Doctoral theses at NTNU, 2019:161

Martin Dorber

Quantifying biodiversity impacts of hydropower electricity production within the framework of Life Cycle Assessment

Doctoral Thesis

Martin Dorber

ISBN 978-82-326-3920-5 (printed version)
ISBN 978-82-326-3921-2 (electronic version)
ISSN 1503-8181

Doctoral theses at NTNU, 2019:161

NTNU
Norwegian University of
Science and Technology
Faculty of Engineering
Department of Energy and Process Engineering

NTNU
Norwegian University of
Science and Technology

NTNU

NTNU
Norwegian University of
Science and Technology

Martin Dorber

Quantifying biodiversity impacts of hydropower electricity production within the framework of Life Cycle Assessment

Thesis for the degree of Philosophiae Doctor

Trondheim, June 2019

Norwegian University of Science and Technology
Faculty of Engineering
Department of Energy and Process Engineering

NTNU
Norwegian University of
Science and Technology

NTNU
Norwegian University of Science and Technology

Thesis for the degree of Philosophiae Doctor

Faculty of Engineering
Department of Energy and Process Engineering

ISBN 978-82-326-3920-5 (printed version)
ISBN 978-82-326-3921-2 (electronic version)
ISSN 1503-8181

Doctoral theses at NTNU, 2019:161

Printed by Skipnes Kommunikasjon as

Preface

The thesis has been submitted to the Faculty of Engineering Science (IV) at the Norwegian University of Science and Technology (NTNU) in partial fulfilment of the degree of Philosophiae Doctor. This work was carried out at the Industrial Ecology Programme (IndEcol) and the Department of Energy and Process Engineering (EPT), in the period 2016-2019. This PhD work was part of the "Towards sustainable renewable energy production (SURE): Developing a Life Cycle Impact Assessment framework for biodiversity impacts" project, funded through the ENERGIX programme by the Research Council of Norway (Grant Number 244109).

Martin Dorber,

Trondheim, March 2019

To my parents

for always believing in me, for their unqualified support and love

Acknowledgments

First of all, I would like to thank my supervisors Francesca Verones and Roel May. Thank you so much for being available at any moment, even when you had little time yourselves. Francesca, all our discussions, your explanations and helpful comments, your open door, and all the motivating words in difficult times, have been without a doubt, indispensable for my gratifying PhD research.

Roel, thanks a lot for your quasi support in times of high uncertainty and for always finding a way to sort out data issues.

I would also express my sincere thanks to Odd Terje Sandlund, for supporting my research and for feedback which was always on the spot and guided me always in the right direction.

In addition, I want to thank John Woods and Koen Kuipers, colleagues and friends at once, for all your discussions, critical reflections and all your honest feedback. I guess it is self-evident that you both contributed a big proportion to my pleasant working environment at IndEcol.

Certainly, it would have been only half as pleasant, without all my colleagues at IndEcol, and I would like to express my special thanks to all of you. I would like to thank especially Cristina Iordan, for all the small sweet gifts, and Eivind Bjelle, for making sure that I have not missed any external meeting.

Furthermore, I would like to thank Erik Gracey, for laying the foundation of the SURE-Project.

Besides, I would also like to thank the whole NINA aquatic department, for a strongly influencing research stay, and especially Ingeborg Palm Helland and Antti Eloranta.

In addition, I would like to acknowledge the whole NTNUI Friidrett group, which had a big influence on my work-life balance. My special thanks go to Magnus Warvik and Svenn-Helge Vatne.

Special thanks to Ronnie Schmit, for always having an open ear.

Last but not least, very special thanks to my family. Sylke and Ralf, your outstanding support with seemingly unlimited energy at every situation, it was and is beyond words. Elisabeth und Wolfgang, ohne euch hätte ich womöglich nie meine Liebe zur Natur entdeck und vielen Dank das ihr mich immer mit so viel lebensfreunde unterstützt. Beatrice, tack så jättemycket for backing me up so lovely in every life situation.

Abstract

The United Nations developed 17 Sustainable Development Goals (SDGs) for the transition into a more sustainable world. One of the central aspects of the SDGs is the provisioning of sustainable energy, covered by SDG 7 (Affordable and clean energy). Hydropower, the largest source of renewable electricity production, has a huge potential to contribute to the fulfilment of SDG 7. As the SDGs can be viewed as a network, fulfilment of the SDG 7 targets can lead to positive synergies and negative trade-offs with other SDGs. Due to relatively low CO_2 emissions, compared to other energy technologies, hydropower electricity production can help to fulfil SDG 13 (Climate action). However, due to land use and land use change, freshwater habitat alteration and water quality degradation, hydropower electricity production may negatively affect terrestrial and aquatic biodiversity. This can lead to negative trade-offs with SDG 6 (Clean water and sanitation) and SDG 15 (Life on land).

Life Cycle Assessment (LCA) is a tool that is used to analyse the environmental impacts of a product or process throughout all its life cycle stages. Hence, LCA can help to identify locations where hydropower electricity production will have the lowest biodiversity impact. However, due to a lack of methods, so far no LCA study has accounted for biodiversity impacts of hydropower electricity production.

This PhD work was part of the "Towards sustainable renewable energy production (SURE): Developing a Life Cycle Impact Assessment framework for biodiversity impacts" project, and aimed to advance and develop operational LCA related methods for the assessment of biodiversity impacts of hydropower electricity production in LCA.

The assessment of biodiversity impacts of hydropower electricity production in LCA requires site-specific Life Cycle Inventory (LCI) data. In Chapter 2, the first net land occupation LCI parameters for existing Norwegian hydropower reservoirs are provided. The underlying model uses satellite images to account for the natural water surface area before dam construction. The newly developed method has the potential for global application to all reservoirs where annual electricity production is reported.

The net land occupation values from Chapter 2 enabled a calculation of net water consumption values for Norwegian hydropower reservoirs in Chapter 3. To quantify this water consumption, an

evaporation model with global coverage was used, having again the potential for global application.
In the Life Cycle Impact Assessment (LCIA) step, characterization factors (CFs) are required, to transform the calculated land occupation and water consumption LCI values into potential biodiversity impacts.
For the LCIA impact category “water stress”, so far no CFs existed that could quantify the aquatic biodiversity impact of water consumption in a recently (in geological time) glaciated region like Norway. Therefore, the first spatially-explicit CFs quantifying biodiversity impacts of water consumption in a post-glaciated region were developed in Chapter 3. The novelty behind these CFs is that they include Species-discharge relationships (SDR), which account for local variation in fish fauna by delineating regions with the same postglacial freshwater fish immigration history.
Inside the LCIA impact category “land stress”, so far no CFs covering land use change from terrestrial to aquatic habitat existed, even though this may be a major environmental change occurring during reservoir creation. Therefore, in Chapter 4, the first global CFs that quantify the potential future biodiversity impact of inundating terrestrial habitat area were developed. To follow current recommendations from the Life Cycle Initiative hosted by UN Environment and to enhance comparability, the CFs are based on an adaptation of the methodology developed by Chaudhary et al. 2015.
In Chapter 5, a global and spatially explicit assessment of terrestrial and freshwater biodiversity impacts of potential future hydropower reservoirs is performed. This is done by combining a high-resolution, technical assessment of the future ecological economic hydropower potential (Gernaat et al. 2017) with the developed LCA models in this thesis and existing methodology. The results reveal that carefully selecting future hydropower reservoir locations can significantly avoid future biodiversity impacts and can in turn help to achieve the development of sustainable renewable energy.

In summary, this thesis contributes models to the research community that now allow the assessment of damages on ecosystem quality from hydropower electricity production (and additional stressors) within LCA, especially regarding the impact categories “water stress” and “land stress”. However, it is not possible to assess all relevant biodiversity impacts (yet), wherefore further methodological developments are needed.

List of publications

Articles included in this thesis:

1. Dorber, M.; May, R.; Verones, F., Modeling Net Land Occupation of Hydropower Reservoirs in Norway for Use in Life Cycle Assessment. *Environmental Science & Technology* 2018, *52*, (4), 2375-2384.

> Author contributions: M.D. designed and carried out the analyses. M.D., R.M., and F.V. wrote the manuscript. M.D. made all the figure and tables.

2. Dorber, M.; Mattson, K. R.; Sandlund, O. T.; May, R.; Verones, F., Quantifying net water consumption of Norwegian hydropower reservoirs and related aquatic biodiversity impacts in Life Cycle Assessment. *Environmental Impact Assessment Review* 2019, *76*, 36-46.

> Author contributions: M.D. and K.M. designed the analyses. M.D. carried out the analyses. M.D., K.M., R.M., O.S. and F.V. wrote the manuscript. M.D. made all the figure and tables.

3. Dorber, M.; Kuipers, K.; Verones, F. (In review), Global characterization factors for biodiversity impacts of land inundation in Life Cycle Assessment. In review: *Nature - Scientific Data.*

> Author contributions: M.D. designed the analyses. M.D. carried out the analyses. M.D., K.K. and F.V. wrote the manuscript. M.D. and K.K. made all the figures. M.D. made all tables.

4. Dorber, M.; Arvesen, A.; Gernaat, D.; Verones, F. (In preparation), The potential to control biodiversity impacts of future global hydropower reservoirs by strategic site selection. Intended for publication in *Nature Communications.*

> Author contributions: A.A., D.G., and M.D. conceived the research idea. M.D. and F.V. designed the analyses. D.G. provided potential new hydropower reservoirs data. M.D. carried out the analyses. M.D., A.A., D.G. and F.V. wrote the manuscript.

Table of Contents

Chapter 1: Introduction

1.1 Renewable electricity production for a more sustainable world

Achieving a good quality of life for a growing human population without using planet earth beyond its boundaries is possibly the biggest challenge facing humanity.[1,2] To reconcile human development needs and the protection of the biosphere that humans are depending on, sustainable development pathways are required.[3] For the transition into a more sustainable world, the United Nations developed 17 Sustainable Development Goals (SDG) (Figure 1), including 169 targets to be fulfilled by 2030.[4]

Figure 1: The United Nations developed 17 Sustainable Development Goals (SDGs). The arrangement of the SDGs highlights the interconnections between the SDGs and that economies and societies depended on the biosphere. Credit: Azote Images for Stockholm Resilience Centre.[5]

One of the central aspects of the SDGs is the provisioning of sustainable energy,[6] covered by SDG 7 (Affordable and clean energy), which has the target to *"ensure universal access to affordable, reliable and modern energy services"*.[4] As the world electricity demand is expected to double by 2050,[7] the fulfilment of this target requires an increase in electricity production.

Renewable electricity production plays an important role in this transition, as a second target within SDG 7 is to "*increase substantially the share of renewable energy in the global energy mix"*.[4]

As the SDGs can be viewed as a network,[8] with interdependent goals,[9] there is a nexus between the goals.[10] This means that fulfilment of the SDG 7 targets will lead to positive synergies with other SDGs.[10] One important synergy stems from the fact that renewable energy sources are likely to have lower CO_2 emissions in comparison to fossil fuel based electricity production.[7] Hence, renewable electricity production is expected to contribute to mitigating climate change and can thus help to fulfil SDG 13 (Climate action) (Figure 2). Further, the IPCC special report "Global Warming 1.5 ºC" highlights, that in all pathways limiting global warming to 1.5 °C (with no or limited overshoot), up to 85% of the total electricity demand has to be produced from renewable energy sources.[11]

In addition, renewable electricity can be indirectly important for the fulfilment of further SDGs. The assessment of the relationships is context and definition specific, but it has been shown that actions inside SDG 7 can lead to synergies with 143 of the 169 SDG targets.[6] For example, housing, cooking, irrigation, transport, health care, financial systems and any information and communications technology (for example: computers, smartphones, internet) rely on electricity. All these aspects are important to fulfil SDG 11 (Make cities and human settlements inclusive, safe, resilient and sustainable), SDG 2 (End hunger, achieve food security and improved nutrition, and promote sustainable agriculture), SDG 4 (Ensure inclusive and equitable quality education and promote lifelong learning opportunities for all), SDG 5 (Achieve gender equality and empower all women and girls), SDG 9 (Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation) and SDG 17 (Strengthen the means of implementation and revitalize the Global Partnership for Sustainable Development). This overview is not meant to be comprehensive, but shows the variety and quantity of possible synergies.

Geothermal, solar power, wind power and hydropower have been identified as key renewable electricity production technologies for the future.[7] All these technologies convert energy from natural processes (e.g. flowing water and wind) into electricity[7] and, together, produced approximately 6,200 TWh in 2015.[12] Globally, hydropower is the largest source of renewable electricity production with a total annual production of 3,889 TWh in 2016, which corresponded to 16.6% of the global electricity supply.[12] Geothermal, solar power, wind power produced in 2015 approximately 650 TWh each.

Geothermal plants require boreholes to use the thermal energy stored in the earth to heat water.[7] Solar power obtains energy from sunlight which can have a high daily and yearly variability. Wind power generates electricity from wind, but the wind speed can highly variable.[7] Even though these technologies have comparably low CO_2 emissions, these technologies are relatively impractical, when it comes to the provision of reliable energy.

Hydropower plants obtain energy from flowing water.[13] The three most common types of hydropower plants are run-of-river, storage and pumped storage.[13] The latter two use dams to store water in reservoirs in times of surplus. For pumped-storage plants water is actively pumped into the reservoir, while the storage water reservoir fully depends on the natural water inflow.[13] The stored water can then be released at flexible times, with short reaction times, and allows for electricity production during periods of peak energy demand.[13] Therefore, storage and pump storage are especially important to fulfil SDG 7 (Affordable and clean energy). In parallel to hydropower electricity production, hydropower reservoirs can also be used for other purposes like for example flood control, water supply or irrigation.[14] Thus, hydropower reservoirs can additionally contribute to fulfilling, for example, SDG 6 (Clean water and sanitation) (Figure 2), which aims to *"achieve universal and equitable access to safe and affordable drinking water for all"*.[4] Run-of-river plants use the natural river discharge, and normally do not have storage dams. Hence, these power plants can have a high variation in electricity production, as they fully depend on the natural river discharge. However, run-of-river plans can be built downstream of a storage or pumped storage hydropower plant, to improve the electricity production of the total hydropower system.[13] Due to unexploited technical hydropower potential, it is estimated that the global hydropower electricity production could be increased between 13,270 TWh yr^{-1} and 30,470 TWh yr^{-1},[15,16] or in other words by almost 10 times. Hydropower electricity production therefore has a huge potential to contribute to the fulfilment of the above-mentioned SGDs.

Despite the mentioned benefits of renewable electricity production, both UN Environment[7] and the IPCC[17] pointed out that there are potential ecological trade-offs related to renewable energy production. They indicate that there is a need to assess the environmental impacts of current and future renewable energy projects to identify trade-offs involved with increasing renewable energy electricity production. At the same time, it has been identified that the SDG nexus can also result in negative trade-offs when fulfilling specific SDG.[10] For the fulfilment of SDG 7 trade-offs with 65 indicators of the SDGs have been identified.[6]

Potential terrestrial and aquatic biodiversity impacts of renewable electricity production thereby may interfere with SDG 6 (Clean water and sanitation) which aims to *"protect and restore water-related ecosystems, including mountains, forests, wetlands, rivers, aquifers and lakes"*[4] and SDG 15 (Life on land) which aims to *"reduce the degradation of natural habitats, halt the loss of biodiversity and, by 2020, protect and prevent the extinction of threatened species"*(Figure 2).[4] Conservation of biodiversity has been identified as a key parameter for sustainable development,[4,18-20] as human-well-being relies on biodiversity and their ecosystem services.[21]

In addition to the SDGs, the Convention on Biological Diversity has adopted the Strategic Plan for Biodiversity for 2011– 2020 and set up the Aichi Biodiversity Targets.[22] The overall mission of these targets is to *"halt the loss of biodiversity"* and one of the five strategic goals is to *"reduce the direct pressures on biodiversity and promote sustainable use"*.[22] To reach their vision of *"Living in harmony with nature"* by 2050[22] a post-2020 global biodiversity framework is currently under development.[23]

Despite the SDGs and Aichi Targets, current species extinction rates are estimated to be 1,000 times higher than background extinction rates.[24] Rapidly increasing human pressures could further increase the species extinction rate[25] and cause a sixth species mass extinction event.[26] The Norwegian government, for example, also pointed out that hydropower (the main electricity source in Norway) has significant environmental impacts on Norwegian rivers that should be assessed and accounted for.[27]

Fostering renewable energy development, with minimized trade-offs between the SDGs,[9,28] thus requires an assessment of the renewable energy – biodiversity – climate nexus,[10] of both current and future renewable energy projects (Figure 2). This is the only way to ensure that electricty production benefits are maximised while at the same time the adverse impacts on the environment and biodiversity are minimsed.[7,17]

It has been pointed out that the tool Life Cycle Assessment can be used to assess nexus relationships and the UN Environment[7] recommends the use of Life Cycle Assessment (LCA) to compare such environmental impacts of renewable energy sources. LCA is an ISO standardized tool, which can be used to analyse the complete environmental impacts of a product or process throughout all life cycle stages.[29] Hence, it could help to identify at which locations renewable electricity can be produced with the lowest biodiversity impact.

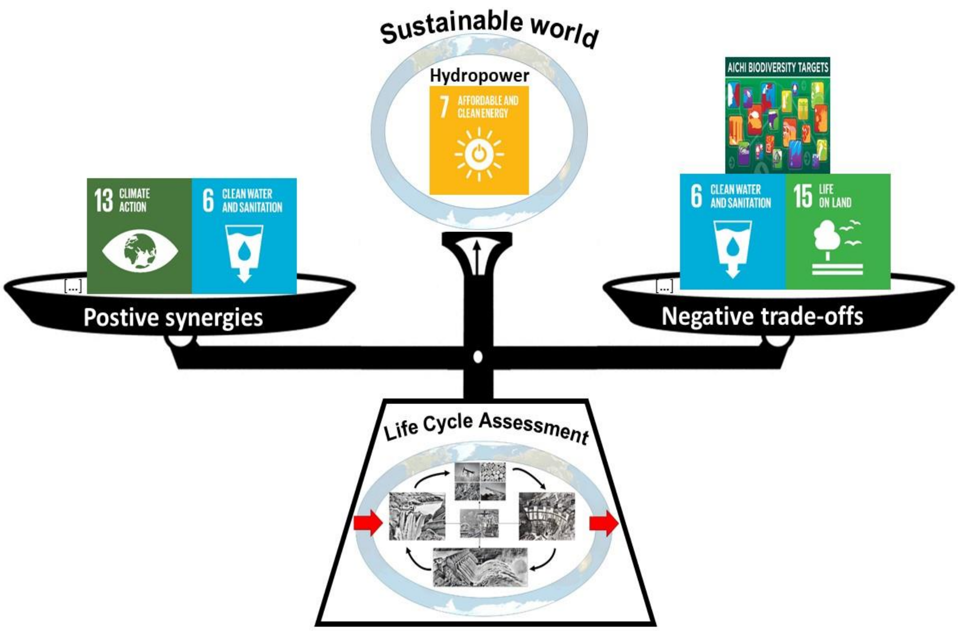


Figure 2: The transition into a sustainable world requires the fulfilment of SDG 7: Affordable and clean energy. Due to a nexus between the SDG goals,[10] fulfilment of SDG 7 can lead to both positive synergies and negative trade-offs with other SDGs.[10] Thereby, Life Cycle Assessment can contribute to find a balance between positive synergies (e.g. SDG 13: Climate action) and negative trade-offs (e.g. SDG 15: Life on land), which is an inevitable requirement for a transition into a more sustainable world. […] represents additional positive synergies and negative trade-offs (these are beyond the scope of this thesis).[6] SDG logos obtained from United Nations[30] and Achi logo from Taylor and Francis Online.[31]

Since hydropower is the largest source of renewable electricity production, this thesis is dedicated to quantifying potential biodiversity impacts of this technology (see overview in section 1.2) within the framework of Life Cycle Assessment (see section 1.3).

1.2 Biodiversity impacts from hydropower electricity production

A literature review by Gracey and Verones[32] identified, land use and land use change, freshwater habitat alteration and water quality degradation as the main cause-effect pathways for hydropower electricity production impacts on terrestrial and aquatic biodiversity.[32] In addition, climate change can be considered as an additional, fourth, cause-effect pathway (see Figure 3). Storage and pumped storage hydropower plants affect all of the four mentioned impact pathways, while run-of-river plants mainly cause freshwater habitat alteration.[32]

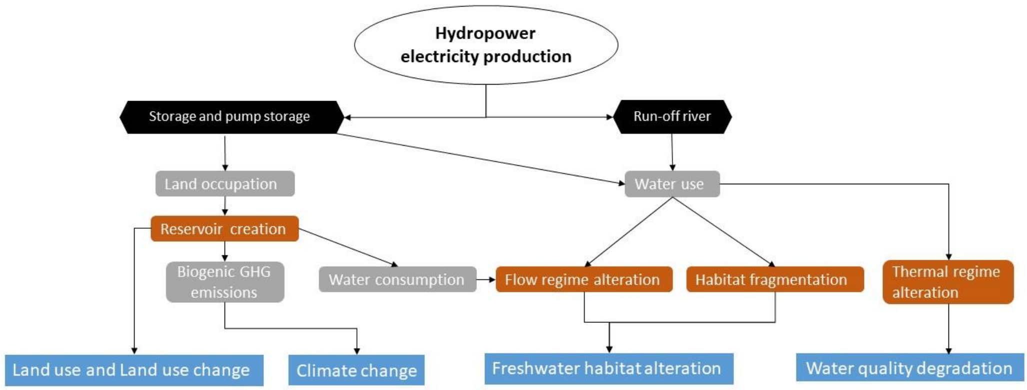


Figure 3: Overview of the main identified cause-effect pathways from hydropower electricity production on biodiversity, based on Gracey and Verones.[32] Black = type of hydropower plant; Grey = environmental stressor; Orange= environmental alteration; Blue= impact category.

1.2.1 Land use and land use change

Land use and land use change are caused by the inundation of land for the reservoirs of storage and pumped storage hydropower plants. The inundation can lead to a direct mortality of resident terrestrial species. Furthermore, the inundated land area becomes inhabitable for most terrestrial species and can subsequently lead to local species extinctions and biodiversity loss.[33] Several cases studies[34,35,36,37] have confirmed that the inundation of land causes habitat loss for resident terrestrial and in part freshwater flora and fauna. In this thesis, I define the inundation of terrestrial land as land occupation, because the recovery to the original biodiversity state is postponed as long as the water is present.[38] The construction of infrastructure like power lines[39] or access roads[33] can cause further land use change and can also affect resident terrestrial flora and fauna. In addition, both reservoirs and associated infrastructure can contribute to terrestrial habitat fragmentation.

Fragmentation can create barriers, lead to the isolation of individual habitat fragments and can reduce the amount of habitat available to species. This could lead to population declines and increased extinction risk of terrestrial species.[40]

1.2.2 Freshwater habitat alteration

Freshwater habitat alteration can be divided into alteration of natural river flow regimes and habitat fragmentation.

Hydropower electricity production can alter the natural flow regime of a river in two ways: reservoir operation and water evaporation. Reservoirs are used to store water in times of surplus (e.g. snow melt, heavy rainfall) and to produce electricity with a release of water during peak energy demand (e.g. heating in winter).[41] This operation regime and the associated non-consumptive water use[42] is not in accordance with the natural flow regime and commonly produces a stabilizing effect on a river's annual discharge by removing flow peaks. In rivers characterized by high snowmelt/runoff variability this non-consumptive water use can lead to a reduction of spring floods due to reservoir filling and to increased winter flows during peak demand.[43]

Reservoir filling of storage and pumped storage can raise water levels and inundate land. Thereby the terrestrial habitats are replaced by one water surface.[44] As long as the new water is ice-free, there will be continuous water evaporation. In contrast, before inundation a terrestrial area has limited water availability,[44] which limits the potential evaporation rate. Due to this increased evaporation,[44] hydropower electricity can lead to substantial increases in water consumption.[45,46]

The evaporation caused by land use change (i.e. inundation) of hydropower reservoirs is considered as water consumption,[42] and I use "water consumption" in this sense throughout the thesis. This water consumption can lead to alteration of the natural river flow regime by reducing the discharge downstream of the reservoir, thus also affecting the natural flow regime.

Although explained independently, flow regime alteration from reservoir operation and water evaporation are occurring at the same time. On a seasonal scale (e.g. monthly), the water use could offset the water consumption or lead to an increased water availability.

The natural river flow regime has been identified as a key variable for many fundamental ecological characteristics of riverine ecosystems.[47] Alteration of the natural river flow regime has been shown to result in a decline in species abundance and community diversity for macroinvertebrates, fish, and riparian species across all river types on a global scale.[48] Fish species are impacted independent of whether the natural flow is increased or decreased.[48] Both Teichert et al.[49] and Puffer et al.[50] showed evidence for a linear relation between growth of juvenile salmon and discharge rates during summer. Furthermore, migratory fish species require a minimum discharge to migrate[51] and a discharge falling below a certain threshold will stop migration.[52] In addition, hydropeaking, the fast release of a pulse of water can lead to stranding of fish or macroinvertebrates.[53,54] This can lead to cumulative mortalities of fish or macroinvertebrates and result in a significant fish loss and decrease of macroinvertebrate density.[53,54] Hydropeaking can additionally affect macroinvertebrate density by causing increased drift of macroinvertebrates or by reducing the habitat quality.[53]

Habitat fragmentation can be caused by run-of-river plants, but especially by dams, and the resulting loss of freshwater habitat connectivity can lead to the loss of isolated freshwater species populations.[55] Furthermore, the loss of connectivity reduces the dispersal possibility of freshwater species.[56] Fragmentation can effect both the upstream and downstream movement of fish species between feeding and spawning grounds.[56,57] The extirpation of fish populations has, inter alia, been attributed to a lack of upstream fish passage for anadromous fish migrations.[56] When fish are swimming downstream, they normally follow the strongest current,[57] which can be the one flowing into the hydropower plant water intake. As the water intake is connected to the hydropower turbine, the current can guide fish directly into the turbine, and especially adult fish have a high probability to get killed by rotating turbine blades.[57]

Furthermore, water-level regulations in the reservoir may not be in line with the natural water level fluctuations. This alteration can lead to erosion in the littoral zone, decreasing benthic invertebrate diversity, a decline in fish biomass, and to a shift in habitat use.[58]

1.2.3 Water quality degradation

Water quality degradation is mainly caused by releasing epilimnetic (warmer and lighter) or a hypolimnetic (colder and heavier) water from reservoirs.[59] This changes the thermal regime of the river and the potentially resulting rapid temperature changes can cause stress for freshwater fish.[60] This stress triggers a cascade of physiological responses, which can lead to reduction in health and fitness or even mortality.[60]

1.2.4 Climate change

Hydropower reservoirs can lead to the accumulation of biomass, which can get into the reservoir during the inundation of resident flora during the creation of the reservoir, be transported into the reservoir (via land and water) or can directly grow in the reservoir.[61] The decomposition of this biomass produces biogenic CO_2 and CH_4. In addition, N_2O forms during the denitrification of nitrogen from the biomass.[61] Therefore, hydropower reservoirs represent a potential source of greenhouse gas (GHG) emissions. The GHG emission can have a high variation between hydropower reservoirs,[62] from carbon sinks to values larger than the ones of fossil fuel power plants.[15] GHG emissions can contribute to climate change[11] and the potential related temperature increase can reduce the available habitat for species. If species are not able to disperse into new suitable habitat, climate change can accelerate the extinction of terrestrial species.[63] Climate change[11] is also expected to lead to a change in precipitation. If the precipitation is reduced,[11] the discharge is reduced, which can affect freshwater species.[64,65]

In summary, the operation of hydropower reservoirs could potentially contribute to habitat change, climate change and pollution; three of the five identified main drivers of human-induced biodiversity loss.[66,67] Hence, confirming that hydropower electricity production may interfere with SDG 6 (Clean Water and Sanitation) and SDG 15 (Life on Land).

1.3 Life Cycle Assessment

LCA is an ISO standardized tool, which can be used to analyse all relevant environmental impacts of a product or process throughout all life cycle stages (resource extraction, construction, operation, decommissioning).[29] LCA is used, for example, by the European Commission[68] and individual companies[69] for decision-making or communication towards consumers.[70] An LCA commonly summarizes the cumulative impact of a product or process, grouped within three areas of protection: human health, ecosystem quality and natural resources (Figure 4).[71] Biodiversity damage is one aspect of the impacts on ecosystem quality, which focuses on the intrinsic value of terrestrial, freshwater and marine ecosystems.[72]

To obtain the impact of a product or process in LCA, the following four steps are required: (1) Goal and Scope definition, (2) Life Cycle Inventory analysis, (3) Life Cycle Impact Assessment, and (4) Life cycle interpretation. In the following these steps are explained, especially pertaining to hydropower electricity production (Figure 4).

(1) The Goal and Scope definition, a Functional Unit for comparison is defined. As impacts of power generation are compared per unit of electricity produced,[73-75] the functional unit is usually one kWh hydropower electricity produced (Figure 4).

(2) In the Life Cycle Inventory (LCI) analysis, resource use (e.g. water consumption, land use, construction materials, energy) and emissions (e.g. CO_2 emissions) are collected over the entire lifespan, and for each life cycle stage (dam and power plant construction, reservoir operation and decommissioning[76]), including required inputs and outputs.[77] The dam and power plant construction requires material, which has to be sourced, transported and requires energy. Reservoir operation can occupy land, use and consume water and release biogenic GHG emissions. Decommissioning requires energy, transport, recycling, and waste treatment of the construction materials. The LCI collects the resource use and emissions of all these processes (Figure 4). It has been shown that the operation phase causes more water consumption and GHG emission then the dam construction.[78,79]

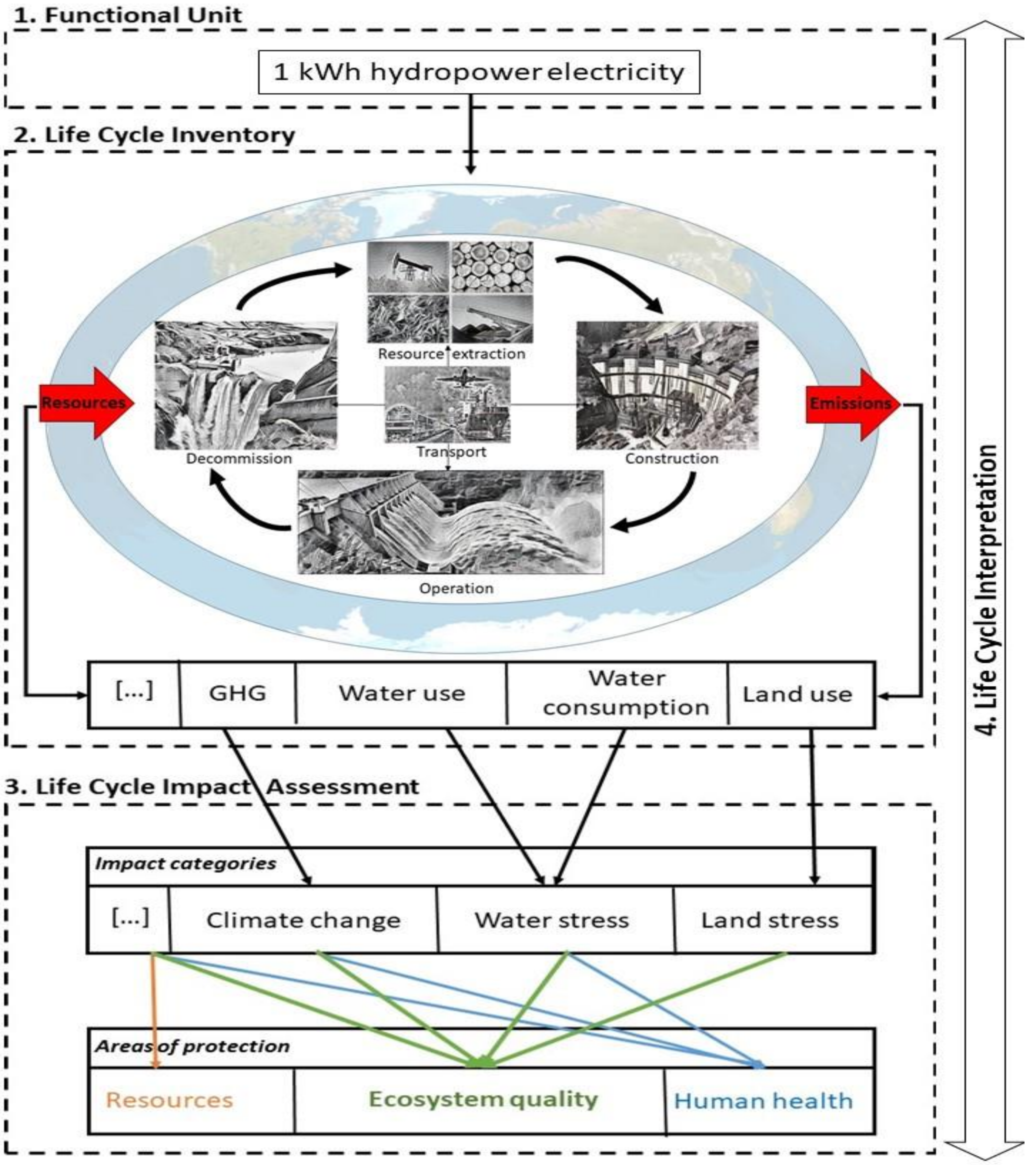


Figure 4: Visualization of the four required Life Cycle Assessment (LCA) steps to assess the environmental impact of hydropower electricity production. Example Life Cycle Inventory (LCI) parameters and impact categories are relevant for hydropower electricity production and ecosystem quality. […] underlines that in an LCA many more LCI parameters can be collected and that more impact categories exist.

For the calculation of inventory data related to the operational phase of storage and pumped storage hydropower reservoirs it is important to consider the natural water surface area before dam construction. This area cannot be inundated again, and cannot contribute to a land occupation. In addition, the evaporation of the natural water surface area before dam construction should not be attributed to water consumption from hydropower electricity production, because it already evaporated water naturally before dam construction. Due to the same reason, also the pre-construction evapotranspiration of the inundated land should be subtracted from water consumption. As reservoirs can be created either by damming a natural river or an already existing natural lake, the natural water surface before dam construction can be highly variable between hydropower reservoirs. Hence, the LCI parameters should be calculated using a so called “net” approach which is accounting for the conditions prior to dam construction.[80]

Relevant LCI data for hydropower electricty production are available in LCI databases, for example Ecoinvent.[32] But because environmental parameters, such as topographic and climatic conditions[76] can vary considerably between hydropower plant locations,[62,80] it is advisable to collect site-specific LCI data.[81,82] The Life Cycle Initiative underlines the need for site-specific LCI data as is encourages model developers to “*prioritize the development of regionalized inventories when high spatial variability is observed or expected*”.[83]

(3) In the Life Cycle Impact Assessment (LCIA) step the LCI data is converted into impact scores for different impact categories.[84] The potential biodiversity impacts from hydropower production (Section 1.2), would be assessed inside the LCIA impact categories: “land stress”, “water stress” and “climate change” (Figure 4). To transform the emission and resource use collected in the LCI step into potential environmental impacts, characterization factors (CF) are required. The CF itself consists of a fate factor (FF) and an effect factor (EF). The FF models the spatial distribution and intensity of a unit intervention (i.e. m^3 water consumed) and is generally obtained from environmental fate models. The EF relates the intensity of a unit of pressure to a quantified effect (i.e. biodiversity loss per m^3 water consumed). The LCIA steps assumes a linear relation between resource use or emissions and impacts.[85]

It is currently recommended, also due to a lack of alternatives, to assess the impacts on ecosystem quality with species-richness related metrics.[86] The Life Cycle Initiative hosted by UN Environment recommends to assess damage to ecosystem quality in the unit of Potentially Disappeared Fraction of Species (PDF).[87] A PDF is a fraction, and is calculated by dividing the number of potential disappeared species with the total number of present species.[88] Following, a PDF of 1 represents a potential loss of all species.

In addition, a regionalised LCIA model development is needed, as the environmental impact can be dependent on local conditions.[83] For example, occupation of 1 m^2 land in Antarctica, with low species richness, most likely has a smaller impact then occupying on 1 m^2 of tropical rainforest. To account for this variability site-specific CFs should be developed.[89] At the same time, the CFs need should have global coverage to ensure a comparison of products and processes on a global scale. The spatial resolution of global CFs, is normally defined by the resolution of global input data,[83] in particular the parameter with data available at the lowest scale. To perform a regionalized LCA, LCI data and LCIA models should match in terms of spatial resolution.[72,83]
It is also recommended to develop CFs for local, regional and global scales to reflect losses in local, regional and global biodiversity.[72] To convert regional PDFs (indicating regional species loss per region) into global PDFs (indicating global species extinctions per region, irreversible) conversion factors, such as the "Vulnerability Score",[90] have been developed. By accounting for geographic range and ICUN threat level of different species, the "Vulnerability Score" describes the ratio between threatened endemic species and total species richness.[90]

(4) The Life Cycle Interpretation stage can occur in parallel to steps 1-3 (Figure 4).[77] The results of an LCA should be used for a relative comparison, but not for an absolute comparison.[72]

Thus, an LCA summarizes the contribution of each life cycle stage to the total environmental impact. The results can be used to compare products and processes,[91] or alternative scenarios to improve the environmental performance of products and processes at various points in their life cycle.[92] For a single renewable energy source, like hydropower, LCA could help to identify at which locations hydropower electricity can be produced with the lowest ecosystem damage and during which life cycle stage most ecosystem damage is occurring. This information can help decision makers with strategic planning or priority setting, to achieve a more sustainable hydropower development. In addition, if an LCA is performed for other renewable energy sources, LCA could help to identify which renewable energy sources should be chosen to reach the highest possible share of renewable energy in the global energy mix, while ensuring the lowest possible biodiversity trade-off.

1.4 Research gap

So far, no operational LCA methodology exists to assess biodiversity impacts of freshwater habitat alteration, water quality degradation, or land use and land use change.[32] As a consequence, until now, most LCA studies dealing with hydropower impacts only account for environmental impacts in the form of GHG emissions.[93-95] Land use changes[96] and water consumption are rarely reported,[55] and no study accounted for biodiversity impacts.[32] Also the report from UN Environment on green energy choices[7] did not consider relevant potential biodiversity impacts of hydropower electricity production, like habitat loss, due to a lack of methods. From a sustainable development perspective this means that all these studies mainly focus on SDG 7 (Affordable and Clean Energy) and SDG 13 (Climate action), but overlook potential trade-offs with SDG 6 (Clean Water and Sanitation) and SDG 15 (Life on Land). The report from UN Environment on green energy choices[7] concludes that the "*assessment can be improved with updated or new impact assessment methods*".

This PhD work was part of the "Towards sustainable renewable energy production (SURE): Developing a Life Cycle Impact Assessment framework for biodiversity impacts" project, funded through the ENERGIX programme by the Research Council of Norway (Grant Number 244109).[97] The aim of the project was to develop methods that allow for the assessment of biodiversity impacts from (1) onshore wind power production and (2) hydropower production, within the framework of Life Cycle Assessment. The focus of this PhD work was on part (2), hydropower production.

An operational LCA model for hydropower electricity production requires site-specific inventory data.[81,82] However, currently, LCI databases (e.g. Ecoinvent[98]) only contain spatially-explicit LCI parameters related to hydropower reservoirs operation, such as land use and water consumption parameters, for two regions: Switzerland and Brazil.[76] Despite the fact that LCI parameters should be calculated with the “net” approach (see section 1.3),[80] all the currently available hydropower LCI operation parameters are related to the actual reservoir surface area[76] and represent so called “gross” parameters. For example, gross land occupation parameters are calculated by dividing the actual reservoir surface area with the annual hydropower electricity production, and gross water consumption parameters are calculated by dividing the evaporation of the actual reservoir surface area with the annual hydropower electricity production. Even outside LCA, most of the published water consumption[46,80] or GHG emission[79] estimates of hydropower electricity production represent gross values. Only a few studies calculated net water consumption values,[45,80] but they assumed that the state before dam construction reservoir was one uniform land type. The main reason for calculating gross values is the lack of information on the natural water surface area before dam construction, due to the age of the existing reservoirs.[99] The Global Reservoir and Dam database for example, collected information about 6,824 reservoirs, but only provides data for the previous existence of a possible natural lake for 104 reservoirs.[14] As a consequence, all currently available hydropower LCI gross parameters most likely represent overestimated values. Bakken et al.[80] reported a discrepancy of up to 60% between net and gross water consumption estimates. Therefore, using gross values in LCA also leads to an overestimation of the total environmental impact.[80] Furthermore, no appropriate LCI data related to water quality degradation (e.g. temperature of used water) exists. This highlights the research need for spatially explicit net LCI parameters related to hydropower electricity production.

Inside the LCIA impact category “land stress”, CFs could so far assess the land use change from one terrestrial habitat to another terrestrial habitat type[90,100,101] and from aquatic to terrestrial habitat.[102] However, no CFs covering land use change from terrestrial to aquatic habitat existed, even if this is a major change occurring during reservoir creation. Therefore, no CFs exist to quantify the land occupation impact of current and future hydropower electricity production. Considering that viable hydropower expansion plans might potentially lead to an inundation of up to 240,000 km^2 terrestrial habitat globally, the development of such CFs seems crucial for supporting sound environmental decision-making.[16]

To quantify water consumption impacts on aquatic biodiversity, inside the impact category "water stress", CFs have been developed for regions between 42° north and south and for Europe with focus on Switzerland.[65,103,104] The main reason for excluding areas at latitudes above 42° north is that these river basins were recently (in geological time) glaciated and have not had sufficient time yet to reach their maximum species richness potential .[43-45] As species richness can vary between regions,[105] these CFs should only be applied within the geographic range to which they pertain.[103] This means that biodiversity impacts from any water consumption occurring outside the so far covered regions cannot be assessed appropriately in LCA. As a result, LCA can presently not assess water consumption impacts of hydropower electricity production in countries such as Canada, Norway, Sweden, Finland and Iceland, which, together, accounted for 11.8% of the global hydropower electricity production in 2016.[12] More spatially-explicit CFs are needed to allow for the quantification of water consumption biodiversity impacts on a gloabal scale in LCA.[68,69,106]

For the cause-effect pathway "freshwater habitat alteration", non-consumptive water use is currently only considered in the LCI[32] and CFs quantifying biodiversity impacts from non-consumptive water use are completely lacking.

Appropriate methodology to assess biodiversity impacts for the cause-effect pathway "water quality degradation" is scarce but a CF assessing the impact on aquatic biodiversity of releasing warm water from a nuclear power plant into two rivers extist.[107]

For the impact category "climate change", global CFs exist assessing the impact of global warming on freshwater fish species[64,65] and terrestrial species.[63,84,108] They have not, however, been applied to quantify ecosystem damages of renewable electricity production.

1.5 Research aim

The main aim of this thesis is to advance and develop LCA related methods, to allow an operational assessment of biodiversity impacts of hydropower electricity production in LCA. To ensure compatibility with existing methods, the developed methods should be harmonized with the existing LCIA models and should follow current recommendations from the Life Cycle Initiative hosted by UN Environment.[87] The focus of this work is on biodiversity impacts related to water consumption and land occupation, i.e. two of the predominant cause-effect pathways of hydropower electricity production on biodiversity, as identified by Gracey and Verones.[32] Furthermore, these impacts are part of the existing LCIA impact categories “water stress” and “land stress”. In addition, are land use change and flow alteration among the main threats for terrestrial[25,109] and aquatic biodiversity globally.[110]

Due to data availability the models are in the first instance modelled for Norway, but all developed models shall be applicable globally, with the prerequisite of additional data availability.

The goals of this thesis are:

I. Calculate spatially-differentiated net land occupation values for Norwegian hydropower reservoirs. The developed method should be applicable globally.

II. Calculate net water consumption values for Norwegian hydropower reservoirs, by using the land occupation values obtained in goal I. The developed method should be applicable globally.

III. Develop spatially-differentiated CFs assessing the impacts of Norwegian water consumption on freshwater biodiversity. These CFs shall account for post-glacial fish migration history and the developed method should be applicable to all previously glaciated areas. The CFs will be primarily designed for hydropower electricity production, but should be applicable for other types of water consumption.

IV. Develop spatially-differentiated CFs with global coverage assessing the biodiversity impact of land inundation on terrestrial biodiversity. The CFs will be primarily designed for hydropower electricity production but should be applicable to other types of land inundation, for example, occurring from sea-level rise or land based aquaculture farming.

V. Apply the developed methods in combination with other existing CFs (e.g. on GHGs) to assess the potential biodiversity impact of the future remaining hydropower potential. This should show the applicability of the developed methods and highlight a potential biodiversity trade-off.

VI. Discuss the relevance and usability of the developed methods for LCA and highlight future research needs.

The research goals are addressed in five thesis chapters (visualised in Table 1):

- Chapter 2: Modeling net land occupation of hydropower reservoirs in Norway for use in Life Cycle Assessment
- Chapter 3: Quantifying net water consumption of Norwegian hydropower reservoirs and related aquatic biodiversity impacts in Life Cycle Assessment
- Chapter 4: Global characterization factors for biodiversity impacts of land inundation in Life Cycle Assessment
- Chapter 5: The potential to control biodiversity impacts of future global hydropower reservoirs by strategic site selection
- Chapter 6: Conclusion and Outlook

Table 1: Connection between the five thesis chapters and the six research goals of the thesis. ✓= covered.

	Research goal					
Chapter	**I**	**II**	**III**	**IV**	**V**	**VI**
2	✓					
3		✓	✓			
4				✓		
5					✓	
6						✓

1.6 References

1. Steffen, W.; Richardson, K.; Rockstrom, J.; Cornell, S. E.; Fetzer, I.; Bennett, E. M.; Biggs, R.; Carpenter, S. R.; de Vries, W.; de Wit, C. A.; Folke, C.; Gerten, D.; Heinke, J.; Mace, G. M.; Persson, L. M.; Ramanathan, V.; Reyers, B.; Sorlin, S., Planetary boundaries: Guiding human development on a changing planet. *Science* **2015,** *347*, (6223), 1259855-1259855.
2. O'Neill, D. W.; Fanning, A. L.; Lamb, W. F.; Steinberger, J. K., A good life for all within planetary boundaries. *Nature Sustainability* **2018,** *1*, (2), 88-95.
3. Cash, D. W.; Clark, W. C.; Alcock, F.; Dickson, N. M.; Eckley, N.; Guston, D. H.; Jäger, J.; Mitchell, R. B., Knowledge systems for sustainable development. *Proceedings of the National Academy of Sciences* **2003,** *100*, (14), 8086-8091.
4. United Nations *Transforming our world: The 2030 agenda for sustainable development - A/RES/70/1*; 2015.
5. Azote Images for Stockholm Resilience Centre, Sustainable development goals. https://www.stockholmresilience.org/research/research-news/2016-06-14-how-food-connects-all-the-sdgs.html
6. Fuso Nerini, F.; Tomei, J.; To, L. S.; Bisaga, I.; Parikh, P.; Black, M.; Borrion, A.; Spataru, C.; Castán Broto, V.; Anandarajah, G.; Milligan, B.; Mulugetta, Y., Mapping synergies and trade-offs between energy and the Sustainable Development Goals. *Nature Energy* **2018,** *3*, (1), 10-15.
7. Hertwich, E.; de Larderel, J. A.; Arvesen, A.; Bayer, P.; Bergesen, J.; Bouman, E.; Gibon, T.; Heath, G.; Peña, C.; Purohit, P. *Green Energy Choices: The benefits, risks, and trade-offs of low-carbon technologies for electricity production*; United Nation Environmental Program 2016.
8. Le Blanc, D., Towards Integration at Last? The Sustainable Development Goals as a Network of Targets. *Sustainable Development* **2015,** *23*, (3), 176-187.
9. Nilsson, M.; Griggs, D.; Visbeck, M., Policy: Map the interactions between Sustainable Development Goals. *Nature* **2016,** *534*, (7607), 320-322.
10. Liu, J.; Hull, V.; Godfray, H. C. J.; Tilman, D.; Gleick, P.; Hoff, H.; Pahl-Wostl, C.; Xu, Z.; Chung, M. G.; Sun, J.; Li, S., Nexus approaches to global sustainable development. *Nature Sustainability* **2018,** *1*, (9), 466-476.
11. Intergovernmental Panel on Climate Change *Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty [V. Masson-Delmotte, P. Zhai, H. O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J. B. R. Matthews, Y. Chen, X. Zhou, M. I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, T. Waterfield (eds.)]*; In Press., 2018.
12. IEA, *Renewables 2017*. 2017.
13. Egré, D.; Milewski, J. C., The diversity of hydropower projects. *Energy Policy* **2002,** *30*, (14), 1225-1230.
14. Lehner, B.; Liermann, C. R.; Revenga, C.; Vörösmarty, C.; Fekete, B.; Crouzet, P.; Döll, P.; Endejan, M.; Frenken, K.; Magome, J.; Nilsson, C.; Robertson, J. C.; Rödel, R.; Sindorf, N.; Wisser, D., High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. *Frontiers in Ecology and the Environment* **2011,** *9*, (9), 494-502.
15. Kumar, A.; Schei, T.; Ahenkorah, A.; Rodriguez, R. C.; Devernay, J.-M.; Freitas, M.; Hall, D.; Killingtveit, Å.; Liu, Z., Hydropower. In *IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation*, Edenhofer, O.; Pichs-Madruga, R.; Sokona, Y.; Seyboth, K.; Matschoss, P.; Kadner, S.; Zwickel, T.; Eickemeier, P.; Hansen, G.; Schlömer, S.; von Stechow,

C., Eds. Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA, 2011.
16. Gernaat, D. E. H. J.; Bogaart, P. W.; Vuuren, D. P. v.; Biemans, H.; Niessink, R., High-resolution assessment of global technical and economic hydropower potential. *Nature Energy* **2017,** *2*, (10), 821-828.
17. Edenhofer, O.; Pichs-Madruga, R.; Sokona, Y.; Seyboth, K.; Matschoss, P.; Kadner, S.; Zwickel, T.; Eickemeier, P.; Hansen, G.; Schlömer, S. *IPCC special report on renewable energy sources and climate change mitigation*; 2011.
18. Secretariat of the Convention on Biological Diversity *Global Biodiversity Outlook 4*; Montreal, 2014.
19. Bennett, E. M.; Cramer, W.; Begossi, A.; Cundill, G.; Díaz, S.; Egoh, B. N.; Geijzendorffer, I. R.; Krug, C. B.; Lavorel, S.; Lazos, E.; Lebel, L.; Martín-López, B.; Meyfroidt, P.; Mooney, H. A.; Nel, J. L.; Pascual, U.; Payet, K.; Harguindeguy, N. P.; Peterson, G. D.; Prieur-Richard, A.-H.; Reyers, B.; Roebeling, P.; Seppelt, R.; Solan, M.; Tschakert, P.; Tscharntke, T.; Turner, B. L.; Verburg, P. H.; Viglizzo, E. F.; White, P. C. L.; Woodward, G., Linking biodiversity, ecosystem services, and human well-being: three challenges for designing research for sustainability. *Current Opinion in Environmental Sustainability* **2015,** *14*, 76-85.
20. Opoku, A., Biodiversity and the built environment: Implications for the Sustainable Development Goals (SDGs). *Resources, Conservation and Recycling* **2019,** *141*, 1-7.
21. Mace, G. M.; Norris, K.; Fitter, A. H., Biodiversity and ecosystem services: a multilayered relationship. *Trends in Ecology & Evolution* **2012,** *27*, (1), 19-26.
22. Secretariat of the Convention on Biological Diversity *Strategic Plan for Biodiversity 2011–2020 and the Aichi Targets*; Montreal, Quebec, Canada 2010.
23. Secretariat of the Convention on Biological Diversity Preparations for the Post-2020 Biodiversity Framework. https://www.cbd.int/post2020/
24. Pimm, S. L.; Jenkins, C. N.; Abell, R.; Brooks, T. M.; Gittleman, J. L.; Joppa, L. N.; Raven, P. H.; Roberts, C. M.; Sexton, J. O., The biodiversity of species and their rates of extinction, distribution, and protection. *Science* **2014,** *344*, (6187), 1246752-1246752.
25. Sala, O. E.; Chapin, F. S.; Armesto, J. J.; Berlow, E.; Bloomfield, J.; Dirzo, R.; Huber-Sanwald, E.; Huenneke, L. F.; Jackson, R. B.; Kinzig, A.; Leemans, R.; Lodge, D. M.; Mooney, H. A.; Oesterheld, M.; Poff, N. L.; Sykes, M. T.; Walker, B. H.; Walker, M.; Wall, D. H., Global biodiversity scenarios for the year 2100. *Science* **2000,** *287*, (5459), 1770-1774.
26. Barnosky, A. D.; Matzke, N.; Tomiya, S.; Wogan, G. O. U.; Swartz, B.; Quental, T. B.; Marshall, C.; McGuire, J. L.; Lindsey, E. L.; Maguire, K. C.; Mersey, B.; Ferrer, E. A., Has the Earth's sixth mass extinction already arrived? *Nature* **2011,** *471*, (7336), 51-57.
27. Norwegian Government Ministries and Offices, Meld. St. 25 (2015–2016) Power to Change - Energy policy 2030 (In Norwegian).
28. Bhaduri, A.; Bogardi, J.; Siddiqi, A.; Voigt, H.; Vörösmarty, C.; Pahl-Wostl, C.; Bunn, S. E.; Shrivastava, P.; Lawford, R.; Foster, S.; Kremer, H.; Renaud, F. G.; Bruns, A.; Osuna, V. R., Achieving Sustainable Development Goals from a Water Perspective. *Frontiers in Environmental Science* **2016,** *4*.
29. ISO *ISO 14044: Environmental management — Life cycle assessment — Principles and framework*; International Organisation for Standardization: Geneva, Switzerland, 2006.
30. United Nations. Communications materials. https://www.un.org/sustainabledevelopment/news/communications-material/

31. Taylor and Francis Online. Aichi Biodiversity Targets Article Collection. http://explore.tandfonline.com/page/est/aichi-biodiversity-targets-hub
32. Gracey, E. O.; Verones, F., Impacts from hydropower production on biodiversity in an LCA framework—review and recommendations. *The International Journal of Life Cycle Assessment* **2016,** *21*, (3), 412-428.
33. McAllister, D. E.; Craig, J. F.; Davidson, N.; Delany, S.; Seddon, M. *Biodiversity impacts of large dams*; 2001.
34. Alho, C. J., Environmental effects of hydropower reservoirs on wild mammals and freshwater turtles in Amazonia: a review. *Oecologia Australis* **2011,** *15*, (3), 593-604.
35. Kitzes, J.; Shirley, R., Estimating biodiversity impacts without field surveys: A case study in northern Borneo. *Ambio* **2016,** *45*, (1), 110-119.
36. Zhang, J.; Zhengjun, L.; Xiaoxia, S., Changing landscape in the Three Gorges Reservoir Area of Yangtze River from 1977 to 2005: Land use/land cover, vegetation cover changes estimated using multi-source satellite data. *International Journal of Applied Earth Observation and Geoinformation* **2009,** *11*, (6), 403-412.
37. Tefera, B.; Sterk, G., Hydropower-Induced Land Use Change in Fincha'a Watershed, Western Ethiopia: Analysis and Impacts. *Mountain Research and Development* **2008,** *28*, (1), 72-80.
38. Dorber, M.; May, R.; Verones, F., Modeling Net Land Occupation of Hydropower Reservoirs in Norway for Use in Life Cycle Assessment. *Environmental Science & Technology* **2018,** *52*, (4), 2375-2384.
39. Gundula, S. B.; Roel, M.; Kjetil, B.; Sigbjorn, S.; Eivin, R., The effects of power lines on ungulates and implications for power line routing and rights-of-way management. *International Journal of Biodiversity and Conservation* **2014,** *6*, (9), 647-662.
40. Ewers, R. M.; Didham, R. K., Confounding factors in the detection of species responses to habitat fragmentation. *Biological Reviews* **2005,** *81*, (01).
41. SSB 08583: Elektrisitetsbalanse (MWh) etter produksjon, statistikkvariabel og måned. https://www.ssb.no/statbank/table/08583/tableViewLayout1/
42. ISO *ISO 14046- Environmental management -- Water footprint -- Principles, requirements and guidelines*; International Organisation for Standardization: 2014.
43. Renöfält, B. M.; Jansson, R.; Nilsson, C., Effects of hydropower generation and opportunities for environmental flow management in Swedish riverine ecosystems. *Freshwater Biology* **2010,** *55*, (1), 49-67.
44. Strachan, I. B.; Tremblay, A.; Pelletier, L.; Tardif, S.; Turpin, C.; Nugent, K. A., Does the creation of a boreal hydroelectric reservoir result in a net change in evaporation? *Journal of Hydrology* **2016,** *540*, 886-899.
45. Scherer, L.; Pfister, S., Global water footprint assessment of hydropower. *Renewable Energy* **2016,** *99*, 711-720.
46. Mekonnen, M. M.; Hoekstra, A. Y., The blue water footprint of electricity from hydropower. *Hydrology and Earth System Sciences* **2012,** *16*, (1), 179-187.
47. Poff, N. L.; Allan, J. D.; Bain, M. B.; Karr, J. R.; Prestegaard, K. L.; Richter, B. D.; Sparks, R. E.; Stromberg, J. C., The Natural Flow Regime. *BioScience* **1997,** *47*, (11), 769-784.
48. Poff, N. L.; Zimmerman, J. K. H., Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. *Freshwater Biology* **2010,** *55*, (1), 194-205.

49. Teichert, M. A. K.; Kvingedal, E.; Forseth, T.; Ugedal, O.; Finstad, A. G., Effects of discharge and local density on the growth of juvenile Atlantic salmon Salmo salar. *Journal of Fish Biology* **2010,** *76*, (7), 1751-1769.
50. Puffer, M.; Berg, O. K.; Huusko, A.; Vehanen, T.; Forseth, T.; Einum, S., Seasonal Effects of Hydropeaking on Growth, Energetics and Movement of Juvenile Atlantic Salmon (Salmo Salar). *River Research and Applications* **2015,** *31*, (9), 1101-1108.
51. Quinn, T. P.; Hodgson, S.; Peven, C., Temperature, flow, and the migration of adult sockeye salmon (Oncorhynchus nerka) in the Columbia River. *Canadian Journal of Fisheries and Aquatic Sciences* **1997,** *54*, (6), 1349-1360.
52. Haro, A.; Castro-Santos, T.; Noreika, J.; Odeh, M., Swimming performance of upstream migrant fishes in open-channel flow: a new approach to predicting passage through velocity barriers. *Canadian Journal of Fisheries and Aquatic Sciences* **2004,** *61*, (9), 1590-1601.
53. Kjaerstad, G.; Arnekleiv, J. V.; Speed, J. D. M.; Herland, A. K., Effects of hydropeaking on benthic invertebrate community composition in two central Norwegian rivers. *River Research and Applications* **2018,** *34*, (3), 218-231.
54. Young, P. S.; Cech, J. J.; Thompson, L. C., Hydropower-related pulsed-flow impacts on stream fishes: a brief review, conceptual model, knowledge gaps, and research needs. *Reviews in Fish Biology and Fisheries* **2011,** *21*, (4), 713-731.
55. Fuller, M. R.; Doyle, M. W.; Strayer, D. L., Causes and consequences of habitat fragmentation in river networks. *Annals of the New York Academy of Sciences* **2015,** *1355*, (1), 31-51.
56. Silva, A. T.; Lucas, M. C.; Castro-Santos, T.; Katopodis, C.; Baumgartner, L. J.; Thiem, J. D.; Aarestrup, K.; Pompeu, P. S.; O'Brien, G. C.; Braun, D. C.; Burnett, N. J.; Zhu, D. Z.; Fjeldstad, H.-P.; Forseth, T.; Rajaratnam, N.; Williams, J. G.; Cooke, S. J., The future of fish passage science, engineering, and practice. *Fish and Fisheries* **2018,** *19*, (2), 340-362.
57. Fjeldstad, H.-P.; Pulg, U.; Forseth, T., Safe two-way migration for salmonids and eel past hydropower structures in Europe: a review and recommendations for best-practice solutions. *Marine and Freshwater Research* **2018,** *69*, (12).
58. Eloranta, A. P.; Finstad, A. G.; Helland, I. P.; Ugedal, O.; Power, M., Hydropower impacts on reservoir fish populations are modified by environmental variation. *Science of The Total Environment* **2018,** *618*, 313-322.
59. Arthington, A. H.; Naiman, R. J.; McClain, M. E.; Nilsson, C., Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities. *Freshwater Biology* **2010,** *55*, (1), 1-16.
60. Donaldson, M. R.; Cooke, S. J.; Patterson, D. A.; Macdonald, J. S., Cold shock and fish. *Journal of Fish Biology* **2008,** *73*, (7), 1491-1530.
61. Hertwich, E. G., Addressing Biogenic Greenhouse Gas Emissions from Hydropower in LCA. *Environmental Science & Technology* **2013,** *47*, (17), 9604-9611.
62. Deemer, B. R.; Harrison, J. A.; Li, S.; Beaulieu, J. J.; DelSontro, T.; Barros, N.; Bezerra-Neto, J. F.; Powers, S. M.; dos Santos, M. A.; Vonk, J. A., Greenhouse Gas Emissions from Reservoir Water Surfaces: A New Global Synthesis. *BioScience* **2016,** *66*, (11), 949-964.
63. Urban, M. C., Accelerating extinction risk from climate change. *Science* **2015,** *348*, (6234), 571-573.
64. Xenopoulos, M. A.; Lodge, D. M.; Alcamo, J.; Märker, M.; Schulze, K.; Van Vuuren, D. P., Scenarios of freshwater fish extinctions from climate change and water withdrawal. *Global Change Biology* **2005,** *11*, (10), 1557-1564.

65. Hanafiah, M. M.; Xenopoulos, M. A.; Pfister, S.; Leuven, R. S. E. W.; Huijbregts, M. A. J., Characterization Factors for Water Consumption and Greenhouse Gas Emissions Based on Freshwater Fish Species Extinction. *Environmental Science & Technology* **2011,** *45*, (12), 5272-5278.
66. Pereira, H. M.; Navarro, L. M.; Martins, I. S., Global biodiversity change: the bad, the good, and the unknown. *Annual Review of Environment Resources* **2012,** *37*.
67. Millennnium Ecosystem Assessment *Ecosystems and human wellbeing: biodiversity synthesis*; Washington, DC, 2005.
68. European Commission. The European Platform on Life Cycle Assessment. http://eplca.jrc.ec.europa.eu/
69. Unilever Lifecycle Assessments. http://www.unilever.com/sustainable-living-2014/our-approach-to-sustainability/our-strategy/lifecycle-assessments/
70. Hellweg, S.; Mila i Canals, L., Emerging approaches, challenges and opportunities in life cycle assessment. *Science* **2014,** *344*, (6188), 1109-1113.
71. Curran, M.; De Baan, L.; De Schryver, A. M.; Van Zelm, R.; Hellweg, S.; Koellner, T.; Sonnemann, G.; Huijbregts, M. A. J., Toward meaningful end points of biodiversity in life cycle assessment. *Environmental Science and Technology* **2011,** *45*, (1), 70-79.
72. Jolliet, O.; Antón, A.; Boulay, A.-M.; Cherubini, F.; Fantke, P.; Levasseur, A.; McKone, T. E.; Michelsen, O.; Milà i Canals, L.; Motoshita, M.; Pfister, S.; Verones, F.; Vigon, B.; Frischknecht, R., Global guidance on environmental life cycle impact assessment indicators: impacts of climate change, fine particulate matter formation, water consumption and land use. *The International Journal of Life Cycle Assessment* **2018**.
73. Hertwich, E. G.; Gibon, T.; Bouman, E. A.; Arvesen, A.; Suh, S.; Heath, G. A.; Bergesen, J. D.; Ramirez, A.; Vega, M. I.; Shi, L., Integrated life-cycle assessment of electricity-supply scenarios confirms global environmental benefit of low-carbon technologies. *Proceedings of the National Academy of Sciences* **2015,** *112*, (20), 6277-6282.
74. Denholm, P.; Hand, M.; Jackson, M.; Ong, S., Land-use requirements of modern wind power plants in the United States. *Golden, Colorado: National Renewable Energy Laboratory* **2009**.
75. Fthenakis, V.; Kim, H. C., Land use and electricity generation: A life-cycle analysis. *Renewable and Sustainable Energy Reviews* **2009,** *13*, (6-7), 1465-1474.
76. Flury, K.; Frischknecht, R., Life cycle inventories of hydroelectric power generation. *ESU-Services, Fair Consulting in Sustainability, commissioned by€ Oko-Institute eV* **2012**, 1-51.
77. Rebitzer, G.; Ekvall, T.; Frischknecht, R.; Hunkeler, D.; Norris, G.; Rydberg, T.; Schmidt, W.-P.; Suh, S.; Weidema, B. P.; Pennington, D. W., Life cycle assessment: Part 1: Framework, goal and scope definition, inventory analysis, and applications. *Environment international* **2004,** *30*, (5), 701-720.
78. Bakken, T. H.; Modahl, I. S.; Engeland, K.; Raadal, H. L.; Arnøy, S., The life-cycle water footprint of two hydropower projects in Norway. *Journal of Cleaner Production* **2016,** *113*, 241-250.
79. Scherer, L.; Pfister, S., Hydropower's Biogenic Carbon Footprint. *Plos One* **2016,** *11*, (9), e0161947.
80. Bakken, T. H.; Killingtveit, Å.; Engeland, K.; Alfredsen, K.; Harby, A., Water consumption from hydropower plants - review of published estimates and an assessment of the concept. *Hydrology and Earth System Sciences* **2013,** *17*, (10), 3983-4000.

81. Mutel, C. L.; Hellweg, S., Regionalized life cycle assessment: computational methodology and application to inventory databases. *Environmental Science & Technology* **2009,** *43*, (15), 5797-5803.
82. Mutel, C. L.; Pfister, S.; Hellweg, S., GIS-Based Regionalized Life Cycle Assessment: How Big Is Small Enough? Methodology and Case Study of Electricity Generation. *Environmental Science & Technology* **2012,** *46*, (2), 1096-1103.
83. Mutel, C.; Liao, X.; Patouillard, L.; Bare, J.; Fantke, P.; Frischknecht, R.; Hauschild, M.; Jolliet, O.; Maia de Souza, D.; Laurent, A.; Pfister, S., Overview and recommendations for regionalized life cycle impact assessment. *The International Journal of Life Cycle Assessment* **2018**.
84. Huijbregts, M. A. J.; Steinmann, Z. J. N.; Elshout, P. M. F.; Stam, G.; Verones, F.; Vieira, M.; Zijp, M.; Hollander, A.; van Zelm, R., ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. *The International Journal of Life Cycle Assessment* **2016,** *22*, (2), 138-147.
85. Huijbregts, M. A. J.; Hellweg, S.; Hertwich, E., Do We Need a Paradigm Shift in Life Cycle Impact Assessment? *Environmental Science & Technology* **2011,** *45*, (9), 3833-3834.
86. Woods, J. S.; Damiani, M.; Fantke, P.; Henderson, A. D.; Johnston, J. M.; Bare, J.; Sala, S.; Maia de Souza, D.; Pfister, S.; Posthuma, L.; Rosenbaum, R. K.; Verones, F., Ecosystem quality in LCIA: status quo, harmonization, and suggestions for the way forward. *The International Journal of Life Cycle Assessment* **2017**, 1995–2006.
87. Verones, F.; Bare, J.; Bulle, C.; Frischknecht, R.; Hauschild, M.; Hellweg, S.; Henderson, A.; Jolliet, O.; Laurent, A.; Liao, X., LCIA framework and cross-cutting issues guidance within the UNEP-SETAC Life Cycle Initiative. *Journal of cleaner production* **2017,** *161*, 957-967.
88. Curran, M.; de Baan, L.; De Schryver, A. M.; van Zelm, R.; Hellweg, S.; Koellner, T.; Sonnemann, G.; Huijbregts, M. A. J., Toward Meaningful End Points of Biodiversity in Life Cycle Assessment†. *Environmental Science & Technology* **2011,** *45*, (1), 70-79.
89. Hauschild, M., Spatial Differentiation in Life Cycle Impact Assessment: A decade of method development to increase the environmental realism of LCIA. *The International Journal of Life Cycle Assessment* **2006,** *11*, (S1), 11-13.
90. Chaudhary, A.; Verones, F.; de Baan, L.; Hellweg, S., Quantifying Land Use Impacts on Biodiversity: Combining Species–Area Models and Vulnerability Indicators. *Environmental Science & Technology* **2015,** *49*, (16), 9987-9995.
91. Klöpffer, W., *Background and future prospects in life cycle assessment*. Springer Science & Business Media: 2014.
92. International Standard Organization *ISO 14044: Environmental management — Life cycle assessment — Principles and framework*; International Organisation for Standardization: Geneva, Switzerland, 2006.
93. Raadal, H. L.; Gagnon, L.; Modahl, I. S.; Hanssen, O. J., Life cycle greenhouse gas (GHG) emissions from the generation of wind and hydro power. *Renewable and Sustainable Energy Reviews* **2011,** *15*, (7), 3417-3422.
94. Varun; Bhat, I. K.; Prakash, R., LCA of renewable energy for electricity generation systems—A review. *Renewable and Sustainable Energy Reviews* **2009,** *13*, (5), 1067-1073.
95. Turconi, R.; Boldrin, A.; Astrup, T., Life cycle assessment (LCA) of electricity generation technologies: Overview, comparability and limitations. *Renewable and Sustainable Energy Reviews* **2013,** *28*, 555-565.

96. Evans, A.; Strezov, V.; Evans, T. J., Assessment of sustainability indicators for renewable energy technologies. *Renewable and Sustainable Energy Reviews* **2009,** *13*, (5), 1082-1088.
97. Reseach Council of Norway. Towards sustainable renewable energy production: Developing a Life Cycle Impact Assessment framework for biodiversity impacts. https://www.forskningsradet.no/prosjektbanken/#/project/NFR/244109/Sprak=en
98. Wernet, G.; Bauer, C.; Steubing, B.; Reinhard, J.; Moreno-Ruiz, E.; Weidema, B., The ecoinvent database version 3 (part I): overview and methodology. *The International Journal of Life Cycle Assessment* **2016,** *21*, (9), 1218-1230.
99. NVE (The Norwegian Water Resources and Energy Directorate). NVE Atlas. http://atlas.nve.no
100. Chaudhary, A.; Brooks, T. M., Land Use Intensity-Specific Global Characterization Factors to Assess Product Biodiversity Footprints. *Environmental Science & Technology* **2018,** *52*, (9), 5094-5104.
101. de Baan, L.; Mutel, C. L.; Curran, M.; Hellweg, S.; Koellner, T., Land Use in Life Cycle Assessment: Global Characterization Factors Based on Regional and Global Potential Species Extinction. *Environmental Science & Technology* **2013,** *47*, (16), 9281-9290.
102. Verones, F.; Saner, D.; Pfister, S.; Baisero, D.; Rondinini, C.; Hellweg, S., Effects of Consumptive Water Use on Biodiversity in Wetlands of International Importance. *Environmental Science & Technology* **2013,** *47*, (21), 12248-12257.
103. Tendall, D. M.; Hellweg, S.; Pfister, S.; Huijbregts, M. A. J.; Gaillard, G., Impacts of River Water Consumption on Aquatic Biodiversity in Life Cycle Assessment—A Proposed Method, and a Case Study for Europe. *Environmental Science & Technology* **2014,** *48*, (6), 3236-3244.
104. Verones, F.; Pfister, S.; Hellweg, S., Quantifying Area Changes of Internationally Important Wetlands Due to Water Consumption in LCA. *Environmental Science & Technology* **2013,** *47*, (17), 9799-9807.
105. McGarvey, D. J.; Ward, G. M., Scale dependence in the species-discharge relationship for fishes of the southeastern U.S.A. *Freshwater Biology* **2008**, 2206-2219.
106. Núñez, M.; Bouchard, C. R.; Bulle, C.; Boulay, A.-M.; Margni, M., Critical analysis of life cycle impact assessment methods addressing consequences of freshwater use on ecosystems and recommendations for future method development. *The International Journal of Life Cycle Assessment* **2016,** *21*, (12), 1799-1815.
107. Verones, F.; Hanafiah, M. M.; Pfister, S.; Huijbregts, M. A. J.; Pelletier, G. J.; Koehler, A., Characterization Factors for Thermal Pollution in Freshwater Aquatic Environments. *Environmental Science & Technology* **2010,** *44*, (24), 9364-9369.
108. LC-Impact Version 0.5 - A spatially differentiated life cycle impact assessment approach. https://lc-impact.eu/about.html
109. Newbold, T.; Hudson, L. N.; Hill, S. L. L.; Contu, S.; Lysenko, I.; Senior, R. A.; Borger, L.; Bennett, D. J.; Choimes, A.; Collen, B.; Day, J.; De Palma, A.; Diaz, S.; Echeverria-Londono, S.; Edgar, M. J.; Feldman, A.; Garon, M.; Harrison, M. L. K.; Alhusseini, T.; Ingram, D. J.; Itescu, Y.; Kattge, J.; Kemp, V.; Kirkpatrick, L.; Kleyer, M.; Correia, D. L. P.; Martin, C. D.; Meiri, S.; Novosolov, M.; Pan, Y.; Phillips, H. R. P.; Purves, D. W.; Robinson, A.; Simpson, J.; Tuck, S. L.; Weiher, E.; White, H. J.; Ewers, R. M.; Mace, G. M.; Scharlemann, J. P. W.; Purvis, A., Global effects of land use on local terrestrial biodiversity. *Nature* **2015,** *520*, (7545), 45-50.
110. Vörösmarty, C. J.; McIntyre, P. B.; Gessner, M. O.; Dudgeon, D.; Prusevich, A.; Green, P.; Glidden, S.; Bunn, S. E.; Sullivan, C. A.; Liermann, C. R.; Davies, P. M., Global threats to human water security and river biodiversity. *Nature* **2010,** *467*, (7315), 555-561.

Chapter 2: Modeling net land occupation of hydropower reservoirs in Norway for use in Life Cycle Assessment

Environmental Science & Technology 2018, *52*, (4), 2375-2384.

Environmental Science & Technology

Article

Cite This: Environ. Sci. Technol. 2018, 52, 2375−2384

pubs.acs.org/est

Modeling Net Land Occupation of Hydropower Reservoirs in Norway for Use in Life Cycle Assessment

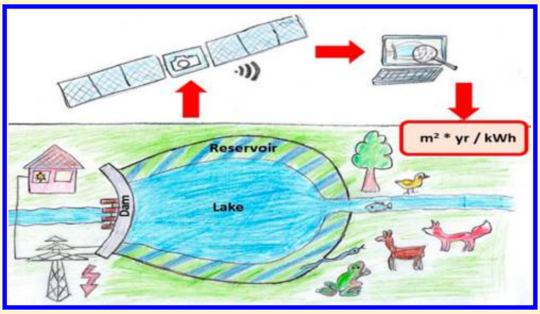
Martin Dorber,*,† Roel May,‡ and Francesca Verones†

†Department of Energy and Process Engineering, Norwegian University of Science and Technology (NTNU), Sem Sælands vei 7, 7491 Trondheim, Norway

‡Norwegian Institute for Nature Research (NINA), Høgskoleringen 9, 7034 Trondheim, Norway

Supporting Information

ABSTRACT: Increasing hydropower electricity production constitutes a unique opportunity to mitigate climate change impacts. However, hydropower electricity production also impacts aquatic and terrestrial biodiversity through freshwater habitat alteration, water quality degradation, and land use and land use change (LULUC). Today, no operational model exists that covers any of these cause-effect pathways within life cycle assessment (LCA). This paper contributes to the assessment of LULUC impacts of hydropower electricity production in Norway in LCA. We quantified the inundated land area associated with 107 hydropower reservoirs with remote sensing data and related it to yearly electricity production. Therewith, we calculated an average net land occupation of 0.027 $m^2 \cdot yr/kWh$ of Norwegian storage hydropower plants for the life cycle inventory. Further, we calculated an adjusted average land occupation of 0.007 $m^2 \cdot yr/kWh$, accounting for an underestimation of water area in the performed maximum likelihood classification. The calculated land occupation values are the basis to support the development of methods for assessing the land occupation impacts of hydropower on biodiversity in LCA at a damage level.



INTRODUCTION

Increasing renewable energy production constitutes a unique opportunity for mitigating climate change impacts.[1] Furthermore, the IPCC has recommended to substantially increase the share of renewable energy in the global energy production.[2] However, even renewable energy sources cause environmental impacts during their life cycle, and these may impact biodiversity.[3−5] Therefore, it is important to assess all relevant impact pathways of renewable energy sources to highlight the main environmental impacts and identify trade-offs between different energy production options and places of operation.

Hydropower electricity production is the largest current source of renewable energy[6] which contributes 16% of the global electricity supply.[7] Its impacts on aquatic and terrestrial biodiversity can be categorized into three main cause-effect pathways.[8] Freshwater habitat alteration potentially affects for example fish, riparian vegetation and macroinvertebrate species,[9−11] water quality degradation can affect (e.g., fish species[12]) and land use and land use change (LULUC) can affect terrestrial and, in part, freshwater flora and fauna.[13−16] All of these pathways may thus lead to local species extinctions and biodiversity loss.[17] However, the three common types of hydropower plants, run-of-river, storage and pumped storage[18] are triggering these impact pathways differently.[8] Run-of-river plants mainly cause freshwater habitat alteration. In contrast, storage and pumped storage hydropower plants affect all of the three mentioned impact pathways.[19]

Concurrently, the IPCCs Special Report on Renewable Energy Sources and Climate Change indicates that there is a need to include long-term environmental consequences from hydropower into current and future projects to identify trade-offs involved with increasing hydropower electricity production.[7] Furthermore, the Norwegian government has pointed out that hydropower electricity production has significant environmental impacts on Norwegian rivers that should be assessed and accounted for.[20]

A particularly suited method for identifying these potential trade-offs between impact pathways is life cycle assessment (LCA). LCA is a commonly used methodology for analyzing the complete environmental impacts of a product or process throughout its life cycle.[21] However, LCA is still developing and can today not assess all relevant biodiversity impacts from hydropower electricity production on a global scale.[8]

In this paper, we address this research gap from a LCA perspective with focus on LULUC, as this is one of the main drivers of global biodiversity loss.[22−24]

Received: October 4, 2017
Revised: December 20, 2017
Accepted: January 12, 2018
Published: January 12, 2018

Storage and pumped storage hydropower plants, which use dams to store water in reservoirs to allow for flexible electricity production, cause LULUC.[18] Reservoir filling causes LULUC by raising water levels and inundating land.[14] Besides reservoir filling, further LULUC is caused by the construction of infrastructure, including power lines[25] and access roads.[17]

The first step for quantifying the biodiversity impacts of this process in LCA is to assess the land occupation per kWh energy produced, in a comprehensive way. In LCA terms, this corresponds to the life cycle inventory (LCI).[26,27] Because globally underlying environmental parameters, such as topographic and climatic conditions[27] vary considerably,[28,29] spatially explicit LCI information is important.[30,31] However, in LCI databases, such as the largest database, Ecoinvent,[32] spatial land occupation information related to hydropower electricity production is only available for Switzerland and Brazil.[27] Consequently for Norway, one of the top-ten hydropower electricity producers worldwide[33] with more than 95% of domestic power production from hydropower,[34] no spatially explicit land occupation information exist.

In addition, all the currently available hydropower LCI parameters do not account for water area of a potential natural lake prior to the inundation of the reservoir[27,35,36] and represent therefore gross parameters.

However, most of the Norwegian hydropower reservoirs are created by impounding natural lakes,[37] thus applying gross parameters to Norway would consequently lead to an overestimation of LCI values[28] and consequently also of the total impact. However, as natural lake surface area was not recorded at the time when most hydropower reservoirs were constructed,[38] information on natural lake surface areas prior to inundation required for estimating the net land occupation is lacking.(Supporting Information 1 (SI1), section S2).

Remote sensing data provides an opportunity for assessing net land occupation in a spatially explicit manner. Remote sensing data is useful for monitoring actual surface area,[39] as well as wetland identification in general.[40] In addition, case studies on land-use transitions from lakes[41] and lake desiccation,[42] have shown that remote sensing data can be used to calculate natural lake surface area prior to inundation.

To identify land cover types, like water, from satellite images, these studies use the different spectral responses of different land cover types, assessed by the satellite sensor.[43]

Therefore, the first aim of this study was to utilize these case study based approaches in combination with remote sensing data providing global coverage[44,45] to quantify spatially explicit inundated land area values due to the installation of storage hydropower plants in a globally systematically applicable approach.

Due to data availability and the domestic importance of hydropower, we applied our work to Norway to validate the applicability of our approach. The second aim is to use the quantified inundated land area to calculate net reservoir-specific land occupation, in m^2·year per kWh of hydropower electricity produced. Land occupation caused by the construction of associated infrastructure, such as roads and power lines, was not considered in this study. This net land occupation can be directly implemented in LCIs. While beyond the scope of this paper, the presented approach is a crucial step toward quantifying impacts of hydropower electricity production on biodiversity in LCA.

■ MATERIALS AND METHODS

Inundated Land Area of Hydropower Reservoirs. Constructing hydropower reservoirs by either damming a river or impounding a natural lake leads to an inundated land area (ILA).[37] Maximal ILA [m^2], for each dammed waterbody x, is the difference between the actual reservoir surface area at highest regulated water level (RSA) and the waterbody surface area before dam construction (WSA), both assumed constant over time (eq 1). This assumption is valid on a long-term perspective with no anthropogenic disturbance.[46]

$$ILA_x = RSA_x - WSA_x \tag{1}$$

In this study we used the actual reservoir surface area at highest regulated water level with commissioning year provided by the Norwegian Water Resources and Energy Directorate (NVE),[38] as this database provides the most detailed information for Norway. We estimate waterbody surface area before dam construction using Landsat data[44] with global coverage, and aerial photographs,[45] as described in the following sections.

Origin of the Remote Sensing Data. Satellite images were downloaded from the NASA-USGS Global Land Survey data set (GLS) provided by the public domain of U.S. Geological Survey.[44] The GLS data set is a collection of freely accessible, orthorectified and cloud-minimized Landsat satellite images with global coverage, which has been used to map global forest cover,[47] as well as historical changes of wetlands.[40,48,49]

Due to the age of most hydropower reservoirs in Norway, we used the oldest assembled epoch, the GLS-1975 data set, with Landsat 1−3 images acquired from 1972 to 1983 with a resolution of 60 m.[50] We extracted 32 multispectral images from the GLS-1975 available for Norway, which were not totally covered with ice and snow (SI1, S3). Images were then sorted by satellite and merged in ArcGIS10.3[51] by path. The path describes the orbital track of the satellite from east to west. Consequently, we avoided overlap of images and ensured the use of all extracted images for Norway. As a result, we obtained 13 merged multispectral images, which are path and satellite specific (SI1, S3).

In addition, seven aerial photographs where obtained from the Internet portal Norge i Bilder[45] as additional data source. This platform provides aerial photographs for Norway with a resolution of 0.2 m, dating back to 1937 (SI1, S4).

Quantifying Water Surface Area before Dam Construction. To assess the water surface area before dam construction, the commissioning year of the hydropower reservoirs has to be equal or younger than the exposure year of the remote sensing data. This means that we can potentially assess water surface area before dam construction of hydropower reservoirs with commissioning year equal or after 1936 from aerial photographs and with commissioning year 1972 and after from Landsat images. Additionally, WSA can only be calculated for hydropower reservoirs that were not covered with ice and snow or clouds during the image exposure date, as this makes identification of water surface areas impossible. To calculate water surface area before dam construction from Landsat images, the water body area was identified with an image classification method. To classify land cover types from satellite images two main methods exist: unsupervised and supervised classification.[43] Unsupervised classification aggregates pixels with similar spectral values in clusters, which are then assigned to a land cover type.[40] In the more commonly

DOI: 10.1021/acs.est.7b05125
Environ. Sci. Technol. 2018, 52, 2375−2384

used supervised maximum likelihood classification,[43] land cover types are identified based on user-defined training areas, consisting of area on the satellite image where the land cover type is known.[40] The maximum likelihood classification, has the advantage that with training areas our desired land cover type "water" can be chosen directly, whereas in the unsupervised classification a single cluster may not correspond with the land cover type "water", because one land cover type can be represented by multiple clusters.[40] Therefore, we performed a supervised maximum likelihood classification in ArcGIS10.3[51] to identify water pixels on the Landsat satellite images.

For each merged multispectral image, we created training areas each comprising either a "water" or "non-water" land cover type. Each land cover type was defined by several training areas. The maximum likelihood classification analyzes the pixel values, defined by the spectral reflectance of the pixels in the different spectral bands of the satellite image, of all training areas in each land cover type. The mean and variance of the pixel values in each land cover type is then used to categorize all pixels of the image in the land cover type with the highest probability of a membership.[40] "Water" training areas consisted of areas clearly identified as water on the true color satellite images. These mainly consisted of lakes and, where available, fjord areas. As certainty of water identification increases with lake size, we used the largest lakes available on the image for training. Additionally, we included, if present, fjords, lakes in mountainous areas and lakes containing algae, as they have different spectral reflectances.[40] Nonwater training areas consisted of the land cover types: land, clouds, and ice and snow. These land cover types have different spectral reflectance, and thus improve the correct categorization of these pixels in the right land cover type, avoiding misclassification of water pixels. The amount and size of training areas depended on the land cover types contained by the merged multispectral images. Scatterplots were used to ensure that the spectral reflectance of water and nonwater training areas did not overlap. The maximum likelihood classification was performed with each of the merged multispectral images (SI1, S3). Water surface area before dam construction from Landsat images was calculated with eq 2, using the identified pixels of the land cover type "water".

$$\mathrm{WSA}_x = \mathrm{WP}_x \times \mathrm{PR} \tag{2}$$

Where WP is the number of water pixels of reservoir x on the GLS-1975 image within the boundary of the reservoir at highest regulated water level and PR is the pixel resolution in m^2.

The images from the different Landsat-paths can overlap, thus partly covering the same area (SI1, S3).

As a result, we calculated up to four different water pixel numbers for each hydropower reservoir. In these cases, we used the maximum number of water pixels as final WP_x, assuming representation of the maximum water level of the natural lake (SI2). With the number of water pixels, we calculated water surface area before dam construction and consequently the net land occupation with eq 1, for ice/snow- and cloud-free hydropower reservoirs on the merged multispectral images.

From 11 hydropower reservoirs on aerial photographs, we obtained WSA directly by using the online measurement tool from Norge i Bilder[45] and used eq 1 to calculate the inundated land area. Direct measurement was possible due to image resolution (SI1, S4).

Inundated land area estimates from remote sensing data were based on a planar surface, thereby assuming that the slope of the terrain is always zero. However, inundated land area around reservoirs, which are usually situated in mountainous regions, will most likely not be on a flat surface. Therefore, we tested the effect of slope by also calculating inundated land area with a sloped terrain. However, there was no significant effect of slope, and thus we have not considered the inundated land area with a sloped terrain in this study (SI1, S6).

Land Occupation Modeling for the Life Cycle Inventory. The inundated land area represents a land use change. For land use, LCA distinguishes between land occupation and land transformation. Land occupation is defined as a use of a land area for a certain human-controlled purpose. The recovery to the original state is postponed by a period of time equal to the duration of the occupation process.[52] Land transformation is defined as a change a of land area in line with requirements of a new occupation process. The recovery to the original state is depending on the severity of transformation, the duration of land occupation and the recoverability of the affected terrestrial habitat.[52] For our purpose, we define the land use caused by the inundation of land as land occupation, as the inundation of land occurs over a specific time for a human-controlled purpose. Further, studies have documented a fast geomorphic floodplain change inherent with an ecological recovery after dam removals.[53,54]

The net land occupation LO_x [$m^2 \cdot yr/kWh$] modeled for the LCI (eq 3), relates the inundated land area of each hydropower reservoir x to yearly average electricity production, assuming that the reservoir is only used for hydropower electricity production.

$$\mathrm{LO}_x = \frac{\mathrm{ILA}_x}{\mathrm{ER}_x} \tag{3}$$

ER is the average annual electricity production of hydropower reservoir x in kWh. ILA is the inundated land area of hydropower reservoir $x$in m^2.

As Bakken et al.[28] pointed out that the power production of several hydropower plants could benefit from the creation and regulation of the uppermost hydropower reservoir in a cascade system, we calculated ER for each reservoir x with eq 4:

$$\mathrm{ER}_x = \sum_{m=1}^{m} \frac{\mathrm{RSA}_x}{\sum_{n=1}^{n} \mathrm{RSA}_{x \in z|\mathrm{upstream}}} \cdot E_z \tag{4}$$

RSA is the actual reservoir surface area at highest regulated water level in m^2. n is the number of upstream reservoirs located in Norway connected to hydropower plant z. E is the average annual electricity production at hydropower plant z in kWh. m is the number of hydropower plants located downstream of reservoir x. We received RSA, n, m and average E (between 1981 and 2010) from the Norwegian Water Resources and Energy Directorate (SI2).[38,55] We assumed that this is the average true electricity production in a normal year and is not significantly fluctuating over the years. Average hydropower electricity production in a normal year is calculated as a function of installed capacity and expected annual inflow in a year with normal precipitation.[34] However, for pumped storage plants, which have usually a negative net electricity output,[56] this methodology is not applicable.[27] Therefore, we did not calculate land occupation values for reservoirs directly assigned to a pumped storage hydropower plant with negative net electricity production. Additionally, due to the resolution of the satellite images (60 m), reservoirs with an RSA smaller than

DOI: 10.1021/acs.est.7b05125
Environ. Sci. Technol. 2018, 52, 2375−2384

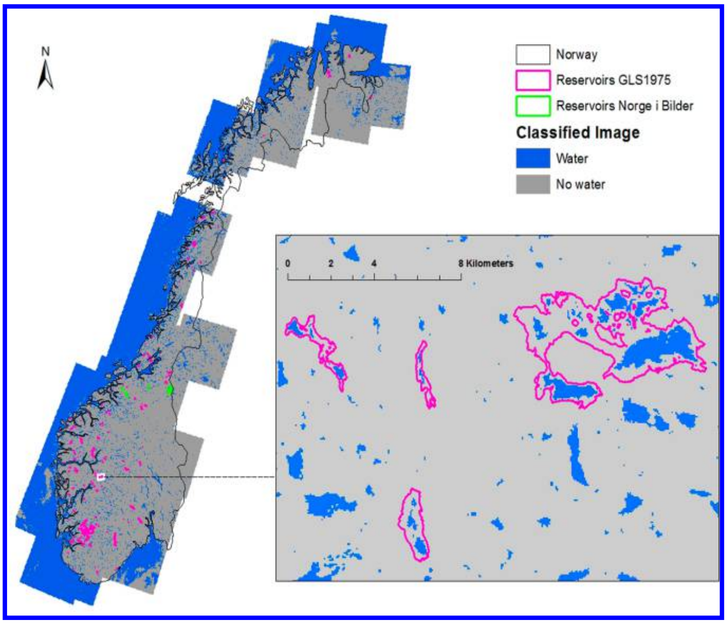


Figure 1. Merged classified image of all 13 Landsat paths, showing water (blue) and nonwater pixels (gray) overlaid with hydropower reservoirs[38] in Norway[58] where inundated land area was quantified from the NASA-USGS Global Land Survey GLS-1975[44] data set (pink) or from aerial photographs from the Internet portal Norge i Bilder[45] (green). The enlarged area shows an example of a more detailed view of the map.

0.5 km^2 were excluded from the calculations and not assigned to a hydropower plant.

Uncertainty of the Land Occupation Calculation. Frazier and Page[57] report that maximum likelihood classification underestimates the amount of water pixels due to mixed pixels, which contain more than one land cover type. In our case, mixed pixels are located at the shore of the reservoirs and contain both water and nonwater land cover types. To assess the uncertainty in the land occupation calculation due to this potential bias, we calculated the water body area of natural lakes with surface area larger than 0.5 km^2 contained by the classified images obtained from the maximum likelihood classification and compared this calculated water body area to the actual natural lake surface area provided by NVE.[38] Here, we assumed that the surface area of natural lakes remains constant over time. We limited our error analyses to Landsat 1 Path 214 and Path 215, as lakes must be manually checked for ice and cloud cover (SI2). We regressed the latter area against the calculated maximum likelihood classification area using a generalized linear model with a quasi-Gaussian distribution to account for the skewness of the data. Based on this model, we estimated, adjusted water surface areas (WSA_{adj}), the subsequent adjusted land occupation (ALO), and associated confidence intervals. In cases where the adjusted land occupation value became negative, due to a larger WSA_{adj} compared to RSA, we set it to zero.

■ RESULTS

Inundated Land Area of Hydropower Reservoirs. We were able to quantify the inundated land area for 184 of the 265 hydropower reservoirs in Norway that have a commissioning year of 1972 or after (SI1, S2 and SI2). The main reason for not quantifying all hydropower reservoirs with commissioning year of 1972 or after is the fact that many GLS-1975 images were acquired in early May or October[50] when lakes are frozen in Norway making identification of some water surface areas impossible. This was also the main reason for the low number of hydropower reservoirs with quantified ILA in the north (Figure 1).

Total ILA from 1972 up to today is 305.3 km^2 with an average of 1.66 km^2 and ranging from 0.003 km^2 to 63.9 km^2 per hydropower reservoir.

We calculated ILA for 173 hydropower reservoirs from 32 GLS-1975 images[44] covering 13 paths, and ILA for another 11 hydropower reservoirs from 6 aerial photographs.[45] We excluded GLS-1975 images that did not contain any ice- or cloud-free hydropower reservoirs. Therefore, the classified image in Figure 1 contains gaps, despite the fact that GLS-1975 images are available for the whole of Norway.

Land Occupation Modeling for the Life Cycle Inventory. Of the 184 hydropower reservoirs identified on the satellite images and aerial photographs, 73 hydropower reservoirs were excluded from the land occupation calculation due to small surface area (<0.5 km^2) and one because related hydropower plant were not available from NVE.[38] In total, three of the 184 hydropower reservoirs were assigned to a pumped storage hydropower plant, but had to be excluded as their net electricity output was negative.

Consequently 107 hydropower reservoirs of storage power plants (96 from GLS-1975 and 11 from Norge i Bilder), were used to calculate the land occupation with the obtained inundated land area (Figure 2; SI2). Thereof, 100 hydropower reservoirs were only used for hydropower electricity production. Five were also used as recreational dams, one as fishing dam, and one for water supply. The average land

DOI: 10.1021/acs.est.7b05125
Environ. Sci. Technol. 2018, 52, 2375−2384

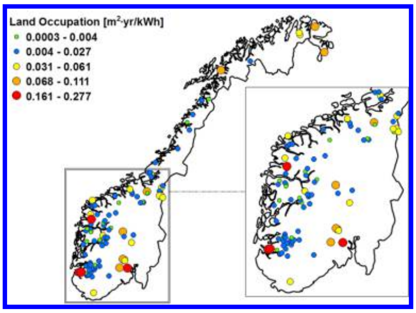


Figure 2. Map showing the land occupation [m^2·yr/kWh] for each hydropower reservoir in Norway.[58] For an easier identification, the inset map shows a more detailed view of southern Norway.

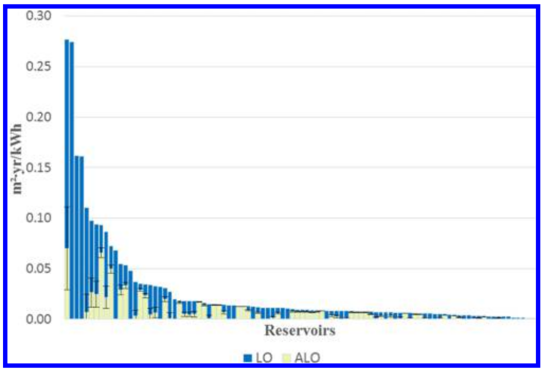


Figure 3. Comparison of land occupation (LO) in blue and adjusted land occupation (ALO) in mint with standard deviation of ALO in black. In 31 cases, the adjusted land occupation was set to zero.

occupation across all investigated hydropower reservoirs was calculated as 0.027 m^2·yr/kWh.

We assessed land occupation for 75% (808 km^2) of the RSA from hydropower reservoirs with commissioning year of 1972 or after in Norway. This represents 13.4% of the total RSA of all hydropower reservoirs in Norway. The 107 hydropower reservoirs have an average annual electricity production of 27.059 GWh, representing 19.6% of the total average annual hydropower electricity produced in Norway between 1981 and 2010.

Land Occupation Uncertainty. For the Landsat 1 Path 214, the natural water surface area (dispersion parameter: 0.077; intercept: −0.303 ± 0.029, $P < 0.001$) was significantly related to the maximum likelihood classification water surface area (1.068 ± 0.011 SD, $P < 0.001$). For the Landsat 1 Path 215, the natural water surface area (dispersion parameter: 0.024; intercept: 0.228 ± 0.013, $P < 0.001$) was significantly related to the maximum likelihood classification water surface area (1.103 ± 0.009 SD, $P < 0.001$). The averaged correction values across both Landsat 1 Paths were used to adjust the water surface area, including 95% confidence intervals (intercept: 0.266 ± 0.022 SD; water surface area: 1.085 ± 0.010 SD). After adjustment, the ratio of the natural to calculated water surface area reduced from 1.40 (95% percentile: 1.04−2.23) to 0.97 (95% percentile: 0.70−1.28); removing underestimation of the water surface area.

In 31 cases, the adjusted land occupation was set to zero, due to a larger WSA_{adj} then RSA, indicating that a natural lake became utilized as reservoir. With this, we calculated an adjusted land occupation with an average of 0.007 m^2·yr/kWh (Figure 3).

Using WSA_{adj} instead of WSA resulted in adjusted land occupation values 2.7−100% smaller than the previous calculated land occupation values. This variation can be explained as the difference between WSA_{adj} and WSA in relation to the ILA. The example of the Reservoir Riskallvatn shows that even if WSA_{adj} is 293% larger (0.24 km^2) than WSA, the difference of adjusted land occupation in relation to land occupation is only 28.9%, because the difference is small in comparison to the previously estimated inundated land area of 0.89 km^2. In contrast, for the Breimsvatn Reservoir, WSA_{adj} is only 110% larger (2.02 km^2) than WSA, but as this value is big in comparison to the previously estimated inundated land area of 2.04 km^2, the difference of adjusted land occupation in relation to land occupation is 98.9%.

DISCUSSION

Calculating Inundated Land Area from Remote Sensing Data. We performed a supervised maximum likelihood classification on GLS-1975 Landsat MSS images[44] to identify water pixels and showed that it is possible to assess the inundated land area on a reservoir level for the whole of Norway. Even though we used a supervised classification method, unsupervised classification techniques have also been used for wetland classification. Unsupervised classification aggregates pixels with similar spectral values in clusters and the step of defining training areas is therefore not necessary.[40] However, the corresponding step in the unsupervised classification is the assignment of clusters to a land cover type[43] and unsupervised classification for wetlands is only effective when a large number of clusters is used.[40]

Additionally in our case, identifying land cover types of clusters only including small lakes or the border area between land and water on a satellite image with a 60 m resolution would have been almost impossible. In the supervised maximum likelihood classification, we defined our desired land cover type "water" directly by using training areas that were clearly identifiable as water. All pixels were categorized in the land cover type with the highest probability of a membership. This is an advantage in comparison to the unsupervised classification method. The selected supervised maximum likelihood classification was therefore deemed the appropriate classification method for our purpose.[40]

Besides the two main classification methods, the normalized difference water index (NDWI) was developed to quantify open water areas.[59] This index performs best when a middle infrared band is used.[60,61] The Landsat MSS data used in this study, however, does not cover this band. We were therefore not able to use the NDWI in this study. However, the NDWI may be applied when calculating the inundated land area of more recently built hydropower reservoirs from more recent satellite images, containing a middle infrared band.

We used the nonslope corrected inundated land area to calculate the land occupation, as there was no statistically significant difference between slope-adjusted ILA and ILA. However, slope-adjusted ILA was always larger than ILA. For the slope-adjusted ILA we assumed that the slope outside and inside the water body was the same. As this is not always the case, the potential error of this assumption might, however, be

DOI: 10.1021/acs.est.7b05125
Environ. Sci. Technol. 2018, 52, 2375−2384

higher than the quantified effect.[62] As such, we recommend using ILA, as done in this study, and not slope-adjusted ILA, but account for the potential underestimation of ILA (SI1, S5).

Land Occupation Modeling for the Life Cycle Inventory. We calculated a land occupation value for 107 of 1289 hydropower reservoirs in Norway. We thereby calculated a land occupation value for 96 of the 250 hydropower reservoirs that have a commissioning year of 1972 or after, with the GLS-1975 data set. The number of 107 is high, because many images were acquired when lakes are frozen in Norway. Considering that the GLS-1975 is the oldest available satellite image set available,[50] the main limiting factor to assess all hydropower reservoirs in Norway is the image acquisition year. However, our number of 107 reservoirs is much higher than the 52 reservoirs assessed for Switzerland and the one reservoir for Brazil in the existing Ecoinvent database.[27] The average land occupation in our study across all investigated hydropower plants is 0.027 $m^2 \cdot yr/kWh$ and is larger than the existing 0.004 $m^2 \cdot yr/kWh$ in the Ecoinvent database,[27] despite the fact that we calculated the net land occupation.

When comparing average values it has to be considered that we only calculated a land occupation value for 107 hydropower reservoirs. Further assessment of land occupation values of neglected reservoirs in this study could therefore change the average net land occupation value. However, a more detailed quantification is not possible, as we have applied all the available information regarding inundated land area of Norwegian hydropower reservoirs.

The land occupation varies between 0.0003 $m^2 \cdot yr/kWh$ and 0.28 $m^2 \cdot yr/kWh$ and therefore 18 reservoir-specific LO values were lower than the Ecoinvent value.[27] The range of our values highlights the importance of site-specific life cycle inventory modeling, even when not all hydropower reservoirs of a country are assessed.

However, further research is needed to asses a land occupation value for hydropower reservoirs with commissioning year of 1972 or before. A promising starting point could be old lake depth maps with lake surface area calculations. The focus of the old analog lake depth maps in Norway[63] was to quantify the depth and water volume to predict ice conditions after a possible regulation. Hence, only depth soundings were conducted as fieldwork, while water surface area estimation is based on the M711 topographic map series.[63] This map series was compiled between 1952 and 1988.[64] Due to this time span, it is not ensured that the estimated water surface area represents the status before the dam construction. Therefore, we did not include this additional information, although it reduced the amount of included hydropower reservoirs.

As impacts of power generation are generally compared per unit of electricity produced,[65−67] we are describing the land occupation per kWh hydropower produced. As the power production of several hydropower plants could benefit from the creation and regulation of the uppermost hydropower reservoir in a cascade system, we used the RSA to reallocate the electricity produced for the land occupation calculation. This assumption might be incorrect, as factors like reservoir volume might also have an influence. However, it is a method to ensure that produced electricity is not double counted.

Seven out of 107 reservoirs used to calculate land occupation are used as multipurpose reservoirs,[38] thus hydropower electricity production is not the only reason causing land occupation. In multipurpose reservoirs the impact should therefore be allocated between use purposes[68] as part of the land occupation should be related to the other purposes. Nevertheless, allocation guidance is still lacking[68] and due to the low number of seven multipurpose reservoirs we have not included an allocation factor. Therefore, our calculated land occupation may overestimate the land occupation for these seven hydropower reservoirs that are used for several purposes.

Due to satellite image resolution, the land occupation is only quantifying the amount of land occupied, but not which type of land cover nor its significance for terrestrial biodiversity or the effect on evaporation rates and the related water consumption. However, when quantifying the land occupation of newly built hydropower reservoirs, newer remote sensing data that have a higher spatial resolution can be used. This higher resolution, for example allows that the habitat quality of the occupied land could be assessed as shown by Zlinszky et al.[69]

Land Occupation Uncertainty. The adjusted average land occupation (0.007 $m^2 \cdot yr/kWh$) is lower than the average land occupation (0.027 $m^2 \cdot yr/kWh$) and therewith closer to the existing 0.004 $m^2 \cdot yr/kWh$ in the Ecoinvent database.[27] Furthermore, the adjusted average land occupation varies between 0 and 0.07 $m^2 \cdot yr/kWh$ showing the importance of spatially explicit life cycle inventory modeling.

Frazier and Page[57] report that maximum likelihood classification underestimates the amount of water pixels, which is in accordance with our uncertainty analysis, as for both Landsat 1 Path 214 and 215 the maximum likelihood classification water surface area was always smaller than the natural water surface.

An underestimation of approximately 5% for water area classification from Landsat MSS data is reported by Smith.[49] Frazier and Page[57] performed a maximum likelihood classification on Landsat TM data to identify water bodies. For lakes with a surface area of 0.06 to 0.18 km^2 an underestimation of 43.5% was reported and for water bodies with surface areas of 50 m^2 to 0.032 km^2 an underestimation of up to 80%.[57]

These studies confirm that our maximum likelihood classification, with an underestimation of 39% and 36% for natural lakes on Landsat 1 Path 214 and Path 215 with surface areas <1 km^2, performs within the conventional range and that uncertainty is indeed depending on lake surface area. The average underestimation of 8% for natural lakes with surface area >5 km^2 Landsat 1 Path 214 is close to the reported 5% by Smith (SI2).[49]

As the number of mixed pixels decreases with higher image resolution,[40] choosing a higher image resolution presents the best way to reduce mixed pixels and therewith the understimation of water surface from small lakes. However, this option is not possible for the Landsat 1−3 images used in this study.

In addition, variability in annual electricity production (ER) should be included in the confidence intervals. These data, however, were not available for this study.

Besides methodological uncertainty,[40] which we quantified in this study, seasonal aspects and geolocation error of the GLS-1975[50] are most likely the largest contributors to uncertainty. However, quantification of these uncertainties in a systematic way is difficult. Therefore, they can be only discussed qualitatively in the following section, nevertheless adding an undefined amount of error to our performed error analysis.

To calculate WSA and WSA_{adj} we assumed that natural lakes have a constant surface area over time. This assumption is correct for a long-term perspective with no anthropogenic

DOI: 10.1021/acs.est.7b05125
Environ. Sci. Technol. 2018, 52, 2375−2384

disturbance.[46,70] Nevertheless, water levels of freshwater lakes can have a seasonal fluctuation.[71] These fluctuations can have different amplitudes.[46,70,72] For lake Atnasjøen in Norway an average seasonal fluctuation of approximately 1 m is reported, characterized by spring-floods due to the snowmelt.[73]

The Landsat images used in this study were acquired between 1972 and 1983 and from early May to October or later.[50] They therefore may include both high and low water levels resulting in different waterbody surface areas.

We accounted for high levels, assuming that these represent the natural maximum range of the lake, by using the maximum number of water pixels obtained for each hydropower reservoir to calculate WSA. However, due to the temporal resolution of the Landsat images even the maximum number of water pixels pertain to a period of low water level, leading to an underestimation of WSA and overestimation of LO.

Moreover, the GLS-1975 data set has a geolocation error of maximal 24.9 m in comparison to the GLS-2000.[50] Therefore, RSA and classified GLS-1975 images may be shifted against each other. As a result, counted pixels on the satellite image may not intersect with RSA in reality, resulting in wrong WSA estimates.

Implementation and Use in LCA. The unit of the modeled land occupation is $m^2 \cdot yr/kWh$. This is in accordance with the unit of $m^2 \cdot$years for land occupation in the land use inventory[74] and therefore our net land occupation values calculated for storage hydropower reservoirs are directly implementable in LCI databases.

We assumed that the average annual electricity production is the "true" electricity production in a normal year. In the long term perspective this assumption is indeed correct,[34] therefore the average annual electricity production can be used to calculate the land occupation per kWh produced. This means that our values are designed to calculate the average land occupation over the complete operational phase. They are, however, not applicable for individual years, as this can either lead to over- or underestimation of the average yearly land occupation, because the annual inflow to hydropower reservoirs in Norway has varied from 1990 to 2013 by about 60 TWh. This, for example, caused the variation in the hydropower electricity production in the whole of Norway from 143 TWh in 2000 to 106 TWh in 2003, the latter being a very dry year.[34] However, if the efficiency of the hydropower plants change over time (e.g., due to changes in precipitation patterns), the inventory has to be updated, as this will reduce the land occupation per kWh. Some land occupation inventory parameters are designed to predict future land occupation impacts.[75] In contrast, our calculated land occupation is only representative for the period from 1972 to 1985 and should not be used to quantify the land occupation of newly built hydropower reservoirs.

For newly built hydropower reservoirs, the inundated land area itself can be directly modeled with digital elevation models.[76,77] In addition, our values do not account for land occupation and hydropower electricity production changes that may occur due to possible precipitation and related hydrological regime changes under different climate change scenarios.[78]

Further LCA does not account for potential positive effects. Therefore, we did not account for potential positive effects of the hydropower reservoirs on limnic and littoral species like fish[79] or water/shore birds.[80] Moreover, we did not assess the habitat quality of the additional water area introduced. Most reservoirs are rather deep and steep water bodies, which may not represent suitable habitat for many species.

Prospective Implementation in LCA. This paper provides important net land occupation parameters for Norway. Due to the fact that the GLS-1975 data set has a global coverage,[50] our proposed method has the potential to assess the land occupation of storage hydropower reservoirs systematic and with spatial variation on a global scale. In addition, the reservoir filling has an impact beyond the LULUC cause-effect pathways as the increased water surface area of the filled reservoirs leads to consumptive water use through increased evaporation from the open water surface. This is causing potential impacts for aquatic ecosystems by decreasing the discharge.[81] Furthermore, the hydropower reservoirs creation can lead to increased greenhouse gas emissions, which arise from the decomposition of organic matter that was either flooded during reservoir filling or flushed into to the reservoir by river runoff or deposed on the reservoir surface after filling.[29,36,82,83] Therefore, our inundated land area values can be further used to calculate net water consumption and net greenhouse gas emission for the LCI.[27] This is the basis to support the development of methods for assessing the land occupation, water consumption, and greenhouse gas emission impacts of hydropower on biodiversity in LCIA at a damage level, as general applicable LCIA methodology for each of these LCI parameters exists.[81,84,85] With reservoir-specific values we are in addition providing the smallest possible resolution. This enhances the development of needed spatially differentiated LCIA methods.[86,87]

When the application of the land occupation values requires a broader spatial resolution for biodiversity life cycle impact assessment we recommend an aggregation on terrestrial ecoregions for land occupation,[88] for water consumption on freshwater ecoregions[89] and for greenhouse gas emissions on a country level. Furthermore, it highlights once again the importance and possibilities of remote sensing data to improve the LCI and LCIA framework spatially.[90]

Further Application Outside LCA. In this study we used the inundated land area of storage hydropower reservoirs, to calculate inventory parameters suitable for LCA, due to its suitability for identifying potential trade-offs between impact pathways.[21] Indeed, LCA is only one application. The ILA values could also be made publically available by integration in the Global Reservoir and Dam Database.[91] Then the ILA values could be, for example, directly used to assess terrestrial biodiversity impacts of hydropower reservoir inundation.[14,92]

ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge on the ACS Publications website at DOI: 10.1021/acs.est.7b05125.

Details on the case study area and remote sensing data is available as PDF (PDF)

Inventory calculations results are available as Excel file (XLSX)

AUTHOR INFORMATION

Corresponding Author

*Phone: +47 73598946; e-mail: martin.dorber@ntnu.no.

ORCID

Martin Dorber: 0000-0001-8477-1934

Author Contributions

M.D. designed and carried out the analyses. M.D., R.M., and F.V. wrote the manuscript. M.D. made all the figure and tables. All authors have given approval to the final version of the manuscript.

Notes

The authors declare no competing financial interest.

■ ACKNOWLEDGMENTS

We thank John Sebastian Woods for helpful comments and discussions, and Sajith Vezhapparambu for comments on image classification methods. Further, we thank the four anonymous reviewers for their helpful comments. This work was funded by the Research Council of Norway through the SURE project (Project Number 244109)

■ REFERENCES

(1) Moomow, W.; Yamba, F.; Kamimoto, M.; Maurice, L.; Nyboer, J.; Urama, K.; Weir, T., Introduction. In *IPCC Special Report on Renewable Energy Sources and Climate change Mitigation*; Edenhofer, O.; Pichs-Madruga, R.; Sokona, Y.; Seyboth, K.; Matschoss, P.; Kadner, S.; Zwickel, T.; Eickemeier, P.; Hansen, G.; Schlömer, S.; von Stechow, C., Eds.; Cambridge University Press: United Kingdom and New York, NY, 2011.

(2) United Nations *Global Sustainable Development Report*, New York, 2016.

(3) Allison, T. D.; Root, T. L.; Frumhoff, P. C. Thinking globally and siting locally – renewable energy and biodiversity in a rapidly warming world. *Clim. Change* **2014**, *126* (1–2), 1–6.

(4) Jackson, A. L. R. Renewable energy vs. biodiversity: Policy conflicts and the future of nature conservation. *Global Environmental Change* **2011**, *21* (4), 1195–1208.

(5) Santangeli, A.; Toivonen, T.; Pouzols, F. M.; Pogson, M.; Hastings, A.; Smith, P.; Moilanen, A. Global change synergies and trade-offs between renewable energy and biodiversity. *GCB Bioenergy* **2016**, *8* (5), 941–951.

(6) Brown, A.; Muller, S.; Dobrotkova, Z. *Renewable Energy Prospects and Technology*; IEA, 2011.

(7) Kumar, A.; Schei, T.; Ahenkorah, A.; Rodriguez, R. C.; Devernay, J.-M.; Freitas, M.; Hall, D.; Killingtveit, Å.; Liu, Z., Hydropower. In *IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation*; Edenhofer, O.; Pichs-Madruga, R.; Sokona, Y.; Seyboth, K.; Matschoss, P.; Kadner, S.; Zwickel, T.; Eickemeier, P.; Hansen, G.; Schlömer, S.; von Stechow, C., Eds.; Cambridge University Press: Cambridge, United Kingdom and New York, NY, USA, 2011.

(8) Gracey, E. O.; Verones, F. Impacts from hydropower production on biodiversity in an LCA framework—review and recommendations. *Int. J. Life Cycle Assess.* **2016**, *21* (3), 412–428.

(9) Poff, N. L.; Zimmerman, J. K. H. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. *Freshwater Biol.* **2010**, *55* (1), 194–205.

(10) Crook, D. A.; Lowe, W. H.; Allendorf, F. W.; Eros, T.; Finn, D. S.; Gillanders, B. M.; Hadweng, W. L.; Harrod, C.; Hermoso, V.; Jennings, S.; Kilada, R. W.; Nagelkerken, I.; Hansen, M. M.; Page, T. J.; Riginos, C.; Fry, B.; Hughes, J. M. Human effects on ecological connectivity in aquatic ecosystems: Integrating scientific approaches to support management and mitigation. *Sci. Total Environ.* **2015**, *534*, 52–64.

(11) Jansson, R.; Nilsson, C.; Renofalt, B. Fragmentation of Riparian Floras in Rivers with Multiple Dams. *Ecology* **2000**, *81* (4), 899.

(12) Donaldson, M. R.; Cooke, S. J.; Patterson, D. A.; Macdonald, J. S. Cold shock and fish. *J. Fish Biol.* **2008**, *73* (7), 1491–1530.

(13) Alho, C. J. Environmental effects of hydropower reservoirs on wild mammals and freshwater turtles in Amazonia: a review. *Oecologia Australis* **2011**, *15* (3), 593–604.

(14) Kitzes, J.; Shirley, R. Estimating biodiversity impacts without field surveys: A case study in northern Borneo. *Ambio* **2016**, *45* (1), 110–119.

(15) Zhang, J.; Zhengjun, L.; Xiaoxia, S. Changing landscape in the Three Gorges Reservoir Area of Yangtze River from 1977 to 2005: Land use/land cover, vegetation cover changes estimated using multi-source satellite data. *ITC J.* **2009**, *11* (6), 403–412.

(16) Tefera, B.; Sterk, G. Hydropower-Induced Land Use Change in Fincha'a Watershed, Western Ethiopia: Analysis and Impacts. *Mountain Research and Development* **2008**, *28* (1), 72–80.

(17) McAllister, D. E.; Craig, J. F.; Davidson, N.; Delany, S.; Seddon, M. *Biodiversity Impacts of Large Dams*, 2001.

(18) Egré, D.; Milewski, J. C. The diversity of hydropower projects. *Energy Policy* **2002**, *30* (14), 1225–1230.

(19) Arthington, A. H.; Naiman, R. J.; McClain, M. E.; Nilsson, C. Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities. *Freshwater Biol.* **2010**, *55* (1), 1–16.

(20) Norwegian Government Ministries and Offices, Meld. St. Twenty-five (2015–2016) Power to Change - Energy policy 2030 (In Norwegian).

(21) ISO Environmental management. *Life Cycle Assessment — Principles and Framework*; International Organisation for Standardization: Geneva, Switzerland, 2006.

(22) Duraiappah, A.; Naeem, S.; Agardi, T.; Ash, N.; Cooper, D.; Díaz, S.; Faith, D.; Mace, G.; McNeilly, J.; Mooney, H. Ecosystems and Human Well-Being: Biodiversity Synthesis; World Resources Institute: Washington, DC, 2005; Vol. *86*.

(23) Newbold, T.; Hudson, L. N.; Hill, S. L. L.; Contu, S.; Lysenko, I.; Senior, R. A.; Borger, L.; Bennett, D. J.; Choimes, A.; Collen, B.; Day, J.; De Palma, A.; Diaz, S.; Echeverria-Londono, S.; Edgar, M. J.; Feldman, A.; Garon, M.; Harrison, M. L. K.; Alhusseini, T.; Ingram, D. J.; Itescu, Y.; Kattge, J.; Kemp, V.; Kirkpatrick, L.; Kleyer, M.; Correia, D. L. P.; Martin, C. D.; Meiri, S.; Novosolov, M.; Pan, Y.; Phillips, H. R. P.; Purves, D. W.; Robinson, A.; Simpson, J.; Tuck, S. L.; Weiher, E.; White, H. J.; Ewers, R. M.; Mace, G. M.; Scharlemann, J. P. W.; Purvis, A. Global effects of land use on local terrestrial biodiversity. *Nature* **2015**, *520* (7545), 45–50.

(24) Sala, O. E.; Chapin, F. S.; Armesto, J. J.; Berlow, E.; Bloomfield, J.; Dirzo, R.; Huber-Sanwald, E.; Huenneke, L. F.; Jackson, R. B.; Kinzig, A.; Leemans, R.; Lodge, D. M.; Mooney, H. A.; Oesterheld, M.; Poff, N. L.; Sykes, M. T.; Walker, B. H.; Walker, M.; Wall, D. H. Biodiversity - Global biodiversity scenarios for the year 2100. *Science* **2000**, *287* (5459), 1770–1774.

(25) Gundula, S. B.; Roel, M.; Kjetil, B.; Sigbjorn, S.; Eivin, R. The effects of power lines on ungulates and implications for power line routing and rights-of-way management. *International Journal of Biodiversity and Conservation* **2014**, *6* (9), 647–662.

(26) Rebitzer, G.; Ekvall, T.; Frischknecht, R.; Hunkeler, D.; Norris, G.; Rydberg, T.; Schmidt, W.-P.; Suh, S.; Weidema, B. P.; Pennington, D. W. Life cycle assessment: Part 1: Framework, goal and scope definition, inventory analysis, and applications. *Environ. Int.* **2004**, *30* (5), 701–720.

(27) Flury, K.; Frischknecht, R. Life cycle inventories of hydroelectric power generation. *ESU-Services Ltd* **2012**, 1–51.

(28) Bakken, T. H.; Killingtveit, Å.; Engeland, K.; Alfredsen, K.; Harby, A. Water consumption from hydropower plants - review of published estimates and an assessment of the concept. *Hydrol. Earth Syst. Sci.* **2013**, *17* (10), 3983–4000.

(29) Deemer, B. R.; Harrison, J. A.; Li, S.; Beaulieu, J. J.; DelSontro, T.; Barros, N.; Bezerra-Neto, J. F.; Powers, S. M.; dos Santos, M. A.; Vonk, J. A. Greenhouse Gas Emissions from Reservoir Water Surfaces: A New Global Synthesis. *BioScience* **2016**, *66* (11), 949–964.

(30) Mutel, C. L.; Hellweg, S. Regionalized life cycle assessment: computational methodology and application to inventory databases. *Environ. Sci. Technol.* **2009**, *43* (15), 5797–5803.

(31) Mutel, C. L.; Pfister, S.; Hellweg, S. GIS-Based Regionalized Life Cycle Assessment: How Big Is Small Enough? Methodology and Case

Study of Electricity Generation. *Environ. Sci. Technol.* **2012**, *46* (2), 1096−1103.
(32) Wernet, G.; Bauer, C.; Steubing, B.; Reinhard, J.; Moreno-Ruiz, E.; Weidema, B. The ecoinvent database version 3 (part I): overview and methodology. *Int. J. Life Cycle Assess.* **2016**, *21* (9), 1218−1230.
(33) Manzano-Agugliaro, F.; Alcayde, A.; Montoya, F. G.; Zapata-Sierra, A.; Gil, C. Scientific production of renewable energies worldwide: An overview. *Renewable Sustainable Energy Rev.* **2013**, *18*, 134−143.
(34) The Ministry of Petroleum and Energy Facts 2015-Energy and Water Resources in Norway 2015.
(35) Scherer, L.; Pfister, S. Global water footprint assessment of hydropower. *Renewable Energy* **2016**, *99*, 711−720.
(36) Scherer, L.; Pfister, S. Hydropower's Biogenic Carbon Footprint. *PLoS One* **2016**, *11* (9), No. e0161947.
(37) Rørslett, B. An integrated approach to hydropower impact assessment. I. Environmental features of some Norwegian hydro-electric lakes. *Hydrobiologia* **1988**, *164* (1), 39−66.
(38) NVE (The Norwegian Water Resources and Energy Directorate). NVE Atlas (15.03.2016); http://atlas.nve.no.
(39) Gao, H.; Birkett, C.; Lettenmaier, D. P., Global monitoring of large reservoir storage from satellite remote sensing. *Water Resour. Res.* **2012**, *48*, (9).10.1029/2012WR012063
(40) Ozesmi, S. L.; Bauer, M. E. Satellite remote sensing of wetlands. *Wetlands Ecol. Manage.* **2002**, *10* (5), 381−402.
(41) Xie, C.; Huang, X.; Mu, H.; Yin, W. Impacts of Land-Use Changes on the Lakes across the Yangtze Floodplain in China. *Environ. Sci. Technol.* **2017**, *51* (7), 3669−3677.
(42) Liu, H.; Yin, Y.; Piao, S.; Zhao, F.; Engels, M.; Ciais, P. Disappearing lakes in semiarid northern China: drivers and environmental impact. *Environ. Sci. Technol.* **2013**, *47* (21), 12107−12114.
(43) Richards, J. A., Remote sensing digital image analysis - An Introduction. *Springer*: **2013**; *Vol. 5*.10.1007/978-3-642-30062-2
(44) USGS (United States Geological Survey). EarthExplorer-Global Land Survey 1975 dataset (01.05.2016); http://earthexplorer.usgs.gov/.
(45) Geodata, A. S. NORGE i Bilder - Image search (01.05.2016); https://www.norgeibilder.no/.
(46) Zohary, T.; Ostrovsky, I. Ecological impacts of excessive water level fluctuations in stratified freshwater lakes. *Inland Waters* **2011**, *1* (1), 47−59.
(47) Giri, C.; Ochieng, E.; Tieszen, L. L.; Zhu, Z.; Singh, A.; Loveland, T.; Masek, J.; Duke, N. Status and distribution of mangrove forests of the world using earth observation satellite data. *Global Ecology and Biogeography* **2011**, *20* (1), 154−159.
(48) Butera, M. K. Remote sensing of wetlands. *IEEE Transactions on Geoscience and Remote Sensing* **1983**, *3*, 383−392.
(49) Smith, L. C. Satellite remote sensing of river inundation area, stage, and discharge: A review. *Hydrol. Processes* **1997**, *11* (10), 1427−1439.
(50) Gutman, G.; Huang, C.; Chander, G.; Noojipady, P.; Masek, J. G. Assessment of the NASA−USGS Global Land Survey (GLS) datasets. *Remote Sensing of Environment* **2013**, *134*, 249−265.
(51) *ArcGIS Release 10.3*; ESRI (Environmental Systems Research Institute): Redlands, CA, 2014.
(52) Milà i Canals, L.; Bauer, C.; Depestele, J.; Dubreuil, A.; Freiermuth Knuchel, R.; Gaillard, G.; Michelsen, O.; Müller-Wenk, R.; Rydgren, B. Key Elements in a Framework for Land Use Impact Assessment Within LCA (11 pp). *Int. J. Life Cycle Assess.* **2007**, *12* (1), 5−15.
(53) Bednarek, A. T. Undamming Rivers: A Review of the Ecological Impacts of Dam Removal. *Environ. Manage.* **2001**, *27* (6), 803−814.
(54) East, A. E.; Pess, G. R.; Bountry, J. A.; Magirl, C. S.; Ritchie, A. C.; Logan, J. B.; Randle, T. J.; Mastin, M. C.; Minear, J. T.; Duda, J. J.; Liermann, M. C.; McHenry, M. L.; Beechie, T. J.; Shafroth, P. B. Large-Scale Dam Removal on the Elwha River River Channel and Floodplain Geomorphic Change. *Geomorphology*, Washington, U.S., 2015, *228*, 765−786.
(55) NVE (The Norwegian Water Resources and Energy Directorate). Vannkraftdatabase (20.11.2017); https://www.nve.no/energiforsyning-og-konsesjon/vannkraft/vannkraftdatabase.
(56) Rehman, S.; Al-Hadhrami, L. M.; Alam, M. M. Pumped hydro energy storage system: A technological review. *Renewable Sustainable Energy Rev.* **2015**, *44*, 586−598.
(57) Frazier, P. S.; Page, K. J. Water body detection and delineation with Landsat TM data. *Photogrammetric Engineering and Remote Sensing* **2000**, *66* (12), 1461−1468.
(58) Thematic Mapping API. World Borders Dataset (11.04.2016); http://thematicmapping.org/downloads/world_borders.php.
(59) McFeeters, S. K. The use of the Normalized Difference Water Index (NDWI) in the delineation of open water features. *International journal of remote sensing* **1996**, *17* (7), 1425−1432.
(60) Ji, L.; Zhang, L.; Wylie, B. Analysis of dynamic thresholds for the normalized difference water index. *Photogrammetric Engineering & Remote Sensing* **2009**, *75* (11), 1307−1317.
(61) Xu, H. Modification of normalised difference water index (NDWI) to enhance open water features in remotely sensed imagery. *International Journal of Remote Sensing* **2006**, *27* (14), 3025−3033.
(62) Sobek, S.; Nisell, J.; Fölster, J. Predicting the volume and depth of lakes from map-derived parameters. *Inland Waters* **2011**, *1* (3), 177−184.
(63) NVE (The Norwegian Water Resources and Energy Directorate). Dybdekart over norske innsjøer 1984 (In Norwegian with English summary) (01.06.2016); https://gis3.nve.no/metadata/tema/DKBok1984/Dybdekart_1984.htm.
(64) Kartverket. Historia om Noreg 1:50 000 (07.08.2017); http://www.kartverket.no/Kunnskap/Historie/Historien-om-Norge-1-50-000/.
(65) Hertwich, E. G.; Gibon, T.; Bouman, E. A.; Arvesen, A.; Suh, S.; Heath, G. A.; Bergesen, J. D.; Ramirez, A.; Vega, M. I.; Shi, L. Integrated life-cycle assessment of electricity-supply scenarios confirms global environmental benefit of low-carbon technologies. *Proc. Natl. Acad. Sci. U. S. A.* **2015**, *112* (20), 6277−6282.
(66) Denholm, P.; Hand, M.; Jackson, M.; Ong, S. *Land-Use Requirements of Modern Wind Power Plants in the United States*; National Renewable Energy Laboratory: Golden, Colorado, **2009**; 10.2172/964608
(67) Fthenakis, V.; Kim, H. C. Land use and electricity generation: A life-cycle analysis. *Renewable Sustainable Energy Rev.* **2009**, *13* (6−7), 1465−1474.
(68) Bakken, T. H.; Modahl, I. S.; Raadal, H. L.; Bustos, A. A.; Arnoy, S. Allocation of water consumption in multipurpose reservoirs. *Water Policy* **2016**, *18* (4), 932−947.
(69) Zlinszky, A.; Heilmeier, H.; Balzter, H.; Czúcz, B.; Pfeifer, N. Remote Sensing and GIS for Habitat Quality Monitoring: New Approaches and Future Research. *Remote Sensing* **2015**, *7* (6), 7987−7994.
(70) Hofmann, H.; Lorke, A.; Peeters, F. Temporal scales of water-level fluctuations in lakes and their ecological implications. *Hydrobiologia* **2008**, *613* (1), 85−96.
(71) Wantzen, K. M.; Rothhaupt, K.-O.; Mörtl, M.; Cantonati, M.; G.-Tóth, L.; Fischer, P. Ecological effects of water-level fluctuations in lakes: an urgent issue. *Hydrobiologia* **2008**, *613* (1), 1−4.
(72) Evtimova, V. V.; Donohue, I. Water-level fluctuations regulate the structure and functioning of natural lakes. *Freshwater Biol.* **2016**, *61* (2), 251−264.
(73) Mjelde, M.; Hellsten, S.; Ecke, F. A water level drawdown index for aquatic macrophytes in Nordic lakes. *Hydrobiologia* **2013**, *704* (1), 141−151.
(74) Koellner, T.; de Baan, L.; Beck, T.; Brandão, M.; Civit, B.; Margni, M.; i Canals, L. M.; Saad, R.; de Souza, D. M.; Müller-Wenk, R. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. *Int. J. Life Cycle Assess.* **2013**, *18* (6), 1188−1202.
(75) Koellner, T.; de Baan, L.; Beck, T.; Brandão, M.; Civit, B.; Goedkoop, M.; Margni, M.; Milà i Canals, L.; Müller-Wenk, R.;

DOI: 10.1021/acs.est.7b05125
Environ. Sci. Technol. 2018, 52, 2375−2384

Weidema, B.; Wittstock, B. Principles for life cycle inventories of land use on a global scale. *Int. J. Life Cycle Assess.* **2013**, *18* (6), 1203−1215.

(76) Merwade, V.; Cook, A.; Coonrod, J. GIS techniques for creating river terrain models for hydrodynamic modeling and flood inundation mapping. *Environmental Modelling & Software* **2008**, *23* (10−11), 1300−1311.

(77) Wang, Y.; Liao, M.; Sun, G.; Gong, J. Analysis of the water volume, length, total area and inundated area of the Three Gorges Reservoir, China using the SRTM DEM data. *International Journal of Remote Sensing* **2005**, *26* (18), 4001−4012.

(78) Hanafiah, M. M.; Xenopoulos, M. A.; Pfister, S.; Leuven, R. S. E. W.; Huijbregts, M. A. J. Characterization Factors for Water Consumption and Greenhouse Gas Emissions Based on Freshwater Fish Species Extinction. *Environ. Sci. Technol.* **2011**, *45* (12), 5272−5278.

(79) Lehtonen, H.; Rask, M.; Pakkasmaa, S.; Hesthagen, T. Freshwater fishes, their biodiversity, habitats and fisheries in the Nordic countries. *Aquat. Ecosyst. Health Manage.* **2008**, *11* (3), 298−309.

(80) Jonsson, M.; Strasevicius, D.; Malmqvist, B. Influences of river regulation and environmental variables on upland bird assemblages in northern Sweden. *Ecol. Res.* **2012**, *27* (5), 945−954.

(81) Tendall, D. M.; Hellweg, S.; Pfister, S.; Huijbregts, M. A. J.; Gaillard, G. Impacts of River Water Consumption on Aquatic Biodiversity in Life Cycle Assessment—A Proposed Method, and a Case Study for Europe. *Environ. Sci. Technol.* **2014**, *48* (6), 3236−3244.

(82) Abril, G.; Guérin, F.; Richard, S.; Delmas, R.; Galy-Lacaux, C.; Gosse, P.; Tremblay, A.; Varfalvy, L.; Dos Santos, M. A.; Matvienko, B. Carbon dioxide and methane emissions and the carbon budget of a 10-year old tropical reservoir (Petit Saut, French Guiana). *Global Biogeochemical Cycles* **2005**, *19* (4), n/a.

(83) DelSontro, T.; McGinnis, D. F.; Sobek, S.; Ostrovsky, I.; Wehrli, B. Extreme Methane Emissions from a Swiss Hydropower Reservoir: Contribution from Bubbling Sediments. *Environ. Sci. Technol.* **2010**, *44* (7), 2419−2425.

(84) Chaudhary, A.; Verones, F.; de Baan, L.; Hellweg, S. Quantifying Land Use Impacts on Biodiversity: Combining Species−Area Models and Vulnerability Indicators. *Environ. Sci. Technol.* **2015**, *49* (16), 9987−9995.

(85) De Schryver, A. M.; Brakkee, K. W.; Goedkoop, M. J.; Huijbregts, M. A. J. Characterization Factors for Global Warming in Life Cycle Assessment Based on Damages to Humans and Ecosystems. *Environ. Sci. Technol.* **2009**, *43* (6), 1689−1695.

(86) Rugani, B.; Rocchini, D. Boosting the use of spectral heterogeneity in the impact assessment of agricultural land use on biodiversity. *J. Cleaner Prod.* **2017**, *140*, 516−524.

(87) Curran, M.; Maia de Souza, D.; Antón, A.; Teixeira, R. F. M.; Michelsen, O.; Vidal-Legaz, B.; Sala, S.; Milà i Canals, L. How Well Does LCA Model Land Use Impacts on Biodiversity?—A Comparison with Approaches from Ecology and Conservation. *Environ. Sci. Technol.* **2016**, *50* (6), 2782−2795.

(88) Olson, D. M.; Dinerstein, E.; Wikramanayake, E. D.; Burgess, N. D.; Powell, G. V. N.; Underwood, E. C.; D'Amico, J. A.; Itoua, I.; Strand, H. E.; Morrison, J. C.; Loucks, C. J.; Allnutt, T. F.; Ricketts, T. H.; Kura, Y.; Lamoreux, J. F.; Wettengel, W. W.; Hedao, P.; Kassem, K. R. Terrestrial Ecoregions of the World: A New Map of Life on Earth. *BioScience* **2001**, *51* (11), 933.

(89) Abell, R.; Thieme, M. L.; Revenga, C.; Bryer, M.; Kottelat, M.; Bogutskaya, N.; Coad, B.; Mandrak, N.; Balderas, S. C.; Bussing, W.; Stiassny, M. L. J.; Skelton, P.; Allen, G. R.; Unmack, P.; Naseka, A.; Ng, R.; Sindorf, N.; Robertson, J.; Armijo, E.; Higgins, J. V.; Heibel, T. J.; Wikramanayake, E.; Olson, D.; López, H. L.; Reis, R. E.; Lundberg, J. G.; Sabaj Pérez, M. H.; Petry, P. Freshwater Ecoregions of the World: A New Map of Biogeographic Units for Freshwater Biodiversity Conservation. *BioScience* **2008**, *58* (5), 403.

(90) Abotalib, M.; Zhao, F.; Clarens, A. Deployment of a Geographical Information System Life Cycle Assessment Integrated Framework for Exploring the Opportunities and Challenges of Enhanced Oil Recovery Using Industrial CO2Supply in the United States. *ACS Sustainable Chem. Eng.* **2016**, *4* (9), 4743−4751.

(91) Fluet-Chouinard, E.; Lehner, B.; Rebelo, L.-M.; Papa, F.; Hamilton, S. K. Development of a global inundation map at high spatial resolution from topographic downscaling of coarse-scale remote sensing data. *Remote Sensing of Environment* **2015**, *158*, 348−361.

(92) Pandit, M. K.; Grumbine, R. E. Potential Effects of Ongoing and Proposed Hydropower Development on Terrestrial Biological Diversity in the Indian Himalaya. *Conservation Biology* **2012**, *26* (6), 1061−1071.

Chapter 3: Quantifying net water consumption of Norwegian hydropower reservoirs and related aquatic biodiversity impacts in Life Cycle Assessment.

Environmental Impact Assessment Review 2019, *76*, 36-46.

Environmental Impact Assessment Review 76 (2019) 36–46

Contents lists available at ScienceDirect

Environmental Impact Assessment Review

journal homepage: www.elsevier.com/locate/eiar

Quantifying *net* water consumption of Norwegian hydropower reservoirs and related aquatic biodiversity impacts in Life Cycle Assessment

Martin Dorber[a,*], Kim Rainer Mattson[a], Odd Terje Sandlund[b], Roel May[b], Francesca Verones[a]

[a] Department of Energy and Process Engineering, Norwegian University of Science and Technology (NTNU), Sem Sælands vei 7, 7491 Trondheim, Norway
[b] Norwegian Institute for Nature Research (NINA), Høgskoleringen 9, 7034 Trondheim, Norway

ARTICLE INFO

Keywords:
Life Cycle Impact Assessment
Hydropower reservoir
Water consumption
Fish
Characterization factor
Species-discharge relationship

ABSTRACT

Compared to conventional energy technologies, hydropower has the lowest carbon emissions per kWh. Therefore, hydropower electricity production can contribute to combat climate change challenges. However, hydropower electricity production may at the same time contribute to environmental impacts and has been characterized as a large water consumer with impacts on aquatic biodiversity. Life Cycle Assessment is not yet able to assess the biodiversity impact of water consumption from hydropower electricity production on a global scale. The first step to assess these biodiversity impacts in Life Cycle Assessment is to quantify the water consumption per kWh energy produced. We calculated catchment-specific net water consumption values for Norway ranging between 0 and 0.012 m^3/kWh. Further, we developed the first characterization factors for quantifying the aquatic biodiversity impacts of water consumption in a post-glaciated region. We apply our approach to quantify the biodiversity impact per kWh Norwegian hydropower electricity. Our results vary over six orders of magnitude and highlight the importance of a spatial explicit approach. This study contributes to assessing the biodiversity impacts of water consumption globally in Life Cycle Assessment.

1. Introduction

Hydropower electricity production has the lowest carbon emissions per kWh of all conventional energy technologies (Barros et al., 2011) and can provide access to affordable and reliable energy (United Nations, 2015; Edenhofer et al., 2011; Hertwich et al., 2016). Therefore, hydropower electricity production can contribute to fulfilling two of the 17 Sustainable Development Goals (SDG), developed by the United Nations for a transition into a sustainable world (United Nations, 2015), namely SDG 7 (Affordable and Clean Energy) and SDG 13 (Climate action). However, both the United Nations Environment Program (UN Environment) (Hertwich et al., 2016) and the Intergovernmental Panel on Climate Change (IPCC) (Edenhofer et al., 2011) point out that there are potential ecological trade-offs related to hydropower electricity. Freshwater habitat alteration, land use change and water quality degradation have been identified as the main cause-effect pathways of hydropower electricity production on biodiversity (Gracey and Verones, 2016). These 3 cause-effect pathways may lead to local species extinctions (McAllister et al., 2001) of, for example, fish and macroinvertebrate species (Poff and Zimmerman, 2010; Crook et al., 2015), as well as terrestrial flora and fauna (Jansson et al., 2000; Alho, 2011; Kitzes and Shirley, 2016; Zhang et al., 2009; Tefera and Sterk, 2008). As the 17 SDGs can be viewed as a network (Le Blanc, 2015), with interdependent goals (Nilsson et al., 2016), the terrestrial and aquatic biodiversity impacts of hydropower electricity production therefore may interfere with SDG 6 (Clean Water and Sanitation) and SDG 15 (Life on Land). Thus, a sustainable hydropower development, with minimized trade offs between the SDGs (Nilsson et al., 2016; Bhaduri et al., 2016), requires an assessment of all relevant biodiversity impacts.

The report from UN Environment on green energy choices (Hertwich et al., 2016) recommends using Life Cycle Assessment (LCA) to assess potential trade-offs between renewable energy sources. LCA is a tool which is commonly used for analyzing the environmental impacts of a product or process throughout its life cycle (ISO, 2006a, b). However, the report from UN Environment does not quantify relevant biodiversity impacts from hydropower production in LCA, due to a lack of mature assessment methods (Hertwich et al., 2016; Gracey and Verones, 2016; Winter et al., 2017).

Our study focuses on freshwater habitat alteration, one of the main threats for aquatic biodiversity (Vörösmarty et al., 2010). Besides, the conservation of aquatic biodiversity has been identified as one of the key parameters for sustainable development (United Nations, 2015; Secretariat of the Convention on Biological Diversity, 2014). For

* Corresponding author at: Sem Sælands vei 7 NO-7491, Trondheim, Norway.
E-mail address: martin.dorber@ntnu.no (M. Dorber).

https://doi.org/10.1016/j.eiar.2018.12.002
Received 25 June 2018; Received in revised form 12 December 2018; Accepted 13 December 2018

freshwater habitat alteration, storage and pumped storage hydropower plants are most relevant, since they store water in reservoirs to allow flexible electricity production (Egré and Milewski, 2002).

The operation of hydropower reservoirs replaces various habitat types like forest, peatlands and water bodies with one large water surface (Strachan et al., 2016). This new water surface will evaporate water permanently during ice-free periods, while the possible inundated terrestrial surface will evaporate water only temporarily (Strachan et al., 2016). Due to this increased evaporation (Strachan et al., 2016), hydropower electricity production has been characterized as a large consumer of water (Mekonnen and Hoekstra, 2012). Following ISO 14046 (ISO, 2014) the alteration in evaporation caused by land use change of hydropower reservoirs is considered as water consumption. We use “water consumption” in this sense throughout the paper.

In LCAs of hydropower electricity production, a prerequisite for quantifying biodiversity impacts of water consumption is to quantify the water consumption per kWh energy produced for the Life Cycle Inventory (LCI) (ISO, 2014; Rebitzer et al., 2004; Flury and Frischknecht, 2012). This has to be done in a spatially explicit way, because underlying environmental parameters (such as precipitation, topography and climatic conditions (Flury and Frischknecht, 2012)) may vary considerably (Bakken et al., 2013; Deemer et al., 2016; Mutel and Hellweg, 2009; Mutel et al., 2012). Global assessments of water consumption values from hydropower reservoirs are not available (Bakken et al., 2016a), and in LCI databases (e.g. (Wernet et al., 2016)) spatially-explicit water consumption parameters related to hydropower reservoirs are only available for Switzerland and Brazil (Flury and Frischknecht, 2012). In addition, the dominant approach for published estimates of water consumption is the *gross* method (Bakken et al., 2013). Compared to the *net* method, the *gross* method does not account for evaporation losses of the natural lake prior to the inundation of the reservoir (Flury and Frischknecht, 2012; Scherer and Pfister, 2016a, b). As a consequence, all currently available hydropower LCI water consumption parameters represent overestimated values. Using these values leads to an overestimation of the total environmental impact. Hence, the *net* water consumption method should be the preferred approach (Bakken et al., 2013).

Water consumption leads to a reduction of the yearly average discharge downstream of the hydropower reservoir (Kumar et al., 2011; Biemans et al., 2011). Further, reservoirs can be used to store water in times of surplus and to produce electricity with a release of water during peak energy demand or drier season. This reservoir operation can in addition change the frequency of the flow magnitude (Richter et al., 1997) downstream of the hydropower reservoir (Kumar et al., 2011). However, this represents water use (ISO, 2014) (not water consumption) and is beyond the scope of this paper.

To quantify biodiversity impacts of water consumption in Life Cycle Impact Assessment (LCIA), characterization factors (CFs) quantifying the Potentially Disappeared Fraction of Species (PDF) per unit of water consumed are required (Rebitzer et al., 2004; Milà et al., 2008; Pennington et al., 2004). PDF is the recommended endpoint from the Life Cycle Initiative hosted by UN Environment to assess ecosystem quality damages (Verones et al., 2017). The existing CFs do not differentiate between the cause of water consumption. They assume that water consumption due to evaporation, water withdrawal for irrigation, industrial production, or residential needs, has in principle the same impact on freshwater biodiversity (Tendall et al., 2014; Hanafiah et al., 2011).

Spatially-explicit CFs for water consumption impacts on aquatic biodiversity have been globally developed for areas below 42°N, and for Europe with a focus on Switzerland (Tendall et al., 2014; Hanafiah et al., 2011). All these CFs are based on Species-discharge relationships (SDR), which relates the discharge rates of given rivers to the associated species richness (Xenopoulos and Lodge, 2006).

The main reason for excluding areas at latitudes above 42°N is that these river basins were recently (in geological time) glaciated and have not had time to reach their maximum species richness potential (Tendall et al., 2014; Hanafiah et al., 2011). This means that for Canada, Norway, Sweden, Finland, and Iceland, which have been glaciated during the last glacial maximum (Clark et al., 2009) and accounted together for 11.8% of the global hydropower electricity production in 2016 (IEA, 2017), no spatially-explicit CFs exist to assess impacts of water consumption on biodiversity.

The first aim of this study is to calculate *net* water consumption values of hydropower electricity production for the LCI. Due to data availability we limit the calculation of *net* water consumption values to Norway, which is one of the top-ten hydropower electricity producers worldwide (Manzano-Agugliaro et al., 2013) and where the government corroborates that hydropower electricity production has significant environmental impacts on rivers that should be assessed (Norwegian Government Ministries and Offices, Meld. St. 25, 2015–2016). However, our suggested framework has the potential to be used in other regions, given that data are available.

The second aim of the study is to develop the first spatially-explicit CFs for water consumption in post-glaciated regions, based on regionally specific SDRs for fish, accounting for local variation in fish fauna by delineating regions with the same postglacial freshwater fish immigration history. Due to data availability, we only develop CFs for Norway. The output is a set of catchment specific CFs that express the fish biodiversity loss in PDF per unit water consumed for Norway. Due to data availability and the complexity to reconstruct the postglacial immigration history of species, we only consider fish species in this study, as they are good indicators of ecosystem health (Schiemer, 2000).

The third aim of this study is to use the provided LCI values and CFs to calculate the impact on aquatic biodiversity of water consumption from Norwegian hydropower reservoirs in LCA. Further, it enhances the development of CFs quantifying the impact on aquatic biodiversity of water consumption in other glaciated regions.

2. Method

2.1. Quantifying water consumption for the Life Cycle Inventory

Water consumption can be divided into three components: green water consumption (consumptive use of rain water), blue water consumption (consumptive use of ground or surface water) and grey water consumption (the volume of water polluted) (Mekonnen and Hoekstra, 2012). The water consumption quantified in this study follows the ISO 14046 (ISO, 2014) and only concerns blue water consumption in the form of evaporation from reservoirs during the use phase for storage hydropower plants (ISO, 2014).

Two main methods exist to calculate water consumption from hydropower reservoirs: *gross* water consumption and *net* water consumption. *Gross* water consumption is the most commonly used method and equates the evaporation of the actual reservoir divided by the annual electricity production. As the reservoir area could originally have been either a natural lake or a terrestrial area the *gross* water consumption does not account for evaporation losses *prior to* the construction of the hydropower reservoir. This is leading to an overestimation of the water consumption (Bakken et al., 2013). In contrast, the *net* water consumption method accounts for the evaporation losses *prior to* the construction of the hydropower reservoir, i.e. the evaporation rates from the actual reservoir surface area minus the evaporation rates *prior* to the reservoir construction divided by annual power production. Because the majority of Norwegian hydropower reservoirs are dammed natural lakes (Dorber et al., 2018), the net water consumption is used in this study. Consequently, calculation of the *net* water consumption requires open water evaporation rates from the actual reservoir surface, as well as land use change information, including evaporations rates of the terrestrial area *prior* to reservoir inundation. To estimate open water evaporation, several methods, including empirical, water budget, energy budget, or mass transfer exits. These methods can all be applied either alone or in combination (Mekonnen and Hoekstra, 2012; Mengistu and Savage, 2010). The Penman-Monteith equation with heat storage, a combination method of energy budget and mass transfer, is often considered most suitable for estimating open water evaporation from hydropower reservoirs (Mekonnen and Hoekstra, 2012; Rosenberry et al., 1993). However, this approach can neither be applied to Norway nor globally, as the necessary in situ data on, for example, water temperature and wind speed, are not available in the required, detailed spatial scale (Finch, 2001). Therefore, we use the potential evapotranspiration (PET) as proxy for the open water evaporation (Lee et al., 2014), as for example done by Pfister et al. (2011) and Scherer and Pfister (2016a). Evapotranspiration (ET) can be defined as the amount of water transferred to the atmosphere by evaporating water from plants or soil surfaces (Mu et al., 2007). PET is the amount of evapotranspiration which occurs when an infinite amount of water is available (Westerhoff, 2015). AET is defined as the amount of evapotranspiration happening under local water conditions (Westerhoff, 2015), affected by annual rainfall, vegetation type and climatic conditions (Zhang et al., 2001).

The validity of this assumptions is confirmed by, e.g., Lee et al. (2014) who report a difference of 5% between satellite based PET estimates and open water evaporation measurements, and Douglas et al. (2009) who report a difference of up to 6% between Penman–Monteith PET estimates and open water evaporation measurements. However, the rates can differ depending on the PET estimation method (Rosenberry et al., 1993; Douglas et al., 2009; Lu et al., 2005).

The evaporation rates from the actual reservoir equals the potential evapotranspiration of the actual reservoir surface area. To calculate the evaporation rate *prior* to the reservoir construction, land use information *prior* to reservoir construction is needed. The evaporation rate *prior* to the reservoir construction is the PET occurring on the natural water surface area plus the AET occurring on the later inundated terrestrial land area. As there is no change in PET from changing 1 m^2 natural water surface area to 1 m^2 reservoir surface area, the net water consumption only considers the difference between PET and AET of the inundated land area. As the water consumption of all hydropower reservoirs in a catchment leads to a discharge reduction in the same main river, the catchment level is chosen as a system boundary. Thus, the *net* water consumption [m^3/kWh] in catchment x for the LCI can be calculated according to Eq. (1).

$$\text{Net water consumption}_x = \frac{\sum_{y=0}^{k} \frac{((\mathrm{PET}_y - \mathrm{AET}_y) \times \mathrm{ILA}_y)}{1000}}{\sum_{y=0}^{k} \mathrm{ER}_y} \tag{1}$$

where k is the number of reservoirs with inundated land data in catchment x, *PET* is the average yearly potential evapotranspiration in mm/year of reservoir y, *AET* is the average actual evapotranspiration in mm/year of reservoir y, *ILA* is inundated land area in m^2 due to the reservoir creation of reservoir y and *ER* is the average annual electricity production in kWh of reservoir y.

The average yearly potential evapotranspiration and average yearly actual evapotranspiration were obtained from the MODIS Global Evapotranspiration Project (MOD16) (Mu et al., 2007, 2011; University of Montana, 2011). MOD16 is based on the Penman-Monteith equation and by using Land Cover Data, the Leaf Area Index and a modified version of the Normalized Difference Vegetation Index, the MOD16 is able to distinguish the evaporation rates of different vegetation types. It offers an average potential evapotranspiration and average actual evapotranspiration for the period 2000–2013 in a 1-km^2 resolution for the whole globe (Mu et al., 2011).

To calculate *PET* we averaged the MOD16 PET values inside the actual reservoir surface area at highest regulated water level (RSA) provided by the Norwegian Water Resources and Energy Directorate (NVE) (NVE (The Norwegian Water Resources and Energy Directorate), 2016) (see Supporting Information 2 (SI2)). *AET* could not be calculated directly, because MOD16 assesses the status after reservoir inundation, and information about the vegetation and soil composition prior to inundation does not exist (Dorber et al., 2018). Therefore, we had to assume that a buffer around the shoreline of the actual reservoir, represents the vegetation and soil composition prior to inundation. Based on this assumption we assessed *AET* by averaging the MOD16 actual evapotranspiration in a 2-pixel buffer around the shoreline of the actual reservoir in ArcGIS10.3 (ESRI (Environmental Systems Research Institute), 2014) (see Supporting Information 2 (SI2)). The sensitivity of this assumption will be tested and discussed in Section 3.2. Inundated land area data are obtained from Dorber et al. (2018).

2.2. Uncertainty and sensitivity of water consumption calculations

Main contributors to uncertainty of the calculated *net* water consumption are evaporation estimates, inundated land area estimates and water-level fluctuations. For evaporation estimation from the MOD16 project, Mu et al. (2011) report an average mean absolute bias of 24.6% for the *AET* value. We account for this uncertainty by calculating a *net* water consumption due to AET using 24.6% higher and lower *AET* values (see Supporting Information 1 (SI1), section S2 and SI2). To account for uncertainty related to inundated land area assessment, we calculate a *net* water consumption with the standard deviation (SD) of the adjusted inundated land area data from Dorber et al. (2018). Further, Dorber et al. (2018) calculated the inundated land area related to the actual reservoir surface area at the highest regulated water level. The common operational scheme for Norwegian reservoirs is characterized by a distinct decline in water level during winter followed by a significant increase in spring, and an almost stable water level during summer and autumn (Mjelde et al., 2012; Eloranta et al., 2018). Additionally, most Norwegian hydropower reservoirs are generally filled to less than 90% of maximum capacity (Norwegian Water Resources and Energy Directorate, 2017). Consequently, the actual reservoir surface area at the highest regulated water level may not be reached over the whole year. Thus, our *net* water consumption values, which do not cover seasonal water-level fluctuations, are most likely overestimations. As the relationship between water level and water surface area is not available for Norwegian hydropower reservoirs (Mekonnen and Hoekstra, 2012), the uncertainty of this temporal aspect cannot be quantified directly. Therefore, we test the sensitivity of water-level fluctuations on the calculated *net* water consumption value by reducing the inundated land area. To test the sensitivity of the assumption that a buffer around the actual reservoir represents the vegetation prior to inundation, we calculate the *net* water consumption in addition with a 1-pixel buffer (see SI2).

2.3. Aquatic species loss per unit change of discharge

To assign biodiversity damage to water consumption from the LCI in LCIA on a damage level, a characterization factor for each catchment needs to be developed. The CF denotes the Potentially Disappeared Fraction of Species per unit of water consumption (Verones et al., 2017). In this study, we used the Species-discharge relationship concept already applied within LCIA for the derivation of water consumption CFs (Tendall et al., 2014; Hanafiah et al., 2011). As species richness is positively correlated with mean annual discharge (Oberdoff et al., 1995; Poff et al., 2001; Xenopoulos et al., 2005), the SDR is a model that relates river discharge to species richness within a catchment (Xenopoulos and Lodge, 2006). This relationship can therefore be used to predict the species loss per unit change of discharge (Xenopoulos and Lodge, 2006).

In regions where SDRs have already been developed, fish species richness variability can be statistically explained as a function of mean annual discharge (Oberdoff et al., 1995). However, in the northern Hemisphere, including Norway, species richness variability is additionally explained by historical glaciation events and postglacial immigration history (Oberdorff et al., 1997; Reyjol et al., 2007; Leprieur et al., 2009), which caused variation on a local scale. An SDR developed for the whole of Norway is weak, because even today postglacial immigration plays an important role for species richness variability (Hanafiah et al., 2011). Therefore, the first step in developing regional SDRs for Norway is to identify catchments with similar glaciation and dispersal history. Within each catchment, species richness is subsequently correlated with mean annual discharge. Consequently, catchment-specific SDRs are calculated.

2.3.1. Identifying catchments with similar glaciation and dispersal history

During the last glacial maximum the northern parts of Europe were covered by ice or permafrost (Reyjol et al., 2007). Many fish species in the northern part of the continent were unable to migrate along a north–south gradient and therefore became locally extinct (Reyjol et al., 2007). The surviving fish species shifted south into so-called glacial refugia (Reyjol et al., 2007; Leprieur et al., 2009; Hänfling et al., 2002; Refseth et al., 1998; García-Marín et al., 1999; Griffiths, 2006). From these refugia, recolonization of all freshwater fish species into Scandinavia occurred when the ice retreated after the last glaciation (approx. 10,000 years ago) (Refseth et al., 1998). As catchments are separated by barriers that are insurmountable for freshwater fish (land masses or oceans), the movement of freshwater fish into Norway is defined by the connectivity of water bodies through rivers and streams (Leprieur et al., 2009). Saltwater-tolerant (anadromous) fish were able to colonize coastal Norway via the sea from the West, while non-anadromous freshwater fish probably colonized Norwegian water courses from the East or Southeast from the Baltic Sea refugium, or from the south following the retreating glacial front (Refseth et al., 1998). Colonization via the seas is considered a fast process in comparison to colonization via land masses (Oberdorff et al., 1997; Reyjol et al., 2007). Fish migration via land masses could only happen during marine regressions when sea levels decreased and new freshwater connections between catchments became possible (Reyjol et al., 2007). During the last glacial maximum a decrease in sea levels by 20 m occurred (Patton et al., 2017). Alternatively, fish migration via land mass occurred when the water of melting glaciers connected catchments located on opposite sides of mountain ridges (Oberdorff et al., 1997; Reyjol et al., 2007).

To account for the colonization history in Norway via the seas, we select catchments according to their associated marine ecoregion (Spalding et al., 2007). This assumes that the distance to the refugia and also the recolonization time is equal for all catchments draining into the same marine ecoregion. Following Reyjol et al. (2007), the selection of catchments by marine ecoregions also accounts for colonization via marine regression, assuming that these catchments experienced the same sea-level lowering. To account for colonization through surface waters in land masses, we select catchments by the freshwater ecoregions they belong to. Freshwater ecoregions are partially defined by geological processes, speciation, glaciation history, climatic and physiographic patterns, and dispersal barriers, with a focus on freshwater fish species (Abell et al., 2008). Thus, a region with similar colonization history is delineated by those catchments located in the same freshwater region and draining into the same marine ecoregion (Fig. 2) (SI1, S3).

2.3.2. Developing regional SDRs for Norway

Species-discharge relationships for each of the identified regions with similar colonization history are derived by curve-fitting the relationship between the discharge rates and the fish species richness of a given catchment. Annual runoff for the period 1961–1990 in each catchment is provided by NVE (NVE (The Norwegian Water Resources and Energy Directorate), 2016). We use the oldest available period, to represent the natural flow situation before hydropower. Fish species occurrence data are obtained through the publicly available database and map services Artsdatabanken (The Norwegian Biodiversity Information Centre and GBIF Norway, 2017) and GBIF (The Norwegian Biodiversity Information Centre and GBIF Norway, 2017; Finstad et al., 2017; Hesthagen and Gravbrøt, 2017; Natural History Museum, University of Oslo, 2017; Vang, 2017a, b). We exclude freshwater fish species classified as introduced from Fishbase (Froese and Pualy, 2018) and obtained 140,311 fish occurrence points, collected between 1869 and 2017 in 1463 catchments (SI1,S4). For reasons of comparability, we use the power function commonly employed in LCA to calculate the SDR (Tendall et al., 2014). The SDR function is solved analytically, as shown in Eq. (2).

$$S = a \times x^{b}$$

$$dS = (b \times a) \times x^{(b-1)} \tag{2}$$

a and b are model coefficients produced by the regression model, whereas x signifies the discharge rate [m^3/y] of the catchment in question. The SDR equates how many species S we would expect within a catchment, whereas dS (the derivative of the SDR power function) tells us how the number of fish species changes as we change the discharge by one unit (m^3/y).

As some sites are more likely to be surveyed than others (Phillips et al., 2009), the number of species occurrence points varies in each catchment. We assume that the accuracy of species richness estimates increases when more occurrences are recorded in a catchment. To account for this assumption we weigh the power function fitting by the total number of occurrence records in each catchment (SI1, S4) (Motulsky and Christopoulos, 2004). Power function fitting was performed in MATLAB version R2015a using the nonlinear least squares method (Mathworks, 2015). We do not calculate SDRs for Norwegian catchments with rivers that flow into Sweden or Finland or catchments in Norway where more than 30% of the area is located outside Norway, because discharge and species richness data for these catchments are not available in an exhaustive and comparable way.

2.3.3. Calculation of the characterization factor

The characterization factor (CF) [$PDF*y/m^3$], consisting of a Fate Factor (FF) [m^3*y/m^3*y] and Effect Factor (EF) [$PDF*y/m^3$], quantifies the downstream impact of water consumption in catchment x on freshwater fish species in Norway, and can be expressed by Eq. (3). The FF models the river discharge reduction of a unit water consumed and the EF relates the intensity of a unit water consumed to a quantified biodiversity effect.

$$CF_x = FF_x \times EF_x = \frac{dQ}{dW} \times \frac{\frac{dS_x}{R_x}}{dQ} \tag{3}$$

The FF is adopted from Hanafiah et al. (2011), where dQ is the marginal change in discharge [m^3/y] and dW is the marginal change in water consumption [m^3/y]. The FF equals one, as one unit change in water consumption (e.g. 1 m^3 evaporation) leads to one unit reduction of river discharge. For EF, dS is the derivative of the SDR power function developed for the related region in Norway (see Eq. (3)), used to find the species loss per unit change of discharge. R is the total fish species richness of catchment x, which is the maximum number of species predicted by the SDR. The ratio of dS to R gives the potentially disappeared fraction of fish species loss per unit water consumption. In our case, dQ is always 1 m^3/y, to link it with the water consumption of the Life Cycle Inventory. We calculated the 95% simultaneous confidence intervals of the fitted power function and the related coefficients in each region with MATLAB version R2015a (Mathworks, 2015) to quantify the uncertainty of the CFs.

Water consumption due to water withdrawal for irrigation, industrial production, or residential needs can in principle have the same impact on the freshwater biodiversity (Tendall et al., 2014; Hanafiah et al., 2011). Therefore, the developed CFs are applicable to all fields of blue water consumption in Norway, with related LCI data, and are not limited to the quantification of water consumption impacts from

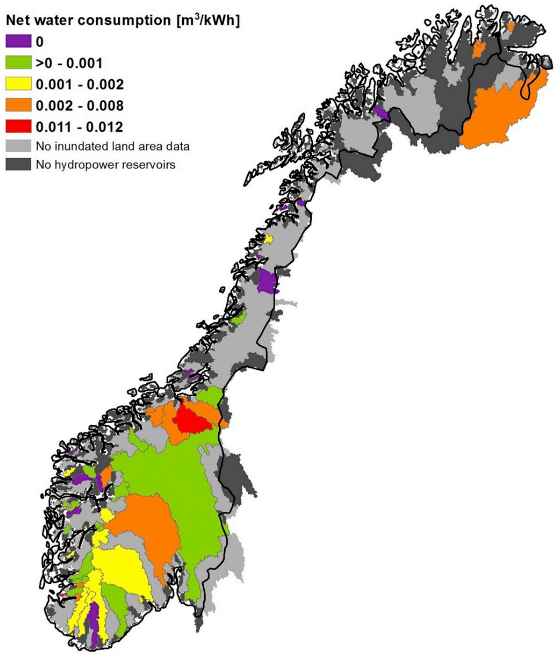


Fig. 1. *Net* water consumption per kWh calculated from the adjusted inundated land area for Norway (Thematic Mapping API World Borders Dataset, 2009). In grey areas no inundated land information was available. In the dark grey areas no hydropower reservoirs exist. Catchment information obtained from the Norwegian Water Resources and Energy Directorate (NVE (The Norwegian Water Resources and Energy Directorate), 2016).

hydropower. To showcase the applicability of our results we calculate the impact on aquatic biodiversity of water consumption from hydropower electricity production in Norwegian catchments in Section 3.5.

3. Results

3.1. Net water consumption

We calculate *net* water consumption values for 63 out of 1833 Norwegian catchments including 107 reservoirs (Fig. 1). For the remaining catchments no *net* water consumption values could be calculated, due to a lack of reservoirs with inundated land area data (Dorber et al., 2018). The average *net* water consumption was 0.0016 m^3/y, with a minimum of 0 m^3/kWh and a maximum of 0.012 m^3/kWh. A value of 0 m^3/kWh indicates that a natural lake existed prior to the dam construction and that its surface area was not increased.

3.2. Uncertainty and sensitivity of water consumption

Accounting for uncertainty in the actual evapotranspiration results in an average *net* water consumption due to AET that differs by 0.0007 m^3/kWh, respectively 42.6% relative to the average *net* water consumption presented before. Hence, the average *net* water consumption due to AET, varies between 0.0009 m^3/kWh and 0.0023 m^3/kWh. Accounting for inundated land area estimation uncertainty results in an average *net* water consumption due to inundated land area that varies between 0.0014 m^3/kWh and 0.002 m^3/kWh, respectively −20.1% and 22.9% relative to the average *net* water consumption. The calculation procedure for the inundated land area uncertainty reveals that a reduction of the inundated land area by 1% results in an average reduction of 0.000016 m^3/kWh, respectively 1% relative to the average *net* water consumption. The difference between the *net* water consumption calculated with actual evapotranspiration within a 2-pixel buffer in comparison to a 1-pixel buffer varies between 11.2% and −9.7%, with an average of 1.2%. For a visualization of the estimated uncertainty and further explanations see Supporting Information 1, Section S2.

3.3. Regional SDRs

For Norway, we identify eight regions where catchments are draining into the same marine or freshwater ecoregion (Fig. 2). We develop an SDR for five of the eight identified regions. It is not possible to develop a SDR for region 4 and region 6, because they only consist of one catchment each. Region 8 includes only catchments with rivers flowing into Sweden and Finland, so no SDR is developed, due to a lack of data. The fit of the power functions, reflected in the R^2, varies between 0.43 and 0.81.

3.4. Characterization factors

Based on the five SDRs, we calculate characterization factors for 1790 of 1833 catchments in Norway varying between $7.1*10^{-12}$ PDF*y/m^3 and $8.0*10^{-7}$ PDF*y/m^3 (Fig. 3). For the remaining 43 catchments, no characterization factors are calculated as these are either situated in region 4 and region 6 or overlapped with Sweden.

The CFs in Fig. 3 do not follow the pattern of the regions identified in Fig. 2. The new pattern can be explained by the fact that we are calculating the Potentially Disappeared Fraction of Species as the species loss per m^3 water consumed divided by the fish species richness of catchment *x*. Even if the relative species loss per m^3 water consumed is the same for a small and a large catchment, the small catchment will get the comparably higher PDF*y/m^3 value, because it has a comparably lower fish species richness. For further explanations, see Supporting Information 1, Section S6.

By using the 95% confidence intervals of the fitted power function we estimate an uncertainty of respectively ± 28% in Region 1, ± 5% in Region 2, ± 18% in Region 3, ± 8% in Region 5, and ± 9% in Region 7, relative to the characterization factors. Therefore, the CFs considering uncertainty vary between $6.00*10^{-12}$ PDF*y/m^3 and $8.35*10^{-7}$ PDF*y/m^3. The CF values are provided in Supporting Information 1, Section S6 and Supporting Information 2.

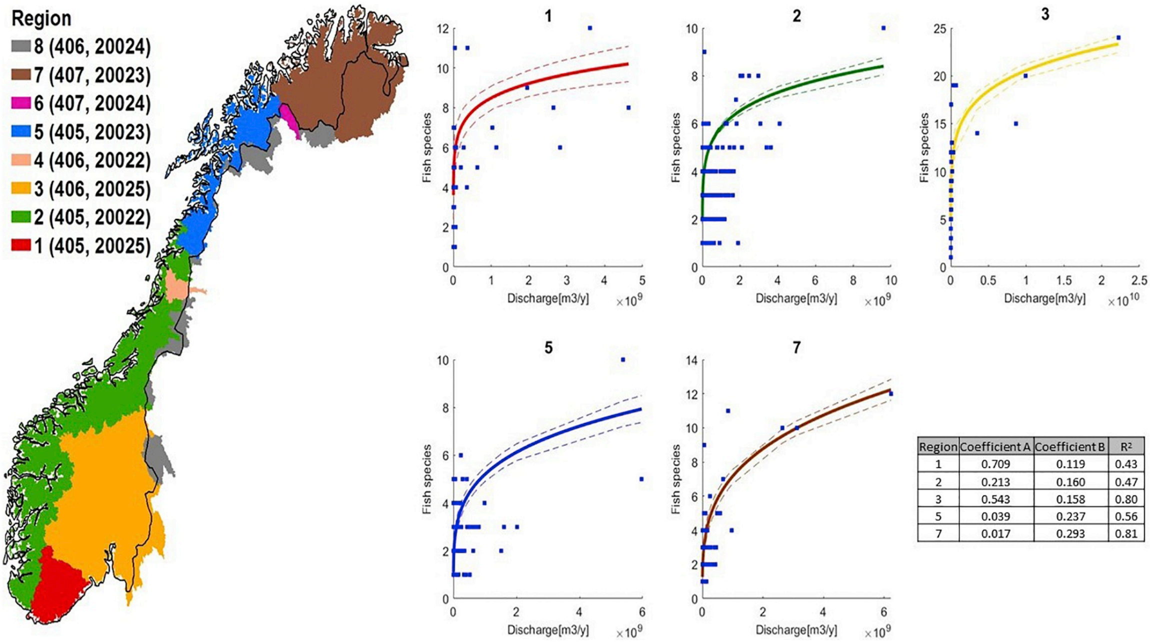
3.5. Application

To showcase the applicability of our results we calculate the impact on aquatic biodiversity of water consumption from hydropower electricity production in Norwegian catchments by multiplying the *net* water consumption LCI values with the regional CFs assessed in this study (Fig. 4). The functional unit is 1 kWh hydropower produced. In cases where no catchment-specific inventory parameter is available we average the available *net* water consumption value on freshwater ecoregions (405 = 0.0014 m^3/kwh; 406 = 0.0023 m^3/kwh; 407 = 0.0038 m^3/kwh) (Abell et al., 2008). We used freshwater ecoregions as they can be used to categorize water bodies (Abell et al., 2008). However as for this purpose no standard LCA methodology exists (Mutel et al., 2012), our approach should be seen as one example.

4. Discussion

4.1. Water consumption for the Life Cycle Inventory

This is the first study providing *net* water consumption values of storage hydropower plants for the Life Cycle Inventory with estimated uncertainty. The unit of the modelled *net* water consumption is m^3/kWh, which is in accordance with the unit of m^3 water consumption in



Region	Coefficient A	Coefficient B	R^2
1	0.709	0.119	0.43
2	0.213	0.160	0.47
3	0.543	0.158	0.80
5	0.039	0.237	0.56
7	0.017	0.293	0.81

Fig. 2. Left: Regions where catchments are draining into the same marine or freshwater ecoregion (3 digit number = Freshwater Ecoregion (Abell et al., 2008) code; 5 digit number = marine ecoregion (Spalding et al., 2007) code); Right: Developed SDRs (solid line) and confidence interval (dashed lines) with corresponding coefficients and R^2 for each of the regions.

the commonly used water consumption inventory (Koellner et al., 2013). This makes our *net* water consumption values calculated for Norwegian catchments directly implementable in LCI databases (Pekel et al., 2016; Lehner et al., 2011). The average *net* water consumption for Norway in our study across all investigated catchments was 0.0016 m^3/kWh, which is 25% smaller than the existing value in the Ecoinvent database (0.002 m^3/kWh) (Flury and Frischknecht, 2012). Thus, current Life Cycle Impact Assessments of water consumption from Norwegian hydropower reservoirs would overestimate a potential impact by 25%. *This* highlights that spatially-explicit inventory modelling is needed (ISO, 2014; Flury and Frischknecht, 2012; Bakken et al., 2013; Mutel and Hellweg, 2009) to assess the impact of water consumption on a global scale in LCA (Núñez et al., 2016). By using remote sensing assessed reservoir inundated land area (Pekel et al., 2016) and global hydropower reservoirs data (Lehner et al., 2011) in combination with the global MOD16 evaporation model, the methodology for Norway developed in this study has the potential to be applied globally. Therefore, this study contributes to providing a method to assess the biodiversity impact of water consumption from hydropower electricty production, which is a requirement for LCA purposes (Núñez et al., 2016).

We choose the MOD16 model with the Penman-Monteith equation, as it provides global evaporation values. It therefore enhances the development of *net* water consumption values for the LCI of hydropower electricity production on a global scale. The basis for our calculated *net* water consumption are the evaporation values under the climatic conditions from 2000 to 2013. These values do not accommodate for the fact that evaporated water may return as precipitation in the same catchment (Bakken et al., 2013). This may lead to an overestimation of the *net* water consumption. Abstraction of water in hydropower tunnels is also not included. If evaporation rates change under further climate change scenarios (Hanafiah et al., 2011), new *net* water consumption values will have to be calculated.

A *net* water consumption value for only 63 of 1833 catchments could be calculated, due to a limited number of reservoirs with inundated land area (Dorber et al., 2018). However, the availability of data on 63 catchments, including 107 reservoirs, adds important information from Norway to the 52 reservoirs assessed to calculate a water consumption for Switzerland in the existing Ecoinvent database (Flury and Frischknecht, 2012). Seven out of the 107 reservoirs are used as multipurpose reservoirs (Dorber et al., 2018; NVE (The Norwegian Water Resources and Energy Directorate), 2016). In these cases hydropower electricity production might not be the only factor causing water consumption, wherefore the resulting water consumption in multipurpose reservoirs should be allocated to all use purposes (Bakken et al., 2016b; Scherer and Pfister, 2016b). For four out of the seven multipurpose reservoirs, a net water consumption of 0 m^3/kWh was calculated. Following, in these cases allocation would not have an influence on the results. As the remaining three reservoirs are only used as flood protection dams in addition to hydropower electricity production, we have not included an allocation factor. Consequently, our calculated *net* water consumption values may overestimate the water consumption caused by electricity production for these three hydropower reservoirs.

During the whole life cycle of a storage power plant, the dam construction and the reinvestment contribute additionally to the total water consumption. For Norway a contribution of 67.8% from the use-phase of storage power plants of the total water consumption has been reported (Bakken et al., 2016a). This is indicating that water consumption of the use-phase is the major contributor to the total water consumption.

4.2. Uncertainty and sensitivity levels in water consumption estimation

The average *net* water consumption considering AET uncertainty varies between 0.0009 m^3/kWh and 0.0023 m^3/kWh. Accounting for

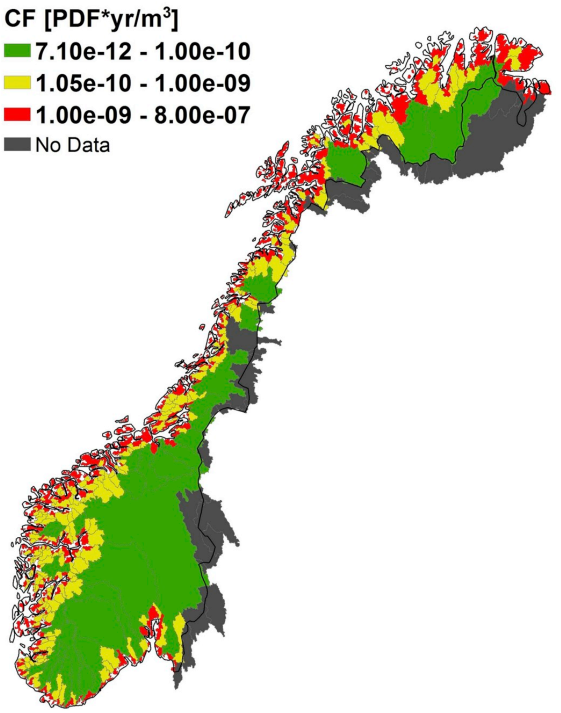


Fig. 3. Results of catchment-specific characterization factors quantifying the marginal impact of net water consumption on freshwater fish species in PDF*y/m^3. Catchment information obtained from the Norwegian Water Resources and Energy Directorate (NVE (The Norwegian Water Resources and Energy Directorate), 2016).

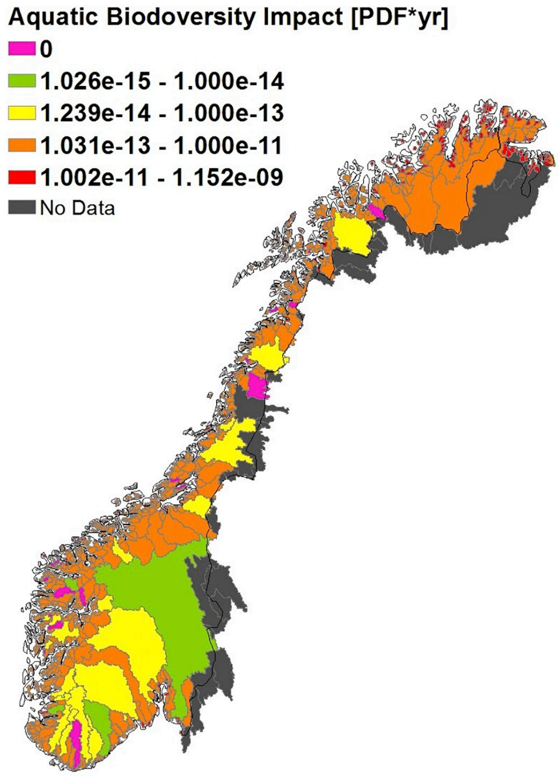


Fig. 4. Impact on aquatic biodiversity of water consumption from 1 kWh hydropower electricity production in Norwegian catchments [PDF*yr]. Catchment information obtained from the Norwegian Water Resources and Energy Directorate (NVE (The Norwegian Water Resources and Energy Directorate), 2016).

uncertainty of the inundated land area results in an average *net* water consumption that varies between 0.0014 m^3/kWh and 0.002 m^3/kWh.

We have investigated evaporation and inundated land uncertainty separately. A combined assessment of both uncertainties is not possible, as the standard deviation of the inundated land area is obtained directly for each reservoir, while the evaporation uncertainty is only available as average mean absolute error based on field stations not located in Norway. A reduction of the inundated land area by 1% results in an average reduction of 1% relative to the average *net* water consumption. This indicates a linear relationship between the calculated *net* water consumption and water-level fluctuations. However, as the relationship between water level and water surface area is not available for Norwegian hydropower reservoirs (Mekonnen and Hoekstra, 2012), the overestimation cannot be quantified directly. This highlights the need for quantifying the relationship of water level and water surface for all Norwegian hydropower reservoirs, to account for water-level fluctuations in *net* water consumption values.

The proportional difference between the *net* water consumption calculated with actual evapotranspiration within a 2-pixel buffer in comparison to a 1-pixel buffer varies between 11.2% and −9.7% with an average of 1.2% (SI2). Our finding, that the average proportional difference between the *net* water consumption calculated with actual evapotranspiration within a 2-pixel buffer compared to a 1-pixel buffer is only 1.2%, shows that vegetation and thus actual evapotranspiration is not sensitive to distance.

4.3. Regional SDRs for Norway

Our five SDRs with an R^2 between 0.43 and 0.81 lie in the range of the R^2 between 0.35 and 0.90 reported by Tendall et al. (2014) for Europe and the R^2 between 0.47 and 0.61 reported by Xenopoulos and Lodge (2006) for the USA, and may indicate that the SDRs presented here are sufficiently good for use in LCA. Further, our results show that regional SDRs for fish can be calculated for rivers above this latitude, even if the fish diversity is lower due to the postglacial history.

To show the importance of regionally developed Species-discharge relationships we compare our SDRs with the global SDR from Hanafiah et al. (2011) and the Central Plains SDR from Tendall et al. (2014) in Supporting Information 1, Section S5. As our SDRs predict the lowest species richness, our results are in accordance with the statement from Hanafiah et al. (2011) that other existing SDR models should not be applied to rivers north of 42° latitude, due to the low species richness per unit of discharge in these river basins. This highlights that spatially-explicitly developed SDRs are an important requirement (Tendall et al., 2014) to assess the impact of water consumption on a global scale in LCA (Núñez et al., 2016).

To develop the regional SDRs, we identify five regions with similar glacial and dispersal history. In accordance with our assumption that the distance to the refugia is an important factor for recolonization, region 3, located in the southeast of Norway closest to the identified glacial refugia, has the highest species richness. Regions 2 and 5 located in the west of Norway and along the coast, have the lowest species richness, as these regions are further away from the refugia, and could predominantly be colonized by saltwater-tolerant species. However, region 7 located in northern Norway has a higher species richness than regions 2 and 5, and the same species richness as region 1 located in the most southern part of Norway. This is due to the topography in northern Norway, and indeed in northern Fennoscandia and Russia, which allowed for the postglacial immigration of a diverse fauna of freshwater fish from the east (Huitfeldt-Kaas, 1924).

4.4. Characterization factors

In this study we develop the first CFs quantifying the impact of *net* water consumption on freshwater fish species in Norway, contributing to spatially-explicit regional LCIA models of water consumption impacts on biodiversity. The unit of the CFs is PDF*y/m^3 and is in accordance with existing characterization factors assessing the impacts of water consumption on biodiversity (e.g. Tendall et al., 2014; Verones et al., 2016). In addition, we use the power function as a regression function to ensure comparability which existing characterization factors assessing the impacts of water consumption on biodiversity (e.g. Tendall et al., 2014; Verones et al., 2016). Therefore, this study provides new regional CFs. Novel to this study is that it develops the first method to calculate SDRs in previously glaciated regions. This further indicates that SDRs for northern Europe and northern America can be calculated and used in connection with newly developed CFs. This enables a more regionally specific Life Cycle Impact Assessment, which is needed to assess the biodiversity impact of water consumption on a global scale (Tendall et al., 2014; Núñez et al., 2018).

Hanafiah et al. (2011) report average CFs between $2.51*10^{-15}$ PDF*y/m^3 and $1*10^{-08}$ PDF*y/m^3 below 42° latitude north. Our CFs varying between $7.1*10^{-12}$ PDF*y/m^3 and $8.0*10^{-7}$ PDF*y/m^3 are therefore generally higher. This shows that the impact per fish species of 1 m^3 water consumption in Norway is comparatively higher than that below 42°N. However, as PDFs are calculated relative to the actual species richness in each catchment, only a few potential fish species lost in one catchment could lead to a high PDF value. As our SDRs report a lower fish species richness than the SDRs from Hanafiah et al. (2011), the absolute number of potentially disappeared fish species in Norway could be lower compared to Hanafiah et al. (2011). Our results highlight that spatially-explicit CFs above 42°N are needed to assess the impact of water consumption on a global scale in LCA (Núñez et al., 2016).

However, the SDRs represent a simplification of the relationship between water consumption and biodiversity loss, as frequency and timing of high and low flows, the rate of energy available in a river (Poff and Zimmerman, 2010; Mittelbach et al., 2001), temperature, (Xenopoulos and Lodge, 2006) trophic interactions or habitat diversity also influences fish species richness. Some migratory fish species, for example, require a minimum discharge to migrate (Quinn et al., 1997) and a discharge falling below a certain threshold will lead to a migration stop (Haro et al., 2004). An additional shortcoming of the SDR is that it assumes that fish species cannot rapidly adapt to an altered flow magnitude. Further, the SDR cannot account for external factors like habitat fragmentation (McKay et al., 2013). Our calculated CFs could therefore either lead to an over- or underestimation of the total fish species richness. However, despite all of these shortcomings it has been shown that exactly this simple relationship can be used to identify general patterns between flow and fish species richness (McGarvey, 2014). Therefore, we infer that the SDR can be applied for LCA purposes, as the goal of LCA is to compare general environmental impact patterns between similar products or processes at a global scale (Tendall et al., 2014; Hanafiah et al., 2011; Huijbregts et al., 2016).

The comparison at the global scale further requires CFs with global coverage (Jolliet et al., 2018). Due to its comparably low parameter demand, the SDR enables the development of regionally specific CFs for water consumption impacts on biodiversity at a global scale. However, if appropriate data would be available, the robustness of the SDRs could be greatly increased by including, e.g., species-specific habitat requirements and habitat-discharge interactions (Xenopoulos and Lodge, 2006).

Further, the developed CFs account only for freshwater biodiversity loss due to loss in magnitude of flow, as they are based on the mean annual discharge. As a result, they are not able to assess the effect of seasonality in magnitude change and the related impact on fish species. Our CFs with annual averages thus likely overestimate the impact, as water consumption during a specific season does not necessarily always lead to an impact for all fish species.

4.5. Uncertainty of characterization factors

We use the 95% confidence intervals of the obtained power function coefficients to quantitatively assess uncertainty. In addition, the obtained fish occurrence contributes to the uncertainty of the CFs. However, this uncertainty cannot be assessed quantitatively and therefore is only discussed in a qualitative way in the following section. The obtained fish occurrence data often reflects a strong spatial bias in survey efforts, because some sites are more likely to be surveyed than other sites (Phillips et al., 2009). Also, occurrence data are often collected without planned sampling schemes (Elith et al., 2006). In addition, the probability of detecting a species depends on features of the local habitat or the surrounding landscape (Gu and Swihart, 2004). As a result, the species richness estimation used for the SDR may represent an underestimation. Although not quantifiable, this underestimation is accounted for by weighing the power function by the total number of occurrence records in each catchment (Motulsky and Christopoulos, 2004).

We used all available occurrence points to develop the SDRs, as reservoir operation in Norway began as early 1800. As a result, the developed SDRs may underestimate the fish species richness because we cannot account for fish species that have gotten extinct before the earliest collection date of an occurrence point in the related catchment. The later reservoir operation started in a catchment, the more likely it is that we were able to obtain occurrences points from before reservoir operation. This leads to a lower probability of an underestimation of fish species richness by the SDR. Because the year of reservoir construction varies between catchments, the probability of an underestimation of fish species by the SDR is lowest in region 5 and 7 and highest in region 1 and 3 (SI1, S4).

5. Application in LCA

This study provides *net* water consumption values of Norwegian hydropower reservoirs in combination with CFs quantifying the impact of water consumption on freshwater fish species in Norway. When the *net* water consumption values are implemented in inventory databases and the CFs in Life Cycle Impact Assessment methods, the impact of water consumption of Norwegian hydropower plants on aquatic biodiversity can be assessed on a damage level. When performing an LCA of the whole-life cycle of a storage power plant, water consumption of dam construction and reinvestment phases also have to be considered (Bakken et al., 2016a). Water consumption values for these processes are available in LCI databases (e.g. Wernet et al., 2016). The fact that the CFs vary substantially between the catchments shows that is important to only apply the CF of the relevant catchment in an LCA study and not use average CFs from other catchments, since this may result in a substantial bias in the results. In addition, the CFs in this study should only be used to quantify the impact of a *decrease* in discharge, due to the uncertain influence of *increased* discharge on fish species richness (Xenopoulos and Lodge, 2006).

Finally, we would like to point out that water consumption is only one of several cause-effect pathways leading to potential biodiversity impacts related to hydropower production (Gracey and Verones, 2016), as dam construction for example can also lead to habitat fragmentation (McKay et al., 2013) or influence food web interactions (Power et al., 1996). A holistic LCA of storage power plants should assess all relevant biodiversity impacts from hydropower electricity production (Gracey and Verones, 2016), thus further model development for the reaming impact pathways is required.

6. Conclusions and future research

This study provides *net* water consumption values of Norwegian hydropower reservoirs in combination with the first developed CFs quantifying the impact of *net* water consumption on freshwater fish species in Norway. Thereby, this study contributes to providing methods and values to assess the biodiversity impact of water consumption. We calculate catchment-specific net water consumption for Norway using reservoir land inundation data in combination with evapotranspiration data. The average net water consumption across all investigated catchments, taking into account evaporation losses *prior* to the inundation of the reservoir, is 0.0016 m^3/kWh. This is 25% smaller than the existing value in the Ecoinvent database (0.002 m^3/kWh) (Flury and Frischknecht, 2012). Further, we develop 1790 catchment-specific characterization factors for Norway, quantifying the aquatic biodiversity impacts of water consumption based on Species-discharge relationships for fish, varying between $7.1*10^{-12}$ PDF*y/m^3 and $8.0*10^{-7}$ PDF*y/m^3. Novel to this CF is that it develops the first method to calculate SDRs in glaciated regions, by delineating regions with similar glacial and fish dispersal history. By using remote sensing assessed reservoir inundated land area (Pekel et al., 2016) and global hydropower reservoirs data (Lehner et al., 2011) in combination with the global MOD16 evaporation model, the methodology for Norway developed in this study has the potential to be applied globally. Further assessment of inundated land area from hydropower reservoirs is thereby most critically needed to allow for the estimation of *net* water consumption values of hydropower reservoirs on a global scale. This study shows that it is possible to calculate regional SDRs and related CFs for fish species in glaciated regions, and therefore additional SDRs for northern Europe and northern America should be calculated and used to develop new CFs. In addition, flow regime alterations have been linked to reduced invertebrate species richness as done by Tendall et al. (Poff and Zimmerman, 2010; Dewson et al., 2007), so developing macro-invertebrate SDRs could be justified in the future. Our CFs developed for Norway can be applied to hydropower projects that aim to include life cycle impacts of existing and planned hydropower reservoirs. Furthermore, a comparison with other energy carriers should be performed, to minimize the highlighted trade-offs between the mentioned SDGs (Nilsson et al., 2016; Bhaduri et al., 2016).

Funding

This work was funded by the Research Council of Norway through the SURE project (Project Number 244109).

Acknowledgment

We would like to thank John Woods, Xiangping Hu and Koen Kuipers for helpful comments and discussions. Further, we thank the two anonymous reviewers for their very helpful comments.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.eiar.2018.12.002.

References

Abell, R., Thieme, M.L., Revenga, C., Bryer, M., Kottelat, M., Bogutskaya, N., Coad, B., Mandrak, N., Balderas, S.C., Bussing, W., Stiassny, M.L.J., Skelton, P., Allen, G.R., Unmack, P., Naseka, A., Ng, R., Sindorf, N., Robertson, J., Armijo, E., Higgins, J.V., Heibel, T.J., Wikramanayake, E., Olson, D., López, H.L., Reis, R.E., Lundberg, J.G., Sabaj Pérez, M.H., Petry, P., 2008. Freshwater ecoregions of the world: a new map of biogeographic units for freshwater biodiversity conservation. Bioscience 58 (5), 403.

Alho, C.J., 2011. Environmental effects of hydropower reservoirs on wild mammals and freshwater turtles in Amazonia: a review. Oecol. Aust. 15 (3), 593–604.

Bakken, T.H., Killingtveit, Å., Engeland, K., Alfredsen, K., Harby, A., 2013. Water consumption from hydropower plants - review of published estimates and an assessment of the concept. Hydrol. Earth Syst. Sci. 17 (10), 3983–4000.

Bakken, T.H., Modahl, I.S., Engeland, K., Raadal, H.L., Arnøy, S., 2016a. The life-cycle water footprint of two hydropower projects in Norway. J. Clean. Prod. 113, 241–250.

Bakken, T.H., Modahl, I.S., Raadal, H.L., Bustos, A.A., Arnoy, S., 2016b. Allocation of water consumption in multipurpose reservoirs. Water Policy 18 (4), 932–947.

Barros, N., Cole, J.J., Tranvik, L.J., Prairie, Y.T., Bastviken, D., Huszar, V.L.M., del Giorgio, P., Roland, F., 2011. Carbon emission from hydroelectric reservoirs linked to reservoir age and latitude. Nat. Geosci. 4 (9), 593–596.

Bhaduri, A., Bogardi, J., Siddiqi, A., Voigt, H., Vörösmarty, C., Pahl-Wostl, C., Bunn, S.E., Shrivastava, P., Lawford, R., Foster, S., Kremer, H., Renaud, F.G., Bruns, A., Osuna, V.R., 2016. Achieving sustainable development goals from a water perspective. Front. Environ. Sci. 4.

Biemans, H., Haddeland, I., Kabat, P., Ludwig, F., Hutjes, R.W.A., Heinke, J., von Bloh, W., Gerten, D., 2011. Impact of reservoirs on river discharge and irrigation water supply during the 20th century. Water Resour. Res. 47 (3).

Clark, P.U., Dyke, A.S., Shakun, J.D., Carlson, A.E., Clark, J., Wohlfarth, B., Mitrovica, J.X., Hostetler, S.W., McCabe, A.M., 2009. The last glacial maximum. Science 325 (5941), 710–714.

Crook, D.A., Lowe, W.H., Allendorf, F.W., Eros, T., Finn, D.S., Gillanders, B.M., Hadweng, W.L., Harrod, C., Hermoso, V., Jennings, S., Kilada, R.W., Nagelkerken, I., Hansen, M.M., Page, T.J., Riginos, C., Fry, B., Hughes, J.M., 2015. Human effects on ecological connectivity in aquatic ecosystems: integrating scientific approaches to support management and mitigation. Sci. Total Environ. 534, 52–64.

Deemer, B.R., Harrison, J.A., Li, S., Beaulieu, J.J., DelSontro, T., Barros, N., Bezerra-Neto, J.F., Powers, S.M., dos Santos, M.A., Vonk, J.A., 2016. Greenhouse gas emissions from reservoir water surfaces: a new global synthesis. Bioscience 66 (11), 949–964.

Dewson, Z.S., James, A.B., Death, R.G., 2007. Stream ecosystem functioning under reduced flow conditions. Ecol. Appl. 17 (6), 1797–1808.

Dorber, M., May, R., Verones, F., 2018. Modeling net land occupation of hydropower reservoirs in Norway for use in life cycle assessment. Environ. Sci. Technol. 52 (4), 2375–2384.

Douglas, E.M., Jacobs, J.M., Sumner, D.M., Ray, R.L., 2009. A comparison of models for estimating potential evapotranspiration for Florida land cover types. J. Hydrol. 373 (3–4), 366–376.

Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Seyboth, K., Matschoss, P., Kadner, S., Zwickel, T., Eickemeier, P., Hansen, G., Schlömer, S., 2011. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation.

Egré, D., Milewski, J.C., 2002. The diversity of hydropower projects. Energy Policy 30 (14), 1225–1230.

Elith, Jane, Graham, Catherine H., Anderson, Robert P., Dudík, Miroslav, Ferrier, Simon, Guisan, Antoine, Hijmans, Robert J., Huettmann, Falk, Leathwick, John R., Lehmann, Anthony, Li, Jin, Lohmann, Lucia G., Loiselle, Bette A., Manion, Glenn, Moritz, Craig, Nakamura, Miguel, Nakazawa, Yoshinori, Overton, Jacob McC M., Townsend Peterson, A., Phillips, Steven J., Richardson, Karen, Scachetti-Pereira, Ricardo, Schapire, Robert E., Soberón, Jorge, Williams, Stephen, Wisz, Mary S., Zimmermann, Niklaus E., 2006. Novel methods improve prediction of species' distributions from occurrence data. Ecography 29 (2), 129–151.

Eloranta, A.P., Finstad, A.G., Helland, I.P., Ugedal, O., Power, M., 2018. Hydropower impacts on reservoir fish populations are modified by environmental variation. Sci. Total Environ. 618, 313–322.

ESRI (Environmental Systems Research Institute), 2014. ArcGIS release 10.3. Redlands, CA.

Finch, J.W., 2001. A comparison between measured and modelled open water evaporation from a reservoir in south-east England. Hydrol. Process. 15 (14), 2771–2778.

Finstad, A.G., Berg, O.K., Ulvan, E., Helland, I., Ugedal, O., 2017. Gillnet Test Fishing and Radiocaesium (Cs-137) of Brown Trout (*Salmo trutta*) and Arctic Char (Salvelinus aplinus) 2008 and 2009 from 20 Central Scandinavian Lakes: occurrences. Version 1.2. NTNU University Museum Occurrence Dataset. In.

Flury, K., Frischknecht, R., 2012. Life cycle inventories of hydroelectric power generation. In: ESU-Services, Fair Consulting in Sustainability, Commissioned by€ Oko-Institute eV, pp. 1–51.

Froese, R., Pualy, D., 2017. FishBase. www.fishbase.org.

García-Marín, J.-L., Utter, F.M., Pla, C., 1999. Postglacial colonization of brown trout in Europe based on distribution of allozyme variants. Heredity 82 (1), 46–56.

Gracey, E.O., Verones, F., 2016. Impacts from hydropower production on biodiversity in an LCA framework—review and recommendations. Int. J. Life Cycle Assess. 21 (3), 412–428.

Griffiths, D., 2006. Pattern and process in the ecological biogeography of European freshwater fish. J. Anim. Ecol. 75 (3), 734–751.

Gu, W., Swihart, R.K., 2004. Absent or undetected? Effects of non-detection of species occurrence on wildlife–habitat models. Biol. Conserv. 116 (2), 195–203.

Hanafiah, M.M., Xenopoulos, M.A., Pfister, S., Leuven, R.S.E.W., Huijbregts, M.A.J., 2011. Characterization factors for water consumption and greenhouse gas emissions based on freshwater fish species extinction. Environ. Sci. Technol. 45 (12), 5272–5278.

Hänfling, B., Hellemans, B., Volckaert, F., Carvalho, G., 2002. Late glacial history of the cold-adapted freshwater fish Cottus gobio, revealed by microsatellites. Mol. Ecol. 11 (9), 1717–1729.

Haro, A., Castro-Santos, T., Noreika, J., Odeh, M., 2004. Swimming performance of upstream migrant fishes in open-channel flow: a new approach to predicting passage through velocity barriers. Can. J. Fish. Aquat. Sci. 61 (9), 1590–1601.

Hertwich, E., de Larderel, J.A., Arvesen, A., Bayer, P., Bergesen, J., Bouman, E., Gibon, T., Heath, G., Peña, C., Purohit, P., 2016. Green Energy Choices: The Benefits, Risks, and Trade-Offs of Low-Carbon Technologies for Electricity Production. United Nation Environmental Program.
Hesthagen, T.F., Gravbrøt, Anders, 2017. Norwegian Freshwater Lake Fish Inventory. Norwegian Institute for Nature Research Sampling_event Dataset In.
Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., van Zelm, R., 2016. ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. Int. J. Life Cycle Assess. 22 (2), 138–147.
Huitfeldt-Kaas, H., 1924. Einwanderung und Verbreitung der Süsswasserfische in Norwegen. Arch. Hydrobiol. 4, 223–314.
International Energy Agency, 2017. Renewables 2017.
ISO, 2006a. ISO 14044: Environmental Management — Life Cycle Assessment — Principles and Framework. International Organisation for Standardization, Geneva, Switzerland.
ISO, 2006b. Environmental management: life cycle assessment: principles and framework. In: Technical Committee ISO/TC 207, Environmental Management. 5 Subcommittee SC Life cycle assessment.
ISO, 2014. ISO 14046- Environmental Management – Water Footprint – Principles, Requirements and Guidelines. International Organisation for Standardization.
Jansson, R., Nilsson, C., Renofalt, B., 2000. Fragmentation of riparian floras in rivers with multiple dams. Ecology 81 (4), 899.
Jolliet, O., Antón, A., Boulay, A.-M., Cherubini, F., Fantke, P., Levasseur, A., McKone, T.E., Michelsen, O., i Canals, L.M., Motoshita, M., Pfister, S., Verones, F., Vigon, B., Frischknecht, R., 2018. Global guidance on environmental life cycle impact assessment indicators: impacts of climate change, fine particulate matter formation, water consumption and land use. The International Journal of Life Cycle Assessment 1–19.
Kitzes, J., Shirley, R., 2016. Estimating biodiversity impacts without field surveys: a case study in northern Borneo. Ambio 45 (1), 110–119.
Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., i Canals, L.M., Saad, R., de Souza, D.M., Müller-Wenk, R., 2013. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. Int. J. Life Cycle Assess. 18 (6), 1188–1202.
Kumar, A., Schei, T., Ahenkorah, A., Rodriguez, R.C., Devernay, J.-M., Freitas, M., Hall, D., Killingtveit, Å., Liu, Z., 2011. Hydropower. In: Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Seyboth, K., Matschoss, P., Kadner, S., Zwickel, T., Eickemeier, P., Hansen, G., Schlömer, S., von Stechow, C. (Eds.), IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
Le Blanc, D., 2015. Towards integration at last? The sustainable development goals as a network of targets. Sustain. Dev. 23 (3), 176–187.
Lee, T.M., Sacks, L.A., Swancar, A., 2014. Exploring the long-term balance between net precipitation and net groundwater exchange in Florida seepage lakes. J. Hydrol. 519, 3054–3068.
Lehner, B., Liermann, C.R., Revenga, C., Vörösmarty, C., Fekete, B., Crouzet, P., Döll, P., Endejan, M., Frenken, K., Magome, J., Nilsson, C., Robertson, J.C., Rödel, R., Sindorf, N., Wisser, D., 2011. High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. Front. Ecol. Environ. 9 (9), 494–502.
Leprieur, F., Olden, J.D., Lek, S., Brosse, S., 2009. Contrasting patterns and mechanisms of spatial turnover for native and exotic freshwater fish in Europe. J. Biogeogr. 36 (10), 1899–1912.
Lu, J., Sun, G., McNulty, S.G., Amatya, D.M., 2005. A comparison of six potential evapotranspiration methods for regional use in the southeastern United States. J. Am. Water Resour. Assoc. 41 (3), 621–633.
Manzano-Agugliaro, F., Alcayde, A., Montoya, F.G., Zapata-Sierra, A., Gil, C., 2013. Scientific production of renewable energies worldwide: an overview. Renew. Sust. Energ. Rev. 18, 134–143.
The MathWorks, Inc., 2015, MATLAB release R2015a. Natick, Ma.
McAllister, D.E., Craig, J.F., Davidson, N., Delany, S., Seddon, M., 2001. Biodiversity Impacts of Large Dams.
McGarvey, D.J., 2014. Moving beyond species-discharge relationships to a flow-mediated, macroecological theory of fish species richness. Freshw. Sci. 33 (1), 18–31.
McKay, S.K., Schramski, J.R., Conyngham, J.N., Fischenich, J.C., 2013. Assessing upstream fish passage connectivity with network analysis. Ecol. Appl. 23 (6), 1396–1409.
Mekonnen, M.M., Hoekstra, A.Y., 2012. The blue water footprint of electricity from hydropower. Hydrol. Earth Syst. Sci. 16 (1), 179–187.
Mengistu, M.G., Savage, M.J., 2010. Open water evaporation estimation for a small shallow reservoir in winter using surface renewal. J. Hydrol. 380 (1–2), 27–35.
Milà, i, Canals, L., Chenoweth, J., Chapagain, A., Orr, S., Antón, A., Clift, R., 2008. Assessing freshwater use impacts in LCA: part I—inventory modelling and characterisation factors for the main impact pathways. Int. J. Life Cycle Assess. 14 (1), 28–42.
Mittelbach, G.G., Steiner, C.F., Scheiner, S.M., Gross, K.L., Reynolds, H.L., Waide, R.B., Willig, M.R., Dodson, S.I., Gough, L., 2001. What is the observed relationship between species richness and productivity? Ecology 82 (9), 2381–2396.
Mjelde, M., Hellsten, S., Ecke, F., 2012. A water level drawdown index for aquatic macrophytes in Nordic lakes. Hydrobiologia 704 (1), 141–151.
Motulsky, H., Christopoulos, A., 2004. Fitting Models to Biological Data Using Linear and Nonlinear Regression: A Practical Guide to Curve Fitting. Oxford University Press.
Mu, Q., Heinsch, F.A., Zhao, M., Running, S.W., 2007. Development of a global evapotranspiration algorithm based on MODIS and global meteorology data. Remote Sens. Environ. 111 (4), 519–536.
Mu, Q., Zhao, M., Running, S.W., 2011. Improvements to a MODIS global terrestrial evapotranspiration algorithm. Remote Sens. Environ. 115 (8), 1781–1800.
Mutel, C.L., Hellweg, S., 2009. Regionalized life cycle assessment: computational methodology and application to inventory databases. Environ. Sci. Technol. 43 (15), 5797–5803.
Mutel, C.L., Pfister, S., Hellweg, S., 2012. GIS-based regionalized life cycle assessment: how big is small enough? Methodology and case study of electricity generation. Environ. Sci. Technol. 46 (2), 1096–1103.
Natural History Museum, University of Oslo, 2017. Fish Collection. Natural History Museum, University of Oslo . Version 1.90. Occurrence Dataset. In.
Nilsson, M., Griggs, D., Visbeck, M., 2016. Policy: map the interactions between sustainable development goals. Nature 534 (7607), 320–322.
Norwegian Government Ministries and Offices, Meld. St. 25, 2015–2016. Power to Change - Energy policy 2030. (In Norwegian).
Norwegian Water Resources and Energy Directorate, 2017. Diagram magasinfyllingen - hele Norge. http://vannmagasinfylling.nve.no/Default.aspx?ViewType=Chart&Tidsenhet=Aar&Omr=NO (1.11.2017).
Núñez, M., Bouchard, C.R., Bulle, C., Boulay, A.-M., Margni, M., 2016. Critical analysis of life cycle impact assessment methods addressing consequences of freshwater use on ecosystems and recommendations for future method development. Int. J. Life Cycle Assess. 21 (12), 1799–1815.
Núñez, M., Rosenbaum, R.K., Karimpour, S., Boulay, A.-M., Lathuillière, M.J., Margni, M., Scherer, L., Verones, F., Pfister, S., 2018. A multimedia hydrological fate modeling framework to assess water consumption impacts in life cycle assessment. Environ. Sci. Technol. 52 (8), 4658–4667.
Norwegian Water Resources and Energy Directorate, 2016. NVE Atlas. http://atlas.nve.no (15.03.2016).
Oberdoff, T., Guégan, J.F., Hugueny, B., 1995. Global scale patterns of fish species richness in rivers. Ecography 18 (4), 345–352.
Oberdorff, T., Hugueny, B., Guegan, J.-F., 1997. Is there an influence of historical events on contemporary fish species richness in rivers? Comparisons between Western Europe and North America. J. Biogeogr. 24 (4), 461–467.
Patton, H., Hubbard, A., Andreassen, K., Auriac, A., Whitehouse, P.L., Stroeven, A.P., Shackleton, C., Winsborrow, M., Heyman, J., Hall, A.M., 2017. Deglaciation of the Eurasian ice sheet complex. Quat. Sci. Rev. 169, 148–172.
Pekel, J.-F., Cottam, A., Gorelick, N., Belward, A.S., 2016. High-resolution mapping of global surface water and its long-term changes. Nature 540, 418.
Pennington, D.W., Potting, J., Finnveden, G., Lindeijer, E., Jolliet, O., Rydberg, T., Rebitzer, G., 2004. Life cycle assessment part 2: current impact assessment practice. Environ. Int. 30 (5), 721–739.
Pfister, S., Saner, D., Koehler, A., 2011. The environmental relevance of freshwater consumption in global power production. Int. J. Life Cycle Assess. 16 (6), 580–591.
Phillips, S.J., Dudík, M., Elith, J., Graham, C.H., Lehmann, A., Leathwick, J., Ferrier, S., 2009. Sample selection bias and presence-only distribution models: implications for background and pseudo-absence data. Ecol. Appl. 19 (1), 181–197.
Poff, N.L., Zimmerman, J.K.H., 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshw. Biol. 55 (1), 194–205.
Poff, N.L., Angermeier, P.L., Cooper, S.D., Lake, P., Fausch, K.D., Winemiller, K.O., Mertes, L.A., Oswood, M.W., Reynolds, J., Rahel, F.J., 2001. Fish diversity in streams and rivers. In: Global Biodiversity in a Changing Environment. Springer, pp. 315–349.
Power, M.E., Dietrich, W.E., Finlay, J.C., 1996. Dams and downstream aquatic biodiversity: potential food web consequences of hydrologic and geomorphic change. Environ. Manag. 20 (6), 887–895.
Quinn, T.P., Hodgson, S., Peven, C., 1997. Temperature, flow, and the migration of adult sockeye salmon (Oncorhynchus nerka) in the Columbia River. Can. J. Fish. Aquat. Sci. 54 (6), 1349–1360.
Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.-P., Suh, S., Weidema, B.P., Pennington, D.W., 2004. Life cycle assessment: part 1: framework, goal and scope definition, inventory analysis, and applications. Environ. Int. 30 (5), 701–720.
Refseth, U., Nesbø, C., Stacy, J., Vøllestad, L., Fjeld, E., 1998. Genetic evidence for different migration routes of freshwater fish into Norway revealed by analysis of current perch (Perca fluviatilis) populations in Scandinavia. Mol. Ecol. 7 (8), 1015–1027.
Reyjol, Y., Hugueny, B., Pont, D., Bianco, P.G., Beier, U., Caiola, N., Casals, F., Cowx, I., Economou, A., Ferreira, T., Haidvogl, G., Noble, R., de Sostoa, A., Vigneron, T., Virbickas, T., 2007. Patterns in species richness and endemism of European freshwater fish. Glob. Ecol. Biogeogr. 16 (1), 65–75.
Richter, B., Baumgartner, J., Wigington, R., Braun, D., 1997. How much water does a river need? Freshw. Biol. 37 (1), 231–249.
Rosenberry, D., Sturrock, A., Winter, T., 1993. Evaluation of the energy budget method of determining evaporation at Williams Lake, Minnesota, using alternative instrumentation and study approaches. Water Resour. Res. 29 (8), 2473–2483.
Scherer, L., Pfister, S., 2016a. Global water footprint assessment of hydropower. Renew. Energy 99, 711–720.
Scherer, L., Pfister, S., 2016b. Hydropower's biogenic carbon footprint. PLoS One 11 (9), e0161947.
Schiemer, F., 2000. Fish as indicators for the assessment of the ecological integrity of large rivers. Hydrobiologia 422, 271–278.
Secretariat of the Convention on Biological Diversity, 2014. Global Biodiversity Outlook 4. Montreal.
Spalding, M.D., Fox, H.E., Allen, G.R., Davidson, N., FerdaÑA, Z.A., Finlayson, M.A.X., Halpern, B.S., Jorge, M.A., Lombana, A.L., Lourie, S.A., Martin, K.D., McManus, E., Molnar, J., Recchia, C.A., Robertson, J., 2007. Marine ecoregions of the world: a bioregionalization of coastal and shelf areas. Bioscience 57 (7), 573.
Strachan, I.B., Tremblay, A., Pelletier, L., Tardif, S., Turpin, C., Nugent, K.A., 2016. Does the creation of a boreal hydroelectric reservoir result in a net change in evaporation?

J. Hydrol. 540, 886–899.
Tefera, B., Sterk, G., 2008. Hydropower-induced land use change in Fincha'a watershed, Western Ethiopia: analysis and impacts. Mt. Res. Dev. 28 (1), 72–80.
Tendall, D.M., Hellweg, S., Pfister, S., Huijbregts, M.A.J., Gaillard, G., 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. 48 (6), 3236–3244.
The Norwegian Biodiversity Information Centre and GBIF Norway, 2017. Species map service 1.6. http://artskart.artsdatabanken.no/Default.aspx (3.04.2017).
Thematic Mapping API World Borders Dataset, 2009. http://thematicmapping.org/downloads/world_borders.php (11.04.2016).
United Nations, 2015. Transforming Our World: The 2030 Agenda for Sustainable Development - A/RES/70/1.
University of Montana, 2011. MODIS Global Evapotranspiration Project (MOD16). http://www.ntsg.umt.edu/project/modis/mod16.php (01.11.2017).
Vang, R., 2017a. National Fish Tag Database. Version 1.5. Norwegian Institute for Nature Research. Occurrence Dataset. In.
Vang, R., 2017b. Ims Fish Tag Database. Version 1.2. Norwegian Institute for Nature Research. Occurrence Dataset. In.
Verones, F., Pfister, S., van Zelm, R., Hellweg, S., 2016. Biodiversity impacts from water consumption on a global scale for use in life cycle assessment. Int. J. Life Cycle Assess. 22 (8), 1247–1256.
Verones, F., Bare, J., Bulle, C., Frischknecht, R., Hauschild, M., Hellweg, S., Henderson, A., Jolliet, O., Laurent, A., Liao, X., 2017. LCIA framework and cross-cutting issues guidance within the UNEP-SETAC Life Cycle Initiative. J. Clean. Prod. 161, 957–967.
Vörösmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S.E., Sullivan, C.A., Liermann, C.R., Davies, P.M., 2010. Global threats to human water security and river biodiversity. Nature 467 (7315), 555–561.
Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21 (9), 1218–1230.
Westerhoff, R.S., 2015. Using uncertainty of Penman and Penman–Monteith methods in combined satellite and ground-based evapotranspiration estimates. Remote Sens. Environ. 169, 102–112.
Winter, L., Lehmann, A., Finogenova, N., Finkbeiner, M., 2017. Including biodiversity in life cycle assessment – state of the art, gaps and research needs. Environ. Impact Assess. Rev. 67, 88–100.
Xenopoulos, M.A., Lodge, D.M., 2006. Going with the flow: using species-discharge relationships to forecast losses in fish biodiversity. Ecology 87 (8), 1907–1914.
Xenopoulos, M.A., Lodge, D.M., Alcamo, J., Märker, M., Schulze, K., Van Vuuren, D.P., 2005. Scenarios of freshwater fish extinctions from climate change and water withdrawal. Glob. Chang. Biol. 11 (10), 1557–1564.
Zhang, L., Dawes, W., Walker, G., 2001. Response of mean annual evapotranspiration to vegetation changes at catchment scale. Water Resour. Res. 37 (3), 701–708.
Zhang, J., Zhengjun, L., Xiaoxia, S., 2009. Changing landscape in the Three Gorges Reservoir Area of Yangtze River from 1977 to 2005: land use/land cover, vegetation cover changes estimated using multi-source satellite data. Int. J. Appl. Earth Obs. Geoinf. 11 (6), 403–412.

Chapter 4: Global characterization factors for biodiversity impacts of land inundation in Life Cycle Assessment.

In review: *Nature - Scientific Data.*

Global characterization factors for biodiversity impacts of land inundation in Life Cycle Assessment

Martin Dorber, †, Koen Kuipers†, Francesca Verones†*

†Department of Energy and Process Engineering, NTNU, Sem Sælands vei 7, 7491 Trondheim, Norway

This article is awaiting publication and is not included in NTNU Open

Chapter 5: The potential to control biodiversity impacts of future global hydropower reservoirs by strategic site selection

Intended for publication in *Nature Communications.*

This article is awaiting publication and is not included in NTNU Open

Chapter 6: Conclusion and Outlook

6.1 Embedding of the thesis into the existing Life Cycle Assessment context

Until now, Life Cycle Assessment (LCA) studies related to hydropower electricity production have only accounted for environmental impacts in the form of greenhouse gas (GHG) emissions,[1-3] due to a lack of operational Life Cycle Inventory (LCI) values and Life Cycle Impact Assessment (LCIA) models for additional impact pathways (Section 1.4).[4] This thesis contributes the first approaches for quantifying potential biodiversity impacts of hydropower electricity production within LCA. The assessment of potential biodiversity impacts of hydropower electricity production was achieved by advancing, developing and applying both Life Cycle Inventory (LCI) and Life Cycle Impact Assessment (LCIA) models for the cause-effect pathways land use and land use change, climate change and freshwater habitat alteration (Figure 1).

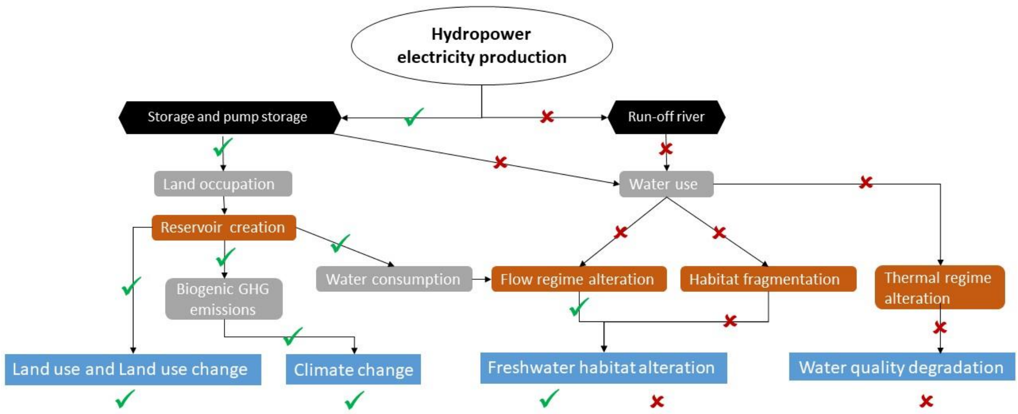


Figure 1: Overview showing which of the cause-effect pathways from hydropower electricity production on biodiversity (identified in Section 1.2) are covered in this thesis. ✓= covered, ✗ = not covered. Black = type of hydropower plant; Grey = environmental stressor; Orange = environmental alteration; Blue = impact category.

6.1.1 Advances regarding the Life Cycle Inventory

This thesis provides the first net land occupation LCI parameters for existing Norwegian hydropower reservoirs (Chapter 2). The developed underlying modelling framework uses open source Landsat satellite images[5,6] as the main input data. Since these images are available for the entire globe, the developed approach has the potential for application to all reservoirs globally where annual electricity production is also known. In addition, the developed model can also be used for prospective assessments, for example, to predict the net land occupation for proposed future hydropower reservoirs on a global scale (Chapter 5).

The net land occupation values from Chapter 2, enabled a calculation of net water consumption values for Norwegian hydropower reservoirs (Chapter 3). To quantify this water consumption, I used an evaporation model with global coverage,[7,8] which again gives the potential for global application if land occupation and annual electricity production data is available. Like for land occupation, this model can also be used for prospective assessments, for example to model potential future net water consumption of prospective reservoirs on a global scale (Chapter 5).

The calculated Norwegian net land occupation and net water consumption values for existing reservoirs have a high variation between the hydropower reservoirs. In both cases the average values are smaller than the existing values in the Ecoinvent database,[9] which are gross values and only representative for Switzerland and Brazil (see Section 1.4). Similarly, the inventory values for prospective reservoirs also show a high variation, showing that environmental parameters, such as topographic and climatic conditions[9] and water area before dam construction, are highly variable. In order to take this variation into account, spatially-explicit net LCI inventory parameters are urgently required,[9,10] which is what I am contributing to the research field.

To assess impacts of Norwegian hydropower electricity production in LCA prior to the developments presented in this thesis, practitioners necessarily could only use existing inventory values (which are not Norway specific). This would very likely have resulted in an overestimation of the biodiversity impact of Norwegian hydropower reservoirs. This highlights that LCI modellers should "*prioritize the development of regionalized inventories when high spatial variability is observed or expected*".[11] This thesis shows the value of incorporating remote sensing data to achieve regionalized inventories.

6.1.2 Novel characterization factor development

In this thesis, the first global CFs that quantify the potential future biodiversity impact of inundating terrestrial habitat area were developed (Chapter 4 and Figure 1). To follow current recommendations from the Life Cycle Initiative hosted by UN Environment and to enhance comparability,[12] the CFs are based on an adaptation of the methodology developed by Chaudhary et al.,[13] as this is the currently recommended LCIA method for quantifying land use and land use change impacts on biodiversity.[14] As the developed CF introduces the land cover class “water”, the developed CF further aligns with the Life Cycle Initiative’s recommendation to expand the number of LCIA land cover classes.[14]

The developed CFs were applied to calculate the terrestrial biodiversity impact of future land occupation from potential hydropower reservoirs (Chapter 5). Thereby, I followed the Life Cycle Initiative recommendation to show the usefulness and practicality of newly developed CFs.[11]

Even though the CFs were developed for hydropower electricity production, they can potentially be applied to any process which is changing terrestrial habitat into aquatic habitat, e.g. sea level rise,[15] as discussed in Chapter 4. A good example for the importance of assessing biodiversity impacts from sea level rise is the recent extinction of the Bramble Cay melomys (*Melomys rubicola*).[41] This small rodent used to live on a small island in the eastern Torres Strait of the Great Barrier Reef. However, the highest point on the island is only 3 meters above sea level and ocean inundation has led to a dramatic habitat loss. Therefore, the extinction of the rodent has just been attributed to ocean inundation of the island, because of climate change.[16,17]

Hence, the developed CFs are advancing the existing LCIA impact category “land stress” beyond hydropower electricity production. With this addition, LCIA is now able to assess the biodiversity impact of land use change from one terrestrial habitat to another terrestrial habitat type,[13,18,19] from aquatic to terrestrial habitat[20] and from terrestrial to aquatic habitat (Chapter 4).

However, since the developed CFs are for future impacts, they cannot be used to calculate the biodiversity impact of already existing hydropower reservoir’s land occupation. Therefore, it was possible to quantify the biodiversity impact of the existing net land occupation from Norwegian hydropower reservoirs in Chapter 2. To quantify the biodiversity impact of present land occupation, additional CFs accounting for historical changes from terrestrial habitat to aquatic habitat are required (see also Section 6.5).

For the LCIA impact category "water stress", no CFs previously existed that could quantify the aquatic biodiversity impact of water consumption in a recently (in geological time) glaciated[21] region like Norway.[22,23] Therefore, in this thesis the first spatially-explicit CFs quantifying biodiversity impacts of water consumption in a post-glaciated region were developed (Chapter 3). The novelty behind these CFs is that they include several Species-discharge relationships (SDR), which account for local variation in fish fauna by delineating regions with the same postglacial freshwater fish immigration history (Chapter 3). The limitations of the SDR concept are discussed in Section 6.2. The comparison of the SDRs developed for Norway with the existing SDRs (Supplementary information, Section 7.2) confirms that the usage of existing SDRs that are used for LCIA purposes and suited to non-glaciated regions would have led to an overestimation of the impact in Norway, due to the lower fish species richness in this recently glaciated region. However, if assessing impacts of Norwegian water consumption in LCA, before the assessments in this thesis, practitioners would have been forced to use exactly these exisiting CFs or would not have been able to quantify the impact because of "no data" values. This highlights the importance of spatially explicit CF development.

Other water consumption processes, such as water consumption for irrigation, industrial production, and residential needs, each can have, in principle, the same impact mechanism on freshwater biodiversity like evaporation from reservoirs. Therefore, the developed water consumption CFs are not limited to the quantification of biodiversity impacts from hydropower production and are applicable to all processes requiring consumptive use of surface water in Norway, given that appropriate LCI values are available.

The developed model shows that it is possible to develop CFs in a post-glaciated region and could therefore enhance development of water consumption CFs for biodiversity impacts in other post-glaciated countries like Canada, Sweden or Iceland.

Both sets of CFs developed in this thesis contribute towards a spatially-explicit assessment of biodiversity impacts on a global scale, improving the operationality of the LCIA impact categories "water stress" and "land stress".

The developed CFs harmonize with existing LCIA models and go along with the following current recommendations from the Life Cycle Initiative hosted by UN Environment:[12]

1) All the developed CFs assess ecosystem quality damages in the recommended unit of Potentially Disappeared Fraction of Species (PDF);[41]
2) For land occupation this work follows the recommendation to develop CFs for both regional and global biodiversity loss.[14] For the water consumption CFs only regional aquatic biodiversity loss could be quantified, because at the time Chapter 3 was developed, no conversion factors to convert regional aquatic PDFs into global aquatic PDFs existed. However, the recent development by Kuipers et al.[24] now enables the estimation of potential global aquatic biodiversity loss from future potential hydropower electricity production.

In summary, this thesis contributes models to the research community that now allow the assessment of damages on ecosystem quality from hydropower electricity production (and beyond) within LCA, especially regarding the impact categories "water stress" and "land stress".

6.2 Limitations and uncertainties

When using LCA results for decision making, the limitations and uncertainties of the results have to be considered. Therefore, the Life Cycle Initiative points out that "*reporting of uncertainties should become a routine practice to avoid over-interpretation and biased decisions*".[12] To follow this recommendation, Chapters 2, 3 and 4 included a quantitative and qualitative assessment of uncertainty. Non-quantifiable uncertainty was discussed in a qualitative way, in line with the recommendation from the Life Cycle Initiative.

The uncertainty of the in this thesis developed models mainly results from parameter uncertainty and model uncertainty. Parameter uncertainty is caused by inaccurate, incomplete, or unrepresentative input data.[25] Model uncertainty is the result of an incomplete understanding and over-simplification of a mechanism.

6.2.1 Uncertainty in Life Cycle Inventory models

The main parameter uncertainty, which affects all in thesis calculated net land occupation and net water consumption parameters, comes from estimates of water surface area before dam construction, which is input data to both models. Uncertainty arises as the applied images classification method in Chapter 2 tends to underestimate the water surface area due to so called mixed pixels, which contain both water and non-water land cover types. By using the surface area of natural lakes as reference, the developed model corrected to a certain extent for this potential bias. However, the potential for an underestimation remains, because the surface area of natural lakes is not constant. This underestimation can lead to an overestimation of the land occupation, followed by an overestimation of the water consumption and finally also an overestimation of the related terrestrial and aquatic biodiversity impact. As the number of mixed pixels decreases with higher image resolution,[26] using satellite images with higher image resolution presents the best way to reduce mixed pixels and therewith the underestimation of the water surface area. For Norway, high resolution satellite images could not be used (as explained in Chapter 2), but the remote sensing data used in Chapter 5 has a higher resolution and therefore a smaller uncertainty arising from mixed pixels. Future remote sensing data with higher resolution provides the opportunity to further lower this type of parameter uncertainty.

At the same time, the net land occupation and net water consumption values can be uncertain because of unrepresentative actual reservoir surface area data (Chapters 2 and 5). Despite the fact that reservoirs are generally filled less than 90% of their maximal capacity,[27] the calculation of the net land occupation and net water consumption values is based on the assumption, that the hydropower reservoirs are always completely full. This assumption was necessary due to a lack of water regulation to surface area curves. Hence, it its most likely that we overestimate the net land occupation and water consumption values of hydropower reservoirs. This can again lead to an overestimation of the related terrestrial and aquatic biodiversity impact. To reduce this uncertainty, reservoir water regulation data and models describing the relationship between water-level regulation and water surface area would be needed (Section 6.4).

Model uncertainty for the LCI parameters comes mainly from the electricity allocation. As one hydropower reservoir can be connected to several power plants, and one power plant can be connected to several reservoirs, we had to estimate the amount of electricity produced by a specific reservoir. This was done by using the reservoir surface area as an allocation parameter (Chapter 2 and 3). As reservoir areas might not be representative for the potential storage of water, using reservoir volume may have been the better choice, but related data was lacking. An underestimation of the electricity production would lead to an overestimation of the LCI parameter and the total impact. This uncertainty could be reduced by the provision of more and more detailed hydropower electricity production data by power producers.

6.2.2 Uncertainty in Life Cycle Impact Assessment models

Species richness data affects the parameter uncertainty of all CFs developed in this thesis. Especially for freshwater fish, occurrence points were unevenly distributed over Norway and between species. In catchments with a low number of occurrence points, the fish species richness estimates most likely represent an underestimation of actual species richness. Depending on in which river systems these underestimations occur, this can flatten or increase the slope of the calculated SDR and lead to either an underestimation or overestimation of the biodiversity impact. This uncertainty can only be decreased by increasing the number of species occurrence points. Furthermore, these occurrence points need to be evenly distributed. The Strategic Plan 2017-2021 from the Global Biodiversity Information Facility (GBIF),[28] which, inter alia, aims to improve the coverage, completeness and resolution of species occurrences points, gives confidence that this data situation will be improved.

Model uncertainties of the calculated CFs are coming from the used Species-area and Species-discharge relationship concepts. For example, the Species-discharge relationship is a simplification of the relationship between river discharge and fish species richness. Factors such as the frequency and timing of high and low flow events, the rate of energy available in a river,[29,30] temperature,[31] and trophic interactions[31] can also influence species richness. The model uncertainty of using these relationships can only be assessed indirectly with, for example, a sensitivity analysis. These relationships, however, have comparably low parameter demand and appropriate input data is globally available, which makes them useful modelling concepts for LCA approaches. These relationships, with comparably low parameter demand, enable the development of regionally specific characterization factors for impacts of biodiversity at a global scale.[22,23,32] To reduce model uncertainty of water consumption CFs, Tendall et al.[22], developed a SDR for specific river sections and accounted for species variability inside one catchment. However, this approach requires a sufficiently large amount of species occurrence points and discharge data for each river section, which was not available for our Norwegian case. The GBIF Strategic Plan 2017-202,[28] however, may allow the application of river section specific SDRs in the future.[22] This further indicates that model uncertainty could be reduced by incorporating further data and knowledge from the field of ecology in CFs (e.g. refs[33,34]).

6.2.3 Uncertainty regarding Life Cycle Assessment applications

In theory, LCA combines regionalized inventory parameters with spatially explicit CFs to obtain the result. In practice, CFs and inventory parameters may not match in spatial coverage, which creates "no data" areas. Such a case occurred in Chapter 5. While net water consumption for hydropower reservoirs could be calculated on the global scale, the water consumption CFs did not have a global coverage. This for example, prevented a direct assessment of water consumption biodiversity impact from hydropower reservoirs in Canada. To handle the "no data" areas, there are four choices: (1) treat the areas as zeros, (2) assign a default value (3) interpolate from neighbouring areas, (4) or exclude the areas from the assessment.[11] Each of these choices has a different uncertainty. In Chapter 5, we decided to interpolate from neighbouring areas. With option (4), reservoirs would have been attributed a "no data" value instead of an interpolated value, to avoid unacceptable uncertainty, which may occur in options 1-3. Option 4, however, automatically implies that the biodiversity impacts of these reservoirs cannot be considered in decision making. By using the interpolated values with parameter uncertainty, the biodiversity impact of hydropower reservoirs in Canada could be quantified, although with uncertainty. One way to minimize the uncertainty of options 1-3 is the development of LCI and LCIA models with global coverage. This highlights once more, that LCA requires models which are able to assess environmental mechanisms on a global scale. As a result, the availability of global data often defines how well an environmental mechanism can be covered in LCA, even though the knowledge about the environmental mechanism may be more elaborate. Adequate LCA models with global coverage must therefore find a balance between parameter uncertainty and model uncertainty.

6.2.4 Model limitations

The models in this thesis assume that the background condition stay the same in the future. However, climate change may change the availability of water for each catchment. Increased precipitation could for example offset water consumption rates, while increased temperatures could lead to increased evaporation. Climate change could further change the electricity demand (e.g. more air conditioning). In addition, hydropower power plants may be upgraded or receive more efficient turbines, which in turn would lead to an increase annual electricity production and fewer impacts per kWh. If these changes occur, the result of this thesis should be recalculated with updated input parameters.

Due to a lack of net emission methods,[35-37] biogenic CO_2 and N_2O emissions could not be calculated. This model uncertainty leads to an underestimation of the total GHG emissions from hydropower reservoirs, and subsequently an underestimation of the total biodiversity impact. However, as Hertwich et al.[38] point out, CH_4 emissions are most relevant from a climate perspective and N_2O emissions of hydropower reservoirs play only a minor role in the total biogenic GHG emissions budget.

6.2.5 Life Cycle Impact Assessment limitations

Besides the described model limitations, limitations can also be a result of the LCIA framework itself. LCIA assesses biodiversity loss independently for each impact category. Therefore, it is possible to sum the terrestrial biodiversity impact of land occupation with the terrestrial biodiversity impact of climate change (Chapter 5). However, species cannot die twice (because of land occupation and climate change).

Furthermore, there may be interactions between these two environmental processes, which are currently not taken into account in LCIA. Both land occupation and climate change can cause habitat loss. These habitat loss mechanisms can interact, for example in a synergistic way.

In addition, this means, that impacts from consumptive and non-consumptive water use need to be assessed individually, even though they are happening in parallel and both can cause a freshwater habitat alteration. On a seasonal scale, non-consumptive water use from hydropower electricity production could therefore offset the water consumption or even lead to an increased water availability. On one hand this highlights that characterization factors assessing the impacts of non-consumptive water use on freshwater biodiversity are needed. On the other hand, once CFs for both water consumption and other water uses exist, it should be ensured that the CFs do not double count a potential biodiversity loss. Indeed, a potential double counting of impacts between CFs affects potentially all impact categories in LCIA.

Furthermore, LCA does not account for the mitigation of negative biodiversity effects, even though it has been shown that some negative effects from hydropower electricity production on for example salmons (*Salmo salar*) can be mitigated.[39] LCIA models accounting for the influence of mitigation strategies may avoid an potential overestimation of the biodiversity impact.

Despite all the mentioned limitations, one should not forget the purpose of an LCA. The results of an LCA should allow a relative comparison of products and processes[40] to improve the environmental performance of products and processes.[41] LCA does not allow an absolute comparison[14] and is not meant to replace a local impact assessment, which is necessary to quantify the absolute biodiversity impact of hydropower reservoirs. Therefore, as long as the uncertainty and limitations of the applied models to each hydropower reservoir are highlighted, they should not restrict interpreting results comparatively. However, when a cross comparison of hydropower to other renewable electricity sources, like wind power, is performed, the limitations and uncertainty of the used models should be in the same range. If this is not the case, it is not clear if the different results arise from different model uncertainty or from varying environmental impacts.

6.3 Practical relevance

We know that an increased renewable energy production for fulfilment of SDG 7 (Affordable and clean energy) can lead to both positive synergies and negative trade-offs.[42] For hydropower electricity production, studies have so far mainly focused on the quantification of the positive synergies only, in relation to SDG 13 (Climate change).[1-3] However, negative trade-offs from biodiversity impacts, that could interfere with SDG 6 (Clean water and sanitation) and SDG 15 (Life on land), have not been assessed in a qualitative way. By advancing and developing LCA related models, the results of this thesis showed that hydropower electricity production can indeed have biodiversity impacts and that spatially-explicit LCAs can be used to quantitatively assess these potential negative trade-offs.

At the same time, Chapter 5 highlights, that stakeholders have choices to reduce the biodiversity impact of future hydropower electricity production, highlighting that LCA is a suitable tool to assess nexus relationships between the SDGs.[43] In particular, LCA can help to identify optimal hydropower locations that balance the positive synergies (e.g. mitigate climate change) and negative trade-offs (e.g. biodiversity impacts), needed for a transition into a more sustainable world (Chapter 1, Figure 2).

Therefore, my results are relevant for all practitioners using LCA to assess the environmental impacts of a product that uses hydropower electricity during its life cycle and for practitioners analysing electricity production processes.

In addition, they are potentially relevant for any stakeholder involved in environmental decision making inside the hydropower sector, such as energy companies, environmental organizations or policy makers/governments. For energy companies, who construct and operate the reservoirs, results can inform where biodiversity impacts have happened and the results can help to inform which new hydropower reservoirs to prioritize. In addition, the results can help environmental organizations to prevent the construction of future hydropower reservoir with a comparably high biodiversity impact. Policy makers/governments can use the results, for example, to give out licenses for hydropower reservoirs. These licenses could for example include regulations about areas in which hydropower reservoirs should not be built to conserve existing biodiversity and how much inundated land area can be inundated when constructing new reservoirs. The decision making can take place on a national scale. For example, in Norway, the government has pointed out that environmental impacts from hydropower electricity production on Norwegian rivers should be assessed.[44] For the IPCC objective to produce 85% of the total electricity demand with renewable energy sources,[45] decision making needs to take place on a global scale, as fulfilment of SDG 7 most likely depends on a mix of different energy technologies on the global scale.[46] The Norwegian "case studies", which were performed in Chapter 2 and 3, can help to enable local decision making. The gained understanding of local processes has helped to developed global methods assess process globally, and may thus help to enable global decision making.

To exploit the full potential of the LCA approach for achieving a transition into a more sustainable world, decision makers have to be able to apply the results of this thesis and add they have to be added to existing LCIA methodology (e.g.[32,47]). All relevant results are provided in the supporting information (Chapter 7), which allows a direct application by "expert users". To ensure the applicability for a wider field of users, the in this thesis calculated LCI parameters need to be included in LCI databases like Ecoinvent.[48] Even though the units of the calculated Norwegian net water consumption [m^3/kWh] and net land occupation [m^2·yr/kWh] are in accordance with the units of existing inventory parameters,[49] they are not directly implementable in existing LCI databases.[48] Even though, for example, the Ecoinvent data format would allow for spatially differentiation, it currently only has sub-country inventories for electricity grids in China, India, Brazil, Canada, and the United States.[50] Hence, it is only the average net water consumption and net land occupation value for Norway, which is implementable in the existing Ecoinvent database.[50] In addition, commonly used LCA software does not support regionalized inventories.

The same is true for the CFs of spatially-differentiated LCIA models that are in most commercially available software only implemented on a country level. However, the Life Cycle initiative points out, that "*regionalized inventories are necessary to unlock the value of data that already exists in regionalized LCIA methods*".[11] A regionally developed CF, for example on a catchment level, cannot use its full potential if the inventory data is only available on a country level. To ensure the applicability of regionalized LCI and LCIA values, beyond the values calculated in this thesis, further LCI database and software development is urgently needed.[11] The software Brightway,[51] the only one so far allowing full regionalisation, can be seen as pioneer towards achieving this.

The developed CFs also need to be included in larger LCIA methods like ReCiPe,[32] ImpactWorld+[52] and LC-IMPACT[47] to ensure user uptake. In order to implement the CFs, all models need to have global coverage.

However, ReCiPe,[32] for example, has the limitation that they again only provide country averages. In contrast, LC-IMPACT[52] provides spatially differentiated CFs (e.g. land stress), which should be the standard for all LCIA data in the future. Once methodologies are included in relevant methods, these LCIA methodologies have to be implemented again in available LCA software, such as GaBi,[53] SimaPro[54] or, in order to maintain full spatial detail Brightway.[51]

Additional details to pay attention to are whether methods take regional or global species loss into account. For example, ReCiPe[32] (which does also not calculate ecosystem damage with the PDF metric) does not calculate global species loss. The only method that currently allows for estimation of global species losses is LC-IMPACT.[52]

Even if the LCIA methodology is included in LCA software, it is the decision maker who decides which areas of protection, and which impact categories are considered to compare products and processes in LCA. In other words, whether biodiversity impacts should be assessed in an LCA or not. In my opinion, assessments of biodiversity impacts in LCAs can only be achieved, if decision makers are aware of the fact that sustainable development and human development needs rely on a protection of the biosphere and therewith on biodiversity. But it is also consumer awareness in combination with a change in consumer behaviours, that could force practitioners to include biodiversity impacts into their assessment.[55]

Global targets, like the SDGs and the Aichi Targets, are one important step to raise this awareness. Moreover, this awareness could be raised by educating present and future decision-makers. This can be done, for example, via the media, LCA user training and the integration of lectures about environmental impacts in schools and universities. The methods developed in Chapters 2 - 4 contribute to raising this awareness, as they were used for education and public dissemination (e.g. ref[56]).

6.4 Conclusion

This thesis contains the first approaches for quantifying potential biodiversity impacts of hydropower electricity production within LCA. Thereby the results confirm, that Life Cycle Assessment is a suitable tool to assess nexus relationships between the SDGs.[43] LCA has now the potential to assess negative biodiversity trade-offs from hydropower electricity production that could interfere with SDG 6 (Clean water and sanitation) and SDG 15 (Life on land). However, it is not possible to assess all relevant impacts (yet) (see Figure 1 in this chapter), for which further developments are needed.

The model development itself highlights the value of incorporating remote sensing data in the calculation of spatially differentiated LCI inventory parameters on a global scale. In parallel, the model development showed, that it is easier to calculate LCI inventory parameters for future hydropower reservoirs than for existing ones, because it is easier to assess the present state than to reconstruct the pre-hydropower state. Even if we may not be able to avoid negative trade-offs from existing hydropower electricity production, stakeholders should be able to account for potential negative biodiversity trade-offs in future hydropower projects. Nevertheless, biodiversity trade-offs are only one important layer of the whole assessment, as other factors like social aspects or human health impacts can have additional negative-trade-offs that need to be assessed in additional studies. Furthermore, the fulfilment of SDG 7 most likely relies on the correct mix of renewable energy technologies.[46] Therefore, stakeholders should cross compare the biodiversity impacts of all renewable energy sources and should not only focus on one renewable energy source like hydropower. This thesis indicates that this comparison could also be done within the field of Life Cycle Assessment, provided that harmonised LCIA methodologies exist for other renewable energy technologies.

6.5 Outlook

A fully operational assessment of biodiversity impacts of hydropower electricity production in LCA requires further development in the following areas:

A. Global net land occupation and net water consumption LCI values

The main input parameter for the land occupation and water consumption values comes from open source Landsat satellite images with global coverage and a global evaporation model. By analyzing more Landsat satellite images, more spatially-explicit land occupation and water consumption values can be calculated. The Global Reservoir and Dam (GRanD) Database represents a good starting point to identify where existing reservoirs are located globally.[57] Furthermore, it would be beneficial to develop a method, that allows for identifying which land cover and/or habitat types are inundated.

B. Accounting for possible water diversion

To enable an even more flexible hydropower electricity production, water can be diverted in tunnels between catchments. In one catchment this can cause an additional water consumption, while in the other catchment this may lead to an increased water availability.[58] In other words, it can offset the water consumption from evaporation. However, this process is not covered in Chapter 3, mainly due to a lack of data. Water diversion may require a new concept, which is able to assess the impacts of increased discharge, as fish species are also impacted when the natural flow is increased.[29]

C. Assessing additional land use change

Besides reservoir creation land use change is caused by the construction of infrastructure, such as power lines[59] and access roads,[60] and resettlement of people, which used to live in the inundated reservoir area.[61] However, these types of land occupation are not covered in the thesis, which may lead to an underestimation of the land occupation and consequently also of the terrestrial biodiversity impact.

D. Development of water level regulation – reservoir surface area curves

Due to a lack in water level regulation – reservoir surface areas curves, the calculation of the LCI values is based on the assumption, that the hydropower reservoirs are always completely full, although reservoirs generally contain less than 90% of their maximal capacity.[27] This assumptions leads to uncertainty for the calculated net land occupation values, but especially for the net water consumptions values. To reduce the uncertainty, models describing the relationship between water-level regulation and reservoir surface area are needed.

E. Characterization factors quantifying biodiversity impacts of present land inundation

To quantify the biodiversity impact of present hydropower reservoir land occupation (as quantified in Chapter 2), new CFs accounting for historical changes from terrestrial habitat to aquatic habitat are required. This could, for example, be done by introducing a land use class "new permanent water" into the existing Chaudhary et al.[13] model framework. The land use class "new permanent water" could be quantified with remote sensing data.[62] A currently ongoing master project at the Industrial Ecology programme, aims to narrow this research gap.

F. Characterization factors quantifying the impacts of water consumption on freshwater biodiversity

Despite the developed CFs in this thesis, LCIA is still not able to quantify spatially-explicit impacts of water consumption on freshwater biodiversity on a global scale. Further spatially-explicit CFs, quantifying the impacts of water consumption on freshwater biodiversity, are needed to achieve a global coverage. Furthermore, the taxonomic coverage of the existing CFs should be increased. For example, by developing more SDRs for macroinvertebrates, as started by Tendall et al.[22]

G. Characterization factors quantifying the impacts of non-consumptive water use on freshwater biodiversity

So far no CF assessing the biodiversity impacts of non-consumptive water use exists. To make the biodiversity impact estimates of the LCIA impact category "water stress" more complete, CFs covering freshwater impacts of non-consumptive water use have to be developed.

H. Accounting for biodiversity impacts of habitat fragmentation

The currently existing CFs for terrestrial and freshwater biodiversity (including the CFs developed in this thesis), are currently not accounting for biodiversity impacts of habitat fragmentation. However, Koen Kuipers, a PhD candidate at the Industrial Ecology Programme, is currently working on the inclusion of habitat fragmentation biodiversity impacts into the "land stress" impact category. For freshwater biodiversity the research gap remains, but ecological concepts (e.g. refs[63-65]) represent a promising starting point.

I. Increase of taxonomic coverage

For terrestrial biodiversity impacts, the developed models cover four taxonomic groups (terrestrial mammals, birds, reptiles and amphibians), while for aquatic impacts the models only cover fish. Even though fish can function as a good indicator for ecosystem health,[66] the aquatic models developed and applied in this thesis may overlook a taxonomic specific impact on, for example, macroinvertebrates. This highlights that further development to cover more taxonomic groups, for both terrestrial (e.g. plants) and aquatic species is needed. This could be done, for example, by developing more SDRs for macroinvertebrates, as done by Tendall et al.[22]

J. Accounting for seasonal aspects

The developed models calculated average annual LCI parameters and CFs quantifying the average annual biodiversity impact. However, the water consumption of hydropower reservoirs can vary between day and night and between summer and winter. At the same time, both Teichert et al.[67] and Puffer et al.[68] showed evidence for a linear relation between growth of juvenile salmon and discharge rates during summer. But during winter, discharge did not seem to affect growth rates.[67,68] Using CFs with annual averages thus likely overestimates the impacts, as water consumption during a specific season does not necessarily always lead to a biodiversity impact. Thus, seasonal CFs should be developed, even though these have a high parameter demand and require a more detailed understanding of the underlying ecological processes.

K. Enabling a comparison to other renewable energy sources

   To increase the share of renewable energy in the global energy mix with a minimum trade-off, and to identify a relatively low or high biodiversity impact of hydropower electricity, the biodiversity impacts of hydropower electricity production have to be compared to other renewable electricity sources like wind power. However, a comparable study for other renewable energy sources, does not exist.[69] For electricity production from onshore wind power, however, the SURE project is currently developing new LCIA methods,[69] which will allow a cross-comparison with hydropower electricity production in the future.

In summary, further research is needed to operationally assess biodiversity impacts of hydropower electricity production from freshwater habitat alteration, water quality degradation, and land use and land use change in LCA.[4] The result of this thesis, the ongoing research and the rapidly increasing amount of global datasets, make me confident that LCA will be able to asses biodiversity impacts from the main cause-effect pathways of hydropower electricity production, and possibly other renewable energy sources, in the near future. Thereby, LCA will be able to contribute to the transition into a more sustainable future.

6.6 References

1. Raadal, H. L.; Gagnon, L.; Modahl, I. S.; Hanssen, O. J., Life cycle greenhouse gas (GHG) emissions from the generation of wind and hydro power. *Renewable and Sustainable Energy Reviews* **2011,** *15*, (7), 3417-3422.
2. Varun; Bhat, I. K.; Prakash, R., LCA of renewable energy for electricity generation systems—A review. *Renewable and Sustainable Energy Reviews* **2009,** *13*, (5), 1067-1073.
3. Turconi, R.; Boldrin, A.; Astrup, T., Life cycle assessment (LCA) of electricity generation technologies: Overview, comparability and limitations. *Renewable and Sustainable Energy Reviews* **2013,** *28*, 555-565.
4. Gracey, E. O.; Verones, F., Impacts from hydropower production on biodiversity in an LCA framework—review and recommendations. *The International Journal of Life Cycle Assessment* **2016,** *21*, (3), 412-428.
5. USGS (United States Geological Survey). EarthExplorer-Global Land Survey 1975 dataset. http://earthexplorer.usgs.gov/
6. Gutman, G.; Huang, C.; Chander, G.; Noojipady, P.; Masek, J. G., Assessment of the NASA–USGS Global Land Survey (GLS) datasets. *Remote Sensing of Environment* **2013,** *134*, 249-265.
7. Mu, Q.; Zhao, M.; Running, S. W., Improvements to a MODIS global terrestrial evapotranspiration algorithm. *Remote Sensing of Environment* **2011,** *115*, (8), 1781-1800.
8. University of Montana, MODIS Global Evapotranspiration Project (MOD16). In.
9. Flury, K.; Frischknecht, R., Life cycle inventories of hydroelectric power generation. *ESU-Services, Fair Consulting in Sustainability, commissioned by€ Oko-Institute eV* **2012**, 1-51.
10. Mutel, C. L.; Pfister, S.; Hellweg, S., GIS-Based Regionalized Life Cycle Assessment: How Big Is Small Enough? Methodology and Case Study of Electricity Generation. *Environmental Science & Technology* **2012,** *46*, (2), 1096-1103.
11. Mutel, C.; Liao, X.; Patouillard, L.; Bare, J.; Fantke, P.; Frischknecht, R.; Hauschild, M.; Jolliet, O.; Maia de Souza, D.; Laurent, A.; Pfister, S., Overview and recommendations for regionalized life cycle impact assessment. *The International Journal of Life Cycle Assessment* **2018**.
12. Verones, F.; Bare, J.; Bulle, C.; Frischknecht, R.; Hauschild, M.; Hellweg, S.; Henderson, A.; Jolliet, O.; Laurent, A.; Liao, X., LCIA framework and cross-cutting issues guidance within the UNEP-SETAC Life Cycle Initiative. *Journal of cleaner production* **2017,** *161*, 957-967.
13. Chaudhary, A.; Verones, F.; de Baan, L.; Hellweg, S., Quantifying Land Use Impacts on Biodiversity: Combining Species–Area Models and Vulnerability Indicators. *Environmental Science & Technology* **2015,** *49*, (16), 9987-9995.
14. Jolliet, O.; Antón, A.; Boulay, A.-M.; Cherubini, F.; Fantke, P.; Levasseur, A.; McKone, T. E.; Michelsen, O.; Milà i Canals, L.; Motoshita, M.; Pfister, S.; Verones, F.; Vigon, B.; Frischknecht, R., Global guidance on environmental life cycle impact assessment indicators: impacts of climate change, fine particulate matter formation, water consumption and land use. *The International Journal of Life Cycle Assessment* **2018**.
15. Church, J. A.; Clark, P. U.; Cazenave, A.; Gregory, J. M.; Jevrejeva, S.; Levermann, A.; Merrifield, M. A.; Milne, G. A.; Nerem, R. S.; Nunn, P. D.; Payne, A. J.; Pfeffer, W. T.; Stammer, D.; Unnikrishnan, A. S., Sea Level Change. In *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A.*

Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 2013.
16. Howard, B. C. First mammal species recognized as extinct due to climate change. https://news.nationalgeographic.com/2016/06/first-mammal-extinct-climate-change-bramble-cay-melomys/
17. Waller, N. L.; Gynther, I. C.; Freeman, A. B.; Lavery, T. H.; Leung, L. K. P., The Bramble Cay melomys Melomys rubicola (Rodentia : Muridae): a first mammalian extinction caused by human-induced climate change? *Wildlife Research* **2017,** *44*, (1).
18. Chaudhary, A.; Brooks, T. M., Land Use Intensity-Specific Global Characterization Factors to Assess Product Biodiversity Footprints. *Environmental Science & Technology* **2018,** *52*, (9), 5094-5104.
19. de Baan, L.; Mutel, C. L.; Curran, M.; Hellweg, S.; Koellner, T., Land Use in Life Cycle Assessment: Global Characterization Factors Based on Regional and Global Potential Species Extinction. *Environmental Science & Technology* **2013,** *47*, (16), 9281-9290.
20. Verones, F.; Saner, D.; Pfister, S.; Baisero, D.; Rondinini, C.; Hellweg, S., Effects of Consumptive Water Use on Biodiversity in Wetlands of International Importance. *Environmental Science & Technology* **2013,** *47*, (21), 12248-12257.
21. Clark, P. U.; Dyke, A. S.; Shakun, J. D.; Carlson, A. E.; Clark, J.; Wohlfarth, B.; Mitrovica, J. X.; Hostetler, S. W.; McCabe, A. M., The Last Glacial Maximum. *Science* **2009,** *325*, (5941), 710-714.
22. Tendall, D. M.; Hellweg, S.; Pfister, S.; Huijbregts, M. A. J.; Gaillard, G., Impacts of River Water Consumption on Aquatic Biodiversity in Life Cycle Assessment—A Proposed Method, and a Case Study for Europe. *Environmental Science & Technology* **2014,** *48*, (6), 3236-3244.
23. Hanafiah, M. M.; Xenopoulos, M. A.; Pfister, S.; Leuven, R. S. E. W.; Huijbregts, M. A. J., Characterization Factors for Water Consumption and Greenhouse Gas Emissions Based on Freshwater Fish Species Extinction. *Environmental Science & Technology* **2011,** *45*, (12), 5272-5278.
24. Kuipers, K.; Hellweg, S.; Verones, F., Potential consequences of regional species loss for global species richness – a quantitative approach for estimating global extinction probabilities. *Manuscript submitted for publication* **2018**.
25. Huijbregts, M. A.; Gilijamse, W.; Ragas, A. M.; Reijnders, L., Evaluating uncertainty in environmental life-cycle assessment. A case study comparing two insulation options for a Dutch one-family dwelling. *Environmental Science & Technology* **2003,** *37*, (11), 2600-2608.
26. Ozesmi, S. L.; Bauer, M. E., Satellite remote sensing of wetlands. *Wetlands Ecology and Management* **2002,** *10*, (5), 381-402.
27. Norwegian Water Resources and Energy Directorate. Diagram magasinfyllingen - hele Norge. http://vannmagasinfylling.nve.no/Default.aspx?ViewType=Chart&Tidsenhet=Aar&Omr=NO
28. GBIF (The Global Biodiversity Information Facility) *Strategic Plan 2017–2021*; 2017.
29. Poff, N. L.; Zimmerman, J. K. H., Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. *Freshwater Biology* **2010,** *55*, (1), 194-205.
30. Mittelbach, G. G.; Steiner, C. F.; Scheiner, S. M.; Gross, K. L.; Reynolds, H. L.; Waide, R. B.; Willig, M. R.; Dodson, S. I.; Gough, L., What Is the Observed Relationship between Species Richness and Productivity? *Ecology* **2001,** *82*, (9), 2381-2396.

31. Xenopoulos, M. A.; Lodge, D. M., Going with the flow: Using species-discharge relationships to forecast losses in fish biodiversity. *Ecology* **2006,** *87*, (8), 1907-1914.
32. Huijbregts, M. A. J.; Steinmann, Z. J. N.; Elshout, P. M. F.; Stam, G.; Verones, F.; Vieira, M.; Zijp, M.; Hollander, A.; van Zelm, R., ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. *The International Journal of Life Cycle Assessment* **2016,** *22*, (2), 138-147.
33. McGarvey, D. J., Moving beyond species-discharge relationships to a flow-mediated, macroecological theory of fish species richness. *Freshwater Science* **2014,** *33*, (1), 18-31.
34. Xu, Z.-H.; Yin, X.-A.; Zhang, C.; Yang, Z.-F., Piecewise model for species–discharge relationships in rivers. *Ecological Engineering* **2016,** *96*, 208-213.
35. Scherer, L.; Pfister, S., Hydropower's Biogenic Carbon Footprint. *Plos One* **2016,** *11*, (9), e0161947.
36. Deemer, B. R.; Harrison, J. A.; Li, S.; Beaulieu, J. J.; DelSontro, T.; Barros, N.; Bezerra-Neto, J. F.; Powers, S. M.; dos Santos, M. A.; Vonk, J. A., Greenhouse Gas Emissions from Reservoir Water Surfaces: A New Global Synthesis. *BioScience* **2016,** *66*, (11), 949-964.
37. Chanudet, V.; Descloux, S.; Harby, A.; Sundt, H.; Hansen, B. H.; Brakstad, O.; Serça, D.; Guerin, F., Gross CO2 and CH4 emissions from the Nam Ngum and Nam Leuk sub-tropical reservoirs in Lao PDR. *Science of The Total Environment* **2011,** *409*, (24), 5382-5391.
38. Hertwich, E. G., Addressing Biogenic Greenhouse Gas Emissions from Hydropower in LCA. *Environmental Science & Technology* **2013,** *47*, (17), 9604-9611.
39. Forseth, T.; Harby, A. *Handbook for environmental design in regulated salmon rivers*; 2014.
40. Klöpffer, W., *Background and future prospects in life cycle assessment*. Springer Science & Business Media: 2014.
41. International Standard Organization *ISO 14044: Environmental management — Life cycle assessment — Principles and framework*; International Organisation for Standardization: Geneva, Switzerland, 2006.
42. Fuso Nerini, F.; Tomei, J.; To, L. S.; Bisaga, I.; Parikh, P.; Black, M.; Borrion, A.; Spataru, C.; Castán Broto, V.; Anandarajah, G.; Milligan, B.; Mulugetta, Y., Mapping synergies and trade-offs between energy and the Sustainable Development Goals. *Nature Energy* **2018,** *3*, (1), 10-15.
43. Liu, J.; Hull, V.; Godfray, H. C. J.; Tilman, D.; Gleick, P.; Hoff, H.; Pahl-Wostl, C.; Xu, Z.; Chung, M. G.; Sun, J.; Li, S., Nexus approaches to global sustainable development. *Nature Sustainability* **2018,** *1*, (9), 466-476.
44. Norwegian Government Ministries and Offices, Meld. St. 25 (2015–2016) Power to Change - Energy policy 2030 (In Norwegian).
45. Intergovernmental Panel on Climate Change *Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty [V. Masson-Delmotte, P. Zhai, H. O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J. B. R. Matthews, Y. Chen, X. Zhou, M. I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, T. Waterfield (eds.)]*; In Press., 2018.
46. Hertwich, E.; de Larderel, J. A.; Arvesen, A.; Bayer, P.; Bergesen, J.; Bouman, E.; Gibon, T.; Heath, G.; Peña, C.; Purohit, P. *Green Energy Choices: The benefits, risks, and trade-offs of low-carbon technologies for electricity production*; United Nation Environmental Program 2016.

47. LC-Impact Version 0.5 - A spatially differentiated life cycle impact assessment approach. https://lc-impact.eu/about.html
48. Wernet, G.; Bauer, C.; Steubing, B.; Reinhard, J.; Moreno-Ruiz, E.; Weidema, B., The ecoinvent database version 3 (part I): overview and methodology. *The International Journal of Life Cycle Assessment* **2016,** *21*, (9), 1218-1230.
49. Koellner, T.; de Baan, L.; Beck, T.; Brandão, M.; Civit, B.; Margni, M.; i Canals, L. M.; Saad, R.; de Souza, D. M.; Müller-Wenk, R., UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. *The International Journal of Life Cycle Assessment* **2013,** *18*, (6), 1188-1202.
50. Mutel, C. Ecoinvent geography definitions. https://geography.ecoinvent.org/
51. Mutel, C. Brightway. https://brightwaylca.org/
52. Bulle, C.; Margni, M.; Patouillard, L.; Boulay, A.-M.; Bourgault, G.; De Bruille, V.; Cao, V.; Hauschild, M.; Henderson, A.; Humbert, S.; Kashef-Haghighi, S.; Kounina, A.; Laurent, A.; Levasseur, A.; Liard, G.; Rosenbaum, R. K.; Roy, P.-O.; Shaked, S.; Fantke, P.; Jolliet, O., IMPACT World+: a globally regionalized life cycle impact assessment method. *The International Journal of Life Cycle Assessment* **2019**.
53. thinkstep GaBi. http://www.gabi-software.com/norway/index/
54. Pré Consultants B.V. SimaPro. https://simapro.com/
55. Ivanova, D. Environmental impacts of household consumption: from demand patterns to mitigation strategies. Norwegian University of Science and Technology, Trondheim 2018.
56. Bazilchuk, N. Land under water: Estimating hydropower's land use impacts. https://geminiresearchnews.com/2018/03/land-water-estimating-hydropowers-land-use-impacts/
57. Lehner, B.; Liermann, C. R.; Revenga, C.; Vörösmarty, C.; Fekete, B.; Crouzet, P.; Döll, P.; Endejan, M.; Frenken, K.; Magome, J.; Nilsson, C.; Robertson, J. C.; Rödel, R.; Sindorf, N.; Wisser, D., High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. *Frontiers in Ecology and the Environment* **2011,** *9*, (9), 494-502.
58. Annys, S.; Adgo, E.; Ghebreyohannes, T.; Van Passel, S.; Dessein, J.; Nyssen, J., Impacts of the hydropower-controlled Tana-Beles interbasin water transfer on downstream rural livelihoods (northwest Ethiopia). *Journal of Hydrology* **2019,** *569*, 436-448.
59. Gundula, S. B.; Roel, M.; Kjetil, B.; Sigbjorn, S.; Eivin, R., The effects of power lines on ungulates and implications for power line routing and rights-of-way management. *International Journal of Biodiversity and Conservation* **2014,** *6*, (9), 647-662.
60. McAllister, D. E.; Craig, J. F.; Davidson, N.; Delany, S.; Seddon, M. *Biodiversity impacts of large dams*; 2001.
61. Chen, J.; Deng, Y.; Liu, H.; Wang, K., Review of and analysis on resettlement induced by hydropower projects in China. In *Proceedings of the 2017 2nd International Conference on Civil, Transportation and Environmental Engineering (ICCTE 2017)*, 2017.
62. Pekel, J.-F.; Cottam, A.; Gorelick, N.; Belward, A. S., High-resolution mapping of global surface water and its long-term changes. *Nature* **2016,** *540*, 418.
63. McKay, S. K.; Schramski, J. R.; Conyngham, J. N.; Fischenich, J. C., Assessing upstream fish passage connectivity with network analysis. *Ecological Applications* **2013,** *23*, (6), 1396-1409.
64. Erős, T.; Olden, J. D.; Schick, R. S.; Schmera, D.; Fortin, M.-J., Characterizing connectivity relationships in freshwaters using patch-based graphs. *Landscape Ecology* **2011,** *27*, (2), 303-317.

65. Eros, T.; Schmera, D.; Schick, R. S., Network thinking in riverscape conservation - A graph-based approach. *Biological Conservation* **2011,** *144*, (1), 184-192.
66. Schiemer, F., Fish as indicators for the assessment of the ecological integrity of large rivers. *Hydrobiologia* **2000,** *422*, 271-278.
67. Teichert, M. A. K.; Kvingedal, E.; Forseth, T.; Ugedal, O.; Finstad, A. G., Effects of discharge and local density on the growth of juvenile Atlantic salmon Salmo salar. *Journal of Fish Biology* **2010,** *76*, (7), 1751-1769.
68. Puffer, M.; Berg, O. K.; Huusko, A.; Vehanen, T.; Forseth, T.; Einum, S., Seasonal Effects of Hydropeaking on Growth, Energetics and Movement of Juvenile Atlantic Salmon (Salmo Salar). *River Research and Applications* **2015,** *31*, (9), 1101-1108.
69. Laranjeiro, T.; May, R.; Verones, F., Impacts of onshore wind energy production on birds and bats: recommendations for future life cycle impact assessment developments. *The International Journal of Life Cycle Assessment* **2018**.

Chapter 7: Supporting information

7.1 Supporting information for Chapter 2

Modeling net land occupation of hydropower reservoirs in Norway for use in Life Cycle Assessment

Supporting Information 1

Martin Dorber*,†, Roel May‡, Francesca Verones†

†Department of Energy and Process Engineering, NTNU, Sem Sælands vei 7, 7491 Trondheim, Norway

‡Norwegian Institute for Nature Research, Høgskoleringen 9, 7034 Trondheim, Norway

*Corresponding author e-mail: martin.dorber@ntnu.no

15 pages

9 figures

5 tables

Environmental Science and Technology

Table of Contents

Supporting Information 1

S1: List of acronyms and abbreviations

ALO	-	Adjusted land occupation
ASL	-	Average slope
ER	-	Average annual electricity production of hydropower reservoir *x*
E	-	Average annual electricity production at hydropower plant *z*
GLS	-	NASA-USGS Global Land Survey Dataset
ILA	-	Inundated land area
LCA	-	Life Cycle Assessment
LCI	-	Life Cycle Inventory
LCIA	-	Life Cycle Impact Assessment
LO	-	Land occupation
LULUC	-	Land use and land use change
MSL	-	Maximal slope
NVE	-	Norwegian Water Resources and Energy Directorate
PR	-	Pixel resolution
RSA	-	Actual reservoir surface area at highest regulated water level
SILA	-	Slope adjusted inundated land area
WSA	-	Water surface area before dam construction
WP	-	Water pixels

S2: Hydropower reservoirs in Norway

Hydropower electricity production started in the early nineteenth century.[1] Norway is particularly suited for hydropower electricity production due to its comparably high precipitation, many lakes, valleys and topographic elevation change that results in an abundance of potential energy that can be converted into electricity.

In 2013, Norway produced approximately 129 TWh of hydropower electricity annually.[2] The ten largest hydropower plants together accounted for approximately 18% of the total Norwegian hydropower electricity production in 2013.[2] According to the Norwegian Water Resources and Energy Directorate (NVE).[1] 1,288 out of the 2,280 reservoirs located in Norway are used for electricity production in 587 power plants (Figure S1). This means that one hydropower plant can be assigned to up to 15 reservoirs. The remaining reservoirs are mainly used for water supply. Most of the hydropower reservoirs consist of dammed natural lakes.[3] The total hydropower reservoir area in Norway is 5,992 km^2.[1] Actual reservoir surface area at highest regulated water level of hydropower reservoirs in Norway varies between 0.01 and 375.95 km^2.[1]

The commissioning year of 265 hydropower plants of these is after 1972 (Figure S2).[1] Thereof 90 hydropower reservoirs are smaller than 0.5 km^2 (Figure S3).

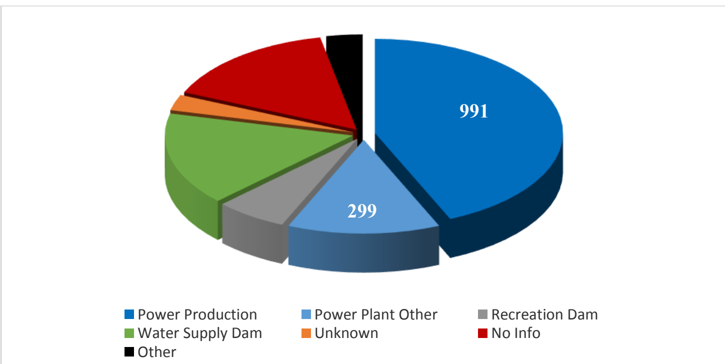


Figure S1: Use of Reservoirs in Norway.[1]

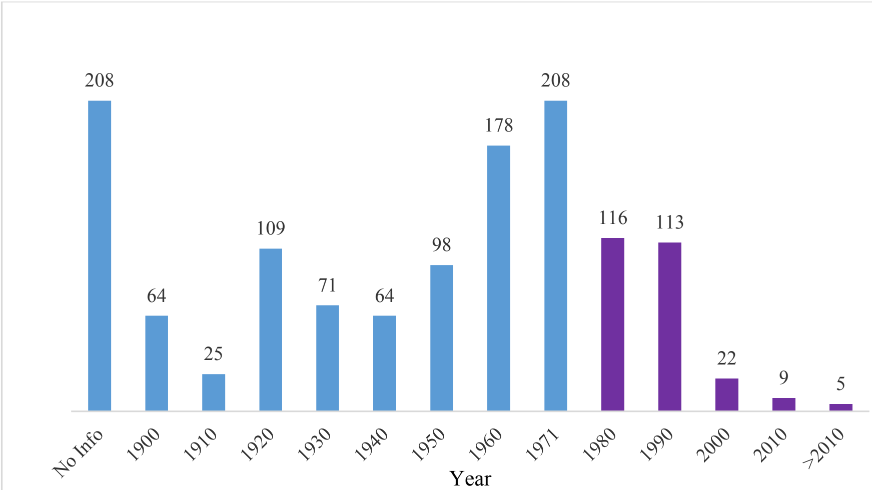


Figure S2: Histogram of commissioning year of hydropower reservoirs in Norway. [1]

Hydropower reservoirs relevant for this study are highlighted in purple.

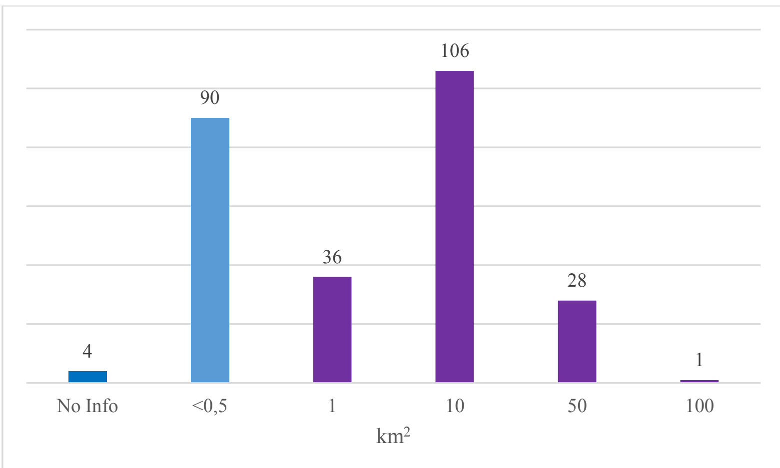


Figure S3: Histogram of actual reservoir surface area at highest regulated water level of hydropower reservoirs with commissioning year after 1972. [1]

Categories containing hydropower reservoirs relevant for this study are highlighted in purple.

S3: Landsat images and maximum likelihood classification

The NASA-USGS Global Land Survey dataset (GLS) is offering a Landsat satellite images collection, assembled in five epochs, with a nearly complete coverage of the global land area and a resolution of 60 m. The oldest assembled epoch, is the GLS-1975 data set. A Multispectral Scanner onboard Landsat satellites 1-3 acquired the GLS-1975 images from 1972 to 1983. The multispectral images consist of four bands with wavelength in micrometers: Band 1(4): 0.5-0.6 Band 2(5): Band 3(6): 0.6-0.7 Band 4 (7): 0.8-1.1.[4] We extracted all multispectral images from the GLS-1975 available for Norway, not totally covered with ice and snow.[4] Images used are shown in Table S1.

Table S1: Overview of satellite images used from the GLS-1975 Dataset.[5]

p208r010_1dm19730721 for example stands for:

Path 208; Row 10; Landsat 1; Image date: 21.07.1973

Satellite	Path	Image 1	Image 2	Image 3	Image 4	Image 5	Image 6	Image 7
Landsat 1	208	p208r010_1dm19730721	p208r011_1dm19730721					
Landsat 1	213	p213r018_1dm19730602	p213r019_1dm19730602					
Landsat 1	214	p214r018_1dm19750611	p214r016_1dm19760921	p214r015_1dm19760921				
Landsat 1	215	p215r018_1dm19760817	p215r016_1dm19760817					
Landsat 2	206	p206r011_2dm19790715						
Landsat 2	209	p209r010_2dm19800817						
Landsat 2	212	p212r011_2dm19800802	p212r010_2dm19800802					
Landsat 2	213	p213r015_2dm19791002	p213r017_2dm19790511					
Landsat 2	214	p214r017_2dm19791003						
Landsat 2	215	p215r011_2dm19780816	p215r013_2dm19750727	p215r015_2dm19750709	p215r017_2dm19770716	p215r019_2dm19770523		
Landsat 2	216	p216r013_2dm19791005	p216r018_2dm19750622	p216r019_2dm19750622	p216r017_2dm19750728	p216r014_2dm19791005	p216r016_2dm19760827	p216r015_2dm19791005
Landsat 2	217	p217r018_2dm19760705	p217r017_2dm19760705	p217r016_2dm19760705				
Landsat 3	216	p216r011_3dm19820928						

Each Landsat satellite flies along its path, while taking images of the area below. Due to the orbital parameters of the satellites, the path is shifted west each day. Therefore, each path of the following day partially overlaps with the path of the previous day.[6] The paths of Landsat satellites for Norway and the paths combination used in this study are shown in Figure S4**.**

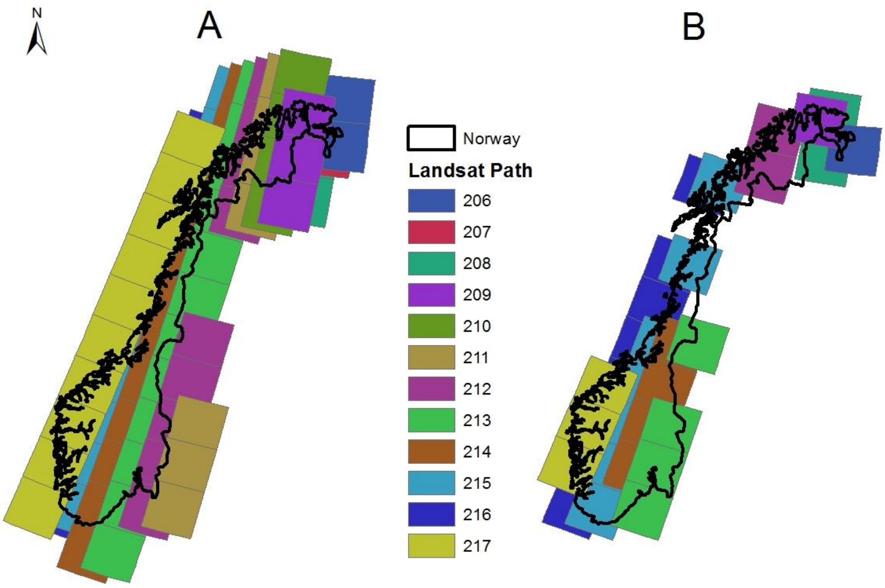


Figure S4: Landsat1-3 path;[7] A shows the path of Landsat satellites for Norway,[8] thereby each square represents a row; B shows the paths used in this study.

Each path itself consists of rows, representing the latitudinal centerline of a frame of imagery (Figure S5).[7]

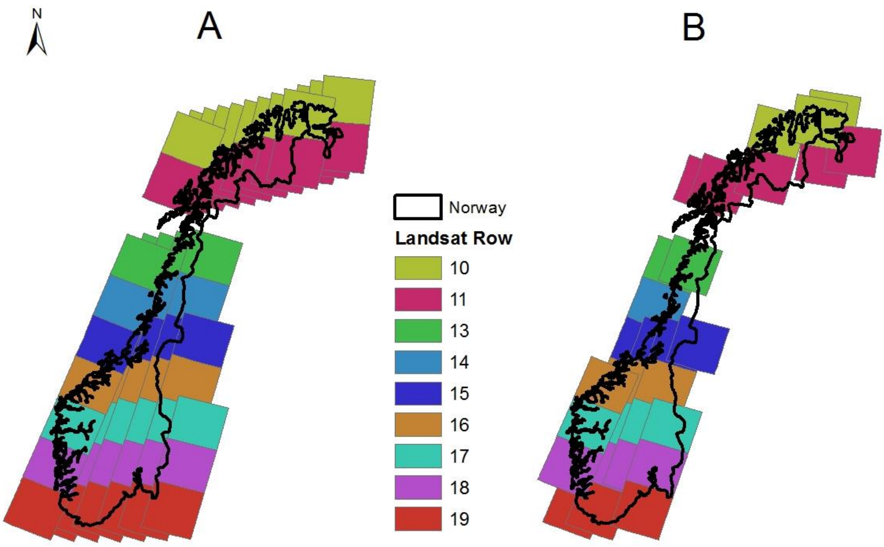


Figure S5: Landsat 1-3 rows;[7] A shows the rows of Landsat satellites for Norway;[8] B shows the rows used in this study.

Consequently, images taken in the same row can overlap, while images taken in the same path do not overlap. However, this statement is only true per Landsat satellite. As each of the three Landsat Satellites used in the GLS1975 flies the same path, images between Landsat satellites can overlap. Therefore, we merged the images by path and Landsat satellite to avoid an overlap of images and to ensure that all images available for Norway are used. As result, we obtained 13 merged multispectral images from the three Landsat satellites for the Maximum Likelihood Classification (Figure S6).

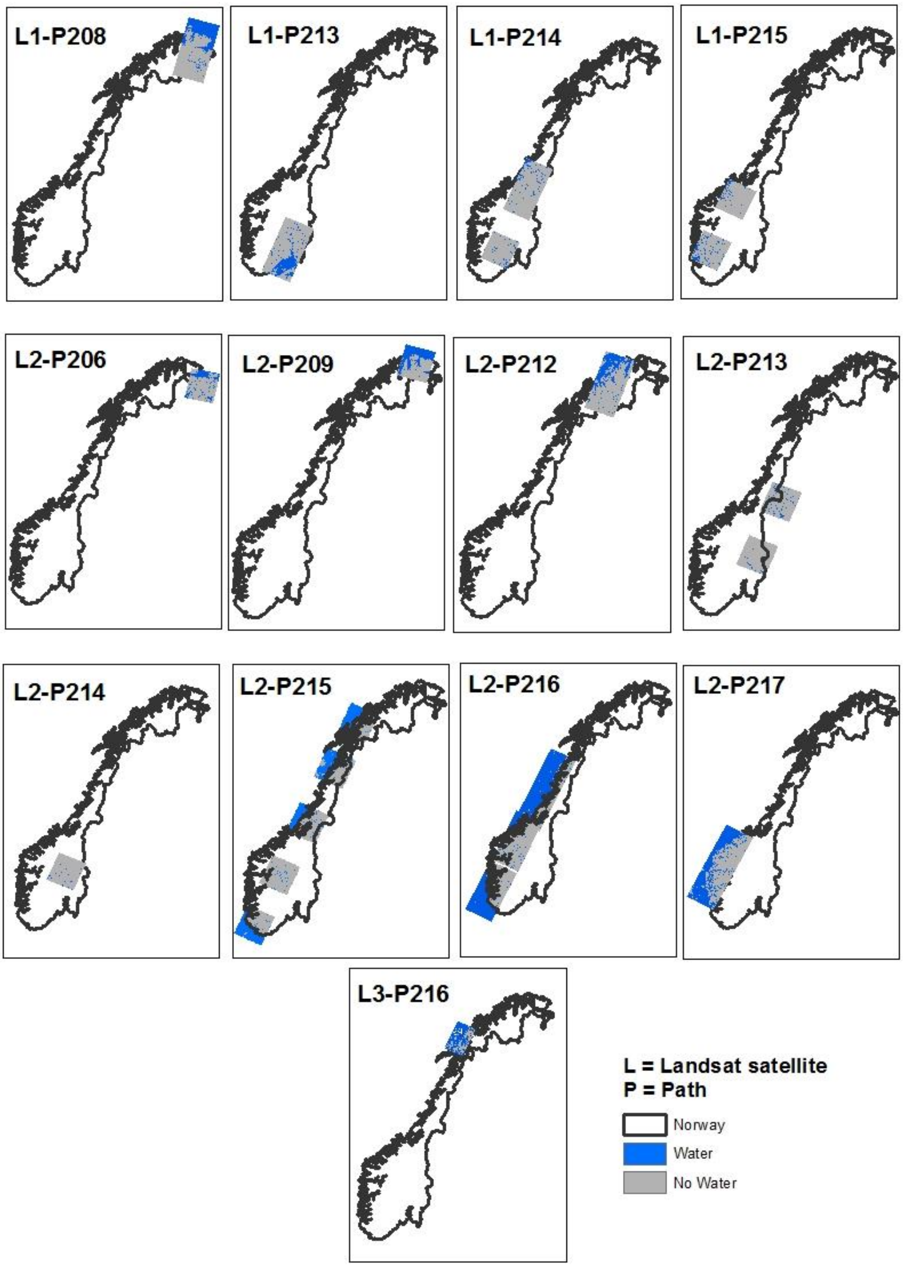


Figure S6: The 13 merged classified multispectral images from the three Landsat satellites that we obtained from merging the 32 images of Table S1.[8]

To perform the Maximum Likelihood Classification we created for each of the 13 merged multispectral images training areas, separating between water and non-water. Classification results are shown in Figure S6. In total 140,240 pixels, were used for water training areas and 133,405 pixels for non-water training areas. As example the scatterplot of the training areas from Landsat 2 Path 215 are shown in Figure S7 and associated statistics in Table S2.

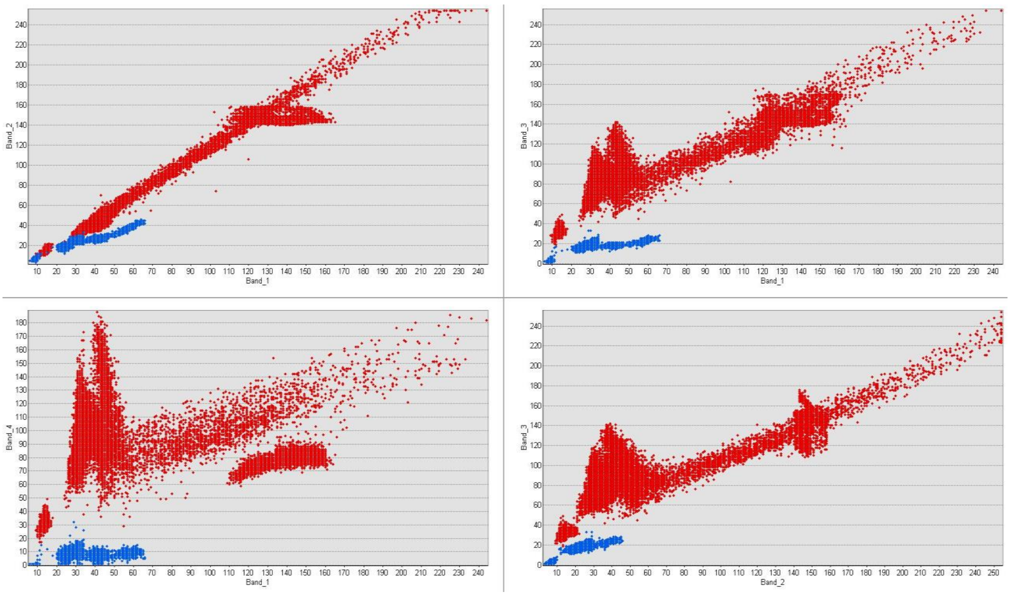


Figure S7 : Scatterplot of water (blue) and non-water (red) training areas of Landsat 2 path 215.

Table S2: Statistic of water and non-water training areas pixel values of Landsat 2 path 215.

Training Area	**Water**	**Non-Water**	**Water**	**Non-Water**	**Water**	**Non-Water**	**Water**	**Non-Water**
Band	1	1	2	2	3	3	4	4
Minimum	6	9	3	9	1	19	1	17
Maximum	66	244	46	254	33	254	32	188
Mean	22	68	17	74	12	101	6	91
Std.dev	12	49	9	53	8	38	4	27
Covariance								
Band 1	156	2360	108	2559	85	1656	40	74
Band 2	108	2559	80	2853	66	1790	32	87
Band 3	85	1656	66	1790	60	1415	30	430
Band 4	40	74	32	87	30	430	18	708

S4: Aerial photographs

From seven aerial photographs we obtained WSA of 12 hydropower reservoirs, directly by using the online measurement tool from the internet portal Norge i Bilder[9] (Table S3). Direct measurement was possible due to image resolution of 0.2 m.[9]

Table S3: Aerial photographs used from the internet portal Norge i Bilder[9] and the related number of hydropower reservoirs where inundated land area (ILA) was quantified.

Aerial photograph	Exposure year	Hydropower reservoirs with quantified ILA
Selbusjøen-Tya-Nea	1952	1
Nea-Essandsjø-Stuesjø	1953	2
Selbu-Tydal-Holtålen nord	1961	1
Melhus-midtre Gauldal	1963	1
Melhus-midtre Gauldal	1964	1
Rindal-Surnadal-Stangvik	1963	5

S5: Slope-adjusted inundated land area calculation

The slope adjusted inundated land area [m^2] (SILA), can be calculated with Eq. S1 and is visualized in Figure S8.

$$\mathrm{SILA}_x = \frac{\mathrm{ILA}_x}{\sin(90 - \alpha_x)} \tag{S1}$$

Where α is the slope [degree] inside the reservoir x and ILA is the inundated land area of hydropower reservoir x in m^2. Eq. S1 shows that every slope above zero will result in an underestimation of the inundated land area: the higher the slope, the greater the underestimation.

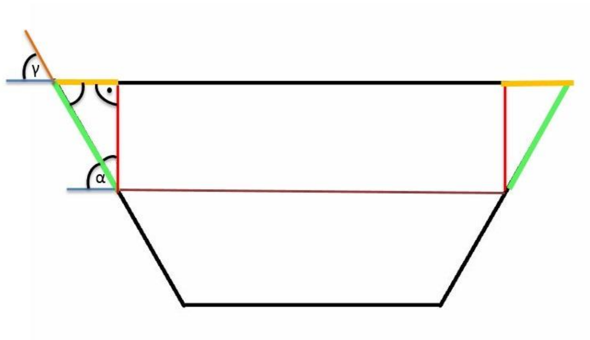


Figure S8: Schematic outline of a lake. The yellow line represents the projected inundated land area measured from remote sensing data (ILA), the green shows the inundated land area considering slope (SILA) and the brown line is the water level before dam construction; α = slope inside reservoir; γ= slope outside reservoir

As the reservoir is already flooded, the slope inside the reservoirs is unknown. However, it can be assumed that the surrounding slope bordering the reservoir is similar to the slope inside the reservoir.[10] We used the 25-m European Digital Elevation Model (EU-DEM, Version 1)[11] to calculate average (ASL) and maximum slope (MSL) within a 25 and 50 m buffer (representing 1 and 2 pixels of the DEM) around the reservoir, assuming these to represent α (Figure S8).

The total estimated slope-adjusted inundated land area from each of the four used scenarios was always larger than the total inundated land area estimated from satellite images, which was based on a planar surface (Table S4).

Table S4: Comparison of inundated land area (ILA) and slope adjusted inundated land area (SILA) in [km^2] for 184 hydropower plants in Norway. ASL= Average slope within a 25- or 50-m buffer around actual reservoir surface area at highest regulated water level; MSL = Maximum slope within a 25- or 50-m buffer around actual reservoir surface area at highest regulated water level

.

		SILA			
	ILA	**ASL25M**	**MSL25M**	**ASL50M**	**MSL50M**
Total	305.3	315	494.5	314.6	509.6
Median	0.30	0.30	0.37	0.30	0.37

We performed a Mann–Whitney U test in IBM SPSS Statistic 24,[12] a nonparametric test to determine statistically significant differences between two non-normal distributed groups,[13] to test for statistical differences between inundated land area with and without sloped terrain, as inundated land area and slope adjusted inundated land area data was not normally distributed (P < 0,000; Shapiro-Wilk). The results in Table S5 show, that there is no signifanct difference between ILA and SILA.

Table S5: Mann-Whitney U test results between inundated land area (ILA) and slope adjusted inundated land area (SILA). ASL= Average slope within a 25- or 50-m buffer around actual reservoir surface area at highest regulated water level; MSL = Maximum slope within a 25- or 50-m buffer around actual reservoir surface area at highest regulated water level

ILA compered to	**SILA(ASL25M)**	**SILA(MSL25M)**	**SILA(ASL50M)**	**SILA(MSL50M)**
Mann-Whitney U	16667	15351	16660	15233
Asymp. Sig. (2-tailed)	0.798	0.122	0.793	0.097

S6: Summary of the method

The inundated land area of a hydropower reservoir is the difference between the actual reservoir surface area at highest regulated water level and the waterbody surface area before dam construction. We received the actual reservoir surface area at highest regulated water level from NVE[1] and calculated the waterbody surface area before dam construction from remote sensing data.[5,9] By dividing the inundated land area with the annaul eletricty prodcution of each hydropower reservoir we cacluated site-specific net land occupation values for the Life Cycle Inventory. Whilst beyond the scope, the presented approach is a crucial step towards quantifying impacts of hydropower electricity production on biodiversity in LCA (Figure S9).

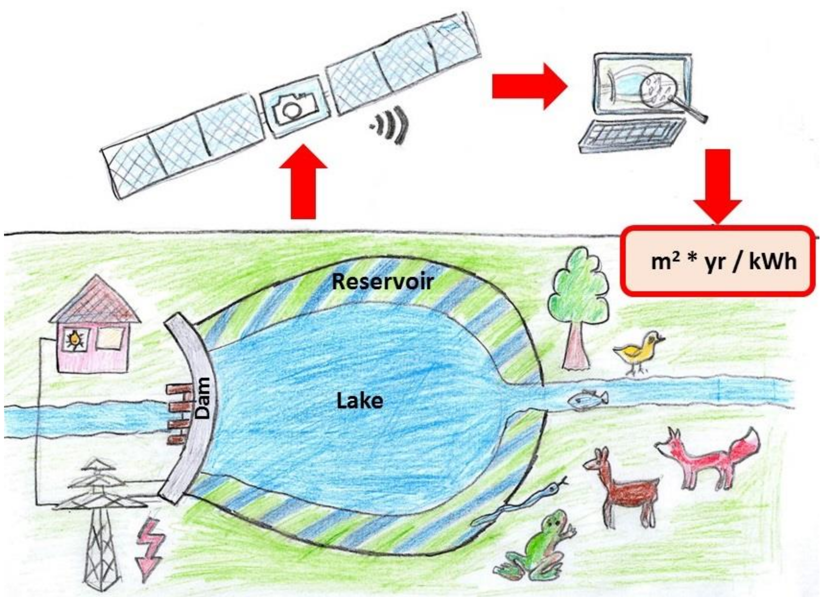


Figure S9: TOC-Art; Schematic visualization of the methodology in this study.

S7: References used within the supplementary materials

1. NVE (The Norwegian Water Resources and Energy Directorate). NVE Atlas (15.03.2016); http://atlas.nve.no.
2. The Ministry of Petroleum and Energy *Facts 2015 - Energy and Water Resources in Norway* 2015.
3. Rørslett, B., An integrated approach to hydropower impact assessment. I. Environmental features of some Norwegian hydro-electric lakes. *Hydrobiologia* **1988,** *164*, (1), 39-66.
4. Gutman, G.; Huang, C.; Chander, G.; Noojipady, P.; Masek, J. G., Assessment of the NASA–USGS Global Land Survey (GLS) datasets. *Remote Sensing of Environment* **2013,** *134*, 249-265.
5. USGS (United States Geological Survey). EarthExplorer-Global Land Survey 1975 dataset (01.05.2016); http://earthexplorer.usgs.gov/.
6. NASA-Landsat Science. The Worldwide Reference System (01.11.2016); http://landsat.gsfc.nasa.gov/the-worldwide-reference-system/.
7. USGS (United States Geological Survey). Path/Row Shapefiles-WRS1 https://landsat.usgs.gov/pathrow-shapefiles.
8. Thematic Mapping API. World Borders Dataset (11.04.2016); http://thematicmapping.org/downloads/world_borders.php.
9. Geodata AS. NORGE i Bilder - Image search (01.05.2016); https://www.norgeibilder.no/.
10. Sobek, S.; Nisell, J.; Fölster, J., Predicting the volume and depth of lakes from map-derived parameters. *Inland Waters* **2011,** *1*, (3), 177-184.
11. European Comission. EU DEM (LAEA) (18.03.2016); http://ec.europa.eu/eurostat/web/gisco/geodata/reference-data/elevation/eu-dem-laea.
12. IBM. SPSS Software (02.10.2017); https://www.ibm.com/analytics/data-science/predictive-analytics/spss-statistical-software.
13. MacFarland, T. W.; Yates, J. M., Mann–Whitney U Test. **2016**, 103-132.

7.2 Supporting information for Chapter 3

Quantifying *net* water consumption of Norwegian hydropower reservoirs and related aquatic biodiversity impacts in Life Cycle Assessment

Supporting Information 1

Martin Dorber[*, †], Kim Rainer Mattson[†], Odd Terje Sandlund [‡], Roel May [‡], Francesca Verones[†]

[†]Department of Energy and Process Engineering, Norwegian University of Science and Technology (NTNU), Sem Sælands vei 7, 7491 Trondheim, Norway

[‡]Norwegian Institute for Nature Research (NINA), Høgskoleringen 9, 7034 Trondheim, Norway

[*]Corresponding author e-mail: martin.dorber@ntnu.no

17 pages

7 figures

4 tables

Environmental Impact Assessment Review

Table of Contents

S1 List of acronyms and abbreviations

AET	Average actual evapotranspiration
CF	Characterization factor
EF	Effect Factor
ER	Average annual electricity production
FF	Fate Factor
ILA	Inundated land area
k	Number of reservoirs with inundated land data in catchment *x*
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact assessment
LO	Land occupation
NVE	Norwegian Water Resources and Energy Directorate
MOD16	MODIS Global Evapotranspiration Project
PET	Average yearly potential evapotranspiration
PDF	Potentially Disappeared Fractions of Species
RSA	Actual reservoir surface area at highest regulated water level
SDR	Species-discharge relationship

S2 Sensitivity and uncertainty levels in water consumption estimation

To account for uncertainty of evaporation estimates, we calculated a lower *net* water consumption with Eq. S1 and an upper *net* water consumption boundary with Eq. S2.

$$\text{Lower } net \text{ water consumption}_{AET\ Catchment\ x} = \frac{\frac{\sum_{y=0}^{k}\left(\left(\text{PET}_y\ - \text{AET}_Y * 1.246\right) \times \text{ILA}_y\ \right)}{1000}}{\sum_{y=0}^{k} \text{ER}_y} \tag{S1}$$

$$\text{Upper } net \text{ water consumption}_{AET\ Catchment\ x} = \frac{\frac{\sum_{y=0}^{k}\left((\text{PET}_Y - \text{AET}_Y \times * 0.774) \times \text{ILA}_Y\ \right)}{1000}}{\sum_{i=0}^{k} \text{ER}_y} \tag{S2}$$

Where *k* is the number of reservoirs with inundated land information in catchment *x*, *PET* is the average yearly potential evapotranspiration in mm/year of reservoir *y*, *AET* is the average actual evapotranspiration in mm/year of reservoir *y*, *ILA is* inundated land area due to the reservoir creation of reservoir *y* in m^2 and *ER* is the average annual electricity production in kWh of reservoir *y*.

Estimated water consumption uncertainty results of evaporation estimates are presented in Figure S1.

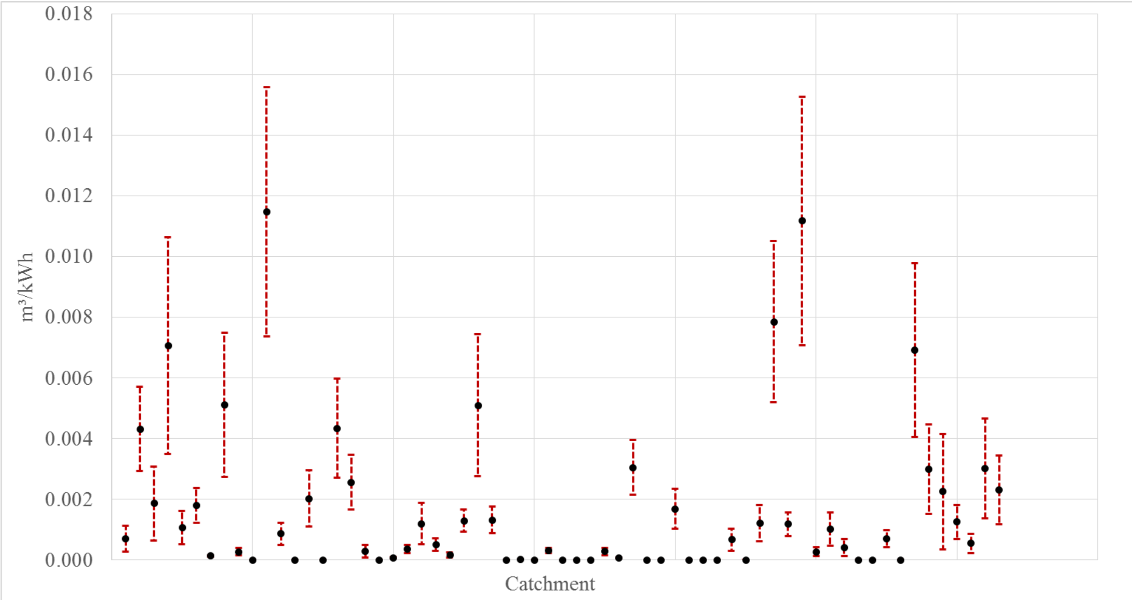


Figure S1: Visualization of the estimated *net* water consumption uncertainty (in red) due to reported evaporation estimation.

To account for uncertainty of inundated land area estimates, we calculated a lower *net* water consumption due to inundated land area with Eq. S3 and an upper *net* water consumption due to inundated land area with Eq. S4.

$$\text{Lower } net \text{ water consumption}_{ILA\ Catchment\ x} = \frac{\sum_{y=0}^{k} \frac{\left(\left(\mathrm{PET}_y - \mathrm{AET}_Y\right) \times \mathrm{ILA}_y - SD_y\right)}{1000}}{\sum_{y=0}^{k} \mathrm{ER}_y} \tag{S3}$$

$$\text{Upper } net \text{ water consumption}_{ILA\ Catchment\ x} = \frac{\frac{\sum_{y=0}^{k}\left(\left(\mathrm{PET}_y - \mathrm{AET}_y\right) \times \mathrm{ILA}_y + SD_y\right)}{1000}}{\sum_{y=0}^{k} \mathrm{ER}_y} \tag{S4}$$

Where k is the number of reservoirs with inundated land information in catchment x, PET is the average yearly potential evapotranspiration in mm/year of reservoir y, AET is the average actual evapotranspiration in mm/year of reservoir y, ILA *is* inundated land area due to the reservoir creation of reservoir y in m^2, SD is the standard deviation of the inundated land estimation of reservoir y in m^2 and ER is the average annual electricity production in kWh of reservoir y. Estimated water consumption uncertainty results of inundated land area estimates are presented in Figure S2.

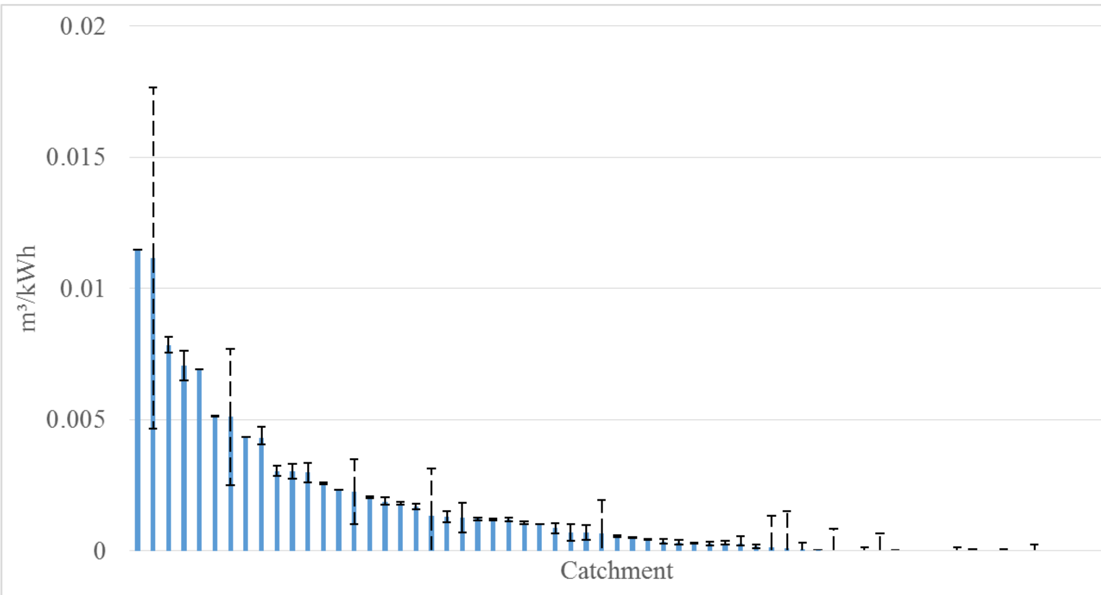


Figure S2: Visualization of the estimated *net* water consumption uncertainty in black, considering the standard deviation of the inundated land area.

The reason for catchments with zero uncertainty is that in these the inundated land area was estimated from aerial photographs. Consideration of the standard deviation could lead to a positive inundated land area value, even if the adjusted land inundated land area was previously set to zero by Dorber et. al,[1] due to a smaller adjusted water surface area before dam construction than water surface area before dam construction. This further explains why accounting for inundated land area uncertainty resulted in an average *net* water consumption due to inundated land area, where the upper and lower boundary did not vary equally (respectively -20.1% and 22.9%) relative to the average *net* water consumption.

Further, we tested the sensitivity of reducing the inundated land area by 1% on the calculated *net* water consumption values with Eq. S5.

$$net \text{ water consumption}_{Sensitivity\ Catchment\ x} = \frac{\frac{\sum_{y=0}^{k}\left(\left(\mathrm{PET}_y - \mathrm{AET}_y\right) \times \mathrm{ILA}_y \times \left(1 - \mathrm{RILA}_y\right)\right)}{1000}}{\sum_{y=0}^{k} \mathrm{ER}_y} \tag{S5}$$

Where k is the number of reservoirs with inundated land information in catchment x, *PET* is the average yearly potential evapotranspiration in mm/year of reservoir y, *AET* is the average actual evapotranspiration in mm/year of reservoir y, *ILA is* inundated land area due to the reservoir creation of reservoir y in m^2, *ER* is the average annual electricity production in kWh of reservoir y and *RILA* is the reduction of the inundated land area of reservoir y in %.

Reducing the inundated land area by 1% resulted in an average reduction of 0.000016 m^3/kWh, respectively 1% relative to the average *net* water consumption. We only calculated the sensitivity of reducing the inundated land area by 1% on the calculated *net* water consumption, as Eq. S5 shows that there is a linear relationship between the reduction of the inundated land area and the *net* water consumption value. Consequently, a reduction of the inundated land area by, for example

5%, would lead to an average reduction of 0.00008 m^3/kWh, respectively 5% relative to the average *net* water consumption.

S3 Identifying catchments with similar glaciation and dispersal history

We accounted for the colonization history of freshwater fish in Norway via the seas by selecting catchments by the marine ecoregion they drain into.[2] To account for colonization through surface waters in land masses, we selected catchments by the freshwater ecoregions they drain into.[3] Figure S3 is visualization the location of marine and freshwater ecoregions in Norway.

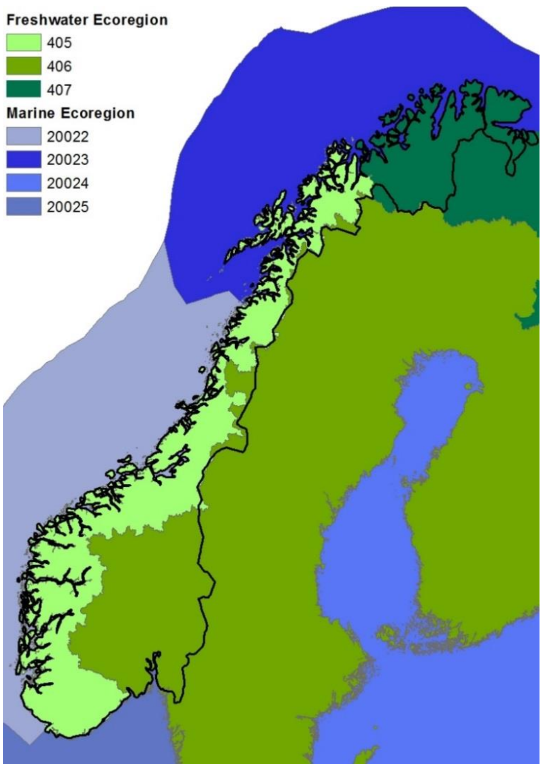


Figure S3: Map of Freshwater Ecoregions[3] (green colors) and Marine Ecoregions[2] (blue colors) of Norway.[4]

S4 Developing regional Species-discharge relationships for Norway

We excluded fish species classified as introduced from Fishbase[5] and include only the 31 fish species shown Table 1 in to calculate the Species-discharge relationships for Norway (Table S1).

Table S1 : Freshwater fish species from Fishbase[5] used to calculate the species discharge relationships for Norway.

Species	**FishBaseName**
Abramis brama	Freshwater bream
Acipenser sturio	Sturgeon
Anguilla anguilla	European eel
Blicca bjoerkna	White bream
Carassius carassius	Crucian carp
Coregonus albula	Vendace
Cottus gobio	Bullhead
Cottus poecilopus	Alpine bullhead
Esox lucius	Northern pike
Gasterosteus aculeatus	Three-spined stickleback
Gobio gobio	Gudgeon
Gymnocephalus cernuus	Ruffe
Lampetra fluviatilis	River lamprey
Lampetra planeri	European brook lamprey
Lethenteron camtschaticum	Arctic lamprey
Leuciscus aspius	Asp
Leuciscus idus	Ide
Liza aurata	Golden grey mullet
Lota lota	Burbot
Osmerus eperlanus	European smelt
Perca fluviatilis	European perch
Phoxinus phoxinus	Eurasian minnow
Pungitius pungitius	Ninespine stickleback
Rutilus rutilus	Roach
Salmo salar	Atlantic salmon
Salmo trutta	Sea trout
Salvelinus alpinus	Arctic char
Salvelinus salvelinoinsularis	Bear Island charr
Sander lucioperca	Pike-perch
Squalius cephalus	Chub
Thymallus thymallus	Grayling

Using this criteria in total 140311 occurrence points, covering 1463 catchment, were obtained through the publicly available database and map services Artsdatabanken[6] and GBIF[6-11] (Table S2).

Table S2: Sources of occurrence points and their collection year

Data Base	Occurrence Points	Collection Year
Artsdatabanken[6]	84001	1869 - 2017
GBIF: Gillnet test fishing and radiocaesium (Cs-137) of brown trout (Salmo trutta) and Arctic char (Salvelinus aplinus) 2008 and 2009 from 20 Central Scandinavian lakes[7]	4538	2008 - 2009
GBIF: Natural History Museum- Fish collection[9]	697	1844 - 2000
GBIF: National fish tag database[10]	18590	1938 - 2016
GBIF: Ims fish tag database[11]	31653	1966 - 2016
GBIF: Norwegian freshwater lake fish inventory[8]	822	2017

As reservoir construction in Norway began in 1800 and for example 662 out of 2297 reservoirs have been built before 1950,[12] we used all available occurrence points to develop the SDRs. As a result, the developed SDRs may underestimate the fish species richness because we cannot account for fish species that have gotten extinct before the earliest collection date of an occurrence point in the related catchment. Figure S4 shows that this underestimation is probably highest for region 1 and 3, as reservoir operation started earliest in these catchments. As a result, we do not have many occurrences points before reservoir operation in these regions, meaning the probability that we do not account for fish species that have gotten extinct before the earliest collection date of an occurrence point is highest in these region. As in region 5 and 7 reservoir operation started later, it is more likely that we were able to obtain occurrence points before reservoir operation. Therefore, the probability of an underestimation of fish species by the SDR is lower in these regions.

Further, we weighed the power function fitting by the total number of occurrence records in each catchment,[13] shown in Figure S5.

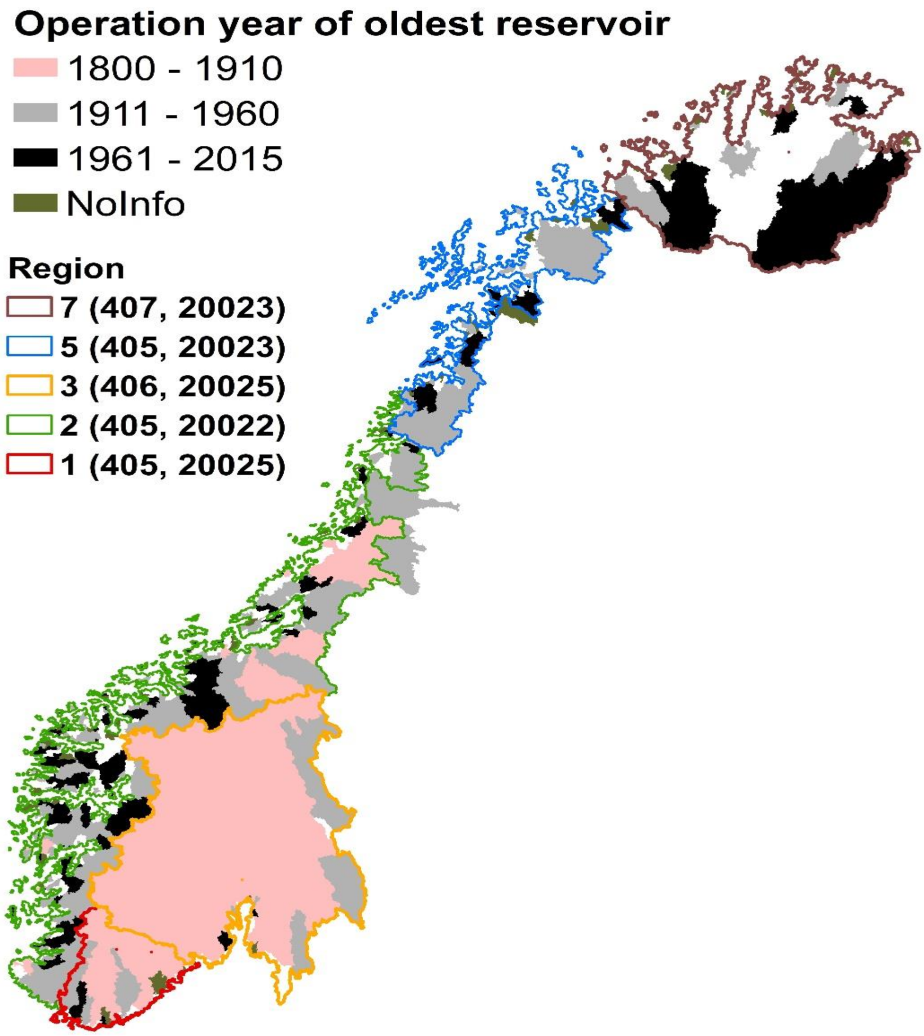


Figure S4 : Map showing the first operation year of the oldest reservoir in each catchment and in each of the five regions where we developed an SDR (3 digit number = Freshwater Ecoregion[3] code; 5 digit number = marine ecoregion[2] code). Catchment information and reservoir operation year obtained from the Norwegian Water Resources and Energy Directorate.[12]

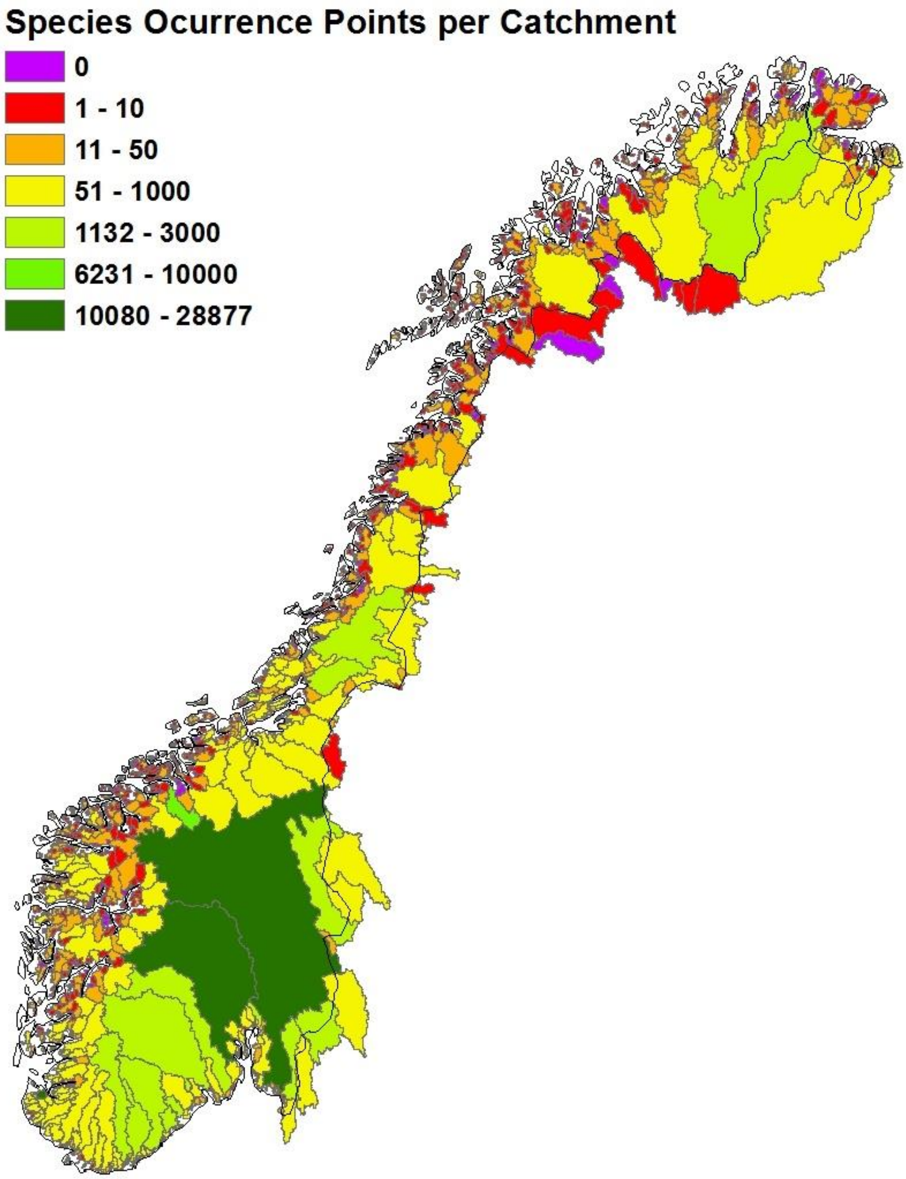


Figure S5: Number of obtained fish occurrence points per catchment. Catchment information obtained from the Norwegian Water Resources and Energy Directorate.[12]

S5 Species-discharge relationships comparison

Our results are in accordance with the statement from Hanafiah et al.[14] that the application of the current Species Discharge Relationships (SDR) would lead to overestimation of the impact of rivers above 42° degree north , due to the lower fish diversity reported for Norway in this study. To show the difference of regional developed SDRs we compared our SDR with the global SDR from Hanafiah et al.[14] and the Central Plains SDR from Tendall et al. (Figure S6).[15] We have chosen the Central Plains SDR, as it is the geographically closest developed SDR from Tendall et al.[15]

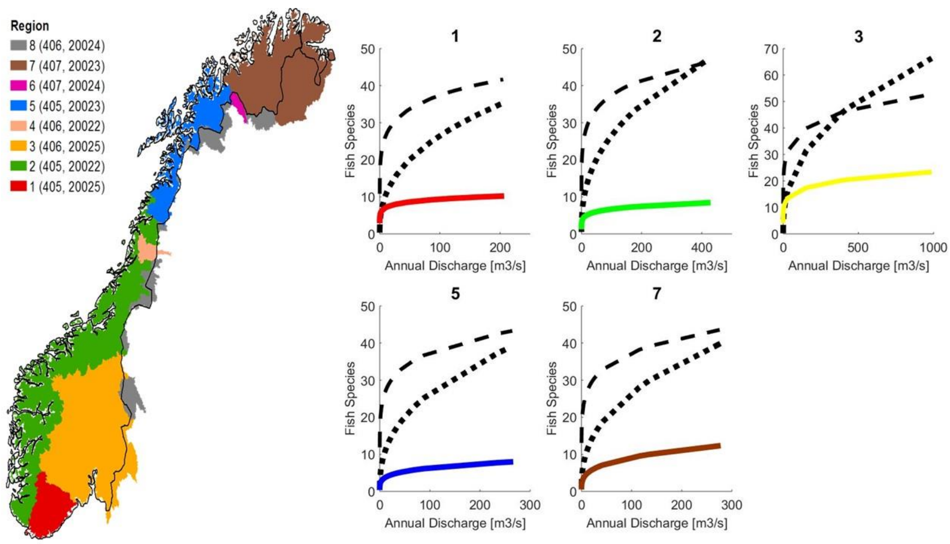


Figure S6: Left: Regions where catchments are draining into the same marine or freshwater ecoregion (3 digit number = freshwater ecoregion[3] code; 5 digit number = marine ecoregion[2] code); Right: Comparison of the SDR developed in this study (solid line) with the global SDR from Hanafiah et al.[14] (dotted line) and the Central Plains SDR from Tendall et al.[15] (dashed line).

S6 Characterization factor

We calculated CFs for 1790 of 1833 catchments in Norway varying between $7.1*10^{-12}$ PDF*y/m^3 and $8.0*10^{-7}$ PDF*y/m^3. The minimum and maximum CF of each region is presented in Table S3.

Table S3: Minimum and maximum Characterization Factors (CF) in each region

Region	Min CF	Max CF
1	$6.00*10^{-12}$	$8.36*10^{-07}$
2	$1.59*10^{-11}$	$7.66*10^{-07}$
3	$6.00*10^{-12}$	$2.78*10^{-07}$
5	$3.69*10^{-11}$	$1.52*10^{-07}$
7	$4.30*10^{-11}$	$1.00*10^{-07}$

To explain the pattern of the CFs in Figure 3, we calculated for each catchment *x* the species loss per m^3 water consumed with Eq. S6 and the fish species richness with Eq. S7. Figure S7 A shows that the relative species loss per m^3 water consumed does not differ much between small and large catchment. As the slope of the used SDR increases with low discharges (Figure 2), the derivative of the SDR power function for small rivers results in a comparable higher number of fish species lost per m^3/y of discharge reduction. This is explaining the pattern of Figure S7 A with higher species loss around the cost. However, the small catchments get the comparably higher PDF*y/m3 value, because they have a comparably lower fish species richness (Figure S7 B).

$$\frac{\text{species loss}}{\text{m}^3}_x = \frac{\text{dQ}}{\text{dW}} \times \frac{\text{dS}_x}{\text{dQ}}$$

(S6)

Where dQ is the marginal change in discharge [m^3/y], dW is the marginal change in water consumption [m^3/y], dS is the derivative of the SDR power function developed for the related region in Norway.

$$\mathrm{R}_x = a \times Q^b \tag{S7}$$

R is the fish species richness of catchment x, a and b are model coefficients produced by the regression model (Figure 2) and Q is the discharge [m^3/y] of catchment x.

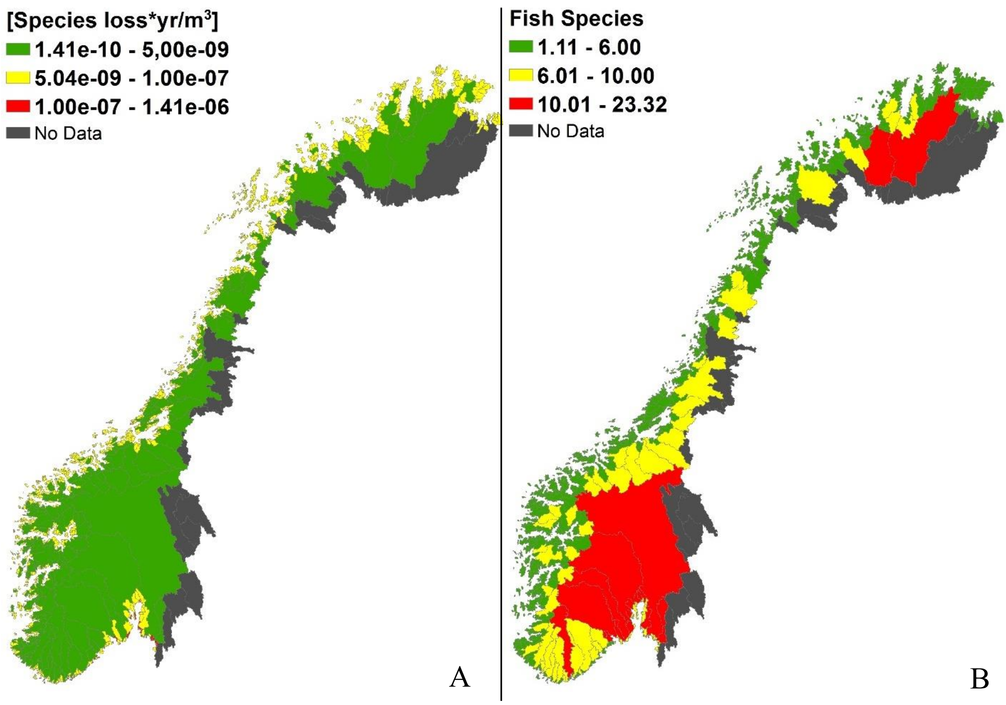


Figure S7: A: Species loss per m^3 water consumed; B: Number of fish species predicted by the Species-discharge relationship.

We used the 95% confidence intervals of the fitted power function coefficients in each region to quantify the uncertainty of the CFs. The obtained upper and lower power function coefficients in each region and the resulting minimum and maximum CF are presented in Table S4.

Table S4: Upper and lower bound power function coefficients, which were used to quantify the uncertainty of the characterization factors and the resulting minimum and maximum characterization factor in each region.

Region	Lower Bound Coefficient A	Upper Bound Coefficient A	Lower Bound Coefficient B	Upper Bound Coefficient B	Min CF [PDF*y/m^3],	Max CF [PDF*y/m^3],
1	0.31	1.33	0.16	0.09	$3.4*10^{-11}$	$1.2*10^{-07}$
2	0.18	0.25	0.17	0.15	$1.7*10^{-11}$	$7.7*10^{-07}$
3	0.25	0.99	0.19	0.13	$8.5*10^{-12}$	$2.8*10^{-07}$
5	0.02	0.06	0.26	0.22	$4.3*10^{-11}$	$1.5*10^{-07}$
7	0.01	0.03	0.32	0.27	$5.2*10^{-11}$	$1.0*10^{-07}$

S7 References used within the supplementary materials

1. Dorber, M.; May, R.; Verones, F., Modeling net land occupation of hydropower reservoirs in Norway for use in Life Cycle Assessment. *Environmental Science & Technology* **2018**.
2. Spalding, M. D.; Fox, H. E.; Allen, G. R.; Davidson, N.; FerdaÑA, Z. A.; Finlayson, M. A. X.; Halpern, B. S.; Jorge, M. A.; Lombana, A. L.; Lourie, S. A.; Martin, K. D.; McManus, E.; Molnar, J.; Recchia, C. A.; Robertson, J., Marine Ecoregions of the World: A Bioregionalization of Coastal and Shelf Areas. *BioScience* **2007,** *57*, (7), 573.
3. Abell, R.; Thieme, M. L.; Revenga, C.; Bryer, M.; Kottelat, M.; Bogutskaya, N.; Coad, B.; Mandrak, N.; Balderas, S. C.; Bussing, W.; Stiassny, M. L. J.; Skelton, P.; Allen, G. R.; Unmack, P.; Naseka, A.; Ng, R.; Sindorf, N.; Robertson, J.; Armijo, E.; Higgins, J. V.; Heibel, T. J.; Wikramanayake, E.; Olson, D.; López, H. L.; Reis, R. E.; Lundberg, J. G.; Sabaj Pérez, M. H.; Petry, P., Freshwater Ecoregions of the World: A New Map of Biogeographic Units for Freshwater Biodiversity Conservation. *BioScience* **2008,** *58*, (5), 403.
4. Thematic Mapping API World Borders Dataset. http://thematicmapping.org/downloads/world_borders.php (11.04.2016),
5. Froese, R.; Pualy, D. FishBase. www.fishbase.org. 2017.
6. The Norwegian Biodiversity Information Centre and GBIF Norway. Species Map Service 1.6. http://artskart.artsdatabanken.no/Default.aspx (3.04.2017).
7. Finstad, A. G.; Berg, O. K.; Ulvan, E.; Helland, I.; Ugedal, O., Gillnet test fishing and radiocaesium (Cs-137) of brown trout (Salmo trutta) and Arctic char (Salvelinus aplinus) 2008 and 2009 from 20 Central Scandinavian lakes: Occurrences. Version 1.2. NTNU University Museum. Occurrence Dataset. In 2017.
8. Hesthagen, T. F., Anders Gravbrøt, Norwegian freshwater lake fish inventory. Norwegian Institute for Nature Research. Sampling_event Dataset. In 2017.
9. Natural History Museum, University of Oslo, Fish collection, Natural History Museum, University of Oslo. Version 1.90. Occurrence Dataset. In 2017.
10. Vang, R., National fish tag database. Version 1.5. Norwegian Institute for Nature Research. Occurrence Dataset. In 2017.
11. Vang, R., Ims fish tag database. Version 1.2. Norwegian Institute for Nature Research. Occurrence Dataset. In 2017.
12. The Norwegian Water Resources and Energy Directorate. NVE Atlas. http://atlas.nve.no (15.03.2016).
13. Motulsky, H.; Christopoulos, A., *Fitting models to biological data using linear and nonlinear regression: a practical guide to curve fitting*. Oxford University Press: 2004.
14. Hanafiah, M. M.; Xenopoulos, M. A.; Pfister, S.; Leuven, R. S. E. W.; Huijbregts, M. A. J., Characterization Factors for Water Consumption and Greenhouse Gas Emissions Based on Freshwater Fish Species Extinction. *Environmental Science & Technology* **2011,** *45*, (12), 5272-5278.
15. Tendall, D. M.; Hellweg, S.; Pfister, S.; Huijbregts, M. A. J.; Gaillard, G., Impacts of River Water Consumption on Aquatic Biodiversity in Life Cycle Assessment—A Proposed Method, and a Case Study for Europe. *Environmental Science & Technology* **2014,** *48*, (6), 3236-3244.

7.3 Supporting information for Chapter 5

This information is not included in NTNU Open due to awaiting publication of the main article