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Abstract

Forest drainage has been extensively used to facilitate forest growth in Fennoscandinavia. However, its impact on the ecosystem carbon (C) and greenhouse gas (GHG) balances is limited, particularly in the large area of hemiboreal and boreal Sweden. This work quantified the effect of drainage on C and GHG fluxes in the northern forests under various geographical settings, soil conditions and drainage regimes. First, this thesis describes an investigation into the initial impacts of ditch cleaning (DC) on the C and GHG balances in two forest clear-cuts using closed chamber measurements. Second, the historical drainage impacts on C and GHG balance were evaluated using eddy covariance and stream discharge measurements for a drained peatland forest and an adjacent oligotrophic mire in boreal Sweden. Results show that DC did not increase CO_2 emissions. Instead, annual CO_2 emission decreased after DC at the dry and fertile clear-cut site, whereas an insignificant DC effect on CO_2 flux was observed at the relatively wet and infertile clear-cut site. The net CO_2 uptake was recorded as being greater in the drained peatland forest relative to the adjacent mire. An effect of DC and historical drainage on mitigating strong CH_4 emission and potentially increasing CH_4 uptake was observed. Ground vegetation growth was identified as a primary mediator of drainage effects on the C and GHG balance, but the interaction between ground vegetation growth and drainage depended on the site soil and hydrological conditions. Other carbon and greenhouse gas components, such as flux of nitrous oxide and loss of C through discharge, did not significantly respond to drainage activities. The conclusion is that forestry drainage activities, including DC and historical activities, do not intensify carbon and greenhouse gas emissions. This work provides novel insights which can support the development of sustainable and climate-friendly forest management strategies.

Keywords: ditch cleaning; CO_2; methane; peatland; boreal forest; clear-cut; eddy covariance; closed chamber

Upptag och avgivning av växthusgaser i dikat skogslandskap i nordliga Sverige

Abstrakt

Under 1900-talet dikades betydande arealer i stora delar av Fennoskandien med syfte att öka tillväxten av skog. Kunskapen om hur dikad beskogad torvmark påverkar ekosystemets kol (C) och växthusgasbalans är begränsad, särskilt i hemiboreala och boreala Sverige. Syftet med de studier som presenteras i den här avhandlingen är att kvantifiera effekten av dikning, under olika mark- och dräneringsförhållanden, på flöden av kol och växthusgaser i olika delar av nordliga skogsområden i Sverige. Avhandlingen behandlar dels den initiala påverkan av dikesrensning på omsättningen av kol och växthusgaser baserat på mätningar med markkammare i två olika kalavverkade ytor. Den andra delen baseras på mätningar med Eddy Covariance teknik och avrinningsstudier och behandlar den långsiktiga effekten av dikning och beskogning på omsättningen av kol och växthusgaser på näringsfattiga nordliga myr. Dikesrensning leder till minskad årlig avgivning av CO_2 under torrare och mer näringsrika förhållanden medan avgivningen från blötare och mer näringsfattig förhållanden inte påverkades. Den dikade och beskogade myren hade ett högre upptag av CO_2 jämfört med den öppna och odikade myren. Dikesrensning såväl som beskogad dikad torvmark leder till minskad avgivning respektive möjligt upptag av metan (CH_4). Markvegetationens sammansättning och biomassa hade en avgörande betydelse för hur utbytet av kol och växthusgaser påverkades av olika dikningsaktiviteter. Effekten påverkas dock starkt av specifika markegenskaper och hydrologi. Inga effekter på vare sig avgivning av lustgas eller transporter av kol med avrinningsvatten noterades från de aktuella studieområdena. Studierna som sammanfattas i avhandlingen visar att vare sig dikesrensning eller dikad beskogad torvmark, under studerade betingelser, leder till ökad avgivning av kol och övriga växthusgaser. Resultaten som presenteras i avhandlingen är av stor betydelse för att utveckla vetenskapligt välgrundade skötselåtgärder av dikade beskogade torvmarker under nordliga förhållanden i Sverige.

Nyckelord : dikesrensning; CO_2; metan; torvmark; boreal skog; kalavverkning; eddy covariance; markkammare

Dedication

This thesis is wholeheartedly dedicated to my beloved parents. Thank you for being my source of endless love, support and encouragement.

I dedicated to the memory of my beloved grandparents, who passed away in 2020 and 2022. Thank you for holding my hands warm before I started my journey.

And lastly, I dedicated this thesis to the Almighty God. Thank you for your guidance, strength and protection. I offer all of the glory to you.

Contents

List of publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I. Tong, C. H. M., Nilsson, M. B., Drott, A., & Peichl, M. (2022). Drainage Ditch Cleaning Has No Impact on the Carbon and Greenhouse Gas Balances in a Recent Forest Clear-Cut in Boreal Sweden. *Forests*, *13*(6), 842.

II. Tong, C. H. M., Nilsson, M. B., Sikström, U., Ring, E., Drott, A., Eklöf, K., Futter, M. N., Peacock, M., Segersten, J. & Peichl, M. Initial Effects of Post-Harvest Ditch Cleaning on Greenhouse Gas Fluxes in a Hemiboreal Peatland Forest. (accepted in *Geoderma,* PII: S0016706122003627)

III. Tong, C. H. M., Nilsson, M. B., Drott, A., Laudon, H., Noumonvi, K. D., Ratcliffe, J., Peichl, M. A comparison of the net ecosystem carbon balances of a nutrient-poor peatland forest and adjacent natural mire in boreal Sweden. (manuscript)

Papers I and II are reproduced by kind permission of the publishers.
The contribution of Cheuk Hei Marcus Tong to the papers included in this thesis was as follows:

I. Carried out field work, all statistical analyses, and led the writing of the manuscript.

II. Carried out field work, all statistical analyses, and led the writing of the manuscript.

III. Carried out field work, all statistical analyses, and led the writing of the manuscript.

1. Introduction

1.1 The boreal forest carbon cycle after a century of drainage activities

The northern forest ecosystem is the global reservoir of an estimated one third of the global terrestrial carbon (C) (367–1716 Pg C) (Bradshaw and Warkentin, 2015; IPCC, 2013; Pan et al., 2011). International efforts to mitigate climate change have largely neglected this area in relation to management of C storage and flux (Moen et al., 2014), as the boreal zone has been widely considered to be a C sink (Jobbágy and Jackson, 2000**;** Ciais et al., 2010**;** Pan et al., 2011). A major contribution to C uptake is the accumulation of C in deep peatlands at a steady rate over long timescales (Clymo, 1984; Clymo et al., 1998; Gorham et al., 2007).

However, biogeochemical cycling and, therefore**,** C storage is sensitive to forest management practices (Lindeskog et al., 2021). Around 15 million hectares of northern wetlands has been drained for forestry since the 19th century to increase biomass production (Laine et al., 1995). Drainage of wetland soils to enhance timber production could directly and indirectly alter the total C storage in these high C systems. There is much speculation concerning the consequences of forestry drainage on the C storage, but the possibility of a potentially large C release into the atmosphere from boreal systems could strongly impact the global C balance and climate system (He et al., 2015; Lavoie et al., 2005; Meyer et al., 2013).

To date, extensive transformation of wetland forestry through ditching activities has essentially stopped in Fennoscandia (Paavilainen & Päivänen, 1995). However, the drainage capacity and efficiency of many original ditches have deteriorated in relation to the lowering of the water table level

(WTL). Furthermore, decreased evapotranspiration after harvesting further causes a rise in the WTL which hampers the establishment, survival and growth of subsequent seedling generations (Dubé et al., 1995; Roy et al., 2000). To mitigate increasing soil wetness, ditch cleaning (DC) following forest harvest is increasingly used in Sweden to restore the drainage function of ditches and thereby support desired tree growth rates (Paavilainen and Päivänen, 1995) (Figure 1). According to government statistics from the Finnish National Forest Programme and the Swedish National Forest Inventory (NFI), about 65,000 ha and 10,000 ha of drained forests have been ditch cleaned every year in Finland (during 2001–2010) and Sweden (during 2015–2019), respectively, at a constantly increasing rate over recent years. As with first-time ditching, a lower WTL following DC may strongly influence the C and GHG balances through interacting processes associated with soil biogeochemistry and vegetation growth.

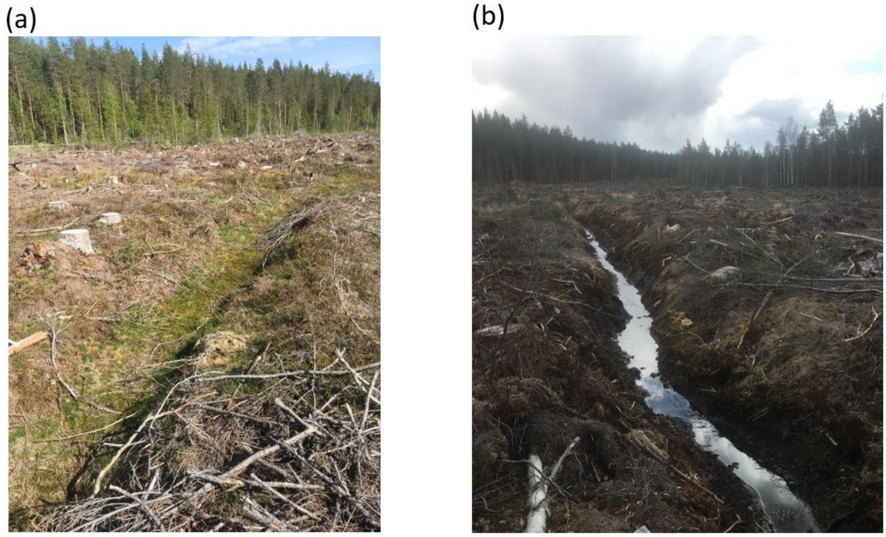

Figure 1. A drainage ditch in Robertsfors, Västerbotten, Sweden (a) before ditch cleaning in 2019 and (b) after ditch cleaning 2020.

Current C and GHG emission estimates for drained and ditch-cleaned forests in Sweden are, however, based on very limited data from croplands and forests on nutrient-rich peat soils in Southern Sweden (He et al., 2015; Meyer et al., 2013) which represent only a few percent of the national land area. In contrast, information is lacking for the much larger areas of nutrient-poor forested peatlands within the Swedish boreal landscape. Furthermore, assessment of the effect of DC on the C and GHG balance is currently lacking. As a consequence, a complete reckoning of the C and GHG balance is urgently required to achieve national (e.g. ‘Swedish Roadmap 2050’) and international (e.g. ‘Paris Agreement 2015’) climate goals.

1.2 Soil CO_2 and CH_4 exchanges

Forestry drainage may have both positive and negative impacts on the C and GHG budget. Specifically, drainage on wet soils could speed up the peat decomposition process in response to higher oxygen availability (Drzymulska, 2016), eventually increasing emissions of CO_2 from the soil (Maljanen et al., 2010; Ojanen et al., 2013; van Huissteden et al., 2006). Meanwhile, wet soil drainage improves root aeration and nutrient availability, thereby facilitating initial development of ground vegetation as well as tree establishment and subsequent growth (Hökkä and Kojola, 2001; Hökkä and Kojola, 2003; Lauhanen and Ahti, 2001; Sikström et al., 2020; Sikström and Hökkä, 2016), eventually enhancing rates of gross primary productivity (GPP) and C sequestration.

Soil type and fertility play an important role in regulating the spatial variation of such drainage impacts on the C balance. In fertile peat soils, CO_2 emission can increase steadily through higher rate oxidative decomposition and mineralisation after drainage (Alm et al., 1999; Silvola et al, 1996; Munir et al., 2017). Soil fertility has also been found to be connected with litter production, modifying litter quality (Laiho et al., 2003; Minkkinen and Laine, 1998; Straková et al., 2012). For instance, drained forest sites in southern Sweden at minerotrophic mires were identified as a net source of C (Kasimir et al., 2018; He et al., 2016; Meyer et al., 2013), while a net sink of C was reported from nutrient-poor sites in southern boreal Finland (Lohila et al., 2011; Ojanen et al., 2014).

Water-saturated conditions in natural mires are typically characterised by an extended anaerobic zone which favours the production of methane (CH_4), potentially meaning the mire produces a net warming effect (Drösler et al., 2008). Such humid soil conditions are also commonly observed after clear-cutting due to loss of evapotranspiration effects from trees in weakened ditch conditions (Korkiakoski et al., 2019). Following drainage, CH_4 emissions are expected to decrease in response to improved soil aeration that potentially switches the soil into a net CH_4 sink (Kasimir et al., 2018; Maljanen et al., 2001; Martikainen et al., 1995; Nykänen et al., 1998; von Arnold et al., 2005). Furthermore, drainage forces the CH_4 production zone down to deeper soil profiles (Borken et al., 2006; Feng et al., 2020; Fest et al., 2017), eventually limiting the substrate supply for methanogenesis as decomposition rates often deteriorate with increasing peat age (Frolking et al., 2001). However, the change in vegetation composition after drainage might further amplify or counterbalance the hydrological consequences from drainage for CH_4 fluxes through various mechanisms, such as substrate supply to methanogens (Minkkinen et al. 2006), as well as improvement of CH_4 transport through stomatal conductance and aerenchyma tissue (Chu et al., 2014; Garnet et al., 2005; Granberge et al., 1997; Long et al., 2010).

Low C:N ratios were typically reported in drained organic soils (Ernfors et al., 2007). Significant production and emission of nitrous oxide (N_2O) are observed when there is high soil nitrogen availability (i.e. C:N ratio < 20), through interacting biological pathways of NH_4^+ and NO_3^- reduction (Firestone and Davidson, 1989; Klemedtsson et al., 2005). Drainage can modify the biological pathways through, for example, initiating N_2O emissions from incomplete denitrification under partially-oxidised conditions (Rubol et al., 2012) and triggering nitrifier activity under fully aerated conditions (Santin et al., 2017). On the other hand, drainage could potentially limit N_2O emission through suppression of denitrifying bacterial activities under dry conditions (Christiansen et al., 2012; Rassamee et al., 2011). Hence, the response of N_2O emission to soil moisture follows an optimum curve which accounts for both positive (e.g. Pihlatie et al., 2004; Rochette et al., 2010) and negative (e.g. Christiansen et al., 2012; Pärn et al., 2018) correlations between WTL and N_2O production seen in previous studies. The optimum for N_2O emission, however, depends on various factors such as initial WTL, drainage efficiency and soil fertility. Empirical data exploring these complex relationships are currently lacking.

Previous studies have indicated that the effect of WTL drawdown on GHG emissions depends on soil conditions (Klemedtsson et al., 2010; Säurich et al., 2019), as well as the timescale of the evaluation (He et al., 2016; Kasimir et al., 2018). This indicates that the impact of drainage observed after DC is expected to be significantly different from the effects of initial ditching. Specifically, long-term drainage significantly transforms plant community composition, and causes major changes on litter input

(Straková et al., 2012; Urbanová and Bárta, 2016). Changes in soil conditions also involve the mineralization of organic C, nitrogen (N), phosphorus (P) and sulphur (S) through long-term drainage (Straková et al., 2011). While most previous studies have focused solely on the effects of long-term drainage on GHG fluxes (e.g. Lohila et al., 2011; Maljanen et al., 2010; Ojanen et al., 2013), the impacts of cleaning existing but deteriorated ditches on the GHG balance of the managed areas are currently not well studied.

1.3 Net ecosystem exchanges of CO_2 and CH_4

Although forestry drainage can increase short-term C emissions from soil, a net increase in C storage per unit area has been identified associated with the enhancement of vascular plant productivity, root growth and soil C storage (Minkkinen et al., 2002). The majority of previous studies have reported strong emissions from drained peatlands by only considering terrestrial soil balances, using peat depth (Armentano and Menges, 1986; Gorham, 1991) or chamber fluxes on the forest floor (e.g. Alm et al., 1999). It is, however, important to highlight the important contribution of forest vegetation to the ecosystem-scale C balance which is rarely accounted for (Kasischke, 2000; Talbot et al., 2010). The application of the eddy covariance (EC) technique allows for quantifying all terrestrial-atmosphere exchanges of CO_2 and CH_4 associated with soil processes and vegetation growth (comprising both tree layer and understory) across half-hourly to annual timescales (Baldocchi, 2003), thus giving the potential to identify the biomass contribution to C cycling at the ecosystem scale (Lohila et al., 2011; Maljanen et al., 2010). Using the EC technique, Lohila et al. (2011) identified a higher average accumulation rate than previously reported for natural northern peatlands, highlighting the important role of forest biomass on increasing the CO_2 uptake rate of nutrient-poor peatland ecosystems.

In the clear-cut sites, the loss of tree biomass and the ability to take up C could potentially cause significant impacts on the C balance. However, previous studies have reported fast recovery of ground vegetation even on nutrient-poor boreal forest clear-cuts (Strömgren et al., 2016; Sundqvist et al., 2014; Uri et al., 2022) in response to the available radiation on the soil surface. The regeneration of ground vegetation and seedlings thus plays an important role in determining the net primary production and, eventually, the time required to switch the site from a net source to net sink of carbon, varying from 1 to 20 years (Vestin et al., 2020; Hyvönen et al., 2007; Kowalski et al., 2004; Rannik et al., 2002; Rebane et al., 2019).

1.4 Aquatic fluxes of C

In addition to the terrestrial land-atmosphere exchanges, the aquatic C flux is also an important component determining the C cycle of the boreal catchments (Cole et al., 2007; de Wit et al., 2015; Kindler et al., 2011; Nilsson et al., 2008). Aquatic C appears in boreal streams primarily in the form of dissolved organic carbon (DOC), dissolved inorganic carbon (DIC) and dissolved CO_2 and CH_4, all of which reflect the connection between the terrestrial and aquatic environment, influenced by terrestrial C content, hydrological regime and vegetation structure (Moore, 2003; Neubauer and Megonigal, 2021). These highlighted factors are strongly influenced by forest drainage and, therefore, the aquatic export of C which has been found to be considerably different from that of natural mires (Evans et al., 2016; Nieminen et al., 2021).

Drainage ditches are potentially significant GHG emission hotpots and contributors to ecosystem-scale GHG budgets. Specifically, ditches can emit large amounts of CH_4 due to their often flooded and anoxic conditions that stimulate methanogenesis (Hyvönen et al., 2013; Minkkinen and Laine, 2006; Peacock et al., 2017; Sundh et al., 2000). Ditches have also been identified as emitting CO_2 (e.g. Sundh et al., 2000; Teh et al., 2011; Hyvönen et al., 2013; Vermaat et al., 2011) and may emit N_2O (e.g. Reay et al., 2003; Teh et al., 2011; Hyvönen et al., 2013) at levels which vary considerably from the surrounding field layer. Ditching or cleaning of deteriorated ditches could produce an immediate effect on the leaching of suspended solids from flooded conditions (Nieminen et al., 2018), which might consequently reduce the availability and quality of substrates to trigger GHG emissions (Hyvönen et al., 2013). Yet, studies of GHG fluxes from drainage ditches, particularly CO_2 and N_2O, are currently not sufficient to draw reliable conclusions (Evans et al., 2016). It is, thus, crucial to improve our understanding of drainage impacts on ditch GHG fluxes and to evaluate their contribution to ecosystem-scale GHG budgets.

2. Objectives

The main aim of the work for this thesis was to investigate and quantify the impact of drainage on C and GHG balances in the northern forests of Sweden under various geographical settings, soil conditions and drainage regimes. Specifically, this was achieved by carrying out field experiments in central boreal nutrient-poor land covered with podzol soil (Paper I), hemiboreal nutrient-rich peatland (Paper II), and central boreal nutrient-poor peatland (Paper III). The thesis describes the investigation of the initial impacts of DC (Papers I and II), and historical drainage impacts (Paper III), on the C and GHG balances. An overview of the studies caried out for this thesis is shown in Figure 2.

For the three papers, the investigation included three aspects: (1) examination of the spatio-temporal dynamics of the C and GHG flux components at the drainage sites, (2) identification of the environmental influences on the C and GHG balance, and (3) estimation of the annual budget of C and GHG balances for national accounting purposes.

The specific objective of each study was:

I. To investigate the impact of drainage DC on the C and GHG balances in a recent forest clear-cut on mineral soil in boreal Sweden (**Paper I**).

II. To determine the initial effects of post-harvest drainage DC on GHG fluxes in a hemiboreal peatland forest (**Paper II**).

III. To assess the net ecosystem C balances of a nutrient-poor peatland forest relative to an adjacent natural mire in boreal Sweden (**Paper III**).

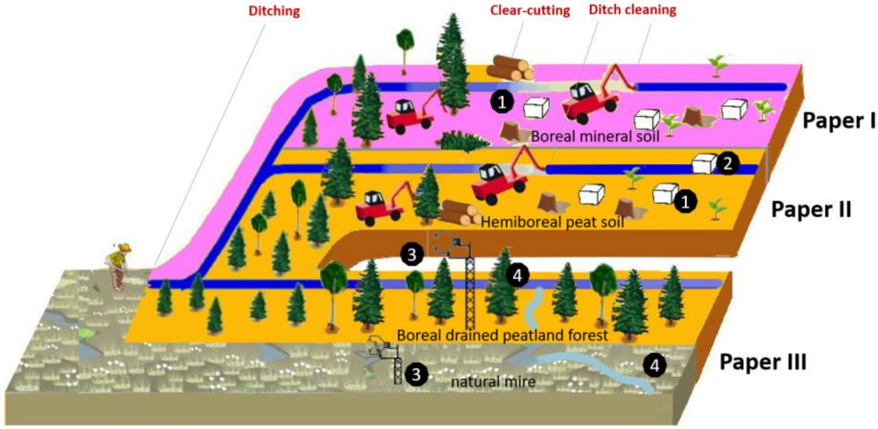

Figure 2. The overview of studies carried out for this thesis (**Papers I–III**) across a timeline. Carbon and greenhouse gas measurements are indicated by numbers on black dots: 1. Closed chambers on field layer (**Papers I and II**); 2. Closed chambers on ditches (**Paper II**); 3. Eddy covariance on drained forest and natural mire (**Paper III**); 4. Stream water sampling on drained forest and natural mire (**Paper III**).

3. Methods

3.1 Study sites

The three studies were carried out in separate regions with different characteristics. Specifically, the work for **Papers I** and **II** was carried out in two different clear-cut sites across boreal and hemiboreal regions of Sweden, whereas the work for **Paper III** was carried out in a forestry drained peatland in boreal Sweden. More detailed information is given in Table 1.

Table 1. Geographical information of the three study sites.

Paper	**I**	**II**	**III**
Site Name	**Pettersson**	**Tobo**	**Hälsingfors**
Coordinates	64°13′N, 20°50′E	60°16'N, 17°37'E	64°09'N, 19°33'E
Elevation	60 m	40 m	290 m
Mean air temperature*	4.0 °C	6.2 °C	2.5 °C
Mean annual precipitation*	701 mm	603 mm	670 mm
Dominant soil type	Podzol	Peat	Peat

*Climate data are based on the 1990–2020 mean values from the nearest weather station provided by the Swedish Meteorological and Hydrological Institute.

The Pettersson and Tobo clear-cut sites (**Papers I** and **II)** represent typical landscapes in boreal and hemiboreal areas of Sweden, respectively, but they differ considerably in terms of soil conditions. A summary of their site characteristics is given in Table 2. The Pettersson site is located in a nutrient-poor region where Podzol is the dominant soil type (Geological Survey of Sweden (SGU)), with soil layers consisting of primarily postglacial sand, gravel and small particles of clay and silt. In the top 10 cm of the soil profile, it was observed that the thin organic layer had been mixed into the upper mineral soil layer by the harvesting machinery. Comparatively, the Tobo site had a higher soil fertility as a result of originally being a minerotrophic mire which was drained with a ditch network for agricultural use. Peat depth was >100 cm within all study plots with a high degree of humification at the top 30cm peat layer (H8–H9 on von Post scale).

Table 2. Information of the two clear-cut sites for **Papers I** and **II**.

Site Name	Pettersson	Tobo
Carbon content*	50 ± 1 %	25 ± 5 %
Nitrogen content*	3.1 ± 0.1 %	0.84 ± 0.19 %
CN ratio*	16.5 ± 0.3	31 ± 1.5
Dominant stand species	*Pinus sylvestris* L (76%), *Picea abies* L (22%) *Betula* spp (3%)	*Picea abies*
Clear-cut	Oct 2016	Sep–Nov 2017
Ditch cleaning	Jan 2020	Nov 2017
Sampling years	2018–2021	2018–2019

*Soil data were sampled during August 2018.

DC was carried out at both the Pettersson and Tobo sites, using a tracked excavator. This left cleaned ditches trapezoidal in shape, about 1 m deep with a width of about 2 m at the top, tapering to 0.5 m at the bottom. Vegetation and other material deposits were removed and piled up along two sides of the ditches. At the Pettersson site, the 3-year period between harvesting (Oct 2016) and DC (Jan 2020) allowed for measurement of pre-DC conditions at the clear-cut. In contrast, the relatively short period between harvesting and DC at the Tobo site did not allow for extensive pre-DC measurements, but the similar soil chemistry and pre-DC water table data across the site suggest that there was no effect of local environmental conditions which could confound the evaluation of DC effects.

The ditch network at the two DC sites is shown in Figure 3. At the Pettersson site, the drainage ditch network was composed of two separate ditches that diverted water in different directions, allowing comparison of DC impacts on GHG fluxes in the two separate areas. At the Tobo site, the ditch network system primarily consisted of a central main ditch running north-south across the entire site, collecting water from the five perpendicular tributary ditches. DC was carried out in the central ditch and three of the perpendicular tributary ditches. The other two perpendicular tributary ditches and an additional drainage ditch at the southern edge of the site remained uncleaned.

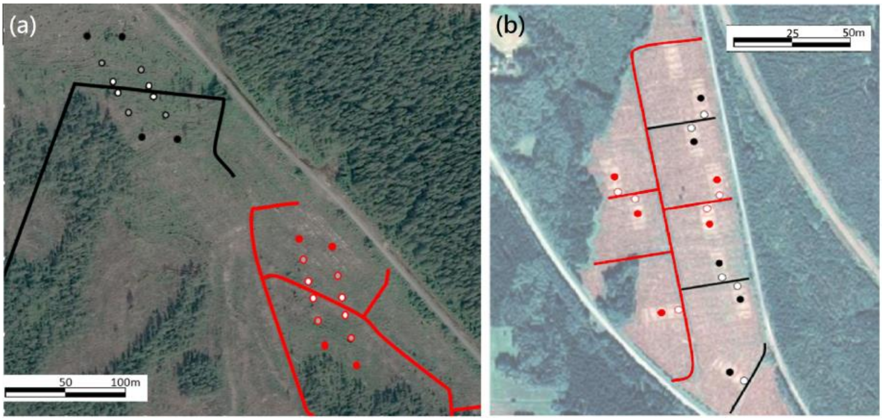

Figure 3. Map of ditch network at the (a) Pettersson and (b) Tobo field sites. Cleaned and uncleaned ditches are marked in red and black, respectively. The locations for manual chamber measurements are marked in red for the cleaned area and black for the uncleaned area (**Papers I** and **II**).

The work for **Paper III** involved studies of a forestry-drained peatland in boreal Sweden (HDPF site) located in Hälsingfors, Västerbotten, Northern Sweden, which is part of the Kulbäcksliden research infrastructure area (fFigure 4). The east and western sides of the site are defined as open and dense forest as they differ in tree density, composition and volume (Table 3). The woody dwarf shrubs *Calluna vulgaris*, *Andromeda polifolia* L., *Empetrum nigrum* and *Vaccinium oxycoccos* L., together with graminoids *Eriophorum vaginatum* L. and *Sphagnum* mosses, were the dominant understory vegetation in the open forest. In the dense forest area, dwarf shrubs including *Vaccinium vitis-idaea*, *Vaccinium myrtillus* L. and forest mosses including *Dicranum sp. and Pleurozium schreberi* dominated the understory vegetation.

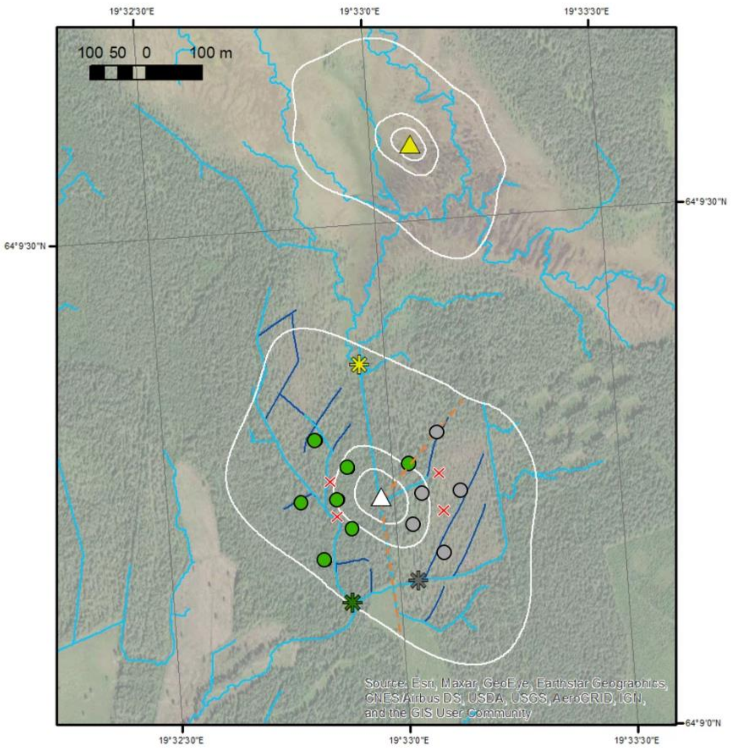

Figure 4. Experimental setup at the Hälsingfors drained peatland forest (HDPF) and mire area. The gray and green circular dots denote the measurement plots (10 m radius in scale) represented for the open and dense forest area, respectively. The white and yellow triangles denote the location of the eddy covariance (EC) towers at the forest and mire, respectively, while the white contours around each tower denote the 50, 70 and 90% footprint contours of the EC system. The orange dotted lines from the forest EC tower denote the wind direction partitioning thresholds between open and dense forest area. The red crosses denote the four monitoring locations of environmental conditions. The light and dark blue lines denote the stream and ditch network, respectively. The three weirs for water sample collections are indicated by the star symbols (white: mire area; gray and green: open and dense forest area) (**Paper III**).

Table 3. Soil and biomass information at the HDPF site (**Paper III**)

	Open forest	**Dense forest**
Carbon content	51.8 ± 0.3 %	50.6 ± 7.9 %
Nitrogen content	1.3 ± 0.1 %	1.7 ± 0.2 %
CN ratio	41 ± 3	29 ± 4

Mean stem density	620 stems ha^{-1}	1870 stems ha^{-1}
Mean stem volume	52 m^3 ha^{-1}	131 m^3 ha^{-1}
Mean tree height	8.7 m	10.0 m
Tree species proportions		
- ***Pinus sylvestris***	90%	7%
- ***Picea abies***	9%	37%
- ***Betula pubescens***	1%	56%

The adjacent Hälsingfors mire was used as a reference study for the comparison of C balance with the HDPF site. The dominant vegetation species on the mire included various *Sphagnum* mosses (e.g. *Sphagnum Cuspidata*, *Sphagnum linbergii*, *Sphagnum majus*, *Sphagnum papillosum*) and graminoids such as *Eriophorum vaginatum*, *Trichophorum sp.*, *Carex limosa* on lawns, carpets and hollows. Woody dwarf shrubs such as *Empetrum nigrum*, *Calluna fuscum*, graminoids along with sparse tree species, mainly *Pinus sylvestris* and *Betula pubescens*, were found on the hummocks.

3.2 Data sampling

3.2.1 Soil CO_2 and CH_4 flux measurements

CO_2 and CH_4 flux measurements at the two clear-cut sites for the studies reported in **Papers I** and **II** were carried out using the closed dynamic chamber method (Livingston et al., 1995). At least three weeks prior to the first measurement, square aluminium frames (48.5 × 48.5 cm) with a frame base extending down to 5 cm below the soil surface were permanently installed at each measurement plot, as indicated at Figure 3, at both study sites. In total, there were 24 measurement plots at the Pettersson site, located at 4 m, 20 m and 40 m from cleaned and uncleaned ditches, whereas 20 measurement plots were installed at the Tobo site**,** at 4 m and 40 m from both cleaned and uncleaned ditches. At each plot, measurements started with the covering of a transparent chamber to quantify the net ecosystem exchange of CO_2 (NEE). An opaque and light-reflective shroud was then put over the chamber to estimate the ecosystem respiration (R_{eco}) under dark conditions. The gross primary productivity (GPP) was then calculated as the difference between R_{eco} and NEE. The chamber closure lasted for 180–240 s with continuous monitoring of CO_2, CH_4 and H_2O concentrations using a closed loop connection to a greenhouse gas analyser. Specifically, a Gas Scouter G4301 (Picarro Inc., Santa Clara, CA, USA) was used at the Pettersson site and an ultraportable Los Gatos Research (LGR; San Jose, CA, USA) was used at the Tobo site. For both analysers, there was a built-in sampling pump which circulated air continuously between chamber and analyser. CO_2 and CH_4 measurements were carried out during daytime with roughly once every two weeks during the snow-free period (May to October at the Pettersson site and May to November at the Tobo site).

The rate of the change in CO_2 and CH_4 concentrations (dC/dt; ppm s^{-1}) was calculated using a simple linear regression over a chosen data range. Qualified gradients for the rate of change for all potential 100 s windows during the 180–240 s chamber closure were calculated and the slope with the highest coefficient of determination (r^2) was selected as dC/dt. The flux rate of CO_2 and CH_4 was then converted from dC/dt using the ideal gas law (Eqn 1):

$$F = \frac{dC}{dt} \times \frac{V \times p}{R \times T_a \times A} \text{ (Eqn 1)}$$

where F is the estimated flux (μmol m^{-2} s^{-1}), dC/dt is the linear slope with the highest r^2 of change in gas concentration per unit time (ppm s^{-1}), V is chamber headspace volume (m^3), p is the atmospheric pressure defined as 101,325 (Pa), R is the universal gas constant defined as 8.3143 (J mol^{-1} K^{-1}), T_a is the mean air temperature (K) during chamber closure, and A is the frame area (m^2).

CO_2 and CH_4 fluxes in ditches were measured at the Tobo site (**Paper II**) on a monthly basis using opaque floating chambers (diameter 31.5 cm, volume 9.56 L) in both cleaned and uncleaned ditches, using a Gas Scouter G4301 (Picarro Inc., Santa Clara, CA, USA) sampling at 1 Hz.

N_2O fluxes were measured using manual static chambers and gas chromatography. A separate set of white and opaque chambers (48.5 × 48.5 × 50 cm) was placed on the frames for 75 minutes, during which four 60 mL gas samples were collected from the chamber headspace using plastic syringes at 0, 25, 50 and 75 minutes after closure. The collected gas was then injected into 20 mL evacuated glass vials for determination of N_2O concentration. A small fan was operated inside the chamber during chamber closure to support air circulation, and a continuous temperature logger (Hobo® pendant, Onset Computers, Bourne, MA, USA) was used to monitor the temperature change inside the headspace. Determination of N_2O

concentration at each time interval involved a headspace sampler and gas chromatograph. After quantifying N_2O concentrations, the rate of change in N_2O concentration inside the chamber headspace was converted into a flux estimate using Eqn 1. N_2O measurements were carried out at the Tobo site at roughly once every two weeks during the snow-free period in 2019 (May to November in Tobo). Measurements were also taken at the Pettersson site in early and late August 2020. However, results suggested the N_2O fluxes of 16 ± 5 µg-N m^{-2} h^{-1} were negligible even considering climate impact, thus these measurements were not included in the general measurement programme.

3.2.2 Ecosystem-scale measurements of CO_2 and CH_4

Ecosystem-scale measurements of CO_2 and CH_4 flux in the drained peatland forest sites for **Paper III** were carried out using the eddy covariance method in 2020 and 2021. A 20.2 m high flux tower was installed at the centre between open and dense forest area (Figure 4). Fluctuations of wind and temperature were measured on the tower using a three-dimensional ultrasonic anemometer (uSonic-3 Class A anemometer, Metek GmbH, Germany). A closed-path CRDS LGR-FGGA sensor (Model 908-0010; San Jose, CA, USA) was used to measure the fluctuations in CO_2 and CH_4 concentrations, housed within a climate-controlled cabin under the tower. The air inlet for the closed-path gas sensor was located 19 cm horizontally from the middle of the sonic anemometer sensor, and fixed at the same height on top of the flux tower. The air was transported and pumped through a plastic tube with an inner diameter of 2.2 mm. Another eddy covariance flux tower was set up in the centre of the mire (Figure 4) equipped with a Metek Usonic-3 Class A to provide a high-frequency wind and temperature sensor, together with a closed-path Picarro G2311-f gas analyser for measuring CO_2, CH_4 and H_2O concentrations.

The high-frequency gas concentrations and environmental data recorded at both sites were processed into flux data using the open source EddyPRO software (v7.0.4, Li-COR Biosciences, Nebraska, USA). Instantaneous CO_2, CH_4 and H_2O fluxes were then averaged into half-hourly estimates which were corrected and quality-controlled using the following standard protocols to ensure consistency across both sites. Corrections included double coordinate rotation of the anemometers' axes along the local wind streamlines (Wilczak et al., 2001), removal of linear trends using block averaging and linear detrending over each 30-minute averaging period (Gash and Culf, 1996), and correction of time lags between vertical wind speed and gas concentration calculated using automatic time lag optimization (Rebmann et al., 2012). Data removed included those relating to statistical outliers, low signal strength of EC instruments and non-steady state or low turbulent conditions (Foken et al., 2004). Due to the difference in wind structure between forest and mire, low turbulence conditions in the forest were defined based on the decoupling of vertical wind speed between the below-canopy and above-canopy layer (Jocher et al., 2018; Thomas et al., 2013), whereas single-level friction velocity (u_*) was defined as the threshold for turbulent mixing in the mire (Papale et al., 2006). After all quality control and filtering processes, 33%, 29% and 42% of all half-hourly CO_2, CH_4 and H_2O values remained for the study years, respectively, of which 58%, 60% and 62% of the filtered CO_2, CH_4 and H_2O samples were classified respectively as values from the dense forest area. The remaining values were defined as being from the open forest area. At the mire, 36%, 50% and 38% of all half-hourly fluxes of CO_2, CH_4 and H_2O respectively remained after all the quality control and filtering.

The half-hourly gaps with missing CO_2 and H_2O flux data were then filled using the REddyProc online gap-filling tool to obtain annual flux sums (Wutzler et al., 2018). Values for filling the gaps were calculated based on the correlation of the fluxes with environmental variables (air temperature, global radiation and vapor pressure deficit) measured continuously at the site. The processed and gap-filled NEE was then partitioned into GPP and R_{eco}, using the Reichstein approach which uses the temperature sensitivity of night-time NEE to predict R_{eco} during the daytime (Reichstein et al., 2005). CH_4 fluxes were gap-filled using the random forests technique which involves a machine learning model evaluated by Irvin et al. (2021). The total uncertainty contributed by random measurement errors and gap-filling errors was evaluated using the Monte Carlo approach as described by Richardson and Hollinger (2007).

3.2.3 Aquatic C export

To evaluate the contribution of aquatic C export to the net ecosystem C balance in the open and dense forest area in HDPF and its adjacent mire for **Paper III**, the rates of aquatic DOC and DIC export were estimated at the three corresponding catchments over the study years by multiplying the stream discharge with the concentration of the DOC and DIC over the lateral flow.

Seasonal dynamics of stream discharge for each catchment were measured at the nearby (~3km) heated shed where the stream water level was continuously recorded and height discharge rating curves established (Ågren et al., 2008). The specific discharge derived from established height-discharge rating curves was then used to estimate the discharge from all catchments, assuming specific runoff was the same in all catchments. The specific runoff was then multiplied with the total amount of discharge estimated using a water balance approach. The annual discharge was estimated by subtracting annual rainfall from i) annual evapotranspiration balance estimated from the gap-filled H_2O flux measured at the forest and mire EC towers, and ii) the annual change in water storage calculated by multiplication of soil porosity and annual WTL change.

Concentrations of DOC and DIC were analysed once every two weeks from stream water samples collected at the end of the three corresponding catchments, with more intensive samplings (up to 12 samples) during spring flood in April and May. DOC was determined using a Shimadzu TOC-CPCH analyser (Ågren et al., 2007; Buffam et al., 2007) following the steps described by Wallin et al. (2010). DIC was analysed using a headspace method as described by Wallin et al. (2013). The study sites were located in the region where negligible particulate organic carbon (POC) concentrations relative to the dissolved fraction have been regularly reported (Ågren et al., 2008; Ivarsson and Jansson, 1995; Laudon and Bishop, 1999). Thus, the concentrations of total organic carbon (TOC) in the collected samples are defined solely as DOC in this thesis.

3.2.4 Environmental data

For all the studies described in this thesis, environmental variables were recorded manually alongside flux measurements over the year. Abiotic variables measured included air temperature and soil temperature at various depths (5 cm, 10cm and 15cm), water table level (WTL), soil moisture (SM), and photosynthetic active radiation (PAR). Biotic variables included normalized difference vegetation index (NDVI), greenness index, and a direct estimation of vegetation areal coverage was also made selectively at different sites. Vegetation greenness index was defined by the green chromatic coordinate (g_{cc}) from aerial pictures, which refers to the relative intensity of green image channels. These variables were recorded manually alongside chamber measurements for **Papers I** and **II**, in order to i) investigate the environmental drivers of the measured gas fluxes through statistical analyses, ii) to develop nonlinear models to estimate annual fluxes (See section 3.5), and iii) to calibrate the continuous data which served as input to the developed models (See section 3.5). These environmental variables were also recorded continuously at hourly intervals over the entire study period for **Papers I** and **II**, in order to provide year-round records as input for estimating the annual fluxes with the developed models. For **Paper III**, all continuous environmental sensors were operated with CR1000 data loggers (Campbell Scientific Inc., Logan, UT, USA), which allowed continuous data recording at minute intervals. The specific equipment used for each parameter in each site is shown in Table 4.

Table 4. Summary of the abiotic data collection for the three study sites in this thesis.

Paper **Site Name**	**I** **Pettersson**	**II** **Tobo**	**III** **Hälsingfors**
Air temperature			
Manual data	Hobo® temperature logger	Handheld thermometer	/
Continuous data	Hobo® temperature logger	Nearby weather station	Campbell HC2S3
Soil temperature			
Manual data	Handheld thermometer	Handheld thermometer	/
Continuous data	TOMST® TMS-4 sensors	Campbell CS655	TOJO TR03
PAR			
Manual data	Hobo® radiation logger	QSO-S Photon Flux Sensor	/
Continuous data	Hobo® pendant radiation logger	Nearby weather station	Li-Cor Li-190
WTL			
Manual data	Direct measurements	Direct measurements	/
Continuous data	Solinst Levelogger 5	TruTrack WT-HR 1000 probes	CS451 pressure transducers
Soil moisture			
Manual data	Campbell GS3	Campbell GS3	/
Continuous data	TOMST® TMS-4 sensors	Campbell CS655	/

Vegetation growth			
Manual data	Aerial images	Aerial images	
Continuous data	TimeLapseCam		Decagon spectral reflectance sensors

3.3 Data analysis

The work for this thesis used various types of statistical analyses on the different data structures produced by the manual chamber measurements (**Papers I** and **II**) and the eddy covariance technique (**Paper III**). Specifically, data from manual chamber measurements were characterised by multiple dimensions from various sampling locations and at different times of the year. Therefore, a set of statistical analyses was used for **Papers I** and **II** in order i) to investigate the impact of drainage or DC on the fluxes through the mediating and moderating effect of the environmental variables, and ii) to provide estimated annual balances of C and GHG fluxes using model extrapolations (Figure 5).

Statistical results from the mixed effect models were considered significant at $p < 0.05$. The standard error (± SE) of the sample averages has been used as a measure of uncertainty throughout this thesis. All statistical analyses were carried out using the Mathworks Matlab software.

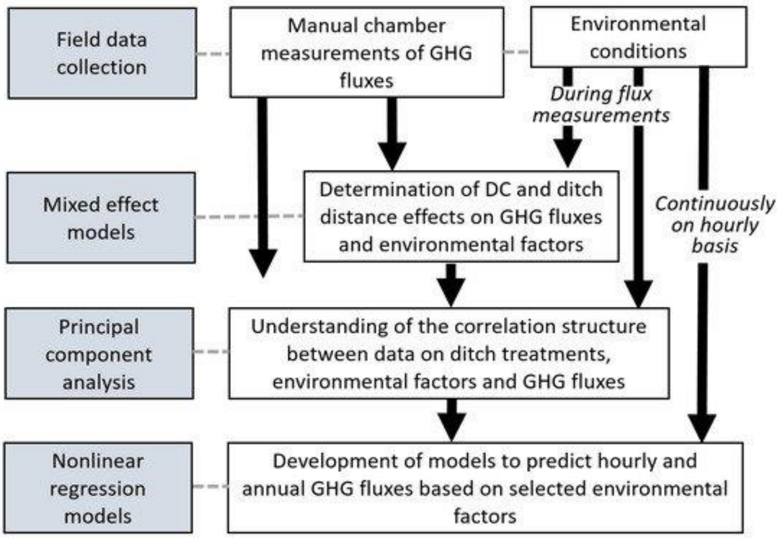

Figure 5. Flowchart of study methodology for **Papers I** and **II** (**Paper I**).

3.3.1 Significance of DC effects on flux and environmental variables

First, mixed effect models with repeated measurements were used for **Papers I** and **II** to quantify the statistical significance level of the DC treatment and the distance effects on the spatio-temporal variation of environmental variables (Table 4) and GHG flux variables (Eqn 2). These models included a spatial covariance structure where correlations weaken over time (Phillips et al., 2001). The statistical models applied were as follows:

$$y = \beta_0 + \beta_1 T + \beta_2 D + S + \varepsilon \quad \text{(Eqn 2)}$$

where y denotes the environmental or GHG flux variable, β_0 denotes the overall mean of the environmental or GHG flux variable, β_1 denotes the fixed effect of the sensitivity to treatment T, β_2 denotes the fixed effect of the sensitivity to distance to ditch D, S denotes the random effect of sampling event presenting a covariance structure where correlations weaken over time (Phillips et al., 2001), and ε denotes the random error. Mixed effect models have been proven to be robust for different data distributions (Schielzeth et al., 2020).

3.3.2 High dimensional structures among variables

Principal component analysis (PCA) was carried out for **Papers I** and **II** to visualise the high dimensional structures of all the variables, including the GHG fluxes, environmental parameters and treatments (DC and distance). The input variables to the PCA were first normalized by subtracting their respective means and dividing by the standard deviation of the variable (Jolliffe, 1990). Significant principal components (PCs) were chosen and displayed using the Kaiser criterion (Kaiser, 1960). The underlying relationships between variables were evaluated using variable loadings, which are defined as the correlation between each variable and PC (Cadima and Jolliffe, 1995).

3.3.3 Estimating annual budgets of C and GHG balances

The manual chamber measurements used for **Papers I** and **II** are only periodic snapshots of the biotic-abiotic conditions affecting the fluxes at a specific location. To provide annual estimates of C and GHG balances that are representative of the entire site, the work described in **Papers I** and **II** was a development of nonlinear models to extrapolate the response of a flux to biotic and abiotic influences. Relevant techniques for such nonlinear models for the prediction of CO_2 component fluxes have been well used in numerous previous studies (e.g. Järveoja et al., 2016a, 2016b; Kandel et al., 2013; Olson et al., 2013). In particular, the measured GPP from each sampling plot was fitted to PAR and frame-specific vegetation greenness data using a hyperbolic PAR function adjusted with normalised frame-specific g_{cc} representing seasonal variations in vegetation biomass (Eqn 3):

$$GPP_{(hr,frame)} = (\alpha \times P_{max} \times PAR \times g_{cc_{\mathrm{norm}}})/(\alpha \times PAR + P_{max} \times g_{cc_{\mathrm{norm}}}) \text{ (Eqn 3)}$$

where GPP denotes the gross primary production (mg m^{-2} h^{-1} of CO_2-C), α denotes model fitted value of the initial slope of the light use efficiency of photosynthesis (mg µmol photons^{-1} of CO_2-C), PAR denotes the mean photosynthetically active radiation (µmol m^{-2} s^{-1}), P_{max} denotes the modelled fitted value of maximum photosynthetic rate under light saturation (mg m^{-2} h^{-1} of CO_2-C), and $g_{cc_{\mathrm{norm}}}$ denotes the frame-specific chromatic greenness index (g_{cc} (JD)) normalised to a scale between 0 and 1.

In the R_{eco} model, an exponential relationship was used with T_a, based on Lloyd and Taylor (1994), adjusted using the normalised frame-specific g_{cc} as the second predictor variable (Eqn 4):

$$R_{eco_{(hr,frame)}} = R_0 \times exp^{b\times T} + \left(\beta \times g_{cc_{\mathrm{norm}}}\right) \times exp^{b\times T} \text{ (Eqn 4)}$$

where R_{eco} denotes ecosystem respiration (mg m^{-2} h^{-1} of CO_2-C), and T denotes temperature of the soil at 5cm depth (T_{s5}; Paper I) or air temperature (T_a; Paper II) (°C). Fitted parameters include R_0 which denotes R_{eco} at 0°C (mg m^{-2} h^{-1} of CO_2-C), b which denotes respiration sensitivity to T_a, and β which is a scaling factor for plant development which refers to the contribution of plant autotrophic respiration (R_a) to R_{eco}.

For **Paper II**, CH_4 fluxes were modelled using an exponential relationship with WTL and temperature of the soil at 10 cm depth (Olson et al., 2013) (Eqn 5):

$$CH_{4_{hr,frame}} = exp^{b_0+b_1 \times WTL+b_2 \times T_{s10}} \text{ (Eqn 5)}$$

where CH_4 denotes CH_4 flux (g m^{-2} h^{-1} of CH_4-C), b_1 and b_2 denote the model fitted sensitivity of CH_4 flux to water table level (WTL, cm) and temperature of the soil at 10 cm depth (T_{s10}, °C), respectively, and b_0 denotes the model intercept. As described in **Paper I**, there was a weak relationship between CH_4 fluxes and environmental variables, thus annual CH_4 balances were based on interpolation from the median of measured CH_4 fluxes.

The continuous hourly environmental data, after calibrating with the manually measured data ($R^2 > 0.9$ for both **Papers I** and **II**), were used as input variables to the respective model equations. The diel hourly fluxes generated from the models were then summed for the entire year to obtain the annual balance estimates.

4. Results and discussion

4.1 Effect on drainage on environmental conditions

4.1.1 DC impact on environmental conditions

The temporal variations of WTL at both clear-cut sites are shown in Figure 6. In general, the Tobo site had a significantly shallower WTL than the Pettersson site over the entire measurement period. Furthermore, the two sites were characterised by a different response to DC. Specifically, during the year before DC, the mean WTL at the Pettersson site was already 5 cm higher in the uncleaned area (−27 ± 1 cm) than in the area to be cleaned (−32 ± 1 cm). After DC in January 2020, however, the mean WTL in the uncleaned area rose to −19 ± 1 cm in 2020 and −15 ± 1 cm in 2021. At the same time, the WTL in the DC area remained at a same level as before DC, being −37 ± 2 cm in 2020 and −32 ± 1 cm in 2021. Thus, following DC, the mean WTL in the cleaned area was 18 ± 2 cm and 17 ± 2 cm lower than in the uncleaned area in 2020 and 2021, respectively. Considering the 5 cm difference of WTL between the two areas before DC, this indicates that DC enhanced the WTL difference between the two areas by 12 ± 2 cm. Over the four years, the differences in WTL between the two areas remained similar over a year, implying insignificant seasonal variations of the DC effects on WTL changes.

There was a lack of comprehensive pre-DC WTL data due to the subsequent DC after harvesting. However, preinstalled WTL sensors recorded an insignificant difference ($p = 0.74$) between WTL in the uncleaned area (-35 ± 3 cm) and area to be cleaned (-33 ± 4 cm) during the period between harvesting and DC in October 2017. In the first year after DC, the mean WTL was −54 ± 1 cm in the uncleaned area and −57 ± 1 cm in the cleaned area, remaining statistically the same across the two areas (Figure 6). The DC effect on WTL became visible only in 2019, when a much deeper mean WTL (−65 ± 2 cm) was recorded in the cleaned area than in the uncleaned area (−56 ± 2 cm) (Figure 6). The largest deviations were observed between the two areas during the peak growing season (June to August) when the WTL was the deepest (< −80 cm) in the cleaned area.

The deep mean WTL of -55 cm at the Tobo site was in contrast to that at the Pettersson site, as well as previous studies at drained peatland forests in Finland with shallower mean WTLs of -30 to -40 cm (e.g. Leppä et al., 2020; Lohila et al., 2011; Ojanen and Minkkinen, 2019) (Figure 6). Such hydrological conditions with low WTL at the Tobo site were also commonly found at the nearby ditched peatland forest areas according to the national SLU Soil Moisture Map (Ågren et al., 2021).

The delayed response of WTL to DC at the Tobo site during 2018 could be attributed to the enhanced transpiration from the earlier and more developed herbaceous vegetation in the uncleaned area which might have counterbalanced the enhanced drainage effect at the cleaned area. Aerial pictures taken in 2018 clearly indicated the higher abundance of in-frame vegetation in the uncleaned area (mean areal coverage: 47 ± 13 %; mean greenness index: 0.37 ± 0.01) than in the cleaned area (mean areal coverage: 13 ± 5 %; mean greenness index: 0.34 ± 0.00). Less vegetation growth in the cleaned area could be explained by the combined effect of meteorological drought stress and additional soil water reduction following DC, which together might have made it more difficult for herbaceous vegetation to establish in the first year. The importance of herbaceous vegetation on the water regulation along the soil profile was also indicated by Ruseckas et al. (2015). In 2019, however, ground vegetation became more abundant across the entire site, without significance differences between the two areas. In comparison, the seasonal maxima of the vegetation growth at the Pettersson site was statistically similar between the uncleaned and cleaned treatment areas, during both pre-DC and post-DC periods.

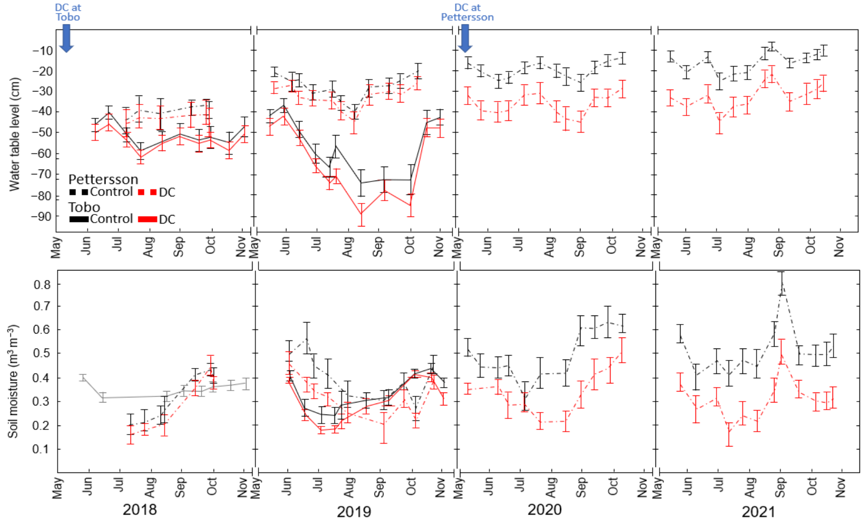

Figure 6. Time series of water table level (WTL) and soil moisture (SM) monitored at the measurement plots during flux measurements at the Pettersson (**Paper II**) and Tobo sites (**Paper II**). Data are averaged for each sampling event and grouped by experimental area (control versus DC) and year (2018–2021 for the Pettersson site and 2018–2019 for the Tobo site). The variable means for each sampling event ± standard error (SE) are presented for both control and DC areas, respectively.

Relative to WTL, consistent spatial and temporal patterns were observed for soil moisture, with significant deviations observed only during the post-DC period at both study sites (Figure 6). It is, however, notable that the overall difference in soil moisture between the two sites was less significant than WTL, based on the data in 2019 when soil moisture remained similar over the two sites.

In addition to the relatively apparent changes in WTL and vegetation growth after DC, the DC treatment was associated with increased soil temperature at Pettersson site, as indicated in the PCA (Figure 7). At the Pettersson site, the linear mixed effect models further suggested significant higher mean soil temperatures of the cleaned area. Drainage has also been reported to lower soil thermal inertia and accelerate soil warm-up, in particular during the warm season (Jin et al., 2008; Prévost et al., 1999).

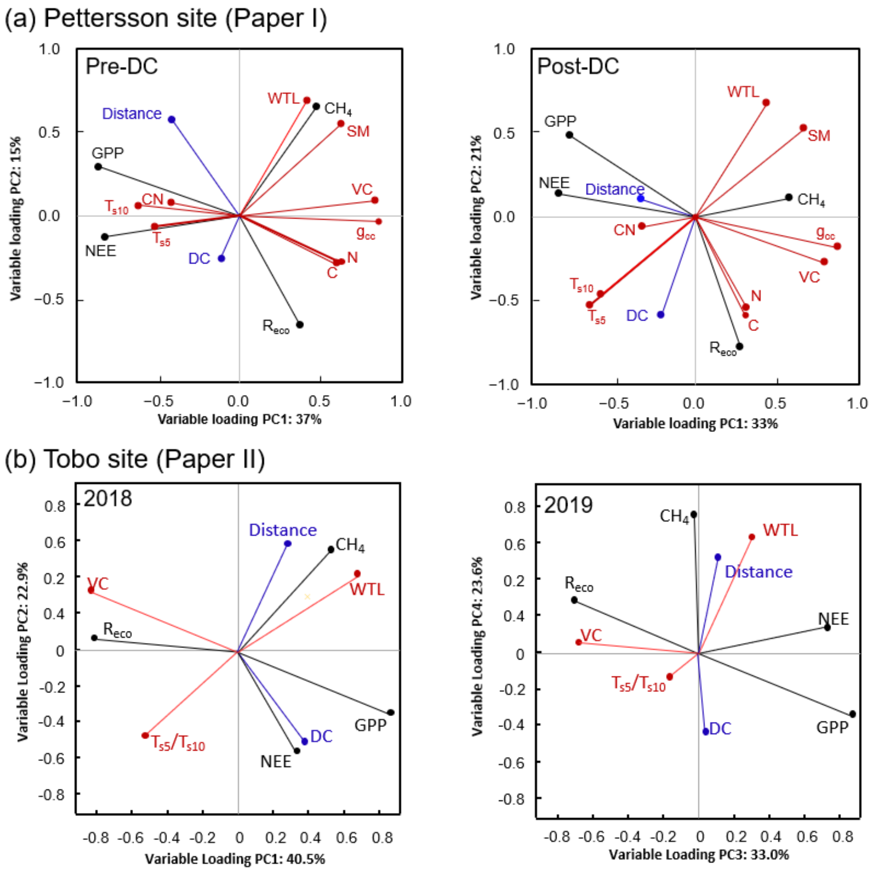

Figure 7. Principal component analysis (PCA) biplots for the (a) Petterson site and (b) Tobo site. For both sites, the first two significant principal components (PC1 and PC2), based on the plot-averaged measurement data displaying variable loadings and object scores, are shown. For the Pettersson site, the two panels denote pre-DC (panel on the left) and post-DC (panel on the right) periods. For the Tobo site, the two panels denote measurement years of 2018 (panel on the left) and 2019 (panel on the right). Loadings, representing the measured variables, are indicated by solid symbols. Solid symbols with lines denote the component loadings of the measured variables. Environmental variables, treatment variables and flux variables are shown in different colours.

4.1.2 Long-term drainage impact on environmental conditions

The change in vegetation composition between the drained peatland forest and the adjacent mire, as reported in Paper III, is described in Section 3.1. Table 5 shows the significant differences in the environmental factors between the study areas. These include the significantly lower soil temperature at 15 cm depth and higher NDVI at the HDPF during the growing season. Lower soil temperatures under the forest canopy were also reported by associated studies (Pihlatie et al., 2010; Solondz et al., 2008), highlighting the role of change of vegetation structure on the soil's physical properties.

Table 5. Environmental variables at HDPF and the adjacent mire area.

Variable	Measure of tendency	Mire	HDPF	
			Open Forest	Dense Forest
T_{s15} (°C)	mean (maximum)	5.7 (19.0)	4.8 (12.9)	4.7 (12.1)
WTL (cm)	mean (range) growing season	−33 (−55 to −12)	−15 (−33 to +3)	−4 (−27 to +25)
NDVI	mean midday growing season	0.41	0.57	0.69

4.2 Effect of drainage on spatio-temporal dynamics of CO_2 flux

4.2.1 DC impact on spatio-temporal dynamics of CO_2 flux

Over the four study years at the Pettersson clear-cut site, the instantaneous daytime NEE switched from net emissions of 72 ± 9 mg C m^{-2} h^{-1} to a net uptake of 168 ± 27 mg C m^{-2} h^{-1}, averaged over all treatment plots. Both the peak growing season GPP and R_{eco} increased significantly over the four measurement years in both the uncleaned and cleaned areas. In relation to the large inter-annual variations, the treatment effects remained small during the four measurement years. Although the mixed effect models suggested that the measured R_{eco} and GPP would increase slightly but significantly ($p <$ 0.02, 5% to 18%) at the cleaned area as compared to the uncleaned area, their changes offset each other resulting in inconsequential changes in their sum of NEE after DC ($p > 0.05$). PCA results further indicated that CO_2 component fluxes were independent of DC effects on soil temperature, WTL and moisture levels. Instead, these fluxes were primarily controlled by vegetation growth which remained positively correlated to soil C and nitrogen contents.

A large decrease in daytime NEE was also observed within the two study years at the Tobo site. As at the Pettersson site, there was a high inter-annual variability as CO_2 component fluxes of R_{eco} and GPP increased substantially over the two years. However, smaller CO_2 component fluxes at the ditch cleaned area were observed at the Tobo site, which contradicts the finding at the Pettersson site. Specifically, the mean GPP and R_{eco} were 16% to 51% lower in the cleaned area than in the uncleaned area. The mixed effect models further suggested that DC significantly reduced R_{eco} and GPP in both years ($p < 0.02$). Yet, their sum NEE was not significantly affected by DC in both years ($p > 0.05$), switching from a mean of 89 to 39 mg mg C m^{-2} h^{-1} at the uncleaned area in the two study years; at the cleaned area, the mean changed from 109 to −15 mg C m^{-2} h^{-1} in the two study years.

The different impact of DC on the CO_2 component fluxes across the two study sites can likely be attributed to the varying WTL and vegetation growth conditions over the two sites, which were identified as primary factors in the spatial variation of CO_2 component fluxes. The Pettersson site was characterised by its wet soil in which the uncleaned area had a shallow WTL of −17 cm during the studied years. The wet soil resulted in a narrow upper aerobic zone which likely limited decomposition and the development of vascular plants, thus explaining the limited R_{eco} and GPP noted in the uncleaned area. The impact of WTL drawdown on the CO_2 flux components began to decrease when WTL was <−20 cm, which was in agreement with another study of a peatland forest clear-cut in boreal Finland (Korkiakoski et al., 2019). At the Tobo site, however, the dry soil, having the lowest WTL reaching −107 cm during summer 2019, might imply that the additional drainage following DC might have led to drought-induced inhibition of microbial activity and reduced heterotrophic components of respiration (Drzymulska, 2016; Manzoni et al., 2012). In addition, plant production and the respiration could be further limited as a more extensive cover of herbaceous plants species (e.g. *Chelidonium majus*) was observed in the uncleaned area. This may have, eventually, supported heterotrophic decomposition by providing additional input of easily decomposable organic matter (Thormann et al., 2001).

It is noteworthy that plots at difference distances from the ditch (4 m to 40 m) were involved in both the Pettersson and Tobo studies, but a clear pattern of the effect of ditch distance on CO_2 fluxes was not observed in either uncleaned or cleaned areas. The lack of a clear distance to ditch effect on CO_2 fluxes is likely explained by the fact that primary sources of CO_2 production and respiration, such as vegetation growth and soil temperature, did not have a consistent relationship with ditch distances in both studies.

Overall, the impact of DC on the R_{eco}, GPP and NEE largely depended on both the initial WTL conditions and the efficiency of DC, which together regulated the change in soil surface aerobic conditions and subsequently the production and decomposition of organic matter in the initial years following clear-cutting.

4.2.2 Long-term drainage impact on spatio-temporal dynamics of CO_2 flux

A comparison of NEE between HDPF and the adjacent mire indicates that historical forestry drainage has substantially increased the net sink of CO_2. During the study period, the NEE at the mire remained small for the majority of the time, except for a net uptake of CO_2 during some periods of the growing season (Figure 8). Altogether, the annual NEE was -1.4 ± 0.1 t-C ha^{-1} year^{-1} during the whole of 2021. The small mire CO_2 uptake reported in this study was different from the earlier study by Nilsson et al. (2008) which reported a larger CO_2 uptake (-4.8 to -5.5 t-C ha^{-1} year^{-1}) at the nearby Degerö mire. A possible reason is that pool areas found in this mire site, which have been reported to constrain emergent vegetation and reduce ecosystem CO_2 uptake capacity (Pelletier et al., 2015), were identified using partitioning analysis as a source of CO_2.

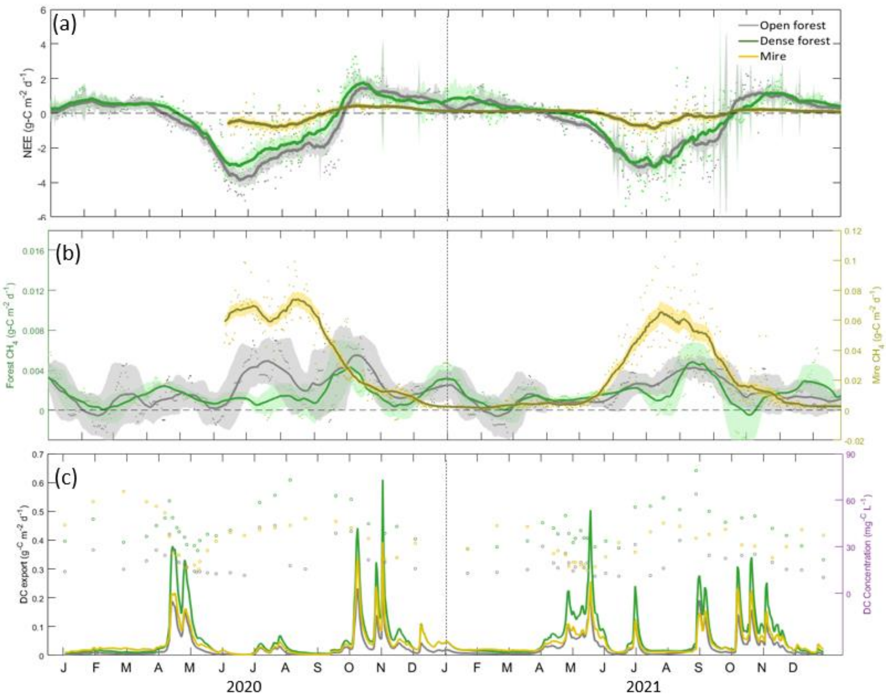

Figure 8. Daily time series for both 2020 and 2021 of the daily sum of (a) net ecosystem exchange (NEE), (b) methane (CH_4) and (c) total dissolved carbon (DC) export for both open and dense forest in HDPF and the adjacent mire. In (a) and (b), the shaded areas represent the standard deviation of hourly fluxes, while the bold dark lines denote the 30-day running means. (**Paper III**)

The forestry-drained peatland was a stronger sink of CO_2 over both study years. Specifically, in the open forest, the annual NEE was −18.4 ± 3.4 and −17.2 ± 3.0 t-C ha^{-1} year^{-1} for 2020 and 2021, respectively, while the annual NEE in the dense forest was −8.3 ± 2.9 and −12.3 ± 3.0 t-C ha^{-1} year^{-1} for the two years, respectively. The larger CO_2 uptake from the open forest relative to the dense forest, despite the lower tree density, has highlighted the role of ground vegetation uptake on the forest C budgets. With higher availability of solar radiation reaching the open forest floor (Baldocchi et al., 2000), the contribution of the forest floor vegetation to momentary CO_2 exchange may also be large. Goulden and Crill (1997) estimated that the ground vegetation of boreal spruce forests can account for up to 50% of the total forest photosynthesis. Therefore, while it is found that historical forestry drainage has increased CO_2 uptake, the forestry structure had substantial influence on its magnitude as both tree biomass and understory vegetation could have potential importance on the spatial heterogeneity of CO_2 balance.

4.3 Effect of drainage on spatio-temporal dynamics of CH_4 flux

4.3.1 DC impact on spatio-temporal dynamics of CH_4 flux

The Pettersson site was characterised by a small CH_4 flux over the entire site before DC (10–90 percentile: −0.06 to +0.07 mg C m^{-2} h^{-1}; median: −0.02 mg C m^{-2} h^{-1}). The mean WTL of −30 cm at the site created a substantial surface oxic layer and provided favourable conditions for CH_4 oxidation, preventing substantial CH_4 emissions from the deeper soil layer (Korkiakoski et al., 2019). After DC, the number and magnitude of CH_4 emission spikes increased over the entire site including the cleaned area, which was likely due to the increased vascular ground vegetation (e.g. *Deschampsia flex* spp.) noted in these DC plots, with deep roots providing substrate to methanogens and supporting the plant-mediated transport of CH_4 into the atmosphere (Gauci et al., 2010; Korkiakoski et al., 2021; Terazawa et al., 2007). Much greater emission spikes were observed in the uncleaned area, up to 3.6 mg C m^{-2} h^{-1}, recorded at the plots with shallow WTL (between 0 and −20 cm) under saturated soil conditions (> 0.45 m^3 m^{-3}). As there was no consistent relationship between ditch distance and WTL across the site, ditch distance had no consistent effect on CH_4 fluxes. Altogether, the median CH_4 flux was 0.08 mg C m^{-2} h^{-1} in the control area and −0.05 mg C m^{-2} h^{-1} in the DC area, with significantly (p < 0.01) higher emissions of CH_4 from the uncleaned area.

The CH_4 flux at the Tobo site was characterised as a consistent uptake by both cleaned and uncleaned areas during the study years, ranging between −0.06 to −0.09 mg C m^{-2} h^{-1} across the entire site. The magnitude of uptake in the cleaned area did not differ substantially from the cleaned area in the first year but the slight increase (~8%) of uptake became apparent in the second year (2019) following DC ($p < 0.05$). This might be explained by the delayed effect of DC on lowering WTL, the latter being a major influence on CH_4 fluxes (Maljanen et al., 2010; Ojanen et al., 2010) as further evident from the PCA results (Figure 6). As WTL was found to deepen with proximity to both cleaned and uncleaned ditches, the net uptake of CH_4 decreased ($p < 0.01$) with distance from both uncleaned and cleaned ditches in both growing seasons. Furthermore, ditch emissions of CH_4 were 0.01 ± 0.01 mg C m^{-2} h^{-1} and 0.12 ± 0.11 mg C m^{-2} h^{-1} at cleaned and uncleaned ditches, significantly lower than the 0.19–6.84 mg C m^{-2} h^{-1} reported by previous studies of ditches in other drained peatland forests (Ball et al., 2007; Peacock et al., 2021; von Arnold et al., 2005). This further highlighted the dry conditions in both cleaned and uncleaned ditches which favoured aerobic CH_4 oxidation and suppressed anaerobic CH_4 production (Nykänen et al., 1998).

Taking the two sites together, it was apparent that DC potentially constrained anaerobic CH_4 production and increased CH_4 uptake by lowering the WTL, which was identified as the primary influence on methanotrophic and methanogenic processes. The initial WTL conditions determined the strength of DC to suppress anaerobic CH_4 production and increase its uptake (Nykänen et al., 1998). The herbaceous species in the uncleaned ditches were also found to enhance potential CH_4 emission through root-derived substrate supply (Zhang et al., 2002).

4.3.2 Long-term drainage impact on spatio-temporal dynamics of CH_4 flux

Continuous measurement data, described in **Paper III,** have indicated that the growing season emission of CH_4 from the mire was, on average**,** 10 times higher than from the drained peatland forest areas, highlighting the significant impact of historical drainage on the reduction of CH_4 emissions. Specifically, the mean CH_4 flux was 6.8 ± 0.06 g C m^{-2} $year^{-1}$ (or 0.78 ± 0.01 mg C m^{-2} h^{-1}) from the mire over 2021. In comparison, the HDPF was a small CH_4 source, being 0.07 ± 1.2 t-C ha^{-1} $year^{-1}$ (or 0.08 ± 0.01 mg C m^{-2} h^{-1}) in the open forest and 0.06 ± 0.01 t-C ha^{-1} $year^{-1}$ (or 0.07 ± 0.01 mg C m^{-2} h^{-1}) in the dense forest, averaged over the two study years.

This spatial heterogeneity further confirmed that WTL drawdown after historical drainage was the major regulator of CH_4 production and the oxidation zone from peat layers (Nykänen et al., 1998). It is, however, notable that with the same WTL and soil temperature, the drained forest still produced significantly lower emissions of CH_4 than the mire, which indicated other factors influencing the spatial variability of CH_4 emissions. Possible explanations include the difference in vegetation types which altered the substrate supply for methanogenesis in the deep anoxic peat layer after drainage (Sundh et al., 1994). For instance, the aerenchymatous plants commonly found in the mire area possibly enhanced CH_4 transport from the anoxic layer through their deep root system (Grosse et al., 1996; Joabsson et al., 1999). Hence, long-term drainage did not only alter CH_4 dynamics through a change in hydrological conditions, but also possibly through changes in vegetation composition and soil nutrient conditions that regulated CH_4 fluxes.

4.4 Effect of drainage on spatio-temporal dynamics of N_2O flux

While negligible N_2O fluxes were reported at the Pettersson site, N_2O fluxes at the Tobo site occurred mostly within the 10–90 percentile range of −14 to 34 $\mu g\ N\ m^{-2}\ h^{-1}$ across both uncleaned and cleaned areas. The spatial variability in N_2O fluxes was caused by sporadic high and low fluxes at a given measurement location. A clear DC effect on N_2O fluxes was not observed based on the statistical analysis. Despite the nutrient-rich conditions at the Tobo site, the small N_2O emissions at this site were possibly also the result of the low soil moisture content relative to previous studies in drained peatland forest clear-cuts (e.g. Saari et al., 2010), as suppressed denitrification was previously reported at WTLs below −30 cm (Hefting et al., 2004). Small N_2O fluxes from soil were also measured in the HDPF area, being 22 $\mu g\ N\ m^{-2}\ h^{-1}$ in the open forest to −22 $\mu g\ N\ m^{-2}\ h^{-1}$ in the dense forest, measured during the peak growing season of July 2020.

The studies described in this thesis were unable to provide a detailed understanding of the production and emission of N_2O. This is because N_2O fluxes are commonly highly variable in both space and time, as they depend on a complex series of different processes and pathways (Robertson and Tiedje, 1987; Webster and Hopkins, 1996). Thus, the limited sampling campaigns in these studies could have failed to capture occasional emission events (Smith and Dobbie, 2001). Therefore, it is recommended that the spatial and temporal resolution for measurements is increased, using, for example, automated chamber or eddy covariance flux systems, in order to study the response of N_2O emissions to drainage in more detail (Pihlatie et al., 2005).

4.5 Effect of drainage on spatio-temporal dynamics of aquatic C fluxes

The total estimates for the aquatic export of dissolved C from the open and dense drained peatland forest, as described in **Paper III**, were below and above the estimates from the adjacent mire, respectively. Specifically, the annual export of dissolved C was estimated at 0.94 ± 0.10 and 2.05 ± 0.21 t-C ha^{-1} $year^{-1}$ in the open and dense forest, respectively, averaged over the two study years. Meanwhile, the total export of dissolved C was 1.46 ± 1.10 t-C ha^{-1} $year^{-1}$ averaged over the two study years.

This difference of dissolved C between the two forest areas suggested that forest structure within the forest area had a greater influence on aquatic DC export relative to the long-term drainage impacts. DOC, the main component of dissolved carbon, was found to have a positive correlation between tree stand volume and increasing DOC trends (Nieminen et al., 2021), which supported the findings given in **Paper III**. The increased amount of litterfall and exudates in a dense forest has been shown to correspond to increasing DOC exports in Norwegian systems (Finstad et al., 2016). However, our results differed somewhat from those of Nieminen et al. (2021) which indicated higher DOC concentrations in drained peatlands than in natural mires. It is, however, notable that the drainage effect on DOC concentrations was only found to be significant in southern latitudes in Finland (Finér et al., 2020), suggesting the limited impacts in the central boreal region studied in this paper.

In contrast to DOC, the concentration of DIC was the highest from the mire area. This is likely due to the wide range of Sphagnum species in the mire, found dominating most of the stratigraphy (60–370 cm depth) from the nearby Degerö mire (Larsson, 2016), making it able to sustain high concentrations of DIC (Leach et al., 2016). The higher DIC concentration from the mire during low runoff seasons further reinforced the role of Sphagnum species on DIC export through subsurface recharge.

As the studies described in **Papers I** and **II** focused primarily on terrestrial-atmospheric exchanges of C and GHG, the lateral discharge of C was not measured in the two clear-cut sites as part of understanding their responses to DC. However, a recent study carried out in drained boreal peatland forests has indicated that DC did not increase export of dissolved carbons (Nieminen et al., 2018). It was suggested that DC would not change the export of dissolved carbons in the short term unless water pathways were switched after DC so that they diverged to a layer of high concentration, easily releasable, recently dead organic matter (Åström et al., 2001).

Taken together, DOC and DIC concentrations were both sensitive but responded differently to land cover heterogeneity, which brought significant variations in their annual export. It is known that, for the flat landscapes described in **Paper III**, the water samples collected for determination of dissolved C concentration could partially have originated from outside the representative area, as a result of the overlapping ditch network running across a flat landscape. Thus, more frequent sampling at different stream orders is suggested for future studies.

4.6 Drainage impacts on the annual C and GHG balances

4.6.1 Contribution of CO_2 to annual C and GHG budget

The ecosystem's C and GHG balance for drained peatland forests and clear-cut areas involved for this work were dominated by CO_2 exchange, being in line with earlier studies of the GHG balance of boreal forest sites (e.g. Ojanen et al., 2013) and clear-cut sites on drained peatlands (e.g. Korkiakoski et al., 2019; Vestin et al., 2020). The model estimates suggested that both clear-cut sites were an annual source of C during the initial years, but a decreasing trend of NEE was observed at the Pettersson site for four years of measurement, from 6.6 ± 1.2 t-C ha^{-1} $year^{-1}$ in 2018 to 1.5 ± 1.3 t-C ha^{-1} $year^{-1}$ in 2021, averaged over all the study plots (Figure 9). At the Tobo site, higher emissions were estimated by the model in the second post-harvest year, 18.0 ± 6.1 t-C ha^{-1} $year^{-1}$ in the uncleaned area and 10.1 ± 3.7 t-C ha^{-1} $year^{-1}$ in the cleaned area (Figure 9).

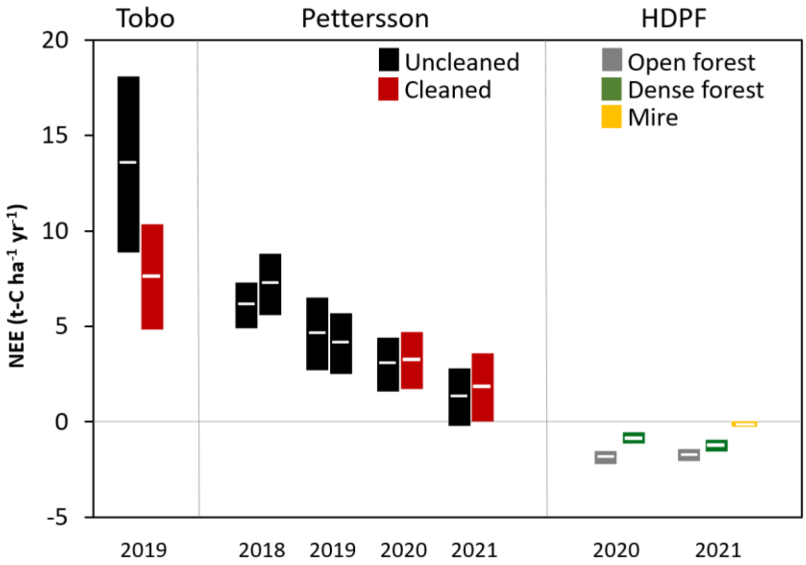

Figure 9. Model estimates for the total annual carbon dioxide (CO_2) balance of various areas at the Tobo site (**Paper II**), the Pettersson site (**Paper I**) and HDPF sites (**Paper III**). Range of bars denotes the standard deviation from the means shown with white lines. NEE estimates for the Tobo and Pettersson sites were based on model interpolation of chamber measurements over the study years. Note that estimates for the Pettersson site were based on field layer measurements, while estimates for the Tobo site were estimated using spatial averaging over field layer and ditches.

In spite of the slightly higher (<10%) modelled R_{eco} and GPP in the ditch cleaned area at the Pettersson site, the model estimates of their annual sums of NEE were statistically the same between the two experimental areas during both post-DC years. The apparent discrepancy in the DC effect between the modelled annual sums relative to the instantaneous daytime measurements (Section 4.2) could be due to the greater vegetation coverage of the surrounding area compared to inside the measurement frames in the uncleaned treatment. While the measured fluxes corresponded to the vegetation coverage inside the measurement frames, the vegetation development in the surrounding area was used as input in the model estimation in order to provide evaluations that were representative of the entire area. The higher vegetation coverage in the surrounding control area could be accounted for by the higher prevalence of Sphagnum moss species which are favoured in shallow WTL conditions. Thus, given the dominating contribution of CO_2 to the C and GHG budgets, it is concluded that the total GHG budget remained statistically similar after DC at the Pettersson site.

The model results for the Tobo site were consistent with the daytime measurements recorded. Specifically, there was a significant decrease in the CO_2 balance and, subsequently, the GHG balance after DC, in response to the decrease in heterotrophic respiration following WTL draw-down. This indicates that the enhanced drought stress after DC activities on dry sites might have suppressed microbial activities and net CO_2 emissions. Overall, CO_2 contributed predominantly to the total GHG budget, by 99% and 98% in uncleaned and cleaned areas, respectively.

The combination of terrestrial and aquatic C flux components in the HDPF enabled estimation of the net ecosystem carbon balance (NECB) of this boreal peatland forest ecosystem (Table 6). Results indicated that the forest was a net C sink of −16.8 ± 2.3 t-C ha^{-1} $year^{-1}$ in the open forest area and −8.2 ± 2.1 t-C ha^{-1} $year^{-1}$ in the dense forest area during the study period, for which the magnitude of the ecosystem-atmosphere CO_2 exchange contributed more than 80% to the NECB in absolute terms. The main proportion of terrestrial CO_2 exchange to NECB was comparable with a study carried out nearby (Chi et al., 2020), but even higher CO_2 uptakes from −23.4 to −57.0 t-C ha^{-1} $year^{-1}$ were reported in other boreal drained peatland forests in Fennoscandia (Lohila et al., 2011; Minkkinen et al., 2018; Ojanen et al., 2013).

Table 6. Annual estimates for the net ecosystem carbon balance (NECB) and its components of net ecosystem balance (NEE), comprising ecosystem respiration (R_{eco}) and gross primary productivity (GPP), methane (CH_4) flux and later export of dissolved organic carbon (DOC) and inorganic carbon (DIC). Estimates are shown as mean ± standard deviation. Note that only 2021 is shown for the gas fluxes from the mire. Units are given in t-C ha^{-1} $year^{-1}$. (**Paper III**)

	HDPF				**Mire**	
	Open Forest		**Dense Forest**			
	2020	**2021**	**2020**	**2021**	**2020**	**2021**
(a) NEE	−18.4 ± 3.4	−17.2 ± 3.0	−8.3 ± 29	−12.3 ± 30	/	−1.4 ± 0.1
- R_{eco}	81.1 ± 2.9	69.3 ± 2.8	74.1 ± 3.4	75.1 ± 2.9	/	19.5 ± 0.1

- GPP	−99.5 ± 5.8	−86.5 ± 4.6	−82.4 ± 5.9	−87.4 ± 5.2	/	−20.9 ± 0.2
(b) CH_4	0.09 ± 0.01	0.06 ± 0.01	0.06 ± 0.01	0.06 ± 0.01	/	0.68 ± 0.01
(c) DOC	0.78 ± 0.11	0.91 ± 0.12	1.76 0.22	2.02 ± 0.26	1.11 ± 0.14	1.17 ± 0.15
(d) DIC	0.09 ± 0.01	0.10 ± 0.01	0.15 ± 0.02	0.15 ± 0.02	0.32 ± 0.4	0.33 ± 0.04
NECB	−17.5 ± 3.4	−16.1 ± 3.0	−6.3 ± 2.9	−10.1 ± 3.0	/	0.78 ± 0.17

Comparatively, the magnitude and contribution of NEE to NECB were relatively lower in the adjacent mire compared to the forest area. The smaller CO_2 uptake of −1.4 ± 3.9 t-C ha^{-1} year^{-1} reported for 2021 contributes to NECB in the same order of magnitude as CH_4 (0.68 ± 0.01 t-C ha^{-1} year^{-1}) and lateral dissolved C export (1.5 ± 0.15 t-C ha^{-1} year^{-1}). The existence of pools could further limit CO_2 exchange but, in turn, enhances the relative contribution of CH_4 to the C budget (Pelletier et al., 2015).

Overall, while previous findings showed that WTL drawdown might increase soil mineralization and net CO_2 emission rates (Maljanen et al., 2010; Ojanen et al., 2013; van Huissteden et al., 2006), both long-term drainage and initial DC were not found to increase the net emission of CO_2. Instead, they resulted in a reduction of emissions or increase in uptake due to the increase in biomass uptake (HDPF and Pettersson site) and reduction of microbial decomposition in the drier soil conditions (Tobo site). DC effects on the CO_2 and, thus, total GHG balance might vary across different sites dependent on the combined effects of the efficiency of the ditch drainage function before DC, site weather and hydrological conditions.

4.6.2 Contribution of CH_4 to annual C and GHG budget

Although the natural mire (**Paper III**) was a significant contributor of CH_4 emissions in terms of the net C balances, the other sites involved in this thesis were characterised by their small contribution of CH_4 fluxes to the total C balances (Figure 10). It is, however, notable that the Pettersson site was characterised by a large spatial-temporal variability: the annual modelled CH_4 flux from the uncleaned area varied from 0% to 25% for different plots and years but, on average, was <1%. The negligible contribution of CH_4 fluxes to the total annual GHG budget noted in this thesis was in line with previous studies in boreal forest sites (e.g. Ojanen et al., 2013) and recent forest clear-cuts (e.g. Korkiakoski et al., 2019; Vestin et al., 2020).

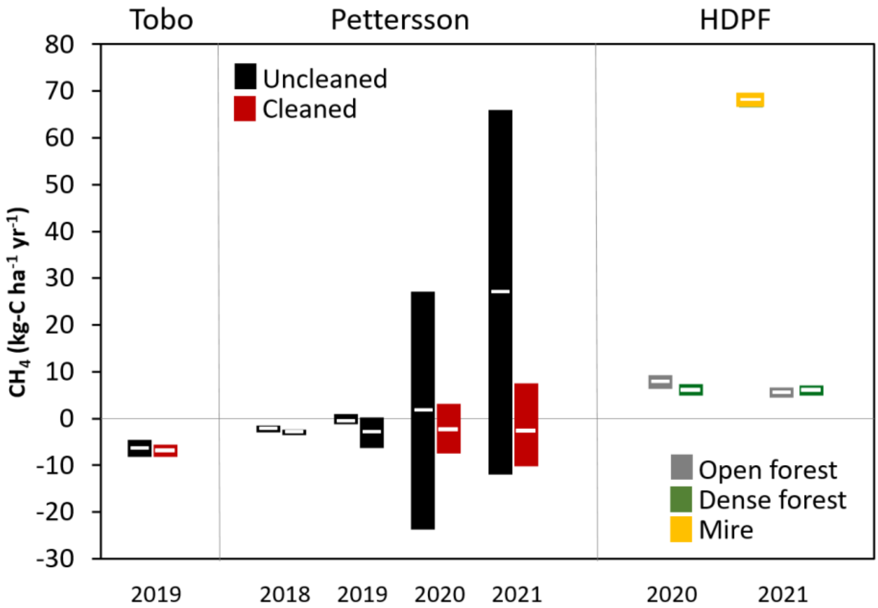

Figure 10. Model estimates for the total annual methane (CH_4) balance of various areas at the Tobo (**Paper II**), Pettersson (**Paper I**) and HDPF sites (**Paper III**). Range of bars denotes the standard deviation from the means shown with white lines. CH_4 estimates for the Tobo and Pettersson sites were based on model interpolation of chamber measurements over the study years. Note that estimates for the Pettersson site were based on field layer measurements, while estimates for the Tobo site were estimated using spatial averaging over field layer and ditches.

As the global warming potential of CH_4 is 86 times higher relative to CO_2 over a 20-year time frame (IPCC, 2013), the median contribution of CH_4 to the annual GHG budget could reach up to 39% in 2021 in the uncleaned area at the Pettersson site. However, the cleaned area remained within <5% after DC, highlighting the role of DC in the suppression of potential CH_4 emissions.

It is also notable that the CH_4 flux from the HDPF and Tobo sites was small in terms of both C and GHG balance even when considering its more powerful warming impact. This indicated that drainage activities, including both DC and historical drainage, had a significant climate-cooling effect through reduction of emissions of the powerful GHG methane.

4.6.3 Contribution of N_2O to annual C and GHG budget

Occasional measurement of N_2O fluxes across the study sites has shown that these fluxes have a negligible contribution to the GHG balance even though they have a 298 times higher warming potential than CO_2. As it is extremely time-consuming and costly to estimate accurately N_2O fluxes, an in-depth evaluation of N_2O fluxes was not a primary goal of this research.

4.6.4 Contribution of lateral C transport to annual C and GHG budget

It is notable that the lateral transport of dissolved C gave the largest contribution to the NECB for the natural mire (42%) relative to CO_2 and CH_4 in absolute terms. In the drained forests, where CO_2 dominated the C dynamics, the dissolved C contributed 5% in the open forest area and 21% in the dense forest in absolute terms (Table 4). This agreed well with the previous studies indicating the importance of aquatic transport of land-derived C in northern landscapes (de Wit et al., 2015; Jonsson et al., 2007).

5. Conclusions

For this thesis, there was an investigation into the impact of drainage on C and GHG balances in three northern forest sites under various geographical settings, soil conditions and drainage regimes. Results indicate that the drainage activities examined in all three studies did not increase C and GHG emissions. CO_2, the dominant gas in C and GHG balances for most conditions, had increased uptake on the historical drained peatland forest relative to the adjacent mire and decreased emissions after initial DC at the dry and fertile clear-cut; insignificant changes were estimated after DC at the relatively wet and infertile clear-cut site. Furthermore, the effect of drainage on mitigating strong emission of CH_4 and potentially increasing CH_4 uptake was observed at all the three studies. Vegetation growth, including both ground vegetation development on clear-cut sites as well as overstory and understory vegetation in peatland forests, was identified as a primary influence on the strength of C uptake and the climate-cooling effect of the site, but the interaction between vegetation and drainage varied between sites with different hydrological and management scenarios.

A specific summary of each study is given below.

Paper I:

- DC had a limited initial effect on the spatio-temporal variations in the net CO_2 exchange and its component fluxes, GPP and R_{eco}, in a recent forest clear-cut in boreal Sweden. Overall, DC had no significant impact on the annual carbon and GHG balance in the initial post-harvest years.
- Despite there being a small proportion of CH_4 in comparison to CO_2, DC reduced the soil water content and thereby mitigated CH_4 emissions during wet post-harvest years.

- The variation in the CO_2 component fluxes was primarily controlled by the spatial variations in ground vegetation growth, likely in correspondence to drainage legacy effects on soil carbon and nitrogen contents. Ditch distance had no consistent effect on CO_2 and CH_4 fluxes.

Paper II:

- DC lowered GHG emissions by 30% from the ditch cleaned area as compared to the uncleaned area. Overall, the ecosystem GHG balance was dominated by the CO_2 exchange.
- The reduction in GHG balance was likely due to a decrease in heterotrophic respiration (by ~34%) in response to enhanced soil water stress at the relatively dry study site.
- In general, soil water dynamics (i.e. water table level and soil moisture) and ground vegetation coverage were identified as the primary influences on the spatial variations of measured CH_4 uptake and CO_2 component fluxes (R_{eco} and GPP), respectively. Specifically, lower soil water content and delayed vegetation growth in the cleaned area corresponded to larger CH_4 uptake and smaller CO_2 component fluxes (i.e. production and respiration), respectively, relative to the uncleaned area.

Paper III:

- Combining vertical atmospheric and lateral aquatic fluxes, a significant net carbon uptake was estimated in the open forest (-16.8 ± 2.3 t-C ha^{-1} year^{-1}) and dense forest area (-8.2 ± 2.1 t-C ha^{-1} year^{-1}), whereas the adjacent mire was close to C neutral (NECB = 0.78 ± 0.17 t-C ha^{-1} year^{-1}). Considering the greater warming impact of CH_4 compared to CO_2, the forest ecosystem had a climate-cooling effect while the mire ecosystem had a strong climate-warming effect.
- The net uptake of CO_2 was the dominating component of the forest NECB, whereas the mire NECB was balanced between CO_2 uptake and emissions of CH_4 and DC exports.
- Forest structure, such as ground vegetation composition and stand volume, had a significant influence on total NECB through different amounts of annual CO_2 uptake and lateral DC export.

- Long-term drainage changed not only WTL but also soil temperature and vegetation cover in terms of NDVI, all of which could have considerable impacts on the NECB components.

6. Future perspectives and implications

The work described in this thesis aimed at capturing the impact of different kinds of drained systems, that is, initial effects of DC and historical effects on forestry drainage, on the C and GHG balances. As they behave differently in terms of their influence on vegetation, hydrology and soil chemistry, more empirical data are needed to fully understand the drainage effects on the forest ecosystem GHG balance over the entire rotation period. For instance, the subsequent tree growth at the Pettersson and Tobo sites (**Papers I** and **II**) might further change the DC impact on the forest C and GHG balance. These impacts may include additional soil water reduction as a result of increased evapotranspiration and photosynthetic CO_2 uptake by trees but also larger canopy shading effects on soil moisture and temperature conditions. In addition, the recently cleaned ditches might deteriorate in their drainage function over time which will feedback to the soil water, C and subsequently GHG dynamics.

Although the work for **Paper III** provides a top-down perspective using eddy covariance measurements of the net ecosystem C balance over the forest and mire areas, further experimental investigations are needed to estimate the C interactions within the ecosystem. These include soil fluxes using chamber measurement, C allocation in trees using biomass measurement and sampling of peat with understory biomass to partition the C storage into various components. A bottom-up approach to the NECB is needed and beneficial from two perspectives. First, a comparison with the existing top-down eddy covariance approach would help validate the consistency of estimates of forest C fluxes. Second, while the work for **Paper III** indicated a net C sink in forests, it might be possible that most C is stored in the form of tree biomass while the soil is losing C through peat decomposition. This indicates that clear harvesting on such forestry-drained peatlands would convert the site into a loss of carbon from the site. Thus, a detailed partitioning would significantly improve our current knowledge of the whole ecosystem exchange of both CO_2 and CH_4 in drained forested peatlands in boreal Sweden.

Over recent decades, many forestry-drained peatlands have been rewetted for biodiversity restoration with the aim of decreasing greenhouse gas emissions. As this thesis indicates that drainage activities did not increase C and GHG emissions, it poses a question as to the efficiency of rewetting from a climate perspective. With the long-term forestry transformation of vegetation composition and soil properties, rewetting of drained forests does not restore their former function, thus a better understanding of the response of GHG fluxes on rewetted sites is required to improve planning and application of peatland drainage, rewetting and subsequent management.

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Popular Science Summary

Since the 19th century, almost three million hectares of wetlands in Sweden have been drained (Paavilainen and Päivänen, 1995). In most cases, ditches were dug to produce cultivable land, and to promote forest growth from the latter half of the 19th century. Over recent years, however, forestry drainage has been the subject of considerable discussion. One main concern is that drainage might cause large emissions of greenhouse gases and intensify global warming. For instance, peatlands cover approximately 16% of Swedish land area (> 30 cm depth of peat) and a further 9% is covered by peat soils (< 30 cm depth of peat) (Hånell, 1988). Peatlands and peat soils are formed when the water level is close to the soil surface. Their waterlogged conditions prevent living material from fully decomposing, thus meaning these peatlands have accumulated large amounts of carbon in the form of peat over thousands of years. Drainage increases the exposure of peat soil to oxygen, and potentially emits a large amount of carbon in the form of carbon dioxide, that is reportedly comparable to the carbon emission from road traffic (Kasimir et al., 2018) in southern Sweden. A large area of northern Sweden exhibits nutrient-poor soil conditions, yet studies in the central boreal region of Sweden are currently limited. Furthermore, ditch cleaning is becoming a common practise after tree harvesting. Such a process removes sediments in deteriorated ditches in order to improve drainage and subsequent tree seedling establishment. However, assessment of the effect of ditch cleaning on the carbon balance is currently lacking. Thus, this work aimed to investigate and quantify the impact of drainage on carbon and greenhouse gas balances in the northern forests at various locations, and under different soil conditions and drainage regimes.

The objectives and methods for each paper are listed as below:

For **Paper I**, the work focused on the impact of drainage DC on the carbon and greenhouse gas balances in a recent forest clear-cut in central boreal Sweden. Emissions or uptakes of carbon dioxide and methane were measured using a closed chamber approach, in which a closed container of known volume and surface area is placed face-down on a patch of soil, and the gas concentration inside the chamber recorded continuously with a gas detector.

For **Paper II**, the initial effects of post-harvest drainage DC on greenhouse gas fluxes in a hemiboreal peatland forest of central Sweden were determined. As with the work for **Paper I**, a closed chamber approach was used.

For **Paper III**, the carbon balances of a nutrient-poor peatland forest and an adjacent natural mire in boreal Sweden were studied, in order to compare and quantify the long-term drainage effects on carbon balances. Fluxes of carbon dioxide and methane were measured using the eddy covariance approach, which provides flux estimates by continuous measurement of high-frequency vertical wind speed and gas concentration at a fixed height above the ecosystem. This work also involved the measurement of carbon loss from surface waters, in order to provide a full net ecosystem carbon balance for the sites.

In addition, environmental conditions, such as temperature, vegetation growth and moisture level, were continuously measured for all three studies, in order to 1) identify the control of gas fluxes and 2) provide flux estimates when measured values were missing.

Results showed that the drainage activities in all three studies did not increase carbon and greenhouse gas emissions. Carbon dioxide was the dominant greenhouse gas in general, and had a significantly higher uptake on the drained peatland forest relative to the adjacent mire (**Paper III)**. At the same time, emissions of carbon dioxide decreased after initial ditch cleaning at the dry and fertile clear-cut (**Paper II)**, but no changes in these emissions were found after ditch cleaning at the relatively wet and infertile clear-cut site (**Paper I)**. In all studies, ground vegetation had a strong influence on the carbon balance. For instance, ditch cleaning was found to suppress vegetation growth under dry conditions (**Paper II**). The higher uptake of carbon dioxide from an open forest with fewer trees, relative to a denser forest, highlighted the importance of understory ground vegetation on the carbon accumulation (**Paper III**). However, ground vegetation development did not only respond to drainage activities, but also to other factors such as soil fertility and initial water conditions. Therefore, the interactions between drainage, vegetation and thus balance of carbon dioxide varied across the study sites.

Methane is more than 86 times as potent as carbon dioxide at trapping heat in the atmosphere over a 20-year time frame (IPCC, 2014). All three studies have indicated that drainage potentially mitigates methane emission and potentially increases methane uptake. Such an impact was particularly clear when comparing the wet and poorly drained sites, such as the natural mire (**Paper III**) and the uncleaned area at the Pettersson site (**Paper I**), where strong and highly variable methane emissions were recorded.

Other carbon and greenhouse gas components, such as nitrous oxide emissions and carbon loss from surface waters, were found to be largely unchanged by drainage activities.

Altogether, the work for this thesis helps to shed light on the effect of forest drainage on carbon and greenhouse gas emissions, and provides insights into supporting the development of appropriate forest management strategies to mitigate climate change.

Populärvetenskaplig sammanfattning

Sedan 1800-talet har ca tre millioner hektar våtmark i Sverige dikats ut (Paavilainen and Päivänen, 1995). Ursprungligen avsåg dikningen i första hand att skapa odlingsmark. Under 1900-talets första hälft och fram till 60-70-talet dikades dock även ansenliga arealer för att öka skogsproduktionen. Dikning av torvmark (våtmark med ett tjockare lager av dött organiskt material) har dock under de senaste decennierna identifierats som en betydande källa för avgivning av växthusgaser till atmosfären. Även i Sverige pågår en intensiv debatt om hur torvmark som är dikad för att öka skogsproduktionen påverkar växthuseffekten. Av Sveriges landyta täcks ca 16% torvmark (>30 cm torv) och ytterligare 9% täcks av torvartade jordar (<30 cm torv) (Hånell, 1988). Torvmarker och torvartade jordar bildas när vattenytan är nära markytan (~0-30(40) cm). Att marken är vattenmättad bidrar till att dött växtmaterial inte bryts ned helt och hållet utan istället bildas det vi kallar torv. I Sverige har stora mängder torv ackumulerats sedan senaste istiden.

Dikning av våtmarker sänker vattennivån och leder till att dött växtmaterial (torv) som tidigare varit vattenmättat nu blir syresatt. Tillgången på syre gör att torven börjar brytas ned vilket bl.a. resulterar i att koldioxid (CO_2), men även lustgas, kan avges till atmosfären. Det finns uppskattningar att dikad torvmark, i framförallt södra Sverige, avger lika mycket växhusgaser som från biltrafiken i Sverige (Kasimir et al., 2018). Förutsättningarna för hur mycket växthusgaser som bildas och avges till atmosfären varierar väldigt mycket beroende på vilken typ av torvmark som dikats. Merparten av all torvmark i Sverige finns i mellersta och norra delen av landet och domineras starkt av näringsfattiga myrar. Samtidigt saknas studier av hur dikning har påverkat upptag och avgivning av växthusgaser nästan helt i dessa regioner.

Avverkning av skog på dikad torvmark leder generellt till att vattenytan stiger. För att bibehålla förutsättningar för skogsproduktion är det därför nästan alltid nödvändigt att restaurera de diken som finns. Förutom att påverka förutsättningarna för skogstillväxt så påverkar dikesrensning även upptag och avgivning av koldioxid och andra växthusgaser. Även kunskapen om hur dikesrensning på verkar upptag och avgivning av växthusgaser i Sverige är ytterst begränsad.

Syftet med den här avhandlingen är att kvantifiera betydelsen av dikning för omsättningen av kol och andra växthusgaser i mellersta och nordliga Sverige under olika klimat, olika markförhållanden och olika dikningsregimer. Avhandlingen bygger på tre olika delstudier.

Syfte och metod för var och en av de tre delstudierna är:

Uppsats I fokuserar på effekten av dikesrensning på omsättningen av kol (C) och växthusgaser efter avverkning i boreala Sverige. Upptag och avgivning av koldioxid och metan mättes med hjälp av speciella mätkammare som placeras med den öppna sidan mot marken. Genom att mäta hur koncentrationen av olika växthusgaser förändras under ett antal minuter inuti mätkammaren kan hastigheter för upptag och avgivning beräknas.

Uppsats II behandlar hur dikesrensning direkt efter avverkning påverkar upptag och avgivning av kol och andra växthusgaser under de första åren efter dikesrensning. Studierna utfördes i de sydligare delarna av den boreala regionen i Sverige. Samma metoder användes som i uppsats I.

Uppsats III behandlar kolbalansen i en dikad beskogad, respektive en odikad, näringsfattig myr i nordliga (boreala) Sverige. Syftet är att bestämma den långsiktiga effekten av dikning och beskogning på upptag och avgivning av kol och växthusgaser på näringsfattig torvmark i norra Sverige. Upptag och avgivning av metan och koldioxid mättes med Eddy Covariance-teknik. Tekniken baseras på att man, med hög hastighet (> 10 ggr/sekund) samtidigt mäter både luftens rörelse i tre riktningar och koncentrationen av, i det här fallet, metan och koldioxid. Mätningarna utförs ett par-tre meter över antingen myrens yta eller över den beskogade dikade myren. Med hjälp av data från mätningarna kan vi sedan beräkna såväl upptag som avgivning av respektive gas. Mätningarna representerar ett medelvärde över en yta från ett fåtal hektar på den öppna myren till ett tiotal hektar för den dikade beskogade torvmarken. Genom att även mäta förlusten av kol och metan via avrinning i bäckar/diken har även den totala balansen för upptag och avgivning av kol kunnat bestämmas.

För samtliga studier gäller att ett större antal miljövariabler också har mätts, bl.a. temperatur och fukt i marken samt växtsammansättning och dess tillväxt. Dessa mätningar har gjorts för att 1) identifiera de viktigaste kontrollerande faktorerna för de olika växthusgaserna och 2) för att kunna konstruera matematiska modeller så att vi kan beräkna upptag och avgivning av respektive växthusgas även för tidsperioder då vi inte mäter eller mätdata saknas.

Sammantaget visade studierna att vare sig dikesrensning eller dikad beskogad torvmark, under de aktuella förhållandena, ledde till ökad avgivning av kol eller växthusgaser. Den totala balansen av växthusgaser i samtliga studier dominerades helt av koldioxid. Upptaget av koldioxid över den dikade beskogade torvmarken var betydligt högre än över den närliggande öppna odikade torvmarken (**Uppsats III)**. Effekten av dikesrensning på upptag och avgivning av koldioxid varierade mellan de olika studieobjekten. Dikesrensning på den mer näringsrika och mindre blöta lokalen ledde till minskad avgivning av koldioxid relativt till om dikena lämnades orensade (**Uppsats II**). Motsvarande studie på den mer näringsfattiga och blötare experimentlokalen visade att dikesrensning inte påverkade utbytet av koldioxid (**Uppsats I**). Samtliga studier visade på förekomst och sammansättning av markvegetation hade en avgörande betydelse för kolbalansen. Under de mer torra för hållandena (**Uppsats II**) så resulterade dikesrensning i en minskad tillväxt av markvegetationen. Studierna av den dikade beskogade torvmarken (**Uppsats III**) visade att de delar av studieområdet som hade relativt sett glesare skog trots detta hade ett högre nettoupptag av koldioxid. Det högre upptaget förklarades av skillnader i sammansättning av markvegetation. Effekten av dikning eller dikesrensning på nettobalansen av upptag och avgivning påverkas alltså mycket tydligt av samspelet mellan ursprungliga vattenförhållanden och markens näringstillgång.

Metan är en växthusgas som vid samma koncentration som koldioxid har betydligt kraftigare påverkan på atmosfärens strålningsbalans än koldioxid. Beräknat för en 20-årig tidsperiod så är metan 86 gånger starkare än koldioxid (IPCC, 2014). Såväl dikning som dikesrensning ledde till minskad avgivning och även till ökat upptag från atmosfären. Avgivningen av metan från såväl den odikade torvmarken (**Uppsats III**) som de icke dikesrensade ytorna på det nordliga näringsfattiga avverkade experimentet (**Uppsats I**) var betydligt högre och mer varierande än på de dikade eller dikesrensade motsvarande ytorna.

Studierna visade vidare att de olika behandlingarna inte hade någon effekt på vare sig växthusgsen lustgas och eller den mängd kol som transporteras bort med avrinningsvatten.

Resultaten från de här studierna kan användas som stöd för beslut om hur dikad beskogad torvmark skall hanteras med avseende på påverkan på atmosfärens strålningsbalans. Dock är antalet studier i mellersta och norra Sverige fortfarande högst begränsat och fler studier är nödvändiga för att erhålla stabila resultat som kan användas som beslutsstöd för berörda myndigheter.

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I

forests

Article

Drainage Ditch Cleaning Has No Impact on the Carbon and Greenhouse Gas Balances in a Recent Forest Clear-Cut in Boreal Sweden

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Abstract: Ditch cleaning (DC) is increasingly applied to facilitate forest regeneration following clear-cutting in Fennoscandinavia. However, its impact on the ecosystem carbon and greenhouse gas (GHG) balances is poorly understood. We conducted chamber measurements to assess the initial DC effects on carbon dioxide (CO_2) and methane (CH_4) fluxes in a recent forest clear-cut on wet mineral soil in boreal Sweden. Measurements were conducted in two adjacent areas over two pre-treatments (2018/19) and two years (2020/21) after conducting DC in one area. We further assessed the spatial variation of fluxes at three distances (4, 20, 40 m) from ditches. We found that DC lowered the water table level by 12 ± 2 cm (mean ± standard error) and topsoil moisture by 0.12 ± 0.01 m^3 m^{-3}. DC had a limited initial effect on the net CO_2 exchange and its component fluxes. CH_4 emissions were low during the dry pre-treatment years but increased particularly in the control area during the wet years of 2020/21. Distance to ditch had no consistent effects on CO_2 and CH_4 fluxes. Model extrapolations suggest that annual carbon emissions decreased over the four years from 6.7 ± 1.4 to 1.6 ± 1.6 t-C ha^{-1} $year^{-1}$, without treatment differences. Annual CH_4 emissions contributed <2.5% to the carbon balance but constituted 39% of the GHG balance in the control area during 2021. Overall, our study suggests that DC modified the internal carbon cycling but without significant impact on the carbon and GHG balances.

Keywords: carbon dioxide; closed chamber; ditch network maintenance; harvest; gas fluxes; methane; forest management

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1. Introduction

The boreal forest biome is an important global reservoir of terrestrial carbon (367–1716 Pg C) [1]. However, its biogeochemical cycling and thus carbon storage is sensitive to forest management practices [2]. Specifically, tree growth is often hampered by waterlogging in humid soils, which is also a typical feature in boreal upland forest areas [3–5]. Therefore, artificial drainage ditching has been performed during the past century in large parts of northern Europe (Finland, Sweden, Estonia and Latvia) to increase tree biomass production [6]. Most of these earlier ditched forests are now reaching the end of the rotation period, in which time, the drainage capacity and efficiency in lowering the water table level (WTL) has deteriorated for many of these old ditches. In addition, decreased evapotranspiration following clear-cutting further supports an increase in the WTL that limits the establishment, survival and growth of subsequent seedling generation [7,8].

As one solution for mitigating increasing soil wetness, ditch cleaning (DC) following forest harvest may help to restore the drainage function of the ditches and thereby regain desired tree growth rates (Paavilainen and Päivänen, 1995) [3]. According to the official statistics from the Finnish National Forest Programme and the Swedish National Forest Inventory (NFI), about 65,000 ha and 10,000 ha of forest lands have been ditch cleaned annually in Finland (during 2001–2010) and Sweden (during 2015–2019), respectively, with a continued increase during recent years. In addition, NFI data suggest that up to two thirds of the DC activities in Sweden have been conducted in upland forests growing on wet mineral soils (i.e., nonpeatland soils with ≤30 cm peat). However, while previous studies have explored DC effects on hydrology [9,10] and tree growth [11–15], the associated impact on the forest carbon and greenhouse gas (GHG) fluxes is poorly understood.

A lower WTL following DC may strongly affect the carbon and GHG balances through various processes related to soil biogeochemistry and vegetation growth, which regulate the uptake and emissions of carbon dioxide (CO_2) and methane (CH_4) [16]. For instance, the drainage of wet soils enhances root aeration and nutrient availability, which facilitates the better growth of ground vegetation and tree seedlings, leading to increased gross primary productivity (GPP) and associated plant carbon uptake [14,15,17–19]. Adversely, drainage might also limit water and nutrient supply for the ground vegetation [20,21]. Previous studies have highlighted the important role of the rapidly developing field-layer vegetation in modifying the water, energy, carbon and GHG balances specifically by increasing transpiration rates [22–24]. Additionally, the WTL drawdown may alter soil CO_2 emissions through greater soil aeration, which accelerates the microbial decomposition of soil organic matter [25].

Apart from its effects on the CO_2 exchange, changes in WTL due to DC may also affect CH_4 production and consumption rates in the forest soil. Specifically, water-saturated conditions after clear-cutting result in an extended anaerobic zone which supports the production of CH_4, possibly turning a wet forest site into a source of CH_4 [26]. Lowering the WTL through DC increases the depth of the surface oxic soil layer and thereby enhances the potential for aerobic CH_4 oxidation. At the same time, the CH_4 production may be suppressed as the anoxic soil layer is forced deeper into the soil [27–29]. In addition, vegetation responses to DC might alter the substrate supply to methanogens [30], which may further amplify or counterbalance the hydrological consequences of DC for CH_4 fluxes.

Thus, the hydrological changes due to DC potentially cause multiple effects and complex interactions with biogeochemical processes that regulate the production and consumption rates of CO_2 and CH_4. This requires an in-depth understanding of how DC affects the forest carbon and GHG balance, both in the short- and in the long-term. To date, however, studies investigating the effects of DC on C and GHG fluxes are lacking, particularly in the nutrient-poor boreal forests growing on mineral soil, which account for the majority of forested area in Fennoscandia [31–33].

The aim of this study was to investigate the impact of DC on CO_2 and CH_4 fluxes in a recent forest clear-cut in boreal Sweden. The specific objectives were to:

(1) Quantify the magnitudes of CO_2 and CH_4 fluxes from seasonal to inter-annual scales;
(2) Investigate the effects of DC on the spatio-temporal variations in CO_2 and CH_4 fluxes;
(3) Identify environmental factors that drive the changes in CO_2 and CH_4 fluxes in response to DC;
(4) Estimate the effect of DC on the annual C and GHG balances.

2. Materials and Methods

2.1. Site Description and Experimental Design

This study was conducted at a forest clear-cut site ("the Pettersson site" 64°12′56″ N, 20°49′32 E″, 60 m.a.s.l.) located approximately 3 km north of Robertsfors town in the county of Västerbotten, Sweden. The local climate according to the Köppen–Geiger classification [34] is characterized as boreal (Dfc; also called subarctic) with persistent snow cover during ~6 months. According to the data from the nearest weather stations (Brände and Bjuröklubb, 15 km and 46 km from the site, respectively) of the Swedish Meteorological and Hydrological Institute (SMHI; www.smhi.se, accessed on 1 February 2022), the 30-year (1991–2020) mean annual temperature (*Rec*orded in Bjuröklubb) is 4.0 °C, and the mean annual precipitation (*Rec*orded in Brände) is 701 mm, of which about 35% falls as snow.

While detailed *Reco*rds for the timing of the original drainage and plantation of the former stand are missing, historical air photos suggest that trees and the ditch network of the study site already existed during the early 1960s. Prior to harvest in October 2016, the former stand had a tree volume of 274 m^3 ha^{-1} and consisted of about 75% pine (*Pinus sylvestris* L.), 22% Norway spruce (*Picea abies* L.) and 3% broad-leaved tree species (e.g., *Betula* spp). Soil preparation by mounding was conducted across the entire site in August 2018, followed by the establishment of tree seedlings in June 2019, which consisted of 30% Norway spruce and 70% pine, planted with density of 2100 plants per hectare.

The drainage ditch network is composed of two main ditches that diverge water into different directions, i.e., south-east and south-west (Figure 1). In January 2020, DC was performed along the ditch network branch draining towards the south-east (its surrounding area is hereafter referred to as the DC area), using a tracked excavator. During DC, both vegetation and deposited material were removed from the ditches and piled up along both sides of the ditches. After DC, the open ditches had a trapezoidal shape, with a depth of about 1 m and a width of about 0.5 at the bottom and towards 2 m at the top. The ditch network branch draining to the south-western side was left uncleaned (hereafter referred to as the control area) and characterised by stagnant surface waters due to sediment deposition and vegetation in-growth, which reduced its drainage functions.

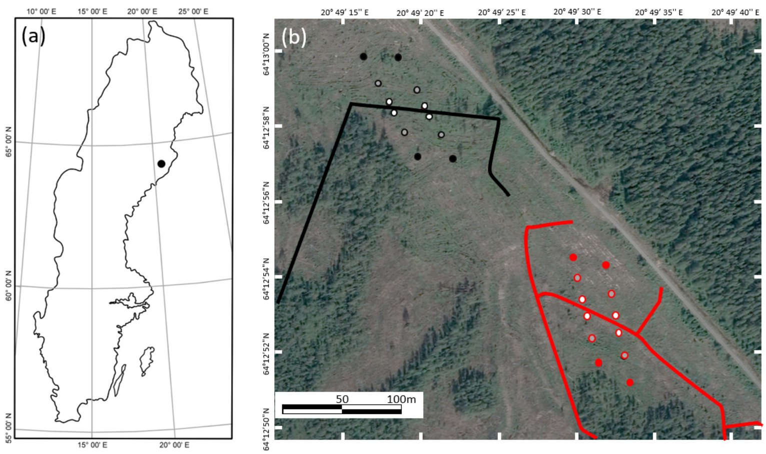

Figure 1. (**a**) Location of the Pettersson study site in boreal Sweden, and (**b**) map of ditch network at the Pettersson field site. Cleaned and uncleaned ditches are marked in red and black, respectively. The locations for flux measurements at 4 m, 20 m and 40 m distance from the ditch, respectively, are marked in red for the cleaned area and black for uncleaned area.

The site is located in a region where Podzol is the dominant soil type (Geological Survey of Sweden (SGU)), with soil layers consisting of thin incoherent surface layers of postglacial sand and gravel, while finer soil particles of clay and silt are dominant at 0.5 m below the ground surface. The thin organic layer at our study site was partly mixed into the upper mineral soil layer by the harvest machinery, which resulted in elevated carbon and nitrogen concentrations of 25.1 $\pm$ 5.3 % and 0.84 $\pm$ 0.19 % (i.e., C:N ratio: 31 $\pm$ 1.5) in the 0 to 10 cm profile, respectively. In comparison, carbon and nitrogen content decreased to 10.6 $\pm$ 4.5 % and 0.40 $\pm$ 0.17 % (i.e., C:N ratio: 31 $\pm$ 1.1), respectively, within 10 and 20 cm depth. There was no significant difference in carbon and nitrogen concentrations between the control and DC areas based on the soil samples taken from 24 measurement plots in August 2018. The main herbaceous plant species that were established across the site during the study years following harvest included *Deschampsia flexuosa* L., *Trichopho-*

rum alpinum L., *Epilobium angustifolium* L., *Potentilla erecta* L., *Vaccinium myrtillus* L. and *Vaccinium vitis-idaea* L.

The experimental design included two parallel transects (about 30 m apart) in each of the two treatment areas with sampling plots at three distances (4, 20, and 40 m) on both sides of the ditch along each transect (i.e., 2 treatment areas × 2 transects × 3 distances × 2 sides = 24 plots) (Figure 1). The 40 m plots were also at similar distance to the surrounding intersecting ditches (Figure 1). Pre-treatment data were collected for two years (2018–2019) followed by two years (2020 and 2021) of measurements after the ditch was cleaned in the designated DC treatment area. This experimental setup allowed us to control for potential confounding effects from both spatial (e.g., topography) and temporal (e.g., weather patterns) factors.

2.2. Greenhouse Gas Flux Measurements

We conducted CO_2 and CH_4 flux measurements using the closed dynamic chamber method [35]. The measurements were carried out during daytime in approximately bi-weekly intervals during the snow-free period (May to October) from 2018 to 2021. In May 2018, square aluminium frames (48.5 × 48.5 cm) with a frame base down to 5 cm below the soil surface were permanently installed at each plot. The observed vegetation species and coverage within the frames were consistent with the surrounding area indicated in Section 2.1 over the study years.

Forest-floor CO_2 and CH_4 fluxes were determined by placing a chamber connected in a closed loop to a portable GHG analyser (Gas Scouter G4301, Picarro Inc., Santa Clara, CA, USA) onto the pre-installed frames. The Gas Scouter has a built-in sampling pump circulating air between chamber and analyser at a flow rate of 0.001 m^3 min^{-1}, and *Reco*rds CO_2 and CH_4 concentrations at a frequency of approximately ~1 Hz, with precision levels of ±150 ppb and ±0.8 ppb, respectively. In the first two years (2018/19), we used a transparent chamber with the dimensions of 48.5 × 48.5 × 30 cm. During the subsequent two years (2020/21), a taller transparent chamber (48.5 × 48.5 × 50 cm) was required to cover the growing herbaceous vegetation within the frames. While effective mixing of the chamber headspace was created by the continuous recirculation pump from the analyser in the smaller chamber used in 2018/19, the larger chamber was equipped with a small fan to further support mixing in the larger and more densely vegetated headspace. A comparison of repeated (i.e., within <5 min) fluxes measured with the two different chamber sizes suggested good agreement for CO_2 (r^2 = 0.95) and CH_4 (r^2 = 0.58) fluxes, respectively. The lower r^2 for the CH_4 flux was likely due to the disturbance from the initial measurement, which may have modified the comparatively small CH_4 concentration gradient before the subsequent measurements. The mixing ratios of CO_2 and CH_4 are reported as dry air mole fractions.

During each flux measurement, a transparent chamber was used at first to measure the net ecosystem exchanges for CO_2 (i.e., NEE) and CH_4 during 90–120 s under natural light conditions. After 2 min of venting, the chamber was placed onto the same frame again and covered by an opaque blanket to estimate the CO_2 flux during dark conditions, i.e., ecosystem respiration (*Reco*). The gross primary productivity (GPP) was then calculated using the difference between NEE and $R_{eco.}$

The average rate of the change in the mixing ratios over time (dC/dt; ppm s^{-1}) was estimated using simple linear regression over a chosen data range. The flux rate was then calculated based on dC/dt as a function of chamber headspace volume, air temperature and pressure according to the ideal gas law. Poor-quality flux data were eliminated under the criteria of the root-mean-square error (RMSE) and r^2 of the chosen dC/dt. Based on visual examination of the data, CO_2 fluxes with RMSE > 2.5 ppm and r^2 < 0.90, and CH_4 fluxes with RMSE > 2.5 ppb and r^2 < 0.90 were removed. These quality control procedures led to the removal of about 2% and 5% of all CO_2 and CH_4 fluxes measured, respectively. The sign convention in this study is such that positive and negative flux values refer to a net source and sink of the GHG, respectively.

We also conducted measurements of N_2O fluxes during early and late August 2020 with static chambers following the methodology described in Appendix A. The results suggested negligible N_2O fluxes of 14 ± 8 µg-N m^{-2} h^{-1} and 18 ± 5 µg-N m^{-2} h^{-1} in the control and DC area, respectively, which accounted for <1.1% of the GHG balance even when accounting for the 298 higher warming potential of N_2O relative to CO_2. We therefore did not include these measurements into the ordinary measurement program.

2.3. Measurements of Abiotic Factors

To investigate the impact of DC on the environmental factors and to explore their relationships with the measured GHG fluxes, we *Reco*rded a suite of environmental variables in parallel during each flux sampling campaign. Specifically, manual WTL measurements were made inside PVC groundwater tubes (Ø = 32 mm external and 26 mm inside, h = 125 cm, 3 mm holes every 2.5 cm) inserted to a depth of 1 m adjacent to each flux measurement frame. Soil moisture within the upper 5 cm was measured at three sides around the frame using a GS3 combined moisture–temperature sensor connected to a handheld data logger (ProCheck, Decagon Devices, Pullman, WA, USA). Soil temperature (T_s) was also *Reco*rded manually next to each frame at 5 and 10 cm depths (T_{s5}, T_{s10}). Air temperature (T_a) was measured using two Hobo® pendant temperature loggers (Onset Computers, Bourne, MA, USA). One was placed at 2 m height above surface and shaded from direct sunlight, *Reco*rding data at 15 min intervals continuously over the year, whereas the other one was operated in the chamber headspace at 5 s intervals during chamber closures. Photosynthetically active radiation (PAR) was continuously measured using two Hobo® pendant radiation sensors with built-in loggers (Onset Computers, Bourne, MA, USA). Specifically, one sensor *Reco*rded PAR at 5 s intervals inside the chamber during the period of a flux measurement, while another sensor was situated at the centre of the study site to log ambient PAR all year round at an hourly interval. Soil temperature at 5 cm and 10 cm depths was also continuously measured with automated TOMST® TMS-4 sensors (TOMST, Prague, Czech Republic) installed at 6 selected plots, with each located 4 m, 20 m and 40 m from the ditches in both the control and DC areas. The hourly ambient T_a and PAR data served as input for the nonlinear regression models to predict hourly CO_2 fluxes (see Section 2.4).

2.4. Vegetation Characteristics

The growth of the field-layer vegetation within each flux measurement frame was monitored during the measurement years by taking overhead images during each flux measurement campaign. These images were then analysed to derive a vegetation greenness index defined by the green chromatic coordinate (g_{cc}) [36–38] (Equation (1)).

$$g_{cc} = G/(R + G + B) \tag{1}$$

g_{cc} refers to the greenness index from the image taken on the frame; *R*, *G* and *B* denote the intensity (0–255) of the red, green and blue image channels. The g_{cc} values were averaged for each image pixel located within the chamber frame. Based on the overhead images, we also calculated the ground vegetation areal coverage, which is defined as the vertical projection of vegetation onto a unit of land.

In July 2019 and July 2021, the ground vegetation g_{cc} and areal coverage for the area surrounding the frames were studied by taking overhead images at 12 spots evenly distributed within 15 m radii around each flux measurement frame. The purposes of these measurements were to (1) evaluate if the vegetation cover inside the flux measurement frames was representative of that within the surrounding area and (2) to increase the sample size to test for DC impacts on ground vegetation development.

Continuous measurements of vegetation growth were also made using two phenocams (TimeLapseCam, Wingscapes, Calera, AL, USA) which *Reco*rded images at hourly intervals with a nadir angle of 15° towards a northernly direction. The g_{cc} and areal coverage calculated from these images were used to calibrate and interpolate the manual overhead images to frame-specific continuous phenology time series. The latter were then used as model input for estimating annual CO_2 flux budgets (see Section 2.4).

2.5. Statistical Analysis

We first applied mixed effect models with repeated measures to test for the significant effects of the DC and distance to ditch treatments on the spatial variation of environmental (i.e., T_a, T_{s10}, T_{s5}, SM, WTL, g_{cc} and vegetation areal coverage) and flux (i.e., GPP, *Reco*, NEE and CH_4) variables (Figure 2). The statistical models applied were as follows (Equation (2)):

$$y_{ijk} = \beta_0 + T_j + \beta_1 d + S_{ij} + \varepsilon_{ij} \quad (2)$$

where y_{ij} denotes the environmental or gas flux variable for sampling occasion i with DC treatment j (j = control or DC); β_0 denotes the overall mean of the environmental or gas flux variable; T_j denotes the fixed effect of DC treatment j; β_1 denotes the fixed effect of the sensitivity to distance to ditch d; S_{ij} denotes the random effect of sampling occasion i; ε_{ij} denotes the random error for sampling occasion i with treatment j. The model takes into account the random effects presenting a covariance structure where correlations decrease with time [39]. Mixed effect models were proven to be robust to various data distributions [40]. We set the statistical error levels to $\alpha = 0.05$ for the mixed effect models, and the standard error (±SE) of the sample means was used to indicate the level of uncertainty.

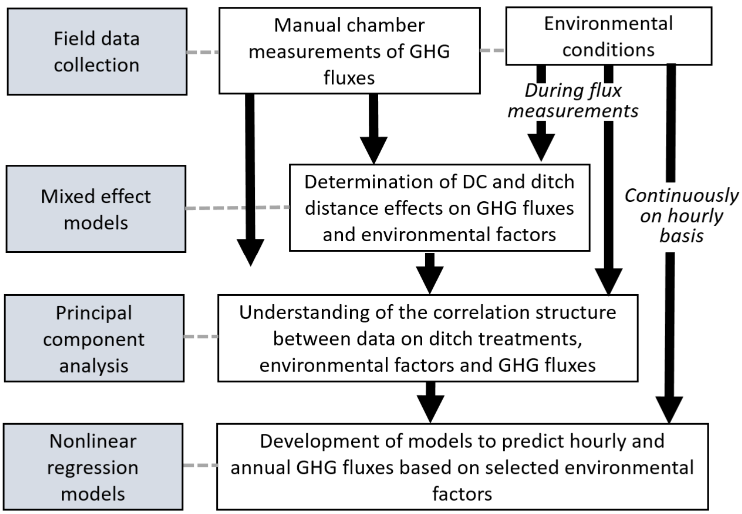

Figure 2. Flowchart of study methodology.

Next, since the mixed effect models did not provide information on the associations between environmental and flux variables, we complemented our statistical analysis by conducting a principal component analysis (PCA) with the goal to identify the three-way interactions between the treatments (DC and distance), environmental factors and flux variables (CO_2 and CH_4) (Figure 2). Principal components (PCs) were calculated using the growing season means of the variables from each measurement plot. Separate PCAs were carried out for the datasets from pre-DC and post-DC treatment periods. Variables were normalised by subtracting each value from the mean and dividing by the standard deviation before inputting them to the PCA models [41]. Significant PCs were selected and presented using the Kaiser criterion [42]. The correlation coefficients (loadings) of the variables with the significant PCs were compared to identify the relationships among variables [43]. All statistical analyses were conducted using the Mathworks Matlab software R2021a.

2.6. Modelling of Annual CO_2 and CH_4 Flux Budgets

Nonlinear regression models following [36,44–46] were used to predict hourly and annual R_{eco} and GPP fluxes based on data of T_a, PAR and g_{cc}. Particularly, GPP from each frame was fitted to hourly PAR and frame-specific g_{cc} data using a hyperbolic PAR function adjusted by normalised frame-specific g_{cc} representing seasonal variations in vegetation biomass (Equation (3)):

$$GPP_{(hr,frame)} = (\alpha \times P_{max} \times PAR \times g_{cc_{\text{norm}}}) / (\alpha \times PAR + P_{max} \times g_{cc_{\text{norm}}}) \tag{3}$$

where GPP denotes the hourly gross primary production (mg m^{-2} h^{-1} of CO_2-C); α denotes the initial slope of quantum use efficiency of photosynthesis (mg μmol photons^{-1} of CO_2-C); PAR denotes the mean of hourly photosynthetically active radiation (μmol m^{-2} s^{-1}); P_{max} denotes the maximum photosynthesis under infinite PAR (mg m^{-2} h^{-1} of CO_2-C); and $g_{cc_{\text{norm}}}$ is the daily frame-specific chromatic greenness index (g_{cc}) surrounding the frames normalised to the scale of 0 to 1.

Hourly estimates of R_{eco} were modelled using an exponential function of soil temperature based on [47] adjusted by the normalised frame-specific g_{cc} (Equation (4)):

$$R_{eco_{(hr,frame)}} = R_0 \times exp^{b \times T_{s5}} + (\beta \times g_{cc_{\text{norm}}}) \times exp^{b \times T_{s5}} \tag{4}$$

where R_{eco} denotes hourly ecosystem respiration (mg m^{-2} h^{-1} of CO_2-C); R_0 denotes soil respiration at 0 °C (mg m^{-2} h^{-1} of CO_2-C); T_{s5} denotes soil temperature at 5 cm depth (°C); b denotes the sensitivity of R_{eco} to T_{s5}; and β is a scaling parameter associated with the contribution of autotrophic respiration $(R)_a$ to $R_{eco.}$ Hourly estimates of GPP and R_{eco} were then summed up for the entire year, and annual NEE was estimated by the difference of the modelled annual GPP and $R_{eco.}$

To increase the robustness of the models, we pooled the available data to develop models based on the two pre-DC (2018/19) and two post-DC (2020/21) years separated for the control and DC areas, respectively. The model parameters of the final models selected based on the highest R^2 are summarised in Table 1. It is noteworthy that despite similar greenness and areal cover (Table S1), the P_{max} in the GPP model and the β in the R_{eco} model were higher in the DC plots during the post-DC period compared to the control plots. This is explained by the greater vascular biomass noted in the DC plots (based on visual observation) at which the same greenness and areal coverage facilitate higher photosynthetic rate compared to moss-dominated ground vegetation. During the snow period, R_{eco} was represented by R_o, which is an iterated parameter in the respiration model that describes the respiration rate when soil temperature was at 0 °C. The modelled GPP values were zero in response to the negligible exposure of ground vegetation during snow coverage (i.e., $g_{cc_{\text{norm}}}$ = 0; Equation (3)).

The uncertainty of the annual CO_2 flux budgets derived by the model extrapolations was evaluated using Monte Carlo simulations. For that purpose, we assigned a normal distribution to each model input parameter (see Table 1) in accordance with its mean and standard deviation derived during model development [48]. Then, a large number (1000) of random draws were taken from the normal distributions of each model parameter. The standard deviation for the set of 1000 predicted estimates was then used to define the uncertainty of the annual CO_2 flux budgets.

Table 1. Model parameters for estimating gross primary production (GPP) (Equation (3)) and ecosystem respiration (R_{eco}) (Equation (4)) for control and ditch cleaning (DC) clear-cut area, applied separately in the pre-DC years (2018/19) and post-DC years (2020/21); α is the initial slope of the light-use photosynthetic efficiency (mg µmol^{-1} photons of CO_2-C); P_{max} is the maximum photosynthetic rate at light saturation (mg m^{-2} h^{-1} of CO_2-C); R_0 is respiration rate (mg m^{-2} h^{-1} of CO_2-C) at 0 °C; b is the sensitivity of R_{eco} to soil temperature (T_s); β represents the contribution of autotrophic respiration to R_{eco}; numbers in parentheses indicate standard error; R^2 denotes the coefficient of determination of the model. Number of observations ranges between 186 and 272 for the development of each model.

Time	Pre-DC (2018/19)		Post-DC (2020/21)	
Area	**Control Area**	**DC Area**	**Control Area**	**DC Area**
GPP Model				
α	−1.3 (0.37)	−1.1 (1.06)	−4.3 (0.72)	−6.9 (2.1)
P_{max}	−1069 (141)	−1250 (215)	−1589 (74)	−1774 (110)
Adjusted R^2	0.47	0.35	0.66	0.58
R_{eco} model				
R_0	34.9 (5.8)	28.2 (4.2)	47.4 (4.9)	57.5 (6.3)
b	0.09 (0.01)	0.12 (0.01)	0.09 (0.01)	0.05 (0.01)
β	72.9 (21.3)	23.9 (7.7)	86.1 (12.8)	288 (35)
Adjusted R^2	0.59	0.65	0.65	0.69

Due to the weak relationship between CH_4 fluxes and environmental variables, annual CH_4 balances were interpolated from the median of measured CH_4 fluxes. We propagated the standard errors to estimate the uncertainty of the annual CH_4 budgets due to spatial variability among sampling plots. A baseline winter CH_4 emission rate was estimated based on the assumption that the low CH_4 fluxes *Reco*rded from the late October measurement campaigns (see Section 3.2) continued during the following winter period (i.e., November–April).

3. Results

3.1. Environmental Data

The annual mean air temperature at the nearby SMHI weather station in Bjuröklubb (46 km from the study site) was close to the 30-year average (4.0 °C in 1991–2020) in 2018 (4.4 °C), 2019 (3.9 °C) and 2021 (4.1 °C), but considerably warmer in 2020 (5.9 °C) due to an unusually warm winter period (−0.4 °C and −2.1 °C in January and February, with reference to 30-year average of −5.1 °C and −6 °C). The annual total precipitation in 2018 and 2019 at the nearby SMHI weather station in Brände (15 km from the study site) was 505 and 652 mm, which was lower than the 30-year long-term average (1991–2020) of 701 mm. In the post-DC years of 2020 and 2021, the annual total precipitation was 958 and 1007 mm, substantially higher than the long-term average.

T_a *Reco*rded during the flux measurement campaigns ranged from a minimum of 5.9 ± 0.4 °C during the beginning or the end of the growing season to a maximum of 29.1 ± 1.3 °C in June or July during the four years (Figure 3a). T_{s5} and T_{s10} measured during flux measurements ranged from 4.4 ± 0.3 °C to 19.3 ± 0.9 °C and from 4.7 ± 0.4 °C to 16.2 ± 0.7 °C, respectively, for the four years (Figure 3a). The difference in T_{s5} and T_{s10} between the control and DC areas was <0.4 °C during the study years. T_{s5} and T_{s10} at 4 m distance from the ditch were 0.2 to 0.9 °C lower than at 40 m from the uncleaned ditches in the control area (Table S1). The magnitudes and temporal dynamics of PAR were similar during the study years (Figure 3b).

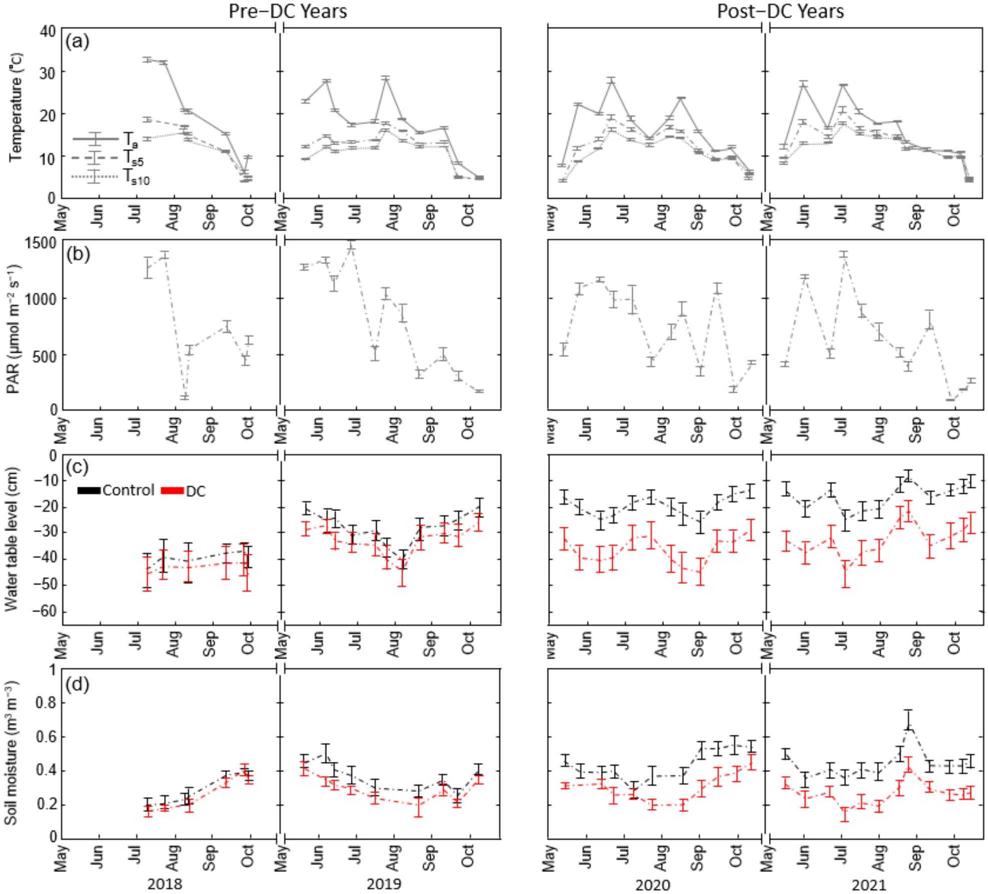

Figure 3. Environmental variables including (**a**) air (T_a) and soil temperature (T_{s5} at 5 cm and T_{s10} at 10 cm depth), (**b**) photosynthetically active radiation (PAR), (**c**) water table level (WTL), (**d**) soil moisture (SM) monitored at the measurement plots (shown as means of 4 m, 20 m and 40 m from ditch) during flux measurements. PAR, WTL and SM data are averaged for each sampling occasion and grouped by experimental area (control versus DC) and year (pre-DC years of 2018 and 2019 with post-DC year of 2020 and 2021). The variable means of each sampling occasion ± standard error (SE) are presented for both control and DC areas (n = 12), respectively.

There was no clear seasonal pattern on the WTL fluctuation, but a shallower WTL was usually observed at the beginning or the end of the growing season. In 2018, the mean WTL averaged over all flux plots and sampling campaigns was −39 ± 2 cm and −43 ± 3 cm in the control and DC areas, respectively (Figure 3c). Thus, the mean WTL was already 5 cm higher in the control area than in the DC area during the pre-DC period. In 2019, the mean WTL rose to −27 ± 1 cm in the control area and −32 ± 1 cm in the DC area, suggesting a significant ($p < 0.05$) but similar increase in both areas during the second pre-DC treatment year (Table S1). After DC, the mean WTL in the control area rose to −19 ± 1 cm in 2020 and −15 ± 1 cm in 2021, whereas the WTL in the DC area remained at a similar level of −37 ± 2 cm in 2020 and −32 ± 1 cm in 2021, relative to the pre-DC year of 2019. Thus, following DC, the WTL in the DC area was 18 ± 2 cm and 17 ± 2 cm lower than in the control area in 2020 and 2021, respectively. This implies that the WTL difference between the two areas increased by 12 ± 2 cm due to the DC effect. In the control area, significantly shallower WTL was observed in the plots closer to the ditch during all four study years. In the DC area, however, plots closer to the ditch were already characterised by deeper WTL in the pre-DC years, and this spatial difference remained significant in the post-DC years (Table S1).

Similar spatial and temporal patterns were observed for the soil moisture content. Specifically, in the first two pre-treatment years, the annual mean soil moisture remained in the range of 0.29 to 0.36 $m^3\ m^{-3}$ in the two treatment areas (Figure 3d; Table S1). The difference between the control and DC area increased substantially in the two post-DC years, where in the control area, the soil moisture increased to 0.44 ± 0.02 and 0.43 ± 0.01 $m^3\ m^{-3}$ in the year 2020 and 2021. In the DC area, the soil moisture remained at 0.30 ± 0.01 and 0.26 ± 0.01 $m^3\ m^{-3}$ in the year 2020 and 2021, respectively, and was not statistically different from the pre-DC period. Altogether, the DC effect on soil moisture content was estimated at 0.12 $m^3\ m^{-3}$ based on the additional differences noted between the control and DC areas after DC. The effect of distance to ditches on the soil moisture was similar to its effect on the WTL, with soil moisture decreasing with distance to ditches in the control area but increasing with distance to ditches in the DC area during all measurement years.

In July 2019 and 2021, the seasonal maxima of the greenness index and vegetation areal coverage were not significantly different between the two treatment areas (Figure 4). Averaged over all frames and treatments, the greenness index increased from 0.24 ± 0.02 in July 2019 to 0.53 ± 0.04 in July 2021, while the areal coverage increased from 0.21 ± 0.04 in July 2019 to 0.54 ± 0.05 in July 2021. In the surrounding area, the maximum greenness index and areal coverage over all treatment plots increased from 0.22 ± 0.01 to 0.46 ± 0.02 and from 0.20 ± 0.02 to 0.57 ± 0.02, respectively, over the two years. The greenness index and vegetation areal coverage in the surrounding area at the 4 m distance from the ditch locations were, however, 15% and 30% greater in the control area than in the DC area in July of 2019 and 2021, whereas no difference between both areas was noted at the 20 m and 40 m plots (Table S1). Furthermore, the greenness index and vegetation areal coverage of frames and their surrounding areas agreed well overall, except the greenness index in the frame was 16% greater (p = 0.047) than in the surrounding areas in the DC area in July 2021. In the model extrapolation for annual gas balances, this difference was accounted for by developing treatment-specific models (see Section 2.4).

3.2. Temporal Variations of CO_2 and CH_4 Fluxes

The magnitudes and seasonal variations of daytime net CO_2 exchange (NEE) and its components (GPP and R_{eco}) increased throughout the four measurement years in both the control and DC areas. When averaged over all plots, the instantaneous daytime NEE switched from net emissions of 72 ± 9 mg C $m^{-2}\ h^{-1}$ in 2018 to a net uptake of 168 ± 27 mg C $m^{-2}\ h^{-1}$ in 2021 (Figure 5a; Table 2). From 2018 to 2021, the peak growing season GPP increased about 13 times in the DC area, relative to a 10-time increase in the control area (Table 2). Similarly, daytime measurements of R_{eco} increased by more than three times in the DC area from 2018 to 2021, whereas in the control area, the increase was by about two times.

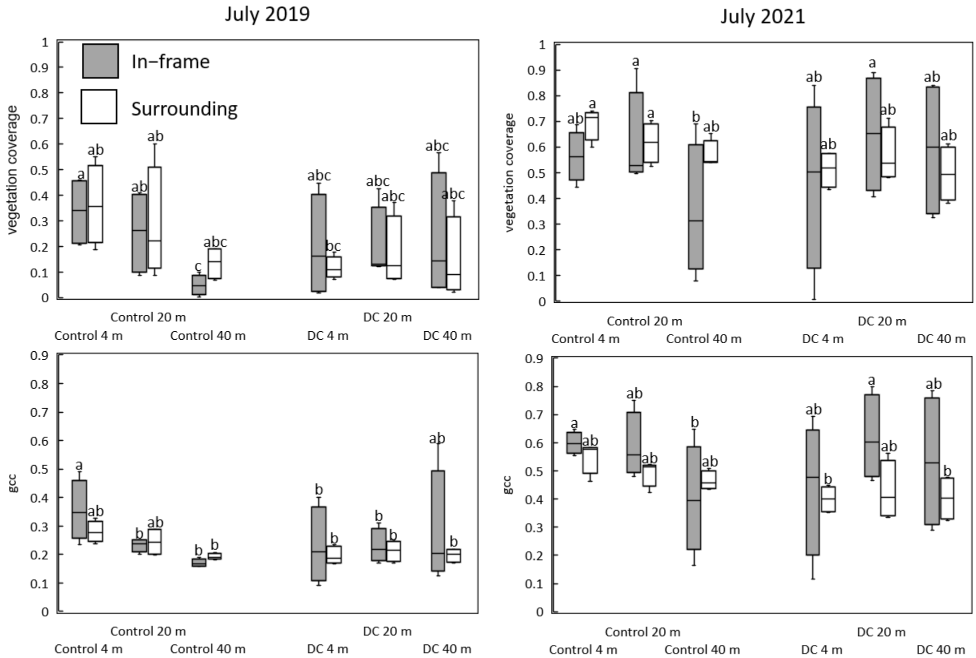

Figure 4. Ground vegetation greenness index inside the frames and in their surrounding areas within 15 m of each flux measurement plot in July 2019 and in July 2021, grouped by ditch cleaning treatment (control versus DC) and distance to ditches (4 m, 20 m and 40 m). The boxes denote the interquartile range with the median indicated inside each box, whereas the lines extending from the boxes indicate the range of each category. Different letters above boxes (a, b, c) refer to compact letter display of one-way ANOVA that indicate significant differences (α = 0.05) among groups, followed by the least significant difference test.

In the two pre-treatment years, CH_4 fluxes varied within the 10–90 percentile range of −0.06 to +0.07 mg C m^{-2} h^{-1} across all plots, with a median of −0.02 mg C m^{-2} s^{-1}, indicating the majority of fluxes were negative (i.e., uptake) at the site (Figure 6). Occasional emission spikes were observed at individual plots across both the experimental areas throughout the four years (Figure 6). In the two years after DC, the number and magnitude of CH_4 emission spikes increased, particularly during the peak season in both the control and DC areas, but with considerably higher ranges and frequencies observed in the control area. Specifically, the 10–90 percentile ranged from −0.07 to 3.6 mg C m^{-2} s^{-1} in the control area and from −0.14 to 0.11 mg C m^{-2} s^{-1} in the DC area during the two years after DC. The median CH_4 flux was 0.08 mg C m^{-2} s^{-1} in the control area and −0.05 mg C m^{-2} s^{-1} in the DC area (Table 2). It is further noteworthy that most emission spikes were Recorded at the plots 4 m from the uncleaned ditch in the control area, with median emissions of 1.2 mg C m^{-2} s^{-1} (Table 2).

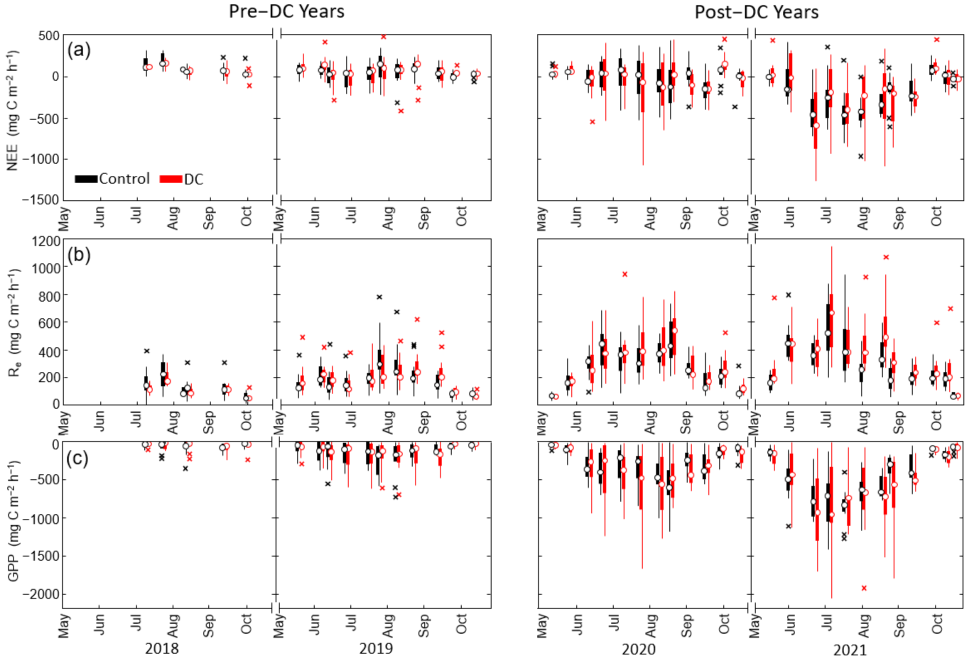

Figure 5. Seasonal variations of carbon dioxide (CO_2) flux components including (**a**) net ecosystem exchange (NEE), (**b**) ecosystem respiration (*Reco*) and (**c**) gross primary productivity (GPP) in control and ditch-cleaned (DC) clear-cut areas during pre-DC (2018/19) and post-DC periods (2020/21). Positive and negative values represent losses and uptakes by the ecosystem, respectively. The circles denote sample (n = 12) medians, whilst the bottom and top edges of the thick bar indicate the 25th and 75th percentiles. The vertical line denotes the range excluding outliers (crosses), which are defined as the values more than 1.5 interquartile range away from the top or bottom of the line. Data shown represent the means of measurement plots at all distances from ditches.

3.3. DC Effect on CO_2 and CH_4 Fluxes

During the pre-treatment period, the mixed effect models suggested that the growing season means of daytime NEE exhibited no significant differences ($p > 0.05$) between the control and DC areas. The growing season means of the measured GPP and R_{eco} fluxes also did not differ significantly ($p > 0.05$) between the two treatment areas during the two pre-treatment years. After DC, both measured R_{eco} and GPP became significantly ($p < 0.01$) greater as compared to the control area. Due to similar changes in R_{eco} and GPP, however, NEE was not significantly affected by DC ($p > 0.05$).

The mixed effect models indicate no significant difference ($p > 0.05$) in CH_4 fluxes between the control and DC areas before the treatment (Table 2). In the two years after DC, emissions from the control area became significantly higher ($p < 0.01$) than from the DC area, consistent with the increased emission spikes in the control area.

Table 2. Annual central estimates ± standard error (SE) and mixed effect model results for ditch treatment effects (control versus DC) and distance to ditch combinations (4 m, 20 m and 40 m) on CO_2 and CH_4 fluxes based on observations during pre-DC (2018/19) and post-DC periods (2020/21). Mean and median values were applied for the central estimates of CO_2 and CH_4 flux values, respectively. Fixed factors of mixed effect models include ditch cleaning treatment (T), distance to ditches (D) and their interaction (TD). Column N refers to the sample size of each model. Significant p values (α = 0.05) are represented in bold face.

Gas Species	Period	N	Central Estimates ± SE								p Values from the Mixed Effect Models	
			Control Area			All Control Plots	DC Area			All DC Plots		
			4 m	20 m	40 m		4 m	20 m	40 m		T	D
CO_2 (mg C m^{-2} h^{-1})												
NEE	Pre DC	395	−10 ± 13	87 ± 12	96 ± 9	58 ± 7	66 ± 12	75 ± 13	30 ± 15	57 ± 8	0.75	0.10
	Post DC	577	−157 ± 26	−118 ± 25	3 ± 19	−90 ± 14	−37 ± 25	−156 ± 39	−133 ± 32	−109 ± 19	0.37	0.16
Reco	Pre DC	395	177 ± 17	168 ± 16	121 ± 10	155 ± 8	167 ± 14	168 ± 14	133 ± 9	156 ± 7	0.78	**<0.01**
	Post DC	577	277 ± 19	296 ± 20	239 ± 17	271 ± 10	302 ± 21	368 ± 23	282 ± 18	317 ± 12	**<0.01**	**0.02**
GPP	Pre DC	395	−187 ± 20	−81 ± 15	−25 ± 7	−97 ± 10	−101 ± 19	−94 ± 15	−103 ± 18	−99 ± 10	0.94	**<0.01**
	Post DC	577	−433 ± 34	−414 ± 31	−235 ± 26	−360 ± 18	−340 ± 37	−525 ± 49	−418 ± 43	−427 ± 25	**<0.01**	**0.02**
CH_4 (mg C m^{-2} h^{-1})												
	Pre DC	392	0.00 ± 0.02	−0.03 ± 0.03	−0.02 ± 0.01	−0.02 ± 0.01	−0.04 ± 0.01	−0.03 ± 0.01	−0.02 ± 0.1	−0.03 ± 0.01	0.24	0.85
	Post DC	567	1.2 ± 0.4	0.08 ± 0.3	−0.03 ± 0.01	0.08 ± 0.9	−0.07 ± 0.01	−0.04 ± 0.01	−0.03 ± 0.2	−0.05 ± 0.06	**<0.01**	**<0.01**

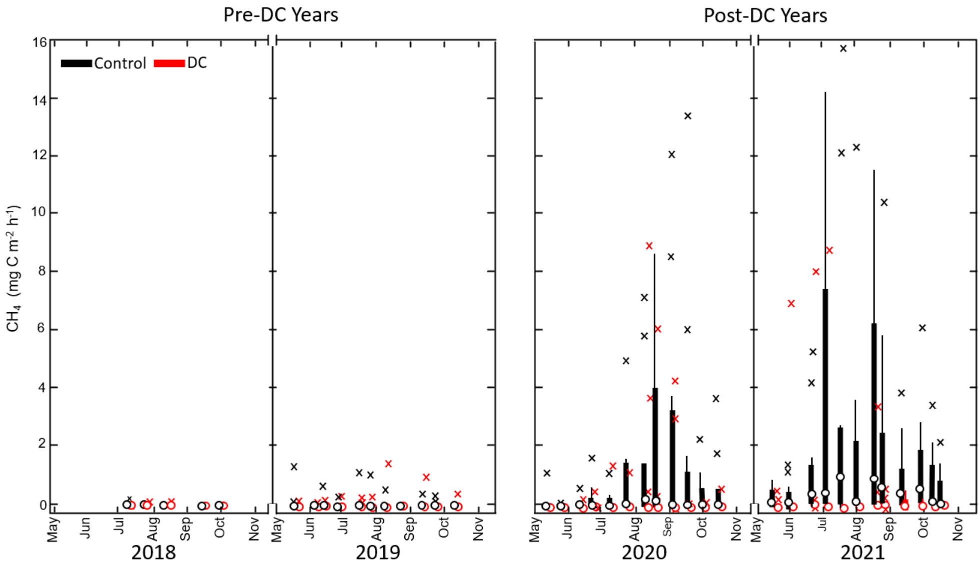

Figure 6. Seasonal variation of methane (CH_4) fluxes in control and ditch cleaning (DC) clear-cut areas during pre-DC (2018/19) and post-DC periods (2020/21). Negative values denote uptake. The circles denote sample (n = 12) medians, whilst the bottom and top edges of the thick bar indicate the 25th and 75th percentiles. The vertical line denotes the range excluding outliers (crosses), which are defined as the values more than 1.5 interquartile range away from the top or bottom of the line. Data shown represent the means across all distances from ditches.

3.4. Distance to Ditch Effect Effects on CO_2 and CH_4 Fluxes

The distance to ditches had contrasting effects on NEE, R_{eco} and GPP. Specifically, NEE shifted from a net CO_2 sink to a net CO_2 source with increasing distance to ditches in the control area, whereas the magnitude of net CO_2 uptake decreased with distance to ditches in the DC area (Table 2). This spatial pattern persisted throughout all four measurement years. Meanwhile, the significant effects of distance to ditches on R_{eco} and GPP were only noted in the control area, suggesting a decrease in their magnitude with increasing distance to the ditches. In the DC area, however, the highest R_{eco} and GPP values were observed at 20 m from the ditch (Table 2). These distance effects were consistent throughout all measurement years.

As the CH_4 emission spikes were concentrated at the plots 4 m from the uncleaned ditches, the mixed effect models also suggest a negative effect of distance to ditch on the CH_4 fluxes. However, additional inspection of the data suggests that this distance to ditch effect was mainly driven by a strong gradient in the control area, where CH_4 emissions, coinciding with highest WTL, remained highest at the plots closest (4 m) to ditches.

3.5. Three-Way Interaction between Ditch Treatments, Environmental Factors and GHG Fluxes

The PCA on all variables *Rec*orded during the pre-DC years showed that in total, 83% of the total variance was explained by five significant PCs (Table S3). Together, the first two most significant PCs explained 52% of the total variance (Figure 7). PC1 revealed that vegetation growth (g_{cc} and areal coverage) was most strongly coherent with the CO_2 component fluxes of R_{eco} and GPP. Vegetation variables were also positively related to soil carbon and nitrogen content and negatively related to soil temperature. PCA results further showed that R_{eco} and GPP were negatively associated with the distance to ditch and CN ratios. CH_4 flux had a strong positive connection with WTL and soil moisture in both PC1 and 2 but was not associated with the distance to ditch. During the pre-DC years, the DC treatment variable had relatively small variable loadings in both PC1 and 2, implying that pre-DC environmental conditions and fluxes were comparable between the two measurement areas.

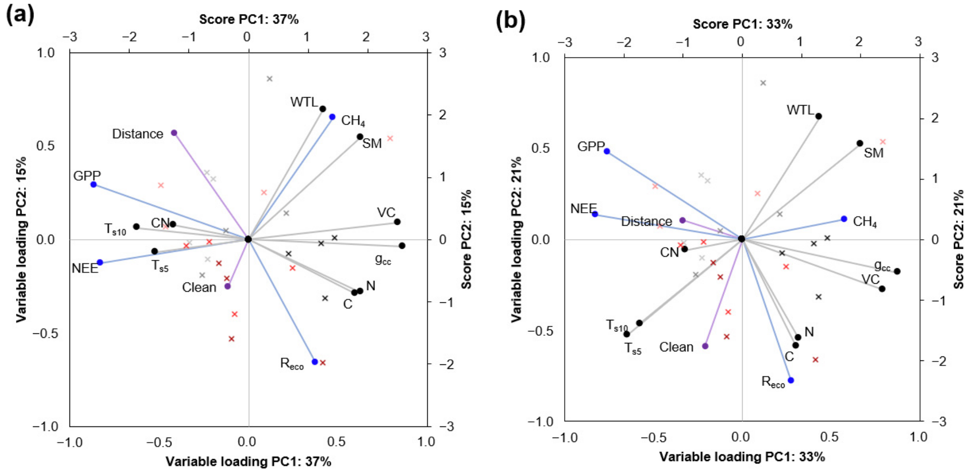

Figure 7. Principal component analysis (PCA) biplots of the first two principal components (PC1 and PC2) based on the plot-averaged measurement data displaying variable loadings and object scores, during (**a**) pre-DC and (**b**) post-DC periods. Variable loadings of all other significant PCs are listed in Table S3. Filled symbols denote the component loadings of the measured variables. Crosses highlighted in black (control area) and red (DC area) denote the component scores of each measurement plot. Colour intensity of component scores decreases with the distance to the ditches. Note that ecosystem uptake in NEE and GPP is given a negative sign, resulting in negative correlation to both PC1s despite positive causal relation. Abbreviations represent greenness index (g_{cc}), soil moisture (SM), air temperature (T_a), soil 5 or 10 cm depth temperature (T_{s5}, T_{s10}), vegetation areal coverage (VC), water table level (WTL), soil carbon (C) and nitrogen content (N) along with their C:N ratio (CN) for environmental variables (black symbols); net ecosystem exchange (NEE), ecosystem respiration (R_{eco}), gross primary productivity (GPP) and methane (CH_4) for flux variables (blue symbols); and DC treatment and distance to ditch variable (purple symbols). Note that the DC variable was fitted as a binary variable for control (0) and DC (1) treatments.

During the post-DC years, 80% of the total variance was explained by four significant PCs (Table S3), while together, the first two most significant principal components explained 54% of the total variance (Figure 7). Compared to the pre-DC years, the DC treatment effect had a larger association with the environmental and flux variables, as reflected by the increased variable loadings in the first two PCs. Specifically, the DC treatment was associated with lower soil moisture, lower WTLs and with increased soil temperature, in correspondence with lower CH_4 fluxes. In contrast, the CO_2 component fluxes were independent from the DC effects on soil temperature and moisture levels but instead were primarily controlled by vegetation growth. The latter also remained positively related to soil carbon and nitrogen contents. The results further show that soil moisture, WTL and thereby CH_4 emissions decreased with greater distance from the ditches. Meanwhile, the small loading of distance to ditch indicated its limited association with the other environmental and CO_2 flux variables relative to the pre-DC period.

3.6. Total Annual Carbon and GHG Balances

Our model estimates suggest that our study site was an annual source of carbon during the four study years, regardless of the DC treatment (Figure 8; Table S2). However, the source magnitude decreased by 78% and 74% in the control and DC areas, respectively, from 6.1 ± 1.2 t-C ha^{-1} year^{-1} and 7.2 ± 1.6 t-C ha^{-1} year^{-1} in 2018 to 13.1 ± 1.4 t-C ha^{-1} year^{-1} and 14.4 ± 1.4 t-C ha^{-1} year^{-1} in 2021. The reduction in the net CO_2 emission occurred because GPP (from zero to −12.1 ± 0.7 t-C ha^{-1} year^{-1}) increased more than R_{eco} (from 6.6 ± 1.4 t-C ha^{-1} year^{-1} to 13.8 ± 1.1 t C ha^{-1} year^{-1}) between 2018 to 2021 in both areas. The interannual variations in the modelled annual CO_2 flux components were similar between the control and DC treatment areas over the four years. Averaged over the four study years, there was no significant difference between the model estimates of annual NEE in the control and DC areas.

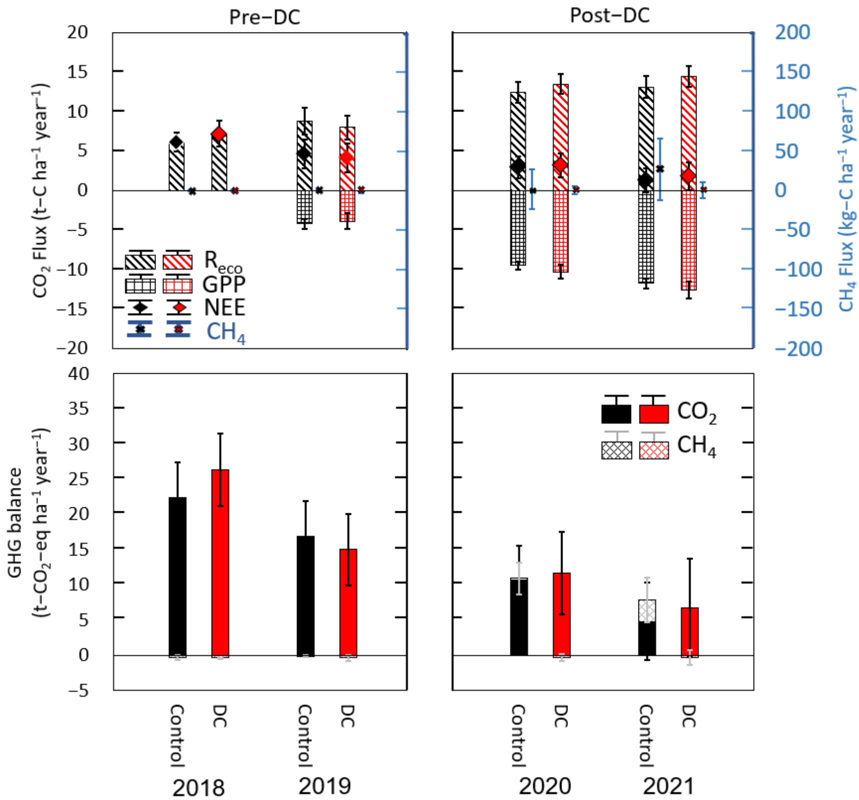

Figure 8. Model estimates for the total annual carbon dioxide (CO_2) and methane (CH_4) balance (upper panels) and total annual greenhouse gas balance (lower panels) in the control and ditch cleaning treatments areas (DC) during the four study years. CO_2 balances comprise the net exchange (NEE) and its component fluxes of gross primary productivity (GPP) and the ecosystem respiration (*Reco*). Scale of CH_4 flux is at 0.1 times scale of CO_2 flux (blue axis to the right). Greenhouse gas balance of CO_2 and CH_4 uses global warming potentials of 86 for CH_4 over a 20−year timeframe (IPCC, 2013).

The interannual variations in the modelled annual CH_4 balance over the four years differed between the control and DC treatment areas. Specifically, in the control area, the annual CH_4 flux increased from −2.2 to 27 kg C ha^{-1} year^{-1} between 2018 and 2021. Within the control area, the observed CH_4 emissions at 4 m from the uncleaned ditches were by 1 and 2 magnitudes higher relative to the 20 and 40 m locations, respectively, leading to a contribution of 6% and 25% to the carbon balance at this location in 2020 and 2021, respectively. In the DC area, however, the inter-annual variation in the CH_4 balance was relatively small and stable within a range of −3.0 to −2.3 kg-C ha^{-1} year^{-1}, which represented < 0.15% of the total carbon balances in all four study years. However, when considering the global warming potential of CH_4 as 86 times higher relative to CO_2 over a 20-year time frame (IPCC, 2014), the median contribution of CH_4 to the annual GHG budget (in t CO_2-eq ha^{-1} year^{-1}) increased from 0.1% in 2018 to 39% in 2021 in the control area, but remained within <5% in the DC area, respectively. The enhanced CH_4 emissions resulted in a 23% increase in the total annual GHG emissions in the control area in 2021; however, given the large uncertainty (from the spatial variation) associated with the annual CH_4 flux estimate, the total annual GHG balances were not significantly different between the control and DC areas.

4. Discussion

4.1. Ditch Cleaning Effects on Hydrology, Vegetation and GHG Fluxes

This study investigated DC's effects on CO_2 and CH_4 fluxes in a two-way experimental set up, i.e., through (i) a direct comparison of fluxes in the control and DC area, respectively, and (ii) a pre- versus post-treatment assessment in the DC area. Both evaluation alternatives indicated only a minor effect of DC on the net CO_2 exchange and its component fluxes GPP and $R_{eco.}$ Instead, both spatial and temporal variations were mainly dependent on the within-site variations in herbaceous ground vegetation during the post-harvest years. The similar spatial pattern of herbaceous ground vegetation before and after DC indicated the minor impact of DC on the ground vegetation development. In addition, GPP and R_{eco} were strongly correlated with variations in soil carbon and nitrogen, which is in line with previous studies suggesting that soil fertility and organic content are the primary controls over spatial variations of CO_2 fluxes [49,50]. This is because the decomposition of organic matter and associated nutrient release are tightly coupled to microbial respiration and plant carbon uptake [51]. Essentially, positive feedback may develop as the increased plant growth provides additional organic matter input, thereby speeding up the carbon cycle [50]. The spatial variations in both ground vegetation together with soil carbon and nitrogen at our study site itself did not correspond to DC effects and instead might be a relict of the historical drainage effects and/or disturbance during harvest and site preparation.

The lack of WTL changes in the DC area after DC appeared surprising at first. However, this could be explained by the increase in precipitation during the post-DC years, which may have counterbalanced the enhanced drainage function following DC. In fact, the concurrent rise in WTL (by 14 cm) in the control area reveals an increased drainage function in the DC area during the two post-DC years. Given the wetter conditions, the control area was characterised by a relatively shallow WTL (−17 cm on average) during these two post-harvest years, resulting in a narrow upper oxic zone which likely supressed decomposition and the growth of vascular plants, thus explaining the limited R_{eco} and GPP noted in the control area during these wet years (Table 2). However, we observed no significant correlation between WTL and CO_2 flux components (i.e., R_{eco} and GPP) during periods when WTL was <−20 cm, which agrees with similar findings from a peatland forest clear-cut in boreal Finland [26]. This indicates that DC's effects on the *Reco*, GPP and NEE largely depend on both the initial WTL and the effectiveness of the DC, which jointly regulate the change in soil surface oxic conditions and subsequently the production and decomposition of organic content in the initial years following harvest.

Contrary to our expectations, we did not observe a clear pattern in the effect of ditch distance on WTL, soil moisture and CO_2 fluxes. Given the constantly shallow WTL conditions (−17 cm on average during the two post-harvest years) at all distances from uncleaned ditches, the R_{eco} and GPP were higher at locations closer to these uncleaned ditches, which could be due to the contribution from the more extensive growth of Sphagnum moss under the wet conditions. In the DC area, however, the highest R_{eco} and GPP values observed at 20 m from the cleaned ditches may have resulted from optimum WTLs of −35 to −40 cm that created a favourable balance between sufficient water and oxygen availability for enhancing vascular plant growth and decomposition processes [52,53]. Such an impact could have dominated over the drainage legacy effects, as observed in the control area with limited WTL variations. Since the year-to-year increase in GPP was greater than that of R_{eco} at 20 m from the cleaned ditches, the most negative measured daytime NEE (-156 ± 39 mg C m^{-2} h^{-1}) indicates that such a WTL condition likely favours net CO_2 uptake. Nevertheless, we cannot rule out the possibility that the high rainfall during both post-DC years may have influenced the moisture condition and eventually the locations where the optimal WTL with highest fluxes were observed. Thus, additional observations during dry years will be required to further elucidate the interactions among ditch distance, soil water levels and CO_2 fluxes.

CH_4 fluxes are known to be strongly related to soil oxic conditions [54,55] and thus are likely modified by a reduction in WTL following DC. Specifically, the deeper mean WTL of -35 ± 2 cm in the DC area retained a substantial surface oxic layer and thereby provided favourable conditions for CH_4 oxidation while limiting CH_4 production to the deeper soil layer [26]. In comparison, the majority (75%) of plots in the control area, including all plots at 4 m distance from the ditches, were characterised by relatively high WTL (between 0 and −20 cm) and close to saturated soil conditions (> 0.45 m^3 m^{-3}), which may have facilitated large CH_4 production and emissions from the enhanced anaerobic soil layer [26,56,57]. However, it is noteworthy that CH_4 emission spikes occurred more frequently at several DC plots given the same or even deeper WTL during the two post-DC years. This is likely due to the enhanced amount of vascular ground vegetation (e.g., *Deschampsia flex* spp.) noted in these DC plots, with roots reaching deep into the CH_4 production zone, providing substrate from root exudates to methanogens and supporting the plant-mediated transport of CH_4 into the atmosphere [58–60]. Alternatively, more frequent rain events might have resulted in additional CH_4 production from waterlogged microsites within the oxic soil layer [61]. Altogether, these findings highlight that, relative to the CO_2 exchange, CH_4 fluxes were more sensitive to DC via its effects on soil hydrological properties and plant growth.

4.2. Ditch Cleaning Effects on the Annual C and GHG Balances of a Boreal Forest Clear-Cut

We observed a rapid decrease in annual carbon emissions during the four measurement years, which indicates that the high emissions commonly expected to occur in boreal clear-cuts [26,62] might be limited to only the initial few years. One reason for this decrease was the rapid development of the herbaceous ground vegetation, which resulted in increasing GPP. The importance of the fast *Reco*very of ground vegetation was previously highlighted in nutrient-poor boreal forest clear-cuts on mineral soils [63–65]. In comparison, the increase in R_{eco} over the four years was relatively smaller, possibly because the wet conditions in 2020/21 might have constrained microbial aerobic decomposition. Previous studies have shown that the *Reco*very rate of the net carbon balance after clear-cutting depends on regeneration management, biomass production and decomposition rate, suggesting that it could take anywhere between 1 and 20 years for GPP to counterbalance R_{eco} [62,66–69].

Despite the higher daytime R_{eco} and GPP measured in the DC area, our model estimates of their annual sums were not different between the two experimental areas during both post-DC years. The apparent discrepancy in the treatment effect between the modelled annual sums relative to the instantaneous daytime measurements is explained by the greater vegetation coverage in the surrounding area compared to the measurement frames in the control treatment (Figure 4). While the measured fluxes corresponded to the vegetation coverage within the frames, the vegetation coverage in the surrounding area was used as input in the model extrapolation with the goal to obtain estimates that were representative of the entire area. The higher vegetation coverage in the surrounding control area relative to the DC area was primarily a result of the higher prevalence of Sphagnum moss species under favourable shallow WTL conditions. We note that our results are confined to the initial vegetation responses after DC, whereas continued vascular species and seedling development will likely result in a greater contribution from plant carbon uptake to the total respiration at the DC area in the future.

Our result that CH_4 had a minor (<2.5%) contribution to the total carbon balance is in agreement with several previous studies conducted in other boreal forest clear-cuts on drained mineral soils [62,63,70] as well as with findings from peatland forest clear-cuts [26]. However, in terms of the total GHG balance where, the GWP of CH_4 is 86 times over CO_2, CH_4 emissions become increasingly important when the CO_2 balance is closer towards carbon neutral, i.e., during the later phase when the site shifts from an annual source to sink (Figure 7). Leaving the ditches uncleaned might further create local hotspots for CH_4 emission within the waterlogged areas near the ditch. In comparison, DC effectively mitigated CH_4 emissions by maintaining low WTLs deeper than −30 cm during the wet post-harvest years, resulting in even slightly negative (i.e., uptake) annual GHG balance. This is in agreement with a previous study suggesting that post-harvest CH_4 emissions increase only if the WTL rises above approximately −20 cm [65]. It is noteworthy that the modelled annual CH_4 flux sums were characterised by considerable uncertainty due to the high spatial variability inherent to our measurement plots. The use of the eddy covariance technique is required to overcome this challenge and to provide more precise estimates for the CH_4 balance of heterogeneous clear-cut areas [71].

We caution that our study lacked flux data from the snow-covered winter periods, and our annual flux budget therefore relied on simple assumptions for estimating the contribution from winter fluxes. However, previous studies based on continuous year-round data collected with the eddy covariance and automated chamber techniques suggest low fluxes during snow-covered winter periods commonly contributing to less than 10% of the annual CO_2 and CH4 budgets in boreal ecosystems [62,72,73]. This indicates a limited potential bias introduced by our assumptions in estimating the winter flux. It is further noteworthy that this study only addressed the initial responses of the environment and fluxes to DC. However, long-term effects on the growth of the tree layer might further enhance the DC impact on the forest carbon balance over an entire rotation period. Moreover, the ditch functions will again deteriorate over time through erosion and sedimentation, which may result in a nonlinear trajectory of DC effects on the soil water dynamics and thus carbon balance. Thus, more empirical data from winter periods and extending over the decadal timescale of post-DC years are required to better understand the DC effects on the forest ecosystem carbon balance across contrasting sites. Given the steady increase in DC activities, particularly within Fennoscandia, it is essential to improve our knowledge of DC effects on the forest ecosystem–atmosphere exchange of carbon and GHGs with the goal to provide an empirical knowledge base that can support the development of sustainable and climate-responsible forest management strategies.

5. Conclusions

This study investigated how the cleaning of degraded drainage ditches affects CO_2 and CH_4 fluxes from a drained forest clear-cut on mineral soil in boreal Sweden over four years (i.e., 2–5 years after harvest). Based on our findings, we conclude that:

(1) The clear-cut area with old and degraded ditches acted as a net carbon source in all four post-harvest years. However, during the study period, the annual total carbon emissions decreased by 76% (from 6.7 ± 1.4 t-C ha^{-1} year^{-1} to 1.6 ± 1.6 t-C ha^{-1} year^{-1}).

(2) Ditch cleaning had a limited initial effect on the spatio-temporal variations in the net CO_2 exchange and its component fluxes, GPP and $R_{eco.}$ The variation in the component fluxes was instead primarily controlled by the within-site variations in ground vegetation development, likely in response to drainage legacy effects on soil carbon and nitrogen contents.
(3) In comparison, ditch cleaning reduced the soil water content and thereby mitigated CH_4 emissions during wet post-harvest years.
(4) Ditch distance had no consistent effect on CO_2 and CH_4 fluxes. While R_{eco} and GPP tended to increase towards uncleaned ditches coinciding with legacy trends in soil carbon and nitrogen content, maximum R_{eco} and GPP occurred at 20 m from cleaned ditches, likely in response to an optimal WTL for vascular plants commonly observed in the DC area. In comparison, high emissions of CH_4 mainly occurred on nearly saturated soil locations near to uncleaned ditches.
(5) Overall, ditch cleaning had no significant impact on the annual carbon and GHG balance in the initial post-harvest years.
(6) There is a critical need for long-term observations to evaluate DC effects on the forest carbon and GHG balances during the entire subsequent rotation period.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/f13060842/s1, Table S1: Annual mean ± standard error (SE) and mixed effect model results for environment variables during manual flux measurements during pre-DC (2018/19) and post-DC periods (2020/21), classified by ditch treatment effects (Control versus DC) and distance to ditches combinations (4m, 20m and 40m). Fixed factors of mixed effect models include clean treatment (T), distance to ditches (D) and their interaction (TD). Column N refers to the sample size of each model. Significant p; Table S2: Model estimates for the total annual carbon dioxide (CO_2) and methane (CH_4) balances in control and ditch cleaning (DC) treatment areas. CO_2 fluxes includes its net exchange (NEE) and component fluxes of gross primary productivity (GPP) and ecosystem respiration (R_{eco}). Columns denotes the three distances (4m, 20m and 40m) from ditches estimated for the four study years (2018 to 2021). Values are in the unit of t-C ha^{-1} year^{-1} for CO_2 and of kg-C ha^{-1} year^{-1} for $CH_{4.}$ Numbers are represented with ± standard error (SE); Table S3: The correlation coefficients (or loadings) of the each significant Principal components (PCs) with the input variables. Significant PCs are defined by the Kaiser criterion where the eigenvalues are greater than one. Five PCs were defined significant in the PCA with sample data for Pre-DC Period shown at the left columns, whereas four PCs were significant for the Post-DC period at the right columns. The total variance explained for each PC (in %) is presented at the first row.

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Appendix A. Additional Information on the Measurement of N_2O Fluxes

N_2O fluxes were measured biweekly with a separate set of opaque chambers (48.5 × 48.5 × 50 cm) in 2020. The chambers were placed on the frames for 75 min, during which, four 60 mL gas samples were taken from the chamber with plastic syringes at 0, 25, 50 and 75 min after closure. During chamber closure, a small fan was operated inside the chamber to maintain air circulation, and a Hobo® pendant temperature logger (Onset Computers, Bourne, MA, USA) was provided to continuously Record air temperature at 5 s intervals in the chamber headspace. The headspace gas in the 60 mL syringe was then injected into 20 mL evacuated glass vials and determined for their N_2O concentration within seven days using a headspace sampler (TurboMatrix 110; Perkin-Elmer, Waltham, MA, USA) and gas chromatograph (Clarus 580, PerkinElmer Inc., Waltham, MA, USA). N_2O was separated by two identical 30 m × 0.53 mm internal diameter megabore capillary porous-layer open tubular columns (Elite PLOT Q) maintained at 30 °C (detection limit: N_2O < 1 ppb). The GC system was equipped with an electron capture detector (ECD) operated at 375 °C for N_2O analysis. The linear increase in N_2O concentrations inside the chamber over time was then converted into a flux estimate using the ideal gas law (Equation (A1)):

$$F = \frac{dC}{dt} \times \frac{V \times p}{R \times T_a \times A} \tag{A1}$$

where F is the measured flux (μmol m^{-2} s^{-1}), dC/dt is the linear slope with the highest r^2 of concentration change over time (ppm s^{-1}), V is chamber headspace volume (m^3), p is the atmospheric pressure (approximated by a constant value of 101,325 Pa), R is the universal gas constant of 8.3143 (m^3 Pa K^{-1} mol^{-1}), T_a is the mean air temperature (K) during the measurement, and A is the frame area (m^2).

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Forest drainage has been extensively used to facilitate forest growth in Fennoscandinavia. This work quantified the effect of historical drainage and recent ditch cleaning on carbon and greenhouse gas fluxes in the northern forests of Sweden under various geographical settings and soil conditions, using closed chamber, eddy covariance and stream discharge measurements. Results show that forestry drainage activities do not intensify carbon and greenhouse gas emissions. This work provides insight for the development of sustainable forest management strategies.

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