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Significant Emissions From Forest Drainage Ditches—An Unaccounted Term in Anthropogenic Greenhouse Gas Inventories?

Mike Peacock[1], Gustaf Granath[2], Marcus B. Wallin[1], Lars Högbom[3,4], and Martyn N. Futter[1]

[1]Department of Aquatic Sciences and Assessment, Swedish University of Agricultural Sciences, Uppsala, Sweden, [2]Department of Ecology and Genetics, Uppsala University, Uppsala, Sweden, [3]Skogforsk, Uppsala, Sweden, [4]Department of Forest Ecology and Management, Swedish University of Agricultural Sciences, Umeå, Sweden

Key Points:

- Emissions of methane and carbon dioxide can be large from forest ditches on both mineral and peat soils
- Ditches are constructed and therefore these emissions are anthropogenic in origin and should be accounted for in inventories
- Ditch methane emissions are particularly important, as they can influence landscape-scale methane budgets

Supporting Information:

Supporting Information may be found in the online version of this article.

Correspondence to:

M. Peacock,
michael.peacock@slu.se

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Author Contributions:

Conceptualization: Mike Peacock, Martyn N. Futter
Formal analysis: Mike Peacock
Funding acquisition: Mike Peacock, Martyn N. Futter
Investigation: Mike Peacock, Gustaf Granath, Marcus B. Wallin, Lars Högbom
Writing – original draft: Mike Peacock

Abstract Forestry is a natural climate solution for removing atmospheric carbon dioxide (CO_2) and reaching net zero emissions. Managed boreal forests typically have extensive drainage ditch networks, and these can be hotspots of anthropogenic greenhouse gas (GHG) emissions, potentially offsetting the terrestrial carbon gain. However, there is a lack of data detailing GHG emissions from ditches on mineral soils, where most boreal forestry occurs. Here, we address this knowledge gap using two approaches. First, we conducted a synoptic campaign to measure summer GHG fluxes from 109 boreal forest ditches draining mineral soils within one local region. We found a clear control of ditch water level on methane (CH_4), with zero emissions from dry ditches and variable, but often high, emissions from water-filled ditches. Almost all ditches acted as sources of CO_2, regardless of water status. Second, we reanalyzed a published data set of boreal forest ditches and streams across three regions where GHG concentrations had been repeatedly measured and detailed catchment information was available. Within this data set we categorized 76 ditches into mineral and peatland catchments and detected no difference in mean CH_4 and CO_2 concentrations between the two soil types. GHG emissions from ditches draining mineral forest soils can be as large as those from peatland forest ditches. Using literature values for forest GHG uptake we demonstrate that ditch CH_4 emissions are particularly important and can offset the terrestrial CH_4 uptake. Ignoring ditch emissions, which are anthropogenic in origin, will lead to incorrect estimates of the landscape-scale forest GHG budget.

Plain Language Summary Managed boreal forests can remove large amounts of carbon dioxide from the atmosphere. It is hoped that this removal will go some way to counteracting anthropogenic greenhouse gas (GHG) emissions. However, forests often contain networks of human-constructed drainage ditches which can be hotspots for emissions of methane, a powerful GHG. Most boreal forests are found on mineral soils but there is almost no information concerning the size of forest ditch GHG emissions from these soils. We measured GHGs in a large number of boreal forest ditches and found that emissions were of a similar average size from ditches draining mineral and peat soils. Dry ditches emitted no methane, but water-filled ditches emitted high amounts. All ditches, dry and wet, emitted carbon dioxide. We show that even though ditches occupy a small proportion of the forest area, their emissions can influence landscape-scale GHG budgets.

1. Introduction

"In the beginning there was a mire, a ditching mattock and Jussi"

The opening lines of the Finnish novel Under the North Star (published 1959) by Väinö Linna, in which the farm hand Jussi is draining a mire.

Drainage ditches are common features of many production landscapes. They are constructed for various reasons but are generally used to lower the terrestrial water table in order to improve agricultural and forest productivity, by increasing both vegetation rooting depth and nutrient availability via the decomposition of organic matter. There is a wide variety of ditch morphologies and characteristics; it is not unusual for

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ditches to sometimes dry up, and these systems therefore occupy a changing interface between the aquatic and terrestrial realms. Perhaps this has led to ditches becoming a scientific "no man's land"? Certainly, they are extremely under-researched when compared to streams; their counterparts of natural origin (Koschorreck et al., 2020). Despite this, ditches are important components in many landscapes and often perform important ecosystem functions, for example, biodiversity provision (Williams et al., 2004), nutrient and pollutant retention (Herzon & Helenius, 2008), and stormwater removal (Needelman et al., 2007).

Ditches often have specific attributes (nutrient-rich sediments, anoxic waters, etc.) that result in the production and emission of the greenhouse gases (GHGs) methane (CH_4) and carbon dioxide (CO_2). Although ditches typically only occupy small fractions of an ecosystem's area, their emissions can be sizeable, and important on landscape scales for both CH_4 (Cooper et al., 2014; Peacock et al., 2017; Roulet & Moore, 1995; Teh et al., 2011) and CO_2 (Hyvönen et al., 2013; Jauhiainen & Silvennoinen, 2012; Xiao et al., 2021). Ditches, particularly those in intensive agricultural environments, can also emit nitrous oxide (N_2O) (Peacock et al., 2017; Teh et al., 2011; Webb et al., 2021), but these emissions are often lower than those from the adjacent terrestrial ecosystem (Evans et al., 2016). Across regional and global scales ditches further exert effects; a recent synthesis calculated that ditches emit 1% of all anthropogenic CH_4 (Peacock et al., 2021) and in countries with large areas of drained land (e.g., The Netherlands) it has been estimated that ditches are responsible for 16% of national emissions (Koschorreck et al., 2020). Furthermore, because ditches are human-constructed their emissions are classed as anthropogenic and need to be included in GHG inventories (IPCC, 2014, 2019).

The widespread adoption of the Paris Agreement, which aims to limit increases in global temperatures to 1.5°C, has fuelled interest in natural climate solutions (NCS). One promising NCS is forestry, which includes reforestation, avoided forest conversion, and improved forest management (Griscom et al., 2017). The possible benefits of forestry as a NCS could be compromised by climate-driven risks (e.g., drought, fire; Anderegg et al., 2020) but forestry has a central role in plans for limiting net GHG emissions in many countries. In Sweden, the government aims to reach net zero (of GHG emissions) by 2045, and CO_2 removal by forests will play a key role (Government Offices of Sweden, 2020). It has recently been argued that GHG emissions from wetlands, lakes, and streams offset ∼50% of the Swedish land-use sink (Lindroth & Tranvik, 2021), but most of these emissions, under IPCC definitions, are natural in origin and therefore irrelevant for current IPCC accounting. However, forest ditch GHG emissions are anthropogenic and, if they substantially offset the forest carbon sink, they become highly relevant for national accounting.

Emissions of GHGs, particularly CH_4, from forest ditches are well represented in the literature (Koskinen et al., 2016; Roulet & Moore, 1995; Schiller & Hastie, 1996). In Finland, where 5.5 Mha of peatlands have been drained for forestry, total ditch CH_4 emissions have been calculated as 0.05 Tg, equivalent to 9% of the total national CH_4 emissions (Peacock et al., 2021). However, past studies have focused almost exclusively on ditches draining forests on mires/peatlands (typically defined as soils with peat depth ≥30 cm) and in our recent CH_4 synthesis (Peacock et al., 2021) of all global ditch studies we identified only one study of forest ditches draining mineral soils (organic horizon <10 cm, or <20 cm and <12% organic carbon content, IPCC, 2014) where annual emissions could be derived (Klaus et al., 2018; although in some cases this may be an issue of semantics, with ditches being called streams in the literature). Generally, fluvial GHG emissions from low-order forest streams are driven by terrestrial production, either through heterotrophic or autotrophic respiration, and with CH_4 being produced in near-stream anoxic environments (Campeau et al., 2018, 2019). Lateral inputs of soil-derived dissolved organic carbon (DOC) with subsequent in situ mineralization are also suggested to contribute to fluvial GHGs (Evans et al., 2016). However, the breakdown of DOC is known to be related to water retention time (Catalán et al., 2016), and DOC can be adsorbed by mineral soils and then mineralized (Kothawala et al., 2009), so where in the landscape this GHG production occurs is system specific and largely related to hydrological conditions. It is possible that both the relative importance of external and internal production, as well as the magnitude of ditch GHG fluxes, may differ between peat and mineral soils. Thus, there is a clear knowledge gap concerning the "true" landscape-scale (i.e., including aquatic fluxes) GHG balance of managed forestry on mineral soils.

This knowledge gap is particularly relevant for Sweden where the majority of managed forests are on mineral soils. Ditching of these soils is commonplace (Ågren & Lidberg, 2019) and is conventionally done to either lead water through the landscape or to lower the water table in wetter areas (Jakobsson, 2013). However,

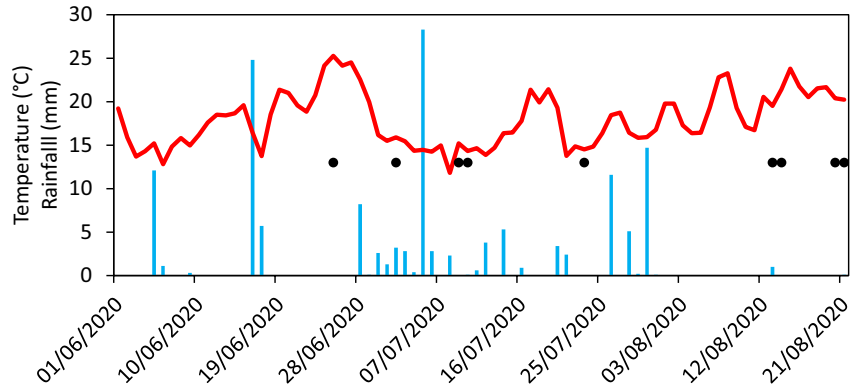

Figure 1. Mean daily temperature (red line) and rainfall (blue bars) for the sampling period, and sampling days (black circles).

ditching in Sweden has been practised for hundreds of years and its history has a convoluted path; for instance, forest drainage has at times taken place because it was thought to reduce frost, to prevent paludification, and because it was considered to be good for the environment (Jacks, 2019). The result of this is that ditch networks in the Swedish landscape are prevalent and extensive. A national analysis of Swedish streams <6 m wide (i.e., not just headwaters) found that, across all land cover types, ditches make up 63% of the network and 4% are straightened streams (Ågren & Lidberg, 2019). In forested headwater catchments the extent of stream network modification further increases; for example, in an individual forested Swedish catchment it was found that there were 149 km of ditches compared to a perennial stream length of 178 km, and that 22% of this perennial stream network had been straightened (Hasselquist et al., 2018).

We initiated a two-part study to investigate GHGs in boreal forest ditches. Firstly, to quantify the magnitude and spatial variation of GHG emissions from forest ditches on mineral soils, we surveyed 109 individual forest ditches in one locality in southern Sweden. We measured growing season fluxes of CH_4 and CO_2, ditch morphology, and water chemistry. This survey measured GHG fluxes from ditches on mineral soils, but some sites may have peat soils upstream in the catchment that release organic matter and dissolved GHGs which are subsequently transported down the drainage network and emitted from areas of mineral soils (i.e., where we measured fluxes). Therefore, for the second part of our study, we compared the influence of mineral and peat soils on forest ditch GHG emissions by analyzing a subset of a published data set of CH_4 and CO_2 concentration measurements from ~300 fluvial forested headwater sites across three large regions of Sweden (Wallin et al., 2014, 2018). In the original publications these sites were described as "streams" but a significant proportion of forested headwater watercourses in Sweden are non-natural or modified (Kuglerová et al., 2017). We identified ditches within this data set and tested whether GHGs differed between ditches on mineral and peat soils.

2. Materials and Methods

2.1. Local Survey of Ditch GHG Fluxes

We sampled 109 forest ditches (Figures S1 and S2) during the 2020 growing season. The sampling took place on 9 days between 25th June and 21st August (Figure 2), between 9:00 and 16:20. The sampled sites were all within an area centered on the city of Uppsala (59°51′29″N 17°38′41″E) with an approximate radius of 40 km. The climate is humid continental, and for 2020 mean annual temperature was 8.9°C and total precipitation was 503 mm (SMHI data from "Uppsala Aut" station—https://www.smhi.se/). During the study period the mean daily temperature was 18.1°C and 101 mm of rain fell (Figure 1). The 1991–2020 daily mean temperatures are 18.2°C and 16.9°C for July and August respectively, and respective mean monthly rainfall is 57.4 and 73.6 mm. The region is dominated by boreal forest; for Uppsala County (where 100 of the ditches were located) 66% of the land area is forest (Statistics Sweden, 2019). The principal trees species are Norway spruce (*Picea abies*), Scots pine (*Pinus sylvestris*), and birches (*Betula pendula* and *Betula pubescens*), with a shrub layer dominated by blueberry (*Vaccinium myrtillus*) and lingonberry (*Vaccinium*

vitis-idaea), and a ground layer of red-stemmed feathermoss (*Pleurozium schreberi*), glittering woodmoss (*Hylocomium splendens*), and *Polytrichum* and *Cladonia* species. Aerial photography from the period 1955–1967 showed that 33 of the 109 ditches were originally on agricultural land, which has since been converted to Norway spruce forestry.

Ditches were chosen using a combination of local knowledge and mapping. Soil types were checked using Geological Survey of Sweden maps (apps.sgu.se/kartgenerator). In four cases, soils were mapped as deep peat, but field observations showed that this was not the case (e.g., rocks were present). The most common soil types were glacial clay ($n = 37$) and sandy till ($n = 37$; see Table S1 for a full list of soil types). At each ditch we measured ditch depth and width, and sediment temperature, and noted the presence or absence of vegetation. If water was present we measured water depth and temperature, made a visual assessment of whether the water was flowing, and took a water sample for analysis. Finally, we measured GHG emissions using the static/floating chamber method.

2.2. GHG Measurements

To measure GHG fluxes we used a chamber adapted from Bastviken et al. (2015) with a circular base (diameter 31.5 cm) and a total volume of 9.56 l. The chamber was covered in silver foil to reflect sunlight and minimize internal heating. In water-filled ditches, the chamber was deployed floating and allowed to drift. In dry ditches, the chamber was gently pushed down into the sediment to create a seal without creating chamber artifacts. The chamber was deployed for 5 min at each ditch and was connected in a closed loop to a Picarro GasScouter measuring real-time concentrations of CH_4 and CO_2. Diffusive fluxes were calculated using linear regression between GHG concentration and chamber deployment time and were corrected for ambient atmospheric pressure and air temperature. We did not set an R^2 cut-off but used the approach of Peacock et al. (2017); all nonsignificant ($p > 0.05$) regressions are classified as zero fluxes, but significant fluxes with low R^2 values are accepted as real fluxes. On nine occasions our chamber deployments caught ebullition events. These were evident in the data as abrupt increases in CH_4 concentration as bubbles entered the chamber. Where these occurred, an ebullitive flux was calculated as the change in CH_4 concentration over time. The ebullition event was not included in the calculation of diffusive flux; instead, the chamber was removed from the water to air it out, then the deployment was repeated to measure the diffusive flux. Ebullitive and diffusive fluxes were summed to calculate total flux; these total fluxes are given in the text of Section 3.1, but figures include only diffusive emissions.

2.3. Water Chemistry Measurements

Water samples were stored in the dark at 4°C before analysis. Samples were analyzed for pH and electrical conductivity (EC), and full UV-visible absorbance scans were measured using a 1 cm pathlength cuvette and an Avantes AvaLight DH-S-BAL light source, on samples filtered at 0.45 μm. We used absorbance measurements at 270 and 350 nm to calculate DOC concentrations using a published model (Carter et al., 2012; Tipping et al., 2009). Furthermore, we used absorbance at 400 nm as a measure of water color and also calculated the E2:E3 ratio (250:365 nm) which is a proxy for dissolved organic matter (DOM) composition and has been related to DOM aromaticity and molecular weight (Peuravuori & Pihlaja, 1997). Total phosphorus (TP) was analyzed by the SWEDAC-accredited Geochemical Laboratory at the Swedish University of Agricultural Science (SLU, 2020).

2.4. Multi-Regional Analysis of Ditch GHG Concentrations

For the second part of our study, we analyzed a published data set of CH_4 and CO_2 concentrations from ~300 fluvial headwaters. The data were collected from watercourses in three boreal or hemi-boreal regions: south east Sweden, central Sweden, and south west Sweden (Figure S1; these regions are SES, DAL, and LAVI in Wallin et al., 2014, 2018). Land use in the sampled catchments was dominated by managed forests, with no urban areas, and agricultural land cover <5%. Each watercourse was sampled three times: in spring, summer, and autumn. Measurements of GHGs were made using a headspace method and analyzed on a gas chromatograph. All samples were above the limit of detection for CO_2 (50 ppm), but for CH_4 some samples were below the limit of detection (1 ppm) and these were treated as zeroes in the analysis (see Wallin

et al., 2014, 2018 for more information). Although the sampled waterbodies were referred to as "streams" we assumed that many of the watercourses in this data set could be ditches, or straightened streams (and we note that others in Sweden have used the term "streams" to refer to what are forest ditches [e.g., Klaus et al., 2018]). We therefore used maps and satellite images to examine all sampled watercourses. We classified watercourses as ditches if they were perfectly straight, if they made unnaturally sharp turns (e.g., 90° turns), or if they were clearly part of ditch networks (i.e., numerous parallel watercourses, geometric drainage networks). This resulted in a total of 168 ditches.

We used data from these 168 ditches (Wallin et al., 2014, 2018), which includes catchment land cover, to test whether there was a difference in GHG concentrations between watercourses draining peat or mineral soil catchments. Firstly, we tested for correlations between catchment peatland area and GHG concentrations in all 168 ditches. Secondly, we designated all catchments with zero peatland area as mineral catchments ($n = 61$), and all catchments with peatland cover ≥20% as catchments with peatland influence ($n = 15$). We calculated mean GHG concentrations for each catchment type (mineral and peat).

2.5. Statistical Analysis

We tested for differences in mean GHGs and other ditch variables (e.g., ditch depth, width) between categories (e.g., water-filled vs. dry ditches, mineral vs. peat soils, vegetated vs. unvegetated ditches) employing *t*-tests. Where our GHG variables displayed non-normal distributions and unequal variances between categories we used Welch's *t*-test (Fagerland & Sandvik, 2009; Kasuya, 2001). CH_4 data are frequently skewed, and therefore we also use median tests where appropriate. For the local GHG survey we used Spearman (rho) and Pearson (r) correlations to test for monotonic and linear relationships between GHGs and ditch characteristics/water chemistry, for water-filled ditches. For the multi-regional analysis of ditch GHG concentrations we used Spearman correlations to test for relationships between GHGs and catchment peatland area. All analyses were run in SPSS 26.

3. Results

3.1. Local Survey of Ditch GHG Fluxes

Of the 109 surveyed ditches, 51 contained water ("water-filled") and 58 were dry. Water-filled ditches were significantly deeper and wider than dry ditches, and sediments temperatures were lower (Table 1). Although the mean water depth was relatively shallow (9 cm), the range was large with water depths up to 1 m. Mean concentrations of both TP (221 μg l^{-1}) and DOC (24 mg l^{-1}) were relatively high, as was mean E2:E3 (4.8); a measure of DOM composition.

Overall mean GHG fluxes were 33.9 (range −1.3 to 1,390) mg CH_4 m^{-2} d^{-1} and 6,016 (range −720 to 32,470) mg CO_2 m^{-2} d^{-1}. For CH_4, fluxes from water-filled ditches were significantly larger than those from dry ditches (Figure 2), and 39 dry ditches acted as small sinks of CH_4 (mean uptake of 0.43 CH_4 m^{-2} d^{-1}). Ebullition events were recorded at nine ditches, with a mean flux of 1,061 (range 3–3,880) mg CH_4 m^{-2} d^{-1}, and ebullition comprised a mean of 75% (range 35%–99%) of total flux. There was no detectable difference in CO_2 fluxes between water-filled and dry ditches (Figure 3) and the majority of ditches were CO_2 sources; three ditches had zero emissions and five ditches were CO_2 sinks (mean uptake of 404 mg CO_2 m^{-2} d^{-1}). More ditches were unvegetated ($n = 63$) than vegetated ($n = 46$), and we found no effect of vegetation on fluxes of CH_4 or CO_2 (see Table S2).

For water-filled ditches, CH_4 and CO_2 fluxes were significantly related to each other (Figure 4, Spearman correlation: rho = 0.57, $p < 0.001$). When considering water chemistry and ditch characteristics, positive correlations with GHG fluxes were detected for ditch water depth, but the relationships were weak (Figure 4, respective rho values of 0.4 and 0.29 for CH_4 and CO_2). There was no significant difference in the magnitude of GHG fluxes between flowing and standing ditches. However, there was a significant positive correlation between GHGs and DOC concentration for ditches with standing water (Figure 5, respective r values of 0.53 and 0.54 for CH_4 and CO_2). These correlations did not exist for ditches with flowing water, or for the combined data set of flowing and standing water.

Table 1
Means (±Standard Error of the Mean, SEM) and Ranges of Ditch Characteristics and Water Chemistry for 109 Sampled Forest Ditches, Grouped According to Whether Ditches Contained Water ("Water-Filled") or Were Dry

	Water-filled (n = 51)		Dry (n = 58)	
	Mean ± SEM	Range	Mean ± SEM	Range
Ditch depth (cm)	118 ± 8	30–250	79 ± 5	20–220
Ditch width (cm)	165 ± 11	35–400	100 ± 6	50–290
Sediment temp (°C)	12.8 ± 1.0	10.4–15.7	14.0 ± 0.2	10.5–18.0
Water temp (°C)	13.7 ± 0.4	10.5–26.1	–	–
Water depth (cm)	9 ± 2	1–100	–	–
pH	6.89 ± 0.07	5.61–7.80	–	–
EC (μS cm^{-1})	157 ± 14.5	30–440	–	–
DOC (mg l^{-1})	24.2 ± 1.6	5.3–52.9	–	–
TP (μg l^{-1})	221.4 ± 43.5	6.9–1,240	–	–
Water color (400 nm)	0.133 ± 0.014	0.003–0.458	–	–
E2:E3	4.8 ± 0.1	3.1–7.6	–	–

Note. n = 35 for sediment temp in water-filled ditches because deep water prevented temperature measurements being taken in 16 ditches. There are significant ($p \leq 0.001$) differences between water-filled and dry ditches for the first three variables. DOM, dissolved organic carbon; EC, electrical conductivity; TP, total phosphorus.

3.2. Multi-Regional Analysis of Ditch GHG Concentrations

Analysis of the published data set of ditch GHG concentrations revealed no significant difference in mean CH_4 or CO_2 concentrations between ditches draining catchments on mineral or peat soil, but median concentrations of CH_4 were significantly greater in peat catchments (Figure 6). There was a very weak but significant correlation between catchment peat cover and CH_4 concentration (rho = 0.18, p = 0.02), but no relationship with CO_2 (Figure 7).

4. Discussion

4.1. Local Survey of Ditch GHG Fluxes

Our synoptic survey of GHG fluxes from 109 forest ditches revealed an overriding control of ditch water status on CH_4 emissions. The mean emission from dry ditches was zero, and CH_4 uptake was often observed. In contrast to this, relatively large emissions were observed from water-filled ditches, with a mean flux of 72 mg CH_4 m^{-2} d^{-1}. For water-filled ditches, we found a weak positive relationship between CH_4 flux and water depth. For other small waterbodies, such as peatland pools, negative relationships have been found (McEnroe et al., 2009), and this has been attributed to deeper waters being colder and suppressing methanogenesis (Pelletier et al., 2007). However, as suggested elsewhere (Peacock et al., 2017), we assume that ditches with deeper water levels during summer (when we sampled) are likely to have reduced water level fluctuations, and less likely to dry out entirely, which will result in both the development of anoxia and the establishment of microbial communities favorable to CH_4 production. As in other studies of fluvial systems (Campeau & Del Giorgio, 2014; Wallin et al., 2014), CH_4 and CO_2 were positively correlated in our water-filled ditches perhaps because, to some extent, methanogenesis and ecosystem respiration are driven by the same environmental factors (Stanley et al., 2016). It is also feasible that anaerobic metabolism could be a shared source, resulting in oversaturation of CH_4 and CO_2 (Wallin et al., 2018). In contrast to CH_4, our CO_2 fluxes did not differ between water-filled and dry ditches but, like CH_4, we observed a positive correlation between water depth and CO_2 flux. We speculate that this could be because a greater depth indicates a stronger soil-water connection, which results in enhanced lateral inputs of carbon. Intriguingly, we found positive relationships between DOC concentration and both GHGs but only for ditches with standing, rather than flowing, water. This suggests that, due to the increased water residence time, in situ breakdown of DOC contributes to emissions of CO_2 and CH_4 in these standing water bodies.

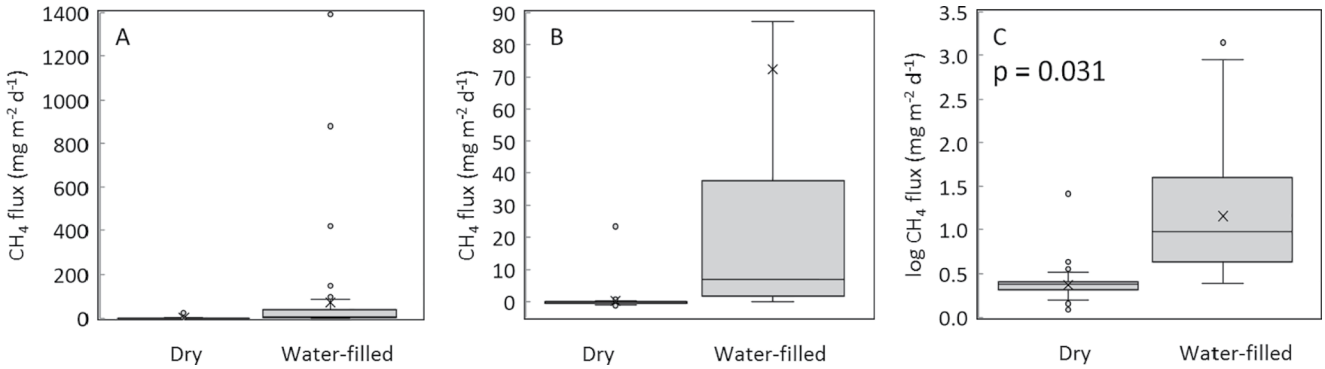

Figure 2. Box plots of diffusive CH_4 fluxes from dry ($n = 58$) and water-filled ($n = 51$) ditches, visualized three ways: (a) all data including outliers; (b) outliers not shown for water-filled ditches; (c) all data, log10 transformed, including outliers. Boxes represent medians and interquartile range (IQR), whiskers mark minimum and maximum values, excluding outliers (calculated as box limits ± 1.5 × IQR). Also shown are mean fluxes (x) and outliers (o). CH_4 fluxes are significantly different between categories (Welch's t-test).

We compared our CH_4 and CO_2 fluxes to other studies of forest ditches (Figure 8). For CH_4, the majority of reported data are from mires and peatlands, but our fluxes are of the same magnitude as most of these. We were only able to find three other studies reporting CO_2 fluxes from forest ditches (on any soil type) but, again, these fall within the same magnitude as our measurements. Assuming a 3-month summer period and upscaling accordingly (i.e., the same flux over the whole period) would give summer emissions of 6.5 g CH_4 m^{-2} and 540 g CO_2 m^{-2}. The annual emissions are likely to be considerably higher, because spring and autumn fluxes can account for a significant proportion of the total flux (Peacock et al., 2017) and winter emissions can be high, even for boreal forest ditches (Minkkinen & Laine, 2006). Nevertheless, the total annual emissions are likely to be lower than those from ditches in more intensively managed landscapes such as grasslands and croplands (Evans et al., 2016). We also acknowledge that there is potential for photosynthetic CO_2 uptake in some of our ditches during the growing season; 60% were unvegetated, but 40% were vegetated to some extent. Von Arnold et al. (2005) used transparent chambers on ditches in a drained spruce forest on mineral soils, and measured a net efflux of CO_2 during the growing season, but the combined use of dark and light chambers would allow a full quantification of photosynthetic uptake versus respiratory emission. We found no effect of vegetation on ditch GHG fluxes but this may be due to the basic nature of our assessment which simply categorized the presence or absence of vegetation. More detailed observations of vegetation type and percentage cover would likely reveal some linkages with GHGs, especially with sedges which have been shown to enhance ditch CH_4 fluxes, either by supplying labile substrates for methanogenesis or by providing a direct conduit for atmospheric CH_4 emission (Cooper et al., 2014; Minkkinen & Laine, 2006).

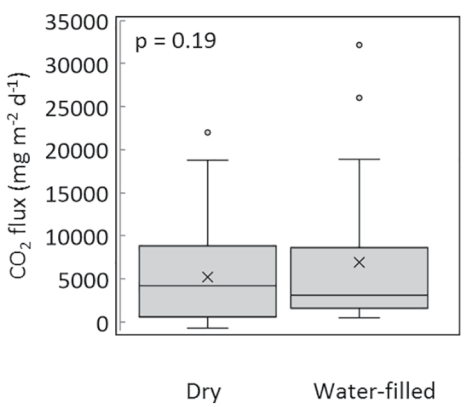

Figure 3. Box plot of CO_2 fluxes from dry ($n = 58$) and water-filled ($n = 51$) ditches. Boxes represent medians and interquartile range (IQR), whiskers mark minimum and maximum values, excluding outliers (calculated as box limits ± 1.5 × IQR). Also shown are mean fluxes (x) and outliers (o). CO_2 fluxes are not significantly different (t-test).

At nine sites our chamber captured ebullition events which, on average, contributed 75% of the total CH_4 flux. This is high compared to the only other report of forest ditch ebullition we are aware of, where CH_4 emissions were dominated by the diffusive pathway (Minkkinen et al., 1997). Data from ditches in other ecosystems provide conflicting evidence, with ebullition being cited as being an important component of total flux (Panneer Selvam et al., 2014; Vermaat et al., 2011) or being negligible (Green et al., 2016; Köhn et al., 2021). The high ebullitive fluxes in our ditches could be due to a nutrient effect (Davidson et al., 2018), as total P concentrations were relatively high.

4.2. Ditches on Mineral Versus Peatland Soils

A comparison with other forest ditch GHG measurements (Figure 8), the majority of which have been taken from peatlands and mires, at least for CH_4, suggests that ditches on mineral soils can emit similar

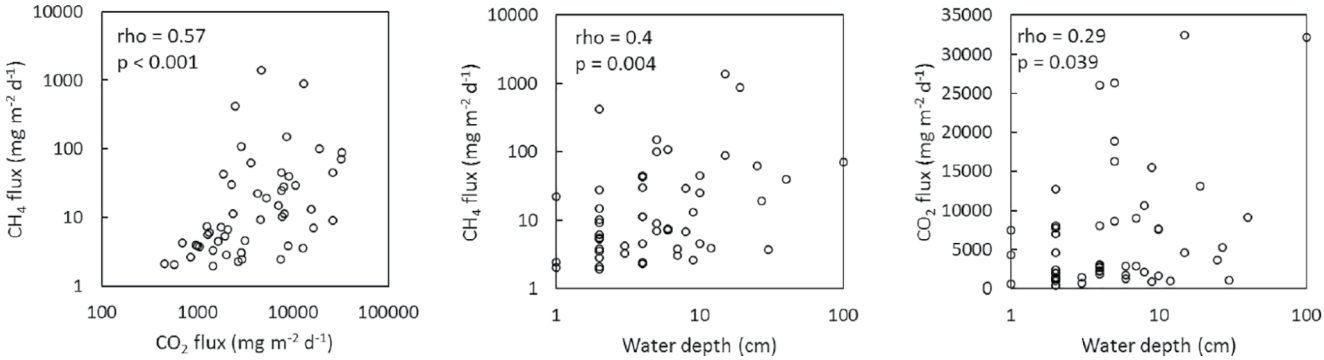

Figure 4. Scatter plots of greenhouse gases and ditch water depth for all water-filled ditches ($n = 51$). Note that CH_4 was transformed by adding 2 to all fluxes so all data points could be displayed on a log axis, but statistical tests were performed on untransformed data.

amounts of GHGs to ditches on peat soils. We further tested this with our multi-regional analysis of GHG concentrations from forest ditches that were repeatedly sampled during spring, summer, and autumn. No significant differences were found in mean CH_4 or CO_2 between ditches draining catchments with exclusively mineral soils, compared to ones that had ≥20% peat cover, but median CH_4 concentrations were significantly greater in peat catchments by a factor of three, and we detected a weak relationship between catchment peat cover and CH_4. The mean CH_4 concentration for mineral soil catchments is beyond the interquartile range (Figure 6) and this (along with the same finding for the CH_4 fluxes in Figure 2) revels the skewed nature of the CH_4 concentration and flux data. The skewed nature of CH_4 data is well known and brings forth the issue of whether to use the median or mean for upscaling and comparisons (Rosentreter & Williamson, 2020). However, assuming that sampling efforts have not been biased toward measuring GHG hotspots, these high emitting sites are representative of the entire population and thus the mean is the better measure (Al-Haj & Fulweiler, 2020; Paneer Selvam et al., 2014). Indeed, our data demonstrate the importance of large-scale, spatially replicated sampling, in order to capture these rare high-CH_4 emitting sites that are important to fully evaluate landscape-scale budgets.

What, then, are the mechanisms for the differences and similarities in GHGs between catchment soil types? First, DOC concentrations have been linked to fluvial CO_2 and CH_4 concentrations (Rasilo et al., 2017; Stanley et al., 2016; Wallin et al., 2014), suggesting a common spatial origin in the landscape of the different

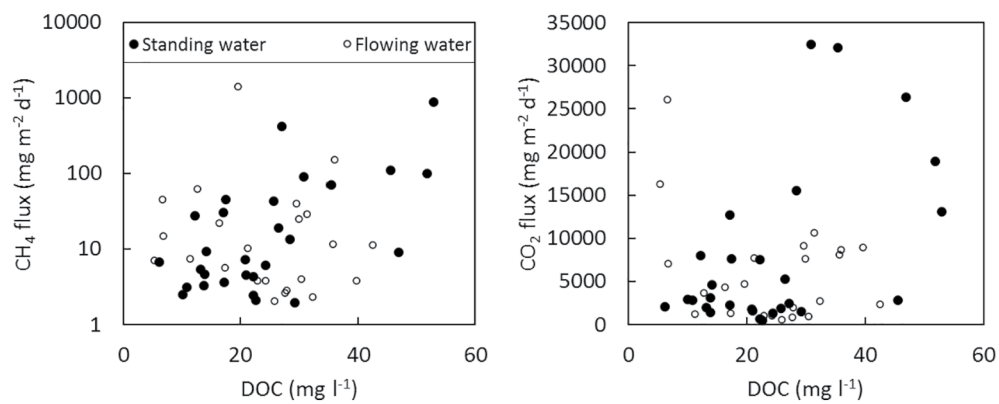

Figure 5. Scatter plots of greenhouse gases and dissolved organic carbon (DOC) concentration, grouped by ditches with standing water (large, black-filled circles, $n = 28$) and flowing water (smaller, white-filled circles, $n = 23$). Pearson correlations are significant for ditches with standing water (for CH_4, $r = 0.53$, $p = 0.004$; for CO_2, $r = 0.54$, $p = 0.003$), but not for ditches with flowing water (for CH_4, $r = -0.08$, $p = 0.7$; for CO_2, $r = -0.3$, $p = 0.16$), or the combined data set of standing + flowing water (for CH_4, $r = 0.19$, $p = 0.17$; for CO_2, $r = 0.27$, $p = 0.052$). Note that CH_4 was transformed by adding 2 to all fluxes so all data points could be displayed on a log axis, but statistical tests were performed on untransformed data.

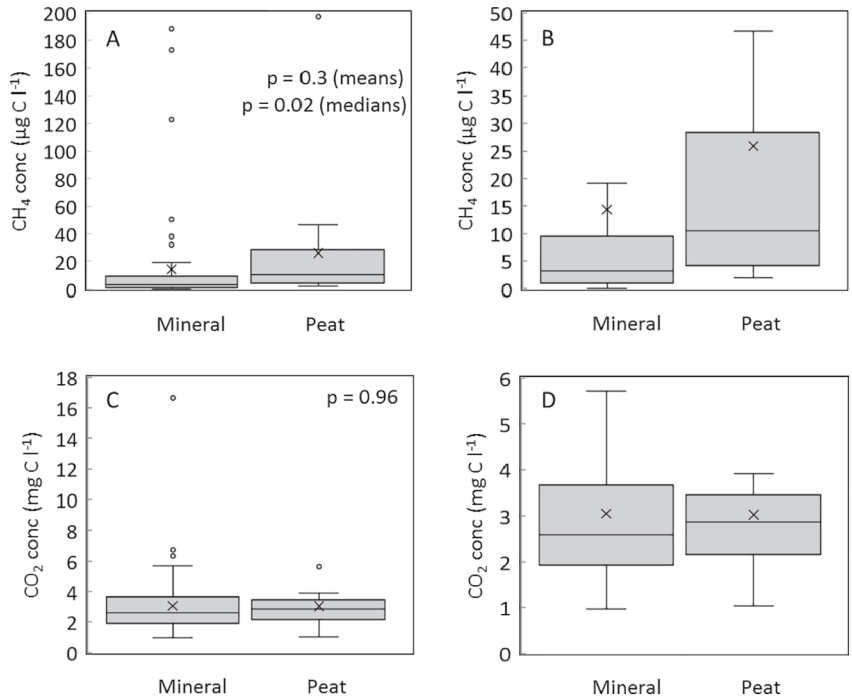

Figure 6. Box plots of dissolved greenhouse gas concentrations in ditches from the multi-regional analysis, grouped according to catchment soil type classification (mineral = 0% peat soil $n = 61$, peat = ≥20% peat soil $n = 15$. Panels (a and c) show all CH_4 and CO_2 data including outliers; panels (b and d) show the same data with no outliers shown. Boxes represent medians and interquartile range (IQR), whiskers mark minimum and maximum values, excluding outliers (calculated as box limits ± 1.5 × IQR). Also shown are mean fluxes (*x*) and outliers (*o*). CO_2 concentrations between categories are not significant (*t*-test). Mean CH_4 concentrations between categories are not significant (*t*-test) but medians are (median test).

carbon species, although not necessarily from a common metabolic source (Campeau et al., 2019). In-keeping with the similarity of GHG concentrations between mineral and peat catchments, DOC concentrations in our multi-regional analysis were also the same in ditches draining peat and mineral catchments (means of ~30 mg l^{-1}) and concentrations in our mineral ditches sampled during the GHG flux survey were also similar (mean = 24 mg l^{-1}). Thus, it appears that, regardless of soil type, lateral inputs of terrestrially derived carbon in shallow groundwater can sustain GHG and DOC concentrations in these ecosystems (Cam-

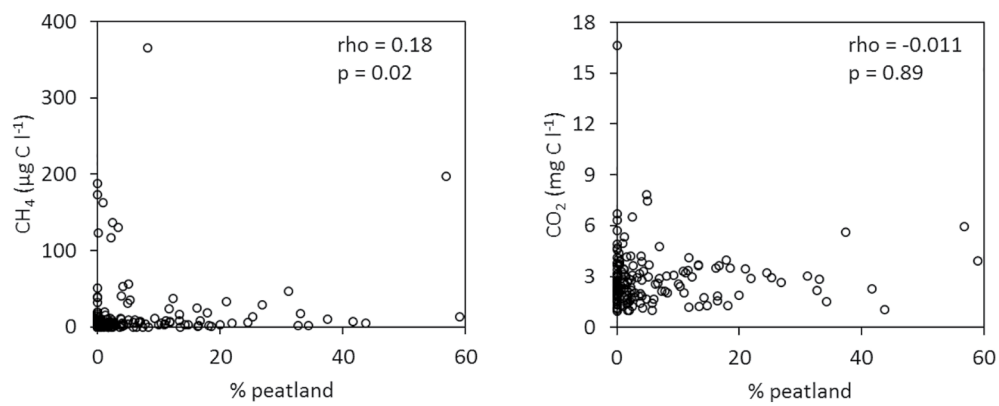

Figure 7. Scatter plots of greenhouse gas concentrations and catchment peatland area ($n = 168$). The relationship for CH_4 is not dependent on the value of ~200 µg C l^{-1} at 57% peatland cover: removing this datapoint results in rho = 0.17, $p = 0.033$.

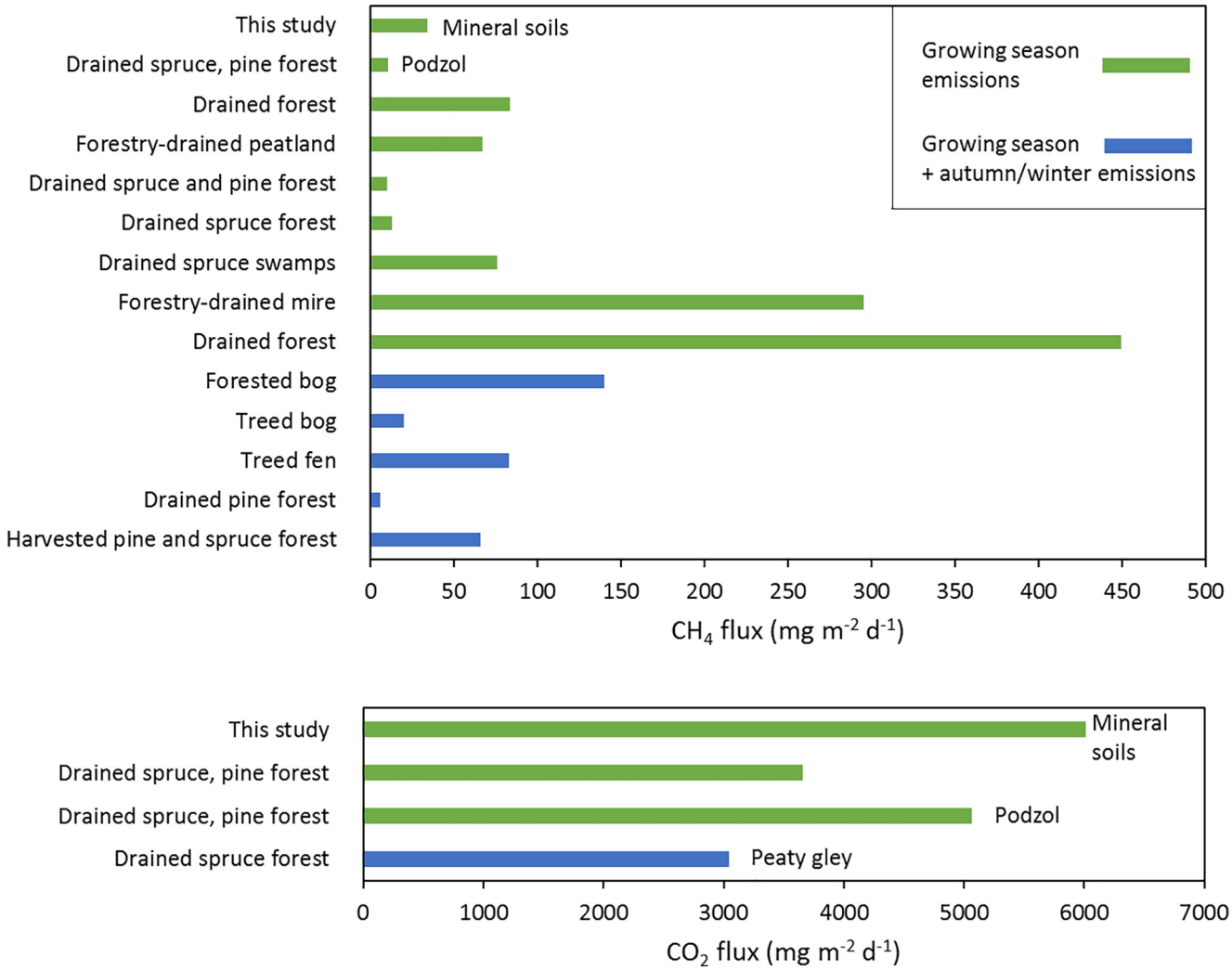

Figure 8. A literature summary of ditch CH_4 (top panel) and CO_2 (lower panel) emissions from ditches in temperate and boreal forests. All studies are on peatland soils unless labeled otherwise. Emissions are color-coded by study period (growing season only, or growing season + autumn/winter measurements). References and additional information (climate zone, measurement period) are in Table S3.

peau et al., 2018). We assume that some DOC removal may occur via mineral soil adsorption but that carbon inputs from small riparian areas, thin organic layers, or even unmapped pockets of peat soils override any adsorption (Leith et al., 2015). When comparing medians and when expressed on a continuous scale (CH_4 vs. % peat cover), the tendency for catchments with more peat cover to have higher CH_4 concentrations is likely an effect of increased lateral inputs from anaerobic soils (Minkkinen & Laine, 2006; Rasilo et al., 2017), which can potentially be sizable even in drained forest peats (Roulet & Moore, 1995). There is also the opportunity for in situ production within the ditch sediments which is likely to be greater in the organic-rich sediments of peatland ditches (Roulet & Moore, 1995). For catchments with greater areas of peat, most of the CH_4 produced in situ and via lateral inputs will likely be emitted rapidly, and to a lesser extent transported down the drainage network. Another possibility is that a difference in DOM composition is implicated; mean E2:E3 in our water-filled mineral ditches was 4.8, considerably higher than values from peatland ditches (3–3.5; Peacock et al., 2018; Strack et al., 2015). In a study of two hemi-boreal headwater forest catchments, Wallin et al. (2015) also found a significantly higher E2:E3 in a stream draining a mineral catchment compared to an adjacent peatland catchment. A lower E2:E3 is indicative of higher aromaticity (Peuravuori & Pihlaja, 1997), which has been shown to be associated with elevated concentrations of aquatic CH_4 (Zhou et al., 2018).

4.3. Implications

Our results show that GHG emissions from ditches on mineral soils in boreal forests are non-negligible. The 2013 IPCC Wetlands Supplement (IPCC, 2014) highlighted peatland drainage ditches as being important sources of CH_4 emission that could influence landscape-scale GHG budgets (Evans et al., 2016) but assumed that drained wetlands on mineral soils had zero CH_4 emissions; that is, it did not consider ditches. Regardless of the pre-drainage status of the soils, the ditches should now be considered for inclusion when calculating the landscape-scale forest GHG balance. To test this, we took values for GHG uptake in boreal forests on mineral soils as 0.52 mg CH_4 m^{-2} d^{-1} (the mean of boreal forest mineral soil fluxes from Dutaur & Verchot, 2007) and 15,200 mg CO_2 m^{-2} d^{-1} (the mean of whole-forest peak summer uptake in Swedish forests, from Chi et al., 2020; Lindroth et al., 1998). We then took our mean ditch CO_2 flux, and our mean CH_4 flux from water-filled ditches only, and calculated what fraction of the landscape water-filled ditches would have to occupy ("$Frac_{ditch}$"; Evans et al., 2016) for their GHG emissions to offset the terrestrial forest uptake. For CH_4, the calculated $Frac_{ditch}$ is 0.007 (0.7%) while for CO_2 $Frac_{ditch}$ is 0.72 (72%). The majority of $Frac_{ditch}$ values, for both peat and mineral soils, are much higher than 0.007, although there are no reported $Frac_{ditch}$ values for forests on mineral soils (Peacock et al., 2021). Care is needed when generalizing our results as they are from summer sampling in one region only. Nevertheless, they suggest, in keeping with research from peatland ditches (Evans et al., 2016), that mineral ditches act as landscape-scale hotspots for CH_4 emission but not CO_2. However, this only holds true for CH_4 when ditches are water-filled. It may therefore be possible to incorporate ditch CH_4 emissions into inventories using simple modeling of water levels (Tucker & Acreman, 2000) or by direct measurements with low-cost sensors (Chapin et al., 2014). These data could be combined with emission factors (which would be zero for dry ditches) for CH_4 accounting, although we note that the current IPCC emission factor for mineral ditches (41.6 g CH_4 m^{-2} yr^{-1}) appears to be inappropriately high when compared to our summer emissions (6.5 g CH_4 m^{-2} for the 3-month period). This is due to a widespread lack of flux data from non-peatland ditches resulting in a single generic emission factor for mineral ditches, regardless of climate zone or nutrient status (IPCC, 2019). Thus, further studies of mineral ditch CH_4, including comprehensive measurements of ebullition, are clearly needed to refine emission factors and enable accurate GHG accounting for managed lands. Future climatic changes in boreal northern Europe are likely to lead to lower amounts of summer rainfall, more frequent droughts, and therefore low water availability (Ruiz-Pérez & Vico, 2020). These changes may have the effect of mitigating growing season ditch CH_4 emissions if watercourses dry out on a regular basis, but higher temperatures may also enhance ditch CO_2 emissions. Furthermore, changes to the timing of precipitation events could increase the occurrence of drought-rewetting cycles which can be hot moments for fluvial GHG emission (Arce et al., 2021; Wallin et al., 2020). Regardless of future changes, it is clear that ditches have the potential to offset, at least to some extent, the NCS of GHG uptake provided by drained forested landscapes.

Data Availability Statement

The data from the survey of ditch GHG fluxes are available in Table S3 in Peacock (2021, https://figshare.com/articles/dataset/Forest_ditch_GHG_data/15152253). The full data (i.e., from all streams) from the multi-regional sampling of ditch GHG concentrations are available at the Uppsala University data repository: http://urn.kb.se/resolve?urn=urn:nbn:se:uu:diva-332472. The specific data analyzed here (i.e., just the watercourses identified as ditches) are available in Table S4 in Peacock (2021, https://figshare.com/articles/dataset/Forest_ditch_GHG_data/15152253).

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