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An integrated framework to assess impacts on ecosystem services in LCA demonstrated by a case study of mining in Chile

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A B S T R A C T

Life Cycle Assessment (LCA) is a tool to quantitatively assess the environmental impacts associated to a product's life cycle. Since its conception, LCA has improved considerably in sophistication and scope. Yet efforts to incorporate ecosystem services (ES) are still at an early stage. We present a novel framework for assessing ES in LCA that integrates models from adjacent fields and partitions the required modeling steps into different phases of LCA. Physical models are first used to determine how physical units of ecosystems are transformed by industrial processes; ES models are then used to determine the losses or gains of ES per ecosystem unit, and economic valuation is used to normalize and weigh the total ES losses/gains. We demonstrate the framework for a case study on water extraction by the mining industry in Chile and compare ES losses that result from the transformation of wetland and coastal ecosystems respectively. The proposed framework advances current efforts to assess ES beyond land use impacts in LCA by presenting a coherent approach to deal with spatial and temporal variability of ES production and by incorporating socioeconomic aspects of ES use. It also facilitates the coupling of LCA with other ES databases currently being developed

1. Introduction

Life Cycle Assessment (LCA) is a widely accepted tool to assess the environmental impacts associated to the production, supply, use, reuse, recycling and/or disposal of products and services (Guinee, 2002). For a given product, LCA calculates the economic and environmental inputs and outputs (flows) that are required for its production, as well as the resulting environmental impacts throughout the product's entire life cycle. It therefore allows for comparison of the sustainability of two or more product systems that fulfill the same function (Goedkoop et al., 2009). It is also particularly strong in identifying environmental tradeoffs, e.g. when the environmental performance of one product life cycle stage is improved at the expense of another (Guinee, 2002). Traditionally, LCA models have used *midpoint* impact categories such as toxicity, acidification and global warming. These midpoint indicators link environmental flows to points in the cause-effect chain of each impact category, e.g. radiative forcing represents global warming potential (Bare et al., 2000). Trade-offs between different impact categories, however, cannot be easily described with the midpoint approach. Therefore some methodologies consider more aggregated *endpoint* impact categories like human health or ecosystem quality (Huijbregts et al., 2017; Goedkoop and Spriensma, 1999; Itsubo and Inaba, 2003). Endpoint indicators describe the relevance of environmental flows at the end of the cause-effect chain, reflecting society's understanding of their final effect (Bare et al., 2000).

Recently, efforts have intensified to incorporate ecosystem services (ES) as an impact category in LCA to describe changes in their supply/availability which could result from the economic and environmental flows in product systems. ES refer to "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life" (Daily, 1997) or more generally "the benefits people obtain from ecosystems" (Millennium Ecosystem Assessment, 2005). Comprehensive reviews of these efforts are provided by Othoniel et al. (2016) and Zhang et al. (2010). However, there are diverging views on whether ES should be considered as midpoint or endpoint impact categories in LCA (Dewulf et al., 2015; Koellner and Geyer, 2013). Other authors have even incorporated them as internal product system flows rather than impact categories (Schaubroeck et al., 2013). Nevertheless, adding ES next to conventional impact categories can provide a more holistic set of results and highlight new perspectives, even though there may be some overlap between indicators. In this paper we aim to provide an operational framework for the incorporation of ES as a midpoint impact category in LCA, using commonly accepted definitions of ES.

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Several challenges must first be overcome in order to achieve this (Othoniel et al., 2016). These challenges include but are not limited to:

- Lack of methods to incorporate impacts on ES resulting from drivers such as water withdrawal and pollution. The methods developed so far mostly focus on land use change as a driver and rely on existing land use change data to calculate the expected losses of ES (Saad et al., 2013; Koellner and Geyer, 2013; Brandão and Milà i Canals, 2013). This limits the assessments to terrestrial ecosystems and requires the use of broad land use types whose definitions convey only limited information about ecosystem function and ES productivity.
- Ecosystem processes and ES are spatially heterogeneous, even at small geographic scales (Carpenter et al., 2009). However, conventional impact assessment modeling in LCA "assumes a global set of average/standard conditions as regards the properties of the source and the receiving environment" (Finnveden et al., 2009). Attempts to deal with the spatial variability of ecosystem processes in recent LCA-ES approaches involve using either globally averaged values (Brandão and Milà i Canals, 2013) or spatially explicit flows at regional levels (Saad et al., 2013; Koellner and Geyer, 2013). It can be argued that both scales are still too coarse to obtain ecologically relevant results, while using finer scales requires data that may be impracticable to collect.
- Only a few studies have attempted to incorporate the socioeconomic aspects of ES demand and use (Bruel et al., 2016; Cao et al., 2015)
- The conventional LCA framework does not yet allow models to incorporate feedback or interdependencies between potentially competing ES. For example, while food provision may be seen as a desirable ES, its increased use may lead to a decrease in other ES like carbon sequestration due to deforestation for agriculture.

The framework for ES assessment in LCA that we present addresses some of these challenges, specifically those related to temporal and spatial variability, non-land use drivers and socioeconomic aspects of ES demand and use. Our framework integrates methodologies from the fields of ecosystem services and environmental sciences within the traditional conventional LCA framework. The sequential application of these methodologies allows the use of state-of-the-art ES concepts and databases within LCA, which offers important advantages for the quantitative assessment of ES.

We demonstrate the framework through a case study of water extraction by mining industries in the north of Chile, where nearly 30% of the world's copper is produced (U.S. Geological Survey, 2016). The case is of utmost relevance because the largest mines are located in extremely arid and semi-arid locations, and copper production requires large volumes of water. This has intensified competition over the scarce resource and caused significant water stress for local human populations and ecosystems (Romero et al., 2012). Because of this, the government started restricting ground and surface water rights while promoting desalination of seawater to satisfy the industry's increasing demand (Montes Prunes and Cantallopts, 2014). To gain more insight on how ES could be affected by this shift in water supply, we apply our framework to this case study and assess the impacts on ES of each alternative water source.

2. Methods

2.1. Conceptual framework

The conventional LCA framework is represented by four phases: Goal and Scope Definition, Life Cycle Inventory (LCI), Life Cycle Impact Assessment (LCIA), and Results and Interpretation (Rebitzer et al., 2004). In the LCI phase, production systems are modelled as interconnected discrete unit processes with economic flows (e.g., inputs and outputs of raw materials, manufactured parts, energy) and environmental flows (e.g., inputs and outputs of natural resources, chemical substance emissions). The flows are quantified for a desired product output or *functional unit* that is delivered by a production system (e.g. 1 kWh of electricity, 1 pair of running shoes). Production systems typically involve hundreds of unit processes, therefore the calculations use a simplified model that scales linearly and assumes a steady state condition (Heijungs and Suh, 2002; Rebitzer et al., 2004). The LCIA phase translates the aggregated environmental flows into selected impact categories by using multiplicative *characterization factors* (CF). In this way, the contributions of two or more different substances to an impact category can be represented using a single indicator (Pennington et al., 2004). For example CO_2 and CH_4 emissions both contribute to global warming, and their contribution is expressed in units of CO_2 equivalents. Characterization factors are calculated from characterization models, which can vary considerably in nature depending on the impact category assessed. A comprehensive review of LCIA characterization models is provided by Hauschild et al. (2013).

The complex ecosystem and socioeconomic dynamics required for ES modeling (see Introduction) are difficult to represent in the conventional LCA framework. As a result, important simplifications and approximations must be made. We adapt the LCA framework so that these simplifications are placed separately in three steps described below, and discuss the advantages of this placement for the assessment of impacts on ES in each step.

2.1.1. Step 1: Ecosystem transformations as environmental flows

We list environmental flows in the LCI phase as the physical transformation of ecosystems (expressed in units of area or volume) by individual unit processes. In other words, one or several types of ecosystems are "consumed" by the processes and replaced for others. These ecosystem transformations can result from three types of human interventions which can be associated to some of the main anthropogenic drivers of biodiversity loss and ecosystem change identified by the Millennium Ecosystem Assessment (2005): land use (associated to *habitat change*), natural resource use (associated to *overexploitation*) and substance emissions (associated to *pollution*). Drivers like habitat change and overexploitation have been considered in previous LCA-ES and LCA-biodiversity frameworks (Koellner and Geyer, 2013; Saad et al., 2013; Verones et al., 2015). Our framework can additionally incorporate ecosystem changes that result from pollution-type drivers. For example, the emission of substances like sulfur, nitrogen, ozone and mercury into the atmosphere may significantly affect several ecosystem processes and change species composition, even at sub lethal concentrations (Lovett et al., 2009). In our framework, these effects can be represented as the replacement of a highly productive ecosystem for a less productive one.

The approach we propose is analogous to the land use inventories commonly used in LCA, but expanded to incorporate many different types of terrestrial and aquatic ecosystems. Modeling certain ecosystem processes in the LCI stage has been proposed in the past. For example De Rosa et al. (2017) incorporated carbon fluxes from forest ecosystems in the LCI phase, yet their model focused on impacts other than ES. Our proposed ecosystem flows in the LCI account for spatial variation yet the transformations are independent of geographical location as such, e.g. 10 ha of tall grass prairie represent the same flow whether in Canada or in Europe. The spatial resolution of the model is therefore linked solely to the classification of ecosystem types which can be defined according to existing databases, e.g. Olson et al. (2001). A detailed discussion on this and other options for classification of land use and ecosystem types in LCA is provided by Koellner et al. (2013).

Building a LCI is a data intensive, time consuming pursuit and our framework is not exempted from this. However, ecosystem transformations can be estimated for many unit processes from existing physical environmental models. In some cases, they can be directly measured as in the case of crop management or static plume dispersions of chemicals in aquatic environments. The total number of relationships to be estimated may in fact be much lower than when relating environmental flows directly to impacts on individual ES as done by Arbault et al. (2014), Saad et al. (2013) and Dewulf et al. (2015). The idea of incorporating parts of the ES modeling in the inventory phase rather than in the impact assessment phase has recently been suggested by Callesen (2016), but the methodological steps required for general application were not yet fully developed (Rugani et al., 2017). Here we provide essential steps into that direction.

2.1.2. Step 2: Losses and gains of ES as life cycle impacts

We use ES provisioning rates (per spatial unit) of different ecosystem types as CFs. These CFs are multiplied by the aggregated ecosystem transformations from the previous step to obtain the total losses or gains in ecosystem services. In this way, each ecosystem service that is potentially affected constitutes an individual impact category with its corresponding indicator. This characterization method is considered to be useful because extensive work has been conducted on determining the provision of ecosystem services for a range of ecosystem types, e.g., GLOBIO (Alkemade et al., 2009) and The Natural Capital Project InVEST (Sharp et al., 2015). These efforts are only expected to increase in time and can be linked directly to our framework. Also, provided that ecosystem types have been identified at an adequate resolution in the inventory stage, our characterization method is not spatially or temporally explicit. In other words, we assume that the average yearly production of ES is constant for a given ecosystem type.

2.1.3. Step 3: Economic valuation as normalization and weighing

Optional normalization and weighing by applying monetary valuation methods is included to incorporate the socioeconomic aspects of ES demand and use. Cao et al. (2015) and Bruel et al. (2016) have already incorporated economic valuation of ES in LCA. The former used it as a characterization method to calculate an endpoint indicator in monetary units, while the latter used it to determine weighing factors. Like Bruel et al. (2016), we maintain that economic valuation is essentially a weighing exercise that can be used to represent ES demand and use by society. This way, we follow the suggestion of Koellner and Geyer (2013) to "try to separate 'natural science-based' [our steps 1 and 2] from 'value-based' approaches' [our step 3]" in weighing. Particularly for ES there is a large body of literature on valuation, its methods and available databases that include global and local estimates (Pearce et al., 2006; de Groot et al., 2012). As with weighing in LCA (Finnveden et al., 2009), monetary valuation of ES has been subject of extensive debate. Concerns have been raised regarding the implications of commodifying nature, as well as the complexity to represent important cultural and intrinsic values in monetary terms (Saarikoski et al., 2016). Because of this, monetary valuation may be particularly useful in a comparative mode to assess potential trade-offs. While not attempting to resolve the problems associated with valuation and weighing, our framework allows the incorporation of these methods, which can be selected based on the study's goal and objectives. This approach is preferable because value choices are separated from the impact assessment calculations and presented in a more transparent way.

Fig. 1 presents the steps required for the adaptations presented in this section graphically and compares them to the conventional LCA framework and existing efforts to incorporate ES in LCA.

2.2. Case study

2.2.1. Background

Most of the largest mining operations in Chile are located in the northern regions, between the Atacama Desert and the Chilean *altiplano* (high plateau). Despite the aridity of these environments, scant water sources are sparsely scattered across the landscape forming wetlands and *salares* (salt lakes) on the surface that host several threatened species like the *Chilean*, *James* and *Andean* flamingoes. In addition to being a natural habitat to conserve, salares provide a variety of ecosystem services which have been widely recognized and reported in literature. These include provisioning services like water for local communities and livestock, regulating services like carbon sequestration and flood control, and cultural services like recreation and ecotourism (Correa-Araneda et al., 2011; RIDES, 2005). Copper mining is the main source of economic output for the region, yet its high water demand is in stark contrast to the availability. In the year 2016, the total water consumption by mining industries -excluding recirculation- was 16.4 m^3/s, of which 79% was freshwater and 21% seawater. Aiming to reduce water stress in the region, the government projects a reduction of 17% in freshwater use and a 173% increase in seawater use by the year 2027 (Montes Prunes and Cantallopts, 2016).

2.2.2. Goal and scope definition

We use LCA to compare the impacts on ES of two alternative water supply systems used by the mining industry in Chile: groundwater extraction and seawater desalination. Four ecosystem services were chosen for analysis: food provision, carbon sequestration, tourism and recreation, and flood protection. These ES have been widely reported in literature as relevant for the study area (Correa-Araneda et al., 2011; RIDES, 2005; Figueroa et al., 2010; Peillard et al., 2011). The functional unit for comparison was 1 m^3 of water delivered to a mine site.

2.2.3. Step 1: Life cycle inventory

A life cycle inventory of ecosystem transformations was constructed for each alternative. To exemplify the framework, only foreground unit processes were considered i.e., those pertaining to the water supply system. A proposal to extend the method to background processes (e.g. those involving raw materials, manufactured parts and energy supply to the production systems) is provided in the discussion section of this paper.

For the groundwater extraction alternative, a simplified hydrogeological model was used to estimate the area of wetlands affected. In the model, a decline in the phreatic level that results from groundwater pumping was calculated by applying Darcy's Law, which describes the flow of fluids through porous media (Bear and Cheng, 2010; de Smedt, 2009). We used average values for the hydrological and operational parameters of mine sites in the region (see Supplementary Material, Table S1). The salt lakes were modelled as irregular cones and geometric relationships were used to calculate the area of lake lost as a function of the decrease in salt lake depth. For simplicity of the LCA model, it was assumed that the wetlands became a barren desert land as they dried out (no ecosystem outflow). Detailed calculations for this model are provided in the Supplementary Material, Section S1. Verones et al. (2013) had previously calculated wetland area loss from groundwater extraction in the context of LCA, using partially analogous hydrogeological modeling. Their method applies the model to calculate regionalized fate factors as part of composite CFs that

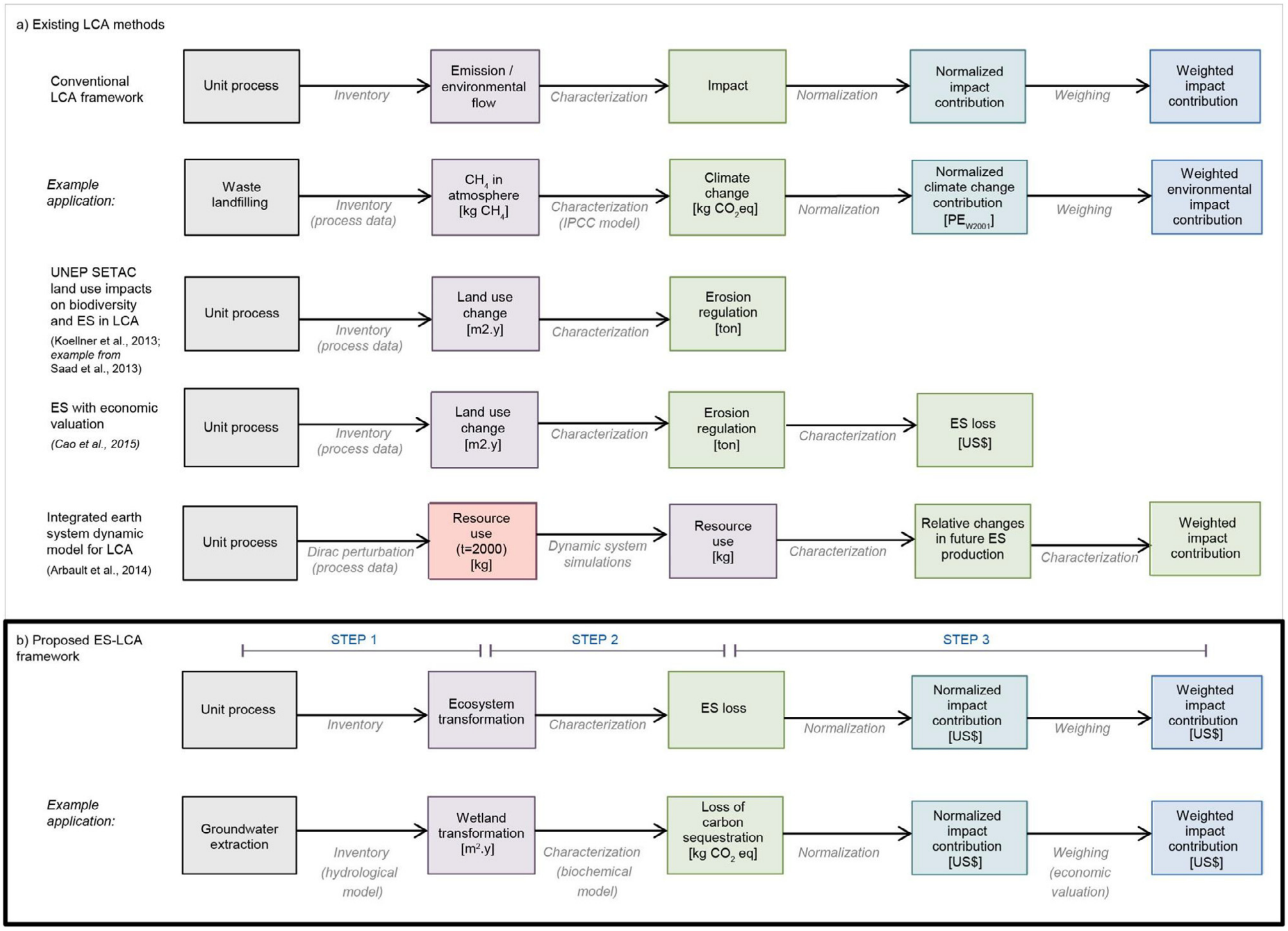

Fig. 1. Comparison of LCA-ES frameworks.

ultimately estimate biodiversity loss in relation to wetland area change. Our modeling of wetland area loss takes place in the LCI. Therefore the parameters and input data are fine-tuned to the specific unit process, in this case groundwater pumping for mining.

In the seawater desalination alternative, the ecosystem transformation involves a three-dimensional volume of seawater intake. Using the desalination plant efficiencies reported by Minera Spence (2015) in their Environmental Impact Assessment for the Spence Desalination Plant, an intake of 2.1 m^3 of seawater would be required to produce 1 m^3 of desalinated water. The brine disposal after desalination also transforms the coastal ecosystem; in this case a plume dispersion model developed by Minera Spence (2015) was used to determine the coastal area affected. The model, which is built on EPA's Visual Plumes application (Frick et al., 2003), is extrapolated linearly to predict a resulting impact area of 4153 m^2 for 1.1 m^3/s of brine outflow. This is equivalent to an ecosystem transformation of 1.32×10^{-4} m^2 y for every 1 m^3 of desalinated water supplied to the mine. The affected coastal waters were assumed to be rendered completely unproductive due to high salinity levels; hence there were no outflows for this case. Detailed calculations are presented in Section S2 of the Supplementary Material. The environmental flows for both production alternatives are summarized in Table 1.

2.2.4. Step 2: Impact assessment

Characterisation factors for four critical ES were determined.

Food production. The dominant food products from artisanal fisheries in the region are brown seaweed ("huiro") and the Chilean abalone or *loco* (a mollusk) which is highly valued in Asian markets (Servicio Nacional de Pesca y Acuicultura, 2016). Loss of food produce can occur in the seawater desalination system through three impact pathways: (1) impingement, (2) entrainment of marine organisms during seawater intake, and (3) effects of high salinity and chemicals discharged with the brine outflow (Elimelech and Phillip, 2011). Abalones and brown seaweed are attached to the bedrock in the intertidal zone, so it is assumed that no abalones or seaweed are impinged to the intake pipe. However, abalone larvae can be entrained. The entrainment impacts were calculated following the simplest form of the adult equivalent method (Horst, 1975), described in (Goodyear, 1978) and using proxy data from a study of 19 different desalination plants along the coast of California (WateReuse Association, 2011). A comprehensive discussion of more refined models is provided by EPRI (2004). For the model, we assumed an average mollusk survival rate of 0.275% (larvae to harvestable adult). Data on brown seaweed and abalone landings per hectare in the fisheries was used to determine the impacts of brine outflow, assuming the affected area was rendered completely unproductive.

Table 1
Life cycle inventory of ecosystem transformations from the supply of 1 m^3 of water to a mine site in north Chile.

Unit process	Inflows	Outflows
Alternative 1: seawater extraction & desalination		
Seawater extraction	2.1 m^3 seawater, North Chile coast	–
Brine disposal	North Chile coast: 1.32×10^{-4} m^2 y	–
Alternative 2: groundwater extraction		
Groundwater extraction	Salt lake: 3.47×10^{-2} m^2 y	–

Carbon sequestration. The rate of CO_2 uptake in the coast of northern Chile was based on primary productivity reported by Marin and Olivares (1999) in a bay on the coast of Antofagasta. For the salt lakes in the altiplano, Figueroa et al. (2010) reported an average carbon sequestration rate of 45 ton CO_2/ha y.

Tourism and recreation. We used tourism data from the *Salar de Surire* salt flat in the north of Chile, given that it is a protected area for which visitor statistics are kept and its main attractions are the salt lake and the endangered species it hosts (RAMSAR, 2017). In order to fit this impact category into the framework, the approximation is made that the amount of tourists visiting the site is proportional to the wetland area. The Chilean forestry authority reported 276 national and 600 foreign tourists in the year 2016 (CONAF, 2017), and the salt lake surface area is 15,858 ha (RAMSAR, 2017). According to surveys conducted by Figueroa et al. (2010), national visitors stay on average 1.1 days and foreign visitors stay 2.38 days. Tourism and recreation of coastal ecosystems were considered negligible because discharge areas are not located in the vicinity of touristic beaches.

Flood protection. The arid and steep conditions in the north of Chile mean that catastrophic flash flood events can occur with very little rain (NASA Earth Observatory, 2015). Even though this is such an important phenomenon in the north of Chile, there is no data available that can be used to quantify the impacts of flash floods events and to correlate wetland areas to flood retention. Instead, we use retention capacity (in cubic meters) per wetland area as an indicator of flood regulation services, which can be valued economically based on the cost per cubic meter of artificial flood retention structures. This indicator is also useful because it can be related to other ecosystem services like denitrification (Lane and D'Amico, 2010). Based on a study of different types of wetlands, Cernohous (1979) reported an average retention of 12 inches per acre (3084 m^3/ha) and Lane and D'Amico (2010) reported an average of 1619 m^3/ha. For simplicity, this study assumes the lower value, but more specific models can be developed for different types of wetlands (Krasnostein and Oldham, 2004).

ES not considered in the case study. Of the ES provided by Chilean wetlands that are reported in literature, several were not considered in the assessment. Some wetlands in the north of Chile provide local communities with fodder for livestock grazing, but its use is scarce and was assumed negligible. Provision of fuel and timber was also disregarded because this use is forbidden by law. Impacts on water provisioning services were not assessed because it is the function of the system under study and as such constitutes an economic flow rather than an impact. Water purification and waste treatment was not considered because there are no agricultural or industrial processes occurring upstream of the wetlands.

Table 2 presents the characterization factors for each impact pathway. Detailed calculations for each CF are presented in the Supplementary Material, Sections S3–S5.

2.2.5. Step 3: Normalizing and weighing

Monetary valuation was applied to obtain monetary values for each ES loss and used as a normalizing and weighing mechanism. Depending on the ES involved, different methods of valuation were used (e.g. market price vs. mitigation and restoration) to provide the most representative monetary estimate for each ES (de Groot et al., 2012). The values for food provision were calculated using direct market pricing methods, which basically take the value of the product or service on the market (Pascual and Muradian, 2010). Average export values for brown algae in the year 2014 (US$1.6/kg) were taken from the Chilean Fishing Promotion Office (Instituto de Fomento Pesquero 2016). The price of Chilean abalone (US$20/kg) was approximated from Chilean Central Bank statistics for 2015 (Banco Central de Chile 2016).

For carbon sequestration, cost based methods were used to estimate mitigation and restoration costs using carbon capture and storage (CCS) technologies. The Carbon Capture and Storage Association reports costs for CCS ranging between €60 and 90 per ton of carbon dioxide abated. However, they expect the costs to reduce to €35–50 per ton in the early 2020s due to improvements in technology (Carbon Capture and Storage Association 2016). The Global CCS Institute reports a range of US$23–92 per ton of CO_2 (Global CCS Institute 2011). For the model, an estimate of US$0.04/kg CO_2 was used, representing a low end estimate for future projections.

A similar approach was used to estimate the cost of wet detention ponds that can perform an equivalent flood protection service. Costs for these type of structures have been reported in the range of US$17.50–$35.00 per cubic meter of storage (EPA 1999).

The economic value of cultural services like tourism and recreation can be represented by travel cost methods, which relate the recreational experiences to the costs they have (e.g. travel expenses, opportunity costs of time) (Pascual and Muradian, 2010). Figueroa et al. (2010) used data based on extensive surveys to estimate the average expenditure per tourist per day in the protected wetlands of northern Chile: US$11.3 (national) and US$53.8 (foreign).

Table 2
Characterization factors for ecosystem services, for each impact pathway.

Characterization factors	CF wetland	CF coast	CF intake
Food provision, mollusks (kg/m^2 y; kg/m^3)	–	8.49E−04	9.29E−03
Food provision, algae (kg/m^2 y)	–	1.01E−02	–
Carbon sequestration (kg CO_2/m^2 y)	4.5	96.8	–
Flood protection (m^3 pond/m^2 y)	6.48E−03		
Tourism & recreation, national (d/m^2 y)	1.90E−06		
Tourism & recreation, foreign (d/m^2 y)	9.00E−06		

2.3. Sensitivity analysis

A sensitivity analysis was applied to understand how the spatial variability of the environmental flows and ES responses affected the outcome. In the LCI, the most spatially variable parameters in the system relate to the physical models used during the inventory phase to determine the area of ecosystems transformed. The estimate of the area is of central importance because the production of ES is directly proportional to it. For the salt lakes, it was determined through trial and error that (1) salt lake perimeter, (2) well distance from lake and (3) aquifer transmissivity present the highest variability and can have the largest impacts on the end results. Using the most extreme cases possible within the reported ranges (perimeter 10 km, well distance 2 km, aquifer transmissivity 7×10^{-3} m^2/s), the maximum salt lake area that can be affected according to the model was estimated to be 0.259 m^2 per cubic meter of water supplied.

We also tested the sensitivity of the LCIA, where a key source of variability is the larval and juvenile survival rate of the entrained species. For some types of mollusks (bivalves), survival from larvae to harvestable adults are found in the range of 0.5–0.05 percent (Menzel 1991). Juvenile survival rates for gastropods are commonly less than 2%, but can be as high as 76% (Gosselin and Qian 1997). We tested the range of 0.05–0.5% survival rate to harvestable adult and compared it to the average case.

3. Results

The results of the impact assessment are presented for each alternative in Table 3. These results indicate the losses of ES per cubic meter of water delivered to a mine site. For the seawater

Table 3
Losses of ecosystem services from water extraction by the mining industry in the north of Chile.

ES losses (unit/m^3)	Seawater desalination	Groundwater extraction
Food provision, mollusks (kg)	1.95E−02	
Food provision, algae (kg)	1.33E−06	
Carbon sequestration (kg CO_2)	1.27E−02	1.56E−01
Flood protection (m^3)		2.24E−04
Tourism & recreation, national (d)		6.58E−08
Tourism & recreation, foreign (d)		3.12E−07

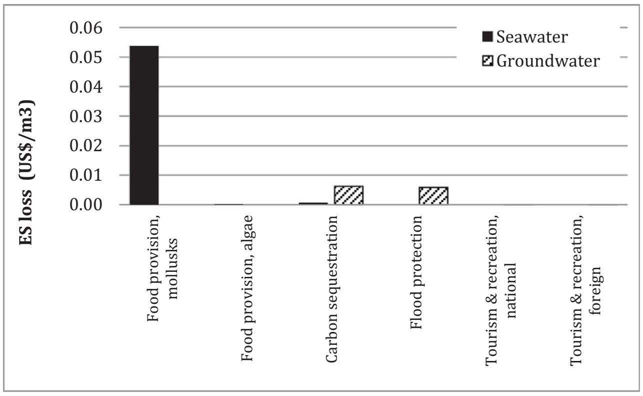

Fig. 2. Losses of ES in US$ per cubic meter of water supplied to a mine.

desalination system, it can be seen that the loss in production of mollusks is much larger than the loss in production of algae. Both the seawater intake and the brine disposal processes contribute to loss of mollusk production, whereas only the latter affects algal production. The reduction in carbon sequestration is one order of magnitude higher for groundwater extraction than for seawater desalination.

The results for each impact category, normalized and weighted according to their economic value are shown in Fig. 2. The market value of abalone mollusks is significantly higher than all other ES, which become almost negligible from an economic perspective.

Fig. 3 shows how both alternatives compare in terms of the losses of ecosystem services, using the government's water demand projections for the year 2025.

The results of the sensitivity analysis for the LCI (Fig. 4) give an overview of the uncertainty and variability of the model. Total losses in the worst case scenario compared to average situation are less than one order of magnitude higher. The results also highlight that the best performing option is different in the average case (groundwater) than in the worst case scenario (mixed).

The LCIA sensitivity analysis sheds some additional light on this, as it can be seen that a key cost driver is the abalone production. Testing the range of mortalities using average LCI parameters for the mixed 2027 projection we can predict a minimum loss of $10,704,157 (high mortality) and a maximum loss of $32,864,098 (low mortality).

4. Discussion

4.1. Process contributions to ES

One of the key applications of LCA is contribution analysis, which identifies the processes that are most relevant to specific impact categories. For the case study, the main contributor to the loss of ES is the abalone larvae that are lost via entrainment in the seawater intake pipes. The abalone is a closely monitored and controlled species, with capture bans during reproductive times of the year. However, the seawater intake mechanisms are entraining larvae year round, irrespective of temporary fishing bans. The high price of abalone increases even further the contribution of this process to the weighted results. The variability in natural mortality and entrainment rates is a highly sensitive aspect of the model; this suggests that this may be an aspect to investigate in more detail and supplement with field data.

The weighted contributions from other ES are small to negligible when compared to food provision. On the basis of this comparison, these results could imply that groundwater extraction is a preferable alternative. This seems contradictory to the intensive efforts to replace groundwater for desalinated seawater. However, this stresses the point that ES are only one aspect to be taken in consideration for decision making. It can also be noted that improvements in the valuation methods are still required. As an example, the tourism figures do not represent the full potential touristic value, since this depends on the efforts that government and local businesses put into promoting the salares.

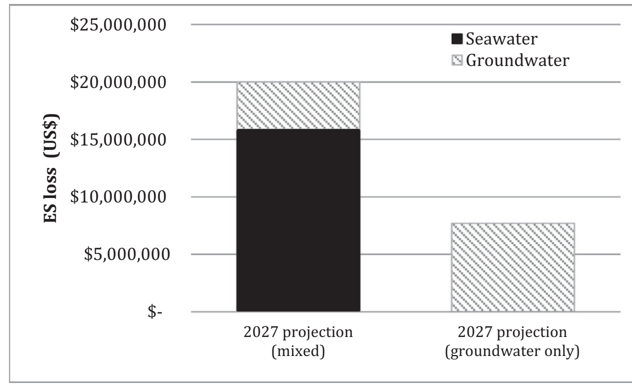

Fig. 3. Total losses of ES in US$ for 2027 projections.

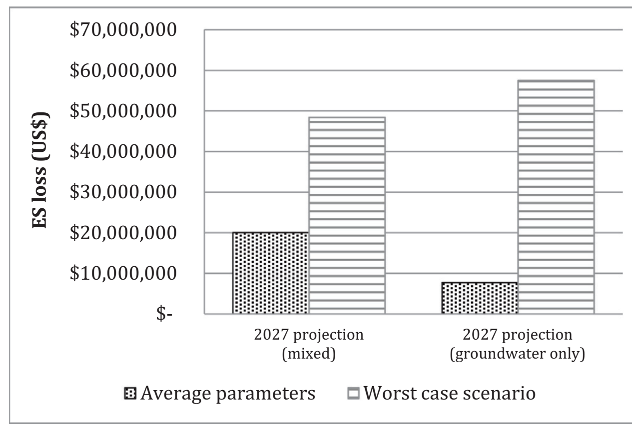

Fig. 4. Total losses of ES, average vs. worst case scenario.

4.2. Methodological advantages of the proposed framework

Enabling LCA to assess impacts on ES may offer more holistic results for decision makers. In the case study, the government and the industry in Chile have so far assessed ecosystem impacts from a site specific perspective, while LCA places the production systems in a wider context of interconnected social, economic and environmental processes. It also facilitates that full life cycles are taken in consideration and hidden trade-offs are recognized and quantified.

The proposed framework allows the consideration of non-land use drivers and the spatial variability therein by using models to determine the spatial quantities of ecosystems that are affected by each unit process. For stakeholders, this may also be a preferable way of presenting the model: it clearly states how much of an ecosystem type is transformed by each particular unit process. In this form, it allows a more open discussion on the accuracy of the assumptions and the adequacy of the selected spatial resolutions in LCA studies. It may also be more easily verifiable by process owners in the industries.

Another interesting outcome of the ecosystem flows approach is an improved interpretation of land use impacts. Conventional LCIA methods like CMLCA (Guinée, 2002) and ReCiPe (Goedkoop et al., 2009) deal with land use impacts by considering land use both as an inventory item (environmental flow) and as an impact. Hence the area transformed and occupied is also an impact indicator with units of m^2 or $m^2 \times y$. This is useful to understand the impacts of product systems on land competition, since land is a finite resource. However, additional information on ecosystem function can be obtained if we represent the land use change -in this case ecosystem transformation- as an inventory flow and express the impacts as changes in ES. A land use change in itself does not convey any specific information on damage to species or ecosystem functions, only a notion that "natural land" may be better for ecosystems than "agricultural land" or "urban land". The ReCiPe LCA methodology (Goedkoop et al., 2009) dealt with this by adding a second characterization step which translates midpoint impact indicators like land use and occupation to an endpoint impact indicator of Potential Disappeared Fraction of species. Yet this is only a limited indicator of ecosystem damage. Some LCA approaches have already proposed refining land use archetypes to more closely match ecosystem classifications (Koellner et al., 2013).

The framework also demonstrates that the socioeconomic aspect of ES can be coupled to LCA; by considering it a normalizing and weighing application, it clearly separates the modeling of the inventory and characterization steps from the valuation that underlies the socioeconomic dynamics and market mechanisms that determine prices. This approach allows the LCA framework to be easily coupled to the many economic valuation models that have been applied to ES. It also provides the framework with greater transparency and flexibility, since the impact results can be weighted in multiple ways (contingency valuation, market pricing, etc.), comparing the different results. Spatial and temporal variations in socioeconomic dynamics of ES demand and use can be dealt with by using e.g. average market data for selected time periods and export prices.

Finally, our framework presents certain advantages for modeling the effects of substance emissions on ES. For example, atmospheric emissions of substances like CO_2 can transform ecosystems by causing a temperature change. If these effects are modeled in the characterization step, then the model could potentially incur in double counting for unit processes that emit additional substances with their own CFs. This is evident in the case study, where the brine outflow is mixed with maintenance chemicals, both having an impact on the same area. Modeling the affected area of ecosystem in the inventory allows a more realistic representation of the areas that are really transformed by the industrial processes as a whole rather than aggregating the effects of each emission.

4.3. Extending the framework to background processes

For the framework to be consistent with the full life cycle perspective of LCA, comprehensive LCI databases of ES flows assigned to economic activities must be built. Direct field data may be collected (or available) for some high-impact and well known industrial processes in addition to physical modelling as required for the case study. The most obvious case for direct field data of ecosystem transformations is land use change, but other field data may exist for industries with directly measurable effluent dispersion plumes in water bodies or other types of natural resource extraction.

For typical production systems, background processes can amount to hundreds or thousands and this makes the task challenging. However, the efforts for each process would only be undertaken once, as LCI data can be reused once added to publicly or commercially available LCA databases. For each process, the ecosystem quantities (as environmental flows) would be recalculated by linear extrapolation, depending on the demand imposed by the functional unit on the product system. A practical way forward to implement this could be to apply a Pareto principle (or 80/20 rule) using existing data for conventional impact categories like ecotoxicity, eutrophication and acidification. A contribution analysis on these categories can highlight hotspots in a product system for which an ecosystem transformation will need to be taken into account. In the case study, for example, only 2 processes account for 87 percent of the marine aquatic ecotoxicity potential impact (see Supplementary Material, Section S6). These two relevant background processes are the generation of electricity from coal fired power plants and the disposal of the spoil from mining the coal. For these two processes, fate and transport models can be developed to estimate the affected coastal ecosystem areas. These are areas where toxicity thresholds will result in a significantly less productive ecosystem in analogy to the salinity gradient of the brine outflow.

The foreground system can therefore be expanded to incorporate the most relevant (formerly background) processes, in order to understand which ecosystems are particularly affected and how. In the case study, the thermal power generation happens in close proximity to the desalination facility, so the affected coastal ecosystem would be of the same type. The coal, however, is imported mostly from Colombia (60%) and the United States (30%), and this may imply transformations of different ecosystem types. In this way, we can significantly reduce the number of pathways and models required.

4.4. Limitations

Of the key challenges in LCA-ES modeling, our framework is still not able to account for feedbacks and interrelatedness in the impact pathways. That is, how the consumption/use of one ES may affect the production of others. Arbault et al. (2014) took an important step forward in this respect by applying an integrated earth system dynamic model and integrating the output over time. Their method, however, requires some important assumptions and data to model the global economy. This aspect of ES has also been an important challenge for other non-LCA models of ES (Rieb et al., 2017).

Data availability can also be an important limitation, particularly for some ES for which information is not available or is qualitative in nature. This is particularly challenging for cultural ES and supporting ES where models may result in double counting (Othoniel et al., 2016). Future efforts to integrate ES in LCA must follow closely how the ES discipline improves in the quantification and valuation of these services.

The issue of spatial variability is tightly linked to the resolution and/or scale at which ecosystem types are classified. In the proposed framework, this decision is made in the inventory stage. For our framework, this implies that spatially dependent parameters are used to determine the environmental flows, while each flow is considered to be representative for an ecosystem type. This differs from other spatialized approaches in LCA, e.g. (Verones et al., 2013), where the parameters are averaged for the receiving compartment in the characterization step.

Another important limitation is that in practice, our framework is only useful to capture large effects of interventions, i.e. those that can observably transform one ecosystem type into another. This limit is imposed by the resolution at which ecosystem types are defined, and also by the quantity of a flow in the case of substance emissions and natural resource use. In the case study, this limitation was overcome by assuming the effects of brine disposal and water extraction were large enough to result in barren ecosystems. But in many cases, substance emissions and natural resource use may result in smaller impacts on certain ecosystem processes, making it difficult to determine whether an absolute ecosystem transformation is an appropriate representation. In principle, this more subtle changes could be modelled by implementing additional characterization factors. However, a more practical way to deal with this could be to apply adjustments to the transformed areas or volumes instead, e.g. if 50% of an ecosystem productivity is affected, then the inventory considers 100% of an ecosystem half the size of the actually affected one. This makes sense in an LCA model because of its incremental nature. Here we do not further explore the implications of this possible approach.

Finally, we note that incorporating ES as impact categories in LCA may induce some overlap with existing indicators. Carbon sequestration, for example, may have an impact on the climate change category. Other authors have proposed biodiversity (as species richness), which may affect the production of ES and vice versa. Ultimately, this may require a reevaluation of endpoint indicators along the lines proposed by Dewulf et al. (2015).

5. Conclusions

Our framework provides an alternative way of using the key strengths of LCA to better understand the relationship between technological production systems and ecosystem services. This was evidenced in the case study, where the systematic LCA approach highlighted the fact that important tradeoffs do occur when shifting from groundwater to seawater sources. It was also possible to identify the entrainment of abalone larvae as one of the most sensitive aspects for ES, which makes a strong case for more comprehensive studies and monitoring of the seawater intake process.

Incorporating ES as an impact category in LCA allows quantitative comparisons of product's ecosystem impacts, which could better inform decision making and discussions between diverse stakeholders. While it introduces some new limitations vs. current state of the art, our approach simplifies the characterization step into one in which consensus is more easily achieved and data is more readily available from the ES field. In doing this, it may facilitate interpretation and communication of LCA results to broader audiences. In the same way, associating ecosystem transformations directly to unit processes in the LCI may be more easily verifiable by process owners. Finally, the framework considers ES impacts resulting not only from land use drivers, but other equally important drivers like resource use and water pollution for a more complete representation.

As the ES field advances in the assessment, quantification and valuation methods, our framework will also become strengthened. ES offer one perspective on the multidimensional question of sustainability, therefore efforts to integrate it with other environmental perspectives must continue. To achieve this, ES and LCA practitioners must continue to work towards consistent datasets and modeling structures.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.ecoser.2017.11.011.

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