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Multiple stressors in small streams in the forestry context of Fennoscandia: The effects in time and space

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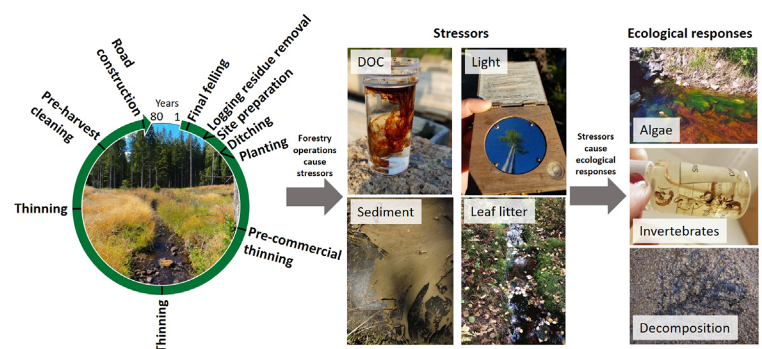
HIGHLIGHTS

- Streams in Fennoscandian production forests are subject to number of disturbances.
- Operations during the forestry rotation cycle introduce stressors to small streams.
- Individual stressors interact in time and space causing multiple stressor phenomena.
- Aquatic multiple stressors are not well understood in forestry dominated landscapes.
- Future research should focus on multiple stressors using experimental approaches.

GRAPHICAL ABSTRACT



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ABSTRACT

In this paper we describe how forest management practices in Fennoscandian countries, namely Sweden and Finland, expose streams to multiple stressors over space and time. In this region, forestry includes several different management actions and we explore how these may successively disturb the same location over 60–100 year long rotation periods. Of these actions, final harvest and associated road construction, soil scarification, and/or ditch network maintenance are the most obvious sources of stressors to aquatic ecosystems. Yet, more subtle actions such as planting, thinning of competing saplings and trees, and removing logging residues also represent disturbances around waterways in these landscapes. We review literature about how these different forestry practices may introduce a combination of physicochemical stressors, including hydrological change, increased sediment transport, altered thermal and light regimes, and water quality deterioration. We further elaborate on how the single stressors may combine and interact and we consequently hypothesise how these interactions may affect aquatic communities and processes. Because production forestry is practiced on a large area in both countries, the various stressors appear multiple times during the rotation cycles and potentially affect the majority of the stream network length within most catchments. We concluded that forestry practices have traditionally not been the focus of multiple stressor studies and should be investigated further in both observational and experimental fashion. Stressors accumulate across time and space in forestry dominated landscapes, and may

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interact in unpredictable ways, limiting our current understanding of what forested stream networks are exposed to and how we can design and apply best management practices.

1. Introduction

In recent decades, our understanding of how multiple stressors in lotic systems link to land-use has increased tremendously. Much of this work has emphasized how agricultural and urban land-use can simultaneously alter multiple physical and chemical properties of streams (e.g., Canobbio et al., 2009; Chará-Serna and Richardson, 2018; Matthaei et al., 2010; Wang et al., 2014). In contrast, forested areas are often used as reference systems against which agricultural and urban stream perturbations are compared (Burdon et al., 2020; Kuglerová et al., 2019; Villeneuve et al., 2018). However, for countries with a large and intensive forestry sector, production forests are not ideal reference systems. For example, Sweden and Finland, although each holding only about 1% of the world's commercial forest areas, together provide 14% of sawn timber, 11% of pulp, and 17% of paper that is traded on the global market (FAO, 2018). More than 50% of the land area in both countries is covered by forests, and over 70% of their forested area is classified as productive forest land that experiences some forestry operations (Luke, 2019; Skogsstyrelsen, 2020). Finally, although forestry practices in the two Fennoscandian countries typically use native tree species and rotation cycles are relatively long (60–100 years), several highly mechanized and intensive management operations are usually applied to the same stand during the rotation (Fig. 1).

These forestry operations cause a number of effects, or stressors, in receiving waters in boreal landscapes. In this context, individual stressors caused by forestry may be similar to those linked to agriculture and/or urbanization, but their effects in streams are likely to be modulated by the inherent physical and chemical constraints imposed at such high latitudes, including low temperatures, extreme seasonality in incident light, and low nutrient concentrations in fluvial systems. Moreover, the edaphic properties of boreal landscapes (e.g., large soil carbon pools) may also give rise to unique stressors under intense forest management, including increased dissolved organic carbon (DOC) and mercury loading (Laudon et al., 2011). Finally, because forestry is so spatially extensive in the Fennoscandian landscape, the nature of interactions among individual stressors arising from these activities likely differ in their spatial and temporal extent when compared to other land-uses in the region. While sometimes viewed as a single stressor

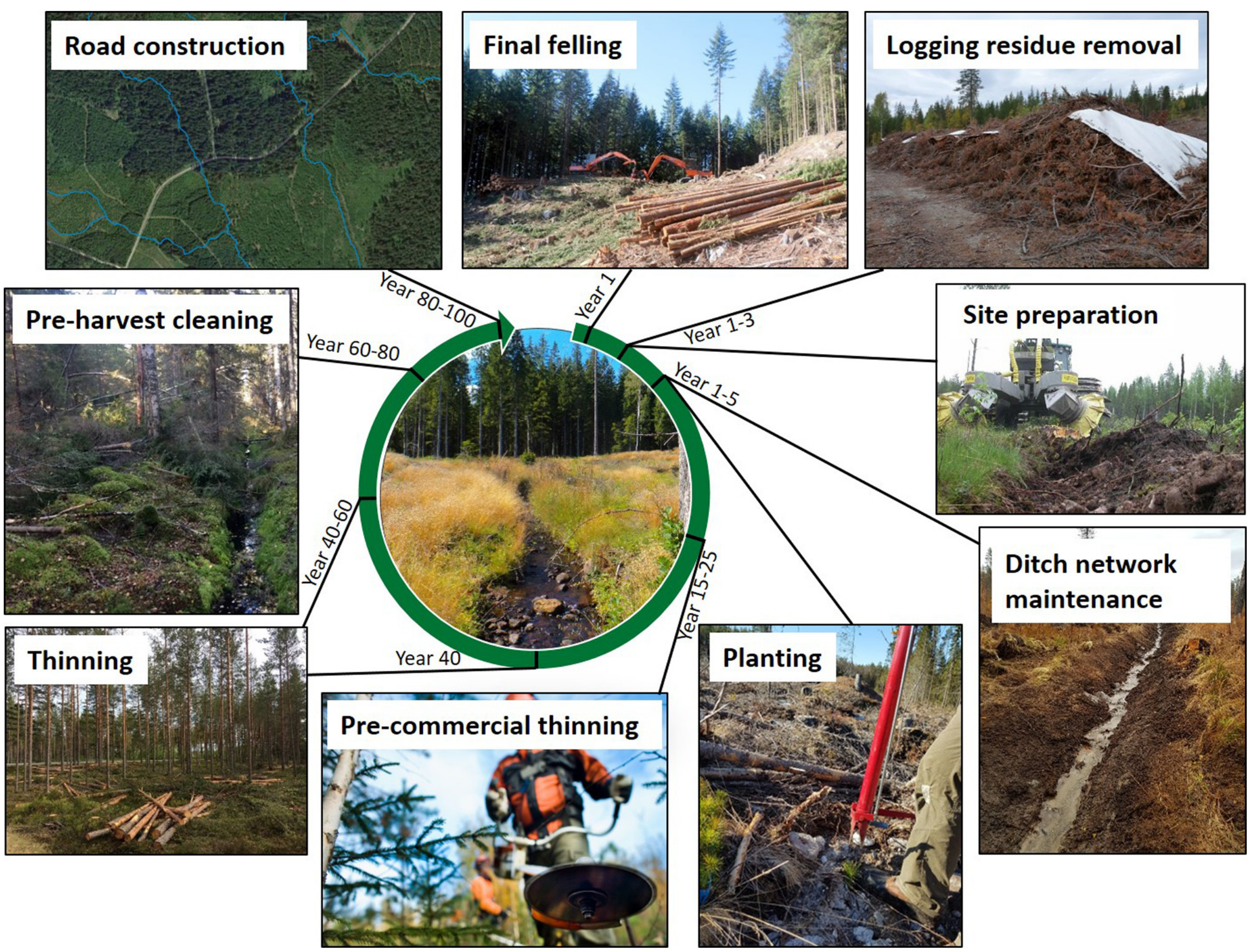

Fig. 1. A diagram of different forestry operations applied to a typical stand in a Fennoscandian production forest. The rotation cycle is assumed to be between 80 and 100 years but in southern parts of the countries it can be as short as 60 years, therefore the year intervals for each operation are approximate. Photos by: L. Kuglerová, E. Hasselquist, G. Hallsby, G. Egnell, and SCA.

(Fausch et al., 2010; Marttila et al., 2020) we argue that forestry in boreal landscapes should be addressed in the framework of multiple stressors to analyse how its combined influence affects ecological processes and organisms in northern watercourses.

2. Objectives and framework

In this paper, we investigate how multiple stressors are addressed in lotic systems in Fennoscandian production forests. We focus on Sweden and Finland because they are both dominated by production forests with a highly mechanized and intensive forest sector; however, we believe that the examples of the stressors are applicable to other managed forests. We first describe the different interventions during the forestry rotation cycle in Fennoscandian countries and what physicochemical effects they have been linked to in the literature (Section 3). Our literature search (using Google Scholar) was based on combinations of key words specifying location (Fennoscandia and/or Sweden and/or Finland and/or boreal), ecosystem (aquatic and/or freshwater and stream and/or lotic), and land-use (forestry and/or all keywords for operations identified in Fig. 1). Here we do not aim to provide a systematic review of all literature, but rather to give a comprehensive overview of the topic, identify gaps in the research, and help to establish a baseline for our conceptual and hypothetical framework. Therefore, after screening the titles and abstracts of studies generated by the keywords search, we incorporated insights from publications which reported abiotic responses to forestry operations in Sweden and Finland (Table A1). If a particular forestry operation and/or stressor was poorly documented in the two countries (e.g., road effects) we draw examples from managed forests elsewhere, prioritizing boreal landscapes. In addition, we present several examples from the reviewed literature of how these abiotic changes can represent stressors that influence aquatic biota (Fig. 2; Table A1).

Second, we elaborate on how individual stressors may interact creating a multiple-stressor situation (Jackson et al., 2016; Piggott et al., 2015), and we generate hypotheses regarding how such interactions in turn affect the ecology of small streams in Fennoscandia (Section 4). Given that the search for 'multiple stressor' combined with the aforementioned keywords yielded only a few studies which explicitly addressed multiple stressor phenomena in Fennoscandian context (Annala et al., 2014; Tolkkinen et al., 2015; Turunen et al., 2018), our analysis is largely based on examples from other biomes

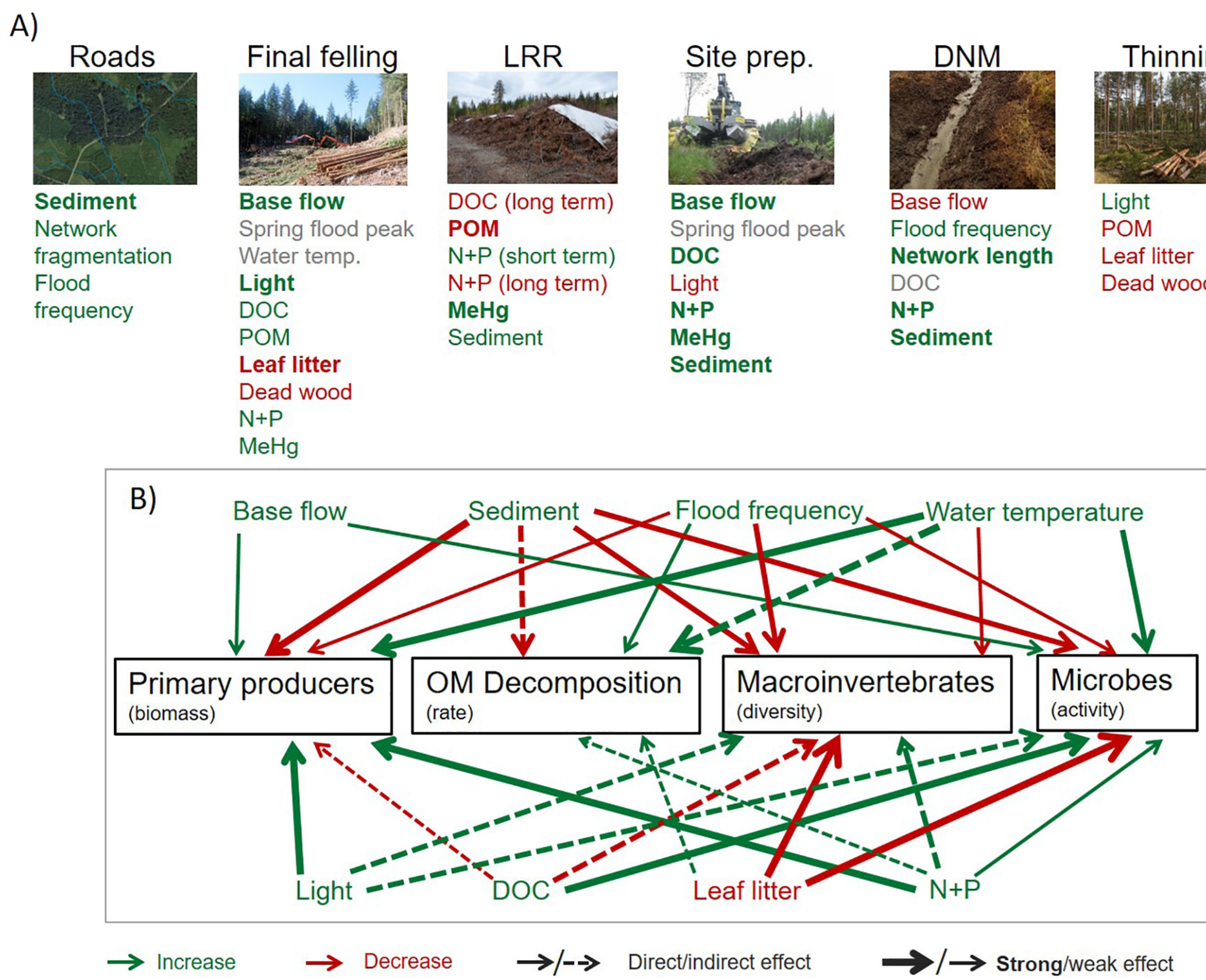

Fig. 2. A conceptual diagram of how different forestry operations cause changes in physicochemical parameters, which in turn can act as stressors for aquatic communities and processes in small boreal streams. A) The physicochemical parameters are listed below each intervention and indicate if they increase (green), decrease (red), or if both trends have been reported (grey) in boreal systems. Relatively larger changes are displayed in bold. B) Changes in the physicochemical parameters can act as stressors for aquatic organisms (e.g., primary producers, macroinvertebrates, microbes) and processes (e.g., decomposition). The color of the stressors corresponds with the increase (green) or decrease (red) with the forestry operations from A), and the arrows indicate if a change in the stressor causes an increase (green) or decrease (red) in an ecological response. Arrow thickness corresponds to the relative strength of the effect; arrow type indicates direct (full line) or indirect (dashed line) effects of the stressor. The mechanisms behind the changes in the physicochemical parameters as well as their effects on aquatic communities and processes are explained in text and in detail in Table A1. DNM = ditch network maintenance, DOC = dissolved organic carbon, LRR = logging residual removal, MeHg = methylmercury, N = nitrogen, P = phosphorus, (P)OM = (particulate) organic matter.

and/or land-uses, and the authors' understanding of boreal waters. Third, we briefly discuss how multiple stressors operate across time and space (Sections 4.1 and 4.2). Finally, based on the identified knowledge gaps we suggest avenues for future research (Section 5).

Our work focuses on responses in small streams (i.e., 1–2 stream order and/or < 3 m width, or headwaters). In recent years, it has been argued that low order streams might be the most affected part of the river network by forestry operations (Hasselquist et al., 2020; Hylander et al., 2002; Kuglerová et al., 2017). This is because *a)* the proportion of affected area (e.g., harvested) in relation to catchment area is generally larger in small catchments, *b)* small streams are more strongly linked to surrounding terrestrial ecosystems compared to larger rivers (Tolkkinen et al., 2020; Vannote et al., 1980), and *c)* they are usually left with minimum protection in the form of riparian buffers (Hasselquist et al., 2020; Kuglerová et al., 2020). Correspondingly, we assume no, or little riparian buffer protection has been implemented within an even-aged production system. Although 'small stream' was not included as a keyword for the literature search, studies on large stream were only used as examples for cumulative effects (Section 4.2). We use the term boreal (both through the text and in the literature search), broadly to include northern, middle and southern boreal zones as well as borealnemoral (or hemiboreal) zone across both countries.

3. The forestry cycle and associated stressors

We define stressors as factors which exceed the range of natural, undisturbed conditions in response to a human activity (Townsend et al., 2008), for example, change in runoff conditions, altered water temperature and/or alteration in solute concentration after a certain forest management action. Below we review multiple stressors in relation to multiple disturbances/pressures (Paine et al., 1998), which are here defined as the human activities triggering stressors, namely the individual management operations within a forest stand over the rotation cycle (Fig. 1). It should be noted that in some forest stands certain practices can appear several times during the rotation period, such as ditch network maintenance (DNM), road construction/maintenance, and/or thinning. On the other hand, in other stands, some practices are not conducted at all (e.g., planting, DNM, logging residue removal).

3.1. Final felling

Final felling is the most studied forestry operation with respect to effects on aquatic ecosystems, especially in small streams (Futter et al., 2016; Laudon et al., 2011; Tolkkinen et al., 2020). In the boreal forest of Fennoscandia, clearcutting of even-aged (60–100 years), single-species dominated forest stands is the most common final felling strategy (Esseen et al., 1997; Hallsby, 2007), although some stands are subject to continuous cover forestry (Kuuluvainen et al., 2012). Due to the decrease in evapotranspiration from upland vegetation, clearcutting is typically associated with elevated groundwater (GW) levels and increased flow to streams, which can temporarily increase discharge and alter flow-peaks dynamics (Ide et al., 2013; Schelker et al., 2013; Sørensen et al., 2009). Such hydrological effects also result in increased nutrients and DOC leaching, as well as increased sediment transport (Futter et al., 2016). The use of heavy machinery can also damage the integrity of forest soils (Ågren et al., 2015), which typically results in further changes in hydrology and soil water chemistry. Removing forests close to the stream channels modifies light and thermal regimes because of increased solar radiation reaching the water surface and changed microclimate in the riparian zone (Johnson and Almlöf, 2016. Oldén et al., 2019). Resource subsidies from riparian vegetation are also changed, with substantial decreases in leaf litter and deadwood entering the streams if riparian forests are partially or completely harvested (Richardson et al., 2005; Lidman et al., 2017). All these individual stressors have been linked to a number of responses by aquatic biota (Table A1, Fig. 2).

3.2. Logging residue removal (LRR)

Removal of all logging residue has emerged as a practice that can contribute to the increasing demands for bioenergy (Bouget et al., 2012; Marttila et al., 2020). In the Fennoscandian countries, this practice has been applied since the 1970s (Egnell, 2017) and it is predicted that further increases will be seen in the next decades to meet the fossil fuel reduction goals (European Commission, 2008). Logging residue removal (LRR) involves collecting of branches, tree tops, and smaller trees left on site during final felling, and/or extracting stumps for bioenergy purpose. This can have an effect on receiving waters due to less organic material (OM) available for decomposition in the upland soils and consequently less OM, nutrients, and base cations delivered to streams (Akselsson et al., 2007; Bouget et al., 2012; Table A1). Although the long-term aquatic responses to this practice are not well understood (Marttila et al., 2020), there is evidence from Finnish and Swedish catchments that LRR can reduce soil nutrient pools to levels too low to sustain the expected growth of the next generation of trees (Egnell, 2017). Such an effect will likely result in more intense fertilization of the LRR treated stands in the future. Forest fertilization is currently applied to <0.2% of the total area of production forests (Skogsstyrelsen, 2020, Luke, 2019) and has not been found to significantly affect nutrient levels and, consequently, organisms in small boreal streams (Gonzalez and Plamondon, 1978; Lucas et al., 2011). Nevertheless, the effects of forest fertilization, if applied more frequently on larger forest areas in the future, should be carefully evaluated and understood before it becomes a common practice.

Physical perturbation of soils during LRR may also cause increased sediment transport (Ukonmaanaho et al., 2016) and is associated with an increase in toxic methyl mercury (MeHg) concentrations in surface waters (Eklöf et al., 2018; Table A1). Due to bedrock composition and atmospheric deposition, levels of Hg in Swedish and Finnish soils tend to be high (Bishop et al., 2009). Anaerobic microbial processes in the small pools left after the stump removal, in combination with access to fresh organic carbon sources, create the potential for toxic MeHg production that can be directly delivered to aquatic ecosystems, where it can be taken up by consumers (Eklöf et al., 2018). Because MeHg bioaccumulates, elevated levels are found in most fish in Swedish freshwaters, causing a direct risk to human health if consumed (Bishop et al., 2009).

3.3. Site preparation

To increase the chances of seedling survival, mechanized site preparation by disk trenching or mounding is typically done in Fennoscandian forests (Esseen et al., 1997). During site preparation, soils are scarified and the top humus layer turned over, exposing mineral layers, increasing nutrient availability, soil aeration, and drainage (Hallsby, 2007; Örlander et al., 1996). Since this practice causes physical soil disturbance, it is not allowed close to surface waters or on wet soils, especially in riparian areas (Skogsstyrelsen, 2019). However, creating continuous furrows in the majority of upland areas cannot be completely mitigated by leaving a small strip of unaffected land next to streams. Thus, soil preparation has been documented to affect a number of chemical and physical characteristics in small streams, and in turn, influence ecological processes and communities (Table A1, Fig. 2). A major concern connected to site preparation is soil erosion, with potentially elevated sediment transport to streams (Palviainen et al., 2014). Further, similar to the LRR, site preparation has also been found to lead to increased risks of MeHg delivery to streams. Depending on site preparation equipment, settings and mode of operation, the impact will range from intermittent scalps leaving the mineral soil intact to deep pits or continuous furrows that result in standing pools of water with fresh OM. Eklöf et al. (2018)

recently showed that stump harvesting triggers higher MeHg concentrations compared to site preparation. However, in the same study, the authors also concluded that the levels of MeHg supply to streams are context dependent and not all Fennoscandian landscapes will be susceptible to increases after individual or even combined forestry practices. Nevertheless, preventing elevated MeHg in surface water should be a priority during forestry operations, and several studies have suggested that direct hydrological connectivity (e.g., via machine tracks) between uplands and streams should be kept to a minimum to reduce these inputs (Bishop et al., 2009; Laudon et al., 2011; Skyllberg et al., 2009).

3.4. Planting

In Fennoscandia, the majority of stands are planted with native commercial coniferous species following site preparation; typically Norway spruce and Scots pine (Hallsby, 2007). Although planting per se might not lead to immediate stressors in aquatic ecosystems, its long-term effect is potentially far reaching (Table A1). Although planting commercial species in riparian zones is not recommended by the agencies today, management prescriptions and silvicultural measures that favor coniferous trees all the way to the water edge of small streams dominated for more than 50 years, up until the 1990s (Enander, 2007). Thus, mature riparian forests around headwater systems are often dominated by coniferous trees of uniform age and structure (Ring et al., 2018). Such forests provide low energy detrital resources (i.e., needles) to aquatic and riparian consumers and prevent solar radiation from reaching the water's surfaces (Jonsson et al., 2017; Lidman et al., 2017; Nieminen et al., 2018). Although regulation of light is one of the functions desired from riparian forests (Skogssyrelsen, 2019), it has been suggested that natural disturbances create canopy gaps in riparian zones that are beneficial to aquatic communities (e.g., by promoting instream productivity), and that forest management should strive to mimic this dynamic (Sibley et al., 2012; Tolkkinen et al., 2020). If more deciduous species and/or larger variation in canopy structure were encouraged by appropriate management, including the planting stage, the subsequent effects of final felling on light, temperature and resource subsidy regimes could be less pronounced.

3.5. Thinning

The forestry cycle of Swedish and Finnish production forests typically includes at least two interventions between planting and final harvesting, during which undesired individuals that compete with desired future crop trees are felled (Fig. 1). Those trees typically include deciduous species (e.g., birch, rowan, aspen), which self-established in the new stand, but also individuals of commercial species which might have regenerated naturally (Hallsby, 2007). Some of the naturally regenerated individuals could be spared as a substitute for missing or injured crop trees. Further, pre-harvest brush cleaning (sometimes preceeding final felling or thinning) removes any undergrowth trees to technically facilite harvester operations. This practice further simplifies the structure and composition of the residual stand and reduces the number of future silvicultural options. A large part of the forest sector agrees that these operations should be avoided in riparian areas (Skogsstyrelsen, 2019). However, the riparian zone around small streams is not well defined, and cleaning and thinning operations occur frequently within <10 m distance from water courses.

Pre-commercial thinning (also called "cleaning") is typically performed about 15–25 years after planting while commercial thinning (or simply "thinning") of more mature stands occurs once to a few times later in the rotation cycle, depending on site fertility (Fig. 1). During cleaning, the number of saplings removed can be two to ten times higher than the number of planted individuals (Hallsby et al., 2015), while during thinning 20–30% of total basal area is harvested (Hallsby, 2007). Importantly, deciduous species are targeted during all thinning operations. Although thinning immature riparian forests has not been studied in relation to changes in boreal aquatic ecosystems, inferences can be drawn from studies addressing partial harvesting in riparian zones elsewhere (e.g., Muto et al., 2009; Oldén et al., 2019; Peura et al., 2020). Accordingly, thinning and extracting deciduous saplings and immature trees in areas surrounding streams is likely to reduce the availability of high quality resources for aquatic consumers (Webster and Benfield, 1986). Indeed, we know that litter used by aquatic consumers can originate from sources up to 30 m away from the streams (Bilby and Heffner, 2016; Kiffney and Richardson, 2010). Assuming that aquatic biota recover from all the previous interventions by the time of thinning, and are adapted to receive both deciduous and coniferous riparian subsidies (Jonsson et al., 2017), thinning could lead to a long-term shortage of resources for aquatic consumers (Table A1, Fig. 2).

Due to interventions like cleaning and thinning that typically create even-aged stands, deadwood is generally limited in production forests of Sweden and Finland (Siitonen et al., 2000). In streams, deadwood is important as a substrate for biofilms, a habitat structure for many aquatic organisms, as well as a source of OM (Mäenpää et al., 2020; Richardson et al., 2005). In unmanaged forests, deadwood is provided to streams more continuously due to small scale disturbances and competition among trees (Bahuguna et al., 2010). In production forests, deadwood typically does not enter small streams until final felling if riparian buffers are saved and subsequently blown down (Grizzel and Wolff, 1998; Peura et al., 2020).

Thinning will also cause more light to reach the riparian forest floor and the water surface of streams compared to pre-thinning conditions (Mallik et al., 2013). Riparian microclimate, especially temperature and humidity, has been also shown to respond when riparian zones were partially harvested in Finland (Oldén et al., 2019), which can have subsequent effects on the stream physical properties. Finally, depending on the volume of thinned trees and the placement of strip roads to allow machine access, small changes in water balance and water quality, similar to final felling, can occur (Kreutzweiser and Capell, 2001). It is however likely that the single and combined effects of commercial and pre-commercial thinning will be more subtle compared to the other forestry operations (Kreutzweiser et al., 2010).

3.6. Road construction and off road driving

In Sweden, the goal is to have the harvested stand located within 500 m of the nearest road (Esseen et al., 1997), thus road building and maintenance is a typical part of harvest operations. Road construction and use, as well as off-road driving during other operations, has been shown to stimulate soil erosion that can cause large amounts of sediment to enter nearby streams (Ågren et al., 2015; Kreutzweiser and Capell, 2001). In many cases, small streams must be crossed with large machinery (Fig. 1), and even permanent and/or temporary bridges cannot completely prevent increased sediment loading (Aust et al., 2011). Further, poorly constructed stream crossings represent barriers for downstream material flow and movement of organims (Luce and Wemple, 2001; Perkin et al., 2013). Surprisingly, the effects of roads and driving on sediment supply to small streams and the further consequences for aquatic organisms have received little attention in Sweden and Finland (Futter et al., 2016; Table A1). This is probably because fluvial systems in Fennoscandian countries are often thought to be sediment-limited, due to geology, relatively low topographical relief, as well as historical channel modifications during the channelization era (Rosenfeld et al., 2011). Nevertheless, excess fine sediment covering stream bottoms nearby recently harvested (and thus trafficked) areas in both Sweden and Finland were recently reported (Kuglerová et al., 2020). Further, sediments washed out from forest roads also have an effect on water chemistry, including oxygen levels, pH, conductivity, and/or heavy metals (Aust et al., 2011; Emilson et al., 2017; Ryan, 1991; Zhang et al., 2014). Construction of roads in small catchments can also change the hydrological pathways and regimes. High density of roads with nearly impervious surfaces will lead to greater hydrological flashiness, which in turn further increases the risk of erosion and sediment transport, as well as export of particulate organic matter (POM) from the affected stream reaches (Luce and Wemple, 2001).

3.7. Ditching and ditch network maintenance (DNM)

In contrast to roads, ditching practices and their effect on sedimentation and water quality have been intensively studied, especially in Finland (Table A1). Together, Sweden and Finland represent the area with the highest density of drained peatlands in the EU as almost 25% of forests in these countries have been artificially drained over the past century to increase timber production (Päivänen and Hånell, 2012). Ditching lowers the GW level and increases aeration of the rooting zone, thus improving tree growth given that other factors, e.g. nutrients, are not limiting (Päivänen and Hånell, 2012; Sikström and Hökkä, 2016). As ditches age, ditch network maintenance (DNM) may be required to sustain drainage and adequate timber production, but this is also a large source of sediment and nutrient loads to receiving water bodies (Nieminen et al., 2017). It was recently estimated that drained areas in Sweden and Finland may account for over 60–70% of the total sediment and nutrient loads from forests (Nieminen et al., 2018). DOC export from newly ditched catchments might be lower, compared to non-ditched areas, because GW table does not reach OM rich top soils (Nieminen et al., 2018). However, the long-term legacy of DNM seems to results in increased levels of DOC (Asmala et al., 2019). Furthermore, ditches lead to higher peak flows and lower baseflow, thus changing important hydrological patterns and water residence time in the system (Holden et al., 2004; Nieminen et al., 2018). Ditches also change the shape of stream networks, in particular extending the total length and density of tributaries, and increasing the number of confluences (Hasselquist et al., 2017).

4. Interaction among stressors

Each forestry operation is individually associated with several physicochemical stressors and it is therefore inevitable that these stressors interact. Their interactive nature is not always additive or reversible, but instead can act synergistically or antagonistically (Townsend et al., 2008; Fig. 3). Consequently the responses of aquatic communities and processes to combinations of stressors are typically non-linear (Piggott

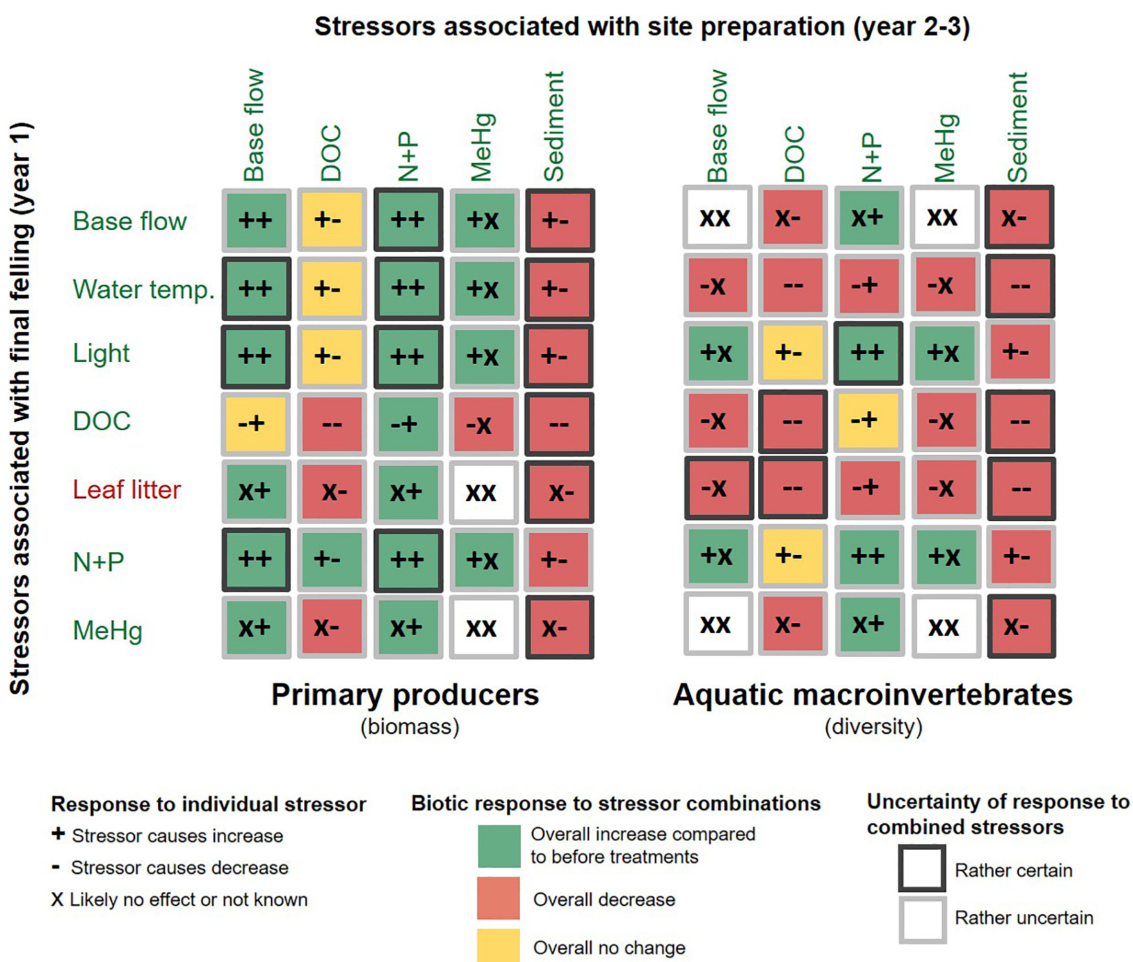

Fig. 3. Hypothesis about potential interactions of selected stressors associated with final felling (y axis) and site preparation (x axis) and how they affect biomass of primary producers (PP) and diversity of aquatic macroinvertebrates when they combine. Response to individual stressor are based on Fig. 2B and are displayed as ++ when both stressors cause increase, −− decrease or +− when one causes increase and the second decrease in the response by biota. The overall change in PP biomass and macroinvertebrate diversity after both practices are implemented is color coded (green = overall increase, red = overall decrease and yellow = overall no change, empty box indicates unknown response). The uncertainty about the responses (increase, decrease, or no change) to the two interacting stressors is displayed as light grey outline if the trend is rather uncertain and as dark grey if rather certain. Here, the two forestry interventions are applied to the same location at different times (1–3 years apart), but they can also be applied simultaneously at sites adjacent to each other (the effects are then cumulative in space). Responses are assumed on a short time scale (within 1–5 years post-treatment). The mechanisms behind the changes in the physicochemical parameters as well as their effects on aquatic communities and processes are explained in text and in detail in Table A1.

et al., 2015) which complicates predictions. For example, removal of trees around small streams should elevate water temperature, which can negatively affect cold-water adapted species, but may concurrently increase primary productivity that provides high quality resources to food webs (Johnson and Almlöf, 2016; Newton and Ice, 2016). However, in the Fennoscandian countries, this effect can be counterbalanced by changes in the evapotranspiration upon harvesting that lead to increases in the contribution of cold GW to the streams (Schelker et al., 2013). Further, temperatures tend to be low even in the summer in boreal biomes (with an average air temperature for July around 15 °C, [Laudon et al., 2013]). Thus, the overall change in water temperature can be nuanced, with even decreases for small streams flowing through recent clearcuts (Jonsson et al., 2017; McKie and Malmqvist, 2009).

Increasing light in the instream environment after canopy removal is commonly associated with greater primary productivity, especially when combined with higher water temperature (e.g., Burrows et al., 2015; Holopainen and Huttunen, 1992). However this response can, to some extent, be lowered by increasing DOC concentration and browning of water (so called brownification) following harvest, site preparation, and/or ditching (Kritzberg et al., 2020; Nieminen et al., 2015). Depending on canopy cover and degree of browning, shallow streams might not experience substantial reduction of incident light to benthic surfaces, yet deeper channels may become light limited in response to elevated color. What effect those contrasting trends in light might have on aquatic organisms and processes is not known (Fig. 3). Further, increases in DOC are typically matched by decreases in pH, which can threaten a number of aquatic organisms in poorly buffered environments (Petrin et al., 2007; Serrano et al., 2008). At the same time, increased DOC and nutrients, together with greater supplies of POM from logging residues (if not removed for biofuels) may offer an alternative subsidy for aquatic consumers (Kritzberg et al., 2020; Lupon et al., 2019), when riparian leaf litter is removed. Vegetation removal near streams is acknowledged as a disturbance to resource-subsidy dynamics, and a number of studies have shown how this is propagated through stream and riparian food webs (Erdozain et al., 2019; Lidman et al., 2017; Nakano and Murakami, 2001; Richardson and Danehy, 2007). However, much less is known about how a shift to alternative subsidies (e.g., DOC and POM) may influence the broader aquatic food web (Fig. 3).

Sedimentation on the bottom of the streams, associated with a number of forestry interventions (Fig. 2), can reverse some of the effects of increased light (after final felling) on algae and/or bryophyte accrual, as this may reduce available surfaces for growth and/or promote scour and burial (Louhi et al., 2017; Turunen et al., 2020; Fig. 3). Further, sediments can regulate access to POM inputs sometimes observed after harvest by burying OM in channel sediment, making it unavailable for many aquatic consumers. Such burial can further counteract the potentially increased OM decomposition if water temperature raises (at least during summer), by reducing the role of shredders in this process (Emilson et al., 2017; Kreutzweiser et al., 2008). Finally, sediments can also burry deadwood recruited to streams after final felling (from blown down riparian buffers), making it unavailable as a substrate and habitat.

Increased base flow due to less evapotranspiration from harvested uplands might provide more stable aquatic habitats in the smallest and intermittent streams, and increased DOC inputs may support higher activity of microbial communities (Lupon et al., 2019). On the other hand, flashy hydrographs due to lower water residence time on harvested catchments pose a stress for the aquatic organisms (including microbes) which cannot cope with a high flood frequency (Holomuzki and Biggs, 2000; Lytle and Poff, 2004). It is also possible that increased discharge due to final felling might partially counteract sediment loading due to faster flow and thus increased export downstream. While this is beneficial for the stream reach within the harvested area, the sediment will cause negative effects when deposited further downstream, due to cumulative effects in space (Seitz et al., 2011). However, the overall outcome of changed hydrology interacting with other stressors introduced by forestry operations is not documented and should be subject of further research (Fig. 3). Such interactions may also be exacerbated by ongoing climate changes that further alter hydrological patters at northern latitudes (Mustonen et al., 2018).

In further respect to hydrological changes, the effects of ditches on aquatic ecology may also be difficult to predict. For example, Annala et al. (2014) found that the additional impact of forest drainage on species richness in naturally acidic streams differed between organismal groups. Here, drainage had no effect on diatoms, a weaker effect than predicted for bryophytes, but an additive effect on invertebrates. Working in the same streams, Tolkkinen et al. (2015) showed that litter decomposition rates were reduced by upstream ditch disturbance in naturally acidic, but not circumneutral streams. The greater sensitivity of decomposition to upstream ditching in acidic streams was attributed to inherent differences in fungal communities, which were simplified and thus potentially more vulnerable to disturbance (Tolkkinen et al., 2015). Similar to the patterns found for MeHg after LRR and site preparation, the effects of ditches seem to be context and taxa dependent.

All of the above interactions demonstrate that addressing the single effects of individual stressors in small streams influenced by forestry will be insufficient for understanding aquatic responses. Further, some interactions have not been studied at all (e.g., ecological effects of elevated MeHg and interactions with other stressors). In Fig. 3, we present hypotheses related to how individual stressors identified in Fig. 2 may interact and consequently influence primary productivity and macroinvertebrate diversity in small boreal streams. This hypothetical framework is poorly resolved in the context of forestry in Fennoscandia and should be a center of future research. We acknowledge that our framework is simplified. For example, we assume no other direct factors, besides stressors, limiting the responses of primary producers and aquatic macroinvertebrates (e.g., grazing, habitat and resource competition and limitation) and we only assess short-term responses on a local scale. Nevertheless, these hypotheses are a starting point for future research that aims to disentangle multiple stressor phenomena in production forests. In the next sections we further elaborate on how multiple stressors should be put in the context of time (Section 4.1) and space (Section 4.2) in order to comprehend ecosystem change after pressures from forestry.

4.1. Compounding effects of different operations over time

Not only are small streams in forestry-dominated landscapes exposed to multiple stressors, but they are also exposed to several doses of the same stressor over time. Many studies about forestry effects on recipient waters neglect the fact that the same stream reaches are subjected to repeated management activities during the forestry cycle (Fig. 1) and thus, become perturbed multiple times over decades. For example, it has been shown that site preparation typically accelerates the effects of final felling and other previous forestry interventions, and that it primarily is the combined effects which pose a threat to water quality and aquatic ecology (Fig. 3). Namely, soil scarification can increase the duration and/or magnitude of hydrological change observed after final felling due to inhibited evapotranspiration from disturbing the early successional vegetation layer (Schelker et al., 2012). Similarly, site preparation can elevate carbon and nutrient export even further than final felling, and thus DOC, N (nitrogen) and P (phosphorus) concentrations in adjacent streams may increase for several years after the combined operations (Piirainen et al., 2007; Schelker et al., 2012). It has been speculated (e.g., Futter et al., 2016; Nieminen et al., 2017) that applying LRR could counteract the effects of final felling and site preparation (increasing DOC, base cations and N leaching), leading to an overall net balance of the carbon and nutrient delivery to streams over a longer time scale. However, Mlambo et al. (2015) and Ukonmaanaho et al. (2016) showed that both DOC and N concentrations were higher in streams adjacent to both only logged and logged

+ LRR treated sites, compared to reference (unharvested sites) in the short term. It has also been shown that if the logging residual material is piled up and left on site for some time before collection, N and P can leach into soils (Nieminen et al., 2018). Therefore, if strong hydrological connections between the upland soils and streams exist, these excess nutrients could be delivered to surface waters (Laudon et al., 2016). In general, for nutrient limited boreal streams, such nutrient additions are unlikely to induce eutrophication, and would likely influence stream communities and food webs through increased rates of aquatic primary production and decomposition (e.g., Burrows et al., 2015, 2017; Fig. 2). It is still unknown what the different consequences would be of a single, acute flushing event of nutrients versus a chronic, prolonged input due to a combination of forestry practices in a boreal context.

Another important aspect of interacting perturbations (and consequent stressors) over time is application of ditch network maintenance on catchments that have undergone final felling, LRR, and/or site preparation. DNM will increase the hydrological connectivity between surface waters and uplands (Laudon et al., 2016), which will accelerate transport of sediments, DOC and potentially MeHg to streams. Overall, water quality in drained and undrained catchments has shown very different responses to additional forestry operations (Nieminen et al., 2017). Therefore, consideration of ditching practices in combination to other anthropogenic disturbances offers a unique opportunity for multiple stressor studies.

An important question remains concerning the scale of additive effects on the volume of transported sediments over time, and how this in turn affects aquatic biota and freshwater processes over longer term. Jyväsjärvi et al. (2014) concluded that even about 20% of the stream bed covered by fine sediments is harmful for a number of aquatic organisms and freshwater processes in Finnish streams. It is possible that this threshold can be exceeded and maintained for a long time in many streams by the numerous forestry-related practices in Fennoscandian production forests. Importantly, we also know from research in other regions that the effects of sedimentation on benthic communities can persist for many decades (Harding et al., 1998).

4.2. Compounding effects of different operations across space

In Fennoscandia, forestry operations are performed in several thousand stands every year. Due to logistical reasons (e.g., maintained roads), closely situated stands can be treated at the same time, or only couple of years apart (Fig. 4). Since the Fennoscandian forests grow slowly and many forestry operations take place within the first couple of years (Fig. 1), streams situated in two clearcuts separated by a just a few years can often be considered as experiencing concurrent stressors. This is of special concern if the two adjacent clearcuts are intersected by the same stream. There are currently insufficient guidelines for time gaps for operations in adjacent stands (The Swedish Forest Agency, personal communication) or consideration of catchment boundaries in forestry planning. This represents a problem for water quality and ecological integrity on a catchment scale because a) if a large number of small streams is impacted within a short time, there is a large area of impaired habitat in the same catchment, and b) many small streams that intersect stands close to each other combine in the same network (Fig. 4) which may cause downstream cumulative effects. The severity of cumulative effects has been considered in urban and agricultural catchments (Jones et al., 2017; Kielstra et al., 2019; Mineau et al., 2015; Seitz et al., 2011), but has not been adequately and empirically addressed in production forests (Kuglerová et al., 2017; Richardson, 2019). This is problematic also because forested headwaters receive few mitigation measures (i.e., buffers, Kuglerová et al., 2020) to prevent impairments and thus the potential for cumulative effects is large.

Several studies have estimated thresholds of forestry operations within a catchment which should not be exceeded. Palviainen et al. (2014) show that a threshold of 30% clearcut area within a catchment should not be crossed in order to keep N, P, suspended sediments and C within normal range in Finnish production forests. Somewhat lower thresholds (11–25%) were found for DOC in northern Sweden (Schelker et al., 2014). Löfgren et al. (2009a) investigated several parameters (water chemistry, aquatic diatoms and benthic macroinvertebrates) indicating ecological status in two areas in Sweden. Although not all differences across the catchments could be associated with forestry land-use, they showed that sampled sites in a catchment with a lower proportion (15%) of clearcut + recently planted areas had better ecological status compared to those with higher proportion (16–35%) of recently harvested and planted area. Based on those and other studies (Burrows et al., 2015; Jyväsjärvi et al., 2014; Jonsson et al., 2017; Kreutzweiser et al., 2008) it seems that the threshold for the spatial extent of harvesting within short (1–10 years) time period in boreal forest catchments should not exceed 15–30%, in order to avoid deterioration of water quality and aquatic ecology on a catchment scale. However, these studies only focused on final felling and/or site preparation. Other practices across different stands and their added effects are

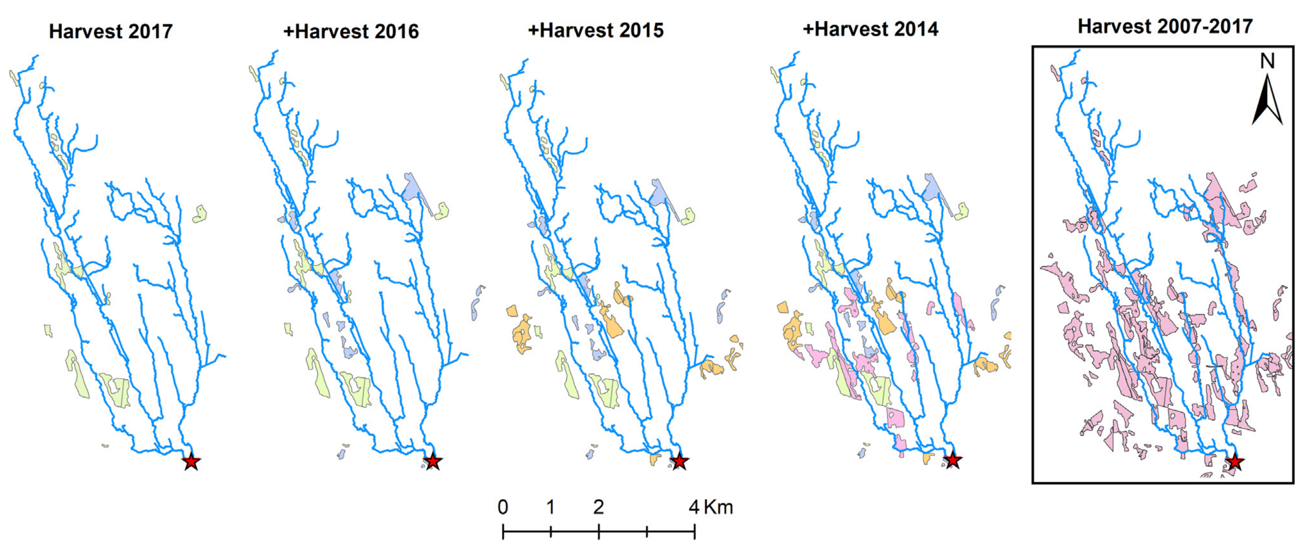

Fig. 4. Final felling (clearcutting) applied within the same catchment (draining to the location marked by the star) in northern Sweden. Final-felled areas are displayed for the year 2017 (to the left) and as added over the previous 4 years in different colors. On the figure to the far right, all harvest within 10 year window (2007–2017) within the catchment is displayed in pink.

typically not evaluated. It is likely that if all forestry operations are considered, the threshold for allowable catchment-scale operations would be lower than 15%.

An important aspect to consider when estimating thresholds for how much harvest is allowed at the catchment-scale is a proximity of a stand to a stream. Stands situated hundreds of meters away from the stream should have lesser influence on stressors compared to stands located closer to channels. The spatial arrangement of treated stands is also related to hydrological connectivity between uplands and surface water, which can be rather variable across forested landscapes (Laudon et al., 2016). Further, artificial hydrological connectivity (e.g., drainage ditches, ruts from machines) can change water flow paths (Ågren et al., 2014; Hasselquist et al., 2017) and this can offset the negative correlation between increasing stand distance and magnitude of a stressors. Similar to cumulative effects, the distance-weighted influences of land-use on stream water quality, quantity, and aquatic ecology have been addressed in agricultural and urban landscapes (King et al., 2005; Urban et al., 2006; McBride and Booth, 2005) but have not previously received much attention in forestry.

Some of the stressors that we described will occur locally in small streams but might not propagate to downstream reaches. For example, increases in nutrients and carbon are typical for small boreal headwaters following forestry practices (Futter et al., 2010), but in higher order streams their magnitude is much smaller (Schelker et al., 2014; Schelker et al., 2016). This likely means that some substances are locally utilized by aquatic organisms within small streams (Lupon et al., 2019). However, as downstream cumulative effects are largely unexplored in the forestry context (Kuglerová et al., 2017), it is difficult to make a general conclusion whether nutrients accumulate or dissipate along the river network. Further, downstream propagation from headwaters can also be minimized for some stressors (e.g., sediments, POM) if segments of streams do not connect to downstream reaches because they are blocked at road crossings or by a collapsed temporary bridge. This will however affect migration and dispersal paths for a number of organisms, especially if organisms with active dispersal attempt to avoid locally disturbed reaches by migrating to nearby undisturbed reaches (Holomuzki and Biggs, 2000). From a river network perspective, this can change the broader metacommunity dynamics (Göthe et al., 2013, Kuglerová et al., 2015, Table A1) and, in worst case scenario, eliminate source populations. In this context, ditches can extend the total network length, alter the shape of tributaries, and increase the frequency of confluences (Hasselquist et al., 2017; Fig. 5). Assuming that ditches can offer similar habitat for aquatic organisms as small streams (a contention which has not been tested) this can positively affect aquatic processes and communities, simply by offering more habitat.

5. Conclusion and future research

Throughout this review, we have discovered that forestry is a relatively underrepresented land-use activity in the multiple stressor literature in lotic systems. Although it has been recognized for decades that forestry operations create a number of physical, biogeochemical, and ecological changes in adjacent streams, little is known about how they combine to influence aquatic ecosystems. Importantly, while some studies clearly investigated stressors in combinations (e.g., Annala et al., 2014; Eklöf et al., 2018; Erdozain et al., 2019; Lidman et al., 2017; Löfgren et al., 2009a; McKie and Malmqvist, 2009; Mlambo et al., 2019; Nieminen, 2003; Palviainen et al., 2014) the term 'multiple stressor' is rarely used in the forestry literature. We conclude that more awareness should be directed to multiple stressor phenomena in relation to different forestry operations, especially in the intensively managed forests of Fennoscandia. Failing to properly acknowledge stressors and their interactions during the forestry cycle inevitably results in poor management decisions targeting ecological mitigations, and lead to ecological surprises (Paine et al., 1998).

Given the interests in multiple stressors in agricultural and urban contexts, many techniques, approaches and tools have been developed to address the types of interactions among single stressors (Jackson et al., 2016). Therefore, forestry research has a great opportunity to use these methods to advance our understanding of interacting stressors in space and time in streams flowing through catchments dominated by production forests. Most studies reviewed here present results from field observations. Although these studies have advanced our understanding, the disadvantage of observational studies (surveys) is that causality cannot be determined (Downes, 2010). To certain extent, causality can be addressed in BACI (before-after-control-impact)

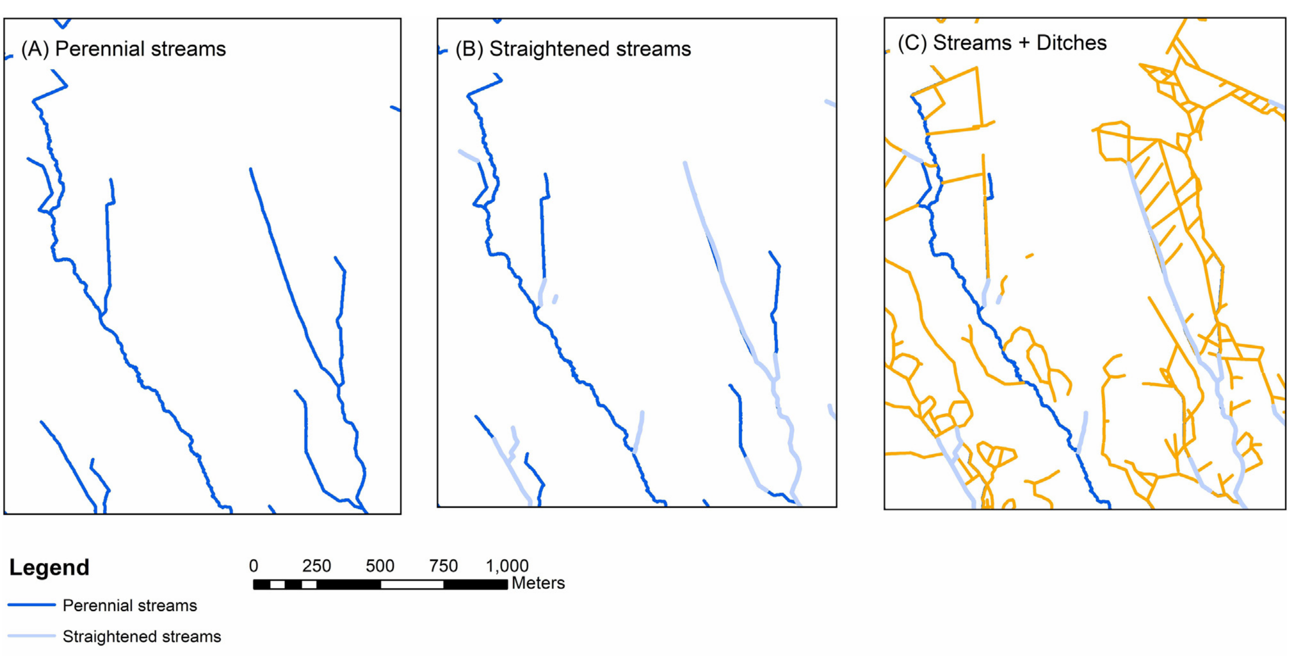

Fig. 5. Waterways within a northern Swedish forest. Perennial streams are shown in (A), perennial streams that have been modified to increase drainage capacity are highlighted in (B), and perennial streams as well as drainage ditches dug to drain wetland forests or peatlands are shown in (C).

studies, where treated and reference sites are followed through time both before and after treatment (e.g., Ide et al., 2013; Löfgren et al., 2009b; Palviainen et al., 2014; Schelker et al., 2013). The disadvantage of BACI is that the intensity of sampling frequency needed is high, and the funding cycles are often not long enough to allow researchers to address anything but short term effects on a limited number of catchments (e.g., Ide et al., 2013; Löfgren et al., 2009b; Schelker et al., 2012). An alternative approach for longer-term studies is to use space-for–time-substitution (Pickett, 1989) that assume that the reference and the treated sites of different ages are similar in all except the treatment conditions. This assumption, however, rarely holds in natural systems (Downes, 2010).

Emerging tools to adequately address the behaviour of multiple stressors are experiments (Stewart et al., 2013). Both lotic and lentic systems have been subjected to mesocosm manipulation, but lentic systems are easier to mimic by using small ponds or aquaria (Nõges et al., 2016; Stewart et al., 2013). To address interactions of stressors in lotic systems, a few studies have experimentally manipulated real streams (Fausch et al., 2010 and references therein; Rosemond et al., 2015; Zhang and Richardson, 2011), with the disadvantage that only some stressors can be addressed, because manipulating certain substances in nature can be unethical (e.g., toxins [Wallace et al., 1982]). Artificial channels are being increasingly utilized to disentangle the individual and combined effects of different stressors, and manipulation of forestry related effects have started to emerge in recent years (Louhi et al., 2017; Melody and Richardson, 2004; Turunen et al., 2018).

Across all possible designs, one challenge has been to appropriately address the reference conditions against which each treatment effect is evaluated. This is especially important in studies situated in production forests. In both Sweden and Finland, forests have been utilized for various ecosystem services for several centuries and true un-impacted forest stands are practically non-existent (Esseen et al., 1997; Östlund et al., 1997). Similarly, water courses have been affected by drainage practices or timber transport practices across Finland and Sweden for centuries (Lõhmus et al., 2015; Törnlund and Östlund, 2002). Therefore streams situated in the mature production forests of Fennoscandia can hardly be considered as unimpacted references against which to compare the various influences of forestry. Addressing stressors in aquatic systems against a set of acceptable threshold or target values, rather than against a chosen reference state, might be an alternative way forward.

While the forestry sector in Fennoscandia is getting more intensive and more mechanized it is also progressively improving methods for protecting water quality and biodiversity (Hasselquist et al., 2020; Kotilainen and Rytteri, 2011). A number of techniques are being tested and broadly applied, such as using logging residues to prevent driving damage, applying overland flow fields or breaks in cleaning before the confluence of drainage ditches and streams, applying forested riparian buffers or operation-free zones adjacent to small streams, and the use of wet area mapping to help plan management activities (e.g., Futter et al., 2016; Kuglerová et al., 2020; Laudon et al., 2011; Lidberg et al., 2019; Nieminen et al., 2017; Öhman et al., 2009). Some protection measures work better than others and more experimental research should be invested into testing their optimal applications. Many of the forestry effects are context dependent and carbon copy applications simply cannot work everywhere (Hilderbrand et al., 2005). Therefore future studies should also focus on the context dependency of forestry practices and how to best predict where in the landscape protection measures are the most effective.

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CRediT authorship contribution statement

Lenka Kuglerová: Conceptualization, Methodology, Investigation, Writing - original draft, Visualization, Funding acquisition. **Eliza Maher Hasselquist:** Methodology, Investigation, Writing - review & editing, Visualization. **Ryan Sponseller:** Investigation, Visualization, Writing - review & editing. **Timo Muotka:** Investigation, Visualization, Writing - review & editing. **Göran Hallsby:** Investigation, Visualization, Writing - review & editing, Supervision. **Hjalmar Laudon:** Investigation, Visualization, Writing - review & editing, 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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