NTNU – Trondheim
Norwegian University of
Science and Technology

Land Use and Land Use Change Impacts on Terrestrial Habitats from Hydropower Development in the LCA framework

Vilde Fluge Lillesund

Master in Industrial Ecology
Submission date: July 2014
Supervisor: Edgar Hertwich, EPT
Co-supervisor: Dagmar Hagen, NINA
Ottar Michelsen, EPT

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

NTNU
Norwegian University
of Science and Technology

Department of Energy
and Process Engineering

EPT-M-2014-63

MASTER THESIS

for

Student Vilde Fluge Lillesund

Spring 2014

Implementation of land use impacts on terrestrial habitats from hydropower production in a LCA framework

Implementering av effekter på terrestriske habitat av arealbruk fra vannkraftproduksjon i et LCA rammeverk

Background and objective

Land use and land use changes are regarded as the most severe threats to biodiversity at present. For decades there has been ongoing work to develop methods to incorporate this aspect in Life Cycle Assessments (LCA), but no consensus is reached. There is a common understanding that impacts must be assessed in three dimensions; time, area, and quality changes, but in particular the measurement of quality is debated, as well as the reference situation for this. Most proposals are based on species richness and/or ecosystem vulnerability/scarcity.
The mitigation hierarchy is gaining more attention and is designed to be a tool for minimizing environmental impacts of projects, e.g. hydropower production. It involves both onsite actions as avoiding and minimizing impact and restoring, as well as offsite actions, in particular offsetting. This can, at least in theory, give a net gain.
So far no attempt has been undertaken to link these two approaches. The aim of this project is to assess the impact of a hydropower project on biodiversity due to land use and land use change, through different approaches. The impact is to be quantified following different quality assessments where pros and cons of the different quality measures must be discussed; data requirement, ecological validity, accuracy etc.

The following tasks are to be considered:

1 Describe the case in Mandal water system, including the present impact and the planned development.
2 Identify other planned hydropower projects that can be assessed for comparison. Criteria for selection of additional cases must be developed.
3 A detailed inventory of the terrestrial habitat changes due to these projects
4 Briefly describe relevant existing methods for assessing land use impacts on biodiversity in LCA and, where possible, use these to quantify the impacts from the included cases
5 Based on available habitat data and the mitigation hierarchy, use this to propose new approaches for evaluation land use impacts on biodiversity in LCA. Restoration costs should be included, but also loss/gain of nature types, biotopes for species of particular interest, selected ecosystem services, etc. should be considered.

6 Discuss the results based on the different approaches. Discuss potential deviation in results; how (biological) relevant and robust are the different approaches.

-- " --

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

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

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

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

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

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

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

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

Department of Energy and Process Engineering, 14. January 2014

Olav Bolland
Department Head

Edgar Hertwich
Academic Supervisor

Research Advisor: Ottar Michelsen, NTNU og Dagmar Hagen, NINA

Preface

This thesis is part of the Center for Environmental Design of Renewable Energy (CEDREN) project '*EcoManage – Improved development and management of energy and water resources*' and belongs to work package 4 "Ecosystem services and biodiversity offsets", which is lead by David N. Barton (NINA). The work package description can be found in Appendix 1.

I would like to thank my supervisor Edgar Hertwich and co-supervisors Ottar Michelsen and Dagmar Hagen for their great guidance, advice and invaluable inspiration. I would also like to thank Anders Foldvik (NINA), without whom the mapping of the land use change would have been difficult.

I am forever grateful to my family and friends for their support, the Industrial Ecology Program for the opportunity to learn, and my classmates for sharing their knowledge. I would especially like to thank my favorite "Englisher" M Samantha Peverill, the best childhood friend I met 20 years late, for her loving presence and invaluable help, and my "family" in Guttormsgate, Eivind and Sigmund. Last but not least, Ine Vabø, for keeping me on my toes.

ABSTRACT
Anthropogenically driven biodiversity loss is one of the major challenges of modern society. The main cause of biodiversity loss is habitat fragmentation and destruction. Hydropower causes transformation and occupation of large water systems and areas of land for infrastructure and reservoirs. The impacts of habitat change and occupation on biodiversity are incorporated into Life Cycle Assessment (LCA) the impact category 'Land Use and Land Use Change' (LULUC). Despite decades of effort there is still no consensus on the proper methodology for LULUC.

The mitigation hierarchy has been designed to be an instrument for minimizing environmental impacts of development projects. The mitigation hierarchy framework has four steps; (1) avoid impacts, (2) minimize impacts, (3) restore impacts on-site, and (4) offset impacts by restoring, preserving, enhancing and/or establishing ecosystems off-site. The mitigation hierarchy and LCA share a common need for a suitable indicator for biodiversity/ecosystem quality.

Within the LCA framework for LULUC four hydropower development cases were analyzed, it was proposed to use three different approaches for measuring impacts to ecosystem/biodiversity quality; indirect indicators representing the current condition of the ecosystem affected, carbon emissions from the affected areas, and, combining the mitigation hierarchy and LCA, using the cost of restoration of the affected areas.

The three approaches provided similar results for all case projects, and were also consistent with basic total land use per kWh. Impacts to wetland were larger than impacts to other ecosystems, for all approaches. Carbon emissions from land use change related to the construction of infrastructure underlined the importance of including total land use change per kWh when assessing hydropower, especially for small-scale hydropower development.

Incorporation of applied restoration ecology in LCA through the mitigation hierarchy is possible and restoration cost performs equally well as the other indicators, but there are uncertainties concerning the absolute impact result, and the methodology should be further developed to address the issues.

SAMMENDRAG

Menneskeskapt tap av biologisk mangfold er en av de store utfordringene i dagens samfunn. Den viktigste årsaken til tap av biologisk mangfold er habitatfragmentering og habitatødeleggelse. Infrastruktur og reservoarer forbundet med vannkraftproduksjon påvirker vassdrag og landområder gjennom store arealbruksendringer og stort arealbruk. Disse påvirkningene er innarbeidet i livssyklusanalyse (LCA) i kategorien 'arealbruk og arealbruksendringer"(LULUC), men til tross for flere tiår med metodeutvikling er det fremdeles ikke enighet om riktig metodikk for LULUC.

Rammeverket 'The Mitigation Hierarchy' er utviklet som et instrument for å minimere negative miljøkonsekvenser forbundet med utbyggingsprosjekter. Rammeverket har fire trinn; (1) unngå naturinngrep, (2) minimere naturinngrep, (3) restaurere naturinngrep på byggestedet, og (4) kompensere for inngrep ved å restaurere, bevare og/eller etablere økosystemer i et annet område enn det opprinnelig berørte. 'The Mitigation Hierarchy' og LCA deler et felles behov for en egnet indikator for biodiversitet.

Miljøpåvirkningene til fire vannkraftutbyggingskasus ble vurdert innenfor LCA-rammeverket for LULUC og grunnet uenigheten rundt metodikken ble det foreslått å bruke tre ulike indikatorer for å måle påvirkningene på økosystemer/biologisk mangfold; indirekte indikatorer som representerer den nåværende tilstanden i økosystemene som ble påvirket, karbonutslipp fra de berørte områdene og, i en kombinasjon av 'The Mitigation Hierarchy' og LCA, restaureringskostnad for de berørte områdene.

De tre tilnærmingene gav lignende resultater for alle kausene, og var også i samsvar med total arealbruksendring per kWh. Våtmarksinngrepene var større enn inngrepene på andre økosystemer for alle tilnærmingene. Karbonutslipp fra arealbruksendringer knyttet til bygging av infrastruktur understreket viktigheten av å inkludere total arealbruksendring per kWh ved vurdering av vannkraft, spesielt for småskala vannkraftutbygging.

Inkorporering av anvendt restaureringsøkologi i LCA gjennom 'The Mitigation Hierarchy' er mulig og restaureringskostnad tjente tilsvarende godt som indikator for biodiversitet som de andre indikatorene. Det er likevel en del usikkerhet forbundet med metoden og den bør videreutvikles for å håndtere disse problemene.

TABLE OF CONTENT

TABLE OF TABLES

TABLE OF FIGURES

ABBREVIATIONS

C – Carbon
CBD – Convention on Biological Diversity
CMB – Conditions for Maintained Biodiversity
CO_2e – CO_2-equivatents
ES – Ecosystem scarcity
EV – Ecosystem Vulnerability
FU – Functional Unit
GHG – Green House Gas
GWh – Giga Watt hour
KF – key factor
kWh – kilo Watt hour
LCA – Life Cycle Assessment
LU – Land Use
LUC – Land Use Change
LULUC – Land Use and Land Use Change
MA – Millennium Ecosystem Assessment
NNI – Norwegian Nature Index
NVE – Norwegian Water Resource and Energy Directorat
OI – Occupation Impact
PIC – Permanent Infrastructure Construction
Q – Ecosystem Quality
ROR – run-of-river (hydropower)
SER – Society for Ecological Restoration
SETAC – The Society of Environmental Toxicology and Chemistry
TI – Transformation Impact
TIC – Temporary Infrastructure Construction
TWh – Terra Watt hour
UNEP – United Nations Environmental Program
yr – year

1 INTRODUCTION

1.1 BACKGROUND

Anthropogenically driven biodiversity loss is one of the major challenges of modern society. Current rates of biodiversity loss are well beyond the planetary boundary identified by Rockström et al. (2009) and it is suggested that we might be in the middle of a sixth mass extinction of species (Barnosky et al. 2011). The Millennium Ecosystem Assessment (2005) found that habitat change, climate change, pollution, invasive species and overexploitation of wild populations are the five most important threats to biodiversity, and of these habitat change is the most severe. Transformation of natural land into agricultural land and fragmentation of previously continuous ecosystems are currently the dominant causes of habitat change, and also multiple minor changes will have a cumulative effect.

Hydropower is a major source of electricity. Globally, hydropower provided 3700 TWh of electricity in 2012, making up 16,5% of global electricity supply and 3,7% of global energy supply. Norway produced 143 TWh in 2012, making Norway the sixth largest hydropower producer, worldwide (REN21 2013). Hydropower causes transformation and occupation of large water systems and areas of land for infrastructure and reservoirs. Approximately 70% of Norwegian watersheds are currently affected by hydropower production (Miljødirektoratet 2013). Further development of hydropower is expected to satisfy growing energy demand, but the associated biodiversity loss associated with the development is unclear.

Life Cycle Assessment (LCA) identifies and assesses the environmental impacts of product and service systems (ISO 14040 2006). Measures of the impacts of habitat change and occupation on biodiversity are incorporated into the impact category 'Land Use and Land Use Change' (LULUC). Despite decades of effort there is still no consensus on the proper methodology for LULUC (Milà i Canals et al. 2007). The result of the lack of consensus is that LCAs of products and services that obviously cause land use change or occupy land often do not include LULUC, e.g. only 9% of assessments on bioenergy systems include LULUC (Cherubini & Strømman 2011). A resent statistical data gap analysis for Life Cycle Inventory for hydropower left out biodiversity loss all together referring to conflicting views as a reason (Moreau et al. 2012).

The reason for the lack of consensus on LULUC in LCA is likely the width and complexity of the concept of biodiversity (Michelsen 2008). The Convention on Biological Diversity (CBD) defines biological diversity as '*the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and among ecosystems*' (UNEP 1992).

Actions to preserve biodiversity and prevent further loss have become widespread following increased awareness of the consequences of habitat destruction. Restoration ecology offers a significant contribution to restoring lost biodiversity. Restoration ecology is defined as 'the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed' (SER 2004) and is emerging, along with conservation of remaining habitats, as one of the most important tools for maintaining biodiversity levels. Restoration ecology focuses on restoring both the abiotic environment and populations. The species and their interactions in a restored system provide crucial services, and restoration efforts may therefore be undertaken to restore ecosystem services (Bullock et al. 2011). The Aichi Targets developed by the CBD include restoration as an important instrument to enhance the benefits from biodiversity end ecosystem services and aims to restore 15% of damaged habitats before 2020 (CBD 2010).

The opportunity to restore habitats directly affected by hydropower will only be available when the hydropower production is terminated. Hydropower plants and the associated reservoirs commonly have long predicted lifetimes (100 years) and the restoration efforts cannot be undertaken in a reasonable timeframe. The mitigation hierarchy has been introduced as a concept in restoration ecology to facilitate implementation of restoration considerations in development projects. The mitigation hierarchy is a four-step concept that aims to avoid, minimize, restoring on-site and offset damage to ecosystems (McKenney & Kiesecker 2010). It is particularly the last two steps of restoring habitats on-site and offsetting that directly links to restoration ecology. Offsetting damage means to compensate for on-site impacts by restoring previous impacts off-site. In total, implementation of the mitigation hierarchy can, in theory, give a net gain to biodiversity. The mitigation hierarchy has the same challenge as LCA, as they share the need for a good measure for biodiversity, but the approaches have yet to be combined.

1.2 LCA – Calculating Impact of Land Use and Land Use Change

The United Nations Environmental Program (UNEP) and The Society of Environmental Toxicology and Chemistry (SETAC) have published the guideline for calculating impacts of Land Use (LU) and Land Use Change (LUC) (Koellner et al. 2013). The guideline (Koellner et al. 2013) distinguishes between two types of land use. The first is land transformation (land use change) where the properties of the area are actively changed to facilitate a different use. The construction of a dam and the subsequent formation of a reservoir is an example of such a land transformation, where natural land is altered for the purpose of hydropower. The second type of land use is land occupation (land use) where the conditions and quality of the land is actively kept in the state required for the intended LU. In relation to hydropower, the land occupation occurs during the lifetime of the hydropower when the dam is maintained and operated to ensure that the reservoir can deliver water for electricity production. The total impact to any ecosystem is calculated by using three key dimensions; A – area occupied or transformed, measured in meters squared or cubed, T – time of occupation or reversal to reference state, measured in years, and Q – ecosystem quality. A conceptual illustration can be seen in Figure 1.

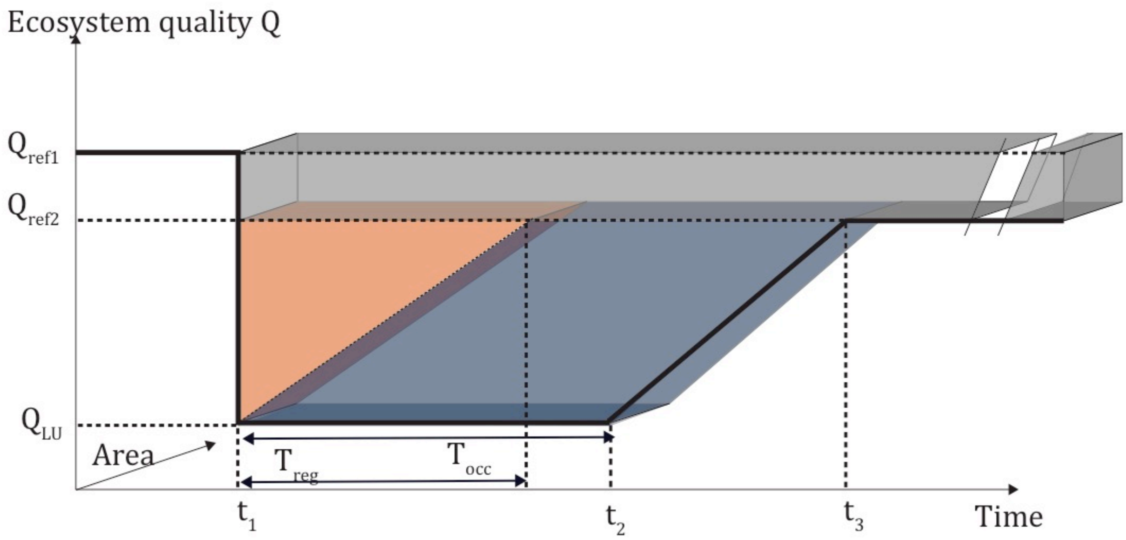


Figure 1: Conceptual illustration of the impact of Land Use (blue area) and Land Use Change (orange area). Transformation is for modeling purposes instant and the quality during occupation constant. When occupation ends the quality will with time linearly revert to a reference state, which may or may not be reached. The system can stabilize in a state of lower quality than the original reference state. If so, there will be permanent impact to the system (the grey area between Q_{ref1} and Q_{ref2}).

Occupation impacts (OI) are calculated using the area occupied, a relative difference in Q and the time of occupation T_{occ}. The relative difference, ΔQ, is the difference in quality between the reference state and the land use state. The occupation time T is the time between the initial transformation (t_1) and either start of regression back to the reference state (t_2) or a new transformation occurs.

$$OI = \Delta Q * T * A \qquad (1)$$

Transformation impacts (TI) are the impacts that occur as a result of the land transformation. If the system would be allowed to return to its natural state after transformation the impact is equal to the reduced quality over that time. ΔQ and A therefore stay the same but T is changed to T_{reg}, which is the restoration time of the area affected. The restoration time will be dependent on the type of area in question.

$$TI = 0.5 * \Delta Q * T_{reg} * A \qquad (2)$$

The relation between quality, area and time in the model allows for a trade-off between the factors. An increase in ΔQ can be compensated for by a decrease in either one or both A and T. In more practical terms, the method allows for reduction of calculated transformation impact by reducing regeneration time by active restoration. Decreasing area occupied or the occupation time can reduce occupation impact.

The restoration time is dependent on type of ecosystem. Different ecosystems have their respective restoration times, and these are based on research in restoration ecology. For example, the restoration in alpine ecosystems is slower than in lowland ecosystems due to cold climactic conditions and a limited growth season. The difficult conditions also require a high degree of species specialization in order to survive and colonization of disturbed habitats will be slow due to the lack of nearby species able that have the necessary traits. Both of these factors will contribute to slow natural succession (Hagen et al. 2014).

Quality is defined in the UNEP-SETAC guideline as '*the capability of an ecosystem (or a mix of ecosystems at a landscape scale) to sustain biodiversity and to deliver services to the human society*' and '*may be measured with different indicators expressing the intrinsic value of biodiversity and natural landscapes of functional value of ecosystems in terms of their goods (i.e. natural resources like timber or food) and services (i.e. life support functions like climate regulation or erosion regulation)*' (Koellner et al. 2013). As both biodiversity and quality has such wide definitions the need for indicators with biological relevance and robustness is evident. Curran et al. (2011) identified that most indicators for Q in LCA are measures of biological composition – the quantity and variety of elements. Indicators of biological structure and function are rare, meaning that the ecological and evolutionary processes among the elements and the physical organization of them are largely missing in LCA. The indicators are also largely measured for species and communities, leaving out hierarchical components on genetic and ecosystem/landscape level. Species richness is the most common indicator for ecosystem quality in LCA (Michelsen 2008), but it is only one of the available measures, covering only part of the concept of biodiversity and sometimes even misrepresents the occurrence of different types of biodiversity (Curran et al. 2011).

Species richness is a seemingly straightforward way of quantifying biodiversity, but there are several limitations with this measure. First, there is the issue of having sufficient sampling sizes in order to find a representative species richness for an area, as most species are rare (Gotelli & Colwell 2001; McGill et al. 2007). Spatial heterogeneity in species occurrence is also a complication when measuring species richness. For example, species richness is often higher in newly disturbed areas, early successional stages and edge zones, than in the undisturbed surrounding habitat (Penariol & Madi-Ravazzi 2013). Also, easily detectable species groups like vascular plants, mammals and birds tend to be over-represented when species richness is measured, while much more species rich groups like insects, bryophytes and fungi are only rarely covered (Wolters et al. 2006). Yet another important factor is that the abundance of species is not included in species richness, but may still be important for ecosystem function (Walker et al. 1999). Lastly, valuing an area based on species richness indicated that species rich areas are intrinsically more valuable than species poor areas, and this form of valuation of nature is problematic, at best.

There is debate on how to determine an appropriate reference state for ecosystem quality. One option is to use the expected state of mature vegetation in the absence of human intervention (Potential Natural Vegetation- PNV) or the natural mix of ecosystems, but it is difficult to quantify and model due to ecosystem dynamics and might allocate past damage to present LULUC (Chiarucci et al. 2010). A third option is to use the current mix of land uses as a reference, and thus disregarding all past impacts ad avoiding allocating these to present LULUC. There might however, be an extinction debt due to past development (Wearn et al. 2012) and this should be taken into consideration. On a regional level there is an option to actually model the current state of the ecosystem. The Norwegian Nature Index (NNI (Nybø 2010)) offers national and regional indices on the state of the major ecosystems in Norway for 1950, 1990, 2000 and 2010. The indices are based on 309 individual indicators spread over the major ecosystem types. Each indicator has been given a reference value and the major ecosystem indices are based on deviation from these reference values.

1.3 MITIGATION HIERARCHY AND RESTORATION ECOLOGY

The mitigation hierarchy has been designed to be an instrument for minimizing environmental impacts of development projects. The framework first emerged in US wetland ecosystems mitigation and has later been adopted in Australia, the European Union, Colombia, Brazil and the UK, among others (McKenney & Kiesecker 2010; Saenz et al. 2013). A recent report from the Norwegian Ministry of Transport and Communication (Samferdselsdepartementet 2013) has investigated the possibility of incorporating the mitigation hierarchy in the planning process for road construction in Norway.

The mitigation hierarchy framework has four steps; (1) avoid impacts, (2) minimize impacts, (3) restore impacts on-site, and (4) offset impacts by restoring, preserving, enhancing and/or establishing ecosystems off-site, (Figure 4: Quintero & Mathur 2011; McKenney & Kiesecker 2010; Saenz et al. 2013). Avoiding impacts is done by situating development projects where they make the least impact to the environment, and should be strived for "to the maximum extent possible, considering cost, existing technology and practical feasibility" (US EPA as read in McKenney & Kiesecker 2010).

Any impact that cannot be avoided should then, in step 2, be minimized considering time, space and intensity, e.g. time of occupation, pollution of air, noise etc. The Norwegian Water Regulation Act prohibits hydropower development in protected rivers and currently 391 water systems, equivalent to 45TWh, are protected by this act. The act further proposes to protect water system that could produce an additional 2,15 TWh every year (St.prp. nr.53, 2008-2009). The water systems are distributed across the whole country and protect water systems that have particular natural value.

These first two steps of the mitigation hierarchy are connected to the planning of a development project, while the last two steps are designed to compensate for damage done during development. Step 3 is restoration of habitat on-site. Wherever possible, the project area should be restored back to, or as close as possible to, the state it was in before commencing the project or to its natural state, depending on the initial condition. The construction phase of a development project often requires more area than the occupation phase. Careful restoration with natural vegetation will lead to smaller impacts, albeit delayed in time, and will prevent vulnerability to invasive species (e.g. Hagen & Evju 2013).

The fourth step of the mitigation hierarchy is the offsetting stage. Biodiversity offsetting is defined as *'… measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been* *taken. The goal of biodiversity offsets is to achieve no net loss and preferably a net gain of biodiversity on the ground with respect to species composition, habitat structure, ecosystem function and people's use and cultural values associated with biodiversity*' (Business and Biodiversity Program 2013). Offsetting thus involves restoration efforts off-site to maintain or enhance the environmental value of an area wherein the development project is situated. McKenney & Kiesecker (2010) identify some key issues with the mitigation hierarchy, of which some will be elaborated below.

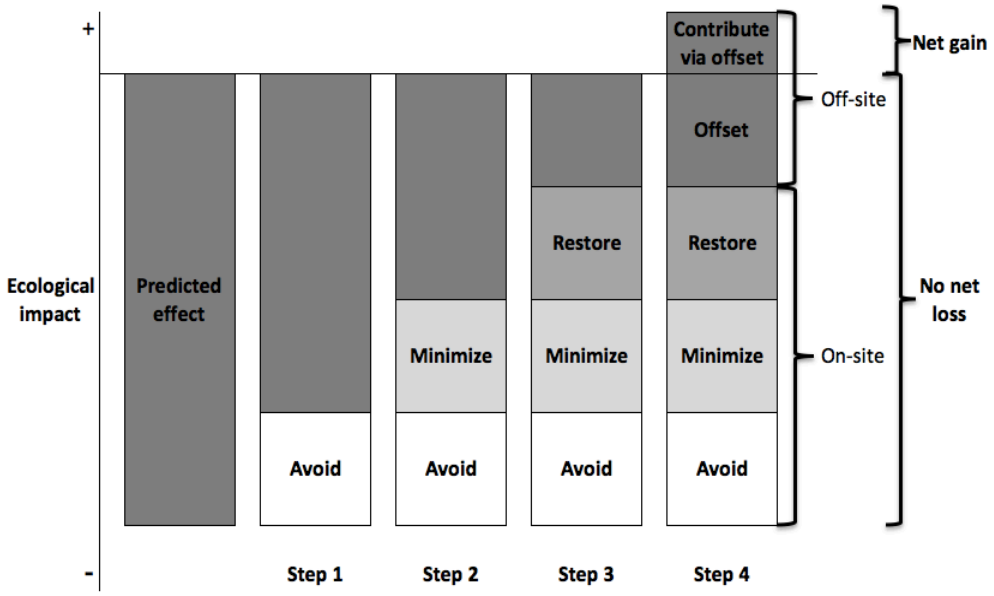


Figure 2: The Mitigation hierarchy – stages leading to a net positive ecological impact (Quintero & Mathur 2011; Business and Biodiversity Program 2013)

A critical issue with the mitigation hierarchy is how impact is measured. There is no single preferred indicator in current offset policies, but the indicator should reflects the value associated with ecological function, quality, and integrity (McKenney & Kiesecker 2010). Establishing an indicator can be difficult and in some cases the consequence is that only the basic land use is taken into account with no other measures of biodiversity, thus leaving out important qualities from the indicator. Failing to establish a good indicator for ecological impact makes it difficult to determine both the extent of the damage of a development project and the success of on-site restoration and offset efforts. As a precaution an offset should always be in-kind, meaning that when a particular ecosystem is affected by anthropogenic development an ecosystem of the same kind should be restored (The Environment Bank 2013; Business and Biodiversity Program 2013). Also, perpetual offsets are preferred to finite offsets. Finite offsets assume that project impacts at the site of development are reversible, which is to say that the area will return to its previous state when the occupation ends. Perpetuating the offset after occupation ends assumes that the impact is permanent.

Offsets replacement ratios are implemented to compensate for the uncertainties related to restoration and offset success. The temporal lag of the offset relative to impact, risk of restoration failure, and offset location relative to the affected site are examples of such uncertainties, in addition to the equivalence of the offset (in-kind/out-of kind) and the perpetuity of the offset. Replacement ratios are established with an assessment of the national and local level of ecosystem rarity, percentage of remaining ecosystems relative to historic distribution, and the rate of loss over the previous years. The highest ratios are given to both severely disturbed areas to stem further loss and close to natural ecosystems to encourage continued conservation of highly intact areas. McKenney and Kiesecker (2010) have found that replacement ratios in offset policies vary, but that they are normally set to fixed ratios, e.g. 1:1, 1:2 or 1:5 depending on the type of offsetting practice. However, Saenz et al. (2013) have determined that higher ratios (1:7-10) might be necessary.

Saenz et al. (2013) suggest that the mapping should be done in the early stage of planning the development project so that the currency can also be used in the early stages of the mitigation hierarchy to situate the project, and facilitate avoidance and mitigation. Starting with the mapping can be beneficial in several ways. It helps with finding a suitable site for the project by identifying areas where the potential damage is the smallest, and at the same time allows for identification of a suitable offset site near to the damage. By coordinating with existing conservation measurements one can ensure that the restoration and offset is coordinated with broader landscape conservation considerations, and early identification can promote offset startup sooner.

Restoration of disturbed, degraded or damaged ecosystem, whether it is done on-site or off-site, involves actively implementing restoration measures. Restoration projects can therefore be associated with cost, and this cost will be the cost of maintaining ecosystem services.

1.4 Ecosystem services

Ecosystem services became an important focus area for environmental research after the Millennium Ecosystem Assessment (MA) was published (2005). It described human dependence on natural systems, unsustainable resource use and the resulting destructive effect on biodiversity and natural ecosystems, decreasing nature's resilience and robustness. The degradation decreases nature's ability to deliver these services and the term ecosystem service was used to underline that nature, in addition to having intrinsic value, preforms services with direct and indirect value for society. Of 24 defined ecosystem services 15 were in decline. There are four main categories of ecosystem services:

- Supporting services – Services necessary for the production of all other ecosystem services: soil formation, nutrient cycling and primary production
- Provisioning services – Products obtained from ecosystems: food, freshwater, fuel wood, fiber, biochemical, genetic resources
- Regulatory services – Benefits obtained from regulation of ecosystem processes: climate, disease and water regulation, water purification and pollination
- Cultural services – Nonmaterial benefits obtained from ecosystems: Spiritual and religious, recreation and ecotourism, aesthetic, inspirational, educational, sense of place, cultural heritage.

The climate regulating abilities of ecosystems have become a focus in light of global warming and climate change. Ecosystems influence climate both locally and globally. At a local scale, changes in land cover can affect both temperature and precipitation. At a global scale ecosystems play an important role in climate by sequestering or emitting green-house gasses (GHG). Globally, carbon stocks in soil and biomass are >3 times the size of the carbon stores on the atmosphere. When natural systems are disturbed the carbon is released to the surrounding environmental compartments.

Hydropower is often depicted as a carbon poor energy source with reported GHG emissions between 2-40 gCO_2e/kWh, but the actual climate benefit of hydropower as opposed to more carbon intensive fuel sources is poorly understood due to biogenic GHG emissions, changes in albedo and increased evaporation rates from reservoirs. Biogenic green house gas emissions are often left out of LCA (Hertwich 2013), and when they are included, only biogenic GHG emissions from reservoirs are included, excluding emission from terrestrial LUC (Houghton et al. 2012).

Following up on the MA, The Economics of Ecosystems and Biodiversity (TEEB) project published reports on the economic consequences of degradation of ecosystem services (TEEB 2010). Lower service levels will increasingly cause economical and societal consequences, causing the greatest impact in developing countries where they, to a greater extent than in developed countries, directly rely on ecosystem services. One of the main causes of the unsustainably high consume of ecosystem services is that the cost of the service is perceived as very low. There is little or no consideration of resource scarcity and nature as a finite resource when production and consumption decisions are made, which makes ecosystem services an externality. One way of internalizing the value of an ecosystem is to put a price on the services it provides, where possible. However, some ecosystem services are difficult to relate to monetary values.

Numbers on cost of ecosystem degradation can help incorporate ecosystem services in our economic system, systems of national accounting, tax systems, laws and regulation.

1.5 Aim of the Study

The aim of this thesis is to assess the impact of hydropower on biodiversity due to land use and land use change. Due to the discord on correct methodology for LULUC in LCA three different approaches were used to indicate biodiversity/ecosystem quality:

- ecosystem scarcity/vulnerability as indirect indicators to represent the current conditions of the ecosystems affected
- biogenic GHG emissions to represent reduction of ecosystem services
- the cost of restoring the affected habitats, in the context of the mitigation hierarchy

Further, the following topics were investigated:

- How do the results from the different methods compare and what are the data requirements, validity and accuracy of the alternative measures for quality?
- Can the incorporation of applied restoration ecology contribute to the implementation of LULUC in future LCAs?

2 MATERIAL AND METHODS

2.1 CASE STUDIES

Four hydropower cases were selected to test the methods. The case projects have all been given, or is in the process of applying for, a permit to expand current production or establish new production. The Norwegian Water Resource and Energy Directorate (NVE) grants the permits and the permit applications are publicly available at www.nve.no. The main case, Skjerkevatn, was selected in cooperation with the hydropower company Agder Energi for its size and representativeness for the required upgrading of the aging stock of Norwegian hydropower plants.

As LULUC methodology is not consistent and restoration cost and LCA has not previously been combined, inclusion of additional cases was needed to assess the validity of the results. Three additional cases were selected with following criteria to ensure consistency and facilitate evaluation of method validity:

1. The hydropower development project should be in the process of applying for or have been given a permit, but not yet started the construction phase. This ensures data availability for both the "before" and ""after" land use change, in the form of current maps and technical drawings, respectively.
2. The hydropower development should be located within the same NNI region as Skjerkevatn, which is Southern Norway (Vest-Agder, Aust-Agder, Telemark and Vestfold Counties). Norwegian regions are highly variable in both geography and biodiversity and comparison of cases in the same region would be more valid due to the minimization of this natural variation and the availability of ecological data for the region.
3. The predicted mean annual production capacity of the planned hydropower project should be close to that of Skjerkevatn – 43 GWh/yr. This will enable the comparison of the impact per energy unit produced for the individual cases. Energy, measured in kWh, is the functional unit (FU) of the hydropower plant.

Table 1 provides key information, and Figure 2 a map of the region and the location of the selected cases. For technical drawings for each case see Appendix 2.

Table 1: Key information for the four selected case projects.

Attributes	Skjerkevatn	Langevatn	Dvergfossen	Kilandsfossen
Location	Åseral, Vest-Agder	Åseral, Vest-Agder	Kvinesdal, Vest-Agder	Åmli/Froland, Aust-Agder
Type of hydropower	Storage	Storage	Run-of-river	Run-of-river
Age	81	62	New	New
Expected yearly production from new development	43 GWh	18 GWh	35,5 GWh	38,5 GWh
Full reservoir level/height upstream (meters above sea level)	627,7	693,6	100	126,7
Draw-down level/height downstream (meters above sea level)	591	667,6	50,4	120,7

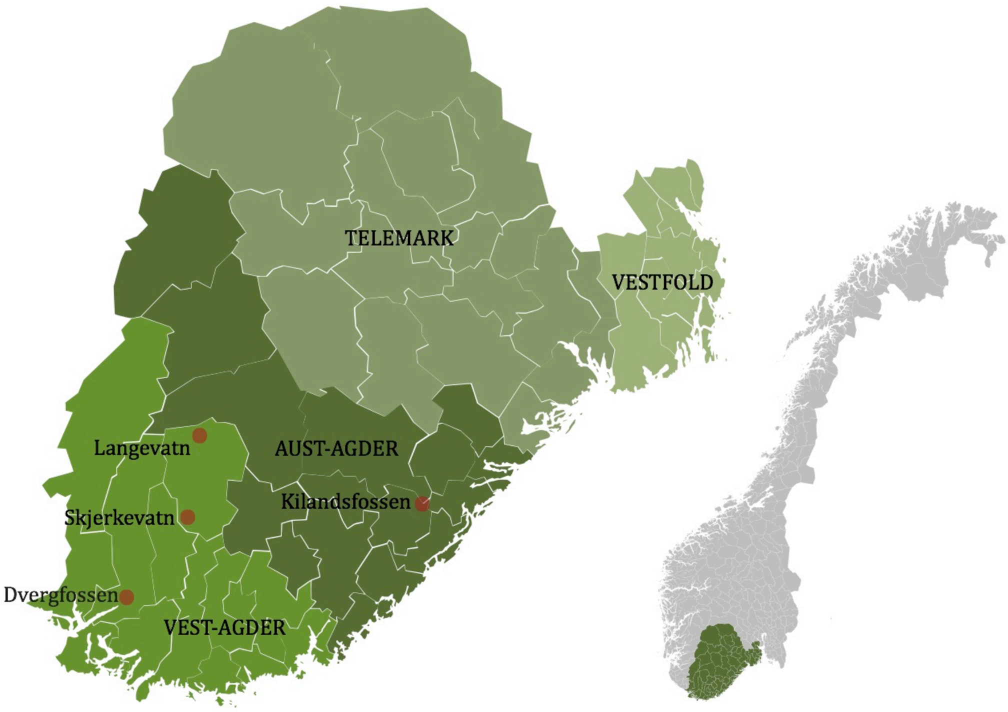


Figure 3: Map of the Norwegian Nature Index (NNI) region Southern Norway, with its counties. The selected case projects are marked in red.

Main Case – Skjerkevatn

The first permit for development of the Skjerka water system was given in 1924 and involved run-of-river hydropower between Sandvatn to Ørevatn. Further development was granted in 1933 and involved a construction of a dam in Skjerkevatn's eastern end with full reservoir level at 605 meters. In 1936 a concession was given for the joining of Hagedalsvann, Nedre and Øvre Skjerkevann, and Nåvatn and Sandvatn.

In 2013 an additional permit was granted for development in Skjerkevatn. The development involves replacing the old dam in Skjerkevatn and construction of an additional dam in Heddersvika, Skjerkevatn's southwestern end. The four dams currently separating Nyvatn and Skjerkevatn will be removed. This will merge Nyvatn and Skjerkevatn, and will raise the water level in Skjerkevatn by 23 meters. The increase in water volume is approximately 50 million m^3. The full reservoir level of the lake will be at 627,70 meters above sea level, and the drawdown level will be 591.00 meters for Skjerkevatn.

The power plant in Skjerka currently produces 612 GWh per year, and after the new dams are finished the plant will be producing and additional 43 GWh per year, giving a total of 655 GWh per year. The increase in in power production of 43 GWh is equivalent to the annual demand from 2150 households. Production of the same amount of electricity from coal would result in the emission of 40,000 tons of CO_2.

Langevatn

Langevatn is the northern most regulated lakes in the Skjerka water system in Åseral municipality in Vest-Agder County. The first regulation permit was issued in 1950. Langevatn is currently regulated between the drawdown level at 667,6 and full reservoir level at 683,6 meters. The operator has applied to NVE for a permit to increase the full reservoir level by 10 meter to 693,6 meters, but the permit has not yet been granted. Further regulation of Langevatn is one of several partial projects in the permit application. The other projects in the application will not be considered in this paper.

The operator has proposed to increase the full reservoir level by 10 meters building a new dam directly below the old dam, and then incorporating the old dam in the new one. The increased full reservoir level will deliver an estimated increase in power production by 18 GWh per year. Langevatn currently covers 2.08 km^2. The added water volume will be 24 million m^3.

Dvergfossen

Dvergfossen is a waterfall in the Litleåna River in Kvinesdal municipality, in Vest-Agder County in southern Norway. In 2013, NVE gave a permit allowing construction of a run-of-river (ROR) hydropower plant at Dvergfossen, utilizing the fall of water from 100,0 to 50,4 meters. The resulting energy production will be 35,5 GWh per year.

The project involves construction of a 40-meter dam for the water intake at contour line 100,0, which will form an unregulated basin upstream. During normal operation of the water level will be kept at level with the dam.

Kilandsfossen

Kilandsfossen is a waterfall in Nidelva River on the border between Åmli and Froland municipalities, in Aust-Agder County, in southern Norway. In 2013 NVE gave a permit allowing construction of a run-of-river hydropower plant at Kilandsfossen, which utilizes the fall of water, directly downstream Flatenfoss hydropower plant, from contour line 126,7 to 120,6, upstream Bøylefoss hydropower plant. The resulting yearly energy production will be 38,5 GWh.

The project involves construction of a hydropower plant at the site of a previous flume dam on the eastern river shore upstream from the waterfall, and 4 retaining dams (sperredam) and 4 artificial weirs (overløpsdam) blocking the natural waterway down the waterfall. This will create an unregulated basin.

2.2 MAPPING LAND USE AND LAND USE CHANGE

Mapping of the area was necessary to determine the effect of LUC caused by hydropower development. Information on the land use "after" the project realization was found in technical drawings in the permit applications for each of the cases (Appendix 2). The planned land use changes were manually geo-referenced in ArcMap 10.1 as either polygons or lines with an added land use change-specific buffer (Table 2). Areas that previously had been affected by land use change were excluded from the geo-referenced areas in order to exclusively consider LUC from the projects assessed in this thesis. Orthographically corrected aerial photos from the Norwegian Mapping Authority's Web Map Service www.norgeibilder.no was used to determine the existence and location of areas that had previously undergone LUC. The hydropower case projects assessed in this thesis have yet to be realized and information on the natural land use "before" realization of the projects was attained from current LU maps and orthophotos. Geo-referenced maps with information on natural land use from the Norwegian Mapping Authority's N50 series were superimposed on the maps of planned LUC to create maps containing information on total area changed, including which natural LU types that were affected and which LU types they were transformed into.

Table 2: Buffer zones - Each type of LUC was given an added buffer to include the associated edge effects of the type of LUC.

Land use change	Roads	Tractor roads	Electricity lines	Dam	Temporary infrastructure	New full reservoir level
Buffer (meters)	10	5	8	20	10	0

The total area occupied and the area occupied by the reservoir was divided by the yearly electricity production to give a value for each total basic LUC per kWh/year and the energy density each reservoir, both with the unit (m^2y/kWh).

2.3 IMPACT CALCULATIONS - ECOSYSTEM VULNERABILITY/SCARCITY AS INDICATOR FOR ECOSYSTEM QUALITY

Based on Milà i Canals (2007), the earlier version of the LULUC guideline by Koellner et al. (2013), Michelsen (2008) proposed using three factors to calculate an indirect quality indicator (Q) for an ecosystem. The three factors are Ecosystem Scarcity (ES), Ecosystem Vulnerability (EV) and Conditions for Maintained Biodiversity (CMB).

$$Q = ES * EV * CMB \quad (3)$$

Ecosystem scarcity

Weidema and Lindeijer (2001) first proposed ES as an indicator for biodiversity. The rational for ES is that it can be assumed that biodiversity´s vulnerability is positively correlated with the scarcity (rareness) of the ecosystem. This is due to smaller populations in less widespread habitats, and thus higher risk of damage caused by stochastic events and increase in edge effects in smaller habitats. ES is expressed as the inverse value of the potential area of the structure (A_{pot}).

$$ES = \frac{1}{A_{pot}} \qquad (4)$$

The structure can be at any hierarchical level, e.g. biome, landscape or ecosystem depending on data availability and the purpose of the study. All the cases assessed in this study were all situated in Southern Norway (as defined by NNI - Nybø 2010b, p.15) and A_{pot} for ecosystems were collected from area data from Statistics Norway (www.ssb.no – Appendix 3). ES was normalized using the total area of the region (A_{max}) with the following equation:

$$ES = 1 - \frac{A_{pot}}{A_{max}} \qquad (5)$$

Ecosystem vulnerability

The EV indicator is introduced to give information on the current pressure on an ecosystem in relation to the potential area of the ecosystem, thus reflecting that as more of an ecosystem that is lost, the vulnerability of the remaining parts increases. This is a consequence of MacArthur and Wilson's (1967) species-area relationship. Peter et al. (1998) first introduced the factor (originally named "area factor") and proposed the formula as follows:

$$EV = \frac{1}{1 - fraction\ lost} \qquad (6)$$

Similar to ES, EV can be normalized and the most vulnerable structure will then have a score of 1. NNI provides region specific information on the condition of distinct ecosystems based on past and predicted LUC and compares the current area to the potential area of the specific ecosystems. This gives a value for the remaining fraction (1 – fraction lost). For the alpine ecosystems the remaining fraction is 0,68, for wetland it is 0,51, and forest has a value of 0,42 (background material in Nybø 2010), indicating that forest is the most vulnerable ecosystem.

Conditions for maintained biodiversity

The last factor that makes up the quality measure is the Conditions for Maintained Biodiversity (CMB) indicator. This indicator gives information on the current condition of the key factors (KF) in the ecosystem, e.g. amount of dead wood in forest ecosystems. As the KF differ from ecosystem to ecosystem, the factors measured need to be ecosystem specific and the numbers of factors may also vary. For this reason the CMB should also be normalized in the same manner as ES and EV, using the key factors from the ecosystem with the highest value.

$$CMB = 1 - \frac{\sum_{i=1}^{n} KF_i}{\sum_{i=1}^{n} KF_{i,max}} \qquad (7)$$

Literature review to identify KF was done by using Scopus and Web of Science databases with the search terms ”key factors” + ”ecosystem” + ecosystem type. Boreal forest KF are well established and Larsson (2001) identifies a range of different key factors, especially amount of decaying wood and loss of diverse forest formations (Hanski & Walsh 2004; Rasche et al. 2013) and for wetlands hydrology and nutrient level are the most important (Laiho 2006; Xu et al. 2006). For alpine environments there are many different ecosystem types and KF will therefore differ. Suitable KF could not be identified for alpine ecosystems.

However, since the areas affected by LUC becomes built environment it was assumed that all the previously existing biodiversity would disappear and that CMB could be set to zero for all key factors. ΔQ can then be calculated as:

$$\Delta Q = ES * EV * (1 - 0) = ES * EV \quad (8)$$

Spatial and temporal impact

The duration of LU is set equal to the lifetime of the hydropower plant – 100 years. The restoration time is set to 500 years for alpine and wetland ecosystems, and 200 years for forest (Drescher et al. 2008; Moreno-Mateos et al. 2012). The spatial impacts are equal to the LUC found in the area mapping.

2.4 GHG Emissions from LUC

Carbon (C) released after disturbance in a natural system depends on the degree, duration and nature of disturbance and the amount of carbon stored in the system. Construction of hydropower plants and reservoirs cause permanent and temporary terrestrial damage, and permanent flooding. In this thesis C emissions were calculated as gross emissions over the lifetime of the hydropower plant, and any C sequestered by biomass regrowth and soil accumulation during the lifetime was not included. Also excluded was any LUC to areas previously covered by freshwater. The lifetime of the hydropower plant was set to 100 years.

There were, as previously mentioned, three distinct types of LUC; permanent infrastructure construction – PIC (construction of dams, roads, power plant etc.), temporary infrastructure construction – TIC (holding and rigging area) and areas permanently flooded for reservoirs. GHG emissions were calculated separately for each type of LUC.

Emissions from each of the natural LU types depend on the C content per square meter of each individual compartment. Data on C content in different natural systems in Norway was collected from the Norwegian Institute for Agricultural and Environmental Research, Bioforsk (Grønlund et al. 2010). The values are averages for each LU type (Table 3). C content of alpine land had a range and are therefore calculated for both the lower and upper limits of this range. More detailed information on C content was available in the report but the area data collected for this thesis could not match the same level of detail. All of the C in the natural LU types was assumed to be released to the atmosphere as CO_2 or CO_2-equivalents and all C emissions were scaled to CO_2 equivalents (CO_2e) with molecular weight ratios for CO_2 to C (1kg C = 3,67 kg CO_2).

Table 3: Carbon content and CO_2 emitted from the different land categories (based on Grønlund et al. 2010)

	Alpine environment		**Wetland**	**Forest**	
	Lower limit	Upper limit		Biomass	Soil
Carbon stored (kgC/m^2)	5	11	55	3,62	13,5
CO_2 emitted (kgCO_2/m^2)	18,35	40,37	201,85	13,2854	49,545

LUC related to PIC are severe in terms of the nature, duration and degree of disturbance and it was therefore assumed that 100% of the C in all compartments was released to the atmosphere over the lifetime. For TIC the LUC is assumed to be less severe and the disturbance and C release was assumed to be comparable to the C release during land transformation from natural to agricultural land. This implies removal of 100% of the above ground biomass (AGB) with 100% C release, and release of 25% of the carbon in all other compartments (Guo & Gifford 2002).

GHG emissions for LUC related to the establishment of reservoirs were calculated according to Tier 1 Guidelines by the IPCC (2003). Tier 1 only accounts for diffusive flows and methane bubbling from the reservoir, and does not include emissions upstream/downstream, or as a result of degassing or decay of above-water biomass. The Tier 1 method gives default emissions for CO_2, CH_4 and N_2O in kg per hectare per day for the boreal zone and these are adjusted according to the United Nation Framework Convention on Global Climate Change (UNFCCC) 100-year global warming potential (GWP) where CH_4 and N_2O are magnified 25 and 298 times, respectively. CH_4 and N_2O-emissions remain stable for the whole lifetime (100 years) of the reservoir, while CO_2-emission from the initial flooding cease after approximately 10 years. However, it was assumed that emissions of CO_2 also remained stable for the remaining 90 years as a result of biological material transferred to the reservoir, mainly from snow melting and/or flooding. Calculations of CO_2e-emission per square meter can be found in Table 4.

Table 4: Lifetime emission of CO_2-equivalents per square meter of reservoir

GHG	**Emission rate**	**Lifetime**	**Radiative forcing ratio**	**Lifetime emission**
	(gGHG/m^2/yr)	(yr)	(GHG/CO_2)	(kgCO_2-eq/m^2)
CO2	58,2175	100	1	5,82
CH4	1,6425	100	25	4,11
N2O	0,0292	100	298	0,87
			Total CO_2e/m^2:	**10,80**

Emissions from PIC, TIC and the reservoir were added together for the individual cases and divided over the individual lifetime production of electricity. This results in a comparative metric in units of CO_2-equivalent GHG emissions per kWh produced.

2.5 Restoration Actions and Cost

The cost of restoration is a reflection of the effort necessary to facilitate the return of a disturbed, damaged or destroyed ecosystem/landscape to a more resilient natural condition. As previously described, in this thesis restoration cost was explored in the context of biodiversity offsetting, meaning restoration of degraded ecosystems off-site in order to compensate for impacts on-site. From the natural LU mapping of the four case projects, it was found that the ecosystems that were affected by LUC were alpine environments, freshwater systems, wetlands and forests. Only restoration of terrestrial ecosystems has been considered here, leaving out freshwater ecosystems, such as smaller lakes and rivers.

Due to lack of available background data from offset sites hypothetical restoration scenarios were created to illustrate a general approach to calculating restoration cost. Historically, the natural environment in Norway has been changed considerably over the centuries and historical LU was set as the starting point for the scenarios. The Norwegian Ministry of Finance's white paper (NOU 2009:16) states that 25% of Norway's wetland has been drained the last century, mainly for forestry, and we assumed that the theoretical offset site for wetland is an afforested former wetland. Forestry has also had great impact on Norwegian natural forests ecosystems and approximately 20% of Norwegian land area is covered by productive forest. It was assumed that the theoretical offset site for forest was a planted forest under management. The alpine environment in Norway covers approximately 46% of the land area (www.statkart.no) and has been highly fragmented by infrastructure development during the last 100 years. The fraction of undeveloped natural areas (INON - defined as area >5km from major infrastructure constructions – www.inon.miljodirektoratet.no) has decreased from 48% to 12% in the period from 1900 to 2003 (NOU 2009:16). It was therefore assumed that the theoretical alpine offset site had been fragmented by road construction.

In order to facilitate a given ecosystem's return to a more resilient natural state, several actions can be implemented. The actions suggested in this thesis were based on recent development in applied restoration of boreal ecosystems from Finnish boreal forest and wetland restoration and Norwegian alpine restoration, where cost of specific restoration actions in each ecosystem were available (Table 5) (Hagen & Evju 2013; Hagen et al. 2014; Similä & Junninen 2012; Aapala et al. 2013). The general aim of these actions are to secure ownership to prevent future unwanted anthropogenic disturbance, remove existing disturbances and unwanted species, and enable ecosystem specific biodiversity's viability, growth and resilience.

Common for all of the ecosystems were the procurement of land and removal of roads in the area. For modeling purposes it was assumed procurement should consist of a total area of equal size to the area affected by LUC in the hydropower development case project in question, and that all the ecosystems were replaced one to one in terms of size. For example, development in Skjerkevatn caused LUC in a certain area of the alpine environment and thus an alpine area of the same size should be procured for restoration and protection. In order to estimate costs for road removal, assumptions were made for the total length of road at the restoration site. It was assumed that the restoration site was circular to reduce edge effects and that there were roads within that area with a total length equal to two times the diameter of the area.

In addition to the procurement and the roads, ecosystem specific restoration actions were added. Due to the cold climactic conditions, natural recovery is a slow process in alpine ecosystems (Jorgenson et al. 2010). Experience from Norwegian restoration projects indicate relevant restoration methods to improve recovery (e.g. Hagen & Evju 2013; Hagen et al. 2014). Based on this it was the following restoration actions were used: add topsoil to 100% of the road, application of fertilizer and native seeds to 30% of the road, and plant shrubs on 5% of the road.

Successful restoration projects by Tolvanen et al. (Simil & Junninen 2012; Aapala et al. 2013) were the basis for the restoration actions suggested for wetland and forest ecosystems. In the wetland and forest, ditches should be filled to undo past disturbance, and the total length of the ditches in both ecosystems was assumed to be 4 times the diameter of the site. In the wetland all trees should be removed and transported out of the area. It was assumed that the trees have an average volume of 0,5 m^3 and that there are 250 trees per decare. In the forest no trees are removed, but instead some are uprooted or girdled to create spatial heterogeneity and dead wood in the ecosystem, both of which are vital to biodiversity. Uprooting creates canopy gaps and dead wood on the forest floor. Girdling (ring-barking) kills the trees but leaves them standing. An old growth forest has approximately 20 m^3 of dead wood per hectare and with an average tree volume of 0,5 m^3 that would suggest that 40 trees should be uprooted or girdled. Forests under management usually have approximately 10 m^3 of dead wood per hectare and this decreases the need for addition of dead and decaying wood during restoration. However, since dead and decaying wood needs to be added consecutively for some time during the restoration process, calculations were done for the uprooting of 40 trees and the girdling of 40 trees, creating in total an additional 40 m^3/ha of dead wood. To further increase spatial heterogeneity and thin the forest, an additional 20 glades were also added.

Finally, the total cost of offsetting the LUC 1:1 can be found by multiplying all required effort with the cost of that effort and the natural land affected by land use change, and summing across all ecosystems. Details for the background for the cost values can be found in Appendix 4.

Table 5: Implemented restoration aims with actions, required effort for success, and cost per unit effort for each ecosystem, based on assumptions listed in the text.

Ecosystem	Restoration aim	Action	Target/effort	Costs
Alpine – previously disturbed by road construction	Secure ownership and protection of land	Procure land	Land should be equal in LU and size to the area affected by LUC in the development project	0,5 NOK per m^2
	Remove existing disturbance from previous LU	Remove roads and add topsoil	All roads in area – area assumed to be circular with roads equal to 2x area diameter	300 NOK per meter road
	Enable biodiversity and biodiversity growth	Add fertilizer and indigenous seeds	Applied to 30% of road area	90 NOK per meter road
		Plant indigenous shrub	Conducted on 5% of road area	15 NOK per meter road
Wetland – former wetland previously turned into managed forest	Secure ownership and protection of land	Procure land	Should be equal in LU and size to the area affected by LUC in the development project	1 NOK per m^2
	Remove existing disturbance from previous LU	Remove roads	All roads in area – area assumed to be circular with roads equal to 2x diameter	300 NOK per meter road
		Fill ditches	All ditches in area should be filled – assumed to be 4x area diameter	6,44 NOK per meter ditch
	Remove unwanted species	Fell trees	100% of the trees in the area should be felled	111,12 NOK per m^3 wood
		Remove trees	All felled trees should be transported away	234,26 NOK per m^3 wood
Forest – old-growth forest previously turned into managed forest	Secure ownership and protection of land	Procure land	Land should be equal in LU and size to the area affected by LUC in the development project	1 NOK per m^2
	Remove existing disturbance from previous LU	Remove roads	All roads in area – area assumed to be circular with roads equal to 2x area diameter	300 NOK per meter road
		Fill ditches	All ditches in area should be filled – assumed to be 4x area diameter	3,44 per meter ditch
	Enable biodiversity growth by creating a heterogeneous environment	Uproot trees	40 trees per ha	117,13 NOK per uprooting
		Create glades/gaps	20 glades per ha	97,61 NOK per glade
		Girdle (ring-barking)	40 m3 of decaying wood per ha	78,09 NOK per m^3 wood

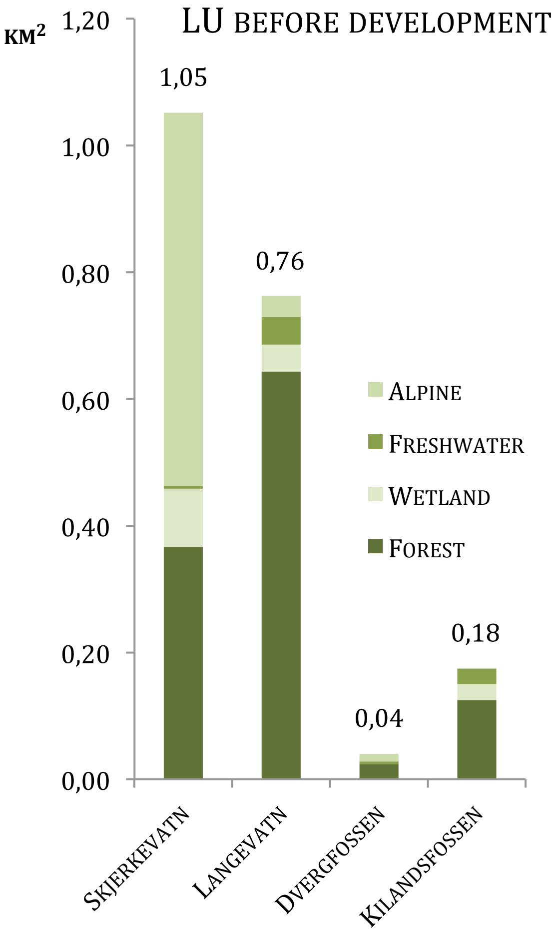
3 RESULTS

3.1 LAND USE CHANGE

The LUC in the cases affected four main ecosystem types; alpine, freshwater, wetland and forest. The total area that will undergo LUC in Skjerkevatn, Langevatn, Dvergfossen and Kilandsfossen is 1,05 km^2, 0,76 km^2, 0,04 km^2 and 0,18 km^2, respectively (Figure 4).

Prior to development, forest was the dominant natural LU type for Langevatn, Dvergfossen and Kilandsfossen covering 84%, 58% and 71% of the total developed area, respectively. Alpine/open LU was dominant in Skjerkevatn covering 56%, with forest a close second at 35%. Wetland and freshwater LU were small with values from 6% to14%.

After development the reservoir was the dominant LU for Skjerkevatn, Langevatn and Dvergfossen, and covered 69%, 74% and 53%, respectively. LUC related to permanent infrastructure construction (PIC) were just below 20% for Skjerkevatn and Langevatn, and 44% and 67% for Dvergfossen and Kilandsfossen. LUC related to temporary infrastructure construction (TIC) was 13% in Skjerkevatn, 6% in Langevatn and 3% in Dvergfossen.



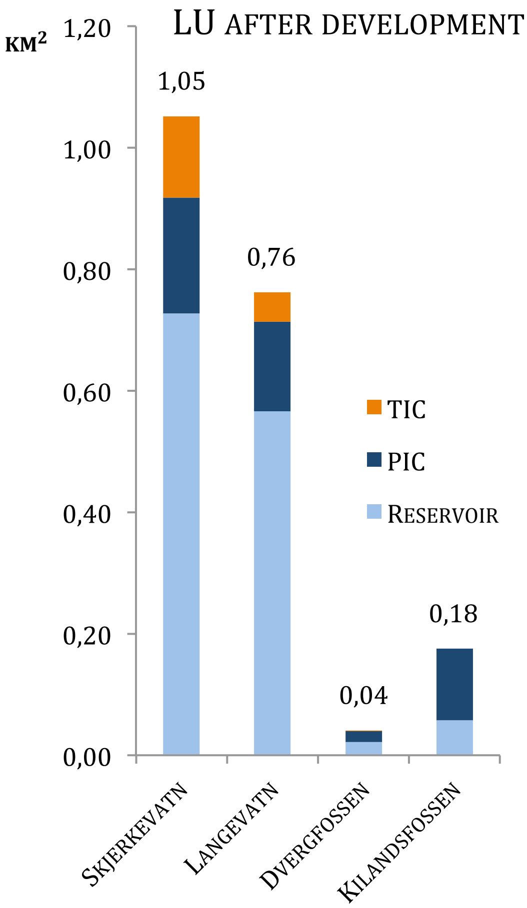


Figure 4: LU before (left) and after (right) development (LUC) for each case. The total LU does not change with development, but LU type does.

The mapping method makes it possible to trace the fate of each natural LU type (Figure 5). This figure is the connection between the two LU figures in Figure 4. The complete table with background data for the mapped LUC is available in Appendix 5. The basic LUC/kWh and the energy density of the reservoir can be found in Table 6.

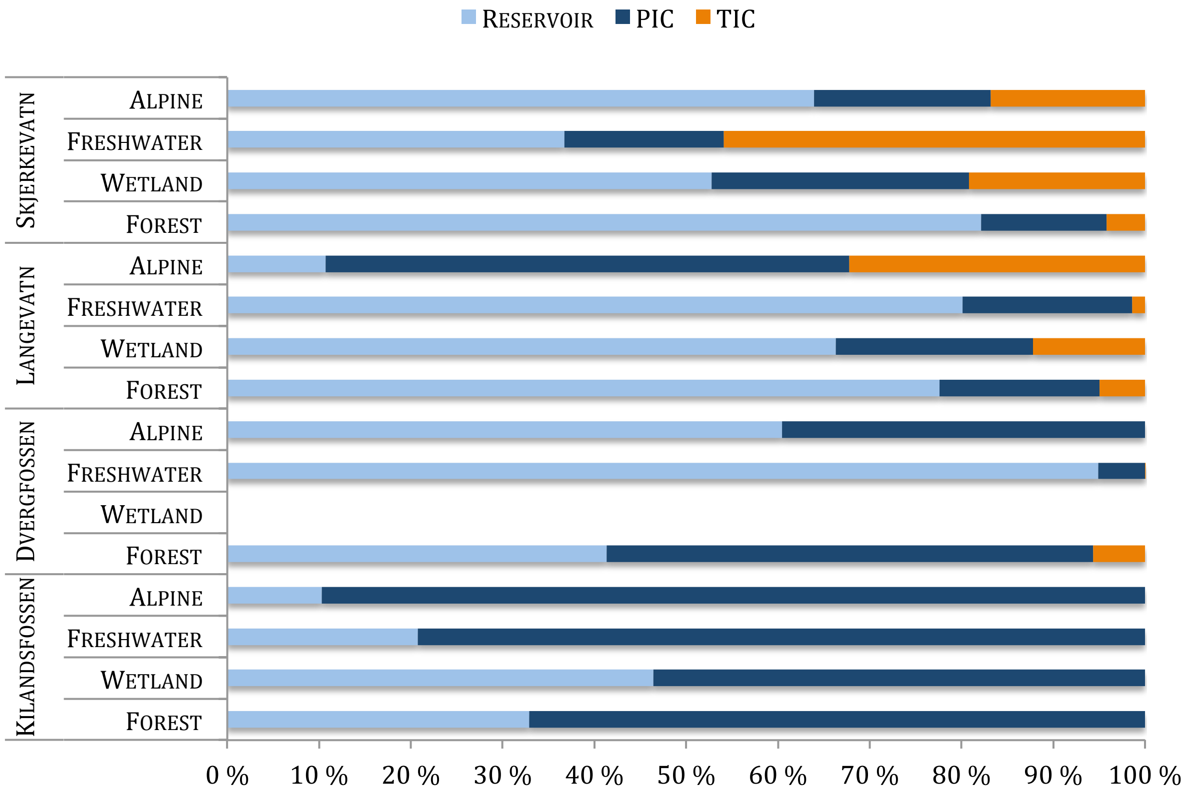


Figure 5: Fate distribution of individual natural LU types for each case. This figure illustrates the connection between the LU before and after development.

Table 6: Basic LU per electricity unit produced for each study case project

	Skjerkevatn	Langevatn	Dvergfossen	Kilandsfossen
Total basic LUC (m^2y/kWh)	0,0245	0,0423	0,0011	0,0045
Energy density reservoir (m^2y/kWh)	0,0169	0,0295	0,0005	0,0014

3.2 Ecosystem Scarcity, Vulnerability and CMB

Separate ΔQ-values were calculated for all terrestrial ecosystems (Table 7). Wetland is clearly the scarcest of the three ecosystems and also has the highest ΔQ value. Forest has the lowest value for ΔQ despite being the most vulnerable ecosystem. The alpine ecosystem has the middle value for all three indicators.

Table 7: Difference in change in ecosystem quality due to difference in ecosystem scarcity and vulnerability

Ecosystem	Potential area (A_{pot}) (m^2)	Ecosystem Scarcity (ES)	Remaining fraction	Ecosystem Vulnerability (EV)	ESxEV = ΔQ
Alpine	9277,07	0,67	0,68	0,62	0,42
Wetland	1136,79	0,96	0,51	0,82	0,79
Forest	18101,89	0,37	0,42	1,00	0,37

Transformation impacts (TI) were larger than occupation impacts (OI) in all cases (Figure 6), due to the difference in T_{occ} and T_{reg} for the ecosystems. The LULUC per FU was similar for Skjerkevatn and Langevatn, the latter slightly larger (Table 8). For Dvergfossen and Kilandsfossen, LULUC per FU was one order of magnitude smaller.

Table 8: Overall LULUC and LULUC per unit of electricity produced over the lifetime of the hydropower plants

	Skjerkevatn	Langevatn	Dvergfossen	Kilandsfossen
LULUC - ΔQ*km²y	138,2	63,6	3,6	16,2
LULUC per FU - ΔQ*m²y/kWh	0,0321	0,0353	0,0010	0,0042

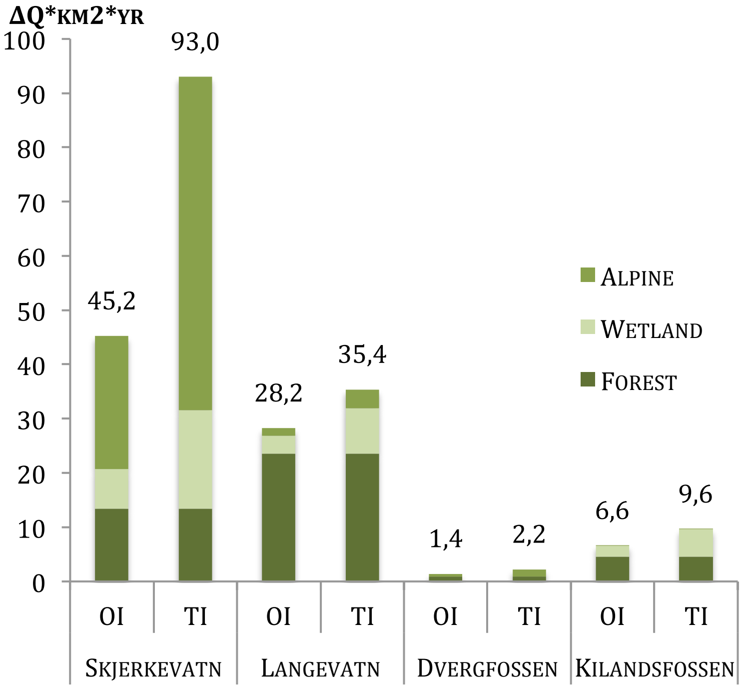


Figure 6: Impacts to ecosystems of occupation impact (OI) and transformation impact (TI) for each case.

3.3 GHG Emissions

The main source of CO_2e came from LUC related to the PIC, followed by LUC related to the reservoir, and lastly related to TIC. Skjerkevatn had the highest gross emission for all types of LUC. Langevatn follows closely with slightly lower emission values for all types of LUC (Table 9). Emissions values from Dvergfossen and Kilandsfossen are mostly lower by one order of magnitude compared to Skjerkevatn and Langevatn, with one clear exception – emissions associated with PIC in Kilandsfossen are one magnitude higher, at 8,03 kT, and almost as high as Langevatn.

CO_2-emissions per kWh over the lifetime of the hydropower plant are clearly lowest for Dvergfossen and highest for Langevatn (Table 9).

The highest contribution to gross emission related to PIC came from wetland in Skjerkevatn, and forest soil in Langevatn, Dvergfossen and Kilandsfossen. Biomass was the largest contributor to gross emissions related to TIC in Langevatn, Dvergfossen and Kilandsfossen. For Skjerkevatn, the main contribution came from wetland or alpine, depending on whether the upper or lower limit for the alpine environment was used.

Table 9: CO_2equivalent emissions for each case.

	Skjerkevatn	**Langevatn**	**Dvergfossen**	**Kilandsfossen**
Gross emissions (kT CO_2e)	20,02-23,06	16,11-16,58	1,10-1,22	8,60
Reservoir (kT CO_2e)	7,84	5,74	0,19	0,57
PIC (kT CO_2)	10,43-12,93	9,25-9,67	0,87-0,99	8,03
TIC (kT CO_2)	1,74-2,29	1,12-1,18	0,03	-
Emissions per lifetime production (gCO2e/kWh)	4,65-5,36	8,95-9,21	0,31-0,34	2,23
Emissions per area (kgCO_2e/m^2)	19,07-21,96	21,20-21,82	27,50-30,50	49,14

3.4 Restoration cost

Restoration costs for each action and case are listed in Table 10. Skjerkevatn had the highest restoration offset cost and Dvergfossen had by far the smallest. Wetland restoration was the largest contributor to the overall cost for all cases where wetland was restored, contributing 66%, 50% and 70% of the total cost for Skjerkevatn, Langevatn and Kilandsfossen respectively. The high total cost of wetland restoration is largely due to the cost of tree felling and transportation, which alone makes up 89–93% of the wetland restoration costs.

The forest restoration actions contribute significantly to the cost in all cases. In Dvergfossen, forest restoration cost was dominant with 58% of the total cost. For Langevatn the forest cost came close to the wetland costs with 1,86 mill, 46% of the total.

The restoration cost per total area restored was approx. 6 NOK/m^2 for Skjerkevatn, Langevatn and Dvergfossen. For Kilandsfossen the cost was 10 NOK/m^2. The cost per kWh produced over the lifetime is highest for Kilandsfossen with 0,00227 NOK/kWh. Restoration cost per kWh for Skjerkevatn is of the same order of magnitude. For Dvergfossen and Kilandsfossen the cost is one order of magnitude less (Table 10). Figure 7 shows the contribution to the total restoration cost by each ecosystem.

Table 10: Restoration costs for each case, based on assumptions and methods described in the text

	Skjerkevatn	Langevatn	Dvergfossen	Kilandsfossen
Total cost of restoration actions	**6 441 785**	**4 078 302**	**266 416**	**1 750 329**
Alpine cost	*996 473*	*182 086*	*110 631*	*14 784*
Procure land	294 738	16 426	6 491	129
Remove roads + add topsoil	519 804	122 711	77 141	10 856
Fertilize and seed	155 941	36 813	23 142	3 257
Plant shrubs	25 990	6 136	3 857	543
Wetland cost	*4 279 914*	*2 033 494*	-	*1 233 600*
Procure land	92 042	42 731	-	25 380
Remove roads	205 399	139 952	-	107 859
Fill ditches	8 821	6 011	-	4 632
Fell trees	1 278 433	593 523	-	352 526
Remove timber	2 695 220	1 251 278	-	743 203
Forest cost	*1 165 398*	*1 862 722*	*155 785*	*501 944*
Procure land	366 262	643 321	23 543	125 135
Remove roads	409 734	543 025	103 881	239 495
Fill ditches	17 597	23 322	4 461	10 286
Uproot trees	143 002	251 175	9 192	48 857
Create glades	171 602	301 410	11 030	58 629
Girdle trees	57 201	100 470	3 677	19 543
Cost per m^2 restored (NOK/m^2)	**6,12**	**5,35**	**6,57**	**10,00**
Cost per FU over LT (NOK/kWh)	**0,00150**	**0,00227**	**0,00008**	**0,00045**

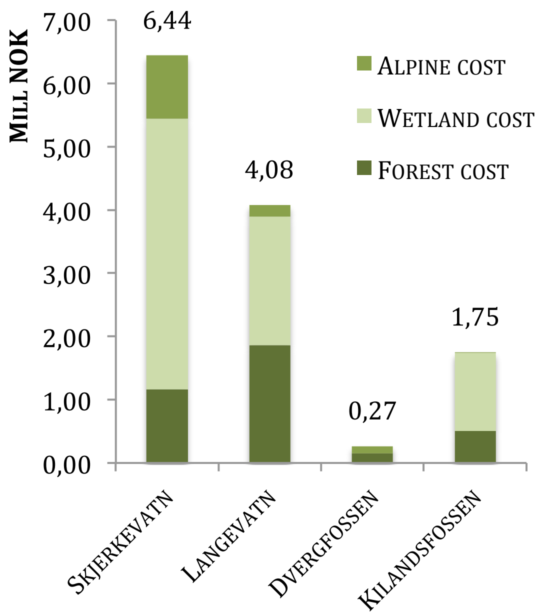


Figure 7: Distribution of restoration cost per ecosystem type for each study case project

3.5 Comparing methods

Normalization of the results from each method enables comparison between them. The results for each method were normalized with the highest value for that particular method (Figure 8). Mean values were used for the carbon results.

Langevatn had the highest impact per FU for all methods, while Dvergfossen had the lowest impacts for all methods with only 3-4% of the impacts compared to Langevatn. The results for Skjerkevatn and Kilandsfossen were more variable. The value for ES/EV in Skjerkevatn was 91%, and restoration cost was 66% of Langevatn's maximum, while the values for the results for CO_2e/kWh and the basic LUC/kWh were 55-58% (Figure 8). In Kilandsfossen, the CO_2e-emission was 25% and restoration cost was 20% of the values for Langevatn, while the basic LUC and ES/EV were 11-12% per kWh produced (Figure 8).

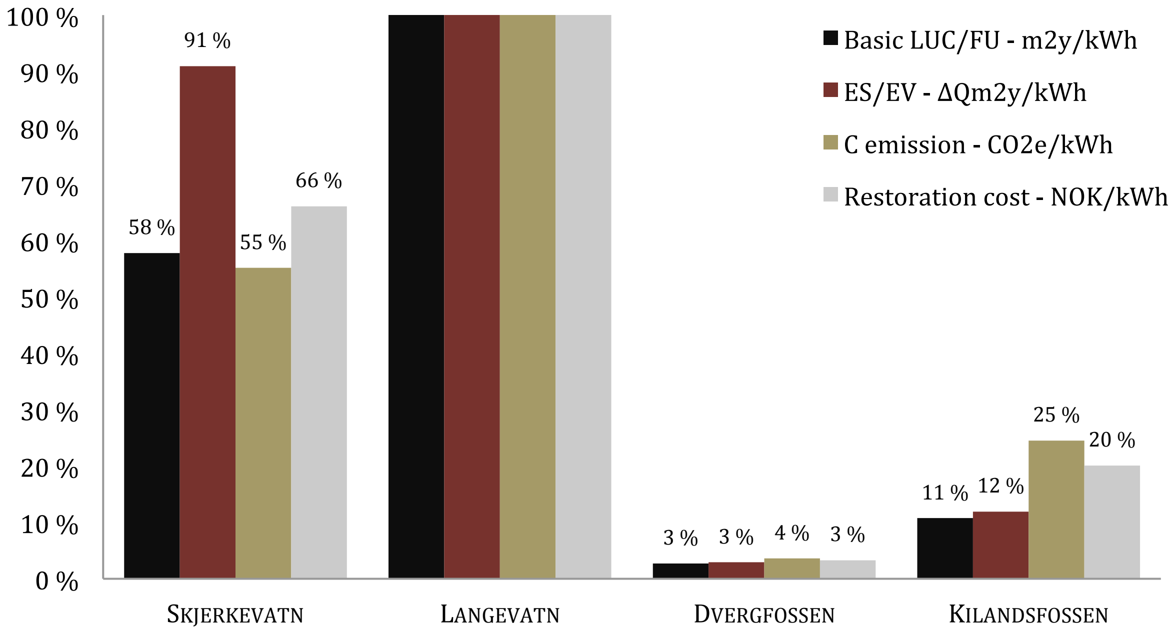


Figure 8: Comparison of normalized results for all methods and cases.

4 Discussion

The aim of this study was to assess the impact of hydropower on biodiversity due to land use and land use change. This section will discuss the influence that case selection had on the results of the mapping. The results for the different approaches will then be discussed in light of the research questions.

4.1 Land Use and Land Use Change

The area affected by LUC is much larger in Skjerkevatn and Langevatn compared to Dvergfossen and Kilandsfossen. LUC in the larger development projects are in large part associated with the construction of the large reservoirs (Figure 4), which is to be expected when comparing reservoir based and run-of-river hydropower (Bakken et al. 2014). Interestingly, Kilandsfossen has a LUC related to PIC of almost equal size to that of Skjerkevatn and Langevatn, which suggests that LUC related to infrastructure is relatively more important for smaller run-of-river hydropower. This is consistent with the findings in Hagen & Erikstad (2013) and is a good argument for including total basic LUC/kWh, and not just the energy density of the reservoir when evaluating smaller run-of-river hydropower development. Bakken et al. (2014) concluded that small-scale hydropower (1-10 MW installed capacity – NVE standard) also has a larger impact on red-listed species then larger reservoir based hydropower.

The LUC in Dvergfossen did not show the same results as Kilandsfossen but a substantial part of the TIC and PIC in Dvergfossen was situated on previously disturbed land and was not included in the mapping. The unexpected results for Dvergfossen illustrates that there are benefits to situating development on previously disturbed land, as the impact of the development in Dvergfossen is consistently smaller than the other case project for every method. By situating development on previously disturbed land further destruction of natural systems are avoided and in doing so the associated carbon emissions, biodiversity loss and ecosystem quality reduction is avoided.

The reservoir energy density of Skjerkevatn and Langevatn are low compared to other hydropower reservoirs (background material in Hertwich 2013) and well below the limits of the Clean Development Mechanism at 0,1 m^2y/kWh (Hertwich 2013). Total basic LU/kWh is also well below the limit (Table 5). However, in this thesis, the energy densities have only been calculated for the expansion of the reservoirs and they were expected to be lower than the energy density of a newly established reservoir due to the relationship between the areas a reservoir occupies and the volume of water it can contain.

4.2 Ecosystem Scarcity and Vulnerability Ecosystem Quality

Based on the indirect indicators for the current ecosystem condition, represented by ecosystem scarcity and vulnerability, the results show that the change in ecosystem quality (ΔQ) is twice as large for wetland compared to alpine and forest, due to the extreme scarcity of wetland in the region. This makes the relative occupation impacts (OI) of wetland twice as large as alpine and forest impact, compared to the basic area of LUC.

The transformation impacts (TI) are greater than OI for all the case projects, and the relative TI of alpine and wetland are 2,5 times as large as TI for forest (Figure 6), due to the difference in restoration time. The importance of the alpine ecosystem for TI is the reason for Skjerkevatn's large impact for ES/EV compared the results for the other methods (Figure 8). This indicates that ES/EV gives more weight to impacts on alpine ecosystems than the other methods and that it is sensitive to restoration time. Careful consideration for determining appropriate values for restoration time is recommended.

The foundation for using ecosystem scarcity as an indirect factor for biodiversity is that we recognize that there are significant knowledge gaps concerning ecosystem composition, structure and function, it is assumed that what is rare is valuable. The values for ecosystem scarcity are highly dependent on the value chosen for $A_{max.}$ In this analysis regional area data for Southern Norway is used as A_{max}. It reflects the regional natural composition of the ecosystems examined, and also fits the data availability on a regional level for the remaining fraction used in the Ecosystem Vulnerability calculations. If a higher value for A_{max} was chosen all ecosystems will be relatively rare (ES close to 1) and the relative difference in scarcity between the ecosystems will be smaller. If A_{max} was set at a lower level, the difference between ES of the ecosystems would be larger and the disturbance in the most common ecosystems would be almost insignificant due to an ES close to or equal to zero. For example, if the most common ecosystem type, forest, was used as A_{max}, the ES for the forest would be zero and any disturbance in forest ecosystems would therefore be unaccounted for in the calculations. The dependency on A_{max} makes comparison between studies difficult, and further development of the method should establish a strict framework.

Total LUC for each ecosystem was used as A (in equation 1 and 2). No differentiation was made for whether the natural LU was changed into reservoir, permanent or temporary infrastructure. Unassisted recovery can be expected in areas affected by temporary impacts, and the reservoir will also contain biodiversity. As a consequence of this, ecosystem quality will vary during occupation. The areas permanently occupied by infrastructure will be the only area where relatively stable ecosystem quality can be expected. The inclusion of freshwater ecosystems would be beneficial from a holistic perspective, but it was elected to do the analysis for terrestrial ecosystems exclusively.

The results from this thesis are difficult to compare to previous studies (e.g. Michelsen 2008; V. Coelho & Michelsen 2014) due to methodological differences, but could set a baseline for future studies conducted with the same method as is used in this thesis, either within the same region or in other NNI regions, given appropriate adjustment of ecosystem scarcity and vulnerability. The coupling of LULUC methodology with the continuous development of NNI and land use surveys by Statistics Norway or the Norwegian mapping authority would ensure consistency and reliability.

4.3 CARBON SEQUESTRATION

The results for GHG emissions show that emissions related to construction of permanent infrastructure are more important than emissions from the reservoir over the lifetime of the hydropower plants, even for the larger reservoir-based case projects. In relation to the relative importance of PIC for small-scale hydropower discussed previously, this is a further argument for including total basic LUC/kWh in evaluation of new hydropower development.

The comparison of the methods (Figure 8) reveals that the C emissions follow the other methods in ranking of impact per kWh for the case projects and thus, in this thesis, preforms equally well as an indicator for biodiversity/ecosystem quality as the other methods. The relatively high results for Kilandsfossen are related to the large share of PIC and that a large part of the carbon rich wetland affected by the development is changed into permanent infrastructure (Figure 4). Using C sequestration as a biodiversity indicator will emphasize carbon rich ecosystems.

Carbon sequestration is only one of many ecosystem services, and is in itself not necessarily a representative indicator for biodiversity and ecosystem quality. For example, the Sitka spruce (*Picea sitchensis*) grows fast and sequesters more carbon than Norway spruce, but it is an invasive species in Norway (Gederaas et al. 2012). Planted Sitka forest tend to be dense and is documented to outcompete and reduce the ecosystem quality for a number of native forest species (Gederaas et al. 2012). If ecosystem services are to be included in LULUC, other services need to be included, in addition to sequestration, to ensure that the indicator is representative of biodiversity.

Ideally, the calculations for the carbon should have been net carbon equivalent fluxes from the area over the lifetime in a consequential LCA, where both emission and sequestration from the whole area over the lifetime could be included. It would also include information on the carbon flux in the area if no development occurs. The forest ecosystems currently sequester more carbon than they emit and wetland and freshwater systems have net GHG emissions if left untouched (Tremblay et al. 2005; Grønlund et al. 2010). There are currently no available carbon flux measurements for alpine ecosystems, but due to low soil respiration and primary production the fluxes are smaller than those found in other ecosystems (Grønlund et al. 2010). The emissions associated with PIC were largest in all the cases and the PIC areas are also the areas where no biodiversity recolonization is expected. The area will therefore not contribute to the future carbon flux in the area. The areas affected by temporary LUC can be expected to be recolonized over the lifetime and therefore contribute to carbon sequestration.

The low emissions from the reservoirs are due to the uncertainty surrounding the fate of the carbon in the soil that is washed out in the reservoir after flooding. The distribution of the soil in the water column and the degree of sedimentation will determine the break-down and emission of the carbon. Some of the particles will also be transported down stream and will therefore be outside the physical system boundaries of this thesis, since emissions down-stream are not included. This suggests an underestimation for emissions from the reservoir.

4.4 RESTORATION COST

The use of restoration cost as an indicator for impacts on ecosystem quality makes no assumptions concerning ecosystem value. It is built on the assumption that a resilient ecosystem has intrinsic quality, whether it delivers services or not, and merely gives an estimation on how much it would cost to get back to a resilient state in the ecosystem (Suding 2011).

Of the ecosystems that were considered for restoration in this thesis, wetland was by far the most costly to restore per square meter. In large part this was due to the felling and removal of unwanted trees in the ecosystem, due to the fact that a drained and afforested wetland was chosen as the theoretical restoration site. If a different theoretical restoration site were chosen e.g. a peat extraction site, other restoration actions would have been required and the cost might have been changed. As afforestation of wetland has historically been common practice in Norway the choice of the afforested site was considered suitable.

The restoration actions that were suggested in this thesis are in no way exhaustive, but the cost of restoration is rarely published in the scientific literature and those that are available are highly variable both within a single ecosystem and between different ecosystem types (Bullock et al. 2011). This is due to high variability in timescales and inconsistencies in methods (Aronson et al. 2010).

Using the suggested method of calculating cost for individual measures based on a desired level of implementation makes it possible to add or subtract any suitable restoration actions to the budget. The calculations for the forest restoration cost can be an example of this. The measures that were suggested for forest ecosystems came from Finnish restoration where fire is frequently used to create spatial heterogeneity and create dead wood (Simil & Junninen 2012). Burning has traditionally been a part of Finnish forest management and managers have the necessary level of expertise to use of forest fires in a controlled manner. Also, as a consequence of the historic use of fire the forest ecosystems have adapted to the management regime and are therefore dependent on the fires to maintain biodiversity levels. Norwegian forest management does not have the same traditional use of controlled burning and therefore has neither the experience nor the same ecological dependence on this type of management. The restoration actions associated with the controlled burning were therefore not included in suggested actions for the forest restoration calculations that were made in this thesis. Norway does not have a tradition for restoring forest ecosystems (Hagen et al. 2013) but as the field of restoration expands its knowledge, more restoration actions could easily be added to the budget and adapted to local conditions.

Restoration projects require planning both pre-realization and during the implementation, and these costs are referred to as transaction costs. These can be hard to define and calculate. In some cases it might also be necessary to conduct additional monitoring and further management following restoration actions. Both management and additional restoration actions will increase the overall cost of the project, but were not included here as the costs were unavailable. Further research into these aspects of cost calculations is needed.

The restoration calculations conducted in this thesis have only been concerned with the *cost* of the restoration actions. However, there could be considerable income related to the restoration actions, particularly related to wetland restoration. The theoretical wetland restoration site is a managed forest and trees taken out of the area can be sold as timber or wood for energy purposes. Income from sales was not included in the calculations in this thesis, however, so as not to camouflage the cost related to restoration. Taking into account the mitigation hierarchy, one could argue that any revenue generated in relation to the restoration effort should be returned back into the restoration project to secure funding for unforeseen cost or to facilitate a net gain from biodiversity offsets.

The restoration cost calculations in this thesis have been conducted with the assumption that the offset site is of equal size with the area that was affected by LUC. As mentioned in chapter 1.3 on the Mitigation Hierarchy, offset ratios of 1:1 are probably not enough to secure no net loss/net gain to biodiversity. Moilanen et al. (2009) points out that this simple model excludes time lag between the impact and the restoration gain, does not consider success rates for restoration (75-86% biodiversity and 80% for ecosystem services (Bullock et al. 2011)) and the proximity of the restoration site. Including these factors may increase the required offset ratios by two orders of magnitude (Moilanen et al. 2009). The offset ratio that was chosen for the calculations in this thesis was therefore probably too small and the total restoration cost for each of the cases were probably underestimated. The total restoration cost should therefore not be interpreted as the true cost of restoration for each case, but rather serves as a measure for comparing between the different cases and ecosystems and cannot be generalized to be valid for other cases and regions.

The restoration actions implemented this thesis are, as mentioned previously, not exhaustive, and illustrate that restoration ratios can vary not only from case to case but also from ecosystem to ecosystem. This is especially true for wetland restoration in this thesis, since no direct actions to facilitate biodiversity establishment and growth were suggested. This is contrary to restoration for alpine and forest ecosystems where addition of fertilizer and addition of dead wood were suggested. Inexperience in boreal ecosystem restoration and lack of information on cost made it difficult to include other restoration actions than the ones already in place in the calculations.

4.5 DATA QUALITY

There is high-resolution data available for carbon content in different ecosystems. The main ecosystem types utilized in this thesis have several specific sub-classes with information on carbon content and area covered, used to estimate total carbon content in Norwegian vegetation and soil (Grønlund et al. 2010). However, the available land cover maps (N50) do not have the same resolution, especially for different soil types and wetland depth. Wetland and soil have the largest carbon stores in the boreal zone and more detailed mapping on their occurrence would give more certain results for the emission estimates.

CO_2e emission per square meter of the reservoir is around the same as the CO_2-emission from one square meter of above ground biomass. This indicates that the estimated GHG emissions for the individual case projects are too low, especially for the reservoir-based hydropower development in Skjerkevatn and Langevatn. The IPCC Tier 1 calculation for GHG emissions for reservoirs does not take into account the soil types that are flooded when reservoirs are established. The carbon content of soil can vary substantially depending on amount of organic content and the GHG emissions would therefore probably vary substantially with soil type.

The buffer zones that were included during the mapping of the development are a source of insecurity as the post-development condition of the ecosystems within the buffer zones is uncertain. The buffer zones will have lower emission rates than the area directly affected by LUC, and are probably causing an overestimation for these particular zones.

The statistics for area data used for calculations of ecosystem scarcity values are sound. Grønlund et al. (2010) compared the SSB data to similar data from The Norwegian Mapping Authority and found only minor differences. The basis for the ecosystem vulnerability calculations from the Nature Index for Norway is also sound, with multiple criteria for each index, and its basis in past and present area pressure for each ecosystem.

4.6 Methodical Limitations

Following is a brief discussion of some of the methodological limitations common for all the approaches. Other methodological limitations have been dealt with in the discussion for the specific approach they concern.

Impact analyses were done exclusively for terrestrial ecosystem and impact to freshwater ecosystem were therefore not part of the analysis. As a consequence of this the analyses do not give a full and complete insight into the impact the case project had on the ecosystems that were impacted.

Residual impacts were not included in the calculations in this thesis. Residual impacts are considered a consequence of the initial transformation, but there is disagreement within the workgroup that has developed the guideline (Koellner et al. 2013) whether the residual impact and the transformation impact should be aggregated. They argue that residual impacts represents diminishing options for future development of an affected area, and that TI and OI on the other hand describes actual temporary impacts during occupation/regeneration, and that aggregating them implies a weighting of present and future impacts. Excluding residual impact from the calculations in this thesis therefore implies that the future development options are considered to not be important.

5 CONCLUSION AND RECOMMENDATIONS FOR FURTHER RESEARCH

The aim of this thesis is to assess the impact of hydropower on biodiversity due to land use and land use change using three different approaches. The three approaches to provided similar results for overall impact/kWh to biodiversity/ecosystem quality, where Langevatn had the highest impact, followed by Skjerkevatn, Kilandsfossen and Dvergfossen. This ranking was consistent with calculated m^2y/kWh using only the basic Land Use Change per kWh.

Impacts to wetland ecosystems were most important, for all methods, relative to the impacts on other ecosystems. Impacts to alpine ecosystems were more important when using ecosystem scarcity/vulnerability as an indirect indicator for the current condition of the affected ecosystems, relative to the other methods. The results for GHG emissions show the importance of including total LUC as a result of construction of infrastructure, and that this is especially important for smaller hydropower development projects, due to the relative importance of Land Use Change connected to infrastructure for small-scale hydropower.

All the approaches require mapping of Land Use both prior to and after Land Use Change. The technical drawings had a high level of detail and were suitable for determining Land Use after development. The maps used to determine the natural Land Use prior to development (N50) was a constraining factor for the analyses due to the low resolution, compared to other data sources. Further research is necessary to provide natural Land Use maps with higher resolution, and this would increase the accuracy and validity of the approaches used in this thesis, especially for the GHG emissions.

Restoration cost preformed equally well as the other approaches, as an indicator for biodiversity/ ecosystem quality. The methodology for calculation of restoration cost is coupled with applied restoration ecology, and is highly adaptable to future development restoration ecology. However, the total restoration cost of the case projects are probably not accurate, due to uncertainty related to offset ratios, the availability of offset sites and the exclusion of transaction costs. Incorporation of restoration cost into LCA, as an indicator for biodiversity/ecosystem quality will require further research, both in applied restoration ecology and appropriate methodology development for LULUC.

6 References

Aronson, J. et al., 2010. Are Socioeconomic Benefits of Restoration Adequately Quantified? A Meta-analysis of Recent Papers (2000-2008) in Restoration Ecology and 12 Other Scientific Journals. *Restoration Ecology*, 18(2), pp.143–154.

Bakken, T.H. et al., 2014. Demonstrating a new framework for the comparison of environmental impacts from small- and large-scale hydropower and wind power projects. *Journal of environmental management*, 140C, pp.93–101.

Barnosky, A.D. et al., 2011. Has the Earth's sixth mass extinction already arrived? *Nature*, 471(7336), pp.51–57.

Bullock, J.M. et al., 2011. Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in ecology & evolution*, 26(10), pp.541–9.

Business and Biodiversity Program (BBOP), 2013. *To No Net Loss and Beyond: An Overview of the Business and Biodiversity Offsets Programme (BBOP), 2nd ed.*, Washington, D.C.

Cherubini, F. & Strømman, A.H., 2011. Life cycle assessment of bioenergy systems: State of the art and future challenges. *Bioresource Technology*, 102(2), pp.437–451.

Chiarucci, A. et al., 2010. The concept of potential natural vegetation: an epitaph? *Journal of Vegetation Science*, 21(6), pp.1172–1178.

V. Coelho, C.R. & Michelsen, O., 2014. Land use impacts on biodiversity from kiwifruit production in New Zealand assessed with global and national datasets. *The International Journal of Life Cycle Assessment*, 19(2), pp.285–296.

Convention on Biological Diversity, 2010. Aichi Biodiversity Targets. Available at: http://www.cbd.int/sp/targets/ [Accessed July 6, 2014].

Curran, M. et al., 2011. Toward meaningful end points of biodiversity in life cycle assessment. *Environmental science & technology*, 45(1), pp.70–9.

Drescher, M. et al., 2008. *Boreal forest succession in Ontario: an analysis of the knowledge space*, Ontario.

Gederaas, L. et al. eds., 2012. *Fremmede arter i Norge - med nosk svarteliste 2012 (alien species in Norway - including Norwegian blacklist 2012)*, Trondheim: Norwegian Biodiversity Center.

Gotelli, N.J. & Colwell, R.K., 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters*, 4(4), pp.379–391.

Grønlund, A. et al., 2010. *CO2-opptak i jord og vegetasjon i Norge- Lagring , opptak og utslipp av CO2 og andre klimagasser. Bioforsk rapport 162.*, Ås, Norway.

Guo, L. & Gifford, R., 2002. Soil carbon stocks and land use change: a meta analysis. *Global change biology*, 8(4), pp.345–360.

Hagen, D. et al., 2013. Ecological and Social Dimensions of Ecosystem Restoration in the Nordic Countries. *Ecology and Society*, 18(4), p.34.

Hagen, D. et al., 2014. To seed or not to seed in alpine restoration: introduced grass species outcompete rather than facilitate native species. *Ecological Engineering*, 64, pp.255–261.

Hagen, D. & Erikstad, L., 2013. Arealbrukens betydning for miljøprofil i småkraftbransjen, med vekt på vei og rørgate. *Kart og Plan*, 73(4), pp.297–308.

Hagen, D. & Evju, M., 2013. Using Short-Term Monitoring Data to Achieve Goals in a Large-Scale Restoration. *Ecology and Society*, 18(3).

Hanski, I. & Walsh, M., 2004. *How much, how to? Practical tools for forest conservation.*, Helsinki.

Hertwich, E.G., 2013. Addressing biogenic greenhouse gas emissions from hydropower in LCA. *Environmental science & technology*, 47(17), pp.9604–11.

Houghton, R. a. et al., 2012. Carbon emissions from land use and land-cover change. *Biogeosciences*, 9(12), pp.5125–5142.

IPCC, 2003. *Good Practice Guidance for Land Use, Land Use Change and Forestry – appendix section 3a.3.3 Flooded Land remaining Flooded Land*, Japan.

ISO 14040, 2006. *ISO 14040:2006 - Environmental management -- Life cycle assessment -- Principles and framework*,

Jorgenson, J.C., Ver Hoef, J.M. & Jorgenson, M.T., 2010. Long-term recovery patterns of arctic tundra after winter seismic exploration. *Ecological applications : a publication of the Ecological Society of America*, 20(1), pp.205–21.

Koellner, T. et al., 2013. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. *The International Journal of Life Cycle Assessment*, 18(6), pp.1188–1202.

Laiho, R., 2006. Decomposition in peatlands: Reconciling seemingly contrasting results on the impacts of lowered water levels. *Soil Biology and Biochemistry*, 38(8), pp.2011–2024.

Larsson, T.B. ed., 2001. *Biodiversity evaluation tools for European Forestes*, Oxford: Ecological Bulletins 50, Blackwell Science.

MacArthur, R. & Wilson, O., 1967. *The theory of island biogeography*, Princeton: Princeton University Press.

McKenney, B.A. & Kiesecker, J.M., 2010. Policy development for biodiversity offsets: a review of offset frameworks. *Environmental management*, 45(1), pp.165–76.

Michelsen, O., 2008. Assessment of Land Use Impact on Biodiversity Proposal of a new methodology exemplified with forestry operations in Norway. , 13(1), pp.22–31.

Milà i Canals, L. et al., 2007. Key Elements in a Framework for Land Use Impact Assessment Within LCA (11 pp). *The International Journal of Life Cycle Assessment*, 12(1), pp.5–15.

Miljødirektoratet, 2013. Påvirkning av vann og vassdrag. Available at: http://www.miljodirektoratet.no/no/Tema/Vannforvaltning/Vann-og-vassdrag/Pavirkning-av-vann-og-vassdrag/ [Accessed July 6, 2014].

Millenium Ecosystem Assessment, 2005. *Ecosystems and Human Well-being: Biodiversity Synthesis*, Washington, D.C.

Moilanen, A. et al., 2009. How Much Compensation is Enough? A Framework for Incorporating Uncertainty and Time Discounting When Calculating Offset Ratios for Impacted Habitat. *Restoration Ecology*, 17(4), pp.470–478.

Moreau, V. et al., 2012. Statistical estimation of missing data in life cycle inventory: an application to hydroelectric power plants. *Journal of Cleaner Production*, 37, pp.335–341.

Moreno-Mateos, D. et al., 2012. Structural and functional loss in restored wetland ecosystems. M. Loreau, ed. *PLoS biology*, 10(1), p.e1001247.

NOU, 2009. *Globale miljøutfordringer - norsk politikk. Hvordan bærekraftig utvikling og klima bedre kan ivaretas i offentlige beslutningsprosesser*, Oslo.

Nybø, S. ed., 2010. *Naturindeks for Norge 2010. DN-utredning 3*, Trondheim: Direktoratet for naturforvaltning.

Penariol, L. V & Madi-Ravazzi, L., 2013. Edge-interior differences in the species richness and abundance of drosophilids in a semideciduous forest fragment. *SpringerPlus*, 2(1), p.114.

Peter, D. et al., 1998. *LCA graphic paper and print products (part 1, long version).*, Infras AG (Zürich), Axel Springer Verlag AG (Hamburg), Stora (Falun, Viersen) and Canfor (Vancouver).

Quintero, J.D. & Mathur, A., 2011. Biodiversity offsets and infrastructure. *Conservation biology : the journal of the Society for Conservation Biology*, 25(6), pp.1121–3.

Rasche, L., Fahse, L. & Bugmann, H., 2013. Key factors affecting the future provision of tree-based forest ecosystem goods and services. *Climatic Change*, 118(3-4), pp.579–593.

REN21, 2013. *Renewables 2013 Global Status Report*, Paris: REN21 Secretariat.

Rockström, J., Steffen, W. & Noone, K., 2009. A safe operating space for humanity. *Nature*, 461(September), pp.472–475.

Saenz, S. et al., 2013. A Framework for Implementing and Valuing Biodiversity Offsets in Colombia: A Landscape Scale Perspective. *Sustainability*, 5(12), pp.4961–4987.

Samferdselsdepartementet, 2013. *Kompensasjon av jordbruks- og naturområder : Litteraturstudie med anbefalinger og vurderinger av kostnader*, Oslo.

Simil, M. & Junninen, K. eds., 2012. *Ecological restoration and management in boreal forests - best practices from Finland*, Helsinki: Metsähallitus.

Similä, M. & Junninen, K. eds., 2012. *Ecological restoration and management in boreal forests*, Helsinki: Metsähallitus.

Society for Ecological Restoration, 2004. SER International Primer on Ecological Restoration. Available at: http://www.ser.org/resources/resources-detail-view/ser-international-primer-on-ecological-restoration [Accessed July 6, 2014].

Suding, K.N., 2011. Toward an Era of Restoration in Ecology: Successes, Failures, and Opportunities Ahead. *Annual Review of Ecology, Evolution, and Systematics*, 42(1), pp.465–487.

TEEB, 2010. The Economics of Ecosystems and Biodiversity (TEEB) Ecological and Economic Foundations. In P. Kumar, ed. *The Economics of Ecosystems and Biodiversity (TEEB) Ecological and Economic Foundations*. London and Washington: Earthscan, p. 456.

The Environment Bank, 2013. Biodiversity Offsetting. , (July). Available at: http://www.environmentbank.com/files/general-guidejuly13-1.pdf [Accessed October 15, 2013].

Tremblay, A. et al., 2005. GHG Emissions from Boreal Reservoirs and Natural Aquatic Ecosystems. In A. Tremblay et al., eds. *Greenhouse Gas Emissions - Fluxes and Processes*. Environmental Science. Springer Berlin Heidelberg, pp. 209–232.

UNEP, 1992. Convention on biological diversity. *Diversity*.

Walker, B., Kinzig, A. & Langridge, J., 1999. Original Articles: Plant Attribute Diversity, Resilience, and Ecosystem Function: The Nature and Significance of Dominant and Minor Species. *Ecosystems*, 2(2), pp.95–113.

Wearn, O.R., Reuman, D.C. & Ewers, R.M., 2012. Extinction debt and windows of conservation opportunity in the Brazilian Amazon. *Science (New York, N.Y.)*, 337(6091), pp.228–32.

Weidema, B. & Lindeijer, E., 2001. *Physical impacts of land use in product life cycle assessment. Final report of the EURENVIRON-LCAGAPS sub- project on land use.*, Lyngby.

Wolters, V., Bengtsson, J. & Zaitsev, A.S., 2006. Relationship among the species richness of different taxa. *Ecology*, 87(8), pp.1886–1895.

Xu, Z. et al., 2006. No TitleEffects of nutrient and water level fluctuation on wetland plants. *Chinese Journal of Ecology*, 25(1), pp.87–92.

Aapala, K., Similä, M. & Penttinen, J. eds., 2013. *Ojitettujen soiden ennallistamisopas (Handbook for the restoration of drained peatlands)*, Helsinki: Metsähallitus.

7 Appendix

7.1 EcoManage WP4: Ecosystem services and biodiversity offsets

WP 4: Eco-system services and biodiversity offsets (Lead: David N. Barton, NINA)

We aim to combine methods of habitat equivalency analysis, ecosystem services valuation using rehabilitation cost approaches, multi-criteria (decision) analysis (MCDA) of offsetting alternatives and stakeholder participatory methods to address the questions of whether biodiversity offsetting in the development of 'peak power' is ecologically and economically feasible. We would aim to conduct an evaluation of habitat equivalency and restoration costs of sites that would offset impacts within watershed that are candidates for peak power regulation and land possibly under transmission lines.

An evaluation of offsets would be carried out for hypothetical alternatives for offsetting within the same catchment, and hypothetical alternatives in a different catchment, most likely in South-western Norway (co-ordinated with submitted HydroBalance proposal). MCDA would be used to structure the habitat equivalency analysis, as well as economic and social impacts at the deteriorated and alternative rehabilitated sites. Stakeholders would participate in the structuring of the multi-criteria problem, in social impact evaluation and in consideration of the relative weighting of criteria using participatory MCDA techniques (Sparrevik et al., 2011). We will compare the MCDA approach to evaluating the impact of ecosystem services with conceptualization used in LCA analysis. Using the evaluation criteria developed by EFTEC (2010) would also assess data, legal and institutional requirements for a ecosystem services and biodiversity offset scheme to work in the Norwegian context. We would also evaluate the distributions of costs and benefits between local stakeholders, developers and national and foreign consumers of a biodiversity offsets scheme in the context of peak power development.

In WP4 different methodologies will be used to quantify and evaluate both potential positive and negative impacts on Ecosystem Services in design/redesign, construction and operation of hydropower plant systems and offset schemes. The LCA approach will develop and test methodologies for quantification of positive and negative impacts on Ecosystem Services based on assessments of the quantity and quality of habitat resources that are in the influence area of hydro power projects.

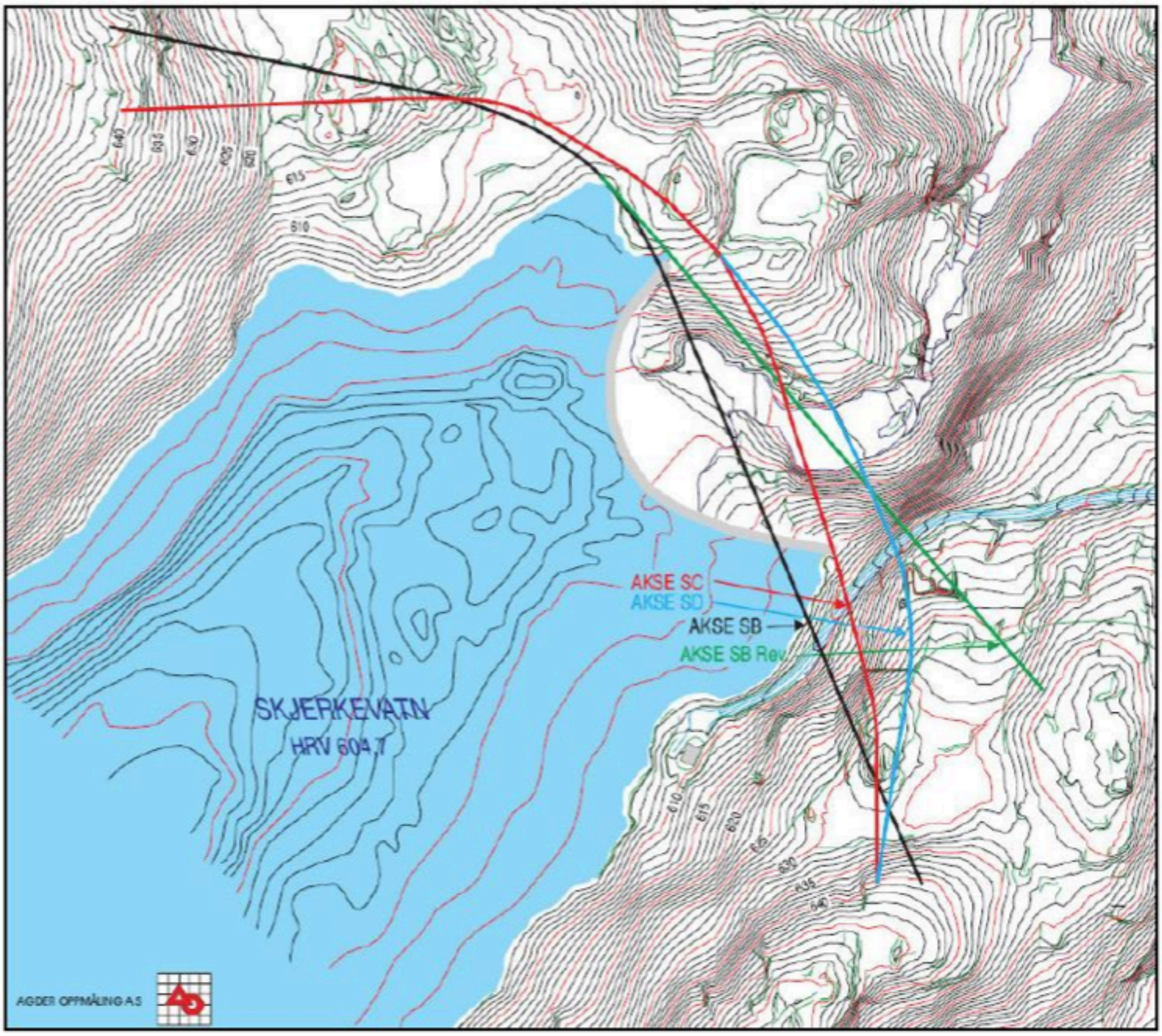
Relevant approaches will be to analyze impacts on vulnerable natural habitats as defined in the newly published reports from Artsdatabanken in Norway (Artsdatabanken, 2010). Another approach will be to use available data from Environmental Impact Assessments and GIS data, to assess the impacts on biodiversity and bioproductivity of the influence area (Geyer et al. 2010a, b).

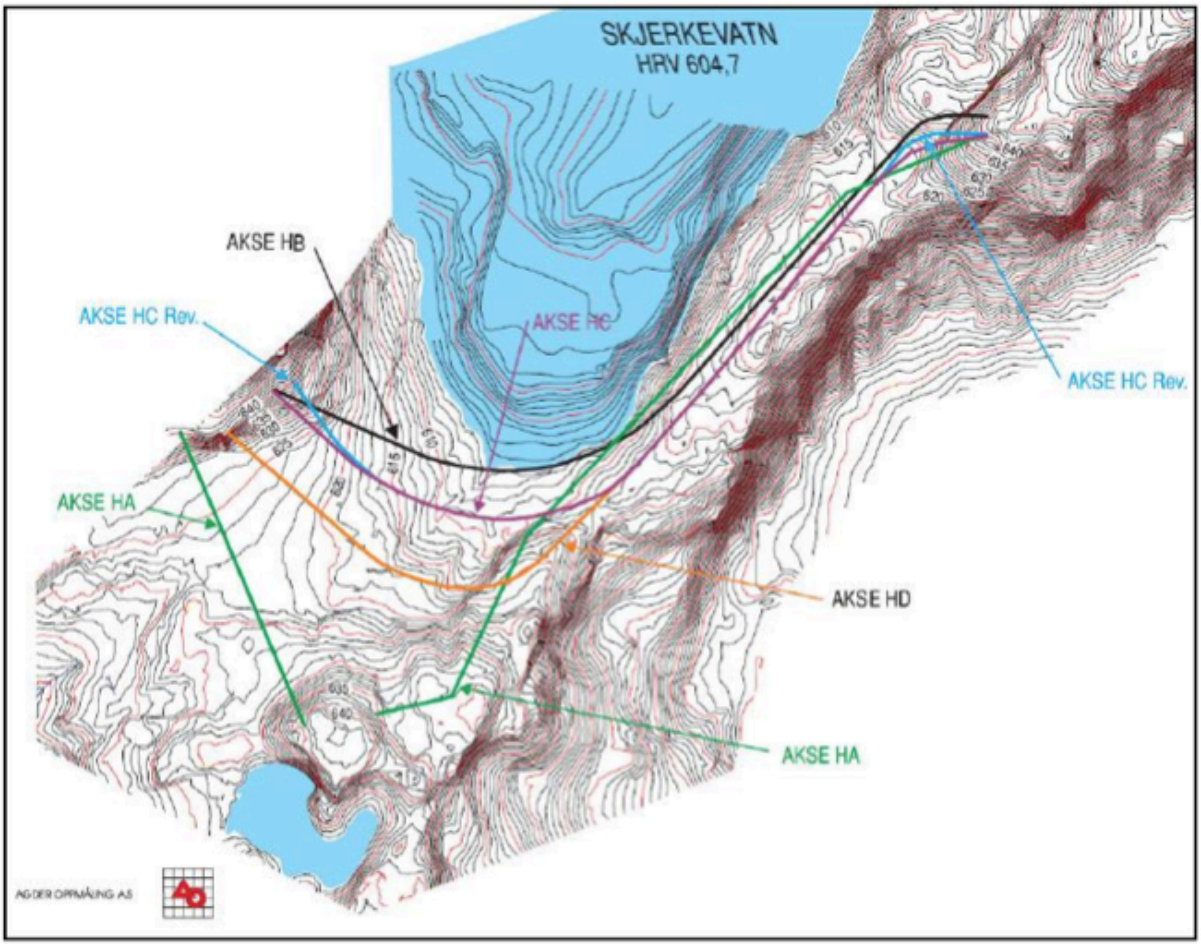
MCDA will be applied in providing a framework for information structuring and analysis when several, conflicting criteria are important for a decision and as a framework for structuring and documenting stakeholder participation. 'Social or participatory' MDCA techniques are therefore an approach to organizing assessments when many stakeholders are likely to be involved. MCDA is deemed a suitable approach to support decision making for regulated rivers, for instance when defining multi-objective environmental goals, prioritizing of mitigating measures and negotiating conflicting interests. The use of MCDA in the ecosystem services study will be carried out in consultation with a reference group of concerned stakeholders.

Deliverables: 3 refereed journal publications on:
1. Habitat equivalency and offsetting of peak power impacts in terrestrial and aquatic ecosystems (lead authors: Finstad/Hagen).
2. Trade-offs and ranking of offset sites using multi-criteria analysis (lead author: Barton)
3. An evaluation of biodiversity offsets and habitat banking in the hydropower sector in Norway: instrument design and policy (lead author: Lindhjem).

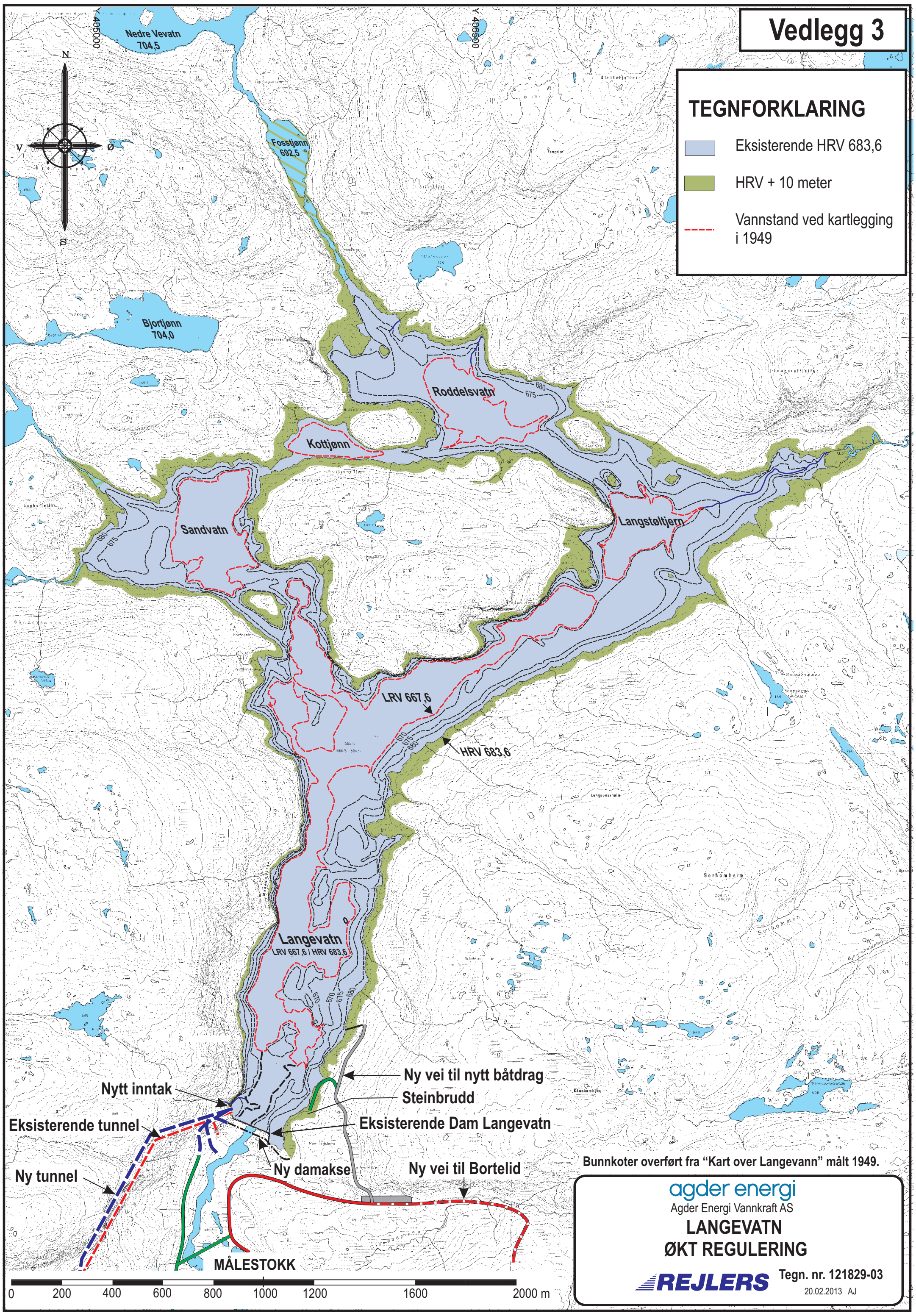
7.2 TECHNICAL DRAWINGS

Skjerkevatn

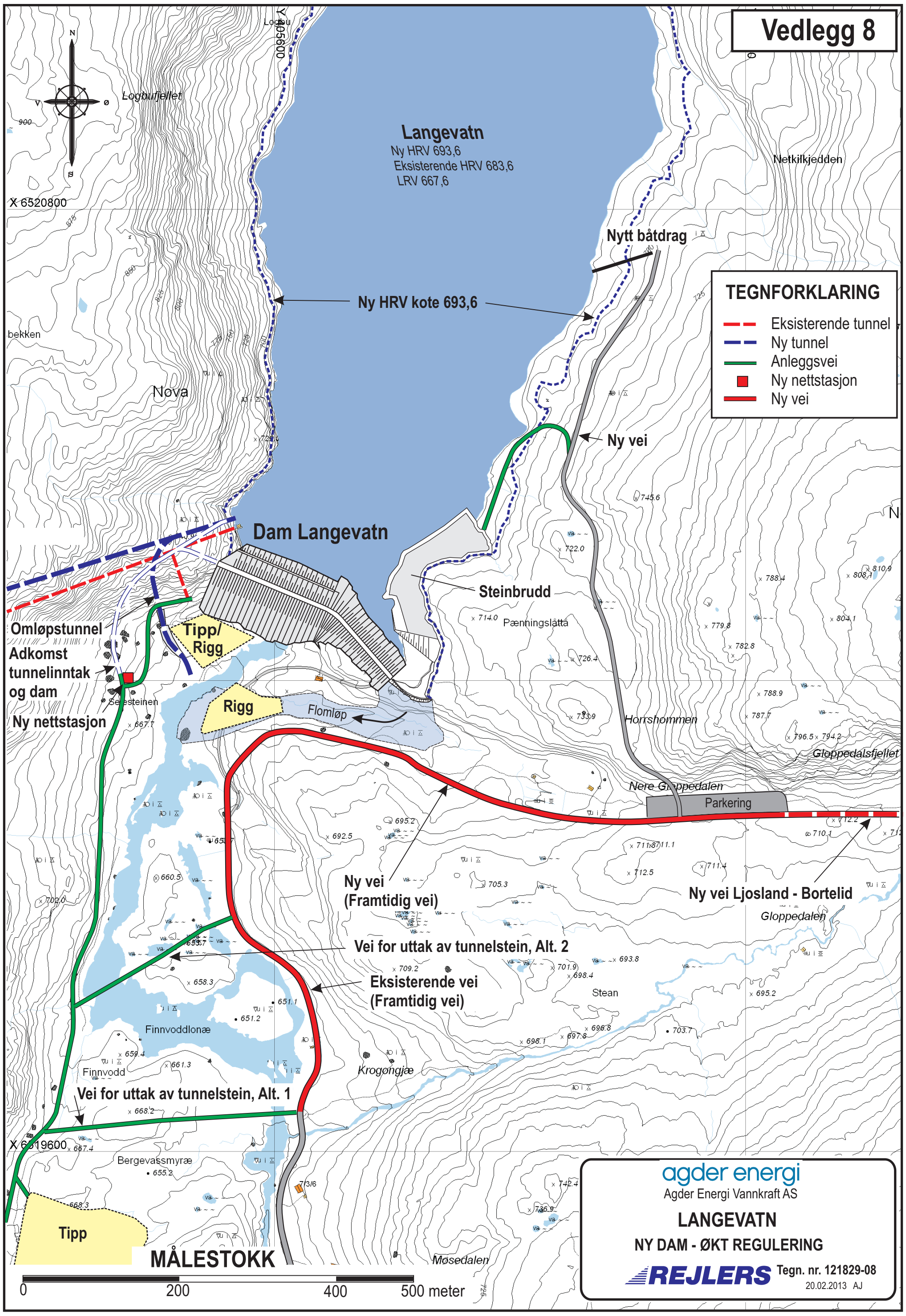



SKJERKEVATN
HRV 604,7
AKSE HB
AKSE HC Rev.
AKSE HC
AKSE HC Rev.
AKSE HA
AKSE HD
AKSE HA
AGDER OPPMÅLING AS

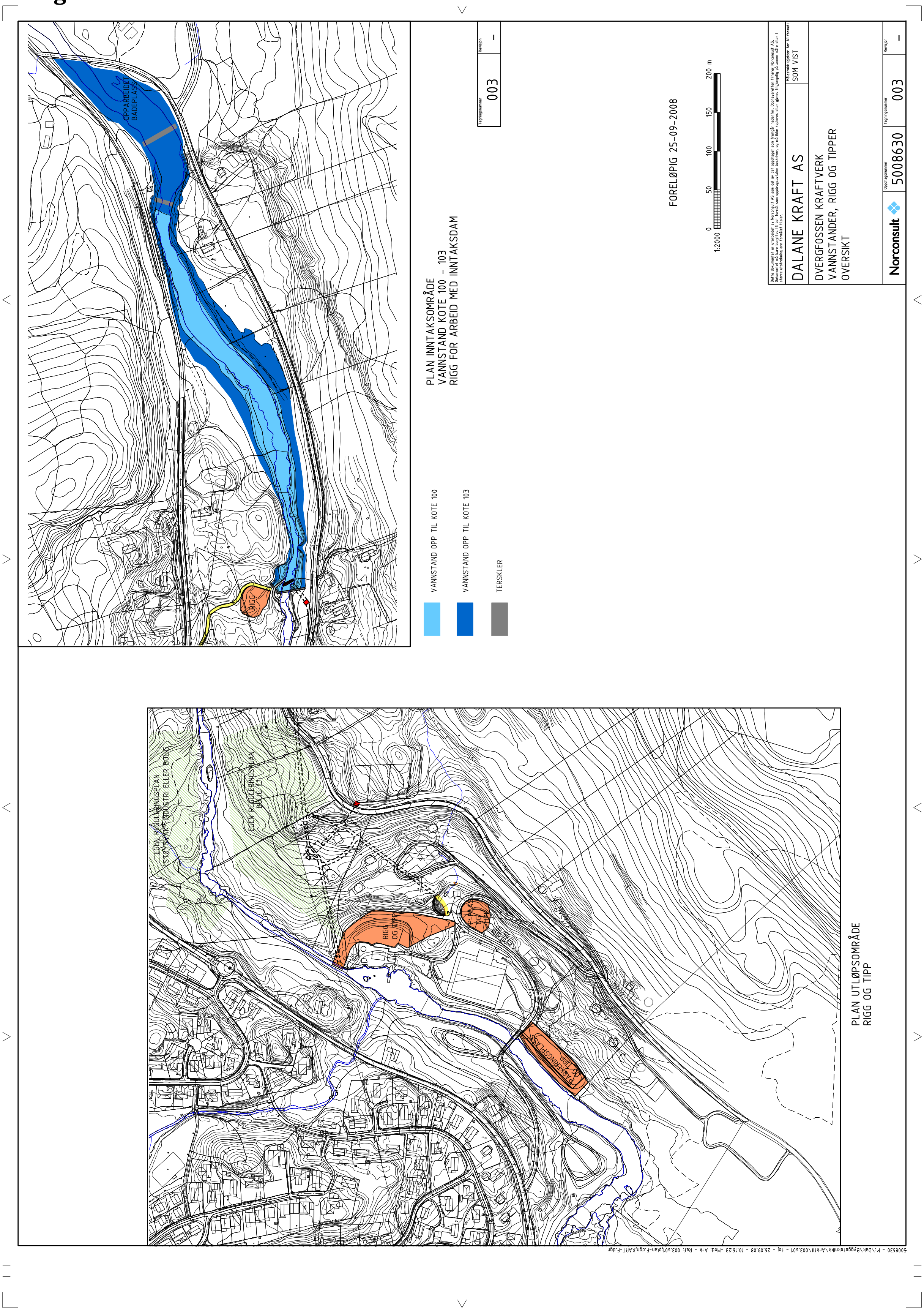
Langevatn



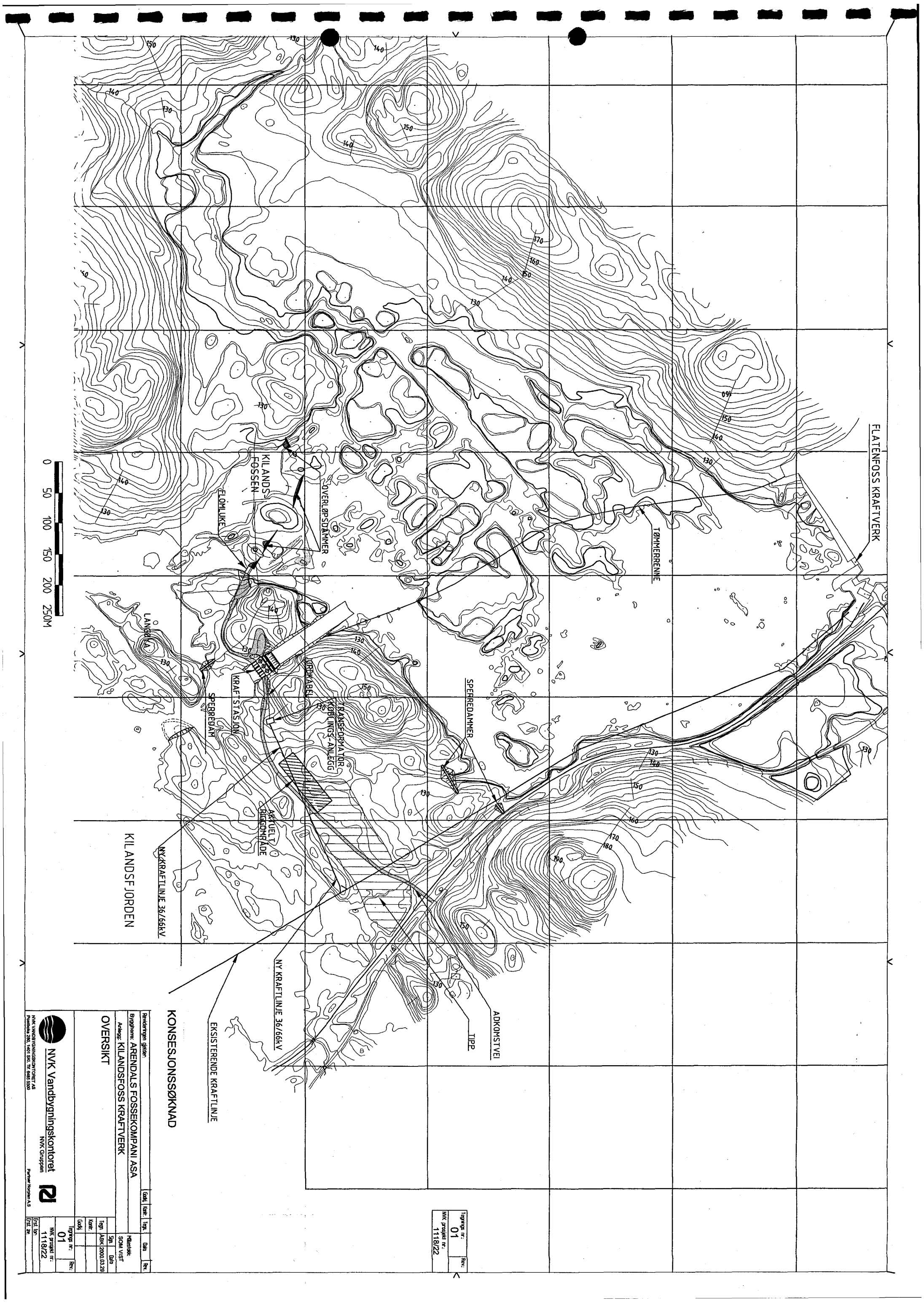
Vedlegg 3
TEGNFORKLARING
Eksisterende HRV 683,6
HRV + 10 meter
Vannstand ved kartlegging i 1949
Nedre Vevatn 704,5
Fosstjønn 692,5
Bjortjønn 704,0
Roddelsvatn
Kottjønn
Sandvatn
Langstøltjern
LRV 667,6
HRV 683,6
Langevatn
LRV 667,6 / HRV 683,6
Nytt inntak
Eksisterende tunnel
Ny tunnel
Ny vei til nytt båtdrag
Steinbrudd
Eksisterende Dam Langevatn
Ny damakse
Ny vei til Bortelid
MÅLESTOKK
0 200 400 600 800 1000 1200 1600 2000 m
Bunnkoter overført fra "Kart over Langevann" målt 1949.
agder energi
Agder Energi Vannkraft AS
LANGEVATN
ØKT REGULERING
REJLERS
Tegn. nr. 121829-03
20.02.2013 AJ


Vedlegg 8
Langevatn
Ny HRV 693,6
Eksisterende HRV 683,6
LRV 667,6
Nytt båtdrag
TEGNFORKLARING
Eksisterende tunnel
Ny tunnel
Anleggsvei
Ny nettstasjon
Ny vei
Ny HRV kote 693,6
Ny vei
Dam Langevatn
Steinbrudd
Omløpstunnel
Adkomst tunnelinntak og dam
Ny nettstasjon
Tipp/ Rigg
Rigg
Flomløp
Parkering
Ny vei (Framtidig vei)
Ny vei Ljosland - Bortelid
Vei for uttak av tunnelstein, Alt. 2
Eksisterende vei (Framtidig vei)
Vei for uttak av tunnelstein, Alt. 1
Tipp
MÅLESTOKK
0
200
400
500 meter
agder energi
Agder Energi Vannkraft AS
LANGEVATN
NY DAM - ØKT REGULERING
REJLERS
Tegn. nr. 121829-08
20.02.2013 AJ
X 6520800
X 6519600
Logburfjellet
Nova
Netkilkjedden
Horrshommen
Gloppedalsfjellet
Nere Gloppedalen
Gloppedalen
Pænningslåtta
Finnvoddlonæ
Finnvodd
Krogongjæ
Bergevassmyræ
Stean
Mosedalen

Dvergfossen



Kilandsfossen



7.3 AREA DATA – SSB 2013

		2011	2013
		Areal	Areal
07 Vestfold	**Barskog**	1226,14	1221,7
	Blandingsskog	111,57	111,21
	Lauvskog	90,71	90,15
	Uklassifisert skog	0	0
	Åpen fastmark	35,75	35,49
	Våtmark	14,61	14,59
	Totalt	1478,78	1473,14
08 Telemark	**Barskog**	6715,07	6708,76
	Blandingsskog	511,89	507,78
	Lauvskog	567,43	566,85
	Uklassifisert skog	372,85	372,43
	Åpen fastmark	4424,96	4427,02
	Våtmark	447,89	446,96
	Totalt	13040,09	13029,8
09 Aust-Agder	**Barskog**	3910,93	3914,05
	Blandingsskog	517,84	512,41
	Lauvskog	349,31	350,98
	Uklassifisert skog	230,67	230,59
	Åpen fastmark	2424,01	2424,51
	Våtmark	389,31	387,75
	Totalt	7822,07	7820,29
10 Vest-Agder	**Barskog**	2036,87	2034,06
	Blandingsskog	478,4	477,64
	Lauvskog	1008,14	1003,28
	Uklassifisert skog	0	0
	Åpen fastmark	2383,24	2390,05
	Våtmark	295,3	287,49
	Totalt	6201,95	6192,52
Totalt	**Barskog**	13889,01	13878,57
	Blandingsskog	1619,7	1609,04
	Lauvskog	2015,59	2011,26
	Uklassifisert skog	603,52	603,02
	Åpen fastmark	9267,96	9277,07
	Våtmark	1147,11	1136,79
	Totalt	28542,89	28515,75

7.4 DETAILS ON COST OF RESTORATION MEASURES

Restaureringskostander for skog og myr i Finland. Kilde: Anne Tolvanen og restaureringshåndbøkene

Peatland (Handbook Peatland restoration)	
Cutting timber	14,23 Euro/m3
Transport to factory	8,16
Cut energywood	20-35
Gjøres manuelt	25-63
Flytte tømmeret gjennom peatland	4-14
Cutting along ditches (using manpower)	0,5-1,5 pr m
Filling ditches on large peatland (excavator)	0,45-1,2 pr m
Forest – adding decaying wood	
Manuelt	5-25 pr m3
Exchavator	4-15 pr m3
Forest – lage små glenner	
Manual	15-40
Exchavator	5-20
Forest – lage kunstige rotvelter (storm-simulation), kun maskinelt	Sannsynligvis ca 5-25

Fra Hjerkinn, Pilot

Lyng-/lavhei (Pilot Mogop)		
Gj.snitt – for hele vegen	0,30 time/ løpemeter veg	Inkluderer fjerning av toppmasser ned til opprinnelig terreng (1,6 m3 pr meter veg), terrengarrondering og utsetting av vegetasjonstorver fra grøfta og inn i vegen på deler av strekningen (ikke transport av torvene)
		Der toppgrusen ikke ble fjernet (bare blandet inn) gikk tidsbruken ned til 0,21. Der den ble fjernet ligger tidsbruken på mellom 0,26 og 0,35.
Fuktdrag med noe vierkratt og overgang til lynghei (Pilot Vier)		
Gj.snitt for hele vegen	0,28 time/ løpementer veg	Her er det større variasjon langs vegen, enhetstida er mye mindre der det ikke er fjernet toppmasser (0,07t/m). Og mye høyere der det er fuktig kratt og mere innflytting av vegetasjon, samt tilførsel av masser for å gjenopprette terreng (0,48t/m)
Grasmyr	1,36 time/m	Tilbakeføring av masser som var lagt å sida da vegen ble bygd, tilpasning til sideterreng og innplanting av torver med stedegen vegetasjon.

*Timebruket er summen av tida for graving og for bortkjøring av overskuddsmassene.

Fordeling av metoder for fjell

- Fjerne veg/tilførte toppmasser 100%
- Gjødsle /så (Fjellfrø) m ref 30%
- Plante med busker 5%

Enhetspriser (=pris pr time m maskin og sjåfør) for anleggsmaskiner (basert på opplysninger direkte fra bransjen)

- 40 tonn gravemaskin (800,-)
- 14 og 20 tonn gravemaskin (700,-)
- Dumper (700,-)
- Lastebil (700,-)
- Hjullaster (700,-)

Dette er priser uten mva, dvs prisen er oppimot 1000,- per time til sammen.

7.5 DETAILED LAND USE DATA FOR THE MAPPED LAND USE CHANGE

All values in square meters

		Permanent infrastructure	**Temporary infrastructure**	**Reservoir**	Total
Skjerkevatn	Total area	189904,60	134202,90	727744,50	**1051852,00**
	Alpine	113334,30	99238,00	376903,90	589476,20
	Freshwater	707,20	1870,30	1494,60	4072,10
	Wetland	25787,70	17660,90	48593,00	92041,60
	Forest	50075,40	15433,70	300753,00	366262,10
Langevatn	Total area	148221,59	48078,41	565835,01	**762135,02**
	Alpine	18762,38	10576,02	3513,12	32851,52
	Freshwater	7986,32	618,35	34627,09	43231,77
	Wetland	9177,81	5210,50	28342,76	42731,07
	Forest	112295,08	31673,54	499352,04	643320,66
Dvergfossen	Total area	17819,90	1336,32	21374,86	**40531,07**
	Alpine	5132,81		7849,56	12982,38
	Freshwater	204,47	0,10	3801,19	4005,76
	Wetland				0,00
	Forest	12482,61	1336,22	9724,11	23542,94
Kilandsfossen	Total area	117006,37	0,00	58026,34	**175032,71**
	Alpine	230,62		26,50	257,12
	Freshwater	19220,17		5039,98	24260,15
	Wetland	13604,70		11775,64	25380,34
	Forest	83950,89		41184,21	125135,10