



Riparian Plant Biodiversity and Community Composition of Ditches and Straightened Streams

Identifying Key Environmental Drivers in a Boreal
Forest Setting

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Riparian plant biodiversity and community composition of ditches and straightened streams. Identifying Key Environmental Drivers in a Boreal Forest Setting

Biodiversitet och artsammansättning av växter i den kantnära zonen hos diken och modifierade bäckar -

Identifiering av underliggande miljöfaktorer i ett borealt skogslandskap

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Abstract

Modified ecosystems are typically seen as degraded and having no ecological value. In many cases, the modification of the ecosystem was done more than 100 years ago and may have had time to recover and become a refugia for species that were once more common in the landscape. Ditches and straightened streams are one such ecosystem that are at risk of being destroyed through forest management practices like ditch cleaning and poor riparian buffer practices. The overall aim with this study was to get new insights in the poorly explored topic of the ecology of boreal forest ditches and straightened streams. With this study, I start filling a knowledge gap regarding the importance of these artificial waterways, by analysing riparian plant biodiversity (i.e., species richness and Hill-Shannon diversity) and community composition. New knowledge gained from this study can help when deciding on how to manage these systems in the future.

In the Krycklan Catchment Study, close to Vindeln, Sweden, I surveyed ground cover vegetation in plots on 0cm, 20cm and 40cm elevations from the water surface along sixteen ditches and straightened streams. These were on till soil across a catchment area size gradient (0.5 – 50 ha), which allowed me to study what effect catchment area has on riparian plant biodiversity and community composition. I included six small ditches (< 5 ha) on peat soil to study differences in species richness and community composition between soil types.

I hypothesised that the species richness, Hill-Shannon diversity, and community composition would all depend on catchment area, soil type, elevation above the water surface and if the waterway was a ditch or a straightened stream. The species richness of ditches and straightened streams was significantly related to catchment area size while the Hill-Shannon diversity was not. Species richness was significantly higher in plots on till soil on 40 cm elevation than plots on peat soil on 40 cm elevation. Furthermore, Hill-Shannon diversity was significantly lower in plots on 0 cm elevation on till soil than plots on 0 cm elevation on peat soil. Riparian plant community composition was significantly different between ditches and straightened streams and between soil types. Catchment area, carbon – nitrogen ratio, proportion of litter and large wood were the variables driving the composition.

I suggest that the underlying factors driving the differences in biodiversity and plant community composition in ditches and straightened streams are a combination of hydrological (related to catchment area) and edaphic. The variation in biodiversity and community composition in these systems across soil types and catchment areas needs to be acknowledged and carefully considered when planning for future management.

Keywords: Artificial waterways, riparian zone, novel ecosystems, forestry, ditch network maintenance, species richness, hill-diversity, drainage ditches, coarse woody debris, ditch cleaning, ecology.

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Abbreviations

CA	Catchment Area
LW	Large Wood
DNM	Ditch Network Maintenance
SMO	Strategic Management Objective
HSDI	Hill-Shannon Diversity Index
KCS	Krycklan Catchment Study
LOI	Loss On Ignition
WFD	European Water Framework Directive

1. Introduction

Much of the landscape in Sweden has been altered by the digging of drainage ditches and the straightening of streams with the goal to increase forest production. Close to 1 million kilometres of ditches and straightened streams exist throughout the country and over half of these artificial waterways are forest ditches (Paul et al. 2023). Yet, the importance of these modified waterways as refugia for wetland species, remains poorly understood. Deeper insight into the value of drainage ditches and straightened streams for biodiversity is critical for making future decisions about the management of these extensive, yet understudied novel ecosystems (Hobbs et al. 2013, Koschorreck et al. 2020).

1.1 Background

1.1.1 History

Drainage ditches in Sweden have transformed the boreal landscape since the 1840's (Norstedt et al. 2021). However, in southern Sweden, drainage ditches have been dug since the middle of the sixteenth century (Jakobsson 2013). Laudon et al. (2022) estimated the total length of the channel network in Sweden to approximately 1.2 million km, a majority being human-made (67%). Most of these human-made channels are forest ditches (56%), dug to drain wetlands and littoral areas along streams and lakes. Natural streams have also been straightened (19%) to increase flow capacity (Paul et al. 2023). In the second half of the nineteenth century and early twentieth century, drainage ditches were dug to make mires arable to produce hay and crops. During this time, the Swedish government introduced subsidies for ditching projects, payments for these kinds of ditching projects were made until the 1930's (Norstedt et al 2021). In the early 1900's, the forest industry in Sweden began to expand and in an effort to increase wood production on peatlands and wetlands, forest ditching began. Forest ditches were dug during most of the twentieth century, until a requirement for a permit to dig new ditches was added in the nature conservation law in 1986 (1964:822) (Naturvårdsverket 2023), and ditching declined to practically nothing (Hånell 1990). Today, ditches are often cleaned to keep the intended effect on forest productivity (i.e., Ditch Network

Maintenance (DNM)), no permit is required for this type of management (Bergquist et al. 2016).

1.1.2 Ditching Objectives and Consequences

Historically, the primary objectives of digging new forest ditches have been to increase the timber growth in wet forests and to promote regeneration on clearcuts experiencing secondary paludification (Lõhmus et al. 2015). Today, few new ditches are being dug in Sweden (Hånell 1990), instead DNM is done mainly after clear-cutting to counteract rises of the ground water level to promote regeneration of trees (Sikström & Hökkä 2015). A high water table affects respiration, photosynthesis, protein synthesis, and the presence of mycorrhizal symbionts negatively, resulting in an impeded tree growth (Kozłowski 2002). Ditching lowers the groundwater table, thus increasing the gas exchange between soil and air, enabling previously impaired tree roots exchange gasses more easily (Sikström & Hökkä 2015). Tree growth increases due to deeper rooting, less mortality (Kozłowski 1982), and greater nutrient availability through the faster decomposition of organic matter (Peacock et al. 2021).

While drainage through ditching, in many cases improves tree growth, it has negative effects on wetland ecosystems such as mires (Paal et al. 2016). Over 300 species in Sweden are threatened due to ditching of mires and forests (Bernes 1994). Studies from Italy, Estonia and Denmark have shown less diverse riparian plant communities in artificial waterways than natural ones (Bolpagni & Piotti 2016; Riis & Sand-Jensen 2001; Remm et al. 2013). Some ecosystem features may not recover at all after draining, such as bulk density of the soil (Holden et al. 2006). According to Holden et al. (2006), ditching has resulted in flashier hydrographs, resulting in a shorter connectivity between land and water. This shorter connectivity has had implications on the plant and animal species that live there (Hasselquist et al. 2018). Additionally, the erosion of suspended solids causes water quality problems (Nieminen et al. 2018). The biogeochemical and ecological consequences of forest ditching discussed above deserves more attention in the scientific community (Koschorreck et al. 2020), and has not been explored in a Swedish context.

1.1.3 Riparian Zones and Their Vegetation

The riparian zone is the land constituting the border between the aquatic and terrestrial environment (Naiman & Décamps 1997). This zone is heterogenous, expressed in a wide array of life-history strategies and successional stages. This makes it difficult to precisely delineate the spatial extent of the riparian zone but generally it covers the area from the low water mark to the terrestrial upland where vegetation may be affected by elevated water tables or flooding (Naiman &

Décamps 1997). These sometimes diffuse riparian zones are seen as key components for regulating linkages between the aquatic and terrestrial systems and are hotspots for biodiversity with a disproportionately species rich flora compared to surrounding ecosystems (Naiman & Décamps 1997). The riparian zones of headwaters, many of which are forest ditches and straightened streams in Sweden (Paul et al. 2023), are different from larger streams in a number of ways that shape a unique biological community (Richardson & Danehy 2007). One key feature differentiating the riparian zones of many headwaters and larger streams is the lack of fish and thus a different food-web. Another key feature is the steep banks of headwaters leading to little terrestrial land in direct contact with water and thus a lack of truly hydrophilic vegetation (Richardson & Danehy 2007). The ecological, hydrological and physical aspects of riparian zones of headwaters and larger streams are all interrelated (Ledesma et al. 2018). Riparian zones around headwaters are an especially important component of the landscape when smaller waterways make up as much as 90% of the total stream network length (Bishop et al. 2008).

In the riparian zone of waterways in boreal forests, there are distinct vegetation belts on different elevations from the water surface (Ström et al. 2011). This elevational gradient exists due to the flooding tolerance and moisture requirement of different species (Johansson & Nilsson 2002). Ström et al. (2011) found that species richness was higher on mid- and high elevations compared to plots on low elevations (90 cm of elevational difference). There are several factors influencing the vegetation community in these vegetation belts in riparian zones (Luke et al. 2007), such as pH that has a significant effect on vegetation near the water surface (Ives et al. 2011). The work by Luke et al. (2007) supports the idea that hydrological factors can have a substantial impact on the vegetative community, such as the positive correlation between maximum stream power and maximum water level range to species richness (Kuglerová et al. 2015). Additionally, Kuglerová et al. (2015) found that species richness in the riparian zone is positively correlated with the catchment area (CA) size of the waterway (CA meaning the area of land which water drains from). Edaphic factors are also key drivers on the plant community composition of riparian zones (Luke et al. 2007), such as the amount of the stable isotope ^{13}C , sedimentation (Kuglerová et al. 2015) and the soil carbon to nitrogen ratio (C/N) (Eskelinen et al. 2009). Furthermore, the ability of the soil to hold water is important in determining the distribution of vegetation (Naiman & Décamps 1997; Zelnik & Čarni 2008; Moeslund et al. 2013). The distribution of plants from the waterway to upland is affected by sediment grain size and microtopography which coincide with the soil conditions, especially the soil water balance (Knapp et al. 2002; Piedallu et al. 2013). Thus, the plant community composition in riparian zones has been shown to differ between till and peat soils (Mc Conigley et al. 2015).

More factors affecting the plant community composition of riparian zones are light, the amount of coarse woody debris and the proportion of litter (Szaro 1990; Harmon et al. 1986; Lenssen et al 2020). Henceforth, I will refer coarse woody debris as large wood (LW) to remove the negative connotation of the word “debris”. Regarding the effect of light on the vegetation of the riparian zone, there are opposing opinions. Studies by MacDougall & Kellman (1992) and Pollock et al. (1998) showed no relationship between seedling densities and diversity to light intensity, suggesting low effects on the vegetative community, while Szaro (1990) explained differences in species composition along west-flowing and east-flowing streams with variations in solar radiation input. LW as an environmental factor affecting the vegetation of riparian zones is often overlooked but is still important (Harmon et al. 1986). Harmon et al. (1986) writes that LW serves both as autotrophic and heterotrophic habitat and is continuously added to stream and forest ecosystems, including the riparian zone. It has been shown that LW is involved in nutrient cycling in many ecosystems and that it is a functional component in streams and forest ecosystems (Harmon et al. 1986). The accumulation and proportion of litter is higher in riparian zones than elsewhere, contributing to the heterogeneous habitat needed to host a diverse flora and fauna community (Xiong & Nilsson 1997). Litter cover with a thickness between 2 and 5 cm is an environmental variable related to the community composition and explains over half of the variation in species richness within certain communities (Lenssen et al. 2000). Despite the strong influence abiotic factors have on the plant community there are also biotic factors that function as drivers. Competition, herbivory, and diseases are shaping the community of riparian zones, a competitive hierarchy exists, albeit reduced due to frequent abiotic disturbances (Naiman & Décamps 1997).

There are indications that there could be different environmental factors affecting the plant community composition depending on the type of waterway. Pielech et al. (2015) showed that environmental factors explaining variations in floristics differed between streamside forests and forests influenced by groundwater discharge (i.e., spring forests). Stream power, distance from source and altitude are the main driving factors in streamside forests, while altitude and solar radiation are the main driving factors in spring forests (Pielech et al. 2015).

1.1.4 Riparian Zones and Vegetation of Ditches and Straightened Streams

Ecosystems are shaped by disturbances over long periods of time, creating ecological legacies and memories of past ecosystem states (Johnstone et al. 2016). A resilience to these historical disturbances is build up, meaning that the essential structures or properties of the ecosystems will recover (Holling 1973). However, with the introduction of novel disturbances the ecosystems can be triggered to be

reorganized into new ecosystem states (Johnstone et al. 2016). Whether or not the ecosystems recover from novel disturbances they can be seen as either historical, where the biota and ecosystem properties have stayed more or less the same throughout history, or novel, where the system has been transformed from its historical state (Hobbs et al. 2013). Hobbs et al (2013) acknowledge that a hybrid between the two systems may exist, where historical characteristics are retained but the composition or function of the system lies outside the historical variability. The ecosystem properties of ditches and straightened streams, such as instream habitat, riparian connectivity, hydrochemistry and nutrient cycling are significantly different from natural streams (Rosenvald et al. 2014), arguably caused by ditching and the straightening of streams as novel disturbances. In the boreal forest of Sweden, ditches were dug and streams were straightened approximately 100 years ago, and some were cleaned 25-30 years after that (Hånell 1990). This means that ditches and straightened streams have had a century to recover past ecosystem properties, making them historical ecosystems. Alternatively, they would not have had time to recover, suggesting them to be novel ecosystems. It has been shown that the digging of drainage ditches can create novel ecosystems, such as drained forests (Remm et al. 2013). Remm et al. (2013) showed that drained forests fail to host site specific species but can provide valuable habitat for less site-specific species. Additionally, drainage ditches themselves can provide suitable habitat for the movement of aquatic and riparian species (i.e., corridors) (Löhmus et al. 2015). The unique chemical and hydrological environment together with a different management regime of ditches than streams can contribute to overall biodiversity of an area (Armitage et al. 2003). Hobbs et al. (2013) stressed the need for more research regarding different novel ecosystems to gain knowledge on how to manage these in the future.

The importance of the riparian zone around natural waterways is unequivocal. However, studies regarding species composition, abundance and distribution along artificial waterways are scarce (Montanari et al. 2022). It has been shown that diversity and plant community composition differ between natural streams and channelised waterways in Denmark (Baattrup-Pedersen et al. 2005). Additionally, there are some indications that show that artificial waterways can form riparian zones as important for biodiversity as the ones in natural waterways. A study conducted in north-central Colorado, USA by Carlson et al. (2019) show that the functional and taxonomic composition of riparian plant communities in human made irrigation canals and natural streams in an agricultural setting do not differ. Carlson et al. (2019) suggested that canal networks have created a riparian habitat typical of streams, despite hydrological differences. There are also studies pointing out differences between artificial and natural waterways. In floodplains mainly surrounded by forests the biodiversity measures, species richness and Shannon-Wiener index were both higher in an artificial transportation canal than in a natural

nearby natural river due to lateral heterogeneity, while functional divergence and landscape diversity was higher in the natural river (Harvolk et al. 2014). It has been shown that ditches can contribute to the overall biodiversity of a floodplain. The unique conditions of ditches give rise to communities different from natural streams and rivers (Armitage et al. 2003). Harvolk et al. (2014) wrote that an artificial waterway can serve as habitat for single endangered floodplain species, but the functions of a natural floodplain cannot be substituted.

Studies show a need of more research regarding ecosystem, ecological and biological functioning of artificial waterways: including forest drainage ditches and straightened streams. The actual or potential biodiversity values of these waterways need to be recognised (Chester & Robson 2013). The removal of ditches or management with no consideration of the habitat value of ditches are two of the biggest threats to the ecosystem quality of ditch networks (Herzon & Helenius 2008).

1.1.5 Current Policies, Legislation, and Management

The knowledge on the biodiversity in the riparian zones of ditches is poor, especially compared to natural systems such as small streams (Koschorreck et al. 2020). Koschorreck et al. (2020) states that only ~10% of the surface water in Europe is covered by the European Water Framework Directive (WFD), where a substantial proportion of the excluded water systems are likely human made (e.g., ditches or straightened streams). In Sweden only streams with larger CA receive sufficient riparian buffers as protection when forestry operations are made, while many smaller streams completely lack this protection (Kuglerová et al. 2020). More research regarding the significance of human made aquatic systems, including the biodiversity of ditch riparian zone is needed to understand what role they play in the biogeochemical cycle and what additional ecosystem services they can provide (Koschorreck et al. 2020).

Strategic management objectives (SMOs) have been set up by the Swedish forestry agency, academia, operational forestry, non-governmental organizations and other relevant authorities to meet environmental consideration goals and legal requirements of forestry in Sweden (Ring et al. 2022). Parts of the SMO's refer to having a functional forest buffer around streams and lakes to protect the water and riparian zone (Andersson et al. 2013). These forested buffers apply primarily to lakes, perennial streams, and even to straightened streams, but exclude human made ditches (Andersson et al. 2013). In contrast to the SMO's for streams that should maintain ecological functions (Andersson et al. 2013), management guidelines for ditches include DNM that cause dramatic changes in habitat conditions, sometimes adverse such as an increase in sediment transport (Nieminen et al. 2018). Ring et

al. (2022) conclude that most ditches have no protective buffer and to develop more efficient protective measurements for all waterways in Sweden, more knowledge regarding the ecological role of ditches is needed.

1.2 Aim

Riparian biodiversity and plant community composition in natural waterways and what environmental variables are significant for the riparian vegetation are well studied (Dybkjær et al. 2012; Kuglerová et al. 2015), while studies in artificial waterways are scarce (Montanari et al. 2022). The overall aim with this study is to gain knowledge on the previously unexplored topic of riparian plant communities in boreal forest ditches and straightened streams. This research will help fill a knowledge gap regarding the ecology of artificial waterways.

CA and soil type are main drivers of biodiversity and riparian plant community composition in natural streams (Kuglerová et al. 2015; Mc Conigley et al. 2015), and an elevational gradient in species richness from the water surface has been shown (Ström et al. 2011). I studied sixteen ditches and straightened streams along a CA gradient (0.5 ha to 50 ha) on till soil to analyse what effects CA have on biodiversity (i.e., species richness and Hill-Shannon diversity index (HSDI)). As 50 % of ditches and straightened streams are on peat soil (Ågren et al. 2024), I included six additional small (< 5 ha) ditches on peat soil, which I then could compare to ditches on till soil with similar CAs (< 5 ha) to analyse what effect soil type has on biodiversity and the riparian plant community composition. To analyse how CA affects riparian plant community composition, the ditches and straightened streams were divided in three size categories (small, medium, and large). Furthermore, I sought to analyse how different edaphic factors and habitat characteristics such as C/N, pH, litter and LW affects community composition in ditches and straightened streams when these have been shown to be significant factors in natural waterways (Harmon et al. 1986; Lenssen et al. 2000; Eskelinen et al. 2009; Ives et al. 2011). New knowledge gained from this study can be used in future research and when deciding how ditches and straightened streams in boreal forests should be managed in the future.

1.3 Research Questions and Hypotheses

1. Does CA and elevation above the water surface affect riparian plant biodiversity (i.e., species richness and HSDI) of ditches and straightened streams on till soil?
 - A. Previous studies on natural streams show that riparian plant species richness is positively correlated with CA (Kuglerová et al. 2015; Mc Conigley et al. 2015). I hypothesize that the same patterns will be shown for riparian plant species richness and the HSDI along ditches and straightened streams. Additionally, plots on higher elevations will have a higher species richness and higher HSDI than plots on lower elevations due to a gradient in soil moisture (Ström et al. 2011).
2. Is riparian plant biodiversity (i.e., Species richness and HSDI along ditches with small (<5 ha) CAs different between peat and till soil and across elevations from the water surface?
 - B. For ditches and straightened streams with small (<5 ha) CAs I hypothesize that that the riparian plant species richness and the HSDI will be generally higher on till soil than on peat soil, similar to what Mc Conigley et al. (2015) found in natural waterways and explained by the lower pH, higher organic matter and higher soil moisture in peat soils. Additionally, I hypothesize that patterns in species richness and HSDI across elevations will be different between peat and till soil as ditches on till soils are more susceptible for erosion (Hasselquist et al. 2018). This erosion will result in more patches of bare soil closer to the water surface in till ditches, thus affecting the vegetation.
3. Does the riparian plant community composition differ between waterways of different CAs (i.e., small, medium, and large), soil type and type of waterway (i.e., ditch or straightened stream) and what edaphic factors and habitat characteristics are significant variables driving the composition?
 - C. I hypothesize that the riparian plant community composition will be significantly different between ditches and straightened streams, this difference will depend on the soil type and CA of the waterway. Peat soils will have an unique environment, with low pH, high organic content and high soil moisture throughout the riparian zone where only a few species are dominant (Mc Conigley et al. 2015). Hydrological differences between small and large CAs such as stream power and maximum water level range (Kuglerová

et al. 2015) will create significantly different community compositions. Furthermore, I hypothesize that LW, litter and C/N will be significantly related to the riparian plant community composition. Previous studies have shown that these factors are significant variables on species composition and biodiversity (Harmon et al. 1986; Lenssen et al. 2000; Eskelinen et al. 2009).

2. Method and Material

2.1 Study Area

Sites were selected around forest ditches and straightened streams located in the Krycklan catchment study (KCS) area close to Vindeln, Sweden, approximately 60 km from the coast of the Baltic Sea (64°14'N, 19°46'E) (Figure 1). The KCS offer a well-developed field research infrastructure with good maps (Laudon et al. 2013). This is an area of boreal forest dominated by Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*) with an understory of a variety of ericaceous shrubs, mostly cowberry (*Vaccinium vitis-idaeus*) and bilberry (*Vaccinium myrtillus*) (Laudon et al. 2013). Besides the dominant conifer forest, patches of deciduous trees such as birch (*Betula pubescens*), alder (*Alnus incana*), aspen (*Populus tremula*) and willows (*Salix spp.*) are found in the more mesic-wet riparian areas. The landscape features patches of mires, drained by a network of rivers, streams and ditches (Laudon et al. 2013). The elevation in the area ranges between 114 to 405 meters above sea level, at the higher altitudes, the soil is dominated by till, while at the lower altitudes it is dominated by sedimentary deposits (Laudon et al. 2013). The organic content increases closer to the streams, forming a riparian peat zone. The climate of the study area is continental subarctic climate with cold summers, with a mean annual temperature of 1.8 Celsius. The average precipitation is 614 mm which of 40 % falls as snow, resulting in a hydrology driven by snowmelt with peak flows during spring (Laudon et al. 2013).

Study area

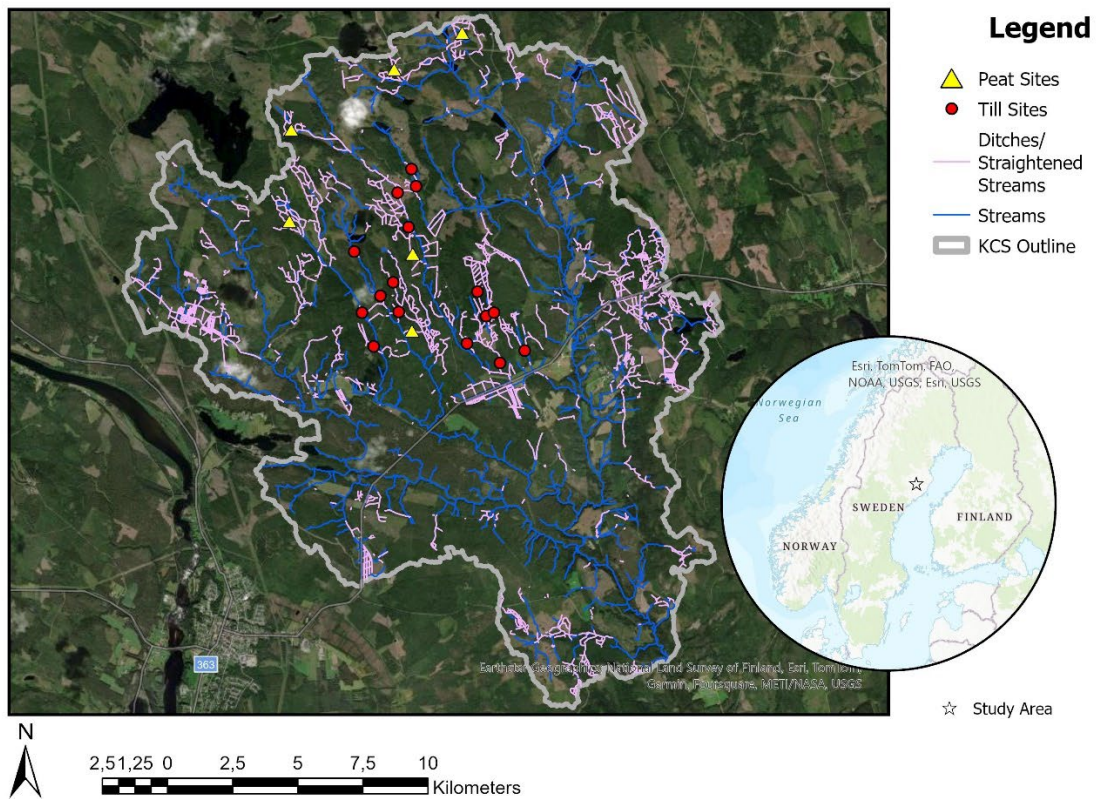


Figure 1. Study area within the KCS.

2.2 Site Selection

Sixteen sites along ditches and straightened streams of varying CAs (0.5 ha – 50 ha) on till soil were surveyed in the autumn of 2023 to study what effect CA have on species richness and HSDI. These were later divided into three size categories to explore differences in riparian plant community composition in waterways with different CAs. Small waterways were defined with CAs under 5 ha, medium waterways were defined with CAs between 5 and 20 ha and large waterways were defined with CAs larger than 20 ha. Six additional peat ditches under 5 ha were included to compare what effect soil type has on species richness and HSDI (Table 1). These small peat ditches were compared with small ditches on till soil. It was important to make this comparison between soil types as 50 % of ditches are on peat soil (Ågren et al. 2024). Ditches and straightened streams within KCS, along with their CA (the area of land water is drained from) were mapped by Paul et al. (2023). Paul et al. (2023) separated ditches from streams by the degree of straightness, the presence of mounds on the side of the channels and how smooth

they appeared in a digital elevation model (DEM). Streams were more meandering and gave a more “fuzzy” impression in the DEM (Paul et al. 2023). A large proportion of ditches in the KCS was found to never support flow, Hasselquist et al. (2018) suggested proportions between 25-50 % and that these were likely on peat soils. In this study, ditches and straightened streams were defined according to methods by Paul et al. (2023). A waterway located near or in a prehistoric channel was defined as a straightened stream, while a waterway that did not overlap a natural stream channel was considered a ditch. In this study, ditches are on both peat and till soil, while the straightened streams only are on till soil. Further, the CA of the ditches are all small, except for one that has a medium CA. All the straightened streams have medium or large CAs (Table 1).

*Table 1. Showing all sites in the study, and which sites were used to answer which research questions. SS denote straightened stream. * Denote the peat ditches that were added to make comparisons between soil types.*

Site Name	Catchment Area (ha)	Size Category	Soil Type	Waterway Type	Research Questions
10Ha_D_Peat_S1*	1.6	Small	Peat	Ditch	2,3,4
10Ha_D_Peat_S2*	0.6	Small	Peat	Ditch	2,3,4
10Ha_D_Peat_S3*	0.7	Small	Peat	Ditch	2,3,4
10Ha_D_Peat_S4*	0.7	Small	Peat	Ditch	2,3,4
10Ha_D_Peat_S5*	0.6	Small	Peat	Ditch	2,3,4
10Ha_D_Peat_S6*	0.7	Small	Peat	Ditch	2,3,4
10Ha_D_S1	2.2	Small	Till	Ditch	1,2,3,4
10Ha_D_S2	1.8	Small	Till	Ditch	1,2,3,4
10Ha_D_S3	5.5	Medium	Till	Ditch	1,3,4
10Ha_D_S4	3.0	Small	Till	Ditch	1,2,3,4
10Ha_D_S5	1.4	Small	Till	Ditch	1,2,3,4
10Ha_D_S6	1.2	Small	Till	Ditch	1,2,3,4
30Ha_D_S1	24.3	Large	Till	SS	1,3,4
30Ha_D_S2	12.1	Medium	Till	SS	1,3,4
30Ha_D_S3	11.2	Medium	Till	SS	1,3,4
30Ha_D_S4	17.3	Medium	Till	SS	1,3,4
30Ha_D_S5	17.1	Medium	Till	SS	1,3,4
30Ha_D_S6	11.3	Medium	Till	SS	1,3,4
30Ha_D_S7	13.2	Medium	Till	SS	1,3,4
60Ha_D_S1	49.8	Large	Till	SS	1,3,4
60Ha_D_S2	29.2	Large	Till	SS	1,3,4
60Ha_D_S3	48.8	Large	Till	SS	1,3,4

Sites were selected based on the structure of the surrounding forest. Sites were only selected where the overstory was dominated by *Pinus sylvestris*, *Picea abies* and *Betula pubescens*. Using a mapping layer, ditches and straightened streams were only chosen from stands with a stand height over ten meters. This was done to ensure that DNM had not occurred recently and that sites would have a similar time since the last major disturbance (i.e., clear cut).

2.3 Habitat and Vegetation Survey

In each site a 10-meter stretch of the ditches and straightened streams was selected to survey the ground cover vegetation. A 20 cm x 80 cm quadrat plot was placed at 0 cm, 20 cm, and 40 cm in elevation from the water surface. Quadrat plots were established from the height of the current water surface (i.e., the water width) at the time of sampling. Three quadrats were placed on the right, at 1 meter, 4 meter and 8 meter and three quadrats were placed on the left at 2 meter, 5 meter, and 9 meter (Figure 2). In the ground cover survey, the percentage coverage of LW, bare soil and litter was estimated to gain habitat information. Furthermore, plant species, including bryophytes, sphagnum, lichen, and tree seedlings were estimated to determine biodiversity and species composition. Three soil samples were taken along each side of the waterway at 20 cm and 40 cm elevation from the water surface to get soil pH and carbon to nitrogen ratio (C/N) and soil organic content. All the 20 cm and all the 40 cm samples from the respective side were collected in one bag each, resulting in a total of 4 sample bags per site (20 cm left, 20 cm right, 40 cm left and 40 cm right).

To further explore the riparian habitat of ditches and straightened streams and analyse how habitat characteristics affect species composition the following measurements were done: The slope of the waterway was measured using an inclinometer, while the bank slope of the riparian zone was calculated using trigonometry. Waterway characteristics measured were total width, high water width, water width, total depth, high water depth, water depth (Figure 3). The canopy cover was estimated using a convex spherical densiometer at three points within the channel along the 10-meter stretch. An average overall canopy cover was then estimated per site to get information about light transmission.

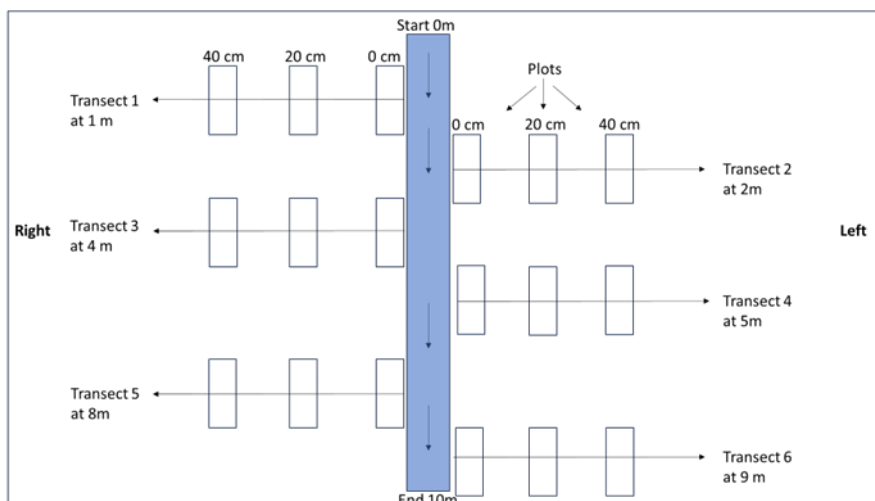


Figure 2. Vegetation plot set up.

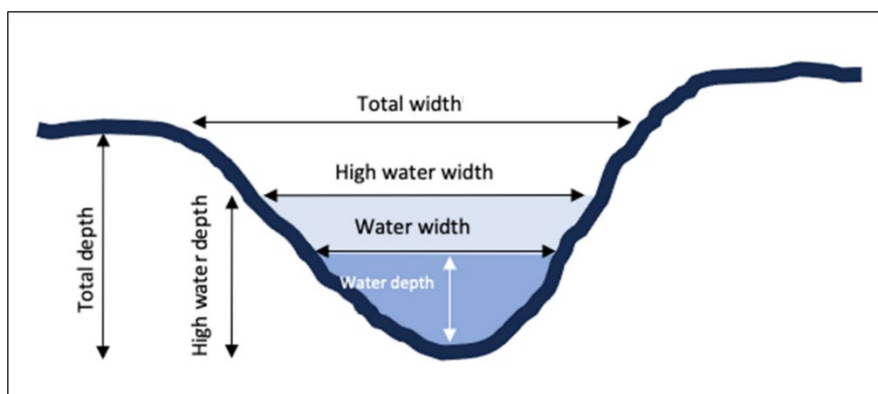


Figure 3. Channel measurements taken at each site.

2.4 Soil Sample Processing and Analysing

The collected soil samples were brought to the lab and put in the drier at 60 degrees for 48 hours before processing. After drying, soils were sieved (2 mm mesh), homogenized, and ground to a fine powder in a rolling mill. To obtain the organic content of the samples a loss on ignition (LOI) procedure was performed (550°C for 4 hours). After burning, when the samples had cooled down to approximately 90 degrees, the samples were transferred from the muffle furnace to a desiccator. Later, the samples were reweighed to obtain the burned weight. The organic content of the soil was calculated by subtracting the burned weight from the dry weight. The weight of the organic content was later divided by the dry weight to obtain the percentage of organic content.

The percentage of organic content was used to calculate sample weight for isotopic analysis. Soil was weighed into tin capsules and were later analysed by the Swedish

university of agricultural science (SLU) Stable Isotope Lab, Umeå, Sweden. The bulk nitrogen (N) and carbon (C) were measured using an elemental analyser (Flash EA 2000) connected to a continuous flow isotope ratio mass spectrometer (DeltaV), both from Thermo Fischer Scientific, Bremen, Germany

2.5 Data Analysis

The biodiversity of ditches and straightened streams was tested with two components, species richness and the Hill-diversity (more specifically the HSDI, a modified version of the original Shannon index (Roswell et al. 2021)). Two components of biodiversity measures were chosen to make the analysis more robust. I chose to include species richness when it is a classic way of measuring biodiversity so I could easily make comparisons with other studies. However, species richness is sensitive to sampling effort and to relative abundance, thus strongly influenced by the rarest species (Roswell et al. 2021). Roswell et al. (2021) writes that because species richness is influenced by the species we know least about, this method of measuring biodiversity should be seen as uncertain. I added the HSDI as a biodiversity measure to include the relative abundance of species, adding robustness to sampling problems related to rare species. According to Roswell et al. (2021) there is a growing consensus that Hill-diversity is the preferred way of biodiversity measurement. This method of measuring biodiversity acknowledges both the number of species and the relative abundance of them, consisting of a single equation that can vary from counting all species equally to focusing on the most common ones. The three forms of Hill-diversity, species richness, Hill-Shannon and Hill-Simpson have the same units and scale intuitively with species gain or loss and only differ in the way they scale rarity (Roswell et al. 2021). I processed and analysed the data using the software program Rstudio, R version 4.3.1. The species richness and the HSDI was calculated using the ```renyi``` function in the `vegan` package (Oksanen et al. 2022) in Rstudio.

A generalised linear mixed model with a ```poisson``` distribution was used to explore differences in riparian plant species richness between ditches of different CAs, if elevation from the water surface affected the richness and the interaction between the two variables (Model 1). The ```poisson``` distribution was selected because it is the bench-mark distribution for counted data in generalised models (McCullagh 2019). All individual plots across sites on till soil was used as replicates. However, these plots are dependent of each other within sites, therefore site (i.e., the individual site) was added as a random effect in the models to deal with the problem of not having true replicates. The generalised model was selected in favour of the linear mixed effects model when the assumption of normally distributed residuals was violated. However, a linear mixed effects model was used

to explore differences in HSDI, depending on CA and elevation from the water surface (Model 2). To meet the assumption of normally distributed residuals in model 2 the HSDI values were square root transformed.

Model 1

*glmer(richness~size * elevation + (1|Site), family = poisson, dat = df*

Model 2

*lmer(diversity~size * elevation + (1|Site), dat = df*

In the models above, I excluded ditches on peat soil because I wanted to remove the influence of soil type and focus the analysis on the effects of CA and elevation.

To test if soil type and elevation had any effect on species richness of small ditches, the ditches on peat soil (under a 5 ha CA) were compared to small ditches on till soils (under a 5 ha CA). A generalised linear model with a ``poisson`` distribution was used (Model 3) in favour of a linear mixed effects model when the assumption of normally distributed residuals was violated. To test if soil type and elevation had an influence on the HSDI of small ditches, a linear mixed effects model was used (Model 4). To meet the assumption of normally distributed residuals in model 4 the HSDI values were log-transformed. In contrast to the square root transformation used for the HSDI values in model 2, log-transformation was the only method showing a normal distribution of HSDI values for this smaller data set, only containing small ditches.

Model 3

*glmer(richness~soil * elevation + (1|Site), family = "poisson", data = df)*

Model 4

*lmer(diversity~soil * elevation + (1|Site), dat = df*

For the models above, richness represents the total number of species present in each plot, size is the CA of the ditch, soil is the type of soil, either till or peat, elevation represents the elevation in cm from the water surface for each plot and diversity is the HSDI. Conditional squared R values (R^2) are presented to show the variance explained by the whole model.

Variation in the riparian plant species composition between size categories (i.e., small, medium, and large), soil type and type of waterway for all sites (Table 1) was explored in non-metric multidimensional scaling ordinations (NMDS) based on a Bray-Curtis distance matrix. CA of ditches and straightened streams was used to define size groups, small < 5 ha, medium 5-20 ha and large >20 ha. To test for

significant differences in community composition between size categories, soil types and types of waterways an analysis of similarities (ANOSIM) was used. I used ANOSIM in favour of permutation tests for multivariate analysis of similarity (PERMANOVA), when PERMANOVA gives up the non-parametric framework and is instead seen as a semi-parametric test in complex designs. ANOSIM is still non-parametric, even with complex designs (Anderson 2005), thus appearing to be a more robust alternative. How the riparian plant community composition varied with other environmental data was tested by fitting the used NMDS with an environmental matrix. The multivariate statistics was performed with the ``vegan`` package (Oksanen et al. 2022) within the software Rstudio, R version 4.3.1.

An alpha value of 0.05 was chosen for significance of all data analysis.

3. Results

3.1 Riparian Plant Floristics

Across twenty-two sites and 396 plots, fifty-seven different taxa were surveyed, including bryophytes, sphagnum species and one species of lichen. Seedlings of *Picea abies*, *Prunus padus*, *Betula pendula* and *Salix spp.* were also surveyed. Below is a table with the 10 most common taxa (Table 2).

Table 2. The ten most common taxa found in vegetation plots sampled along ditches and straightened streams in the Krycklan Catchment.

Taxa	Presence in Plots	Percentage of Plots
<i>Sphagnum spp.</i>	273/396	69
<i>Polythricum spp.</i>	238/396	60
<i>Vaccinium vitis idaea</i>	193/396	49
<i>Vaccinium myrtillus</i>	163/396	41
<i>Hypnales spp.</i>	134/396	34
<i>Equisetum sylvaticum</i>	133/396	34
<i>Mnium spp.</i>	111/396	28
<i>Deschampsia flexuosa</i>	103/396	26
<i>Carex globularis</i>	92/396	23
<i>Linnaea borealis</i>	89/396	22

The ten most uncommon taxa were only surveyed in one plot each, 0.25% of the plots, including the red-listed vulnerable species *Goodyera repens* (SLU art databanken 2020). Almost all the surveyed taxa were represented along ditches and straightened streams on till soil, while only *Salix spp.*, *Rhododendron tomentosum* and *Vaccinium uliginosum* were exclusively surveyed along ditches on peat soil.

3.2 Riparian Plant Species Biodiversity

Riparian plant species richness along ditches and straightened streams was positively correlated with CA ($p = 0.046$) (Figure 4a). Furthermore, species richness was higher on plots 20 cm ($p < 0.001$) and 40 cm ($p < 0.001$) above the water surface compared to plots on 0 cm from the water surface (Figure 5a). HSDI along ditches and straightened streams was not significantly related to CA ($p = 0.076$) (Figure 4b). However, HSDI was higher on plots 40 cm from the water surface than plots on 0 cm and 20 cm ($p < 0.001$ and $p = 0.013$ respectively), and plots on 20 cm had a higher diversity than plots on 0 cm ($p < 0.001$) (Figure 5b).

There were no significant interactive effects between CA and elevation, neither explaining variation in species richness or HSDI. CA and elevation together with site as a random effect were able to explain 27% of the variation in species richness ($R^2 = 0.270$) and 40% of the variation in HSDI ($R^2 = 0.400$).

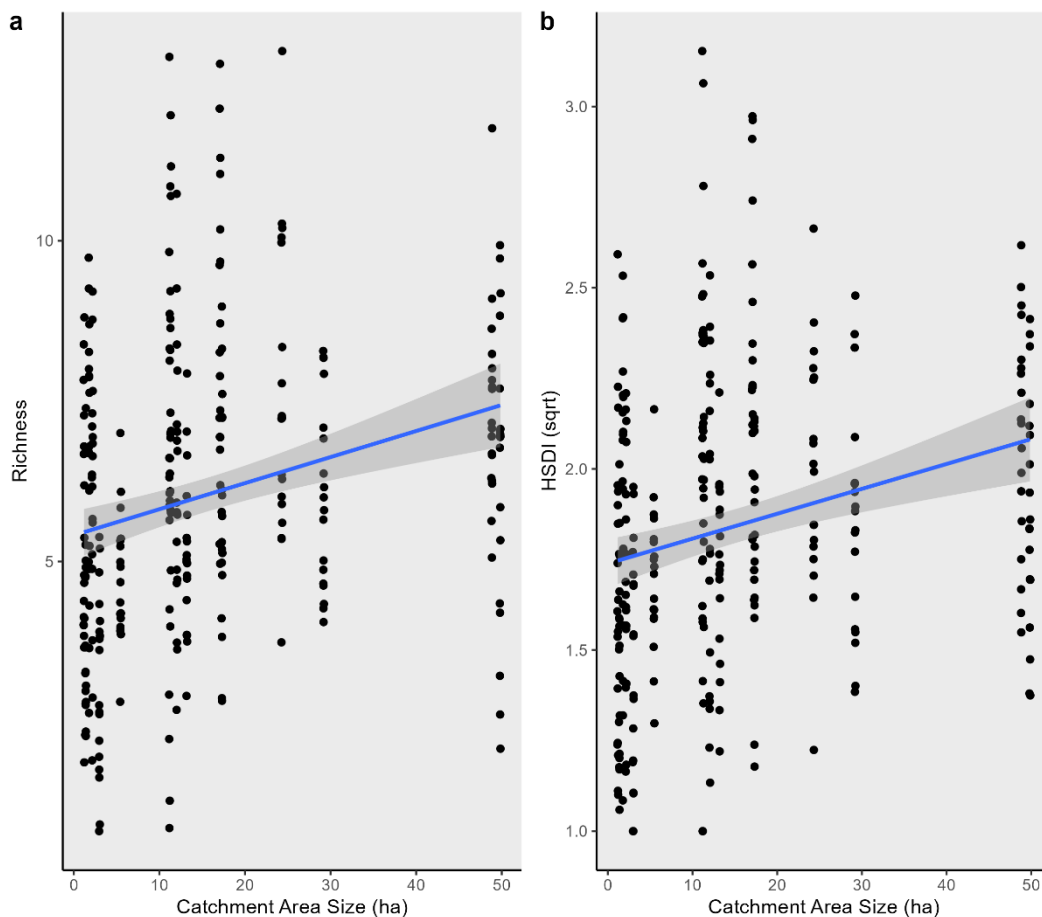


Figure 4. (a) Relationship of riparian plant species richness with CA. The black dots represent individual plots from all sites with till soil. (b) Relationship of the Hill-Shannon diversity index that has been square root transformed with CA. The black dots represent individual plots from all sites with till soil.

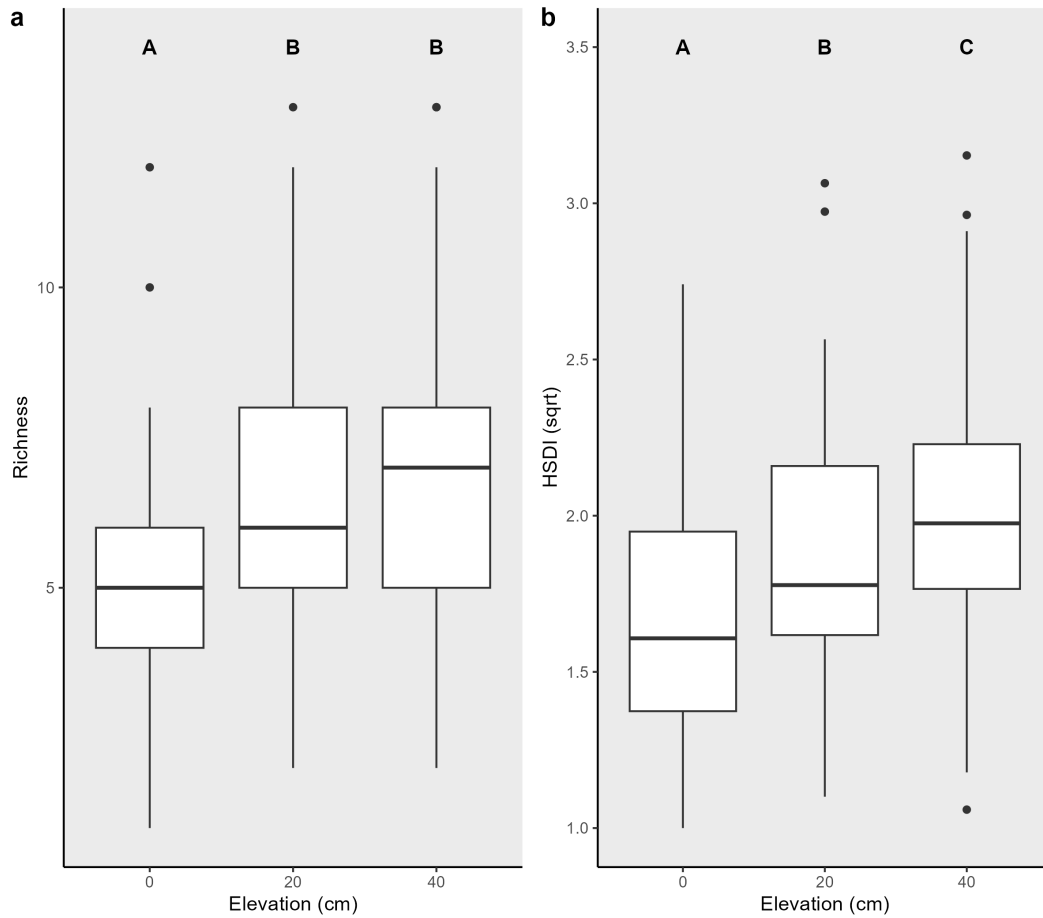


Figure 5. (a) Riparian plant species diversity across elevations within the riparian zone of ditches and straightened streams. Box plots denote median, 25th and 75th percentiles together with potential outliers. (b) Hill-Shannon diversity index that has been square root transformed across elevations within the riparian zone of ditches and straightened streams. Box plots denote median, 25th and 75th percentiles together with potential outliers. Different letters represent significant differences at $p < 0.05$.

Around smaller ditches < 5 ha there were significant interactions between soil type and elevation, riparian plant species richness was higher along ditches on till soil than ditches on peat soil in plots 40 cm from the water surface ($p = 0.031$) (Figure 6a). However, the HSDI was lower in plots on 0 cm elevations on till soil than all other plots (Table 3, Figure 6b).

Table 3. Plots showing significant differences in the Hill-Shannon diversity index across soil type and elevation.

Significant HSDI differences between plots	P-values
Till Soil 0 cm – Peat Soil 0cm	0.045
Till Soil 0 cm – till Soil 20 cm	0.007
Till Soil 0 cm – Peat Soil 20 cm	0.021
Till Soil 0 cm – till Soil 40 cm	<0.001
Till Soil 0 cm – Peat Soil 40 cm	0.040

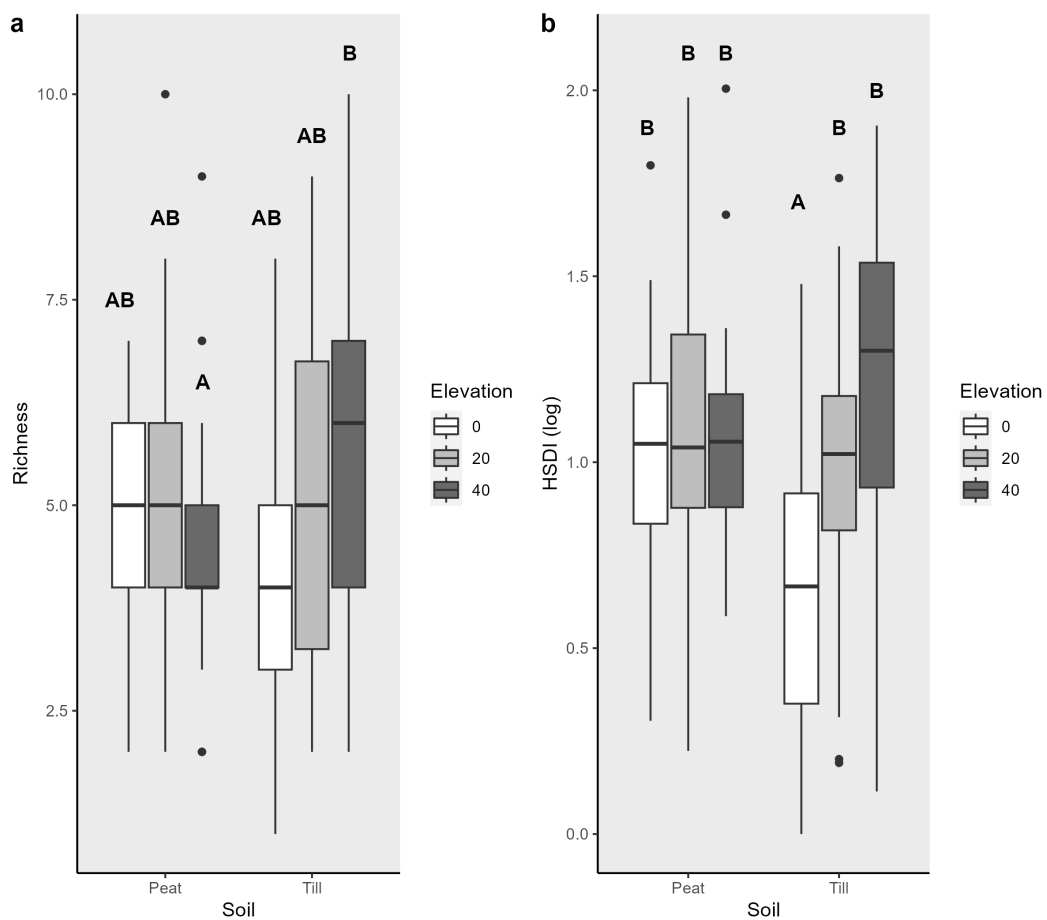


Figure 6. Riparian plant species richness (a) and Hill-Shannon diversity index that has been log-transformed (b) for small ditches on peat and till soil at different elevations. Box plots denote median, 25th and 75th percentiles together with potential outliers. Different letters above the box plots represent significant differences at $p < 0.05$.

3.3 Riparian Plant Community Composition

The riparian plant community composition of ditches and straightened streams was not significantly different between size groups (i.e., small, medium & large) ($R = -0.051$ $p = 0.638$) (Figure 7a). There was a significant difference between the community composition of ditches on different soil types ($R = 0.455$ $p = 0.003$) and waterway type ($R = 0.136$ $p = 0.027$) (Figures 7b & 7c). Furthermore, The proportion of litter ($p = 0.003$), C/N ($p = 0.002$), LW ($p = 0.033$) and CA as a continuous variable ($p = 0.025$) were significantly correlated with the plant community composition (Figures 7a, 7b & 7c).

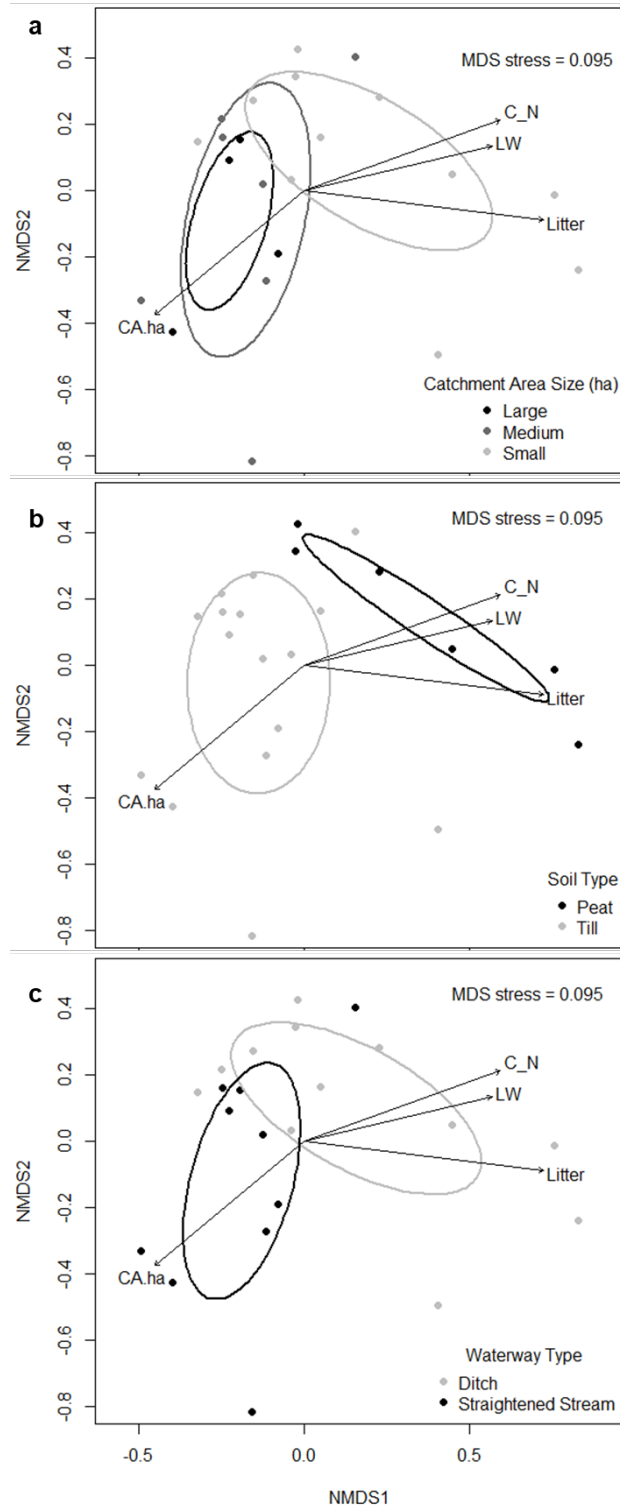


Figure 7 Riparian plant community composition of ditches and straightened streams. Points are mean values for each site and are colour-coded by CA (a), soil type (b) and type of waterway (c). Circles denote the standard deviation of points. Environmental variables that are significantly ($p < 0.05$) correlated with the community composition are shown with arrows (C/N, proportion of litter in plots, size (i.e., CA) as a continuous variable and the proportion LW in plots).

4. Discussion

To summarize the results, I found that the species richness was positively related to CA, similar to what Kuglerová et al. (2015) and Dybkjær et al. (2012) found. However, the HSDI was not significantly correlated with CA. Furthermore, these two biodiversity measures of ditches and straightened streams were positively correlated with elevation above the water surface. For small ditches I found significant interactions between soil type and elevation. Species richness was higher in plots on till soil on 40 cm elevation than plots on peat soils on 40 cm elevation, while the HSDI was lower in plots on till soil on 0 cm elevation than all other plots on both till and peat soil. The riparian plant community composition differed between ditches and straightened streams and depending on what soil type there was. With key environmental characteristics driving the species assemblage (i.e., CA as a continuous variable, proportion of litter, proportion of LW and the C/N). I show that there is variation in biodiversity and species composition of ditches and straightened streams dependent on several environmental factors.

4.1 Biodiversity and Catchment Area

It is well established that community composition is related to the downstream position in river networks (i.e., CA) (Vannote et al. 1980), but the number of studies regarding the ecology of artificial waterways are scarce (Montanari et al. 2022). Kuglerová et al. (2015) and Dybkjær et al. (2012) showed that species richness increased with CA in natural waterways. This is also what I found for ditches and straightened streams, supporting hypothesis (A). Species richness was significantly higher with an increasing CA. That the species richness would be higher in larger CAs was expected and hypothesised, as relationships between riparian vegetation and CA have been found (Dunn et al. 2011). Dybkjær et al. (2012) demonstrated that the riparian vegetation is closely linked with CA, and Kuglerová et al (2015) concluded that the increase in species richness with CA was caused by an increase in species density. The increase of flood disturbance with the size of the waterway, edaphic factors and the accumulation of species downstream as a result of hydrochory are all factors that have been linked to an increasing species density with CA size (Kuglerová et al. 2015). It is possible that these factors are also driving the increase of species richness with CA size found in my study,

even though I studied artificial waterways. However, Kuglerová et al. (2015) studied waterways with CAs ranging from under 1 hectare to 12,000 hectares, while I studied artificial waterways at a small scale (i.e., CAs below 1 hectare to 50 hectare). The same conclusions may not be possible for my study due to the different study designs.

I hypothesized (A) that HSDI would correlate to CA like the correlation between species richness and CA. I found similar positive relationships, but HSDI was not significantly correlated with CA. This opposing result on how species richness and HSDI changes with CA could reflect how well the two different biodiversity measures are working. There is a consensus that Hill-diversity (i.e., HSDI) is better at describing the biodiversity than species richness. Species richness as a biodiversity measure can overestimate the biodiversity gained with size of the waterway due to its leverage to rare species (Roswell et al. 2021). Even though I showed a positive relationship between species richness and CA, care is needed when drawing conclusions about biodiversity gain with CA when I found no significant relationship between HSDI and CA.

Elevation above the water surface had an influence on both species richness and the HSDI, supporting hypothesis (A). Plots higher above the water surface had higher species richness and HSDI, suggesting an increase in biodiversity with elevation from the water surface. Floods, high water tables and inundation create a unique environment near the stream edge (Naiman & Décamps 1997) where only a few specialist species can survive (Kozłowski 2002). At the same time, plots on higher elevations are also affected by these hydrological factors but not to the same extent, relating to the idea of "vegetational belts" (Ström et al. 2011). This, "just right" degree of hydrological disturbance such as flooding and inundation on higher elevations creates an environment where many species can coexist, leading to a high biodiversity and fits in the Intermediate disturbance hypothesis first proposed by Grime (1973). Across elevations I suggest that there is a flooding and moisture gradient driving the difference in species richness and HSDI. Furthermore, I expect that if I had surveyed plots on even higher elevations I would eventually be seeing a decrease in biodiversity, due to a drier environment where competition becomes a major factor affecting the species composition (Zelnik & Čarni 2008).

4.2 Biodiversity and Soil Type

Riparian plant species richness was higher on till than on peat soil along small ditches, but only in plots on 40 cm elevation from the water surface, partly supporting hypothesis (B). Species richness is likely higher in plots on 40 cm elevations on till than on peat soil due to a combination of lower pH, higher soil

wetness and higher organic content in peat soil, influencing what species are found (Mc Conigley et al. 2015). However, it must be acknowledged that several of the non-vascular taxa (e.g., *Sphagnum spp.*, *Hypnales spp.* and *Mnium spp.*) dominated plots on peat soil were not identified at a species level. This is a limitation in the study that likely results in a lower species richness in plots on peat soil. There were no differences in species richness in the mid and low elevations between soil types. Hydrological factors such as maximum stream power and maximum water level range may be additional key factors affecting species richness (Luke et al. 2007; Kuglerová et al. 2015).

HSDI was lower in plots on 0 cm elevations on till than on peat soils and there was no significant difference in plots on 20 cm and 40 cm elevations. Higher proportions of bare soil and rocks in plots closest to the water surface in ditches on till soil than on peat soil is a potential explanation for these results. Johansson & Nilsson (2002) discuss in a similar context, that the vegetation can be affected indirectly when substrate is disturbed by erosion. Additionally, Hasselquist et al. (2018) writes that erosion may be more severe in ditches on till soil than on peat soil, thus leading to more bare patches and less vegetation closer to the water surface. The lack of disturbance measures is a limitation in my study, as this factor has been shown to be important for diversity and community composition (Ström et al. 2011; Kuglerová et al. 2015). The difference in disturbance on low-elevation plots between soil type along ditches and straightened streams remains to be explored.

As I hypothesised (B), there is higher variation in species richness and HSDI across elevations on till than on peat soils. Plots on peat soil have a consistent biodiversity (i.e., species richness and HSDI) across elevations while biodiversity on till soil increases with elevation. Peat soils have often a continuously high water table throughout the riparian zone, while till soils have a periodically high water table (Mc Conigley et al. 2015). This difference in moisture regimes and hydrological disturbances, together with till soils being more susceptible to erosion are likely factors driving this pattern (Piedallu et al. 2013, Ström et al. 2011). Therefore defining riparian vegetation as distinct vegetational belts on different elevations (Ström et al. 2011) around small ditches, is more applicable and apparent on till soil.

4.3 Riparian Plant Community Composition

I confirmed parts of my hypothesis (C), showing a distinct community composition depending on if the waterway was defined as a ditch or straightened stream and if there was till or peat soil. However, I could not support the part of my hypothesis (C), suggesting that community composition would be different between size

categories (i.e., small, medium, and large). Carbon to nitrogen ratio (C/N), proportion of litter and proportion LW are all key environmental variables driving the riparian plant community composition in small ditches on peat soil.

Litter accumulation has been shown to be greater in riparian zones than upland because it is carried by water from upstream, redistributed from upland and is generated in the riparian zone itself (Xiong & Nilsson 1997). This litter affect the riparian plant community, shaping an environment where only certain plants are successful, such as plants with big seeds or plants with vigorous growth (Xiong & Nilsson 1997). Like litter, LW is also transported by water from upstream. Where LW is deposited it serves as both autotrophic and heterotrophic habitat (Harmon et al. 1986). This suggest that litter and LW accumulation may be higher in waterways on peat soil, shaping a unique environment hosting a different riparian plant community than on till soils. Additionally, it is also likely that a moisture gradient between the soil types is a main variable driving the community composition. This fits with previous studies that show different plant communities on till and peat soil, and that it is a mixture of factors affecting riparian plant communities (Mc Conigley et al. 2015). The edaphic properties of peat soils, such as lower pH, higher C/N and higher soil wetness than till soil are all factors that are linked with controlling the community composition (Eskelinen et al. 2009; Ives et al. 2011; Moeslund et al. 2013). Zelnik & Čarni (2008) showed that in drier plots the community had more competitors than stress tolerant species, while in wetter plots there were more stress tolerant species.

Surprisingly, I did not find differences in the riparian plant community composition when arranging the waterways in size categories (i.e., small, medium, and large). However, I found that when using a continuous scale with true CAs it was a significant driver of the community composition for straightened streams. In the small size scale of my study (CAs ranging from 0.5 to 50 ha) there are not distinct riparian plant communities in different size categories, the composition is rather gradually following changes in the habitat. The habitat is progressively shifting downstream with an increase in pH, a decrease in C/N and an increasing disturbance from flooding (Kuglerová et al. 2015). This together with an accumulation of species due to hydrochory gives rise to a gradient in riparian species composition with CA size (Nilsson et al. 2010).

4.4 Conclusions

The two biodiversity measures, species richness and HSDI did not respond to increasing CA of ditches and straightened streams in the same way. CA had only a significant effect on species richness, not on the HSDI. With support from Carlson

et al. (2019), suggesting ecological similarities between artificial and natural waterways, I suggest that the positive relationship between species richness and CA in artificial waterways is explained by the same factors explaining an increase in species richness with CA in natural waterways (i.e., edaphic factors, an increase in hydrological disturbance and species transportation via hydrochory (Dybkjær et al. 2012; Kuglerová et al. 2015)).

With support from previous studies, I conclude that the biodiversity along ditches and straightened streams are primarily driven by a combination of hydrological factors, such as flooding and high water tables (Luke et al. 2007; Moeslund et al. 2013) and soil type (Mc Conigley et al. 2015). In till soils, a gradient in biodiversity across elevations was found which I explain with hydrological factors such as flooding and high water tables. No such gradient was seen in peat soils suggesting that the idea of "vegetational belts" (Ström et al. 2011) is more distinct on till soils.

The riparian plant community composition differs between soil types and if the waterway is a ditch or straightened stream. Additionally, I show that the proportion of litter, the proportion of LW and the C/N are significant drivers of community composition in small peat ditches, while CA as a continuous variable is a significant driver of community composition in straightened streams on till soil. With these findings showing variability between these waterways, I suggest that ditches and straightened streams cannot be seen as trivial systems that all can be treated the same way. These heterogenous systems must rather be evaluated one by one when deciding on how to manage them. Future studies in this topic are needed to further explore the communities that are linked with ditches and straightened streams on different soil types and CAs.

Löhmus et al. (2015) suggested that drainage ditches could provide suitable habitat for aquatic and riparian species. I show that ditches and straightened streams host numerous riparian plant species, in different compositions depending on a variety of environmental factors. Previous studies suggest that the novel and hybrid ecosystems of artificial waterways are important for the overall biodiversity of the landscape (Armitage et al. 2003; Chester & Robson 2013), and that the management of old ditches and straightened streams with low consideration is one of the biggest threats to these drainage networks (Herzon & Helenius 2008). I conclude that variation in biodiversity and species composition in ditches and straightened streams must be acknowledged and that these systems need to be carefully managed for a sustainable future.

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Popular Science Summary

Ditches and straightened streams are artificial waterways that in total stretches almost 1 million kilometres in Sweden. Despite being a common part of the landscape, we know little about the ecology of these artificial waterways. Here I will summarize my study that aim to start filling this knowledge gap.

Ditches have been dug and streams have been straightened for drainage purposes in Sweden for a long time. In central Sweden this has been going on since the days of Gustav Vasa in the sixteenth century, while in the north of Sweden it started in the middle of the nineteenth century. So, why dig ditches and straighten streams in the first place? The answer to this is that ditches and straightened streams drain water from the land and thus allowing plants to grow better. In later days this has mainly been done to promote tree growth for the forest industry. However, since the 1980s ditching and straightening of streams have stopped due to new legislation. Today, existing drainage waterways are instead cleaned to keep their intended functions.

When ditches and straightened streams function as they should the water table in the surrounding ground is lowered, making the environment drier. This environment near waterways is called the riparian zone. The ecology of this zone has been extensively studied along natural streams and has been shown to be one of the most biodiverse environments in the landscape. However, we know little about the ecology of the riparian zone along artificial waterways, such as ditches and straightened streams.

I studied the riparian plant biodiversity and community composition in twenty-two ditches and straightened streams near Vindelån, 60 km west from Umeå in the north of Sweden. Previous studies on natural waterways have found that the biodiversity was higher along larger than small waterways. I expected to find the same, despite the previously made studies were on natural waterways and I studied artificial ones. However, I found that the size of the waterway only influenced species richness, not the other biodiversity measure I studied, the Hill-Shannon diversity. I suggest that it rather is a combination of hydrological factors and if the waterway is on peat or till soil that is affecting the biodiversity. So, there are ecological differences between natural and artificial waterways! I suggest that the novel riparian

ecosystems along artificial waterways have great variability and must be treated with care for the protection of the biodiversity.

As I explained earlier, the cleaning of ditches and straightened streams is the only management of these waterways today. However, guidelines on how, when and on which ditches and straightened streams to clean is inadequate. I show that the community composition differs depending on soil type and if the waterway is a ditch or straightened stream. Further, I show that several variables are affecting what kind of species are found at which sites. This is telling us that there is ecological variation across different drainage waterways and that the same management procedures may not be possible everywhere.

A summary of the take home messages from my study is that the variation of the ecosystems of ditches and straightened streams needs to be carefully considered when planning for future management. By taking this consideration in management, the overall biodiversity of the landscape will increase.

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