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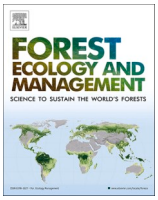

Ditch cleaning in boreal catchments: Impacts on water chemistry and dissolved greenhouse gases in runoff

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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 %).

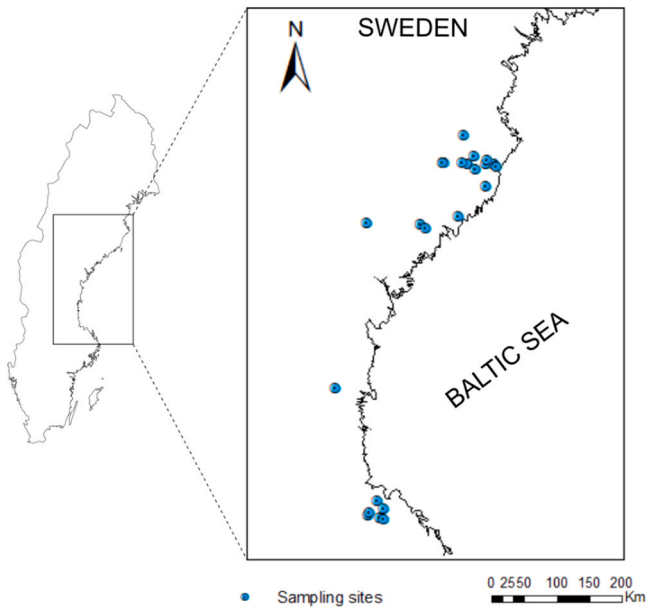

Fig. 1. Location of the sampling sites where ditches were cleaned (DC, n=25). The paired reference sites (REF, n=25) are situated in close vicinity (ca 2 km on average) to each of the cleaned sites and overlap on the map. The map was created in ArcGIS (ESRI).

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

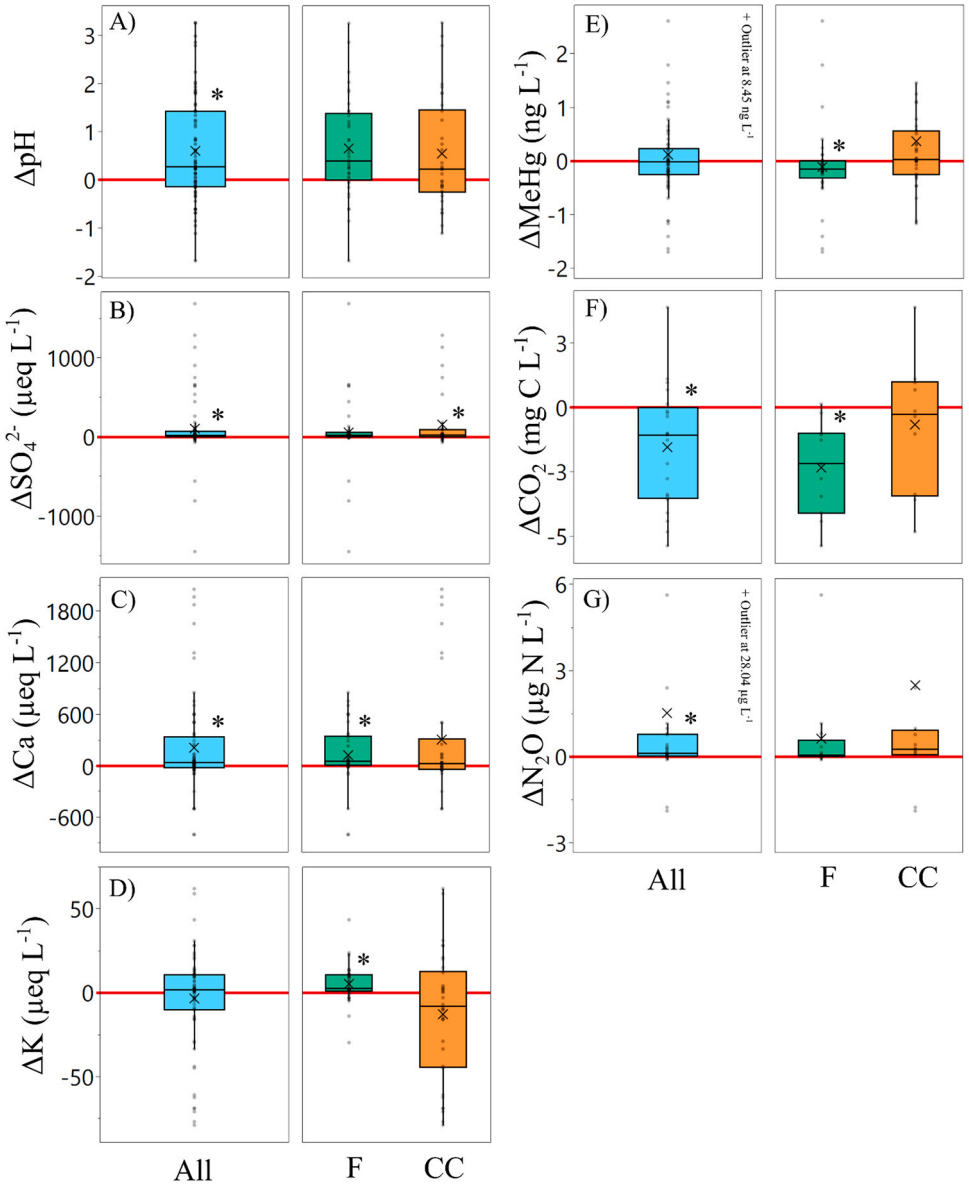

Fig. 2. Boxplots illustrating the distribution in paired Δ values (DC - REF) for each of the variables that were significantly different from zero according to the GLMM. Data is presented for the entire data set (All) and by the forested (F) and clear-cut (CC) sites separately. A value higher than zero 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 GHG's, 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 GHG's 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 GHG's. 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 GHG's (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 GHG's

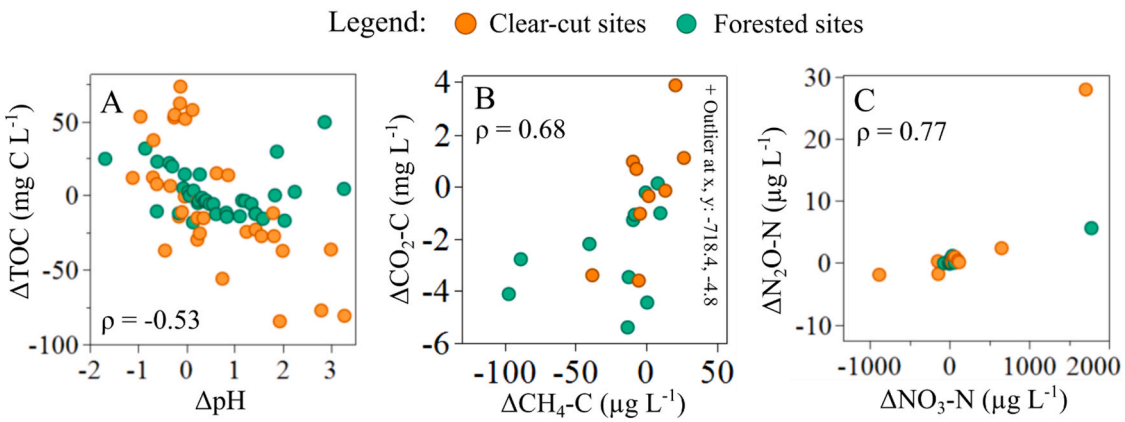

Fig. 3. Selected scatterplots of A) ΔpH vs ΔTOC, B) ΔCH_4-C vs ΔCO_2-C and C) ΔN_2O-N vs ΔNO_3-N, based on the correlation statistics of the variables that were significantly affected by DC listed in Table S3.

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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