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Impacts of ditch cleaning on hydrological processes in a drained peatland forest

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Abstract. One fourth of the forests in Finland are growing on drained peatlands. Forestry operations such as ditch network maintenance increase the export of suspended solids and nutrients, and deteriorate water quality in lakes and rivers. Water protection presupposes an understanding of how forestry operations affect peatland hydrology. The objective was to study the hydrological impacts of ditch cleaning on the basis of water table level and runoff measurements from two pairs of artificially delineated catchments in drained peatland forests in Finland. Data from treated and control catchments indicated that ditch cleaning lowered the level of the water table in sites where a shallow peat layer was underlain by mineral soil. In sites with deep peat formation, the water table showed no detectable response to ditch cleaning. Runoff data suggested that annual runoff clearly increased after ditch cleaning, which was in conflict with the previously reported small impacts of ditch network maintenance. The hydrological model FEMMA was calibrated and applied to assess the conformity of the data and the experimental setup. In the model application, the catchments were assumed to behave as independent hydrological units. However, assessment of the model results and the measurements suggested that ditch cleaning had an impact on hydrological measurements in both treated and control catchments. It appeared that the independence assumption was violated and there was a hydrological connection between the artificial catchments and, therefore, the results of the data analysis were considered misleading. Finally, a numerical experiment based on the model simulations was conducted to explain how the assumed relationship between soil moisture and transpiration is reflected in the modelled runoff. Modelled runoff decreases and evaporation increases when ditches are cleaned in poorly drained sites, where the initial ditch depth is small and the depth of a highly conductive topsoil layer is low. The numerical experiment can be applied to assess when ditch cleaning does not improve evapotranspiration and is unnecessary.

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1 Introduction

Forest cuttings in the middle of the 20th century exceeded the annual growth of the stock volume in Finland. In order to increase timber production, ditch drainage was introduced in peatland areas where poor aeration in the rooting zone restricts the growth of the trees. In the late 1960s' and early 1970s' about 2500 km^2 of pristine peatlands were drained annually (Kenttämies, 2006). The drainage activities subsequently gradually decreased toward the end of the 1990s', when pristine peatlands were no longer drained. The ditches deteriorate over time (Robinson, 1986; Hökkä et al., 2000), thereby decreasing the growth of the tree stands. The problem can be rectified by maintaining the ditch network, i.e. digging complementary ditches or cleaning old ditches. The need for ditch network maintenance has increased and is currently estimated to be about 1600 km^2 per year (Tomppo, 2005). According to the 9th National Forest inventory, one fourth of the managed forests in Finland are located on peatlands (Tomppo, 2005). The share of peatlands drained for forestry is about 54% of the total peatland area (100 000 km^2) in Finland.

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The hydrological effects of draining pristine peatlands have been studied widely in Finland (e.g., Kaitera, 1955; Mustonen and Seuna, 1971; Seuna, 1980; Starr and Päivänen, 1981; Ahti, 1987) and elsewhere (e.g., Robinson, 1986; Lundin, 1994; Prévost et al., 1999; Holden et al., 2006). Holden et al. (2004) provide an extensive review on the effects of drainage on the hydrological and hydrochemical processes of peatlands. The drainage of peatlands has both short- and long-term effects on hydrological processes, and the magnitude and direction of the effects are dependent on local conditions. Many studies report increases in low flow rates after draining peatlands (Mustonen and Seuna, 1971; Seuna, 1980; Ahti, 1987; Johnson, 1998; Prévost et al., 1999), but the impact of drainage on the peak flows reported in different studies varies. Peak flows after drainage have been found to increase (Mustonen and Seuna, 1971; Seuna, 1980; Ahti, 1987), decrease (Kaitera, 1955; Lundin, 1994), or to be unchanged (Prévost et al., 1999). Field studies have demonstrated that differences in soil hydraulic properties, meteorological conditions, vegetation cover and drainage design, can affect the direction of change in the hydrological response. Robinson and Rycroft (1999) reviewed the mechanisms underlying the impacts of drainage and stated that, in addition to site characteristics, also catchment scale properties, such as the location of the drained area and alterations in the main channel are reflected in the runoff response. In the long-term the drainage of peatlands increases growth of the tree stands (Seppälä, 1969; Hökkä, 1997). The increased height and leaf area index (LAI) of the tree stand result in higher canopy interception and transpiration, which gradually leads to a decrease in runoff volumes (e.g., Koivusalo et al., 2006). A well-growing, densely-stocked tree stand may play a decisive role in the water balance of a drained peatland (Ahti and Hökkä, 2006). Päivänen and Sarkkola (2000) suggested that maintenance of a ditch network is not necessarily required when the volume of the growing stock is sufficiently large to maintain efficient interception and transpiration. An important question for practical forestry is to assess whether ditch network maintenance is required, or whether the water uptake of the forest stand is sufficient to maintain favourable moisture conditions in the rooting zone. In addition to the growth of the tree stand, the vegetation colonisation in the ditches can reduce runoff in the long term, owing to the decrease in ditch depths and drainage efficiency (Robinson, 1986). Peatland drainage also has long-term effects on the structure of the topmost peat layers (Silins and Rothwell, 1998; Holden et al., 2006). The decreasing soil moisture content enhances decomposition and subsidence of the peat layer, and alters the bulk density, water retention characteristics, and pore-size distribution of the peat. These changes can lead to a decrease in the hydraulic conductivity of peat (Silins and Rothwell, 1998).

Although the hydrological effects of draining pristine peatlands have been comprehensively studied, the hydrological effects of ditch network maintenance are not fully known. Joensuu et al. (1999, 2001, 2002) investigated how ditch maintenance affects runoff and water quality by comparing infrequent (biweekly – monthly) measurements from a large set of small catchment pairs. Clear effects of ditch maintenance on water quality were detected, but the effects on measured runoff were not visible. Päivänen and Sarkkola (2000) also suggested that ditch network maintenance combined with stand thinning has minor impacts on the hydrology of peatlands in terms of water table elevation. Ahti and Päivänen (1997) reported that ditch network maintenance alone results in a drop of only 0.05 m in the highest levels of the water table. Lundin (1994) studied the effects of forest clear-cutting and remedial drainage on flow regimes in catchments partly covered with peat or shallow layers (<0.3 m) of organic material and partly covered with mineral soil (till). Mean annual runoff clearly increased after clear-cutting, and to a lesser extent after the subsequent remedial drainage of the ditch network.

The data and methods used in earlier experimental studies do not support an evaluation of the effects of ditch maintenance on peatland hydrology as a whole. Combining experimental data with hydrological modelling provides one option for assessing and understanding how forest management practices affect hydrological processes (see e.g., Kokkonen et al., 2006; Koivusalo et al., 2006; Laurén et al., 2005). Meteorological and hydrological measurements are bound together in the hydrological model, which allows the user to assess the suitability of the model structure, the quality of the data, and the consistency of the experimental setup. Poor performance of the model can indicate deficiencies in the model structure. An abrupt or gradual change in the model performance, on the other hand, can be an indicator of inconsistency in the data or in the experimental setup. A successful model application produces information for understanding the processes behind the observations. In a hydrological model, the drainage of forested peatlands can be described using routines implemented in agricultural water management models (e.g., Skaggs, 1980; Jarvis, 1994; Oztekin et al., 2004; Oosterbaan et al., 1996). Amatya et al. (1997) and Skaggs et al. (2006) implemented a ditch drainage scheme of this sort in a forest ecosystem model. Dunn and Mackay (1996) presented an application of a catchment scale simulation model, which showed that ditch drainage has counteracting effects on the generation of surface runoff and subsurface flow. The counteracting effects can change the direction of the drainage impact on runoff response.

This study exploits experimental data from a drained peatland area in central Finland, where four artificial forested catchments were created using ditch delineation. Two of the catchments were subjected to ditch cleaning twenty years after the initial drainage. The objectives of the study were 1) to identify the impact of ditch cleaning on water table levels and runoff on the basis of experimental data, 2) to calibrate a hydrological model against the data, and to assess how model performance reflects the data and the conformity

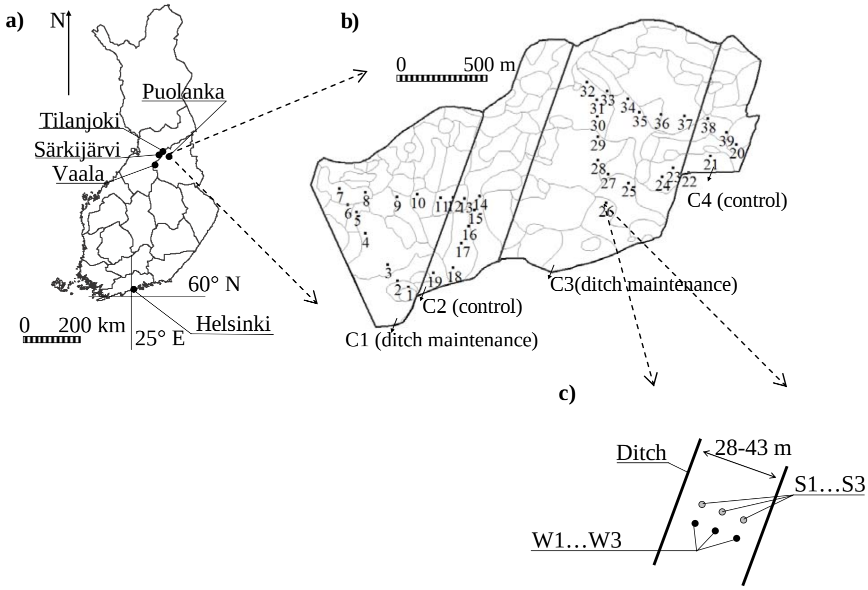

Fig. 1. Location of Tilanjoki and the weather stations in Särkijärvi, Puolanka, and Vaala **(a)**, layout of 4 research catchments (C1…C4) and 39 measurement sites **(b)**, and layout of a measurement site between two ditches. W1…W3 refer to water table measurement locations and S1…S3 to snow measurement points. Forest compartments with different tree stand properties are delineated with grey lines in (b).

of the experimental setup, and 3) to derive the effect of ditch cleaning on annual runoff using a model-based, numerical experiment. The model calibration is based on the approach of Generalised Likelihood Uncertainty Estimation (GLUE) outlined in Beven and Binley (1992) and Beven and Freer (2001). GLUE is introduced to identify multiple parameter sets that produce acceptable predictions of water table level and runoff, to study the sensitivity of model performance to calibration parameters, and to identify the limitations of the model structure.

2 Site description and field data

The experimental peatland area of Tilanjoki is located on the border between two municipalities, Utajärvi and Puolanka, in Finland (Fig. 1a). The long-term (1971–2000) mean annual temperature in the area was 1°C and precipitation 550 mm/a. The peatlands were drained for the first time in 1969, four experimental catchments were delineated in 1983, and ditch cleaning was conducted in two of the catchments (C1 and C3) in autumn 1989 (Fig. 1b). The catchments in which ditch cleaning was carried out, and the control catchments where the ditches were not treated, were set up in Tilanjoki to facilitate a paired catchment analysis of runoff and the export of solids and solutes. The areas of catchments C1, C2, C3, and C4 were 0.74, 0.50, 0.99, and 0.28 km^2, respectively. The spacing of the ditches ranges from 28 to 43 m, the depth of the ditches prior to the cleaning was 0.3–0.5 m, and 0.8 m after the cleaning.

Runoff at the outlet of each catchment was measured using v-notched weirs and limnigraphs plotting the height of the water level at the weir. Inside the four catchments there were altogether 39 measurement sites, where snow depth and depth of the level of water table were measured at three points in each site (Fig. 1b, c). In addition to the snow depth measurements, snow water equivalent (SWE) was monitored at one location in each site. The snow and groundwater level measurements were made once every 1–2 weeks.

Daily meteorological data, including precipitation, air temperature, relative humidity, wind speed, and cloudiness, were available from nearby weather stations operated by the Finnish Meteorological Institute in Särkijärvi (Utajärvi), Vaala and Puolanka (Fig. 1a). In addition to these data, chart records of air temperature were available from two on-site stations near the outlets of catchments 1 and 4. The hydrometeorological variables were measured from 1983 to 1994.

Tree stands in the study catchments are dominated by Scots pine (*Pinus sylvestris* L.), with a minor admixture of pubescent birch (*Betula pubescens* Ehrh.) and Norway spruce (*Picea abies* L. Karst.). Stand characteristics, as well as the characteristics of the dominant trees (100 largest trees per ha), including height, diameter at breast height (DBH), and tree density, were measured on sample plots established in the 39 measurement sites. In 1983, 1989, and 1995 the proportion of Scots pine out of the total stand volume was 91%, 90%, and 89%, respectively. The stand volumes in the measurement sites ranged from 1.6 to 154 m^3/ha, and the median volume was 30 m^3/ha in 1983. The bottom layer vegetation consisted of *Spaghnum* moss and the field layer vegetation of sedges and dwarf shrubs.

The Tilanjoki area is characterized by a shallow peat thickness in most of the measurement sites. In this study, topsoil refers to the shallow peat layer, except in two sites (7 and 8) with a deep peat formation, where the topsoil refers to the peat layer above the depth of the ditches. Subsoil refers to all the material below the topsoil. Subsoil is composed of peat, till, or sand, and their spatial distribution within the catchments was interpreted from ground penetrating radar data.

3 Methods

3.1 Analysis of water table level and runoff data

Water table and runoff measurements were analysed in order to detect the main hydrological impacts of ditch cleaning. In each catchment the water table data were classified according to the three subsoil types (peat, till, and sand). For each subsoil type and catchment, the median water table level was computed for all measurement occasions. Median was used instead of average, because the water table frequently dropped below the depth of the measurement tubes, resulting in censored observations. The median value is not as sensitive to censored data as an average value. The annual mean values of the median water table levels before and after ditch cleaning were computed, and the non-parametric Mann-Whitney U test was applied to assess whether the level of the water table changed between the pre- and post-treatment periods. The non-parametric test was also applied to determine whether the difference between the catchment pairs (C2-C1 or C4-C3) changed between the pre- and post-treatment periods for each subsoil type. The significance of the change was assessed using a risk level of 5%. In the computation of annual water table depth, the censored values were replaced with the measured maximum water table depth of the observation tube. The maximum value was used because the effective length of the measurement tubes was not known. The data from the first year (1983) were omitted from the analysis due to the sedimentation of soil material at the bottom of the tube after installation.

A paired catchment analysis between C1 and C2, and C3 and C4 (Fig. 1) was conducted to determine changes in the annual runoff volumes following ditch cleaning. The pre-treatment (1983–1988) data were used to form a regression model between the time series from the control and treatment catchments (e.g. Nieminen, 2004). For the post-treatment period, the regression model and the data from the control catchment were used to predict runoff for the treatment catchment under the assumption that the treatment had not occurred (e.g. Watson et al., 2001). The difference between the observed and predicted values was assumed to be a measure of the treatment effect. In the computation of annual runoff the data were accumulated only for those days when records were available from both the treated and control catchments. Because of the missing data, the estimate of annual runoff in this case is an underestimate compared to the actual runoff from the catchments.

In a paired catchment analysis, the treated and control catchments need to be relatively similar in terms of area, topography, geology and vegetation, and the catchments must not have a connection linking their hydrological behaviour. When these criteria are met, the catchments can be assumed to have a similar response to manipulation (Cosby et al., 1996).

3.2 Hydrological model

FEMMA (model for Forestry Environmental Management) consists of sub-models for interception and transpiration in the overstorey and understorey vegetation layers, snow accumulation and melt, soil- and ground water interactions, and stream discharge (Koivusalo et al., 2005; Laurén et al., 2005). In the current study, daily time series of air temperature, precipitation, relative humidity, wind speed, and downward short and long-wave radiation were used as input data. For the current study, FEMMA was modified in order to 1) improve the description of the canopy model for young and sparse peatland forests, 2) facilitate the computation of the drainage flow as a water balance component, 3) improve prediction of the water table level, and 4) formulate a spatial description of a modelling domain for a drained peatland forest. The following paragraphs briefly address these modifications, together with the general description of FEMMA.

3.2.1 Canopy and snow models

Based on input data characterising the meteorological conditions above the canopy, the canopy model simulates downward short and long-wave radiation, wind speed, and throughfall beneath the forest canopy. Relative humidity and air temperature are assumed to be unaffected by the canopy. The process descriptions are given in detail in Wigmosta et al. (1994), Koivusalo and Kokkonen (2002), and Koivusalo et al. (2006).

The canopy model accounts for the interception of rainfall and snowfall in the overstorey vegetation (trees), and for the interception of rainfall in the understorey vegetation (field and bottom layer). Whenever the ground is snow-covered, interception in the understorey is disregarded. The stand density gives the proportion of the ground that is covered by the overstorey. In the current version of the canopy model, the method presented by Raupach (1994) and Schaudt and Dickinson (2000) was applied to parameterize the zero plane displacement height and the roughness height as a function of the stand density (canopy closure) and crown ratio. The parameterization ensures that aerodynamic resistance decreases when canopy closure approaches either full coverage in a dense forest or zero in a very sparse forest. The density of the understorey canopy was set to the value of one.

Potential evaporation of intercepted water is computed separately for the overstorey and understorey vegetation according to a combination equation of the Penman-Monteith type, where the stomatal resistance is set to zero. Evaporation of the intercepted water occurs at the potential rate until all the intercepted water is depleted. Transpiration, which is initiated after the canopy has become dry, is controlled by the stomatal resistance. The stomatal resistance is controlled by the leaf area index (LAI), soil temperature, water vapour pressure deficit, photosynthetically active radiation (PAR), and soil moisture (see Sect. 3.2.2). Evaporation from the soil surface was neglected, because the moss vegetation and undecomposed litter covering the ground were assumed to block evaporation from the peat surface.

The snow model simulates the snow surface energy balance, heat conduction through the snowpack into the soil, snowmelt, liquid water retention in the snow, melt water discharge out of the snowpack, and compaction of the snow. The snow model is described in more detail in Koivusalo et al. (2001, 2006).

3.2.2 Characteristic profile model

Soil and ground water interactions in FEMMA are described on the basis of the characteristic profile approach of Karvonen et al. (1999). In the case of a drained peatland, the characteristic profile is a vertical one-dimensional column residing between the drainage ditch and the midpoint between two parallel ditches. Soil water movement and runoff generation processes are simulated using daily series of throughfall/snowmelt available from the canopy and snow sub-models. The characteristic profile model is quasi-two-dimensional in the sense that vertical and lateral water fluxes are computed alternately. The soil column is divided vertically into soil layers and the water fluxes between the layers are computed according to the Richards equation (Richards, 1939). Transpiration is extracted from the soil layers residing within the rooting zone. Infiltration into a soil column is controlled either by the current air-filled pore volume or the hydraulic conductivity of the topsoil layer. Water that cannot infiltrate is transported laterally to the ditch as surface runoff.

In order to simulate the effect of drainage on transpiration, the relationship between soil moisture and transpiration was changed from earlier applications of FEMMA. Schwärzel et al. (2006) studied moisture dynamics and evapotranspiration in a drained peatland and presented a relationship between the rooting zone pressure head and the ratio of actual and potential evapotranspiration. The relationship was adopted in FEMMA to characterise how excessive soil moisture or soil drying in the rooting zone decrease transpiration. The stomatal resistance r_s is given by

$$r_s = r_{s\min} f_1^{-1}(T_{\mathrm{soil}}) f_2^{-1}(\Delta e) f_3^{-1}(\mathrm{PAR}) f_4^{-1}(\theta) \tag{1}$$

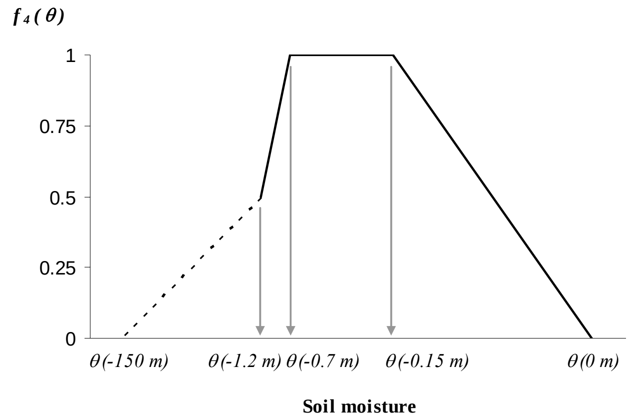

Fig. 2. Relationship between θ (pressure head) and function $f_4(\theta)$, where θ is the soil moisture content. Soil moisture does not limit transpiration when $f_4(\theta)$ is equal to 1.0.

where $r_{s\min}$ is the minimum stomatal resistance, $f_1(T_{\mathrm{soil}})$ is a function describing the influence of the soil temperature T_{soil} on r_s, $f_2(\Delta e)$ defines the influence of the vapour pressure deficit Δe on r_s, $f_3(\mathrm{PAR})$ defines the influence of the photosynthetically active radiation (PAR) on r_s, and $f_4(\theta)$ depicts the influence of the soil moisture θ on r_s. The functions $f_1(T_{\mathrm{soil}})$, $f_2(\Delta e)$, $f_3(\mathrm{PAR})$ are given in Nijssen et al. (1997) and the soil moisture function is illustrated in Fig. 2. When the pressure head of a computation node in the rooting zone is between –0.15 m and –0.70 m, soil moisture does not limit transpiration. Schwärzel et al. (2006) sketched the relationship down to a pressure head of about –1.2 m, where the ratio of actual and potential evapotranspiration is about 0.5. In this study, the function $f_4(\theta)$ was assumed to further decrease toward zero when the pressure head approaches the wilting point (dashed line in Fig. 2).

After the vertical water fluxes and the resulting groundwater level of a column are solved, the lateral water flows to drainage ditches are computed. When the soil is fully saturated and surface runoff is generated, surface runoff entering the ditch is delayed using a linear store. Lateral drainage flow within the soil column is computed according to Hooghoudt's drainage equation (e.g., El-Sadek et al., 2001). The method assumes steady state recharge and drainage fluxes, and allows a description of soils with different values of an effective saturated hydraulic conductivity above and below the ditch depth. The effective saturated hydraulic conductivity is computed by dividing the total transmissivity above or below the drain depth with the depth of the corresponding saturated layer. The water level in the ditch is set equal to the elevation of the ditch bottom, and it prescribes a boundary condition for the drainage flow computation. Ditch cleaning changes the boundary condition when the ditches are dug deeper. Channel flow processes and dynamics of the water level in the ditches, as well as gradual

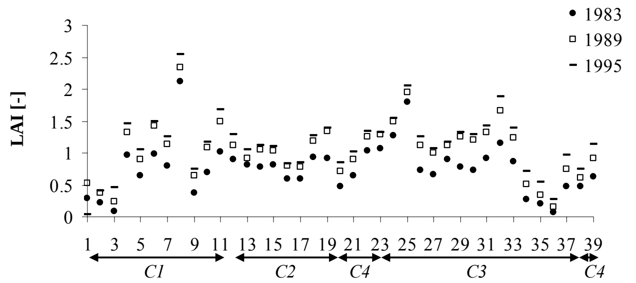

Fig. 3. Estimated values of LAI in years 1983, 1989, and 1995 in the 39 measurement sites at Tilanjoki. Catchments C1 and C3 were subjected to ditch cleaning in 1989, and catchments C2 and C4 were control catchments.

decline of the ditch depth resulting from vegetation colonisation, are disregarded in the model.

In earlier applications, the lateral groundwater flow was included in the model to account for subsurface flow in saturated soil (e.g. Kokkonen et al., 2006). In the current study, drainage flow was assumed to be the only lateral subsurface flow mechanism in peatlands drained using open ditches. After the drainage flow ceases no groundwater flow occurs. The sum of the two runoff components entering a ditch – surface runoff and drainage flow – forms the total runoff.

3.2.3 Assumptions behind the parameterisation of ditch cleaning

In FEMMA only the direct hydrological effects of ditch cleaning were considered. It was assumed that subsidence of the peat mainly occurred during the years following the initial drainage, and that the temporal change in peat structure during the five-year periods preceding and following the ditch cleaning is small and does not have a large impact on the hydrology of the peatland. The saturated hydraulic conductivity was assumed to be significantly higher in the topmost soil layers compared with the subsoil, and this difference was not influenced by ditch cleaning. The effect of ditch cleaning on forest growth was not simulated. Temporal changes in forest properties, such as LAI, canopy density, and tree height, were estimated on the basis of the stand measurements (see Sect. 3.4.). Understorey vegetation was assumed to adapt immediately to the changed soil moisture conditions, i.e. there was no degeneration of old species or invasion of new species. Both overstorey and understorey transpiration were limited by excessive soil moisture conditions or soil drying, as described in Fig. 2. Changes in channel flow processes caused by ditch cleaning were ignored, because the flow delay caused by the ditch network is likely to be shorter than the daily modelling time step used in the study.

3.3 Parameterisation of the experimental catchments

One characteristic profile, i.e. a soil column between two parallel ditches, was parameterised for each measurement site where three snow depth and water table level measurements, and one SWE measurement were available. As the small number of snow observation points per site did not warrant the separate calibration of the canopy and snow models for each site, the sub-models were not calibrated against the snow data. Instead, the parameters for the snow model were adopted from Koivusalo et al. (2006).

The input data for the snow and canopy models were compiled from the closest weather stations. Downward short- and long-wave radiation fluxes were estimated on the basis of air temperature, simulated clear-sky radiation, and cloudiness index (see e.g. Tarboton and Luce, 1996). Daily air temperature was derived from both temperature graphs measured on-site and from the closest weather station. The validity of the on-site temperature measurement was assessed on the basis of the snow model results as explained in Sect. 4.2.1.

Forest stand characteristics at each site were inventoried in 1983, 1989, and 1995. The stand properties between the measurement times were estimated with linear interpolation. In order to derive LAI at each site, the needle biomass of Scots pine was computed from the stand properties in the following way. A two-parameter Weibul distribution characterizing the stand DBH distribution was fitted against the measured arithmetic mean DBH of the stand and the mean DBH of the dominant trees. Once the Weibul distribution was created, the needle biomass was computed for ten discrete DBH classes using the biomass function proposed by Hakkila (1979). The biomass for different DBH classes was subsequently multiplied by the stem number and the specific needle area to produce the estimate of LAI. Finally, the relationship between the effective winter leaf area index and the forest density (Pomeroy et al., 2002) was applied to derive canopy closure directly from the LAI estimate. Figure 3 illustrates the distribution of LAI in the measurement sites. The estimated values of LAI and the canopy closure were used in the parameterisation of the overstorey vegetation. The LAI for the understorey vegetation was fixed to a value of 1.0.

The parameters of the functions controlling stomatal resistance were fixed to the values reported in Nijssen et al. (1997), except for the parameter defining the minimum stomatal resistance (r_{smin}), which was calibrated as explained in Sect. 3.4. The rest of the canopy model parameters were set according to Koivusalo et al. (2006), with the exception of the new parameter, the crown ratio, which was set to a value of 3.5 (see e.g., Schaudt and Dickinson, 2000).

Separate water retention curves were described for the peat layer from the surface down to a depth 0.3 m and for the layer below the depth of 0.3 m. The water retention characteristics for the peat layers were adopted from Päivänen (1973), who tabulated water retention characteristics for *Sphagnum* peat with different bulk densities and degrees of humification. The water retention characteristics of samples 173–178 (Päivänen, 1973) with a bulk density of 47 kg/m^3 were adopted for the upper peat layer, and the water retention characteristics of samples 97–100 with a bulk density of 108 kg/m^3 for the lower peat layer. In the current study, parameters of the van Genuchten (1980) function were fitted against the data from Päivänen (1973). Temporal changes in peat characteristics were neglected. The water retention characteristics for mineral soils were derived by using the on-site measurements of particle size distribution and the relationship presented by Jauhiainen (2004).

Peat is characterised by a high hydraulic conductivity in the top soil layer, and the conductivity typically decreases with depth (e.g. Päivänen, 1973; Ahti, 1987; Lundin, 1994; Skaggs et al., 2006). In each measurement site, the depth of an interface between a highly conductive upper soil layer and a less conductive lower soil layer was deduced from the water table data in each site. During excessively wet periods, such as the summer of 1987 in Tilanjoki, the water table remained near the bottom level of the highly conductive topsoil layer. The depth of an interface between the top soil layer having high hydraulic conductivity and the lower layer having low conductivity was set equal to the median level of the measured water table in the wet summer of 1987 (May–September). The depth of the rooting zone was set to a value of 0.2 m.

In the model setup the depth of the drainage ditches was set to 0.5 m prior to ditch cleaning and 0.8 m after cleaning in autumn 1989. Changes in the depth of the ditches caused by erosion, sedimentation, and vegetation colonisation were disregarded.

3.4 Calibration of FEMMA

FEMMA was calibrated against both water table and runoff data. Water table data for the calibration were from site 7, where the subsoil is peat, from site 26 with sandy subsoil, and from site 27 with till subsoil. The three calibration sites represented the three different subsoil types within the catchments. Measured runoff for the calibration was from catchment C3, where the ditches were cleaned in 1989. The modelled runoff that was compared against the measured runoff during the calibration was computed as the average runoff from the two sites (26 and 27) located in catchment C3. In order to reduce the computation time during the model calibration, not all the sites (23–37) located in catchment C3 were used in computing runoff.

The six calibration parameters are listed in Table 1. The minimum stomatal resistance controls the level of transpiration during the growing season, and its range was set following the values presented for coniferous trees in earlier studies (e.g. Wigmosta et al., 1994; Nijssen et al., 1997; Whitaker et al., 2003; Koivusalo et al., 2006). The hydraulic conductivity of the saturated topsoil layer, and the hydraulic conductivities of peat subsoil, sand subsoil, and till subsoil control the dynamics of the water table and runoff. The range of the hydraulic conductivity of the topsoil was prescribed to be larger than the ranges for the subsoil hydraulic conductivities (Table 1). The ranges were chosen according to preliminary testing of the model parameterisation during the wet summer of 1987. It is noteworthy that the saturated hydraulic conductivity of the topsoil layer was assumed to be the same in all three sites used for the calibration. The retention coefficient of surface runoff affects the runoff dynamics during the highest peak flows in spring. The calibration range for the retention coefficient was a physically meaningful range from a small value close to zero to a value close to unity.

Table 1. Ranges of the calibration parameters. Calibration parameters are the retention coefficient of surface runoff (a), the saturated hydraulic conductivity of the topsoil layer (K_{top}), the conductivity of sandy subsoil (K_{sand}), the conductivity of till subsoil (K_{till}), the conductivity of peat subsoil (K_{peat}), and the minimum stomatal resistance (r_{smin}). The values of the parameters used in the numerical experiment are also shown.

	Min	Max	Numerical exp.
a [–]	0.05	0.95	1
K_{top} [cm/h]	0.1	500	124
K_{sand} [cm/h]	0.0001	10	4.6
K_{till} [cm/h]	0.0001	10	2
K_{peat} [cm/h]	0.0001	2	0.08
r_{smin} [s/m]	50	1000	508

The model calibration was based on the GLUE (Generalised Likelihood Uncertainty Estimation) approach of Beven and Binley (1992). In the application of the GLUE methodology, prior distributions of the calibration parameters were assumed to be uniform within the predefined parameter ranges. For each parameter set, a likelihood measure, E_T, was computed as

$$E_T = E_{\mathrm{gwp}} E_{\mathrm{gwt}} E_{\mathrm{gws}} E_r \tag{2}$$

where E_{gwp} is the Nash and Sutcliffe (1970) efficiency between the measured and modelled depth to the water table in site 7 with peat subsoil, E_{gwt} is the efficiency between the measured and modelled water table depth in site 27 with till subsoil, E_{gws} is the efficiency between the measured and modelled water table depth in site 26 with sandy subsoil, and E_r is the efficiency between measured and modelled daily runoff in catchment C3. The parameter set was determined to be 'behavioural' when the individual efficiencies (E_{gwp}, E_{gwt}, E_{gws}, and E_r) were greater than the prescribed threshold (See Sect. 4.2.1). After a large set of "behavioural" parameter sets were identified, the posteriori distributions of the calibration parameters were derived as likelihood-weighted cumulative distributions. The 5th and 95th percentiles of the model parameters, simulated water table level, and simulated runoff were derived as outlined in Beven and Binley (1992) and Beven and Freer (2001). It should be noted that the uncertainty estimation in the GLUE methodology includes subjectivity, because the selection of calibration parameters, their prior distributions, and the form of the likelihood function are fixed on the basis of the modeller's deduction (see e.g., Freni et al., 2008). Here the uncertainty limits were applied to see how the equifinality of the calibration parameters was reflected in the model results and whether there were reasons for a rejection of the model structure.

After model calibration, runoff from each catchment was computed for each parameter set at a time as follows. The hydrological model was applied in each measurement site to simulate the runoff input that enters the ditch network. Total runoff from the catchments was computed as an equally weighted average runoff from the measurement sites located in each of the catchments C1–C4 (Fig. 1). The simple averaging scheme implies that the distribution of soil and vegetation properties is assumed to be similar in the catchment and among the water table measurement sites inside the catchment.

3.5 Numerical experiment

In the numerical experiment, FEMMA was applied to demonstrate how the key assumptions behind the model structure are reflected in the simulated ditch cleaning impacts on the water balance. The numerical experiment focused on identifying changes in annual runoff, when the ditch depth varied in the range typically found in Finnish drained peatlands before and after ditch network maintenance. The hypothetical drained peatland forest used in the numerical experiment was hydrologically isolated from its surroundings, and the most important water fluxes out of the system were the water flow to the ditches and evapotranspiration of the vegetation. Model simulations were conducted to visualise the effect of forest dimensions (LAI) and soil conductivity structure (depth of the highly conductive layer) on the annual runoff volume. LAI and conductivity depth are the variables, which show large variability between the 39 measurement sites in Tilanjoki.

Table 1 lists the model parameters for the numerical experiment. In each model simulation, the ditch depth, forest dimensions, subsoil type, and soil structure, were set to prescribed values, and the model was run for a four-year period from 1990 to 1993. The ditch depth varied from a depth of 0.3 m to a depth of 1.2 m, LAI of the initial tree stand in 1990 was either 0.5 or 2.0, the subsoil type was either peat or till, and the depth of the highly conductive layer ranged from 0.1 to 0.4 m. The spacing of the ditches was set to a value of 40 m.

4 Results and discussion

4.1 Analysis of the experimental data

4.1.1 Water table level

The response of the annual water table level to ditch cleaning depended on the subsoil type (Fig. 4). The annual water table level showed no change from the pre-treatment to the post-treatment period in the sites with a deep peat layer in catchment C1 (Fig. 4a, P-value=0.15). In the sites with till subsoil in catchment C1, the water table level decreased after ditch cleaning (Fig. 4b, P-value=0.016). However, the difference between the annual water table levels between catchments C1 (ditch cleaning) and C2 (control) did not change when moving from the pre-treatment period to the post-treatment period (Fig. 4b, P-value=0.056). The results for catchments C3 (ditch cleaning) and C4 (control) with till subsoil (Fig. 4c) suggest that the difference between the water table levels in the two catchments increased after ditch maintenance (P-value=0.008). The impact of ditch cleaning on the water table level was smaller in catchment C1 compared with catchment C3. The mean depth of the surface peat layer is thin (0.22 m) in catchment C3 compared with the mean peat depth (0.57 m) in catchment C1. The impact of ditch cleaning could be related to the thickness of the surface peat layer on the till subsoil. In the sites with sand subsoil the impact of ditch cleaning on the water table level was clear in both catchment pairs C1–C2 and C3–C4 (Fig. 4d–e, P-value=0.008). The peat depth in the sites with sand subsoil was 0.29 and 0.33 m in catchments C1 and C2, and 0.12 and 0.16 m in catchments C3 and C4, respectively.

The decrease of the water table in the control catchment C4 was significant in Fig. 4c (P-value=0.008) and in Fig. 4e (P-value=0.016). This indicates that 1) the meteorological conditions driving the water table dynamics changed from the pre-treatment to the post-treatment periods, 2) the control catchment was not fully isolated from the treated catchment, or 3) the observed change resulted from the different number of water table measurements in the pre- and post treatments periods. This issue is analysed further in the modelling exercise in Sect. 4.2. The small response of the water table level in sites with a deep peat formation is line with earlier results of Ahti and Päivänen (1997).

4.1.2 Runoff

Runoff from the treated catchment C1 was, on the average, 12% lower than runoff from the control catchment C2 before ditch cleaning but, after ditch cleaning, the runoff of C1 was 23% higher than that from C2 (Fig. 5a). The paired catchment analysis yielded a regression between the two catchments for the pre-treatment period with an R^2 value of 0.93 (P-value=0.0017). The resulting treatment effect suggested that runoff increased by 38% after ditch cleaning.

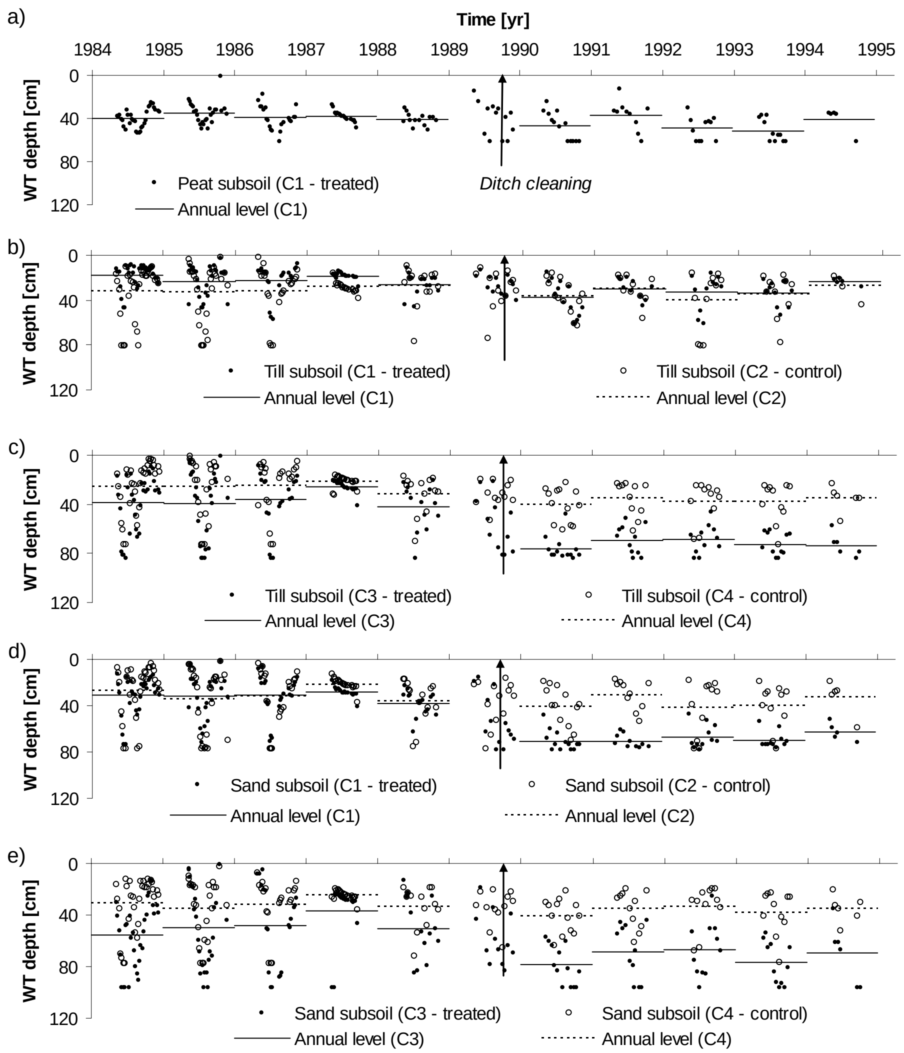

Fig. 4. Measured median depth of the water table level (WT) in sites with peat subsoil **(a)**, median WT depth in sites with till subsoil located in catchments C1 (treated) and C2 (control) **(b)**, median WT depth in sites with till subsoil located in catchments C3 (treated) and C4 (control) **(c)**, median WT depth in sites with sand subsoil located in catchments C1 and C2 **(d)**, and median WT depth in sites with sand subsoil located in catchments C3 and C4 **(e)**. The horizontal lines show the mean annual WT levels.

A similar analysis was conducted for the catchment pair C3–C4, where the number of missing data was lower than in catchment pair C1–C2 (Fig. 5b). Before ditch cleaning the runoff from the treated catchment C3 was 17% higher than that from the control catchment C4, and after cleaning this difference was 83% (Fig. 5b). The regression between the catchments C3–C4 for the pre-treatment period had an R^2 value of 0.95 (P-value=0.0010). The paired-catchment analysis again showed a clear increase (37%) in runoff after ditch cleaning. The increase of runoff is in contradiction with earlier results, where ditch network maintenance was reported to have only a small impact on runoff (Joensuu, 2002). The results can also be viewed against those of Lundin (1994), who studied the impacts of subsequent clear-cutting and drainage on the runoff regime in catchments. Remedial drainage was estimated to increase the mean annual runoff from 9 to 36% compared to the clear-cut catchment before the remedial drainage. The estimate of the impact of ditch cleaning in Tilanjoki is at the upper end of this range, even though no cuttings were conducted in Tilanjoki. In the next Section the Tilanjoki case is further discussed in the application of the hydrological model.

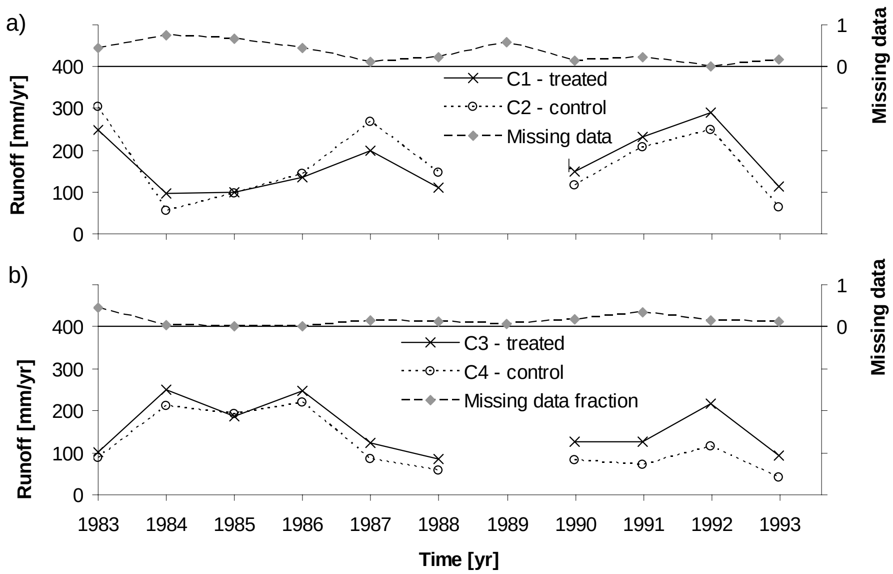

Fig. 5. Measured annual runoff in catchments C1 (treated) and C2 (control) **(a)**, and measured annual runoff in catchments C3 (treated) and C4 (control) **(b)**. The proportion of days per year when measurements are missing for the catchment pair are also presented. Ditch cleaning occurred in 1989.

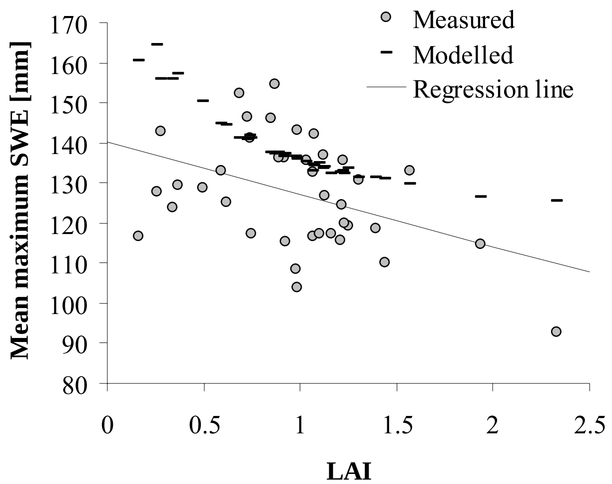

Fig. 6. Measured and computed annual maximum snow water equivalent (SWE) plotted against leaf area index (LAI) in the forest stands of Tilanjoki. The regression between measured SWE and LAI has a P-value of 0.009.

Table 2. Performance of the snow model in the sites located in four study catchments (C1–C4) before ditch cleaning (1983–1989) and after cleaning (1990–1994). NS is median Nash and Sutcliffe (1970) efficiency, MAE is the mean absolute error, and WBE is the bias between the measured and modelled snow water equivalent.

	C1	C2	C3	C4
NS before	0.56	0.62	0.72	0.92
NS after	0.59	0.64	0.74	0.96
MAE before [cm]	13	13	11	11
MAE after [cm]	11	11	8.5	8.5
WBE before %	13	13	8.7	8.7
WBE after %	18	18	12	12

4.2 Assessment of model performance against the measurements

4.2.1 Snow modelling

The canopy and snow models were run separately in all 39 measurement sites, where the tree height, LAI, and canopy closure developed with time according to the data gathered from the sites. The snow model application is an efficient test for the consistency of wintertime meteorological data. Large errors in precipitation and air temperature become highlighted as mismatches between measured and modelled snow accumulation and snowmelt, respectively. Because of the clear mismatch between on-site snow data and model simulation during 1989–1992, the daily air temperature during this period was estimated as the mean temperature of the two closest weather stations (Särkijärvi and Puolanka) instead of on-site temperature chart measurements.

The performance of the snow model is better in the sites located in catchments C3 and C4 compared to the pair C1 and C2 (Table 2). The difference between the computed and measured annual maximum SWE was negatively correlated with the measured annual maximum SWE, which indicates that SWE was underestimated by the model in the sites with a high maximum SWE and overestimated in the sites with a low maximum SWE. The annual maximum SWE decreased with LAI, as shown in Fig. 6, but the annual highest SWE was recorded in small seedling stands instead of nearly open sites, where the model produced the highest annual maximum SWE. The higher accumulation of snow in seedling stands compared with the open could be a result of lateral wind-driven snow transport, which was not included in the snow model.

Model performance for the periods preceding and following ditch cleaning (Table 2) was similar, suggesting that the compiled meteorological variables did not contain errors that could affect snow processes and change systematically over the study period. After the snow model was found to have a consistent performance for the periods preceding and following ditch cleaning, the model analysis was continued with water table and runoff simulations.

4.2.2 Application of GLUE methodology

In the application of GLUE methodology, the prior distribution of the calibration parameters was assumed to be uniform within their ranges (Table 1). Monte Carlo sampling was applied to produce 21 000 sets of parameters, and the likelihood function (Eq. 2) was computed for each parameter set. The parameter set was determined to be "behavioural" when E_{gwt}, E_{gws}, and E_r were greater than 0.4, and E_{gwp} was greater than 0.1. A less strict threshold efficiency was used for simulation of the water table level in site 7 (E_{gwp}), because the range of the water table level variation in the site with a peat subsoil was small compared with the range in the sites with a till or sandy subsoil. The highest efficiency value in site 7 was 0.22 and the corresponding mean absolute error was 4.0 cm. In contrast to these values, the best efficiencies in sites 26 and 27, where the water level fluctuations were dynamic, were 0.72 and 0.65, respectively, and the mean absolute errors were 10.9 and 11.8 cm.

After parameter sampling and model computations, a total of 738 "behavioural" parameter sets were identified. Prior and posterior distributions of the calibration parameters revealed the degree of equifinality of the calibration parameters (Fig. 7). The prior and posterior distributions of the surface runoff retention coefficient (Fig. 7a) were approximately the same, which suggests that the parameter had no influence on the performance of the model. The posterior distribution of the saturated hydraulic conductivity of the top soil layer (Fig. 7b), which was same for all calibration sites (7, 26, and 27), clearly changed from the prior distribution, but 90 % of the values were still within a large range of 24 to 54 m/d. The saturated hydraulic conductivity of the sandy subsoil (0.6 … 2.26 m/d) showed only a small change between the prior and posterior distributions, which suggests that the predefined range of the prior distribution may have been too narrow (Fig. 7c). The saturated hydraulic conductivities for till subsoil (0.28 … 1.3 m/d) and for peat subsoil (0.002 … 0.19 m/d) showed less equifinality compared with the sandy subsoil (Fig. 7d–e). Finally, 90% of the minimum stomatal resistance values (Fig. 7f) were in the range 440 to 930 s/m, which was large and indicated a considerable degree of equifinality for this parameter.

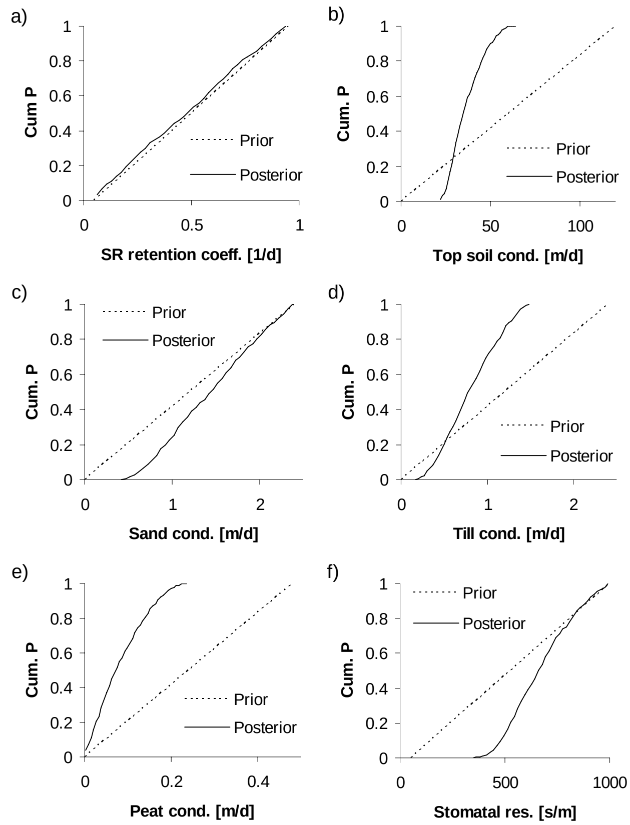

Fig. 7. Cumulative prior and posterior distributions (Cum. P) of the calibration parameters including the retention coefficient of surface runoff **(a)**, the saturated hydraulic conductivity of the topsoil layer **(b)**, the conductivity of sandy subsoil **(c)**, the conductivity of till subsoil **(d)**, the conductivity of peat subsoil **(e)**, and the minimum stomatal resistance **(f)**.

The large degree of equifinality in the retention coefficient was explained by the fact that the modelled generation of surface runoff was minimal due to the high hydraulic conductivity of the saturated soil in the topmost layer. The low volumes of simulated surface runoff and the retention scheme of the surface runoff had no significant effect on model performance. The ranges of the saturated hydraulic conductivities indicated that the conductivity decreased when moving from the topsoil to sandy subsoil, to till subsoil, and to peat subsoil in this order. The calibration results of the conductivity values reflected the dynamic water table behaviour in sites with sandy and till subsoil, and a more stagnant behaviour in sites with a peat subsoil (see Fig. 4). The underlying groundwater aquifer probably affected the water table level in the

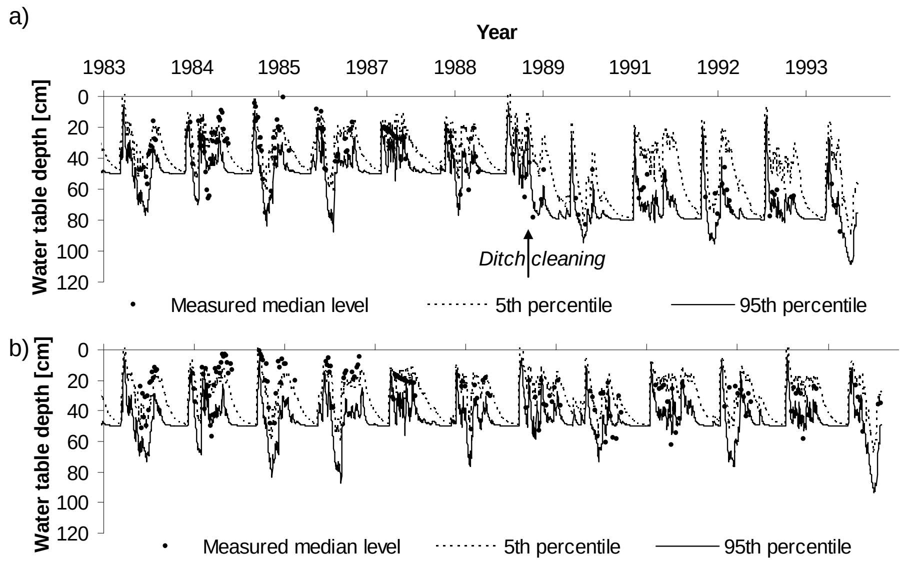

Fig. 8. Measured and modelled median depth of the water table level (WT) in sites with till subsoil in catchment C3 (treated) **(a)**, and in catchment C4 (control) **(b)**. Modelled depth is presented in terms of the 5th and 95th percentiles (90% uncertainty limit).

sites with a conductive subsoil. For example, water table levels below ditch depth were frequently observed during the pre-treatment period, as seen in Fig. 4b–e. The effect of not including the groundwater aquifer in the model could lead to poor identifiability of the model parameters controlling the water table behaviour. The equifinality present in the calibration of the stomatal resistance was influenced by the role of the spring flood, because it is the highest annual runoff event in the calibration catchment. The spring flood in catchment C3 was measured over a period of 9 years, and the maximum daily runoff occurred during the spring in 8 of these years. The efficiency coefficient E_r was prone to errors in simulating the largest event, and the insensitivity of the stomatal resistance, which most strongly affected the summer and autumn runoff, resulted from the tendency of E_r to sacrifice the performance of summer and autumn peaks for the reproduction of spring high peaks.

4.2.3 Assessment of model performance and data quality

The snow simulations already indicated that the meteorological input was not fully consistent for the daily simulation of the snow processes, but that the errors in the meteorological variables were likely to be similar in the periods preceding and following ditch cleaning. In order to assess the conformity of the hydrological data preceding and following ditch cleaning and the consistency of the experimental setup, the 'behavioural' parameter sets and their associated likelihood values were applied to produce the 5th and 95th percentiles (90% uncertainty limit) of the simulated water table levels at the sites with a till subsoil (Fig. 8) and the simulated runoff in control catchments C2 and C4 (Fig. 9).

The uncertainty limits of the simulated water table levels were wide during the growing season and decreased in the end of the winter low flow period, when the simulated water level reached the depth of the drainage ditch (Fig. 8). Several water table level observations in catchments C3 and C4 were outside the 90% uncertainty limits which, according to the GLUE methodology, suggests that the model structure for simulating the water table dynamics should be rejected. In control catchment C4 the measured water table level was commonly above the 5th percentile during 1983–1987, within the uncertainty range during 1987–1989, and mostly within or below the 90% limits during the rest of the years. The 90% range in Fig. 8b seems to be shifting over time when compared with the measurements. There was subjectivity in producing the uncertainty limits in the GLUE methodology, because the uncertainty limits are affected by the selection of calibration parameters, the ranges and types of prior distribution of the parameters, and the form of the likelihood function. The uncertainty limits become wider when, for instance, increasing the number of calibration parameters, or lowering the thresholds that define the "behavioural" parameter sets in the likelihood function (Freni et al., 2008). Increasing the uncertainty ranges decreases the prediction ability of the simulation model. However, increasing the uncertainty limits is likely to result in a similar change for the periods preceding and following ditch cleaning. The larger uncertainty limits were not expected to lead to systematic temporal changes in the model results.

In order to attain a clearer picture of the model performance and the measurement setup during the time periods preceding and following ditch cleaning, the paired catchment

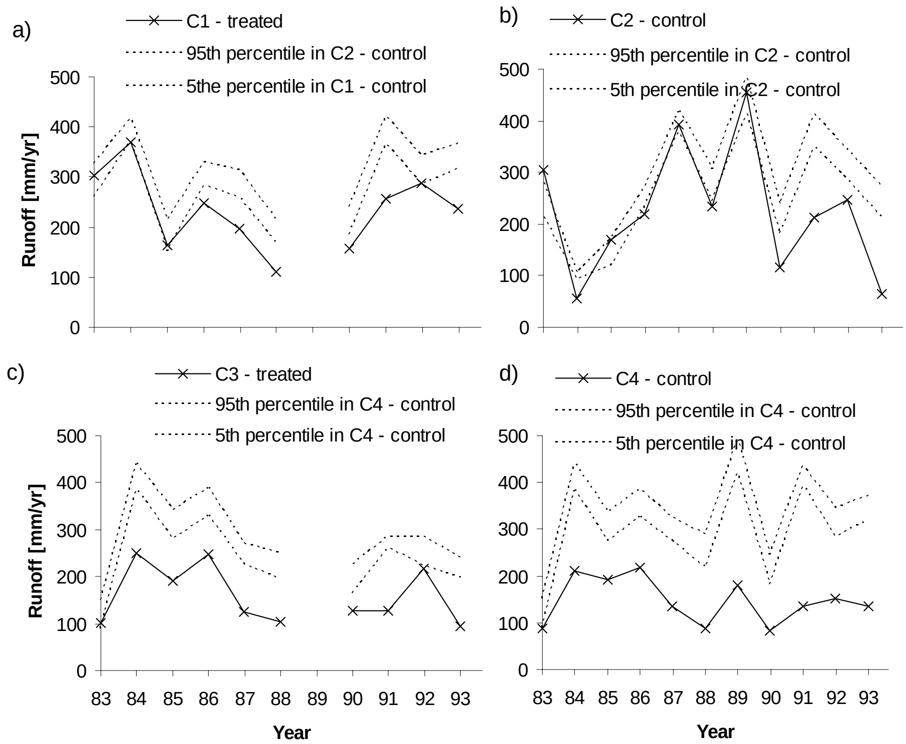

Fig. 9. Measured annual runoff in catchment C1 (treated) and modelled runoff in C2 (control) **(a)**, measured and modelled runoff in C2 **(b)**, measured runoff in catchment C3 (treated) and modelled runoff in C4 (control) **(c)**, and measured and modelled runoff in C4 **(d)**. Modelled runoff is presented in terms of the 5th and 95th percentiles (90% uncertainty limit). Ditch cleaning occurred in 1989.

analysis of the annual runoff volumes presented in Sect. 4.1.2 was repeated by using the simulation results as a control dataset. The uncertainty limits in the annual runoff volumes presented in Fig. 9 are relatively narrow, which is explained by the fact the only one of the calibration parameters (minimum stomatal resistance) exerted a strong control on the water balance in the annual time scales. The rest of the calibration parameters had a larger impact on the short-term dynamics of the runoff hydrograph. Again, changing the number of calibration parameters and the likelihood function would have increased the uncertainty limits, but these changes were not deemed to be necessary for the subsequent assessment of the pre- and post-treatment periods. When the measured runoff in the treated catchment C1 was compared against the modelled runoff in the control catcment C2 (Fig. 9a), the relationship between the time series was found to be opposite to that between the measured runoff time series in Fig. 5a. The results of paired catchment analysis for the measured runoff in C1 and the modelled mean runoff in C2 suggest that runoff in C1 decreased by 16% after ditch cleaning. In the regression for the pre-treatment period, the value of R^2 and its P value were 0.86 and 0.008, respectively. In Fig. 9b, the measured runoff in the control catchment C2 was found to decrease with respect to the modelled runoff in catchment C2. According to the paired catchment computation, the decrease was estimated to be 43%. The results presented in Fig. 9c–d suggest that the measured runoff in the treated catchment C3 decreased by 1% and the runoff in the control catchment decreased by 27% compared with the modelled mean runoff in the control catchment C2. The runoff results supported rejection of the simulation model for describing the hydrology of the Tilanjoki catchments.

The change in the behaviour of the control catchments was not a result of changes in the meteorological conditions. If there were systematic errors in the climatic variables between the pre- and post-treatment periods, then the errors would be reflected in the snow simulation results. Therefore, the hydrological behaviour of the control catchments was affected by the ditch cleaning in the adjacent catchments. The control and treated catchments were not isolated from each other, which in this case violated the use of paired catchment analysis for studying the management impacts on hydrology. The treatment effect cannot be estimated using the paired-catchment analysis and, therefore, the ditch cleaning impact estimated from the measurements alone in Sect. 4.1 is misleading. Rejection of the model implied that there was no structure in the model that could link the control and treated catchments together. The inconsistency of the experimental setup was not detected without an application of a hydrological model that binds the meteorological and hydrological

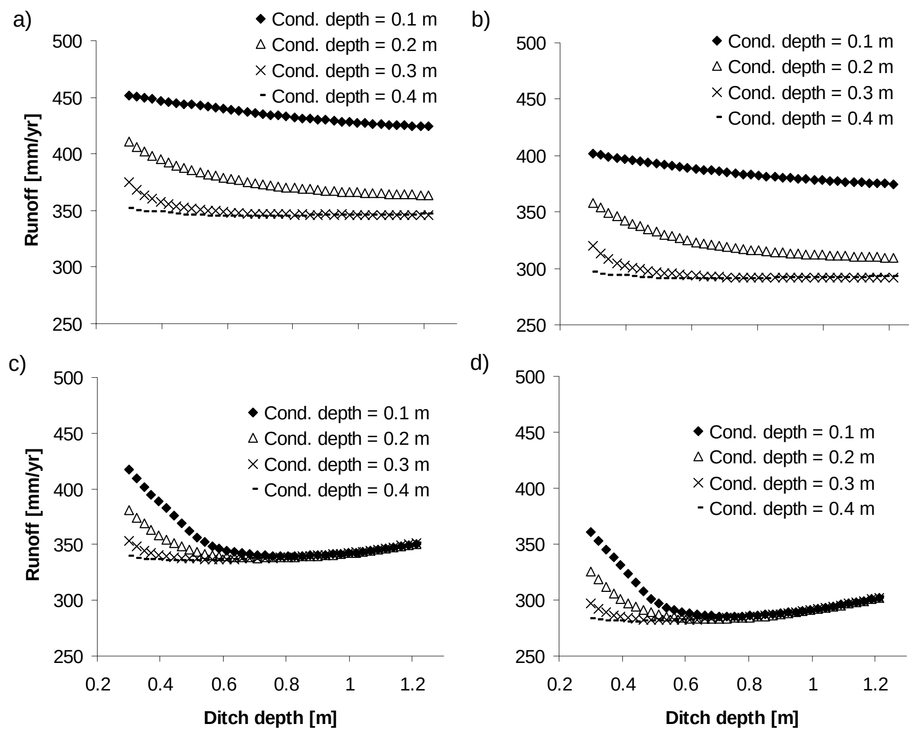

Fig. 10. Effect of ditch depth on the mean annual runoff in hypothetical sites where the subsoil type is peat and LAI is 0.5 **(a)**, the subsoil type is peat and LAI is 2.0 **(b)**, the subsoil type is till and LAI is 0.5 **(c)**, and the subsoil type is till and LAI is 2.0 **(d)**.

fluxes together.

Beven (2006) noted that a hydrological model is often not structurally correct, because it is not possible to arrive at a single correct representation of the hydrological system given the particular set of observations for the model calibration. This study confirms that the rejection of the model is not a deadlock, but provides an opening for identifying unexpected controls of the studied case, which shows the way to competing model structures.

4.3 Numerical experiment

A numerical experiment was conducted to apply the model alone and to demonstrate the impacts of ditch cleaning on annual runoff in sites with a different vegetation cover and soil conductivity structure (Fig. 10). In the case with peat as the subsoil type (Fig. 10a–b) the volume of runoff decreased as the ditch depth increased. The decrease in the rate of runoff levelled out in the deeper ditches. The volume of annual runoff was dependent on the size of the tree stand and on the depth of the highly conductive top soil layer. The effect of ditch depth on runoff decreased when the thickness of the highly conductive layer increased. The runoff response to changing ditch depth was explained by the soil moisture limit shown in Fig. 2, where an excessively high soil moisture restricts transpiration. In soils with a small depth of the highly conductive layer, the soil moisture in the rooting zone is often close to saturation. In this case deepening the ditches made the soil moisture regime drier and shifted it toward the optimum transpiration range, where soil moisture did not limit transpiration. The effects of ditch depth and conductivity depth on runoff were similar for the two tree stands in Fig. 10a and b, but the level of runoff was lower in the vegetation with a higher LAI.

When the subsoil type was till (Fig. 10c–d) with a hydraulic conductivity that was higher than the conductivity of the peat subsoil, ditch cleaning increased or decreased runoff depending on the initial ditch depth. Runoff decreased with increasing ditch depth, when the ditch depth was small ($<$0.6 m) and the thickness of the highly conductive layer less than about 0.3 m. The runoff response in this case was related to the soil moisture limit in near-saturated soil, as explained earlier (Fig. 2). When ditch depth increased in the range above about 0.6 m, runoff started to increase. The increase was again explained by the soil moisture restriction in Fig. 2. In well-drained conditions and with a conductive subsoil type, the soil moisture regime was mostly within the range where soil moisture did not limit transpiration. In this case deepening the ditches shifted the soil moisture to a dryer regime where, according to Fig. 2, soil moisture starts to decrease transpiration. The increase of runoff was larger for the larger stand (Fig. 10d) than for the smaller stand (Fig. 10c).

The model suggests that, in most cases, the annual runoff volume decreases and evapotranspiration increases when ditches are cleaned down to a greater depth. If the increase in evapotranspiration is assumed to be related to forest growth, then excessive deepening of the ditches, or cleaning of the ditches in sites with deep a surface layer of highly conductive soil, would not be necessary.

Finally, it should be noted that the results of the numerical experiment only demonstrate how the hydrological impact of ditch cleaning varies and depends on certain vegetation and soil characteristics, but the modelling results are not a generalization of ditch cleaning impact. Earlier studies have shown that field, catchment, and meteorological characteristics have complex and counteracting effects on the hydrological impact of drainage (Lundin, 1994; Dunn and Mackay, 1996; Robinson and Rycroft, 1999). Since long term processes, such as vegetation colonization in ditches and subsidence of the peat (Robinson, 1986; Holden et al., 2006) were disregarded in the model, their effects on runoff could not be assessed on the basis of the numerical experiment.

5 Conclusions

Available experimental data on the water table level showed that the effect of ditch cleaning was not seen in sites with a deep peat formation. The effect of ditch cleaning became clear in sites with a shallow peat layer underlain by till or sand. The results of a paired catchment analysis suggested that annual runoff volume from the two treated catchments, where ditch cleaning was performed, strongly increased compared with the control catchments, where no cleaning was implemented. The increase estimated from the data was in conflict with the results of earlier studies, and it was concluded that a further assessment of the data quality and experimental setup was required.

The application of a simulation model that binds together meteorological and hydrological measurements was a useful aid in understanding the difficulties related to the experiments. The GLUE methodology allowed an assessment of the uncertainty propagation from parameter estimation to the model simulations, and provided a means to identify deficiencies in the model structure. The measured water table level was frequently outside the 90% uncertainty limit of the simulated water table level, which suggested that the model could not predict the water table dynamics. The comparison between the modelled and measured runoff confirmed the result that the model was not able to predict the measured hydrological response to ditch cleaning.

Assessment of the model behaviour and experimental data during the pre- and post-treatment periods revealed a shift in the measured annual runoff in the control catchments compared to the modelled runoff from the control catchments. Since the change in runoff from the control catchments was not explained by the meteorological conditions, it was concluded that ditch cleaning must have had an impact on runoff in both the treated and control catchments. When paired catchments are artificially delineated side-by-side using relatively shallow ditches, there is the possibility that the catchments may not in fact be fully isolated from each other. Derivation of the water balance in a separated sub-area of a peatland is a more demanding task than in pristine headwater catchments. In Tilanjoki, the hydrological connection between the treated and control catchments was clearly reflected in the runoff measurements and less clearly in the water table measurements. In the design of hydrological experiments in drained peatlands the possibility of dependence between artificial catchments should be considered. The study catchments should preferably be natural catchments delineated by topography. When artificial catchments are used, the side-by-side location of differently managed (e.g., control and treated) catchments should be avoided. Runoff data from a natural catchment that includes the artificial study catchments (measurements in nested catchments) would be useful for the data analysis.

A numerical experiment based on model simulations demonstrated how the function between soil moisture and transpiration efficiency controls the impact of ditch cleaning on the total annual runoff. According to the model, runoff decreases and evapotranspiration increases, when ditches are dug deeper in poorly drained sites, where the initial ditch depth is small and the depth of highly conductive top soil layer is low. From the point of view of evapotranspiration efficiency, ditch cleaning is not necessary in sites where the highly conductive surface soil layer extends to a depth of about 0.3 m and the initial ditch depth is about 0.5 m or deeper. Information about the necessity of ditch network maintenance is valuable for outlining practical guidelines for forest management on drained peatlands.

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