Pass the salt please! From a review to a theoretical framework for integrating salinization impacts in food LCA

Sandra Payen[1,2,*], Claudine Basset-Mens[2], Stéphane Follain[3], Olivier Grünberger[4], Serge Marlet[5], Montserrat Núñez[6], Sylvain Perret[7]

[1] ADEME, 20 avenue du Grésillé, F-49004 Angers, France
[2] CIRAD, UPR Hortsys, Boulevard de la Lironde, F-34398 Montpellier, France
[3] Montpellier SupAgro, UMR LISAH, place Pierre Viala, F-34060 Montpellier, France
[4] IRD, UMR LISAH, place Pierre Viala, F-34060 Montpellier, France
[5] CIRAD, UMR G-Eau, INRGREF, Rue Hédi Elkarray, Tunis, Tunisia
[6] IRSTEA, UMR Itap, 361 Rue Jean François Breton, F-34196 Montpellier, France
[7] CIRAD, UMR G-Eau, F-34398 Montpellier, France
[*]Corresponding author. E-mail address: *sandra.payen@cirad.fr*

ABSTRACT

Food LCA should include salinization. Salinization is a threat not only to arable land but also to freshwater resources. Nevertheless, salinization impacts have been rarely and partially included in LCA so far. First, a comprehensive overview of salinization mechanisms is presented and highlights its multiple causes, which affect soil and water, and ultimately human health, ecosystems and resources. Second, adopting the ILCD analysis grid, we analyzed the scientific relevance and accuracy of existing published methods addressing salinization in LCA. Although interesting, these seminal approaches are often incomplete with regards to both the salinization pathways they cover and their geographical validity. Third, we analyzed how to consistently integrate salinization within the methodological frameworks for impacts modeling in LCA, and raised questions to address towards a comprehensive integration of salinization.

Keywords: Salinization, Life Cycle Assessment, Agriculture, Soil, Water.

1. Introduction

Salinization is a global and major environmental concern. Although we commonly think this issue is limited to arid and semi-arid regions, it appears that no climatic zone is free from salinization (Rengasamy 2006). Furthermore, both agricultural and non-agricultural areas, both irrigated and non-irrigated lands can be prone to salinization (Wood et al. 2000). The FAO estimates that 34 million hectares of irrigated land are salt-affected worldwide, and an additional 60-80 million hectares are affected by waterlogging and related salinity (FAO 2011). Although reports of secondary salinization, i.e salinization due to human beings, continue to appear in the literature, there is a lack of recent assessment of the level of anthropogenic salinization (Flowers 1999). Salinization is a threat not only to arable land (EEA, 1997) but also to water resources (freshwater lakes and wetlands, rivers and streams), particularly in the arid and semi-arid regions of the world (Williams 1999). Salinity becomes a major issue in global agriculture when it adversely affects crop production (Rengasamy 2010).

As a global and multicriteria environmental assessment tool, Life Cycle Assessment should account for one of the major threats of food production. Research on including salinization in LCA is of high priority (JRC-IES 2011). Evaluating salinization impacts is particularly relevant in food LCA, because agricultural systems are both the main affected targets and causes of salinization. Sustainable irrigated agricultural production is seriously threatened by salinization (Aragüés et al. 2011).

So far, a few methods evaluated salinization impacts in the LCA framework: Amores et al. (2013); Feitz and Lundie (2002); Leske and Buckley (2003; 2004a; 2004b). However, these methods were rarely or never applied and focus on one salinization type or on a specific location. A comprehensive approach to salinization in the LCA framework is lacking.

The goals of this work are (1) to provide a comprehensive and structured overview of secondary salinization mechanisms and cause-effect chains, (2) to review LCIA methods modeling salinization impacts and (3) to identify methodological issues to build a consistent framework for salinization impacts in LCA.

2. Salinization: environmental mechanisms, cause and effect chains

2.1. A definition of salinization(s)

Salinization is the accumulation of salts, not exclusively NaCl as it is frequently assumed, but also many other types of salts such as carbonates, sulphates and other salts (calcium, magnesium and potassium) (Rengasamy 2010). Salinity is described as Total Dissolved Salts (TDS) (Leske and Buckley 2003). Although common ions are classified as non-toxic inorganic constituents, they may cause toxic effects at high concentration. Soil and water salinization are often studied separately: "*Salinization is the process that increases the salinity of inland waters*" (Williams 1999). "*Salinization is an accumulation in the soil of dissolved salts*" (Wood et al. 2000). But soil and water salinization are inter-related, water being the main vector of salts. Salts are conservative and resistant to degradation (Schnoor 2013) but they are mobile: they can either stay in a soil at a given location or they can migrate with water.

Soil salinization includes also sodization and alcalinization, while salt-affected soils include saline soils, saline–sodic soils and sodic soils (Ghassemi et al. 1995). Salinization is an accumulation of salts, and sodisation is an accumulation of sodium on the soil exchange complex causing soil clays dispersion, thus altering soil structure. Saline conditions are usually characterized by measuring the Electrical Conductivity (EC) and the Sodium Adsorption Ration (SAR) in soil extracts (Rengasamy 2010). EC measures the ability of water to conduct electrical current and is correlated with the total dissolved solids in soil water (Corwin and Lesch 2005) and thus salinity. SAR is the ratio of sodium ion on calcium and magnesium ions and is correlated with sodicity. Both salinity and sodicity are associated with more alkaline (basic) soils (Wood et al. 2000). Several salt-affected soil classifications exists (e.g. Rengasamy (2010)), and it is important to note that SAR and EC thresholds values are not the same in different soil classification systems (Rengasamy 2006).

Water quality in relation to salt abundance can be classified in 4 classes: fresh, brackish, saline and eventually brine water. On an operational point of view, farmers usually classify irrigation water according to EC and SAR measurements because it represents an easy way to estimate the salinization and alcalinization hazard of soil associated with its use (Richards 1954).

2.2. Causes of salinization in space and time

This paper does not consider natural salinization and focuses on anthropogenic salinization because LCA addresses impacts of human interventions. Salinization results from various causes which are often inter-related (Williams 2001). Salinization involves physico-chemical mechanisms at local scale and hydrological mechanisms at regional (catchment) scales. We distinguish for analysis between salinization associated with land use change and salinization associated with irrigation.

2.2.1. Salinization associated with land use change

Land Use Change (LUC) modifies hydrological processes and then water cycle at the catchment level. In particular, clearance of deep-rooted perennial native vegetation with high transpiration rates, and replacement with shallow-rooted crops with lower transpiration rates, will increase the water infiltration rate and mobilize salts stored in soil. Thus, the underlying groundwater tables can rise and reach the near soil surface in lowland, leading to temporary surface waterlogging and then deposition of salts through capillary action after evaporation (Williams 1999). In addition, the infiltrated salts increase the salinity of the aquifer. Many examples have been documented in Australia (Scanlon et al. 2007; Williams, 1999). Classical factors, as topography, acts on soil and water salinization at catchment scale. Specificities associated to LUC are directly linked to modifications of the water balance at the landscape scale: a modification of amounts of precipitations, a modification of groundwater table level, a modification of evapotranspiration rates, a modification of soil geochemical and hydrodynamic profiles, and salt stock variation in soil.

2.2.2. Salinization associated with irrigation

Salinization associated with deposition of ions - Irrigation is "*the salt concentration and mobilization machine*" (Smedema and Shiati 2002). The development of irrigation influences the local geohydrological regime, mobilizes salts stored in the underlying substrate, and contributes to the concentration of salts in land and water resources (Smedema and Shiati 2002). Irrigation water always contains some salts that may eventually accumulate in the soil, unless the irrigation management allows the salt leaching (Flowers 1999). However, the leached salts from the root zone may induce salinization of water bodies, such as the underlying groundwater, if drainage is not carefully managed (Mateo-sagasta and Burke 2010). Salts have a higher tendency to accumulate in semi-arid and arid areas because of the conjunction of low rainfall and high evapotranspiration rates (Marlet and Job 2006). The combined effect of the withdrawal of fresh irrigation water from a water body and the return of saline drainage water to this water body leads to salinity increase (Smedema and Shiati 2002). Furthermore, changing climatic conditions are worsening salinization with the use of low quality water to compensate the increased scarcity of freshwater (Duan and Fedler 2013). Irrigation's controls on salinization are those associated to local dependencies: salinization associated with deposition of ions through irrigation. Salts are also derived from fertilizers (Scanlon et al. 2007). Embedded key parameters are salt content in irrigation water, fertilizer use intensity, precipitation levels and irrigation doses, evapotranspiration rates, soil hydrodynamic profile, and salts reservoirs in the soil.

Salinization associated with shallow groundwater table or poor drainage - When an area is irrigated, the hydrologic balance is modified, and is often causing groundwater table rising. This rise may be noticeable after only a few years (Smedema 1993) cited by (Marlet and Job 2006). Areas having shallow water tables usually have soil salinity issues (Corwin and Lesch 2005). In this case, good drainage is required to avoid waterlogging and salts deposition through capillary action. More generally, poor irrigation management and inadequate drainage often lead to salinization and waterlogging (Wood et al. 2000). When the groundwater rising is saline, it may in turn induce salinization of some fresh waters (Williams 2001). Nested scales are involved in this salinization context: if spatial extend and structure of groundwater tables are regional, associated salinization processes are expected to act at a local scale; the key parameters are water table depth, drainage rates, salts content in irrigation water, precipitation levels, evapotranspiration rates, irrigation doses, and salts reservoirs in the soil.

Salinization associated with overuse of a water body: saline intrusion - In many coastal regions, excessive withdrawal of groundwater leads to marine intrusion: the decreasing aquifer table level and the proximity of seawater induces sea-water inflow in the aquifer. Some aquifers are already permanently salinized (FAO 2011; Flowers 1999; Scanlon et al. 2007). Deltas are also prone to marine intrusion when the freshwater flow of the river is reduced because of excessive water withdrawal upstream or the construction of impoundments (FAO 2011; Williams 2001). Sea-level rise induced by climate change is an aggravating factor of marine intrusion (FAO 2011). In non-coastal areas, saline intrusion may result from saline water transfer from a saline aquifer to an overused aquifer. Salinization associated with saline intrusion involves mechanisms at catchment scale; the key parameters are volume of freshwater withdrawal, distance to the coast, water body (river or aquifer) exploitation rate and presence of saline aquifer.

2.2.3. Salinization causes are often inter-related

Although we can establish a typology of salinization contexts, in many cases the situation is complex because salinization results from several causes. Many freshwater lakes, wetlands and rivers become saline because of the replacement of natural vegetation by agricultural crops upstream, together with the discharge of saline agricultural wastewater (Williams 2001). Groundwater salinization can be due to both sea-water intrusion and the agricultural return flows (Bouchaou et al. 2008). In addition, water and soil salinization are intimately related. The degradation of freshwater sources (surface or groundwater) has concomitant effects on the systems using these sources, and soil salinity affects in return water resources (D'Odorico et al. 2013).

2.3. Water and soil salinization damages to ecosystems, human health and resources

Salt-affected soils and waters have many effects and ultimately damage the areas of protection (AoP) as defined in LCA: ecosystems (or natural environment), human health and natural resources. Figure 1 depicts the salinization cause and effect chains, emphasizing the inter-relation between the multiple causes.

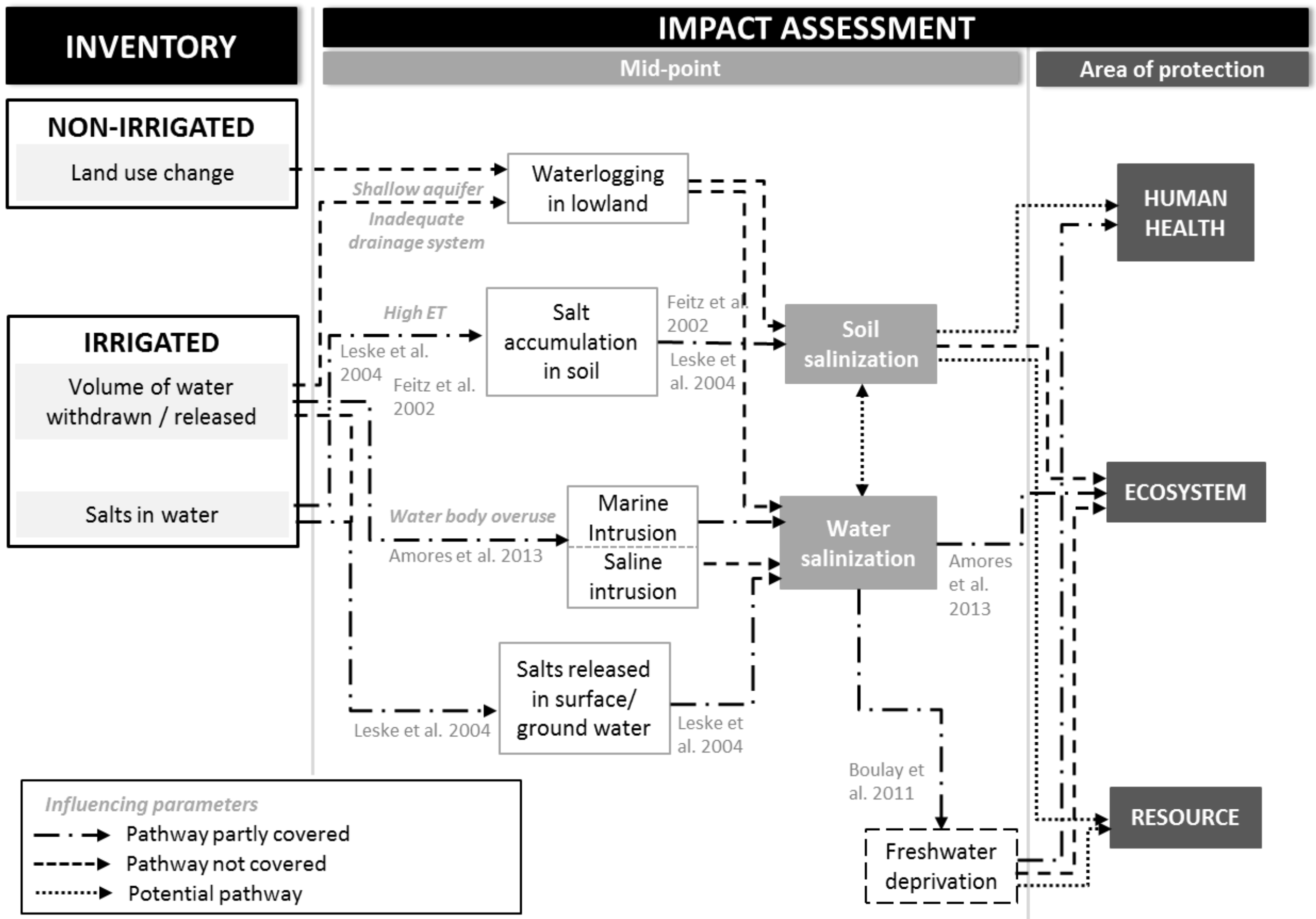

Figure 1. Human-driven salinization cause and effect chains and positioning of approaches proposed in the literature

Soil salinization not only reduces vegetation or crop growth, but also degrades land more or less permanently (D'Odorico et al. 2013). Salt-affected soils have a lower productivity through three potential effects on plants: i) reduction of plant water uptake by lowering the osmotic potential, ii) toxic effect by different ions depending on the pH, and iii) plants nutrients uptake imbalance (Flowers and Flowers 2005). Sodic soils also have indirect effects due to soil structure degradation, permeability reduction and possible waterlogging (D'Odorico et al. 2013; Rengasamy 2010; Suarez et al. 2006). Thus, by reducing crop yield, salinization of crop land could result in malnutrition for poor populations who rely on this production to feed themselves. Soil salinization is also considered as a driver of desertification because it affects terrestrial ecosystems and is closely related to land degradation processes such as soil erosion (D'Odorico et al. 2013). Land degradation might be considered as a damage to soil resource. It should be noticed that ecosystems may not lose diversity per se but simply replace the halosensitive biota with a halotolerant one (Williams 1999). Measures to reduce soil salinity and sodicity exist, but salinization can be irreversible: in arid regions where there is not enough freshwater available to leach the accumulated salts (Rozema and Flowers 2008), or in lowland areas of endorheic basins with shallow and saline groundwater (D'Odorico et al. 2013).

Salinization of a water-body not only affects the aquatic and riparian ecosystem, but also reduces the water availability for further use. An increase of water salinity causes change in species composition of algae, zooplankton, and benthic communities and leads to the disappearance of macrophytes and riparian trees (Schnoor 2013; Williams 1999). In addition, saline freshwater lakes, wetlands, rivers or aquifers are unfit to serve as supplies for domestic, agricultural and other uses (FAO 2011; Williams 1999), thus resulting in water deprivation for humans and ecosystems. This water resource quality alteration may be irreversible, for example for a permanently saline aquifer, and thus affects the water resource for future generations.

3. Review of salinization impact assessment methods in LCA

3.1. Overview of the methods

Three methods have been developed assessing salinization impacts in the LCA framework so far (Table 1). These approaches are either mid-point oriented (Feitz and Lundie 2002), end-point oriented (Amores et al. 2013), or near-endpoint oriented (Leske and Buckley 2003; 2004a; 2004b).

Mid-point oriented approaches - The soil salinization potential developed by Feitz and Lundie (2002) assesses the propensity of irrigation water to damage soil structure and the accumulation of sodium in the soil. The volume of irrigation water and sodium concentration are multiplied by a soil sodisation hazard characterization factor (CF). The soil sodisation hazard is assessed through the ratio between the EC threshold representing the limit of soil structure integrity for a given SAR, and the EC of the irrigation water. The SAR calculation is based on the irrigation water composition.

Table 1. Salinization impacts assessment methods overview

Publication	**Feitz and Lundie 2002**	**Leske and Buckley 2003;2004a; 2004b**	**Amores et al. 2013**
Inventory requirement	Irrigation water volume (L), [Na] (mg/L).	kg TDS released	Evapotranspiration of the crop: ET_{crop} (m^3)
Characterization factor (CF)	$CF=EC_{threshold}/EC_{iw}$ With: $EC_{threshold}=0.121xSAR+0.033$ linear equation representing the clay flocculation - dispersion threshold. SAR calculation requires: [Ca], [Mg], [SO4], [$CaCO_3$], pH, EC of irrigation water. If CF<1, no soil degradation hazard from sodisation.	Total salinity potential (TSP) for salt release in a given compartment: TSP= Σ potential effects on environmental target. With: potential effect $=\Sigma_i^N PEC_i-PEC_i^0 / PNEC.M$ PEC_i=predicted concentration in the compartment during day i after an emission of total mass M; PEC_i^0=predicted concentration in the compartment during day i without an emission; PNEC=predicted no-effect concentration; N=number of days in the simulation	Change in PAF of species due to a change in groundwater consumption: - Fate Factor $=\Delta FGW/\Delta ET_{crop} \times \Delta C_N.V_N/\Delta FGW$ With: FGW=fresh groundwater inflow to Lagoon, C=salinity and V=volume of the lagoon, ET_{crop}=crop ET, Δ symbolizes the change between years. - Effect Factor $=\Delta PAF_{sal}/\Delta C_N.V_N$ $=0.5/HC50_{sal}$ With: $HC50_{sal}$=concentration at which ≥50% of the species are exposed to concentrations above their EC50
Indicator	$\sum_i CFi.[Na]i.Vi$ For irrigation water i. **Unit:** kg $Na+_{eq}$	TDS released x TSP **Unit:** kg TDS $_{eq}$	ET_{crop}. CF **Unit:** PAF.m^3.year, converted into species.year considering a 7.89×10^{-10} species.m^{-3} freshwater species density

Near-endpoint oriented approaches - Leske and Buckley (2003; 2004a; 2004b) developed a salinity impact category for South African LCA. They provide salinity potential CFs for salts release in atmosphere, surface water, natural surfaces, and agricultural surfaces compartments, and account for potential effects on aquatic ecotoxicity, materials, natural wildlife, livestock, aesthetic, natural vegetation and crop. Inspired by a risk assessment approach, salts fate factors are calculated with an atmospheric and hydrosalinity catchment model, thus assessing the predicted salt concentration in the different compartments. Effect factors are calculated using the predicted no-effect concentration for each environmental target.

End-point oriented approaches - Amores and colleagues (2013) evaluated the impacts on biodiversity associated with a salinity increase in a Spanish coastal wetland caused by the use of groundwater for agriculture. The water consumed by the crop (withdrawn from the aquifer) is multiplied by fate and effect factors. The fate factor represents the freshwater that lacks in the wetland system due to withdrawal, leading to increased concentration of salt due to increased sea water infiltration into the wetland. It is calculated from seasonal water and salts balances for the wetland Albufera de Adra. The effect factor is obtained from the fitted curve of the potentially affected fraction of native wetland species due to salinity. It is focused on plants, fishes, algae, and a crustacean.

Apart from the applicability test done by the authors themselves (Amores et al. 2013; Feitz and Lundie 2002), literature provides only one case study implementing Feitz and Lundies's method. Muñoz and colleagues (2010) calculated soil salinization potential as one indicator for soil quality impacts (besides soil organic carbon deficit), to compare the impacts of systems using treated wastewater, groundwater, or desalinated water for irrigation.

3.2. Critical analysis of the methods

We analyzed the methods against criteria of the ILCD Handbook procedure for methods analysis (JRC-IES 2011): completeness of scope, environmental relevance, scientific robustness and certainty, documentation, transparency and reproducibility, applicability and potential stakeholder acceptance.

Completeness of scope - Feitz and Lundie (2002) and Amores et al. (2013), focus on one single salinization pathway: soil sodisation and salinization from poor irrigation practices and seawater intrusion in a groundwater-fed wetland, respectively. Leske and Buckley (2003; 2004a; 2004b) cover several pathways of water and soil salinization induced by salts release. The methods are providing good methodological approaches to salinization impact modeling but with a limited geographical validity. Moreover, the methods cover only a part of salinization mechanisms within the pathway they are modeling. Figure 1 positions the contributions of these approaches on the global salinization cause and effects chains.

Feitz and Lundie (2002) do not account for waterlogging. Leske and Buckley (2003, 2004a, 2004b) do not consider salinization induced by a LUC or a saline intrusion, and Amores et al. (2013) do not consider groundwater salinization. All methods have site-specific CFs, emphasizing that salinization impacts are highly site-dependent; especially regarding the hydrology, the climate and irrigation water quality. However, their geographical extend validity is limited. The soil salinization impact of Feitz and Lundie (2002) depends on the validity domain of the electrolyte threshold curve which «*may not be appropriate for some soils*», and the estimation of the Sodium Adsorption Ratio of the soil drainage water is assumed for an Australian red-brown earth. Leske and Buckley (2003; 2004a; 2004b) fate factor is calculated with a South African catchment atmospheric deposition-hydrosalinity model, and the effect factor is based on the South African Water Quality Guidelines. Amores et al. (2013) fate factor is based on water and salts balance relying on the specific hydrologic functioning of the wetland and local hydro-climatic parameters, and the effect factor is based on specific native species of the Albufera de Adra wetland. It would be time-consuming and data-intensive for one who wants to apply the methods in other contexts.

Environmental relevance - Feitz and Lundie (2002) indicator is based on a relatively ancient approach but very common and generally well accepted. However, the soil type is not accounted for, although soil texture is a key parameter in the sodicity sensitivity. Indeed, sandy soils do not have soil structural problems caused by high SAR, whilst clayey soils are likely to be sodic with soil structural problems (Rengasamy 2010). Besides, the quality of the soil solution is buffered by slow physico-chemical mechanisms occurring over several years (Condom et al. 1999). Leske and Buckley (2003; 2004a; 2004b) fate model predicts environmental concentrations in all the compartments relevant to the calculation of salinity potentials. The land use distribution of the model is simplified through one single urban area, one single rural area and one single rural agricultural area. The calculated characterization factors for salts emissions onto the agricultural surface by far outweigh the CFs for releases into other compartments. This warrants further research to better model agricultural systems. The salts effect factors are based on the Predicted No-Effect Concentrations, a conservative approach compared with the HC50 (50% hazardous concentration), assuming that sensitivity of an ecosystem depends on the most sensitive species. Effect factors are not calculated as a function of the background salt concentration, except for aquatic ecotoxicity. Amores et al. (2013) fate and effect factor calculation are simplifying the salinization mechanisms. The effect factor is linear: calculated as the average gradient at the HC50 but does not account for background concentration. The Species Sensitivity Distribution are not based on EC50s describing the same effects (e.g., survival or growth inhibition). The fate factor is not utilizing any model but is based on simple water and salt balance equations.

Scientific robustness and certainty - These methods were published in peer-reviewed journals. Model uncertainty is not provided by Feitz and Lundie (2002), but is provided through a sensitivity analysis of the fate model by Leske and Buckley (2003, 2004 a&b), and through confidence intervals and CF standard error by Amores et al. (2013).

Documentation, transparency and reproducibility - Model documentation, characterization model and results published are available for the three methods. Reproducible for LCA studies located in areas where CFs are available.

Applicability - CFs are not straightforward to apply for the three methods. CFs have to be calculated by the practitioner because they are either irrigation-water dependent (Feitz and Lundie, 2002), or require re-developing the whole modeling approach because of its narrow geographical validity. The indicator units Na^{+}_{eq} and TDS cannot be compared with other methods as they are salinity-specific units. In contrast, the common end-point unit PAF from Amores et al. (2013) can be compared with other methods.

Potential stakeholder acceptance - The acceptance among LCA practitioners is limited with no or a single application (Leske and Buckley, 2003; Feitz and Lundies, 2002). Amores et al. (2013) method is young and remains to be implemented for validation in other contexts; further developments are required to ascertain if CF with global coverage can be calculated.

4. Perspectives toward a consistent framework for salinization impacts assessment in LCA

The purpose in this section is to analyze how salinization impacts should be modeled within the methodological framework of LCA. Answering this question raises several methodological issues: where to set the boundary between technosphere and ecosphere? What is the status of the AoP resource? A holistic approach is adopted to identify the methodological questions and risks of double counting to be addressed for future developments.

4.1. Methodological framework of LCA

We can distinguish two environmental impact modeling types in LCA: the first in relation to impacts due to the emission of a substance and the second in relation to the consumption of a resource. If tracing a substance in the environment has been historically addressed in LCA through the modeling of his fate and effect on the receiving environment, the modeling of impacts due to biotic resource consumption, especially for water and land resources, has only been studied recently (Finnveden et al. 2009).

Water and land as resources are inventory flows, which may be consumed or altered by the human activity under study. Modeling the damages caused by a resource use is complex because its function depends on the quality required by the user. Regarding water, if impacts from pollutant emissions into water are accounted for in LCA, impacts from water unavailability are not yet fully quantified (Boulay et al. 2011b). Boulay and colleagues (2011a; 2011b) developed a method to fill this gap: a functionality-based water impact assessment considering that water quality degradation can lead to water deprivation if not suitable anymore for specific users. In this method, the water quality is related to a functionality; assessing to which users the water is functional. This method assesses the impacts from water deprivation on the AoP human health. The same approach could be adopted to assess impacts of water deprivation on the AoP ecosystems, but the water categories should be based on ecosystems uses instead of human uses. Regarding the impact of water deprivation on the AoP resource, the framework proposed by the UNEP-SETAC “Water Use in LCA” (WULCA) working group considers that only fossil water use or renewable water overuse can affect the resources, thus water quality alteration is not addressed (Kounina et al. 2013). Yet, a permanently degraded freshwater represents a loss of water resource for future generation. There is no risk of double counting if we clearly define the AoP resource as the protection of a resource (in sufficient quality and quantity) for future generation, while the AoP human health and ecosystems reflect the protection of current people and ecosystems. The fact that the new IMPACT World^{+TM} LCIA methodology is considering an AoP "resource and ecosystems services" raises the question of the status of this AoP.

Although it has sometimes been considered as an impact indicator, land use is an inventory flow that leads to a group of impact categories, directly affecting ecosystems and resources, and indirectly affecting human health (Finnveden et al. 2009; Koellner and Geyer 2013a). The open research question is which midpoint categories should be defined to support an integrated impact assessment accounting for land use impacts alongside those from chemical emissions, water use and climate change (Koellner and Geyer 2013b). The UNEP-SETAC Life Cycle Initiative project LULCIA recommends for each impact pathway the determination of the optimal resolution of land cover types and regionalization, and the development of CFs for all relevant combinations of land use type and location (Koellner et al. 2013a). Several pathways were recently developed, within a structure of the LCIA in accordance with the typology of ecosystem services of the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005). Indeed, the framework distinguishes between two intermediate end-points: biodiversity damage potential and ecosystem services damage potential. Brandão and Milà i Canals (2012) characterized land use impacts on biotic production potential (representing an ecosystem services damage potential) using deficit of Soil Organic Carbon (SOC) as an indicator. They developed CFs for eight land use types at the climate region level. Saad et al. (2013) address land use impact on three ecosystem services: erosion regulation potential, freshwater regulation potential, and water purification potential. They calculated CFs per land use type (seven) specific to each biogeographic region (tested at different regionalization scales), using LANCA® model which assesses the influence of different land use activities on soil ecological functions. Developing such CFs requires the availability of land-use-specific and biogeographically differentiated data on the indicator selected: SOC for Brandão and Milà i Canals (2013) or soil properties, landscape and climatic conditions for Saad et al. (2013).

4.2. Perspectives for integrating salinization in LCA

Regionalization - The fate of salts depends on climate, agricultural practices, soil, and hydrological context, and their effects depend on the sensitivity of the target (species, capacity to desalinate water). Therefore a regionalized approach is paramount. The methods developed by Leske and Buckley and Amores and colleagues are regionalized, but with a very limited coverage. Regionalized impacts in LCA are supported by geographic information systems (e.g. (Boulay et al. 2011b; Núñez et al. 2012; Núñez et al. 2009; Saad et al. 2013).

Midpoint - For water and soil salinization, two pathways modeling can be differentiated: on the one hand salinization induced by a LUC, which inventory flow is a land use transformation, and on the other hand irrigation-driven salinization, which inventory relies on a volume of water withdrawn (for saline intrusion) or released (waterlogging) and salts contents in water (salts released in soil and water bodies) (Cf. Figure 1). While LUC and saline intrusion result from a modified hydrologic (and thus salt) balance at regional or catchment scale, direct salts emissions through irrigation water occur at local scale. The system limit definition, and thus the consistency between LCI and LCIA modeling, is a crucial issue because salinization mechanisms occur at different spatial and temporal scales. In particular, the soil status is critical because it is both an environmental target and a part of the agricultural system. If soil is excluded from the technosphere, a salt emission is the salts content in irrigation water, while if soil is partially included in the system, a salt emission depends on soil conditions and practices. If soil is totally included in the system, no soil salinization would be accounted for and salt emissions would occur in the aquifer or river. Setting the system limit will determine which parameter has to be accounted for in the inventory or to be part of the CF.

We analyze in the following the modeling options for each AoP.

Ecosystems - Salinization affecting ecosystems could be modeled through aquatic ecotoxicity assessment for water salinization. Although salts are not classified as toxic substances, salts have toxic effects at high concentration. Amores et al. 2013 adopted this type of modeling in a specific context. Salts are not yet modeled in the USEtox model (Rosenbaum et al. 2008), but future developments of this tool could include freshwater ecotoxicity CFs for salts. Another future improvement of USEtox model is to develop regional versions because no spatial differentiation of location of the emission was considered so far (Henderson et al. 2011). Water salinization may also damage ecosystems through reduced water availability: a functionality-based approach could be adopted for this pathway. Similarly to water salinization, soil salinization affecting ecosystems could be modeled through terrestrial ecotoxicity assessment. The development of terrestrial ecotoxicology CFs is part of future developments of the USEtox model (Henderson et al. 2011).

Human health - The modeling of water salinization affecting human health could be approached by Boulay and colleagues framework. Total Dissolved Solids, Bicarbonate, Chloride, Chlorides/nitrites, Sodium and Sulfate in water are already parameters accounted for in this method to define the water categories for users (Boulay et al. 2011a). However, Boulay et al. (2011) method cannot be applied in its present form due to a scale modeling issue: the water salinized should be an inventory flow which is the result of a balance between water input and water output (with associated salinity increase). Boulay's method can only be applied in the case of salinization of drainage water induced by irrigation, and if the soil is included in the technosphere; i.e the saline drained water is considered an "emission". Soil salinization affects human health through reduced food supply. This is linked with the ecotoxicological effects of salts on crops, but also to soil physical degradation. Feitz et al. (2002) method is the mid-point step contributing to this pathway.

Resource - Water and soil salinization affecting water and soil resources are debatable pathways. In the case of permanently saline aquifers, we can consider that future generations will be deprived of water in that specific location. But as noticed previously, water quality alteration affecting resource is not considered in the water use impact framework. Regarding soil salinization, permanent degradation of soil or a loss of soil through erosion reduces soil availability as a future resource (Núñez et al. 2012). The UNEP SETAC Land use working group considers that a high salinity area in a very dry climate could be barren for an indefinite time period and corresponds to a permanent impact (Koellner et al. 2013b).

The land use impacts framework could cover soil salinization pathways, but within this framework, the inventory flux is a land use type, and not a salt release. According to JRC-IES 2011 "*soil salinization may be included in a completed/revised land use category"*. Soil salinization and waterlogging could be included in the global land use impact assessment on biodiversity and ecosystem services. Salinization, together with other mid-points, has an impact on the potential for biotic production (Brandão and Milà i Canals 2012). Also, salinization may impact freshwater regulation, erosion regulation and water purification, but this soil parameter is not accounted for in the method developed by Saad et al. (2013). To include salinization impact potential, a refinement of the land use types would be required to account for soil and agricultural practices and their effects on salinization. However, it is important to notice that accounting for salts emission through the land use framework reduces the frontier with the salt emission modeling framework, thus increasing double counting risks. Moreover, it is important to notice the link between land use and water use categories, especially for irrigated agriculture. Indeed, irrigation is part of the land use practices, and LUC can lead to changes in the water cycle (Koellner et al. 2013c).

5. Conclusion

Including salinization impacts in LCA is of high priority, especially for agricultural systems. Although the existing methods addressing salinization in LCA are important and relevant contributions, they are incomplete in terms of spatial and environmental mechanisms coverage. The modeling complexities lie in the inter-relations between salinization mechanisms, at both local and regional scales, and the status of soil and water in LCA which are both resources and living environments. An analysis of the modeling options in agreement with the LCA framework has been proposed in this paper. Much research effort is still required to include salinization impacts in a global, consistent and operational manner in LCA.

6. References

Amores MJ, Verones F, Raptis C, et al. (2013) Biodiversity impacts from salinity increase in a coastal wetland. Environ Sci Technol 47:6384–92. doi: 10.1021/es3045423

Aragüés R, Urdanoz V, Çetin M, et al. (2011) Soil salinity related to physical soil characteristics and irrigation management in four Mediterranean irrigation districts. Agric Water Manag 98:959–966. doi: 10.1016/j.agwat.2011.01.004

Bouchaou L, Michelot JL, Vengosh a., et al. (2008) Application of multiple isotopic and geochemical tracers for investigation of recharge, salinization, and residence time of water in the Souss–Massa aquifer, southwest of Morocco. J Hydrol 352:267–287. doi: 10.1016/j.jhydrol.2008.01.022

Boulay A-M, Bouchard C, Bulle C, et al. (2011a) Categorizing water for LCA inventory. Int J Life Cycle Assess 16:639–651. doi: 10.1007/s11367-011-0300-z

Boulay A-M, Bulle C, Bayart J-B, et al. (2011b) Regional characterization of freshwater Use in LCA: modeling direct impacts on human health. Environ Sci Technol 45:8948–57. doi: 10.1021/es1030883

Brandão M, Milà i Canals L (2012) Global characterisation factors to assess land use impacts on biotic production. Int J Life Cycle Assess 18:1243–1252. doi: 10.1007/s11367-012-0381-3

Brandão M, Milà i Canals L (2013) Global characterisation factors to assess land use impacts on biotic production. Int J Life Cycle Assess 18:1243–1252. doi: 10.1007/s11367-012-0381-3

Condom N, Kuper M, Marlet S, et al. (1999) Salinization, alkalinization and sodification in Punjab (Pakistan): characterization of the geochemical and physical processes of degradation. L Degrad Dev 10:123–140. doi: 10.1002/(SICI)1099-145X(199903/04)10:2<123::AID-LDR321>3.0.CO;2-V

Corwin DL, Lesch SM (2005) Apparent soil electrical conductivity measurements in agriculture. Comput Electron Agric 46:11–43. doi: 10.1016/j.compag.2004.10.005

D'Odorico P, Bhattachan A, Davis KF, et al. (2013) Global desertification: Drivers and feedbacks. Adv Water Resour 51:326–344. doi: 10.1016/j.advwatres.2012.01.013

Duan R, Fedler CB (2013) Salt management for sustainable degraded water land application under changing climatic conditions. Environ Sci Technol 47:10113–10114. doi: 10.1021/es403619m

FAO (2011) The state of the world's land and water resources for food and agriculture (SOLAW) - Managing systems at risk. London

Feitz AJ, Lundie S (2002) Soil Salinisation: A Local Life Cycle Assessment Impact Category. Int J Life Cycle Assesment 7:244–249.

Finnveden G, Hauschild MZ, Ekvall T, et al. (2009) Recent developments in Life Cycle Assessment. J Environ Manage 91:1–21. doi: 10.1016/j.jenvman.2009.06.018

Flowers TJ (1999) Salinisation and horticultural production. Sci Hortic (Amsterdam) 78:1–4.

Flowers TJ, Flowers S a. (2005) Why does salinity pose such a difficult problem for plant breeders? Agric Water Manag 78:15–24. doi: 10.1016/j.agwat.2005.04.015

Henderson AD, Hauschild MZ, Meent D, et al. (2011) USEtox fate and ecotoxicity factors for comparative assessment of toxic emissions in life cycle analysis: sensitivity to key chemical properties. Int J Life Cycle Assess 16:701–709. doi: 10.1007/s11367-011-0294-6

JRC-IES (2011) International Reference Life Cycle Data System (ILCD) Handbook - Recommendations for Life Cycle Impact Assessment in the European context. doi: 10.278/33030

Koellner T, Baan L, Beck T, et al. (2013a) Principles for life cycle inventories of land use on a global scale. Int J Life Cycle Assess 18:1203–1215. doi: 10.1007/s11367-012-0392-0

Koellner T, Baan L, Beck T, et al. (2013b) UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. Int J Life Cycle Assess 18:1188–1202. doi: 10.1007/s11367-013-0579-z

Koellner T, Baan L, Beck T, et al. (2013c) UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. Int J Life Cycle Assess 18:1188–1202. doi: 10.1007/s11367-013-0579-z

Koellner T, Geyer R (2013a) Global land use impact assessment on biodiversity and ecosystem services in LCA. Int J Life Cycle Assess 18:1185–1187. doi: 10.1007/s11367-013-0580-6

Koellner T, Geyer R (2013b) Global land use impact assessment on biodiversity and ecosystem services in LCA. Int J Life Cycle Assess 18:1185–1187. doi: 10.1007/s11367-013-0580-6

Kounina A, Margni M, Bayart J-B, et al. (2013) Review of methods addressing freshwater use in life cycle inventory and impact assessment. Int J Life Cycle Assess 18:707–721. doi: 10.1007/s11367-012-0519-3

Leske T, Buckley C (2004a) Towards the development of a salinity impact category for South African life cycle assessments : Part 2 – A conceptual multimedia environmental fate and effect model. 30:241–252.

Leske T, Buckley C (2004b) Towards the development of a salinity impact category for South African life cycle assessments : Part 3 – Salinity potentials. 30:

Leske T, Buckley C (2003) Towards the development of a salinity impact category for South African environmental life-cycle assessments : Part 1 – A new impact category. 29:289–296.

Marlet S, Job J-O (2006) Processus et gestion de la salinité des sols. In: Tiercelin J-R, Vidal A (eds) Trait. d'irrigation, Second. Ed., Ted & Doc . p 1266

Mateo-sagasta J, Burke J (2010) SOLAW Background Thematic Report - TR08 Agriculture and water quality interactions: a global overview. Rome

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: synthesis, Island Pre. Washington, DC

Muñoz I, Milà-i-Canals L, Fernández-Alba AR (2010) Life Cycle Assessment of Water Supply Plans in Mediterranean Spain. J Ind Ecol 14:902–918. doi: 10.1111/j.1530-9290.2010.00271.x

Núñez M, Antón A, Muñoz P, Rieradevall J (2012) Inclusion of soil erosion impacts in life cycle assessment on a global scale: application to energy crops in Spain. Int J Life Cycle Assess 18:755–767. doi: 10.1007/s11367-012-0525-5

Núñez M, Civit B, Muñoz P, et al. (2009) Assessing potential desertification environmental impact in life cycle assessment. Int J Life Cycle Assess 15:67–78. doi: 10.1007/s11367-009-0126-0

Rengasamy P (2006) World salinization with emphasis on Australia. J Exp Bot 57:1017–23. doi: 10.1093/jxb/erj108

Rengasamy P (2010) Soil processes affecting crop production in salt-affected soils. Funct Plant Biol 37:613. doi: 10.1071/FP09249

Richards L. (1954) Diagnosis and improvment of saline and alkali soils.

Rosenbaum RK, Bachmann TM, Gold LS, et al. (2008) USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. Int J Life Cycle Assess 13:532–546. doi: 10.1007/s11367-008-0038-4

Rozema J, Flowers T (2008) Crops for a salinized world. Science (80-) 322:1478–1480. doi: 10.1126/science.1168572

Saad R, Koellner T, Margni M (2013) Land use impacts on freshwater regulation, erosion regulation, and water purification: a spatial approach for a global scale level. Int J Life Cycle Assess 18:1253–1264. doi: 10.1007/s11367-013-0577-1

Scanlon BR, Jolly I, Sophocleous M, Zhang L (2007) Global impacts of conversions from natural to agricultural ecosystems on water resources: Quantity versus quality. Water Resour Res 43:18p. doi: 10.1029/2006WR005486

Schnoor JL (2013) Salt: The Final Frontier. Environ Sci Technol 4004312. doi: 10.1021/es4004312

Smedema LK, Shiati K (2002) Irrigation and salinity : a perspective review of the salinity hazards of irrigation development in the arid zone. Irrig Drain Syst 16:161–174.

Suarez DL, Wood JD, Lesch SM (2006) Effect of SAR on water infiltration under a sequential rain–irrigation management system. Agric Water Manag 86:150–164. doi: 10.1016/j.agwat.2006.07.010

Williams W. (2001) Anthropogenic salinisation of inland waters. Hydrobiologia 466:329–337.

Williams WD (1999) Salinisation: A major threat to water resources in the arid and semi-arid regions of the world. Lakes Reserv Res Manag 4:85–91. doi: 10.1046/j.1440-1770.1999.00089.x

Wood S, Scherr SJ, Sebastian K (2000) Pilot Analysis of Global Ecosystems.

This paper is from:

Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector

LCA Food 2014

8-10 October 2014 - San Francisco

Rita Schenck and Douglas Huizenga, Editors
American Center for Life Cycle Assessment

The full proceedings document can be found here:
http://lcacenter.org/lcafood2014/proceedings/LCA_Food_2014_Proceedings.pdf

It should be cited as:

Schenck, R., Huizenga, D. (Eds.), 2014. Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2014), 8-10 October 2014, San Francisco, USA. ACLCA, Vashon, WA, USA.

Questions and comments can be addressed to: staff@lcacenter.org

ISBN: 978-0-9882145-7-6