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# **16 Years of River Restoration: Effects of Restoration Measures on Macroinvertebrates and Salmonids in Bognelv, Northern Norway**

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## Preface

This thesis was written at the Faculty of Environmental Sciences and Natural Resource Management (MINA) at Norwegian University of Life Sciences (NMBU). This work completes our master's degree in Natural Resource Management.

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## Summary

Lotic ecosystems worldwide are being degraded by human activities, with severe consequences for stream biota. River restoration has gained momentum over the last decade, but many projects lack the long-term monitoring needed to efficiently evaluate restoration outcomes.

One river restoration project operating with a long-term perspective is the project in Bognelv in northern Norway. The river was channelized and secured against erosion and flooding between 1930 and 1990, with a resulting decline in fish stocks of Atlantic salmon (*Salmo salar*), brown trout (*Salmo trutta*) and Arctic charr (*Salvelinus alpinus*). Restoration began in 2006 and is now going on its 16<sup>th</sup> year, the latest measures being conducted in 2019.

This thesis is the eighth study investigating effects of river restoration on stream biota in Bognelv. Earlier studies show that Atlantic salmon and brown trout have responded well to the restoration measures, while Arctic charr have been absent from studies since 2013. Macroinvertebrates have been sampled in 2015, 2019 and 2021, to better understand how the restoration process affects the river biota.

In August and September 2021, we registered environmental variables and conducted electrofishing and kick-sampling in Bognelv. We followed the study design of earlier studies with a total of 56 stations spread from the lower to the middle stretches of the river. Our analysis focused on three main effects; type of restoration measure, time since last restoration measure, and distance from estuary. Only one Atlantic salmon was caught during electrofishing, and was therefore excluded from analysis. Due to unusual small body sizes and the resulting impaired catchability, 0+ for brown trout were difficult to sample in 2021 and were also excluded from our analyses. Our results show a sharp decline in brown trout densities from previous years for the age classes included in our analyses, but an increase in macroinvertebrate abundance.

The effect of type of restoration measure on macroinvertebrate diversity was not statistically significant, but our results showed a tendency to favor weirs as the most successful measure. We found a significantly positive effect of side channels and channelized stations on macroinvertebrate abundance. We were not able to find a statistically significant effect of

type of measure on brown trout densities, but AIC model selection favored weirs and riparian modifications as the most successful measure.

The effect of time since last restoration measure on macroinvertebrates yielded statistically significant effects on abundance, but not on diversity. AIC model selection predicted a peak in abundance eight to ten years post-restoration. The effect of time on brown trout densities showed a decrease in the density the first ten years post-restoration, and a possible bottom-point being reached at about 14 years post-restoration.

The effect of distance from estuary did not prove statistically significant for diversity or abundance of macroinvertebrates. However, combined with type of measure and distance from estuary, the effect on macroinvertebrate abundance was significant. AIC model selection predicted a peak in abundance at three to four kilometers upstream in the river. For brown trout, predicted density was highest close to estuary.

Combined with results from earlier studies on restoration effects in Bognelv, this study provides valuable knowledge for future river restoration projects, as well as for the continued implementation of measures in Bognelv. Even so, more research on the effects of river biota in Bognelv is needed to properly understand the processes at play, and to further improve the ecological condition in the river.

## Sammendrag

Elvesystemer over hele verden er forringet av menneskelige aktiviteter, med alvorlige konsekvenser for elvebiotaen. Til tross for stort fokus på elverestaurering i løpet av de siste tiårene, mangler mange prosjekter den langsiktige overvåkingen som er nødvendig for å evaluere effektene av restaureringstiltakene. I Bognelv i Nord-Norge pågår det et restaureringsprosjekt med langsiktig overvåking. Mellom 1930 og 1990 ble elva kanalisert og sikret mot erosjon og flom, med en påfølgende nedgang i fiskebestandene av atlantisk laks (*Salmo salar*), sjørøret (*Salmo trutta*) og sjørøye (*Salvelinus alpinus*). I 2006 startet restaureringen av Bognelv og den har nå pågått i 16 år. De siste tiltakene ble utført i 2019.

Denne oppgaven er den åttende studien som undersøker effektene av elverestaurering på bunndyr og fisk i Bognelv. Tidligere studier viser at atlantisk laks og sjørøret har respondert godt på restaureringstiltakene, mens sjørøye ikke har vært fanget i Bognelv siden 2013. I 2015, 2019 og 2021 har bunndyr blitt innsamlet for å få en bedre forståelse av hvordan restaureringen har påvirket elvebiotaen. I august og september 2021 registrerte vi miljøvariabler, elfisket og tok sparkeprøver av bunndyr i Bognelv. Vi fulgte samme studiedesign som tidligere studier har brukt, med totalt 56 stasjoner spredt fra utløpet til midtre strekninger av elva. Vår analyse fokuserte på tre hovedeffekter: type restaureringstiltak, tid siden forrige restaureringstiltak, og avstanden fra elvemunningen. Atlantisk laks er ekskludert fra analysen vår, ettersom vi kun fikk én atlantisk laks under feltarbeidet vårt. Vi ekskluderte også 0+ sjørøret fra våre analyser, grunnet uvanlig små fiskestørrelser som påvirket fangbarheten. Våre resultater viser en kraftig nedgang i ørrettettheter sammenlignet med tidligere år, men en økning i forekomst av bunndyr.

Effekten av type restaureringstiltak på bunndyrdiversiteten var ikke statistisk signifikant, men våre resultater viste at terskler var det mest vellykkede tiltaket. Vi fant en signifikant og positiv effekt av åpning av sideløp og kanaliserte stasjoner på forekomsten av bunndyr. Vi fant ingen statistisk signifikant effekt av type tiltak på tetthetene av sjørøret, men AIC modellseleksjon favoriserte terskler og modifisering av kantsonen som de mest vellykkede tiltakene.

Effekten av tid siden forrige tiltak ga statistisk signifikant effekt på forekomst av bunndyr, men ikke på bunndyrdiversitet. AIC modellseleksjon predikerte et toppunkt for

bunndyrforekomster åtte til ti år etter forrige restaureringstiltak. Effekten av tid på ørrettettheter viste en nedgang i tetthet de første ti årene etter restaurering, og et mulig bunnpunkt rundt 14 år etter restaurering.

Effekten av avstand fra elvemunningen var ikke signifikant for verken diversitet eller forekomst av bunndyr. Vi fant derimot en signifikant effekt på bunndyrforekomsten når type restaureringstiltak og avstand fra elvemunningen ble kombinert. AIC modellseleksjon predikerte et toppunkt i bunndyrforekomster tre til fire km oppstrøms i elva. Den predikerte tettheten av sjøørret var høyest nedstrøms i elva, nær elvemunningen.

Denne studien, samt de tidligere studiene på effektene av restaureringstiltak i Bognelv, gir verdifull kunnskap for fremtidige elverestaureringsprosjekter. Til tross for dette er det behov for flere undersøkelser på elvebiotaen i Bognelv, for å skape en større forståelse av prosessene i elva, og for å videre forbedre den økologiske tilstanden i elva.

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# 1 Introduction

As human populations grow ever larger, pressure on our stream ecosystems is increasing. Altering rivers and streams for hydropower, agriculture and fishing has led to habitat degradation, pollution, overfishing, and numerous other challenges (Allan et al., 2021; Pander & Geist, 2013; Wohl et al., 2015). The result is widespread decline in the ecological status of rivers and more homogenous stream habitats, with severe effects on stream biota (Poff et al., 2007). The importance of healthy water bodies cannot be stressed enough, as they provide habitat, food and shelter for both humans and non-humans, as well as ensuring vital ecosystem services (Allan et al., 2021).

The ecological status of rivers and streams in Europe is monitored through the European Water Framework Directive (WFD). Established in 2000, the goal is to ensure “very good/good status” for all bodies of water in Europe (Water Framework Directive, 2000). River restoration is one method for achieving this goal where degradation has occurred, and ecological status is below “good”. Restoration has many definitions, and according to Wohl et al. (2005, p. 2) it can be defined as “*assisting the recovery of ecological integrity in a degraded watershed system by re-establishing the processes necessary to support the natural ecosystem within a watershed*”. This definition considers the importance of focusing on entire river systems, along with understanding the linkages connecting different parts of the river and how they affect each other (Wohl et al., 2005). Importantly, many factors influence how, why, and when restoration actually progresses, and testing the success of specific measures is vital for future efforts.

In the Nordic countries as in the rest of the world, river restoration is gaining traction, and there are several ongoing restoration projects (Hagen & Skrindo, 2010; Kristensen et al., 2014; Louhi et al., 2011; Nilsson et al., 2017). In Norway, restoration measures have mostly focused on liming projects and improving conditions for anadromous fish (Hagen et al., 2013). However, one quarter of Norwegian water courses do not have a satisfactory ecological status, and intact river systems are classified as “near threatened” on the Norwegian Red List (Artsdatabanken, 2018; Miljødirektoratet, 2021). This highlights the need for more focus on river restoration in Norway, along with studies testing effects of different measures in different rivers.

Although the number of river restoration projects have grown rapidly the past decades, few effect studies have been conducted. Most studies on the effects of river restoration to date lack documentation of positive responses on the biota (Arango et al., 2015; Bernhardt & Palmer, 2011; Roni et al., 2008). River restoration is rarely monitored for more than a couple of years, meaning we lack information on effect development covering a long-term perspective (Brederveld et al., 2011; Pander & Geist, 2013). This makes it difficult to properly assess success rates of such projects, as stream ecosystems often use years or decades in response to hydromorphological change. For instance, the study of the river Vindel in Sweden detected no positive responses for the stream biota five years post-restoration (Nilsson et al., 2017). One study even suggested that projects should have a time span of at least 20 years in a river to be able to document relevant responses in stream biota (Louhi et al., 2011).

The ongoing project in Bognelv, Finnmark (Figure 1), is one of a few of its kind. It was initiated in 2006 and is now in its 16<sup>th</sup> year, with the latest restoration measures conducted in 2019 (Bjordal & Sæle, 2019). The effects of river restoration on macroinvertebrates and fish in Bognelv have been studied since 2008 (Austvik, 2012; Bjørngaard, 2020; Nordhov & Paulsen, 2016; Schedel, 2011; Sødal, 2014; Solvang Strand, 2020). This long-term series of data represents a unique opportunity to study river restoration effects on stream biota over several years, and to compare results between years and for specific measures.

Macroinvertebrates and fish are among the most commonly used bioindicators for measuring success of river restoration projects (Lasne et al., 2007; Metcalfe, 1989), which emphasizes the importance of monitoring these species.

Bognelv used to be a naturally meandering river with thriving stocks of Atlantic salmon (*Salmo salar*), brown trout (*Salmo trutta*) and Arctic charr (*Salvelinus alpinus*) (Colman, 2011; Hoseth & Josefsen, 2005). However, large spring floods were damaging the surrounding agricultural land, and this led to Bognelv being channelized in the late 1930s. Several follow-up control measures were conducted during the next 60 years that greatly altered the river. Building of the new highway (E6) resulted in further channelization and erosion protection measures. By the late 1990s, the lower 3.5 km of the anadromous stretch of Bognelv was channelized. This had severe consequences for the river biota, resulting in

restoration being initiated in 2006. Since then, various in-stream restoration measures have been implemented to make the river more heterogenous and improve habitats for salmonid species. These include opening of side channels, removal of erosion protection and migration barriers, placement of large boulders, and digging out pools (Colman, 2011) (Appendix 3).

The first study investigating restoration effect on fish in Bognelv was conducted by Schedel (2011), with field work in 2008. This study revealed that restoration efforts improved conditions for young salmonids, leading to an increase in the population density, especially for brown trout. Austvik (2012) followed this up and concluded with similar results, adding that the macroinvertebrate community also benefitted from restoration by increased density. Both studies favored opening of side channels as the most successful restoration measure for increasing salmonid densities. Sødal (2014) further investigated how environmental variables interacted with restoration measures and affected the fish and macroinvertebrate communities. Her results indicated that shallow, slow-flowing areas with low amounts of algae and moss and large amounts of macroinvertebrates had the highest 0+ brown trout densities. She also found that 1+ brown trout densities were highest in areas with coarse gravel. Nordhov and Paulsen (2016) found that macroinvertebrate diversity was greatly affected by distance from estuary and water velocity, and that diversity was highest in areas with reopened side channels and riparian modifications. They also found that brown trout and Atlantic salmon densities in 2015 were at their highest since the restoration process in Bognelv began in 2006. In 2019, Bjørngaard (2020) tested macroinvertebrate diversity against time since last adjustment of the restoration measure. She found that diversity peaked approximately six years post-restoration. The same year, Solvang Strand (2020) found that side channels function as nursing habitats for salmonids in Bognelv, and that the river has not yet reached its carrying capacity for brown trout. These findings reveal that the restoration process in Bognelv has begun, and that it is starting to show relevant effects on macroinvertebrates and salmonids. However, the variability in effects and findings also point to the importance of monitoring and investigating river restoration projects for several years. Our study picks up where the last studies left, trying to piece together how macroinvertebrates and salmonids in Bognelv are responding to the restoration measures after 16 years of restoration efforts.

The effect of distance from estuary is one of our main topics for investigation. Habitat conditions and productivity in a river follow a gradient, and these factors are highly influenced by properties in the surrounding catchment (Foldvik et al., 2017). For instance, proportion of agricultural land surrounding the river has been shown to have a positive effect on production of salmonids, up to a certain threshold (Jonsson et al., 2011). This means that carrying capacity for brown trout in Bognelv will likely differ across the spatial gradient depending on the properties of the catchment, and distance from estuary is therefore an important topic for investigation.

During three weeks of August and September 2021, we conducted field work in Bognelv to study fish and macroinvertebrates. First, we registered environmental variables and habitat conditions for the stream biota at our sampling sites. Next, we sampled macroinvertebrates to investigate their diversity and abundance. Finally, we electro-fished juvenile salmonids so that we could calculate salmonid density and age composition. Our data could then be compared with data from previous years, thus providing a platform to better understand how the restoration processes in Bognelv are developing.

We investigated three specific effects: type of restoration measure (tom), time since last measure (tslm), and distance from estuary. Our overarching research aim was to investigate how the macroinvertebrate and salmonid communities have changed over time, and how the restoration measures have affected these communities. To answer this, we have four sub-questions:

1. What types of restoration measures have proven most successful in improving conditions and why?
2. What effect does the time perspective have?
3. What effect does distance from estuary have?
4. Looking back over the past 16 years, how has the entire system responded to restoration?

## 2 Materials and methods

### 2.1 Study area

Field work for this study was conducted over the course of three weeks from 23rd of August to 10th of September 2021, in the river Bognelv (Bávnnjajohka) in Langfjordbotn, Troms and Finnmark county.

Bognelv runs from south to north through the valley Bognelvdalen and enters Langfjordbotn (Figure 1). Bognelv's water course number is 211.8A0. The river drains a large mountainous area south-west of the municipal center Alta, with a catchment area of 88.5 km<sup>2</sup> located in an alpine zone 500-600 meters above sea level, and 80 % of this area is above the tree line (Appendix 1). The alpine conditions in the catchment area cause large spring floods in the end of June (Hoseth & Josefsen, 2005).

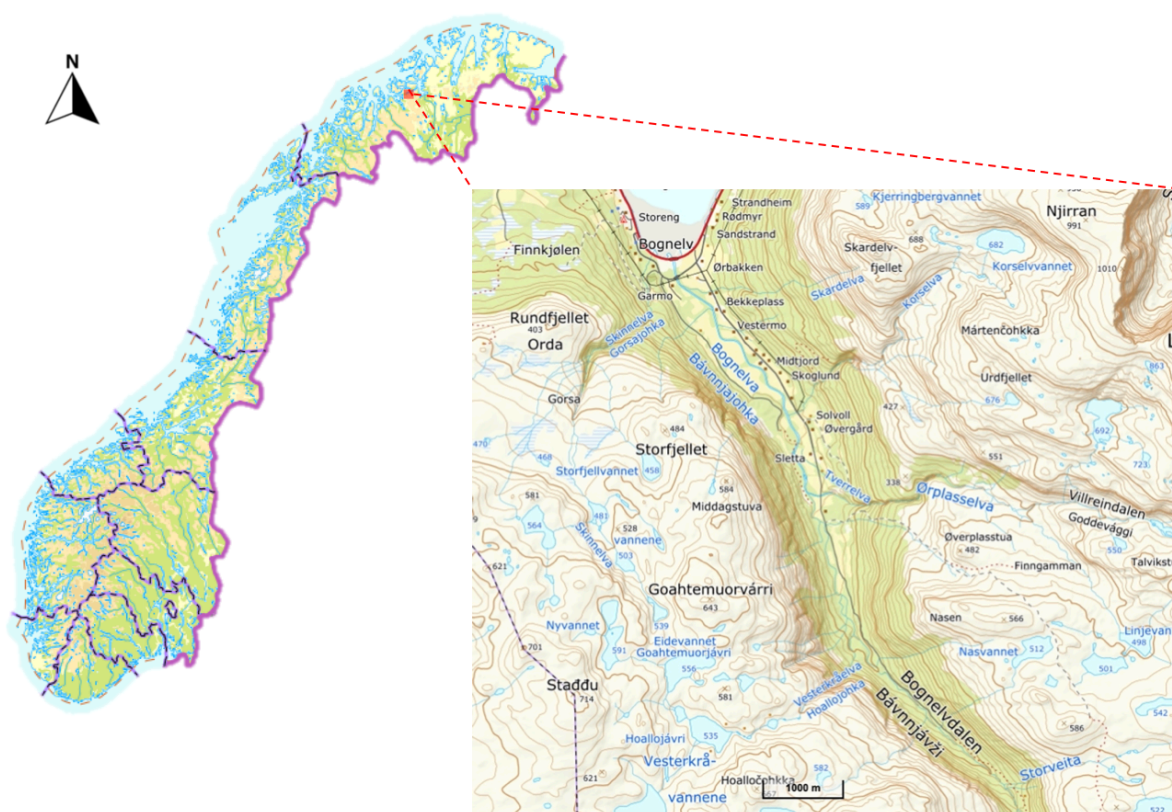


Figure 1. Map of the study area ©norgeskart.no. Map of zones and stations in Appendix 2.

Bognelvdalen is a typical U-shaped valley, with steep hillsides, a flat valley bottom and lush vegetation. There is extensive agricultural activity and scattered settlements along the river.

Bognelv used to be a free-flowing river with dynamic erosion and sedimentation processes forming a braided river pattern (Figure 2). At this time, the river kept thriving stocks of Atlantic salmon, brown trout and Arctic charr. However, in the late 1930s, measures were implemented to secure the surrounding agricultural land from floods. In 1956, a two km stretch in the lower part of the river was channelized to further secure the surrounding land from erosion. By 1972, the lower part of Bognelv had become streamlined and the natural erosion and sedimentation processes were lost (Figure 3).



*Figure 2. The lower part of Bognelv in 1946. It is free flowing and still has dynamic erosion and sedimentation processes (Hoseth & Josefsen, 2005).*



*Figure 3. The lower part of Bognelv in 1972. The river has been channelized and streamlined and the natural erosion and sedimentation processes were lost (Hoseth & Josefsen, 2005).*

The abundance of salmonids drastically declined following these measures (Dønnum & Colman, 2004; Hoseth & Josefsen, 2005). Concerns were raised by the local hunting and fishing association in 1972 regarding the decline of salmonid species in the river due to the conducted measures. As a response, boulders and weirs were placed in the river as an attempt to improve habitat conditions (Hoseth & Josefsen, 2005). An inspection of Bognelv in 2003 revealed that approximately 40 % of the anadromous stretch of the lower reaches were unsuitable as spawning areas for salmonids. Most of the side channels were also deemed unsuitable as spawning habitats, and some of these were even detached from the river (Dønnum & Colman, 2004). In 2006, the Norwegian Water Resources and Energy Directorate (NVE) began restoration measures in Bognelv to improve its ecological condition. Measures such as re-opening side channels, recreating the natural flow of the river and removal of barriers for salmonid species were implemented (Bjordal & Hoseth, 2009; Colman, 2011). Restoration has continued until recently, with the latest measures being implemented in 2019. See map of conducted measures in Bognelv in Appendix 2, and table of conducted measures in Appendix 3.

## 2.2 Study species

### 2.2.1 Macroinvertebrates

Macroinvertebrates are one of the biological elements used for classification of ecological status of rivers according to the WFD (Water Framework Directive, 2000). They are a diverse group with many different requirements and sensitivities to pollutants and can therefore function as indicators of water quality (Direktoratsgruppe Vanndirektivet, 2018; European Commission, 2021). Because they are relatively easy to sample and identify, and because they often respond quickly to environmental changes, they are commonly used when assessing river restoration projects (Kilgour & Barton, 1999; Metcalfe, 1989; Miller et al., 2010). Macroinvertebrates have a range of different functional roles in the ecosystem and are an important source of food for salmonids (Wallace & Webster, 1996).

### 2.2.2 Salmonids

Several studies have investigated fish stocks in Bognelv following the restoration measures, and Arctic charr has not been observed in the river since 2013 (Austvik, 2012; Bjørngaard,

2020; Nordhov & Paulsen, 2016; Sødal, 2014; Solvang Strand, 2020). It will therefore not be included in the study species.

Many abiotic factors are important in the growth of both Atlantic salmon and brown trout. Factors such as water velocity, depth, substrate size, shelter availability and vegetation cover can affect the spatial distribution of these species. It can also affect the growth and size, which are important for reproduction and recruitment. Atlantic salmon and brown trout are morphologically similar, but their habitat preferences are somewhat different (Jonsson & Jonsson, 2011).

Availability of pools and shelter affect the growth of salmonid species. Slow-flowing habitats such as pools are critical in the early stages of life for both species, where they can grow and feed whilst not draining their energy (Armstrong & Nislow, 2006; Nislow et al., 2000). Both species prefer to feed on drifting macroinvertebrates, and holding position in fast-flowing stretches is energy draining. Therefore, presence of shelter in these areas of the river is essential (Jenkins Jr, 1969; Teichert et al., 2010).

Atlantic salmon are more adapted to strong currents and will spawn in deeper and more fast-flowing water than brown trout. Juvenile Atlantic salmon prefer areas with gravel, pebbles and cobble, and abundance will be low in areas where the substrate is of silt and sand (Baglinière & Champigneulle, 1986; Heggenes, 1990; Jonsson & Jonsson, 2011). The smallest parr (<7 cm) prefer a diameter of substrate particles between 1.5 and 20 cm, while longer individuals (>7 cm) prefer substrate diameter to be 10-50 cm or larger (Backiel & Le Cren, 1978; Degraaf & Bain, 1986; Jonsson & Jonsson, 2011).

Brown trout prefer slower flowing habitats than Atlantic salmon and are more sensitive to strong currents. Brown trout are usually abundant close to the riverbank. They can be found in streams with fine-grained bottom substrate but prefer more coarse substrates. The smallest parr (<7 cm) use river mosses for shelter, while longer individuals (>7 cm) find shelter between boulders (Bohlin, 1977; Heggenes, 1988b; Jonsson & Jonsson, 2011). Brown trout favor boulders and coarse substrate as it will protect against strong currents. The juveniles save energy by being in low-velocity water whilst being close to the drift of invertebrates (Jonsson & Jonsson, 2011).



Canopy cover is important for the growth of salmonid species and absence of vegetation has been found to cause chronic stress (Jonsson & Jonsson, 2011; Pickering et al., 1982).

Boulders, large woody debris, and canopy cover are all used for shelter by juveniles. Input of leaves from riparian vegetation also increases invertebrate activity, and thus, increases food supply for juvenile salmonids (Jonsson & Jonsson, 2011).

### 2.3 Data collection

This study was part of a continuous project of river restoration effects on fish and macroinvertebrates in Bognelv (Austvik, 2012; Bjørngaard, 2020; Nordhov & Paulsen, 2016; Schedel, 2011; Sødal, 2014; Solvang Strand, 2020). To ensure our results were comparable to previous studies, the method for data collection was replicated from Nordhov and Paulsen's study from 2016, the most recent study conducted on fish *and* macroinvertebrates in Bognelv. The studies conducted in 2019 had a different study design with fewer stations in mostly the lower section of the river. Data from 2019 will therefore be comparable only on a system level, not by station.

The data collection was split into three parts: 1. Measurements of environmental variables, 2. Macroinvertebrate sampling, and 3. Electrofishing. The river was divided into 12 zones and 56 stations (Nordhov & Paulsen, 2016, Appendix 2). This zonation was used for all three parts of the data collection. Each station measured 15 meters long and 2 meters wide (from the riverside towards the river center). All stations were divided into three cross-section transects at 0, 7.5, and 15 meters along the riverside. Zone 1 was located in the estuary, while zone 12 was furthest upstream (station coordinates can be found in Appendix 4). Zones 11 and 12 were used as control zones, as no channelization or restoration measures have been conducted here. They represent therefore an undisturbed state. Due to challenging conditions in the river, we were unable to conduct electrofishing at all stations. Station 34 had dried out, while stations 15, 19, 20, 48 and 68 were excluded because of high discharge and strong currents.

#### 2.3.1 Registration of environmental variables

Environmental variables were recorded at all three cross-section transects for each station. The recorded variables were percentage canopy cover of river, riverbank and riverside

vegetation, substrate grain-size composition, river width and depth, water velocity, mean percentage cover of algae and moss. Percentage canopy cover of river, riverbank and riverside vegetation was classified into six percentage cover groups. Substrate grain-size composition was classified into five grain-size groups and visually estimated to a percentage (Figure 4). River width was measured across the entire river for each transect and depth was measured at five points along each cross-section transect. Water velocity was calculated by how many seconds it took for a leaf to drift one meter downstream (m/s). Mean percentage cover of algae and moss was classified into four percentage cover groups. Total number of pools (areas with still water larger than 2 m<sup>2</sup>) and number of large woody debris (branches with diameter larger than 10 cm and more than 1 m long and large concentrations of small woody debris) was counted for each station. Detailed method for measuring environmental variables can be found in Appendix 5.



*Figure 4. Measuring substrate size.*

As in previous master theses, distance from E6 was used to measure distance from estuary. Data on implemented restoration measures in Bognelv up until 2014 were collected from Nordhov and Paulsen (2016) and implemented measures after 2014 was retrieved from Johansen (2020) and Anders Bjordal at NVE. This data was categorized into five categories, (Nordhov & Paulsen, 2016); channelized stations (n=11), stations with weirs implemented (n=7), stations with reopened side channels (n=22), stations with riparian modifications (n=8) and stations with no restoration measures conducted (n=6).

Data on snow depth going back to 1998 were retrieved from the nearest weather station, Sopnesbukt (SN92910) (Norwegian Centre for Climate Services, 2021). Average monthly snow depth was calculated by adding all the measured snow depths from each station where data was available. Snow depth was the only climate variable that was available dating back to 1998, which is why we used this as an indicator for “spring arrival” (Figure 5).

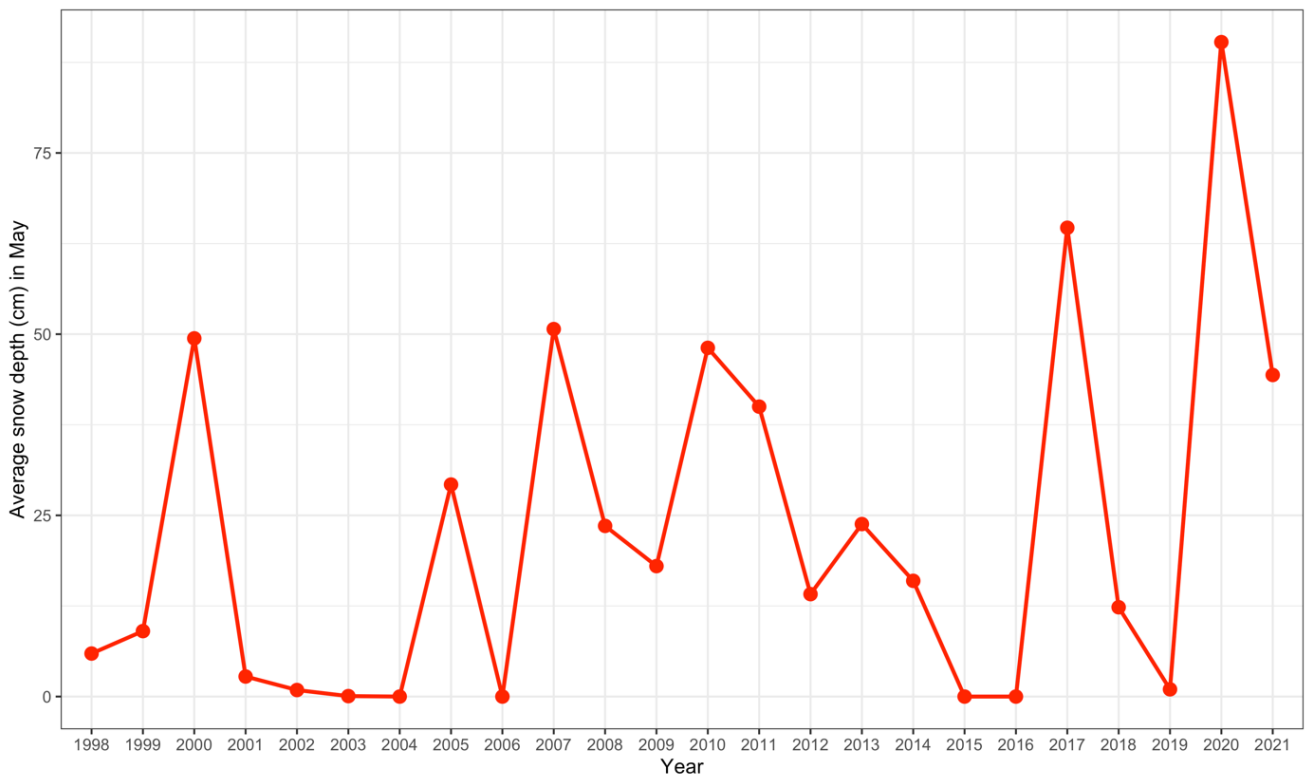


Figure 5. Average snow depth (cm) in Bognelv in May. Data was retrieved from Norwegian Centre for Climate Services ([seklima.met.no](http://seklima.met.no)) from the weather station Sopnesbukt in Langfjordbotn, station number SN92910. Studies in Bognelv have been carried out in the fall.

### 2.3.2 Macroinvertebrate sampling

Macroinvertebrate sampling was conducted at three cross-section transects for each station. We followed the “kick-sampling” method used in previous master theses from Bognelv, as defined by Hynes (1961). A quadratic net with a 30 x 30 cm opening and mesh size of 450 µm and a 1.2 m handle was placed in the river downstream by the person sampling. For 20 seconds, the person kicked the riverbed right upstream the net whilst moving vertically along the cross-section transect so that macroinvertebrates would loosen and drift into the net. All three cross-section transects were sampled without emptying the net between sampling, and the accumulated contents from the three sampling rounds were emptied into a white tray for

investigation (one sample per station). Larger debris like leaves and sticks were inspected for macroinvertebrates and disposed whilst the rest was emptied into plastic bags with 2 dl of 96 % ethanol and a waterproof tag with sample identification details. The samples were transported to Department of Ecology and Natural Resources at NMBU, to be identified in the laboratory. To save time, large samples were divided into subsamples either 2, 4 or 8 times in the lab. This method is in accordance with the method used in previous studies on macroinvertebrates (Bjørngaard, 2020; Lungrin, 2020; Nordhov & Paulsen, 2016). The sample was emptied into a tray and mixed thoroughly, before it was divided into the chosen number of subsamples. A random number-generator was used to pick out the subsample for investigation. When identification of the subsample was complete, the number of individuals of each species in the sample was multiplied with the number of subsamples (2, 4 or 8). All macroinvertebrates belonging to the EPT-order (Ephemeroptera, Plecoptera, Trichoptera) were classified to species, or lowest taxonomic level possible. Individuals belonging to other orders were classified to family (Arnekleiv, 1995; Krogvold & Sand, 2008; Lillehammer, 1988; Nilsson, 1996; Nilsson, 1997; Rinne & Wiberg-Larsen, 2016).

### 2.3.3 Electrofishing

Electrofishing for juvenile salmonids was carried out using a GeOmega FA-4 portable backpack generator produced by Terik Technology (Figure 6). Electrofishing was conducted along the entire station (15 m upstream), from the riverside and 2 m out in the river (covering the cross-section). The method is based on Bohlin et al. (1989). One person operated the electrofishing backpack with the anode, while the other walked behind with a small net to catch the stunned fish and put them in a bucket filled with water. The anode also had a small net mounted around the anode ring to assist in catching stunned individuals. While slowly moving upstream, the person handling the anode sent electric pulses through the water for an interval of 5 to 10 seconds, after which the team moved a few meters upstream and continued. We followed the method of a “three pass system”, to use the Zippin removal method for estimation of population densities (Bohlin et al., 1989; Zippin, 1958). This entails electrofishing each station three times, with a time interval of 30 minutes between the start of each pass. However, in stations with no catches or only one fish caught in the first pass, only one pass was conducted. The salmonids were placed in dark 10 L buckets holding fresh river water, one bucket for each pass. They were identified to species and measured to the closest

millimeter (total length, Figure 7). After all passes were conducted, the salmonids were released back into the river in the same station as where they were caught.



*Figure 6. Electrofishing in Bognelv.*



*Figure 7. Measuring length of captured brown trout.*

## 2.4 Statistical analyses

Earlier studies on Bognelv have focused on three salmonid species: Atlantic salmon, Arctic charr and brown trout. However, overall catches of Atlantic salmon have been low ever since the studies began (n=266), and only four Arctic charr have been sampled during the entire study period from 2008 to 2021. In comparison, a total of 3415 trout have been caught since the Bognelv project was initiated. Only one Atlantic salmon was caught in 2021, and no Arctic charr. Atlantic salmon and Arctic charr were therefore excluded from statistical analysis, and the focus is on brown trout only.

Data were prepared for analysis in Microsoft Excel (Microsoft Office, 2022), and then imported into the statistical software program R version 4.1.2 (R Core Team, 2022). The R package “vegan” (Oksanen et al., 2020) was used for ordination analyses, to assess the effects

of environmental variables and restoration measures on macroinvertebrates and fish. Count data was  $\ln(x+1)$ -transformed to avoid  $\ln(0)$ .

Environmental data was imported into Microsoft Excel and the categories were simplified to ease the analysis process. The percentage-groups were made into numbers that corresponded to each category (i.e., percentage cover of algae and moss: Category 1: 0%, Category 2: 1-33%, Category 3: 34-66% and Category 4: >66%). For more detailed information, see Appendix 5. Mean values for each environmental variable were calculated to compare cross-section transect values with total number of pools and large woody debris, which was only counted once per station.

When analyzing the effect of time since last restoration measure (tslm) on salmonids and macroinvertebrates, the control zones (11 and 12) were removed. No restoration measures have been conducted here, which means  $tslm=0$ . The channelized stations that are unrestored also have  $tslm=0$ . So, to avoid the control zones furthest upstream from being juxtaposed with the channelized zones, we decided to remove the control zones. In addition, the channelization measure was excluded when conducting analysis on  $tslm$ , because no restoration measures were conducted here ( $tslm=0$ ).

Zone 1 was excluded from analysis due to brackish conditions in the estuary (see zone map in Appendix 2). These conditions led to an excess of *Gammarus* species which strongly affected our diversity and abundance analysis. Hence, zone 1 was not representative for the macroinvertebrate community. We therefore decided to exclude that zone completely from the model selections of both macroinvertebrates and fish.

Shannon Wiener diversity index was used to estimate the relative macroinvertebrate alpha-diversity in the river. The formula is

$$H' = - \sum_{i=1}^s p_i \ln p_i$$

where  $H'$  is the species diversity index,  $S$  represents the number of species, and  $p_i$  is the amount each species contribute to the entire sample (Nolan & Callahan, 2006). The higher the Shannon Wiener index number, the higher species diversity (Heip et al., 1998). In this thesis, only species diversity within the entire sample for each station was analyzed, i.e., as a

comparison between stations for the river as a whole ( $\alpha$ -diversity) and not among-station difference in species composition ( $\beta$ -diversity).

ANOVA-test and parameter effect test were used to test for statistical significance.  $\alpha=0.05$  was set as the level of significance.

#### 2.4.1 Ordination analysis

When dealing with a large dataset with many variables, ordination is a useful tool to simplify the analysis process without compromising accuracy. It is typically divided into constrained and unconstrained ordination. Unconstrained ordination helps you make sense of a large data set by spotting overall patterns, using Principal Component Analysis (PCA) for linear data and Correspondence Analysis (CA) for unimodal data. If there are one or more predictor variables and factors that can be used to explain the variation in the response data, constrained ordination is used. In this case, an unconstrained Detrended Correspondence Analysis (DCA) was used on the dataset for environmental variables, to figure out which analysis was most relevant for our data. The DCA yielded the first axis length between 1 and 3, which led to the conclusion that PCA was the most suitable method of analyzing the environmental data (Lepš & Šmilauer, 2003).

We conducted two separate ordination analyses. First to test correlation of environmental variables and a separate analysis to test effect of restoration measures on the macroinvertebrate data. For the environmental data, we ran a PCA to test the effect of these variables and how they grouped together, to explain the variation in the data. A principal component analysis consists of independent principal components (PC) that are fitted to the environment data (Abdi & Williams, 2010). To begin with, our dataset consisted of 13 environmental variables, but through PCA these were reduced to ten principal components, with the top three principal components explaining 61 % of the variation. Principal component scores and corresponding biplots of principal components 1, 2 and 3 are presented in the results section, while species scores are presented in Appendix 6. The top three principal components from our environmental data were used as predictors in a constrained redundancy analysis (RDA) for the macroinvertebrate data. They were also used as predictors in the analysis of salmonid data. The PC-scores were used to account for the effect of habitat composition in each station.

Residual plots were used for model validation to ensure that the necessary conditions were met (Appendix 7). Ideally, the residuals should be randomly spaced around the horizontal axis of the fitted values and should display no clear patterns. If this is the case, it is a sign that the data fits well for regression analysis (Sokal & Rohlf, 2012).

#### 2.4.2 Salmonid density data

High discharge and small fish led to reduced catchability of 0+. This led us to exclude 0+ from analysis, as the catchability was impossible to estimate. Instead, we decided to analyze densities of 1+. The 1+ density data was analyzed by fitting candidate linear models where fish densities (both when used as response and predictors) were  $\ln(x+1)$ -transformed to secure homogenized residuals and to avoid problems with  $\ln(0)$  (Sokal & Rohlf, 2012).

#### 2.4.3 Analytical approach

Akaike's Information Criteria (AIC) was used for model selection (Akaike, 1974). AIC is a metric that finds models that most effectively balances model precision and model bias. It is calculated by taking the sum of the models' deviance plus twice the number of parameters included in the model. The model with the lowest AICc value is the one with the most support in the data. The R package AICcmodavg (Mazerolle, 2020) was used to perform the model selection. Models with a  $\Delta AIC$  level below 4 were considered for discussion of results (Anderson & Burnham, 2002).

Densities of 1+ were analyzed against densities of >1+. We separated >1+ into categories of low (0), middle (20) and high (80) densities. The numbers of >1+ in each category were chosen based on calculated averages of >1+ from our catches.



### 3 Results

#### 3.1 Influence of environmental variables

A Detrended Correspondence Analysis (DCA) on the environmental variables yielded a first axis length between 1 and 3 (DCA1 = 1.453), which supported linear ordination for further analysis of the data (Table 1). Principal component analysis was used to analyze correlation of environmental variables with macroinvertebrate diversity and abundance, and brown trout densities.

*Table 1. Results from a detrended correspondence analysis on the environmental variables. Axis length for DCA 1 is between 1 and 3 which supports linear ordination for further analysis.*

	<b>DCA1</b>	<b>DCA2</b>	<b>DCA3</b>	<b>DCA4</b>
<b>Eigenvalues</b>	0.213	0.036	0.018	0.017
<b>Decorana values</b>	0.213	0.009	0.005	0.004
<b>Axis lengths</b>	1.453	0.542	0.310	0.356

PC1, PC2 and PC3 accounted for 61% of variation (Table 2).

*Table 2. Importance of components of PCA-analysis for environmental variables. PC1, PC2 and PC3 combined accounts for 61% of the variation in the data. See full table in Appendix 6.*

	<b>PC1</b>	<b>PC2</b>	<b>PC3</b>
<b>Eigenvalue</b>	2.677	1.880	1.573
<b>Proportion explained</b>	0.268	0.188	0.157
<b>Cumulative proportion</b>	0.268	0.455	0.613

The PC1 axis was associated with vegetation and accounted for ~ 27% of the variation in the data. Canopy cover, riverbank vegetation and moss all correlated positively (Figure 8). Algae and riverside vegetation also correlated positively with PC1, but with a weaker association than the other vegetation variables.

PC2 accounted for ~ 19% of the variation in the data. On this axis, pools and large woody debris (lwd) correlated positively, and water velocity, depth and substrate correlated negatively. The higher velocity, the fewer pools and large woody debris items.

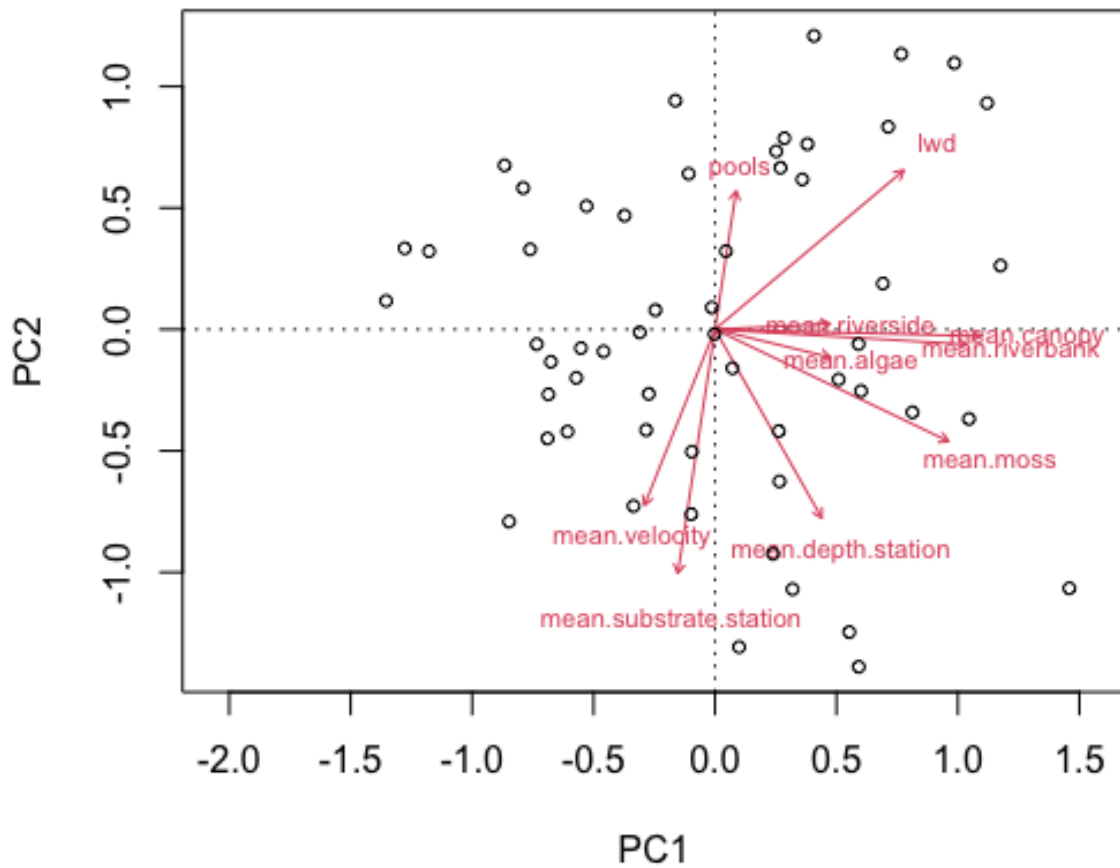


Figure 8. Biplot of PC1 and PC2 showing correlation of environmental variables in Bognelv. All values are averages from the three transects per station, except for pools and large woody debris (lwd) which have one value per station. White dots represent stations.

PC3 was more difficult to interpret than the two other principal components, with few clear patterns (Figure 9). Pools and algae seemed to group together, and there was a weak grouping of the different vegetation variables. It accounted for ~ 16% of the variation in the data.

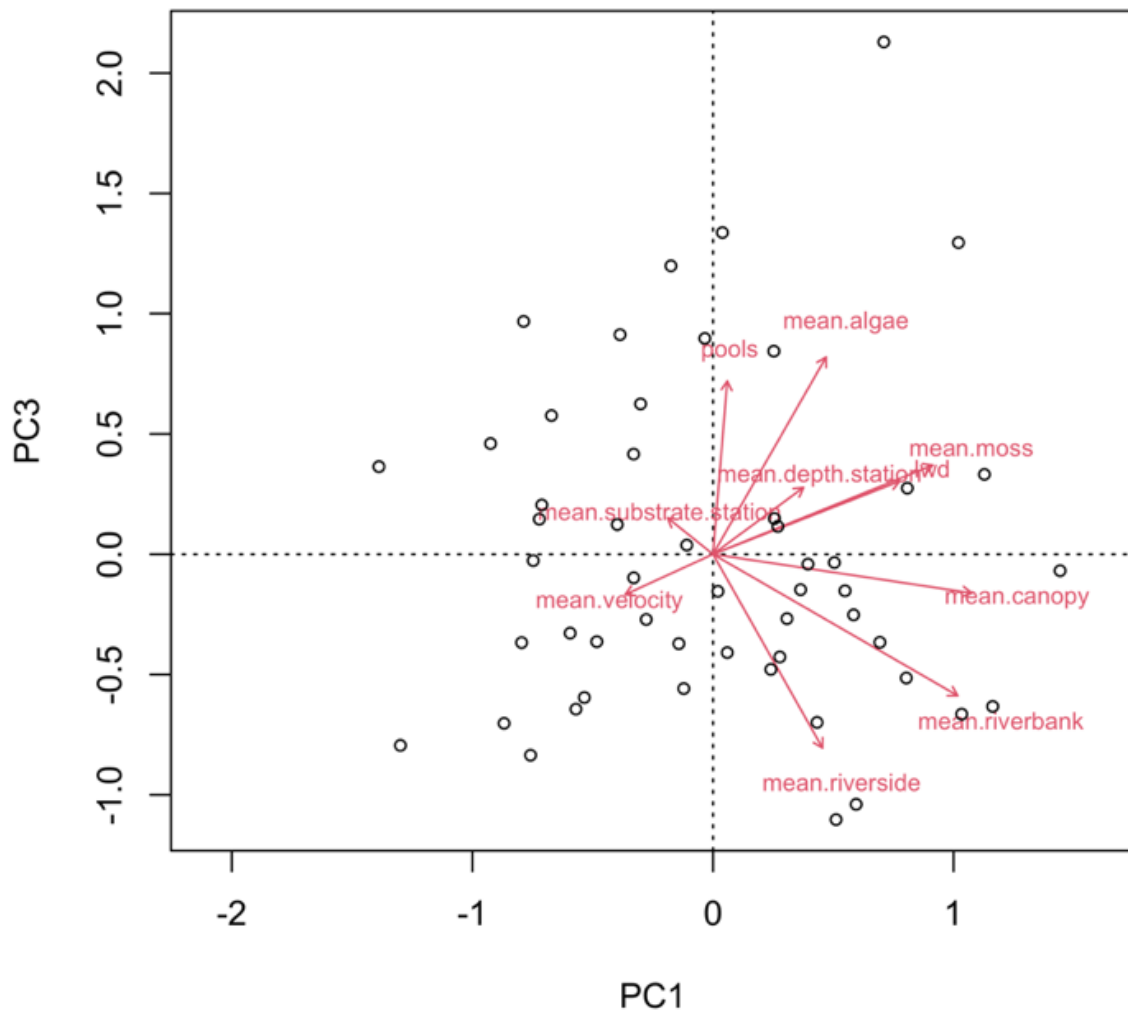
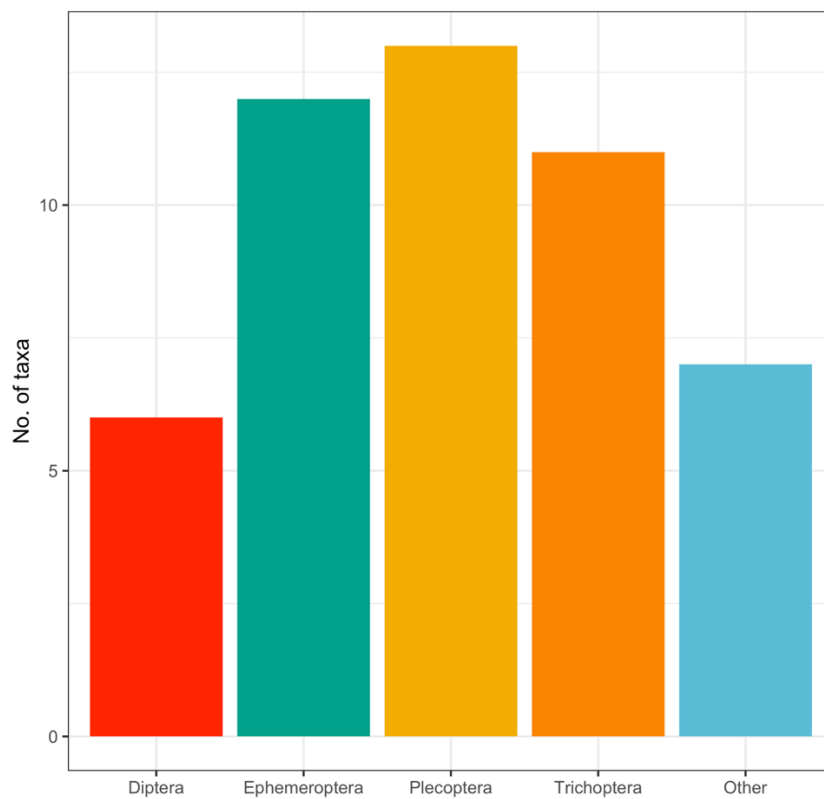


Figure 9. Biplot of PC1 and PC3 showing correlation of environmental variables in Bognely. All values are averages from the three transects per station, except for pools and large woody debris (lwd) which have one value per station. White dots represent stations.

## 3.2 Macroinvertebrates

### 3.2.1 Species composition

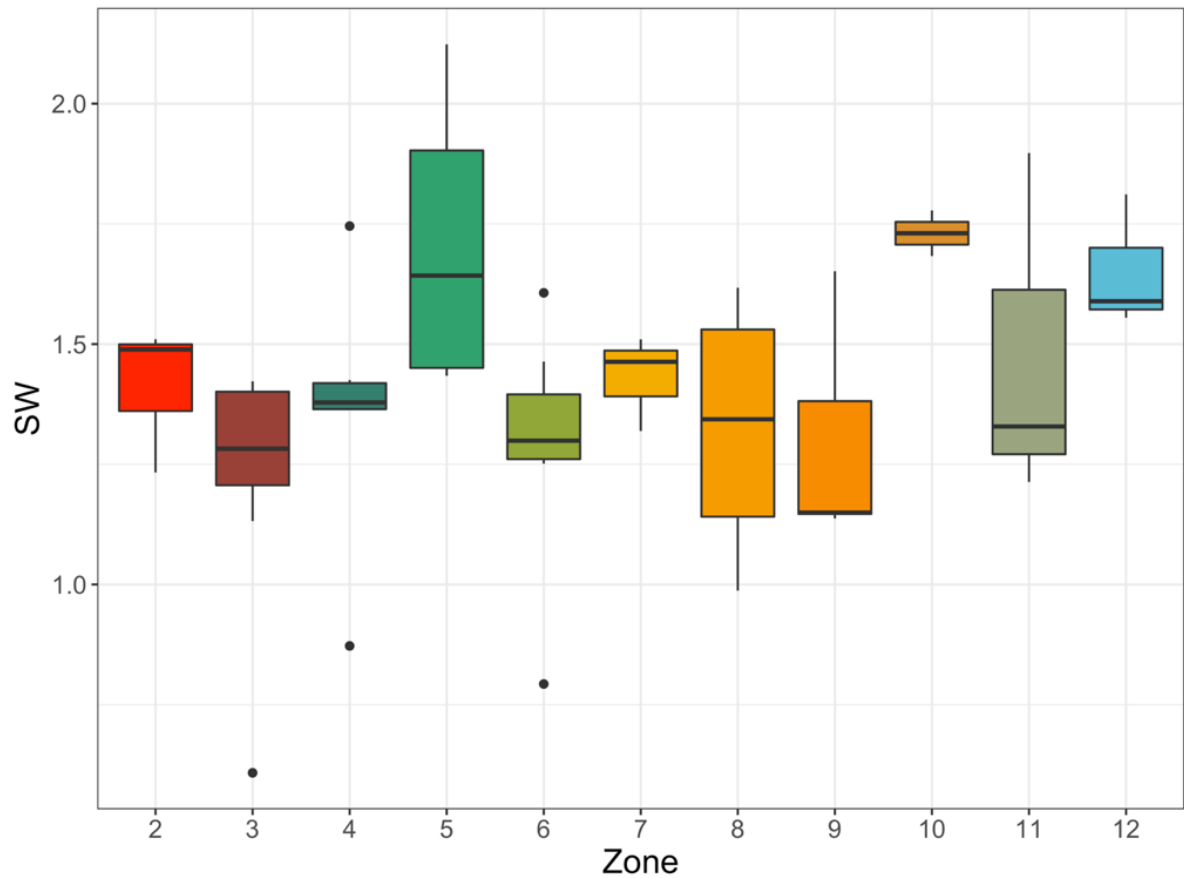
We sampled in total 12 211 macroinvertebrate individuals, covering 50 different taxonomic levels (Figure 10, raw data in Appendix 8). We identified six taxa of the order Diptera (true flies), 12 taxa of Ephemeroptera (mayflies), 13 taxa of Plecoptera (stoneflies), 11 of Trichoptera (caddisflies), and eight other taxonomic levels: Empididae (dagger flies), Turbellaria (flatworms), Collembola (springtails), Hydrachnida (water mites), Coleoptera (beetles), Copepoda (crustacea), Gastropoda (snails) and Gammarus (amphipoda).



*Figure 10. The species composition and number of sampled taxa of macroinvertebrates sampled in Bognelv 2021.*

## 3.3 Macroinvertebrate diversity

There was substantial among-zone and within-zone variation in Shannon Wiener diversity (SW), with a seemingly positive upstream tendency in species diversity. Zone 10 had the highest median value (Figure 11).



*Figure 11. Shannon Wiener diversity index for different zones in Bognelv. The black horizontal line represents the median value, while the colored box indicates the 50 % interquartile range, and 95 % confidence interval is shown with black vertical lines.*

Type of measure did not have a significant effect on macroinvertebrate diversity (Figure 12). However, stations with no measure yielded highest median value.

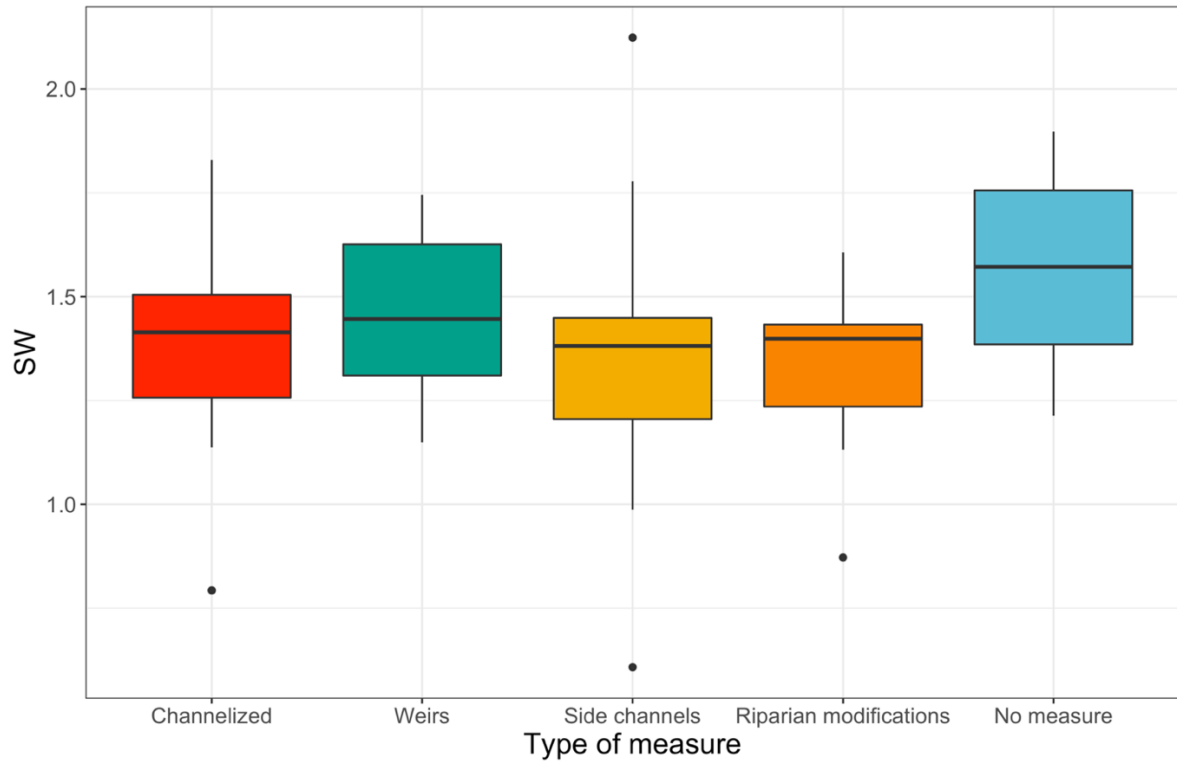


Figure 12. Shannon Wiener index and type of measure. The black horizontal line represents the median value, while the colored box indicates the 50 % interquartile range, and 95 % confidence interval is shown with black vertical lines.

### 3.3.1 Correlates and effects on macroinvertebrate diversity

Shannon Wiener model selection favored a model with time since last restoration measure (tslm) as an effect (Model 1, Table 3).

Table 3. AICc-based linear model selection for the 10 candidate models fitted to predict Shannon-Wiener (SW) index for macroinvertebrate data from Bognelv 2021. *tslm* = time since last measure (years); *tslm*<sup>2</sup> = a quadric effect of time since last measure; *PC1*, *PC2* and *PC3* = principal components of environmental variables (Table 2); *dist.estuary* = distance from the estuary (km); *tom* = type of restoration measure. *K* = number of estimated parameters; *AICcWt* = the model AICc weight; *Cum.Wt* = cumulative weight; *LL* = model log likelihood. All models were fitted using log-likelihood method. The top model with *tom* included as an effect is provided in the table to address all hypotheses.

Model no.	Model structure	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
1	<i>tslm</i>	3	15.22	0.00	0.18	0.18	-4.32
2	<i>tslm</i> <sup>2</sup>	4	15.94	0.72	0.13	0.31	-3.47
3	<i>tslm</i> + <i>PC2</i>	4	16.05	0.82	0.12	0.42	-3.52
4	<i>dist.estuary</i> + <i>PC2</i>	4	17.16	1.94	0.07	0.49	-4.08
5	<i>tslm</i> + <i>PC3</i>	4	17.22	2.00	0.07	0.56	-4.11
6	<i>tslm</i> + <i>PC1</i>	4	17.56	2.34	0.06	0.62	-4.28
7	<i>dist.estuary</i> + <i>PC3</i>	4	17.69	2.47	0.05	0.67	-4.35
8	<i>dist.estuary</i> + <i>PC1</i>	4	17.90	2.68	0.05	0.71	-4.45
9	<i>dist.estuary</i> * <i>PC3</i>	5	18.50	3.28	0.03	0.75	-3.48
10	<i>tslm</i> * <i>PC2</i>	5	18.56	3.34	0.03	0.78	-3.51
	...	...	...	...	...	...	...
11	<i>tom</i> + <i>tslm</i> <sup>2</sup>	7	20.54	5.32	0.01	0.91	-1.76

Model 1 predicted a slope with an increase in Shannon Wiener diversity with 0.004 per years post-restoration with a standard error of 0.008 (parameter estimates in Appendix 9). The effect of time in this model was negligible. The second most supported model, model 2, predicted diversity to decrease post-restoration and reach a bottom point after approximately six years, after which diversity was predicted to increase (Figure 13). The effect of time since last measure was not statistically significant (Table 4, bottom).

Table 4. Parameter estimates and results from ANOVA-test for model 2, the second most supported model according to AIC model selection (Table 3). *tslm* = time since last measure; *tslm*<sup>2</sup> = a quadric effect of time since last measure.

Parameter estimates			
Parameters	Estimate	Std. Error	t value
Intercept	1.370	0.081	16.848
<i>tslm</i>	-0.033	0.030	-1.082
<i>tslm</i> <sup>2</sup>	0.0029	0.002	1.270

ANOVA					
Effect	Df	Sum Sq	Mean Sq	F value	Pr(>F)
<i>tslm</i> <sup>2</sup>	2	0.141	0.070	0.960	0.391
Residuals	42	3.074	0.0732		

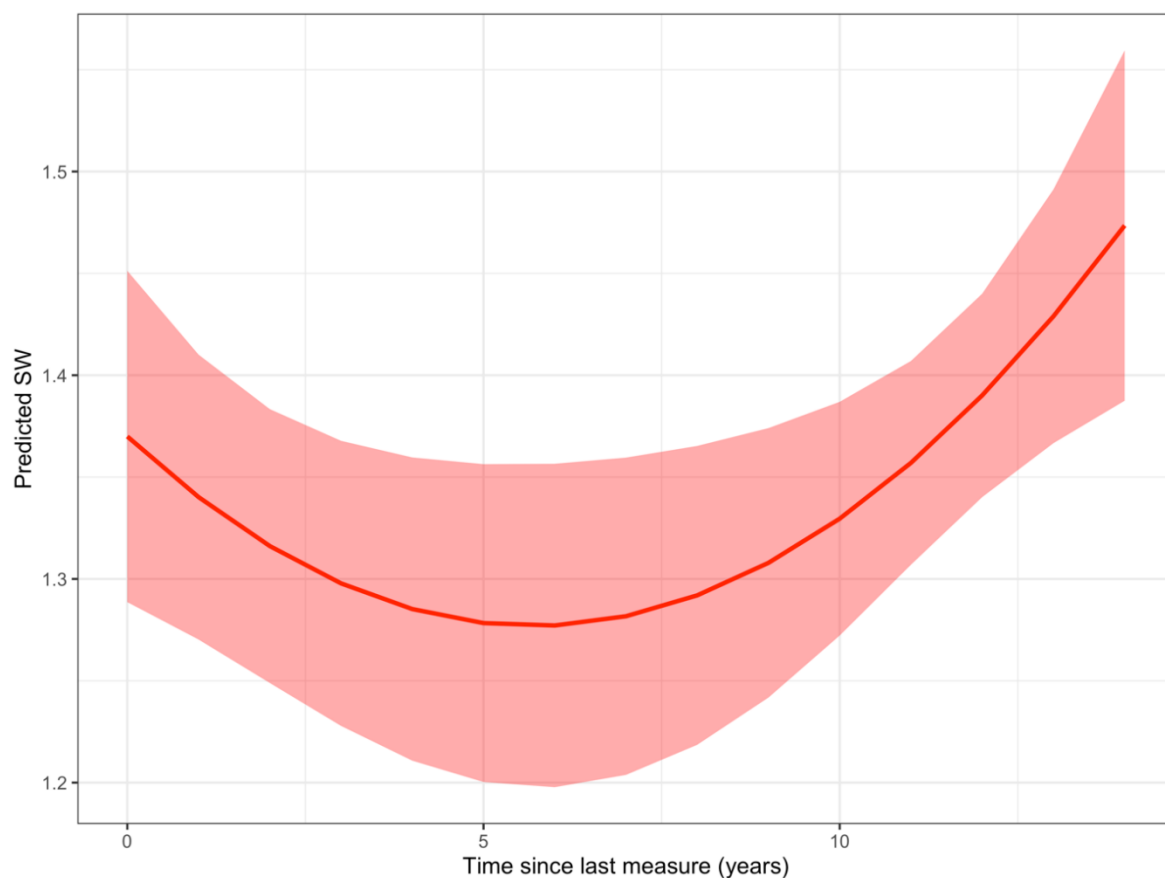


Figure 13. Model 2, the second most supported model for macroinvertebrate diversity (Table 3). The figure shows how time since last restoration measure affects predicted Shannon Wiener index (SW). The ribbon represents the 95 % confidence interval.

Model 4 is the top model with distance from estuary included as an effect (Table 3). It predicted that PC2 had a negative effect on the SW macroinvertebrate diversity (Figure 14). For any given PC2 value, the Shannon Wiener index decreased with increasing distance from



estuary. ANOVA-test revealed that neither the effect of distance nor PC2 was statistically significant (Table 5, bottom).

Table 5. Parameter estimates and results from ANOVA-test for model 4, the most supported model with distance from estuary according to AIC model selection (Table 3). *dist.estuary* = distance from estuary (km); PC2 is principal component 2 from our PCA analysis (Table 2).

Parameter estimates					
Parameters	Estimate	Std. Error	t value		
Intercept	1.379e+00	8.091e-02	17.040		
dist.estuary	-2.882e-06	3.665e-05	-0.079		
PC2	-5.409e-02	6.273e-02	-0.862		
ANOVA					
Effect	Df	Sum Sq	Mean Sq	F value	Pr(>F)
dist.estuary	1	0.000	0.000	0.001	0.971
PC2	1	0.056	0.056	0.744	0.393
Residuals	42	3.159	0.075		

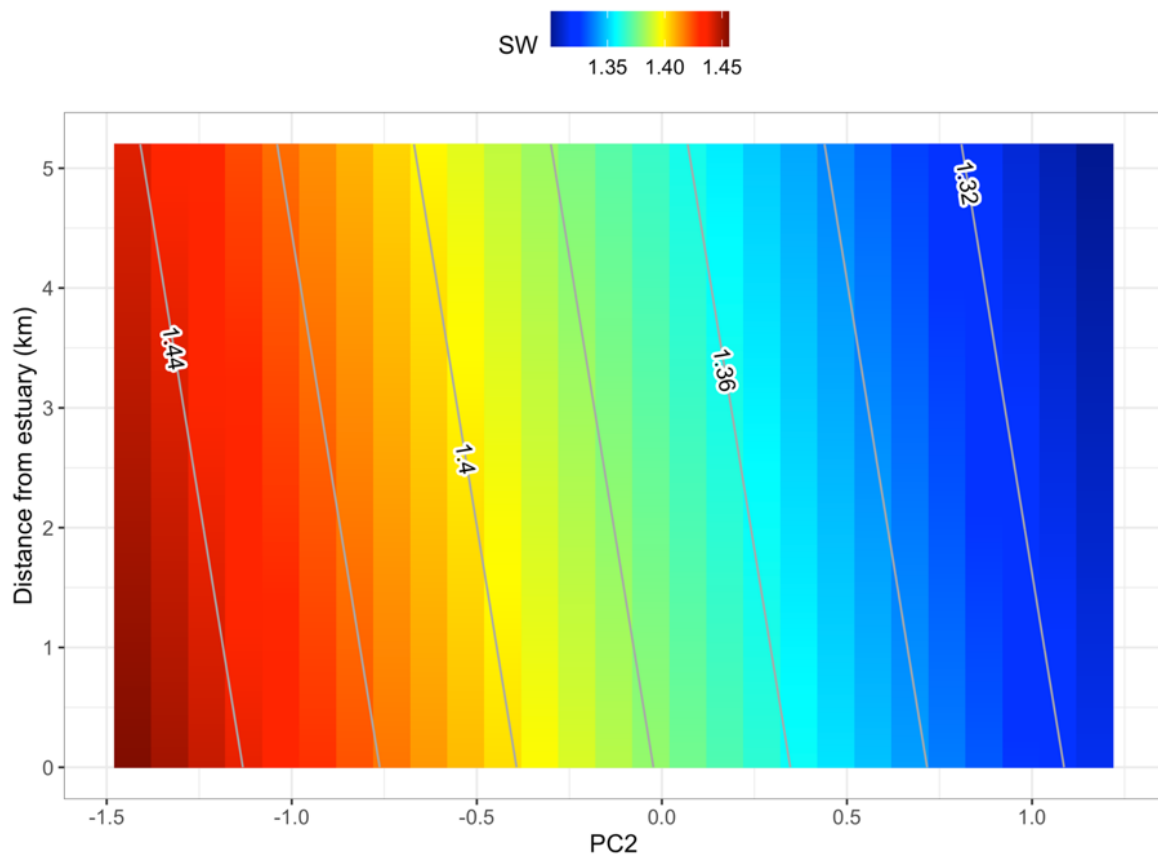


Figure 14. Model 4, the most supported model for macroinvertebrate diversity with distance from estuary included as an effect (Table 3). The effects of both PC2 and distance from estuary (km) on the Shannon Wiener (SW) diversity index is shown in this figure.

Candidate model 11 (Table 3, Figure 15) showed an initially higher SW diversity for weirs, but all measures showed a similar increasing slope. Channelization was predicted for 0 years since last measure (as explained in method-section, chapter 2.7). Results from an ANOVA-test yielded the effects of type of restoration measure and time since last measure not statistically significant (Table 6, bottom).

*Table 6. Parameter estimates and results from ANOVA-test for model 11, the most supported model with type of measure (tom) and time since last measure (tslm) included as effects according to AIC model selection (Table 3). tslm = time since last measure;  $tslm^2$  = a quadric effect of time since last measure; tom = type of restoration measure.*

<b>Parameter estimates</b>						
<b>Parameters</b>	<b>Category level</b>	<b>Estimate</b>	<b>Std. Error</b>	<b>t value</b>		
Intercept		1.366	0.086	15.979		
tomWEI	Weirs	0.040	0.275	0.145		
tomSID	Side channels	-0.191	0.308	-0.618		
tomRIP	Riparian modifications	-0.163	0.310	-0.528		
tslm		-0.025	0.059	-0.416		
$tslm^2$		0.0033	0.003	0.999		
<b>ANOVA</b>						
<b>Effect</b>		<b>Df</b>	<b>Sum Sq</b>	<b>Mean Sq</b>	<b>F value</b>	<b>Pr(&gt;F)</b>
tom		3	0.083	0.028	0.380	0.768
$tslm^2$		2	0.283	0.141	1.936	0.158
Residuals		39	2.849	0.073		

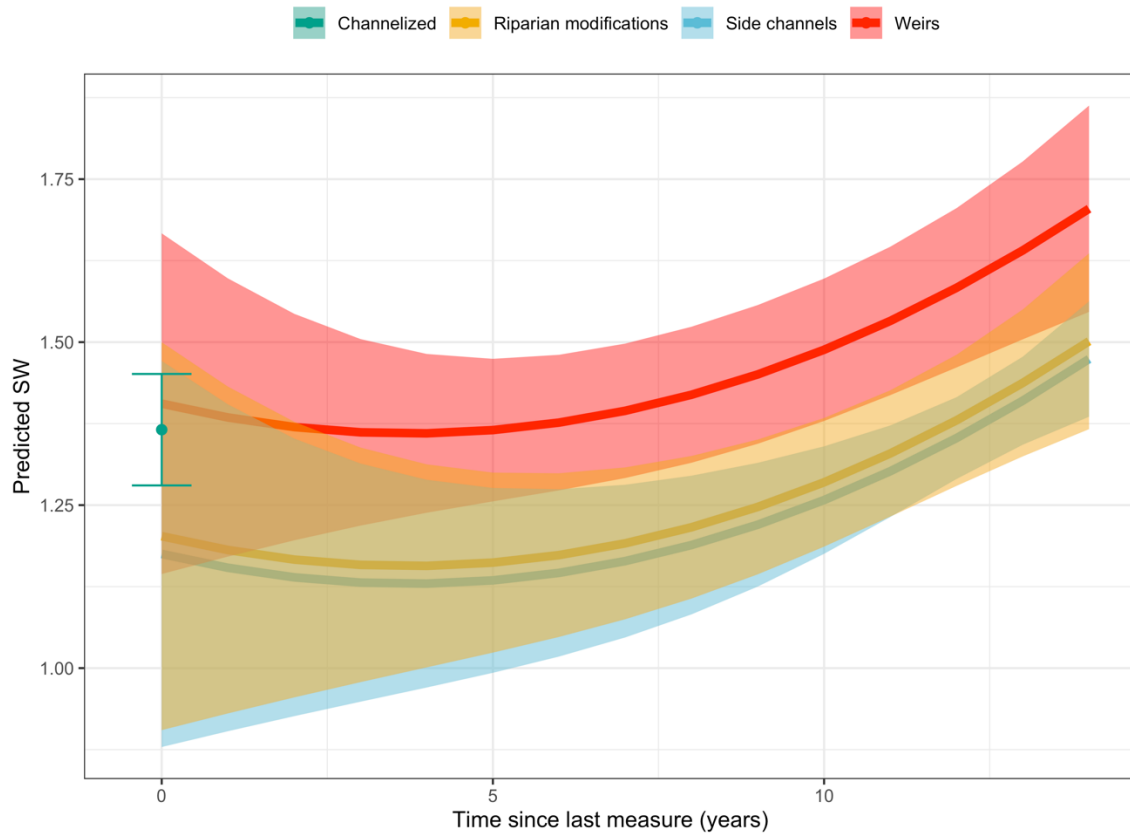


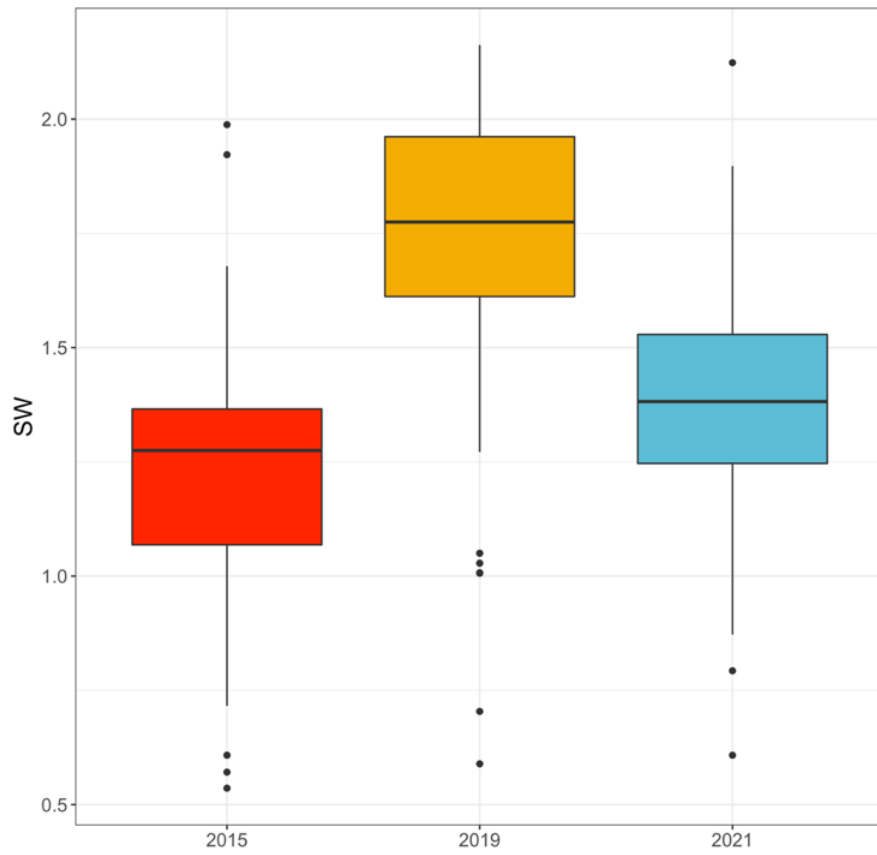
Figure 15. Model 11 (Table 3), showing the predicted Shannon Wiener index and time since last measure in years (*tslm*), sorted by type of measure. The shaded ribbon shows the standard error. Channelization have *tslm*=0 and can therefore not be predicted beyond 0 years. Thus, it is shown as a point in this plot.

### 3.3.2 Macroinvertebrate diversity 2015-2021

The SW-index in 2015, 2019 and 2021 were significantly different ( $F = 35.137$ ,  $p < 0.0001$ , Table 7). The mean SW-index in 2015 was 1.22, it peaked in 2019 with a mean of 1.76 and decreased in 2021 to 1.38 (Figure 16).

Table 7. Effect of year on Shannon Wiener diversity index for the years 2015, 2019 and 2021 (ANOVA).

ANOVA					
Effect	Df	Sum Sq	Mean Sq	F value	Pr(>F)
year	2	6.958	3.479	35.137	<0.0001
Residuals	160	15.842	0.099		



*Figure 16. Macroinvertebrate diversity in Bognely for all years sampled (measured as Shannon Wiener diversity index, SW). The black horizontal line represents the median value, while the colored box covers the 50 % interquartile range (IQR), and 1.5\*IQR is shown with black vertical lines.*

### 3.4 Macroinvertebrate abundance

The abundance of macroinvertebrates was lower in the control zones (11 and 12) and higher in the mid-zones (Figure 17), and there was substantial within-zone variation.

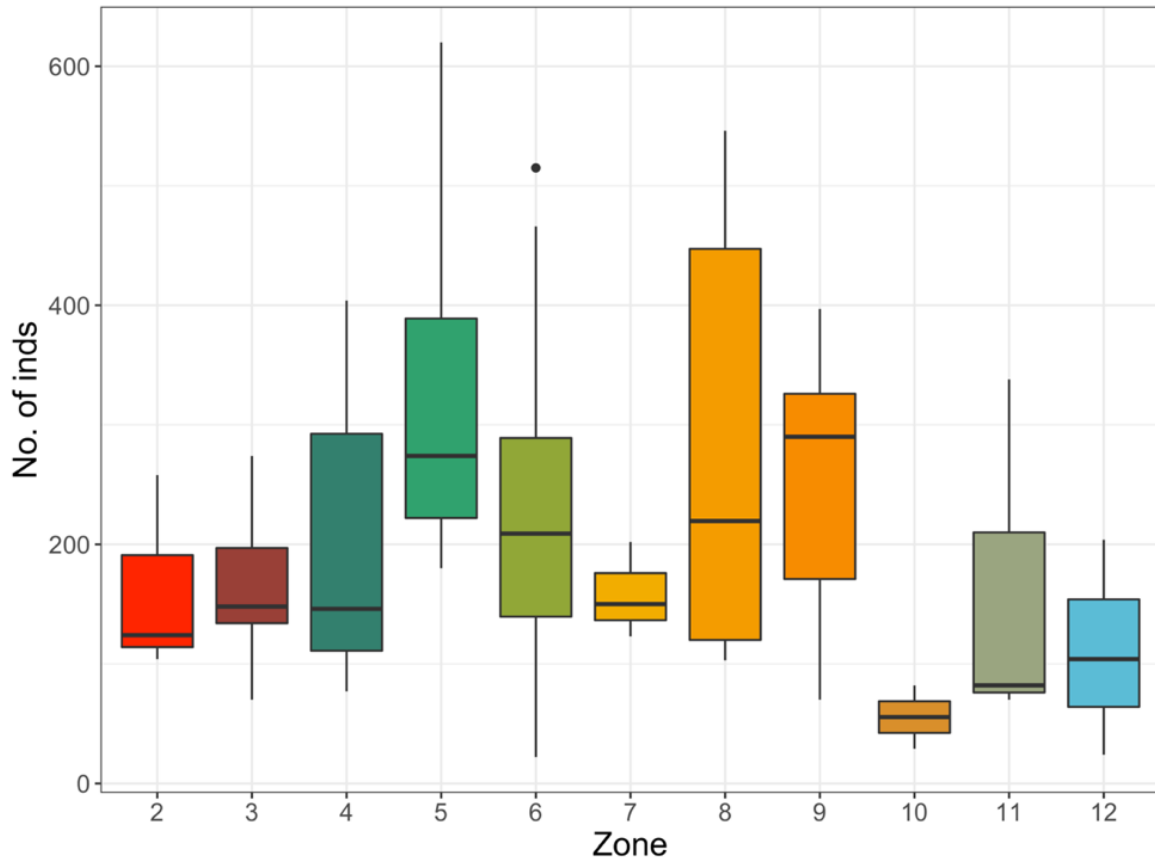


Figure 17. Boxplot of macroinvertebrate abundance per zone in Bognely. The black horizontal line represents the median value, while the colored box covers the 50 % interquartile range (IQR), and  $1.5 \cdot IQR$  is shown with black vertical lines.

#### 3.4.1 Correlates and effects of macroinvertebrate abundance

AIC model selection among candidate models fitted to macroinvertebrate abundance data favored an interaction model with type of measure (tom), a quadric effect of time since last measure (tslm<sup>2</sup>) and distance from estuary as predictors (model 1, Table 8). This model attained substantially higher AICc support in the data than other candidate models.

Table 8. AICc-based generalized linear model selection for the 10 candidate models fitted to predict macroinvertebrate abundance (number of individuals=No..Ind) based on data from Bognely 2021. tom = type of measure; tslm = time since last measure (years); tslm<sup>2</sup> = a quadric effect of time since last measure; dist.estuary = distance from estuary (km); PC1, PC2 and PC3 = environmental variables (Table 2). K = number of estimated parameters; AICcWt= the model AICc weight; Cum.Wt=cumulative weight; LL=model log likelihood. All models were fitted using log-likelihood method.

Model no.	Model structure	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
1	tom * tslm <sup>2</sup> * dist.estuary	20	2696.43	0.00	1	1	-1310.72
2	tom * PC1 + PC3	9	3020.67	324.23	0	1	-1498.76
3	tom * PC1	8	3038.92	342.49	0	1	-1509.46
4	tom * tslm <sup>2</sup> * PC2	20	3059.94	363.51	0	1	-1492.47
5	dist.estuary * tom + PC2	9	3318.19	621.76	0	1	-1647.53
6	dist.estuary * tom + PC3	9	3331.47	635.04	0	1	-1654.16
7	dist.estuary * tom + PC1	9	3381.59	685.15	0	1	-1679.22
8	tslm * PC1 + PC3	5	3404.17	707.74	0	1	-1696.32
9	tslm * PC1	4	3411.64	715.21	0	1	-1701.32
10	tom * tslm <sup>2</sup> + PC2	11	3487.12	790.68	0	1	-1728.56

The selected macroinvertebrate model (model 1, Table 8) predicted weir measure stations to have a sharp increase in abundance approximately eight to ten years after the measure was conducted (Figure 18). For side channel measures the abundance peaked between two and three kilometers upstream and approximately ten years post-measure. In stations that had undergone riparian modifications, predicted abundance sharply increased from one kilometer upstream, and at five years post-measure. The predictions in areas with few observations are less reliable (white dots in Figure 18 represent observations). An ANOVA-test yielded the effect of type of restoration measure (tom), time since last restoration measure (tslm) and distance from estuary as statistically significant (Table 9, bottom).

Table 9. Parameter estimates and results from ANOVA-test for model 1, the most supported model according to AIC model selection (Table 8). *tslm* = time since last measure (years); *tslm2* = a quadric effect of time since last measure; *tom* = type of measure; *dist.estuary* = distance from estuary (km).

<b>Parameter estimates</b>					
<b>Parameters</b>	<b>Category level</b>	<b>Estimate</b>	<b>Std. Error</b>	<b>z value</b>	
Intercept		5.687e+00	3.313e-02	171.637	
tomWEI	Weirs	1.887e+01	3.207e+00	5.884	
tomSID	Side channels	2.829e+01	2.049e+00	13.808	
tomRIP	Riparian modifications	-1.997e+01	6.316e+00	-3.162	
<i>tslm</i>		3.494e+00	1.431e+00	2.442	
<i>tslm</i> <sup>2</sup>		-1.599e-01	8.198e-02	-1.951	
<i>dist.estuary</i>		-2.796e-05	1.809e-05	-1.546	
tomWEI* <i>tslm</i>	Weirs	-1.819e+01	2.990e+00	-6.082	
tomSID <i>tslm</i>	Side channels	-9.543e+00	1.481e+00	-6.444	
tomRIP <i>tslm</i>	Riparian modifications	NA	NA	NA	
tomWEI <i>tslm</i> <sup>2</sup>	Weirs	1.874e+00	3.210e-01	5.836	
tomSID <i>tslm</i> <sup>2</sup>	Side channels	4.581e-01	8.389e-02	5.461	
tomRIP <i>tslm</i> <sup>2</sup>	Riparian modifications	NA	NA	NA	
tomWEI* <i>dist.estuary</i>	Weirs	-1.807e-02	3.042e-03	-5.939	
tomSID* <i>dist.estuary</i>	Side channels	-1.881e-02	1.090e-03	-17.265	
tomRIP* <i>dist.estuary</i>	Riparian modifications	2.499e-02	5.342e-03	4.678	
<i>tslm</i> * <i>dist.estuary</i>		-4.248e-03	1.067e-03	-3.981	
<i>tslm</i> <sup>2</sup> * <i>dist.estuary</i>		1.812e-04	5.470e-05	3.313	
tomWEI* <i>tslm</i> * <i>dist.estuary</i>	Weirs	1.146e-02	1.647e-03	6.955	
tomSID* <i>tslm</i> * <i>dist.estuary</i>	Side channels	8.283e-03	1.090e-03	7.597	
tomRIP* <i>tslm</i> * <i>dist.estuary</i>	Riparian modifications	NA	NA	NA	
tomWEI* <i>tslm</i> <sup>2</sup> * <i>dist.estuary</i>	Weirs	-8.441e-04	1.298e-04	-6.505	
tomSID* <i>tslm</i> <sup>2</sup> * <i>dist.estuary</i>	Side channels	-3.836e-04	5.584e-05	-6.870	
tomRIP* <i>tslm</i> <sup>2</sup> * <i>dist.estuary</i>	Riparian modifications	NA	NA	NA	
<b>ANOVA</b>					
<b>Effect</b>	<b>Df</b>	<b>Deviance</b>	<b>Resid. Df</b>	<b>Resid. Dev</b>	<b>Pr(&gt;Chi)</b>
NULL			44	3967.8	
tom	3	246.38	41	3721.5	<0.0001
<i>tslm</i> <sup>2</sup>	2	35.71	39	3685.8	<0.0001
<i>dist.estuary</i>	1	153.94	38	3531.8	<0.0001
tom* <i>tslm</i> <sup>2</sup>	4	320.68	34	3211.1	<0.0001
tom* <i>dist.estuary</i>	3	352.98	31	2858.2	<0.0001
<i>tslm</i> <sup>2</sup> * <i>dist.estuary</i>	2	229.48	29	2628.7	<0.0001
tom* <i>tslm</i> <sup>2</sup> * <i>dist.estuary</i>	4	322.02	25	2306.7	<0.0001

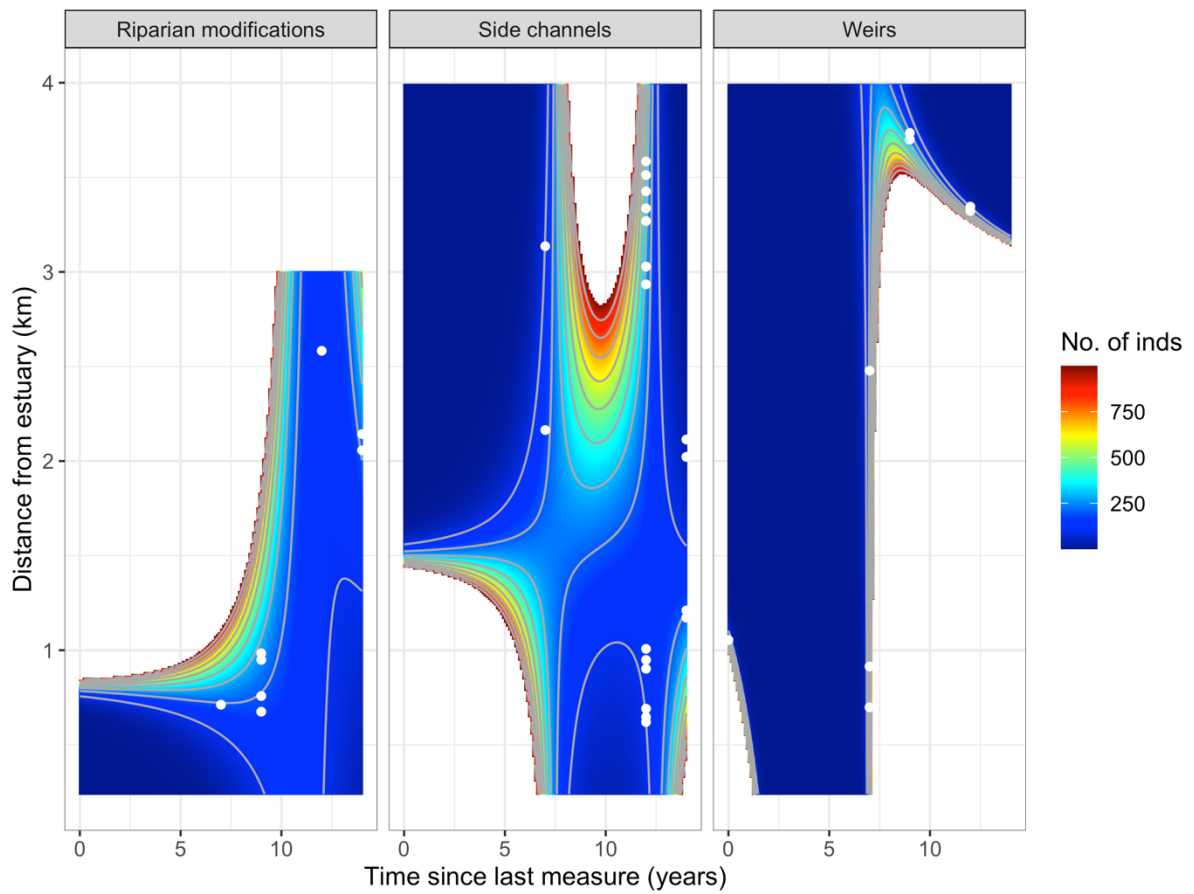
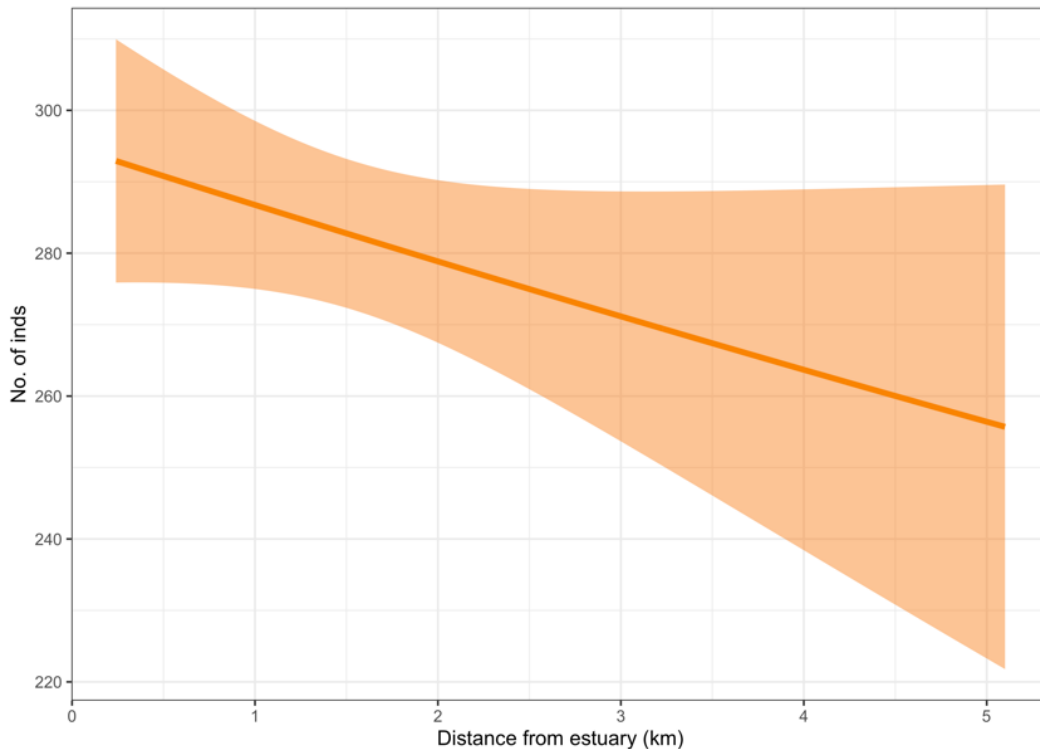


Figure 18. Contour plot of model 1 (Table 8), the most supported model for predicted macroinvertebrate abundance in our data. The figure shows the effects of time since last measure in years, the distance from the estuary in km and type of restoration measure on the predicted macroinvertebrate abundance. White dots represent observations.

Predicted abundance for channelized stations is shown in a separate plot, with distance from estuary as the predictor variable (Figure 19). This figure is also an illustration of model 1 (Table 8). As distance from estuary increased, abundance decreased in channelized parts of the river.





*Figure 19. Model 1 (Table 8) illustrating the effect of distance from estuary on macroinvertebrate abundance in channelized stations only, as the time aspect is not relevant in channelized stations. The ribbon represents the 95 % confidence interval.*

A prediction plot of model 2 (Table 8), the second most supported model for macroinvertebrate abundance, showed that predicted abundance increased in channelized stations with increasing PC3 and PC1 scores. Side channel measure stations showed a high abundance for low PC1 scores and high abundance for all PC3 scores (Figure 20). In addition, side channel measures and channelization yielded highest predicted abundance out of all four restoration measures. Weir measure stations showed an increase in abundance with increasing PC1 and PC3 scores but had overall low abundance. Riparian measure stations had lowest predicted abundance but showed a slight increase with higher PC1 and PC3 scores. These effects are statistically significant according to the ANOVA-test (Table 10, bottom).

Table 10. Parameter estimates and results from ANOVA-test for model 2, the second most supported model from our AIC model selection (Table 8). tom = type of restoration measure; PC1 and PC3 = principal components of environmental variables (Table 2).

<b>Parameter estimates</b>				
<b>Parameter</b>	<b>Category level</b>	<b>Estimate</b>	<b>Std. Error</b>	<b>z value</b>
Intercept		5.641	0.019	290.371
tomWEI	Weirs	-0.241	0.032	-7.653
tomSID	Side channels	-0.278	0.026	-10.720
tomRIP	Riparian modifications	-0.292	0.032	-9.044
PC1		0.327	0.026	12.593
PC3		0.077	0.017	4.654
tomWEI*PC1	Weirs	-0.157	0.047	-3.356
tomSID*PC1	Side channels	-1.078	0.036	-29.632
tomRIP*PC1	Riparian modifications	-0.253	0.049	-5.148

<b>ANOVA</b>					
<b>Effect</b>	<b>Df</b>	<b>Deviance</b>	<b>Resid. Df</b>	<b>Resid. Dev</b>	<b>Pr(&gt;Chi)</b>
NULL			44	3967.8	
tom	3	246.38	41	3721.5	<0.0001
PC1	1	64.14	40	3657.3	<0.0001
PC3	1	3.92	39	3653.4	0.04765*
tom*PC1	3	970.66	36	2682.7	<0.0001

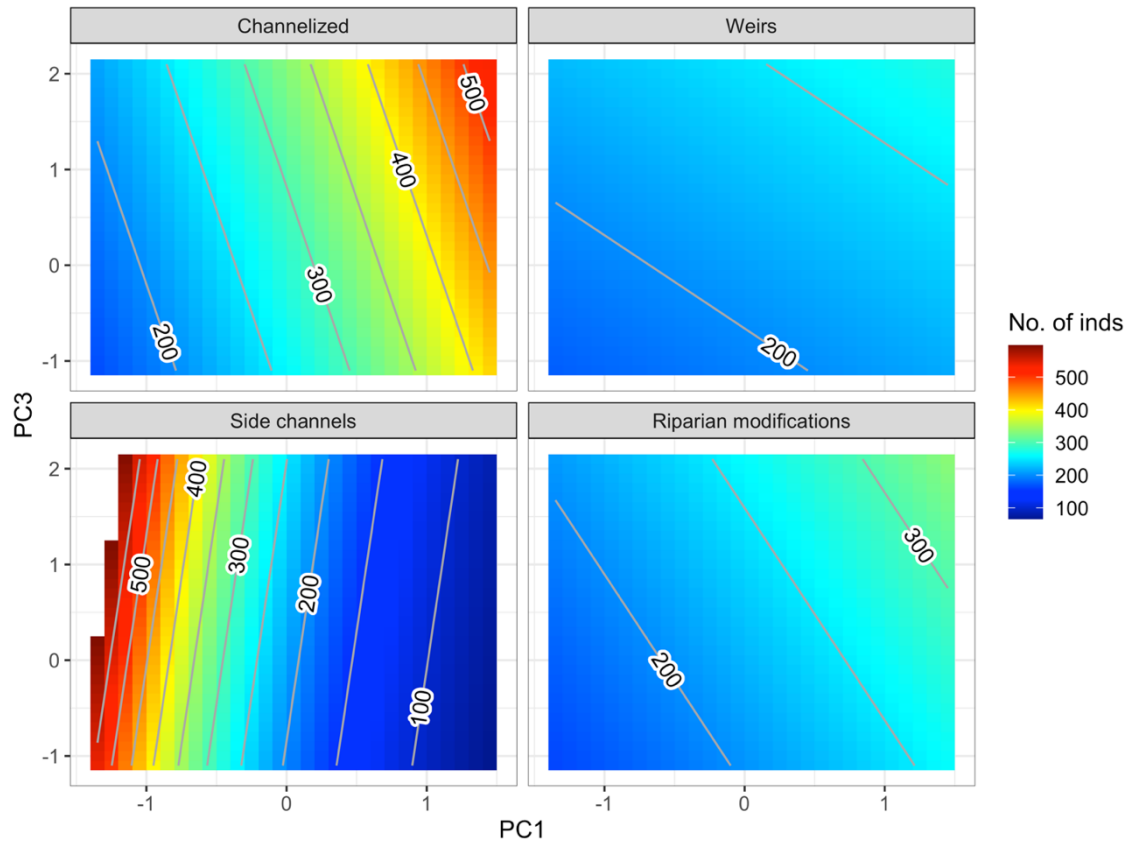


Figure 20. Model 2 (Table 8), the second most supported model for macroinvertebrate abundance. It shows the effects of PC1 and PC2 (Table 2, Figure 8) on predicted abundance for each type of restoration measure.

### 3.4.2 Macroinvertebrate abundance 2015-2021

The macroinvertebrate abundance and variance has increased since 2015, with 2021 being the peak year for abundance so far (Figure 21).

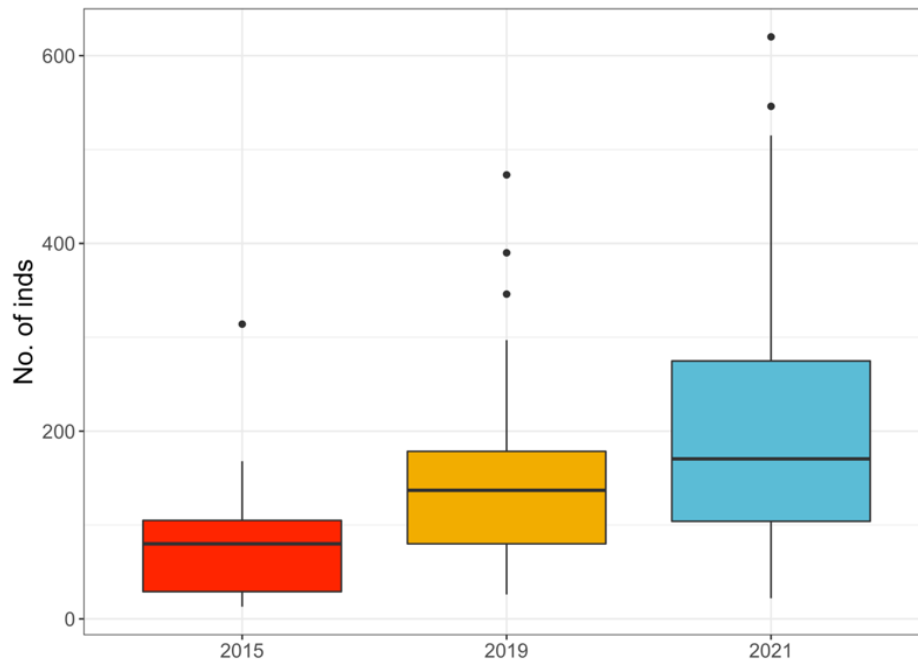


Figure 21. Macroinvertebrate abundance for all years sampled. The black horizontal line represents the median value, while the colored box covers the 50 % interquartile range (IQR), and 1.5\*IQR is shown with black vertical lines.

### 3.5 Ordination analysis of macroinvertebrates

A DCA on the macroinvertebrate data yielded a first axis length of 1.78, which supported linear regression for further analysis of the data (Table 11). We proceeded with a constrained redundancy analysis (RDA) to reveal the most supported explanatory variable.

Table 11. Results from a detrended correspondence analysis on the environmental variables on the macroinvertebrate data. Axis length for DCA 1 is between 1 and 3 which supports linear regression for further analysis.

	DCA1	DCA2	DCA3	DCA4
<b>Eigenvalues</b>	0.108	0.084	0.054	0.038
<b>Decorana values</b>	0.141	0.079	0.053	0.037
<b>Axis lengths</b>	1.779	1.309	1.417	0.940

A biplot for the selected RDA model showed that the macroinvertebrate community in stations with no measure differed significantly from stations with measures (Figure 22). Stations with side channel measures were positively associated with PC1 and PC2, while channelized stations were negatively associated with both principal components. Stations with weir measures showed an overlap in species composition with channelized, side channels and riparian modification stations.

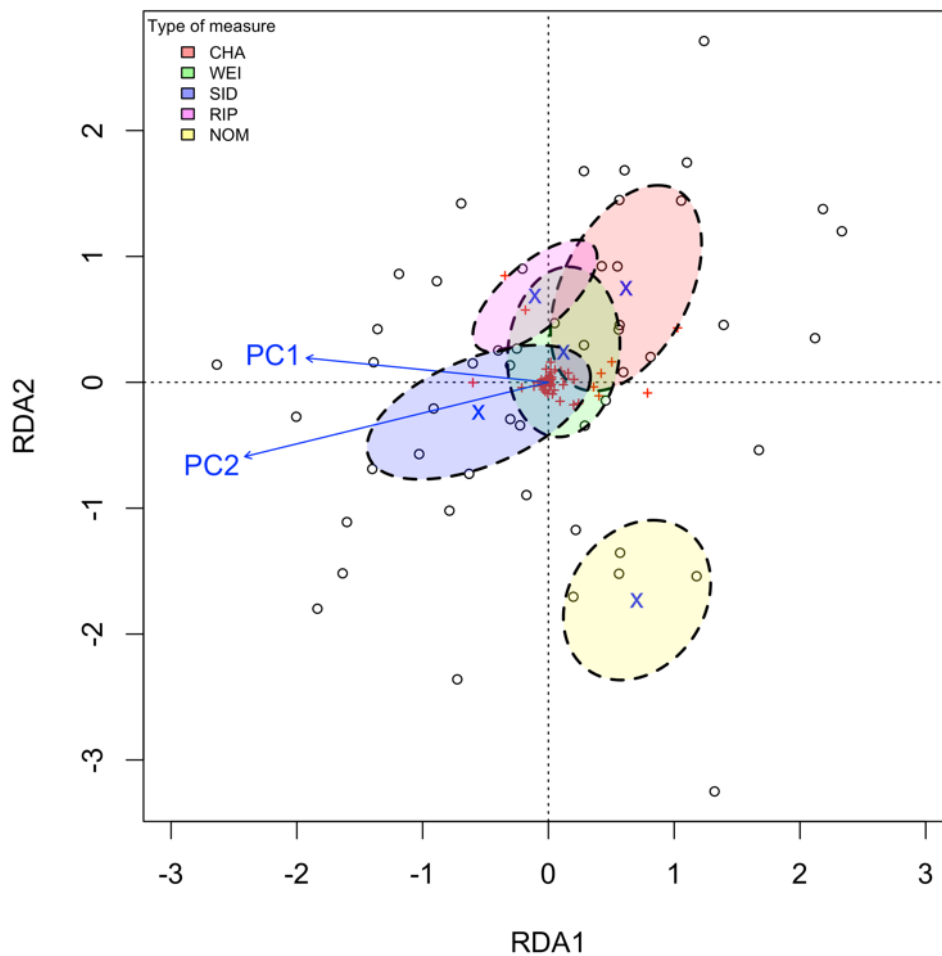


Figure 22. Biplot of the selected RDA model with principal components 1 and 2 (blue vectors). Type of measure is shown as 80 % centroids. Macroinvertebrate species are shown as red plus signs, and white dots represent stations.

### 3.6 Salmonids

Electrofishing yielded a total catch of 294 salmonids. Out of these, only one individual was Atlantic salmon, and the rest brown trout. No Arctic charr were caught.

#### 3.6.1 Age distribution

Age classes for sampled brown trout were defined by the length distribution (Figure 23). In total, we caught 30 individuals of 0+, 57 individuals of 1+ and 207 individuals of >1+. Compared to previous years, the body length of the 0+ age class was at its smallest in 2021 (Table 12).

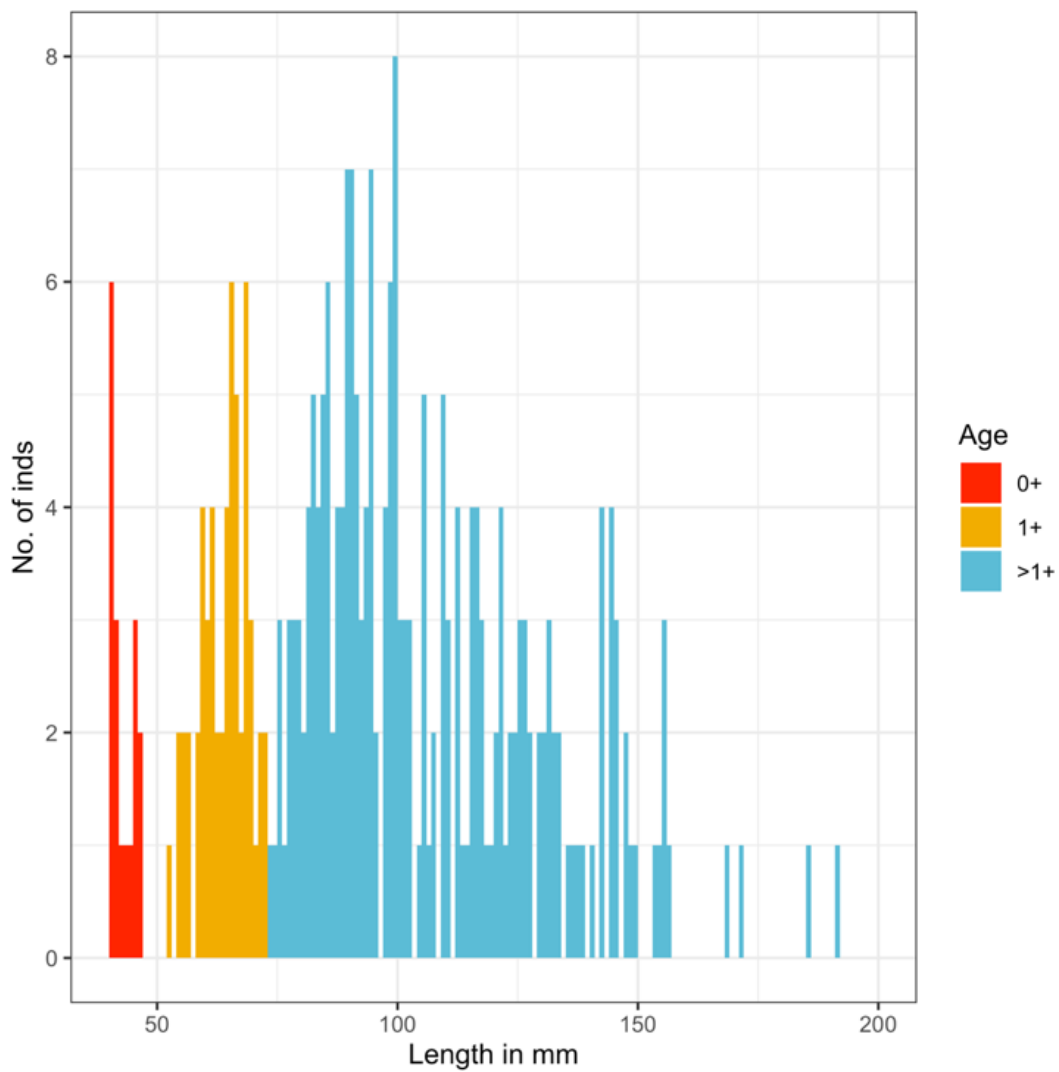


Figure 23. Length distribution of age groups for salmonid species captured in 2021.

Table 12. Length interval of captured fish and age groups from 2015, 2019 and 2021, measured in millimeters. For 2021, one Atlantic salmon is included in the length interval.

Year	Age groups		
	0+	1+	>1+
2015	31-57	58-88	>88
2019	25-50	51-90	>90
2021	32-47	48-74	>75

Figure 24 shows the distribution of fish lengths (mm) in the different zones. The majority of the 0+ age class was captured in zone 8. Most of the 1+ age class was captured in zone 3. The >1+ age class was more evenly distributed among zones. Zones 1, 6 and 9 had the highest number of captured >1+.

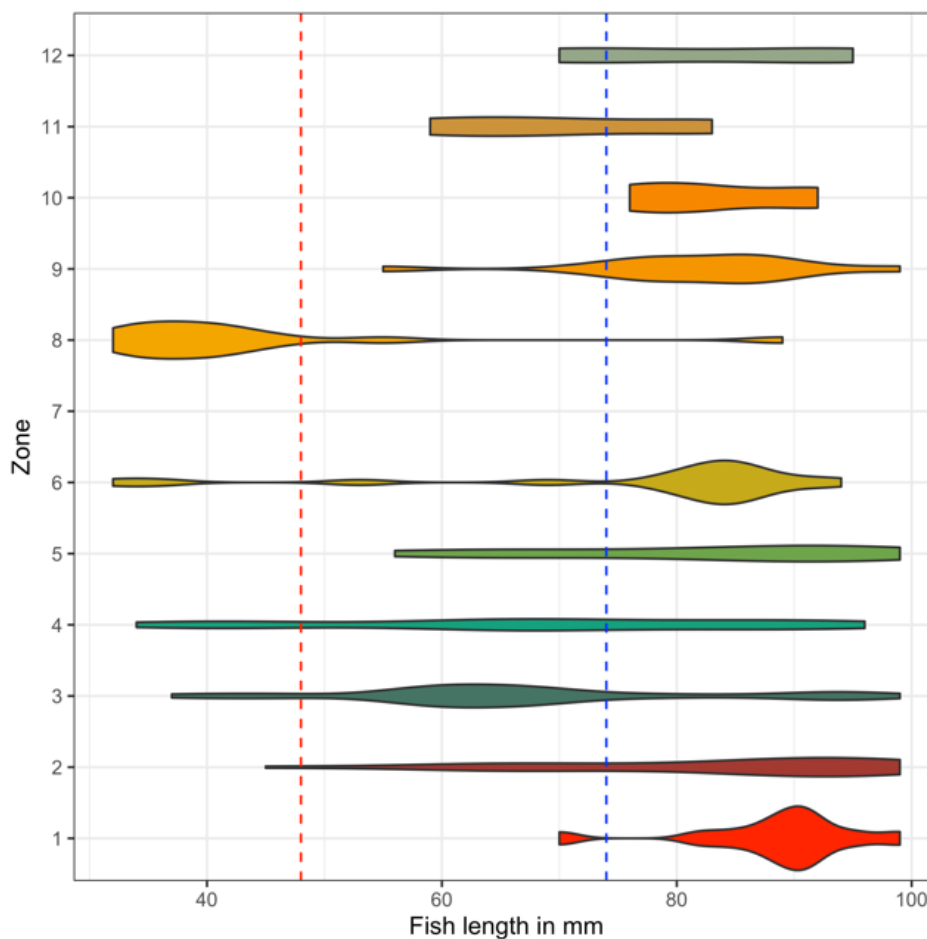


Figure 24. Fish length distribution by zone. To the left of the red dashed line is 0+, between the red and blue dashed line is 1+ and >1+ is to the right of the blue dashed line.

### 3.6.2 Density before and after restoration

The 0+ age class had increasing densities after 2004, peaking in 2011 (Figure 25). Since 2015, the densities of 0+ has decreased, and in 2021, 0+ densities were almost as low as before restoration. Densities of 1+ generally increased post-restoration and peaked in 2019. Year 2021 showed a drastic decline in 1+ densities. Ages >1+ showed a more stable trend post-restoration, with higher densities in 2008, 2013 and 2015 than in 2011, 2019 and 2021. The >1+ have shown a reduction in densities since 2015.

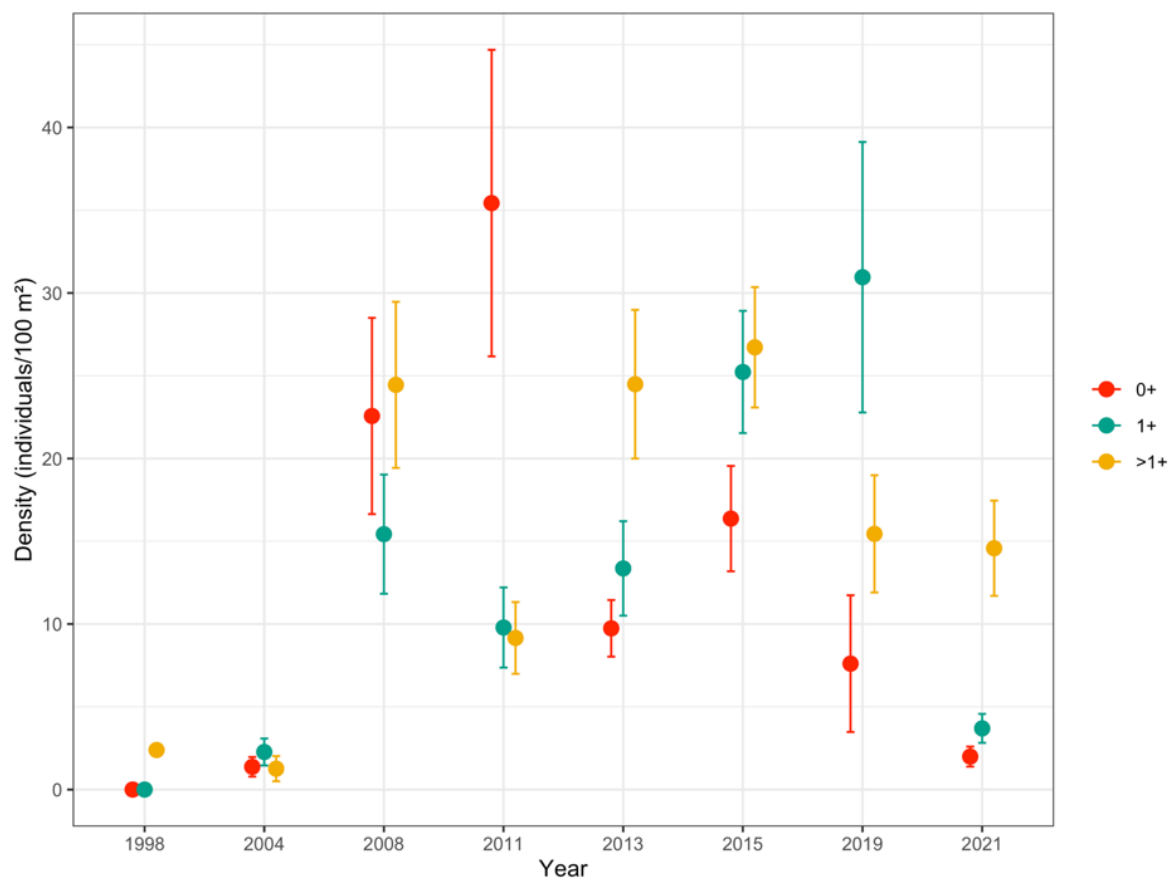


Figure 25. Juvenile brown trout densities before (1998-2004) and after (2008-2021) restoration, separated by age class. Standard error bars are included.

Density of 0+ in the control zones 11 and 12 have been extremely low for almost all years sampled, with 2015 being the exception (Figure 26). The middle zones have had a generally high density of >1+, compared to the other age classes.



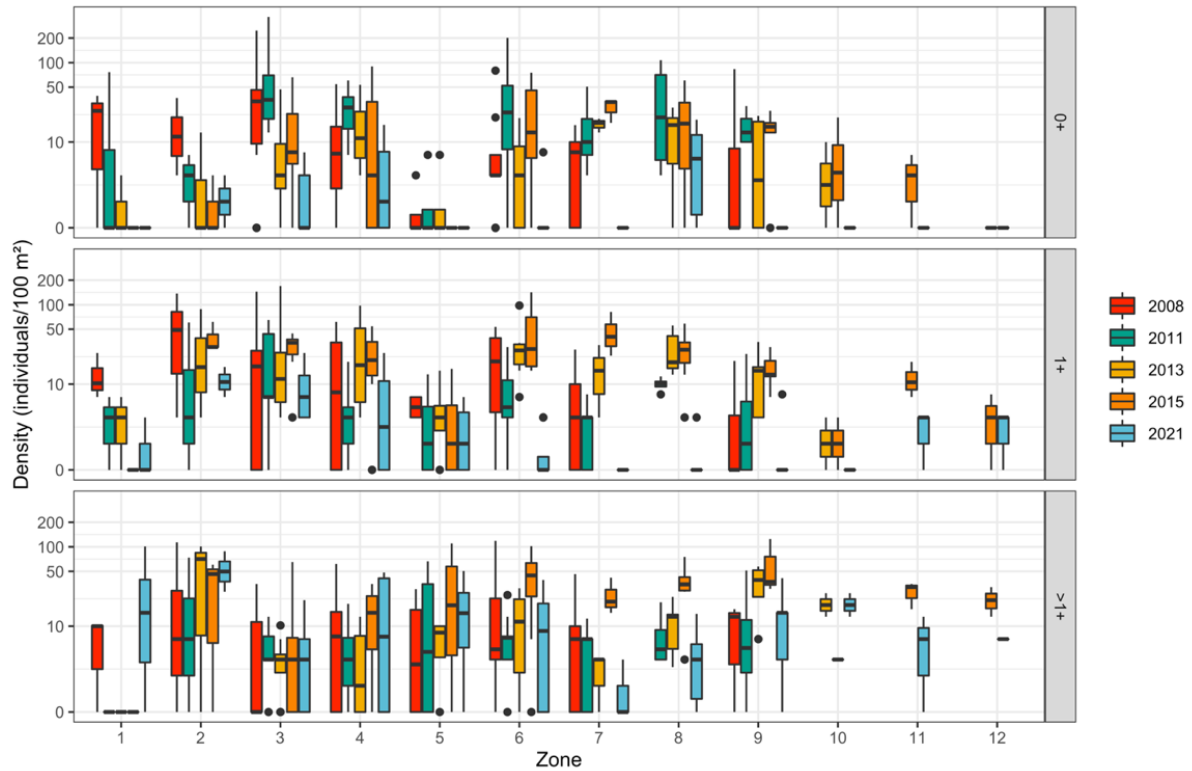


Figure 26. Age classes of brown trout separated by zone and year. Zone 10 was added in 2013, zone 11 and 12 were added in 2015. Plotted in log-scale. The black horizontal line represents the median value, while the colored box covers the 50 % interquartile range (IQR), and  $1.5 \times IQR$  is shown with black vertical lines.

### 3.6.3 Correlates and effects for salmonid density

The model selection for predicted density of 1+ salmonids favored an additive model with density of >1+, distance from estuary and PC3 as predictor variables (Table 13, Model 1). The difference in AICc score between the top model and the second most supported model was only 1.15 points, and consecutive models followed close behind. The top model with time since last measure included as an effect (model 11) attained an AICc score of 122.98, which is only 3.87 points behind our top model.

Table 13. AICc-based linear model selection for the 10 candidate models fitted to predict density of 1+ salmonids based on brown trout catch data from Bognelv in 2021. tom=type of measure; tslm=time since last measure (years); tslm<sup>2</sup> = quadric effect of time since last measure; dist.estuary=distance from estuary (km); PC1, PC2 and PC3 = principal components of environmental variables (Table 2). K = number of estimated parameters; AICcWt= the model AICc weight; Cum.Wt=cumulative weight; LL=model log likelihood. All models were fitted using log-likelihood method. The top model with tslm as an effect is included in the table to address all hypotheses (model 11).

Model no.	Model structure	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
1	dist.estuary + PC3	5	119.11	0.00	0.22	0.22	-53.76
2	dist.estuary + PC1	5	120.26	1.15	0.12	0.34	-54.34
3	dist.estuary + PC2	5	120.42	1.31	0.11	0.45	-54.42
4	dist.estuary * tom + PC3	11	120.51	1.40	0.11	0.56	-45.13
5	dist.estuary * PC1	6	121.24	2.13	0.08	0.64	-53.48
6	dist.estuary * PC3	6	121.57	2.46	0.06	0.70	-53.65
7	dist.estuary * tom + PC1	11	121.88	2.78	0.05	0.76	-45.82
8	dist.estuary * tom + PC2	11	122.04	2.93	0.05	0.81	-45.90
9	dist.estuary * PC2	6	122.39	3.28	0.04	0.85	-54.06
10	dist.estuary * PC2 + PC3	7	122.89	3.78	0.03	0.88	-52.89
...							
11	tom+tslm <sup>2</sup> + dist.estuary	9	122.98	3.87	0.03	0.91	-49.84

Model 1 predicted 1+ densities to increase with increasing >1+ densities (Figure 27). Highest 1+ densities were found furthest downstream in the river and at high PC3 values. The effects of distance from estuary and density of >1+ was statistically significant, while PC3 was not statistically significant (Table 14, bottom).

Table 14. Parameter estimates and results from ANOVA-test for model 1, the most supported model according to AIC model selection (Table 13). *older.one.pluss* = >1+; *dist.estuary* = distance from estuary (km); *PC3* = principal component 3 from our PCA of environmental variables (Table 2).

Parameter estimates					
Parameters	Estimate	Std. Error	t value		
Intercept	1.649	0.317	5.205		
log( <i>older.one.pluss</i> + 1)	0.209	0.091	2.293		
<i>dist.estuary</i>	-0.001	0.000	-4.857		
PC3	0.244	0.200	1.219		
ANOVA					
Effect	Df	Sum Sq	Mean Sq	F value	Pr(>F)
log( <i>older.one.pluss</i> + 1)	1	7.634	7.634	10.291	0.003
<i>dist.estuary</i>	1	19.013	19.013	25.633	<0.0001
PC3	1	1.102	1.102	1.486	0.230
Residuals	40	29.670	0.742		

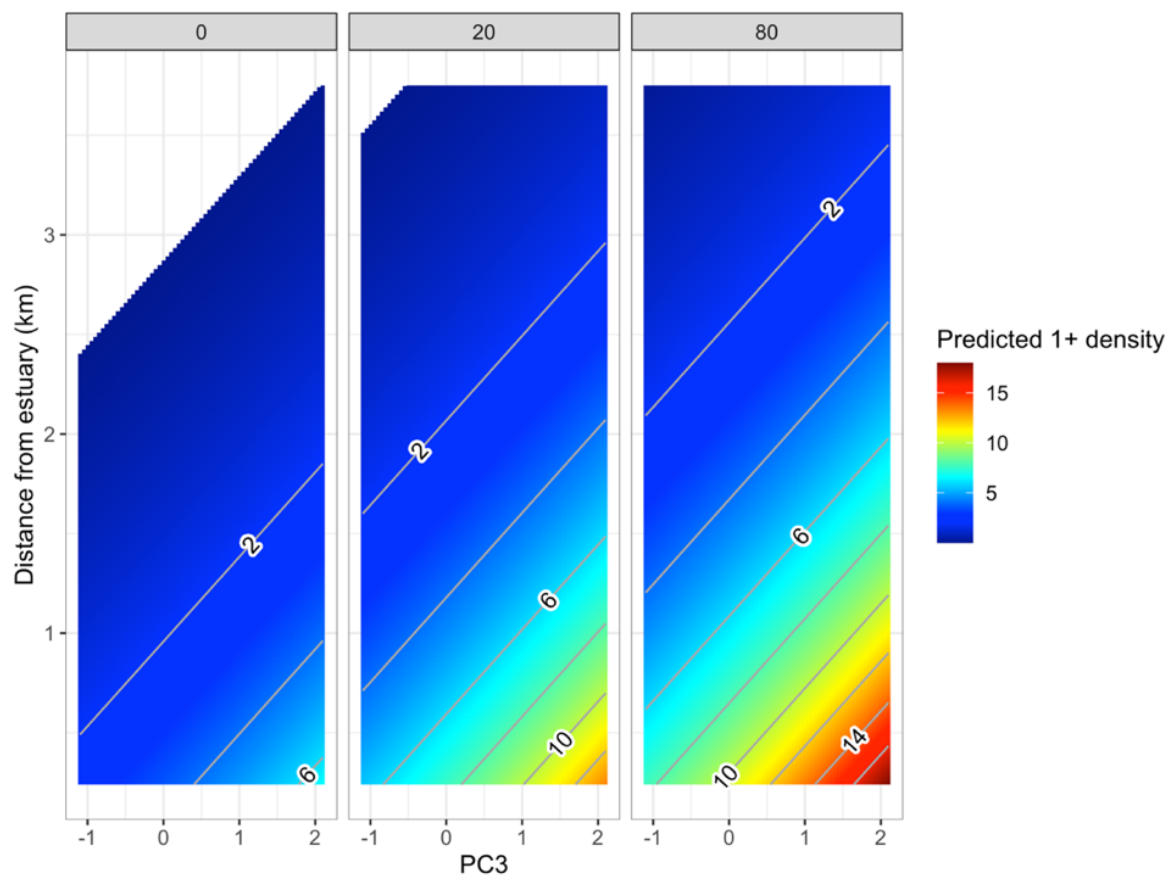


Figure 27. Model 1, the most supported model for 1+ brown trout density (Table 13). The model shows effects of distance from estuary (km) and PC3 and densities of >1+ brown trout on predicted densities of 1+. The three plots represent different >1+ densities: low (0), middle (20) and high (80).

The most supported model for 1+ density with type of restoration measure (tom) as an included effect showed that the effect changed with increasing distance from estuary, >1+ densities and PC3 values (model 4, Table 13). Highest density of 1+ was found closest to estuary, at high densities of >1+ and at high PC3 values (Figure 28). Weirs and riparian modification measures yielded higher 1+ densities than side channels and channelization. In addition, 1+ densities increased with higher >1+ densities. An ANOVA-test yielded statistically significant results for the effects of >1+ density, distance from estuary and type of restoration measure on 1+ density (Table 15, bottom).

*Table 15. Parameter estimates and results from ANOVA-test for model 4, the most supported model with type of restorations measure according to AIC model selection (Table 13). older.one.pluss = >1+; dist.estuary = distance from estuary (km); tom = type of restoration measure; PC3 = principal component 3 from our PCA of environmental variables (Table 2).*

<b>Parameter estimates</b>					
<b>Parameters</b>	<b>Category level</b>	<b>Estimate</b>	<b>Std. Error</b>	<b>t value</b>	
Intercept		9.776e-01	5.883e-01	1.662	
log(older.one.pluss + 1)		2.293e-01	9.237e-02	2.482	
dist.estuary		-4.867e-04	2.657e-04	-1.832	
tomWEI	Weirs	1.609e+00	7.881e-01	2.042	
tomSID	Side channels	3.681e-01	6.872e-01	0.536	
tomRIP	Riparian modifications	2.060e+00	7.736e-01	2.663	
PC3		2.508e-01	1.941e-01	1.292	
dist.estuary*tomWEI	Weirs	-2.631e-04	3.407e-04	-0.772	
dist.estuary*tomSID	Side channels	3.016e-05	3.251e-04	0.093	
dist.estuary*tomRIP	Riparian modifications	-1.104e-03	4.678e-04	-2.361	
<b>ANOVA</b>					
<b>Effect</b>	<b>Df</b>	<b>Sum Sq</b>	<b>Mean Sq</b>	<b>F value</b>	<b>Pr(&gt;F)</b>
log(older.one.pluss + 1)	1	7.634	7.634	12.953	<0.0001
dist.estuary	1	19.013	19.013	32.262	<0.0001
tom	3	5.651	1.884	3.196	0.036 *
PC3	1	0.364	0.364	0.618	0.437
dist.estuary*tom	3	4.719	1.573	2.669	0.063
Residuals	34	20.038	0.589		

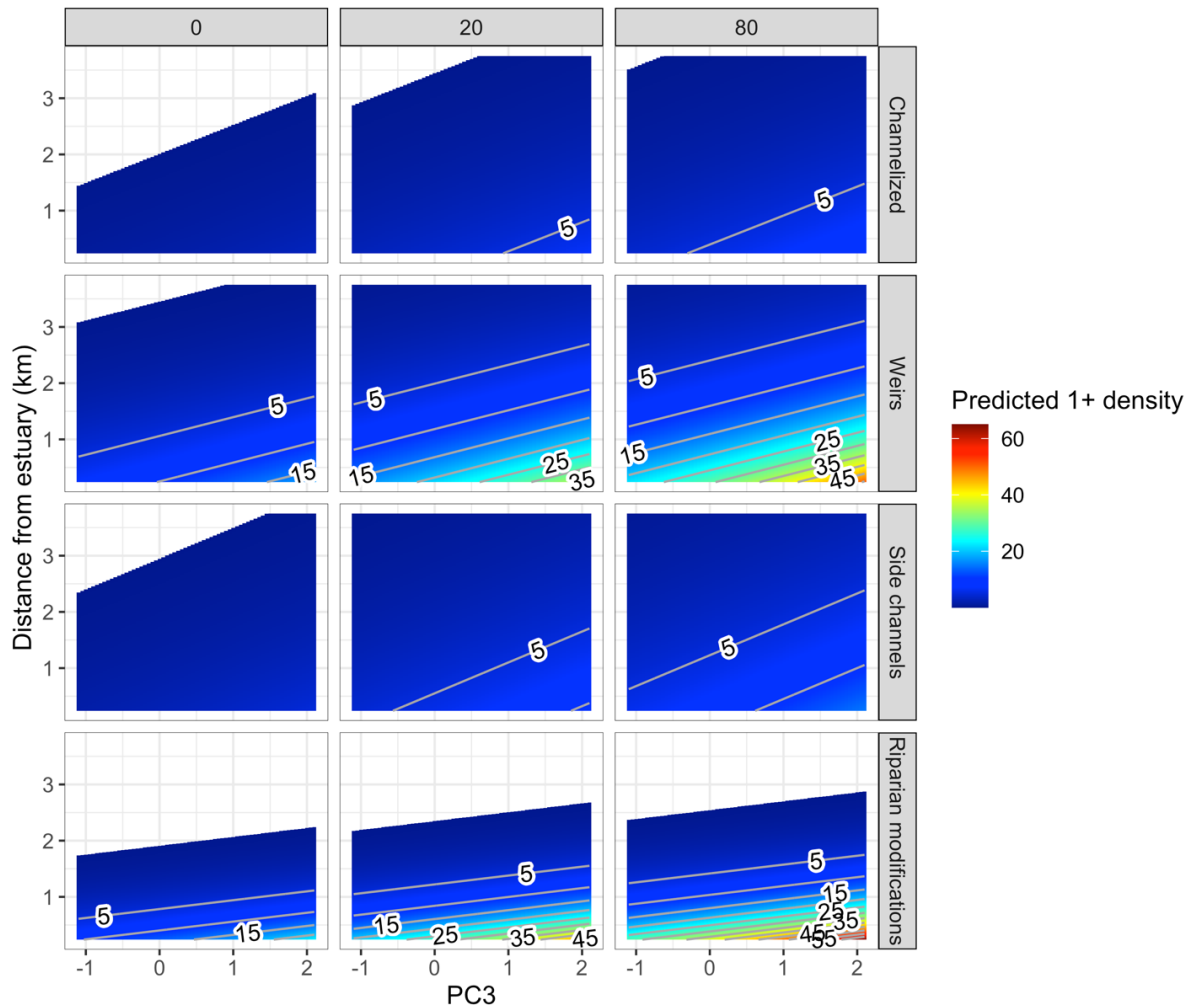


Figure 28. Model 4, the most supported model for 1+ brown trout density with type of restoration measure (tom) included as an effect (Table 13). The plot shows effects of distance from estuary (km), PC3, density of >1+ and type of restoration measure on 1+ density in Bognelv. The three columns of plots represent different >1+ brown trout densities: low (0), middle (20) and high (80). Each row of plots represents type of measure.

The highest densities of 1+ were found in weir measure stations with high densities of >1+ and short distance from estuary (Figure 29, model 11). Stations with side channels and riparian modifications showed a similar pattern of decreasing abundance of 1+ with increasing distance from estuary. Stations with low numbers of >1+ (plot column 0 in Figure 29) had very low abundance of 1+ for all measures. For all types of restoration measures, density of 1+ decreased with time since last measure, but this decrease seems to be leveling off. Effects of time and type of restoration measure were not statistically significant, but effect on densities of >1+ of distance was significant (Table 16, bottom).

Table 16. Parameter estimates and results from ANOVA-test for model 11, the most supported model with time since last restoration measure according to AIC model selection (Table 13). tom = type of restoration measure; tslm = time since last restoration measure (years); tslm<sup>2</sup> = quadric effect of time since last restoration measure; dist.estuary = distance from estuary (km).

<b>Parameter estimates</b>					
<b>Parameters</b>	<b>Category level</b>	<b>Estimate</b>	<b>Std. Error</b>	<b>t value</b>	
Intercept		1.154	0.383	3.046	
log(older.one.pluss + 1)		0.248	0.090	3.081	
tomWEI	Weirs	1.632	0.873	1.958	
tomSID	Side channels	1.181	0.994	0.753	
tomRIP	Riparian modifications	1.090	1.034	0.546	
tslm		-0.090	0.192	0.541	
tslm <sup>2</sup>		0.003	0.011	-0.640	
dist.estuary		-0.001	0.000	-5.405	
<b>ANOVA</b>					
<b>Effect</b>	<b>Df</b>	<b>Sum Sq</b>	<b>Mean Sq</b>	<b>F value</b>	<b>Pr(&gt;F)</b>
log(older.one.pluss + 1)	1	7.634	7.634	11.070	0.00203 **
tom	3	3.089	1.030	1.493	0.233
tslm <sup>2</sup>	2	3.490	1.745	2.530	0.094
dist.estuary	1	18.382	18.382	26.658	<0.0001
Residuals	36	24.824	0.690		

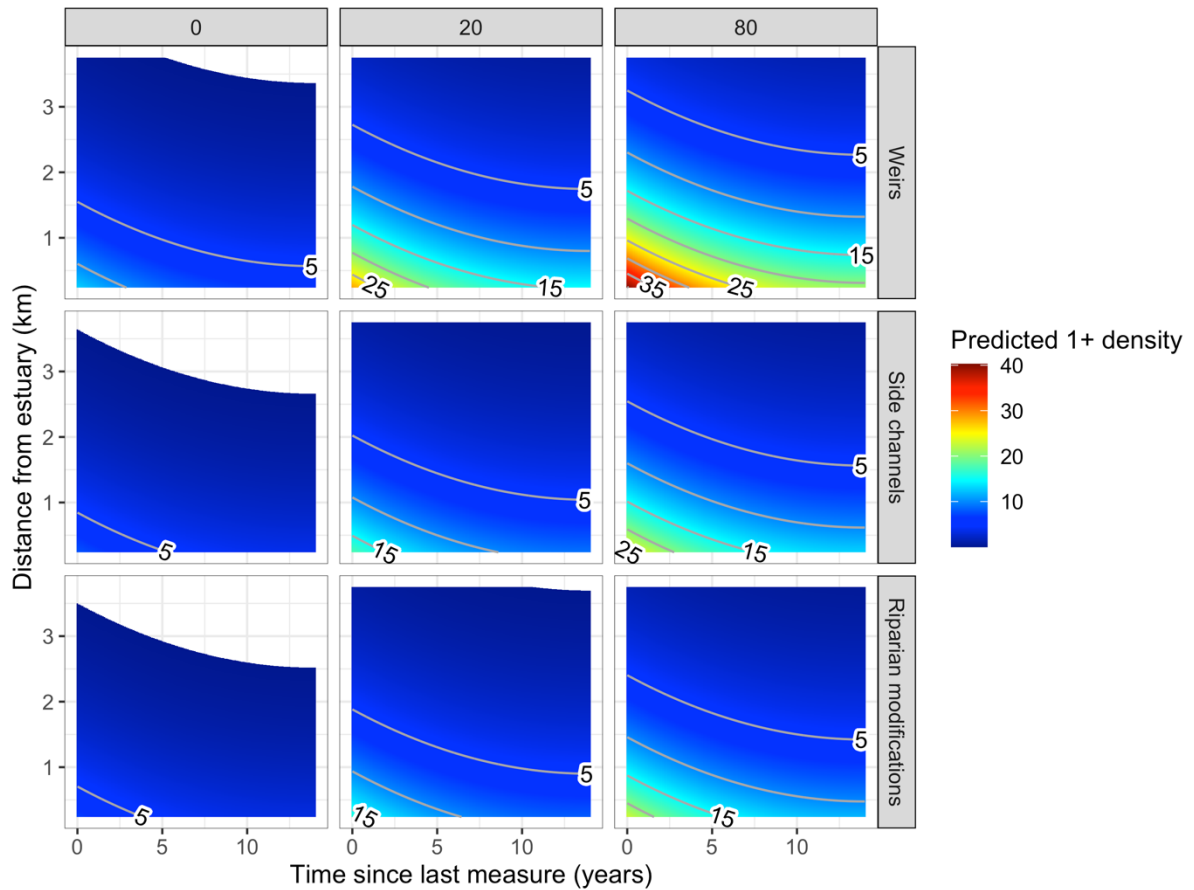


Figure 29. Model 11 (Table 13), the most supported model for 1+ density of brown trout with time since last restoration measure included as an effect. The plot shows effects of distance from estuary (km), time since last restoration measure (years), >1+ density and type of restoration measure on 1+ density in Bognelv. The three columns of plots represent different >1+ brown trout densities: low (0), middle (20) and high (80). Each row of plots represents type of measure.

## 4 Discussion

The starting point for this thesis was to investigate changes in the macroinvertebrate and brown trout community in Bognelv, and their responses to the restoration measures. Our results paint a complicated picture, with mixed responses that can be difficult to interpret. The main research questions will lead the way when discussing the results, with focus on three factors: *type of restoration measure*, *time since last measure*, and *distance from estuary*. Many of our results on the effects of these three factors on river biota are not statistically significant ( $p > 0.05$ ). This does not mean that they are not biologically significant, and we will discuss our results in the light of our research aims and AIC model selection. Furthermore, a non-significant result can be an important result in itself.

Other important aspects of the results that are not covered by our research questions will be discussed in the last section of this chapter. As mentioned in the method section, zero-inflation in our 0+ age-group data led to exclusion of 0+ from the analyses. The discussion will attempt to uncover reasons for the low amounts of 0+ in our sample.

### 4.1 What type of restoration measure has proven most successful in improving conditions?

#### 4.1.1 Effects of restoration measures on macroinvertebrate diversity and abundance

##### Diversity

Our results indicate that weirs have been the most successful restoration measure for increasing macroinvertebrate diversity. Weir measures in Bognelv include placement of rock clusters and groin dikes in the river to make the water flow more heterogenous and create new habitats. Figure 15 shows that diversity was initially higher in weir measure stations than other measures, and had a similar trend of increase as the other measures. However, the effect was not statistically significant (Table 6), and candidate models with type of measure included as an effect attained a high AICc score. This indicates that type of restoration measure is not a main driver for macroinvertebrate diversity. Still, these findings are interesting when comparing them to previous years.



The result of weirs having the highest macroinvertebrate diversity contradict earlier studies on macroinvertebrates in Bognelv. In 2015, macroinvertebrate diversity was predicted to be highest in stations with riparian modifications and second highest in side channel stations (Nordhov & Paulsen, 2016). Nordhov and Paulsen furthermore found that weirs had similar diversity as unrestored stations, in contrast to our results from 2021.

### Abundance

Macroinvertebrate abundance showed a different tendency, with highest predicted abundance in side channels and channelized stations (Figure 20). These results were statistically significant with a p-value  $< 0.05$  (Table 10). Interestingly, the two measures responded completely opposite of each other to PC1 and PC3. The PC1 axis was associated with vegetation, both riparian and in-stream, while PC3 was associated with pools and algae (Figure 8 & Figure 9). While channelized stations showed an increased abundance with increasing PC1-values, side channels went in the opposite direction. The abundance increased with decreasing PC1-values. This means that in channelized stations, the macroinvertebrate abundance increased with increasing vegetation, while the opposite is true for side channels. The response of channelized stations to PC3 is that abundance is higher at higher PC3-values, which means more pools and algae. Side channels did not respond strongly to PC3. Stations with weirs and riparian modifications showed a significantly lower abundance overall, and did not respond strongly to either PC1 or PC3.

These findings are in accordance with other studies investigating the effects of in-stream restoration measures on stream biota. Kail et al. (2015) found that instream measures such as placement of boulders and large woody debris tended to be more successful than other measures in increasing diversity and abundance of macroinvertebrates. Such measures diversify the water flow and create new habitats which are easily colonized, hence increasing macroinvertebrate abundance (Kail et al., 2015; Miller et al., 2010). Our results predicted higher abundance of macroinvertebrates in side channels, and lower in weir measure stations. Several side channel stations in Bognelv had high amounts of large woody debris. This creates pools with slow flowing, or still-standing water in front of the clusters of wood, where organic matter deposits. Chironomids and the mayfly *Baetis rhodani* were the dominant species in many of our samples, especially downstream zone 6. Chironomids are deposit collectors and prefer slow-flowing water (Lennox III & Rasmussen, 2016), while *B. rhodani*

prefers more fast-flowing stretches (Timm, 1997). The habitat preference of Chironomids might explain why we found a large abundance in side channels, while the abundance of *B. rhodani* might be due to other factors, like their tolerance to organic pollution from agriculture (see section 4.3.1 on the effect of agricultural runoff on the macroinvertebrate community).

Our findings contradict the results of Bjørngaard (2020), where she found side channels to have the lowest macroinvertebrate abundance out of all the conducted measures. This could mean that the habitats are changing from year to year, and that the side channels were more favorable for macroinvertebrate abundance in 2021 than in 2019.

Both in 2019 and in 2021, channelized stations were predicted to have some of the highest abundances of macroinvertebrates. This contrasts with the literature, and our own expectations. We expected channelized stations to have a lower abundance than the non-channelized stations, due to altered habitat conditions. Loss of habitat and increased flow velocity are major reasons for reduction in macroinvertebrate biomass due to channelization (Lennox III & Rasmussen, 2016). However, channelized stations in the lowermost part of Bognelv were generally shallow and with slower flowing water. The preference of Chironomids to shallow, slow-flowing stretches might explain the high abundance in channelized stations, at least those close to the estuary.

The study in 2015 did not investigate abundance of macroinvertebrates against type of restoration measure (Nordhov & Paulsen, 2016).

#### 4.1.2 Effects of restoration measures on brown trout density

Results from electrofishing in 2021 yielded highest predicted density of 1+ brown trout in weir measure and riparian modification stations, and very low predicted density in channelized and side channel stations (Figure 28). The effect was not statistically significant (Table 15), but the candidate model with type of measure included as an effect attained an AICc-score only 1.4 points behind our top model for 1+ density (Table 13). This indicates that type of measure did influence 1+ densities in Bognelv in 2021.

These findings on 1+ brown trout are interestingly similar to findings from 2015 and 2019, only for density of 0+ brown trout. Since we had to exclude 0+ from our sample in 2021, we are unable to compare the same age classes, and we should be careful with applying the same results to a different age class. However, 1+ brown trout in 2021 were generally very small (Table 12). One could therefore argue that some of the 1+ in 2021 would have had similar habitat preferences to the 0+ in 2015 and 2019. It is therefore relevant to compare 1+ results from 2021 with the 0+ results for brown trout from these two years.

In 2015, Nordhov and Paulsen found the greatest density of 0+ brown trout in stations which had undergone weir measures and riparian modifications. In 2021, 1+ brown trout densities were higher in weir measure and riparian modification stations. In 2019, Solvang Strand found highest densities of 0+ in areas with pools, while density of 1+ in 2021 increased with increasing presence of pools (increasing PC3-values) (Figure 28).

Our findings are supported by other studies on the effect of restoration measures on river biota. Instream measures, such as placement of boulders and large woody debris have been shown to have a substantial positive effect on fish densities (de Jong & Cowx, 2016; Kail et al., 2015; Louhi et al., 2016). Weirs are a type of instream measure, and placement of boulders and rock clusters have been commonly used as a restoration measure in Bognelv. Kail et al. (2015) found that instream measures yielded higher densities than riparian modifications, because these measures diversify the habitat by altering stream velocity and substrate conditions. More specifically, placement of boulders in combination with large woody debris have been shown to yield the greatest response on salmonid densities compared to other restoration measures (de Jong & Cowx, 2016; Louhi et al., 2016). Placement of large woody debris has not been used as a restoration measure in Bognelv as of yet, but our findings suggests that this should be a focus of future restoration efforts to further improve habitat conditions for brown trout.

Schedel (2011) and Austvik (2012) found reopening of side channels to be the most successful restoration measure, in contradiction to our results and the results from 2015 and 2019. This indicates that the effect of restoration measures on densities of brown trout is dynamic, and that it is not given which measure is most successful from year to year.

## 4.2 What effect does time have?

Time since last restoration measure varied between two and 14 years in our analyses, as some stations have had measures conducted in 2019, whereas other stations have not had any measures conducted since 2007 (Appendix 3).

### 4.2.1 Effects of time on macroinvertebrate diversity and abundance

#### Diversity

Our model selection yielded a model that predicted diversity to reach a bottom point at *six years post-restoration*, after which it was estimated to increase (Figure 13). The effect was not statistically significant (Table 4), but our top two AIC candidate models for macroinvertebrate diversity included time since last measure as the only predictor variable (Table 3). This signifies that the time needed by macroinvertebrates for recovery after disturbance is important in determining diversity in Bognelv.

These results are interesting when comparing them with results from Bjørngaard's study from 2020. She found that macroinvertebrate diversity in Bognelv peaked after approximately six years. Our findings suggest the complete opposite, with a sharp decline shortly after the measure was conducted and a bottom-point in diversity approximately six years post-restoration. Our results are in accordance with literature suggesting that diversity decreases rapidly after disturbance (Ruaro et al., 2016). The process of restoration creates a disturbance and habitat conditions post-restoration will likely get worse before they get better (Laasonen et al., 1998; Nilsson et al., 2017). One study conducted in channelized, cold-water streams found very little improvement in habitat diversity even 80 years after channelization (Lennox III & Rasmussen, 2016), suggesting that several decades are needed for recovery after disturbance in certain streams. Moreover, high latitude streams generally support fewer species than streams at low latitudes (Allan & Flecker, 1993). This suggests that there are fewer species to re-colonize habitats in cold-water streams and that the re-colonization time is therefore longer.

However, when comparing macroinvertebrate diversity with previous years, the predictions of a low point six years post-restoration is contradictory to the observed development of diversity. From 2015 to 2019, diversity increased, followed by a decrease between 2019 and

2021 (Figure 16). Restoration measures have been carried out in Bognelv in 2006, 2007, 2009, 2012, 2014, 2018 and 2019 (Appendix 3). The measures that were implemented in 2014 were substantial and covered an area up to zone 8 (Bjordal & Hoseth, 2014, Appendix 2). It is highly likely that these measures created a large disturbance that would have affected the macroinvertebrate community negatively. Bjørngaard (2020) suggested that the 2014 measures could have reduced the availability of flow refugia and therefore slowed down re-colonization, resulting in a low diversity in 2015 when sampling took place in Bognelv. Taking our six-year perspective into consideration, we would expect macroinvertebrate diversity to decline the following six years, resulting in a lower diversity in 2019 than in 2015. However, this is not the case. It might mean that the restoration measures conducted in 2014 improved conditions and therefore led to a more rapid recovery than expected. The decrease in diversity in 2021 compared to 2019 are more in line with our findings, as the measures conducted in 2018 created a disturbance that could account for the decline in macroinvertebrate diversity.

Another explanation for the decline in diversity between 2019 and 2021 could be that the sampling of macroinvertebrates in 2019 took place much later in the season than in 2021 (Bjørngaard, 2020). The macroinvertebrates would therefore have had more time to grow and would likely have been easier to identify to species. Small macroinvertebrates posed a challenge for identification down to species in 2021, and it is probable that this resulted in a lower diversity index. It is also important to keep in mind that the stations in 2019 are not the same as the stations in 2015 and 2021, and this might have affected the differences in diversity.

### Abundance

Our results showed a statistically significant effect of time since last restoration measure on macroinvertebrate abundance (Table 9). While diversity models predicted a bottom-point six years post-restoration, our top model for abundance predicted a peak for side channel measures at ten years post-restoration and for weirs at approximately eight years post-restoration (Figure 18). Several studies have found that macroinvertebrate abundance has the potential to increase rapidly after disturbance, especially with availability of flow refugia (Laasonen et al., 1998; Negishi et al., 2002). Laasonen et al. (1998) studied macroinvertebrate re-colonization in several rivers with varying time since restoration. They

found lowest abundance in the most recently restored streams, due to the disturbance of restoration. However, a rapid increase followed in the first few years post-restoration. Furthermore, they found that the increase in abundance leveled off after approximately eight years, which is similar to our findings of an eight to ten-year peak in abundance.

Our predictions of an eight to ten-year peak in macroinvertebrate abundance are coherent with the increase of abundance from 2015 to 2021 (Figure 21). The restoration measures conducted in Bognelv in 2014 were substantial, and probably led to an immediate loss of abundance. Consequently, 2015 was likely a year with low macroinvertebrate abundance, so soon after the conducted measures. One might expect the same to have happened in 2019, with the latest measures being conducted in 2018. However, the 2018 measures were less substantial than in 2014 and would likely not have resulted in an equally large loss of abundance. Also, as earlier mentioned, the 2019 measures were likely insignificant to the macroinvertebrate community (Bjørngaard, 2020), accounting for the increase in abundance from 2015 to 2019. No measures have been conducted since 2019, which explains the continued increase in abundance in 2021. If our prediction of an eight to ten-year peak in abundance is correct, the macroinvertebrate abundance in Bognelv will continue to increase for a couple of years before leveling off, and eventually decrease again.

#### 4.2.2 Effects of time on brown trout density

The effect of time on density of 1+ brown trout proved not to be statistically significant (Table 16). However, the top candidate model with the time aspect included as an effect attained an AICc score only 3.87 points behind our top model (Table 13). This indicates that time since last measure is an important factor for 1+ brown trout density in Bognelv. Our model predicted 1+ brown trout density to decrease following restoration, but after approximately ten years the curve appears to be flattening (Figure 29).

When comparing brown trout densities before restoration began in Bognelv, with the years following the first restoration measures in 2006, we see a sharp increase post-restoration (Figure 25). From 2004 to 2008, densities of brown trout went from almost zero to around 20 individuals per 100 square meter. After 2008, densities of brown trout have generally been high for all years sampled. However, 2021 represents a low-point, with densities of 0+ and 1+ being down to almost the same as before restoration began.

Upon closer inspection of brown trout densities in the years following 2015, the results are interesting. Our predictions of brown trout density showed a decrease for ten years post-restoration, with a subsequent flattening of the curve. Our sampling happened six years after Nordhov and Paulsen (2016) and the densities of both 0+ and >1+ have been steadily decreasing since then (Figure 25). Louhi et al. (2016) argued that monitoring should exceed ten years post-restoration to evaluate the success of a restoration project. Although ten years have passed since the initial restoration began in Bognelv, there have been continuous smaller restorations projects in the river to improve habitat conditions. In 2014, several measures were conducted, causing a substantial disturbance to the river system. This correlates with our predictions of a steady decline in brown trout densities ten years post-restoration. These results strengthen our belief that we are approaching a bottom-point of brown trout densities in Bognelv.

In addition to a decline in density, brown trout caught in 2021 were generally smaller than previous years (Table 12). In 2019, fishing was conducted later in the season, providing a longer growth season for the sampled fish. However, sampling in 2015 took place around the same time as in 2021, and so the large differences in size between these two years warrants a closer look. Since the macroinvertebrates sampled in 2021 were also very small, we hypothesized that something was limiting density and growth of brown trout and macroinvertebrates that particular year. One explanation could be a longer winter, which results in a late onset of spring.

Late onset of spring has the potential to limit brown trout growth, as their growth is positively correlated with temperature (Vøllestad et al., 2002). Compiled data on average snow depth in Langfjordbotn in May from the last 20 years showed a high average snow depth in 2021 compared to previous years, most notably 2015 and 2019 (Figure 5). From these data we can assume that spring arrived later in 2021 than in previously studied years, which provides one explanation for the unusual small and low amounts of 0+ in our sample. Another explanation is inter-cohort competition, which leads to older age classes limiting growth of 0+ brown trout. This will be further explained in chapter 4.4.

## 4.3 What effect does distance from estuary have?

### 4.3.1 Effects of distance from estuary on macroinvertebrate diversity and abundance

#### Diversity

We expected our results to show a difference in diversity between the lower and the upper sections of Bognelv, due to the different habitat conditions. The lower sections have been channelized and the upper sections can be seen as “undisturbed”, thus creating more habitat heterogeneity suitable for high macroinvertebrate diversity upstream (Garcia et al., 2012). However, our data revealed no such pattern (Figure 14), and the effect was not statistically significant (Table 5). Distance has been previously shown to be one of the most important variables for explaining macroinvertebrate diversity (Nordhov & Paulsen, 2016; Sødal, 2014). Although species composition varied between stations, our analysis did not detect significant changes in the Shannon Wiener diversity index.

#### Abundance

The distance effect is more noteworthy when looking at macroinvertebrate abundance. Our data showed a pattern of higher abundance in the middle stations (Figure 17). We expected the abundance of macroinvertebrates to follow a gradient according to the river continuum concept, with increasing abundance closer to the outlet (Muneepeerakul et al., 2008; Vannote et al., 1980). This was not the case, however. One explanation could be that agricultural runoff from the surrounding land in the middle zones creates generalist communities tolerant to agricultural pollutants (Karaouzas et al., 2007). This explanation is substantiated by the species composition in 2021. As earlier mentioned, we found an abundance of Chironomids and the mayfly *B. rhodani* in our samples. Chironomids are a pollution-tolerant family and are often the dominant taxa at sites with high levels of organic pollution. Mayflies are in general more sensitive to pollution, but Baetidae is one of the more pollution-tolerant families (Armitage et al., 1983; Arslan et al., 2016). *B. rhodani* was abundant in almost all stations, but the majority was found in the middle and lower parts of the river. This is reasonable, as the uppermost stretches of the river are not influenced by surrounding cultivated land and will therefore have a different species composition, with less dominance of *B. rhodani*.

Our top model for macroinvertebrate abundance showed how the effect of distance from estuary changed with time since last measure and with type of restoration measure (Figure



18). If we focus on distance from estuary, we found stations with side channels and riparian modifications to have a high abundance in the middle stretches of the river. This is in accordance with the areas of cultivated land surrounding Bognelv. For weir measures, the distance effect is difficult to interpret, due to few data points. However, there is a tendency of higher abundance three to four km upstream. It is important to note that these findings are based on few data points and are therefore highly uncertain. For channelized stations, predicted abundance was highest furthest downstream and decreased with increasing distance from estuary (Figure 19). In the case of channelized stations, we found that the river continuum concept can be applied, with highest abundance furthest downstream (Vannote et al., 1980). The further upstream, the more uncertain our predictions of abundance in channelized stations are, because most of the channelized stations are located in the middle and lower stretches of Bognelv.

#### 4.3.2 Effects of distance from estuary on brown trout density

Analyses of densities of 1+ brown trout predicted highest density close to estuary. The results were statistically significant (Table 14). Depth have been an important variable to explain Atlantic salmon and brown trout densities in earlier studies (Nordhov & Paulsen, 2016; Sødal, 2014), and the results from previous years have both similarities and differences with our study. Nordhov and Paulsen found that distance from estuary was an important variable for density and length of 0+ Atlantic salmon, and that the density and length increased with increasing distance from estuary (Nordhov & Paulsen, 2016). For brown trout, however, distance from estuary was not among the most important factors explaining density of 0+ in their study, in contrast to our findings. Sødal (2014) found that 0+ brown trout densities decreased with increasing distance from estuary at low depths (<25 cm), while density increased with increasing distance from estuary in deeper areas (>25 cm). In our results, density increased closest to estuary with higher PC3-values, which was positively associated with depth (Figure 27 and Figure 28). The PC3 component was difficult to interpret and only accounted for 16 % of the variation in our data (Table 2 and Figure 9). The effect of depth is therefore uncertain, and we believe land use and the abundance of macroinvertebrates are more important factors affecting the spatial dispersal of brown trout in Bognelv.

The middle and lower stretches of Bognelv are surrounded by cultivated land, which influence brown trout densities through agricultural runoff. Agricultural land use has been

shown to be both positive and negative for fish densities. In nutrient-poor rivers, such as Bognelv, nutrient enrichment from agricultural runoff can be beneficial for young salmonids. (Jonsson et al., 2011; Kail et al., 2015). Jonsson et al. (2011) studied juvenile salmonids in small Norwegian streams and found that density increased with increasing percentage of agricultural land use, up to a certain point. They found that 20 % cultivated land in the catchment area was a threshold for salmonid production. In Bognelv, only 1.4 % of the catchment area is cultivated land (Appendix 1). This suggests that agricultural runoff should not be a limiting factor for brown trout densities in Bognelv. Also, nutrient rich areas support a higher abundance of macroinvertebrates, which is important food for brown trout. They feed on drifting macroinvertebrates, which could explain the high densities of brown trout furthest downstream, as they would benefit from the high abundance of macroinvertebrates in the middle stretch of the river.

Another explanation might be that the downstream stretches of Bognelv are in contact with sea water, which also provides a plentiful food source for young brown trout.

When comparing brown trout densities from 2008 to 2021, the control zones (11 and 12) stand out with extremely low densities of 0+ (Figure 26). The stations were only added in 2015, but for the three years sampled, the densities of 0+ have been low compared with other age classes. One would expect there to be a higher density in areas that have been unaltered by channelization and restoration than in river stretches that have been “disturbed”, but this is not the case in Bognelv. Initially, we hypothesized that the habitat was less suitable for 0+ juvenile brown trout further upstream, with higher velocity and colder waters (Heggenes, 1988a). However, there might be a more specific reason for the low 0+ densities in the uppermost zones. This will be explained in the following chapter.

#### 4.3.3 Possible effects of an old mine upstream

One summer about 120 years ago, copper was extracted from a mine in Bognelvdalen, located between zone 11 and 12 (Rapp, 2019). Our results and results from previous years have all yielded low catches of brown trout in these two uppermost zones, especially of 0+ (Figure 26). This led us to believe that the mining operations might influence fish densities in the uppermost zones. It is highly unlikely that the water quality in Bognelv would still be influenced by runoff from the mine, as it would have been replenished countless times since

the mining occurred. The river also divides into two stretches below zone 11. This means that fresh water from both river stretches runs into the main river stretch below zone 11, which would further replenish the water quality and dilute the possible influence of the mine. However, another possible effect of the mining is that runoff and particulate matter could have been stored in the sediment (Lewin et al., 1977; Macklin et al., 2006). This could mean the sediment is unsuitable for spawning, and it could cause the roe to die in the sediment. The catches of 1+ and >1+ in these zones indicate that the older fish are able to utilize the habitat, possibly because they are more mobile and can swim up or down from other stretches. It is important to note that this will have to be further investigated before any conclusions can be drawn regarding possible effects of the mine on 0+ brown trout in Bognelv.

#### 4.4 Inter-cohort competition between years?

As earlier mentioned, studies conducted in Bognelv have not been able to find evidence of density dependence between 0+ and 1+ brown trout. However, when comparing the overall strength of age classes for different years, the pattern is quite clear. Except for a couple of good years (2008 and 2011), the 0+ brown trout age class has been consistently lower in density than the other two age classes since 2013 (Figure 25). This indicates that inter-cohort competition might be a limiting factor for 0+ brown trout, on river scale. Interestingly, the sampling of salmonids in Bognelv has been carried out only in odd years since 2011. Inter-cohort competition can occur in cycles, with high and low densities of 0+ every other year (e.g. Cantin & Post, 2018). This might mean that studies on Bognelv the past nine years have consistently missed out on good years for 0+ brown trout, as they might be occurring in even years. Conducting fieldwork in even years could reveal the years where 0+ densities are higher. This is something to consider for further research on brown trout in Bognelv.

#### 4.5 Change in macroinvertebrate diversity and abundance 2015 – 2021

Ordination analysis revealed that the macroinvertebrate community in stations with no measures were significantly different than stations which had undergone restoration measures or had been channelized (Figure 22). This is to be expected, as these stations are the uppermost stations in our sample, and they have not been affected by restoration measures or channelization further downstream. The location of these stations can be another explanation for the significant difference in macroinvertebrate species composition. According to the river continuum concept, headwater streams have a different species composition than further

downstream, with shredders and collectors as the dominating functional groups (Vannote et al., 1980).

When comparing species lists from all years macroinvertebrates have been sampled, a couple of things stand out. As earlier mentioned, abundance was at its highest in 2021 since macroinvertebrate sampling began. Another important finding is that Chironomids and the mayfly *B.rhodani* was much more abundant in 2021 than in 2015 and 2019. Chironomids and *B.rhodani* are generalist species, and it can therefore be a sign that the habitats in Bognelv are changing in favor of the generalists, rather than the specialists. Bjørngaard (2020) predicted that macroinvertebrate diversity would peak in Bognelv six years after restoration. However, our models did not predict a peak in macroinvertebrate diversity, even 14 years after restoration (Figure 15). This indicates that the carrying capacity for macroinvertebrate diversity has not yet been reached.

#### 4.6 Potential effects of pink salmon

During our field work we observed high numbers of pink salmon (*Oncorhynchus gorbuscha*). The ecological impacts of pink salmon on native species, such as brown trout and Atlantic salmon, are still uncertain due to limited data availability. Although the pink salmon spawn earlier than Atlantic salmon and brown trout, they prefer similar spawning sites and competition can occur (Sandlund et al., 2019). Additionally, the pink salmon only spawn in odd years, which coincides with low-density years of 0+ brown trout. This could be another factor limiting densities of 0+ brown trout in Bognelv.

After spawning, the pink salmon starts to rot and this process can have several effects on the river and on other species. Decomposition of the pink salmon tissue can decrease oxygen levels and negatively affect fish egg survival. On the other hand, studies have found that the decomposition of pink salmon can largely increase macroinvertebrate abundance (Chaloner et al., 2002). Many groups of invertebrates thrive in low-oxygen conditions, and they are an important part of juvenile salmonid diets (Bailey et al., 2018).

The pink salmon were numerous up until zone 4, above which we would only spot them sporadically. Large efforts were made by the local fishing association and volunteers to stop pink salmon from spawning in Bognelv. They took out approximately 350 pink salmon from

the river, but this is only a fraction of the pink salmon entering Bognelv (Mikalsen, 2021). Effects of the high numbers of pink salmon on densities of Atlantic salmon and brown trout in Bognelv should be further investigated. As should effects of the extreme nutrient enrichment to the system that comes with a large number of decomposing pink salmon.

#### 4.7 Potential sources of error

This study was conducted on a large river system with many influencing variables and it is therefore natural that it faces certain limitations. The first group of limitations is related to problems with the data collection, i.e. field work issues. Bognelv is a relatively large river, and some places the current was too strong to conduct electrofishing safely. Other places were shallow, but with low catchability because of high stream velocity. This led to six stations being excluded from electrofishing and a resulting reduction in total catches. A related limitation is the low catches of Atlantic salmon in our sample. This might be explained by the fact that the stations were located by the riverside and not in the middle of the river where the habitat is more suitable for Atlantic salmon.

Classification of macroinvertebrates presents another limitation in our study. The fact that many of the individuals sampled were too small to detect identifying features, even with a stereo microscope, presented us with a challenge. We got a lot of help from macroinvertebrate expert Trond Bremnes at the Natural History Museum at the University of Oslo. Still, the lion's share of the work was done by us at the lab at NMBU. Considering our limited background knowledge on macroinvertebrate identification, it is safe to assume that some individuals have been incorrectly identified. It is also safe to assume that as our knowledge and experience grew, the identification became more accurate. This means that there is likely an imbalance in our classification accuracy between the beginning and the end of the laboratory period.

The second group of limitations has to do with the analysis. We made a choice to exclude 0+ brown trout from our analysis, because of zero-inflation in our data. We could therefore not compare our results on 0+ to results from previous years. Furthermore, our analysis on effects of restoration measures assumed that the different measures (weirs, side channels, riparian modifications and channelization) were independent of each other. It is highly likely that the different measures affect each other in various ways, and that they are interlinked in their

effect on macroinvertebrates and brown trout. In that way, our analysis of the restoration measures paints a simplified picture of the situation, when the picture might be much more complex.

#### 4.8 Recommendations for further research

To counteract some of the limitations mentioned above, we will now make some suggestions for further research.

We collected our data during the end of August and beginning of September. The small sizes of both fish and macroinvertebrates proved a challenge. Conducting field work later in the season might give macroinvertebrates and fish more time to grow, thus yielding larger individuals. However, going too late might present problems with snow and ice in the river, so a trade-off will have to be made. Field work should also be conducted in an even year, to detect year-to-year variations in the age classes of brown trout, as mentioned in chapter 4.4. As the catches of Atlantic salmon have been relatively low in recent years compared to brown trout, other sampling methods should be considered in the future. A method that takes into account Atlantic salmon preference for placement in the river should be considered. Arctic charr have been missing from studies since 2013. It might be that arctic charr simply have not responded to the restoration measures yet, but it might also be that the stations are not placed in suitable habitat for arctic charr (Nordhov & Paulsen, 2016). Choosing stations in areas more suitable for arctic charr should therefore be something to consider for future research. Moreover, finding a reference river to compare effects of restoration with effects of no restoration should be a focus in the future. The uppermost stations in Bognelv (zones 11 and 12) can be used as reference sites, but looking at an entire river will provide a more complete picture of the differences between restored and not restored, as it includes the gradient from lower to upper.

Lastly, a different study design that analyzes effects on a zone scale instead of on a station scale might be worth considering. Effects of restoration measures are rarely restricted to stations and looking at effects on a larger scale might yield different results.

## 5 Conclusion

Our first aim was to determine the most successful restoration measure in Bognelv, in terms of improving conditions for brown trout and macroinvertebrates. Our results showed that weirs were most successful for both macroinvertebrate diversity and brown trout densities, in contrast to previous years findings. For macroinvertebrate abundance, side channels were the most successful restoration measure, but channelized stations (not restored stations) also yielded high macroinvertebrate abundance. The side channels in Bognelv were often slow flowing with pools and large woody debris, which also provides new habitats for macroinvertebrates. Although we have looked at these measures separately, we would like to emphasize the fact that these measures are not independent of each other and combined probably have a large impact on the river ecosystem that is not investigated in this thesis.

Our second aim was to investigate the effect of time on the macroinvertebrate and brown trout communities in Bognelv. Macroinvertebrate composition in Bognelv in 2021 compared to previous years shows that diversity is decreasing, while abundance is increasing. This suggests that the macroinvertebrate community in Bognelv is becoming more generalist. A peak in macroinvertebrate abundance eight to ten years post-restoration suggests that it takes almost a decade for macroinvertebrates to re-colonize and stabilize the habitat. For brown trout, on the other hand, predicted densities decreased the first ten years post-restoration, with a subsequent flattening of the curve. Our results furthermore suggest that it takes more than 14 years for brown trout densities to stabilize post-restoration, and that densities of brown trout in Bognelv are approaching a low point.

Testing the effect of distance from estuary was our third aim. For macroinvertebrate diversity, distance from estuary had little effect. Abundance showed a higher effect of distance, and the highest abundances were found in the middle parts of the river. The highest densities of brown trout were found close to the estuary, contradictory to our expectations. Influence of agricultural land in the middle zones might increase abundance of macroinvertebrates and thus increase brown trout densities downstream these zones. The sediment in the upmost zones of Bognelv might be negatively affected by a runoff from an old copper mine, explaining the low densities in these zones.

The final aim of our study was to evaluate how the whole system has responded to the restoration. Looking back over the last nine years, we discovered that studies have only been carried out in odd years, and that we likely have missed out on “good” 0+ years, due to inter-cohort competition. Although our catches were low in 2021 compared to previous years, we believe this is because of dynamic changes occurring in nature, and not due to a “failure” of restoration. Very low catches of Atlantic salmon and Arctic charr suggest that the methods are not fit for catching these species, or that the habitat in Bognelv is currently more suitable for brown trout. This might change as the restoration progresses, as it takes time for lotic communities to get back to a “natural” state. We conclude that the system has not reached its carrying capacity for either macroinvertebrates or brown trout.



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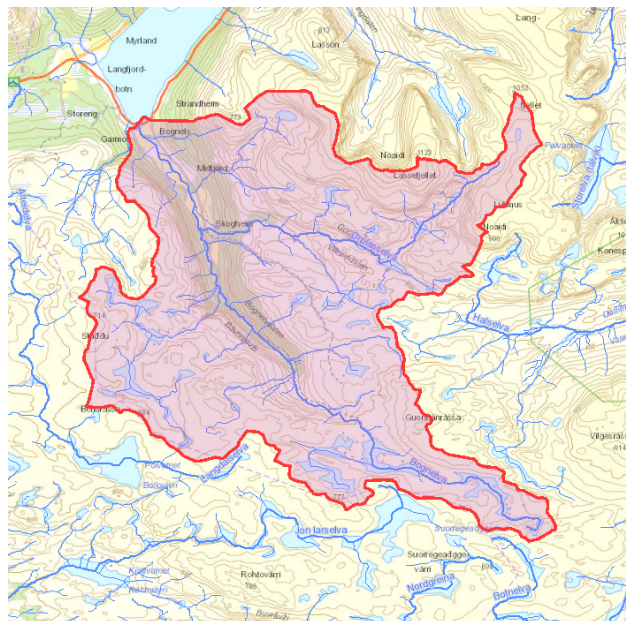
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# Appendix 1

Weather data generated from [nevina.nve.no](http://nevina.nve.no)



Norges  
vassdrags- og  
energidirektorat

NVE

Kartbakgrunn: Statens Kartverk  
Kartdatum: EUREF89 WGS84  
Projeksjon: UTM 33N  
Beregn.punkt: 777617 E  
7785045 N

Nedbørfeltgrenser og feltparametere er automatisk generert og kan inneholde feil. Resultatene må kvalitetssikres.

## Nedbørfeltparametere

Vassdragsnr.: 211.8A0  
Kommune.: Alta  
Fylke.: Troms og Finnmark  
Vassdrag.: Bognelva

### Feltparametere

Areal (A)	88.7	km <sup>2</sup>
Effektiv sjø (A <sub>SE</sub> )	0.28	%
Elvleengde (E <sub>L</sub> )	19.5	km
Elvegradient (E <sub>G</sub> )	34.5	m/km
Elvegradient <sub>1085</sub> (E <sub>G,1085</sub> )	44.1	m/km
Helning	12.8	°
Dreneringstetthet (D <sub>T</sub> )	1.4	km <sup>-1</sup>
Feltleengde (F <sub>L</sub> )	16	km

### Arealklasse

Bre (A <sub>BRE</sub> )	0	%
Dyrket mark (A <sub>JORD</sub> )	1.4	%
Myr (A <sub>MYR</sub> )	0.0	%
Leire (A <sub>LEIRE</sub> )	0	%
Skog (A <sub>SKOG</sub> )	11.6	%
Sjø (A <sub>SJO</sub> )	5.6	%
Snau fjell (A <sub>SF</sub> )	80.1	%
Urban (A <sub>U</sub> )	0	%
Uklassifisert areal (A <sub>REST</sub> )	1.3	%

### Hypsografisk kurve

Høyde <sub>MIN</sub>	1	m
Høyde <sub>10</sub>	261	m
Høyde <sub>20</sub>	448	m
Høyde <sub>30</sub>	523	m
Høyde <sub>40</sub>	564	m
Høyde <sub>50</sub>	601	m
Høyde <sub>60</sub>	626	m
Høyde <sub>70</sub>	660	m
Høyde <sub>80</sub>	695	m
Høyde <sub>90</sub>	796	m
Høyde <sub>MAX</sub>	1115	m

### Klima- /hydrologiske parametere

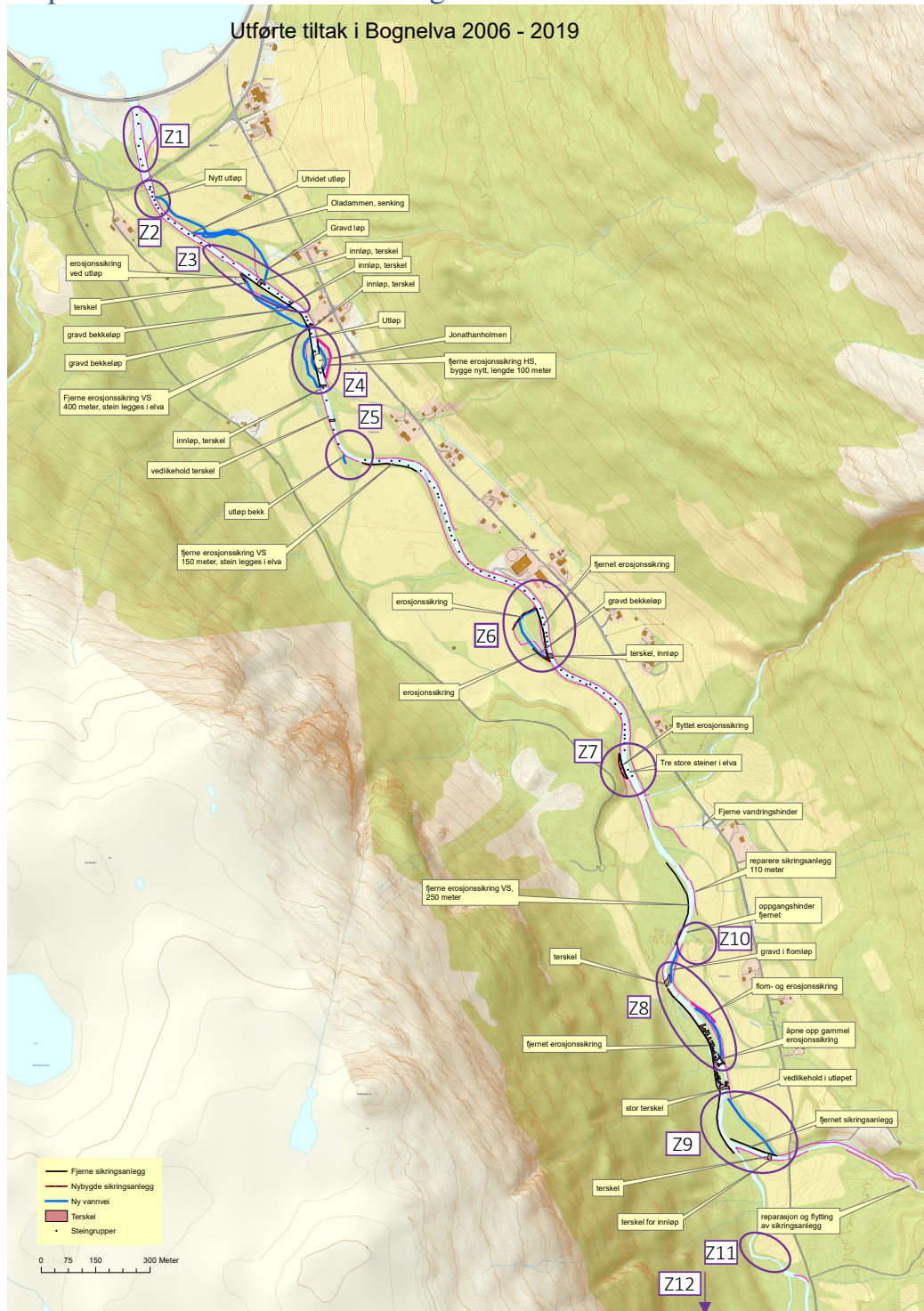
Avrenning 1961-90 (Q <sub>N</sub> )	32.4	l/s*km <sup>2</sup>
Sommernedbør	282	mm
Vinternedbør	499	mm
Årstemperatur	-0.5	°C
Sommertemperatur	5.5	°C
Vintertemperatur	-4.9	°C

Rapportdato: 19.11.2021

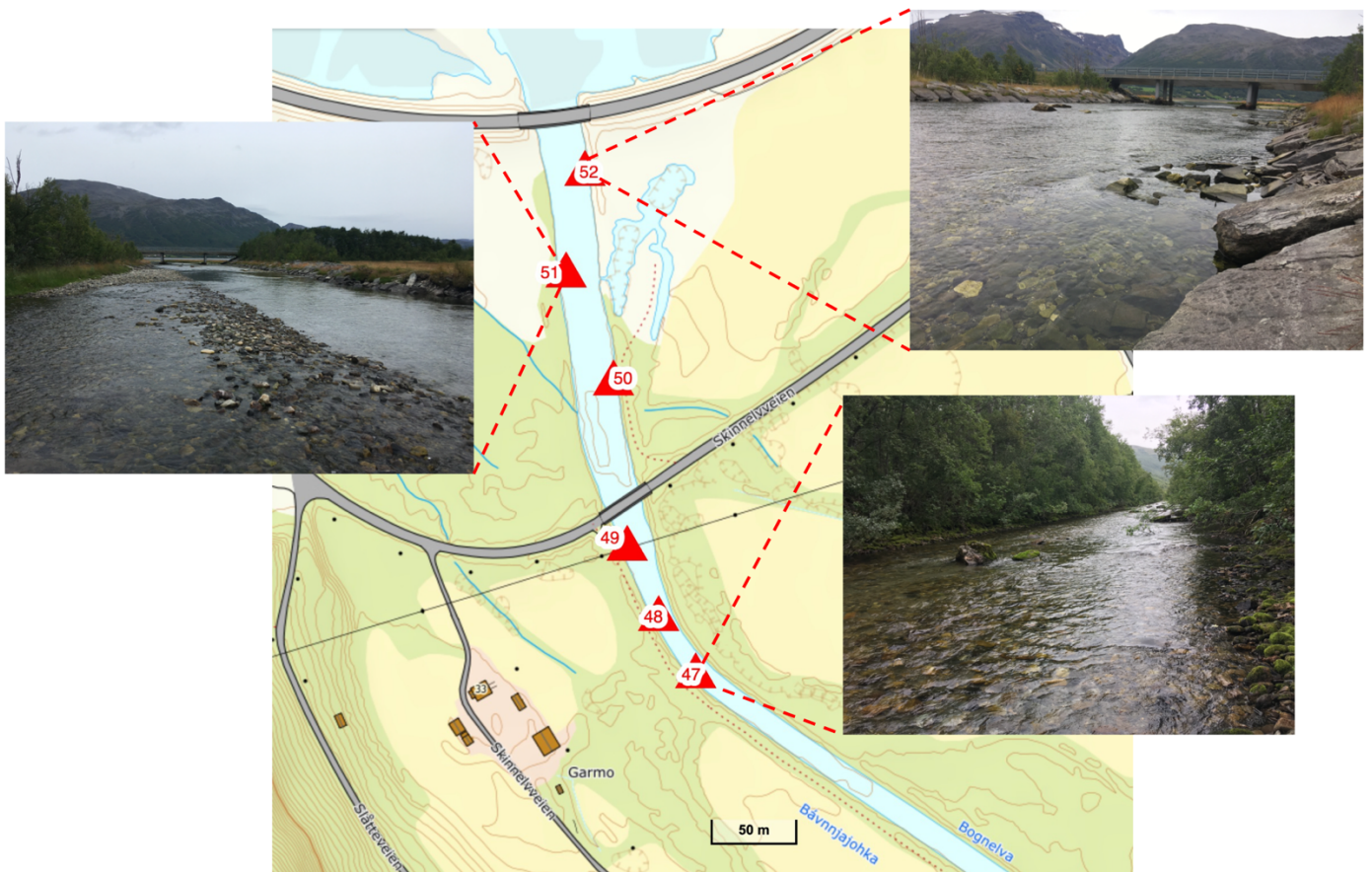
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## Appendix 2

### Map of conducted measures in Bognelva

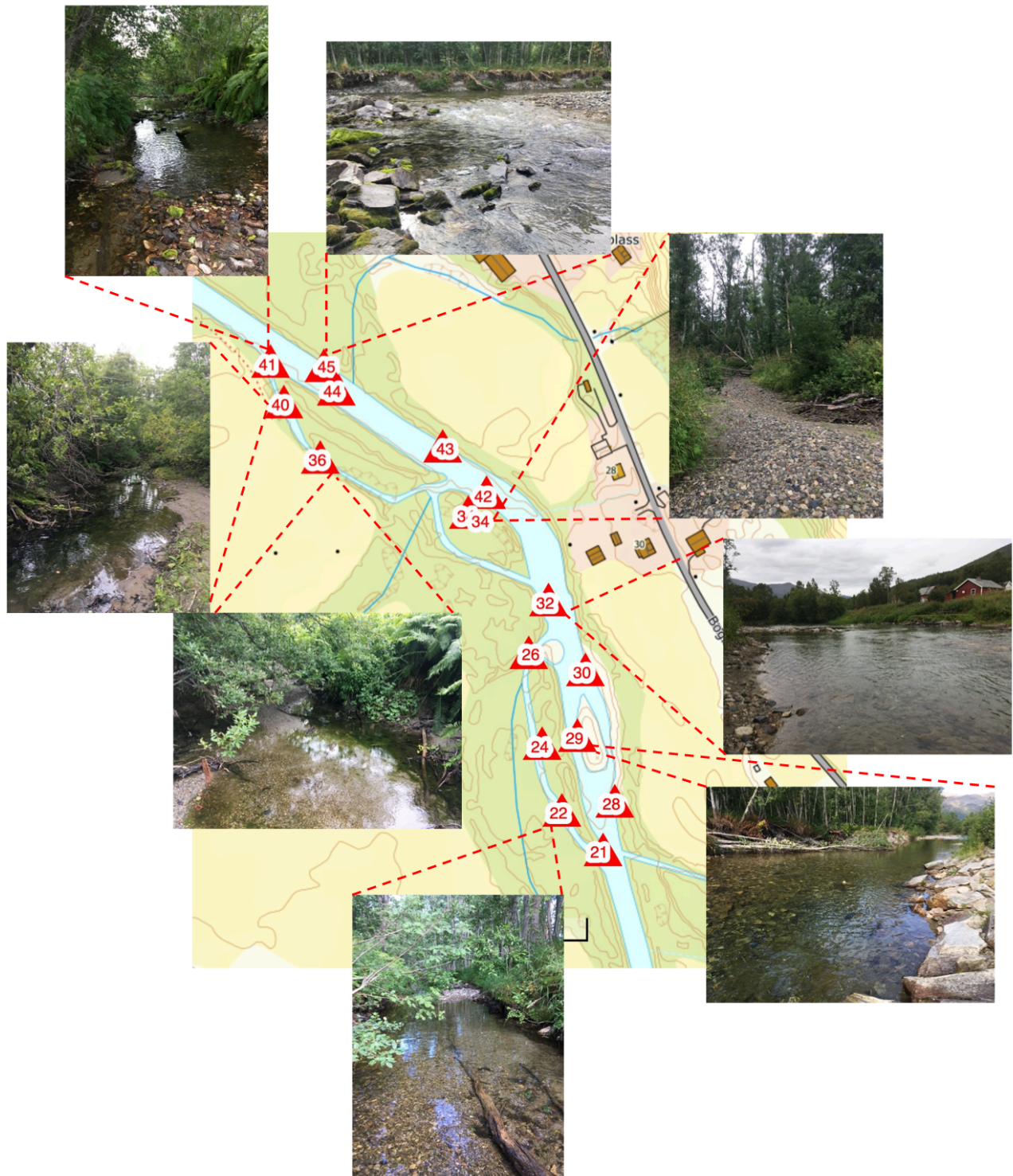


Aerial map of stations with photos from our field work.

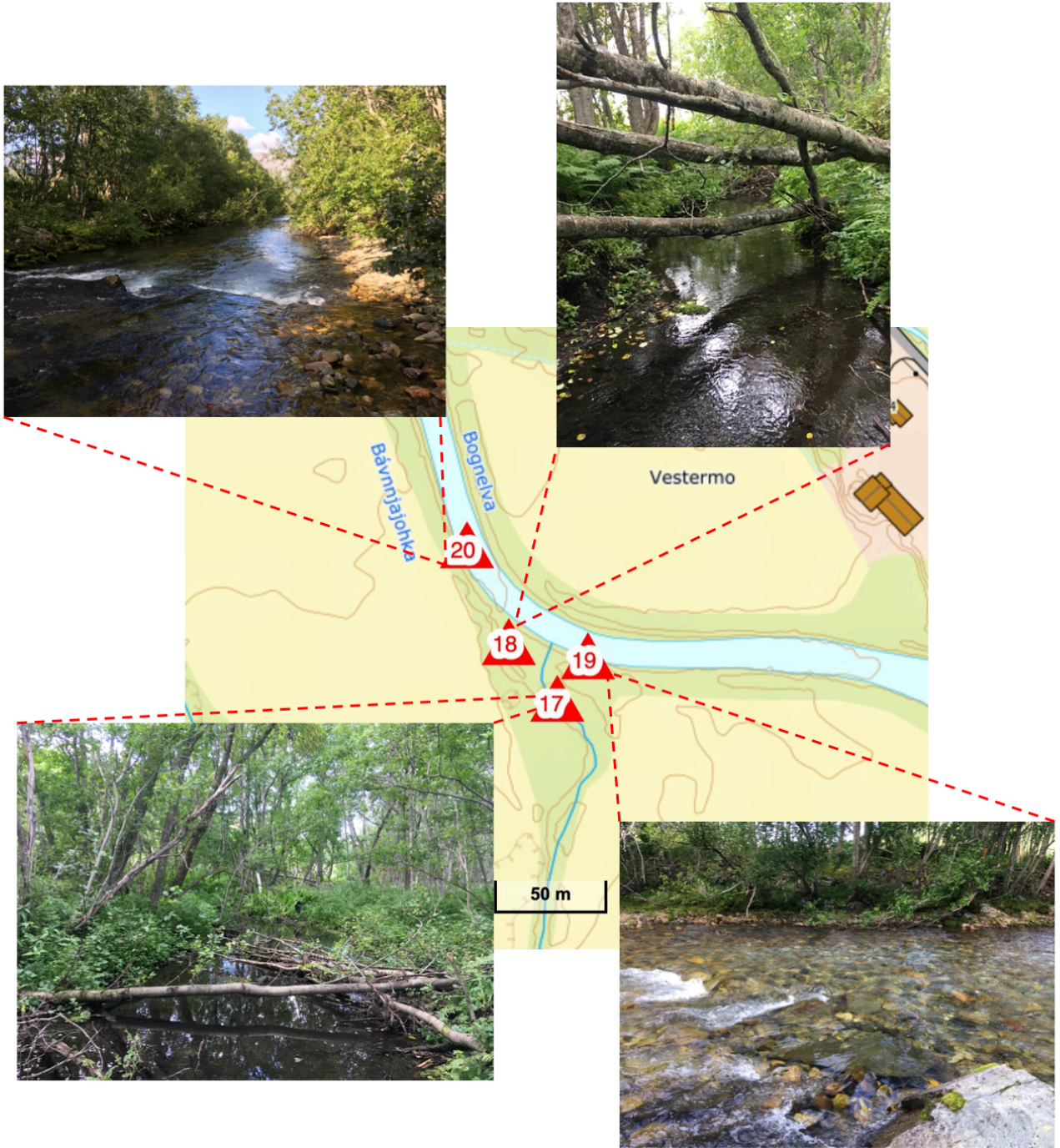


**Zone 1: station 52, 51 and 50**

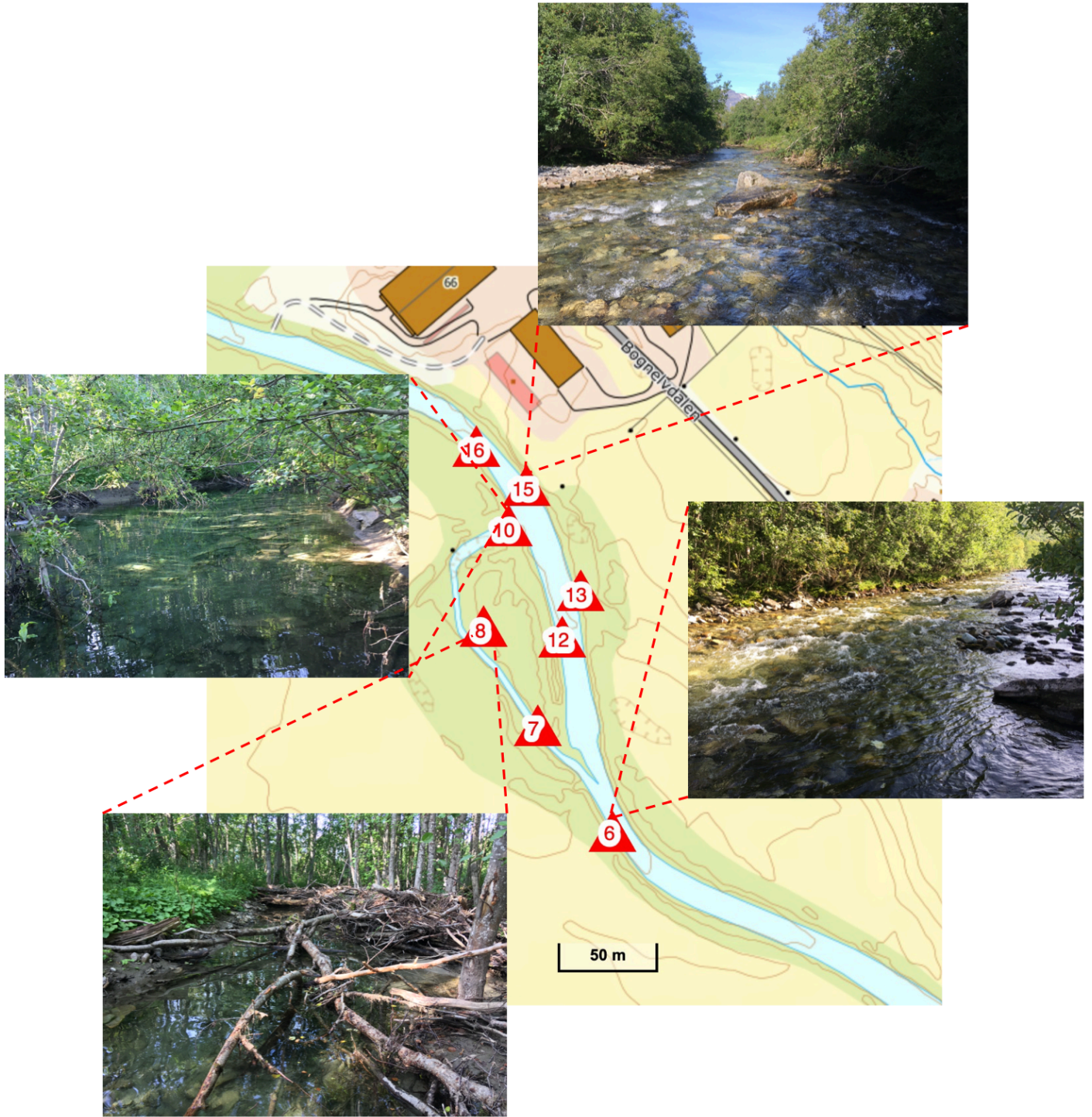
**Zone 2: station 49, 48, 47**



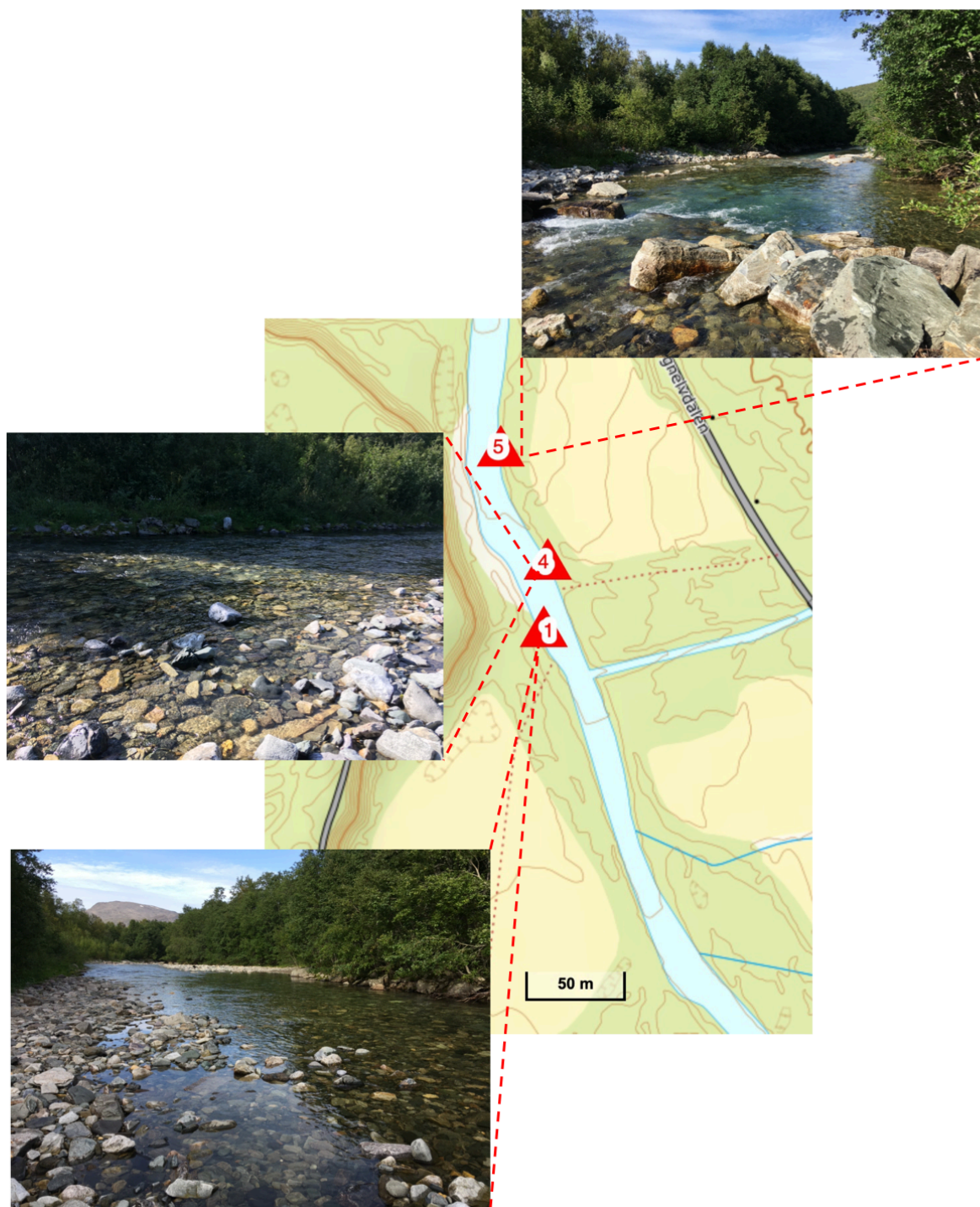
**Zone 3: station 41, 45, 44, 43, 42, 40, 36, 34**  
**Zone 4: station 32, 30, 29, 28, 26, 24, 22, 21**



**Zone 5**



**Zone 6**

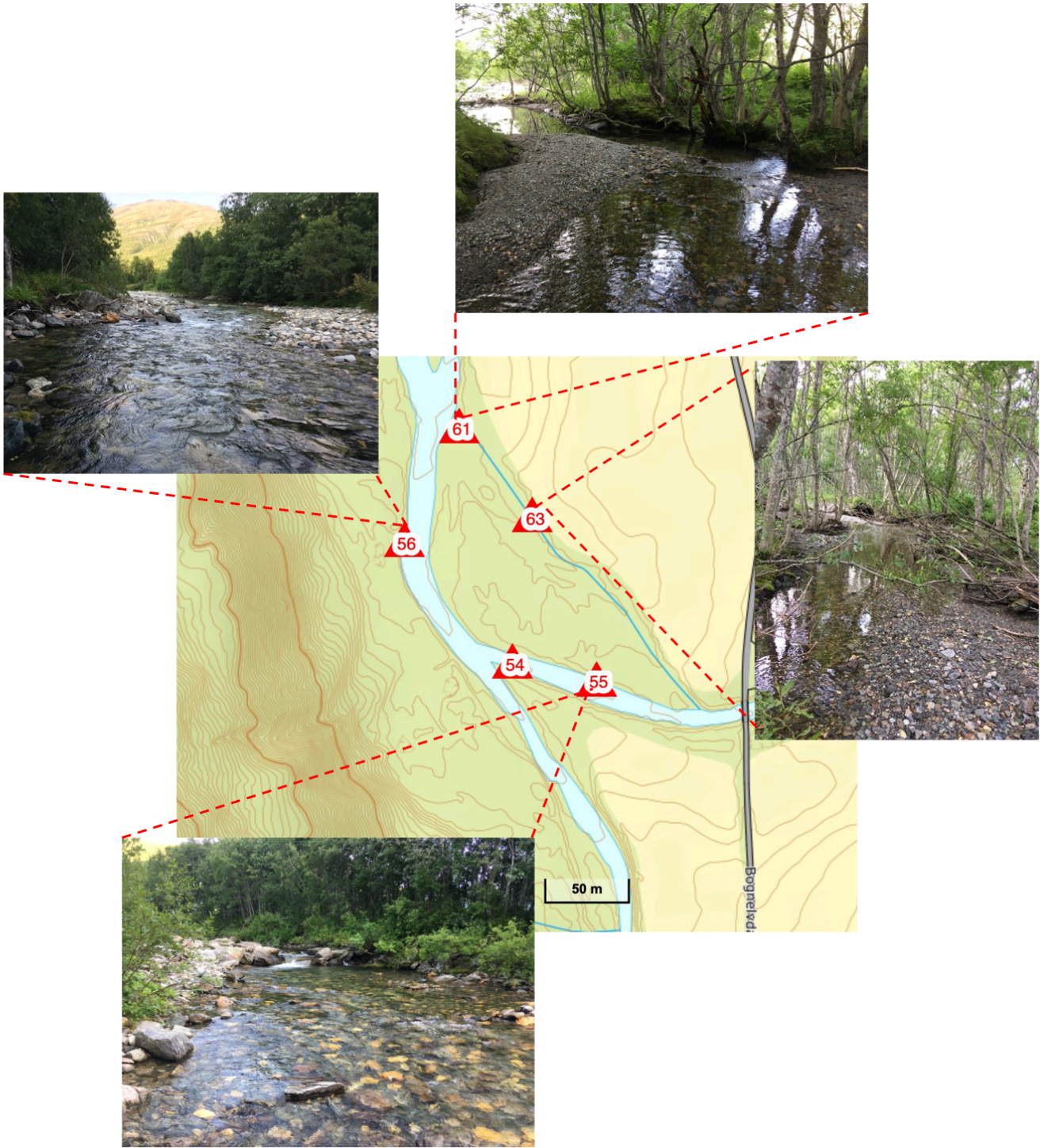


**Zone 7**

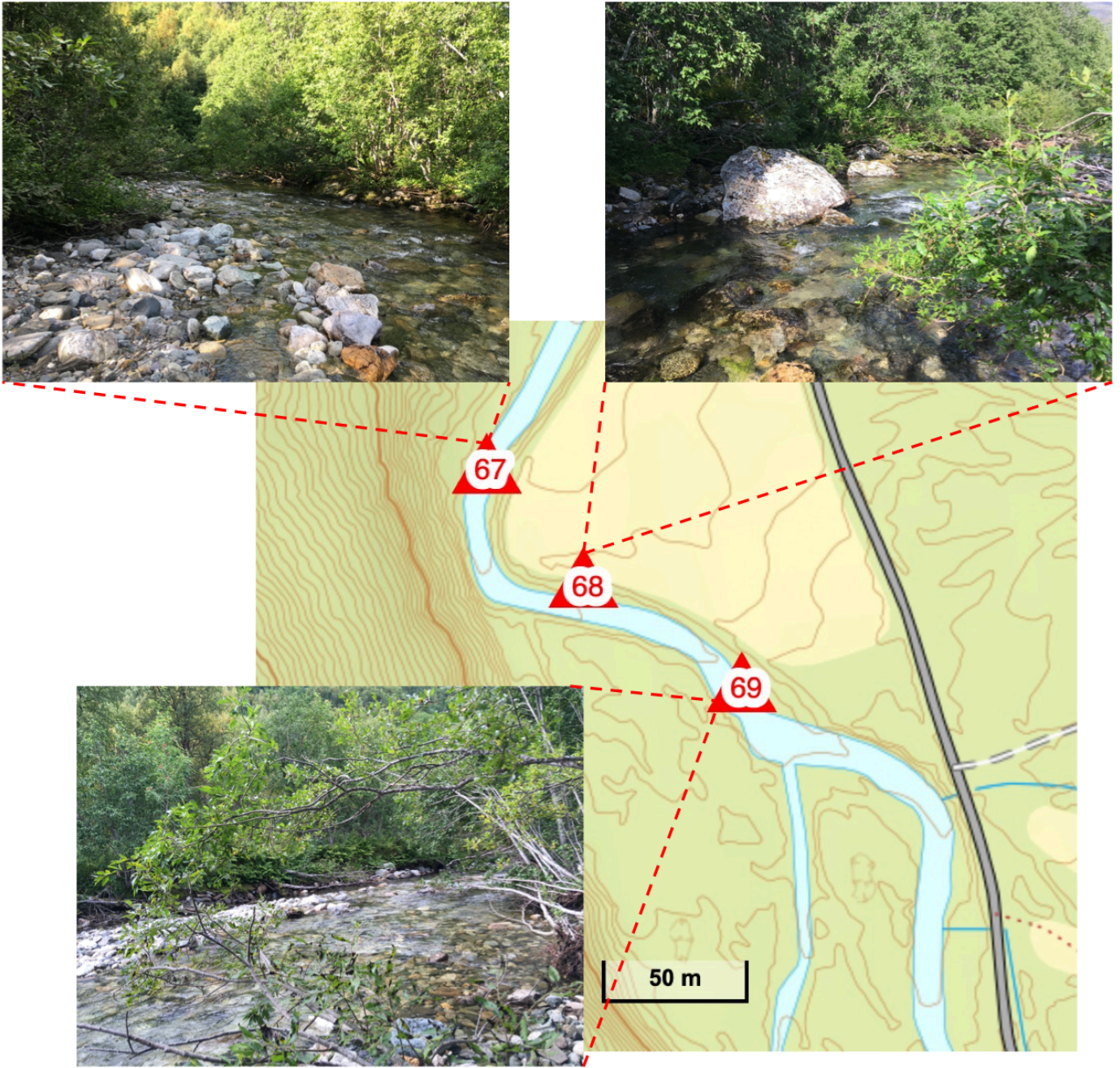


**Zone 10: station 65, 64**  
**Zone 8: station 60, 59, 59b, 58, 57, 57b**

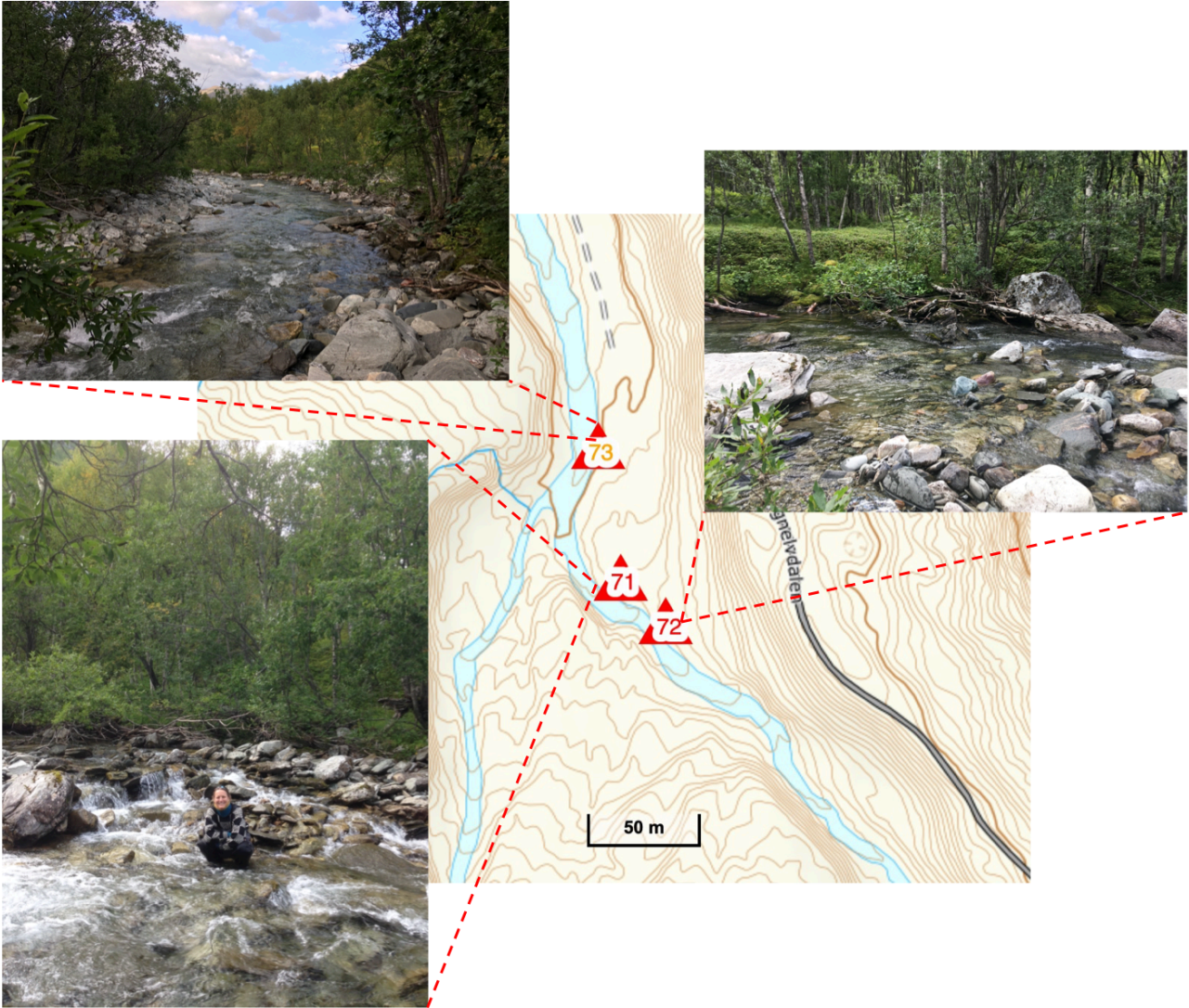




**Zone 9**



**Zone 11**



**Zone 12**

## Appendix 3

### Table of conducted measures in Bognelv, 2006 - 2019

Adapted from Nordhov & Paulsen (2016) and Johansen (2020), supplemented with information from Hoseth & Josefsen (2005), Bjordal & Hoseth (2009), Hoseth & Josefsen (2012), Bjordal & Hoseth (2014), Johnsen & Bjordal (2018) and Bjordal & Sæle (2019).

Measures are categorized by type of restoration measure. *CHA*=channelized, *WEI*=weirs, *SID*=side channel, *RIP*=riparian modifications, *NOM*=no measure. *CHA* with numbers represent the stations that have been channelized within the zone.

Zone	2006	2007	2009	2012	2014	2018	2019
1	<p>CHA Whole zone channelized in the 1930s and never restored</p>						
2							<p>SID</p> <ul style="list-style-type: none"> <li>Expand both outflows from Oladammen</li> </ul>
3	<p>CHA</p> <ul style="list-style-type: none"> <li>Opening of side channel, two inflows and one outflow</li> </ul> <p>WEI</p> <ul style="list-style-type: none"> <li>Placement of rock clusters downstream the inflows to increase the water levels</li> <li>Placement of weir in outflow of side channel to increase the water level</li> </ul>	<ul style="list-style-type: none"> <li>Supplementary work to improve water flow</li> </ul>	<p>SID</p> <ul style="list-style-type: none"> <li>Reinforce and increase weirs by the inflows of the side channel.</li> </ul>	<p>RIP</p> <ul style="list-style-type: none"> <li>Removal of erosion control systems in the main river</li> <li>Placement of rock clusters in the main river</li> </ul>	<p>SID</p> <ul style="list-style-type: none"> <li>Re-opening of in- and outflow to Oladammen</li> <li>Establish weir by the inflow and rock clusters to increase the water level.</li> <li>Dig deeper inflow ditch to ensure constant water flow.</li> </ul> <p>WEI</p> <ul style="list-style-type: none"> <li>Placement of rocks in the river to vary the water flow.</li> </ul> <p>RIP</p>	<p>WEI</p> <ul style="list-style-type: none"> <li>Adjusted weir by inflow to Oladammen</li> <li>Extended inflow to Oladammen by 1 meter</li> <li>Lowered Oladammen with 1 meter to maintain surface water throughout the summer</li> </ul>	<ul style="list-style-type: none"> <li>Adjustment of earlier conducted measures in Oladammen</li> </ul>

					<ul style="list-style-type: none"> <li>• New erosion control system to protect farmed area</li> </ul>		
4 CHA: 21	<p>SID</p> <ul style="list-style-type: none"> <li>• Opening of side channel, two inflows and one outflow.</li> </ul> <p>WEI</p> <ul style="list-style-type: none"> <li>• Placement of rock clusters downstream the inflows to increase the water levels.</li> <li>• Placement of weir in outflow of side channel to increase the water level.</li> </ul>	<ul style="list-style-type: none"> <li>• Supplementary work to improve water flow</li> </ul>	<p>SID</p> <ul style="list-style-type: none"> <li>• Reinforce and increase weir by the inflow of the side channel</li> </ul>	<p>RIP</p> <ul style="list-style-type: none"> <li>• Removal of erosion control systems in the main river</li> <li>• New erosion control system to protect farmed area</li> </ul> <p>WEI</p> <ul style="list-style-type: none"> <li>• Placement of rock clusters in the main river</li> </ul>	<ul style="list-style-type: none"> <li>• Removal of some rocks to increase water velocity in pool upstream Oladammen</li> <li>• Building of an island.</li> </ul> <p>WEI</p> <ul style="list-style-type: none"> <li>• Groin dike up- and downstream new island.</li> </ul>	<p>SID</p> <ul style="list-style-type: none"> <li>• Between zone 3 and 4: Adjustment measures to side channel, placement of groin dike to increase water flow into side channel</li> </ul>	

5 CHA: 20, 19		<p>SID</p> <ul style="list-style-type: none"> <li>Opening of the tributary Mikkeltveita</li> </ul> <p>WEI</p> <ul style="list-style-type: none"> <li>Two weirs were improved and repaired</li> </ul>			<p>WEI</p> <ul style="list-style-type: none"> <li>Upstream zone 5, placement of rocks in the river to vary the water flow.</li> </ul>		
6 CHA: 15, 13, 16	<p>RIP</p> <ul style="list-style-type: none"> <li>Upgrade and removal of flood protection, and establishment of new flood protection.</li> </ul> <p>SID</p> <ul style="list-style-type: none"> <li>Opening of side channel.</li> <li>Placement of rock clusters downstream the inflow and by the outflows of side channel to increase the water levels.</li> </ul>	<p>RIP</p> <ul style="list-style-type: none"> <li>Upgrade flood protection</li> </ul> <p>SID</p> <ul style="list-style-type: none"> <li>New side channel</li> </ul>			<p>SID</p> <ul style="list-style-type: none"> <li>Removal of deposited sand from inlet to tributary.</li> </ul>		
7 CHA: 4		<p>RIP</p> <ul style="list-style-type: none"> <li>Relocation and improvement of flood protection</li> </ul> <p>WEI</p> <ul style="list-style-type: none"> <li>Split a big rock into several pieces.</li> </ul>	<p>RIP</p> <ul style="list-style-type: none"> <li>Relocation of flood protection systems</li> </ul>		<p>WEI</p> <ul style="list-style-type: none"> <li>Groin dike from both sides to concentrate water flow.</li> <li>Removal of deposits to re-open pools.</li> </ul>	<p>SID</p> <ul style="list-style-type: none"> <li>Upstream zone 7, opening of side channel</li> </ul>	
8			<p>WEI</p> <ul style="list-style-type: none"> <li>Four new weirs were made.</li> </ul>		<p>SID</p> <ul style="list-style-type: none"> <li>Placement of rocks in the river</li> </ul>		

			<p>SID</p> <ul style="list-style-type: none"> <li>• Opening of an old river course.</li> </ul> <p>RIP</p> <ul style="list-style-type: none"> <li>• Removal of erosion control systems.</li> <li>• New erosion control systems to protect farmed area.</li> </ul>		to better water flow into tributary.		
9 CHA: 56			<p>WEI</p> <ul style="list-style-type: none"> <li>• Maintenance of a weir.</li> <li>• Construction of a weir to get water into the original river course.</li> </ul> <p>RIP</p> <ul style="list-style-type: none"> <li>• Removal of erosion control systems.</li> </ul> <p>SID</p> <ul style="list-style-type: none"> <li>• Opening of the original river course for Ørplasselva.</li> </ul>	<p>WEI</p> <ul style="list-style-type: none"> <li>• Repairing of a weir in Ørplasselva</li> <li>• Removal of gravel</li> </ul>			
10			<p>SID</p> <ul style="list-style-type: none"> <li>• Removal of a migration barrier</li> </ul>				
11 & 12	<p>NOM (no measure) Control zones, uninfluenced by channelization or restoration</p>						

Rock clusters	<ul style="list-style-type: none"> <li>• Zone 6. Rock clusters to increase diversity in water flow.</li> </ul>	<ul style="list-style-type: none"> <li>• Zone 1 – 7, from the new E6 up to Korselva. 2-3 rocks are added to each of the 78 originally single rocks, to create rock clusters. In addition 60 new rock clusters were made.</li> </ul>	<ul style="list-style-type: none"> <li>• Zone 8 and 9. Rock clusters to increase diversity in water flow.</li> </ul>	<ul style="list-style-type: none"> <li>• Zone 3 and 4. Placement of bigger rock clusters in the main river.</li> </ul>			
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## Appendix 4

### Station coordinates

<b>Zone</b>	<b>Station</b>	<b>UTM Zone 33 E</b>	<b>UTM Zone 33 N</b>
1	52	777634.510	7785008.158
1	51	777623.286	7784948.713
1	50	777650.601	7784886.036
2	49	777659.053	7784790.316
2	48	777677.033	7784750.855
2	47	777698.407	7784717.764
3	45	777966.210	7784542.063
3	44	777973.663	7784527.712
3	43	778039.552	7784491.980
3	42	778066.509	7784463.464
3	41	777932.749	7784544.791
3	40	777941.285	7784519.484
3	36	777963.753	7784485.496
3	34	778055.639	7784451.326
4	32	778105.273	7784396.889
4	30	778127.631	7784353.837
4	29	778122.603	7784313.109
4	28	778145.760	7784272.148
4	26	778092.387	7784364.436
4	24	778100.985	7784307.955
4	22	778113.087	7784266.908
4	21	778138.654	7784242.278
5	20	778183.988	7784088.848
5	19	778237.114	7784039.785
5	18	778202.263	7784046.399
5	17	778223.846	7784021.375
6	16	778710.152	7783670.126
6	15	778735.227	7783650.476
6	13	778762.553	7783597.858
6	12	778753.762	7783574.861
6	10	778726.140	7783630.468
6	8	778714.029	7783579.988
6	7	778741.061	7783530.357
6	6	778778.446	7783477.727
7	5	778983.315	7783226.404
7	4	779005.956	7783170.305
7	1	779004.322	7783135.947
8	60	779105.718	7782639.044
8	59	779170.725	7782530.806
8	59b	779219.276	7782487.324
8	58	779243.393	7782416.284
8	57	779172.177	7782495.746
8	57b	779187.291	7782475.112
9	63	779303.631	7782234.152

<b>9</b>	61	779261.265	7782286.291
<b>9</b>	56	779228.618	7782219.700
<b>9</b>	55	779341.264	7782138.296
<b>9</b>	54	779291.976	7782148.513
<b>10</b>	65	779206.307	7782720.394
<b>10</b>	64	779169.339	7782789.158
<b>11</b>	67	779325.432	7781891.317
<b>11</b>	68	779358.452	7781852.336
<b>11</b>	69	779412.290	7781816.417
<b>12</b>	71	779868.727	7780393.045
<b>12</b>	72	779888.723	7780373.898
<b>12</b>	73	779858.938	7780451.420

## Appendix 5

### Method for measuring environmental variables

#### **Cover of branches (canopy):**

**Canopy cover:** Percentage cover of branches measured 2 meters from the edge of the riverbank out over the river (only wet area).

**Riverbank:** Percentage cover of branches hanging over the riverbank. Category 1: 0% cover, category 2: 1- 25% cover, category 3: 26- 50% cover, category 4: 51- 75% cover, category 5: 76- 90% cover, category 6:  $\geq 91\%$  cover.

**Riverside vegetation:** Percentage cover of branches **on the top** of the riverbank. Category 1: 0% cover, category 2: 1- 25% cover, category 3: 26- 50% cover, category 4: 51- 75% cover, category 5: 76- 90% cover, category 6:  $\geq 91\%$  cover.

**Substrate composition:** The different types of substrates (sand, silt, gravel, rocks) were divided into five categories based on their size. Category 1: 0-2mm, category 2: 2-20 mm, category 3: 20- 100 mm, category 4: 100-250 mm, category 5: >250 mm. Each category was visually estimated to a percentage of the whole.

**Algae:** Mean percentage cover of algae. Biofilm and small periphytic algae covering the substrate were classified as algae. Category 1: 0%, category 2: 1-33%, category 3: 34-66%, category 4: >66%.

**Moss:** Mean percentage cover of moss. Threadlike algae were classified as moss. Category 1: 0%, category 2: 1-33%, category 3: 34-66%, category 4: >66%.

**Water velocity:** Water velocity was measured by the time it took a leaf to float one meter mid-stream (meter per second).

**Depth:** River depth was measured at five points along each cross-section transect (10%, 25%, 50%, 75% and 90%).

**Number of pools:** Number of areas with still water larger than 2 m<sup>2</sup>

**Number of large woody debris:** Branches with diameter more than 10 cm and length more than 1 meter and large concentrations of small woody debris.

## Appendix 6

Principal component scores and species scores from principal component analysis (PCA)

Table 6.1. Principal component scores.

	PC1	PC2	PC3	PC4	PC5	PC6	PC7	PC8	PC9	PC10
<b>Eigenvalue</b>	2.677	1.877	1.57	1.132	0.750	0.613	0.493	0.395	0.312	0.180
<b>Proportion Explained</b>	0.268	0.188	0.157	0.113	0.075	0.061	0.049	0.040	0.031	0.018
<b>Cumulative Proportion</b>	0.268	0.455	0.613	0.726	0.801	0.862	0.911	0.951	0.982	1.000

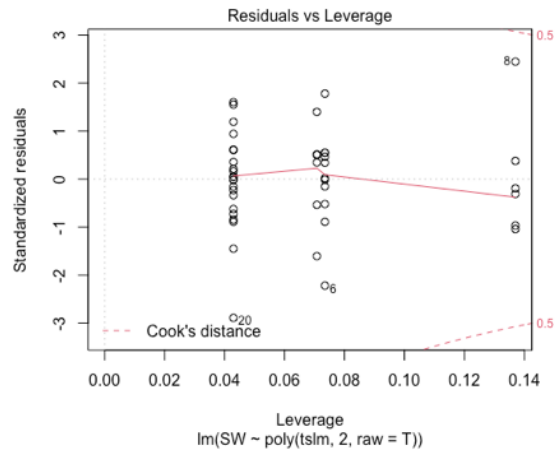
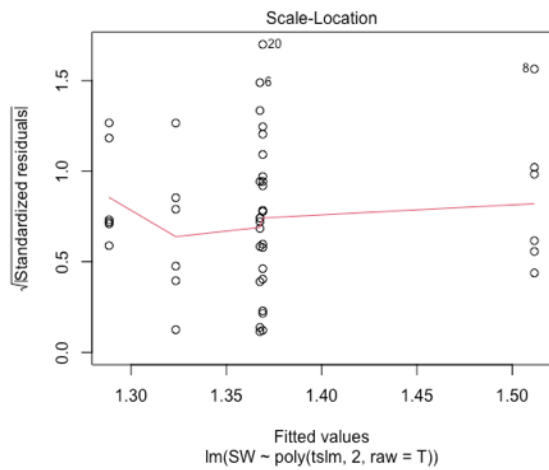
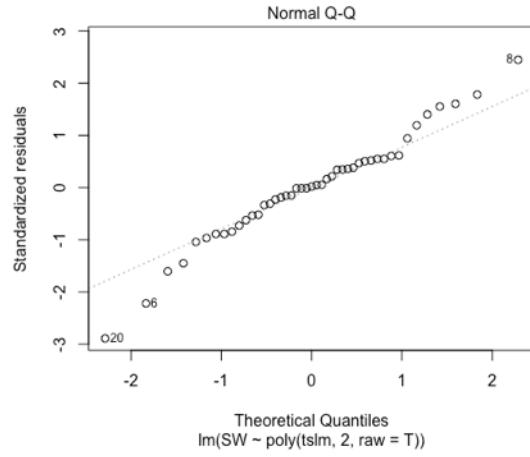
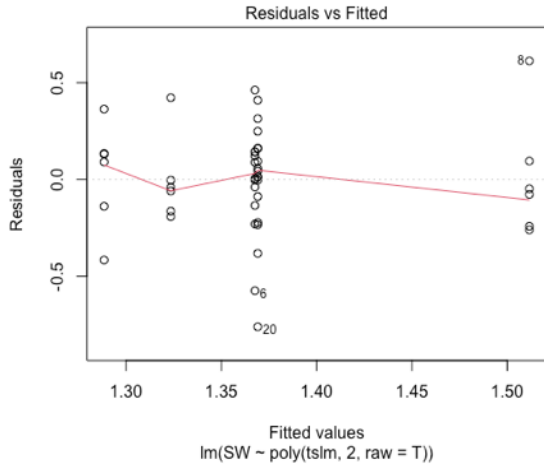
Table 6.2. Species scores from principal component analysis.

	PC1	PC2	PC3	PC4	PC5	PC6
<b>mean.depth.station</b>	0.443	-0.914	0.327	-0.684	0.429	-0.415
<b>mean.substrate.station</b>	-0.222	-1.173	0.178	-0.362	0.106	0.527
<b>mean.velocity</b>	-0.429	-0.814	-0.197	0.900	-0.460	0.263
<b>mean.algae</b>	0.553	-0.212	0.964	0.242	-0.660	-0.550
<b>mean.moss</b>	1.074	-0.627	0.436	0.173	-0.085	0.166
<b>mean.canopy</b>	1.265	-0.058	-0.190	0.279	0.234	0.225
<b>mean.riverside</b>	0.536	0.008	-0.947	-0.644	-0.695	-0.162
<b>mean.riverbank</b>	1.197	-0.039	-0.692	-0.018	-0.086	0.191
<b>pools</b>	0.070	0.634	0.846	-0.648	-0.434	0.644
<b>lwd</b>	0.913	0.721	0.357	0.381	0.332	0.075

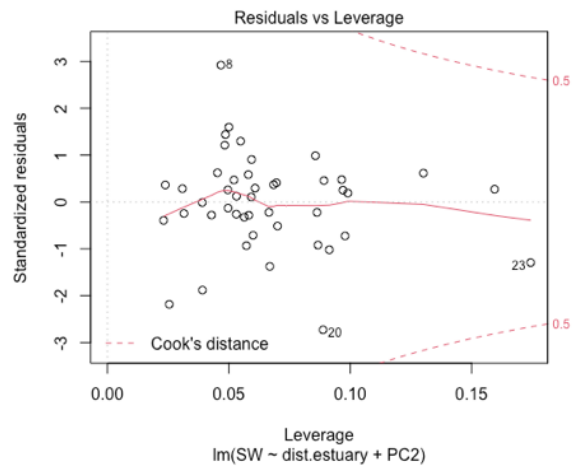
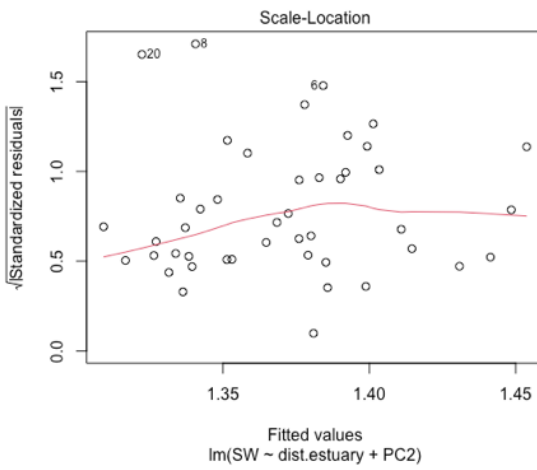
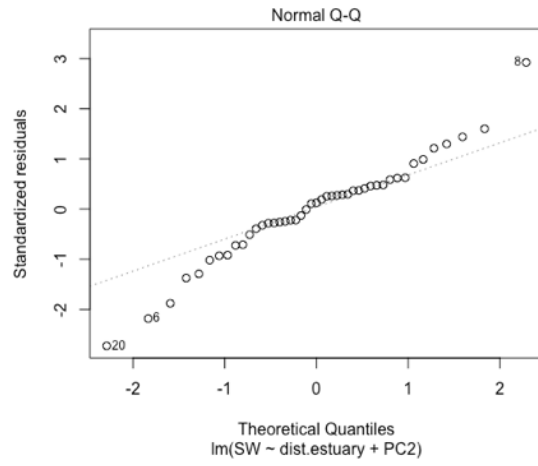
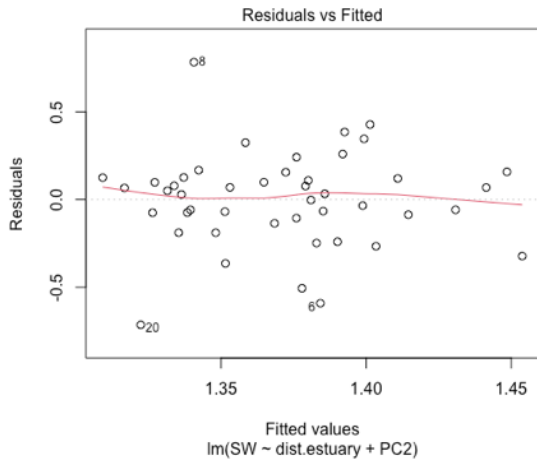
# Appendix 7

## Residual plots for model verification

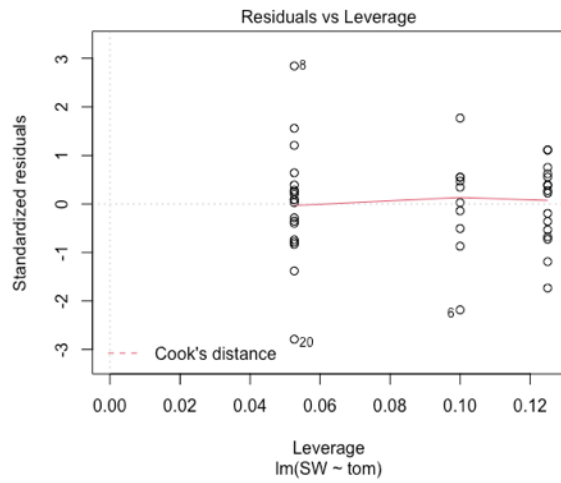
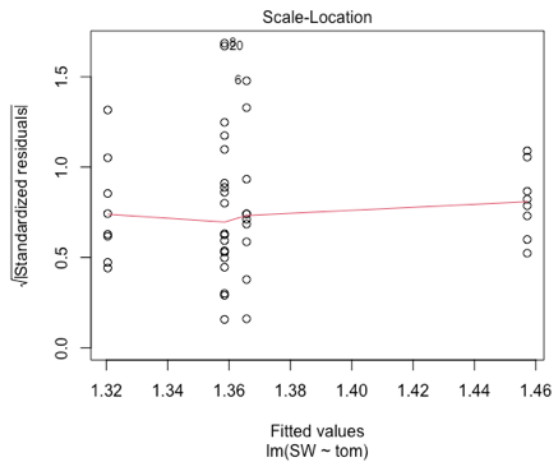
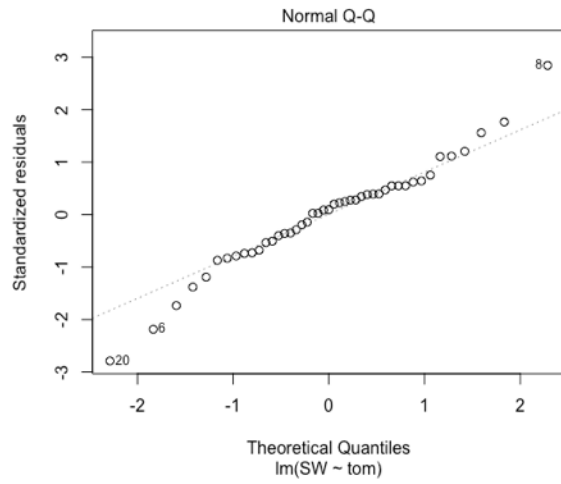
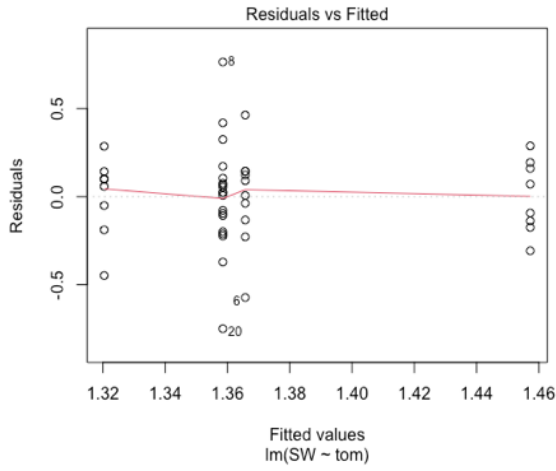
Residual plots for macroinvertebrate diversity model 2.



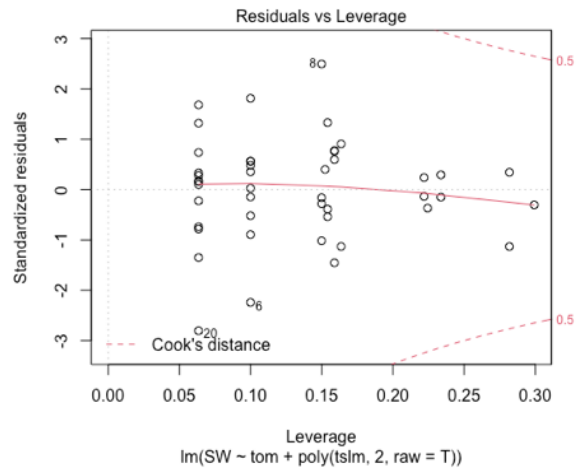
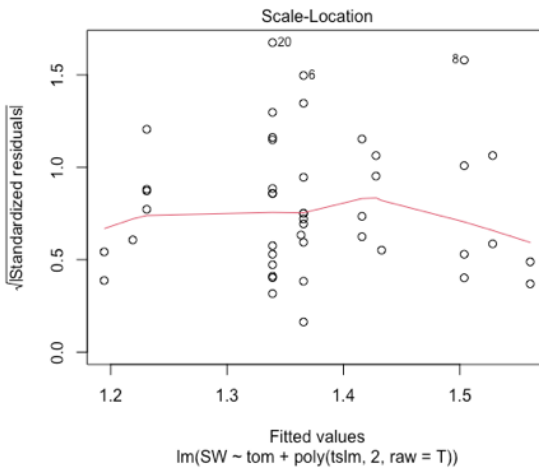
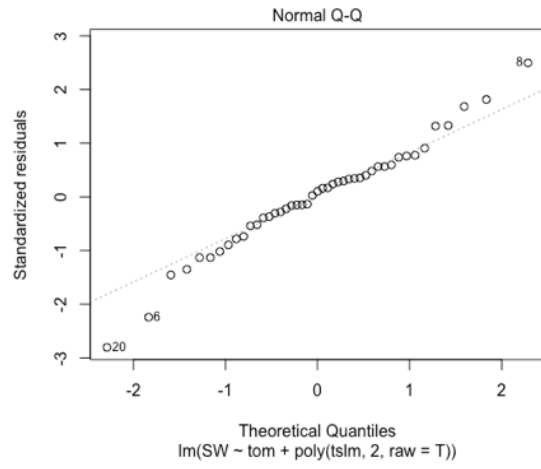
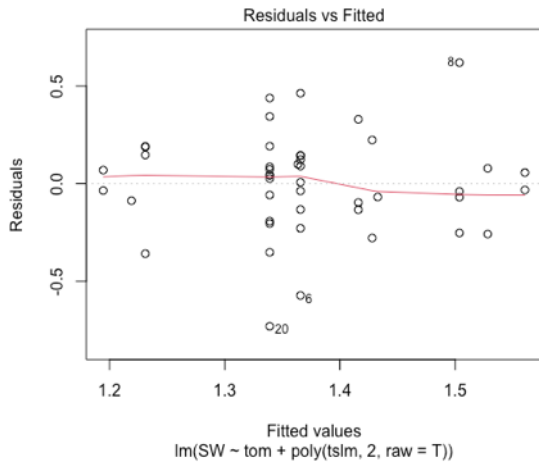
Residual plots for macroinvertebrate diversity model 4.



Residual plots for macroinvertebrate diversity model 11.

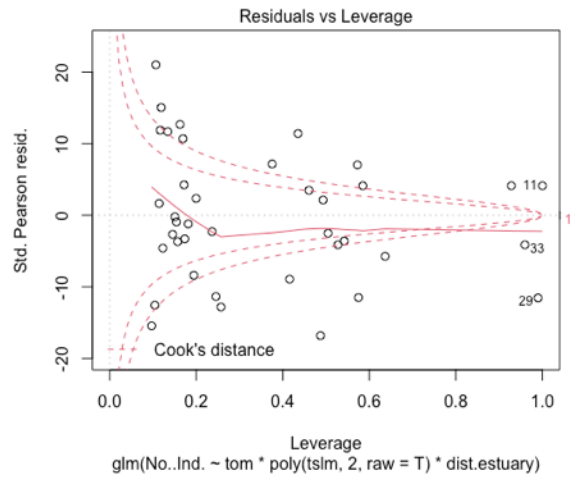
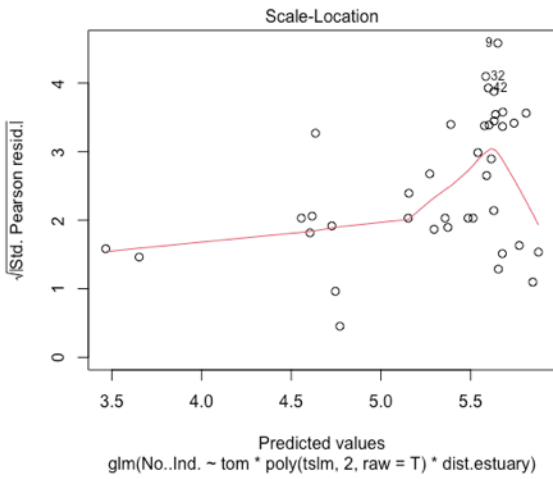
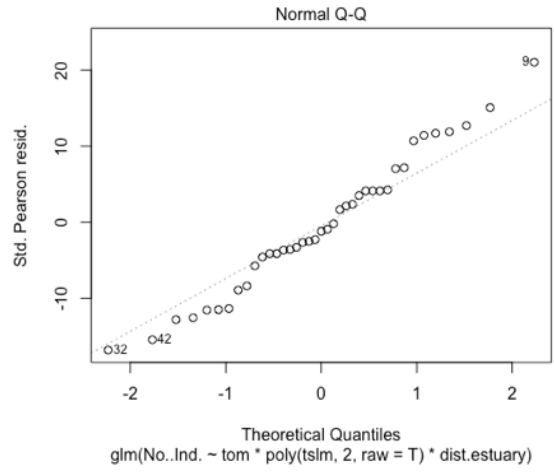
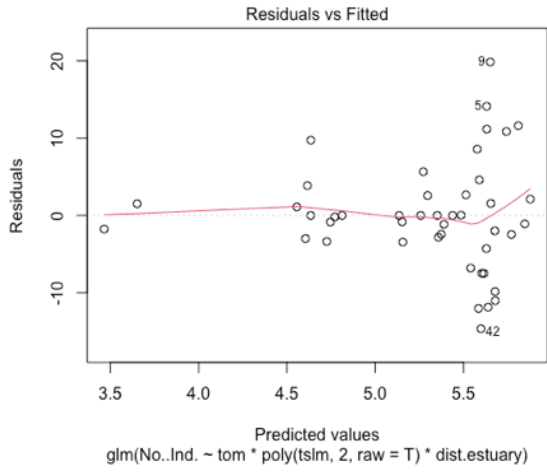


Residual plots for macroinvertebrate diversity model 12.

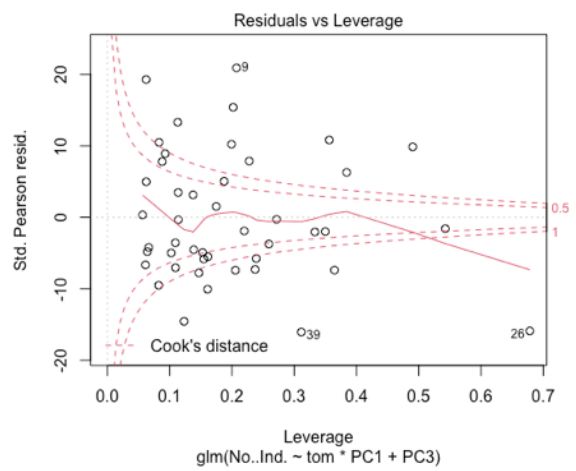
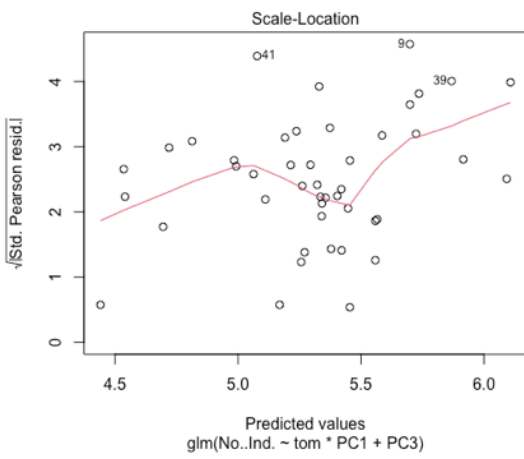
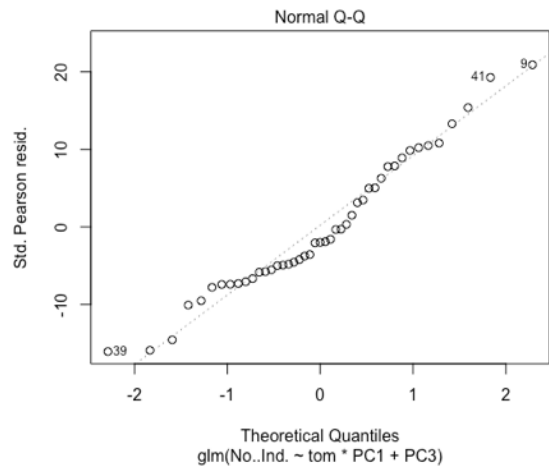
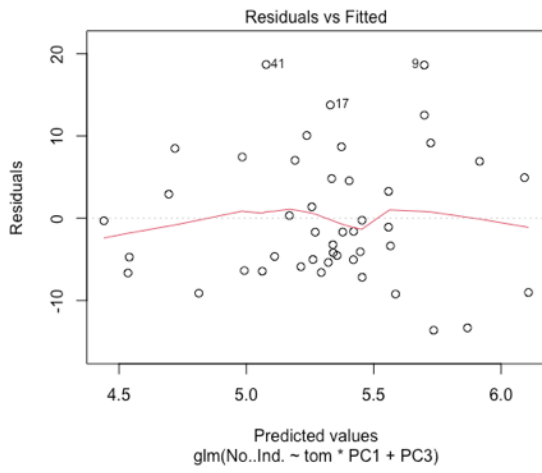




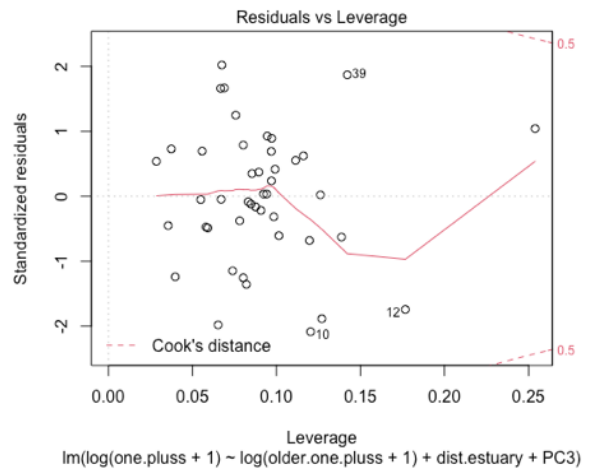
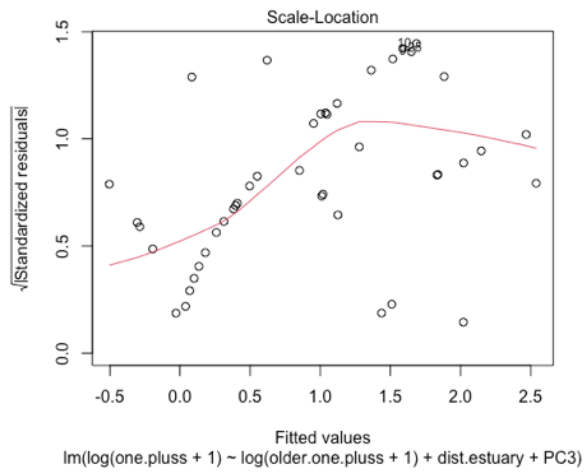
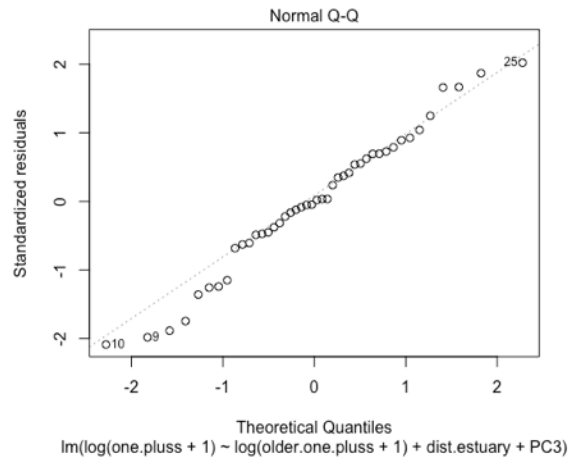
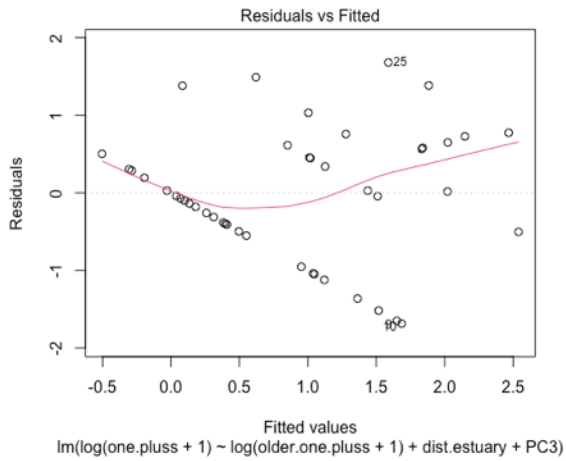
Residual plots for macroinvertebrate abundance model 1.



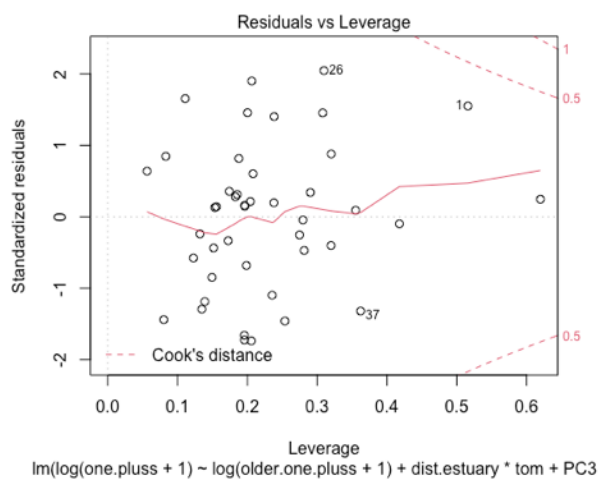
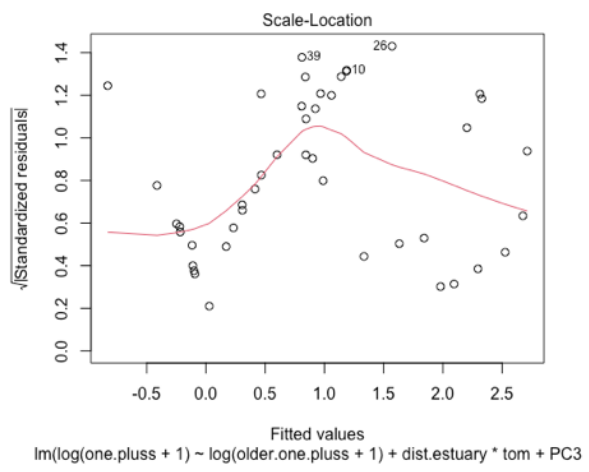
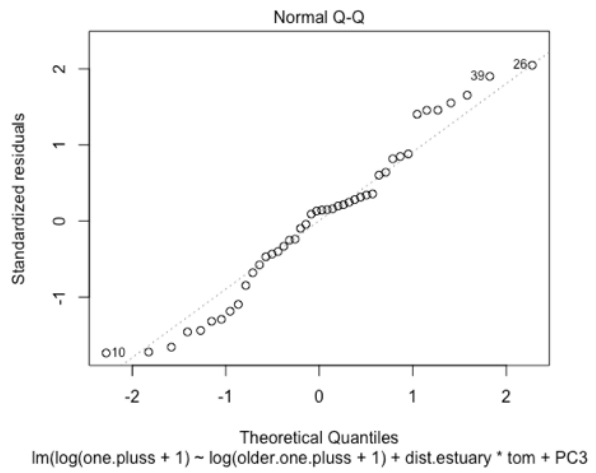
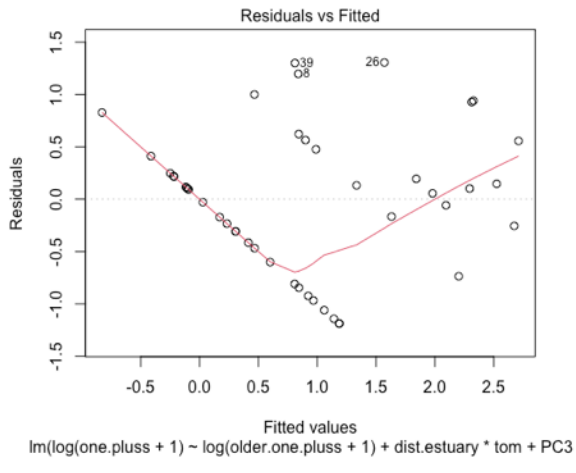
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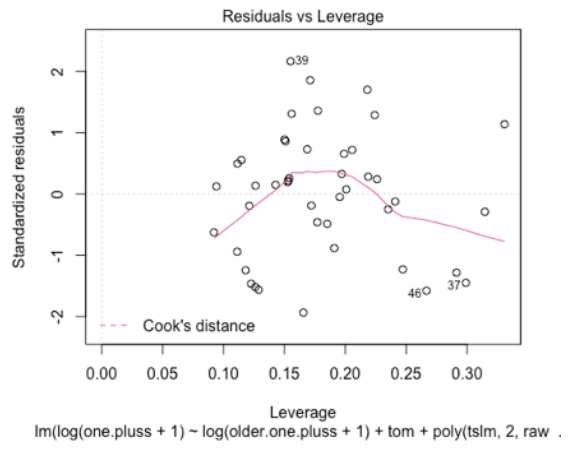
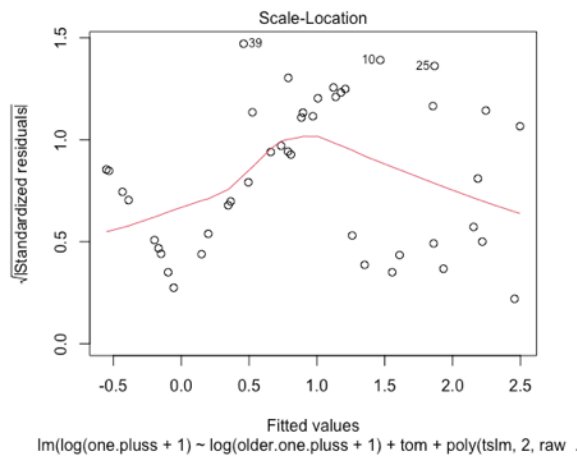
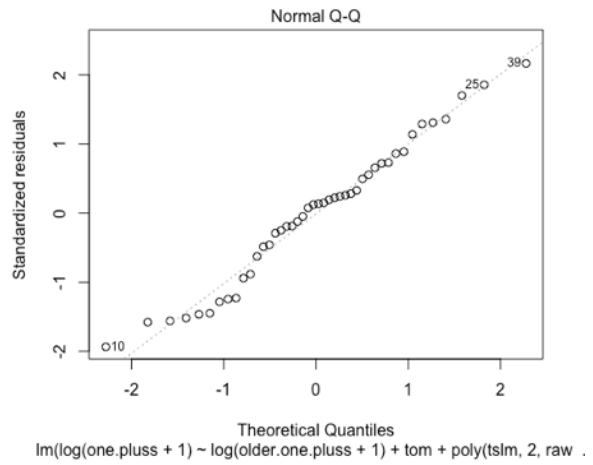
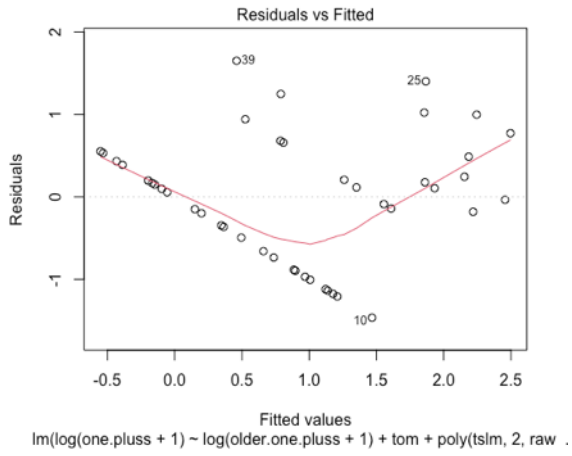
Residual plots for 1+ brown trout density model 1.



Residual plots for 1+ brown trout density model 4.



Residual plots for 1+ brown trout density model 11.



## Appendix 8

### Raw data from macroinvertebrate identification

Table 8.1. Overview of species composition in Bognely.

zone	station	No. D. ind.	No. D. fam.	No. E. ind.	No. E. sp.	No. P. ind.	No. P. sp.	No. T. ind.	No. T. sp.	No. Other ind.	No. Other. O.	No. Species	No. Ind.
1	52	4	1	0	0	0	0	0	0	423	2	3	427
1	51	104	3	0	0	10	3	2	1	398	2	9	514
1	50	144	3	2	1	8	3	0	0	430	3	10	584
2	49	46	1	40	1	13	3	0	0	5	1	6	104
2	48	39	2	22	3	18	1	0	0	45	1	7	124
2	47	62	1	120	2	28	5	4	2	44	2	12	258
3	45	60	4	34	2	30	1	0	0	4	1	8	128
3	44	66	2	16	1	56	1	4	1	6	2	7	148
3	43	90	2	88	2	4	1	0	0	10	3	8	192
3	42	114	2	92	1	42	2	24	2	2	1	8	274
3	41	58	3	4	1	6	1	2	1	0	0	6	70
3	40	126	2	0	0	0	0	0	0	14	1	3	140
3	36	120	2	34	1	38	1	0	0	10	1	5	202
4	32	14	2	1	1	19	1	0	0	1	1	5	35
4	30	162	4	128	2	78	5	14	2	22	2	15	404
4	29	105	1	2	1	28	1	1	1	10	2	6	146
4	28	190	4	90	2	41	2	7	4	15	2	14	343
4	26	9	2	8	2	46	2	2	1	12	1	8	77
4	24	9	1	28	1	49	2	3	2	17	2	8	106
4	22	14	2	29	3	57	2	2	1	14	1	9	116
4	21	92	1	26	1	66	1	0	0	58	2	5	242
5	20	46	3	124	3	58	4	8	3	76	2	15	312
5	19	301	3	197	5	51	4	29	3	42	1	16	620
5	18	106	4	16	4	10	4	6	2	42	3	17	180
5	17	164	2	8	1	28	3	4	1	32	2	9	236

6	16	26	1	376	2	52	3	8	1	4	1	8	466
6	15	71	4	344	5	56	3	22	1	22	1	14	515
6	13	28	4	144	5	25	3	5	3	5	2	17	207
6	12	28	1	132	4	39	2	5	4	7	1	12	211
6	10	10	1	6	2	3	2	1	1	28	1	7	48
6	8	8	2	0	0	4	1	1	1	9	2	6	22
6	7	10	2	112	4	21	3	2	2	25	1	12	170
6	6	74	2	90	3	34	2	0	0	32	3	10	230
7	5	20	3	132	3	31	4	3	1	16	3	15	202
7	4	30	1	60	1	48	4	2	1	10	3	10	150
7	1	28	1	54	3	31	1	0	0	10	2	7	123
8	60	7	1	32	1	58	3	1	1	5	1	7	103
8	59	18	3	376	5	66	3	22	2	22	2	15	504
8	59b	14	2	208	2	39	3	4	1	12	2	10	277
8	58	54	2	320	2	112	4	14	2	46	2	12	546
8	57	29	2	59	1	62	4	8	1	4	2	10	162
8	57b	22	4	12	2	57	2	3	2	12	4	14	106
9	63	54	3	24	2	299	4	1	1	19	3	13	397
9	61	78	3	12	1	206	3	0	0	30	2	9	326
9	56	4	1	46	3	17	2	1	1	2	1	8	70
9	55	16	1	12	2	123	4	20	1	0	0	8	171
9	54	72	2	50	2	134	4	4	1	30	3	12	290
10	65	48	2	4	1	14	3	4	1	12	3	10	82
10	64	14	3	0	0	5	2	0	0	10	2	7	29
11	67	4	1	60	3	14	2	2	1	2	1	8	82
11	68	12	3	34	2	16	2	4	2	4	2	11	70
11	69	26	2	228	6	77	3	3	2	4	2	15	338
12	71	16	1	64	3	124	4	0	0	0	0	8	204
12	72	4	3	5	2	12	1	0	0	3	2	8	24
12	73	24	1	32	1	36	3	8	2	4	1	8	104
		3094	119	4137	114	2599	137	260	63	2121	96	530	12211

Table 8.2. Overview of species composition of Diptera species in Bognelv.

zone	station	Chironomidae spp	Pediciidae spp	<i>P.dicranota</i>	Psychodidae spp	Simuliidae spp	Diptera spp	Empididae
1	52	4						
1	51	100		2				2
1	50	140		2		2		
2	49	46						
2	48	37		2				
2	47	62						
3	45	54		2		2		2
3	44	64						2
3	43	86				4		
3	42	108				6		
3	41	40		14				4
3	40	114		12				
3	36	94		26				
4	32	13		1				
4	30	146		4		10		2
4	29	105						
4	28	183		2		4		1
4	26	8		1				
4	24	9						
4	22	13					1	
4	21	92						
5	20	42			2	2		
5	19	297		2		2		
5	18	60		10	4	32		
5	17	140		24				
6	16	26						
6	15	60	2			8	1	
6	13	20		4		1		3



6	12	28					
6	10	10					
6	8	7					1
6	7	9				1	
6	6	70					4
7	5	18	1	1			
7	4	30					
7	1	28					
8	60	7					
8	59	10				4	4
8	59b	12		2			
8	58	46					8
8	57	25					4
8	57b	18		1		1	2
9	63	47		3			4
9	61	66		2			10
9	56	4					
9	55	16					
9	54	70					2
10	65	44				4	
10	64	9		2		3	
11	67	4					
11	68	6				4	2
11	69	22				4	
12	71	16					
12	72	2			1	1	
12	73					24	

Table 8.3. Overview of species composition of Ephemera species in Bognelv.

zone	station	Baetidae	<i>B.rhodani</i>	<i>B.macani</i>	<i>B.bundyaе</i>	<i>B.subalpinus</i>	<i>C.luteolum</i>	<i>A.lapponica</i>	Ephemerellidae			Siphonuridae	<i>A.inopinatus</i>
		<i>B.muticus</i>							<i>E.aurivillii</i>	<i>E.mucronata</i>	<i>E.ignita</i>	<i>Siphonuridae spp</i>	
1	52												
1	51												
1	50		2										
2	49		40										
2	48		18				2		2				
2	47		118						2				
3	45		32										2
3	44		16										
3	43		86						2				
3	42		92										
3	41		4										
3	40												
3	36		34										
4	32		1										
4	30		124						4				
4	29		2										
4	28		89							1			
4	26		7										1
4	24		28										
4	22	1	27										1
4	21		26										
5	20	2	118						4				
5	19	3	190			2			1	1			
5	18	2	10	2	2								
5	17		8										
6	16		374										2
6	15	4	328					4	6	2			
6	13	2	139	1				1					1
6	12	2	127						1				2
6	10		5										1
6	8												

6	7	2	108						1				1
6	6		86						2	2			
7	5	1	130			1							
7	4		60										
7	1	3	48										3
8	60		32										
8	59		368			2		2	2				2
8	59b		206						2				
8	58		314						6				
8	57		59										
8	57b		11									1	
9	63		22						2				
9	61		12										
9	56		44					1	1				
9	55		10					2					
9	54		46										4
10	65		4										
10	64												
11	67	4	54					2					
11	68		30					4					
11	69	4	214			3		4	1				2
12	71		52						4				8
12	72		4						1				
12	73		32										

Table 8.4. Overview of species composition of Plecoptera species in Bognelv.

zone	station	Capnidae	Nemouridae					Perlodidae	Perlodidae spp	D.nanseni	L.gramatica	Isoperla spp	Taeniopterygidae	Leuctridae
		Capnia spp	N.cinerea	N.sahlbergi	P.meyeri	A.sulcicollis	A.strandfussi	Nemouridae spp				T.nebulosa		
1	52													
1	51	2							6				2	
1	50	4							2			2		
2	49	9						1	3					
2	48	18												
2	47	18		2					2	2		4		
3	45							30						
3	44	56												
3	43	4												
3	42	40							2					
3	41	6												
3	40													
3	36	38												
4	32	19												
4	30	58	2		4				2			12		
4	29	28												
4	28	33										8		
4	26	44							2					
4	24	48							1					
4	22	56							1					
4	21	66												
5	20	2	6		18							32		
5	19	28						1	5			17		
5	18	2	2			4		2						
5	17	4	20			4								
6	16	40							8			4		
6	15	14							2			40		
6	13	20							1			4		



Table 8.5. Overview of species composition of Trichoptera species in Bognelv.

		Apatanidae					Polycentropodidae			Glossosomatidae		
zone	station	<i>Apatania</i> <i>spp</i>	<i>A. zonella</i>	Limnephilidae	<i>C.</i> <i>sahlbergi</i>	<i>Cheatopteryx</i> <i>spp</i>	<i>P. falvamaculatus</i>	<i>R.</i> <i>nubila</i>	<i>Rhyacophila</i> <i>spp</i>	<i>G. intermedium</i>	Lepidostomatidae	<i>L. hirtum</i>
1	52											
1	51	2										
1	50											
2	49											
2	48											
2	47							2	2			
3	45											
3	44									4		
3	43											
3	42							20		4		
3	41				2							
3	40											
3	36											
4	32											
4	30				8			6				
4	29	1										
4	28	1				1		2		3		
4	26			2								
4	24			2				1				
4	22										2	
4	21											
5	20	2						4		2		
5	19							24	1	4		
5	18							4				2
5	17				4							
6	16							8				

6	15						22				
6	13				1			2	2		
6	12	1		1			2				1
6	10			1							
6	8								1		
6	7						1				1
6	6										
7	5						3				
7	4						2				
7	1										
8	60			1							
8	59						16				6
8	59b						4				
8	58						8	6			
8	57			8							
8	57b	2		1							
9	63			1							
9	61										
9	56			1							
9	55						20				
9	54						4				
10	65						4				
10	64										
11	67						2				
11	68						2	2			
11	69			2			1				
12	71										
12	72										
12	73				4		4				

Table 8.6. Overview of species composition of other species in Bognelv.

zone	station	Collembola spp	Dystiscidae spp	Hydrachnidia spp	Hydraena spp	Oligochaeta	Coleoptera	Gammarus	Copepoda	Gastropoda	Turbellaria
1	52					38		385			
1	51					30		368			
1	50			4		26		400			
2	49			5							
2	48			45							
2	47			42		2					
3	45					4					
3	44			2		4					
3	43			6		2	2				
3	42			2							
3	41										
3	40			14							
3	36			10							
4	32			1							
4	30			18		4					
4	29			7		3					
4	28			12		3					
4	26			12							
4	24	1		16							
4	22			14							
4	21			54		4					
5	20			70		6					
5	19			42							
5	18	6		30		6					
5	17					20			12		
6	16			4							
6	15			22							
6	13			3		2					



6	12			7						
6	10			28						
6	8			8		1				
6	7			25						
6	6			28		2	2			
7	5			13		1			1	1
7	4	2		6		2				
7	1	1		9						
8	60			5						
8	59			20			2			
8	59b			6		6				
8	58			8		38				
8	57	1		3						
8	57b	1		4	1	6				
9	63	1		9		9				
9	61			22		8				
9	56						2			
9	55									
9	54	2		12		16				
10	65		4	4		4				
10	64			5		5				
11	67			2						
11	68			2		2				
11	69	1		3						
12	71									
12	72			1		2				
12	73					4				

## Appendix 9

Parameter estimates for model 1 macroinvertebrate diversity (table 3).

	<b>Estimate</b>	<b>Std. Error</b>	<b>t value</b>
Intercept	1.335	0.077	17.332
tslm	0.004	0.008	0.551



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