

## NOTE

13.02.2020

Recipient(s)	Norges forskningsråd
Prepared by:	Benoît Demars, Jonas Persson, Nikolai Friberg
Copy:	n/a
Reference number:	0066/20
Project number:	17068

**Subject:** Development of a holistic conceptual framework to assess the effects of intensified forestry on aquatic ecosystem services and biodiversity in Norway

Summary

### **1. Introduction**

### **2. Literature review**

### **3. General framework**

- 31. Forestry
- 32. Forest state
- 33. Catchment descriptors
- 34. Resources and pollutants
- 35. Ecological responses

### **4. Ecological concepts**

- 41. Cross ecosystems material and energy flows and autotrophs–heterotrophs interactions
- 42. Meta-ecosystems and resilience

### **5. Recommendations for research and management (policy note)**

Supplementary information: Endnote database

## Summary

1. *General hypothesis.* Intensification of forestry as an effective climate mitigation measure affects freshwater biodiversity and ecosystem functioning through changes in natural resources for aquatic ecosystems. These changes will be highly dependent on management practices, forestry types and biogeographical context (e.g. acid sensitive areas).

2. *Review.* Best management practices were generally mitigating the impact of forestry but were not systematically implemented. The effects of (intensification of) forestry on aquatic ecology have only been investigated on short time scale during the logging period. So far most studies focused on simple indicators to estimate ecological responses. These indicators were not very sensitive, and the ecological responses were very variable between studies, suggesting a more mechanistic and context dependent approach was needed.

3. *General framework.* We developed a general causative framework amenable to Bayesian Belief Network to quantify the probabilistic effects of forestry activity on aquatic ecology through context dependent factors and the use of our general knowledge.

4. *Ecological concepts.* Two ecological concepts were found useful to structure the Bayesian Belief Network: (i) cross-ecosystem material and energy flows and autotrophs–heterotrophs interactions, (ii) meta-ecosystems and resilience across spatial and temporal scales.

5. *Recommendations.* ● The implementation of the Programme for the Endorsement of Forest Certification (PEFC 2016) should be assessed for small streams and ponds, particularly dependent on the riparian forest.

● Current bioindicators are known to have very low sensitivity. The recommendation is to use a context dependent general and causative framework using our general knowledge.

● Most experiments and studies focused on short time scale prior to and immediately after logging. More studies need to consider the whole rotation period, and quantify flows of material, energy and species across ecosystems.

## 1. Introduction

*General hypothesis:* Intensification of forestry as an effective climate mitigation measure affects freshwater biodiversity and ecosystem functioning through changes in natural resources for aquatic ecosystems and possible habitat degradation. These changes will be highly dependent on management practices, forestry types and biogeographical context (e.g. acid sensitive areas).

A recent review has questioned the use of forestry in Europe to reduce global warming due to trade-offs such as surface albedo, and concluded that forests could be adapted to climate change with neither positive nor negative climate effects (Luysaert *et al.*, 2018). Even if forestry is not as effective as initially thought for climate mitigation, the continuous drive to increasing productivity in a market economy will likely put more pressure on biodiversity and ecosystem functioning. Therefore, the initial general hypothesis of the project remains highly relevant.

In Norway, forests are dominated by Norway spruce (*Picea abies*, 40%), Scots pine (*Pinus sylvestris*, 30%) and birch (*Betula pendula* and *Betula pubescens*, all deciduous trees 25%) – Futter *et al.*, 2019. Commercial forestry focuses mostly on spruce and pine. In southern Norway, (re-)growth of the forest to canopy closure takes about 20-25 years, at which time thinning is applied (about 30% of the trees are felled). Final harvesting takes about 70-100 years. In northern Norway forest density is much lower (25-50% of national average).

Intensification of forestry as climate mitigation measure may come from afforestation on new areas, densification of existing forests, fertilisation prior to harvest, whole-tree-harvest to produce biofuels (Futter *et al.*, 2019). In Norway, intensification of forestry may have substantial effects on fine sediment supply and surface water quality, notably in acid sensitive areas, at least within the harvested area (100% clear cut, no buffer strip – Futter *et al.*, 2019, Valinia, Kaste & Wright, *in preparation*). A catchment model study in an acid sensitive area of Norway (Birkenes) showed the effects of N fertilisation 5-10 years prior to harvest were only detected in surface waters following felling. Stem harvest had a more immediate effect on NO<sub>3</sub> leaching, while whole-tree harvest had a long-term effect on acidification Valinia, Kaste & Wright, *in preparation*). These predictions may only apply to cut areas without buffer strips.

Best management practices have been effective in preventing fine sediment loads to streams (Cristan *et al.*, 2016), and this is enshrined in the Programme for the Endorsement of Forest Certification (PEFC 2016) adopted in 70% of forest cover in Norway (Futter *et al.*, 2019). Many small streams draining recently cut areas lack buffer strips and other mitigation measures (e.g. preferential flow paths) in Norway (personal observations, Jackson-Blake & Clayer, 2020). Current forestry practices form a fragmented landscape with possible remaining riparian corridors within clear cut patches. Small water bodies are likely to be the most affected during the initial stages (road construction, afforestation) and throughout the whole rotation period (thinning, fertilisation, felling and replanting). Moreover, as small water bodies have a larger proportional connection with the terrestrial ecosystem than larger systems, their structure and functioning will be highly influenced by riparian vegetation and vegetation driven processes across scales.

Here we first reviewed the literature, present a general framework, explore ecological responses to changes in material and energy flows due to (intensification of) forestry within the context of our general framework. We also tested a simple bioindicator (macroinvertebrate ASPT metric used as the national WFD compliant standard in Norway) against riparian land cover using NIVA data. Finally, we suggest some recommendations for research and management (policy note).

## **2. Literature review**

We first reviewed the world literature specific to forestry and aquatic ecology. We searched the Web of Science database in October 2018 using the following search criteria covering the period 1990-2018:

[forestry OR afforestation OR “forest harvest”]  
AND [“aquatic ecosystem” OR river OR lake OR stream OR pond]  
AND [productivity OR diversity OR subsid\* OR riparian OR food web OR phytoplankton  
OR zooplankton OR macroinvertebrate OR “fish AND boreal”]

This returned 1139 papers. From reading the abstracts, studies judged relevant were saved in an Endnote database. A few additional key studies (e.g. Friberg, 1996) were included in the database to refer to work done prior 1990 and post 2018 (e.g. Erdozain *et al.*, 2019), as well as a few reports. Some studies cited in the selected papers were also included. In total, the Endnote database hosts 261 entries. This is a sample of the literature on which to base our thinking. We briefly highlight a few recent key studies to present the state of knowledge, without pretending to cover all aspects of forestry on aquatic ecosystems.

*Streams.* Forestry best management practices (BMPs) can protect water quality when constructed correctly and in adequate numbers (Cristan *et al.*, 2016). The effects of varying forest management intensities on the integrity of 15 forest headwater streams in Canada (New Brunswick) showed streams in catchments with highest management intensity (especially road density) tended to have higher fine inorganic sediment deposition and entrainment, carbon, dissolved organic matter, and water temperature (Erdozain *et al.*, 2018). BMPs were, however, mostly effective at preventing the impairment of benthic macroinvertebrate communities (Erdozain *et al.*, 2018). The proportion of aquatic primary production contributing to the food webs decreased as forest management intensity increased, and as canopy openness decreased (Erdozain *et al.*, 2019). This suggests the quality of the consumers may decrease because primary producers are rich in essential compounds (e.g. amino acids, polyunsaturated fatty acids) not found through the detrital pathway. A global review of the literature about the direct impacts of timber harvesting on the abundance of stream biota (macroinvertebrate, fish, amphibians, birds, mammals) identified large gaps in knowledge with most of the literature focusing on macroinvertebrates in North America (Lunn, Munks & Carver, 2017). This review also highlighted the need for gaining more mechanistic understandings of biotic responses to disturbance, through experimentation and more powerful statistical approaches, considering broader spatial and temporal scales to better assess population resilience. A meta-analysis of 18 studies showed riparian forest harvest significantly increased benthic macroinvertebrate density (Richardson & Beraud, 2014). Water chemistry and other macroinvertebrate metrics showed considerable variation in their responses calling for a more context dependent site analysis (Richardson & Beraud, 2014). While logging is often associated with an increase of in-stream production, riparian deforestation in 16 North American streams reduced stream width which nullified any potential advantages of deforestation regarding abundance of fish, quality of dissolved organic matter, and pesticide degradation (Sweeney *et al.*, 2004).

*Lakes.* The pelagic food webs of Swedish humic lakes (DOC > 15 mg/L) might be resilient to a moderate form of forest clear-cutting, at least two years after tree removal, before mechanical site preparation (e.g., mounding, plowing) and when leaving buffer strips along lakes and incoming streams (Deininger *et al.*, 2018). A comparison of these results with the literature showed stronger responses in nutrients (C, N, P) than phytoplankton and even less zooplankton (Deininger *et al.*, 2018), but this may also reflect the short time scale of most experiments.

*Pollution.* Mobilisation of Hg can be increased with forestry activity (logging, site preparation, forest machinery operations) by altering redox conditions in soils and creating preferential flows. A review of the evidence showed considerable differences in treatment effects on total Hg and methylmercury (MeHg) at different sites (Eklof *et al.*, 2018). This also called for a more mechanistic and context dependent approach. Other key factors promoting bioaccumulation of MeHg in fish across both streams and lakes included suppressed individual growth of consumers, low rates of primary and secondary production and acidification (Ward, Nislow & Folt, 2010).

There is a dearth of studies considering the effect of forestry on aquatic ecosystems during the whole rotation period (70-100 years) – Warren *et al.*, 2016. Bayesian belief networks (BBNs) could be developed from a general framework based on our general ecological knowledge using evidence from expert judgment, observational and experimental data (e.g. Allan *et al.*, 2012).

### **3. General framework**

From this review and broad knowledge in aquatic ecology, we sketched a general framework (**Fig. 1**) on which cause-to-effect links can be drawn from forestry practices through to catchment descriptors, natural resources and ecological responses. The individual nodes (boxes) were characterised with categories to facilitate the building of Bayesian belief networks (e.g. Marcot *et al.*, 2001; Allan *et al.*, 2012).

Forests and forestry shape the landscape of Norway. Forestry practices can lead to large land use changes, such as afforestation (grass or bog to forest) or changes in forested species (e.g. coniferous *versus* broad leaved deciduous trees, fast growth spruce *versus* slow growth Scot pine, old growth forest to new plantation). Intensification of forestry may be realized by densification (may be in northern Norway), N fertilization 5-10 years prior to harvest, harvest of branches and tree stumps in addition to stems. New regulations to protect the environment (PEFC 2016) promote the use of best management practices such as riparian corridors and other mitigation measures against pollution (e.g. fine sediments clogging river beds, peaks of suspended solids clogging macroinvertebrate and fish gills).

### 32. Forest state

Road construction and increase in road density (also for access to cabins), notably unmetalled roads cut into the hill slope and on steep gradients may be a source of fine sediments. Forest age structure largely reflects the type of forest and rotation stage. For example, less than 2 years reflect the felling period, 2-25 years replanting to canopy cover and thinning, 25-100 years growing plantation and over 100 years old growth forest. The years can be adjusted to the location of the rotation and forest type. The composition of the forest (e.g. coniferous evergreen *versus* deciduous trees), and how it differs from the expected natural vegetation zone in Norway, will determine different patterns in material flow and energy availability to aquatic ecosystems.

### 33. Catchment descriptors

The impact of forestry on forest state will be sensed by aquatic biota through changes in resources and pollution. The impacts may be buffered or amplified by catchment characteristics, so understanding the local context is vital to make meaningful scenarios. The impacts will also largely depend on the area of forestry relative to total catchment area. While a small clear cut may damage greatly a first order headwater stream, it may not be possible to detect its downstream effect on a large river. In southern Norway, N fertilisation may affect the reproduction and survival of fish in river catchments sensitive to acidification (with low acid neutralizing capacity – ANC). N supply to surface waters may particularly affects water bodies where N is a limiting factor for ecological processes, or change the C:N:P stoichiometry of resources to consumers. High precipitation and run off with low base flow index (BFI) combined with steep slopes will lead to denser river network and higher potential for erosion along the road network. The hydraulic retention of stream can be affected by

changes in large woody debris supply and clogging by fine sediment. This in turn can affect nutrient cycling and respiration rates (self-purification power).

#### 34. Resources and pollutants

The supply of material and energy from the catchment (donor system) can drive a large part of aquatic processes (the recipient system), the more so with increased connectivity (high soil-water interface relative to volume or water flows of the aquatic ecosystem). Some reciprocal exchanges between aquatic ecosystems and riparian corridor can be qualitatively important (e.g.  $\Omega 3$  fatty acids, bioaccumulated pollutant). A short list of potential key resources and toxic elements are presented in **Fig. 1**. Ecological responses to these proximate causes are difficult to predict because of interactions and time lags. There can be interactions among the drivers (between resources and toxicants) as well as part of the ecological response (trophic interactions, change in community structure) which are time dependent due to variability in life history (e.g. body size).

The most obvious resource is **light** availability. The type of forestry and forest composition will alter light availability differently within a year as well as over the whole rotation period. Larch and broad-leaved deciduous trees will provide a window of opportunity for aquatic primary production in spring and autumn, that will be absent in mature spruce and pine tree plantation forest. The amount of light reaching the aquatic ecosystem will depend on tree density and proportion of canopy closure, as well as the size of the aquatic systems and aspect (slope orientation, e.g. facing south or north). Light interacts positively with **temperature**. The fate of the organic matter released by the forest into the aquatic ecosystem will depend on its nature (e.g. C:N:P stoichiometry, phenolics) and structure: while bacteria can equally use **dissolved (DOM) and particulate organic matter (POM)**, fungi will thrive on stable substrate such as **coarse particulate organic matter (CPOM, e.g. leaf litter) and large woody debris**. Large woody debris supply can also generate habitat heterogeneity, notably woody debris dams creating pools. In Norway the concentrations of **soluble reactive phosphorus (SRP)** and **dissolved inorganic nitrogen (DIN=NO<sub>3</sub>+NH<sub>4</sub>)** can be extremely low (near or below detection limits), and thus are likely to limit both photosynthesis and respiration. Internal cycling of N and P are likely to be high relative to fluxes going through the aquatic ecosystems, and the use of organic P and N may be substantial. Thus, small N leaching from forest fertilisation (possibly delayed until tree felling) could have profound effects on ecological processes and trophic dynamic. In the absence of buffer strips and blockage of preferential flow paths, aquatic ecosystems strongly connected to the land will be



severely perturbed. **Fine sediments** from tracks and road erosion, **methyl mercury** (MeHg) from soil water solution with low redox potential (e.g. resulting from soil compaction), and **soluble aluminium** (Al) in acid sensitive catchment are known sources of pollution for aquatic ecosystems. Fine sediment may interact negatively with light availability, mercury bioaccumulation through the trophic chain may also be transferred through to riparian habitats. Although intensification of forestry may lead to an increase in DOM (notably fulvic and humic acids) and corresponding decrease in pH, DOM forms complexes with MeHg and Al in surface waters, limiting their availability and toxicity.

### 35. Ecological responses

There are many individual chemical and biological indicators available in Norway for rivers and lakes (Veileder 02:2018, Klassifisering av miljøtilstand i vann, [www.vannportalen.no](http://www.vannportalen.no)), but it is not known how these indicators respond to multiple stressors, as generated by the effects of (intensification of) forestry on aquatic ecosystems. Previous studies have used macroinvertebrate general indicators of water quality. Here we tested the effect of riparian vegetation cover (scale 1-5), visually assessed when undertaking field sampling, on the average macroinvertebrate average score per taxa (ASPT, range 1-10), an indicator initially designed to characterise the degree of organic pollution from sewage effluents but also used as a general indicator of water and habitat quality. Here we used 2067 macroinvertebrate samples collected in Norway by NIVA (including polluted sites). The ASPT was somewhat lower (9-16%) in river sections with the highest riparian cover (overall regression test:  $p=1 \times 10^{-7}$ ), which is relatively surprising as the assumption based on the literature would be that riparian cover would influence macroinvertebrate community structure positively. The coefficient of variation (standard deviation / average) of ASPT within riparian cover classes were large however (21-36%). Overall the ASPT did not respond much to riparian vegetation cover (**Fig. 2**), indicating that other factors were responsible for the ASPT value and hence the macroinvertebrate community structure. In addition, neither ASPT nor the sampling were designed to test for the effect of riparian vegetation (or forestry), so the methodology is not likely to be very sensitive to changes in riparian vegetation even without other stressors present.

Macroinvertebrate bioindicators may be used as an end point in Bayesian Belief Network models, but this requires testing within a range of environmental settings relevant to forestry

activities and its sensitivity is likely to remain low (Allan *et al.*, 2012). Alternatively, we can use our general ecological knowledge and past experiments manipulating multiple pressures to investigate the cascading probabilities of impacts from forestry activities to ecological responses. We expand on this approach using ecological concepts.

#### **4. Ecological concepts**

##### **41. Cross-ecosystems material and energy flows and autotrophs–heterotrophs interactions**

The interactions between autotrophs and bacteria are difficult to study (Amin *et al.*, 2015), because primary producers, decomposers and organic matter (allochthonous and autochthonous) are intricately connected both in benthic biofilms (Kamjunke, Herzsprung & Neu, 2015; Battin *et al.*, 2016) and pelagic aggregates (Grossart, 2010; Gärdes *et al.*, 2011). In theory, bacteria and autotrophs could compete for limiting nutrients (Currie & Kalff, 1984), notably when bacteria have lower C:nutrient biomass ratios than autotrophs (Daufresne & Loreau, 2001). However, bacteria can have high C:nutrient ratios similar to autotrophs (Cotner *et al.*, 2010) supporting the co-existence of autotrophs and heterotrophs (Daufresne & Loreau, 2001). Correlative analyses in streams seem to support the idea that positive interactions between autotrophs and bacteria increase with nutrient (N, P) limitations (Carr, Morin & Chambers, 2005; Scott *et al.*, 2008). Whole-stream metabolism in open streams under low flow conditions (hydrologically disconnected from catchment soils supplying allochthonous organic carbon) showed ecosystem respiration to be tightly correlated to gross primary production (Demars *et al.*, 2016), suggesting a strong indirect mutualistic interaction between autotrophs and bacteria (Demars *et al.*, 2011), with autotrophs providing detritus C, N, P to bacteria and bacteria regenerating N and P by mineralisation (Cotner *et al.*, 2010), in agreement with theory (Daufresne & Loreau, 2001). Under changing light availability (canopy cover in deciduous forest, high loads of fine sediments) and changing carbon supply (dissolved organic matter during high flow conditions and leaf fall in the autumn) the mutualistic interaction weakens with heterotrophy dominating total metabolism (Roberts, Mulholland & Hill, 2007; Demars, Friberg & Thornton, 2020) and the role of nutrients becomes more difficult to predict (Roberts & Mulholland, 2007).

In forested streams, bacterial respiration and production will depend on both primary production and allochthonous supplies of carbon (dissolved and particulate organic matter). Fungal respiration and production will be more dependent on coarse particulate organic matter (leaf litter) and large woody debris. Primary production, unlike fungal and bacterial production, is rich in essential biomolecules (e.g. amino acids, polyunsaturated fatty acids) preferentially transferred along the food chain (macroinvertebrate, fish) and across systems into the riparian corridor *via* predation and insect emergence.

**Cross-ecosystems concept (Fig. 3) – example of application:** Decrease in light availability and increase in supply of organic matter and fine sediments (i) switches autotrophs–heterotrophs interaction from mutualism to competition, (ii) alters nutrient cycling, (iii) increases biotic CO<sub>2</sub> emissions (sensitive to temperature), promotes detrital pathways in food-webs, (iv) decreases food quality to consumers (higher C:N:P stoichiometric ratios, less Ω<sub>3</sub> fatty acids and more Hg), and (v) decrease secondary production.

*Further reading:* Erdozain *et al.*, 2019; Gounand *et al.*, 2018b

#### 42. Meta-ecosystems and resilience

Forested landscape is now a dynamic mosaic of forest patches at different stages of development. Forest patches may also be embedded in another land cover type (agriculture, wetlands) or *vice versa*. The resulting landscape is fragmented. This has important consequences for the dynamic of the impacts of forestry on aquatic ecosystem over time. Some impacts may be short term (e.g. nutrient peak following logging, Valinia, Kaste &

Wright, *in preparation*) and other longer term (e.g. canopy closure control on light availability, Warren *et al.*, 2016). There may also be a long spell of lessened impacts on (mostly) receiving aquatic ecosystems. During this time window, there may be a recovery of former populations and associated ecosystem functioning, that is aquatic ecosystems may be somewhat resilient to forestry impact. Over one rotation period (70-100 years), less obvious long-term stress (e.g. diffuse pollution) may have larger effects on aquatic ecosystems than short term disturbances (e.g. light effect). This calls for the cross-ecosystem concept to be assessed at multiple spatial and temporal scales.

The concept of reciprocal material and energy exchanges between freshwater and terrestrial ecosystems is now well established (Nakano & Murakami, 2001; Baxter, Fausch & Saunders, 2005; Gounand *et al.*, 2018b). In terms of carbon flows, aquatic ecosystems are largely receiving systems (Gounand *et al.*, 2018b), nonetheless aquatic ecosystems can also be donor of essential biomolecules (e.g. polyunsaturated fatty acids; Gladyshev, Arts & Sushchik, 2009) and limiting nutrients (including anadromous fish; Naiman *et al.*, 2002).

Lakes, streams draining forested catchments at different stage of development and riparian corridors form a complex network of exchange of material and energy flows between ecosystems, we call it meta-ecosystem (Loreau, Mouquet & Holt, 2003). The diversity and dynamics of population form a meta-community (Mouquet & Loreau, 2003). Meta-community and meta-ecosystem may be more resilient to large scale environmental changes (e.g. climate change, atmospheric deposition) if species and their associated traits have time to move (disperse and establish) through this dynamic landscape (Peterson *et al.* 1998). The systems most at risk of species loss are those most connected to the forest ecosystem (e.g. pools, streams) and most isolated (relative to species dispersal ability) from each-others. In streams it can take 25 years for the system to recover following the felling of riparian trees, but this will be context dependent.

*Further reading:* Truchy *et al.*, 2015; Gounand *et al.*, 2018a

## **5. Recommendations for research and management (policy note)**

- The Programme for the Endorsement of Forest Certification (PEFC 2016) present best management practices, including the presence of a riparian buffer zone, generally resulting in mitigated impacts of even intensified forestry. However, the implementation of the guidance towards riparian zone is not obvious in the current Norwegian landscape, at least for small streams and ponds which tend to be the most connected to the riparian zone and the most at risk of forestry.
- Water framework directive (WFD) indicators have not been designed to test the effect of (intensification of) forestry and this type of indicators are known to have very low sensitivity. The recommendation is to use a context dependent general framework linking causes to effects using our general knowledge.
- Most experiments and studies focused on short time scale prior to and immediately after logging, ground preparation for replanting. More studies need to consider the whole rotation period, and quantify flows of material, energy and species across ecosystems (such as streams, riparian zone, pools) is lacking in Norway.

### **Acknowledgments**

This work was supported by the Research Council of Norway through the SURFER project (NFR project number 267481) and the Norwegian Institute for Water Research NIVA.

## References

- Allan J.D., Yuan L.L., Black P., Stockton T., Davies P.E., Magierowski R.H. & Read S.M. (2012) Investigating the relationships between environmental stressors and stream condition using Bayesian belief networks. *Freshwater Biology*, **57**, 58-73.
- Amin S.A., Hmelo L.R., Van Tol H.M., Durham B.P., Carlson L.T., Heal K.R., Morales R.L., Berthiaume C.T., Parker M.S., Djunaedi B., Ingalls A.E., Parsek M.R., Moran M.A. & Armbrust E.V. (2015) Interaction and signalling between a cosmopolitan phytoplankton and associated bacteria. *Nature*, **522**, 98-101.
- Battin T.J., Besemer K., Bengtsson M.M., Romani A.M. & Packmann A.I. (2016) The ecology and biogeochemistry of stream biofilms. *Nature Reviews Microbiology*, **14**, 251-263.
- Baxter C.V., Fausch K.D. & Saunders W.C. (2005) Tangled webs: reciprocal flows of invertebrate prey link streams and riparian zones. *Freshwater Biology*, **50**, 201-220.
- Carr G.M., Morin A. & Chambers P.A. (2005) Bacteria and algae in stream periphyton along a nutrient gradient. *Freshwater Biology*, **50**, 1337-1350.
- Cotner J.B., Hall E.K., Scott J.T. & Haldal M. (2010) Freshwater bacteria are stoichiometrically flexible with a nutrient composition similar to seston. *Frontiers in Microbiology*, **1**, 132.
- Cristan R., Aust W.M., Bolding M.C., Barrett S.M., Munsell J.F. & Schilling E. (2016) Effectiveness of forestry best management practices in the United States: Literature review. *Forest Ecology and Management*, **360**, 133-151.
- Currie D.J. & Kalff J. (1984) Can bacteria outcompete phytoplankton for phosphorus? A chemostat test. *Microbial Ecology*, **10**, 205-216.
- Daufresne T. & Loreau M. (2001) Ecological stoichiometry, primary producer-decomposer interactions, and ecosystem persistence. *Ecology*, **82**, 3069-3082.
- Deininger A., Jonsson A., Karlsson J. & Bergström A.K. (2018) Pelagic food webs of humic lakes show low short-term response to forest harvesting. *Ecological Applications*, doi.org/10.1002/eap.1813.
- Demars B.O.L., Friberg N. & Thornton B. (2020) Pulse of dissolved organic matter alters reciprocal carbon subsidies between autotrophs and bacteria in stream food webs. *Ecological Monographs*, **90**, e1399.
- Demars B.O.L., Gislason G.M., Olafsson J.S., Manson J.R., Friberg N., Hood J.M., Thompson J.J.D. & Freitag T.E. (2016) Impact of warming on CO<sub>2</sub> emissions from streams countered by aquatic photosynthesis. *Nature Geoscience*, **9**, 758-761.

- Demars B.O.L., Manson J.R., Olafsson J.S., Gislason G.M. & Friberg N. (2011) Stream hydraulics and temperature determine the metabolism of geothermal Icelandic streams. *Knowledge and Management of Aquatic Ecosystems*, **402**, 05.
- Eklof K., Bishop K., Bertilsson S., Bjorn E., Buck M., Skyllberg U., Osman O.A., Kronberg R.M. & Bravo A.G. (2018) Formation of mercury methylation hotspots as a consequence of forestry operations. *Science of the Total Environment*, **613**, 1069-1078.
- Erdozain M., Kidd K., Kreutzweiser D. & Sibley P. (2018) Linking stream ecosystem integrity to catchment and reach conditions in an intensively managed forest landscape. *Ecosphere*, **9**, e02278.
- Erdozain M., Kidd K., Kreutzweiser D. & Sibley P. (2019) Increased reliance of stream macroinvertebrates on terrestrial food sources linked to forest management intensity. *Ecological Applications*, **29**, e01889.
- Friberg N. (1996) *Biological structure of forest streams and effects of afforestation*. PhD, National Environmental Research Institute, Silkeborg.
- Futter M., Clarke N., Kaste Ø. & Valinia S. (2019) The potential effects on water quality of intensified forest management for climate mitigation in Norway. Vol. 7363-2019. NIVA, Oslo.
- Gladyshev M., Arts M. & Sushchik N.I. (2009) Preliminary estimates of the export of omega-3 highly unsaturated fatty acids (EPA+ DHA) from aquatic to terrestrial ecosystems. In: *Lipids in aquatic ecosystems*. (Eds M. Kainz & M.T. Brett & M.T. Arts), pp. 179-210. Springer, New York, NY.
- Gounand I., Harvey E., Little C.J. & Altermatt F. (2018a) Meta-ecosystems 2.0: Rooting the theory into the field. *Trends in Ecology & Evolution*, **33**, 36-46.
- Gounand I., Little C.J., Harvey E. & Altermatt F. (2018b) Cross-ecosystem carbon flows connecting ecosystems worldwide. *Nature communications*, **9**, 4825.
- Grossart H.P. (2010) Ecological consequences of bacterioplankton lifestyles: changes in concepts are needed. *Environmental Microbiology Reports*, **2**, 706-714.
- Gärdes A., Iversen M.H., Grossart H.P., Passow U. & Ullrich M.S. (2011) Diatom-associated bacteria are required for aggregation of *Thalassiosira weissflogii*. *Isme Journal*, **5**, 436-445.
- Jackson-Blake L. & Clayer F. (2020) Assessment of risks to drinking water provision in Glitrevann from forest fertilization and harvesting. Vol. 7458. Norwegian Institute for Water Research, Oslo.

- Kamjunke N., Herzsprung P. & Neu T.R. (2015) Quality of dissolved organic matter affects planktonic but not biofilm bacterial production in streams. *Science of the Total Environment*, **506**, 353-360.
- Loreau M., Mouquet N. & Holt R.D. (2003) Meta-ecosystems: a theoretical framework for a spatial ecosystem ecology. *Ecology Letters*, **6**, 673-679.
- Lunn T., Munks S. & Carver S. (2017) The impacts of timber harvesting on stream biota - An expanding field of heterogeneity. *Biological Conservation*, **213**, 154-166.
- Luyssaert S., Marie G., Valade A., Chen Y.Y., Djomo S.N., Ryder J., Otto J., Naudts K., Lanso A.S., Ghattas J. & Mcgrath M.J. (2018) Trade-offs in using European forests to meet climate objectives. *Nature*, **562**, 259-+.
- Marcot B.G., Holthausen R.S., Raphael M.G., Rowland M.M. & Wisdom M.J. (2001) Using Bayesian belief networks to evaluate fish and wildlife population viability under land management alternatives from an environmental impact statement. *Forest Ecology and Management*, **153**, 29-42.
- Mouquet N. & Loreau M. (2003) Community patterns in source-sink metacommunities. *American Naturalist*, **162**, 544-557.
- Naiman R.J., Bilby R.E., Schindler D.E. & Helfield J.M. (2002) Pacific salmon, nutrients, and the dynamics of freshwater and riparian ecosystems. *Ecosystems*, **5**, 399-417.
- Nakano S. & Murakami M. (2001) Reciprocal subsidies: Dynamic interdependence between terrestrial and aquatic food webs. *Proceedings of the National Academy of Sciences of the United States of America*, **98**, 166-170.
- Richardson J.S. & Beraud S. (2014) Effects of riparian forest harvest on streams: a meta-analysis. *Journal of Applied Ecology*, **51**, 1712-1721.
- Roberts B.J. & Mulholland P.J. (2007) In-stream biotic control on nutrient biogeochemistry in a forested stream, West Fork of Walker Branch. *Journal of Geophysical Research-Biogeosciences*, **112**, G04002.
- Roberts B.J., Mulholland P.J. & Hill W.R. (2007) Multiple scales of temporal variability in ecosystem metabolism rates: Results from 2 years of continuous monitoring in a forested headwater stream. *Ecosystems*, **10**, 588-606.
- Scott J.T., Back J.A., Taylo J.M. & King R.S. (2008) Does nutrient enrichment decouple algal-bacterial production in periphyton? *Journal of the North American Benthological Society*, **27**, 332-344.
- Sweeney B.W., Bott T.L., Jackson J.K., Kaplan L.A., Newbold J.D., Standley L.J., Hession W.C. & Horwitz R.J. (2004) Riparian deforestation, stream narrowing, and loss of



stream ecosystem services. *Proceedings of the National Academy of Sciences of the United States of America*, **101**, 14132-14137.

Truchy A., Angeler D.G., Sponseller R.A., Johnson R.K. & Mckie B.G. (2015) Linking biodiversity, ecosystem functioning and services, and ecological resilience: towards an integrative framework for improved management. In: *Ecosystem Services: From Biodiversity to Society, Pt I*. (Eds G. Woodward & D.A. Bohan), pp. 55-96. *Advances in Ecological Research*.

Ward D.M., Nislow K.H. & Folt C.L. (2010) Bioaccumulation syndrome: identifying factors that make some stream food webs prone to elevated mercury bioaccumulation. *Annals of the New York Academy of Sciences*, **1195**, 62-83.

Warren D.R., Keeton W.S., Kiffney P.M., Kaylor M.J., Bechtold H.A. & Magee J. (2016) Changing forests-changing streams: riparian forest stand development and ecosystem function in temperate headwaters. *Ecosphere*, **7**, e01435.

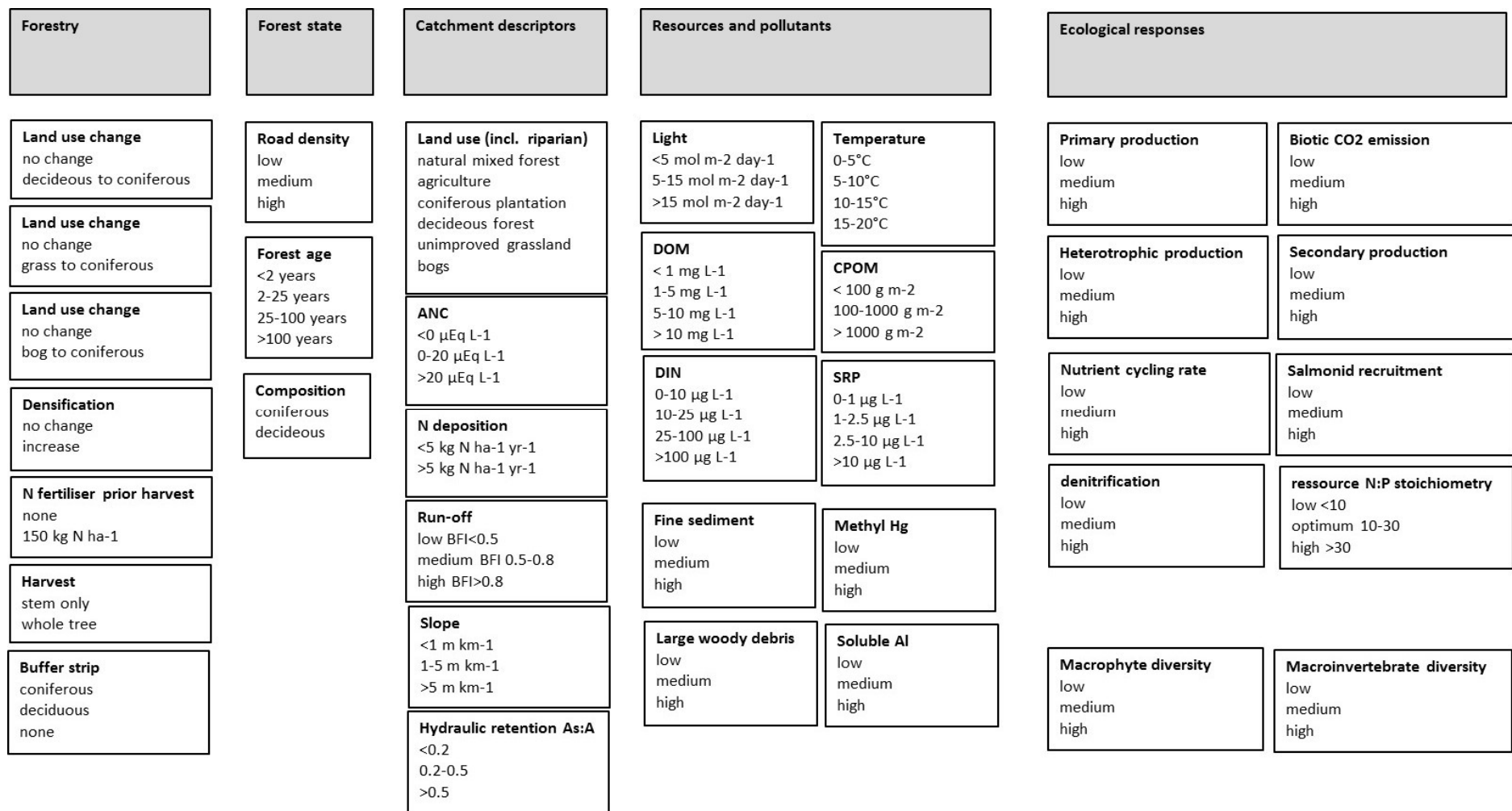
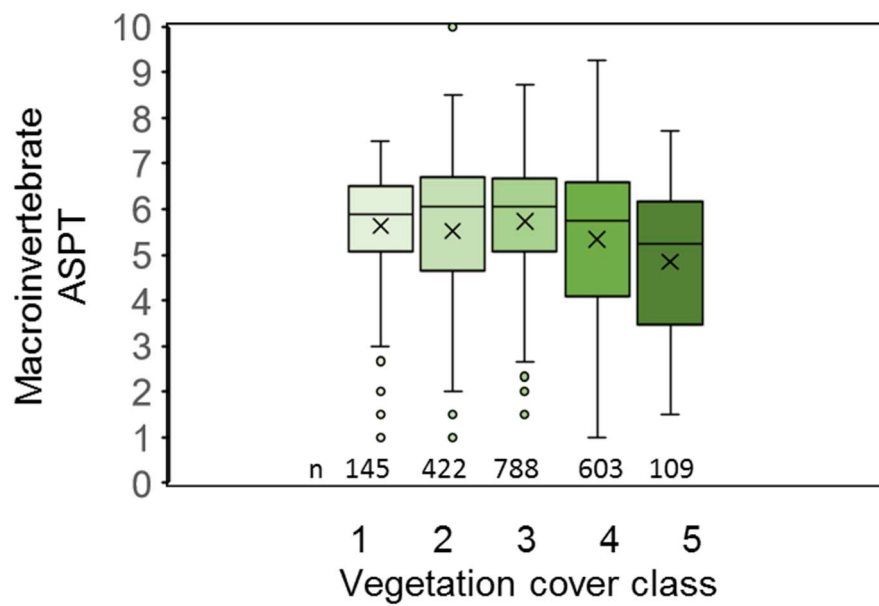
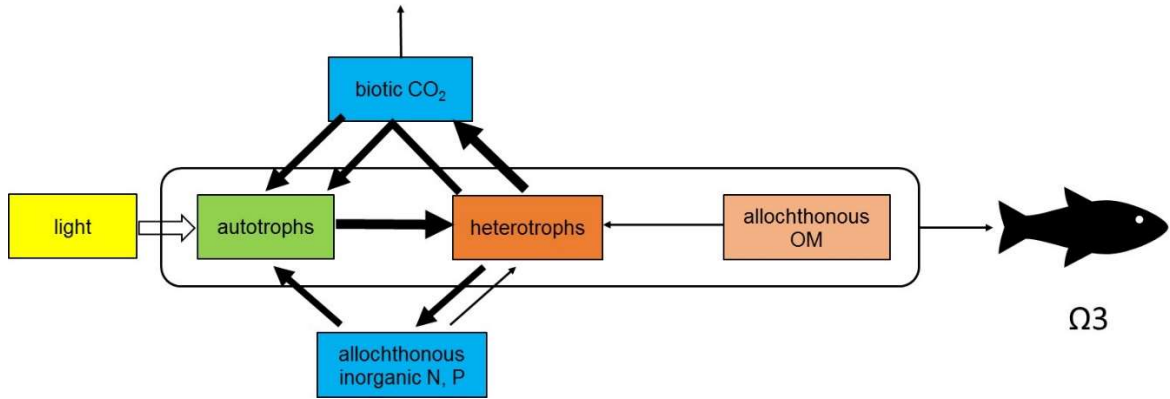


Fig. 1. Framework linking forestry practices to aquatic ecology.

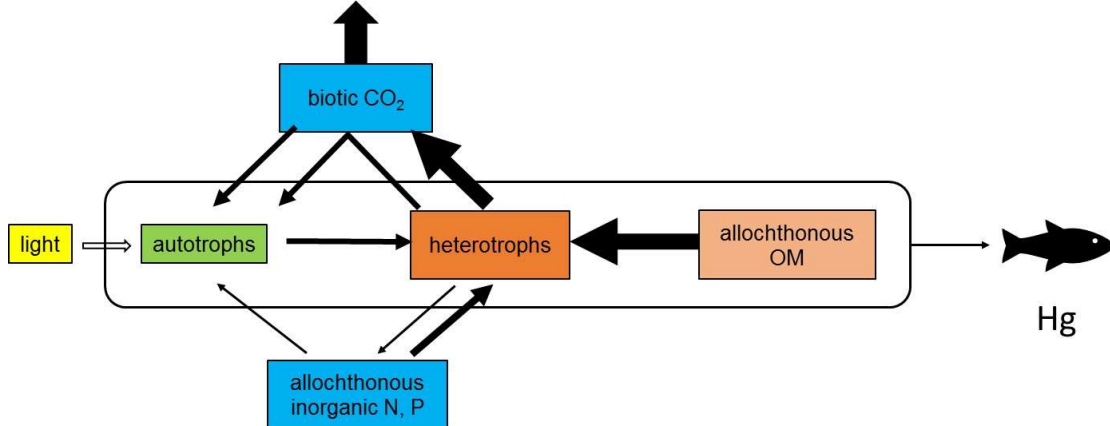


**Fig. 2.** Macro-invertebrate average score per taxa (ASPT) within riparian vegetation cover classes (1-5, low to high). n = number of samples. Box indicate 25-75<sup>th</sup> centiles, line is median, and cross is average.

(A) High light availability, low allochthonous organic matter (OM) supply



(B) Low light availability, high allochthonous organic matter (OM) supply



**Fig. 3.** Impact of forestry on aquatic ecosystem functioning (nutrient cycling and food webs) in nutrient (N, P) poor ecosystems. **A.** In the absence of canopy cover, light availability and low supply of organic matter drive a strong mutualistic interaction between autotrophs and bacteria. **B.** In the presence of thick canopy cover, lack of light availability and high supply of organic matter (wood, leaf litter, dissolved organic matter) heterotrophs outcompete autotrophs. This shift in the base of the food web can reduce the availability of unsaturated fatty acids (e.g. Ω3) and other essential compounds (e.g. amino acids) to the consumers, and may increase the bioaccumulation of mercury (Hg) through the detrital pathway into the consumers.