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Review

The threat of soil salinity: A European scale review

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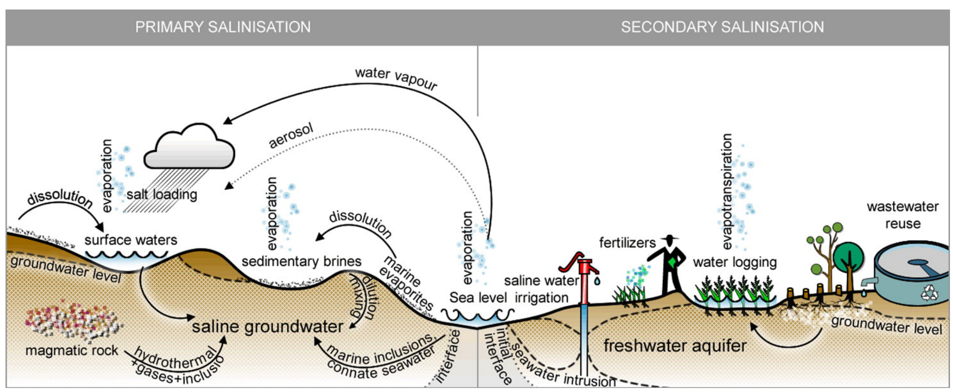
HIGHLIGHTS

- State of the art regarding drivers, effects, indicators, monitoring, modeling and management of soil salinity at European scale is presented.
- Current state of soil salinity in Europe is introduced by compiling a variety of sources.
- Knowledge gaps and aspects beyond the state of the art regarding the soil threat of salinisation are highlighted.

GRAPHICAL ABSTRACT



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ABSTRACT

Soil salinisation is one of the major soil degradation threats occurring in Europe. The effects of salinisation can be observed in numerous vital ecological and non-ecological soil functions. Drivers of salinisation can be detected both in the natural and man-made environment, with climate and the foreseen climate change also playing an important role. This review outlines the state of the art concerning drivers and pressures, key indicators as well as monitoring, modeling and mapping methods for soil salinity. Furthermore, an overview of the effect of salinisation on soil functions and the respective mechanism is presented. Finally, the state of salinisation in Europe is presented according to the most recent literature and a synthesis of consistent datasets. We conclude that future research in the field of soil salinisation should be focused on among others carbon dynamics of saline soil, further exploration of remote sensing of soil properties and the harmonization and enrichment of soil salinity maps across Europe within a general context of a soil threat monitoring system to support policies and strategies for the protection of European soils.

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1. Introduction

Soil salinisation is a term that includes saline, sodic and alkaline soils (van Beek and Tóth, 2012), respectively defined as (a) high salt concentration, (b) high sodium cation (Na^+) concentration, and (c) high pH, often due to high CO_3^{2-} concentration, in the soil. Soil salinisation leads to the alteration or even disruption of the natural biological (Smith et al., 2015), biochemical (Decock et al., 2015), hydrological (Keesstra et al., 2012) and erosional (Berendse et al., 2015) Earth Cycles. High salinisation levels can thus result to the loss of the emerging resources, goods and services of soil, impacting agricultural production and environmental health (Rengasamy, 2006), eventually evolving into a sociocultural and human health issue (Brevik et al., 2015) that hinders economic and general welfare.

Soil salinisation is a widespread phenomenon, with saline and sodic soils covering 932.2 Mha globally (Rengasamy, 2006), and one of the major soil degradation threats worldwide, with mismanaged irrigation affecting 34.19 Mha (Mateo-Sagasta and Burke, 2011) or over 10% of the total irrigated land (Aquastat, 2016). Europe contributes about 30.7 Mha or 3.3% of the global saline and sodic soils (Rengasamy, 2006). Global soil salinisation hotspots include Pakistan, China, United States, India, Argentina, Sudan and many countries in Central and Western Asia (Aquastat, 2016; Ghassemi et al., 1995), while at European scale the Mediterranean coastline stands out (Geeson et al., 2003). Effectively, this soil threat has gained worldwide attention in the State of the Art, as concern has grown about irrigation mismanagement (Young et al., 2015), organic (Drake et al., 2016; Singh et al., 2016; Srivastava et al., 2016; Wu et al., 2014) and inorganic amendment selection and quantification (Ahmad et al., 2016; Mao et al., 2014), and the role of plant tolerance (Singh et al., 2015) and soil fauna (Oo et al., 2015) in the adaptation and soil reclamation process.

A wide range of traditional and state-of-the-art amelioration methodologies against soil salinisation has been documented (Panagea et al., 2016), nevertheless, they can be very case specific. In order for reclamation studies to be efficiently upscaled or effectively adapted to local problems, a review of the state of soil salinity in Europe is essential. The objective of this review is to show the State of the Art on soil salinisation in Europe based on scientific publications and reports. Each chapter describes the State of the Art at global scale and concludes with findings and discussion at the European level.

2. Drivers and types of salinisation

2.1. Primary salinity

Primary salinisation is the development of salts through natural processes, mainly including physical or chemical weathering and transport from parent material, geological deposits or groundwater (Fig. 1). Soil may be rich in salts due to parent rock constituents such as carbonate minerals and/or feldspar. Closely related to this, geological events or specific formations can increase salt concentration in groundwater and therefore in superimposed soil layers. This can occur when, after capillary effects or evapotranspiration cause salinity affected groundwater to rise, previously dissolved salts accumulate at or near the surface (Chari et al., 2012; Geeson et al., 2003). These drivers affect the soil depending on aquifer architecture and hydraulic conductivity of geological layers and soil characteristics such as porosity, structure and texture, clay mineral composition; compaction rate, infiltration rate, water storage capacity, saturated and unsaturated hydraulic conductivity and finally potential salt content (Chesworth, 2008; van Beek and Tóth, 2012). In total, the types of saline or saline prone soil formed as listed by WRB (2014) are shown in Table 1. Naturally saline soils occur in Spain, Hungary, Slovakia, Greece, Austria, Bosnia, Serbia, Croatia, Romania, Bulgaria, Ukraine and the Caspian Basin (Geeson et al., 2003; Jones et al., 2008; Trnka et al., 2013; van Beek and Tóth, 2012; van Camp et al., 2004).

Apart from the long-term accumulation of salts in the soil profile, natural soil salinisation can also pre-exist due to once submerged soils under seawater. During this period, seawater fills the voids of the sediments (connate water, e.g. Edmunds et al. (1987)) and remains enclosed within the marine deposits (Wendland et al., 2008), even after the seawater incursion. Besides historical marine waters, contemporary sea level rises may cause seawater to flood coastal land, either for long (marine transgressions) or short (storm flood events, tsunamis) periods. In addition, these rises may boost lateral seawater intrusion into coastal areas that are hydraulically connected to the sea, causing wide-spread soil salinity problems across regions near the coast, as observed in Western Netherlands, Denmark, Belgium, North-eastern France, and South-eastern England (Raats, 2014; Trnka et al., 2013; van Weert et al., 2009).

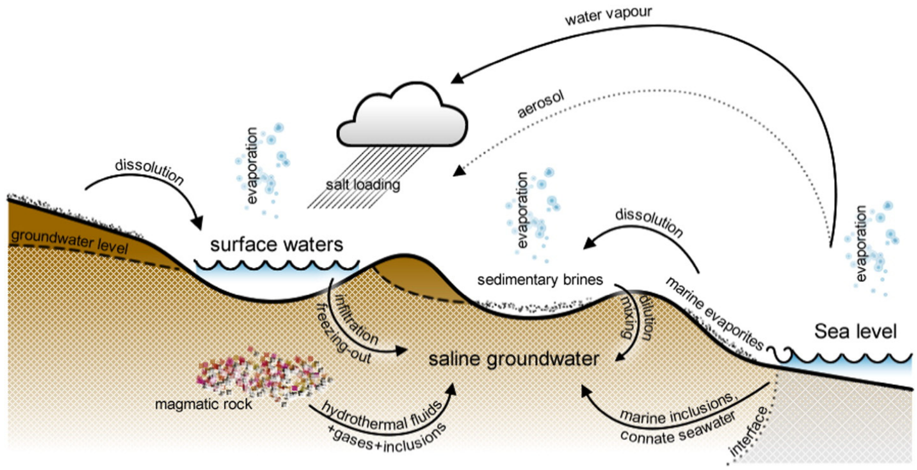

Fig. 1. Primary soil salinity mechanisms.

2.2. Secondary salinity

Contrary to primary salinisation, secondary salinisation is introduced by human interventions; mainly irrigation with saline water or other ill-suited irrigation practices often coupled with poor drainage conditions (Fan et al., 2012; Trnka et al., 2013) (Fig. 2). With a climate predominated by little rainfall and adverse evapotranspiration rates, and soil characteristics that restrain salt leaching, arid irrigated lands are prominent salinisation hotspots. While constant or increasing salt accumulation in the upper soil layers is primarily the result of irrigation sourced from highly saline water such as seawater contaminated groundwater, moderate problems are observed even when sufficient quality water is used (Dubois et al., 2011; Geeson et al., 2003; Mateo-Sagasta and Burke, 2011; Tóth and Li, 2013; van Camp et al., 2004). As such, salinisation it is a major factor limiting crop production and land development in arid coastal areas (Li et al., 2012; Sparks, 2003).

Interventions that increase time of ponding or limit sufficient drainage can also lead to salinisation. An increased water table lever due to filtration from unlined canals, reservoirs and waterlogging (Barros et al., 2012), uneven distribution of irrigation water, land clearing, and improper drainage may mobilise salts that have accumulated in the soil layers (Chesworth, 2008; Eckelmann et al., 2006). Salty groundwater may reach the upper soil layers and, thus, supply salts to the root zone. Additional hurdles to good drainage may be posed by coastal protection infrastructure aiming to reducing seawater encroachment into the aquifers but ergo blocking natural drains of rich in salts discharges. In arid regions, poorly drained soils, also allow for too much evaporation leading to salt residuals on the soil surface (Mateo-Sagasta and Burke, 2011; van Beek and Tóth, 2012).

Salinisation origins can also be relevant to soil pollution. The use of fertilizers (Moreira Barradas et al., 2014) and other inputs in association with irrigation and insufficient drainage cause soil salinisation, markedly in cases of intensive agriculture in compacted and limited leaching soils (Eckelmann et al., 2006). Wastewater treatment (Moral et al., 2008), industrial (Lefebvre and Moletta, 2006) or mining operation effluents are often rich in salts, therefore their mismanaged subsurface injection, surface disposal or use for irrigation, can also lead to soil salinisation. Finally, the use of traditional salt based de-icing agents in excess contributes to the accumulation of salt in the soil and water (Mateo-Sagasta and Burke, 2011).

According to Stanners and Bourdeau (1995), secondary salinisation affects around 3.8 Mha in Europe. Using expert judgement, van Camp et al. (2004) assessed that approximately 4 Mha of European soils have a moderate to high level of degradation due to secondary salinisation. Artificially induced salinisation is affecting significant parts of Italy (e.g. Campania and Sicily), Spain (e.g. the Ebro Valley), Hungary (e.g. Great Alfold), Greece, Cyprus, Portugal, France (West coast), the Dalmatian coast of the Balkans, Slovakia and Romania. In addition, North Europe countries (e.g. Denmark, Poland, Latvia, and Estonia) are facing similar issues. Road and bridge snow and ice control in Europe contribute 20 to 25 Mt of de-icing salt per year (Houska, 2007). Soil salinity is a major cause of desertification along the Mediterranean coast, mainly due to human activities (Abu Hammad and Tumeizi, 2012; Domínguez-Beisiegel et al., 2013), especially with the extension of irrigation and undisciplined use of saline water which has caused over-pumping, and the consequent sea-water infiltration into the groundwater layer. In the Mediterranean region, soil salinisation affects 25% of irrigated agricultural land at a significant level (Geeson et al., 2003; Mateo-Sagasta and Burke, 2011). For example, about 3% of the 3.5 Mha of irrigated land in Spain now has a significantly reduced agricultural potential due to soil salinity, with another 15% facing the same risk. Also, about 9% of the 1.4 Mha of irrigated land in Greece is affected by soil salinisation due to seawater intrusion (Jones et al., 2003;

Table 1
Types of saline or saline prone soil formed as listed by WRB (2014).

Soil types	Main characteristics	Symbol	Saline	Saline Prone
Solonetz	Subsurface clay accumulation, rich in sodium	SN	×	
Solonchak	Strongly saline	SC	×	
Acrisol	Subsurface accumulation of low-activity clays and low base saturation	AC		×
Alisol	Subsurface accumulation of high-activity clays, rich in exchangeable aluminium	AL		×
Calcisol	Accumulation of secondary calcium carbonates	CL		×
Fluvisol	Relatively young in alluvial deposits	FL		×
Gleysol	Permanent or temporary wetness near the surface	GL		×
Luvisol	Subsurface accumulation of high-activity clays	LV		×
Vertisol	Dark-coloured cracking and swelling clays	VR		×

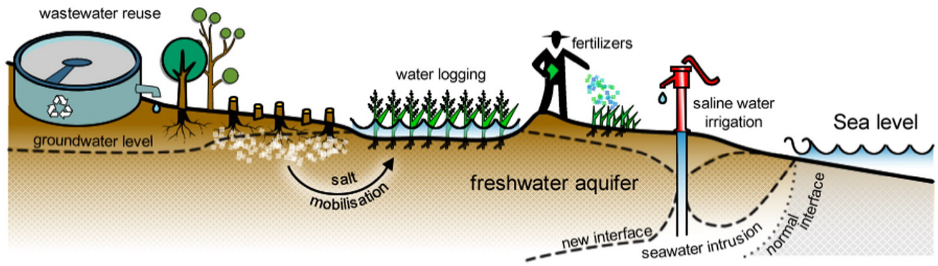

Fig. 2. Secondary soil salinity mechanisms.

OECD, 2009). Fig. 3 depicts the locations of saltwater intrusion, compiled from (Daskalaki and Voudouris, 2008; EEA, 1999). In addition to seawater intrusion, in several areas like Cyprus, the excess use of fertilizers and municipal wastewater has contributed to the soil salinity (FAO, 2011; Huber et al., 2008; Mateo-Sagasta and Burke, 2011).

2.3. Climate and climate change

A future warmer climate will cause variations in the hydrological cycle (Sterling et al., 2012; Vautard et al., 2014) and rising sea levels (Hinkel et al., 2014), and in turn will significantly increase soil salinity resulting in the expansion of areas affected by this form of the problem.

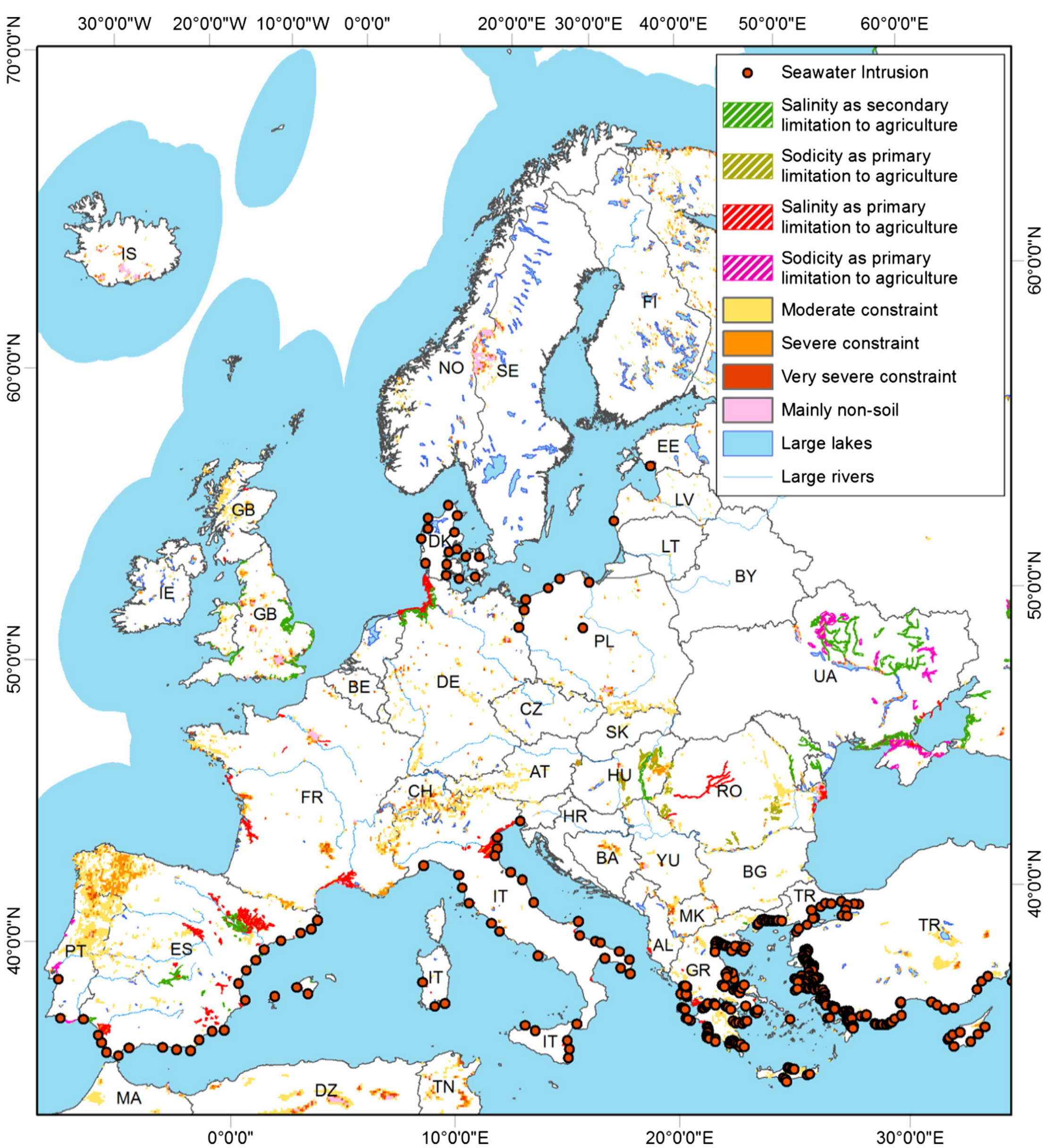

Fig. 3. Saline (EC > 4 dS m^{-1} within 100 cm of the soil surface) and sodic (ESP > 6% within 100 cm of the soil surface) soils as agricultural constraint and as primary and secondary limitations to agricultural use, and areas of seawater intrusion in Europe. Compiled from SGDBE, EEA (1999), Daskalaki and Voudouris (2008) and Fischer et al. (2008).

Sea level rise in combination with groundwater overexploitation is expected to intensify the saltwater intrusion in coastal (Taylor et al., 2012) and inland aquifers from neighbouring saline aquifers (Chen et al., 2004). Increase in evapotranspiration is likely to increase salinisation of shallow groundwater especially in semi-arid and arid areas (Bates et al., 2008). Globally, there is a high potential of "salinity refugees" in vulnerable areas due to climate change, sea level rise and coastal land subsidence (Hallegatte et al., 2013), that can potentially affect European coasts and islands. Especially areas with little rainfall and high evapotranspiration such as the Mediterranean (Koutroulis et al., 2013) will face a reduction of the extent of their water courses and a transition to a more arid environment (Koutroulis et al., 2011; Vrochidou et al., 2013). Irrigation water requirements are generally projected to escalate with higher global mean temperature (Haddeland et al., 2014) and are likely to have an even higher salt content, due to solute concentration following evaporation. An intensified hydrological cycle may also trigger an increase of floods and flash floods (Daskalaki and Voudouris, 2008; Hallegatte et al., 2013), thus causing an increased transfer of dissolved salts to the soil in areas with geological substrates prone to releasing large amounts of salts (Mateo-Sagasta and Burke, 2011; Trnka et al., 2013; van Weert et al., 2009).

The research by Szabolcs (1990) was one of the first studying the implications of impact of climate induced (i) temperature increase, (ii) sea level rise and (iii) irrigation water shortage on the salinisation of European soils. Selecting three different regions that cover a large part of Europe, Szabolcs examined the influence of several major processes under a different future climate and indicated that the extent of salinity could potentially be doubled. In this context, human induced salinity in combination with climate constitutes a potential hazard much greater than primary salinity. Várallyay (1994) summarized the principal preconditions for salt accumulation along with the most important potential repercussions of a changing climate for selected climatic scenarios. The quantification of the effect of climate change on salinisation extend is rather a difficult task (Yeo, 1998) that needs complex approaches in order to achieve predictable responses of the soil processes related to the threat (Schofield and Kirkby, 2003). Table 2 includes a number of the few studies dealing with the potential implications of a changing climate on the threat of soil salinity of pan-European or regional focus.

3. Mechanisms and impacts

3.1. Physicochemical properties

In sodic soils, high concentration of Na^{+} displace cations such as Ca^{2+} and Mg^{2+} and persist bound to clay particles causing significant structural degradation. As exchangeable sodium is hydrolysed to a greater extent, bonds between soil particles weaken, swelling and often becoming detached, resulting in increased dispersibility and susceptibility to erosion by water and wind (Paix et al., 2013). On drying, sodic soils become dense, cloddy and structureless because natural aggregation is destroyed. At the soil surface, dispersed clay can act as adhesive, forming relatively dense crusts that impede seedling rooting and emergence. The degree of crusting depends on soil texture, the mineralogy of the clay, the exchangeable sodium content, the energy of raindrop impact, and the rate of drying. Soils with high montmorillonite clay contents will crack on drying. Moreover, the genesis of some soils has resulted in sodic subsoils, often with a columnar structure. Sodic subsoils may be dense, with reduced water storage capacity, poor aeration and increased shear strength, and can be susceptible to tunnel erosion. In this way, small clay particles move through the soil due to their small size, eventually clogging pore spaces thus reducing hydraulic conductivity (DNR, 1997).

3.2. Ecological functions

Soil salinisation primarily affects ecological soil functions. The adverse impact of increased *EC* on important soil processes such as respiration, residue decomposition, nitrification, and denitrification through the decrease of soil biodiversity and microorganism activity is well known (Singh, 2015). Depending on its form and stage, soil salinisation degrades fertile and productive land to the degree of eliminating all vegetation (Chesworth, 2008; Jones et al., 2012; Trnka et al., 2013). Saline soils enter a negative feedback of Soil Organic Carbon (SOC) loss as decreased fertility and microbial and enzyme activities (Singh, 2015) lead to less biomass production which adversely affects distribution and stability of soil aggregates (Six et al., 2000) and promotes a higher fraction of plant input in the accumulated organic matter. These changes increase dispersion of clay particles and higher wind and water erosion rates (Paix et al., 2013) that further aggravate SOC losses. Regarding C accounting, there is currently limited evidence on the C stock trends in salt affected soil, whereas C dynamics studies are contradictory (Wong et al., 2010). Besides ecological limitations posed by secondary salinity, naturally saline soils are unique gene reservoirs (Zechmeister, 2005) that deserve protection for this very value. As a feedback of its effects on ecological functions, salinisation impacts a series of environmental interactions thus undermining water infiltration and soil water storage capacity resulting in increased water runoff and erosion.

Table 2
List of studies with climatic change projections, related implications and threats.

Region	Projected driving change	Resulting threat	Reference
Central and eastern Europe	Reduced groundwater recharge	Soil salinisation in marginal areas	Falloon and Betts (2010)
Pan European, mainly North, Central and Eastern Europe	Increased risk of flood and flash flood hazards – heavy precipitation	Salinisation due to increased water loss past crop root zone	Falloon and Betts (2010)
Greece	Increased temperature and evapotranspiration in combination with reduced groundwater recharge	Intensification of salinisation due increased evapotranspiration under brackish water irrigation	Daliakopoulos et al. (2016)
Netherlands	Sea level rise - land subsidence - changes in recharge	Intensification of salinisation of shallow groundwater and surface waters	Oude Essink et al. (2010)
France	Sea level rise – reduced discharge	Inland displacement of the salinity front affecting lagoon ecology and agriculture.	Paskoff (2004)
Spain	Reduced precipitation and river flow	Salinisation due to sea water intrusion	Martín-Rosales et al. (2007)
Italy	Shifting of water availability as a result of change in precipitation and evapotranspiration	Change in the seasonal pattern of rainfall-induced leaching of the salt accumulated in the soil due to irrigation during the cultivation season.	Zanchi and Cecchi (2010)
Italy	Increase in evapotranspiration - increase in the frequency of extreme high rainfall events - extreme drought conditions	Increase in aquifer salinity due to the upward movement of hyper-saline groundwater during the cold season	Colombani et al. (2015)

3.3. Effects on vegetation

The mechanisms of the toxic effects of salinity to vegetation growth can be described by various theories including osmotic inhibition (Koorevaar et al., 1983), plant mineral nutrition imbalance (Verbruggen and Hermans, 2013), saline ion toxic action (Munns, 2005, 2002), and nitrogen metabolism impediment theory (Lovatt, 1986). Relevant studies have demonstrated that saline ion concentrations in soil can result in physiological hydropenia phenomenon, reduced nutrient absorption, plant dysplasia, output reduction, and death (Bernstein, 1963). Na^{+} and Mg^{2+} can destroy cell morphology, restrain plant photosynthesis, and reduce chlorophyll production. In addition, soil saline ions can produce some toxic intermediates in the process of nitrogen metabolism, which may hinder metabolic processes (Epstein, 1980). For alkaline soils, toxicity and deficiency reactions due to plant altered element availability are also the major problems.

Among the highest production cereals in Europe (EC, 2016), wheat, barley, triticale and rye are salt tolerant whereas maize is sensitive. Other crops, commonly cultivated in Europe, including sunflower, potato, and sugar beet are considered tolerant (>3 dS m^{-1}) or moderately salt tolerant (2–3 dS m^{-1}) and maintain or slightly ameliorate their yield with an increase of soil salinity (Higton and Akeroyd, 1991; Katerji et al., 2008, 2003). Cotton, that has a strong economic importance for Spain and Greece (EC, 2016) is also salt tolerant. Olive tree and grapes, as well as mango grown in the few coastal subtropical areas of Europe can be considered as moderately salt tolerant (Chartzoulakis, 2005; Paranychianakis and Chartzoulakis, 2005; Zuazo et al., 2004). Also, the remarkably saline- and drought- tolerant caper plant, native to the Mediterranean region, has been gaining agricultural interest (Rhizopoulou and Psaras, 2003). Vegetable crops demonstrate higher sensitivity to soil salinisation than grains and forages with the notable exceptions of asparagus, artichoke, red beet, and zucchini squash (Shannon, 1997). Most fruit trees, such as stone fruits, citrus, and avocado, have shown specific sensitivity to foliar accumulations of sodium chloride from saline soils, which hinders tree growth and fruit yield.

Crops may demonstrate different salt tolerance depending on soil properties, types of rhizobacteria, growth stage, and agronomical practices including salt-resistant rootstocks. Furthermore, various plant adaptive responses to salinity stress at molecular, cellular, metabolic, and plant physiology levels have been identified (Gupta and Huang, 2014), and the selection of salt-tolerant species, salt-tolerant genotypes and symbiotic biological agents are currently in the focus of international research projects to reduce yield losses under saline conditions (Cabot et al., 2014; Koubouris et al., 2015; Roy et al., 2014). For example, Negrão et al. (2013) investigated salt-related genes in rice in order to take advantage of genetic variations and improve salinity tolerance of this this major food crop. Finally, Navarro et al. (2014) investigated the salinity resistant effects of arbuscular mycorrhizal fungi in citrus, and Wagner et al. (2016) quantified the benefits of trichoderma fungi on salt-stressed tomatoes, which are two of the most dominant fruit crops in the Mediterranean region.

3.4. Non-ecological soil function

Additional non-ecological soil function perturbation include damages to water supply infrastructure as well as transport infrastructure from shallow saline groundwater (Montanarella, 2007) thus hindering the functions of soil a physical medium for build development. As a general result, land value depreciates, with some studies estimating agricultural land depreciation at 50%, and supply of raw material such as sand, gravel and peat being hindered (Schiefer et al., 2016; Tóth et al., 2008). Significant damages to water supply and infrastructures from salinisation have occurred in Europe, especially in Hungary and Spain, with annual costs reaching 18.23 M€ and 12.08 M€ respectively (Aquastat, 2016). Finally, cultural value is also affected thus impacting tourism and inhabitant livelihood. Reports of positive effects of saline soil are sporadic and are mostly relevant to herbivore diet (Kreulen, 1985) and fruit taste enhancement (Cuartero and Fernández-Muñoz, 1998) which may not have strictly ecological character.

3.5. Cost of soil salinisation

Though difficult to estimate, studies in 3 countries (Spain, Hungary and Bulgaria) have demonstrated annual costs of soil salinisation due to mainly agricultural yield losses, but also damages to infrastructure and the environment in the range of 158–321 M€ (Montanarella, 2007). A more recent study (Bosello et al., 2012; Richards and Nicholls, 2009) focused on selected rivers and deltas, estimates that the current EC economic impact exclusively in agriculture due salinity is in the area of 600 M€ (mostly borne by Germany, the Netherlands and France), assuming that saline agricultural land is half as valuable as is non-saline land.

4. Key indicators

4.1. Field symptoms

Field symptoms such as the poor condition or absence of vegetation, areas that take longer to dry or the presence of unnatural colour soil crusts or stains (white or dark) can reveal salinity in affected areas visually. Nevertheless, some symptoms are not always related to soil salinity (Lin and Bañuelos, 2015), as in the case of high soil pH (Table 6). Poor vegetation response to nutrients and water, recession or extinction of floral sensitive bioindicators or the presence of salt-tolerant plants, also mentioned as floral accumulative bioindicators (Pandolfini et al., 1997; Rana and Parkash, 1987) can also serve as a field indicator. At a European level, a set of indicator values developed by Ellenberg (1950) (EIV) and undated several times (Ellenberg et al., 2001) include different levels of salt tolerance per plant species. The system has been widely used and it has been reportedly adapted to a number of central European countries (Kollmann and Fischer, 2003), The Netherlands (Ertsen et al., 1998) and the UK (Hill et al., 2000). However, Godefroid and Dana (2006) explored the relationships between EIVs for Greece and Italy and showed that they should not be used outside the region for which they are defined. Besides EIV, several European studies have used statistical methods to classify floral bioindicators relevant to soil salinity at the local scale (González-Alcaraz et al., 2014; Piernik, 2003). The richness of potential bioindicators extends further, including parameters such as microbial biomass and/or respiration, microflora and microfauna composition and abundance, pathogens, enzymes (Pankhurst et al., 1997). Nonetheless, their efficiency may be limited due to the contradictory results found in the literature (Wong et al., 2010) and our limited understanding of soil ecosystem composition and dynamics in relation to salinity and salinity gradients (Rath and Rousk, 2015). Due to these uncertainties, further investigation is typically carried out to confirm soil salinity evidence, mainly with the combination of physical and chemical indicators. In that direction, Huber et al. (2008) documented a list of soil salinisation indicators applicable at European scale:

4.2. Electrical conductivity of solution (EC)

Electrical conductivity (EC) is typically measured in deci-Siemens per meter (dS m^{-1}) at 25 °C to avoid the influence of temperature, and determines the concentration of all the soluble salts in soil or water. At the usual environmental sampling ranges temperature can also be compensated linearly (Hayashi, 2004). EC is desirably measured at field capacity (EC_f) from soil samples as this provides the soil's true salt concentration. However, salinity can be also estimated in a standard saturation extract (EC_e) obtained by adding water to a dry soil. The relationship between EC_f and standard saturation extract EC_e can be

Table 3
Classification of electrical conductivity with regard to salinity effects on crops (Richards, 1954).

EC_e [dS m^{-1}]	Class	Effect
0–2	Non saline	Negligible
2–4	Slightly saline	Yield reduction of sensitive crops
4–8	Moderately saline	Yield reduction of many crops
8–16	Strongly saline	Normal yields for salt tolerance crops only
>16	Very strongly saline	Reasonable crop yield for very tolerance crops only

obtained from soil texture using tables (Whitney, 2012) or regression equations (Sonmez et al., 2008). The prerequisite is typically, a 1:n solution (1 part soil n parts distilled water) preparation from field soil samples using standard procedures (He et al., 2012). The derived EC_e can be used to compare across different soils and is classified depending on the salinity hazard and its effects on the yield of field crops according to the general scheme of (Richards, 1954) presented in Table 3.

4.3. Sodium adsorption ratio (SAR) and exchangeable sodium percentage (ESP)

Each soil type is characterized by its ability to adsorb the positively charged components of dissolved salts (e.g. Na^+, Ca^{2+}, Mg^{2+}, etc.). The process involves the exchange of cations at an extent that depends mainly on (a) the relative concentrations in the soil solution, (b) the energy level and size of the cation involved, and (c) the nature and amounts of other cations present in the solution or the exchange complex. Sodicity expresses the Na^+ ions in soil or water compared to Ca^{2+} and Mg^{2+} ions. It is expressed by means of sodium adsorption ratio (SAR) or by means of exchangeable sodium percentage (ESP). SAR is a measure of both, water suitability for use in agricultural irrigation and a measure of the soil sodicity. The former is determined based on the concentrations of dissolved solids in the water and the latter from analysis of water extracted from the soil (Shahid et al., 2013; van Beek and Tóth, 2012), in terms of:

$$SAR = \frac{Na^+}{\sqrt{\left(Ca^{2+} + Mg^{2+}\right)/2}} \quad (1)$$

where the concentrations of Na^+, Ca^{2+} and Mg^{2+} are in milliequivalents per litre (meq L^{-1}) in soil extract from saturated paste, and SAR is expressed as (mmoles $L^{-1})^{0.5}$. In case of high carbonate (CO_3^{-2}) or/and bicarbonate (HCO_3^-) concentration in irrigation water, an "adjusted" SAR_{ADJ} may be calculated to account for the calcium carbonate ($CaCO_3$) or magnesium carbonate ($MgCO_3$) that will precipitate in the solid phase (Bauder et al., 2011; Lesch and Suarez, 2009). ESP, on the other hand, is defined as the amount (percent of the cation exchange capacity in milliequivalent per 100 g of soil) of adsorbed Na^+ on the soil exchange complex. ESP is calculated using the relationship (Shahid et al., 2013):

$$ESP = \frac{\text{Exchangeable Na (me/100g soil)}}{\text{Cation exchange capacity(me/100g soil)}} \times 100 \quad (2)$$

If the SAR of the soil equals or is >13 (mmoles $L^{-1})^{0.5}$ or ESP equals or is >15, the soil is termed sodic (Richards, 1954).

4.4. Total dissolved solids, (TDS)

TDS is a measure of the total ionic concentration of dissolved minerals in water. This indicator is directly related to EC. An important classification of EC and TDS is that of USDA Salinity Laboratory (Richards, 1954), that is still commonly used (Table 4).

pH: pH is an indicator of the acidity or alkalinity of the soil. Specifically, if pH is >8.5 the soil is more likely to be saline – alkaline. Table 5 represents salinity/alkalinity/sodicity classification schemes for the above commonly used indicators.

Salt profile: Salinity does not always affect the entire soil profile; therefore a salt profile can provide a complete representation of the vertical salt distribution in terms of content, composition (Huber et al., 2008). As soluble salts are more mobile than carbonates, the soil profile can provide insight to solute movement and salt accumulation processes (Lin and Bañuelos, 2015) in a single instance in time.

4.5. Remote sensing indices

Under salinity stress, plant health is hindered showing symptoms similar to that of water deficit (Hamzeh et al., 2013). Remote sensing indices pertinent to vegetation health such as the reflectance of Near InfraRed (NIR) or water stress such as the water absorption bands in the Short-Wave InfraRed (SWIR) have been used as a proxy for soil salinity estimation in agricultural fields both outside (Poss et al., 2006; Zhang et al., 2011) and in Europe (Ceccato et al., 2001; Leone et al., 2007). High salt concentrations can also be inferred through the detection of spectral signatures of salt resistant plants (e.g. halophytes), distinguishable growth patterns or by the salt efflorescence and crust that may be present on bare soil. Similar to vegetation indices, a range of salinity indices exist (Allbed and Kumar, 2013; Hamzeh et al., 2013) for detecting and mapping soil salinity from multispectral (low cost or free) and hyperspectral (higher resolution information) satellite sensors (Dehaan and Taylor, 2002). Nevertheless, surface reflectance is largely influenced by soil moisture content, salt content, colour and roughness. Since these indices have not yielded consistent results, selecting a single index may not be suitable for all cases. Drawing from the above, a no-regrets and robust index seems to be the Normalized Difference Vegetation Index (NDVI) that can quickly be used to assess vegetation health spatial patters. Based on this criterion, statistical methods such as principal component analysis (PCA) can be used to correlate soil properties and different indices (Levi and Rasmussen, 2014). In addition, (Ivits et al., 2013) used remote sensing derived productivity indicators to characterise productivity limitation of salt-affected soils in different regions of Europe.

5. Monitoring, mapping

Monitoring and mapping soil salinity is crucial for effective adaptation and mitigation through land reclamation actions. The appropriate mapping methods are directly related to the spatial scale of interest. The need of regional soil salinity mapping was also one of the first published geostatistical applications (Hajrasuliha et al., 1980). Macroscopic

Table 4
Classification of the water EC and TDS regarding the hazard of adverse salinity effects.

EC (dS m^{-1})	TDS (ppm)	Salinity hazard
0–0.25	<160	Low – water use is safe
0.25–0.75	160–480	Medium – water quality is marginal
0.75–2.25	480–1470	High – water unsuitable for use
>2.25	>1470	Very high

Table 5
Salinity/alkalinity/sodicity classification scheme (van Beek and Tóth, 2012).

Soil type	Soil property			
	EC (dS m^{-1})	SAR	ESP	pH
Nonsaline, nonalkaline	<4	<13	<15	<8.5
Saline	>4	<13	<15	<8.5
Alkaline	<4	>13	>15	>8.5
Saline – alkaline	>4	>13	>15	>8.5

Table 6
Potential symptoms of salinity, sodicity and high soil pH, after Ali (2011).

Soil problem	Potential symptoms
Saline soil	White crust on soil surface
	Water stressed plants
	Leaf tip burn (chlorosis/necrosis)
Sodic soil	Dark powdery residue on soil surface
	Poor drainage, crusting or hardsetting
	Low infiltration rate; high runoff and erosion
	Stunted plants with leaf margins burned
Saline-sodic soil	Generally, same symptoms as in saline soil
High soil pH	Stunted yellow plants
	Dark green to purplish plants

maps of salt affected soils at global scale (Li et al., 2014; Szabolcs, 1985) may roughly illustrate the extent of the environmental problem, however regional or greater level assessments are based on remote sensing and geographic information systems coupled with ground measurements. Monitoring at farm or field scale can be accomplished through local salinity sensors and sampling or non-invasive geophysical techniques (Domra Kana et al., 2015) such as Electromagnetic Induction (EMI), Vertical Electrical Sounding (VES) and Electrical Resistivity Tomography (ERT). Typically, soil profile surveys can lead to soil classification labelling (Table 1) whereas surface measurements can lead to quantifications according to Table 3.

Remote observations using satellite sensors and aerial photography offers efficient techniques for salinity mapping and monitoring, outperforming traditional ground methods at large spatial scales. The remote detection of soil salinity can be performed directly through salt features on the soil surface in the visible spectrum (Farifteh et al., 2008), or through multispectral/hyperspectral remote sensing indices that depict soil properties or vegetation health that can serve as a proxy (Setia et al., 2013). Remote geophysical measurements (Metternicht and Zinck, 2003) such as airborne electromagnetic (Ganjegunte et al., 2014), magnetic, and gamma-ray sensors also have the ability to directly map subsurface soil information when combined with ground data. Multi-scale integrated assessments that use a combination of remote sensing, field data and various modeling approaches can improve the development of soil salinisation risk maps useful to land managers and users (Bouksila et al., 2013; Douaoui et al., 2006; Farifteh et al., 2006; Metternicht and Zinck, 2003). Finally, exploratory data analysis can be employed to associate physical and socioeconomic variables with the risk of soil salinisation thus providing comprehensive risk maps (Salvati and Ferrara, 2015).

At the European level, a number of characteristic examples of mapping and monitoring innovations and applications regarding soil salinity highlight the regions where the problem is most significant to scientists. Mapping of the spatial variations of soil salinity applying geostatistical methods on EC_e, pH, and ion concentrations were first demonstrated in the UK (Burgess and Webster, 1980), and more recently in Alicante (Juan et al., 2011) and Murcia (Martínez-Sánchez et al., 2011), SE Spain, as well as in Rhodope (Pisinaras et al., 2010) and the Peloponnese (Alexakis et al., 2015), in Greece. Imaging spectroscopy techniques to map saline soils have been applied in SE Spain using ASTER (Melendez-Pastor et al., 2010) and Crete, Greece using WorldView-2 multispectral satellite imagery at multi-temporal stages (Alexakis et al., 2016). Mapping of saline soil physical properties has also been supported by geological and geophysical mapping at irrigated fields in Lerma, Spain (Aragüés et al., 2011), coastal Alt Empordà, NE Spain, where two decades of monitoring were integrated (Zarroca et al., 2011), vineyards in Cetona, Italy (Costantini et al., 2010), the Island of Terschelling in the Netherlands (Groen et al., 2008), and in 3-dimensional soil salinity representations in Quinto Basin, NE Italy (Greggio et al., 2012).

The need of harmonized soil mapping and monitoring at a Pan-European level (Eckelmann et al., 2006; Morvan et al., 2008; van Beek et al., 2010) motivated the initiation of several projects. Monitoring harmonization may be achieved by defining and evaluating (Stephens et al., 2008) top indicators (Huber et al., 2008) for salinisation at field level and geo-statistical upscaling at regional, national (Arrouays et al., 2008) and European level (Morvan et al., 2008) based on specific procedures and protocols (Jones et al., 2008). Recent coordinated efforts towards this goal include (Morvan et al., 2008) and (van Lynden et al., 2014). Finally, Montanarella et al. (2016) have presented global maps of soil threats from the first State of the World's Soil Resources Report, identifying salinisation and solidification as the 2nd main threat in Europe. Several studies (e.g. Paz et al., 2004) have shown that salinisation levels in soils in several European countries (e.g. Spain, Italy, Greece, Cyprus, Romania and Hungary) are increasing, but systematic data on trends across Europe are not documented in literature. JRC (IES) has recently developed an updated version of the Soil Geographical Database of Europe (SGDBE) which among other threats presents the limitations to agricultural use posed by salinity and sodicity (Fig. 3).

Table 7
List of models relevant to salinity including main modelled interactions and published field applications in Europe.

Model name	Type	Key processes	European studies	Key reference
AQUACROP	C	Y	–	Raes et al. (2012)
BUDGET	C	WB/V	–	Raes (2002)
CATSALT	C	RR/LU/CK	–	Tuteja et al. (2001)
DRAINMOD-S	C	WB/L/V	–	Kandil et al. (1995)
ENVIRO-GRO	P	GW/ST/CK/V	–	Pang and Letey (1998)
FEFLOW	P	GW/ST/CK	–	Diersch (2014)
FLOWTUBE	C	GW	–	Dawes et al. (2000)
HYDRUS	C/P	GW/ST/CK/V	ES, PT	Kool and Van Genuchten (1991)
LEACHM	P	GW/ST/CK	–	Hutson and Wagenet (1989)
MODFLOW	P	GW	PL	McDonald and Harbaugh (1984)
MOCDENS3D	P	GW	IT, NL	Oude Essink (2001)
MOPECO	C	Y	SP	López-Mata et al. (2010)
MT3DMS	P	ST/CK	–	Zheng and Wang (1999)
OASIS_MOD	C	WB	–	Askri et al. (2010)
PHREEQC	P	ST/CK	ES, PT	Parkhurst and Appelo (2013)
SAHYSMOD	C/P	GW/WB/L/V	–	Oosterbaan (2005)
SALTIRSOIL	C	WB/CK/V	ES	Visconti et al. (2011)
SALTMED	C/P	ST/L/V	GR, PT, IT, DK, RS	Ragab (2002)
SALTMOD	C	WB/L/V	–	Oosterbaan (2001)
SALSODIMAR	C	L	ES	Visconti et al. (2012)
SWAGMAN	C	WB/LU/V/E	–	Khan et al. (2008)
SWAGSIM	C	GW/WB	–	Prathapar et al. (1996)
SWAP-WOFOST	P	SVAT	NL	Kroes and Supit (2011)
SWATRE	P	GW/WB/V	–	Belmans et al. (1983)
UNSATCHEM	C/P	GW/ST/CK/V	–	Šimůnek et al. (1996)
WATSUIT	C	L	ES	Rhoades and Merrill (1975)
WAVES	P	SVAT	–	Zhang and Dawes (1998)

Model type and key process abbreviations: P: physically based; C: conceptual; SVAT: soil-vegetation-atmosphere; RR: rainfall-runoff; GW: groundwater flow; LU: land use; CK: chemical kinetics, ST: solute transport, V: vegetation component; L: leaching; WB: water balance; Y: yield response to water.

6. Modeling and assessment

Salinity is a dynamic and transient condition in saline soils. Chemical reactions in root zone (solubility, precipitation, cation-exchange reactions) in irrigated fields affect soil salinity and sodicity and salt leaching to drainage water. Many studies use models to evaluate salinity, sodicity, and environmental hazards of drainage water as a result of irrigation (Oster and Rhoades, 1975; Rhoades and Suarez, 1977; Shahid et al., 2013) and others calculate the effect of chemical reactions in the soil solution composition for transient conditions within the root zone (Jury et al., 1978; Robbins et al., 1980). The First Expert Consultation on

Advances in Assessment and Monitoring of Salinisation for Managing Salt-Affected Habitats (Aquastat, 2016), concluded that salinity models may face various limitations and vulnerabilities is not properly designed and developed. While state of the art physically based models of water and solute transport can be considered, calibration and validation considering soil and crop field data as well as a solid understanding of the dynamic nature of salinity are required to produce reliable soil salinity management scenario results (Shahid et al., 2013). A major constraint to these models is usually the lack of input data (Ranatunga et al., 2008), therefore simple more robust forms are advantageous.

The application of the concept of Leaching Requirement (*LR*) – the amount of water that must infiltrate the root zone to retain soil salinity within acceptable levels – can be expressed by means of easily measurable and robust properties (Rhoades, 1974), such as the water content of soil at field capacity and in the saturated paste. *LR* is defined by the following equation (van Beek and Tóth, 2012):

$$\mathrm{LR} = \frac{\mathrm{D_{DW}}}{\mathrm{D_{IW}}} \approx \frac{\mathrm{w_{FC}}}{\mathrm{w_{SP}}} \times \frac{\mathrm{EC_{IW}}}{\mathrm{EC_e^*}} \tag{3}$$

where D is the amount of water inputs in mm/year, w the soil water content by weight, and $\mathrm{EC_e}$ the electrical conductivity. Symbol subscripts DW, IW, FC, and SP represent drainage water, irrigation water, field capacity, and saturation extract, respectively, whereas the asterisk (*) represents the maximum electrical conductivity of the saturated paste (Corwin et al., 2007). The LR component has motivated research on drainage improvements as a direct way to simulate the necessity of drainage.

Based on this concept, various transient LR models (e.g. WATSUIT, TETrans) (Corwin et al., 2007) have been developed as well as more advanced software that while focussing on salinity/sodicity problem also includes other complex key process (e.g. UNSATCHEM) (Shahid et al., 2013; Šimůnek et al., 1996). This advanced code has been used successfully to understand both salinity and sodicity process dynamics at a very local scale (Jalali et al., 2008). Other software, such as LEACHM, PHREEQC, HYDRUS, and ORCHESTRA are less focussed to soil salinity issues (van Beek and Tóth, 2012). Table 7 gives an overview of the most relevant models, their main characteristics, a respective key reference, and the European countries where published field applications exist. Apart from these models, a range of mostly black box data driven models have been applied for case specific studies (Eklund, 1998; Patel et al., 2002; Zou et al., 2010). The systematic review of models reveals that Mediterranean countries attract scientific attention for secondary salinity applications, with the SALTMED LR model (Ragab, 2002) being the most popular in field applications. On the other hand, fewer N European country applications are focused on primary soil salinisation modeling, mainly due to sea level rise.

7. Conclusions and outlook

In Europe, salinisation is not considered as important as in areas that have suffered for long for its consequences, such as continental Australia. Nevertheless, in coastal southern Europe the problem is intensified by the increase of ground water abstractions that grease the wheels of seawater intrusion. As such, salinisation can become part of the desertification paradigm, enhancing the rate of soil degradation but also the social conflict over the sparse natural resources of some semi-arid lands. Moreover, important knowledge gaps such at the carbon dynamics in saline soils still exist in order to fully map the effects of salinisation on soil function. Regarding the monitoring of soils, remote sensing and especially satellite imagery is a promising and cost effective technology for estimating soil properties. Nevertheless, current research indicated that methods are extremely case specific and a no-regrets index for quantifying salinisation does not exist yet. Undoubtedly, remote sensing could be a staple in the direction of harmonized soil monitoring and mapping much as the (EEA, 2000) land cover strategy. Regarding monitoring, literature contains a significant amount of work, possibly lead by the efforts of (Morvan et al., 2008), for the harmonization of measurements at least for a limited spatial support. Nevertheless, when it comes to mapping, European scale datasets are lacking consistency and possibly comprehensiveness. Here we make an attempt to aggregate traceable spatial information on salinity and sodicity. Finally, as demonstrated here, estimates of the cost of salinisation in Europe are either rough or based on a subset of cases.

This lack in data, cost assessments and monitoring is reflected by the deficient EU policy on soils. A number of policy and soft law texts (Quevauviller and Olazabal, 2003) indicate the intention of EU for further and more specific protection of the soil, nevertheless a hard law text (directive, regulation) is vitally important in order to set the limit values of the salinisation soil threat. To underline this, (Eliasson et al., 2010) indicate eight essential climate, soil and terrain criteria that can be elaborated for the future delimitation of the Intermediate Less Favoured Areas (LFAs) support measure of the CAP, of which soil salinity is critical. Nevertheless, one of the strongest policy instruments, the Water Framework Directive (WFD), treats soil merely as a medium of achieving "good status" of waters and member states policy implementation being inherently weaker (Kutter et al., 2011). After failing to adopt the proposed Soil Framework Directive (Naidu et al., 2015), it falls upon the broad umbrella of the 7th Environment Action Programme to turn steer towards efficient soil protection and remediation. In this effort soil salinisation should be a priority.

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