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REVIEW

Climate change impacts on soil salinity in agricultural areas

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Abstract

Changes in climate patterns are dramatically influencing some agricultural areas. Arid, semi-arid and coastal agricultural areas are especially vulnerable to climate change impacts on soil salinity. Inventorying and monitoring climate change impacts on salinity are crucial to evaluate the extent of the problem, to recognize trends and to formulate irrigation and crop management strategies that will maintain the agricultural productivity of these areas. Over the past three decades, Corwin and colleagues at the U.S. Salinity Laboratory (USSL) have developed proximal sensor and remote imagery methodologies for assessing soil salinity at multiple scales. The objective of this paper is to evaluate the impact climate change has had on selected agricultural areas experiencing weather pattern changes, with a focus on the use of proximal and satellite sensors to assess salinity development. Evidence presented in case studies for California's San Joaquin Valley (SJV) and Minnesota's Red River Valley (RRV) demonstrates the utility of these sensor approaches in assessing soil salinity changes due to changes in weather patterns. Agricultural areas are discussed where changes in weather patterns have increased root-zone soil salinity, particularly in areas with shallow water tables (SJV and RRV), coastal areas with seawater intrusion (e.g., Bangladesh and the Gaza Strip) and water-scarce areas potentially relying on degraded groundwater as an irrigation source (SJV and Murray-Darling River Basin). Trends in salinization due to climate change indicate that the infrastructure and protocols to monitor soil salinity from field to regional to national to global scales are needed.

Highlights

- Climate change will have a negative impact on agriculture, particularly in arid regions.
- Proximal/remote sensors are useful to assess climate change impact on soil salinity across scales.
- Salt-water intrusion, shallow water tables and degraded water reuse will increase soil salinity.
- Infrastructure and protocols to monitor soil salinity across multiple scales are needed.

KEYWORDS

airborne sensor, apparent soil electrical conductivity, electrical resistivity, electromagnetic induction, proximal sensor, remote sensing, satellite sensors, soil salinity mapping, soil spatial variability

1 | INTRODUCTION

Climate change is defined by: high atmospheric CO_2 (i.e., >400 ppm); increasing air temperatures; abrupt and significant changes in daily, seasonal and between-year temperature; changes in wet and dry cycles; intensive rainfall events; extended drought periods; extreme frost; and hot, dry spells that elevate fire hazard levels. Changes in climatic patterns are expected to significantly impact terrestrial systems, soil properties, surface waters and stream flows (Patterson, Lutz, & Doyle, 2013). High levels of uncertainty exist in climate model projections, particularly at regional and local scales, but climate change models do agree on some basic global trends. The models indicate that climate change is expected to affect primarily precipitation, potential evapotranspiration (ET) and temperature. Changes will occur in irradiance, ultraviolet irradiance and evaporative demand, as well as on the secondary factor ozone (Yeo, 1999). Climate change models suggest (a) an increase in the average global temperature, (b) altered weather patterns with shifts in rainfall patterns, and (c) an increase in climatic extremes within localized areas (Yeo, 1999). Increased global temperatures will raise ocean levels as polar ice caps melt and will bring more extreme weather conditions. Droughts and flooding are expected to increase in frequency and intensity. Hot dry areas are expected to become hotter and drier, some wet areas wetter, and isolated cold areas will be colder.

There is an expectation that climate change will bring increased frequency of extreme weather events around the globe, with unusually high rainfall events leading to floods and low precipitation, higher temperatures, and higher potential ET resulting in longer, harder and more frequent droughts. The National Center for Atmospheric Research released a collection of drought severity maps projecting drought levels using the Palmer Drought Severity Index for the time period 2000–2099. The drought maps show that much of the Western Hemisphere and large parts of Eurasia, Africa and Australia will experience extreme drought, whereas higher-latitude regions from Alaska to Scandinavia are likely to become wetter and experience flooding.

Evidence of the impact of the change on climate patterns is found worldwide. For instance, the USA experienced an increase in moderate to severe levels of drought, particularly in the southwest, but other areas of the USA are not exempt (e.g., the midwest and southeast). Arguably, the most notable in the USA from a public-awareness perspective, due to its impact on agricultural productivity, was the California drought of 2011–2015. This caused drastic reductions in irrigation water allocations to farmers in the agriculturally productive San Joaquin Valley (SJV) and heightened water conservation measures in urban areas. Other recent worldwide droughts include: a 1 in a 1000-year drought in Australia (e.g., lower portion of the Murray-Darling River Basin), which began in 1995 and continued until 2009; Spain's drought in Catalonia; northern India's drought in the first decade of the new millennium; and droughts in northern China, Syria and southeastern Brazil. Recent floods include: Queensland, Australia; Tennessee, Arkansas, Texas and Wisconsin in the USA; Pakistan; and India. Even though there is no short-term extreme weather event that can be conclusively attributed to climate change, there is a statistical record of these events showing that they clearly occur with increased frequency and/or intensity (Dai, 2011).

Ironically, some of the most crop-productive areas of the world occur in water-scarce regions, such as the arid southwestern USA (e.g., California's San Joaquin and Imperial-Coachella Valleys) and other arid regions of the world, including the Middle East, the Hai He, Huang He and Yangtze basins in China, and along the Nile River in Egypt and Sudan. In most cases these areas owe their successful crop productivity to mild year-round climates and available sources of surface water and/or groundwater for irrigated agriculture. Climate change will influence global rainfall patterns, affecting both the amount and distribution of rainfall. Global climate change model predictions indicate decreased precipitation for drier regions of the world, with annual average precipitation decreases likely to occur in most of the Mediterranean, north and south Africa, northern Sahara, Central America, the American southwest and the southern Andes, as well as southwestern Australia (Collins et al., 2013). Arid regions are the most prone to desertification and salinization (Geist, 2005; Szabolcs, 1990).

Agriculture is directly linked to climate change. Crop yield, water use, biodiversity and soil health are directly affected by changes in the climate. Changes in the frequency and intensity of rainfall, temperature and other extreme weather events will impact agricultural productivity, with the net effect of climate change on world agriculture most likely to be negative. Even though some regions and crops may benefit from increased CO_2, many will not. Increases in atmospheric CO_2 are likely to increase organic matter in soil and stimulate growth and improve water-use efficiency in some crops, but heat waves, droughts and flooding may dampen these potential yield increases. Recent research has indicated that increased atmospheric CO_2 may not have as large an influence on plant productivity as once thought (Jarvis, Ramirez, Anderson, Leibing, & Aggarwal, 2010; Körner, 2006; Long, Ainsworth, Leakey, & Morgan, 2005; Poorter & Navas, 2003; Zaehle, Friedlingstein, & Friend, 2010; Zavaleta et al., 2003). Ostensibly, increasing ozone levels may counteract the CO_2 effect (Long et al., 2005). The negative effects of increased temperature on plant growth may also counteract the CO_2 effect (Jarvis et al., 2010). Furthermore, indirect climatic impacts, such as greater competition by insects, weeds and pathogens, will decrease yield.

A recent paper by Brevik (2013) provides an informative review of the potential impact of climate change on soil properties and processes influencing plant productivity. Nitrogen (N) and phosphorus (P) limitations may play a role. Niklaus and Körner (2004) found in a long-term elevated-CO_2 grasslands experiment that after 2 years N and P were limiting biomass production expected from high CO_2 levels. There is also evidence that increased CO_2 levels may not necessarily increase C sequestration. Carney, Hungate, Drake, and Megonigal (2007) found that due to increased microbial activity, soil organic C levels declined under increased CO_2 levels. Temperature can also exacerbate matters. Increased temperatures cause increased CO_2 to be released from soil into the atmosphere, which increases ambient temperatures and continues until a new equilibrium is reached.

Higher atmospheric CO_2 concentrations, higher temperatures, more intensive rainfalls and extended droughts and heat waves will accelerate weathering of rocks and minerals in soils. In a 44-year field study by Gislason et al. (2009), weathering rates were found to be already increasing because of global warming. There are both positive and negative effects of climate-change-induced accelerated weathering. Accelerating weathering can increase the inorganic carbon pool in soils due to carbonate mineral formation, which will help decrease atmospheric CO_2 levels. In addition, the dissolution of elements that serve as nutrients for microbes and plants will stimulate microbial and plant growth and biotic C sequestration, which will also help decrease atmospheric CO_2 levels (Qafoku, 2014). In contrast, accelerated weathering may perturb the balance of the biotic and abiotic C cycles within soils. This could increase contaminant mobilization, which may: alter soil microbial activity, plant productivity, and C and elemental cycling; influence the distribution of C in less stable soil pools; and create elemental imbalances in aquatic systems (Qafoku, 2014).

Global projections of climate change paint a rather dark picture with respect to the occurrence and frequency of extreme weather events. Extreme climates, such as droughts, floods and hurricanes will destroy fish, livestock and crops, as well as the infrastructures that support them. Extreme climates will contribute to ecosystem degradation and loss, including soil erosion, declining rangeland quality and salinization of soils (FAO, 2016). There is a cascading effect of climate change, in that it directly impacts agroecosystems, which impacts agricultural production, which drives economic and social impacts that impact food security and livelihoods.

Global projections have less associated uncertainty than regional predictions; consequently, effort is needed to monitor regional- and local-scale impacts. Continuous monitoring of climate change impacts on soil health and condition at regional and local scales is essential to identify and quantify trends requiring the development of management strategies that will ameliorate detrimental impacts on crop productivity, especially in highly productive agricultural areas. The sustainability of agriculturally productive arid and semi-arid areas depends upon a timely knowledge of the geospatial impacts that changes in climate patterns will have on soil properties influencing crop yield because of the predicted susceptibility of arid regions to extended and recurring droughts.

In arid and semi-arid regions, soil salinity and irrigation management go hand in hand because salinity control is generally a consequence of leaching. Soil salinity refers to the concentration of salts in the soil solution, consisting of four major cations (i.e., Na^+, K^+, Mg^{+2} and Ca^{+2}) and five major anions (i.e., HCO_3^-, Cl^-, NO_3^-, SO_4^{-2} and CO_3^{-2}). Soil salinity is characterized in terms of the concentration and composition of the soluble salts and is most commonly measured in the laboratory as the electrical conductance of the saturation extract in dS m^{-1} (Corwin & Yemoto, 2017). The accumulation of soil salinity can result in reduced plant growth, reduced yields, and in severe cases, crop failure. Salinity limits water uptake by plants by reducing the osmotic potential, making it more difficult for the plant to extract water. Salinity may also cause specific ion toxicity effects (e.g., Na^+ ion toxicity) depending on the soil pH and upset the nutritional balance of plants. The salt composition of the soil water influences the composition of cations on the exchange complex of soil particles, which influences soil permeability and tilth. Sodic soils have soil structure degradation and permeability reduction. Furthermore, soil salinization is regarded as a driver for desertification and has a close association with land degradation processes such as arable land abandonment and soil erosion (D'Odorico, Bhattachan, Davis, Ravi, & Runyan, 2013). Because of the potential detrimental impacts of soil salinity accumulation and its ubiquitous association with irrigated agriculture, salinity is a soil chemical property that is crucial to soil health.

For centuries farmers have been well aware of the inexorable link between soil salinity and crop yield. Scientists have quantitatively described the relationship between salinity and yield through the piecewise linear salt tolerance equation (Maas & Hoffman, 1977):

$$Y_r = 100 - b\,(EC_e - a), \tag{1}$$

where Y_r represents the relative crop yield, EC_e is electrical conductivity of the saturation extract (dS m^{-1}), a is the salinity threshold (dS m^{-1}) and b is the slope expressed as % per dS m^{-1}. Representative salt-sensitive, moderately sensitive, moderately tolerant and tolerant crops include carrot, broccoli, wheat and cotton, with salinity thresholds (a; dS m^{-1}) and slopes (b; % per dS m^{-1}) of 1.0 and 14.0, 2.8 and 9.2, 6.0 and 7.1, and 7.7 and 5.2, respectively (Maas & Hoffman, 1977). In short, beyond a specific soil salinity threshold, which is characteristic to each plant and variety of plant, crop yield will decrease linearly, where the yield decrement is also plant specific.

The impact of climate change on soil salinity levels in the root zone has been far less studied than other soil properties, such as organic matter, N and P. This may be due to the fact that salinity is among the most spatially complex and temporally dynamic soil properties, with a coefficient of variation generally over 60% (Corwin et al., 2003). Another reason may be the lack of quantitative spatial data over large spatial extents to make comparisons of changes over time. Salt-affected soils are estimated to comprise 23% of the cultivated land, approximately 3.5×10^8 ha (Massoud, 1981). In actuality, however, there are no directly measured global inventories of soil salinity. All known global inventories of soil salinity and, with only one exception (i.e., California's San Joaquin Valley), all known regional-scale inventories are gross approximations based on qualitative and not quantitative data (Lobell, 2010; Lobell et al., 2010). Until the recent development of proximal and remote sensors with associated protocols and guidelines for measuring soil salinity from field to regional scales, the ability to map and monitor soil salinity across multiple scales has been too formidable due to the high spatial and temporal variability of soil salinity (Corwin & Scudiero, 2016). Or, it may be due to the fact that, unlike many other soil properties, soil salinity can be easily managed by the addition of water to leach salts. As long as the water source is plentiful and sufficiently good in quality, then salinity is generally not regarded as a problem of concern. However, if droughts become more frequent as climatologists predict, then salinity is likely to become a growing issue of concern.

The objective of this paper is to evaluate the impact climate change has had on root zone soil salinity development in selected agricultural areas throughout the world that either have experienced weather pattern changes or have been affected by climate-driven sea level changes (e.g., California's SJV; Minnesota's Red River Valley, RRV; Bangladesh; the Gaza Strip; and Australia's Murray-Darling River Basin, MDRB). To meet this objective, proximal and remote sensing techniques are reviewed as background material for assessing soil salinity in the root zone across multiple scales. These geophysical techniques provide rapid, reliable and detailed spatiotemporal georeferenced measurements through the use of an integrated system of proximal and satellite sensors, protocols and guidelines, and statistical spatial software. The integrated system assesses salinity development across multiple scales to evaluate climate change impacts by inventorying and temporally monitoring soil salinity changes due to alterations in climate patterns and to anthropogenic salinization processes associated with irrigated agriculture that are a consequence of climate change. Five case studies are discussed (i.e., California's SJV, Minnesota's RRV, Bangladesh, the Gaza Strip and Australia's MDRB), which demonstrate impacts on soil salinity due to disparate salinization processes influenced by climate change. For two of the case studies (i.e., California's SJV and Minnesota's RRV) proximal and remote sensors were used to assess the impact of climate change on soil salinity accumulation in the root zone. The ramifications of these soil salinity impacts are presented and how best to mitigate and monitor future trends in salinization due to climate change.

2 | MAPPING ROOT ZONE SOIL SALINITY WITH PROXIMAL AND SATELLITE SENSORS ACROSS MULTIPLE SCALES

Historically, three methods have been developed for determining soil salinity at field scales and larger spatial extents: (a) visual crop observations, (b) geospatial measurements of apparent soil electrical conductivity (EC_a), and most recently (c) multi- and hyperspectral imagery. Visual crop observation is the oldest method of determining the presence of soil salinity in a field. It is a quick method, but it has the disadvantage that salinity development is detected after crop damage has occurred. For obvious reasons, the least desirable method is visual observation because crop yields are reduced to obtain soil salinity information. Remote imagery, including multi- and hyperspectral imagery, is increasingly becoming a part of agriculture and represents a quantitative approach to the antiquated method of visual observation. Multi- and hyperspectral remote imagery offers tremendous potential but are still in their infancy, with an inability at the present time to differentiate osmotic from matric or other stresses, which is key to the successful application of remote imagery as a tool to map salinity and/or water content. Currently, the most common method of mapping salinity at field (i.e., < 3 km^2) and landscape (i.e., 3–10 km^2) scales is geospatial measurements of EC_a.

Apparent soil electrical conductivity is a measure of the electrical conductivity of the bulk soil. It measures anything conductive in the soil. Geospatial EC_a measurements are particularly well suited for establishing within-field spatial variability of not only soil salinity, but a range of soil properties (e.g., water content, texture, organic matter and bulk density) because they are quick and dependable measurements that integrate the influence of several soil properties contributing to the electrical conductance of the bulk soil. At present, no other measurement provides a greater level of spatial soil information than that of geospatial measurements of EC_a when used to direct soil sampling (Corwin & Leach, 2005). The characterization of spatial variability using EC_a measurements is based on the hypothesis that spatial EC_a information can be used to develop a directed soil sampling plan that identifies sites that adequately reflect the range and variability of soil salinity and/or other soil properties correlated with EC_a at the study site (Corwin & Lesch, 2003, 2005b). Maps of the variability of EC_a provide the spatial information to direct the selection of soil sample sites to characterize the spatial variability of those soil properties correlated, either for direct or indirect reasons, with EC_a. In essence, EC_a serves as a surrogate to map those properties that correlate with EC_a at that specific field site. This is referred to as *EC_a-directed soil sampling*. This hypothesis has repeatedly held true for a variety of agricultural applications (Corwin, 2005; Corwin, Kaffka, et al., 2003; Corwin, Lesch, et al., 2003; Corwin & Lesch, 2003, 2005b, 2005c; Johnson et al., 2001; Lesch, Corwin, & Robinson, 2005; Lesch, Rhoades, Lund, & Corwin, 1992).

The field-scale mapping of salinity shifted in the 1970s and 1980s from the measurement of the EC of soil solution extracts to the measurement of EC_a. The geophysical techniques of electrical resistivity (ER) and electromagnetic induction (EMI) are most commonly used to measure EC_a. The shift to measuring EC_a occurred because the time and cost of obtaining soil solution extracts prohibited their practical application at field scales due to the high spatial variability of soil. In contrast to EC measurements of soil solution extracts, EC_a is a very rapid and easy to take measurement that can be readily mobilized to obtain tens of thousands of geo-referenced measurements of EC_a in a comparatively short time period, providing field-scale maps of EC_a. The use of EC_a to measure salinity has the further advantage of increased volume of measurement. However, EC_a suffers from the complexity of measuring EC for the bulk soil rather than being restricted to the solution phase where salts reside, making interpretation of the spatial EC_a data difficult. Furthermore, EC_a must be converted to EC_e because EC_e is the reference point for salinity, requiring a calibration between EC_a and EC_e. Even with these challenges, the practical application of EC_a far outweighs the negative aspects. Corwin and Leach (2005), Doolittle and Brevik (2014) and Heil and Schmidhalter (2017) review the use of EC_a in agriculture.

Since 1980, the U.S. Salinity Laboratory (USSL) within USDA-ARS has been the centre of research related to mapping and monitoring soil salinity at the field scale and larger spatial extents using proximal (i.e., EMI and ER) and remote sensors (Corwin, 2008; Corwin & Scudiero, 2016, 2019). Over that time, USDA-ARS scientists and scientists visiting USSL have developed three approaches for mapping soil salinity at three distinct spatial scales: field (<3 km^2), landscape (3–10 km^2) and regional (10–10^6 km^2) scales. Each approach is based on EC_a-directed soil sampling. The three approaches are: (a) field-specific regression using EC_a-directed soil sampling (field scale), (b) analysis of covariance (ANOCOVA) approach using EC_a-directed soil sampling (landscape scale), and (c) satellite imagery combined with EC_a-directed soil sampling (regional scale). Corwin and Scudiero (2019) provide a comprehensive review of these three approaches and various applications for which these approaches have been used.

2.1 | Field-scale mapping of soil salinity from geospatial EC_a measurements

The protocols for an EC_a-direct soil sampling survey to measure soil salinity at the field scale include eight basic elements: (a) EC_a survey design, (b) geo-referenced EC_a data collection, (c) soil sample design based on geo-referenced EC_a data, (d) soil sample collection, (e) physical and chemical analysis of pertinent soil properties, (f) spatial statistical analysis, (g) determination of the dominant soil properties influencing the EC_a measurements at the study site, and (h) geographic information system (GIS) development (Corwin & Lesch, 2005b). A detailed discussion of the protocols for mapping soil salinity at field, landscape and regional scales can be found in Corwin and Scudiero (2016, 2019). Scientists at the USSL have developed an integrated system for the measurement of field-scale spatial variability, particularly salinity, consisting of (a) guidelines and protocols for the characterization of soil spatial variability using EC_a-directed soil sampling presented by Corwin and Lesch (2003, 2005b) and protocols specific to soil salinity assessment presented by Corwin and Lesch (2013), (b) mobile EC_a measurement equipment (Rhoades, 1993), and (c) sample design software (Lesch, Rhoades, & Corwin, 2000). The integrated system and procedure for mapping soil salinity at the field scale is schematically illustrated in Figure 1.

2.2 | Landscape-scale mapping of soil salinity from geospatial EC_a measurements

Multiple-field EC_a survey data often exhibit an abrupt change in magnitude across field boundaries, which presents a challenge to the conversion of EC_a to EC_e at spatial extents of thousands to tens of thousands of hectares (i.e., landscape scale). The abrupt change has various causes: (a) between-field variation in field average water content due to irrigation method, frequency and timing; (b) between-field variation in soil texture; (c) condition of the soil surface (e.g., till vs. no-till) due to management practices that effect soil compaction; (d) surface geometry (i.e., presence or absence of beds and furrows); (e) temperature differences (i.e., EC_a surveys conducted at different times of the year); and (f) between-field spatial variation in salinity (Corwin & Lesch, 2014).

Calibration models are often used to make an adjustment for an abrupt change. Consider the case of surface geometry, for example, presence and absence of beds and furrows in a field, where an EC_a survey has been conducted. In the absence of any surface geometry, a simple power model describes the deterministic component of the EC_e–EC_a relationship, for example, $EC_{e,i} \quad \beta \cdot EC_{a,i}^{\alpha}$, where β is a coefficient and $i = 1, 2, 3, ..., n$. To account for the surface geometry effect, an additional dummy variable (x) and associated scaling parameter (θ) are used,

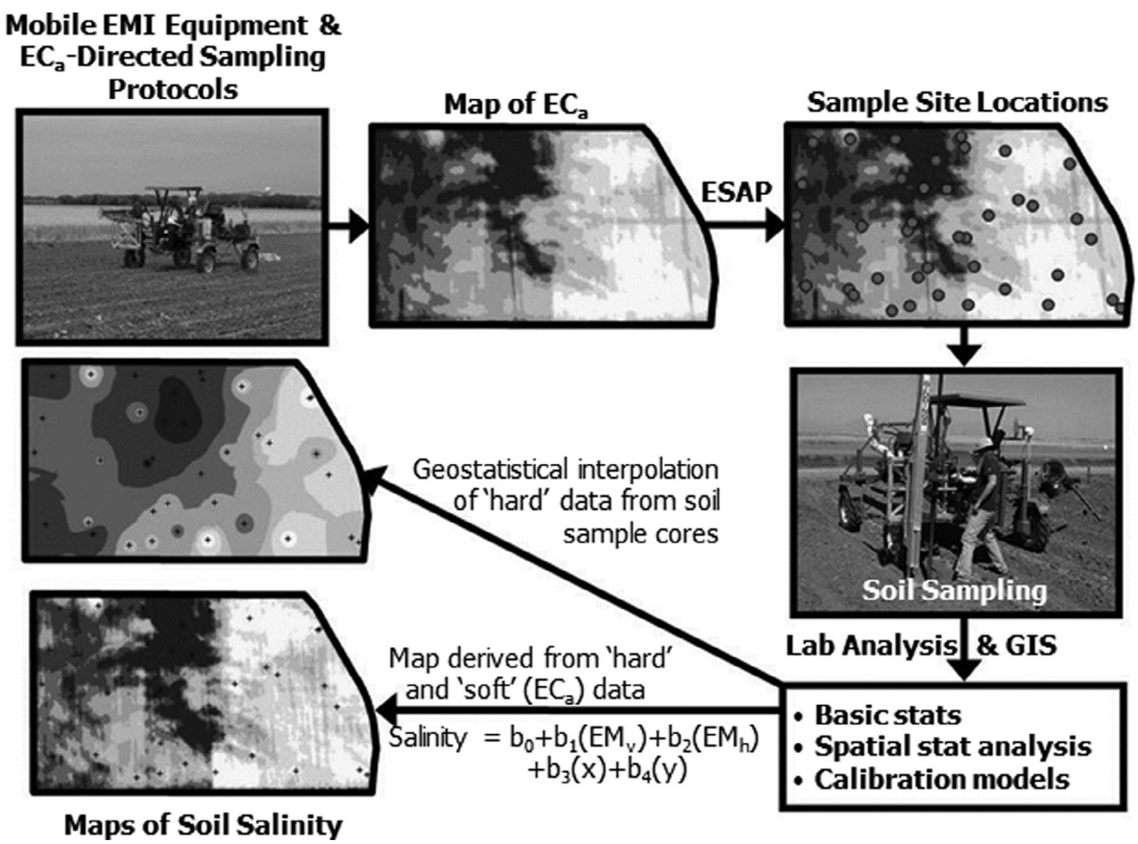

FIGURE 1 Schematic of the integrated system to assess field-scale soil salinity using apparent electrical conductivity (EC_a) directed soil sampling protocols, a mobile electromagnetic induction (EMI) rig, ESAP software, and geographic information system (GIS). *Source*: Corwin (2015) with permission. EM_v refers to the measurement of EMI in the vertical coil configuration and EM_h refers to measurement of EMI in the horizontal coil configuration

for example, $EC_{e,i} \quad \theta^{x_i} \cdot \beta \cdot EC^{\alpha}_{a,i}$, where $x_i = 1$ if there is a surface geometry effect and $x_i = 0$ otherwise. Under a log transformation, this multiplicative parameter becomes additive, as shown in Equation (2):

$$\ln(EC_{e,i}) \approx x_i \ln(\theta) + \ln(\beta) + \alpha \ln(EC_{a,i}) = \beta_{01} + \beta_{02}(x_i) + \alpha \ln(EC_{a,i}). \tag{2}$$

On a log–log scale, a simple linear regression model with an additional blocking (shift) parameter can make an adjustment for an abrupt change in any multiplicative EC_a effect within a field. Equation (2) is a type of ANOCOVA model. In principle, this type of ANOCOVA modelling approach could be used to calibrate multiple-field EC_a surveys to EC_e, provided the assumptions in Equation (2) are reasonable.

If geo-referenced EC_a survey data are acquired across multiple fields and assuming that the number of soil sampling locations collected in any given field is minimal (i.e., $n \leq 10$), then basic regression modelling techniques are used, such as ANOCOVA. An ANOCOVA model for EC_a–EC_e calibration is defined by Equation (3):

$$\ln\left(EC_{e,ijk}\right) = \beta_{0,jk} + \beta_{1,j} \ln(EM_{v,ik}) + \beta_{2,j} \ln(EM_{h,ik}) + \varepsilon_{ijk}, \tag{3}$$

where i refers to the soil sample site within a field (i = 1, 2, 3, ..., n_k), j is the sample depth (j = 1, 2, 3, ..., p), k is the field (k = 1, 2, 3, ..., M), EM_v is the EC_a measured with EMI in the vertical coil configuration (dS m^{-1}), and EM_h is the EC_a measured with EMI in the horizontal coil configuration (dS m^{-1}). In the ANOCOVA model, the intercept parameter is uniquely estimated for each sampling depth and field, but the slope coefficients are assumed to change across sampling depths (not across fields).

The ANOCOVA approach for EC_a–EC_e calibration has been validated at the regional scale (Corwin & Lesch, 2016). However, the practical application of the ANOCOVA approach is best used at the landscape scale (i.e., 3–10 km^2) (Corwin & Lesch, 2016; Scudiero, Skaggs, & Corwin, 2016).

2.3 | Regional-scale mapping of soil salinity from combining remote sensing and geospatial EC_a measurements

At the regional scale, spatial patterns of soil salinity are influenced by several factors, including pedogenic, meteorological, hydrological, topographical, agronomic, anthropogenic and edaphic factors. For instance, agronomic management influences local scale salinity, whereas pedogenesis influences landscape-scale salinity. To model such multiscale variations, covariates offering continuous spatial coverage, such as remote sensing data, are ideal. In the past three decades, two remote sensing approaches have been developed for mapping soil salinity. The most popular approach includes a variety of spatial analyses of surface (bare) soil reflectance. The other consists of the indirect assessment of root-zone soil salinity through the study of plant canopy reflectance.

Salt accumulation at the soil surface often results in the formation of white salt crusts. Such crusts are easily identifiable with remote sensing, as their reflectance properties are different from those of soils not affected by soil salinity (Mougenot, Pouget, & Epema, 1993). One way to identify crusts is through image classification (e.g., Metternicht, 1998). Often, salt efflorescence is partial, making the identification of salt-affected bare land more problematic. This is because of confounding effects from different soil types (e.g., texture and colour), soil roughness, presence of vegetation and surface soil water content. However, most of these confounding effects can be accounted for (e.g., Xu, Zeng, Huang, Wu, & van Leeuwen, 2016). Unfortunately, this approach has limited relevance in agricultural applications because crop growth and yield are influenced by the salinity in the root zone. In agriculture, information on surface soil salinity is often only relevant for evaluation of plant germination. Indeed, several studies show that there is no direct correlation between root-zone and surface-soil salinity (e.g., Zare, Huang, Santos, & Triantafilis, 2015).

Spectral reflectance properties of salt-affected vegetation are different from those of non-stressed plants. Differences can be seen in the spectral signature of crops, especially in the visible (e.g., 450–700 nm) and near-infrared (e.g., 770–900 nm) spectra. Plants stressed by soil salinity are characterized by higher visible and lower near-infrared range reflectance than non-stressed plants. Unfortunately, the use of surface reflectance (i.e., multi- and hyperspectral), from a single satellite scene, to model soil salinity is site specific, for reasons including: (a) the spectral signature of a crop changes with phenological stages; (b) different crops are characterized by different spectral signatures; (c) other stress sources, such as nutrient deficiency or water stress, trigger similar responses in plants' reflectance properties; and (d) surface reflectance is influenced by different soil backgrounds. Due to these confounding effects, regional-scale mapping of soil salinity with remote sensing has often yielded unsatisfactory and inconsistent results in the past.

Recent research showed that salinity stress can be isolated from other types of within-season and season-wide transient stressors (e.g., water stress and mismanagement) by analysing multi-year canopy reflectance data (Lobell et al., 2010; Scudiero, Corwin, & Skaggs, 2015). Scudiero et al. (2015) considered annual average values of Landsat 7 (united states geological survey and national aeronautics and space administration, USA) vegetation indices over a 7-year period and used the year with the highest vegetation index value (i.e., year with maximum average plant performance) to build a regression model from ground-truth fields located in California's western SJV. The regional-scale salinity model included covariate information on land use (i.e., cropping system) and meteorology. Salinity assessment models can be improved by adding information on agronomic practice (e.g., crop type) (Lobell et al., 2010), meteorology and soil type (Scudiero et al., 2015), reflectance in other spectral ranges (e.g., thermal) (Wu et al., 2014), landscape position, and other factors that are known to influence crop growth in a given study region. Soil salinity maps based on multi-year analysis of canopy reflectance are accurate, especially in areas characterized by homogeneous soils, where salinity is the primary permanent stressor. However, the approach has some limitations, including uncertainty in the depth of the modelled soil profile (because different crops have different root-zone depths) and uncertainty of predictions at very low salinity values. At low salinity levels, most crops are not influenced by soil salinity (Maas & Hoffman, 1977), making it impossible to assess the underlying soil salinity through canopy reflectance. Moreover, the growth of halophyte (salt tolerant) weeds is not optimal at low salinity levels (BOSTID, 1990), making it hard to discriminate between low halophyte plant performances at low (i.e., not stressed) and high (i.e., stressed) salinity levels (Scudiero et al., 2015; Zhang et al., 2015). The use of multi-year canopy reflectance data for regional-scale monitoring of salinity over a growing season or from one year to the next may be problematic unless the model relating the satellite imagery to soil salinity is stable. Currently, scientists at the USSL are looking into the year-to-year stability of their regional-scale salinity model for the SJV developed by Scudiero et al. (2015). Further discussion on this approach for mapping soil salinity, including considerations of the selection of satellite platform and ground-truth sites, can be found in Scudiero, Corwin, Anderson, and Skaggs (2016).

3 | PROCESSES OF SOIL SALINIZATION

To understand clearly the interrelationship and interaction of the climate-driven factors that influence development of salinity in the root zone it is necessary to have a clear picture of the processes involved with soil salinization. There are two types of soil salinity: primary and secondary. Primary soil salinity accumulates by natural phenomena, whereas secondary soil salinity is a consequence of the management of natural resources during anthropogenic activities such as urbanization and agriculture (irrigated land and dryland). Even though soluble salts are inherent in soils there are a variety of processes that influence the accumulation of salts within the soil profile. The processes that contribute salts to soil include: (a) weathering of soil minerals, (b) salts added from rain or irrigation with freshwater, (c) salts added from recycled degraded waters, (d) application of fertilizers and pesticides, (e) saline groundwater intrusion from fluctuating water tables or overdrafting of wells, (f) seawater intrusion along coastal areas, and (g) dumping of industrial or municipal wastes. These salt-contributing processes, combined with edaphic (e.g., texture, permeability, pH and organic matter), climatic (e.g., relative humidity, temperature and rainfall), hydrologic (e.g., groundwater quality and depth to groundwater), anthropogenic (e.g., leaching efficiency and cropping strategy), topographic (e.g., slope, elevation, relief and landforms) and biological (e.g., soil–water–plant interactions, earthworms and microbes) factors, determine the extent and distribution of soil salinization.

Salts accumulate in soil primarily due to the process of ET. In irrigated soils when drainage does not meet leaching requirements (i.e., the quantity of extra irrigation water that must be applied above the amount required by the crop to maintain an acceptable root-zone salinity), then the quality and quantity of the irrigation water will result in salt accumulation. However, soil salinity can also result from poor drainage or a shallow water table, physical and chemical soil conditions that reduce leaching of salts from the soil profile, poor irrigation quality, topography and salt-water intrusion. As a consequence of transpiration, plant roots extract pure water, leaving salts behind. In a similar fashion, evaporation from the soil surface removes soil water into the atmosphere, leaving salts behind. In cases where a shallow water table (e.g., < 2 m) exists, particularly in arid and semi-arid environments, evaporation at the soil surface (and transpiration from plants) acts as a driver in combination with the adhesive and cohesive forces of capillary rise to draw water upward to the soil surface. Salts in the water are then deposited as the water evaporates, causing salts to accumulate at the soil surface. Physical and chemical conditions of the soil can reduce leaching of salts from the soil profile, causing salts to build up. For instance, a claypan or other low-permeability or impermeable layer (e.g., caliche layer) will slow the leaching of salts, resulting in their buildup. The chemical composition of the exchange sites on clays can also reduce the permeability of a soil. Sodium on exchange sites causes clay soils to disperse, reducing their permeability. Poor irrigation water quality or the reuse of degraded water (e.g., drainage water or municipal waste water) will add salts to agricultural soil. Generally, poor irrigation water quality occurs in areas where saline groundwater is the only source of water, often due to drought conditions. Degraded water reuse also often occurs during drought conditions and often involves the use of drainage water from agricultural areas where drainage tiles are present, which provides an alternate water resource and a means of reducing the need for disposal of drainage water (Corwin, 2012; Corwin, Lesch, Oster, & Kaffka, 2008). In the case of salt accumulation due to topography, an upslope recharge leaches salts downslope until they reach a shallow impermeable layer, where a downslope discharge occurs due to capillary rise and evaporation, causing salts to accumulate. Salt-water intrusion occurs at regional (e.g., fluctuating sea level elevation) and local (e.g., well) scales (Werner et al., 2013). In non-coastal areas, salt-water intrusion results from a transfer of saline water from a saline aquifer to an overused non-saline aquifer, generally due to well-water extraction to meet agricultural use. Along coastal areas, a rise in sea level due to climate change causes seawater intrusion (FAO, 2011). As elevations in sea level are driven by rising global temperatures, ocean water penetrates deeper into coastal estuaries and groundwater aquifers. Saline estuaries cause salt accumulation in the root zone of soils near the estuary. Seawater intrusion into coastal aquifers is aggravated by the extraction of well water from the aquifers and use of these degraded saline waters for agriculture and urban purposes, salinizing the soils. Seawater intrusion simulations using various combinations of sea-level rise and groundwater extraction have established that groundwater extraction is the primary driving force of seawater intrusion (Loáiciga, Pingel, & Garcia, 2012).

4 | GENERAL IMPACTS OF CLIMATE CHANGE ON AGRICULTURE

Global average temperature is projected to increase from 0.3 to 0.7°C for the period from 2016 to 2035 as compared to the period from 1986 to 2005, with increased temperature greater on land than over the ocean and more frequent high-temperature episodes on land (Kirtman et al., 2013). Meehl et al. (2007) project that over the next two to three decades global warming will increase by 0.2°C per decade, with greater increases for cultivated lands. Collins et al. (2013) indicate that global annual mean surface air temperatures are projected to increase by 1–2°C from 2046 to 2065. Subsequently, soil moisture will be depleted more rapidly in both irrigated and non-irrigated agricultural areas (FAO, 2013). More rapid depletion of soil moisture places more demand on surface and groundwater supplies for irrigated agriculture and reduces crop production in non-irrigated agriculture. The impact of climate change on crop productivity is the consequence of a variety of physical and chemical parameters, including temperature, patterns of precipitation and increases in atmospheric ozone and CO_2 (FAO, 2016; Porter et al., 2014).

Since 1950, global average temperatures have risen by approximately 0.13°C per decade as a result of increasing atmospheric CO_2 levels (Meehl et al., 2007), yet the impact this has had on agriculture, especially irrigated agriculture, is only now becoming apparent (Lobell, Schlenker, & Costa-Roberts, 2011). Many cultivated plants will respond favourably to an increase in atmospheric CO_2. When CO_2 is the only experimental variable, then elevated levels of CO_2 tend to enhance plant growth and water use efficiency in the short term and sometimes the long term (Idso & Idso, 1994; Idso & Kimbaall, 1997). For every 100 ppm increase in CO_2, cereal crops average an increase of 4–5% in yield (Teh & Koh, 2016). However, neither CO_2-enhanced plant growth nor increased water use efficiency is certain to outweigh the effect that elevated CO_2 will have on temperature, water availability, nutrients, evaporative demand, salinity and other stresses (Derner et al., 2003; Yeo, 1999). The expectation is for climate change to have a net increase in the proportion of semi-arid land. Higher temperatures are likely to benefit some crops while placing others at a disadvantage due to increased ET and thermal damage. Yeo (1999) points out that factors enhancing crop growth under osmotic stress, such as decrease in water use, decrease in leaf salt concentration and increase in fixed carbon, may not occur from elevated CO_2. Concomitantly, many weeds will also react favourably. Subsequently, the increase or decrease in productivity of a crop will depend on how competitive weeds are for nutrients and water. Diseases and pests will follow climate change into areas that are less well prepared to combat them from both a biological and institutional perspective.

Shifts in plant zonation are also likely under climate change, with changes in the composition of both natural and agricultural systems likely. Agricultural systems are more likely to be independent of rapid climate change conditions than natural systems, but agricultural systems are less adaptable and there are fewer options for change. It is likely that the gains and losses due to shifts in zonation will balance out globally, but cropping system changes in a localized area will be inevitable (Yeo, 1999). For example, rice cultivation is predicted to increase in the more northerly latitudes, with less production occurring closer to the equator due to heat damage and lower water availability (Matthews et al., 1995). However, globally the rice cultivation area is expected to have little change (Kropff, Matthews, van Laar, & ten Berge, 1995; Solomon & Leemans, 1990).

Chakraborty and Newton (2011) estimate that 10–16% of global crop productivity is lost to pests, which is estimated to cost USD 220 billion in lost revenue, with weeds estimated to be the greatest cause of loss (36%) (Oerke, 2006). Climate change and changes in CO_2 concentration will increase the impact of pests by increasing their competitiveness and enhancing their distribution (FAO, 2016). Research has shown that changes in temperature result in changes in geographic range, which could extend pests toward the poles and higher altitudes (Porter et al., 2014; Svobodová et al., 2014). This could open the door for the spread of invasive plant species such as water hyacinth. Pests may also appear earlier in the season due to the higher temperature that accompanies climate change. Because weeds in arid and semi-arid regions tend to be more drought and salt tolerant than most crops grown under irrigation, changes in soil salinity due to altered weather patterns will favour the presence of weeds over crops that are less salt tolerant, and weeds will even be competitive with crops that are highly salt tolerant.

Global food security focuses on four major crops that account for 85% of the world's cereal exports: wheat, rice, maize and soybean (Teh & Koh, 2016). Past climate change trends in crop production are evident in several regions across the globe (Porter et al., 2014). Lobell et al. (2011) provided evidence that climate change had already affected wheat and maize yields both regionally and globally. The expectation is that climate change will fundamentally alter the patterns of global food production, with negative impacts on crop productivity of wheat, rice and maize in low latitude and tropical regions (FAO, 2016). Temperate zones will also be impacted, such as for maize in the USA and wheat in the European Union, due to increased water scarcity, more frequent and intense heat events, and accelerated phenology (FAO, 2016). Even though most climate change impacts on crop productivity are expected to be negative, there are studies that predict positive impacts in areas where increases in precipitation are expected to occur. For instance, the combined use of climate and crop models for the grain-producing zone of Central Eurasia indicates an increase in yield as a result of higher atmospheric CO_2, warmer temperatures and longer growing seasons with less frost (Lioubimtseva, Dronin, & Kirilenko, 2015). However, none of the modelling or trend analysis studies has looked at climate change impacts on soil salinity, which has a direct impact on crop yield.

5 | IMPACTS OF CLIMATE CHANGE ON ROOT-ZONE SOIL SALINITY DEVELOPMENT

Changes in climate patterns influence the salinization process. Soil salinity development in the root zone can occur due to reduced water availability in arid and semi-arid irrigated agricultural regions, upward movement of salts from shallow water tables, the reuse of degraded waters and salt-water intrusion. Too much or too little rainfall can have significant impacts on soil salinity in the root zone. Too much rainfall raises the water table. When the water table is 2 m or less from the soil surface, then capillary rise causes the upward movement of salts from the water table to the soil surface. This results in the accumulation of salinity at or near the soil surface during the dry portion of the year when insufficient rainfall is available to leach the salts from the root zone. For instance, in the RRV of the USA's midwest region, higher than average annual rainfalls for most of the past two decades, along with the growth of more shallow rooted crops, have caused the water table to rise. During low-rainfall years, water moves upward from the shallow water table due to ET increasing salinity in the root zone. Concurrently, topographic effects have occurred where increased upslope recharge due to higher than average rainfall has caused increased downslope discharge, resulting in the formation of saline seeps.

Drought can have somewhat similar end results. When droughts occur in agricultural areas, such as the recent 5-year California drought from 2011 to 2015, those areas with shallow water tables, such as the west side of California's SJV, will accumulate salts, once again due to capillary rise. The west side of the SJV has shallow water tables due to local and basin hydrological effects. Under normal conditions, applications of irrigation water leach salts from the root zone. However, during reduced water availability from drought conditions there is limited or no surface water available for irrigation, causing significant portions of the land to be fallow. Upward water movement resulting from capillary rise from the shallow water table causes salts to accumulate at the soil surface. Another instance of drought causing salinity accumulation in the root zone occurs during the overdrafting of well water for agricultural purposes from a non-saline aquifer located near a saline aquifer or near coastal areas. In this case the overdrafting will transfer saline water to the overused non-saline aquifer or cause seawater intrusion. In addition, irrigated agricultural areas experiencing drought conditions are likely to shift to the reuse of degraded waters, such as municipal or drainage waters, which are higher in salinity than rain water. The reuse of degraded water will increase salinity levels in the root zone. Rising ocean levels due to the melting of the polar icecaps results in greater seawater intrusion.

Model simulations clearly show that salinity development in the root zone can increase nitrogen leaching from soil and decrease crop yield. Simulations conducted by Pang and Letey (1998) explain the underlying mechanism of salt stress on plants, resulting in increased nitrogen leaching, which is qualitatively described as follows: 'Salinity leads to reduced plant growth, which leads to more leaching, which leads to salt removal from the root zone … Reduced N leads to reduced plant growth, which leads to less ET, which leads to more leaching, which leads to even less N in the root zone'.

Case studies are presented that reflect some of these salinity-development scenarios resulting from altered weather patterns, including in the SJV (influence of drought on shallow water tables) and RRV (influence of extended higher than average precipitation on shallow water tables and topographical recharge-discharger), seawater intrusion in Bangladesh and the Gaza Strip due to rising ocean levels, and in Australia's MDRB (combined influences of increased crop water requirement, lower irrigation water quality and reduced rainfall).

6 | CASE STUDIES REFLECTING THE IMPACT OF CLIMATE CHANGE ON SALINIZATION

Two agricultural regions within the USA are presented: California's SJV and Minnesota's RRV. The altered weather patterns differ in the two agricultural regions, with a 5-year drought in the SJV (2011–2016) and prior to 2010 nearly two decades of above average rainfall for the RRV. The detailed procedures for measuring and monitoring salinity development for SJV and RRV are found in the papers by Corwin, Carrillo, Vaughan, Cone, and Rhoades (1999), Corwin (2012), Scudiero, Skaggs, and Corwin (2014) and Scudiero et al. (2015, 2017) for the SJV and Lobell et al. (2010) for the RRV. The multiscale EC_a-directed soil sampling methodologies described in Corwin and Scudiero (2016) were the basis of the SJV and RRV studies. The two case studies show the impact of climate change on soil salinity development for approximately 900,000 ha of the west side of California's San Joaquin Valley (WSJV) and Kittson County (284,000 ha) in Minnesota's RRV.

Bangladesh and the Gaza Strip within the Mediterranean Basin are examples of where climate change caused salinity issues, but in both instances the salinity issues are due to seawater intrusion from rising ocean levels caused by the melting of ice sheets primarily above the Arctic Circle. Australia's MDRB illustrates a complex interaction of factors (e.g., higher crop water requirement, lower irrigation water quality and reduced rainfall) that are projected to increase soil salinity.

6.1 | West side of California's San Joaquin Valley

The California drought, resulting from changes in climate patterns, has exacerbated soil salinity levels. Three studies provide insight into the impact of climate change on soil salinity accumulation in the WSJV. Each study used the described EC_a-directed soil sampling methodology as a basis for monitoring soil salinity in the root zone (i.e., 0–1.2-m depth increment) at three different scales: field, multiple-field or landscape, and regional scales.

Recent studies by Corwin and his colleagues have shown that reuse of saline irrigation water can reclaim saline-sodic soil (Corwin et al., 2008; Corwin, Lesch, Oster, & Kaffka, 2006). In a long-term, field-scale study of the reclamation of a saline-sodic soil using 3–5 dS m^{-1} drainage water on a 32.4-ha field located on Westlake Farm in the WSJV, Corwin (2012) found that from 1999 to 2009 there was steady improvement in soil quality due to decreases in salts, molybdenum (Mo), boron (B) and the sodium adsorption ratio (SAR), resulting from leaching. Sodic soils have very low permeability due to the dispersive effect of Na on clay particles, which causes the clay to flocculate, reducing the hydraulic conductivity. The decrease in salinity, Mo, B and SAR was attributed to the presence of Ca in the low-quality drainage water used for reclamation. The Ca displaced the Na on clay exchange sites, causing the soil particles to aggregate, thereby improving the permeability. The improved soil quality from 1999 to 2009 is shown in Figure 2. However, in 2011 when the California drought began there was no fresh or drainage water available for irrigation; consequently, the field was left fallow, receiving only rainfall. Within 18 months after irrigation with drainage water had ceased and the field was left fallow, the levels of salinity (i.e., electrical conductivity of the saturation extract in dS m^{-1} abbreviated as EC_e), Mo, B and SAR returned to nearly the original 1999 levels and in some instances the levels were even higher (compare 1999–2011 for EC_e, SAR, B and Mo in Figure 2). This field is typical of fields located in the WSJV near the San Joaquin River, where the soils are generally fine textured and the

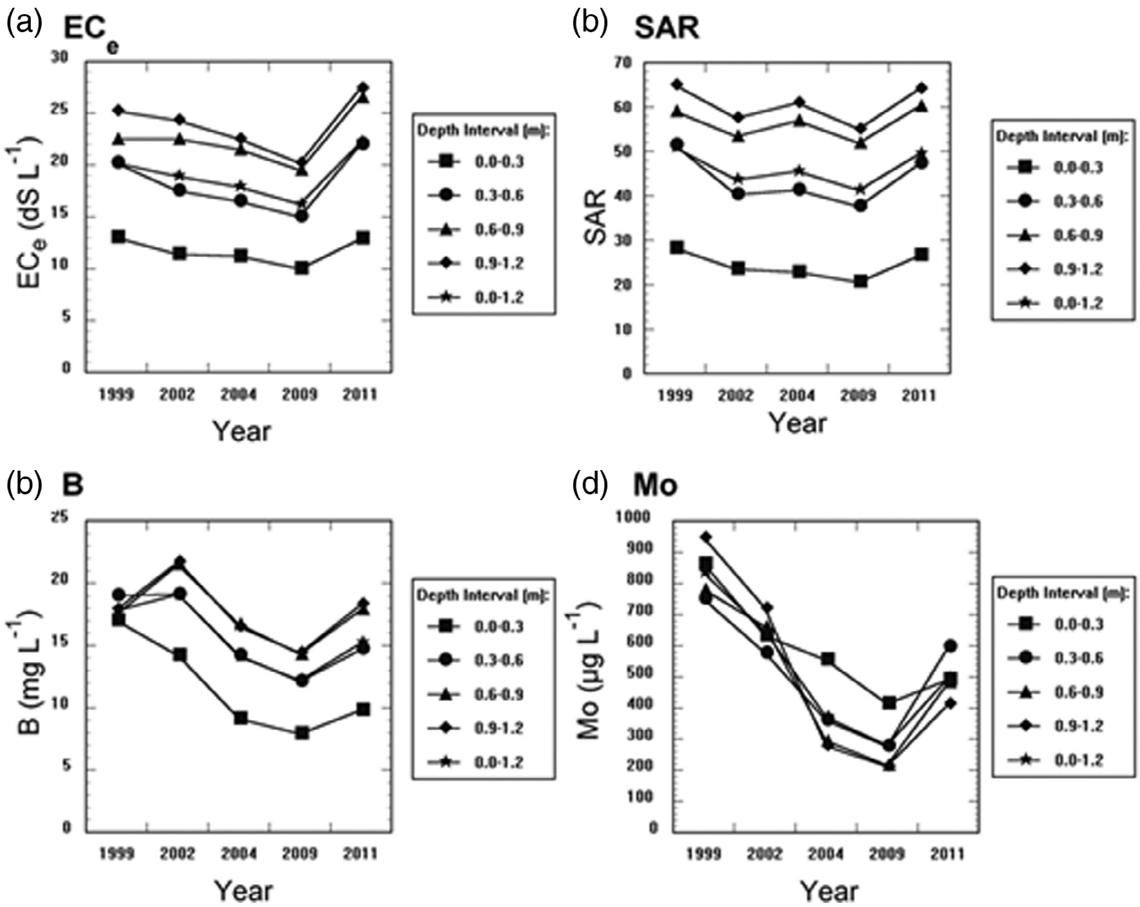

FIGURE 2 Graphs showing the improvement in soil quality from 1999 to 2009 for (a) salinity (EC_e), (b) sodium adsorption ratio (SAR), (b) boron (B), and (d) molybdenum (Mo) followed by the return to original levels by 2011 after drought interrupted further application of drainage water to reclaim the saline-sodic soil. *Source*: Corwin (2012) with permission

water table is perennially shallow (i.e., within 2 m of the soil surface). It can be expected that all fallow lands in the WSJV with shallow water tables would experience a similar increase in salinity and trace elements due to the capillary rise effects from the shallow water table causing the upward movement of salts and trace elements.

A comparison of salinity levels for 2,400 ha of the former Broadview Water District in the WSJV from 1991 (Figure 3a) to 2013 shows an increase in soil salinity has occurred (Figure 3b). In February 2005, Broadview Water District land became fallow as a result of their water allocation being sold to an adjacent water district, Westlands Water District (Wichelns & Cone, 2006). Figure 3a shows a map of salinity for the Broadview Water District in 1991 obtained from the EC_a-directed soil sampling approach of Corwin et al. (1999), Corwin and Lesch (2003) and Corwin and Leach (2005). Figure 3b shows the increase in soil salinity from 1991 to 2013 as indicated by the percentage of the total acreage falling within salinity classes of 0–2, 2–4, 4–8, 8–16 and >16 dS m^{-1}. From 1991 to 2013 the 2–4 dS m^{-1} salinity class showed a substantial decrease from 27 to 3%, whereas the 8–16 dS m^{-1} class significantly increased from 3 to 33%. The field average soil salinity increased by 43% within the root zone during this time period. Essentially, all the non-saline and slightly saline soils in 1991 have become moderately (4–8 dS m^{-1}) and strongly saline by 2013, which is primarily due to the upward movement of salts from the shallow water table resulting from capillary rise.

At a regional scale there are strong indications that soil salinity for the root zone (0–1.2 m) has increased for the WSJV over the past three decades. Estimates of salt-affected soils (i.e., $EC_e > 4$ dS m^{-1}) for the WSJV calculated from data presented by Backlund and Hoppes (1984) were approximately 0.45×10^5 ha for 1984. Scudiero et al. (2014, 2015, 2017) estimated salt-affected soils for the WSJV were approximately 0.6×10^5 ha for 2013. The increased salinization of the WSJV from 1984 to 2013 may or may not be attributed solely to changes in climate patterns because there are no reliable estimates of salt-affected soils for the WSJV in 2011 when the California drought began. Nevertheless, salinization of the

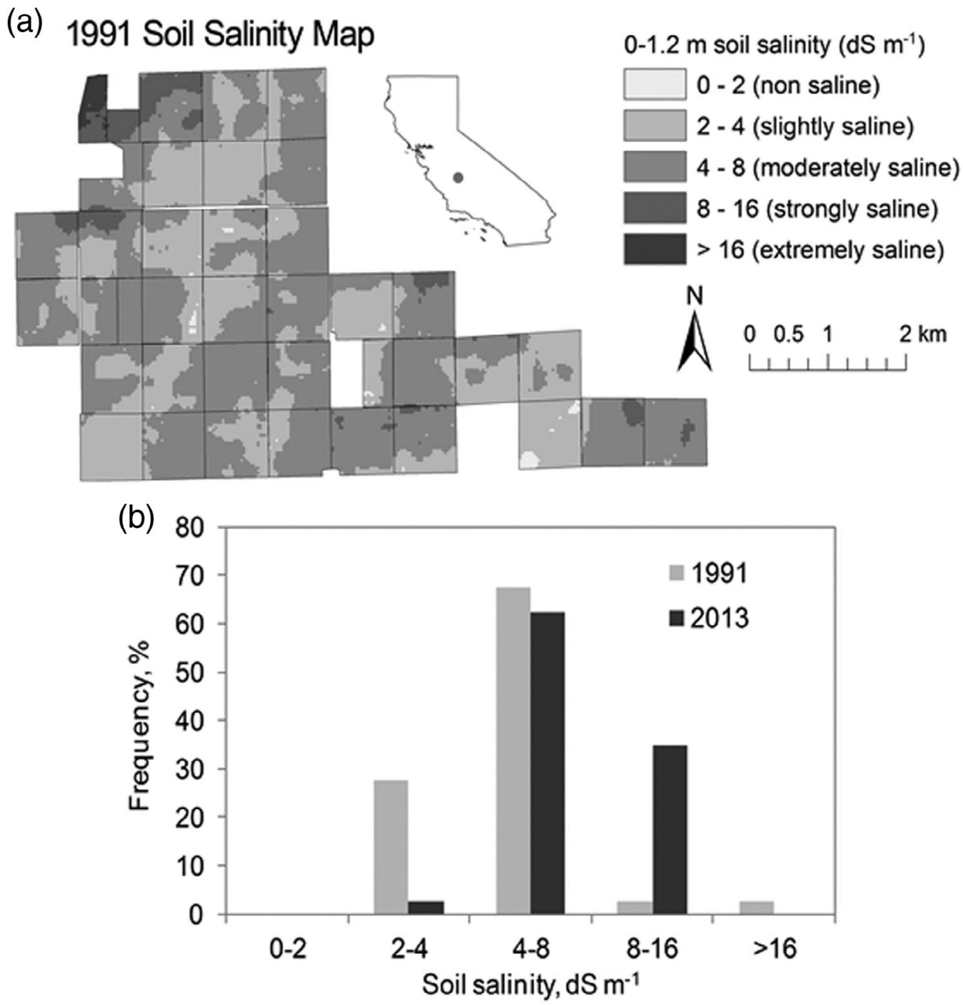

FIGURE 3 Broadview Water District: (a) map of soil salinity levels within the root zone (0–1.2 m) for Broadview Water District, CA, in 1991 (*Source*: Corwin et al., 1999) and (b) histogram of soil salinity categories as predicted for 2013 using the soil salinity assessment model of Scudiero et al. (2014)

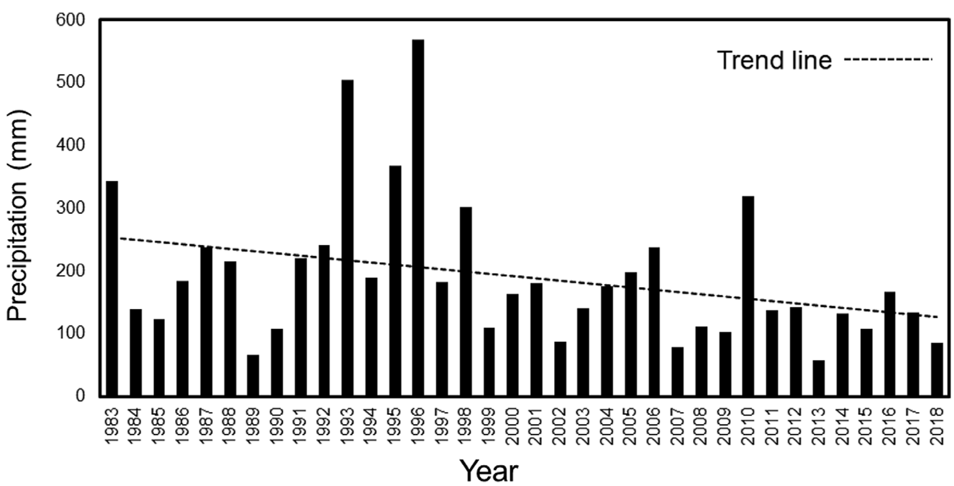

FIGURE 4 Graph of the average annual precipitation (mm) for the west side of the San Joaquin Valley obtained from the California Irrigation Management Information System (CIMIS) from 1983 to 2018 and the associated calculated linear trend line

WSJV occurred from 1984 to 2013. There are inferential data relating climate change to the increase in soil salinization for the WSJV from the trend in reduced rainfall over the past 35 years. Figure 4 shows the average annual precipitation for the WSJV derived from WSJV California Irrigation Management Information System (CIMIS) stations from 1983 to 2018 and the associated calculated linear trend line. The decreasing trend of the average annual rainfall in the WSJV from 1983 to 2018 indicates a trend of decreasing precipitation available to leach soil salinity. A similar general trend is found for the water available for irrigation in the SJV from the Central Valley

Project (CVP) due to the reduced snowpack in the Sierra Nevada mountains, which is the source of the CVP water supply.

Droughts are common in California, but 5-year droughts, such as the drought from 2011 to 2015, are not common. The increased frequency and severity of drought in California is what is associated with climate change. The 2011–2015 drought was the worst in the history of agriculture for the SJV since record keeping began in 1895. California's expansive and well-designed water delivery system has repeatedly dealt with routine droughts but was pushed to its limits by the 2011–2015 drought. Surface water supplies in the network of reservoirs used for drinking water and agriculture were reduced by 60% of their capacity and water allocations to irrigation districts were reduced by 90–95% before the drought ended. There is no doubt that salinity development at a regional scale in the SJV is attributable to numerous interacting factors and processes in addition to climate change. However, there is concrete field-scale evidence of the impact of climate change on soil salinity in the SJV, such as Figure 2, which graphically shows the direct and dramatic impact of drought on a reclaimed soil with a shallow water table that has been left fallow due to the lack of irrigation water. Soil with a shallow water table is typical of the west side of the SJV. In the case of the SJV, drought has a direct impact on salinity development.

The implications of the drought and its impact on soil salinity are self-evident. Less available fresh water for irrigation necessitates a shift to use of impaired water (e.g., municipal, ground and drainage waters) as an alternative source of water and to high-efficiency irrigation systems (e.g., drip, buried drip and micro-sprinkler irrigation systems). Management guidelines for reuse of low-quality water are needed. The potential accumulation of soil salinity in the root zone requires spatial knowledge of soil salinity levels for site-specific management of soil salinity at farm levels to optimize scarce water resources, for the development of water-use and regulatory guidelines at state and federal levels, and for assessing climate change impact trends at global levels. To obtain the necessary information on spatial soil salinity a multiple-scale infrastructure is needed.

6.2 | Minnesota's RRV

Unlike the WSJV, the RRV, which is located in eastern North and South Dakota and western Minnesota, has experienced rainfall that has exceeded the average rainfall in 17 of the last 20 years prior to 2007. The increased rainfall plus a shift in crops from deeper-rooted to more shallow-rooted crops has resulted in rising water tables. Because of the extremely high clay content of the soil in many areas of the RRV (e.g., Kittson County), the capillary rise of moisture from shallow water tables results in the accumulation of salt in the root zone. In addition, salts accumulate in the RRV due to topography. An upslope recharge causes a downslope discharge in downslope areas where a shallow layer of low permeability exists. Lobell et al. (2010) assessed the level of impact of climate change on salinity development in Kittson County, located on the north border of Minnesota in the RRV. The increase and redistribution of salinity within RRV's Kittson County in northwest Minnesota from 1979 to 2007 is shown in Figure 5. Figure 5 indicates how salinity has changed from 1979 to 2007 in western Kittson County of Minnesota's RRV. Areas within the thick solid black line indicate zones of salinity >2 dS m^{-1} obtained from a 1979 salinity survey conducted by hand by the Soil Conservation Service, which is currently the Natural Resource Conservation Service. The grey, blue, yellow and red areas indicate salinity levels of <2, 2–4, 4–8 and > 8 dS m^{-1} from the approach used by Lobell et al. (2010) combining MODIS imagery with ground-truth salinity from EC$_a$-directed soil sampling. From 1979 to 2007 there was an approximately 30% increase in agricultural land with soil salinity greater than 2 dS m^{-1}. If this trend in salinity accumulation continues, then some

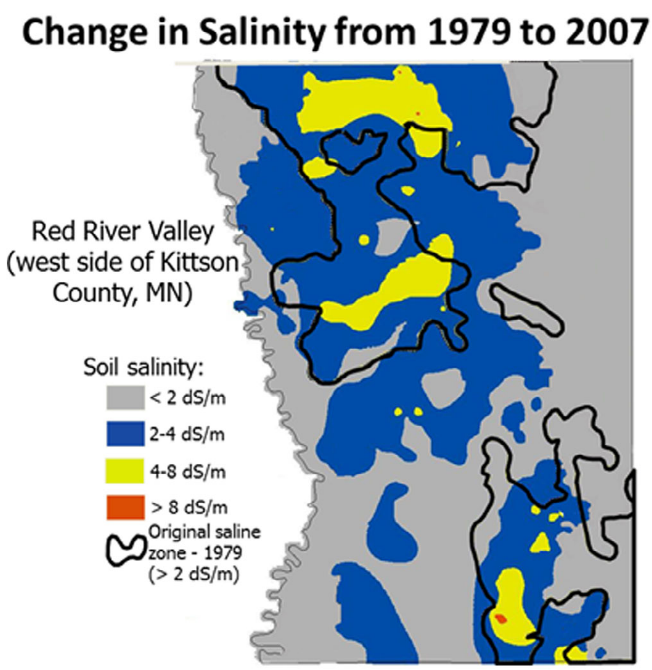

FIGURE 5 Map of the change in soil salinity from 1979 to 2007 for the west side of Red River Valley's Kittson County in northwest Minnesota. *Source*: unpublished data from Lobell et al. (2010)

means of salinity control (e.g., tile drain system) and/or shifts to more salt-tolerant crops will be necessary.

6.3 | Coastal Bangladesh

The coastal areas of Bangladesh are among the most threatened regions by sea level rise and saltwater intrusion. Approximately 30% of Bangladesh's cultivable land is in coastal areas where salinity in rivers, estuaries and soil is influenced by tidal flooding, storm surges and movement of saline groundwater (Haque, 2006). Considerable research has been completed that documents salinity changes in rivers and estuaries in coastal areas of Bangladesh, whether by model simulations (Aerts, Hassan, Savenije, & Khan, 2000; Bhuiyan & Dutta, 2012; Nobi & Das Gupta, 1997) or surveys (Haque, 2006; Mahmood et al., 2010; Sarwar, 2005). The most comprehensive study of seawater intrusion in coastal areas of Bangladesh is a combined modelling and salinity survey study by Dasgupta, Hossain, Huq, and Wheeler (2014, 2015).

Dasgupta et al. (2014, 2015) provide projections of river salinity up to the year of 2050 and provide the most comprehensive assessment of soil salinity influenced by seawater intrusion in Bangladesh. Measured soil salinity distributions bear resemblance to the river salinity distributions, particularly in central Khulna where concentrations of river and soil salinity are high. However, there are instances in coastal areas of Bangladesh where marked differences occur between river and soil salinity, such as in Barrisal where soil salinity is higher than river salinity. The projected trend in soil salinity is for the number of upazilas (upazilas are administrative regions in Bangladesh, which are sub-units of districts) falling into higher soil salinity classes of 3.01–4.00 and >4.01 dS m^{-1} to increase from 12 to 24 and 18 to 27, respectively, whereas those falling into lower salinity classes of 1.01–2.00 and 2.01–3.00 dS m^{-1} will decrease from 21 to 1 and 18 to 17, respectively (Dasgupta et al., 2014). The trend in rising soil salinity within Bangladesh's coastal areas will be accompanied by decreased rice yield.

6.4 | Mediterranean Basin and the Gaza Strip

According to climate models the Mediterranean Basin will increase in winter temperatures combined with altered rainfall patterns and changes in rainfall amount (Mimi & Jamous, 2010). The Gaza Strip, like most arid and semi-arid agricultural areas within the Mediterranean Basin, has serious water-deficit problems regarding both quantity and quality (Qahman, Larabi, Quazar, Naji, & Cheng, 2009; Shomar, Fakher, & Yahya, 2010). Over 70% of the total groundwater extracted in the Gaza Strip is used for agriculture (Ashour & Al-Najar, 2012). Simulations of seawater intrusion by Loáiciga et al. (2012) showed that groundwater extraction is a significant factor in seawater intrusion. The relationship between Na^+ and Cl^- and the spatial variation of ionic ratios of $rCa^{2+}=(rHCO_3^- + rSO_4^{2-})$ in coastal areas of the Gaza Strip show that the aquifer currently used as an irrigation water source exhibited seawater intrusion (Al-Khatib & Al-Najar, 2011; Qahman et al., 2009). It is expected that seawater intrusion in the Gaza Strip and throughout the Mediterranean Basin will be exacerbated by several climate change projections: (a) air temperature will increase from 2.2 to 5.1°C, (b) precipitation will decrease from 4 to 27%, (c) drought periods will increase, with a higher frequency of days exceeding 30°C, and (d) sea level will increase by around 0.35 m, with a concomitant increase in seawater intrusion (Rosa, Marques, & Nunes, 2012). These projected changes in climate for the Mediterranean Basin will have an impact on salinity accumulation due to seawater intrusion, which will impact crop productivity and directly impact crop yield. Ashour and Al-Najar (2012) found that in the Gaza Strip the impact of an increase in salinity on irrigation requirements is considerably higher than the impact of climate change.

6.5 | Australia's Murray-Darling River Basin

The Murray-Darling River Basin (MDRB) extends across 1.06×10^6 km^2 of southeastern Australia and accounts for more than 40% of Australia's agricultural production, estimated at $USD 19 billion (Murray-Darling Basin Authority, 2015). Even though it is dominated by dryland agriculture, irrigated agriculture in the MDRB comprises about 70% of all water used in Australia, with an estimated 1.3×10^6 ha of irrigated land (Crabb, 1997), resulting in $USD 3–5 billion annually in revenues (Bryan & Marvanek, 2004; Cape, 1997).

Observations of climate change in the MDRB include (Schofield, 2011): (a) a general upward trend in temperature since the 1950s, with the greatest warming occurring in spring (about 0.9°C) and lowest in summer (about 0.4°C) and with strongly rising annual maximum and minimum temperatures across the MDRB; (b) there is no clear temporal trend in precipitation across the entire MDRB, but there is a trend of increased drying in the southeastern corner; and (c) there is no clear trend in annual pan evaporation for the entire MDRB, but there is a spatial trend of increasing pan evaporation in the south and decreasing in the north. Projected climate changes in the MDRB include (Schofield, 2011): (a) an expected increase in average surface temperatures throughout the MDRB, with a maximum surface temperature increase of 1–2°C by 2030 and up to 7°C by 2,100; (b) a projected decrease in the future mean annual rainfall in 2030 relative to 1980 by 2% in the north and 5% in the south; and (c) by 2030 ET is predicted to increase by 75–100 mm annually in the far northeast, 25–50 mm in the northwest and 50–75 mm in the central and southeastern parts of the MDRB. Subsequently, climate change will reduce flows in MDRB rivers due to greater water demand to meet increased ET and reduction in rainfall and surface water availability for the southern MDRB. Projections indicate this trend will continue in the future, which will exacerbate the challenges of water over-allocation that currently exist in the MDRB.

Increased water scarcity in the MDRB due to lower rainfall will place greater demand on degraded water supplies, such as saline groundwater. Increased ET and higher salinity concentration of irrigation water will increase the crop water requirement. The combined impact of greater water demand from increased crop water requirement and reduced water allocations from lower precipitation will inevitably reduce agricultural production, which can impact global food supplies and security (Hanjra & Qureshi, 2010).

Surprisingly, no field-, landscape- or regional-scale soil salinity monitoring studies have been conducted in the MDRB to relate soil salinity to altered weather patterns in Australia due to climate change. Only hydrologic and economic modelling studies have been conducted to look at the impact of climate change, which show an increase in soil salinity in the root zone.

Hydrologic modelling studies for the MDRB, such as those by Austin et al. (2010) using the Biophysical Capacity to Change model coupled to climate change scenarios from the CSIRO DARLAM 125 and Cubic Conformal regional climate models and by Beare and Heaney (2002) using the SALSA model, investigated the impacts of climate change on catchment water and salt balances and stream salt concentration. Austin et al. (2010) predicted up to 25% reduction in mean annual rainfall with a similar magnitude of increase in potential ET by 2070, resulting in reductions in mean annual runoff of up to 45% in wetter/cooler southern catchments and up to 64% in drier/hotter western and northern catchments, and reductions in salt yield of up to 34% in southern catchments and up to 49% in western and northern catchments.

More specific to soil salinity-related issues, the economic modelling of Connor, Schwabe, King, and Knapp (2012) indicated that understanding the impact of climate change on agricultural production requires an understanding of how production may adapt to water supply variability and salinity. Without considering salinity, Qureshi, Connor, Kirby, and Mainudin (2007) and Connor, Schwabe, King, and Kaczan (2009) found that adaptation to water scarcity due to climate change in the MDRB included a decreased irrigated area (i.e., more fallow areas) and decreased water application rates. However, Connor et al. (2012) found the water-scarcity incentive to increase water-use efficiency is counteracted by an incentive to leach salts out of the root zone to avoid crop yield decrements due to soil salinity accumulation stemming from higher irrigation efficiency and the use of low-quality irrigation water due to water scarcity.

7 | CONCLUSIONS

There is strong quantifiable evidence indicating that climate change patterns are impacting the salinization of agricultural lands in the SJV and RRV. From the field-scale observations for Westlake Farm, landscape-scale observations for the Broadview Water District and regional-scale estimates of salinity for the WSJV it is surmised that significant increases in root-zone soil salinity are occurring due to impacts from extended drought conditions. Even though the occurrence of above-average annual rainfall in northern California in 2016 ameliorated water-scarcity conditions that existed in California's SJV from 2011 to 2015, there are clear indications that salt accumulation in soil is occurring and is likely to continue to occur on lands with the right conditions (i.e., fallow, shallow water table and fine-textured soil). In contrast, the RRV has experienced excessive rainfall, which has contributed significantly to a rise in the water table and subsequent salinization of the soil profile.

Model simulation efforts that combine biophysical, hydrological and climate change models have helped to further identify arid and semi-arid agricultural regions at risk of developing soil salinity due to alterations in weather patterns, as shown for Bangladesh, the Gaza Strip and the MDRB. However, this approach has the inherent weakness of uncertainty in the models, which affects the reliability of the simulations. Schofield (2011) provided a list of the sources of uncertainty associated with models predicting the impact of climate change: (a) key processes selected for and omitted from the model, (b) different representations of the processes, (c) selection of model parameter values, (d) representation of the interactions between the processes modelled, (e) inclusion of feedback mechanisms, (f) processes not fully understood, (g) model structure influences, (h) influence of different precursor conditions, and (i) downscaling errors. Downscaling global climate models to a regional scale is problematic.

The impact of climate change on salinization has been previously overlooked and needs to be monitored. As a consequence of changes in climate patterns, salt accumulation is most likely to occur in irrigated agricultural areas around the world subjected to extended drought conditions where shallow water tables and fine-textured soils exist and in areas subjected to extensive rainfall where salinity accumulates due to upslope recharge and downslope discharge or where shallow water tables and fine-textured soils exist. Seawater intrusion due to sea level rise and salt-water intrusion due to over-pumping will undoubtedly continue to salinize coastal agricultural areas.

To mitigate the trend of increased root-zone salinity that is expected as a consequence of global climate change, four recommendations are proposed for semi-arid and arid agricultural areas with potential problems due to a shallow water table: (a) develop the technological infrastructure (i.e., software, GIS, spatial database, proximal and remote sensor network, data fusion protocols, etc.) to inventory and monitor soil salinity (and soil properties influencing salt accumulation and movement) across multiple scales, (b) utilize site-specific irrigation management and, where cost effective, site-specific micro-irrigation systems to control salinity identified from inventory and monitoring efforts, (c) couple solute transport models and proximal and remote sensors to develop site-specific irrigation management recommendations, and (d) genetically engineer crops with enhanced salt tolerance.

Water and food security are threatened under climate change because both are vulnerable to altered weather patterns. As pointed out by Teh and Koh (2016), factors including declining freshwater resources, soil salinization, land degradation, population growth, inadequate agricultural infrastructures, plant disease, poor soils and unfavourable climate threaten food security. All of these factors, except population growth, are in turn exacerbated by climate change. Currently, proximal and remote sensors provide the most reliable means to assess salinity from field to regional scales in a timely manner (Corwin & Scudiero, 2016). The methodology to assess soil salinity from field to regional scales is clearly available (Corwin & Scudiero, 2016). However, greater research is needed to provide national and global inventories and to monitor national and global changes in soil salinity over time and space. Once the infrastructure is in place to map and monitor changes in soil salinity locally, nationally and globally, then the mechanism to identify where a salinity problem exists, to what extent and how it is trending provides a targeted means of dealing with a detrimental worldwide environmental and agricultural problem that has plagued civilization for millennia.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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