (Paper III).

Soil parameterization

The structure of FLUSH enables water flow simulations to be carried out with the dual-permeability (DUP) or single porosity (SP) concepts. In Paper II, model parameterizations were implemented with both SP and DUP modeling approaches. In the SP model, the parameterization describes properties of an average soil core. In the DUP model, both soil matrix and macropore systems were parameterized separately.

Saturated hydraulic conductivity of the soil $K_{\text{sat}}$ [m h⁻¹] is the weighted sum of soil matrix $K_{\text{sat,M}}$ [m h⁻¹] and macropore $K_{\text{sat,F}}$ [m h⁻¹] conductivities, taking into account the total pore space of the two pore systems (Eq. 5). Saturated hydraulic conductivity of the macropore system is calculated with Eq. 6 in which the fraction of the macropore system of the total porosity $w$ [-] affects the conductivity of the system ($K_{\text{sat,F}}$).

$$K_{\text{sat}} = (1 - w) K_{\text{sat,M}} + w K_{\text{sat,F}}$$

$$K_{\text{sat,F}} = w K_{\text{FS,MUL}}$$

where $K_{\text{FS,MUL}}$ [m h⁻¹] is the macropore saturated hydraulic conductivity multiplier.

In Paper II, soil parameterizations for the SP and DUP models were derived from the measured soil properties that were available from five soil sample cores and three layers from each core (Section 2.1.2). The three layers, where the soil samples had been collected, represented the top soil, plow pan, and subsoil layers (Table 3). After creating the five soil parameterizations (soil sets C1–C5) for the SP and DUP models, additional parameterizations for both models were created using randomized soil properties (see Paper II for details). The drain trenches were parameterized according to the properties of the backfill material in the original and supplementary trench segments. In addition to the five soil parameterizations, three different drainage schemes were formed describing (1) the original drainage with 16 m drain spacing, (2) the supplementary drainage with 8 m drain spacing, and (3) the supplementary drainage without the drain trenches (original or supplementary). The varying soil parameterizations and drainage schemes enabled the possibility to assess the roles of the surrounding soil and drainage design on water flow (see sections 3.2 and 4.3).
In Paper III, soil parameterization followed the approach in Paper II. Old and new drain trenches were parameterized separately according to the amount of gravel used above the drains. The parameterization of the surrounding soil (Table 3) was set according to the average of the measured soil sample data.

In Paper IV, the soil properties were parameterized (Table 3) based on the average values of the soil sample data in each layer. The N model was parameterized according to the collected data and with values found in the literature (see Paper IV for more details).

**Model evaluation criteria**

Paper II tested the model performance with soil parameterizations derived from the soil samples. The goodness of the model output was assessed with the Nash-Sutcliffe efficiency coefficient (NSE) (Nash and Sutcliffe, 1970).

Paper III simulated the timing and magnitude of the observed drain discharge and tillage layer runoff. As the focus was on capturing the annual water balance, the total water volumes (mean absolute deviation, MAD) were more important in this case than accuracy of the daily hydrographs (e.g. Rankinen, 2006). In Paper III, the modified NSE (e.g. Krause et al., 2005) was used instead of the NSE applied in papers II and IV. De Schepper et al. (2017) used weekly average values to calculate the NSE to avoid the dominating effect of the higher differences between the simulated and observed values that were raised to the power of two in the NSE calculation. Here, the modified NSE was chosen, as it is not as sensitive to higher values as the normal NSE and is therefore a better overall indicator for under- or overpredictions (Krause et al., 2005).

**Table 3.** Soil parameterization of the simulated Nummela field sections B, C, and D. For B and D, soil parameterization was based on the average values of the soil properties. For C, the range (minimum to maximum) of the soil parameter values is presented.

<table>
<thead>
<tr>
<th>Soil layer</th>
<th>Field section</th>
<th>( \theta_b ) [m(^3) m(^-3)]</th>
<th>( w ) [-]</th>
<th>( K_{sat} ) [m h(^{-1})]</th>
<th>( \alpha ) [m(^{-1})]</th>
<th>( n ) [-]</th>
<th>( K_{SMA} ) [m h(^{-1})]</th>
</tr>
</thead>
<tbody>
<tr>
<td>0–0.25</td>
<td>B</td>
<td>0.54</td>
<td>0.033</td>
<td>0.1</td>
<td>1.40</td>
<td>1.13</td>
<td>90</td>
</tr>
<tr>
<td></td>
<td>C</td>
<td>0.48–0.56</td>
<td>0.01–0.1</td>
<td>0.03–0.6</td>
<td>0.65–13.0</td>
<td>1.12–1.17</td>
<td>10–80</td>
</tr>
<tr>
<td></td>
<td>D (* )</td>
<td>0.56</td>
<td>0.034</td>
<td>0.11</td>
<td>2.77</td>
<td>1.12</td>
<td>90</td>
</tr>
<tr>
<td></td>
<td>D (** )</td>
<td>0.56</td>
<td>0.034</td>
<td>0.11</td>
<td>3.56</td>
<td>1.15</td>
<td>90</td>
</tr>
<tr>
<td>0.25–0.45</td>
<td>B</td>
<td>0.55</td>
<td>0.0014</td>
<td>0.00028</td>
<td>0.11</td>
<td>1.17</td>
<td>90</td>
</tr>
<tr>
<td></td>
<td>C</td>
<td>0.51–0.59</td>
<td>0.0002–0.006</td>
<td>0.000004–0.26</td>
<td>0.12–0.96</td>
<td>1.10–1.30</td>
<td>9–80</td>
</tr>
<tr>
<td></td>
<td>D (* )</td>
<td>0.56</td>
<td>0.001</td>
<td>0.0002</td>
<td>0.13</td>
<td>1.17</td>
<td>90</td>
</tr>
<tr>
<td></td>
<td>D (** )</td>
<td>0.56</td>
<td>0.001</td>
<td>0.0002</td>
<td>0.33</td>
<td>1.15</td>
<td>90</td>
</tr>
<tr>
<td>0.45–1.0</td>
<td>B</td>
<td>0.55</td>
<td>0.0038</td>
<td>0.0014</td>
<td>0.11</td>
<td>1.17</td>
<td>90</td>
</tr>
<tr>
<td></td>
<td>C</td>
<td>0.50–0.55</td>
<td>0.001–0.008</td>
<td>0.00005–0.03</td>
<td>0.44–0.96</td>
<td>1.12–1.14</td>
<td>5–80</td>
</tr>
<tr>
<td></td>
<td>D (* )</td>
<td>0.56</td>
<td>0.002</td>
<td>0.0005</td>
<td>0.13</td>
<td>1.17</td>
<td>90</td>
</tr>
<tr>
<td></td>
<td>D (** )</td>
<td>0.58</td>
<td>0.002</td>
<td>0.0005</td>
<td>0.58</td>
<td>1.12</td>
<td>90</td>
</tr>
</tbody>
</table>

\( * \) in Paper IV

\( ** \) in Paper III
3 Results

3.1 Field drainage performance

In Paper I, comparison of groundwater levels between field sections drained with the trencher (T1) or trenchless (T0) drainage methods in the Sievi experimental field revealed that both the drainage installation method and soil texture influenced the differences between the sections. The measured groundwater level range in sections where the same drainage method had been applied indicated that the drainage method alone did not explain the differences (Fig. 4a). For the T0 sections, the groundwater level variability was greater during periods when the groundwater table was near the field surface (autumn 2015, summer 2016, and spring 2017, Fig. 4a). But for the T1 sections, the variability was the greatest during periods when the groundwater table was below the drain depth (winter and early spring, Fig. 4a). A graphical analysis showed that the greatest absolute differences between T0 and T1 (T0 was 0.38–0.6 m higher than T1) were at the observation locations nearest to the drain line (at 0.2 m). The average difference between T0 and T1 sections, in terms of the maximum groundwater level time series, ranged from 0.11 m to 0.16 m.

When the Sievi field sections were grouped according to the soil texture in the drain depth, it was noticed that the loam field sections had more variability in groundwater level results near the drain line, and the variability decreased when moving away from the drain line (Fig. 4b). The variability of the groundwater level in the sandy loam and loamy sand field sections was the greatest at the middle of the drain lines (7.5 m away from the drain line) and decreased closer to the drain line (Fig. 4b). This indicated that the soil type of the neighboring section was affecting the groundwater level. Sandy loam and loamy sand field sections were located in the middle of the field and surrounded by the loam sections (see Fig. 1 and Table 2).

The experimental setup aimed to minimize the effect of the drainage installations of the neighboring field sections, as each field section had three drains and the observations were made from the middle drain. However, this approach did not completely rule out the possible effect of the soil texture of the neighboring sections on the results. When the field sections were organized according to the soil type, Sen’s slope for depth-filtered time series (see Section 2.2) showed that differences in groundwater levels occurred between the different soil types and the same soil types (Fig. 7 in Paper I). Sen’s slope showed also that groundwater level was higher in the T0 sections than in the T1 sections. Cumulative sums
showed that To section 4 was an exception, which demonstrated the effect of the coarser soil type (see Table 2) with descending (T1 higher than To) or slowly rising cumulative sums in the To–T1 comparison (Fig. 8 in Paper I).

Figure 4. Groundwater level range for (a) trenchless and trencher drainage methods, and (b) loam and sandy loam, loamy sand field sections. The ranges are presented near the drain line location (at 0.2 m) and at the midpoint between the drain lines (at 7.5 m). Paper I.

Paper III showed that improved drainage practice (decreasing drain spacing from 32 m to 10.7 m) increased the drainage performance of the poorly drained clay field section in the Nummela field in terms of groundwater depth during the spring snowmelt period (April to May). Long-term FLUSH simulations (2008–2017) of the field hydrology showed that after the supplementary drainage installation, GWD dropped to 0.6 m over a week earlier compared to the original drain spacing scenario (Fig. 7 in Paper III). In Sievi, drainage efficiency differences between the field sections—where trenchless and trencher drainage methods had been applied—were not observed in spring before sowing (Paper I). However, at the time of the harvest, the groundwater level was 0.2–0.47 m higher in the trenchless drainage sections than in the trencher drainage sections, depending on the distance to the drain line (Fig. 2 in Paper I).

3.2 Modeling water flow

3.2.1 Model parameterization

The soil data from the Nummela experimental field enabled a detailed, spatially distributed soil parameterization of the 3D FLUSH model to describe the field. The model parameterization was done with data gathered from field section C (Fig. 1). The parameters derived from soil samples collected from five different
locations and three different depths in Paper II (Section 2.1.2) described large heterogeneities in the soil properties. The range of simulated drain discharge events produced with the different soil parameter sets nearly covered the measured drain discharge events during three autumn periods (NSE 0.05–0.76 and Fig. 5 in Paper II). Performance was weaker with the DUP modeling approach (NSE < 0.65), and negative NSE values occurred during high rainfall events (4–6 mm h⁻¹). One explanation to the weaker model performance is that drain discharge in the model was not restricted by the drain pipe capacity (see Section 4.3), as the drains were described as sinks, i.e. water was removed from the system immediately and pipe flow was not simulated.

In Paper III, a soil-data based model parameterization approach was tested in field-scale simulations (D section in the Nummela field) to describe the effects of improved drainage on water outflow and groundwater depth. The 2D computational grid was well-suited for the purpose because it could describe the spatial features, including the steep slope and the drain trenches in the simulated domain (Figs. 1 and 2 in Paper III). The parameterization approach enabled simulation of spatial differences in drain discharge generation, and a larger share of drain discharge (84%) originated from the supplementary drains. From the drain-spacing scale (Paper II) to the field-section scale (Paper III), the average values of the soil sample measurements in each layer (topsoil, plow pan, and subsoil layers) were appropriate for the soil surrounding the drain trenches because a similar parameterization approach has worked well in the past to simulate the whole field section (Turunen et al., 2013). Models for the original and supplementary drainage settings were separately calibrated against hourly drain discharge and topsoil layer runoff during snowless periods (see Paper III for details). Modified NSE for the calibration period drain discharge were 0.53 and 0.5 in the case of original and supplementary drainage, respectively. For the validation period drain discharge, the corresponding values were 0.43 and 0.52. Still, simulated annual water volumes were close to the field measurements (Figs. 4 and 5 in Paper III) with absolute deviations of 35 mm and 26 mm on average for drain discharge and topsoil layer runoff, respectively. The model was better able to simulate the effects of supplementary drainage on field water balance than discharge dynamics.

The autumn 2008 period was used to test the effects of soil macropore parameterization and the existence of drain trench on discharge volumes with the Nummela field section C setup (see Section 2.3.4). Simulations with and without macropore system descriptions (DUP and SP models) revealed the dominant nature of the macropores on water outflow pathways (Fig. 5). The results of different soil parameterizations (soil samples C1–C5 and randomized soil) showed that the soil properties had a more pronounced role in the formation of drain discharge than the existence of a highly conductive drain trench (DUP model in Fig. 5). However, without the macropore network (SP model in Fig. 5), this was not the case. The simulations with the SP model and explicit parameterization of the drain trench indicated that a highly permeable drain trench has a similar effect to drain discharge as the soil macropores.
Results

Figure 5. Simulated water balance components (16.10.–7.11.2008) using the single pore system (SP) and dual-permeability (DUP) models for the 8 m drain spacing (a) with and (b) without drain trenches. Paper II.

3.2.2 Model applications with drainage scenarios

Model applications with different drainage setups for Nummela field sections B, C, and D (Fig. 3) showed that groundwater outflow was an important part of the field section water balance. In section D, annual groundwater outflow was on average 26% of precipitation, and improved drainage installation decreased the share of groundwater outflow to 22%. However, simulations for the 2008 and 2011 autumn periods (Paper IV) showed that groundwater outflow in autumn was even greater (35–38% of precipitation) from the poorly drained section D. Autumn simulations in sections B (Paper IV) and C (Paper II) showed that groundwater outflow was around 28–35% of precipitation (Table 4) when drain spacing was 16 m. Decreasing the drain spacing from 16 m to 8 m in section C decreased the share of groundwater outflow to 23%. In the case of poorly functioning drainage (i.e. the no-trench scenario with 8 m drain spacing), the share of groundwater outflow was approximately the same as in the 16 m drain spacing scenario (Fig. 6 in Paper II). In the absence of soil macropores (SP model results in Table 4), groundwater outflow was clearly smaller (5–13% of precipitation), which points out the role of macropores in the rate of groundwater outflow.

Simulation results from the autumn 2008 and 2011 periods showed that drain discharge accounted for 34–40% of the B section water balance, but only 20% of the section D water balance (Table 4). Table 4 shows that drain discharge in section C was clearly higher (53–85%) than in the B and D sections during autumn periods. The high discharge in C was explained by the small drain spacing (8 m) during the 2008 and 2012 simulations. Larger drain spacing in the B and D sections (16 m and 32 m) resulted in the higher groundwater outflow and topsoil layer runoff compared to section C.
In papers II and IV, short-term water flow (weeks to months) was simulated, while long-term simulations in Paper III exposed the seasonal differences in water balance components (Fig. 6 in Paper III). For an average year, the monthly outflow (drain discharge + topsoil layer runoff + groundwater outflow) was the same regardless of the drainage scenario (Fig. 6). The supplementary drainage changed the share of the three outflow components. The highest changes occurred during autumn and spring. The summer period was dominated by evapotranspiration (Fig. 6 in Paper III), and discharge from the field was mainly groundwater outflow (Fig. 6).

![Figure 6. Monthly average water outflow components for (a) the original drainage scenario and (b) the supplementary drainage scenario. Simulation results are from Paper III.](image)

**Figure 6.** Monthly average water outflow components for (a) the original drainage scenario and (b) the supplementary drainage scenario. Simulation results are from Paper III.

<table>
<thead>
<tr>
<th>Year</th>
<th>Drain discharge [%]</th>
<th>Topsoil layer runoff [%]</th>
<th>Groundwater outflow [%]</th>
</tr>
</thead>
<tbody>
<tr>
<td>2007</td>
<td>3.10-3.11. 63 / 53</td>
<td>12 / 6</td>
<td>13 / 35</td>
</tr>
<tr>
<td>2008</td>
<td>14.10-7.11. 85 / 76</td>
<td>9 / 1</td>
<td>5 / 23</td>
</tr>
<tr>
<td>2011</td>
<td>B 1.9-31.12. 34</td>
<td>15</td>
<td>28</td>
</tr>
<tr>
<td>2012</td>
<td>B 1.9-31.12. 40</td>
<td>22</td>
<td>35</td>
</tr>
<tr>
<td></td>
<td>C 14.10-7.11. 84 / 69</td>
<td>22</td>
<td>38</td>
</tr>
</tbody>
</table>

Table 4. Simulated water outflow components (share of precipitation) for autumn periods 2007, 2008, 2011, and 2012. Results for section C are presented separately for the SP and DUP models (SP% / DUP%). Simulation results for section C are from Paper II, and simulations for sections B and D are from Paper IV.

* Calculated from the mean simulation results of the five soil parameterizations.

### 3.3 Nitrogen transport and leaching

In Paper IV, the developed solute transport component was tested by simulating N transport and processes (Fig. 3) with the calibrated and validated water flow component of FLUSH (field sections B and D in the Nummela field). The model sensitivity analysis in Paper IV showed that, initialization of soil N content had a notable effect on autumn N loads. The initial values of the soil N content affected the modeled drain discharge loads (Paper IV) and loads via groundwater outflow, but not topsoil layer runoff loads (Table 5). Nearly all of the soluble N (NO3-N +
NH4-N) was transported by advective water flow in the macropore system to field discharge points; less than 0.0025% originated from the soil matrix system in the case of drain discharge. In addition, the dominant nature of advective transport was seen through the water flow parameters, including drain entrance resistance and water exchange coefficient, which had a notable effect on the leaching amounts (full sensitivity analysis in Salo, 2014).

Differences were detected between the 16 m (B) and 32 m (D) drain spacing field sections in terms of N concentrations at the outflow pathways. In autumn 2008, the highest drain discharge NO3-N concentration was 30% greater from the B section than from the D section. The mean concentration of section B drain discharge was 1.5 times the mean concentration of section D. Also, the total measured NO3-N load for the autumn (September to December) was three times greater from the B section than from the D section (0.54 and 0.18 kg N ha⁻¹), and the simulated total load from the B section was also three times the amount from the D section (0.95 and 0.33 kg N ha⁻¹). The difference in NO3-N load between sections B and D was caused by drain discharge volume differences, as the concentration difference in drain discharge was clearly smaller.

The main controlling factors behind load generation after the growing period were soil N storage after harvest and the timing of precipitation events. Paper IV showed that during autumn 2011 (the validation period), the concentrations of NO3-N in drain discharge were notably higher (2–12 mg N l⁻¹ in B and 1–9 mg N l⁻¹ in D) than during autumn 2008 (< 3 mg N l⁻¹ in B and < 2 mg N l⁻¹ in D). The differences were explained by the soil N storages after the growing periods. In 2011, crops were not harvested due to a rainy autumn that led to wet soil conditions (Äijö et al., 2014). The effect due to the timing of precipitation was seen in 2011, when there was a drier period in the middle of the autumn that led to an accumulation of NO3-N in the soil (Fig. 8). The accumulated NO3-N caused the high concentration peak in drain discharge after the low discharge period (Fig. 7 in Paper IV).

*The effect of sorption*

Compared to the NO3-N concentrations, the NH4-N concentrations in drain discharge, topsoil layer runoff, and groundwater outflow, as well as the resulting loads, were less affected by the drain spacing, soil N storage, and water outflow volumes during the two studied autumn periods (2008 and 2011). Drain discharge NH4-N concentrations and loads were 5–10 times smaller compared to the NO3-N results (Table 5, and figs. 4 and 7 in Paper IV). On the other hand, simulated NH4-N loads via topsoil layer runoff were 25–60% higher than NO3-N loads (Table 5), which can be explained by the sorptive nature of NH4-N. Sorption caused a lower infiltration rate of NH4-N by delaying NH4-N transport from the soil surface to the subsurface domain, and therefore there was more NH4-N mass at the surface available for leaching via topsoil layer runoff. Solute transport in the overland domain was purely advective (Section 2.3.2).
Decay restrictions with soil temperature and moisture

The main mass balance component for NO$_3$-N and NH$_4$-N was decay (Table 5). All the decay reactions were restricted by soil moisture and temperature factors (Section 2.3.3), but the effect of soil moisture content was clearer based on the simulations. The higher soil moisture condition in section D increased denitrification (i.e. decreasing NO$_3$-N storage) compared to section B. In contrast, lower soil moisture content resulted in higher nitrification in section B compared to D (i.e. increasing NO$_3$-N storage). According to the N balance calculations in Numnela, section D had the lowest N leaching rate and also the highest residual of the measured N balance (Äijö et al., 2014). The measured N balance did not take into account N leaching via groundwater outflow or denitrification of NO$_3$-N into gaseous compounds. The negative storage changes (figs. 7 and 8) include NO$_3$-N leaching via subsurface topsoil layer runoff, drain discharge, groundwater outflow, and denitrification. Positive storage change includes infiltration of NO$_3$-N from soil surface to subsurface domain and nitrification.

The simulated soil temperature was affected by the energy submodel initialization (based on Turunen et al., 2013) and meteorological input data series which were the same for sections B and D. Therefore, the effect of the temperature restriction was not clear from the comparison of N balances between the field sections (Table 5). The temperature factor affected the efficiency of N reactions (i.e. the mass of soil N storages). During low temperatures with minimal reaction rates, the already existing N fractions were still transported in the soil via water flow routes. Also, the higher decay in 2011 than in 2008 simulations was caused by the greater N storages after the growing period in 2011. N left in the field after harvest was greater in 2011 than in 2008 (Äijö et al., 2014).

Table 5. Simulated NO$_3$-N and NH$_4$-N balances for calibration (2008) and validation (2011) periods. The measured mass per area is presented in parentthesis.

<table>
<thead>
<tr>
<th>N balance components</th>
<th>2008 autumn</th>
<th>2011 autumn</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>B</td>
<td>D</td>
</tr>
<tr>
<td>NO$_3$-N</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Initial storage</td>
<td>3.91</td>
<td>3.91</td>
</tr>
<tr>
<td>Air deposition $^c$</td>
<td>0.78</td>
<td>0.78</td>
</tr>
<tr>
<td>Decay</td>
<td>1.25</td>
<td>1.42</td>
</tr>
<tr>
<td>Drain discharge</td>
<td>0.94 (0.54)</td>
<td>0.33 (0.18)</td>
</tr>
<tr>
<td>Topsoil layer runoff</td>
<td>0.09 (-)</td>
<td>0.16 (0.12)</td>
</tr>
<tr>
<td>Groundwater outflow</td>
<td>0.65</td>
<td>0.50</td>
</tr>
<tr>
<td>End storage</td>
<td>3.40</td>
<td>3.10</td>
</tr>
<tr>
<td>NH$_4$-N</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Initial storage</td>
<td>3.91</td>
<td>3.91</td>
</tr>
<tr>
<td>Air deposition $^c$</td>
<td>0.63</td>
<td>0.63</td>
</tr>
<tr>
<td>Decay</td>
<td>1.64</td>
<td>0.83</td>
</tr>
<tr>
<td>Drain discharge</td>
<td>0.18 (0.05)</td>
<td>0.06 (0.02)</td>
</tr>
<tr>
<td>Topsoil layer runoff</td>
<td>0.12 (-)</td>
<td>0.20 (0.01)</td>
</tr>
<tr>
<td>Groundwater outflow</td>
<td>0.09</td>
<td>0.07</td>
</tr>
<tr>
<td>End storage</td>
<td>8.89</td>
<td>9.40</td>
</tr>
</tbody>
</table>

$^c$ From the Finnish Environmental Institute
Figure 7. Simulated soil NO3-N daily storage change, and the simulated soil temperature and groundwater table level in field sections (a) B and (b) D for the autumn 2008 period. The storage change is relative to the initial soil NO3-N storage (i.e. daily change is divided by the initial storage mass).

Figure 8. Simulated soil NO3-N daily storage change, and the simulated soil temperature and groundwater table level in field sections (a) B and (b) D for the autumn 2011 period. The storage change is relative to the initial soil NO3-N storage (i.e. daily change is divided by the initial storage mass).
4 Discussion

4.1 Performance of subsurface drainage

Factors that affect performance of subsurface drainage

The drainage installation method and the soil type in the drain depth explained the differences in groundwater level time series between the sections in the Sievi field (Paper I). Previous studies have documented differences in drainage performances with the trencher (T1) and trenchless (To) drainage installation methods (e.g. Mirjat and Kanwar, 1992; Kanwar et al., 1986) but did not investigate the role of other factors (e.g. soil texture, topography, or installation conditions) that may have influenced groundwater levels in the experimental field plots. The novelty of the Sievi experimental setup was that the soil type differences between the field sections enabled the comparison of the field sections that had the same and different soil type in the drain depth (see Section 2.2). Results from the Sievi field suggested that the drainage installation method (in favor of T1 over To) had greater influence on the drainage performance with finer soil texture (loam) than with coarser soil texture (sandy loam and loamy sand) (Section 3.1 and Fig. 4). Site-specific drainage solutions (e.g. Tuohy et al., 2016) indicate that the success of the drainage system is related more to the site properties than the applied installation method.

Assessing field drainage capacity with groundwater level

Measured (Paper I) and simulated groundwater depths (Paper III) were used in comparing the performances of the drainage systems and quantifying the drainage efficiency during spring after snowmelt (Paper III). Mante et al. (2018) and Kornecki and Fouss (2001) used soil moisture content to represent the soil trafficability and the field drainage efficiency. However, soil moisture data includes large variability, depending on the applied technique (e.g. Hakojärvi, 2015). Groundwater depth has been reported to be well-suited in evaluating clay field drainage (Aura, 1990). A larger spatial area is covered using groundwater level observations compared to soil moisture (e.g. Smedema et al., 2004), and data can be used in deciphering spatial water flow directions above and below the drain level (e.g. Vereecken et al., 2008). In addition, capillary upflux can be very slow in clay soils (Aura, 1990), which influences the soil water content. The preferential flow pathways enable groundwater table depth to react quickly to rain events (e.g.
Abbaspour et al., 2001); but according to Rye and Smettem (2018), this might not be visible in topsoil soil moisture.

The long-term FLUSH simulations in the Nummela field section D (Paper III) revealed that the simulated groundwater depth showed drainage efficiency differences between the original and supplementary drainage scenarios more clearly than the topsoil layer soil moisture content. In terms of soil moisture, the average difference was 0.01 m$^3$/m$^3$, and in terms of groundwater depth it was 0.17 m. It can be difficult to discern the effects between treatments when the differences in soil moisture content are near to measurement accuracy, e.g. 0.05 m$^3$/m$^3$ (Vereecken et al., 2008).

The Sievi experiment showed that the average difference in groundwater levels between field sections was small (around 0.1–0.15 m on average) and had no practical effect on cultivation (Paper I). However, the continuous simulated groundwater depth time series (Paper III) showed differences in drainage efficiency important for cultivation practices, in terms of the beginning of the growing period (e.g. Karvonen, 1988). When assessing hydrological fluxes in the field and the resulting drainage efficiency, it is important to compare the rate of change in soil moisture or groundwater depth (Fig. 6 in Paper III) to detect differences that might not be visible from the average difference of the observations (e.g. Vereecken et al., 2008).

**Effect of improved drainage on water outflow pathways**

Supplementary drainage installation in Nummela field section D increased the drain discharge volume more than it reduced the topsoil layer runoff volume (Section 3.2.2), and a similar result was reported previously by Seuna and Kauppi (1981). The simulation results in Paper III provided a rare quantification of drainage installation impact on all water outflow components (see Fig. 6). The evidence of the additional outflow pathway (groundwater outflow below the drain depth) was verified through the model simulations, even though indirectly seen in the data, as measured runoff coefficient (topsoil layer runoff + drain discharge) was remarkably lower in the D section than in the B and C sections (Vakkilainen et al., 2010; Turunen et al., 2013 and Äijö et al., 2014), i.e. the missing water had to flow somewhere. In the annual water balance, the missing water could belong to evapotranspiration, but this was not probable, based on the low crop yields from section D (Äijö et al. 2014). Also, the lower runoff coefficient in section D was seen during autumn periods when evapotranspiration was low (see Table 5).

**4.2 Field hydrology assessment**

**From statistical analysis to modeling**

Field hydrology has typically been studied using either statistical analysis (e.g. Tuohy et al., 2016; Patil et al., 2010; Turtola et al., 2007; Mirjat and Kanwar, 1992) or mathematical models (e.g. Koivusalo et al., 2017; Frey et al., 2016; Nousiainen et al., 2015). In Paper I, statistical tests provided a quick way of testing hypotheses about drainage performance differences between two drainage installation meth-
ods, but numerical modeling produced a mechanistic description of the field hydrology (papers II–IV). Sen’s slope and cumulative sums were the most promising analysis methods to present evidence that groundwater levels were higher for the trenchless (T0) sections than for the trencher (T1) sections, which was not evident when using only statistical tests (Paper I). The study setup in Sievi did not allow testing both the drainage installation method and soil type at the same time, as this would have required a greater number of tested samples (see Section 3.1). The findings in Paper II showed that in macroporous clay soil, variability of soil properties had a greater effect on water outflow pathways than the applied drainage scheme (see Section 3.2.1).

**Assessing the hydrological effects of drainage**

Papers I–IV showed how the field-specific features are important to consider when assessing the hydrological effects of field drainage. It is crucial to understand local processes in order to assess larger scale hydrological and environmental effects (e.g. Hansen et al., 2013). The agricultural practice in Finland is mostly subsidized, and practices that increase productivity and reduce the environmental load are supported (agri-environmental measures). Previously, agri-environmental measures have not adequately reduced agricultural nutrient losses (e.g. Ekholm et al., 2007; Granlund et al., 2005), but recent monitoring studies have reported a decreasing trend in nutrient losses (Aakkula and Leppänen, 2014). Papers I–IV demonstrated the use of statistical methods and mathematical modeling to decipher the functioning of field water management in individual locations. The modeling studies showed that the effect of spatial features on field drainage can be simulated with 1D, 2D, and 3D models, but the field drainage description in the process-based models needs to be tested against field data (e.g Filipović et al., 2014; Yousfi et al., 2014). Afterwards, the models can be used for assessing different drainage scenarios to identify their functioning on site (Paper III). The simulation results are more reliable if the model structure (i.e. field drainage description) has been validated for multiple drainage systems (papers II–IV) during hydrologically different years and seasons.

Paper III showed water balance of the field being an efficient way to analyze the effects of drainage on field hydrology. Water balance has been used to investigate the effects of annual or seasonal climate on field drainage (e.g. Koivusalo et al., 2017; Jin and Sands, 2001). In Paper III, monthly simulated water balance showed the effect of supplementary drainage on seasonal changes in water outflow routes (Fig. 6), while groundwater depth was used to assess the field drainage efficiency before and after supplementary drainage.

**4.3 Simulating agricultural water management**

**Parameterization of 1D, 2D, and 3D field drainage**

In sections 3.2 and 3.3, field drainage was described in 1D, 2D and 3D simulations, which all together offered new and diverse results about the impacts of field water management on water flow routes. Model applications in each dimension described the in situ field drainage differently through modifications in the structure
of FLUSH where drain discharge is calculated with Darcy’s law and adjusted with drainage entrance resistance (Warsta et al., 2013) as well as soil hydraulic properties around the drains.

Field drainage can be described with the drainage equation (e.g. DRAINMOD, Daisy), Darcy’s law (e.g. Hydrus, FLUSH), and a pipe flow component (e.g. HydroGeoShere). The Hooghoudt drainage equation is popular in 1D models (e.g. Mollerup et al., 2014; Morrison et al., 2014) and even in some 2D applications (Abbaspour et al., 2001), but it does not take into account the effect of unsaturated water flow on drain discharge (Yousfi et al., 2014). Mollerup et al. (2014) showed that the modeling results were similar when using the Hooghoudt equation in 1D and the Darcy approach in 2D to simulate drain discharge. In Paper IV, the Darcy approach was used for simulating drain discharge in 1D soil columns, but required adjusting the model parameters to take into account the effect of the lateral flow on drainage. The adjustments included changes in the values of drainage parameters, hydraulic parameters of the soil matrix-macropore interface, and the horizontal hydraulic conductivity multiplier of the macropore system from previous 3D applications (Turunen et al. 2013). 1D and 2D models in other studies have used soil parameters (e.g. van Schaik et al., 2010), drainage coefficients (e.g. Youssef et al., 2018; Kohler et al., 2001), or model structure (e.g. Yousfi et al., 2014; Frey et al., 2012b; Gärdenäs et al., 2006) to account for the effects of lateral subsurface and surface flow on field drainage.

Papers II and III demonstrated that the soil parameterizations can be changed to take into account the effect of lateral water flow directions on field drainage. The 3D applications simulated all the water flow directions explicitly. The directly measured soil properties were applicable in the 3D simulations (Paper II) but not in the 2D simulations (Paper III). The average values of the soil samples in each layer (Table 3) were more suitable for the 1D and 2D simulations, in which the effect of soil spatial heterogeneity on drainage had to be described through the drainage and soil parameters.

Drainage entrance resistance describes the drainage system effectiveness (e.g. De Schepper et al. 2017; Nousiainen et al., 2015; Turunen et al., 2013), including drain pipe features and characteristics of the envelope material (e.g. Kohler et al., 2001). The effectiveness of the drain pipe must be simulated with the drainage parameters when the computational grid cells are clearly larger compared to the spatial features (such as drain trench) in the vicinity of the pipe (Nousiainen et al., 2015; Turunen et al. 2013). Paper II demonstrated that the trench-scale soil heterogeneities could be simulated with the soil properties. Even though the drain trenches were parameterized in the 2D model applications of Paper III, the drain discharge had to be adjusted with the drainage entrance resistance to avoid over-prediction of the discharge volume. This was probably due to the fact that in the 2D simulations, water was only flowing laterally in one direction compared to the two directions in 3D simulations.

Boundary conditions

Simulated drain discharge occurs only when the water table rises above the drain depth, and the soil around the pipe becomes saturated (Varvaris et al., 2018; De
Schepper et al., 2017). In 3D models, the water table can rise unevenly along the drain lines, which enables simulation of non-equilibrium drainage conditions in the computational area. Therefore, it is possible to describe the drain pipe clogging and exceeding of the pipe capacity spatially within the drainage system (e.g. Kohler et al., 2001). Henine et al. (2010) took into account the drainage capacity by simulating the effect of a pressurized drain pipe on water flow. The approach used by Henine et al. (2010) restricted the drain discharge and facilitated a dynamic boundary condition for the drain discharge. Some modeling approaches take the exceeding of field drainage capacity into account with a constant value of a drain entrance resistance parameter (Nousiainen et al., 2015), pressure in the drain pipe (Paper III), or through the resistance of the porous medium (De Schepper et al., 2017). In Paper II, the simulated drain discharge dynamics indicated that the field drainage capacity was occasionally exceeded as the measured hydrographs were flatter compared to the taller and sharper simulated drain discharge peaks.

The 1D and 2D field drainage simulations typically apply no-flow boundary conditions at the bottom and sides of the computational domain (Frey et al., 2012b; Gärdenäs et al., 2006; Abbasbour et al., 2001). The hydrological effects of the lateral flow are taken into account by considering only drain discharge as an outflow component in the subsurface domain. However, groundwater outflow has been shown in many studies to be an important part of the field hydrology (e.g. Turunen et al., 2013, 2015a; Rozemeijer et al., 2010; van der Velde et al., 2010). In Papers III and IV the drainage effect of lateral outflow in 1D and 2D applications was simulated by adjusting the conditions at the field boundaries: In Paper III, the groundwater outflow was simulated by an open ditch drainage at the field boundary, whereas in Paper IV, the outflow was calculated based on the hydraulic gradient at the boundary. However, depending on the ratio of the computational domain volume and the area of the boundary, soil hydraulic conductivity had to be adjusted in Paper IV to avoid excess outflow from the larger boundary area. In other 2D studies, the drainage effect of the groundwater outflow has been described with a constant percolation coefficient that calculates the groundwater recharge volume (e.g. Mollerup et al., 2014).

The lack of surface water flow directions in 2D and 1D models needed to be taken into account by adjusting the parameterization related to the upper boundary condition, i.e., the interface between the surface and subsurface domains. A higher infiltration rate was applied in the 2D (Paper III) and 1D (Paper IV) simulations by increasing the depression storage capacity compared to the 3D simulations (Paper II, Turunen et al., 2013).

The drainage effect of macropores

Comparing model results with and without a preferential flow description has shown the hydrological effect of the macropore system on the generation of drain discharge (e.g. Varvaris et al., 2018; Gärdenäs et al., 2006; Haws et al., 2005). The simulations with a macropore system seem to result in faster changes in the soil profile moisture conditions (van Schaik et al., 2010). Paper II showed that the drain discharge dynamics were the most different between the case where
macropores and parameterization of drain trenches did not exist (SP model without trenches in Fig. 5) and the case where macropores and drain trenches were parameterized into the computational domain (DUP model with trenches in Fig. 5). Varvaris et al. (2018) reported that similar results were obtained from the simulations with (DUP) and without (SP) a macropore system after both models were separately calibrated to the data. The drainage effect of a highly permeable drain trench can be taken into account by using a macropore system (DUP model) or parameterization of a soil with a trench (SP model). Larger catchment-scale simulations might fail to simulate local scale dynamics of tile drain discharge if heterogeneities in soil hydraulic properties (e.g. drain trenches) are missing from the description of the field (e.g. Hansen et al., 2013).

In Paper II, the SP model with a trench simulated drain discharge dynamics successfully without a macropore system. However, Gerke et al. (2007) showed that it was not possible to simulate solute transport correctly with a single pore system model in a macroporous soil. The solute transport simulations in Paper IV were only conducted with the dual-permeability model. The lack of a soil heterogeneity description in 1D models has been seen as a problem in simulating solute transport (Vogel et al., 2000). However, the results in Paper IV showed that it was possible to simulate N transport using a 1D model application when the hydraulic effect of spatial heterogeneity (e.g. drain trench) was taken into account through dual-permeability modeling.

4.4 Nitrogen leaching from cultivated fields

Factors affecting the N leaching during autumn

Comparison of N transport simulation results between the two autumn periods (Paper IV) pointed out the effect of initial N storages on N leaching at the beginning of the autumn. Gustafson (1988) reported that between-year variation in N leaching rates could be explained by the amount of NO3-N remaining in the soil profile after harvest. In Paper IV, initial values of the N storages were set according to findings of Pietola et al. (1999), but adjusted based on the N balance calculations in the Nummela field (Äijö et al., 2014). Harvest was not fully successful in 2011 due to heavy rains occurring at the end of the growing period (Äijö et al., 2014), which was described in the simulations as higher N storages in the beginning of autumn.

The timing of the precipitation events was another explanatory factor for the NO3-N concentration peaks in drain discharge during autumn (Fig. 7 in Paper IV). More precisely, the explanation seemed to be related to the dry phases between the rain events (Paper IV). Gustafson (1988) concluded that some of the N leaching was explained by mineralization outside of the growing season, which could also explain the NO3-N concentration dynamics in Nummela. During the dry phase, NO3-N was accumulated in the soil due to mineralization and nitrification (Fig. 8) and then washed away by the following precipitation event. Patil et al. (2010) reported the short-term effects of soil warming and precipitation patterns on N leaching based on lysimeter experiments and found warmer winters to decrease N leaching even though precipitation amounts increased. The finding of
Discussion

Patil et al. (2010) can be an indication of higher amounts of gaseous N leaving the soil in wet and warm conditions, which are optimal for denitrification of NO₃-N.

The effect of field practices on N transport and leaching was noted from the drain discharge concentrations during autumn 2011 (Fig. 7 in Paper IV). Repairing of the topsoil layer runoff collectors took place just before the second concentration peak in drain discharge. The maintenance of the collectors likely influenced the NO₃-N transport in the soil, which was not taken into account in the model simulations showing weaker performance in section D (Fig. 7 in Paper IV). Field studies have reported that after drainage installation, N concentrations increased during the following year, but then gradually decreased close to the original level (Äijö et al., 2017). Soil ploughing and digging can mobilize soil N fractions, i.e. increase mineralization of organic N, as a result of the disturbance of the soil aggregates (Turtola and Paajanen, 1995).

Subsurface drainage changes the N loading pathways

Soil disturbance might have a short-term effect on N leaching, but the long-term increase in N loads due to subsurface drainage installation (e.g. Äijö et al. 2017; Seuna and Kauppi, 1981) can be explained by the increased drain discharge volume (e.g. Bednorz et al., 2016). Many studies have found out that controlled drainage could reduce N loads by lowering the drain discharge volume (e.g. Wesström et al., 2014; Salazar et al., 2009; Paasonen-Kivekäs et al., 1996). On the other hand, the autumn period simulations (Section 3.4) showed that a lower NO₃-N load via drain discharge resulted in a higher NO₃-N load via groundwater outflow (i.e. load via drain discharge was 1.4 and 0.7 times the load via groundwater outflow in field sections B and D, respectively). Rozemeijer et al. (2010) showed that even though drain discharge was the main N leaching route, groundwater outflow also contributed to the total load to surface waters. Hansen et al. (2019) showed that subsurface drainage decreases nitrate reduction in the subsurface domain due to increased nitrate outflow via drains.

Even though water outflow routes seem to determine the N leaching pathways, and decreasing water outflow is recommended by using controlled drainage, most studies pay little attention to the remaining nitrogen (Carstensen et al., 2016). It is very likely that the higher moisture content in the soil, resulting from the less intensive drainage, increases gaseous N losses and N transport to the groundwater. Frey et al. (2016) observed transport of NO₃-N into the shallow groundwater when drainage was less intensive. Based on the simulation results in Paper IV, the main outputs for N transport were drain discharge and groundwater outflow (Section 3.3) as well as denitrification. The poorly drained field section D had higher denitrification rates (Table 5), which indicates that better soil drainage can decrease gaseous N losses. According to the simulations in papers III and IV, subsurface drainage improvements could transfer the N load from the groundwater outflow to drain discharge, i.e. the total N load from the field would not change.
For the autumn simulations, it was necessary to restrict the N processes according to temperature and moisture factors, which are used to describe the seasonal differences in the N processes. The moisture restriction factor resulted in N balance differences between the field sections (Table 5), as higher soil moisture in section D resulted in higher denitrification (i.e. decreasing the NO3-N soil storage). Lower soil moisture in section B increased NO3-N storage because of lower denitrification and higher nitrification rates. In this study, the soil moisture limiter allowed nitrification to occur if the soil water content was below saturation. However, Kroes and van Dam (2003) used a threshold value allowing decay only when hydraulic head was -1 m. Therefore, the volume of nitrification might have been overestimated as nitrifying bacteria are more active when soil is drier than -60 m pressure head (Stark and Firestone, 1995).

Patil et al. (2010) reported that soil warming resulted in higher soil NO3-N storage after harvest when crop N uptake no longer occurred. Rankinen et al. (2016) pointed out the increase of N leaching due to higher temperatures during autumn. The two autumn period simulations in Paper IV could not confirm the role of changes in autumn temperature on N leaching (Section 3.3). In the simulations the decrease of soil NO3-N storage occurred at the end of the autumn period when temperature was low (figs. 7 and 8). The other main driver for the NO3-N leaching and losses via drain discharge and groundwater outflow were the precipitation events. On the other hand, Rankinen et al. (2016) reported that an increasing trend in winter runoff did not increase N leaching. The simulation results in Paper IV and the literature on the effects of soil moisture and temperature on N leaching indicate that the soil moisture content could have a more pronounced effect on N losses (from the soil to surface water or to air) compared to soil temperature.

The performance of the N transport model

The developed N transport model was able to quantify the chemical reaction chain from organic to nitrate N. Many N simulation studies do not report the transport of NH4-N due to its small effect on total N load (e.g. Bednorz et al., 2016). Most likely, the minor effect is caused by the small quantities of NH4-N in the soil and the sorptive nature of NH4-N. Still, NH4-N loads estimated from field data were up to 10% of the NO3-N loads (Table 5). Salazar et al. (2009) reported that mineralization of organic N to inorganic N increased due to higher soil water content, which at the same time decreased nitrification. The findings of Salazar et al. (2009) would mean that NH4-N storage increases as a result of higher soil water content favoring mineralization over nitrification reaction. This would highlight the need to take into account transport of both NO3-N and NH4-N.

Simulated cumulative NO3-N and NH4-N loads were overestimated compared to loads estimated from measurements (see Table 5). Measured loads were calculated based on concentrations of flow-weighted bulk samples that flatten the highest peak values in the measurements. Even a small difference (0.5 mg N l\(^{-1}\)) between the simulated and measured concentrations caused 60% overestimation of
a short-term NO₃-N load via drain discharge. However, when considering periodical or annual volumes, the simulation results can be seen to be reliable (e.g. Rankinen, 2006).

4.5 Future considerations

*FLUSH development*

The model applications revealed further model development and testing issues. Currently, FLUSH does not simulate the effect of soil freezing on hydraulic conductivity. Simulating the effect of these winter time processes on soil water movement could enhance the groundwater level predictions during the periods of soil frost. In both Nummela (Paper III) and Sievi (Paper I) the wintertime groundwater level descended below the drain level as a result of groundwater outflow (Turunen et al., 2013) or soil frost (Sheng et al., 2013).

The next step for the N transport studies is to simulate the year-round N cycle processes, which requires developing a crop N uptake processes and a simple crop growth submodel. Furthermore, the solute transport component should be tested and re-calibrated for 2D or 3D model applications. This study did not consider the role of discontinuous macropores in terms of drainage and N loads, which would require developing routines for simulating flow and transport in discontinuous macropore networks.

Water management practices such as controlled drainage and sub-irrigation could be implemented in the FLUSH model enabling the assessment of these practices on water balance and N processes in Nordic conditions. Controlled drainage has been shown to decrease N leaching, but field experiments and modeling studies have not reported where the remaining amount of N fractions end up (e.g. Carstensen et al., 2016).

Increasing the size and resolution of the computational domain increases the simulation times from minutes (in Paper IV) and hours (in Paper II) to days (in Paper III). Other complicated water flow and solute transport models have experienced similar obstacles (e.g. Hwang et al., 2019; Šimůnek et al., 2017; De Scheper et al., 2015), underlining the need for more efficient algorithms for numerically solving the governing partial differential equations.
5 Conclusions

The conclusions regarding the objective of understanding the effects of field drainage installation methods and supplementary drainage on water flow and nitrogen transport are:

1. Improved drainage by installing new subsurface drains between the old drains increased drain discharge more than it decreased surface runoff. According to the results, the additional water originated from groundwater outflow, which was decreased by the narrower drain spacing (Fig. 6).

2. The model simulations revealed that in macroporous clay soil, variability of soil properties had a greater effect on water outflow pathways than the applied drainage scheme (Fig. 5).

3. The drainage performance within the same field was affected by the drainage installation method as well as the soil type differences between the field sections. In the studied conditions (loam, loamy sand and sandy loam), the trencher method produced slightly more efficient drainage.

4. The site-specific features had a high impact on the results. It is relevant to design locally suitable solutions and to understand the site-specific features affecting water and N management at the field.

5. The increased N outflow via drain discharge after supplementary drain installation does not necessarily affect the total load from the field; it moves the load from the groundwater outflow to the drain discharge route.

6. The autumn period simulations showed that the main factors affecting NO3-N leaching were the initial N storages at the beginning of the autumn and the timing of the precipitation events.

For the objective of selecting appropriate methodology for studying field drainage and N transport at field scale, the conclusions are:

1. The applied statistical analysis could not be used to decipher the reasons for the observed drainage performance differences. The benefit of 1D, 2D, and 3D modelling with varying soil parameterizations was the systematic testing of the effects of drainage design and soil properties on drainage performance.

2. When the hydrological model was tested with application of measured soil parameters, the DUP model structure produced smaller variability in the simulation results with different parameterizations than the SP model structure. These results reflected the dominating nature of
macropores in water flow processes for structured soils with or without drain trenches. DUP model structure can be used to describe the drain trench impact on field drainage in 1D, 2D, and 3D model applications, but in SP models the effect of drain trench can be included only in 2D and 3D applications.

3. The dual-permeability model combined with the novel solute transport model and the configured N model was able to realistically reproduce N concentrations at the field outlets including subsurface drains. The successful solute transport results with the 1D DUP model indicate a proper flexibility of the model structure and parameterization when simulating water flow and solute transport. Inclusion of temperature and moisture restrictions on the N processes is important. Cold temperatures outside the growing season and wet conditions in the autumn and spring influence the N processes in the simulations.

Field drainage can be an indirect measure to reduce overall N loads from the field and efficient drainage is a way to increase crop production. Narrower drain spacing enables earlier start for cultivation after the snow melt period. Poor drainage conditions during wet growing periods might result in low crop yields leaving nitrogen in the field which has a leaching risk during the following autumn rains. Based on the simulation results presented here, there might not be a conflict between environmental benefits (reducing the nitrogen loads) and increasing the crop production (draining the fields).
Appendix A Maps of the study fields

Figure 9. Nummela field. Field borders are marked with black dash line.

Figure 10. Sievi field. Field borders are marked with black dash line.
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