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WATER USE IN LCA

Salinisation impacts in life cycle assessment: a review of challenges and options towards their consistent integration

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Abstract

Purpose Salinisation is a threat not only to arable land but also to freshwater resources. Nevertheless, salinisation impacts have been rarely and only partially included in life cycle assessment (LCA) so far. The objectives of this review paper were to give a comprehensive overview of salinisation mechanisms due to human interventions, analyse the completeness, relevance and scientific robustness of existing published methods addressing salinisation in LCA and provide recommendations towards a comprehensive integration of salinisation within the impact modelling frameworks in LCA.

Methods First, with the support of salinisation experts and related literature, we highlighted multiple causes of soil and water salinisation and presented induced effects on human health, ecosystems and resources. Second, existing life cycle impact assessment (LCIA) methods addressing salinisation were analysed against the International Reference Life Cycle Data System analysis grid of the European Commission. Third, adopting a holistic approach, the modelling options for salinisation impacts were analysed in agreement with up-to-date LCIA frameworks and models.

Results and discussion We proposed a categorisation of salinisation processes in four main types based on salinisation determinism: land use change, irrigation, brine disposal and overuse of a water body. For each salinisation type, key human management and biophysical factors involved were identified. Although the existing methods addressing salinisation in LCA are important and relevant contributions, they are often incomplete with regards to both the salinisation pathways they address and their geographical validity. Thus, there is a lack of a consistent framework for salinisation impact assessment in LCA. In analysing existing LCIA models, we discussed the inventory and impact assessment boundary options. The land use/land use change framework represents a good basis for the integration of salinisation impacts due to a land use change but should be completed to account for off-site impacts. Conversely, the land use/land use change framework is not appropriate to model salinisation due to irrigation, overuse of a water body and brine disposal. For all salinisation pathways, a bottom-up approach describing the environmental mechanisms (fate, exposure and effect) is recommended rather than an empirical or top-down approach because (i) salts and water are mobile and theirs effects are interconnected; (ii) water and soil characteristics vary greatly spatially; (iii) this approach allows the evaluation of both on- and off-site impacts and (iv) it is the best way to discriminate systems and support a reliable eco-design.

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Conclusions This paper highlights the importance of including salinisation impacts in LCA. Much research effort is still required to include salinisation impacts in a global, consistent and operational manner in LCA, and this paper provides the basis for future methodological developments.

Keywords Irrigation · Land use change · Life cycle impact assessment · Life cycle inventory · Salinisation · Soil · Resource · Water

1 Introduction

Salinisation is the process leading to the accumulation of salts, not exclusively sodium chloride as it is frequently assumed, but also many other types of salts (Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}, Cl^{-}, SO_4^{2-}, HCO_3^{2-}, CO_3^{2-} and NO_3^{-}) (Rengasamy 2010). The salinisation process is commonly categorised in primary and secondary salinisation. Primary salinisation refers to salinisation processes mobilising natural salts (naturally present in the environment), while secondary salinisation refers to salinisation processes increased or induced by human activity (also called anthropogenic salinisation). Although we commonly consider salinisation problems to be limited to arid and semi-arid regions (Rengasamy 2006), no climatic zone is free from salinisation. Salinisation is a worldwide problem affecting various land use types: both agricultural and non-agricultural areas and both irrigated and non-irrigated lands can be prone to salinisation (Wood et al. 2000). FAO estimates that 83×10^5 km^2 are affected by salinity, including 34×10^4 km^2 of irrigated land, and 60–80 $\times 10^4$ km^2 affected by waterlogging and related salinity (FAO 2011). According to FAO (2003), soil salinisation is considered the second largest cause of land degradation from (and for) agricultural production. Soil salinity is a major issue because it adversely affects crop production, threatening agricultural sustainability (Aragüés et al. 2011). Moreover, salinisation is a threat not only to arable land but also to water resources (freshwater lakes and wetlands, rivers and streams) (Williams 1999). Indeed, secondary salinisation is impacting water resources in almost one third of the world's land area. This extent is likely to increase, in particular because of global climatic change: notably through higher evaporation rates and temperatures increasing surface water salinity, and higher water demand for crop production increasing the salts brought in the soil profile due to irrigation (Duan and Fedler 2013). Since global climatic change causes are anthropogenic, the associated salinisation can be considered as secondary salinisation. Whatever the cause, the effects lead to harmful economic, social and environmental impacts (Williams 1999).

Life cycle assessment (LCA) is a method to quantitatively assess the environmental impacts of goods and processes from "cradle to grave" (Hellweg and Milà i Canals 2014). The strength of this environmental assessment tool is to identify possible burden shifting from one environmental impact category to another, by addressing all impacts occurring throughout the entire value chain. However, salinisation is missing in the range of impact categories of most LCA case studies (Finkbeiner et al. 2014). Because of the important environmental damages of salinisation, including this impact in LCA is considered a high priority for research (JRC-IES 2011). Many LCA studies highlight this methodological gap for the environmental impact assessment of many technological processes: brine disposal from water desalination (Muñoz and Fernández-Alba 2008; Zhou et al. 2013a), water treatment processes (Friedrich and Pillay 2009), micro-algae cultivation (Grierson et al. 2013) and, especially, agricultural products (Bartl et al. 2012).

Yet, only four methods are available to assess salinisation impacts in LCA: Amores et al. (2013); Feitz and Lundie (2002); Leske and Buckley (2003; 2004a; 2004b) and Zhou et al. (2013b). All methods were applied at least once in an applicability test performed by the authors; only the methods of Feitz and Lundie, valid in Australia, and the one of Amores et al. (2013), developed for a specific case in Spain, were applied in other case studies (Tangsubkul et al. 2005; Muñoz et al. 2010; Antón et al. 2014). Overall, they either focus on one salinisation type or on one specific geographical location. Therefore, there is a lack of a comprehensive approach to assess salinisation impacts due to human interventions in the LCA framework.

Our objective is to provide the scientific basis to build a relevant and complete model to assess salinisation impacts in LCA. This was done following the guidelines from Cucurachi et al. (2014), Huijbregts (2013) and Jolliet et al. (2014) for the development and critical evaluation of life cycle impact assessment (LCIA) methods. A three-step approach was adopted: (1) the setup of a comprehensive and structured overview of anthropogenic salinisation mechanisms and cause and effect chains. For this overview, we collected evidence from the literature as a measure of the importance and priority of salinisation, with support from specialists in salinisation outside the LCIA field; (2) the critical analysis of the LCIA methods modelling salinisation impacts according to the criteria proposed by the Institute for Environment and Sustainability from the European Commission (JRC-IES 2011); and (3) the identification of the methodological issues and recommendations to build a consistent framework for including salinisation impacts in LCA. At this stage, recommendations are mostly of conceptual nature; operationalisation will be the aim of future research.

2 Salinisation environmental mechanisms

The detailed analysis of the salinisation environmental mechanisms is relevant to highlight the processes involved in salinisation impacts due to human interventions. LCA addresses impacts of human interventions. Therefore, we decided to focus on anthropogenic salinisation (secondary salinisation).

2.1 Salinity

Salinisation is the accumulation of salts. The major cations involved are sodium, calcium and magnesium and the major anions are chloride, sulfate and carbonate (Rengasamy 2010). Salinity refers to the total concentration of these salts in both soil and water samples and is measured with the electrical conductivity (EC, in siemens per metre) of water or a soil-saturated extract. EC is strongly correlated with the ion charges and total dissolved solids (TDS) in soil water (the liquid phase of soils) (Corwin and Lesch 2005). The nature of salts involved is also important: when the sodium is in excess, an additional process may occur, which is soil sodification (Ghassemi et al. 1995). Sodification is the accumulation of sodium on the soil exchange complex causing soil clay dispersion, responsible for soil structure degradation. Sodic conditions are characterized by the exchangeable sodium percentage (ESP, dimensionless): the amount of sodium held in exchangeable form on the cation exchange complex, measured in soil extracts, and the sodium adsorption ration (SAR, dimensionless) measuring the relative preponderance of dissolved sodium in water compared to the amounts of dissolved calcium and magnesium (Rengasamy 2010; USDA 1954). Several classifications exist for salt-affected soils (e.g. Rengasamy 2010). Depending on the classification system, the SAR and EC thresholds values are not the same (Rengasamy 2006). From an operational point of view, farmers usually classify irrigation water according to EC measurements and crop sensitivity (USDA 1954).

2.2 Human interventions causing soil and water salinisation

Soil and water salinisation are often studied separately: “Salinisation is the process that increases the salinity of inland waters” (Williams 1999). “Salinisation is an accumulation in the soil of dissolved salts” (Wood et al. 2000). But soil and water salinisation are inter-related, water being the agent for salt movement (Grundy et al. 2007). 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 migrate with water. We distinguished four main patterns of salinisation due to human interventions associated with land use change, irrigation, brine disposal and overuse of a water body. Herein, we identified the biophysical and human management factors responsible for both soil and water salinisation for each type.

2.2.1 Salinisation associated with land use change

Land use change (LUC) modifies hydrological processes and therefore the water cycle at the catchment scale, in particular, clearance of deep-rooted perennial native vegetation and replacement with shallow-rooted crops that decrease transpiration rates and increase water infiltration rate in the vadose zone. As a consequence, saline groundwater tables can rise and reach the near-soil surface in lowlands. This leads to soil salinisation through capillary rise (Williams 1999) or artesian flow (Hammecker et al. 2012). In addition, percolation of salts can contribute to the increase of the salinity of the aquifer as it was observed in Australia (Williams 1999; Grundy et al. 2007; Scanlon et al. 2007) or in Thailand (Williamson et al. 1989; Hammecker et al. 2012) and the USA (Black et al. 1981; Scanlon et al. 2007). This salinisation type involves specific biophysical factors such as topography, precipitations, groundwater table level, soil geochemical and hydrodynamic profiles and salt stock in soil, but also management factors such as a land use change modifying the evapotranspiration rates (Table 1).

2.2.2 Salinisation associated with irrigation

Irrigation and fertilisation can cause soil and water salinisation. Salts provided by irrigation water have a higher tendency to accumulate in the soil in semi-arid and arid areas because of the conjunction of low rainfall and high evapotranspiration rates (Marlet and Job 2006). Irrigation water always contains some salts, but the use of low-quality water (e.g. treated wastewater) to compensate the increased scarcity of freshwater might worsen salinisation (Duan and Fedler 2013). Salts are also present in fertilisers (Scanlon et al. 2007), so fertiliser applications influence the salinity of the soil (Boman and Stover 2012). On the one hand, fertilisers may cause salinity increase; on the other hand, their appropriate management helps cope with saline conditions. The development of irrigation affects the local geohydrological regime, mobilises salts stored in the underlying substrate (Smedema and Shiati 2002) and favors salts leaching from the root zone to waterbodies or underlying groundwater (Mateo-sagasta and Burke 2010). Finally, if irrigation overcomes drainage capacities, the rise of the groundwater table causes soil salinity issues through capillary action (Corwin and Lesch 2005). The rise of saline groundwaters may in turn induce salinisation of some freshwaters (Williams 2001). Subsequently, poor irrigation management and inadequate drainage often lead to salinisation and waterlogging (Wood et al. 2000). Thus, there are tradeoffs, notably between salinisation of the soil in the case of insufficient salt leaching and salinisation of the underlying

Table 1 Key management and biophysical factors involved in secondary salinisation, per salinisation type.

Salinisation type	Spatial scale	Management factors	Biophysical factors
Land use change	Hydrogeological catchment	Land use transformation: ET rates modification	-Topography -Soil geochemical and hydrodynamic profile -Salt reservoir in soil -Water table depth -Precipitation
Irrigation (w or w/o shallow groundwater or poor drainage)	Local (field) within regional context (groundwater)	- Volume of irrigation water - Salts in water - Salts in fertilizers - ET rates - Drainage rates (and irrigation mode)	-Soil hydrodynamic profile -Precipitation -Salt reservoir in soil -Water table depth
Brine disposal	Local (discharge location) within regional context	- Salts in water - Discharge location	-Many factors involved, relying on the geographical features of the discharge context
Overuse of a water body	Surface and underground water catchments	- Volume of water withdrawn - Water body exploitation rate	-Distance to the coast or estuary -Presence of saline aquifer

aquifer if salts are leached. There are two scales involved in this salinisation context: a regional and a local one. If the spatial structure of groundwater tables is regional, associated salinisation processes act at a local scale (plot/farm). The embedded biophysical factors of this salinisation context are the water table depth, soil hydrodynamic profile, precipitation, evapotranspiration rates and salt reservoirs in the soil. The management factors are salt content in irrigation water and fertilisers, irrigation volume and drainage rates (Table 1).

2.2.3 Salinisation associated with brine disposal

Many activities generate saline wastewater, e.g. mining, pumping of shallow saline aquifer and seawater desalination, therefore the problem of brine disposal is raised (Williams 2001). This is a topical question to address while many countries need to complement their water supply with seawater desalination (Zhou et al. 2013b). In coastland desalination plants, brine may be discharged in seawater (impacting the marine ecosystem), whereas in inland areas, brine discharge is more problematic because diluting brine in a water stream or discharging it directly in the soil may lead to water and soil salinisation (Sánchez et al. 2015). New alternatives are studied, such as the use of brine water for agricultural use, in combined scheme (e.g. microalgae cultivation, fish production and halophyte forage scrub irrigation). But these alternatives do not prevent the gradual salinisation of land (Sánchez et al. 2015). Brine disposal is a major cause of aquatic ecotoxic impact, and the subject is of growing interest in research (Zhou et al. 2014). This salinisation type highly depends on the salt composition of the brine and the discharge location. Salinisation due to brine disposal will be driven by many biophysical factors relying on the geographical features of the discharge context, actually all biophysical factors identified for the other salinisation types.

2.2.4 Salinisation associated with overuse of a waterbody

In many coastal areas, excessive withdrawal of groundwater and/or river streams leads to seawater intrusion: the decrease of the coastal aquifer table level induces seawater inflow in the aquifer, leading groundwater to long-term salinisation (Flowers 1999; Scanlon et al. 2007; FAO 2011). The depth of the interface between freshwater and seawater is reduced when the aquifer table is decreased as illustrated by the Ghyben-Herzberg formula, a linear relationship often used to simulate seawater intrusion (for a review of methods investigating seawater intrusion processes, see Werner et al. (2013) and Sreekanth and Datta (2015)). In the estuaries and deltas, seawater intrusion happens when the freshwater flow of the river is reduced because of excessive water withdrawal upstream or the construction of impoundments (Williams 2001; FAO 2011). Sea level rise induced by climate change is an aggravating factor of seawater intrusion (FAO 2011). In non-coastal areas, saline intrusion may result from saline water transfer from a saline aquifer to an overused aquifer. This type of salinisation happens when too much water is withdrawn from a waterbody, independently of the usage. However, irrigation is the principal cause because 70 % of all water extraction worldwide is devoted to agricultural use (World Water Assessment 2009). Salinisation associated with saline intrusion involves mechanisms at the regional (e.g. fluctuating sea level) and local (e.g. well) scales (Werner et al. 2013). The biophysical factors involved are the distance to the coast or estuary and the presence of saline aquifer. The management factors are the volume of freshwater withdrawal and the exploitation rate of the waterbody (river or aquifer) (Table 1).

2.3 Water and soil salinisation damages to ecosystems, human health and resources

Salinisation of soils and waters affects and ultimately damages the so-called areas of protection (AoP) commonly used in LCA, i.e. ecosystems (or natural environment), human health and resources (Dewulf et al. 2015). Figure 1 depicts the salinisation environmental mechanisms (or cause-effect chains).

Soil salinisation not only affects terrestrial ecosystems and crop growth but also degrades land more or less permanently (D'Odorico et al. 2013). Salt-affected soils have a lower fertility through three potential effects on plants: (i) reduction of plant water uptake or dryout by lowering the osmotic potential, (ii) toxic effect by different ions depending on the soil pH and (iii) plant nutrient uptake imbalance (Flowers and Flowers 2005). Sodic soils also have effects due to soil structure degradation and permeability reduction (Suarez et al. 2006; Rengasamy 2010; D'Odorico et al. 2013). Decrease of the soil fertility and thus of the yield production potential could result in malnutrition for poor populations (UNESCO 2003). Impacts of salinity on ecosystems and human health also include increased flood risk and increased infrastructure failure risk (Grundy et al. 2007; Zhou et al. 2013a). Soil salinisation is also considered a driver for desertification, and it is closely related to land degradation processes such as soil erosion and arable land abandonment (D'Odorico et al. 2013). Although some measures to reduce soil salinity and sodicity can be employed, salinisation is considered irreversible in arid regions where there is not enough freshwater available to leach out the accumulated salts (Rozema and Flowers 2008) or in lowland areas of endorheic basins (i.e. closed drainage basin) with shallow and saline groundwater (D'Odorico et al. 2013). Land degradation due to salinisation might then be considered a damage to the soil resource. It is noteworthy that salinisation management techniques are simply shifting the problem by moving salts from one compartment (e.g. root zone) to another (e.g. ground water).

Salinisation of a waterbody not only affects the aquatic and riparian ecosystems but also reduces the water availability for further use. An increase of water salinity causes a change in the species composition of algae, zooplankton and benthic communities and leads to the disappearance of macrophytes and riparian trees (Williams 1999; Schnoor 2013). It should be noticed that ecosystems may not lose diversity per se but evolve from a halosensitive biota (organisms sensitive to high salinity conditions) to a halotolerant one (organisms adapted to high salinity conditions) (Williams 1999). In addition, saline freshwater lakes, wetlands, rivers or aquifers are unfitted to serve as supplies for domestic, agricultural and other uses (Williams 1999; FAO 2011), thus resulting in water deprivation for humans and ecosystems. This quality alteration of the

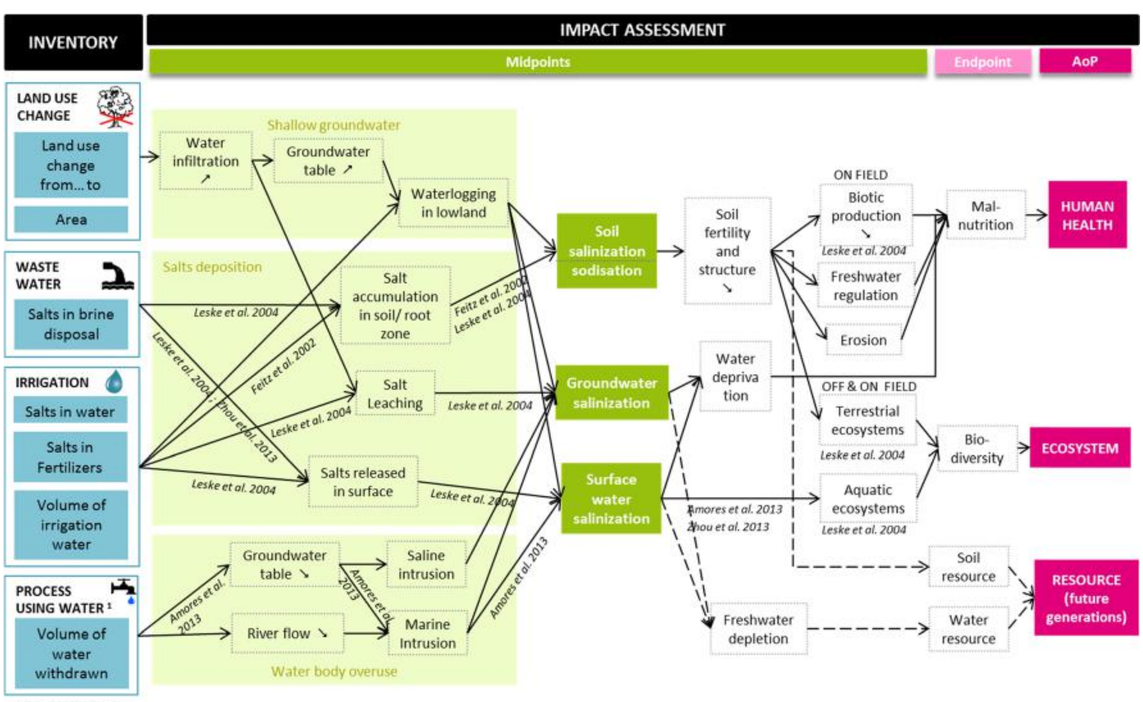

Fig. 1 Human-driven salinisation environmental mechanisms and positioning of approaches proposed in the literature. *Long dash lines* represent controversial pathways in the scientific community

water resource may be irreversible, for example for a permanently saline aquifer, and thus affects the water resource for present and future generations.

2.4 Complexities related with salinisation in space and time

Salinisation processes are often inter-related (Williams 2001) and involve environmental mechanisms from different nested scales. Physico-chemical mechanisms stand at local scale and hydrological mechanisms at catchment scales. Nevertheless, the hydrological processes causing the salt mobilization are similar for all salinisation types (Zhou et al. 2013a). Although we can establish a typology of salinisation contexts, in many cases the situation is complex because salinisation results from several causes. The combined effect of the replacement of natural vegetation by agricultural crops upstream and the discharge of saline agricultural wastewater can lead to the salinisation of many freshwater lakes, wetlands and rivers (Williams 2001). Groundwater salinisation can be due to both seawater intrusion and the agricultural return flows (Bouchaou et al. 2008). In addition, water and soil salinisation are intimately related. The degradation of freshwater resources (surface or groundwater) has concomitant effects on the systems using these sources, and soil salinity affects in turn water resources (D'Odorico et al. 2013). The identification of the actual cause of salinisation is also difficult because its effects can be transferred in time and space from its causes (Grundy et al. 2007). It should be mentioned that other salinisation contexts with a narrower extent were not presented here (e.g. sea spray, deicing salt spreading on roads, industrial wastes). It is also important to specify that the descriptions of salinisation mechanisms provided in this article focus on the main processes involved, for the sake of clarity and concision. Since salinisation mechanisms are intimately related with the water cycle, all water flows can potentially have an influence on salinisation.

Salinisation impacts are demonstrated in literature. They are of crucial concern, and the link between different human interventions and the salinisation impacts on the three AoP has been clearly established.

3 Critical analysis of salinisation impact assessment methods in LCA

Four methods have been developed to assess salinisation impacts in the LCA framework so far (Table 2). These approaches are either midpoint-oriented (Feitz and Lundie 2002), endpoint-oriented (Zhou et al. 2013b; Amores et al. 2013), or near-endpoint-oriented (Leske and Buckley 2003; 2004a; 2004b). To highlight their strengths and flaws, we analysed the methods against the criteria defined in the International Reference Life Cycle Data System Handbook procedure proposed by the Institute for Environment and Sustainability from the European Commission (JRC-IES 2011). The criteria are as follows: completeness of scope, environmental relevance, scientific robustness and certainty, documentation, transparency and reproducibility, applicability and potential stakeholder acceptance. The detailed assessment is available in the Electronic Supplementary Material (Table S1).

3.1 Salinisation associated with irrigation: Feitz and Lundie (2002)

The midpoint soil salinisation 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, expressed in $Na^{+}{}_{eq}$. The inventory data requirements are the volume of irrigation water (V_i) and the water sodium concentration ([Na]). These parameters are multiplied by a soil sodisation hazard characterization factor (CF). The soil sodisation hazard is assessed through the ratio between the electrical conductivity threshold ($EC_{threshold}$) representing the limit of soil structure integrity for a given SAR and the EC of the irrigation water (EC_{iw}). This method presents the advantage of assessing both salt accumulation in the soil and soil structure degradation. Apart from the applicability test done by the authors, two case studies applied this method: Tangsubkul et al. (2005) and Muñoz et al. (2010). Muñoz and colleagues (2010) calculated, besides soil organic carbon deficit, soil salinisation potential as one indicator for soil quality impacts, to compare different water sources and water qualities for irrigation purposes. Tangsubkul et al. (2005) calculated the relative soil salinisation potential of the use of irrigation water from different water recycling technologies.

The main limitations of this method are as follows: its restricted scope; its limitation to irrigated cropping systems, not prone to waterlogging; and its limited validity domain for the CF, in spite of the detailed inventory data requirement. The 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. For example, 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). The indicator is only valid for soils within the validity domain of the electrolyte threshold curve, and the estimation of the SAR of the soil drainage water is assumed for an Australian red-brown earth. The applicability of the method may be hampered by the data requirement. Indeed, CFs have to be calculated by the practitioner because they depend on the quality (thus the composition) of the irrigation water used (Table 2).

Table 2 Inventory requirement, characterization factors and category indicator results of salinisation impact assessment methods in LCA

Article	Life cycle inventory (LCI)	Characterization factor (CF)	Category indicator result
Feitz and Lundie 2002	**Vi**: Irrigation water volume (L), [**Na**]: water sodium concentration (mg/L).	**CF** = $EC_{threshold}/EC_{iw}$ With: $EC_{threshold} = 0.121 \times SAR + 0.033$ equation representing the clay flocculation - dispersion threshold. SAR calculation requires: [Ca], [Mg], $[SO_4]$, $[CaCO_3]$, pH, EC_{iw} of irrigation water. If CF < 1, no soil degradation hazard from sodisation.	**Midpoint** $\sum_i CFi \times [Na]i \times Vi$ For irrigation water i. **Unit**: kg Na^+_{eq}
Amores et al. 2013	$\mathbf{ET_{crop}}$: Crop groundwater consumption (m^3/yr) (= Evapotranspiration of the crop: ET_{crop} in Amores et al. 2013)	**CF** = FF ∗ EF = change in Potentially Affected Fraction of species (PAF) due to a change in salinity due to a change in groundwater consumption: With: FF = **Fate Factor** = $\Delta FGW/\Delta ET_{crop} \times \Delta C_N.V_N / \Delta FGW$ FGW: fresh groundwater inflow to Lagoon, C: salinity, V: volume of the lagoon, ET_{crop}: crop ET, Δ: change between years EF = **Effect Factor** = $\Delta PAF_{sal} / \Delta C_N.V_N = 0.5/HC50_{sal}$[1]	**Endpoint** ET_{crop}. CF **Unit**: PAF m^3.year, converted into species.year considering a 7.89×10^{-10} species m^{-3} freshwater species density
Zhou et al. 2013b	$\mathbf{m}_{\mathbf{salinity\ group}}$: mass of chemical group "salinity" (Cl^-, Na^+, SO_4^{2-}, Mg^{2+}, Ca^{2+}, K^+, HCO^{3-}) in 1 m^3 of brine, (kg)	For chemical group "salinity", based on a "whole effluent approach": $\mathbf{CF_{salinity\ group}}$ = FF ∗ XF ∗ EF = 4.62E − 01 PAF m^3 day/kg With: FF = **Fate Factor** = 37 days; residence time of Cu^{2+}, based on USEtox fate model XF = **eXposure Factor** = 1; salts 100 % dissolved in water EF = **Effect Factor** = 0.5/EC50 = 1.25E − 02; $EC50_{salinity}$ = 40,000 mg/L	**Endpoint** $\sum_j m_{salinity\ group} \times CF_{salinity\ group}$ **Unit**: PAF m^3 day
Leske and Buckley 2003;2004a; 2004b	**TDS released**: Total Dissolved Salts released (kg) in a compartment	**CF** = total salinity potential (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: days in the simulation	**Near-endpoint** TDS released × TSP **Unit**: kg TDS $_{eq}$

[1] HC50: concentration at which ≥ 50 % of the species are exposed to concentrations above their EC50 (concentration where a 50 % reduction in a given endpoint (e.g. growth) is observed compared to the control)

3.2 Salinisation associated with overuse of a water body: Amores et al. (2013)

Amores and colleagues (2013) evaluated the damages on biodiversity associated with a salinity increase in a Spanish coastal wetland. This salinity increase is caused by seawater infiltration in the wetland, after groundwater overexploitation for irrigation. The inventory data required is the volume of groundwater consumed by the studied crop (ET_{crop}). To calculate the indicator result, the inventory is multiplied by the CF, composed by a fate and an effect factor. The fate factor represents the change in salt concentration in the wetland due to a change in groundwater consumption. It is calculated from yearly averaged seasonal water and salt balances for the wetland Albufera de Adra in the South of Spain (Table 2). The effect factor stands for the change in potentially affected fraction of native wetland species due to salinity increase. Species included in the assessment are plants, fishes, algae and a crustacean. Apart from the applicability test done by the authors, the literature provides one case study: Antón et al. (2014) calculated damages on biodiversity from water consumption for tomato grown in the specific area where Amores and colleagues developed their method.

This method addresses a specific context of salinisation associated with overuse of a waterbody: seawater intrusion in a wetland (fed by the groundwater), but does not consider a seawater intrusion in the aquifer. Although the approach could certainly be applied in other contexts, the main limitation of this method is its limited geographical validity. The fate factors are based on water and salt balances relying on the specific hydrological 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. The common endpoint unit PAF allows comparison with other methods and impacts: in the tomato case study, the biodiversity loss in the wetland due to salinisation has the same order of magnitude than terrestrial acidification and terrestrial ecotoxicity impacts (Antón et al. 2014).

3.3 Salinisation associated with brine disposal: Zhou et al. (2013b)

Zhou et al. (2013b) propose a method for assessing aquatic ecotoxicity of brine disposal from seawater desalination plants. The aquatic ecotoxic impact is the sum of the impacts generated by groups of influential chemicals. The method consists in deriving specific aquatic ecotoxic potential CFs for each group of chemicals: metals, organic and inorganic chemicals. This approach is supported by freshwater ecotoxic CFs from the USEtox database. Due to the lack of fate models for the inorganic salt group, a whole effluent approach is used instead of a chemical-specific approach: CFs are estimated based on a worst case scenario.

The inventory data required is the mass of chemical group "salinity" ($m_{salinity\ group}$) in 1 m^3 of brine, then multiplied by the CF, composed by a fate and an effect factor, to calculate the indicator result. The residence time of most persistent chemicals is used as a fate factor. But since the persistence time of Na^+ ions (millions of years) exceeds the range of the acute test (100 years), the residence time of the second most persistent chemical in the brine mixture, Cu^{2+}, is used instead (Table 2). The effect factor is calculated based on a worst case scenario EC50 (Yoon and Park 2012). To our knowledge, no case study has used this method yet, apart from the applicability test done by the authors.

This group-by-group approach presents the advantage of including the contribution of inorganic chemicals such as salts, which are suffering from a lack of CFs in the usual LCIA models. However, this whole effluent approach may be associated with high uncertainty if the composition of the inorganic group is highly variable. The fate factor is based on the residence time of Cu^{2+} which does not belong to the salinity group: this metal is assessed with a chemical-specific approach. Regarding the effect factor, the EC50 corresponds to the salinity concentration threshold for acute toxicity of brine on four phytoplanktons (Yoon and Park 2012). The EC50 (referring to growth rate) values reported in this experiment range from 40.2 to 78.7 g/L. This high variability of the EC50 and the limited number of species considered (all marine) warrant the need to use a HC50 based on a wider range of aquatic species. Recent publications in the field may now allow the use of HC50, a better alternative to EC50 (Zhou, personal communication).

3.4 Salinisation associated with salt release: Leske and Buckley (2003; 2004a; 2004b)

Leske and Buckley (2003; 2004a; 2004b) developed a salinity impact category for South African LCA. They provide salinity potential CFs for salt release in the atmosphere, surface water, natural surfaces and agricultural surface compartments. The CFs stand for potential effects on aquatic ecotoxicity, materials, natural wildlife, livestock, aesthetic effects, natural vegetation and crop. Inspired by a risk assessment approach, salt fate factors are calculated with an atmospheric and hydrosalinity catchment model. Effect factors are calculated using the predicted no-effect concentration for each environmental target (Table 2). This method is covering several salinisation contexts and pathways, accounting for both water and soil salinisation (Fig. 1). However, it does not cover salinisation induced by a LUC or a saline intrusion. The main limitation of this method is its geographical validity restricted to South Africa. Indeed, the fate factors are calculated with a catchment model for South Africa, and the effect factors are based on the South African Water Quality Guidelines. It is noteworthy that the calculated CFs for salt emissions onto the agricultural soil by far outweigh the CFs for releases into other compartments. This warrants further research to better model agricultural systems. The near-endpoint indicator units TDS cannot be compared with other methods as it is specific to salinity.

3.5 Lack of consistent frameworks

The methods provide relevant methodological approaches to salinisation impact modelling and could certainly be inspiring in other contexts. Their main limitations though are their restricted scope in terms of pathways covered (Feitz and Lundie 2002), their intensive inventory data requirement (Feitz and Lundie 2002) or their restricted geographical validity (Amores et al. 2013; Feitz and Lundie 2002; Leske and Buckley 2003, 2004, 2004b). All methods have site-specific CFs, emphasising that salinisation impacts are highly site-dependent, especially regarding the hydrology, the climate and irrigation water quality, but are not globally applicable. Applying these methods in other contexts requires to redevelop the whole characterization approach: a new effect factor specific of the species in the studied wetland (Amores et al. 2013), using an equivalent country-specific catchment hydrosalinity model of the studied country (Leske and Buckley 2003; 2004a; 2004b). As a result, it would be time-consuming and data-intensive to adapt the methods to other contexts. Figure 1 positions the contributions of these approaches on the global salinisation cause-effect chains identified, showing that a consistent framework is missing (Finkbeiner et al. 2014).

4 Towards a consistent framework for salinisation impacts assessment in LCA: methodological issues and recommendations

4.1 Context of LCIA for assessing salinisation impacts

The purpose in this section is to analyse how salinisation impacts could be modelled within the methodological framework of LCA. Answering this question raises topical methodological issues.

Since salinisation may affect soil and water resources, both often considered as limited resources at least locally, there is a need to analyse the status of the AoP resources in LCA. Dewulf et al. (2015) identified and discussed the different status of the AoP resources, in between the natural environment (their cradle) and the human-industrial environment (their application), depending on the viewpoint. This fundamental on-going debate is presented in the Electronic Supplementary Material (S.2). In the following, we define the AoP resources as the protection of a resource (in sufficient quality and quantity) for future generations, while the AoP human health and ecosystems reflect the protection of current people and ecosystems. Since salinisation can potentially be assessed through several LCIA modelling approaches, in relation to a salt emission and/or water use and/or land use/land use change (LULUC), we analysed in the following the different modelling options for each salinisation types.

4.2 Modelling options for the different salinisation types

4.2.1 Midpoint indicators

It is challenging to define the optimum midpoint indicator because "Midpoints concern all elements in an environmental mechanism of an impact category that fall between environmental intervention and endpoints" (Udo de Haes et al. 2002). What are the best midpoints: soil and water salinisation, or soil fertility loss and freshwater deprivation? (Fig. 1) On the one hand, soil fertility loss and freshwater deprivation as midpoints represent a real impact and not just a concentration increase. On the other hand, soil and water salinisation as midpoints represent the end of the fate modelling. In addition, soil and water salinisation could be expressed in the same unit (e.g. TDS or EC) so we can sum up the two midpoints into one single indicator representing salinisation impacts for both soil and water. A conversion factor exists between TDS (mg L^{-1}) and EC (μS cm^{-1}): TDS = 640 EC (USDA-NRCS 2015). For these reasons, we propose to define the midpoints as the soil and water salinisation.

4.2.2 Salinisation associated with land use change

According to JRC-IES (2011), a completed or revised land use framework may include soil salinisation in LCA. However, we believe that the LULUC framework can only partially model salinisation associated with a land use change. The human intervention is a land transformation of a given area (e.g. from forest to arable), matching perfectly with the inventory flow requirement: the area of a land use cover transformed from one type to another. Koellner et al. (2013b) provided a tiered typology of the land use and cover categories: they can be global (e.g. "arable") or more refined (e.g. "arable, irrigated, intensive"). These inventory flows are regionalized because the same type of land use may trigger different impacts at different locations of the globe. Regarding the CF, it reflects a difference in quality as well as a regeneration time for a LUC. In some cases, the regeneration time is exceeding the modelling horizon, thus corresponding to permanent impacts.

The LULUC framework assesses impacts on-site (at the location of the intervention) but does not model accurately in practice off-site impacts (not at the location of intervention). This is a limitation to account for the subsequent waterlogging and salinisation of soil and aquifer occurring off-site (from the LUC) in lowlands. For example, among the operational methods available, Saad et al. (2013) assesses the land use impacts on freshwater regulation, erosion regulation and water purification. However, the hydrological balance alteration downstream of the location of a LUC is not modelled in Saad et al. (2013). Furthermore, soil salinity may impact freshwater regulation, erosion regulation and water purification, but this soil parameter is not accounted for in the LANd use indicator value CAlculation (LANCA®) model used by Saad et al. (2013). LANCA® is a calculation tool model assessing the influence of different land use activities on soil ecological functions (Beck et al. 2010; Saad et al. 2011). It is important to notice the strong link between land use and water use impacts, especially for irrigated agriculture because irrigation is part of the land use practices and LUC can lead to changes in the water cycle at the catchment scale (Koellner et al. 2013b). We thus suggest the use of a hydrological model accounting for the key parameters listed in Table 1: linking the inventory flows (LUC) with a mechanistic fate modelling of water and salts (i.e. describing environmental mechanisms). The LANCA model is a good basis, but several limitations should be overcome (e.g. accounting for soil salinity, not considering a constant depth of aquifer). The use of a globally valid and reliable model such as SWAT (modelling the movement of pesticides, sediments or nutrients, and driven by the water balance of the watershed) (Neitsch et al. 2009) could be investigated.

In the LUC salinisation context, the main issue is related to the time frame: how to allocate the impacts when the land use change occurred many years ago. In many areas, as in Australia, dryland salinity is an on-going threat but is not easily linked to on-site agricultural management (Renouf et al. 2014).

4.2.3 Salinisation associated with irrigation, brine disposal and overuse of a waterbody

The LULUC framework is not appropriate to model salinisation associated with irrigation, brine disposal and overuse of a waterbody. The land use and land cover types are not refined enough to account for the key parameters affecting salinisation such as the irrigation mode or the crop type. Nevertheless, defining refined land cover types, such as "citrus crop, drip irrigation, water EC = 4dS m^{-1}", is not feasible. In addition, accounting for salt emission through the land use framework reduces the frontline with the salt emission modelling framework, thus increasing double counting risks. Koellner and Geyer (2013a) highlighted this difficulty to define an integrated impact assessment where land use impacts are accounted for alongside chemical emissions, water use and climate change impacts. This overlapping risk increases with the evolution of the land use framework (more detailed land use types are claimed), in parallel with the improvement of methods based on environmental mechanisms such as for water use impact modelling (e.g. Verones et al. 2013a; 2013b). The land use framework is a good way to assess multifactorial impacts that are complex to model and could be a good strategy in LCA where the agricultural phase is of minor importance. For LCA including an important agricultural phase, modelling complex cause-and-effect chains with a fine description of environmental mechanisms is preferable and is gradually being developed (e.g. pesticides, water).

The inventory flow requirement for each salinisation type will vary depending on the boundary between the technosphere and the ecosphere (this will be discussed in Section 4.2.4). The minimum inventory requirements are as follows: for salinisation associated with irrigation, the inventory flow consists of a salt emission (from water and fertilisers) and a volume of irrigation water, considering both water quantity and quality like Boulay et al.'s method (2011a); for salinisation associated with brine disposal, the inventory flow consists of a salt emission in brine disposal; and for salinisation associated with overuse of a water body, the inventory flow is a volume of water withdrawn (Fig. 1).

For the LCIA, a bottom-up approach will focus on the stressors (interventions responsible for the impacts) and allow to better discriminate specific human interventions and contexts, in comparison with the alternative top-down approach organising impacts according to which AoPs are affected (Udo de Haes et al. 2002). Again, we recommend an approach modelling the environmental mechanisms, in line with the ISO standard which states that category indicator shall use identifiable environmental mechanisms. The cause-effect consists in three steps: the fate of the substance in the environment until the final compartment, the exposure of the target and the effect of this substance on the target. But following salts implies following water. The fate of salts and water, ultimately reaching soil, aquifers and surface compartments, should be calculated based on salt and water balances, involving different scales (field and/or catchment), and accounting for the key parameters listed in Table 1. The fate modelling should allow for the discrimination of systems according to key parameters such as soil type, irrigation mode and water quality. For example, in agricultural LCA, discriminating systems according to the irrigation mode would be relevant because there is a paradoxical effect between water saving and salinisation: switching from a surface to a drip irrigation system helps save water but increases the soil salinisation. Indeed, the adoption of drip irrigation results in the reduction of the amount of irrigation water, but salt leaching is reduced as well, thus increasing soil salinity. Another advantage of adopting an approach based on environmental mechanisms lies in the assessment of both on-site and off-site impacts thanks to the fate factor which follows the substance in the environment. However, on-site impact assessment for agricultural systems is dependent on the boundary definition between the technosphere and the ecosphere as shown for pesticide emissions by Van Zelm et al. (2014) (Cf. Part 4.2.4).

A geographical differentiation of CF is required because the fate of salts depends on climate, soil and the hydrological context and their effect depends on the sensitivity of the target (e.g. species, capacity to desalinate water). It is paramount to develop highly spatially explicit CF supporting aggregation over the whole life cycle (Hauschild et al. 2013; Cucurachi et al. 2014). Geo-referenced CFs should be supported by geo-referenced databases, which availability may hamper a operationalisation method (Cf. Part 4.3). Regionalised impacts in LCA can be supported by geographic information systems (e.g. Núñez et al. (2010); Boulay et al. (2011b); Saad et al. (2013); Núñez et al. (2013)). The time horizon definition is of crucial importance as well for the calculation of the salt fate factor, as it can modify the outcome of an LCA, especially when quantifying the impact of substances with a long lifetime (De Schryver et al. 2011; De Schryver et al. 2012; Huijbregts 2013). The time horizon is related to the spatial scale as it will determine which final compartment is reached at this time. If salinity varies on a daily basis and can have visible effects within a short period (crop cycle scale), in contrast, sodicity has an effect after many years of inadequate management (several crop cycle scale). In the model selection, the challenge will be to find a tradeoff between feasibility and accuracy. As an example for agricultural systems, the AquaCrop model (Steduto et al. 2012) is a salt and water balance model including a crop growth model, which can be coupled with GIS. This model is already valid for all herbaceous crops and should be updated to progressively include perennial crops (E. Fereres, pers, comm.).

The disadvantage of modelling each pathway following the fate and effect factors (bottom-up approach) lies in the subsequent impact weighting (Goedkoop and Spriensma 2001): we can only add the impacts and miss the potential combined effects, in opposition to the top-down approach.

4.2.4 From midpoint to endpoint

Water and soil salinisation affect the three AoP. We analyse in the following the modelling options for each AoP. The fate factor being part of the previous step (from human intervention to midpoints), this section refers strictly to effect factors.

4.2.5 Ecosystems

Water salinisation affecting ecosystems could be modelled through aquatic ecotoxicity assessment. Amores et al. (2013) adopted this type of modelling in a specific context. Salts are not yet modelled in the USEtox model, a scientific consensus model providing CFs for both human health and freshwater ecotoxicity impacts (Rosenbaum et al. 2008), but future developments of this tool could include freshwater ecotoxicity CFs for salts. Another future improvement of the USEtox model is to develop regional versions because no spatial differentiation of location of the emission was considered so far (Henderson et al. 2011). Similarly, soil salinisation affecting ecosystems could be modelled through terrestrial ecotoxicity assessment. The development of terrestrial ecotoxicology CFs is also part of future developments of the USEtox model (Henderson et al. 2011).

Soil salinisation occurs on agricultural land and on non-agricultural land. It is crucial to differentiate it because non-agricultural land refers to biodiversity everywhere except in the agricultural soil (AoP ecosystems), whereas agricultural land refers to agricultural soil biodiversity which has an intrinsic value (AoP ecosystem) but also a crop support value through the support of soil fertility and a potential effect on malnutrition (AoP human health) (Fig. 2). Thus, salinisation on agricultural land refers to damages on biodiversity (AoP ecosystem) and fertility of the soil (AoP human health). LCA should be able to reflect the importance of agricultural soil biodiversity and its paramount role in land fertility. There are no double-counting risks if the discrimination between the two pathways—agricultural land salinisation effects on agricultural soil ecosystem (AoP Ecosystem) and on soil fertility (AoP Human health)—is done properly.

The effects of soil salinisation on agricultural land biodiversity/ecosystems can be accounted for only if the technosphere (system studied) and ecosphere (environment) boundary allows for it, i.e. if the agricultural soil is not completely included in the technosphere or is included into it only momentarily. Indeed, several soil status options are possible because soil is both an environmental target and a part of the agricultural system. Setting the technosphere boundary will determine which parameters have to be accounted for in the inventory or to be part of the impact assessment (Fig. 3). Following the recommendations of Rosenbaum and colleagues (2015), this boundary should be defined according to the goal and scope of the study. In the case of salinisation associated with irrigation, if the objectives are to (i) distinguish the management practices and (ii) put the stress on the human intervention on which we can act to reduce impacts, we recommend including in the technosphere the part of the soil that is influenced by the practice. Moreover, if one does not want to miss the potential impact on the agricultural land, soil should be included in the technosphere only during the time it is being used by the system and supporting its function, and considered as returned to the ecosphere in a potentially modified state afterwards (not shown in Fig. 3). From an operational viewpoint, this means that the discriminating factors such as drainage, irrigation mode and soil hydrodynamic profile should be accounted for in a dynamic way in the inventory stage rather than in the characterization stage assuming a steady state (Fig. 3). Setting the lower boundary at the root zone limit and the temporal limit at the beginning/end

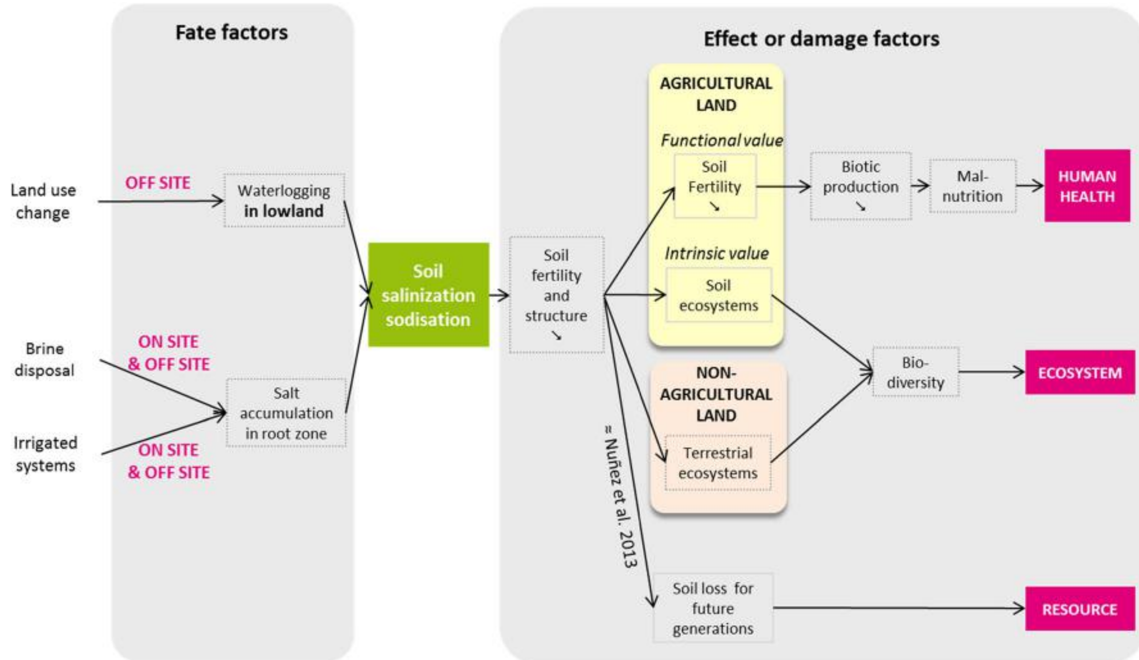

Fig. 2 Soil salinisation impacts on human health, ecosystems and resource: fate and effect factors positioning on the cause-effect chain and relations between agricultural and non-agricultural lands

of the cropping cycle is consistent with crop and water balance models.

Ecological effect factors are usually based on species sensitivity distributions (SSD), relating the concentration of a pollutant in the environment with the Potentially Affected or Disappeared Fraction of species (PAF or PDF) (Huijbregts et al. 2011). This non-linear relationship requires that the fate and exposure assessment provide background dose

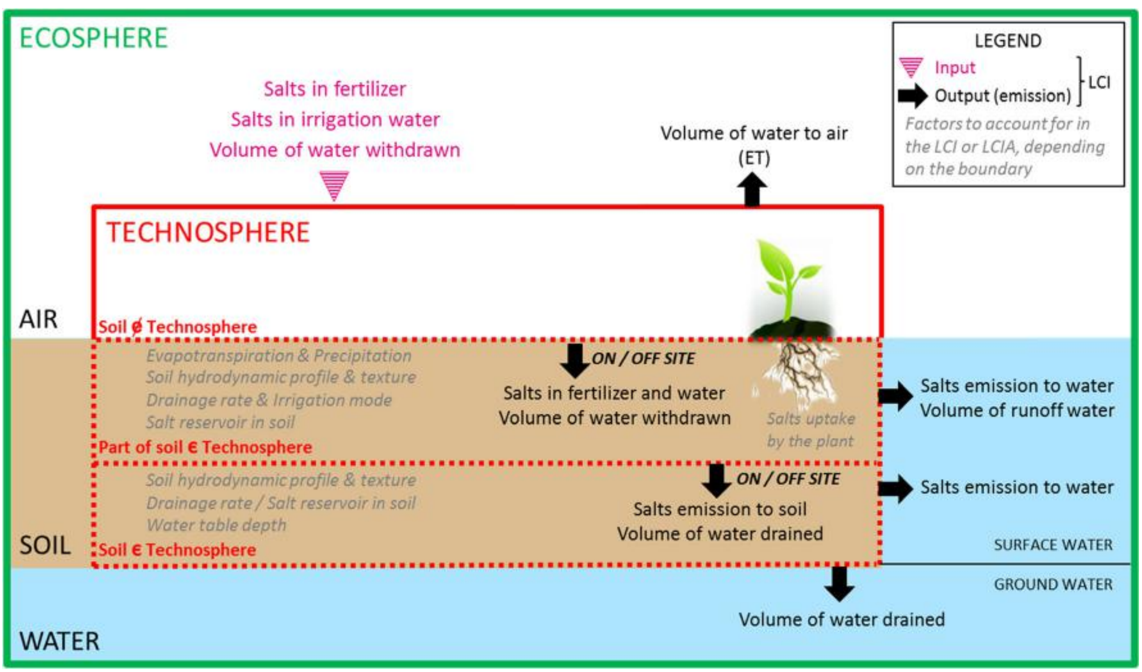

Fig. 3 Salinisation associated with irrigation and deposition of salts: technosphere and ecosphere boundary options and corresponding parameters to account for in the inventory or in the impact assessment. If soil is not included in the technosphere, the inventory requirements are the inputs to the technosphere: salts in fertilizer and in irrigation water, and volume of water withdrawn. If part or the whole soil is included in the technosphere, additional parameters (in *grey italic*) have to be accounted for at the inventory stage to calculate the output (emissions) flows, thus allowing to account for management practices

information (Udo de Haes et al. 2002), except if the effect factor is linear (e.g. Amores et al. 2013). But there is a limited updated knowledge about the background exposure levels of salinisation: although reports of secondary salinisation abound in the literature, there is a lack of recent assessment of the levels of salinisation (Flowers 1999). An open research question is whether the effect factors should be derived following a marginal approach or an average approach (Electronic Supplementary Material S.3). Salinity dose-response information abound in the literature with heterogeneous spatial coverage, e.g. freshwater fishes of south-western Australia (Beatty et al. 2011), freshwater small crustacean (Gonçalves et al. 2007), freshwater mussels in Canada (Gillis 2011) and aquatic plants in Australia (Kim et al. 2013). Thus, the development of spatially explicit effect factors on ecosystems with global coverage will be challenging and should compile all publications in the field.

Damage to ecosystem quality can be expressed as species diversity, the recommended endpoint indicator by JRC (2010). But function-related parameters, such as the biomass production of the ecosystem often estimated through the Net Primary Production (NPP), might also be good endpoint indicators (Núñez et al. 2013). However, when using a NPP-based indicator, one should specify which production of biomass is considered: either the biomass production of ecosystems referring to the AoP ecosystem or the biomass production of agricultural land referring to the AoP human health.

4.2.6 Human health

Soil salinisation reduces productivity of agricultural and pasture lands, leading to a reduction in food availability. This, in turn, can cause malnutrition and damage human health if compensation scenarios are not possible (i.e. in developing countries). The reduction in biotic production potential is due not only to ecotoxicological effects of salts on crops but also to soil physical degradation. Many studies report the salt tolerance of crops: salinity thresholds (Maas and Hoffman 1977; FAO 1985) and sodicity thresholds (Qadir et al. 2001). Soil structure alteration affects not only the biotic production potential but also freshwater regulation and erosion potential. It is noteworthy that other impact pathways exist: a review of the implications of salinity on human health shows effects on respiratory health, vector-borne disease and mental health (Jardine et al. 2007). The pathway soil salinisation damaging human health concerns impacts on agricultural land, which are on-site in the salinisation context of irrigated systems and off-site in the context of a LUC (Fig. 2). As discussed in the previous section, a proper accounting of impact on agricultural land strongly relies on the technosphere boundary: soil should therefore be included only partially in the technosphere, and only the time it is being used to support the studied function.

Water salinisation affecting human health does not refer to toxicological effects of drinking saline water. It refers to the water quality degradation making the water inappropriate for certain usages, thus corresponding to a water deficit if compensation scenarios are not possible. The modelling of water salinisation affecting human health requires a functional approach such as the one suggested by Boulay and colleagues, which assesses damages of a water functionality loss, accounting for the adaptation capacities (Boulay et al. 2011b). 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, the method proposed by Boulay et al. (2011a, 2011b) cannot be applied in its present form due to a scale modelling issue: the water salinised 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 salinisation of drainage water induced by irrigation (Fig. 4) and if the soil is included in the technosphere (at least temporarily); i.e. the saline-drained water is considered an "emission". Furthermore, the CF is based on a water stress index, not referring to any environmental process. Damage to human health can be expressed as disability-adjusted life years (DALY): the most used unit by current LCIA methods addressing damages to human health (e.g. Pfister et al. (2009); Motoshita et al. (2011); Boulay et al. (2011b)).

4.2.7 Resource

Water and soil salinisation affecting water and soil as resources are debatable pathways because the AoP resources are not always considered as an intrinsic AoP (Cf. part 4.1). However, for the sake of completeness and to put forward soil and water resource preservation stake (Renouf et al. 2014), we suggest considering that permanent water or soil quality degradation represents a damage to resource for future generations. There is no risk of double counting if one clearly defines the AoP resources as the protection of a resource (in sufficient quality and quantity) for future generations, while the AoP human health and ecosystems reflect the protection of current people and ecosystems. Permanent degradation of soil or a loss of soil through erosion reduces soil availability as a future resource (Núñez et al. 2013). 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). In particular, the soil structure alteration in case of sodisation is almost irreversible. Soil salinisation damages on resources is close to the pathway modelled by Brandão and Milà i Canals (2013) (a slightly modified version of Milà i Canals et al. (2007) recommended by ILCD (2010)),

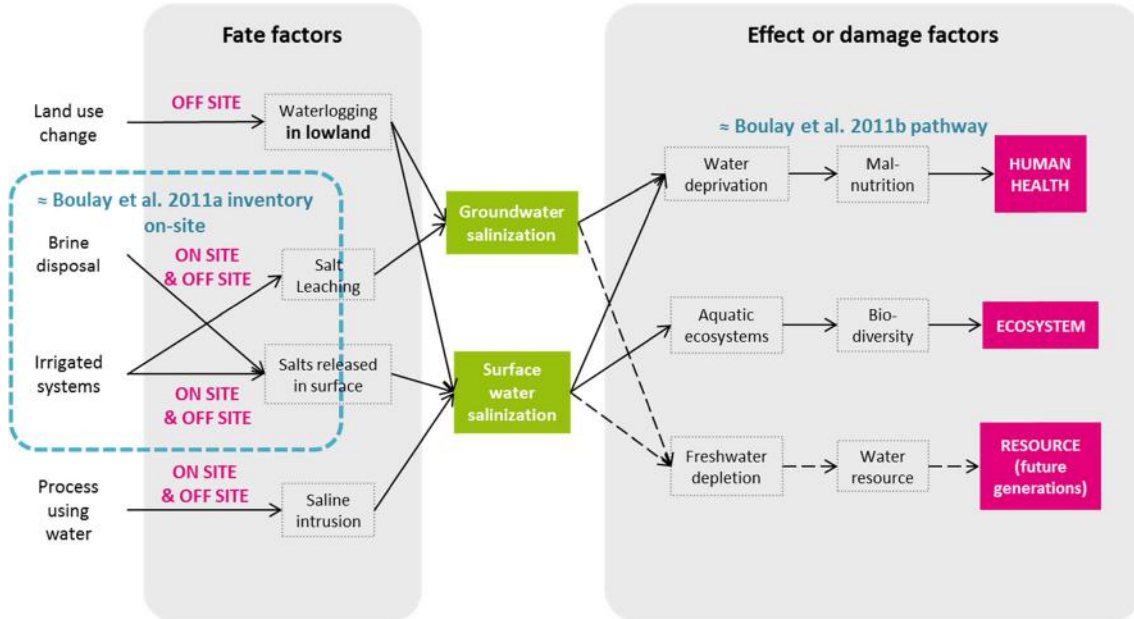

Fig. 4 Water salinisation impacts on human health, ecosystems and resource: fate and effect factor positioning on the cause-effect chain and Boulay et al. inventory and damage to human health method positioning

consisting, in a midpoint indicator, of soil quality based on soil organic carbon. This impact is damaging the AoP resources, but there is no endpoint indicator developed so far, and salinisation is not accounted for, in spite of its influence on biomass production potential. Salinisation damages on resources should rather be modelled through a framework based environmental mechanisms modelling, close to the pathway developed by Núñez et al. (2013) for soil erosion: an intermediate between LULUC and an approach based on environmental mechanisms, involving damage and effect factors (Fig. 2). Water quality alteration affecting resources is not considered in the water use impact framework (Kounina et al. 2013), although a quality alteration is affecting the availability of these resources for further uses. 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 (Cf. Fig. 1 in Kounina et al. 2013), although a permanently degraded freshwater represents a loss of water resource for future generations. For example, in the case of permanently saline aquifers, we can consider that future generations will be deprived of water in that specific location. It is therefore paramount to address this question in future research. Damage to resources can be expressed as energy needed to make the resource available in the future: in energy units (megajoule equivalents), such as in EcoIndicator99, or emergy units (megajoule solar equivalents), such as in Núñez et al. (2013) for soil depletion. Other approaches express damage to resource in monetary equivalent, such as in ReCiPe, but this unit is confronted to the cost variability of a technology.

4.3 Toward operationalisation

The recommendations provided in this article are mostly conceptual. The aim is to build a consistent and comprehensive framework which is not available through the existing methods addressing salinisation impacts. The need of a common framework regarding the technosphere and ecosphere boundary, the status of the AoP and the modelling approach (top-down vs. bottom up) are paramount. Our recommendations aim at overcoming limitations that existing methods where confronted with: regarding the need for a global coverage (the characterisation model should be applicable globally and to not miss any important parameter involved) and regarding the need for accounting for all potential impact pathways (without gaps or overlapping). That is why we recommend starting from an understanding of the environmental mechanisms, driven by the water cycle, at a scale going beyond the plot. Such a bottom-up and mechanistic (i.e. describing environmental mechanisms) approach is the best way to discriminate the studied systems, therefore allowing eco-conception, one of the core application of LCA. Nevertheless, in the operationalisation process, we will have to cope with the lack of globally available data for the development of characterisation models. Indeed, spatial explicit impact assessment requires the use of local, regional or country-specific information. This concerns all impact categories. The lack of data available with a global coverage may hamper the operationalisation of a method, both for the inventory and the impact assessment. Thus, method developer should provide geo-referenced databases for inventory parameters (according to the technosphere boundary) and background system assessment and build geo-referenced CF. Among the parameters involved in salinisation impact assessment (Table 1), reliable geological and groundwater level data are lacking (Zhou et al. 2013a). Generally, data availability and accuracy vary according to the country. But the development of geo-referenced databases is significant, notably thanks to remote sensing data acquisition (Bastiaanssen et al. 2007). Thus, we can reasonably think that in the medium term these objectives of global modelling are reachable. As a next step of this article, on-going work is developing an inventory tool operational for agricultural systems: water and salt flow model, compliant with the recommendations provided in this article.

5 Conclusions

Including salinisation impacts in LCA is of high priority. Assessing salinisation impact is particularly relevant in food LCA, because agricultural systems are both the main affected targets and causes of salinisation, but not exclusively: water body overexploitation, brine disposal or a land use change are also major contributors to salinisation worldwide.

Although the existing methods addressing salinisation in LCA are important and relevant contributions, they are incomplete in terms of spatial and environmental mechanisms coverage. The modelling complexities lie in the diversity of salinisation 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 modelling options in agreement with the LCA framework has been proposed in this paper. We identified and categorized the key biophysical and management factors involved for each salinisation types and discussed the inventory and impact assessment boundary options. The land use framework might be suitable to partially model salinisation impacts from a LUC but should be completed with a mechanistic approach (based on environmental mechanisms modelling) to account for off-site impacts. An approach modelling environmental mechanisms (i.e. based on fate, exposure and effect factors) should also be preferred to model salinisation related with irrigation, brine disposal and waterbody overexploitation. For all salinisation pathways, a bottom-up approach (rather than empirical or top-down approach) is recommended because of the following: (i) salts and water are mobile and theirs effects are interconnected; (ii) this is in line with the ISO norm stating that one shall relate a consequence with a cause (i.e. model environmental mechanisms); (iii) this approach allows the evaluation of both on- and off-site impacts; and (iv) it is the best way to discriminate systems in which the agricultural stage is predominant and supports a reliable eco-design which is the core aim of LCA. Regarding the boundary between the technosphere and ecosphere (i.e. inventory and impact assessment), we recommend to consider the part of the soil that is influenced by farmer practices in the technosphere and only during the time it is being used by the system, because it allows discriminating the agricultural practices and their effects (including impacts on agricultural land) and it is consistent with many crop and water balance models (that we recommend to use within a fate modelling).

By discussing paramount methodological issues, this paper provides the basis for future method developments and shows that much research effort is still required to include salinisation impacts in a global, consistent and operational manner in LCA. Next steps include the testing of fate and exposure models, fed by global databases, with the background issue to find a tradeoff between accuracy and feasibility. To do so, it is important that LCIA scientists join their efforts together with salinisation experts to build a consensual model (Huijbregts 2013).

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