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Linking land-use management and aquatic carbon

Effects of forestry and peatland rewetting

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Linking land-use management and aquatic carbon: effects of forestry and peatland rewetting

Abstract

To manage the forested landscape in a sustainable way while maintaining or even improving its carbon (C) sequestration capacity is a key challenge in climate change mitigation strategies. An often neglected aspect in this context is how different management practices affect the export of terrestrial C via the aquatic pathway. In intensively managed boreal landscapes like Sweden, a dense network of man-made drainage ditches designed to promote forest growth receives high amounts of terrestrial C, with implications for C balance estimates and downstream water quality. This thesis presents field-based observations on the effects of clear-cutting, ditch cleaning (DC), and rewetting of drained wetlands on aquatic C of different forms. First, I show that clear-cutting can shift CO_2 concentration dynamics in a boreal ditch from being mostly hydrologically driven to being mainly controlled by light- and temperature-dependent metabolic processes. Second, I explore the effect of DC on runoff chemistry by comparing cleaned and uncleaned ditches. Some results were consistent with earlier studies (higher pH), while others show divergent (similar total organic C (TOC)) or novel (higher nitrous oxide (N_2O) and lower CO_2) patterns in cleaned compared to uncleaned ditches. These patterns are suggested to stem from deeper flow paths and altered redox conditions after DC. Finally, I assess the effects of rewetting of drained wetlands on aquatic C at local and regional scales. Among the results, it was evident that rewetting increased the dissolved concentrations of organic and inorganic C (DOC and DIC), and methane (CH_4) in runoff, which in turn caused enhanced lateral C exports. However, the effects were site-, C form- and season-specific and were influenced by e.g. presence of open-water areas, latitude and nutrient status. The thesis provides an important knowledge basis to be used when implementing the examined management practices, to ensure intended outcomes with minimal negative consequences on runoff C.

Keywords: Sweden, boreal, ditches, DOC, DIC, methane, ditch-cleaning, clear-cut

Kopplingen markanvändning och akvatiskt kol: effekter av skogsbruk och återvätning av torvmarker

Sammanfattning

Att hållbart bruka skogslandskapet och samtidigt behålla eller till och med förbättra dess förmåga att binda kol (C) är en utmaning i strategier för att minska pågående klimatförändring. En ofta försummad aspekt i detta sammanhang är hur olika skötselmetoder påverkar exporten av terrestert C via akvatiska system. Intensivt brukade boreala landskap som i Sverige präglas ofta av ett tätt nätverk av dräneringsdiken som är utformade för att främja skogstillväxt men tar samtidigt emot stora mängder terrestriskt C, vilket har konsekvenser för skattningar av landskapets kolbalans samt nedströms vattenkvalitet. Denna avhandling presenterar fältbaserade observationer rörande effekter av kalhuggning, dikesrensning (DC) och återvätning av dikade våtmarker på olika former av akvatiskt C. Först visar jag att kalhuggning kan förändra dynamiken av koldioxid (CO_2) i ett skogsdike från att vara mestadels hydrologiskt driven till att styras av ljus- och temperaturberoende metaboliska processer. För det andra utforskar jag effekten av DC på avrinningskemin genom att jämföra rensade och orensade diken. Vissa resultat överensstämmer med tidigare studier (högre pH), medan andra visar skilda (liknande totalt organiskt C) eller nya (högre dikväveoxid (N_2O) och lägre CO_2) mönster i rensade jämfört med orensade diken. Dessa skillnader förklaras av djupare flödesvägar och förändrade redox-förhållanden efter DC. Slutligen undersöker jag effekten från återvätning av dikade våtmarker på akvatiskt C på lokal och regional skala. Resultaten visade tydligt att återvätning ökade den lösta koncentrationen av organiskt och oorganiskt C (DOC och DIC) samt metan (CH_4) i avrinning, vilket i sin tur orsakade ökad lateral C-export. Effekterna var dock plats-, C form- och säsongsspecifika och påverkades av t.ex. förekomst av öppna vattenspeglar, latitud och näringsstatus. Avhandlingen ger en viktig kunskapsbas att använda vid olika brukande av skogslandskapet, så att avsedda effekter nås med minimala negativa konsekvenser för C i avrinnande vatten.

Keywords: Sverige, boreal, diken, DOC, DIC, metan, dikesrensning, hygge

This is an adventure.

—*Steve Zissou*

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List of publications

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

I. **Zannella, A.**, Eklöf, K., Lannergård, E., Laudon, H., Maher Hasselquist, E., & Wallin, M. B. (2023). Metabolic processes control carbon dioxide dynamics in a boreal forest ditch affected by clear-cut forestry. *Frontiers in Water*, 5, 1250068. https://doi.org/10.3389/frwa.2023.1250068

II. **Zannella, A.**, Wallin, M. B., Sikström, U., Arvidsson, E., & Eklöf, K. (2024). Ditch cleaning in boreal catchments: Impacts on water chemistry and dissolved greenhouse gases in runoff. *Forest Ecology and Management*, 569, 122146. https://doi.org/10.1016/j.foreco.2024.122146

III. **Zannella, A.**, Eklöf, K., Maher Hasselquist, E., Laudon, H., Garnett, M. H. & Wallin, M. B. Changes in aquatic carbon following rewetting of a nutrient-poor northern peatland (submitted).

IV. Wallin, M. B., **Zannella A.** & Eklöf, K. Enhanced carbon in runoff following rewetting of previously drained forest wetlands (manuscript).

Papers I and II published open access.

The contribution of Alberto Zannella to the papers included in this thesis was as follows:

I. The respondent was responsible for the study design and the majority of the fieldwork. In addition, he analyzed the data and led the writing and publication process.

II. The respondent led the writing and contributed to the interpretation of data and results.

III. The respondent was responsible for a part of the data collection. In addition, data analysis, interpretation and writing were mainly done by the respondent.

IV. The respondent was responsible for all data collection. In addition, he was involved in data handling, analysis, interpretation and writing of the paper.

Additional publications

In addition, the author has contributed to the following papers, which are not included in the thesis:

I. Laudon, H., Mosquera, V., Eklöf, K., Järveoja, J., Karimi, S., Krasnova, A., Peichl, M., Pinkwart, A., Tong, C.H.M., Wallin, M.B., **Zannella, A.**, & Hasselquist, E.M. (2023). Consequences of rewetting and ditch cleaning on hydrology, water quality and greenhouse gas balance in a drained northern landscape. *Sci. Rep.*, 13, 1–13. https://doi.org/10.1038/s41598-023-47528-4

II. Olid, C., **Zannella, A.**, & Lau, D. C. P. (2021). The role of methane transport from the active layer in sustaining methane emissions and food chains in subarctic ponds. *Journal of Geophysical Research: Biogeosciences*, 126, e2020JG005810. https://doi.org/10.1029/2020JG005810

Abbreviations

C	Carbon
^{14}C	Radiocarbon
CH_4	Methane
CO_2	Carbon Dioxide
DC	Ditch Cleaning
DIC	Dissolved Inorganic Carbon
DOC	Dissolved Organic Carbon
GHG	Greenhouse Gas
GWT	Groundwater Table
LCE	Lateral Carbon Export
MeHg	Methyl Mercury
N_2O	Nitrous Oxide
NECB	Net Ecosystem Carbon Balance
Q	Discharge
THg	Total Mercury
TOC	Total Organic Carbon

1. Introduction

"Attento al rivetto!" ("Mind the ditch!")
—My mom, when my dad was reversing the car

The Land Use, Land Use Change and Forestry (LULUCF) sector is vital to achieve climate neutrality (European Commission, 2021) and is currently estimated to offset about two-thirds of the total greenhouse gas (GHG) emissions in Sweden (SEPA, 2024; Figure 1). A national strategy is to manage forest land to meet the growing demand for forest products while also mitigating climate change by increasing carbon (C) sequestration (Swedish Climate Policy Council, 2022). Of Sweden's total land area, 58 % is accounted for as productive forest (SLU, 2023), defined as forests producing ≥ 1 m^3 ha^{-1} $year^{-1}$ of wood. Forests that fall below this productivity threshold are considered unproductive and are often found in open areas such as peatlands, rocky outcrops, and at high altitudes (Swedish Forest Agency, 2020). Current production-oriented forestry in Sweden, as in most boreal regions, is primarily based on even-aged clear-cutting, followed by interventions to aid forest regeneration (Laudon and Hasselquist, 2023). This land-use strategy has been proposed as a way to increase forest biomass production, providing a renewable energy source to address future environmental challenges (Egnell et al., 2011). However, given the close connection between terrestrial landscapes and surface waters, this approach also poses risks to water quality and aquatic ecosystems, which may be negatively affected by the intensification of this forestry practice (Laudon et al., 2011a). Another land-use method historically used to promote forest growth and facilitate agriculture in waterlogged areas is the drainage of forest peatlands by digging ditches. However, this practice has significantly altered the present-day forest landscape (Norstedt et al., 2021). Recent estimates suggest that Sweden has a total of 1 million km of man-made ditches, twice

the length of all Swedish natural streams (Ågren and Lidberg, 2019). Today, ditching is strictly regulated as it led to widespread degradation of associated ecosystems. The decline or loss in original drainage efficiency of numerous forest ditches, caused by the accumulation of vegetation and sediments over time, calls for remedial actions like ditch cleaning (DC) (Sikström and Hökkä, 2016). While DC has the potential to restore forest growth, it also has serious environmental impacts. These include erosion, nutrient losses as well as changes in the transport of carbon, mercury and sediments to aquatic ecosystems (Nieminen et al., 2018; Wesström et al., 2017). Furthermore, DC in peat-rich areas may increase peat mineralization, resulting in elevated carbon dioxide (CO_2) emissions (Ojanen et al., 2013). Hence, careful consideration of downstream effects is necessary when conducting such efforts. Contrary to DC, an alternative approach is soil rewetting (more generally known as *rewetting*), which involves blocking ditches to reduce their drainage capacity while simultaneously rewetting drained unproductive areas by raising the groundwater tables (Landry and Rochefort, 2012). From a climate perspective, the goal of rewetting the soil is to restore the ability of wetlands to retain peat, thereby capturing C and reducing CO_2 emissions (Hiraishi et al., 2014). Moreover, rewetting is expected to have positive effects on biodiversity and water retention in the forest landscape (Parish et al., 2008). Thus, there is a significant and growing interest in rewetting, with substantial government funding allocated to relevant stakeholders. In June 2024, the Nature Restoration Law was approved as part of the EU national long-term climate strategy with the aim to restore (including to rewet) degraded ecosystems, particularly those with active roles in GHG cycling and C sequestration, such as peatlands (European Commission, 2022). However, similar to DC, rewetting may also result in unintended environmental impacts, such as an elevated risk of enhanced organic C export to surface waters and methane (CH_4) formation due to the creation of oxygen-depleted environments (Wilson et al., 2016).

Whether it is better to rewet areas where no production has occurred after ditching, and whether to clean deteriorated ditches or to leave them alone are currently big questions for landscape managers. There are major considerations that are required before action, balancing the potential benefits and drawbacks. DC could enhance drainage and promote forest growth, while rewetting could reduce the risk of floods and droughts,

enhance biodiversity, and positively influence the climate. However, these actions can also lead to lower water quality by releasing buried organic material into surface waters or creating hotspots for GHG production. Our understanding of the impacts of each practice on surface waters ecosystems is still limited, making it difficult for stakeholders to make informed decisions on managing existing ditches in the forested landscape. Therefore, empirical field studies can provide valuable insights to fulfil knowledge gaps and improve our understanding of aquatic C dynamics in the forest.

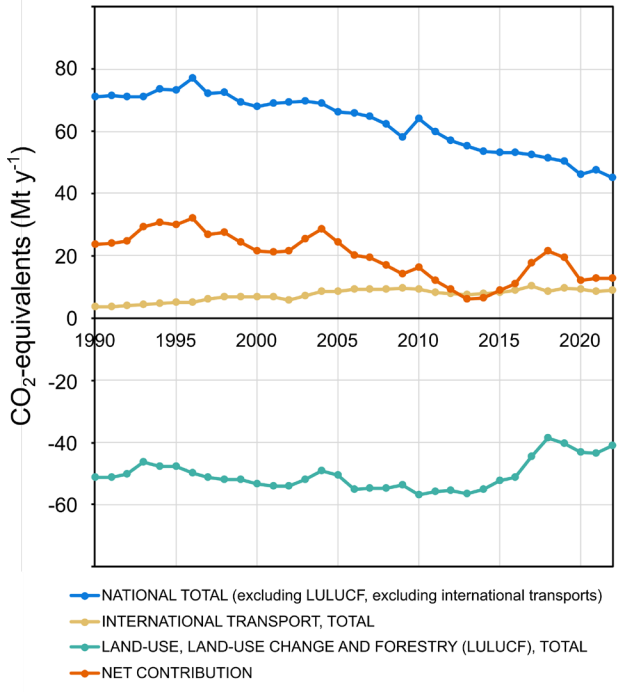

Figure 1. The graph shows the total annual emissions and removals of greenhouse gases (in Mt CO_2-eq y^{-1}) in Sweden. The orange line is the sum of total emissions (national + international transport) and uptake by land use, land use change and forestry (LULUCF). This sector currently offsets around two-thirds of the national territorial emissions, highlighting the key role of Swedish forests in balancing these emissions. Source data: www.scb.se.

2. Objectives

In this thesis, by presenting field-based observations, I aim to advance our understanding of the effects of land-use practices such as clear-cutting, ditch cleaning, and rewetting on water chemistry, with a particular focus on aquatic C and dissolved GHG. The specific objectives for each study in this thesis collection were as follows:

I. To explore CO_2 dynamics in a forest ditch affected by clear-cut forestry and identify major drivers for these dynamics compared to patterns in a non-harvested ditch (**Paper I**).

II. To assess, on a regional scale, whether DC caused significant differences in water chemistry and dissolved GHGs between cleaned and uncleaned ditches and explore any differences in DC effects between forested and clear-cut sites (**Paper II**).

III. To determine the effects of peatland rewetting on aquatic C chemistry in runoff and pore water at the catchment scale; to further quantify the impact of rewetting on lateral C exports; and finally identify whether rewetting caused any change in the age composition of runoff C (**Paper III**).

IV. To explore the impact of rewetting drained wetlands on runoff C and dissolved GHGs across large geographical and climatic ranges of boreal and hemi-boreal Sweden; and to evaluate whether the rewetting effects were related to the site-specific nutrient statuses (**Paper IV**).

3. Background

3.1 Carbon cycling of the boreal forest

Boreal forests play a critical role in the global C cycle by taking up CO_2 from the atmosphere through photosynthesis. The fixed C is used by vegetation for growth, ultimately contributing to soil litter formation and the accumulation of soil C stocks when vegetation falls onto the ground. While some of this C is released back into the atmosphere through decomposition by soil microbes and other organisms, boreal ecosystems primarily function as a C sink. It is estimated that boreal forests alone contribute around 20 % to the global net forest C sink, helping to regulate the climate by offsetting a considerable portion of human-induced CO_2 emissions (Pan et al., 2011).

Furthermore, the boreal forest landscape is complex, featuring a multitude of interconnected freshwater bodies such as lakes, ponds, rivers, and streams (including ditches), all actively participating in C cycling and balance (Allen and Pavelsky, 2018; Downing and Duarte, 2006). At boreal latitudes (50-70° N), the lateral C export (LCE) must be considered in the C mass balance, with surface waters receiving substantial loads of dissolved organic and inorganic C (DOC and DIC) from adjacent soils (Laudon et al., 2011b; Öquist et al., 2009). As a result, these surface waters are often supersaturated with CO_2 (Raymond et al., 2013) and CH_4 (Bastviken et al., 2011), potent GHGs which are rapidly returned to the atmosphere by physical gas exchange, contributing positively to atmospheric climate forcing.

While roughly 20 % of the C stored in boreal forests is found in forest biomass, a significantly larger proportion, approximately 60 %, is stored in the soils (Pan et al., 2011). The cold and wet conditions typical of the boreal

biome slow down the decomposition of organic material, leading to the accumulation of a significant amount of soil organic C since the last glaciation period (Frolking and Roulet, 2007). Presently, this buried C reservoir accounts for one-third of the global soil organic C, roughly equivalent to the total atmospheric CO_2 pool (200-500 Pg of C) (Bonan et al., 1992). Most of the C in boreal soils is found in waterlogged environments like peatlands, where the anaerobic conditions further limit decomposition activity (Gorham, 1991). Disturbing and mobilizing C stocks in boreal environments through land-use and forest management practices that significantly disrupt natural conditions could have significant impacts for the climate as well as for the C balance in recipient aquatic ecosystems. However, our understanding of how responsive peatlands, and boreal environments at large, are to climate and land-use changes is still limited and requires further investigation.

3.2 Land-use management practices: impacts on aquatic chemistry and carbon balance

3.2.1 Clear-cutting

Clear-cutting alters catchment hydrology by raising the groundwater table (GWT) and increasing runoff discharge (Sørensen et al., 2009). This occurs because tree removal reduces evapotranspiration and increases snow accumulation and melt in open clear-cut areas. Additionally, clear-cutting has been shown to cause more significant and rapid runoff responses during hydrological events (Schelker et al., 2013). Such altered hydrology is recognized as a major driver of increased concentrations of DOC (Laudon et al., 2009; Schelker et al., 2012), particles and nutrients (Nieminen, 2004; Schelker et al., 2016) as well as mercury (Bishop et al., 2009) in affected catchment waters. Moreover, factors such as higher soil temperatures, increased decomposition of organic matter or reduced nutrient uptake by vegetation are found to enhance these effects (Laudon et al., 2009; Liski et al., 1998). Following clear-cutting, logging residues left on site (Figure 2) can further increase soil C availability and in turn the LCE to surface waters (Hazlett et al., 2007; Hyvönen et al., 2000).

While the impact of clear-cutting on C and other solute concentrations (e.g., nitrogen (N) and phosphorus (P)) has been quantified, there is limited knowledge concerning the influence of clear-cutting on dissolved CO_2 concentrations in connecting aquatic systems (e.g. drainage ditches; **Paper I**). Additionally, more research is needed to investigate the combined effects of clear-cutting and forestry practices such as ditching and ditch cleaning, on water chemistry and dissolved GHGs (**Paper II**).

Figure 2. Clear-cut in Trollberget, northern Sweden with some left "tree islands" to preserve flora and fauna habitats (photo credit: Alberto Zannella).

3.2.2 Ditch cleaning

Ditch cleaning (DC) is typically conducted in connection to clear-cutting to lower the GWT and promote aerobic conditions in the upper soil layers for tree establishment. During the DC operations, degraded ditches are cleaned down to their original depth (approx. 1 m) by removing accumulated vegetation and sediment with the aim to restore the drainage capacity (Figure 3). While promoting forest growth (Sikström et al., 2020; Sikström and Hökkä, 2016), DC can have undesirable effects on downstream surface water chemistry (Nieminen et al., 2018). Removal of ditch vegetation and sediments can increase hydrological mobilization of particles by exposing

bare soils to erosion (Joensuu et al., 1999). Additionally, lowering the GWT may alter groundwater flow paths, potentially affecting the chemical composition of runoff (Joensuu et al., 2002; Manninen, 1998; Nieminen et al., 2010). One of the most recognized effects of DC is an increase in the aquatic concentration and export of suspended solids (Joensuu et al., 1999, Finér et al., 2010). In addition, increases in mineral N (Joensuu et al., 2002) and particulate P concentrations as well as elevated pH in downstream waters have been observed following DC (Nieminen et al., 2010). Lowering the GWT with DC can also increase sulfate (SO_4) transport to downstream waters by oxidizing reduced sulfur in the upper soil layers (Fanning et al., 2017). However, the effects of DC on runoff water chemistry in drained boreal peatland forests are poorly understood and the literature often shows divergent results (Nieminen et al., 2018).

While some DC studies have shown substantial (15-30 %) reductions of DOC in runoff (Hansen et al., 2013; Joensuu et al., 2002; Nieminen et al., 2010), other studies (Manninen 1998) have reported no DC-related change in DOC concentrations. A common hypothesis for reduced DOC export from DC areas is that lowering the GWT and altering the flow paths limit the mobilization of easily releasable, recently decomposed organic matter from the top humic layer (Åström et al., 2001). In contrast, negligible changes in DOC export may result from unaltered groundwater paths after DC when the GWT is equally low before DC (Sarkkola et al., 2012). Another concern is that DC could further elevate surface water concentrations of total mercury (THg), together with its bioavailable and highly toxic form methylmercury (MeHg). Generally, forestry can be an important contributor of MeHg in aquatic ecosystems (Eklöf et al., 2016), with estimates suggesting that 9–23 % of Hg in Swedish freshwater fish is linked to clear-cutting (Bishop et al., 2009). Finally, limited information exists on how DC influences GHG concentrations and emissions. Studies show mixed results: Tong et al. (2022) found no significant change in CO_2 and CH_4 emissions within two years of DC, while Rissanen et al. (2023) reported higher CH_4 emissions from moss-free compared to moss-covered ditches. The limited studies and inconsistent results regarding the effects of DC on downstream water chemistry and ditch GHG concentrations and emissions underscore the need for more comprehensive data basis and research (**Paper II**).

Figure 3. Excavator cleaning a ditch in Trollberget, northern Sweden (photo credit: Andreas Palm).

3.2.3 Rewetting

Rewetting unproductive drained wetlands to restore their natural state is typically done by blocking and filling the ditch network to raise the GWT (Landry and Rochefort, 2012; Figure 4). From a climate mitigation perspective, this reduces aerobic peat degradation and mineralization of organic C, thereby increasing C sequestration and lowering CO_2 emissions (Hiraishi et al., 2014). However, the anaerobic conditions produced leads to increased CH_4 emissions—up to 25-30 times (Koskinen et al., 2016)—and, considering the high global warming potential (GWP) of CH_4, there is a clear risk of overestimating the climate benefit of rewetting. Previous assessments of C balance changes following rewetting have focused solely on vertical atmospheric exchange, overlooking potential changes in LCE via runoff (**Paper III**). Peatland ditches link the drainage network to surrounding soils, transporting substantial amounts of terrestrial-derived C downstream. These C inputs are largely driven by hydrology (Billett et al., 2006; Wallin et al., 2015), which can be substantially altered by rewetting, along with the associated C source areas. The few existing studies on the rewetting impact

on C runoff in boreal peatland only focus on DOC, which typically makes up 60-80 % of the LCE (Dinsmore et al., 2013; Leach et al., 2016), while other C forms (DIC and CH_4) are often overlooked, thus preventing a complete assessment of the net ecosystem C balance (NECB).

In addition to the amount of LCE, the impact of rewetting on the age composition of the exported C is a critical but underexplored factor in assessing its effects on peatland C balance and the stability of soil organic C stores. Radiocarbon dating of C from pristine boreal peatlands reveals that runoff is primarily composed of recently fixed C, even though the drained peat itself can be up to 8,000 years old (Campeau et al., 2017). Rewetting, which raises the GWT and alters flow paths, may shift terrestrial C sources and release C of different ages into the ditch network, but this has yet to be proven (**Paper III**).

Another important but scarcely studied aspect of rewetting boreal peatlands is its effects on downstream water quality (**Paper III** and **IV**). A major water quality concern is the brownification of boreal surface waters, driven by rising DOC levels (Evans et al., 2005; Monteith et al., 2007; Roulet and Moore, 2006). Increased DOC and browner waters have chemical, physical, and ecological impacts on aquatic ecosystems and pose challenges for drinking water production (Kritzberg et al., 2020). However, the impact of rewetting drained peatlands on the quantity and characteristics of organic C exported to surface waters remains uncertain, with limited and conflicting literature on the topic. In boreal Finland, peatland rewetting has led to reduced DOC in peat pore water, though surface water color often remained the same or increased (Menberu et al., 2017). Nutrient status also plays a significant role: nutrient-rich fens showed a five-fold increase in runoff DOC during the first year post-rewetting, which then stabilized after five years, while nutrient-poor systems exhibited little change in DOC runoff (Koskinen et al., 2017). Elevated DOC levels in boreal waters, combined with the observed inconsistency in findings, underscore the urgent need for a deeper understanding of rewetting impacts on dissolved C and GHG dynamics at both catchment (**Paper III**) and regional scales (**Paper IV**). This should also account for spatial variability across different climates and wetlands with varying nutrient statuses (**Paper IV**). Closing this knowledge gap would improve the know-how of rewetting as well as the ability to select the best

locations to maximize climate benefits while minimizing negative effects on surface water quality.

Figure 4. Rewetted peatland in Trollberget, northern Sweden. The increase of the groundwater table following rewetting efforts led to the expansion of a pond-like open water area that was covering a much smaller area before the rewetting (photo credit: Alberto Zannella).

4. Methodology

4.1 Study design

This thesis project employed a unique approach to field sampling, which combined two distinct methods: catchment-scale studies (**Paper I** and **III**) and large-scale synoptic surveys (**Paper II** and **IV**) of forest ditches across various regions in Sweden (Figure 5). By adopting this study design, my aim was to gain a comprehensive spatial understanding of the effects of certain land-use practices on aquatic C and dissolved GHG, while also investigating the finer details of small-scale processes and their temporal variability.

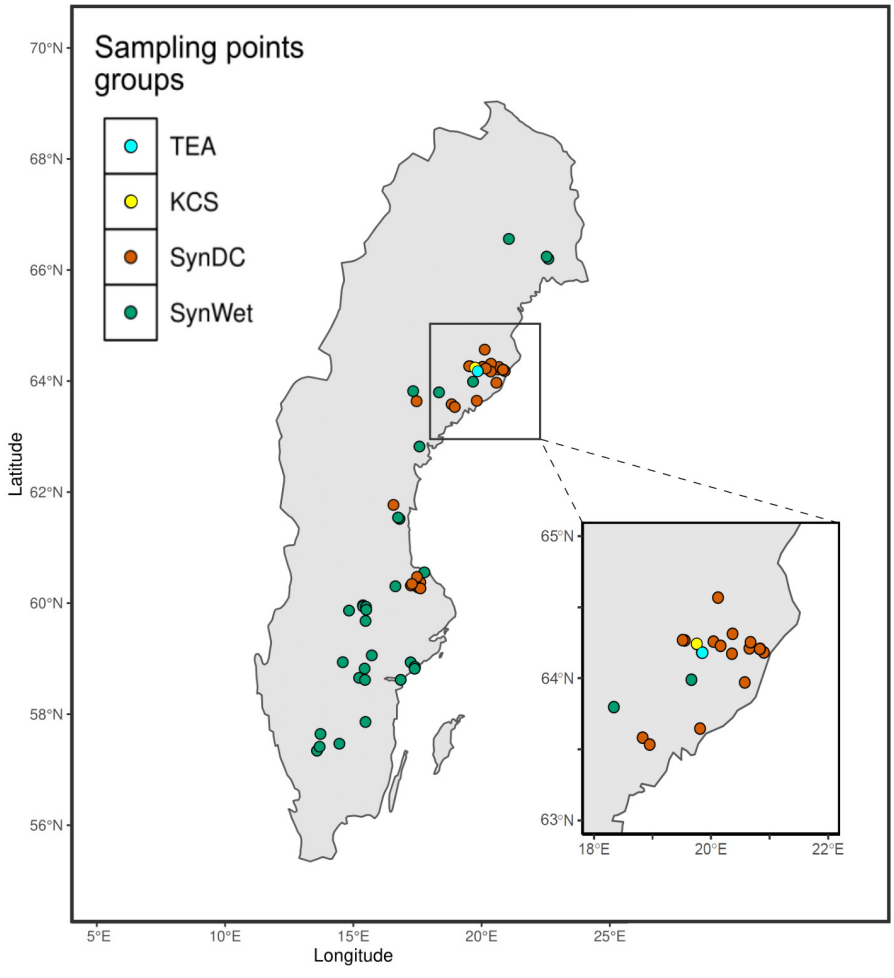

Figure 5. Location of sample sites from all papers included in this thesis—the detailed catchment-scale studies (**Paper I** and **III**) conducted at Trollberget Experimental Area (TEA) and Krycklan Catchment Study (KCS) and the two regional sampling campaigns of DC sites (SynDC, **Paper II**) and rewetted sites (SynWet, **Paper IV**). For the synoptical surveys, only cleaned or rewetted sampling points are illustrated as their control sites would overlap on the map.

4.2 Catchment-scale studies at Trollberget Experimental Area

The catchment-scale studies of **Paper I** and **Paper III** were conducted in the Trollberget Experimental Area (TEA), a state-of-the-art site for field-based research located 50 km northwest of Umeå (64°10' N, 19°51' E), in the Swedish boreal biome (Figure 5). TEA was established in 2018 as part of the Krycklan Catchment Study (KCS, www.slu.se/Krycklan, Laudon et al., 2021) to study the impacts of forestry practices and peatland rewetting on freshwater environments. In this thesis, three sub-catchments within TEA were studied (Figure 6): one that was clear-cut (DC2, **Paper I**) and two peatland areas that were rewetted (R1 and R2, **Paper III**). Detailed information on the TEA catchments can be found in Laudon et al. (2023).

The forest at DC2 was clear-cut in summer 2020 (20 July–24 August). Whilst clear-cutting, only trunk wood was collected, leaving branches and tops on the ground to reduce damage during transport. A forested sub-catchment (C2) located 10 km from DC2 in KCS was used as an unmanaged control site for comparative analysis (Laudon et al., 2013). The two sub-catchments have similar soil and forest types. Soils are primarily composed of podzols with pockets of histosols and the forest is mainly dominated by Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*), with an understory of *Vaccinium myrtillus*, *Vaccinium vitis-idaea*, and *Deschampsia flexuosa* (Leith et al., 2015). Both DC2 and C2 were impacted by drainage from historical ditching activities in the early 20th century.

The rewetting of R1 and R2 took place in November 2020 and was concluded on the 30th. Before rewetting, the sparse tree population was harvested and removed. Some of the felled trunks were reutilized along with peat as backfill material for blocking the ditches to rewet the peatland. Two natural peatlands, C4 within KCS and C18 in the nearby Kulbäcksliden Research Infrastructure (KRI, Noumonvi et al., 2023), were used as controls for R1 and R2. The peatlands of KCS and KRI, as the one within TEA, are characterized as oligotrophic minerogenic mires, predominantly covered by *Sphagnum spp.*, with sparse sedges, dwarf shrubs and occasional slow-growing Scots pine (*Pinus sylvestris*). Soil types are predominantly humic podzols, although humo-ferric podzols are found in drier areas and histosols in wetter areas.

The climate is typical of the northern boreal zone, with short, cool, and bright summers followed by long, cold, and dark winters. Snow covers the catchments for an average of 167 days per year, usually from late October to early May. The mean annual air temperature is 2.1°C, with an average annual precipitation of 614 mm per year (30-year average from SMHI, 1986–2015).

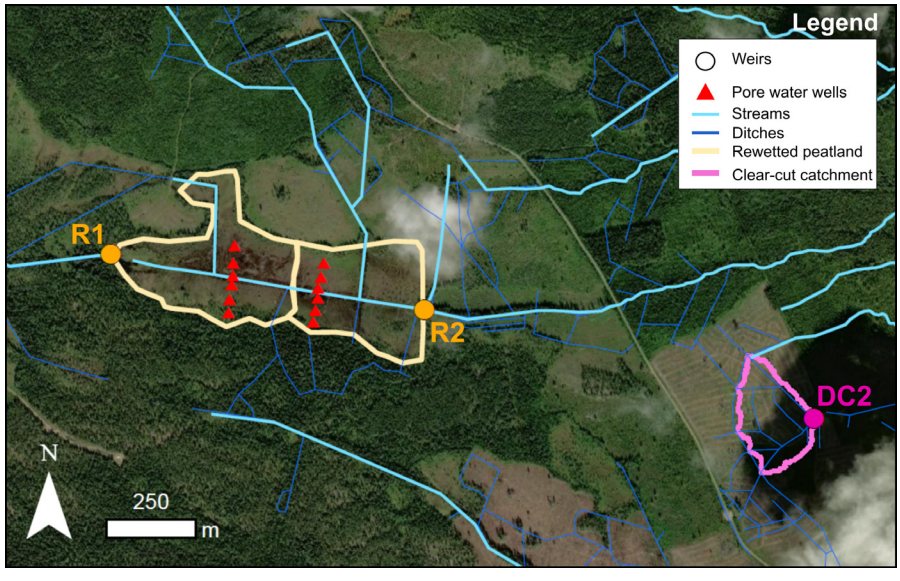

Figure 6. Map of Trollberget Experimental Area (TEA) with the studied sites: the two peatland areas that were rewetted (R1 and R2), and the clear-cut catchment (DC2).

4.2.1 Sampling program and data collection

The sampling program for water chemistry analyses started in the winter of 2019 at TEA, while it is part of a long-term monitoring effort started decades ago at the reference sites of KCS and KRI. On each sampling occasion, all watercourses were sampled synoptically (± two days). Sampling regularly occurred every second week during the growing season, once per month during winter base-flow conditions, and twice per week during high flow periods such as spring flood, or other important hydrological events and forestry operations. Samples were collected upstream of V-notch weirs installed at each site following sampling and analytical procedures from qualified protocols used at KCS and KRI by the SLU Biogeochemical Analyses Laboratory (BAL), Umeå (Laudon et al., 2021; Noumonvi et al.,

2023). Pore water (0–1 m) sampling was also conducted in two transects of the mire in Trollberget—one in R1 and one in R2—distributed perpendicular to the main ditch channel (Figure 6). Each transect consisted of six pore water wells installed 4, 10 and 20 m from each side of the ditch. It was conducted at four occasions, one before (09/2020) and three after rewetting (06/2021, 10/2021 and 10/2022).

4.2.2 Aquatic carbon chemistry and radiocarbon analysis

Water samples for DOC analysis were collected in acid-washed high-density polyethylene bottles and kept dark and cool during transport and storage. Samples were filtered (0.45 µm mixed cellulose ester syringe filters, Millipore®) within 24 h and kept refrigerated at 4°C until analysis (< 7 days after filtration). DOC analysis consisted of acidification of the sample to remove inorganic C, followed by combustion using a Shimadzu TOC-VCPH (Laudon et al., 2011b). Samples for DIC and CH_4 were collected using a 10 mL syringe and injecting 5 mL of ditch water into a sealed 22.5 mL glass vial. Vials were evacuated prior to sampling, filled with N_2 at atmospheric pressure and prefilled with 0.1 ml 85 % H_3PO_4 to shift the carbonate equilibrium toward CO_2. Headspace CO_2 and CH_4 concentrations were analyzed on a gas chromatograph equipped with a methanizer and flame ionization detector (GC-FID). *In-situ* ditch concentrations of DIC and CH_4 were calculated from headspace concentrations using water and headspace volumes and temperature-dependent equations. For more details on DIC and CH_4 sampling and analysis see Wallin et al. (2010, 2014).

Additionally, water samples from the peatland sites were collected for radiocarbon analysis of DOC (^{14}C-DOC) on 10 occasions: three before and seven after rewetting (**Paper III**). Grab samples were taken in 500 mL acid-washed plastic bottles, pre-rinsed with sample water. They were filtered the same day through pre-baked glass-fiber filters (GF/F, 0.7 µm) and kept dark and cold until shipped to the National Environmental Isotope Facility Radiocarbon Laboratory (East Kilbride, UK). There, samples were freeze-dried, acid-fumigated, and combusted to CO_2 using the sealed quartz tube method (Ascough et al., 2024). The CO_2 was cryogenically purified and split: one part was analyzed for $\delta^{13}C$ using isotope ratio mass spectrometry (Delta V, Thermo-Fisher) to normalize ^{14}C results to a $\delta^{13}C$ = -25 ‰, while the second part was converted to graphite and measured for ^{14}C content using

accelerator mass spectrometry (AMS) at the Scottish Universities Environmental Research Centre. ^{14}C results are reported as % modern C and conventional radiocarbon years (years BP, where 0 BP = AD 1950).

4.2.3 Discharge measurements and carbon export estimates

In each catchment outlet of the TEA, water level was continuously calculated upstream of the weirs using pressure transducers (Expert 3400, MJK A/S, Denmark) connected to data loggers (Solinst Levelogger 5). Frequent manual measurements were also taken (monthly during winter and at least bi-weekly during the rest of the year) to calibrate the automatic water level data. A correlation between manual water level and flow measurements was used to determine continuous discharge data from logger readings (Karlsen et al. 2016). Specific discharge, defined as discharge per unit area, was calculated for each studied catchment (**Paper I** and **III**). Daily aquatic C exports—expressed in g C m^{-2} d^{-1} or g CO_2-eq m^{-2} d^{-1} (IPCC, 2021)—were estimated by multiplying mean daily discharge by daily concentration of each C component (**Paper III**). Full time series of daily aquatic C concentrations were derived using linear interpolation between sampled values. The daily exports were also summed per year and divided by catchment area.

4.2.4 High-frequency measurements

In addition to the regular sampling program described above, I measured at high-frequency CO_2 concentration, water temperature, electrical conductivity (EC) and water level in the outlet of the clear-cut DC2 catchment for a full growing season (May–October 2021, n=174 days) (**Paper I**). CO_2 concentration was measured with an eosGP sensor. Water temperature and EC were monitored using a thermocouple (Type T) and a CS547A-L conductivity sensor, respectively. Sensors were connected to a CR1000X data logger recording at 1 min intervals and storing mean values at 30 min resolution. Water level was recorded every 60 min using a capacitance sensor (TruTrack Logger Type WT-HR 64K). Moreover, precipitation, air temperature, and short-wave radiation were measured in a nearby sub-catchment within TEA. Stream CO_2 concentrations are also measured hourly in the forest stream site C2 of the KCS and were used for

comparative analysis. Measurements are performed with a Vaisala CARBOCAP GMP221 NDIR sensor with water levels recorded at 5 min resolution. More details are included in **Paper I**.

4.3 Regional samplings

The impacts of ditch cleaning and rewetting on ditch water chemistry and dissolved GHG concentrations were studied in synoptic sampling campaigns (Figure 5) at the regional scale in **Paper II** and **Paper IV**, respectively.

4.3.1 Regional ditch cleaning study

Sites selection and description

The regional DC study (**Paper II**) was conducted in eastern central Sweden, covering 600 km from north to south (Figure 5). Using GIS data from forestry companies, 25 ditches cleaned within the past one to four years were selected. Only ditches in forested areas were chosen, excluding those in agricultural or urban settings. Each cleaned ditch had a corresponding reference (REF) ditch that was not cleaned. REF sites were identified through GIS analysis from nearby yet hydrologically disconnected catchments, and were finally selected during field visits, ensuring the paired ditches matched in size, vegetation, peatland coverage, and showed no signs of recent ditching. Ultimately, 25 pairs were sampled: 25 cleaned ditches and 25 not cleaned ditches. As the study also aimed to assess the impact of DC in both forested and clear-cut areas, out of the 25 pairs, 13 were located in forested areas and 12 in clear-cut areas. A detailed description of the paired catchments is found in **Paper II**. It shows that the 25 cleaned areas average 37 ha (2–136 ha), while the 25 reference areas average 44 ha (5–176 ha). They predominantly consist of coniferous forest (92 % in cleaned areas, 89 % in reference areas) and till soils (63 % in cleaned areas, 58 % in reference areas).

Sampling and water chemistry analyses

Water samples were collected from the 50 ditches during three field campaigns: June 2021, September 2021, and June 2022. Each campaign lasted 2–3 weeks and was conducted during stable hydrological (baseflow) conditions, e.g. avoiding sampling during spring floods. Sampling started in the South in June and moved north to ensure consistent seasonal conditions. In September, it occurred in the opposite direction due to earlier autumn onset in the North. To minimize the impact of discharge variations on water chemistry, paired areas were sampled on the same day or within a day's interval. Water samples were analyzed for a wide range of water chemistry variables. General water chemistry, including total organic carbon (TOC), pH, nutrients, cations, and anions, was analyzed using accredited methods at the geochemical laboratory at SLU, Department of Aquatic Sciences and Assessment. Mercury samples—both THg and MeHg—were also taken, using an ultra-clean sampling protocol. Single use plastic gloves were used, and water samples were collected in acid-washed Teflon bottles that were sent urgently on the same day or the day after to the Swedish Environmental Research Institute (IVL) for immediate preservation before analysis. For further analytical details, see **Paper II**.

4.3.2 Regional rewetting study

Sites selection and description

A second synoptic survey was conducted sampling rewetted and non-rewetted but drained sites across Sweden (**Paper IV**). Rewetted forest areas were identified through collaboration with county administrations, municipalities, and private landowners. These sites were selected to represent a diverse geographical (latitudinal range 63°–74°N) and climatic range (mean annual temperatures 1–8°C) within Sweden. The rewetted sites were chosen based on specific criteria: (i) they are situated in forested landscapes with minimal agricultural or urban influence (<5 % coverage), (ii) have catchment areas <10 km², and (iii) were historically drained. Furthermore, the wetlands vary in characteristics such as nutrient status, peat content, and presence of open water. Each rewetted site was matched with a nearby drained and non-rewetted site for comparative analysis using a paired

design. The reference drained sites were selected based on similar catchment size, soil type, and land cover, typically within a 5 km radius, to ensure comparable hydrometeorological conditions while draining different catchments. In total, 33 rewetted and 33 drained sites were studied. The rewetting occurred between 2010 and 2022; however, most sites (n=25) were rewetted within four years from sampling. GIS analysis (see **Paper IV** for details) showed that catchment sizes ranged 1-1075 ha, with average sizes <100 ha. Drained catchments were slightly larger (averaging 99 ha) than rewetted ones (averaging 65 ha). Both rewetted and drained sites shared similar characteristics, including approximately 60–70 % forest cover (mostly coniferous), around 15 % wetland, and 10–20 % clear-cut areas. Soil types varied among sites but averaged around 65 % mineral and 35 % peat soils, with similar distributions between rewetted and drained references.

Sampling and analyses

Water samples were collected from the outlets of rewetted and drained sites during spring and autumn 2022, following a latitudinal design that followed the timing of seasonal changes across Sweden. Spring sampling lasted three weeks (May 24–June 15) and was conducted from south to north, while autumn sampling took two weeks from north to south (October 4–15). In the south, dry conditions prevented sample collection from eight site pairs, resulting in 25 sites sampled during autumn.

Grab samples were collected using low-density polyethylene bottles rinsed beforehand and analyzed for TOC, absorbance at 420 nm (Abs_{420}) and for a large set of additional chemical variables (nutrients, cations, anions, etc.) as in **Paper II** (§ 4.3.1). All samples were ensured to be kept dark and cold before regularly being shipped (every second/third day) to SLU geochemical laboratory for analysis. Dissolved gas samples were also collected using a headspace equilibration method (Hope et al., 2004), where 30 mL of water and 30 mL of ambient air were drawn into a 60 mL polypropylene syringe equipped with a directional flow valve that was shaken vigorously for one minute to reach equilibration. Then, 15 mL of the equilibrated air was transferred into 22 mL evacuated exetainer vials (Wilkinson et al., 2018) that were analyzed for CO_2 and CH_4 using a GHG analyzer (Gas scouter, Picarro) with a closed-loop system. For more detailed analytical information see **Paper IV**.

During this campaign, additional samples for analysis of the stable isotopic composition of DIC (δ^{13}C-DIC) were collected. A volume of 2 mL of sample water was collected in a syringe and injected into 12 ml septum-sealed glass vials (Labco Limited) pre-filled with N_2 gas, and pre-injected with phosphoric acid in order to convert all DIC species to CO_2. δ^{13}C-DIC was analyzed using a Gasbench II and a Thermo Fisher Delta V mass spectrometer at the SLU Stable Isotope Lab (SSIL), Umeå. The δ^{13}C values are reported in terms of deviation from the standard Vienna Pee-Dee Belemnite (VPDB). The repeated measurements of the standard indicated a standard deviation <0.2 ‰ on each sampling occasion.

4.4 Data analysis

In **Paper I**, linear regression analysis was used to identify significant relationships ($p < 0.05$) between ditch CO_2 concentration (or diel amplitude) and discharge, water temperature, or shortwave radiation. The response of C concentrations to discharge variations was analyzed using concentration-discharge (C-Q) relationships to assess hydrological control variations related to seasonal changes. The slope values obtained from the C-Q regressions were interpreted as in Meybeck and Moatar (2012). Briefly, sites were considered "source limited" when the slope < -0.2, "chemostatic" when the slope ranged between -0.2 and 0.2, and "transport limited" when the slope was ≥ 0.2. In addition, the hydrological control on ditch CO_2 was analyzed using CO_2-Q hysteresis loops (Evans and Davies, 1998) of hydrological events that were identified with the method of Lannergård et al. (2021). Hysteresis indexes were calculated (as in Lloyd et al., 2016) and combined with characteristics of each event and environmental factors. A principal component analysis (PCA) was then performed on the entire event dataset to examine and assess the different temporal controls on ditch CO_2 dynamics.

In **Paper II**, the impact of DC on water chemistry and dissolved GHGs was assessed by calculating the difference (Δ) within each DC-REF variable pair on each sampling occasion and testing significant deviation from zero ($p < 0.05$) using site pair identity and sampling occasion as random factors in a generalized linear mixed model (GLMM). For this approach, all response variable values are first Log-transformed to achieve normal distribution. This analysis was conducted on the entire dataset (n=25), as well

as separately for forested (n=13) and clear-cut (n=12) sites. Correlations between the Δ of variables were further analyzed using a multiple Spearman's rank correlation test. Only significant correlations with $|\rho| > 0.5$ were considered, and only for variables influenced by DC according to the results of the GLMMs.

In **Paper III**, GLMMs were used to test significant deviations from zero for the Δ in runoff, pore water, and C age concentrations before and after rewetting between the rewetted sites and the mean of the controls. In the GLMMs, the pre- and post-rewetting periods were treated as a fixed factor, and a repeated structure (date) was incorporated to accounted for multiple samples at each site. Moreover, similarly to **Paper I**, C-Q relationships were analyzed to assess hydrological control variations related, in this case, to rewetting. Specifically, ANCOVAs were used to test significant changes in C-Q slopes before and after rewetting, using Log-Q as the covariate and treatment time (i.e., before or after treatment) as a categorical factor.

In **Paper IV** the Δ in log-transformed concentration of C and dissolved GHG for the paired sites (rewetted-drained) were tested for significant deviations from zero using GLMMs first on the entire dataset and then divided by season. When data from both seasons were used, sampling occasion and site pair were added as random factors. Significant relationships ($p < 0.05$) between Δ concentrations of studied C forms and latitude or nutrient status (using C/N ratio as proxy) were explored using linear regressions.

All data and statistical analyses were performed using JMP Pro 15 to 17 (SAS Institute Inc., Cary, NC, USA).

5. Results and discussion

5.1 Ditch CO_2 dynamics in a clear-cut (Paper I)

Mean CO_2 concentration (2.47 mg C L^{-1}, Figure 7) in the studied boreal clear-cut ditch (DC2, Figure 6) was higher compared to other boreal and temperate forest headwaters (Crawford et al., 2017; Dinsmore et al., 2013). The concentration dynamics (Figure 7) showed a seasonal pattern, with CO_2 peaking in summer, which closely correlated with water temperature, suggesting ecosystem respiration as the main source of CO_2 (Yvon-Durocher et al., 2012). Removal of shading trees after clear-cutting raises soil temperatures, which strongly affects respiration, and in turn increases microbial mineralization of organic matter and thus CO_2 production (Schelker et al., 2013). In addition to a seasonal pattern, distinct daily CO_2 cycles were identified, with day-night differences that were particularly pronounced in June-July and decreased in the autumn (Figure 7). These daily CO_2 amplitudes (mean and max Δ: 0.35 and 1.11 mg C L^{-1}) were higher than those observed in similar studies of low-productive arctic and alpine streams (Rocher-Ros et al., 2020; Peter et al., 2014), and only lower than nutrient-rich agricultural streams (Wallin et al., 2020). The temporal control on ditch CO_2 changed after clear-cutting due to elevated DOC and nutrient concentrations (Schelker et al., 2016, 2012), as well as higher temperature and light availability. The daily CO_2 cycles in my study were likely driven by aquatic primary production, which consumes CO_2 during daytime (Gómez-Gener et al., 2021; Rocher-Ros et al., 2021, 2020). These dynamics are typical of open canopy systems, where increased light availability drives primary production (Crawford et al., 2017; Gómez-Gener et al., 2021).

Discharge variation typically regulates ditch CO_2 dynamics in boreal regions by driving the hydrological export of CO_2 from catchment soils, although relationships may differ between sites (Striegl and Michmerhuizen, 1998; Riml et al., 2019). In my study, the slopes of the monthly CO_2-Q relationships suggested a chemostatic system response, indicating a limited hydrological influence on ditch CO_2. This could imply a stable terrestrial CO_2 source and/or non-hydrological processes counteracting the influence of discharge variations on CO_2 (Rehn et al., 2023). The lack of clear CO_2 responses to most identified hydrological events suggests that light- and temperature-induced metabolic processes may dominate over discharge-induced effects. Finally, the importance of *in-situ* metabolic processes in regulating CO_2 dynamics was supported by the event-based and control-integrated PCA analysis. Both CO_2 concentration and diel amplitude closely correlated with daily total shortwave radiation (SR) and mean water temperature, while hydrological metrics showed minimal influence on CO_2.

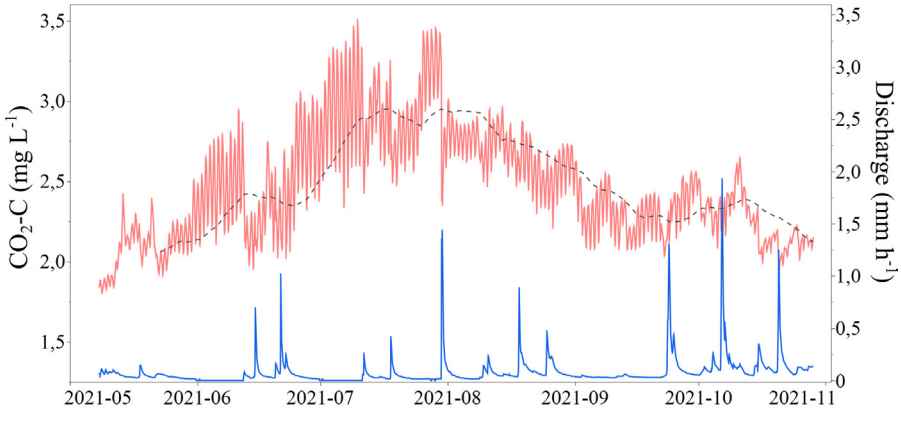

Figure 7. High-resolution monitoring (hourly) time series of dissolved CO_2 (mg L^{-1}; in red) and discharge (mm h^{-1}; in blue) for the study period (growing season) from May 8 to October 28, 2021, in the clear-cut ditch at DC2, TEA. The black dotted line shows the 15-day moving average.

The comparison of continuous high-res data with the nearby forested catchment (C2 in KCS) revealed clear differences in post-clear-cut dynamics (Figure 8). These differences, seen in diel CO_2 concentration and amplitude (Figure 8A) as well as in water temperature patterns (Figure 8B), suggested altered *in-situ* ecosystem function due to increased incoming SR, elevated DOC (46 vs 22 mg C L^{-1}, in DC2 vs C2, respectively) and nutrient levels post-clear-cutting (NH_4-N: 115 vs 8 µg L^{-1}; NO_3-N: 40 vs 5 µg L^{-1}, in DC2 vs C2 respectively). This adds further support that *in-situ* metabolism is an important driver for CO_2 dynamics in clear-cut ditches.

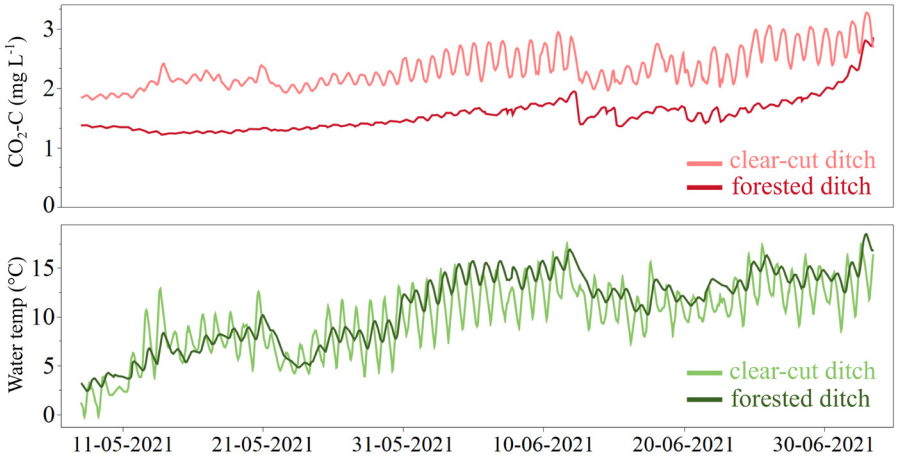

Figure 8. Hourly time series of CO_2 (panel A) and water temperature (panel B) for the clear-cut ditch at DC2 and the reference forested ditch at C2, KCS, during the period from May 8 to July 5, 2021 (2 months).

5.2 Effects of ditch cleaning on water chemistry and greenhouse gases in boreal ditches (Paper II)

Ditches in DC sites had higher pH than in REF sites, along with elevated calcium (Ca) and SO_4 concentrations (Table 1), suggesting that lowered GWTs directed the groundwater flow paths through the deeper mineral soil layers rich in weathering products (i.e., base cations) before feeding into the ditches (Ledesma et al., 2016; Ukonmaanaho et al., 2014). pH would typically increase after DC due to decreased DOC inputs from the more superficial organic-rich soil strata (Joensuu et al., 2002; Nieminen et al.,

2018, 2010) and the removal of vegetation and accumulated organic material from the ditch (Hansen et al., 2013). However, TOC concentrations in my study showed no differences between DC and REF ditches (Table 1), although the observation of a significant negative correlation between ΔTOC and ΔpH suggests that changes in TOC after DC were still linked to the observed pH increase.

Moreover, while DC can increase THg in runoff (Wesström et al., 2017), my study found no differences in THg levels between DC and REF ditches (Table 1). However, ditches in forested DC sites had lower MeHg concentrations compared to forested REF sites (Table 1). Lowering the GWT via DC may reduce the mobilization of Hg, which is often concentrated in the topsoil (Bishop et al., 2020), and decrease MeHg formation in waterlogged areas, thereby reducing its export to connected ditches. Surprisingly, no differences in MeHg levels between DC and REF ditches was observed in clear-cut areas, possibly as the increase in GWT from reduced transpiration, due to tree removal, counteracts the decrease in GWT due to DC.

Boreal ditches are recognized as GHGs hotspots (Audet et al., 2020; Wallin et al., 2018), but the impact of DC on their concentrations is currently poorly explored. In my study, dissolved CO_2 was lower in DC than REF ditches (Table 1). Since CO_2 is part of the DIC pool, and its speciation is closely linked to pH (Stumm and Morgan, 1996), the pH increase following DC likely shifted the DIC speciation toward carbonate forms (CO_3^{2-}), which may explain the lower CO_2 levels in the cleaned ditches. Notably, lower ditch CO_2 concentrations were found in forested, but not in clear-cut DC sites (Table 1). This difference may be due to increased mineralization of organic matter in the clear-cut areas (Nieminen, 2004; Schelker et al., 2016), which could counterbalance the CO_2 reduction that was instead observed in forested sites. This is further supported by the findings from **Paper I**, which showed higher CO_2 in clear-cut compared to forested ditches. Furthermore, ditch CH_4 concentrations did not differ between DC and REF sites (Table 1). Nonetheless, a correlation between ΔCH_4 and ΔCO_2 suggests both these gases likely originate from similar sources or metabolic processes (Campeau et al., 2018), and that the lowering of the GWT after DC may similarly influence the formation and mobilization of both GHGs. Finally, ditch nitrous dioxide (N_2O) concentrations were higher in DC compared to REF

sites (Table 1). DC operations, particularly when combined with clear-cutting, result in significant nutrient losses—either as particulate matter or in dissolved forms like ammonia (NH_4) and nitrate (NO_3)—through the ditch network (Åström et al., 2004). This, along with enhanced oxygen availability in riparian soils after DC, promotes denitrification and N_2O production (Beaulieu et al., 2011). While no direct effect of DC on NO_3 was observed, a strong positive correlation between ΔN_2O and ΔNO_3 suggests a potential indirect impact of N availability on N_2O production.

Table 1. Simplified table from **Paper II**: Water chemistry and dissolved GHG data in ditch waters collected from the 25 ditch cleaned (DC) and 25 reference (REF) sites. Data are presented as mean values based on all sampling occasions, while the test of significance is based on GLMMs analysis, which tests the difference between paired sites. The dataset was further sub-grouped dependent on whether the DC operations were conducted in forested areas (13 pairs) or clear-cut areas (12 pairs). Arrows are used to highlight when Δ-values for each pair are significant different from zero according to the GLMMs ($p < 0.05$): ↑ indicates significantly higher values in the DC sites compared to their REF sites, while ↓ indicates significantly lower values in the DC sites compared to their REF.

		All		Forested		Clear-cut	
		DC	REF	DC	REF	DC	REF
Acidity and Ions	pH (-)	↑ **5.8**	5.2	5.9	5.3	5.7	5.2
	SO_4^{2-} (μeq L^{-1})	↑ **211.3**	105.4	215	151.5	↑ **207.1**	53.8
	Ca^{2+} (μeq L^{-1})	↑ **808.6**	600.1	↑ **563.8**	440.3	1082.1	778.8
	K^+ (μeq L^{-1})	19.8	22.9	↑ **16.2**	10.5	13.9	36.8
Mercury	THg (ng L^{-1})	9.82	9.37	7.15	6.36	12.81	12.74
	MeHg (ng L^{-1})	0.76	0.63	↓ **0.63**	0.71	0.89	0.53
Carbon and GHGs	TOC (mg L^{-1})	38.8	39.5	28	26	50.9	54.6
	CO_2-C (mg L^{-1})	↓ **2.3**	3.9	↓ **1.7**	4	3.1	3.8
	CH_4-C (μg L^{-1})	9	32.4	6.7	18.0	11.6	48.9
	N_2O-N (μg L^{-1})	↑ **2.3**	0.8	1.1	0.5	3.6	1.1
Nutrients	NO_3-N (μeq L^{-1})	18.4	4.3	4.6	1.4	33.9	7.5

5.3 Changes in aquatic carbon following rewetting of a nutrient-poor northern peatland (Paper III)

The results suggest that the rewetting-induced increase in annual LCE was primarily driven by increases in runoff C concentrations. As mentioned above (§ 3.2.3), there is limited and contrasting literature describing how rewetting nutrient-poor peatlands affects total runoff C (including DOC, DIC, and CH_4) in boreal regions. In the current study, DOC in pore water decreased in R1 after rewetting and showed a similar trend in R2, likely due to dilution from catchment water inputs with lower DOC. On the other hand, ΔDOC concentrations in the outlets of R1 and R2 increased after rewetting. The mismatching pattern between ditch and pore water concentrations observed after rewetting was attributed to changes in groundwater flow paths and connectivity between peat soils and ditches. I here suggest that rewetting shifted the system from a high and evenly distributed terrestrial-aquatic connectivity due to a dense ditch network to one where connectivity converged, both horizontally and vertically, toward a newly created ditch initiation point near the catchment outlets (Figure 9). As a result, water travelled longer distances through the peat and potentially through different soil layers, resulting in altered chemical signals in the runoff (Laudon and Sponseller, 2018).

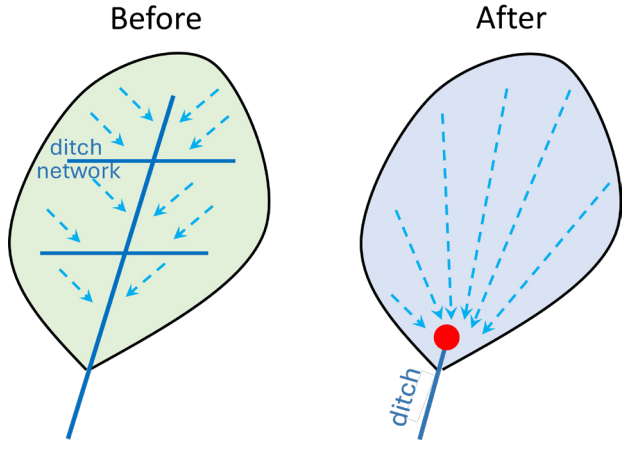

Figure 9. Conceptual figure of altered terrestrial-aquatic connectivity with rewetting. The arrows represent the catchment water flowing through the peatland towards the ditch network (before rewetting) and to a newly created ditch initiation point (red dot) near the catchment outlet (after rewetting).

Higher runoff ΔDIC concentrations were observed after rewetting at R1 (Figure 10), where the expansion of a pond upstream of the outlet (Figure 4) likely contributed to the elevated DIC levels. Peatland ponds, having favorable conditions for microbial and photochemical processes, are hotspots for GHG formation, including CO_2 (DIC) and CH_4 (Arsenault et al., 2024; Dean et al., 2024). This is further supported by the massive 990 % increase in runoff ΔCH_4 concentrations at R1 after rewetting, which was instead not seen at R2. In addition, shallower GWTs promote anoxic conditions, stimulating CH_4 production and emission (Koskinen et al., 2016; Laudon et al., 2023).

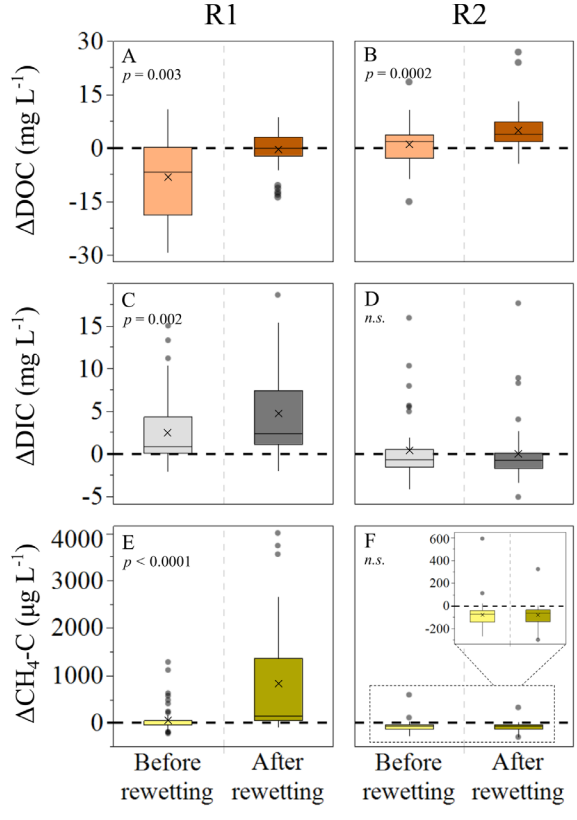

Figure 10. Differences (Δ) between the treatment and control sites before and after rewetting for runoff concentrations of DOC, DIC and CH_4 at R1 and R2. The Δ was calculated as treatment minus control, thus positive values indicate higher C concentration at treatment site compared to the control site, while negative values indicate the opposite. Mean Δ values are shown with a "×". Statistically significant differences from the GLMMs are reported with corresponding *p*-values or "*n.s.*" if not significant.

The LCE significantly increased at R1 in response to rewetting, while it remained largely unchanged at R2 (Figure 11). In the two years after rewetting, average LCE at R1 was lower than typical export rates in pristine boreal and temperate peatlands (Billett et al., 2004; Dawson et al., 2002) but aligned with the values typically observed in the controls of my study (Leach et al., 2016; Wallin et al., 2013). DOC was the major contributor to LCE across all catchments. After rewetting, DOC export increased by 68 % at R1, which was comparable with the results of Koskinen et al. (2017) for rewetted oligotrophic peatlands, but still lower than the increases they observed in more fertile rewetted peatlands. DIC export at R1 rose by 120 %, a significant increase compared to the controls (~20 %). The DIC export at R1 (~3.0 g C m^2 y^{-1}) was relatively high for peatland streams (Rantakari et al., 2010; Rehn et al., 2023). After rewetting, CH_4 export decreased by ~60 % at R2 but increased markedly by ~770 % at R1. Post-rewetting CH_4 export at R1 was an order of magnitude higher than at R2 and the control sites, with concentrations (mean and max: 997 and 4184 µg C L^{-1}) reaching the upper range globally for surface waters (Stanley et al., 2023).

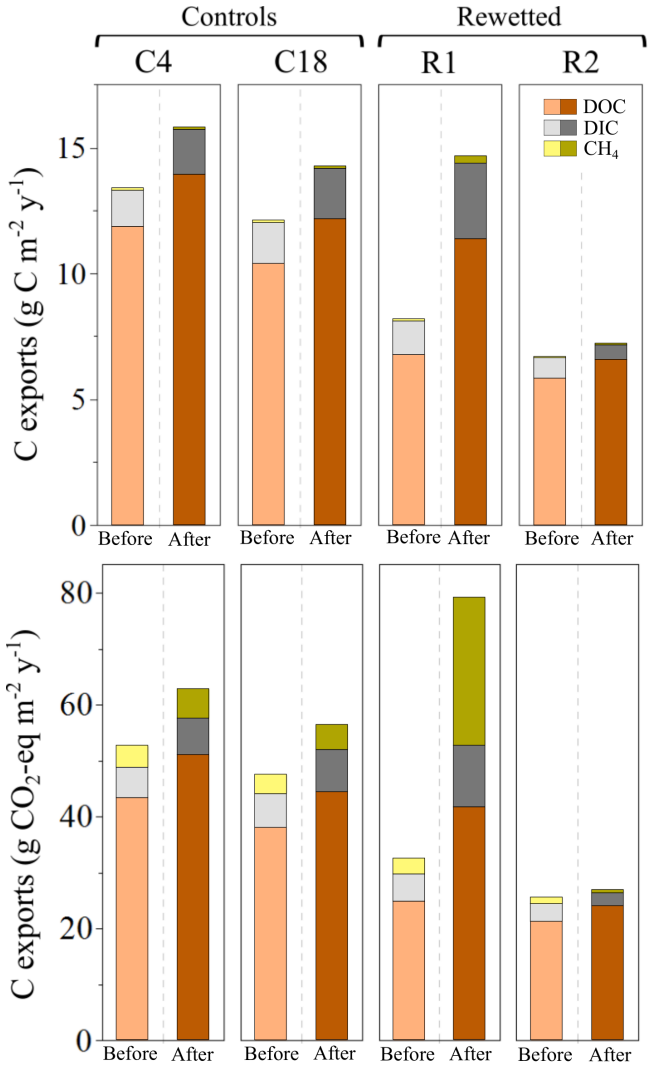

Figure 11. Yearly export of DOC, DIC and CH_4 from the control sites (C4, C18) and rewetted sites (R1, R2) before and after rewetting based on averages of the two respective years. Upper panel shows yearly exports in g C m^{-2} y^{-1} and lower panel shows the yearly exports expressed in CO_2-equivalents (g CO_2-eq m^{-2} y^{-1}).

The slopes of the C-Q relationships for DOC, DIC, and CH_4 did not change before and after rewetting across all sites, suggesting that rewetting did not affect the hydrological source control. DOC-Q relationships were chemostatic, suggesting a stable DOC source weakly related to hydrological variations (Fork et al., 2020). This stability may be due to increased runoff from superficial peat layers with lower (diluted) DOC after rewetting. In contrast, the C-Q patterns for DIC and CH_4 were source-limited and indicated that their production cannot keep pace with increased mobilization at higher discharge rates. Although C-Q slopes were unchanged before and after rewetting, the offsets between the regression lines increased, particularly at R1, indicating higher baseflow concentrations. The increased baseflow DIC and CH_4 concentrations at R1 after rewetting points again towards the increased importance of the pond upstream the outlet as the main source.

Radiocarbon analysis showed that the exported DOC from both rewetted and control catchments was predominantly modern, and consistent with other boreal peatlands (Billett et al., 2012; Campeau et al., 2017). After rewetting, R2 showed an increase in ^{14}C-DOC content, indicating greater mobilization of recently fixed C from shallow and hydrologically connected peat layers. However, a similar change was also observed at one control site, suggesting that intra-annual variability could partly explain the shift at R2. In contrast, no significant change in ^{14}C-DOC was observed at R1.

5.4 Enhanced carbon in runoff following rewetting of drained forest wetlands (Paper IV)

Mean TOC concentration was higher in rewetted (51.4 mg L^{-1}) compared to drained sites (30.4 mg L^{-1}), both when analyzing the entire dataset and when considering spring and autumn data separately (Figure 12). A similar pattern, with enhanced runoff DOC following rewetting, has been observed in local studies in boreal Sweden (e.g. Laudon et al. 2023) and Finland (e.g. Koskinen et al. 2017; Menberu et al. 2017) but this is among the first studies to assess rewetting effects on aquatic C at such large geographical scale. Rewetted sites had also higher values of Abs_{420} than drained sites, indicating browner runoff and thus potential implications for downstream water quality. Mean CO_2 concentration was higher in rewetted sites (6.1 mg C L^{-1}) compared to drained sites (3.8 mg C L^{-1}), across both seasons and also when

considered separately (Figure 12). Mean CH_4 concentration was higher in rewetted sites (352 µg C L^{-1}) than in drained sites (110 µg C L^{-1}), though this difference was significant only in the spring (Figure 12). The shallower GWTs at the rewetted sites likely promoted anoxic conditions that stimulated CH_4 production and in turn higher runoff concentrations (Koskinen et al., 2016). In addition, it could be claimed that the clear and high differences in concentration between rewetted and drained sites—ranging from 60 % to 320 % higher depending on C form—are likely to correspond to an increase in the annual LCE, as seen in **Paper III**.

The δ^{13}C-DIC values showed no difference between rewetted (-20.0 ‰) and drained sites (-19.6 ‰), regardless of season (Figure 12), and suggested a predominant biogenic source of the DIC pool (Campeau et al., 2017), which mainly consists of CO_2 in such low pH (mean pH < 5.5) environments (Wallin et al., 2010). In addition, this similarity in δ^{13}C-DIC values suggested that rewetting did not change the metabolic pathways that sustain runoff CO_2 (and potentially also CH_4) (Campeau et al., 2018). The specific absorbance (Abs_{420}/TOC) also did not differ between rewetted and drained sites (Figure 12), suggesting that rewetting, while increasing runoff TOC concentrations, did not affect the characteristics of its coloured fraction (Eklöf et al. 2021; Köhler et al., 2013). Collectively, these results indicate that the higher C in runoff of rewetted sites stem from shallower GWTs and an increased hydrological connectivity to more superficial terrestrial sources rich in organic C, rather than from significant changes in the metabolic source patterns.

Furthermore, the rewetting effect (Δ in concentration) for CO_2 was related to nutrient status (C/N ratio in runoff) and latitude, with greater ΔCO_2 found in more nutrient-rich systems (lower C/N ratio) and at southern latitudes (Figure 13). However, while the study by Koskinen et al. (2017) showed that the rewetting response on DOC export was higher in nutrient-rich than nutrient-poor wetlands, this study showed that rewetting effect on TOC and CH_4 was not related to C/N ratio or latitude (Figure 13).

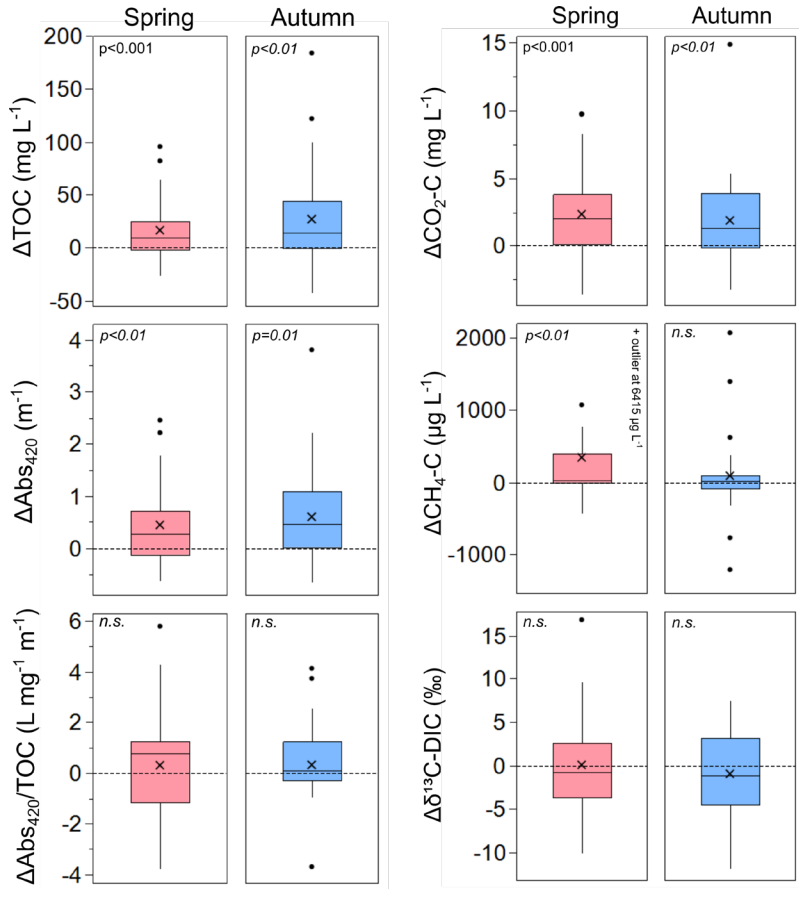

Figure 12. Distributions in delta (Δ) values of TOC, Abs_{420}, Abs_{420}/TOC, CO_2, CH_4 and $\delta^{13}C$-DIC collected in runoff separated by season. The Δ values were calculated as rewetted minus control (drained), thus positive values indicate higher concentration or absorbance metric at rewetted compared to the control site, while negative values indicate the opposite. *p*-values are reported for statistically significant differences from zero (dashed line) according to the GLMMs.

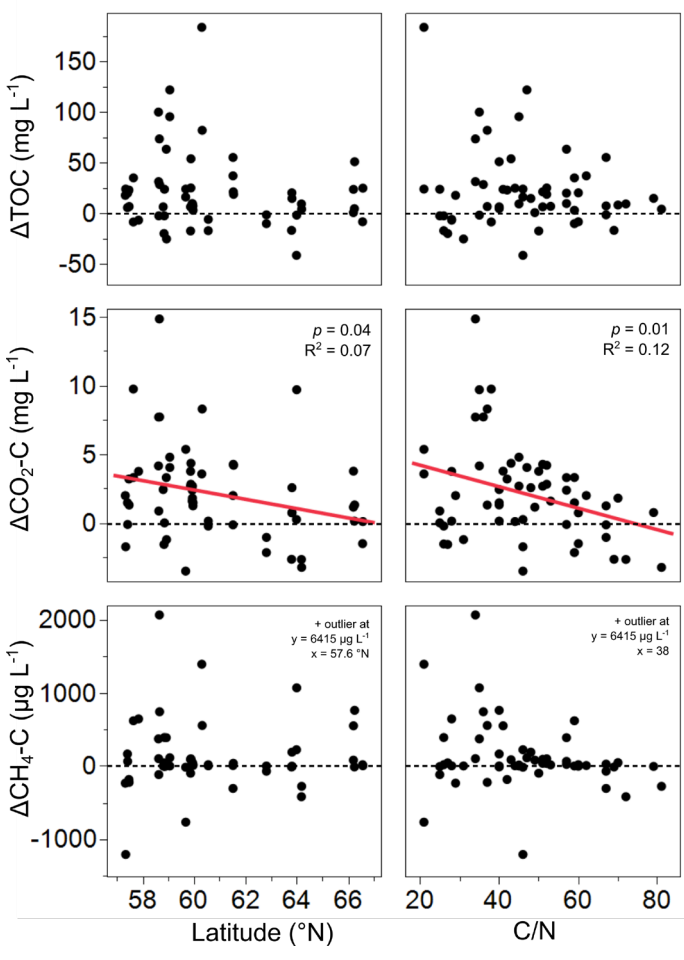

Figure 13. Delta (Δ) concentrations (difference between rewetted and drained) of TOC, CO_2 and CH_4 collected in runoff as a function of latitude and aquatic C/N ratio.

6. Conclusions and future perspectives

The main conclusions of the studies conducted within this thesis are summarized as follow:

Paper I

- Growing season CO_2 dynamics in forest ditches affected by clear-cutting are high and mainly controlled by light- and temperature-induced biological factors. In contrast, hydrology has a comparatively less dominant influence on CO_2 concentration changes.
- The comparative analysis with the forested catchment shows clear differences in CO_2 concentration magnitudes and diel amplitudes, but also in the water temperature patterns. Collectively, the conditions after clear-cutting, with elevated light exposure and increased DOC and nutrients, stimulate primary production as major control in driving ditch CO_2 dynamics.

Paper II

- DC has a significant impact on water chemistry and dissolved GHGs in draining ditch networks as shown by the differences between cleaned and uncleaned ditches. Some of these match previous findings (increased pH and Ca), while others differ (unaffected TOC and nutrients). Notably, DC leads to decreased MeHg (only in forested ditches) and CO_2 but increased N_2O.
- Changes in water chemistry and GHGs following DC are likely driven by (i) deeper groundwater flow paths, (ii) altered soil redox conditions, and (iii) the removal of organic-rich sediments and vegetation from the ditches.

Paper III

- Rewetting of a nutrient-poor boreal peatland increased the lateral export of DOC, DIC, and especially CH_4, which, when converted to CO_2-equivalents, became a major component in the NECB. Neglecting these findings could result in overestimating the climate benefits of peatland rewetting.
- The expansion of open-water areas and altered hydrological flow paths were identified as key drivers for the enhanced C export.
- Radiocarbon analysis showed an overall dominance of contemporary DOC with an indication of even younger DOC being exported via runoff following rewetting.

Paper IV

- Paired sampling of rewetted and drained wetlands across large geographical and climatic ranges of boreal and hemi-boreal Sweden showed that rewetted sites had higher runoff concentrations of all major C forms (TOC, CO_2 and CH_4) compared to drained sites.
- The rewetting effect on runoff CO_2 was higher in nutrient-rich (relatively lower in C/N ratio) wetlands and at lower latitudes. In contrast were no relationships found between C/N ratio and TOC or CH_4.

This thesis also raised a number of considerations for future research:

Forestry impact on ditch CO_2 emissions

- Future studies should combine measurements of aquatic C exports/emissions with *in-situ* metabolism for a more complete understanding of processes regulating surface water CO_2 dynamics following forestry operations.
- The high-resolution monitoring of ditch CO_2 and other relevant variables should be conducted over multiple years in order to explore intra-annual variabilities in external and *in-situ* C sources/sinks.
- The observed CO_2 dynamics and the associated controls need to be considered when scaling ditch CO_2 emissions across boreal landscapes affected by clear-cut forestry.

Ditch cleaning

- Continued monitoring is essential to assess the impacts of DC on the biogeochemistry of forest ditches across temporal and spatial scales, ensuring that DC operations effectively mitigate negative effects on downstream surface waters.
- Future research efforts should capture both long- (season to years) and short-term (hours to days) effects following DC, as different water quality effects might operate at very different time scales.

Rewetting

- More long-term monitoring of the biogeochemical impacts of rewetting is essential, as these effects can evolve or change over time. This thesis primarily focuses on the initial outcomes within two (**Paper III**) to four (**Paper IV**) years from rewetting, while the desired climate benefits or potential water quality changes may take decades or even centuries to fully manifest.
- The results highlight the importance of selecting optimal rewetting sites (e.g., considering the landscape position of new ditch initiation points) to minimize negative impacts on water quality and GHG. One recommendation would be to avoid rewetting areas to form new open-water areas, or where already present open-water areas could expand and function as hotspots for increased GHG production and export.
- Further research is needed to investigate changes in the age of exported C following rewetting, including the analysis of ^{14}C content in DIC and CH_4 to provide a complete age characterization of the LCE.
- More spatially distributed studies are needed to assess the effects of rewetting on runoff C across wetlands with varying nutrient statuses. This is important because nutrient-rich wetlands are currently being targeted for rewetting in climate mitigation strategies.
- Future rewetting studies should address the complete GHG balance, including N_2O (particularly in nutrient-rich wetlands), and account for both lateral and vertical fluxes.

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Popular science summary

If you have ever been to Sweden, you may have noticed the endless stretches of forest. It is no surprise that the Swedish forest industry plays a key role in the economy of the country. Forests not only provide renewable products and energy, but they also absorb carbon dioxide (CO_2) from the atmosphere, regulating the global climate. It is therefore essential to manage the forests sustainably so that they can provide resources while also acting as carbon "sinks". This aspect could be crucial to achieve the EU climate goals of reducing greenhouse gas emissions.

As you explore these forests, you may come across large areas where trees have been harvested. This is a most common practice in the Swedish forestry cycle. As you leave the trail—always be careful!—you may also come across ditches dug into the forest floor to drain the land of water for better tree growth. This may happen more often than you think: these drainage ditches are one of the most significant human alterations to the boreal forest landscape, stretching nearly 1 million kilometers. Yet, many of these ditches are now degraded and filled with sediment and vegetation. This raises big questions for sustainable forest management: Should these ditches be cleaned to restore their drainage function? Should they be left alone? Should they be blocked to restore the original wetland conditions? There is no easy answer—and while each of these actions may improve various ecosystem services (e.g. increased forest productivity, positive impacts on climate, etc.), they could also harm connected aquatic ecosystems by, for example, increasing the mobilization of soil material into surface waters and reducing water quality. There is not a clear understanding of the impacts each practice has on aquatic ecosystems and carbon cycle. This highlights the urgent need for field-based studies of forest and drainage ditch management to help fill

these knowledge gaps and better understand the carbon dynamics of forests and associated surface waters.

My thesis addresses this knowledge gap. Over the course of my PhD, I studied the effects of three land-use management practices—*clear-cutting*, *ditch cleaning*, and *rewetting* of drained wetlands—on ditch water chemistry, focusing on aquatic carbon and dissolved greenhouse gases. The collection of papers in my thesis is based on four studies—two conducted at the catchment scale (**Papers I** & **III**) and two at a regional level (**Papers II** & **IV**).

In **Paper I**, I investigated how CO_2 dynamics change in a ditch after clear-cutting. I monitored CO_2 levels during the snow-free season, where one would typically expect moderate daily CO_2 changes in these nutrient-poor ecosystems where biological activity is low. Instead, I was surprised to see high CO_2 levels and large daily changes in concentrations. It seems that the tree removal increased light exposure, temperature, and dissolved organic carbon and nutrients in the ditch, triggering metabolic processes that significantly enhanced CO_2 production and consumption.

The research question in **Paper II** addressed how water chemistry differs between cleaned and uncleaned ditches. Ditch cleaning entails removing vegetation and sediment from the ditch to restore the drainage capacity. Although ditch cleaning is common practice today, there is a surprising lack of data regarding how water chemistry is affected. Water samples were collected from 25 clean and 25 uncleaned ditches located in east-central Sweden during three sampling occasions. Some results were expected, such as increased pH in cleaned ditches, but others were in contrast with existing studies, such as no change in total organic carbon content. In addition, I show that ditch cleaning affects greenhouse gases by potentially reducing CO_2 but increasing the potent greenhouse gas nitrous oxide (N_2O) in the ditches.

The remaining two studies of **Paper III** and **Paper IV** focus on rewetting, which entails flooding of drained soils by blocking the ditches and raising the water table. Rewetting is currently being widely discussed at the EU management level as a potential climate change mitigation measure. **Paper III** uses data from an experimental peatland site in northern Sweden two years before and two years after rewetting. Results show that rewetting increased the concentrations and water exports of all major forms of carbon and especially methane (a powerful greenhouse gas), raising concerns about the short-term climate impacts of this practice. In **Paper IV**, I expanded my

research on rewetting vs. drained wetlands to the national scale, providing, to the extent of my knowledge, one of the first large-scale assessments of the effects of rewetting on aquatic carbon in Sweden. My results show higher levels of organic carbon and dissolved greenhouse gases in rewetted areas, especially in nutrient-rich wetlands. To summarize the findings of **Papers III** & **IV**, rewetting is currently recognized as good practice for the environmental health, yet it should be implemented with careful consideration of its potential initial effects on water quality and climate.

It is my hope that this thesis findings on the impacts of clear-cutting, ditch cleaning, and rewetting on aquatic carbon will provide a solid evidence base to support informed decisions for future implementation of these practices.

Populärvetenskaplig sammanfattning

Om du någonsin har varit i Sverige har du kanske lagt märke till de oändliga mängden skog. Det är ingen överraskning att den svenska skogsindustrin spelar en stor roll i landets ekonomi. Skogen ger inte bara förnybara produkter och energi, utan absorberar också koldioxid (CO_2) från atmosfären. Det är därför viktigt att förvalta skogarna på ett hållbart sätt så att de kan tillhandahålla resurser samtidigt som de fungerar som ”kolsänkor”. Därmed kan denna förvaltning vara avgörande för att uppnå EU:s klimatmål om att minska utsläppen av växthusgaser.

När du utforskar dessa skogar kan du stöta på områden där träd har huggits. När du lämnar stigen—var alltid försiktig!—du kan också stöta på diken som grävts i skogsbotten för att dränera marken på vatten för bättre trädtillväxt. Detta kan hända oftare än du tror: dessa dräneringsdiken är några de största mänskliga förändringarna av det boreala skogslandskapet, då dessa diken kan sträcka sig nästan 1 miljon kilometer. Ändå är många av dessa diken nu förstörda och fyllda med sediment och växtlighet. Detta väcker stora frågor för ett hållbart skogsbruk: Bör dessa diken rensas för att återställa deras dräneringsfunktion? Ska de lämnas i fred? Bör de blockeras för att återställa de ursprungliga våtmarksförhållandena? Det finns inget enkelt svar—och även om var och en av dessa åtgärder kan förbättra olika ekosystemtjänster (t.ex. ökad skogsproduktivitet, positiva effekter på klimatet etc.), kan de också skada anslutna akvatiska ekosystem genom att t.ex. öka mobiliseringen av jordmaterial i ytvatten och minska vattenkvaliteten. Det finns ingen tydlig förståelse för vilken inverkan varje metod har på akvatiska ekosystem och kolcykeln. Detta understryker det akuta behovet av fältbaserade studier av skogs- och dikesrensning för att fylla dessa kunskapsluckor och bättre förstå kolets dynamik i skogar och tillhörande ytvatten.

Min avhandling handlar om denna kunskapslucka. Under min doktorandtid studerade jag effekterna av tre metoder för markanvändning—kalhuggning, dikesrensning och återvätning av dränerade våtmarker—på dikesvattnets kemi, med fokus på akvatiskt kol och lösta växthusgaser. Artikelsamlingen i min avhandling baseras på fyra studier—två som genomfördes på avrinningsområdesnivå (**Paper I** och **III**) och två på regional nivå (**Paper II** och **IV**).

I **Paper I** undersökte jag hur CO_2-dynamiken förändras i ett dike efter kalhuggning. Jag övervakade CO_2-nivåerna under den snöfria säsongen, där man vanligtvis skulle förvänta sig måttliga dagliga CO_2-förändringar i dessa näringsfattiga ekosystem där den biologiska aktiviteten är låg. Istället blev jag förvånad över att se höga CO_2-nivåer och stora dagliga förändringar i koncentrationerna. Det verkar som om borttagningen av träden ökade ljusexponeringen, vilket i sin tur påverkade temperaturen och det lösta organiska kolet och näringsämnena i diket, vilket utlöste metaboliska processer som avsevärt ökade CO_2-produktionen och konsumtionen.

Forskningsfrågan i **Paper II** handlade om hur vattenkemin skiljer sig mellan rensade och orensade diken. Dikesrensning innebär att vegetation och sediment avlägsnas från diket för att återställa dräneringskapaciteten. Trots att dikesrensning är vanligt förekommande idag finns det en förvånansvärd brist på data om hur vattenkemin påverkas. Vattenprover samlades in från 25 rensade och 25 orensade diken i östra Mellansverige under tre provtagningstillfällen. Vissa resultat var förväntade, t.ex. ökat pH i rensade diken, men andra stod i kontrast till befintliga studier, t.ex. ingen förändring av den totala halten organiskt kol. Dessutom visar jag att dikesrensning påverkar växthusgaserna genom att potentiellt minska CO_2 men öka den potenta växthusgasen dikväveoxid (N_2O) i dikena.

De återstående två studierna i **Paper III** och **Paper IV** fokuserar på återvätning, vilket innebär att dränerad mark översvämmas genom att dikena blockeras och grundvattennivån höjs. Återvätning diskuteras för närvarande flitigt på EU-nivå som en potentiell åtgärd för att begränsa klimatförändringar. I **Paper III** används data från en experimentell torvmark i norra Sverige två år före och två år efter återvätning. Resultaten visar att återvätning ökade koncentrationerna och vattenexporten av alla former av kol och särskilt metan (en kraftfull växthusgas), vilket väcker farhågor om de kortsiktiga klimateffekterna av denna metod. I **Paper IV** utvidgade jag min forskning om återvätning kontra dränerade våtmarker till nationell nivå

och gjorde, såvitt jag vet, en av de första storskaliga utvärderingarna av effekterna av återvätning på vattenkol i Sverige. Mina resultat visar högre halter av organiskt kol och lösta växthusgaser i återvätade områden, särskilt i näringsrika våtmarker. För att sammanfatta resultaten i **Paper III** och **IV** är återvätning för närvarande erkänt som god praxis för miljöhälsan, men det bör genomföras med noggrant övervägande av dess potentiella initiala effekter på vattenkvalitet och klimat.

Jag hoppas att resultaten från denna avhandling om effekterna av kalhuggning, dikesrensning och återvätningens effekter påakvatiska kolsorter kommer att ge en solid bevisbas för att stödja välgrundade beslut för framtida implementering av dessa metoder.

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As I write these acknowledgments, I realize how difficult it is to recall all of you who have entered my life over the past four years, but as I close my eyes for a moment, you all come to me, like a wave of shared experiences. Of course, I got to see some of you almost every day: Svante, Kong, and Carlo. Without a doubt, office life wouldn't have been as much fun without you. You're the "worst"—in the best possible way—office mates I could have ever asked for, and for that, I love you. I was also lucky enough to share a short but precious time with some of you: Claudia, you were an amazing buddy when I started. Maura, for a few months, the *"Italianità"* reigned in the department. Whether for a long or a little time, it didn't matter—you all made the time during my PhD most memorable: a big shoutout to all the community of PhD students, postdocs, staff, and visitors at IVM.
The first two and a half years of my PhD were often spent out in the field, frequently in miserable conditions between clear-cuts and flooded peatlands, rained-on, snowed-on, or eaten alive by mosquitoes. But I have to admit, that is exactly why I love this job: just being out there in the elements. On those days, it was a pleasure to have the company of other amazing fieldworkers like Moritz, Bella, Shirin, Vicky, Viktor, Lucas, Hannah and even my father. A big thank you also goes to the staff at Svartberget Research Station, and especially to Johan Westin, for being available and having a warm room ready for me to crash in at the end of an exhausting field day.

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These years have also made me realize even more how much my friends back home mean to me. Always ready for every *"andirivieni"*, whether it's some adventure in our backyard mountains or just hanging out together: Alex, Anc, Skip, Piro, Scola, Madda, Pippo, Tommy Bi, Pelo, Soma, Jonny, Peter... again, you're too many to list. A special thank you to Tommy—without you, some moments would have been much harder. Thanks for all our talks, my friend.

I've also been thinking more often about my family, and I've realized that I'll never be able to fully repay everything you've done for me. Thank you for everything, Mamma, Papa, and Ste. You are the three most important people in my life, and I just want you to be proud of me.

I'll stop now. They say we shouldn't linger too much in the past, and we should live in the present. Even though, lately, I've been thinking a lot more about the future, and it can feel a little intimidating sometimes. In those moments, it's enough to have you by my side, because you make me realize that the future can be anything we want it to be, and, suddenly, everything seems more exciting. I can't wait. Grazie di cuore, Vale.

> If the point at which you immerse yourself in the river is the present, I thought, then the past is the water that has flowed past you, that which has gone downstream and where there is nothing left for you; whereas the future is the water that comes down from above, bringing dangers and surprises. The past is in the valley, the future is in the mountains.
>
> —*Paolo Cognetti*, *Le otto montagne*


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Metabolic processes control carbon dioxide dynamics in a boreal forest ditch affected by clear-cut forestry

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Boreal watercourses are large emitters of carbon dioxide (CO_2) to the atmosphere. For forestry intensive areas of the Nordic and Baltic countries, a high share of these watercourses are man-made ditches, created to improve drainage and increase forest productivity. Previous studies have suggested that terrestrial sources sustain the CO_2 in these ditches and variability in hydrology is the main temporal control. However, few studies have explored ditch CO_2 dynamics and its associated controls in catchments being exposed to forest harvest. An altered hydrology, increased nutrient export and light availability following forest harvest are all factors that potentially can change both levels, dynamics, and source controls of ditch CO_2. Here, high-frequency (30 min) CO_2 concentration dynamics together with other hydrochemical variables were studied in a forest ditch draining a fully harvested catchment in the Trollberget Experimental Area, northern Sweden. We collected data during the snow-free season from May to October. Ditch CO_2 concentrations displayed a clear seasonal pattern with higher CO_2 concentrations during summer than in spring and autumn. Concentrations ranged from 1.8 to 3.5 mg C L^{-1} (median: 2.4 mg C L^{-1}, IQR = 0.5 mg C L^{-1}). Strong diel cycles in CO_2 developed during early summer, with daily amplitudes in CO_2 reaching up to 1.1 mg C L^{-1}. These pronounced daily cycles in CO_2 were closely related to the daily sum of shortwave radiation and water temperature. Variations in hydrology had generally a low impact on the CO_2 dynamics but did vary among seasons and between individual hydrological events. It was evident from our study that growing season CO_2 concentrations in a forest ditch affected by clear-cut harvest were highly variable and mainly controlled by light and temperature induced metabolism. These high dynamics and the associated controls need to be considered when scaling up ditch CO_2 emissions across boreal landscapes affected by intensive forestry.

KEYWORDS

CO_2, drainage ditches, clear-cut forestry, metabolic processes, land use change

1. Introduction

Headwater streams and ditches are important sources of atmospheric CO_2 emissions, estimated to contribute more than 70% of the total global fluvial CO_2 emissions (Raymond et al., 2013). The importance of headwaters for emitting CO_2 holds particularly true for boreal ecosystems, as they are rich in soil carbon and often closely connected with dense aquatic networks (Wallin et al., 2018). For forestry intensive areas of the Nordic and Baltic

countries, a high share of these watercourses are man-made ditches, created to improve drainage and increase forest productivity during the 20th century (Päivaänen and Hånell, 2012; Norstedt et al., 2021). Given the widespread occurrence of ditch networks combined with high concentrations and emission rates of CO_2, boreal ditches are important sources of atmospheric CO_2 (Peacock et al., 2021). As an intrinsic characteristic, boreal headwaters show close hydrochemical connectivity with adjacent soils and receive continuously high inputs of terrestrial derived carbon from which inflows are largely controlled by variations in hydrological inputs (Billett et al., 2006; Öquist et al., 2009; Crawford et al., 2013; Leith et al., 2015; Wallin et al., 2015). This carbon can enter the aquatic network directly as CO_2 that is produced in the soil via the mineralization of organic matter or from root-associated respiration (Campeau et al., 2019). In addition to terrestrial CO_2 inputs, CO_2 could also be produced *in-situ* in the watercourse from microbial decomposition or photochemical oxidation of organic matter transported from catchment soils (Köhler et al., 2002; Schelker et al., 2016a; Crawford et al., 2017). Aquatic CO_2 can also serve as the carbon source in primary production (photosynthesis), hence being consumed during daytime. For boreal headwaters, *in-situ* decomposition of organic matter is often found to be of minor importance for the observed CO_2 (Winterdahl et al., 2016) due to low residence time of the water (Catalán et al., 2016), limited light availability caused by dense tree canopies (Burrows et al., 2021), and low water temperatures (Tank et al., 2010). In addition, due to often unproductive conditions with low nutrient levels as well as restricted light availability, primary production typically also has low influence on CO_2 in boreal watercourses (Lamberti and Steinman, 1997; Roberts et al., 2007).

A critical aspect to consider when unraveling the different controls and their relative importance on CO_2 in watercourses is the time scale of interest. Different processes that control aquatic CO_2 are operating from hourly to seasonal scales (Riml et al., 2019). Thus, interpreting the primary controls on CO_2 concentration dynamics in watercourses requires continuous data collected at sufficient frequency (e.g., hourly) covering complete seasons (Wallin et al., 2020; Gómez-Gener et al., 2021). The development of sensors that monitor high-frequent CO_2 data has enabled studies that explored controls on dissolved CO_2 in watercourses draining various ecosystems (e.g., forest, agriculture, and wetlands) and across different climatic zones (e.g., boreal, alpine, subtropical, etc.) (e.g., Johnson et al., 2010; Dinsmore et al., 2013; Peter et al., 2014; Crawford et al., 2017; Rocher-Ros et al., 2020; Wallin et al., 2020). Studies performed in streams draining nutrient-poor forest ecosystems have shown that CO_2 dynamics are generally driven by variability in stream discharge (Johnson et al., 2007; Dinsmore et al., 2013; Crawford et al., 2017; Riml et al., 2019). The hydrological response in stream CO_2 is dependent on catchment-specific characteristics and variations in groundwater flow paths controlling connectivity to terrestrial CO_2 sources (Leith et al., 2015; Campeau et al., 2018). In contrast, for ditch or stream systems draining landscapes with open canopies that are fully exposed to light, *in-situ* metabolic processes have been found to have stronger control on aquatic CO_2. In such systems diel cycles in CO_2 can be particularly pronounced, with large concentration differences between day and night (Nimick et al., 2011; Crawford et al., 2017; Attermeyer et al., 2021; Gómez-Gener et al., 2021). These diel cycles reflect the interplay of *in-situ* primary production (photosynthesis) and respiration within the watercourse (or in the adjacent soils).

Previous findings of low *in-situ* contribution to CO_2 dynamics in boreal watercourses stem mainly from studies in low-intensively or unmanaged forest systems (Marx et al., 2017; Campeau et al., 2019). In contrast, few studies (e.g., Klaus et al., 2018) have explored CO_2 dynamics in headwaters draining areas affected by clear-cut forestry, which is a common management operation for many production forests in the boreal countries. In Sweden, about 1% (or ca 230,000 ha, based on a 5-year mean 2016-2020) of the productive forest land is harvested every year (Swedish Forest Agency, 2020), and as a consequence, many forest drainage networks are affected by this practice. Following harvest, the catchment hydrology is altered due to reduced evapotranspiration leading typically to higher groundwater levels and increased runoff (Andréassian, 2004; Sørensen et al., 2009; Schelker et al., 2013). Also, dissolved organic carbon (DOC) concentrations are often increased following harvest due to higher rates of decomposition of organic matter in soils and due to greater lateral mobilization from terrestrial sources (Nieminen, 2004; Laudon et al., 2009; Schelker et al., 2012). Similarly, forest harvest often results in enhanced export of nutrients, especially nitrogen, caused by reduced uptake in vegetation and increased mineralization of organic matter (Nieminen, 2004; Schelker et al., 2016b). There is limited literature concerning the influence of forest harvesting on dissolved CO_2 concentrations and emissions in connecting aquatic systems. However, Klaus et al. (2018) found that harvest increased dissolved CO_2 concentrations in groundwater of the surrounding catchment soils but did not affect CO_2 emissions from recipient streams. To what extent this discrepancy in observed patterns between groundwater and watercourses stems from changes in the *in-situ* stream CO_2 controls following harvest is currently unknown.

This study aims to investigate the impact of clear-cut forestry on the dynamics of dissolved CO_2 concentrations in draining watercourses. We hypothesize that dissolved CO_2 dynamics following forest harvest are altered and become more variable on short time scales (daily) due to changes in light and nutrient regimes which in turn increase the potential for *in-situ* metabolism. To test this hypothesis we (1) quantified ditch CO_2 concentration levels and dynamics for a full growing season in a forest ditch within a catchment recently being clear-cut harvested, (2) identified and explored the main temporal controls and how they vary with season, and (3) compared observed CO_2 concentration patterns from the clear-cut ditch with patterns observed in a stream draining an unmanaged forest catchment in close proximity.

2. Study area

The study was conducted in the Trollberget Experimental Area (TEA) (64°10'N, 19°46'E), located 50 km northwest of the city of

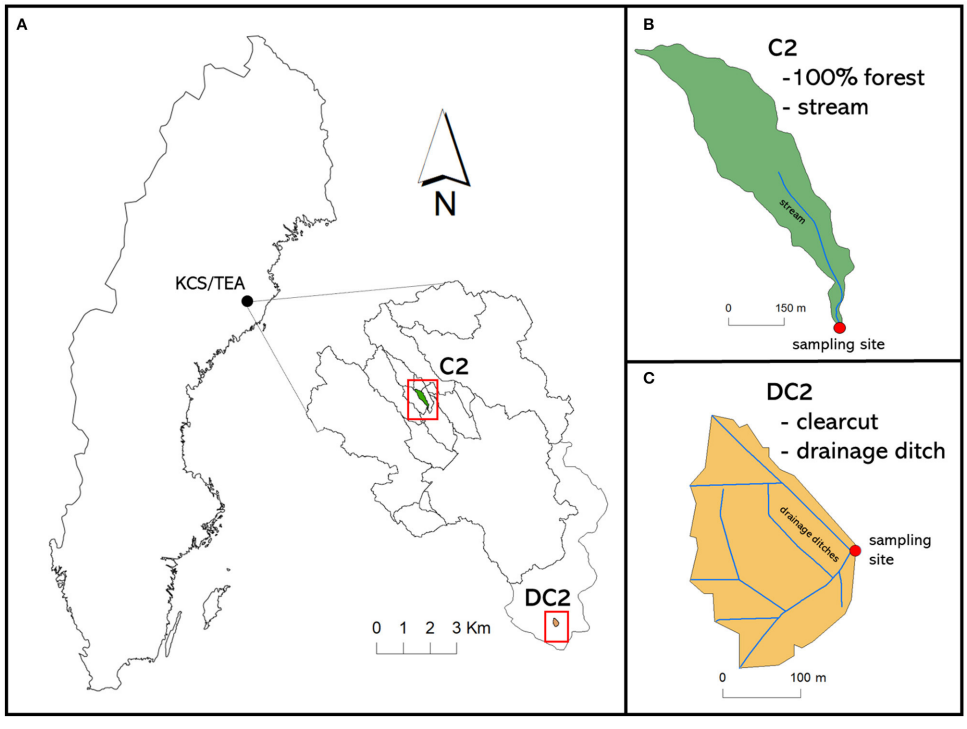

FIGURE 1
The **(A)** Krycklan catchment study including the TEA, with the **(C)** DC2 and **(B)** C2 sub-catchments highlighted.

Umeå, Sweden (Figure 1A)[1]. The experimental area was set up in 2018 to study the environmental impacts of different types of forest management practices on aquatic ecosystems and has been embedded within the framework of the Krycklan Catchment Study (KCS) (Laudon et al., 2021). The mean annual air temperature for the area is 2.4°C and with a mean annual precipitation of 623 mm (about 30% as snow, based on data from 1980–2020 collected at the nearby Svartberget Climate station) (Laudon et al., 2021). Across the period of the growing season, the number of sunlight hours changes drastically at these northern latitudes (from about 20 h in early June to about 8 h in October).

Within TEA, we studied a 4.4 ha large catchment (DC2) characterized by a dense ditch network (total length, 1.1 km, density, 0.025 m m^{-2}) (Laudon et al., 2021, Figure 1C). The DC2 catchment is dominated by till soils (almost 100%) with well-developed podzols including a 10–20 cm humic/partly humic layer on top. Until July 2020, DC2 was completely forest covered, mainly by Norway spruce (Picea abies) and Scots pine (Pinus sylvestris). In July 2020, the catchment was completely harvested.

A second unmanaged sub-catchment (C2, located about 10 km from DC2) of the Krycklan catchment study was used in the study for comparative analysis with the clear-cut catchment of DC2 (Laudon et al., 2013; Leith et al., 2015; Figure 1B). C2 is 100% forested and slightly larger (12 ha) than DC2 but is otherwise similar in terms of soil types and forest composition. Both DC2 and C2 have been affected by historical ditching activity that occurred in the early 20th century to improve drainage. Catchment characteristics of DC2 and C2 are shown in Supplementary Table 1.

3. Methods

3.1. Sensor measurements

Measurements in DC2 were conducted during the snow-free period from 8 May to 28 October 2021 (in total 174 days), encompassing a full growing season. CO_2 concentration was continuously measured together with water temperature, and electrical conductivity (EC) just upstream of a V-notch weir installed at the outlet of DC2. The sensors were deployed

1 Laudon, H., Mosquera, V., Eklöf, K., Järveoja, J., Karimi, S., Krasnova, A., et al. (2023). Consequences of rewetting and ditch cleaning on hydrology, water quality and greenhouse gas balance in a drained northern landscape. (Manuscript submitted for publication).

underwater attached to a wooden structure of the weir. CO_2 concentration was monitored using an eosGP sensor (range 0–2%, Eosense, Dartmouth, Canada) wrapped with copper tape to prevent biofouling. Inspection and cleaning of the sensors were performed monthly. Sensor accuracy is, according to the manufacturer, <1% of the calibrated range (0–2% CO_2) + 1% of the reading corresponding to a maximum error of ca 0.1 mg C L^{-1} based on the maximum CO_2 measured in the current study. Water temperature and EC were monitored using a thermocouple (Type T) and a CS547A-L conductivity sensor (Campbell, UK), respectively. All sensors were connected to a CR1000X data logger (Campbell, UK) measuring at a 1 min interval and storing mean values at a temporal resolution of 30 min.

Volume fraction outputs (ppmv of CO_2) from the sensor were corrected for variations in water temperature, water level and atmospheric pressure (Johnson et al., 2010; Wallin et al., 2020) and expressed in milligrams of carbon per liter (mg C L^{-1}). Finally, sensor derived CO_2-C concentration data were calibrated against manually taken CO_2 measurements (n = 35, see water sampling section) that were carried out in the ditch during the entire study period. Calibration was conducted through the application of a regression equation, encompassing nearly the entire range of measurements (Supplementary Figure 1). Water discharge was measured at the outlet V-notch weir using an established stage height-discharge relationship. Stream discharge gauging for rating curve definition was done using time-volume (bucket) measurements covering a wide discharge range (n = 10). Stage height was continuously recorded (60 min) using a capacitance sensor (TruTrack Logger Type WT-HR 64K). Discharge per unit of catchment area (Q) was calculated and reported in mm h^{-1} or mm d^{-1}. Precipitation and air temperature were measured nearby (300 m) the DC2 catchment using an ARG100 tipping bucket rain gauge (Campbell, UK) and a shaded thermocouple Type T. Short-wave radiation (SR) was measured at 30 min intervals within DC2 about 150 m from the catchment outlet using a Huskeflux NR01 net radiometer. Atmospheric pressure was monitored at the meteorological station of Svartberget (located ca. 8.5 km from DC2) and data were downloaded from the ICOS carbon portal (https://www.icos-sweden.se/Svartberget). Stream CO_2 concentrations at C2 were measured (at 5 min resolution) using a Vaisala CARBOCAP GMP221 non-dispersive infra-red (NDIR) sensor (range 0–5 %), that was hermetically sealed and covered with a gas-penetrable membrane (Johnson et al., 2010; Leith et al., 2015; Campeau et al., 2018). The outlet of C2 is equipped with a V-notch weir in a heated dam house and with stage height recorded at 5 min resolution. As in DC2, discharge in C2 was determined according to a known stage height-discharge relationship based on volume-time measurements.

3.2. Water sampling

Manual sampling for water chemistry at the outlets of DC2 and C2 was conducted at biweekly intervals, as part of a regular monitoring program of KCS/TEA, for a total of 15 occasions. In addition, during the 3 weeks between September 20 and October 10, DC2 was manually sampled daily as part of an intensive sampling campaign in all the ditches within the TEA. For DC2, a total of 35 grab samples were taken during the study period with a mean sampling time at ca 10:45. At C2, the mean sampling time for the grab samples was ca 13:30. For analysis of dissolved CO_2, a headspace method was used where a 5 mL sample of bubble-free water was injected in a 22.5 ml glass vial sealed with a bromobutyl rubber septa (Wallin et al., 2010; Åberg and Wallin, 2014). The injection was made by using a sterile syringe which was flushed with stream water before sampling. The vial was pre-filled with 0.1 ml 85 % H_3PO_4 and N_2 at atmospheric pressure. The samples were stored dark and cold (4°C) for a maximum of 1 week prior to analysis. Samples were analyzed on a gas chromatograph equipped with a methanizer and flame ionization detector (GC-FID). Water samples for pH and EC analysis were collected in 50 mL polyethylene bottles tightened avoiding the formation of air bubbles. In the lab, pH and EC were measured with pH and conductivity electrodes (MP220, Mettler Toledo). Grab samples for DOC analysis were collected in 250 mL polyethylene bottles, filtered [0.45 µm mixed cellulose ester (MCE) syringe filters, Millipore®] within 24 h and then acidified to remove inorganic carbon prior to analysis. Analysis was performed with a Shimadzu Total Organic Carbon Analyzer TOC-VCPH, following storage at 4°C for 2–3 days' periods (Leach et al., 2016; Campeau et al., 2018). Finally, filtered subsamples were stored at a temperature of −20°C for later analysis of nitrogen and phosphorus and their respective fractions (for more analytical details see Blackburn et al., 2017 and Mosquera et al., 2022).

3.3. Statistical analysis

All sensor data were visually inspected to examine their patterns and a 15 day moving average of CO_2 concentrations was calculated to better visualize the seasonal dynamics. Linear regression analysis was used to investigate the relationships between ditch CO_2 concentration or amplitude in diel CO_2 concentration and discharge, water temperature and daily accumulated SR. Linear regressions were considered significant if $p < 0.05$. Dynamics in ditch CO_2 concentration were explored and visualized on a diel (24 h) basis for the full study period, as well as for individual months, using box plots with a 30 min resolution. The response in CO_2 concentration to variable discharge was analyzed by constructing C-Q relationships [log daily median C (mg L^{-1}) vs. log specific discharge (mm d^{-1})]. Such C-Q plots were created on a monthly basis in order to assess whether the hydrological control was changing throughout seasons. The values of the slopes obtained from C-Q regressions were interpreted as done in Rehn et al. (2023) and following Meybeck and Moatar (2012).

To distinguish the different controls on CO_2 dynamics, significant hydrological events were identified according to the method described in Lannergård et al. (2021). The method adopted for the event definition was based on the change in daily discharge (mm d^{-1}). The events started on (1) the rising limb of the hydrograph (the previous observation should be on the falling limb), (2) with an increase <3% from one observation to the next (x_1 = 0.03), (3) no threshold was set for excluding events during low flow conditions, meaning an event could start during the full study period (x_2 = 0). However, events with a magnitude lower than 0.4 mm h^{-1} were omitted. To mark the end of an event, the

decrease in discharge from one observation to the next was set to 20% ($x_3 = -0.2$) and the observed discharge was less than the baseflow decay function. The baseflow decay function is a baseline, starting at the 1st day of the event, with a starting value of the discharge during that day. It is then decreasing with 0.1% per day (further explained in Lannergård et al., 2021). The identification of the events was done in Python 3.9.

The hydrological control on ditch CO_2 concentrations was further explored using CO_2-Q hysteresis analysis and where the shape and direction of the hysteresis loops of each hydrological event were identified (Evans and Davies, 1998; Wallin et al., 2020). The shape of the hysteresis loops has been related to the timing of CO_2 and discharge responses depending on catchment characteristics and hydrological pathways (Evans and Davies, 1998). A clockwise shape indicates a system where CO_2 peaks before discharge, and could indicate a transport limited source of CO_2 but that eventually reaches a source limitation in the available catchment soil or stream bed CO_2 pool. An anti-clockwise pattern typically indicates a diluting effect on CO_2 suggesting a source limitation occurring already at low discharge increases (Wallin et al., 2020). A complex CO_2-Q loop, instead, indicates that the CO_2 pattern is not related to the changes in hydrology or that any relationship is interfered by additional controlling processes. Hysteresis indexes were calculated according to Lloyd et al. (2016) using a 20% increment of the discharge range for each event. For further analysis, the hysteresis indexes were combined with event characteristics (duration of event, season, $Q_{max,\ mean,\ range,\ peakhour}$, $CO_{2mean,\ range,\ peakhour}$, shape) and environmental conditions (water temperature, precipitation, $EC_{mean,\ range}$, SR_{tot}) (Supplementary Table 2). A principal component analysis (PCA) was used to explore the entire event data set (Supplementary Table 2) and to evaluate the different temporal controls on ditch CO_2 concentration dynamics.

Finally, statistical differences in chemical variables between the DC2 and C2 catchments were assessed using the non-parametric Wilcoxon test and were considered significant if $p < 0.05$. The software JMP Pro 15 (SAS Institute Inc., Cary, NC, USA) was used for all statistical calculations.

4. Results

4.1. Seasonal variation in hydrochemical variables and CO_2 concentrations

Mean air temperature and total precipitation registered at TEA for the full study period (8 May—28 October 2021) were 11.7°C and 571 mm, respectively (Figure 2A). Precipitation was distributed relatively evenly throughout the months of the study period (June–October, ~20%/month), with July and October being the months with the highest precipitation (118 mm each) and July 30 the day with the highest daily precipitation (57 mm). Mean ditch water temperature over the study period was 10°C and ranged from −0.4 to 19.7°C (Figure 2B). High diel variability in water temperature was evident, with daily temperature amplitudes being closely related to the daily mean SR ($R^2 = 0.88$, $p < 0.0001$). Mean and median daily discharge (Q) for the study period were 1.86 and 0.98 mm d^{-1}, ranging from 0 to 17 mm d^{-1} (Figure 2C). The total number of dry days (i.e., days without any registered water flow over the V-notch weir) was 19 out of 174, or 11% of the study period. According to frequency analysis, 73% of the days had a daily mean discharge below the overall mean for the study period (1.86 mm d^{-1}), but the accumulated discharge during days with a discharge higher than the overall mean accounted for 71% of the total discharge (324 mm). Daily precipitation and discharge were positively related ($R^2 = 0.53$, $p < 0.0001$, Supplementary Figure 2) with an average response time between precipitation event and discharge peak of 2 h. The mean pH was 4.6 ($n = 15$) and the electrical conductivity was on average 35.5 µS cm^{-1} (range: 22.9–66.0 µS cm^{-1}) and was positively related to variations in discharge ($R^2 = 0.38$, $p < 0.0001$).

The mean CO_2 concentration at DC2 for the whole study period was 2.47 mg C L^{-1} (IQR = 0.51 mg C L^{-1}) [corresponding to a partial pressure (pCO_2) of 2,848 µatm] and were ranging from 1.81 to 3.50 mg C L^{-1} (pCO_2 range: 461–7,183 µatm) (Figure 2D). This should be compared with an estimated atmospheric equilibrium concentration of dissolved CO_2 of 0.23 mg C L^{-1} (assuming an atmospheric CO_2 concentration of 417 ppm and an average water temperature of 10°C representing the entire study period). Ditch CO_2 concentrations displayed a clear seasonal pattern with higher CO_2 during summer than in spring and autumn. CO_2 increased from May until approximately the beginning of August and then started to decrease. The highest measured CO_2 concentrations occurred at midnight on 9 July. The CO_2 concentration displayed a bimodal distribution with frequency peaks at ~2.1 and ~2.7 mg C L^{-1} (Supplementary Figure 3). The higher peak ($n = 1,194$) represents data collected during spring and late summer as well as on many days during the autumn period, whereas the lower peak ($n = 797$) was attributed to some days in early June and the summer period (July–August).

4.2. Light and temperature controls on variation in CO_2

Mean daily CO_2 concentration was positively related to daily mean water temperature for the full study period ($R^2 = 0.75$, $p < 0.0001$, Figure 3A). On a diel scale, CO_2 concentrations commonly displayed a cyclic pattern that were developed during early summer, with daily CO_2 amplitudes reaching maximum 1.07 mg C L^{-1} in July, and then progressively decreasing to reach its minimum of 0.04 mg C L^{-1} in October (Supplementary Figure 4). The amplitude of the diel CO_2 concentration was related to the daily accumulated shortwave radiation ($R^2 = 0.24$, $p < 0.0001$, Figure 3B) as well as to daily mean water temperature ($R^2 = 0.59$, $p < 0.0001$, Figure 3C). During the full period of study, daily CO_2 concentrations were higher during night-(22:00–7:00) than day hours (Supplementary Figure 5), with the highest and lowest concentrations within a diel cycle at around 1:00 and 14:00, respectively (mean values: 2.59 and 2.32 mg C L^{-1}). By separating the diel analysis by month (Figure 4), the highest monthly mean CO_2 concentration (3.18 mg C L^{-1}) was measured at 00:30 in July. July also showed the largest mean daily amplitude in CO_2 concentration ($\Delta CO_2 = 0.52$ mg C L^{-1}), with the minimum values recorded at 13:30. Among the studied months (May–October), a shift in the hour of the day when

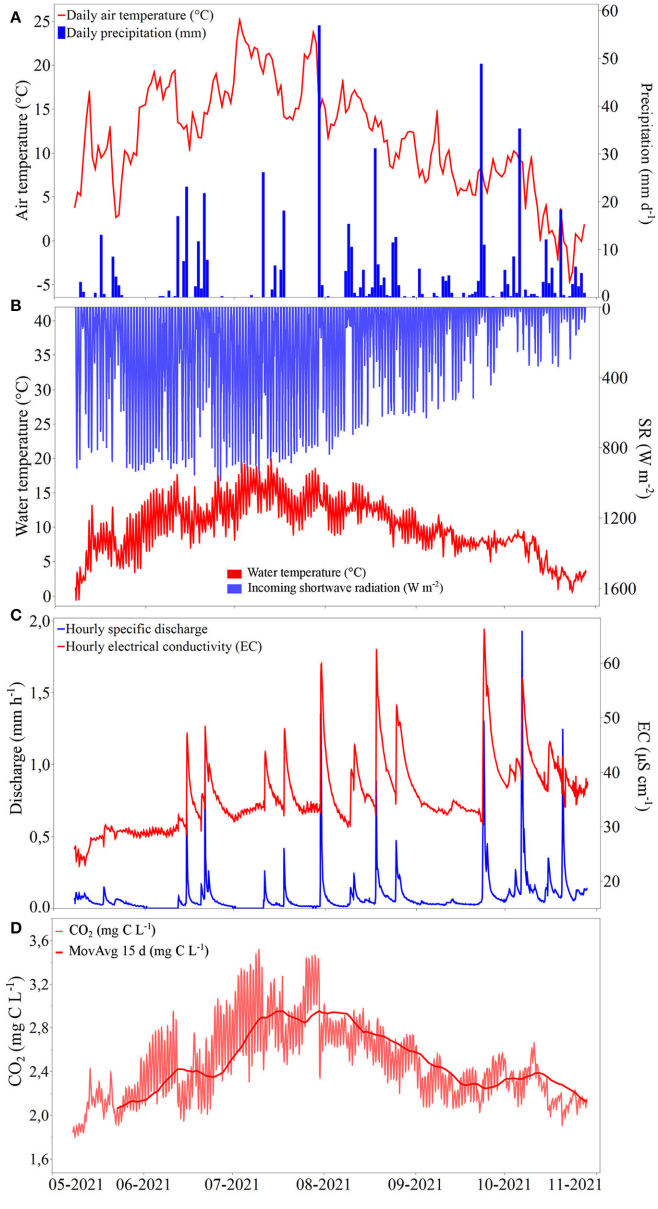

FIGURE 2
Time series of **(A)** daily mean air temperature (line) and daily precipitation (bars) registered at TEA; **(B)** water temperature (blue) and shortwave incoming radiation, SR (red); **(C)** hourly discharge and electrical conductivity (EC); **(D)** dissolved CO_2 for the study period 8 May–28 October 2021 in DC2. Note the reverse axis for SR.

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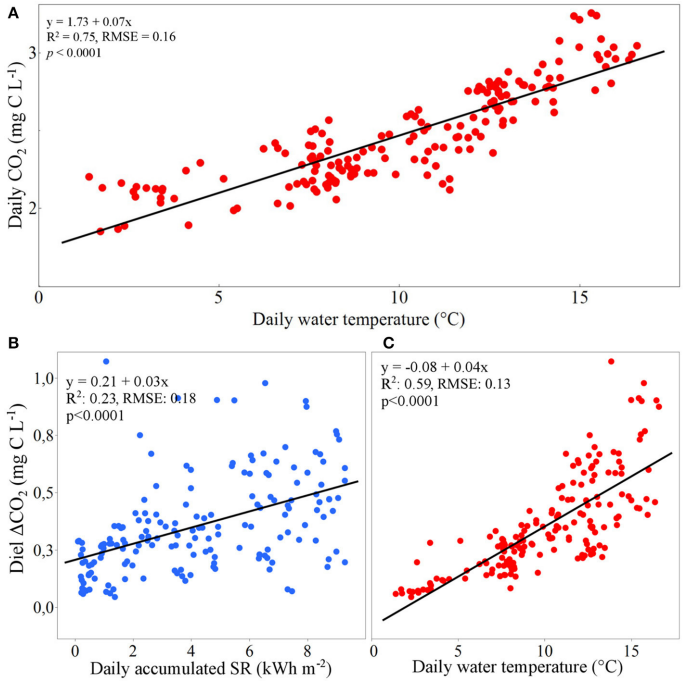

FIGURE 3
(A) Daily mean ditch concentration of CO_2 at DC2 as a function of daily mean water temperature; Diel amplitudes in CO_2 concentrations at DC2 as functions of **(B)** daily accumulated incoming short-wave radiation, SR and **(C)** water temperature.

CO_2 peaked was noticed. Daily maximum peak, $[CO_2]_{max}$, was recorded at around 23:30 during May and June, at 00:30 in July, 2:30 in August, 6:00 in September and finally occurred at 7:30 in October. Daily minimum peak, $[CO_2]_{min}$, was recorded at 12:00 in May but gradually shifted in time, from 13:00 in June, to 13:30 in July, 14:30 in August and, finally, 16:00 in September and October.

4.3. Hydrological control on variation in CO_2

Significant negative logCO_2-logQ relationships were found on a monthly basis from May to August with variable explanatory power (R^2 = 0.18–0.71) (Figure 5), with highest R^2 in May and lowest in June. The slope of the logCO_2-logQ relationships were classified as chemostatic for the entire study period, but progressively became less negative for every month from −0.09 in May to −0.02 in August. In contrast, during September and October no significant logCO_2-logQ relationships were identified suggesting low influence of variations in runoff on CO_2 at the monthly basis during autumn.

4.4. Event based evaluation of controls on CO_2

Based on the event identification (see method section above), 19 hydrological events were identified during the full study period (Supplementary Figure 6) with different characteristics (Supplementary Table 2). The duration of each event varied between 2 and 10 days, with an average of 5.6 days. The events further covered a wide discharge range (between 0.04 and 1.8 mm h^{-1}, representing 92% of the monitored Q range). Three different shapes of CO_2-Q loops were identified by the calculated hysteresis indexes, (1) clockwise (CW) loop with positive indexes during the full event, (2), anticlockwise (AW) loop with negative indexes during the full event, (3) complex (complex) loop that contained both positive and negative indexes during the event (Supplementary Figure 7). Out of all the CO_2-Q loops, nine displayed CW shape, four displayed the AW shape and six displayed the complex "figure eight" shape (Figure 6). CW and

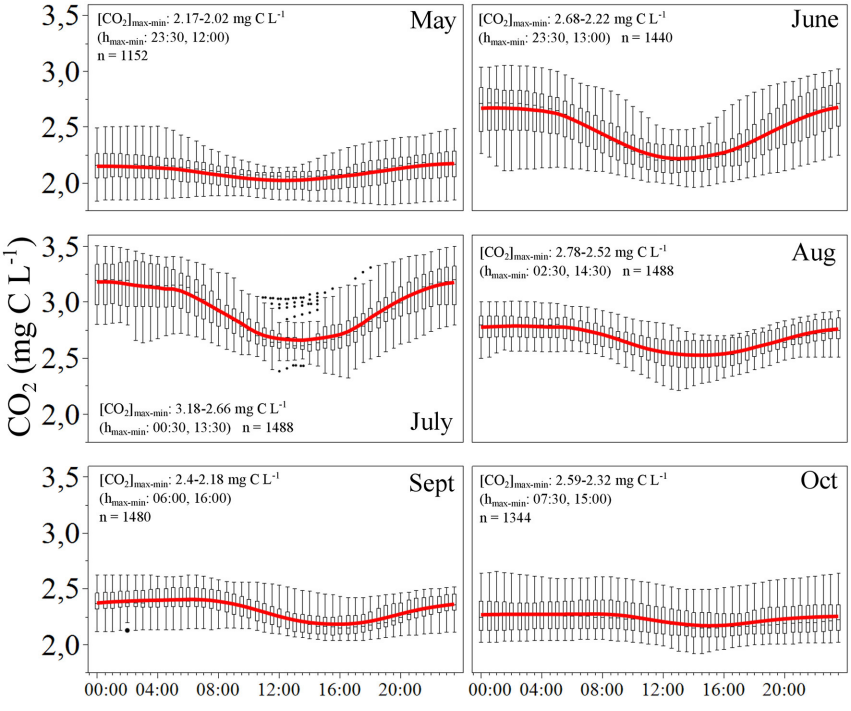

FIGURE 4
Monthly distribution in ditch CO_2 concentrations at DC2 over the full study period presented on a diel (24 h) basis with each boxplot representing a 30 min period. The red line displays the mean diel CO_2 concentration pattern.

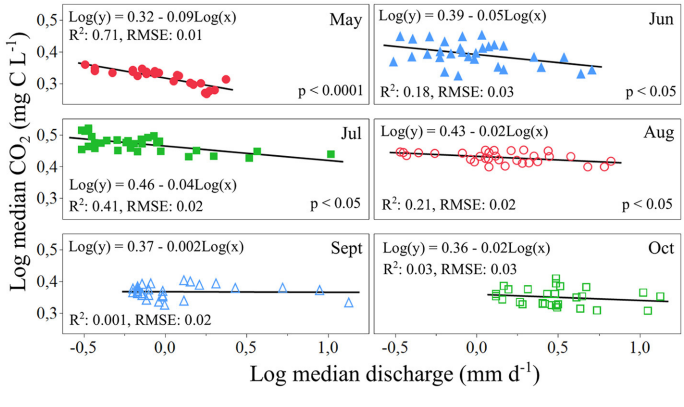

FIGURE 5
Log median CO_2 concentrations at DC2 as a function of Log median discharge for the different months of the study period.

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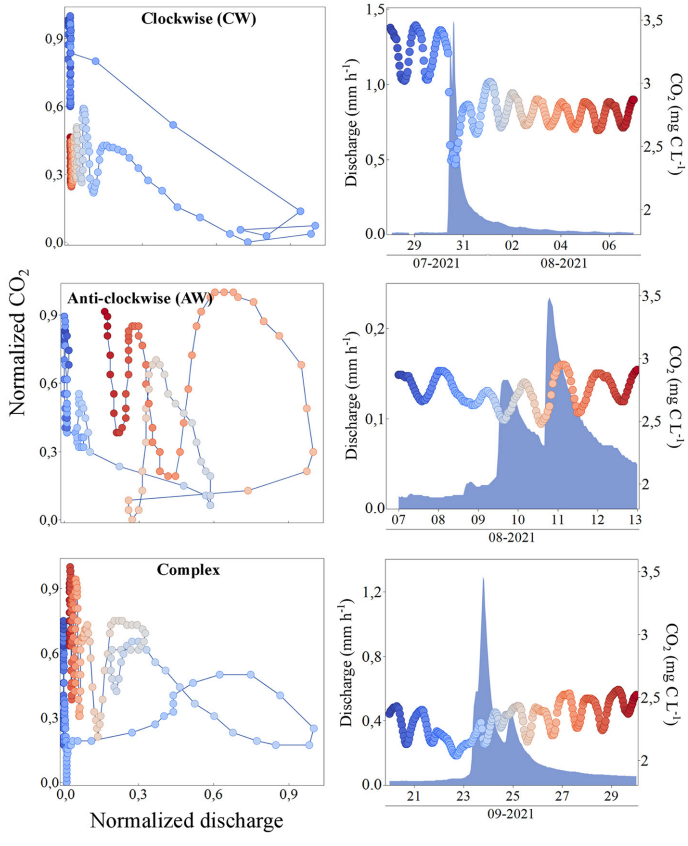

FIGURE 6
Examples of the three different identified event types (CW, event no. 8, AW, event no. 9 and complex, event no. 13, see Supplementary Table 2). To the **left**, the three types of hysteresis loops with normalized CO_2 concentration (y-axis) and normalized discharge (x-axis). To the **right**, the time series of discharge (**left** y-axis) and CO_2 concentration (**right** y-axis) over time for each event (x-axis). Cold colors (blue/light blue) represent the beginning of the event and warm (red/orange) colors the end of the event.

complex hysteresis patterns occurred independent of season or size of the hydrological event. AW loops, on the other hand, were more common during the summer period and at medium discharge ($0.07 < Q < 0.12\ mm\ h^{-1}$, Supplementary Figure 7).

Results of the PCA showed that the first two principal components (PC1 and PC2) accounted for 69% of the variation (Figure 7). The PCA showed a good assemblage of the events based on their seasonality (spring, summer, and autumn), but not according to the shape of the hysteresis loops. Summer events were generally positively associated with mean CO_2 and range of CO_2 variation, total SR, and mean water temperature. In contrast, these descriptive characteristics were generally negatively related to autumn events. Flow related characteristics (max, range and mean Q) were not related to CO_2 describing characteristics (or summer events). The total precipitation generating each event was closely related to the duration of the event, but also to the range in measured EC. The mean EC was closely associated to both the maximum (Q_{Max}) and range (Q_{Range}) in discharge generated at each event. Finally, no correlation was found between the CO_2 and Q peak hours. The shapes of the hysteresis loops were not clearly related to any of the descriptive characteristics during events.

4.5. Comparison with forested catchment

The CO_2 concentration time series from DC2 where further compared with a corresponding time-series collected simultaneously from the completely forested catchment (C2) to explore any differences in the observed CO_2 patterns between catchments with distinct land cover (Figure 8). It was evident from

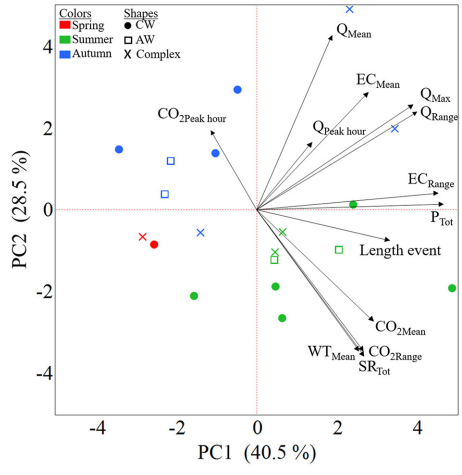

FIGURE 7
Biplot of the principal component analysis (PCA) based on event-specific data (Supplementary Table 2). The arrows represent the event characteristics and markers in color represent the single events. Each event is presented by (1) season, with different marker colors (red: spring, green: summer, blue: autumn) and (2) shape of the hysteresis loop, by marker type, circles: CW, squares: AW and crosses (×): Complex.

the 2-month comparison (May–June) that the two catchments differed in water chemistry (Table 1). DC2 and C2 displayed similar pH (4.6) but DC2 had generally higher EC than C2. Mean DOC concentrations in DC2 were twice the concentrations in C2. Nutrient levels were 4–14 times higher in DC2 than C2. Both the overall magnitude in CO_2 concentration and its associated diel dynamics were different between the two headwater catchments. The mean CO_2 concentration in DC2 for the 2 months was 2.32 mg C L^{-1} (range: 1.81–3.27 mg C L^{-1}), to be compared with 1.51 mg C L^{-1} (range: 1.22–2.86 mg C L^{-1}) for C2. On average, the amplitude of the diel cycle recorded at DC2 was 0.41 mg C L^{-1}, or four times as high as in C2 (0.10 mg C L^{-1}). All comparison between water chemistry variables at the two sites, except for pH, were significantly different ($p < 0.05$). The frequency distribution of CO_2 concentration for DC2 showed a left-skewed unimodal distribution peaking around 2.05–2.15 mg C L^{-1} accounting for 20% of the observations, while C2 has a left-skewed bimodal distribution (peak values at around 1.25–1.40 and 1.65 mg C L^{-1} representing 32% and 9% of the total, respectively) (Supplementary Figure 8). The LogCO_2 vs. LogQ relationship of C2 exhibited a stronger linear fit ($R^2 = 0.82$, $p < 0.0001$) and a steeper slope (i.e., −0.12) than DC2 ($R^2 = 0.25$, slope: −0.08, $p < 0.0001$) (Supplementary Figure 9). The water temperature time series of the two sites showed an overall similar seasonal pattern but with a much more pronounced diel water temperature amplitude (5.0°C) in DC2 than in C2 (2.0°C). It is to be noted that in C2 the initial discharge peak in May is attributed to the snowmelt that was still ongoing in the forested catchment for the two 1st weeks of the comparison. In contrast, for the clear-cut dominated DC2, the snow had already melted, and the discharge peak already passed prior to the comparing 2-month period. Despite the discharge peak induced by the snowmelt at C2, CO_2 stream concentrations were relatively stable.

5. Discussion

Headwater streams and ditches are known hotspots for atmospheric CO_2 emissions, and the hydrological export of CO_2 from catchment soils is commonly found as the main source in boreal regions (Striegl and Michmerhuizen, 1998; Rasilo et al., 2012; Riml et al., 2019). However, current large-scale estimates suffer from limited information regarding how these emissions are affected by human induced disturbances. Forestry is one such major disturbance, which is known to alter a wide range of hydrological and biogeochemical processes, but the effect on CO_2 concentration dynamics and associated emissions in connected drainage networks are largely unknown.

Here we observed a mean clear-cut ditch CO_2 concentration (2.47 mg C L^{-1}) that was relatively high compared to both what was observed in the comparing forested catchment (C2), but also compared to other high-resolution monitoring studies of forested headwaters found in the literature. For example, mean CO_2 concentration levels found in a study of streams draining different boreal and temperate forest ecosystems were generally lower (range of means: 0.73–2.13 mg C L^{-1}) than the mean of the current study (Dinsmore et al., 2013). Furthermore, expressed as partial pressure (pCO_2), the range found in our study (461–7,183 µatm) encompassed the full pCO_2 range found by Crawford et al. (2017) covering multiple ecosystem types from alpine tundra (434–536 µatm) to temperate forests (2,815–6,225 µatm). The overall seasonal CO_2 concentration pattern found, characterized by a summer peak in CO_2, is typically observed across different types of ecosystems (i.e., arctic tundra, boreal forest, temperate forest, temperate peatlands, and alpine regions) (Crawford et al., 2017). The observed seasonal CO_2 concentration pattern suggests a respiratory source further supported by the close relationship between mean daily CO_2 concentration and water temperature (Figure 3A). Respiration is strongly controlled by temperature (Del Giorgio and Williams, 2005; Yvon-Durocher et al., 2012), and microbial mineralization of soil organic material is known to increase after clear-cut harvest due to increased soil temperatures caused by the absence of shading trees (Liski et al., 1998; Schelker et al., 2013).

In addition to the observed overall seasonal patterns, the high-frequency measurements allowed us to capture ditch CO_2 concentration dynamics on short timescales (hourly or daily). For a majority of the study period, a clear diel signal was recorded with large day-to-night differences in CO_2 concentration (mean and medium Δ: 0.35 and 0.30 mg C L^{-1}, respectively corresponding to mean and median ΔpCO_2 of 1,105 and 885 µatm). These diel CO_2 cycles were particularly pronounced in amplitude during June and July (reaching up to 1.1 mg C L^{-1} or 4,078 µatm) but became more constrained toward the autumn. The mean of observed daily CO_2 amplitudes was comparatively high in relation to other

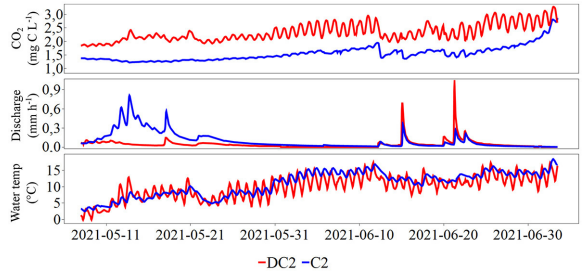

FIGURE 8
Time series of CO_2, discharge, and water temperature for DC2 (red) and C2 (blue) during the period 8 May-5 July 2021 (2 months).

TABLE 1 **Water chemistry at the outlets of DC2 and C2 catchments manually collected during the period 8 May-5 July 2021 ($n = 6$).**

	Median		Mean		Min-Max	
	DC2	C2	DC2	C2	DC2	C2
EC ($\mu S\ cm^{-1}$)	32.4	25.4	34.9	25.4	28.8–50.7	23.6–28
pH	4.6	4.5	4.6	4.5	4.4–4.8	4.4–4.6
DOC ($mg\ C\ L^{-1}$)	38.7	21.3	45.9	22	32.5–83.8	15.8–32.7
CO_2 ($mg\ C\ L^{-1}$)	2.2	1.2	2.0	1.2	1.6–2.2	0.9–1.9
NO_2-N + NO_3-N ($\mu g\ N\ L^{-1}$)	18.7	4.4	40.1	4.8	8.3–148.4	3.9–7.3
NH_4-N ($\mu g\ N\ L^{-1}$)	56.9	7.8	114.6	8	6.9–433.2	6.7–10
PO_4-P ($\mu g\ P\ L^{-1}$)	10.9	2.1	17.4	1.8	4.5–57.2	0.7–2.4

continuous CO_2 measurements in low-productive arctic and alpine streams exhibiting diel fluctuations. For example, Rocher-Ros et al. (2020) registered a mean summer amplitude of about 900 µatm in a Swedish arctic tundra stream, and Peter et al. (2014) found a mean diel CO_2 amplitude of about 370 µatm in an alpine stream, with amplitude values that peaked at 845 µatm in the summer during extended base flow. In contrast, much higher diel CO_2 amplitudes were found in a nutrient rich agricultural stream in Sweden (medium amplitude: 2.03 mg C L^{-1}, 2,974 µatm) (Wallin et al., 2020). We suggest that the pronounced diel cycles found in the ditch of the current study were driven by aquatic primary production consuming CO_2 during daytime, as the minimum concentrations were recorded during mid-day and with a gradual shift in timing toward the afternoon during autumn (Rocher-Ros et al., 2020, 2021; Gómez-Gener et al., 2021). The pronounced diel patterns we observed further suggests that the temporal control on ditch CO_2 has changed after clear-cut as a consequence of the elevated DOC and nutrient concentrations (compared to the forested catchment, C2) as well as due to increased light availability. Diel dynamics of the observed amplitude are typically observed in open canopy systems and is attributed to primary production largely driven by high light exposure (Crawford et al., 2017; Gómez-Gener et al., 2021).

Hydrology (i.e., variations in discharge) commonly plays an important role in regulating stream CO_2 dynamics across different environments although with site-specific CO_2-Q relationships (Dinsmore et al., 2013; Riml et al., 2019; Wallin et al., 2020). In our case study, the influence of variations in discharge on growing season ditch CO_2 concentrations was complex and not easy to disentangle from the metabolic diel dynamics. Median daily ditch CO_2 concentration was found negatively related to median daily discharge during spring and summer (May to August) but not during autumn (September to October) (Figure 5). However, the slopes of the monthly CO_2-Q relationships indicate a general "chemostatic" response in relation to variable discharge, implying a relatively low hydrological influence on ditch CO_2 concentrations. This suggests that (1) the terrestrial (or in-ditch) source for CO_2 is relatively stable in its hydrological connectivity, or (2) that non-hydrological processes counterbalance any variations in CO_2 caused by a variable discharge (Rehn et al., 2023). The response in CO_2 following individual hydrological events was in contrast highly variable. The CO_2-Q hysteresis plots were in many cases influenced by the diel CO_2 fluctuations leading to tangled hysteresis shapes, making it hard to extract information. Only one extreme hydrological event (i.e., event no. 8, Figure 6) showed an unequivocal and straightforward response in CO_2 concentrations from the analysis of the hysteresis loops. It is worth noting that this event registered both the highest incoming SR as well as the second highest total precipitation and had a runoff peak during day hours (Supplementary Table 2), when CO_2 is consumed due to high primary production rates. Other runoff events (some with comparable intensity) had a significant impact on the CO_2 level,

but either had runoff peaks during night hours (e.g., events no. 4 and 13) or occurred in the autumn (e.g., events no. 16 and 18), when the SR is low. As a result, the simultaneous metabolic signals made the hysteresis plots complex. We suggest that the different hydrological responses on CO_2 are related to the timing of an event, during what season the event occurs, whether the event follows an extended dry period, or when during the day (day or night) the runoff peaks. The absence of a clear response in CO_2 concentration for most of the events, suggests that variations in runoff did not have a major control on ditch CO_2 dynamics and were instead overridden by the stronger light and temperature induced metabolic control operating at the diel timescale (Bernal et al., 2022).

The importance of *in-situ* metabolic processes controlling CO_2 dynamics was finally supported by the event based and control integrated PCA analysis (Figure 7). Both the magnitude in CO_2 concentration and range in diel CO_2 concentration amplitude were closely related to both daily total SR and mean water temperature. In contrast, the PCA displayed low influence on CO_2 by any of the hydrological metrics. The elevated *in-situ* control on ditch CO_2 following forest harvest was further evident when comparing continuous data collected from DC2 with the completely forested catchment C2 included in the KCS and located within 10 km from DC2. C2 is representative for the conditions at DC2 as they were prior to the forest harvest and is used as one of two forest control catchments within the experimental design of the TEA. From the comparative analysis between DC2 and C2 clear differences were observed, both in concentration magnitude and amplitude of the diel CO_2 cycles, but also in diel water temperature patterns. This suggests that the collective conditions after clear-cut, with elevated solar radiation exposure and increased DOC and nutrient concentrations alter the *in-situ* ditch ecosystem function, and by that enhancing the importance of temperature- and light-induced metabolic control on the CO_2 dynamics. Furthermore, the slope of the logCO_2-logQ relationship observed in C2 was more negative than in DC2 (−0.12 and −0.08, respectively) suggesting a higher runoff control on CO_2 concentration (Supplementary Figure 9). This comparative part of the study further supports our hypothesis that *in-situ* metabolism is a key driver of aquatic CO_2 dynamics in clear-cut catchments.

We acknowledge that the current study only represents a single ditch and observed patterns are likely site-specific. However, we believe our finding of an increased metabolic control on CO_2 dynamics in forest ditches and streams following clear-cut harvest should be valid across regions with similar climatic conditions and forest management. The increased short-term CO_2 concentration dynamics following forest harvest will also lead to altered emissions patterns. To what extent these altered patterns will influence total annual emissions is uncertain and will require detailed investigations. Klaus et al. (2018) found that despite significant increases of CO_2 in groundwater of clear-cut affected catchment soils, no change in GHG (including CO_2) fluxes in adjacent streams were detected within 3 years after the treatment. The authors explained the mismatch in patterns between ground- and stream water with that the trees left in the riparian zones most likely acted as an effective buffer zone mitigating stream GHG emissions. The findings of the current study suggest that increased aquatic productivity might play a role in consuming the elevated soil CO_2 export following forest harvest. Thus, inorganic C will be converted into organic forms shortly after being transported across the soil-water interface.

We conclude that CO_2 concentration dynamics in forest ditches affected by clear-cut harvest are driven by a complex interplay of light and hydrologically induced processes. Despite the common perception of forest ditches as nutrient-poor systems typically showing low metabolic rates, our findings suggest that metabolism, and primary production specifically, exerts significant control on short-term ditch CO_2 concentration dynamics. In contrast, variations in discharge displayed a comparatively less dominant influence on the variation in CO_2 concentrations. The high CO_2 concentration dynamics and the associated metabolic controls should be considered when scaling CO_2 emissions across boreal landscapes impacted by clear-cut forestry. To improve our understanding of these processes, we recommend that future studies combine measures of C export/emission with *in-situ* metabolism and that these are conducted over longer time scales (i.e., >single growing season). Overall, our results emphasize the need for more comprehensive and detailed investigations of the factors regulating CO_2 dynamics in forest ditches and their implications for the landscape-scale C budgets.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

AZ and MW designed the study. AZ carried out most of the fieldwork, analyzed the data, and wrote the first draft of the manuscript. MW, KE, EL, EM, and HL further provided scientific insight to the analysis and interpretation of the data. All authors commented on earlier versions of this paper.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/frwa.2023.1250068/full#supplementary-material

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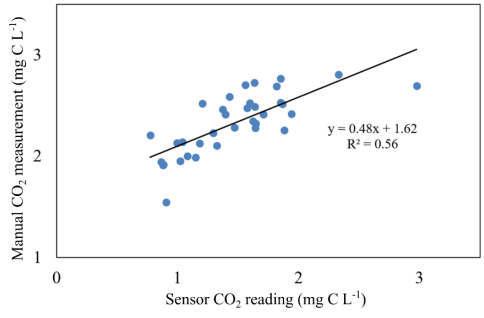
SUPPLEMENTARY INFROMATION

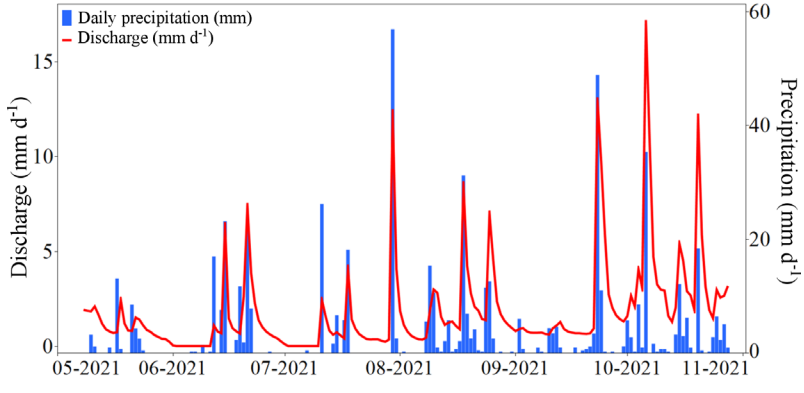
Metabolic processes control carbon dioxide dynamics in a boreal forest ditch affected by clear-cut forestry

Supplementary Table 1. Catchment characteristics of DC2 and C2 (Laudon et al., 2021).

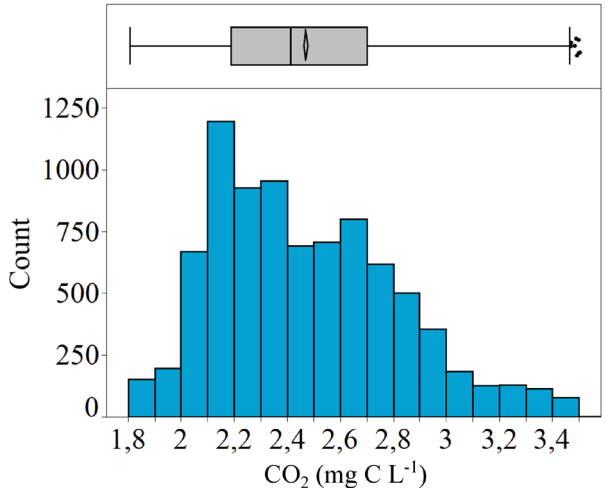
Catchment	Area (ha)	Altitude range (masl)	Forest (%)	Clear-cut (%)	Peat (%)	Till soils (%)	Thin soils (%)	Ditch length (m)	Ditch density (m ha^{-1})	Forest manag. type
DC2	4.4	200-175	0	100	0	100	0	1117	252	Clear-cut
C2	12	310-235	100	0	0	84	16	800	67	Unmanaged



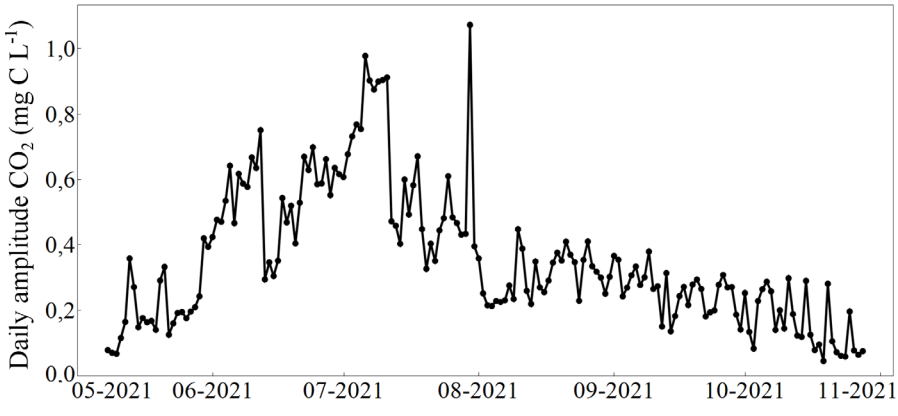
Supplementary Figure 1. Linear regression of sensor CO_2 readings against manual CO_2 measurements. The equation was used to obtain calibrated results of CO_2 concentration.



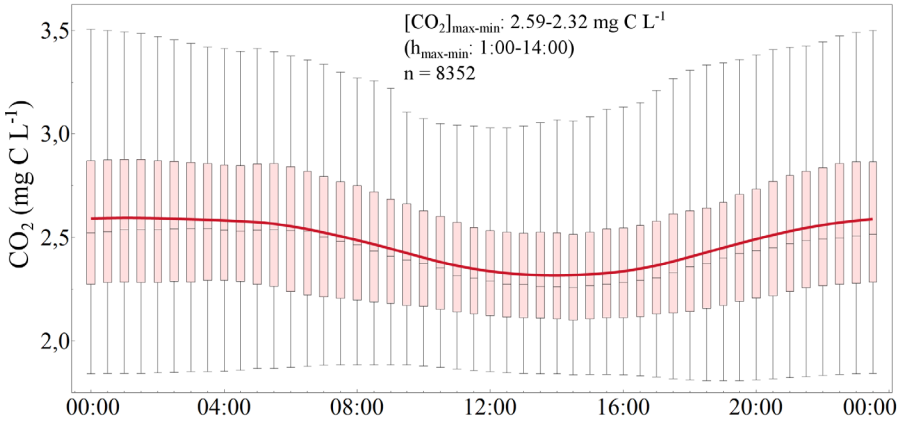
Supplementary Figure 2. Daily discharge and precipitation for the full study period 8 May - 28 October 2021 in the study catchment DC2.



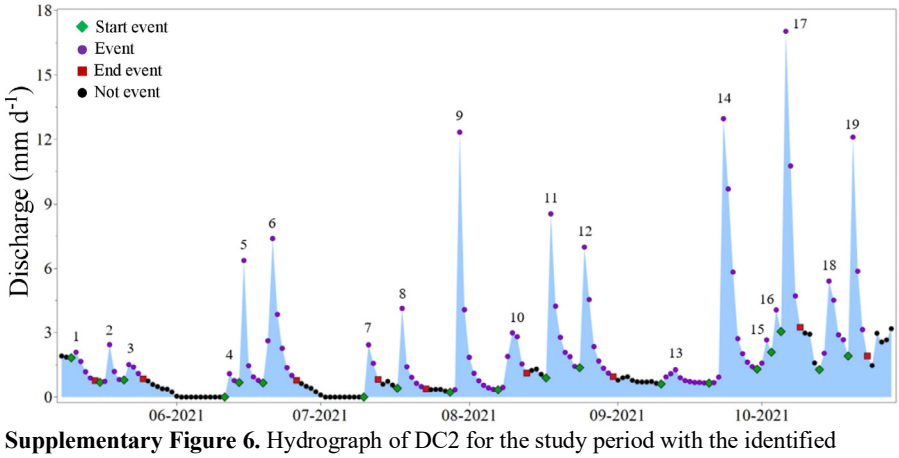
Supplementary Figure 3. Distribution in ditch CO_2 concentration data at DC2 for the study period (n = 8352). Upper boxplot shows mean (diamond), median (line), IQR (box), 10th and 90th percentiles (whiskers).



Supplementary Figure 4. Daily amplitude in the ditch CO_2 concentration in DC2 for the full study period 8 May - 28 October 2021



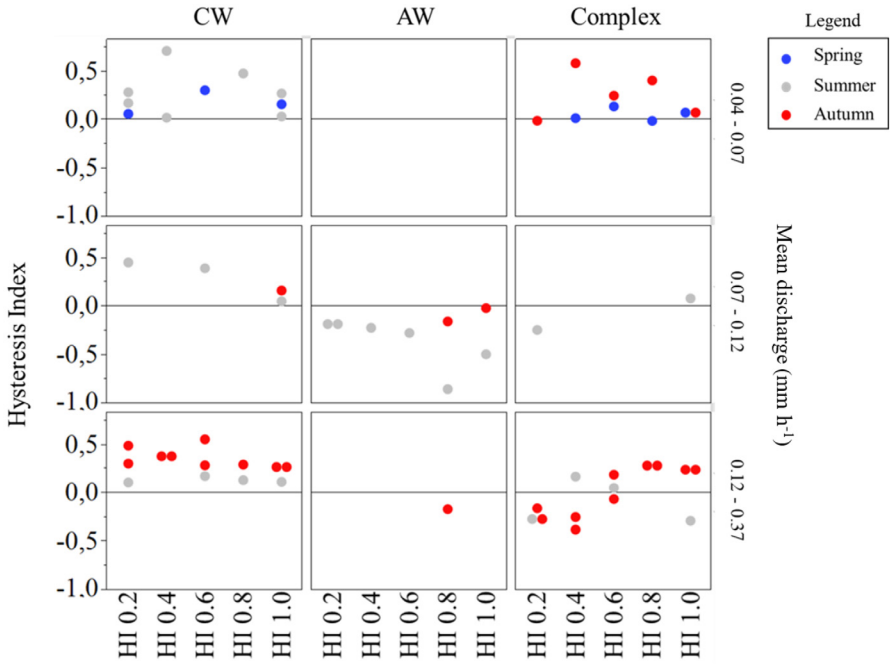
Supplementary Figure 5. Distribution in ditch CO_2 concentrations at DC2 over the full study period presented on a diel (24 hr) basis with each boxplot representing a 30 min period. The red line displays the mean diel CO_2 concentration pattern.



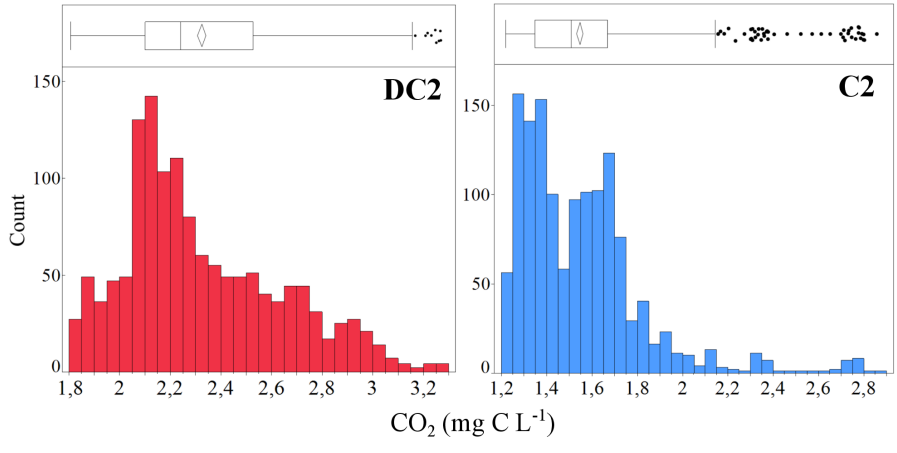
Supplementary Figure 6. Hydrograph of DC2 for the study period with the identified hydrological events (n=19).

Supplementary Table 2. Characteristics of the different hydrological events. Note: discharge (Q) is expressed in mm h^{-1}, CO_2 in mg C L^{-1}, water temperature (WT) in °C, electrical conductivity (EC) in µS cm^{-1}, total short-wave radiation (SR) in kWh m^{-2}, accumulated precipitation (P) in mm. Shape (CW: clockwise, AW: anti-clockwise, Complex: both CW and AW).

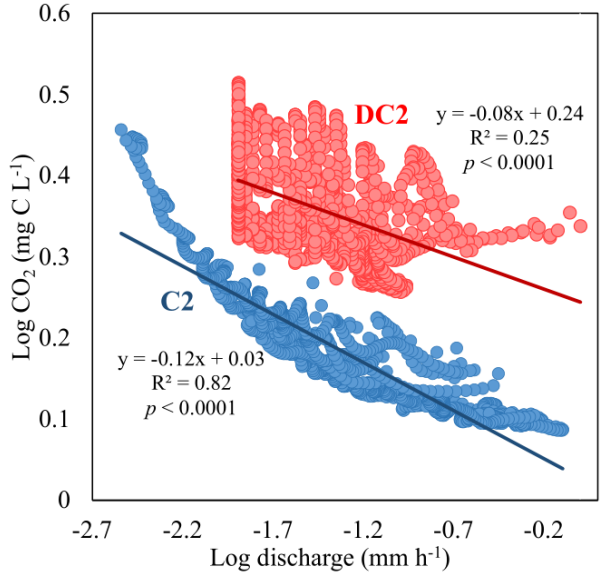
No.	Start	End	Days	Season	Q_{max}	Q_{Mean}	Q_{Range}	$CO_{2\,Mean}$	$CO_{2\,Range}$	WT_{Mean}	EC_{Mean}	EC_{Range}	SR_{Tot}	P_{Tot}	Shape	Q_{Peak_hour}	CO_{2Peak_hour}
1	16-May	21-May	5	Spring	0.15	0.05	0.12	2.16	0.41	8.2	28.9	3.7	27.83	13.8	CW	01:00	21:00
2	21-May	25-May	4	Spring	0.06	0.05	0.04	2.04	0.45	6.2	29	1.5	19.72	15.7	Complex	07:00	00:00
3	11-Jun	14-Jun	3	Spring	0.08	0.04	0.08	2.37	0.96	12	31	2.6	22.72	17.2	CW	10:00	01:00
4	14-Jun	19-Jun	5	Spring	0.68	0.08	0.66	2.29	0.79	11.1	35.1	18.2	31.35	33	Complex	02:00	02:00
5	19-Jun	26-Jun	7	Summer	1	0.11	0.98	2.42	0.99	11.8	37	18.7	37.2	43.1	AW	07:00	00:00
6	10-Jul	14-Jul	4	Summer	0.25	0.05	0.25	2.89	1.06	15.7	37.2	10.4	31.15	26.2	CW	13:00	00:00
7	17-Jul	23-Jul	6	Summer	0.41	0.06	0.39	2.79	0.76	13.3	37.3	15.4	45.45	23.9	CW	02:00	22:00
8	28-Jul	07-Aug	10	Summer	1.43	0.09	1.43	2.88	1.1	13.4	37.7	28.2	63.34	59.6	CW	14:00	01:00
9	07-Aug	13-Aug	6	Summer	0.23	0.07	0.22	2.74	0.47	13.2	36.3	14.9	26.8	32.5	AW	19:00	02:00
10	17-Aug	24-Aug	7	Summer	0.87	0.13	0.84	2.61	0.56	12.1	42.8	30	23.28	47.5	CW	08:00	03:00
11	24-Aug	31-Aug	7	Summer	0.46	0.11	0.42	2.55	0.51	10.4	42.3	17.7	29.59	26.8	Complex	02:00	00:00
12	10-Sep	20-Sep	10	Autumn	0.06	0.03	0.04	2.25	0.35	8.1	33.3	2.6	21.38	14.7	Complex	22:00	07:00
13	20-Sep	30-Sep	10	Autumn	1.29	0.16	1.27	2.32	0.52	7.6	42.4	35.2	11.18	65.5	Complex	19:00	06:00
14	30-Sep	03-Oct	3	Autumn	0.13	0.08	0.08	2.4	0.33	7.9	38.4	3.9	1.08	9.6	AW	10:00	09:00
15	03-Oct	05-Oct	2	Autumn	0.26	0.13	0.18	2.31	0.23	8.4	40.3	4	1.29	8.8	AW	13:00	08:00
16	05-Oct	09-Oct	4	Autumn	1.92	0.37	1.82	2.31	0.34	8.1	46.7	18.1	4.04	38	Complex	18:00	23:00
17	13-Oct	19-Oct	6	Autumn	0.35	0.13	0.3	2.17	0.35	4.1	40.5	8.8	6.24	25.3	CW	15:00	00:00
18	19-Oct	24-Oct	5	Autumn	1.23	0.21	1.18	2.09	0.28	2.7	38	9.1	3.51	19	CW	13:00	05:00
19	24-Oct	26-Oct	2	Autumn	0.13	0.09	0.07	2.15	0.2	2	35.9	4.9	1.27	9.2	CW	04:00	21:00



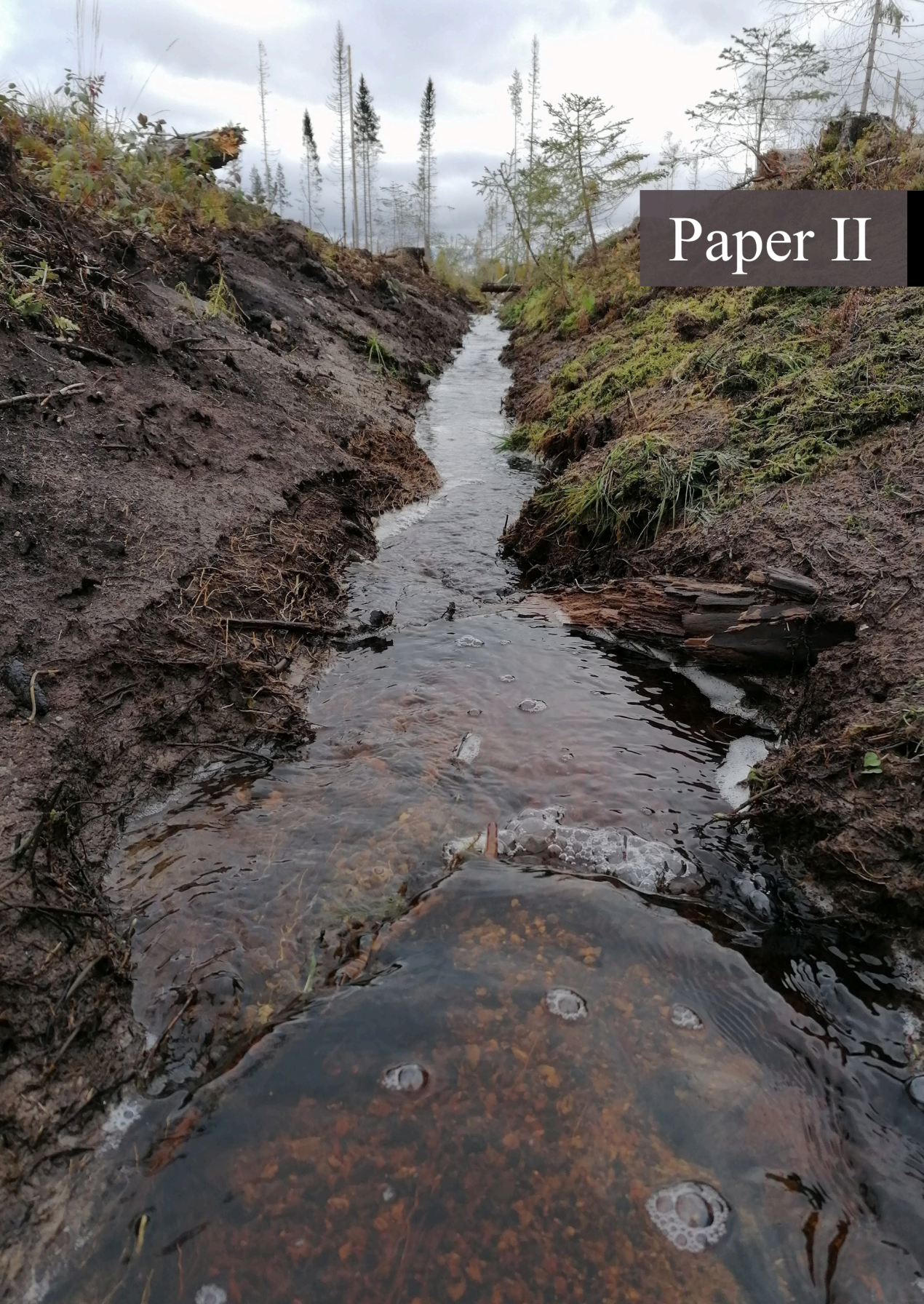
Supplementary Figure 7. Events divided by shape and mean discharge, showing the varying hysteresis indexes (HI) over the events; seasons are indicated by color.



Supplementary Figure 8. Distribution in CO_2 concentration data for DC2 (left-red histogram) and C2 (blue-right histogram) over the 2-month comparison (May-June) (n = 1357). Upper boxplot shows mean (diamond), median (line), IQR (box), 10^{th} and 90^{th} percentiles (whiskers).



Supplementary Figure 9. Log discharge vs Log CO_2 for the sites DC2 (red) and C2 (blue) based on the data collected during the first two months of the study period (May-June). Regression equations and R2 values are also indicated.



Paper II

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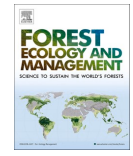

Ditch cleaning in boreal catchments: Impacts on water chemistry and dissolved greenhouse gases in runoff

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ABSTRACT

Ditch cleaning (DC) is a forestry practice commonly conducted in boreal regions that aim to lower groundwater tables (GWT) in waterlogged soils, thereby maintaining or improving forest growth. However, there is limited information on the impact of DC on water chemistry and dissolved greenhouse gases (GHG) in draining ditch networks. Based on a repeated synoptic sampling of a paired catchment design we here evaluated water chemistry and GHG data in ditch waters from 25 cleaned sites (being cleaned 1–4 years prior to sampling) and 25 non-cleaned reference sites (REF). The sampled sites were further selected to test whether there were any differences in the DC effect if the operations were conducted in forested or clear-cut areas. Across all sites, we found that DC sites exhibited higher pH, sulfate and calcium concentrations than REF sites. Also, lower dissolved carbon dioxide and higher nitrous oxide concentrations were observed in DC sites. In forested areas, DC sites exhibited significantly higher calcium and potassium concentrations, along with reduced levels of methylmercury and carbon dioxide. In clear-cuts, sulfate concentrations were significantly elevated in the DC sites. We suggest that the observed differences in water chemistry and GHGs between DC and REF sites were induced by the DC and largely driven by lower GWT following DC, resulting in deeper groundwater flow paths through more mineral-rich soil layers, and altered redox conditions. Also, the removal of organic rich sediments and vegetation from the ditches themselves may affect water chemistry and GHGs e.g. by decreasing the formation of methylmercury and carbon dioxide.

1. Introduction

Extensive areas in Sweden have been historically drained to overcome the prevalent waterlogged conditions of forested peatlands and wet mineral soils, and in turn increase forest growth (Sikström and Hökkä, 2016; Norstedt et al., 2021). The Swedish forest landscape is among the most drained in the world, with a man-made network of ditches that spans the entire country, excluding the mountainous regions (Peatland and Climate Change, 2023). About 1.5 million hectares (ha) of peat soils are estimated to have been drained in Sweden by ditching since the mid-19th century (Wesström et al., 2017). However, over time a large share of those ditches have lost the original drainage capacity due to sediment accumulation and vegetation overgrowth (Hånell, 2009). In this context may ditch cleaning (DC) work as a measure to lower the groundwater table (GWT) and maintain or improve forest growth. Approximately 10,000 ha of forest land was ditch cleaned annually in Sweden between 2013 and 2017 (Swedish NFI, 2020; Tong et al., 2022b).

Clear-cutting is the most common harvesting method in Sweden. When a large share of the standing vegetation is removed, the resulting loss in tree transpiration significantly affects the catchment hydrology by raising the GWT and increasing runoff rates (Bosch and Hewlett, 1982). Although increased sunlight exposure on the exposed ground can lead to higher evaporation rates, this effect is relatively minor compared to the reduction in transpiration and the increase in snow accumulation when trees are removed (Murray and Buttle, 2003). Moreover, the absence of trees reduces the nutrient uptake, increasing the availability of nutrients and other elements in the root zone, which can be hydrologically mobilized with shallower groundwater flow paths following harvest. Additionally, higher nutrient levels in streams after clear-cutting are not only due to the loss of vegetation uptake but also result from the decomposition of logging residues left on site (Nieminen, 2004; Schelker et al., 2016).

DC is conducted as a way to counteract a rising GWT when the tree

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layer is removed, and by that promote the establishment of the new tree generation (Sikström and Hökkä, 2016). Historically, both peat and mineral soils have been ditched in Sweden, thus DC may occur on either soil type. DC is typically undertaken in direct connection to harvest although it can be performed anytime during the rotation period. During the DC operations ditches are typically cleaned down to their original depth by removing vegetation and sediment with the aim to restore the drainage capacity. A lowering of the GWT maintain or promote aerobic conditions in the upper soil layers, which is required for new trees to be established and for organic material to be mineralized. To what extent the GWT is affected by DC is related to how the cleaning is conducted, but also to the original design of the individual ditch networks as well as catchment specific characteristics such as soil types (including thickness of peat layer and peat type). Studies have shown that DC operations lower the GWT in areas with shallow peat underlain by mineral soil (Ahti and Päivänen, 1997; Koivusalo et al., 2008; Laudon et al., 2023; Tong et al., 2022a), while no, or small effects on the GWT have been observed in areas with thick (> ca.1 m) peat layers (Koivusalo et al., 2008; Tong et al., 2022b), or where the GWT is already low due to e.g. high tree water uptake or deep ditches.

Although DC can be an efficient measure to maintain forest productivity (Sikström and Hökkä, 2016; Sikström et al., 2020), it can also have unexpected impacts on water chemistry in downstream surface waters (Nieminen et al., 2018). The removal of ditch vegetation and sediments may intensify the hydrological mobilization of particles through erosion of the exposed bare soils (Joensuu, 2013). Also, lowering the GWT by DC change the groundwater flow paths which in turn could affect the chemical composition in runoff (Joensuu et al., 2002, 1999; Mäkitalo, 2009; Manninen, 1998; Nieminen et al., 2010; Wesström et al., 2017). Nieminen et al. (2018) synthesized findings from the available literature (mainly derived from Finland) to assess the impact of ditch network maintenance operations (i.e., DC and supplementary ditching in rare cases) on runoff water chemistry in drained boreal peatland forests. The literature review underlined how the impact of DC on surface water chemistry is poorly understood. Only a few recurring patterns induced by DC were recognized, such as increases in suspended solids (SS) and particulate nutrients (specifically nitrogen (N) and phosphorus (P)) (Joensuu et al., 2002; Nieminen et al., 2010; Finer et al., 2010). Furthermore, post DC concentrations of dissolved organic N have been shown to decrease, but in contrast, inorganic fractions, and especially ammonia (NH_4), was found to increase after DC. As a consequence, the total dissolved N concentration following DC operation remained unchanged (Joensuu et al., 2002). In a more recent study conducted in boreal Sweden, Laudon et al. (2023) reported DC-induced decreases in total N and P as well as no effect on SS, further highlighting contrasting findings compared to previous research. Divergent results have also been observed when assessing the impacts on dissolved organic carbon (DOC). While some studies have found significant decreases in DOC after DC (e.g. Nieminen et al., 2010; Joensuu et al., 2013; Hansen et al., 2013; Laudon et al., 2023), other studies report non-significant effects on DOC (e.g. Manninen, 1998; Manninen, 1995). DC may also have consequences on the transport of sulfate (SO_4) from soils to downstream aquatic ecosystems. When lowering the GWT, reduced sulfur forms in the upper soil layers are oxidized leading to enhanced runoff SO_4, acidity and mobilization of metals (Fanning et al., 2017).

Another water quality concern following DC that is even less studied, is effects on total mercury (THg) and its bioavailable form methylmercury (MeHg). MeHg are prone to bioaccumulation and biomagnification in aquatic food webs, elevating contaminant levels in aquatic resources and posing a risk to human consumption (WHO, 2017, 2020). Forestry operations in general can be an important contributor of MeHg in aquatic biota (Eklöf et al., 2016). Bishop et al. (2009) suggested that 9–23 % of the Hg in Swedish freshwater fishes is a consequence of forest harvest. Concerns have been raised that DC may further raise the concentrations of Hg and MeHg in surface waters and biota (Wesström et al., 2017). The few existing studies exploring DC effects on THg and MeHg in ditch waters have explored different time scales but show divergent results. While Hansen et al. (2013) found initial increases in concentrations of THg and MeHg in the ditch water in one of the studied sites during a few days following DC, Laudon et al. (2023) observed decreased THg concentrations within two years following the DC operation. Bitenieks et al. (2022) also found lower sediment MeHg concentrations in cleaned ditches compared to non-cleaned ones.

In addition to effects on water chemistry, DC operations might have other consequences for the function of ditch networks. For example, boreal forest ditches and streams are known to emit greenhouse gases (GHG) (carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)) to the atmosphere, emissions that are found significant (<20 % of net terrestrial C uptake) when estimating complete landscape GHG balances (Audet et al., 2020; Wallin et al., 2018; Butman et al., 2016). Any information on how DC may influence these emissions is currently limited. The only studies we are aware of found either no significant impact on ditch CO_2 and CH_4 emissions in the first two years following DC (Tong et al., 2022b) or augmented CH_4 emissions from moss-free ditches compared to moss-covered ditches (Rissanen et al., 2023).

Due to the limited number of studies, and sometimes divergent results regarding DC and its influence on downstream surface water chemistry and ditch GHG emissions, there is a clear need for a more comprehensive data basis and knowledge. Such information is required for making well-founded decisions on how to manage the boreal landscape including a dense ditch network. This study aims to contribute to this knowledge gap by evaluating data collected in a paired design of ditches across central Sweden, where half of the ditches has been ditch cleaned (DC) and half not (REF). The specific research objectives are to 1) assess whether there are any significant differences in ditch water chemistry and dissolved GHG concentrations between DC and REF ditches, and 2) explore whether any DC induced effects differed if the DC operations were conducted in forested or clear-cut sites.

2. Methods

2.1. Sampled ditches and catchment characteristics

The study was based on a regional sampling effort conducted according to a paired design with sampling in 25 ditches that were cleaned one to four years prior to first sampling (DC), and 25 reference ditches with no signs of recent DC activities (REF). All ditches were located in east-central Sweden (Fig. 1). The 25 pairs were further selected to investigate any effects of whether the DC was conducted in forested (henceforth referred to as “forested sites”; 13 pairs) or clear-cut parts of the catchments (henceforth referred to as “clear-cut sites”; 12 pairs). The identification of suitable DC sampling sites was based on information from various landowners, mainly forest companies. In these practical DC operations, ditches are normally cleaned down to the original ditch depth and in accordance with national guidelines to ensure good environmental consideration (Swedish Forest Agency, 2019). The corresponding REF catchments were selected to be as similar as possible to the DC catchments in terms of catchment size, land use, and vegetation composition. They were further selected to be near-by located (ca 2 km on average) but hydrologically disconnected from the DC catchment within the pair. Ditch sampling points were selected to be representative of each specific site. For cleaned ditches, sampling was conducted at the most downstream location to maximize DC effects while avoiding sections of high turbulence and contamination from particles and algae.

The 50 sampled catchments had a mean area of 40 ha (range 2–176 ha) and are typical for the region with domination of coniferous forest on till soils. The elevation at the sampling ditch sites varied from 30 masl to 287 masl with the overall mean of 133 masl. Mean annual precipitation (MAP) was 735 mm (range: 631–844) mm and mean annual temperature (MAT) was 3.9 °C (range: 1.9–6.3 °C) across the catchments. The average main land cover was coniferous forest (90 %).

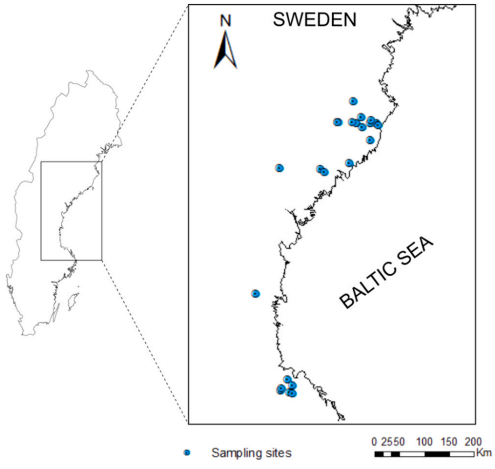

Fig. 1. Location of the sampling sites where ditches were cleaned (DC, n=25). The paired reference sites (REF, n=25) are situated in close vicinity (ca 2 km on average) to each of the cleaned sites and overlap on the map. The map was created in ArcGIS (ESRI).

Soil types varied across the catchments pairs but consisted on average by till (61 %) followed by peat (11 %), sand (7 %) and clay (4 %). Catchments were characterized using ArcGIS software (version 10.8.2) with data from SLU geodata extraction tool and Swedish authorities such as Lantmäteriet (Swedish land registry), Swedish EPA (Environmental Protection Agency), SGU (Swedish geological survey), and the Swedish Forestry Agency. Catchments were then visually inspected and, when needed, manually adjusted using SMHI datasets SVAR2016 and SVAR2022, national hydrography surveys, and accumulation lines from SLU VIVAN2 model. Thirty-year temperature and precipitation averages were obtained from the SMHI open data API using their PTHBV model. Land cover and soil type data were extracted from the 2018 Swedish National Land Use Layer and the SGU base soil type layer, respectively, and reclassified according to the categories in Table 1. Peat depth data was derived from the SLU peat map, based on an elevation-derived soil moisture map. The overall catchment characteristics for all, forested and clear-cut catchments, divided by DC and REF, respectively, are given in Table 1, while detailed information for each study catchment are provided in Table S1.

2.2. Analyses and calculations

Grab samples were collected from the 50 ditches on three different occasions: June 2021 (1), September 2021 (2), and June 2022 (3). These samples were analysed for 17 different water chemistry variables, two optical measurements and three dissolved GHGs, as detailed in Table 2. Each sampling campaign lasted for ca. 12 days during periods when flow did not differ too much i.e. by avoiding periods of high flows due to spring floods or rainstorm events. For the dissolved GHGs, CH_4 was only sampled on occasion 2 and 3, N_2O was only sampled on occasion 2, and CO_2 was only sampled on occasion 3.

For all water chemistry (except for mercury), grab samples were collected in low-density polyethylene bottles that were rinsed three times with ditch water prior to sampling. Samples were stored dark and cold from sampling until analysis ($\leq$ five days). Water chemistry was analysed using standard methods at the accredited laboratory of the department of Aquatic sciences and assessment at the Swedish University of Agricultural Sciences. The unitless absorbance measure at 420 nm (Abs_{420}) was converted to the absorbance coefficient (α) as follows:

$$\alpha = \frac{A}{L}$$

where α is expressed in m^{-1}, A is the unitless absorbance measure, and L is the length of the quartz cuvette used (5 cm). The α value was further divided by the total organic carbon (TOC) concentration to calculate specific absorbance at 420 nm (Abs_{420}/TOC, L mg C^{-1} m^{-1}) as a measure of organic carbon characteristics (Köhler et al., 2013; Weyhenmeyer et al., 2012).

Samples for THg and MeHg were collected using an ultra-clean sampling protocol. Single use plastic gloves were used, and water samples were collected in acid-washed Teflon bottles and sent, on the same day or the day after sampling, to the Swedish Environmental Research Institute (IVL) for immediate preservation. Analyses of THg followed the EPA 1631 method version E, which involves oxidation with BrCl, reduction to Hg0 with $SnCl_2$, double amalgamation, and subsequent determination through atomic fluorescence spectrometry. Analyses of

Table 1
Catchment characteristics, land use and soil type distributions for the DC and REF catchments expressed as mean (min-max) values.

	ALL		FORESTED		CLEAR-CUT	
	DC	REF	DC	REF	DC	REF
Catchment area (ha)	37 (2–136)	44 (5–176)	41 (3–136)	38 (13–96)	33 (2–81)	50 (5–176)
Elevation at sampling point (m)	132 (30–279)	134 (32–287)	149 (37–265)	148 (40–281)	114 (30–279)	119 (32–287)
MAP (mm)	732 (631–844)	738 (662–844)	750 (631–844)	752 (665–844)	713 (657–820)	724 (662–822)
MAT (°C)	3.9 (2.1–6.3)	3.9 (1.9–6.3)	3.7 (2.1–6.3)	3.6 (1.9–6.2)	4.2 (2.1–6.3)	4.2 (2.1–6.3)
*Land cover (%)**						
Coniferous forest	92 (71–100)	89 (72–98)	92 (71–100)	88 (72–97)	93 (82–100)	89 (77–98)
Deciduous forest	3 (0–15)	4 (0–15)	4 (0–15)	3 (0–15)	2 (0–10)	4 (0–14)
Open wetland	2 (0–19)	4 (0–21)	2 (0–19)	5 (0–21)	2 (0–9)	3 (0–8)
Exploited land	1 (0–6)	2 (0–5)	1 (0–5)	2 (0–4)	2 (0–6)	2 (0–5)
Other	1 (0–7)	2 (0–8)	1 (0–3)	2 (0–6)	1 (0–7)	2 (0–8)
*Soil type (%)**						
Till	63 (6–100)	58 (0–99)	58 (6–100)	53 (0–96)	68 (29–100)	62 (0–99)
Peat	8 (0–41)	14 (0–45)	8 (0–41)	10 (0–45)	9 (0–25)	18 (0–35)
Sand	6 (0–42)	7 (0–62)	11 (0–42)	8 (0–43)	2 (0–18)	5 (0–62)
Silt	5 (0–44)	5 (0–89)	10 (0–44)	9 (0–89)	0 (0–0)	0 (0–0)
Clay	5 (0–70)	2 (0–14)	0 (0–2)	1 (0–12)	9 (0–70)	3 (0–14)
Other	12 (0–60)	16 (0–61)	12 (0–40)	19 (0–61)	12 (0–60)	13 (0–55)
Peat depth (cm)						
Catchment mean	23 (12–44)	26 (12–51)	21 (12–44)	26 (13–51)	24 (12–40)	25 (12–48)
Sampling point	46 (11–78)	44 (13–81)	39 (11–70)	39 (15–57)	53 (13–78)	48 (13–81)

* Percentage of catchment area

Table 2
Water chemistry and dissolved greenhouse gas (GHG) data in ditch waters collected from the 25 ditch cleaned (DC) and 25 reference (REF) sites. Data are presented as mean values based on all sampling occasions with standard deviation in brackets as well as min-max, while the test of significance is based on GLMM analysis, which tests the difference between paired sites. The dataset was further sub-grouped dependent on whether the DC operations were conducted in forested areas (13 pairs) or clear-cut areas (12 pairs). Color-coding is used to highlight when Δ-values for each pair are significant different from zero according to the GLMM ($p < 0.05$): green indicates significantly higher values in the DC sites compared to their REF sites, while red indicates significantly lower values in the DC sites compared to their REF sites. All p-values from the GLMM are reported in Table S2.

		All		Forested		Clear-cut	
		DC	REF	DC	REF	DC	REF
Acidity and Ions	pH (-)	5.8 (1.2) 3.8-8.1	5.2 (1.2) 3.8-7.7	5.9 (1.2) 3.8-8.1	5.3 (1.1) 3.8-7.6	5.7 (1.2) 4.1-7.5	5.2 (1.2) 3.8-7.7
	EC (µS cm^{-1})	96.7 (110.5) 13-491	88.2 (116) 16-516	77 (75.8) 13-318	70.5 (81.3) 16-262	118.7 (137.4) 13-491	108.1 (144.1) 22-516
	SO_4^{2-} (µeq L^{-1})	211.3 (352.3) 14-1700	105.4 (287.7) 5-1600	215 (339.2) 14-1700	151.5 (387.3) 15-1600	207.1 (371.6) 16-1400	53.8 (69.2) 5-270
	Cl^- (µeq L^{-1})	29.7 (19.2) 9.6-100	61.8 (153.2) 3.5-990	25 (13.9) 9.6-73	30.6 (15.3) 9.6-71	34.9 (22.9) 12-100	96.7 (218.8) 3.5-990
	F^- (mg L^{-1})	0.27 (0.27) 0.03-1.3	0.26 (0.28) 0.03-1.3	0.18 (0.09) 0.03-0.4	0.20 (0.11) 0.03-0.5	0.36 (0.35) 0.03-1.3	0.33 (0.38) 0.03-1.3
	Ca^{2+} (µeq L^{-1})	808.6 (1234.2) 38-5490	600.1 (1145.3) 31-4990	563.8 (741) 38-3100	440.3 (759.4) 31-2500	1082.1 (1586.1) 43-5490	778.8 (1453.6) 45-4990
	Mg^{2+} (µeq L^{-1})	125.5 (111.5) 17-520	97.5 (99.5) 16-470	115.5 (105.8) 17-520	95.0 (107.3) 16-470	136.6 (118.2) 29-400	100.3 (91.4) 24-370
	Na^+ (µeq L^{-1})	104.7 (66.2) 29-400	122 (147.4) 28-960	86.5 (35.6) 29-170	100.4 (67.7) 37-370	125.1 (84.8) 40-400	146.1 (201.2) 28-960
	K^+ (µeq L^{-1})	19.8 (15.2) 2.8-87	22.9 (25.7) 0.5-100	16.2 (10.7) 2.8-46	10.5 (10.5) 0.5-38	13.9 (18.4) 4.9-87	36.8 (30.4) 3.1-100
Mercury	THg (ng L^{-1})	9.82 (7.43) 1.5-39	9.37 (8.53) 1.3-48	7.15 (3.73) 1.5-15	6.36 (2.15) 2.7-11	12.81 (9.27) 1.9-39	12.74 (11.37) 1.3-48
	MeHg (ng L^{-1})	0.76 (1.26) 0.03-8.9	0.63 (0.67) 0.03-3	0.63 (0.85) 0.03-2.9	0.71 (0.69) 0.03-2.7	0.89 (1.59) 0.03-8.9	0.53 (0.65) 0.03-3
Absorbance, carbon and GHGs	Abs_{420} (5 cm^{-1})	0.74 (0.56) 0.1-2.8	0.81 (0.64) 0.1-3.1	0.52 (0.33) 0.1-1.2	0.51 (0.26) 0.1-1.1	0.98 (0.66) 0.1-2.8	1.15 (0.76) 0.2-3.1
	TOC (mg L^{-1})	38.8 (25.8) 4.6-108	39.5 (29.7) 8.5-165	28 (26.5) 4.7-62.7	26 (10.1) 8.5-50.7	50.9 (29.1) 4.6-108	54.6 (36.6) 10.0-165
	Abs_{420}/TOC (L mg C^{-1} m^{-1})	0.37 (0.10) 0.2-0.7	0.40 (0.10) 0.1-0.6	0.36 (0.07) 0.2-0.5	0.38 (0.11) 0.1-0.6	0.38 (0.12) 0.2-0.7	0.41 (0.08) 0.2-0.6
	CO_2-C (mg L^{-1})	2.3 (1.3) 1-6.1	3.9 (1.7) 1.6-7.2	1.7 (0.5) 1-2.7	4 (1.7) 1.6-7.2	3.1 (1.6) 1.2-6.1	3.8 (1.7) 2.2-6.8
	CH_4-C (µg L^{-1})	9 (9.8) 0.7-54.8	32.4 (105.8) 1.3-723.3	6.7 (5.8) 0.7-18.8	18.0 (28.1) 1.9-105.8	11.6 (12.6) 1.8-54.8	48.9 (151.9) 1.3-723.3
	N_2O-N (µg L^{-1})	2.3 (5.6) 0.4-28.5	0.8 (1.1) 0.3-5.7	1.1 (1.6) 0.4-6.3	0.5 (0.1) 0.4-0.7	3.6 (7.9) 0.5-28.5	1.1 (1.6) 0.3-5.7
Nutrients	TN (µg L^{-1})	1521 (1846) 132-11800	1146(1117) 307-6080	844 (761) 132-4210	671 (282) 307-1300	2279 (2358) 144-11800	1676 (1431) 373-6080
	NH_4-N (µg L^{-1})	93.0 (431.4) 1.5-2630	126.6 (365.9) 5-2190	13.8 (12.9) 1.5-67	17.9 (25.0) 5-159	181.5 (620.5) 1.5-2630	248.0 (508.5) 5-2190
	NO_3-N (µg L^{-1})	18.4 (90.6) 0.5-10500	4.3 (9.7) 1-910	4.6 (20.8) 0.5-1800	1.4 (2.7) 1-210	33.9 (129.2) 0.5-10500	7.5 (13.2) 1-910
	TP (µg L^{-1})	50.1 (67.6) 4.6-353	68.8 (162.1) 4.5-981	30.7 (23.2) 5.8-113	20.1 (13.0) 4.5-76.4	71.7 (91.3) 4.6-353	123.2 (224.8) 5.9-981
	PO_4-P (µg L^{-1})	11.8 (30.2) 0.5-195	35.4 (113.5) 0.5-694	4.7 (5.5) 0.5-27	4.1 (5.4) 0.5-25	19.9 (42.5) 0.5-195	70.4 (159.1) 0.5-694

MeHg followed the EPA 1630 method, which involves primary separation through distillation followed by ethylation in the aqueous phase, gas chromatographic separation, and atomic fluorescence spectrometry. Sampling for dissolved concentrations of CO_2, CH_4 and N_2O was conducted using a headspace equilibration method (Hope et al., 2004), where bubble-free water samples were drawn into a 60 mL polypropylene syringe equipped with a three-way-stopcock. 30 mL of sample water and 30 mL of ambient air was equilibrated in the syringe by shaking it vigorously for one minute. The equilibrated headspace gas was extracted into a 12 mL evacuated exetainer tube (for CO_2 and CH_4) or a 22 mL GC vial (for N_2O). Analyses for CO_2 and CH_4 were made on a portable greenhouse gas analyser (Gas scouter, Picarro) using a closed loop system (Wilkinson et al., 2019), and for N_2O on a gas chromatograph (Perkin Elmer Clarus 500) equipped with an autosampler (Turbomatrix 110) and FID/ECD detectors. Dissolved concentrations for each gas were calculated using the gas-specific Henry's constants (Weiss, 1974; Wiesenburg and Guinasso, 1979; Weiss and Price, 1980) after correcting for lab/water temperatures, air pressure, water/headspace volumes and gas concentrations in ambient air.

2.3. Statistical analysis

In total, 22 chemical, optical and GHG variables in the ditch water were evaluated, and those were further grouped into four main variable categories: 1) acidity and ions, 2) mercury, 3) absorbance, organic carbon, and dissolved greenhouse gases (GHG), and 4) nutrients (N and P species) (Table 2). To evaluate the effect of DC, the difference in concentration (or other variable value for pH and absorbance) (Δ) for each DC-REF variable pair was calculated. This was done by first Log-transforming data to achieve normal distribution, and then subtracting the variable concentration in REF from the DC value for all variables and sampling occasions. Mean Δ values for each variable were divided by the mean REF values to calculate the difference in percentage. The Δ values were then tested whether they significantly differed from zero using a generalized linear mixed model (GLMM) approach with site pair number and sampling occasion as random factors. The effect of DC was both tested on the entire data set (n=25) and separately on the forested (n=13) and clear-cut (n=12) sites. Any correlations between how variables differed between DC and REF sites were tested using a multiple Spearman's rank correlation test on the Δ values. Only significant correlations with any of the variables influenced by DC were considered and only if the statistical power was $|\rho| > 0.5$. Significant levels for all statistical evaluations were set to $p < 0.05$. Statistical analyses were carried out using JMP Pro 16 software.

3. Results

3.1. Acidity and Ions

For all pairs, mean ΔpH exhibited a statistically significant ($p = 0.04$) value greater than zero (Fig. 2A), with the mean pH in DC sites being 0.6 pH units higher than in REF sites (Table 2). Mean ΔpH was also positive when considering forested and clear-cut catchments separately, but these were not significantly different from zero. Among the ions, mean ΔSO_4 (Fig. 2B) was significantly different from zero for all pairs ($p = 0.004$), with 101.5 μeq L^{-1} (or 96 %) higher mean SO_4 concentration in the DC sites compared to the REF sites. When the dataset was analysed separately, mean ΔSO_4 were significantly different from zero in the clear-cut sites (153.2 μeq L^{-1} or 285 % higher in DC than REF, $p = 0.02$), but not in the forested sites (Fig. 2B). For the base cations, mean ΔCa was significantly different from zero when evaluating the entire data set (208.9 μeq L^{-1} or 35 % higher in DC than REF, $p = 0.009$) and the forested catchments separately (122 μeq L^{-1} or 28 % higher in DC than REF, $p = 0.02$) (Fig. 2C). Mean ΔK was significantly different from zero for forested sites only (5.2 μeq L^{-1} or 50 % higher in DC than REF, $p = 0.05$, Fig. 2D). The mean Δ-value was not significantly different from zero for electrical conductivity (EC) or any of the ions magnesium (Mg), sodium (Na), chloride (Cl) and fluoride (F).

3.2. Mercury

Mean ΔTHg were not significantly different from zero, neither for the full dataset, nor when considering forested and clear-cut sites separately (Table 2). In forested sites, mean ΔMeHg was significantly different from zero, with 0.12 ng L^{-1} (or 17 %) lower MeHg mean concentrations in the DC sites compared to REF sites ($p = 0.02$) (Fig. 2E). In contrast, mean ΔMeHg were not significantly difference from zero for the entire data set, or the clear-cut sites only (Table 2).

3.3. Absorbance, carbon and GHGs

There were no significant differences between DC and REF sites in concentration or character of the organic carbon i.e., TOC, Abs_{420}, Abs_{420}/TOC, both when evaluating the entire data set, and when evaluating the forested and clear-cut sites separately (Table 2). For the GHGs, mean ΔCO_2 were significantly different from zero for all pairs, with 1.5 mg L^{-1} (or 40 %) lower mean CO_2 concentrations in DC compared to REF sites ($p = 0.005$) as well as in forested sites only, with 2.3 mg C L^{-1} (or 59 %) lower mean CO_2 concentrations in DC compared to REF ($p = 0.001$) (Fig. 2F). No significant difference in CO_2 was observed in clear-cut sites. Mean ΔCH_4 were not significantly different from zero, either for the entire dataset, or in forested and clear-cut sites separately (Table 2). Finally, mean ΔN_2O was significantly different from zero only when analyzing the entire data set, with 1.5 μg N L^{-1} (or 190 %) higher concentrations in DC than REF ($p = 0.01$) (Fig. 2G).

3.4. Nutrients

There were no significant differences between DC and REF sites for any of the nutrient variables, TN, TP, NH_4, NO_3, PO_4 independent on if the entire data set was evaluated or if the evaluation was made on the forested and clear-cut sites separately (Table 2).

3.5. Correlations among DC-REF differences

Any correlations between concentration differences (Δ) of the DC-REF pairs were tested for all variables of the entire data set (Figure S1, Table S3). ΔpH was negatively correlated with ΔTOC (Fig. 3A), ΔAbs_{420} and ΔTP ($\rho = -0.53$, -0.54 and -0.51, respectively, $p < 0.0001$). Additionally, several of the ions were strongly positively correlated, for example, ΔSO_4 was correlated with the Δ-values of both the base cations Mg and Ca ($\rho = 0.69$ and 0.50, respectively, $p < 0.0001$), and ΔMg was correlated to ΔCa ($\rho = 0.70$, $p < 0.0001$) (Table S3). Furthermore, ΔCO_2 and ΔCH_4 exhibited a strong positive correlation ($\rho = 0.68$, $p = 0.001$, Fig. 3B, Table S3), as did ΔN_2O with ΔNO_3 ($\rho = 0.77$, $p < 0.0001$, Fig. 3C, Table S3). The significant correlations identified in the separate datasets for forested and clear-cut sites are listed in Table S4 and S5, respectively.

4. Discussion

The results from this study suggest that DC has a significant impact on water chemistry and dissolved GHGs in draining ditch networks. We found that seven out of the 22 analyzed water chemistry and GHG variables were significantly different between DC and REF sites (Table 2, Fig. 2). The elevated pH (i.e., less acidic water) following DC found in the current study (on average 0.6 units higher in DC compared to REF sites) is in line with findings from other studies in boreal and hemiboreal areas (typical increases of 0.5–1.0 pH-units) (Joensuu et al., 2002; Hansen et al., 2013; Manninen, 1998). The higher pH in DC sites may be attributed to deeper GWTs with groundwater flow paths that, to a higher degree, traverse mineral soil layers that are richer in weathering

products (i.e., base cations) than more organic-rich soils (Ledesma et al., 2016; Ukonmaanaho et al., 2014). This is further supported by the higher Ca concentrations observed in the DC compared to in the REF sites. pH may also increase following DC as organic carbon concentrations (as TOC or DOC) typically decrease, either as a result of deeper groundwater flow paths through less organic-rich soil layers (Joensuu et al., 2002; Nieminen et al., 2010; 2018), and/or due to the removal of excessive organic material (plants and sediments) from the ditches (Hansen et al., 2013). The reduced hydrological connectivity to organic sources should decrease the concentration of organic acids in the ditch waters. The organic acidity should be related to the TOC concentration; however, our observations revealed no significant difference in concentration or characteristics (Abs_{420}, Abs_{420}/TOC) of TOC between DC and REF sites (Table 2). Although there was no significant difference in the TOC concentrations between DC and REF, the ΔTOC of the paired DC-REF sites was negatively correlated with ΔpH (Fig. 3A), indicating that changes in TOC following DC were still related to the observed pH increase.

The substantially higher SO_4 concentrations in DC sites compared to REF sites of the current study may be primarily attributed to the oxidation of reduced S forms due to lower GWTs following DC, which create more aerobic soil conditions. In addition, SO_4 concentrations typically increase with soil depth in boreal forest soils (Ledesma et al., 2016; Ukonmaanaho et al., 2014), while deeper groundwater flow paths after DC may also contribute to the elevated SO_4 levels. The observed higher SO_4 concentrations following DC is in line with findings from other DC studies (Joensuu et al., 2002, Hansen et al., 2013), whereas the magnitude of increase was particularly high in the current study (mean Δ: 101.5 μeq L^{-1}) if compared for example to the results of Hansen et al. (2013) (mean Δ: 5.8 μeq L^{-1}, with mean post-DC concentrations of 21.8 μeq L^{-1}, as opposed to 211.3μeq L^{-1} in this study). The eastern coastal areas of northern Sweden are known to have local pockets of acid sulfur soils (Nyman et al., 2023). A drop in GWT following DC may oxidize these soils and could be a reason for the enhanced SO_4 export (Fanning et al., 2017; Karimian et al., 2018). However, according to available map information, only two of the DC catchments in the current study had

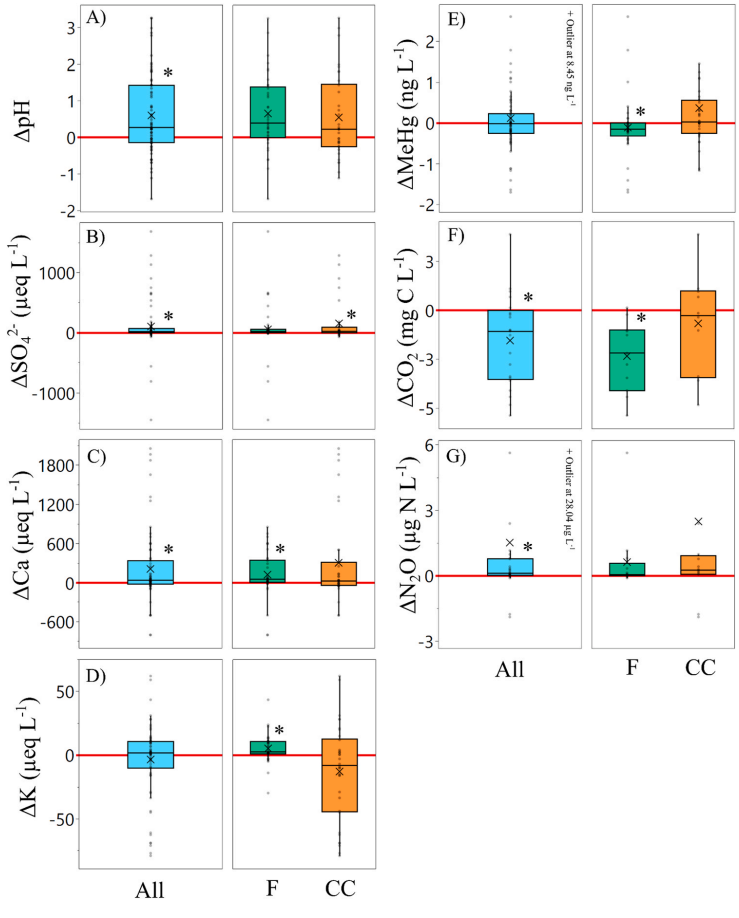

Fig. 2. Boxplots illustrating the distribution in paired Δ values (DC - REF) for each of the variables that were significantly different from zero according to the GLMM. Data is presented for the entire data set (All) and by the forested (F) and clear-cut (CC) sites separately. A value higher than zero means higher concentration in DC sites, or vice versa. Variables and treatments that were significantly different from zero according to the GLMM (see colored cells in Table 2) are marked by *. Mean Δ values are shown with a ×.

identified areas of acid sulfate soils, and the areal coverage was less than 2 %. Hence, we believe that acid sulfur soils were not the main reason to the observed patterns in SO_4 when comparing DC and REF catchments.

Elevated Hg in runoff have been pointed out as a potential risk when cleaning ditches (Wesström et al., 2017). However, in this study no significant difference in THg concentrations was found between DC and REF sites, whereas MeHg concentrations were lower in the forested DC sites compared to the forested REF sites. High Hg concentrations are most often found in the top soils (Bishop et al., 2020), and deeper flow paths may thereby mobilize less Hg following DC. Furthermore, DC-induced lowering of the GWT would make previously water logged riparian areas less favorable for MeHg formation leading to reduced export to adjacent ditch networks. It have also been found that cleaned ditches contain less Hg methylating microorganisms in their sediments when organic matter and vegetation is removed (Bitenieks et al., 2022), which may result in lowered *in-situ* MeHg formation in the ditch network. In contrast to the forested sites, no difference in MeHg between DC and REF sites were observed for the clear-cut sites. This difference may be explained by the reduced transpiration after tree removal in the clear-cut areas which partly counteracted the DC-induced groundwater lowering and in turn the MeHg formation.

Boreal forest ditches are known hotspots in the landscape for GHG emissions to the atmosphere (Wallin et al., 2018; Audet et al., 2020), but there is currently very limited information on how DC might affect these emissions. Here we found that dissolved CO_2 concentrations in the ditch water were significantly lower in DC sites compared to REF sites (Table 2), which suggest lower atmospheric emissions. CO_2 in boreal ditch waters is mainly sustained by input from catchment soils (Campeau et al., 2019), and where the soil CO_2 concentrations typically increase with soil depth (Winterdahl et al., 2016). Hence, deeper groundwater flow paths following DC may not be the cause for the lower CO_2 observed in the ditch waters. However, CO_2 is part of the dissolved inorganic carbon (DIC) pool, and the internal speciation is tightly connected to pH (Stumm and Morgan, 1996). Thus, the pronounced effect of pH following DC (mean 0.6 units higher) shifted the DIC speciation which could explain the lower ditch CO_2 observed. Another potential reason for the lower CO_2 following DC is the removal of ditch vegetation and excess of organic material leaving less substrate available for decomposition and *in-situ* CO_2 production. However, Peacock et al. (2021) and Tong et al. (2022b) found that whether a ditch is vegetated or not had no influence on atmospheric CO_2 fluxes from ditch networks, indicating that the removal of vegetation by DC is not the main cause for the lower CO_2 concentrations observed here. We further found only lower CO_2 concentrations following DC in the forested sites. The non-difference in CO_2 among the clear-cut DC and REF sites could be explained by increased mineralization of organic matter (both in soil and stream water) following clear-cut harvest (Nieminen, 2004; Schelker et al., 2016), which may have counteracted the decrease in CO_2 concentrations that was observed in the forested sites. This is further supported by findings showing higher ditch water CO_2 concentration in clear-cut compared to forested sites (Zannella et al., 2023).

We found that ditch waters CH_4 concentrations did not differ significantly between DC and REF sites. This finding aligns with the results from Tong et al. (2022b), that found no differences in CH_4 emissions between DC and non-DC ditches. On the other hand, the results of our study were in contrast with those from Rissanen et al. (2023), which found increased CH_4 emissions from moss-free ditches compared to moss-covered ditches. Collectively, this underscores the difficulty in generalizing the effects of ditch cleaning on CH_4 concentrations and dynamics. These effects are highly dependent on multiple factors, such as hydrological conditions and the prior extent and quality of plant and moss coverage. Still, ΔCH_4 were correlated with ΔCO_2 (Fig. 3B) which suggests that ditch CO_2 and CH_4 are originating from same source areas and/or sustained by similar metabolic processes (Campeau et al., 2018). It further suggests that lowered GWTs following DC influence the formation and/or mobilization of CH_4 and CO_2 in a similar way.

In contrast to CO_2, dissolved N_2O concentrations in ditch waters were significantly higher in the DC compared to the REF for all sites. DC operations, especially if conducted in connection with clear-cut harvest (Åström et al., 2004), are commonly recognized to cause relatively high nutrient losses via ditch network runoff (Nieminen et al., 2018), either in particulate (Manninen, 1998) or in inorganic dissolved forms (ammonia (NH_4) or nitrate (NO_3)) (Joensuu et al., 2002; Hansen et al., 2013). Together with higher oxygen availability in the riparian soils due to lower GWTs and enhanced aeration in the ditch network following DC, this collectively favor denitrification and production of N_2O. Although no DC effect on NO_3 was observed, the higher ΔN_2O in sites with greater ΔNO_3 suggests a potential indirect impact of N availability on N_2O production (Fig. 3C).

DC may cause rapid changes on water quality in direct connection to when the actual DC operations were conducted. Studies have found elevated spikes of particles, organic carbon, THg and MeHg in ditch water during the initial DC phase, which have then stabilized at lower levels within a few days or weeks following DC (Hansen et al., 2013). It was not possible to detect these immediate effects as this study design, with sampling conducted one to four years post DC did not aim to catch the initial impact of DC on water chemistry and dissolved GHGs, but rather focused on assessing more long-term effects. Nevertheless, an increased export of particulate matter (e.g., SS, particulate N and P) is considered as one of the most detrimental effects of DC on recipient surface waters, especially in areas with a relatively thin peat layer and where the cleaning has reached the mineral soil. In such areas, Nieminen et al. (2018) recommended avoiding disturbance of the mineral soil when conducting DC operations to reduce the elevated export of particulates that may harm downstream aquatic ecosystems. In the current study, suspended solids were not measured, but the total concentrations of C, N and P (which includes particulate fractions) were not significantly different between DC and REF sites indicating a low overall mobilization of particles one to four years after the DC operation. Still, the increased pH and base cation concentrations as well as decreased levels of MeHg or CO_2 collectively suggests deeper flow paths through more mineral-rich soils post DC. Recognizing these implications as potentially beneficial or at least not negative from an ecosystem service perspective, the big challenge lies in effectively balancing any positive and negative effects on forest production, ditch water quality and ecological functions when translating field-based research findings into actual implementation of DC practices. This is especially true since DC not only affects the chemistry and GHGs in ditch water, but may also influence, in addition to other aspects, terrestrial GHG emissions and biodiversity.

It is important to highlight that the results of the current study represent an exemplary regional snapshot of the baseflow effects of DC one to four years after the operations. The repeated synoptic data collection was not intended to allow analyses of seasonal variation or effects of hydrological events, but rather to retrieve site-comparable and non-biased data. For example, to what extent DC cause enhanced particle mobilization at higher flows is unclear, but beyond the scope of the current study. In this context, the results of the current study provide a solid basis that can be used for future research efforts towards more sustainable DC practices.

In summary, this study contributes to the evidence base of DC-related changes in water chemistry and dissolved GHGs. While some of the results are in line with findings from previous literature (increased pH and Ca concentrations), other findings show divergent patterns compared with earlier efforts e.g. unaffected TOC and nutrient concentrations. Furthermore, we provide novel results on how DC affect MeHg (decreasing in forested sites) and dissolved GHGs (decreasing CO_2 and increasing N_2O) in runoff. Lower MeHg in forested sites may lower the bioavailability of Hg in downstream aquatic ecosystems. While lower CO_2 after DC could be considered positive, higher N_2O concentration following DC is an unwanted effect that requires careful consideration when conducting DC. The changes in water chemistry and GHGs

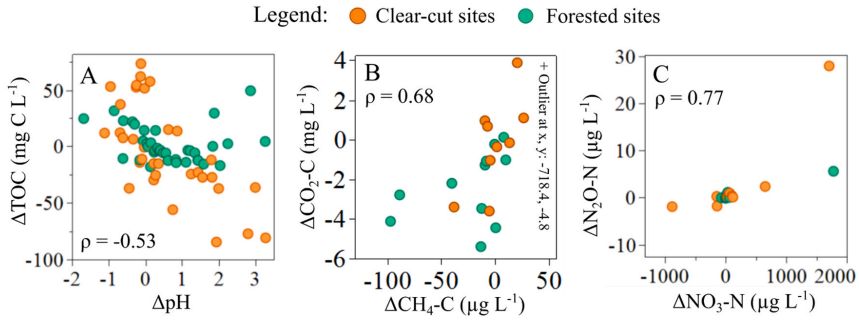

Fig. 3. Selected scatterplots of A) ΔpH vs ΔTOC, B) ΔCH_4-C vs ΔCO_2-C and C) ΔN_2O-N vs ΔNO_3-N, based on the correlation statistics of the variables that were significantly affected by DC listed in Table S3.

following DC is suggested to be governed by a combination of i) deeper groundwater flow paths, ii) changed redox conditions in the soil as GWTs are lowered, and iii) removal of organic rich sediments and vegetation from the ditches themselves. These findings emphasize the necessity for continued monitoring to detect the impacts of DC on the biogeochemistry of forest ditches at both temporal and spatial scales. Such studies are essential for conducting DC operations that effectively mitigate any negative impacts on headwater- and associated ecosystems.

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CRediT authorship contribution statement

Marcus B. Wallin: Writing – review & editing, Supervision, Methodology, Formal analysis, Conceptualization. **Ulf Sikström:** Writing – review & editing, Supervision, Conceptualization. **Emeli Arvidsson:** Writing – original draft, Methodology. **Karin Eklöf:** Writing – review & editing, Supervision, Project administration, Methodology, Funding acquisition, Formal analysis, Conceptualization. **Alberto Zannella:** Writing – review & editing, Writing – original draft, Visualization, Methodology, Formal analysis.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data Availability

Data will be made available on request.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2024.122146.

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SUPPLEMENTARY INFROMATION

Ditch cleaning in boreal catchments: impacts on water chemistry and dissolved greenhouse gases in runoff

Table S1. Detailed overview of catchment characteristics reported by pair (DC and REF). F or CC: Forested or Clear-Cut catchments; MASL: Meters Above Sea Level; MAP: Mean Annual Precipitation (30-years mean, 1991-2020); MAT: Mean Annual Temperature (30-years mean, 1991-2020); CON: Coniferous-dominated forest; CC-CON: Clear-Cut of Coniferous-dominated forest (i.e., percentage of harvested forest out of the Coniferous-dominated share of the catchment. Note: In Sweden, forest harvesting primarily involves coniferous-dominated forests. Deciduous trees are typically present only as part of a mixed forest with conifers); DEC: Deciduous-dominated forest; OW: Open Wetland; OTH: Other; TIL: Till soils; PET: Peat soils; SAN: Sandy soils; CLA: Clay soils; SIL: Silty soils.

									Land-cover (%)						Soil types (%)						Peat depth (cm)	
Pair no.		F or CC	Year CC	Year DC	Area (ha)	MASL (m)	MAP (mm)	MAT (°C)	CON	CC-CON	DEC	OW	OTH	EXP	TIL	PET	SAN	CLA	SIL	OTH	Catchment mean	Sampling point
1	DC	F	--	2018	114	44	670	6.3	92	29	6	0	0	1	88	4	0	2	0	6	22	53
	REF	F	--	--	35	40	670	6.2	97	1	0	0	0	3	71	1	0	12	0	16	14	27
2	DC	F	--	2019	25	37	691	6.2	96	9	0	0	0	4	55	0	42	0	0	3	14	23
	REF	F	--	--	40	46	691	6.2	87		10	2	0	1	82	18	0	0	0	0	25	35
3	DC	F	--	2020	7	90	631	5	85		15	0	0	0	92	0	0	0	0	8	12	11
	REF	F	--	--	57	92	683	4.9	95		2	1	1	2	38	5	0	0	12	44	13	22
4	DC	F	--	2017	7.9	247	828	2.6	100		0	0	0	0	100	0	0	0	0	0	20	31
	REF	F	--	--	20	251	828	2.6	95		5	0	0	0	23	11	0	0	13	54	20	23
5	DC	F	--	2018	3	265	844	3.3	96		0	4	0	0	9	1	19	0	42	29	34	57
	REF	F	--	--	13	245	844	3.3	90		0	6	0	4	66	20	0	0	0	14	24	57
6	DC	F	--	2020	136	52	807	3.7	91		4	0	3	1	6	9	26	0	18	40	25	62
	REF	F	--	--	96	48	777	3.7	89	2	1	7	2	1	0	0	11	0	89	0	34	55
7	DC	F	--	2020	21	55	807	3.7	98		0	1	1	0	11	2	24	0	28	35	21	51
	REF	F	--	--	35	45	807	3.7	75		15	0	6	3	20	6	10	0	2	61	37	47
8	DC	F	--	2020	108	53	807	3.7	95	16	0	0	3	1	51	41	0	0	0	8	25	40
	REF	F	--	--	76	82	819	3.4	88		1	4	4	3	53	45	0	0	0	2	17	47
9	DC	F	--	2018	34	241	728	2.7	71	1	8	19	1	0	77	0	0	0	0	23	44	29
	REF	F	--	--	13	203	718	3	76		0	21	3	0	96	3	0	0	0	2	51	47
10	DC	CC	2019	2019	4	279	735	2.4	91	0.5	3	0	0	6	29	11	0	0	0	60	12	13
	REF	CC	2019	--	9	287	735	2.4	90	2	5	0	4	2	79	11	0	0	0	10	12	13
11	DC	CC	2017	2020	6	117	763	4	99	0	0	0	0	1	72	9	0	0	0	19	13	76
	REF	CC	2020	--	51	77	822	3.7	93	8	3	3	0	1	38	8	0	0	0	55	31	81
12	DC	CC	2019	2020	81	34	820	3.4	95	21	0	3	1	1	54	0	0	0	0	46	30	13
	REF	CC	2018	--	26	121	820	3.4	96	3	0	3	1	0	77	23	0	0	0	0	25	76
13	DC	F	--	2018	20	247	693	2.5	93		4	0	0	2	49	0	0	0	44	7	12	23
	REF	F	--	--	38	248	695	1.9	72		7	14	3	4	33	0	43	0	0	25	37	47
14	DC	F	--	2018	16	101	822	3.5	93		4	0	2	1	100	0	0	0	0	0	25	37
	REF	F	--	--	15	110	822	3.5	94		1	0	5	0	83	6	0	0	0	11	22	15
15	DC	F	--	2017	20	246	755	2.4	97		3	0	0	0	37	35	28	0	0	0	12	21
	REF	F	--	--	23	229	755	2.4	97		1	0	0	1	33	13	43	0	0	11	18	49
16	DC	CC	2019	2019	17	30	717	4.4	90	3	0	9	0	1	100	0	0	0	0	0	34	78
	REF	CC	2017	--	17	34	774	4.2	93	6	0	5	0	2	80	20	0	0	0	0	24	13

Table S1. Continued.

17	DC	F	--	2019	25	260	665	2.1	88		6	1	0	5	84	14	0	0	0	2	12	70
	REF	F	--	--	32	281	665	2.1	95		1	4	0	0	96	0	0	0	0	4	28	38
18	DC	CC	2017	2019	67	276	665	2.1	92	3	1	5	2	0	99	0	0	0	0	1	26	57
	REF	CC	2020	--	56	277	665	2.1	98	24	0	0	1	1	0	31	62	0	0	7	14	35
19	DC	CC	2018	2019	11	46	757	4	97	1	0	0	0	3	62	19	18	0	0	0	40	60
	REF	CC	2018	--	20	37	757	4	92	4	3	3	0	1	33	25	0	0	0	42	48	57
20	DC	CC	2017	2018	31	119	759	3.3	91	4	1	8	0	0	96	0	0	0	0	4	34	70
	REF	CC	2018	--	72	130	767	3.3	84	9	2	8	4	2	62	12	0	0	0	27	33	65
21	DC	CC	2017	2019	46	276	692	2.5	98	4	2	0	0	1	70	25	0	5	0	0	16	51
	REF	CC	2019	--	33	268	692	2.5	97	4	1	0	0	2	58	35	0	1	0	6	20	81
22	DC	CC	2019	2020	31	59	662	6.1	87	2	2	0	7	3	61	15	0	24	0	0	20	76
	REF	CC	2017-2020*	--	176	54	662	6.1	81	18	4	1	8	5	54	31	0	14	0	1	25	13
23	DC	CC	2017	2017	35	33	667	6.3	82	4	10	0	6	1	74	18	0	7	0	1	19	40
	REF	CC	2018	--	99	32	667	6.3	86	23	7	1	0	5	74	10	0	13	0	3	22	35
24	DC	CC	2018-2020*	2020	67	51	657	6.2	94	10	5	0	0	1	75	7	0	5	0	13	21	70
	REF	CC	2020	--	34	55	663	6.1	81	1	11	0	4	5	90	6	0	2	0	2	14	51
25	DC	CC	2019	2019	2	51	663	6.1	100	1	0	0	0	0	30	0	0	70	0	0	27	28
	REF	CC	2019	--	5	51	663	6.2	77	3	14	7	0	2	99	0	0	1	0	0	36	60

*In connection to sampling point there were areas clear-cut in different times

Table S2. *p*-values resulting from the generalized linear mixed model (GLMM) analyzing the DC - REF difference across all variables for all the paired sites of the entire data set as well as separately on the forested sites and sites which included clear-cuts where DC was conducted. Significant differences ($p > 0.05$) are highlighted in bold style and indicated by arrows, denoting whether values were higher or lower in DC compared to REF.

	Variable	DC vs REF (All)	DC vs REF (Forested)	DC vs REF (Clear-cut)
Acidity and Ions	**pH**	**0.04** ↑	0.17	0.14
	EC	0.27	0.71	0.29
	Alkalinity	0.17	0.3	0.41
	SO_4	**0.004** ↑	0.12	**0.02** ↑
	Cl	0.21	0.09	0.64
	F	0.63	0.67	0.34
	Ca	**0.009** ↑	**0.02** ↑	0.19
	Mg	0.06	0.24	0.1
	Na	0.92	0.62	0.64
	K	0.49	**0.05** ↑	0.36
Mercury	**Hg-tot**	0.59	0.91	0.64
	MeHg	0.36	**0.02** ↓	0.57
Absorbance, carbon and GHGs	**Abs_{420}**	0.64	0.95	0.53
	TOC	0.82	0.98	0.75
	Abs_{420}/TOC	0.24	0.62	0.23
	CO_2-C	**0.005** ↓	**0.001** ↓	0.36
	CH_4-C	0.26	0.3	0.27
	N_2O-N	**0.01** ↑	0.05	0.1
Nutrients	**Tot-N**	0.69	0.71	0.8
	NH_4-N	0.28	0.43	0.31
	NO_3-N	0.8	0.95	0.77
	Tot-P	0.74	0.28	0.71
	PO_4-P	0.77	0.57	0.45

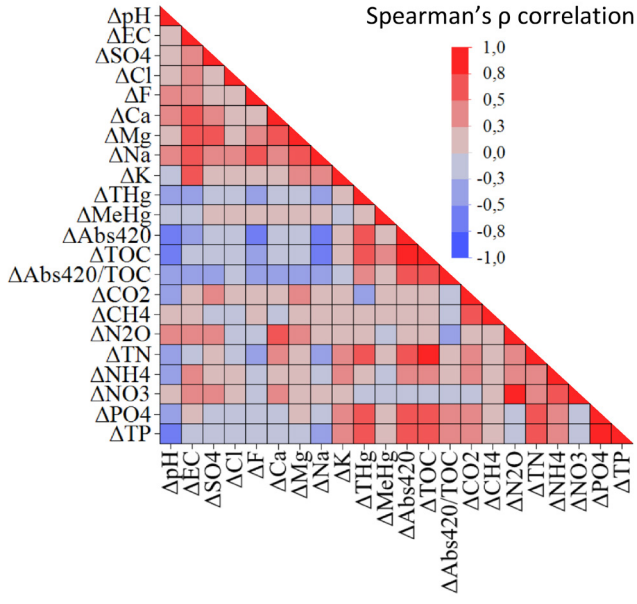

Figure S1. Spearman rank correlation between the 22 variables analyzed. Red colors indicate positive correlation while blue colors indicate negative correlation. The stronger the intensity of the color the stronger the relationship considered.

Table S3. Significant correlations between the difference (Δ) between DC and REF sites of the variables measured in ALL sites. The table shows only significant relationships containing at least one of the variables where Δ was significantly different from zero in the GLMM analysis and with a |ρ| value > 0.5.

	ALL		
Variable	**by Variable**	**Spearman ρ**	**Prob>|ρ|**
ΔN_2O	ΔNO_3	0.77	>0.0001
ΔMg	**ΔCa**	0.7	>0.0001
ΔMg	**ΔSO_4**	0.69	>0.0001
ΔCH_4	**ΔCO_2**	0.68	0.001
ΔN_2O	**ΔCa**	0.6	0.001
ΔK	ΔEC	0.6	>0.0001
ΔCa	ΔEC	0.59	>0.0001
ΔAbs_{420}	**ΔpH**	-0.54	>0.0001
ΔTOC	**ΔpH**	-0.53	>0.0001
ΔTP	**ΔpH**	-0.51	>0.0001
ΔCa	**ΔSO_4**	0.5	>0.0001

Table S4. Significant correlations between the difference (Δ) between DC and REF sites of the variables measured in FORESTED sites. The table shows only significant relationships containing at least one of the variables where Δ was significantly different from zero in the GLMM analysis and with a |ρ| value > 0.5.

FORESTED			
Variable	**by Variable**	**Spearman ρ**	**Prob>|ρ|**
ΔN_2O	ΔNO_3	0.83	0.001
ΔK	ΔNa	0.76	>0.0001
ΔK	ΔEC	0.72	>0.0001
ΔK	ΔMg	0.72	>0.0001
ΔCO_2	ΔNH_4	0.68	0.02
ΔMg	**ΔSO_4**	0.65	>0.0001
ΔCO_2	**ΔK**	0.63	0.04
ΔK	**ΔSO_4**	0.6	>0.0001
ΔSO_4	ΔEC	0.6	>0.0001
ΔTP	**ΔpH**	-0.58	>0.0001
ΔN_2O	**ΔCa**	0.58	0.04
ΔN_2O	ΔTN	0.55	0.05
ΔPO_4	**ΔpH**	-0.55	0
ΔCa	ΔEC	0.53	0.001
ΔK	ΔF	0.51	0.001

Table S5. Significant correlations between the difference (Δ) between DC and REF sites of the variables measured in CLEAR-CUT sites. The table shows only significant relationships containing at least one of the variables where Δ was significantly different from zero in the GLMM analysis and with a |ρ| value > 0.5.

CLEAR-CUTS			
Variable	**by Variable**	**Spearman ρ**	**Prob>|ρ|**
ΔMg	**ΔCa**	0.89	>0.0001
ΔCa	**ΔSO_4**	0.83	>0.0001
ΔNa	**ΔpH**	0.76	>0.0001
ΔN_2O	ΔNO_3	0.75	0.005
ΔMg	**ΔSO_4**	0.74	>0.0001
ΔAbs_{420}	**ΔpH**	-0.73	>0.0001
ΔTOC	**ΔpH**	-0.68	>0.0001
ΔNa	**ΔCa**	0.66	>0.0001
ΔN_2O	ΔEC	0.65	0.02
ΔCH_4	**ΔCO_2**	0.65	0.04
ΔN_2O	**ΔCa**	0.62	0.03
ΔMg	**ΔpH**	0.61	>0.0001
ΔCa	ΔNO_3	0.61	>0.0001
ΔCa	**ΔpH**	0.61	>0.0001
ΔCa	ΔEC	0.6	>0.0001
ΔK	ΔNH_4	0.6	>0.0001
ΔAbs_{420}/TOC	**ΔpH**	-0.59	>0.0001
ΔTP	**ΔpH**	-0.59	>0.0001
ΔF	**ΔpH**	0.55	0.001
ΔK	ΔPO_4	0.55	0.001
ΔSO_4	ΔNO_3	0.54	0.001
ΔPO_4	**ΔpH**	-0.52	0.002

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How do different land-use management practices affect the export of terrestrial carbon via the aquatic pathway? This thesis presents field-based observations on the effects of clear-cutting, ditch cleaning, and rewetting of drained wetlands on various forms of aquatic carbon. The results provide a valuable knowledge base to support informed decision-making for future implementation of these practices, helping to ensure the intended outcomes with minimal negative consequences on runoff carbon.

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