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Institutionen för skogens ekologi och skötsel

Evaluating abundance of deciduous trees in production forests along small streams

– can Sweden meet current policy goals without intensive management

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Abstract

Riparian zones are important for many ecological functions such as for providing shade to streams and leaf litter to instream organisms, as well as acting as filters for sediments and excess nutrients that are released during forestry operations. Riparian areas are also important for preservation of biodiversity. Over time, forestry policies in Sweden have changed and this has had effects on the management of forests. For example, in 2013 the Strategic Management Objectives (SMO) were introduced, aiming to act as a guideline for an expected level of environmental consideration. Within the SMOs, there is a goal to preserve and manage functional riparian buffer zones adjacent to streams. One way of reaching this goal is to encourage deciduous tree species in the riparian zones at all stages of forestry operations. The aim with this thesis was to investigate how forestry policy actions have changed management of riparian zones and if historic forest management has impacted how we can meet the SMOs today. The main focus was to understand if deciduous tree species already exist within the riparian zones of small streams or if there is a need for changed or intensified management to encourage deciduous trees. My study was performed in Vindeln, Västerbotten County, northern Sweden. I measured stand characteristics of 16 riparian buffer zones of four different age classes, i.e., <1975, <1993, <2013 and <2019. Each of the age classes has been subject to different forest policies and thus different management within buffers. I found that the 1993 Forestry Act has had a positive effect on deciduous cover in riparian buffers of small streams. On the contrary, the introduction of SMOs in 2013 has not yet improved the buffer management, at least with regard to deciduous tree species and some riparian functions, i.e. dead wood provision and shading. A reoccurring pattern in my results was that there is a time lag from the time the buffer is created until the riparian buffer is functioning as intended. It seems that the most recently created buffers have not yet had time to recover from historical management. If the goal of the SMOs is to have ecologically functioning buffers and stable conditions, e.g. canopy cover adjacent to streams, there is likely a need to change the forestry practices in riparian zones. For example, leaving wider buffers and potentially selectively logging coniferous trees within those buffer strips can encourage deciduous tree species and compensate for economic losses.

Keywords: Riparian forests; Small streams; Strategic Management Objectives; Deciduous; Forest policy change

Sammanfattning

Strandzoner kring bäckar och vattendrag är viktiga för många ekologiska funktioner så som beskuggning och tillförsel av lövförna till organismer i vattendrag. De kan också fungera som filter för sediment och läckage av näringsämnen som frigörs vid skogsskötselåtgärder. Strandzoner är också viktiga för bevarande av biologisk mångfald. Den svenska skogsbrukspolitiken har förändrats över tid och detta har påverkat skogsskötseln. Ett exempel på detta är de branschgemensamma målbilderna som introducerades 2013 med syfte att fungera som riktlinjer för hur skogsbruket ska agera för att främja god miljöhänsyn. I målbilderna finns det ett mål att bevara och sköta funktionella kantzoner mot vattendrag. Ett sätt att nå detta mål är att främja andelen lövträd i kantzoner vid alla skogsskötselåtgärder under rotationsperioden. Syftet med den här studien var att undersöka hur skogsbrukspolitiken har förändrat skötseln av strandzoner och om historiskt brukande har påverkat uppfyllandet av målbilderna idag. Fokus låg främst på att förstå om det redan finns lövträd i strandzonen intill små bäckar eller om skötseln måste förändras eller intensifieras för att främja lövträd. Min studie genomfördes i Vindeln, Västerbottens län, norra Sverige. Jag mätte beståndsvariabler på 16 kantzoner fördelat på fyra olika åldersklasser: <1975, <1993, <2013 och <2019. Varje åldersklass har avverkats under olika rådande skogspolitik och därmed också haft olika skötsel inom kantzonerna. Jag fann att skogsvårdslagen från 1993 har haft en positiv effekt på lövträdsandelen i kantzoner kring små bäckar. I motsats till detta visade det sig att introduktionen av målbilderna 2013 inte ännu har förbättrat skötseln av kantzoner, åtminstone med avseende på andelen lövträd och vissa ekologiska funktioner som t.ex. tillförsel av död ved och beskuggning. Ett återkommande mönster i mina resultat var att det är en tidsförskjutning från skapandet av kantzonen tills att den uppnår avsedd funktion enligt målbilderna. Det verkar som att de nyligast lämnade kantzonerna inte ännu haft tid att återhämta sig från historisk skötsel. Om syftet med målbilderna är att ha funktionella kantzoner och stabila förhållanden t.ex. kronslutenhet intill bäckar, finns det troligen ett behov av förändrad skogsskötsel i strandzoner. Till exempel att lämna bredare kantzoner och potentiellt plockhugga barrträd ur dessa kantzoner kan främja andelen lövträd samt kompensera för ekonomiska förluster.

Nyckelord: Strandskog; Små bäckar; Målbilder; Lövträd; Skogspolicy

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Abbreviations

AIC	Akaike’s Information Criterion
BA	Basal Area
DBH	Diameter at Breast Height
DRIP	Discrete Riparian Inflow Point
FSC	Forest Stewardship Council
GLM	General Linear Models
ha	hectare
KCS	Krycklan Catchment Study
m3sk/ha	Cubic meter standing volume per hectare/ forest cubic meter per hectare
PEFC	Program for the Endorsement of Forest Certification Schemes
SEPA	Swedish Environmental Protection Agency
SFA	Swedish Forest Agency
SMO	Strategic Management Objectives

1. INTRODUCTION

Swedish forests and forest landscapes have been a resource for humans for a very long time, but during the last 100-150 years the forest land use has intensified due to mechanization of the forest industry (Östlund 1993; Bengtsson et al. 2000; Josefsson & Östlund 2011). The technical advancement in forest management has had large positive effects on the economy (Lundmark et al. 2013) and today, forestry in Fennoscandia is among the most mechanized and effective in the world. In fact, nearly all productive forest land is used for production of sawn-timber and pulp-wood (Esseen et al. 1997). However, the advancement in forestry measures has also had effects on the environment and biology. Natural disturbances such as fire and seasonal flooding (Linder et al. 1997, Johansson & Nilsson 2002, Hellberg et al. 2009), as well as many features of old forests such as old trees and deadwood have declined dramatically in the Swedish forest landscape. Instead, these natural disturbances and characteristics have been replaced by anthropogenic disturbances related to forestry operations (Östlund et al. 1997; Bengtsson et al. 2000).

The standard forestry practices in Sweden can be illustrated as a forestry cycle (Figure 1). It starts off with clear-felling of the forest stand, followed by soil preparation (typically scarification) and planting within three years after the felling. After about 16 years the site is normally cleaned (pre-commercially thinned), commercially thinned at 20-70 years and finally cut again after about 60-120 years (Lundqvist et al. 2014; Agestam 2015). At each of these time points, decisions are made about how to manage the forest for future utilization, however the decisions depends on the prevailing forest and environmental policy of the time.

The advancement in forestry and fire management has changed the succession and composition of the forests. Forest management operations such as cutting of old forests, planting, and thinning results in, for example, fragmentation, creation of even-aged monocultures, and a decreased amount of old growth forests. This has in turn been shown to decrease the populations for several hundreds of plant and animal species (Esseen et al. 1997; Bengtsson et al. 2000; Aune et al. 2005; Widenfalk & Weslien 2009). Commercial dominance of coniferous tree species has led to a decrease of broadleaf-dominated forests (Esseen et al. 1997). An important factor leading to the decrease in deciduous trees was the use of herbicides to control spontaneous deciduous regeneration in clear cut areas. Spraying of deciduous trees was introduced in the 1950's and peaked in the 1960's (Östlund et al. 1997). The suppression of deciduous trees has become a threat to biodiversity since deciduous forests constitute one of the most species-rich types of forests in Fennoscandia (Berg et al. 1994; Esseen et al. 1997, Jonsell et al. 2007). It is therefore crucial to retain or encourage deciduous trees in the forest landscape (Ring et al. 2018). Another consequence of forest management is that the occurrence of old and dead trees in Swedish boreal forests has decreased dramatically since the 19th century (Linder & Östlund 1992). Lack of old trees and deadwood poses a great threat to many forest species, including plants, animals and fungi (Berg et al. 1994; Siitonen 2001).

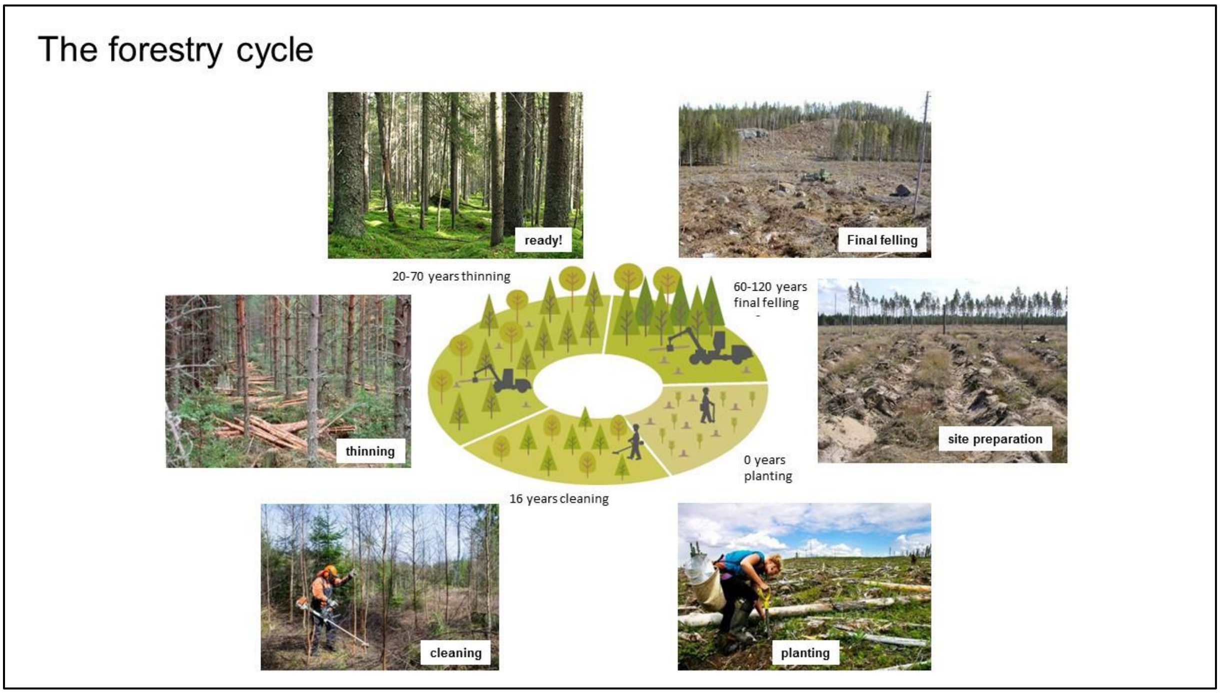


Figure 1: A figure describing the common forestry actions in Sweden during a rotation period starting at clear felling of the old forest, followed by site preparation, planting, cleaning and thinning.

One part of the forest landscape that has been proposed to act as an important area for preservation of biodiversity and hotspots for climate change adaptation is the riparian forests (Capon et al. 2013). Riparian forests are the interface between streams or other freshwater ecosystems and forests (Naiman & Décamps 1997). In Sweden, riparian forests along streams, rivers and lakes was estimated to be about 2.5 % of the total forest area, considering a 10 m wide forested zone on each side of a water body according to Gundersen et al. 2010. Recent research has suggested that simple delineation of fixed 10 m wide riparian forests across the landscape is not adequate, and in fact riparian forests can be narrower as well as much wider than 10 m (Richardson at al. 2012; Kuglerová et al. 2014b; Kuglerová et al 2016). Riparian forests and streams account for numerous important ecological functions in the landscape and have a tight link between each other (Capon et al. 2013). Small streams and their surrounding riparian forests can act as corridors for species dispersal (Grant et al. 2007; Clarke et al. 2008; Kuglerová et al. 2016). Furthermore, riparian forests around small streams also affect factors such as stream water temperature and quality, stream bank stability and sediment influx to the stream (Naiman & Décamps 1997; Moore et al. 2005; Gundersen et al. 2010). The link between riparian and aquatic ecosystems is strongest along small streams (Gregory et al. 1991) and therefore, riparian forests of small streams should be given special attention. Riparian ecosystems along small streams are therefore key areas for ensuring downstream water quality as well as biodiversity on a landscape-scale (Sabo et al. 2005; Kuglerová et al. 2014a). However, in managed landscapes, stressors such as forestry may negatively affect these links (i.e. Wallace et al. 1997; Hjältén et al. 2016, Kuglerova et al 2017).

Observed effects of forestry operations in riparian areas may be increased erosion, sediment transport, nutrient and dissolved organic carbon leakage and water temperature change (Lisle 1989; Ahtiainen 1992; Munthe & Hultberg 2004; Ågren et al. 2015). This may negatively affect flora and fauna in the riparian zone. Forestry operations may lead to increased sediment transport into streams through change of natural flow-paths or erosion of mineral soil (Ågren et al. 2015). Sediment transport may cause siltation in stream beds (Lisle 1989) and by that, decrease the reproductive success of freshwater fish, but also affect macroinvertebrates (Lemly 1982; Soulsby et al. 2001). Furthermore, soil compaction by forestry machines can result in increased leakage of e.g. mercury to streams (Munthe & Hultberg 2004). Reduction of tree cover due to clear-cutting has been shown to increase water temperature (Ahtiainen 1992; Cole & Newton 2013). Also, removal of trees increase incoming light (Holopainen & Huttunen 1992) which stimulates algae growth that can change the entire aquatic food web (Hill et al. 1995, Holopainen & Huttunen 1998). In addition to changed light conditions and temperatures, it has been shown that clear-cutting followed by soil scarification leads to increased concentrations of phosphorous (P), nitrogen (N) and iron (Fe) in small streams (Ahtiainen 1992). It has been shown that the increased temperature and concentrations of nutrients in small streams could be reduced by leaving a forested riparian buffer along the edges of the streams (Peterjohn & Correll 1984; Ahtiainen 1992).

Riparian forested buffer zones, a strip of vegetation not managed for production forestry, along streams and other freshwater ecosystems were developed with the aim to mitigate negative effects of forestry operations on riparian ecosystems (Broadmeadow & Nisbet 2004; Gundersen et al. 2010). The buffer zones can mitigate increased loadings of nutrients, pollutants and sediments to streams and today it is suggested practice in the Nordic countries to leave a forested buffer zone adjacent to all streams and rivers (Ahtiainen 1992; Gundersen et al. 2010; Andersson et al. 2013; Ring et al. 2017). Riparian buffers can also be important for preservation of biodiversity. A study from northern Sweden indicated that riparian forests have higher biodiversity than surrounding forests and higher occurrence of deciduous then coniferous tree and shrub species (Kuglerová et al. 2014b). Therefore, changed tree species composition from deciduous to conifers in riparian forests may pose a threat to both riparian biodiversity as well as instream communities and processes by reducing the quality of leaf litter that enters the stream (Friberg 1997; Lidman et al. 2017). Furthermore, broadleaved trees have been shown to reduce risk of windthrows (Peltola et al. 2000; Valinger & Fridman 2011) and thus contribute to the stabilization of streambanks. Therefore, recommendations include retention of deciduous tree species in the riparian buffers and selective cutting of conifers (Hylander et al. 2004; Gundersen et al. 2010; Andersson et al. 2013; Felton et al. 2016). However, even though the importance of riparian forests for biodiversity and stabilizing stream banks and other ecosystem functions is high, it may be exceeded by their role in regulating stream water quality (Kuglerová et al. 2014b; Kritzberg et al. 2019). Small streams have been researched with focus on hydrological and biogeochemical functions for a long time, however, they have not been given the same attention when it comes to ecological research (Wohl 2017; Richardson 2019) or how they are affected by commercial forestry (Kuglerova et al. 2017). In both North America and Fennoscandia, forested buffer zones are often missing along small streams (Richardson et al. 2012; Hasselquist et al. 2019). Due to the large total length and organization of stream networks (Blyth & Rodda 1973; Bishop et al. 2008); increased protection of small, temporary streams could affect large areas of forest land and thus also lower the available land for timber production. Thus, there is a conflict between environmental advocates which stress that small streams are essential for the integrity of stream networks and foresters that argue that it will be too costly to protect them all (Acuña et al. 2014).

The ecological status of riparian forests depend on abiotic and biotic factors, but also on forest history. The forest history in Sweden, including change of management systems and several policy changes, have influenced the riparian forest landscape. History can explain when and how forest stands regenerated and how they have been treated during thinning operations or harvest (Ring et al. 2018). Between 1920 and 1940 there was a period of financial crisis in Sweden and during this period, clearcutting was abandoned. Instead, the predominant method for timber harvest was selective logging, especially in northern Sweden (Lundmark et al. 2013). Thus, at present, stands older than about 75 years are more likely to have originated from natural regeneration within selection or shelterwood systems (Ring et al. 2018). After the Great Depression, the profitability of forestry increased and clearcutting became the dominant silvicultural system in Sweden (Östlund et al. 1997). In 1979, a new Forestry Act was introduced (Figure 2). It aimed at guaranteeing a long-term forest production with the first, albeit very little, considerations for biodiversity and nature conservation (SFS 1979:429). From the advancement in clearcutting until the introduction of the new Forestry Act in 1993 (Figure 2) (SFS 1993:1096) that stated that production goals and environmental goals should be equal, the trees in the riparian zone of small streams were normally logged (Ring et al. 2018). Consequently riparian areas were planted with commercial (coniferous) species, thinned and cleaned during the rotation. Hence, forests in this age (26-75 years old) will likely have mature spruce all the way to the waters' edge and very little shrubs, small trees and deciduous species. During the 1990's, forest certification was implemented in Sweden and right now, the majority of Swedish forest land is certified according to one or both of the two standards, FSC (Forest Stewardship Council 2015) and PEFC (Program for the Endorsement of Forest Certification Schemes 2018). Both the FSC and PEFC national standards for Sweden demand protection of riparian forest buffers adjacent to streams and other freshwater ecosystems. Therefore, sites that are at present younger than 20-25 years old often have a retained forested buffer from the previous stand (Ring et al. 2018).

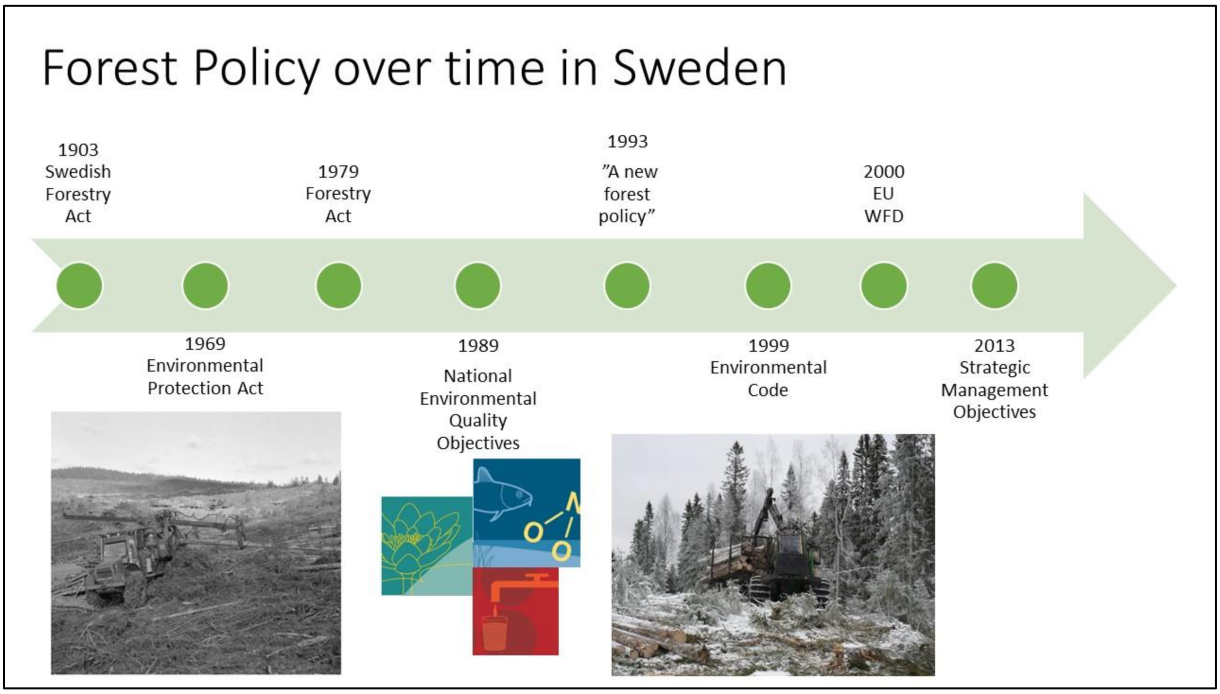


Figure 2: Illustration of the development of forest policy in Sweden.

In 2011, the Swedish Forest Agency (SFA, skogsstyrelsen) and the Swedish Environmental Protection Agency (SEPA, naturvårdsverket) presented a government mandate that aimed at developing “a knowledge platform for sustainable forest management”, with focus on the environmental goals (SFA & SEPA 2011). From this, SFA started a dialogue with the forestry sector about goals, legal demands and methods for follow-up for environmental consideration (miljöhänsyn). From this dialogue process, the SFA developed new Strategic Management Objectives (SMO, målbilder) for environmental considerations in 2013 (Figure 2). The goal with the SMOs was to reflect an expected level of consideration based on the principal of sector responsibility (Andersson et al. 2013). The Swedish forestry policy is usually summarized with the motto “freedom with responsibilities” where there are a number of rules, but forest owners still have a responsibility to reestablish forests and maintain or develop environmental values on their own (Appelstrand 2012). Thus, very little legal enforcement of the rules takes place in Sweden (Ring et al. 2018).

Within the SMOs (2013), there is the goal to preserve and manage riparian buffer zones along streams adjacent to harvested forest in order to:

1. **Preserve important soil chemical processes,** such as denitrification and nutrient uptake.
2. Act as a **filter** for sediment transport from upland and **stabilize** shoreline to prevent erosion.
3. Contribute **food to aquatic organisms** through falling leaves and insects.
4. Provide **stable shade** of streams over time.
5. Contribute **deadwood** to the water.
6. Preserve **biodiversity.**

One way of reaching these objectives that has been promoted by the SFA is to encourage deciduous tree species in riparian zones at all stages of the forestry cycle (Figure 1), i.e. pre-commercial and commercial thinning and at final felling (Andersson et al. 2013). It is, however, unclear if encouragement of deciduous species is possible without planting or seeding due to historic even-aged forestry practices and policies that favored conifers, and if it has actually been done since the implementation of the SMOs. Further, very few studies have evaluated site characteristics of riparian forests along small streams in Sweden (Ring et al. 2018).

Objective

The aim of this master's thesis was to investigate if riparian buffers can meet the SMOs without additional management practices. Primarily, I investigated the occurrence of deciduous tree species in riparian forests of different ages, i.e., stages within the rotation cycle (Figure 1). The occurrence of deciduous trees was used to determine whether managers already have deciduous tree species on site or if there is a need of more intensive management practices to meet the SMOs.

Questions

Q1: Do managers have deciduous tree and shrub species on site at different riparian forests management stages?

Q2: What is the biodiversity of woody species in riparian zones along small streams?

Q3: Can differences in deciduous tree and shrub species composition be attributed to past management of riparian buffers?

Q4: Has implementation of the SMOs affected other characteristics of riparian buffers zones, such as contributing deadwood or providing shade?

Hypothesis

H1: There will be more deciduous tree and shrub species present in the riparian zones of buffers that were created after the SMOs were introduced (2013).

H2: There will be higher biodiversity of woody species in the riparian zones of buffers that were created after the SMOs were introduced (2013).

H3: The amount of deciduous trees and shrubs in riparian zones in stands generated during the even-aged clear-felling era are likely to be lower than in stands generated after the 1993 Forestry Act or the SMOs from 2013. The introduction of SMOs will have increased the amount of deciduous trees and shrubs more than the 1993 Forestry Act.

H4: The introduction of the SMOs in 2013 will make a significant positive shift in the ability for riparian zones to provide services such as deadwood or shade. Buffers created after the introduction of SMOs to have similar shading as older buffers (before 1993 Forestry Act). Further, I also hypothesize that the SMOs will have a larger impact on the services (shading & deadwood) provided by riparian zones than the 1993 Forestry Act.

2. MATERIALS AND METHODS

2.1 Study area

All sites were in Vindeln municipality (Figure 3) and the majority of the inventoried sites included in this study were found within or adjacent to the Krycklan Catchment Study (KCS), a research catchment in northern Sweden that is situated approximately 50 km north-west of the city Umeå (Laudon et al. 2013). The KCS covers 6790 ha of land and the primary land use in the area is production forestry. The till soils are dominated by well-developed iron podzols in forests, but near the stream channels, the inorganic content increases which forms a riparian peat zone along streams (Laudon et al. 2013).

The sites not within or adjacent to the KCS are found within another study area called Trollberget which is located about two of kilometers southeast from the KCS (Figure 3). Trollberget is a part of the KCS water quality monitoring network and was established to study the restoration of a drained peatland, best management practices for cleaning of forest ditches, and riparian buffer design (Laudon 2019). The streams in Trollberget are tributaries to the Krycklan river and join it below the boundary of the KCS.

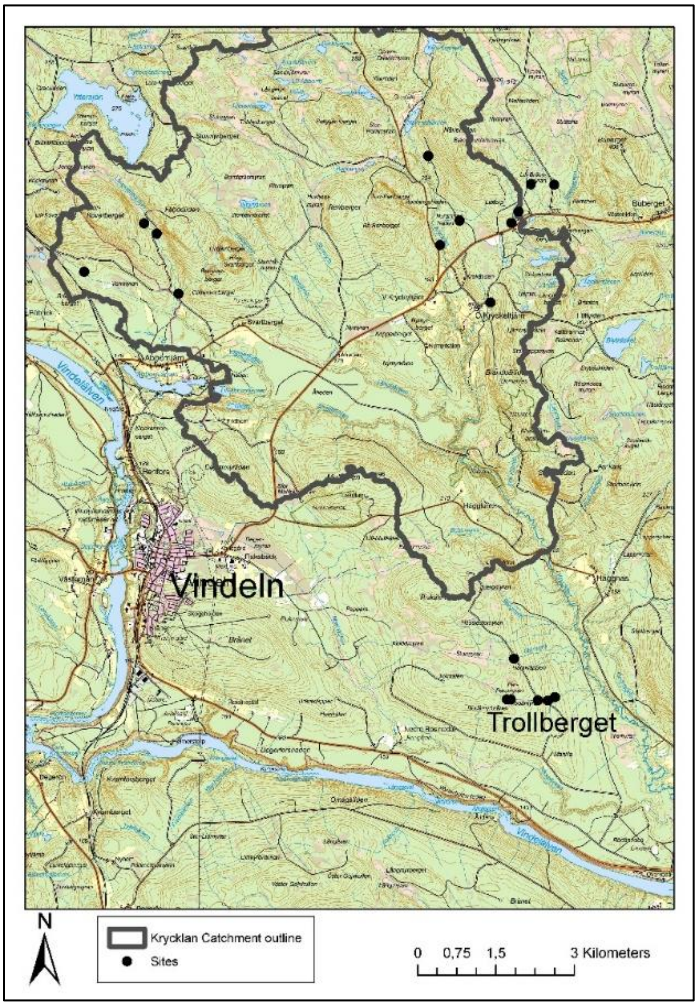


Figure 3: Map displaying the inventoried sites within Vindeln municipality. The black dots represent each inventoried site and the black border shows the outline of the Krycklan catchment. Map by Jessica Åström based on Lantmäteriets landskapskarta.

2.2 Experimental design

I set-up belt transects along 16 small streams, < 3 m wide, within forests of four different age classes. The age classes were based on shifts in management recommendations or policies in Sweden over the past 100 years (Hasselquist et al. 2019), with the year of final felling as the baseline. The age classes were 0-5 years, 6-15 years, 26-44 years and >44 years since final felling. The first class (harvested 0-5 years ago) represents the years 2014-2019 and are linked to the time after the creation of the SMOs in 2013. The second class (harvested 6-15 years ago), years 2004-2013 represents the period after the Forest Act of 1993 that stated that environmental goals should be equal to the production goals, but before the SMOs. The third age class (harvested 26-44 years ago), years 1975-1993, represents the period between the Forestry Act of 1979 and the introduction of the new Forestry Act of 1993. Finally, the last class is >44 years old, or pre 1975, that was the oldest record of management actions in our available data material. This age class is supposed to reflect forestry before the forestry act in 1979. I will from now on refer to the age classes as following: <2019 (2014-2019), <2013 (2004-2013), <1993 (1975-1993) and <1975. It is important to note that age of the riparian buffer relates to different policy and management rules and thus different operations may also have occurred within the riparian forests. Also, factors like ownership can affect timing of forestry operations. Therefore, the age of the riparian trees and time since creation of the buffer is not the only factor which varies with the differing age classes of the riparian zones.

2.3 Site selection:

Using orthophotos of the KCS from 1963, 1975, 1985, 1993, 2004 and 2014, loaded into in standard GIS software (ArcMap 10), I searched for areas that were cut within the span of the different age classes. For each age class, I selected four sites where a small stream (< 3 m) with a minimum reach length of 60 m intersected with at least 20 m of the stand on each side of the stream. Using a georeferenced PDF I created in GIS, loaded into the free version of the Avenza maps app (version 3.7.2, Avenza system Inc., Toronto, Canada), I could easily find my sites in the field using my smartphone. During fieldwork, I first checked that the stream was in fact a flowing stream and not a ditch or a DRIP (discrete riparian inflow point, sensu Ploum et al. 2018). I also looked in the surrounding stand to confirm that it was the correct age class. If these criteria were fulfilled, I performed measurements.

Along each stream within a stand, I placed four belt transects (similar to Ring et al. 2018) with a fixed width of 7 m perpendicular to the stream, starting about 15 m from the edge of the stand and following edge to edge from each other on alternate sides of the stream (Figure 4). The side I started on was randomly chosen in the field with a coin toss. I divided each transect into three, 5 m wide zones, 0-5 m, 5-10 m and 10-15 m and these zones were the basis for three plots with 7 m plot width (Figure 4). For the first transect on each site, I noted the GPS coordinates. In total I inventoried 16 sites, each with four belt transects with three plots each, which sums up to a total of 192 plots. The inventories were carried out between 23 September and 11 October 2019.

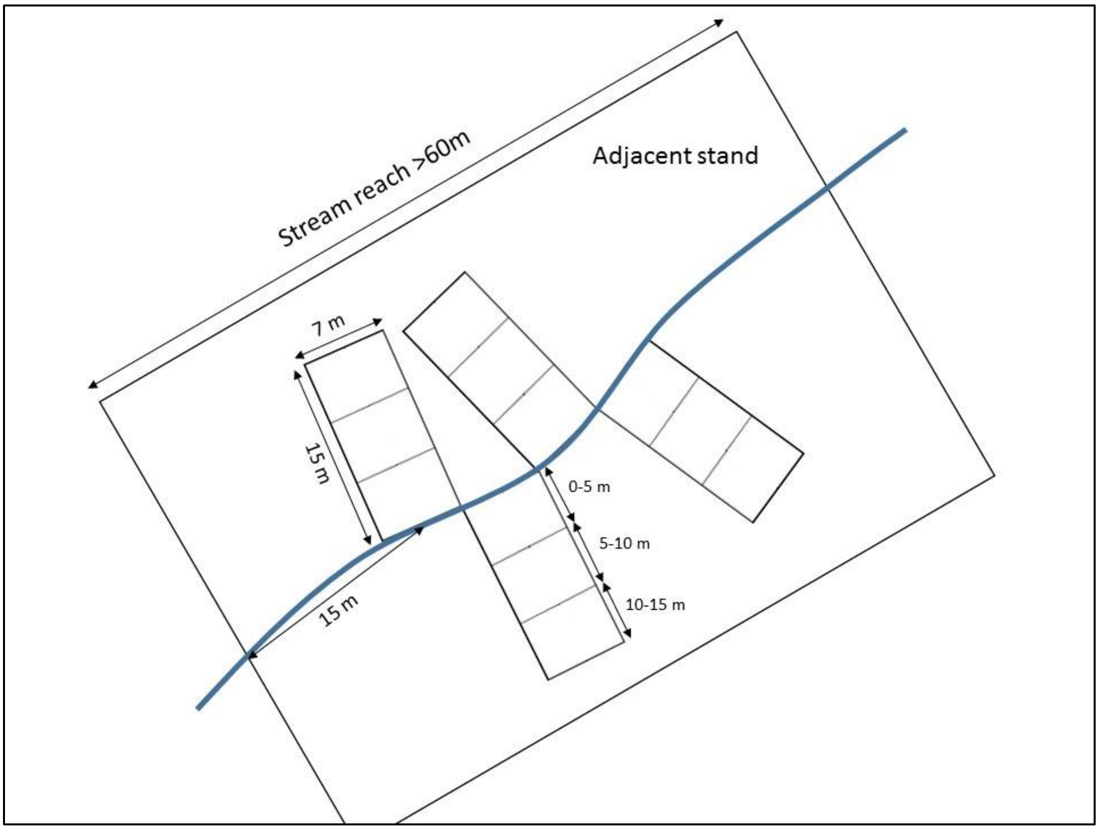


Figure 4: Sampling design for a single stream reach within a forest stand with four, 15 m long transects placed perpendicular to the stream. The distance between the edge of the stand and the first transect is 15 m and the following transects is laid out edge to edge with the previous one. Each transect is divided into three zones, 0-5, 5-10 and 10-15 meters and these form three plots with the width of seven meters.

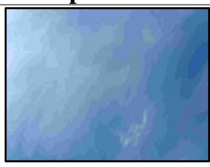
2.4 Variables used for site description

In order to describe the site, I evaluated different variables for each transect. These variables were: width of stream, ground conditions and bottom layer. I measured average wetted width of the stream channel in the middle of each transect. I inventoried ground conditions in order to describe the sites and provide information on factors that might affect the composition of the riparian forest. For each plot, I evaluated whether the ground was wet (peat-like) or solid, I also noted if the plot lied within a DRIP and the DRIP's was determined visually by characteristics described in Ploum et al. (2018). I also noted the dominant species (bryophytes) in the bottom layer vegetation. The species was divided into five soil moisture classes: very dry, dry, mesic, moist and wet (according to Arnborg 1990). I measured the bottom layer vegetation cover to strengthen the evidence for DRIP's.

2.5 Response variables

The response variables for the study were the number of stems, height, basal area and volume of different species of trees > 1.3 m, number of stems of trees and shrubs < 1.3 m, number and diameter of deadwood, species richness of shrubs and small trees and canopy cover. Variables related to the stand were number of trees differentiated into species and the diameter at breast height for all trees > 1.3 m. I calipered deadwood at the middle of the trunk if it had been rooted within the plot and had a length of > 1 m and diameter > 5 cm in the middle of the trunk, and I also determined the length. Other deadwood within the plot where I could not determine the origin was treated the same way. I noted what probable reason the dead tree died/fell over (e.g. blow-down, rot or snow) and also identified whether it was coniferous or deciduous deadwood. Species richness was calculated as the number of species of shrubs and small trees recorded in each plot with the assumption that the shrub and small tree species would also be occurring in the > 1.3 m trees. I determined the canopy cover above the stream through visual analysis of a photo. I took a photo of the canopy in the center of the stream at the middle of the transect 1.3 m above the water level and classified the canopy coverage using a scale of 1-5, where 1 was no canopy cover and 5 was a full, closed canopy (Table 1). The photos were taken with an ordinary phone camera and with no wide angle objective. Each photo was saved so that it was possible to re-evaluate the canopy cover after the field inventories to ensure that they were all equally assessed.

Table 1: canopy cover class with the corresponding canopy cover percentage

Canopy cover class	**Percentage of coverage**	**Example**
1 (no canopy cover)	0 %	
2	~25 %	
3 (intermediate canopy cover)	~50 %	
4	~75 %	
5 (almost full canopy cover)	>90 %	

2.6 Data management and preparations

All the data that I collected in field was entered and processed in MS Excel.

For each site, I measured tree heights on four trees within each transect. In total I measured 16 tree heights per site and regressed them against their respective DBHs to find a function that could best explain their relationship. The function with highest R^2 was the one that best explained the correlation between height and DBH and was therefore the one I used to reconstruct the height of the rest of the measured trees in the plots. In this example a 2nd order polynomial, was used to reconstruct the heights:

$$h_i = -cx_i{}^2 + \ cx_i - c$$

Where h_i is the estimated height in meters of tree number i, c is a coefficient determined by the polynomial function, x is the diameter in centimeters for tree number i.

I calculated basal area for each stem:

$$BA_i = \left(\frac{DBH_i}{2}\right)^2 \times \pi$$

Where BA_i is basal area for tree i (m^2), DBH_i is the diameter in breast height (m) for tree i.

I then used the basal area and reconstructed heights to calculate the volume for each tree:

$$V_i = BA_i \times h_i \times 0{,}47$$

Where V_i is the volume (m^3sk) for tree number i, BA_i is the basal area for tree number i, h_i is the height of tree number i and 0.47 is a coefficient that accounts for the conical shape of the trees.

Basal area, volume and number of stems was summed up for each 5 x 7 m plot and scaled up to number per hectare because this is typically how wood volume is presented in forestry science.

2.7 Statistical analysis

I performed all statistical analyses and created all graphs in R, a free software for statistical computing and creating graphics (The R Foundation 2019). To understand how time since buffer creation (age class) and distance from the stream (plot) affected my response variables, but to also include the effect of random variables of stream identity or transect, I used the lme4 package (Bates et. al. 2015). When data were normally distributed, I used linear mixed effects models (lmer) and if the data where not normally distributed, I either used a general mixed effects model (glmer) with a Poisson distribution or transformed the data before analysis. I then did a backward elimination of non-significant effects, starting from the random effects, taking the model with the lowest AIC value (Akaike's information criterion) and proceeding to the interactions. In some cases, the random effect model overfitted the data, and in those cases I used General Linear Models (GLM). For the GLM models, I used the Quasi-Poisson distribution due to the presence of so many zeros in my data. I only measured canopy cover at the stream, thus I used a GLM because I did not test for the effect of distance from stream. Due to too little data, I used only the summed values for each site for deadwood volume, total amount of deadwood, and blowdown trees. For these models, I used GLM models to test if there were any significant differences among the age classes.

Common for all statistical tests was to calculate the estimated marginal means, similar to the least squared means for a model. The estimated marginal mean takes predictions from a model and averages them together in order to summarize the primary factor effects. Since the estimated marginal means summarize the model that was fitted to the data, I used these for presentation in figures instead of traditional means (Searle et al. 1980). Within emmeans package, I used the "tukey" method to compare all possible pairwise differences of means, to determine which means were significantly different from each other (Sokal & Rohlf 1995).

3. RESULTS

The bottom layer vegetation was dominated by mosses such as *Pleurozium schreberi*, *Hylocomium splendens*, *Dicarnum scoparium* (Table 2). In total, I inventoried nine different species (or species groups, e.g. *Salix* spp.) of trees and woody shrubs.

Table 2: Mean (± 1SE) characteristics of 16 sites, with four sites within each age class

Site characteristics	Overall	Age class <1975	Age class <1993	Age class <2013	Age class <2019
Mean stream width (m)	0,7 ± 0,05	0,6 ± 0,07	0,9 ± 0,14	0,7 ± 0,12	0,6 ± 0,07
Dominating soil moisture type	Mesic	Mesic	Mesic	Mesic	Mesic
Mean number of deadwood pieces/ha	192 ± 18	202 ± 34	125 ± 31	179 ± 37	262 ± 43
Mean volume of deadwood/ha	30 ± 8	12 ± 4	3 ± 1	49 ± 25	55 ± 20
Mean number of blowdown deadwood/ha	28 ± 7	12 ± 8	0	54 ± 22	48 ± 18

Notes: The age classes are based on the year of clear felling where age class <1975 represents sites cut before 1975, <1993 represents sites cut between 1975-1993, <2013 represents sites cut between 2004-2013 and <2019 represent sites cut after 2014. The soil moisture type "mesic" is dominated by *Pleurozium schreberi*, *Hylocomium splendens*, *Dicarnum scoparium* mosses.

3.1 Deciduous tree and shrub species

The total number of tree stems per hectare differed among age classes ($p < 0.0001$) as well as with distance from the stream ($p < 0.0001$; Figure 5). Age class <1993 had a significantly higher number of tree stems than all other age classes ($p < 0.0001$; Figure 5A). Across all sites, the number of tree stems per hectare was significantly higher in the first five meters from the stream compared to both 5-10 m ($p = 0.0038$) and 10-15 m ($p = 0.0007$; Figure 5B) from the stream. For both deciduous and coniferous trees, the number of stems was significantly higher in riparian buffers from <1993 compared to all other age classes ($p < 0.0001$; Figure 6A). The number of deciduous trees was, in fact, significantly higher in buffers from <2013 compared to <2019 ($p = 0.0036$; Figure 6A). Furthermore, when I analyzed the percentage of deciduous stems per hectare, I found significant differences among age classes ($p = 0.001$; Figure 7). The percent of deciduous stems was higher in age class <2013 than both <1975 and <2019 ($p = 0.1$; Figure 7).

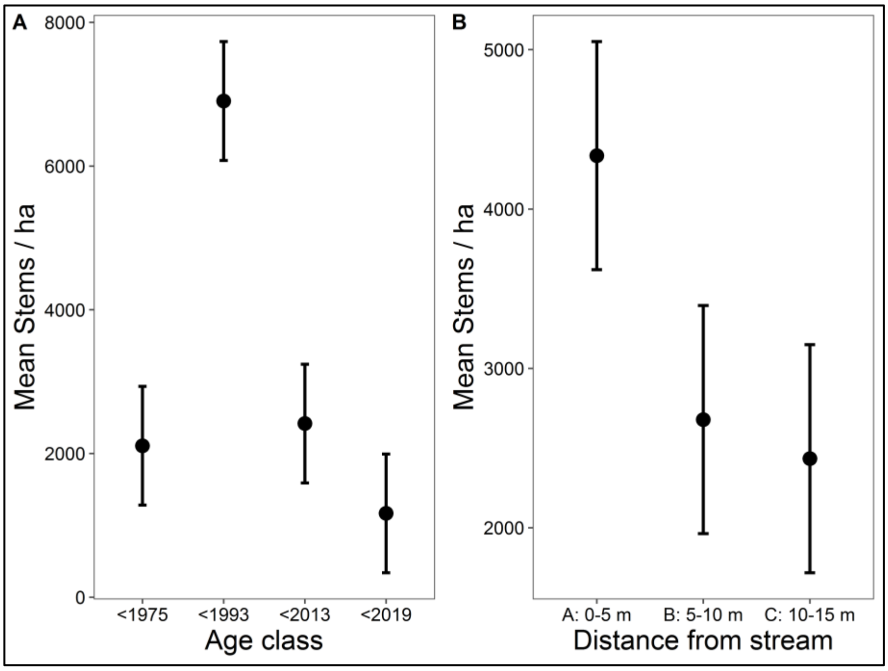


Figure 5: Estimated marginal means for total number of stems (>1.3 m) per hectare separated by age class and distance from the stream (plot). The age classes are based on the year of clear felling where age class <1975 represents sites cut before 1975, <1993 represents sites cut between 1975-1993, <2013 represents sites cut between 2004-2013 and <2019 represent sites cut after 2014.The plots A, B, C represent the three different zones with a fixed width 0-5 m, 5-10 m and 10-15 m away from the stream. The error bars show the 5% and 95% confidence interval for the mean values.

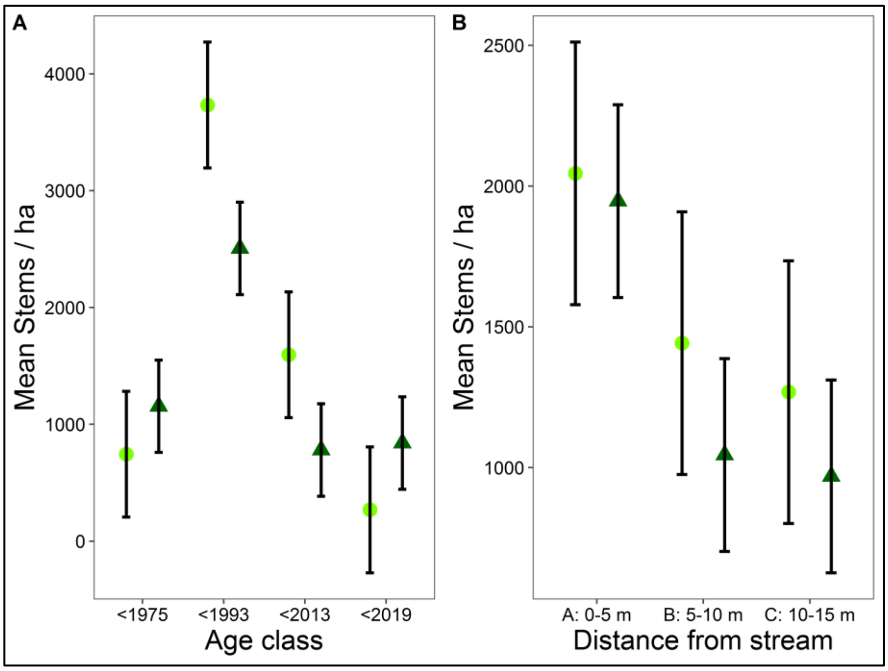


Figure 6: Estimated marginal means for total number of tree stems (>1.3 m) per hectare divided into deciduous (light green, points) and coniferous (dark green, triangles) species separated by age class and distance from the stream (plot). The age classes are based on the year of clear felling where age class <1975 represents sites cut before 1975, <1993 represents sites cut between 1975-1993, <2013 represents sites cut between 2004-2013 and <2019 represent sites cut after 2014.The plots A, B, C represents the three different zones with a fixed width 0-5 m, 5-10 m and 10-15 m away from the stream respectively. The error bars show the 5% and 95% confidence interval for the individual mean values.

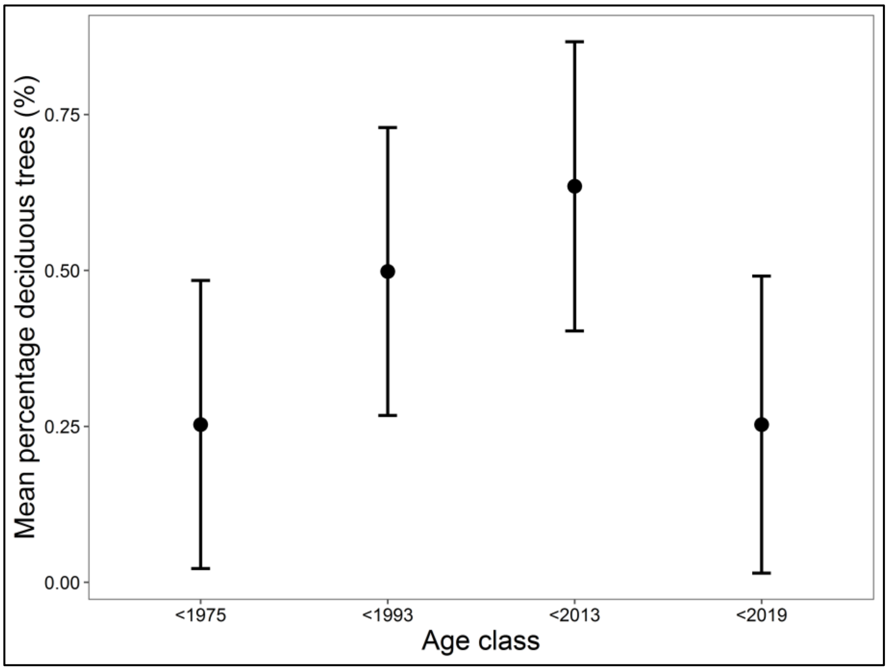


Figure 7: Estimated marginal means for percentage of deciduous tree stems (>1.3m) per hectare separated by age class. The age classes are based on the year of clear felling where age class <1975 represents sites cut before 1975, <1993 represents sites cut between 1975-1993, <2013 represents sites cut between 2004-2013 and <2019 represent sites cut after 2014. The error bars show the 5% and 95% confidence interval for the individual mean values.

The total number of shrubs and small trees (< 1.3 m) per hectare differed significantly among age classes ($p < 0.0001$; Figure 8A), with riparian buffers from <2013 having a significantly higher number of shrubs and small stems than both <1975 ($p = 0.0015$) and <1993 ($p = 0.0016$). Riparian buffers from <2019 also had higher number of shrubs and small trees than both <1975 and <1993 ($p = 0.175$; Figure 8A). Furthermore, the total number of shrubs and small trees differed significantly with regard to distance from the stream ($p < 0.0001$; Figure 8B). The number of shrubs and small trees was significantly higher in the first 5 meters from the stream compared to both 5-10 m and 10-15 m from the stream ($p < 0.0001$).

The total number of deciduous shrubs differed significantly with age class ($p = 0.05$), as well as with distance to the stream ($p < 0.0001$; Figure 9). Riparian buffers from <2013 had higher counts of deciduous shrubs than those of age class <1975 ($p = 0.0443$) as well as <2019 ($p = 0.1575$). The first 0-5 m from the stream had lower numbers of deciduous shrubs and small trees than both 5-10 m and 10-15 m from the stream ($p < 0.0001$). Additionally, the deciduous shrub count was lower within 5-10 m compared to 10-15 m from the stream ($p < 0.0001$). The interaction between age class and distance to the stream significantly affected the count of deciduous shrubs per hectare ($p < 0.0001$; Figure 9). Within the first 5 m from the stream, age class <2013 was higher than <1975 ($p = 0.0083$) and <2019 ($p = 0.0764$). At 10-15 m from the stream, riparian buffers of age class <2013 had higher number of deciduous shrubs than both 1975 ($p=0.0341$) and <1993 ($p=0.0668$; Figure 9). No significant differences in percentage of deciduous shrub counts was found among age classes or with distance to the stream ($p > 0.05$).

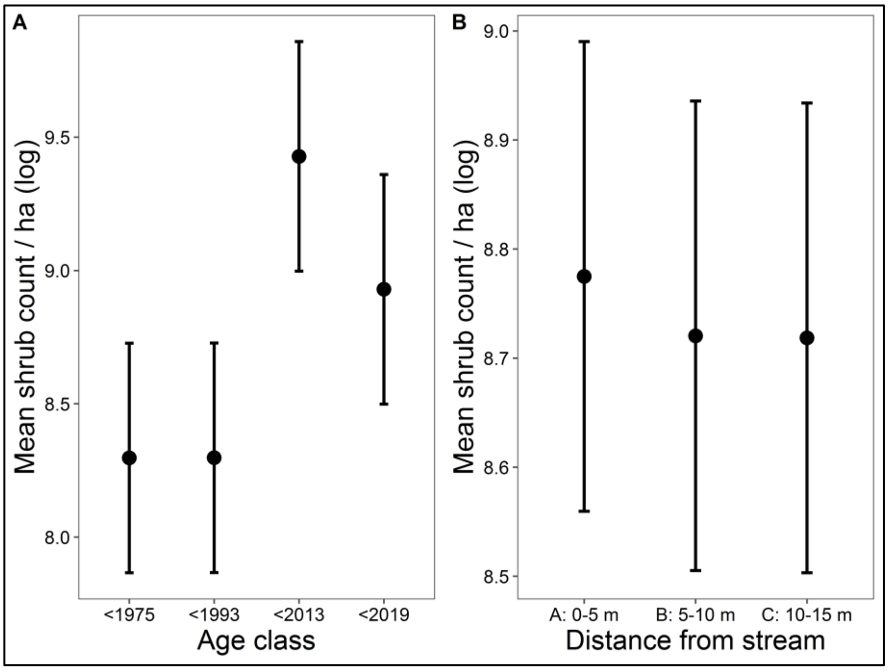


Figure 8: Log transformed estimated marginal means for total number of shrubs and small trees (<1.3m) per hectare separated by age class and distance from the stream (plot). The age classes are based on the year of clear felling where age class <1975 represents sites cut before 1975, <1993 represents sites cut between 1975-1993, <2013 represents sites cut between 2004-2013 and <2019 represent sites cut after 2014. The plots A, B, C represents the three different zones with a fixed width 0-5 m, 5-10 m and 10-15 m away from the stream respectively. The error bars show the 5% and 95% confidence interval for the individual mean values.

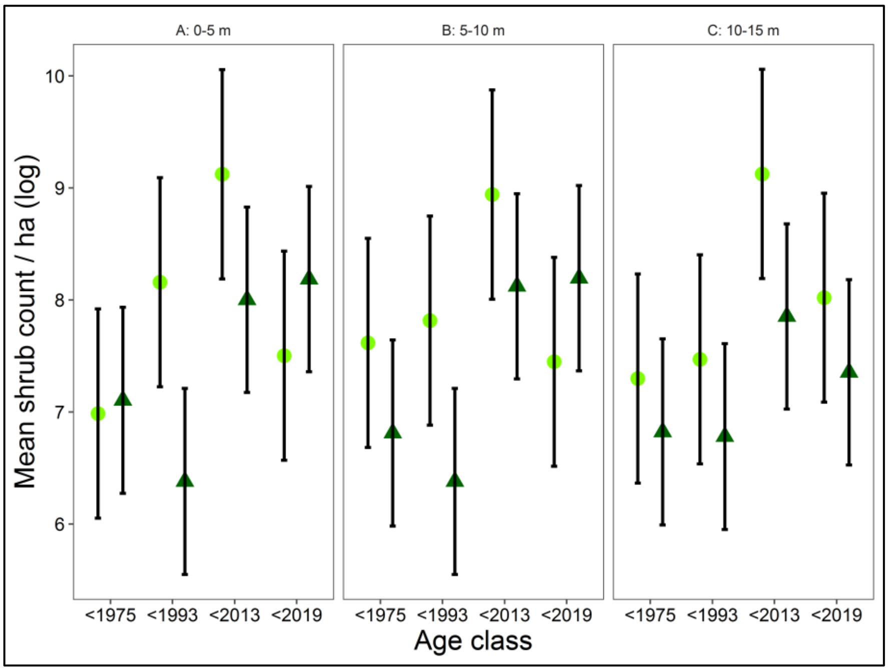


Figure 9: Log transformed estimated marginal means for total number of shrubs and small trees (<1.3 m) per hectare divided into deciduous (light green, points) and coniferous (dark green, triangles) species separated by age class and distance from the stream (plot). The age classes are based on the year of clear felling where age class <1975 represents sites cut before 1975, <1993 represents sites cut between 1975-1993, <2013 represents sites cut between 2004-2013 and <2019 represent sites cut after 2014. The plots A, B, C represents the three different zones with a fixed width 0-5 m, 5-10 m and 10-15 m away from the stream respectively. The error bars show the 5% and 95% confidence interval for the individual mean values.

In addition to the number of stems, I also calculated volume to determine if there were not only more stems, but bigger deciduous trees in the riparian zones of the youngest age class (<2019). The volume of all trees per hectare differed among age classes ($p = 0.001$; Figure 10A) as well as with distance to the stream ($p = 0.01$; Figure 10B). Total volume per hectare was highest in riparian buffers of age class <1975 that was significantly higher than both <1993 ($p = 0.0701$) and <2013 ($p = 0.0362$; Figure 10A). The total volume per hectare was highest in the first five meters from the stream, and it was significantly higher than both 5-10 m ($p = 0.0336$) and 10-15 m from the stream ($p = 0.0292$; Figure 10B). Divided into leaf types, the total volume of deciduous trees differed among age classes and with distance to the stream ($p < 0.0001$; Figure 11). Riparian buffers of age classes <1975 had higher volume of deciduous trees than those of <2013 ($p = 0.0268$; Figure 11A). Additionally, those from <1993 had higher deciduous volumes than <2013 ($p = 0.0798$; Figure 11A). The deciduous volume was highest closest to the stream (0-5 m), and significantly higher than both 5-10 m and 10-15 m away ($p = 0.0103$, $p = 0.0035$; Figure 11B).

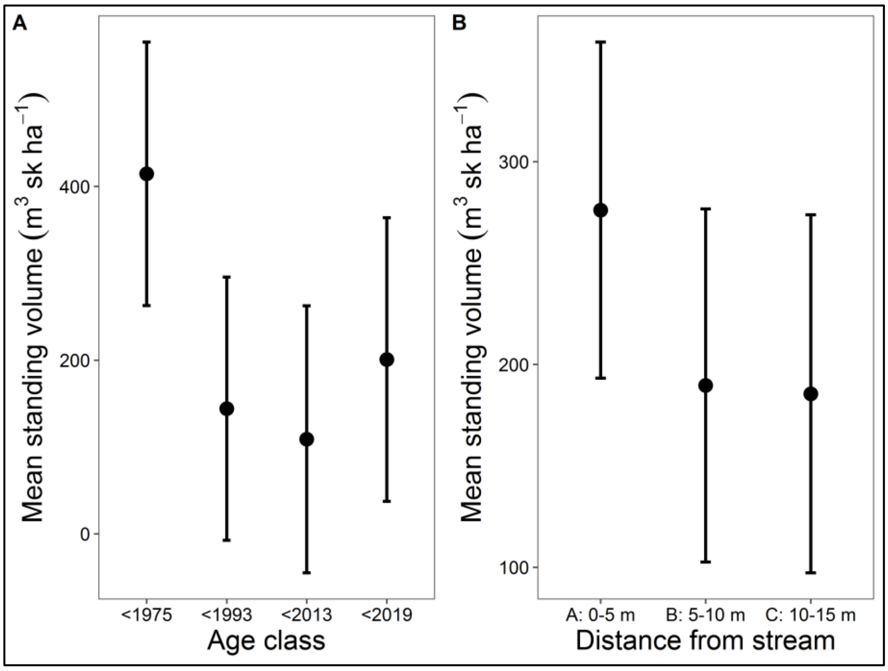


Figure 10: Estimated marginal means for total volume (m^3sk) of trees per hectare separated by age class and distance from the stream (plot). The age classes are based on the year of clear felling where age class <1975 represents sites cut before 1975, <1993 represents sites cut between 1975-1993, <2013 represents sites cut between 2004-2013 and <2019 represent sites cut after 2014. The plots A, B, C represents the three different zones with a fixed width 0-5 m, 5-10 m and 10-15 m away from the stream respectively. The error bars show the 5% and 95% confidence interval for the individual mean values.

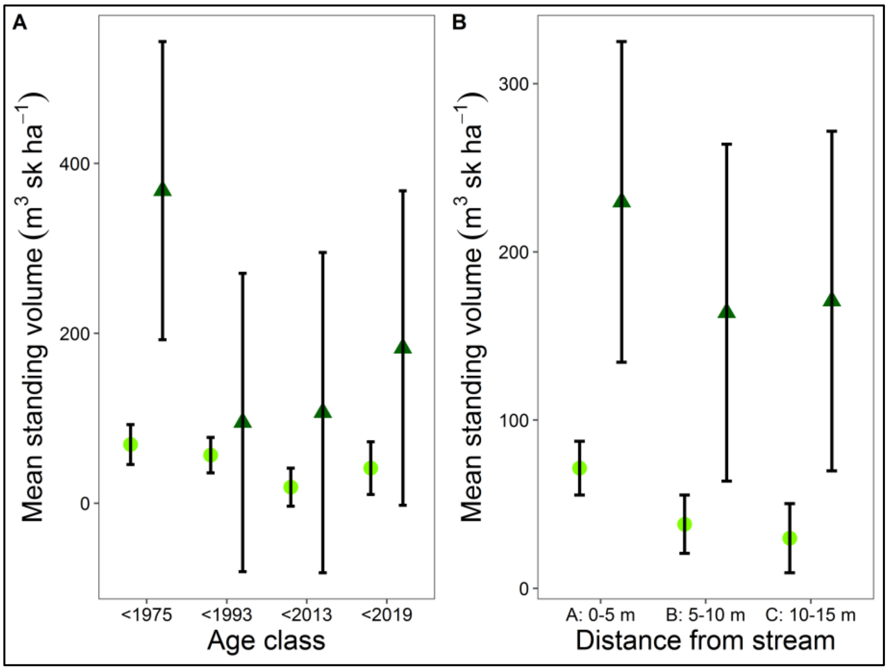


Figure 11: Estimated marginal means for total tree volume (m^3sk) per hectare divided into deciduous (light green, points) and coniferous (dark green, triangles) species separated by age class and distance from the stream (plot). The age classes are based on the year of clear felling where age class <1975 represents sites cut before 1975, <1993 represents sites cut between 1975-1993, <2013 represents sites cut between 2004-2013 and <2019 represent sites cut after 2014. The plots A, B, C represents the three different zones with a fixed width 0-5 m, 5-10 m and 10-15 m away from the stream respectively. The error bars show the 5% and 95% confidence interval for the individual mean values.

3.2 Species richness

In total I inventoried nine different species of woody shrubs and small trees within the plots. They were: birch (*Betula pubescens* Ehrh.), alder (*Alnus incana* L.), aspen (*Populus tremula* L.), S*alix* spp., rowen (*Sorbus aucuparia* L.), pine (*Pinus sylvestris* L.), spruce (*Picea abies* (L.) H. Karst), juniper (*Juniperus communis* L.) and raspberry (*Rubus idaeus* L.).

Species richness of shrubs and small trees was highest in the two intermediate age classes (p <0.0001; Figure 12A), the riparian buffers of both <1993 and <2013 had significantly higher species richness than those of <1975 (p<0.0001). Additionally, the species richness was higher in both age class <1993 and <2013 than <2019 (p = 0.0003, p = 0.0001; Figure 12A). The highest species richness for all age classes was found closest to the stream (0-5 m; p = 0.01; Figure 12B). The species richness in the distance of 0-5 m from the stream was higher than both 5-10 m and 10-15 m (p = 0.0375, p = 0.0820; Figure 12B).

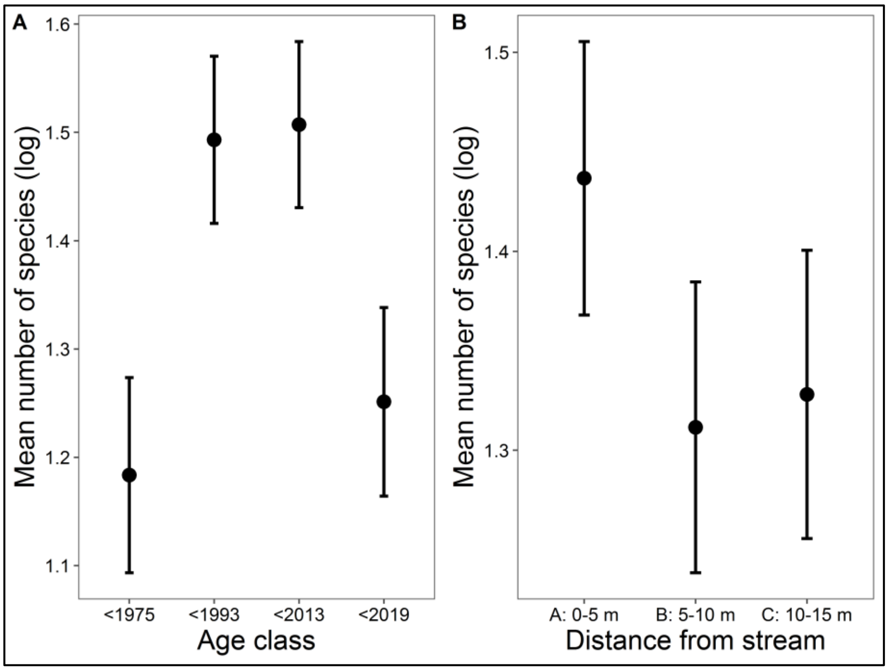


Figure 12: Log transformed estimated marginal means for species richness of shrubs and small trees separated by age class and distance from the stream (plot). The age class are based on the year of clear felling where age class <1975 represents sites cut before 1975, <1993 represents sites cut between 1975-1993, <2013 represents sites cut between 2004-2013 and <2019 represent sites cut after 2014. The plots A, B, C represents the three different zones with a fixed width 0-5 m, 5-10 m and 10-15 m away from the stream respectively. The error bars show the 5% and 95% confidence interval for the individual mean values.

3.3 Other services provided by riparian buffers

I found that the two oldest age classes had higher canopy cover, while the two youngest age classes had less dense canopies ($p < 0.05$; Figure 13). The canopy cover in age classes <1975 and <1993 was significantly higher than age class <2013 ($p= 0.0003$) and <2019 ($p= 0.0001$; Figure 13).

For deadwood volume, there were no statistically significant difference among age classes ($p > 0.05$). However, the highest volume of deadwood was found within the two youngest age classes (<2013 & <2019) and the lowest volume in the age class <1993 (Table 2). Furthermore, results from total deadwood count partly agrees with this (Table 2). There were no significant difference in total count of deadwood pieces among age classes ($p > 0.05$). However, I found the highest amount of deadwood pieces per hectare in buffers of the youngest and oldest age classes (<2019 & <1975) and lowest amounts in buffers from the next oldest age class (<1993; Table 2). Additionally, my analysis of blowdown deadwood did not show any differences among age classes ($p > 0.05$), but I found that the highest amounts of blowdown deadwood was located within buffers of the two youngest age classes (<2013 & <2019), and in the <1993 buffers I found no blowdown deadwood at all (Table 2).

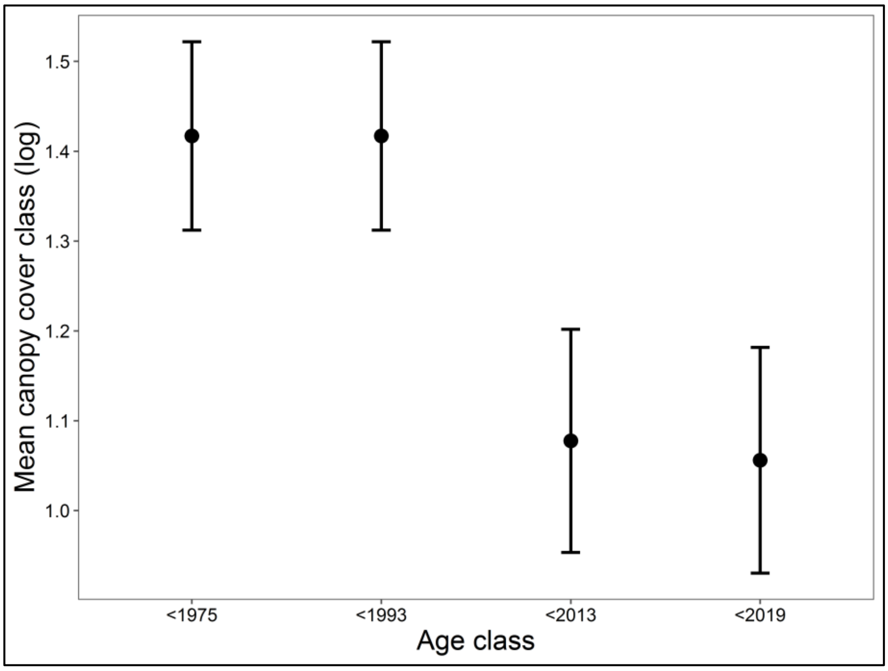


Figure 13: Log transformed estimated marginal means for canopy cover class (1-5) separated by age class. The age classes are based on the year of clear felling where age class <1975 represents sites cut before 1975, <1993 represents sites cut between 1975-1993, <2013 represents sites cut between 2004-2013 and <2019 represent sites cut after 2014. The error bars show the 5% and 95% confidence interval for the individual mean values.

4. DISCUSSION

4.1 Differences in deciduous tree and shrub species in riparian zones of different ages

In general, I found that the introduction of the SMOs have not yet had a positive effect on the deciduous tree species composition or other measured functions in riparian zones. The time since buffer creation was found to affect both the number of deciduous tree stems and shrubs as well as their volume, but deciduous tree and shrub count as well as percentage of deciduous trees did not positively change in age class corresponding to the implementation of the SMOs (<2019). Hence, I rejected my first hypothesis that there would be higher number of deciduous tree and shrubs in the riparian buffers that were created after the introduction of the SMOs.

I found that buffers created after the introduction of the SMOs had, in fact, lower deciduous stem densities than buffers of both <1993 and <2013. But when the percentage of deciduous stems was analyzed, I found that buffers of age class <2013 had higher proportions of deciduous stems than buffers of age class <1975 and <2019 which rejected my first hypothesis but partly supported my third hypothesis. The lack of deciduous trees in buffers of the oldest age classes could be related to the suppression of deciduous trees using herbicides in the 1950’s to 1970’s (Östlund et al. 1997). Further, smaller amounts of deciduous trees in the most recently created buffers could be that the majority of the trees in the younger buffers (<2013 & <2019) may not have reached 1.3 m in height, but also the history of the old (previous) stand before clear-cutting. According to Ackzell (1994), naturally regenerated deciduous trees in planted spruce stands in northeastern Sweden will only have reached a height of 107 cm after 10 growing seasons post-planting. When exploring the role of forest history on my results, it may be that the trees in the most recent buffers are still experiencing a legacy effect of previous forestry actions that suppressed deciduous trees. Examples of these forestry actions could be cleaning and thinning before SMOs and/or implementation of the 1993 Forestry Act. Thus, the probability that there would be large amounts of deciduous trees >1.3 m in the buffers of <2019 is, in fact, quite low.

Given that deciduous species take more than 10 years to reach heights of > 1.3 m (Ackzell 1994), is not surprising that deciduous trees were not proportionally greater in the younger age classes. But, using deciduous shrub counts I found that riparian buffers from the age class <2013 had a higher amount of deciduous shrubs than buffers of age class <1975 and just slightly higher than those of <2019. The difference in age between the two younger age classes (<2013 & <2019) will have had an effect on the succession. The more recently cut sites would have had less time to colonize with new trees after the clear-cutting event. The coniferous trees in the stands created after 1975 will most likely have been regenerated through planting (Esseen et al. 1997, Hallsby 2013), but broadleaved tree species will all have been regenerated naturally since planting of deciduous trees is not common practice in Sweden (Hallsby 2013). According to Karlsson (2001), seedling establishment takes about 3-5 years after germination. Further, wind throws could affect the deciduous species success as the felling of trees creates canopy gaps and microclimates favorable for regeneration (Kuuluvainen 1994). The risk of wind throws are higher in newly created buffers (Zeng et al. 2004), but it could be that the sites cut very recently haven’t yet been exposed to high wind loads or that time is needed for seedlings to establish and grow. Hence, because of seedling establishment and wind throws, the most recently cut stands (<2019) may not have had the time to establish new trees while the <2013 sites have had the time.

Stand history will also determine the outcome of the new stand (Ring et al 2018). It is likely that the <2019 riparian buffer age class is similar in structure and function to the <1975 sites riparian forest. The forest remaining in buffers of the <2019 age class were established a long time ago (70+ years) and likely underwent similar management steps as the forests in the <1975 age class. Further, since the <2019 age class sites were harvested so recently they have not had time to recover (Figure 1). Already at the time of clear felling, small trees and shrubs (but also large trees in the buffer) in the old stand will be legacy trees in the new stand (given that they survive changed environmental conditions and large trees are not blown down; Gustafsson et al. 2012). These seedlings will be the ones that can withstand competition for light (mostly shade tolerant species, i.e. spruce; Leemans 1991) and not light demanding pioneer species such as birch and aspen (Esseen et al. 1997). This is because succession in boreal forests typically proceeds from shade-intolerant species in early successional stages towards shade-tolerant species in the latter successional stages (Leemans & Prentice 1987; Bergeron & Dubue 1989). Thus, since the <2019 stands will still be more similar to the old stand, and have not had time to establish new seedlings, it is logical that the two younger sites have different amounts of deciduous shrubs and small trees.

In addition to number of stems, I also analyzed volume to determine if there were not only more stems, but bigger deciduous trees in the riparian zones of the most recently created buffers (<2019). I rejected my first and third hypotheses because deciduous volume per hectare was highest in the two oldest age classes. There was a trend that the volume of deciduous trees was higher in buffers from <2019 than <2013, suggesting that the introduction of the SMOs have had a positive effect on the deciduous volume in riparian buffer zones (Figure 11A). However, this visual pattern was not statistically supported. Ring et al. (2018) found a lower volume of deciduous stems in younger sites, but their findings were also not significantly different. The most likely explanation to the patterns that both Ring et al. (2018) and I found is that the older age classes will have older, and most likely bigger trees than the younger age classes due to harvesting within the riparian zones.

Altogether, I found that there are in fact deciduous tree species present in riparian zones that can be encouraged during future management steps. Based on my results, the introduction of the SMOs have not yet showed any significant increase in deciduous tree species in the riparian zones. But, as proposed by other authors (e.g. Hylander 2004; Oldén et al. 2019) and the SMOs (Andersson et al. 2013), active forestry operations such as thinning and pre-commercial thinning could make it possible to increase the amount of deciduous trees by selective cutting of coniferous trees in riparian zones.

As there were no significant increases in deciduous tree species after the introduction of the SMOs, it would be interesting to do a follow up study at least 10 years after 2013 to see if the results differ from what I found. I believe that there is a time lag, and that the most recently cut sites will have higher amounts of deciduous trees in a couple of years when the new stand has been established. Another likely reason for the lack of deciduous trees in the youngest buffers is that they still are affected by historical management, such as cleaning and thinning of deciduous trees. A study by Lidman et al. (2017) confirms that short-term effects of forest management as well as the longer-term results of successional change in riparian zones shape broad-scale supply of services to streams (in their case litter supply). In order to encourage deciduous trees in future management, deciduous stems need to already exist in the stands at the time of management. As my results show, the oldest sites (<1975) are the ones with the least number of deciduous trees. Hence there may still be a legacy effect on historical management that decreases the potential to encourage deciduous species in riparian zones.

I found no significant shifts in deciduous tree and shrub composition with regard to distance from stream related to age class. I can therefore not draw any conclusions whether the implementation of riparian buffer zones have affected the composition of the inventoried buffers.

4.2 Diversity of woody species

In general, species richness tends to be high in riparian zones (Gregory et al. 1991; Naiman & Decamps 1997). Differences in richness of woody species in riparian zones could be related to different factors such as disturbance, productivity (Pollock et al. 1998), dispersal of propagules in streams (Nilsson et al. 2010) or position in the river network (Kuglerová et al. 2015). I hypothesized that the SMOs would have had a positive impact on species richness, and that the most recently created buffers (<2019) would have highest species richness. However, this was not what I found. The species richness was highest in the two intermediate age classes (<1993 and <2013). Again, I found a pattern suggesting that the sites from the oldest and youngest age class are more similar in structure. A study by Hasselquist et al. (2015) investigated recovery of plant species richness after restoration and found that species richness increased with time. It may be fair to draw the conclusion that time also affects the recovery of woody species in riparian buffers. My findings suggest that it would take a minimum of 6 years for the woody species in riparian zones to “recover” since the species richness was found to be high already in my age class that was 6-15 years after cutting (Figure 12). My findings of high species richness in the riparian buffers of the two intermediate age classes (<1993 & <2013) can be related to a study by Pollock et al. (1998). They found that riparian areas with intermediate amounts of disturbance and productivity had the highest species richness and that these results supported Michael Huston’s dynamic-equilibrium model of species diversity (Huston 1979, 1994). Further, the intermediate disturbance hypothesis (Connell 1978) states that the highest species richness is obtained in intermediately disturbed systems. The level of productivity on my sites is unknown, while the amount of disturbance can be related to the time since forest management operations (Figure 1). It is probable that the most recently created buffers (<2019) still have site conditions very similar to relatively old growth forests (<1975). The buffers in the oldest age class are presumably in a more steady successional and non-disturbed state than the <1993 and <2013 buffers since they are in the state between thinning and final felling (Figure 1). It is however important to understand that my analysis of species richness was only focused on the shrubs and small trees and hence only comprise the newly established or understory stand.

In addition to differences among age classes, I found that the distance to stream also affected the species richness in riparian zones. The zone of 0-5 m from the stream had the highest species richness. The higher species richness closer to streams has also been shown by many other studies that also point out that the increased diversity is due to the dynamic conditions in riparian zones. (Gregory et al. 1991; Naiman & Décamps 1997; Sabo et al. 2005; Ström et al. 2012; Kuglerová et al. 2014).

4.3 Other services provided by riparian buffers

I hypothesized that the introduction of the SMOs from 2013 would have made a significant positive shift in the ability for riparian zones to provide services such as deadwood or shade. Also, I hypothesized that the older forests would currently provide shade due to their age. But because the SMOs now require stable shade, I expected the newest buffers to provide similar canopy cover as older buffers. I also hypothesized that the SMOs would have had a larger impact on these services provided by riparian zones than the 1993 Forest Act.

I found no significant differences for deadwood volume among age classes or distance to the stream but I found the highest volume of deadwood within the two youngest age classes (<2013 & <2019) and the lowest volume in the age class <1993 (Table 2). This supports my hypothesis and suggests that the 1993 Forestry Act and SMOs of 2013 have had a positive impact on the ability of riparian buffers to provide deadwood. Other studies that have investigated downed deadwood in riparian zones found that age class had a significant effect on the volume. Both Fridman & Walheim (1999) and Ring et al. (2018) found an increasing volume of deadwood with increasing age in managed forests, contradicting my findings. It is important to understand that the buffers have originated from forests that are in similar age or older than the <1975 buffers which would mean that there is a legacy effect also here. Hence, high volumes of deadwood in mature stands and a legacy effect in buffers could explain higher volume of deadwood in the oldest (<1975) and youngest (<2019) sites. Furthermore, results from the total deadwood count partly agrees with my hypothesis. As for deadwood volume, there were no significant differences in total count of deadwood pieces among age classes. However, I found that the highest number of deadwood pieces per hectare occurred in buffers of the youngest and oldest age class (<2019 & <1975) and lowest amounts in buffers from the next oldest age class (<1993) (Table 2). Hence, the SMOs seem to have had a positive effect on the total number of deadwood but not the 1993 Forestry Act. Even though the results for blowdown deadwood were not significantly different among age classes, the trends supported my hypothesis. I found the highest amounts of blowdown deadwood within buffers of the two youngest age classes (<2013 & <2019), and in the <1993 buffers I found no blowdown deadwood at all (Table 2). The pattern of higher amounts of blowdown deadwood in the two youngest age classes is likely related to the fact that the risk of wind throws are higher in newly created buffers (Zeng et al. 2004). Additionally, the patterns shown are likely also an effect of past management. The <1993 age class did most likely not have any riparian buffer left after harvest (Ring et al. 2018) and hence no trees left that could be recruited as deadwood through wind throws. Also in the <1975 sites, foresters most likely harvested all the large trees in the riparian zones because of no regulation that demanded riparian buffer retention at the time (SFS 1979:429) and thus there was very little to be blown down. The blown down trees in the <1975 sites are the ones which have been planted, grew large and had time to blow down in the closed forest. It could be that forestry operations such as thinning have caused wind throws in the oldest sites since thinning increases the risk of wind throws (Valinger & Fridman 2011).

I hypothesized that riparian buffers created after the introduction of the SMOs would provide shade to the streams similar to older sites. I have evaluated canopy cover that is the density of the canopy which determines the available light reaching the stream (Lecerf et al. 2016). The ability of riparian buffers to provide light conditions similar to old stands (buffers from <1975) to streams was not fulfilled in buffers created after the introduction of the 1993 Forest Act (buffers from <2013) or the SMOs (buffers from <2019) and hence my hypothesis was rejected. As a number of studies have shown, clearcutting may reduce the canopy cover even within the riparian zone and thereby decrease the strength of the forest to act as a light filter to the stream (Ahtiainen 1992; Groom et al. 2011). The two older age classes had mature forest developed and thus had higher canopy cover above the stream.
One factor that influenced the canopy cover measurements in this study was the timing of the inventories. It would have been favorable to do the inventories during the summer where there are no risk that the deciduous trees will have dropped their leaves.

The goal within the SMOs to have buffers that provide stable shade to streams is currently not being fulfilled. There is a gap of at least 6 years after clear-cutting where the canopy cover in the buffers are significantly lower than the relatively old forest buffers (<1975). In order to fulfill the goals of the SMOs that buffers should provide stable shade, managers need to create wider, or denser buffers or settle for a window of time where the canopy is not dense enough to provide stable shade to streams. In addition to canopy cover, it seems as the introduction of the 1993 Forestry Act and SMOs in 2013 have positively affected the ability of riparian buffers to provide deadwood to streams. Note that management proposals are not for trying to mimic what old growth forests would look like, the goal with the SMOs is only to target the buffer important functions.

4.4 Conclusion

It appears that the forest management after the introduction of the 1993 Forestry Act has had a positive effect on deciduous cover in riparian buffers. However, there is no evidence so far that the introduction of the SMOs has contributed to improved buffer management and therefore the ecological functioning of those buffers directly after clear-cutting. The <2013 age class seems to be delivering most of the services that were outlined in the SMOs, except for standing volume. However, if the <2019 sites are representative of what the <2013 sites were like right after their creation (clear-cutting), there is a time lag of at least 6 years where the goals of the Objectives are not met. Hence, the SMOs have not yet had a positive effect on deciduous tree species composition or other functions in riparian zones.

A pattern that was reoccurring in my results was the similarity in characteristics between the youngest and oldest sites (<2019 & <1975). This pattern could be evidence that there is, in fact, a lag in time where the buffers have not had time to recover from historical management. If the goal is to have stable conditions and continually have ecologically functioning buffers adjacent to streams, I would suggest a change in current management practices of riparian zones. One way to do this could be to leave wider buffers and selectively log coniferous trees within that buffer strip to compensate for economic loss. Leaving a wider buffer provides shade, and logging of some coniferous trees creates a diverse riparian buffer zone that encourage deciduous tree species.

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SENASTE UTGIVNA NUMMER

2018:8	Författare: Rasmus Behrenfeldt Vindens inverkan på höjdtillväxten i ett tallbestånd (*Pinus sylvestris*) längs en sluttning
2018:9	Författare: Erik Sundström Brandhårdhetens påverkan på knäckesjukans omfattning på brandfältet i Sala
2018:10	Författare: Jenny Dahl How is soil carbon stock in old-growth boreal forests affected by management?
2018:11	Författare: Johannes Larson Know the flow – spatial and temporal variation of DOC exports and the importance of monitoring site specific discharge
2018:12	Författare: Sanna Nilsson Hur tidpunkten för och samordningen av föryngringsåtgärder påverkar föryngrings-resultatet och konkurrenstrycket i plantskogen
2019:1	Författare: Lina Arnesson Ceder Skogshistoria kommer upp till ytan – en akvatisk inventering efter samiskt påverkad död ved i tjärnar kring Mattaur-älven
2019:2	Författare: Linda Norén "Det var ett äventyr" – en studie om livet som flottare efter Piteälven
2019:3	Författare: Elin Edman Bladyta och virkesproduktion i fullskiktad granskog skött med blädningsbruk
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2019:6	Författare: Elias Hannus Beslutsstöd för att finna diken och bedöma behov av dikesrensning
2019:7	Författare: Jan Lindblad The future of retention forestry – the historical legacy in stands and its impact on retention in the next generation
2019:8	Författare: Hilda Mikaelsson Alternative oxidase respiration in the mycorrhizal fungus *Laccaria bicolor*
2019:9	Författare: Joel Jensen Above- and belowground carbon stocks and effects of enrichment planting in a tropical secondary lowland dipterocarp rainforest
2019:10	Författare: Josefin Runesson Total carbon sequestration during an entire rotation period of oil palm in northern Borneo
2020:01	Författare: Mikaela Rosendahl Fysiska och psykiska hälsoeffekter av att vistas i naturen – En pilotstudie utförd på Stora Fjäderägg, Västerbottens län