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9	Author	Organization	Norwegian Institute for Nature Research
10		Division	
11		Address	Sluppen, P.O. Box 5685, Trondheim 7485
12		e-mail	dagmar.hagen@nina.no
13		Family Name	**Lillesund**
14		Particle	
15		Given Name	**Vilde Fluge**
16		Suffix	
17	Author	Organization	Norwegian University for Science and Technology
18		Division	Department of Energy and Process Engineering, Industrial Ecology Program
19		Address	Sem Sælands vei 7, Trondheim 7491
20		e-mail	
21		Family Name	**Michelsen**
22		Particle	
23		Given Name	**Ottar**
24		Suffix	
25	Author	Organization	Norwegian University for Science and Technology, NTNU Sustainability
26		Division	
27		Address	Trondheim 7491
28		e-mail	

AUTHOR'S PROOF!

Author	Family Name	**Foldvik**
	Particle	
	Given Name	**Anders**
	Suffix	
	Organization	Norwegian Institute for Nature Research
	Division	
	Address	Sluppen, P.O. Box 5685, Trondheim 7485
	e-mail	
Author	Family Name	**Barton**
	Particle	
	Given Name	**David N.**
	Suffix	
	Organization	Norwegian Institute for Nature Research
	Division	
	Address	Sluppen, P.O. Box 5685, Trondheim 7485
	e-mail	

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Abstract

Purpose: Habitat destruction is today the most severe threat to global biodiversity. Despite decades of efforts, there is still no proper methodology on how to assess all aspects of impacts on biodiversity from land use and land use changes (LULUC) in life cycle analysis (LCA). A majority of LCA studies on land extensive activities still do not include LULUC. In this study, we test different approaches for assessing the impact of land use and land use change related to hydropower for use in LCA and introduce restoration cost as a new approach.

Methods: We assessed four hydropower plant projects in planning phase (two upgrading plants with reservoir and two new run-of-river plants) in Southern Norway with comparable geography, biodiversity, and annual energy production capacity. LULUC was calculated for each habitat type, based on mapping of present and future land use, and was further allocated to energy production for each power plant. Three different approaches to assess land use impact were included: ecosystem scarcity/vulnerability, biogenic greenhouse gas (bGHG) emissions, and the cost of restoring affected habitats. Restoration cost represents a novel approach to LCA for measuring impact of LULUC.

Results and discussion: Overall, the three approaches give similar rankings of impacts: larger impact for small and new power plants and less for larger and expanding existing plants. Reservoirs caused a larger total area affected. Permanent infrastructure has a more

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similar absolute impact for run-of-river and reservoir-based hydropower, and consequently give relatively larger impact for smaller run-of-river hydropower. All approaches reveal impacts on wetland ecosystems as most important relative to other ecosystems. The methods used for all three approaches would benefit from higher resolution data on land use, habitats, and soil types. Total restoration cost is not accurate, due to uncertainty of offset ratios, but relative restoration costs may still be used to rank restoration alternatives and compare them to the costs of biodiversity offsets.
Conclusions: The different approaches assess different aspects of land use impacts, but they all show large variation of impact between the studied hydropower plants, which shows the importance of including LULUC in LCA for hydropower projects. Improved data of total restoration cost (and cost accounting) are needed to implement this approach in future LCA.

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LCIA OF IMPACTS ON HUMAN HEALTH AND ECOSYSTEMS

Comparing land use impacts using ecosystem quality, biogenic carbon emissions, and restoration costs in a case study of hydropower plants in Norway

Vilde Fluge Lillesund [1] · **Dagmar Hagen** [2] · **Ottar Michelsen** [3] · **Anders Foldvik** [2] · **David N. Barton** [2]

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Keywords bGHG emission · Ecosystem scarcity/vulnerability · Land use change impact · Life cycle assessment (LCA) · Mitigation hierarchy · Restoration cost

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✉ Dagmar Hagen
dagmar.hagen@nina.no

Q1

[1] Department of Energy and Process Engineering, Industrial Ecology Program, Norwegian University for Science and Technology, Sem Sælands vei 7, 7491 Trondheim, Norway

[2] Norwegian Institute for Nature Research, P.O. Box 5685, Sluppen, 7485 Trondheim, Norway

[3] Norwegian University for Science and Technology, NTNU Sustainability, 7491 Trondheim, Norway

1 Introduction

Habitat destruction, climate change, pollution, invasive species, and overexploitation of wild populations are the five main threats to biodiversity, and of these, habitat destruction is the most severe (Millenium Ecosystem Assessment 2005).

Transformation of natural land into agricultural land and fragmentation of previously continuous ecosystems for development of, e.g., infrastructure and energy production, are currently the dominant causes of habitat change and loss of biodiversity, and multiple minor changes will have a cumulative effect (Thorne et al. 2009).

Hydropower development causes transformation and occupation of water systems and large areas of land for infrastructure and reservoirs. Approximately 70% of Norwegian watersheds are currently affected by hydropower production (Norwegian Environmental Agency 2013). Hydropower is a major source of electricity, making up 16.5% (3700 TWh) of global electricity supply in 2012 (IEA 2015). Norway produced 143 TWh in 2012 and is the sixth largest hydropower producer worldwide (REN21 2013). To satisfy growing energy demand, further development of hydropower is expected in Norway (www.nve.no) and other hydropower-producing countries (IEA 2015), but biodiversity loss and land use impact associated with the development of hydropower infrastructure and operation are unclear.

Land use change (LUC) in the larger hydropower projects is primarily associated with the construction of the large reservoirs, which is to be expected when comparing to smaller run-of-river hydropower (Bakken et al. 2014). In addition, both reservoir-based and run-of-river plants cause various level and range of permanent and temporary constructions. Permanent constructions are those needed during the lifetime of the project, such as permanent roads, dams, the power station, and parking areas. Temporary constructions are those needed only during the construction phase, such as storage areas for gravel and construction equipment, access roads, and parking areas, and these can be removed before the operational phase of the power plant.

1.1 Life cycle assessment and land use change

Life cycle assessment (LCA) identifies and measures the environmental impacts of product and service systems (Finnveden et al. 2009). Measures of habitat change and occupation on biodiversity are, when incorporated, included in the impact category "land use and land use change" (LULUC). Despite decades of effort, there is still no consensus on a proper methodology on how to assess impacts on biodiversity from LULUC in LCA (Milà i Canals et al. 2007; Koellner et al. 2013; Curran et al. 2016; Teixeira et al. 2016). As a consequence, a significant number of LCA studies on land extensive activities still do not include LULUC (Cherubini and Strømman 2011; Moreau et al. 2012; Michelsen et al. 2014). When included, the most common indicators are based on species richness (Curran et al. 2011; Michelsen and Lindner 2015; Curran et al. 2016) which only cover a limited part of the concept of biodiversity (Gotelli and Colwell 2001; Wolters et al. 2006; McGill et al. 2007; Penariol and Madi-Ravazzi 2013).

The calculation of changes in biogenic carbon stocks and changes in biogenic greenhouse gas (bGHG) emissions can be another approach to assess land use changes in hydropower development. The actual climate benefit of hydropower as opposed to more carbon intensive fuel sources is poorly understood due to biogenic GHG emissions, changes in albedo, and increased evaporation rates from reservoirs. The bGHG emissions are often left out of LCA (Hertwich 2013), and when included, they only address bGHG emissions from reservoirs, excluding emission from terrestrial LUC (Houghton et al. 2012). Carbon content has been defined for most terrestrial habitat types in Norway (Grønlund 2010) and can be used to improve the calculation of total emission from terrestrial LUC.

1.2 Ecological restoration and offsetting

Quantifying offsetting and restoration costs can be a third approach to assessing land use and land use changes in LCA. This offers an opportunity for calculating cost of lost biodiversity and is a complementary approach to assess and compare losses and gains of biodiversity, independent of normative judgments often found in present approaches on LULUC in LCA (Michelsen and Lindner 2015).

Actions to preserve biodiversity and prevent further loss have become widespread following increased awareness of the consequences of habitat destruction. Ecological restoration offers a significant contribution to mitigating and restoring biodiversity loss as a restored system can provide crucial ecosystem services (Bullock et al. 2011). Ecological restoration is today considered a most important tool for maintaining biodiversity at all levels, and it is a global aim to restore 15% of damaged habitats before 2020 (Convention on Biological Diversity 2010; EU 2010).

The mitigation hierarchy has been introduced as a concept in ecological restoration to facilitate implementation of restoration considerations in development projects, and the framework has four steps: (1) avoid impacts; (2) minimize impacts; (3) restore impacts on-site; and (4) offset impacts by restoring, preserving, enhancing, and/or establishing ecosystems off-site (McKenney and Kiesecker 2010; Business and Biodiversity Program 2013). In relation to hydropower, the opportunities to restore habitats are most obviously available when a hydropower plant is terminated, or by mitigating non-permanent infrastructure during construction or operation stage. Restoration for off-site compensation is another opportunity, however disputed, mainly related to the time lags, uncertainty, and risk of restoration failure (see, e.g., Curran et al. 2016; Souza et al. 2015). However, restoration for biodiversity offset gives new and relevant input to the calculation of restoration cost and quality (Moilanen et al. 2009) and contributes to make restoration cost a relevant approach for LCA.

1.3 Aim

The aim of this study is to test different approaches for assessing the impact of land use and land use change (LULUC) related to hydropower for use in LCA. The main purpose is to explore the different approaches for LULUC, considering only the foreground system with the dam construction. Three different approaches to assess land use impact were included: (1) ecosystem scarcity/vulnerability as indirect indicators to represent the impact on biodiversity in the ecosystems, (2) biogenic greenhouse gas (bGHG) emissions to represent reduction of ecosystem services, and (3) the cost of restoring affected habitats, in the context of the mitigation hierarchy. We use four hydropower plant projects in South Norway as our model case examples and compare the results, data requirements, validity, and accuracy of the different approaches to quantify the impact caused by LULUC. In particular, we look at whether the use of restoration cost adds relevant information, since this is a new approach to assess land use in LCA.

2 Material and methods

2.1 Case hydropower plant projects

To ensure consistency and enable comparison, the following criteria were used to identify and select hydropower plant case projects for this study, as they should all:

1. be in the planning phase (applied for or approved) to ensure data availability for both the "before" and "after" land use change (using current maps and technical specifications for the projects, respectively)
2. be located within the same region (Southern Norway; Vest-Agder, Aust-Agder, Telemark, and Vestfold Counties) to allow for geography and biodiversity comparison (Fig. 1)
3. have a predicted mean annual production capacity within a comparable range, enabling a relevant comparison of the impact per energy unit produced (kWh as the functional unit) for the individual projects.

Four case projects were identified, two were upgrading of existing plants (Skjerkevatn and Langevatn), and two were new plants (Dvergfossen and Kilandsfossen) (Fig. 1). Skjerkevatn will merge two previously regulated lakes by demolishing old dams, construction of one new, and expansion of one old dam and will raise the water level by 23 m and increase energy production by 43 GWh/year. Langevatn involves the expansion of one old dam, raising of the water level by 10 m, and increase of energy production by 18 GWh/year. Dvergfossen and Kilandsfossen are new run-of-river hydropower plants with smaller dams and unregulated basins upstream with an estimated production of 35.5 and 38.5 GWh/year, respectively. For further key information about the case projects, see Appendix I (Electronic Supplementary Material).

2.2 Mapping land use and land use change

Land use data were obtained from technical drawings in the permit applications for each project, and planned changes in land use were manually geo-referenced in ArcMap 10.1 as either polygons or lines with an added land use change-specific buffer ranging from 0 to 20 m (Appendix II, Electronic Supplementary Material). The buffers were based on distances from physical installations using orthophotos (www.norgeibilder.no) and were included to incorporate direct effects from the visual physical features around roads and other constructions. We excluded areas affected by previous land use to exclusively consider the land use impacts caused by expansion or new projects. Present land use and distribution of main habitat types were based on Norwegian Mapping Authority's N50 series (including alpine, freshwater, wetland, forest, and built-up areas). By comparing present and planned land use, we calculated total area changed, which habitat types were affected, and what they were transformed into. Total area occupied includes all types of permanent and temporary infrastructure, such as dams, roads, buildings, parking space, storage areas, and other areas used during construction phase. Total area also includes area covered by reservoir in the reservoir-based projects. Total area occupied and area occupied by the reservoir were divided by the yearly electricity production to allocate the land use to kWh/year and the energy density for each of the reservoir (m^2y/kWh).

2.3 Calculating impact of land use and land use change

Impacts from land use and land use change are traditionally divided between the impact caused by the actual transforming of the area from one type of use to another (transformation impact—TI) and the actual use which keeps the area in a new, and often assumed steady state, and prevents it to recover to the original state (occupational impacts—OI). OI is traditionally calculated using three key dimensions: the area (A) occupied, the relative difference in ecosystem quality between the defined use and a reference state (ΔQ), and the time (T) of occupation. Present situation is used as reference state. This choice put emphasizes on the new impacts and expansion of existing conditions (cf. Michelsen and Lindner 2014; Souza

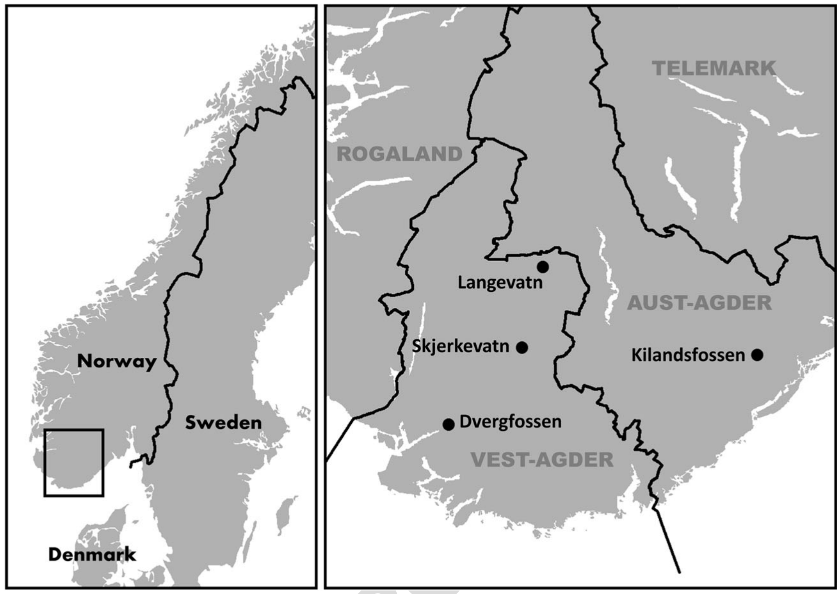

Fig. 1 The four hydropower plant projects used in the study: Skjerkevatn, Langevatn, Dvergfossen, and Kilandsfossen situated in Vest-Agder, Aust-Agder, Telemark, and Vestfold Counties, Southern Norway

et al. 2015). The duration of occupation is set equal to the lifetime of the hydropower plant (100 years; EPD 2007).

$$OI = \Delta Q^* T^* A \tag{1}$$

The TI is depending on the time it would take for a piece of land to recover (either from natural recovery or from the use of assisted restoration measures) to its natural state if occupation stopped. Assuming a linear recovery, the total impact of the transformation is given by ΔQ caused by the transformation, the area A transformed, and the time needed for restoration (t_{res}), divided by two. Restoration time depends on ecosystem type (see, e.g., Milà i Canals et al. (2007) and Curran et al. (2014) for more details).

$$TI = 0.5^* \Delta Q^* t_{res}{}^* A \tag{2}$$

Data on recovery time are based on general ecology and restoration ecology for different ecosystems. Colonization of disturbed habitats depends on factors like climatic condition, species growth rates, rate of soil development, and level of degradation (Aradottir and Hagen 2013); hence, the natural recovery in alpine ecosystems is slower than in lowland ecosystems due to harsh climactic conditions and a shorter growth season, in particular when the degradation is severe. There is no consensus or total answer to restoration time for Northern ecosystems. The restoration time in our study was set to 500 years for alpine and wetland ecosystems and 200 years for forest (Drescher et al. 2008; Moreno-Mateos et al. 2012). We are aware that these numbers will affect the results, and improved data on recovery and restoration time must always be considered when applying restoration cost as an approach for LCA.

2.3.1 Using ecosystem scarcity, vulnerability, and quality for land use impact assessment

To assess impacts on ecosystem quality (Q), a combination of ecosystem scarcity (ES), ecosystem vulnerability (EV), and conditions for maintained biodiversity (CMB) has been proposed (Michelsen 2007; Coelho and Michelsen 2014):

$$Q = ES^* EV^* CMB \tag{3}$$

ES represents the inherent scarcity or rareness of an ecosystem, assuming that scarce ecosystems have a higher risk of damage caused by stochastic processes due to smaller populations and thus need extra attention (Weidema and Lindeijer 2001; Lande et al. 2003; IUCN 2012). Values for ES can be calculated at any hierarchical level, e.g., biome, landscape, or ecosystem depending on data availability and the purpose of the study, and a normalized value for ES is proposed given by the following:

$$ES = 1 - \frac{A_{pot}}{A_{max}} \tag{4}$$

where (A_{pot}) represents the potential area of the ecosystem in focus (Michelsen 2008) and A_{max} is the total area included and used to normalize A_{pot}. In this study, we use data from South Norway and A_{max} is then equal to the total area of Southern Norway, while A_{pot} are areas of alpine ecosystems, wetlands, and forests in the region. All area data was collected from Statistics Norway (www.ssb.no).

EV represents the current pressure on an ecosystem and is calculated based on the proportion of the ecosystem still remaining following the equation

$$EV = \frac{1}{1-\text{fraction lost}} \quad (5)$$

(Peter et al. 1998; Michelsen 2008; Coelho and Michelsen 2014). This is a consequence of the species-area relationship (MacArthur and Wilson 1967). When the fraction lost is approaching 1, EV will go towards infinity. Michelsen (2008) suggested normalizing the values based on the most vulnerable ecosystem, giving scores between 0 and 1, where 1 represents the most vulnerable ecosystem.

CMB is an indicator for how well conditions for biodiversity are maintained in an ecosystem given the land use in focus (Michelsen 2008; Coelho and Michelsen 2014). This will be ecosystem dependent, and key factors must be identified (Michelsen 2008), e.g., in a managed boreal forest tree cover, tree species composition and dead wood are important, in a bog the water level is important etc. In our case, all affected ecosystems are occupied and replaced into either built or inundated environment, giving CMB equal 0 since there are no biodiversity given with the new land use, and no further development of ecosystem specific key factor is needed. Changes in ecosystem quality in terms of biodiversity (ΔQ) are then given by the difference in quality before and after the land use change, given by the equation

$$\Delta Q = ES*EV*\left(CMB_{ref}-CMB_{t1}\right) = ES*EV*(1-0) = ES*EV \quad (6)$$

where CMB_{ref} is the CMB for pristine environment (1 per definition) and CMB_{t1} is the CMB after the land use change.

2.3.2 Biogenic greenhouse gas emissions (bGHG) from LUC

Amount of carbon released after disturbance in natural systems depends on the type and duration of the disturbance and the amount of carbon stored in the system (Zummo and Friedland 2011). Construction of hydropower plants and reservoirs causes permanent and temporary changes in terrestrial ecosystem and rivers and permanent flooding in limnic and terrestrial systems. Change of areas previously covered by freshwater was excluded from this study, as these only covered minimal area. Emissions were calculated as gross emissions over the lifetime of the hydropower plant. bGHG emissions were calculated separately for permanent construction, temporary construction, and the reservoir. Data on carbon content has been collected for different natural systems in Norway (Grønlund et al. 2010), and for calculations, all carbon released from terrestrial areas was assumed released as CO_2.

LUC related to permanent construction are severe and assumed to cause 100% carbon release in above ground biomass and 75% in below ground (root) and soil biomass during the lifetime. For temporary construction, the LUC is assumed to be less severe and comparable to the carbon release during land transformation from natural to agricultural land, implying 100% carbon release in above ground biomass and 25% in remaining compartments (Guo and Gifford 2002).

bGHG emissions from new reservoirs were calculated according to Tier 1 Guidelines by the IPCC (2003) with default values for CO_2, CH_4, and N_2O emissions per m^2 and year. These were adjusted with a 100-year global warming potential to CO_2e. The guidelines predict stable CH_4 and N_2O emissions for the whole lifetime of the reservoir, while CO_2 from the initial flooding cease after approximately 10 years. However, in this study, we assumed stable emissions also for CO_2 for the whole lifetime as a consequence of biological material transferred to the reservoir, mainly from snow melting/flooding. Total lifetime emission from the reservoir is then 10.80 $kgCO_2e/m^2$.

Emissions caused by permanent construction, temporary construction, and the reservoir were added together for each case and divided over the individual lifetime production of electricity, giving a comparative metric in units of CO_2e bGHG emissions per kWh produced.

2.3.3 Restoration actions and cost

The cost of restoration will reflect the effort and capacity for recovery of disturbed or destroyed ecosystem to a resilient natural condition. In the context of biodiversity, offsetting restoration cost is the calculated cost off-site to compensate for impacts on-site (ICFGHK 2013). In this study, restoration of alpine, wetland, and forest ecosystems has been considered, leaving out freshwater ecosystems. Due to lack of available background data from offset sites, we developed restoration scenarios to illustrate a general approach to calculating restoration cost and calculated restoration cost based on case studies and literature review in the relevant ecosystem types.

The toolbox for restoration is diverse and what methods and actions to apply depends on factors like nature conditions, type of disturbance (range and intensity), logistics, traditions, and experiences (Aradottir and Hagen 2013). The actions used for our purpose are based on applied restoration of boreal ecosystems from Finnish boreal forest and wetland and Norwegian alpine restoration, where cost of specific restoration actions in each ecosystem were available (Hagen and Evju 2013; Hagen et al. 2014; Simil and Junninen 2012; Aapala et al. 2014).

For modeling purposes, it was assumed that restoration would take place in an area of equal size to the area affected by land use changes in each case project and that all the ecosystems were restored one to one in terms of size. To minimize edge effects, it was assumed that the restoration site was circular and a total length of roads to be removed was set to two times the diameter of the area. For alpine restoration, the following restoration actions were used for the calculations: adding topsoil to 100% of the restoration area, application of fertilizer and native seeds to 30%, and plant shrubs on 5% (for details, see Appendix III, Electronic Supplementary Material; Hagen and Evju 2013; Hagen et al. 2014). In wetland and forest ecosystem restoration, the following actions are used: filling ditches, removing trees in wetland, uprooting, girdling, and creation of forest gaps (for details, see Appendix III, Electronic Supplementary Material; Simil and Junninen 2012; Aapala et al. 2014). Details about actual cost for different restoration measures are listed in Appendix III (Electronic Supplementary Material). The total cost found by multiplying all required effort with the cost of that effort and the area of natural land affected by land use change, and summing across all ecosystems.

3 Results

3.1 Land use change

The affected areas at Skjerkevatn, Langevatn, Dvergfossen, and Kilandsfossen are 1.05, 0.76, 0.04, and 0.18 km^2, respectively (Fig. 2). Forest was the dominant ecosystem for Langevatn (84% of total area), Dvergfossen (58%), and Kilandsfossen (71%). Alpine was the dominant ecosystem in Skjerkevatn (56%). Wetland, and freshwater covered small areas in all case studies (6 to 14%). The mapping method makes it possible to track the changes for all land use classes and ecosystems (Appendix IV and V, Electronic Supplementary Material). After development, the reservoir was the dominating land use for Skjerkevatn (69%) and Langevatn (74%), and land use related to permanent construction was below 20% for both, while for Kilandsfossen, permanent constructions were 67% of the land use (Fig. 2).

3.2 Ecosystem scarcity, vulnerability, and CMB

Separate ΔQ-values were calculated for all terrestrial ecosystems (Appendix VI, Electronic Supplementary Material). The total land use impact (measured in $\Delta Q \times km^2 \times y$) was 138.2 for Skjerkevatn, 63.6 for Langevatn, 3.6 for Dvergfossen, and 16.2 for Kilandsfossen. For all cases, TI were larger than OI (Fig. 3), since t_{res} for the ecosystems is twice the lifetime of the installations or more. The total impact caused by LULUC, the sum of both TI and OI, per FU ($\Delta Qm^2y/kWh$) was similar for Skjerkevatn and Langevatn with 3.2×10^{-2} and 3.5×10^{-2}, respectively, while much smaller for Dvergfossen (1×10^{-3}) and Kilandsfossen (4.2×10^{-3}).

3.3 bGHG emissions

The main source of CO_2e came from LUC related to the permanent construction, followed by the reservoir, and least related to temporary construction. Skjerkevatn had the highest gross emission, followed closely by Langevatn (Table 1). Emissions from Dvergfossen and Kilandsfossen were much lower compared to Skjerkevatn and Langevatn, with one clear exception; emissions associated with permanent construction in Kilandsfossen were almost as high as Langevatn (6.02 kT). CO_2e per kWh over the lifetime of the hydropower plant was lowest for Dvergfossen and highest for Langevatn (Table 1). The highest contribution to permanent construction gross emission came from wetland in Skjerkevatn and forest soil in Langevatn, Dvergfossen, and Kilandsfossen. Removal of above ground biomass in forest ecosystems was the largest contributor to gross emissions related to temporary construction in Langevatn, Dvergfossen, and Kilandsfossen. For Skjerkevatn, the main contribution came from wetland and alpine ecosystems.

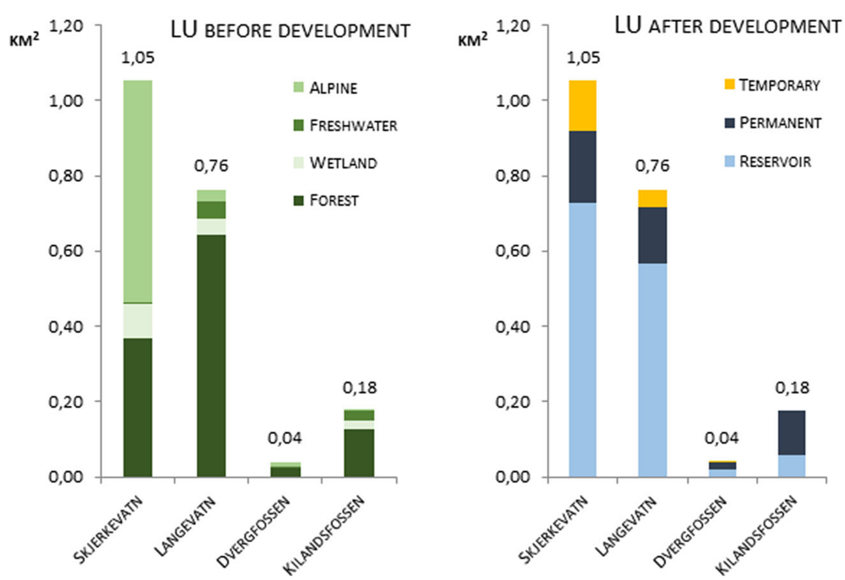

Fig. 2 Land use (LU) before (*left*) and after (*right*) development (LUC) for each case, illustrating that total LU is unchanged while LU type changes

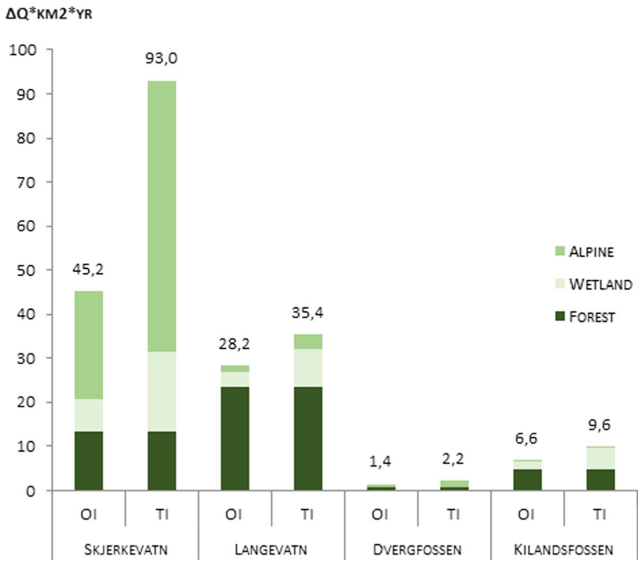

Fig. 3 Occupation impacts (OI) and transformation impacts (TI) on alpine, wetland, and forest ecosystems for Skjerkevatn, Langevatn, Dvergfossen, and Kilandsfossen. TI is 2.5 times larger than OI for alpine and wetland. The impacts on forest ecosystems are equal for OI and TI

3.4 Restoration cost

Restoration costs for all action and ecosystems are listed for each case project (Table 2). Total restoration cost is highest at Skjerkevatn and by far the lowest for Dvergfossen (Fig. 4). However, restoration cost per total area restored was approx. 0.90 USD/m^2 for Skjerkevatn, Langevatn, and Dvergfossen and significantly higher for Kilandsfossen with 1.57 USD/m^2 (Table 2). The cost per kWh produced over the lifetime was highest for Langevatn with 3.52×10^{-4} USD/kWh, and quite the same for Skjerkevatn. For Dvergfossen and Kilandsfossen, the cost was lower (Table 2).

Wetland restoration was the largest overall cost, contributing 66, 50, and 70% of the total cost for Skjerkevatn, Langevatn, and Kilandsfossen, respectively. The high total cost of wetland restoration was largely due to the cost of tree felling and transportation, which alone makes up 89–93% of the wetland restoration costs. The forest restoration actions contributed significantly to the cost in all cases, and in Dvergfossen, forest restoration cost was dominant with 58% of the total cost (Table 2).

3.5 Comparing methods

The results for the cases were normalized based on the highest value for each method, enabling comparison between them (Fig. 5). Mean values were used for GHG emissions. All three approaches give the same ranking of projects; Langevatn had the highest impact per kWh for all methods, while Dvergfossen had the lowest impacts for all methods (only 3% compared to Langevatn). The results for Skjerkevatn and Kilandsfossen showed more variation. The value for ES/EV in Skjerkevatn was 91%, and restoration cost was 66% of Langevatn's maximum, while the values for the results for CO_2e/kWh and the basic LUC/kWh were 56–58% (Fig. 5). In Kilandsfossen, the CO_2e emission was 22% and restoration cost was 20% of the values for Langevatn, while the basic LUC and ES/EV were 11–12% per kWh produced (Fig. 5).

4 Discussion

4.1 Does permanent infrastructure have larger ecosystem impact in small power plants?

Reservoirs caused a larger total area affected in reservoir-based hydropower, but permanent infrastructure has similar absolute impact for both run-of-river and reservoir-based hydropower. Land use related to infrastructure is consequently relatively more important for smaller run-of-river hydropower, and consistent with the findings for assessment of a large number of Norwegian small-scale plants (Hagen and Erikstad 2013). Small-scale hydropower plants are also reported to have larger impact on red-listed species (Bakken 2014). This indicates that total impact from land use per kWh, and not just

Table 1 Estimated CO_2-equivalent emissions for planned land use change (LUC) in Skjerkevatn, Langevatn, Dvergfossen, and Kilandsfossen

	Skjerkevatn	Langevatn	Dvergfossen	Kilandsfossen
Gross emissions (kT CO_2e)	17.41–19.83	13.80–14.17	0.88–0.93	6.59
Reservoir (kT CO_2e)	7.84	5.74	0.19	0.57
PIC (kT CO_2)	7.82–9.70	6.94–7.25	0.66–0.74	6.02
TIC (kT CO_2)	1.74–2.29	1.13–1.18	0,03	–
Emission per area (kgCO_2e/m^2)	16.55–18.85	18.11–18.59	21.78–23.03	37.66
Emission per lifetime production (gCO2e/kWh)	4.05–4.61	7.67–7.87	0.25–0.26	1.71

The gross emissions in kTCO_2-equivalentes are calculated for three categories of LUC: the reservoir, permanent infrastructure construction (PIC), and temporary infrastructure construction (TIC). The emissions are based on average carbon content in natural land use types (Grønlund 2010) affected by land use change. See Appendix V for land use in each power plant, distributed in habitat types: alpine, wetland, and forest

Table 2 Estimated offset restoration costs for Skjerkevatn, Langevatn, Dvergfossen, and Kilandsfossen based on ecosystem specific restoration measure for a hypothetical offset restoration site

Total cost of restoration actions (USD)	Skjerkevatn 830,307	Langevatn 577,745	Dvergfossen 38,196	Kilandsfossen 274,003
Alpine cost	*105,510*	*20,667*	*15,688*	*2173*
Procure land	10,874	494	288	6
Remove roads + add topsoil	70,100	14,943	11,408	1605
Fertilize and seed	21,030	4483	3422	482
Plant shrubs	3505	747	570	80
Wetland cost	*553,825*	*285,252*	–	*195,919*
Procure land	10,999	5549	–	3753
Remove roads	27,305	19,393	–	15,950
Fill ditches	1270	902	–	742
Fell trees	165,448	83,459	–	56,455
Remove timber	348,802	175,949	–	119,019
Forest cost	*170,973*	*271,826*	*22,508*	*75,911*
Procure land	51,881	90,450	3284	18,505
Remove roads	59,301	78,301	14,920	35,416
Fill ditches	2758	3642	694	1647
Uproot trees	21,936	38,244	1388	7824
Create glades	26,323	45,892	1666	9389
Girdle trees	8774	15,297	555	3130
Cost per m^2 restored (USD/m^2)	0.96	0.83	0.98	1.57
Cost per FU over LT (USD/kWh)	$2.34*10^{-04}$	$3.52*10^{-04}$	$1.12*10^{-05}$	$7.12*10^{-05}$

Restoration costs are based on active restoration projects and literature review (Hagen and Evju 2013; Hagen et al. 2014; Simil and Junninen 2012a; Aapala et al. 2014)

the energy density of the reservoir, should be included in LCA when evaluating smaller run-of-river hydropower development. New hydropower development is often considered based on the energy density of the reservoir (Hertwich 2013), which might be adequate for assessing energy purposes but does not capture ecosystem impacts.

In Dvergfossen, most of the development is situated on previously disturbed land, and the new impact does not add new disturbed areas. By locating development projects to previously disturbed land, further destruction of natural systems was avoided as were further carbon emissions, biodiversity loss, and ecosystem quality reduction. This, however, depends on using the present state as reference, as other choices (e.g., potential natural vegetation) would give different results (Koellner et al. 2013; Coelho and Michelsen 2014; Michelsen and Lindner 2015).

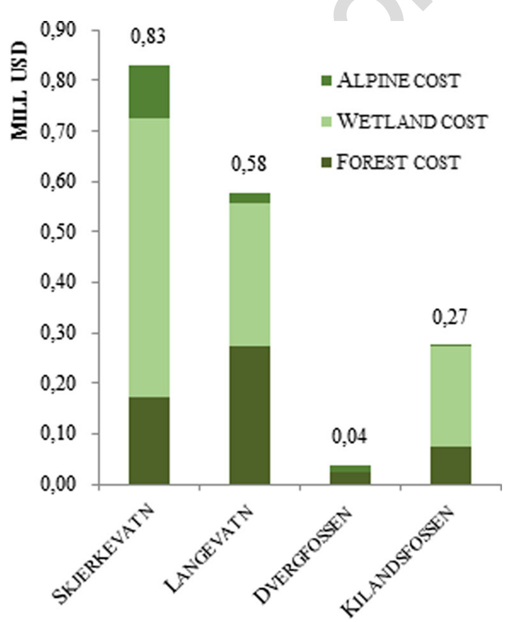

Fig. 4 Distribution of restoration cost per ecosystem type (alpine, wetland, and forest) for each study case project. For details of restoration costs per measure, see Appendix III (Electronic Supplementary Material)

4.2 Are ecosystem scarcity and vulnerability sensitive to normalization of values?

When using ecosystem scarcity and vulnerability as indicator for ecosystem value, transformation impact gives higher total contribution than occupation impact for all cases. This is because restoration time is assumed to be more than twice the lifetime of the installations in the cases included. In LCA studies, land occupation is more frequently included than land transformation (Cherubini and Strømman 2011), partly due to better methodologies, but also based on an assumption that occupation impacts are more important than transformation

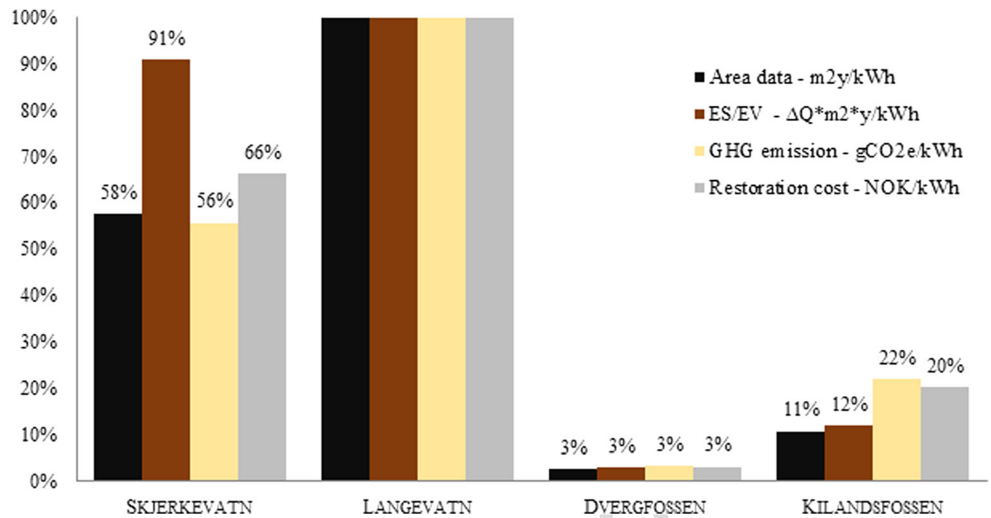

Fig. 5 Comparison of ecosystem impact between all methods assessed in the study for each case hydropower plant project. For comparison, the values are normalized, and the highest value for each method was set to 100%

impacts. However, in this study, the overall impact would be severely underestimated if only occupation impact were included. The relative impact of transformation of alpine and wetland areas were 2.5 times larger than for forested areas due to longer restoration times. The large proportion of alpine ecosystem causes the high transformation impact at Skjerkevatn compared to the other cases. The method used here is sensitive to restoration time, and a better justification of restoration time is recommended to increase the validity of the method (cf. Curran et al. 2014).

The use of ecosystem scarcity and vulnerability as a quality indicator for biodiversity implicitly assumes that what is rare is valuable and makes calculation possible despite significant knowledge gaps concerning ecosystem composition, structure, and function. However, the values for ecosystem scarcity depend on the value chosen for A_{max}, as it is used for normalizing the results (Coelho and Michelsen 2014). In this analysis, regional area data for Southern Norway is used as A_{max}. It reflects the regional natural composition of the ecosystems examined and fits the data availability at a regional level for the remaining fraction used in the ecosystem vulnerability calculations. If instead total area of Norway was used as A_{max} and the total distribution of the relevant ecosystems in Norway, this would have changed the scarcity scores for the ecosystems.

4.3 Are carbon calculations different for permanent and temporary constructions?

The bGHG emissions follow the other methods in ranking of impact per kWh for the case projects. The relatively high values for Kilandsfossen are consequences of the large share of permanent construction and a large part of carbon-rich wetlands that are changed into permanent infrastructure. Ideally, the calculations for carbon should have been net carbon equivalent fluxes from the area over the lifetime in a consequential LCA, where both emission and sequestration from the whole area over the lifetime could be included. It would also include information on the carbon flux in the area if no development occurs. Forest ecosystems currently sequester more carbon than they emit and wetland and freshwater systems have net bGHG emissions if left untouched (Tremblay et al. 2005; Grønlund et al. 2010). There are currently no available carbon flux measurements for alpine ecosystems, but due to low soil respiration and primary production, the fluxes are smaller than those found in other ecosystems (Grønlund et al. 2010). The emissions associated with permanent construction were largest in all cases and are also the areas where no biodiversity recolonization is expected and will therefore not contribute to the future carbon sequestration. The areas affected by temporary constructions will be recolonized and therefore contribute to carbon sequestration over the lifetime.

Emissions from reservoirs are complicated and uncertain (Hertwich 2013). After flooding, carbon in the soil is washed out, and distribution of the soil in the water column and the degree of sedimentation will determine the breakdown and emission of the carbon. Soil particles are transported downstream and outside the physical system boundaries used in the presented cases, and most likely gives an underestimation for emissions from the reservoir.

High-resolution data are available for carbon content in different ecosystems, including several specific sub-classes with information on carbon content and area covered, used to estimate total carbon content in Norwegian vegetation and soil (Grønlund et al. 2010). However, the available land cover maps (N50) do not have the same resolution, especially for different soil types and wetland depth. The carbon content of soil can vary substantially depending on amount of organic content, and the GHG emissions would therefore probably vary substantially with soil type, and the IPCC Tier 1 calculation does not take into account the soil types that are flooded when reservoirs are established. Wetland and soil have the largest carbon stores in the boreal zone (IPCC 2014), and more detailed mapping on their occurrence would give more specific results for the emission estimates.

4.4 Will calculation of restoration cost contribute to the calculation of LULUC in LCA?

Present proposals on how to include impacts from LULUC in LCA are all to a certain degree based on normative choices on which aspects of biodiversity (e.g., rareness, endemism, structural diversity, etc.) that is to be included and the relative emphasize on these (Michelsen and Lindner 2015; Curran et al. 2016). The use of restoration cost offers a complementary approach that takes into account different aspects of biodiversity without any need of weighting the different aspects to each other. This is because full restoration of ecosystem function has a cost which is independent of the ecosystem services delivered (Suding 2011). Adding restoration cost to LCA makes it possible to include common nature under pressure by most development projects, rather than emphasizing only rare and particularly valuable ecosystems. Assessments are more complex when restoration of ecological function is incomplete, or the degradation partial, requiring an approach to assess cost-effectiveness of degraded states relative to a reference state. Assessments are further complicated if restoration is conducted for the purpose of compensation, in which case interim damages from the time of degradation until restoration should also be considered. With the exception of compensation situations, restoration costs offer an approach free from subjective assessments of values of environmental impacts.

Wetlands were by far the most costly to restore in this study, in large part due to the felling and removal of unwanted trees. If other restoration techniques had been required, the cost might have been different. Afforestation of wetland has historically been common practice in Norway, and the choice of the afforested site was considered relevant. The restoration actions suggested in this paper are in no way exhaustive, but the cost of restoration is rarely published in the scientific literature. Those limited data that are available are highly variable both within a single ecosystem and between different ecosystem types (Bullock et al. 2011) and with high variability in timescales and inconsistent methods (Aronson et al. 2010). Improved data of total restoration cost is needed to implement this approach in LCA. Future improvements in restoration cost methodology should include cost of single restoration actions and techniques under different conditions, as well as other types of costs for developers and regulatory authorities, such as cost of acquiring land and transaction costs (planning, monitoring, and reporting the actions). Guidance on cost accounting can be found in the literature on habitat banking (ICFGHK 2013). Today, restoration actions according to the mitigation hierarchy are most often mandated by legal requirements on the developer (Vatn et al. 2011). In this case, a benefit-cost rationale is not required to justify restoration, only an assessment of the most cost-effective way of achieving no net loss (with whatever offset ratios that are required by the licensing or EIA process).

The restoration calculations conducted in this paper have only been concerned with the *cost* of the restoration actions, and assumes an offset site of equal size as the area affected by LUC. Factors like time and uncertainty make a large difference for the offset ratio (Hilderbrand et al. 2005; Currain et al. 2016), and by including these factors, the ratio might increase by a hundred-fold (Moilanen et al. 2009). Loss of habitat is immediate and occurs as the development is carried out, while gain (from mitigation/restoration) is uncertain and takes time. In this case, an offset ratio of 1:1 is most unlikely to secure no net loss/net gain to biodiversity in any ecosystem. The offset ratio chosen for the calculations in this paper is underestimates if restoration measures have a compensation purpose. On the other hand, direct income and other benefits related to the restoration actions should also be a part of such calculation. Cost-benefit analysis of restoration may potentially indicate a net benefit for some types of restoration (e.g., de Groot et al. 2013). For example, trees logged as part of wetland restoration in Finland yield an income from timber or biomass energy, which can in some cases cover the cost of restoration (Anon. 2015). The total restoration cost should therefore not be interpreted as the absolute cost of restoration for each case, but rather a relative measure for comparing between the different cases and ecosystems.

4.5 Outlook for further methodological development

All methods used in this study represent a contribution on how to implement land use impact in LCA. In the case projects, all methods provided comparable results for overall impact/kWh, where the power plant at Langevatn had the highest impact, followed by Skjerkevatn, Kilandsfossen, and Dvergfossen. Impacts on wetland ecosystems were identified as most important relative to impacts on other ecosystems by all methods. Impacts on alpine ecosystems were more important when using ecosystem scarcity/vulnerability as indicator compared to the other methods. The results for GHG emissions show the importance of including total LUC as a result of construction of infrastructure, and this is especially important for smaller hydropower development projects, due to the relative high importance of such infrastructure for small-scale hydropower.

All methods provide results that can be used to compare the impact from the included case studies. Still, all methods have a potential for further development to improve their accuracy for use in LCA, and it is important to have in mind that they all only cover elements of the land use impacts (see Curran et al.

2016). A combination of more methods is consequently advisable, but do of course increase the data demand.

In this, study we have used the present situation as reference. This put more weight on present natural areas than potential vegetation and was considered most relevant in this case (cf. old growth (OG) sites in Curran et al. 2016). However, this will influence the results (Coelho and Michelsen 2014), and further emphasis on the choice of reference situation is needed (Michelsen and Lindner 2015; Souza et al. 2015).

All the methods require improved mapping of land use both prior to and after land use change. The technical drawings had a high level of detail and were suitable for determining land occupation after development, but maps used to determine land use prior to development (N50) were a constraining factor for the analyses due to the low resolution, compared to other data sources. Maps with higher resolution will increase the accuracy and validity of all methods used in this paper and would improve consistency and reliability considerably. All parts of the scarcity/vulnerability model would benefit from more data and higher resolution, including setting value for A_{max}. More detailed mapping of nature and soil types with high carbon stores will give more specific results for the emission estimates.

Restoration cost is not mentioned in the review of land use methods in LCA presented by Curran et al. (2016) and represents a new approach to modeling impacts which is complementary to LCA. Calculation of restoration cost is essential as a basis for cost-effectiveness analysis of restoration alternatives. While total restoration cost of the case projects is probably not accurate, due to uncertainty of offset ratios (Maron et al. 2012; Hilderbrand et al. 2005), relative restoration costs may still be used to rank restoration alternatives. Restoration costs on-site may be compared to the costs of biodiversity offsets (off-site). Incorporation of restoration cost into LCA, as an indicator for biodiversity/ecosystem quality, seems promising, but will require further research, both in applied restoration ecology and appropriate methodology development for LULUC.

5 Conclusions

In this study, we have compared three different methods to approach impacts from land use and land use changes for implementation in LCA and exemplified these with case studies on hydropower projects. We conclude that all three methods can be used to measure impact from LULUC in LCA and actually compare impact from LULUC for the different cases. Overall, they give similar rankings of impacts in our study, larger impact for small and new power plants, less for larger and expanding existing plants. However, more case studies are needed to verify if this is an overall valid conclusion. The different models assess different aspects of land use impacts, but all methods show large variation of impact between the case power plants, which motivate the importance of including LULUC in LCA for hydropower projects. We introduced a novel approach in LCA using restoration cost for measuring impact of LULUC. This approach avoids most normative choices in existing methods on implementation of land use in LCA. We recommend that this approach in particular should be used on more cases to show its potential applicability. All methods used give a high resolution in impacts, but are demanding in terms of on-site data, and at least in the short terms, it is challenging to include background processes.

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AUTHOR QUERIES

AUTHOR PLEASE ANSWER ALL QUERIES.

Q1. Please check if affiliations are captured correctly.

Q2. Houghton et al. 2012; Grønlund 2010; Michelsen and Lindner 2014; Michelsen 2008; Bakken 2014; Currain et al. 2016 are cited in the body but its bibliographic information is missing. Kindly provide its bibliographic information. Otherwise, please delete it from the text/body.

Q3. The citation “Michelsen 2008” has been changed to “Michelsen, 2007” to match the author name/date in the reference list. Please check if the change is fine in this occurrence and modify the subsequent occurrences, if necessary.

Q4. References [Chaudhary et al, 2015, de Baan et al, 2013, Moen, 1999, Skarpaas et al, 2012] were provided in the reference list; however, this was not mentioned or cited in the manuscript. As a rule, all references given in the list of references should be cited in the main body. Please provide its citation in the body text.

Q5. Please provide complete bibliographic details of references Weidema and Lindeijer 2001; Simil and Junninen 2012; REN21 2013; Peter et al. 1988; Moreno-Mateos et al. 2012; Moen 1999; Gronlund et al. 2010; Dreshercher et al. 2008; Aapala et al. 2014.

Q6. Please provide atleast the first 3 names of authors for reference Vatn et al. 2011.