The role of agricultural drainage in controlling the effectiveness of two-stage ditches in Sweden

Neerajaa Nagarajan

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Neerajaa Nagarajan

Supervisor:	Magdalena Bieroza, SLU, Department of Soil and Environment
Assistant supervisor:	Lukas Hallberg, SLU, Department of Soil and Environment
Examiner:	Ingrid Wesström, SLU, Department of Soil and Environment

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Swedish University of Agricultural Sciences
Faculty of Natural Resources and Agricultural Sciences
Department of Soil and Environment

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Abstract

Increased agricultural activities are causing eutrophication in downstream water bodies. To mitigate nutrient leaching from cropland, various in-field and edge-of-field practices have been implemented. One such measure, the two-stage ditch (SD), with wider vegetated terraces than a traditional trapezoidal ditch (TD), is meant to slow down water flow, thus enabling sediments to settle and nutrients to be biogeochemically processed. The stability and nutrient removal of the SDs are being studied recently, but the influence of existing in-field drainage on the functionality of SDs, especially during high flows, has not been considered yet. It is especially important to learn the source and transport of nutrients and suspended sediments (SS) in the drains so that suitable in-field practices can be implemented.

This study included collecting grab samples from all the active subsurface drains and tributaries, lab analyses for concentration (mgl^{-1}) and loads (mgs^{-1}) of four: ammonia-nitrogen (NH_4N), nitrate-nitrogen (NO_3N), orthophosphate (PO_4P), total phosphorous (TP), SS and dissolved organic matter (DOM) indices. Contribution from individual drains and tributaries were estimated with an attempt to explain the source of the inputs by SS and DOM indices. The retention of the nutrients and SS in the SDs for each month and the influence of the drains and tributaries in terms of input % were also calculated.

The results showed that drains in clayey soil had loads (mgs^{-1}) accounting for 87% of PO_4P (SD1) and 78% of SS (SD3) of the total input to the SD. However, in terms of concentrations (mgl^{-1}), land use played an important role where sites with higher agricultural land use contributed with NO_3N concentration up to 300% higher than (SD8) than the SD.

DOM indices of drain water showed that the months following snowmelt between January and March had highly humified, aromatic compounds from terrestrial sources. The DOM stored nutrients and they were decomposable. This period saw a net nutrient/SS addition in SD downstream to an extent of -256% (SD3: January NO_3N) and -387% (SD5: March SS) in several SDs, potentially due to less vegetation, high flows, decreased denitrification rates, and erosion from the banks. April and May had a change in DOM in drains with more microbial-like, low MW compounds. While the months included cropping season with fertilization and its direct effects on the drain concentration, the removal rate of SDs was high in most cases, possibly due to active terrace biota.

The impact from drains and tributaries were detrimental in two situations: when high loads of nutrients and SS interfered in subsequent removal by the SD (SDs 1 and 3), when high concentrations were combined with unstable SDs that couldn't effectively reduce the inputs (SDs: 3, 5, and 10). To avoid further leaching to downstream ecosystems, future studies should consider techniques to reduce erosion by strengthening the terraces and reduce leaching from drains by effective in-field management.

Keywords: Subsurface drainage, Absorbance fluorescence spectroscopy, Suspended sediments, NO_3N, NH_4N, PO_4P, TP values, DOM indices

Popular scientific summary

The Baltic Sea is undergoing eutrophication by multiple sources, agriculture being one of them. The net result of this nutrient loading has led to decreased stability in the aquatic life cycle and the communities dependent on it. The European government has passed many initiatives to deal with this problem, notably the Water framework directive, which directly targets nutrients such as nitrogen (N) and phosphorus (P), that is being leached from the agricultural fields.

This study focused on one of the ways N and P can leach from a field and end up in the rivers and seas: artificial drainage. The study included measurement of N (Nitrate NO_3N and Ammonia NH_4N), P (orthophosphate PO4P and total phosphorus TP), suspended sediments (SS) and dissolved organic matter (DOM) indices. Out of these, SS and DOM can store and transport the nutrients and thus, gave information about the source and transport of the nutrients through the drains. The study was done between January and May. The two-stage ditch (SD) is an upcoming idea to be constructed in agricultural fields as an outlet for the drains and tributaries, in addition to transporting the water downstream. An SD would contain a wide, vegetated terrace, which would slow down the water, giving enough time for the SS to settle down or the nutrients to be removed by plants and microbes. This study also tried to evaluate if the SDs can achieve this by calculating the amount of NO_3N, NH_4N, PO_4P, and TP, which is reduced by 7 SDs in central and southern Sweden.

The results showed that SDs 1 and 3 had the highest influence on the functioning of an SD with 87% of PO_4P in SD1 and 78% of SS in SD3 inputs coming from the drains and tributaries. When the loads (mgs^{-1}) were less, the drains and tributaries can still interfere in the functioning of an SD with high concentrations (mgl^{-1}). For example, SD 8 had a 300% difference between concentrations of drains and the SD. This could put high pressure on the functioning of the SD. When the nutrient and SS reduction was calculated, SDs 3, 5, and 10 had net negative removal with an increase of -256% (SD3: January NO_3N) and -387% (SD5: March SS).

From the DOM, it was seen that January leached highly complex (humic) compounds which have been accumulating in the field over winter and leaching out nutrients in the form of snowmelt. April and May had comparatively smaller compounds, that could be easily decomposed to release nutrients. The smaller compounds are formed because of cropping in the neighbouring fields.

The study showed that the drains and tributaries can be an interference to functioning of the SD if they contain high concentrations of the nutrients and SS or if they bring in high loads themselves. The ways to avoid the high leaching must be done on the field, by either decreasing the quantity of fertilizers used or by reducing harmful management practices, such as tillage, which would cause more DOM and SS to be leached.

Table of contents

List of tables

List of figures

Abbreviations

BIX	Freshness index
DOM	Dissolved organic matter
DON	Dissolved organic nitrogen
EEM	Excitation emission
FI	Fluorescence index
FNU	Formazin nephelometric unit
HIX	Humification index
MW	Molecular weight
N	Nitrogen
NH_4N	Ammonia nitrogen
NO_3N	Nitrate nitrogen
NTU	Nephelometric turbidity unit
P	Phosphorous
PO_4P	Orthophosphate/ dissolved reactive phosphorous
SD	Two-stage ditch
SDup	Upstream (start) of an SD
SDdn	Downstream (end) of an SD
SOC	Soil organic carbon
S_R	Spectral slope
SS	Suspended sediments
TD	Traditional trapezoidal ditch
TP	Total phosphorus
TSS	Total suspended sediments

1. Introduction

The onset of water pollution is generally linked to the rapid population growth following the industrial revolution in the mid-19th century (Markham 2019). There were various concerns that the direct effects of fossil fuels, growth in food production, and intensive manufacturing were seen in the degradation of the air and water quality. However, there was no action taken until a century later in the US. The Clean Water Act of 1972 (US EPA 2013) was the first piece of official legislation that came as a victory to the decade-long battle initiated by an environmental movement. Since then, most countries have come up with their own environmental laws and amendments notably the EU's Water Framework Directive (2000/60/EC) and the Chinese Water Law of 2002 (Ravesteijn *et al*, 2009).

The continuous monitoring and research of the world's water bodies since the above legislation, have led to a much better understanding of the sources of pollution. On one hand, urbanization and rapid growth of the manufacturing sector have caused a multitude of problems such as domestic and industrial effluent discharges, smokestacks-borne particulate matter, and excessive freshwater abstraction (Likens et al. 1979; Wang et al. 2008; Talabi & Kayode 2019). However, the remediation for these point sources could be implemented locally and on a smaller scale. The challenge lies in the activities where there are no single source of pollution and thus, no simple solution for a safe countermeasure. One of the toughest issues regarding the water pollution crisis has been found in the farming sector (Evans et al. 2019).

Agriculture has been experiencing a production boom since the introduction of mechanical tools, fertilizers and improved crop breeding in mid 1900s (Passioura 2002). Combined with the industrial revolution, technological advancements in terms of health and sanitation, improvement in people's standard of living, the world has seen an increase in population and food demand (Smil 1999; Steckel & White 2012). As a result, land use has changed dramatically in the world and led to an intensification of agriculture, both in terms of farming area and methods. For example, Davidson (2014) showed that about 56.3% of wetlands in Europe have been drained since 1900. Wetlands have been known to have fertile soils and by laying effective drainage systems (open and subsurface), the land becomes highly productive (Verhoeven & Setter 2010). The same principle could then be applied to areas of high seasonal precipitation to unlock even more potential farming sites (Valipour 2014).

The positive and negative effects of drainage have been extensively studied. While drainage pipes/ditches improve water infiltration, they have been also responsible for increased nutrient leaching from soils to the downstream water bodies (Janse & Van Puijenbroek 1998). Although in-field drainage systems have become crucial in a commercial agriculture, the focus has now turned to conservation ideas. For reducing nutrient leaching, both ‘in-field’ and ‘edge-of-field’ practices have been established (Christianson 2018). For example, cover crops and non-inversion tillage or even limed tile drains are some in-field practices (Cooper et al. 2017).

Edge-of-field practices are applied on the waterways that run along multiple fields. Some measures include field border, filter strips, or riparian forest buffers (Dabney et al. 2006). The most effective form of remedy would be to combine the two types of measures as the approach converts farming from being a big, non-point source of nutrients to multiple steps of quantifiable point sources.

The two-stage ditch (SD) is an edge-of-field remediation method for improving water quality. It is a modification of a traditional ditch (TD) where the in-field drainage ends. An SD has wider vegetated terraces (floodplain) compared to the TD. The idea is to give enough retention time for the water flow to slow down, causing the sediments to settle down on the terrace and the nutrients to be removed (Davis et al. 2015). For nitrogen, it occurs by an accumulation of organic matter leading to enhanced denitrification rates, flooding of water on the terraces causing a rise in absorption by plants and other organisms. For phosphorous, it occurs by settling of sediment-bound P, leading to an increased availability for plants and other organisms (Powell & Bouchard 2010).

1.1. Aim

The main aim of this study is to understand the role of the drainage systems in controlling the water quality and functioning of the two-stage ditches.

The study took place in 10 catchments spread around central and southern Sweden, varying in climate, soil type, land use, and crop management practices. To understand the role of drainage systems on the functioning of the SDs, the following questions were evaluated:

- What is the role of drainage inputs on nutrient (N and P), sediment, and organic matter concentrations and loads in studied SDs?

- Can the concentrations of suspended sediments (SS) or the character of dissolved organic matter (DOM) provide any information on the sources of pollutants?
- Are there any spatial or temporal variations between the studied sites, and if so, can they be explained by observed patterns in SS or DOM?

2. Literature Review

2.1. In-field drainage systems

The concept of agricultural drainage is very old. Valipour *et al, 2020* investigated the evolution of agricultural drainage and stated that the earliest evidence of artificial irrigation and drainage was found in Iran dating around 4000 BC. The design underwent various developments over time in different parts of the world based on purpose, geography, and level of technological expertise. In Europe, subsurface tile drainage was introduced in Maubeuge, France in 1620 (Valipour et al. 2020). As of 2019, about 200 Mha in the world was artificially drained and a further 450 Mha was expected to benefit from improved drainage (Castellano et al. 2019). Agricultural drainage has a multitude of benefits ranging from prevention of waterlogging, improved infiltration, improvement in crop yield, field trafficability, reduction in nitrous oxide emissions, and control of soil salinization (Sims et al. 1998; Castellano et al. 2019). While drainage is also linked to losses of soil organic carbon (SOC) and increased nitrogen and phosphorous leaching, it can be ameliorated by better management practices (Castellano et al. 2019).

Drainage in Sweden has existed since the 1500s. It became prominent at the end of the 1800s when the lake Hjälmaren was lowered by 1.3 m creating 15000 ha of cultivable land (Jacks 2019). While open ditches were preferable in the initial days, subsurface tiles are gathering more popularity now. Between 1927- 2016, the land drained by tile drainage increased from 910 960 ha to 1 221 680 ha. That is about 47% of the total arable land (Grönvall 2017). According to a survey by the Swedish Board of Agriculture, 12% of the Swedish arable land needs new tile drainage and a further 12% needs refurbished tiles (Grönvall 2017).

2.1.1. Drainage design

Artificial drainage in each field is designed based on specific conditions to ensure the best results in terms of productivity and the least impact on the surrounding environment (Gramlich et al. 2018). Most countries have laws regarding drainage as it involves a transfer of water from the owner’s land to a common area, such as streams or rivers. Once the permit is obtained, the most common factors to be considered before installation are the climate (precipitation and evapotranspiration), field topography, water table depth, soil properties, and the planned farming activity (crops and growing season). The drainage capacity (mm/day) is then fixed based on how moist the soil is required to be and the drain design parameters, including the subsurface drain material, size, depth, and spacing are finalized (Schiechtl 1985). It is interesting to note that one or more of the installation factors can have contradictory effects on the drain parameters and hence, the decision is made to achieve a suitable balance. For example, while shallower drains might drain excessively causing drought in the crop root zone, one study found that deeper drains proved to be ineffective in clayey soil since a compacted top layer blocked the water from being drained (Harris et al. 1984). Here, the decision for drain depth and spacing would be made based on where the water table lies.

Figure 1 Drain design at the outlet from SD3

A typical design at the outlet is shown in Figure 1. The drain exits from the field, surrounded by bedding material, typically made of gravel, which slows down the flow and prevents erosion of the terrace. The drain lengths are adjusted so that water flows directly on a grassed strip (here, vegetated terrace of the SD), which is expected to reduce some nutrient and pesticide concentrations before they enter the SD (Haddaway et al. 2016).

From the standpoint of water quality in the recipient streams, drains are an important connection to the field and understanding the drain water composition can also be a good way to understand the effects of in-field management practices. For a fully installed drainage system, the parameters that are directly linked to nutrient transport include the discharge, SS, and dissolved organic matter (DOM) found in the water samples, and their roles will be discussed in the following chapters.

2.2. Composition of the drainage water

2.2.1. Nutrients and discharge

Drainage improves water infiltration from the field leading to a higher total annual discharge than undrained soils (Gramlich et al. 2018). An increase in subsurface drainage, however, affects the nutrients differently. A study conducted on clay soils showed that an increase in subsurface drainage discharge decreased the amount of orthophosphate and total phosphorous through sediment retention but increased the amount of nitrate leached as dissolved fraction (Turtola & Paajanen 1995). The amount of nitrogen leached was especially higher in dry years during precipitation, which showed that soil nitrate levels could accumulate from soil mineralization or fertilization and flush out at once (Randall & Mulla 2001). The amount of phosphorous leached varied with respect to different forms. The major phosphorous compounds in drain water broadly fell under 6 categories including dissolved (filtered) and undissolved (unfiltered) forms of reactive, unreactive, and total P (Haygarth et al. 1998). One study found that an increase in subsurface discharge led to an increase in dissolved phosphorous but a reduction in total phosphorous (TP) levels (Sharpley & Syers 1979). Particulate phosphorous, a part of the undissolved fraction, was adsorbed to or embedded within sediments and as there was a reduction in sediment movement through the soil profile, it led to reduced P leaching (Hansen et al. 2002). However, the level of TP transported through open drains/runoff generally increases with an increase in discharge (Algoazany et al. 2007).

2.2.2. Nutrients and suspended sediments

The drain water conveys nutrients in two forms: particulate and dissolved fractions (Ritzema et al. 1996). SS are an important part of water quality analysis as a medium of transport for adsorbed nutrients and harmful toxins. SS can include flocculation of microbes, organic and inorganic particles (Droppo 2001). In addition to the potential of pollution, the loss of sediments may also affect the flow of the receiving stream over a period of time (Schwab et al. 1980). In large surface waterways, regular maintenance by sediment dredging is needed to ensure continuous water flow, leading to high costs (Powell et al. 2007). In order to take effective control measures for sediment loss, quantification and source tracking are important steps to be considered (Rügner et al. 2013).

SS can be measured from water samples by many methods including filtration, centrifuge, and electron microscopy (Schwab et al. 1980; Ball Coelho et al. 2012). One of the convenient and inexpensive ways to calculate it on-site or periodically is by measuring turbidity instead (Gippel 1995). Turbidity and SS have been found to be linearly correlated to each other (Packman et al. 1999; Skarbøvik & Roseth 2015; Villa et al. 2019).

Open drains (tributaries) usually contain higher sediment concentrations in comparison to subsurface drains due to erosion of the ditch's sides through quick overland runoff (Blann et al. 2009). But subsurface drains also have a potential for high sediment delivery, especially during high flows by means of macropores in the soil and regions with high clay content, by means of clay shrinkage pores (Chapman et al. 2005). The net effect of increased sediment loss in drains is usually associated with an increased total phosphorous (TP) (Lannergård et al. 2019) and ammonium transport (Wang et al. 2010). This increase is expected, given the highly adsorptive property of phosphates (Agudelo et al. 2011) and ammonium (NH_4) ions (Ghane et al. 2016).

2.2.3. Dissolved Organic Matter

The Dissolved organic matter (DOM) fraction is very versatile, and often responsible for leaching highly reactive and bioavailable nutrients to the downstream water bodies (Heinz et al. 2015). The source of DOM can be from existing soil organic matter, release from sediments, exudation from macrophytes, or microbes such as algal cells and bacteria (Stedmon & Markager 2005). Depending on the type of organic matter, it can further degrade to release the bound compounds or store valuable plant nutrients in recalcitrant form for a long time (Carlson & Hansell 2015).

DOM is generally classified as protein-like or humic-like compounds (Hudson et al. 2007). Protein-like compounds include labile, small-sized amino acids such as tryptophan, tyrosine, and phenylalanine, which are considered to be freshly-produced, of microbial origin, or at least in easily bioavailable form (Cammack et al. 2004). That makes this type of DOM easily decomposable to release nutrients. Humic-like compounds include big, complex organic molecules that were produced by oxidation of carbohydrates, lignin, proteins of dead organisms, and exudates from living organisms (Elkins & Nelson 2001). Humic substances are less biodegradable than protein-like compounds but in aquatic systems, they can be gradually degraded by light or microbes (Hutchins et al. 2017). They are also less prone to leaching as they are hydrophobic (Cleveland et al. 2004). By a process called humification, the protein-like compounds can become converted to humic-like compounds (Schnitzer & Monreal 2011).

Types of DOM have been distinctly analysed and studied using absorption emission spectrometry for many years (Coble et al. 2014). DOM contains compounds called chromophores, that absorb light and fluorophores that absorb and re-emit light. Depending on the specific wavelength of the light, that is absorbed and emitted, it is possible to identify different carbon fractions (Hudson et al. 2007). For example, compounds containing benzene rings have a UV absorbance peak at 254 nm, making this wavelength an indirect measure of aromaticity (Weishaar et al. 2003; Holc et al. 2018). In terms of absorbance, humic-like compounds have absorbance peaks at a longer wavelength than their protein-like counterparts, probably due to an increase in molecular weight (MW) and aromaticity (Baker & Inverarity 2004). Surprisingly, Helms et al. (2008) also saw a decrease in low MW compounds in terms of microbial degradation, which could mean that microbes prefer to degrade smaller aromatics first. This study focused on three absorption ratios: Spectral slope (S_R), E2:E3, A254 and three fluorescence indices: Fluorescence index (FI), Freshness index (BIX) and Humification index (HIX), whose values and brief interpretations are summarised in Table 4.

2.2.4. Nutrients and Dissolved Organic Matter

Different combinations of absorption and fluorescence indices indicate different origins of DOM, which are useful in understanding nutrient interactions. Studies have shown that a high A254 ratio (Castan et al. 2020) and a high HIX (Hudson et al. 2007) are an indication of increased adsorption sites. As phosphates can adsorb to organic surfaces and make up particulate phosphorous (PP) (Hansen et al. 2002), PP may correlate to samples with increased aromatic compounds. Dissolved reactive phosphorous (PO_4P) is usually found to be associated with more protein-like compounds (high FI) with low aromaticity (Coble et al. 2016). A similar trend is also found in the nitrogen cycle. A study done in a forested stream found that DOM of terrestrial origin (low FI) and high A254 led to increased transport of ammonia (Coble et al. 2016). Nitrate, on the other hand, is correlated more with DOM of microbial origin (high FI) and high A254 (Tiefenbacher et al. 2020), because these labile carbon compounds are rapidly mineralized by microorganisms. Additionally, one study found that dissolved organic nitrogen (DON) is released from soil during fertilization (Kalbitz & Geyer 2002) which constitutes an additional source from agricultural fields.

2.3. Spatial and temporal variation

Spatial variation in nutrient transport can be attributed to various factors such as catchment properties (size, geomorphology, intensity of land use), physical conditions (climate, soil type, runoff) or management conditions (proportion of arable land, fertilization, regional laws; Gelbrecht et al., 2005). It is difficult to include all the factors, so this study focused mainly on the influence of soil properties, % of agriculture, and drain type on the turbidity, DOM, and nutrients of drain water (Table 2, Table 6).

On a general basis, SS in runoff water correlates with both turbidity and clay % of soil (Udeigwe et al. 2007). Higher agricultural land use increased the release of nitrogen and phosphorous in comparison to forests (Correll et al. 1992).

DOM indices show the temporal differences that can be linked to a change from a period of low to high activity in terms of living organisms. HIX is generally low (terrestrial sources) during snowmelt and the value increases (microbial sources) in summer (Miller et al. 2009). E2:E3 ratio, which relates to molecular size, is expected to increase from winter to summer (Macdonald & Minor 2013). A254 is high around snowmelt and is expected to start decreasing towards summer as more labile compounds are produced and high decomposition begins (Miller et al. 2009). With respect to turbidity and nutrients in the drains, generally the concentrations follow the discharge values (high during snowmelt and rainfall events) but a detailed study has found individual hysteresis curves specific to each nutrient and turbidity (Ulen 1995). Additionally, nutrient concentrations in individual sites can go up rapidly during storm events following long periods of drought, especially when the field soil has a high stored nutrient level from past management practices (Bieroza et al. 2019). Nutrient concentrations can also go up when the neighbouring field is fertilized (Di & Cameron 2002).

2.4. Two-stage ditches

The two-stage ditch was a part of the best management practices (BMP) to replace a traditional ditch that already existed (Figure 2). One of the oldest designs for the two-stage ditches was first introduced in the US Midwest in around 2003 (Ward et al. 2004). A part of the existing riparian zone, which included a grass buffer strip was dug out to make a vegetated channel bed (Mahl et al. 2015). During high flows, it was expected to spread out the water to a bigger area, slow down the water flow, reduce the shear stress on the bank sides and thus reduce erosion (Powell et al. 2007).

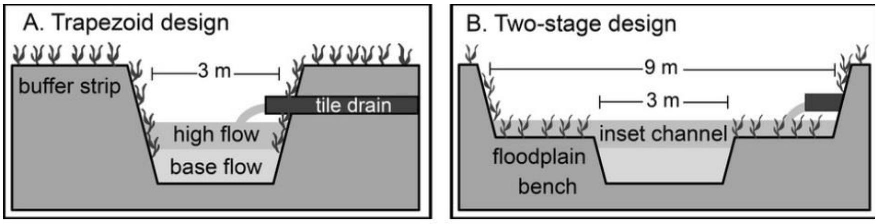


Figure 2 Difference between a TD and an SD (Mahl et al. 2015)

The advantages of the SD can be summarized as following (Table 1)

Table 1 Design changes made in the SDs and its improvements in terms of maintenance, stability and water quality

Changes made to TD	Expected improvement
Wider ditch	Increased water-holding capacity, less flooding during high flows (Hodaj et al. 2017)
Gradual sloping- introduction of terraces	More stability, less erosion and prevention of bank collapse (D'Ambrosio et al. 2015)
Providing by-pass for tile drains on the vegetated terraces	Adsorption of N and P by DOM, SS Intake of nutrients by plants and microbes Settling of sediments by vegetation (Davis et al. 2015; Mahl et al. 2015)
Baseflow during dry periods and high flow during wet periods	Self-cleaning: Redistribution of sediments during the high flow, less accumulation in the furrow, lesser need to dredge sediments (A.D. Ward et al. 2004; Powell et al. 2007)

One study found that some of the biggest factors that could affect nutrient removal in an SD include a change in oxygen concentration, organic matter content, residence time, and discharge (Hodaj et al. 2017). The concentration of nutrients and turbidity in the drains and tributaries comparing with the SD could also influence the removal by an SD, especially during a low-flow period, when the terrace is not flooded and the pollutants could escape as base flow (Mahl et al. 2015). Agricultural drains can make changes to organic matter composition, daily nutrient, and sediment input in a way that it can burden the SD at peak flow or dry periods.

3. Materials and Methods

3.1. Site Description

The study was conducted in constructed two-stage ditches in ten agricultural catchments of central and southern Sweden as shown in Figure 3. The regions varied in soil type, land use, climate, and the age and design of the SDs (Table 2). SDs 1-5 had a clayey silt soil, which changed to silty sand in SDs 6-9 and predominantly sandy soil in SD10. Average annual precipitation was around 600mm in most sites but SD10, which is situated in south-west coast, had a higher precipitation of 853mm. While SDs 3, 6, 7, 8, and 9 had intense agriculture in the catchment level, all the SDs were located in agricultural fields. The design of SDs were with terraces that was one-sided (SD 4), two-sided (SDs 1, 2, 8) or mixed (5, 6, 7, and 10).

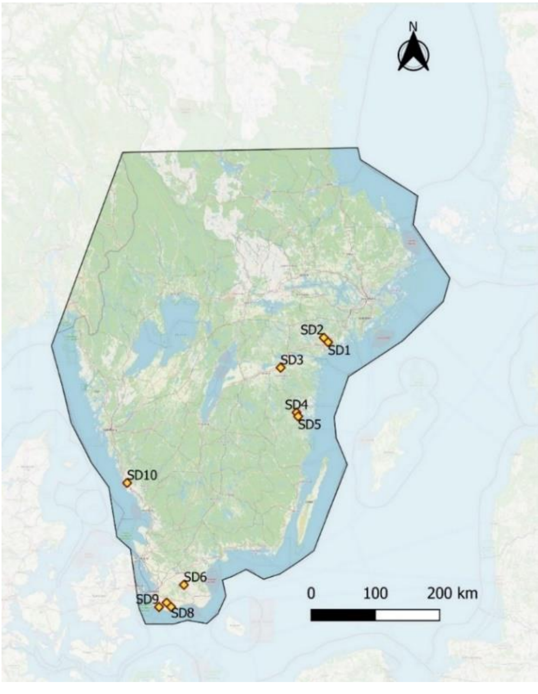


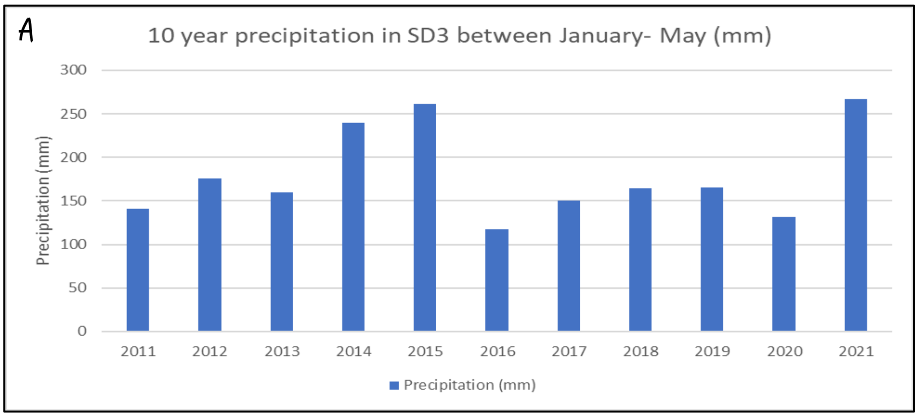
Figure 3 Ten catchments SD1 to 10 that was studied in this project

Out of the ten sites, samples were not taken in three, where the subsurface tiles were submerged under the stream (SD2, SD4 and SD6) and not included in this study. SDs 1, 5, 7, and 9 had tributaries joining the SD. All the other drains were subsurface drains of varying diameters (between 10 and 48 cm). They were made of brick tile, concrete, or PVC. All the drains were conducting water by gravity and there was no pumping system in any of the visited sites.

*Table 2 Information on catchment properties, including length and median flow of each SD and sampling months. The SDs are ordered from North to South. Stream flow are from measured values between 2020 and 2021. Annual Precipitation was from 1990-2020. * (Source: https://www.smhi.se/data/meteorologi/nederbord/)*

SD	**Annual precipitation (mm)***	**Soil texture (%)**			**Land Use (%)**		**SD length (m)**	**SD Q50 (m^3s^{-1})**	**Sampling Months**
		Clay	**Silt**	**Sand**	**Agri.**	**Others**			
SD1	597	43	43	14	16	84	340	0.025	Feb, Mar, Apr, May
SD2	597	34	48	18	27	73	730	0.032	None
SD3	577	40	40	20	70	30	1500	0.011	Jan, Mar, Apr, May
SD4	628	36	37	27	35	65	350	-	None
SD5	628	29	40	31	38	62	750	0.023	Mar, Apr, May
SD6	698	18	41	41	84	16	400	0.077	None
SD7	691	23	38	39	81	19	750	-	Mar, May
SD8	691	19	35	46	81	19	890	-	Mar
SD9	597	18	32	50	86	14	630	0.079	Mar, May
SD10	853	8	19	73	58	42	1760	-	Mar, May

The sampling period in 2021 included a snowmelt (January) and a large rainfall event (May). To quantify the magnitude, precipitation for SD3 was plotted (Figure 4*B*). The other sites, while differing in exact quantity, had similar flow pattern. In addition, the long-term precipitation for the season between January and May was plotted to check if the year of study, 2021 was representative in terms of wetness.



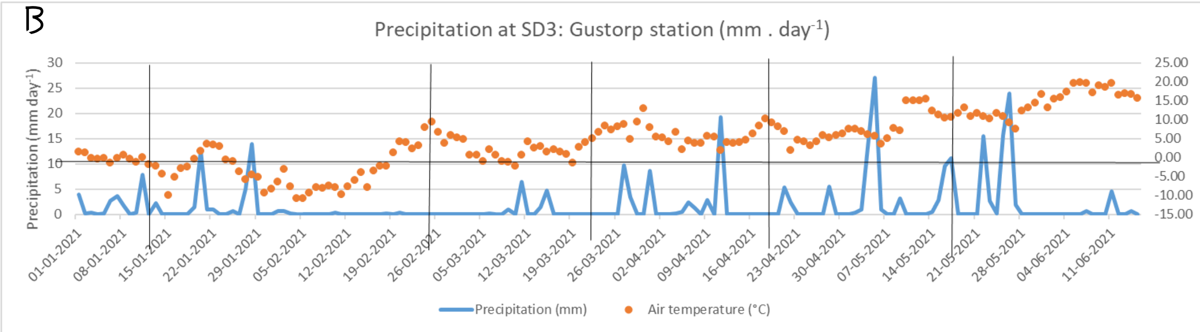


Figure 4A) 10-year seasonal precipitation (between January and May) in SD3 (Nearest weather station Gustorp) B) Precipitation and air temperature between January and June 2021. The sampling dates are marked with a line. A precipitation with negative temperature meant snow fall and intact snow cover. A precipitation with a positive temperature would have led to high flow in the SDs. 0°C line is drawn from the secondary axis (Source: https://www.smhi.se/data/meteorologi/nederbord/)

3.2. Sampling and field measurements

The fieldwork was carried out in specific SDs between January and May 2021 (Table 2) and the grab samples were collected and analysed for NH_4N, NO_3N, PO_4P, TP, turbidity, and absorbance fluorescence. Flow (ls^{-1}) for the drains was calculated by taking the time taken to fill a 250ml bottle. For one drain SD3_D23.5, which was too big and had a very high flow (<1s to fill the bottle), the depth of water level, the diameter of the drain, and velocity of flow (ms^{-1}) were measured, and the flow was calculated using the AUTOCAD web app (Figure 5).

$$Discharge(m^3s^{-1}) = Velocity\ (ms^{-1}) * Cross\ sectional\ area\ (m^2)$$

$$Cross\ sectional\ area(m^2) = \frac{1}{8} * (\theta - sin\theta) * Diameter\ of\ the\ pipe^2(m)$$

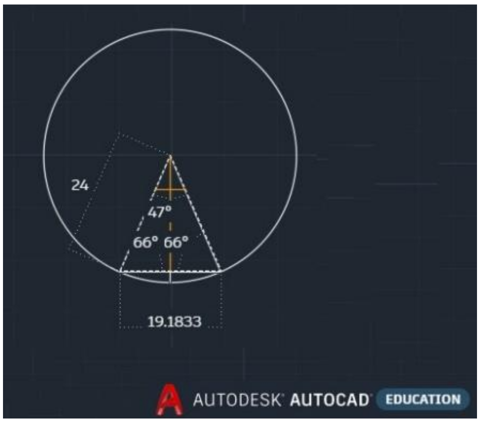


Figure 5 Calculation of angle for measured water depth (Source: AutoCAD web app)

Flow in all the SDs and tributaries was measured using a portable acoustic doppler velocimeter (Flowtracker 2, Sontek). The turbidity of the SDs (in FNU) and tributaries were measured using a multiparameter handheld sensor (ProDSS, YSI). The SDs concentration and flow values were measured at the upstream (SDup) and downstream points (SDdn) on the sampling dates, which were taken as the inlet and outlet readings for the mass balance. The tributaries were sampled just before the intersection with the main SD, to account for complete input to SD.

3.3. Laboratory measurements

The concentrations of nitrate-nitrogen (NO_3N) and nitrite-nitrogen (NO_2N) were measured together by ISO 1996, but this study took the whole value as NO_3N as the amount of nitrite present was negligible. The concentration of ammonia-nitrogen NH_4N (ISO 2005), dissolved reactive phosphorous PO_4P (ECS 1996), and total phosphorous TP (ISO 2003) were analysed within 7 days of fieldwork. Turbidity (in NTU) was measured from a 40 ml with a spectrophotometer (2100AN turbidimeter, Hach Lange). The turbidity measurement is listed only for the unfiltered sample as the turbidity values measured for the filtered solution (0.45μm filter) were typically <0.5 NTU, which falls within the error range. The turbidimeters used in this study, works on the principle of measuring the scattering of light by suspended particles (Gippel 1995). The Pro-DSS measured turbidity in Formazin nephelometric unit (FNU) and the turbidimeter used Nephelometric turbidity unit (NTU), which are equivalent units and vary only depending on the instrument's technology (Dogliotti et al. 2015) and so are considered equal in this study. Absorbance and fluorescence spectroscopy were analysed using an optical spectrofluorometer (Aqualog, Horiba) for both the unfiltered and filtered (0.45μm filter) samples. Corrections were made for inner filter effect, Raman scattering, and first and second order Rayleigh scattering with the Aqualog software before further data analysis (Coble et al. 2014).

3.4. Data analyses

The drain locations were plotted in the open-source software QGIS 3.10 (*QGIS.org* 2021) and all the graphs and statistical analyses was done in Microsoft Excel 16.0 software.

3.4.1. Flow and Suspended Sediments

To calculate SS, turbidity was used as a proxy. The relationship is site-specific and can be derived from previously measured, known values of turbidity and SS. All measured turbidity and SS values between January and June 2021 from the SDs and TDs were regressed, to get a linear relationship for each individual SD (Figure 6-Figure 12).

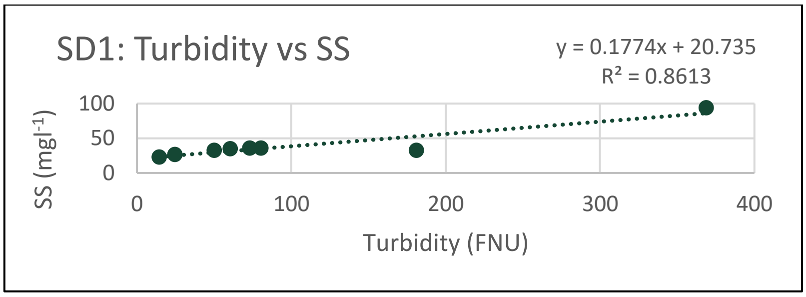


Figure 6 Turbidity vs SS (mgl^{-1}) for SD1 between January and May 2021

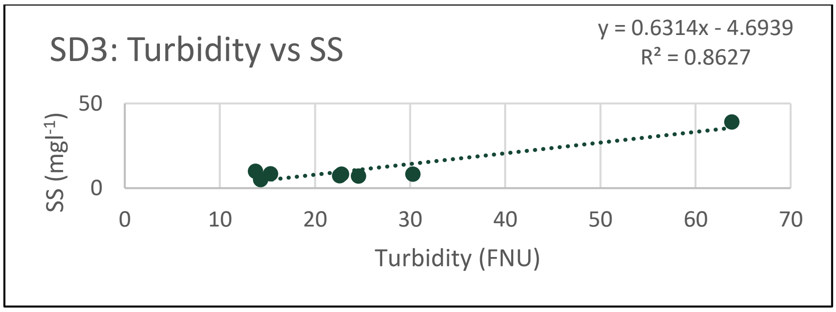


Figure 7 Turbidity vs SS (mgl^{-1}) for SD3 between January and May 2021

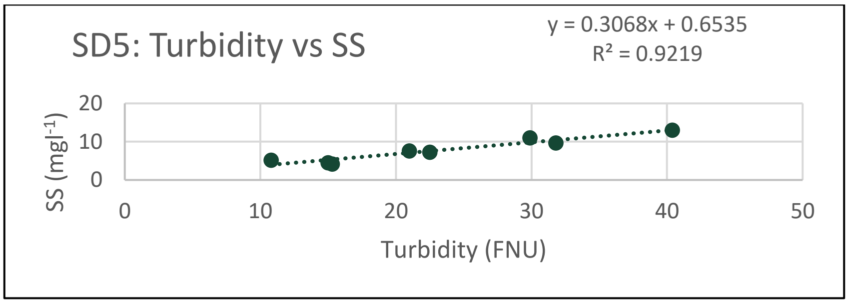


Figure 8 Turbidity vs SS (mgl^{-1}) for SD5 between January and March 2021

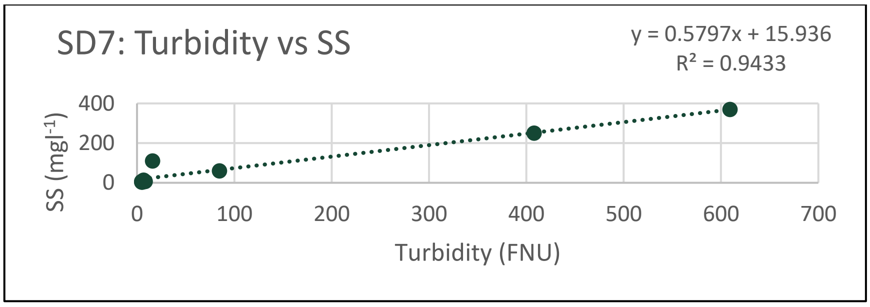


Figure 9 Turbidity vs SS (mgl^{-1}) for SD7 between January and June 2021

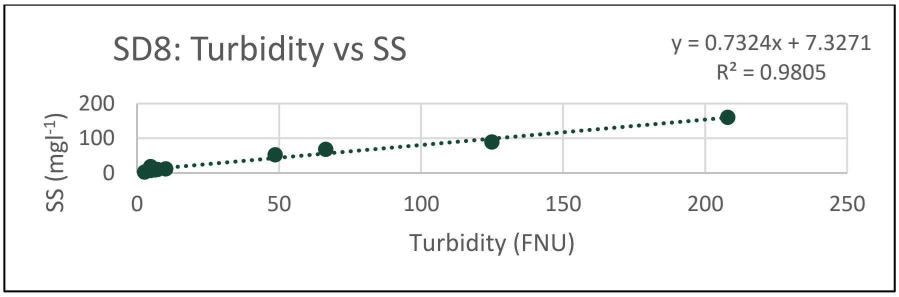


Figure 10 Turbidity vs SS (mgl^{-1}) for SD8 between January and June 2021

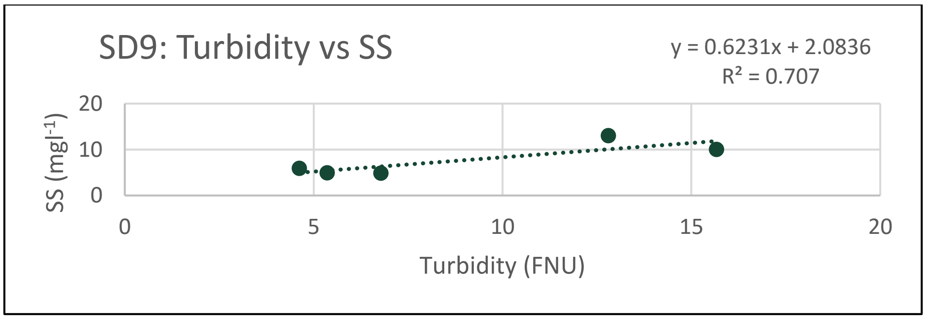


Figure 11 Turbidity vs SS (mgl^{-1}) for SD9 between January and April 2021

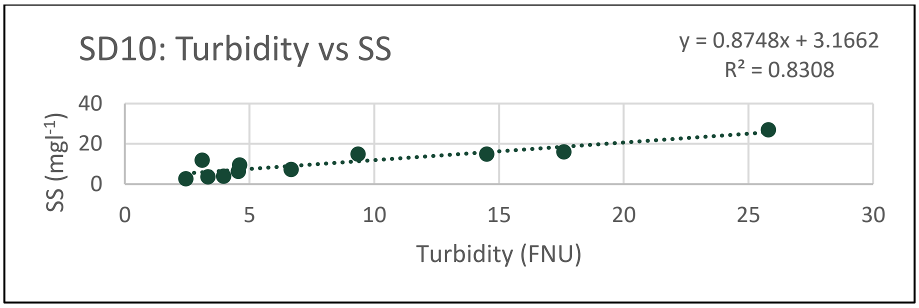


Figure 12 Turbidity vs SS (mgl^{-1}) for SD10 between February and May 2021

3.4.2. DOM indices

Excitation emission matrix (EEM), which was transferred from the Aqualog software was then processed using a MATLAB script to obtain the DOM indices. Common excitation (absorption) and emission wavelengths for some compounds are given in Table 3.

Table 3 Excitation and emission wavelength of common DOM compounds with humic-like compounds transforming at longer wavelengths than protein-like compounds: Taken from (Coble et al. 2014)

DOM-Type	Component	Ex (nm)	Em (nm)
Humic-like	Humic-like	330-350	420-480
Protein-like	Tyrosine-like	270-280	300-320
	Tryptophan-like	270-280	320-350

One example is the humification index (HIX) which is calculated by taking the ratio of emitted light intensity between specific wavelength ranges at the excitation wavelength of 254 nm.

$$HIX = \frac{Em\ 435 - 480nm}{Em\ 300 - 345nm\ and\ Em\ 435 - 480nm}\ at\ Ex254nm$$

This ratio gives the proportion of humic-like compounds in comparison to total (humic-like and protein-like) compounds (Table 3). As both compounds are aromatic, a common excitation wavelength of 254 nm is chosen. An example of absorption ratio, Spectral slope (S_R) is a ratio calculated accordingly (Helms et al. 2008).

$$S_R = \log transformed\ slope(\frac{275 - 295nm}{350 - 400nm})$$

This ratio showed evidence of photochemical degradation in such a way that the S_R increases on exposure to light, showing that there is a decrease in absorption at a longer wavelength. For the specific compounds that was of interest in this study, the ranges: Excitation Ex: 240-480 nm, Emission Em: 270-525 nm was used. The other indices were calculated and interpreted in a similar way (Table 4).

*Table 4 DOM indices and their interpretations, *Autochthonous: DOM produced in-situ; allochthonous: DOM transported from elsewhere*

Index	Name	What is it?	Values	Interpretation
Fluorescence Indices	Fluorescence Index (FI)	Indicator of DOM origin	1.4 (microbial) or 1.9 (terrestrial)	If the DOM is labile or recalcitrant
	Freshness Index (BIX)	Ratio of newly produced compounds to humic compounds	0.6 (allochthonous*) or >1 (autochthonous*)	If the DOM is produced there (new) or transported from somewhere else (old)
	Humification Index (HIX)	Indicator of degree of Humification of DOM	0 (labile) or 1 (humified)	If the DOM is protein-based or humic-based
Absorbance ratios	A254	Compounds that undergo absorption at 254nm	Higher value -> More aromaticity	If the DOM can adsorb a lot of nutrients
	E2:E3 ratio	Compounds that undergo absorption at 250nm to 365nm	Higher value -> Smaller compounds, Low MW	Smaller compounds are easy to decompose and carry less nutrients
	Spectral Slope (S_R)	Derived from log transformed values of absorption data	Higher slope -> Decrease in absorption with increasing wavelength	If the DOM prefers to absorb at higher wavelength or not -> indication of MW

In addition, the indices were obtained based on different conditions, which are briefly summarized in Table 5. Each index was calculated by setting a fixed wavelength range suitable for different water samples: excitation-emission wavelength for fluorescence index and absorbance wavelength for absorption index. The analysis was done in MATLAB.

Table 5 Basic wavelength used to extract DOM indices

Index	Parameter	Based on
Fluorescence Index	Em 470nm/Em 520nm at Ex 370nm	(Cory & McKnight 2005)
Freshness Index	Em 380nm/ Em between 420- 435nm at Ex 310nm	(Wilson & Xenopoulos 2009)
Humification Index	Between Em 435- 480nm/ Em 300-345nm and Em 435-480nm	(Ohno 2002)
A254	Abs 254nm	(Weishaar et al. 2003)
E2:E3 ratio	Abs 250nm/ 365nm	(Peuravuori & Pihlaja 1997)
Spectral Slope	log transformed slope (275-295nm/ 350-400nm)	(Helms et al. 2008)

3.4.3. Mass Balance- Nutrients

The removal rate in % (Table 8) for all the nutrients and SS was calculated with the input load (mgs^{-1}) at SDup, load from the drains and tributaries (mgs^{-1}), and output load at SDdn.

$$Removal\ rate(\%)\ for\ each\ nutrient = \frac{((SDup\ (mgs^{-1}) + Avg.drains\ load\ (mgs^{-1}) - SDdn\ (mgs^{-1})\ )}{(SDup\ (mgs^{-1}) + Avg.drains\ load\ (mgs^{-1})} * 100\%$$

The influence of drain in % was calculated based on the proportion of the total input load that was contributed by the drains/tributaries.

$$Influence\ of\ drains\ (\%) for\ each\ nutrient = \frac{Avg.drains\ load\ (mgs^{-1})}{(SDup\ load\ (mgs^{-1}) + Avg.drains\ load\ (mgs^{-1}))} * 100\%$$

3.4.4. Statistical Analysis

ANOVA

The one-way analysis of variance (ANOVA) was done using the data analysis tool pack on Excel. Alpha value α was fixed at 0.05 and significance between two (or more) groups were checked to confirm if they varied between each other.

Correlation

The Pearson correlations for the nutrient concentrations (mgl^{-1}) with SS (mgl^{-1}) and DOM indices were calculated for all the SDs using the 'Correlation' data analysis tool pack on Excel. The significance of each correlation was checked using the regression analysis tool.

In addition, the Pearson correlations between the nutrient concentrations (mgl^{-1}) with each other were checked for all the SDs.

4. Results

4.1. Drain location

The GPS locations of the identified drains and tributaries were marked. The maps could be used to trace the source of drains with highest nutrient and sediment concentrations. The map in Figure 13 is from SD1, which had one subsurface drain (D1) and three tributaries (T1, T2, T3). The remaining maps are listed in Appendix 1.

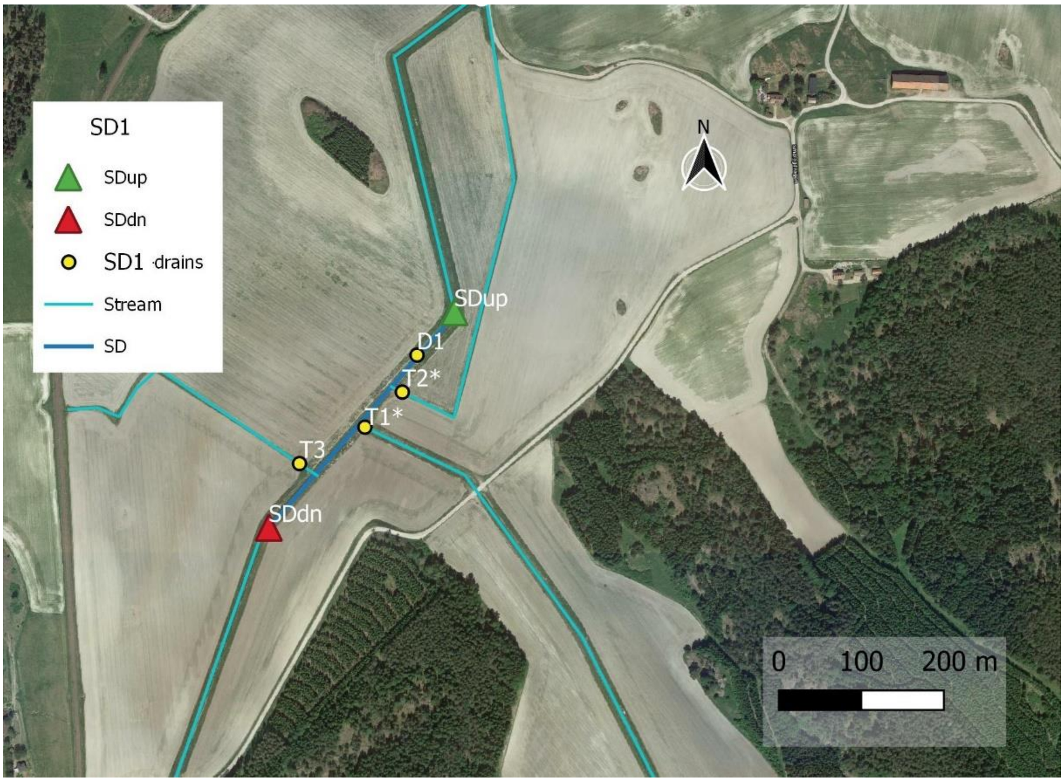


Figure 13 GIS map showing the drains of SD1 and the immediate field next to it. SDup and SDdn points mark the beginning and end point of the SD. The drains/tributaries with high nutrient concentrations are marked with ***** *(Source: QGIS 3.10.14)*

4.2. Discharge

The active subsurface tiles (drains) and tributaries during each visit were listed along with average discharge values in Table 6. It is to be noted that different drains were active during the visits, depending on the weather, topography, and the activity on the adjacent fields. SD3 had the greatest number of active drains in all the sampled months. SD1 has highest average discharge values in the drains due to high flow in the tributaries with highest values recorded in February and May (58.5 ls^{-1} and 14.08ls^{-1}).

Table 6 Average discharge values of all the drains and tributaries that were active during sampling. The discharge values of the SD (average of flow at SDup and SDdn) on the same day as sampling were also included. On a comparison between the SDs, the highest discharge was seen in SDs 8, 9, and 10.

SD	Month	Total no. of identified drains		No. of active drains		Average discharge of the drains (ls^{-1})	Average discharge of the SD (ls^{-1})
		Tributary	Subsurface (Pipes)	Tributary	Subsurface (Pipes)		
1	Feb.	2	0	2	0	58.50	68.6
1	March	3	1	2	1	4.78	25.5
1	April	3	1	2	1	1.10	11
1	May	3	1	2	1	14.08	108
3	Jan.	2	32	0	15	0.12	23.7
3	March	2	32	0	12	0.12	14
3	April	2	32	0	14	0.08	22
3	May	2	32	0	12	0.11	85
5	March	3	11	1	3	0.64	87
5	April	3	11	0	3	0.11	33
5	May	3	11	0	3	0.07	98
7	March	1	34	1	11	0.15	58.5
7	May	1	34	0	3	0.05	14
8	March	0	7	0	3	0.23	307.5
8	May	0	7	0	0	0	120.5
9	March	1	8	1	1	0.16	340.5
9	May	1	8	1	3	2.53	95.5
10	March	0	33	0	5	0.16	182
10	May	0	33	0	4	0.05	214.5

4.3. Spatial Variation: Turbidity and nutrient concentrations

4.3.1. Turbidity

The turbidity values measured from drains, tributaries, SDup and SDdn were studied. The turbidity values were used instead of SS as an overall measure of sediments, coloured chemical, and biological sources. As the turbidity varied between 0.18NTU (in SD8) and 1306NTU (in SD3), the graph was shown on a logarithmic scale (Figure 14). It was seen that the drains in SD3 and SD1 had the highest mean turbidity values in comparison to the other sites (Mean: 45.47 and 153.68NTU). In addition, SD3 drains still showed the highest variation but there was also high variation in drains and tributaries of SDs 1, 7, and 9. Also it was possible to see that the mean turbidity within all the SDs was higher than the drains. However, the variation in the drains (and tributaries) were higher consistently (Figure 14). Turbidity values differed significantly between the SDs (one-way ANOVA, $F = 13.49$, $p < 0.01$).

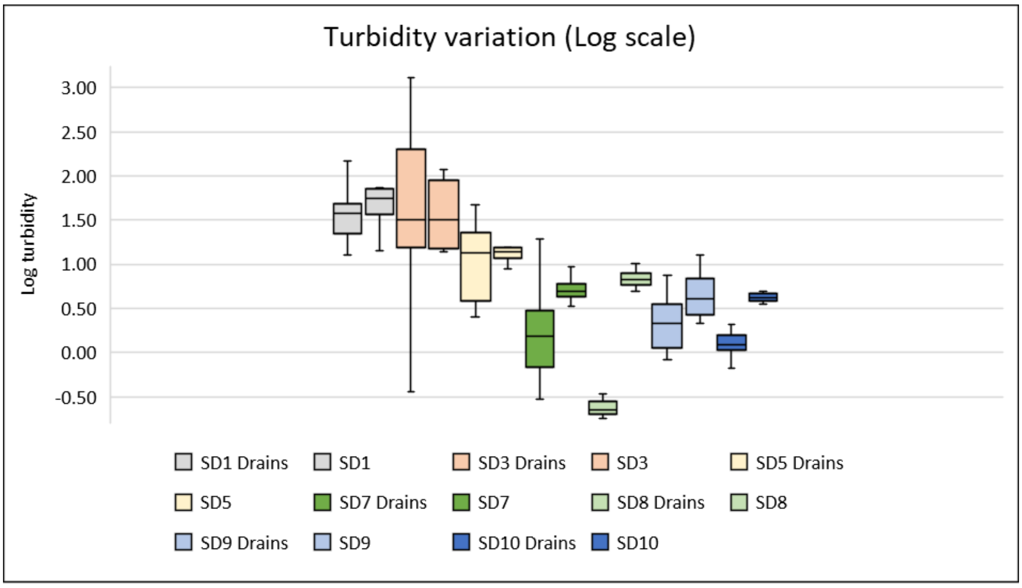


Figure 14 The turbidity values of all the drains and tributaries, SDup and SDdn points were plotted as boxplots to show the variation in the values, both within the SD and between each other. All samples taken between January and May 2021 in all the drains and tributaries were included; SD values include the corresponding turbidity values of both SDup and SDdn; Negative log values indicate turbidity values between 0 and 10 NTU

4.3.2. Nutrient concentrations

The average concentration of NO_3N, NH_4N, PO_4P, and TP from March 2021 for all the drains and SDs were studied. The month March was chosen to do the comparison as it was the only sampling month with samples from all the SDs. In addition, it included the start of fertilization and planting in many sites. From the concentration values of drains and tributaries in comparison to the SD-value, it was clearly seen that N values (NO_3N) were generally higher in SDs 7-10 (Figure 15) and P-values were higher in SDs 1 and 3 (Figure 17 and Figure 18).

From Figure 15, the average NO_3N concentrations of the drains for March in SDs 1, 5, 7, 8, 9, and 10 were higher than the SD by 4%, 51%, 202%, 274%, 123%, and 234% respectively. Some drains reported values as high as 34.7mgl^{-1} (SD7_D12), 33.9mgl^{-1} (SD8_D3), and 27.5mgl^{-1} (SD10_D8). SD3 was the only SD with lower average drain concentrations for NO_3N in the drain than the SD.

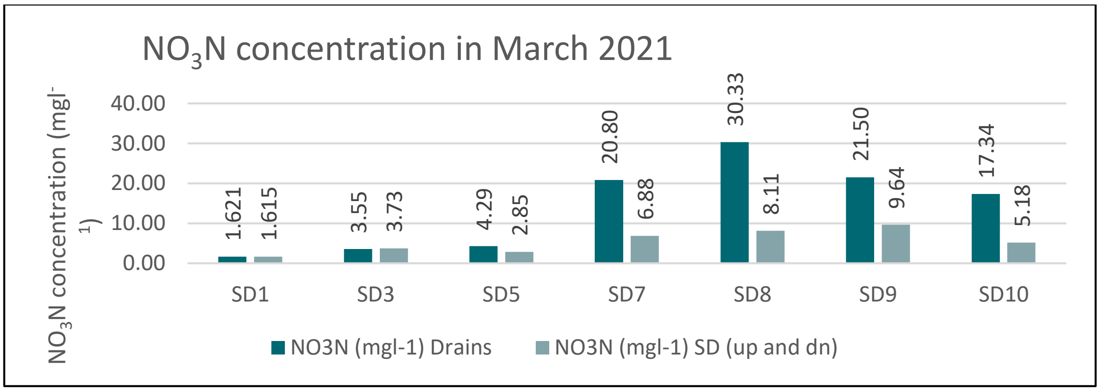


Figure 15 Average NO_3N concentration (mgl^{-1}) for the all the drains in March 2021 (dark blue); NO_3N values of the SD is the average of SDup and SDdn (light blue); Label indicates the corresponding values.

The average NH_4N concentration of the drains for SDs 3 and 9 were higher than that of the SD by 24% and 316% (Figure 16). Individual values were as high as 0.38mgl^{-1} (SD3_D23.5), 0.19mgl^{-1} (SD10_D7), 0.12mgl^{-1} (SD9_T1).

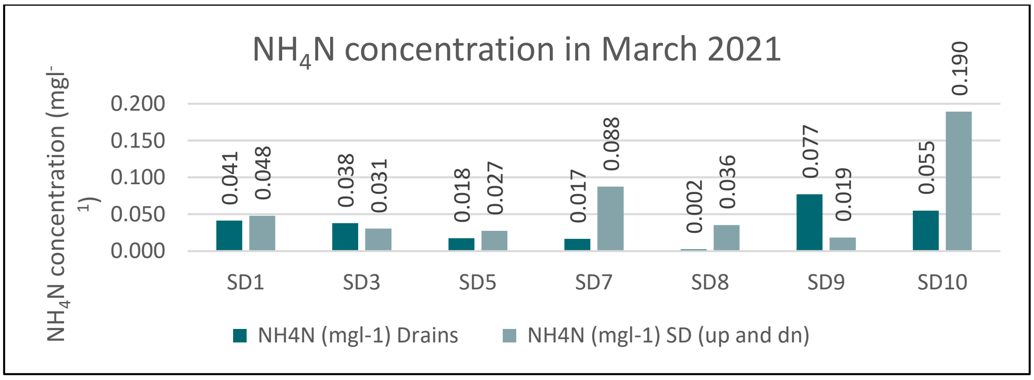


Figure 16 Average NH_4N concentration (mgl^{-1}) for all the drains in March 2021 (dark blue); NH_4N values of the SD is the average of SDup and SDdn (light blue); Label indicates the corresponding values

The average PO_4P concentration of the drains for SDs 1, 3, 8, and 9 were higher than the SD by 41%, 34%, 86%, and 63% (Figure 17). Individual values were as high as 0.38mgl^{-1} (SD3_D3) and 0.035mgl^{-1} (SD8_D3).

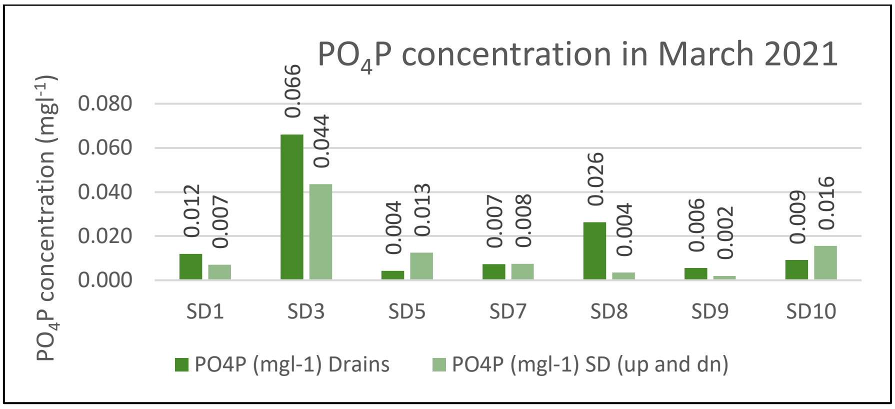


Figure 17 Average PO_4P concentration (mgl^{-1}) for the all the drains in March 2021 (dark green); PO_4P values of the SD is the average of SDup and SDdn (light green); Label indicates the corresponding values

The average TP concentrations of the drains were higher than the SD only in SD3 by 32% (Figure 18). Individual values were as high as 0.46mgl^{-1} (SD3_D3) and 0.12mgl^{-1} (SD1_T1).

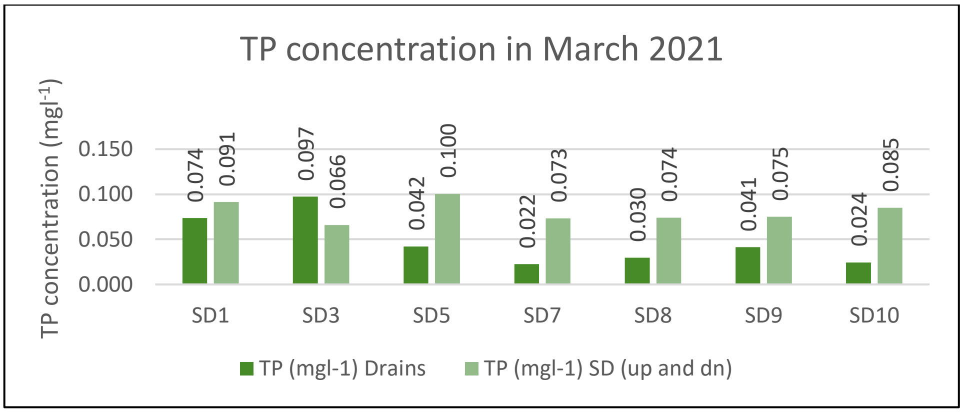


Figure 18 Average TP concentration (mgl^{-1}) for all the drains and in March 2021; (dark green); TP values of the SD is the average of SDup and SDdn (light green); Label indicates the corresponding values

4.3.3. Phosphorous forms

As PO_4P was a part of TP, its proportion of total leached P (mgl^{-1}) was checked in all the SDs (Table 7). Higher value denotes a greater presence of dissolved phosphorous fraction. While the SDs 3 and 8 had most of the phosphorous in reactive form (68% and 89%), SDs 1, 5, 9, and 10 (<40% each) showed the presence of other P forms.

Table 7 Proportion of PO_4P in TP for all the SDs. Average PO_4P and TP values from all the drains and tributaries were taken for the calculation

D/T	PO_4P (mgl^{-1})	TP (mgl^{-1})	Proportion of PO_4P (%)
SD1	0.01	0.07	16.33
SD3	0.07	0.10	67.92
SD5	0.01	0.05	10.63
SD7	0.01	0.02	32.23
SD8	0.026	0.03	88.96
SD9	0.01	0.04	13.32
SD10	0.01	0.02	37.89

4.4. Temporal Variation: Turbidity and nutrient concentrations

The log turbidity from SD3 in different months was studied (Figure 19). The variance and the mean turbidity value of the drains and tributaries were higher than the SD in January, March, and May. However, the difference was the greatest in January (mean of 2.3 in drains and 2.01 in the SD) and May (mean of 2.28 in the drains and 1.86 in the SD). Additionally, the range of turbidity was the greatest in May (log values between 1.31 and 3.11).

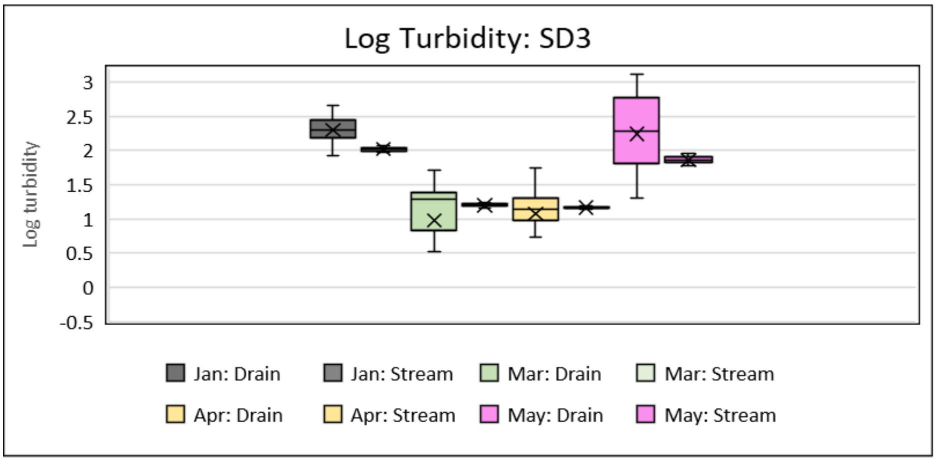


Figure 19 Log. Turbidity values of all the drains and tributaries from January to May in SD3 were plotted as boxplots to show the change in variation between different months, SD value for each month includes both SDup and SDdn values

The average loads (mgs^{-1}) in SD1 for all the months were studied (Figure 20). The values of all the nutrients were higher in the drains (and tributaries) than the SD in February, indicating a direct source. In terms of individual months, February had the highest load values from drains followed by May (except for NH_4N values).

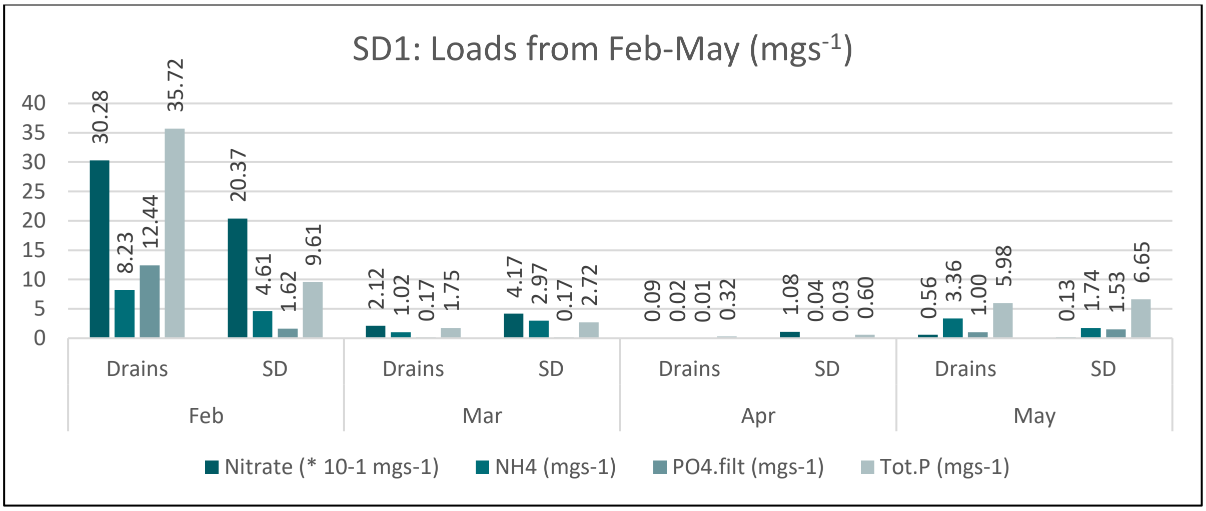


*Figure 20 The average loads (mgs^{-1}) for NO_3N, NH_4N, PO_4P, and TP were plotted for SD1 from February to May 2021. SD value is the average of SDup and SDdn. NO_3N was expressed in 10^{-1} * mgs^{-1} so that it can be comparable with the other nutrients. Label denotes the respective values*

4.5. Mass Balance- SS and Nutrients

The removal rate in % (Table 8) for all the nutrients and SS with the influence of drain in % was calculated for all the SDs on the sampling day each month. The value helped to identify the functionality of the SD as a response to high loads.

Different SDs had negative removal for different nutrients. SD3 had increased NO_3N and TP values in all the sampled months. SDs 1, 3, 5, and 10 had a net negative reduction of SS in different months. SDs 8 and 10 had a consistent negative removal only for PO_4P. However, a general trend that was noticed was an improvement in removal % from the snowmelt months (between January and March) towards April and May. In terms of influence of drains (and tributaries) in terms of input load into the SD, SDs 1 and 3 had the highest influence percentage. SD9 had a high influence with respect to the loads of NH_4N and SS.

Table 8 Removal rate (%) of the total input in each SD is given on the left; A positive value corresponded to a net reduction of the nutrient/SS at SDdn, and a negative value meant a net addition of the corresponding nutrient/SS. The SDs with a negative removal rate are marked in red. The Influence of the drains and tributaries on total input is given on the right. Higher values mean greater influence and these values are marked in green.

SD	Month	Removal rate (%)					Influence of drain/ tributaries (%)				
		NH_4N	NO_3N	PO_4P	TP	SS	NH_4N	NO_3N	PO_4P	TP	SS
SD1	February	54	51	90	79	77	69	63	87	79	65
	March	-103	-13	37	-92	-21	44	43	54	71	50
	April	24	0	50	-2	1	36	8	16	43	29
	May	72	-	59	38	31	63	-	35	50	23
SD3	January	-74	-256	-	-165	1	64	9	-	50	78
	March	17	-165	-	-199	-307	36	23	-	38	21
	April	43	-28	-	-28	-66	61	6	-	15	4
	May	17	-	20	-	22	2	-	6	-	8
SD5	March	-56	-45	-24	-51	-387	2	3	1	1	0.8
	April	89	-	70	-38	-150	0.1	-	0.3	1	0.3
	May	-73	-	10	-	1	0.1	-	0	-	0.1
SD7	March	14	16	-9	3	5	1	7	3	1	8
	May	75	-	89	91	48	0.2	-	1	0.3	4
SD8	March	57	15	-95	20	29	0	1	2	0.1	0.2
	May	35	-	-96	38	52	0	0	0	0	0
SD9	March	12	7	4	3	10	16	6	3	2	1
	May	39	-	38	7	14	29	-	1	7	14
SD10	March	-29	-43	-19	6	73	0.2	2	0.3	0.1	0.1
	May	-17	-	-196	-32	-311	0.1	-	0.2	0.1	0.1

4.6. DOM Indices

The spatial and temporal variation in the DOM indices for all SDs were studied (Figure 21 to Figure 27). The corresponding boxplot for the filtered samples is given in Appendix 2.

FI values (Figure 21) were the lowest in SD3 for samples taken in January with a mean of 1.4 (terrestrial source). The FI in the months April and May had an increasing trend with SDs 3, 5, and 9 having a mean FI of 1.7 each in May. For the other sites, the mean values in March and May were not significantly different.

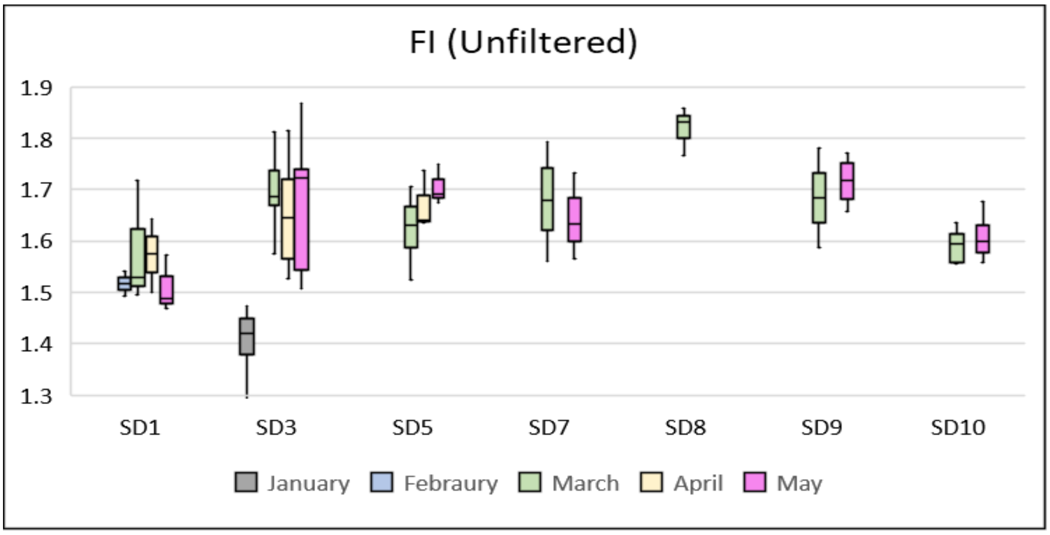


Figure 21 FI for the unfiltered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), April (yellow, SDs 1–5) and May (pink, All SDs) was expressed as boxplots; The boxplot was drawn including the median value

BIX values (Figure 22) were highly variable for all the sites in the drains. The mean value for all the months was 0.6 in SDs 1 and 10, 0.7 in SDs 3, 5, 7, and 9, and 0.8 in SD8. BIX also ranged from a mean of 0.6 (predominantly allochthonous sources) in February to 0.7 the rest of the months (mixture of both autochthonous and allochthonous).

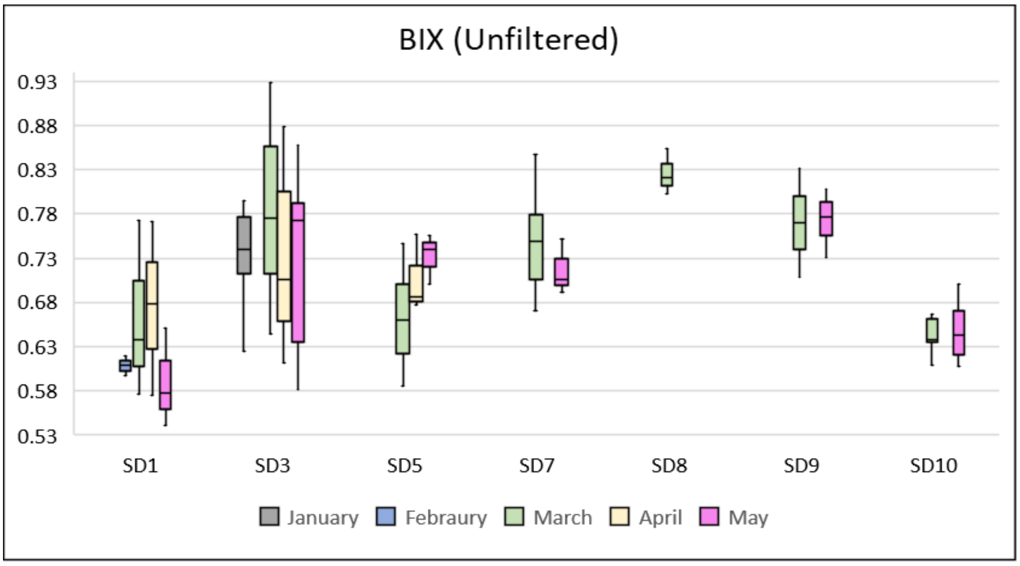


Figure 22 BIX for the unfiltered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), April (yellow, SDs 1-5) and May (pink, All SDs) was expressed as boxplots; The boxplot was drawn including the median value

HIX (Figure 23) for SD3 in the months January and May ranged between 0.3 (labile) to 0.9 (humified). However, the mean HIX in all the other sites SDs 1, 5, 7, 8, 9, 10 in all the months for the unfiltered samples were 0.9 each, which showed the predominant sites had highly humified DOM sources in all the sampled months.

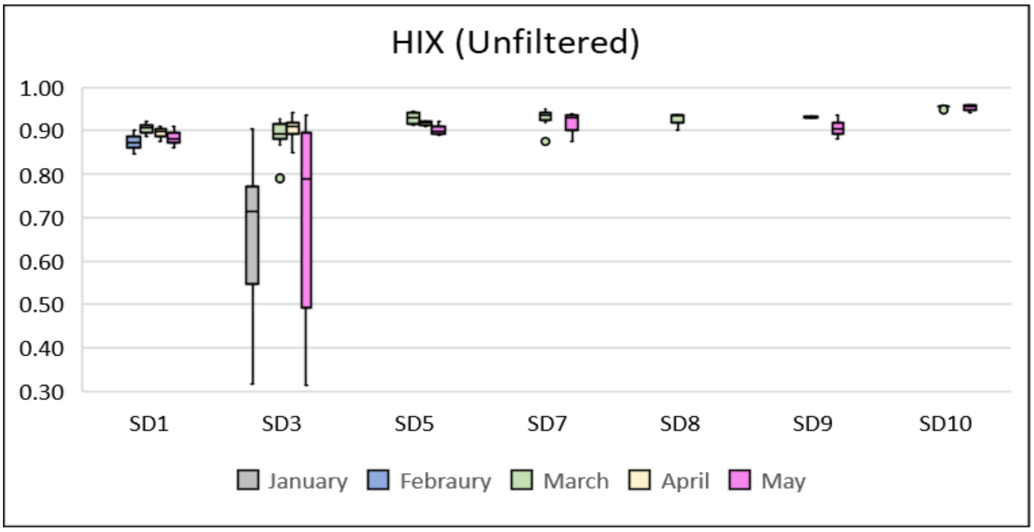


Figure 23 HIX for the unfiltered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), April (yellow, SDs 1-5) and May (pink, All SDs) was expressed as boxplots; The boxplot was drawn including the median value

A254 (Figure 24) for SD3 in the months of January and May varied between 0.8 (low aromaticity) to 4.2 (high aromaticity). SD1 had a temporal variation with mean values of 1 and 1.3 in February and May and 0.5 in March and April. The other sites SDs 5-10 had low values from March to May (Mean of 0.4 in SDs 5 and 10, 0.2 in SDs 7, 8, 9).

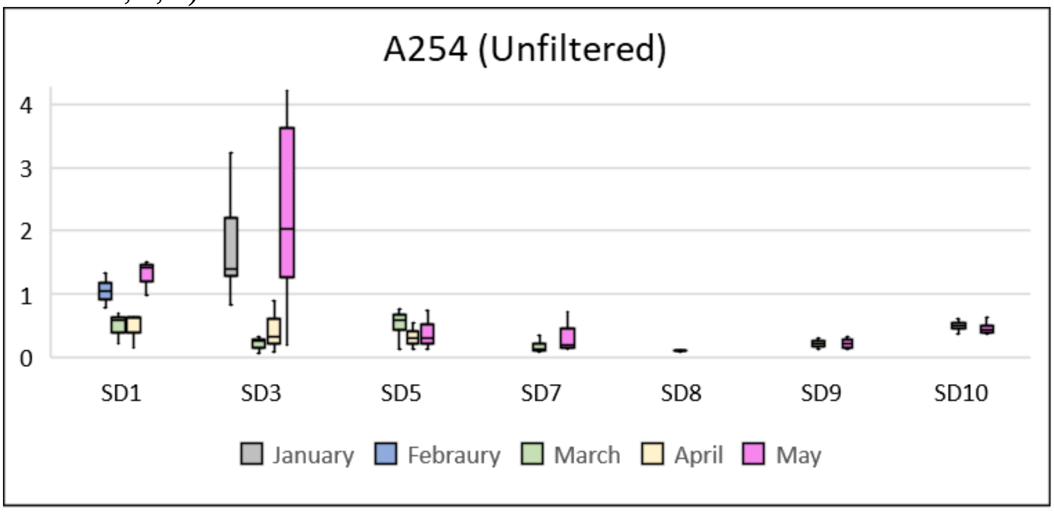


Figure 24 A254 for the unfiltered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), April (yellow, SDs 1-5) and May (pink, All SDs) was expressed as boxplots; The boxplot was drawn including the median value

From SDs 1, 3, and 5 (Figure 25), it was seen that E2:E3 in April (mean: 2.03) was higher than March (mean: 1.17) for unfiltered samples. SDs 1 and 5 had a high variation in April (ranges from 0.4 to 3.9). SD3 had generally higher values in January (mean:2.6), but its drain ratios had high variations in March and April (ranged from 0.39 to 3.49).

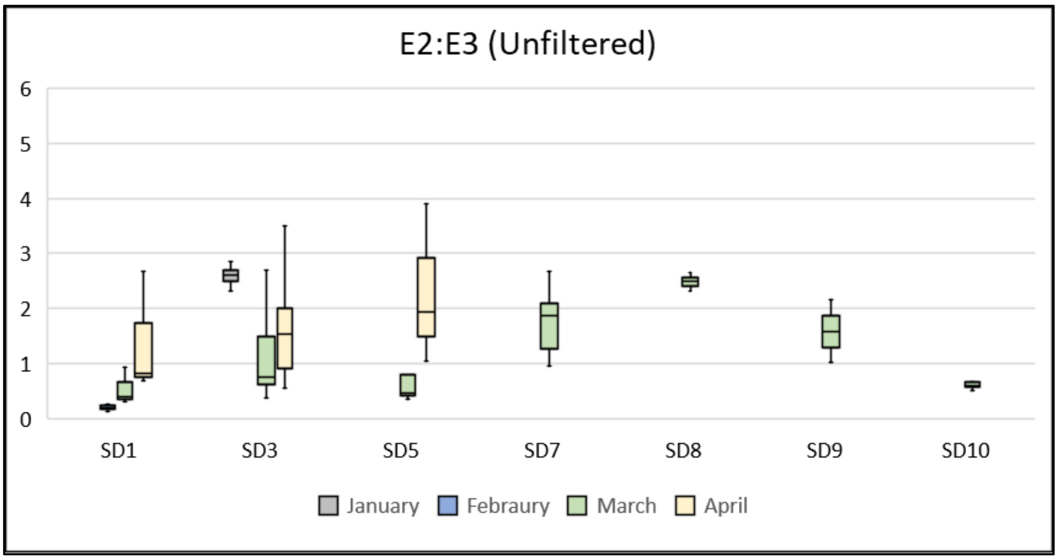


Figure 25 E2:E3 ratio for the unfiltered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), and April (yellow, SDs 1-5) was expressed as boxplots; The boxplot was drawn including the median value.

In the month of May (Figure 26), SD9 had the highest mean E2:E3 ratio of 33.72 and SD5 had the highest variation (ranged from 6.5 to 41.6).

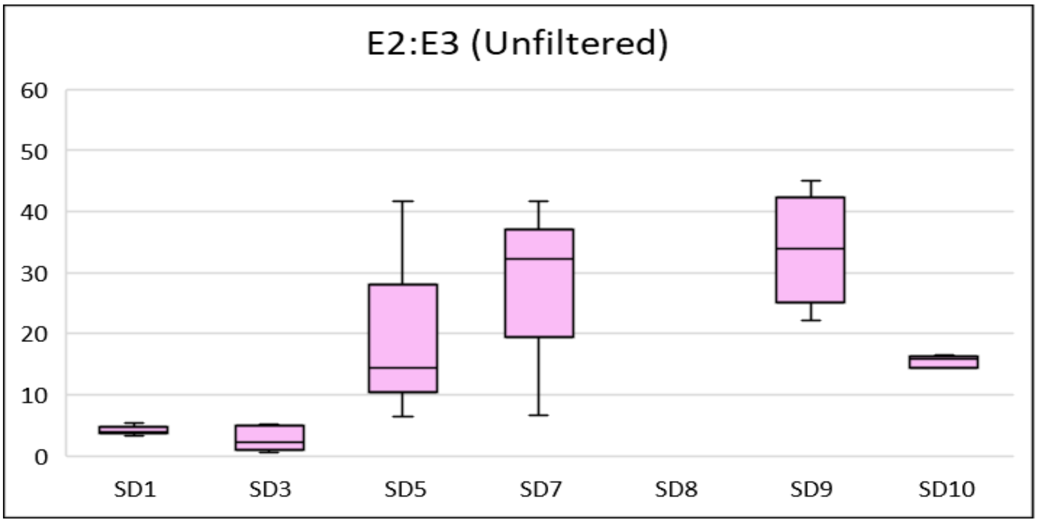


Figure 26 E2:E3 ratio for the unfiltered samples in May (SDs 1-5) was expressed as boxplots; The ratio for May was shown separately as it had an increased value and required a different scale The boxplot was drawn including the median value

The S_R (Figure 27) was highly site-specific. SDs 3,5,7 and 8, showed high variation in March (ranged from 1.03 to 2.25). SD9 had the highest mean value of

2.4 in May. When compared between March and May, SDs 1, 5, and 9 had an increase in S_R (decrease in absorption at long wavelength: formation of smaller compounds). SDs 3, 7, and 10 had a decrease in S_R (formation of larger compounds).

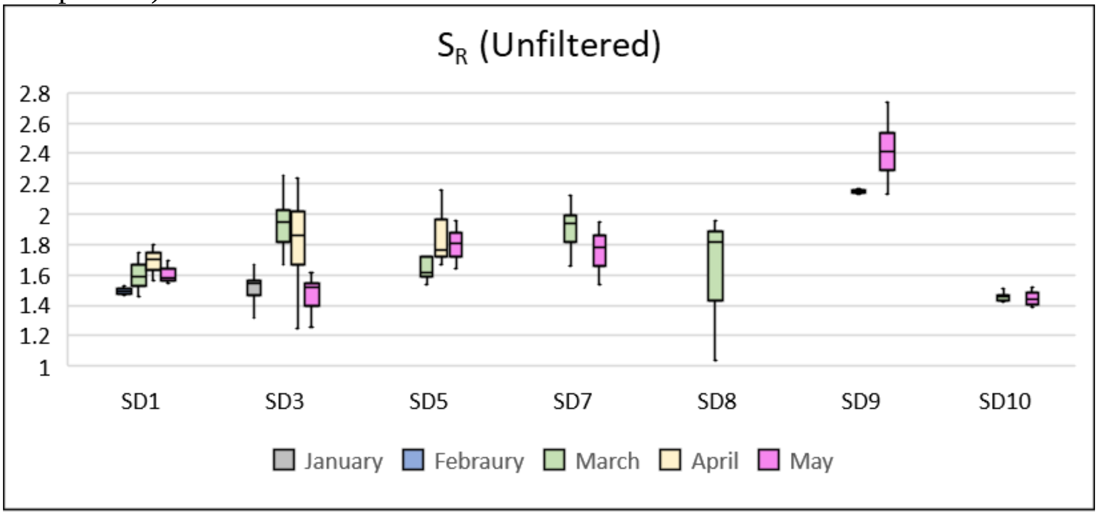


Figure 27 A254 for the unfiltered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), April (yellow, SDs 1-5) and May (pink, All SDs) was expressed as boxplots; The boxplot was drawn including the median value

4.7. Correlations

4.7.1. Correlations-DOM

The correlation results for both unfiltered (Table 9) and filtered samples (Table 10) for SD7 with the DOM indices were shown here. SS was used instead of turbidity to separate out the adsorbed nutrient pathway. The correlations, both significant and insignificant, for the remaining SDs are shown in Appendix 3.

In SD7, for the unfiltered solution, PO_4P correlated with HIX and A254, TP correlated with FI, BIX, and SS. The differences in correlations between the unfiltered and filtered samples were noted. In filtered samples, NO_3N correlated with S_R, and TP correlated additionally with HIX and A254.

Table 9 Correlations made for unfiltered drain water samples in SD7; Significant correlations ($p<0.05$) is marked in bold text; Provided there was significant correlation, the values up to 0.50 were taken as weak correlation, values between 0.50 and 0.75 as moderate correlation and above 0.75 as strong correlation.

	NH_4N (mg l^{-1})	*NO_3N (mg l^{-1})*	*PO_4P (mg l^{-1})*	*TP (mg l^{-1})*
FI	-0.51	0.12	-0.16	**-0.71**
BIX	-0.47	0.19	-0.15	**-0.65**
HIX	0.26	-0.03	**-0.52**	-0.04
A254	0.09	-0.07	**0.71**	0.47
E2_E3	-0.11	-0.44	0.09	-0.15
S_R	0.18	-0.34	-0.36	0.00
SS (mg l^{-1})	0.11	-0.10	**0.52**	**0.77**

Table 10 Correlations made for filtered drain water sample in SD7; Significant correlations ($p<0.05$) is marked in bold text; Provided there was significant correlation), the values up to 0.50 were taken as weak correlation, values between 0.50 and 0.75 as moderate correlation and above 0.75 as strong correlation; Correlation with SS was not checked for filtered solution as it was assumed that all the sediments>0.45µm

	NH_4N (mg l^{-1})	*NO_3N (mg l^{-1})*	*PO_4P (mg l^{-1})*	*TP (mg l^{-1})*
FI	-0.50	0.23	-0.24	**-0.73**
BIX	-0.45	0.20	-0.31	**-0.67**
HIX	0.44	-0.19	0.17	**0.65**
A254	0.39	0.07	0.31	**0.56**
E2_E3	-0.20	-0.45	0.26	-0.18
S_R	0.07	**-0.70**	0.20	0.34

4.7.2. Correlations- Nutrients

The significant correlations between the nutrient’s concentrations were studied (Table 11). In SDs 1 and 3, there was a positive correlation between PO_4P and TP. In SDs 7 and 10, there was a positive correlation between NH_4N and TP. Additionally, there were also positive correlations between NO_3N with TP values in SD5.

Table 11 Significant Pearson correlations ($p<0.05$) found between the nutrients in all the SDs; Provided there was significant correlation), the values up to 0.50 were taken as weak correlation, values between 0.50 and 0.75 as moderate correlation and above 0.75 as strong correlation

Site	Correlations	R-value
SD1	PO_4P and TP	0.88
SD3	PO_4P and TP	0.67
SD5	NO_3N and TP	0.75
SD7	NH_4N and TP	0.55
SD10	NH_4N and TP	0.73

5. Discussion

One of the main objectives of this study was to evaluate the composition of the water quality in the drains and tributaries and check if its interactions with SS or DOM indices could provide any information about the source/transport of the nutrients. While the concentration of the nutrients (mgl^{-1}) could explain how the drains contribute to the SD, the input loads (mgs^{-1}) decide if the SD could remove them at the required rate. As the study included visits to the SDs, it was possible to consider the activity observed in nearby areas as potential sources. In addition, the behaviour of each nutrient differs from the other based on its physical and chemical properties and thus will be discussed separately in the following sections.

5.1. Nitrogen Cycling

5.1.1. Ammonia

The influence of specific drains in transporting high amounts of ammonia to the SD could be noticed in SD1, SD3, and SD9 (Figure 16). SD1 and SD9 had high contributions from 2 tributaries with the tributary SD1_T1 having an average concentration of 0.06 mgl^{-1} (SDup mean: 0.04 mgl^{-1}) and SD9_T1 having an average concentration of 0.11 mgl^{-1} (SDup mean: 0.03 mgl^{-1}). SD3 had high contributions primarily from 4 drains: D6, 13, 18, and 23.5 with a mean concentration of 0.41 mgl^{-1} (SDup mean: 0.03 mgl^{-1}).

The observation of the area at a proximity to the sources with high concentration showed some possible explanations. SD9_T1 was connected to a constructed wetland (Figure 28), where a condition of low pH and ammonia build-up could have occurred, making it a potential source (Clarke & Baldwin 2002).

Figure 28 Wetland close to SD9_T1

In SD3, D23.5 was a ϕ48 cm drain that was not part of the planned subsurface pipes (which were between ϕ10-15 cm) and was suspected to have been set up as water management from the nearby household and roads, which could be the source of nutrients (Gray & Becker 2002). The drains D6, 13, and 18 were linked to the same field (Figure 32), which also had an animal barn. If animal manure was stored or spread on the field, elevated ammonia levels could be possible (Hernandez-Ramirez et al. 2011).

The sediments and DOM indices gave more information about how ammonia could have been transported. Ammonia concentration in SD1 correlated negatively with S_R (Appendix 3: Table 13) but positively with SS. The negative correlation with S_R became weaker with the filtered solution. This could suggest the mineralisation of ammonia in the sediments by decomposition of organic matter (Arango & Tank 2008). Ammonia concentration in SD3 (Appendix 3: Table 14) did not correlate with sediments or any of the DOM indices suggesting a presence of less-soluble inorganic forms of ammonia (Bridger et al. 1962). Ammonia concentration in SD9 (Appendix 3: Table 17) correlated negatively with BIX but correlated positively with A254 and SS. While the A254 suggests a preferred bonding to aromatic compounds, the mean A254 value of 0.30 was low and the mean humification in the samples was high at 0.91 suggesting that ammonia from the surrounding area must have accumulated in the wetland in the past years, got humified and is now getting transported through the tributary.

5.1.2. Nitrate

The influence of drains/tributaries on the SD NO_3N values (mgl^{-1}) was quite different from that of ammonia. From Figure 15, it could be observed that all the sites (except SD3) showed a higher concentration of NO_3N in the drains than the SD for the month of March. The difference was more prominent in the SDs 7, 8, 9, and 10 with a mean of 20.80, 30.3, 21.5, and 17.3mgl^{-1} (SD mean: 6.88, 8.1, 9.64, and 5.18mgl^{-1}). From the direct observation of the sites, the major source of NO_3N could be linked to fertilization in the neighbouring fields, which usually begin in the month of March for the spring cropping (*SCB* 2020).

The DOM indices were considered for more information on the source and transport of NO_3N. It was interesting to note that there were very few significant correlations between NO_3N concentrations and DOM indices in the SDs. In SDs 5 and 7, the 0.45µm filtered samples correlated negatively to S_R (Table 10), Appendix 3: Table 15), which suggested a presence of compounds with high aromaticity and MW (Helms et al. 2008). However, the lack of correlations in most SDs made it difficult to draw conclusions to the influence of DOMs in nitrate transport. This also corroborates the idea that mineral fertilizers would have been a source of NO_3N values. In addition, a study found that high HIX values, as seen in the SDs (Mean: 0.9 each) in fertilized fields could be due to a disturbance of organo-mineral aggregates by tillage practices (Graeber et al. 2012).

SD3 had a high NO_3N concentration in the stream, especially in January (Mean: 6.36 mgl^{-1}). Considering no uptake from vegetation due to the cold season, the source of the NO_3N could be from the runoff (Ohte et al. 2004) or groundwater (Guimerà 1998).

5.1.3. Nitrogen Removal

The mass balance of the nitrogen species in the SDs: NH_4N and NO_3N was studied together using the removal % (Table 8). SDs 1, 3, 5, and 10 were the sites of net positive addition (negative removal %) of NO_3N and ammonia at SDdn. While SDs 1 and 3 had a high influence from the drains in terms of input, these four SDs also had a common trait in terms of narrow terrace width. Two of the most important means of nitrogen removal in SDs were by means of absorption by biota or denitrification (Hodaj et al. 2017). Assuming that the role of plants to take up nitrogen was negligible during the study period (January-April), denitrification rates were expected to play an important role. One study found that, for increased denitrification rates, a wide terrace of 10 m for the tile drainage outlet would be ideal (Mahl et al. 2015). In addition, the study also stated that an increased concentration of NO_3N in the input to the SD increased denitrification rates only up to 5 mgl^{-1} beyond which the rates flat lined (Mahl et al. 2015). This suggested a fixed capacity for the SDs to remove nitrate at a given time. SDs 1, 3, 5, and 10 had small terraces, either by design or due to erosion, which could have led to lower an increased load at SDdn. By comparing it to the high nitrate removal % in SDs 7, 8, and 9, this could be attributed to their wider terraces (Figure 29).

Figure 29 Terrace from SD8 taken from May2021

Another interesting observation was in sites with positive ammonia removal but negative nitrate removal. This raised an important question about nitrification rates from ammonia to nitrate. The ideal pH range for nitrifying bacteria is around 6.6-9.7 (Odell et al. 1996). Most of the SDs were in agricultural sites with a mean pH of about 8, and so it was possible that some of the ammonia got converted to nitrate. This could be beneficial if it led to a simultaneous increase in denitrification but considering nitrate could be easily leached in comparison to ammonia (Ball Coelho et al. 2012), nitrification and denitrification rates should be monitored in future studies.

5.2. Phosphorous cycling

Phosphorous exists in many forms depending on its oxidation state and it is hard to quantify due to its unique associations. While this study included direct measurements of dissolved reactive phosphorous (PO_4P), and undissolved total phosphorous (TP), the other forms, such as particulate phosphorous, could be detected in specific sites. Since PO_4P is a fraction of TP, they were considered together for the discussion.

5.2.1. Source and transport

The mean concentration for PO_4P in SDs 1, 3, 8, and 9 was higher in the drains than in the SD but the TP concentration for the drains was only higher for SD3 (Figure 17, Table 6). PO_4P, a reactive, readily bioavailable form was a big proportion of the TP in SDs 3 and 8 (Foy 2007). The source of PO_4P in the sites from direct observation seemed to be agricultural activities, but the drain SD3_D23.5, which led to the nearby household and roads, also pointed at domestic waste.

To understand the P sources further, its DOM interactions were considered. A study done on different streams stated that higher PO_4P values were observed in sites with lower aromaticity (low A254), more microbial-like DOM (FI closer to 1.9) (Coble et al. 2016). Comparing this with the current study, it was seen that drains with high concentrations in SD3, SD8, and SD9 had higher FI (mean of 1.84 and 1.79 respectively) and higher E2:E3 ratio (10.67 and 9.25 respectively). But the SDs 1, 8 and 9 showed no significant correlations between PO_4P and any DOM indices, and so, the PO_4 could be inorganic fraction leaching from the neighbouring fields. However, SD3 PO_4P concentration correlated negatively to S_R and positively to A254 (Appendix 3: Table 14), indicating preferential association to highly aromatic compounds and considering that HIX was high (mean of 0.9 each), it could be an indication other fractions of P, namely colloidal fraction that usually ends up in filtered solution (Haygarth et al. 1997).

TP values in most streams were already high compared to the drains. Looking at the DOM associations in the SDs, SD3 TP correlated positively with SS and negatively with HIX (Appendix 3: Table 14). While the correlation with SS showed that some phosphorous is adsorbed to sediments (Davis et al. 2015), a negative correlation to HIX suggests that phosphorous did not adsorb to DOM surfaces (Hansen et al. 2002). Another trend that was seen in SDs 1, 3, and 9 was a positive correlation with A254 and a negative correlation to S_R (Appendix 3: Table 13). Despite a strong positive correlation with A254, the mean A254 ratio value for all the SDs was low, suggesting that the presence of some unreactive organic phosphorus forms, that are typically in lower MW substances (Darch et al. 2016).

5.2.2. Phosphorous removal

The mass balance on removal % of phosphorous forms gave some understanding into the functioning of the SDs (Table 8). SDs 5, 7, 8, and 10 had negative removal (net addition at SDdn) for PO_4P loads. The influence of drains in the input load was low (0.2- 2%). However, one study did a long-term evaluation on dissolved reactive phosphorous and stated that the biggest source for it was runoff from fields, with a combination of fertilization application and storm events (Daloğlu et al. 2012). This study was conducted in the months between March and May, which included a growing period in the sites with fertilization. In addition, there was also precipitation in the period (Figure 30), making the field activity a plausible reason for high P values.

TP removal was the lowest in SDs 1, 3, 5, and 10. Here, it was interesting to note that most of the sites with net negative TP removal in the SD were also accompanied by an increase in SS (Table 10). One of the ideas behind constructing the SD with wide terraces was to slow down the streamflow, causing the sediments to settle down, causing a simultaneous reduction in phosphorous values due to the association of particulate phosphorous with sediment surfaces. SDs 1 and 3 had a high influence from the drains but SD 1 showed a positive trend in May for both TP and SS, which could be an indication that the growing vegetation was trapping the sediments and/or taking up the phosphorous (Lee et al. 2000). A similar result could not be confirmed for SDs 3 and 5 as the TP from May wasn't measured, but there was a successive reduction in SS from March to May, so a similar trend might have continued for TP. SD10 had an increasing SS and TP in the month of May, even in the presence of vegetation, so the erosion in this SD and instability in the terraces could be due to other factors, possibly its soil properties (73% sandy soil) or high rainfall (853mm).

5.3. Turbidity and SS

The role of SS in nutrient transport and retention had been previously discussed. In addition, sediments also provide additional surfaces for microbes to grow and interact (Rehmann & Soupir 2009). In SDs, the sediments can be beneficial if they settled along the terraces but a very high sediment level in subsurface drainage tiles was an indication of erosion in the field level. This study measured turbidity as a proxy for SS (Figure 6- Figure 12). The dependence of turbidity on soil properties could be seen in Figure 13, where SDs 1 and 3 had the highest turbidity values. One study found that clayey soils tended to leach more clay-sized particles (1nm-1µm) than sandy soil resulting in more phosphorus leaching (Poirier et al. 2012). As stated in the study, SDs 1-5 had more clay %, had higher turbidity and higher TP concentrations (Figure 3, Figure 14, Figure 17) in the drains and tributaries in comparison to SDs 7-10. However, turbidity within the drains of almost all SDs showed big variation, even when they originated from the same field (Figure 14), which could be due to the varying sediment movement in the field by tillage, preferential flow through macropores, or in the case of clayey soil, shrink-swell pores (Coelho et al. 2010).

As SD3 had the maximum variation, it was interesting to see if there were any seasonal variations so, the monthly turbidity values were plotted (Figure 19). January and May were the months with the highest variations, which coincided with snowmelt and precipitation. A study found that the sediment values were higher in the tile drains when storm events followed a dry period (Simard et al. 2000). In May samples of SD3, turbidity in some drains went as high as 1306NTU, which was undesirable both from field productivity, as most of the SS were fine sediments and functioning of SD, as a strong correlation seen in SD3 with TP meant that high turbidity also increased TP load.

5.4. Correlation between nutrients

Correlations between nutrients were studied to predict if they could behave similarly. This would be beneficial in terms of suggesting mitigation measures targeting several nutrients. From Table 11, it could be seen that PO_4P and TP correlated with each other for SDs 1 and 3. This was to be expected as the source for both these nutrients could be the same. However, in SDs 7 and 10, there was a positive correlation between NH_4 and TP. This could be linked to the usage of manure on fields, which included struvite, a soluble compound containing ammonium phosphate. However, since the proportion of PO_4P fraction in TP was low in both the sites (Table 7), it could just be that both nutrients were transported in a similar way by adsorbing to sediments and DOM. This could explain why a sediment settling could reduce both TP and NH_4 in the SDs. SD5 had a correlation between NO_3 and TP (Appendix 3:Table 15) and considering both these nutrients also correlated positively with HIX and negatively with S_R, and SD5 had a high % of forest (Appendix 1: Figure 34), this could be high-resistant humified forms leaching from there (Coble et al. 2016).

5.5. Spatial and temporal variation in DOM

The general influence of DOM indices in transporting high level of nutrients has been discussed, but the variation in DOM with respect to different sites were also studied (Figure 21- Figure 27). SDs 1 and 3, where samples were taken in January and February showed DOM with more terrestrial (low FI), allochthonous (low BIX) origin compared to later months, which was expected following a period of snow cover. E2:E3 ratio increased generally from January to May, corresponding to the crop season, and SDs 7-10 with higher cropland, had a higher ratio. The surprising value was that of HIX which showed highly humified sources in most of the SDs, even in May, when the cropping system had begun. Combined with the fact that A254 was low until May confirmed with the studies that suggested that an increase of agricultural practices had led to a unique DOM signature of highly humic compounds with microbial-like aromaticity (Stedmon & Markager 2005; Graeber et al. 2012). This is a combination that can adsorb nutrients to transport it yet decompose easily to release it downstream, which could potentially lead to increased eutrophication.

SD3 stood out in some values during January and May, coinciding with high discharge periods (Figure 4A). For example, both HIX (between 0.3 and 0.9; Figure 23) and A254 (0.1 and 4.2; Figure 27) showed great variations in the drains. So, when the individual drains were considered in that periods, it was seen that labile, highly aromatic compounds were linked to drains SD3_D3, 3.5, and 4, which seemed to originate from the same area (Figure 32). While this was expected amidst cropping period in May, it was surprising in January, when the fields were fallow. Hence, it was assumed that the DOM could have accumulated and washed away from the neighbouring farm or the forest (Figure 32).

5.6. Implications and Limitations

5.6.1. Implications

Evaluating the impact of drains on nutrient concentrations and loads is difficult because of its direct and indirect dependency on constantly changing variables such as pH, temperature, climate, redox status, vegetation type, and management practices. In addition, determining the exact origin of nutrients and sediments in SDs requires appropriate tracing techniques, which can be complex. Despite better accuracy, it is not always possible to take daily samples due to the extensive resources required for individual drains. Also, periodic grab sampling is an efficient way of measurement when the final goal is fixed and quantifiable. This study, which tried to provide information on the inputs from the drains relative to concentrations and loads within SDs was able to accomplish that based on specific sampling dates. While monthly sampling in all sites was not possible, it was quite sufficient to understand the impact. For example, in SD3, where the sampling was done in January, March, April, and May, January and March accounted for the snowmelt period, April was relatively dry with low flow and May included a precipitation event following the dry period (Figure 4B). Additionally, the period also had no vegetation in January and tracked the influence of a vegetated terrace by May. By considering the values of measured nutrients and sediments in these conditions, it was possible to understand its effect on the functioning of the SD.

From the results, it was possible to see that the drains mostly had a higher concentration of N, P, and SS in comparison to the SD. This is dangerous as it could leach downstream during high flows when the discharge is high, and removal is limited. It could also leach downstream during low flows when the SD only has base flow and the terraces aren't flooded, leading to a reduced removal (Mahl et al. 2015). While the loads from drains were much lower than the SD (Table 6), its role in NH_4N and PO_4P was especially high in many sites. In addition, the study included only the drains within the SD's length and might have missed contributions from many upstream.

On comparison with previous years, this year 2021 had a high precipitation (Figure 4A), which could have contributed to an increased leaching of highly humic substances in the drains and tributaries. However, an increased water flow in the SDs would have compensated for this by more frequently inundated terraces with an enhanced plant and microbial activity (Roley et al. 2012).

A high concentration of nutrients from the drains is generally expected in places of intensive crop production. One study done on the uncertainty of plant response to fertilizers in Europe found that farmers might over-fertilize to avoid crop failure and the nutrient leaching would be seen as an unavoidable effect (Lemaire et al. 2021). However, if the water quality issue is to be addressed, measures must start at the field level. The ways to reduce leaching through drains must start with quantity, type, and timing of fertilization, which is a part of precision agriculture (Cao et al. 2018). There is also research about using microbial symbioses to reduce/remove fertilization, as most of the fields already have legacy nutrients from previous inputs, just in inaccessible forms (Bolduc & Hijri 2011). There are also possibilities to improve the existing drainage system into a controlled drainage, which would give an option to control the amount of drainage, thus the amount of nutrients leached (Wesström et al. 2001).

5.6.2. Design Inconsistencies

The primary design of SDs had design parameters for subsurface drains, for example, that the drains should flow out on the terraces to increase the residence time. But that was not always observed and in some places the drains directly flowed into the SD, bypassing the vegetation. Other problems included sediment deposition, broken/leaky drains, and built too low (always flooded) to be included in the evaluation. Some of the commonly encountered design/maintenance problems are shown in Figure 30 A-F.

Figure 30 Design Inconsistencies A) SD5 long drain by-passing the terrace B) Broken drain in SD1 C) Low, flooded drain in SD4 D) Sediment deposition in old drain SD7 E) SD3 big drain by-passing the terrace F) Leaking drain in SD5

6. Conclusions

The influence of agricultural drains on the effectiveness of two-stage ditches was seen in two ways depending on the sampling month. On one hand, the drains and tributaries of SDs 1, 3, and 9 had the potential to be the direct source of nutrients and SS into the SD. This could increase the load (mgs^{-1}) to be reduced by the SD, causing a lower retention. On the other hand, the drains and tributaries of nearly all the SDs had a higher concentration of nutrients/ SS (mgl^{-1}), which could have caused a strain on the functioning of the SD.

To explain the sources and transport of high nutrient concentration in the drains and tributaries, their correlation to DOM indices and SS were studied. The results showed that ammonia preferred to associate with sediments or highly humified DOM. Nitrate did not correlate with DOM indices in most SDs suggesting predominance of the inorganic forms that are most likely linked to fertilization. PO_4P correlated positively with A254 but considering most drains had a low A254 value, indicated a possibility of colloidal phosphorous forms. TP correlated with SS, indicating a high fraction of particulate phosphorous, but in SD 3, it correlated negatively with S_R, indicating labile forms.

Nutrient removal was the lowest between January and March, probably due to flush accompanying the snowmelt. For SDs 3, 5, and 10, the nutrient reduction was low, despite vegetation in May. As the SS retention was also low in these places, the cause of nutrient reduction is assumed to due to an instability in the terraces and the SD, as a whole. SD8 had a low PO_4P retention in both sampled months, but as the influence from drains was low, its source needs to be studied in the future.

SDs 1, 7, 8 (except PO_4P), and 9 showed net positive nutrient and SS retention in May, despite high input from the drains, when the SD was fully vegetated and inundated. Considering that the SDs 7 and 8 were the widest terraces in this study, it showed that with good design, the SDs can be a good mitigation measure for nutrients and SS.

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I will end by thanking my parents, for giving me this second-chance at academics (still unbelievable!), sister and relatives, who have directly and indirectly helped me be at peace in a new country far, far away!

Appendix 1 Drain Locations and Mass balance data

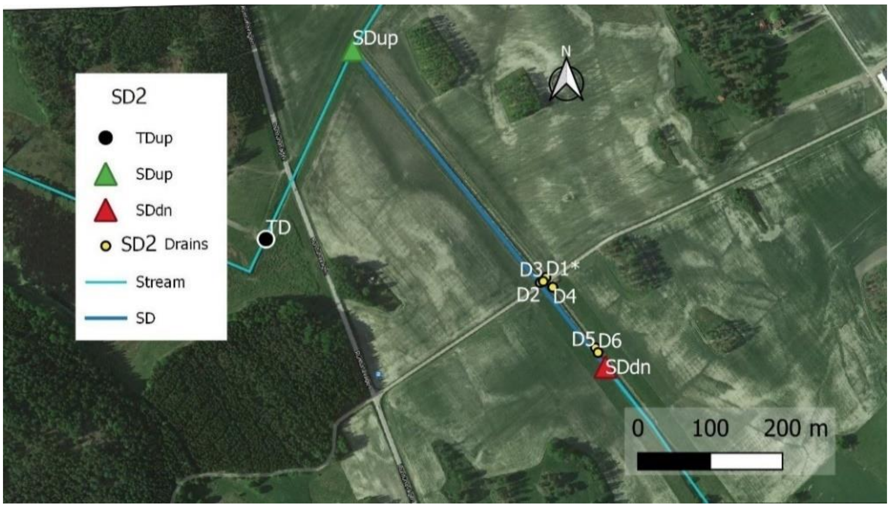

Figure 31 GIS map showing the drains of SD2; SDup and SDdn points mark the beginning and end point of the SD (Source: QGIS 3.10.14)

*Figure 32 GIS map showing the drains of SD3; SDup and SDdn points mark the beginning and end point of the SD; The drains/tributaries with high nutrient concentrations are marked with * (Source: QGIS 3.10.14)*

*Figure 33 GIS map showing the drains of SD4; SDup and SDdn points mark the beginning and end point of the SD; The drains/tributaries with high nutrient concentrations are marked with * (Source: QGIS 3.10.14)*

*Figure 34 GIS map showing the drains of SD5; SDup and SDdn points mark the beginning and end point of the SD; The drains/tributaries with high nutrient concentrations are marked with * (Source: QGIS 3.10.14)*

Figure 35 GIS map showing the drains of SD7; SDup and SDdn points mark the beginning and end point of the SD; The drains/tributaries with high nutrient concentrations are marked with ***** *(Source: QGIS 3.10.14)*

Figure 36 GIS map showing the drains of SD8; SDup and SDdn points mark the beginning and end point of the SD; The drains/tributaries with high nutrient concentrations are marked with ***** *(Source: QGIS 3.10.14)*

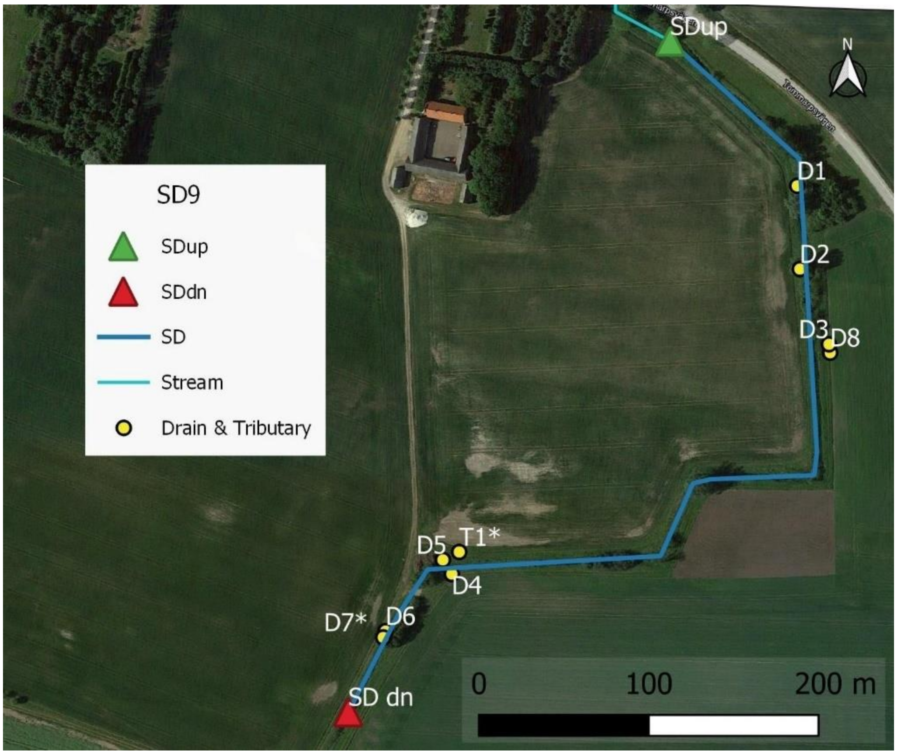

*Figure 37 GIS map showing the drains of SD9; SDup and SDdn points mark the beginning and end point of the SD; The drains/tributaries with high nutrient concentrations are marked with ***** (Source: QGIS 3.10.14)*

*Figure 38 GIS map showing the drains of SD10; SDup and SDdn points mark the beginning and end point of the SD; The drains/tributaries with high nutrient concentrations are marked with ***** (Source: QGIS 3.10.14)*

The removal rate (%) and influence of the drains (%) from Table 8 was thus calculated using the following data (Table 12)

Table 12 Sample concentration and load values for some drains and tributaries obtained for all the SDs; Load values (mgs^{-1}) were calculated using the concentration values (mgl^{-1}) and flow (ls^{-1}) from the same day. SDup and SDdn values are only listed for SD1

Sampling date	**Site**	**Location**	**NH_4N (mgl^{-1})**	**NO_3N (mgl^{-1})**	**PO_4P (mgl^{-1})**	**TP (mgl^{-1})**	**Normalised flow (ls^{-1})**	**NH_4N load (mgs^{-1})**	**NO_3N load (mgs^{-1})**	**PO_4P load (mgs^{-1})**	**TP load (mgs^{-1})**
22-03-2021	SD1	D1	0.03	2.54	0.003	0.04	0.25	0.01	0.63	0.001	0.01
22-03-2021	SD1	T1	0.07	1.46	0.01	0.12	14.00	1.01	20.44	0.17	1.74
22-03-2021	SD1	T2	0.02	0.86	0.02	0.06	0.10	0.0021	0.09	0.0021	0.01
23-03-2021	SD3	D2	0.01	13.00	0.02	0.03	0.17	0.001	2.17	0.003	0.005
23-03-2021	SD3	D3	0.01	3.50	0.38	0.46	0.17	0.002	0.58	0.06	0.08
23-03-2021	SD5	D5	0.08	0.33	0.00	0.02	0.11	0.01	0.04	0.0	0.002
23-03-2021	SD5	T2	0.01	1.33	0.003	0.01	2.00	0.01	2.66	0.01	0.02
25-03-2021	SD7	D12	0.01	34.70	0.01	0.02	0.17	0.001	5.78	0.0	0.004
25-03-2021	SD7	D16	0.02	30.50	0.00	0.02	0.11	0.002	3.45	0.0	0.002
25-03-2021	SD8	D3	0.004	33.90	0.04	0.04	0.15	0.001	5.05	0.01	0.01
25-03-2021	SD8	D6	0.003	28.10	0.03	0.03	0.25	0.001	6.96	0.01	0.01
25-03-2021	SD8	D7	0.00	29.00	0.01	0.02	0.31	0.0	8.87	0.004	0.01
23-03-2021	SD9	D7	0.04	22.90	0.01	0.02	0.21	0.01	4.86	0.002	0.004
23-03-2021	SD9	T1	0.12	20.10	0.002	0.06	10.00	1.18	201.00	0.02	0.63
26-03-2021	SD10	D7	0.19	13.30	0.01	0.04	0.18	0.04	2.45	0.002	0.01
26-03-2021	SD10	D8	0.01	27.50	0.01	0.02	0.17	0.002	4.58	0.002	0.004
22-03-2021	SD1	up	0.07	1.55	0.01	0.04	18.00	1.28	27.90	0.14	0.71
22-03-2021	SD1	dn	0.14	1.68	0.01	0.14	33.00	4.65	55.44	0.20	4.72

Appendix 2 DOM indices of filtered samples

FI values (Figure 21) were the lowest in SD3 for the filtered samples taken in January with a mean of 1.4 (terrestrial source). For the other sites, the mean values in March and May were not significantly different with a mean value of 1.5 in SD1, 1.6 in SDs 5,7,9, and 10 (terrestrial) and 1.8 in SD8 (microbial).

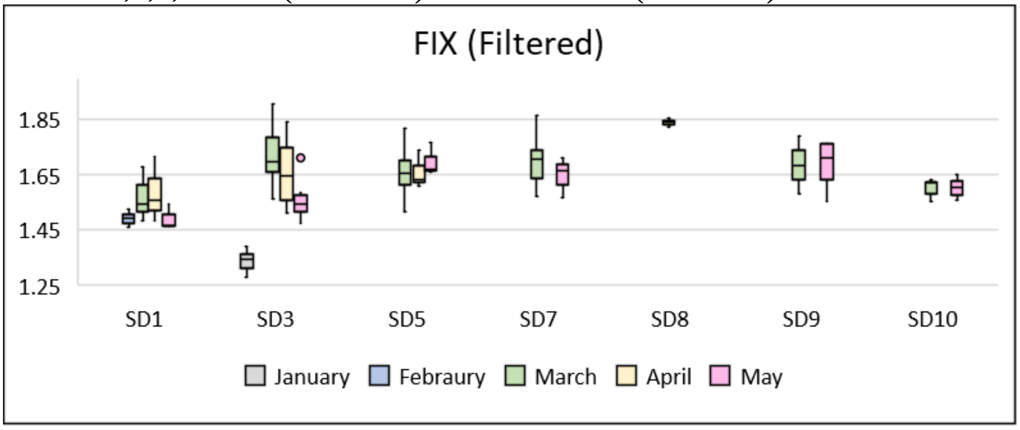

Figure 39 FI for the filtered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), April (yellow, SDs 1–5) and May (pink, All SDs) was expressed as boxplots; The boxplot was drawn including the median value

BIX values (Figure 22) for the filtered samples were not different compared to the unfiltered. The mean value for all the months was 0.6 in SDs 1, 3 and 10, 0.7 in SDs 5, 7, and 9 and 0.8 in SD8. The mean value were ranging from 0.6 in February (allochthonous sources) to 0.7 the rest of the months (mixed between allochthonous and autochthonous sources).

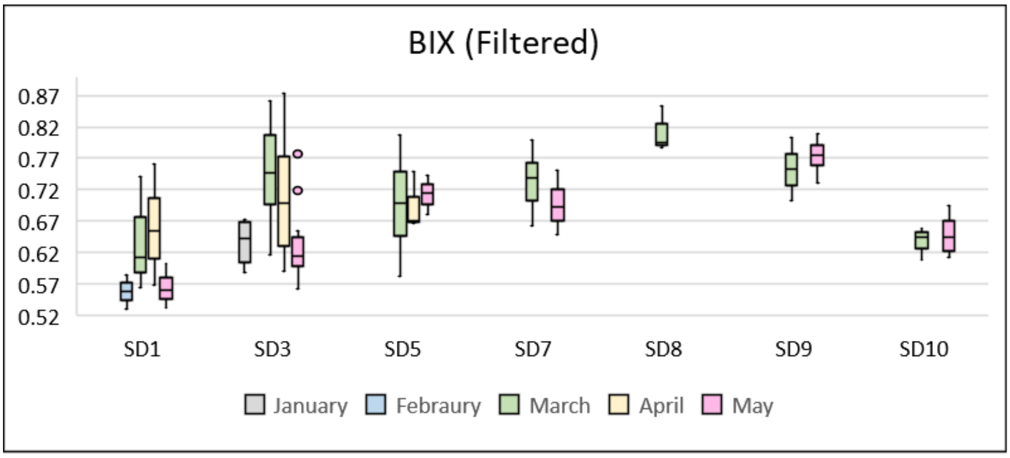

Figure 40 BIX for the filtered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), April (yellow, SDs 1–5) and May (pink, All SDs) was expressed as boxplots; The boxplot was drawn including the median value

HIX (Figure 23) for SD3 did not show as much variation as unfiltered samples with the values just ranging between 0.7 and 0.9. The mean HIX in all the other sites SDs 1, 5, 7, 8, 9, 10 in all the months for the filtered samples were close to 0.9 each, which showed the predominant sites had highly humified DOM sources.

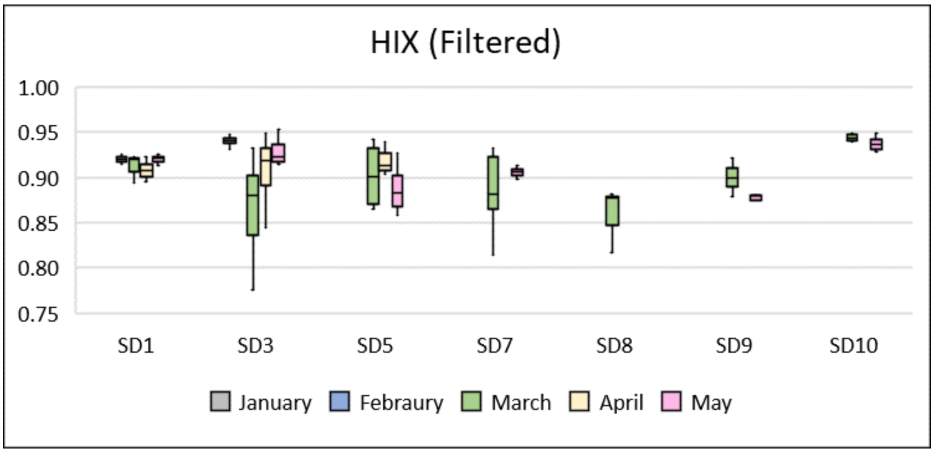

Figure 41 Humification index for the filtered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), April (yellow, SDs 1–5) and May (pink, All SDs) was expressed as boxplots; The boxplot was drawn including the median value

A254 (Figure 24) for both SD3 and SD1 (Mean of 0.3 and 0.5) reduced in comparison to unfiltered samples, suggesting less-aromatic compounds. The other sites SDs 5-10 had low values from March to May (Mean of 0.4 in SD10, 0.2 in SDs 5 and 9, 0.1 in SDs 7 and 8).

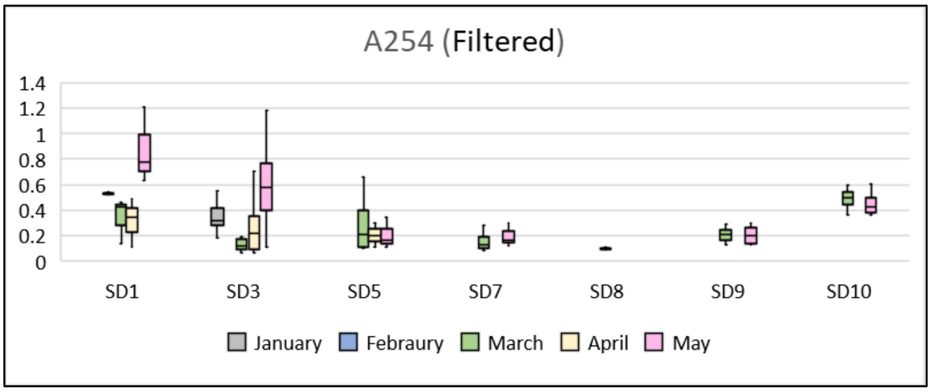

Figure 42 A254 ratio for the filtered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), April (yellow, SDs 1–5) and May (pink, All SDs) was expressed as boxplots; The boxplot was drawn including the median value

S_R (Figure 27) followed a similar pattern to unfiltered samples. SDs 1, 3, 5, and 8, showed high variation in March (ranged from 1.14 to 2.5). When compared between March and May, SDs 5, and 9 had an increase in S_R (decrease in absorption at long wavelength: formation of smaller compounds). SDs 1, 3, 7, and 10 had a decrease in S_R (formation of larger compounds).

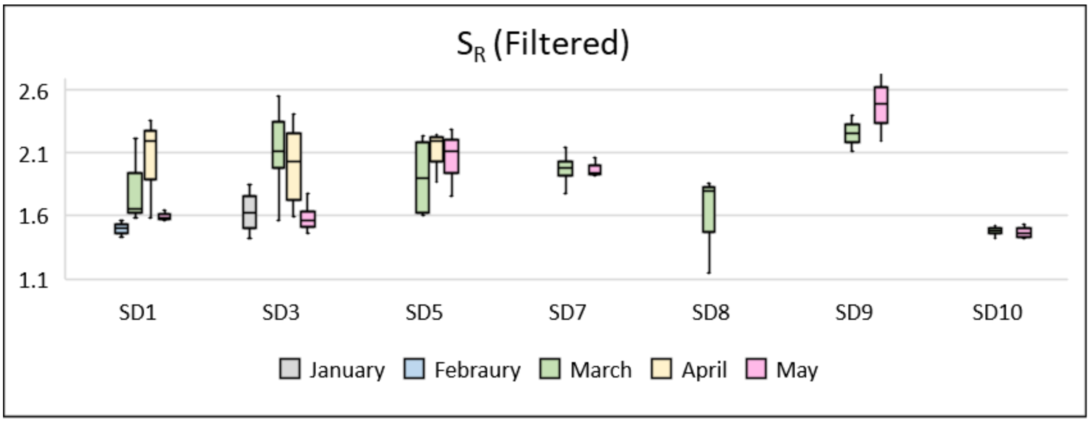

Figure 43 Spectral slope for the filtered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), April (yellow, SDs 1–5) and May (pink, All SDs) was expressed as boxplots; The boxplot was drawn including the median value

E2:E3 ratio (Figure 44 and Figure 45) for the samples was shown separately for the months of January to April and May. The mean values for filtered samples were higher than unfiltered in all the months. Specifically, in May, the variation in SD3 for the filtered samples was high (Ranged between 5.8 and 52.3). SD7 had the highest mean ratio of 37.3 and SD1 was the lowest.

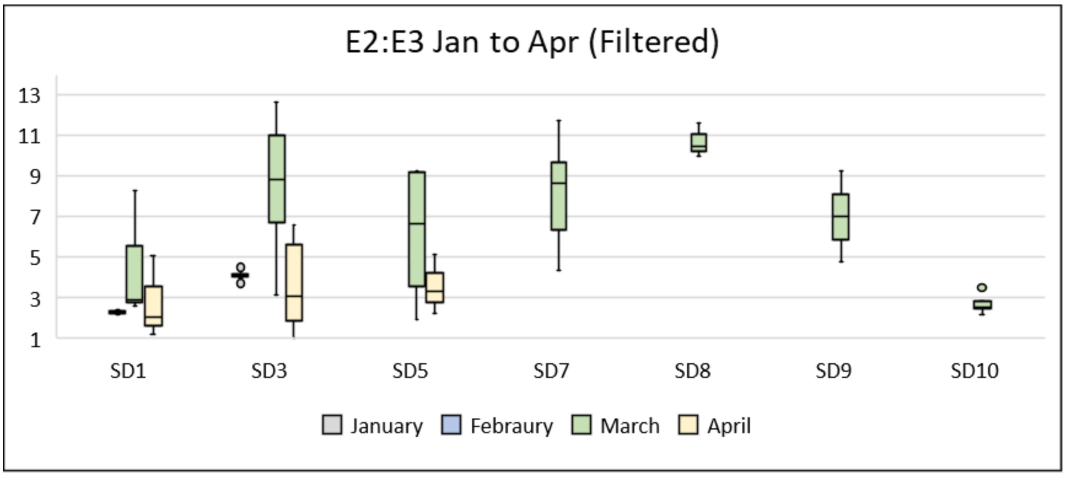

Figure 44 E2:E3 ratio for the filtered samples ratio for the unfiltered samples in January (grey, SD3), February (blue, SD1), March (green, All SDs), and April (yellow, SDs 1-5) was expressed as boxplots; The boxplot was drawn including the median value

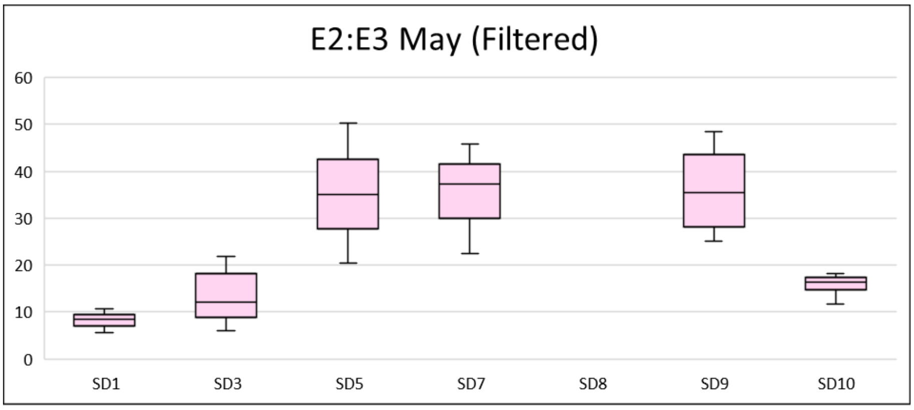

Figure 45 E2:E3 ratio for the filtered samples in May (SDs 1-5) was expressed as boxplots; The ratio for May was shown separately as it had an increased value and required a different scale The boxplot was drawn including the median value

Appendix 3 Correlation between SDs and DOM indices

The Pearson correlations for the nutrients with SS and DOM indices were calculated for all the SDs (Table 13-Table 18).

Table 13 Correlations made for unfiltered and filtered drain water samples in SD1; Significant correlations ($p<0.05$) was marked in bold text; Provided there was significant correlation, the values up to 0.50 were taken as weak correlation, values between 0.50 and 0.75 as moderate correlation and above 0.75 as strong correlation.

SD	*UF/F*	*Index*	*NH_4N (mgl^{-1})*	*NO_3N (mgl^{-1})*	*PO_4P (mgl^{-1})*	*TP (mgl^{-1})*
SD1	UF	FI	-0.25	0.42	-0.15	-0.32
		BIX	-0.18	0.35	-0.19	-0.35
		HIX	-0.17	-0.37	0.24	-0.08
		A254	0.31	0.01	0	0.41
		E2:E3	0.06	-0.38	-0.23	-0.08
		SR	**-0.6**	-0.27	-0.43	**-0.63**
		SS	**0.59**	0.37	0.17	0.42
	F	FI	-0.28	0.28	-0.17	-0.38
		BIX	-0.31	0.18	-0.22	-0.43
		HIX	0.37	-0.05	0.12	0.34
		A254	0.11	-0.32	0.06	0.35
		E2_E3	0.1	0.21	-0.25	-0.11
		SR	-0.42	-0.11	-0.31	-0.5

Table 14 Correlations made for unfiltered and filtered drain water samples in SD3; Significant correlations ($p<0.05$) is marked in bold text; Provided there was significant correlation, the values up to 0.50 were taken as weak correlation, values between 0.50 and 0.75 as moderate correlation and above 0.75 as strong correlation.

SD	*UF/F*	*Index*	*NH_4N (mgl^{-1})*	*NO_3N (mgl^{-1})*	*PO_4P (mgl^{-1})*	*TP (mgl^{-1})*
SD3	UF	FI	-0.18	0.14	-0.2	-0.03
		BIX	-0.12	0.08	-0.32	-0.01
		HIX	0	-0.08	-0.23	**-0.75**
		A254	0.05	0.05	0.38	**0.89**
		E2:E3	-0.03	0.08	-0.11	-0.15
		SR	-0.08	-0.03	-0.32	**-0.7**
		SS	0.02	0.11	0.29	**0.89**

SD	UF/F	Index	NH_4N (mgl^{-1})	NO_3N (mgl^{-1})	PO_4P (mgl^{-1})	TP (mgl^{-1})
	F	FI	-0.17	0.05	-0.33	**-0.44**
		BIX	-0.08	-0.01	**-0.44**	**-0.53**
		HIX	0.15	-0.05	0.36	0.39
		A254	0.07	-0.15	**0.46**	**0.49**
		E2_E3	-0.08	0.28	-0.15	0.01

Table 15 Correlations made for unfiltered and filtered drain water samples in SD5; Significant correlations ($p<0.05$) is marked in bold text; Provided there was significant correlation, the values up to 0.50 were taken as weak correlation, values between 0.50 and 0.75 as moderate correlation and above 0.75 as strong correlation.

SD	*UF/F*	*Index*	*NH_4N (mgl^{-1})*	*NO_3N (mgl^{-1})*	*PO_4P (mgl^{-1})*	*TP (mgl^{-1})*
SD5	UF	FI	-0.02	-0.19	-0.58	-0.4
		BIX	0.09	-0.21	-0.56	-0.36
		HIX	0.13	-0.2	0.01	-0.26
		A254	-0.19	0.56	0.46	**0.73**
		E2:E3	-0.18	-0.26	-0.33	-0.24
		SR	0.16	-0.62	-0.17	-0.58
		SS	-0.16	**0.67**	0.32	**0.94**
	F	FI	-0.28	-0.29	**-0.57**	-0.61
		BIX	-0.1	-0.41	**-0.64**	**-0.69**
		HIX	-0.19	**0.64**	**0.69**	**0.77**
		A254	0.07	0.46	0.52	**0.75**
		E2_E3	-0.13	-0.22	-0.4	-0.11
		SR	0.05	**-0.83**	-0.53	**-0.87**

Table 16 Correlations made for unfiltered and filtered drain water samples in SD8; Significant correlations ($p<0.05$) is marked in bold text; Provided there was significant correlation, the values up to 0.50 were taken as weak correlation, values between 0.50 and 0.75 as moderate correlation and above 0.75 as strong correlation.

SD	*UF/F*	*Index*	*NH_4N (mgl^{-1})*	*NO_3N (mgl^{-1})*	*PO_4P (mgl^{-1})*	*TP (mgl^{-1})*
SD8	UF	FI	-0.86	-0.92	-0.85	-0.93
		BIX	-0.99	-0.66	-0.99	**-1**
		HIX	0.97	0.36	0.97	0.92
		A254	0.94	0.27	0.94	0.87
		E2:E3	-0.96	-0.78	-0.96	-0.99
		SR	0.93	0.24	0.93	0.86
		SS	0.48	0.99	0.47	0.62
	F	FI	-0.25	-0.93	-0.24	-0.4
		BIX	-0.99	-0.49	-0.99	-0.96
		HIX	0.98	0.42	0.99	0.94
		A254	0.87	0.1	0.88	0.78
		E2_E3	-0.85	-0.07	-0.86	-0.75
		SR	0.95	0.31	0.96	0.89

Table 17 Correlations made for unfiltered and filtered drain water samples in SD9; Significant correlations (p<0.05) is marked in bold text; Provided there was significant correlation, the values up to 0.50 were taken as weak correlation, values between 0.50 and 0.75 as moderate correlation and above 0.75 as strong correlation.

SD	*UF/F*	*Index*	*NH_4N (mgl^{-1})*	*NO_3N (mgl^{-1})*	*PO_4P (mgl^{-1})*	*TP (mgl^{-1})*
SD9	UF	FI	-0.76	-0.06	0.25	**-0.99**
		BIX	**-0.85**	0.11	0.46	**-0.89**
		HIX	-0.33	0.67	0.02	-0.35
		A254	**0.85**	-0.19	-0.39	**0.88**
		E2:E3	-0.63	-0.8	0.5	-0.39
		SR	0.36	-0.68	-0.11	0.3
		SS	0.73	-0.4	-0.43	0.57
	F	FI	**-0.9**	0.16	0.51	**-0.84**
		BIX	**-0.92**	-0.11	0.58	**-0.85**
		HIX	0.72	0.55	-0.47	0.72
		A254	**0.86**	-0.16	-0.39	**0.9**
		E2_E3	-0.67	-0.78	0.52	-0.44
		SR	0.54	-0.62	-0.12	0.63

Table 18 Correlations made for unfiltered and filtered drain water samples in SD10; Significant correlations (p<0.05) is marked in bold text; Provided there was significant correlation, the values up to 0.50 were taken as weak correlation, values between 0.50 and 0.75 as moderate correlation and above 0.75 as strong correlation.

SD	*UF/F*	*Index*	*NH_4N (mgl^{-1})*	*NO_3N (mgl^{-1})*	*PO_4P (mgl^{-1})*	*TP (mgl^{-1})*
SD10	UF	FI	**0.69**	-0.43	-0.25	0.34
		BIX	0.61	-0.3	-0.31	0.21
		HIX	**-0.69**	0.56	0.38	-0.23
		A254	-0.07	0.08	0.61	0.43
		E2:E3	-0.06	-0.52	0.25	0.03
		SR	-0.3	0.18	0.55	0.14
		SS	0.53	-0.47	-0.49	0.03
	F	FI	0.6	-0.28	-0.35	0.21
		BIX	0.6	-0.42	-0.31	0.2
		HIX	-0.38	0.63	0.24	-0.03
		A254	-0.1	0.16	0.6	0.4
		E2_E3	-0.04	-0.53	0.18	-0.01
		SR	-0.07	-0.02	0.42	0.25