



Effects of Forest Ditch Cleaning on Surface Water Quality

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Abstract

Ditch cleaning (DC) is recommended by authorities to Swedish landowners to maintain forest productivity after forest harvest by lowering groundwater tables of catchment soils. Knowledge regarding the impact of this practice on surface water quality is limited, and results from previous research vary. Therefore, this master thesis aims to contribute to the knowledge gap by evaluating the impact of DC in relation to the impact of clear-cutting (CC) on ditch water quality. Additionally, I attempt to investigate whether impacts of DC on water quality can be predicted from catchment characteristics. The study was based on synoptic sampling of water chemistry conducted according to a paired design with 25 cleaned and 25 uncleaned reference ditches located along the east coast of mid-Sweden. The sites were further selected to investigate effects of CC forest harvest, with 25 sites located in direct connection to CC areas and 25 sites located in forested areas (unpaired). Sampling was conducted at three different occasions for 25 chemical variables including standard water chemistry, mercury and dissolved greenhouse gases. The results showed that six out of the 25 chemical variables analysed were significantly different between DC and reference (R) sites, namely pH, sulphate (SO₄), SUVA₂₅₄, carbon dioxide (CO₂-C), methane (CH₄-C), and nitrous dioxide (N₂O-N). A lowered groundwater table and more deeper flow paths following DC is suggested as the main cause for the observed differences in chemical composition. In contrast, following CC higher concentrations of many chemical variables could be linked to more superficial groundwater flow with many of the variables typically linked to organic matter and nutrients. No interaction effect was observed between treatments, meaning that DC does not enhance the impact of CC. However, CO₂-C was significantly lower in DC compared to R, but significantly higher in CC compared to F, indicating that DC counteracts CC in some respects. Finally, the predictability of water quality following DC based on catchment characteristics proved weak. Only Δ SO₄ and Δ CO₂-C models showed predictive relevance, but the low explanatory power of the models suggest that these models should be used with caution and seen as indications rather than used for predictions. The results of this thesis provide important information on how water quality could be affected by DC. The multi-variable inclusion of the current study is an important knowledge basis that can be used as a starting point for more in-depth evaluations of the effects of DC on different water quality aspects.

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Abbreviations

CC	Clear-cutting
CH ₄	Methane
CO ₂	Carbon dioxide
DC	Ditch cleaning
DOC	Dissolved organic carbon
F	Forested
GHG	Greenhouse gas
GIS	Geographical Information Systems
GWT	Groundwater table
Hg	Mercury
K	Potassium
MAT	Mean annual temperature
MAP	Mean annual precipitation
MeHg	Methylmercury
N ₂ O	Nitrous oxide
NH ₄	Ammonium
NO ₂	Nitrite
NO ₃	Nitrate
PO ₄	Phosphate
R	Reference
SO ₄	Sulphate
SUVA ₂₅₄	Specific UV absorbance at 254 nanometers
THg	Total mercury
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphor
VIP	Variable Importance for the Projection

1. Introduction

The practice of draining peatlands to improve tree growth has long been an important practice in Swedish forestry (Eliasson 2008). In areas where groundwater tables (GWTs) and soil water contents are high, roots tend to establish in the upper part of the soil horizon, where gas exchange between soil and air is more rapid (Sikström and Hökkä 2016). A shallow rooting system covers less area, leading to a lower nutrient uptake and reduced growth. Additionally, this increases the risk of drought stress during dry spells (Sikström and Hökkä 2016). To counteract this issue and improve forest productivity, over 1.5 million hectares of peatlands have been artificially drained by ditching in Sweden since the mid-19th century (Wesström et al. 2017). Maintaining or increasing productivity levels within the Swedish forestry sector have primarily been driven by economic benefits and access to wood-based products. During more recent years, increased forest productivity has also been promoted as a mitigation measure to combat climate change (Government Offices of Sweden 2020). The strategy is based on that an increased productivity will lead to more carbon being sequestered in soils and biomass, thereby increasing the forestry sector's potential as a greenhouse gas (GHG) sink (Government Offices of Sweden 2020). Over time, however, the drainage capacity of forest ditches has deteriorated due to sediment accumulation and overgrowth. As forests reach the end of their rotation period, the need to clean the ditches may be essential for seedling survival and to maintain desired productivity levels (Tong et al. 2022).

Ditch cleaning (DC), alternatively referred to as ditch network maintenance (DNM), involves the removal of vegetation and sediment in drainage ditches to lower water table levels and thereby restore their original efficiency (Skogskunskap 2016). The practice is performed in connection to clear-cutting (CC) or tree thinning and ditches are only allowed to be cleaned down to the original depth of the ditch (Skogsstyrelsen 2019). Furthermore, DC is only performed where tree growth is expected to be maintained or increased, otherwise ditches are filled in or left uncleaned (Skogsstyrelsen 2019). In recent years, DC has become increasingly common, and between 2015 and 2019 approximately 10 000 ha of forest lands were cleaned annually (Tong et al. 2022). Though cleaning ditches may be essential to maintain forest productivity, the increased management operations may come at a cost. Stream nutrient concentrations in natural forest and wetland catchments are

generally low and their effect on water quality for downstream recipients can thereby be positive (Djodjic et al. 2021). However, lowering water table levels and removing vegetation can alter the natural hydrology and cause changes in water quality (Mäkitalo 2009; Wessström et al. 2017). In fact, deforestation and other forestry operations have been shown to affect water quality and freshwater biota to a large extent, with final felling being the most studied operation (Nieminen 2003; Palviainen et al. 2014; Kuglerová et al. 2021). Palviainen et al. (2014) found that annual runoff, phosphate ($\text{PO}_4\text{-P}$), total nitrogen (TN), nitrate ($\text{NO}_3\text{-N}$), total organic nitrogen (TON), and suspended solids (SS) loads increased in catchments with >30% clear-cutting and soil preparation. Schelker et al. (2016) reported a 15-fold increase in $\text{NO}_3\text{-N}$ after forest harvest, together with an increase in ammonium ($\text{NH}_4\text{-N}$). Schelker et al. (2012) found that clear-cutting significantly increased average dissolved organic carbon (DOC) concentrations in headwater streams by 3 mg L^{-1} , and that the following site preparation further increased concentrations by 6.2 mg L^{-1} . Furthermore, Wu et al. (2018) found that mercury (Hg) concentrations in large fish increased with 26% in five lakes three years after clear-cutting.

The impact of DC on surface water quality is, however, unlike effects of final felling, poorly understood and has primarily been studied in Finland (e.g., Joensuu et al. 2002; Koivusalo et al. 2008) and with only a few Swedish examples (Hansen et al. 2013). From a hydrological point of view, Koivusalo et al. (2008) reported that whether DC affects the water table level or not depends on the soil type, more specifically the thickness of the peat layer. The water table level was lowered in areas with shallow peat underlain by mineral soil, while no effect was detected in areas with thick peat layers. Additionally, Koivusalo et al. (2008) noted clear increases in annual runoff from sites that had undergone DC compared to control sites. Joensuu et al. (2002) found that DC reduced DOC concentrations by more than 10 mg L^{-1} compared to control areas, and that mean pH increased from 5.6 to 6.3. The same study also found that DC increased the concentration of suspended solids (SS) and base cations in runoff. Similarly, Hansen et al. (2013) reported initial increases in SS, as well as increases in pH, inorganic nitrogen (N) and potassium (K) following DC. They also reported extremely high concentrations of total Hg (THg) during the first couple of days after DC.

Typically, ditches in managed forests form extensive networks and the water they collect sustain downstream rivers and lakes. According to Kuglerová et al. (2021), headwater streams and ditches are the most affected part of river networks following forestry operations. This is because 1) small catchments have a higher proportion of affected area when harvested, 2) the hydrochemical connectivity to catchment soils is high in headwaters, and 3) riparian buffers are often minimal or non-existing in small streams and ditches (Kuglerová et al. 2021). According to Bol et al. (2018), research should be focused on these headwater catchments due to their significant influence on the initial chemical composition of larger river catchments,

where management interventions can be implemented more effectively. The impact forestry and its ditches have on water quality might also depend on catchment variables such as soil type, land use, and vegetation in these headwater catchments. For example, Kortelainen et al. (2006) demonstrated that in a significant majority (~86%) of unmanaged boreal catchments in Finland, the proportion of peat was the best predictor of TOC, iron (Fe) and nitrogen compounds concentrations in first order streams.

The primary objective of DC is to lower the GWT of catchment soils and by that enhance forest productivity. However, this activity can cause altered groundwater flow pathways which in turn affect the chemical composition of water being discharged. There is currently limited knowledge on how DC affects the water quality of surface waters, and the results from the few existing studies vary. Despite these uncertainties, DC is recommended by authorities to landowners to maintain high forest productivity. Hence, there is a strong need for improved understanding on the consequences of DC on water quality. This master thesis aims to contribute to this knowledge gap by evaluating data collected in a paired design of forest ditches (DC and non-DC) in Sweden. I specifically aim to answer the following research questions:

- 1) Does ditch cleaning in boreal catchments influence water quality?
- 2) What are the effects of ditch cleaning on water quality in relation to effects from clear-cut harvest?
- 3) If ditch cleaning influences water quality, is it possible to predict where in the landscape these effects will occur based on catchment characteristics?

2. Methods and materials

2.1 Site description

The study was based on sampling conducted according to a paired design with 25 cleaned and 25 uncleaned reference ditches located across the Västerbotten (n=30), Västernorrland (n=6), Gävleborg (n=2), and Uppsala (n=12) counties (Fig. 1). These 50 sites were further selected to investigate effects of forest harvest, with 25 sites located in direct connection to clearcut areas and 25 sites located in forested areas (unpaired). The sampled sites and data can thereby be divided into 4 groups, i.e., based on the sampled site; ditch cleaned (DC) and reference (R) or clear-cut (CC) and forested (F).

The catchments vary in size from 1 ha to 176 ha, with an overall mean of 30 ha (table 1). Mean annual precipitation (MAP) varied from 400 mm to 600 mm across the catchments with an overall mean of 557 mm. Mean annual temperature (MAT) varied from 2 to 6°C with 3.6°C being the overall mean (table 1). Finally, elevation at the sampling points varied from 30 masl to 287 masl with the overall mean of 133 masl.

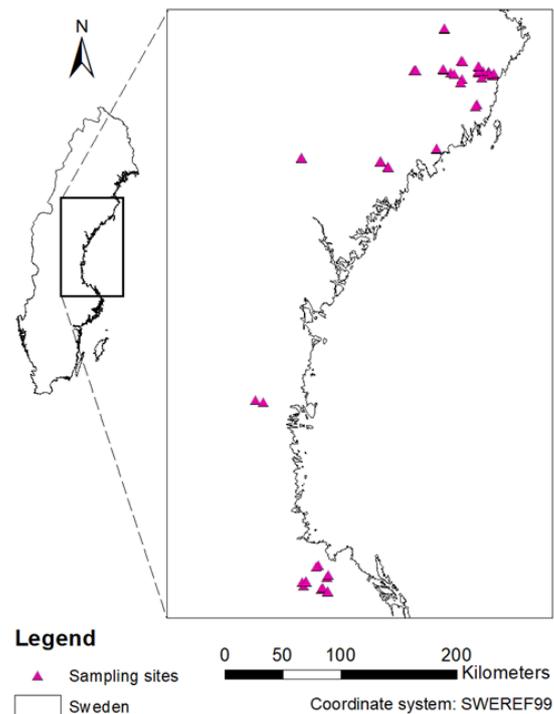


Figure 1. Distribution of sampling sites across the Västerbotten (n=30), Västernorrland (n=6), Gävleborg (n=2), and Uppsala (n=12) counties. The map was created in ArcGIS (ESRI).

Table 1. Mean (min-max) of catchment characteristics for ditch cleaned (DC), reference (R), clear-cut (CC), and forested (F) catchments.

Variable	All	DC	R	CC	F
Number of samples	144	72	72	71	73
Catchment area (ha)	30 (1–176)	25 (1–114)	35 (2–176)	29 (1–176)	30.2 (1–114)
MAP (mm)	557 (400–600)	560 (400–600)	553 (400–600)	513 (400–600)	600 (600–600)
MAT (°C)	3.6 (2–6)	3.6 (2–6)	3.6 (2–6)	3.8 (2–6)	3.3 (2–6)
Elevation (m)	133 (30–287)	132 (30–279)	134 (32–287)	123 (30–287)	143 (37–265)

Soil type and land use varied across catchments and groups (table 2). The main soil type for all four catchment groups was till (52–65%) followed by peat (R and CC) and clay (DC and F). The main land use/vegetation type for all four groups was coniferous forest on wetland (59–65%) followed by clear-cut areas (8–22%).

Table 2. Percentual cover of soil types and land use calculated for all ditch cleaned (DC), reference (R), clear-cut (CC), and forested (F) catchments. Other soil types include water surfaces, shingle, wave-washed sediments (gravel), and bedrock. Other land use includes e.g. arable land and water surfaces.

Variable	DC	R	CC	F
Till (%)	61	57	65	52
Peat (%)	9	19	22	8
Sandy soil (%)	9	3	3	8
Clay soil (%)	10	7	2	14
Other soil types (%)	11	14	8	18
Open wetland (%)	2	3	2	2
Exploited land (%)	1	3	3	1
<i>Forest on wetland</i>				
Coniferous (%)	59	65	61	65
Deciduous (%)	2	3	2	3
<i>Forest outside wetlands</i>				
Coniferous (%)	4	7	8	4
Deciduous (%)	1	1	1	1
CC in catchment (%) *	18	12	22	8
Other land use (%)	13	6	1	16

* The percentual cover is calculated on the catchment scale. The percentual distribution regarding clear-cutting in direct connection to the sampling site is 48%, 52%, 100% and 0%, respectively.

2.2 Geographical information

Geographical information concerning catchment areas, soil type, land use, and climate variables was extracted from data sources using the ArcGIS (version 10.8.2) software. The data basis was collected from different Swedish authorities i.e. Swedish Land Survey, Environmental Protection Agency, Swedish Geological Survey and Forest Agency.

An automatic watershed analysis was conducted to identify catchment areas of the individual sampling sites. In cases where ditches were too small for identifying their associated catchments, or when the automatic analysis did not capture the correct flow direction, catchments were manually delineated from digital elevation models. Information regarding aerial coverage of variables was extracted using zonal tabulation and zonal statistics.

2.3 Analyses and calculations

Water grab samples were taken from the 50 ditches synoptically on three different occasions, June 2021 (1), September 2021 (2), and June 2022 (3). Methane (CH₄) was only sampled on occasion 2 and 3, N₂O was only sampled on occasion 2, and CO₂ was only sampled on occasion 3. See table 3 for all analysed variables.

For general water chemistry, samples were collected in low-density polyethylene bottles that were rinsed three times with stream water prior to sampling. Samples were stored dark in coolers during transport to the laboratories. For descriptions on how general water chemistry was analysed, see the list of accredited analytical methods at the geochemical laboratory at SLU (SLU 2022).

Absorbance at 254 nm was measured on a filtrated sample (0.45 µm) on a Horriba Aqualog spectrophotometer. The unitless absorbance measure (A) at 254 nm was converted into the absorbance coefficient (α) as follows:

$$\alpha = \frac{A}{L} \quad (1)$$

where α is expressed in m⁻¹, A is unitless, and L is the length of the quartz cuvette used (1 cm). Secondly, α was divided by the total organic carbon (TOC) concentration to calculate Specific UV absorbance at 254 nm (SUVA₂₅₄, L mg C⁻¹ m⁻¹) (Weishaar et al. 2003; Wallin et al. 2015). Absorbance at 254 nm was only analysed from sampling 3 and, thereby, SUVA₂₅₄ was only calculated for this occasion. SUVA₂₅₄ can be used to estimate the dissolved aromatic C content in aquatic ecosystems and is thereby a measure of aromaticity (Weishaar et al. 2003). It can therefore also act as a measure of the “quality” of carbon, as aromatic compounds are considered recalcitrant (Kögel-Knabner 2002).

The analyses of THg and MeHg were performed at the Swedish Environmental Research Institute (IVL). Analyses of THg in water followed the EPA 1631 method version E, which involves oxidation with BrCl, reduction to Hg⁰ with SnCl₂, double amalgamation, and subsequent determination through atomic fluorescence spectrometry. Analyses of MeHg in water 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.

The concentrations of CO₂, CH₄ and N₂O were sampled using a headspace equilibration method, where samples were drawn into 60 mL polypropylene syringes and sealed with three-way-stopcocks after removing any bubbles. 30 mL of sample water and 30 mL of ambient air was equilibrated in the syringe by shaking it vigorously for one minute. 15 mL of the equilibrated air was extracted into a 12 mL evacuated exetainer tube (for CO₂ and CH₄) or a 22 mL GC vial (N₂O). Analyses were for CO₂ and CH₄ made on a portable greenhouse gas analyser (Gas scouter, Picarro) using a closed loop system, and for N₂O on a Gas chromatograph equipped with an ECD detector. Concentrations for each gas were calculated using the gas-specific Henry's constant after correcting for temperature, water/headspace volumes as well as gas concentrations in ambient air.

2.4 Statistical analysis

Statistical analyses were carried out using the JMP Pro 16 and the SIMCA17 software's. The first two research questions were tested using a Mixed Model approach in JMP using treatment type, both separately and in interaction with each other, as fixed variables (fixed effects) and measured values as dependent variables (role variable). Sample number (1-3) was set as a repeated structure. The confidence interval was set to 95% and all data was log transformed. In total, 25 dependent variables were screened, including general chemistry, Hg and the different greenhouse gases. To visualize where differences occurred in the Mixed Model, bar charts were created using the R software (version 4.2.2).

In the next step of the analysis, concentration differences (Δ) for all chemical variables between cleaned and reference ditches were calculated and used in a Partial Least Square (PLS) regression (SIMCA17). PLS is suitable when there are interdependent explanatory variables in the dataset (Eriksson et al. 1999 see Wallin et al. 2014). The Variable Importance for the Projection (VIP) value, calculated by summing the squares of the PLS loading weights for each explanatory variable, was used to determine the importance of each explanatory variable in the PLS model. VIP values ≥ 1 was considered to be of importance for explaining variability explanatory variables whereas VIP values < 1 were considered unimportant.

3. Results

3.1 Overall ranges in observed water chemistry

There was a wide range in water chemistry found across all sampled sites. pH varied between 3.8 and 8.1, alkalinity/acidity between -0.56 and 5.29 meq L^{-1} and SO_4 concentrations between 0.01 and 1.70 meq L^{-1} (table 3). Conductivity varied between 1.3 and 51.6 mS/m^2 and showed similar mean values across all treatments. Absorbance at 420 and 436 nm varied between 0.06 - 3.07 and 0.04 - 2.41 , respectively. TOC concentrations varied between 4.6 and 165 mg L^{-1} , and SUVA_{254} between 2.6 and $5.7 \text{ L mg C}^{-1} \text{ m}^{-1}$. A large span was seen in nutrient concentrations, where TN varied between 132 - 11800 , $\text{NH}_4\text{-N}$ between 1.5 - 2630 , nitrite and nitrate (NO_2^- and $\text{NO}_3\text{-N}$) between 0.5 - 10500 , TP between 4.5 - 981 , and $\text{PO}_4\text{-P}$ between 0.5 - $694 \mu\text{g L}^{-1}$. Regarding base cation concentrations, Ca varied between 0.03 - 5.49 , Mg between 0.02 - 0.52 , Na between 0.03 - 0.96 , and K between 0.001 - 0.10 meq L^{-1} . Anion concentrations, Cl and F, varied between 0.004 - 1.0 meq L^{-1} and 0.03 - 1.3 mg L^{-1} , respectively. Total and methyl mercury varied between 1.3 - 48 and 0.03 - 8.9 ng L^{-1} , respectively. Finally, regarding GHG concentrations, $\text{CO}_2\text{-C}$ varied between 1 - 7.2 mg L^{-1} , $\text{CH}_4\text{-C}$ between 0.7 - $723.3 \mu\text{g L}^{-1}$, and $\text{N}_2\text{O-N}$ between 0.3 - $28.5 \mu\text{g L}^{-1}$.

Table 3. Mean (min-max) of all analysed variables for ditch cleaned (DC), reference (R), clear-cut (CC), and forested (F) catchments.

Variable	All	DC	R	CC	F
pH	5.5 (3.8-8.1)	5.8 (3.8-8.1)	5.2 (3.8-7.7)	5.5 (3.8-7.7)	5.6 (3.8-8.1)
Alk./Acid (meq L ⁻¹)	0.42 (-0.56-5.29)	0.47 (0.47-4.14)	0.38 (-0.56-5.29)	0.55 (-0.56-5.29)	0.31 (-0.47-3.21)
SO ₄ (meq L ⁻¹)	0.16 (0.01-1.70)	0.21 (0.01-1.70)	0.10 (0.01-1.60)	0.13 (0.01-1.40)	0.19 (0.01-1.70)
Conductivity (mS/m25)	9.3 (1.3-51.6)	9.8 (1.3-49.1)	8.7 (1.6-51.5)	11 (1.3-51.6)	7.6 (1.3-31.8)
Abs ₄₂₀	0.77 (0.06-3.07)	0.74 (0.06-2.83)	0.81 (0.08-3.07)	1.05 (0.06-3.07)	0.51 (0.07-1.18)
Abs ₄₃₆	0.61 (0.04-2.41)	0.58 (0.04-2.29)	0.64 (0.06-2.42)	0.82 (0.04-2.42)	0.40 (0.05-0.96)
TOC (mg L ⁻¹)	39.2 (4.6-165)	38.9 (4.6-108)	39.4 (8.5-165)	51.8 (4.6-165)	26.9 (4.7-62.7)
SUVA ₂₅₄ (L mg C ⁻¹ m ⁻¹)	4.4 (2.6-5.7)	4.1 (2.6-5.7)	4.6 (3-5.6)	4.4 (2.6-5.7)	4.3 (3.3-5.2)
TN (µg L ⁻¹)	1333.7 (132-11800)	1533.4 (132-11800)	1139.4 (307-6080)	1918.7 (144-11800)	764.7 (132-4210)
NH ₄ -N (µg L ⁻¹)	109.8 (1.5-2630)	93.8 (1.5-2630)	125.4 (5-2190)	206.1 (1.5-2630)	16.2 (1.5-159)
NO ₂ - & NO ₃ -N (µg L ⁻¹)	159.2 (0.5-10500)	261.7 (0.5-10500)	59.5 (1-910)	277.7 (0.5-10500)	43.9 (0.5-1800)
TP (µg L ⁻¹)	59.5 (4.5-981)	50.5 (4.6-353)	68.1 (4.5-981)	93.9 (4.6-981)	26 (4.5-113)
PO ₄ -P (µg L ⁻¹)	23.6 (0.5-694)	12 (0.5-195)	34.9 (0.5-694)	43.3 (0.5-694)	4.5 (0.5-27)
Ca (meq L ⁻¹)	0.70 (0.03-5.49)	0.82 (0.04-5.49)	0.59 (0.03-4.99)	0.90 (0.04-5.49)	0.52 (0.03-3.10)
Mg (meq L ⁻¹)	0.11 (0.02-0.52)	0.13 (0.02-0.52)	0.10 (0.02-0.47)	0.12 (0.02-0.40)	0.11 (0.02-0.52)
Na (meq L ⁻¹)	0.11 (0.03-0.96)	0.11 (0.03-0.40)	0.12 (0.03-0.96)	0.13 (0.03-0.96)	0.09 (0.03-0.37)
K (meq L ⁻¹)	0.02 (0.001-0.10)	0.02 (0.003-0.09)	0.02 (0.001-0.10)	0.03 (0.001-0.10)	0.01 (0.001-0.05)
Cl (meq L ⁻¹)	0.05 (0.004-1.0)	0.03 (0.01-0.1.0)	0.06 (0.004-1.0)	0.06 (0.004-1.0)	0.03 (0.01-0.07)
F (mg L ⁻¹)	0.3 (0.03-1.3)	0.3 (0.03-1.3)	0.3 (0.03-1.3)	0.3 (0.03-1.3)	0.2 (0.03-0.5)
Si (mg L ⁻¹)	4.8 (0.8-15)	5.0 (0.9-15)	4.5 (0.8-13)	4.5 (0.8-12)	5.0 (1.1-15)
THg (ng L ⁻¹)	9.6 (1.3-48)	9.9 (1.5-39)	9.3 (1.3-48)	12.5 (1.3-48)	6.7 (1.5-15)
MeHg (ng L ⁻¹)	0.7 (0.03-8.9)	0.8 (0.03-8.9)	0.6 (0.03-3)	0.7 (0.03-8.9)	0.7 (0.03-2.9)
CO ₂ -C (mg L ⁻¹)	3.1 (1-7.2)	2.4 (1-6.2)	3.9 (1.6-7.2)	3.6 (1.2-6.8)	2.7 (1-7.2)
CH ₄ -C (µg L ⁻¹)	20.7 (0.7-723.3)	9.1 (0.7-54.8)	31.8 (1.3-723.3)	29 (1.3-723.3)	12.8 (0.7-105.8)
N ₂ O-N (µg L ⁻¹)	1.6 (0.3-28.5)	2.4 (0.4-28.5)	0.8 (0.3-5.7)	2.3 (0.3-28.5)	0.8 (0.4-6.4)

There were differences in the effects of CC and DC on water quality, i.e. the variables with significant differences seen between DC and R sites where different

from the variables with significant differences between CC and F sites. Only one variable (CO₂) was significantly affected by both treatments (table 4). Furthermore, no interaction effect between treatments was observed, i.e. the combination of both treatments had no significant effect on variables, indicating that DC does not enhance the effect of CC.

Table 4. P-values for all variables and treatments together with the combination of both treatments (interaction). Where there is a significant difference, arrows indicate whether values were higher or lower in CC compared to F and in DC compared to R.

Variable	CC vs F	DC vs R	Interaction
pH	0.3918	0.0029	↑ 0.8519
Alk/acid	0.3791	0.6916	0.5782
SO ₄	0.0683	0.0001	↑ 0.1822
Conductivity	0.0885	0.4322	0.967
Abs ₄₂₀	<0.0001	↑ 0.4014	0.564
Abs ₄₃₆	<0.0001	↑ 0.3857	0.5451
TOC	<0.0001	↑ 0.667	0.9426
SUVA ₂₅₄	0.8772	0.0452	↓ 0.8683
TN	<0.0001	↑ 0.403	0.5206
NH ₄ -N	<0.0001	↑ 0.0555	0.8196
NO ₂ - & NO ₃ -N	<0.0001	↑ 0.4372	0.1995
TP	<0.0001	↑ 0.5792	0.3085
PO ₄ -P	0.0002	↑ 0.7196	0.3673
Ca	0.2577	0.0508	0.8801
Mg	0.3246	0.0695	0.7149
Na	0.122	0.8389	0.3078
K	<0.0001	↑ 0.0909	0.0751
Cl	0.089	0.1847	0.5389
F	0.0589	0.6144	0.2251
Si	0.2068	0.2372	0.276
THg	0.0003	↑ 0.5788	0.6308
MeHg	0.7919	0.4516	0.0876
CO ₂ -C	0.038	↑ 0.0002	↓ 0.0623
CH ₄ -C	0.2028	0.0487	↓ 0.3487
N ₂ O-N	0.0973	0.0225	↑ 0.5273

3.2 Forest harvest effects

Absorbance, both at 420 nm and 436 nm, was higher in CC sites (1.05 and 0.82) compared to F sites (0.51 and 0.40) ($p < 0.0001$) (Fig. 2). TOC concentration was also higher in CC sites (51.8 mg L⁻¹) than in F sites (26.9 mg L⁻¹) ($p < 0.0001$). CC sites showed significantly higher K and THg concentrations (0.03 meq L⁻¹ and 12.5 ng L⁻¹) than F sites (0.01 meq L⁻¹ and 6.7 ng L⁻¹) ($p < 0.0001$ and $p = 0.0003$,

respectively). Lastly, CO₂-C levels were higher in CC sites (3.6 mg L⁻¹) than in F sites (2.7 mg L⁻¹) (p=0.038).

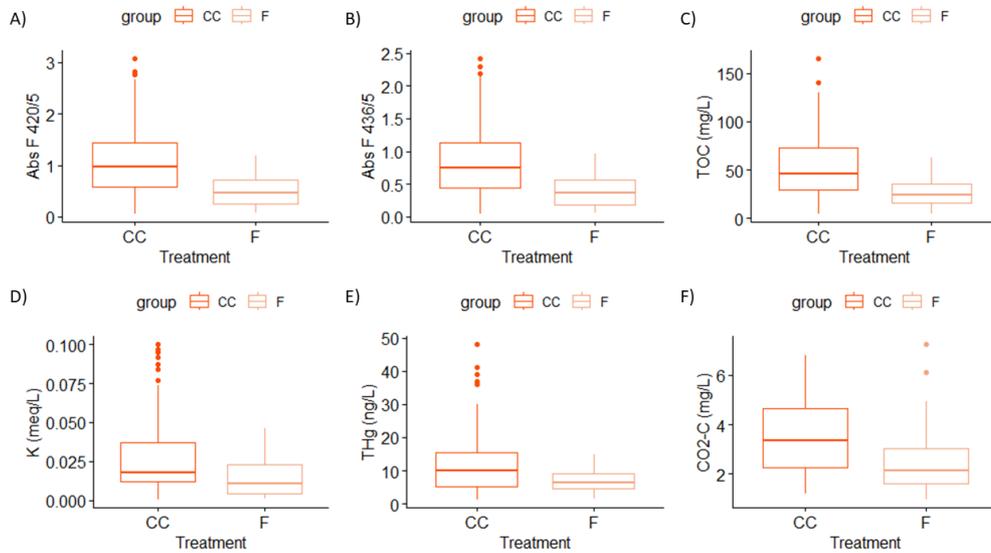


Figure 2. Concentrations (Y-axis) of variables significantly differing between clear-cut (CC) and forested (F) sites, A) Abs₄₂₀, B) Abs₄₃₆, C) TOC (mg L⁻¹), D) K (meq L⁻¹), E) THg (ng L⁻¹), and F) CO₂-C (mg L⁻¹).

All nitrogen compounds tested (TN, NH₄, and NO₂⁻ & NO₃⁻-N) were significantly higher in CC sites (1918.7, 206.1 and 277.7 μg L⁻¹) than in F sites (764.7, 16.2 and 43.9 μg L⁻¹) (p<0.0001) (Fig. 3). Furthermore, both total phosphorous (TP) and PO₄ concentrations were higher in CC sites (93.9 and 43.3 μg L⁻¹) compared to F sites (26 and 4.5 μg L⁻¹) (p=0.0002 and p<0.0001, respectively).

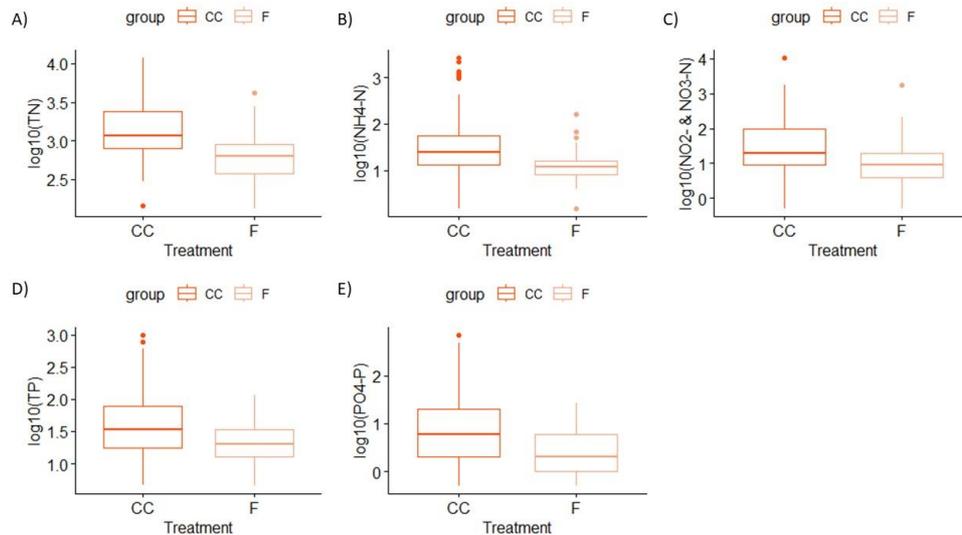


Figure 3. Log-scaled nitrogen and phosphorus concentrations (total and fractions) (Y-axis) significantly differing between clear-cut (CC) and forested (F) sites, A) TN (μg L⁻¹), B) NH₄-N (μg L⁻¹), C) NO₂⁻ & NO₃⁻-N (μg L⁻¹), D) TP (μg L⁻¹), and E) PO₄-P (μg L⁻¹).

3.3 Ditch cleaning effects

pH and SO_4 was significantly higher in DC sites (5.8 and 0.21 meq L^{-1}) than in R sites (5.2 and 0.10 meq L^{-1}) ($p=0.003$ and $p=0.0001$, respectively) (Fig. 4). SUVA_{254} was significantly lower in DC sites (4.1 $\text{L mg C}^{-1} \text{m}^{-1}$) than in R sites (4.6 $\text{L mg C}^{-1} \text{m}^{-1}$) ($p=0.452$) though TOC was not affected. All GHGs tested were significantly affected by DC. CO_2 and CH_4 concentrations were both lower in DC sites (2.4 mg L^{-1} and 9.1 $\mu\text{g L}^{-1}$) compared to R sites (3.9 mg L^{-1} and 31.8 $\mu\text{g L}^{-1}$), whereas N_2O concentrations were higher in DC sites (2.4 $\mu\text{g L}^{-1}$) than in R sites (0.8 $\mu\text{g L}^{-1}$) ($p=0.0002$, $p=0.0487$, and $p=0.0225$, respectively).

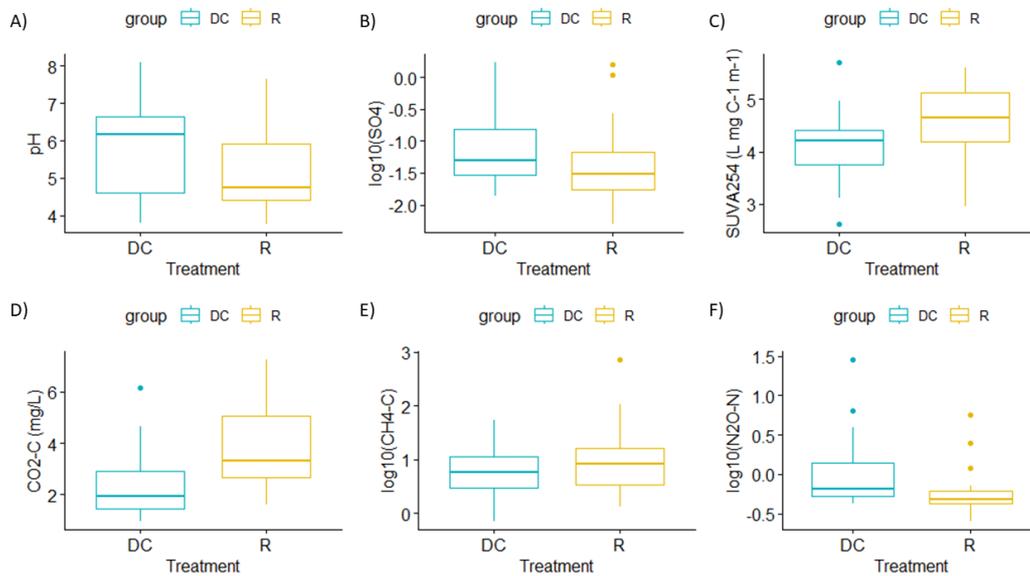


Figure 4. Concentrations (Y-axis) of all variables significantly affected by ditch cleaning, A) pH, B) $\log_{10}[\text{SO}_4]$, C) SUVA_{254} ($\text{L mg C}^{-1} \text{m}^{-1}$), D) $\text{CO}_2\text{-C}$ (mg L^{-1}), E) $\log_{10}[\text{CH}_4\text{-C}]$ ($\mu\text{g L}^{-1}$), and F) $\log_{10}[\text{N}_2\text{O-N}]$ ($\mu\text{g L}^{-1}$). Note that some of the variables are log-scaled

3.4 Predicting water quality effects of ditch cleaning

Solute-specific PLS models were constructed for the chemical variables where an effect of DC was observed, i.e. that displayed significant concentration differences between DC and R sites. The six PLS models displayed cumulative R^2Y -values between 0.18-0.69, but all models except SO_4 and CO_2 displayed negative Q^2 -values indicating low to no predictive relevance (table 5).

The catchment variables that explained most of the variability in pH in ditch water are %deciduous forest outside wetlands (VIP=1.74), %deciduous forest on wetland (VIP=1.56), %sandy soil (VIP=1.38), elevation (VIP=1.26), %CC in the catchment (VIP=1.18), and %open wetland (VIP=1.18) (table 5). pH was positively correlated with %deciduous forest outside wetlands, elevation and %open wetland

and negatively correlated with %deciduous forest on wetland, %sandy soil, and %CC in the catchment.

The catchment variables that explained most of the variability in SO_4 concentrations in ditch water are %deciduous forest on wetland (VIP=2.15), %clay soil (VIP=1.71), MAP (VIP=1.40), MAT (VIP=1.39), %CC in the catchment (VIP=1.15), and %coniferous forest outside wetlands (VIP=1.02) (table 5). All variables were positively correlated with SO_4 concentrations except for MAP, which showed a negative correlation.

The catchment variables that explained most of the variability in SUVA_{254} concentrations in ditch water are MAT (VIP=1.55), %exploited land (VIP=1.47), %open wetland (VIP=1.32), MAP (VIP=1.30), elevation (VIP=1.25), area (VIP=1.18), %CC in the catchment, and %CC at the sample site (VIP=1.01) (table 5). SUVA_{254} was positively correlated with MAT, %exploited land, elevation, and area, and negatively correlated with %open wetland, MAP, %CC in the catchment, and %CC at the sample site.

The catchment variables that explained most of the variability in CO_2 concentration were MAP (VIP=1.82), %peat (VIP=1.79), %CC at the sample site (VIP=1.50), MAT (VIP=1.35), %coniferous forest on wetland (VIP=1.14), and %CC in the catchment (VIP=1.05). All variables once again showed positive correlations with CO_2 concentrations except for MAP, which showed a negative correlation.

The catchment variables that explained most of the variability in CH_4 concentrations in ditch water are %coniferous forest on wetland (VIP=2.18), MAP (VIP=1.37), %CC in the catchment (VIP=1.25), elevation (VIP=1.25), %coniferous forest outside wetlands (VIP=1.16), MAT (VIP=1.07), and %CC at the sample site (VIP=1.04) (table 5). CH_4 was positively correlated with %coniferous forest on wetland, elevation, and MAT, and negatively correlated with MAP, %CC in the catchment, %coniferous forest outside wetlands, and %CC at the sample site.

The catchment variables that explained most of the variability in N_2O concentrations in ditch water are %clay soil (VIP=1.90), MAP (VIP=1.33), MAT (VIP=1.30), elevation (VIP=1.24), %coniferous forest on wetland (VIP=1.11), and %CC in the catchment (VIP=1.05) (table 5). All variables were positively correlated with N_2O except for MAP which showed a negative correlation. For PLS figures including all catchment variables, see appendix 1-6.

Table 5. Statistics of the PLS models that show X variables with VIP values >1 for all Y variables significantly affected by DC.

Y Variable	X Variable	VIP	Coefficient	R ² Y	Q ²
Δ pH	Deciduous (%)	1.74	Positive	0.357	-0.0937*
	Deciduous on wetland (%)	1.56	Negative		
	Sandy soil (%)	1.38	Negative		
	Elevation (m)	1.26	Positive		
	CC catchment (%)	1.18	Negative		
	Open wetland (%)	1.18	Positive		
Δ SO ₄	Deciduous on wetland (%)	2.15	Positive	0.321	0.0652
	Clay soil (%)	1.71	Positive		
	MAP (mm)	1.40	Negative		
	MAT (°C)	1.39	Positive		
	CC catchment (%)	1.15	Positive		
	Coniferous (%)	1.02	Positive		
Δ SUVA ₂₅₄	MAT (°C)	1.55	Positive	0.543	-0.21*
	Exploited land (%)	1.47	Positive		
	Open wetland (%)	1.32	Negative		
	MAP (mm)	1.30	Negative		
	Elevation (m)	1.25	Positive		
	Area (ha)	1.18	Positive		
	CC catchment (%)	1.12	Negative		
	CC sample site (%)	1.01	Negative		
Δ CO ₂	MAP (mm)	1.82	Negative	0.352	0.0769
	Peat (%)	1.79	Positive		
	CC sample site (%)	1.50	Positive		
	MAT (°C)	1.35	Positive		
	Coniferous on wetland (%)	1.14	Positive		
	CC catchment (%)	1.05	Positive		
Δ CH ₄	Coniferous on wetland (%)	2.18	Positive	0.18	-0.21*
	MAP (mm)	1.37	Negative		
	CC catchment (%)	1.25	Negative		
	Elevation (m)	1.25	Positive		
	Coniferous (%)	1.16	Negative		
	MAT (°C)	1.07	Positive		
	CC sample site (%)	1.04	Negative		
Δ N ₂ O	Clay soil (%)	1.90	Positive	0.689	-0.21*
	MAP (mm)	1.33	Negative		
	MAT (°C)	1.30	Positive		
	Elevation (m)	1.24	Positive		
	Coniferous on wetland (%)	1.11	Positive		
	CC catchment (%)	1.05	Positive		

4. Discussion

Ditch cleaning is a common practice to enhance forest productivity by lowering the groundwater table of catchment soils. However, by altering the groundwater flow pathways, this activity can have unintended consequences that impact the chemical composition of water being discharged. The impact of DC on surface water quality is currently not well understood, with limited research available and inconsistent findings. Despite these uncertainties, DC is recommended for landowners in order to maintain or increase high forest productivity.

4.1 Ditch cleaning effects on water quality

The results showed that six out of the 25 variables analysed were significantly different between DC and R sites (pH, SO_4 , SUVA_{254} , CO_2 , CH_4 , and N_2O).

The elevated pH in DC sites compared to R sites is consistent with the findings of Joensuu et al. (2002) and Hansen et al. (2013). The observed mean pH was 0.6 units higher in DC compared to R sites (5.8 and 5.2, respectively). One explanation for the higher pH in DC sites is that when the ditch is cleaned, the groundwater table is lowered leading to deeper flowpaths in soil layers where pH is typically higher, which subsequently leads to a higher pH in runoff (Joensuu et al. 2002, see Lundin 1996). Furthermore, the pH of water is partly controlled by the proportions of dissolved inorganic carbon (DIC) constituents (Cole and Prairie 2014). When pH is 5.3, CO_2 is the only DIC constituent present in significant quantities, but as pH rises both CO_2 and bicarbonate (HCO_3^-) is present in significant quantities (Cole and Prairie 2014). The higher pH in DC sites could thereby be explained by the lower CO_2 concentrations seen in DC compared to R sites.

Mean SO_4 concentrations were twice as high in DC sites (0.21 meq L^{-1}) compared to R sites (0.10 meq L^{-1}), supporting the findings of Hansen et al. (2013), who reported an increase in $\text{SO}_4\text{-S}$ concentrations after DC from 0.44 to 0.51 and 1.12 to 1.60 mg L^{-1} in two separate Swedish catchments. Ukonmaanaho et al. (2014) conducted a study in seven coniferous forest sites in southern and northern Finland and found a positive correlation between soil depth and SO_4 concentration. Hence, the higher SO_4 concentrations observed in DC compared to R sites might therefore, alike pH, be attributed to the lowering of GWTs leading to dominating flow paths through deeper soil layers that store more sulphur-rich parent material.

DOC did not significantly differ between DC and R sites which, according to Klavina et al. (2021), could be because groundwater levels were not so much affected by DC. However, SUVA₂₅₄, i.e., the measure of aromaticity, was significantly lower in cleaned ditches compared to reference ditches (mean SUVA₂₅₄ of 4.1 and 4.6 L mg C⁻¹ m⁻¹, respectively). This suggests that the groundwater flow paths did change due to DC and thereby TOC of different character is exported, although at similar concentrations. Leach et al. (2016) found, however, that SUVA₂₅₄ increased in runoff from a boreal peatland during low flow summer conditions as a result of runoff originating from deeper, more decomposed peat layers. Instead of shifting flow paths, removal of vegetation in the ditches may have caused the lower SUVA₂₅₄ in DC sites, as the removal of vegetation subsequently leads to less decomposed material in ditches.

CO₂-C concentrations were significantly lower in DC sites compared to R sites (p=0.0002), with mean values of 2.4 and 3.9 mg L⁻¹, respectively (table 2). Removing ditch vegetation leaves less substrate available for decomposition, thereby less C and subsequently less CO₂ in the water. However, Peacock et al. (2021) found that whether the ditch is vegetated or not makes no difference on atmospheric CO₂ fluxes from the ditch, indicating that it is not the removal of vegetation by DC that causes concentrations to be lower. If the higher pH observed in DC sites is in fact attributed to runoff/leaching from deeper soil layers as discussed previously, the lower observed CO₂ concentrations could be an effect of a shift in the DIC speciation towards lower CO₂ at higher pH. It should be noted that CO₂ was only sampled on occasion 3 (June 2022), which might have affected the results due to seasonal variation.

CH₄-C concentrations were significantly lower in DC sites compared to R sites (p=0.05), with mean values of 9.1 and 31.8 µg C L⁻¹, respectively (table 2). Higher CH₄ concentrations in R sites is likely due to higher water table levels and thereby more widespread anaerobic conditions in soils, as these conditions are fundamental for CH₄ production (Wallin et al. 2014). Furthermore, undisturbed sites have higher microbial diversity, carbon abundance, and decomposition rates, favouring methanogenesis (Bitenieks et al. 2022). It should be noted that CH₄ was only sampled on occasion 2 and 3 (September 2021 and June 2022).

N₂O-N concentrations were higher in DC compared to R sites (p=0.02), with a mean difference of 1.6 µg L⁻¹ (2.4 and 0.8 µg L⁻¹, respectively). A mean concentration of 2.4 µg L⁻¹ in DC sites is 1.0 µg L⁻¹ higher than the overall mean concentrations across Swedish low-order streams in both forested and agricultural catchments found by Audet et al. (2020). The authors further found that N₂O concentrations were negatively correlated with pH and suggested that in more acidic soils, N₂O is more likely to be the terminal product of denitrification whereas N₂ is more likely in soils with a higher pH (Čuhel et al. 2010). This is in contrast to the results of the current thesis as both pH and N₂O-N were higher in DC compared

to R sites. Higher N₂O-N concentrations could also be attributed to C and N availability (Mosier et al. 1998). However, no significant difference was seen in TN, NO₂-N and NO₃-N, or TOC. This opens the question of whether the higher N₂O-N concentrations found in DC sites could be attributed to the different character of the organic matter as suggested by the difference in SUVA₂₅₄ between DC and R sites. An additional potential explanation could be enhanced chemodenitrification (abiotic reduction of NO₂⁻ and NO₃⁻ by ferrous iron) due to higher iron (Fe) concentrations in DC sites (Grabb et al. 2017). However, Fe was not measured in this study, so this is only speculative. Moreover, it should be noted that N₂O was only sampled on occasion 2 (September 2021), which might have affected the results due to seasonal variation.

Furthermore, previous research has reported very high levels of Hg (in early stages) after ditch cleaning (e.g. Hansson et al. 2013). This effect was not seen in this study, which might be a result of the synoptic sampling not being done “early enough”, as the high spikes in Hg leaching has previously been observed within a few days after DC while the first synoptic sampling in this study was done up to three years after DC (Hansson et al. 2013).

4.2 Clear-cut harvest effects in relation to ditch cleaning effects

The results showed that CC affected other variables than DC did. Variables that significantly differed between CC sites and F sites were related to organic matter (OM) (absorbance and TOC), nutrients (TN, NH₄, NO₂ and NO₃, TP, PO₄), and THg. All variables showed higher levels or concentrations in CC sites compared to F sites.

Specifically, mean TOC concentrations were higher in CC compared to F sites, with values of 51.8 and 26.9 mg L⁻¹, respectively. DOC concentrations generally decrease with soil depth, meaning that transport from soils and riparian zones typically increase when flow paths become more superficial, e.g., after CC (Siebert et al. 2009). Also, residue from forest harvest decompose and further increase DOC concentrations in the lateral water transport to ditches (Schelker et al. 2012). Additionally, mean absorbance was two times higher in CC compared to F sites for both measured wave lengths (420 and 436 nm), 1.05 and 0.51, and 0.82 and 0.40, respectively. This can be related to the increase seen in TOC.

Nutrients showed between two- and 12-times higher concentrations in CC sites compared to F sites. This is likely due to a combination of well-known factors, including 1) an increase in ground water table levels as water is no longer taken up by trees, leading to more superficial flow paths through soil strata with higher

nutrient content, 2) nutrients no longer being taken up by trees, leading to higher concentrations being mobilized from soils to surface waters, and 3) increased input of nutrients due to logging residues on the soil surface, which decomposes and releases nutrients for infiltration (Hyvönen et al 2000; Schelker et al. 2012; Eklöf et al. 2016).

The concentrations of THg were almost two times higher in CC compared to F sites ($p=0.0003$), with mean concentrations of 12.5 and 6.7 ng L⁻¹, respectively. This is in line with previous research, which has shown that forestry operations increase mobilization of Hg from soils to surface waters (Eklöf et al. 2016). Organic material has a great complexing capacity for metals and DOC/TOC and absorbance often correlates with an increase in Hg, which was also seen in this study (Hansen et al. 2013; Ukonmaanaho et al. 2016). No difference was seen in MeHg concentrations ($p=0.7919$) between CC and F or DC and R.

In contrast to the difference found between DC and R sites, CO₂-C concentrations were higher in CC compared to F sites ($p=0.038$), 3.6 and 2.7 mg L⁻¹, respectively. CO₂ is produced via degradation of organic matter and as CC resulted in higher TOC concentrations it is likely that elevated TOC can increase production of CO₂ (Cole and Prairie 2014).

In general, the observed concentration differences found for many variables between the CC and F sites are likely an effect of the altered hydrological pathways. CC typically raises the GWT thereby creating more superficial flow which in turn leads to mobilization of many solutes from more shallow soil strata (Koivusalo et al 2008; Eklöf et al. 2012; Schelker et al 2012). The causative pattern for DC versus R sites effect on water quality is thereby opposite the pattern for CC versus F sites. One might speculate that the older and more overgrown the ditch is, the larger will the effect of DC be, as an older ditch with decreased hydrological function likely has had a shallower GWT during a longer time.

4.3 Predicting water quality effects of ditch cleaning

Only two out of six variable-specific PLS models, ΔSO_4 and $\Delta\text{CO}_2\text{-C}$, demonstrated predictive relevance ($Q^2>0$) in explaining the observed differences seen between DC and R sites. However, the predictive power was still low for both ΔSO_4 and $\Delta\text{CO}_2\text{-C}$ (0.0652 and 0.0769, respectively) meaning that results should be taken with caution.

The proportion of deciduous forest on wetland explained most of the variability in ΔSO_4 concentrations (VIP=2.15) between DC and R sites. The correlation was positive suggesting that areas with a higher proportion of deciduous forest on wetlands corresponded with a rise in ΔSO_4 concentrations due to DC. Why this variable is of importance for the model is unclear, however, and further

investigation is needed. Additionally, MAP showed a negative correlation with ΔSO_4 (VIP=1.40). A short-term increase in water table levels in riparian zones are associated with high pulses in SO_4 concentrations (Ledesma et al. 2016). A higher overall precipitation and the following superficial groundwater table and flow paths, however, suggests a higher dilution factor as water flows through less S rich soil (Ukonmaanaho et al. 2014). Furthermore, it is unclear why some of the variables were of importance for the model, including the proportion of clay soil (VIP=1.71), MAT (VIP=1.39), %CC in the catchment (VIP=1.15), and %coniferous forest outside wetlands (VIP=1.02). These findings suggest that a higher proportion of deciduous forest on wetlands and/or clay soil within a catchment, along with decreased MAP and increased MAT, may lead to elevated SO_4 concentrations after DC. However, these variables only explained 32% of the variability in ΔSO_4 concentrations.

MAP was negatively related to $\Delta\text{CO}_2\text{-C}$ concentrations (VIP=1.82), indicating a dilution effect as the underlying explanation. In contrast, the proportion of peat in the catchment and the proportion of CC at the sample site correlated positively with $\Delta\text{CO}_2\text{-C}$ (VIP=1.79 and VIP=1.50, respectively). Both peat and CC contribute to increased OM, which subsequently could influence CO_2 levels. These results are supported by the findings of Wallin et al. (2010), who found peatland coverage to be the catchment characteristic which best explained spatial variability of CO_2 concentrations in streams, with the strongest positive correlations in headwater streams. Furthermore, MAT also exhibited a positive correlation with $\Delta\text{CO}_2\text{-C}$ concentrations. Higher temperatures are associated with enhanced production and decomposition processes in soils, resulting in greater carbon export into the water (Wallin et al. 2014). However, how this is somehow linked to deeper flow paths following DC is unclear. Moreover, both %coniferous forest on wetland (VIP=1.14) and %CC in the catchment (VIP=1.05) showed positive correlations with $\Delta\text{CO}_2\text{-C}$ concentrations but the reason for this correlation is unclear. These findings suggest that areas with high availability of OM, either due to CC or peatlands, coupled with low precipitation and high temperatures, are sensitive to enhanced export of CO_2 in runoff. However, these variables only explained 35 % of the variability in $\Delta\text{CO}_2\text{-C}$ concentrations.

The remaining PLS models with the aim to explain variability in ΔpH , ΔSUVA_{254} , $\Delta\text{CH}_4\text{-C}$, and $\Delta\text{N}_2\text{O-N}$, did not demonstrate predictive relevance. However, some catchment variables in these models still displayed interesting connections to the Δ variables, suggesting their potential use as indicators. For instance, the ΔpH model indicated a positive correlation with the percentage of deciduous forest (VIP=1.74). Coniferous forest are known to enhance acidity more than deciduous forest and thereby have a lowering effect on pH, but the connection to DC is unclear (Billett et al. 1988) However, the model did not indicate any relationship between coniferous forests and ΔpH . The negative correlation between

ΔpH and the percentage of deciduous forest on wetland (VIP=1.56) is also partly reasonable, as peatland connected headwater streams have been shown to be more acidic (Wallin et al. 2010). Moreover, indications were seen of a positive correlation between ΔSUVA_{254} and MAT (VIP=1.55), likely due to higher decomposition rates at higher temperatures, which alter the carbon composition in water. The percentage of open wetland indicated a negative (VIP=1.32) correlation with ΔSUVA_{254} , possibly due to low decomposition rates in anoxic environments. Furthermore, the percentage of coniferous forest on wetland indicated a positive correlation with $\Delta\text{CH}_4\text{-C}$ (VIP=2.18), likely due to the wetland aspect of this catchment variable, as anaerobic conditions are fundamental for CH_4 production (Wallin et al. 2014). MAP indicated a negative correlation with $\Delta\text{CH}_4\text{-C}$, possibly explained by increased soil moisture also leading to anoxic conditions.

Overall, the predictive relevance between land use, soil type and water quality parameters proved weak. Three variables, CO_2 , CH_4 and N_2O , were not sampled on all three occasions, which might have affected the power of the models. Also, it is important to note that GIS data on surface cover, soil type and land use are partially interpolated and that the small-scale variability within a headwater channel is high, especially regarding GHGs (Wallin et al. 2011). This emphasizes the need for more precise, high resolution geographical information for GIS to properly be used as a tool in prediction analyses.

5. Conclusions

The study revealed that DC does influence ditch water quality. Significant differences were seen between DC and R sites across six variables: pH, SO₄, SUVA₂₅₄, CO₂, CH₄, and N₂O. The higher pH in DC sites is consistent with previous research, suggesting that the lowering of the groundwater table during DC leads to deeper flow paths in soil layers with higher pH, resulting in elevated pH in runoff. The higher SO₄ concentrations in DC sites can be attributed to dominant flow paths through deeper soil layers containing higher SO₄ concentrations. While TOC did not differ significantly, the lower SUVA₂₅₄ in DC sites indicates that removal of ditch vegetation affects the quality of C in ditch water. The lower CO₂ concentrations in DC sites could be attributed to a shift in DIC proportions, with lower CO₂ at higher pH. DC sites exhibited lower CH₄ concentrations, likely due to higher water table levels and thereby anaerobic conditions in R sites. On the other hand, N₂O concentrations were higher in DC sites, but the reason behind this effect is still unclear. In addition to previous research findings, these results further add GHGs and carbon quality to the existing list of chemical variables affected by DC and the accompanied changes in groundwater flow paths.

The mechanism behind ditch water chemistry seem to be the same for DC and CC sites, but with opposite directions. It is fairly clear that the differences between DC and R sites can be attributed to a shift towards deeper GWTs in DC, while differences between CC and F sites can be attributed to a shift towards shallower GWTs in CC sites. The controlling factor is then, in both cases, the shift in hydrological pathways and higher OM content in shallower soil layers. As the only variable that differed in both treatment groups (CO₂-C) was lower in DC sites but higher in CC sites, DC activities could counteract some of the effects of forest harvesting.

It proved difficult to predict which catchment characteristics lead to water quality effects due to interpolated geographical data coupled with high small-scale variability within a headwater channel. Indications can still be made, however, and e.g. pH is likely to be elevated to a higher degree in deciduous forests.

As forestry ditches make up extensive headwater networks, relatively small measures such as DC can have large impacts on the overall water quality. This study further highlights the need for more extensive research on the effects of DC, as the results added new variables to the list of affected water quality parameters.

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Popular science summary

In order for forests to grow in a productive manner the groundwater should not be too shallow, as tree roots will not be able to grow deep enough. In Swedish forest, the natural state is often shallow groundwater, which is why ditches have been implemented since the 19th century. The quality of these ditches do, however, degrade over time due to sedimentation and vegetation overgrowth. Today, a common practice is to clean the ditches to restore the draining capacity and in that way maintain or increase the productivity of the forest stand. It has previously been shown that forest clear-cutting (CC) has a large impact on the ditch water quality, with increasing nutrient, sediment, and metal loads. However, how ditch cleaning (DC) impacts water quality, on its own and in relation to CC, is not well studied and therefore information on the topic is limited. Therefore, I investigated how water chemistry differs between DC and reference (R) sites, and if it is possible to predict which chemical variables that might be affected by DC by looking at catchment variables such as land use, soil type and vegetation.

I found that pH, sulphate, carbon quality, and greenhouse gases significantly differed between DC and R sites. pH, sulphate, and nitrous oxide concentrations were higher in DC sites whereas carbon quality, carbon dioxide, and methane concentrations were higher in R sites. These results are thought to mainly be attributed to the change towards a deeper groundwater table at DC sites.

DC and CC were found to affect different variables, except for carbon dioxide which was lower in CC sites compared to forested sites. The differences between ditch cleaned vs reference sites and clear-cut vs forested sites were also thought to mainly be attributed to changes in groundwater flow paths, which are more shallow after CC and thereby passes through organic rich layers in the soil.

Because geographical information is interpolated between sampling points and chemical variables can vary a lot spatially, predicting affected water chemistry variables from catchment characteristics proved difficult. Only two out of six models showed predictive relevance, sulphate and carbon dioxide, but the models only explained 32 and 35% of the variability, respectively.

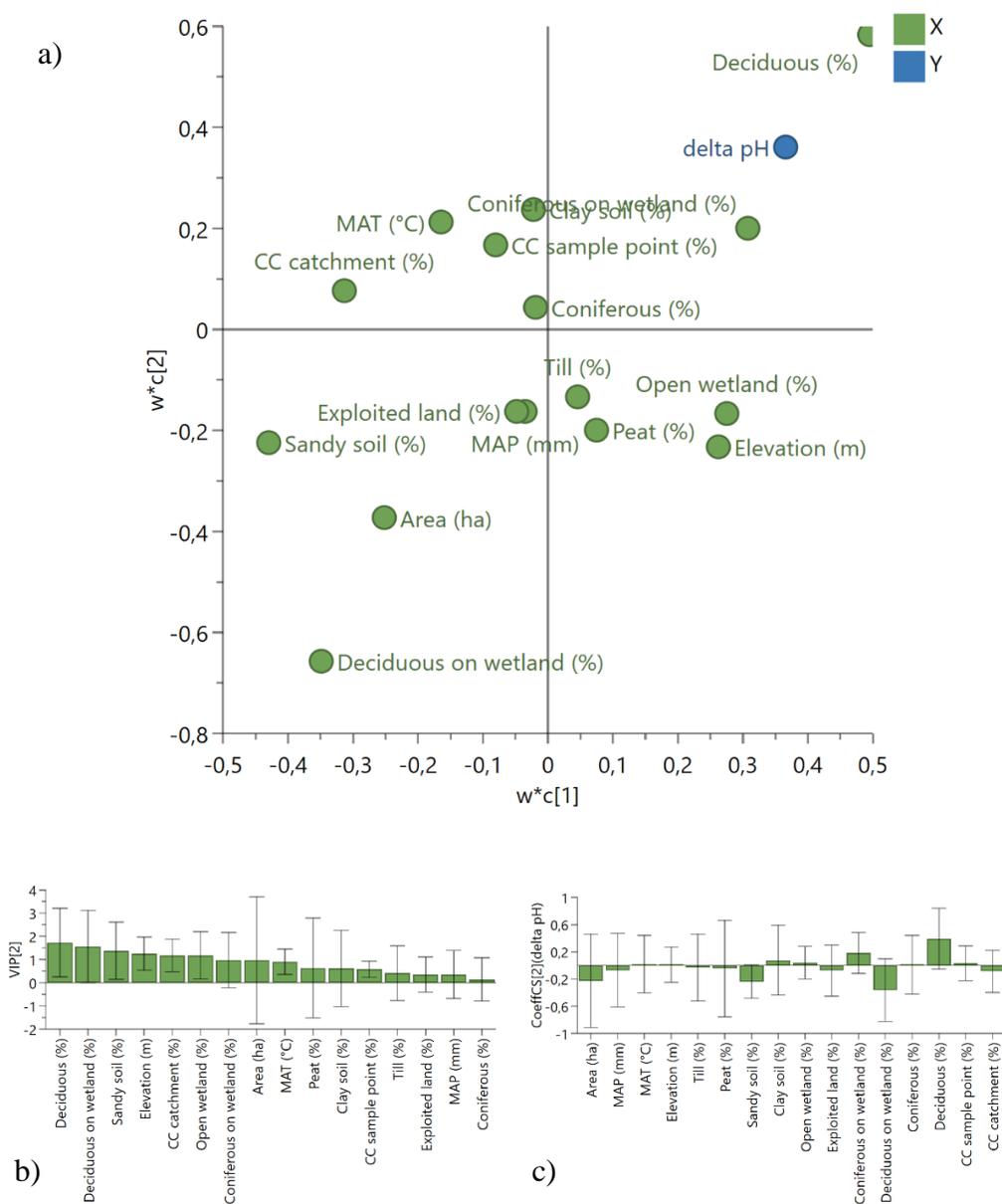
The results of this thesis further adds chemical variables to the existing list of water quality parameters potentially affected by DC, leading to the conclusion that more extensive research on this topic is still needed.

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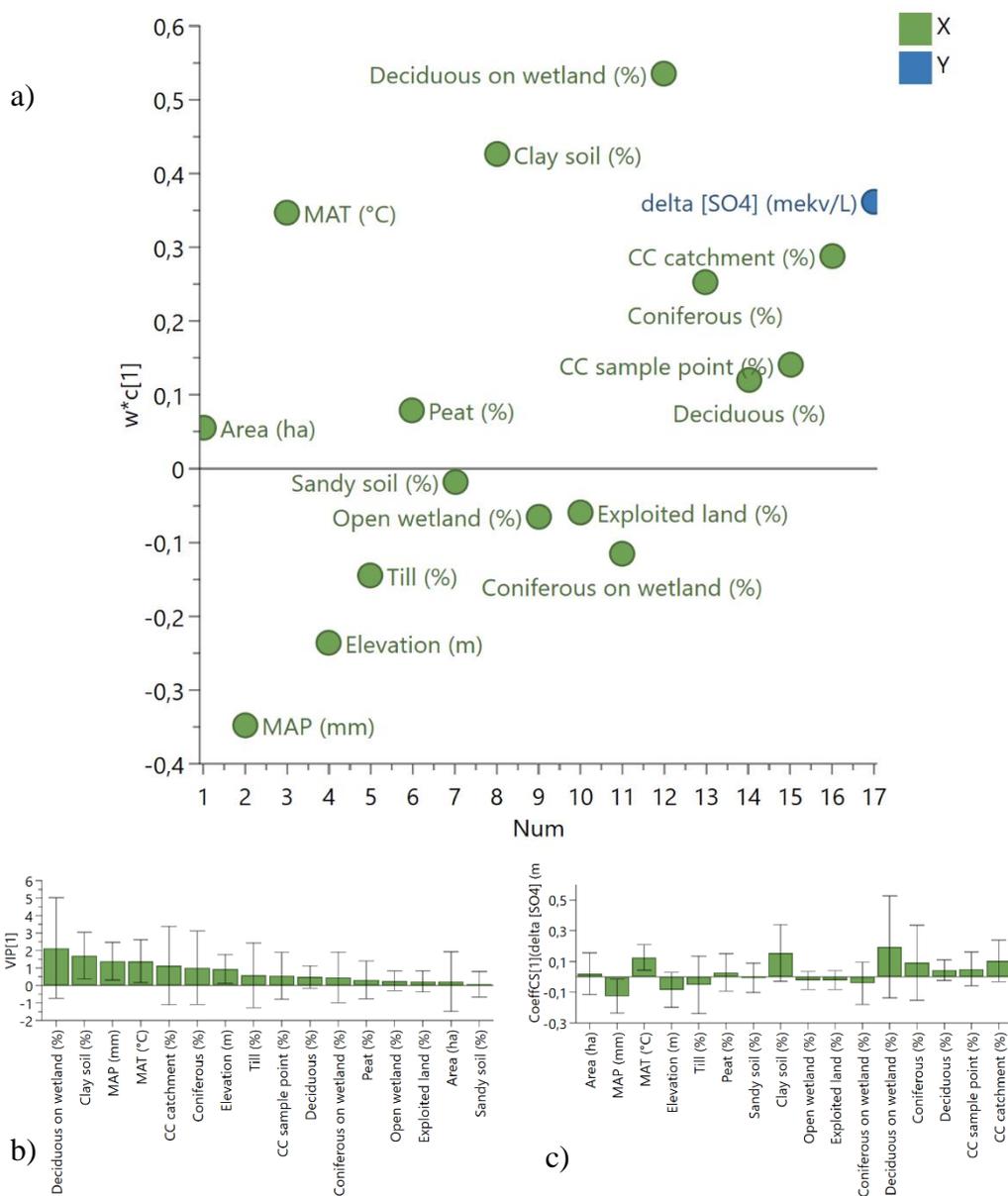
Appendix 1

Partial Least Square analysis for pH (R2Y=0.251, Q2=0.151) with a) scatter plot of the default loading vectors for the first two components, b) variable importance for the prediction (VIP), and c) regression coefficients.



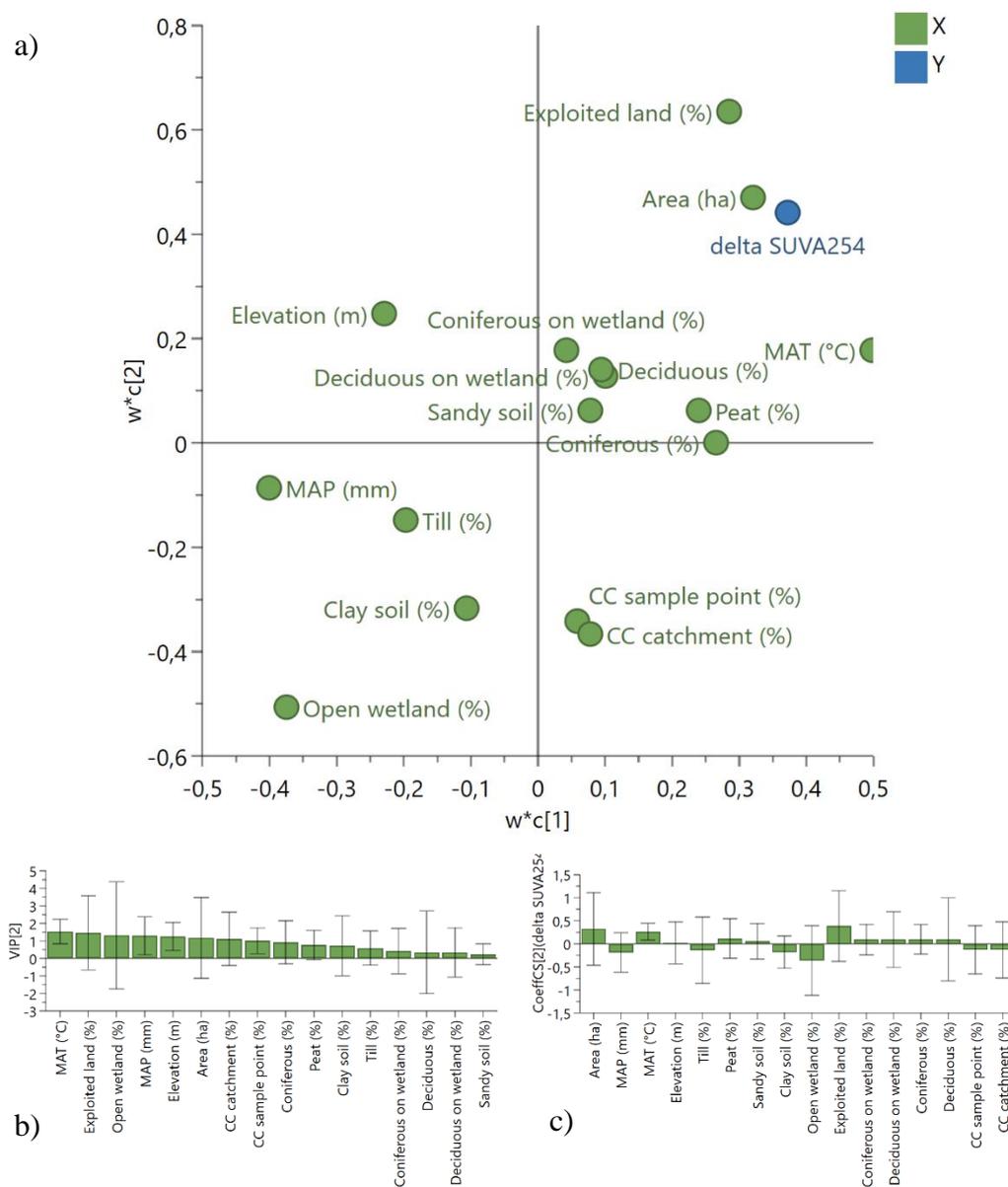
Appendix 2

Partial Least Square analysis for SO_4 ($R^2Y=0.251$, $Q^2=0.151$) with a) scatter plot of the default loading vectors for the first two components, b) variable importance for the prediction (VIP), and c) regression coefficients.



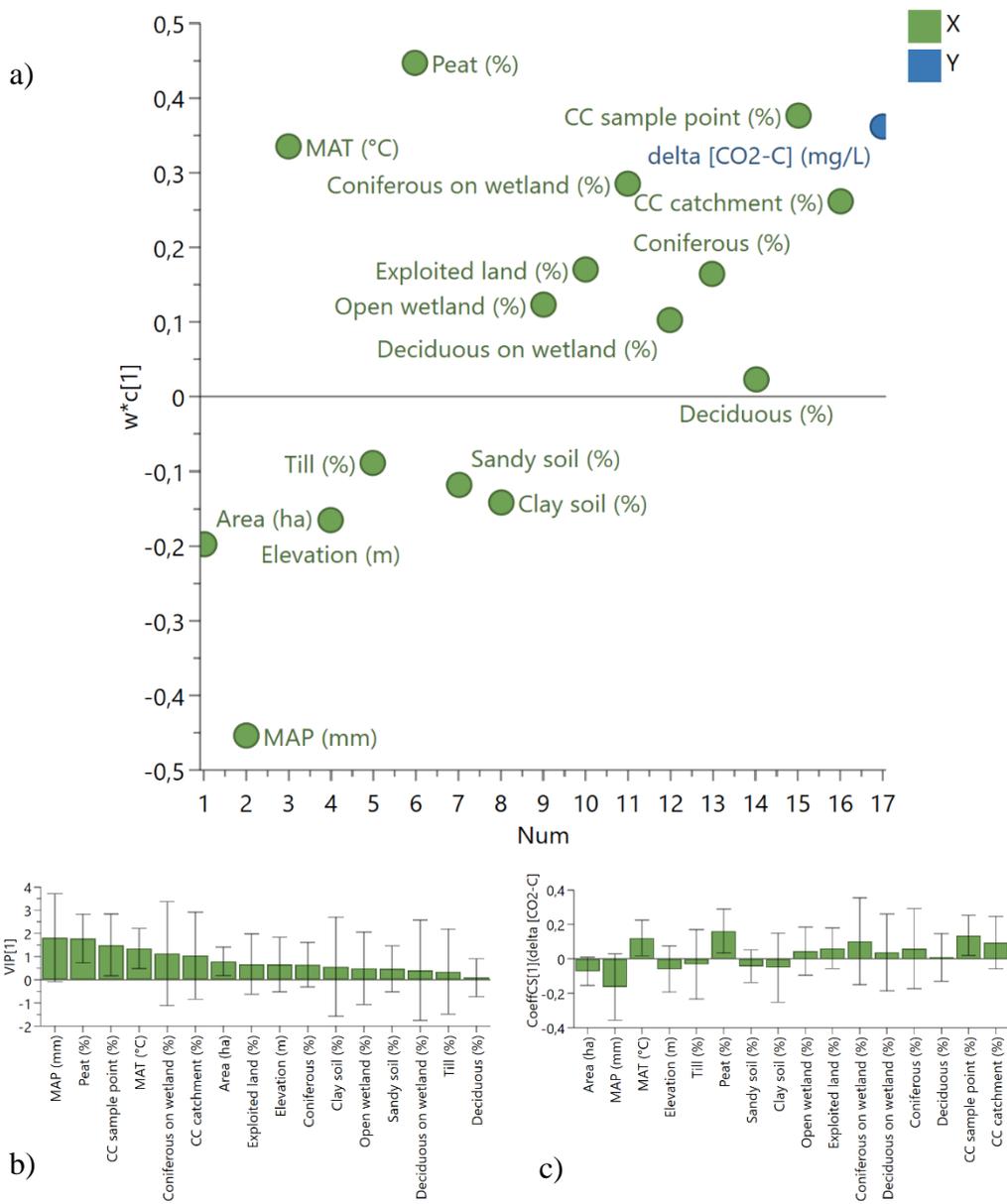
Appendix 3

Partial Least Square analysis for SUVA₂₅₄ (R²_Y=0.251, Q²=0.151) with a) scatter plot of the default loading vectors for the first two components, b) variable importance for the prediction (VIP), and c) regression coefficients.



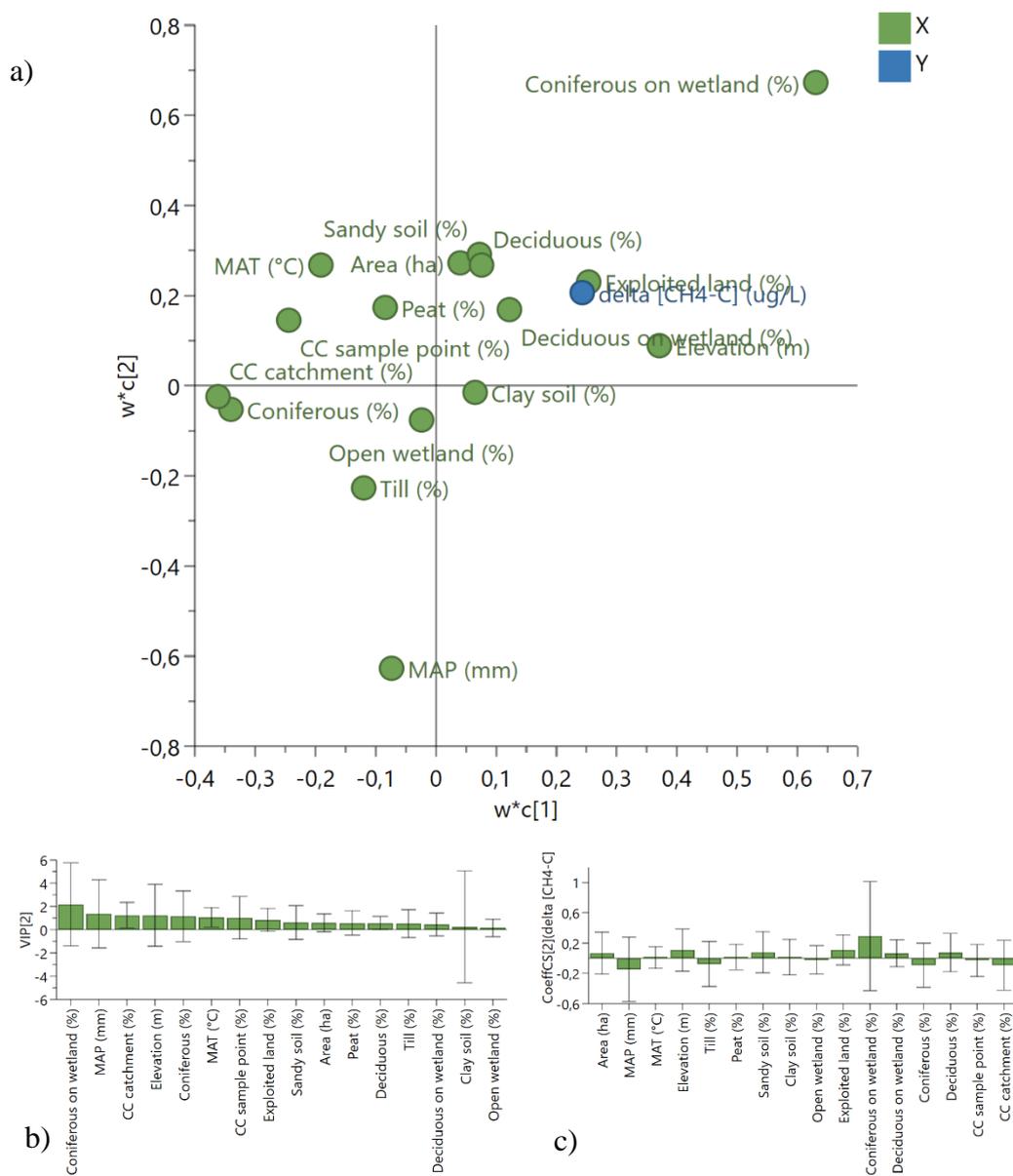
Appendix 4

Partial Least Square analysis for CO₂ (R²_Y=0.251, Q²=0.151) with a) scatter plot of the default loading vectors for the first two components, b) variable importance for the prediction (VIP), and c) regression coefficients.



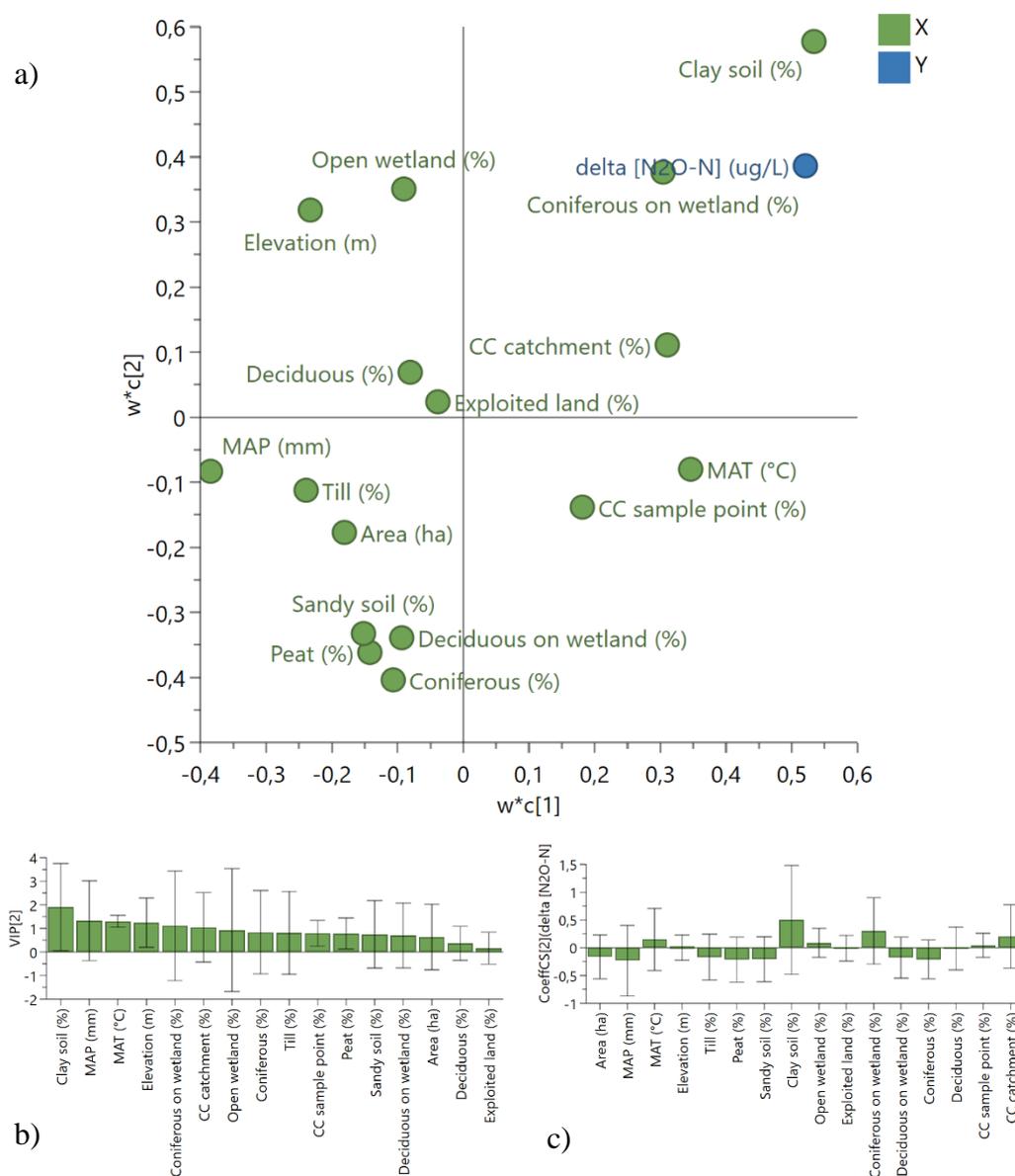
Appendix 5

Partial Least Square analysis for CH₄ (R²Y=0.251, Q²=0.151) with a) scatter plot of the default loading vectors for the first two components, b) variable importance for the prediction (VIP), and c) regression coefficients.



Appendix 6

Partial Least Square analysis for N₂O (R²_Y=0.251, Q²=0.151) with a) scatter plot of the default loading vectors for the first two components, b) variable importance for the prediction (VIP), and c) regression coefficients.



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