Water footprint approaches in Life Cycle Assessment: State-of-the-art and a case study of hydroelectric generation in the Høyanger area.

Vannfotavtrykkstilnærminger i LCA: State-of-the-art og et case studie av vannkraft produsert i Høyanger.

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Preface

This Master thesis is the result of half a years research done at the Norwegian University of Life Sciences. I was first introduced to Life Cycle Assessment through a University course. My interest for the subject grew after completing a summer internship at Ostfold Research.

A number of people have helped me along the way to finish this thesis, a way that has at times seemed impossible to finish. Ole Jørgen Hanssen: as my supervisor you have been inspirational and supportive. Thank you for quick feedback, good input, educational meetings, and for always meeting me with warmth and motivation. Kaja Henny Engebrigtsen, dear friend and co-student: thank you for the coffees, the input and this experience. Jan Riise at Statkraft: thank you for quick replies to my emails and good answers. My housemates: thank you for keeping me equipped with clean socks and a nutritious diet. Family and friends: thank you for picking up the phone, for supportive text messages and general tender love and care. I would like to direct a special thanks to my mom for all that she does and all that she is.

It has been a fruitful and giving experience for me to complete this work.

Abstract

Life Cycle Assessment (LCA) currently fails to include impacts of freshwater use, and specification of water accounts in terms of geography and quality. Water footprints can be included in LCA to account and assess freshwater use, in combination with traditional methods such as Environmental Impact Assessment (EIA) and Risk Assessment (RI). A literature review is utilized to present the state-of-the-art for water footprint methods than can be combined with LCA. Based on identified methods, a theoretical framework for LCA of hydroelectric generation including water footprints is presented. Water footprint studies based on global averages have presented high water footprint values. The Høyanger hydropower scheme is used as a case study, assessing the change in water footprint values resulting from an impending upgrading and expansion of the power scheme in the area. This Master thesis calculate the water footprint of electricity produced in the Høyanger power scheme, using two methods. The first method (water footprinting according to Hoekstra) considers water losses through evaporation only. The second method (WF-3 developed by Herath and colleagues) accounts for both water inputs through precipitation and water losses through evaporation. Both methods produced significantly lower results than what have been presented in other water footprint studies of hydroelectric generation, with global average values of 68 and 22 m^3/GJ. Employing the first method, weighted average water footprints decreased in value from 1.21 to 1.05 m^3/GJ, indicating benign impacts of freshwater use changes as a result of the upgrading and expansion. The second method produced negative values for all studied power plants, indicating that the Høyanger region collects more precipitation than it loses through evaporation. Accounting for water inputs in addition to water outputs provides information of the water stress in the region. To add such information to the water footprints produced utilizing the first method, these were characterized according to a stress water index provided by Pfister and colleagues. The characterized values decreased from 0.013 to 0.012 m^3/GJ. LCA can provide accounts for potential impacts of freshwater use. For complete qualitative environmental assessment, LCA should be used in a combination with EIA and RI. The EIA for the upgrading and expansion project was reviewed to add qualitative environmental information to the analysis.

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1 Introduction

1.1 Background

Freshwater is a basic necessity of life, with nearly all-human activity depending on it. Securing freshwater access is therefore imperative to human life. Although freshwater is a renewable resource, its form, quantity and quality vary with geographical locations (Pindoria, 2010). In many regions of the world freshwater is regarded as a scarce resource. This is in part due to changing climates, part poor water management strategies, and parts of the scarcity problem find explanation in natural factors. For regions suffering from freshwater scarcity it is imperative that water is sustainably managed in order to secure access. Global trade leads to virtual water trading, impacting the freshwater footprint of all countries (Allan, 2011). There is a common responsibility to secure freshwater access for all the peoples of the world; even regions not suffering water scarcity have to manage their water in a sustainable manner. Freshwater can only be managed sustainably if decision-makers make sustainable decisions. Life cycle Assessment (LCA) is a tool developed to assist sustainable decision-making processes.

Another necessity of modern life is energy. There are strong links between freshwater resources and energy, making the two highly connected and interdependent (Koehler, 2008). Freshwater resources are inputs to energy production systems, and energy is necessary for operating modern water systems (Gleick, 1994).

The world is experiencing tremendous population growth. With increasing populations come increasing freshwater demand. Population growth intensifies the need for freshwater resource management in order to supply the growing population with sufficient freshwater. In 2010 the Institute of Electrical and Electronics Engineers (IEEE) stated that competition for freshwater and energy will become one of the central challenges for this century (IEEE, 2010). Predicted population growth will induce increased water and energy demands by 30 and 50%, respectively (Beddington, 2009). It will be a substantial challenge to meet these demands without affecting natural capital stocks and ecosystem services that flow from them.

Freshwater scarcity may induce significant problems for human health, social and political stability, and ecosystems (Humbert et al., 2010). Methodologies to identify effective action are lacking and there is an identified need to measure, act and communicate the effects of freshwater use (Humbert et al., 2010 op. cit.). The UNEP-SETAC Life Cycle Initiative and the International Organization for Standardization (ISO) have initiated projects to develop consistent frameworks for the assessment of freshwater use in LCA based on scientific review of available methods and international agreement. Sustainable water management is also part of the UN millennium goals under the section for Environmental Sustainability.

1.2 Life Cycle Assessment (LCA)

Life Cycle Assessment is a methodology aspiring to measure the aggregated environmental impacts of a product or service, from cradle to grave. It is a widely accepted method for environmental assessment. Historically the focus of LCA was energy use along the supply chain, and emissions of climate gasses and toxic substances such as acidification, eutrophication, POCPs etc. (Berger & Finkbeiner, 2010). Consumption and quality alterations of water resources, upstream and downstream, have to a large extent been ignored in LCA. This can be attributed to several reasons, but can to some extent be explained by its historical uses: LCA was developed in industrialized countries, often without water shortages. Furthermore, LCA was initially used to assess industrial product chains, where water consumption is a low fraction of the total input (Koehler, 2008). In a constantly developing methodology, with broadening scopes for the assessments, not accounting for fresh water resources is a limitation for LCA. Scientific research is continuously carried out, and there is consensus within the field that methods for including freshwater use in LCA must be developed. Not being able to account for fresh water resources is a serious lack for a method developed to assist sustainable decision-making processes, and is even claimed to be in conflict the ISO 14040:2006 ‘comprehensiveness’ requirement (Berger & Finkbeiner, 2010).

1.3 Water footprinting

As an indicator of freshwater use, the water footprint concept has gained interest since its introduction by Hoekstra in 2002. Product/service water footprints are defined as the volume of freshwater used, directly and indirectly, in the production of a good or service (Mekonnen & Hoekstra, 2011a). Freshwater use, in this context, are freshwater resources not returned to the system it belonged to originally, and/or freshwater that is integrated into a good or service during production. How to define freshwater use, and what elements to include in this definition, is source for discussion within the research field. This will be explored in greater detail later in this thesis.

Current state-of-the-art has presented water footprint results for hydroelectric generation as high as 68 m^3/GJ (Mekonnen & Hoekstra, 2011a), with an often-cited world average of 22 m^3/GJ (Gerbens-Leenes et al., 2009). Herath et al. (2011) challenged these high values, claiming that the methodology behind the averages contains thoughtful lacks. The results of the study by Herath and colleagues present significantly lower values for studied hydropower plants in New Zealand, in the range of 1.55-6.05 m^3/GJ.

In light of available research and method, it could be reasonable to assume that Norwegian hydroelectric generation, due to climatic factors and water availability, has a significantly lower water footprint than the global average values.

1.4 Hydroelectricity

Dam construction has played a historic role in human development, facilitating significant social and economic progress. 30-40% of irrigated land worldwide depends on dam water (World Commission on Dams, 2000). In 2009 hydropower accounted for 16% of the world's electric generation (IEA, 2009). Dam construction and subsequent hydroelectric generation impacts surrounding areas, positively and negatively. (Sternberg, 2008) listed the following as potential effects on people and ecosystems downstream of large hydropower dams: positive effects include regulation of river flows; storage of water to guarantee supply in dry periods; flood controlling; irrigation of agricultural lands, and provision of navigation and electricity supply. Negative effects include displacement of people; loss of land, and alteration of river flows and water quality.

Hydropower is an attractive energy source primarily because of its low operating costs per unit produced, low CO_2 emissions, and renewable character. Compared to other energy sources, not counting bio crops, the established water footprint for hydropower is large (Gerbens-Leenes et al., 2008; Mekonnen & Arjen Y. Hoekstra 2011b). For all countries depending on hydropower, or energy intensive goods or services where the energy source is hydropower, the accuracy of water footprint calculations is very important. Imprecise or incorrect water footprints can potentially impact the competitiveness and reputation of produced goods and services. This aspect is specially prominent for so-called green goods and services (Herath et al., 2011).

In some regions of the world water is readily available and highly renewable. Such regions often depend on hydroelectricity as their source of energy. With increasing energy demands it is likely that wet regions will continue to depend on, and further develop hydroelectric generation (Mekonnen & Hoekstra, 2011a). Predicted future energy shortages may also lead to hydropower developments in regions with less readily available water due to low operation costs and minimal associated CO_2 emissions, the focus of today's climate debate. It is important to assess such hydropower development in in terms of environmental impact and water footprint to secure sustainable allocation of existing freshwater resources. Incorporating water footprinting methods into the current LCA framework can assist comprehensive environmental assessment of hydropower development.

1.5 The case: Eiriksdal

The Norwegian state owned Energy Company, and Europe's largest producer of renewable energy, Statkraft AS, is planning an extensive upgrading and expansion process of the Høyanger hydropower scheme, situated on the West Coast of Norway. The project sets out to be an environmentally positive process returning substantial areas to their pre-development state, and increasing current electricity generation. The constructing company Skanska Norway AS will carry out a Life Cycle Assessment of the upgrading process. To get a more comprehensive LCA, it would be beneficial to include freshwater use and related environmental impacts in this the impending LCA. Water footprints could, in this context, be employed as a resource indicator of freshwater use.

2 Objectives

The central aim of this Master thesis is to present the state-of-the-art for water footprinting methodologies. Through a discussion of identified methods, the methods best suited for Life Cycle Assessment of hydropower production will be presented in a theoretical methodological framework, and data requirements of this framework will be accounted for. The Environmental Impact Assessment (EIA) conducted for the case study will be reviewed in conjunction to the data requirements, to explore potential data links between the two. Employing one or more of the identified methods, this Master thesis furthermore aspires to collect and analyze availability of relevant data in order to estimate operational water footprint values for the hydroelectric generation in the Høyanger area. By evaluating the difference between operational water footprints of the current hydropower scheme and the forthcoming, this Master thesis will attempt to indicate if impacts of freshwater resource use increase or decrease with the impending upgrading and expansion of the Høyanger production facilities. Research questions for this master thesis are:

1. Give an overview of water footprint methodologies in available literature and evaluate how these methods are included in present methodologies for LCA in general and for hydroelectricity specifically.
2. Present a theoretical methodological framework that can be used to assess water footprints of hydroelectric generation.
3. Evaluate how assessments of relevance for water footprints have been accounted for in the present Environmental Impact Assessment and permit for reconstruction of the Eiriksdal Hydropower plant.
4. Evaluate if the data necessary to calculate the water footprint for generation of electricity from Eiriksdal Power plant is available.
5. Discuss further needs for research in the field, to be able to include the water footprint as a resource indicator in LCA of hydroelectric generation.

3 Life Cycle Assessment (LCA)

3.1 Method[1]

Life Cycle Assessment is a methodology that systematically describes pollution and resource use related to delivering a specific product or service. As the name Life Cycle Assessment implies, the environmental assessment is based on pollution and resource use along the complete supply chain, from raw material input to production, throughout the 'use phase' of the product, until waste or recycling. Methodologically the assessment is done in four steps. It is common to divide between descriptive LCA and change or effect-oriented LCA. Descriptive LCA focus on mapping environmental impacts from the product or service being analyzed, whereas effect-oriented LCA maps the consequences of changes (Ekvall & Finnveden, 2001). This is further elaborated in the ILCD handbook (2010).

A Life Cycle Assessment is undertaken through a characterization of several impact categories. A product's life cycle is referred to as all its life phases, from input raw materials, through production and use, to the end of the life cycle. It is a systematic and robust tool that identifies and quantifies potential environmental impacts from a good or service. The system approach makes it possible to identify manifold processes and elementary flows related to the product system and the effects these have, categorized in different impact categories. This approach makes the assessments a good tool for identifying cause-effect chains, hot-spot processes, and delivering sound mapping of environmental effects. Life Cycle Assessments are used to analyze a broad range of activity, from environmental declarations and eco design, to food production and transportation alternatives (Goedkoop et al., 2009). Assisting environmentally sustainable decisions is central to LCA.

The International Organization for Standardization (ISO), under the ISO 14040:2006, has standardized the methodological principles and framework for conducting LCA (ISO, 2006a). The ISO 14044:2006 has standardized the requirements and provide guidelines for the assessment process (ISO, 2006b).

[1] Kaja Henny Engebrigtsen presented a similar approach in her master thesis: 'Arealbrukseffekter i livsløpsvurdering.' This section is inspired by Engebrigtsen (2012).

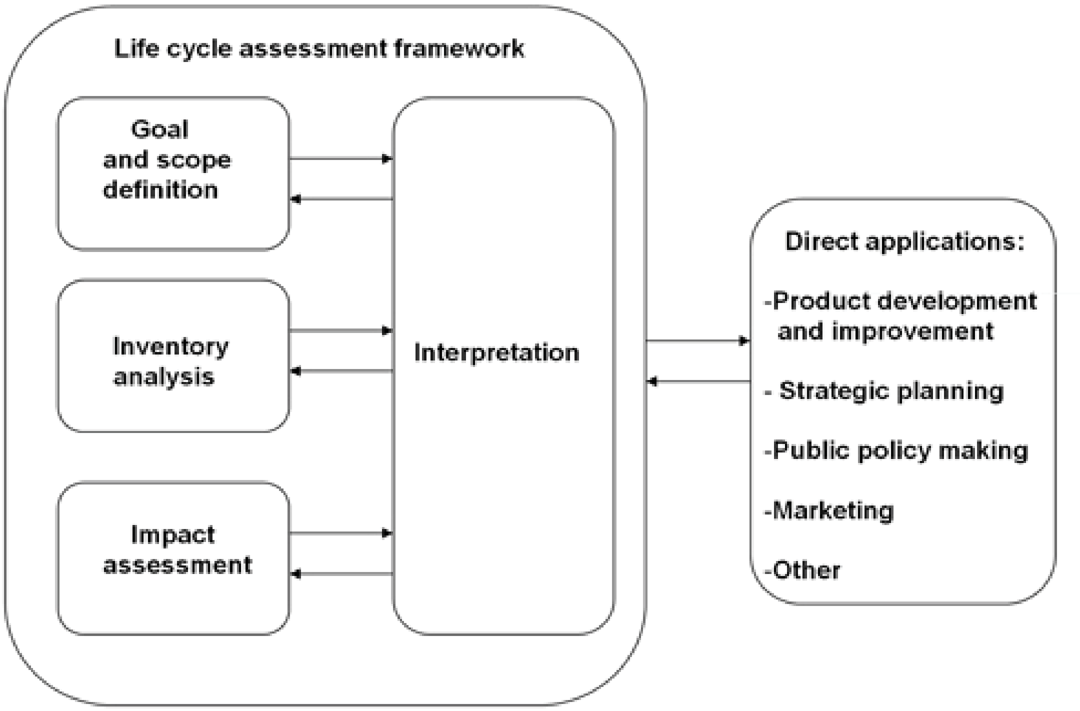

Figure 1: Schematic Diagram of the Life Cycle Assessment phases (ISO, 2006a).

The methodology focuses on quantifying potential environmental impacts of a defined product or service and is structured around a defined functional unit (FU). The functional unit is a quantitative measure of the function delivered by the system, and it provides a reference to which the input and output flows can be related. This enables comparison of two essentially different systems, with the same functional unit. If the purpose of an LCA is to compare two product systems it is imperative that the system boundaries are identical for the comparison to be made on an equal basis. After defining the functional unit, a quantification of the amount of input product needed to fulfil this function is required. The result of this quantification is the reference flow (Baumann & Tillman, 2004). Life Cycle Assessments are carried out through four phases (see figure 1):

1. Goal and scope definition;
2. Inventory analysis;
3. Life cycle impact assessment;
4. Interpretation and presentation of results.

3.1.1 Goal and scope definition

The goal and scope definition is the first step in conducting a LCA. The ISO 14040:2006 states that the goal definition "shall unambiguously state the intended application, the reasons for carrying out the study and the intended audience." Through defining the goal and scope, the modeling requirements for the assessment are specified, and these are defining for the remaining steps and outcome of the assessment (Baumann & Tillman, 2004). As far as possible all choices, specifications, and assumptions should be clearly stated in this phase. This will leave few to none value choices for the following phases, and ensures the transparency and verifiability of the assessment (Baumann & Tillman 2004, op. cit.).

The product system to be analyzed has to be described, including which options to model. Under this section LCA practitioners have to define which specific products, product designs or process options to be investigated. Hereafter it can be beneficial to make a general flowchart of the system. The functional unit (FU) of the study is decided on in this step and has to be defined in accordance with the function of the system. The FU works as a reference for the flows related to the system, and is key to the analysis. Furthermore, based on which environmental impacts the assessment is focusing on, relevant impact categories have to be decided on, and impact assessment method has to chosen. The ISO 14040 provides headlines for impact categories: 'resource use', 'ecological consequences', and 'human health' (ISO, 2006a). It is not specified what should be accounted for under the impact headlines. For each specific assessment practitioners therefore have to choose which impact categories to focus on, and interpret the ISO-headlines in operational categories, e.g., acidification, toxicity, and resource depletion (Baumann & Tillman, 2004). Definition of impact categories is done in the life cycle assessment phase.

System boundaries have to be set in the goal and scope definition, and are defined for several dimensions including the time horizon considered, delineation geographically, division between the technological system and nature, and boundaries within the technical system (Tillman et al., 1994; Baumann & Tillman, 2004). When setting boundaries within the system aspects to consider are production capital, personnel, cut-off criteria and allocation procedures. Division between background and foreground systems is made, if applicable. All choices and assumptions made during this phase are done to ensure transparency throughout the assessment.

3.1.2 Inventory analysis (LCI)

The inventory analysis is the second step of a LCA. In this phase data is collected and analyzed, and ultimately a value is given for the damage load per functional unit. The first step in this phase is the construction of a flow model to illustrate the technical system under study. The flow model is an illustration of the mass and energy flows in the system, but is incomplete as only environmentally relevant flows are considered. Life cycle inventory models are static, linear models, where time is not used as a variable. As relationships are expressed linearly within LCI models, relationships are often simplified (Baumann & Tillman, 2004).

The flow model is constructed in accordance with the system boundaries defined in the goal and scope definition. Data has to be collected for all the processes in the product system, and accounts are made for relevant elementary flows in and out of the system. The data for each process within the system boundaries are classified under the following:

- Energy need, raw material need, construction equipment and other elementary flows;
- Product, residual product and waste;
- Emissions to air, water and soil;
- Other environmental aspects;

The dataset supporting Life Cycle Assessment is big and complex. The collection process is often a time consuming activity as a thorough LCA includes many small and big processes far from the products system's core activities, depending on the system boundaries. When data is collected for all relevant system flows, this data is related to the functional unit defined in the goal and scope definition. Practitioners may encounter situations in which impact has to be allocated between different flows. Allocation within LCA is a research topic in itself, and will not be explored in greater detail. The final result of an LCI is an inventory list, where all the inputs and outputs of the modeled system are presented.

3.1.3 Life cycle impact assessment (LCIA)

The third step of a LCA is the life cycle impact assessment. The purpose of impact assessment step is three-fold. Primarily, the LCIA increases the environmental relevance, comprehensiveness and communicativeness of the results. Secondly, the LCIA improves the readability of the results by grouping them and reducing result parameters. Finally, LCIA aspires to make the LCI results readily available for easy communication (Baumann & Tillman, 2004).

LCIA interprets the environmental loads quantified in the LCI, and describe the potential related environmental consequences. The environmental loads are interpreted in operational impact categories such as acidification, ozone depletion, and eutrophication, under the ISO-headlines ‘resource use’, ‘ecological consequences’ and ‘human health’. After potential impacts are interpreted, relevant cause-effect chains are identified (Baumann & Tillman, 2004). The ISO 14044:2006 defines LCIA as the “phase in the life cycle assessment where one evaluates the scope of the potential environmental impacts of a product system.” Impact assessment is a four-fold process: classification, characterization, normalization and weighting, of which the first are obligatory, and the two last are optional:

1. Classification: All substances (LCI result parameters) have to be sorted into classes according to their environmental impact, hence operational impact category.
2. Characterization: Quantitative step in which the relative contribution of the classified environmental flows are calculated. Sizes of environmental impacts are calculated per category using equivalency factors, defined while modeling relevant cause-effect chains. The environmental impact per category is calculated and aggregated, and reflected through one value for each operational impact category.
3. Normalization: Relation of the characterization results to a reference value, for example relating the impacts of the studied product to the impacts of the total amount of pollutants emitted in a region. The quantified impact is compared to a certain reference value, for example the average environmental impact of a European citizen in one year.
4. Weighting: Giving each impact category a relative weight and thereafter ranging them. Different values are related to impact categories to generate a single score. Aggregation of characterization results across impact categories.

The methodological framework for LCA divides between impacts on mid-point level and end-point level. Impact categories on mid-point level use a problem-oriented approach and translate into impact category indicators such as climate change, acidification, human toxicity and so on. End-point indicators are damage-oriented and translate environmental impacts into issues of human concern such as human health, ecosystem quality, and resource availability. Endpoint indicators have a lower level of certainty compared to midpoint indicators, but are easier to understand, hence better for communicative purposes.

3.1.4 Interpretation and presentation of results

Interpretation and presentation of results is the fourth and last phase of LCA. ISO 14040:2006 defines this phase as the "… phase of life cycle assessment in which the findings of either the inventory analysis or the impact assessment or both, are combined consistent with the defined goal and scope in order to reach conclusions and recommendations." Interpretation and presentation of results is a systematic technique to identify, quantify and evaluate the results from the inventory and impact assessment. The interpretation should result in a conclusion and recommendation for further investigations. According to the ISO 14040:2006 the interpretation and presentation of results should contain the following:

- Uncertainty analysis;
- Sensitivity analysis;
- Contribution analysis:

The fourth LCA phase should also present a summary of relevant findings from the LCI and LCIA, and provide an evaluation of the validity and reliability of the analysis. Finally, a conclusion, description of limitations, and recommendations should be given.

3.1.5 Critical Assessment

After a complete LCA is presented, critical assessment of the study can be conducted, voluntarily. This is done in order to examine the study's conformity with ISO-requirements and to increase the reliability of the study. Critical Assessment is preformed externally, by peer review. It is generally an optional activity, but is required for LCA studies intended to support comparative assertions made public (ISO, 2006b). Environmental Product Declaration (EDP) is always subject to critical assessment.

3.2 Methodological limitations

The LCA methodology is, despite being a robust tool for environmental assessment, not complete, and contains some methodological limitations. A thorough review of methodological limitations is outside the scope of this study. Site-specific impacts are generally not accountable in LCA (Milà i Canals et al., 2009). Such impacts are claimed to be better dealt with through Environmental Impact Assessment (EIA), and/or Risk Assessment (RI). However, site-specific impacts require increased attention as e.g., conservation of biodiversity and sustainable management of freshwater resources are gaining ground on the international agenda. Both impacts on biodiversity as a result of land use changes, and impacts of freshwater resource use, are not sufficiently covered by current LCA methods. LCA is a tool utilized to describe potential impacts of production systems or human development in natural environments. LCA can never be complete in terms of describing qualitative aspects. But enhanced completeness can be achieved by expanding the current LCA framework to enable assessment of site-specific impacts together with EIA and RI.

4 Methodology

After effort in method development to properly address water use in LCA, methods for accounting and assessing water use in LCA have improved and operational methods are available. The methodological development has made considerable progress after initiatives from the World Business Counsel on Social Development (WBCSD) and the UNEP/SETAC Life Cycle Initiative, both focusing on operational methods that can be used in the inventory and impact assessment phases of LCA (Berger & Finkbeiner, 2010). In addition to this the International Organization for Standardization (ISO) is in the process of establishing a new standard for water use: the ISO 14046 on water footprinting.

Connecting the LCA methodology with methods for measuring freshwater use, can give near to exact accounts of freshwater used in the production of a good or service, and provide indicators related to potential impacts of this water use. True to the life cycle perspective this provides accounts of the freshwater use of a product from cradle to grave: freshwater use in the mining of input materials, for each step of production, and for the disposal or recycling process (Berger & Finkbeiner, 2010 op. cit.).

Several methods aiming at quantifying and/or assessing freshwater use exist. Within the LCA community there is debate regarding the correctness of the methods developed. Herath et al., (2011) states: “The science of water footprinting is still in its infancy, and methodologies are still being developed and revised. There is no well-documented and accepted methodology yet to quantify the WF of hydroelectricity.” This indicates a need for a standardized framework for quantifying water footprints, generally, and more specifically for the case of hydropower. A number of what has been regarded as the “most relevant methods” have therefore been identified through a literature search, and will in the following be described. Subsequently, an assessment of the methods will be carried out based on their advantages and limitations. The relevance of the chosen methods has in great part been based on the methodological discussion by Berger and Finkbeiner (2010) and Bayart et al., (2010). The methodological state-of-the-art will be drawn upon to establish a methodology that is considered valid and sound for determining water footprints of hydroelectric generation.

4.1 Concepts

To make the discourse in this chapter clearer the concepts basic to the understanding of water footprinting will be defined according to their use in this paper. This terminology is in accordance with the terminology proposed by the second phase of the UNEP/SETAC Life Cycle Initiative (Bayart et al., 2010). In addition, a flow model is presented, to ease interpretation for the reader.

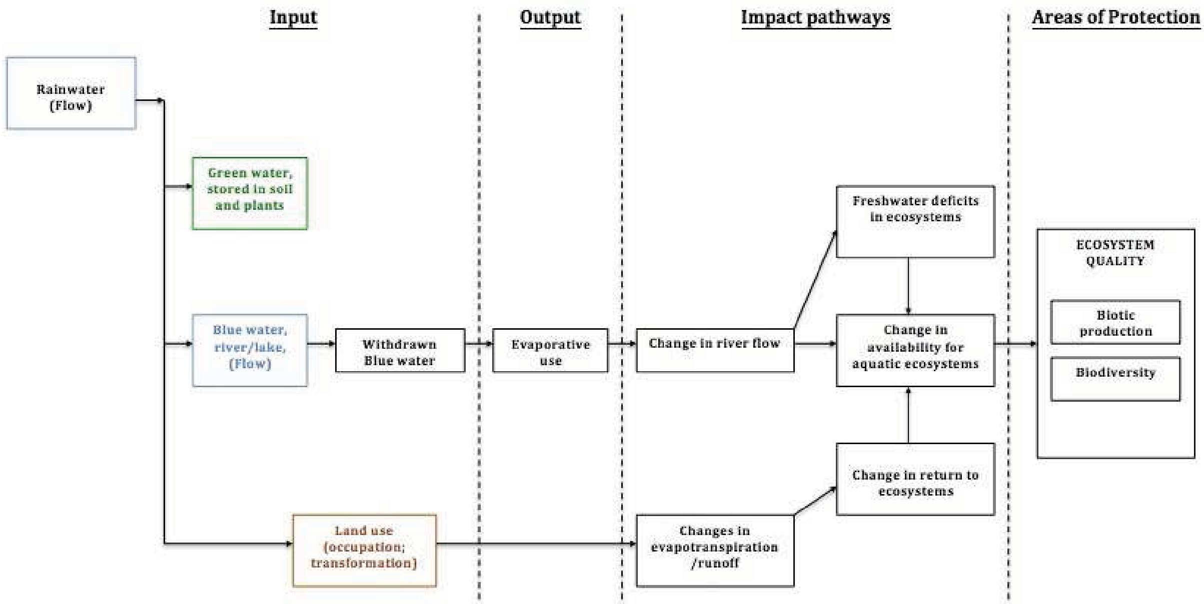

Figure 2: Inventory requirements and impact pathways from different types of water use

4.1.1 Water Footprinting

Berger and Finkbeiner (2010) points out an important distinction when defining water footprinting, namely that the definition takes on two meanings. One is the concept coined by Hoekstra in 2002, which is employed by the water footprinting assessment manual, developed and maintained by the Water Footprint Network (Hoekstra et al., 2011). The other refers to all methods aiming at describing water use in LCA. The terminology in this thesis will follow the same logic.

4.1.2 Freshwater use

Generic term that comprises all human uses of freshwater resources (Bayart et al., 2010), and is thus regarded as the total input of freshwater into a product or service system (Berger & Finkbeiner, 2010).

4.1.3 In-stream freshwater use

Term for water used in its original location. This includes shipping activities on rivers and hydroelectric generation (Bayart et al., 2010).

4.1.4 Off-stream freshwater use

Term for water used in other places than its original location. For water use to be classified as off-stream it is required that the water was removed from its natural freshwater body or groundwater aquifer. This term comprises pumping or diversion of freshwater for various municipal, agricultural or industrial purposes (Bayart et al., 2010).

4.1.5 Consumptive freshwater use

Term used for water withdrawn from a freshwater body and not returned to its original system. This occurs due to product integration, evaporation or discharge into water systems different from the original (Bayart et al., 2010; Berger & Finkbeiner, 2010). Consumptive freshwater use is essentially evaporation of water, in-stream or off-stream.

4.1.6 Degradative freshwater use

Term used for water that is withdrawn and released back into the same freshwater system with altered quality, in-stream or off-stream (Berger & Finkbeiner, 2010).

4.1.7 Water withdrawal

Term traditionally used for measuring water use. Water withdrawal is defined as the quantity of water removed from available resources for use of any purpose. The withdrawn water is not necessarily entirely consumed and some portion may be retuned for further use downstream. This is not reflected in water withdrawal figures and are therefore imprecise volumetric measures (Hoekstra et al., 2011).

4.1.8 Virtual water

In the 1960s Tony Allan developed the virtual water concept and according to Hoekstra and Hung (2002) this was the first attempt to develop product water footprints. Within the framework of this methodology the sum of the freshwater consumed along the supply chain of a product is quantified and both direct and indirect water use is accounted for. With the introduction of the virtual water concept Allan was also the first to introduce the division of water into blue, green and grey (see sections 4.1.9, 4.1.10, 4.1.11).

4.1.9 Blue water

Blue water is defined as surface and groundwater. Consumption of blue water occurs when water from the available surface and groundwater in a catchment area is reduced. Reduction will occur due to evaporation, product integration, or release of water to other water systems, or the sea (Hoekstra et al., 2011).

4.1.10 Green water

Green water is defined as the water stored in plants and soil. Consumption of green water occurs when this stored water evaporates through evapotranspiration[2] (Hoekstra et al., 2011).

4.1.11 Grey water

Grey water refers to pollution of water during the production of a good or service and is defined as the volume of freshwater required to dilute the pollutants emitted to the water system for it to reach a quality required by minimum water quality standards, given natural concentrations of pollutants in a water body (Hoekstra et al., 2011).

4.1.11 Withdrawal-to-availability ratio (WTA)

Withdrawal-to-availability ratios indicate regional water stress by measuring how much water is withdrawn from a basin in relation to the renewable availability in that same basin. The higher this ratio is, the higher is the possibility of the resource to be polluted or depleted (Alcamo et al., 1997).

4.2 State-of-the-art

According to Bayart et al (2010) and Koehler (2008) freshwater is, following standard resource classification for LCA, defined as an abiotic resource, and can be found as water deposits (fossil groundwater), water funds (groundwater aquifers and lakes) and water flows (streams and rivers). Freshwater cannot be substituted for, and is also the only abiotic resource than can be renewable and finite at the same time (Koehler, 2008).

In the literature search a series of issues connected to freshwater use have been identified: Water is a scarce resource is in need sustainable management (Humbert et al., 2010). In order to support such sustainable management development of methods to assist decision makers is imperative (Bayart et al., 2010). Furthermore, the world is experiencing tremendous population growth with estimates reaching nearly 9 billion people by 2050 (Beddington, 2009). With increased populations come increased demand for freshwater, and if consumption rates exceed regeneration rates, freshwater resources will deplete. Another issue is competition for water resources as a result of changing supplies making less water readily available for other users (Bayart et al., 2010). Moreover, water is a strategic resource, politically and economically. Global trade is a fundament of society, as we know it, requiring

[2] ET is the total evaporation from the surface of vegetation. It consists of evaporation from physical objects (surface water, earth and wet rocks) and transpiration from the surface of living plants.

huge inputs of freshwater both in production and for transportation (Hoekstra & Hung, 2002). In relation to hydropower generation it is debated weather hydropower is a mere in-stream water user or if it also consumes water (Mekonnen & Hoekstra, 2011a). The relationship between water and energy has also been underlined (Gleick, 1994).

The following section will provide a description of various methods aiming at accounting and assessing water use. Methods include stand-alone methods and methods developed and intended for use within LCA. Literature reviews carried out by Bayart et al. (2010) and Berger and Finkbeiner (2010) have been helpful in determining appropriate methods.

4.2.1 Water Inventories

According to Berger and Finkbeiner (2010) defining water footprints is most simplistically done by creating water inventories for the supply chain of a product or service. Water inventories are calculated by withdrawing the output flow of water from the freshwater inputs. The result of this determines the freshwater consumption during production due to evaporation, product integration and leakages.

By constructing water inventories for each production step and aggregating these complete water footprints can be determined, and be used as single-score resource use indicators (Berger og Finkbeiner 2010). Existing LCA databases such as ecoinvent and GaBi, in addition to tools such as the Global Water Tool, can be used to determine water inventories. Berger and Finkbeiner (2010) stress that inventories may vary depending on what database or tool that is used to establish them, with their complexity being what distinguish them.

4.2.2 Virtual water

In the 1960s Tony Allan developed the virtual water concept and according to Hoekstra and Hung (2002) this was the first attempt to develop product water footprints. Within the framework of this methodology the sum of the freshwater consumed along the supply chain of a product is quantified and both direct and indirect water use is accounted for. With the introduction of the virtual water concept Allan was also the first to introduce the division of water into blue, green and grey.

4.2.3 Water footprinting according to Hoekstra

Hoekstra and Hung (2002) introduced the water footprint concept, and Arjen Y. Hoekstra is regarded as the developer of the concept. The concept has gained interest as an indicator of freshwater use not merely focusing on direct use, but also indirect use. Because of this the water footprint is claimed to be a comprehensive indicator, measuring different types of water and water quality, in addition to including both direct and indirect use (Hoekstra et al., 2011). This last attribute is much like the virtual water concept, and is in clear accordance with a life cycle perspective. In addition, all components of a total water footprint are specified geographically and temporally (Hoekstra et al., 2011 op. cit). Another similarity the water footprint concept share with the virtual water concept is the division of water into blue, green and grey. To calculate a water footprint according to Hoekstra several methods are available, essentially all building on the same concept; accounting for the virtual water content of a product or service.

Mekonnen and Hoekstra (2011a) calculated the water footprint of electricity generated by hydropower (WF, m^3/GJ), by dividing the amount of water evaporated from the surface areas of hydropower reservoirs annually (WE, m^3/GJ), by the amount of energy generated annually (EG, GJ/year):

(1) $$WF = \frac{WE}{EG}$$

The total volume of evaporated water (WE, m^3/year) from the hydropower reservoir over the year was calculated according to eq. (2). Yearly evaporation (E, mm/day) was multiplied by the surface area of the reservoir (A, ha), and 10 to account for the evaporation that can be attributed to the surface area of the reservoir measured in m^3:

(2) $$WE = (10 \times \sum_{t=1}^{365} E) \times A$$

There are a number of methods for estimating/calculating evaporation. This is beyond the scope of this paper, and is therefore not discussed further.

Mekonnen and Hoekstra (2011a) compared the water footprint of hydropower with the water footprint of electricity generated from bio crops. To calculate the water footprint of bioelectricity, they multiplied the water footprint of the crop (m^3/ton) by the harvest index for that crop to get the water footprint in m^3/ton of total biomass harvested (Mekonnen & Hoekstra, 2011b). Harvest indices were taken from Gerbens-Leenes et al. (2008) and

Gerbens-Leenes et al. (2009). The water footprint of total biomass was then divided by the bioelectricity output per unit crop (GJ/ton) as reported by (Gerbens-Leenes et al., 2008).

4.2.4 Water footprinting according to Herath and colleagues

Herath and colleagues (2011) introduced a methodology building on the water footprint according to Hoekstra, used to calculate the water footprint of hydroelectricity generated in New Zealand. Unlike the water footprint according to Hoekstra this methodology operates with three concepts: WF-1, WF-2, and WF-3. The WF-1 measures consumptive freshwater use and is identical to the water footprint method applied by Hoekstra. It is calculated as the evaporative loss from the surface are of the reservoir, divided by the energy produced by that hydropower plant. The WF-2 measures net consumptive freshwater use, and in addition to considering the consumptive freshwater use like the WF-1, it also compares the consequences of land use changes created by the dam as dam construction necessarily replaces some of the area vegetation by a water surface. The WF-2 therefore replaces the evaporation from the surface of the reservoir with evapotranspiration and considers the net evaporative water loss from the area occupied by the reservoir:

(3) $$\text{WF-2:} \frac{(E_0 - ET_C)}{P}$$

ET_C is the amount of water lost by evapotranspiration (m^3/year) from the prior-to-dam vegetation that would have been at the site were it not for the construction of the dam. Inclusion of evapotranspiration enables accounting of the difference between natural and human induced evaporation. The research team determined the evapotranspiration using a soil-water balance that was calculated using daily rainfall, runoff and soil-water deficit data from the "National Institute of Water and Atmospheric Research" (NIWA) database. It was assumed that the vegetation prior to dam construction was constructed was pasture, but it was also found that the type of original vegetation did not affect the calculations significantly.

The WF-3 employs a net-water balance for the calculation of water footprints. The net-water balance was used in order to consider both the water inputs and outputs from the reservoir. Though evaporation is the most obvious consumptive use of water, loss through seepage in the porous geology underlying hydroelectric dams may also be considered consumptive use of water. However, it is likely that the water lost through the seepage becomes available again downstream, or it may recharge underlying ground water resources ((Gleick, 1994) in (Herath et al., 2011). Water loss through seepage was not considered. Likewise, the water used in turbines was not considered as it returns to the river after use, and was considered a through-flow. The difference between the WF-3 and the other methods (WF-1 and WF-2) is the fact that it accounts for water entering the system, not merely the evaporated water, or water loss through evapotranspiration:

(4) $$\text{WF-3}: \frac{E_0 - RF}{P}$$

RF is the annual volume of rainfall entering the reservoir, measured in m^3.

The ET_0, ET_C, and the RF volumes were calculated on an annual basis by multiplying the annual average surface evaporation, evapotranspiration and rainfall by the respective reservoir area.

4.2.5 LCI accounts and midpoint indicators for freshwater consumption

Owens (2001) presented a set of indicators for water quantity and quality, and identified the related inventory data necessary to construct the indicators. Water quantity indicators include indicators for in-stream water use, in-stream water consumption, off-stream water use, off-stream water consumption and off-stream water depletion. For water quality he proposed to use dissolved oxygen demand, eutrophication, thermal, pathogenic microorganisms, suspended solids, toxic hazards and effluent toxicity as indicators.

4.2.6 LCIA procedure with resource groups as areas of protection for South Africa

Brent (2004) suggested an assessment method, providing a characterization procedure for available LCIA midpoint categories, except land use. Site-specific characterization factors were suggested for South Africa. A distance-to-target approach was utilized for normalization of midpoint categories, with focus on set ambient quality and quantity objectives for four resource groups: air, water, land and mined abiotic resources.

Brent (2004) proposed the following procedure for the resource group ‘water’: use of ground and surface water should be aggregated without characterization in a sub-resource group called ‘water quantity’. The proportion of the water polluted during production of a good/service should be accounted for in a sub-resource group called ‘water-quality’, by normalizing the results for the impact categories eutrophication, acidification, human and eco toxicity. The normalization should be based on set objectives for site-specific ambient quantity and quality objectives. After obtaining results for both sub-resource categories these should be merged within the resource group ‘water’ by a distance-to-target weighting. In this weighting the results of the ‘water quantity’ and ‘water quality’ should be multiplied by a factor reflecting the ratio of current ambient state to target ambient state. The weighted results will provide a site specific resource impact indicator (RII), calculated based on normalization and weighting for four South African regions (Berger & Finkbeiner, 2010). The calculation of the RII follows the precautionary approach. Other approaches are used to calculate RIIs for the other resource groups and the results for all the RIIs are merged into a single-score environmental resource impact indicator (EPRII) (Berger & Finkbeiner 2010, op. cit.). Subjective weighting values are used to calculate the EPRII. The EPRII facilitates comparison of different types of resources.

4.2.7 Ecological Scarcity Method

Frischknecht et al. (2008) introduced the Swiss Ecological Scarcity Method where eco-factors are provided for a range of resources expressing their environmental impact, including freshwater (Berger & Finkbeiner 2010; Bayart et al., 2010). To use it in as an impact assessment method the elementary flows collected in the accounting phase are multiplied by corresponding eco-factors. The results of these are then expressed in eco-points that can in turn be aggregated into a singe score indicator. The calculation uses an equation containing characterization, normalization and weighting:

$$Eco - factor\left[\frac{eco-points}{unit}\right] = K * \frac{1*eco-points}{F_n} * \left(\frac{F}{F_e}\right)^2 * c \tag{5}$$

The method employs two concepts: the relationship between water scarcity and the rate of consumption, and how this relationship varies depending on site-specific water use. If this relationship between water consumption and renewable water resources exceeds 40%, pressure on freshwater in that area is high. Equally, if it is equal or lower than 20%, it is medium pressure. Medium pressure is ranged as the limit for sustainable pressure. From this, equations for weighting are obtained. Frischknecht et al. (2008) proposed eco-factors pinpointing six categories of water stress, calculated by comparing the current pressure on the freshwater resources (expressed by the water consumption-renewable water resource ratio) in a specific area to critical values defined by the OECD. The method does not only provide eco-factors for evaporation, but also water use.

4.2.8 Recommendations from the first phase of the UNEP/SETAC Life Cycle Initiative

In an unpublished document Bauer et al. made recommendations regarding method development for a framework to assess freshwater use in life cycle impact assessments, on the background of being part of the Task Force 2 of the UNEP–SETAC Life Cycle Initiative (phase 1). Bayart et al. (2010) reproduce and summarize these recommendations: (1) the assessment method should be regionalized in reference to the hydrological context; (2) resource depletion can be considered as a midpoint, while human health and reduction in biodiversity seem to be appropriate endpoints; (3) natural resource damage categories may not be considered if the cause–effect chain is modeled up to the human health and ecosystem quality categories; (4) impact pathways should be considered that highlight human health damages through the use of lower quality water for domestic purposes and reductions in food production; (5) impacts of food-compensation production and those on biodiversity through desiccation and loss of habitat should also be addressed.

4.2.9 LCI and LCIA modeling of water use according to Milà i Canals and colleagues

Milà i Canals et al. (2009) introduced a methodology focusing on impacts caused by evaporative use of freshwater. The method differentiates between different types of water and suggests two midpoint categories for LCIA. For the accounting phase it is recommended that practitioners differentiate the inventory parameters into green water (stored as soil moisture), blue water (surface and ground water), fossil blue water (non-renewable ground water) and water use as a result of land changes (Berger & Finkbeiner, 2010). Four main impact pathways were distinguished: (1) changes in freshwater causing impacts on human health; (2) changes in freshwater availability for ecosystems causing impacts on ecosystem quality; (3) abstraction of groundwater leading to (fund and stock) freshwater depletion; (4) land use changes affecting the water cycle causing impacts on ecosystem quality.

The methodology recommends ignoring effects on human health (1), as impacts of this nature, according to the authors, are not as such caused by water shortage, but by degradative freshwater use. Freshwater ecosystem impact (FEI) is recommended as a midpoint impact category comprising the two impact pathways where freshwater use causes impact on ecosystems (2), (4). In this category evaporative use of blue water (surface and groundwater) and water use due to land changes are accounted for. The authors argued that only evaporative use leads to reduced water availability for other users and it is stressed that in an LCA context only the impacts resulting from less available water can be assessed. Site-specific local effects are usually not accountable in LCA and should, according to the authors, be dealt with through Environmental Impact Assessment (EIA).

The second recommended impact category is freshwater depletion (FD), (3). Depletion of freshwater resources occurs when use exceeds renewability rates. FD assesses the reduced availability of freshwater resources for future generations. High renewability rates are assumed for surface water, therefore only consumption of water from aquifers (evaporative) and fossil water (evaporative and non-evaporative) can contribute to the impact category. Depletion may be an issue for funds and stocks of water. Using groundwater may reduce future supply and so have effects on natural resources.

4.2.10 Impact assessment of freshwater use according to Pfister and colleagues

Pfister et al., (2009) proposed a LCIA method that enables comprehensive impact assessment of freshwater consumption on midpoint and endpoint level. The authors suggest a midpoint impact category called 'water deprivation'. For the calculation of 'water deprivation' a regional water stress index (WSI) is introduced and the WSI is used as a characterization factor for the category. The WSI is calculated as a function of freshwater scarcity but in difference to other water stress indexes, it has a variation factor (VF) accounting for variation in the hydrological conditions due to seasonal differences in precipitation. The blue water consumption is multiplied by the regional specific WSI to calculate characterized results that are aggregated into the midpoint category. This enables damage assessment according to Eco-indicator 99-method in the areas of protection (AoP) human health, ecosystem quality and resources.

For the endpoint indicator 'human health' the impact pathway 'malnutrition due to lack of irrigation water' is accounted for in additional disability adjusted life years (DALY). Damage is quantified in this category by modeling the entire cause-effect chain. For the endpoint indicator 'ecosystem quality' modeling of the entire ecological cause-effect chain is required. It is assumed within this context that withdrawals of blue water reduce the availability of green water, which is crucial to vegetation in many ecosystems and so accounts for the net primary production affected by freshwater deficits are made. Damage to 'freshwater resources' is accounted for by modeling the energy used for desalinating seawater to replace depleted freshwater. When the damage resulting from freshwater consumption is classified in the different endpoint categories, normalization and weighting based on weighting factors from the Eco-indicator-99 can be used to calculate a singe-score indicator. The single-score will indicate complete damage caused by the freshwater consumption. Because this methodology gives regionalized characterization factors on watershed level, calculated by the global WaterGAP2 for more than 10 000 watersheds globally, it is a fully operational method for assessing consumptive freshwater use (Berger & Finkbeiner, 2010).

4.2.11 LCI and LCIA modeling of water use according to Bayart and colleagues

As a continuation of the LCIA modeling recommendations provided by Bauer et al., Bayart et al. (2010) proposed a framework containing recommendations for LCI modeling and descriptions of possible impact pathways for LCIA, including indicators for midpoint and endpoint assessment. In the methodology the authors pinpoint key elements affected by changes in freshwater availability and suggest three midpoint categories linked to common areas of protection (AoP).

For the accounting phase a set of water types were defined and these in turn represented the elementary flows. Making water balances for each water type enabled quantification of changes in freshwater availability. The result of this quantification is recommended as results for the LCI. Furthermore the authors identified three elements of environmental concern that should be in the cause-effect chain assessment: (1) Sufficiency of freshwater resources for contemporary human users and existing ecosystems; (2) Sufficiency of freshwater resources for existing ecosystems; (3) Sustainable freshwater resource base for future generations and the future use of present-day generations (depletion). It is suggested that the three midpoint indicators are expressed in 'cubic meters of freshwater equivalent', and that a weighting of the physical cubic meter is done by parameters that differentiate the value of the resource according to water type based on indicators such as water resource type or freshwater quality. Within this framework hydropower will be treated in the following manner: the yield indicator will be generated power measured in Mega Joules. Two scenarios were proposed: a deficiency scenario (D) and a compensation scenario (C). The D scenario is illustrated by no access to electricity and the C scenario is illustrated by changes in the power generation processes. For the D scenario the LCA impact category potentially affected at midpoint level is freshwater deficits for human users, which will, at endpoint level potentially affect human health and labor. For the C scenario potential impacts will require generation of a new LCI, assessed with full LCIA methodology accounting for all impacts at midpoint and endpoint level.

4.3 Discussion of methods

In the pervious section a number of methodological approaches for water footprint assessments were described. In terms of scope, information value, relevance and data requirements the methods vary substantially.

Initially, when attempting to identify relevant methods for determining a water footprint for the case of this study, water footprinting according to Hoekstra was the main methodological focus. The aim of the literature search was to identify of methods that could supplement that specific methodological framework, adding relevance and soundness for determining the water footprint of the case. Through the literature search, it became clear that there are several methodological approaches available and that a full assessment of freshwater use requires a combination of methods to be employed. In addition, substantial amounts of data hard to obtain have to be collected. In the following an assessment of the described methods will be attempted, based on methodological advantages and limitations. LCA is a tool developed to account for the difference between human influenced systems and natural systems. This will be methodologically explored and discussed.

Successful accounting and assessment of freshwater use within the LCA framework requires sound schemes for LCI and LCIA on both midpoint and endpoint level. Endpoint level damage is hard to predict within the LCA format, as endpoint damage resulting from freshwater use always are site-specific. The LCI/LCIA scheme to be developed will therefore focus on midpoint indicators, but still discuss endpoint indicators.

Water footprints, in their simplest way through water inventories have because of their simplicity substantial drawbacks when being used as part of a LCA. Water inventories based on commercially available databases, such as the ecoinvent database or the GaBi database, contain limited information value as the data available in these databases do not contain information for neither geographical nor quality-related aspects (Berger & Finkbeiner, 2010). Furthermore the correctness of the data provided in the databases can be questioned as it is unclear whether all relevant water flows are included or not (Berger & Finkbeiner, 2010 op. cit). According to Berger and Finkbeiner (2010) the data on freshwater use and consumption of materials have large disparities, varying with a factor of as high as ten. Moreover, because no characterization scheme for the water use and consumption data is provided, the water inventories provide simplistic volumetric measures with no information value beyond the volume of blue water consumed during production of a good or provision of a service.

As an extension of water inventories, the virtual water concept and the water footprint according to Hoekstra can be defined as more advanced water inventories (Berger & Finkbeiner, 2010). The development reflected in these concepts is their differentiation of water into the types blue, green and grey. On LCI level the water footprint of product corresponds to the output of an LCI: the quantification of the elementary flow 'freshwater' crossing the system boundary from nature into the technosphere (Bayart et al., 2010). The water footprint according to Hoekstra includes spatial information on where water is withdrawn, but only the volumes of the different types of water consumed, and where the consumption have taken place geographically, is revealed. Due to the spatial information, water consumption in water scarce areas can be identified, but as the concept is not related to water scarcity, no information about environmental damage as a result of the freshwater consumption is given. A clear advantage of the virtual water concept and water footprint according to Hoekstra is their accounting of green water, especially important in relation to LCA studies of agricultural goods and generation of bioelectricity (Berger & Finkbeiner, 2010). On the LCIA level, grey water can be used as a midpoint impact for degradative freshwater uses (Berger & Finkbeiner 2010, op. cit.). But as several authors point out, the lack of a common ambient standard for water quality implies that gray water use will vary substantially, making the concept somewhat vague. An advantage accounting for grey water holds is making aggregation of consumptive and degradative uses available already on the inventory/midpoint level (Berger & Finkbeiner 2010, op. cit.). Double accounting can become an issue when accounting for gray water use: water pollution is already covered by common impact categories, such as eutrophication, acidification, and human toxicity. Because of the limited information provided by the virtual water concept and the water footprint according to Hoekstra, they can in terms of life cycle assessment be meaningless or even misleading as a large water footprint in areas with substantial amounts of renewable water supplies may have significantly lower environmental impact than small water footprints in areas with scarce renewable water supplies (Berger & Finkbeiner, 2010 op. cit.).

The water footprint according to Herath and colleagues builds on the water footprint method described by Hoekstra, but introduces two additional concepts: the WF-2 and the WF-3. Similar to the framework for water footprints provided by Hoekstra, this methodological framework focuses on consumptive freshwater use. The difference in the water footprint calculations by Herath and colleagues is their introduction of evapotranspiration (consumption of green water) to account for land use changes as a result of dam construction (WF-2), and net blue water consumption (WF-3). The WF-3 accounts for the difference between water inputs to a reservoir (precipitation) and water outputs (evaporation) from a reservoir. This is an advantage of the method as it relates the water footprint to water availability in the area, and it is suggested by the authors that this is the most hydrological rational method of the three they present. On LCI level the water footprint according to Herath and colleagues, like the virtual water concept and the water footprint according to Hoekstra, lack characterization schemes for the inventory, resulting in volumetric measures of the consumption of the different water types considered. On LCIA level the methods do not provide an assessment scheme, and suffers the same disadvantage for life cycle assessment as the water footprint according to Hoekstra: the size of the water footprint does not reveal environmental impact.

In the framework for LCI accounts and midpoint indicators for freshwater consumption by Owens (2001) methodological advances are identified. Indicators for water quantities and qualities are proposed, and the LCI data necessary to construct these indicators are identified. The framework can work as an appropriate basis for the assessment of the water balance in the LCI phase, but for the LCIA phase the framework lacks ways to assess environmental mechanisms and related pathways caused by the freshwater use Bayart et al., 2010.

The distance-to-target method for site-specific LCIA in South Africa developed by Brent (2004) accounts for water use rather than consumption and impacts in terms of water use can be aggregated and compared to other environmental impacts (Berger & Finkbeiner, 2010). Water as a main resource group is subdivided into ‘water quantity’ and ‘water quality’. Aggregation of use of ground and surface water is proposed and normalization is suggested for the proportion of polluted water in common impact categories. Advantages of this method is the possibility of comparison between different types of resources, and that it can be transferred to other regions even if it was developed site specific to South Africa. The method does though lack methods for modeling environmental mechanisms in freshwater use (Bayart et al., 2010), and as the method is subject to subjective weighting in the distance-to-target normalization, it cannot, according to the ISO 14044:2006, be used in LCA studies that contain comparative results disclosed to the public (Berger & Finkbeiner, 2010).

Another distance-to-target method is the Ecological Scarcity Method. This method provides eco-factors for water use, and though it is site specific to Switzerland, it can be adapted to the hydrological conditions in any country (Berger og Finkbeiner 2010). On LCIA level the method provides site-specific assessment of water use or consumption, based on a water-to-availability rate and a regionalized water stress index based on OECD recommendations. The authors recommend accounting for water use instead of water consumption in the accounting phase to better reflect the water intensity of a product system, as consumption can be can be the same for different amounts of withdrawn water. A disadvantage of the method, like the distance-to-target method for South Africa, is that it cannot be applied in LCA studies with results meant for comparison disclosed to the public due to subjective weighting. A clear advantage is that the ecological threat of water use can be aggregated and compared with other environmental impacts resulting from raw material extraction or emissions (Berger og Finkbeiner 2010 op. cit.).

The method for LCI and LCIA modeling by Milà i Canals et al., (2009), proposes a detailed scheme for accounting for water use on LCI level, and two impact categories on LCIA level; ecosystem quality and resource use. In the inventory phase it is recommended to differentiate water according to type. For the life cycle assessment, ecosystem impacts are related to surface and aquifer blue water. A water stress index is suggested as a characterization factor, but this is only available for main river basins, restricting the global applicability of the method. The method also lacks characterization factors describing relevant impacts of freshwater deprivation on human health (Berger & Finkbeiner, 2010). A clear advantage of the method is that is accounts for water losses due to changes in evapotranspiration and runoff as a consequence of land use change.

The LCIA method by Pfister et al. (2009) is a wide-ranging LCIA method on both midpoint and endpoint level. The method only accounts for off-stream blue water consumption, which is a drawback and limits its applicability. Also this method employs a water stress index serving as a characterization factor based on a withdrawal-to-availability ratio of the area under study. As for other methods using the withdrawal-to-availability ratio, it can be a misleading concept, as it does not express the vulnerability of a region of additional water withdrawals. An example used by Frischknecht et al. (2008) illustrates this. If water use is low in a region with low renewable water supplies, the characterization factors can be relatively low as well compared to countries with high water use and a high renewable supply. As a withdrawal-to-availability ratio only considers renewable water supplies, non-renewable water resources are not taken into account. These water resources can temporarily be used if renewable supplies decline. As described in the previous section, the method operates with three endpoint categories; damage to human health, ecosystem quality and resources. The damage pathways are thoroughly explored, but for human health only damage resulting from malnutrition is taken into account. The damage to resources is accounted for by calculating the energy needed to desalinate seawater to replace depleted freshwater. This becomes a rather vague indicator, and is hard to compare with other indicators of freshwater depletion (Berger & Finkbeiner, 2010). An advantage of the method is that it enables aggregation of the three damage categories to one singe-score eco-indicator (Berger & Finkbeiner, 2010 op. cit.). The authors have developed a layer that can be added to the open-access software Google Earth, facilitating easy site-specific characterization factors for the midpoint and endpoint categories for manifold water systems around the world. The map layer is a great advantage for the applicability of the method. A drawback is that the method contains subjective weighting and therefore cannot be used in published LCA studies with comparative results.

The second phase of the UNEP/SETAC Life Cycle Initiative published a paper summarizing the initiative's framework recommendations for assessing for off-stream freshwater consumptive uses of blue water was (Bayart et al., 2010). On LCI level this framework provides spatial information, and suggests that the inventory should denote the quality of water inputs and outputs as well as the type of water system water is withdrawn from and released into. The spatial information enables accounting of local scarcity conditions. The LCIA consists of three elements of environmental concern and based on links between the inventory results and the elements of environmental concern, LCI flows are established and modeled in cause-effect chains comprising midpoint indicators and endpoint indicators. For all the midpoint indicators the characterization factors should account for regional aspects. The framework is a comprehensive framework, but requires large amounts of data and may suffer from high uncertainties along the cause-effect chain, as impacts become subject to uncertainties the further along this chain they are from the start. The paper recommendations, alongside the recommendations from the first phase of the UNEP/SETAC initiative, indicate what the LCA community attributes importance in the method requirements and in the continuation of water footprint method developments.

The reviewed methods are subject to advantages and drawbacks, and vary in methodological scope, relevance, detail level, and information value. A trade-off between scientific correctness and detail level of the methods and applicability is identified as methods develop. Methods requiring such amounts of detailed data that these cannot be met by scientific or commercial databases will be difficult to employ, whereas methods too simplistic will suffer from their lack of information value for freshwater use beyond volumetric measures. The need for a standardized method is identified, and hopefully the efforts of the UNEP/SETAC Life Cycle Initiative and the forthcoming ISO 14046 on water footprinting, in addition to general method development, will assist this need.

4.3 Method to be applied the case

The previous sections described and assessed two methods with frameworks directly liked to hydroelectric generation: the water footprint according to Hoekstra, and the water footprint according to Herath and colleagues. Mekonnen and Hoekstra focused on accounting for consumptive blue water use as a result of hydroelectric generation. Herath et al. (2011) also accounted for consumptive blue water use, but in addition explored two new ways to determine the water footprint of hydropower: inclusion of evapotranspiration and a net-water balance.

Landscape characteristics prior to construction of reservoirs serving hydropower plants can be subject for debate when determining a sound methodology for calculating the water footprint of hydropower (Mekonnen & Hoekstra, 2011a). This is not included in the study by Hoekstra and Mekonnen as they argue that the purpose of the water footprint is quantifying the volume of water consumption that can be linked to a specific human *purpose*, and the purpose of their study was generation of electricity. As water is released back to the atmosphere from all surfaces, be it evaporation from water surfaces, transpiration from pants, or evapotranspiration from vegetation surfaces, the methods developed by Herath and colleagues appear more sound for determining water footprints of hydroelectric generation. Herath and colleagues argue that the WF-3 is the most appropriate method according to hydrological models. This thesis is by no means based on deep hydrological knowledge, and it is therefore difficult to pass judgment on how appropriate the methods are from a hydrological perspective. Further development where hydrologists and LCA experts work transdisciplinary should address this aspect in order to determine the hydrological relevance of methods, as attributing all freshwater consumption to evaporation from the surface of hydropower reservoirs could appear to be incorrect. A net-evaporation balance, accounting for the difference between evaporation from the surface before construction of the dam, and evaporation after construction of the dam could be a more correct measure. The WF-2 and WF-3 accounting for net-evaporative freshwater loss and net-consumptive freshwater use, respectively, stand out as the most logical ways to account for freshwater consumption. Using the WF-2 and WF-3 in combination could potentially be a sound way of accounting for blue and green consumptive freshwater use as a result of hydroelectric generation. By employing both concepts, changes in evaporation as a result of land use, and inclusion of freshwater input, imperative to the hydrological cycle, can be accounted for:

$$WF = \frac{RF-(E_0-ET)}{P} \tag{6}$$

Establishing water footprints according to eq. (6) can, despite accounting for both water inputs and net-evaporation, complicate the life cycle assessment of the determined water consumption. Relating the water footprint to a water-to-availability ratio (WTA) and a water stress index (WSI) might cause double counting of precipitation as this is an input variable to those indexes. In LCA studies where a WSI approach is to be employed, water footprint calculations according to Hoekstra, or WF-2 calculations, are more appropriate, as these can be characterized by a WSI. Due to data shortages, the water footprint estimations presented in chapter 6 of this thesis are calculated according to Hoekstra and the WF-3.

The figure revealed by eq. (6) is the so-called operational water footprint of hydroelectric power generation (Mekonnen & Hoekstra, 2011a). LCA as a tool is supposed to assess the environmental impacts resulting from man-made structures. Withdrawing evapotranspiration from the antecedent vegetation (ET) (possibly evaporation from a natural reservoir) from the surface evaporation of the reservoir (E_0) is done to ensure that what is being studied is the part of the consumptive freshwater use that can be attributed to human interference with nature. Moreover, combining the two methods so that rainfall, freshwater input, is accounted for, secures accounting of a fundamental part of the hydrological cycle.

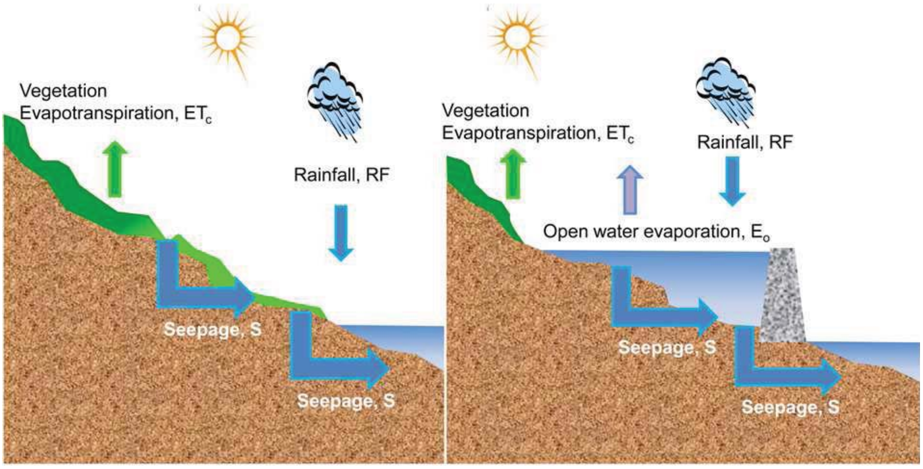

Figure 3: Schematic diagram showing different hydrological components and landscape features before (left) and after (right) a hydroelectric dam. Through-flow is ignored (Herath et al., 2011).

By calculating water footprints according to eq. (6), the water footprint is differentiated into the blue and green water types. Yet, according to the framework for water footprinting provided by Hoekstra, one water type is not accounted for – grey water. Grey water within a hydropower scheme would be degradative water use as a result of alterations in temperature, turbidity, or chemical status (Herath et al., 2011). Both the study by Hoekstra and Mekonnen and Herath and colleagues leave out the inclusion of grey water. Herath et al. (2011) argue that this is correctly done because the part of the water anticipated to be grey is very low, as pollution from the operational phase of hydroelectric generation is minimal. This logic is superimposed into this framework, but should be subject to further research and method development. The lack of ambient standards related to grey water is another reason for leaving grey water out of the calculations, as inclusion of grey water would require development of such a standard.

The operational water footprint estimated without the inclusion of precipitation does not disclose much in terms of information value, and their relevance in relation to LCA is therefore limited. Neither Hoekstra and Mekonnen, nor Herath and colleagues account for the full supply chain of hydroelectricity. The full supply chain includes accounting for the freshwater use of all the life cycle stages of a hydropower plant. The Product Category Rules (PCR) for electrical energy classifies the life cycle stages of a hydropower plant: maintenance, batteries, gates, turbines, generators, transformers, internal net, station hall, tunnels, dam, and inundation of land (Schmincke et al., 2011). To establish a water footprinting method with relevance for LCA studies of hydropower the full supply chain of processes and utilities being used for hydroelectric generation has to be accounted for, and the operational water footprint has to gain relevance through a number of steps.

For determining a sound methodology for water footprints of hydropower, a list of criteria are made in order to check that the method will include aspects imperative to LCA accounting and assessment of hydroelectric generation. These are in line with the UNEP/SETAC Life Cycle Initiative:

- ✓ The water footprint of hydropower needs to be characterized in terms of water type, spatial location area water scarcity.
- ✓ The water footprint of hydropower needs to account for damage within the impact category ecosystem quality, and if applicable to the study context, impact within the categories human health and resource use.
- ✓ The water footprint of hydropower needs sound modeling of cause-effect chains and/or damage impact pathways.

Pfister et al. (2009) proposed a regional water stress index (WSI) in order to characterize water footprints for the midpoint category 'water deprivation', serving as a requirement for regionalized LCIA of freshwater use. This enables accounting of spatial factors such as freshwater availability and use patterns at the specific location under study (Pfister et al., 2009). To obtain regionalized data, the authors used a geographic information system (GIS).

The withdrawal-to-availability ratio (WTA) commonly defines water stress and measures the total annual freshwater withdrawals to hydrological availability:

(7) $$WTA_i = \frac{\sum_j WU_{ij}}{WA_i}$$

WA_i is the annual freshwater availability, and WU_{ij} is the water withdrawal for different users for each watershed i. WTA_i is WTA in watershed i, and user groups j are industry, agriculture and households. This calculation is based on the WaterGAP2 global model (Alcamo et al., 2003), containing modeled data based on both hydrological and socio-economic conditions, describing the WTA for more than 10 000 individual watersheds (Pfister et al., 2009).

As in the Ecological Scarcity Method, OECD figures for moderate and severe stress are also employed by Pfister and colleagues to define thresholds, 20 and 40% respectively. Pfister et al. (2009) further develop the WTA-ratio by introducing a variability factor VF to account for seasonal variations. This leads to a WTA* which in turn is used for calculating the WSI. The WSI indicates the portion of the consumptive water use ($WU_{consumptive}$) that deprives other users of freshwater. The authors adjust the WSI to a logistic function to achieve continuous values between 0.01 and 1:

(8) $$WSI = \frac{1}{1+e^{-6,4\times WTA^*}(\frac{1}{0.01}-1)}$$

Equation (8) gives results in the range 0.01 to 1, where 0.01 is minimal water stress. The curve turns at 0.5 corresponding to a WTA of 0.4, which is the threshold between moderate and severe stress. Normal water stress lies in the region 0 to 0.2 according to OECD standard, and this corresponds of a WSI of 0.09. 0.6 denotes severe stress and results in a WSI of 0.91. Pfister et al., (2009) proposes the WSI be used for a general screening indicator or characterization factor for water consumption in LCIA, e.g., as a separate impact category in methods such as CML2001[3]. The authors also use the WSI for assessing damages to human health.

By characterizing the water footprint calculations according to Hoekstra or the WF-2, valuable information about freshwater consumption as a result of hydroelectric generation can be revealed, adding relevance to the operational footprint measure. By relating the characterized water footprint to relevant midpoint and endpoint categories, LCIA can be conducted. Utilizing GIS-software for the generation of WSI data adds relevance to this method as this is software available to LCA practitioners, and can be substituted for with other software containing similar data. The researches provide a Google Earth layer that can be employed to easily obtain characterization factors for numerous watersheds globally. Pfister et al. (2009) relate the WSI to the category 'damage to human health' via a damage pathway that explores malnutrition as a result of reduced access to irrigation water. The calculation of this includes the percentage of agricultural water use to total water use. This will not be applicable in the context of the case because the studied reservoir water does not

[3] LCIA method

share functionality with agricultural purposes. In other contexts where a reservoir serves multifunctional purposes such as water storage for irrigation, inclusion of this should be further explored. The WSI is also related to 'damage to ecosystem quality' and 'resource depletion'. Damage to ecosystem quality requires the modeling of the ecological cause-effect chain, assuming that blue water consumption reduces availability of green water which is essential to vegetation in many ecosystems (Pfister et al., 2009). Damage to ecosystem quality as a result of hydropower should be included in LCA studies of hydropower. Pfister and colleagues assess the effects of freshwater consumption on terrestrial ecosystem quality ($\Delta EQ\ (m^2 \times yr)$) by following the Eco-indicator 99 method, with units of potentially disappeared fraction of species (PDF), a measure for the vulnerability of vascular plant species biodiversity (VPBD). To assess vegetation damage related to water shortage a net primary production (NPP) was considered as a proxy for ecosystem quality for two reasons. First, the authors adopted results from Nemani et al. (2003) who found a significant correlation between VPBD and NPP. Second, there are spatial data globally available assessing constraints to net primary production due to water shortage ($NPP_{wat\text{-}lim}$) by means of indices ranging from 0 to 1 (Nemani et al., 2003). As shown in eq. (9) damage to ecosystem quality (ΔEQ) is determined by multiplying $NPP_{wat\text{-}lim}$ by the ratio of water consumption ($WU_{consumptive}$) to precipitation (P). Ecosystem damage is given by CF_{EQ}:

(9) $$\Delta EQ = CF_{EQ} \times WU_{consumptive} = NPP_{wat-lim} \times \frac{WU_{consumptive}}{P}$$

It is unlikely that hydroelectric generation will cause resource depletion, as storage hydropower collects rainfall and does not normally utilize groundwater, though this is theoretically possible.

The conceptual framework proposed by Bayart et al. (2010) provides a wide-ranging scheme LCA for accounting and assessment, as described in the previous sections. This framework is in accordance with the recommendations of the first and second phase of the UNEP/SETAC life cycle initiative. The provision of characterization factors is outside the scope of the paper, hence it is not an operational method, but a set of principles and three related qualitative parameters that should be accounted for calculation characterization for midpoint assessment are given. The qualitative parameters focus on spatially explicit modeling, implicitly implying regionalization of the midpoint category, hence accounting for regional freshwater scarcity. Furthermore, depending on the quality assessment method chosen within the inventory, a functionality or distance-to-target approach has to be determined. If a compensation scenario is to be included in the assessment, the impacts generated by a backup technology has to be integrated into the system boundaries and accounted for in the LCI to account for the technological changes induced by freshwater use. Although Bayart and colleagues model damage pathways linked to human health and resource use, it is assumed here that in a Norwegian climatic context, these will not have significant impact. The impact pathway linked to insufficiency for existing ecosystems could have significant impact. Bayart et al. (2010) propose that a method developed by Maendly and Humbert (2009) could be used. The method proposed by Maendly and Humbert is an empirical damage assessment model at endpoint level, assessing the impacts of water use for hydropower production on biodiversity. The method is based on empirical observations of the fraction of fish (species) that disappear after the construction of a dam on a given affected area (PDF*m^2) due to a certain amount of water used per year (m^3/yr) (Bayart et al., 2010).

For a Norwegian context this paper suggests the following: Water footprints should account for freshwater use along the complete supply chain, including the freshwater input into construction of all life cycle stages, in addition to the operational water footprint. This should as far as possible by differentiated into water type and characterized by employing the WSI, as suggested by Pfister and colleagues to account for regional water scarcity. In cases where the reservoir servers multifunctional purposes, this should be accounted for, and assessed in an appropriate manner. For midpoint and endpoint assessment cause-effect chains should be modeled reflecting relevant damage pathways. Relevant damage pathways are considered to be freshwater deficits in ecosystems affecting the resource category ‘ecosystem quality’ assessed in the operational impact categories ‘biotic production’ and ‘biodiversity’. Sound schemes for modeling cause-effect chains are provided by Milà i Canals et al. (2009) and Bayart et al. (2010). These suggestions are illustrated in figure 2. The assessment should focus on midpoint impacts. Describing endpoint impact in LCA is problematic, and such impacts should be assessed using Environmental Impact Assessment and/or Risk Assessment.

Figure 2 depicts inventory requirements and impact pathways related to different types of water use. The model is based on Milà i Canals et al. (2009) and Bayart et al. (2010). The framework provided by Canals and colleagues does not provide an impact pathway for green water use. As this is a thesis focusing on hydropower in a Norwegian context, green water use as a result of hydroelectric generation is not further explored. In other contexts, particularly those of agricultural production, green water will be the largest contributor to the water footprints and will therefore have to be included and explored in such analyses.

4.4 Application of method

Through the course of writing this thesis, I have realized the extent to which the complexity of water footprinting reaches. The method range is considerable, with each method having varying methodological scopes and focuses, information value and data requirements. In conjunction to this all the current method developments have very specific focuses, ranging from specific water-type analyses, specific LCI methodology and LCIA methodology. A need for a generic methodology is identified by this master thesis, in accordance with much of the current literature on the subject. As methods are neither standardized, nor subject to accordance within the research community, connecting exciting water footprint methods with the current framework for LCA provides challenges. This in turn provides challenges for applying the water footprinting methods to the case of hydroelectric generation. This master thesis will, despite these limiting factors, attempt to theoretically model how the method suggestions from the previous section can be employed in LCA studies of hydropower.

4.4.1 Application of water footprinting methods to LCA studies of hydroelectric generation

LCA studies of hydroelectric generation require extensive modeling of foreground and background systems. Normally the functional unit of such studies will be a unit of generated electricity delivered to the electrical transmission grid, e.g., 1 kWh, and the reference flow will be the various input flows needed to produce this unit. Conducting a full LCA study of hydropower plants in accordance with ISO 14040/14044:2006 requires that the study be organized in four phases (see chapter 3): (1) Goal and scope and definition; (2) Inventory analysis (LCI); (3) Life cycle impact assessment (LCIA); (4) Interpretation and presentation of results.

In addition to defining the purpose of the study hereunder choices, specifications and assumptions, the first phase of a LCA of hydropower will include defining the system boundaries for several dimensions according to the model requirements of the system. Specific products, product designs or process options have to be described in order to reveal the input flows needed to create the reference flow, and to collect the associated data. Relevant impact categories have to be decided on, and depending on which environmental consequences the assessment is focusing on, operational impact categories under the headlines ‘human health’, ‘ecological consequences’ and ‘resource use’ have to be determined. In addition, a relevant impact assessment method has to be chosen. After initial choices are made, creating a flow chart of the modeled system can be beneficial for identification purposes. Such a flow chart is presented in figure 4. This figure is adapted from (Ribeiro, 2004) and illustrates the inputs on which data needs to be collected.

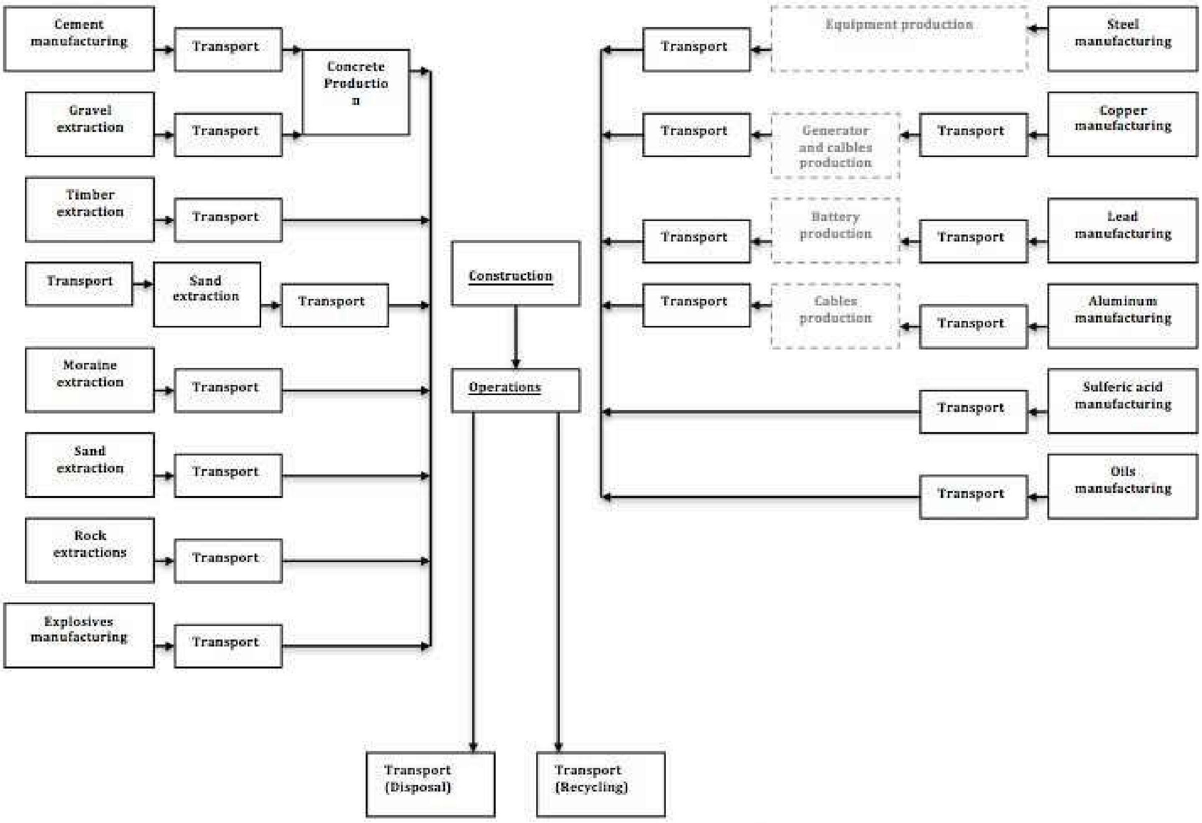

Figure 4: Schematic diagram of input and output flows.

LCA data/studies can be used for creating Environmental Product Declarations (EPD). These are often made for easy communication of environmental product impact and used as a basis for comparison of products. Product Category Rules (PRC) have been drawn up, to establish common and harmonized calculation rules, in order to ensure transparency of EPD in their comparative uses (Schmincke et al., 2011). Hydroelectric generation falls under the product group 'electrical energy', covered in the PCR 2007:08, version 2.0. This master thesis will not attempt to provide an entire review of these rules, but will use the PCR and an EDP made for hydropower produced at the Trollheim hydropower plant for a simplistic system modeling of elementary inputs to the various life cycle stages hydropower plants. The modeling is in main part done to reveal the data required in the accounting of freshwater use for all life cycle stages of hydroelectric generation.

The PRC refers to a declared unit, which translates into functional unit (FU) in LCA terms. The FU should be 1 kWh net of electricity generated and thereafter distributed. For LCA studies of hydropower this will normally translate into 1 kWh delivered to the electrical transmission grid. This is not in line with the PCR, but is done due to difficulty in assessing the power station's share of the distribution net (Askham, 2007). Environmental impact should be expressed per FU during the technical life of the energy conversion plant, based on a defined reference period. In a hydropower context the machinery (turbine, generator, etc.), power stations building, and dams and waterways have expected life times of 60, 100, and 100 years, respectively. Environmental impacts from materials used for construction of the power station are allocated to the power station production volume over 60 years for the power station and internal net, whereas the impacts of the materials used for the construction of waterways and dams are allocated over 100 years (Askham, 2007 op. cit). The system boundaries are set in order to account for the total contribution to environmental impacts for electricity delivered to the net. Hence, decommissioning of the dam and/or hydropower plant, and grid construction, is not included in the LCA. Life cycle stages include construction of the dam and tunnels, in addition to the construction, maintenance, and operation of the power station, including manufacture and transport of production equipment. Life cycle stages of the of the power station includes the station hall, cables, batteries and internal net, transformer, turbine, generator, gates and valves and pressure shaft (Askham, 2007 op. cit). Allocation of raw materials and production processes are included for virgin materials. The recycling processes are included for recycled materials. Environmental impacts should be allocated between power stations within the power plant based on average production. The EPD of Trollheim provides a schematic modeling of the main inputs to a hydropower plant. This has been reproduced here:

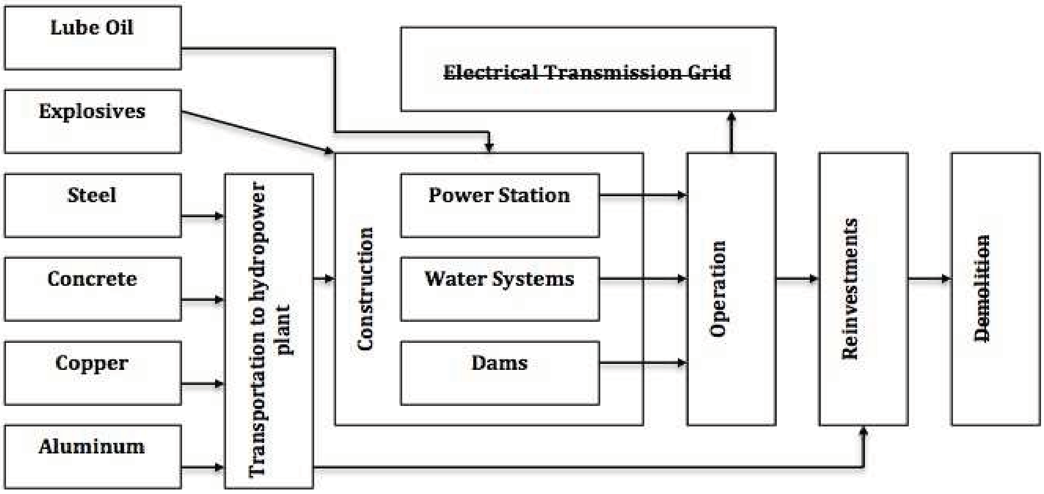

Figure 5: Schematic diagram of system boundaries and input flows for hydropower production.

In order to ensure that freshwater use from the full supply chain of hydropower is accounted for, freshwater accounts have to be made for the freshwater use of all the material input flows. These in turn have to be aggregated and added to the operational water footprint. Only then can the total freshwater measured over the whole life cycle of hydroelectricity be determined. Obtaining data for all relevant processes can be an exhaustive process: existing databases do not provide sufficient data, and the various input components are manufactured at spatially different locations, requiring characterization of the water input according to site-specific factors where the water is withdrawn. If LCA practitioners had information on the exact watershed where water was withdrawn, a tool such as the Google Earth layer provided by Pfister et al. (2009), could be used to obtain characterization factors for the that watershed, and this process could in turn be done for all freshwater inputs. It is unlikely that such information could easily be attained, and the process of the required calculations would not be economic in terms of time consumption. To overcome such challenges development of country specific characterization factors could be recommended. In practice this could mean that freshwater input to steel production produced in China be characterized by a weighted factor according to national water scarcity averages, accounting for seasonal variations. Water inputs to steel produced in the US characterized in in the same manner and so on. This will simplify the data collecting, and calculation procedures required for LCA studies; while at the same time provide data with reasonably good indicator value. There will in the first phase of including freshwater use in LCA studies be an almost constant trade-off between scientific correctness of methods and data, and applicability. The provision of 'reasonable' data could in the first round of applying methods therefore be regarded as more important that exact site-specific data. In the PCR for 'electrical energy' there is a section on data quality rules and rules for generic data. It is here stated, "As a general rule, specific data shall always be used if available. Generic data may be used if specific data is lacking and in cases where generic data are representative for the purpose of the EDP, e.g., for bulk and raw material from a spot market." Even if freshwater resources are not traded on a spot marked, it could seem reasonable that while methods are developing a simplified approach is used to ensure that use of freshwater resources *is* accounted for. This is not an expert view, and the argumentation could (very well) be too simple. As with many of the challenges related to water footprinting methodologies, and their subsequent data requirements, it is suggested that this topic be subject to further research.

4.4.1 Data requirements and data sources for water footprinting methods included in LCA studies of hydroelectric generation

This section will attempt to describe data requirements and sources of data when accounting for freshwater use in LCA. As stated previously, the data collecting process for LCA can be exhaustive and time-consuming, and the inclusion of freshwater accounts may initially intensify this process.

Through the previous discussions the need for site-specific assessment of freshwater has been stressed. In order to construct freshwater accounts for the full supply chain of hydroelectric generation, data needs to be spatially explicit, and reflect water quality. The EPD for Trollheim used literature data for the inputs flows: lube oil, explosives, steel, concrete, copper, aluminum, and reinvestments. For the other input flows data obtained from Statkraft was used. Material input flows used for the construction of the hydropower plant are categorized into virgin renewable, recycled non-renewable, and virgin non-renewable material resources. Biomass and water, steel/iron, aluminum and copper, coal, oil, fossil gas, nickel, iron, aluminum, copper and other metals, calcium/limestone, and minerals, sand and rock fall under these categories, respectively. Energy input flows include fossil fuels, hereunder oil, coal and natural gas, and nuclear, and renewable energy, hereunder biomass and hydropower, and unspecified energy inputs.

As argued previously, current databases such as ecoinvent and GaBi do neither specify water withdrawals geographically, nor quality of the freshwater data in their databases. Moreover the databases do not specify which water related processes that have been accounted for, and they contain large variations in the data provided (Berger & Finkbeiner, 2010; Owens, 2001). The databases furthermore only provide data for water inputs, not outputs. Obtaining data from commercial databases will therefore not be possible within the suggested framework, as it is required that the freshwater used reflect the regional water stress of where the data was withdrawn. The suggested framework does not contain quality related data requirements, but provision and inclusion of such data would improve the suggested method further. As freshwater input data of input flows to the hydropower production cannot be obtained from databases, practitioners either have to obtain information on amounts of freshwater input and spatially explicit information of the water withdrawals, and calculate this themselves, or leave this part out of the analysis. This exclusion will impair the quality of the studies.

Geographical information system (GIS) analyses can be used to obtain data on the surface areas of hydropower reservoirs. My knowledge of GIS is limited, but it is to my understanding GIS can be used for manifold purposes, and that such systems contains large amounts of data including historic records of registered biodiversity in areas, climatic data and so on. In this perspective obtaining required input data from GIS for many of the processes could be possible. A number of interactive map applications are provided in Norway, presumably providing GIS data. The Climate and Pollution Agency (KLIF) provides a map service containing registrations and analyses of watersheds and water quality (KLIF, 2012). The Water Resources and Energy Directorate (NVE) provides an online atlas where data on hydropower facilities, risk zones, hydrological data and more is provided (NVE, 2012). A collaborative effort between NVE, KLIF and the Meteorological Institute provides a map application where snow, water, weather and climatic data can be obtained (Meteorologisk Institutt, NVE, & Statens Kartverk, 2012).

Public/national databases can be used to acquire some of the necessary input data. National meteorological institutes commonly collet meteorological data and present these in databases. eKima is the Norwegian database, where amplitude amounts of meteorological data are provided (Meteorologisk Institutt, 2012). Additionally, forecasting authorities and research communities concerned with water, presumably hold freshwater data that can be utilized in water footprint studies. In Norway NVE operates the HBV-model for various aspects related to water management. The HBV-model contains data on evaporation and evapotranspiration, which can be used in water footprint studies.

The Google layer provided by Pfister and colleagues can be employed to get WSI characterization factors for more than 10 000 watersheds around the world. If LCA practitioners possess knowledge of where freshwater resources are extracted, they can characterize water footprints obtained through simplistic methods in order for the water footprint to reflect water stress in the region where the water was withdrawn.

LCA of energy commonly divides the established water footprints on generated energy. Such data can be obtained from the energy producers, or, when conduction global average studies, from literature.

When the ISO 14046 on water footprinting is finalized it can be expected that commercial databases will improve the quality of their data on freshwater sources, hereby easing the collecting process.

5 Environmental Impact Assessment of the Upgrading and Expansion Process

This chapter will present a description of the case, and important aspects from the Environmental Impact Assessment (EIA) will be accounted for. Hydrological data and impacts will be paid particular attention. The information in this section is gathered from the EIA conducted in relation to the license application, and the expert report on hydrological consequences (supporting information for the EIA). These reports were written in Norwegian. Some information may therefore be 'lost in translation', but will as accurately as possible be described. The term 'regulated area' is referred to frequently. A 'regulated area' is the area for which the license has been granted. The license sets specific requirements for the degree of allowed human intervention, including minimum and maximum water levels in the reservoirs, minimum and maximum water flow levels in the surrounding rivers, and so on. The Norwegian Water Resource and Energy Directorate (NVE) is the state authority granting hydropower licenses. For proper names these will at first mention be followed by a parenthesis with the English translation. After first mention, proper names in Norwegian will be used.

5.1 Description of the case[4]

Statkraft has planned an expansion and upgrading of the Høyanger hydropower plant in Sogn og Fjordane (Sogn and Fjordane County). The current Høyanger hydropower plant scheme consists of five hydropower plants: K2, K3, K4, K5A, and K5B (figure 6). The K2, K3 and K4 power plants are, after several decades of operation, in need of upgrading. Several alternatives for this upgrading were discussed, two of which were subject to impact assessment. The upgrading and expansion will, according to Statkraft, lead to a better use of the water resources in the area by reducing losses through flooding.

5.1.1 Current hydropower scheme in Høyanger

The current hydropower scheme in Høyanger consists of the power stations K2, K3, K4, and K5. In stretch the stations affect the municipalities of Gaular, Høyanger and Balestrand (Figure 6):

[4] Kaja Henny Engebrigtsen presented a similar approach in her master thesis: 'Arealbrukseffekter i livsløpsvurdering.' This section is inspired by Engebrigtsen (2012).

- K5 consists of two power stations: K5A and K5B. K5A utilizes the head between Bergsvatnet (the Berg lake) and the Høyanger fjord. K5B utilizes the head between Nedre Breiddalsvatnet (the lower Breidal lake) and the Høyanger fjord;
- K4 utilizes the head between Norddalsvatnet (the Norddal lake) and Høgsvatnet (the Høg lake);
- K3 utilizes the head between Høgsvatnet and Roesvatnet (the Roes lake);
- K2 utilizes the head between Roesvatnet and Ekrene, situated at the innermost point of the Dalsdalen (the Dal valley).

Bergsvatnet is the intake reservoir for K5A and has water transferred through a tunnel from Fossvatnet (the Foss lake) and the Gautingsdal (the Gauting valley)-tunnel transferring large parts of the water inflow from Eiriksdal and Gautingsdal. The transmission capacity is relatively small and at times there are occurrences of large overflows in the affected waterways. Flood losses and limited transmission capacity through water tunnels lead to losses in production because the head in K5A is higher than the K2, making the water value in Bergsvatnet higher than inn Roesvatnet. From K4 a 12kV transmission cable runs via K3 and K2 further on to K5. The switching station in the K5 connects the plants to the central electrical transmission grid, which runs through Stølsdalen (the Støl valley) and Langedal (the Lange valley). Table 1 presents key data for the current hydropower scheme in Høyanger.

Table 1: Data for the current hydropower scheme in Høyanger, hydrological period 1971-2000.

Power Plant	Unit	K2	K3	K4	K5A	K5B	Total
Catchment	km^2	73.3	74.4	24.2	208.4	20.1	228.5
Mean runoff	l/s/Km2	99.5	99.7	108.2	102.6	20.1	102.9
Mean inflow	mm^3/yr.	231.2	227.7	82.6	612.5	67.3	
Reservoir volume	mm^3	62,9	62.3	24.8	228.4	25.7	253.5
Head	m	472	63	82	573	710	
Max. Operating flow	m^3/s	6.3	6.6	2.8	19.4	2.9	
Production potential	GWh/yr.	95.6	25.7	13	710	111	1088.1
Operating time	Hours	3720	7139	6500	7516	6314	
Nominal capacity	MW	25.7	3.6	2	93.5	17.2	142

In large parts of the regulated area for the current Høyanger scheme large flood losses are registered. This is mainly attributed to congestions in the transfer system between the catchment areas and narrow intake channels to the power stations. In addition an increase in runoff has been recorded in the later years. After several years of operation, the K2, K3 and K4 plants are in need of upgrading. This fact, in addition to the flood losses, increases in runoff and aims to better utilize the existing energy potential in the regulated area brought on Statkraft's upgrading and expansion plans.

According to the Norwegian Water Resources and Energy Directorate (NVE), the process of upgrading hydropower plants includes reducing head losses, e.g., by expanding the cross-section in the waterways, and modernization and automation of power stations to increase total efficiency, reduce operation costs and improve the operational reliability of power plants. The process of expanding hydropower plants includes transferring water from unutilized watersheds, increasing existing reservoirs or establishing new reservoirs, better utilization of the head, and increasing the installed machinery and intake capacity to get more available effect during peak loads and to reduce flood losses.

Research by Statkraft shows that utilizing the water that drains to the Eiriksdal in a new power station, in stead of transferring water to Bergsvatnet via the Gautingsdal-tunnel, will reduce flood loss in the whole regulated area.

5.1.1 Alternative upgrading projects

Six alternatives were discussed when initiating the upgrading and expansion plans. Of these, three were unrealizable due to technical and economic conditions, one did not require Environmental Impact Assessment (EIA), and two were subject to EIA. The two options subject to EIA were the Eiriksdal project, and the Lånefjord (the Låne fjord) project. The three alternatives with realization potential were:

1. Rehabilitation of K2 and K3 stations. No EIA was required for this alternative.
2. Demolition of the K2 and K3 stations and building a new power plant; Eiriksdal, with intake from the Høgsvatnet and discharge in Daleelva, with and without water transfer from Isvotni. Construction of a new grid connection for the power plant.
3. Rehabilitation of the K2 and K3 stations. Building a new power plant; Lånefjord, with intake from the Høgsvatnet and discharge in Lånefjord, with and without water transfer from Isvotni. Building a new grid connection for the power plant.

The license was granted for alternative 2; Eiriksdal power plant.

5.1.2 The Eiriksdal power plant

The Eiriksdal power plant project received construction license in 2009. This came after extensive discussion within Statkraft, and by stakeholder consultations. The Eiriksdal project implies the demolition of the K2 and K3 power plants, which will be replaced by a new power plant named Eiriksdal. The Eiriksdal plant will be built into Tungefjell (the Tunge mountain), at the south side of the Eiriksdal. The aim of this is restoring natural landscapes to as close to pre-development state as possible. In conjunction to this, the air transmission grid will be replaced by underground cabling restoring close to 70% of the natural flow in Daleelva (the Dale river). Also the Makkoren power plant (replacing the current K4 plant) will be built into the mountain, with presumed beneficial environmental impacts. The power generation in the Høyanger hydropower scheme will increase by approx. 110 GWh, making annual average generation about 1055 GWh/year. Increased power production and restored natural environments makes Eiriksdal a win-win project. Figure 6 illustrates the current power plant scheme (in yellow) and the forthcoming scheme (in blue):

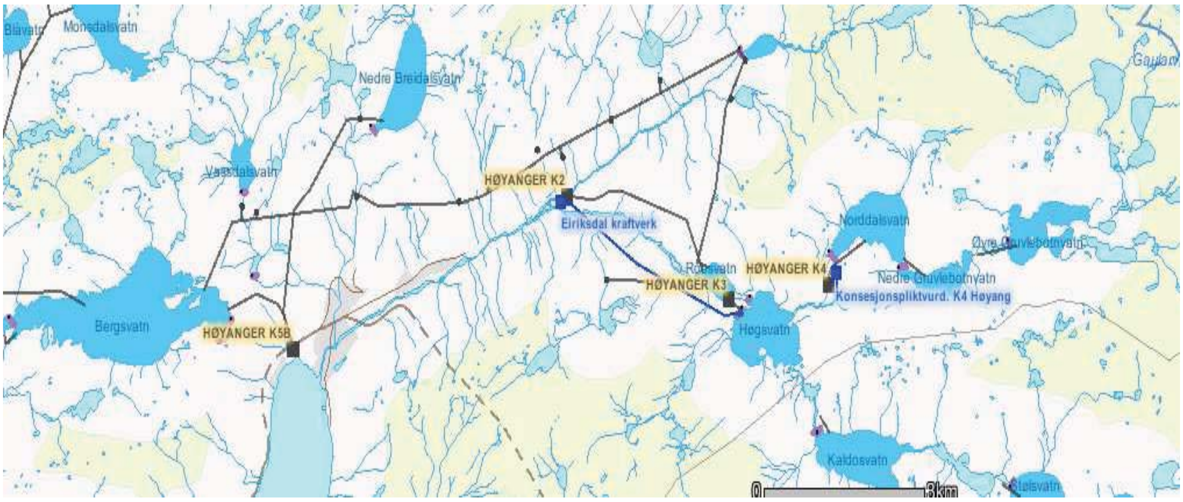

Figure 6: Map section of the current, and forthcoming hydropower plants in Høyanger.

Høgsvatnet will be utilized as the intake reservoir for the Eiriksdal plant, with discharge in Daleelva, just downstream of todays K2 plant. Høgsvatnet today has regulated limit allowing water fluctuating levels of seven meters (between 694 meters above sea level 687 meters above sea level).

Maximum water flow through the power plant will be in the range 12-16 m^3/s. One or two aggregates (generators) will be built, with a corresponding turbine performance in the range

60-80 MW. A separate draining plant will be built to secure the flow requirements for the minimum water flow in Daleelva in case the power plant has production stops.

The waterway for the Eiriksdal power plant will consist of an intake in the Høgsvatnet. Through a tunnel measuring approx. 2230 m water will be lead to the power station, and from the power station through an approx. 1800 m longs discharge tunnel. The discharge will finish in an approx. 80 m long canal leading to the Daleelva. The power plant will be connected to the grid through a 123 kV transmission line that will be connected with the central electrical transmission line towards Fardal (the Far valley). This will imply that one of the 12 kV lines that runs from the K2 stations to Høyanger will be decommissioned and removed. The remaining line will be placed in the ground. Wastes from construction of the underground tunnels will be put into landfills and these will be located on the area of the power plant adjacent to Ekrene and at the old dump down of Roesvatnet.

Table 2: Data and production estimates for the Høyanger hydropower scheme, with the construction of Eiriksdal power plant, hydrological period 1971-2000.

Power Plant	Unit	Eiriksdal	K4	K5A	K5B	Total
Catchment	km^2	74.3	24.2	136	20.1	230.4
Mean inflow	mm^3	233.2	82.6	447.1	67.3	747.7
Reservoir volume	mm^3	62.3	24.8	165.5	25.7	253.5
Head	m	563.4	82	573	710	
Production potential	GWh/yr.	323	17.4	629.1	113.5	1083.1
Mean operating time	Hours	4707	3953	6523	6314	
Max. Performance*	MW	64.5	4.2	95.8	17.2	181.7
Production simulation	GWh/yr.	304.8	17	619	108.6	

* In the simulations it is assumed that the turbine effect in the Eiriksdal power plant is 63,5 MW and 4,2 in the K4 station

Working locations and rig areas

In the area surrounding the power plant a rig area will be established adjacent to Ekrene and the K2 station containing all necessary functions for operation in the construction period. Access roads to the rig areas will also be built. Totally the rig areas will have an area of approx. 10-15 ha.

Tips and canalization

Establishing the Eiriksdal power plant requires blasting and removal of debris during the construction period. The debris mass will be placed in a tip area next to the K2 station, between Eiriksdalselva (the Eiriksdal river) and Daleelva. The tip will be adjusted to the surrounding terrain. The debris mass that will be extracted at Høgsvatnet is to be placed next to Roesvatnet, and as road filling on the mountain.

Cost estimates for the Eiriksdal power plant

The total construction costs for the Eiriksdal power plant is calculated to approx. 327.3 million NOK, including 6% interest rate during the construction period.

5.2 Hydrologic consequences

The new Eiriksdal power station will have twice the intake capacity compared to todays K2 station. This, in addition to a significant reduction in the frequency of overflow and the amount of water in the overflows, will dimension the water flow in Daleelva. Constructing the Eiriksdal power station will generally increase the water flow in Daleelva compared to the current situation in all years, and reduce the variation in the water flow. Minimum water flow will occur at less frequency than today, and the expert report on the hydrological consequences from the IA, even states that minimum water flow will occur seldom.

5.2.1 Water temperature

No well-suited measurements of the water temperature exist for the affected waterways. It is assumed that the water temperature in the K2/Eiriksdal power station will depend on the temperature at the intake reservoir in Høgsvatnet. The water temperature in Daleelva currently varies somewhat on a yearly basis. In some years the river is defined as summer-cold, with temperatures reaching maximums of 10-11°C. In other years maximum measurements between 15-17°C are registered. The river is also relatively cold in the spring. During winter the water temperature reaches zero in periods with cold weather, and the river is frozen in periods where the K2 station is run on the minimum flow requirements.

Construction of the Eiriksdal power station will create a larger dependency between the water temperature in the plant, and that of Daleelva. This implies that the temperature in the river downstream of the power station will be more similar to the current temperature measurements in the K2 station. In Daleelva the temperature will decrease faster during fall and keep low during a longer period in the spring, compared to the current situation. In the early winter small changes are expected in some years, but it is also expected that the temperature will be higher in several years. In late winter it is expected that the temperature will decrease somewhat compared to the current situation. During summer small changes are expected. In some years the length of the period with water temperatures exceeding 7-8 °C at the discharge will be the same as in the current situation, whereas some years may experience reduced periods of high temperatures in the range of three weeks to a month during spring.

5.2.2 Ice conditions

Construction of the Eiriksdal power station will lead to reduced intervals where the river is frozen. Higher water flows and an expected increase in water temperatures early in the winter will reduce the likelihood of the river freezing. This will in turn reduce problems associated with drifting ice.

5.2.3 Sedimentation and erosion

Currently there are large variations in the water flow in Daleelva causing large transfers of mass in the river. At water flows above 60 m^3/s, the river transfers mass. Erosion problems associated with embankments along the riverbank will occur at even larger water flows. When the river transfers mass, large parts of this mass will settle in threshold pools. Construction of the Eiriksdal power station will reduce both the frequency and size of the over flows into Daleelva. The frequency and size of floods will also be reduced, in turn reducing problems associated with mass transfers and erosion in Daleelva. The reduced frequency and level of larger floods may lead to increased sedimentation of fine materials in Daleelva. In the connecting river systems small changes are expected in terms of sedimentation and erosion. The new transfer from Isvotni will lead to erosion in the new river delta established, and in the existing stream the water will be lead to.

5.2.4 Sedimentation and erosion

The expected water flow changes may cause altered recipient capacity and alter the water quality of the impacted rivers. Construction activity may also lead to risks for pollution. The measures will not affect any known drinking water sources. The expert report have assessed the recipient capacity based on calculated water flow changes, water chemical conditions of the recipient, in addition to a general assessment of existing emission sources in the catchment area. Construction of the Eiriksdal power station will lead to increased water flows, in turn increasing the recipient capacity in the waterways. The measurement in itself will not lead to an increased strain in terms of nutrients in the waterways, and there are no measurements that are expected to increase emissions in the waterways. The ongoing remediation of the municipal sewer system will contribute to increased water quality in terms of impacts from thermo-tolerant coliform bacteria and nutrients. Increased flows of acidic water from Eiriksdal will give a maximum decrease of 0.1 pH units in a wet year. In a dry and normal year, the decrease is in the range 0.04-0.07 pH units. Pollution from construction activities can be prevented through thorough planning, in addition to close monitoring of the operation of the power plant. It is presumed that necessary wastewater treatment plants will be established for the gathering and treatment of polluted runoff.

6 Results

6.1 Data collection

It became clear, early on in the process of writing this thesis, that collecting and analyzing data for the whole supply chain of the hydroelectric generation in the Høyanger area would be an immensely time consuming activity, if at all possible. The aim of this master thesis with regard to the case study in Eiriksdal is thus to provide a conceptual LCA framework including accounting and assessment of freshwater use, in the LCI and LCIA phases of LCA. The output of an LCI will contain operational water footprint values, in addition to water footprint values for all inputs to construction and maintenance of a hydropower plant. The aggregated sum of these will be the total water footprint of hydroelectric generation. I have therefore, for demonstrational purposes, used data from available sources on evaporation, and precipitation in the Høyanger area, in addition to production data, and sizes of the surface areas of the intake reservoirs at the different power plants. This data construct the input figures needed to estimate the operational water footprint according to Hoekstra, and water footprint methods provided by Herath and colleagues. This chapter will provide a brief description of the data collecting and estimation processes, and present the results. All the data was measured at the nearest meteorological station, Høyanger verk.

6.1.1 Evaporation data

Obtaining evaporation data proved to be difficult. It was hard to obtain data series from official databases, and the input variables to estimate equations for evaporation are many, resulting in time consuming calculations. As stated in chapter four, estimating evaporation is outside the scope of this paper. I therefore used a simple method to estimate evaporation, resulting in low quality data. Collaboration between the Norwegian Water Resources and Energy Directorate (NVE), the Norwegian Mapping Authority, and the Norwegian Meteorological Institute provides an interactive map application online. Various map layers can be applied to a default map of Norway, expressing different meteorological and hydrological data. One of these map layers express evapotranspiration. The map layer displays a color-coding scheme for interval values. The color-coding was interpreted for the 1st day of each month at the meteorological station Høyanger verk, and then multiplied this figure by the number of days for the corresponding month. This procedure was done for the 10-year period 2000-2010 summed, and divided by number of years to find the yearly average evaporation. Yearly average area evapotranspiration is used as a proxy measure for evaporation, as data for the reservoir evaporation was difficult to obtain. Yearly average evapotranspiration at the meteorological station Høyanger verk was 714.05 mm.

6.1.2 Precipitation data

The Norwegian Meteorological Institute provides an online database containing meteorological data. From this database monthly values for precipitation for the 10-year period (2000-2010) was obtained, and the yearly average measured at Høyanger verk meteorological station was calculated. Yearly average precipitation was 2127 mm.

6.1.3 Surface area data

Data on the surface areas of the reservoir areas was difficult to find. Statkraft did not have exact information. The use of Geographical Information Systems (GIS) is suggested by Herath et al. (2011), and if one has access to such software, it would be the most appropriate and easy way to determine the surface areas of hydropower intake reservoirs. Information on the surface areas of the intake reservoirs was gathered from a web site supplying information for recreational fishing in Høyanger (Ravnestad, 2012). This site contained a register of surface areas of all the lakes in the area. To quality check the figures, the figures obtained was cross-checked with imprecise figures drawn up on an interactive map service for watersheds and quality registrations, provided by the Norwegian Climate and Pollution Agency. The data for surface areas of intake reservoirs are presented in table 3.

Table 3: Surface area of intake reservoirs.

Power plant	Intake reservoir	Surface area (km^2)
K2	Roesvatnet	0.2104
K3	Høgsvatnet	0.9039
K4	Norddalsvatnet	0.8066
K5A	Bergsvatnet	3.0896
K5B	Nedre Breiddalsvatnet	0.7635
Eiriksdal	Høgsvatnet	0.9039

6.1.4 Production data

Data on mean annual production for the current hydropower scheme in the hydrological period 1971-2000, and estimated annual mean production for the new hydropower scheme in the same hydrological period was gathered from the environmental impact assessment. Figures are provided in tables 4 and 5.

Table 4: Mean production in the existing hydropower scheme, hydrological period 1971-2000.

Power plant	Mean production (GWh/yr)
K2	95.6
K3	25.7
K4	13.0
K5A	702.8
K5B	108.6
Sum	945.7

Table 5: Estimated mean production in the hydropower scheme, with construction of the Eiriksdal power station, hydrological period 1971-2000.

Power plant	Estimated production (GWh/yr)
K4	17.0
Eiriksdal	304.8
K5A	619.0
K5B	108.6
Sum	1049.4

6.2 Data analysis

6.2.1 Operational water footprint estimates

Calculating the operational water footprint according to the frameworks provided by Hoekstra and Herath and colleagues requires the input variables collected and calculated in the previous section. Based on these I calculated estimates for the water footprint according to Hoekstra, which is identical to the WF-1, and estimates of the WF-3 from the framework provided by Herath and colleagues. The calculations are, due to imprecise evaporation figures, indicative of the water footprint of hydroelectric generation in Eiriksdal, but should be used with care due to uncertainty in data. Results are shown in tables 6 and 7.

Estimates were calculated employing eq. (1 and 3), for the water footprint according to Hoekstra, and eq. (4) for the WF-3. In the method provided by Hoekstra, input figures were required to be measured (m^3/GJ) for evaporation and (GJ/year) for production, providing WF figures in (m^3/GJ). As environmental load is given per function unit in LCA studies, and this unit is kWh in studies of hydropower, the figures were converted into (m^3/kWh). Evaporation and data was obtained as (mm/year). In order to convert the (mm/year) figures into m^3, the figures are multiplied by 10 and the corresponding surface area of the reservoir measured in (ha), according to eq. (2).

To determine the WF-3, data on both precipitation and evaporation had to be converted from (mm/year) to m^3, in the same manner as for the calculation of the water footprint according to Hoekstra. These figured were also converted from (m^3/GJ) to (m^3/kWh) to fit a LCA study. The results are also provided per GJ, in order to compare them with figures from other studies.

Table 6: Operational water footprint results for the current hydropower scheme.

Hydropower plant	Weather station	Annual energy output (GWh)	Surface area (ha)	Rainfall (mm/yr)	Surface evaporation (mm/yr)	WF according to Hoekstra (m^3/kWh)	WF-3 (m^3/kWh)
K2	Høyanger verk	95.60	21.04	2127.00	714.05	0.00157	-0.00311
K3	Høyanger verk	25.70	90.39	2127.00	714.05	0.02511	-0.04970
K4	Høyanger verk	13.00	80.66	2127.00	714.05	0.04430	-0.08767
K5A	Høyanger verk	702.80	308.96	2127.00	714.05	0.00314	-0.00621
K5B	Høyanger verk	108.60	76.35	2127.00	714.05	0.00502	-0.00993
Weighted-average						0.00436	-0.00863

Table 7: Operational water footprint results for the hydropower scheme, with the construction of Eiriksdal power station.

Hydropower plant	Weather station	Annual energy output (GWh)	Surface area (ha)	Rainfall (mm/yr)	Surface evaporation (mm/yr)	WF according to Hoekstra (m^3/kWh)	WF-3 (m^3/kWh)
Eiriksdal	Høyanger verk	304.80	90.39	2127.00	714.05	0.00212	-0.00419
K4	Høyanger verk	17.00	80.66	2127.00	714.05	0.03388	-0.06704
K5A	Høyanger verk	619.00	308.96	2127.00	714.05	0.00356	-0.00705
K5B	Høyanger verk	108.60	76.35	2127.00	714.05	0.00502	-0.00993
Weighted-average						0.00379	-0.00749

Table 8: Operational water footprint results measured per GJ for the current hydropower scheme.

Hydropower plant	WF according to Hoekstra (m^3/GJ)	WF-3 (m^3/GJ)
K2	0.44	-0.86
K3	6.98	-13.80
K4	12.31	-24.35
K5A	0.87	-1.73
K5B	1.39	-2.76
Weighted-average	1,21	-2,40

Table 9: Operational water footprint results measured per GJ for the hydropower scheme, with the construction of Eiriksdal power station.

Hydropower plant	WF according to Hoekstra (m^3/GJ)	WF-3 (m^3/GJ)
Eiriksdal	0.59	-1.16
K4	9.41	-18.62
K5A	0.99	-1.96
K5B	1.39	-2.76
Weighted-average	1.05	-2.08

6.2.2 Water stress index estimations

As argued in the discussion of methods in chapter three, the water footprint calculations according to Hoekstra do not reveal significant aspects related to freshwater consumption, neither in terms of disclosing site-specific water scarcity, nor in terms of relating damage-pathways to water consumption. Transforming the water footprint estimates according to the WSI provided by Pfister and colleagues could add to the information value of the water footprint estimates. By utilizing the map layer developed by Pfister and colleagues provided for Google Earth, I was able to obtain the WSI characterization factor for the Høyanger area. The figure obtained would be the result of eq. (8). WSI figures are given in the range 0.01-0.1 where 0.01 denotes minimal water stress. As table 10 shows, the water stress value for the Høyanger area is 0.0109, indicating minimal water stress in the region. LCA impact category results are also provided by the Google Earth map layer, and automatically analyzed using Eco-indicator 99 method. The results obtained via Google Earth are presented in table 10.

Table 10: Water scarcity index and LCA impact factors for the Høyanger area

WSI	**0.0109**
LCA impact factors:	
DALY (E-06 years)	0
M2YR	0.0738
MJ	0
Human health EI99_PTS	0
Ecosystem quality EI99_PTS	0.0058
Resources EI99_PTS	0
Aggregated EI99_HA_PTS	0.0058

I proceeded to characterize the water footprints estimated according to Hoekstra, for the current hydropower scheme, and the forthcoming hydropower scheme. This characterization was done by multiplying the results from tables 7-9 by the characterization factor 0.0109. The characterized water footprint estimates relates blue water consumption as a result of hydroelectric generation in the Høyanger area to the water scarcity/stress of the area. Results of the characterized water footprint estimates are shown in tables 11 and 12.

Table 11: Characterized water footprint according to Hoekstra for the current hydropower scheme

Hydropower plant	Characterized WF according to Hoekstra (m^3/GJ)	Characterized WF according to Hoekstra (m^3/kWh)
K2	0.00476	0.00002
K3	0.07604	0.00027
K4	0.13414	0.00048
K5A	0.00950	0.00003
K5B	0.01520	0.00005
Weighted-average	0.01320	0.00005

Table 12: Characterized water footprint according to Hoekstra for the hydropower scheme, with the construction of Eiriksdal power station.

Hydropower plant	Characterized WF according to Hoekstra (m^3/GJ)	Characterized WF according to Hoekstra (m^3/kWh)
Eiriksdal	0,00641	0,00002
K4	0,10258	0,00037
K5A	0,01079	0,00004
K5B	0,01520	0,00005
Weighted-average	0.01146	0.00004

7 Discussion

7.1 Local impact of hydroelectric generation the Høyanger region

By the two methods used to estimate water footprints, evaporation was considered as a loss (consumption) of freshwater from intake reservoirs, and the ecosystem in the Høyanger hydropower scheme. Herath and colleagues point out that evaporation and evapotranspiration also are major driving forces of the hydrological cycle. Hence, evaporation and evapotranspiration has to be seen as factors with both positive and negative connotations.

The water footprint values for the different hydropower plants in Høyanger are shown in Table 6 and Table 7. Estimation of the water footprint values follows definitions used by Mekonnen and Hoekstra (2011a) and Herath et al. (2011). In the current hydropower scheme, the weighted average water footprint is 0.0044 m^3/kWh, using the method provided by Hoekstra. Individual power plant footprints were in the range 0.0016-0.04 m^3/kWh. The weighting was done based on percentage contribution to total production. For the forthcoming hydropower plants the weighted average water footprint was 0.0038 m^3/kWh (estimated), with individual power plant footprints in the range 0.0021-0.034 m^3/kWh. The K2 and K3 stations that will be demolished and replaced by the Eiriksdal station have an aggregated water footprint of 0.027 m^3/kWh, compared to 0.021 m^3/kWh of the Eiriksdal station. Both in terms of the water footprint of the replacing station, and for the whole hydropower scheme, the water footprint values decrease in the forthcoming scheme, indicating that Statkraft's upgrading and expansion plans are beneficial in terms of the freshwater resources consumed as a result of hydroelectric generation.

Much of the reduction in the water footprint can be attributed to the upgrading/reconstruction of the K4 plant, which holds the highest individual water footprint both in the current and the forthcoming hydropower scheme, with water footprints of 0.044 and 0.034 m^3/kWh, respectively. This is because of relatively low electricity generation in conjunction to a relatively large surface area. In contrast, the K2 power plant has the third largest contribution to total electrical generation, but has the smallest surface area, resulting in the lowest water footprint in the current hydropower scheme, with a water footprint of 0.002 m^3/kWh. The data assumes increased production in the forthcoming scenario for the K4 plant (Makkoren); hence its contribution to the water footprint is reduced. Both Hoekstra and Mekonnen, and Heath and colleagues identified a relationship between the surface area of hydropower reservoirs, and the production volume in terms of water footprint sizes. This is also observed in the preliminary results of this study. The aggregated water footprint of the K2 and K3 plants compared to the water footprint of the Eiriksdal station can be used to illustrate. The K2 and K3 plants' aggregated production is 121.3 GWh, and the aggregated surface area of their intake reservoirs is 111.43 ha. In comparison, the estimated production at the Eiriksdal plant is 304.8 GWh, and the surface area of its intake reservoir is 90.39 ha. The water footprint of the Eiriksdal plant is lower than the aggregated water footprint of the K2 and K3 station, because the Eiriksdal plant produces more energy in relation to a smaller intake reservoir. The K2 station accounts for a substantial part (0.025 m^3/kWh) of the aggregated water footprint of the K2 and K3 stations as it produces only 25.7 GWh in relation to the same intake reservoir as will now be used as the intake reservoir for the Eiriksdal station.

The WF-3 estimations provided the lowest water footprint values of the two methods considered for all the hydropower plants, both in the current and the forthcoming scheme in Høyanger. The results are presented in Table 6 and Table 7. Water footprint values are negative for all the power plants in the range -0.088-(-0.003) m^3/kWh, with a weighted average water footprint equal to -0.0087 m^3/kWh in the current hydropower scheme. For the forthcoming hydropower plants the values are also negative for all plants in the range -0.067-(-0.004) m^3/kWh, with a weighted average water footprint equal to -0.0075 m^3 per produced kWh. This means that precipitation actually exceeds evaporation in the Høyanger area; hence there is negative blue water consumption per produced kWh. The WF-3 estimates for the forthcoming hydropower scheme are reduced in negativity, compared to the current scheme. As total surface area has been reduced due to demolishing the K2 and K3 plants, the total surface area in the forthcoming hydropower scheme will evaporate less, but it will also collect less input water, following assumptions made previously. These factors result in less negative freshwater consumption measured in m^3 per estimated produced kWh. Water footprint estimations according to the WF-3 method reflect freshwater supplies in the area by incorporating water inputs, and thus give insightful information, reflecting volumetric blue water consumption in a more appropriate manner than the water footprint estimations according to Hoekstra. To understand the differences in the hydrological impacts of water footprints in different locations provides meaningful information because it reflects that all regions are different in terms of freshwater resource availability. Another method reflecting this fact is the WSI provided by Pfister and colleagues.

To add information value to the water footprints estimated according to Hoekstra, these were characterized according to the WSI factor obtained through applying the map layer developed for Google Earth. The WSI and LCA impact category results are shown in Table 10. The characterization factor indicated minimal water stress in the Høyanger region, with a value of 0.0109. The M2YR figure equals 0.0738, which is the eco-indicator that is multiplied by the weighted PDF to obtain the eco-points in the resource category 'ecosystem quality'. Damage to ecosystem quality scores 0.0058 eco-points. This is a low value indicating low environmental impact of freshwater use in the Høyanger region. As assumed in chapter four, damage in the impact categories 'human health' and 'resources' is negligible, with values equaling zero. The characterized water footprints according to Hoekstra are shown in Table 11 and Table 12. The weighted average values y decrease significantly, with values equaling 0.0005 m^3/kWh in the current hydropower scheme, and 0.0004 m^3/kWh in the forthcoming scheme. This is a reduction of nearly 90% compared to the uncharacterized water footprints. The difference between the characterized and uncharacterized water footprint estimates illustrates the importance of relating water footprints to water scarcity in the area. Even if the results presented here are imprecise in various manners, it is clear that when assessing potential environmental loads resulting from freshwater losses, it makes a significant difference if the assessment is based on 0.0044 m^3/kWh, compared to 0.0005 m^3/kWh.

Local environmental impacts in the Høyanger region, as a result of freshwater use by hydropower, appear to be insignificant, and will be reduced through the impending upgrading and expansion development, according to the water footprint values obtained in this study. It is still important to remember that data analysis can neither predict, nor describe environmental state, or predicted state, in exact terms. Water footprints integrated into LCA can describe potential effects, but Environmental Impact Assessment or Risk Assessment should be used for more accurate description of current and expected consequences. The EIA of the Eiriksdal project (chapter 5) does not expect severe environmental impacts as a result of the upgrading and expansion in the Høyanger hydropower scheme. In terms of the hydrological impacts, it is even expected that more water will be available downstream of the new power plant, resulting in less flooding, less run through and better biological conditions for salmon and other species.

7.1.1 Comparison of results with results of other studies

Gerbens-Leenes et al. (2009) calculated the blue water footprint of hydroelectric generation by dividing global evaporation from artificial surface water reservoirs by the hydroelectric generation for the year 1990, resulting in a global water footprint average of 22 m^3/GJ. This has been used as an accepted proxy value for the water footprint of hydroelectric generation. In 2011 Mekonnen and Hoekstra calculated the water footprint of 35 selected hydropower plants on a global basis, as described in detail in chapter four. This resulted in a global average water footprint of 68 m^3/GJ, a water footprint value that is significantly higher than the value provided by Gerbens-Leens et al., and is also high in comparison with the water footprint of other energy sources. The New Zealand case study by Herath et al. (2011) provided significantly lower water footprint values for hydroelectricity produced in New Zealand. These were in the range 32.48-0.75 m^3/GJ, with weighted-average values in the range 1.55-6.05 m^3/GJ, depending on the method employed.

This study provided data in the format of m^3/kWh values for use in LCA. These values were converted into m^3/GJ and presented in tables 8 and 9. The weighted average water footprint for the current hydropower scheme is 1.21 m^3/GJ, and for the forthcoming scheme the weighted average is 1.05 m^3/GJ. Compared to the global average values of 68 m^3/GJ and 22 m^3/GJ, these values are significantly lower. Also in comparison with Herath and colleagues, who obtained a weighted average water footprint for hydroelectric generation in New Zealand calculated according to Hoekstra of 6.05 m^3/GJ, these are low.

Initially, when starting the procedure of writing this master thesis, I assumed that the water footprint of Norwegian hydroelectric would be similar, or smaller compared to that of New Zealand. This assumption turned out to be correct. The global average values are significantly higher than those of both New Zealand, and the Høyanger region of Norway. These figures are not incorrect, as they include hydroelectric generation in areas with significantly higher evaporation rates than those of New Zealand and Norway, but as these water footprints do not relate the water footprint values to regional water availability, they can hardly reveal valuable and needed information about the impact of the freshwater consumption of hydropower. Impacts related to freshwater use will, in difference e.g. CO_2 emission, always be local, and therefore have to be calculated and assessed based on local climatic values.

7.3 Discussion of preliminary results

Data used as proxy data for evaporation are in fact data on evapotranspiration, a mix of blue and green consumptive water use. This data was used, as it was hard to obtain data on the surface evaporation of the reservoirs only. In the studies by Mekonnen and Hoekstra (2011) and Herath et al. (2011) surface evaporation from the reservoirs were used as input variables to the water footprint estimations, resulting in a water footprint reflecting blue consumptive freshwater use. The WSI provided by Pfister and colleagues was developed to characterize blue consumptive water uses. The estimations done by proxy evaporation data is therefore meant to depict blue water consumption, even if it in fact, following the definition of evapotranspiration, is a mix of green and blue consumptive water. In the method recommendation provided in chapter four, it was suggested that a merging of the WF-2 and WF-3 method (eq. (6)) could account for both water inputs, and changes in evaporation related to retaining water in reservoirs. This was not employed in the water footprint estimations in the previous chapter, due to insufficient data material. For the same reason calculations of the WF-2 were not attempted.

The WF-3 estimations suffer the same disadvantages as the estimations according to Hoekstra, in regards to the data material. These estimations hold an advantage though, in the fact that they consider both input to the hydroelectric system (rainfall) *and* output (evaporation). This appears to be a hydrological rational argument. As stated in the methodology chapter, I hold no extensive hydrological knowledge, and this is therefore a value judgment, more than a scientific one.

Water used and consumed by a hydropower plant does not solely depend on precipitation collected in the reservoirs. Water in the whole catchment area can find its way to the reservoir through seepages through the porous geology underlying hydropower reservoirs (Herath et al., 2011), and water can drain into the reservoir. Water losses can occur in the same manner, finding its way out of the reservoir. Because calculations and modeling of this requires knowledge beyond my range, this has been ignored in the analysis. Furthermore, both the current and the forthcoming hydropower scheme in Høyanger in reality has more than one reservoir, besides the intake reservoir, as the schemes consist of several water transfers as described in chapter five. This is not accounted for, and the evaporation and precipitation rates are estimated for, and attributed to, the surface area of the intake reservoir only. The water footprinting calculations provided are meant to be indicative of the freshwater consumption that can be attributed to hydroelectric generation by the Høyanger power plant, but holds no exact values. This is due to impreciseness in the data collected and modeled, as described above and should be kept in mind when interpreting the results. Another aspect that should be kept in mind is that the environmental impact assessment assumes increased production in the K4 station in the forthcoming hydropower scheme as this is to be upgraded/reconstructed due to wear and tare. This increased production contributes to the reduced water footprint estimates for the forthcoming scheme.

7.4 Methodological gaps

Assessing freshwater use, according to the methodological framework provided in chapter four, implies differentiating water use into water types. In the water footprint estimations provided in this study, only blue water has been accounted for, and this is arguably a limitation for the method. Green water has been included in several water footprint studies, also water footprint studies of hydropower, though green water is a particularly prominent factor in assessment of freshwater inputs to agricultural production. Grey water is, as argued in chapter four, a vague concept. Grey water is measured with basis in ambient standards, and can therefore fluctuate from study to study, depending on the ambient standard chosen for that particular study. Grey water consumption as a result of hydroelectric generation can occur due to water temperature alterations, and quality alterations as a result of turbidity or chemical composition. The exclusion of green and grey water was done purposely, as these water types did not appear to have significant impact for the case study. This may have been cutting the method short, and inclusion of green and grey water in water footprint studies of hydroelectric generation should be subject to further research.

LCA of freshwater use requires LCI and LCIA of the full supply chain of hydroelectric generation. Even if the methodological framework suggested in chapter four, include LCI and LCIA, the results in the previous chapter only presents operational water footprint values. This is a severe methodological gap. Water footprints for all life cycle phases of a hydropower plant needs to be added to the operational water footprints, in order to obtain total freshwater use. Input water footprints are expected to be smaller than the operational water footprint, but in order to preform LCA in a proper manner, they are still required to reflect total water freshwater use. As a research extension, building on the research presented in this thesis, input water footprints should be established, resulting in the LCI data needed for estimating the aggregated water footprint of the upgrading and expansion in Høyanger.

8 Concluding remarks

This Master thesis has given an overview of the state-of-the-art for water footprinting methods and presented a conceptual framework for life cycle assessment of hydroelectric generation that includes accounting and assessment of freshwater resources. The research presented in this thesis was done in order to demonstrate the importance of utilizing local climatic variables when estimating water footprints.

A review of the Environmental Impact Assessment (EIA) of Eiriksdal was done, with particular focus on hydrological consequences for freshwater resources. Water footprints were not considered in the EIA, but both the EIA and the water footprint values indicate that the upgrading and expansion project in Høyanger will produce beneficial environmental impacts. Negative impacts will also be present, but surpassed by less water consumption, less flooding and less run-off, resulting in more efficient energy generation and more water in the impacted waterways.

The conceptual framework for full LCA and the water footprint methods can be applied in any hydropower context, providing meaningful information, and assisting sustainable water resource management. The data requirements to produce water footprints for the full supply chain of hydropower cannot currently be met by commercial databases. LCA practitioners will have to use substantial amounts of time collecting and calculating data to be able to produce water footprints for the full supply chain of hydropower. This is a significant drawback for the framework presented. To facilitate LCA including accounting and assessment of freshwater use, excising LCA databases should improve the quality of their water data by including site-specific information on water withdrawals and water quality related aspects.

Negative water footprint values for all power plants utilizing the WF-3 method, suggest that reservoirs in wet regions collect more water than they lose through evaporation. Development of hydropower in wet regions is therefore beneficial. Water footprints of hydroelectric generation depend on local climatic conditions, surface areas of reservoirs and energy generation. Electricity generated in the Høyanger hydropower scheme has a very low water footprint, due to beneficial climatic conditions (high precipitation rates and low evaporation rates), and energy generation related to relatively small surface areas of intake reservoirs. In an international context the weighted average water footprint values, obtained for the

Høyanger scheme, are the smallest (to my knowledge), and small compared to often-cited global average values.

Water footprints are gaining both popularity and importance in the international discourse of water management strategies. Water is a globally traded commodity (virtual water), and is the responsibility of all nations to manage sustainable, even in regions with water abundance. Because the water footprint may become important for the competitiveness of goods marketed as green, and nations dependent on hydroelectric generation, it is imperative that local climatic data is used for the calculation of water footprints. Incorrect and imprecise water footprint values could be detrimental not only to the competitiveness of water intensive products, but also for further development of hydroelectricity. With changing global climates affected by fossil based energy resources, it is important that hydropower, which is connected with low CO_2 emissions, retains its position as an environmental friend.

Hydropower currently represents 20% of total global electric generation, supplying about one billion people with electricity. The International Hydropower Association (IHA) and the International Energy Agency (IEA) estimate that the economically realizable potential for further development is 8000 TWh, with almost 80% of this potential for development situated in Africa, Asia and Latin America. In Norway current hydroelectric generation accounts for 99%, with 33 TWh estimated economic potential for further development. Energy demand is expected to rise, requiring a 50% increase in energy supplies by 2030. Water footprints should be included in environmental assessment, if the remaining potential is to be developed. The results of this case study, and other studies, demonstrating a connection between local climatic variables, production efficiency and reservoir surface areas, can facilitate the environmental assessment of remaining hydropower developments, in addition to Environmental Impact Assessment and Risk Assessment for the evaluation of exact and predicted impacts.

It is a shared international responsibility that water resources are managed sustainably. The methodological findings and results of the case study of this Master thesis can assist sustainable decision-making in hydropower developments.

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