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Soil Salinity and Sodicity in Drylands: A Review of Causes, Effects, Monitoring, and Restoration Measures

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Soil salinization and sodification are common processes that particularly characterize drylands. These processes can be attributed either to natural conditions or anthropogenic activities. While natural causes include factors such as climate, lithology, topography, and pedology, human causes are mostly related to agricultural land-use, and specifically, to irrigated agriculture. The objective of this study was to thoroughly review this topic, while highlighting the major challenges and related opportunities. Over time, the extent of saline, sodic, and saline-sodic croplands has increased, resulting in accelerated land degradation and desertification, decreased agricultural productivity, and consequently jeopardizing environmental and food security. Mapping and monitoring saline soils is an important management tool, aimed at determining the extent and severity of salinization processes. Recent developments in advanced remote sensing methods have improved the efficacy of mapping and monitoring saline soils. Knowledge on prevention, mitigation, and recovery of soil salinity and sodicity has substantially grown over time. This knowledge includes advanced measures for salt flushing and leaching, water-saving irrigation technologies, precision fertilizer systems, chemical restoration, organic and microbial remediation, and phytoremediation of affected lands. Of a particular interest is the development of forestry-related means, with afforestation, reforestation, agroforestry, and silvopasture practices for the recovery of salt-affected soils. The forecasted expansion of drylands and aggravated drying of existing drylands due to climatic change emphasize the importance of this topic.

Keywords: climate change, crop selection, drought stress, environmental sustainability, precision agriculture, primary salinity and sodicity, secondary salinity and sodicity, water logging

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INTRODUCTION

Salinity and sodicity of soils can be caused by either natural or anthropogenic factors. While natural salinization and sodification processes occur regardless of anthropogenic activities, agriculture, specifically irrigated cropping systems, accelerates these processes without a doubt (Ayers and Westcot, 1985; Ghassemi et al., 1995; Forkutsa et al., 2009; Litalien and Zeeb, 2020). Over time, salinization and sodification of lands put environmental sustainability and agricultural crop productivity at risk (Jamil et al., 2011; Shrivastava and Rajesh, 2015; Singh, 2015; Ivushkin et al., 2019). Soil salinization and sodification processes mostly characterize dryland regions (Brown et al., 1982; Artzy and Hillel, 1988; Ghassemi et al., 1995; Cuevas et al., 2019 (**Figure 1**). Yet climatic changes, with forecasted increasing temperatures and growing frequency and magnitude of droughts

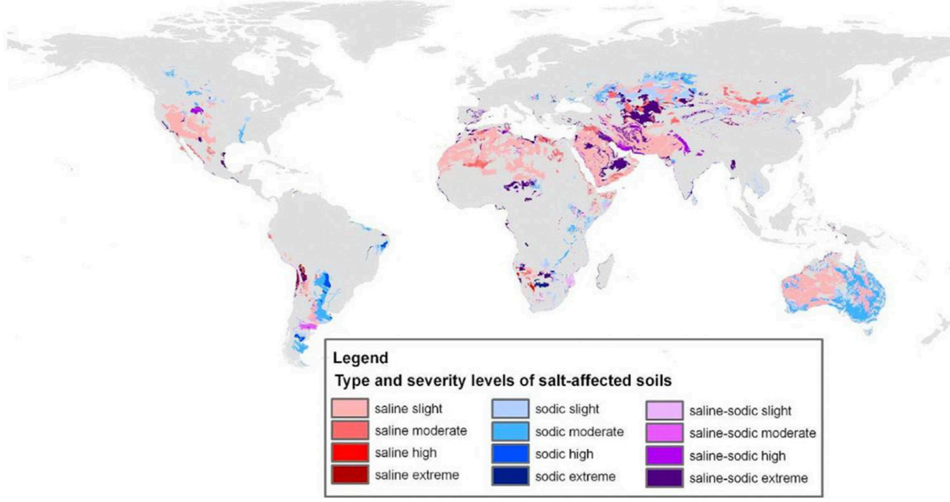

FIGURE 1 | Global distribution of saline, sodic, and saline-sodic soils. Source: Encyclopedia of the environment. https://www.encyclopedie-environnement.org/en/zoom/land-salinization/.

in moister climatic regions, alongside with the aggravated drying and expansion of the world's drylands (Cook et al., 2014; Huang et al., 2017a), make soil salinization and sodification a global challenge.

Soil salinity refers to the presence of water-soluble salts, including sodium (Na^+), potassium (K^+), chloride (Cl^-), and sulphate (SO_4^{2-}). Some ions, such as K^+ and SO_4^{2-}, also act as plant nutrients, while Na^+ and Cl^- are not considered plant nutrients. Therefore, soil salinity often focusses on Na^+ and Cl^-. Sodicity refers to an excess of Na^+ among the exchangeable cations in the soil solution (Qadir et al., 2007). Salinity and sodicity impact plant growth directly through the effects on water uptake by plants, nutrient availability for plants, and by imposing plant toxicity (Ayers and Westcot, 1985; Litalien and Zeeb, 2020), and indirectly, through the deterioration of soil physical conditions (Driessen et al., 2001).

The movement of water from the soil into the roots, through the stem into the leaves, and finally to the atmosphere, is mainly driven by a decreasing water potential (ψ) along this track. When water transpires from a plant's leaves to the atmosphere, both the plant's water content and water potential decrease. This results in movement of water through the plant tissues towards the leaves. When water potential in the root cells is lower than that of the soil's rhizospherous zone, this water is replenished by water that the plant takes from the soil. The presence of water-soluble salts in the soil increases the osmotic potential, and hence, reduces the water potential of the soil, bringing it closer to the water potential in the plant roots. Therefore, water uptake by the roots slows down, causing drought stress for the plant. To some extent, plant roots are able to actively pump water from the soil and transport it further upwards. However, this process is energy-intensive and lowers the growth rate of plants (Ayers and Westcot, 1985).

Plant macro- and micro-nutrients, such as the cations K^+, calcium (Ca^{2+}), magnesium (Mg^{2+}) and ammonium (NH_4^+), and the anions nitrate (NO_3^-), phosphate (PO_4^{3-}) and SO_4^{2-}, are absorbed through carrier proteins, which are located in the outer cell membranes of root epidermal cells. Among these carrier proteins, most are able to channel different cations or anions through the cell membrane. However, a limited number of carrier proteins is specified for one particular ion (Reid and Hayes, 2003). In the less specific carriers, nutrients compete among each other, as well as with other ions, for transport into the root cells. Regarding salinity and sodicity, Na^+ competes with other cations, particularly with NH_4^+ and K^+, while Cl^- competes with NO_3^-. In essence, the presence of Na^+ decreases the uptake of NH_4^+ and other cations, while the presence of Cl^- decreases the absorption of NO_3^- from the soil's exchange complex (Reid and Hayes, 2003).

The Cl^- is not absorbed in the soil, and therefore, it moves with the soil-water and is easily absorbed by plants. Within the plant, Cl^- is transported with the sapflow to the leaves, where it increases the osmotic potential of the sap and reduces water availability in the leaf tissue for plant metabolism. Symptoms of Cl^- toxicity are leaf burn and dry leaf tissue. At the same time, Na^+ ions are absorbed by the soil, and therefore, are not absorbed as readily as Cl^- ions. Also, Na^+ competes with other cations for absorption by plants. Therefore, high concentrations of K^+, Ca^{2+}, and Mg^{2+} reduce Na^+ uptake by the plants. However, if Na^+ concentration exceeds the plant specific threshold in the leaf tissue, the effects and symptoms of Na^+ toxicity are the same as for Cl^- toxicity (Ayers and Westcot, 1985).

Plants have several mechanisms to cope with excess Na^+ and Cl^-, all of which require additional energy, and therefore, adversely impact plant growth (Litalien and Zeeb, 2020). The most common mechanism is the prevention of salt uptake into the plant, by enzymes in the root cell membranes that constantly pump Na^+ and Cl^- back to the soil (Munns et al., 2019). Another mechanism is the secretion of salts through glands on the leaf. Although the plant absorbs salt, potentially impacting plant metabolism, salt secretion controls its concentration. An

TABLE 1 | Classification of salt-affected soils.

Type	Soil EC (dS m^{-1})	Soil pH	Soil ESP	Soil SAR	Soil physical condition
Saline	>4.0	<8.5	<15	<13	normal
Sodic	<4.0	>8.5	>15	>13	poor
Saline-sodic	>4.0	<8.5*	>15	>13	normal

*Notes: EC–electrical conductivity; ESP–exchangeable sodium percentage; SAR–sodium adsorption ratio; * despite many exceptions of saline-sodic soils with pH > 8.5. Source: FAO/UNESCO (1974).*

additional mechanism is the accumulation of salts in the plants' vacuoles. This mechanism requires steady activity of enzymes that pump the salts against a concentration gradient in the vacuole (Litalien and Zeeb, 2020).

Soil sodicity, or the soil's imbalance with regards to Na^+ is expressed through the sodium adsorption ratio (SAR: **Eq. 1**) or the exchangeable sodium percentage (ESP: **Eq. 2**) (Qadir et al., 2007):

$$SAR = \frac{cNa^+}{\sqrt{\frac{1}{2}\left(cCa^{2+} + cMg^{2+}\right)}} \quad (1)$$

where: c–concentration ($mmol_c$/l); Na–sodium; Ca–calcium; Mg–magnesium.

$$ESP = \frac{100\,(eNa^+)}{CEC} \quad (2)$$

where: eNa^+—exchangeable sodium ($mmol_c$/kg); CEC–cation exchange capacity ($mmol_c$/kg).

Soils with an ESP of 15 and more (which corresponds to SAR ≈13) are considered sodic (Driessen et al., 2001; Qadir et al., 2007). Sodic soils often have a pH higher than 8.5 and up to 11 (**Table 1**), imposing alkaline conditions that are toxic for plants. These conditions are caused by the presence of bicarbonate (HCO_3^-), from evaporation of water containing HCO_3^-, or from biological reduction of SO_4^{2-} under water-saturated conditions.

Sodic soils contain dispersed clay materials, which clog soil pores so that infiltration and aeration of the root zone is hampered. The processes that cause clay dispersion are explained in full details in Qadir et al. (2007). In essence, divalent cations bind the negatively charged clay compounds stronger than monovalent cations. Cations are bound to the negatively charged clay surfaces, while at the same time and according to a concentration gradient, the cations tend to diffuse into the soil solution. This results in a positively charged cloud of cations around the clay compounds. If these clouds are large, neighboring clay compounds are driven away from each other, resulting in swelling or increased dispersion. Monovalent cations, in particular Na^+, build larger positively-charged clouds than divalent cations. Na^+ carries a larger hydrate shell than K^+, which enhances clay dispersion.

The objective of this study was to thoroughly review the topic and implications of soil salinity and sodicity in drylands. In the following sections, we review selected aspects related to these processes, which are relevant for land managers and policy makers. These aspects include: natural causes (primary salinity or sodicity); anthropogenic causes (secondary salinity or sodicity); mapping techniques of saline lands; and the prevention, mitigation, and restoration of saline, sodic, and saline-sodic soils.

NATURAL CAUSES

Primary salinization and sodification are the processes of salt and sodium accumulation due to natural causes, namely mineralogy of the parent material, topography, and water table quality. Both sodium and other ions are products of primary mineral weathering. The weathering process releases soluble cations and anions, of which the most common cations are Ca^{2+}, K^+, Mg^{2+}, and Na^+, and the most common anions are HCO_3^-, Cl^-, and SO_4^{2-}. Certain minerals and rocks are more susceptible to chemical weathering than others, and release cations and anions more easily (Sverdrup and Warfvinge, 1988).

In humid climates, cations and anions are generally leached from the soil system and transported by water movement to low-lying landforms or groundwater aquifers (Zinck and Metternicht, 2009). In arid, semi-arid, and sub-humid climates, such cations tend to remain in the soil exchangeable complex, or to precipitate as secondary minerals when the ionic concentration in the solution reaches saturation of a certain salt. The less soluble salts, calcium carbonate ($CaCO_3$), gypsum ($CaSO_4.2H_2O$), and magnesite ($MgCO_3$), can easily precipitate under arid and semi-arid conditions. The result is a relative increase in the proportion of Na^+ ions in solution, and, consequently, a replacement of some exchangeable Ca^{2+} and Mg^{2+} by Na^+ on the exchange complex (Bui, 2017). This process increases ESP values, and leads to soil sodification. In arid conditions, where high evaporation rates tend to concentrate the water solution, salts that are more soluble than gypsum may precipitate in the soil. These salts include sodium carbonates (trona, nahcolite, thermonatrite), sodium sulphates (e.g., thenardite), magnesium sulphates (e.g., epsomite), potassium chloride (sylvite), magnesium chlorides (e.g., bischofite), and the most soluble and common salt, sodium chloride (NaCl, halite). Salts that are more soluble than gypsum are called 'soluble salts' and encompass the diagnostic minerals that define saline soils (IUSS Working Group WRB, 2015). Soils that have a high concentration of these soluble salts at some time in the year, either at the surface or at a certain depth, are classified as "solonchacks" according to WRB international classification (IUSS Working Group WRB, 2015). The presence of abundant Na-bearing minerals in the parent rocks, namely amphiboles (e.g. glaucophane, riebeckite, pargasite, arfvedsonite), Na-pyroxenes (e.g. augite, aegirina), and Na-plagioclases (e.g. albite, oligoclase, andesine) contribute to the high release of Na^+ anions and then to sodification and/or salinization processes (Monteiro et al., 2012). The recharge and throughflow on sloping lands and discharge at lower topographic positions also contribute to land sodification (Fitzpatrick et al., 1994).

Some lithologies release relatively high quantities of cations and anions also in non-dryland regions. For example,

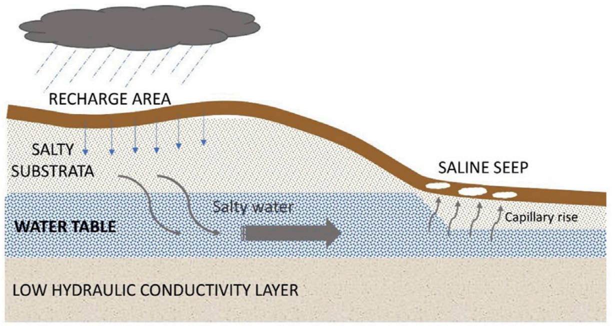

FIGURE 2 | Schematic illustration of saline seep formation.

sedimentary rocks deposited in lagoon or marine environments may include evaporite strata associated with shales and marls. Weathering of evaporite strata highly increases the content of dissolved salts in the circulant water. Often, such geological formations of evaporites are relatively plastic and tend to form domes through the tectonic process of diapirism. Usually, the formations of such evaporites occur on higher landscape positions, making them a source of salts for the surrounding landscapes (Zinck and Metternicht, 2009).

In drylands, salt-rich groundwater bodies downslope of recharge areas, called "saline seeps" (Brown et al., 1982) (**Figure 2**), elevate soil salinity enough to inhibit vegetation growth. The United States Department of Agriculture (USDA) describes several types of saline seeps, based on different geological, morphological, and hydrological contexts, including "slope change seep," "soil texture change seep," "geologic outcrop seep," and "hydrostatic pressure seep" (Brown et al., 1982). Alluvial plains and wetlands in arid and semiarid regions are often very sensitive to primary (and secondary) salinization as they accumulate overland water flow due to their low relative elevation. The processes of salinization in these landforms are driven by the dynamics between ground- and surface-water. During dry periods, low-salinity groundwater has beneficial effects because its discharge can replace evaporating surface water, maintaining moderate salinity in the alluvial plain or wetland. On the contrary, when groundwater is saline and the water table level rises because of land-use change or river regulations, the impact on ground surface salinity is detrimental (Jolly et al., 2008). In extreme cases, salt marshes or highly saline discharge playas may form. Yet, where groundwater is absent or very deep, such as in the case of recharge playas, salinization processes are negated. While agricultural land-use is impossible in discharge playas, it can be successful in recharge playas (Stavi et al., 2017).

Another natural source of soil salinity and sodicity in arid and semi-arid regions is linked to volcanic activity, and specifically, to wind-blown ashes rich in sodium-rich minerals such as plagioclases and Na-pyroxenes (Rodríguez-Rodríguez et al., 1993). Secondary volcanic activities, like fumaroles and thermal springs, are sources of chlorine and sulfur components that increase the salinity of groundwater and soils (Di Liberto et al., 2002).

Coastal areas in arid and semi-arid climates are at high risk of primary soil salinization. Salts transported by winds directly from the sea surface through marine spray can be deposited on the inland ground surface. This is a common primary salinity process in oceanic islands, such as Hawaii (Whipkey et al., 2000). Aeolian salt deposition can affect areas several kilometers from the coastline. For example, in the Western Australia Wheatbelt, Pannell (2001) reported on considerable salt deposition (20–200 kg ha^{-1} year^{-1}) by wind and rainfall at considerable distances from the coastline. Another type of primary salinity in coastal areas subjected to tides is the intrusion of saline water into rivers and groundwater. Specifically, the movement of high salinity backwater from the river delta inland is responsible to this process. The major cause of the backwater effect is the rise in sea level. Therefore, due to global warming, the extent of this process is expected to increase in the future. An example for such processes is evident in the Ganges, Brahmaputra, and the Meghna delta in Bangladesh (Mahmuduzzaman et al., 2014). This issue is relevant to many other large river deltas such as Mississippi, Orinoco, Niger, Indus, Mekong, and Yellow River, all of which have a huge impact on food and water security at local and regional levels (Rahman et al., 2019).

ANTROPOGENIC CAUSES

Secondary soil salinization is attributed either to water logging or to irrigation without proper leaching and drainage. These conditions leave dissolved salts in the soil profile or on the ground surface. Over time, if salts are not washed out, they will accumulate up to levels which impede plant growth (Ayers and Westcot, 1985) (**Table 2**). Secondary sodification is mostly attributed to the continuous use of sodic water for irrigation, resulting in the formation of dense clay-sodic layers in the sub-surface, which impede hydraulic conductivity of the soil profile (Shahid et al., 2018b).

Already in ancient times, secondary soil salinization affected irrigated agriculture, and resulted in the collapse of civilizations, such as in the case of Mesopotamia (Artzy and Hillel, 1988). Obviously, oases with a history of hundreds or thousands of years of irrigation avoided salinization because salts were washed out of their soils. Most of these oases are located in the forelands of mountains, such as the oases that are dotted around the Tarim Basin in China, or the Ferghana Valley in Central Asia. The oases, located on slightly inclined lands, allow some of the irrigation

TABLE 2 | Soil salinity classes and effects on crop growth.

Soil salinity class	EC (dS m^{-1})	Effect on crops
Non-saline	0–2	No negative effects
Slightly saline	2–4	Reduction of yields of sensitive crops
Moderately saline	4–8	Yields of most crops are restricted
Strongly saline	8–16	Only tolerant crops produce good yields
Very strongly saline	>16	May be suitable for halophytes only

Note: EC–electrical conductivity. Modified from: https://www.salineagricultureworldwide.com/uploads/file_uploads/files/More%20info%20on%20Salinization-%20Salt%20Farm%20Foundation%202018(1).pdf.

TABLE 3 | Salinity thresholds (in µS/cm) for different irrigation water qualities.

Salinity hazard				References
Low	Medium	High	Very high	Soil Survey Staff (1954)
100–250	250–750	750–2,250	>2,250	
None	Some	Moderate	Severe	Follett and Soltanpour (2002); Bauder et al. (2011)
<750	750–1,500	1,500–3,000	3,000–7,500	
None	Slight to moderate		Severe	Ayers and Westcot (1985)
<700	700–3,000		>3,000	

water to percolate through the soil profile, transporting the salts into the groundwater, which drains into drainage ditches or directly into the river further downstream. This process removes drainage water from the oasis area. The associated salt load is often transported to a terminal lake or wetland downstream of the oasis, forming salt-swamps, salt-lakes, or saltpans, often associated with sodic salts. Where drainage water or groundwater from irrigated lands drains into rivers, the salt concentration in the river water increases downstream. In these events, the use of this water for irrigation in downstream croplands accelerates the risk of soil salinization and sodification (Forkutsa et al., 2009). Salinity thresholds of irrigation water of different qualities are detailed in **Table 3**.

Large parts of the world's irrigated lands are located in flat river plains, with very low or no inclination. Under such circumstances, it is difficult to enable drainage that adequately lowers the groundwater level and allows water percolation that sufficiently washes the salts from the soil profile. Under such topographical conditions, salinity is further aggravated by the gradually rising groundwater level. In these cases, capillary rise reaches the soil surface, depositing salts from the groundwater, and aggravating the severity of soil salinization and potential sodification (Forkutsa et al., 2009). Obviously, high volume irrigation methods, such as flood irrigation and furrow irrigation, aggravating the risk of soil salinization due to rising groundwater levels (Burt and Isbell, 2005). Yet, soil salinization may also be caused by water-saving irrigation methods, such as drip or sprinkler irrigation, which must be coupled with regular leaching (Stavi, 2020).

Secondary soil salinization and sodification have become urgent challenges in drylands, being a major driver of land degradation, adversely affecting agricultural production and food security. In the 1990s, 20% of irrigated lands across the world were affected by salinization, with Egypt and Iran having 33 and 29%, respectively, of their irrigated lands being affected. In absolute numbers, the largest areas of salt-affected irrigated lands were in India and China, with 7.0 and 6.7 million ha, corresponding respectively to 17 and 15% of their cropland area (Ghassemi et al., 1995). During the past 30 years, global soil salinization has been increasing, as shown by Ivushkin et al. (2019), who mapped global saline lands through remote sensing. According to this study, the world's saline land (including both primary and secondary salinization) was 915.5 million ha in 1986 and 1,069.3 million ha in 2016. Other studies showed that globally, 33% of all irrigated lands and 20% of all croplands are affected by salinization (Thenkabail, 2010; Shrivastava and Rajesh, 2015; Singh, 2015). It is expected that by 2050, 50% of the global cropland areas will be affected by salinization of varying degrees (Jamil et al., 2011), with the consequent extending of sodic soils as well (Qadir et al., 2007). A contemporary report on the global status of human-made salt-affected soils will soon be released (http://www.fao.org/global-soil-partnership/areas-of-work/soil-salinity/en/).

Salinization of agricultural land significantly reduces economic benefits, as shown by Welle and Mauter (2017) for California, where salinization reduced overall agricultural revenues by 7.9%. Globally, Wichelns and Qadir (2014) calculated the annual loss of agricultural crop yields to be 27.3 million USD, due to reduced productivity of salinized lands or due to outmigration of local populations from salt-affected regions.

Under secondary salinity conditions, plant nutrient management needs to counteract the nutrient imbalance caused by excess Na^+ and Cl^-, while not straining the ability of plants to uptake soil-water. In a review study, Hu and Schmidhalter (2005) showed that Na^+ competes with other cations, mainly NH_4^+ and K^+, for exchange places in the soil and for carrying molecules that transport nutrients through the plant cell membrane into the root. Therefore, the application of NH_4^+ and K^+ helps to address the plants' demand for nutrients, and reduces the absorption of Na^+. The application of Ca^{2+} and Mg^{2+} acts in a similar way, and also helps to preserve the soil structure, as Ca^{2+} and Mg^{2+} are more likely to displace Na^+ from the exchange places in the soil than K^+ or NH_4^+. With regards to anions, Cl^- competes with NO_3^-. Therefore, the application of NO_3^- helps satisfy the plants' demand for N, while reducing the absorption of Cl^-. One way or another, fertilizer must be applied carefully, as fertilizers are water-soluble salts themselves, and therefore may increase the osmotic potential of soil, negatively affecting the ability of plants to uptake soil-water. Furthermore, excess Ca^{2+} may immobilize phosphate (Hu and Schmidhalter, 2005).

Sodification deteriorates the soil structure, resulting in the decrease in infiltration of water and penetration of air into the soil profile (Qadir et al., 2007). This limits the availability of water and oxygen for plants, as well as for the entire soil-food web (European Communities, 2009). In addition, very high Na^+ contents may have negative effects on plant nutrition (Alexandre et al., 2018). Therefore, management of sodic lands needs to address these soil physical constraints.

TECHNIQUES FOR SOIL SALINITY MAPPING AND MONITORING

Calculating Soil Salinity

To plan effective restoration strategies and combat land degradation in the face of future climate change scenarios, the spatiotemporal distribution and likelihood of reoccurrence of salt-affected soils must be known (Hassani et al., 2020). Visual indications of soil salinization include a white salt crust or salt stains on the ground surface, fluffy soil surface, patchy or lack of seed germination, leaf burn and reduced plant vigor, naturally growing halophytes, worsening of affected areas after rainfalls, and waterlogging (Shahid et al., 2018a). Yet, mapping and monitoring of soil salinity at wide scales is challenging due to the dynamic feature of this property, which is highly variable in space and time (Aldabaa et al., 2015; Corwin and Scudiero, 2019).

Soil salinity is generally measured in laboratory through the analysis of aqueous extracts from disturbed soil samples (EC_e), or of saturated soil paste (EC_s) (Soil Survey Staff, 2014, Method 4F). Temperature has an effect on EC, because EC increases at approximately 1.9% per centigrade over the range of 15–35°C (Rhoades et al., 1989). Consequently, EC is expressed at a reference temperature of 25°C for purposes of comparison (Corwin and Lesch, 2003). An empiric equation to calculate the amount of soluble salts from electrical conductivity was developed by Marion and Babcock, 1976: (**Eq. 3**):

$$Log\,(\mathrm{TSS}) = 0.990 + 1.055 Log\,(EC) \tag{3}$$

where TSS is the total soluble salt concentration ($\mathrm{mmol{\cdot}l^{-1}}$), and EC is electrical conductivity ($\mathrm{mS{\cdot}cm^{-1}}$).

Several rapid and inexpensive methods for mapping and monitoring soil salinity have been developed during recent decades. The methods include both proximal and remote sensing technologies.

Proximal Soil Sensing

Proximal soil sensing (PSS) refers to field-based techniques that directly or indirectly measure soil properties, employing one or more sensors close to, or in a direct contact with the soil surface (Viscarra Rossel et al., 2010). The sensors may be hand-held, or pulled by vehicles. PSS can efficiently acquire high frequency soil information, allowing the assessment of soil spatial variability by geostatistical analysis. PSS technologies include contact electrodes designed for electrical resistivity (ER) methods, electromagnetic induction (EMI), mechanical sensors, gamma-ray spectroscopy, vis-NIR diffuse reflectance spectroscopy (VNIR-DRS), and laser induced breakdown spectroscopy (LIBS) (Viscarra Rossel et al., 2011). Among these technologies, the sensors based on soil electrical resistivity (or conductivity), namely ER and EMI sensors, are the most common for mapping soil salinity (Scudiero et al., 2013), texture (Doolittle and Brevik, 2014), moisture (Martini et al., 2017), and depth (Priori et al., 2013). ER techniques involve contact electrodes that directly inject electrical current into the soil, and measure the electrical potential drop due to the electrical resistivity of a determined volume of soil. ER measurements require at least four electrodes, two for injecting the current into the soil (current electrodes), and two for measuring the resulting potential difference (potential electrodes). Available ER apparatus usually have other electrodes that measure the potential differences at different depths. Both the penetration depth of the electrical current and the volume of the measured soil grow as the spacing between inter-electrodes increases (Samouelian et al., 2005).

Electrical conductivity of the bulk soil, also called apparent electrical conductivity (ECa), is the inverse of ER and can be measured by EMI without direct contact with the soil (Sudduth et al., 2003; Doolittle and Brevik, 2014). A transmitter coil located at one end of the EMI sensor produces an electromagnetic field, which induces circular eddy current loops in the soil; the magnitude of the loops is directly proportional to ECa. Such current loops produce a secondary electromagnetic field that is intercepted by the receiver coils of the instrument (Corwin and Yemoto, 2020). The differences in amplitude and phase between the secondary and the primary electromagnetic fields are related to the soil's properties, as well as to the coils' spacing and orientation, EM field frequency, and distance of the sensor from the ground surface (Doolittle and Brevik, 2014). Simultaneous assessments at multiple depths by commercial EMI and ER sensors make multi-layer inversion modeling possible (Mester et al., 2011). Mapping both the lateral and vertical variations of soil properties produce highly detailed 3D models (Huang et al., 2017b; Jiang et al., 2019). Another sensor generally used for monitoring soil ECa is the time domain reflectometer (TDR). The TDR, a stationary sensor generally installed within the soil at determined depths, monitors temporal variations of moisture content and ECa. The TDR measures the propagation time of an electromagnetic wave to travel down a soil probe and back, which is a function of the dielectric constant of the porous media (Noborio, 2001). Several TDR technologies for soil monitoring (including moisture, ECa, and temperature) are available.

ECa, or its inverse ER is a function of several soil properties, including soil texture, amount and pattern of voids, soil-water content, fluid electrical conductivity (solute concentration), and temperature (Corwin and Yemoto, 2020). In particular, the amount and type of clay minerals strongly affects ECa, because of their high ionic conductivity and high surface area. In general, 1:1 clay minerals (kaolinite, halloysite) have a relatively low surface area and lower ion exchange capacity than 2:1 interlayer clay minerals (illite, vermiculite, montmorillonite), which provide more spaces into which cations can be adsorbed (Kriaa et al., 2014). At the same time, the electrical current in soils is mainly based on the displacement of ions in pore-water, and is therefore greater with the presence of dissolved salts. Thus, the electrical current in soils strongly depends on the amount of water in the pores and on the concentration of soluble salts. Several papers reported that although direct causal relationships do not exist between exchangeable sodium and ECa, indirect and site-specific relationships are commonly found, and can be used to map SAR spatial variability (Amezketa, 2007; Heilig et al., 2011; Ganjegunte et al., 2014). High content of exchangeable sodium decreases soil aggregation and increases bulk density. Soil bulk density is another important feature influencing ECa measurements, because high soil porosity and aggregation increase the soil-air content, and thus lower the ECa (Corwin and Lesch, 2003; Heilig et al., 2011). Further studies are needed to assess the usefulness of mapping the ECa for identifying spatial variability of soil sodicity in a wider range of saline and non-saline soils (Amezketa, 2007; Ganjegunte et al., 2014).

VNIR-DRS is another promising proximal sensing technique for estimating EC and soil salinity. Aldabaa et al. (2015) compared several models for predicting EC of saline soils in Texas, United States, using VNIR-DRS spectra, and found good accuracy for all models, as shown by high residual prediction deviation (RPD; from 2.49 to 3.1). Similar performance indices of EC prediction in saline soils from VNIR-DRS spectra were found by Farifteh et al. (2007) and Peng et al. (2016). At the same time, other experiments studying the relationships between VNIR-DRS and EC_e in different soil types, from very low to medium EC and of different textures, reported much lower prediction performance, and site-specific relationships (Zornoza et al., 2008; Zovko et al., 2018). The spectral response of saline soils can be explained by vibrational absorption due to water molecules chemically bound as part of the crystal structure of the evaporate minerals (Howari et al., 2002). Sensitive bands for soils affected by Na_2SO_4 type salts were identified from 1,920 to 2,230 nm, for NaCl type salts between 1,970 and 2,450 nm, and for Na_2CO_3 type salts from 350 to 400 nm (Wang et al., 2012). The results reported from these studies demonstrated that VNIR-DRS can determine not only EC, but also the type of salt and mineralogy in saline soils and crusts (Howari et al., 2002). Regarding the use of VNIR-DRS for ESP estimation, a recent paper by Zhao et al. (2021) on cotton plantations in Australia, reported poor accuracy of predictive models (RPD <1.4), and recommend not using this technique for ESP prediction. The same results of low accuracy of VNIR-DRS prediction for ESP estimation were reported by Chang et al. (2001), as well as by Zornoza et al. (2008).

Remote Sensing Methods

Various remote sensing data acquisition techniques have been applied for identifying and monitoring soil salinity in salt-affected soils. Indeed, remotely sensed data can potentially be used for low-cost mapping and temporal monitoring of topsoil salinity across extensive areas. Therefore, the topic of remote sensing applications for soil salinity assessment has received tremendous attention over the last decade. The availability of new free multispectral images, such as Landsat 8, Sentinel 2 and Hyperion, promote this trend.

The earliest review of remote sensing of salt-affected soils, by Mougenot et al. (1993), included the detection of calcite and various salts on the ground surface. Successive thematic reviews on remote sensing methods for monitoring salt-affected soils were published by Metternicht and Zinck (2003), Farifteh et al. (2006), Ben-Dor et al. (2008), Corwin and Scudiero (2019), and Mahajan et al. (2020).

In salt-affected areas, soil moisture is very low during the dry season, and salts may precipitate around the soil surface, creating efflorescence, crusts, puffy structures, and cracks. These surface features in bare soils can be easily detected by aerial and satellite images, because they accentuate the light color and affect the roughness of topsoil (Metternicht and Zinck, 2003). Several authors have determined key spectral absorption bands of salt-affected soils in the range of visible (Vis: 450–740 nm), near infrared (NIR: 780–1,000 nm), and short-wave infrared (1,550–1,650 nm and 2,100–2,300 nm) wavelengths (Khan et al., 2005; Douaoui et al., 2006; Nield et al., 2007; El Harti et al., 2016; Wang et al., 2019). These VNIR-SWIR absorption peaks are mainly associated with the internal vibration status of anions or water molecules, which are trapped in the crystal structure of the salts (Farifteh et al., 2008). The major spectral indices commonly used to monitor salt-affected soils are summarized in **Table 4**. The use of remote sensing for the assessment of soil sodicity is less common than for assessing soil salinity. A relatively rare example for such an assessment is a study by Bai et al. (2016), who reported good correlation between Landsat 8-OLI bands 1, 2, 3, and 4 and soil pH in alkali soils of northeastern China.

Often, topsoil figures of salinity or sodicity can be blurred by vegetation, which creates spectral confusions with the salt reflectance properties. Moreover, salt that does not form clear surface features, or salinity/sodicity in deeper soil layers, cannot be directly differentiated by remote sensing. In these cases, indirect drivers, based on the response of vegetation to salinity stress, are often used (Metternicht and Zinck, 2003). The negative impact of soil salinity/sodicity on plant growth was monitored by several vegetation indices, such as the normalized difference vegetation index (NDVI) (Scudiero et al., 2013; Gorji et al., 2017), the enhanced vegetation index (EVI) (Lobell et al., 2010), and the soil-adjusted vegetation index (SAVI) (Allbed et al., 2014). Scudiero et al. (2014) proposed a specific empirical index for plant salinity stress, named canopy response salinity index (CRSI), which includes the RGB (red-green-blue, the Vis band) and the NIR bands of Landsat 7. Vegetation indices show better performance when corrected using soil indices, a finding that demonstrates both the effect of soil on the spectral response of the surface and the presence of a discontinuous or weakly-developed cover (Metternicht and Zinck, 2003). For this reason, multivariate (PCA, PLS) and computational methods (Support Vector Machine, Artificial Neural Networks) for predicting soil salinity from several remote sensing indices, both pedological and vegetal, are currently considered to be the best approach for salinity mapping (Scudiero et al., 2014; Fan et al., 2015; Taghadosi et al., 2019; Wang et al., 2019; Mahajan et al., 2020).

Thermal infrared images, either obtained from satellites or airborne sensors, are also used for estimating both moisture and salt content of soil, although moderate results were obtained when using the thermal range spectrum alone for soil salinity prediction (Metternicht and Zinck, 2003). At the same time, it was demonstrated that the inclusion of thermal infrared bands into Vis-NIR bands can strongly improve the prediction of soil salinity (Verma et al., 1994; Aldabaa et al., 2015). Particularly, it was shown that thermal bands help to distinguish Na^+ from other salts in alkaline soils (Farifteh et al., 2006).

Satellite microwaves, both passive and active, are also used for mapping and monitoring salt-affected lands (Bell et al., 2001; Wu et al., 2018). The high penetration of microwaves through clouds and its sensitivity to dielectric properties make this possible (Gong et al., 2013). In general, microwave bands at lower frequencies, especially the lowest L-bands (1.25 GHz), are considered suitable for detecting salinity under different settings, although soil moisture strongly affects the dielectric properties and the microwave images (Metternicht and Zinck, 2003). Details regarding microwave technology for monitoring soil salinity are available in Gong et al. (2013) and in Wu (2019).

TABLE 4 | Spectral indices used for soil salinity mapping and monitoring.

Rs indices	Satellite images	Equation	References
Soil index 1	SPOT2	$SI = \sqrt{G \times R}$ [(1)]	Douaoui et al. (2006); Gorji et al. (2017)
Soil index 2	SPOT2	$S2 = \sqrt{G^2 + R^2 + NIR^2}$ [(1)]	Douaoui et al. (2006)
Soil index 3	SPOT2	$S3 = \sqrt{G^2 + R^2}$ [(1)]	Douaoui et al. (2006)
Salinity index	IKONOS	$S1_T = \frac{R}{NIR} \times 100$ [(2)]	Allbed et al. (2014)
Brightness index	IKONOS	$BI = \sqrt{R^2 \times NIR^2}$ [(2)]	Allbed et al. (2014)
Canopy response salinity index	Landsat 7	$CRSI = \sqrt{\frac{(NIR \times R)-(G \times B)}{(NIR \times R)+(G \times B)}}$ [(3)]	Scudiero et al. (2014)
Natric index	Landsat 7	$NI = \frac{SWIR-NIR}{SWIR+NIR}$ [(3)]	Nield et al. (2007)
Salinity index	IRS-1B	$SI = \sqrt{B1 \times B3}$ [(4)]	Khan et al. (2005); Abbas et al. (2013)
Brightness index	IRS-1B	$BI = \sqrt{B3^2 + B4^2}$ [(4)]	Khan et al. (2005)
Normalized differential salinity index	IRS-1B	$NDSI = \frac{B3-B4}{B3+B4}$ [(4)]	Khan et al. (2005); Abbas et al. (2013)
OLI salinity index	Landsat 8 OLI	$OLI_{SI} = (CB^2 \times 50) - (B + G + R)$ [(5)]	El Harti et al. (2016)
Three bands index	Sentinel 2	$TBI = \frac{(SWIR2-G)}{(G-SWIR1)}$ [(6)]	Wang et al. (2019)

Notes: [(1)] G: green band (510–590 nm), R: red band (610–680 nm), NIR: near infrared band (790–890 nm); [(2)] R: red (630–690 nm), NIR: near infrared (760–900 nm); [(3)] B: blue (450–520 nm), G: green (520–600 nm), R: red (630–690 nm), NIR (775–900 nm), SWIR (1,550–1750 nm); [(4)] B1: blue band (450–520 nm); B3: red band (620–680 nm); B4: NIR band (770–860 nm); [(5)] CB: coastal blue band (430–450 nm), B: blue band (450–510 nm), G: green band (530–590 nm), R: red band (640–670 nm); [(6)] G: green band (543–578 nm), SWIR1 band (1,565–1,655 nm), SWIR2 (2,100–2,280 nm).

PREVENTION, MITIGATION, AND RESTORATION OF SALINE AND SODIC SOILS

Site Selection

Judicious site selection of new agricultural lands minimizes the risk of water logging and soil salinization. First, coastal sites that are highly prone to seawater intrusion, as well as inland sites prone to saltwater intrusion (see: Duan, 2016), should be avoided. Second, sites with a shallow aquifer, with a risk of secondary salinization by capillary force of groundwater (see: Devkota et al., 2015), should also be avoided. Third, challenging physical settings including certain climatic, lithologic, topographic, and pedologic conditions that increase the risk of water logging and prevent effective drainage of excess water from the target land unit (see: Gebrehiwot, 2018) should be excluded.

However, challenging terrains are often the only available option, necessitating their use for agriculture. In these events, special attention should be paid for establishing artificial drainage systems, which must effectively control the groundwater table, lowering the salt-affected soil layer far below the crops' root zone. According to the physical settings, agro-technical level of development, and available funds, this could be implemented using open canal or tunnel systems around the fields' circumference, or by installing a subsurface drainage system across the field (Cuevas et al., 2019). Both circumferential and subsurface drainage systems can be effective in minimizing the fields' risk of soil salinization. However, in sodic lands, drainage systems may not effectively remove the Na^+ from the soil (NDSU Extension Service–Subsurface tile drainage). Similarly, in saline-sodic lands, drainage will cause the preferential removal of Ca^{2+} and Mg^{2+}, making Na^+ dominant in the soil (NDSU Extension Service, 2014—Saline and sodic soils). Regardless, to prevent potential damage further downstream, effluent water flow should be constrained in retention ponds, where it can be monitored and treated before being released to the environment or reused for agriculture.

Means for prevention, mitigation, and restoration of saline and sodic soils are discussed in the following sub-sections, with an emphasis on aspects related to both recharge (where net movement of water is downward) and discharge (where net movement of water is upward) processes, and their consequences to the upper soil layers and underground aquifer.

Salt Flushing and Leaching

Apparently, the uppermost salt-affected soil layer can be mechanically removed using heavy tractor-scraper machineries. However, this practice is rather expensive, and its efficacy for reducing soil salinity is usually short-term (Cuevas et al., 2019). At the same time, flushing with sufficient volume of high-quality water is suitable for heavy textured soils with salt crusts, where the flushed salts from the soil surface enter the drainage system and are thus removed from the target land (Shahid et al., 2018b).

Regardless, saline soils can be restored by leaching the salts out of the soil profile through excess irrigation, coupled with a drainage system deep enough to prevent capillary rise of the water back into the root zone or soil surface (Cuevas et al., 2019). For example, in west-central Kazakhstan, pre-season technical inundation lasting several days is common in furrow irrigation agricultural systems. Also, throughout the cropping season, irrigation water is applied beyond the crops' requirements, leaching the salts from the rooting zone (Devkota et al., 2015). This excess water, referred to as the leaching fraction, must increase along the growing season, accordingly with the growth of the crops' root system (Cuevas et al., 2019). Often, the economic and environmental costs of intentional flushing and leaching of salts are rather high, questioning their viability and sustainability over the long-run. Yet, the partial or full subsidy of agricultural water in many countries, alongside ignorance regarding the environmental footprint of agriculture at all levels (i.e., from the farm level to the national and international levels), allow these practices to persist in extensive parts of the world. Regardless, as shown by Driessen et al. (2001), frequent leaching may remove Ca^{2+} and Mg^{2+} from the soil profile, increasing SAR if Na^+ is present, and consequently increasing sodicity and forming of solonetz.

Chemical Remediation

Saline soils cannot be restored by chemical methods (Shahid et al., 2018b). At the same time, sodic soils can be restored by applying Ca^{2+} to displace the excess Na^+ from the exchange complex. Soluble sources of Ca^{2+} are released to the soil solution, replacing the excess Na^+ in the soil's exchange complex, thus soluble Na^+ becomes available for leaching. Because lime is only slightly soluble in water, it cannot provide enough Ca^{2+} to displace Na^+. Therefore, the most widely used Ca^{2+} sources are highly soluble minerals such as gypsum ($CaSO_4 \cdot 2H_2O$) (Qadir et al., 2007) and gypsum-like byproducts–e.g., phosphogypsum (a byproduct of phosphoric acid manufacturing), coalgypsum (a byproduct of coal power plants), and lactogypsum (a byproduct of lactic acid and lactate manufacturing) (Amezketa et al., 2005). Following the application of such amendments, and after the Na^+ is solubilised, the soluble salts must be leached (Paz et al., 2020) by excess irrigation. The rate of gypsum application varies greatly, and depends on the magnitude of sodification, and the prevailing biophysical conditions, of which most important are the prevailing climate, parent material, soil type, quality of irrigation and underground water, and the coming crop variety (Amezketa et al., 2005; Qadir et al., 2007; Paz et al., 2020). Chemical remediation of saline-sodic soils is similar to that of sodic soils, followed by salinity remediation practices through improving soil drainage, lowering the water-table level, and leaching of salts (other than Na^+) (NDSU Extension Service, 2014—Saline and sodic soils).

Organic and Microbial Remediation

Application of organic amendments, such as livestock manures and composts, increases the soil organic carbon concentration and improves the soil's macro-aggregation processes and structure formation. These processes improve soil permeability, increasing the leaching capacity of salts from the rhizosphere, and simultaneously decrease evaporation rates, reducing salt accumulation in the surface soil layer (Lakhdar et al., 2009). In addition to livestock-derived organic amendments, this practice can also utilize waste materials, such as municipal solid wastes, transforming these environmental burdens into highly valorized resources (Lakhdar et al., 2009). Yet, special attention should be paid when using highly-saline solid wastes (or manures), which their application may aggravate soil salinity. In sodic lands, the application of organic amendments combined with gypsum was reported to better alleviate soil sodicity than the application of gypsum alone (Gonçalo et al., 2020). The low cost of organic amendments makes them widely utilized for alleviation of soil salinity and sodicity around the world. In addition to these "regular" organic additives, specific, designated organic soil conditioners may also be applied, despite being comparatively expensive. Similar to regular organic amendments, these conditioners are more effective in sodic soils when combined with gypsum (Cuevas et al., 2019).

Tillage Schemes

Soil tillage is a regular practice in croplands, aimed at seedbed preparation. Yet, the intensity of tillage depends not only on the prevailing physical conditions and cropping history, but also on traditions, viewpoints, and beliefs. Alongside the benefits of conventional tillage, such as soil loosening and aeration, several unintentional disadvantages are known, such as the damage to soil structure, increased oxidation of soil organic carbon, breakdown of macro-aggregates, decreased hydraulic conductivity, and increased soil erodibility (Blanco-Canqui and Ruis, 2018). The degraded permeability of the soil lowers its salt leaching efficacy, leading to salt accumulation in the rhizosphere. In recent decades, tremendous advancements in the conservation agricultural sector have led to the development of reduced tillage practices. It was reported that minimum tillage, and particularly no-till systems, improve leaching and negate soil salinity (Botha, 2013). Alongside with the tillage system, the management of crop residue–i.e., the removal vs retention of crop residue–has a tremendous impact on the susceptibility of soil to salinization and sodification (Bezborodov et al., 2010). It was demonstrated that surface mulching (e.g., by the crop residue) decreases evaporation rates, consequently lowering soil salinity (Stavi, 2020) and sodicity (Verhulst et al., 2009). Ground surface mulching is especially relevant in drylands, and where saline water is regularly used for irrigation (Stavi, 2020).

Irrigation Schemes

To minimize aquifer recharge with excess salts and thus limit soil salinization (Finlayson et al., 2010), flood and furrow irrigation should be negated wherever possible (Cuevas et al., 2019). Yet, under certain conditions, technical inundation is required to reduce soil salinity before or during the cropping season (Devkota et al., 2015). Similarly, in non-flooding irrigation methods that use routine, frequent high dose irrigation to leach salts from the rhizosphere increase the risk of groundwater salinization (Devkota et al., 2015). This risk is particularly relevant in drylands, where the risk of soil salinization is high even when high-quality water is used for irrigation (Cuevas et al., 2019).

Conversely, water-saving irrigation methods, such as micro-irrigation systems, and specifically drip irrigation systems, can substantially decrease the risk of groundwater recharge, while effectively leaching salts from the dripper-formed "wet bulbs", from which the plant roots extract water (Stavi, 2020). Yet, depending on the quality of irrigation-water, as well as on associated agronomic practices, soil salinization and sodification can still occur even under drip irrigation systems (Liu et al., 2020; Stavi, 2020). Recent developments in precision irrigation technologies significantly increase the crop's water-use efficiency, and may decrease agriculture's environmental footprint. For example, the irrigation-on-demand (IOD) technology is based on drippers with built-in tensiometers that are directly activated by the crop's roots. It provides the plant with precise water requirements, specified to its variety and phenological stage (Tal, 2016).

Fertilizing and Manuring Schemes

Use of chemical fertilizer may exacerbate rhizosphere salinization. Moreover, excess fertilizing can increase nutrient recharge into the groundwater (Finlayson et al., 2010). This is not only wasteful in terms of fertilizing efficiency, it also increases the salinity of aquifers (Finlayson et al., 2010), and contaminates them with hazardous materials (Cuevas et al., 2019). Specifically, drinking water from aquifers contaminated with residual nitrate (NO_3^-) may cause infant methemoglobinemia, certain cancers, and neural tube defects (Ward et al., 2018). Similar effects were reported for organic nutrient management systems, where the use of livestock manure salinizes the soil's upper layer and increases the leaching of salts and contaminants to the aquifers (Manitoba, 2013). Specifically, manuring may enrich the underground water with pathogenic microorganisms and viruses, imposing a high risk to human health (Ward et al., 2018).

In addition to harming aquifers, runoff processes taking place on either chemically or organically fertilized agricultural lands, may remove excess fertilizers and displace them into surface waterbodies, increasing their salinity and enriching them with nutrients. This may cause eutrophication by algae and degradation of waterbodies (Withers et al., 2014). Recent advances in precision fertilizer systems may increase fertilizer use-efficiency, while reducing the risk of salinization and contamination of both underground and surface waterbodies. Specifically, fertigation technologies, in which dissolved fertilizers are distributed through drip irrigation systems, were shown to optimize fertilizer use-efficiency and reduce agriculture's environmental footprint (Azad et al., 2020).

Phytoremediation

In sodic lands, crops can contribute to soil restoration through two mechanisms. Plant roots release CO_2 and protons into the soil, which both increase acidity in the ambient soil. The CO_2 is released by root respiration and bacterial decomposition of root exudates, while protons are released during cation uptake by plant roots. The increased acidity partially dissolves Ca-bearing soil minerals, such as lime, and therefore provides Ca^{2+}, which displaces Na^+ from the exchange complex.

Overall, crops with a high tolerance to both drought and salinity should be selected. Specifically, deep-root crop varieties that extract water from depths of several meters (Cuevas et al., 2019) reduce the risk of groundwater discharge and the upward migration of salts to the ground surface (Finlayson et al., 2010). Among grain crops, barley (*Hordeum vulgare* L.) is highly tolerant to dry conditions and soil salinity. Further, some relatively new barley cultivars proved to be exceptionally salt tolerant (Katerji et al., 2006). Among vegetable crops, potato (*Solanum tuberosum* L.), carrot (*Daucus carota* subsp. Sativus (Hoffm.) Schubl. & G. Martens), onion (*Allium cepa* L.), lettuce (*Lactuca sativa* L.), and cabbage (*Brassica oleracea* L.) were reported to have a comparatively high tolerance to moderately saline conditions (de Vos et al., 2016). Among forage crops, alfalfa (*Medicago sativa* L.) is known to tolerate soil salinity well, with some specific cultivars having exceptionally high tolerance (Scasta et al., 2012). Kallar grass (Leptochloa fusca (L.) Kunth.), a fodder crop, has been widely used in salt-affected and water-logged lands in Pakistan, and has proved to effectively ameliorate saline, sodic, and saline-sodic soils (Nadeem et al., 2017).

Regardless, crop rotation has been reported to effectively prevent soil salinization, where crops with high tolerance to salinity are planted during the dry period, while more sensitive crops are planted in the wet season, when rains wash the salts from the upper soil layers to the subsoil layers. Other agronomic practices, such as cover cropping and green manuring–where certain herbaceous crops are grown in the off-season or in rotation with main crops, and can either be left on the ground (cover crops) or inverted into the soil (through inversion tillage, for green manures)—were reported to decrease soil salinity while improving the bio-physio-chemical quality of soil (Cuevas et al., 2019). Among these herbaceous crops, salt-tolerant legumes, such as alfalfa and riverhemp (*Sesbania* spp.) are rather common for increasing the system's nutrient availability and decreasing the soil's salinity and solidity levels. Additionally, such legumes can be used as feed for livestock, either through on-site grazing or as harvested forage (Qadir et al., 2007). Overall, phytoremediation is a comparatively cheap method of land restoration. It is also environmentally friendly, as it avoids the use of synthetic products and improves carbon sequestration (Paz et al., 2020). Major practices of phytoremediation are summarized in **Table 5A**.

Afforestation and Reforestation

In addition to the "regular" agronomic means and practices, planting tree and shrub species with deep root systems and high water demand proved to be effective in controlling the discharge of underground water, limiting salinization or sodification of the upper soil layers, and subsequently restoring saline, sodic, and saline-sodic soils (Qadir et al., 2007; Finlayson et al., 2010). These trees and shrubs should be planted following certain procedures that facilitate them to overcome the upper and most saline or sodic layers. For example, furrow system, where the trees and shrubs are planted in the furrows' salt-washed micro-habitats, may be suitable. Alternatively, sub-surface planting, where the trees and shrubs are planted in deep holes or trenches whose depth exceeds the most severely salt-affected surface layer, may be necessary (Dagar et al., 2001; Banyal et al., 2017).

In west-central Uzbekistan, afforestation of saline croplands with Euphrates poplar (*Populus euphratica* Oliv.) and Siberian elm (*Ulmus pumila* L.) proved to be highly effective in reducing soil salinity, while improving the soil's aggregate stability and organic carbon pools (Hbirkou et al., 2011). In northern India, mesquite (*Prosopis juliflora* (Sw.) DC.), tamarisk articulata (*Tamarix articulate* Vahl.), and goma arábiga (*Acacia nilotica* (L.) Delile) were reported to be highly effective in restoring sodic soils. At the same time, forest red gum (*Eucalyptus tereticornis* Sm.), Indian rosewood (*Dalbergia sissoo* Roxb.), Manila tamarind (*Pithecellobium dulce* (Roxb.) Benth.), arjuna (*Terminalia arjuna* (Roxb.) Wight & Arn.), sausage tree (*Kigelia pinnata* (Jacq.) DC.),

TABLE 5 | Major practices of phytoremediation **(A)**, afforestation and reforestation **(B)**, and agroforestry and silvopasture **(C)**, for soil salinity and sodicity prevention and restoration.

Practice	Country/region	Target soils	Plant species	References
A. Phytoremediation	Australia	Saline soils	Grass species, such as wheat grass (*Thinopyrum ponticum* Podp.Z.W. Liu + RRC Wang cv. Largo), barley, puccinellia (*Puccinellia maritima* jacq. Parl.), salt water couch (*Paspalum vaginatum* Sw.), Russian wildrye (*Psathyrostachys juncea* Fisch. Nevski), western wheatgrass (*Pascopyrum smithii* Rydb. A.Love), bulbous canary-grass (*Phalaris aquatic* L.), reed canary grass (*Phalaris arundinacaea* L.), Kallar grass, and Rhodes grass (Chloris gayana Kunth)	Rogers et al. (2006)
	Australia	Saline soils	Legume species, such as alfalfa, lucerne (*Medicago sativa* L.), burr medic (*M. polymorpha* L.), subterranean clover (*Trifolium subterraneum* L.), balansa clover (T. michelianum L.), strawberry clover (*T. fragiferum* L.), Persian clover (*T. resupinatum* L.), Egyptian clover (*T. alexandrinum* L.)	Rogers et al. (2006)
	Not specified	Saline soils	Barley	Katerji et al. (2006)
	Not specified	Sodic and saline-sodic soils	Forage crops, such as Bermuda grass, Kallar grass, riverhemp, and sesbania	Qadir et al. (2007)
	Not specified	Saline soils	Forage crops, such as alfalfa	Scasta et al. (2012)
	India	Sodic soils	Forage crops, such as Kallar grass, Rhodes grass, Para grass (*Brachiaria mutica* (Forssk.) Stapf), blue panicgrass (*Panicum antidotale* Retz.), buffalo grass (*Panicum maximum* Jacq.), land grass (*Panicum laevifolium* Hack.), and vetiver grass (*Vetiveria zizanioides* Roberty, 1960)	Dagar (2014)
	India	Saline soils	Forage crops, such as Kallar grass, Sporobolus (*Sporobolus helvolus* (Trin.) T.Durand & Schinz), mangrove grass (*Aeluropus lagopoides* L. Thwaites), *Panicum* spp., *Atriplex* spp., and lovegrass (*Eragrostis*) spp.	Dagar (2014)
	Not specified	Moderately saline soils	Vegetable crops, such as potato, carrot, onion, lettuce, and cabbage	de Vos et al. (2016)
	Pakistan	Saline, sodic, and saline-sodic soils	Forage crops, such as Kallar grass	Nadeem et al. (2017)
B. Afforestation and reforestation	Australia	Saline soils	Shrub species, such as river saltbush (*Atriplex amnicola* L.), oldman saltbush (*A. nummularia* Lindl.), creeping saltbush (*A. semibaccata* R.Br.), grey saltbush (*A. cinerea* Poir.), and small-leaved bluebush (*Maireana brevifolia* R.Br. Paul G. Wilson)	Rogers et al. (2006)
	Not specified	Sodic and saline-sodic soils	Shrub and tree species, such as saltbush, mesquite, and tamarisk	Qadir et al. (2007)
	Uzbekistan	Saline soils	Euphrates poplar and Siberian elm	Hbirkou et al. (2011)
	India	Saline and sodic soils	Arjuna, neem, and lebbeck	Singh et al. (2012)
	India	Sodic soils	Mesquite, tamarisk articulate, goma arábiga, arjuna, and forest red gum	Dagar (2014)
	India	Saline soils	Mesquite, goma arábiga, Parkinsonia, umbrella thorn (*Vachellia tortilis* (previously *Acacia tortilis*) Forssk. Galasso & Banfi), sweet acacia (*Acacia farnesiana* L.) Wight et Arn.), swamp oak (*Casuarina glauca* Sieber), river red gum (*Eucalyptus camaldulensis* Dehnh), Indian beech (*Millettia pinnata*, syn. *Pongamia pinnata* (L.) Panigrahi), Bada Peelu (*Salvadora oleoides* Decne.), toothbruch tree (*Salvadora persica* L.), *Tamarix* spp., and *Atriplex* spp.	Dagar (2014)
	Mediterranean basin	Sodic and saline-sodic soils	Saltbush	Walker et al. (2014)
	Uzbekistan	Saline soils	Russian olive (*Elaeagnus angustifolia*L.), Euphrates poplar, and Siberian elm	Vargas et al. (2018)
	Russia	Saline and sodic soils	Planting common oak (*Quercus robur* L.), green ash (*Fraxinus pennsylvanica* Marshall), Siberian elm, and ash-leaved maple (*Acer negundo* L.)	Vargas et al. (2018)
C. Agroforestry and silvopasture	India	Highly saline soils	Mesquite trees in combination Kallar grass	Singh (1995)
	India	Moderately saline soils	Mesquite trees in combination with Persian clover and Egyptian clover	Singh (1995)
	India	Sodic soils	Goma arábiga, north Indian rosewood, and mesquite trees along with grass species of Halfa grass and poolongi (Sporobolus marginatus)	Kaur et al. (2002)
	India	Sodic soils	Different combinations of the tree species of mesquite, tamarisk articulate, goma arábiga, arjuna, and forest red gum, with the grass species of Kallar grass, Rhodes grass, Para grass, blue panicgrass, buffalo grass, land grass, and vetiver grass	Dagar (2014)

(Continued on following page)

TABLE 5 | (*Continued*) Major practices of phytoremediation **(A)**, afforestation and reforestation **(B)**, and agroforestry and silvopasture **(C)**, for soil salinity and sodicity prevention and restoration.

Practice	Country/region	Target soils	Plant species	References
	India	Saline soils	Fruit trees, such as bael, aonla, and karonda, in combination with barley and cluster bean	Sharma et al. (2014)
	India	Sodic soils	Mesquite trees in combination with Kallar grass and goma arábiga; north Indian rosewood trees in combination with Halfa grass	Sharma et al. (2014)

Parkinsonia (*Parkinsonia aculeate* L.), and grey-leaved saucer berry (*Cordia rothii* Roem. & Schult.) showed good survival but relatively low productivity (Dagar et al., 2001). In another northern Indian study, mixed reforested lands–comprising arjuna, neem (*Azadirachta indica* A. Juss. 1830), and lebbeck (*Albizia lebbeck* (L.) Benth.), among other tree species, were reported to restore sodic croplands, with specific improvement of the physio-chemical, biological, and bio-chemical properties of soil (Singh et al., 2012). Halophyte shrub species that are prevalent across the Mediterranean basin, such as saltbush (*Atriplex halimus* L.), are able to remove salts through epidermal glands on both surfaces of their leaves. This shrub species is widely utilized as feed for livestock animals (Walker et al., 2014). Major practices of afforestation and reforestation are summarized in **Table 5B**.

Further, in addition to regular afforestation and reforestation practices, agroforestry and silvopasture systems–in which trees or shrubs are planted at certain spatial patterns, and combined with grain, vegetable, or forage crops–are known to restore saline, sodic, and saline-sodic soils, while providing food, feed, and fiber. These systems provide food, feed, construction materials, fuelwood, and many other products, which can be utilized either for domestic consumption or for selling as a source of income (Dagar, 2014). In saline soils of northern India, fruit trees such as bael (*Aegle marmelos* (L.) Corrêa), aonla (*Phyllanthus emblica* L.), and karonda (*Carissa carandas* L.) are planted in combination with barley and cluster bean (*Cyamopsis tetragonoloba* (L.) Taub.). In sodic soils of the same region, mesquite trees are planted in combination with Kallar grass and goma arábiga, and north Indian rosewood trees are planted in combination with Halfa grass (*Desmostachya bipinnata* (L.) Stapf) (Sharma et al., 2014). Major practices of agroforestry and silvopasture systems are summarized in **Table 5C**.

CONCLUSION

Soil salinization and sodification affect extensive areas worldwide, but mostly characterize drylands. While primary salinity or sodicity occurs in a combination of certain physical settings, secondary salinity or sodicity is predominantly determined by agricultural activities, specifically irrigation. Climatic change exacerbates this hazard, and active interventions are required to safeguard dryland soils. Judicious selection of sites for the establishment of croplands, alongside with the use of water-saving irrigation technologies, can reduce the risk of waterlogging and soil salinization and sodification processes. In saline soils, routine measures should be taken to allow effective drainage of excess water, alongside with flushing and leaching of salts. In sodic and saline-sodic soils, chemical remediation measures, and specifically the use of gypsum-based soil amendments, could be effective in restoring affected lands. Further, the use of conservation agricultural practices, such as reduced tillage systems, crop residue management, manuring/composting, precision fertilizer schemes, and proper crop selection can considerably reduce the risk of soil salinity and sodicity. Specifically, the use of advanced forestry and agroforestry systems has a tremendous potential in restoring saline and sodic soils.

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct, and intellectual contribution to the work and approved it for publication.

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