Does the buffer width matter: testing the effectiveness of forest certificates in the protection of headwater stream ecosystems

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Running title: Effectiveness of stream buffers

Keywords: forestry, buffer management, decomposition, ecosystem processes, benthic invertebrates

Abstract

Forest harvest has multiple impacts on adjoining freshwater ecosystems, particularly headwater streams which typically receive minimal protection against forestry. However, evidence on the effectiveness of differently sized riparian buffers remains limited. Using data from two discrete regions of Finland, we assessed the effectiveness of riparian buffers in providing protection for the riparian and stream environment, benthic invertebrate diversity and species composition, and ecosystem functioning of boreal headwater streams. Our study included streams with both wide (> 15 m) and narrow (< 15 m) riparian buffers, enabling comparison of the two dominant forest certificates (FSC and PEFC). Compared to unharvested reference streams, nutrient concentrations as well as stream and riparian light intensity and temperature were higher at forestry-impacted sites. The amount of woody debris, cover of aquatic mosses and particulate organic matter standing stock were strongly reduced in streams draining harvested forests, especially in narrowly buffered streams. Changes in light and nutrient conditions induced a transition towards more autotrophic conditions. Organic matter decomposition rates were elevated in forestry-impacted sites only in the southern region. Forest harvest decreased macroinvertebrate diversity and evenness, and altered community composition in the northern region, but much weaker changes were observed in the southern region. Our findings support the retention of riparian buffers, but also confirm that their effectiveness depends on the environmental context and thus remains poorly predictable. Our results also suggest that the widely applied PEFC certification does not provide sufficient protection for stream ecosystems and more stringent protocols are needed to ensure ecological sustainability of forestry.

1. Introduction

Global efforts to reduce the consumption of non-renewable energy have provoked a worldwide surge of bioeconomy (EC, 2018). This is placing increasing pressure on the exploitation of natural resources, and in forest-rich countries a prominent environmental impact of this development is the loss and modification of forests. Intensified forestry not only degrades forest ecosystems but also imposes a substantial threat to adjacent freshwater ecosystems (Laudon et al., 2011; Jonsson et al., 2017).

The uppermost parts of fluvial watercourses (hereafter, headwater streams) represent up to 80% of the total stream network length (Bishop et al., 2008) and are key habitats for biological diversity (Finn et al., 2011). Headwater ecosystems also regulate the availability of resources to downstream habitats (Alexander et al., 2007). Given their small size and close intertwining with the surrounding forests, headwaters are often considered the most vulnerable part of riverine networks (Wallace and Eggert, 2015), yet recent studies imply that they are largely neglected during forestry planning and management (Richardson et al., 2012; Kuglerová et al., in review).

Various protective measures have been taken to mitigate the effects of forestry on adjacent stream ecosystems. The most conventional method is to leave an unmanaged strip of vegetation (henceforth, riparian buffer) between the stream and the harvest (Richardson et al., 2012). Retention of riparian buffers reduces nutrient and sediment transport from land, maintains natural in-stream thermal and light regimes and provides terrestrial resource subsidies to stream food webs (Kreutzweiser et al., 2009; Richardson and Sato, 2015).

In most jurisdictions the retention of buffer strips is voluntary (Ring et al., 2017). For example, the Finnish Forest Act requires protection of riparian forests, but does not provide any specifications for the width, type or quality of the riparian buffers. In production forests, similar statutory protection does not exist; instead, their conservation is based on forest certificates. To verify that their forests are ecologically, economically and socially sustainably used, forest owners are required to obtain a certificate for the wood they produce.

Globally, there are two major forest certification systems: Program for the Endorsement of Forest Certification (PEFC) and the Forest Stewardship Council (FSC). Both require some level of protective buffer between the harvested forest and the adjoining freshwater ecosystem. According to the Finnish FSC, the protective riparian buffer must comprise at least 15 meters of intact forest (Finnish FSC Association, 2010). The principles of PEFC are less stringent and only require that the riparian buffer should be at least 5 meters wide, and careful thinning can be allowed (PEFC Finland, 2014). While both certificates are currently available the PEFC certification is much more widely applied in Finland (95 % of certified forests; see Lopatin et al., 2016).

The riparian buffer retention measures of either certificate are, however, poorly supported by scientific evidence. Several lines of evidence suggest that the protection of environmental conditions, key ecosystem processes and stream biodiversity requires 30-m wide riparian buffers (Sweeney and Newbold, 2014) and safeguarding the riparian plant and wildlife biodiversity may necessitate even more extensive (> 40 m) buffers (Marczak et al., 2010; Selonen and Kotiaho, 2013). A recent comparison of contemporary riparian buffer practices among major forestry countries (Canada, Sweden and Finland) revealed that the average width of the riparian buffer around small forest streams was far less than recommended by scientific consensus (Kuglerová et al, in review). These results also indicated considerable inconsistencies in riparian retention practices within each of the three countries, reflecting the lack of clear guidelines for buffer retention protocols. The discrepancies in riparian buffer management stem from region-specific differences but also indicate the shortage of systematic research regarding the effectiveness of riparian buffers on adjoining freshwater ecosystems. Given the noticeable contribution of Nordic countries, particularly Sweden and Finland, to global forestry and forest economy (FAO, 2019), scientific evidence on the importance of riparian

93 buffers and concrete guidelines for ecologically justified buffer widths is imperative for steering

94 sustainable use of forests in northern Europe.

95 Here, we evaluated the importance of riparian buffer width to the protection of headwater stream

96 ecosystems, particularly macroinvertebrate diversity and species composition, as well as key

97 ecosystem functions. We focused on small streams draining recent clear-cut areas. Previous research

98 has demonstrated that forestry impacts on adjoining streams are often complex and dependent on the

99 environmental context (see Richardson and Béraud, 2014). To examine whether and how the

100 responses differ within a single jurisdiction, we repeated our monitoring scheme in two regions

101 differing in geographical location, climate, forest management practices and land use. In both regions,

102 riparian buffer widths varied between $0 - 35+$ m, thus demonstrating the adaptation of buffer retention

103 principles of either the PEFC or FSC certificate.

104 We hypothesise that, regardless of the region, streams receiving minimal protection (buffer < 15 m)

105 deviate from the unharvested reference streams in terms of macroinvertebrate assemblages and

106 ecosystems functioning while streams with wider riparian buffers (> 15 m) remain largely unaffected

107 by forest harvest, resembling the reference streams. We anticipate that the environmental conditions

108 inherent to near-pristine headwater streams and their riparian forests (e.g. high shade, low

109 temperature, low nutrient levels) are compromised in poorly buffered streams. We anticipate that a

110 narrow buffer decreases the quantity of particulate organic matter of terrestrial origin and intensifies

111 light availability and primary production, resulting in transition towards autotrophy. Finally, the two

112 study regions are expected to differ in their responses to nearby forestry; while the abiotic responses

113 (light, temperature, organic matter standing stocks) are expected to be coherent across the regions,

114 we expect that more pervasive land use in the southern region results in lower-quality reference sites.

115 This is expected to obscure the signal of any individual land use type and therefore results in less

116 pronounced biological differences between the reference and forestry-impacted streams.

117 ## 2. Material and methods

118 *2.1 Site selection*

119 We selected 23 small streams (orders $1 - 2$) for the study, twelve in northern-central Finland (hereafter

120 "northern" region) and 11 in south-western Finland ("southern" region; Fig. 1). The two regions differ

121 in terms of climate, topography and intensity of land use (Table 1). The northern region constitutes

122 headwaters of River Iijoki basin. The area represents a transitional zone between middle boreal and

123 northern boreal ecoregions and is characterised by mires and mixed forests (Table 1). The southern

124 region combines headwater tributaries of Rivers Karvianjoki, Isojoki and Kyröjoki. It is characterized

125 by a flat coastal landscape dominated by a mixture of peatlands, forests and arable land (Table 1).
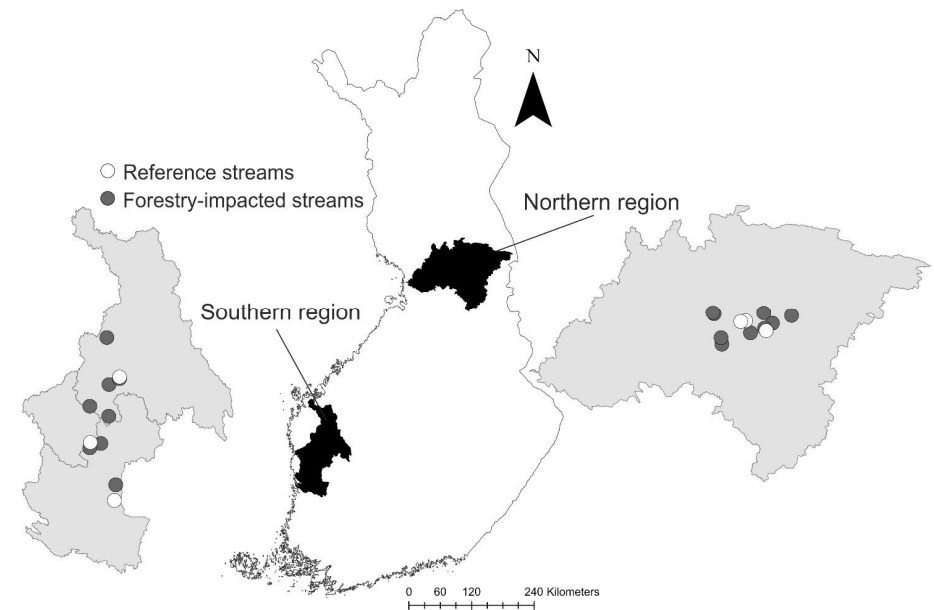
127 **Fig. 1**. *Geographical locations of the study regions and of reference and forestry-impacted streams*

128 *within each region.*

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131 ***Table 1** Geographic, climatic and catchment-scale characteristics of the two study regions.*

	Northern region	Southern region
Latitude (°N)[1]	65°34' N	62°12' N
Longitude (°E)[1]	27°51' E	22°16' E
Altitude (m, a.s.l.)[1]	117	275
Annual mean air temperature (°C)	0	4.1
Annual mean precipitation (mm)	567	681
Built areas (%)[2]	1.0	3.3
Open mires (%)[2]	11.4	4.1
Coniferous forests (%)[2]	57.4	49.4
Deciduous forests (%)[2]	0.4	1.3
Mixed forests (%)[2]	8.6	11.5
Arable land (%)[2]	0.9	15.5

[1] Central position of the study sites

[2] Mean across the river basin(s)

132

133 Three of the study sites in each region did not have any recent (< 50 – 60 y) forestry actions nearby

134 (> 300 m from the stream or at least 1.5 km upstream of a study site) and were used as reference

135 streams. The remaining streams drained recent (3 – 6 y old) clear-cuts and sampling sites were located

136 at the downstream end of a clear-cut. The streams were selected from a larger number of candidate

137 sites (see Kuglerová et al., in review) so that the width of the forested riparian buffer between the

138 stream and the clear-cut ranged from 0 to > 35 m. This range is well representative of Finnish riparian

139 buffer practices (Kuglerová et al., in review). Typical of Finnish forestry, a great majority of our study

140 streams (> 90 %) had clear-cuts and riparian buffers on one side of the stream while the other side

141 remained intact. Buffer width was calculated from three measurements taken within a 50-m study

142 section. For the one-sided buffers, only the forestry-impacted side was considered and for the few

143 two-sided buffers, the buffer width was measured on both sides and average of these measurements

144 was used in the analyses. The following additional criteria were used for site selection: i) each site

145 should contain a 50-m long continuous riffle section, ii) reference and forestry-impacted sites were

146 spatially interspersed (Fig. 1), and iii) none of the sites were in the vicinity of an open mire, lake or

147 spring outlet.

2.2 Environmental measurements

149 Substratum structure was determined in 15 randomly distributed 0.25 m^2 plots using a modified

150 Wentworth scale (see Mykrä et al., 2008). Water samples were collected once (August) and they were

151 analysed for electrical conductivity, total phosphorus (TP), phosphate (PO$_4$), total nitrogen (TN),

152 nitrate-N (NO$_3$), dissolved organic carbon (DOC) and pH following national standards. Catchment

153 characteristics were determined using ArcMap Desktop 10.5 and CORINE 2018 land use data.

154 Amount of large woody debris (LWD) was measured by summing the volume of wood particles (Ø

155 > 5 cm) within the bankfull channel for each 50 m study section. We recorded temperature (°C) and

156 light intensity (lux) in the stream channel and in the riparian forest (1 m from the stream, 1 m above

157 the ground) in 1–h intervals with waterproof loggers (HOBO Pendant, Onset, Massachusetts) from

158 late June to early October.

2.3 Sampling of benthic invertebrates and organic matter standing stock

160 Benthic macroinvertebrates and fine and coarse organic matter (FPOM and CPOM) were sampled by

161 eight Surber samples (0.1 m^2, 500 µm mesh size) distributed randomly along the 50 m study reach.

162 Each sample was preserved in 70% ethanol and invertebrates were later sorted from the other material

163 and identified in the laboratory, mostly to species or genus level. Six of the eight samples were

164 randomly selected for macroinvertebrate identification. The remaining material in each Surber sample

165 was fractionated into FPOM and CPOM using 0.2 mm and 1 mm sieves, respectively, and the sieved

166 material was dried for 48 h at 60 °C and weighed. The samples were combusted then for 4 h at 550

167 °C and reweighed to obtain ash-free dry mass (AFDM, g m^{-2}).

168 *2.4 Algal biomass accrual and organic matter decomposition*

169 Stream biofilm was sampled twice (July and August) in 2018 by incubating 10 cm × 10 cm

170 unglazed ceramic tiles (n = 10 per site) in the stream for 4 – 5 weeks. After incubation, biofilm was

171 scraped off into 100 ml distilled water and the suspension was filtered to 0.45 µm filter papers

172 (Whatman GF/F, Kent, UK). Chlorophyll *a* concentration (n = 5) was determined according to

173 Huttunen et al. (2012). Another five samples were dried at 60 °C for 24 h, weighed, combusted for

174 4 h at 550 °C and reweighed to convert dry mass to AFDM. Autotrophic index (AI) was calculated

175 as the ratio of biofilm AFDM and chlorophyll *a*. Mean values of the two samplings were used in

176 statistical analyses.

177 Total decomposition rate (including both macroinvertebrates and microbes) was measured using

178 coarse mesh (8 mm) leaf bags which allowed leaf-shredding invertebrates to enter. Four grams of

179 dried birch (*Betula pendula*) leaves were enclosed in 15 × 15 cm nylon bags. Five bags were placed

180 onto the stream bed in early September and were removed after 5 – 6 weeks. In the laboratory,

181 litter bags were gently cleaned under tap water to remove any accumulated sediments. The

182 remaining leaf material was dried 48 h at 60 °C and ashed (4 h at 550 °C).

183 To measure microbial-mediated organic matter (cellulose) decomposition we used standardized

184 cotton-strip assays (Tiegs et al., 2013). The cotton strips were prepared using unprimed 12–oz.

185 cotton fabric, following Tiegs et al. (2013). One pre-weighed strip was inserted into each leaf bag.

186 After incubation the strips were gently cleaned, and decomposition was terminated by submerging

187 the strips into 96% ethanol for 60 sec. The strips were dried for 48 h in 50 °C and weighed. Tensile

188 strength loss (a surrogate for microbial decomposition rate) was measured by placing a 1-cm-long

189 portion of a cotton strip in the grips of a tensiometer (Zwick/Roell Z010, Germany) mounted on a

190 motorized test stand; the strip was then pulled at a fixed rate (2 cm min^{-1}) until the strip tore. The

191 initial tensile strength (N) was determined as an average of 10 control strips processed identically

192 to the treatment strips. Tensile-strength loss was expressed as percent of the initial tensile-strength

193 lost per incubation period. No temperature correction was done for the decomposition data as the

194 potential within- and among-group differences in stream water temperature were considered

195 inherent to our study design.

196 *2.5 Data analyses*

197 The forestry-impacted sites were classified into two groups based on their average buffer width

198 measured at three locations within the 50 m study reach. The sites with less than 15 m buffers

199 (hereafter "Narrow") represent the typical buffer width of the PEFC certified forests whereas the sites

200 with wider buffers (> 15 m; "Wide") follow the guidelines of the FSC certificate.

201 Principal component analysis (PCA) was used to characterize environmental variation among the site

202 groups (Reference, Wide and Narrow). For individual variables, the differences between the reference

203 (baseline) and the two forestry-impacted site groups were determined with Generalized Linear

204 Models (GLMs) with a gaussian link function. If test assumptions were not satisfied, log$_{10}$-

205 transformation was applied to response variable. For each benthic macroinvertebrate sample, we

206 calculated total density (individuals m^{-2}), total species richness, number of EPT taxa (Ephemeroptera,

207 Plecoptera, Trichoptera) and Pielou's evenness. Linear mixed effects-models (LMM) were applied to

208 macroinvertebrate metrics and other biotic response variables with multiple measurements per stream

209 (chlorophyll *a*, Autotrophic index, CPOM, FPOM and decomposition rates). LMM models were

210 constructed using lme4 and lmerTest packages in R (R Core Team, 2019) and included buffer

211 treatment as fixed effect and individual samples nested within stream as random effects. For count

212 data (species and EPT number), we used generalized linear mixed models (GLMMs) with poisson

213 error distribution with a log link function.

214 Patterns in macroinvertebrate species composition were visualized with non-metric multidimensional

215 scaling (NMDS) using *metaMDS* function of vegan package (Oksanen et al., 2019) of the R.

216 Differences in species composition among stream groups were tested using nonparametric

217 permutational multivariate analysis of variance (PERMANOVA) with *adonis* function in vegan.

218 PERMANOVAs were run using the Bray-Curtis similarity coefficient on log_{10}-transformed

219 abundance data, and statistical significance was estimated based on 9999 permutations. All six Surber

220 samples per site were used in PERMANOVA and the argument *strata* in the *adonis2* function was

221 used to circumvent spatial autocorrelation among replicates within each stream. A significant global

222 test was ensued by pairwise PERMANOVAs. The among-group differences in environmental

223 conditions were also tested with PERMANOVA, based on a Euclidean distance matrix.

3. Results

3.1 Environmental conditions

226 The median width of narrow riparian buffers was 6.75 m for the northern region (n = 4) and 7.0 m

227 for the southern region (n = 5), whereas the medians for wide buffers were 24.5 m (n = 5) and 23 m

228 (n = 3) for the northern and southern region, respectively. For the northern region, the first two PCs

229 explained 58.6% of the variance of environmental data. The stream groups were well separated along

230 the first PC, which correlated positively with light availability and water and air temperature (Fig. 2a;

231 Table 2), thus representing a buffer width gradient. The second PC represented a water chemistry

232 gradient with TP, NO_3 and DOC concentrations correlating positively with PC2 (Fig. 2a; Table 2).

233 Environmental conditions differed among the stream groups (PERMANOVA; $F_{2,9} = 2.75$, $P = 0.016$),

234 with the narrowly-buffered sites differing from the reference ($P = 0.05$) based on, for example, lower

235 moss cover and a less large woody debris (Fig. 2a, Table 2).

238 ***Table 2*** *Mean values of the measured environmental variables for the three stream groups (reference,*

239 *wide buffer [> 15 m] and narrow buffer [<15 m]) in each study region.*

	Northern region			Southern region		
	Reference	Wide	Narrow	Reference	Wide	Narrow
Catchment area (km^2)	2.7	4.9	2.9	10.3	8.9	9.3
Open mires (%)	5.2	5.0	0.0	8.6	6.3	3.3
Coniferous forests (%)	57.9	55.9	33.1	75.9	66.2	68.9
Mixed forests (%)	26.3	4.0	28.8	2.4	0.0	3.9
Arable land (%)	0.0	0.0	0.0	2.2	3.4	0.9
pH	7.0	6.8	7.1	6.9	7.1	6.8
Electrical conductivity (mS m^{-1})	2.4	2.0	2.8	3.4	4.9	3.5
TP ($\mu g\ L^{-1}$)	11.5	13.6	12.0	24.9	50.8	71.5
PO_4 ($\mu g\ L^{-1}$)	6.2	5.2	3.6	12.5	15.7	19.3
NO_3 ($\mu g\ L^{-1}$)	14.4	28.6	26.2	88.2	158.7	205.1
DOC (mg L^{-1})	5.8	11.1	10.9	4.9	5.3	14.2
Channel width (cm)	227	152	112	248	265	212
Channel depth (cm)	17.3	18.3	15.7	19.7	21.3	12.4
Substrate size (Wenthworth scale)	4.7	5.1	5.1	4.8	4.9	4.8
Large woody debris ($m^3\ a^{-1}$)	0.032	0.011	0.006	0.459	0.010	0.011
Current velocity (m s^{-1})	19.2	24.5	16.3	31.5	16.6	18.0
Moss cover (%)	49.6	31.4	14.8	14.0	8.0	8.8

243 PCA for the southern region followed largely the same pattern. The first two PCs explained 53.7% of

244 the variance and the stream groups were separated only along PC1, which represented clear-cut

245 impact and water quality gradients (Fig. 2b; Table 2). The among-group difference bordered on

246 significance ($F_{2,9} = 1.77$, $P = 0.052$) and only the pairwise comparison between the reference and

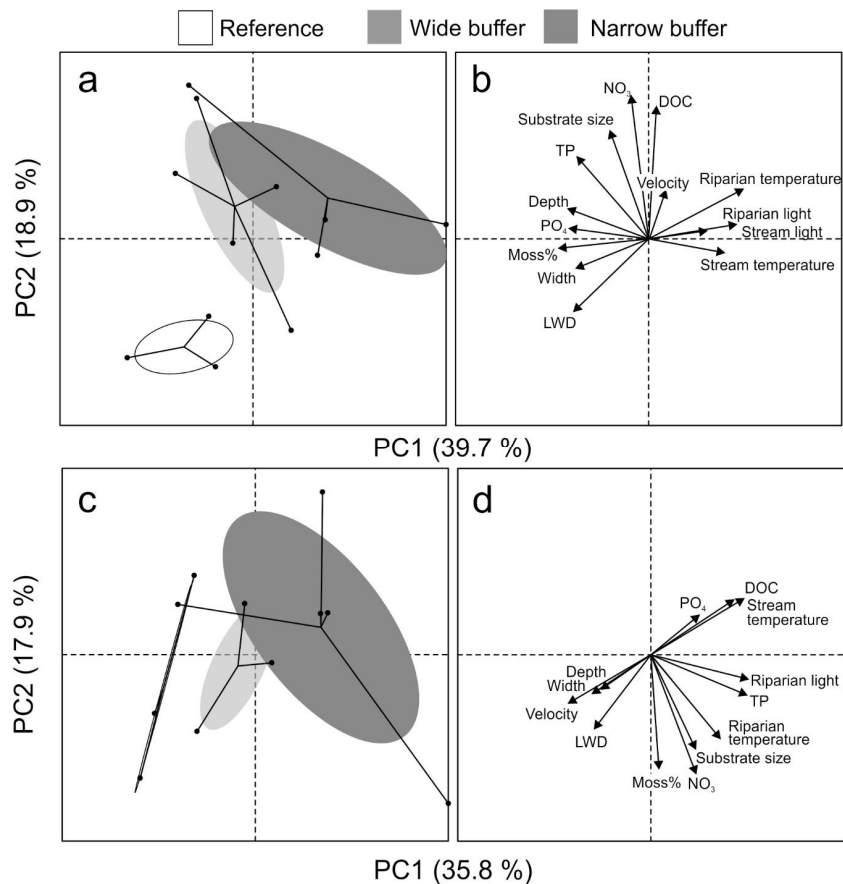247 narrowly buffered streams approached significance ($P = 0.076$).

248

Fig. 2. PCA ordinations of the environmental characteristics of the northern (upper panel) and southern (lower panel) streams, showing (a and c) separation of the three stream groups in the ordination space and (b and d) correlations (i.e., arrow lengths) between the principal components and environmental variables. Ellipses depict ±95% CI around group centroids.

253

254 Narrowing of the riparian buffer altered the light and thermal regimes of the stream-riparian ecotone.

255 Riparian light intensity was higher in narrowly buffered sites compared to reference sites in both

256 regions (GLM; $t = 2.33$, $P = 0.048$ for northern; $t = 2.78$, $P = 0.022$ for southern sites; Fig. 3a).

257 Riparian air temperature differed between the reference and narrow-buffered streams only in the

258 northern region ($t = 4.57$, $P = 0.001$; Fig. 3b). Stream light intensity did not differ among the stream

259 groups (Fig. 3c), whereas water temperature tended to increase in forestry-impacted streams, albeit

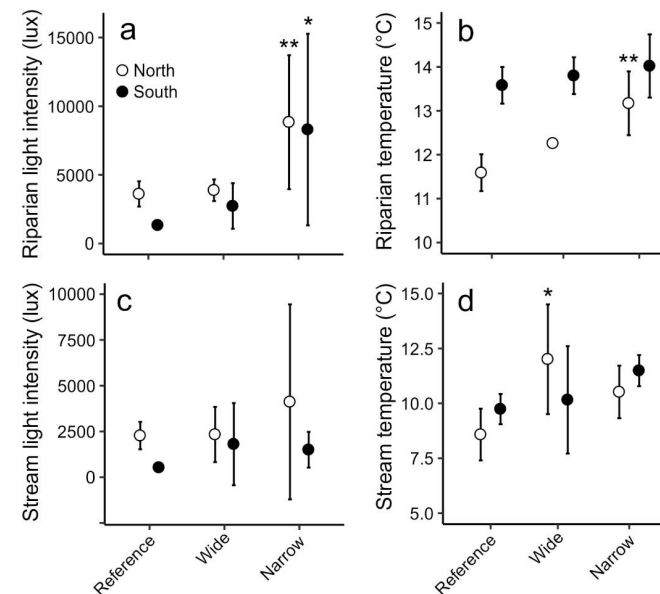260 significantly so only in the northern widely buffered streams (Fig. 3d).

261

262 *Fig. 3. Mean (±95% CI) a) riparian light intensity, b) riparian air temperature, c) stream light*

263 *intensity and d) stream water temperature in late June to early October in the three buffer width*

264 *categories in each study region. All comparisons are made against the reference (GLM); significant*

265 *differences are denoted by asterisks (* P < 0.05; ** P < 0.01).*

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3.2 Primary productivity, organic matter standing stock and decomposition processes

Chlorophyll *a* accrual rates were generally higher in forestry-impacted than in reference streams (Fig. 4a). The autotrophic index showed an opposite pattern, with lower values than reference for both buffer types in the southern streams (LMM: t = -2.88, P = 0.02 for wide buffers; t = -3.10, P = 0.015 for narrow buffers) and for wide buffers in the northern region (t = -1.96, P = 0.01) (Fig. 4b).

In both study regions, CPOM standing stock showed a weak (P = 0.09 – 0.17) decreasing trend towards the wide- and narrow-buffered streams (Fig. 4c). The amount of FPOM showed a similar negative trend, but significantly so for the northern streams (Fig. 4d). Unexpectedly, both CPOM and FPOM were nearly equal in widely- and narrowly buffered streams (Fig. 4c and 4d).

Leaf decomposition rate was enhanced towards narrowly buffered streams in the southern region (t = 2.34, P = 0.047 for wide buffers; t = 2.47, P = 0.038 for narrow buffers), but not in the northern region (t = 0.90, P = 0.39 for wide buffers; t = -0.29, P = 0.77 for narrow buffers) (Fig. 4e). Likewise, microbial-mediated decomposition (loss of cotton strip tensile strength) was enhanced in the southern region's forest-impacted streams, but significantly so only for narrow buffers (t = 2.38, P = 0.044; t = 1.75, P = 0.11 for wide buffers), whereas no differences were observed in the northern region (t = -1.81, P = 0.10 for wide buffers; t = -0.25, P = 0.80 for narrow buffers) (Fig. 4f).
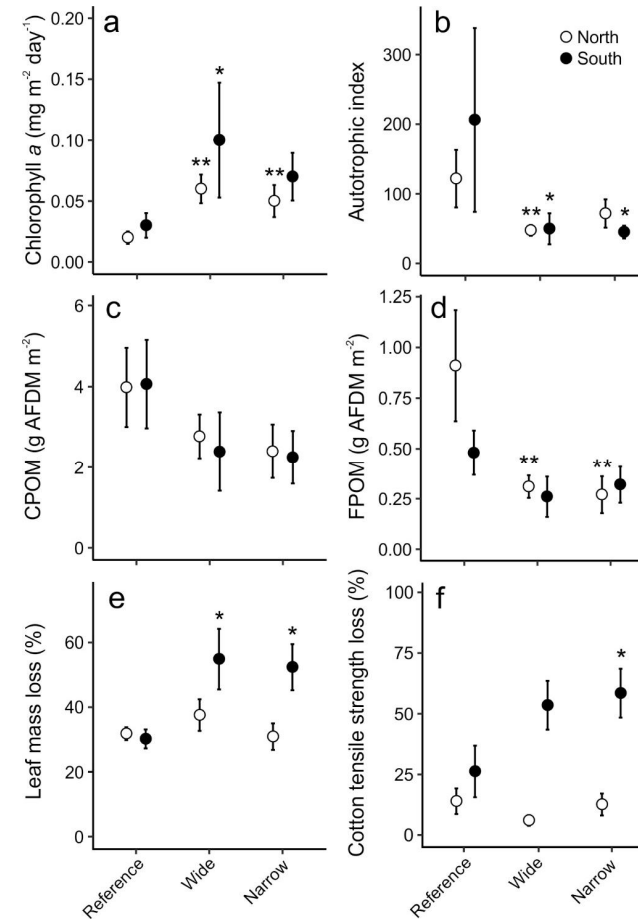
Fig. 4. *Mean (±95% CI) a) chlorophyll-a accrual and b) autotrophic index (AFDM:Chl a), ash-free dry mass of c) coarse (CPOM; > 1 mm) and d) fine (FPOM; 0.2 – 1 mm) particulate organic matter and e) leaf mass loss and f) cotton strip tensile strength loss during the decomposition assay in the three stream groups in each region. All comparisons are made against the reference (LMM); significant differences are denoted by asterisks (* P < 0.05; ** P < 0.01).*

3.3 Benthic macroinvertebrate density, diversity and community composition

Benthic macroinvertebrate data of the northern region comprised 60 taxa, whereas the southern region's data contained 49 taxa. Total macroinvertebrate densities ranged between 617 – 6599 and 872 – 3401 individuals per m^2 for the northern and southern regions, respectively. Total densities did not differ between the reference and forestry-impacted streams in either region (all $P > 0.05$). For the northern region, total number of taxa was marginally lower in the narrowly buffered sites compared to reference (t = -1.77, P = 0.07; Fig. 5a), while no such pattern was observed in the southern region (Fig. 5a). In the northern region, the narrowly buffered stream lacked a notable proportion of EPT taxa (t = -2.61, P = 0.009; Fig. 5b). Macroinvertebrate evenness was likewise significantly lower in the narrowly buffered streams in the northern region (t = -3.51, P = 0.007; Fig. 5c), but not in the southern region (t = -0.74, P = 0.47; Fig. 5c). All community metrics were lower in the southern reference sites than in the northern ones (see Fig. 5), indicating lower reference-site quality in the southern region.
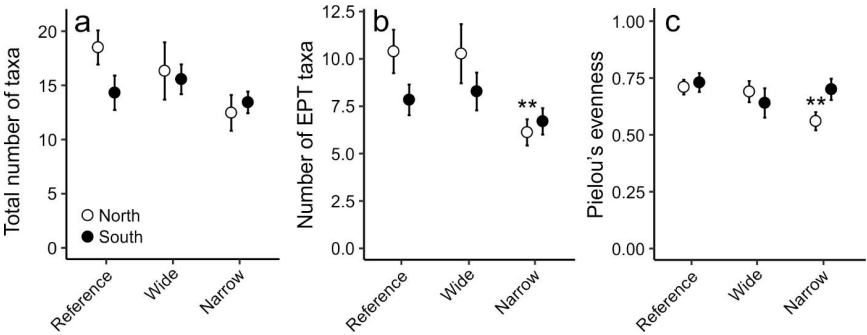

Fig. 5. *Mean (±95% CI) a) number of macroinvertebrate taxa, b) number of ephemeropteran, trichopteran and plecopteran (EPT) taxa and c) Pielou's evenness index in the three buffer width categories in each region. All comparisons are made against the reference; significant differences are denoted by asterisks (** P < 0.01).*

Macroinvertebrate species composition differed among the stream groups in the northern region (PERMANOVA; $F_{2,69}$ = 9.27, P = 0.029), but not in the southern region ($F_{2,69}$ = 3.26, P = 0.84). Pairwise PERMANOVAs for the northern streams indicated that the species composition of the narrowly buffered streams differed marginally from the reference assemblages ($F_{2,69}$ = 11.75, P = 0.08), whereas wide buffer assemblages did not ($F_{2,69}$ = 8.26, P = 0.12). NMDS ordinations indicated for both study regions that the streams with very narrow (≤ 10 m) riparian buffers deviated most from the reference assemblages (Fig. 6a and b).
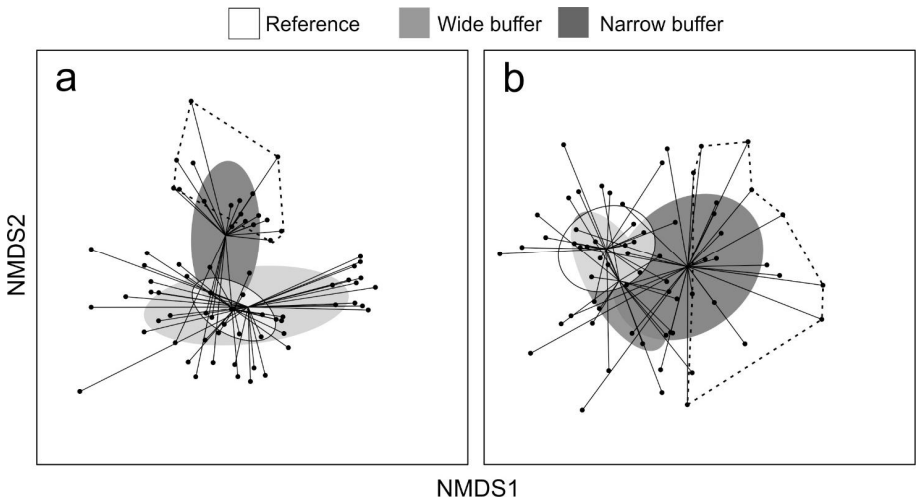

Fig. 6. *Nonmetric multidimensional scaling (NMDS) ordinations of macroinvertebrate assemblages in the a) northern and b) southern streams. Solid lines connect each sample to the corresponding group centroid and ellipses depict ±95% CI around the centroids. Dashed polygons delineate samples taken from streams with ≤ 10 m riparian buffers.*

4. Discussion

4.1 Environmental conditions

Our findings highlight that narrow (< 15 m) riparian buffers are insufficient in mitigating the forestry-induced alteration of headwater streams and their riparian forests. It should be noted that we used space-for-time (SFT) substitution design in our study. SFT assumes that drivers of ecological change through time are the same as those that drive changes in space. We acknowledge that this design has inherent weaknesses as it may confound temporal and spatial effects (Damgaard, 2019) but it was unfeasible to conduct a well replicated before-after-control-impact (BACI; Underwood, 1994) design within the limited project period. The amount of large woody debris (LWD) was reduced in forestry-impacted streams, especially those with the narrowest buffers. LWD affects the movement and storage of organic material, nutrients and streambed sediments and maintains the diversity of meso-scale habitats. As a result of clear-cuts, the amount of wood in the stream channel typically decreases (Wallace et al., 2001).

Moss cover was lower in forestry-impacted streams. Aquatic mosses influence stream processes and biodiversity mainly by providing sheltered microhabitats and fine detritus for benthic animals (Turunen et al., 2020). Bryophytes typically thrive in headwater streams where canopy shading limits algal growth (Stream Bryophyte Group, 1999). Clear-cutting of riparian forests increases the amount of light and thus, algal and vascular plant productivity (Kiffney et al., 2003), potentially outcompeting bryophytes (González-Mancebo et al., 2004). In our data, light intensities in the stream channel and the riparian forest increased, especially at narrowly buffered sites (see also Kiffney at al., 2003).

The pattern of increased light intensity was less pronounced in the stream channel. This is due most likely to increased DOC content in the narrowly buffered sites. Terrestrially derived DOC typically makes water brown-colored, thus enhancing light attenuation (Karlsson et al., 2009). Forest clear-cutting and subsequent site preparations have been observed to increase DOC input into adjacent streams (Piirainen et al., 2007; Laudon et al., 2009), which was also suggested by our results.

Water temperature in the stream channel was higher in the narrowly buffered than in the reference sites. This is in accordance with Kiffney et al. (2003), who reported an increase of insolation, thermal radiation and stream water temperature in recently harvested and buffered (10 m) streams compared to unharvested streams. On the other hand, Gomi et al. (2006) showed that in some cases even a 10 m buffer can mitigate the logging-induced increase of water temperature, but the efficiency of the buffer was related to north-south orientation of the stream, and thus differences in exposure to insolation. The retention of riparian buffers is not critical only for protecting streams from local forestry-induced impacts but also for adapting fluvial ecosystems to climate change (Thomas et al., 2016). By providing shade, thermal refugia and detrital input, wide and longitudinally continuous riparian buffers may contribute importantly to climate change adaptation of riverine networks through suppressing the warming of the riparian soil and stream water and maintaining energy flow (Palmer et al., 2009).

4.2 Organic matter decomposition

We observed strongly divergent patterns in both microbial and invertebrate-mediated organic matter decomposition rates between the study regions. While in the southern region decomposition rates were accelerated towards the narrowly buffered streams, data from the northern region did not suggest any differences between the reference and forestry-impacted streams. This observation underlines the context-specificity of responses to forestry. For example, Kreutzweiser et al. (2008) and Lecerf and Richardson (2010) noted that forest harvest resulted in lowered decomposition rates compared to reference streams, whereas Kreutzweiser et al. (2010) did not observe any effect of forestry on decomposition, and McKie and Malmqvist (2009) documented an accelerated rate of both microbial and detritivore-mediated decomposition in streams draining recent (3 – 5 y old) clear-cuts. Our southern streams exhibited considerable variation in NO_3 and PO_4 concentrations among differently

375 buffered streams (see Table 2) and, given the well-known role of nutrients in regulating particularly

376 microbial decomposition rates (Ferreira et al., 2015; Gulis and Suberkropp, 2003), the increased

377 nutrient runoff is the most likely explanation for the stronger response of decomposition rates to forest

378 harvest in the southern region. Also increased stream temperatures in the forestry-impacted streams

379 may have contributed to enhanced decomposition processes as organic matter decomposition tends

380 to be higher in warmer conditions (Martínez et al., 2014). In our decomposition analyses, we did not

381 correct decomposition rates for the temperature as we expected forestry actions to influence stream

382 temperatures, thereby rendering within- and among-group differences in temperature inherent to our

383 study design. However, we tested also the temperature-corrected data and the results remained the

384 same. This implies that thermal conditions played a minor role in regulating decomposition rates and

385 verifies that the decomposition patterns were primarily driven by the differences in nutrient levels.

386 *4.3 Benthic organic matter standing stocks*

387 Our observation of reduced biomass of FPOM and CPOM in forestry-impacted streams is in line with

388 some earlier evidence (Kiffney and Richardson, 2010). However, responses of CPOM and FPOM to

389 forest harvest have been highly variable (Göthe et al., 2009; Richardson and Béraud, 2014). The

390 reasons for this inconsistency may be related to site-specific differences in logging practices, soil

391 preparation and processing of harvest residues that can greatly modify the amount, quality and fate

392 of organic material delivered to the stream (Jackson et al., 2007). In our forestry-impacted streams,

393 the quantities of CPOM and FPOM virtually equaled, regardless of the buffer width. One potential

394 explanation for the reduced organic matter standing stock in both forestry-impacted stream groups

395 could be the intensified discharge dynamics due to nearby forestry actions. Finnish forest harvests

396 are often accompanied by intensive peatland drainage (Nieminen et al., 2017), resulting in more

397 frequent and extreme spates during high rainfalls, which is a key determinant of CPOM/FPOM export

398 in headwater streams (Cuffney and Wallace, 1989). The observed reduction of organic matter

399 standing stock potentially alters the physical structure of the stream ecosystem (Eggert et al., 2012)

400 and is likely to further intensify the reliance of stream biota on autochthonous production (see below).

401 *4.4 Algal productivity and level of autotrophy*

402 As expected, increased insolation stimulated algal productivity in forestry-impacted streams. Along

403 with the concurrent reduction in CPOM/FPOM, this induced a transition towards autotrophy (see also

404 Wootton, 2012; Kaylor and Warren, 2017). A classic paradigm in stream ecology dictates that

405 forested headwater streams are heterotrophic and predominantly fueled by allochthonous material

406 from the riparian forest (e.g. Tank et al., 2010). However, recent evidence challenges this paradigm,

407 suggesting that autochthonous production is of disproportionate importance to stream biota, even

408 when in low levels in apparently donor-controlled ecosystems (e.g. Brett et al., 2017).

409 *4.5 Benthic invertebrates*

410 Forest harvest in the vicinity of a stream typically reduces benthic diversity, causes uneven abundance

411 distributions and alters community composition (e.g. Zhang et al., 2009). While all these community-

412 level responses were detected in our northern streams, the responses were much weaker in the

413 southern region. This result underlines the fact that the impacts of human disturbance, including those

414 related to forestry, are highly context specific. The lack of congruence in macroinvertebrate responses

415 may be linked to region-specific differences in historical and contemporary land use, which has a

416 strong influence on the quality of the regional reference condition. In our northern regions, forestry

417 is the dominant land use and our reference streams mainly drain relatively mature forests. In contrast,

418 land use in the southern region is a mixture of more intense land use types (see Table 1), which has

419 likely resulted in compromised stream biodiversity even in regional reference conditions (see Harding

420 et al., 1998), thereby obscuring the detection of local forestry impacts

421 *4.6 Towards a better optimization of riparian buffers*

Buffer width requirements must be economically practicable as exceedingly wide buffers would impose economic costs on forest owners. However, defining an 'ecologically optimal riparian buffer' is challenging because of i) inconsistent scientific evidence, ii) focus on a single or a limited set of indicators, and iii) the traditional separation of terrestrial and freshwater realms in ecological research and ecosystem management. Future research should evaluate the optimization of riparian buffers in a more holistic way, by integrating biodiversity values and ecosystem processes of both the stream and the riparian forest to measure and protect cross-ecosystem multidiversity (Allan et al., 2014) and multifunctionality (Hector and Bagchi, 2007) across a gradient of riparian buffers of varying widths.

Recent scientific literature has offered novel solutions to riparian buffer management to more meaningfully compromise between economic profit and ecological change. Rather than the conventional 'one size fits all' -type fixed-width buffers, flexible buffers would enable allocation of conservation efforts to locations of particularly high biodiversity or otherwise of special ecological, hydrological or biogeochemical importance (Richardson et al., 2012; Kuglerová et al., 2014). Oldén et al. (2019) showed that careful thinning (30%) within 30-m wide riparian buffers can be allowed for without threatening the diversity of riparian plant communities. Similarly, Kreutzweiser et al. (2010) concluded that partial harvesting of > 30 m wide riparian buffers does not pose any major risks to stream biodiversity and ecosystem functioning. Partial harvesting of riparian buffers also decreases the predominance of coniferous trees, diversifying the allochthonous base of stream food webs and thus potentially enhancing stream biodiversity (Jonsson et al., 2017). Intentional small-scale harvesting of riparian forest may be advisable as it emulates the patchy disturbance typical of natural forest succession (Sibley et al., 2012); however, from the perspective of stream biodiversity, it is unlikely to be useful, or even acceptable, unless the riparian buffer is at least 15 m, and preferably 25 – 30 m wide.

4.7 Comparison of the forest certificates

Our results suggest that PEFC -compliant buffer retention measures will not support stream ecosystem processes and biodiversity at a level comparable to reference conditions. Albeit some deviation from reference conditions was observed also in the streams complying with the FSC certificate, we can safely conclude that while the FSC certificate performs reasonably well in providing protection for stream biodiversity and ecosystem functioning, PEFC does not. Dias et al. (2015) reached a similar conclusion for Mediterranean stream corridors and Kuuluvainen et al. (2019) for boreal forests, but neither of these directly tested for the biological effectiveness of the certificates. The responses of stream ecosystems to forestry impacts are context-dependent and vary across biological groups (see Richardson and Béraud, 2014; Lunn et al., 2017), presenting a major challenge to forest management at the stream-forest interface. In our streams, diversity responses to the apparent but complex environmental alterations caused by adjacent forest harvest varied between regions, implying that biological consequences of forest harvest remain to some extent unpredictable and therefore forests adjoining headwater streams should be managed with extreme caution.

Author contributions

JJ and TM conceived the idea and designed the study. JJ, IK and TM collected the data. IK identified macroinvertebrates and was responsible for most of the laboratory work. JJ analyzed the data. JJ led the writing of the manuscript and all authors contributed to drafts and edited the paper and gave final approval for publication.

Declaration of competing interest

The authors declare no conflicts of interest

Acknowledgements

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