Master's thesis

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Biodiversity Revisited: Impacts of Hydropower Activities on Aquatic Insect Fauna on Decadal Timescales

Master's thesis in Natural Resources Management
Supervisor: Anders G. Finstad
Co-supervisor: Caitlin Mandeville, Gaute Kjærstad & Ivar Herfindal
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NTNU
Kunnskap for en bedre verden

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Hanne Bjørnås Krogstie
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Abstract

Earth's ecosystems are under great pressure from anthropogenic activities, and freshwater ecosystems are particularly vulnerable. A major source of impact in river ecosystems are hydropower interventions, which affect biodiversity through habitat alterations such as changes to temperature and flow regimes, sedimentation, and habitat fragmentation. There is pressure to further increase hydropower capacities globally due to growing demands for renewable energy. However, there is still little knowledge about the long-term effects hydropower development and operations have on biodiversity.

The goal of this study was to investigate the long-term effects of hydropower development on river biodiversity. Three metrics of biodiversity were used: total abundance, species richness, and species abundance. The latter was used as a measure of change in community evenness. The taxa in focus were aquatic insects belonging to the orders Ephemeroptera, Plecoptera and Trichoptera. These taxa are commonly used as bioindicators as they are the dominating invertebrate taxa in running water and are sensitive to water quality and disturbances. The study used existing data gathered by the NTNU University Museum using kick-sampling in the period 1973 to 2019, originating from nine rivers in Trøndelag, Central Norway. This data was supplemented by resampling of eight rivers during June 2021. A Before-After-Control-Impact design (BACI) was utilised, with five control rivers (located within protected watercourses) and four impact rivers (impacted by development of hydropower in the study period). The data was analysed to answer the question of whether changes in biodiversity indicators from before to after time of impact differed between control and impact rivers, and whether this differed with taxon order.

This study found indications of a general increase in total abundance and species richness from before to after time of hydropower impact, with the strength of this temporal trend varying among treatments and among taxa. Community evenness did also seem to increase. Environmental contributors to these trends could be recovery from past degradations or temperature increases caused by climate change. Habitat alterations caused by hydropower activities could also have contributed. This is supported by indications of species-specific responses to habitat alterations, such as evening out of flow. Impacts of hydropower are likely local, and the variation in data and environmental conditions suggests that increased statistical power may be necessary to register the effects on biota.

Sammendrag

Økosystemer globalt er under stort press fra menneskelige aktiviteter, og ferskvannsøkosystemer er spesielt sårbare. En viktig kilde til påvirkning av ferskvann er vannkraft, som påvirker biologisk mangfold gjennom habitatendringer som endrede temperatur- og vannføringsregimer, sedimentasjon og habitatfragmentering. Lite er imidlertid fortsatt kjent om hvilke langsiktige effekter vannkraft har på det biologiske mangfoldet. Dette er kunnskap er nødvendig for langsiktig bærekraftig utvikling og produksjon av fornybar energi fra vannkraft.

Formålet med denne studien var å undersøke langtidseffekter av vannkraftutbygging på det biologiske mangfoldet i elver. Tre ulike mål på biologisk mangfold ble brukt som respons: samlet tetthet, artsrikhet og tallrikhet av arter. Studieorganismene i fokus var akvatiske insekter tilhørende ordenene Ephemeroptera, Plecoptera og Trichoptera som ofte er brukt som bioindikatorer siden de er blant de vanligste virvelløse gruppene i rennende vann, samt er følsomme for vannkvalitet og forstyrrelser. Studien utnyttet eksisterende data fra ni elver i Trøndelag fylke, Midt-Norge. Data ble samlet inn gjennom ulike prosjekter ved NTNU Vitenskapsmuseet gjennom bruk av sparkeprøver i perioden 1973 til 2019, samt med supplerende prøvetaking i løpet av juni 2021. Det ble benyttet et før-etter-kontroll-påvirkning-design (BACI) hvor data fra fem kontrollelver (lokalisert i vernede vassdrag) ble sammenliknet med data fra fire påvirkede elver (som har gjennomgått vannkraftutbygging i studieperioden). For hver av de tre målene på biologisk mangfold ble det testet for forskjeller før og etter påvirkningstidspunktet, og om forskjeller i dette varierte mellom kontroll- og påvirkede elver, samt om dette varierte mellom insektordener.

I denne studien ble det funnet indikasjoner på en generell økning i samlet tetthet og artsrikhet mellom før og etter vannkraftpåvirkning. Styrken på denne økningen varierte imidlertid mellom kontrollelver og elver påvirket av vannkraft, samt mellom taksa. Den relative tallrikheten av arter så også ut til å bli jevnere fordelt. Mulige årsakssammenhenger kan være forbedring i vannkvalitet over tid, eller temperaturøkninger forårsaket av klimaendringer. Habitatendringer forårsaket av vannkraftvirksomhet kan også ha bidratt. Det siste støttes av ulike indikasjoner på artsspesifikke responser på habitatendringer. Vannkraft har trolig hatt lokale effekter på biologisk mangfold. Kombinert med variasjon i data og miljøforhold, er det antakelig nødvendig med økt statistisk styrke for å registrere disse effektene.

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1. INTRODUCTION

Earth's ecosystems are under great pressure from anthropogenic activities, and biodiversity is being lost at unprecedented rates (IPBES, 2019). Since biodiversity is fundamental to Earth's life support system, the rapid loss could lead to irreversible damage and move ecosystems beyond a point of no return. This is illustrated by Steffen et al. (2015) with the concept of the Earth's planetary boundaries, in which social and economic interests must operate to achieve sustainability. When anthropogenic activities move outside the boundary of what is considered safe, the risk of destabilising the Earth's systems increase. Human existence is built upon the benefits that nature provides, known as ecosystem services (Millennium Ecosystem Assessment, 2005) or nature's contributions to people (Díaz et al., 2015). Without sustainable management, ecosystems will lose their ability to provide these services partially or completely. In order to predict the consequences of human activities on ecosystem functioning and prevent degradation, more knowledge is needed on the impact these activities have on biota.

One of the biggest threats to biodiversity is currently land-use change (IPBES, 2019a). Land-use change can have diverse ecological consequences through habitat alterations, i.e., changes to the living (biotic) and non-living (abiotic) factors in an organism's environment (Davison et al., 2021). Commonly observed consequences of land-use change include reductions in the biodiversity measures species richness (Beckmann et al., 2019; Gerstner et al., 2014; Newbold et al., 2015) and species abundance (Barzan et al., 2021; Newbold et al., 2015). Habitat fragmentation typically follows land-use change and negatively affects biodiversity (Rybicki et al., 2020).

Freshwater ecosystems are particularly vulnerable to anthropogenic activities. While freshwater ecosystems only cover approximately 1 % of Earth's surface, they support around 10 % of known species (Reid et al., 2020). These ecosystems provide a number of essential ecosystem services that our societies depend upon (Kaval, 2019). This includes provisioning (drinking water, food, soil fertilisation), regulating (water filtration, flood regulation), cultural (recreation, sense of place), and supporting services (habitat provisioning, primary production). The water flowing through riverine ecosystems is also a heavily utilised resource, providing services such as irrigation, water supply, and electricity. As a result, only 23 % of large rivers worldwide are estimated to be uninterrupted by fragmentation and flow alterations (Grill et al., 2019). Pressures on these ecosystems are only increasing (Sendzimir & Schmutz, 2018), for instance, because growing energy demands and climate change have led to planned expansions in hydropower production (Reid et al., 2019).

Hydropower interventions are a major source of impact on riverine ecosystems, as they affect river biodiversity through habitat alterations (Gracey & Verones, 2016). The development of hydropower often affects temperature- and flow regimes, changes sedimentation, and leads to habitat fragmentation. Changes in flow often lead to an evening out of flow throughout the year, with reduced flood peaks. Rapid changes in water flow are often made to match demands for energy production, referred to as hydropeaking. Hydropower structures like dams and turbines create migration barriers, which can negatively affect reproduction and survival in fish (Algera et al., 2020). In Norway, hydropower is the major contributor to electrical energy production (NVE, 2022). Consequently, 70 % of watercourses are affected by hydropower development (Norwegian Environment Agency, 2020). Still, there is pressure to further increase Norway's capacity for hydropower production due to increasing demands for renewable energy both nationally and internationally (Moe et al., 2021).

As a result of the negative impact of hydropower on freshwater ecosystems, Norwegian watercourses have since 1973 been regulated by national protection plans in order to maintain a representative range of the country's watercourses and the values they contain (NVE, 2021c). The protection is mainly against hydropower development, but can also apply to other interventions (*Rikspolitiske Retningslinjer for Vernede Vassdrag [National Policy Guidelines for Protected Watercourses]*, 1994). Activities in the 1950s-1990s – including increased wastewater pollution from urban areas, large-scale regulations, and increased use of fertilizers in agriculture – negatively impacted the state of national freshwater ecosystems. Large management efforts were put into place after the 1990s to reverse the negative trends, with conditions stabilising in the period 1990-2019 according to the Nature Index for Norway (Jakobsson & Pedersen, 2020). Further improvements are still expected to follow from these efforts. However, 366 species with freshwater as their main habitat were on the Norwegian Red List for Species in 2021 (Norwegian Biodiversity Information Centre, 2021). This exemplifies that even if water quality in general has shown a positive development, more focus is needed on anthropogenic habitat alterations.

The effects of habitat alterations on biodiversity are usually divided into being density independent or dependent. Density independent (abiotic) effects of land-use change can on an individual level lead to changes in fitness (ability to survive and reproduce). Rivers are generally considered unstable ecosystems with high levels of disturbance. This high degree of stochasticity has led to suggestions that density independent factors are a major determinant for abundance and composition of species in these systems (Shiozawa, 1983). Density dependent effects include biotic interactions which can alter species communities. As an example, changes in abundance of predators or primary producers can have cascading effects through food webs. The role of density-dependent effects has received more attention in studies of riverine systems with time. This has been paralleled with an increase in knowledge of biotic interactions among riverine organisms like benthic invertebrates (Holomuzki et al., 2010).

One of the issues in inferring consequences of land-use changes on biodiversity is the lack of knowledge about long-term changes in biodiversity metrics (Cardinale et al., 2018). For example, the response of benthic invertebrates to habitat changes may depend on whether a community is still recovering from large-scale and long-term disturbances such as the last glaciation period (Saltveit et al., 1994). Norwegian watercourses may still be experiencing slow migration of species following the deglaciation after the last ice age, due to dispersal limitations of many freshwater organisms (Brabrand, 2006).

It follows that knowledge about the effects of habitat impacts, particularly in unstable systems such as running water with a high degree of stochasticity in species composition and abundance, depends on long-term investigations. However, despite that there is dire need for this knowledge in order to make sustainable management decisions about such valuable resources, there is a general lack of long-term studies in applied ecology (Willis et al., 2007). This also holds for river ecosystems and hydropower impacts. Furthermore, within environmental research the prevalence of robust study designs with randomization and/or controls is low (Christie et al., 2020). Using such designs has been recommended in order to reduce bias introduced by study design and strengthen causal inference (Christie et al., 2020; Josefsson et al., 2020).

The overall objective of the study included in this thesis was to investigate the long-term effects of hydropower development on river biodiversity. Sites including biodiversity data sampled both before and after hydropower development were used as impact treatments. The period from the first to the last sample ranged almost fifty years (1973-2021), but with variation among rivers. Sampling sites with data from the same time period, located within protected watercourses, were chosen as controls. The taxonomic focus was on aquatic insects belonging to the orders Mayflies (Ephemeroptera), Stoneflies (Plecoptera) and Caddisflies (Trichoptera).

These taxonomic groups, often shortened to “EPT”, are commonly used as bioindicators as they are common invertebrate taxon in running water and are sensitive to water quality and disturbances (Bongard et al., 2018). Previous studies have indicated that habitat changes following hydropower development negatively affect these taxa (Kjaerstad et al., 2018; Mihalicz et al., 2019).

Biodiversity can be measured in different ways. In this study, I looked at three metrics: total abundance, species richness, and species abundance curves. The latter was used as a measure for change in community evenness. Long-term data from both control and impact rivers in Trøndelag (Central Norway) was then used to investigate the trends in biodiversity before and after development of hydropower plants in the rivers. The overall objective of this thesis was answered by investigating three research questions, each focusing on one of the metrics of biodiversity:

- ***Total abundance:*** Does the total EPT abundance found per sampling event vary with treatment (*TR*; *Control, Impact*) and time (*BA*; *Before, After*)? Furthermore, does the effect of time on abundance vary with treatment, i.e., an interaction between treatment and time, and does the interaction depend on taxon order (*Order; Ephemeroptera, Plecoptera, Trichoptera*)?
- ***Species richness:*** Does the number of Ephemeroptera and Plecoptera species found per sampling event vary with treatment (*TR; Control or Impact*) and time (*BA; Before or After*)? Is there an interaction between treatment and time, and does this interaction depend on taxon order (*Order; Ephemeroptera, Plecoptera*)?
- ***Species abundance curves:*** Does community evenness, i.e., the abundance of a given species relative to other species, of the orders Ephemeroptera and Plecoptera change with time (*BA; Before, After*)? Is there a difference in the change in evenness over time between treatments (*TR; Control, Impact*)? Is there an interaction between treatment and time, and does this interaction depend on taxon order (*Order; Ephemeroptera, Plecoptera*)?

2. MATERIALS AND METHODS

The study used data from nine rivers in Trøndelag, Norway. Data was gathered by the NTNU University Museum in the period from 1973 to 2019 (Daverdin, 2022). In addition, supplementary resampling replicating existing methodology was conducted during June 2021. A Before-After-Control-Impact design (BACI) was utilised, with five control rivers and four impact rivers selected based on available biological data. The BACI design allowed testing for the effect of an interference while accounting for a possible effect of seasonality.

2.1 STUDY DESIGN

The study followed a Before-After Control-Impact (BACI) design, based on existing recommendations and principles for assessing impacts of interventions on biodiversity (Christie et al., 2020; Underwood, 1994). As insect populations tend to show large fluctuations in population size, having both before- and after-data from multiple rivers spanning several decades decreases the risk of mistaking short-term variation for long-term population trends. Including rivers not impacted by hydropower (controls) made it possible to account for other large scale environmental effects that could affect the observed results in rivers impacted by hydropower (impact), such as climate change. The analyses were further strengthened by having replicates of control- and impact rivers. This makes it possible to distinguish between environmental conditions specific to a given river and the effect of the impact. Even though there are different biotic and abiotic conditions in the nine rivers, they are expected to be similar enough for comparison because of their spatial proximity. The most interesting comparison in the BACI analysis is the interaction between time (*BA; Before, After*) and treatment (*CI; Control, Impact*). This can be interpreted as a difference in the average change in the response in impact rivers over time compared to the average change in control rivers.

2.2 STUDY AREA

The nine study rivers were located within watercourses in Trøndelag County, in Central Norway (Figure 1, Table 1). Trøndelag contains a range of different nature types, including productive agricultural areas, boreal forests, and alpine tundra (Moen, 1999). Both control- and impact rivers were found in watercourses that included these nature types. The region is situated in a temperate climate with distinct differences between seasons, where the spring flood typically marks the transition between winter and spring. Investigation of the trends in monthly abundances showed quite similar patterns for all study rivers. No distinctions were thus made regarding differences in timing of spring floods, and seasonality was controlled for in the analysis by including it as a factor. In Trøndelag, there were 36 protected watercourses in 2021 (NVE, 2021d). Most rivers do however experience some degree of anthropogenic impact regardless of protection status, as seen for both control and impact rivers in the study (Table 2). The largest impacts are still found outside of protected areas, with hydropower being the main reason waterbodies in the study area are categorized as severely modified according to the Norwegian implementation of the Water Framework Directive (Trøndelag Vannregion, 2021).

The five control rivers included in the study were Forra, Gaula, Homla, Nordelva-Holvasselva, and Verdalselva-Helgåa (Table 1, 2). Protection of the included watercourses happened at different stages in the process of developing a national protection plan, with the oldest ones being protected in 1973 (Protection plan I) and the most recent ones in 2009 (Final supplementation) (NVE, 2021d). The four impact rivers included in the study were Dalåa, Nea, Skauga and Stjørdalselva (Table 1), and these were impacted to varying degrees by hydropower (Table 2). Individual maps of each river including sampling locations are found in appendix A.

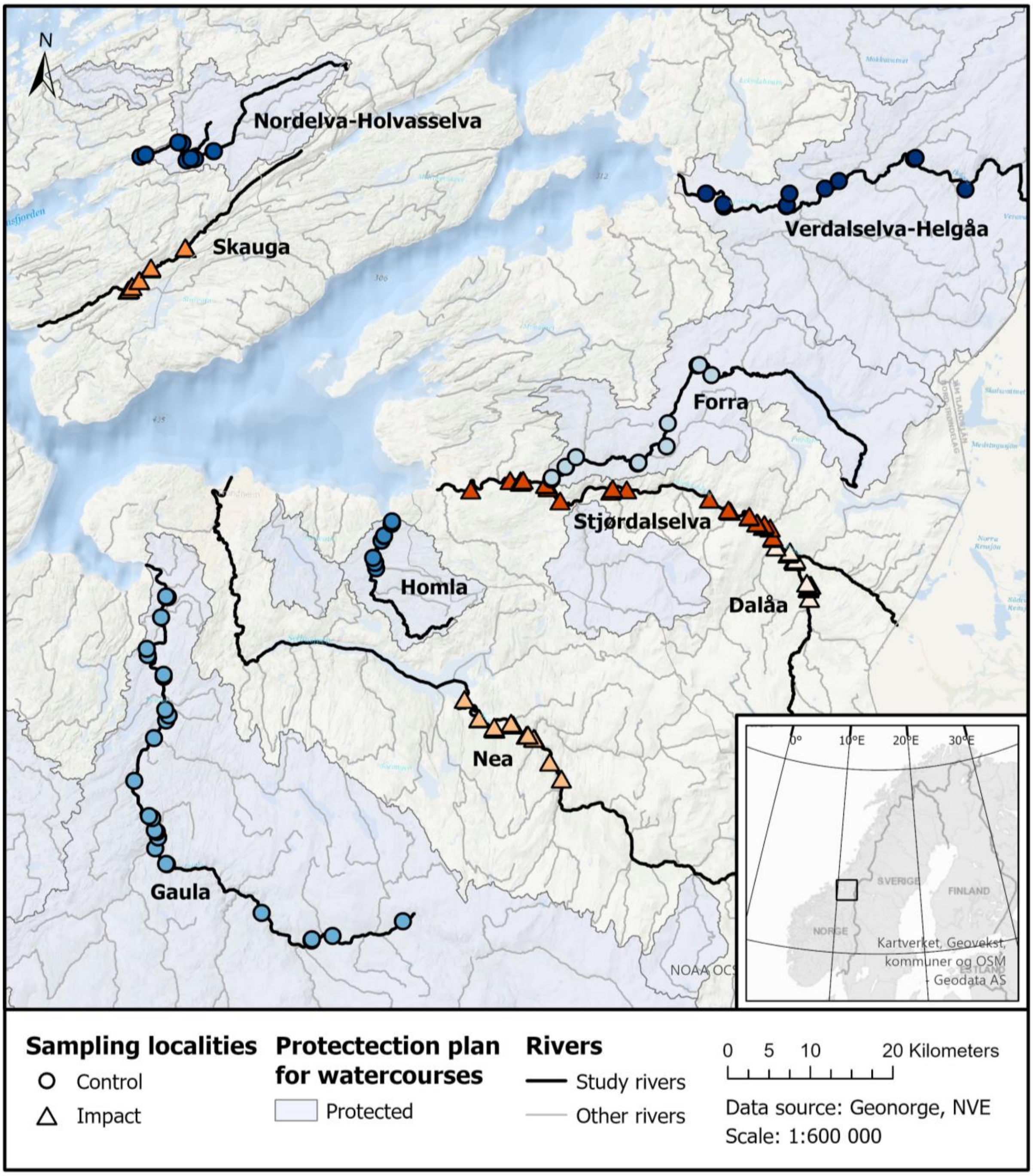

Figure 1. Overview of the study area in Trøndelag, Norway. The nine rivers included in the study, and their sampling localities. Localities within the five control rivers are marked with circles, and within the four impact rivers localities are marked with triangles. The map was drawn using ESRI ArciGIS Pro Desktop v2.8.

Table 1. Summary of the nine rivers in the study. Impacted rivers are highlighted in grey. Catchment codes are stated following the national REGINE system. Data source: Vann-nett.no (https://vann-nett.no/portal/) and NVE Atlas (https://atlas.nve.no/) accessed 10.11.2021.

River name(s)	Catchment (REGINE)	Catchment area (km^2)	River length (km)	Climate zone (moh.)	No. of localities	No. of years sampled	Sampling period
Forra	Forra (124.AZ)	604.23	47	Middle 200-800	15	22	1973-2021
Gaula	Gaula (122.Z)	3660.24	153	Low < 200	27	11	1978-1998
Homla	Homla (123.3Z)	156.09	10	Low < 200	12	4	1985-2021
Holvasselva-Nordelva	Nordelva (133.3Z)	213.62	38	Middle 200-800	12	2	1978-2021
Helgåa-Verdalselva	Verdals-vassdraget (127.Z)	1468.29	67	Middle 200-800	12	2	1979-2021
Dalåa	Dalåa (124.DZ)	187.06	25	Middle 200-800	23	14	1979-2021
Nea	Nidelv-vassdraget (123.Z)	3118.41	80	Middle 200-800	15	8	1986-2021
Skauga	Skaudals-vassdraget (132.Z)	300.6	43	Low < 200	11	7	1985-2021
Stjørdalselva	Stjørdals-vassdraget (124.Z)	2110.7	70	Low < 200	29	25	1979-2018

Table 2. Overview of all major anthropogenic sources of impact on affecting the study rivers. Impacted rivers are highlighted in grey. Level of impact is divided into small, medium, large, and unknown. The table includes all impacts occurring along the rivers, and many may have local effects only. Data source: Vann-nett.no (https://vann-nett.no/portal/), fact sheets for watercourses accessed 02.03.2022.

Impact	Forra	Gaula	Homla	Nordelva-Holvasselva	Verdelselva-Helgåa	Dalåa	Nea	Skauga	Stjørdalselva
Agriculture	-	Small, Medium	Medium	-	-	Small	Unknown	Small, Medium	-
Urban development	-	Medium, Unknown	-	-	-	-	-	Small	-
Hydropower	-	Small, Medium	-	-	-	Medium	Small-Large (SMVF)	Large (SMVF)	Medium
Fisheries & aquaculture	Small, Medium	Small, Medium	Small, Medium	Medium, Large	Small, Large	-	-	Small, Medium	Small, Medium
Introduced species & diseases	-	Medium, Unknown	-	-	-	-	Medium	-	-
Wastewater	-	Small, Medium	Medium	Small	-	-	Unknown	Small, Medium	Small, Medium
Flood protection	-	Medium, Large	Large	-	-	-	-	Large	Medium
Mining	-	Medium, Large	-	-	-	Large	Unknown	-	-
Industry	-	Medium	-	-	-	-	-	Unknown	-
Road transport	-	Large, Unknown	Unknown	-	-	-	-	-	-
Forestry	-	Medium	-	-	-	-	Unknown	-	-
Tourism & recreation	-	-	-	-	-	-	Large	-	-
Other or unknown	-	Small, Medium, Unknown	-	-	-	Large	-	-	-

2.3 BIODIVERSITY DATA

Data used in the analyses were a combination of data from the "*Limnic freshwater benthic invertebrates biogeographical mapping/inventory*" collection of the NTNU University Museum sampled in the period 1973-2019 (Daverdin, 2022), and supplementary resampling of selected localities performed during June 2021. Resampling was done based on preliminary screening of existing data in order to increase the statistical power of the comparisons. See Appendix B for the full taxon list of the final dataset.

The data contained information of the occurrence and abundance of all taxa belonging to the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT) gathered with kick-sampling. Kick-sampling is a common method used for collecting aquatic organisms found in the sediment surface (Frost et al., 1971). Sampling is done by moving perpendicular to the waterflow in a straight line while kicking the substrate and collecting loosened material with a net (frame 25 x 25 cm, mesh 0.5 mm). Each transect starts a distance from the shore to avoid sampling from recently dry areas and lasts a given interval, usually 1 minute. Big debris is sorted out in a container and aquatic organisms are preserved in ethanol (70 %). In the lab, collected material is identified to species level where possible with the use of taxonomic keys and preserved in ethanol (90 %).

Figure 2. Resampling of the rivers Verdalselva (A, B), Homla (C), and Skauga (D). During a transect (A), the upper layer of the river sediments is disturbed by kicking, and loosened benthic invertebrates are caught by the net (B). The contents in the net are place transferred to a bucket (C), and living organisms are sorted out and placed in a container with ethanol (D). Photo: Gaute Kjærstad.

2.3.1 Existing data

I accessed existing data through the Global Biodiversity Information Facility, with the search criteria being Trøndelag administrative area and scientific names being the orders Ephemeroptera, Plecoptera, and Trichoptera (GBIF.org, 2022). The majority of datapoints (80.1 %) belonged to one dataset "Limnic freshwater benthic invertebrates biogeographical mapping/inventory NTNU University Museum" (Daverdin, 2022). In order to reduce sources of variation, such as differences in sampling methodology, all other datasets were excluded.

The data was subsequently filtered with respect to quality and context. Full details of the data filtering are given in Appendix C. In brief; rivers where selected so that a minimum of five sampling localities where present in each river, with minimum of two sampling events before and two sampling events after impact (see section 2.4 for determination of impact time in control rivers). Also, rivers should be possible to categorize in either control or impact, where control rivers are located within protected watercourses unaffected by hydropower activities, and impact rivers are located outside protected watercourses and affected by hydropower activities. The rivers should not have any known major impacts from other sources. Erroneous localities (e.g., inaccurate coordinates) and winter samples were removed.

The data collected by the NTNU University Museum originates from different projects (Appendix D). Several of the older datapoints for protected watercourses belong to the dataset "10-year protected watercourses [10-års vernede vassdrag]". This data was collected following the Norwegian Parliament's decision in 1973 of temporarily protecting 57 watercourses in Norway for ten years, to allow more detailed assessments of interests before substantial impacts were implemented (NVE, 2021e).

2.3.2 Resampling

Resampling using kick-sampling was done in eight rivers with five localities each between the 7th and the 11th of June 2021 (Figure 3, appendix E). Sites were selected for resampling based on existing data and on an assessment of whether resampling could increase power of the subsequent analyses. Also, logistical constraints such as safety or distance to road were considered. See appendix C for the full set of details regarding selection of localities for resampling. The methodology of resampling replicated the original methods used. Accordingly, to reduce sources of variation, each transect of the resampling lasted either 1 minute or 5 minutes depending on previous sampling at the locality. The following literature was used for identification in the lab: Ephemeroptera (Arnekleiv, n.d.; Nilsson, 1996), Plecoptera (Lillehammer, 1988; Nilsson, 1996), and Trichoptera (Rinne & Wiberg-Larsen, 2017). Because of time limitations, material from river Gaula was not sorted.

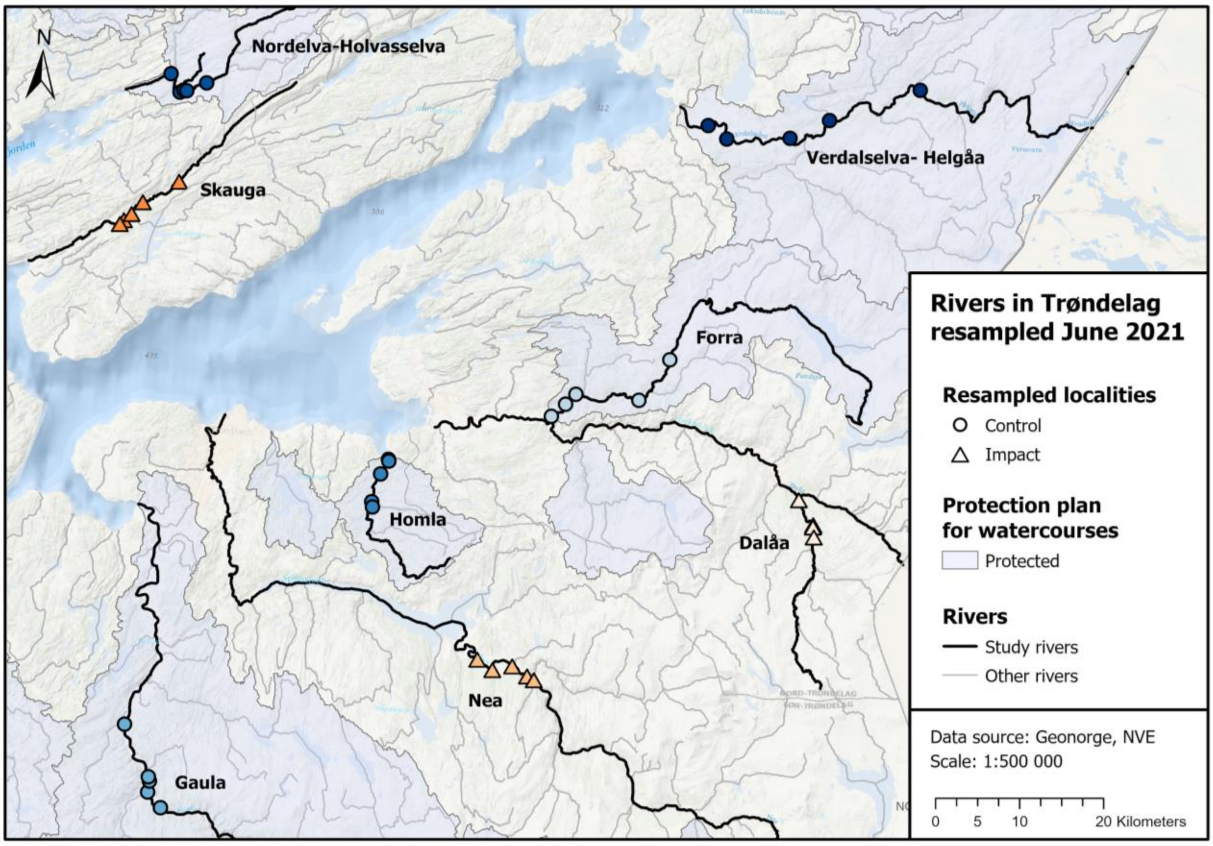

Figure 3. Overview of the 40 localities in Trøndelag resampled during June 2021, five in each river. Localities in protected watercourses are shown as circles while those in unprotected areas are shown as triangles. The map was drawn using ESRI ArciGIS Pro Desktop v2.8.

2.4 DATA ANALYSIS

Data was filtered and analysed in R version 4.1.1 (R Core Team, 2021). After filtering, the data was further analysed to investigate how changes in biodiversity from before to after impact differed between control and impact rivers. I did this by looking at the three different metrics of biodiversity: total abundance, species richness, and species abundance curves. For total abundance, I ran the analysis on the full dataset including all taxa. However, for the analyses on species richness and abundance curves, only records identified for the orders Ephemeroptera and Plecoptera were included. Trichoptera were excluded from the last two analyses as older datapoints were not identified to the finest taxonomic resolution (1970s-1980s to order only).

In order to utilise data from rivers with different years of hydropower impact in the study (establishment of hydropower regulation in the rivers: 1989, 1994, 1994, 2012) – sampling time was transformed to a number relative to year of impact. For control rivers, the median year of impact in regulated rivers (1994) was set as the impact year. I then grouped the data into the categories before or after, with before < 0 and after ≥ 0. With this method one must keep in mind that time (*BA*) does not reflect a strictly linear timeline but is relative to time of impact.

In order to assess which variables that best explained the observed variation in the data, I created a set of candidate models and compared their fit using the Akaikes Information Criterion corrected for small sample sizes (AICc, Burnham & Anderson, 2002). This gives each model a score relative to the other candidate models. The score is based on rewarding goodness of fit (based on log-likelihood), and penalising model complexity to avoid overfitting. The most parsimonious model is identified as the one with the lowest AICc score. I followed Burnham and Anderson (Burnham & Anderson, 2004), and considered models with ΔAICc < 2 to have substantial support. Candidate models were obtained with the "dredge" function ("MuMIn" package in R, Barton, 2020) when possible, and when not possible a set of candidate models was created manually.

Due to interdependence in the data caused by repeated samples for each river and time, all statistical models for total abundance and species richness included sampling localities nested within rivers as a random effect. Preliminary analyses indicated that there were strong seasonal patterns in abundance and species richness. Therefore, season was also included as a main effect in candidate models for these two analyses. In the statistical models for species abundance curves, only river was included as random effects because these analyses utilised data aggregated across sampling localities within river as the response variable.

To investigate if the interaction between time and treatment differed between the orders, an alternative global model was created for each analysis, where order was included as a factor with three (two) levels (*Order; Ephemeroptera, Plecoptera, Trichoptera*). To control for order-specific variations in abundances between seasons, an interaction between seasonality and order (*Seasonality*Order*) was added to the global model for the analyses of total abundance and species richness. All estimated differences and confidence intervals were given on the log scale, with spring as reference level in analyses including seasonality (*Seasonality; spring, summer, fall*).

2.4.1 Total abundance

Abundance data (sum of individual counts) is commonly analysed with a Poisson distribution (Hilbe, 2014), but because my data was highly overdispersed (mean = 200, variance = 61149) I used a negative binomial distribution with a log-link function. A global mixed effects model (I) was fitted using the function "glmer.nb" (package "lme4" in R, Bates et al., 2015) to estimate the dispersion parameter theta (θ).

(I) *Total abundance ~ BA*TR + Seasonality*

A set of candidate models was created based on the global model (I), all fitted with the "glmer" function (from "lme4" in R, Bates et al., 2015) with a negative binomial distribution and a fixed dispersion parameter from the global model (θ = 1.243). The set of models included all combinations of time (*BA; Before, After*) and treatment (*TR; Control, Impact*), and each model included the additive effect of *Seasonality* due to preliminary analyses supporting its importance. If an interaction was included, so were all individual terms of the interaction. Model selection with AICc was performed, and models with ΔAICc < 2 were refitted with "glmer.nb" without fixing the dispersion parameter and evaluated further.

To investigate if the interaction between time and treatment differed between orders, I split the data on abundance into the three orders and included *Order* as an explanatory variable. The global model (II) now included a three-way interaction between *BA*, *TR* and *Order* and a two-way interaction between *Order* and *Seasonality*. The fixed dispersion parameter θ was 1.119 for the global order-specific model for abundance.

*(II) Total abundance ~ BA*TR*Order + Seasonality*Order*

I then ran model selection as described previously, retaining *Seasonality*Order* in all models. If an interaction was included, so were all lower order interactions and terms.

2.4.2 Species richness

I analysed the species richness with a generalised linear mixed effects model ("glmer" function from the package "lme4" in R, Bates et al., 2015). The number of species was count data with almost equal mean and variance (mean = 6.9, variance = 6.7), therefore a Poisson distribution with a log-link function was used. I first analysed species richness for all species in Ephemeroptera and Plecoptera pooled, with a global model including an interaction between *BA* and *TR* and an additive effect of *Seasonality*.

*(III) Species richness ~ BA*TR + Seasonality*

Next, I included order to the global model (IV) to investigate whether the number of species belonging to Ephemeroptera and Plecoptera differed in their response to treatment over time.

*(IV) Species richness ~ BA*TR*Order + Seasonality*Order*

2.4.3 Species abundance curves

The abundance of each species found in a river at a given time (*Before* or *After* treatment) was calculated as the natural logarithm of the average number of individuals found per sampling event (ln(abundance+1/events)). Adding 1 to the abundance was done to avoid taking ln(0) which is undefined. This allowed me to include species that were present during only one of the two periods in a river. Dividing on the number of sampling events in the time-period was done to control for differences in sampling effort.

In a stable community, each species should in general be as common before impact as after impact. Such a community should therefore have a slope describing the relationship between abundance of species at two timepoints equal to one, with intercept of zero. This can be tested with a normal linear regression between abundance *After* against abundance *Before*. If the slope is greater than one, species that were rare before impact could have decreased in abundance, species that were common before impact could have increased in abundance, or both. A slope smaller than one could indicate that the community has become more even, with an increase in abundance for species that were rare before impact, a decrease in abundance for species that were common before impact, or both. If all species increased in abundance over time, the intercept would be greater than zero while keeping the slope at one. If all species decreased in abundance, the intercept would be smaller than zero while keeping the slope constant.

To test whether there was a difference between control and impact rivers in the change in species abundances from before to after impact, I included the interaction between treatment (*TR*) and abundance *Before* in a global linear mixed-effects model (V). River was included as a random effect, and AICc-based model selection was performed on models fitted using the “lmer” function from the package “lme4” (Bates et al., 2015), with REML = false. The best model was refitted with REML = true to get parameter estimates.

(V) *After ~ Before*TR*

Lastly, I tested if the relationship between species abundances before and after impact varied with order and treatment by fitting a global model including a three-way interaction between abundance *Before*, treatment (*TR*) and order (*Order*) (VI).

(VI) *After ~ Before*TR*Order*

3. RESULTS

The final selected data provided information on a total of 4452506 individuals of Ephemeroptera, Plecoptera, and Trichoptera (EPT) from nine rivers, gathered during 2263 sampling events at 156 localities from 1973 to 2021. The mean number of EPT individuals caught per event (abundance) was 147 ± 8.5 (SE) in control rivers and 214 ± 6.1 (SE) in impact rivers. The mean abundance per event varied with order, with Ephemeroptera being the most abundant group (Figure 4).

A total of 45 unique Ephemeroptera and Plecoptera species were found in control rivers while 53 were found in impact rivers. The mean number of species found per event was 7 ± 0.1 (SE), with higher averages for Ephemeroptera than Plecoptera (Figure 5). Within the order Ephemeroptera, the genus *Baetis* dominated in all rivers, both before and after impact. Within the order Plecoptera, *Amphinemura* and *Capnia* were the most commonly occurring genera in control rivers. In impact rivers, the two most common genera shifted from *Amphinemura* and *Diura* to *Amphinemura* and *Capnia* (Figure 18 & 19, appendix F).

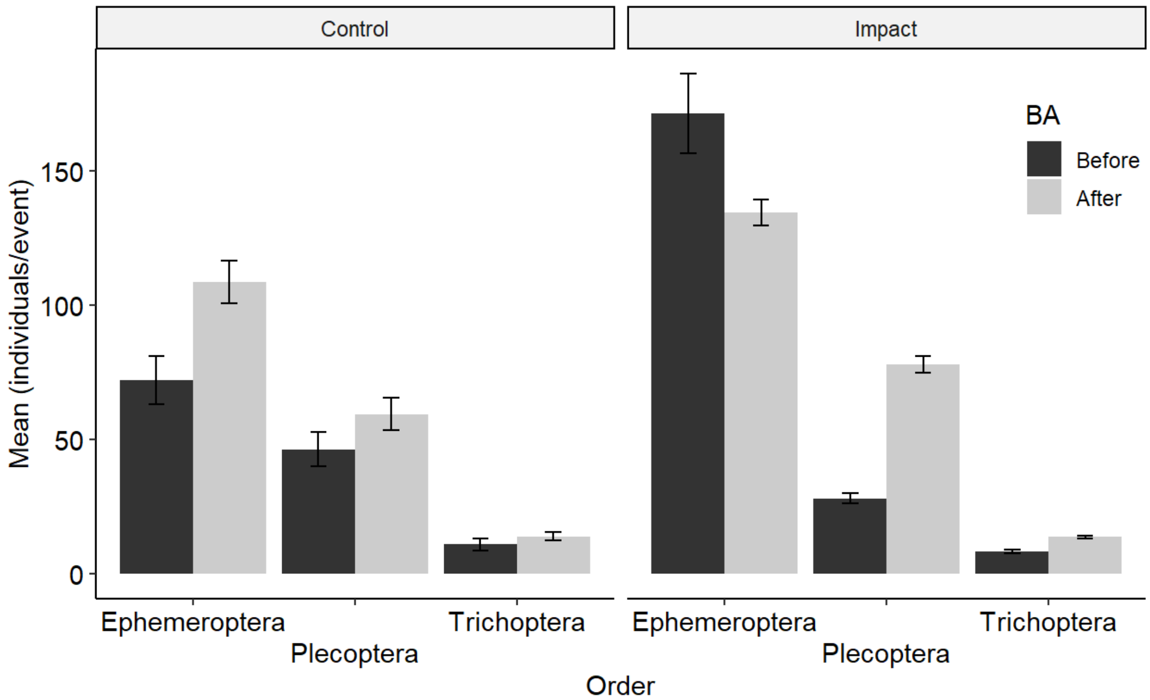

Figure 4. Mean abundance (mean number of individuals per sampling event ± SE) for the orders Ephemeroptera, Plecoptera, and Trichoptera before and after impact of hydropower development. The figure is based on data gathered in nine rivers in Trøndelag, from 1973 to 2021.

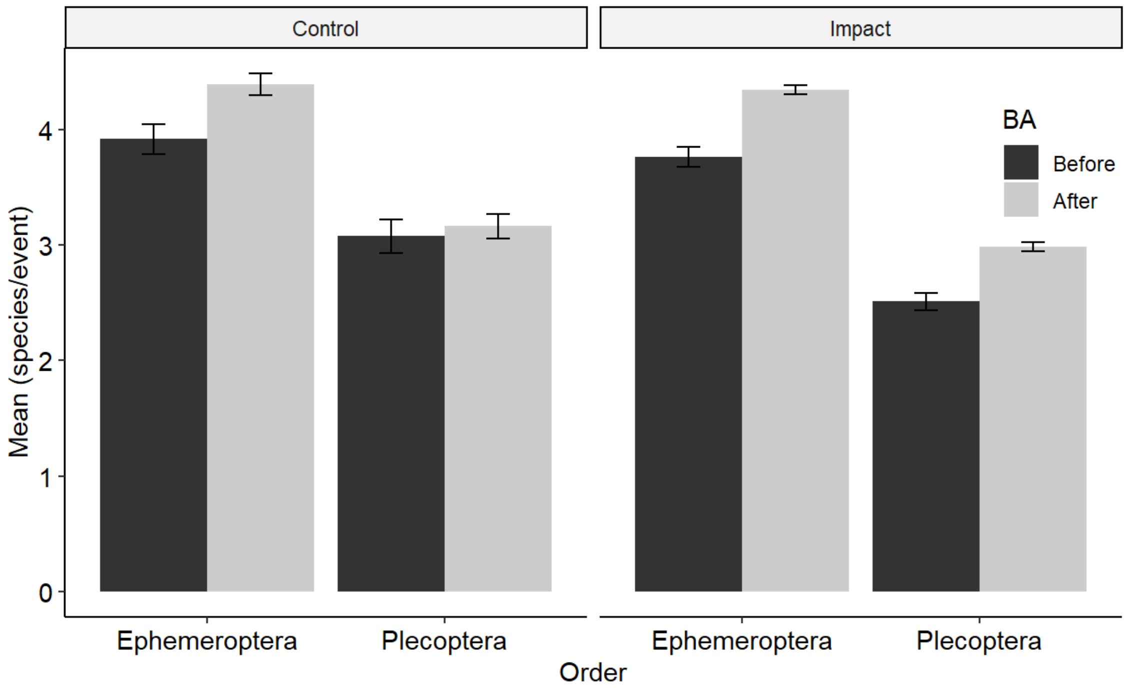

Figure 5. Mean number of species per event (± SE) shown for the orders Ephemeroptera and Plecoptera, before and after impact of hydropower development. The figure is based on data gathered in nine rivers in Trøndelag, from 1973 to 2021.

3.1 Total abundance

The total abundance of EPT varied with time (*BA*) and treatment (*TR*). Abundances were highest during fall and lowest during summer. Model selection gave support to two candidate models (ΔAICc < 2, Table 5). The highest ranked model included time (*BA*), treatment (*TR*), the interaction between time and treatment, and seasonality (*Seasonality*) (Figure 6A-B). The second-highest ranked model did not include the interaction between time and treatment. The evidence ratio pointed to solid support for the highest ranked model compared to the second highest (0.495/0.288 = 1.7), and I therefore chose the highest ranked model for further inference.

According to the highest ranked model, the total abundance increased from before to after treatment in control rivers, with strong support (95 % CI [0.10, 0.75]). In impact rivers the abundance also increased, but only with weak support (95 % CI [-0.01, 0.24]).

When taxon order was included into the full model, it became evident that the interaction between time and treatment also depended on order. The highest ranked model was the full model, including a three-way interaction between time (*BA*), treatment (*TR*) and order (*Order*) as well as a two-way interaction between order (*Order*) and season (*Seasonality*) (Table 5,

Figure 6C-D). There was strong support for an increase in the total abundance of Ephemeroptera and Trichoptera in control rivers (95% CI: Ephemeroptera [0.07, 0.73]; Trichoptera [0.09, 0.80]). Abundances of Plecoptera remained stable in control rivers (weak support for decrease: 95 % CI [-0.34, 0.33]). In impact rivers, there was strong support for an increase in abundance of both Plecoptera and Trichoptera from before to after the impact (95 % CI: Plecoptera [0.51, 0.78], Trichoptera [0.47, 0.76]). Ephemeroptera did however appear to decrease in abundance, but this trend had only weak support (95 % CI [-0.22, 0.03]).

Table 5. Model selection tables for models describing the relationships between time (*BA*), treatment (*TR*), seasonality (*Seasonality*), and total abundance. The model selection was conducted using two sets of full models, with or without taxon order (*Order*) included as an explanatory variable. The best model was selected based on Akaikes Information Criterion adjusted for small sample sizes (AICc). Only models with ΔAICc < 2 are presented. The best model according to AICc is shown in bold. Number of model parameters (*K*) and the Akaike weight of evidence (W_i) in support of model *i* are also given. Models were fitted with a negative binomial distribution using the function "glmer" in R, and a fixed dispersion parameter (θ = 1.243 for total abundance, θ = 1.119 for order-specific model for abundance).

Model	AICc	ΔAICc	*K*	w_i
Total abundance				
BA*TR + Seasonality	**27603.9**	**0.00**	**9**	**0.495**
BA + TR + Seasonality	27605.0	1.08	8	0.288
Order-specific model of abundance				
BA*TR*Order + Seasonality*Order	**58641.8**	**0.00**	**21**	**> 0.999**

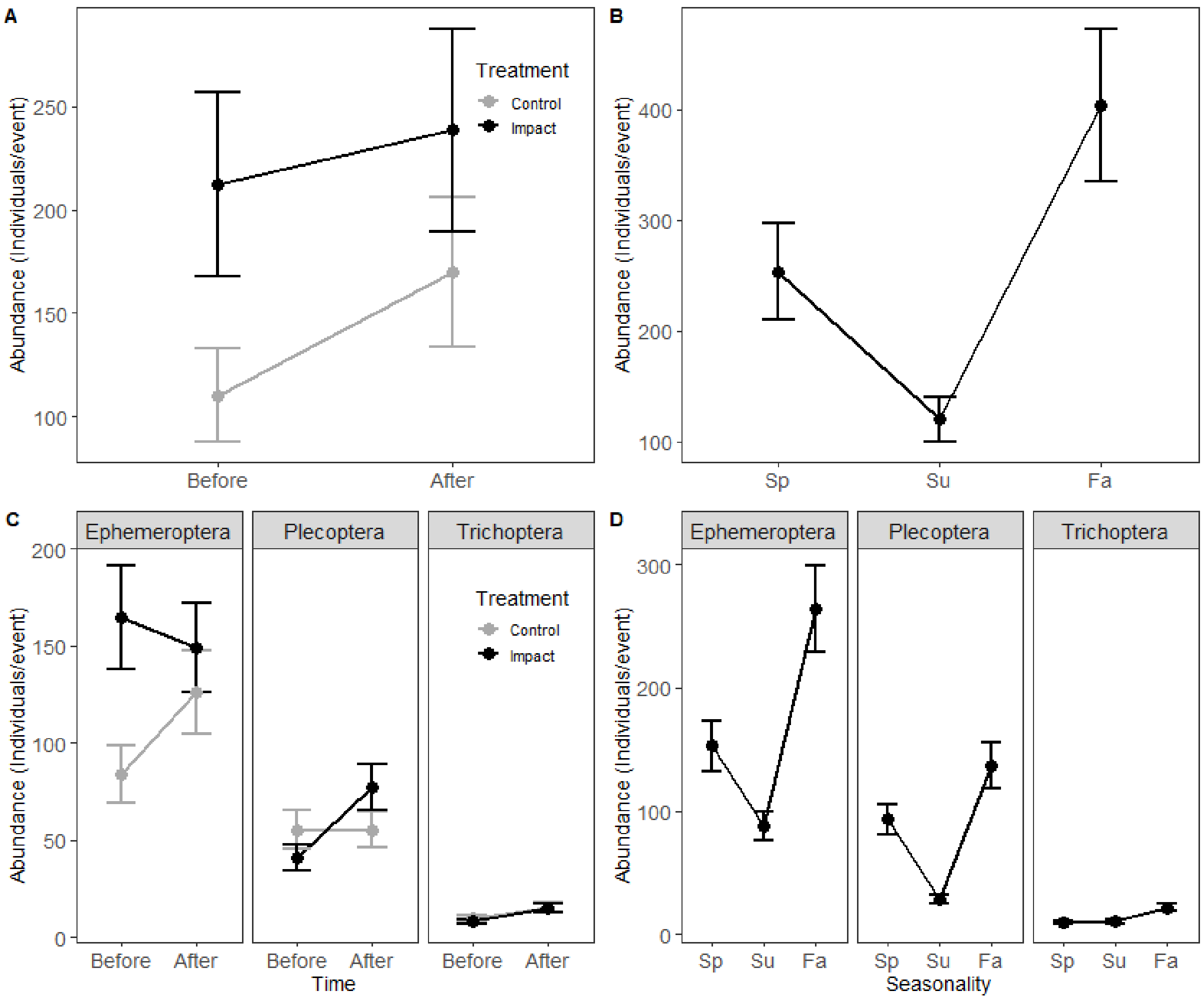

Figure 6. Effect plots for the best models describing the total abundance of Ephemeroptera, Plecoptera, and Trichoptera as the mean number of individuals/event (± SE). The first two figures show how the total abundance varies with time and treatment (A), and with seasonality (B) where sampling periods are categorized into spring (Sp), summer (Su), and fall (Fa). The last two figures show how order affects the relationship between time and treatment (C), and how the effect of seasonality on abundance varies with order (D).

3.2 Species richness

Species richness generally increased over time. The highest species richness was observed in spring and the lowest during summer. Model selection gave support to three candidate models (ΔAICc < 2, Table 6). The highest ranked model included only time (*BA*) and seasonality (*Seasonality*). The second-highest ranked model was the full model including time (*BA*), treatment (*TR*), their interaction and seasonality (*Seasonality*) (Figure 7A-B). The third-highest ranked model included time (*BA*), treatment (*TR*) and seasonality (*Seasonality*). Although the evidence ratio supported the highest-ranked model (0.538/0.245 = 2.2), also the second-highest ranked model was included in the final results since a possible difference between treatments was of interest even if there was not as strong confidence for the effect.

There was strong support for an overall increase in species richness from before to after impact according to the highest ranked model (95% CI [0.05, 0.15]). However, the second-best model indicated that while there was strong support for increased species richness in impact rivers (95 % CI [0.07, 0.17]) there was only weak support for such an increase in control rivers (95 % CI [-0.11, 0.14]). When including taxon order (*Order*) into the full model for species richness, there was evidence suggesting that the interaction between time (*BA*) and treatment (*TR*) depended on order (*Order*). Model selection supported two candidate models (ΔAICc < 2, Table 6). The highest-ranked model was the full model that included a three-way interaction between time (*BA*), treatment (*TR*), and order (*Order*) and a two-way interaction between order (*Order*) and seasonality (*Seasonality*) (Figure 7C-D). The second-highest ranked model differed from the highest by not including this three-way interaction. The evidence ratio of the highest ranked model compared to the second highest was high (0.475/0.190 = 2.5), and I therefore chose the highest ranked model for further inference. Both Ephemeroptera and Plecoptera increased in species richness in the impact rivers (95 % CI: Ephemeroptera [0.06, 0.19], Plecoptera [0.01, 0.18]). Species richness did also increase for Ephemeroptera in control rivers, but with weaker support (95 % CI [-0.05, 0.20]). In contrast, Plecoptera did decrease in control rivers, but this trend only had weak support (95 % CI [-0.28, 0.004]).

Table 6. Model selection tables for models describing the relationships between time (*BA*), treatment (*TR*), seasonality (*Seasonality*), and species richness. The model selection was conducted using two sets of full models, with or without taxon order (*Order*) included as an explanatory variable. The best model was selected based on Akaikes Information Criterion adjusted for small sample sizes (AICc). Only models with ΔAICc < 2 are presented. The best model according to AICc is shown in bold. Number of model parameters (*K*) and the Akaike weight of evidence (W_i) in support of model *i* are also given. Models were fitted with a Poisson distribution using the function "glmer" in R.

Model	AICc	ΔAICc	*K*	w_i
Species richness				
BA + Seasonality	**9775.8**	**0.00**	**6**	**0.538**
BA*TR + Seasonality	9777.3	1.58	8	0.245
BA + TR + Seasonality	9777.6	1.82	7	0.217
Order-specific model of species richness				
BA*TR*Order + Seasonality*Order	**14885.2**	**0.00**	**14**	**0.475**
BA*Order + BA*TR + Order*TR + Seasonality*Order	14887.0	1.83	13	0.190

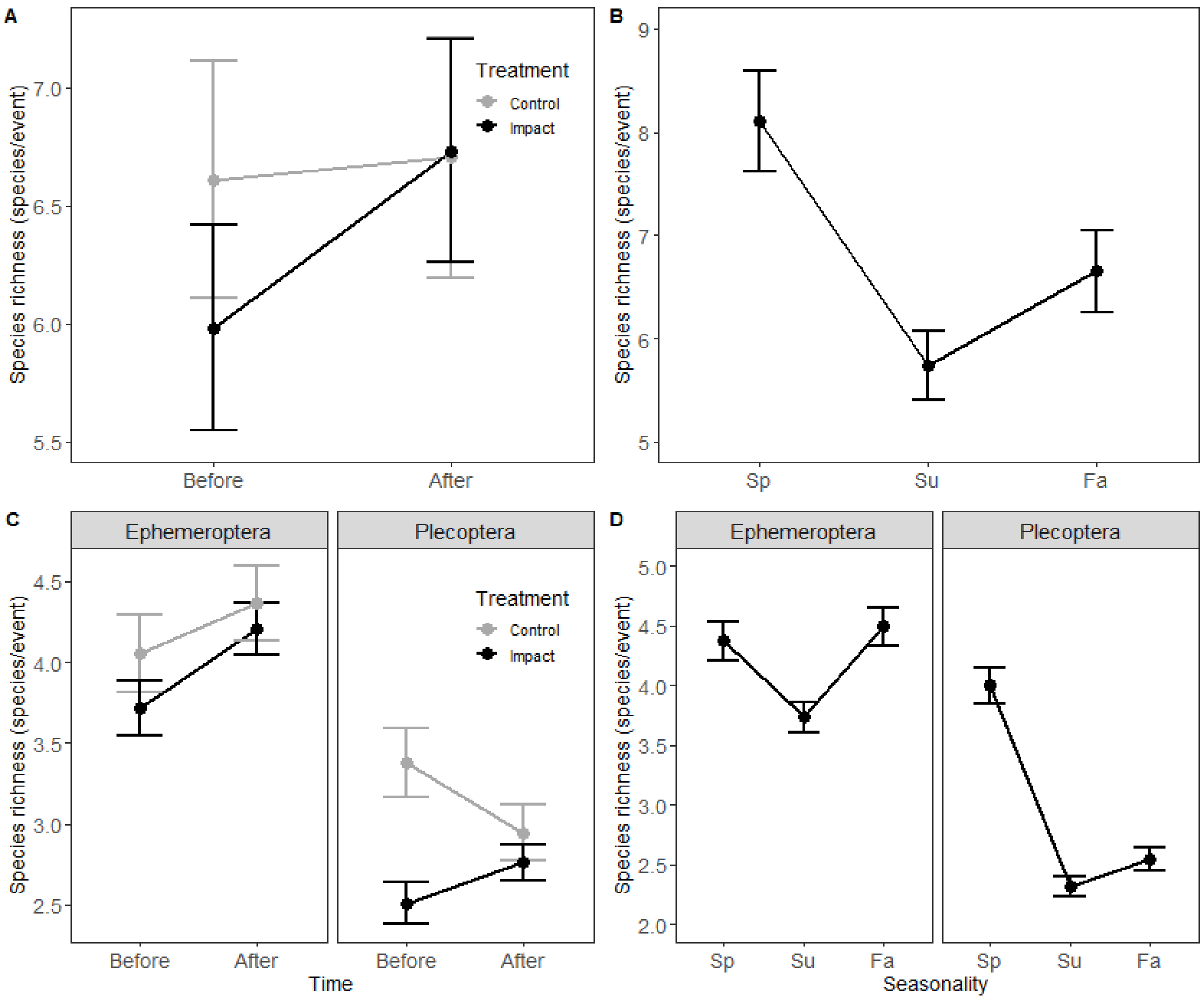

Figure 7. Effect plots for the second-best and best models describing the species richness of Ephemeroptera and Plecoptera as the mean number of species/event (± SE). The first two figures show how species richness varies with time and treatment (A), and with seasonality (B), where sampling periods are categorized into spring (Sp), summer (Su), and fall (Fa). The last two figures show how order affects the relationship between time and treatment (C), and how the effect of seasonality on species richness varies with order (D).

3.3 Species abundance curves

Insect communities in both the control- and impact rivers appear to become more even over time. This is indicated by the overall slopes of species abundance before treatment regressed against species abundance after treatment being below 1. Model selection supported three candidate models (ΔAICc < 2, Table 7). The highest ranked model included the interaction between abundance before impact (*Before*) and treatment (*TR*), supporting that the relationship between species abundances before and after impact differed between treatments (Figure 8). I chose the highest ranked model for further inference, based on the fairly strong support from the evidence ratio between the highest- and second-highest ranked model (0.435/0.289 = 1.5).

The highest ranked model indicated that insect communities in control rivers became more even over time (slope = 0.66) compared to communities in impact rivers (slope = 0.82), but with weak support (95 % CI [-0.02, 0.35]). Slopes for both treatments were lower than one, with strong support (95 % CI: control [-0.49, -0.19], impact [-0.29, -0.06]). Furter, intercepts were not different from 0.

Lastly, model selection for the alternative model including taxon order (*Order*) as explanatory variable supported five candidate models (ΔAICc < 2, Table 7). There was weak support for a three-way interaction between abundance before impact (*Before*), treatment (*TR*) and order (*Order*). The model including this three-way interaction was only the fourth-highest ranked. Therefore, whether the relationship between abundance before impact (*Before*) and treatment (*TR*) varied with order was not investigated further.

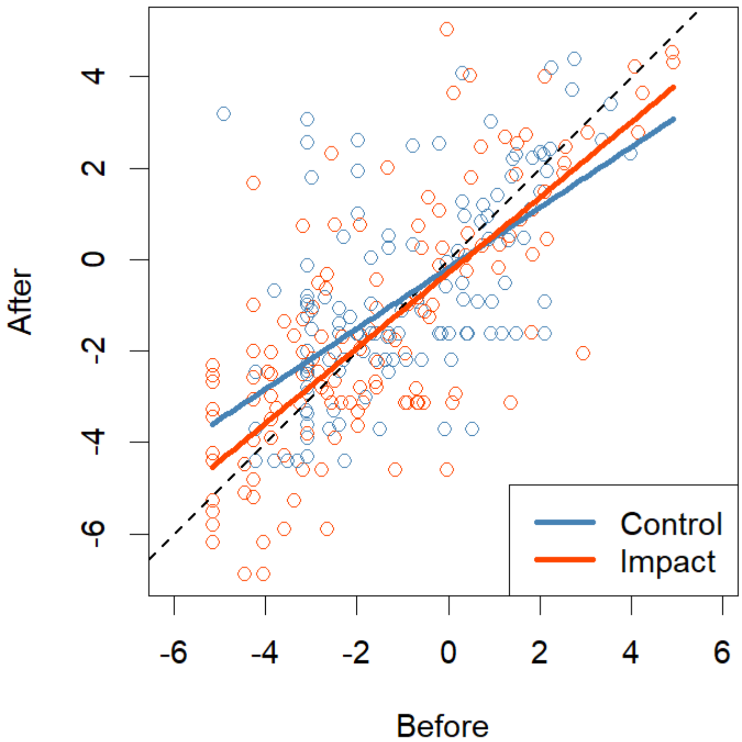

Figure 8. The abundance of individual species found in each river, with abundance after impact plotted against abundance before impact. Abundance was calculated as log(abundance+1/events), data from control rivers is shown in blue, and impact rivers is shown in red. If there is no change in abundance, the slope should follow the dotted 1:1 line. Regression lines were extracted from the linear mixed-effects model.

Table 7. Model selection tables for models describing the relationships between abundance of individual species before impact (*Before*), treatment (*TR*), and abundance of individual species after impact. The model selection was conducted using two sets of full models, with or without taxon order (*Order*) included as an explanatory variable. The best model was selected based on Akaikes Information Criterion adjusted for small sample sizes (AICc). Only models with ΔAICc < 2 are presented. The best model according to AICc is shown in bold. Number of model parameters (*K*) and the Akaike weight of evidence (W_i) in support of model *i* are also given. Models were fitted with a normal distribution using the function "lmer" in R.

Model	AICc	ΔAICc	*K*	w_i
Species abundance curve				
Before*TR	**1169.8**	**0.00**	**6**	**0.435**
Before	1170.6	0.82	4	0.289
Before + Treatment	1170.7	0.90	5	0.277
Species abundance curve with order				
Before*TR + Before*Order	**1165.1**	**0.00**	**8**	**0.240**
Before*Order + TR	1165.2	0.10	7	0.229
Before*Order	1166.3	1.21	6	0.131
Before*TR*Order	1166.8	1.70	10	0.103
Before*TR + Before*Order + Order*Treatment	1166.9	1.78	9	0.099

4. DISCUSSION

In this thesis, I investigated the effects of hydropower on the biodiversity of aquatic insects, asking the question of whether hydropower have long-term effects on the total abundance, species richness, and species abundance curves for the orders Mayflies (Ephemeroptera), Stoneflies (Plecoptera) and Caddisflies (Trichoptera). There were indications of an increase in total abundance from before to after the time of hydropower impact, in both control rivers and impact rivers (affected by hydropower development). However, the strength of this temporal trend in abundance varied among treatments and taxa. Species richness did also exhibit an overall increase from before to after impact, yet there were clear differences between taxon orders and treatments where Plecoptera in control rivers displayed a decrease in species richness. In addition, there was a general tendency for community evenness to increase in both impact and control rivers in the period from before to after impact.

There were some methodological issues having the potential to affect the observed trends in biodiversity. Firstly, improvements in taxonomic identification keys over time and differences between personnel doing the identification could affect temporal trends in species richness. I controlled for such issues by ensuring that the same protocol was used throughout the data material and checked that if certain species appeared after time of impact, they had in fact been identified before time of impact in the data material. Trichoptera did not fulfil these criteria and was therefore excluded from some of the analyses. Secondly, sampling effort was lower in control rivers compared to impact rivers. As species richness is expected to increase with sampling effort until a threshold is reached (Magurran, 2004), there is a risk of erroneously reporting species missing if sampling effort is too low. Low sampling effort also results in a reduced ability to properly account for seasonal variation and variation along the river continuum. Some species may have seemingly disappeared if several seasons were not covered. This includes species belonging to the Plecopteran genus *Capnia* which typically swarm early in spring. Both analyses of total abundance and species richness accounted for seasonality, but species abundance curves did not. Unequal sampling effort along the river continuum (i.e., not all stations were sampled both before and after impact) is not expected to be an issue for detecting most species, as the majority is found throughout the river.

The findings of this study contrast global observations of contemporary insect declines (Sánchez-Bayo & Wyckhuys, 2019). However, a recent meta-analysis did suggest that trends in abundances may differ between terrestrial and aquatic insects (van Klink et al., 2020). The paper reported a decline in terrestrial insect populations, but an increase in aquatic. A recovery from past degradations due to clean water efforts was presented as one possible explanation for the positive trend of aquatic insect abundance. This could also be the case in Norway, as large efforts were made from around 1990 to improve water quality (Jakobsson & Pedersen, 2020). A limitation to many studies investigating biodiversity impacts of different types of human interventions is the lack of data describing undisturbed, pristine conditions. This also applies for my study. A range of different anthropogenic disturbances have affected all of my study rivers also before sampling for this study began, with the exception of river Forra which has faced little disturbance. It is therefore difficult to verify if the observed increase in species richness and abundance is due to a recovery from earlier degradation – as there is no baseline to compare it to. There are also reported weaknesses to several global meta-analyses of insect trends. This includes that the majority of data tend to originate from certain regions like North America and Europe, a lack of data for regions and areas with extensive land-use pressures, and inconsistent methodology (Saunders et al., 2020; Welti et al., 2021). A more nuanced view of spatiotemporal patterns in insect trends have therefore been argued for.

Contemporary time-series of community dynamics also face general challenges with lack of knowledge on underlying long-term baselines in species abundances and species turnover rates (Cardinale et al., 2018). In the context of Norway, watercourses may be experiencing successional changes following deglaciation (Brabrand, 2006). These changes are however expected to be happening over a longer time scale, and thus assumed to be of little relevance for explaining the decadal changes of this study.

An increase in species richness in temperate regions is an expected consequence of global warming, mainly because previous thermal restrictions will disappear (Pecl et al., 2017). This is supported by a predictive study looking at the consequences of climate change on invertebrate assemblages in Finland (Mustonen et al., 2018). The study indicated that thermal and hydrologic responses to climate change would lead to increased species richness due to northwards shifts in distributions. Furthermore, a long-term study of a river residing in a nature reserve in Germany suggested that the observed increase in species richness and community evenness could be linked to rising temperatures (Baranov et al., 2020). Rising temperatures then allowed several new species to establish and existing rare species to increase population sizes. In Norway, annual temperatures increased by approximately 1°C from 1900 to 2014 (Nilsen et al., 2022), which can have important ecological implications (Parmesan et al., 2022).

Large parts of these temperature changes have occurred during the time period of the current study. It is therefore likely that the general increase in species richness over time observed in this study can be a result of improved water quality, climate warming, or a combination of the two.

Out of the three biodiversity metrics used, total abundance showed the strongest support for treatment- and order-specific effects on the response over time. The indication of a negative trend in abundance of Plecoptera in control rivers is in correspondence with findings in global meta-analyses (35 % decline, Sánchez-Bayo & Wyckhuys, 2019). Plecoptera is particularly sensitive to habitat alterations and pollution (Fochetti, 2020), and diversity of European Plecoptera is predicted to decrease following climate change (de Figueroa et al., 2010). In contrast, the abundance of Plecoptera increased in impact rivers. A more detailed analysis of the spatiotemporal overlap between sampling localities and possible impact factors may be necessary to identify potential sources of impact which could explain the difference in trends of Plecoptera abundance among treatment and control rivers.

There could be specific habitat alterations caused by hydropower development explaining the observed responses in abundance. The observed increase in abundance of Plecoptera and Trichoptera in rivers impacted by hydropower corresponds with previous findings from river Stjørdalselva, where an increase in abundance the first years of regulation was observed (Arnekleiv et al., 2018). The authors suggested that the increase could be due to an evening out of water flow, increased sedimentation, and an increased supply of nutrients. It follows that species known to favour slow-flowing lentic conditions, like *Leptophlebia marginata* and *Siphlonurus lacustris* increased in abundance in several impact rivers included in my study. For the order Ephemeroptera in general, there was indicative evidence for a decrease in abundance in impact rivers. A possible contribution to the negative trend was the observed reduction in abundance of the dominant species *Baetis rhodani* which favours fast-flowing waters. Other studies have also reported declines in abundances of *B. rhodani* following hydropower regulation (Koksvik & Reinertsen, 2008; Ugedal et al., 2014). Ugedal et al. (2014) proposed that one explanation for the reduced abundance could be related to their affinity to the uppermost substrate layer, making them vulnerable to reductions in water levels following hydropower regulation.

There was a general increase in species richness from before to after time of hydropower impact with some evidence for a difference between treatments. The differences between treatments became more apparent when taxon order was included in the model. In addition to environmental factors like temperature increase and water quality efforts affecting impact and control rivers alike, it is likely that habitat changes following hydropower development could have increased species richness. River systems are generally extreme, and a reduction in flood-peaks which commonly follows hydropower regulation could bring river conditions closer to an intermediate level of disturbance. According to the intermediate disturbance hypothesis (Connell, 1978; Townsend et al., 1997), this could allow for a higher number of species to coexist. The observed increase in abundance of species favouring more lentic conditions also supports that flow regulation may contribute to the diversity trends of benthic invertebrates in impact rivers. It is interesting to note that there are indications of opposing trends in species richness among taxa and treatments pointing against any overall effect of changes in taxonomic identification. Furthermore, the indicated decrease in species richness as well as total abundance of Plecoptera in control rivers points towards an underlying factor negatively affecting Plecoptera in these rivers. To see if there is further support for this negative trend, more sampling from control rivers should be conducted to increase the statistical power.

Species abundance curves combined information on abundance and species richness, and my result where the slope was shallower than the expected 1:1 relationship, indicates that community evenness generally has increased from before to after the time of hydropower impact. This evening out of species abundance in communities from rivers impacted by hydropower is supported by my findings of reduced abundance of the dominant species *Baetis rhodani*, and the increase in abundance of species favoring slow-flowing conditions that were rare before regulation. An increase in evenness irrespective of treatment could also be anticipated by the before mentioned environmental factors that are expected to positively affect species richness and abundance of rare species. The limited amount of data from control rivers should be taken into consideration when interpreting the difference in evenness between control and impact rivers. Erroneous losses of fairly common species could make it appear like communities in control rivers have become more even than they really have. Further, because of the way data was structured, the effect of seasonality could not be controlled for in the model – thereby accentuating the issue of low sampling effort. Based on the introduced uncertainty, no inferences about differences in evenness between control and impact rivers were made. Still, the trends in total abundance and species richness for control rivers are in accordance with an increase in evenness. The ecosystem consequences of increased evenness is linked to the traits of species that decrease or increase, and particularly dominant species (sensu Hillebrand et al., 2008). Species traits are not investigated in the current study but could be of interest for future studies aiming to predict the ecological consequences of human disturbances.

The general trend of increased evenness does match expectations from analyses of trends in total abundance and species richness. This supports the impression that there were no drastic effects of hydropower on the evenness of the insect communities in the rivers studied. Both density independent and density dependent factors are likely at play in the studied rivers, but disentangling their relative contributions to the observed biodiversity trends is not possible with the data at hand.

5. CONCLUSION

Monitoring trends in biodiversity with metrics like abundance, species richness and community composition can be a tool for detecting changes in ecosystem structure resulting from anthropogenic impacts such as hydropower development. This study did not find evidence for strong effects of hydropower activities on the diversity of benthic invertebrates from the orders Ephemeroptera, Plecoptera, and Trichoptera. One possible explanation is that there are none, and that the few observed temporal changes in diversity were caused by other environmental effects such as recovery from past degradations or temperature increases due to climate change. There may also be methodological issues with the study, which is a known challenge with long-term monitoring efforts (Welti et al., 2021). For instance, higher statistical power in the form of increased sampling effort could be necessary to register the effects because of large variation in the data and environmental conditions within and among rivers.

There are most likely effects of hydropower on the benthic invertebrates in the study rivers, even if this study did not find a clear trend. The effects are likely local, as other studies have found (Jones, 2013; Kjaerstad et al., 2018). Both my findings and other studies (Bruno et al., 2010; Kakouei et al., 2017) support that reactions to habitat changes can be order- and species-specific. This further points to the need to not only understand the response of different species of benthic invertebrates, but also take into account the effects on other taxa, for instance fish, which face different challenges related to river regulations and hydropower (see for instance Algera et al., 2020; Ugedal et al., 2008).

One of the strengths of this study is that it assesses biodiversity impact of hydropower on multiple rivers. Still, including more rivers that experience large hydropower-induced changes to for instance flow and temperature regime could also make potential effects more apparent. Future studies working with long-term datasets of benthic invertebrates should consider performing a control of the taxonomic identification of a subset of the older material if this is available to ensure consistency in taxonomic resolution. Creating a standardised taxonomic list has also been suggested to ensure consistency (Petrin et al., 2016). In addition to using traditional measures of biodiversity like abundance and species richness, including trait-based measures like functional feeding groups could help researchers and natural resource managers better predict ecological consequences of anthropogenic impacts.

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APPENDIX A – Locality maps

All rivers included in the analysis are presented in the maps below. They show both included and excluded sampling localities for each river, hydropower infrastructure, and other relevant details. Information on levels of impact came from vann-nett, a web-portal run by the Norwegian Water Resources and Energy Directorate[1]. Maps were drawn using ESRI ArciGIS Pro Desktop v2.8.

Impact rivers

River Dalåa

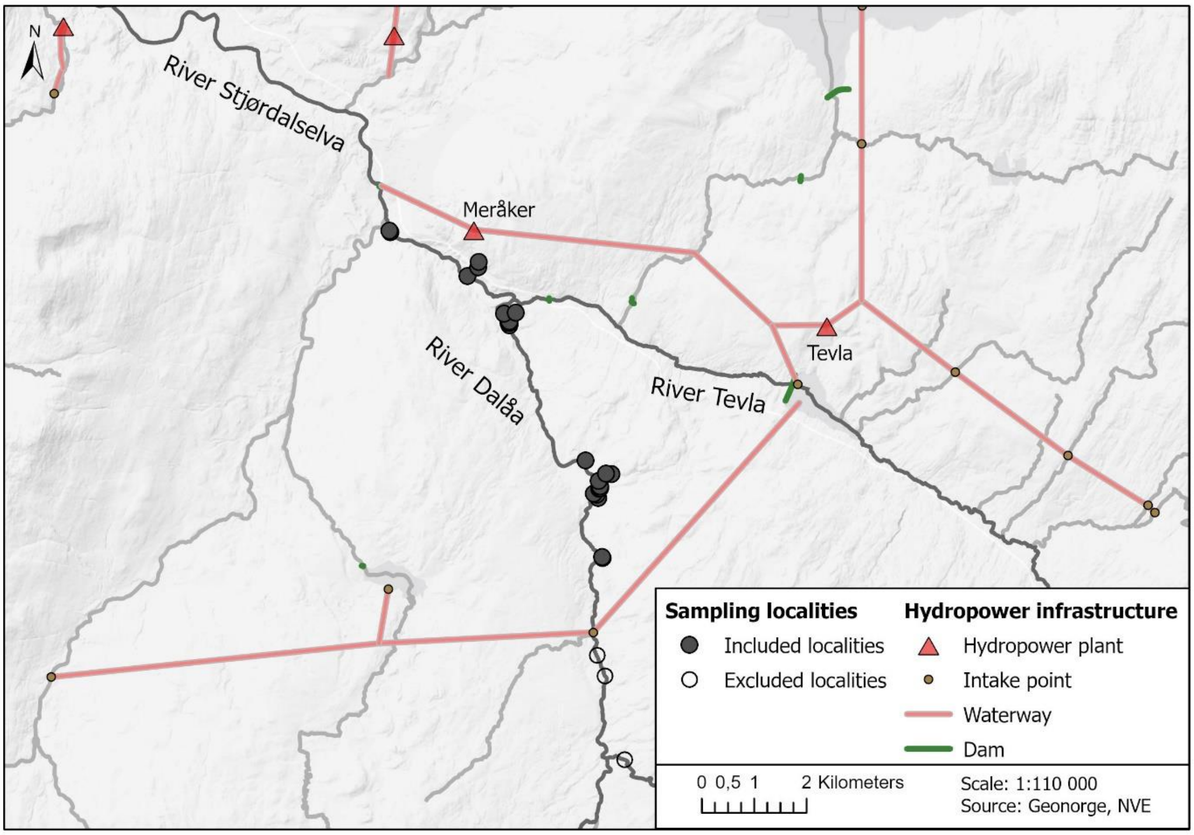

Figure 9. Map showing river Dalåa which meets river Tevla and then changes name to river Stjørdalselva at Meråker. All points upstream from the village of Meråker are defined as belonging to Dalåa. The three localities in Dalåa upstream for the intake point were assumed to be unaffected by hydropower activities, and therefore excluded from analyses. These contained a total of 51 sampling events.

In 1994, both Meråker and Tevla hydropower plants were put into operation. Water was then directed from Dalåa and nearby rivers to lake Grønbergdammen. The majority of Dalåa is categorized as experiencing medium impact from hydropower.

[1] NVE (2021, April 16). Temakart - Vannkraft. Påvirkninger med størst påvirkningsgrad [Vann-nett: Thematic map -Hydropower. Impacts with the greatest degree of influence]. Norwegian Water Resources and Energy Directorate. https://vann-nett.no/

River Nea

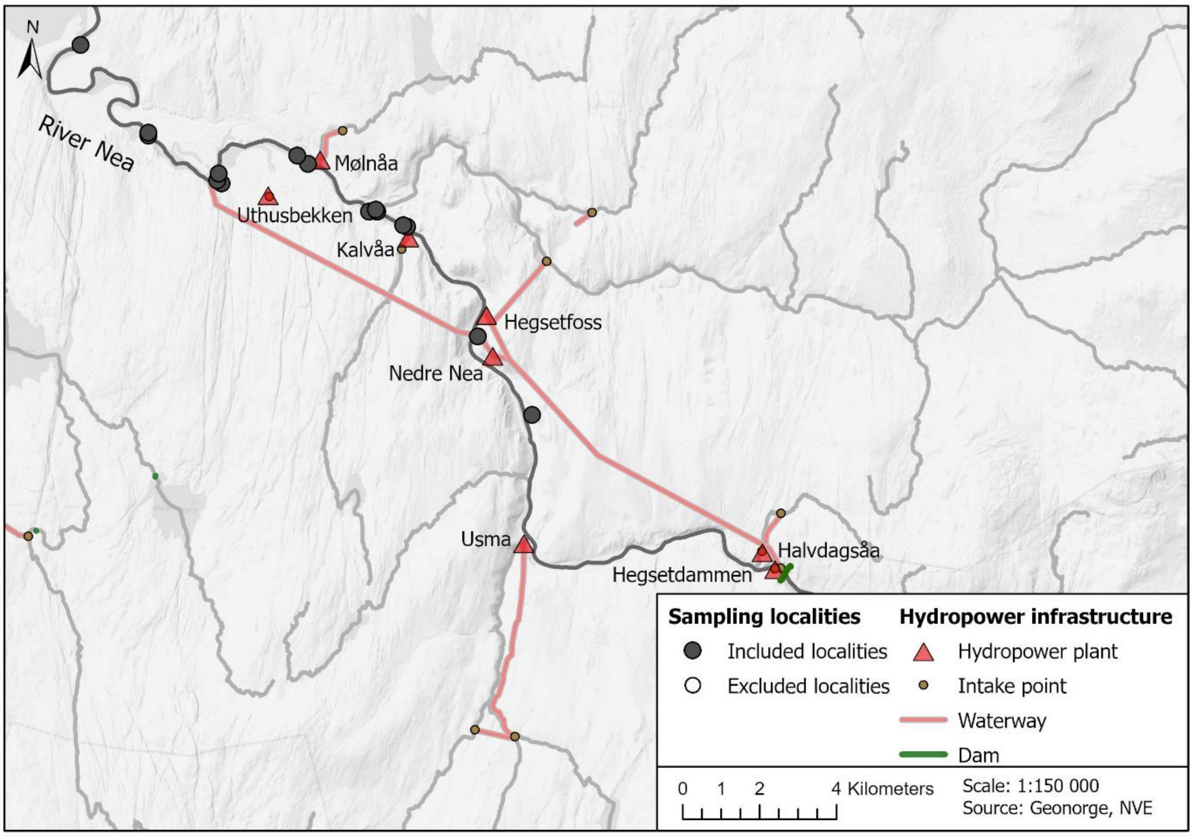

Figure 10. Map showing river Nea, from the dam at lake Heggesetsjøen until it reaches lake Selbusjøen. The major hydropower plants are Nedre Nea and Hegsetfoss. The rest are smaller plants, which mostly affect tributary rivers and streams. Hegsetfoss hydropower plant was put into operation in 1962, which led water to be transported through tunnels from lake Hegsetsjøen. A number of weirs have been constructed in Nea. First to mitigate the reduced waterlevels caused by the operation of Hegsetfoss hydropower plant, where water was transported from Hegsetdammen and to the plant through tunnels. More weirs were added downstream from Hegsetfoss due to the building of Nedre Nea hydropower plant, put into operation in 1994, to mitigate the building of additional tunnels[23]. The building of Nedre Nea reduced the role of Hegsetfoss hydropower plant, which today mainly is in use during periods of flooding. The majority of Nea is classified as being impacted to a large degree, with the uppermost section categorized as facing medium impact.

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[2] Arnekleiv, J. V., Hellesnes, I., Jensen, A., & Lindstrøm, E. A. (1991). *Vannkvalitet, begroing og bunndyr i Nea 1988 og 1989. Del I. Forholdene før regulering, uten Nedre Nea kraftverk [Water quality, vegetation and benthic fauna in Nea 1988 and 1989. Part I. Conditions before regulation, without Nedre Nea power plant] (LFI 83).* (Report 2 1991). NTNU University Museum.
[3] Arnekleiv, J. V., Hellesnes, I., Lindstrøm, E. A., & Bongard, T. (1997). *Vannkvalitet, begroing og bunndyr i Nea 1993 1995. Del II. Forholdene etter regulering [Water quality, vegetation and benthic fauna in Nea 1993 1995. Part II. Conditions after regulation]. (LFI 109)* (Report 19 1997). NTNU University Museum.

River Skauga

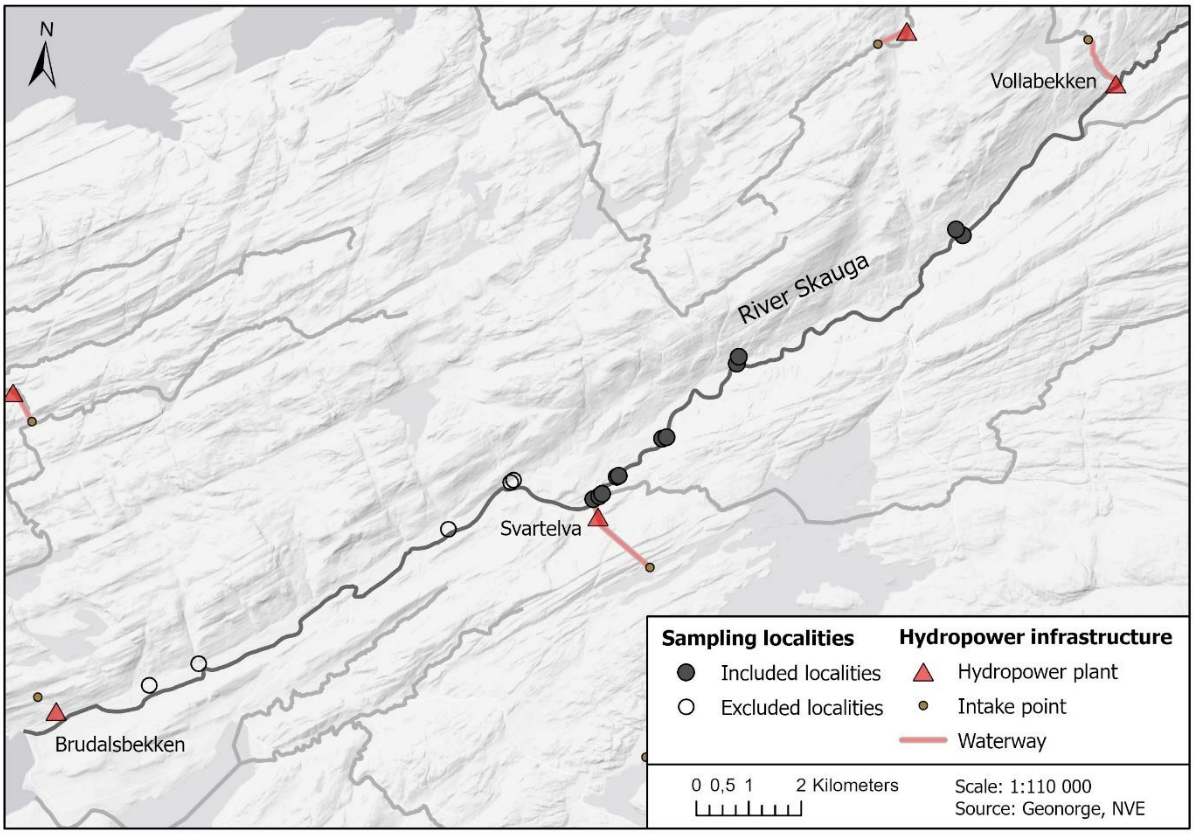

Figure 11. Map showing river Skauga from Vollabekken hydropower plant and down to the Trondheimsfjord.

All localities downstream of Svartelva hydropower plant were excluded, as no sampling were done before the plant was put into operation in 1959. The impact of Vollabekken hydropower plant, initiation year 2012, on the middle section of Skauga was therefore investigated. The entire river section from Vollabekken and down to the fjord is categorized as being impacted by hydropower to a large degree.

River Stjørdalselva

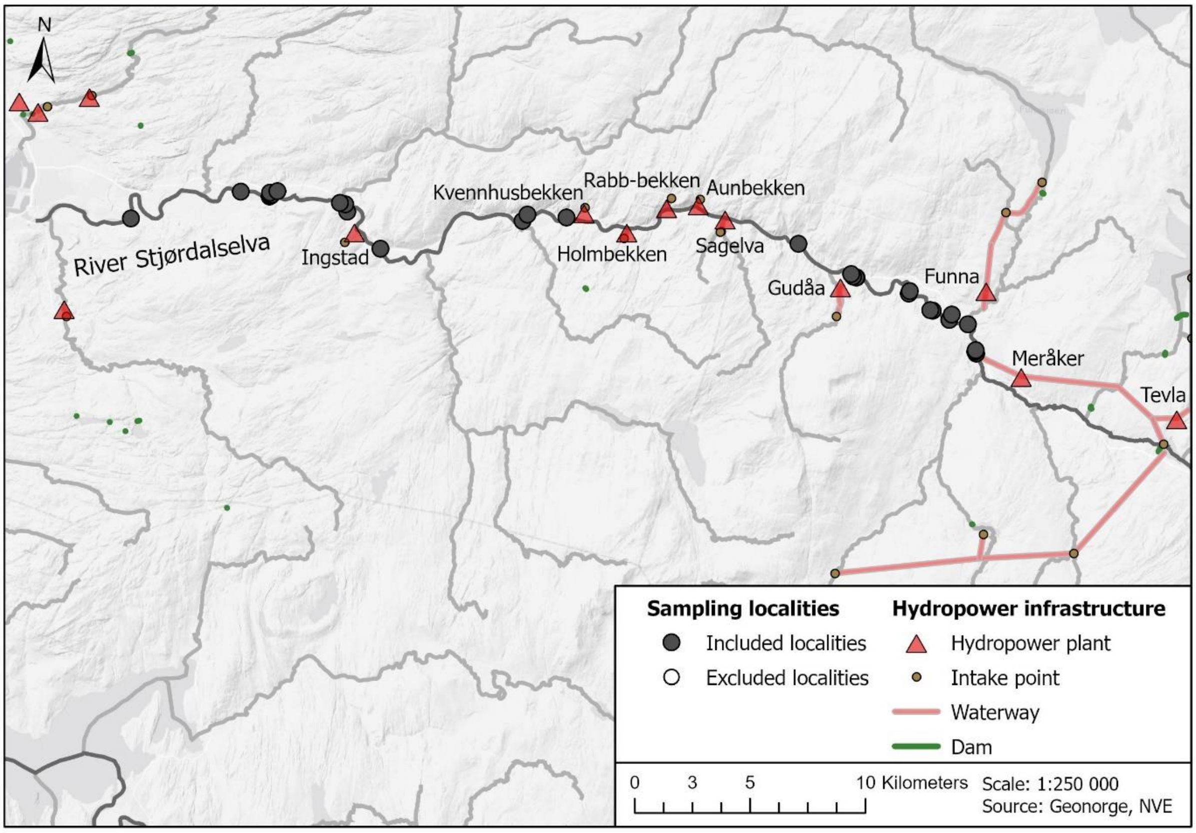

Figure 12. Map showing river Stjørdalselva, which starts at the village Meråker and reaches the Trondheimsfjord at Stjørdal.

Meråker and Tevla hydropower plants, both put into operation in 1994, have the largest effect on Stjørdalselva. Several small plants are found in rivers and streams leading to the main river. The river section below Meråker until Stjørdalselva meets river Sona categorized as facing medium impact. No information on hydropower impact is available for the last river stretch, but it is still assumed to be affected in the analyses.

Control rivers

River Forra

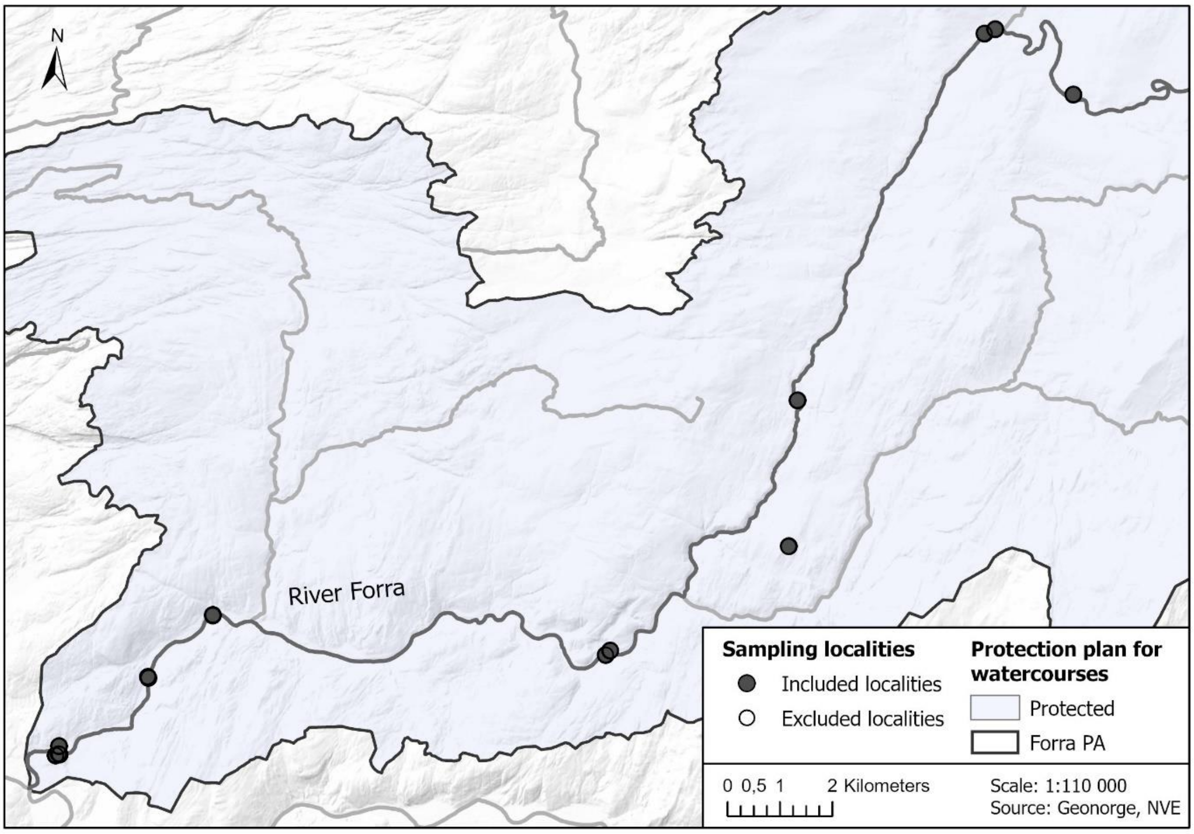

Figure 13. Map of river Forra, starting some kilometers downstream lake Feren and ending where Forra reaches Stjørdalselva.

All sampling localities in the dataset were included. Forra lies within Forra protected area (124/1, protected 1986 with protection plan III), which was put into place because of the watershed's pristine conditions and is a recommended reference river[4].

[4] NVE. (2021, June 15). *124/1 Forra*. Protection Plan for Watercourses. Norwegian Water Resource and Energy Directorate. https://www.nve.no/vann-og-vassdrag/vassdragsforvaltning/verneplan-for-vassdrag/trondelag/124-1-forra/

River Gaula

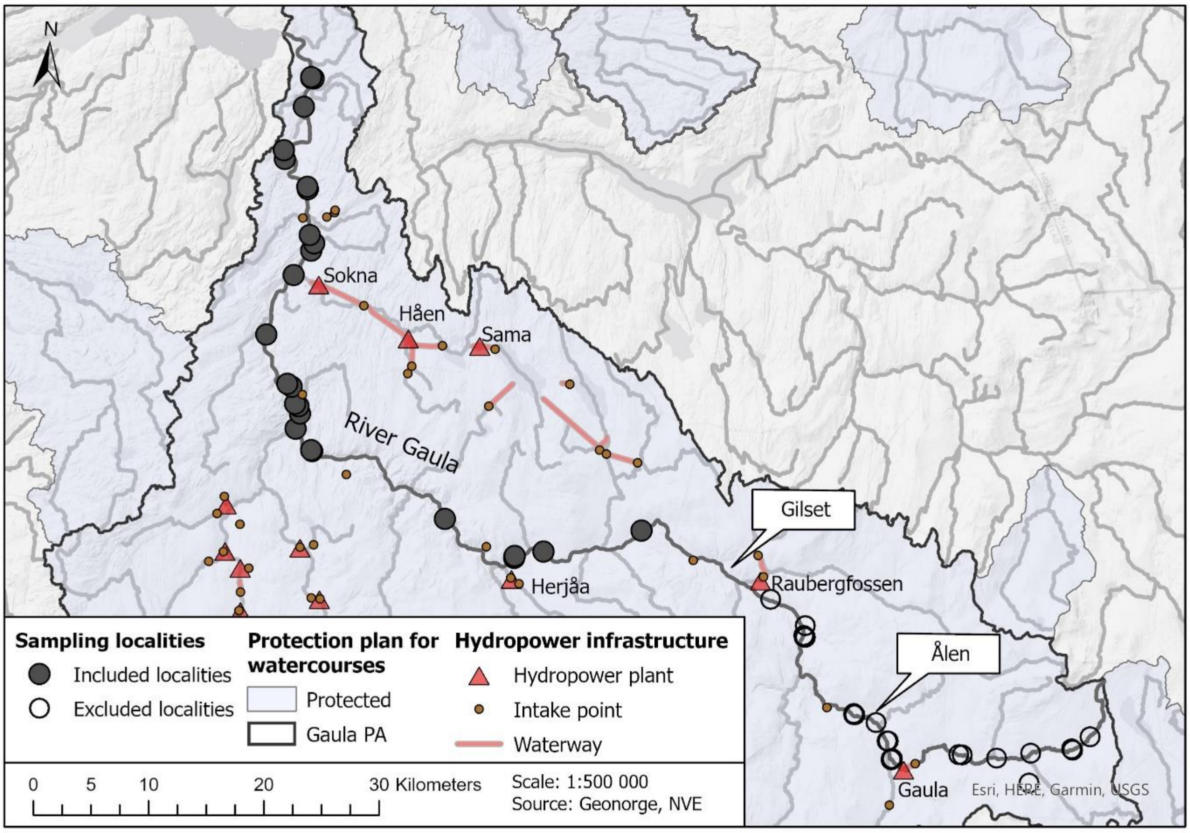

Figure 14. Map over river Gaula, from its beginning close to the county Innlandet to it reaches the Trondheimsfjord at Gaulosen.

The upper parts of river Gaula are affected by heavy metal runoff, and localities in this area are therefore excluded. A report from 1981 detected significant effects of heavy metals on organisms until Ålen but found the benthic fauna to normalize downstream[5]. Therefore, all points upstream from Gilset were excluded. The river is prone to rapid flooding as there are few lakes to dampen the effects. Gaula and its catchment were included in national protection plans in 1986 due to its large size and central placement[6]. It is considered to hold great cultural and recreational values. There are several hydropower plants within Gaula protected area, but none are expected to affect the main river Gaula which is in focus. (NVE, 2021b)

[5] Koksvik, J. I., & Nøst, T. (1981). *Gaulavassdraget i Sør-Trøndelag og Hedmark fylker, ferskvannsbiologiske undersøkelser i forbindelse med midlertidig vern* (No. 24; Rapport Zool.). NTNU University Museum.
[6] NVE. (2021, June 15). *122/1 Gaula*. Protection Plan for Watercourses. Norwegian Water Resources and Energy Directorate. https://www.nve.no/vann-og-vassdrag/vassdragsforvaltning/verneplan-for-vassdrag/trondelag/122-1-gaula/

River Homla

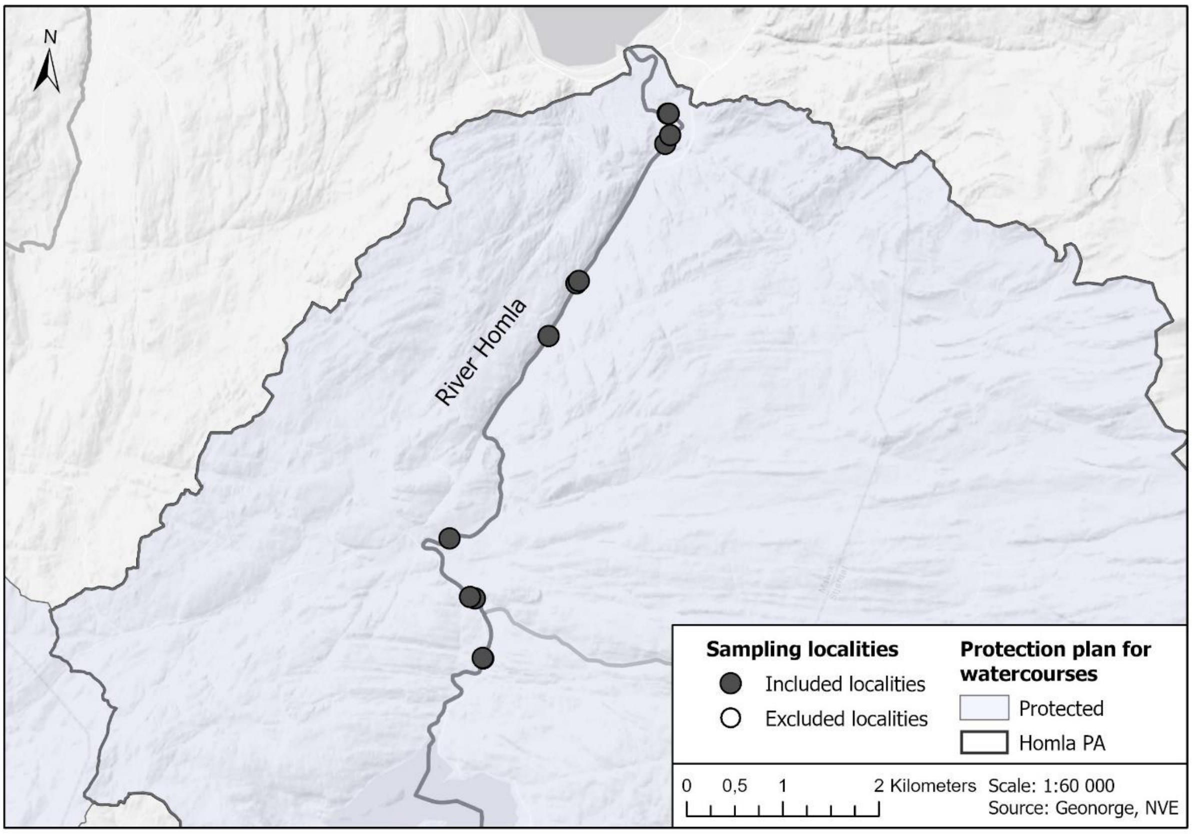

Figure 15. Map of river Homla, which runs from lake Foldsjøen and reaches the fjord in the town of Hommelvik.

All localities along Homla were included in the analysis. Homla protected area is a small lowland protected area that was protected in 2005[7]. It is considered important for recreation, and provides habitat for both Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*)[8].

[7] NVE. (2021, June 15). *123/2 Homla*. Protection Plan for Watercourses. Norwegian Water Resources and Energy Directorate. https://www.nve.no/vann-og-vassdrag/vassdragsforvaltning/verneplan-for-vassdrag/trondelag/123-2-homla/

[8] Arnekleiv, J. V., & Nøst, T. (1987). *Fiskeribiologiske undersøkelser i Homlavassdraget, Sør-Trøndelag [Fishery biological investigations in Homlavassdraget, Sør-Trøndelag]*. (No. 68; Rapport Zool.). NTNU University Museum.

Rivers Nordelva-Holvasselva

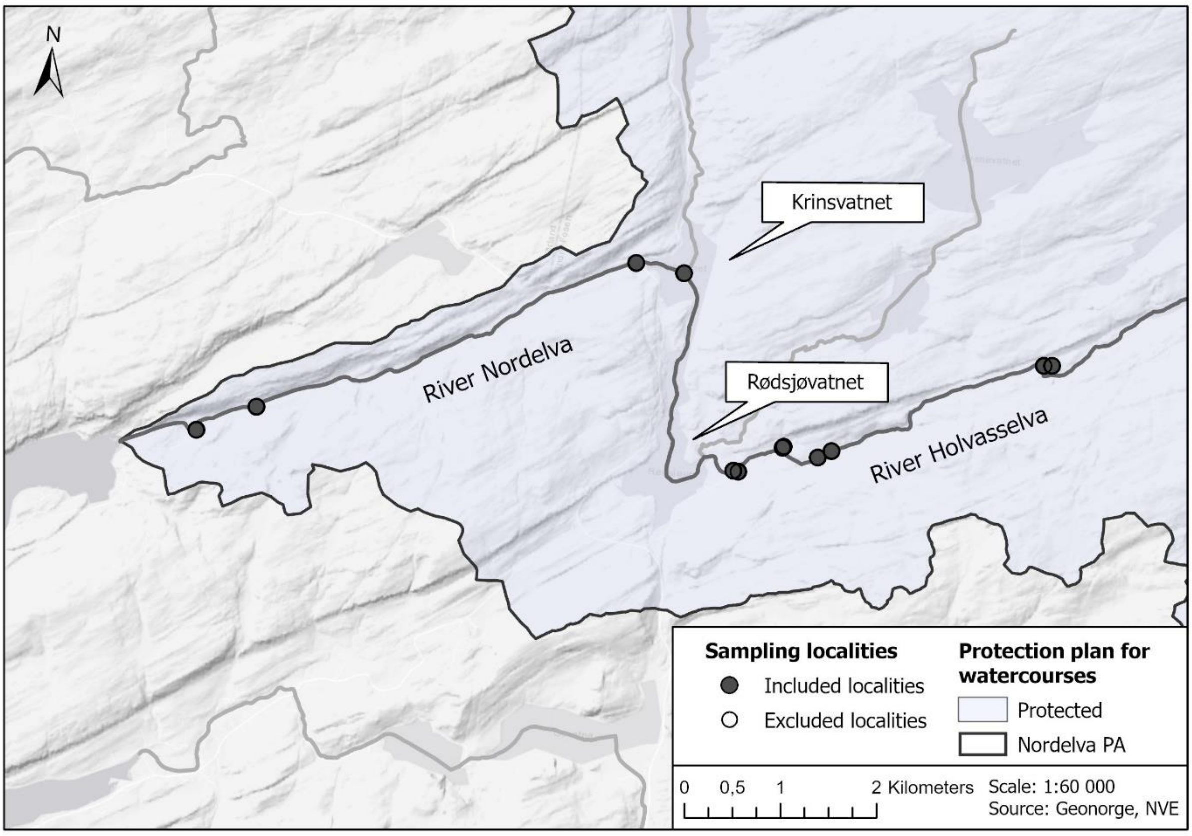

Figure 16. Map of Nordelva protected area, showing river Holvasselva which originates from lake Holvatnet and changes name to river Nordelva after running through the lakes Rødsjøvatnet and Krinsvatnet before it reaches the fjord.

All localities were included. Nordelva protected area is found in the district called Fosen and was added to the national watercourse protection plan in 2006. The area has a characteristic nature type, and large parts are INON-registered. Atlantic salmon (*Salmo salar*) and anadromous brown trout (*Salmo trutta*) are found in the lower parts of Nordelva, and the area is important for recreation and has not faced many large human disturbances[9].

[9] NVE. (2021, June 15). *133/1 Nordelva*. Protection Plan for Watercourses. Norwegian Water Resources and Energy Directorate. https://www.nve.no/vann-og-vassdrag/vassdragsforvaltning/verneplan-for-vassdrag/trondelag/133-1-nordelva/

Rivers Verdalselva-Helgåa

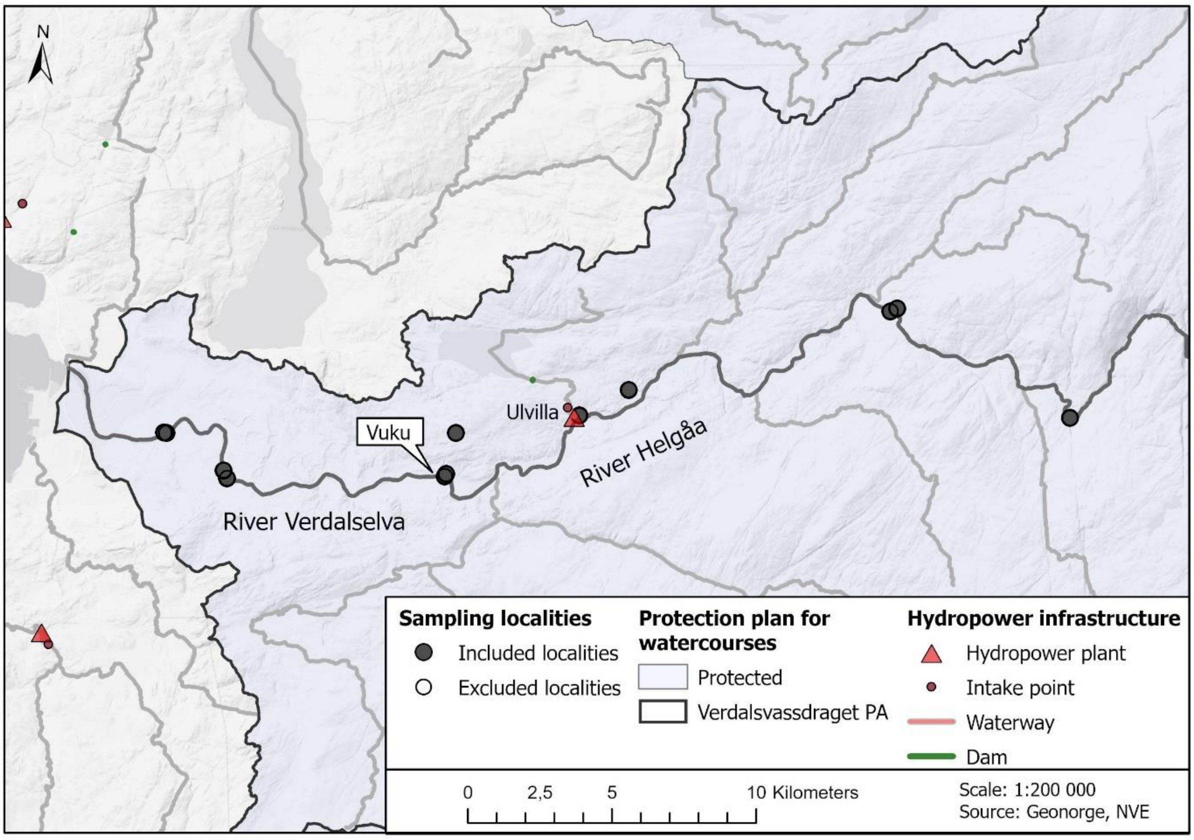

Figure 17. Map of Verdalsvassdraget protected area, showing river Helgåa from the waterfall Fagerlifossen until it changes name to river Verdalselva at Vuku. Verdalselva reaches the Trondheimsfjord at Verdal.

Ulvilla hydropower plant which lies within the protected area was put into operation in 1914 and affects a tributary river to Helgåa but is assumed not to have a significant affect the main river. The nearby area Skjækra was first included in national protection plans in 1986, followed by the additional protection of the watercourse Verdalsvassdraget in 2005[10].

[10] NVE. (2021, June 15). *127/1 Verdalsvassdraget*. Protection Plan for Watercourses. Norwegian Water Resources and Energy Directorate. https://www.nve.no/vann-og-vassdrag/vassdragsforvaltning/verneplan-for-vassdrag/trondelag/127-1-verdalsvassdraget/

APPENDIX B – Taxon list

Table 10. Taxon list for the final dataset, including taxa belonging to the orders Ephemeroptera, Plecoptera, and Trichoptera. Individual counts are given for the 4 BACI groups: control before (CB), control after (CA), impact before (IB) and impact after (IA).

Scientific name	*CB*	*CA*	*IB*	*IA*
Acentrella lapponica Bengtsson, 1912	13	0	1	8
Agapetus Curtis, 1834	0	5	0	726
Agapetus ochripes Curtis, 1834	0	3	0	152
Ameletus inopinatus Eaton, 1887	1186	715	531	6834
Amphinemura borealis (Morton, 1894)	1562	4065	2632	58278
Amphinemura Ris, 1902	715	443	128	8382
Amphinemura standfussi (Ris, 1902)	2	82	8	372
Amphinemura sulcicollis (Stephens, 1836)	282	97	162	387
Annitella obscurata (McLachlan, 1876)	0	0	0	11
Apatania dalecarlica (Forrslund, 1934)	0	0	0	2
Apatania hispida (Forsslund, 1930)	1	0	0	0
Apatania Kolenati, 1848	11	27	7	437
Apatania stigmatella (Zetterstedt, 1840)	31	36	7	68
Apatania wallengreni McLachlan, 1871	0	0	0	21
Apatania zonella (Zetterstedt, 1840)	0	0	10	3
Arctopsyche ladogensis (Kolenati, 1859)	4	74	32	472
Arthroplea congener Bengtsson, 1908	0	0	0	1
Athripsodes Billberg, 1820	0	12	8	221
Athripsodes cinereus (Curtis, 1834)	0	0	0	1
Athripsodes commutatus (Rostock, 1874)	0	29	0	9
Baetidae	0	7	2	4
Baetis digitatus Bengtsson, 1912	10	19	5	4
Baetis fuscatus (Linnaeus, 1761)	248	4844	4449	2302

Baetis Leach, 1815	196	99	1409	1598
Baetis macani Kimmins, 1957	0	0	1	4
Baetis muticus (Linnaeus, 1758)	1219	1566	228	7817
Baetis niger (Linnaeus, 1761)	89	51	21	366
Baetis rhodani (Pictet, 1843)	6370	13075	30046	105604
Baetis subalpinus Bengtsson, 1917	2	6	657	150
Baetis vernus Curtis, 1834	0	0	2	0
Brachycentridae	0	0	0	1
Brachyptera risi (Morton, 1896)	42	232	172	1346
Caenis horaria (Linnaeus, 1758)	0	0	0	4
Caenis luctuosa (Burmeister, 1839)	0	0	1	0
Caenis Stephens, 1835	0	1	0	0
Capnia atra Morton, 1896	842	257	26	198
Capnia bifrons (Newman, 1838)	123	1	0	0
Capnia Pictet, 1841	2083	4400	1068	20094
Capnia pygmaea (Zetterstedt, 1840)	0	1974	79	67
Capniidae	71	1	0	1349
Capnopsis schilleri (Rostock, 1892)	10	4	3	56
Centroptilum Eaton, 1869	0	1	0	0
Centroptilum luteolum Müller, 1776	30	42	53	5795
Ceraclea annulicornis (Stephens, 1836)	0	0	0	1
Ceraclea nigronervosa (Retzius, 1783)	1	0	5	0
Ceraclea Stephens, 1829	0	0	0	1
Ceratopsyche Ross & Unzicker, 1977	0	1	0	29
Chaetopteryx Stephens, 1829	17	0	0	54
Chloroperlidae	0	4	0	91
Dinocras cephalotes (Curtis, 1827)	0	8	1	10
Diura bicaudata (Linnaeus, 1758)	0	0	0	4

Diura Billberg, 1820	1	2	21	0
Diura nanseni (Kempny, 1900)	722	1007	2147	4956
Ecclisopteryx dalecarlica Kolenati, 1848	0	1	0	10
Ephemera danica Müller, 1764	0	0	1	5
Ephemera Linnaeus, 1758	0	1	0	6
Ephemera vulgata Linnaeus, 1758	0	3	0	0
Ephemerella aurivillii (Bengtsson, 1909)	936	1000	4775	18234
Ephemerella mucronata (Bengtsson, 1909)	575	1128	1247	15225
Ephemerella Walsh, 1863	6	136	514	3048
Ephemeroptera	1224	0	10753	82
Glossosoma Curtis, 1834	12	55	0	185
Glossosoma intermedium (Klapalek, 1892)	2	0	0	0
Glossosoma nylanderi McLachlan, 1879	0	0	0	1
Glossosomatidae	1	0	82	3
Halesus digitatus (von Paula Schrank, 1781)	0	0	0	1
Halesus radiatus (Curtis, 1834)	7	1	1	21
Halesus Stephens, 1836	0	0	12	21
Halesus tesselatus (Rambur, 1842)	0	0	0	2
Heptagenia dalecarlica Bengtsson, 1912	1382	2967	1016	15179
Heptagenia fuscogrisea (Retzius, 1783)	2	0	5	2
Heptagenia sulphurea (Müller, 1776)	92	48	49	3579
Heptagenia Walsh, 1863	24	408	339	2373
Holocentropus dubius (Rambur, 1842)	184	0	0	0
Hydropsyche angustipennis (Curtis, 1834)	1	0	0	0
Hydropsyche newae Kolenati, 1858	59	1243	274	3228
Hydropsyche pellucidula (Curtis, 1834)	0	12	0	156
Hydropsyche Pictet, 1834	32	4	0	26
Hydropsyche silfvenii Ulmer, 1906	22	12	0	48

Hydropsyche siltalai Doehler, 1963	36	104	0	20
Hydropsychidae	0	77	0	82
Hydroptila Dalman, 1819	367	199	108	1091
Hydroptilidae	0	0	3	0
Isoperla Banks, 1906	64	102	143	1113
Isoperla difformis (Klapálek, 1909)	1	0	1	0
Isoperla grammatica (Poda, 1761)	83	68	45	137
Isoperla obscura (Zetterstedt, 1840)	51	2	60	146
Ithytrichia lamellaris Eaton, 1873	0	10	0	10
Lepidostoma hirtum (Fabricius, 1775)	2	22	3	180
Leptoceridae	0	2	0	14
Leptophlebia marginata (Linnaeus, 1767)	13	3	8	412
Leptophlebia vespertina (Linnaeus, 1758)	0	0	2	13
Leptophlebia Westwood, 1840	1	1	0	13
Leptophlebiidae	1	8	23	288
Leuctra digitata Kempny, 1899	31	6	27	25
Leuctra fusca (Linnaeus, 1758)	359	298	990	2535
Leuctra hippopus Kempny, 1899	37	79	17	572
Leuctra nigra (Olivier, 1811)	6	6	4	201
Leuctra Stephens, 1836	31	240	128	4987
Limnephilidae	9	29	59	546
Limnephilus fuscicornis (Rambur, 1842)	0	0	0	6
Limnephilus Leach, 1815	0	0	0	2
Metretopus Eaton, 1901	0	0	0	4
Micrasema setiferum (Pictet, 1834)	0	0	1	22
Mystacides azureus (Linnaeus, 1761)	0	0	0	1
Mystacides Berthold, 1827	0	0	1	2
Nemotaulius punctatolineatus (Retzius, 1783)	1	0	0	0

Nemoura avicularis Morton, 1894	2	3	1	40
Nemoura cinerea (Retzius, 1783)	35	4	27	34
Nemoura Latreille, 1796	27	30	35	287
Nemouridae	1	0	1	2
Nemurella pictetii (Klapálek, 1900)	1	2	0	4
Neureclipsis bimaculata (Linnaeus, 1758)	0	29	0	0
Oxyethira Eaton, 1873	2	7	21	691
Paracinygmula joernensis (Bengtsson, 1909)	54	18	1071	709
Paraleptophlebia Lestage, 1917	0	0	11	23
Paraleptophlebia werneri Ulmer, 1920	0	0	0	1
Parameletus Bengtsson, 1908	0	0	27	1
Parameletus chelifer Bengtsson, 1908	502	123	0	3
Perlodidae	5	13	0	73
Philopotamidae	0	1	0	0
Philopotamus montanus (Donovan, 1813)	3	5	0	11
Phryganeidae	0	0	0	1
Plecoptera	1072	1	979	1
Plectrocnemia conspersa (Curtis, 1834)	3	9	26	88
Polycentropodidae	0	1	185	98
Polycentropus flavomaculatus (Pictet, 1834)	206	56	206	5316
Potamophylax cingulatus (Stephens, 1837)	0	3	3	25
Potamophylax latipennis (Curtis, 1834)	2	0	9	34
Potamophylax Wallengren, 1891	0	2	6	9
Procloeon bifidum (Bengtsson, 1912)	1	0	6	6
Protonemura meyeri (Pictet, 1841)	20	33	20	204
Psychomyia pusilla (Fabricius, 1781)	1	36	7	2
Psychomyiidae	0	3	0	0
Rhyacophila nubila Zetterstedt, 1840	135	379	681	2507

Rhyacophila Pictet, 1834	0	0	0	4
Sericostoma personatum (Kirby & Spence, 1826)	2	26	19	458
Sericostomatidae	0	4	0	1
Serratella ignita (Poda, 1761)	2	0	201	3
Silo pallipes (Fabricius, 1781)	0	0	0	1
Siphlonuridae	0	2	0	1
Siphlonurus aestivalis (Eaton, 1903)	0	0	1	0
Siphlonurus Eaton, 1868	71	11	518	798
Siphlonurus lacustris Eaton, 1870	4	0	4	209
Siphonoperla burmeisteri (Pictet, 1841)	111	84	86	994
Taeniopteryx nebulosa (Linnaeus, 1758)	314	36	96	437
Trichoptera	577	335	537	89
Wormaldia McLachlan, 1865	0	20	1	26
Wormaldia subnigra McLachlan, 1865	2	0	0	9
Xanthoperla apicalis (Newman, 1836)	224	66	2	36

APPENDIX C – Data filtration & selection

This appendix contains a description of the process of filtering raw data and selecting rivers and the localities within for analysis and resampling.

First, occurrence data was downloaded from GBIF (GBIF.org, 2022) through the following url: https://api.gbif.org/v1/occurrence/download/request/0172532-210914110416597.zip. Data belonged to the dataset "Limnic freshwater benthic invertebrates biogeographical mapping/inventory NTNU University Museum" (Daverdin, 2022). Occurrences sampled with kick-sampling were selected and stored as a spatial file where all occurrences were grouped by "locality" "decimalLatitude" and "decimalLongitude". The spatial file was transformed to projected coordinates (WGS84 / UTM zone 33N).

1. Locating rivers for resampling and analysis

Rivers which satisfied the following minimum criteria were located:

- Impact rivers: Outside of protected areas and affected by hydropower activities
- Control rivers: Within protected areas not affected by hydropower activities
- 5 sampling localities in each river
- 2 datapoints in time before impact and 2 after impact
    - Rivers with 1 year of data after impact were included only if they were to be resampled in 2021.
    - Time of impact: For impacted rivers, this is set to the initiation year for the hydropower plant. For control rivers, a common year of impact was set to 1994.
- No major known impacts from other sources, for instance heavy metal runoff or rotenone treatment

This led to a selection of 9 rivers: The four impact rivers Nea, Stjørdalselva, Dalåa and Skauga, and the 5 control rivers Gaula, Homla, Nordelva-Holvasselva, Forra and Verdalselva-Helgåa.

2. Locating suitable localities for analysis & removing unwanted data

- Removing localities with large coordinate errors (not possible to find correct placement)
- Removing occurrences from January, February, March and December. There were very few sampling events from these months.

3. Locating suitable localities for resampling

First, rivers were selected. The main criteria for selecting rivers for resampling were as follows:

- Resampling allows the river to be included in analyses, as the criteria of minimum 2 datapoints in time After Impact are fulfilled
- Resampling adds datapoints After Impact to a river considered especially important/interesting, that already fulfils minimum criteria.

Second, locations within each river were selected. Selection of the five sites within a river that should be re-sampled was done in accordance with the following criteria:

- Number of existing datapoints: more data = higher prioritization
- Accessibility: distance and time needed to reach each point, difficulty of moving through terrain/vegetation
- Safety: placement in relation to waterfalls, depth and strength of waterflow. These decisions were largely made in the field.
- Distribution along river: if possible, sites were spread out along the river to cover more of the intraspecific variation

4. Categorizing localities by treatment: Impact or Control

All nine rivers were visually inspected to identify possible problems, and to determine which localities were affected by hydropower activities (downstream from plants) or not (upstream from plants or in protected rivers). There were only four points within unprotected rivers that were unaffected by hydropower activities, and no affected points within protected areas. Therefore, the four points were removed to simplify the analysis.

A factor called treatment (TR) with the levels "Impact" and "Control" was created. Impact represented all localities affected by hydropower outside of protected areas, and Control represented all localities within protected areas.

APPENDIX D – University Museum datasets

Table 8. Overview of the datasets from NTNU History Museum which are included in the analyzed material, and the number of sampling events in each dataset. For Øvre Gaulva overvåking, only localities below Gilset were included.

Dataset name	*Number of sampling events*
Stjørdalsvassdraget overvåking	790
Stjørdalselva transekt	420
Dalåa, Tevla, Torsbjørka	410
Nea, før og etter regulering	139
Gaula grusgraving	103
Gaula flomprosjekt	77
Skauga/Skaua	70
Stjørdalselva, Dalåa, Forra Ungfisk-Driv	57
10-Års vernedevassdrag	52
Homla	24
Færen med Forra	18
Stjørdalsvassdraget forundersøkelse	17
Øvre Gaula overvåking	16
Nordelva-vassdraget og Osavatna	15
Gaula E6	9
Stjørdalselva renseanlegg	9
Rotla før og etter kraftutbygging	2

APPENDIX E – Resampled localities

Table 9. Overview of sites that were re-sampled during June 2021. Each river was resampled at five localities, with the resampling method previously used. Station-position 1 is the uppermost sampling locality and 5 is the lowest one. The temporary number was a number given for internal reference to each point in the dataset which was used during fieldwork. The station number was the final number used to identify the station for Museum collections, which reflected the number of the existing station. If several station names existed for the same locality, the most common or most logical was chosen.

Watercourse name	*locality*	*Station-position*	*Temp-number*	*Station number (label)*	*Date*	*Treatment*	*Old sampling method(s)*	*New sampling method(s)*
Nidelv-vassdraget	Nea	1	890	16	07.06.21	Impact	Rot (1 min)	Rot (1 min)
	Nea	2	892	15			Rot (1 min)	Rot (1 min)
	Nea	3	895	14			Rot (1 min)	Rot (1 min)
	Nea	4	894	12			Rot (1 min)	Rot (1 min)
	Nea	5	896	11			Rot (1 min)	Rot (1 min)
Homla	Homla	1	586	7	07.06.21	Control	Rot (1 min)	Rot (1 min)
	Homla	2	587	6			Rot (1 min)	Rot (1 min)
	Homla	3	590	3			Rot (1 min)	Rot (1 min)
	Homla	4	591	2			Rot (1 min)	Rot (1 min)
	Homla	5	592	1			Rot (1 min)	Rot (1 min)
Forra	Forra	1	342	5	08.06.21	Control	Rot (5 min)	Rot (5 min)
	Forra	2	340	4			Rot (5 min)	Rot (5 min)
	Forra	3	341	3			Rot (1 min)	Rot (1 min)
	Forra	4	339	2			Rot (1 min)	Rot (1 min)
	Forra	5	337	1			Rot (1 min)	Rot (1 min)
Dalåa	Dalåa	1	129	5	08.06.21	Impact	Rot (1 min)	Rot (1 min)
	Dalåa	2	132	4I			Rot (1 min)	Rot (1 min)
	Dalåa	3	134	4E			Rot (1 min)	Rot (1 min)
	Dalåa	4	135	4			Rot (1 min)	Rot (1 min)
	Dalåa	5	144	3			Rot (1 min)	Rot (1 min)
Skaudals-vassdraget	Skauga	1	1217	9	09.06.21	Impact	Rot (1 min)	Rot (1 min)
	Skauga	2	1216	8			Rot (1 min)	Rot (1 min)
	Skauga	3	1215	7			Rot (1 min)	Rot (1 min)
	Skauga	4	1214	6			Rot (1 min)	Rot (1 min)
	Skauga	5	1210	5			Rot (1 min)	Rot (1 min)
Nordelva	Holvasselva	1	580	1	09.06.21	Control	Rot (1 min)	Rot (1 min)
	Holvasselva	2	578	2			Rot (1 min)	Rot (1 min)
	Holvasselva	3	579	3			Rot (1 min)	Rot (1 min)
	Holvasselva	4	577	4			Rot (1 min)	Rot (1 min)
	Nordelva	5	959	1			Rot (1 min)	Rot (1 min)

Gaula	Gaula	1	418	8	10.06.21	Control	Rot (5 min)	Rot (5 min)
	Gaula	2	420	5			Rot (5 min)	Rot (5 min)
	Gaula	3	421	101			Rot (5 min) Rot (1 min)	Rot (5 min)
	Gaula	4	422	102			Rot (5 min) Rot (1 min)	Rot (5 min)
	Gaula	5	426	3			Rot (5 min)	Rot (5 min)
Verdals-vassdraget	Helgåa	1	528	7	11.06.21	Control	Rot (5 min)	Rot (5 min)
	Helgåa	2	525	4			Rot (5 min)	Rot (5 min)
	Verdalselva	3	1574	3			Rot (5 min)	Rot (5 min)
	Verdalselva	4	1573	2			Rot (5 min)	Rot (5 min)
	Verdalselva	5	1575	1			Rot (5 min)	Rot (5 min)

APPENDIX F – Results

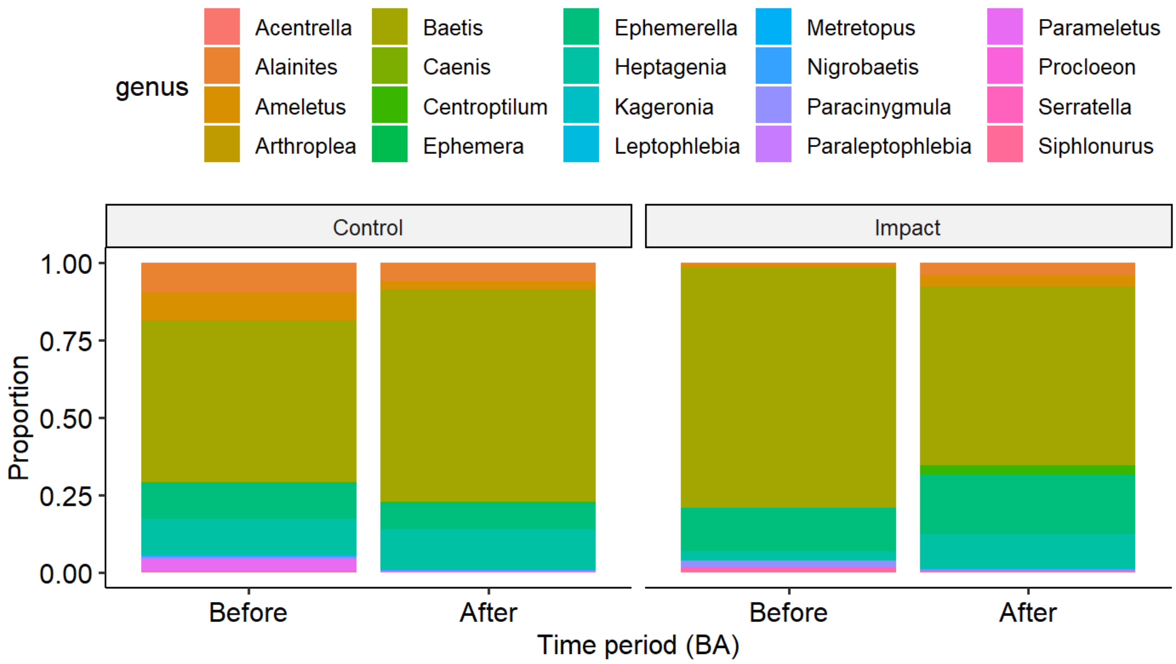

Figure 18. Proportion of individuals specified to different genera within the order Ephemeroptera, before and after impact in control and impact rivers. The genus *Baetis* dominates in all groups.

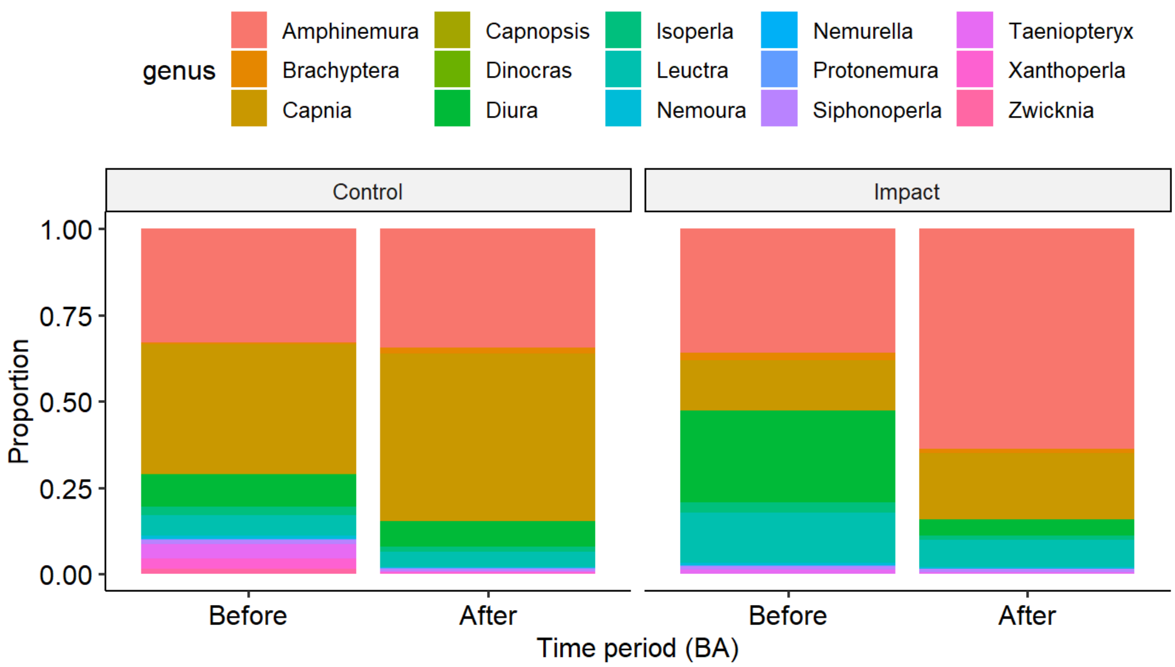

Figure 19. Proportion of individuals specified to different genera within the order Plecoptera, from before to after impact in both control and impact rivers. In control rivers, *Amphinemura* and *Capnia* are the most commonly occurring genera, while in impact rivers the two most common genera shift from *Amphinemura* and *Diura* to *Amphinemura* and *Capnia*.

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