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Spatial variation in benthic macroinvertebrate community structures in tributaries of Verdal river: Effects of biotic and abiotic environmental factors and restoration measures

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Environment and Natural Resources

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Abstract

Rivers and streams are one of the most degraded ecosystems in the world, including Norway. Human activities have long impacted rivers and streams, both directly, and indirectly, and this ultimately affects the streams health. Benthic macroinvertebrates and fish are the most common biological variables to use as bioindicators for ecological condition. Macroinvertebrates as a group play an important role in stream ecosystems, they can affect important stream processes, such as nutrient cycling, primary production, decomposition, and translocation of materials. In addition, they are important part of stream food webs. Verdal watershed located in Trøndelag county in Norway is considered an important Atlantic salmon (*Salmo salar*) and anadromous brown trout (*Salmo trutta*) river. Runoff from agriculture and migration barriers are one of the main threats to the biodiversity of Verdal river. To prevent further degradation of the Verdal watershed and further loss of biodiversity, restoration measures, mainly connectivity and addition of spawning gravel, were conducted in 2016 and 2017 in several of the streams to improve area and productivity of fish. The aim of this study is to examine the responses of benthic macroinvertebrates to restoration measures conducted in the tributary streams of Verdalselva, and shed light on the research questions: 1) Did the in-stream measures affect the macroinvertebrate community, and what determines the variation in macroinvertebrate assemblages? 2) Does the reintroduced presence of salmonids affect macroinvertebrate community structure? And 3) Which variables determines the variation in the organic stressor metric ASPT index scores? The collection of macroinvertebrate data was obtained by kick-sampling method in 12 tributaries of Verdal river. Analysis of the data was undertaken with ordination analysis. The results indicate that the restoration measures have no effect on macroinvertebrate assemblages, as to yet. It is likely that the macroinvertebrates need longer time to respond. Allochthonous input of the streams appears to have a significant effect on macroinvertebrates, shifting towards a pollution-tolerant community structure. Density of salmonids had a significant effect on the macroinvertebrate community structure. There was a small, but insignificant, difference in diversity between the upper and lower reaches of the streams. ASPT index scores were mostly determined by distance to the fjord, distance to the main river and fish densities. However, the positive effect fish densities have on the ASPT index is most likely due to covariance where both groups respond similarly to favourable conditions. In conclusion, long-time monitoring of the benthic macroinvertebrate community is needed to detect long-time responses to the recently conducted habitat measures.

Sammendrag

Elver og bekker er en av det mest degraderte økosystemer i verden, Norge inkludert. Menneskelig aktivitet har lenge påvirket elver og bekker både ved direkte og indirekte påvirkning. Bunndyr og fisk er det mest brukte biologiske variablene for å evaluere den økologiske statusen til vannforekomster. Bunndyr som gruppe spiller en viktig rolle i bekkers økosystem. De kan påvirke viktige bekkeprosesser, som næring syklusen, primærproduksjonen, nedbrytning og translokasjon av materialer. I tillegg er bunndyr en viktig del av bekkers næringskjede. Verdalsvassdraget i Trøndelag fylke i Norge er ansett som et nasjonalt viktig laksevassdrag i Norge. Avrenning fra jordbruket og vandringshindre er mulige trusler for biodiversiteten i Verdalselva. For å forhindre videre degradering av Verdalsvassdraget og øke fiskeproduksjonen har det blitt gjort flere tiltak i sidebekkene, for det meste fokus på konnektivet og utlegg av gytegrus. Målet med denne studien er å undersøke hvordan disse tiltakene har påvirket bunndyrsamfunnet i bekkene, og svare på spørsmålene 1) Hadde bekketiltakene noen påvirkning på bunndyrsamfunnet, og hvilke variabler er med på å bestemme variasjonen i bunndyrsammensetningen? 2) Har den gjeninnførte fiskeproduksjonen noe å si for sammensetningen på bunndyrene? Og til slutt 3) Hvilke variabler bestemmer variasjonen i ASPT indeks verdiene? Det ble tatt to bunndyrprøver ved sparkeprøvemetoden i hver stasjon i 12 sidebekker til Verdalselva. Ordinasjonsanalyser ble gjennomført for å analysere dataene. Resultatene indikerer at restaureringstiltakene utført i sidebekkene til Verdalselva hadde lite effekt på bunndyrsamfunnet. Bunndyrprøvene ble tatt kort tid etter tiltakene, og det er derfor sannsynlig at bunndyrsamfunnet trenger lengre tid på å vise en effekt av tiltakene. Den alloktone tilførselen til bekken viste å ha størst effekt på strukturen i bunndyrsamfunnet, sammen med fisketetthet. Det viste seg å være en ikke signifikant forskjell i bunndyrdiversitet mellom øvre liggende stasjoner sammenlignet med lavere liggende stasjoner. Kun en av 12 bekker viste seg å ha minimum god økologisk status. Variasjon i ASPT indeksen var best forklart av avstand til fjorden, avstand til hovedelv og fisketetthet. Den positive effekten fisketetthet har på ASPT indeksen skyldes mest sannsynlig samvariasjon der begge grupper responderer likt på gunstige miljøforhold. Langtids overvåkning av bunndyrsamfunnet er nødvendig for å se hvordan bunndyrene responderer over tid på nylig utført habitat tiltak.

Contents

1 Introduction

Ecosystems, habitats and species experience an increased pressure from human activities around the world, including Norway. Habitat loss is the main threat to biodiversity today, and rivers and streams are some of the most degraded habitats in the world (Jourdan et al., 2018; Miller et al., 2010; Stoll et al., 2016). Human activities have long impacted streams and rivers, both directly by altering the hydromorphology of the streams (Stoll et al., 2016) making them homogeneous (Nakano & Nakamura, 2006), pollution spills, introduced species, and indirectly by altering the catchment use to agriculture or urbanization (Schneider & Petrin, 2017). These alterations impact the overall health of the streams and the biodiversity in and around the streams (Miller et al., 2010).

Norway is committed to stop the degradation of river systems by EU`s Water Framework Directive (WFD) (Vannportalen, 2019). Norway has implemented EU`s Water Framework directive into national and regional water management plans. According to the Water Regulations every waterbody should achieve minimum good ecological condition within 2020 (Anonym, 2015). The dynamic nature of streams poses a challenge in measuring the ecological condition. Measurements of chemical variables are insufficient in assessing streams health, because they might miss recent pollution events. Biological indicators on the other hand, might not escape these events, and therefore are excellent to use for assessing stream ecological condition. Benthic macroinvertebrates and fish are the most common biological variables to use as bioindicators for ecological condition (Ruaro et al., 2015). However, fish have shown avoidance behaviour to pollution, and will possibly not give a correct picture of the ecological condition. Macroinvertebrates consist of insect larvae, leeches, snails, and other invertebrates which lives at the bottom of streams and rivers (Våge, 2018). The abundances and inability to avoid changes to stream water quality makes them excellent bioindicators (Feeley et al., 2012). Additionally, the cost-effective sampling method makes it easy to get a representative picture of the community structure (Feeley et al., 2012). Measurements of multiple bioindicators might give a more accurate representation of the ecological condition (Larsen et al., 2012).

Macroinvertebrates as a group play an important role in stream ecosystems, they can affect important stream processes, such as nutrient cycling, primary production, decomposition, and translocation of materials (Wallace & Webster, 1996). Benthic invertebrates have evolved due to the heterogeneous physical environment of streams and are separated in to five functional groups based on method of finding and obtaining food; scrapers, shredders, gatherers, filterers and predators (Wallace & Webster, 1996). Aquatic macroinvertebrates in their juvenile stages are important prey for fish, other larger aquatic insects, birds and amphibians, and therefore are important part of stream ecosystems (Pope et al., 2009).

Verdal watershed located in Trøndelag county in Norway is considered as an important Atlantic salmon (*Salmo salar*) and anadromous brown trout (from now on called seatrout) (*Salmo trutta*) river in Norway. However, the salmonid populations have decreased substantially since the 1970s. In 1985, 14 of 26 Verdal river tributaries were assessed as heavily polluted by agricultural and urban runoff, and about half of the productive habitat for sea trout had non-living conditions for fish (Kristiansen, 2007). Runoff from agriculture was assumed the main reason for the disappearance of sea trout. In 1992, the runoff from agriculture was reduced considerably, and a new study of the tributaries showed that there was a positive development in sea trout production. However, the streams were still heavily polluted by agricultural runoff. A study conducted in 2005 by Kristiansen (2007) showed that there was fish in 23 of 29 streams examined, despite most of the streams having poor ecological conditions. Only two of the streams achieved good ecological status in 2016 (Vårhus, 2016). According to Hol (2018) there have been a reduction of 35 % of available habitat, and an 80 % reduction in fish production. Eight of 34 streams were considered empty of fish, and of 25 examined streams, only two achieved minimum good ecological condition (Hol, 2018).

To prevent further degradation of the Verdal watershed and further loss of biodiversity, restoration measures were conducted in 2016 and 2017 in several of the streams to improve fish productivity. The aim of these restoration efforts was to increase the available area for salmonid fish production by reducing migration barriers such as culverts under roads and railways, and to increase spawning opportunities by improving the bottom substrate with addition of spawning gravel (see Table 1). Restoration ecology is an important tool in river management to prevent further loss of biodiversity and increase the overall health of the waterbody (Bernhardt et al., 2005). In-stream measures are the most common restoration efforts in river management. These measures aim to increase diversity and abundance of aquatic organisms by increasing habitat heterogeneity, complexity and increasing food availability. In-stream restoration measures are often performed at reach scale of the stream by adding boulders, gravel or changes in the water course. These types of restoration measures are based on the “field of dreams” theory (Palmer et al., 1997), which assumes that local species diversity is controlled by the physical habitat heterogeneity. Therefore, by improving the habitat, the species diversity will increase (Miller et al., 2010). Increased heterogeneity can allow more species to coexist, by increasing range of niches and reduce competition for resources. Complex habitats might also provide increased refugia for predation, flood risk and increased availability of food (Barnes et al., 2013). There is still an amount of uncertainty of macroinvertebrate responses to restoration measures (Barnes et al., 2013; Miller et al., 2010). A few studies have found that restoration increased the diversity and/ or the abundance of macroinvertebrates, however, there have been more reports of negative responses or no response at all (Jahnig et al., 2010). The lack of responses by macroinvertebrates to restoration measures can reflect insufficient restoration intensity, inappropriate design or method (Li et al., 2018), limited scale or a lack of adjacent source populations for colonisation (Brederveld et al., 2011).

Several of the tributary streams of Verdal watershed, which previously had reaches that were considered empty of fish, have now regained populations of salmonids. The presence and absence of fish predation on macroinvertebrate community in running waters are poorly understood (Williams et al., 2003). Fish predation might determine invertebrate community structure, however Allan (1982) suggest that macroinvertebrates are adapted to the presence of fish, and that changes in fish densities does not affect invertebrate community structure. Williams et al. (2003) suggested that because smaller streams are less stable, disturbance play a greater role in determining community structure, than biotic interactions.

1.1 Objectives

The aim of this study is to examine the responses of benthic macroinvertebrates to restoration measures conducted in the tributary streams of Verdalselva, mostly increased connectivity and supply of spawning gravel. In addition, the streams ecological health will be assessed based on ASPT index scores. The goal is to answer the questions:

1) Did the instream restoration measures affect the macroinvertebrate community structure, and which environmental variables determine the variation in the invertebrate assemblages?
   a) The timescale might be too short, and the restoration intensity might be too low for detection of macroinvertebrate responses, therefore it is possible that the restoration measures have little effect on the macroinvertebrate assemblages, as to yet.
   b) Habitat heterogeneity support more species to co-exist and therefore, the variation in macroinvertebrate assemblages may be determined by number of woody debris, and type of substrate.
2) Does the reintroduced presence of salmonids, due to the restoration measures undertaken in some tributaries of Verdal river affect benthic macroinvertebrate community structure?
   a) The foraging pressure from salmonids may change the community structure of the macroinvertebrates to predation tolerant species.

3) Which variables determines the variation in ASPT index scores?
    a) Streams closer to the fjord are closer to Verdal city and might be more affected by human activities than streams further up the watershed. Thus, increased distance to the fjord may yield higher ASPT index scores.

2 Materials and method

2.1 Study area

This study examines 12 tributary streams of Verdal river located in Verdal municipality in Trøndelag county (Figure 1). The river Verdalselva is a designated national wild salmon river in Norway, and the Verdal watershed is permanently protected against hydropower development (Anonym, 2004). The source and catchment reach the border of Sweden, and the outlet runs out into Trondheimsfjorden. The catchment is 1471 km^2 and most of the catchment are under the post-glacial marine border, 171 meters above sea level. The soils therefore mostly consist of marine clay (Berger & Bremset, 2011; Kristiansen, 2007). The upper part of the watershed consists of open areas, and steep hillsides. The size of the watershed contributes to high biological and geological diversity, with several rich marshes, and riparian vegetation consisting of alder (*Alnus Incana*)- and lime pine forest (Anonym, 2018). The lower part runs through heavily cultivated areas consisting of agriculture and urbanization. Verdal watershed is the only large watershed in middle Norway with few or no water regulations for hydropower. The watershed is mainly used for recreational activities. The catchments geology is from the Caledonian orogeny, with greenschist, schist, and phyllite. The mountain areas toward the Swedish border reaches 1000 meters above sea level, and large parts of the valley are below the marine border, with fjord- and sea deposits. The riverbanks adjacent to the river course consist mainly of alluvial deposits (Anonym, 2018).

The lower parts of Verdal river tributaries drain through intensive cultivated landscape, and many of the streams have been straighten, lowered, and trenched to prevent erosion (Hol, 2018; Vårhus, 2016). All the selected streams are affected by roads crossing the stream course. The migration barriers are because of these roads, and poorly built culverts, with no consideration for fish migration. Recently, many of these culverts were improved to decrease barriers and increase available space for fish. The upper part of the streams mainly runs through forest, and less cultivated landscape, and therefore are less impacted by human activities. Previously, the upper catchments of the streams consisted of marshes, but due to cultivation of the land, marshes disappeared. This resulted in shifts in the hydrological regime (Hol, 2018).

The selected streams in this study are chosen based on location, degree of degradation and restoration measure. Seven of the streams are considered streams with restored sections, and three of the streams (Bjørkbekken, Skjørdalsbekken and Rossvollbekken) are considered control streams (Table 1). They are not qualified as reference streams based on the water directive, however they are used as control streams in consideration of statistical tests in this study.

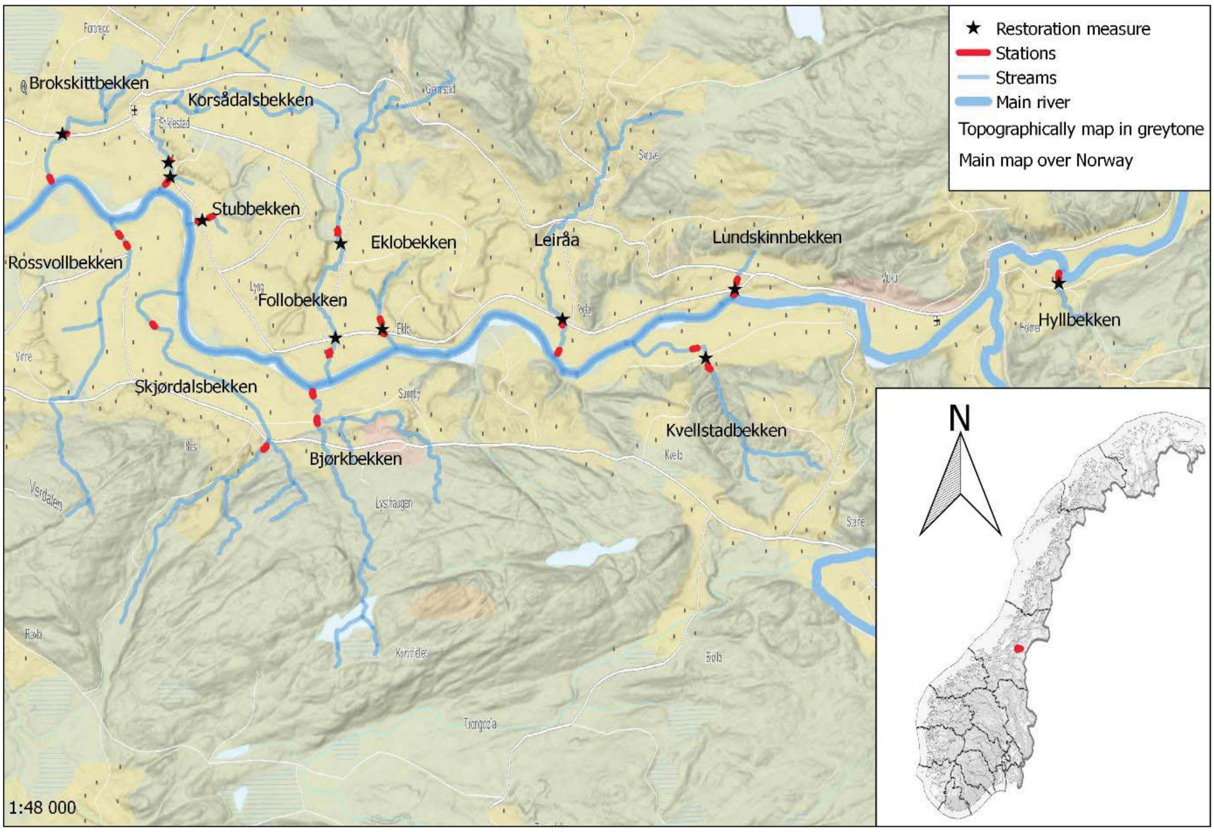

Figure 1. Map over macroinvertebrate sampling sites and restoration measure located in Verdal, Norway.

In addition, Leiråa and Hyllbekken are also considered as control streams, because both samples were taken downstream the restoration measure (Table 1).

Table 1. Description of study streams with type of restoration measure (Hol, 2018; Kristiansen, 2007; Vårhus, 2016). LoS= total length of stream.

Stream name	LoS (km)	Description	Res. measures	Year
Brokskittbekken	5.1	Drains mainly through cultural landscape. Some reaches are characterized by erosion. The stream was heavily polluted in 1994.	Spawning gravel	N/A
Rossvollbekken	1.86	Drains through intensive agricultural landscape. Little riparian vegetation, part from the downstream. Bottom substrate consist of sand/silt with outlet of stone.	Control	
Korsådalsbekken	4.62	The stream is piped at the museum area and characterized by erosion downstream Rv757. Bottom substrate are mainly alluvial deposits with gravel in the stream	Improvement of culvert	2017
Stubbekken	5.9	Drains mainly through cultural landscape. Bottom substrate consist of sand and silt. Digs on the side and the bottom.	Improvement of culvert	2017
Skjørdalsbekken	7.38	Lower part of the stream drains through agricultural landscape with parts characterized by heavy erosion. Upstream drains through forest. Records of sewage runoff from a pig farm. Bottom substrate consist of gravel and sand/silt.	Control	
Bjørkbekken	7.5	Drains through agricultural landscape with some forest. Good riparian vegetation on both sides. Bottom substrate mainly consist of gravel and stone.	Control	
Follobekken	6.06	Drains through agricultural landscape with grass and grain production. Upstream consist mainly of sand/silt. The stream lack some places riparian vegetation.	Thresholds up to culvert	2017
Eklobekken	1.6	The bottom substrate downstream Fylkesvei 757 consist mainly of gravel. The outlet is laid with stone. Upstream Fylkesvei 757 consist mainly of sand/silt and gravel. Characterized by erosion.	Culvert	2017
Leiråa	7.45	Heavily polluted in 2006. Characterized by erosion. Upstream have an older garbage deposit. Runoff from silage effluent by hay balls. Bottom substrate consist mainly of sand/ silt and gravel.	Control	
Kvellstadbekken	6.84	Drains through agricultural landscape. Stream was redirected due to conflict with gravel pit. Bottom substrate consist of sand/ silt and gravel downstream and upstream stone and gravel.	Thresholds and culvert	2016
Lundskinnbekken	2.19	Bottom substrate consist of gravel, stone and block. Drains through agriculture landscape upstream Fylkesvei 757.	New culvert	2017
Hyllbekken	3.6	Lower part of the stream drains through agricultural landscape with no or little vegetation. Bottom substrate consist mainly of gravel and stone.	New threshold up to culvert.	2017

2.2 Habitat description

The registration of habitat characteristics in the selected study streams were performed in late May 2018. The habitat characteristics were described at each station. The length of the stations was measured and then divided into five cross sectional transects. At each transect was water depth in meters recorded at five points on a line (10, 25, 50, 75, 90 % of width) from bank to bank, and the average depth was calculated afterwards. Water velocity (m/sec) was registered by a simple method of timing how fast a leaf travelled 1 meter downstream. Streambed substrate was registered along the transect by how much percentage of each substrate type (sand/silt, cobbles, gravel, blocks and stones) covered the bottom. Streambank and water-surface canopy were registered visually by percentage covered. Moss and algae were also registered visually by percentage covered in the station transect. Number of pools and number of dead wood items (longer than 1m and wider than 0.1m) were counted for the whole stations, and the length of the stations was measured (see Appendix 4).

2.3 Data collection

The benthic macroinvertebrate data from 2018 was collected between 29.10.18 to 3.11.18. To obtain data, a standard stream kick-sampling method was conducted to obtain a good representation of the benthic macroinvertebrate communities present in the streams. Streams examined had one up-stream and one down- stream station. Two samples were collected in each station by 2×30 seconds kick-sampling with a 45 mm mesh hand net. The hand net was placed on the stream bottom upstream and walking against the current while kicking the substrate. This allows any invertebrates to swirl from their hiding places and flow downstream to be collected by the hand net.

After the minute has past, the content in the hand net was placed in a plastic tray to organize the sample. Larger objects such as rocks and vegetation were removed from the sample before emptied into a double plastic bags containing 95 % ethanol to preserve the macroinvertebrates.

Macroinvertebrate data from 2017 were obtain from a previous study conducted by Hol (2018). Only a selection of streams (Hyllbekken, Lundskinnbekken, Kvellstadbekken, Skjørdalsbekken, Rossvollbekken, Stubbekken and Korsådalsbekken) were sampled in both 2017 and 2018.

The fish density data were collected by Hanne Marie Richenberg by electrofishing in august 2018 (Richenberg, 2019). Each station was sampled in three rounds. The Zippin-method were used to estimate the fish densities (Zippin, 1958).

2.4 Macroinvertebrate identification

The samples were brought to Ås, Akershus for further sorting and identification. The samples were emptied into a square plastic tray and divided into four equally sized subsamples, to easier collect and pick out the invertebrates in each sample. The macroinvertebrates was pick out using a tweezer and stored in small glass bottles containing 95 % ethanol for preservation.

Identification of the macroinvertebrates was completed using a Leica MS5 stereo loupe with 4x magnification. Literature used for the identification consisted of Stoneflies (Plecoptera) of Fennoscandia and Denmark (Lillehammer, 1988), Aquatic Insects of North Europe volum 1 (Nilsson, 1996), Trichoptera larvae of Finland: A key to the Caddis Larvae of Finland and Nearby Countries (Rinne & Wiberg-Larsen, 2017), Guide to Freshwater Invertebrates (Dobsen, 2012), and Insektslære for fluefiskere (Krogvold, 2008).

The macroinvertebrates were identified to the lowest possible taxonomic level determined by the available literature and knowledge about them, their role as bioindicators and the difficulty level of identifying them. Organisms such as oligochaetes were only identified to their classes, whilst Coleroptera and Diptera were identified to family level. The EPTs – Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies) – were identified to species level. Some of the organisms were damaged due to handling and sampling that it was identified to the lowest possible taxonomic level.

The completed species list was quality checked by the institute of zoology at the Natural history museum, University of Oslo, by Trond Bremnes.

2.5 ASPT index

The average score per taxon (ASPT) is an index system based on different benthic macroinvertebrate tolerance to organic pollution. The score system ranges from 10 – 1, where 10 is low tolerance to pollution, and 1 is high tolerance to pollution. The index has 85 scoring taxa (see Appendix 1). The families without scores are ignored in the calculation. The ASPT index scores are calculated by the following formula:

$$ASPT\ score = \frac{\sum family\ scores}{number\ of\ scoring\ families}$$

The ASPT index score indicate the ecological state of the waterbody, in this case, the streams, and if measures are needed to achieve acceptable ecological state (Table 2).

Table 2. Five ecological classes with colour code based on EU`s water framework directive. “Very good”, and “good” are acceptable ecological classes, anything below need restoration measures to obtain acceptable ecological status of the water body.

<table>
<tr><td>Ecological state</td><td>ASPT limits</td><td></td></tr>
<tr><td>Very good</td><td>>6.8</td><td rowspan="2">Acceptable ecological state</td></tr>
<tr><td>Good</td><td>6.8-6.0</td></tr>
<tr><td>Moderate</td><td>6.0-5.2</td><td rowspan="3">Unacceptable ecological state (measures are needed)</td></tr>
<tr><td>Poor</td><td>5.2-4.4</td></tr>
<tr><td>Very Poor</td><td><4.4</td></tr>
</table>

2.7 Statistical analysis

The statistical analysis preformed in this study was conducted using the software program R version 3.5.2 (R Development Core Team, 2018). Microsoft Excel was also used to visualize some of the results. All statistical tests are based on a significant level of alpha = 0.05.

The macroinvertebrate species data constitute the main response data used in this study. Two main predictors were used in this study 1) effects such as restoration measure, station (upper/lower), fish density, and 2) habitat characteristics such as wood debris, pools, water velocity, depth, width, algae, moss and substrate. The original species names were altered to modified codes for the ordination analysis (see Appendix 2).

2.7.1 Ordination

Topics of community ecology have often large and complex datasets with numerous variation in species richness and abundance across a collection of different environmental gradients (Smilauer, 2014). These types of datasets with different variables and variation within, might be difficult to organize. By looking at each variable separately, to find the most significant explanatory factor makes little statistical sense. Ordination is an analytical technique that account for the multidimensionality of the data in as few tests as possible, this also reduces the chance of false positives (Type I errors) (Smilauer, 2014). The ordination analysis in this study was conducted using the R package “vegan”. The vegan package provides tools for descriptive community ecology, including basic functions for community ordination analysis.

Vegan divides these techniques into two analysis – unconstrained ordination and constrained ordination. Methods of unconstrained ordination involves Principal Components Analysis (PCA) or Correspondence analysis (CA). The goal of these is to find the axes that are the most influential in shaping the observed structure in the response data. Constrained ordination is introduced where there are one or more accompanying explanatory variables that can be used to explain the variation in the response data. The two most common constrained ordination methods are the Redundancy Analysis (RDA), and the canonical correspondence analysis (CCA).

How to choose a linear or unimodal ordination model depends on the amount of turnover SD units of the response data, by conducting a DCA analysis, to obtain the length of the longest DCA axis in turnover units. If the turnover units are higher than 3, the unimodal ordination model is best fitted, in this study the turnover units in the DCA are lower than 3, therefore the linear ordination model is best fitted to the data (Smilauer, 2014).

Because of the linearity in the data, only PCA and RDA was used for the ordination in this study. The eigenvalues of the different axes represent the variation in the data: the higher the eigenvalue of an axis is, the more variation in the data is explained by the variables that particular axis represents (Smilauer, 2014).

In an ordination diagram, the relative distribution of cases and arrows signifies their correlation. For examples, arrows going in opposite directions are negatively correlated, which in the case of this study is indicative of opposing environmental requirements. The same interpretation applies to cases; the further away from each other, the fewer environmental and ecological attributes they have in common, and vice versa. The longer the arrow, the more important that response data is (Smilauer, 2014).

In order to explore and quantify effects of both human-induced effects, fish density, and habitat characteristics on benthic macroinvertebrate communities, linear effects candidate models were fitted using ASPT and Shannon-Wiener diversity index (Searle, 1971). Effects of habitat characteristics were fitted using principal component scores (PC) from a preceding PCA-analysis using habitat characteristics as responses. Numbers of PCs to use in the analysis was determined from the numbers needed to explain at least 50 % of the habitat variation.

2.7.2 Model selection

Candidate model selection of both multivariate (constrained ordination) and linear univariate candidate models were determined by the Akaike`s Information Criterion, AIC. AIC is estimated as the sum of a fitted model`s deviance and the number of parameters (K) times two included in the model (AIC=deviance + K×2). The background for this is to find models that most efficiently balances parameter estimation precision and bias. The model with the lowest AIC- value is selected as the model with the highest AIC support among the candidates. In this study a corrected version of the AIC (AICc) were used, that penalized complex models to a larger degree when n is small: AIC=deviance + 2K×(n/(n-K-1)) (Akaike, 1974; Anderson, 2008).

3 Results

3.1 General macroinvertebrate compositions

The total of 56 taxa were found in the river Verdal tributaries. 97.4 % of these were EPT species, others were Diptera (true flies), and Oligochaeta (worms). The remaining other taxa (2.6 %) were a mixture of Coleroptera (beetles), Collembola (springtails), Bivalvia (molluscs), Gastropoda (snails), Megaloptera (mud flies), Acari (mites) and Amphipoda (crustacea). Almost all the taxa occurred as larvae, however some pupae did occur mostly in the order Coleroptera. The most common species found overall, were in the order Ephemeroptera, the species *Bäetis rohdani* (30.74 % of the total proportions) followed by the family Simuliidae (13.44 % of the total proportions) in the order Diptera (Figure 2).

In the order Ephemeroptera the three most common species were *Bäetis rhodani* with the highest relative abundance (89.84 %), followed by *Bäetis nigris* (6.36 %), and *Bäetis muticus* (2.21 %). In the order Plecoptera the three most common species were *Nemoura cinerea* (56.26 %), *Brachyptera risi* (17.31 %), and *Capnia bifrons* (13.68 %). The three most common species in the order Trichoptera were *Rhyacophila nubile* (57.55 %), *Silo paliplus* (15.09 %), and the family Limnephilidae (14.62 %) (Table 3).

Table 3. Total proportions and abundance of the major macroinvertebrate groups with their most common species or family from samples taken 2018 in tributary streams of Verdalselva. (See appendix 3 for raw data).

Order	**n**	**Total prop. %**	**Family/ species**	**n**	**Relativ prop. %**	**Total prop. %**
Ephemeroptera	3899	34.21	*Bäetis rhodani*	3503	89.84	30.74
			Bäetis nigris	248	6.36	2.18
			Bäetis muticus	86	2.21	0.75
Plecoptera	855	7.50	*Nemoura cinerea*	481	56.26	4.22
			Brachyptera risi	148	17.31	1.30
			Capina bifrons	117	13.68	1.03
Trichoptera	212	1.86	*Rhyacophila nubila*	122	57.55	1.07
			Silo paliplus	32	15.09	0.28
			Limnephilidae	31	14.62	0.27
Diptera	3918	34.38	Simuliidae	1532	39.10	13.44
			Chironomidae (*Tanypodinae sp.*)	1332	34.00	11.69
			Pediciidae (*Dicronata sp.*)	326	8.32	2.86
Others	300	2.63	*Gammarus lacustris*	166	55.33	1.46
			Total	**8093**		**71.01**
Oligochaeta	2213	19.42				
Total	**11397**	**100**				

Eklobekken had the highest abundance of all tributaries with 1959 individuals. The order Ephemeroptera as the dominant group followed by Diptera. The stream with the lowest abundance was Stubbekken with a total of 136 individuals, with Diptera dominating (63.2 %) (Figure 2).

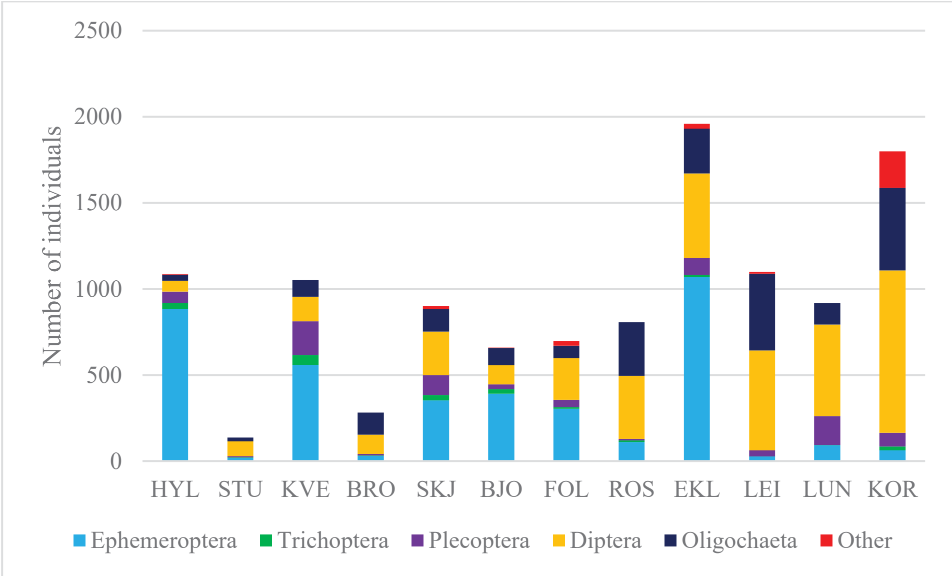

Figure 2. Number of individuals per sample of each stream sorted by macroinvertebrate groups. (HYL = Hyllbekken, STU = Stubbekken, KVE = Kvellstadbekken, BRO = Brokskittbekken, SKJ = Skjørdalsbekken, FOL = Follobekken, ROS = Rossvollbekken, EKL = Eklobekken, LEI = Leiråa, LUN = Lundskinnbekken, and KOR = Korsådalsbekken).

3.2 Macroinvertebrate composition, environmental variables and annual variation

The Detrended Correspondence Analysis (DCA) analysis had axis lengths lower than 3 (Axis lengths = 1.855) (Table 4), therefore a linear ordination analysis was chosen for the analysis of the environmental variables.

Table 4. Detrended correspondence analysis (DCA) analysis to determine which model approach to use in the analysis of the habitat characteristics (linear or unimodal). If the axis lengths are lower than 3, a linear ordination approach is suitable. DCA1 explains 30 % of the variation in the habitat characteristics.

	DCA1	**DCA2**	**DCA2**	**DCA4**
Eigenvalues	0.3001	0.14034	0.10980	0.12614
Decorana values	0.3020	0.04315	0.02466	0.01444
Axis lengths	1.855	1.34739	0.96240	1.10607

To examine the environmental variables a principal component analysis (PCA) was performed to organize the data (Figure 3).

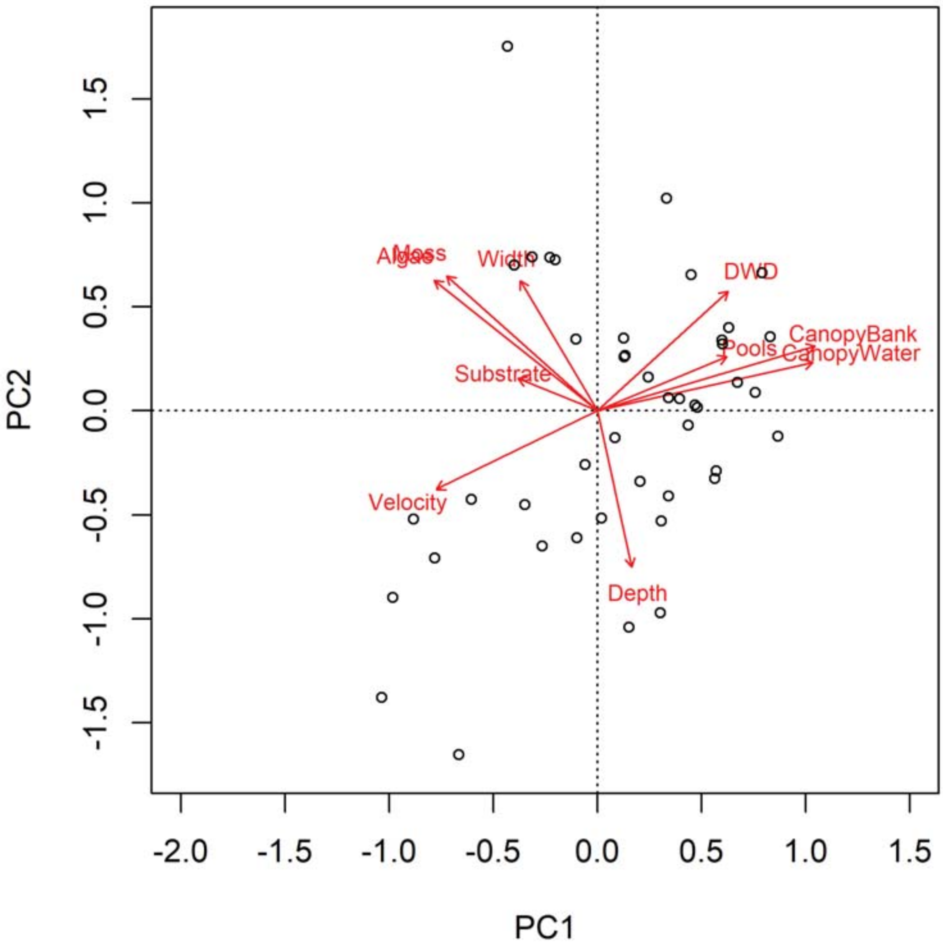

Figure 3. Biplot of Principal Components Analysis (PCA) of the habitat characteristics. Where PC1 (explains 7.93 % of the variation) is the allochthonous production, and PC2 (explains 4.44 % of the variation) is the autochthones production.

The PCA analysis of the habitat characteristics (Figure 3) in the streams showed that depth negatively correlates with width, moss, algae and substrate (i.e., pertinent to the autochthone production in the stream). With increasing depth of the stream, less moss and algae will be present, the substrate will be in a finer form (sand/silt), and the streams will become narrower. On the other side, water velocity negatively correlates with dead woody debris, side vegetation, canopy vegetation, and pools (i.e., pertinent to allochthone production). With increasing water velocity, the less dead woody debris, side vegetation, canopy vegetation, and pools will be present in the stream. Water velocity and depth are predicted to have low or near to zero correlation to each other, the same is accounted for the autochthone, and the allochthone side (near-to-zero correlation to each other).

Table 5. Detrended correspondence analysis (DCA) analysis to determine which model approach to use in the analysis of macroinvertebrate data (linear or unimodal). If the axis lengths are < 3, a linear ordination approach is suitable. DCA1 explains 13,83 % of the species variation?

	DCA1	DCA2	DCA3	DCA4
Eigenvalues	0.1383	0.2545	0.1386	0.1078
Decorana values	0.2902	0.2597	0.1493	0.1067
Axis lengths	2.6548	2.2893	1.6914	1.6144

The axis lengths of the DCA analysis indicated that a linear analysis (axis lengths = 2.6548) (Table 5) approach was suitable to examine the variation in the macroinvertebrate community data, therefore a Redundancy Analysis (RDA) was performed.

Table 6. Selection of model by a forward selection routine permutation tests in constrained ordination. The selected model (PC2 and fish.density) had $R^2_{adj} = 0.133$.

Predictor	Df	AIC	F	Pr(>F)
-UpperLower	1	133.18	0.7806	0.640
-Res.measure	2	132.57	0.9706	0.480
-PC3	1	133.52	1.0659	0.345
-PC1	1	134.01	1.4696	0.115
PC2	1	134.83	2.1631	0.025
fish.density	1	137.71	4.7152	0.005

R estimates AIC after removal of variables. If an important variable is removed (fish density) will the AIC value increase and if an unimportant variable is removed, then AIC will decrease (Res.measure). Even though PC2 and fish density had the highest AIC score, they were the only predictors with significant impact on the macroinvertebrate variation (Table 6). Therefore, PC2 and fish density were selected based on the forward selection routine permutation test. Both PC2 and fish density affect the variation in the invertebrate with significant correlation (P-value = 0.025 and 0.005 respectively). Therefore, an RDA analysis was conducted with PC2 and fish density as effects. The restoration measure has weak effect on macroinvertebrate variation sampled from the streams (P-value = 0.480).

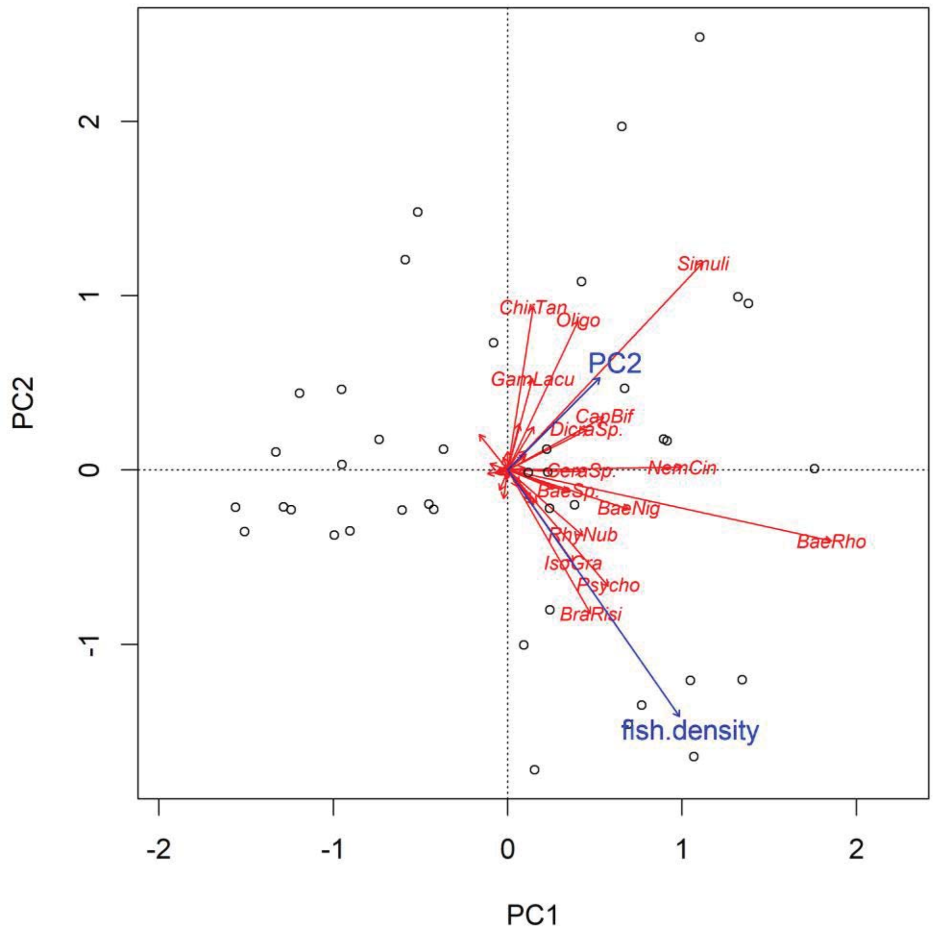

Figure 4. Biplot of the Redundancy Analysis (RDA) of the benthic macroinvertebrate community with PC2 (allochthonous) and fish density as predictors. Invertebrates sampled in 2018.

The RDA analysis indicates that PC2 (allochthone production) positively correlates with pollution tolerant species such as such as Simuliidae, Chironomidae and Oligochaeta, indicating that increased production of allochthone materials in the streams induces a shift in the macroinvertebrate species diversity abundance towards tolerant species. However, the biplot shows that there is weak correlation between PC2 and less tolerant species such as *Rhyacofila nubile* and *Isoperla grammatica,* according to the perpendicular angle of the arrows. This indicate that increased allochthonous production negatively affect intolerant species less. Fish density positively correlates with intolerant species, and giving the length of the arrow, fish density as predictor have a stronger correlation than PC2, indicating that the presence of fish increases the abundance of species such as *Rhyacofila nubile*, *Isoperla grammatica* and

Brachyptera risi, and decreases the abundance the abundance of Chironomidae, Oligochaeta and *Gammarus lacustris,* though, this negative correlation is weaker than the positive correlation on intolerant species (Figure 4).

RDA performed on the 2018 data and Hol (2018) 2017 data yielded most support for a model including tributaries and year as predictors. The RDA analysis showed that there was a significant difference in macroinvertebrate species composition between years. However, there were no significant difference in species composition between the tributaries between years. The streams as shown in Figure 5, have clumped together by distance to fjord, the upper streams (Kvellstadbekken, Lundskinnbekken, and Hyllbekken), and the streams lower down (Skjørdalsbekken, Rossvollbekken, Stubbekken and Korsådalsbekken). The distance between the upper and lower streams indicate that there is a difference in species composition between upper and lower streams, however this difference is not significant. The presence of *Bäetis rohdani* seems to not be affected by season, while several species of the family Chironomidae are affected by season (Figure 5).

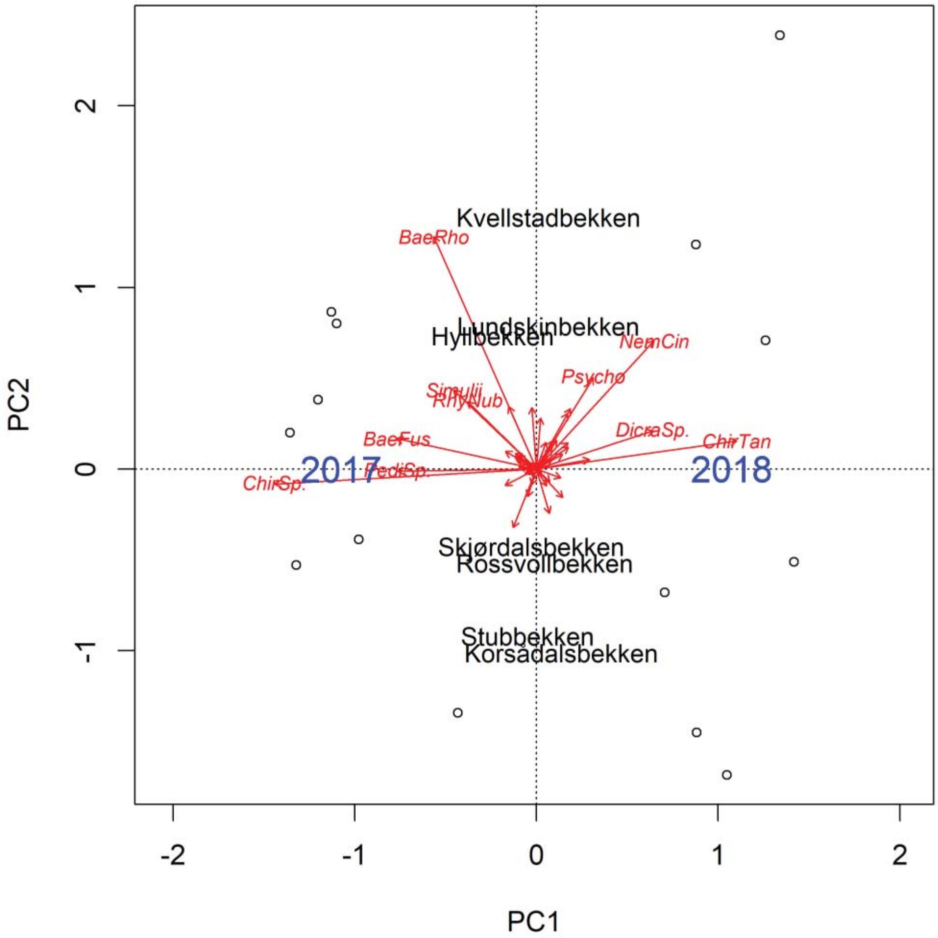

Figure 5. Biplot of Redundancy analysis (RDA) of difference in species composition between the years 2017 and 2018, with a selection of the most common invertebrate species. The ten most heavily loading invertebrate taxon are displayed with abbreviated names.

3.3 Shannon Wiener diversity index

The restoration measure effect candidate models on the Shannon-Wiener diversity index received little AIC-support, indicating little or no effect on macroinvertebrate diversity in restored versus unrestored streams. The model with highest support was just the average (intercept model) Shannon Wiener diversity index. Based on table x, the model selected for an ANOVA test was UpperLower with the second lowest AIC score (AICc =36.10) (Table 7).

Table 7. Ranked model selection table for candidate linear models fitted to predict Shannon-Wiener index values. K = number of fitted values, AICc = corrected Akaike`s Information Criterion, Delta AICc = difference between AICc for a given model and the one with the lowest AICc score, AICcWt = AICc weigth (relative support), and LL = log likelihood value.

Fixed effect model structure	K	AICc	Delta_AICc	AICcWt	Cum.Wt	LL
SW~1	2	34.55	0.00	0.19	1.00	-15.11
UpperLower	3	36.10	1.56	0.09	0.81	-14.72
PC1	3	36.64	2.09	0.07	0.73	-14.99
fish.density+PC1	4	36.69	2.15	0.06	0.66	-13.78
fish.density+UpperLower	4	36.88	2.33	0.06	0.60	-13.87
PC2	3	36.88	2.34	0.06	0.54	-15.11
Dist.fjord	3	36.89	2.34	0.06	0.49	-15.11
UpperLower*PC2	5	36.69	2.42	0.06	0.43	-12.60
Res.measure+fish.density	5	37.00	2.45	0.05	0.37	-12.62
Dist.fjord+fish.density	4	37.01	2.46	0.05	0.32	-13.93
fish.density+PC2	4	37.04	2.49	0.05	0.26	-13.95
Dist.fjord+UpperLower	4	38.55	4.00	0.03	0.21	-14.70
UpperLower+PC2	4	38.58	4.03	0.02	0.19	-14.72
Dist.fjord+PC1	4	38.87	4.32	0.02	0.16	-14.86
Dist.fjord+fish.density+PC1	5	39.22	4.67	0.02	0.14	-13.73
fish.density+UpperLower+PC1	5	39.25	4.71	0.02	0.12	-13.74
fish.density+PC1+PC2	5	39.31	4.77	0.02	0.10	-13.77
Dist.fjord+Res.measure	5	39.81	5.26	0.01	0.09	-14.02
Res.measure+UpperLower	5	39.95	5.41	0.01	0.07	-14.09
Res.measure+PC1	5	39.97	5.42	0.01	0.06	-14.10
Res.measrue+PC2	5	39.97	5.43	0.01	0.05	-14.10
UpperLower*PC1	5	40.52	5.97	0.01	0.04	-14.38
UpperLower+PC1+PC2	5	41.11	6.56	0.01	0.03	-14.67

There is a difference in macroinvertebrate diversity between upper and lower stations, the upper stations tend to have a higher diversity, than the lower stations, however, this difference is not significant (P-value = 0.391) (Table 8) (Figure 6).

Table 8. Parameter estimates and corresponding test statistics for the selected linear model in table x fitted to predict Shannon- Wiener index values as function of upper and lower stations within streams. Upper = upper stations, Lower = lower stations.

Parameter estimates				Effect test					
Term	Level	Estimate	SE	Effect	df	SS	MS	F	p
(Intercept)	Lower	1,5859	0,0802	UpperLower	1	0,0969	0,096877	0,0753	0,391
UpperLowerUpper	Upper	0,09843	0,11343						

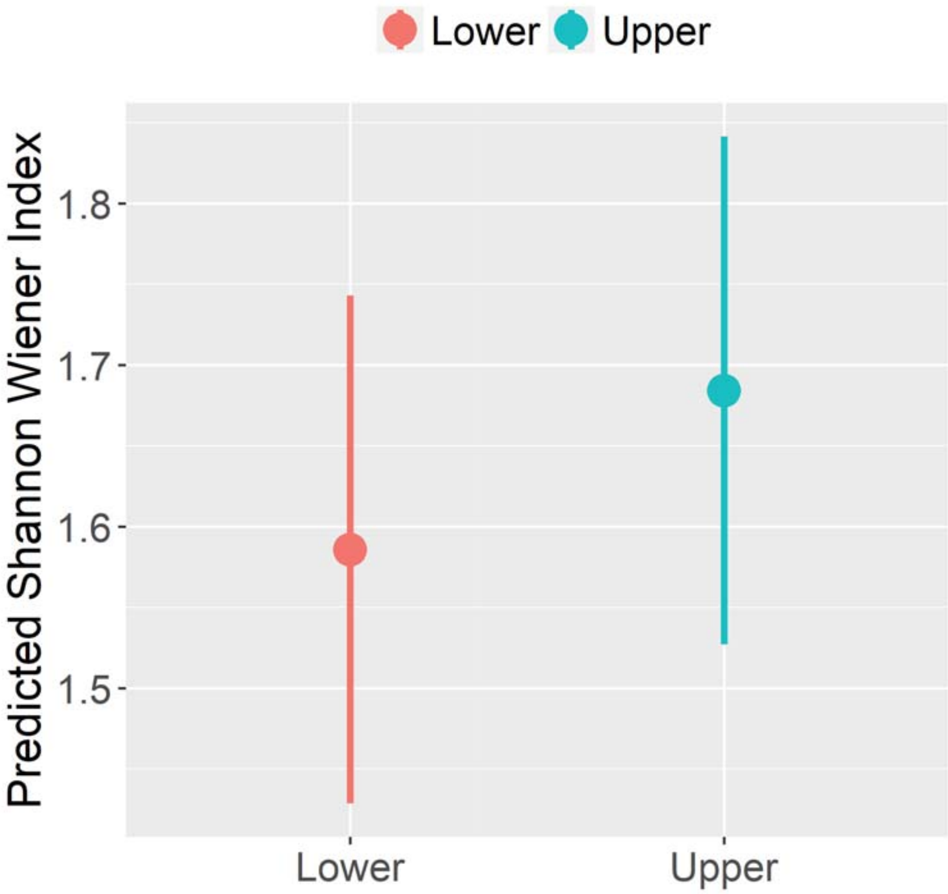

Figure 6. Predicted macroinvertebrate Shannon-Wiener index as a function of upper and lower stations. Predictions were retrieved from the most supported linear model from Table 8.

3.4 ASPT index

The ASPT index scores (Average Score per Taxon) show that only 1 in 12 streams achieved minimum good ecological status based on EU`s water framework directive. Brokskittbekken, Rossvollbekken, Stubbekken, Follobekken and Leiråa had very poor ecological status. Eklobekken, Lundskinnbekken and Hyllbekken have poor ecological status. Stubbekken yielded lowest ASPT index score (ASPT= 3.25). Skjørdalsbekken, and Bjørkbekken have moderate ecological status. Only Kvellstadbekken achieved good ecological status (ASPT =6,01). None of the stream achieved very good ecological status (Figure 7).

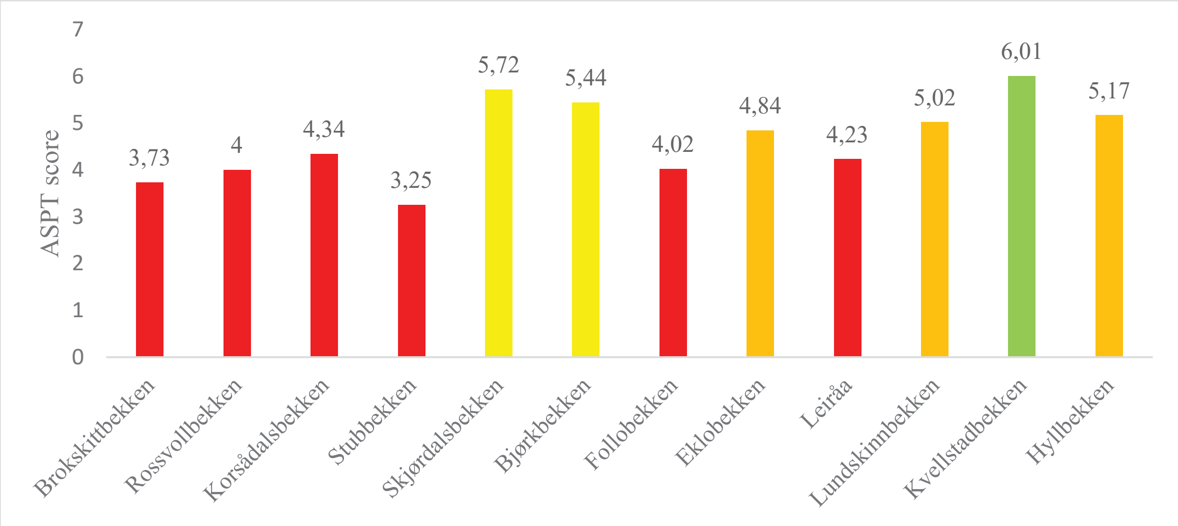

Figure 7. ASPT score for each stream, based on EU`s water framework directive. Very poor= <4.4, poor= 4.4 – 5.2, moderate = 5.2 – 6.0, good= 6.0– 6.8, very good= >6.8.The colours represent the ecological state (Table 2).

When comparing the ASPT scores between 2017 and 2018, the results showed that Rossvollbekken and Stubbekken went from poor to very poor ecological status, while Kvellstadbekken went from very poor in 2017 to good ecological status in 2018. Skjørdalsbekken had good ecological status in 2017 but, got moderate ecological status in 2018. Korsådalsbekken (very poor) and Lundskinbekken (poor) had no difference in ecological status. Only Kvellstadbekken achieved better ecological status, while the other streams got reduced or unchanged ecological status (Table 9).

Table 9. Comparisons of ASPT index score between 2017 and 2018.

Stream	**2017 (Hol, 2018)**	**2018**
Rossvollbekken	Poor	Very poor
Korsådalsbekken	Very poor	Very poor
Stubbekken	Poor	Very poor
Skjørdalsbekken	Good	Moderate
Kvellstadbekken	Very poor	Good
Lundskinnbekken	Poor	Poor
Hyllbekken	Moderate	Poor

According to Table 9, the restoration measures have little effect on the ASPT scores of the streams. The model with highest support (28 %) is distance to fjord + fish.density + UpperLower (Table 9). This candidate model was selected for an ANOVA test. As seen in table x, the restoration measure model had only 2 % support, and had no significant effect on the ASPT index of the streams.

Table 10. Ranked model selection table for candidate linear models fitted to ASPT index scores. K = number of fitted values, AICc = corrected Akaike`s Information Criterion, Delta AICc = difference between AICc for a given model and the one with the lowest AICc score, AICcWt = AICc weigth (relative support), and LL = log likelihood value.

Fixed effects model structure	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Dist fjord + fish density + upperLower	5	120.71	0.00	1.00	0.28	-54.47	0.28
Dist fjord + fish density	4	121.17	0.46	0.79	0.22	-56.02	0.49
Dist fjord * fish density	5	121.85	1.14	0.57	0.16	-55.04	0.65
Dist fjord + fish density + PC1	5	122.86	2.15	0.34	0.09	-55.55	0.75
Dist fjord + fish density * UpperLower	6	123.14	2.43	0.30	0.08	-54.30	0.83
Dist fjord* UpperLower + fish density	6	123.39	2.68	0.26	0.07	-54.42	0.90
fish density + UpperLower	4	125.24	4.52	0.10	0.03	-58.05	0.93
fish density + PC1	4	125.65	4.94	0.08	0.02	-58.25	0.95
fish density + PC2	4	126.27	5.56	0.06	0.02	-58.57	0.97
fish density + UpperLower+ PC1	5	126.47	5.76	0.06	0.02	-57.35	0.98
Res measure + fish density	5	128.20	7.49	0.02	0.01	-58.22	0.99
fish density + PC1 + PC2	5	128.22	7.51	0.02	0.01	-58.23	1.00

The distance to fjord + fish density + Upper Lower model yielded highest support based on the AIC score (Table 10), explaining 28 % of the variation in ASPT index data. Therefore, this model was selected for an ANOVA test. Restoration measures, allochthone and autochthone production yielded little support based on the AIC, explaining less than 2 % of the variation in the ASPT index data (Table 10).

Table 11. Parameter estimates and corresponding test statistics for the selected linear model in Table 10 fitted to predict ASPT index scores as function of distance to fjord, fish density, and upper and lower stations. Dist.fjord = distance to fjord, fish.density = fish density, Upper = upper stations, Lower = lower stations.

Parameter estimates				Effect test					
Term	**Level**	**Estimate**	**SE**	**Effect**	**df**	**SS**	**MS**	**F**	**p**
Intercept	Lower	2.6470	0.5239	Dist.fjord	1	8.086	8.0856	8.1575	0.007078
Dist.fjord		0.1059	0.0399	fish.density	1	16.409	16.409	16.555	0.000247
fish.density		0.0070	0.0023	UpperLower	1	2.861	2.8609	2.8864	0.097961
UpperLower	Upper	0.5869	0.3454						

Both distance to fjord (P-value =0.0007) and fish density (P-value = 0.0002) have significant effect on the ASPT score. UpperLower (P-value = 0.097) is not significant (Table 11).

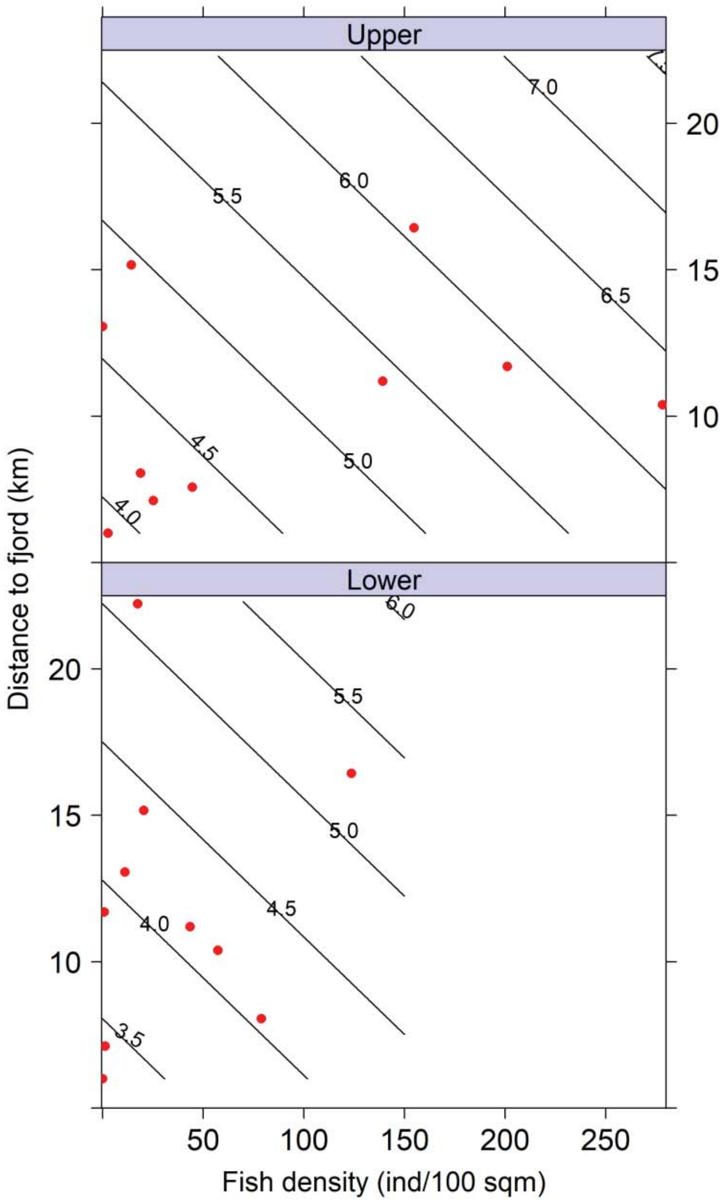

Figure 8. Contour plot of predicted ASPT score responses to increased distance to the fjord and increased fish density in streams, between upper and lower stations within streams. Predictions are shown as isocline (lines) and were derived from the selected model presented in Table 11. Red dots represent raw data.

Streams located further up the watershed tended to have better ASPT score, than the streams located closer to the fjord. According to the ANOVA test, the correlation between ASPT score and distance to fjord is significant. The correlation between fish density and ASPT score was also significant, indicating that streams with higher densities of fish, also had better ecological status. The lower stations (closer to the main river) inclined a poorer ecological status than the stations located further up the stream, however, this was not significant (Figure 8).

4 Discussion

4.1 Effect of restoration measures and environmental variables on macroinvertebrate assemblages

The results indicate that the restoration measures have no or little effect on macroinvertebrate assemblages, as to yet. Several other studies have found similar results (Jahnig et al., 2010; Lepori et al., 2005; Louhi et al., 2011), that restoration efforts have no effect on macroinvertebrate communities. There are several possible explanations to why restoration measures have weak or no effect on macroinvertebrate assemblages. Miller et al. (2010) and Kail et al. (2015) found that it was easier to increase the number of individuals, rather than establishing new taxa in restored reaches of a stream, since restoration had a greater effect on macroinvertebrate abundance than diversity. In-stream measures had greatest effect on macroinvertebrate diversity, because in-stream measures include placement of large woody debris and creating riffles which results in increase of habitat and substrate complexity (Kail et al., 2015). This is consistent with a study done by Barnes et al. (2013), that substrate complexity had a positive correlation with macroinvertebrate richness and abundance, when habitat type was excluded. They found that rough substrate supported greater macroinvertebrate diversity, and that richness increased with larger substrata (Barnes et al., 2013). In addition, many degraded streams lack appropriate oviposition for aquatic insects. Many caddisfly species require specific conditions to oviposition. Some caddisfly species need boulders, submerged vegetation or wood to lay their eggs (Blakely et al., 2006). This might be an explanation for why rough and larger substrates support greater richness, because more space is available, and therefore more microhabitats, which allow species to co-exist. Restoration efforts often include improvement of fish migration barriers, and addition of spawning gravel, which might not be suitable substrates for benthic macroinvertebrates.

Many of the streams in this study have finer substrate consisting of sand and silt. Human activities in the catchments have resulted in increased fine sediment loading, both inorganic and organic sediment loads to waterbodies. Increased loads of fine sediment can be detrimental to aquatic biota, especially benthic macroinvertebrates (Jones et al., 2012). Aquatic biota itself contributes to the production of fine sediments, however anthropogenic sources exceeds the background conditions, resulting in harmful conditions for benthic macroinvertebrates, such as abrasion, burial, clogging, unstable substrates, food availability, refugia, predation, and toxic substances (Jones et al., 2012). The restoration measures conducted in the tributaries of Verdal river had little focus on decrease of sedimentation load, therefore, sedimentation loads might be contributing factor resulting in weak responses by macroinvertebrates to the restoration measures. Decrease of fine sediment loads, and reconstruction of riffles with coarser substratum should be a priority in restoration of benthic macroinvertebrate assemblages in streams in agricultural landscapes (Li et al., 2018).

Intensity of restoration measures might also determine the success of restoration on macroinvertebrate diversity and abundance. Many of the previous restoration projects have only restored shorter reaches of a stream, and this have often resulted in weak responses by macroinvertebrates. However, A study done by Li et al. (2018) examined the response of macroinvertebrates in a restored stream where the stream was densely placed with in-stream restoration measures on a 1000m scale. They found great positive effect on macroinvertebrate recovery in the restored versus the unrestored stream (Li et al., 2018). This highlights the importance of correct restoration intensity and placement of measures to achieve successful recovery of stream biota. The restoration measures in the tributaries of Verdal river have only focused on point restoration measures, therefore, the intensity of the restoration might be too low to achieve successful recovery of stream macroinvertebrates.

The sampling of invertebrates was conducted in 2017 and 2018, which was shortly after the stream measures was undertaken. It can therefore be argued that it is too early to evaluate responses by the invertebrate community. Rubin et al. (2017) pointed out that studies with increased project age found significantly positive responses of invertebrate diversity. Additionally, Kail et al. (2015) found similar results, and drew attention to the need for long-tome monitoring to fully understand the effects on stream biota. Louhi et al. (2011) examined the responses to river restoration on macroinvertebrates in a time-span of 20 years after the restoration efforts was conducted and argued that 15 - 17 years might still be too early for all species to arrive at the restored reaches. They found that macroinvertebrates had weak responses to in-stream restoration efforts, and concluded that the macroinvertebrates were not limited by habitat heterogeneity, however might be dispersal limited. Dispersal is found to be one of the limiting factors of recolonization by benthic macroinvertebrates to restored reaches of a stream (Brederveld et al., 2011). A study done by Tonkin et al. (2014) found that the recolonization of macroinvertebrate diversity is limited by distance to nearest source and the pool of taxa present. They found that in-stream barriers affected colonisation less but, was significant. However, according to Blakely et al. (2006) barriers was found to act as a limiting factor contributing to recolonization of restored reaches versus unrestored reaches of a stream. Road culverts acted as a partial barrier for aerial flight migration upstream, were adult insects encountered many obstacles such as predation in- and traffic above culverts (Blakely et al., 2006). Numerous watersheds might be too degraded, that several taxa might gone extinct in nearby source populations, and therefore not enough to re-establish the previous species diversity to restored reaches (Haase et al., 2013). Nevertheless, this might not be the case in Verdal watershed, as the upper reaches of the streams are less affected by anthropogenic sources, in addition the upper part of the watershed is likewise less impacted by human activities (Hol, 2018; Kristiansen, 2007; Vårhus, 2016). Furthermore, Verdal watershed are also one of the few watersheds in Norway with little (only a small hydropower in Ulvilla) hydropower development (Anonym, 2018). In summary, the intensity of the restorations, and that the samples were taken shortly after the measures was undertaken, might be an explanation for the weak responses by macroinvertebrates to restoration measures conducted in tributaries of Verdal river.

According to the results, the allochthone production of the streams is one of the main determinants of variation in macroinvertebrate assemblages, and therefore the prediction that substrate complexity is the main determinant of macroinvertebrate assemblages have little support. The allochthonous (PC2) production includes inputs from riparian vegetation and runoff from the catchment. This indicate that land use in the catchments of the streams influence macroinvertebrates. The land use in Verdal watershed consist mainly of agriculture and some urban development. Increased production of allochthone material results, in a shift towards pollution tolerant species in the macroinvertebrate composition. Similar results were found by Herringshaw et al. (2011). They found that urban runoff had significantly stronger negative impact on macroinvertebrates, than runoff from agriculture use. This is consistent with the ASPT results in this study. The ecological condition of the streams appears to worsen closer to the fjord. The land use closer to the fjord consist mostly of urban development. This suggest that land use in the catchment influence the water quality, and thus signify the importance of restoration in the catchments in addition to in-stream measures (Palmer et al., 2010).

4.2 Does the presence of salmonids affect macroinvertebrate community structure?

One of the main objectives of this study was to see if the regained populations of salmonids in tributaries of Verdal river, which until recently were considered empty of fish, had an any impact on the macroinvertebrate community. The results revealed that there was a weak negative correlation between fish density and tolerant macroinvertebrate species, such as Chironomids, Oligochaeta and Simuliidae, implying that fish density may have an impact on macroinvertebrate community structure. Previous studies on effects of fish presence on benthic macroinvertebrates have yielded mixed results, from both experiment-, and in-situ studies.

Several studies have reported that fish predation had a significant impact on the macroinvertebrate diversity (Diehl, 1992; Gilinsky, 1984; Reice, 1991; Williams et al., 2003). Indicating that vertebrate predation plays an important role in determining the community structure of macroinvertebrates. Other studies have found little or no effect from fish predation on benthic macroinvertebrates (Allan, 1982; Flecker, 1984; Nicola et al., 2010; Thorp & Bergey, 1981). Revealing that other determinants affect the benthic community structure, rather than top-down forces of fish predation.

A study conducted by Flecker (1984) found that the family Chironomidae was significantly impacted by predation, and therefore concluded that fish might impact the macroinvertebrate community structure, since Chironomidae was one of the most abundant families present in the stream (Flecker, 1984). Chironomids have low recolonization rates compared to other families, which can explain why they are more affected by fish predation than other families (Flecker, 1984). This has likewise been reported in a study conducted by Gilinsky (1984), who found strong negative responses by Chironomids to fish predation. On the other hand, Meissner and Muotka (2006) found no effect on Chironomids, and rather opposite effect than previous studies (Flecker, 1984; Gilinsky, 1984). It is arguable that the fish density might just prefer the same habitat conditions as intolerant species such as *Rhyacofila nubile* (caddisfly)*, Isoperla Grammatica* (stonefly) and *Brachyptera risi,* (stonefly) and therefore find higher fish densities in streams with less abundance of tolerant species.

In the present study, the results indicated that fish density had no effect on macroinvertebrate diversity. Similar results were described by Thorp and Bergey (1981) who found no relation between macroinvertebrate diversity and fish predation. Likewise, Nicola et al. (2010) found that fish predation did not affect macroinvertebrates. Moreover, they reported that in detritus-based food chains might be influenced by bottom-up forces, while in algae-based food webs, top-down forces play a significant role. As mentioned in the previous sub-chapter, the macroinvertebrate assemblages in the tributaries of Verdal river are mostly determined by allochthonous production. Therefore, it is possible that the macroinvertebrates assemblages are controlled by bottom-up forces, rather than top-down forces in the tributaries of Verdal river. Another explanation is that macroinvertebrates are adapted to presence of fish predation, and therefore not influenced by changes in fish densities (Allan, 1982). Allan (1982) concluded that fish predation does not affect invertebrate numbers. On the other hand, fish predation may be a limiting factor in forage activity by invertebrates. It has been shown that invertebrate species increases their nocturnal activity when body size increases. Brown trout are visual hunters, meaning they prefer larger prey over smaller prey, because they are easier to detect (Meissner & Muotka, 2006). Meissner and Muotka (2006) argued that the trout prey on the most convenient and abundant invertebrate species present in the stream. However, the species *Bäetis rohdani* were the most abundant species found in the tributaries of Verdal river, and the results indicating no correlation between fish densities and *Bäetis rohdani*. Mayflies increases its drift and movement during the night (McIntosh et al., 2002), this might explain the why fish densities have no effect on *Bäetis rohdani,* since trout is mainly active during the day (Young et al., 1997). Richenberg (2019) found that there is little competition between brown trout in the tributaries of Verdal river, since 1+ have no negative effect on 0+. The density of brown trout may therefore be lower than the carrying capacity of these systems. This indicate that the predation by brown trout on macroinvertebrates in the tributaries of Verdal river may be lower than it potentially can become.

The results yielded little support for the prediction that the foraging pressure from salmonids changes the species structure in the benthic macroinvertebrate community to predation tolerant species. Since fish densities had no significant effect on the macroinvertebrate diversity, it is possible that the invertebrate community are determined by other variables than the presence of fish. In addition, that the possible explanation to why higher fish densities are found in streams with higher densities of intolerant species, is that fish prefer the same habitat conditions as intolerant species.

4.3 Which variables contribute to the variation in ASPT index scores?

Assessment of the ecological state of the tributaries of Verdal river indicated that the ecological condition of the streams based on the ASPT index scores, are generally poor. Only Kvellstadbekken achieved minimum good ecological condition. Two streams had moderate ecological conditions, four streams had poor ecological state, while the remaining six streams had very poor ecological condition. This indicate that there is high degree of pollution impacting the streams. Seeing as the streams mostly run through cultivated landscape the main source of pollution is most likely runoff from agriculture leading to increased organic sedimentation loads. Many of the streams additionally lack supporting riparian vegetation, thus increased erosion risk. As seen in Leiråa, Stubbekken and Rossvollbekken the stream course is heavily characterized by erosion on the stream banks (Table 1), leading to increased sedimentation loads, which in turn affect the ecological condition. The abundant presence of the species *Bäetis rohdani* indicate that there is little degree of acidification impacting the streams, because *Bäetis rohdani* have low tolerance to acidification (Bergan et al., 2007).

The last objective of this study was to examine what determines the variation in ASPT index scores. The results showed that distance to fjord and fish densities had a significant effect on ASPT index scores. This is in correlation with previous studies conducted by Hol (2018), who found similar results. Further, there was a difference in ASPT index scores between the upper and lower reaches of the tributaries, however, this difference was not statistically significant. The ecological state tended to be slightly better in the upper reaches of the streams, and in the upper part of the watershed. An explanation for this is that the upper reaches mostly drains through forested areas and less impacted areas by human activities. Lower reaches have a tendency to accumulate more organic materials closer to the main river (Hol, 2018). The hydrological regime in the upper reaches are slightly better and closer to a natural state, which contributes to a much coarser substratum. This supports a higher diversity of macroinvertebrates (Barnes et al., 2013). Likewise, in the upper part of the watershed, the streams are less affected by human activities, and to higher extent runs through forested areas with less agriculture activity. The streams further up the watershed tend to have greater riparian vegetation.

Streams with higher densities of salmonid fishes, tended to have higher ASPT score. However, due to the high mobility of salmonids and that they are known to show avoidance behaviour to unfavourable habitat conditions, there are no evidence that the presence of salmonids in itself increase the ecological conditions. Another explanation for this is that salmonid fish tend to choose streams with better ecological conditions, and therefore streams with higher ecological condition have higher densities of salmonid fish.

The ASPT index between the years 2017 and 2018 indicated little annual change in ecological state. Only one of the streams achieved higher ecological condition than the previous year and one stream had unchanged condition. However, most of the streams gained worse ecological condition than the previous year. This suggest that there either have been episodes of pollution, or alterations in the catchment impacting the streams condition. On the other hand, the samples collected in 2017 might be taken in different location within the streams, therefore yield different ASPT score. Samples collected from Skjørdalsbekken was taken further downstream then the samples taken in 2017. Skjørdalsbekken drains mostly through intense agricultural areas downstream, whilst upstream drains mainly through forest. According to the results, the ASPT index is somewhat higher in the upper reaches of a stream, this might explain why Skjørdalsbekken had good ecological state in 2017, and moderate ecological state in 2018, because the samples were only taken in the upper reaches in 2017.

The summer period (May-August) of 2018 was the fourth driest summer in Norway since the 1900s, with temperatures 3.1 °C above normal conditions. The decreased precipitation and high temperatures resulted in decreased discharge to rivers and streams (Skaland et al., 2019). Reduced discharge results in low water velocity, water depth and channel width of the streams. Changes in abiotic factors can affect the biodiversity in streams. Reduced water velocity in streams results in decrease of benthic macroinvertebrate richness, due to decrease in habitat availability (Dewson et al., 2007). A study conducted by Lessard and Hayes (2003) found that increased temperatures in streams resulted in a shift in macroinvertebrate community structure. This might be a possible explanation for the annual difference in benthic macroinvertebrates community structure between 2017 and 2018, as the water flow in the streams was severally reduced, due to the high temperatures and low precipitation in the summer months of 2018.

4.4 Study limitations

Sampling of macroinvertebrates should be undertaken in two sampling rounds (one during spring, and one during late summer) to obtain a representative collection of macroinvertebrates with seasonal changes. This study did only have one sampling round in late October to early November. Therefore, seasonal changes have not been covered by this study. According to Bergan et al. (2007) the best period of sampling is during March and October, when macroinvertebrates are the most abundant, and have favourable sizes for easier identification. Benthic macroinvertebrates are not evenly distributed across a stream. Certain habitats and patches may contain higher abundances than others. The kick-sampling method might not reach all the different macrohabitats and hence fail to obtain a representative picture of the macroinvertebrate community, and therefore the data can be misleading (Feeley et al., 2012).

Inexperience in identification of macroinvertebrate species may potentially be a weakness. However, the ASPT index the species are determined to family level, and the probability of wrong identification are small. Moreover, determination to species level have a higher likelihood of incorrect identification, hence this may influence the ordination and diversity data. Then again, the species list was checked by an experienced identifier of macroinvertebrates, therefore this is of smaller concerns.

The habitat characteristics were based on visual estimation, and therefore are a potential source of error, in addition the habitat was measured by three different people, and differences in visual estimation must be considered.

In addition to sampling bioindicators, measurements of chemical and physical variables of the streams should be conducted. Thermometers were placed in the streams in this study. However, these were not retrieved after the field study, due to failure to place GPS points on the location, and hence not found.

4.5 Further management for conservation

Further restoration of the tributaries of Verdal river should prioritize reduction of organic sedimentation loads resulting from agricultural and urban runoff. The most effective restoration measure in reducing sedimentation loads and agricultural runoff, is to re-establish riparian vegetation (Kaase & Katz, 2012). Riparian vegetation influences the stream water quality by controlling the nutrient runoff to adjacent streams. In addition, riparian vegetation contributes to stabilize soils resulting in less erosion risks (Dosskey et al., 2010). Therefore, it is important to re-establish riparian vegetation where this is lacking. Palmer et al. (2010) concluded that the restoration of water quality, connectivity and riparian vegetation are important to further restoration of benthic macroinvertebrates, as they respond to habitat heterogeneity when larger scale conditions are in order (i.e. quality of water and flow, riparian conditions) (Palmer et al., 2010).

Previous studies have shown that restoration intensity is important to successful recovery of macroinvertebrates (Li et al., 2018). Further restoration measures should be undertaken on a smaller scale over a longer stretch of the stream to create complex substrates, including addition of boulders and placement of groins. Placement of groins have proven effective in recovery of macroinvertebrates diversity (Li et al., 2018). Further adjustments of culverts to improve the ability of macroinvertebrate dispersal up- and downstream and increased connectivity, are recommended.

Another restoration measure needed in the tributaries of Verdal river is cleaning the stream course of garbage. During the study a lot of garbage including plastic, car batteries, and bottles was found along the stream course. This should be removed to avoid further plastic pollution and dangerous chemical spills into the streams. Stubbekken showed signs of iron precipitation by a red/orange colour of the stream bed. Oxidized iron is concentrated, and harmful to organisms sensitive to high concentrations (Hol, 2018).

To understand the effects of restoration, and seasonal changes in the macroinvertebrate assemblages in the tributaries of Verdal river, long-time monitoring of bioindicators are needed.

5 Conclusions

In conclusion, the restoration measures preformed in tributaries of Verdal river, have no effect on macroinvertebrate assemblages, as to yet. Possible explanations for this result are that the intensity of the restoration measures undertaken are too low to influence macroinvertebrate assemblages, and that the type of restoration does not affect macroinvertebrates (connectivity and substrate placement). The recolonization from adjacent source populations need longer time to recover the restored sections, stressing the need for long-time monitoring of the effectiveness of restoration measures. Allochthonous production appears to influence the macroinvertebrate community structure the most out of the environmental variables. Increased allochthonous inputs into the streams positively correlates with pollution tolerant species, indicating that sedimentation loads are important determinants of macroinvertebrate assemblages.

The regained presence of salmonids in some streams may have an impact on macroinvertebrate community structure. However, this may be due to salmonids preference to habitat ecological conditions. Further, there were no effect of fish densities on macroinvertebrate diversity. High dispersal and colonisation rates are a possible explanation for little effect on macroinvertebrate diversity. Moreover, macroinvertebrates may be highly adapted to presence of salmonids, and therefore show no influence of changes in fish densities. Another explanation might be that the macroinvertebrate community in the tributaries of Verdal river are determined by bottom-up, rather than top-down forces.

The overall ecological state of the tributaries of Verdal river are generally poor. Organic inputs from agriculture and urban sources are believed to be the main explanation to the poor ecological condition of the streams. Acidification appears to not be a problem, due to the high abundance of the low acidification tolerant species *Bäetis rohdani*. Distance to the fjord, distance to the main river and fish densities seems to be the main determinants of ASPT index. The ASPT index are higher with increasing distance to the fjord, and increased distance to the main river. Indicating that there is an ASPT index gradient in the streams.

Recommended future restoration measures include re-establishment of riparian vegetation, where this is lacking, increase the intensity of the in-stream restoration measures, and clean-up of garbage from the stream course. Long-time monitoring is needed to fully understand the responses by macroinvertebrates to in-stream measures.

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7 Appendix

7.1 Appendix 1

Overview of the ASPT index scores for each family (Anonym, 2013).

Hovedgrupper	Familier	Verdi
Døgnfluer	Siphlonuridae, Heptageniidae, Leptophlebiidae Ephemerellidae, Potamanthidae, Ephemeridae	10
Steinfluer	Taeniopterygidae, Leuctridae, Capniidae, Perlodidae, Perlidae, Chloroperlidae	10
Teger	Aphelocheridae	10
Vårfluer	Phryganeidae, Molannidae, Beraeidae, Odontoceridae, Leptoceridae, Goeridae, Lepidostomatidae, Brachycentridae, Sericostomatidae	10
Kreps	Astacidae	8
Øyenstikkere	Lestidae, Agriidae, Gomphidae, Cordulegasteridae, Aeshnidae, Corduliidae, Libelluiidae	8
Vårfluer	Philopotamidae	8
Døgnfluer	Caenidae	7
Steinfluer	Nemouridae	7
Vårfluer	Rhyacophilidae, Polycentropidae, Limnephilidae	7
Snegler	Neritidae, Viviparidae, Ancylidae	6
Vårfluer	Hydroptilidae	6
Muslinger	Unionidae	6
Krepsdyr	Corophiidae, Gammaridae	6
Øyenstikkere	Platycnemididae, Coenagriidae	6
Teger	Mesoveliidae, Hydrometridae, Gerridae, Nepidae, Naucoridae, Notonectidae, Pleidae, Corixidae	5
Biller	Haliplidae, Hygrobiidae, Dytiscidae, Gyrinidae, Hydrophilidae, Clambidae, Helodidae, Dryopidae, Elmidae, Chrysomelidae, Curculionidae	5
Vårfluer	Hydropsychidae	5
Stankelbein/Knott	Tipulidae, Simuliidae	5
Flatormer	Planariidae, Dendrocoelidae	5
Døgnfluer	Baetidae	4
Mudderfluer	Sialidae	4
Igler	Piscicolidae	4
Snegler	Valvatidae, Hydrobiidae, Lymnaeidae, Physidae, Planorbidae	3
Småmuslinger	Sphaeriidae	3
Igler	Glossiphoniidae, Hirudidae, Erpobdellidae	3
Ferskvannsasell	Asellidae	3
Fjærmygg	Chironomidae	2
Fåbørstemark	Oligochaeta (hele klassen)	1

7.2 Appendix 2

Complete species list with modified codes for the ordination.

Order	Family	Species	Code
Mayflies (Ephemeroptera)	Baetidae	*Bäetis sp.*	BaeSp.
	Baetidae	*Bäetis rhodani*	BaeRho
	Baetidae	*Bäetis niger*	BaeNig
	Baetidae	*Bäetis fuscatus*	BaeFus
	Baetidae	*Bäetis muticus*	BaeMut
	Baetidae	*Centroptilum sp.*	BaeCen
	Baetidae	*Procloeon bifidum*	BaePro
Caddisflies (Trichoptera)	Rhyacophilidae	*Rhyacophila nubila*	RhyNub
	Rhyacophilidae	*Rhyacophila fasciata*	RhyFas
	Polycentropodidae	*Plectrocnemia conspersa*	PleCon
	Glossosomatinae	*Glossosoma sp.*	GlosSp.
	Goeridae	*Silo palipus*	SilPal
	Limnephilidae		Limne
	Sericostomaidae	*Sericostoma personatum*	SeriPers
	Lepidostomatidae		Lepido
Stoneflies (Plecoptera)	Perlodiae	*Diura nanseni*	DiuNan
	Chloroperlidae	*Siphonoperla burmeisteri*	SipBur
	Perlodiae	*Isoperla grammatica*	IsoGra
	Perlodiae	*Isoperla obscura*	IsoObs
	Nemouridae	*Nemoura cinerea*	NemCin
	Nemouridae	*Amphinemura borealis*	AmpBor
	Capniidae	*Capina bifrons*	CapBif
	Taeniopterygidae	*Brachyptera risi*	BraRisi
	Capniidae	*Capnopsis schilleri*	CapSchi
	Leuctridae	*Leuctra hippopus*	LeuHip
Diptera (True flies)	Simuliidae		Simuli
	Chironomidae	*Tanypodinae sp.*	ChirTan
	Chironomidae	*Orthocladiinae*	ChirOr
	Chironomidae	*Chironominae sp.*	ChirChir
	Ceratopogonidae	*Dasyheleinae sp.*	DasySp.
	Ceratopogonidae	*Ceratopogoninae sp.*	CeraSp.
	Pediciidae	*Dicranota sp.*	DicraSp.
	Tabanini		Taba
	Psychodidae		NeoSp.
	Limoniidae	*Neolimnomyia sp.*	Psycho
	Limoniidae	*Scleroprocta sp.*	ScleSp.
	Limoniidae	*Helius sp.*	HeliSp.
	Stratiomyidae	*Beris clavipes*	BerCla
	Tipulidae		Tipuli
	Limoniidae		Limon
	Empididae	*Chelifera sp.*	CheliSp.
	Tabanidae	*Tabaninae*	Tabani
	Limoniidae	*Eloeophila sp.*	EloeSp.

Diptera	Culicidae pupae		CuliPup
Annelida	Oligochaeta		Oligo
Coleroptera (Beetles)	Dytiscidae	*Agabus type*	DytAga
	Curculionidae		Curcul
	Haliplidae		Halipli
	Hydraenidae	*Hydraena sp.*	HydraSp.
Collembola			Collem
Bivalvia	Sphaeriidae		Sphae
Gastropoda	Lymnaeidae	*Radix baltica*	RadBal
	Planorbidae		Plano
Megaloptera	Sialidae	*Sialis lutaria*	SiaLut
Acari (mites)			Acari
Amphipoda	Gammaridae	*Gammarus lacustris*	GamLacu

7.3 Appendix 3

Overview of the raw data of the macroinvertebrate sampled in 2018, for complete species names see appendix 2. KOR = Korsådalsbekken, FOL = Follobekken, BRO = Brokskittbekken, ROS=Rossvollbekken, SKJ=Skjørdalsbekken, BJO = Bjørkbekken, LEI=Leiråa, KVE= Kvellstadbekken, EKL=Eklobekken, STU=Stubbekken, LUN= Lundskinnbekken, and HYL= Hyllbekken.

SampleID	BaeSp.	BaeRho	BaeNig	BaeFus	BaeMut	BaeCen	BaePro	RhyNub	RhyFas	PleCon
HYL-1-18	2	62	8	0	0	0	0	0	0	2
HYL-1-18	4	216	8	0	16	0	0	4	0	0
HYL-2-18	12	210	2	0	8	0	0	14	0	0
HYL-2-18	4	292	16	0	24	0	0	8	0	0
STU-1-18	0	0	0	0	0	0	0	0	0	0
STU-1-18	0	12	0	0	0	0	0	0	0	0
STU-2-18	0	0	0	0	0	0	0	2	0	0
STU-2-18	0	8	0	0	0	0	0	0	0	0
KVE-1-18	6	112	0	0	0	0	0	6	0	0
KVE-1-18	8	204	0	0	0	0	0	12	0	0
KVE-2-18	0	100	0	0	0	0	0	32	0	0
KVE-2-18	0	126	0	0	2	0	0	8	0	0
BRO-1-18	0	0	0	0	0	0	0	0	0	0
BRO-1-18	0	2	0	0	0	0	0	0	0	0
BRO-4-18	0	12	0	0	0	0	0	0	0	0
BRO-4-18	0	14	2	0	0	0	0	0	0	2
SKJ-1-18	0	0	0	0	0	0	0	0	4	0
SKJ-1-18	4	28	0	0	0	0	0	0	0	0
SKJ-6-18	0	92	8	0	0	0	0	0	0	0
SKJ-6-18	0	192	28	0	0	0	0	0	0	0
BJO-1-18	0	44	0	0	0	0	0	8	0	0
BJO-1-18	0	76	0	0	0	0	0	0	0	0
BJO-4-18	2	116	4	0	0	0	0	6	0	0
BJO-4-18	0	137	1	0	12	0	0	2	0	0

FOL-1-18	0	0	0	0	0	0	0	0	0	0
FOL-1-18	0	0	0	0	0	0	0	0	0	0
FOL-4-18	0	44	24	0	0	0	0	0	0	0
FOL-4-18	1	129	86	1	18	1	0	1	0	1
ROS-1-18	0	2	0	0	0	0	0	0	0	0
ROS-1-18	0	28	0	0	0	0	0	0	0	0
ROS-2-18	0	44	0	0	0	0	0	4	0	0
ROS-2-18	0	36	0	0	0	0	0	0	0	0
EKL-1-18	2	64	8	0	0	0	0	4	0	0
EKL-1-18	1	251	5	0	0	8	1	0	0	0
EKL-2-18	4	476	0	0	0	0	0	0	0	0
EKL-2-18	0	240	8	0	0	0	0	8	0	0
LEI-1-18	0	4	0	0	0	0	0	0	0	0
LEI-1-18	0	12	0	0	0	0	0	0	0	0
LEI-2-18	0	5	1	0	0	0	0	1	0	0
LEI-2-18	0	4	0	0	0	0	0	0	0	0
LUN-1-18	0	8	0	0	6	0	0	0	0	0
LUN-1-18	0	1	1	0	0	0	0	0	0	0
LUN-2-18	0	32	8	0	0	0	0	2	0	2
LUN-2-18	0	24	32	0	0	0	0	0	0	0
KOR-1-18	0	5	0	0	0	1	0	1	0	0
KOR-1-18	0	11	2	0	0	0	0	0	0	1
KOR-2-18	0	40	0	0	0	0	0	0	0	0
KOR-2-18	0	4	0	0	0	0	0	0	0	0

SampleID	GlosSp.	SilPal	Limne	SeriPers	Lepido	DiuNan	SipBur	IsoGra	IsoObs	NemCin	AmpBor
HYL-1-18	0	0	0	0	0	0	0	0	0	6	0
HYL-1-18	0	0	0	0	0	0	0	0	0	12	0
HYL-2-18	0	0	0	0	0	2	0	4	0	22	0
HYL-2-18	0	4	4	0	0	0	0	0	0	12	0
STU-1-18	0	0	0	0	0	0	0	0	0	0	0
STU-1-18	0	0	0	0	0	0	0	0	0	0	0
STU-2-18	0	0	0	0	0	0	0	0	0	2	0
STU-2-18	0	0	0	0	0	0	0	0	0	2	0
KVE-1-18	0	0	0	0	0	0	0	8	0	36	0
KVE-1-18	0	0	0	0	0	0	0	12	0	36	0
KVE-2-18	0	0	0	0	0	0	0	8	0	4	0
KVE-2-18	0	0	0	0	0	0	0	4	0	4	0
BRO-1-18	0	0	0	0	0	0	0	0	0	0	0
BRO-1-18	0	0	0	0	0	0	0	0	0	0	0
BRO-4-18	0	0	0	0	0	0	0	0	0	4	0
BRO-4-18	0	0	0	0	0	0	0	0	0	4	0
SKJ-1-18	0	4	0	0	0	0	0	0	0	0	0
SKJ-1-18	0	12	0	0	0	0	0	0	0	4	0

SKJ-6-18	0	4	0	0	0	0	4	4	0	12	4
SKJ-6-18	4	0	4	0	0	0	0	4	0	4	4
BJO-1-18	0	0	0	0	0	0	0	0	0	4	0
BJO-1-18	0	0	0	0	0	0	0	0	0	0	0
BJO-4-18	0	2	0	0	0	0	2	0	0	2	0
BJO-4-18	0	4	3	0	0	0	0	2	0	0	3
FOL-1-18	0	0	0	0	0	0	0	0	0	0	0
FOL-1-18	0	0	0	0	0	0	0	0	0	0	0
FOL-4-18	0	0	0	0	0	0	0	0	0	2	0
FOL-4-18	0	0	0	7	0	0	0	4	1	24	0
ROS-1-18	0	2	0	0	4	0	0	0	0	2	0
ROS-1-18	0	0	0	0	0	0	0	0	0	4	0
ROS-2-18	0	0	0	0	0	0	0	0	0	4	0
ROS-2-18	0	0	0	0	0	0	0	0	0	0	0
EKL-1-18	0	0	0	0	0	0	0	0	0	8	0
EKL-1-18	1	0	1	0	0	0	0	0	0	16	1
EKL-2-18	0	0	0	0	0	0	0	0	0	0	0
EKL-2-18	0	0	0	0	0	0	0	0	0	8	0
LEI-1-18	0	0	0	0	0	0	0	0	0	0	0
LEI-1-18	0	0	0	0	0	0	0	0	0	16	0
LEI-2-18	0	0	0	0	0	0	0	0	0	0	0
LEI-2-18	0	0	0	0	0	0	0	2	0	0	0
LUN-1-18	0	0	0	0	0	0	0	0	0	10	0
LUN-1-18	0	0	0	0	0	0	0	0	0	0	0
LUN-2-18	0	0	0	0	0	0	0	0	0	18	0
LUN-2-18	0	0	0	0	0	0	0	0	0	128	0
KOR-1-18	0	0	1	0	0	0	0	0	0	0	0
KOR-1-18	0	0	6	0	0	0	0	0	0	8	0
KOR-2-18	0	0	4	0	0	0	0	0	0	72	0
KOR-2-18	0	0	8	0	0	0	0	0	0	0	0

SampleID	CapBif	BraRisi	CapSchi	LeuHip	Simuli	ChirTan	ChirOr	ChirChir	Oligo	DasySp.	CeraSp.
HYL-1-18	2	0	0	0	8	12	0	0	4	0	4
HYL-1-18	0	0	0	0	16	12	0	0	20	0	0
HYL-2-18	0	4	0	0	2	6	0	0	4	0	4
HYL-2-18	0	0	0	0	16	4	0	0	8	0	0
STU-1-18	0	0	0	0	0	6	0	0	10	0	0
STU-1-18	0	0	0	0	0	14	0	0	4	0	0
STU-2-18	0	0	0	0	2	10	0	0	2	0	0
STU-2-18	0	2	0	0	4	6	0	0	6	0	0
KVE-1-18	0	10	0	0	0	10	0	0	22	2	2
KVE-1-18	0	32	0	0	8	4	0	0	60	0	4
KVE-2-18	0	20	0	0	4	4	0	0	12	0	0
KVE-2-18	0	20	0	2	0	0	0	0	4	0	0

BRO-1-18	0	0	0	0	2	10	0	2	62	0	2
BRO-1-18	2	0	0	0	4	14	0	0	58	0	0
BRO-4-18	0	0	0	0	6	8	0	0	2	0	0
BRO-4-18	0	0	0	0	12	10	0	0	6	0	0
SKJ-1-18	0	0	0	0	0	4	0	0	4	0	0
SKJ-1-18	0	0	0	0	20	4	0	0	16	0	4
SKJ-6-18	0	24	12	0	20	4	0	0	8	0	8
SKJ-6-18	8	24	4	4	0	4	0	0	104	0	0
BJO-1-18	4	0	0	0	4	28	0	0	36	0	0
BJO-1-18	0	4	0	0	12	12	0	0	48	0	0
BJO-4-18	0	0	0	0	0	10	0	0	6	0	0
BJO-4-18	0	1	0	7	4	2	0	0	11	0	0
FOL-1-18	0	0	0	0	0	12	0	0	36	0	0
FOL-1-18	0	0	0	0	2	4	0	0	6	0	0
FOL-4-18	0	0	0	0	6	30	0	0	12	0	10
FOL-4-18	7	0	0	5	34	39	0	0	18	0	8
ROS-1-18	0	0	0	0	0	14	0	0	10	2	0
ROS-1-18	0	0	0	0	8	80	0	0	56	0	0
ROS-2-18	0	0	0	0	20	136	0	0	164	0	8
ROS-2-18	0	0	0	0	0	12	16	0	80	0	0
EKL-1-18	12	0	0	0	20	34	0	6	18	0	0
EKL-1-18	8	0	0	0	41	9	3	2	3	0	0
EKL-2-18	20	0	0	0	124	24	0	0	88	0	4
EKL-2-18	24	0	0	0	56	28	0	0	152	0	8
LEI-1-18	0	0	0	0	20	44	0	4	160	0	0
LEI-1-18	12	0	0	0	348	72	0	0	224	0	0
LEI-2-18	6	0	0	0	1	15	0	0	19	0	0
LEI-2-18	0	0	0	0	0	24	0	0	44	0	4
LUN-1-18	4	0	0	2	14	32	0	0	22	0	2
LUN-1-18	0	0	0	0	6	8	0	0	9	0	0
LUN-2-18	0	6	0	0	36	54	0	8	148	0	0
LUN-2-18	8	4	0	0	100	192	0	24	20	0	0
KOR-1-18	0	0	0	0	1	13	0	1	6	0	0
KOR-1-18	0	0	0	0	5	9	0	1	19	0	0
KOR-2-18	0	0	0	0	560	68	0	0	256	0	0
KOR-2-18	0	0	0	0	4	208	0	0	200	0	0

SampleID	DicraSp.	Taba	NeoSp.	Psycho	ScleSp.	HeliSp.	BerCla	Tipuli	Limon	CheliSp.	Tabani	EloeSp.
HYL-1-18	2	4	0	2	0	0	0	0	0	0	0	0
HYL-1-18	4	0	0	0	0	0	0	0	0	0	0	0
HYL-2-18	4	0	0	8	0	0	0	0	0	0	0	0
HYL-2-18	8	0	0	0	0	0	0	0	4	0	0	0
STU-1-18	0	0	0	0	0	0	0	0	2	0	0	0
STU-1-18	8	0	0	0	10	0	0	0	0	0	0	0
STU-2-18	0	0	0	0	6	0	0	0	0	0	0	0
STU-2-18	6	0	0	0	10	0	0	2	0	0	0	0

KVE-1-18	14	0	0	12	2	0	0	0	0	2	0	0
KVE-1-18	4	0	0	24	0	0	0	0	0	0	0	0
KVE-2-18	12	0	0	20	4	0	0	0	0	4	0	0
KVE-2-18	6	0	0	0	0	0	0	0	0	0	0	0
BRO-1-18	6	0	0	0	4	10	0	0	0	0	0	0
BRO-1-18	0	0	0	0	2	0	0	2	0	0	0	0
BRO-4-18	6	0	0	2	6	0	0	0	0	0	0	0
BRO-4-18	0	0	0	0	2	0	0	0	0	0	0	2
SKJ-1-18	8	0	0	0	0	0	0	0	0	0	0	0
SKJ-1-18	8	0	0	8	0	0	0	0	0	0	0	0
SKJ-6-18	4	0	0	24	0	0	0	4	0	0	0	0
SKJ-6-18	4	0	0	64	56	0	0	0	0	4	0	0
BJO-1-18	8	0	0	4	0	0	0	0	0	0	0	0
BJO-1-18	4	0	0	4	0	0	0	0	0	0	0	0
BJO-4-18	2	0	0	6	0	0	0	0	0	0	0	0
BJO-4-18	7	0	0	1	0	0	0	1	0	0	0	1
FOL-1-18	4	0	0	4	4	0	0	4	0	0	0	4
FOL-1-18	4	0	0	10	4	0	0	2	0	0	0	8
FOL-4-18	0	0	0	10	0	0	0	4	0	0	0	0
FOL-4-18	13	0	1	20	0	0	0	0	0	1	0	0
ROS-1-18	4	0	0	4	6	0	0	0	0	0	0	0
ROS-1-18	8	0	0	0	16	0	0	0	0	0	0	0
ROS-2-18	20	0	0	0	0	0	0	0	0	0	0	0
ROS-2-18	4	0	0	4	0	0	0	0	0	0	0	0
EKL-1-18	8	0	0	0	22	0	0	0	0	0	0	0
EKL-1-18	2	0	0	0	0	0	0	0	0	0	0	0
EKL-2-18	36	0	0	16	16	0	0	4	0	0	0	4
EKL-2-18	12	0	0	0	8	0	0	0	0	0	0	0
LEI-1-18	4	0	0	0	0	0	0	0	0	0	0	8
LEI-1-18	16	0	0	0	0	0	0	0	0	0	0	0
LEI-2-18	4	0	0	0	0	1	0	6	0	0	0	0
LEI-2-18	2	0	0	0	0	0	0	4	0	0	2	0
LUN-1-18	2	0	0	2	0	6	0	0	0	0	0	0
LUN-1-18	0	0	0	0	0	1	0	0	0	0	0	0
LUN-2-18	12	0	0	14	0	0	0	0	0	0	0	0
LUN-2-18	16	0	0	44	16	4	0	0	0	0	0	0
KOR-1-18	1	0	0	0	2	0	0	0	0	0	0	1
KOR-1-18	7	0	0	0	3	1	0	0	0	0	0	10
KOR-2-18	20	0	0	0	0	0	0	0	0	0	0	0
KOR-2-18	4	0	0	4	0	0	4	0	0	0	0	8

SampleID	CuliPup	DytAga	Curcul	Collem	Sphae	RadBal	Plano	Halipli	SiaLut	HydraSp.	Acari	GamLacu
HYL-1-18	0	0	0	0	0	0	0	0	0	0	0	0
HYL-1-18	4	0	0	0	0	0	0	0	0	0	0	4

HYL-2-18	0	0	2	0	0	0	0	0	0	0	0	0
HYL-2-18	0	0	0	0	0	0	0	0	0	0	0	0
STU-1-18	0	0	0	0	0	0	0	0	0	0	0	0
STU-1-18	0	0	0	0	0	0	0	0	0	0	0	0
STU-2-18	0	0	0	0	0	0	0	0	0	0	0	0
STU-2-18	0	0	0	0	0	0	0	0	0	0	0	0
KVE-1-18	0	0	0	0	0	0	0	0	0	0	0	0
KVE-1-18	0	0	0	0	0	0	0	0	0	0	0	0
KVE-2-18	0	0	0	0	0	0	0	0	0	0	0	0
KVE-2-18	0	0	0	0	0	0	0	0	0	0	0	0
BRO-1-18	0	0	0	0	0	0	0	0	0	0	0	0
BRO-1-18	0	0	0	0	0	0	0	0	0	0	0	0
BRO-4-18	0	0	0	0	0	0	0	0	0	0	0	0
BRO-4-18	0	0	0	0	0	0	0	0	0	0	0	0
SKJ-1-18	0	0	0	4	0	0	0	0	0	0	0	0
SKJ-1-18	0	0	0	4	4	0	0	0	0	0	0	0
SKJ-6-18	0	0	0	0	0	0	0	0	0	0	0	0
SKJ-6-18	0	4	0	0	0	0	0	0	0	0	0	0
BJO-1-18	0	0	0	0	0	0	0	0	0	0	0	0
BJO-1-18	0	0	0	0	0	0	0	0	0	0	0	0
BJO-4-18	0	0	0	0	0	0	0	2	0	0	0	0
BJO-4-18	0	0	0	0	0	0	0	0	0	1	0	0
FOL-1-18	0	0	0	0	0	16	0	0	0	0	0	0
FOL-1-18	0	0	0	0	0	2	0	0	0	0	0	0
FOL-4-18	0	0	0	0	0	0	0	0	0	0	0	0
FOL-4-18	0	0	0	0	0	6	0	0	1	2	1	0
ROS-1-18	0	0	0	0	0	0	0	0	0	0	0	0
ROS-1-18	4	0	0	0	0	0	0	0	0	0	0	0
ROS-2-18	0	0	0	0	0	0	0	0	0	0	0	0
ROS-2-18	0	0	0	0	0	0	0	0	0	0	0	0
EKL-1-18	0	0	0	0	0	0	0	0	0	0	0	0
EKL-1-18	0	0	0	0	0	0	0	0	0	0	0	0
EKL-2-18	4	0	0	0	24	0	0	0	0	0	0	0
EKL-2-18	0	0	0	0	4	0	0	0	0	0	0	0
LEI-1-18	0	0	0	0	0	0	0	0	0	0	0	0
LEI-1-18	0	0	0	0	0	0	0	0	0	4	0	0
LEI-2-18	0	3	0	0	2	0	0	0	0	0	0	0
LEI-2-18	0	2	0	0	0	0	0	0	0	0	0	0
LUN-1-18	0	0	0	0	0	0	0	0	0	0	0	0
LUN-1-18	0	0	0	0	0	0	0	0	0	0	0	0
LUN-2-18	0	0	0	0	0	0	0	0	0	0	0	0
LUN-2-18	0	0	0	0	0	0	0	0	0	0	0	0
KOR-1-18	0	0	0	0	0	0	1	0	0	0	0	5
KOR-1-18	0	0	0	0	4	1	0	0	0	0	0	13
KOR-2-18	0	0	0	0	0	32	0	0	0	0	0	104
KOR-2-18	0	4	0	0	8	0	0	0	0	0	0	40

7.4 Appendix 4

Overview of habitat characteristics from the study streams. KOR = Korsådalsbekken, FOL = Follobekken, BRO = Brokskittbekken, ROS=Rossvollbekken, SKJ=Skjørdalsbekken, BJO = Bjørkbekken, LEI=Leiråa, KVE= Kvellstadbekken, EKL=Eklobekken, STU=Stubbekken, LUN= Lundskinnbekken, and HYL= Hyllbekken.

				Depth (cm)					
Stream	**Transect**	**Width**	**Length (m)**	**10 %**	**25 %**	**50 %**	**75 %**	**90 %**	**Mean depth**
KOR-1	1	2,3	65	5	10	15	10	5	0
	5	2,3	65	20	25	20	15	10	0
KOR-2	1	0,9	82	5	5	7	2	5	0,048
	5	4,3	82	8	8	12	15	10	0,106
FOL-1	1	2,4	47,6	3	3	2	5	5	0,036
	5	1,4	47,6	2	10	15	10	5	0,084
FOL-4	1	3,2	43	2	1	5	3	7	0,036
	5	3,4	43	8	5	5	10	5	0,066
BRO-1	1	1,4	58	5	15	15	15	5	0,11
	5	1,8	58	1	1	3	1	3	0,018
BRO-4	1	2,1	50	1	5	10	5	2	0,046
	5	1,8	50	10	20	20	15	10	0,15
ROS-1	1	1,65	54	5	5	5	1	1	0,034
	5	1,4	54	30	25	20	15	5	0,19
ROS-2	1	2,1	32,7	1	1	5	2	3	0,024
	5	2,2	32,7	20	20	30	20	15	0,21
SKJ-1	1	2,4	41	18	21	22	24	13	0,196
	5	1,5	41	32	33	32	30	21	0,296
SKJ-6	1	1,6	50	3	5	7	5	4	0,048
	5	1,3	50	5	15	30	20	10	0,16
BJO-1	1	0,9	53,5	10	15	15	15	1	0,112
	5	2,5	53,5	10	20	30	30	15	0,21
BJO-4	1	2,9	63	8	2	1	1	2	0,028
	5	1,45	63	5	8	8	5	5	0,062
LEI-1	1	3,05	50	2	10	18	15	5	0,1
	5	4,1	50	6	19	12	12	5	0,108
LEI-2	1	1,8	49	34	35	39	47	42	0,394
	5	2,9	49	6	10	16	16	11	0,118
KVE-1	1	2,1	60	10	11	13	5	2	0,082
	5	1,8	60	3	14	15	14	12	0,116
KVE-2	1	2,1	62	11	15	17	13	4	0,12
	5	2,1	62	5	19	15	9	6	0,108
EKL-1	1	2,5	39	6	11	11	9	5	0,084
	5	2,65	39	1	29	15	10	5	0,12
EKL-2	1	1,45	65	4	7	22	18	9	0,12
	5	1,8	65	5	6	9	10	8	0,076

STU-1	1	1,05	48	14	20	15	9	2	0,12
	5	1,2	48	1	3	4	9	6	0,046
STU-2	1	2,1	62	3	5	8	10	5	0,062
	5	2,1	62	3	4	5	22	25	0,118
LUN-1	1	1,1	35	8	6	15	13	7	0,098
	5	2,7	35	8	30	35	28	19	0,24
LUN-2	1	1,2	47	6	7	9	10	10	0,084
	5	1,1	47	10	16	24	24	32	0,212
HYL-1	1	4,85	79	12	9	0	4	8	0,066
	5	1,25	79	4	8	3	5	2	0,044

		Substrate (%)					
Stream	**Transect**	**0- 2 mm**	**2- 20 mm**	**20- 100 mm**	**100- 250 mm**	**>250**	**mean sub**
KOR-1	1	45	35	10	0	0	10,3
	5	100	0	0	0	0	1
KOR-2	1	0	5	80	10	5	97,3
	5	5	30	50	5	10	104,6
FOL-1	1	10	30	50	0	10	95,9
	5	80	10	10	0	0	7,9
FOL-4	1	5	40	50	5	0	43,2
	5	5	20	25	40	10	149,75
BRO-1	1	10	0	10	20	60	416,1
	5	5	50	40	5	0	38,3
BRO-4	1	50	40	10	0	0	10,9
	5	50	20	0	0	30	190,2
ROS-1	1	20	20	30	30	0	72,9
	5	50	45	0	5	0	14,2
ROS-2	1	15	10	40	30	5	109
	5	90	0	5	5	0	12,65
SKJ-1	1	30	30	40	0	0	27,6
	5	40	20	40	0	0	26,6
SKJ-6	1	0	10	70	20	0	78,1
	5	20	10	10	10	0	24,8
BJO-1	1	0	5	0	45	50	391,8
	5	10	80	5	5	0	20,65
BJO-4	1	5	25	60	10	0	56,3
	5	5	35	55	5	0	45,65
LEI-1	1	0	10	40	30	10	140,1
	5	0	5	15	50	30	284,55
LEI-2	1	34	35	39	47	42	372,34
	5	6	10	16	16	11	107,51
KVE-1	1	15	60	25	0	0	21,75
	5	10	20	70	0	0	44,3

KVE-2	1	20	35	40	5	0	36,8
	5	0	15	60	20	5	103,9
EKL-1	1	6	11	11	9	5	54,87
	5	1	29	15	10	5	60,95
EKL-2	1	50	30	20	0	0	15,8
	5	15	15	67	3	0	47,25
STU-1	1	30	10	60	0	0	37,4
	5	0	0	10	60	30	298,5
STU-2	1	20	60	20	0	0	18,8
	5	90	10	0	0	0	2
LUN-1	1	5	5	20	40	10	145,1
	5	5	20	40	20	5	92,5
LUN-2	1	3	7	40	50	0	112,3
	5	30	30	40	0	0	27,6
HYL-1	1	5	25	40	20	10	124,3
	5	0	10	20	20	50	360,6

Stream	Transect	Canopy Water	Canopy bank	Woody debris	Pools	Velocity	Algea	Moss
KOR-1	1	100	92	26	4	0,15	0	0
	5	100	92	26	4	0,15	0	0
KOR-2	1	35	33	0	1	0,3	16	0
	5	0	12	0	1	0,07	50	0
FOL-1	1	90	92	29	3	0,4	16	0
	5	75	83	29	3	0,25	0	0
FOL-4	1	40	83	19	3	0,5	16	0
	5	75	63	19	3	0,15	16	16
BRO-1	1	95	92	8	4	0,25	16	0
	5	95	92	8	4	0,5	16	0
BRO-4	1	95	92	10	7	0,25	0	0
	5	95	92	10	7	0,12	0	0
ROS-1	1	100	92	12	6	0,3	0	0
	5	100	92	12	6	0,05	0	0
ROS-2	1	95	92	4	3	0,3	0	0
	5	100	92	4	3	0,01	0	0
SKJ-1	1	40	0	0	0	0,8	0	0
	5	0	33	0	0	0,35	0	0
SKJ-6	1	90	83	9	4	0,4	16	0
	5	90	92	9	4	0,2	0	0
BJO-1	1	100	92	12	5	0,25	0	0
	5	100	92	12	5	0,01	0	0
BJO-4	1	100	92	13	4	0,25	0	0
	5	75	83	13	4	0,25	0	0
LEI-1	1	100	83	7	4	0,6	16	0

	5	100	83	7	4	0,5	16	0
LEI-2	1	95	92	7	5	0,45	0	0
	5	0	0	7	5	0,6	0	0
KVE-1	1	60	63	0	0	0,6	16	0
	5	70	83	0	0	0,5	16	0
KVE-2	1	100	92	2	2	0,4	0	0
	5	0	0	2	2	0,5	0	0
EKL-1	1	90	83	3	5	0,6	0	0
	5	60	63	3	5	0,1	0	16
EKL-2	1	30	12	7	8	0,5	0	0
	5	100	83	7	8	0,5	0	0
STU-1	1	100	92	8	2	0,5	0	0
	5	45	33	8	2	0,9	0	0
STU-2	1	100	92	8	2	0,3	0	0
	5	100	92	8	2	0,2	0	0
LUN-1	1	90	63	5	2	0,4	1	0
	5	70	63	5	2	0,35	1	0
LUN-2	1	100	83	2	2	0,5	16	0
	5	90	83	2	2	0,5	0	0
HYL-1	1	0	12	0	1	0,5	50	16
	5	0	0	0	1	0,7	50	16

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