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Salt-affected soils, reclamation, carbon dynamics, and biochar: a review

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Abstract
Purpose This paper reviews chemical, physical, and biological problems of salt-affected soils and different reclamation methods applied to rehabilitate these soils.
Methods Methods to increase C stocks in these lands are discussed with a focus on biochar application as a potential new approach to not only to increase the C content but also to improve soil properties. Gaps in research knowledge in this field are then identified.
Results Given the concern on the continued worldwide expansion of salt-affected lands and the focus on C sequestration processes, this review has evaluated current knowledge on salt-affected soils and their remediation with organic materials and plants. The review of the published literature has highlighted important gaps in knowledge, which limit our current understanding of rehabilitation of salt-affected soils with organic amendments specially biochar and the associated carbon dynamic. Knowledge about application of biochar in salt-affected soils is scant, and to date, most studies have evaluated biochar use only in nonsalt-affected soils.

Keywords Biochar · Reclamation · Salt-affected soils · Soil carbon

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1 Introduction

Globally, 75 countries have been recognized as having vast areas of salt-affected lands. Martinez-Beltran and Manzur (2005) estimated that nearly 831 million hectares of land are salt-affected worldwide. Salt-affected soils mostly exist in arid and semiarid regions of the world, and many salt-affected wastelands have been productive lands in the past (Qadir et al. 2000). Worldwide, about 95 million hectares of soils are under primary salinization (salt accumulation through natural processes in soils and water) whereas 77 million hectares suffer from secondary salinization (as a result of human activities and ever-rising groundwater) (Metternicht and Zink 2003). Of major concern is that 23 % of the arable lands in the world are affected by salinity and a further 10 % are saline sodic soils while 340 million hectares of lands suffer sodicity (NLWRA 2001; Szabolcs 1994). The high salt concentration negatively affects soil microbial activity as well as soil chemical and physical properties, thus causing a decline in soil productivity. Decline in vegetation growth due to salt toxicity and detrimental osmotic potential results in lower carbon (C) inputs into these soils and further deterioration of their physical and chemical properties (Wong et al. 2009). Therefore, over a long period of soil salinization, C storage decreases at a significant rate (Wong et al. 2010).

Application of organic matter (such as green manures, compost, and food processing wastes) can both ameliorate and increase the C stocks and fertility of saline soils. Biochar is a C-rich organic material produced during slow exothermic thermal decomposition of biomass at temperatures ≤700 °C under zero or low oxygen conditions (Lehman and Joseph 2009; Kwapinski et al. 2010). In recent years with increasing concern about climate change due to elevated anthropogenic CO_2 emissions, research interest on biochar production and its application to soils has gained importance (Woolf et al. 2010). Apart from being a source of carbon, biochar as an organic soil amendment, has been shown to alter and improve physical, chemical, and biological properties of soils and as a result increase plant productivity (Rondon et al. 2007; Asai et al. 2009; Thies and Rilling 2009; Atkinson et al. 2010; Solaiman et al. 2010; Jones et al. 2012). Most biochar studies around the world have been carried out on nonsalt-affected soils (Cross and Sohi 2011; Jones et al. 2012; Herath et al. 2013; Xu et al. 2015). However, a small number of studies carried out on the application of biochar to salt-affected soils have indicated that biochar may also have positive effects of the chemical and biological properties of these problematic soils (Artiola et al. 2012; Lashari et al. 2013; Thomas et al. 2013; Akhtar et al. 2014; Lashari et al. 2014; Akhtar et al. 2015a, b, c; Chaganti et al. 2015a, Chaganti and Crohn 2015b; Hammera et al. 2015). Many review papers have already been published with topics of salt-affected soils, reclamation, and carbon dynamics (Rengasamy 2006, 2010; Wong et al. 2010); however, there is a lack of literature to review the research undertaken on biochar application in salt-affected soils. This article is the first review of exiting literature in this field. The review not only presents an overview of the extent and problems of salt-affected soils, their C content and dynamics, and reclamation through chemical and organic amendments but also specifically examines literature on the behavior of these soils following biochar application. The paper concludes by identifying research gaps in this field.

2 Definition and classification of salt-affected soils

Salt-affected soils are soils with high concentrations of dissolved mineral salts in their profiles such that these dissolved salts adversely affect crop production (Rengasamy 2006; Wong et al. 2010). The salts are primarily composed of carbonates, chlorides, sulfates, and bicarbonates of calcium (Ca^{2+}), magnesium (Mg^{2+}), and sodium (Na^{+}) (Qadir et al. 2000; Manchanda and Garg 2008). Rain, weathering of rocks, application of soil amendments and soluble fertilizers, saline irrigation water, and capillary rise of saline ground water and seawater cause salt accumulation in the soil profile (Rengasamy 2010). Soils generally become salt-affected through primary and secondary salinization processes. Primary salinization includes soils with naturally high amounts of inherent salts from sources such as rock weathering (Rengasamy 2006). In contrast, secondary salinization is a consequence of human activities such as irrigation with saline water without sufficient leaching of salt, thus increasing salt concentration in the root zone (Ghasemi et al. 1995). Secondary salinization also occurs as a result of shallow groundwater tables together with poor drainage and high evaporation rate (Rhoades 1987; Smedema and Shiati 2002; Brinck and Frost 2009). Other contributors to soil salinity include excessive use of chemical fertilizers, overgrazing, and deforestation (Brenstein 1975; Lakhdar et al. 2009)

Salt-affected soils are generally classified on the basis of their electrical conductivity of the saturated extract (EC_e), sodium adsorption ratio (SAR) and exchangeable sodium percentage (ESP), and pH (Richard 1954; Bohn et al. 2001; Rengasamy 2010). Based on these properties, salt-affected soils are thus classified as follows: (1)saline soils, containing high levels of soluble salts and characterized by having high EC_e values (>4 dS m^{-1}); (2) sodic soils, having high levels of exchangeable sodium with SAR >13 and/or ESP (>15); and (3) saline-sodic soils, in which both soluble salts and exchangeable sodium are high, i.e., EC_e>4 dS m^{-1}, SAR>13, and ESP>15 (Ghasemi et al. 1995; Richard 1954). Due to the combined effects of salinity and sodicity on soil properties and plant growth in saline-sodic soils, these soils are considered to be the most degraded form of salt-affected soil (Rengasamy 2002) (Table 1).

3 The effects of salinity and sodicity on different properties of soils

3.1 Physical properties

Elevated levels of exchangeable Na^{+} cause structural deterioration of the affected soils resulting in low pore volume and poor soil-water and soil-air relations in salt-affected soils (Rengasamy and Olsson 1991). Sodicity affects soil hydraulic properties such as hydraulic conductivity and infiltration rate due to aggregate breakdown. Slaking, clay swelling, and dispersion are the main mechanisms involved in aggregate breakdown in sodic soils (Rengasamy and Sumner 1998). Slaking happens during soil wetting when the entrapped air expands and results in the breakup of individual low-strength aggregates. Furthermore, clay swelling shrinks pore sizes, as swelling exceeds the attractive forces of the clay particles. Consequently, they are dispersed into individual clay particles (Frenkel et al. 1978; Rengasamy and Sumner 1998). Dispersion is an irreversible process and can cause the translocation of individual soil particles and consequently the permanent blockage of water navigating pores although swelling is a reversible process (Sumner 1993). The mechanism can be explained by diffuse double layer (DDL) theory whereby intermolecular and electrostatistic forces result in attraction and repulsion between ions in soil solution and soil particles. Soil pore systems and structural stability are influenced by these forces (Quirk 1994; Rengasamy and Olsson 1991). When sodium increases on the exchange sites of the soil particles, the repulsive forces are increased which in turn enlarges the

Table 1 Chemical characteristics of salt and sodium affected soils and the main problems associated with these soils

Soil salinity class	EC_e (dS m^{-1})	pH	ESP	SAR_e	$SAR_{1:5}$	Problems
Normal soil	<4.0	6–8	<5	<3	<5	Can have problems other than salts
Saline soil	>4.0	6–8	<15	<13	<5	- Osmotic effects - Possible toxicity of dominant anion or cation at high EC
Saline-sodic soil	>4.0	>8	>15	>13	>5	- Osmotic effects - HCO_3^- and CO_3^{2-} toxicity - Nutrient deficiency high pH - Organic matter losses - Decline in plant growth - Slaking, swelling, dispersion, surface crust, and hard setting - Increased carbon loss
Sodic soil	<4.0	>8	>15	>13	>5	- HCO_3^- and CO_3^{2-} toxicity - Nutrient deficiency high pH - Organic matter losses - Decline in plant growth - Slaking, swelling, dispersion, surface crust, and hard setting - Increased carbon loss - Seasonal waterlogging - Na^+ toxicity

$SAR_{1:5}$ sodium adsorption ration measured in 1:5 soil/water extract, *SAR_e* sodium adsorption ration measured in saturated extract

interparticulate distance in the DDL and brings about dispersion and breakdown of soil aggregates which negatively affects soil structure (Oster and Shainberg 2001). Soil hydraulic conductivity and infiltration also decrease due to swelling and dispersion in Na^+-dominated soils.

Total electrolyte concentration (TEC) of irrigation water and soil solution also affects clay swelling and dispersion. Soil flocculation is favored by high electrolyte concentration of the soil solution; therefore, higher levels of salinity can improve permeability and positively affect soil structure on. Conversely, low salinity and high sodicity result in considerable reduction of infiltration and hydraulic conductivity by inducing swelling and dispersion (Quirk and Schofield 1955; McNeal et al. 1968; Frenkel et al. 1978; Shainberg and Lety 1984; Quirk 2001; Dikinya et al. 2006).

Following slaking and dispersion in the surface layer of sodic soils, a thin layer of high shear strength is formed after drying which is known as a "surface crust" (Agassi et al. 1981). This surface crust makes the surface soil susceptible to extreme erosion and also to waterlogged conditions (Moore and Singer 1990; Shainberg et al. 1992). "Hard setting" is another important characteristic of saline-sodic soils which functionally is very similar to surface crusting, but its formation processes take place in the lower depths of the soil profile rather than on the soil surface, thus creating an impermeable subsoil layer with high bulk density (Qadir and Schubert 2002). Loss of structure and reduction in transmitting capacity for air and water in soils with high sodium concentration will therefore result in the formation of massive structure, poorly aerated and waterlogged soils. Such conditions are unfavorable for the establishment and the growth of the plants (Nelson et al. 1998).

3.2 Chemical properties

Soils high in salinity and sodicity have high values of EC_e, ESP, SAR, and pH (Table 1). Salt-affected soils generally suffer from deficiencies of nitrogen (N), phosphorus (P), and potassium (K). However, their high pH also adversely affects the availability of micronutrients such as Fe, Al, Zn, Mn, and Cu (Pessarakli and Szabolcs 1999; Lakhdar et al. 2009).

Decline in vegetation growth due to salt toxicity, high osmotic suction, and degraded soil structure of salt-affected soils results in lower C inputs in these soils and further deterioration of their physical and chemical properties (Wong et al. 2009). Surface crusting and sealing, as discussed previously, also cause considerable erosive losses of organic matter so organic matter is low in these soils due to both low input and high losses (Qadir et al. 1997; Nelson and Oades 1998). Chander et al. (1994) showed that with increasing soil sodicity, due to irrigation with sodium-rich water, total N and organic C decreased. Adu and Oades (1978) also proposed that increased dispersion of aggregates in sodic soils exposes their locked up organic matter to rapid decomposition by soil microbes.

Gandhi and Paliwal (1976) reported that by increasing salinity, N mineralization decreased and gaseous NH_3 losses increased. Pathak and Rao (1998) found that increasing salinity and sodicity of arid soils treated with organic amendments led to decreased C and N mineralization. Nelson et al. (1996) however reported an increase in C decomposition in salt-affected soils with increasing sodicity, due to solubilization of organic matter, whereas salinity decreased C mineralization. Frankenberger and Bingham (1982) found that the activity of soil enzymes with a role in C, N, P, and S cycles decreased with increasing salinity. High ion concentration, primarily Na^+ and Cl^-, reduces the ability of plants to take up water, which in turn affects the plant cells and growth (Muneer and Oades 1989a).

3.3 Biological properties

Soil microbial and biochemical processes, which are important for maintaining soil ecological functions, are negatively affected by changes in soil chemistry (Rietz and Haynes 2003). Microbial growth and activity are adversely impacted by increasing soil salinity, as high salt concentrations in soils causes' osmotic stress and dehydration of microbial cells (Oren 1999; Wichern et al. 2006). Furthermore, in addition to salt stress, Na^+ toxicity; nutritional deficiency such as Ca^{2+} deficiency; toxic levels of other ions such as carbonate, bicarbonate, and chloride; and loss of organic matter because of structural degradation all significantly contribute to decreasing microbial populations and activities in salt-affected soils (Zahran 1997; Nelson and Oades 1998). Garcia et al. (1994) studied the effects of salinity on the composition of microbial community in soils collected from arid regions of south east Spain and found increasing salinity had a negative effect on soil microbial communities. McClung and Frankenberger (1985) reported that enzyme activities and C and N mineralization rates decreased at high salinity levels. In their studies, nitrification rate increased up to 83 % with increasing salinity to 20 dS m^{-1}, which stimulated ammonia losses through volatilization. Wichern et al. (2006) proposed that fungal communities are more exposed to increasing salt concentration than bacterial populations. Rietz and Haynes (2003) also reported that soil microbial and biochemical activities were negatively affected by irrigation-induced salinity and sodicity. They observed an exponential decrease in microbial biomass C with increasing soil EC_e and a linear decrease in biomass C with increasing ESP and SAR. They also evaluated various biochemical enzyme activities and reported that enzyme activities linearly decreased with increasing EC_e, ESP, and SAR. Several other studies have also reported that salinity and sodicity significantly decrease the soil microbial biomass and related enzyme activities (Garcia and Hernandez 1996; Tripathi et al. 2006, 2007). Rietz and Haynes (2003) and Wong et al. (2008) showed that with increasing salinity and sodicity, metabolic quotient (respiration per unit biomass) increases, indicating a more stressed microbial community. Gollarata and Raiesi (2007) also found an increase in metabolic quotient with increase in soil salinity. Similarly, Yuan et al. (2007) observed that with increasing salinity, a shift in soil microbial community takes place, with lower metabolisms, which can be an adaptive mechanism to reduce salt stress. Soil ecological functions are significantly affected by microbial and biochemical activities. These activities also play a central role in enhancing soil structure by increasing and improving formation and stabilization of soil aggregations (Six et al. 2004).

4 The effects of salinity and sodicity on soil carbon (C) dynamics

4.1 Soil organic C pools

The largest organic C pool of the continental biosphere is soil organic matter (SOM), with 1550 Pg accumulated in soil over thousands of years. It is about three times that of vegetation C (560–650 Pg) and higher than the atmospheric C pool (750 Pg) (Lal et al. 2003). The soil C pool comprises two parts: soil organic carbon (SOC) and soil inorganic carbon (SIC). The SOC is the most common form of C in humid regions while the SIC pool dominates in soils of arid and semi-arid regions. Based on the turnover time of C in soil, SOC can be partitioned into three main pools (Jenkinson and Raynor 1977; Parton et al. 1987):

1. The active pool which is largely controlled by residue inputs and climate. This can be a nutrient source for plants. This pool is made up of readily oxidizable materials, and its turnover time is in the order of weeks (Schnurer et al. 1985; Wong et al. 2009).
2. The slow pool with a turnover time that is in the order of decades. It contains particulate organic C and moderately decomposable material within aggregates (Parton et al. 1987; Wong et al. 2010).
3. The passive pool with a turnover time in the order of millennia. Group 3 is also called the recalcitrant pool and contains stable C that is chemically resistant to further microbial degradation. Charcoal is one of the largest sources of this pool (Schimel et al. 1994; Skjemstad et al. 1996; Clough and Skjemstad 2000; Skjemstad et al. 2002; Lehmann et al. 2008).

The SIC pool includes carbonate minerals such as gypsum and calcite. This pool comprises two predominant components:

1. Lithogenic or primary carbonates which are formed after weathering of parent material
2. Pedogenic or secondary carbonates formed after dissolution of CO_2 in soil water which produces carbonic acid and re-

precipitation with Ca^{+2} or Mg^{+2} added to soil from different sources (Lal et al. 2006). In Australia, pedogenic carbonate often occurs in southern and inland regions and is estimated to occur in 50 % of the landscape. It is usually found in conjunction with sodic soils (Fitzpatrick and Merry 2000). The formation of pedogenic $CaCO_3$ is linked to the development of sodicity in soils (Pal et al. 2000).

4.2 Carbon in salt-affected soils

It is generally accepted that once organic matter becomes an integral part of soil aggregates through the processes of aggregation, it is physically protected from rapid decomposition. Thus, any processes which contribute to the breakdown of soil aggregates also contribute to organic matter decomposition. For instance, tillage increases the disruption of aggregates and consequently raises the organic C mineralization rate (Wong et al. 2010). The loss of SOM is directly linked to the factors which affect its accessibility by the microbial population and enzymes (Dalal and Mayer 1986). In sodic soils, wetting and drying processes affect C accessibility in different ways. When sodic soils are wetted, dispersion of aggregates increases the availability of SOM and accelerates C loss, while on drying, SOM availability decreases due to an increase in soil bulk density and a decrease in water-holding capacity (Muneer and Oades 1989b). After solubilization of SOM, the initial increase in substrate availability can put some environmental stress on soil microbial communities and decrease the adverse effects of NaCl on microorganisms (McCormic and Wolf 1980; Pathak and Rao 1998). Wong et al. (2008) reported a higher soil microbial biomass (SMB) in soils with high salinity treatments. They related this increase in SMB to increased decomposability and accessibility of SOM which resulted in an increase in substrate. In another study, Wong et al. (2009) found that an initial increase in SMB and respiration rate happened after the addition of organic material to a highly saline-sodic soil. However, after the initial flush in available substrate and therefore increasing SMB, the salt and Na stress decrease substrate decomposition and availability. Consequently over time with microbial adaptation to the high saline environment, microbes continue to mineralize the remaining SOM (Poloneko et al. 1981; Zahran 1997). In another study, Setia et al. (2011) found EC to have a negative impact on CO_2 emission in salt-affected landscapes, so in estimating CO_2 release from these lands, this parameter needs to be taken into account.

In general, high salinity has a similar effect on both plant health and microbial activity through ion toxicity and osmotic effects (Batra and Manna 1997). Therefore, over a long period of time, where the salinization and sodicity processes are taking place, C storage decreases (Wong et al. 2010). In a study of arid saline soils in Spain (Garcia et al. 1994), low microbial activity was observed in a degraded land where high EC levels inhibited respiration. Similarly, they found that salinity decreased soil microbial biomass and concluded that this decline in soil microbial activity was due to a change in microbial community structure from fungi-dominated to bacteria-dominated. The latter are less diverse, competitive, and active (Garcia et al. 1994; Pankhurst et al. 2001; Sadinha et al. 2003). Chander et al. (1994) observed that SMB decreased with increasing sodicity, while the rate of organic matter mineralization increased. They concluded that increasing sodicity places stress on plants and decreases plant carbon inputs. In contrast, a slightly negative effect of sodicity on mineralization was reported by Nelson et al. (1997). This contradiction may be caused by the varying quantity and quality of added substrate in these two studies.

Soil enzyme activities have also been reported to decrease with increasing levels of salinity (Batra and Manna 1997). The inhibition of microbial and enzymatic activity is higher with NaCl than with Na_2SO_4 or $CaCl_2$ (Frankenberger and Bingham 1982). Laura (1973) also found that increasing Na_2CO_3 concentration led to a decrease in total carbon (TC). Exchangeable Na increases due to an increase in Na_2CO_3 concentration which results in higher ESP values. On the other hand, precipitation of $CaCO_3$ and $MgCO_3$ increases pH values. These two processes (increased ESP and precipitation of $CaCO_3$ and $MgCO_3$) decrease SOC stocks in salt-affected soils. In contrast, Pathak and Rao (1998) found that even in highly saline and sodic conditions, biochemical mineralization can occur by soil enzymes. Wong et al. (2008) showed that SOC stocks in scalded and eroded soils were about a third of those in vegetated soils due to very low inputs of plant biomass carbon. Similarly, Pankhurst et al. (2001) attributed low SOC levels in saline soils to reduce presence of plant cover. Wong et al. (2008) reported that revegetation of scalded and eroded areas can increase SOC stocks to the similar levels of native pastures after about 10 years (Table 2). Leguminous trees can decrease ESP and pH and increase SMB of the salt-affected soils once they are established in such soils (Bhojvaid and Timmer 1998; Garg 1999; Mishra and Sharma 2003).

Erosional processes are very common in salt-affected landscapes. During these processes, SOC can be removed from the surface layer. The particulate fraction of SOC is the most prone to being removed because of its low density (Lal 2001). In salt-affected scalded soils, the less fertile B horizon often gets exposed after removal of the A horizon by erosion (Murphy et al. 1998). The eroded materials either get deposited in the downslope section of the landscape or are transported to aquatic environments such as lakes and reservoirs causing water pollution. Such transported organic matter can be protected from decomposition in the water bodies, thus contributing to C sequestration (Izaurralde et al. 2001; McCarty and Ritchie 2002). Table 2 presents data on C

Table 2 Soil organic carbon content (SOC) in some salt-affected soils

EC (dS m^{-1})	ESP	SOC (%)	Soil depth (m)	References
0.5 (EC_e)	2.0	0.78	0–0.15	Pathak and Rao (1998)
0.5 (EC_e)	3.8	0.40	0–0.15	
1.5 (EC_e)	17.7	0.42	0–0.15	
2.5 (EC_e)	65.1	0.26	0–0.15	
6.4 (EC_e)	88.8	0.32	0–0.15	
6.5 ($EC_{1:5}$)	13	0.7	0–0.15	Pankhurst et al. (2001)
4.0 ($EC_{1:5}$)	25	1.9	0–0.15	
1.0 ($EC_{1:5}$)	11	2.3	0–0.15	
0.3 ($EC_{1:5}$)	10	0.8	0–0.15	
0.1 ($EC_{1:5}$)	6	1.4	0–0.15	
0.1 ($EC_{1:5}$)	3	1.5	0–0.15	
nd	15.7	0.63	0–0.25	Tejada et al. (2006)
2.2 (EC_e)	nd	14.1	0–0.20	Tripathi et al. (2006)
3.5 (EC_e)	nd	8.7	0–0.20	
14.7 (EC_e)	nd	10.2	0–0.20	
16.3 (EC_e)	nd	6.9	0–0.20	
15.7 (EC_e)	nd	5.2	0–0.20	
6.4 (EC_e)	nd	8.9	0–0.20	
7.1 (EC_e)	nd	10.4	0–0.20	
3.1 (EC_e)	nd	10.7	0–0.20	
5.3 (EC_e)	nd	10.3	0–0.20	
0.2 ($EC_{1:5}$)	20.3	0.3	0–0.05	Wong et al. (2008)
0.2 ($EC_{1:5}$)	7.9	2.3	0–0.05	
0.2 ($EC_{1:5}$)	5.2	1.5	0–0.05	
0.2 ($EC_{1:5}$)	3.6	2.7	0–0.05	
0.1 ($EC_{1:5}$)	61.2	0.2	0–0.05	
0.1 ($EC_{1:5}$)	3.3	2.0	0–0.05	
0.1 ($EC_{1:5}$)	1.2	2.4	0–0.05	

EC_e electrical conductivity of saturated extract, *$EC_{1:5}$* electrical conductivity in 1:5 soil/water extract, *ESP* exchangeable sodium percentage, *nd* no data exist, *SOC* soil organic carbon

content of salt-affected soil in some different studies on these soils. Based on these data, SOC has decreased with increasing EC and ESP/SAR.

5 Carbon dynamics and reclamation of salt-affected soils

Different reclamation processes are used for saline and sodic soils. Leaching of salts from saline soils is the most commonly used method of reclaiming these soils (Jury et al. 1979; Abrol et al. 1988; Tanton et al. 1988; Djedidi et al. 2005; Letey et al. 2011; Yan and Marschner 2013). This involves infiltration of good quality water through the soil profile, together with the provision of an adequate drainage system to remove the leached salts away from the reclaimed area.

On the other hand, maintaining sustainability of water resources is a worldwide priority for a growing population. Population growth and urbanization have imposed extreme pressures on water resources resulting in reduced availability of fresh water for irrigated agriculture (Levine and Asano 2004; Qadir and Oster 2004). Therefore, current and future generations need to use water resources very efficiently. However, maintaining or even increasing agricultural productivity is also very important to meet the food demands of the planet's growing population. Therefore, reducing the acreage under irrigation is not an option as a considerable amount of land is already lost due to various land degradation problems. Due to this fact, alternatives to using potable water resources for irrigation have received increased attention. As a consequence, treated waste waters from municipal treatment plants (reclaimed water), agricultural drainage waters, and agricultural and urban runoff are often considered as viable options for irrigation waters or to meet other nonpotable water demands (Corwin and Bradford 2008; Oster 1994). These waters are generally referred to as "degraded waters" due to their deterioration in physical, chemical, and biological properties (O'Connor et al. 2008). Reuse of such waters helps to reduce or prevent their discharge into water bodies, thus reducing their degradation effects on receiving environments (Grattan et al. 2008; Toze 2006). In some countries such as the USA, there is high reuse of agricultural drainage and reclaimed waters to irrigate agricultural lands (Kinney et al. 2006; Wu et al. 2009). However, when such waters are diverted for agricultural irrigation the chemical constituents, most importantly salts, can accumulate in soils causing soil degradation such as the loss of soil structure and reduction in soil infiltration and permeability (Stevens et al. 2003). Constraints involved in using these low-quality waters are mostly with respect to their salinity and sodicity (SAR) (Suarez et al. 2006).

Application of waters with high salinity and sodicity and their consequent effects on soil properties have been long studied and are well documented (Chander et al. 1994; Beltran 1999; Grattan and Oster 2003; Rietz and Haynes 2003; Emdad et al. 2004; Suarez et al. 2006; Choudhary et al. 2011), but some research gaps still remain unsolved. For example, while many researchers have studied the effects of high SAR (>15) waters on soil properties, the use of relatively moderate SAR (<8) waters, such as reclaimed waters, has been less investigated especially for leaching salt-affected soils (Mace and Amrhein 2001; Mandal et al. 2008). Understanding the effect of using water with moderate SAR to leach a saline-sodic soils treated with organic amendments is important with respect to soil and water management and thus warrants further study.

5.1 Chemical reclamation

Reclamation of saline-sodic soils requires the replacement of Na^+ on the soil exchange sites by divalent cations such as Ca^{2+}, followed by a salt leaching process (Gupta and Abrol

1990). Na-organic compounds are very soluble in saline-sodic soils, resulting in dissolved Na-humates in percolating and runoff waters which increase dispersion, mobilization, and losses of clay particles (Sumner et al. 1998). In highly alkaline soils with low base saturation status and high Na_2CO_3contents, Na^+ must first be replaced by Ca^{2+} to form a stable linkage between organic matter and Ca^{2+}-saturated particles (Rengasamy and Olsson 1991). This is because organomineral interactions mainly depend on Ca^{2+} bridges rather than Na^+, and sodic soils are not able to retain products of decomposition due to the higher proportion of Na^+ on their exchange sites compared to Ca^{2+} (Naidu and Rengasamy 1993).

Gypsum is commonly used to supply Ca^{2+}. The effect of gypsum on the reclamation of saline-sodic soils has been extensively studied (Armstrong and Tanton 1992; Ilyas et al. 1997; Oster et al. 1999; Ghafoor et al. 2001; Mace and Amrhein 2001; Qadir et al. 2001a; Lebron et al. 2002; Choudhary et al. 2004; Gharaibeh et al. 2009, 2010). Sulfur and sulfuric acid are the other commonly used inorganic amendments which upon application to calcareous soil increase soil Ca^{2+} levels by dissolving native calcium carbonate in these soils (Niazi et al. 2001; Amezketa et al. 2005; Sadiq et al. 2007; Vance et al. 2008).

5.2 Phytoreclamation

Phytoremediation or vegetative bioremediation of salt-affected soils was introduced as early as 1937 by Kelly (1937) when a sodic soil was ameliorated by bermuda grass. Since then, many researchers have studied the potential of crop and crop-based approaches in the reclamation of salt-affected soils (Robbins 1986; Qadir et al. 1996; Qadir et al. 2001b; Qadir and Schubert 2002; Ahmad et al. 2006; Ammari et al. 2008; Gharaibeh et al. 2011). Phytoremediation occurs through several mechanisms in salt-affected soils (Qadir et al. 2007) including the following:

1. Plant roots increase the dissolution rate of calcite, resulting in enhanced levels of Ca^{2+} in soil solution and increased exchange with Na^+. Many sodic and saline-sodic soils have an inherently good source of Ca^{2+}, typically in the form of calcite ($CaCO_3$), at varying depths within the profile. However, unlike other Ca^{2+} sources used in the amelioration of sodic and saline-sodic soils, calcite is not sufficiently soluble to effect the displacement of Na^+ from the cation exchange complex, unless remediated by plants. Production of Ca^{2+} in the soil solution by formation of carbonic acid and solubilization of native calcite occurs due to an increase in partial pressure of CO_2 (PCO_2) in the root zone. Enhanced levels of P_{CO2} and H^+ therefore assist in increasing the dissolution rate of calcite (Qadir et al. 2005).
2. The generation of protons (H^+) released by roots of certain plant species enhances Na^+ uptake by plants and its subsequent removal from the field at harvest.
3. Improving drainage and salt leaching through the pores formed by roots and root penetration. Ghaly (2002) found that after the second year of a field experiment the role of native grass species in reducing salt content was more effective than the gypsum only treatment. He attributed this to enhanced salt uptake by plant. This was evidenced firstly by higher carbon inputs to soil resulting in reclamation of these soils within 2 years and secondly by increased amount of Na^+ uptake and accumulation in the grass shoots.

5.3 Reclamation by organic amendment

Organic matter application is another method used to ameliorate salt-affected soils as organic matter helps binding soil particles into aggregates (Nelson and Oades 1998). Deprotonation of humic and fluvic acids leads to formation of large organic polyanions. These can bind clay particles into microaggregates by forming $[(Cl\text{-}P\text{-}OM)_x]_y$ complexes where Cl, P, and OM are clay particles, polyvalent cations, and organic matter, respectively (Edward and Bremner 1967; Tisdall and Oades 1982). Chemical, physical, and biological properties of soils can be affected by organic material addition. For instance, poultry manure and compost addition to a salt-affected soil can affect its chemical properties, enhancing both CEC and the soluble and exchangeable K^+. Potassium competes with Na^+ in terms of being adsorbed in sodic soils and will limit the entry of Na^+ onto the exchange sites (Walker and Bernal 2008). Chorom and Rengasamy (1997) added green manure to an alkaline sodic soil and observed that decomposition and microbial respiration decreased pH and increased the solubility of $CaCO_3$ due to increase in partial pressure of CO_2. Similarly, Liang et al. (2003) reported significant stimulation of soil respiration, urease, and alkaline phosphatase activity after incorporation of organic manure to a soil derived from alluvial and marine deposits with 3.3 g kg^{-1} total salts. Also, Barzegar et al. (1997) observed that dispersible clay decreases after the addition of pea straw irrespective of SAR. They concluded that organic material addition to sodic soils can enhance structural stability even without initial reclamation of sodicity. Combined application of gypsum and organic matter has also been successful in decreasing ESP and reducing adverse effects of high Na^+ ions. Vance et al. (1998) observed that combined application of organic matter and gypsum to the surface of saline-sodic soils reduced dispersion and EC more than gypsum alone.

Application of organic amendments is considered as an effective management strategy for both amelioration of salt-affected soils and improvement of plant growth. Therefore, organic amendments such as green manures, farmyard manures, and municipal solid wastes provide both soil nutrients and organic matter for lasting improvements in soil fertility. Soil salinity effects on organic matter turnover and the mineralization of C and N depend on the type of organic material incorporated (Walpola and Arunakumara 2010). For example, combined application of rice straw and pig manure affected enzymatic and microbial activity more significantly than applying these amendments alone (Liang et al. 2005). Application of organic material can therefore provide an effective approach to reducing toxic saline conditions. However, excessive use of some organic amendments, for instance chicken manure, can increase the risk of secondary salinization in regions with abundant rainfall. Therefore, important factors to be considered prior to using organic amendments are the following: the proper selection of organic amendment and appropriate timing and correct method of application to the soil (Diacono and Montemurro 2010).

6 Biochar as an organic soil amendment

6.1 Biochar's definition and characteristics

Biochar is a C-rich organic material produced during slow exothermic decomposition of biomass at temperatures ≤700 °C under zero oxygen or low oxygen conditions (Lehman and Joseph 2009; Kwapinski et al. 2010). In recent years, with increasing concern regarding climate change due to elevated anthropogenic CO_2 emissions, research interest in biochar production and application has gained importance (Woolf et al. 2010). This product is highly recalcitrant due to its high aromaticity and can sequester C for very long periods (Fang et al. 2014; Kuzyakov et al. 2014; Lehmann et al. 2006). Therefore, biochar addition can be a potential pathway to improve and enhance native SOC and black C stabilization in cropping systems. For instance, Zhang et al. (2015) studied the effects of crushed corncob biochar application on aggregate-associated C concentrations in a 1-year field experiment. They observed higher C accumulation in large macroaggregates, suggesting that biochar-derived C can be physically protected within these aggregate sizes. Many studies have been undertaken on the effect of biochar application on SOC decomposition to find solutions for enhancing soil C sequestration (Wardle et al. 2008; Jones et al. 2011; Luo et al. 2011). However, previous studies have shown that biochar may both suppress and stimulate native SOC decomposition (Liang et al. 2010; Cross and Sohi 2011; Luo et al. 2011). Differences in the nature of soil and biochar, and incubation conditions used in different studies have led to these inconsistent results (Jones et al. 2011). Apart from being a C source, biochar as an organic soil amendment, has been shown to enhance plant growth either directly or indirectly. Biochar acts as a direct source of mineral nutrients such as Ca, Mg, K, and P, which promotes plant growth. Biochar also has a high adsorption capacity itself (determined by source feedstock and production conditions) and can be used in remediating contaminated soils (Uchimiya et al. 2011; Zhang et al. 2013; Paz-Ferreiro et al. 2014). However, biochar has indirect effects on mechanisms that alter physical, chemical, and biological properties of soils and consequently increase plant productivity (Rondon et al. 2007; Asai et al. 2009; Chan and Xu 2009; Spokas et al. 2009; Thies and Rilling 2009; Atkinson et al. 2010; Laird et al. 2010; Solaiman et al. 2010; Van Zwieten et al. 2010a; Lehman et al. 2011; Jones et al. 2012). Most biochar studies have been carried out on nonsalt-affected soils, where biochar application has resulted in the improvement of soil physical properties such as, infiltration, hydraulic conductivity, degree of aggregation, aggregate stability, bulk density, and water-holding capacity (Asai et al. 2009; Verheijen et al. 2010; Karhu et al. 2011; Uzoma et al. 2011; Liu et al. 2012; Jien and Wang 2013). For instance, Jien and Wang (2013) observed that application of 5 % biochar to a highly weathered soil increased macroaggregation of the soil and its saturated hydraulic conductivity (K_s). Similarly, Asai et al. (2009) observed a twofold increase in soil saturated hydraulic conductivity after the application of biochar at a rate of 16 t ha^{-1}. Mukherjee and Lal (2013), Goerge et al. (2012), and Busscher et al. (2011) reported that soil aggregation increased after the addition of biochar. Piccolo et al. (1996) and (1997) showed that the primary binding agents responsible for enhancing soil aggregation are coal-derived humic substances. In their experiment, soil macroaggregate stability increased between 20 and 130 %, when coal-derived humic substances were added to four experimental soils from southern Nigeria.

Biochar influences different properties of soils under various mechanisms of fragmentation, oxidation, decay, and carboxylation (Kookana et al. 2011). For instance, biochar addition may decrease N availability by increasing N immobilization due to its effects on soil microbial activity, pH and CEC (Van Zwieten et al. 2010b; Reverchon et al. 2014, 2015). On the other hand, biochar may have no effect on soil N availability (DeLuca et al. 2006) or may even increase N availability (Hosseini Bai et al. 2015). The latter studied N cycling (using $\delta^{15}N$) and nutrient availability in soil and plant after 5 years of applying poultry litter and green waste biochar. A significant increase in both soil and foliar $\delta^{15}N$ was observed with poultry litter biochar addition compared to green waste biochar and control treatments.

Biochar also has variable effects on phosphorus (P) availability. Slavich et al. (2013) observed an increase in available P after application of biochar derived from feedlot manure, while in another study, wood-based biochar addition noticeably reduced the availability of P (Van Zwieten et al. 2010b). This confirms that both soil properties and biochar feedstock are the important factors affecting biochar performance in soils (Macdonald et al. 2014).

6.2 Biochar and salt-affected soils

As previously stated, the addition of divalent cations such as Ca^{2+} and Mg^{2+} is critical in the reclamation of saline-sodic soils to offset excessive exchangeable Na^+ and biochar may be able to play a positive role in this respect. In one study, Major et al. (2010) observed that the availability of Ca^{2+} and Mg^{2+} increased after the addition of biochar at a rate of 20 t ha^{-1} to a Columbian savanna oxisol. Likewise, Laird et al. (2010) observed that Ca^{2+} levels increased after the addition of oak-derived biochar to a Midwestern agricultural soil. Similarly, other researchers have also shown that divalent cation concentrations in soils increase after biochar addition (Chan et al. 2008; Gaskin et al. 2010; Novak et al. 2009). However, very few studies have investigated if biochar also has positive physical and chemical effects if applied to salt-affected soils.

Lashari et al. (2013) in a study on a salt-stressed cropland used biochar-manure compost in conjunction with pyroligneous solution. The results showed that soil pH and salt and sodium contents significantly decreased in amended treatments compared to the control. Also, they found significantly higher SOC and available P in amended soils. They concluded that biochar can be used as an ameliorant in salt-affected soils to reduce salinity (sodicity) stress by adsorption of Na^+. Also, Lashari et al. (2014) in a similar study reported that a combined amelioration of manure compost and crop straw biochar plus pyroligneous solution could help reduce salinity stress to maize and improve productivity in salt-affected croplands. Furthermore, Akhtar et al. (2015a) studied the effects of biochar on growth, physiology, and yield of pot-grown wheat under salinity stress and also the underlying mechanism on reducing Na^+ uptake. They reported positive effects of biochar on growth and yield of wheat. Also, biochar significantly affected the concentrations of Na^+, K^+, Ca^{2+}, and Mg^{2+} in the leachate. The concentration of Na^+ was not affected by biochar addition in nonsaline soils; however, in saline conditions, significant reduction of Na^+ concentration was observed, indicating higher Na^+ adsorption potential by biochar compared to control treatments. Also, increased amounts of Ca^{2+} and Mg^{2+} were reported in biochar-amended treatments. They concluded that the enrichment of exchangeable sites of soil profile with Ca^{2+} and Mg^{2+} can decrease the exchangeable Na^+ concentration in these sites and improve soil physical properties in salt-affected soils which are relevant to reclamation of these soils. Similarly, another study by Akhtar et al. (2015b) reported that application of biochar to salt-affected soils can mitigate the salinity stress in potatoes due to biochar's high Na^+ adsorption potential. Thomas et al. (2013), in a glass house experiment, added salts on the surface of biochar (derived from lignocellulosic material) and concluded that biochar could ameliorate salt stress and mitigate its effect on plant performance through salt sorption. In this study, biochar material showed significantly increased electrical conductivity following salt addition. They also concluded that the observed increase in water-holding capacity of the biochar-amended treatments is another important factor in mitigating osmotic effects and ionic toxicity of excessive salts in soils. Hammera et al. (2015) reported similar results when they applied pellets of coniferous wood chips biochar along with arbuscular mycorrhizal (AM) fungi to salt-stressed soils in a greenhouse experiment with *Lactuca sativa*. They indicated that the biochar reduced salt stress by its ion sorption capacity, but additionally, it improved the nutritional condition of the soil which may be important variables in explaining the positive effects of biochar in salt-affected soils. They also noted the dependency of plants on symbiotic microorganisms to fully utilize the potential of biochar benefits in soils. The authors suggested a joint application of AM fungi and biochar in agricultural soils, after observing the positive effect of their combined use in increasing plant growth.

Biochar addition also promotes biological activity in soils. Although biochar is an organic amendment, it is more recalcitrant than any other organic substrate; thus, it less likely supports microbial growth. However, it has been shown that the release of temporary labile pyrolysis products can significantly promote soil biological activity after biochar addition to soils, although this effect could be temporary (Lehman et al. 2011). Biochar as a pure C source can also increase soil C stocks once added to soil. Providing a rich source of C for microbes is another possible reason for observed increases in biological activities in salt-affected soils after biochar addition. However, previous studies on the effects of biochar on microbial biomass carbon (MBC) are inconsistent. In some studies, there was no significant effect of biochar application on soil MBC (Castaldi et al. 2011; Zavalloni et al. 2011) while in others, biochar addition significantly decreased soil MBC (Dempster et al. 2012). In contrast, some other studies (Kolb et al. 2009; Liang et al. 2010; Lehman et al. 2011; Zhang et al. 2014) have found positive effects of biochar addition on microbial biomass. However, in general, MBC reflects any changes in SOC content and decomposition. Thus, any processes and materials which increase or decrease C content in soil can affect biomass and activity of microbial community. Therefore, biochar, as a pure C source, can provide more C for the microbial community and increase MBC.

In reclamation of salt-affected soils with biochar, it is critical to consider both the soil and biochar properties. Soil texture, the level of salinity (and sodicity), nutrient concentrations, and soil native C content are important soil properties which need to be considered before commencing reclamation processes. Also, the type of biochar (acidic or alkaline) and the source feedstock used for producing biochar are key factors determining the effectiveness of biochar as an organic

Table 3 Potential biochar effects on different properties of salt-affected soils

Soil salinity class	EC_e (dS m^{-1})	pH	ESP	SAR_e	Potential biochar effects or unknowns
Saline soil	>4.0	6–8	<15	<13	1. Decreasing/increasing electrical conductivity (EC) depending on the nature of the applied biochar 2. Increasing SOC stocks 3. Improving plant establishment and growth and increasing crop yield
Saline-sodic soil	>4.0	>8	>15	>13	1. Increasing SOC stocks 2. Increasing/decreasing soil pH and SAR, depending on the nature and the source of biochar which affects the biochar pH 3. Increasing water-holding capacity (WHC) and hydraulic conductivity (K_s)
Sodic soil	<4.0	>8	>15	>13	1. Increasing total and aggregate associated SOC 2. Increasing WHC and hydraulic conductivity (K_s) 3. Increasing microbial biomass carbon (MBC)

EC_e electrical conductivity in saturated extract, *$EC_{1:5}$* electrical conductivity in 1:5 soil: water extract, *ESP* exchangeable sodium percentage, *SOC* soil organic carbon

amendment for reclamation of salt-affected soils. The current data on reclamation of salt-affected soils with biochar addition are inconsistent and it is difficult to compare across literature studies. This is probably because of the wide variety of biochar and soil types used in these studies. Moreover, there is a lack of relevant long-term field experiments to verify the mechanisms observed in existing pot or incubation studies. With such a dearth of information on the effects of biochar on saline and saline-sodic soils, further research on the suitability and the functioning mechanism of biochar in salt-affected soils is needed. Table 3 presents a summary of possible changes biochar may induce in salt-affected soils.

7 Conclusions

Given the concern on the continued worldwide expansion of salt-affected lands and the focus on C sequestration processes, this review has evaluated current knowledge on salt-affected soils and their remediation with organic materials and plants. The above review of the published literature has highlighted important gaps in knowledge, which limit our current understanding of rehabilitation of salt-affected soils with organic amendments and the associated carbon dynamic. These knowledge gaps include the following:

1. Assessment of the effects of rehabilitation processes on C cycling and C stocks in salt-affected lands. Of particular importance is how to maximize the accumulation of C stocks in salt-affected lands, where SOC stocks are very small.
2. More studies over longer periods of time are needed to fully understand the mechanism through which biochar affects the properties of saline soils.
3. The benefits of biochar incorporation on reclamation of degraded lands especially salt-affected soils. Knowledge about this is scant, and to date, most studies have evaluated biochar use only in nonsalt-affected soils.
4. The functioning of biochar in different saline soils and the responses of soil to different biochars need to be further investigated in the field.
5. Further studies using long-term field and pot trials with different types of biochar made from various sources and pyrolysis temperatures and different soil types are required. This will enable determination of the ideal biochar to be applied to a specific soil type and the required length of time to effect changes. This will provide ideal information on how biochar can rehabilitate saline-sodic soils or increase SOC stocks permanently.

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