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Critical knowledge gaps and research priorities in global soil salinity

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Abstract

Approximately 1 billion ha of the global land surface is currently salt-affected, representing about 7% of the earth's land surface. Whereas most of it results from natural geochemical processes, an estimated 30% of irrigated lands globally are salt-affected through secondary human-induced salinization. Application of lower quality, alternative irrigation water is further threatening expansion of the areal extent of soil salinity, in addition to climate change causing increases of salt-water intrusion in coastal areas and increasing crop water requirements. The reduced availability of freshwater resources for irrigation, the continued reduction of the world's cultivated agricultural area by land degradation and urbanization, in conjunction with a growing world population further complicates the problem seeking sustainable solutions. This scoping review prioritizes critical knowledge gaps and makes recommendations for 10 priorities in soil salinity research toward a sustainable and productive agricultural system for a food-secure future world.

We also include basin-specific case studies that illustrate progress of the world's major irrigated areas in addressing impacts of soil salinization. By identifying research priorities, we seek to accelerate enhanced research funding to bring new knowledge and innovative solutions toward mitigation of soil salinity impacts. We further want to inspire the science community to develop new directions in salinity research.

1. Introduction

Soil is vital to humankind and our livelihood. Soil processes affect the quality of the food we eat, the water we drink, the air we breathe, and is the foundation of our living and transportation infrastructures (e.g. buildings, parks, roads). As the world's population continues to grow and society expects a wider range of food selections, to provide this more selective world with nutritious food and feed will largely depend on our ability to maintain and sustain productive agricultural soils. Recognizing that soils have a central place in achieving food security, we note that the available arable land resource is decreasing at an alarming pace. In fact, we are at a point in time of what could be designated as a decade of peak agricultural land globally, indicating that the world's area of productive arable land is nearing its maximum. This is so because the annual expansion rate of new farmland is becoming less than the land area removed from agriculture. Causes for reduction in productive farmland are (1) its conversion to urban and industrial development, (2) taken out of production because of it being degraded such as by soil erosion, compaction or salinization, and (3) threatening public health because of soil contamination. It is estimated that about 15% of the world's total land area has been degraded (Wild, 2003).

In addition to the acreage of productive agricultural land decreasing, freshwater resources are also becoming scarce as populations increase, demanding additional water for domestic and industrial use. Moreover, while diverting increasing volumes of water for maintaining healthy freshwater environments and ecosystems, water for irrigated agriculture is becoming restricted in many arid and semi-arid regions. We note that whereas only about 15% of the world's agricultural land is irrigated, it produces about 45% of global food production and even more for fruit and vegetables. As high-quality freshwater availability is becoming a major constraint globally, increasing water use efficiency of irrigated agriculture is becoming essential. This form of agricultural intensification means to do more with less while simultaneously minimizing its environmental footprint and mitigating its contributions to climatic changes and/or adapting to it.

Additional constraints on agricultural production include public debates and policy changes regarding its environmental impacts on soil, air, and water quality, the use of genetically modified (GM) foods, as well as the threat of a changing climate. Among various mitigation and adaptation options, one calls for sustainable intensification of agriculture, water- and climate-smart agricultural practices, as well as for conservation agriculture to improve soil health and to minimize environmental impacts on soil, water, and air quality. In addition, other non-soil related practices are suggested, such as closing crop yield and nutrient gaps and reducing food waste (Foley et al., 2011). Collectively, any of these land and water management practices serve to enhance soil quality, reduce the environmental footprint, conserve freshwater resources, reduce soil degradation while sustaining food production.

Hence, the preservation of our soils is crucial. It is no wonder then that we must address causal factors of soil degradation, such as by water and wind erosion, soil contamination and soil salinity. We note that the room to expand cropland beyond the estimated 12% of the terrestrial land surface is limited, because most productive lands are already in agricultural use, whereas converting additional land would lead to either increasing environmental impacts (e.g. erosion) of marginal lands or destruction of the world's richest natural ecosystems. The importance of sustainable land management was recently acknowledged in the IPCC (2019) Special Report on Climate and Land (IPCC, 2019), highlighting interactions and feedbacks between our changing climate, land degradation, sustainable land management and food security, stating: "Land provides the principal basis for human livelihoods and well-being including the supply of food, freshwater and multiple other ecosystem services, as well as biodiversity. Human use directly affects more than 70% (likely 69-76%) of the global, icefree land surface (high confidence). Land also plays an important role in the climate system."

Among the most prevalent forms of soil degradation, in addition to air and water erosion and soil contamination, is human-induced soil salinization. Soil salinization occurs by the accumulation of water-soluble salts in the plant rooting zone, thereby impacting water and soil quality, and inhibiting plant growth. Osmotic changes in soil water caused by total salinity reduce the ability of plants to take up water from the soil. In addition, specific ions such as Na and Cl negatively impact plant physiology and become toxic when absorbed by the plant at higher than beneficial amounts. Besides, Na accumulation in surface clay-mineral soils cause soil swelling and dispersion thereby reducing water infiltration and soil drainage and causing waterlogging and flooding in sodic soils.

The geological salinization is by far the largest fraction of the approximately 1 billion ha (Bha) of salinized land, making up about 7% of the earth's land surface. In addition, approximately one-third of the world's irrigated land is salt-affected in some way (FAO and ITPS, 2015), equal to about 70 Mha. It is estimated that the area of salinized soils is expanding with a rate of about 1.0–2.0 Mha/year. However, recent data are scarce, and reported data are greatly outdated (Omuto et al., 2020). As freshwater resources become more scarce, alternative irrigation waters are tapped into, further threatening soil degradation in many arid regions. Furthermore, climatic change is causing sea level rise and more rapid saltwater intrusion in coastal areas, whereas increased evaporative demands require larger irrigation water amounts.

Examples of soil salinization by ancient societies are documented widely, caused by overirrigation, flooding and associated rising water tables, specifically in Iraq across the Euphrates and Tigris, but also in Pakistan and India along the Indus plains and in the Americas (Ghassemi et al., 1995; Hillel, 1992; Shahid et al., 2018). In most if not all of these cases, salts have accumulated in the soil rooting zone over hundreds to thousands of years, because of capillary transport from the rising water tables invading the crop's rootzone, thereby necessitating cultivating increasingly salt-tolerant crops (e.g. from wheat to barley), eventually leading to hunger and wars, ending those early agricultural civilizations. More recently in the last 50 years or so, salinization has degraded lands in the Aral Sea basin in Central Asia, the Yellow River basin in China, the Murray-Darling Basin in Australia, and the San Joaquin Valley in California (Chang and Brawer Silva, 2014) at a much faster pace. Although estimates vary widely, salinized lands are growing at an approximate rate of 10%/year (Nachshon, 2018) or about, mostly by human-induced agricultural practices (10 Mha/year), according to Szabolics (1989).

The intent of writing this synthesis paper was triggered after a literature review on soil salinity over the past few decades. It was found that most recent publications are applied and hardly added new basic research, because most presented concepts dated back to before 2000, when large-scale irrigation projects largely expanded the world's irrigated area. Moreover, we believe that funding for salinity research has declined as soil research priorities changed. Though not comprehensive but believed to be indicative, a search using Google Scholar on publications that have soil salinity in their title showed that such publications have stagnated over the past 10 years (Fig. 1). Similar conclusions were presented in Li et al. (2014) for China,

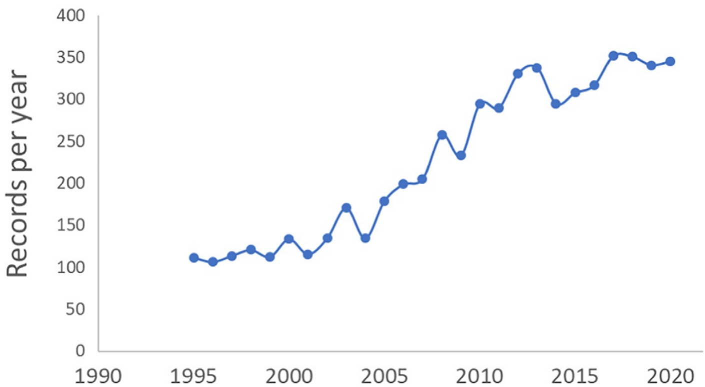

Fig. 1 Number of publications from 1995 to 2019, as obtained from a Google Scholar search of articles with soil salinity in their title.

where soil salinization is becoming a major threat for its food production, with about 5% of its total land area salt-affected with irrigation used on about 75% of its cropland.

We start the review with a presentation of terminologies and the most important concepts in soil salinity management in Section 2. To avoid the repetition of introductory material, we summarized soil salinity measurements and modeling approaches in Section 3, followed by Sections 4–13 that identify the most important soil salinity research priorities. Whereas the first three of these represent improved management of soil salinity using mostly existing knowledge, the additional seven priorities are identified as critical knowledge gaps in salinity management for the future. Each of those 10 sections (a) briefly review past research accomplishments through about 2000, (b) highlight changes in knowledge and practices since then with a summary of recent research, and (c) conclude with identified research priorities that address shortcomings to plan for a food-secure future. Section 14 reports on additional research needs that were not included with the 10 knowledge gaps.

Section 15 presents case studies across the major irrigated regions in the world, such as Australia, California, China, Israel, the Indus-Ganges basin (Pakistan, India), the Euphrates-Tigris basin in the Middle East, the Nile basin, as well as Latin America and the Netherlands with neighboring countries, to illustrate their progress in addressing impacts of soil salinization. Moreover, these studies list additional requirements that need to be achieved to limit continued land degradation and loss of prime agricultural lands by soil salinity in the future. We conclude the report with a final Section 16 that summarizes our mutual findings and discusses a future perspective on the sustainability of irrigated agriculture in the context of societal issues of water and food security.

By identifying the most critical knowledge gaps in soil salinity, we intent to accelerate new research funding to generate new knowledge and innovative solutions. We further want to inspire the science community in developing new directions of salinity research that addresses the identified knowledge gaps presented.

2. Concepts of soil salinity and salinity management

The purpose of this section is to comprehensively review the soil's literature on soil salinity and its relevance, and to present proven soil salinity management practices. We will do this through a review of established handbooks and articles. Specifically, we note the following key references: Salinization of Land and Water Resources (Ghassemi et al., 1995), the ASCE Manual on Agricultural Salinity Assessment and Management (Tanji, 1990) with revision by Wallender and Tanji (2012), Soil Salinity under Irrigation (Shainberg and Shalhevet, 1984), and Saline and Sodic Soils (Bresler et al., 1982). Other references that are relevant are Kamphorst and Bolt (1976) and Sposito (2016).

2.1 Sources of salinity

Soil salinity issues occur under a wide range of climatic conditions, both under natural and human-induced conditions, but are especially widespread in arid and semi-arid climates where rainfall is inadequate to leach accumulated salts below the plant's rooting zone, whether irrigated or rainfed.

The key factors associated with soil salinity are geology and its chemistry, climate, and local hydrology. Rock mineral weathering of parent geological material is the primary source of all salts. It is the main source of salt in seawater and irrigation water taken from streams, lakes, and groundwater. Salts in seawater arrive on land, via atmospheric deposition either by rain or wind, or via seawater intrusion such as by tsunamis or hurricane winds along coastal areas. When formed, soils may already contain high amounts of salts, due to the parent rock material from which it is derived, such as through the weathering of carbonate minerals (sedimentary rocks) or feldspars (granitic rock). Sedimentary rocks typically contain high amounts of carbonates and sulfates, so that their weathering leads to high alkaline soils containing significant amounts of gypsum and/or calcite. In contrast, weathering of granitic rock dominated by primary minerals such as quartz, feldspars, and micas result in more acidic soils. Climate dictates the rate of both chemical and physical weathering through its temperature regimes, dissolution, and precipitation of salts, leaching of dissolved ions/salts (high rainfall) or accumulation of salts (low rainfall).

One distinguishes between primary and secondary, human-induced salinization. Primary salinization occurs by natural processes, such as by atmospheric deposition through rainfall or wind or by rock weathering, accumulating soluble minerals in soils, geological deposits and groundwaters. For example, fossil groundwaters originate from marine depositions, from which salts become available through seepage to near the land surface or through groundwater pumping. Natural soil salinization occurs widely in seawater-submerged soils and geologic formations and in coastal areas with shallow saline groundwaters. For example, much of the salinity in US Northern Great Plains is associated with saline seepage through marine shales and derived weathered regolith, originating from a shallow ocean overlying the region some 100 million years ago (Miller et al., 1981). The changing salinity of the Plains in recent times is largely attributed to the change in land-use from prairie grassland to cropland and changing weather patterns (Nachshon, 2018) with extreme summer rains and associated flooding. In the Netherlands, saline seepage from rising seawater in their coastal areas below sea level (polders) always threatens its freshwater availability and are causing main concerns for their agricultural landuse (Raats, 2015; Section 15.10).

Secondary salinization is caused by human activities, principally by irrigation of agricultural crops under poor drainage conditions and while using marginal irrigation waters. In addition, soil salinity can be caused by removal of deep-rooted vegetation and thereby increasing groundwater recharge (dryland salinity; Holmes, 1981), and by addition of chemicals to soils such as through fertilizers and waste waters. The specific cause of soil salinization depends on local soil and groundwater transport processes relevant to the landscape and thus varies with climate, landscape type, agricultural activities, irrigation method and associated soil and water management practices. Groundwater related salinity occurs when saline groundwaters rise to reach close to the plant rooting zone, followed by upwards transport into the near-surface soil through capillary forces that are triggered by soil evaporation and plant transpiration. This can occur through both primary and secondary salinization, for example, through seepage in low-lying areas or when irrigation-induced by rising groundwater tables. In the latter case, either through excess irrigation or native perennial deep-rooted vegetation removal in dryland agriculture (Australia and Latin America, Sections 15.1 and 15.7). Non-groundwater associated salt accumulation occurs in landscapes with groundwaters that are too deep for upwards capillary action to the plant rooting zone. It is prevalent when drainage of rain or irrigation waters is limited, such when largely controlled by soil textural variations in the landscape or with soil depth. Specifically, coarse-textured soils allow for adequate drainage and salt leaching, whereas soils containing low-permeable soil textural layers restrict deep percolation such as in sodic soils, causing water-logging conditions and shallow water tables.

Approximately 6% of the world's terrestrial land is believed to be salinized by primary salinization. In addition, some 20% of all cropland and between 1/4 and 1/3 of irrigated land is salinized by secondary salinization, totaling about 1 Bha globally.

2.2 Definitions of salinity and sodicity

To quantify soil salinity, one commonly estimates the concentration of total soluble salts through the electrical conductivity or EC, expressed in dS/m or mmho/cm. We note that 1 dS/m corresponds to a salt concentration of approximately 680 mg/L of total dissolved solids (TDS) in soil solution (seawater is about 50 dS/m). However, the effective concentration will depend on the ion activity coefficients, as affected by many factors such as the presence of ion pairs, other complex formations, and temperature (Bresler et al., 1982). Whereas field measurements of EC represent the bulk soil, the more accepted measurement of soil salinity is using the EC of the extracted solution of a saturated soil paste, defined by EC_{ex} (US Soil Salinity Laboratory Staff, 1954). This is so, because plants are dominantly affected by soil salinity through the concentration of salts in the soil's solution. While other extraction methods may quantitatively be more reproducible and have shown good correlations with the chemistry of the saturated paste for Cl^- dominated systems (Sonmez et al., 2008), the US Salinity Laboratory promoted using EC_{ex} because (1) the chemistry of the saturated soil extract is close to that of the soil water (EC_{sw}) and (2) the chemistry could vary due to dissolution and precipitation of sulfate and carbonate minerals, should larger soil water dilutions be employed. The widely accepted classification of what constitutes different levels of soil salinity was defined by the US Laboratory Staff (1954), with EC_{ex} values smaller than 2 dS/m classified as non-saline soils, whereas EC_{ex} values between 2 and 4, 4–8, 8–16 dS/m are defined as slightly, moderately, and strongly saline soils, respectively. Though widely accepted, there are limitations to its use. Firstly, it is a laboratory measurement that underestimates the in situ salinity for unsaturated soils. Second, soil wetting in the laboratory will often lead to dissolution of precipitated salts (gypsum, calcite), thereby overestimating EC of the natural soil (Section 8).

The other relevant soil salinity property is related to the amount of sodium (Na) in soils, as expressed by the Exchangeable Sodium Percentage (ESP) or the Sodium Adsorption Ratio (SAR). The weathering of primary rock minerals results in the generation of individual soil particles that are negatively charged, thereby leading to the electrostatic adsorption of cations from soil solution to counterbalance the total charge along the particle's interstitial hydrated surfaces. Much of the ability to adsorb cations will depend on soil mineral type and varies widely between clay minerals. However, all soils do adsorb ions at a certain level with the type of cations absorbed largely controlled by the composition of the soil solution. The magnitude of adsorptive capacity and level of negative charge is quantified by the soil's Cation Exchange Capacity (CEC) and varies between near zero for pure sands to 100 meq/L or larger for smectite clay soil minerals. Most destructive to soils are large amounts of Na adsorbed in place of other divalent cations such as Mg and Ca. Therefore, the SAR is defined as

$$SAR = \frac{Na^{+}}{\sqrt{Ca^{2+} + Mg^{2+}/2}}, \tag{1}$$

with all concentrations expressed in meq/L and measured from a soil saturation paste extract or for irrigation water. Analogously, the ESP is defined as the ratio of soil exchangeable Na to soil CEC (Section 12), with all values expressed in meq/100 g of soil and computed as a percentage (x100%). Both Na content indicators are used interchangeably and can be derived from each other using the Gapon coefficient (Bresler et al., 1982; Oster and Sposito, 1980; US Soil Salinity Laboratory Staff, 1954) that quantifies the Ca—Na exchange in soils. For the western US, this coefficient is assumed to be around 1.5 for ESP values up to 40% but varies among soil types. Some additional factors will need to be considered when using these definitions. First, soil CEC is highly pH dependent, as hydroxyl groups along soil mineral and organic matter surfaces may deprotonate and become negatively charged at high soil pH, such as for alkaline soils (pH > 8.5). Second, Ayers and Westcot (1985), Suarez (1981), and Rhoades (1982) have discussed adjusting SAR to account for the changes in ionic concentrations in soil solutions due to increasing levels of bicarbonate and carbonate ions in irrigation water, causing Ca or Mg ions to precipitate thereby increasing the sodicity hazard. Alternatively, in the presence of significant concentrations of K^+, it may have to be included in the calculation (Section 12).

It is noted that the chemistry of salt-affected soils is affected by many factors, especially for sedimentary soils that contain calcite or gypsum. For example, soil production of CO_2 by soil and plant root respiration increases the solubility of calcite, thereby creating more alkaline conditions, and even more so at higher solute concentrations of other ions causing the so-called ion strength effect. As emphasized by Suarez and Jurinak (2012), the solution chemistry can become very complex and requires the application of geochemical together with soil hydrological models to incorporate soil-mineral chemistry. Specifically, this was done in Schoups et al. (2006), using the UNSATCHEM hydro-salinity model (Section 3) to predict the long-term soil salinity for irrigated soils in California's San Joaquin Valley, necessitating the accounting for both cation exchange and gypsum dissolution-precipitation.

According to the US Laboratory Staff (1954), sodic soils are defined for SAR or ESP values larger than 13 or 15, respectively. Soils are classified as saline-sodic when $EC_{ex} > 4$ dS/m and ESP > 15. When considering total salinity of agricultural land, sodicity is almost twice as prevalent than salinity-affected, with 412 Mha of saline soils and 618 Mha of sodic soils (Oldeman et al., 1991) globally.

2.3 Salinity impacts on soils, plants, and the environment

Soils—Much of the impact by on soils is caused by relatively high exchangeable sodium levels (ESP), through its adsorption from soil solution thereby largely affecting soil physical properties such as the bulk density and the water retention and hydraulic conductivity characteristics. As compared with divalent cations such as Ca and Mg, the sodium ion (Na) is less strongly adsorbed to soil particle surfaces. When hydrated, soil particles surrounded with sodium dominated water film tend to repel each other, thereby leading to soil dispersion. This causes soil aggregates to break down into individual soil particles, thereby clogging interstitial pore spaces and forming depositional soil crusts upon drying. Particle dispersion is further accelerated by soil swelling, driven by osmotic gradients, forcing pore water into the interlayers of clay minerals, especially pronounced at low salinity. Upon soil drying, these soils will shrink, creating soil cracks that can go very deep into the soil profile. When wetting, these types of soil structural degradation will largely reduce water infiltration and soil drainage, causing waterlogging and flooding and making soil prone to water and wind erosion. Much more detail will be presented in Section 12.

Plants—Increasing levels of total salinity in the soil water solution will reduce the ability of plants to take up water from the soil because of osmotic effects, whereas specific ions such as Na, Cl, or B negatively impact plant physiological processes and can become toxic when adsorbed by the plant (Lauchli and Grattan, 2012). In addition, saline soils can reduce plant nutrient uptake or cause ion imbalances as specific ions such as Na can compete with other essential plant nutrients, causing mineral nutrition disorders and further the plant's ability to survive and produce. Any of these effects vary among plant species and crops. For that reason, empirical crop salt tolerance response functions have been developed (Maas and Hoffman, 1977), defining yield reduction as a function of total soil solution salinity based on EC_{ex} data. However, such data for developing salt tolerance parameters were collected assuming constant (steady state) and high soil moisture conditions, both during the growing season and with crop rooting depth. However, in real field conditions soils wet and dry depending on irrigation frequency so that soil salinity conditions are typically non-uniform with time and soil depth. These factors along with the exclusion of specific ion effects (e.g. Na and B) on plant stress and yield limits their applicability (Section 8). Yet, more detailed additional information is often not available. The osmotic effect on crop growth, as quantified by the soil water osmotic potential (OP, $\text{kPa} = \pm -36 \times EC_{sw}$, when expressed in dS/m) is often considered simultaneously with soil water stress, as determined by soil water matric potential. Both are abiotic stresses that can be added, such as illustrated in Fig. 2 (Rengasamy, 2006a), to reflect the combined additional energy required for plant root water uptake (kPa), as a function of soil water content and soil solution salinity.

Plants can adapt to salinity stress through physiological mechanisms such as salt exclusion by the plant root's apoplast and salt sequestration in specific plant organs such as in vacuoles of plant cells, or through osmoregulation. Breeding programs as well as genetic engineering approaches are widely developed for important food crops to be more salt tolerant. These aspects will be further reviewed in Sections 8–10.

Environmental—In addition to soil and plant effects, soil degradation by salinization can have significant environmental and ecological consequences. Foremost, elevated soil salinity levels can become so toxic that it eliminates

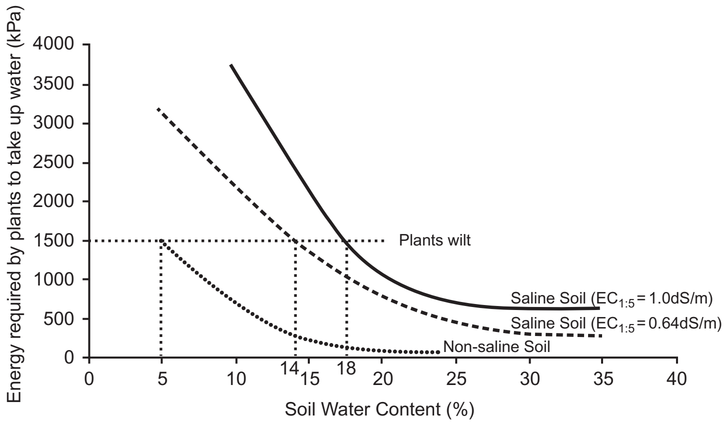

Fig. 2 Energy (kPa) required by plants to take up water as a function of soil water content and salinity, expressed by $EC_{1:5}$ dilutions (Rengasamy, 2006a).

native vegetation or transforms fertile lands to salty swamps or lead to desertification. Salt leaching and disposal of saline drainage waters may lead to groundwater, streams and rivers to become toxic for both human consumption and wildlife, with elevated concentrations of specific trace elements such as As, Cd, and Se (Dudley et al., 2008a; Tanji et al., 1986). Collection of drainage waters may provide a temporary and local solution; however, final disposal remains an issue because of their potential toxic constituents. Elevated soil salinity levels have shown to reduce soil microbial diversity and thus relevant soil microbial processes (Rath et al., 2017; Section 14).

2.4 Hydro-salinity modeling

Early on, in the second half of the past century, mathematical models were developed to optimize irrigation amounts for maximum yield, considering both water and salinity stress effects. These earlier traditional models applied a steady state approach based on the assumption that soil water content with corresponding salt concentration remained approximately constant for given time (irrigation season, irrigation interval, diurnal) and soil type (rooting depth, soil horizon, field). In combination with crop production functions that relate crop yield to applied water for different salinity values (Letey and Dinar, 1986), this relatively simple approach was acceptable as long as availability of freshwater did not limit excessive leaching water amounts. Steady state models apply the mass balance principle, stating that changes in soil water or salt concentration over time are a consequence of differences in water or salt moving in (rain, irrigation) versus out of the soil (drainage, evaporation, root uptake). In the context of this approach one defines the Leaching Fraction (LF) as:

$$LF = {}^{D_d}\!/_{D_i} = {}^{EC_i}\!/_{EC_d}, \tag{2}$$

stating that the mass of salts introduced into the soil must equal to the amount of salt leaving the soil, otherwise salts will accumulate. Here, D_d and D_i represent the depth of drainage and irrigation water applied, respectively, and D defines the volume of water per unit area of soil (cm of water). Though very simple indeed as it does not allow for chemical reactions or crop salt removal and can only be valid for long time periods (irrigation season or year), this steady state expression can be expanded to allow for crop evapotranspiration (ET), soil water stress and allowed salinity stress level, to define the Leaching Requirement (LR) that minimizes salinity buildup and salinity stress for specific crops (Oster, 1984). It can be defined by replacing EC_d in Eq. (2) by the maximum allowed EC value, as dictated by crop salt tolerance. Furthermore, assuming a rootzone mean salinity, Hoffman and van Genuchten (1983) used such an approach to illustrate the relationship between the salinity of the applied water, the salt-tolerance of a specific crop and the LR through a simple expression. Though such simplified models provided for unified concepts of irrigation water management, they did not account for soil heterogeneities, non-uniformly applied irrigation water or improved irrigation water management practices that can reduce the required minimum of leaching while maintaining crop productivity at acceptable levels. Later, and especially so in the past few decades during which irrigation water and salinity management has become more relevant and computer-intensive algorithms have been developed, more sophisticated numerical transient models became available. These process-based unsaturated water flow models using Richards equation (e.g. Simunek et al., 1999) allow for simulation of changes in soil water content and salinity at any moment in time and accounting for soil heterogeneities both with soil depth (one-dimensional) and across the irrigated field (multi-dimensional; Raij et al., 2016), as well as for non-uniform water applications such as for micro-irrigation (drip and sprinkler). As evidenced by Letey et al. (2011), application of steady state models for increasingly efficient high-frequency micro-irrigation systems typically overestimate LR values and soil salinity effects on crop yield.

The more precise transient state hydrosalinity models are hardly limited by complexity and may include as many relevant physical, chemical, and biological processes as desired. In principle though, most salinity models are derived from soil physical and hydrological simulation codes, and account for temporal and spatial changes in input data at any specific level and are multi-dimensional (Minhas et al., 2020a; Section 3.1). They solve for coupled highly non-linear partial differential equations that calculate soil water matric and osmotic potential, water content and soil water fluxes, root water uptake, as well as for solute (salt) content and fluxes, nutrient uptake, and other sink terms to proxy for soil chemical and biological reactions. This modeling approach requires orders of magnitude more input parameter values than steady state models, and therefore can create much uncertainty. Multi-dimensional models are increasingly applied to simulate and test improved management practices for micro-irrigation systems that allow for precision application of water and fertilizers. A regional integrated hydro-salinity model was applied by Schoups et al. (2005), to reconstruct historical changes in salt storage by irrigated agriculture through 2000 for the San Joaquin Valley. A comparison of steady-state and transient salinity management models is given by Corwin et al. (2012), suggesting that the dynamic uptake of plant root water enables the plant to tolerate higher rootzone salinities than the available salt tolerance values thus favoring transient model applications. Section 3 will detail the various modeling approaches in much greater extent, especially those currently applied and their needs for improvement are further presented.

2.5 Salinity management

Unquestionably, soil and water salinization will occur when practicing irrigation as well as for dryland cropping systems in (semi)-arid environments. Therefore, to mitigate and/or to adapt to soil salinization, a wide range of management options have been developed over time, yet none may guarantee long-term sustainability. Such practices vary widely and depend on soil type, landscape positioning, geohydrology, climate, and other local factors and may vary from field to field. Most irrigation engineering projects historically mandated adequate leaching and associated drainage capabilities to prevent rising shallow groundwaters that move salt into the crop rooting zone, and instead discharged accumulated salts away from the cropped field through drainage. Most of these surface irrigation projects required field water applications at low frequency because of the water delivery infrastructure, thus requiring refilling of deep soil water storage to reduce crop water stress between irrigations. Though still highly recommended, the leaching of salts further increase salinization of deep groundwaters whereas salt-containing drainage waters pose environmental threats through discharge of toxic trace elements such as Se (Tanji et al., 1986).

Leaching with required drainage management practices is particularly relevant for surface irrigation methods driven by gravity such as through basin, border, and furrow irrigation. These irrigation systems require adequate soil surface leveling to ensure reasonable water application uniformities across the irrigated fields. For gravity driven irrigation, water application control is limited leading to excess applications to ensure adequate wetting of the whole rooting zone across the irrigated fields, thus necessitating drainage. Either through ditches or perforated drain tubing, groundwater tables are kept sufficiently low to prevent upward salt transport into the rooting zone. However, although the associated high leaching fractions reduce soil salinity buildup, drainage flows create downstream water quality problems. Moreover, as the salts are leached, other applied substances such as agrochemicals and fertilizers such as nitrates move into the groundwater, further complicating irrigation water management.

More advanced micro-irrigation systems such as through sprinkling, surface and subsurface drip irrigation are pressurized allowing controlled application of water and fertilizers with time and location along the crop row but require almost continuous availability of irrigation water. Pressurized irrigation systems therefore often use pumped groundwater thereby increasingly depleting high-quality groundwater aquifers. Typically, drip and sprinkler irrigation are high frequency systems, applying relatively smaller volumes and therefore allow control of wetted soil volume, root zone salt concentration and minimize deep percolation below the plant root zone. For example, Taylor and Zilberman (2017) analyzed trends in irrigation systems from 1972 to 2010 for California as presented by Tindula et al. (2013), showing that irrigated land with low-volume (drip and micro-sprinkler) irrigation increased by approximately 38%, whereas the amount of land irrigated by surface methods had decreased by approximately 37% (Fig. 3). Their historical analysis explained that adoption of pressurized irrigation was triggered by water pricing and increased yields, occurring early on for soils with lower water holding capacities and for high-value crops.

Also, in cases of salinity buildup in soils, surface irrigation systems are converted to pressurized irrigation systems, in concert with changing to higher value cash crops for increased profitability. Hanson et al. (2008) concluded that such systems can be viable even under saline shallow groundwater conditions. The much higher irrigation frequency creates favorable

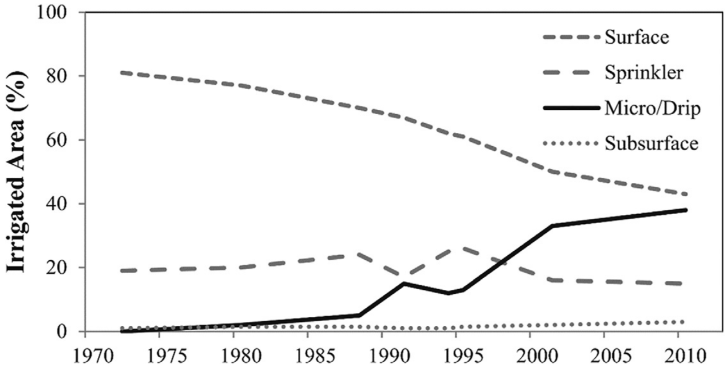

Fig. 3 Trends in irrigated area (%) by irrigation system category in California (Tindula et al., 2013), with permission from ASCE.

plant-soil-water conditions with salt concentrations in the wetted root zone near that of the irrigation water, thereby minimizing both water and salinity stress, though salinity can build up away from the wetted soil zones driven by soil evaporation. Such salt buildup can be reduced by rainfall between growing seasons. Also, using very high irrigation frequencies, more saline irrigation waters can be used if the soil solution salinity does not exceed the salt tolerance of the crop.

As an additional irrigation water management option, in case of limited irrigation water availability or to reuse drainage or wastewater, use of blended irrigation waters or cycling of saline with non-saline water may be feasible using micro-irrigation. There are many crop and soil water management factors that come into play when considering such practices including consideration of adverse effects of increasing soil sodicity and heavy metal concentrations (see Section 13). Innovative cycling strategies include using different quality irrigation waters for different crops on a farm or at different growth stages for one cropped field, and their sequential use. In a sequential reuse system, subsurface drainage water is collected for a series of fields, with increased saline drainage waters applied for more salt tolerant crops on the farm, including salt tolerant forages. Such an integrated on-farm drainage management system reduces the volume of drainage water that requires final disposal. Near-future technologies that further advance micro-irrigation include gravity drip systems, precision irrigation systems that control water volume and fertilizers amounts at the individual tree/vine scale and across fields using zoning and farm-scale desalinization (Section 7).

In addition to available irrigation and drainage water management options, one may have to adapt. For example, this can be done by selection of crops or varieties that produce satisfactory yields at higher soil salinity conditions by developing more salt tolerant crop species. As discussed in Section 11, progress in this area has been slow and much more research work is needed. Soil management options include soil reclamation such as by using halophytes to remove salts (phytoremediation) and chemical amelioration such as by application of gypsum to sodic soils. Many of such practices are discussed in Tanji (1990), including drainage water disposal and treatment.

More recently, to advance more sustainable solutions, there are increasing efforts to apply improved on-farm soil, irrigation, and crop management practices that reduce salt accumulation rather than seeking ways to cope with salinized soils (Section 6). For that purpose, the application of numerical process-oriented computer models is increasingly beneficial, as it allows for sensitivity analysis across defined salinity management options, selecting those that are most desirable. Computer model outcomes can be merged with field soil and crop monitoring and water application control devices, to allow for real-time crop-water-soil salinity management (Section 7).

3. Soil salinity modeling and measurements

In addition to the general salinity review, in this section we give an update on existing soil salinity modeling approaches, as well as established methodologies for in situ soil salinity measurements (Section 3.4). This will avoid duplication of introductory materials in subsequent sections. Regarding current model developments, we distinguish between models that focus on soil chemistry (Section 3.1), plant-soil water relations (Section 3.2) and those that evaluate soil salinity management practices (Section 3.3).

3.1 Soil chemistry

The primary source of soil and water salinity is the geochemical weathering of rocks throughout geologic times, releasing salts of various chemical compositions into surface and groundwaters. In addition to added salts by irrigation water, other controlling factor are dissolution and precipitations reactions of soil minerals, predominantly of gypsum and calcite, as affected by soil pH, alkalinity but also by soil mineralogy, CEC and organic constitutions (e.g. wastewater), soil redox reactions, gas exchange, etc. Clearly, the underlying complexity of salt chemistry requires geochemical computer models, that can be linked with soil water flow, solute transport, and plant growth models. Extensive reviews of relevant chemical processes of salt-affected soils are presented by Oster and Tanji (1985) and more recently by Suarez and Jurinak (2012). Of the various hydro-salinity models available, the most comprehensive are HP1 (Šimůnek et al., 2006) and UNSATCHEM (Simunek et al., 1996; Suarez and Simunek, 1997), simulating the chemistry and transport of the major ions, such as Ca, Mg, Na, K, SO_4, Cl, NO_3, alkalinity, and CO_2 in unsaturated soils. Both models account for various equilibrium chemical reactions, such as complexation, cation exchange and precipitation-dissolution such as for calcite and gypsum and includes the effects of solution chemistry on the soil's hydraulic properties. The UNSATCHEM module was used by Schoups et al. (2005, 2006) to evaluate the relevance of the complex salinity chemistry when considering the sustainability and long-term regional salt balance in California's San Joaquin Valley, including groundwater salinity. Unfortunately, most applications for impacts on plant growth or salinity management do not include specific ion chemistry and consider total soil solution salinity only.

We believe this may be a major shortcoming of future soil salinity research as specific ion effects may be relevant for improving on plant salt tolerance data (Sections 8 and 9), crop salt tolerance breeding (Sections 9–11) as well as for ion effects on soil hydraulic and transport characteristics (Sections 12 and 13).

3.2 Plant-soil water relations

To evaluate the effect of different levels of salinity on vegetation and on water fluxes between soil and atmosphere, simulation models are often used. In soil-hydrological models, plant stress effects by both osmotic and matric potentials on the water uptake and plant transpiration are considered. In their review, Hopmans and Bristow (2002) defined both type I and type II models of plant root water uptake, to simulate water flow in soil-root systems in a mechanistic manner. Type 1 models are based on computation of water potential gradients along a flow line in the soil-plant system (Nimah and Hanks, 1973). Macroscopic flow simulation models that describe plant water uptake by type II models compute plant root zone stress by macroscopic values of soil root zone water content and salinity through stress response functions with values between zero and one, representing reduction in plant transpiration relative to potential transpiration (Section 9.2). The major advantage of the type I modeling approach is that local processes between the bulk soil and the soil-root interface and their hydraulic connections in the multi-dimensional root architecture are simulated explicitly based on principal laws of water flow in porous media, acknowledging that plant root tissues can also be described as a porous material. Type I models therefore avoid empirical parameterizations of root water uptake, uptake compensation, and of stress functions that are used in type II models. In addition to analytical type I models that provide for a simplified representation of the root system, Javaux et al. (2008) developed a transient numerical model that considers the 3D detailed root structure and combines flow and transport of the soil and root system. This model was extended to account for salt accumulation at soil root surfaces and its effect on root water uptake (Jorda et al., 2018; Schröder et al., 2014). These simulations highlighted the importance of the difference between bulk and root surface water potentials and suggest that salt accumulation at the root surface needs to be considered, necessitating the need for small-scale transport simulations (De Jong van Lier et al., 2009; Section 9.3).

For type II models, empirical plant water stress response functions were derived from experiments that relate plant response to rootzone salinity and water content values. Though in principle simpler, several problems arise with the use of these empirical functions. First, the empirical functions were derived by relating plant responses over an entire growing season, to the averaged soil root zone matric and/or osmotic potentials (Feddes et al., 1976; van Genuchten and Hoffman, 1984), irrespective of changing meteorological conditions during the growing season. Yet, soil water flow models resolve flow and root water uptake processes at cm-scale spatial and hourly or smaller scale temporal resolutions. In unsaturated water flow models this is typically accounted for by computing the whole plant response from a composite of local stress responses, derived from local soil matric and osmotic potentials and root distributions. However, local reduction of water uptake may be compensated for by increased water uptake elsewhere in the rooting zone where conditions are more favorable (Jarvis, 2011; Simunek and Hopmans, 2009). Another issue arises from the fact that salt and water stress response functions have often been developed independently. Different approaches have been proposed to quantify the combined stresses, but it has been much subject of discussion and remains unresolved (Feddes and Raats, 2004; Homaee et al., 2002b; Shani and Dudley, 2001), and will be treated in depth in Section 9. Most importantly, type II models use bulk soil potential values for the macroscopic stress response functions, whereas plants respond to potential gradients at the soil-root interface of the rhizosphere. Consequently, salts are expected to accumulate in the rhizosphere, thus resulting is total soil water potentials that are different from those of the bulk soil (Simha and Singh, 1976). Estimation of the stress response function parameters for type II models can be obtained from in situ measurements of soil water content, salinity, root distributions, plant transpiration and root development. However, since root water uptake cannot be measured directly in the soil, the parameters of these functions are derived using inverse modeling, by which model parameters are optimized such that simulated and the measured variables are sufficiently close (Vrugt et al., 2009).

For combined matric and osmotic potential stresses, Cardon and Letey (1992a) compared the sensitivity of type I and type II models (see also Section 9). They used the type I water uptake model of Nimah and Hanks (1973) and concluded that it was insensitive to osmotic stress, while the type II model produced more reasonable results when compared with experimental data. Models that tend to focus on plant water relations for saline soil environments include SWAT and ENVIRO-GRO. The latter model was used by Feng et al. (2003) to simulate relative yields of corn and compared with experimentally measured yields for a range of irrigation water salinity and irrigation frequency values. Ben-Asher et al. (2006) applied the SWAT model to evaluate its ability to account for soil salinity effects for grapevines using both fresh and saline irrigation waters.

3.3 Salinity management

Many conventional salinity management practices have focused on ensuring adequate leaching of salts imported by irrigation water while maintaining sufficiently deep groundwater tables, mostly to mitigate crop yield losses by accumulated salts in the rooting zone (Ayars et al., 2012). However, recent research has focused much more on alternative options as dictated by limited available irrigation water resources. Most prominently, this has been the development of micro-irrigation systems, allowing accurate control of water application volumes and frequency. For example, Hanson et al. (2008) showed that subsurface micro-irrigation can be used even for relatively shallow groundwater table conditions, when properly managed and if seasonal rainfall is adequate to leach the accumulated salts above the dripline (Fig. 4). Ramos et al. (2019) evaluated the threat of increasing soil salinity when using deficit irrigation. In another study, Skaggs et al. (2006b) studied the effects of reusing saline drainage waters on alfalfa yield, exemplifying much recent focus of the need to apply soil salinity models to better understand the long-term effects of using marginal irrigation waters on soil

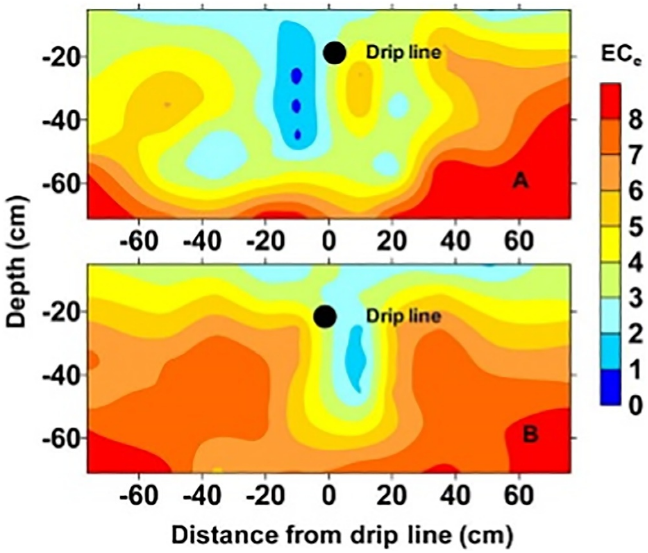

Fig. 4 Effect of amount of applied irrigation water on the measured distributions of soil salinity extracts (dS/m) around the drip line for (A) 589 mm of applied water (about equal to the seasonal evapotranspiration of processing tomato) and (B) 397 mm of applied water. Irrigation water electrical conductivity (EC) = 0.52 dS/m and groundwater EC = 8–11 dS/m. (Hanson et al., 2008).

salinity and plant growth. Others, such as Assouline and Shavit (2004) and Lyu et al. (2019) evaluated the use of reclaimed irrigation water on groundwater quality.

In addition to effects of marginal waters such as treated wastewaters on soil salinity, specific prevalent solution ions can interact with the soil matrix. Specifically, Na is affecting soil pore distribution, soil structure and thus the flow-controlling hydraulic properties such as soil water retention and permeability (Assouline et al., 2020; Assouline and Narkis, 2011). Such effects were simulated by Russo (2013), showing that exchangeable Na in treated wastewater may considerably reduce the soil's hydraulic conductivity, thus impacting infiltration rates of irrigated soils. Other needs for detailed soil salinity modeling include the evaluation of remediation of saline-sodic soils as presented by Chaganti et al. (2015). Though one can likely refer to many different soil salinity management models, those most widely used are HYDRUS (Simunek et al., 2016) and SALTMED (Ragab et al., 2005). Particularly because of its extensive documentation, the modeling environment of the HYDRUS software packages is widely used and offers diverse use of its computer simulation tools, with one- and multi-dimensional codes, integrated with other modules such as UNSATCHEM, PHREEQC, MODFLOW, and WOFOST, among others, also allowing for evaluation of irrigation, salinization and sodification management practices (https://www.pc-progress.com/en/Default.aspx?hydrus-3d).

3.4 Soil salinity measurements

Many local-scale sensors are available for measuring in-situ soil salinity. Each sensory method has its advantages and disadvantages, whereas new developments come to market all the time, reducing technological and economical barriers for more cost-effective and efficient applications. Excellent introductions are presented in Hendrickx et al. (2002) and Corwin and Yemoto (2017). For almost all soil salinity sensors, soil or solution EC is determined from electrical resistance (DC) or impedance (AC) measurements.

Discrete direct sampling of the soil water solution using suction cups is widely used in agricultural and environmental research assuming that the sample's chemistry represents the soil pore water solute composition at the cup location, whereas saturation extracts (or higher dilutions) taken from sampled soil cores are typically used to measure soil water EC_{sw} (Section 2.2). Most other methods measure the bulk soil, EC_b, which is a function of the volumetric soil water content (θ), EC_{sw}, a soil-specific transmission coefficient and the soil's surface conductance (EC_s). To illustrate their dependency, we present the Rhoades et al. (1976) model, which shows that the bulk soil and solution EC are related according to:

$$EC_b = c_1 EC_{sw}\theta^2 + c_2 EC_{sw}\theta + EC_s \tag{3}$$

whereas c_1, c_2, and EC_s are soil specific, typically this expression needs to be calibrated from field measurements of EC_{sw} and θ, so that soil solution EC_{sw} can be determined from bulk soil EC measurements.

Resistivity methods introduce an electrical current by way of inserting electrodes into the soil and measure potential at other in-line electrodes. From the soil resistance, the EC_b is computed from a known cell constant which value depends on the electrode configuration and spacings. Most widely, the four-electrode Wenner array is used in various configurations, including a point-scale soil conductivity probe that can be inserted into the soil at selected depth increments with both manual or automated readings, as well as those that can be pulled by a tractor to obtain field-scale soil salinity information when combined with a GPS system. Field-scale monitoring of soil salinity is also possible using electromagnetic induction (EMI). By way of this noninvasive method, an AC current is passed through a transmission coil. The electromagnetic field produced generates smaller current loops with magnitude depending on soil EC. These smaller currents produce an induced secondary magnetic field of which the voltage is measured through a receiver coil. A significant advantage of the EMI as well as the Wenner array probe is that the representative depth interval of the measurement can be varied by changing resistor or coil configurations.

Alternative sensors are based on measurement of the soil's dielectric permittivity (Corwin and Yemoto, 2017), such as TDR and capacitance sensors, mostly used for measurement of soil moisture but can be used for soil EC information as well. In Time Domain Reflectometry (TDR), a voltage signal is propagated along a set of soil-inserted wave guides, with both soil moisture (θ) and salinity (EC) affecting the shape, duration, and magnitude of the reflected waveforms. Capacitive soil salinity sensors are based on the measurement of the imaginary component of the complex permittivity. Both TDR and capacitive sensors require good contact between the soil and the sensor probes with no airgaps.

Geophysical methods offer the possibility to image noninvasively three-dimensional subsurface structures of soil properties and associated flow and transport processes at spatial scales ranging from soil columns to field-scale. Using electrical methods such as electrical resistivity tomography (ERT), images of the spatial distribution of the bulk soil electrical conductivity can be derived non-invasively. ERT methodology is based on the same principle as the Wenner array described above, however, consisting of large electrode arrays and are DC-based or at low frequency AC. As defined in Eq. (3), bulk soil electrical conductivity is strongly related to water content, so that ERT can also be used to map root water distributions. Resulting current flows are computed from numerical models, after which the soil electrical resistance distribution is mapped after model inversion, so that soil EC or other soil characteristics can be determined. A major difficulty though is the non-uniqueness of this reconstruction. Coupling the ERT data inversion with a process-based hydro-salinity model to inversely estimate process model parameters instead of spatial distributions of bulk soil ECs offers a way to improve this approach (Hinnell et al., 2010). Elaborate reviews on the application of ERT are presented by Furman et al. (2013) and Vanderborght et al. (2013).

Lysimeters are tools for accurately calculating water and solute balances and are successfully used in research as well as to guide decision-making for soil reclamation, fertilization or irrigation with low quality water (Raij et al., 2016). Under certain conditions, including those often found when irrigating with high salinity water in dry climates, changes in soil water storage in lysimeters are negligible for a fixed time period, so that the water balance can be calculated from irrigation and drainage only (Tripler et al., 2012). When relying on lysimeter data for salinity management with low quality water, either drainage volumes or drainage concentration can be used (Eq. 2). The drainage amount allows estimation of crop evapotranspiration, while the drainage EC measurement enables calculation of actual LF (Raij et al., 2018). The use of such lysimeter data has been attractive for decision-making purposes for hydroponics and in greenhouses using water recycling.

4. Priority 1: Need for soil salinity mapping

4.1 Introduction

Approximately 25% of total cropland is irrigated (Nachshon, 2018), producing 40% of all agricultural crops and 80% of nuts and vegetables, and accounting for near 80% of the world's total freshwater use. In addition, it is estimated by FAO that 30+ Mha of dryland agriculture is salt affected. A widely accepted number of human-induced salinized soils is about 76 Mha, with about 45 Mha by irrigation. However, the most recent global soil salinity information dates back to 1980–1990.

4.2 Past information

Although salt-affected soils are widespread and are increasingly listed as a major threat for a food-secure world, the core data still widely used originate from an outdated soil map with data collected in the 1970s (Abrol et al., 1988; FAO-UNESCO, 1980). Derived from it, the Global Assessment of Soil Degradation (GLASOD) was the first attempt to publish a world map on the status of human-induced soil degradation (UNEP, 1992). It led to a global map at a scale of 1:10 million, defining physiographic units, themselves based on expert judgment, in which type, degree, extent, rate and main causes of degradation were characterized. Among the four soil degradation classes, it included chemical deterioration defined by loss of nutrients and/or organic matter, salinization, acidification, and pollution. The GLASOD map was primarily intended as a guide for policymakers to illustrate regions of concern, and not as a highly accurate technical product.

The GLASOD data suggested that about 1 Bha of the world's soils are salt affected. These and other available estimates suggest that about 412 million ha are affected by salinity and 618 million ha by sodicity (Oldeman et al., 1991; UNEP, 1992), but these figures do not distinguish areas where salinity and sodicity occur simultaneously. Estimates of secondary salinization vary and range from 45 to 80 Mha, comprising some 20–30% of all irrigated land and 5–10% of the global salinized area, with about half of it located in the four countries of India, Pakistan, China and the United States. The global irrigated area is estimated to be around 300 Mha (FAO and ITPS, 2015; Ghassemi et al., 1995). In addition, 2% of dryland agricultural area (1500 Mha), equal to about 30 Mha is estimated to be salt-affected. Similar numbers on the extent of salt-affected soils are widely used by various reports (e.g. Ghassemi et al., 1995; Shahid et al., 2018; Szabolics, 1989), listing that 25–30% of all irrigated lands are salt-affected and that 10% of all the world's arable land is affected by soil salinity and/or sodicity. A more recent regional review of salt-affected soils was provided by Shahid et al. (2018), reporting global areas of salt-affected soils ranging from 45 to 77 Mha. It is not clear whether these numbers also include the area of agricultural land that has been permanently lost to salinization, which was estimated to be 76 Mha (IPBES, 2018). Although salt-affected soils are widespread and occur in more than 100 countries, recent statistics on their global extent are absent.

4.3 Recent information

To support the development of national strategies for food and water security, economic development and resource conservation, the need for updated soil information on global degradation was widely recognized. For this purpose, the Harmonized World Soil Database was developed (FAO/IIASA/ISRIC/ISSCAS/JRC, 2012) to improve on the FAO-UNESCO (1980) soil map. The new map comprised over 15,000 different soil mapping units that combined updated regional and national updates of soil information world-wide but was nevertheless largely based on the outdated FAO-UNESCO Soil Map of the World (FAO-UNESCO, 1980). The soil salinity map derived from this updated Soils database is presented in Fig. 5 (FAO/IIASA/ISRIC/ISSCAS/JRC, 2012) and is available on the FAO website. This updated information was largely needed to plan for land use changes that came about because of rising urban cities and growing rural populations, and to curb associated land degradation by erosion, pollution, salinity, as well as biodiversity losses.

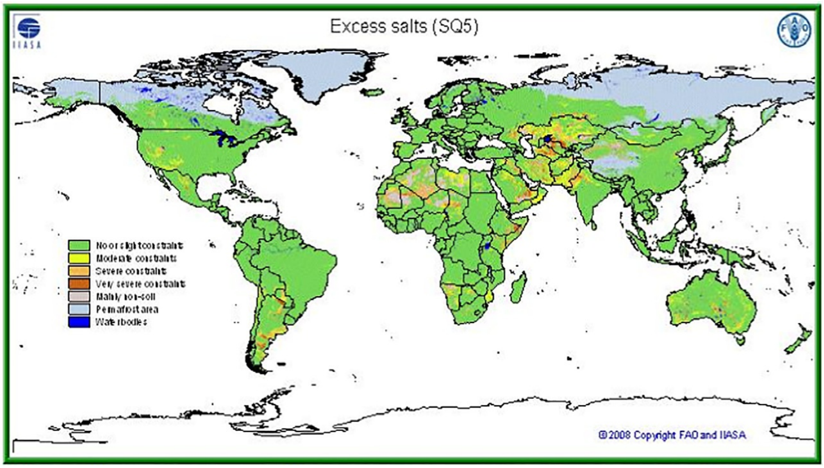

Fig. 5 Soil salinization map, as derived from harmonized world soil data base (http://www.fao.org/soils-portal/soil-survey/soil-maps-and-databases/harmonized-world-soil-database-v12/en/). An improved derivation of this map was published in Wicke et al. (2011).

More recently, FAO through the Intergovernmental Technical Panel on Soils (ITPS) published the Status of the World's Soil Resources (SWSR) report (FAO and ITPS, 2015), intended to serve as a reference document on the status of global soil resources to support studies of regional assessment of soil change. It also contains a synthesis report for policy makers that summarizes its findings, conclusions, and recommendations. The SWSR report identifies the likely rapid increase of salt-affected soils globally and estimates that currently each year some 0.3–1.5 Mha of farmland is taken out of production because of soil salinity problems. The SWSR report also states that about half of the total currently salt-affected soils are further decreasing their production potential. Annual economic costs were estimated to be about US $440 per ha of salt-induced agricultural land.

Currently available maps continue to be out-of-date and too coarse for predicting trends on soil salinization. Global estimates of salinization combine different regional estimates that are not necessarily compatible. It is already noted that percentages vary widely between various literature sources. Across the world, countries and regions typically apply different soil classification systems, and as a result the definition of saline or sodic soils varies, thus changing the acreage of salt-affected lands. A harmonized soil salinity classification system is needed that is universally applied. Gathering accurate, up-to-date information is critical for developing policies to halt the trend of increasing soil salinity across the world and regionally. Efforts to develop an updated and harmonized global soil salinity map were recently initiated by FAO through the Global Soil Partnership or GSP (Omuto et al., 2020), through mapping of soil EC, SAR, and pH using existing country-level data.

4.4 Future priorities

Soil salinity and the increase in areal extent is a serious global threat to agricultural production as soil degradation jeopardizes reaching a food-secure world. The only database that currently provides soil salinity data with global coverage is the Harmonized World Soil Database, but it is outdated and has several limitations when assessing changes in soil salinity and its areal extent. Except for a few country-focused reports, there is limited information on the world's changing extent of salinized soils. Therefore, we recommend taking steps toward a new assessment.

There are various reasons to suggest that the areal extent of soil salinization is increasing as well as becoming more severe. Information on such trends is extremely relevant as global and national policies on landuse are being developed to advance Sustainable Development Goals (https://www.un.org/sustainabledevelopment/sustainable-development-goals/) and to mitigate and/or adapt to climate change (IPCC, 2019; https://www.ipcc.ch/srccl/). Moreover, areas of salt-affected irrigated lands are inconclusive and vary between 25% and 50% (Shahid et al., 2018) depending on the data source.

Soil salinization may be accelerating for several reasons including the changing climate. Rising temperatures increase soil evaporation and crop water requirements, enhancing soil salinization in areas already prone for salinity. Especially, coastal regions will be subjected to increasing risk of salinization by rising seawater levels, thereby pushing more saltwater into coastal aquifers, and increasing groundwater salinity. In addition, the likelihood of extreme storms and tsunamis can cause flooding of seawater, resulting in saltwater infiltration into soils and contaminating groundwater resources (Illangasekare et al., 2006). In his analysis of climate change impacts on soil salinization processes, Corwin (2020) states that the consequences of climate change have been overlooked and that changes in soil salinity extent will need to be monitored and mapped. He suggests that both proximal and remote sensors are the best methods to achieve this in a timely manner.

Another reason that the area of saline soils is expanding relates to the increased use of marginal waters for irrigation, as decreased freshwater availability encourages application of treated wastewater or low salinity water for irrigation. Also, changing land uses from prime agricultural land to residential development promotes cultivation of more marginal lands, thereby enhancing the potential for land degradation. Furthermore, the decreasing availability of freshwater promotes more efficient irrigation methods such as drip and sprinkler irrigation, leading to reduced leaching of accumulated soil salts in regions with limited winter rains. Yet, to meet the world's demand for nutritious food with the rising population, one may expect a further increase in irrigated area, especially in regions where freshwater availability is adequate. Lastly, salts accumulate over extended periods of continuous irrigation, thus further causing more salinity-prone areas over time.

A universal global soil salinity map can be achieved using satellite imagery, soil properties maps, other land surface information, and advanced data analysis methods such as machine learning techniques (Section 5). A recent example of such an approach was taken by Ivushkin et al. (2019), supported by the International Soil Reference and Information Centre (ISRIC, Wageningen, the Netherlands). In their work, a total of six soil salinity maps were produced for 1986, 2000, 2002, 2005, 2009, 2016, using thermal IR imagery data from Landsat satellites. Their analysis presented a clear trend over this 20-year period, indicating that the global area of salt-affected soils increased from about 900 to 1000 Mha, at an annual rate of about 2–5 Mha/year (Fig. 6). Various limitations of their methodology were given, including the need for higher spatial resolution, more ground truth data for regions with sparse data, uncertainty associated with temperature response due to plant variations in salt tolerance, and potential improvement using machine learning techniques.

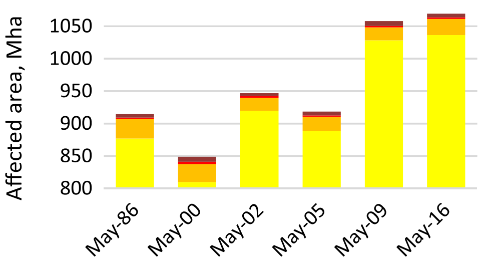

Fig. 6 Salt affected lands area between 1986 and 2016 (Ivushkin et al., 2019).

Summary: Salt-affected soils have significant impacts on the environment, freshwater availability, and agricultural production. Soil salinity maps are outdated and are not harmonized between regions or countries. Updated maps are needed to quantify soil salinization rates and to inform country level and new international policies and strategies to protect soils from further salinization.

We urge prioritizing development of remote sensing instruments for future satellite missions that focus on observing spatial and temporal changes in land degradation, including soil erosion and salinity, at a global scale.

5. Priority 2: Application of remote sensing to detect and map soil salinity

5.1 Introduction

Detecting and monitoring soil salinity across agricultural regions is needed for inventorying soil resources; for identifying trends and drivers in salinization; and for judging the effectiveness of reclamation and conservation programs. Due to the impracticality of directly measuring root zone EC_{ex} over large areas (Section 3), most regional-scale salinity assessment research has focused on alternative measures of salinity obtained through aerial photography and satellite remote sensing (RS). Despite being developed many decades ago, remote detection of salinity has not been widely used in salinity monitoring programs and has achieved only limited success to date. However, methodological and technological advances made over the last 20 years suggest the routine use of remote sensing for monitoring agricultural salinity may be possible.

Two approaches to remote salinity detection have been used: indirect and direct. With indirect methods, the level of root zone salinity is inferred based on crop growth and health, usually as indicated by canopy spectral reflectance or thermographic data. The reflectance of certain visible or infrared spectra generally differs for healthy and stressed leaves (Carter, 1993). Thus, if a correlation between root zone EC_{ex} and spectral response can be established, regression or classifier models can be developed to quantify or label soil salinity levels in a remote sensing image.

Direct methods detect salinity in bare soils based on the reflectance properties of surface salts and crusts. Sections of landscapes with and without surface salts can be distinguished due to the high reflectance of salt covered areas in the visible part of the spectrum. Within salt covered areas, salinity levels and salt types may be differentiated because of the effects that salt abundance, mineralogy, moisture, color, and surface crusting and roughness all have on reflectance (Mougenot et al., 1993). The direct approach is useful for assessing salt marshes and other highly saline, non-agricultural landscapes, as well as for tracking encroachment or appearance of barren, high salinity areas in dryland pastures and rangelands (Furby et al., 2010). However, it has less utility for agricultural regions because of the presence of extensive vegetation. Therefore, we focus on indirect RS methods for soil salinity monitoring.

5.2 Past information

By the middle of the 20th century, aerial photography and image analysis were touted as a means of inventorying crops and detecting disease (Colwell, 1956). Portable or airborne spectral reflectance instruments did not exist, but laboratory measurements made on tissues from leaves in varying states of distress could reveal, for a given crop and development stage, the portion of the spectrum most sensitive to variations in leaf health. Aerial photographs sensitive to the identified spectral range could then be made using an appropriate combination of film and lens filter. Through analysis of the aerial images, it was proposed that areas with healthy and diseased plants could be distinguished.

Myers et al. (1963) were the first to connect aerial images of crops with root zone salinity. Working in Texas cotton fields, Myers et al. (1963) found that the salinity level in the 0.3–1.2 m soil layer could be correlated with the spectral reflectance of cotton leaves, determined from aerial photographs using infrared film and a dark red filter that was sensitive at 675–900 μm wavelengths. In a subsequent paper, Myers et al. (1966) reported it was possible to distinguish five levels of salinity and to estimate with reasonable accuracy the degree of salinity in the soil profile. It was also found that soil salinity could be predicted with reasonable accuracy from leaf temperatures measured with an infrared radiometer.

Thomas et al. (1967) examined in greater detail the spectral reflectance of salt-affected cotton leaves and found that they changed during the growing season. At most wavelengths, percent reflectance from individual leaves was negatively correlated with salinity early in the year and positively correlated later. Multiple regression analyses of aerial image density indicated that under field conditions reflectance was influenced by soil salinity and percentage ground cover.

The Landsat program and launch of the first operational Landsat satellite in 1972 spurred interest in using multi-spectral satellite imagery for natural resource management (Westin and Frazee, 1976). Notable early examples of using spaceborne aircraft to detect salinity include identifying salt flats in Imperial Valley, California from photo images taken aboard Apollo 9 (Wiegand et al., 1971) and distinguishing saline from non-saline rangelands in South Texas using Skylab satellite imagery (Everitt et al., 1977). The review of Metternicht and Zinck (2003) covers advances made during this period with respect to direct observation of visible surface salts.

With the growing availability of multi-spectral reflectance data from satellites and other platforms, it became common from the 1970s onward to quantify multi-band canopy reflectance using vegetation indices such as the Normalized Difference Vegetation Index, NDVI = (NIR − R)/(NIR + R), where R and NIR are spectral reflectance in the visible red and near-infrared bands, respectively. Wiegand et al. (1992) used imaging data from the SPOT-I satellite to evaluate the relationship of NDVI and the Greenness Vegetation Index (GVI) to plant growth and yield in a single salt-affected, irrigated cotton field in Texas. Later, Wiegand et al. (1994) determined NDVI and GVI for four cotton fields in San Joaquin Valley (SJV), California using airborne photographic imagery made with multiple lens filters. Regression equations with NDVI and GVI as predictor variables were used to estimate salinity at about 100,000 pixels per field.

5.3 Current information

The last 2 decades have seen a steady increase in the availability of remote sensing data, in the capabilities of various sensors and platforms, and in remote sensing applications. Even with improved technologies, a major problem with indirect salinity detection methods is that a single image generally cannot differentiate salinity-induced crop stress from stress caused by other factors such as weather, pests, and water management. Lobell et al. (2007, 2010) addressed this difficulty by evaluating multi-year data, hypothesizing that soil salinity is relatively constant compared to other more transient stressors. Lobell et al. (2007) found that using 6 years of reflectance data greatly improved the correlation between salinity and wheat yield, whereas Lobell et al. (2010) successfully evaluated regional-scale salinity using a 7-year average enhanced vegetation index (EVI) derived from satellite

MODIS (Moderate Resolution Imaging Spectroradiometer) data. Multi-temporal data was also used by Caccetta (1997) and Furby et al. (2010) for improved soil salinity classifications. Along the same lines, Zhang et al. (2015) used interpolated and integrated vegetation index time-series data as an explanatory variable rather than analyzing single-date data. Whitney et al. (2018) later applied the same integrated index method to the SJV and concluded that multi-year data further enhanced correlations with soil salinity.

The use of environmental covariates as additional predictor variables in regression equations and classifiers has also improved accuracy (Caccetta, 1997; Furby et al., 2010; Taghizadeh-Mehrjardi et al., 2014). Scudiero et al. (2015) developed a linear regression equation for estimating soil salinity (expressed as EC_e) using spatial precipitation and temperature data, crop-type data, and multi-temporal Landsat 7 ETM+ canopy reflectance data. They calibrated their model using data for thousands of Landsat 7 pixels at 30 m resolution across 22 fields for which ground truth salinity data were available (Scudiero et al., 2014). For each 30×30 m Landsat pixel, average root zone (0–1.2 m) EC_e for a 6-year period was modeled using the Canopy Response Salinity Index, CRSI, which combines spectral reflectance in the green, blue, red, and near-infrared bands.

Rather than spectral reflectance, Ivushkin et al. (2017) used satellite thermography to assess soil salinity in salt-affected cropped areas in a semi-arid province of Uzbekistan. They found that correlations between soil salinity and canopy temperature varied depending on the time of year with the strongest relation occurring for cotton in September. The thermographic approach has also been applied to larger regional- (Ivushkin et al., 2018) and global-scales (Ivushkin et al., 2019).

5.4 Future priorities

Remote sensing of salinity has moved beyond proof-of-concept, but few salinity monitoring programs utilize satellite RS. One exception is the Land Monitor under the National Dryland Salinity Program (https://landmonitor.landgate.wa.gov.au/info.php) in Australia, which tracks salinity in Western Australia. However, further research is needed to establish that RS is sufficiently accurate and cost effective for more general use. We recognize several priorities:

Data integration—With satellite imagery, trade-offs exist among spatial, temporal, spectral, and radiometric resolutions. Satellites and instruments used for indirect remote salinity detection include Landsat 7 ETM+ (30 m resolution, ~16 d return time, 8 bands, 8 bits) and Aqua/Terra MODIS (250–1000 m, 1–2 d, 36 bands, 12 bits). The most recent iterations of long-operating open satellite platforms (e.g. Landsat 8, Sentinel-2) offer improved imaging capabilities while commercial satellites such as WorldView-3 offer spatial resolutions approaching 1 m. Research is needed to integrate data from these varied platforms and technologies because each potentially captures information important for salinity detection. Canopy thermographic imagery may contain information not found in spectral reflectance images. High temporal resolution is important because spectral and thermal response varies according to phenological stage. High spatial resolution is important because salinity often varies substantially over very short distances. However, the finest possible resolution is not necessarily optimal, as correlation between remotely sensed data and soil properties may be highest at coarser resolutions. For instance, Scudiero et al. (2017) used data from the WorldView-2 satellite to examine salinity correlations in a 34 ha fallow field and determined that the relationship between multi-temporal maximum EVI and soil salinity was strongest at a resolution of about 20 m. Future research should develop multi-spatial, multi-temporal, multi-sensor data analysis pipelines to improve accuracy (Wu et al., 2014).

Crop-specific information—Research should prioritize regression and classifier models that integrate crop-specific data. As noted, spectral and thermal response to salinity stress differs by vegetation type and growth stage, but very few RS salinity models have used crop specific crop data. Exceptions include the work by Scudiero et al. (2015) who used the Cropland Data Layer (Han et al., 2012) to incorporate cropping status (fallow or cropped) into their model, and Zhang et al. (2011) who explored the possibility of incorporating crop-specific reflectance properties in their regional salinity assessments. Future research should investigate the use of crop type and growth stage as predictor variables.

Two crop categories that create difficulties for indirect remote sensing methods are (i) salt-tolerant halophytes (Section 10) and (ii) orchards and vineyards (Scudiero et al., 2016). Halophytic vegetation complicates image analysis because in contrast to the monotonically decreasing salinity response function of most agronomic crops, halophytes achieve maximal growth at intermediate salinity levels (Scudiero et al., 2015; Zhang et al., 2015). While most true halophytes have little agronomic value, there is growing interest in their use as biofuels. Orchards and vineyards are mostly excluded in salinity RS studies. For example, the salinity map of western San Joaquin

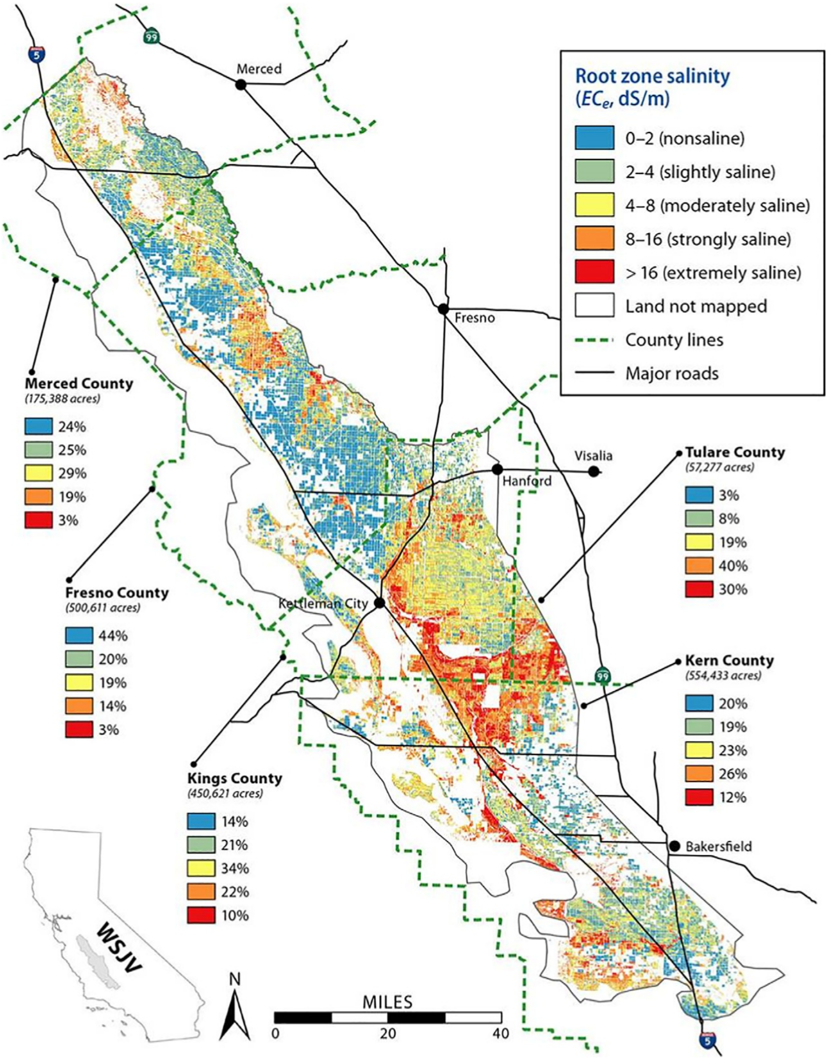

Fig. 7 Remote-sensing estimations of root zone soil salinity for agricultural soils (orchards not included) of the west side of the San Joaquin Valley (Scudiero et al., 2017).

Valley produced by Scudiero et al. (2017) covered only row and field crops because insufficient information existed for orchards (Fig. 7).

Hyperspectral imagery—Multi-band vegetation indices have been the predominant measure of canopy reflectance in RS studies. However, hyperspectral imagery potentially provides a more informative measure of crop status as potentially 100s of wavelength bands can be analyzed simultaneously.

Zhang et al. (2011) investigated hyperspectral reflectance in their salinity evaluations, but the topic is relatively unexplored. Adopting or developing new sensor technologies for salinity detection should also be encouraged, such as terahertz radiation spectroscopy (Browne et al., 2020).

Environmental covariates—Including environmental covariates in regression and classifier models improves accuracy. Emphasis should be placed on developing and validating higher resolution covariate databases. Scudiero et al. (2015) added soil texture to their regression model but found no improvement because the spatial resolution of the textural data was inadequate. Among other benefits, high resolution covariate data may improve salinity predictions at lower salinity levels where the impact of salinity on crop growth is minimal. Recently, several continental- and global-scale digital soil maps have been produced such as SoilGrids250m (Hengl et al., 2017), SoilGrids100m (Ramcharan et al., 2018), and POLARIS (Chaney et al., 2019). With resolutions of 250, 100, and 30 m, respectively, these databases potentially offer a rich source of covariate data. However, their accuracy must be assessed for different world regions.

Summary—Routine monitoring of soil salinity via remote sensing is within reach. Researchers and funding agencies should prioritize the development of: (i) multi-temporal, multi-scale, multi-instrument data analysis pipelines that integrate available satellite data and fully extract the salinity signal; (ii) new remote sensing technologies for canopies and salinity; and (iii) high-resolution covariate and ground-truth databases.

6. Priority 3: Improved soil salinity management practices

6.1 Introduction

Irrigating with water that is high in salt content requires special management practices to mitigate salinity buildup in the crop rooting zone, to minimize reduction in crop yield with associated economic losses and to mitigate environmental degradation. In addition, saline-sodic irrigation water can cause breakdown of soil aggregates, followed by the swelling and dispersion of clays particles which leads to soil crusting, loss of porosity and reduced permeability especially after rainfall or irrigation with low salinity (Rhoades et al., 1992). The degradation of alkali soils using high quality irrigation waters has been documented early on, resulting in reduction in soil infiltration (Fig. 8). We will discuss the historical evolution of improved soil salinity

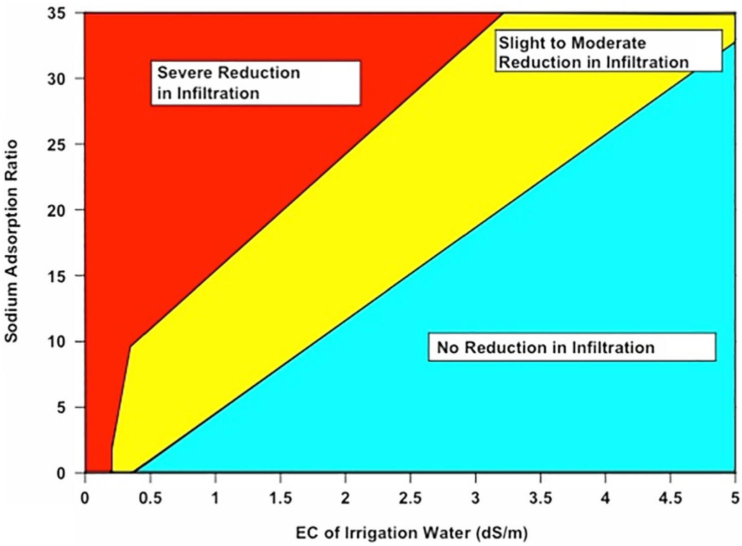

Fig. 8 Effect of irrigation water salinity and sodium adsorption ratio (SAR) on soil infiltration (Pedrero et al., 2020).

management practices in irrigation projects, followed by changes in soil salinity management strategies that have occurred in the past few decades.

6.2 Past information

Early on with the development of irrigation projects, there was general recognition that soil salinity issues had to be addressed at both the on-farm scale and at the basin or irrigation district scale. On the farm scale the focus was on agronomic and engineering practices that minimized soil salinity buildup in the root zone, while at the basin or regional scale the focus was mostly on engineering structures for water delivery and drainage. In our review we will focus on the farm-scale only, though it is realized that with few exceptions (Gill and Terry, 2016) soil salinity issues will persist when regional efforts to ensure adequate drainage facilities are lacking, and will eventually lead to the demise of civilizations and land abandonment (Hilgard, 1886; Wichelns and Qadir, 2015). We also note that most irrigation projects were designed for surface irrigation by flooding the field using gravity (furrow and border irrigation), allowing for over-irrigation to ensure that the whole field receives adequate amounts of water while satisfying the annual leaching requirement (Section 2.4). However, this has led to rising groundwater tables worldwide, further necessitating drainage capabilities. At the same time, these shallow groundwater tables can be beneficial when irrigation water supplies are limited such as in drought periods (Grismer and Gates, 1988). A succinct review by Ayers et al. (2006a) lists main criteria to assess whether in situ crop water use from shallow ground water is suitable.

To prevent the buildup of salts in the root zone, agronomic recommendations would apply irrigation water in excess of crop evapotranspiration. The excess water was commonly referred to as the leaching requirement, maintaining a field salt balance with soil salinity levels to not exceed the crop salt tolerance (Section 2.3). In situations when leaching was inadequate to prevent salt buildup in the root zone, salt tolerant crops were selected.

Seedbed preparation by tillage and higher frequency irrigation were used for sodic soils to mitigate the effects of surface crusting and to promote stand establishment. However, tillage can reduce soil infiltration through formation of a plow layer. For that purpose, deep plowing is used to break the plow pan and to increase leaching and soil water storage in the deep rooting zone (Rhoades et al., 1992). Other soil salinity management strategies included sanding, by mixing clay layers with sand from further down below, thereby improving the effectiveness of leaching, or by creating artificial subsurface barriers (Ityel et al., 2012, 2014).

Flood irrigation, though suited for irrigation with saline water because of its leaching benefit, is often associated with problems such as soil crusting and soil aeration. These are minimized using furrow irrigation, however, because of its partial wetting of the soil surface it tends to accumulate salts in the seedbed. For that purpose, annual preplant irrigations by flooding or sprinklers were applied to flush salts from the shallow root zone before or during seedling establishment.

Chemical amendments are used to replace the excess exchangeable sodium (Na) with calcium (ESP, Section 2.2) in sodic soils to improve soil infiltration (Fig. 8). In addition to gypsum, other amendments include calcium chloride, sulfur, and lime. Addition of such amendments is typically followed with a leaching irrigation to move Na and other reaction products downwards away from the rooting zone.

Soil conditioners continue to be used for management of saline-sodic soils, particularly at seedling establishment in high ESP soils or when crops are irrigated with high SAR water. Soil conditioners such as sulfate lignin were reported to improve soil aggregate stability and permeability and prevent crust formation (Rhoades et al., 1992). Also, organic manures are used to manage saline-sodic soils irrigated with lower quality water, as these promote soil aggregation and increase soil permeability. Organic manures are also used to lower soil pH by releasing CO_2 and organic acids as it decomposes, whereas the lower pH helps in solubilizing $CaCO_3$ when present, thereby increasing soil EC and replacing the exchangeable Na with Ca which lowers ESP.

6.3 Current information

In the last few decades, substantial changes have occurred in irrigation technology, irrigation water sources and cropping systems. Also, public awareness on environmental issues and their regulations have increased. Consequently, soil salinity management is changing as well.

Leaching—Leaching remains an effective management strategy to prevent salt build in the root zone. However, more recent research is showing that soil salinity leaching requirements developed decades ago (Hoffman, 1980) were based on steady state conditions and that the transient models developed later (Section 2.4) improved the prediction of the complex physio-chemical-biological dynamics in an agricultural system (Letey et al., 2011). They concluded that the current guidelines overestimated leaching requirements (LR), especially if LR are low. Most importantly, the salt concentration at a given depth is not constant with time as assumed by steady-state models, but is continually changing as water is added or extracted by the plant. Furthermore, under monsoonal conditions, rainwater mobilizes accumulated salts downwards and restores high quality soil water in the rooting zone during the growing season, thus further reducing the LR as computed by the steady state model (Minhas, 1996). The concentrated salts near the soil surface are "flushed" by the irrigation water thereby moving the salts downwards and reducing the concentration at a given depth. As a result, the concentration after irrigation near the soil surface would be close to the concentration of the irrigation water for high-frequency irrigation systems. Such findings indicated irrigation water amounts could be reduced and that more saline waters and marginal waters (drainage water, recycled water) could potentially be used for irrigation. These results were affirmed by Corwin et al. (2007) and Corwin and Grattan (2018). In addition, using both field experiments and transient numerical modeling studies, Hanson et al. (2008) showed that there is considerable localized leaching around drip systems, even at applied water volumes less than potential crop ET, as drip systems only partially wet the soil surface.

Deficit irrigation (DI)—DI consists of application of irrigation water below potential crop requirements. DI strategies such as partial root zone drying and regulated DI are used to save water and increase water productivity but will increase soil salinity when annual LF values are less than one. In a 5-year field study on peach trees, Aragüés et al. (2014) determined that this increase was counteracted by salt leaching by high LFs attained during the non-irrigation seasons and proved to be sustainable for the climatic conditions of their study area. However, in a similar study (Aragüés et al., 2015) using low-quality irrigation water they determined that long-term application of moderate saline waters would increase soil salinity in the long-term, unless unusual large volumes of irrigation water were applied in the non-irrigation season. Clearly, long-term outcomes of DI will largely depend on crop salt tolerance and climatic conditions (Dudley et al., 2008b).

Crop selection—Selecting salt tolerance crops continues to be used as a simple strategy to deal with saline-sodic soils irrigated with low quality water. For example, in the western San Joaquin valley cotton production has been replaced by pistachio, which is both salt tolerant and a high value specialty crop. However, in general there are not that many crop choices that are both salt tolerant and high value as most fruit and vegetables tend to be salt sensitive, such as lettuce and strawberries. Boron and chloride ion toxicity on woody perennials is occurring more frequently as acreage of this crops is expanding in California. Typically, more water is needed to leach boron than other salts because it is tightly adsorbed on soil particles (Hoffman and Shannon, 2006), whereas tolerances vary among species and rootstocks (Section 8). Boron concentrations in the irrigation water exceeding 0.5–0.75 mg/L have been reported to reduce plant growth and yield (Grattan and Oster, 2000; Section 10). Unlike boron, chloride moves readily with the soil water, is taken up by the plant roots, translocates to the shoot, and accumulates in the leaves. If irrigation water that is high in chloride is applied via sprinkler irrigation it can cause foliar injury (Grattan et al., 1994) and reduce yields in hot climates. Options to reduce foliage injury include (a) irrigation at night or early morning when evaporation rates are low and (b) using infrequent and large irrigation applications (Hoffman and Shannon, 2006).

Effect of irrigation systems on soil salinity management—The soil salinity pattern that develops in the root zone is a function of the water distribution pattern of a given irrigation system. Over the last 2 decades, there has been a rapid conversion from surface irrigation to pressurized irrigation systems particularly drip irrigation in places like California (Section 2.5). The rapid increase in adoption of drip irrigation has been driven by both the demonstrated ability to improve productivity and water use efficiency, as well as it is incentivized by governments.

Surface irrigation systems remain the most widely used method of irrigation around the world. Recent advances in automation and real-time data analytics for surface irrigation have demonstrated improved water use efficiency in Australia and California (Bali et al., 2014; Koech et al., 2010). Distributing applied water more uniformly across the field results in leaching of salts with less water. But traditionally, surface irrigation systems such as flood have typically had lower leaching efficiencies than microirrigation systems, because under soil saturation large fractions of applied water move through macropores thereby bypassing the salts in the smaller pore spaces of the soil matrix and aggregates. However, automated gates and SCADA (supervisory control and data acquisition) control systems can now allow flood irrigation systems to achieve leaching efficiencies like pressurized irrigation systems.

Microirrigation systems are largely preferred when irrigating with more saline waters. They have been successfully used in orchards, vineyards, and vegetable crops in many regions around the world with salinity problems, including Australia, Israel, California, Spain, and China. They are well suited because of their use of high frequency irrigation, thereby preventing dry soil conditions so that soil solution salinities remain close to that of the irrigation water, especially in the vicinity of the emitters where root density is highest (Fig. 9). The salt distribution that develops around a micro-irrigation system depends on system type, but typically salts concentrate on the periphery of the wetted bulb for a surface drip irrigation, whereas salt concentrations typically increase with soil depth for sprinkler systems. The upwards capillary movement of water from the wetted soil depth near the subsurface drip

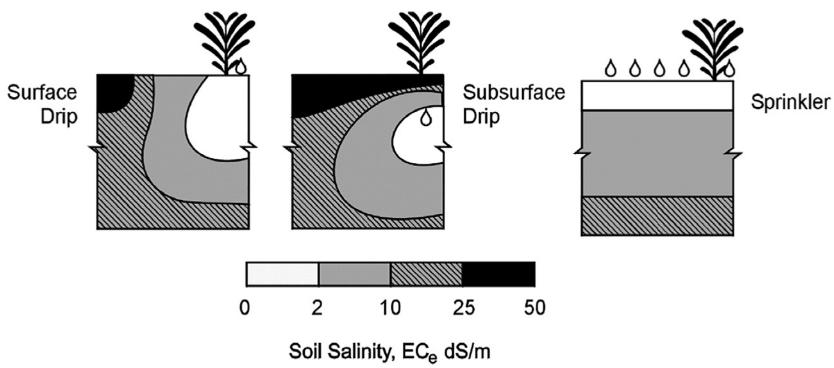

Fig. 9 Salinity distribution under drip, subsurface drip and sprinkler irrigation. *Adapted from Hoffman, J.G., Shannon, M.C., 2006. Salinity. In: Microirrigation for Crop Production. Elsevier Science, 131–161.*

emitter results in soil surface salt accumulation as water is lost through root water uptake and soil evaporation (Roberts et al., 2009). For conditions where seasonal rainfall is inadequate to push those salts near the soil surface further down, options include preseason flood irrigation or sprinkling, moving drip lines every so many years when replacing or change crop rows between seasons (Hanson and May, 2011).

However, anecdotal evidence in the San Joaquin valley orchards has shown that salinity around drip irrigation systems can limit the volume of the root zone thereby limiting nutrient uptake, particularly nitrogen. The residual nitrogen ends up being leached to groundwater either by excess irrigation or winter recharge causing environmental degradation of groundwater quality. The complex interactions between soil salinity stress and water and nitrate applications were discussed in a model simulation study by Vaughan and Letey (2015). Libutti and Monteleone (2017) suggested that since soil salinity management is bound to increase the leaching of N, best practices should optimize the volume of water needed to reduce salinity and that required to avoid or minimize NO_3 contamination of groundwater. They suggested to "decouple" irrigation and fertigation. Abating this salinity-N paradox with coupled nutrient-salt management will requires site specific considerations.

Because of the potentially high control of irrigation amount and timing, it has been shown by Hanson et al. (2009), that subsurface drip directly below the plant row can effectively be used for irrigation under shallow water table conditions as long as the groundwater salinity is low. They showed that converting from furrow or sprinkler to subsurface drip is economically attractive and can achieve adequate salinity control through localized leaching for moderately salt-sensitive crops such as processing tomatoes, eliminating the need for drainage water disposal if so relevant.

Controlled drainage (CD)—Whereas conventionally drains are installed in conjunction with irrigation systems in arid regions, controlled drainage systems originate in humid regions by control of the field water table using more shallow depth drainage laterals and control structures in the drainage ditches or sumps. In controlled drainage systems, irrigation and drainage are part of an integrated water management system where the drainage system controls the flow and water table depth in response to irrigation (Ayers et al., 2006b). Depending on objectives of the CD system, it can reduce deep percolation and nitrate concentrations in drainage water, augment crop water needs by shallow groundwater contribution, and reduce drainage water volume and salt loads for disposal.

Use of marginal waters—When freshwater resources are limited, salt tolerant crops can be irrigated with more saline water to be reused, for example by treated wastewater or drainage water. Management options include to apply irrigation water that is a mixture of saline with fresh water (blending) or cycle saline water with fresh water depending on growth stage (e.g. use freshwater for germination), by using crop rotations between salt sensitive and salt tolerant crops, depending on when more saline water is available or through the use of sequential cropping as described in Ayars and Soppe (2014). In addition to reducing freshwater requirements, it decreases the volume of drainage water required for disposal or treatment. A series of articles that present use of marginal waters has been edited by Ragab (2005). In general, research results in this issue demonstrate that waters of much poorer quality than those usually classified as "suitable for irrigation" can, in fact, be used effectively for the growing of selected crops under a proper integrated management system, as long as there are opportunities for leaching to prevent detrimental effects, such as by sodicity. Studies have shown that drip irrigation gives the greatest advantages, whereas sprinkling may cause leaf burn. Cycling strategies are generally preferred, but beneficial effects decreased under DI (Bradford and Letey, 1992). In addition, blending does not require added infrastructure for mixing the different water supplies in the desired proportions (Minhas et al., 2020b).

6.4 Future research priorities

Priority 1—For irrigation to remain sustainable, drainage must be provided. As estimated by Ritzema (2016), currently only about 22% of irrigated lands worldwide are drained. Therefore, drainage continues to deserve high prioritization and proper investments at both farm and regional scale are required. Whereas no accepted design criteria are available for controlled drainage systems, there is a pressing need for design criteria and management methods for such improved systems.

Priority 2—Development of low-cost sensors for real-time monitoring of soil water content, nutrient and salinity, and crop stress, leveraging progress in artificial intelligence and cloud computing in decision-making (see also Section 7 on precision irrigation).

Priority 3—Among the scientific tools necessary for efficient guidance of future management scenarios are advanced models capable of simulating the complex interactions between physical, chemical and biological processes taking place in the soil-plant-atmosphere system that would enable hypothesis and scenario testing to provide reliable predictions of outcomes such as negative impacts of water reuse on soils, crops and environmental quality, improved guidelines on crop tolerances to salinity at different growth stages, and response to the combined events of climate change and salinity.

Priority 4—Leveraging the soil microbiome to mitigate the negative impacts of saline soils on crop production and the environment.

Summary: To sustain irrigated agriculture optimum soil salinity management practices are key. Salinity management options have gone beyond just providing for essential field drainage and have largely expanded because of emerging technologies on irrigation method, drainage, soil and plant monitoring, and model prediction, among others. Many knowledge gaps exist in successfully applying these optimally, to benefit agricultural production, the environment and society.

7. Priority 4: Soil salinity management using precision irrigation

7.1 Introduction

This section addresses means of irrigation water and soil salinity monitoring to allow for real-time best management practices (BMPs) that maintain acceptable crop yields while minimizing environmental impacts using precision irrigation techniques. We review the early concepts of Precision Agriculture (PA) first, and define it using the Precision Agriculture's Journal definition (https://www.springer.com/journal/11119): "*Precision agriculture is a management strategy that gathers, processes and analyzes temporal, spatial and individual data and combines it with other information to support management decisions according to estimated variability for improved resource use efficiency, productivity, quality, profitability and sustainability of agricultural production.*" Subsequently, we will present the development of precision irrigation (PI) and suggest future advances that allow for real-time soil water and salinity monitoring in conjunction with adaptive management.

7.2 Past research

Precision agriculture (PA) is increasingly becoming an established farming practice that optimizes crop inputs by striving for maximum efficiencies of those inputs thus increasing profitability while at the same time reducing the environmental footprint of those improved practices. While farming has always been about maximizing yield and optimizing profitability, precision farming has allowed for differential application of crop inputs (water, fertilizers, pesticides) across the farmer's field, leading to more sustainable management. PA became possible through the broad availability of global positioning system (GPS) and geographical information system (GIS) technologies with satellite imagery in the 1980s. It was focused on achieving maximum yields, despite spatial variations in soil characteristics (soil texture, nutrient content, soil moisture) across agricultural fields. It enabled farmers to vary fertilizer rates across the field, guided by grid or zone sampling (map-based approach of PA). Therefore, inherent to precision agriculture is the use and refinement of the field soil map, in combination with soil and/or plant sensors.

Whereas early PA applications depended solely on the soil map and its refinement, more sophisticated approaches have been introduced because of the parallel development of on-the-go sensor technologies, allowing for real-time soil and/or plant monitoring during the growing season thus expanding PA toward spatio-temporal applications. For a review of a broad range of such on-the-go-sensors, we refer to Adamchuk et al. (2004), including electrical/electromagnetic (EM) and electrochemical sensors for soil salinity and sodium concentration measurements. Whereas specific electrode sensors are available to measure Na concentration in soil solution, most of the EM sensors were developed to indirectly measure soil moisture by correcting for salinity interference, or to measure bulk soil EC_b (Rhoades et al., 1976). The sole exception is the porous matrix sensor that was originally designed by Richards (1966) and reviewed by Corwin (2002), measuring directly the electrical conductivity of in-situ soil pore water through an electrical circuit with the electrodes embedded in a small porous ceramic element that is inserted in the soil. The EC measurement is solely a function of the solution salinity (EC_w) because the air entry value of the ceramic is such that it will not desaturate beyond 1 bar. Corrections are required for temperature and response time for ions to diffuse from the soil solution into the ceramic.

7.3 Current research

In their synthesis of high priority research issues in PA, McBratney et al. (2005) addressed the need to consider temporal variations, as yields typically vary from year to year. For irrigation applications, knowledge of within season variations are critical for BMP's that minimize crop water and salinity stress. This has led to the term and application of Precision Irrigation

(PI), adhering to the definition of PA but applied to irrigation practices. Whereas traditional irrigation management strives for uniform irrigation across the irrigated field, it is the goal of PI to apply water differentially across the field to account for spatial variation of soil properties and crop needs, thus to also minimize adverse environmental impacts (Raine et al., 2007) and maximize efficiencies. Moreover, PI advances allows for temporal adjustments of irrigation during the growing season because of changing weather conditions, including accounting for rainfall. PI can adjust water/fertilizer amounts because of differential tree/crop needs (e.g. deficit irrigation), by controlling both application rate and timing at the individual tree/crop level or for larger management units (zones).

PI uses a whole-systems approach, with the goal to apply irrigation water and fertilizers using the optimal combination of crop, water, and nutrient management practices. As defined by Smith and Baillie (2009), precision irrigation meets multiple objectives of input use efficiency, reducing environmental impacts, and increasing farm profits and product quality. It is an irrigation management approach that includes four essential steps of data acquisition, interpretation, automation/control and evaluation (Fig. 10). Typically, data acquisition is achieved by sensor technologies, while data interpretation would occur by evaluating simulation model outcomes, e.g. of crop response and salt leaching. Control is achieved by automatic controllers of the irrigation application system using information from both the sensors and simulation models, whereas evaluation closes the loop through adjusting the PI system.

In addition to electrochemical sensors such as specific electrodes, optical reflectance devices such as near- and mid-infrared spectroscopy methods have been developed to quantify specific soil ion concentrations, particularly soil nitrate content (Chambers et al., 2018; Ehsani et al., 2000). Over the past 20 years or so, many new soil moisture and salinity sensors have come to market, most of them being able to be included in wireless data acquisition networks (e.g. Kizito et al., 2008). Selected reviews and sensor comparisons include Robinson et al. (2008) and Sevostianova et al. (2015). Shahid et al. (2009) showed the field results of a real-time automated soil salinity monitoring and datalogging system, tested at the ICBA Dubai Center for Biosaline Agriculture. Recently there has been increased use of geophysical techniques (e.g. electromagnetic induction and electrical resistivity tomography) for delineation of PI irrigation zones and for in-season irrigation and soil salinity management (Fulton et al., 2011). For example, Foley et al. (2012) demonstrated the potential of using ERT and EM38 geophysical methods for measuring soil water and soil salinity in clay soils although they emphasized the need for calibration (see also Section 11).

Whereas traditionally, one would consider only drip or microsprinkler irrigation as a PI method, the broader definition can apply to most pressurized irrigation methods. Specifically, Variable Rate Irrigation (VRI) is applied to center pivot, lateral move, and solid set systems, as reviewed recently by O'Shaughnessy et al. (2019). Many of the aspects of PI equally apply to such sprinkler systems, however, it is noted that their inherent complexity has precluded the required development of user-friendly interfaces for decision support, lagging the engineering technology. Specifically, the need to fuse GIS, remote sensing, and other temporal information with the DSS, allowing management zones to change over the growing season (Fontanet et al., 2020). Recent evaluations on impacts of using VRI on crop yield, water productivity were presented by Barker et al. (2019) and Kisekka et al. (2017), showing potential improvements when using VRI or MDI (mobile drip irrigation), but that additional research is strongly advocated especially because of the significant increased investments required. Another limitation to date of adoption of PI is that large-scale VRI systems require many sensors which can be cost-prohibitive, whereas determining their placement and number of sensors needed is not straightforward. It is worth noting that PI can also be applied to surface irrigation systems as described in Smith and Baillie (2009). For example, automated gates coupled with SCADA (Supervisory Control and Data Acquisition) systems and real-time data analytics can be used to optimize flow rates, and advances times to ensure infiltration rates match variable soil conditions.

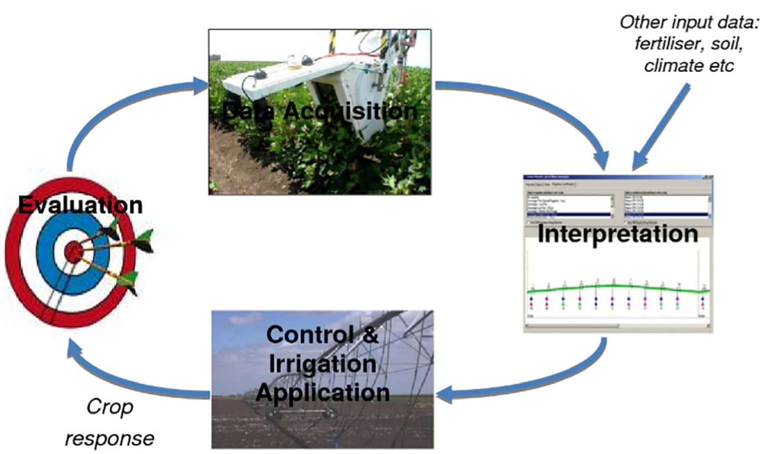

Fig. 10 Precision irrigation cycle (Smith and Baillie, 2009).

The application of PI to maintain plant-tolerable soil salinity levels was introduced by Raine et al. (2007), identifying research priorities at the time that allows for PI to be effective and pointing out that the level of precision, water application uniformity and efficiencies of most irrigation practices is suboptimal. Among identified knowledge gaps was the lack of agreement between field and model-simulated data, especially for multi-dimensional model applications such as required for drip irrigation and for spatially-variable salt and water distributions at the individual plant root zone scale. This puts into question the usefulness of computer modeling for soil salinity management purposes, especially if there is general absence of soil salinity measurements to validate model simulations. Another limitation of successful PI is the lack of information on crop root response to salinity when considering the whole rooting zone in multiple dimensions as well as on crop growth stage.

7.4 Future research priorities

A central component of a roadmap toward precision irrigation is moving from a single management point within an agricultural field toward defining management zones across the field and eventually close to a plant-by-plant level of resolution were appropriate. It requires cost-effective sensors, wireless sensing and control networks, automatic valve control hardware and software, real-time data analytics and simulation modeling, and a user-friendly and visual decision support system.

Many sensor types and technologies are being developed and are introduced for soil moisture sensing; however, few applications include soil salinity sensing in concert with soil moisture monitoring (Section 3.4). For PI to advance further, there is great need for much improved and cost-effective multi-sensor platforms that combine measurements of soil salinity with soil moisture and nitrate concentration. For a recent review of contemporary wireless networks and data transfer methods, we refer to Ekanayake and Hedley (2018), that includes the use of cloud-based databases with smart phone apps and webpages. The same authors also showed that, while development of wireless networks has focused mostly on integrating sensor technologies, there is limited research done on integrating control systems with sensor data acquisition, aimed at automating smart valve systems at the plant or tree scale. Much of this is required for advanced PI systems, allowing for high resolution control of water and nutrient application such as presented by Coates et al. (2012). In addition to ground-based sensors, there is great potential for use of airborne instruments, with the development of commercial airplane based remote sensing and UAVs (Unmanned Aerial Vehicles or drones) for agricultural applications. Especially non-contact platforms such as Electro Magnetic Induction (EMI) could potentially be used with drones for soil salinity monitoring. In addition, hyperspectral and thermal cameras can be used for plant monitoring of water or salinity stress and diseases (Jin et al., 2018).

In addition to using wireless and new sensor technologies for improved soil salinity management and control, integrating real-time sensor and control data with soil and crop growth simulation models allows for real-time management at the plant/tree scale, when combined with visualization tools and decision support systems (DDS). Recent examples of such an approach that integrates sensor information with a combined irrigation application and biophysical crop simulation model was presented by Sperling et al. (2014) and Gonzales Perea et al. (2018). However, other applications have shown the successful application of machine-learning, ANN and AI algorithms, training the DDS system using past information to improve forecasting of soil and plant status, as well as for calibration and validation (Meyers et al., 2018).

Advancing PI even further, is to combine sensing and modeling information in a single DDS system (Goap et al., 2018), to allow for adaptive irrigation water and soil salinity management. With specific attention to soil salinity, such an integrated management system would allow real-time and plant-scale or zone control of water and fertilizer application, minimizing crop water and salinity stress and optimizing yield and water use efficiency (Fig. 11).

Summary: The general absence of intensive soil salinity measurements and monitoring prohibits development of improved soil salinity management practices that maintain crop productivity while minimizing soil and water degradation. Knowledge gaps to advance PI are mostly associated with the need for cost-effective technologies that integrate soil moisture, salinity, and nutrient measurements within a cloud-based multi-sensor platform. When included to an IoT cloud-based system that integrates a wireless monitoring and control network with real-time computer simulations of soil water, salinity and crop growth, PI will allow for real-time adaptive management close to the individual plant/tree or zone scale. Soil moisture and salinity monitoring networks are cost prohibitive, as sensors are still too expensive for application at small management zones.

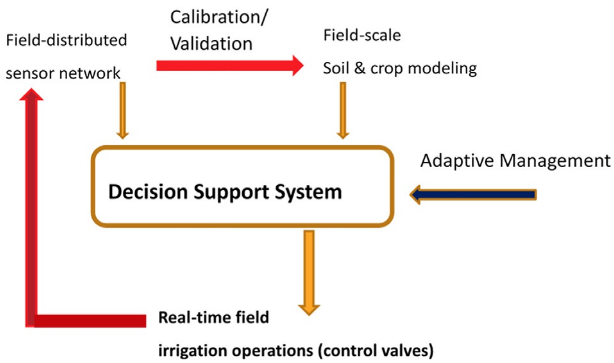

Fig. 11 Real-time sensor data integrated with simulation model forecasting are input to the Decision Support System for real-time irrigation water and nitrate application control to improve efficiencies and for adaptive management. *From Hopmans ppt to Microsoft, Seattle, 2015.*

8. Priority 5: Re-assessment of crop salt tolerance

8.1 Introduction

Crops vary widely in their tolerance to soil salinity. The physiological response of crops to salinity is related to two processes: osmotic and specific ion effects (Sections 9 and 10). These processes are dependent on each other and often impact the crop collectively (Lauchli and Grattan, 2012). Salinity reduces the osmotic potential of the soil solution (Section 2.3) thereby requiring the plant to osmotically adjust by concentrating solutes inside their cells to readily extract water via osmosis. This concentration process requires metabolic energy (ATP), but its ultimate cost to plant growth depends on ion transport efficiencies across membranes and energy requirements to synthesize organic solutes, which differs among species and varieties within a species (Munns et al., 2020a,b). As such, the efficiency of transport processes involving specific ions (e.g. Na^+) will affect the overall osmotic response. As a result, salt-stressed plants are stunted, even though they may appear healthy in all other regards (Bernstein, 1975). Both adjustment processes, i.e., accumulation of ions and synthesis of organic solutes occur but the extent by which one process dominates over the other is dependent on plant type (e.g. glycophytes versus halophytes) and level of salinity (Lauchli and Grattan, 2012). At the plant cell level, compartmentalization is critical to keep toxic ions away from locations of sensitive metabolic processes in the cytoplasm

(Hasegawa et al., 2000; Munns and Tester, 2008). Such compartmentation is controlled by transport processes across the plasma membrane and the tonoplast (i.e. vacuolar membrane), as explained in Section 10.

Specific ion effects can be directly toxic to the crop, due to excess accumulation of Na, Cl or B in its tissue, or cause nutritional imbalances. While specific ions reduce the osmotic potential of the soil solution, ion toxicities are rarely observed in annual crops grown in the field (with the exception of certain beans and soybeans), provided the ion ratios (e.g. Na^+/Ca^{2+}; Cl^-/SO_4^{2-}) are not extreme or salinity is too high. However, when Na^+ dominates the cations or Cl^- concentrations are sufficiently high, these constituents can accumulate in older leaves and produce plant injury. Specific ion toxicities are particularly prominent in tree and vine crops and injury becomes more prevalent over the years, sambut can be controlled by rootstock selection (Bernstein, 1975; Grieve et al., 2012). Specific ions can also induce nutritional disorders due their effect on nutrient availability, competitive uptake, transport, and partitioning within the plant (Grattan and Grieve, 1999). For example, excess Na^+ can cause sodium-induced Ca^{2+} or K^+ deficiency in many crops (Bernstein, 1975).

While osmotic and specific ion effects can occur concurrently, typically osmotic effects occur at early times while specific ion effects occur later (Munns and Tester, 2008). In the field, Na^+ and Cl^- toxicities can be observed in salt-affected fields after several years of tree or vine growth. Often Cl^- toxicity occurs in tree crops sooner than Na^+ toxicity as Na^+, unlike Cl^-, is retained in woody tissue, only to be released when sapwood converts to heartwood (Bernstein, 1975). The mechanisms of boron toxicity are largely unknown but the most sensitive crops to boron tend to be those classified as boron mobile plants (e.g. almonds, plums, peaches, grapes).

8.2 Past research

Rootzone salinity has traditionally been characterized by the electrical conductivity of the saturated soil paste (EC_{ex}), as promoted by the US Laboratory Staff (1954). Because crops vary in their tolerance to salinity, their tolerance was characterized by developing simplistic models to predict their relative yield in the field as a function of seasonal average rootzone salinity. The most comprehensive survey was made in the 1970s by USDA Salinity Laboratory scientists (Maas and Hoffman, 1977) by analyzing research from throughout the world that described well-executed field salinity studies on a wide range of crops. When comparing studies, they understood that absolute yield was an unreliable parameter to compare crop types grown under a range of different conditions. Rather, they described crop salt tolerance as a function of relative yield (RY) decline across a range of salt concentrations as measured using EC_{ex}. Maas and Hoffman (1977) assessed salt tolerance on the basis of two parameters: (1) a "threshold" parameter (t) which is the maximum root zone salinity, expressed as EC_{ex}, (dS/m) that the crop can tolerate above which yields decline and (2) the "slope" (s) which describes the rate by which yields decline with increased soil salinity beyond the "threshold," expressing the percentage of expected yield reduction per unit increase in salinity above the threshold value (Fig. 12). Therefore, for soil salinities exceeding the threshold of any given crop, RY can be estimated using the following expression:

$$\mathrm{RY}\ (\%) = 100 - s(EC_{ex} - t) \tag{4}$$

The greater the threshold value and smaller the slope, the higher the salt tolerance. The salinity coefficients are determined by non-linear least-squares statistical fitted to a set of experimental data. A current up-to-date listing of "salinity coefficients" t and s is published in Grieve et al. (2012), showing that many agronomic grain crops are more salt tolerant than most horticultural tree and vine crops.

8.3 Current research

As indicated earlier, soil salinity adversely affects plants by a combination of mechanisms, including osmotic influences, toxic ion effects (i.e. chloride,

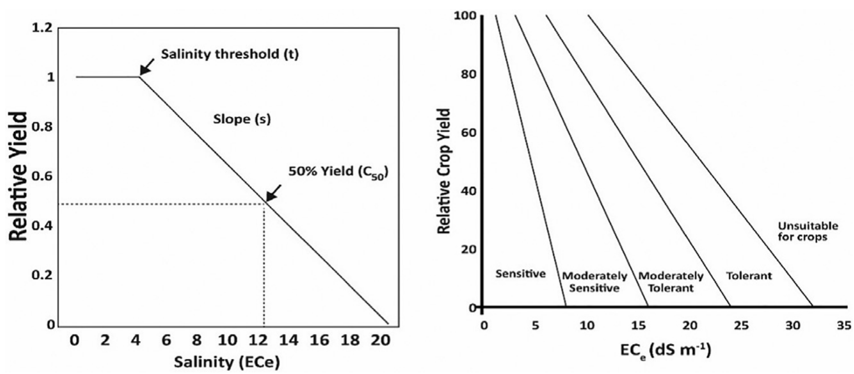

Fig. 12 Salt tolerance parameters (left) and salt tolerance categories (right) described by Maas and Hoffman (1977). *Adapted from Shannon, M.C., Grieve, C.M., 1999. Tolerance of vegetable crops to salinity. Sci. Hortic. 78, 5–38.*

sodium, and boron) and nutritional imbalances (Lauchli and Grattan, 2012). The most relevant one depends on the crop, its growth stage, duration of salinity exposure and environmental conditions (Munns and Tester, 2008), so that salt tolerance is difficult to quantify. For example, ion toxicity in tree and vine crops becomes more pronounced over the years with foliar injury particularly prominent later in the season.

Because of the many factors affecting soil salinity tolerance, there is considerable uncertainty regarding the yield threshold values, as they lack physiological justification. Despite investigators controlling salinity and minimizing all other stresses that could affect yield, the standard errors associated with the 'threshold' values can be 50–100% (Grieve et al., 2012). Obviously, these large percentages represent considerable uncertainty and suggest that true "threshold" values do not exist (Steppuhn et al., 2005a,b). Instead, it has been suggested by van Straten et al. (2019b) to substitute it with a soil salinity parameter, ECe_{90}, that equates to 90% yield. Others have developed non-linear expressions to improve on the physiological response of plants to salinity stress (Van Genuchten and Gupta, 1993; Steppuhn et al., 2005a, b).

We note that most crop salt tolerance data come from field plot studies or greenhouse experiments. Crops in most these studies were irrigated frequently, using high leaching fractions to avoid crop water stress. This was done intentionally to create a uniform soil salinity profile across the rooting zone that remained approximately constant during the growing season. In this way, one could compare salinity tolerances among crop species and rank their sensitivity and explains why most salt tolerance models fit such data very well.

While creating uniform, steady state rootzones experimentally, such uniform profiles are uncharacteristic for an irrigated field (Homaee and Schmidhalter, 2008). Field soils develop characteristic salt distribution patterns that vary with soil depth and irrigation type (Fig. 9). These patterns are a result of water movement via gravitational and capillary action and subsequent root water extraction and soil evaporation. Under sprinkler or border irrigation, the salinity increases with soil depth while under furrow or drip, salinity increases horizontally in the direction of water flow in addition to their increases in the depth direction. Furthermore, soil salinity is affected by rainfall patterns during the growing season, whereas crop salt tolerance can be affected by soil structural changes due to sodic conditions. Under such conditions, three-fold variations in wheat yield were determined for similar soil profile (Minhas and Gupta, 1993). The accumulation of salts *vis-à-vis* their osmotic effects is further modified as a function of soil texture, agro-climatic conditions, ionic constituents of salinity, and soil-irrigation-crop management strategies which impact salt tolerance limits of crops under field conditions (Minhas, 1996).

The current salt tolerance data are based on crop response to saturated soil extract (EC_{ex}) measurements, whereas the crop is responding to the salinity of the soil water (EC_{sw}) in situ, which is continuously changing over space and time. Over the past several decades it is noted that agricultural irrigation is increasingly shifting from conventional surface irrigation methods to pressurized systems that are more efficient (i.e. drip and sprinkler; Section 2.5). Studies have shown that crops with high-frequency irrigation are more tolerant to salinity than using conventional irrigation methods (Bernstein and Francois, 1973; Hillel, 2005; Rawlins and Raats, 1975). Though the wetted root zone is typically much smaller than for low frequency surface irrigation, under high frequency drip irrigation, the salinity of the soil water near the dripper is close to that of the irrigation water with the water content close to field capacity. Therefore, the roots are exposed to a lower soil salinity than for conventional irrigation practices. While the wetted soil volume is smaller, high frequency irrigation allows the crop to maintain its crop water needs.

8.4 Future outlook

The more recent change to pressurized irrigation puts into question the current validity of historical soil salinity tolerance data using the concepts of Eq. (4) developed for conventional surface irrigation systems (Letey et al., 2011), as the root-accessible soil water is near that of the irrigation water salinity using high frequency irrigation. Alternatively, one may think about measuring real-time salinity in situ at multiple soil depths based on the depth-dependent root distribution.

The non-uniform conditions of the irrigated soil complicates how best to characterize the rootzone in their response to soil salinity, as the roots are exposed to changes in soil water content and salinity in different parts of the profile. It has been recognized for decades that the major root activity is found in the least saline portions of the soil profile (US Soil Salinity Laboratory Staff, 1954). Consequently, it has been shown that shoot biomass can be 3–10-fold higher in heterogeneous soil profiles than under equivalent homogeneous salinity conditions, equal to the average rootzone salinity of the heterogeneous soil (Bazihizina et al., 2012).

Experiments with alfalfa indicated that root water uptake rate reacts to soil salinity, but that additional factors such as root activity and evaporative demand can become more important in controlling uptake patterns (Homaee and Schmidhalter, 2008). Roots will grow and develop in the most favorable portions of the rootzone considering factors such as salinity, water content, nutrients, pH, oxygen availability, soil strength, and disease pressure. For example, soil salinity may be low in the upper portion of the soil profile but soil water content (i.e. matric potential) will vary widely due to higher root length density there. In the lower portion of the soil profile the salinity can be substantially higher (i.e. low osmotic potential) but water content is higher and fluctuates less due to lower root activity. Other experimental and modeling studies have shown that the sensitivity of plants to salinity depends on the evaporative demand (Groenveld et al., 2013; Perelman et al., 2020). When multiple stresses occur simultaneously, the dominant stress largely controls crop growth and response (Maas and Grattan, 1999; Shani et al., 2007). Likewise, release of the most dominant stress will promote the most growth.

The root's developmental response to a combination of variable stresses is remarkable (Rewald et al., 2013), yet there is considerable uncertainty how the plant integrates multiple stresses over space and time (Section 9) and it remains a huge knowledge gap. More research is needed to better understand the physiological mechanisms underlying plant water relations and shoot ion regulation in plants under heterogenous salinities (Bazihizina et al., 2012) and how roots can adapt over the growing season with changing soil conditions. While there will likely be complex interactions, it is nonetheless an important area of future research.

Summary: Crop salt tolerance data are urgently needed for micro-irrigated crops, rather than using historical information developed for surface irrigation. Though of tremendous value in the past, soil saturation extracts do not necessarily represent in-situ root zone salinity. In addition, there is considerable uncertainty how the plant integrates multiple stresses across the rooting zone and during its growing season and it is a huge knowledge gap. As new cost-effective sensor technologies are being developed, they may be applied across field trials, thereby much better representing real-time and in situ information on the plant's response to soil salinity, together with other relevant abiotic and biotic soil and plant measurements.

9. Priority 6: Improved understanding of the combined and interactive effects of crop drought and salinity stress

9.1 Introduction

Drought and salinity are the two most common abiotic stressors in agricultural crops and their simultaneous occurrence is relatively common in irrigated fields. In addition, the use of saline or brackish water or the reuse of treated effluents for irrigation is expanding (Hamilton et al., 2007), particularly in arid and semi-arid regions with an increased pressure on water resources (Kan and Rapaport-Rom, 2012). Although the proper understanding of crop response to combined water and salinity stressors is a key question for hydrological and crop modeling, little is known about how the combination of these two stressors affects plant health and crop development, transpiration, dry matter accumulation and yield. In this section we will focus primarily on plant root water uptake, whereas the associated impacts of soil salinity on crop yield is treated in Section 10.

Plant root water uptake is controlled by potential gradients across the soil-root interface, and is generally described by a Darcy-type flow equation, with flow into the roots driven by a combination of matric potential and osmotic potential gradients (Δh_m and Δh_o, respectively), multiplied by a conductance coefficient. Water fluxes into the plant root will reduce both because of decreasing potential gradients, for example due to salt accumulation in the rhizosphere, and by decreasing soil and plant conductance, for example, as the soil dries (Hamza and Aylmore, 1992). At low soil matric potentials, Gardner (1960) and Cowan (1965) showed that water uptake is reduced by a local drop in soil hydraulic conductivity at high water potential gradients. This physical process of root water uptake is summarized by the radial flow equation across the soil-root interface:

$$J_r = AL(\Delta h_m + \sigma \Delta h_0), \tag{5}$$

where J_r is the radial water flux ($cm^3 d^{-1}$), L denotes the effective conductance (d^{-1}), A is the root surface area (cm^2) and σ defines the reflection coefficient (−) representing the effectiveness of the osmotic potential gradient as a driving force to move water across root cell membranes without the dissolved ions. Plants respond to diminished water uptake by stomatal closure in the leaves thereby reducing transpiration rate. Consequently, photosynthesis, plant primary production and crop yield will also decrease (de Wit, 1958). A specific mechanism of plants to cope with either soil salinity or water-stress is through osmotic adjustment or osmoregulation, by way of either accumulating salts into the plant cells, or by synthesizing organic solutes within the plant such as done by halophytes (salt tolerant plant species), thus osmotically decreasing the internal total water potential and increasing the capacity to take up water from a dry or saline soil (Section 10).

The mechanism leading to salinity stress shows two important differences when compared to drought stress. First, although a high salt content leads to an additional osmotic tension analogous to a reduced matric potential in a drying soil, osmotic tension does not affect soil hydraulic conductivity, but may change flow paths in the root conducting vessels. Second, when salt concentrations rise in plant tissue, ion-specific physiological toxicity might occur (Munns and Tester, 2008; Section 8). This toxic response is much slower than its rapid osmotic response (Ben-Gal et al., 2009a; Shani and Ben-Gal, 2005). Plant water stress by these ion-specific mechanisms are not represented by Eq. (5). Additional mechanisms include ion exclusion by the root, partitioning of salts to older leaves, and other salt isolation/sequestration processes outside the plant cell (Lauchli and Grattan, 2012; see Section 10). Much of past research considers water and salinity stress separately, whereas there remains much uncertainty about the plant's response and interactions to both stresses when they occur simultaneously. Moreover, when studying salinity stress, one most often takes account of the total soil salinity without including specific ion effects on the crop's plant physiology which typically are more long-term (e.g. Na and Cl).

9.2 Past research

As transient one-dimensional potential-based unsaturated water flow models were developed (Section 2.4), there was need to include time and soil depth dependent plant root water uptake models that simulate the combined impacts of drought and salinity stresses on crop transpiration and yield as a function of soil depth and during the growing season. These models were classified as either process-based (Type I) or empirical (Type II), as introduced in Section 3. The process-based models (also defined as microscopic, mesoscopic, additive, or Type I models) included the Darcy-type approach of Eq. (5) to represent root water uptake (Whisler et al., 1968) for vertical-dominated soil water flow (z-direction), which can be written as follows, when expressed per unit bulk soil volume, V_s (cm^3):

$$S(z) = J_r/_{V_s} = -AL/_{V_s} = \kappa(z)/_{V_s}[(\Delta h_m + z + h_o)], \tag{6}$$

where S is the root water uptake rate (cm^3 water cm^{-3} soil d^{-1}) and $\kappa = AL$ denotes the depth-dependent effective root-soil conductance (cm^2 d^{-1}), being a function of the soil hydraulic conductivity and the depth-dependent relative root distribution. Their model neglected the osmotic potential in the plant root and assumed that the root reflection σ was equal to one. Nimah and Hanks (1973) further assumed that the root resistance term was represented by the gravitational potential (positive upwards, cm), whereas others proposed to include the radial root resistance in the soil-plant conductance (Grant, 1995). This approach has been successfully included in numerical soil water flow models to assess the effect of combined drought and salinity stresses on crop yield (Bresler and Hoffman, 1986; Childs and Hanks, 1975; Lamsal et al., 1999).

Empirical Type II approaches define a reduction function, $\alpha(h_m, h_o)$ that varies between 0 (zero water uptake) and 1 (potential water uptake, no stress) to express the effects of soil stressors on uptake and transpiration. Effectively, the reduction function is used to estimate root water uptake rate, S (d^{-1}), as a fraction of the respective potential uptake rate, S_p (d^{-1}), in a soil layer according to

$$S(z) = \alpha(h_m, h_o)S_p, \tag{7}$$

where S_p is usually defined as a function of the depth distribution of root density and potential plant transpiration (Feddes and Raats, 2004). However, the functional form of the reduction function that combines both soil water matric and osmotic potential effects has been a longstanding topic of debate. Most frequently one would consider a multiplicative impact of drought and osmotic stresses, such as

$$\alpha(h, h_o) = \alpha_w(h_m) \times \alpha_s(h_o), \tag{8}$$

where $\alpha_w(h_m)$ and $\alpha_s(h_o)$ define the drought and osmotic stress functions, respectively. Several α_w functions have been proposed in literature, both as a function of h_m (Feddes et al., 1976) or of water content (Vanuytrecht et al., 2014). Salinity reduction functions α_s typically depend on the soil water electrical conductivity (Maas and Hoffman, 1977) or the osmotic potential (van Genuchten and Hoffman, 1984), and are further discussed in Section 8.

Alternatively, single stress response function models have been proposed in which the effects of h_m and h_o are weighted through either adding (van Genuchten, 1987) or multiplying (Dirksen and Augustijn, 1988; Homaee, 1999; Homaee et al., 2002a,b,c,d) the weighted potentials relative to their potential values at which their corresponding α-values are equal to 0.5. We note here that such approach is very similar to that of the functional expression that describes soil salinity tolerance by Maas and Hoffman (1977) as presented in Section 8. However, assumptions between these two approaches are very different, which explains discrepancies between them (Skaggs et al., 2006b).

Some of these reduction functions have been used with potential-based (Richards equation) hydrological models (Simunek et al., 2016) to investigate the combination of drought and saline stress (e.g. Homaee et al., 2002c; Pang and Letey, 1998). Comparisons between the different approaches sometimes led to contrasting conclusions. Cardon and Letey (1992a,b) integrated mechanistic and empirical combined stress models into a numerical solution of the Richards equation and showed that a weighted sum reduction function performed better than the additive model. They showed process-based approaches to be insensitive at low salinity levels. Homaee et al. (2002c) performed a comparison of six empirical reduction functions, either multiplicative or of other mathematical shapes and showed that a combination of linear stress functions performed best, although all tested empirical functions performed satisfactorily.

It must be noted that, besides these process-based and empirical models developed in the soil hydrology literature, different approaches have been developed in other disciplines. For example, Castrignanò et al. (1998) proposed empirical functions in their crop model that link soil water salinity level and availability a to predawn plant water potential, which is non-linearly related to a plant stress index.

9.3 Recent research

With the rising of computational capabilities over the last two decades, numerical models have become increasingly powerful and complex (Section 2.4), allowing for simulating multi-dimensional process-based unsaturated water flow (Richards equation) and chemical transport (convection-dispersion equation), including multi-dimensional root water uptake using either Type I (single root scale) or II (root zone scale) approaches. Despite their ability to reproduce experimental data (Homaee et al., 2002c; Skaggs et al., 2006b), the use of empirical reduction functions (Type II) has been subject to much criticisms (Skaggs et al., 2006a).

Issues with empirical approach—First, empirical uptake functions are difficult to verify experimentally. They are integrated within a specific hydrological model and are difficult to evaluate because their parameterization thin dependent experimental setups are time-consuming. Consequently, alternative functional forms of the reduction functions are hard to discriminate and are likely non-unique.

Second, although plant responses to drought and salinity stresses show similarities (Munns and Gilliham, 2015) to decreasing soil water matric or osmotic potentials, summing or multiplying the effects of matric and osmotic potentials on plant transpiration reduction has experimentally been shown unfeasible (Homaee et al., 2002c). Such experiments focusing on abiotic plant stressors usually address a single stress factor as they are time-consuming and expensive, thereby avoiding other stressors to act simultaneously. The physiological response of a crop to a combination of stressors cannot simply be deduced from the combined responses to each of the individual stresses (Ahmed et al., 2015; Iyer et al., 2013; Sun et al., 2015) as physiological mechanisms to different stressors act simultaneously (e.g. Rollins et al., 2013).

Third, empirical functions often assume h_m and h_o to be bulk variables, while important single-root-scale gradients exist in the soil's rhizosphere. Rather than bulk variables, plants are sensitive to water potential values at the soil-root interface. In addition, the magnitudes of these gradients are a function of the transpiration demand, root length density and soil type, thus water uptake reduction functions should depend on these variables as well (see Fig. 13A). In this context, Schröder et al. (2014) used a 3D mechanistic model to demonstrate that the relation between transpiration and bulk water potentials (i.e. the sum of gravimetric, osmotic and matric potentials) cannot be extrapolated and is affected by soil type, evaporative demand, salt content or root density, which further question the use of additive models (Fig. 13B).

Fourth, the stress functions have been determined in experiments with homogeneous conditions in terms of water and/or salt, performed in greenhouse or in other controlled environments. However, for field conditions water and salt fluxes may be very variable temporally and spatially, and their distribution patterns are dependent on irrigation type, soil properties, plant properties, and climate.

An additional issue is related to the ability of plant roots to compensate the reduction of uptake in one root zone layer by increased water uptake in another layer where soil hydraulic conditions are more favorable

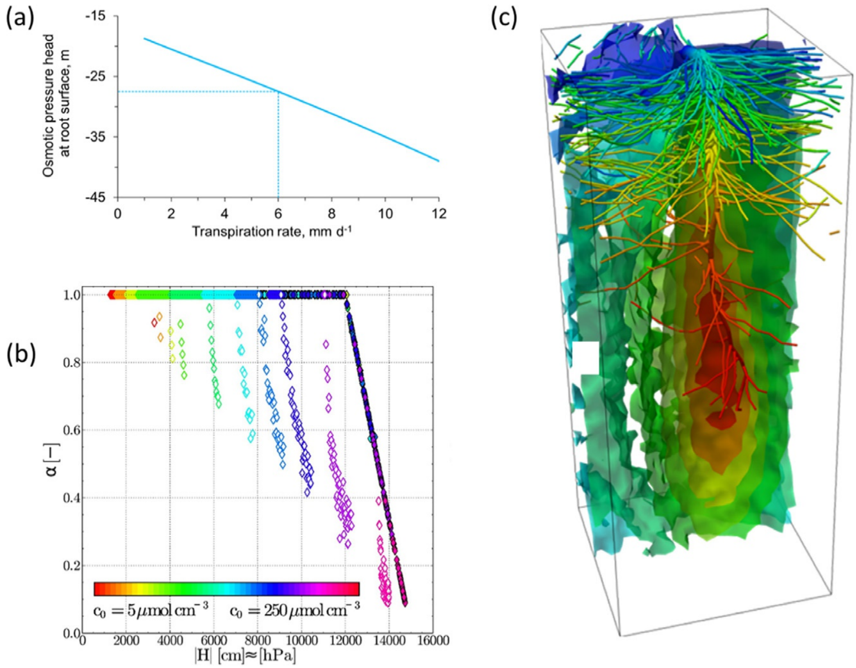

Fig. 13 (A) Impact of osmotic potential at the root-soil interface on plant transpiration rate (De Jong van Lier et al., 2009); (B) additive reduction function, depending on solute concentration (color scale) of bulk soil (open diamonds) or at soil-root interface (closed diamonds), as shown by Schröder et al. (2014); and (C) salt distribution (color scale) as simulated by R-SWMS (Javaux et al., 2008), based on spatially variable root segment hydraulics and soil water content.

(Jarvis, 1989; Lai and Katul, 2000; Li et al., 2006). Despite significant attempts (Jarvis, 2011), compensated empirical models have conceptual limitations (Skaggs et al., 2006a), because they link root water uptake compensation with stress parameters, while these are intrinsically different processes (Javaux et al., 2013). From a process-based point-of-view, plant roots are continuous and hydraulically connected so that the uptake by each root segment in the root zone depends on the status of water potential across the entire root system. Process-based approaches whether one-dimensional (de Jong van Lier et al., 2008, 2013) or multi-dimensional (Huber et al., 2015; Javaux et al., 2008) inherently account for root water compensation, as they implicitly consider the spatial distribution of the water potential gradients across the whole soil-root system.

Another relevant issue comes about because flow paths of water taken up because of matric potential gradients (convection) are different from those driven by osmotic gradients (diffusion). Therefore, values of hydraulic conductance must be considered separately between uptake mechanisms of convection and diffusion (see review by Hopmans and Bristow, 2002). Moreover, it is expected that the magnitude of the reflection coefficient as defined in Eq. (5) is plant species dependent. When salinity levels are high, it is most probable that many crops are not able to sufficiently reflect ions, thus exposing themselves to toxic effects (Shani and Ben-Gal, 2005; Sheldon et al., 2017). Under such conditions, physiological toxicity most likely dominates the crop stress, rather than osmotic stress.

Considering crop physiological mechanisms and responses to drought and salinity stress, plants have shown to avoid toxic levels of salinity in plant cells by regulation of root hydraulic conductivity through aquaporins (Boursiac, 2005), in addition to ion exclusion by the roots (Munns and Tester, 2008). Aquaporins are membrane proteins that form pores in the root cell membrane, thus increasing its conductance and facilitating the transport of water between cells (Javot and Maurel, 2002). It has been shown that osmotic stress can cause aquaporin down-regulation in many species, allowing the plant to protect itself from physiological damage from excessive salt uptake (Carmen Martínez-Ballesta et al., 2003; Carvajal et al., 1999; Martre et al., 2002; Vaziriyeganeh et al., 2018).

Argument for process-based—As a consequence of these shortcomings, a gradual shift has occurred moving from empirical-experimental research focusing on multiplicative or additive reduction functions in macroscopic root water uptake modules (Dudley and Shani, 2003; Skaggs et al., 2006a; Wang et al., 2012, 2015) toward the development of process-based numerical models for root water uptake, allowing for increased insight into the combined effects of drought and osmotic stresses.

At present, one of the challenges in the modeling and prediction of crop drought and salinity stress is the upscaling from a single rootlet to a root system (Feddes and Raats, 2004). To address this challenge, De Jong van Lier et al. (2009) developed a mesoscopic mechanistic model for root water uptake, showing that using the matric flux potential with a lower integration bound defined by the bulk soil osmotic potential was a powerful approach for computing relative transpiration under combined drought and osmotic stress without having to include compensation mechanisms. Both Javaux et al. (2008) and De Jong van Lier et al. (2009) demonstrated the plant's sensitivity to water and salinity stress at the soil-root interface in the rhizosphere, rather than of the bulk soil, especially at high transpiration rates. Similar findings were experimentally and numerically corroborated by Riley and Barber (1970), Simha and Singh (1976), and Perelman et al. (2020).

Plants are continuously facing changing distributions of water, nutrient and salts in the field and respond to them by continuously adapting their uptake, growth, and conductance. Functional-structural plant models (Fig. 13C) have therefore been built to represent plant development and functions in three-dimensions (Dunbabin et al., 2013; Javaux et al., 2011). These complex models have been used to investigate combined stresses and derive effective stress functions (Schröder et al., 2014). For instance, Jorda et al. (2018) attempted to reconcile mechanistic 3D modeling to plant macroscopic stress functions and showed that the parameters defining $\alpha_s(h_o)$ of van Genuchten and Hoffman (1984) were not unique and highly dependent on root length density and potential transpiration rates at the same root zone salt concentration, with local soil-root interface concentration values depending on uptake rates.

9.4 Future research priorities

Physiological understanding of the impact of the combined occurrence of water and salinity stress on plants is still in its early stages. Toxicity, drought, and salinity affect plants through different hydraulic and chemical signals, producing different metabolites that generate separate physiological reactions (Suzuki et al., 2014). In addition, plants react to stresses by altering their membrane permeability (Gambetta et al., 2017) and rhizosphere hydraulic properties (de la Cantó et al., 2020). These processes will change the soil-plant conductance and the membrane reflection coefficient and impact plant transpiration and growth dynamics. Integration of these processes into plant-specific functional-structural models is required for improved versatility and performance of hydrological and crop growth models.

In addition, innovative experiments are needed to better understand the integrated plant response to water and salinity stress. For that purpose, high resolution geophysical methods (such as magnetic resonance imaging and neutron tomography) open new avenues to better quantify local concentration and potential gradients around roots and in plant tissues (Koch et al., 2019; Sidi-Boulenouar et al., 2019). Their application to plants subjected to combined stresses may help to visualize the spatio-temporal evolution of local gradients as a function of root and soil properties, evaporative demand, and soil salinity. Clearly, significant progress in this research area requires collaborations between soil and plant scientists by conducting joint studies of the soil-plant system.

Summary: Despite their simplicity, empirical reduction functions are useful tools to study the impacts of drought and salinity stresses for large-scale hydrological models. However, they present important shortcomings when combined stresses are considered because both rhizosphere salt concentration and water potential play an important interactive role. Currently, detailed process-based approaches are available, which can be used to improve our understanding on how combined water and salinity stresses impact plant transpiration and growth. These complex models could also be used to parameterize simpler, effective reduction functions.

10. Priority 7: Need for broader understanding of physiological mechanisms for adaptation to saline soils

10.1 Introduction

Plant physiologists for decades have been seeking reasons for why some crops and pasture species can grow in saline soils and produce a profitable yield, while others are severely injured or die. A comprehensive survey and comparison of yield responses to increasing soil salinity reported by Maas and Hoffman (1977) was cited in Section 8 (see also: http://www.ars.usda.gov/Services/). Here we focus on the genetic and physiological aspects of soil salinity tolerance, and suggest that knowledge gaps still remain, including cell-specific functions of transporter genes, specific effects of boron, and adaptions to soils that are sodic or have very high pH.

10.2 Past research

Explanations for differences in plant response to saline soil have been studied for many decades, focused on responses to the osmotic effects of a saline soil and the specific ion effects (Bernstein, 1975; Greenway and Munns, 1980). A brief discussion of these two responses is in the Crop Tolerance section earlier. Bernstein and colleagues of the US Salinity Laboratory (https://en.wikipedia.org/wiki/U.S._Salinity_Laboratory) showed that for many of the more salt-sensitive crop species, such as rice and wheat, cultivars with lower rates of salt accumulation in the leaves yield better in saline soils, leading to the concept of ion exclusion as a mechanism of salt tolerance. Grafting experiments showed that salt accumulations in leaves was controlled by roots. Yet many of the more salt-tolerant crop species, such as barley, beet, and cotton, accumulated much more salt in their leaves compared to salt-sensitive species. Plant physiology research provided the explanation for this paradox, as outlined below.

For a plant to continue growing in saline soil, osmotic adjustment in all plant cells is essential. Osmotic adjustment is a reduction in the osmotic potential within cells to match the reduction in osmotic potential of the salts in the soil solution. Osmotic potential is a colligative property meaning that the reduction in osmotic potential is due to an increased number of total solutes in plant cells; the solutes can be individual ions that are taken up from the soil, or organic solutes that are manufactured by the plant. The osmotic potential is the same in all parts of the cell, but the proportion of ions to organic solutes differs in different cell compartments. The vacuoles adjust mainly with ions, and the cytoplasmic compartments mainly with organic solutes. Fig. 14 shows a typical plant cell with a central vacuole surrounded by a thin protein-rich (matrix) layer of cytoplasm which contains the nucleus, mitochondria, chloroplasts, etc., and a high concentration of enzymes that carry out general metabolic processes. The vacuole is contained by the tonoplast membrane with transporters that remove Na^+ and Cl^- from the cytoplasm for osmotic adjustment. A key tonoplast transporter is NHX1 (Na^+/H^+ exchanger) that transports Na^+ into the vacuole using energy supplied by proton pumps. Cl^- may follow passively and not require further energy.

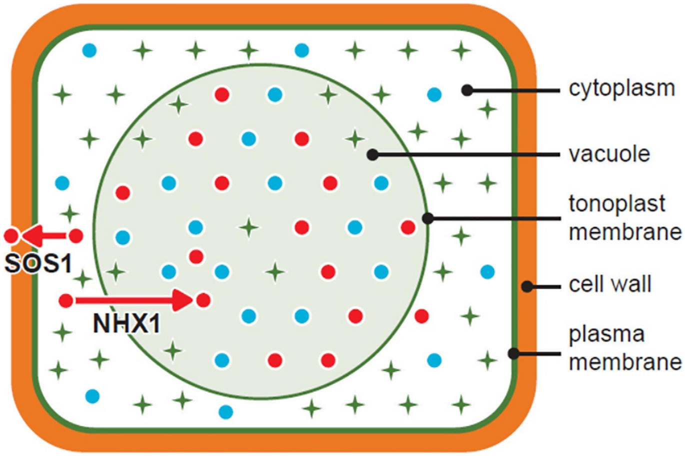

Fig. 14 A typical plant cell showing a central vacuole surrounded by the cytoplasm which contains the nucleus, mitochondria, chloroplasts, etc. The vacuole is contained by the tonoplast membrane. Na^+ is pumped out of the cytoplasm into the cell wall by SOS1 and into the vacuole by NHX1 (see Table 1). Red dots indicate Na^+, blue dots Cl^-, and crosses indicate organic solutes such as sucrose.

The maximum Na^+ concentration in the cytoplasm and particularly the cytosol (the part of the cytoplasm surrounding the mitochondria and chloroplasts) should be only 10–30 mM; concentrations above this are considered toxic (Munns and Tester, 2008). The cell is surrounded by the plasma membrane containing ion channels that restrict Na^+ from entering passively; influx of Na^+ is passive as the cytoplasm is negatively charged with regard to the cell wall. Excess Na^+ that leaks in is pumped back out by SOS1 (Fig. 14).

This adjustment allows cells to maintain turgor and volume, so that the plant can continue to function. With osmotic adjustment, all cells in roots and shoots can continue to grow and expand, and leaves continue to carry out photosynthesis.

Plants vary in their ability to control the uptake of Na^+ and Cl^- for osmotic adjustment, in their efficiency at excluding or transporting ions across membranes, and on how they partition and transport ions within cells, tissues, and organs. There is a delicate balance between excluding salts to avoid excessive concentration in the leaves, while taking up sufficient ions for osmotic adjustment. Too little salt and the plant may suffer from water deficit or will have to use energy-expensive organic solutes (either of which will reduce growth), while too many salts will cause salt toxicity and kill the leaves (Greenway and Munns, 1980).

The more salt-tolerant crop species like barley, sugarbeet and cotton have Na^+ and Cl^- concentrations in leaves and roots that are close or equal to that of the external solution, thus allowing energy-efficient osmotic adjustment (Munns et al., 2020a,b). If the cell Na^+ and Cl^- concentration is not equal to the external solution, energy-rich organic solutes such as sucrose balance the external osmotic potential. Cell turgor and volume is maintained; however, these sugars are then no longer available for the synthesis of new cell walls and cell constituents such as proteins and so the plant grows slower.

A plant avoids salt toxicity by two independent mechanisms that exist in all plant species but are effective to varying extents in different species. These are ion exclusion by roots and ion compartmentation within all cells throughout the plant.

Ion exclusion by roots—Roots exclude nearly all the salt in soil solution while taking up water. All plants, including halophytes, exclude 90–95% of the salt in the soil solution from the transpiration stream. Salts concentrate in leaves because plants evaporate about 95% of the water taken up by roots through leaf surfaces. It therefore follows that approximately 95% of the salt in the soil solution must be excluded by the roots to maintain a steady

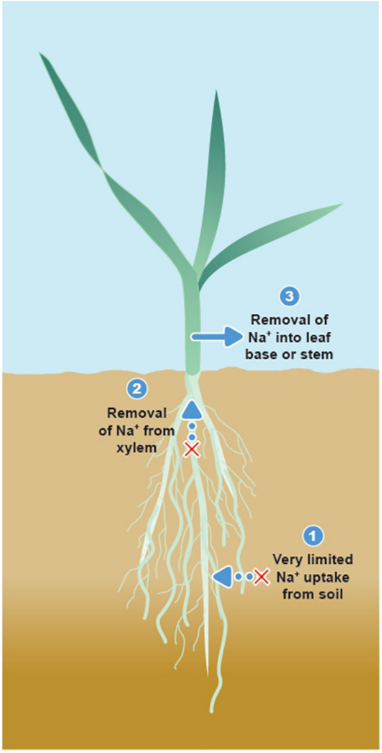

Fig. 15 Main control points for long distance transport of Na^+ in plants. Control point 1 is at the root surface where about 95% of the salt in the soil solution is excluded when the roots take up water. This is achieved by a plasma membrane that is virtually impermeable to Na^+, with a Na^+/H antiporter like SOS1 to efflux Na^+ that leaks in. Control point 2 is the ability of some species to retrieve another 1–4% of the Na^+ from the transpiration stream and prevent it moving to the shoot, using the cell-specific Na^+ uptake transporter HKT1;5. Control point 3 is a further removal of Na^+ in the leaf base and stem by HKT1;4 (Table 1).

concentration in the leaves (Munns et al., 2020a,b). Some species exclude up to 99% from their leaves via additional control points in the upper part of the root system, and the leaf bases and stems (Fig. 15).

Cellular ion compartmentation and "tissue tolerance"—The concept of "tissue tolerance" was based on observations that halophytes can accumulate NaCl in their leaves at very high concentrations (>800 mM), yet their enzymes that play a key role in essential metabolic processes are as sensitive to salt as are non-halophytes. Therefore, halophytes must effectively compartmentalize salt in the cell vacuole (which occupies are large percentage of the cell volume), thereby preventing their interference in key metabolic compartments within the cell (Fig. 14). The strategy of sequestering Na^+ and Cl^- in vacuoles and keeping concentrations low in the cytoplasm is critical to tissue tolerance (Munns et al., 2016).

10.3 Current research

Membrane transporters responsible for the control of Na^+ and Cl^- movement were studied intensively in the 1990s and their functions determined by electrophysiologists, as summarized in reviews focusing on membrane transport of Na^+ in relation to salt tolerance by Apse and Blumwald (2007), Munns and Tester (2008), and Ismail and Horie (2017).

In the following section we consider tolerance for saline and sodic soils separately and devote special attention to boron toxicity.

Saline soils—Realizing that salt tolerance is determined by the control of Na^+ and Cl^- uptake by roots and the transport of these ions within the plant, research has focused on identifying and cloning the genes responsible for this control. The main two approaches are (1) to look for natural variation within crop species and (2) create mutants in a model species that is amenable for genetic transformation. The model plant Arabidopsis has been used extensively, as its small genome speeds up gene discovery and its short life cycle and ease of transformation speeds up functional analysis of a candidate gene.

Membrane transporters that are important in controlling Na^+, Cl^- and K^+ transport in relation to improving crop salt tolerance via molecular breeding have been the subject of several extensive and authoritative reviews, most recently by Ismail and Horie (2017). For the control of Na^+ transport within plants, three membrane transporter genes have received the most attention. These are SOS1, NHX1, and HKT1 family members, summarized in Table 1.

The first two transporters are highly conserved across all species, and little natural genomic variation has been found. Although there are clear differences between species in the ability to accommodate these ions in vacuoles in their leaves, it remains unknown whether this is due to genetic variation in NHX1 leading to differences in levels of activity, to variation in the leakiness of the tonoplast, or to the efficiency of the proton pumps or ATPases that energize these transporters. The third transporter exhibits a degree of natural genetic variation especially in rice and is known to affect Na^+ accumulation in leaves and hence salt tolerance in rice (Platten et al., 2013) and wheat (Munns et al., 2012).

Table 1 For the control of Na^+ transport and improvement of salt tolerance within crop species, three membrane transporter genes have received the most attention.

Transporter	Location	Function	References
SOS1 (Salt Overly Sensitive):	Sodium-proton antiporter on the plasma membrane	Effluxes Na^+ out of plant cells (Fig. 14)	Shi et al. (2000)
NHX1 (Na^+/H^+ exchanger)	Sodium-proton antiporter on the vacuole membrane	Transports Na^+ from the cytoplasm into the vacuole (Fig. 14)	Apse et al. (1999)
HKT (High affinity K^+ Transporter) in rice and wheat: HKT1;4 and HKT1;5	Unusual Na^+ uptake transporters on the plasma membrane found mainly in xylem parenchyma cells	Take up Na^+ from the xylem as it travels to the transpiring leaves, so reducing its transport to the shoot (Fig. 15)	Reviews by Ismail and Horie (2017) and Munns and Tester (2008)

The first two are energized by ATPases and the second also by a H^+-pyrophosphatase proton pump such as AVP1 on the vacuole membrane (Gaxiola et al., 2001). The third transporter does not require energy as the cell cytoplasm is negatively charged.

Less work has been done on the control of Cl^- uptake as it does not enter the root passively (the root cell has a negative electrical potential). It may not be as toxic to metabolism as Na^+ but this is difficult to know as we cannot measure the concentrations of Na^+ or Cl^- in the cytosol where most of metabolism takes place, or in the mitochondria. All the same, Cl^- exclusion is especially relevant for perennials such as citrus and grapevine, which exclude Na^+ well but over time Cl^- can accumulate to high levels in leaves. Grafting scions with stocks for Cl^- exclusion has been shown to improved yield for saline soil. Candidate genes for the control of Cl^-transport in salt-affected plants are reviewed by Ismail and Horie (2017).

Sodic soils—Sodic soils are those that have a high exchangeable sodium percentage and are described in more detail in Section 12. Soil sodicity can directly affect plant growth, such as by sodium-induced Ca^{2+} deficiencies (Lauchli and Grattan, 2012), as well as indirectly due its adverse effect on soil structure. Under sodic conditions, soil aggregates are dispersed, leading to reduction of large soil pores thus affecting water flow and gas diffusion, increased soil strength and soil crusting. The increased soil strength reduces root proliferation and seedling emergence, and promote waterlogging thereby affecting plant growth by reducing oxygen diffusion to the roots and CO_2 away from the root (Barrett-Lennard, 2003). Waterlogging not only creates anoxia, but also reduces Fe^{3+} to Fe^{2+} and Mn^{4+} to Mn^{2+}, sulfate to sulfide, and promotes denitrification, produces toxic constituents, and aggravates waterborne diseases (Kozlowski, 1997). Therefore, a plant under sodic stress is likely encountering additional abiotic or biotic stresses. A survey of genetic variation in waterlogging and salt tolerance of many fodder plants for salt-affected soils is given by Rogers et al. (2005).

Boron toxicity—Boron is often present in saline environments in excess amounts (Section 8) and can cause injury to susceptible crops. While an essential element, there is a small concentration range in the soil solution between what is deficient for plant growth and what is excessive. Boron uptake by plant roots occurs by (1) passive diffusion across the plasma membrane, (2) facilitated transport through intrinsic proteins in the membrane, and (3) energy-dependent transport through a high affinity uptake system (Takano et al., 2008). Boron transporter genes that control the uptake of B have been identified in wheat as alleles of the transporter Bot-B5 (Pallotta et al., 2014). Boron remains immobile in most species after it enters the leaf but in some, particularly stone-fruits, it can remobilize via the phloem to fruits and growing parts of the plant. Boron forms complexes with polyols that allow for its mobility (Brown and Shelp, 1997), therefore making it difficult to use tissue diagnosis for B deficiency and toxicity (Nable et al., 1997).

Salinity-boron Interactions—Despite the common occurrence of salinity with boron, very little research has addressed the complex interaction of these two abiotic stresses on plant growth, which can be antagonistic or synergistic (Läuchli and Grattan, 2007). Wimmer et al. (2003) found that combined salinity and boron stresses significantly increased the B-soluble fractions and that these soluble fractions were an indicator of B-toxicity. Soil pH can also influence the salinity-B interactions (Smith et al., 2013), and could affect membrane transport characteristics (Läuchli and Grattan, 2007).

10.4 Future research priorities

Roots do most of the work protecting the plant from excessive salt uptake by excluding salts in the soil solution while taking up water, but we do not know whether this occurs in all parts of the root system, or whether it is confined to young roots or lateral branch roots.

A more directed molecular breeding approach needs certainty about which cells or cell layers within the root anatomy are the site of Na^{+}

exclusion. Recent analysis suggests that the epidermis is the main site of Na^+ exclusion, not the endodermis as was traditionally thought (Munns et al., 2020a,b). Efflux of Na^+ that has leaked into the root is expensive, possibly consuming over 10% of the total ATP produced by root respiration (Munns et al., 2020a,b) so it is important to know where in the root this occurs, and whether this is due entirely to SOS1 or other transporters. The other expensive process could be the maintenance of high concentrations of Na^+ and Cl^- in cell vacuoles, and we need to know the "leakiness" of the tonoplast membrane, which could result in a major costs to cells that need to keep pumping Na^+ back into the vacuole (Shabala et al., 2020).

Sodic soils, which often have high pH, have received little attention by physiologists, yet they are more widespread than saline soils of neutral pH (Section 12). Many sodic soils have very high pH of 9–10 or more, which alters the speciation and thereby solubility and uptake of many minerals including Al. Research is needed to better understand multiple stress interactions including the effects of soil compaction or waterlogging (Läuchli and Grattan, 2007; Mittler, 2006). In addition, B toxicity like many other abiotic stresses, causes the formation of reactive oxygen species, yet little is known about the actual mechanism of B toxicity in plants or how B toxicity affects the plant's antioxidant defense system (Cervilla et al., 2007). Salinity-B interactions are complex and merit further scientific investigation.

Summary: Mechanisms of adaptation to saline soils of neutral pH have been thoroughly studied by physiologists. A plant avoids salt toxicity by two independent mechanisms that are effective to varying extents in different species. These are ion exclusion by roots, and ion compartmentation within all cells. It is essential to keep Na^+ concentrations low in the cell cytoplasm but at the same time minimize the energy costs of accumulating high concentrations of organic solutes for osmotic adjustment. Key genes for control of Na^+ and Cl^- uptake from a saline soil, their transport throughout the plant, and their sequestration within cells have been identified. Genomic variation exists but has not yet been fully explored and its usefulness exploited.

11. Priority 8: Plant breeders need genes proven for salt tolerance without yield penalty

11.1 Introduction

Despite significant biotechnological efforts toward the development of crop plants that can increasingly tolerate salinity and water stress, progress has been slow and remains a huge challenge. In most countries, genetic modification (GM) of staple food products (wheat, rice, maize) is not acceptable. Advances in "gene editing" have the potential to overcome the objections to previous GM technology (Zaidi et al., 2019) but gene editing is still not widely accepted by regulators. Therefore, genetic approaches should continue looking for natural variation *within* species, rather than introducing genes from *other* species. Vast natural variation exists within the genome of crop species and their close relatives which is under-utilized for breeding salt tolerance. This biodiversity is contained in large international seed collections and should be used to provide new germplasm with improved yield on salt-affected land. Why is this genetic variation not used more by breeders?

To answer this question, we need to understand plant breeding methods and the requirements for commercial release of a new cultivar. The over-riding criteria for a new release are yield potential and quality of product. If yield on the best soil is reduced by the introduction of a salt-tolerant gene, it will be of no interest to breeders, even though yield on saline soil might be improved. There are two practical reasons for this: (1) breeding trials are always conducted on typical soils of the regions, not on the more saline and (2) saline soil is always heterogenous within a field, so total yield is largely determined from yield of the less saline parts of the field (Richards et al., 1987). These authors concluded that the most efficient way to increase yields at high salinity was to select for the best performing lines at low salinity. Not all breeders agree with this, but for most commercial breeding companies, yield on saline soil is subordinate to yield potential.

Conventional breeding for salt tolerance starts with new germplasm with known variation for a specific quantitative trait, crossing into current breeding lines (elite parent) to introduce the trait, then a number of rounds of back-crossing to the elite parent to remove the unintended and undesirable traits that have been introduced with the new germplasm. The new breeding lines are then tested in different soil types in different climatic zones within the regions of release, to ensure no yield penalty of the salt-tolerance gene. This approach of crossing and selection is usually done using molecular markers: DNA fragments that are associated with the trait. Selection for the trait itself is more laborious and expensive.

11.2 Past research

Conventional breeding—For centuries, farmers in countries with extensive soil salinity have long been selecting best yielding crops for their land, as have the more recent commercial breeding companies. If their soil contains salt, they have selected salt-tolerant material without specifically intending to do so. An example is the salt-tolerant bread wheat Kharchia, which forms the basis of most of the salt-tolerant bread wheat germplasm released in India and Pakistan. Kharchia 65 is a landrace developed from selections in farmers' fields in the sodic-saline soils of the Kharchi-Pali area of Rajasthan (Rana, 1986). We do not yet know the physiological or molecular basis of the salt tolerance of Kharchia. For bread wheat a summary by Naeem et al. (2020) listed 14 varieties or landraces under commercial production in India, Pakistan, Egypt and China. All of these were produced by conventional breeding.

For rice, derivatives of the landraces Pokkali or Nona Bokra which occur in the coastal regions of southern India have formed the basis of salt-tolerant rice cultivars. Ismail and Horie (2017) list 27 cultivars that have been released for salt tolerance between 2007 and 2014 for Bangladesh, the Philippines and India. These have been developed by conventional selection and breeding. The two most significant cultivars are CSR 36 for salt-affected soils in India, and BRRI Dhan 10 for soils inundated by seawater in coastal Bangladesh. We know (retrospectively) the molecular basis of some of this salt tolerance: the presence of specific alleles of the Na^+ transporter OsHKT1;5 that enhance Na^+ exclusion (Table 1). These were identified in Nona Bokra as the QTL SKC1 and identified in Pokkali as the genomic region Saltol which encompasses OsHKT1;5 (Ismail and Horie, 2017). Molecular markers are now being used to accelerate breeding and to pyramid salt tolerance with other traits relevant to saline soils such as waterlogging tolerance.

Trait-based breeding—A lack of fast and reliable screening methods has been the major limitation to exploring large germplasm collections, selecting genotypes with greater salt tolerance than the current cultivars, and introducing the salt tolerance into breeders' advanced breeding lines for release of a new salt-tolerant cultivar. Munns and James (2003) summarized the various methods used in the laboratory or glasshouse to select for salt tolerance, along with their advantages and disadvantages. The simplest method is that of screening at germination as it is such a quick and easy test for large numbers of genotypes. However, for most species there is little or no correlation between genotypic differences in germination and genotypic differences in later growth or yield. The most reliable and useful method has been to measure rates of Na^+ or Cl^- accumulation in leaves, selecting individuals with low rates of accumulation.

Ideally, biomass or grain yield should be the ultimate criterion for salt tolerance. Selections of various genotypes of pasture species like clover or alfalfa can conveniently be done in hydroponics or sand cultures with added salt, as cuts can be made every 6–8 weeks for replications. Cereals are more difficult to assess as grain yield needs to be measured in saline soil in the field, as does the yield of perennial horticultural species like citrus and grapevine. However, field experiments are plagued by heterogeneities in soil texture and surface elevation and its associated effect on soil salinity and compaction over short distances by influencing soil water deficits or waterlogging. This heterogeneity makes validation of breeding trials difficult as soil salinity varies greatly over area and depth (Fig. 4). Soil salinity under each of a thousand or so breeding plots (1 m × 2 m) needs to be measured by electromagnetic induction with a simple-to-use meter such as Geonics EM38 after calibration (Section 3.4). Incorporation of plot EC as a co-variant in the statistical analysis was essential to finding durum wheat genotypes (Munns et al., 2012) and bread wheat and barley genotypes (Setter et al., 2016) with higher yield in saline soil.

11.3 Current research

Over the last 20 years, selection of new salt-tolerant germplasm and its use in subsequent breeding has depended on traits and molecular markers for traits, which can be obtained from genetic analysis as Quantitative Trait Loci (QTL) (e.g. Lindsay et al., 2004) or by Genome Wide Association Studies (GWAS) (e.g. Saade et al., 2016).

For many crop species, genetic variation in ion exclusion correlates highly with salt tolerance, and screening based on the measurement of ion accumulation in leaves is the most precise and effective form of selection, being quantitative and non-destructive. Examples include Na^+ exclusion from leaves of durum wheat (Munns and James, 2003) and rice (Platten et al., 2013; Yeo and Flowers, 1986). As an example, we describe a successful project on introduction of genes for salt tolerance from a wheat relative into a durum wheat cultivar, using molecular markers for the trait of Na^+ exclusion.

Durum wheat (*Triticum turgidum* ssp. *durum*, tetraploid) lacks the gene for Na^+ exclusion found in bread wheat (*Triticum aestivum*, hexaploid). Using the screening method of Na^+ exclusion from leaves among 60 durum wheat relatives, Na^+ exclusion equal to bread wheat was found in an unusual durum genotype named Line 149 (Fig. 16). Line 149 was crossed with

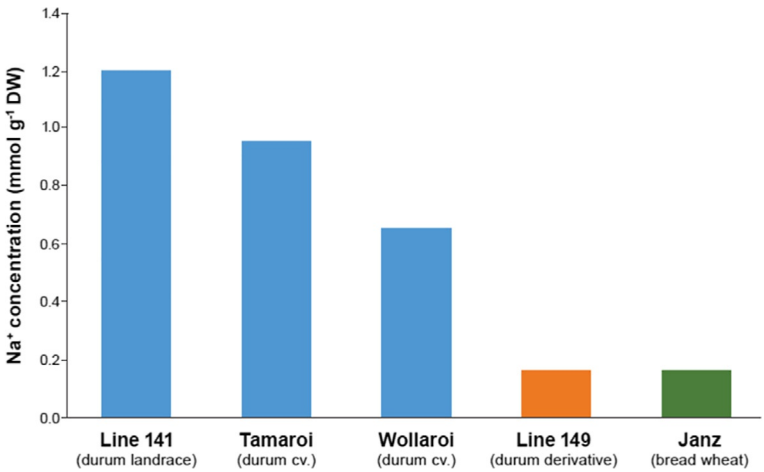

Fig. 16 Sodium concentration after 10 d in leaves of bread and durum wheat genotypes grown in 150 mM NaCl with supplemental Ca^{2+}. Tamaroi and Wollaroi are Australian durum wheats; Janz is a bread wheat. For genetic analysis, Line 149 was crossed with the other three high-Na^+ durum genotypes (Munns et al., 2003). This revealed two genes named *Nax1* and *Nax2*.

the durum cultivar Tamaroi which had five times the leaf Na^+ concentration and subsequent genetic analysis showed that Na^+ exclusion was due to two genes that were named *Nax1* and *Nax2* (Munns et al., 2003). Further crossing enabled separation of the two genes, which were identified as HKT1 transporters (see below). Field trials in multiple sites showed that *Nax2* increased yield on highly saline soil by 25% without affecting yield on better soils (Munns et al., 2012). However, *Nax1* had a yield penalty that outweighed its advantage as a Na^+ excluder. This yield penalty had not been obvious in glasshouse trials but became significant in the field (James et al., 2012).

Phenomics—For crop species where a trait is multi-genic and covering different chromosome regions, molecular markers have limited value and selection is driven by phenomics. High-throughput phenotyping methods, now employed in the field as well as in the laboratory, allow large numbers of plants to be screened efficiently with limited handling and labor. Screening for salt tolerance in species which do not have a selectable salt-specific trait is only feasible using non-destructive methods. Such methods include biomass growth as assessed by photosynthesis, stomatal conductance, chlorophyll fluorescence and spectral reflectance. Using color imaging along with nondestructive measurements of the leaf area and growth rate of each plant, it is possible to separate the effects of salinity on new leaf production from the acceleration of senescence and death of old leaves (Negrão et al., 2017). Imaging allows the short-term osmotic effects on plant growth to be distinguished from the longer-term ionic effects. Infrared thermography is a widely used phenomic tool to detect differences between genotypes in soilless culture, pots, and field plots (Esmaeili et al., 2017). In addition, hyperspectral imaging is used to quantify differences in water status and photosynthetic capacity and to detect genotypic differences in salinity tolerance, for example, among wheat cultivars after anthesis (Hu et al., 2017).

Gene discovery—Research into genes for salt tolerance follows one of several approaches:

- Fine mapping from a QTL. This approach was used to discover genes for retrieving Na^+ from the xylem, namely the *Nax* genes in durum wheat, and *SKC1/Saltol* in rice (Table 1). These genes are being used by wheat and rice breeders.
- Mutagenesis and high-throughput screens such as root length in saline media. In Arabidopsis this led to the discovery of SOS1 (Shi et al., 2000).
- Applying principles of plant physiology, biochemistry, and electrophysiology. This led to discovery of NHX1 and AVP1 (Table 1).
- "Omics" methodology, where global changes in transcripts or proteins or metabolites in response to stress treatments are listed. Comparisons are made between control and stress treatment, or between two genotypes known to different in salt tolerance. So far this has not led to discovery of a new gene, but rather lists of hundreds of known genes or proteins that are up-regulated or down-regulated under stress. When proteomics and metabolomics are combined with flux analysis, changes in metabolic pathways can be seen, for instance in respiratory efficiency and the GABA shunt (Che-Othman et al., 2020).

Most candidate genes so far discovered and proven to be part of the mechanism of salt tolerance are membrane transporters for Na^+, K^+ or Cl^-. Few transcription factors have a known function, either in the downstream target genes, or the cells or tissues in which they operate. Genes involved in signaling pathways are not known to be specific for salinity but have commonalities with other abiotic stresses that reduce growth rate like drought, heat and cold.

Transgenics—Use of the Arabidopsis genome has greatly accelerated the sequencing and functional analysis of candidate genes. In total there have been about 7300 papers on salt tolerance involving Arabidopsis (*Web of Science*). For the six main crop plants (wheat, rice, maize, barley, soybean and canola) there are 9200. How much of this work has led to improving salt tolerance of crops in the field?

A summary of 27 genes that have been overexpressed in various crop species with "reported plant transgenic performance during salt stress" is listed by Roy et al. (2014) in their table 1, but with three exceptions, these transgenics have not been tested in the field or handed over to commercial plant breeders. In a review of genetic engineering for salinity tolerance in wheat (Mujeeb-Kazi et al., 2019), a list of 45 publications on wheat transformed with genes from other species, or other species transformed with genes from wheat, showed only one that included performance in the field; overexpression of AtNHX1 improved grain yield of bread wheat (Xue et al., 2004). A notable success story is with barley: overexpression of AVP1 increased biomass and yield in both non-saline and saline soil (Schilling et al., 2014). Overexpression of genes for accumulation of organic molecules that act as osmolytes such as proline have been studied for decades, but no cultivar has been released with enhanced proline accumulation that improves yield on saline soils.

11.4 Future priorities

To date, QTL continues to be the main tool of genetic analysis for breeders, yet very few pre-breeding efforts have led to production of salt-tolerant cultivars (Mujeeb-Kazi et al., 2019). Similarly, the early optimism for GWAS (genome wide association studies) to discover new loci for salinity tolerance and their subsequent utilization in varietal development is still not realized. Success in has been hampered by lack of (1) quantitative and repeatable measurements of the value of the trait to plant growth and yield in saline soil and (2) selection of the best parents for QTL analysis or genotype array. Further research into selection techniques and germplasm diversity is needed.

Key genes for Na^+ transporters presented in Section 10.2 should be studied using species other than Arabidopsis. Crop species that are amenable to transformation and do not have complex genomes (such as rice and barley) should be used. Omics methodologies should use relevant treatments, such as a gradual and moderate salt stress, not a severe and sudden one (e.g. 200 mM NaCl in one hit). Osmotic shocks cause plasmolysis and induce the synthesis of enzymes that repair the trauma caused to cells by their sudden shrinkage which may take at least 24 h to repair. Gene expression patterns are very different when the stress is imposed gradually compared to a salt shock (Shavrukov, 2013). Cell-specific and tissue-specific expression is critical for the function of transporters and transcription factors, so studies should consider this should, for example, separately analyze growing from mature tissues.

As take-up of genes for salt tolerance by commercial crop breeders has been so slow, and few studies arising with model plants such as Arabidopsis have been validated in the field, there is a high priority to engage plant breeders at an early stage of the project, working along with physiologists, molecular biologists and agronomists. Only then will molecular biology translate to the field and reach crop production targets (Passioura, 2020).

There are clear opportunities to make substantial yield gains by targeting basic strategic research, especially by utilizing pre-breeding results of undomesticated varieties, to improve abiotic stress tolerance of crops. Additional recommendations for future research include to use pre-breeding approaches seeking salt tolerance traits, rather than focus on model plants such as Arabidopsis. Also, while research at the cell level is likely to advance our physiological understanding of salt tolerance mechanisms, in parallel significant investments should be made at the field-level, employing the latest in phenotyping methodology.

Summary: Unexplored and under-utilized biodiversity exists within crop species and their close relatives, which could be used to improve germplasm for crop production on salt-affected land, without resorting to GM methods that are at present unaccepted in many countries. Molecular and genomic tools are becoming more widely available to breeders. Ongoing advances in rapid generation turnover, improved phenotyping, envirotyping and analytical methods can increase the rate of genetic gain in breeding. Further understanding of mechanisms at the molecular and physiological level will complement these new technologies and provide farmers with alternatives to increasing crop production on saline land. While genetic improvements cannot provide a permanent solution to increasing soil salinity, and salt-tolerant crops cannot de-salinize the land, a 10% increase in yield may double the famer's profits, where the profit margin is small.

12. Priority 9: Salinity and sodicity effects on soil physical properties

12.1 Introduction

In most of the salt-affected regions with dominance of sodium salts, salinity and sodicity are related, but they are different in terms of their effects on soil environments. "Salinity," usually measured as total soluble salt concentration, affects plant growth and productivity through osmotic effects and ion toxicity or deficient effects on plant physiological processes (Section 10). "Sodicity," generally defined by soil ESP (exchangeable sodium percentage) or SAR (sodium adsorption ratio) of soil solution, causes constraints to plant growth through its effects on soil physical properties (Section 2). Natural climatic and soil processes can lead to the formation of sodic soils from saline soils. In irrigated agriculture, the use of sodium containing waters leads to sodic soils by the adsorption of sodium by soils. Sodic soils with low salt concentration undergo structural degradation when wet because of swelling and clay dispersion, causing reduced water and air transport in near-surface soils (Shainberg and Letey, 1984; Sumner and Naidu, 1998) and to limitations in soil aeration and infiltration. The effects of sodicity on soil physical properties are modified by soil salinity levels (Section 6.1). Drastic reductions in hydraulic conductivity (K) have been reported when low electrolyte rainwater infiltration conditions were simulated using de-ionized water ($EC < 0.03\,dS/m$) following the saline-sodic waters. It has been shown that the "washed-in" dispersed clays moved into the subsoil, blocking soil pores and permanently restricting downward movement of water (Minhas et al., 2019). Therefore, salinity-sodicity interactions are considered very important in understanding and managing soil physical processes.

12.2 Past research

Dispersive soils with poor physical properties have been investigated since the earlier part of 20th century (e.g. Puri and Keen, 1925). The earlier terminology for sodic soils, to differentiate from saline soils, were "Solonetz" and "Alkali soils" (Kelley, 1951; Szabolics, 1989). These dispersive soils were characterized by high sodium content and alkaline pH. Even though, in colloidal and clay mineralogical studies, monovalent cations, K and Na, have been implicated in the macroscopic swelling and dispersion of clays, only sodium was considered as a factor in soil investigations because of its prevalence in salt-affected soils. The general assumption was that both calcium and magnesium were helping in the promotion of soil structural stability. In the first half of the 20th century debates on soil pH and cation adsorption and exchange were intense (Bolt, 1997; Raats, 2015).

The exchangeable sodium percentage (ESP) of soil expresses the level of sodicity (adsorbed sodium) according to:

$$ESP = 100[Na_{ex}]/CEC, \tag{9}$$

where [Na_{ex}] represents the exchangeable sodium and CEC represents cation exchange capacity, both expressed as meq/100 g. Several reviews (Bresler et al., 1982; Qadir and Schubert, 2002; Shainberg and Shalhevet, 1984; Sumner and Naidu, 1998, among others) evaluated the effects of increasing ESP levels on soil structural deterioration by increasing clay dispersion, soil crusting, soil strength and soil erosion; while decreasing saturated and unsaturated hydraulic conductivities (K_s and K_{uns}, respectively), infiltration and drainage rates, and aeration porosity.

The sodium adsorption ratio (SAR) model was originally derived based on the "ratio law" of Schofield (1947) to predict the adsorption of sodium on exchange sites from soil solutions or irrigation water in relation to divalent cations present, and is used as a criterion to classify irrigation water quality (Section 2.2). Irrigation water SAR does not account for changes in cationic concentrations in soil solution from the solubility of soil minerals. Ayers and Westcot (1985), Suarez (1981), and Rhoades (1982) have discussed adjusting SAR to account for the changes in ionic concentrations in soil solutions due to increasing levels of bicarbonate and carbonate ions in irrigation water, causing Ca or Mg ions to precipitate.

The definition of sodicity using ESP varies in different parts of the world. For example, while the threshold ESP level is 15 in the USA (US Soil Salinity Laboratory Staff, 1954), it is 6 in Australia (Isbell, 2002), likely because critical ESP varies widely depending on several soil factors. Specifically, clay content and mineralogy, organic matter, soil electrolyte concentration and composition, pH, types of exchangeable cations including K, Mg and Al, presence of Al and Fe oxides and cementing agents such as calcium carbonate. All these soil factors, individually or in combination, determine the ESP level at which soil structure and physical properties are affected (Rengasamy and Sumner, 1998).

In the middle of 20th century, scientists understood the role of soil solution or irrigation water salinity in reducing the effects of sodicity on soil physical properties. In their widely cited paper, Quirk and Schofield (1955) defined the "threshold electrolyte concentration" (TEC) as the concentration that led to a 10–15% decrease of sodic soil permeability from its initial value measured for non-sodic conditions. This awareness of sodicity effects on soil physical properties led to the practical field application of electrolytes such as gypsum, and to distinguish between sodic (dispersive) soils and saline (flocculated) soils. Subsequently, several models were published, relating the ESP (or SAR) and the total water electrolyte concentration to determine the TEC value for any sodicity level at which soil physical properties were not detrimental to crop production, based on studies that assessed hydraulic conductivity, infiltration rate and clay dispersion of soils (Ayers and Westcot, 1985; Bennett and Raine, 2012; McNeal, 1968 among others). However, these studies also confirmed that the TEC model is not universal but varies depending on soil factors other than salinity and sodicity (Sposito et al., 2016).

There have been many attempts to simulate the effect of sodicity and salinity on soil water retention and hydraulic conductivity for sodic soils, using both theoretical and empirical models. Theoretical models are based on diffuse double layer theory (Russo, 1988; Russo and Bresler, 1977a,b), whereas empirical models (Dane, 1978; McNeal, 1968; Simunek et al., 1999) are based on laboratory experiments. However, predicting changes by sodicity on hydraulic conductivity, porosity, water retention and soil leaching capabilities (Assouline et al., 2015; Ben-Gal et al., 2008; Russo et al., 2009) has proven to be difficult, because of their dependency on so many soil factors.

12.3 Recent research

Earlier research focused on the diffuse double layer theory and the various electrostatic forces operating in colloidal suspensions (van Olphen, 1977) to explain soil structural changes in sodic soils. To understand the mechanisms of slaking and dispersion of soil aggregates, it is necessary to consider all microscopic processes that occur during initial wetting of dry aggregates. Of particular importance is the interaction of the clay-ionic bonds with polar water molecules resulting in swelling in the first stage to the final stage of aggregate disintegration, followed by dispersion of soil clays when completely wet (Rengasamy and Sumner, 1998). Based on their investigations, Marchuk and Rengasamy (2011) proposed to replace SAR with the CROSS index (cation ratio for soil structural stability):

$$\mathrm{CROSS} = (\mathrm{Na} + 0.56\,\mathrm{K}) / \left[(\mathrm{Ca} + 0.6\,\mathrm{Mg})^{0.5} \right], \tag{10}$$

where all cation concentrations are expressed in $\mathrm{mmol\,L^{-1}}$. Sposito (2016) has also given examples of aggregation of clay particles based on these ionicity indices. As expected, the relationship between CROSS and clay dispersion or the soil's hydraulic conductivity varied with soil factors (Farahani et al., 2018; Jayawardane et al., 2011; Oster et al., 2021).

Rengasamy et al. (2016) proposed the concept of net dispersive charge, responsible for clay dispersion. The inherent negative charge of a soil is

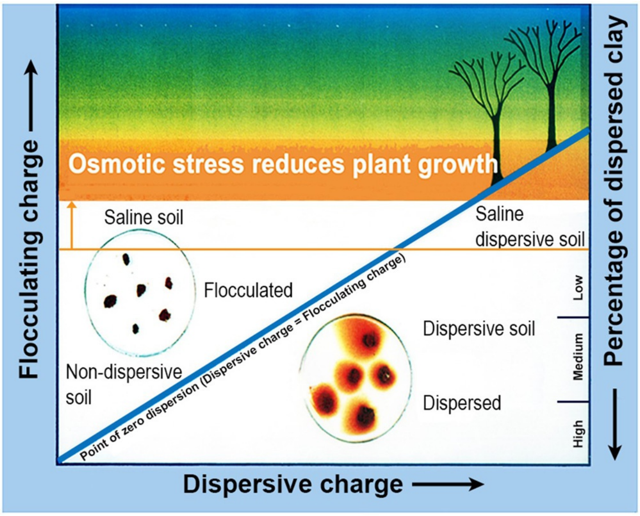

Fig. 17 Distinction between saline and sodic (dispersive) soils based on dispersive charge, flocculating charge, and percentage of dispersed clay. Point of zero dispersion represents threshold electrolyte concentration (TEC). *After Rengasamy, P., 2016. Soil chemistry factors confounding crop salinity tolerance—a review. Agronomy 6 (4), 53. https://doi.org/10.3390/agronomy6040053, with permission from MDPI CC BY 4.0.*

usually measured as CEC at a given soil pH. This negative charge attracts cations, which bind to soil particles with varying degree of ionicity and valency, defining their dispersive and flocculating powers, as calculated by Rengasamy et al. (2016). When the dispersive charge of a soil is more than the flocculating charge, clay dispersion occurs (Fig. 17), with the force difference defined as net dispersive charge. Alternatively, flocculation occurs if the flocculating charge is equal or more than the dispersive charge. Introduction of this conceptual approach has resolved many controversies caused by the roles of organic matter, clay mineralogy, exchangeable cation composition, and electrolyte concentration and composition in relation to clay dispersion, and especially the role of K and Mg on soil structural stability. Based on this concept, Rengasamy (2018) redefined the point of zero dispersion, representing TEC, accounting for the effect of individual cations on the flocculation processes.

It is important to note that laboratory results need proper interpretation for field conditions. Under arid and semi-arid conditions, irrigation water contains appreciable amounts of Na, Ca, and Mg salts and their interactive effects on hydraulic conductivity of soils greatly depend on their relative proportions (Chaudhari et al., 2010). Hamilton et al. (2007) have shown that the impact of sodicity depends on the initial soil water content before irrigation and on the duration of wetting and drying cycles. In addition, van der Zee et al. (2014) showed that the impact of sodic soils on soil hydraulic properties and infiltration of low salinity rainfall depends on the temporal structure of the wetting events. They found that the impact is negligible for rainfall regimes that cause small variations in soil wetness and soil salinity, whereas the impact is more significant for seasonal rainfall patterns. Russo et al. (2004) analyzed flow and transport in montmorillonitic clay soils for a Mediterranean climate where a long dry season requiring irrigation is alternated by a distinct rainy winter period. They analyzed long-term effects, demonstrating a minor decrease in soil hydraulic conductivity function during the irrigation season, but a significant decrease of it during the rainfall season as caused by soil salinity dilution of low salinity rainfall.

12.4 Future priorities

There is a need for better understanding of the salinity and sodicity interactions to develop appropriate and efficient approaches to combat and mitigate the adverse effects on soil properties and the design of proper restoration procedures for saline and sodic soils. Under field conditions, the chemical and physical effects of sodic irrigation water vary greatly as determined by drainage provisions, leaching fraction, soil tillage, and irrigation method. Consideration of the combination of all these factors will be necessary to arrive at practical approaches that mitigate irrigation water effects on soil structure.

Several gaps and tasks are as follows:

– Improved approaches and models for representing the interactions between monovalent and bivalent cations in soil solution and CEC, and on their impact on the clay fraction as related to soil slaking, dispersion and swelling.
– Improved conceptual and physically-based semi-empirical models expressing the impact of sodicity and salinity on soil hydraulic properties.
– Investigating the CROSS-dispersive charge relationship and developing CROSS-based models as guidelines for structural stability of irrigated soils.
– Developing methods to estimate net dispersive charge of soils so that they can be applied toward the modeling of reclamation of sodic (or dispersive) soils.

– Identifying the critical level of clay dispersion that affects crop productivity.

Summary: We point out that the current knowledge of the salinity and sodicity effects on soil physical properties is limited and there is a need for more basic research. Relatively little work has been done to quantify dynamic soil hydraulic properties such as soil water retention, hydraulic conductivity, solute transport and soil aeration properties and processes, as caused by changing soil salinity and sodicity. Research gaps are particularly evident when realizing that the magnitude of soil structure degradation is complex and controlled by many other soil properties other than simply by soil salinity and sodicity only. Soil flow and transport models that simulate irrigation water and soil salinity in concert with soil and plant management practices must include such information.

13. Priority 10: Limitations and opportunities of using non-conventional water sources for irrigation

13.1 Introduction

Irrigated agriculture must expand and new water sources, previously considered "marginal" (e.g. saline, treated waste waters, and desalinated water) need to be utilized to meet the growing demands in the future (Assouline et al., 2015; Gleick, 2000; Grant et al., 2012; Tal, 2006). But for such an endeavor to be successful, a careful balance of agronomic, economic and environmental factors, including long-term risks to soil hydro-ecological functioning, must be considered.

Expansion of irrigated agriculture relying on unconventional water sources of marginal quality, especially in highly populated arid regions where water resources are limited, would invariably enhance the risk of salinization. Worldwide, about 7.1 billion m^3 of treated municipal wastewater are reused mainly for irrigation (about 50%) and industrial purposes (about 20%) (Vergine et al., 2017). While this practice has expanded the overall water supply for irrigation and industrial processes, there are many parts of the world that do not take advantage of its potential.

Nevertheless, the past few decades brought an increase in wastewater use in agriculture in developing countries and in semi-arid and arid areas of industrialized countries. While salinity management strategies to minimize root-zone salinity by leaching are needed (Section 6), such "good" agronomic practices could contaminate groundwater supplies inducing a vicious cycle, where crop production is maximized at the cost of increased groundwater contamination threatening the sustainability of such practices. In addition to salinization, the use of treated wastewater may pose a risk to public health due to exposure to microbial pathogens or chemical compounds (heavy metals, toxic organics, and anthropogenic compounds), thus requiring appropriate regulations (Aiello et al., 2007; Qadir et al., 2010; Scheierling et al., 2010; Shuval et al., 1986; Toze, 2006; Vidal-Dorsch et al., 2012). It could also induce various environmental risks to soil ecology and function in addition to increasing groundwater pollution.

13.2 Past research

Reuse of treated effluent (TE) in agriculture has a long tradition (Shuval et al., 1986), particularly on lands located near urban centers where the wastewater is treated. High concentrations of salts, most particularly those dominated by sodium (Na^+) and the presence of organic compounds were identified as the primary risks associated with TE irrigation (Balks et al., 1998; Feigin et al., 1991). The addition of these constituents has shown to increase the exchangeable sodium percentage (ESP) of the irrigated soils (Halliwell et al., 2001; Shainberg and Letey, 1984), thereby affecting soil structural stability. The effects and mechanisms of salinity and sodicity on plants and soils have been extensively investigated between the 1960s and1980s (Ayers and Westcot, 1985; Bresler et al., 1982; Maas and Hoffman, 1977; Rhoades, 1999); see also Sections 2 and 12.

Unique to irrigation with TE is the combination of organic matter (OM), particularly dissolved organic content (DOC), with high concentrations of sodium. Clay dispersion was found in many studies to be enhanced in the presence of DOC (Frenkel et al., 1992; Quirk and Schofield, 1955; Tarchitzky et al., 1993, 1999). Nelson and Oades (1998) reviewed the literature on the effects of OM on soil sodicity. They concluded that, when irrigating with water of a given salinity, ESP was augmented in soils with lower OM content because the exchange selectivity for Na^+ in soils decreases as their OM content increases. However, it was shown that OM can be either a bonding or a dispersing agent, depending on the level of the ESP, the particular chemical properties of the OM constituents, and the degree of mechanical disturbance of the soil. In particular, dissolved OM was found to disperse soil clay particles in the presence of anionic constituents, high ESP, and mechanically disturbed soil (Churchman et al., 1993; Nelson and Oades, 1998), conditions that are typical following irrigation with TE.

Tarchitzky et al. (1999) showed that the hydraulic conductivity of a soil leached with TE decreased sharply relative to the small decrease observed when the soil was leached with a similarly composed electrolyte solution, but lacking the DOC. This result was explained by the interaction of anionic OM with positively charged edge surfaces of 2:1 clay mineral particles, preventing the edge-to-face association of the particles involved in flocculation (Tarchitzky et al., 1999). Therefore, the organic fraction from TE, particularly the dissolved fraction, is not always beneficial in regards to sodicity and soil structural stability. Such complexity of the relationships between OM and soil permeability properties were reviewed by Churchman et al. (1993).

13.3 Recent research

Over the last 2 decades, studies have advanced our knowledge regarding the impacts of irrigation with TE on soil properties. Long-term TE irrigation of clayey soils was shown to cause significant deterioration in the physical and chemical properties of soils (Aiello et al., 2007; Assouline et al., 2016; Assouline and Narkis, 2011; Lado et al., 2005, 2012; Levy and Assouline, 2011). The ESP in TE-irrigated soils often is higher than the SAR of the soil solution (Assouline et al., 2016; Levy et al., 2014). One possible explanation for this is the absence of equilibrium in the soil between the SAR of the irrigation water, the SAR of the soil solution, and the soil ESP (Keren, 2012; Nelson and Oades, 1998). The cation ratio for soil structural stability (CROSS) has been suggested as a more appropriate index that could replace SAR as it considers potassium's (K) role in dispersion while simultaneously discounting Mg′s role in flocculation (Sposito et al., 2016). For example, the $CROSS_{opt}$ expression below is a further modification of the SAR expression in Eq. (10), to include coefficients that were optimal using the soils tested by Smith et al. (2015):

$$CROSSopt = \frac{Na + 0.335K}{\sqrt{(Ca + 0.0758Mg)}} \tag{11}$$

A detailed quantitative description of the changes in the soil physical and hydraulic properties, and consequently on infiltrability, of a clayey soil following long-term irrigation with TE can be found in Assouline and Narkis (2011), Coppola et al. (2004), and Aiello et al. (2007). An interesting finding in Assouline and Narkis (2011) was the depth-dependence of the extent of soil deterioration. The decrease in the K_s values induced by TE irrigation was maximal in the upper soil layer and decreased gradually with depth. Additionally, the amplitude of the impact on the water retention and the hydraulic conductivity functions was different at each depth, suggesting that long-term use of TE for irrigation will differentially affect zones in the soil profile, depending on soil properties, water quality, irrigation management, plant uptake, and climatic conditions. The changes in soil properties echo the fluxes of the main flow processes (infiltration, drainage, and evaporation) in the soil, and consequently, affect water and nutrient availability to plants.

Sufficient concentrations of root zone oxygen are crucial for healthy plant behavior (Armstrong, 1979; Glinski and Stepniewski, 1985). Assouline and Narkis (2013) demonstrated that the changes in the hydraulic properties of TE-irrigated soil impact not only the soil water regime but also root zone aeration. Irrigation with TE additionally affects soil microbial activity (del Mar Alguacil et al., 2012; Elifantz et al., 2011) and composition of the bacterial community (Frenk et al., 2013). Soil aeration and oxygen diffusion rates are likely reduced because of increased input of organic substrates and concurrent changes in water retention properties associated with TE irrigation.

The short duration of most funded research projects (rarely exceeding 3 years) limits our knowledge with respect to long-term impacts of irrigation with TE. Most studies report no significant statistical differences between TE and local fresh water (FW) irrigation in terms of crop yields, with the exception of specific ion toxicity issues, for example as a result of high boron concentrations (Pedrero et al., 2010). Recent long-term studies in Israel have shown systematic decreases in yields of orchards planted on clayey soils (~50% clay) drip-irrigated with TE (Assouline et al., 2015). Following more than 10 years of consecutive TE irrigation, avocado and citrus yields dropped approximately 20–30% in comparison with yields resulting from irrigation with local FW (Fig. 18). Mechanisms explaining the loss of productivity under TE irrigation are yet unknown and likely involve multi-faceted interactions between chemical, physical and biological soil characteristics affecting plant function.

13.4 Future priorities

One way to promote the success of utilizing water resources of marginal quality is to adopt irrigation methods appropriate to local soil and climate conditions and to develop appropriate site-specific irrigation methods and fertigation management protocols (Assouline et al., 2020). Pressurized irrigation methods, and especially drip irrigation, currently globally

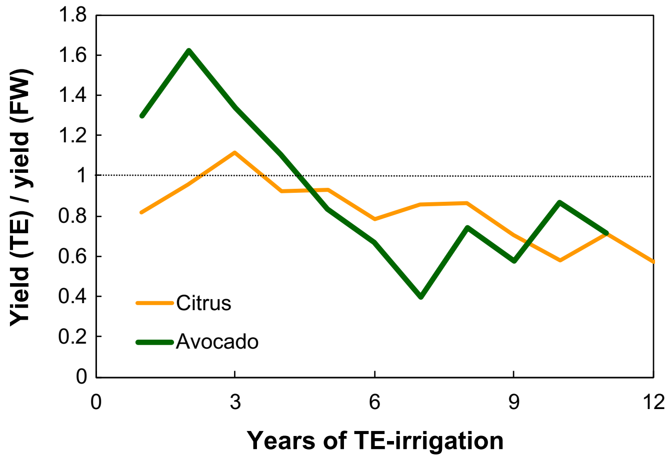

Fig. 18 Impact of long-term TE irrigation on yields: ratio between yields from TE-irrigated and FW-irrigated avocado and citrus trees vs duration of TE-irrigation.

under-utilized, are more efficient than traditional surface irrigation and can minimize environmental impacts and health risks. These advantages come at a cost in terms of infrastructure, knowledge, maintenance and potential vulnerability to crop failure or soil degradation (Assouline et al., 2006; Assouline and Ben-Hur, 2003; Phene and Sanders, 1976; Schneider et al., 2001). In contrast with the large body of knowledge related to the performance of irrigation methods with respect to efficiency and crop response, very little is known about the long-term effects of different irrigation methods using marginal water on soil health and ecological function. Evidence suggests that the extrapolation of knowledge gained from FW irrigation with the various methods to their long-term performance with marginal water is unreliable and that specific monitoring of below ground soil ecological and hydrologic responses for cases of TE irrigation are needed.

Increasing utilization of TE from sources including municipal, industrial, mining, and irrigation drainage waters dictates a need to consider the multiple effects of various ions and DOC on chemical speciation in the soil solution and exchanger phase as a function of irrigation water composition, water movement and solute transport through the soil profile, and crop water uptake (Sposito et al., 2016). Moreover, models need to address physical transport processes as well as geochemistry (Visconti, 2016). Beyond this, the impacts of TE, which is typically high in Na, K, Mg and DOC, on infiltration and hydraulic conductivity of the soil must be understood and quantified.

Current knowledge regarding water quality—soil characteristic relationships is mostly limited to chemical sodicity and salinity factors and largely uncertain relating to other parameters. Evaluation of the impacts of pH, SOM, texture, clay mineralogy, tillage, and irrigation methods, for example, depends for now on field experience. Refinement and further development of current approaches to understanding and managing TE irrigation water, including these additional factors, are therefore important challenges and opportunities.

Along with the expansion of TE, large scale desalination of sea and brackish water (Grant et al., 2012) is rapidly becoming feasible as desalination techniques advance and its costs are continuously and substantially reduced (Beltran et al., 2006; Elimelech and Phillip, 2011; Tal, 2006). Desalinated water (DS) is becoming a competitive source for irrigation, especially for high-value, salt-sensitive cash crops (Kaner et al., 2017). A study on banana irrigation demonstrated that application of DS water can result in a yield increase of approximately 20% for the same amount of allocated FW water or to a significant reduction of about 30% of the irrigation amount if the goal is to achieve a prescribed commercial yield (Silber et al., 2015). However, it has been shown also that there is a need to adapt special fertilization protocols to this mineral-free water (Ben-Gal et al., 2009a,b; Yermiyahu et al., 2007). Desalination has obvious positive impacts on water resources and the environments including augmentation of availability of good quality water and increased quality of TE following its municipal use and recycling. But it also presents several negative impacts for the environment, mainly: brine disposal from desalination process, chemical additives used for antifouling and anticorrosivity; and high consumption of energy that may increase emission of greenhouse gases.

Soil salinization is practically inevitable when low quality water is used for irrigation in dry areas. That said, the actual impact is dependent on the irrigation method, the vertical and spatial distribution of soil properties, topography, cultural practices, weather, and regional hydrological conditions (depth and water quality of local water table). Techniques for improving the quality of available irrigation water by mixing water sources of different qualities have been considered and could be adapted to the irrigation method (Assouline et al., 2015; Ben-Gal et al., 2009a,b; Russo et al., 2015). The appropriate mixing ratio becomes an operational state variable depending on the specific soil properties, climate conditions, and crop characteristics of the system under interest.

Summary: The projected intensification of irrigated agriculture in areas utilizing marginal quality water will undoubtedly affect pre-existing fragile environments and threaten the overall sustainability and functionality of these agro-ecosystems. The future challenge is to devise strategies that increase food production while simultaneously preserving soil ecological functionality, minimizing human health risks, and promoting sustainable use of our land and water resources for agricultural use.

Some of the most critical knowledge gaps, that must be addressed for sustainable and environmentally-responsible intensive agriculture utilizing low or marginal quality irrigation water are: (1) risks to public health, for example by antibiotic resistance induced by wastewater use, or to the long-term ecological functioning of the soil system; (2) interactions between marginal quality water with biological and ecological components; and (3) impacts of future conditions such as climate extremes on agroecosystem sustainability.

14. Additional soil salinity research needs

14.1 Introduction

In this section, we identify several other research areas that are relevant in the context of soil salinity impacts and deserve attention but have not explicitly been covered through the 10 identified priorities of Sections 4–13. These research topics are only briefly discussed, mostly because the available literature is limited, but are likely worthy of further scrutiny. This includes interactions with climate change, soil microbiology, and plant nutrient availability, possible beneficial effects of amending saline soils with biochar, and the potential of planting bioenergy crops on saline soils. We conclude with reviewing the socio-economic impacts and estimated monetary losses associated with salinity.

14.2 Climate change

Climate change is likely to accelerate soil salinization, specifically because of the increased crop water requirements by elevated temperatures, through sea level rise and additionally driven by further limiting freshwater availability for irrigation (Daliakopoulos et al., 2016). It was suggested by Szabolics (1990) that climatic changes can double the areal extent of saline soils. The global impact of the changing climate on land degradation was recently recognized by the Intergovernmental Panel on Climate Change in their report on Climate Change and Land (IPCC, 2019), analyzing interactions and feedbacks between climate, land degradation and food security. The most important direct impacts of climate change on land degradation are the results of increasing temperatures, changing rainfall patterns, and intensification of rainfall. Changes in evapotranspiration and rainfall regimes exacerbate soil salinization, in addition to the intrusion of sea water into coastal areas, both because of sea level rise and land subsidence by groundwater overdraft. Many important indirect linkages between land degradation and climate change occur by way of agriculture. Yield reduction by soil degradation (including salinization) may trigger cropland expansion elsewhere, either into natural ecosystems, marginal arable lands or by intensification, with possible consequences for increasing land degradation. In addition, precipitation and temperature changes will trigger changes in land and crop management, such as changes in planting and harvest dates, type of crops, and type of cultivars. As pointed out earlier (Sections 8–10), much research has been done to understand how plants are affected by a particular stressor, for example, drought, salinity, heat, or waterlogging, but research on how plants are affected by several stressors simultaneously is limited. It is the latter which is more realistic within the context of climate change.

Climate change is causing sea levels to rise worldwide, particularly in tropical and subtropical regions. Assessing the extent of salinization due to sea water intrusion at a global scale has remained challenging. Seawater intrusion in coastal areas is generally caused by increased tidal activity (storm surges, hurricanes), increased groundwater extraction or land-use change, causing contamination of nearby freshwater aquifers (Uddameri et al., 2014). The Indus delta in Pakistan (Rasul et al., 2012), the San Joaquin Valley (SJV) in California (Section 14.2) and coastal countries around the North Sea (Section 14.7) are clear examples of increased soil salinization by seawater intrusion.

The direct impacts of a changing climate on soil salinization have only been recently explored. In Hopmans and Maurer (2008) potential regional-scale impacts of global climate change on sustainability of irrigated agriculture were examined, focusing on California's western SJV. The modeling study (regional-scale hydro-salinity model) analyzed potential changes in irrigation water demand and supply, and quantified impacts on cropping patterns, groundwater pumping and groundwater levels, soil salinity, and crop yields, based on General Circulation Model (GCM) climate projections through 2100 and using three greenhouse gas emission scenarios. Crop water demand was expected to change very little, due to compensating effects of rising temperature on evaporative demand and crop growth rate. This simulation study projected that reductions in surface water supply are going to be offset by groundwater pumping and land fallowing, whereas soil salinity is expected to increase in downslope areas, thereby limiting crop production. The results also showed that technological adaptation, such as through improvements in irrigation efficiency, may partly mitigate these effects. Another recent computer modeling study (Haj-Amor et al., 2020) for the Tunisian coastal region, simulated changes in coastal aquifer salinity and the associated increased groundwater pumping required to offset the increased irrigation requirements and soil salinity levels. Corwin (2020) evaluated various climate change impacts on soil salinity through analysis of various case studies in selected countries with different soil salinization processes with a focus on methods (Sections 3 and 5) for monitoring soil salinity development.

14.3 Microbial processes

In addition to climate parameters affecting soil microbiological processes directly, specifically relevant as to their contribution to greenhouse gas emissions of CO_2, N_2O and methane by soil respiration and redox reactions, respectively, soil scientists are considering secondary soil salinity effects on soil microbiological processes. For example, Egamberdieva et al. (2010) reported reduced soil microbial biomass with increased soil salinity, comparing a wide range of salinity levels for field grown cotton in Uzbekistan where salinity has significantly increased after the expansion of irrigated agriculture in the 1960s. They suggested that the lower microbial population was caused by increased microbial stress by both osmotic and toxic effects. In a subsequent review article (Egamberdieva et al., 2019), the isolation of salt-tolerant plant growth promoting rhizobacteria (ST-PGPR) from both saline and sodic soils evidenced that these could mitigate both biotic and abiotic stresses. It is suggested that selected rhizobacteria can be inoculated to reclaim saline agro-ecosystems, enhancing their productivity and soil fertility. Furthermore, it is proposed to prioritize gene-level studies of ST-PGPR, parallel to that of seeking salt-tolerant crop species. Similarly, Shrivastava and Kuman (2015) proposed that microorganisms could play a significant role toward soil salinity stress management, and pointed to the need to further exploit selected unique properties such as salt tolerance and other interactions with crop plants such as the production of plant growth promoting hormones and bio-control potential. In the last decade,

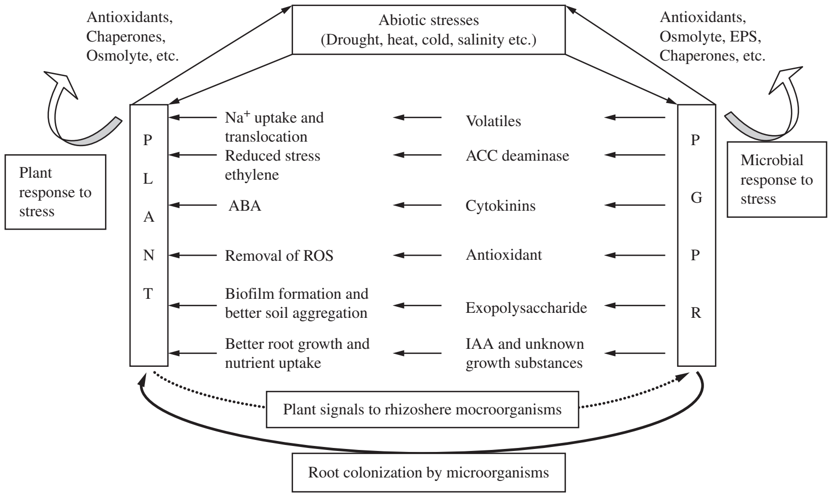

Fig. 19 Conceptual diagram, on the plant–microbe interactions under abiotic stress (Grover et al., 2011).

many different genera of bacteria have shown to provide tolerance to host plants under different abiotic stress environments (Grover et al., 2011; Fig. 19). As pointed out in this review of the role of microorganism to mitigate abiotic plant stresses, their use can open new and emerging applications in agriculture and also provide excellent models for understanding stress tolerance, potentially to be engineered into crop plants to cope with abiotic stresses such as soil salinity.

In another study by Marks et al. (2016) it was demonstrated that dramatic changes in salinity of salt marsh soils as caused by storm surges or freshwater diversions can greatly affect denitrification rates, which is especially relevant for nutrient removal management of eutrophic waters such as for the Mississippi delta. Rath et al. (2017) studied such dynamic conditions by the bacterial response to drying-rewetting in saline soils and concluded that increased soil salinity prolonged the time required by soil microbes to recover from drought, both in terms of their growth and respiration.

14.4 Biochar amendment

Biochar is defined as organic matter that is carbonized by heating in an oxygen-limited environment. The properties of biochar vary widely, dependent on the feedstock and the conditions of production. Biochar is relatively resistant to decomposition compared with fresh organic matter or compost, and thus represents a long-term carbon store. Biochar stability is estimated to range from decades to thousands of years, but its stability decreases as ambient temperature increases. It has been shown that application of biochar to soil can improve soil chemical, physical and biological attributes, enhancing productivity and resilience to climate change, while also delivering climate-change mitigation through carbon sequestration and reduction in GHG emissions (IPCC, 2019).

Chaganti et al. (2015) evaluated the potential of using biochar to remediate saline–sodic soils in combination with various other organic amendments using reclaimed water with moderate SAR. Results showed that leaching with moderate SAR water was effective in reducing the soil salinity and sodicity of all investigated soils, irrespective of amendment application. However, it was shown that combined applications of gypsum with organic amendments were more effective to remediate saline–sodic soils, and therefore could have a supplementary benefit of accelerating the reclamation process. Akhtar et al. (2015) used a greenhouse experiment to show that biochar amendment for a different soil salinity levels could alleviate the negative impacts of salt stress in a wheat crop through reduced plant sodium uptake due to its high adsorption capacity, decreasing osmotic stress by enhancing soil moisture content, and by releasing mineral nutrients into the soil solution. However, it was recommended that more detailed field studies must be conducted to evaluate the long-term residual effects of biochar.

14.5 Plant nutrient availability

The application of marginal waters to augment irrigation water supplies particularly has led to investigations to evaluate plant nutrient uptake impact of saline-sodic soils. It has been shown that soil salinity can induce elemental nutrient deficiencies or imbalances in plants depending on ionic composition of the soil solution, due their effect on nutrient availability, competitive uptake, transport, and partitioning within the plant (Grattan and Grieve, 1999; Section 8; Fageria et al., 2011). Most obviously, soil salinity affects nutrient ion activities and produces extreme ion ratios in soil solution. As a result, for example, excess Na^+ can cause sodium-induced Ca^{2+} or K^+ deficiency in many crops (Grieve et al., 2012). Nutrient uptake and accumulation by plants is often reduced under saline soil conditions because of competition between the nutrient in question and other major salt species, such as by sodium-induced potassium deficiency in sodic soils. Soil salinity is expected to interact with nitrogen both as competition between NO_3^- and Cl^- ions in uptake processes as high chloride concentrations may reduce nitrate uptake and plant development (Chen et al., 2010; Jadav et al., 1976; Yasuor et al., 2017), and indirectly through disruptions of symbiotic N_2 fixation systems (Fageria et al., 2011).

Interactions with phosphorus vary with plant genotype and external salinity and P concentrations in soil solution, which are highly dependent on soil surface properties. There is general evidence of reduced P uptake in salt affected soils. Calcium magnesium and sulfur as well as micronutrients all interact with soil salinity, Na and one another. Imbalance of these elements cause various pathologies in plants including susceptibility to biotic stresses (Bar-Tal et al., 2015).

14.6 Biosaline forestry

Among potential alternative land uses of saline soils is their economic potential for biomass production using forestry plantations (biosaline forestry, various case studies in Section 14), as many tree species are less susceptible to soil salinity and sodicity than agricultural crops. A thorough review of the economic potential of bioenergy from salt-affected soils has been presented by Wicke et al. (2011). Using the FAO soil salinity database, they estimated that the global economic potential of biosaline forestry is about 53 EJ (exajoule) y^{-1} (close to 10% of global primary energy consumption), when including agricultural land, and to 39 EJ y^{-1} when excluding agricultural land. Plantation forestry has been advocated to control dryland salinity conditions, with fast growing versatile Eucalyptus species to lower shallow groundwater tables, however, salinity/sodic stresses in the long-term prohibit significant economic returns (Minhas et al., 2020a,b). Much will depend on regional production costs. Studies have shown that biosaline forestry may contribute significantly to energy supply in certain regions, such as sub-Saharan Africa (SSA) and South Asia, and has additional benefits of improving soil quality and soil carbon sequestration (carbon forestry), thus justifying investigating biosaline forestry in the near future.

14.7 Socio-economic impacts

Economic losses of productive land by salinization are difficult to assess, however, various evaluations have reported annual costs of US \$250–500/ha (Qadir et al., 2014), suggesting a total annual economic loss of US\$30 billion globally (Shahid et al., 2018). As pointed out by Qadir et al. (2014), a large fraction of salt-affected land is farmed by smallholder farmers in Asia and SSA, necessitating off-farm supplemental income activities, with others leaving their land for work in cities. Given that much of the projected global population growth is in those regions, prioritization of research and infrastructure investments to mitigate agricultural production impacts there is extremely relevant.

A thorough analysis of the production losses and costs (including employment losses) of salt-induced land degradation was done by Qadir et al. (2014), based on crop yield losses, however, they point to the need to also consider additional losses such as by unemployment, health effects, infrastructure deterioration, and environmental costs. Their calculations compared economic benefits using cost-benefit analysis of "no action" vs "action" for various case studies. A yield gap analysis by Orton et al. (2018) for wheat production in Australia showed that soil sodicity alone represented 8% of the total wheat yield gap, representing more than AUS $1 billion. In their sustainability assessment of the expanding irrigation in the western US, comparing real outcomes with those predicted by Reisner (1986) in this book Cadillac Desert, Sabo et al. (2010) included an economic analysis of agricultural revenue losses as a result of the increased soil salinity for the western US (west of the 100th meridian). Using the USDA NRCS soil's data base, and available crop salt tolerance information, they estimated a total annual revenue loss by reduced crop yields of 2.8 billion US dollars. In all, land values of salinized lands depreciate significantly and incur huge economic impact, putting into question the sustainability of agricultural land practices that induce soil salinization (Section 16).

15. Case studies

In this section, we present case studies across the major irrigated regions in the world, in alphabetical order. Each study will include an introductory paragraph summarizing the historical development of the specific region or country that has led to soil salinization, followed by their more recent progress in addressing impacts. Each section will conclude with an outlook, listing additional requirements that will need to be achieved to further limit land degradation and loss of prime agricultural lands by soil salinity in the future.

15.1 Australia

15.1.1 Historical development

Australia is the world's driest inhabited continent with an average annual rainfall of 420 mm with a high potential for the formation of salt-affected landscapes. Development of agricultural practices in Australia began after the European settlement and was widely adopted during 20th century. Earlier, the indigenous population found their food by hunting and foraging. They indirectly depended on soils for plant food, but they did so without soil management. The European settlers were unaware of the soil characteristics they had to work with.

Salt has been accumulating in the Australian landscape over thousands of years through small quantities blown in from the ocean by wind and rain. In addition to mineral weathering, salt accumulation is also associated with parna, a wind-blown dust coming from the west and the south-west of the continent (Munday et al., 2000). Many soils of the arid to subhumid regions of Australia contain significant amounts of water-soluble salts, dominantly as sodium chloride. Their dense subsoils are frequently characterized by moderate to high amounts of exchangeable sodium and magnesium (Hubble et al., 1983), and are generally named duplex soils. Discussing the genesis and distributions of saline and sodic soils in Australia, Isbell et al. (1983) concluded that salts from a variety of sources have probably contributed to the present saline and sodic soils.

In the early part of 20th century, the Australian government initiated a nation-wide soil survey with soil analysis. As early as the 1930s, soil surveys in the Salmon Gums district, Western Australia, found that salt accumulation in surface and subsoils (up to 60 cm depth) occurred in more than 50% of the 0.25 million ha surveyed (Burvill, 1988). These surveys also found that virgin areas had higher accumulations of salts in the upper meter than in vegetation-cleared areas for the major soil types. In one of his earlier observations in the Mallee region of Southern Australia, Holmes (1960) found a salt bulge that was more than 4 m below the surface in a virgin heath community.

Northcote and Skene (1972), examining numerous data relating to the morphology, salinity, alkalinity, and sodicity of Australian soils presented the areal distribution of saline and sodic soils in Australia, using the classification of salt-affected soils of Table 2. While 32.9% of the total area in Australia is salt-affected, sodic soils occupy 27.6% of this area. Hence, most of the research during the middle of the 20th century focused on sodic soils and their management. Northcote and Skene (1972) defined sodic soils as those having an ESP between 6 and 14, and strongly sodic soils as those having an ESP of 15 or more. The recent Australian soil classification (Isbell, 2002) defined "Sodosols" (sodic soils) as soils with an ESP greater than 6. However, soils with ESP 25–30 were excluded from sodosols, because of their very different land-use properties.

Table 2 Classification and area of salt-affected soils in Australia (Northcote and Skene, 1972).

Map unit	Salt-affected soil category	Area (km^2)	% of total national land area
SS	Saline soils	386,300	5.3
AS1	Alkaline strongly sodic to sodic clay soil with uniform texture profile	666,400	9.2
AS2	Alkaline strongly sodic to sodic coarse- and medium-textured soils with uniform and gradational texture	600,700	8.3
AS3	Alkaline strongly sodic to sodic duplex soils	454,400	6.3
NS1	Non-alkaline sodic and strongly sodic neutral duplex soils	134,700	1.9
NS2	Non-alkaline sodic acid duplex soils	140,700	1.9
	Total	2,383,200	32.9

Note: 1000,000 km^2 is equivalent to 100 Mha.

15.1.2 Current progress

Salinity in Australian landscapes has developed under different environmental conditions over many geological periods. More recent agricultural activities have caused additional types of salinity. Local communities and governmental agencies in Australia are concerned about the impact of salinity on agricultural production, land values and water resources. Consequently, the major salinity foci in Australia has been on (a) irrigation-induced salinity in the Murray Darling Basin and (b) dryland salinity associated with shallow groundwater, particularly in Western Australia.

Reviewing salinization processes with a focus on Australia, Rengasamy (2006b) concluded that salt accumulation in the landscape is governed by specific processes in combination with climatic and landscape features as well as human activities. It is therefore that he identified three major types of salinity commonly found in Australia (see Fig. 20), with total area affected in Table 2. His classification is different from the usual classification of "Primary" or "Secondary" salinity (Ghassemi et al., 1995), but could be applied outside Australia as well.

1. *Groundwater associated salinity*—It characterizes salt accumulation in discharge areas where water exits from groundwater, bringing dissolved salts toward the soil surface through upward movement of water, driven

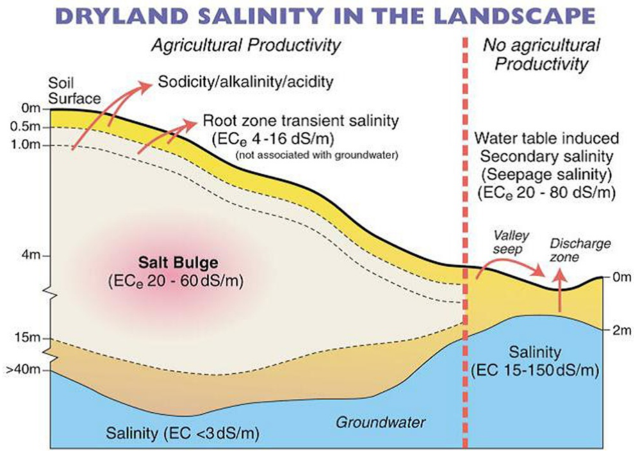

Fig. 20 Different types of salinity in Australian landscapes (Rengasamy, 2002).

by soil evaporation and plant transpiration. Salt accumulation is generally higher when the water table is less than 1.5 m below the soil surface. In Australia, leaching of salts from upper layers areas under native perennial vegetation, led to salt storage in deep regolith or in the shallow groundwater. Because of the clearance of this native vegetation through the introduction of agriculture, groundwater levels have risen toward new equilibrium levels (Hatton et al., 2003). As an unforeseen result of the clearing of deep-rooted native vegetation, saline groundwaters approached the surface, with topsoil layers being salinized and waterlogged. This type of salinity, associated with shallow groundwater, became a major focus in Western Australia early on (George et al., 1997). The National Land and Water Resources Audit (2001) warned that unless effective solutions are implemented, this form of salinity could increase to 17×10^4 km^2 (17 Mha) in Australia by 2050.

2. *Non-groundwater-associated salinity (transient salinity)*—This type of salt accumulation occurs predominantly in landscapes where the water table is deep and drainage is poor, as caused by unfavorable hydraulic properties of near-surface soil layers. The levels of salinity and the depths at which it occurs vary according to climatic conditions and is therefore defined as "transient salinity" (Rengasamy, 2002). This type of salinity is extensive in many landscapes dominated by subsoil sodicity and low rainfall, such as in northern Australia (Shaw et al., 1998), Western Australia (Barrett-Lennard et al., 2016; McArthur, 1991) and southern Australia (Rengasamy, 2002).

Table 3 Distribution of different types of salinity in Australia.

Types of salinity	Approximate area (Mha)	Percentage of total land area
Water table-induced salinity	5.66	0.070
Transient salinity (non-water table-associated)	253.00	30.00
Irrigation associated	1.23	0.002

After Rengasamy, P., Tavakkoli, E., McDonald, G.K., 2016. Exchangeable cations and clay dispersion: net dispersive charge, a new concept for dispersive soil. Eur. J. Soil Sci. 67, 659–665.

3. *Irrigation associated salinity*—Salts introduced by irrigation water are stored within the root zone because of insufficient leaching. Most of the areas developed for irrigation are within the Murray-Darling Basin in Australia, covering about 1.23 Mha (Table 3). Although irrigated agriculture is limited in Australia, representing only 1% of the total land used for agriculture, the gross value of irrigated agricultural production in 2006–2007 was 34% of the country's total gross value of agricultural production (ABS, 2010). Irrigation of saline-sodic soils using low-salinity stream water has led to the formation of sodic soils. Paradoxically, irrigation of sodic soils led to the salt accumulation, causing the formation of saline-sodic soils. Hence, major efforts of management of soil sodicity are focused on irrigation water management. However, the recent use of waste and effluent waters has introduced significant amounts of potassium and magnesium ions into the soils in addition to sodium, creating problems associated with soil structure and crop production (Laurenson et al., 2010).

Currently, limited research efforts are ongoing related to irrigation-induced salinity, and most focus is on salinity in dryland cropping regions. As noted in Table 2, sodic subsoils with high pH are prevalent in most of the states. These soils have moderate to severe constraints to agricultural productivity imposed by salinity, sodicity (dispersivity), alkalinity, acidity, and elemental toxicities and deficiencies. Soil and agronomic management practices are undertaken to mitigate these constraints (McDonald et al., 2013). In addition, there is significant focus on adaptation, with screening, selecting, breeding and genetic engineering of crops for tolerance to salinity, sodicity and ion toxicity (Munns, 2005).

15.1.3 Future outlook

Soil salinity affects more than 33% of the total land area in Australia, with most of it (27.6%) being sodic soils with the potential to develop transient salinity (Rengasamy, 2002), and is therefore a major concern limiting agricultural production and the nation's economy. The following activities are necessary to mitigate salt-induced problems:

1. Salinity issues are site specific and vary widely within a paddock. Further, these variations can also occur between different soil layers of a soil profile. A major focus should be on developing remote-sensing techniques to characterize salinity variations between soil layers, within a paddock, as well as between soil layers. Moreover, regional soil salinity mapping is necessary for planning and implementing regional strategies.
2. Multiple problems occur in sodic soils due to variations in soil pH (acidity and alkalinity). In addition to soil structural degradation, acidic and alkaline pH induce elemental toxicities. Developing soil management strategies that address both soil structural degradation and elemental toxicities are important.
3. In addition to soil management research, additional efforts are needed in plant breeding to manipulate root adaptation to Australia's subsoil limitations. High priority research includes developing plants that modify the rhizosphere and adapt to edaphic conditions. For such research to be successful, one must first identify the dominating soil factor limiting crop yield, among a wide range of soil constraints. Thus, collaborative research efforts across disciplines in soil and plant sciences is needed, working in a real field setting rather than in idealized environmental conditions.
4. Currently, total EC of the soil solution is used to quantify the salinity effects on plants. Similarly, ESP or SAR is used to assess the sodicity effects on soil physical properties. Yet, recent studies have shown the importance of individual ions in soil solution on salinity and sodicity effects (Section 12). To further this type of research, developing simple methods using modern techniques to characterize the ionic composition of soil solutions is required.

15.2 California

15.2.1 Historical development

California's natural geology, hydrology and geography create different forms of salinity problems across the state, ranging from sea water intrusion induced salinity along the central coast to concentration of salts in closed basins such as the Tulare Lake basin in the Central Valley (Fig. 21). In addition, some of the most productive soils in California such as in the western San Joaquin Valley originate from ocean sediments that are naturally high in salts. Irrigation water dissolves that salt and moves it downstream or it is

Fig. 21 California water distribution map. *Source: https://sites.uci.edu/energyobserver/files/2015/04/California-Aqueducts.gif*

infiltrated to groundwater increasing its salt content. California's extensively modified water distribution system (Fig. 21) including the state and federal water projects also carry large amounts of salt into and out of different waters.

Salinity issues were exacerbated in California starting around the second half of the 19th century when commercial irrigated agriculture was introduced to the state (Kelley and Nye, 1984). Historically, both surface water and groundwater are used for irrigation in California. Surface water used for irrigation is relatively low in salinity, especially when derived from snow melt from the Sierra Nevada mountains.

The salinity in the Colorado river used for irrigation in the Imperial Valley is higher than that of surface water from the snow melt. Although salinity problems can be found in various locations around California as shown in Fig. 22, historically the major salinity issues are found in the Western San Joaquin Valley and the Imperial Valley. A thorough review of the history of irrigation in California was presented by Oster and Wichelns (2014). Today, California's interconnected water system irrigates over 3.4 Mha of farmland (USDA, NASS, 2018).

The Imperial Valley in southern California has experienced salinity problems for many decades, since the Colorado river was tapped for irrigation in the early 1900s. By 1918 salinity had forced approximately 20,234 ha out of production and damaged thousands more hectares (Kelley and Nye, 1984). The rapidly deteriorating agricultural lands from salinization forced the Imperial irrigation District (agency responsible for water delivery) to construct open ditch drainage channels. However, due to high salinity in

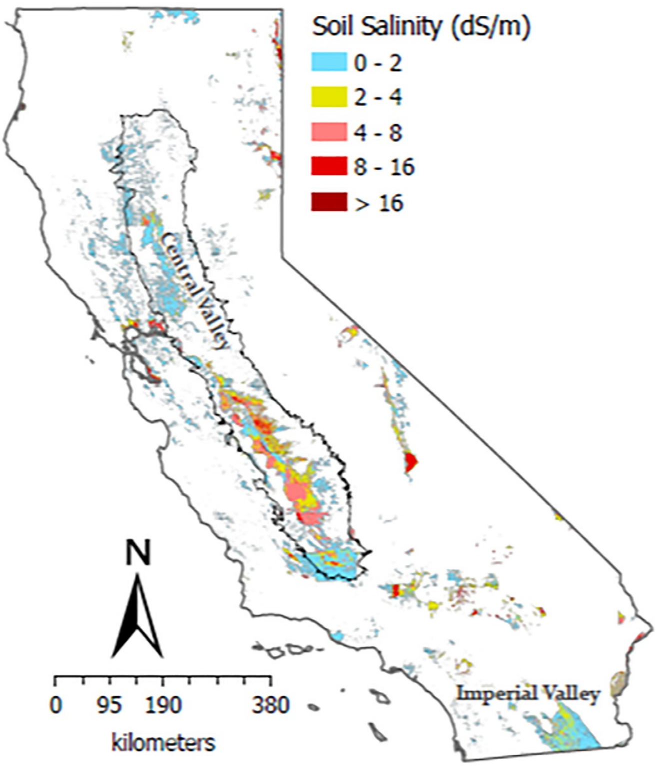

Fig. 22 Salt affected soils in California.

the Colorado river water, heavy soils and poor on-farm water management at the time, the drainage system did not prevent continued salinization of the Imperial Valley. To address the problem, partnerships between the federal government, and the Imperial irrigation district were formed in the early 1940s that resulted in installation of underground concrete and tile drainage on thousands of hectares of farms. The subsurface drainage system and improved on-farm water management led to a reduction in the rate of soil salinization, resulting in flourishing agricultural production in the Imperial Valley. The water from the subsurface drainage tiles was routed to the Salton Sea. However, agricultural runoff and drainage flows with high salt content have affected the elevation of Salton Sea and increased its salinity threatening various wildlife species. On the positive side, the salinity load coming into the Imperial Valley as measured by salinity levels at the Imperial dam have not increased as previously projected. A report from the US Bureau of Reclamation (2013) reported a flow weighted salinity of 680 mg/L in 2011 at the imperial dam and had remained constant for past decades.

Another major region in California significantly impacted by salinity is the western San Joaquin Valley (SJV), comprising the southern half of the Central Valley (Fig. 22). From the second half of the 19th century to the early 1900s the SJV experienced rapid development of irrigated agriculture, along with it came drainage and salinity problems. The salinity problems on the Westside of the valley can be attributed to (i) high water tables near the valley trough caused by an expansion of irrigated agriculture upslope from the valley, (ii) soils on the Westside are derived from alluvium originating from coastal mountains and other marine environments, and (iii) degradation of water quality in the San Joaquin river (Fig. 21). In 1951, some of the fresh water in the San Joaquin river was diverted to irrigate agricultural lands on the eastside north of Friant dam. The diverted water was replaced with saltier water from the Central Valley project.

These changes coupled with agricultural return flows led to increased salinity downstream of the San Joaquin river, the main conduit draining the valley. Drainage and salinity problems on the Valley's Westside were exacerbated by the construction of the San Luis Unit (Fig. 23). The San Luis Unit authorized by the Luis Act of 1960 is part of both the federal Central Valley Project and the State Water Project. The primary purpose of the San Luis Unit was to supply irrigation water for over 400,000 ha of prime farmland (US Bureau of Reclamation, n.d.). The Luis Act of 1960 as part of a comprehensive basin salinity management plan required that drainage be constructed either as a master drain constructed by the state

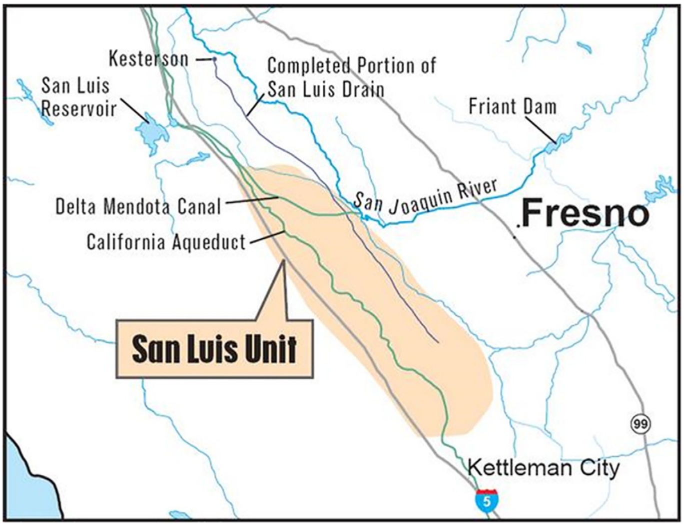

Fig. 23 Location of the San Luis Unit on the Westside of San Joaquin Valley. *Source: USBR.*

of California to serve the entire valley or an interceptor drain constructed by the federal government to serve the San Luis Unit service area. The idea was that either of these two drainage systems would convey brackish water northward in a concrete canal into the Sacramento-San Joaquin delta (Kelley and Nye, 1984).

In mid-1960s the federal and state governments started planning for a master drain that would drain and transport salts out of the entire valley from Bakersfield on the southern end of the valley to the delta. However, in the early stages of the project the state of California withdrew after failing to get assurances from irrigators that they would pay state expenditures for the project (Kelley and Nye, 1984). In 1968 the federal government through the Bureau of reclamation started construction of a drain system to collect and transport subsurface drainage water from the San Luis Unit service area to the Sacramento-San Joaquin Bay-Delta. However, of the planned 302 km of drain, only 140 km were completed from Kettleman City, near Fresno County, to Kesterson Reservoir in Merced County. Construction was halted in 1975 because of mounting costs and water quality concerns. The main water quality issue was selenium in the Kesterson National Wildlife Refuge which caused various ecological concerns including wildlife birth defects and other toxicities (Chang and Brawer Silva, 2014). These instances had a major impact on irrigated agriculture in California. To date the project has stalled due in part to ecological and environmental concerns. In terms of salinity management, the failure to complete the drainage system resulted in reduced agricultural productivity on many farmlands particularly in the Western San Joaquin due to shallow water tables and evapo-concentration of salts in the root zone. Lack of a system to export drainage water and salts out of the valley has stimulated innovative management practices to reduce drainage waters and to find "in valley" solutions for disposal.

15.2.2 Current progress

The threat that salinity poses to California's economy is widely acknowledged by both public and private stakeholders. For example, a study by Howitt et al. (2009) reported that Central Valley salinity accumulations would cause an estimated loss of $2.167 billion in California's value of goods and services produced by 2030, if they remain unmanaged. Incomes would decline by $941 million while employment would reduce by 29,270 jobs. Potential benefits of implementing salinity management strategies in the Central Valley were estimated at over $10 billion. It is reasonable to assume that improved salinity management could bring economic benefits to other regions of California that experience salt problems including the Imperial Valley and the Central Coast, while neglecting this problem would bring dire consequences. In California current efforts to address salinity management have included both traditional and contemporary strategies. Traditional salinity management strategies have included source control (mostly for point source), dilution, and displacement (e.g. leaching management). While contemporary strategies have included salinity management such as treatment (e.g. desalination of brackish water), storage, export, real-time management, and recycling as described in a 2016 inter-agency report by the California Department of Water Resources (2016). The following sections describe recent salinity management case studies in California ranging from on-farm to basin-wide efforts.

On-farm salinity management—In California providing environmentally and politically acceptable disposal of drainage water from irrigated agricultural lands is a major challenge for growers. Ayars and Soppe (2014) reported to have successfully used a technique called Integrated On-Farm Drainage Management (IFDM) to significantly reduce drainage water to 0.7% of field-applied irrigation water, eliminating the need for evaporating ponds. IFDM was demonstrated on four 65-ha fields located at Red Rock Ranch on the Westside of the San Joaquin Valley of California, by sequentially using saline drainage water for supplemental irrigation. In this study three of the 65 ha blocks were used to grow salt sensitive crops (tomatoes and garlic) and drainage from these blocks was used to irrigate a salt-tolerant crop (wheatgrass). IFDM has been successfully used on other farms in the San Joaquin Valley, e.g., Andrews Ag farm, located in Kern County where IFDM was implemented on 486 ha (State Water Resources Control Board, 2004). At Andrews Ag, salt-sensitive crops (lettuce, bell peppers, melons, carrots, garlic, and onions) were irrigated using drip and sprinkler irrigation. A subsurface drainage system collected the drainage water that was subsequently used to irrigate salt-tolerant crops such as cotton. Halophytes (native salt grass and iodine bush) were grown using the drainage water coming from the salt-tolerant crop. The salt grass volatilizes selenium as it grows, removing it from the drainage water and rendering it harmless. By 2005 the farm reported that it was able to reduce drainage by 90% and selenium by 80%.

Eliminating the need for an evaporating basin provides several benefits, including minimizing the size of land taken out of production and need to mitigate environmental impacts associated with evaporating basins such as leaching of salts to groundwater. However, it is worth noting that management practices such as IFDM only provide short term solutions, and that long-term sustainable irrigation requires exporting salts out of the basin to maintain a salt balance, for example through a brine line.

Regional salinity management—A major regional initiative to address the salinity problem in California is the Central Valley Salinity Alternatives for Long-term Sustainability (CV-SALTS). CV-SALTS is focused on sustainable salinity management. CV-SALTS is a collaborative effort that was initiated in 2006 to find solutions to the salt problem in the Central Valley. It includes several stakeholders such as the State Water Resources Control Board, the Central Valley Regional Water Quality Control Board, agricultural coalitions, cities and municipalities, growers, academics, and environmental justice groups. The goals of CV-SALTS are multifaceted and include sustaining the Central Valley's lifestyle, support regional economic growth, sustain agricultural economy, maintain a reliable and high-quality urban water supply, and protect and enhance the environment. Because of the seriousness of salt and nitrate issues in the Central Valley, the California State Water Resources Control Board voted in 2019 to approve a Central Valley-wide Salt and Nitrate Control Program that was submitted by CV-SALTS. Subsequently, the Regional Central Valley Water Board started sending out Notices to Comply for the Nitrate Control Program in late May 2020. The Salt and Nitrate Control Program includes both short- and long-term strategies to address salt issues in the central valley. Dischargers can participate in the program as individuals or as a part of a group of dischargers organized in form of a management zone. This is significant because the Salt and Nitrate Control Program provides a framework for the Central Valley Water Board to regulate salt and nitrate discharges for an area covering 46,619 km^2.

15.2.3 Future outlook

Issues of salinity in California have tremendous consequences, as there is a lot at stake in terms of economic losses, environment degradation and livelihood disruptions. Therefore, to not proactively address salinity is not an option. In California salt moves statewide through the interconnected waterways across different basins. Salinity management should carefully integrate water flows and salt loadings. Sustainable salinity management decisions in any basin involves a wide range of stakeholders such as water managers, regulators, facility operators, policy makers, landowners, growers, agricultural coalitions, environmental justice groups and others.

To successful manage salinity in California these entities must strive to coordinate their efforts to use resources efficiently, develop solutions to local and regional problems, optimize funding opportunities, and seek to achieve a salt balance in any given basin. Sustainable salinity management in California will require collaborative efforts to build consensus on scientifically proven solutions that meet multiple objectives for its diverse regions. Both short and long strategies will need to be considered, for example, to achieve a salt balance in a closed basin such as the Tulare Lake basin, discussions must include options for exporting salts out of the basin using a brine line. Water conservation in the Salton Sea basin should be integrated with salinity management. Integrated approaches should be pursued to mitigate sea saltwater intrusion including substituting groundwater pumping in agricultural regions along the central coast with recycled water.

15.3 China

15.3.1 Historical development

Salt-affected soils have a broad distribution and a rich variety of types in China, totally accounting for approximately 100 Mha, or about 1/10th of the entire land area of the country (Li, 2010). Climatic conditions, landform and geomorphology, and agricultural practices are key factors influencing

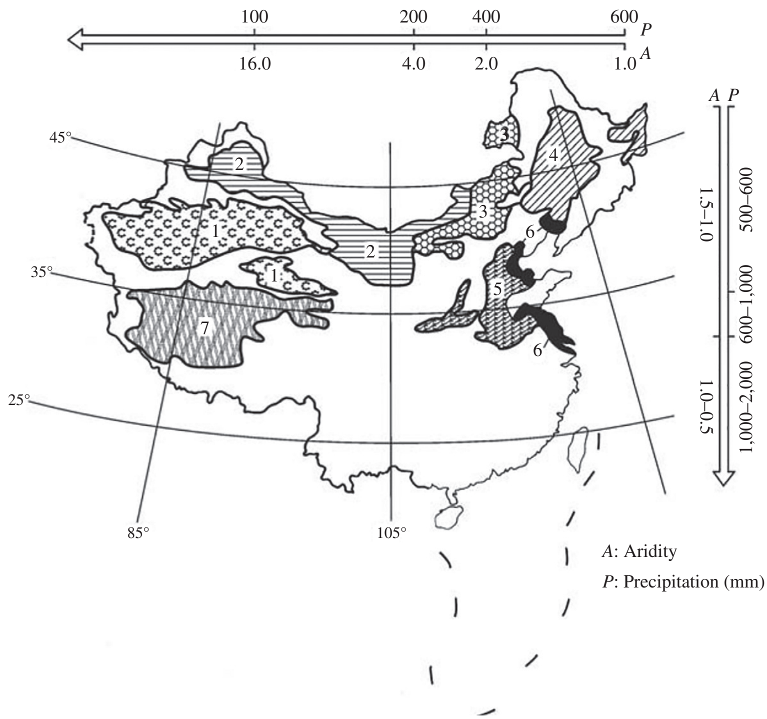

Fig. 24 Salt-affected soil zone in China (Shi, 1986). (1) salt-affected soil zone in extremely arid desert; (2) salt-affected soil zone in arid desert and desert steppe; (3) salt-affected soil zone in arid and semi-arid steppe region; (4) soda salt-affected soil zone in semi-arid and semi-humid climates; (5) salinized soda alkalized salt-affected soil zone in semi-humid monsoon climates; (6) coastal salt-affected soil zone in semi-humid and humid monsoon climate; and (7) salt-affected soil zone in high altitude cold deserts, lakes and basins.

soil salinization in this country (Meng et al., 2016a; Wang, 1993; Yang et al., 2015). The ratio of evaporation to precipitation is often more than one in most regions of Northern China. According to the formation characteristics and geological distribution, salt-affected soils are clarified into seven major zones (Fig. 24; Shi, 1986). In Northwest China, the closed inflow basins (e.g. the Tarim-, Turpan- and Qaidam-basins, and the Hexi Corridor), provide the physical base for the development of soil salinization, which associated with the localized arid and hot climate conditions eventually result in the formation of salt-affected soils. In Northeast China and North China Plain, controlled by the monsoon climate, 60–70% of the precipitation occurs in summer, resulting in a cycle of summer waterlogging and spring drought. Therefore, salt is frequently exchanged between the soil and groundwater. China also has a large area of coastal low plains distributed with salt-affected soils that are mainly due to the seawater encroachment (Li, 2010).

Modern integrated investigations on land resources at large-scales in China were started in the 1950s, which have provided an important foundation for improved understanding of the geographical and genetic classifications of salt-affected soils. From then on, research and management practices have been focusing on the reclamation, improvement and sustainable utilization of salt-affected soils regionally, typically located in Xinjiang, Ningxia, Inner Mongolia, and the Songnen plain (in region 1, 2, 3, and 4 in Fig. 24). Key measures to control soil salinity included artificial salt-leaching, paddy rice sowing, fodder rotation, and application of drainage and irrigation systems. Relying on these measures, China has promoted agricultural development in some of its salt-affected soil zones, especially in Northwest China (Li, 2010). However, the inadequate (often primitive) irrigation and drainage systems resulted in a dramatic rising of the groundwater table across these regions, and eventually led to secondary soil salinization (Nurmemet et al., 2015). Nevertheless, agricultural development led to research and use of water-conserving agricultural technologies, including those that control groundwater depth, prevent water losses of irrigation canals, and building of open ditch and subsurface drainage systems (Huang and Wei, 1962). Additional engineering and agronomic practices developed during this period, including land leveling, flooding sedimentation, green manure planting, organic manure application, and salt-tolerant crop selection (Yang, 2008).

After mid-1970s, key state research projects were launched aiming for integrated management of drought, waterlogging, and soil salinization (Li, 2010). Typically, a national project was initiated in Huang-Huai-Hai Plain in 1978. To systematically study the interrelation and regularities of drought, waterlogging and soil salinization, a system for monitoring and predicting the regional water and salt (PWS) was developed (Li et al., 1993). The focus of soil management in this region was on shallow groundwater exploitation. The shallow groundwater water was extracted from tube wells and used for irrigation, which simultaneously lowered the groundwater table. In addition, the low-pressure water transport technique, deep ditches, optimized fertilizer, and shelterbelt were used to improve the basic conditions of agricultural production. By 1995, the agricultural total output value in this region was raised by 20–56% (Shi, 2003). Meanwhile, important progresses were made in the other regions, such as drainage-based rice sowing in Xinjiang and Ningxia, soda-saline soil improvement in Jilin Province (in region 4 in Fig. 24), coastal saltmarsh development, and improved agricultural drainage systems in Inner Mongolia (Li et al., 2014).

15.3.2 Current progress

Food security is a long-lasting challenge for China because it needs to feed 20% of the world's population, relying on only 7% of the world's arable land. The implementation of the integrated salt-affected soil management projects has improved nearly 1.67 Mha of saline-alkali land and has increased nearly 4 million tons of grain production since 2000.

Since 2000, China has prioritized the application of water-saving irrigation (WSI) techniques, especially in arid and semi-arid regions, including the use of pressurized irrigation such as drip and sprinkler irrigation, as well as subsurface irrigation. By the end of 2015, the total area of water-saving irrigated cropland was about 31 Mha in China, including 9 Mha with sprinkler and drip irrigation (Yao et al., 2017). In addition, China has initialized multiple policies to facilitate intensive implementations of WSI technologies, e.g. to mobilize local governments by providing additional funds for WSI investment, and to promote the Water Users Associations to take on the irrigation management responsibilities in rural areas (Yao et al., 2017). Furthermore, China has launched a comprehensive water management plan (CWMP) in 2006 to improve agricultural water use efficiency. By the end of 2015, it was reported that the average agricultural water-use efficiency increased from 0.53 to 0.58 (Yao et al., 2017). On the other hand, the agricultural water consumption estimated in 2017 accounted for more than 62% of China's total annual water consumption, announcing limited potential for continued application of high-quality irrigation water to meet food demand. Therefore, the development and utilization of unconventional water resources has been on the rise (Cui et al., 2019). This includes reclaimed wastewater, saline water, and rainwater collection. By the end of 2017, China's unconventional water resources projects provided for 1.17×10^{10} m^3 of additional water, accounting for 1.93% of the country's total water supply. Wastewater can be used in the reclamation of severely saline coastal soils and promote plant growth to some extent due to its nutrient content (Li et al., 2019). The annual exploitable saline water resource (with salinity of 2–5 g L^{-1}) is 1.3×10^{10} m^3, which is widely distributed in Northern China. Through field experiments and numerical simulations, recommendations have been developed to apply the saline water using a variety of ways, such as by direct irrigation, rotation irrigation, and blended irrigation. As an example, fresh water is used after sowing, alternated with saline water in the flowering stage of cotton (Sun et al., 2014). Alternatively, the low salinity water can be applied for biological production, and high salinity water can be used for landscaping (Zhang et al., 2019).

In Northwest China, crop growth is totally dependent on irrigation. The key points for soil management in this region are water-saving irrigation and groundwater table control. Drip irrigation was introduced in Xinjiang province in 1996 and was subsequently used in conjunction with mulching technology by covering the soil with plastic film. The use of mulched drip irrigation can simultaneously raise soil temperature while limiting salt accumulation near the soil surface (Qin et al., 2016), and has been widely applied in the Xinjiang, Ningxia, and Inner Mongolia provinces. Development and implementation of mulched drip irrigation has promoted crops production, whereas salt usually accumulates along the wetting fronts and the soil surface between the film rows (Wang et al., 2019). To ensure seedling emergence rate, flood irrigation is used in the fallow period to leach soil salts out of the rooting zone (Wang et al., 2014), which should be supported with sufficient drainage system.

In coastal regions, the improvement of salt-affected soil is mainly through building dikes, raising fields, perfecting river drainage systems, separating field irrigation and drainage channels, sowing rice, and adopting the "raised field planting-shallow pond fishery" system. As an additional water resource, sea ice in coastal areas can be transformed into low salinity water by gravity and centrifugal desalination technologies and has been successfully used for irrigation and aquaculture (Shi et al., 2010).

15.3.3 Future outlook

Although irrigation technologies are fastly increased, there is still huge amount of water that is inefficiently applied, and thus improved and more efficient water application and distribution methods are urgently needed. More than 80% of the irrigation facilities in the medium to large scale irrigation districts have been continuously operated for over 30 years in China. Many of them were not properly maintained, resulting in low water use efficiency (Zhu et al., 2013). In addition, drought has become more severe in Northern China due to global warming and precipitation decrease, which further limits freshwater resources.

Improved irrigation and drainage methodologies and more efficient water resources management practices are still the key measures for mitigation of salt affected soils. Breeding drought and salt tolerant crops, adjusting planting dates based on temporal climate change effects, and adjusting crop distribution and structure would also help to reduce water consumption and promote sustainable agriculture (Zhu et al., 2013).

Coastal zones (e.g. the Yangtze River Estuary, the Yellow River Delta, and the coastal regions of Jiangsu) are important development area in China. Coastal salt marshes are vulnerable ecosystems. Therefore, the reclamation of coastal zone should pay attention to the potential environmental risks, such as secondary soil salinization, eutrophication of offshore waters, and accumulation of heavy metals and pollutants in soils (Li et al., 2014).

Due to the extreme shortage of fresh water, unconventional water resources should be given special attention in the North China Plain. However, there is increasing concern about food safety and environmental risk. For example, reclaimed wastewater for irrigation can increase salinity and NO_3^- concentration in shallow groundwater (Lyu et al., 2019), and long-term sewage irrigation can increase heavy metal concentrations in soils and vegetables (Meng et al., 2016b). Therefore, programs for rational water utilization, long-term monitoring, and evaluation of use of unconventional water resources need to be further strengthened.

As the water resources of Nenjiang River and Songhua River are relatively abundant, Northeast China has the advantage of using river water to improve its saline-alkali lands. The area of saline-alkali wasteland that can be reclaimed in Songnen Plain is more than 1.3 Mha. Conservation tillage and agro-animal husbandry agroecosystem should be considered for the sustainable utilization of saline-alkali soils in this region.

In northwest China, mulched drip irrigation should be combined with salt leaching technologies and adequate drainage systems, to ensure agricultural sustainability for this region. In specific areas with poor drainage conditions, land fallowing should be considered as a way to accumulate and store salts.

Climate change continues to be relevant in China's agriculture. Both carbon dioxide and nitrous oxide are major greenhouse gases and their emissions may be significantly affected by soil salinity and moisture conditions (Maucieri et al., 2017; Zhang et al., 2018). The change of land-use to agriculture is likely to increase soil respiration (Mahowald et al., 2016; Yang et al., 2019). It may also influence salt transport via catchment runoff, aggravating the risk of downstream soil salinization. Therefore, the scale dependent relationship between land use and the risk of soil salinization needs to be quantified (Li et al., 2014). In addition, soil salinity can enhance heavy metal mobility (Acosta et al., 2011; Zhao et al., 2013). Another consequence of increased soil salinity is the potential for enhanced leaching of nitrate into the groundwater, for example by flood irrigation in the fallow period (Feng et al., 2005). It is suggested to specifically pay attention to the eco-environmental effects of salt-affected soils, in addition to impacts on agricultural production.

15.4 Euphrates and Tigris Basin

15.4.1 Historical background

Both the Tigris and the Euphrates are transboundary rivers, originating in Turkey (Fig. 25). Before their confluence at Lake Hammar, the Euphrates flows for 1000 km and the Tigris for about 1300 km within Iraq. The area of the Tigris River Basin in Iraq is 253,000 km^2, which is 54% of the total river basin area. Its average annual runoff is estimated at 21 Bm^3 as it enters Iraq. All the Tigris tributaries are on the east bank. The average yearly flow of the Euphrates is estimated at 30 Bm^3, ranging between 10 and 40 Bm^3 (Al-Layla, 1978). Unlike the Tigris, the Euphrates receives no

Fig. 25 Map of Iraq with location of Euphrates and Tigris rivers.

tributaries during its passage in Iraq. About 10 Bm^3/year are drained into the Hawr al-Hammar, a salty and swampy lake at the confluence of the Euphrates and Tigris rivers. Much of the lake was drained in the early 90s and was reflooded during the Iraqi war. Further downstream, the lake drains into the Shatt Al-Arab river that flows into the Persian Gulf. The Karun River from Iran merges with the Shatt Al-Arab, releasing about 24.7 Bm^3 of fresh water before it drops into the Persian Gulf. The major part of the river flow occurs during February through June on the Tigris River and from March through July on the Euphrates River. On the Tigris, the flow during this period is 60–80% of the total annual flow and on the Euphrates 45–80% (FAO, 2000). During the low flow period (July–September), the flow does not exceed 10% of the annual amount under normal conditions.

Irrigation in Iraq started 7500 years ago in the land between the Tigris and the Euphrates (*Mesopotamia*), when the Sumerians built the first canal to irrigate wheat and barley. The ancient Babylonian Culture fully exploited the lands of these two rivers. The first dam on the Tigris river (Namrood dam) was built about 3000 years ago but was destroyed during the floods of 623 AD (Al-Layla, 1978). The Sassanian Empire (226–640 AD) built a huge canal network to promote irrigated agriculture in the region which was well maintained by the Arabs subsequently (FAO, 1994). The total cultivated area of Iraq is 6 Mha, of which about 50% in northern Iraq is rain-fed while the other 50% is irrigated. The total irrigated area by surface water is estimated at 3.3 Mha, of which 105,000 ha (3%) is in the Shatt Al-Arab river basin, 2.2 Mha (67%) is in the Tigris river basin and 1.0 Mha (30%) is in the Euphrates river basin. The irrigated area with groundwater is estimated at 220,000 ha, using some 18,000 wells (FAO, 2000). Surface irrigation methods are widely used for irrigating crops. Currently, about 50 Bm^3 of water annually is used for irrigation in Iraq, of which a large proportion is returned to the rivers due to the low water use efficiencies. The crop yields are low with wheat, barley and corn yields estimated at 2100, 1900, and 3159 kg ha^{-1}, respectively (Qureshi and Al-Falahi, 2015).

Due to lack of drainage infrastructure, most irrigated areas have suffered from rising groundwater levels and associated soil salinity problems (Pitman and Narisma, 2004). Salinity problems were first recognized in the southern parts of the country at around 3000 BC and continued to spread to other parts over time (Al-Layla, 1978). The central and southern irrigated areas of the twin-river basin produce more than 70% of the total cereal production (Qureshi et al., 2013), where soil salinity is most prevalent. Out of the total salt-affected irrigated areas, 4% is severely saline, 50% moderately saline and 20% slightly saline. Soil salinization and waterlogging problems are damaging 5% of the cultivated lands annually (USAID, 2004), with this twin menace taking away some 70% of the production potential while lands haven been taken out of production for the remaining 30% (Qureshi et al., 2013). The increasing salinity of the river water has mostly contributed to the high soil salinity levels in the irrigated areas. Specifically, the salinity of Tigres River increases from 0.44 dS/m at the Turkish–Iraqi border to more than 3.0 dS/m in Ammarah province (south of Iraq), whereas the salinity of the Euphrates River rises from 1.0 dS/m at the Syrian–Iraqi border to 4.6 dS/m at the Shatt Al-Arab (Al-Zubaidi, 1992; FAO, 2011). The salinity increase in the Euphrates River is higher than the Tigris River because most of the drainage water is discharged into the Euphrates River. In the southern coastal areas, the intrusion of sea water into the irrigated lands further compounded the salinity problems (Wu et al., 2013). Information on the extent and characterization of saline soils in Iraq is minimal and scattered. However, the available literature does provide insight into the extent and characteristics of salt-affected soils (Al-Jeboory, 1987; Al-Layla, 1978; Al-Taie, 1970; Al-Zubaidi, 1992; Dieleman, 1963; Wu et al., 2013). The results of an extensive soil survey conducted from 1955 to 1958 estimated that even if all salts could be leached from the upper few meters of the soil, only 20% of the Mesopotamian plain would be highly productive, 40% would be medium productive, and 40% would be marginal land (Al-Layla, 1978). The estimates of 1970 (Fig. 26) reveal that about 20–30% of the cultivated area is affected by salinity of various levels, resulting in yield reductions of up to 20–50% (Al-Layla, 1978).

15.4.2 Current progress

Salinity has always been a major issue in Iraq, and it was recognized as a cause of low crop yields already some 3800 years ago. Historically, soil salinity has increased from the north to the south of the basin, primarily because of the increasing salinity of the two main river waters. In 1970, half of the irrigated areas in central and southern Iraq were already degraded (FAO, 1994), mostly because of the absence of drainage facilities.

The need for drainage was first realized in the first quarter of the last century when a drainage and salinity investigation was conducted in 1927 (Qureshi and Al-Falahi, 2015). This led to the execution of a few drainage projects in the second half of the last century. However, these were limited to the excavation of a main and some lateral collector drains, without

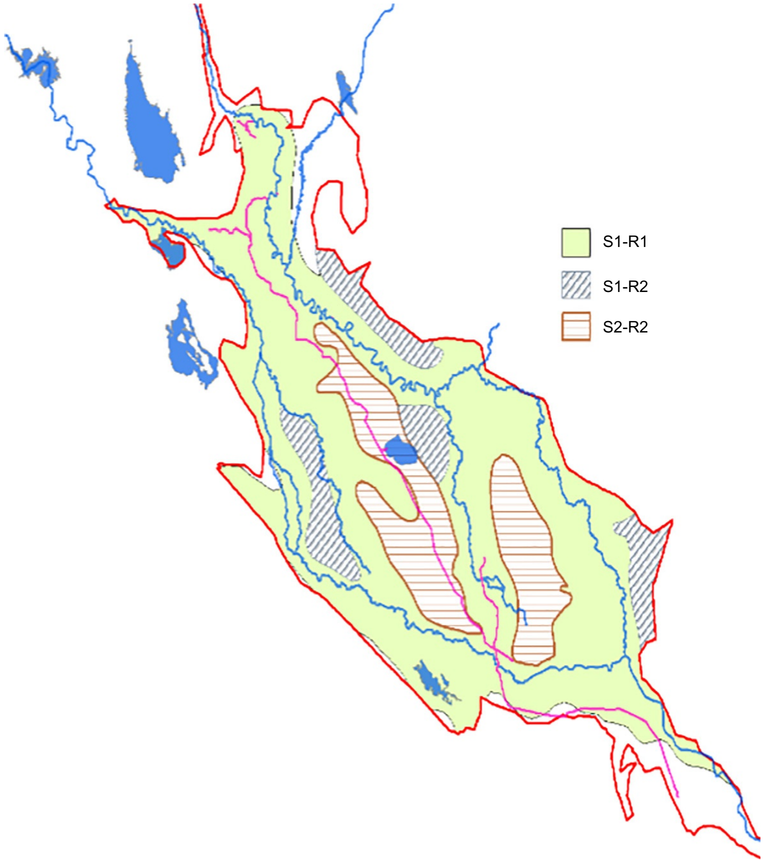

Fig. 26 Salinity in Mesopotamian plain. S1 indicates soil salinity between 4 and 15 dS/m; S2 indicates soil salinity greater than 15 dS/m; R1 indicates soil salinity increase by 2–3 dS/m/year; R2 indicates soil salinity increase by 3–5 dS/m/year.

installation of field drains because of lack of finances. Drainage waters were pumped and discharged into the rivers thereby increasing river salinity. This approach was only a partial solution, as soil salinity continued to grow. Most of these drainage projects are now 40–50 years old, with many abandoned due to poor maintenance. Some additional tile drain projects were executed later. However, they soon became dysfunctional due to the deposition of silt and gypsum precipitates into the drainage pipes. Drainage systems further deteriorated due to the Iraq war of the last 2 decades, further aggravating salinity issues and finding solutions even more critical (Qureshi et al., 2013).

Reclamation of salt-affected soils has largely been done through the lowering of the groundwater table. Historically, groundwater tables were controlled using wheat-fallow system or by limited irrigation water application. This practice, however, was abandoned because of the need for land development and agricultural intensification. Instead, a land rehabilitation program was initiated in 1978 by concrete lining of irrigation canals and installation of field and collector drains, thus reclaiming a total area of 700,000 ha by 1989 at a cost of around US$ 2000/ha (FAO, 1994). Nevertheless, drainage waters continued to be discharged into Euphrates and Tigris rivers deteriorating their water quality and increasing soil salinity.

In 1953, construction began on the Main Outfall Drain (MOD), also known as the Third River, starting NW of Baghdad, ending in Basrah where it delivers drainage water to Shatt Al-Arab and eventually into the Persian Gulf. It was designed for a carrying capacity of 6.9 Bm^3, although total annual flows currently are not more than 3.8 Bm^3 (Licollinet and Cattarossi, 2015). The remaining capacity was designed assuming that the East Tigris Drain (ETD) and the Razzaza Drainage System will be routed to the MOD, when fully developed by 2020. It is estimated that MOD will provide 4.6 Bm^3 per year for drainage water reuse (Licollinet and Cattarossi, 2015).

15.4.3 Future outlook

The sustainability of irrigated agriculture in Iraq is vital to ensure future security for the rising population, which is expected to reach 50 million by 2030 from the present value of 40 million (FAO, 2012). This requires major reforms regarding water usage and allocation, disposal, and reuse of drainage water. The productivity of the irrigation sector largely depends on the management of its drainage waters and soil salinization. Due to the inherent complexity of salinity issues, a multi-dimensional approach that considers biophysical and environmental conditions, as well as livelihood aspects of the people will need to be adopted. Realizing this challenge, Iraq has developed a "Strategy for water and Land Resources in Iraq (SWLRI)" in 2014 (Licollinet and Cattarossi, 2015). It has identified projects for optimizing land and water resources, for the primary purpose to address the need for food and energy security and to sustain the environment. SWLRI has proposed extensive reclamation measures, including subsurface on-farm drainage in all irrigated lands in the center and south of Iraq.

SWLRI also emphasize the importance of the re-use of drainage waters for irrigation to help meet Iraq's 2035 development goals. Therefore, SWLRI strategy must be implemented in both letter and spirit.

Despite widespread soil salinization, no comprehensive monitoring network to record spatial and temporal changes in soil and water salinity is available in Iraq and is badly needed. In addition, restoring existing drainage systems that were destroyed during the Iraq war should be given high priority.

Under the current geo-political circumstances, large-scale investments for the rehabilitation of existing drainage systems and the installation of new drainage systems will be a huge challenge. Therefore, alternate approaches such as irrigation management to control percolation losses and reusing drainage water for salt-tolerant crops need to be encouraged (Qureshi et al., 2013). Drainage water can also be used for the promotion of aquaculture especially in those areas which are not suitable for conventional agricultural production systems.

15.5 India, Indo-Gangetic Basin

15.5.1 Historical development

The salt affected soils form an important ecological entity in the Indus-Ganges Basin (IGB). The IGB covers an area of about 225 Mha, and include all of Nepal, large parts of India, Pakistan, Bangladesh and small areas of China and Afghanistan. The history of salt-affected soils in this basin dates to 1500 BC in Indus Valley when Aryans started crop cultivation using tank and well irrigation and distinguished lands as *urvara* (fertile) and *anurvara* (infertile). They also made efforts to understand the cause of *anurvara* and thus designated salt-affected soils as *usara*. But salinity was recognized as a potential threat to agriculture only during the middle of the 19th century (Singh, 2005). After firmly establishing themselves in India, the British spread irrigation as a revenue-earning proposition and constructed several canal networks. Soon after, the occurrence of salt-affected soils and their further spread attracted the attention of the government. Early complaints from the Munak village in Karnal, near to the Yamuna canal, in 1855 and in 1876 by an Indigo planter from Sikandra Rao village in Aligarh district, about the deterioration of his land after the introduction of Ganga Canal, led to formation of the "*Reh*" Committee to investigate the causes of soil deterioration in canal-irrigated areas. Continued research efforts were conducted in various parts of the Sindh and Punjab states, directed mainly on distinction between saline and alkali soils using salt crust color and their physical traits like hardness and permeability. Testing of gypsum use for reclaiming alkali soils started in the beginning of the 20th century.

Due to the rapid commissioning of several major and medium irrigation projects, many areas became waterlogged and saline during the post-independence period. As a result, the ICAR Soil Salinity Research Institute was established in 1969 at Karnal to conduct research and develop technologies for salt reclamation and management. In addition, research at several state agricultural universities and other research centers through the All Indian Coordinated Research Project on "Management of Salt-affected Soils and Use of Saline Water in Agriculture" have led to improved understanding and development of techniques using multi-disciplinary approaches across the biological, agricultural and engineering sciences. Since then, these technologies have been adopted and upscaled through departments like State Land Reclamation and Development Boards, Department of Agriculture & Cooperation (DAC), State Agriculture and Irrigation Departments and NGOs (non-government organizations).

15.5.2 Current progress

The total salt-affected area in the country is currently about 6.7 Mha, with 2.7 Mha in the IGB (Mondal et al., 2011; Fig. 27; Table 4). Among these, Uttar Pradesh has the maximum land area of 1.37 Mha followed by West Bengal (0.44 Mha) and Rajasthan (0.37 Mha).

Successful agricultural practices that have restored the extent of salt affected soils are:

Reviving alkali lands: Gypsum additions of 10–15 Mg/ha, equivalent to 50% of gypsum requirements for the surface 0.15 m soil were adequate to reclaim alkali soils (Abrol et al., 1988; Gupta and Abrol, 1990). A surge in groundwater irrigation and a shift to paddy-wheat cropping systems further helped through de-sodification and leaching of the reaction products. Several other ameliorative additives like pyrites, sulfuric acid, sulfur and others were not comparable with gypsum in terms of efficiency and costs. Agronomic packages included increasing fertilizer N by 25%, applying Zn, increase irrigation frequency for upland crops, integrated nutrient management, and green manuring. With development of gypsum-based technologies and their implementation at the farm scale, close to 2.1 Mha of alkali soils were rehabilitated (Mandal et al., 2018). Some would argue that the alkali land reclamation efforts in states of Punjab (0.8 Mha), Haryana (0.35 Mha) and Uttar Pradesh (0.85 Mha) created its own "mini revolution"

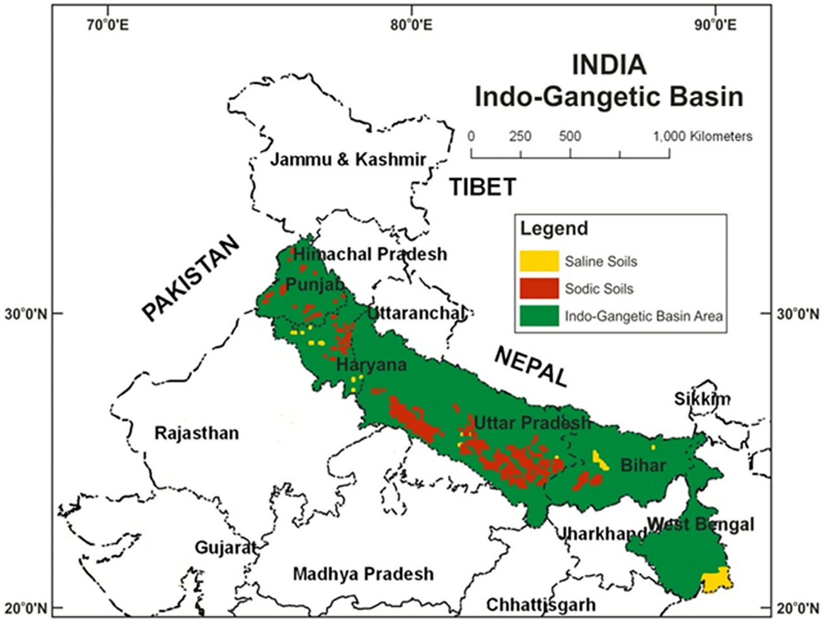

Fig. 27 Distribution of salt affected soils in India IG Basin. *Adapted from Mondal, A.K., Obi-Reddy, G.P., Ravisankar, T., 2011. Digital database of salt affected soils in India using Geographic Information System. J. Soil Salinity Water Qual. 3, 16–29. Map of Indus-Ganges Basin (IGB).*

Table 4 Current extent of salt affected soils and area reclaimed in states of the IG Basin, India.

	Extent (10^3 ha)			Reclaimed area (10^3 ha)	
State	**Alkali soils**	**Saline soils**	**Total**	**Alkali soils**	**Saline soils**
Punjab	152	–	152	797	4
Haryana	183	49	232	352	11
Uttar Pradesh	1347	22	1369	851	–
Bihar	106	47	153	2	–
West Bengal	–	441	441	–	–
Rajasthan	179	196	375	22	17
All India	3770	2957	6727	2071	70

as part of the India's Green Revolution, as it now contributes about 17 million ton annually of additional food grain, in addition to other environmental benefits.

Salinity control of waterlogged soils: Pilot projects on developing guidelines for surface and subsurface drainage (SSD), in conjunction with groundwater pumping have been effective in controlling waterlogging and salinity (Kamra, 2015). If followed, these guidelines facilitated the growing of crops within 2–3 years after implementation of SSD for land not suitable for agriculture before. For those SSD projects cropping intensity increased by 25 to more than 100% with crop yields increasing by 45% (paddy rice), 111% (wheat) and 215% (cotton). However, high capital costs, issues on operation and maintenance and safe drainage water disposal has limited further expansion. Use of farm ponds and land-shaping technique for paddy-cum-fish cultivation such as deep furrow and high ridge have been shown to be viable technologies to address the twin problem of drainage congestion and salinity in the degraded coastal lands (Bandyopadhyay et al., 2009).

Sustaining irrigation with saline waters: About 32–84% of groundwater in the north-western states of IGB are rated as either saline or alkali (Minhas and Gupta, 1992). Long-term field experiments have identified key parameters that control plant responses to soil and groundwater salinity, with optimal conjunctive use irrigation water application practices (Minhas and Samra, 2003). Similarly, irrigation practices have been standardized for sustainable use of alkali groundwaters, including chemical amelioration of soils and irrigation waters, water quality driven conjunctive uses, mobilizing in-situ calcite, use of salt tolerant crops, and by other specialized tillage, fertilizer use and irrigation practices. Based upon the experiences on their use for different agro-ecological zones, highly conservative water quality standards have been replaced with site-specific guidelines.

Improved plant adaptations: Recent results have shown that breeding of high producing and salt tolerant crop varieties should focus on trait-based crop varieties, e.g. CS-52 of mustard, CSR-30 of rice and KRL-219 of wheat. Very successful has been the breeding of the rice variety Basmati CSR-30, which is now grown on about 1.96 Mha of salt affected soils over 15 years (Singh et al., 2021). As an additional benefit, the planting of salt tolerant rice varieties reduced half of the gypsum required for reclamation of alkali soils.

Alternate land uses: For saline soils that cannot support agriculture, other viable landuses are explored, such as growing salt tolerant trees, grasses and other halophytes (Dagar and Minhas, 2016). Several salt tolerant trees have been identified for reforestation of alkali/saline lands, such as *Prosopis julilora, Acacia nilotica, Casurina equisetifolia*. Specific planting techniques, irrigation methods for saline-water logged soils, and post-planting management practices have been developed that assist in the establishment of tree plantations on these salt-affected soils. Some grasses like *Leptochloa fusca* are not only well adapted to highly alkali conditions, these assist in bioremediation of these soils through their extensive and deep root systems.

15.5.3 Future outlook

The IGB is among the most populated river basins in the world with a current population of around 1 billion, and with more than 50% of its area cultivated, largely through extensive irrigation practices by surface water diverted through canals, as well as through groundwater pumping (Cai et al., 2010). The basin at large has witnessed a boom in aquifer withdrawals and currently about two-third of irrigated land is groundwater-dependent. Its intensive use for irrigating crops like rice-wheat and sugarcane in north-western states has led to the lowering of water levels at such an alarming rate that is now endangering their potential for future use. It has further generated multiple negative externalities, including salinity, contamination with arsenic and fluoride, stream depletion, or land subsidence. These are now pushing the IGB toward unsustainable agriculture, raising risks for the farmers, and promoting extreme inequity with respect to water availability.

Despite the large research and developmental efforts on salt-affected soils, knowledge gaps remain, and new research and tools should provide resilience of agriculture. These are discussed below.

Alkali soils: The pace of alkali soil reclamation accelerated due to access to good quality ground water and a shift to paddy-wheat systems. Simultaneously with the rise in land productivity, organic-C inputs through rhizo-depositions, root-biomass, and stubbles further stimulated bio-reclamation processes for otherwise non-cultivated lands. However, further research is needed to better understand the environmental consequences of gypsum addition and its chemistry, specifically through the development and application of hydrochemistry models (see Section 3.1), enabling prediction of both short- and long-term impacts of gypsum-based technology and fate of reaction products *vis-a-vis* groundwater quality. In addition, fine-textured soils with calcareous layers at shallow depths are difficult to reclaim and therefore require appropriate modifications of existing reclamation practices. Together with the increased demand for gypsum by other non-agricultural sectors, availability and costs make their future application limited, and requires consideration of alternatives sources such as through by-products of thermal plants, the sugar industry, and urban wastes. Their potential use to supplement gypsum will lead to win-win scenarios as these waste products would otherwise involve disposal costs.

Saline-waterlogged soils: Although the SSD technology has been standardized for most projects, this was done when irrigation water supplies were abundant whereas surface water is becoming limited and ground waters are often highly saline. Moreover, many of these lands serve as recharge sites under dryland salinity conditions. It is therefore that refined SSD guidelines need to be developed when water resources are constrained and for dryland salinity conditions. Although water-table management through controlled drainage helps in decreasing irrigation demands and drainage outflows, many anticipate that it will reduce the rate of land reclamation. Leaching plans to reduce salt accumulation by SDD need testing to analyze long-term consequences. Additional research is required to (1) evaluate integration of SSD with groundwater pumping control at the regional level, (2) the effectiveness of plantation forestry in reducing water-logged areas, as well as (3) the use of pumps in concert with SSD to better control water-table depths.

Use of saline waters: Micro-irrigation systems such as drip-fertigation are the most efficient in utilizing saline irrigation water, especially for high value horticulture, but their large-scale evaluation is absent. There is lack of understanding of salinity-sodicity interactions when irrigating with brackish water (Section 13), depending on factors like ion chemistry of irrigation water, clay mineralogy, cropping system and climate. This is needed to analyze impacts of long-term application of these irrigation waters on soil physical and hydrological behavior. Furthermore, the proposed amendment applications of gypsum through specially designed "beds" or that of sulfur through "sulfurous acid generators" requires further research for their cost-effectiveness. Remedial strategies should be evolved for fluorine and arsenic contamination in groundwater which have emerged as major toxicological problems across the IGB. Finally, detailed long-term investigations are needed to assess ways by which conservation agriculture practices can use poor-quality irrigation waters.

Alternative land use: A major role of forestry is usually defined in terms of modifications in salt and water dynamics at the field and catchment scale, thereby aiding in the control of water-tables and salinity. Nevertheless, arguments against plantation forestry practices are emerging because of the long time between planting and harvest, the high land requirements, and inevitable soil salinity build-up, affecting their growth and beneficial water withdrawals. To overcome these constraints, research is needed (1) to evaluate the shifting of plantations in between discharge and recharge areas, (2) toward reforestation with salt tolerant species, and (3) combining plantation forestry with engineering measures in saline discharge areas. Specific halophyte species like Chenopodium and Salicornia have potential for commercial production, but much more research is required to successfully apply biosaline agriculture as an alternative landuse for otherwise non-productive lands.

Looking back, several research and developmental organizations have contributed significantly to the reclamation and management of salt-affected lands. But they have been mostly working in isolation without interdisciplinary efforts. Considering the magnitude and complexity of the salinity problem, a holistic multidisciplinary and networking approach is required using a systems approach to tailor technologies across scales from the field to the district and the whole ecosystem. Moreover, key policy impediments must be addressed for rapid technology dissemination. These include effective involvement of stakeholders at the community level, provision of incentives such as subsidies and cost sharing, and enacting new laws that enforce reclamation requirements for maintenance and operation of SSD. Web-based platforms should be created to interface among policy planners, researchers, state agricultural departments and development boards, farmer's associations, self-help groups and NGO's. These will serve principally to ensure multi-stakeholder input when making decisions on the development and implementation of technologies, thereby accelerating the reclamation rate of saline-sodic soils.

15.6 Israel

15.6.1 Historical development

The State of Israel was established in 1948 and Israel's recent history has been heavily influenced by the 1950 Law of Return, granting Jewish people the right to immigrate to and settle in the country. Israel's population has increased from about 650,000 in 1947 to 9 million today. Israel's climate is arid to semi-arid, with two-thirds of its area being desert. The average annual precipitation ranges from 25 mm in the Negev Desert, to about 300 mm in the coastal plains to 800 mm in the Upper Galilee region, occurring almost exclusively in the winter, between November and March.

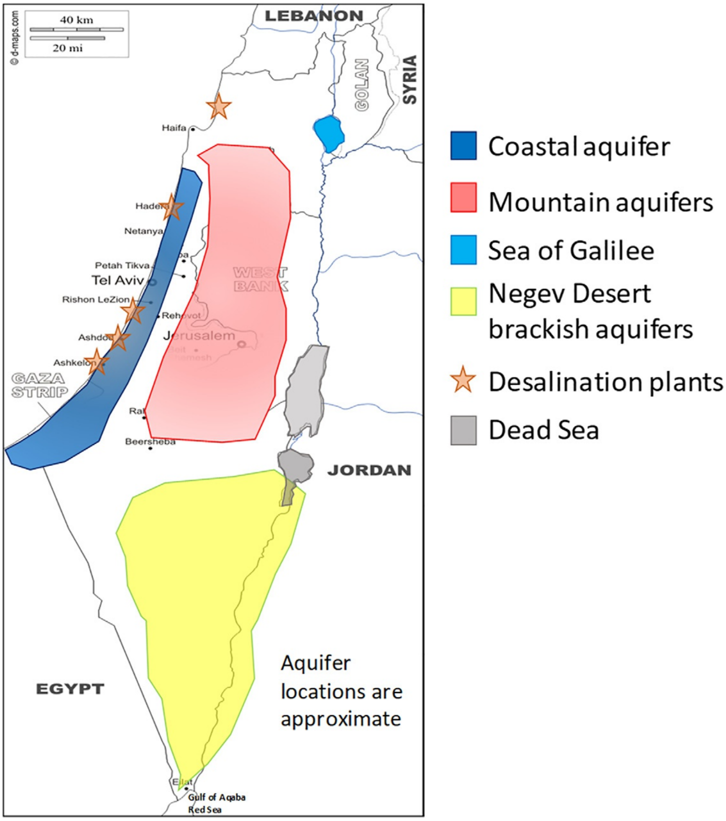

Fig. 28 Water map of Israel.

About two-thirds of the country's fresh water supply has traditionally come from groundwater pumped from two major aquifers (Western Mountain and Coastal Aquifers), with the other one-third coming from the Sea of Galilee, fed largely from the upper Jordan river (Fig. 28). To ensure equitable distribution and efficient use of the available water resources, already in 1949 Israel enacted a legislative code that made water a public property that is under State control, with water licensing issued by its Water Commission. In order to supply water to Israel's south, the National Water Carrier (NWC) was built in the 1960s. About 50–55% of total consumed water is used for irrigation. However, to meet domestic and industrial freshwater demands, the fraction of natural freshwater used for irrigated agriculture has decreased from about two-thirds (90s) to currently about one-third. To supplement irrigation water needs, some 60%

of the irrigation water supply now comes from treated wastewater and brackish groundwater. Finally, to ensure an adequate future water supply, Israel has embarked on building large-scale seawater desalination plants.

In Israel, interest in soils and salts comes mostly from water scarcity and subsequent irrigation-induced salinity. The Israeli experience in salinity management of soils involves three unique intersecting aspects making the lessons learned of interest globally. The three aspects are: (1) early and full adoption of highly efficient irrigation technologies including drip irrigation and knowledge driven scheduling, (2) considerable amounts of relatively high salinity water from brackish groundwater and recycled municipal wastewater utilized for irrigation, and (3) the recent large-scale move to desalination of seawater to insure national municipal water security that has led to reduction of salts in the water system, especially in recycled wastewater.

The lessons learned from Israel's historical irrigation water policies and practices have been reviewed and discussed by Assouline et al. (2015), Tal (2016), Siegel (2015), and Raveh and Ben-Gal (2016, 2018). Here we summarize in terms of salinity and soils.

15.6.2 Current progress

Israel is a small country with a relatively solid economic base, but isolated due to geo-political reality, and unique as a water-scarce country with successful agricultural development. Water consumption from all sources and for all sectors in Israel increased tenfold from 230 MCM (million cubic meters) in 1948 to 2200 MCM in 2018 (Israel Water Authority, 2019). It is estimated that only 55–65% of the present amount of the country's water needs is renewed annually in its natural surface and groundwater resources. The remaining water supplied comes from groundwater mining, allocation of reclaimed wastewater, or by seawater desalination. While per capita consumption in the domestic and industrial sectors has remained essentially the same during these last decades, per capita water available for agricultural uses is less than half today than it was in the 1960s. Despite the reduction in water allocation, agricultural production per capita today is more than 150% of that produced 40 years ago (Ben-Gal, 2011; Tal, 2016). The success can be credited to several central driving principles including: (i) intensification and modernization of agricultural systems; (ii) development and adoption of efficient water application technologies and methodologies; and (iii) establishment of reliable water sources for irrigation.

Intensification and modernization of agriculture were accomplished in Israel by strong research and development programs, knowledge transfer to farmers by means of a solid extension service, and strong government economic support of national strategies. Drip irrigation was developed in Israel where this inherently efficient technology is used at rates higher than anywhere else in the world. Technologies and practices promoting water efficiency have further been encouraged by national water pricing and allocation strategies (Tal, 2006). Utilization of low-quality water has been encouraged (or compensated) through a water for irrigation pricing structure where cost to farmers goes down as irrigation water salinity increases.

The third principle stimulating success, a reliable source of water for irrigation, has been more difficult to accomplish. The NWC has historically conveyed water from the Sea of Galilee in the north to the south of Israel, seasonally mixing it on the way with various ground and floodwater sources. Average EC of the NWC water has historically ranged from 0.8 to 1.1 dS/m. Freshwater use in agriculture dropped from 950 MCM in 1998 to around 490 MCM today. Total water to agriculture has been maintained via the utilization of brackish and recycled water (Fig. 29).

Israel's agriculture directly uses some 80 MCM of brackish groundwater with EC of more than 2 dS/m for irrigation, mainly in arid regions including along the Jordan Valley and the Arava and the Negev Highlands. Wastewater recycling has become a central component of Israel's water management strategy. A master plan presented in 1956 envisioned the ultimate recycling of 150 MCM of sewage, all of which would go to agriculture.

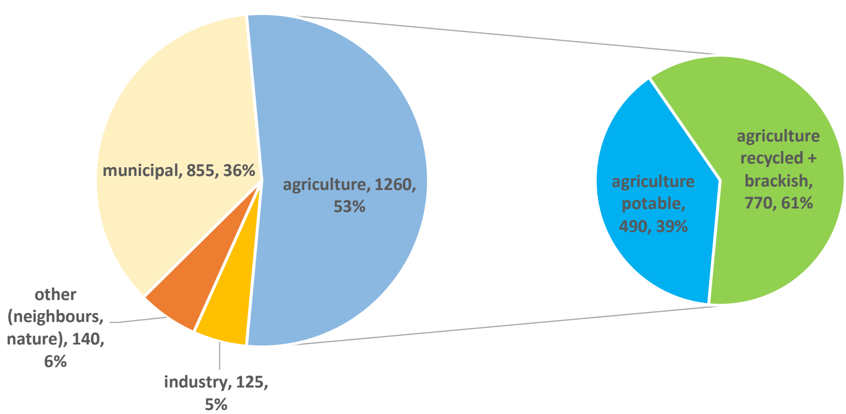

Fig. 29 Average 2015–2018 annual water use (MCM, %) in Israel by sector and source (Israel Water Authority, 2019).

Today four times that level is recycled, representing around 85% of all domestic wastewater produced. Treated effluents today contribute roughly 25–30% of Israel's total water supply and, depending on annual rainfall, up to 40% of the irrigation supply for agriculture. Salinity of recycled wastewater, depending on its type and origin, can range dramatically, but no matter what, salinity increases as the wastewater stream advances. In Israel, municipal recycled wastewater typically ranges from EC of ~1 to more than 3 dS/m (Tarchitzky et al., 2006).

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Unfortunately, due to the high concentrations of salts in the irrigation water, Israel's strategy for agricultural success seems to be not sustainable. Long-term application of salts to agricultural soils in a region where seasonal rainfall is low, unpredictable, and often insufficient to systematically mobilize and remove problematic salts, must include application of water designated to leach the accumulating salts out of the root zone (Russo et al., 2009). The water applied for leaching and leaving the root zone contains not only the salts that must be leached, but also various other contaminants, found naturally in the water, added in agricultural processes (fertilizers, pesticides and herbicides), or mobilized from soil and subsoil (Ben-Gal, 2011; Ben-Gal et al., 2008, 2013).

An example of problematic sustainability stemming from policy and practice of irrigation with water high in salts is found in the Arava desert where brackish groundwater is used to irrigate green and nethouse protected vegetables. It is estimated that irrigation to leach salts in the region can be beneficial to yields and profits at rates as high as twice those necessary to satisfy crop evapotranspiration requirements (Ben-Gal et al., 2008, 2009a,b).

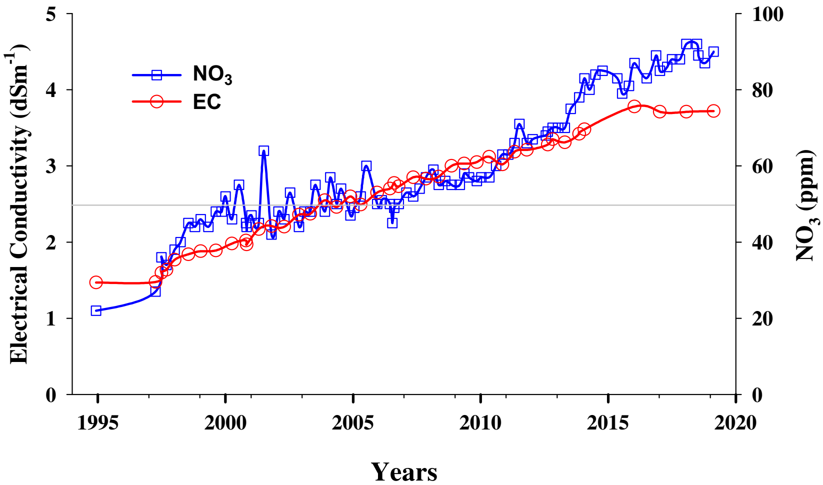

Fig. 30 Electrical conductivity and nitrates (NO_3) in groundwater serving for irrigation in the Arava Valley (Hazeva) since 1995. Data provided by Dr. Effi Tripler, Central Arava R&D. The horizontal gray line indicates the allowable NO_3 concentration in drinking water of 50 ppm according to the WHO (2011).

The most obvious threatening contaminant and best indicator of pollution accompanying the leaching practices is nitrates. Nitrates, as well as salinity in general, have risen from less than 20 to more than 90 ppm in wells of groundwater downstream from local areas of intense vegetable cultivation (Fig. 30).

15.6.3 Future outlook

Regarding continued use of effluents or other salt-rich sources for irrigation water, additional indications of problems are found. These include the long-term increases in sodium adsorption ratio (SAR) and exchangeable sodium percentage (ESP) in soils (Assouline et al., 2016; Assouline and Narkis, 2011, 2013; Erel et al., 2019; Raveh and Ben-Gal, 2016; Segal et al., 2011), affecting soil structure and water infiltrability, a trend of increasing sodium and chloride found in irrigated plant tissues, and the tendency for Israeli fresh produce to have higher than international standards of sodium (Raveh and Ben-Gal, 2016). In addition, there are increasing concerns regarding possible yet undiscovered detrimental long-term repercussions due to trace level (particularly persistent organic) contaminants in agricultural systems and the food chain (Goldstein et al., 2014).

Despite all this, the latest responses of Israel to insure reliable municipal water supply to its growing population may coincidentally provide opportunity for a more sustainable solution for agriculture. Starting in 2007, Israel has added desalinated seawater to its water distribution stream. Desalination currently provides around 25% of Israel's total water supply, as more than 40% of the country's municipal water, often incidentally bringing very good quality water to agricultural areas and consistently reducing the salinity of recycled wastewater (Assouline et al., 2015; Raveh and Ben-Gal, 2018; Yermiyahu et al., 2007). Planned large-scale desalination in The Red Sea, as part of a project to stabilize Dead Sea water level by transporting the brine, would bring a significant amount of good quality water to replace current irrigation with brackish water to Israel. The Red-Dead conduit project, if funded and built, would additionally promote regional strategies for treating water scarcity and salinity together with Jordan and the Palestinian Authority (Aggestam and Sundell, 2016; Hussein, 2017).

The turn to desalination as a strategy for water security is a positive opportunity to reverse the maybe dangerous and apparently non-sustainable trends consequential to irrigation with water containing high concentrations of salts (Assouline et al., 2015, 2020; Raveh and Ben-Gal, 2018; Tal, 2016). Treatment of brackish groundwater and of water specifically destined for irrigation may in the future benefit from technologies that, contrary to the current popular reverse osmosis based desalination, will selectively remove problematic monovalent ions while leaving agricultural desirable bivalent ions like calcium and magnesium (Cohen et al., 2018).

Israel is projecting that by 2050, two-third of its water supplies will come from treated effluent, desalinized or brackish water. Sustainable, healthy, economical, irrigated agriculture in Israel and other semi-arid and arid regions should be possible if the salts are taken out before application, instead of being allowed to negatively affect soils, crops, produce, and the environment (Raveh and Ben-Gal, 2018; Silber et al., 2015).

15.7 Latin America

15.7.1 Historical development

Latin America is a cultural entity extending from the Rio Grande in North America, to Tierra del Fuego, at the southernmost tip of South America. It is a vast area, spanning for 19.2 million km^2 and home for approximately 650 million inhabitants, including countries with diverse availability of natural resources and economies. The Latin languages Spanish and Portuguese are the main tongues in the region, although English, French and Dutch are also spoken. This extensive territory features a huge variety of climates and soils, which lead to a great variability of ecosystems, and support an array of agricultural, livestock and forestry activities. Tropical to temperate/cold crops are cultivated in it. Globally, the region is a net food exporter of a variety of primary products like grains (soybean, maize, wheat, and others), coffee, vegetables, and fruits, etc., and industrialized derivatives as sugar, vegetable oil, and wine.

Latin America ranks third in the world with land surface area of salt affected soils. Unfortunately, estimations of the extension and distribution of salt-affected soils in Latin America are neither updated nor very precise, and partially based on expert judgment. Soil salinity and alkalinity are found in diverse environments throughout the region and include both primary and secondary salinity. Some estimations indicate that an area of about 7×10^5 km^2 is affected by salinity and 6×10^5 km^2 by sodicity, for a total salinized area of 1.3×10^6 km^2, however, other area estimations suggest a total area of 1.7×10^6 km^2. The total irrigated area is around 25–30 Mha. It is estimated that 25–50% of that area is affected by human-induced secondary salinization and sodification, adding approximately 4–5 Mha of recent human-induced salinization processes in non-irrigated areas (Taleisnik and Lavado, 2020).

Primary salinization processes occur in the humid and sub-humid regions where natural saline, but mainly sodic soils are found. They are found in large plains with shallow saline or sodic ground-waters like the Chaco-Pampas regions, which are among the flattest sedimentary plains of the planet and a major grain exporter of the continent. Natural vegetation in the northern part of this plain is composed of xerophytic forests (Chaco and Espinal subregions). The plain hosts shallow water tables, which, combined with negative climatic water balances, makes it prone to salt accumulation both in its deep sediments and in the surface of its low landscape sectors (Contreras et al., 2013). Grasslands dominate in the South (Pampas) of the sedimentary plain, showing some areas with salt affected soils. Besides those areas also few coastal and swampy areas with saline-acid soils are found as well as large internal saltmarshes elsewhere, among them the Pantanal in southern Brazil, one of the larger wetlands of the world (Freitas et al., 2019).

Overview of anthropic salinity problems—In general terms, no country in Latin America is completely free from salinization, and we will focus our analysis on human-induced salinization, mainly caused by irrigation, though not exclusively. Most of the secondary salinization occurs in irrigated areas in arid and semi-arid zones, where intensive agriculture is practiced. This process is mainly due to non-efficient water management, poor drainage conditions, and low irrigation water quality.

Irrigation mode and extension as well as type of crops vary considerably within the region. Fruit and vegetable production are mostly irrigated. In some area's extensive crops such as sugarcane, rice, cotton, maize, and wheat are partially grown under irrigation, using modern technologies. In others, irrigated areas are populated by small holder farmers who generate most of the locally consumed food. The ratio between these two ways of production varies among countries and regions within them but does not to appear to be related to soil salinization processes.

Arid and semi-arid areas under full irrigation are common in Mexico, Peru, Chile, and Argentina. While irrigated production value exceeds that of rainfed agriculture in Mexico, Chile and Peru, the reverse occurs in Argentina. In most cases furrow and flooding irrigation systems are used, but sprinkler, micro-sprinkler or drip irrigation systems are being increasingly adopted. Waters from various origins are used, from surface to ground waters, and quality ranges from good to bad. Irrigation in non-graded and uneven lands has led to low water use efficiencies and rising groundwater levels. Low efficiencies are additionally caused by non-lined water distribution ditches and poor drainage conditions. Besides the typical effects of salts (due to presence of sodium, chlorides, carbonates), boron is an additional problem in several irrigation districts (Pla Sentís in Taleisnik and Lavado, 2020).

The Brazilian semiarid region in the northeast of the country is one of the largest semiarid regions of the world. It features tropical climate conditions with variable rainfall associated with high temperatures during much of the year. The region is also characterized by shallow soils, low quality irrigation waters, lack of drainage and often shallow groundwater. Irrigation has improved the economy of the region through diversified cropping practices, stimulation of agroindustry and export of products, but has also led to large areas being degraded by salinization because of poor water quality and deficient or absent drainage schemes. In general, those degraded areas are left fallow so that agricultural production is moved to other areas. The return of vegetation of these deserted areas would initiate a slow reclamation process (Lacerda et al., 2011; Santos et al., 2020).

In semiarid/subhumid zones, as in Colombia, Venezuela, Cuba, the Dominican Republic and in some of the other countries in the region, similar salinization processes have occurred where sugarcane, rice and other tropical crops are irrigated with waters of varied quality and drainage is poor or nonexistent. Further south, in the temperate area of Pampas region of Argentina, field crops are usually grown under rainfed conditions but are exposed to occasional drought events. Supplementary sprinkler irrigation allows farmers to increase and stabilize grain yields. Exchangeable sodium has increased sharply but no consistent impacts on soil physical degradation have been detected (Costa and Aparicio, 2015).

Human-induced salinization has also developed in the above-mentioned Chaco-Pampas plain, mainly promoted by land use and land cover changes, by cropping or overgrazing. In the Chaco and Espinal regions in the North of this plain, deforestation and cultivation have altered the hydrologic balance, mainly because cropped areas present lower evapotranspiration rates. The resultant water excess infiltrates and slowly causes the rise of deep groundwater tables that bring salts to the surface, thereby damaging crops and soils. This process of salinization is somewhat like the "dryland salinization" in Australia (Fan et al., 2017; Glatzle et al., 2020). The Southern part of this Chaco-Pampas is mostly devoted to field crops, but alkaline and to a lesser extent saline soils predominate in an area known as the "Flooding Pampa," where livestock production activities prevail. There, intensive cattle grazing removes vegetation and high evapotranspiration causes salts from the water table to reach the soil surface in the summer. Subsequent rains leach the salts, but this man-made process affects the composition of the plant communities (Chaneton and Lavado, 1996). A similar process has been observed in grazed salt wetlands (Di Bella et al., 2015).

15.7.2 Current progress

Research on salt affected soils was very active in the 1960–1980 period, when several countries experienced large agricultural development through investments in large irrigation schemes. Research on soil salinity research at that time was mainly applied and based on results published by the US Laboratory Staff (1954). One consequence of such effort was the organization of regional and international conferences, such as those that took place in 1971 in Colombia and in Venezuela in 1983 (Pla Sentís in Taleisnik and Lavado, 2020). However, advances in research and evaluation of salt-affected soils subsequently faded. In most irrigated areas attention was focused on the engineering aspects of irrigation infra-structures (dams, distribution canals) rather than on the installation of effective drainage systems and adequate preparation of irrigation fields (leveling, irrigation ditches, etc.) at the farm-scale. This has led to problems of drainage, waterlogging, and salinization.

More recently, the development of large and expensive irrigation schemes has diminished, whereas new irrigation developments have been for small local irrigation units, using nearby surface and groundwater resources but most often done without consideration of regional impacts. In some extreme cases, due to competition for alternative uses of scarcely available good-quality water resources, non-treated residual waters of urban and industrial origin are used for irrigation. This is valid for small irrigation units, mainly dedicated to production for local markets, but not for larger irrigation units.

Research on soil, water, and crop management in saline areas in Brazil is concentrated in universities and research organisms in NE Brazil. Approaches include the development of soil and water management strategies, appropriate cropping systems, the sustainable use of brackish waters, cultivation of halophytes and salt-tolerant crops, application of mineral and organic amendments, phytoremediation and plant/microorganism interactions (Andrade et al., 2019; Leal et al., 2019; Miranda et al., 2018). A specific concern is the mitigation of socio-economic impacts of soil salinity in agricultural lands, which translate into loss or reduction of crop yields, profit margins, increased unemployment, and reduction of commercial land value. Technologies are being developed to provide a source of income for impacted smallholder farmers to provide for water and food security. This include the desalination of brackish water and its use in an integrated production system involving reject brine for farm-raised fish and the use of fish-pond water to grow organic salt-tolerant vegetables and forage crops for small ruminants (Antas et al., 2019; Moura et al., 2016).

Argentina has active research on soil, water and crop management and salt tolerance mechanisms. Technologies on salt affected soils of humid/subhumid areas are aimed mainly at increasing biomass productivity without altering soil properties. They include grazing management, afforestation, agro-hydrological management, plant introduction, among others. Salinization in semiarid deforested areas has been studied in the great Chaco area and is focusing on ways to mitigate soil and water quality degradation (Garcia et al., 2018), such as by changing cropping systems. Salt tolerance mechanisms are being studied in plants (Pittaro et al., 2016; Taleisnik et al., 2009) and microorganisms. The use and management of native woody species for degraded and salinized areas is considered (Fernández et al., 2018). The characterization, collection and multiplication of both native and introduced species, and their incorporation into breeding program are prominent activities (Marinoni et al., 2019). Traditional breeding efforts have produced new salt-tolerant forage plant cultivars, such as Epica INTA Peman (https://peman.com.ar/en/products/%C3%A9pica-inta-pem%C3%A1n%C2%AE), and new breeding alternatives have been explored to increase salt tolerance in *Melilotus albus* (Zabala et al., 2018). Research on *Lotus* species for alkaline and sodic soils has contributed to their expansion in the Flooding Pampa (Bordenave et al., 2019). Molecular components of signaling chains and salt tolerance mechanisms have been successfully incorporated into commercial crops, soybean for example (Ribichich et al., 2013).

15.7.3 Future outlook

Research interest in the region on salinity-related agricultural aspects has gained new momentum in this century, mainly in Brazil, Mexico, Argentina and Chile. In addition to many publications, this is also reflected by recent national salinity conferences in Argentina and in Brazil (https://redsalinidad.com.ar/inicio/reuniones-ras/). The First Latin American Salinity Symposium was held in Fortaleza, Brazil, in 2019 (https://inovagri.org.br/programacao/). Books on regional salinity issues have been published in Spanish and in Portuguese (Gheyi et al., 2016; Taleisnik et al., 2008; Taleisnik and Lavado, 2017), including a recent comprehensive book (Taleisnik and Lavado, 2020).

The subcontinent is re-awakening to its saline perspective. Social impacts of this problem are being addressed, particularly because of food security issues. The process of salinization in irrigated areas is continuing, although in some cases drainage and improved irrigation technologies improved the situation noticeably. However, in many Latin-American regions soil salinization is still expanding. Deforestation has been extensive, and the consequences of these land use changes will further cause land degradation and affect the sustainability of its land and water resources. It is expected that the coming decade will provide more certain quantification of its increasing spatial extent, as FAO and various organizations from Latin American countries are on the way to develop a contemporary soil salinization map, following unified protocols.

15.8 Netherlands and neighboring lowland countries

15.8.1 Historical development

Problems with salinity in the Netherlands mainly occur in the North Sea coastal regions. Fig. 31 shows in blue colors areas either above or below sea level protected by dikes and in orange color areas below sea level not protected by dikes. The white areas along the coast, including islands in the north and southwest, are dunes in which fresh water floats on sea water.

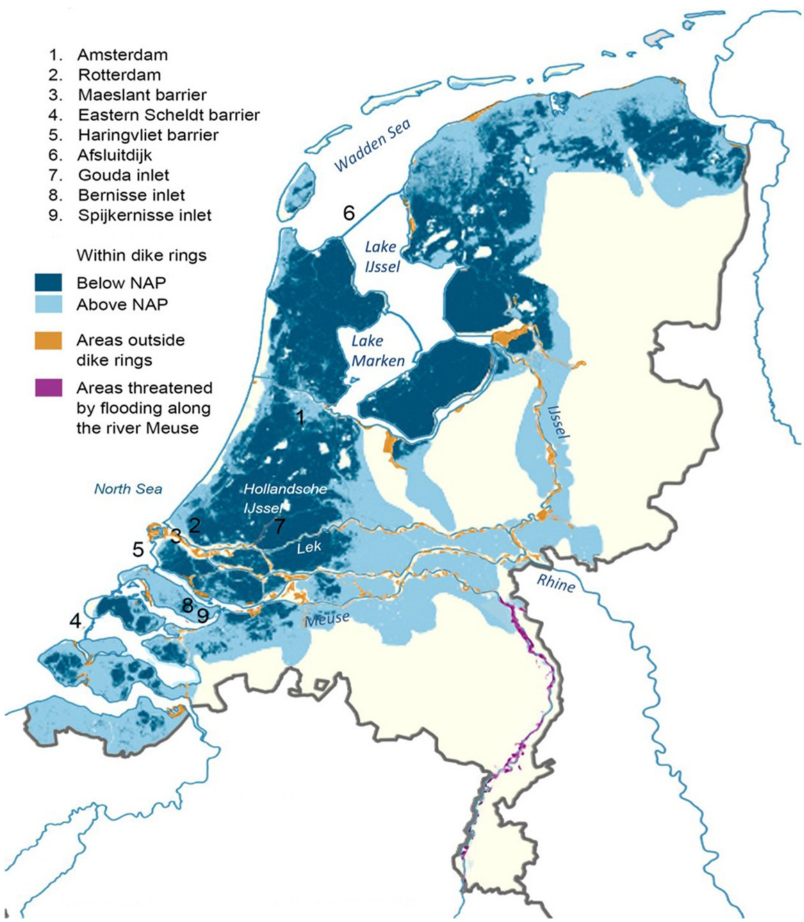

Fig. 31 Current map of the Netherlands. The nine numbers denote locations of the two largest cities, four seawater barriers, and three inlets of fresh river water. NAP stands for sea-level. Protection from flooding by seawater along the coasts is by either dunes or dikes. *Used with permission from: PBL Netherlands Environmental Assessment Agency/ Rijkswaterstaat-Waterdienst (2010), http://www.pbl.nl.*

Coping with salinity from major floods in the North Sea Dutch coastal regions—Throughout history, various aspects of salinity were recognized and dealt with, specifically causes of salinization and sodification leading to soil structure deterioration, desalinization and rehabilitation by de-sodification, and crop salt tolerance/intolerance (Raats, 2015). Originally, experience of water managers and farmers formed the basis in their decision-making. From 1850 onwards, traditional opinions gradually evolved into scientific understanding. First, these were mainly based on chemical analysis of soils, later combined with physico-chemical concepts, and more recently through inclusion of analyses of flow and transport processes and plant physiology.

In the first half of the 20th century, salinization and sodification arose from both natural floods (1906, 1916) and wartime strategic inundations (1939/1940, 1944/1945). In the aftermath of the large 1916 flood, plans were made for the Zuiderzee Works, resulting in completion of the Afsluitdijk (=Enclosure Dam, No 6 in Fig. 31) in 1932, changing the former tidal and saline Zuiderzee in a freshwater environment. Behind this dam are now the freshwater Lakes IJssel and Marken, surrounded by a series of new polders with a total area of 165.000 ha (see Fig. 2 in Raats, 2015). In other words, where formerly was the Zuiderzee are now two lakes and the Wieringermeer and Flevo polders. The two lakes serve as fresh water reservoirs for the northern provinces, including replenishment of water pumped for domestic use in the coastal dunes.

The early salinity research in the Netherlands was linked to consequences of the major floods and the Zuiderzee Works. Much of this specific research was presented in Raats (2015) and includes pioneering studies by Dutch scientists through the later 1800s and into the first half of the 20th century. Specifically, attention was paid to acid sulfate soils, application of gypsum to remedy soil structural degradation, analysis of plant salt tolerance, planting of salt tolerant vegetation to reclaim lands below sea-level (polders), and understanding seepage from saline open water into lower lying land.

Immediately following the devastating February 1953 Stormflood, the Delta Plan was launched, aimed at preventing recurrence of damage from such rare, huge storms in the future. The plan included upgrading all dikes along the entire coastline and build a series of barriers in the Southwest to close off all tidal inlets (locations 4 and 5, Fig. 31), except the Western Scheldt. The original main aims were protection of life and property and reduction of costs of maintenance of dikes. A reduction of saline seepage into many polders on the islands in the Southwest would also have resulted.

During the execution of the Delta Plan, pressure from environmentalists and fisherman ultimately led to drastic changes in the plans. While construction of a dam in the Eastern Scheldt had started already in 1960, it was not until 1979 that parliament approved a novel type of storm surge barrier, with gates that can be closed when necessary (http://www.deltawerken.com/English/10.html?setlanguage=en). This barrier was completed in 1986. Earlier, in 1974 it was decided to keep the planned fresh water Grevelingen Lake saline by means of a sluice in the dam, which was completed in 1978.

15.8.2 Current progress

Density stratified flows—Already in the 1950s W.H. Van der Molen (https://edepot.wur.nl/350617) noted the occurrence of high salinities in the North-East Flevo Polder at depths of 10–15 m in places where a highly permeable Pleistocene deposit reached the land surface. He speculated that these high salinities were probably due to convection currents caused by the small difference in density between the freshwater present in the soil and the supernatant seawater of the former Zuiderzee between 1600 and 1931 AD. More generally, on the time scale of centuries, marine transgression may cause rapid salinization of entire aquifers. In Western Europe Holocene transgressions of a few thousands of years have brought salt water of corresponding age to a depth of over 200 m. Nevertheless, at many places all around the world fresh and brackish waters have been found on the continental shelves (Post et al., 2013). Numerical modeling by Post and Simmons (2009) illustrates how low-permeability lenses protect fresh water from mixing with downward invading overlying saline ocean waters with higher density. Van Duijn et al. (2019) gave a general, modern stability analysis of such density stratified flows below a ponded surface.

Saltwater intrusion by tides in the mouths of rivers—The Zuiderzee Works and Delta Plan stopped salinization from tidal motion in the North. In the Southwestern Delta, tidal motion was only partly eliminated and no major freshwater reservoirs are available, like the Lakes IJssel and Marken for the northern provinces (Fig. 31). Instead, fresh water supply in the southwest comes more directly from diversions of water from the major rivers. In the 20th century the quality of the Rhine water gradually deteriorated, until a series of international treaties brought improvement. The river water quality was further reduced by an inward directed flow of high-density saline water underneath the outward directed flow of lighter runoff water. Traditionally, the tides had free play and salinized the river water far inland, particularly in periods of low river flows (Van Veen, 1941). As a result of this salinization, in the 1970s the surface water in the important Westland greenhouse district between Rotterdam and the Hague was hardly suitable for use as irrigation water. The growers themselves made it even worse using drainage return flows, resulting from high leaching fractions combined with high application of fertilizers. The RAND corporation did a policy analysis of water management for the Netherlands (e.g. Abrahamse et al., 1982), balancing engineering ambitions and agricultural interests, specifically regarding the desired irrigation water quality for use in greenhouse horticulture. The Delta Works have provided some relief from saltwater intrusion in river mouths; however, conflicting agricultural and environmental interests continue to dominate the discussion about seawater blockage as related to the desire to maintain brackish aquatic ecosystems.

Saltwater intrusion by inward flow of water to land below sea-level—Fig. 32 shows the depth of the brackish-fresh interface in the coastal regions of the Netherlands. Similar maps are available for the coastal region of Belgium (Vandenbohede et al., 2010). Because fresh water is floating on top of saline groundwater in the dunes area along the west coast, saline intrusion is strongest in the North and Southwest, where coastal dunes are absent.

At numerous locations in the dunes, fresh dune water is pumped as a source for preparing drinking water for the western part of the country, where the groundwater is too saline because of continued saltwater intrusion. For example, a dune area of 3400 ha along the western coast supplies fresh drinking water to Amsterdam, already since 1853. To keep the floating bodies of fresh water in the dunes intact, the freshwater pumping is compensated for by excess rainfall and infiltration of river water, partly after having been stored in the Lakes IJssel and Marken.

Fresh water floating on top of salt water in agricultural fields—Recently fresh water lenses floating on top of saline groundwater have been fully recognized as being of great importance, not only in the dunes, but also in farmer fields along coastal regions where upward seepage of saline groundwater occurs (see Fig. 32). These freshwater lenses can come from rain, melted snow, and increasingly also from irrigation of agricultural lands.

Eeman et al. (2011) made a detailed analysis of the thickness of a freshwater lens and the transition zone between this lens and the upwelling saline water. Starting from a fully saline condition between drains or ditches and assuming constant rates of saltwater upwelling and freshwater infiltration, they showed that a freshwater lens will grow until it reaches a maximum size. Moreover, they concluded that the fresh/saline ratio of the

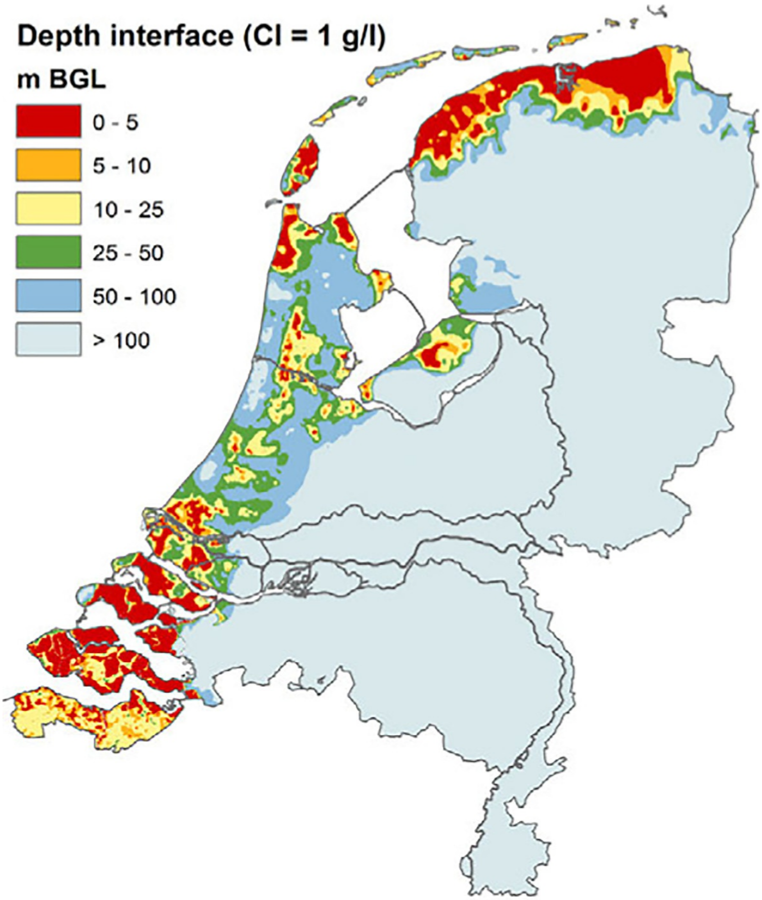

Fig. 32 Depth of the fresh-saline interface in meters below ground level. The interface is set arbitrarily at 1 g L^{-1} chloride (fig. 1.1 in De Louw, 2013). *Used with permission from the author.*

drainage water will change from zero to the infiltration/upward seepage ratio. However, as shown by others (De Vos et al., 2002; Delsman et al., 2014; Eeman et al., 2012), seasonal variations of infiltration and plant root withdrawal of fresh water will cause temporal fluctuations of the thickness of the lenses and the fresh-saline ratio of the drainage water.

Salt tolerance in a generally humid and cool climate—Most salt tolerance data for field crops and flower species date from before 2000 and were reviewed by Van Bakel et al. (2009) and Stuyt et al. (2016). The latter compilation in Dutch is the most complete, providing salt tolerance thresholds for

35 individual crops or groups of crops. Salt tolerance data for greenhouse horticultural crops were brought together by Sonneveld (2000) and Sonneveld and Voogt (2009), and included interactions between plant nutrition and salinity.

In the last decade, salt tolerance tests have been carried out at Salt Farm Texel (De Vos et al., 2016; Van Straten et al., 2016, 2019a). The 160 m^2 experimental plots were irrigated, using eight replications of seven different salt concentrations, obtained by mixing saline seawater with fresh water. Because of the high hydraulic conductivity of the soil, it was possible to maintain the desired concentration throughout the rootzone, irrespective of the weather in the growing season. Salt tolerance was tested for six crops: potato (5 varieties), carrot (7), onion (4), lettuce (3), cabbage (2), and barley (2). The goal was to identify crop varieties that have high salt tolerance. The data were analyzed using the Maas and Hoffman (1977) and Van Genuchten and Gupta (1993) models. An alternative model based on the Dalton-Fiscus model for simultaneous uptake of water and solutes was explored by Van Ieperen (1996).

Salinization in the countries around the North Sea—In principle, the lowland coastal regions of Belgium, Germany, the Netherlands, Sweden, and the United Kingdom face similar threats from salinity as in the Netherlands. For example, there was widespread flooding of farmland along the UK east coast during the Southern North Sea storm of December 5, 2013 (Spencer et al., 2015). Due to different economic and political priorities, the responses to such events have varied. The Netherlands was saved potential disastrous flooding in 2013, thanks to the Delta Plan response to the 1953 Storm Flood. Gould et al. (2020) analyzed the impact of coastal flooding on agriculture in Lincolnshire, UK. They noted that flood risk assessments typically emphasize the economic consequences of coastal flooding on urban areas and national infrastructure and tend to omit the long-term impact of salinization of agricultural land. Considering this long-term salinization, they calculated financial losses ranging from £1366/ha to £5526/ha per inundation, which would be reduced by between 35% up to 85% by post-flood switching to more salt-tolerant crops.

15.8.3 Future outlook

1. Reaching the goal of operating greenhouses as closed, recirculating systems, with minimal periodic refreshment of the nutrient solution in 2027, will be an important milestone after more than a half century of R&D in sustainable greenhouse production systems. The best way to

avoid inputs of sodium and chloride is to harvest and store greenhouse roof runoff and use that as irrigation water. From 2027 onward, disposal of water containing nitrogen or phosphate is no longer permitted and this requires good data on N and P crop requirements. In glasshouse horticulture, drainage used to be 10 cm yr^{-1} ($=1000\ m^3\ ha^{-1}\ yr^{-1}$), now growers have reduced that to 1 cm yr^{-1} ($=100\ m^3\ ha^{-1} yr^{-1}$), and in a recent pilot study this was brought down to 1 mm yr^{-1} ($=10\ m^3\ ha^{-1} yr^{-1}$), an amount that can easily be desalted for reuse.

2. At present there are several groups advising on short- and long-range allocation of freshwater resources. The spatial/temporal distribution of the summer water balance is monitored closely by the KNMI (Royal Meteorological Institute), focusing on growing season water shortage. To complement the KNMI data, it is a big challenge for agronomists to develop tools for location specific, timely advice for growers, as summer water shortage will largely depend on location (soil type, root zone salinity, land use, water management).
3. Despite the large annual rainfall surplus in the Netherlands, cumulative evapotranspiration in the growing season is always larger than precipitation. Consequently, there is need for increasing temporary storage of excess precipitation. Inspired by the success of using the dunes for drinking water storage, there are currently numerous initiatives to develop surface and subsurface storage capacity to retain excess precipitation outside the growing season, rather than losing it as drainage water. To counter salinity damage as shown by the patterns in Fig. 32, there is great interest for good quality irrigation water in the coastal region in the north and for the islands in the southwest to counter salinity damage. Ideally, designing such storage capacity should be based on detailed geohydrological surveys and salinity monitoring, in combination with advanced multi-dimensional computational codes of subsurface flows.
4. There is hope that funds will become available for continuation of experimentation such as done at Salt Farm Texel in the last decade. In addition for the need of classical salt tolerance data for specific crops/varieties, studies are desired to evaluate the performance of those most promising crops/varieties under normal growing conditions. Computer simulation models such as SWAP (Kroes et al., 2009; Van Dam et al., 2008) or similar computational models (Section 3) could be used to plan, evaluate, and extrapolate such experiments.
5. In recent years, the availability of funds from the European Union has fostered cooperation. For the period 2017–2022 the EU Interreg

North Sea Region and several other organizations are financing the project Saline Farming (SalFar) (De Waegemaeker, 2019; Kaus, 2020). This project inspired the international Saline Futures Conference addressing climate change and food security (Saline Futures, 2019). The presentations included not only results from the EU project, but also from many other projects in the North Sea countries, and from related environments in Africa, Asia, and the eastern USA. Commonalities between the lowlands of the US and the Netherlands were already noted by Edelman and Van Staveren (1958), who at the invitation of the SCS toured the US Gulf and Eastern Coasts. The full publication of the papers presented at the Saline Futures Conference is in preparation and no doubt will inspire future studies of salinity issues in coastal regions around the world.

15.9 Nile Basin

15.9.1 Historical background

The Nile is a major north-flowing river in northeastern Africa. With its length of 6650 km, it is the longest river in Africa as its drainage basin covers 11 countries: Tanzania, Uganda, the Democratic Republic of Congo, Rwanda, Burundi, Ethiopia, Eritrea, Kenya, South Sudan, Republic of Sudan and Egypt (FAO, 2009). The Nile has two major tributaries—the White Nile and the Blue Nile (Fig. 33). The White Nile is considered as the headwaters and primary stream of the Nile itself. The Blue Nile, however, is the main source of the water, containing 80% of the water and silt. The White Nile is longer and rises in the Great Lakes region of central Africa and flows north through Tanzania, Lake Victoria, Uganda and then to South Sudan. The Blue Nile starts from the Lake Tana located in Ethiopia and flows into Sudan. The two rivers meet just north of the Sudanese capital of Khartoum. The northern part of the river flows almost entirely through the Sudanese desert to Egypt where it finally ends into a large delta and then drops into the Mediterranean Sea.

The drainage basin of the Nile covers 3.3 million km^2, about 10% of the area of Africa. The Nile basin has complex hydrology; therefore, the discharge at any given location depends on multiple factors such as weather, diversions, evaporation and evapotranspiration, and groundwater flow. Considering the basin area of the Nile, Sudan has the largest size (1.9 million km^2) whereas, of the four major tributaries of the Nile river, three originate from Ethiopia—the Blue Nile, Sobat and Atbara. However, Sudan and Egypt are the major Nile water users (Mohamed et al., 2019).

Fig. 33 Map of Nile basin (Wikimedia).

Egyptians have practiced irrigated agriculture for about 5000 years in the Nile River valley, using basin irrigation dependent on the rise and fall of flows in the Nile river. Since 3000 BCE, the Egyptians used to construct earthen banks to form flood basins of various sizes, filled with the Nile water to saturate soils for crop production. Egyptian irrigated agriculture has been sustainable for thousands of years, in contrast to other civilizations in Mesopotamia. Reasons were provided by Hillel (1992), pointing to (i) the annual natural flooding that deposited nutrient-rich soil material, (ii) annual cycles of rising and falling of the Nile river that created fluctuations of the groundwater table and yearly flushing of salts of its narrow irrigated flood plains, and (iii) the annual inundations that occurred in the late summer and early fall, after the spring growing season.

With the construction of the Aswan High Dam, most of the land was converted to perennial irrigation and the irrigated area increased from 2.8 to 4.1 Mha. The year-around irrigation and lack of leaching by annual pulsing of the Nile river triggered soil salinization (El Mowelh, 1993). More than 80% of Egypt's Nile water share (55.5 Bm^3/year) is used in agriculture. Water- saving in agriculture is a major challenge because annual per capita water availability in Egypt is expected to decrease to 560 m^3 from a current level of 950 m^3.

The salts of the Nile basin are either of intrinsic origin, sea water intrusion (coastal regions) or from irrigation with saline groundwater. Since the climate of Egypt is characterized as arid with annual rainfall ranging from 5 to 200 mm compared to evaporation rates of 1500–2400 mm, crop production is not possible in most parts of Egypt without irrigation. Salinity problems in the irrigated areas are widespread and about 1 million ha are already affected. At present only 5.4% of the land resources in Egypt is of excellent quality, while about 42% is relatively poor due to salinity and sodicity problems. Soils in the Nile valley and the Delta are Vertisols, characterized by substantial expansion by wetting and shrinking by drying. In Egypt, productive lands are finite and irreplaceable and thus should be carefully managed and protected against all forms of degradation (Qadir et al., 2007).

Other countries of the Nile basin also have salinity problems. Kenya has about 5 Mha of salt-affected lands. In Tanzania, about 30% area is characterized by poor drainage and soil salinity problems. The soil salinity problems in countries such as DR Congo, Uganda, Burundi, and Rwanda are less prevalent however soils are low in fertility (FAO, 2009). The salt-affected lands in South Sudan and Sudan are in the White Nile irrigation schemes. This area has hardly been utilized for agricultural production despite having great potential due to the availability of water from Nile (Qureshi et al., 2018). In other parts of South Sudan, low soil fertility and lack of good quality seeds for crops and forages are the major bottlenecks in the development of agriculture.

Ethiopia stands first in Africa in the extent of salt-affected soils with an estimated 11 Mha of land exposed to salinity (Ashenafi and Bobe, 2016; Frew et al., 2015). This corresponds to 9% of the total land area and 13% of the irrigated area of the country. These soils are concentrated in the Rift Valley, Wabi Shebelle River Basin, the Denakil Plains and other lowlands and valleys of the country, where 9% of the population lives (Frew et al., 2015). Currently, soil salinity is recognized as the most critical problem in the lowlands of the country resulting in reduced crop yields, low farm incomes and increased poverty (Gebremeskel et al., 2018). The insufficient drainage facilities, poor-quality groundwater for irrigation and inadequate on-farm water management practices are usually held responsible for the increasing salinity problems.

Despite the widespread occurrence of salt-affected soils, Ethiopia does not have an accurate data base on the extent, distribution, and causes of salinity development. Most of the saline soils are concentrated in the plain lands of the Rift Valley System, Somali lowlands in the Wabi Shebelle River Basin, the Denakil Plains and various other lowlands and valley bottoms throughout the country (Ashenafi and Bobe, 2016). The introduction of large-scale irrigation schemes without the installation of appropriate drainage systems have also resulted in the rapid expansion of soil salinity and sodicity problems in the lower Wabi Shebelle basin of Gode (Somali Region). The distribution of surface salinity in the four largest regions of Ethiopia is given in Table 5.

15.9.2 *Current progress*

Sudan has built four dams on the Nile during the last century to provide irrigation water to an additional 18,000 km^2 of land. This has made Sudan the second most extensive user of the Nile river water, after Egypt.

Table 5 Distribution of surface (0–30 cm) soil salinity in different regions of Ethiopia.

	Afar region		Amhara region		Oromia region		Tigray region	
	Area		Area		Area		Area	
Soil salinity levels	km^2	%	km^2	%	km^2	%	km^2	%
Non-saline/ Waterbody	40,787	42	137,4328	88	287,768	89	48,067	97.39
Low saline (2–5 dS/m)	26,916	28	4903	3	17,292	5.3	0	0
Medium saline (5–10 dS/m)	9798	10	11,892	8	17,152	5.3	1339	2.7
Highly saline (10–15 dS/m)	5618	5	1230	0.8	1577	0.5	0	0
Extremely saline (>15 dS/m)	14,085	15	202	0.2	714	0.3	0	0
Total	97,204	100	155,648	100	324,429	100	49,406	100

Despite these arrangements, Sudan has not achieved full production potential due to lack of water infrastructure for equitable water distribution among farmers, lack of farm inputs and low soil fertility conditions. In Egypt, about 85% of the available water resources are consumed by the agriculture sector. The completion of Aswan dam increased the intensity of irrigation, which created waterlogging problems in many parts contributing to the pollution of land and water resources.

In Egypt, surface and subsurface drainage systems have been installed to control rising water tables and soil salinity. Besides, crop-based management is used to combat soil salinization (Qadir et al., 2007). Farmers were encouraged to use agricultural drainage water to irrigate crops thereby reducing disposal problems. However, the unregulated application of drainage water for irrigation has reduced crop yields and polluted soil and water resources. In addition to agricultural chemical residues and salts, drainage waters include treated and untreated domestic wastewater. The use of organic amendments and the mixed application of farmyard manure and gypsum was useful in reducing soil salinity and sodicity (Mohamed et al., 2019). Recently, phytoremediation or plant-based reclamation has been introduced in Sudan, for example to reduce soil sodicity instead of using gypsum (Mubarak and Nortcliff, 2010).

In the absence of surface and subsurface drainage systems, farmers in Ethiopia continue to manage salt-affected soils by adopting traditional salt management solutions. These include: (1) direct leaching of salts, (2) planting salt-tolerant crops, (3) domestication of native wild halophytes for agro-pastoral systems, (4) phytoremediation, (5) chemical amelioration, and (6) the use of organic amendments such as animal compost. Farmers have also used various drainage designs, allowing salts to settle before its reuse for irrigation water. However, all such practices have failed to mitigate salinity problems in the long-term. Hence crop yields continue to decline, resulting in reduced farm incomes, food shortage and increased poverty. Many of the smallholder farmers are also working as daily laborers, causing unprecedented farmer migration to nearby urban areas and exacerbating prevalent problems of urban unemployment (Kitessa et al., 2020).

15.9.3 Future outlook

The increasing demand for food for the rising population in Egypt (expected to reach 95 million in 2025 from the current value of 85 million), the country is trying to expand its irrigated agricultural area. The FAO, in collaboration with the Ministry of Water Resources and Irrigation (MWRI) and the Ministry of Agriculture and Land Reclamation (MALR), has recently launched the project "*Support sustainable water management and irrigation modernization for newly reclaimed areas*." This project will increase the efficient use of resources to achieve high productivity at low input level, while minimizing adverse external factors. They also focus on managing the ecological, social, and economic risks associated with production systems in the agricultural sector, including disease and climate change. The project will also focus on identifying and increasing the role of ecosystem services, especially regarding their effects on resources utilization, risk response and preserving the environment.

As the freshwater availability from the Nile river is decreasing, farmers are using low-quality groundwater for irrigation instead. It results in increased soil salinization thereby negatively impacting crop yields and quality (Gorji et al., 2017). Therefore, it is essential that Egypt regulates the reuse of drainage water to control soil salinization. This will require a robust salinity monitoring program that can provide updated information on the quality and quantity of drainage water and groundwater. Most importantly, these data are vital in developing strategies for the safe use of these waters. Like many other countries, Egypt needs to prepare comprehensive guidelines for the use of poor-quality drainage and groundwater for irrigation, considering soil types, climatic conditions, and crops to be grown. There is also a need to increase water use efficiency at the farm and basin levels. For the coastal areas where salinity levels are very high, the use of salt-tolerant crops and halophytes must be encouraged.

Despite vast salt-affected areas in Ethiopia, research and development projects that address salinity are mostly absent. Consequently, the current and future extent of salt-affected soils are unknown, whereas economic implications are not brought to the attention of policy makers. No country organization monitors, evaluates, and permits for expanding irrigation or to discontinue existing irrigated farms. Available information is limited and is based on preliminary studies that are incomplete in most cases or comes from outside Ethiopia. The country lacks a systematic analysis of salt-affected areas and its strategic plan that addresses soil salinization and sodification. Such a project should lead to sustained funding of soil salinity research that assesses the quantification of its extent and damage, as well as the development of technologies and management practices that reclaim and prevents further expansion of soil salinity in the country. Specifically, the introduction of adequate drainage systems must be considered, and irrigation water conveyance channels should be lined to reduce water losses, especially in areas of saline groundwater. In addition, the selection of salt-tolerant forages, crops and legumes could largely improve the productivity of salt-affected lands. In summary, Ethiopia must develop a long-term national policy and strategic plan that leads to lasting solutions for its irrigated agriculture.

Another significant development in this region is the construction of the world's largest dam on the Nile River by Ethiopia. The Grand Ethiopian Renaissance Dam (GERD), on the River Nile near the Sudan border will have a reservoir capacity of 70 Bm^3 (equivalent to the entire annual flow of Blue Nile at the Sudan border) and an electricity generation capacity of 6000 MW. It is estimated that GERD will irrigate 1680 km^2 forest land in the northwest of Ethiopia. Ethiopia claimed that this dam would also benefit the downstream countries mainly Sudan and Egypt by removing 86% of their silt and sedimentation load and conserving water by regulating flow that will allow reliable all-season water supply to Sudan and Egypt (Tesfa, 2013). Although Ethiopia claims that there will be no consequences for downstream users such as Egypt (Sherien et al., 2019). There are concerns that GERD will reduce 12–25% (10 Bm^3) of Nile flow into Egypt especially during the dam filling period of 5–7 years (Ibrahim, 2017). This will have severe consequences for optimal crop production and management of soil salinization in Egypt. Therefore, the cooperation between the Nile water-sharing countries is essential for the management and protection of this vital water resource to ensure future food security and livelihood of the 280 million people living in the Nile basin.

15.10 Pakistan

15.10.1 Historical development

Irrigated agriculture in Pakistan is mainly confined to the Indus plains where it has been developed by harnessing major water resources available to the country. The agriculture in the arid and semi-arid areas of Pakistan largely depends on sustained irrigation supplies, as the evapotranspiration demand is high, and rainfall is either inadequate or unreliable. The contiguous Indus basin irrigation system irrigates an area of about 16 million ha (Mha), diverting annually 131 billion m^3 (Bm^3) of surface water to 43 main canal systems. The perennial water supply is available to 8.6 Mha while the remaining area receives water only during the summer season. About 93% of the total water withdrawal is allocated to the agricultural sector, 4% is used for domestic purposes and the rest 3% goes to industrial use (Bakshi and Trivedi, 2011; Qureshi and Husnain, 2014).

The large-scale irrigation development in the Indus Basin was initiated in the second half of the 18th century to expand the settlement opportunities, avoid crop failure and famine. At that time, the groundwater levels were below 30 m from the soil surface, and therefore drainage needs were not considered (Fahlbusch et al., 2004). Due to persistent seepage from unlined canals and percolation from irrigated fields, the groundwater table rose to within 1.5 m of the soil surface, creating waterlogging and, consequently, soil salinity problems (Wolters and Bhutta, 1997). The problems of soil salinity became more noticeable in areas where groundwater was saline (Fig. 34).

Most of the soil salinity in the Indus basin comes from primary salinization (Section 2.3). However, secondary salinization using poor-quality groundwater for irrigation has further compounded the problem. The Indus basin is faced with a considerable salt balance problem. The average annual salt inflow by the Indus river water is estimated to be 33 million tons (Mt), while the outflow to the sea is only 16.4 Mt. The average annual salt storage of around 16.6 million tons is equivalent to 1 ton of salts per hectare of irrigated land. Therefore, saline soils have become an important ecological conundrum with 4.5 Mha (27% of the total area) already afflicted (WAPDA, 2007).

As illustrated in Fig. 35, the salinity problems in Sindh are most severe where about 50% of the irrigated area is affected. This is mainly due to poor

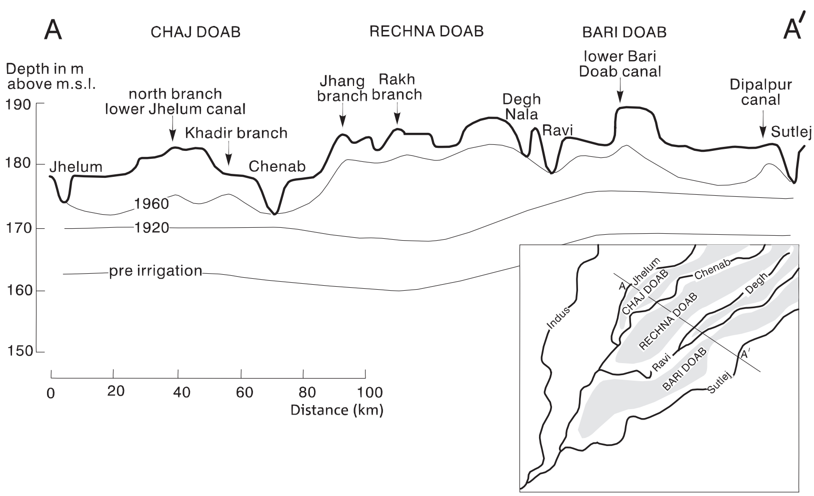

Fig. 34 Rise of the groundwater table after the introduction of canal irrigation in the Punjab, Pakistan. The groundwater profiles are shown for the years 1920 and 1960.

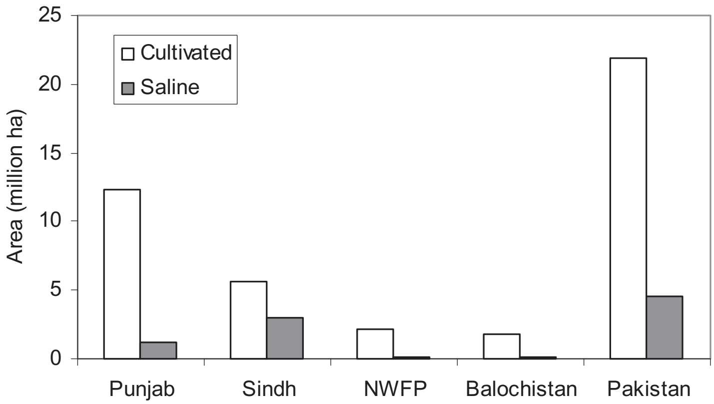

Fig. 35 Distribution of cultivated and salt-affected area in Pakistan by Province.

Table 6 Areas of salt-affected soils in the Indus basin by salinity classification.

Soil classifications	Area affected (million ha)	Characteristics
Slightly saline-sodic	0.4	Slight salinity-sodicity, occurring as patches (about 20% of the area) in cultivated fields
Porous saline-sodic	1.2	Saline-sodic throughout the root zone, porous and pervious to water
Severely saline-sodic	1.0	High groundwater tables, dense and nearly impervious to water
Soils with sodic water	1.9	Severely sodic due to application of sodic water

drainage conditions, shallow saline groundwater, and the use of poor-quality groundwater for irrigation, as surface water supplies are far less than the actual crop water requirements (Bhutta and Smedema, 2007). In addition to total soil salinity in the Indus basin, sodicity is a major problem because 70% of all groundwater wells in the basin pump sodic water, affecting soil structure and infiltration rates (Sections 2.3 and 12). Salt-affected soils of the Indus basin are usually classified into four types as shown in Table 6.

15.10.2 Current progress

The combined threats of waterlogging and soil salinization were recognized as early as 1870, and since then various remedies have been undertaken to overcome this twin menace. These include engineering solutions, reclamation strategies, and biological interventions. These are briefly discussed below.

Engineering solutions—The first detailed survey of groundwater table depth and salinity was conducted in the 1950s with the collaboration of the US Geological Survey. It formed the basis for public sector vertical drainage program through Salinity Control and Reclamation Projects (SCARPs). As a result, in both fresh and shallow groundwater areas, 14,000 tubewells with an average capacity of $80\,L\,s^{-1}$ were constructed between 1960 and 1970, covering 2.6 Mha of irrigated land with an estimated cost of US$ 2 billion (Qureshi et al., 2008). This program was aimed at lowering the groundwater table and increasing irrigation supplies at the farmgate by mixing pumped groundwater with fresh canal water. The SCARPs were partially successful in arresting waterlogging and salinity by lowering groundwater tables below 1.5 m in 2.0 Mha and below 3 m in 4.0 Mha. As a result, areas with soil salinity decreased from 42% in 1960 to about 32% in 1977–79, and improved irrigation supplies allowing increased cropping intensities from 84% to 125% in most SCARP areas (Qureshi et al., 2010).

In the 1970s, one realized that circulating salt-contaminated water through vertical drainage aggravated the salinity problem, thereby shifting to constructing horizontal drainage systems that were 10 times more expensive. The main argument in favor of horizontal drainage was that drainage water quality would improve over time, allowing more of it to be used for irrigation as well as reducing disposal problems. Since then, about 10 major horizontal drainage projects (12,600 km of pipe drains) have been completed in different parts of Pakistan. The major bottleneck in the successful operation of these drainage systems was the safe disposal of saline drainage effluent. To overcome this, Pakistan constructed a 2000 km long surface drain on the East side of the Indus River, moving drainage waters of more than 500,000 ha of land to the sea (Qureshi et al., 2008).

Reclamation strategies—The salinity management in Pakistan remained focused on lowering of groundwater table and leaching of salts, without a national action plan for reclaiming sodic and saline-sodic soils. Efforts by local governments were mainly confined to supporting field-level research and providing subsidies to the farmers for gypsum application. The use of gypsum, acids, and farmyard manure, in combination with surface scarping and deep plowing were extensively applied. Agricultural and industrial wastes such as farmyard manure and sugar industry byproducts have also been used to improve sodic soils. A large range of acid materials was tested in Pakistan including sulfur, sulfuric acid, and aluminum sulfate (Ghafoor et al., 2004). However, due to their cost and management complexities, farmers deemed these less attractive. Instead, gypsum was considered the most cost-effective additive for the reclamation of sodic soils and is heavily subsidized by the government (Shah et al., 2011).

Biological interventions—The biological approach emphasizes the use of highly saline water and lands on a sustained basis through the profitable and integrated use of the genetic resources embedded in plants, animals, and improved agricultural practices. In Pakistan, a considerable amount of work has been done to use highly saline waters for growing salt-tolerant crops (Ghafoor et al., 2004; Shah et al., 2011). This includes the planting of salt-tolerant plants, bushes, trees, and fodder grasses. Plants, particularly trees, are commonly referred to as biological pumps and play an important role in the overall hydrological cycle for a given area. In Pakistan, bioremediation was promoted as a valuable tool for controlling rising water tables and salinity, through enhanced evapo-transpiration (Dagar et al., 2011). During the last 20 years or so, many salt-tolerant species and varieties have been developed in Pakistan, such as poplar, eucalyptus, tamarix, maskit and acacia. Similarly, nonwoody plants such as bushes, sedges, grasses, and herbs can develop deep-rooted systems that can use shallow groundwater (Choudhry and Bhutta, 2000). However, their ability to maintain low water tables is expected only when these plants occupy a sufficiently large area.

15.10.3 Future outlook

During the last 2 decades, Pakistan has made significant efforts to control soil salinization, which has reduced the saline area from over 6 Mha in 1970s to 4.5 Mha in 2007 (WAPDA, 2007).

Despite these massive investments over the last 3 decades, soil salinization remains the biggest challenge for the Indus basin. It continues threatening the sustainability of its agricultural system and the capacity of Pakistan to feed its growing population. Much discussion is focused on future water shortages and the need for adequate drainage of the Indus basin.

The salt management issues in Pakistan are complex, and therefore an integrated approach is a key for sustainable irrigated agriculture. The provision of drainage should therefore be a required complimentary activity to irrigation. Irrigation and drainage are closely linked because excess irrigation is the main cause of waterlogging while the level of irrigation management dictates the amount of effluent disposal. Drainage water disposal will remain a major issue for effective salinity management in Pakistan. Disposal of saline effluent in rivers merely transports the salts to irrigated lands at the tail end of the irrigation system. It is therefore neither a practical nor environmentally friendly long-term solution.

Due to the siltation of main reservoirs, the water storage capacity of Pakistan is expected to reduce by 57% by the year 2025 and to meet the future water requirements, 22 Bm^3 of more water will be needed (World Bank, 2008). Furthermore, due to climate change effects, future unmet water demand is likely to reach 134 million m^3 by 2050 (Amin et al., 2018). Consequently, unless Pakistan significantly increases its freshwater use efficiencies, it will have to use more poor-quality irrigation water in the future. Also, it will need to seek sustainable re-use of drainage water to minimize drainage effluent. Timely availability of farm inputs such as salt-tolerant germplasm and promotion of saline agriculture through crop diversification can improve the capacity of individual farmers as well. Most importantly, farmers will need to have access to new information about improved irrigation management and reclamation approaches.

16. Challenges, knowledge gaps and recommendations

The case studies of Section 15 illustrate the need for application of improved management practices for the major irrigated regions of the world. In this final section, we will synthesize the identified research priorities with these region-specific challenges and needs. Despite the large research and developmental efforts on salt-affected soils in the past, knowledge gaps remain, for new and innovative research and tools that will provide increasing resilience to salt-affected agriculture.

Water moves through hydrologic cycles and always carries salts and other elements with it naturally, as it moves through the landscape into the oceans. It is therefore that salinity and waterlogging have impacted agricultural production in arid areas for more than 2000 years. Long ago, Hilgard (1886) described the inevitability of salinity problems in arid areas and the measures required to prevent or overcome those problems, and he warned of impending salinization in California's Central Valley, based partly on his understanding of salinity and waterlogging problems in India 100 years ago. Despite that the required methods and investments to manage salt-affected soils are well-known, problems persist across the world. Many of the large irrigation projects such as in the Indus-Ganges (IGB) and Nile basins, Iraq, and China were developed in the 19th and early 20th centuries, using gravity to distribute fresh river water through canals and ditches for surface irrigation. However, as their efficiencies have declined for reasons of lack of infrastructure maintenance and regional wars, large-scale investments for the rehabilitation of existing drainage systems and installation of new drainage systems will be a huge challenge.

Alternative approaches such as irrigation management to improve irrigation water efficiencies (both distribution and field application) and to control percolation losses and reusing drainage water need to be prioritized (Oster et al., 2021). Drainage waters can also be used for the promotion of aquaculture or biosaline agriculture to grow feed crops for livestock, especially in those areas which are not suitable for conventional agricultural production systems. Especially for coastal areas where salinity levels may be high, use of salt-tolerant crops and halophytes must be encouraged, including by distribution of salt-tolerant germplasm and crop diversification. For plant breeding to be successful in finding more as well as increased salt-tolerant crop species, advanced breeding lines should be crossed in field settings for a range of salinity and water deficit levels. There is a sense of urgency here as the rate of soil salinization may be greater than the genetic gain of breeding increased salt tolerance into a conventional crop or pasture species. Introductions of non-conventional species into the production system should be considered. For plant breeding to be successful in increasing salt-tolerance within a conventional crop species, new germplasm should be crossed into advanced breeding lines as soon as feasible and the effects on grain yield should be evaluated in field settings for a range of salinity and water deficit levels. Moreover, rather than focusing on transgenic approaches, the natural variation in plants should be explored further and used to improve germplasm for production of major food crops on salt-affected land, with the option of applying gene editing. A major challenge is to engineer plants that use the prevalent Na and Cl ions as a way of osmotic adjustment without causing long-term toxicity. This is especially relevant when seeking for genetic improvement of crops for sodic soils, such as in regions where they are prevalent, with or without irrigation (Australia and Latin America).

Increasingly, the question that remains to be answered is whether irrigated agriculture is sustainable, as irrigation in semi-arid regions will almost always degrade soil and water quality, irrespective of applying sound salinity management practices. Concerns are widespread, with increasing trepidations about constrained freshwater availability globally, as irrigated agriculture consumes some 75% of the available freshwater resources, increasingly so from good quality aquifers that are not being replenished, while at the same time contaminating both good quality surface and groundwaters. As populations continue to rise, especially in emerging economies and resource-limited regions of Sub-Saharan Africa (SSA) and Asia, water scarcity is further threatening food security, caused by decades-long applications of irrigation water with extreme low water use efficiencies as well as by climatic changes that are increasing crop water requirements. To compensate for the decreasing freshwater availability, non-conventional water sources are being applied, much of these being of marginal quality such as drainage waters and treated wastewaters, possibly threatening more fragile environments and the functionality of productive agro-ecosystems. Concerns are not only for common salts, but increasingly so for other chemicals that become mobile when soils are irrigated, such as trace elements, heavy metals, emerging organic contaminants, and nitrogen fertilizers, threating food safety or causing soil degradation and sodification and other long-term unknown impacts on soil and water quality. These associated challenges provide for tremendous opportunities in soil salinity research, seeking more sustainable solutions for productive agriculture.

Despite that irrigation is vital to ensure food security in many arid or semi-arid countries, some may argue that much of irrigated agriculture is just not feasible, especially not so for relatively low value crops and/or when eco-environmental effects of salt-affected soils are taken in consideration. Irrigated agriculture may become a value proposition that must include the real cost of water and its disposal, as done for other sectors. One may also wonder about the costs of water used by agriculture if available for other uses such as for industry or residential use paying orders of magnitude more for close of the same quality water. For example, it was estimated that losses in the USA in 2016 because of droughts and shortages cost businesses US $14 billion (Davies, 2017). Much of irrigation water supplies are heavily subsidized, and do not charge for the use of the water, only for the cost of making it available. Increasingly, alternative production systems are introduced such as vertical or soilless farming, claiming order of magnitude more efficient water use. However, at the same time, one must not look only at the highly developed world countries where many can afford more expensive food. One must consider the fact that about 50% of the projected future world population will come from SSA where food shortages are forecasted but could be eliminated if investments were made in irrigated agriculture. For example, Van Van Schilfgaarde (1994) reported by that the potential for irrigated land in SSA is about 30 Mha, which is about 3 times larger than currently irrigated, but instead through large-scale irrigation schemes, they be planned from a farmer's point-of-view. In other resource-limited regions such as in the IGB, for governments to not invest for abating soil salinity will particularly raise risk for smallholder farmers and promote extreme inequity. Annual economic losses of productive land by salinization suggest have been estimated to be US$30 billion globally (Shahid et al., 2018), much of which is incurred by those smallholder farmers.

So, in the end, we need to evaluate the needs of irrigated agriculture carefully, including the risk of soil salinization, costs for management, and remediation or disposal, as soil salinization is becoming a global ecological issue. One needs to assess the socio-economic costs related to irreversible groundwater pumping and degrading freshwater resources and consider the true economic value of water. Water quantity and quality are closely linked, so that water degradation criteria must be considered when using water such as for irrigation. Alternative landuses may economically be feasible, such as plantation forestry practices through reforestation with salt tolerant species in saline discharge areas and coastal areas, or other ways to successfully apply biosaline agriculture as an alternative landuse for otherwise non-productive lands. Farm-level and policy decisions will vary among regions and over time, with differences in economic development and public preferences regarding the impacts of irrigation on its society and on the environment. Societies must determine the environmental consequences they are willing to incur and how to allocate costs between those who benefit directly or indirectly. Wichels and Oster (2006) have described the inevitable environmental impacts of irrigation, but agree that irrigation can likely be sustained, realizing that the cost of reducing environmental impacts to an acceptable level might be substantial in some areas.

We argue that adopting sound salinity and drainage management practices can sustain irrigated agriculture, but likely in different ways than in the past. In his concluding sentence, van Schilfgaarde (1994) states: "The technology is there or waiting to be discovered. The need is there. The potential is there. Do we have the will?" The future challenge is to devise strategies that increase food production while simultaneously preserving soil ecological functionality, minimizing human health risks, and promoting sustainable use of our land and water resources for agricultural use. The turn to desalination as a strategy for water security such as is done is some of the most water-scarce countries in the Middle East is a positive opportunity to reverse the non-sustainable trends of applying irrigation with water containing high concentrations of salts. Treatment of brackish groundwater and of water specifically destined for irrigation may in the future benefit from technologies that will selectively remove problematic ions before application. Such technologies may become economically feasible, instead of being allowed to negatively affect soils, crops, produce, and the environment. In addition, comprehensive salinity monitoring program are needed in regions with widespread soil salinization such as in the Euphratus-Tigris and Nile basins and the IGB. Such long-term monitoring will provide updated information on soil salinity and the quality and quantity of drainage water and groundwater, to allow for effective salinity management guidelines to be designed and executed across scales from the field to the district and the whole ecosystem.

Wichelns and Qadir (2015) reviewed various perspectives with an outlook toward a sustainable future coinciding with the goal of intensifying agriculture by agronomic practices that meet the nutritional food demands for an expanding global population. They proposed five actions that collectively would address the institutional and policy shortcomings that have hindered public or private investments in salinity management and drainage infrastructures in most irrigated regions of the world. Specifically, they suggest using financial incentives for the farmer, to invest in improved salt and water management practices at the farm scale, for example, through reimbursement by public or regional agencies based on their salt management activities.

Historically, research and developmental organizations have contributed significantly to the reclamation and management of salt-affected lands. But, they have been mostly working in isolation without interdisciplinary efforts. Considering the magnitude and complexity of the salinity problem, a multidisciplinary system's approach is required. Key policy impediments must be addressed to allow for rapid technology dissemination, especially in SSA and south Asia so that their farmers will have access to new information about improved irrigation management and reclamation approaches. New policies must include multi-stakeholder input (interface among policy planners, researchers, state agricultural departments and development boards, farmer's associations, self-help groups and NGO's) at the community level and provide incentives such as for subsidies and cost sharing. Creation of web-based platforms ensure multi-stakeholder input when making decisions on the development and implementation of reclamation technologies of saline-sodic soils. This particularly applies to regions with outdated irrigation infrastructures that were designed for surface irrigation but are now greatly inefficient. To successfully manage salinity anywhere, a multitude of stakeholders must strive to coordinate their efforts to use resources efficiently, develop solutions to local and regional problems, optimize funding opportunities, and seek to achieve a salt balance in any given basin. Such collaborative and multi-stakeholder efforts are currently also applied in California as they help build trust and consensus-building in the regional development of sustainable salinity management practices that meet multiple objectives for its diverse regions. A current example is the implementation of the Central Valley-wide Salt and Nitrate Control Program that includes both short- and long-term strategies to address salt and nitrate discharge issues in its Central Valley, covering 46,619 km^2.

Generally, countries lack a systematic approach of analysis of salt-affected areas and strategic plans will need to be developed that address soil salinization and sodification. Such plans should lead to sustained funding of soil salinity research that assesses the quantification of its extent and damage, as well as the development of technologies and management practices that reclaim and prevents further expansion of soil salinity in the respective regions. To evaluate lasting impacts of alternative soil salinity management may require sustained funding for longer than the typically short duration of most funded research projects (rarely exceeding 3 years). A major focus should be on developing remote-sensing techniques for regional soil salinity mapping that is necessary for planning and implementing regional strategies. FAO with various organizations and national governments are partnering with the GSP (Global Soil Partnership) to develop a contemporary global soil salinization map, following harmonized protocols between regions.

This scoping review was written for the purpose that our collective thoughts of knowledge gaps and priority needs in saline agriculture will accelerate added research funding for new knowledge and innovative solutions. We further want to inspire the science community to develop new directions of salinity research that address the gaps identified by our synthesis.

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