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Impact assessment of Norwegian hydropower on freshwater fish species

an LCA approach

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MASTER THESIS

for

Student
Kim Rainer Mattson

Spring 2016

Impacts from hydropower production on aquatic ecosystems – an LCA approach

Påvirkninger fra vannkraftproduksjon på akvatiske økosystemer - en livssyklusanalyse tilnærming

Background and objective

The impact of hydropower production on aquatic species (e.g., fish, macroinvertebrates) and ecosystems in LCA is today largely neglected. However, hydropower production, especially with large dams, has a considerable influence on the aquatic ecosystem downstream, due to, for example, large reductions in flow volumes and timings of flow.
Methods exist in LCA that quantify the impact of water consumption on aquatic ecosystems (Hanafiah et al. 2011; Tendall et al. 2014) and the aim of this thesis is to investigate the potential to apply these methodologies to hydropower production. Instead of water consumption, the flow of the water volume is decreased due to the necessary storage. The spatial detail of this thesis can be either Norway or expanded to the whole world. SDRs (species-discharge relationships) for Norway or the world shall be developed for fish and possibly other taxonomic groups, in order to quantify the impact of a reduction in flow discharge.

Literature
Hanafiah, M. M., M. A. Xenopoulos, S. Pfister, R. S. Leuven and M. A. J. Huijbregts (2011). "Characterization Factors for Water Consumption and Greenhouse Gas Emissions Based on Freshwater Fish Species Extinction." Environ. Sci. Technol. **45**(12): 5572-5278.
Tendall, D. M., S. Hellweg, S. Pfister, M. A. J. Huijbregts and G. Gaillard (2014). "Impacts of River Water Consumption on Aquatic Biodiversity in Life Cycle Assessment - a proposed method, and a case study for Europe." Environ. Sci. Technol.: DOI: 10.1021/es4048686.

The following tasks are to be considered:

1. Develop an SDR for Norwegian Rivers.

2. Improve and refine the characterization factor for connectivity impacts.

3. Identify where in Norway the largest impacts from hydropower are created.

4. Compare the impacts of one case (river/dam) in a complete LCA of hydropower.

Within 14 days of receiving the written text on the master thesis, the candidate shall submit a research plan for his project to the department.

When the thesis is evaluated, emphasis is put on processing of the results, and that they are presented in tabular and/or graphic form in a clear manner, and that they are analyzed carefully.

The thesis should be formulated as a research report with summary both in English and Norwegian, conclusion, literature references, table of contents etc. During the preparation of the text, the candidate should make an effort to produce a well-structured and easily readable report. In order to ease the evaluation of the thesis, it is important that the cross-references are correct. In the making of the report, strong emphasis should be placed on both a thorough discussion of the results and an orderly presentation.

The candidate is requested to initiate and keep close contact with his/her academic supervisor(s) throughout the working period. The candidate must follow the rules and regulations of NTNU as well as passive directions given by the Department of Energy and Process Engineering.

Risk assessment of the candidate's work shall be carried out according to the department's procedures. The risk assessment must be documented and included as part of the final report. Events related to the candidate's work adversely affecting the health, safety or security, must be documented and included as part of the final report. If the documentation on risk assessment represents a large number of pages, the full version is to be submitted electronically to the supervisor and an excerpt is included in the report.

Pursuant to "Regulations concerning the supplementary provisions to the technology study program/Master of Science" at NTNU §20, the Department reserves the permission to utilize all the results and data for teaching and research purposes as well as in future publications.

The final report is to be submitted digitally in DAIM. An executive summary of the thesis including title, student's name, supervisor's name, year, department name, and NTNU's logo and name, shall be submitted to the department as a separate pdf file. Based on an agreement with the supervisor, the final report and other material and documents may be given to the supervisor in digital format.

☐ Work to be done in lab (Water power lab, Fluids engineering lab, Thermal engineering lab)
☐ Field work

Department of Energy and Process Engineering, 13. January 2016

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Olav Bolland
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Acknowledgements

First and foremost, I would like to thank my supervisor Francesca Verones, associate professor at the industrial ecology program, NTNU. She provided excellent feedback and help not only with the bigger picture of my thesis, but also the small things that are easy to miss. I would also like to thank PhD candidate Martin Dorber at the industrial ecology program for help gathering species data, and for general constructive criticism on multiple key assumptions made in the thesis. The help of Trine Fjeldstad, from the hydrology division at The Norwegian Water Resources and Energy Directorate (NVE), acquiring the large amount of hydrological data from NVE, was pivotal for doing a large scale and thorough assessment of Norwegian hydropower. In regard to data availability, I would like to acknowledge the importance of the public access data that NVE provide. Much of the data I have used in my thesis is taken directly from NVE, and I think we often take for granted how important it is that data of this scale and detail is readily available to the public. The thesis work does not consider the role of public information, so I would like to make a general point, as an overall take away message. In order for large scale analysis and further knowledge generation regarding our resource use, the importance of large scale public access databases cannot be neglected. In order to do science and improve society, transparency is key.

Abstract

Accounting for the total environmental impacts associated with energy technologies are becoming increasingly important due to large scale development of renewable resources. In order to assess the trade-offs between large scale development of various technologies, there needs to exist a transparent and efficient quantitative method for such analysis. The goal of this thesis has been to develop an impact assessment of Norwegian hydropower, by constructing a characterization factor that models the relationship between water use for energy production and impacts on freshwater fish species. The thesis presents the importance of hydropower as a renewable energy technology, but focus exclusively on quantifying the negative biodiversity impacts from hydroelectricity production, using the life cycle assessment method. Species-discharge-relationships are calculated for Norway, showing a lower species density per unit of discharge for rivers with high development of hydropower compared to rivers with low development of hydropower. Discharge rates from 97 Norwegian rivers, water efficiency scores, and energy production data, are used to assess the impacts of hydropower. Results single out northern and south-eastern regions of Norway as the main contributors to freshwater fish impacts. The yearly impact of hydropower production from the rivers included in this thesis is estimated to be 0.14 species lost per year. The validity of this estimate is discussed.

In order to evaluate the compatibility of the characterization factor with life cycle assessment, the life cycle inventory data from two EPDs on hydropower stations are used to calculate species impact scores on a per kWh basis. From this we see that the characterization factor is applicable to LCA and provides a species loss estimate relevant for local freshwater fish species. Further development of a connectivity index directed towards including habitat fragmentation into the impact assessment is done and applied to 35 rivers. The inclusion weigh the impact scores of rivers based on the difficulty level of migration due to barriers, as a function of dam development. The applicability of this index is discussed, and further investigation highlighted. Lastly a basic framework for constructing regionally specific characterization factors for species impacts by hydropower is presented, this framework is based on the importance of the parameters that are identified as the most essential for the analysis.

Keywords: Life cycle assessment, LCA, Impact assessment, Hydropower, Norwegian, environment, freshwater fish, species-discharge-relationship.

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1. Introduction

1.1 The importance of hydropower in global energy production

The energy supply sector, is the main contributor to global greenhouse gas emissions. The Intergovernmental Panel on Climate Change (IPCC) 2014 report, estimates that 35% of the total anthropogenic greenhouse gas emissions (GHG) in 2010, originate from the energy sector, which is the largest fraction of all sectors.

In Work Package 3 report of the IPCC, which deals with the pressing issue of mitigating future global warming, a substantial part highlights the increasing need for research and deployment of renewable energy technologies. The report also focuses on the potential for up-scaling the energy production of established renewable energy technologies, in order to mitigate future GHG emissions (IPCC, 2014). There is an expected 85% increase in energy production from renewable energy sources over the next 30 years (Prado et al. 2016).

Figure 1 shows the recent global growth in total installed capacity of renewable energy, and shows the relative contributions of technologies like hydropower, photovoltaic solar power, and wind power (IPCC, 2014). The figure highlights that the largest installed capacity is from hydropower. Energy from hydropower represented 16% of the total global energy production in 2008 (IPCC, 2011).

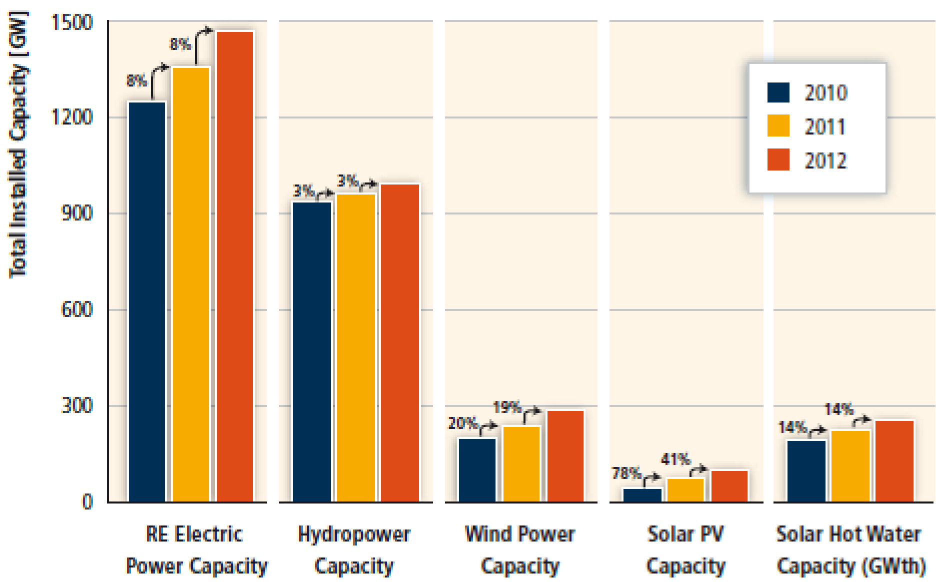

Figure 1:Growth in total installed capacity [GW] of renewable energy (RE) from 2010 to 2012. We also see the breakdown of the various contributing production technologies (Source: IPCC, 2014).

The International Energy Agency (IEA) produced a roadmap report for hydropower development where they estimate that approximately 85% of all energy from renewable sources is from hydropower (IEA, 2012). Both the IEA report and the WG 3 report by the IPCC highlights that the increasing need for renewable energy in the energy sector, is an opportunity for increasing development of hydropower (IEA, 2012; IPCC, 2014), however, regarding other environmental issues the synthesis of the reports differ slightly. The IEA report states in their key findings that

"... hydropower projects must be designed and operated to mitigate or compensate impacts on the environment and local population. The hydropower industry has developed a variety of tools, guidelines and protocols to help developers and operators address the environmental and social issues in a satisfactory way" (IEA, 2012).

The IPCC report however, clearly communicates that there can be trade-offs involved with increasing hydropower development (and other renewable energy sources), and that these need to be accounted for in scenarios projecting an increase in hydropower (IPCC, 2014). The IPCC report does not consider the potential issues resulting from hydropower development as being currently addressed in a satisfactory way.

The IPCCs Special Report on Renewable Energy Sources and Climate Change Mitigation (SRREN)(IPCC, 2011) goes further into detail on the potential of hydropower. Chapter 5 of the report is exclusively dedicated to hydropower. The report states that the "*[m]ain challenges for hydropower development are linked to a number of associated risks such as poor identification and management of environmental and social impacts"* (IPCC, 2011). According to the SRREN report, there is a need for including long-term environmental consequences from hydropower into current and future projects (IPCC, 2011).

The SRREN report focus mainly on the potential impact hydropower can have on the biodiversity of rivers, due to alterations in hydrological conditions. It also highlight the impacts on local populations forced to resettle due to reservoir creation (IPCC, 2011). The overlaying theme of mitigating impacts on the environment, and the importance of biodiversity in this respect, was clearly conceptualized by the Millennium Ecosystem Assessment (2005). The assessment points out that the changes in biodiversity due to human activity has been more rapid the last 50 years than other periods of human history (MA, 2005). Most of the MEA deal with both the multiple losses of economic value incurred by destroying biodiversity, and the ethical and negative human health aspects of continuing the degradation of the environment (MA, 2005). The importance of these topics set the precipice for including biodiversity impacts into the assessment of hydropower development.

1.2 Current trends in hydropower research and inclusion of LCA

A study by Jiang et al. (2016) investigated the research trends within the fields related to hydropower over the last 19 years (1994 - 2013). They used meta-data algorithms in order to create a topic analysis of 1726 scholarly articles highly related to hydropower, which allowed them to identify research development, current trends, and the intellectual structure of hydropower literature (Jing et al. 2016). The findings from this study are helpful for understanding some of the current topics most relevant within hydropower research. The study clearly shows that scientific publications related to hydropower development have increased substantially from 1994 to 2013, from an annual publication rate of 28 articles in 1994, to an annual rate of 238 in 2013 (Jiang et al. 2016). Two of the topics that show a clear growing trend are focused on energy security and climate related issues, and the topic of ecosystem impacts from hydropower (Jiang et al. 2016).

Environmental topics related to fish ecology, species habitat and ecosystem degradation represented 40.9% of the 1726 peer reviewed articles included in the study (Jiang et al. 2016). This clearly shows that there exists a significant research interest on the environmental impacts of hydropower. This could further be exemplified by looking at studies published after 2013. Gaudard & Romerio (2014) for instance focus on the potentially large increase of the installed technical potential for hydropower in Europe, where 51.5% of the technical hydropower potential has already been developed (Gaudard & Romerio, 2014). Gaudard & Romerio (2014) define the technical potential for hydropower to be the total hydropower potential of all sites that could be developed, excluding economic or environmental restrictions, while the developed potential is the fraction of the technical potential that is actually developed.

Gaudard & Romerio (2014) point out that towards the end of the century the potential for hydropower production in Western Europe could decrease with a variety of estimates ranging from 15% to 6%, due to a decrease in precipitation. Even though, there will be a likely increase in precipitation in the northern regions of Europe, it does not make up for the losses incurred in the rest of Europe (Gaudard & Romerio, 2014). The general message is thus to increase the focus on developing the technical potential of European hydropower, in order to ensure that the energy supply does not decrease in renewable energy. However, as a consequence, economic and environmental restrictions will need to be lifted.

Prado et al. (2016) present a case study of the policy side of the projected energy security measures of Brazil, where 30 new large hydropower dams are planned to be constructed in the Amazon River over the next 30 years. They highlight that although more renewable energy is needed, the socio-environmental issues associated with increased supply-side energy development must not be neglected, and that focusing on the demand-side of energy is important for understanding the dynamics of energy use and the development of policies.

Other recent studies fit more into topics that are focusing more exclusively on ecosystem services, compensation mechanisms, and increasing standards for mitigating ecosystem impacts. Yu & Xu (2016) focus on a need for compensation mechanisms in order for hydropower projects to internalize the externalities associated with changing hydrological regimes and geomorphology of rivers, in order to ensure a truly sustainable future development. Yu & Xu (2016) highlight the need for a proper quantitative framework in order to evaluate the socio-economic costs of hydropower development . Another study by Schramm et al. (2016) looks at licensing or relicensing of mitigation plans for 300 hydropower plants in the US from 1998 to 2013. They point out that although federal requirements for environmental protection has been included in hydropower development since the 1970ties, the clarification on what mitigation activities have been implemented, are found lacking. Schramm et al. (2016) notes that increasing mitigation in areas of environmental flows, fish passage, and water quality are needed to ensure environmentally sustainable hydropower development in the US.

We can infer from these studies and reports that the analysis of both the positive and the negative consequences of hydropower has significantly increased over the last decade, largely due to the pressing issue of climate change and energy security. Due to this increased attention to the total environmental consequences of technological development, the IPCC stress the need for a universal and transparent methodology for performing impact assessments of technologies, and a framework for effective comparison between technologies (IPCC, 2014). The SRREN report singles out life cycle assessment (LCA) as a potential method for thorough investigation of hydropower projects, and other renewable technologies (IPCC, 2011).

The objective of a life cycle assessment is to perform a consistent comparison of technical systems, with considerable attention to the environmental impacts these systems contribute to. This means including the different stages of production, use and maintenance, and end of life treatment that the systems require. Accounting for emissions occurring in these stages provides a thorough quantification of the possible environmental impacts due to production or use of some product or process (Strømman, 2010).

The general framework for performing an LCA involves a goal and scope phase, followed by an inventory analysis of all inputs required to produce a functional unit of the product, and the impact assessment of these inputs and/or emissions (Rebitzer et al. 2004). Figure 2 illustrate this framework. For a further introduction to LCA, section 3 of the appendix goes more into detail.

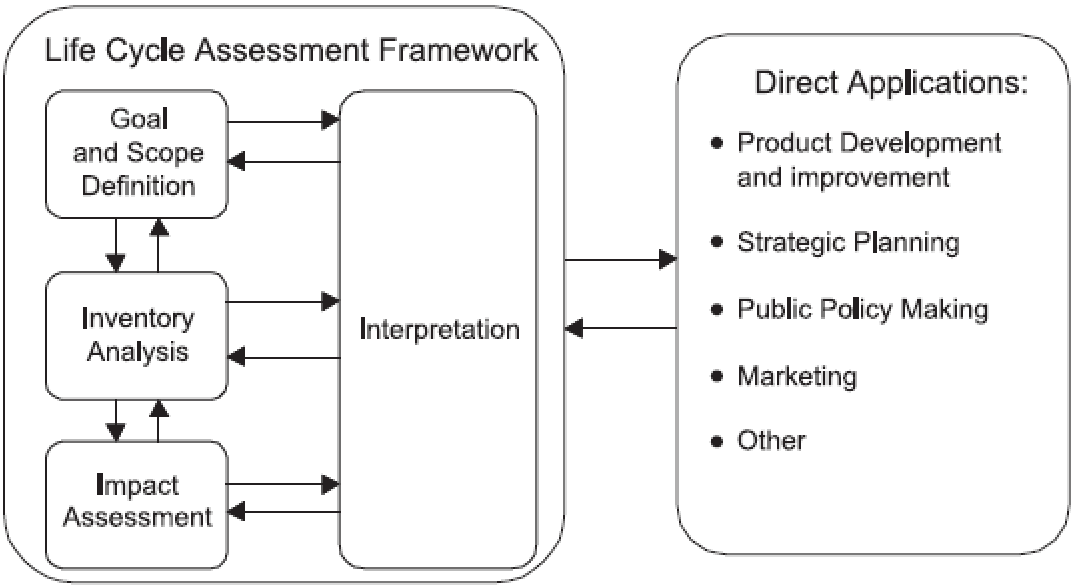

Figure 2:All the phases of the LCA, and the resulting interacting direct applications of performing the analysis (Source: ISO 14040, 1997).

A review of LCA studies performed on hydropower plants and wind power farms was conducted by Raadal et al. (2011), showing that GHG emissions from hydropower and wind power vary between 0.2-152 gCO_2-eq per KWh, where both technologies have very low impacts on global warming potential compared to carbon intensive energy technologies. However, taking the total environmental impact of these technologies into account, could clarify which technology carries the lowest environmental burden. A more recent study by Hertwich et al. (2015) used hybrid-LCA to compare both low-carbon energy technologies and high-carbon energy technologies, including multiple impact categories such as particulate formation and eco-toxicity into the analysis. This study provides evidence of multiple benefits to society with adopting cleaner energy sources, as well as giving indicators towards which of the energy technologies incur the lowest total environmental impact.

Life cycle impact assessment is an essential part of the LCA. Here one tries to formulate cause and effect models from interventions on the surrounding environment, in order to predict the consequences of these interventions (Bengtsson & Steen, 2000). Assessment of water use has only recently become an area of research within LCA, but as stated in a comprehensive review article by Kounina et al. (2013), there exists no method yet which describes all potential impacts from freshwater use.

The issue of water use is important when considering hydropower. Hydropower reservoirs have been characterized as large consumers of water (Mekonnen & Hoekstra, 2012), and multiple review studies have highlighted the multiple in-stream impacts of hydropower installations on biodiversity (Puffer et al. 2014; Schmutz et al. 2014; Anderson et al. 2015). In order to quantify the negative impacts of hydropower we need models that predict its impacts. A recent review study by Gracey & Verones (2016) gives an overview of the multiple ways hydropower may damage biodiversity and how water consumption has been incorporated into LCA, and provides a general framework towards including water consumption when assessing hydropower. Currently, no impact assessment for quantifying species impacts of hydropower production exists (Gracey & Verones, 2016).

Increasing attention towards water use and consumption within LCA was the main topic of discussion during the Water Use in LCA (WULCA) working group (Boulay et al. 2015). Here, the focus on greater spatial and temporal resolution for consumptive water footprints and water stress-based indicators for LCIA, was highlighted. The working group points out that large monthly variation in water availability and consumption can lead to over-or underestimation of impacts when using mean annual hydrological data (Boulay et al. 2015). These points are also covered by Gracey & Verones (2016) when addressing water consumption in the context of hydropower, since hydropower impacts will be subject to specific locations and seasonal variability.

1.3 Impacts of hydropower on biodiversity

Various factors contribute to the impacts generated by hydropower, and I will to this end reiterate a short overview of results from the preceding project thesis' literature review. The impacts generated by hydropower is tightly connected to the type of hydropower plant, and the resulting changes to river hydrology the different types contribute to. Therefore the physical changes to the geomorphology of the river system due to hydropower is first introduced in section 1.3.1, followed by the direct impacts on biodiversity due to these physical changes in section 1.3.2.

1.3.1 Hydrological alterations due to hydropower

The plants that have been most widely studied in regard to large changes to river hydrology are pump-storage hydropower and reservoir dams with hydropower plants (IPCC, 2011). Pump storage and reservoir hydropower plants are based on the principle of catching water at sufficient height, and releasing it when demand for electricity peaks. The reservoirs store water for later consumption, thus reducing the variability of river flows (IPCC, 2011). Major hydrological implications are changes in timing, magnitude, and frequency of flows compared to natural flow regimes- (Church, 1995; Magilligan & Nislow, 2005), that is to say that the average river discharge ([m^3/s]) is decreased. The reversal of hydrographs in certain seasons (increased discharge during winter and decrease during spring/summer) is a prevalent feature of strongly regulated river systems (Magilligan & Nislow, 2005). This is evident in figure 3 of the River Orkla in Sør-Trøndelag, Norway, a strongly regulated river since the 1980s.

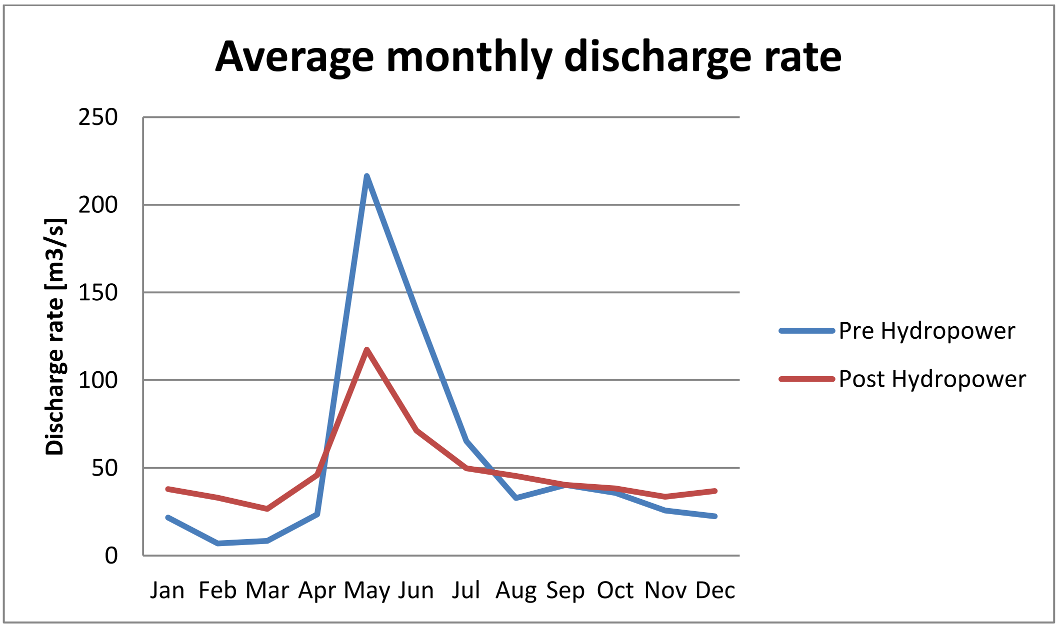

Figure 3: Hydrograph of monthly discharge rates for the River Orkla. The blue line represent the data collected in the period 1972 to 1977, before the development of hydropower in the river. The red line represents the average discharge rates in the period 1978 to 2015 in the River Orkla, after hydropower has been developed in a large scale throughout the river (Source: Mattson, 2015).

Other changes to the river system due to reservoir creation is the flooding of large areas when reservoirs are built, which can affect local and regional weather and climate (Wondmagegn & Faisal, 2015). It also dramatically changes the flooded zone from a river valley or wetland area, to a reservoir (Braatne et al. 2008). Added effects to the reservoir is decreasing sediment transport and deposition downstream from the plant/dam. The sediment transport decreases with lower discharge rates and physical barriers (Church, 1995).

Another type of hydropower plant is the Run-of-River plant. A run of river (ROR) hydropower plant uses the available flow of a river system to produce energy, which means that this kind of hydropower scheme can have a variety of flow regimes at different seasons, due to changes in precipitation and runoff variability (IPCC, 2011). The use of weirs makes it possible to regulate a fraction of the flowing water into a secondary channel and through a turbine (or multiple turbines), and afterwards direct the water into the river stream again (Anderson et al. 2015). These structures can also alter river hydrology. Weirs alter the physical nature of the river by reducing flow variability, velocity and turbulence, which creates a lentic environment upstream of the weirs that can extend several kilometers, that differs from natural flows (Anderson et al. 2015). The effects downstream of the weir are higher velocity, more turbulent flows, and a flow with less sediments changing the geomorphology of the river (scour holes, bar formation) (Csiki & Rhoads, 2010; Anderson et al. 2015).

1.3.2 Biodiversity impacts from habitat alteration

The hydrological changes incurred by these large scale hydropower installations affect multiple species and ecosystems in various ways. For instance, the flooding of areas upstream of a dam and the resulting decreasing in river discharge downstream of the dam can have drastic effects on the riperian ecosystems on both side of the dam (Poff et al. 2010; Poff et al. 2011). Where the riperian ecosystem is the transition zones between land and water ecosystems, and represents a very rich vascular plant diversity that vary with the size of the rivers (Nilsson, 2002). Nilsson et al. (1994) reviewed hydropower impacts on mammals and birds, for instance documenting severe impacts from fluctuating water levels on Eurasian beaver populations, as well as reduction in otter populations due to changing food webs. There is also evidence of changing migration patterns for birds in Scandinavia (Nilsson et al. 1994). There also exist evidence of micro and macro-invertebrate impacts due to decreasing river flows (Dewson et al. 2007).

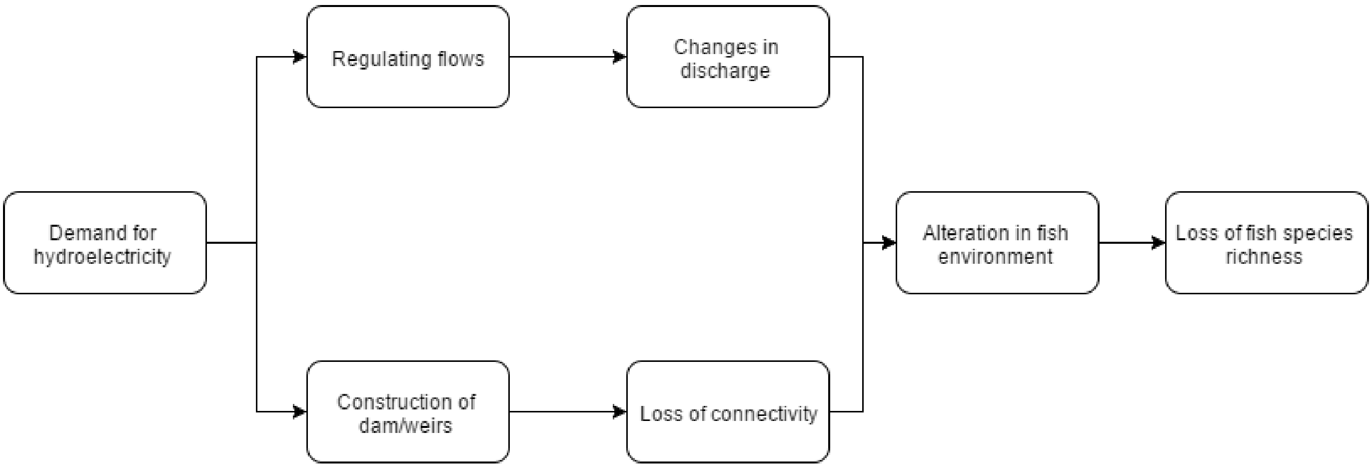

Figure 4: The most important impact pathways for fish species as a consequence of hydropower development (Source: Autumn project, 2015).

Figure 4 represents the main impact pathways that where identified for freshwater fish in the project work. Although multiple species are negatively affected by hydropower development, the most studied and considered within the scientific literature are the various types of freshwater fish species. The heightened focus on fish species is linked to the increased value we place on fish species compared to other species, the greater data availability for fish, and that fish species richness has been considered a good indicator of overall ecosystem quality (Belpaire et al. 2000; Gassner et al. 2003).

Hydropower installations have been directly linked to extinction of certain fish populations (Johnsen et al. 2010; Dudgeon et al. 2011). The main impact pathways identified are loss of connectivity and changes in discharge rates (figure 4). Although there are potential other impacts occurring, such as changes in temperature (Jensen, 1987; Donaldson et al. 2008) and stranding of migrating fish due to large flow variability during migration (Nagrodski et al. 2012).

The loss of connectivity between lakes and rivers represents a clear hindrance for species that need to move longitudinally up and down the river. Much of the research on hydropower impacts has focused on the challenge faced by migratory fish species when large dams and weirs are built (IPCC, 2011). The migration of fish is essential for many fish species' life history stages.

The effects of smaller dams and weirs has also been investigated. Gauld et al. (2013) studied the effects of downstream migration of brown trout smolts over small weirs in the River Tweed in the UK. They found that smolts exhibited major losses while migrating, especially when the flow regime was low (Gauld et al. 2013). In a study where weirs were removed in order to restore local populations of salmon in a Norwegian river, results showed that salmon spawning sites could be successfully recreated, with reduction in egg mortality and marked increase in the density of juveniles (Fjeldstad et al. 2012).

The problems of migrating fish species is not new, mitigation of these impacts have been under consideration and development in Europe as early as the 1900s (Katopodis & Williams, 2012). However, relatively little quantitative assessments of these techniques' efficiency has been performed (Oldani & Baigun, 2002; Noonan et al. 2012). An extensive review by Noonan et al (2012), assessed articles from 1960 to 2011, provides an overview of the general efficiency of fish ladders and fishways for up and downstream movement. On average, downstream passage efficiency was 68.5%, and upstream efficiency was 41.7%, based on estimates from 65 papers. Estimates also show that salmonids are better at using the ladders than non-salmonids for both upstream and downstream passing (61.7% vs. 21.1% upstream, and 74.6 vs. 39.6% downstream) (Noonan et al. 2012). It is clear, that dams and weirs represent a threat to many fish species, and that the mitigation efforts used today to address these issues do not prevent species impacts.

Changes in discharge rates are associated with increases and decreases in flow magnitudes, where both changes has shown to impact fish abundance, diversity and demographic rates negatively (Poff et al. 2010). These are impacts that reflect the change in natural flows due to regulation of rivers, averaging out natural flow variation, but also reduction in river volume due to reservoir creation. Water flow regime and food availability are assumed to be the main variables affected by discharge (Arnekleiv et al. 2006).

Rivers with high discharge rates have over evolutionary time provided river ecosystems with larger variability of biologically preferable environments, like comfortable temperature ranges, solute concentrations, pools, runs, and sandy or silty substrates. This is the underlying assumptions made by Xenopoulos and Lodge (2006) when creating a regression model to forecast biodiversity loss as a function of the loss of river discharge rates, creating species-discharge relationships (Xenopoulos & Lodge, 2006). McGravey and Ward (2008) showed that zonal data is important when estimating within-basin habitat diversity, pointing out that diversity of regions can be large and thus more regional data is necessary for robust assessments of the environment.

Teichert et al. (2010) performed a controlled study testing the linear assumption between reduction in discharge rates and fish density, and showed evidence for a linear relation between growth of juvenile salmon and discharge rates during summer. A similar study was performed by Puffer et al. (2014), where they simulated a hydropeaking flow regime under controlled conditions. They found the same general trend, where low discharge rates during summer reduced growth in juvenile salmon (Puffer et al. 2014). Another study by Schmutz et al. (2014) show that greater magnitude of fluctuations in flows compared to natural flows produce negative impacts on the growth of fish species. These studies enhances the legitimacy of using species-discharge relationships for producing models of fish biodiversity effects. However, they also show that discharge rates can vary in impact depending on seasonal variations.

In summary, the impacts from discharge changes and connectivity barriers should be the first pathways that are modeled into full life cycle impact assessments of hydropower. This is therefore the main concern of this thesis.

1.4 The importance of hydropower in Norway

Hydropower is responsible for 95% of all electricity produced in Norway, with an average of 130 TWh yearly (Flåten et al. 2014). The energy from hydropower originates from approximately 600 power plants all over Norway (Saha et al. 2016), with the regions representing the largest technical potential for hydropower being the north of Norway (Nord-Norge) and the west coast (Vestlandet) (Flåten et al. 2014). Figure 5 shows the concentrations of hydropower plants in Norway.

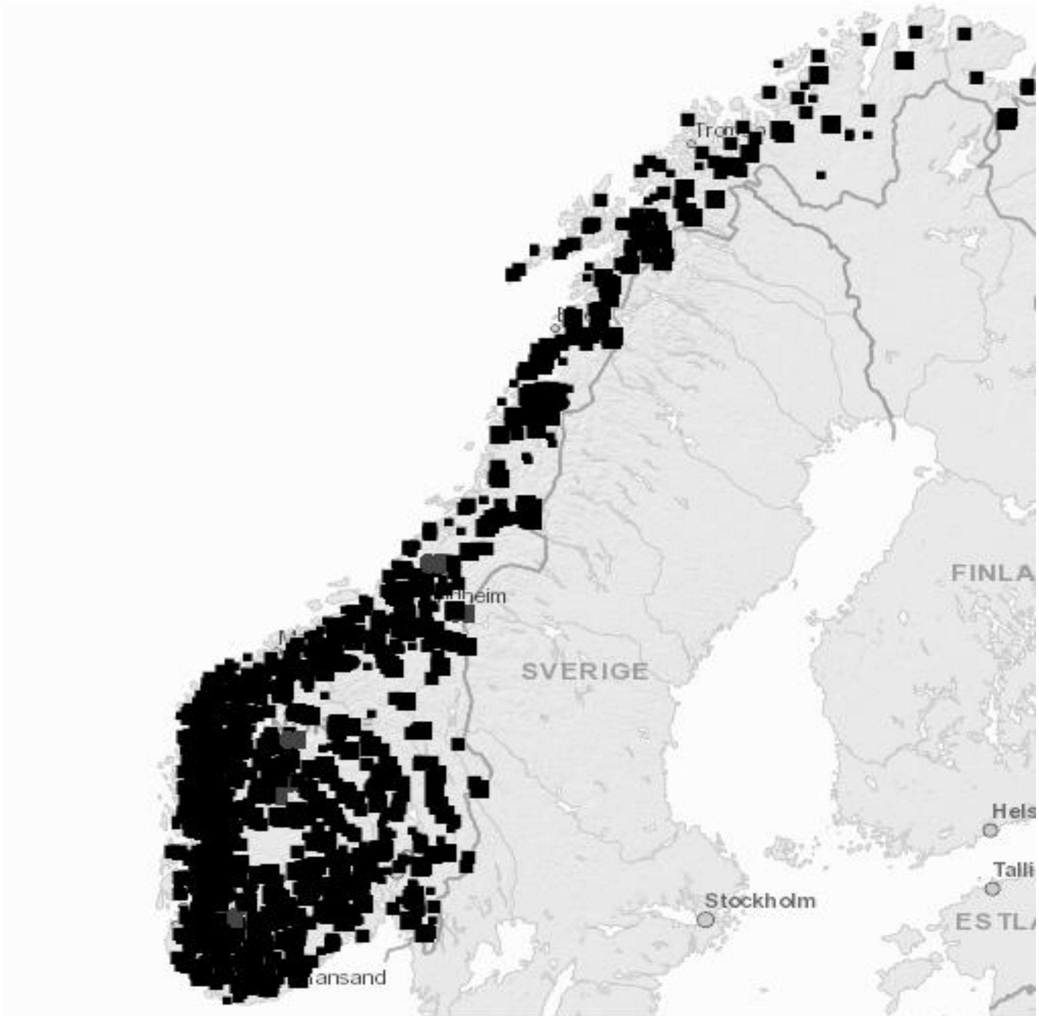

Figure 5: The black squares represent hydropower plants, this illustrates the concentration of hydropower plants in Norway (Source: NVE 2016).

The total potential energy production is estimated at 214.8 TWh per year. Here 60.8% (130,5 TWh) of the technical potential has been developed, where 23.6% (50.8 TWh) is protected against further development (Flåten et al. 2014).

Much of the energy produced supply energy intensive industries (Flåten et al. 2014). However, a large part of the energy goes to satisfy the household energy demand, which is the highest in Europe, at an average annual energy consumption of 17 000 kWh per household (SSB, 2012). This level of energy consumption has been linked to both colder climate and especially low electricity prices (SSB, 2012). The Norwegian Directorate for Water and Energy (Norges vassdrags og energidirektorat, NVE) is the governmental institution responsible for providing concessions to hydropower projects (NVE, 2016). These concessions have multiple environmental criteria associated with them. For instance, maintenance of minimal flow regimes and fish passage mechanisms where this is relevant. In order to get approval, companies need to document all potential positive and negative consequences of the hydropower plant (NVE, 2016).

Although environmental considerations have seemingly been a priority for hydropower development in Norway, there have been multiple observations of impacts on fish species due to hydropower. Out of 45 Norwegian salmon populations that have been lost, 42% have been attributed to the development of hydropower (Hansen et al. 2008). Fjeldstad et al. (2015) performed an analysis of 344 fishways in Norway, estimating that only 66% where functioning well. However, the criteria for assessing the functionality of the fishways were purely qualitative and dependent on the expert opinion of the individual responsible for Fishery Management at the County Governors office (Fjeldstad et al. 2015). A review study by Trussart et al (2002) tried to identify and evaluate mitigation measures from multiple hydropower project around the world (Norway included). The study concluded that the rate of follow-up and publication of these measures effectiveness was simply too low to provide any useful assessment of different mitigation measures (Trussart et al. 2002).

With the large amount of hydropower installations in Norway and the multiple observation of impacts referenced from Norway specifically and hydropower more generally, none question that hydropower has a potentially negative environmental burden, although being a much needed renewable resource. Accounting for environmental impacts needs to be put into a quantitative framework, if the real cost and benefit to increasing hydropower development is to be assessed. This is especially important for Norway, where both governing politicians (Regjeringen, 2016), academic researchers (Brende et al. 2016), as well as the EU through the "Green Electricity Certificate" system (IEA, 2003) are pushing for more renewable energy by increasing hydropower production. To further highlight the focus on hydropower in Norway, the Environmental-friendly Energy Research (FME) center recently received 1,3billion NOK for further research on renewable energies, hydropower representing a large portion of the centers research areas (Universitetsavisa.no, 2016). The main focus for the Norwegian Research Centre for Hydropower Technology is seemingly directed towards value creation, and does not specify direct research on local environments (Forskningsrådet.no, 2016).

1.5 Impact assessment of Norwegian hydropower production

In order to perform impact assessments of hydropower, a good starting point is the framework developed by Gracey & Verones (2016). They highlight that modification to some existing in-stream characterization factors (namely the once developed by Hanafiah et al. (2011) and

Tendall et al. (2014), taking account of more seasonal variability of river flows as well as using multiple ecological response curves to model site specific impacts, should be a priority. The studies conducted by Hanafiah et al (2011) and Tendall et al. (2014) were the starting point for the project work, Since their work is also relevant for this master thesis, I will shortly reiterate some of the important points.

The study by Hanafiah et al. (2011) developed a characterization factor for potential freshwater fish losses from water consumption, building on the concept of species discharge relationships (SDR) provided by Xenopoulos and Lodge (2006). These SDRs are based on regression models predicting a relationship between the amount of discharge within a river, and the number of fish species living within the river. The study provides a way to estimate the change in potentially disappeared fraction of freshwater fish species, as a function of the marginal change in river discharge rates. Marginal changes in discharge are attributed to marginal changes in water consumption for the river basin (Hanafiah et al. 2011). The source of water consumption could be any form of human activity that takes water from a river basin at the cost of the environment. Hanafiah et al. (2011) exclude river basins above 42 degrees latitude, due to these being recently glaciated and not having had "enough time to evolve to their maximum species richness potential" (Hanafiah et al. 2011).

With this study as a basis, Tendall et al. (2014) developed a more regionalized approach, and included impacts on macro-invertebrates. They developed region specific SDR's by acquiring species and river discharge data on a country level, for multiple countries in Europe. Species got assigned a threat or rarity factor, weighting the effects of vulnerable freshwater species. They also included longitudinal river zones, in order to account for the propagation of water consumption downstream of the area of consumption, summing the effects of the change in rivers discharge (Tendall et al. 2014). The characterization factor provided by Tendall et al. (2014) are in absolute numbers of global species extinction equivalents [GSE*y]/[m^3], whereas the characerization factor of Hanafiah et al. (2011) is in potentially disappeared fractions [PDF*m^3 *yr*m^{-3}].

In the project work the characterization factor developed by Hanafiah et al. (2011) was modified to represent water use as a direct consequence of hydropower production. In order to account for the impacts this use of water can have on species, we used SDRs, however, they were not specific to Norway. The project used two rivers as a case study, the River Orkla and the River Gaula. The results showed that impacts where lower for the river with significantly less hydropower development. Therefore this thesis aims at increasing the scope of analysis to all hydropower in Norway, with SDRs specific to Norway. In order for this impacts pathway to be useful for LCA purposes, we want to apply it to LCI data on hydropower plants.

The analysis above only relates the consequences of using the river discharge and potentially changing river flow parameters in river flow to species impacts. As highlighted by the project review, and Gracey & Verones (2016), the connectivity of the river system is not assessed in any way. The project therefore tried to implement some novel developments within the scientific literature on river network ecology. Specifically, the development of indexes of connectivity, that use graph theoretic approaches to movement within a network of interconnected nodes. Further information on the basic theory can be found in section 3 of the appendix. A connectivity index was incorporated into a characterization factor, this index took direct inspiration from the Dendritic Connectivity Index (DCI) by Cote et al. (2009), and the habitat connectivity index for upstream passage (HCIU) by Mckay et al. (2013). In this thesis we want to further assess the applicability of such a network approach to LCIA.

2. Materials and Methods

In this section the materials and data used in order to construct the impact assessment is presented first. We then introduce the construction of the species-discharge-relationship for Norway and the construction of the connectivity index, which are the most important parameters of our characterization factor. We then lastly go through the construction of the full characterization factor and its conversion to impact values.

2.1. Data collection

Discharge rates

Discharge rates from 136 out of 162 rivers was provided by the Norwegian Water Resource and Energy Directorate (NVE) on request. In the 26 missing rivers the hydrological stations were decommissioned, thus no data was available. The discharge data is a measure of the flow rate ([m^3 /s]) at the hydrological station closest to the river mouth. The time period of measures varied a great deal from case to case, with some rivers having discharge data as far back as 1900 to the present day, and some rivers having only a few years of data. The rivers requested where picked based on amount of hydropower installed, and the existence of hydrological stations. In the cases where discharge data was not available for a period longer than 10 years, the river was excluded (2 rivers). This is due to hydrological regimes being highly varied in nature, thus a sufficient time period for assessing the average flow of a river should use data extending over long periods of time (Hunger et al. 2008). Some rivers were not included in the analysis due to only measuring production discharge from the hydropower plant, and not actual river discharge. The large amount of data was handled using Excel pivot tables, where the average yearly and average monthly discharge rates, along with their standard deviation, were calculated in order to assess the temporal variation in discharge rates and to identify possible changes to discharge rates due to hydropower development.

Identification of hydropower installations

Using the Norwegian Water Resource and Energy Directorates web based GIS database (atlas.nve.no), we were able to identify all developed hydropower installations in Norway. This GIS map is open to the public, and provides the user with information regarding the geographical location of hydropower installations, their total yearly energy production in GWh, and information regarding the concessions for approved construction provided by NVE. All hydropower projects are legally obliged to apply for these concessions (NVE, 2009). The identification of energy production was also taken from this GIS database, where most plants were double-checked with energy production estimates from either the owning company's webpage, SNL.no (Online Norwegian Lexicon) or Norwegian Wikipedia.

Passability estimates of hydropower Plants

These estimates are based on information gained from the GIS database (atlas.nve.no), and in some cases the information was provided by concessions and values provided by the scientific literature on the efficiency of fish ladders. Very few concessions specify if there exists fish ladders, but the owners of the hydropower installation are obliged to avoid obstruction of fish migration. Examples of this are the concessions for Ryånda (NVE, Konsesjon Ryånda kraftverk AS, 2005) and Gautvella (NVE, Konsesjon Gautvella kraftverk, 2006) that specify mitigation measures to limit impacts on the movement of fish species. The efficiency of fish passage, however, is not covered in any degree by these concessions. In order to assess this we use passage efficiencies provided in a meta-study by Noonan et al. (2012). Where there exists dams, and there is no information about considerations towards migrating fish, or no concession providing information about the construction of the dams, a passage efficiency of zero was assumed. This means that fish have no chance of upstream passage over the barrier. Generally an upstream passage efficiency of 60% was used in the majority of rivers, following Noonan et al.'s (2012) average estimates for salmon barrier passage. These data were used in order to construct the connectivity index in section 2.2.

Calculation of water use per kWh of energy production (α)

The notation used for this parameter is α, and the calculation was done using the efficiency of production estimates given by the GIS database (atlas.nve.no). Most hydropower plants have a measure of kWh/m^3 as a measure of plant efficiency, which is usually calculated with momentum and mass-conservation equations for pipe hydraulics, including multiple factors like head, pipe size, gravity, and density of water (Bryan et al. 1992). Not all hydropower plants provide an estimate of these efficiency measures. In this case, the average efficiency of all the plants that do provide estimates are assumed for these plants. Since the yearly energy production of each plant is provided by NVE, we can calculate backwards to gain the yearly cubic meters of water needed to produce the given amount of energy.

$$m^3 = \frac{KWh}{KWh / m^3}$$

The total water use and total energy production of all plants in the river network is then divided with each other in order to gain an average m^3/kWh value for α. This is multiplied with the fate factor used by Hanafiah et al (2011) and Tendall et al. (2014), and produces the fate factor that we use in our analysis. If we wish to change the temporal aspect of this estimate, going from annual use of water, to monthly use of water, we can identify the percentage of monthly hydroelectricity production per month by using data provided by SSB for each county. This was done in the project work, however, we then provided a monthly estimate for the entire characterization factor. It would make more sense, as Gracey & Verones (2016) argue in their review, to only change the timestamp of the fate factor. This is due to the effect factor becoming more uncertain when attributed to monthly flow variation, since the SDR will show a large variation of species over the year.

Weighting

Weighting is done by multiplying the characterization factor with the ratio of energy production of the specific river with the total amount of energy produced from hydropower in Norway. This is done to ensure that the impacts of energy production in each respective river, is not overestimated, and that the rivers producing large amounts of energy are emphasized.

The use of average yearly energy production does not take into account the variability of seasonal energy production. Since potential impacts are larger during periods of low discharge, smaller rivers will gain superficially high impacts due to most of the energy production occurring in months with sufficient flows. Since the association assumed here is that energy production leads to water use, and this water use generates an impact on the fish species in the river, the weighting factor also aims at reflecting the conditions in each individual river relative to the total amount of energy being produced.

Species Count

Species counts where performed for a total of 42 rivers, these rivers where selected to represent a distribution of rivers stretching from the southern to the northern coast. The data was gathered using the publicly available database and map service Artsdatabanken (2015). Species counts where only performed for freshwater fish species, there exists data for macro-invertebrates as well, however, acquiring these data was constrained by available time. In order to count the species we maneuver through the map manually, counting each observation as far back as 1993. Species observation before 1993 are not included due to greater uncertainty surrounding the probability of the species still inhabiting the river. The rivers were split into 21 rivers representing rivers of none-to-low development of hydropower, and 21 rivers of medium-to-high development of hydropower. This classification was made in order to test if the species density of rivers with large development of hydropower was different from rivers with low development of hydropower, see section 2.2 on species discharge relationship development. The classification are showed in table 1.

Table 1: Classification used to characterize the scale of hydropower developed in a given river.

Criteria for classification		
Scale of hydropower	Numer of Hydro plants	Energy production
None - Low	0 to 1	0 to < 5,00E+08 kWh/y
Medium	1 to 5	5,00E+08 to < 1,00E+09 kWh/y
High	5 and upwards	1,00E+09 kWh/y and upwards

The data collected on species and discharge rates was used to calculate species discharge relationships (SDR) for Norway.

Watersheds and Norwegian energy production

Information on the Norwegian watersheds was collected from the European Environmental Agency and the publicly available "Water exploitation index for river basin districts" (EEA, 2016). Table 2 shows the counties included in the analysis within each watershed. Data on Norwegian energy production and use was collected from the Central Statistical Agency (SSB), this data was used to assess the share of energy production covered in this analysis.

Table 2: Shows the different counties represented in the EEA watershed categorization of Norway (Source: EEA, 2016)

County	Watershed
Finnmark	Finnmark
Troms	Troms
Nordland	Nordland
Nord-Trøndelag Sør-Trøndelag	Trøndelag
Møre og Romsdal	Møre og Romsdal
Sogn of Fjordane Hordaland	West
Rogaland Vest-Agder Aust-Agder	SE South West
Telemark Vestfold Buskerud	West Bay
Østfold	Glomma

Life cycle inventory data

LCI data on Norwegian hydropower stations was collected using environmental product declarations (EPDs) of Trollheim power station in the River Surna (Østfoldforskning, 2007), and E4 power station in the River Drammensvassdraget (Østfoldforskning, 2012). These EDPs show the material use and emissions generated per kWh of energy production from the power stations. These data were used in order to assess the usefulness of the characterization factor (CF) developed here.

The EPDs do not estimate species impacts, therefore we use the general framework for calculating ecosystem impacts by using the endpoint characterizations by ReCiPe (Goedkoop et al. 2013). This endpoint characterization associate emissions of GHGs to an increase in temperature, and further multiplying with the damage factor this increase in temperature has on terrestrial and freshwater species. This estimates the potential fraction of lost species due to greenhouse gas (GHG) generation at the hydropower plant, which can be converted to an absolute measure of species loss by multiplying with the number of affected species. We add our CF to the water use the power plants contribute to, in order to compare the share of impacts by the different byproducts of hydropower (namely GHG emissions and water use). It is important to assess if the impacts generated by our CF makes sense in comparison to the impacts generated by GHG emissions. In order to calculate the number of fish species in the River Surna and Drammensvassdraget, we use the SDR model equations (section 3.1, table 3), inserting the specific discharge of the rivers as x. Calculating the absolute species loss was done using species estimates based on Mora et al. (2011) for terrestrial and freshwater species.

2.2 Species-Discharge-Relationship and Connectivity index

Species-Discharge-Relationship for Norway

A Species-Discharge-Relationship (SDR) was calculated by curve fitting the relationship between the discharge rates of a given river, and the species count of the same river. Freshwater fish counts and river discharge rates were taken from 42 rivers and several regression models were tested in order to explain the relationship between these parameters. We tested the Weibull function from Tendall et al. (2014) and the power function used by Hanafiah et al. (2011), other function were also assessed in order to find the best fit to the data. For a river for instance, we would expect that increasing the river discharge to the point where there are no rivers in Norway (or the world) reflecting such a large discharge rate, will no longer increase the species density. If we want to be able to assess how freshwater fish are affected by water use, we need to know how the species density is related to the marginal change in discharge rates. This is what the SDR seeks to address.

The rivers were split into two categories in order to assess the difference in species density due to hydropower development. Thus SDRs were developed for 21 rivers in the category none to low hydropower development, and 21 rivers in the medium to high category, and one SDR model including all 42 rivers. This was done using MatLab R2014a, section 2 of the appendix provides the MatLab script used to curve fit the species and discharge data. The results are shown in section 3.1.

Connectivity index

The HCIU (Habitat Connectivity Index Upstream) gives an index of the cumulative probability of a flow of some kind to move from zone to zone within an entire river network (McKay et al. 2013). The HCIU represents the flow of upstream migrating fish species, with a value ranging from 0 to 1 (0 meaning no passabillity between the nodes in the network and 1 meaning complete passabillity). In order to adopt this to a LCIA framework, we first use the HCIU as an index for connectivity:

$$HCIU = \frac{\sum H_{Accessible}}{\sum H_{Total}} \tag{1}$$

Where H denotes number of habitats. Here we count total habits by counting the total amount of upstream habitats in the river which will amount to n nodes +1. Accessible habitats are calculated by cumulative score of the passabillity between the various habitat regions in the network. We then subtract this index from 1, in order to get the fraction of habitat that is on the average unreachable by migrating species.

$$\mathrm{Index} = 1 - \mathrm{HCIU} \tag{2}$$

We can illustrate the concept by showing how the river network was conceptualized in the project work (Mattson, 2015). Creating the network illustration was done by using data provided by the NVE GIS database (atlas.nve.no) of the River Orkla, combined with the salmon migration maps from the River Orkla, provided by the Environmental Agency (Miljødirektoratet, 2015).

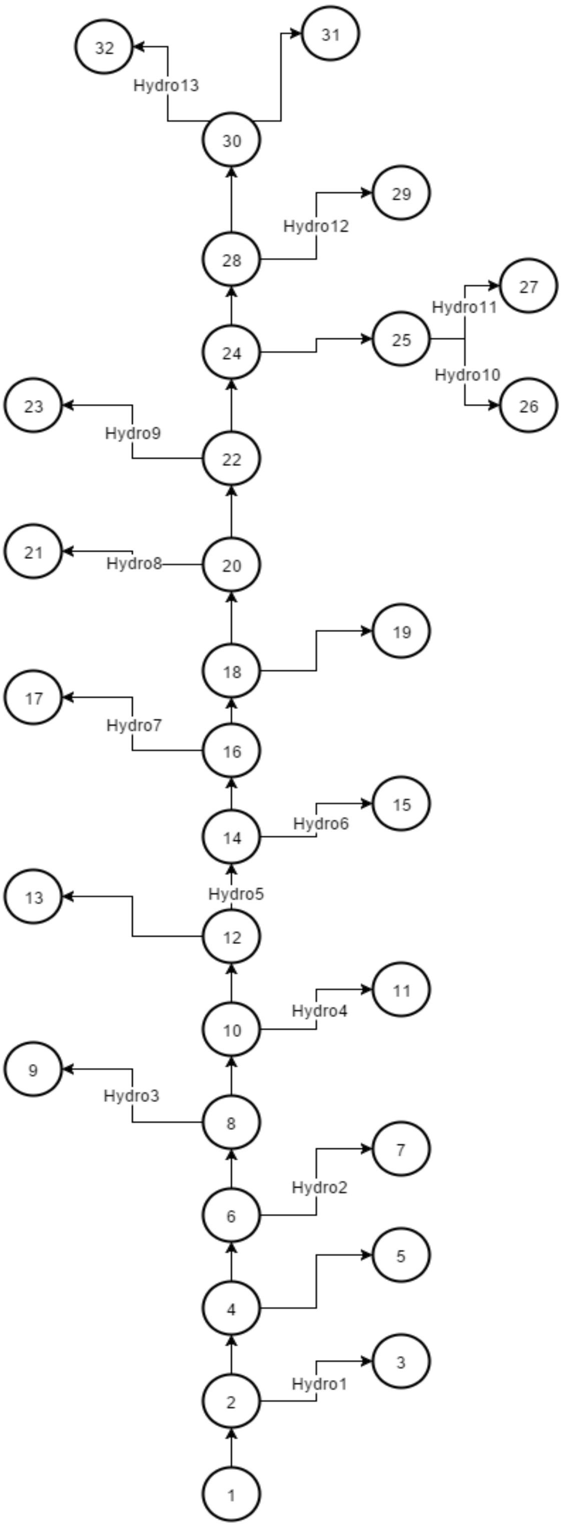

Figure 6: The River Orkla and the multiple nodes that fish can migrate to, starting from the river mouth (node 1), and ending at the node representing the habitat zone where fish migration has been identified to cease (node 31). Each hydropower plant in the river network, where it blocks migration to some degree has been assigned a passability estimate.

The starting node (Node 1) is the mouth of the rivers and illustrates the paths possible for upstream migration by fish within the river network. This was done in order to get a clearer overview of the migration options of the fish within each river, and to guide the collection of relevant data for calculating the HCIU.

In order to check the relationship between the HCIU and the hydropower development in a given river, 35 HCIU indexes were collected using the framework above, and assessed in connection to the yearly energy production of the respective rivers. Table 3 in the Appendix show the values used. Various statistical models were tested in order to check the relationship between energy production and the value of the HCIU. Section 3.1 display the results of this investigation. The relationship between the HCIU and the impacts calculated for the rivers was also investigated with the same approach.

2.3 Characterization factor

In order to make water use by hydropower fit into a life cycle assessment, we need it to fit the relationship between the use of water and the resulting impacts on species, into a life cycle inventory framework. In order for our results to serve as a basis for comparison between different technologies, the unit we want to work towards is [Species*yr/kWh] or [PDF*yr/kWh], in order for technologies to be compared on a per kWh basis. The standard form of characterizing impacts in LCIA is shown in equation (3):

$$CF = FF \bullet EF \qquad (3)$$

Here we multiply a fate factor (FF) with an effect factor (EF) in order to gain a characterization factor that quantifies the potential impact per unit of output (Pennington et al. 2004). We can in the case of discharge models use the fate factor (FF) by Hanafiah et al. (2011) and Tendall et al. (2014), which is unitless, and multiply this with an estimate of water used per KWh electricity produced. This latter part produces an efficiency score which we denote α and has the units [m^3/KWh]:

$$FF = \frac{dQ}{dW} \bullet \alpha \qquad (4)$$

In this equation dQ is the marginal change in discharge [m^3/y], due to water use, dW is the marginal water consumption rate [m^3/y].The fate factor accounts for the water used, but not necessarily consumed from the river, it simply tells us how much water is needed to produce 1

KWh of electricity. The data needed for these estimates would be per power plant based measures of efficiency. NVE provide estimates that reflect these parameters, as shown in section 2.1.

The fate factor is multiplied with the effect factor in order to relate the use of water to the potential effect this incurs on the species in a given river system. The effect factor (EF) takes the form:

$$EF = \frac{dSDR}{Q} \tag{5}$$

dSDR is the derivative of the SDR function, which we use in order to find the species loss per unit change of discharge. Q is the annual discharge rate of the river. This effect factor is slightly different from the one used by Hanafiah et al (2011). Here we solve the SDR function analytically, while they solved it numerically. The equation bellow is the analytical solution to SDR power function.

$$SDR = a \bullet x^{b} \tag{6}$$

$$dSDR = (b \bullet a) \bullet x^{(b-1)} \tag{7}$$

Here *a and b* are model coefficients produced by the regression model, where *x* is the discharge rate [m3/s] of the river in question. In order to perform a regression model of this kind, one needs the average discharge rates of multiple rivers, accompanied with an estimate of the number of species present in the rivers (Xenopoulos and Lodge, 2006).

This equation tells us how many species we would expect to find when we move one unit up or down in discharge, and is a result of what we set out to find in section 2.2. If we were to take one m^3 of water out of the cross-section of the river this dSDR would provide us with the number of species being impacted by that water use, however, this use of water needs to be distributed to the entire volume of the river. We therefore use Q (annual discharge rate) to act as an approximation for the river volume, or the amount of water running through the cross-section within a year.

Multiplying these factors provides us with the characterization factor, which is represented in unit form here:

$$CF = \frac{[m^3]}{[KWh]} \bullet \frac{[Species \bullet y]}{[m^3]} = \frac{[Species \bullet y]}{[KWh]} \qquad (8)$$

For calculate the final impacts over a year of energy production, we can multiply the CF with the energy production within a year:

$$Species\ \mathrm{Impact} = \mathrm{CF} \bullet \mathrm{KWh/\ year} \qquad (9)$$

This gives us the impacts in terms of species lost within the specific river. We also assume here that the water used for energy production, is what drives a fraction of the discharge rates in the river system, since these streams usually are strongly regulated for energy production. In order to calculate the average impacts occurring within a given county or watershed, the yearly impacts of the rivers in this geographical region is calculated, and an average of these impact scores are taken to reflect the general impact on fish species.

In order to account for the connectivity loss hydropower plants contribute to within river systems, we add the index (equation 2) for connectivity to the characterization factor. The index works as a weighting factor for the potential impacts, were it lowers the impacts in proportion to the amount of connectivity that is lost for migratory fish. The CF including the connectivity index (Index) and the energy weighting explained in section 2.1, takes the final form:

$$CF = FF \bullet EF \bullet Index \bullet Weight \qquad (10)$$

Since the Index and weight are both unitless, we still retain our unit of [Species*yr/kWh]. When multiplied with a specific rivers yearly energy production, the final impact score reflect an absolute number of species lost from that river as a function of water use for energy production.

3. Results

Results will be split in four sections, covering the parameters used in order to construct the Characterization factor (CF) first, then showing the impacts generated by the CF, followed by a comparison of impacts on a kWh basis applying the CF to LCI data from two hydropower plants. Lastly the connectivity index (equation 2) is added to the CF as a weighting.

3.1 Species discharge relation and HCIU

Figures 7, 8, and 9 shows the various species discharge relationships calculated for the 42 rivers where freshwater fish species data was collected.

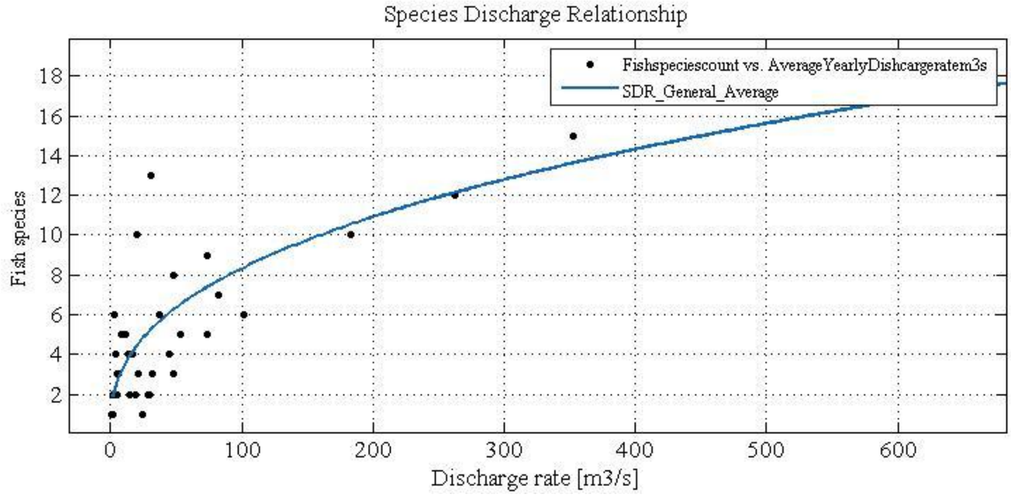

Figure 7: Power function displaying the relationship between discharge rates and species in all of the 42 rivers, adjusted $R^2 = 0.64$.

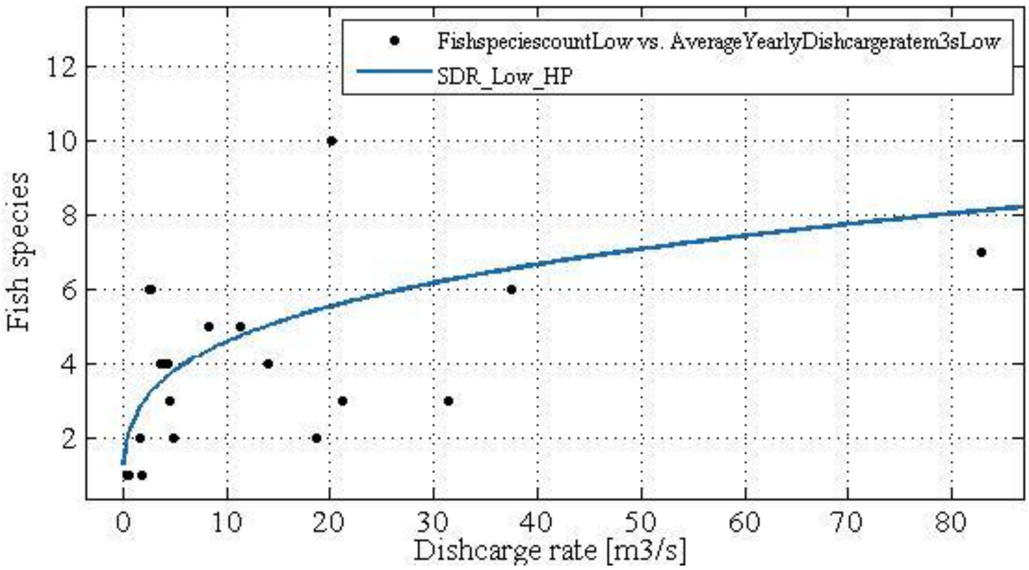

Figure 8:Power function of the rivers classified with low hydropower development, Adjusted $R^2 = 0.28$.

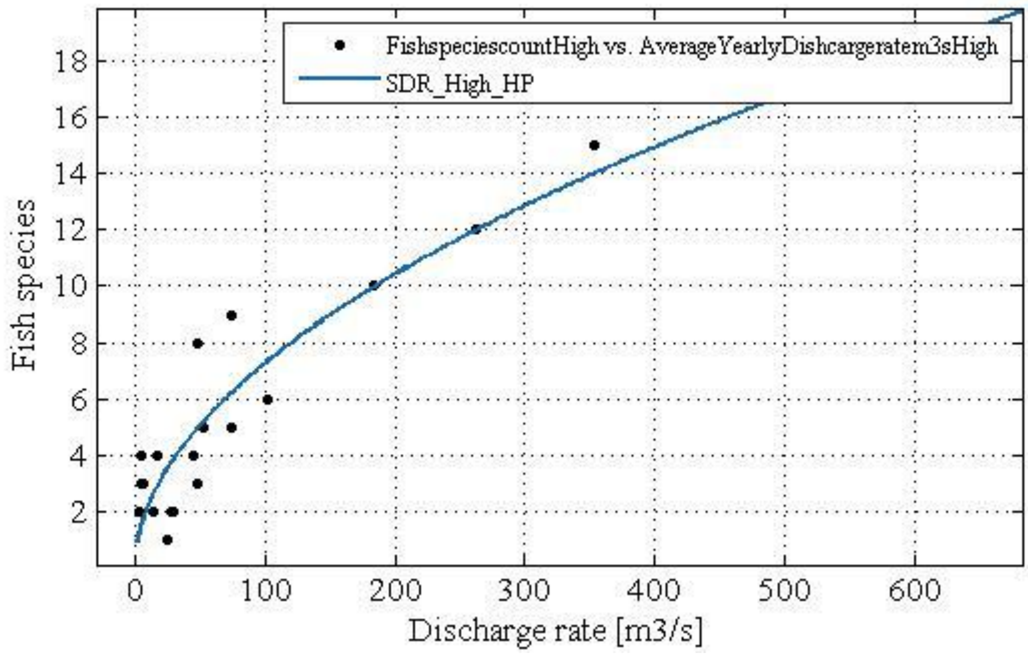

Figure 9: Power function of the rivers classified as medium to high impact, Adjusted R^2 = 0.88.

Multiple regression models were fitted to the data, in all cases the power function was the best fit. The fit, reflected in the Adjusted R^2 measure, show that for the high hydropower development case, the fit is very good at R^2 = 0.88. In the none to low case the fit is much worse, with Adjusted R^2 at 0.28. As a consequence of this the average model reflecting all the rivers with all categories of hydropower development has an Adjusted R^2 at 0.64. This means that the relationship between the variables observed in the regression model, discharge rates on the x-axis and number of fish species on the y-axis, predict 64% of the variation observed in the data. In other words, 64% of the change in species density is due to a change in discharge rate.

Table 3 shows the model equations. These equations where used to calculate the effect factor for all 97 rivers included in the analysis. In the impact assessment the model representing all rivers was used, since this model uses the most data and would be more likely to represent the average species counts in Norwegian rivers.

Table 3: Model equations for SDRs

Class of river	Model equation	First Derivative of the equation
Low hydropower (n =21)	$y = 2.475x^{0.2686}$	$y' = 0.668x^{-0.73}$
High hydropower (n=21)	$y = 0.652x^{0.523}$	$y' = 0.34x^{-0.477}$
All hydropower (n=42)	$y = 1.374x^{0.39}$	$y' = 0.536x^{-0.61}$

These model equations show that the rivers with low amounts of hydropower provide the largest density of fish species, however, the uncertainty of this model is much greater than the high hydropower model. The model equation of all the rivers estimate a moderate species density compared to the low and high hydropower model equations.

To illustrate the usefulness of the SDRs, the impacts of changing discharge rates over time can be performed. We identified three rivers in our data set that show a marked change in discharge rates after the development of hydropower. The River Mossa is one of these rivers (figure 10).

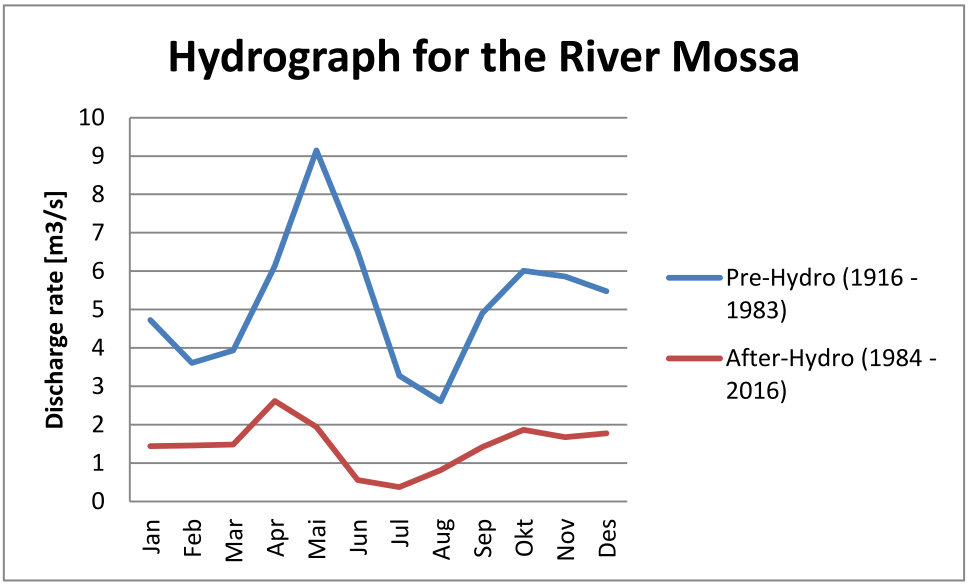

Figure 10: hydrograph for the River Mossa, showing marked decrease in river discharge rates after the development of hydropower.

In order to assess the species loss in this river due to decreasing discharge rates, the SDR function for Low hydropower development was applied to the discharge rate before and after the decrease, the resulting difference in species is shown in table 4.

Table 4:Loss of freshwater fish species due to decreasing discharge rates in the River Mossa.

River Mossa	Impacts	Units
Species before Hydropower	3,850204328	[Species]
Species after Hydropower	2,734951321	[Species]
Loss of species	1,115253007	[Species]
% decrease	29 %	

We can see that based on the SDR function we would expect a 29% decrease in freshwater fish species due to reducing the discharge of the river.

In order to assess the usefulness of the HCIU for making the connectivity index, the HCIU for 35 rivers were calculated. These estimates were plotted with the energy production for the respective rivers in order to check for a relationship between the amount of energy production and the scores of the HCIU.

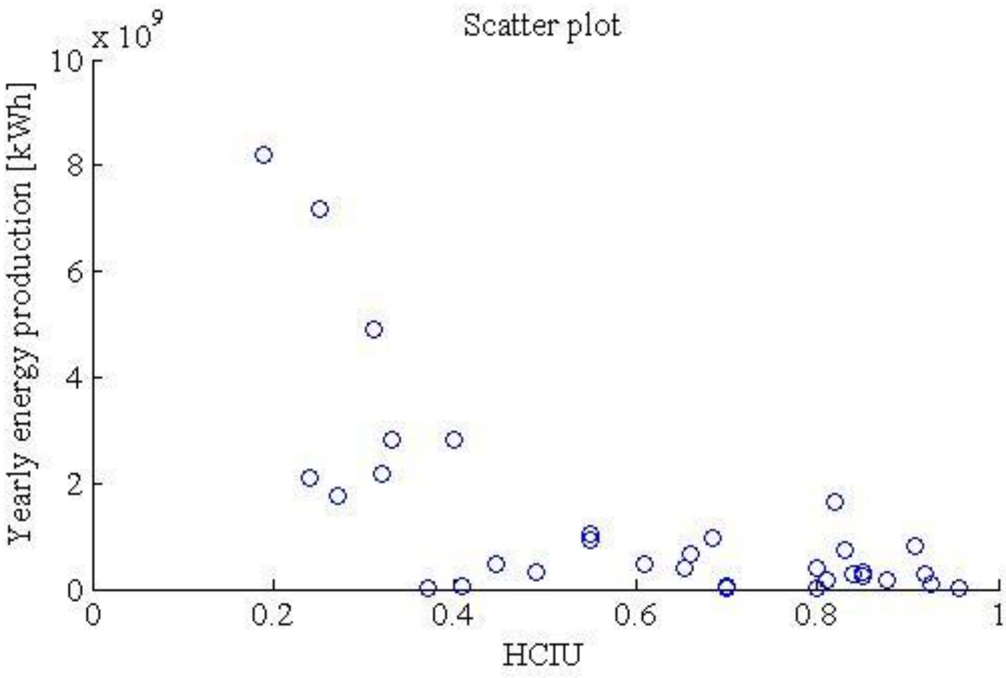

Figure 11: Scatter plot showing the data gathered on HCIU and energy production in 35 rivers, y-axis is in kWh, x - axis represents the index from 0 to 1, where 1 means fully passable river network and 0 means impassable.

The tendency observed from the scatter plot is that low HCIU scores predict larger energy production within the river system.

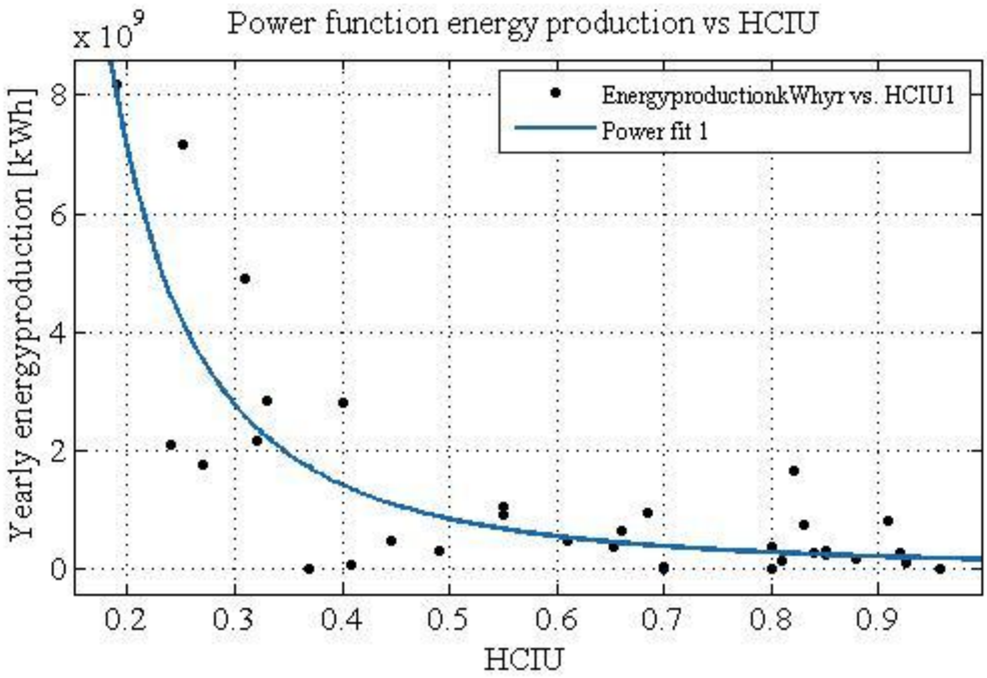

Figure 12: Power function relating energy production of the river system (y) to the connectivity index (x). Adjusted R^2 = 0.71, based on 36 rivers

When curve fitting the data, we find that the power function explain the relationship fairly well (Adj-R^2 = 0.71). The model equation for this relationship takes the form $y= 0.0002558*x^{-1.816}$, were y is the yearly energy production and x is the value of the HCIU. In order to further investigate the relationship between the HCIU and the conditions of the river we plotted the HCIU with the yearly impacts calculated with the characterization factor (presented in section 3.2).

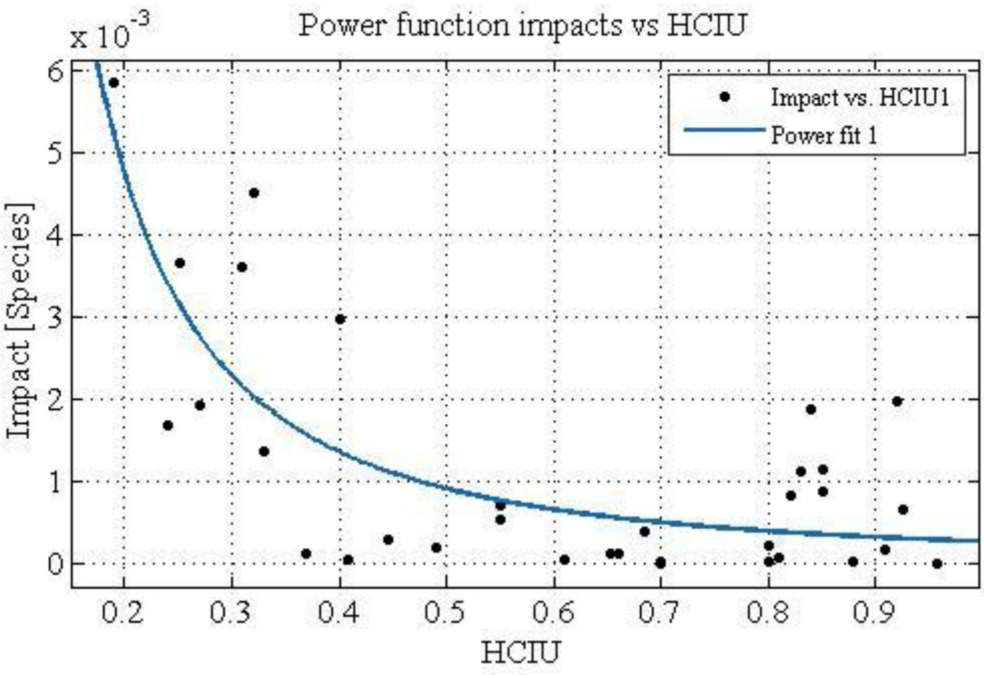

Figure 13: Power function relating the HCIU to the yearly impacts estimated using the CF (equation x), Adjusted $R^2 = 0.576$. Based on 35 rivers.

The relationship between the HCIU and the impacts show that there exists a correlation between the scores of the HCIU and the impacts of the respective rivers (fig. 13).

3.2 Characterization Factor and Impact on species

Section 1 of the appendix shows all the parameters used in order to calculate the potential impacts occurring in the Norwegian rivers used for hydropower production. Here we present the key results on a county and watershed level.

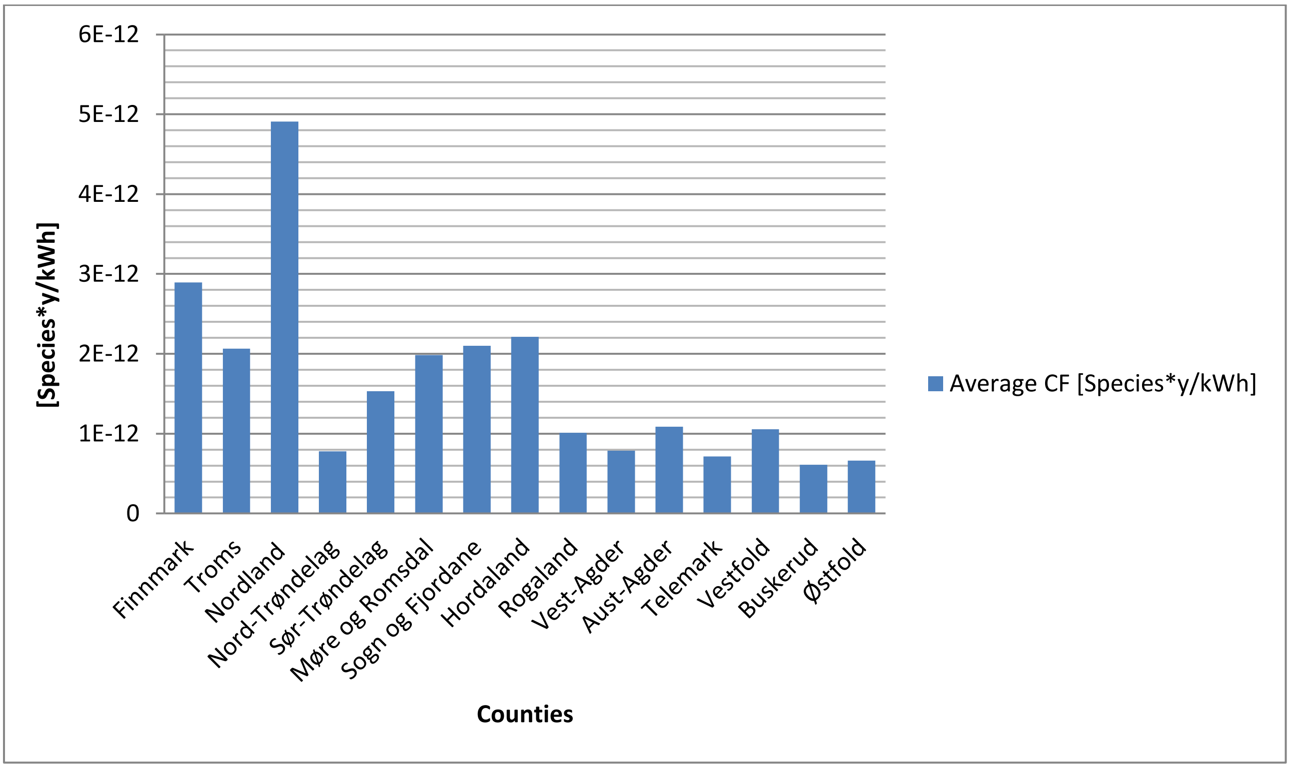

Figure 14: Average characterization factors of all counties included in the impact assessment of Norwegian hydropower.

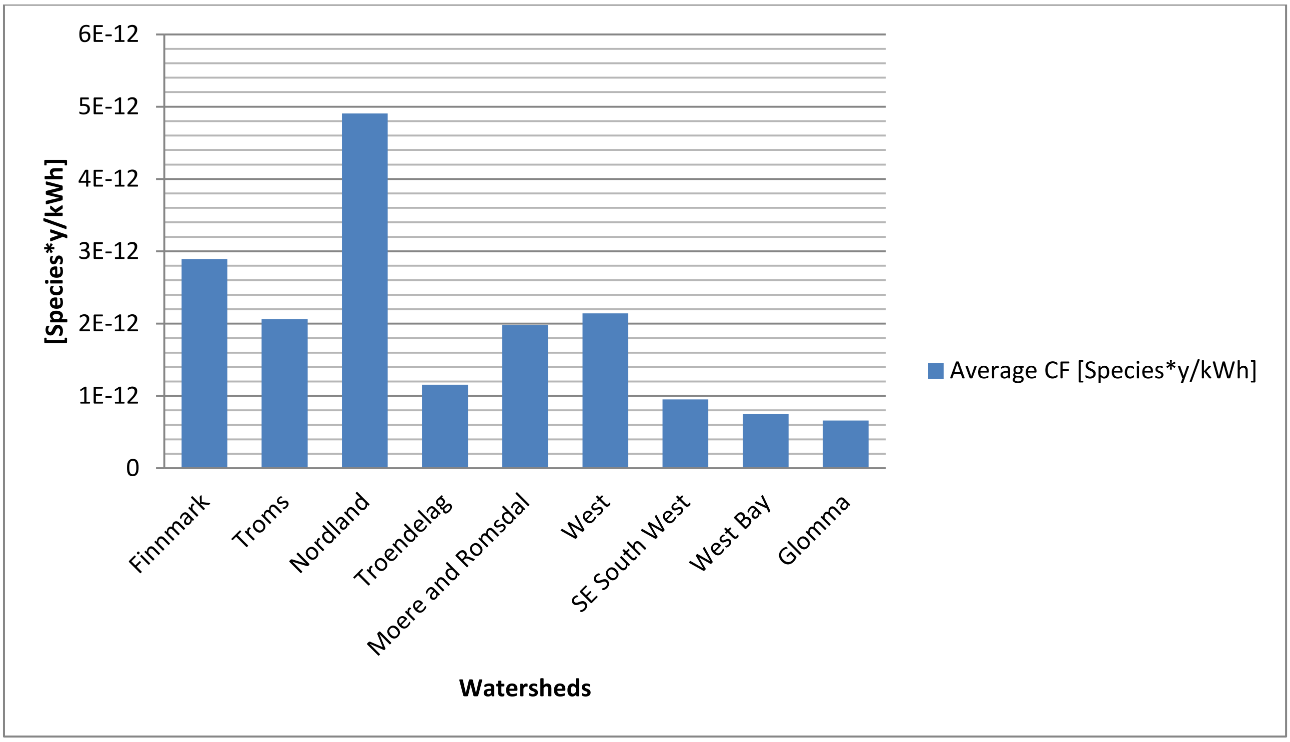

Figure 15: Average characterization factors of the watersheds included in the impact assessment of Norwegian hydropower.

The average CF is an indicator of the potential impacts one kWh of energy produced by hydropower within each county (fig. 14) and each watershed (fig. 15). The potential impacts are largest in the northern and western counties, with Nordland showing significantly higher potential impacts than the rest. It is important to note that the average CF should not be used to calculate impacts, its function is to show a general impacts potential within each county and watershed. To calculate the impact, one needs to account for the specific CF of each river, since the energy production in different rivers can vary significantly.

Figure 16 and 17 highlight the average impacts occurring within a year, on a per county level (fig. 16) and per watershed level (fig. 17).

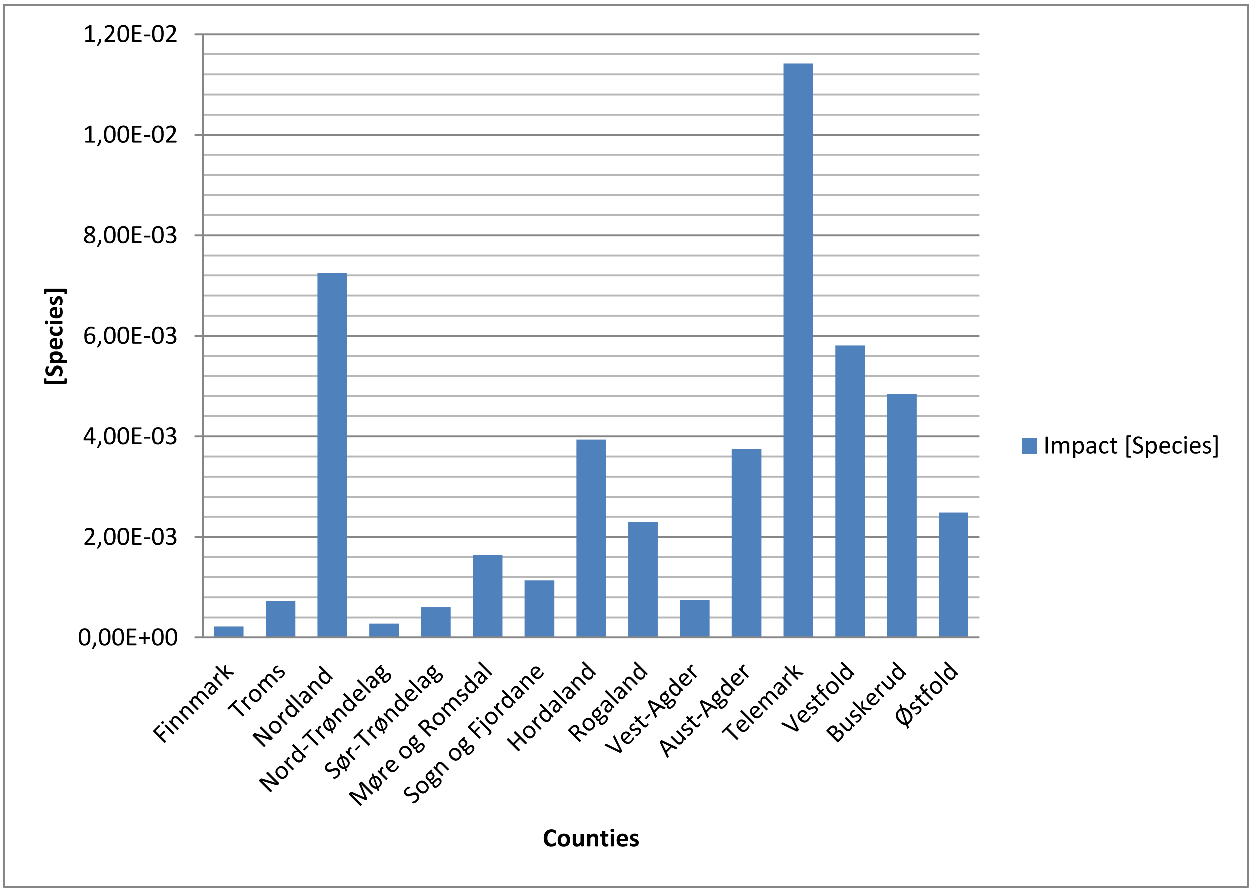

Figure 16: The y-axis shows the impacts on freshwater fish species, the x-axis show the county the impacts are originating from.

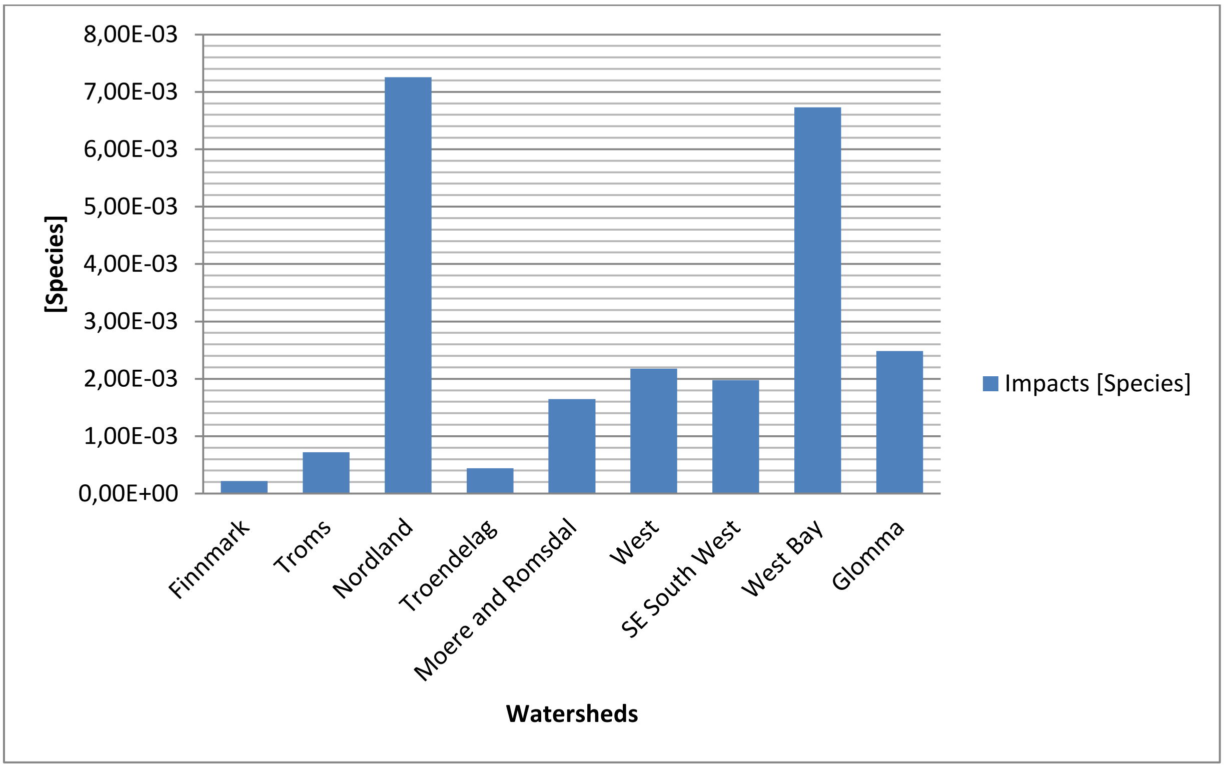

Figure 17: Y-axis shows species impacts per year, x-axis shows the watersheds these impacts are originating from.

We see that the county of Nordland and the counties associated with the West Bay watershed are the once displaying the highest impact. Energy production in the Nordland watershed is responsible for 10% of the energy production covered in this analysis, and the West Bay watershed accounts for 26%. The share of impacts between these two regions show that Nordland accounts for 31% of the impacts, and the West Bay for about 28% of the average total impacts (fig. 18 and 19).

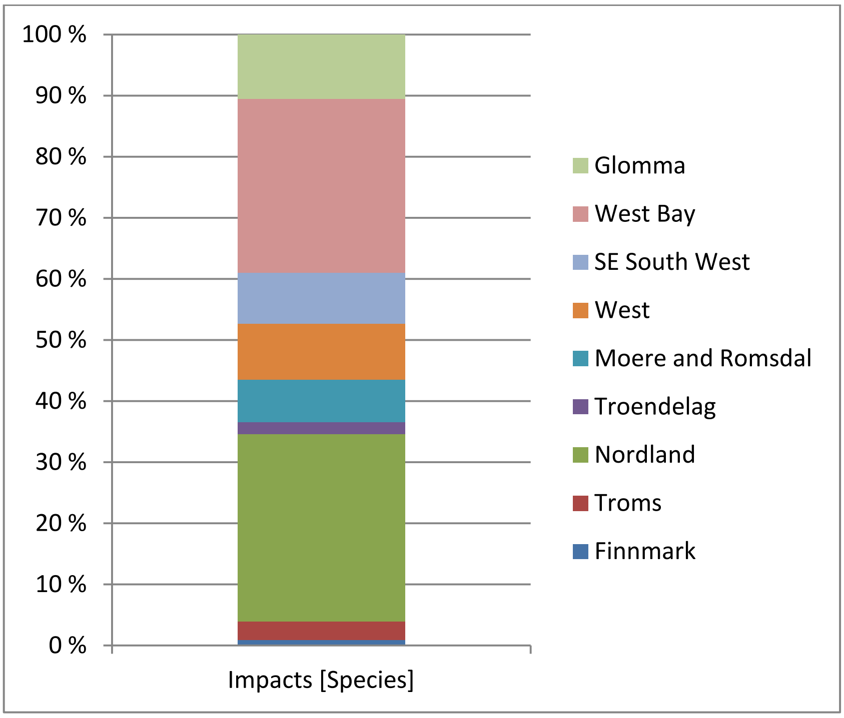

Figure 18: Share of total impact between the watersheds, Nordland and West Bay occupy the largest shares of the total impacts

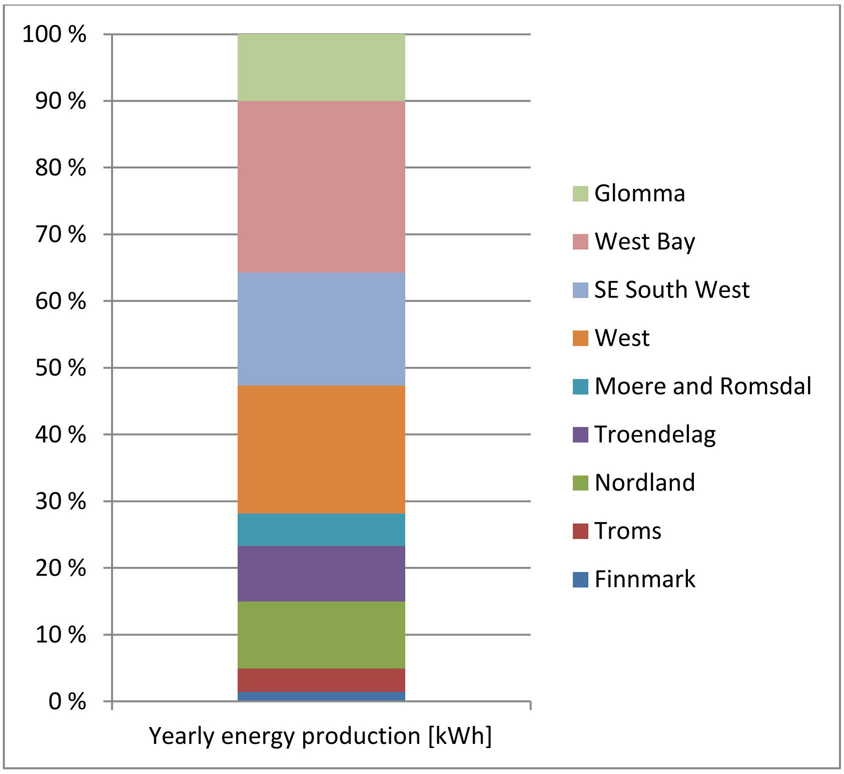

Figure 19: Share of total energy production between the watersheds.

Figures for county based energy and impact shares can be found in section 1 of the appendix. This analysis covers approximately 73 TWh of energy production from hydropower, based on the production levels of 2014 (NVE, 2014). In 2014 the electricity production from hydropower was at 142 TWh (SSB, 2015), which means that the share of energy production from hydropower covered in this analysis is at 51%. The analysis has an average county coverage of 65% for hydroelectricity generation. The reason for the higher % in covered counties, is that some counties were not included in the analysis due to having low hydropower development.

3.3 Comparison of impacts per kWh

Using the EPDs (Østlandsforskning, 2007; 2012) and the characterization factors for calculating impacts on terrestrial and freshwater species from greenhouse gasses (Goedkoop et al. 2013), we apply the characterization factor developed for the River Surna and Drammensvassdraget. Section 1 of the appendix gives the values used for calculating these impacts. In figure 20 we see the impacts on local freshwater species from GHG emissions per kWh of energy production, and the impacts of water use by hydropower per kWh.

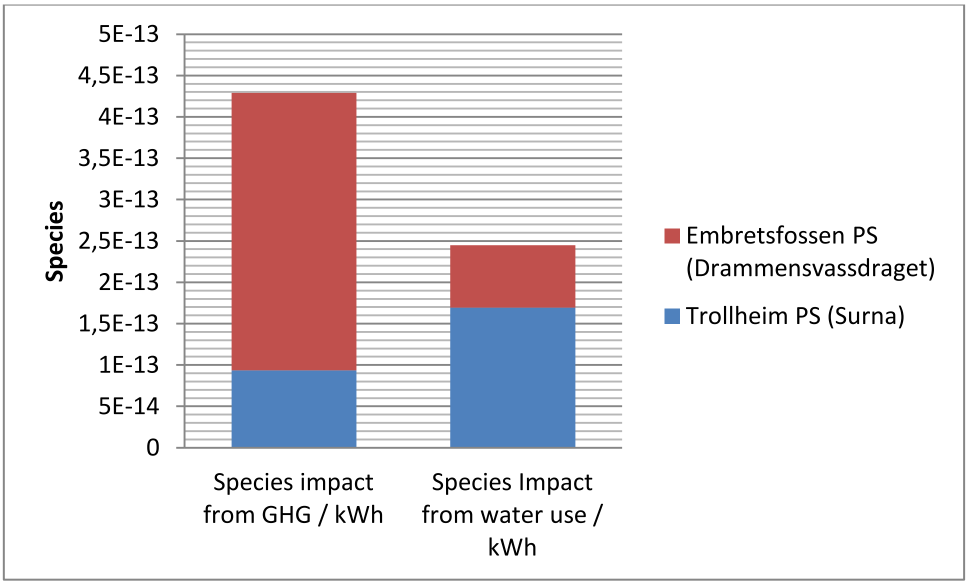

Figure 20: Local freshwater fish impacts from water use and GHG emissions per kWh of energy

We see that the impacts generated by water use in Trollheim power station is higher than the impacts generated by GHG emissions, but for the E4 (Embretsfossen) power station the impacts of GHG emissions are significantly higher than the water use impacts. This is due to the GHG emissions from the E4 power station being one order of magnitude larger than the emissions from Trollheim (Østfoldforskning, 2012).

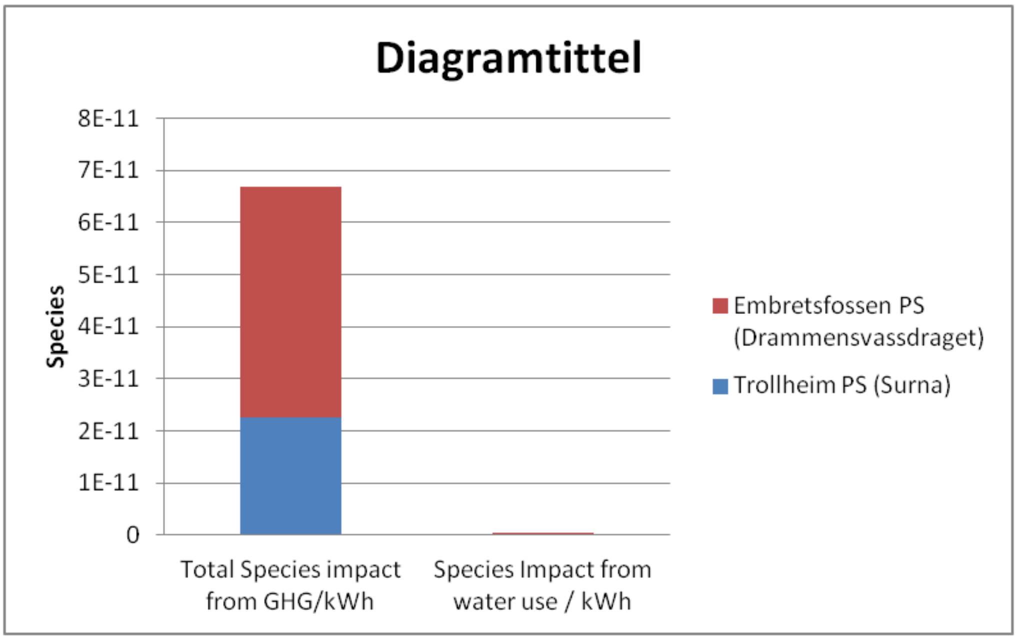

Figure 21: Global species impacts from water use and GHG emissions per kWh

Figure 21 show that the global impacts generated by the GHG emissions per kWh is much higher than the impacts of the water use. This is due to the GHG emissions impacting a fraction of the global terrestrial and freshwater species, which means that the absolute number of species impacted by GHG emissions are significantly higher than the local freshwater species.

3.4 Inclusion of connectivity index

In order to see how the impacts change due to inclusion of the connectivity index, we apply the index (eq. 2) to the yearly impacts of the rivers where the HCIU was calculated.

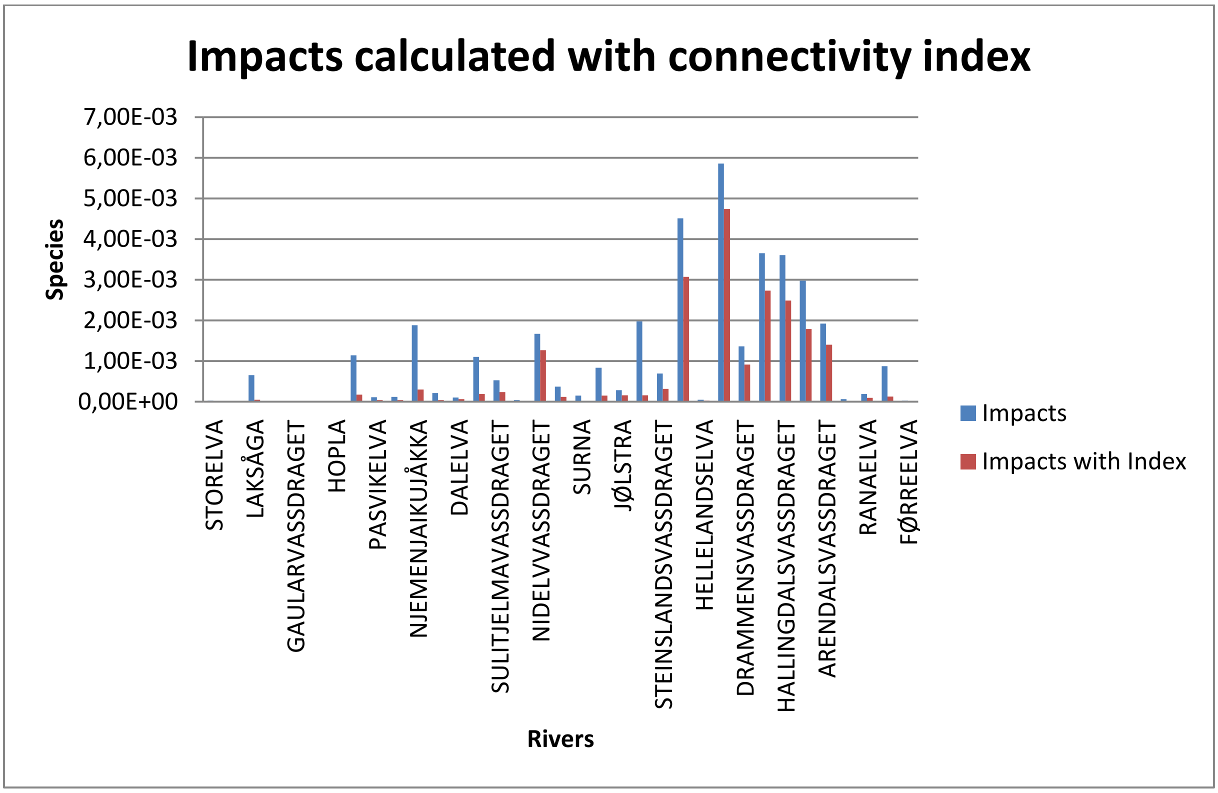

Figure 22: The red bar shows the change due to inclusion of the connectivity index, rivers with a high index show the lowest decrease in impacts.

Including the connectivity index lowers all impacts, but it does so in proportion to the connectivity of the river. We see that rivers with high index scores remain high on impacts compared to rivers with very low index scores. Values used to calculate these new impacts are given in section 1 of the appendix (table 3).

4. Discussion

4.1 Use of Species-Discharge-Relationship

The average SDR estimate in figure 7 is being used to calculate the effect factor for the impact assessment, when we compare the fit of this model (adjuster R^2 = 0.64) with the freshwater fish SDR power functions made by Tendall et al. (2014) we see that the fit in our model is better than the average fit of the models used by Tendall et al. (average adjusted R^2 = 0.49). The power function also has the theoretical advantage of having clear cut-off points, i.e. starting at 0, and displaying a gradual decreasing relationship. The flattening of the effects towards higher discharge rates indicate that species densities does not correlate with discharge rates for this region. When we classify the data to the two groups reflecting hydropower development, the R^2 changes quite drastically to 0.88 for rivers with high hydropower development and 0.28 for rivers with low hydropower development.

A possible explanation for the large difference between the estimates, is that the annual flow variation in the rivers with low levels of hydropower is much greater than the variation in the rivers with high levels of hydropower. Rivers with large amounts of hydropower usually have strongly regulated flows in order to optimize the availability for energy production (Poff et al. 2007). This is not the case for non-regulated rivers, and they usually have far greater variation in the discharge rates. Another factor that could bias the results are the measurements of species occurrences in the rivers with little hydropower. Rivers with more hydropower are usually larger and the interest of conservation more present in these rivers, species counts might simply be more frequent in rivers with more hydropower.

In the paper by Hanafiah et al. (2011) they exclude rivers above 42˚ latitude, stating that the SDRs for these rivers would be weaker than the SDRs for rivers below 42˚ latitude due to fewer species per unit discharge. However, our results show that SDRs can be calculated for rivers above this latitude, all the rivers included in the model are above 42˚. Using these SDRs to calculate the marginal loss of species due to marginal changes to discharge provides a relatively robust estimate of potential impact. This means that collecting SDRs for Northern Europe, Northern America and Canada can be done, and used in order to introduce regional specificity to various impact assessments. It would however, be important to increase the robustness of the model reflecting rivers with natural flow regime and low development of hydropower. This would be very useful for addressing the potential impact of hydropower in these rivers, since the average SDR estimate is likely underestimating the species density. It is also important to point out that if the SDRs developed by Hanafiah et al. (2011), was used to calculate the species density of Norwegian rivers, we would greatly overestimate the number of fish species. This exemplifies the importance regionalized data when performing impact assessments.

To illustrate how SDRs of rivers with no hydropower development could be utilized, the Low-hydro SDR was applied to the River Mossa, in the county of Sør-Trøndelag, where hydrological data extending over a 30 year period was available before and after the construction of a hydropower plant in 1984 (Rosvold, 2010). In table 4 we see that the decrease in fish species at approximately 29% can be attributed to the marked decrease in discharge rate in the period after hydropower development. If one were to introduce hydropower into a river system, estimating the potential decrease of discharge this introduction could contribute to, and applying the SDR function to the old and new discharge, would provide an estimate of lost fish species. More robust SDR models for non impacted rivers would be a good tool for analyzing the consequences of developing a site for hydropower. However, if one wishes to investigate the impacts occurring in rivers with already existing hydropower, using SDRs from a variety of impacted rivers would provide an average estimate of species density. An argument against using average species density instead of low impacted species density estimates, is that the low impacted SDR would provide a worst case scenario of impacted species. Availability of data could be an issue when calculating low impact SDRs, in which case using average SDRs is the only viable option.

4.2 Impact assessment of Norwegian hydropower

In order to explain the yearly impacts that we calculate, we need to understand how the characterization factor (CF) accounts for the vulnerability of the rivers within each watershed and county. We see from figure 15 that the average CF for Nordland is significantly higher than the CFs for the other watersheds. This is due to the sensitivity of the effect factor (equation 5). The effect factor is heavily dependent on the discharge of the rivers, small rivers will have lower species estimates and a higher concentration of species, this we can observe from the SDR (figure 7). With a high concentration of species per unit of discharge, and low discharge rate, the impact of using one unit of discharge is much larger. The sensitivity of organisms inhabiting rivers with low flow magnitude to alteration in flows is well documented (Poff et al. 2010, Ashton, 2012). The discharge rates observed for the rivers of Nordland is on average much lower than the other counties, a table of the average discharge rates per county has been added to section 1 of the appendix (Table 2). Nordland also has the most rivers that have been included in this analysis, (19 out of 97). Due to these factors it would make sense that the potential for impacts is much higher in Nordland.

The CF only accounts for potential impacts as a function of the Fate factor and effect factor, the severity of the actual impacts occurring is directly related to the amount of energy being produced in each specific river. It is important that when calculating the impacts, one does not use the average CF, since this could potentially over- or underestimate the results substantially. Also the energy per year value needs to originate from the river that the CF is specific to, this is again do to the significance of the effect factor. Only energy production originating from the river in question should be assessed. The impacts from the 97 rivers are shown in figure 17, and here we see that some watersheds with relatively low CFs scores very high on fish species impacts. The impacts are still largest in the Nordland watershed. The introduction of the weight to the CF lowers the impacts from this watershed, due to Nordland only accounting for 10% of the energy production (Figure 19), impacts are however, quite large comparatively. When we sum the impacts scores from all the rivers, we gain a species loss estimate of 0.14 species each year.

In order to assess the validity of these results, the impacts were cross-referenced with the nature index for salmon produced by the Norwegian Environmental Agency for 2014 (Miljødirektoratet, 2016). The index is shown in figure 23.

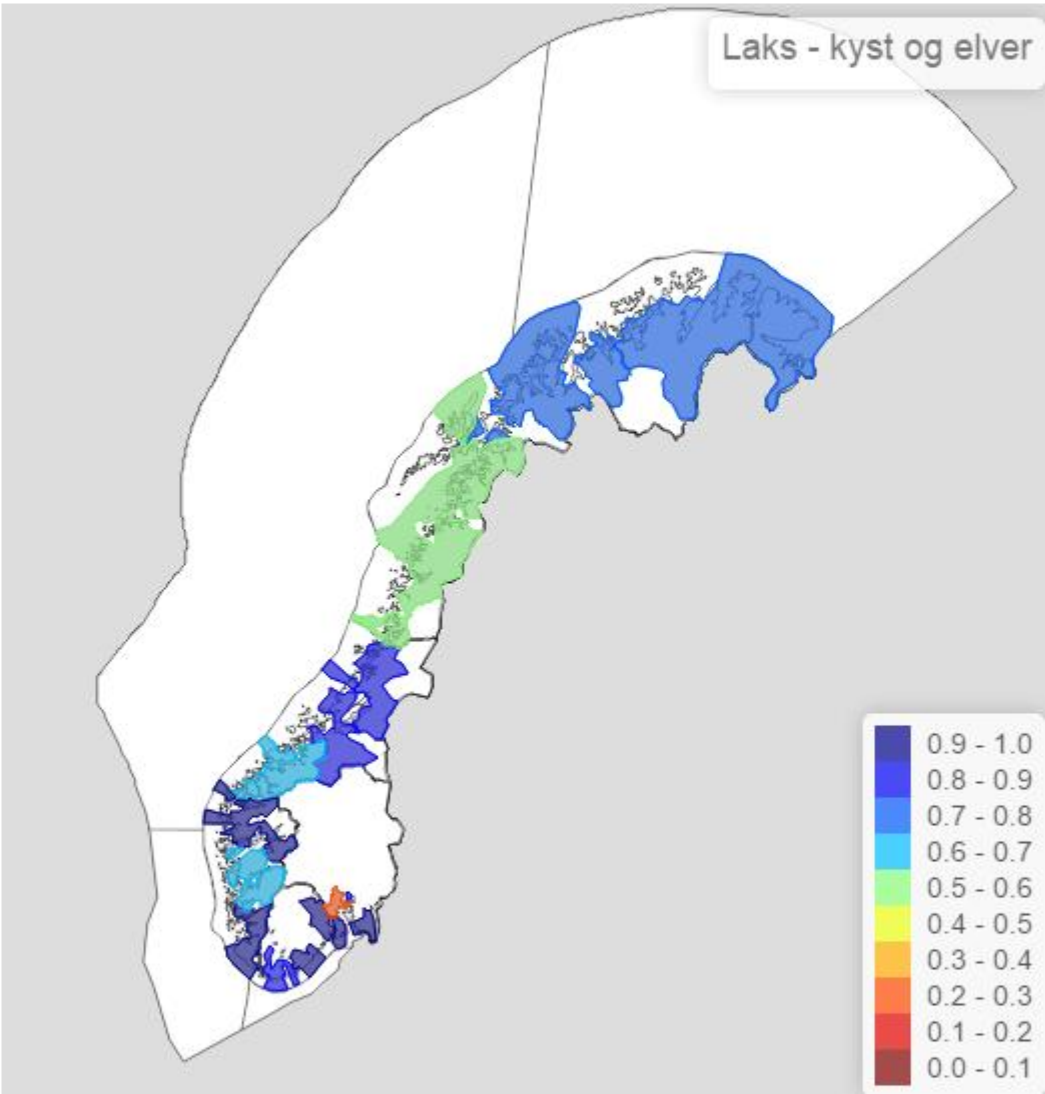

Figure 23: Nature index for salmon populations in the Norwegian coast and rivers, the index relates observed spawning rates compared to expected/desired spawning rates in 2014 (Miljødirektoratet, 2016).

Comparing the spatial distribution of the index to the distribution of impacts by our CF, we see that the watersheds that have the lowest salmon population growth are Nordland and the West Bay watershed. It is important to note that the index does not attribute the low growth rates of salmon population to any specific mechanism, it only gives a general overview of the condition of multiple populations of salmon. These results are encouraging, since they show that the impact assessment of Norwegian hydropower display the most significant impacts on freshwater fish in the same regions that fish populations are growing slowly or are in decline.

In an effort to further validate our results, we compare the cumulative impact generated by energy production each year from the rivers, to the predicted freshwater fish extinction rates for North America in the period 1900-2010. Burkhead (2012) estimate that within the last 100 years 52 taxa of freshwater fish have been driven to extinction in North America, at the rate of 0.52 fish species per year. Our impacts indicate 0.14 fish species lost each year. These two cases are not directly comparable, however, we see that the impacts from our model does not grossly overestimate the loss of species. This we will discuss further in section 4.3.

Although the orders of magnitude of the results are not of immediate concern, they are however, overestimations. This is due to the potential for double counting in the fate factor. In rivers with multiple power plants, the water use of the power plant furthest upstream generate an effect on the water use in the river, some of this water (unless the water is consumed from the river) will be used by the next power plant downstream. Thus the water use is being accounted for, multiple steps down the river. Lobet et al. (2013) show how integrating downstream cascade effects of water consumption into LCA can account for impacts occurring at different location in a river basin, modifications to this approach could be utilized in order to reduce the fate factor. The issue in respect to hydropower is that water is often not directly consumed (Bakken et al. 2013).

4.3 Comparison of impacts per kWh

In order to further assess the applicability of the characterization factor to perform impacts assessments in LCA, we added the CF to two hydropower plants life cycle inventory data obtained from Environmental Product Declarations (EPD) (Østfoldforskning, 2007; 2012). In the results shown in section 3.3 we see that the impacts due to water use and greenhouse gas emissions per kWh produced at the two plants differ greatly on the global level. This is due to GHG emissions are integrated over a period of 100 years, and the fact that GHG emissions actually impact all living species to some extent due to temperature changes (Goedkoop et al. 2013). Water use on the other hand, occur only on the local level and without time integration.

Including the CF for assessing the impacts of a hydropower plant, focusing on the local species impacts from GHG emissions when comparing it to water use impacts would be best. Comparing global impacts of GHG to local impacts water use would only negate the local impacts. This is not to say that global GHG emissions are of no concern, these results do show that the impacts of GHG emissions are a serious issue. However, they also show that accounting for local impacts makes it possible to compare technologies that are invasive to the environment on the local level. For analysis of impacts on species of special concern, one could adapt the CF to the approach taken by Tendall et al. (2014), adding a threat and rarity weighting factor. This could be done when comparing the potential development of hydropower in different regions with low growth rates for salmon. For instance introducing a weighting factor increasing the impacts of freshwater fish when developing hydropower in Nordland, compared to hydropower development in Hordaland, where growth rates of salmon populations are stable (Miljødirektoratet, 2015).

The main message from this analysis is that the CF can be integrated with LCI data, due to the units being compatible with the standard for accounting materials and emissions per kWh of production (Rebitzer et al. 2004). However, the CF needs to be developed for the river basin under study, leading to sufficient regionalization being necessary for the fate and effect factor to provide meaningful results.

4.4 Inclusion of Connectivity Index

From section 3.1 we see that the HCIU score shows an inverse relationship to the yearly energy production of hydropower within a river basin (Adjusted $R^2 = 0.71$). Meaning that the lower the HCIU score is, the larger the energy production is likely to be. We also see that the relationship is lower (Adjusted $R^2 = 0.58$) for impacts and HCIU. Some of the rivers included in this model have relatively little hydropower, but score high on impacts due to being rivers with smaller discharge rates. The results does display a general tendency for rivers with low HCIU scores to have higher impacts.

Creating the HCIU is a very time consuming process, which is why only 35 rivers were included. This makes the potential use of a model equation for predicting the HCIU very appealing. However, every HCIU will be different due to the different topology of every river basin. There could exist a general relationship between the largest possible energy production and an average optimal for the HCIU, however, in order to conclude this, we would need a much larger sample of river basins and a more thorough index. Thus, we would not recommend using the model equation to estimate HCIU values as of this time.

A further weakness of the HCIU is that it only accounts for upstream connectivity, and is only relevant for fish species that migrate through large stretches of the river, such as salmon

(McKay et al. 2013). Furthermore, the probability of passing a barrier is based on the most optimistic estimates from Noonan et al. (2012), which is again most applicable to salmon. Creating the HCIU is however, a good starting point for assessing the integration of river connectivity indexes into impacts assessment.

We see that including the connectivity index (eq. 10) lowers the impacts of the respective rivers in proportion to the level of connectivity calculated using the HCIU estimates. This lowering of impact scores can help to mitigate the double counting of the fate factor, as well as representing the average connectivity characteristics, increasing the spatial resolution of the characterization factor. The connectivity index was not included in the impacts calculated in section 3.2 since this would effectively mean that we use a slightly different CF for the impacts calculated in a few rivers. Unless all rivers apply the connectivity index, the results will be much lower for the rivers with the index and the basis for comparison between a river with and a river without the index would be skewed in favor of the river with the index. Including the index weighs the CF in order to account for the average amount of the river volume that is unreachable to migrating fish.

4.5 Further research and framework

The most important parameter to improve in this model is the fate factor (FF). Two central steps should be further researched in this regard. First, assessing the results when variation in water use is accounted for with a monthly time horizon, and second, decreasing the double counting in rivers with multiple power stations. A way to account for monthly variation in the FF could be to use the monthly production of energy, and re-calculate the water efficiency score. This will differentiate the water use with respect to the energy production. In order to reduce the double counting, the FF has to be reduced as a function of the number of plants within the river system. As of right now, there exists uncertainty regarding the best way of expressing this mathematically. It would be preferable with data representing the proportion of the rivers total volume or length that is being used by the power station furthest up the river. Alternatively one could assume a reduction in river discharge from the top hydropower plant, and gradually reduce the effect of the fate factor. This issue is important to focus on moving forward, since it can have implications for the generation of impacts when assessing single hydropower plants within a river system.

Improving the connectivity index could be done by adopting the framework developed by Cote et al. (2009) and Boume et al. (2011). Here they include both up- and downstream connectivity of river systems, in conjunction with the length of river stretches. This provides a greater representation of the physical nature of the river system, and the environmental challenges incurred on freshwater fish species by constructing barriers within the river. The data requirements for creating the index by Cote et al. (2009) is greater, therefore it would be interesting to test if the inclusion of this index over the index used in this study, changes the results in a substantial way. If the differences are marginal, opting for the HCIU index by McKay et al. (2013) would be preferred when assessing multiple rivers.

In order to use the connectivity index to assess the impacts generated by one power plant/station, one can calculate the index for the given river system, and then re-calculate it without the hydropower plant under study. Subtracting the new result from the original result would provide us with the contribution to the index by the plant. One could use this estimate to reflect the index for individual plants.

Although there is room for improving the characterization factor, and to further assess the parameters of the model, we can derive a basic structure for how to proceed when performing an impact assessment of hydropower within life cycle assessment. The framework consists of (1) obtaining discharge rates for the relevant river(s), (2) using SDRs that are regionally specific, (3) estimating the water use per kWh of energy production, (4) obtaining data on total energy production, and lastly, (5) constructing the connectivity index. Given these parameters, one can construct the characterization factor (eq. 10). Table 5 provides a more detailed overview.

Table 5: Framework for constructing the characterization factor for species impacts from hydropower production

Step	Role in Characterization factor	Data sources
1.	Collection of average yearly discharge rates, this is needed to calculate the species density and the vulnerability of water use.	Public access database, distributed by NVE.
2.	Apply SDRs from the respective region (or eco-region). Use the model equations to calculate species density of rivers, essential for developing the EF (eq. 5). This increases the spatial resolution of the assessment substantially.	Calculated using public access database, distributed by Artsdatabanken. Alternatively use SDRs published in scientific literature (Hanafiah et al. 2011, Tendall et al. 2014).

3.	Calculation of water use per kWh of energy production, important in order to know how hydropower plants use water, and potentially displace fish species.	Public access data distributed by NVE, based on estimates supplied by industry (NVE, 2016).
4.	Collecting energy production data for a year, or on a monthly basis. Needed to assess how much water is being used, and subsequently the impacts this water use is contributing to.	Public access data distributed by NVE, alternatively taken directly from energy companies web pages or national statistical databases.
5.	Accounting for the barriers to fish migration by estimating a connectivity matrix. This is applied to the CF, weighting the impacts to reflect the level of habitat fragmentation within a given river system.	Public access data distributed by NVE, alternatively use of ArcGIS or other software handling satellite data. Passabillity estimates based on published meta-studies (Noonan et al. 2012).

A key area of concern is how easily accessible this kind of data is. The most challenging points in this regard is (1),(2) and (3). Energy production (4) is usually something energy companies publish regularly, and the connectivity index (5) could be constructed using satellite data (McKay et al. 2013). Access to discharge data could be dependent on public access to hydrological stations, and access to efficiency measures depends on the level of transparency between industry and the public. SDRs could be calculated on a country or eco-region level (Tendall et al. 2014), and these results should be readily published in scientific literature and open source databases.

4.6 Conclusion

In the introduction to this thesis we highlight the importance of accounting for biodiversity impacts of hydropower development through the use of life cycle assessment. In order to do this we build on some key developments from the preceding project work, focusing on calculating SDRs for Norway, and further applying the characterization factor (CF) to multiple rivers in Norway. Our results show that there is a clear statistical relationship between the discharge of a river, and the likely density of freshwater fish in the respective river. These findings was used to create a regional impact assessment of hydropower production in Norway, which found impacts to be largest within the watersheds Nordland and the West bay (fig. 17).

Applying the connectivity index to the characterization factor clearly changes the impacts in proportion to the state of hydropower development within a river system, differentiating rivers with many barriers to fish upstream migration as the rivers generating the largest impacts (fig. 22), this is a first step towards including river habitat fragmentation impacts into LCA. Applying the CF to LCI data from two hydropower stations estimating the local and global impacts on species on a per kWh basis, shows that in comparison to GHG generation from these power stations, the global impacts are marginal, but on the local level the water use impacts from hydropower contribute substantially to the local impacts. Through this case study we can conclude that the CF developed here can be applied to LCA studies on hydropower.

Further research should be focused on the fate factor, and further assessment of habitat fragmentation by using statistical approaches like the connectivity index used in this thesis. Table 5 highlight the steps taken in order to obtain the results in this thesis, with one of the most important parameters being reliable discharge rate data for the respective rivers.

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Appendix

Section 1: Essential parameters

Excel tables, includes all essential data for impact assessment used in this analysis. Figures and additional tables show values not included in the results section due to taking up too much space or being redundant.

Table 1: Values for calculating the impact scores for each river.

River	FF [m3/kWh]	EF [Species*y/m3]	Weighting [kWh/kWh]	CF [Species*y/kWh]	Yearly Impact [Species]
PASVIKELVA	27,3	3,85E-12	2,73E-03	2,87E-13	1,12E-04
DUDDAJÅKKA	2,56	3,02E-08	1,40E-04	1,08E-11	2,17E-04
LAKSELVVASSDRAGET	16,4	2,24E-10	7,02E-05	2,58E-13	2,58E-06
ALTAVASSDRAGET	2,31	1,67E-11	4,60E-03	1,78E-13	1,16E-04
NJEMENJAIKUJÅKKA	2,07	1,67E-09	1,95E-03	6,76E-12	1,88E-03
KÅFJORDVASSDRAGET	0,59	4,04E-10	2,18E-03	5,19E-13	1,61E-04
SKIBOTNVASSDRAGET	1,1	1,76E-10	2,79E-03	5,40E-13	2,15E-04
BARDUELVA	3,08	1,15E-11	9,69E-03	3,45E-13	4,75E-04
DIVIELVA	1,61	9,24E-11	9,13E-04	1,36E-13	1,77E-05
SKØELVVASSDRAGET	4,29	1,51E-09	5,88E-05	3,80E-13	3,18E-06
DALELVA	9,29	3,47E-09	1,49E-04	4,80E-12	1,02E-04
SKODDEBERGVASSDRAGET	11,1	1,10E-09	2,48E-04	3,03E-12	1,07E-04
TARALDSVIKELVA	0,96	4,32E-07	7,02E-05	2,91E-11	2,91E-04
STORELVA	1,74	9,49E-10	7,72E-04	1,28E-12	1,40E-04
SKJOMAVASSDRAGET	2,61	1,40E-10	1,14E-03	4,16E-13	6,73E-05
STORELVA	0,676	2,84E-09	8,92E-03	1,71E-11	2,17E-02
BØRSELVA	5,71	6,63E-10	1,05E-04	3,99E-13	5,98E-06
ELV FRA NIINGSVATNET	0,86	2,15E-09	5,05E-04	9,35E-13	6,73E-05
BOTNELVA	1,11	1,45E-09	1,90E-03	3,06E-12	8,26E-04
STORELVA	3,07	5,36E-08	3,16E-05	5,20E-12	2,34E-05
SOLBJØRNELVA	7,04	6,20E-09	4,56E-05	1,99E-12	1,30E-05
SAGELVVASSDRAGET	1,13	2,60E-10	1,68E-05	4,95E-15	1,19E-08
KOBBELVVASSDRAGET	3,54	1,06E-09	1,44E-03	5,39E-12	1,10E-03
LAKSÅGA	0,696	2,43E-10	5,21E-03	8,80E-13	6,53E-04
SULITJELMAVASSDRAGET	2,04	3,33E-11	7,37E-03	5,00E-13	5,25E-04
VATNVASSDRAGET (special case)	3,61	5,57E-09	3,65E-04	7,34E-12	3,81E-04
OLDEREIDELVA	1,28	3,89E-09	4,21E-04	2,10E-12	1,26E-04

ARSTADÅGA	**1,38**	**2,27E-10**	**3,37E-04**	**1,05E-13**	**5,06E-06**
KVASSTEINÅGA	**2,72**	**5,57E-09**	**1,05E-04**	**1,59E-12**	**2,39E-05**
RANAELVA	**10,9**	**2,63E-11**	**2,13E-03**	**6,13E-13**	**1,86E-04**
RØSSÅGA	**2,17**	**3,43E-10**	**2,05E-02**	**1,52E-11**	**4,45E-02**
NAMSEN	**10**	**2,39E-12**	**1,37E-02**	**3,27E-13**	**6,37E-04**
SNÅSAVASSDRAGET	**11,6**	**1,52E-11**	**1,42E-03**	**2,50E-13**	**5,04E-05**
VERDALSVASSDRAGET	**4,93**	**4,97E-11**	**3,37E-05**	**8,25E-15**	**3,96E-08**
MOSSA (Multiple Discharge rates)	**2,1**	**3,06E-09**	**5,26E-04**	**3,38E-12**	**2,54E-04**
HOPLA	**1**	**1,60E-09**	**4,21E-07**	**6,74E-16**	**4,04E-11**
LEVANGERVASSDRAGET	**7,22**	**2,61E-09**	**7,37E-05**	**1,39E-12**	**1,46E-05**
STJØRDALSVASSDRAGET	**1,56**	**1,66E-11**	**3,40E-03**	**8,84E-14**	**4,29E-05**
SKAUDALSVASSDRAGET	**4,01**	**3,40E-10**	**3,92E-04**	**5,35E-13**	**2,99E-05**
HASSELVASSDRAGET	**7,52**	**2,25E-08**	**2,53E-05**	**4,28E-12**	**1,54E-05**
TEKSDALSELVA	**12,2**	**1,18E-09**	**8,42E-05**	**1,22E-12**	**1,46E-05**
VIKELVA	**4,16**	**9,14E-09**	**9,06E-05**	**3,44E-12**	**4,44E-05**
NIDELVVASSDRAGET	**5,48**	**9,95E-12**	**1,47E-02**	**8,00E-13**	**1,67E-03**
ORKLA	**1,68**	**3,34E-11**	**6,82E-03**	**3,82E-13**	**3,71E-04**
GAULARVASSDRAGET	**4,3**	**1,39E-11**	**1,17E-03**	**6,96E-14**	**1,16E-05**
SURNA	**1,13**	**2,80E-11**	**5,85E-03**	**1,85E-13**	**1,54E-04**
DRIVA	**0,825**	**1,97E-11**	**5,26E-03**	**8,55E-14**	**6,40E-05**
LITLEDALSELVA	**0,926**	**2,77E-10**	**1,19E-02**	**3,06E-12**	**5,20E-03**
GRYTÅA	**2,52**	**9,89E-09**	**1,53E-04**	**3,81E-12**	**8,31E-05**
VIKELVA	**1,98**	**3,20E-09**	**4,56E-05**	**2,89E-13**	**1,88E-06**
TUSSELVA	**0,66**	**3,65E-09**	**1,83E-03**	**4,40E-12**	**1,14E-03**
STANDALELVA	**1,99**	**7,23E-09**	**1,62E-04**	**2,33E-12**	**5,39E-05**
NULL	**1,03**	**9,45E-09**	**1,75E-04**	**1,71E-12**	**4,27E-05**
FORTUNVASSDRAGET	**0,56**	**7,77E-11**	**1,16E-02**	**5,04E-13**	**8,32E-04**
JOSTEDØLA	**0,575**	**2,68E-11**	**9,34E-03**	**1,44E-13**	**1,91E-04**
KAUPANGERELVI	**1,08**	**5,54E-08**	**8,21E-05**	**4,91E-12**	**5,75E-05**
GAULARVASSDRAGET	**8,07**	**4,30E-11**	**9,48E-05**	**3,29E-14**	**4,44E-07**
JØLSTRA	**4,46**	**3,74E-11**	**3,45E-03**	**5,77E-13**	**2,84E-04**
ÅNGEDALSELVA	**1,58**	**4,45E-10**	**9,06E-05**	**6,37E-14**	**8,22E-07**
NAUSTA	**5,05**	**1,24E-10**	**3,86E-05**	**2,43E-14**	**1,33E-07**
OSELVVASSDRAGET	**4,57**	**1,45E-10**	**5,26E-04**	**3,48E-13**	**2,61E-05**
ÅSKORELVA	**0,68**	**1,56E-09**	**3,73E-03**	**3,97E-12**	**2,11E-03**
STORELVA	**1,33**	**2,43E-09**	**1,71E-04**	**5,53E-13**	**1,35E-05**
BREIMSVASSDRAGET	**8,09**	**3,95E-11**	**1,28E-03**	**4,11E-13**	**7,52E-05**
BREIMSVASSDRAGET	**4,72**	**1,90E-10**	**9,34E-04**	**8,37E-13**	**1,11E-04**
DYRNESLIELVA	**1,43**	**2,61E-09**	**3,30E-04**	**1,23E-12**	**5,79E-05**
KLYVTVEITELVA	**1,09**	**3,87E-08**	**2,93E-04**	**1,24E-11**	**5,17E-04**
STORELVA	**7,41**	**1,06E-09**	**1,40E-04**	**1,10E-12**	**2,21E-05**
RISEELVA	**1,47**	**2,61E-09**	**1,90E-03**	**7,30E-12**	**1,98E-03**

AURLANDSVASSDRAGET	**1,31**	**5,03E-11**	**1,97E-02**	**1,30E-12**	**3,63E-03**
MATREVASSDRAGET	**1,1**	**8,14E-10**	**9,83E-03**	**8,80E-12**	**1,23E-02**
STEINSLANDSVASSDRAGET	**1,58**	**7,39E-11**	**6,47E-03**	**7,56E-13**	**6,97E-04**
VOSSOVASSDRAGET	**1,1**	**1,25E-10**	**1,02E-02**	**1,40E-12**	**2,04E-03**
EIKELANDSVASSDRAGET	**0,87**	**7,03E-09**	**5,62E-04**	**3,44E-12**	**2,75E-04**
KVERSELVA	**2,64**	**2,46E-09**	**1,68E-04**	**1,09E-12**	**2,63E-05**
AUSTREPOLLELVA	**1,02**	**1,01E-09**	**2,40E-04**	**2,47E-13**	**8,45E-06**
TVEITELVA	**1,92**	**1,42E-08**	**7,02E-05**	**1,91E-12**	**1,91E-05**
JONDALSELVI	**4,07**	**9,01E-10**	**2,74E-04**	**1,00E-12**	**3,91E-05**
TYSSO	**1,4**	**9,66E-11**	**1,53E-02**	**2,07E-12**	**4,51E-03**
SØRELVA	**1,92**	**5,82E-10**	**1,25E-03**	**1,40E-12**	**2,49E-04**
SULDALSVASSDRAGET	**1,32**	**1,06E-11**	**4,58E-02**	**6,38E-13**	**4,16E-03**
FØRREELVA	**4,57**	**2,54E-10**	**3,58E-04**	**4,16E-13**	**2,12E-05**
ÅRDALSELVA	**0,97**	**1,84E-10**	**8,77E-03**	**1,57E-12**	**1,96E-03**
LYSEVASSDRAGET	**0,49**	**2,37E-09**	**2,30E-03**	**2,66E-12**	**8,71E-04**
BJERKREIMVASSDRAGET	**2,31**	**2,97E-11**	**8,14E-04**	**5,59E-14**	**6,48E-06**
HELLELANDSELVA	**6,9**	**2,32E-10**	**4,56E-04**	**7,30E-13**	**4,75E-05**
SIRA	**6,95**	**6,71E-12**	**5,56E-03**	**2,59E-13**	**2,05E-04**
LITLEÅNI	**9,62**	**2,64E-10**	**6,06E-04**	**1,54E-12**	**1,33E-04**
MANDALSELVA	**4,24**	**1,33E-11**	**9,97E-03**	**5,62E-13**	**7,98E-04**
ARENDALSVASSDRAGET	**10,9**	**8,03E-12**	**1,24E-02**	**1,09E-12**	**1,92E-03**
SKIENSVASSDRAGET	**5,76**	**2,16E-12**	**5,75E-02**	**7,15E-13**	**5,85E-03**
NUMEDALSLÅGEN	**6,96**	**7,67E-12**	**1,98E-02**	**1,06E-12**	**2,98E-03**
DRAMMENSVASSDRAGET	**17,9**	**1,35E-12**	**1,99E-02**	**4,80E-13**	**1,36E-03**
HALLINGDALSVASSDRAGET	**2,78**	**7,73E-12**	**3,43E-02**	**7,37E-13**	**3,61E-03**
GLOMMAVASSDRAGET	**20,17**	**5,03E-13**	**5,03E-02**	**5,10E-13**	**3,65E-03**
MOSSEVASSDRAGET	**18,2**	**1,34E-10**	**9,76E-05**	**2,39E-13**	**3,32E-06**
HALDENVASSDRAGET	**12,72**	**1,03E-10**	**9,48E-04**	**1,24E-12**	**1,67E-04**

Table 2: Average discharge rates per county, and the share of total discharge from all the counties.

County	**Average river discharge**	**Share of total**
Finnmark	**68,175**	**5 %**
Troms	**20,30875**	**2 %**
Nordland	**11,37246715**	**1 %**
Nord-Trøndelag	**63,88428571**	**5 %**
Sør-Trøndelag	**35,91857143**	**3 %**
Møre og Romsdal	**17,87**	**1 %**

Sogn og Fjordane	19,80764706	1 %
Hordaland	10,80796446	1 %
Rogaland	32,96	2 %
Vest-Agder	76,1	6 %
Aust-Agder	116,3	9 %
Telemark	263	20 %
Vestfold	119,7	9 %
Buskerud	235,9	18 %
Østfold	231,3696119	17 %

Table 3: Data used to calculate the statistical functions related to the connectivity index in section 3.1.

River	Energy production [kWh/yr]	HCIU	Impact	Index	impact with index
STORELVA	4,50E+06	0,8	2,34E-05	0,2	4,68E-06
SOLBJØRNELVA	6,50E+06	0,8	1,30E-05	0,2	2,59E-06
LAKSÅGA	1,10E+08	0,925	6,53E-04	0,075	4,90E-05
VERDALSVASSDRAGET	4,80E+06	0,957	3,96E-08	0,043	1,70E-09
GAULARVASSDRAGET	1,66E+08	0,878	1,16E-05	0,122	1,41E-06
MOSSEVASSDRAGET	1,39E+07	0,7	3,32E-06	0,3	9,96E-07
HOPLA	6,00E+04	0,7	4,04E-11	0,3	1,21E-11
TUSSELVA	2,60E+08	0,85	1,14E-03	0,15	1,71E-04
PASVIKELVA	3,89E+08	0,653	1,12E-04	0,347	3,87E-05
ALTAVASSDRAGET	6,55E+08	0,66	1,16E-04	0,34	3,96E-05
NJEMENJAIKUJÅKKA	2,78E+08	0,84	1,88E-03	0,16	3,01E-04
SKIBOTNVASSDRAGET	3,98E+08	0,8	2,15E-04	0,2	4,30E-05
DALELVA	2,12E+07	0,37	1,02E-04	0,63	6,42E-05
KOBBELVVASSDRAGET	7,42E+08	0,83	1,10E-03	0,17	1,88E-04
SULITJELMAVASSDRAGET	1,05E+09	0,55	5,25E-04	0,45	2,36E-04
STJØRDALSVASSDRAGET	4,85E+08	0,61	4,29E-05	0,39	1,67E-05
NIDELVVASSDRAGET	2,09E+09	0,24	1,67E-03	0,76	1,27E-03
ORKLA	9,71E+08	0,684	3,71E-	0,316	1,17E-04

			04		
SURNA	**8,33E+08**	**0,909**	**1,54E-04**	**0,091**	**1,40E-05**
FORTUNVASSDRAGET	**1,65E+09**	**0,82**	**8,32E-04**	**0,18**	**1,50E-04**
JØLSTRA	**4,92E+08**	**0,446**	**2,84E-04**	**0,554**	**1,57E-04**
RISEELVA	**2,71E+08**	**0,92**	**1,98E-03**	**0,08**	**1,58E-04**
STEINSLANDSVASSDRAGET	**9,22E+08**	**0,55**	**6,97E-04**	**0,45**	**3,14E-04**
TYSSO	**2,18E+09**	**0,32**	**4,51E-03**	**0,68**	**3,07E-03**
HELLELANDSELVA	**6,50E+07**	**0,409**	**4,75E-05**	**0,591**	**2,80E-05**
SKIENSVASSDRAGET	**8,19E+09**	**0,19**	**5,85E-03**	**0,81**	**4,74E-03**
DRAMMENSVASSDRAGET	**2,84E+09**	**0,33**	**1,36E-03**	**0,67**	**9,14E-04**
GLOMMAVASSDRAGET	**7,16E+09**	**0,251**	**3,65E-03**	**0,749**	**2,73E-03**
HALLINGDALSVASSDRAGET	**4,89E+09**	**0,31**	**3,61E-03**	**0,69**	**2,49E-03**
NUMEDALSLÅGEN	**2,82E+09**	**0,4**	**2,98E-03**	**0,6**	**1,79E-03**
ARENDALSVASSDRAGET	**1,77E+09**	**0,27**	**1,92E-03**	**0,73**	**1,41E-03**
SKJOMAVASSDRAGET	**1,62E+08**	**0,81**	**6,73E-05**	**0,19**	**1,28E-05**
RANAELVA	**3,04E+08**	**0,49**	**1,86E-04**	**0,51**	**9,50E-05**
LYSEVASSDRAGET	**3,27E+08**	**0,85**	**8,71E-04**	**0,15**	**1,31E-04**
FØRREELVA	**5,10E+07**	**0,7**	**2,12E-05**	**0,3**	**6,36E-06**

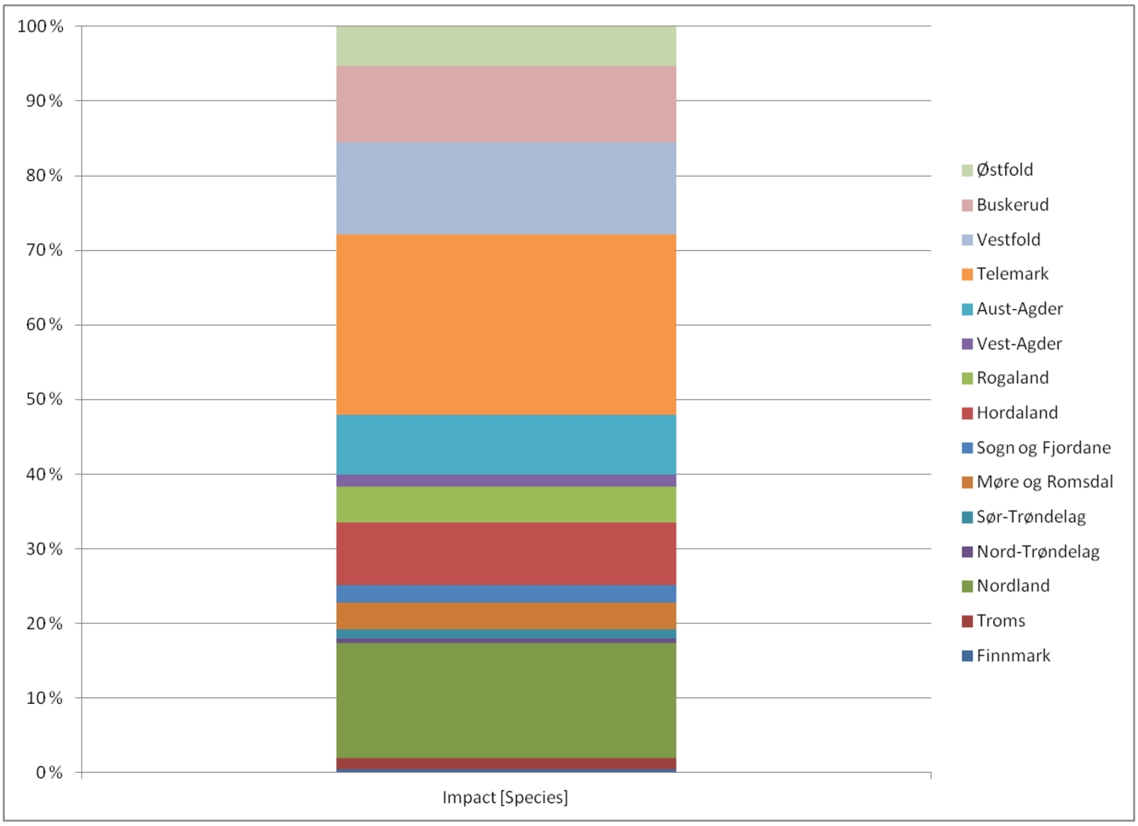


Figure 1: Share of impacts per county

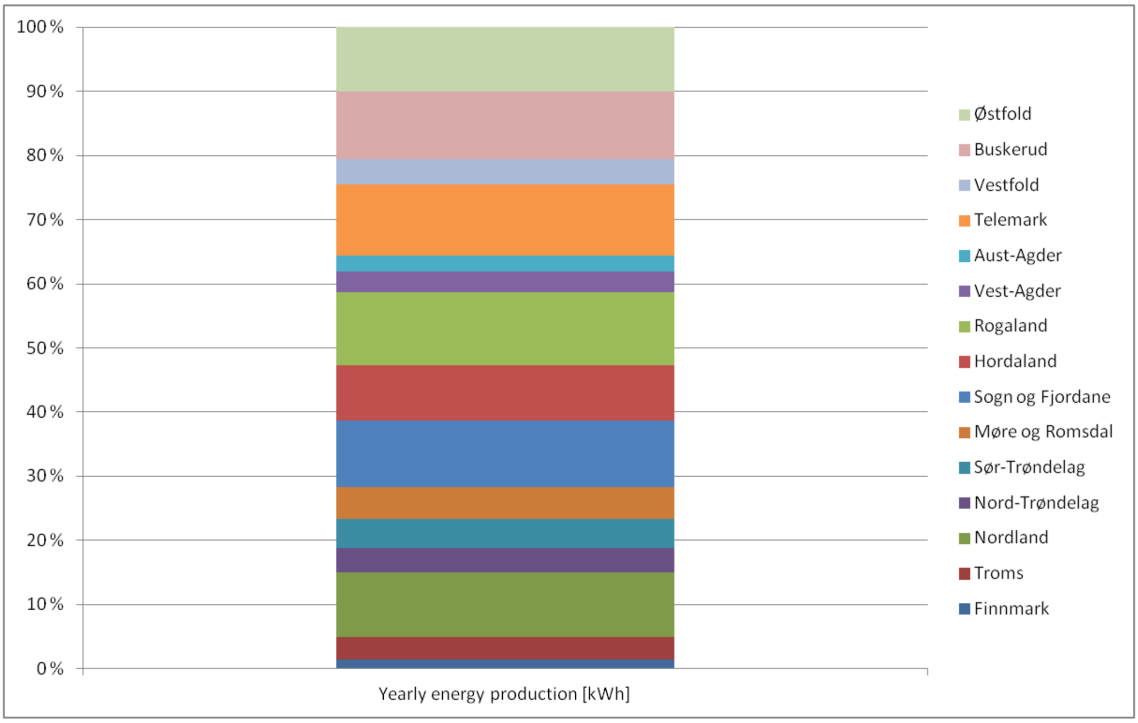


Figure 2: Share of energy production per county

Table 4: Average Characterization factors and impact scores per county

County	Average CF [Species*y/Kwh]	County	Impact [Species]
Finnmark	2,89205E-12	Finnmark	2,18E-04
Troms	2,06411E-12	Troms	7,22E-04
Nordland	4,90778E-12	Nordland	7,25E-03
Nord-Trøndelag	7,78231E-13	Nord-Trøndelag	2,78E-04
Sør-Trøndelag	1,53169E-12	Sør-Trøndelag	6,02E-04
Møre og Romsdal	1,98344E-12	Møre og Romsdal	1,64E-03
Sogn og Fjordane	2,09917E-12	Sogn og Fjordane	1,14E-03
Hordaland	2,21179E-12	Hordaland	3,94E-03
Rogaland	1,01161E-12	Rogaland	2,30E-03
Vest-Agder	7,85794E-13	Vest-Agder	7,39E-04
Aust-Agder	1,08741E-12	Aust-Agder	3,76E-03
Telemark	7,14749E-13	Telemark	1,14E-02
Vestfold	1,05609E-12	Vestfold	5,81E-03
Buskerud	6,08809E-13	Buskerud	4,85E-03
Østfold	6,61442E-13	Østfold	2,48E-03

Table 5: Average characterization factor and impact scores per watershed

Watershed	Average CF [Species*y/Kwh]	Watershed	Impacts [Species]
Finnmark	2,89205E-12	Finnmark	2,18E-04
Troms	2,06411E-12	Troms	7,22E-04
Nordland	4,90778E-12	Nordland	7,25E-03
Troendelag	1,15496E-12	Troendelag	4,40E-04
Moere and Romsdal	1,98344E-12	Moere and Romsdal	1,64E-03
West	2,14088E-12	West	2,17E-03
SE South West	9,51443E-13	SE South West	1,97E-03
West Bay	7,47116E-13	West Bay	6,73E-03
Glomma	6,61442E-13	Glomma	2,48E-03

Section 2: MatLab script

Script to generate the three SDR models

SDR_General_Average includes all the 42 rivers, SDR_Low_HP includes the rivers with no or low amounts of hydropower development (Total of 21 rivers), SDR_High-HP includes the rivers with medium to high hydropower development (Total of 21 Rivers).

```
function [fitresult, gof] = createFits(AverageYearlyDishcargeratem3s,
Fishspeciescount, AverageYearlyDishcargeratem3sLow, FishspeciescountLow,
AverageYearlyDishcargeratem3sHigh, FishspeciescountHigh)
%CREATEFITS(AVERAGEYEARLYDISHCARGERATEM3S,FISHSPECIESCOUNT,AVERAGEYEARLYDIS
HCARGERATEM3SLOW,FISHSPECIESCOUNTLOW,AVERAGEYEARLYDISHCARGERATEM3SHIGH,FISH
SPECIESCOUNTHIGH)
%  Create fits.
%
%  Data for 'SDR_General_Average' fit:
%      X Input : AverageYearlyDishcargeratem3s
%      Y Output: Fishspeciescount
```

```
%  Data for 'SDR_Low_HP' fit:
%      X Input : AverageYearlyDishcargeratem3sLow
%      Y Output: FishspeciescountLow
%  Data for 'SDR_High_HP' fit:
%      X Input : AverageYearlyDishcargeratem3sHigh
%      Y Output: FishspeciescountHigh
%  Output:
%      fitresult : a cell-array of fit objects representing the fits.
%      gof : structure array with goodness-of fit info.
%
%  See also FIT, CFIT, SFIT.

%  Auto-generated by MATLAB on 01-Apr-2016 13:42:31

%% Initialization.

% Initialize arrays to store fits and goodness-of-fit.
fitresult = cell( 3, 1 );
gof = struct( 'sse', cell( 3, 1 ), ...
    'rsquare', [], 'dfe', [], 'adjrsquare', [], 'rmse', [] );

%% Fit: 'SDR_General_Average'.
[xData, yData] = prepareCurveData( AverageYearlyDishcargeratem3s,
Fishspeciescount );

% Set up fittype and options.
ft = fittype( 'power1' );
opts = fitoptions( 'Method', 'NonlinearLeastSquares' );
opts.Display = 'Off';
opts.StartPoint = [1.45634506603136 0.414849571910074];

% Fit model to data.
[fitresult{1}, gof(1)] = fit( xData, yData, ft, opts );

% Plot fit with data.
figure( 'Name', 'SDR_General_Average' );
h = plot( fitresult{1}, xData, yData );
legend( h, 'Fishspeciescount vs. AverageYearlyDishcargeratem3s',
'SDR_General_Average', 'Location', 'NorthEast' );
% Label axes
xlabel( 'AverageYearlyDishcargeratem3s' );
ylabel( 'Fishspeciescount' );
grid on

%% Fit: 'SDR_Low_HP'.
[xData, yData] = prepareCurveData( AverageYearlyDishcargeratem3sLow,
FishspeciescountLow );

% Set up fittype and options.
ft = fittype( 'power1' );
opts = fitoptions( 'Method', 'NonlinearLeastSquares' );
opts.Display = 'Off';
opts.StartPoint = [1.60444713403024 0.48680084076186];

% Fit model to data.
[fitresult{2}, gof(2)] = fit( xData, yData, ft, opts );

% Plot fit with data.
figure( 'Name', 'SDR_Low_HP' );
```

```
h = plot( fitresult{2}, xData, yData );
legend( h, 'FishspeciescountLow vs. AverageYearlyDishcargeratem3sLow',
'SDR_Low_HP', 'Location', 'NorthEast' );
% Label axes
xlabel( 'AverageYearlyDishcargeratem3sLow' );
ylabel( 'FishspeciescountLow' );
grid on

%% Fit: 'SDR_High_HP'.
[xData, yData] = prepareCurveData( AverageYearlyDishcargeratem3sHigh,
FishspeciescountHigh );

% Set up fittype and options.
ft = fittype( 'power1' );
opts = fitoptions( 'Method', 'NonlinearLeastSquares' );
opts.Display = 'Off';
opts.StartPoint = [1.53581712344879 0.319382272626591];

% Fit model to data.
[fitresult{3}, gof(3)] = fit( xData, yData, ft, opts );

% Plot fit with data.
figure( 'Name', 'SDR_High_HP' );
h = plot( fitresult{3}, xData, yData );
legend( h, 'FishspeciescountHigh vs. AverageYearlyDishcargeratem3sHigh',
'SDR_High_HP', 'Location', 'NorthEast' );
% Label axes
xlabel( 'AverageYearlyDishcargeratem3sHigh' );
ylabel( 'FishspeciescountHigh' );
grid on
```

Section 3: Additional background information

Additional information on life cycle assessment.

In LCA accounting for emissions over the life cycle and life stages of a product is done by adopting the mathematical framework of the Leontief model. Originally this model was developed by Leontief in order to capture the interconnected nature of the economic sectors within a country, and is generally termed the Input-Output model (Leontief, 1936). This model uses the representation of a linear system of equations, show in equation (1).

$$x = Ax + y$$

In equation (1) x denotes the output of a system, A denotes the coefficients matrix of the system, and y denotes a specific external demand from the system. This framework is efficient for analyzing economic systems, and also work well when analyzing production system, using resource flows instead of monetary flows (Ebiefung & Kostreva, 1994, Strømman, 2010). With a sufficient amount of data representing the supply-chain of a product, the product often being denoted the term "functional unit", we can investigate how much energy and materials went into producing one functional unit, and provide additional emission equivalents to the resource use (Strømman, 2010). A functional unit (the y term in equation (1)) could for instance be 1 KWh of electricity produced by a specific system. Providing enough data and transparency of the supply-chain, we can estimate the amount of emissions and potential impacts of these emissions, attributed to the production of 1 KWh of electricity.

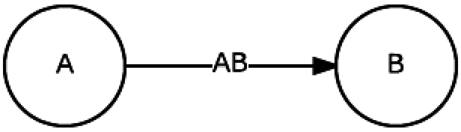
Impact assessments are based on characterization factors in multiple impacts categories that aim to assess the consequences of various emissions from products on human health, ecosystem quality, and resource depletion. (Jolliet et al. 2003, Rosenbaum et al. 2008). The figure below gives a short overview of the many different processes that an LCA tries to incorporate in order to perform a total assessment of a system. Both the inclusion of primary resources and the waste handling of a product is essential for a complete understanding of the many processes involved (Rebitzer et al. 2004).

The general framework of LCA follows the continuously updating ISO standards (Rebitzer et al. 2004, Finnveden et al. 2009), where the latest update within environmental management systems is represented in the ISO 14001 standard (ISO, 2015). Figure (x) shows the most common representation of the framework.

These standards aim to help and guide practitioners, using best practice principles related to relevance, validity, modeling, and transparency of the LCA (Rebitzer et al. 2004). This helps to create a standardized method for comparison of technologies. A critique of the ISO standards is that the guiding principles are to general and criteria for characterization factors to lax, potentially leading to confusion among practitioners (Hauschild et al. 2013). Although there exist large variability in designs, results, and system definitions in LCA studies, contributing to uncertainties of the models, they have proven a valuable asset to policy developers and decision makers when evaluating technological systems (IPCC, 2011).

Additional information on the connectivity index (Graph theory approach)

River streams have been characterized as dendritic networks, where statistical indices can provide information on the overall connectivity of the "nodes" and "edges" of the network (Cote et al. 2009: Grant et al. 2007). A "node" can represent a river zone, simply an area of habitat, and the "edge" represent the flow between one zone and another. One can think of the node A as a river zone, and node B as a river zone upstream of node A, where the edge AB represent a connectivity barrier between zone A and B (for instance a hydropower dam), illustrated in Figure 1. Zone A could be from the mouth of the river, while zone B is the area above the hydropower dam AB.



Figur 1: Circles are nodes, representing river zones. Arrow represent connection between node A and B, AB represents a barrier (Mattson, 2015).

One of the first graph theoretical approaches to analysis of river ecosystem quality and impacts, was developed by Cote et al. (2009). They constructed an index relating barriers to movement between nodes in a network, to the problems faced by migrating fish within river system. This index was named the Dendritic Connectivity Index (DCI), and estimate the probability of fish to move from one end of the river system, to another. The DCI was applied by Cote et al. (2009) and Bourne et al. (2011) in Terra Nova National Park, USA. They used the index to assess the connectivity of rivers, where scores from 0 to 100 indicate the ease of migrating fish species to move up and down the river system. Using this method one can identify crucial points of connectivity within a river network, by removing barriers one by one, and assessing how this affects the index. Using this framework, one can develop an effective strategy for improving the conditions for species in the river (Cote et al. 2009, Bourne et al. 2011).

These studies were the basis for McKay et al. (2013) and multiple other studies using graph theoretical approaches for assessment of river ecosystems (Fagan, 2002, Eros et al. 2011, Eros et al. 2012). McKay et al. (2013) used a simplified DCI index to prioritize barrier improvement in the Truckee River, USA, assessing connectivity issues for migratory fish (McKay et al. 2013). They created a habitat connectivity index for upstream passage (HCIU) for diadromous fish, which is the index adopted in this study, and further explained in section 2.2 of the methods.