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Are riparian buffers surrounding forestry-impacted streams sufficient to meet key ecological objectives? A Swedish case study

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ABSTRACT

In many national guidelines and policies regarding protection of freshwater systems from stressors associated with forestry, riparian buffer width is a commonly prescribed strategy, typically with no other refinements of protection measures. In Sweden, the Strategic Management Objectives (SMOs) were developed to ensure that riparian buffers that are left after harvesting sustain important ecosystem attributes in aquatic systems, referred to as objectives, namely shading, biodiversity, reduction of sedimentation, and provision of deadwood and food. However, little specification is given on threshold targets or how to manage riparian zones to effectively provide these objectives. In this paper, we evaluated whether existing riparian buffers of different widths along small, recently harvested (<8 years) streams were able to provide proxies of these targeted objectives, and further compared harvested streams to counterparts situated in mature unharvested production forests (reference) in northern and southern Sweden. The influence of buffer width varied with objective and geographic location. In both regions, canopy cover (proxy for shading) increased with riparian width, and riparian deadwood was highest in no buffer sites. Organic matter (OM; proxy for food) was highest in the northern no buffer streams, while in the south OM increased with buffer width. All other parameters tested had no relationship to buffer width. These differing responses even in streams subjected to similar land-use and management within a close vicinity and region, suggest that the contemporary strategy of prescribing fixed buffer widths and/or stating objectives without defined guidelines for what constitutes an effective riparian buffer is insufficient given the large variability of stream ecosystems across small spatial scales. More comprehensive consideration synergistically accounting for site-specificity and land mosaic planning are needed to develop functionally effective buffers that can mitigate forestry impacts on stream ecosystems.

1. Introduction

Riparian zones are vital to regulate ecological functions and connections between terrestrial and aquatic environments (Naiman and and Décamps, 1997; Richardson and Sato, 2015). They are important habitats for maintaining biodiversity, regulating sediment and nutrient transport and providing shading and resource subsidies for organisms (Chellaiah and Yule, 2018; Moore et al., 2018). Forestry operations such as harvesting including removal of riparian trees, and/or driving heavy machineries within riparian zones can cause negative impacts on aquatic ecosystems with documented alterations to temperature regimes, sediment fluxes, nutrient runoff and biodiversity (Moore et al., 2005; Löfgren et al., 2009; Richardson and Béraud, 2014; Oldén et al., 2019a). As such, retention of riparian buffers along water bodies such as streams, rivers and lakes in managed landscapes is often advocated to mitigate the impact of forestry practices on freshwater ecosystems (Richardson et al., 2012). Of these, small streams are typically neglected due to their small size and underestimation of their relative importance in maintaining river health and water quality (Richardson et al., 2012; Kuglerová et al., 2020). However, as small streams represent 70–80% of total river network length in forested regions (Ågren et al., 2015) and provide water, energy, biogeochemical constituents and biodiversity to downstream ecosystems (Moore et al., 2018; Coats and Jackson, 2020), improper riparian management surrounding small streams can significantly alter ecology at local and whole-catchment scales.

In accordance with the European Union (EU) Water Framework Directive (WFD, 2000/60/EC) adopted in 2000, all waters within the EU should experience no further degradation and achieve good ecological and chemical status by 2027. To meet this, each EU country employs varying policy approaches concerning riparian buffers around surface

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waters. In Sweden, the majority of production forests are affiliated to forest certifications like PEFC (Programme for Endorsement and Forest Certification) and FSC (Forest Stewardship Council) that requires protective buffer zones along surface waters. However, the required buffer width is not specified neither in the certification nor other legal documents (The Swedish Forest Act). As Sweden is dominated by managed forests (~51% of land surface area is productive forest land; Skogsstyrelsen, 2020) for the production of timber, pulp and bio-energy, its forested landscape is a mosaic of single-species dominated stands of even-aged trees that are subjected to forest rotation practices such as thinning, clearcutting and site preparation for the establishment of the next tree generation (Angelstam et al., 2020; Kuglerová et al., 2021). Hence, the establishment of riparian buffers are vital to reduce forestry impacts (Boothroyd et al., 2004; Jyväsjärvi et al., 2020). With increased demands on forest productivity to increase extraction of woody biomass (Swedish Forestry Act, 2018), voluntarily set-aside areas such as riparian buffers are particularly vulnerable to reduced protection. This is seen in current practices as riparian buffers along small streams are typically composed of 1–2 rows of trees (<5 m wide) and they are mostly designed with fixed-widths (see Kuglerová et al., 2020). The ecological functionality of such buffers is questionable, especially because narrow buffers tend to blow down (Mäenpää et al., 2020).

There are several guidelines set by the Swedish Strategic Management Objectives (SMO, (SMO, Andresson et al., 2013) regarding the establishment of riparian zones and what functions they should provide, namely; to a) preserve important soil chemical processes and element transformation; b) act as a filter for groundwater-transported substances and for sediment transport from upland, in addition to stabilizing shoreline to prevent erosion and sediment transport downstream; c) contribute food to aquatic organisms through falling leaves and insects; d) provide stable shade to streams over time to minimize summer temperature fluctuations in water; e) contribute deadwood to the water; and f) preserve aquatic and riparian biodiversity. However, criterions for the management of such ecologically functional buffers and target values for the required objectives are not specified by the SMOs, raising concern on adequate riparian protection through current guidelines. The effectiveness of riparian buffers to protect stream ecosystem function and structure depends on riparian conditions such as buffer width, slope, vegetation composition, density and structure (Broadmeadow and Nisbet, 2004; Chellaiah and Yule, 2018) as well as catchment-scale differences (e.g, % of harvest within a catchment and geographic location) that influence soils and sediment delivery processes, and water chemistry, hence biological communities and ecosystem functioning (Bowker et al., 2020; Zhang et al., 2009). Nevertheless, fixed-width buffers are the standard prescribed strategy for riparian management in many national guidelines and policies for protection of freshwater systems from stressors associated with forestry (Richardson et al., 2012). The establishment of fixed-width buffers as the go-to management strategy is based on little evidence with many uncertainties due to the scarcity of actual tests, evaluations and monitoring of the effectiveness of implemented widths to achieve ecological functions. Although this is not prescribed in the SMOs, fixed-width are the prevailing strategy in the Swedish forests too (Kuglerová et al., 2020). As our understanding of the best management practice to achieve the SMOs targeted ecosystem objectives is limited, it will be challenging to elucidate if the existing regulations and practices are sufficient to preserve stream ecosystem function and processes.

In this paper we investigate the ability of riparian buffer width to mitigate forestry impacts on headwater ecosystems compared to unharvested, mature and managed production forests (reference). We aim to assess how existing riparian buffers of varying widths along recently harvested (3–8 years) small streams in Sweden are able to achieve the SMOs targeted objectives in terms of shading (and temperature fluctuation responses), sedimentation, food, deadwood provisions and biodiversity. We measure six parameters which we use as proxies for these five objectives: Namely riparian canopy openness is used as a reverse measure of shading, proportion of stream bottom covered by fine sediments is a snapshot of sedimentation, volume of in-channel particulate organic matter (POM) is a measure of food provision, volume and percentage of large deadwood in stream channels and riparian areas respectively are measures of deadwood provision, and diversity of aquatic macroinvertebrate species is a measure of biodiversity. We selected proxies that we deem can be practicable by foresters for field-site evaluation to estimate stream conditions based on simple methods and where possible, "snapshot" evaluations. Although wider riparian buffer width is unarguably important to sustain higher levels of several riparian ecosystems attributes e.g. microclimate, stand structure, and biodiversity of plants, vertebrates and invertebrates (Brosofske et al., 1997; Less and Peres, 2007), this is not always true for the adjacent stream ecosystems due to upstream connections in the drainage network and catchment-scale influences. Therefore, our hypothesis is that not all objectives stated in the SMOs will be satisfied only by increased buffer width. More specifically, we predict that shading, POM and deadwood will increase with wider buffer widths since their major source is the adjacent riparian forests. In contrast, sediments and aquatic biodiversity may have no improvement from wider buffers since sediment is mostly related to catchment-scale properties, also having deleterious effects for the aquatic fauna (Juvigny-Khenafou et al., 2021). We also predict that shading, POM and deadwood will be higher in reference and buffered sites in comparison to no buffer sites, while sediments and aquatic biodiversity might not differ between reference, buffer and no buffer sites due to the upstream effects on these sites. This paper is the first that we know off to investigate the influence of buffer width in providing the targeted objectives highlighted in the SMOs, established by the Swedish Forestry Agency in 2013 (Andresson et al., 2013).

2. Materials and methods

2.1. Study sites

Stream and riparian conditions were measured from Aug - Oct 2018 at 24 locations (Table 1) in northern and southern Sweden within the Västerbotten and Jönköping counties, respectively (Fig. 1). The annual average temperature and annual precipitation were 4.2 °C and 7.5 °C and 477 mm and 606 mm respectively for the northern and southern sites in 2018 (Swedish Meteorological and Hydrological Institute, 2021). The geology of our studied area within northern and southern Sweden is dominantly made up of postglacial till (moraine) and medium-grained glaciofluvial sediments with some sites situated on clay-silt deposits or bedrock (Table 1; Sveriges Geologiska Undersökning, 2020). In each county, twelve small streams (1st – 3rd order; 0.18–6.06 km^2 catchment area) were selected to represent no buffer, buffer and reference streams (mature unharvested sites). The streams were on average 0.57–3.18 m wide (mean channel width: north = 0.96 m, south = 1.34 m). All harvests were performed between 2010 and 2017, and most buffer left behind are remnants of previous mature production forests that were 60–80 years old. Streams were selected to be as similar as possible in their physical conditions including size (width), channel form, substrate cover, gradient and upstream source (absence of lake or wetland within 1 km upstream), and no harvesting within buffer (if present). The main variable, which was selected to vary was buffer width and the emphasis was put into finding sites with buffer of different widths but similar tree species and age composition (that is dominated by mature spruce, Table 1). Nevertheless, some variations in the physical conditions were unpreventable in order to find comparable and recently harvested sites with buffers (see Table 1) that met the above criterion. We note that variation in stream, riparian and catchment-scale conditions can influence measured parameters tested in this study, however we focused on buffer width as an explanatory variable in our analyses, rather than on a suite of physicochemical effects reported elsewhere (Burrows et al., 2017; Jonsson et al., 2017; Lidman et al., 2017) given that buffer width is a common one-size-fit-all strategy to mitigate land-use change impacts

Table 1
Site characteristics of each studied streams.

Site	Lat	Long	Catchment area (km^2)	Site elevation (m)	Channel slope(°)	Velocity (m/s^{-1})	Clearcut year	% harvest in catchment	Buffer width (m), mean (min, max)	Basal area (m^2/ha)	Conifers (%)*	Sapling count§	Underlying geology
No buff 1N	7,134,570	729,984	0.56	273	8	4.88	2016	0.14	1.93 (0, 6.6)	2.49	80	3	Till
No buff 2N	7,107,696	754893.8	1.04	90	2	4.52	2012	0.27	0.30 (0, 2.4)	0.63	100[a]	120	Clay-silt
No buff 3N	7,116,804	760626.2	0.23	104	3	5.75	2013	0.01	0	0	0	8	Till
No buff 4N	7,123,301	713283.9	1.00	235	3	4.29	2014	0.13	1.25 (0, 7)	3.26	38	28	Till
Buff 1N	7,074,344	737,654	2.33	63	4	11.53	2012	2.02	3.96 (1.2, 5.6)	11.04	100	24	Glaciofluvial sediments
Buff 2N	7,129,930	767642.1	0.90	136	5	7.72	2017	0.01	3.38 (1, 6.4)	12.95	82	32	Clay-silt
Buff 3N	7,080,087	734618.4	0.68	124	4	9.91	2014	0.02	5.95 (2.8, 8.3)	27.02	81	27	Glaciofluvial sediments
Buff 4N	7,118,458	716183.5	1.33	214	8	7.34	2015	0.02	4.21 (1, 7.6)	10.45	94	33	Till
Buff 5N	7,137,541	726871.9	0.66	281	5	3.00	2016	0.09	13.05 (7.1, 19.9)	25.99	64	21	Till
Ref 1N	7,076,529	739086.5	1.13	81	4	3.38	NA	0.06	NA	49.63	92	11	Till
Ref 2N	7,116,637	704881.6	0.18	217	4	8.54	NA	0.00	NA	27.81	67[a]	58	Till
Ref 3N	7,133,310	732966.1	0.52	211	4	5.16	NA	0.00	NA	55.52	91	63	Till
No Buff 1S	6,399,270	428005.9	1.33	229	2	2.67	2013	0.35	0.15 (0, 0.8)	0	0	88	Glaciofluvial sediments
No Buff 2S	6,384,548	432075.9	1.87	244	4	2.05	2010	0.34	0	0.15	50[a]	120	Till
No Buff 3S	6,364,885	463789.1	1.93	222	4	7.33	2014	1.12	0.13 (0, 0.5)	0.22	75[a]	63	Till
No Buff 4S	6,363,954	420605.6	6.06	178	5	1.33	2015	5.30	1.14 (0, 3.7)	15.87	67	28	Bedrock
Buff 1S	6,391,863	425953.3	1.03	313	3	9.39	2010	0.48	18.90 (0, 40.5)	17.94	93[ab]	78	Till
Buff 2S	6,375,730	457,451	2.66	196	7	1.87	2017	0.71	3.20 (0, 10)	14.75	22	54	Glaciofluvial sediments
Buff 3S	6,403,182	435222.9	1.17	207	3	2.72	2016	0.01	20.95 (0, 47.9)	21.34	40	16	Glaciofluvial sediments
Buff 4S	6,392,489	427663.5	3.95	234	6	1.73	2011	3.89	26.70 (13.1, 38.4)	34.62	89	41	Bedrock
Buff 5S	6,362,652	421136.5	1.40	172	4	3.97	2015	0.54	35.40 (0, 80)	21.29	70	8	Till
Ref 1S	6,386,782	423050.2	1.30	276	4	1.62	NA	0.16	NA	33.06	100	50	Till
Ref 2S	6,388,691	447394.5	2.73	219	4	3.12	NA	0.83	NA	24.13	32	19	Glaciofluvial sediments
Ref 3S	6,383,388	474,671	0.50	320	4	3.42	NA	0.02	NA	42.76	88	18	Till

Note: Sites with N in the title represent sites in the northern part of Sweden while sites with S are from the south. Buff = buffer. Latitude and longitude are in SWEREF99 coordinate system. We categorised some sites as No buff even though they have a few riparian trees as these trees were only randomly distributed and did not form a continuous riparian cover. Buff 3S and 5S had wider buffers only on one side of the stream. Conifers are predominantly mature trees (60–80 years old) unless denoted otherwise.

* Remaining % are deciduous trees.
§ Total sampling count for the 4 riparian plots (400 m^2).
a young spruce saplings only.
ab mix of young and mature conifers.

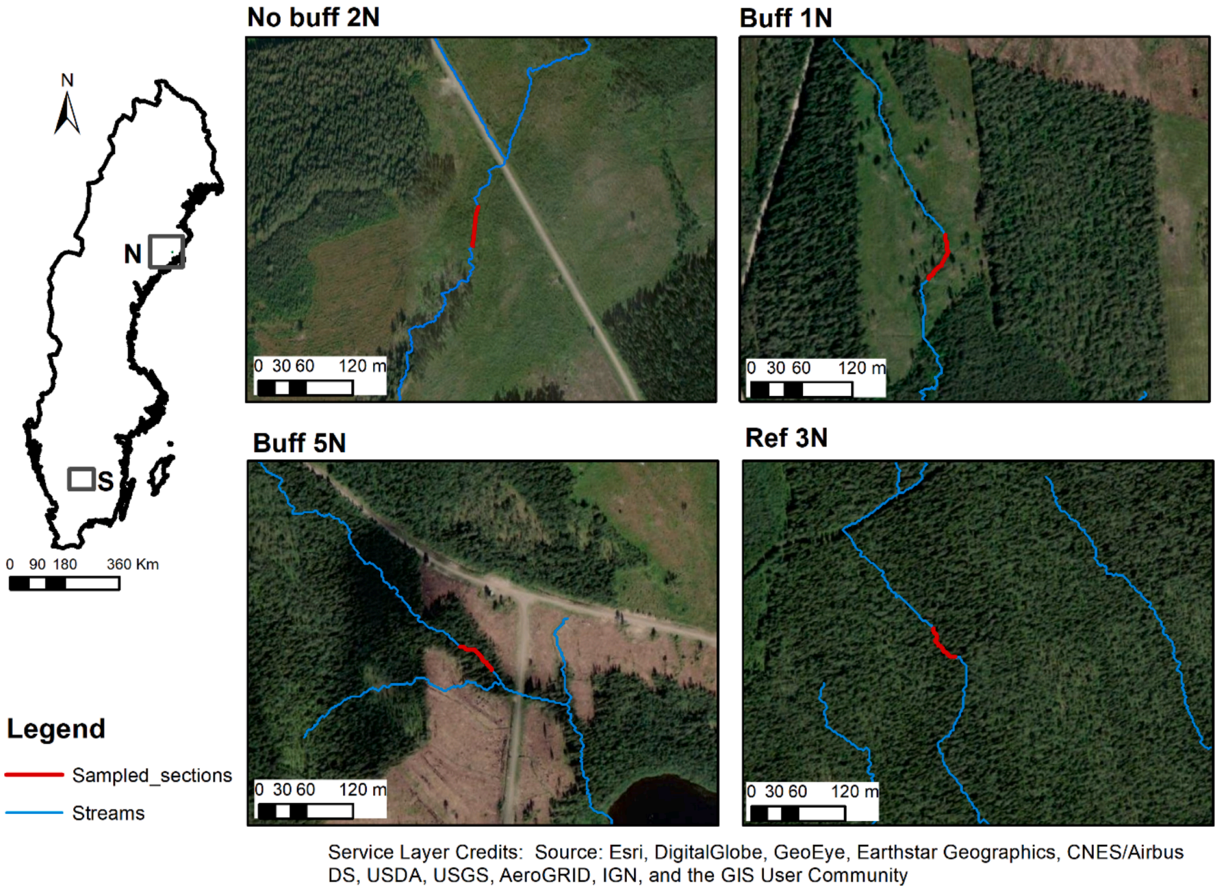

Fig. 1. Locations of study sites within the Västerbotten and Jönköping counties in the north and south of Sweden are indicated by the black squares on the map to the left. Four examples of sampled stream sections (in the red) in the north and the buffers around them are displayed over satellite images. All catchments and sampled stream sections can be found in Supplementary material (S).

on streams (Ring et al., 2017). The buffer conditions for this study were specifically selected to cover typical buffer practices in the two counties (see Kuglerová et al., 2020) from no buffers to a range of buffer widths (Table 1). Buffers were rarely wider than 15 m on each side of small streams, therefore to include wider buffer conditions, we also included some streams that were harvested only on one side of the stream (2 sites, Table 1), while the other side was unharvested. Reference streams selected were within managed production forests, not harvested in the past 60+ years, with riparian zones dominated by homogenized mature coniferous forests that have been historically subjected to forestry practices such as clear-cutting and thinning (Hasselquist et al., 2021). Given the scarcity of old growth forest sites within most parts of Fennoscandia (Östlund et al., 1997), we intentionally placed our reference sites in mature production stands as it is necessary to compare recently harvested sites to managed reference sites because mature productive forest stands are the most common type of forests (>70% of forested area (Kuglerová et al., 2021) throughout Sweden. Further, the harvested streams are situated in managed production stands and therefore more closely resembled our chosen references prior to harvesting, rather than old-growth forests.

2.2. Sampling design

A 50 m long sampling reach (Fig. 1) situated in the most downstream part of the streams (but before passing under a road to avoid the effects of road-side ditches) intersecting with the clear-cut was selected. Along this reach, riparian buffer widths were measured at 8 transects perpendicular to the stream, 4 at each bank, 10 m apart. Riparian plots of 10 m × 10 m (n = 4 plots) were established alternatively on the left and right bank (2 plots per bank). In each riparian plot we measured DBH (diameter at breast-height) of all standing trees and categorized them into species. We also counted all saplings (<2 m tall, < 2 cm DBH) in each plot (all tree species together). We counted the number of deadwood pieces (>5 cm diameter) present in each riparian plot, and expressed it as % of downed riparian wood out of all retained wood in the riparian buffer (Mäenpää et al., 2020). Further, we summed the volume of all large wood (Polvi et al., 2014) within the stream channel at each reach (henceforth in-stream deadwood) and expressed in as volume per area (m^2). At every 5 m along the stream reach, a snapshot visual estimate of fine sediment was assessed as the proportion fine material (<2 mm) within 50 × 50 cm quadrats placed on the stream bottom (n = 10). At these 10 locations, we also measured channel width. To measure canopy openness, a spherical densiometer was used at every 2.5 m intervals, taking one measurement for each cardinal direction per interval (n = 76) from the centre of the stream. Surber samples (n = 5) were collected from stream bottom at riffles at each site to measure stream standing stock particulate organic matter (POM) and aquatic macroinvertebrates that were stored in 70% ethanol until further analysis. Total POM were measured from filtering the surber sample through a 1 mm sieve and determined as ash-free dry mass (AFDM) after drying (60 °C for 48 h) and ashing (550 °C for 4 h). Aquatic macroinvertebrates were sorted and identified to the lowest taxonomic level, typically species or genus. At each stream, air and water temperatures were continuously measured hourly from Jul-Oct 2018 using the HOBO® pendant loggers (Onset Computer Corporation, Bourne,MA, USA) attached at ~10 cm stream depth to a metal rod embedded into the stream bed. Due to the unexpected heat wave and limited rain in the summer of 2018, most selected streams underwent phases of drought during Jun-Aug 2018, however by sampling time, water returned to normal flow levels for the season. We also recorded current velocity as the time it took for fluorescent fluid to travel a 3 m distance in each stream.

2.3. Statistical analysis

Due to climatic, geologic and forest management differences across Sweden (Kuglerová et al., 2020), we analyzed the southern and northern streams separately. We conducted linear regression analyses to test the relationships of buffer width (continuous explanatory variable) with six measured stream and riparian parameters (dependent variable) used as proxies for the five targeted objectives (we measured deadwood in the stream and riparian area). Each parameter was analyzed separately (in total 12 separate regression models for the five objectives). In these analyses, mature, unharvested sites were not included in the regression analysis as buffer width was non-applicable. We were looking for a regression type that fits our data and best explains the trend between buffer width and the specific parameter. Typically, we tested liner, quadratic and polynomial trends and report here the relationships with the highest r^2 (best fit).

Additionally, we categorised our sites into 3 categories based on buffer width as no buffer, buffer and reference (Table 1). For each of the 4 parameters (canopy openness, POM, riparian deadwood, sediment cover) we created linear mixed effects models (LMM) using buffer type as a fixed factor to test if these parameters differed between categories. We used LMMs because we had multiple samples for these measured parameters at each site, and kept 'site' as a random factor. All corresponding P-values and degrees of freedom were estimated using the likelihood ratio test. We only had one measurement at each site for 2 parameters, total in-stream wood volume (recorded along the entire stream reach) and aquatic macroinvertebrate Shannon diversity (Ḣ) (to account for taxon richness in relation to their abundance), hence we used one-way ANOVA, followed by Tukey post-hoc test, to test for differences between the categories. We also ran ANOVA analyses to assess daily fluctuation of summer stream water temperatures for selected streams from each region that represents no, narrow and wide buffer as well as reference sites. Only selected streams were used because many of the streams experienced drought during the summer months. All data and models were checked for normality and homogeneity of variances, and logged transformed prior to analysis when necessary. All statistical analyses were performed in RStudio (R Development Core Team, 2019) with the packages lme4 (Bates et al., 2014), and vegan (Oksanen et al., 2016), while ggplot2 (Wickham, 2016) was used to plot the results.

3. Results

3.1. Objective 1: Shading

In both regions, canopy openness (reverse proxy for shading) significantly decreased with wider buffers (Tables 2 & 3; Fig. 2), the best regression fit being polynomial in the northern regions and logarithmic in the south (Table 3). Generally, canopy openness in both regions were similar in the reference site with 8.0–16.3% of canopy openness in the north and 5.3–13.9% in the south. Highest canopy openness were found

Table 3
Results of the regression models for the relationships between buffer width and selected riparian and aquatic parameters in the north and south of Sweden.

	Variable	Line of best fit	F	Adjusted R^2	p-value
North	Canopy openness	Polynomial	$27.70_{(2,6)}$	0.87	**< 0.001**
	POM	Polynomial	$1.77_{(2,6)}$	0.16	0.249
	Sediment	Polynomial	$0.09_{(2,6)}$	−0.30	0.918
	In-stream wood volume (m3/m2)	Polynomial	$0.27_{(2,6)}$	−0.22	0.772
	Riparian deadwood (%)	Linear	$3.94_{(1,7)}$	0.27	0.088
	Biodiversity (Ḣ)	Linear	$0.0012_{(1,7)}$	−0.14	0.974
South	Canopy openness	Log	$17.18_{(1,7)}$	0.67	**0.004**
	POM	Log	$9.14_{(1,7)}$	0.50	**0.019**
	Sediment	Polynomial	$0.47_{(2,6)}$	−0.15	0.646
	In-stream wood volume (m3/m2)	Polynomial	$0.35_{(2,6)}$	−0.20	0.720
	Riparian deadwood (%)	Polynomial	$0.79_{(2,6)}$	0.72	**0.010**
	Biodiversity (Ḣ)	Polynomial	$4.54_{(2,6)}$	0.47	0.063

Note: P-value in bold is significant.

Table 2
Mean ± SE of measured parameters selected as proxies for the targeted objectives in the stream and riparian area in studied sites. Values are displayed for each site separately.

Site		Shading	Resource subsidies	Deadwood		Sediment cover	Biodiversity
		Canopy openness (%)	POM (g/m^2)	In-stream wood vol (m^3/m^2)	Riparian Wood (%)	Fine sediment (%)	Shannon diversity (Ḣ)
North	No buff 1N	79.44 ± 2.50	143.07 ± 19.97	0.0042	79.00	20.50 ± 1.57	1.94
	No buff 2N	78.82 ± 2.35	176.43 ± 46.41	0.0003	25.00	24.00 ± 8.06	1.36
	No buff 3N	89.77 ± 0.59	68.42 ± 20.84	0.0012	75.00	11.00 ± 2.50	2.30
	Buff 1N	49.34 ± 2.67	26.21 ± 7.87	0.0038	44.89	42.22 ± 9.40	1.62
	No buff 4N	66.28 ± 2.13	142.02 ± 91.00	0.0122	67.86	0	1.69
	Buff 2N	38.06 ± 2.37	102.20 ± 45.22	0.0025	33.44	2.00 ± 1.33	0.75
	Buff 3N	18.18 ± 2.82	39.04 ± 10.47	0.0012	19.37	24.50 ± 8.18	1.18
	Buff 4N	36.43 ± 2.24	39.10 ± 4.30	0.0020	51.97	0.50 ± 0.50	1.90
	Buff 5N	8.86 ± 1.08	91.80 ± 17.62	0.0006	19.10	12.50 ± 2.81	1.95
	Ref 1N	7.98 ± 0.60	43.39 ± 7.26	0.0015	2.50	35.50 ± 4.44	1.22
	Ref 2N	8.43 ± 0.64	84.29 ± 18.99	0.0001	0	24.00 ± 8.09	1.82
	Ref 3N	16.32 ± 1.40	42.89 ± 8.26	0.0025	9.95	29.50 ± 7.10	2.11
South	No Buff 1S	80.51 ± 1.22	72.08 ± 25.65	0.0002	79.17	15.5 ± 3.02	1.87
	No Buff 2S	76.19 ± 2.72	91.12 ± 30.00	0.0010	83.93	31 ± 7.52	2.09
	No Buff 3S	77.42 ± 5.02	98.82 ± 46.38	0.0021	82.50	6 ± 3.40	1.93
	Buff 1S	48.3 ± 4.89	151.74 ± 46.85	0.0009	11.87	12 ± 5.54	2.38
	No Buff 4S	45.21 ± 3.65	149.29 ± 62.40	0.0004	48.81	2 ± 1.33	2.36
	Buff 2S	30.25 ± 5.30	142.08 ± 65.89	0.0014	31.84	43.2 ± 6.19	2.68
	Buff 3S	21.1 ± 2.92	135.35 ± 24.79	0.0003	26.49	50 ± 0	2.40
	Buff 4S	12.11 ± 1.27	118.11 ± 31.35	0.0006	26.75	18.5 ± 3.17	2.43
	Buff 5S	42.77 ± 1.77	154.88 ± 42.26	0.0014	54.04	15.5 ± 7.76	1.65
	Ref 1S	13.92 ± 1.41	101.58 ± 28.16	0.0000	5.00	27 ± 4.67	1.31
	Ref 2S	5.32 ± 0.41	137.67 ± 36.46	0.0012	4.58	47 ± 5.97	1.12
	Ref 3S	8.56 ± 0.58	249.59 ± 43.46	0.0000	21.10	16.5 ± 4.09	2.22

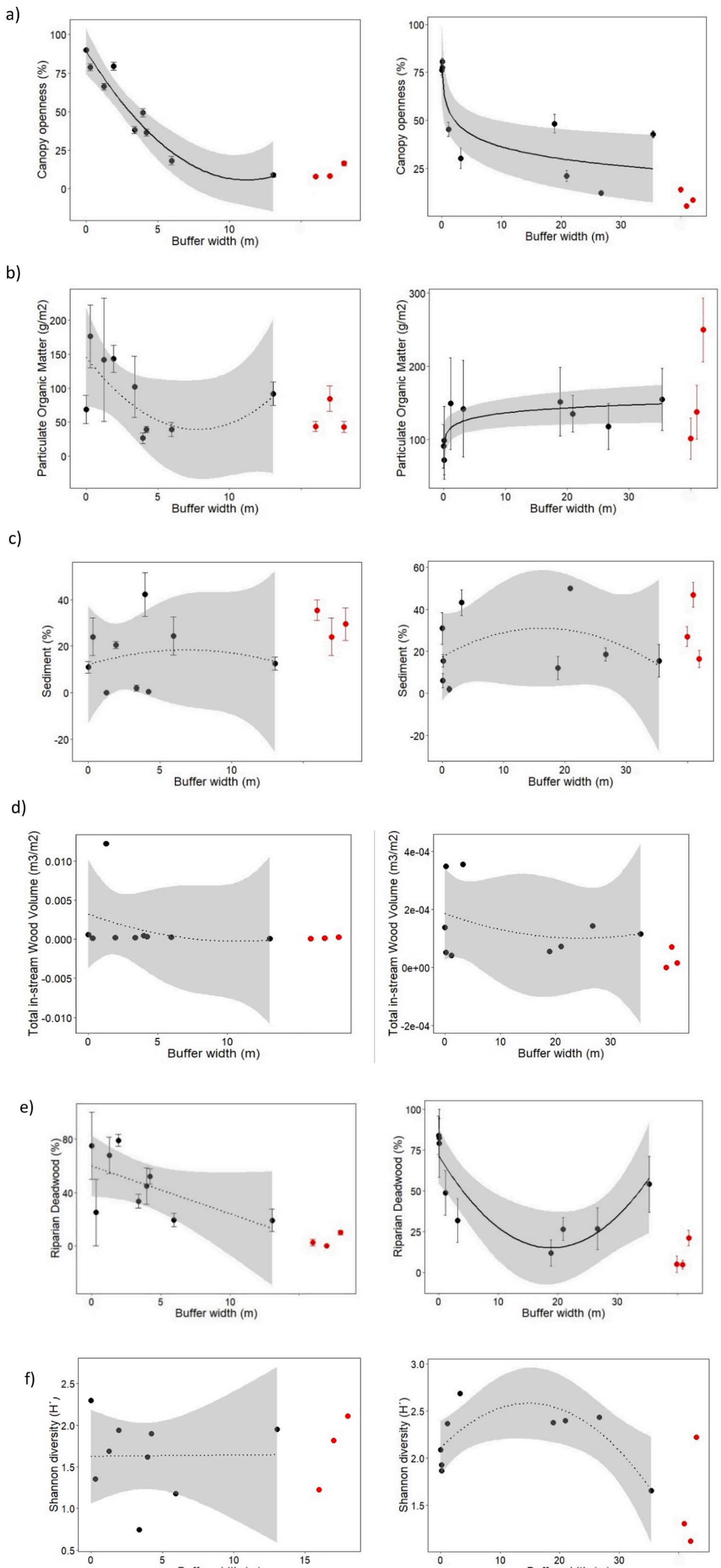

Fig. 2. Mean of (a) canopy openness (±SE), (b) particulate organic matter (±SE), (c) sediment cover (±SE), (d) in-stream wood volume (m^3/m^2), (e) riparian wood debris (±SE) and (f) macroinvertebrate Shannon diversity index (H́) for northern (i) and southern (ii) sites plotted against average buffer width (m). The red dots represent the values at the reference sites but were not used for the regression trends. The solid (significant trend) and dashed lines (insignificant trend) as well as confidence interval are presented for the best-fit regression models for each targeted parameter and buffer width. Test results for the models are presented in Table 3.

in streams without a buffer at a range of 78.8–89.8% in the north and 76.2–80.5% in the south (Table 2). LMM analyses revealed that in both the northern and southern streams, the no buffer sites had significantly higher canopy openness compared to the buffer sites (North: $p = 0.0001$; South: $p = 0.0062$) followed by reference sites (North: $p < 0.0001$; South: $p = 0.0008$) (Table S1). No significant differences were detected between buffer sites and reference sites in both north and south.

The differences in shading provided by the different riparian buffers were also translated onto stream water temperatures (Fig. 3; Table S2) in both regions. We found significantly higher summer diurnal temperature fluctuations in streams with no buffer compared to the "narrow" (Buff 2 N: ANOVA, p = <0.001) and "wide" (Buff 5 N: ANOVA, p = <0.001) buffered streams and reference streams (Ref 2 N: ANOVA, p = <0.001) in the northern sites (Fig. 3a). For example, in the No buffer 2 N streams, diurnal temperature dropped maximally from 25.4 °C to 12.4 °C (~13 °C difference) from day to night in one instance. In contrast, reference stream 2 N had minor diurnal temperature fluctuation from 11.3 °C to 10.8 °C (~0.5 °C difference) on the same date (8th July 2018). Daily temperature fluctuation in the "narrower" buffer (Buff 2N) was also significantly higher (ANOVA, $p = 0.008$) than the reference sites (Ref 2N) and dropped from 16.0 °C to 11.5 °C (~4.5 °C difference) on the same summer day. Southern reference streams also had relatively more stable summer stream temperature fluctuation (~2.1 °C maximum difference for Reference 2S; 3rd July 2018) compared to the wider (4.5 °C maximum difference for Buff 5S) and thinner buffer sites (7.0 °C maximum difference for Buff 2S) on the same date (Fig. 3). ANOVA results show that "wider" buffer (Buff 5S) had significantly larger diurnal temperature fluctuations compared to "narrower" buffer (Buff 2S: ANOVA, p = <0.001) and reference streams (Ref 2S: ANOVA, p = <0.001). Due to the drought in 2018, the southern no buffer site dried out during summer, hence we only have water temperature recordings from mid-Aug 2018 (Fig. 3) and was not included in the ANOVA analysis (Table S2).

3.2. Objective 2: Organic matter

Regression analyses showed a significantly increasing (polynomial) trend between buffer width and particulate organic matter (POM; measure of food provision) in the south and a non-significant relationship in the north (Table 3, Fig. 2 i.b). In the southern streams, POM ranged from 72.1 g/m^2 to 249.6 g/m^2 (mean = 133.5 g/m^2), highest in one of the reference sites (Table 2). In the northern streams, POM ranged from 26.2 g/m^2 to 176.4 g/m^2 (mean = 84.7 g/m^2) with highest POM recorded in one of the no buffer site (Table 2). In general, the presence of buffers regardless of their widths seem to fall within the POM range of reference sites in the southern region (Fig. 2 ii.b). When the sites were categorised into buffer groups, in the north, the no buffer sites had significantly higher POM (Table S1) compared to the buffered ($p = 0.0296$) and reference streams ($p = 0.0471$) but the buffered and reference sites did not differ. In the south, there were no significant

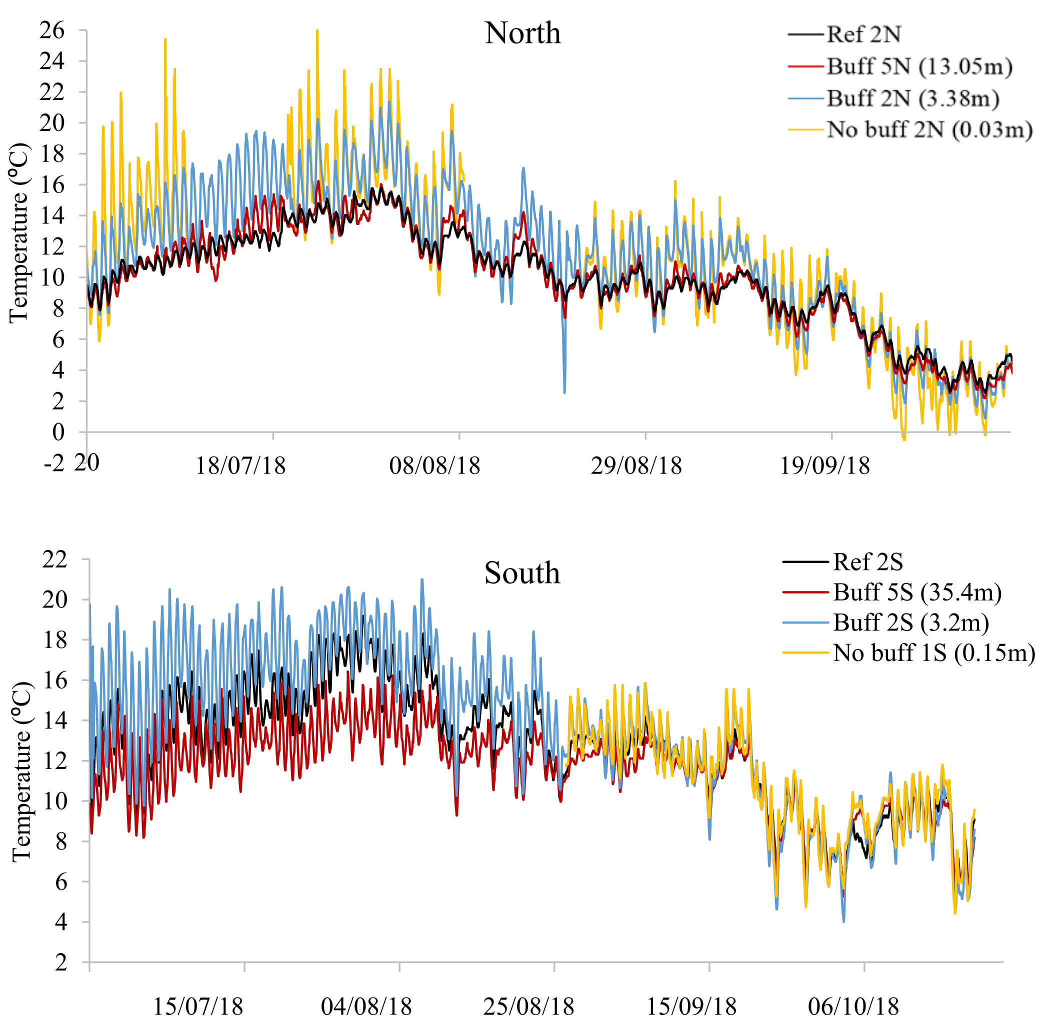

Fig. 3. Hourly stream water temperature for selected northern and southern sites. For better visual clarity and due to some streams drying out, we only reported temperature data from selected stream sites that represents no buffer, "narrow" and "wide" buffer as well as reference sites from each region. Average buffer width (if present) is given in parenthesis in the figure legends.

differences in POM across all groups (Table S1).

3.3. Objective 3: Sediment cover

Both regions show weak and non-significant relationships between buffer width and snapshot estimates of sediment cover (% of fine bottom sediments, Table 3). Overall, sediment covers were highly variable across sites regardless of buffer width in both regions (Fig. 2 i.c, ii.c). In the northern streams, sediment cover ranged from 0% to 42.2% (mean = 18.9%) while in the southern streams, it ranged from 2% to 29.9% (mean = 23.7%) (Table 2). When categorised into buffer groups, sediment cover in no buffer and buffered streams did not differ from reference sites (Table S1) within each region, neither did the buffered streams differed from the no buffered ones.

3.4. Objective 4: Deadwood – Riparian and in-stream

The volume of in-stream deadwood (m^3/m2) shows non-significant relationships with buffer width in both regions (Table 3, Fig. 2 i.d, ii.d). Riparian deadwood (%) follows a non-significant relationship in the north, but a significant polynomial distribution in the south (Table 3, Fig. 2 i.e, ii.e) with sites of intermediate buffer width having the least amount of riparian deadwood (excluding the reference sites). When categorised into buffer groups, the volume of the in-stream deadwood at the no buffer and buffered sites did not differ from reference sites, and nor from each other. However riparian deadwood was higher in the no buffer sites (North: 61.71% ± 8.81; South: 73.6% ± 7.34) compared to reference streams in both north and south (North: 4.15% ± 10.17 ($p = 0.0016$); South: 10.2% ± 8.48 ($p = 0.0001$); Table S1). In the south, the buffered sites also differed significantly from no buffer sites (30.2% ± 6.57 ($p = 0.0012$) but were similar to the reference sites. In the north, no differences were detected between buffered sites to no buffer or reference.

3.5. Objective 5: Biodiversity

Generally, southern streams had higher macroinvertebrate Shannon diversity ranging from (1.12–2.68, mean = 2.04) compared to the northern streams (0.75–2.30, mean = 1.65) (Table 2). In both regions, macroinvertebrate diversity shows non-significant relationships with buffer width (Table 3; Fig. 2 i.f, ii.f). When categorised into buffer groups, neither no buffer sites nor buffered sites differed between each other or to the reference streams in either regions (Table S1).

4. Discussion

In this study, we aimed to evaluate how riparian buffers of varying widths along small Swedish streams provide ecological objectives specified by the recently developed Strategic Management Objectives (SMOs) for protecting freshwater ecosystems during forestry operations (Andresson et al., 2013). We documented the relationships between the measured parameters to buffer width, and compared how different buffers provide each parameter compared to reference (unharvested) streams, separately for northern and southern Sweden. Based on both approaches, we found that buffer width significantly influenced several of the measured parameters (e.g., shading, POM, riparian deadwood) but not others, and deviate from reference conditions in several aspects discussed below.

Shading. The SMOs state that riparian buffers should target to maintain stable shading in streams over time (Andresson et al., 2013). Shading provided by riparian vegetation is crucial to maintain stream and riparian ecosystem integrity because shading maintains microclimate and controls air and water temperatures (Moore et al., 2005; Oldén et al., 2019b). Both light and water temperatures determine biological and ecosystem processes, e.g., primary production, stream metabolism, decomposition and solubility of gasses (Martínez et al., 2014). Our findings show strong negative trends between riparian buffer width and canopy openness indicating that more shading was provided to stream and riparian environments as the buffer widened. To provide similar shading as reference streams, buffers of at least 8 m and 25 m should be kept along northern and southern streams respectively. Kuglerová et al. (2020) showed that buffers were on average 5.3 m and 2.3 m wide in northern and southern Sweden respectively, therefore majority of streams in the two regions do not provide sufficient shading. The difference in buffer width mimicking reference conditions in northern and southern streams is most likely due to differences in riparian forest composition and density of trees as well as different climatic variables. In the north, dominance of denser spruces provide better shading compared to the south with more birches that have larger distances between individual trees. Although we have not specifically measured tree density, we recorded higher proportion of birch in the southern, compared to northern streams. This illustrates that buffer width alone is an insufficient measure for riparian protection. More specific parameters, including e.g., tree species composition and density, should be included in riparian management guidelines for the establishment of an ecologically functional buffer. We also found that buffers affected water temperatures, likely corresponding to the different shading capacities. Although we were not able to analyse all water temperature data (due to logger failures and severe draught) we saw that streams without a buffer had higher diurnal temperature fluctuations. Wider buffers (>10 m) were able to reduce this fluctuation to a larger degree compared to narrow buffers (<5 m) and were closer to emulate reference conditions (see also Oldén et al., 2019b). However, even the presence of a narrow buffer was able to provide some moderation of diurnal temperature fluctuations compared to the no buffer sites (see also Jyväsjärvi et al., 2020). Question remains, what level of shading is the most effective to protect aquatic and riparian microclimate but at the same time maximize understory vegetation diversity and in-stream producitivity? Some suggest that riparian buffer management should emulate natural disturbances, so called END management (e.g., Kreutzweiser et al., 2012; Sibley et al., 2012), and allow canopy gaps to achive the most optimal riparian forest dynamic seen in old-growth forests. Our reference streams were situated in managed mature forests stands, where natural gap canopy dynamic is supressed by management, which explains the high levels of shading.

Particulate organic matter (POM). Several of our other results support the idea that our unharvested reference streams may not be the most ecologically optimal as they differ (lower) in quality compared to e.g., old-growth forest. For example, our results show that standing stock of POM as well as riparian deadwood were higher in no buffer streams compared to reference streams in southern Sweden. A similar result was found for riparian deadwood in the northern Swedish sites. Mature, managed spruce forests have been shown to contribute low amounts of poor-quality litter (Benfield, 1997) and deadwood (Siitonen et al., 2000) because of reduced natural dynamic due to thinning operations throughout the rotation (Hasselquist et al., 2021). As such, our findings of higher aquatic POM and higher volume of riparian deadwood in the no buffer sites correspond with the idea that riparian interventions via management can increase some ecosystem functions and can emulate natural disturbance (Sibley et al., 2012). In the northern streams, high POM within the no buffer sites could be due to the residual impact of harvesting (twigs, branches) and limited filtering of leaves delivered from further upland due to the absence of buffers. In the southern streams, the POM results suggest that volume of standing stock increases with buffer width but is similar to reference sites. We deduce that the high variability in POM responses across both regions could be due to upstream and upland conditions of our studied streams as organic matter can be transported downstream, and delivered to streams from at least a 30 m lateral distance (Bilby and Heffner 2016). Our northern results are in contrast to a similar study done by Jyväsjärvi et al., (2020) in Finland where they found highest standing stock POM in reference streams (old-growth forest), followed by recently harvested streams with wide then narrow buffers. Similarly, Göthe et al., 2009 found that coarse POM was significantly higher in old-growth Swedish streams than recently harvested streams with no buffer. These opposing scenarios indicate a potential for two alternative states of organic matter dynamics in streams, runoff from upstream catchment or runoff from localised harvesting and riparian practices. It is also important to note that POM abundance, as we measured it here, does not necessarily translate to preferred subsidy for all aquatic organisms because conifers provide low quality litter, especially for macroinvertebrates (Lidman et al., 2017). Knowing the proportion of deciduous and coniferous litter in the streams would help to understand how the provided riparian subsidy is utilized by aquatic consumers, however were not assessed in this study. Nevertheless, riparian management should include such riparian buffer composition that is able to provide organic resource subsidies vital for aquatic microbes and invertebrates, and specific thresholds should be presented and targeted in the SMOs.

Riparian and in-stream deadwood. The riparian deadwood being higher in no buffer sites compared to the unharvested references in both regions are likely due to windthrows. We observed that in many of the no buffer sites, forestry practitioners originally left narrow buffers (<5m) but majority of those trees were blown-down by the time of our inventories. Post-harvest windthrows within riparian strips are expected and are more common in narrower (<15 m) than wider (30 m) buffers (Bahuguna et al., 2010; Mäenpää et al., 2020). The U-shaped relationship between buffer width and riparian deadwood in the south does not correspond with this contention likely because we only had one site with buffer wider than 30 m. Deadwood in both riparian and aquatic ecosystems are important as sources of organic matter, nutrients and habitats for terrestrial and freshwater organisms (Hylander et al., 2005). In addition, in-stream deadwood helps to influence stream flow, channel morphology, water depth as well as sediment transport (Broadmeadow and Nisbet, 2004). However, recruitment of riparian windthrows to aquatic ecosystems is delayed as the downed wood might enter the stream at an advanced state of decay (Rossetti de Paula et al., 2020). Our inventories were conducted < 8 years after harvesting and thus riparian deadwood did not have enough time to enter stream channels. This was reflected in the fact that we found no differences in the volume of in-stream deadwood across the harvested and reference sites. The low quantity of wood in our reference sites further illustrates that mature production forests may not be the best indicator for optimal riparian conditions, and SMOs may need to present specific target values for deadwood provision derived from measurements in old-growth forest sites. Continuous management of reference streams used in this study therefore may not represent the best management/riparian targets for small streams as they might be impaired due to historical legacy of past management and likely do not resemblé traditional‘ reference sites such as old-growth unharvested forests (Kuglerová et al., 2021). This is supported by Dahlstrom and Nilsson, 2004 that show that wood volume was four times higher in near-natural streams compared to streams within managed forest landscapes in Sweden.

Sediment cover. We found no trends for sediment cover and buffer width, and unharvested reference sites were similar to the buffer and no buffer sites. Forestry activities are commonly reported to increase erosion and deposition of fine sediments in headwater streams, degrading water quality and physical habitats within streams (Marttila and Kløve, 2010). Retaining riparian buffers along streamside has shown to effectively reduce sediment loading into waterways (Nieminen et al., 2005; Piggott et al., 2012). As all of our sites, including the reference streams, are situated within managed production forests, snapshot of sediment cover within the studied reaches could derive from various sources within the catchment. It is likely that upstream forestry activities e.g., roads, clear-cuts, and/or drainage ditches cause cumulative sediment transport mirroring catchment-scale rather than local sources (Gomi et al., 2006). Riparian buffers left only on small parts of the stream probably cannot offset catchment-scale influences. Moreover, as our streams were harvested between 3 and 8 years ago, sedimentation caused by the local activity might have peaked just shortly post-harvest (when soils are lose, everything is fresh and disturbed) and stabilised a couple of years post-harvest. A study by Macdonald et al. (2003) found increased suspended sediment in small streams following harvest in British Columbia that recovered to levels comparable to or below pre-harvest conditions within 3 years or less. Nevertheless, we show that sediment cover is similar between recently harvested and unharvested streams several years post-harvest, providing similar bottom habitat (see Berg, 2019) important for aquatic organisms including macroinvertebrate communities.

Macroinvertebrate diversity. Macroinvertebrates are reliant on several environmental and habitat variables, hence forest harvesting immediately adjacent to streams or clear-cuts in upstream catchments have been reported to have negative effects on aquatic macroinvertebrate communities in small streams (Reid et al., 2010; Erdozain et al., 2019). We found no differences in macroinvertebrate diversity across the harvested and unharvested reference streams regardless of buffer width in both regions. Correspondingly, other boreal headwater studies have reported similar macroinvertebrate diversity between forested and clear-cut streams (Liljaniemi et al., 2002; McKie and Malmqvist, 2009). This similarity has been attributed to increased benthic organic matter subsidies in recently harvested boreal streams that may compensate for the changes in habitat structure and water quality following forest harvest (Liljaniemi et al., 2002), an explanation also supported by our POM results. Additionally, stream water acidity, one of the main drivers of macroinvertebrate communities has been shown to be comparable between recently harvested (<11 years old) and mature boreal forests (Jonsson et al., 2017; Liljaniemi et al., 2002) and this is also the case of our harvested and reference sites (see Berg, 2019). Another explanation could be that long term and multiple disturbances during rotation period in productive forest stands (Kuglerová et al., 2021), as in all our streams including unharvested reference streams, may have modified and reduced aquatic biodiversity over time that cannot be offset by reforestation of riparian zone but may require large-scale conservation of the watershed (see Harding et al., 1998).

Overall, our results show that buffer width does not relate to a number of ecological parameters stated in the Swedish SMOs to be protected by riparian buffers. This was likely caused by context-dependency of our study sites (see also Jyväsjärvi et al., 2020) as well as the differences in riparian buffer conditions besides width as we show that even streams within the same vicinity and/or region responded differently. Therefore, results from this study should be applied cautiously when reporting the efficacy of buffer width in mitigating forestry impacts on small streams. Firstly, the lack of differences between unharvested reference and harvested streams highlights the aforementioned question as to whether streams within managed production forests (reference) are suitable to be used as a proxy for "best management practice" (Kuglerová et al., 2021). However, given the lack of achievable target values described in the SMOs, comparisons to mature, unharvested production forests are most appropriate to understand what most of the forest landscape in Sweden looks like. On the other hand, using the unharvested production forests as target for post-harvest conditions is not appropriate at all, given the reasons we discussed. Secondly, we have chosen buffer width as an indicator of protection level. However, buffer width may not be equivalent to buffer quality as there are large variations in terms of composition, density, structure and age of riparian trees as well as riparian connectivity/continuity across the study sites. Thirdly, several of the parameters we measured suggested the influence of catchment- or landscape-scale forestry activities that may obscure riparian-related changes. Additionally, our sampling reach of 50 m may be too small to capture spatial variability in stream ecosystems, however 50 m reaches were the best standard distance we could apply for all sites before reaching other obstructions such as roads, the end of the clearcut, or a tributary stream.

4.1. Conclusions and implications

The targeted objectives outlined in the SMOs are crucial to maintain stream ecosystem processes following forestry activities. We show that the effects of riparian buffer width following clear-cut activities are complex as they vary depending on the geographical location and riparian conditions. These results contribute to the debate on the ecological effectiveness of fixed-width streamside buffers to protect riparian and stream environments (Richardson et al., 2012). Further, it raises the question, whether buffer width, currently the most commonly prescribed parameter, should be used at all when delineating riparian reserves. Others have suggested that conditions of the riparian area, such as slope, wetness and/or composition of riparian forests should be used instead (Kuglerová et al., 2014; Oldén et al., 2019b). Although foresters are strongly encouraged to ensure that riparian buffers surrounding surface waters sustain several targeted objectives, buffers rarely exceed 5 m in width (Kuglerová et al., 2020), which is most likely ecologically unsatisfactory. As the provision of buffers involve tradeoffs between ecological and economic benefits, we call for comprehensive consideration synergistically accounting for site-specificity and land mosaic planning to sufficiently meet the targeted objectives outlined in the SMOs (Hasselquist et al., 2021).

CRediT authorship contribution statement

Darshanaa Chellaiah: Formal analysis, Investigation, Data curation, Writing - original draft, Writing - review & editing, Visualization. **Lenka Kuglerová:** Conceptualization, Methodology, Investigation, Writing - review & editing, Visualization, Supervision, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2021.119591.

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