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REVIEW ARTICLE

Phytoremediation of salt-affected soils: a review of processes, applicability, and the impact of climate change

João M. Jesus · Anthony S. Danko · António Fiúza · Maria-Teresa Borges

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Abstract Soil salinization affects 1–10 billion ha worldwide, threatening the agricultural production needed to feed the ever increasing world population. Phytoremediation may be a cost-effective option for the remediation of these soils. This review analyzes the viability of using phytoremediation for salt-affected soils and explores the remedial mechanisms involved. In addition, it specifically addresses the debate over plant indirect (via soil cation exchange enhancement) or direct (via uptake) role in salt remediation. Analysis of experimental data for electrical conductivity (ECe)+sodium adsorption ratio (SAR) reduction and plant salt uptake showed a similar removal efficiency between salt phytoremediation and other treatment options, with the added potential for phytoextraction under non-leaching conditions. A focus is also given on recent studies that indicate potential pathways for increased salt phytoextraction, co-treatment with other contaminants, and phytoremediation applicability for salt flow control. Finally, this work also details the predicted effects of climate change on soil salinization and on treatment options. The synergetic effects of extreme climate events and salinization are a challenging obstacle for future phytoremediation applications, which will require additional and multi-disciplinary research efforts.

Keywords Phytoremediation · Saline soils · Salt-affected soils · Phytoextraction · Climate change

Introduction

Salt-affected soils can be defined as soils with high levels of dissolved salts and/or high concentrations of adsorbed sodium ions in the soil matrix (Qadir et al. 2000). They can be divided into three classes based on salinity and sodicity values, represented by electrical conductivity (ECe) and sodium adsorption ratio (SAR) or exchangeable sodium percentage (ESP): saline, saline-sodic, and sodic soils. Saline soils are characterized by an ECe value of over 4 dS m^{-1} and SAR value below 13 or ESP values below 15. Sodic soils, on the other hand, are characterized by ECe values under 4 dS m^{-1} and SAR values above 13 or ESP values above 15. Saline-sodic soils show both ECe over 4 dS m^{-1} and SAR values above 13 or ESP values above 15 (Qadir et al. 2000).

The effects of high salt concentrations in soils are marked in plants, which exhibit physiological changes including stomata closure, hyper osmotic shock, inhibition of cell division, and photosynthesis; however, the most common effects are nutrient imbalance, low osmotic potential and toxicity of specific ions such as Na^+ and Cl^-, resulting in plant growth

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J. M. Jesus · A. Fiúza
Department of Mining Engineering, Geo-Environment and Resources Research Centre (CIGAR),
University of Porto-Faculty of Engineering (FEUP),
Rua Dr. Roberto Frias s/n, 4200-465 Porto, Portugal

J. M. Jesus · A. Fiúza
Centre for Natural Resources and the Environment (CERENA),
Instituto Superior Técnico, UL, Av. Rovisco Pais,
1049-001 Lisboa, Portugal

M.-T. Borges
Biology Department, Science Faculty, Porto University (FCUP),
Rua Campo Alegre s/n, 4169-007 Porto, Portugal

M.-T. Borges
CIIMAR, University of Porto,
Rua dos Bragas 289, 4050-123 Porto, Portugal

A. S. Danko (✉)
CIGAR/CERENA, Department of Mining Engineering,
University of Porto-Faculty of Engineering (FEUP),
Rua Dr. Roberto Frias s/n, 4200-465 Porto, Portugal
e-mail: asdanko@fe.up.pt

inhibition or death (Aslam et al. 2011). Salinity, and especially sodicity, also contribute to soil degradation by destabilizing soil aggregation due to slaking, swelling, and dispersion (in particular of the clay aggregates), which ultimately leads to hard setting, reduced hydraulic conductivity, impaired air and water movement, runoff, and exposure to erosion. These effects on soil stability are shown in the lower water availability for plants and reduced root penetration, oxygen content, and seedling emergence (Qadir and Schubert 2002).

It is estimated that 1–10 billion ha of salt-affected soils exist worldwide (Yensen and Biel 2006) in over 100 countries (Qadir and Oster 2002), with a potential of 10 to 16 % increase/year (Aydemir and Sünger 2011). Soil salinization is particularly relevant in irrigated lands where 20 to 50 % are considered salt affected and has been shown to result in a decrease of crop yields (Pitman and Läuchli 2004).

The rate of expansion of soil salinization worldwide is expected to increase due to climate change. This will lead to the use of lower-quality water, to increased irrigation-induced salinization, and to the expansion of dryland salinization (by the increase of arid and semi-arid areas and desertification) and to sea level rise, directly contaminating nearby soils or indirectly affecting soils though saline intrusion in aquifers.

Leaching and chemical or organic amendments are the most frequently used methods for salt-affected soil remediation. Leaching involves the application of excess water to promote the movement of soluble salts from the surface soil to deeper soil strata. However, this technique is restricted to saline soils as its effect on SAR is limited and is even counterproductive, since it reduces soil stability. Leaching depends on water availability and quality, as well as soil drainage and water table depth (Qadir et al. 2000). Another disadvantage of leaching is that such treatment reduces total nitrogen (TN), total organic carbon (TOC), and microbial activity and overall soil fertility (Laudicina et al. 2009). Chemical amendments are required for most sodic soil remediation. This process works by promoting ion exchange through the dissolution of existing $CaCO_3$ in the soil or by the addition of calcium cations, followed by leaching. Chemical amendments, such as gypsum ($CaSO_4$) as well as several other compounds ($CaCO_3$, $CaCl_2.2H_2O$, H_2SO_4, S, HCl, FeS_2, CaS_5, $FeSO_4{\cdot}7H_2O$, $Al_2(SO_4)_3{\cdot}18H_2O$), are able to reduce soil salinity and sodicity (Qadir et al. 2001). However, some of these chemical amendments are limited to calcareous soils and all of them still depend on leaching and all of the limitations that it ensues (Qadir et al. 2003). Lastly, organic amendments can also be employed, increasing native calcite dissolution as well as soil structure and aggregation and, by extension, drainage and hydraulic conductivity for improved leaching (Wong et al. 2009). However, increasing prices of chemical amendments have forced farmers to look for alternatives (Qadir et al. 2001), namely in the form of phytoremediation.

Phytoremediation or vegetative bioremediation of salt-affected soils can simply be defined as the cultivation of salt accumulating or salt-tolerant plants for the reduction of soil salinity and/or sodicity (Qadir and Oster 2002). Phytoremediation has several unique advantages over other salt remediation techniques. For instance, it can provide a more uniform removal of salts and at higher depths than gypsum (Qadir, et al. 2001), while also presenting the opportunity to treat salt and other pollutants simultaneously (Greenberg et al. 2007; Manousaki and Kalogerakis 2011a; Shelef et al. 2012). Plants may be used to lower the water table and enhance drainage (Stirzaker et al. 1999). Salt uptake into the shoots also prevents their leaching to groundwater (Rabhi et al. 2009). However, there are still many aspects about this process that need clarification, including the mechanisms (indirect or direct) by which plants are able to contribute to salt remediation, the testing conditions used (e.g., leaching) and their implications, the performance of specific plant species, and how environmental conditions may affect their performance. For instance, it is essential to identify which parameters are more relevant for practical applications, such as phytoextraction potential per dry weight versus quantity and quality of plant biomass produced (whether a plant with high bioaccumulation and low biomass is preferred over a plant with low bioaccumulation and high biomass production) as well as the uptake of specific ions over others (namely sodium) and, most importantly, whether it is technically feasible to enhance the desired traits to expand the applicability of phytoremediation and its efficiency.

It is also imperative to ascertain the impact of climate change not only on all the parameters mentioned above but also on the in situ conditions of soil salinization.

This review paper aims to evaluate phytoremediation as a viable treatment option for salt-affected soils by exploring the mechanisms involved in the process and comparing its performance with the most widely used remediation techniques, especially under climate change scenarios, while also suggesting future research approaches.

Mechanisms involved in salt removal by plants

The mechanisms by which plants remove salt from the soil and the consequences of this process to soil properties are diverse (Fig. 1). Although this complete and holistic approach on plant (and associated rhizosphere microorganisms) impacts in the soil system is not fully explored in the literature, the main mechanisms behind the actions shown are well established and recognized. Aside from increasing leaching conditions (Qadir et al. 2000; Qadir et al 2005), there are two main mechanisms for the role of plants in salt-affected soils remediation. The first one is pH reduction, which increases the dissolution of $CaCO_3$ and, therefore, the available

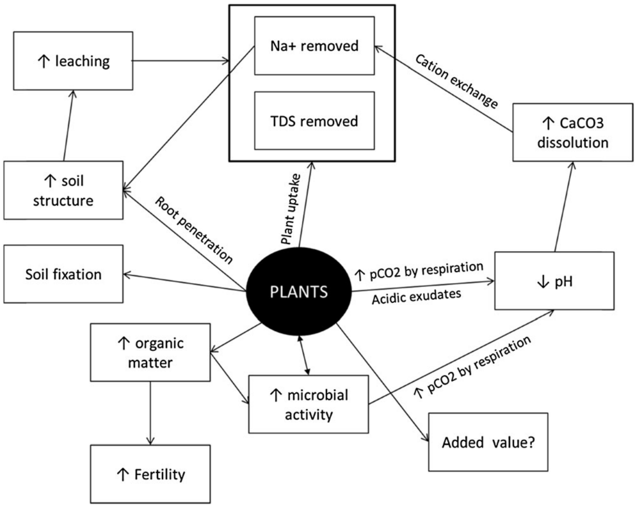

Fig. 1 Role of plants in salt-affected soil remediation and possible variations in soil properties as a result of this process (based on Qadir et al. 2000; Qadir et al. 2006; Rabhi et al. 2009)

Ca^{2+} for cation exchange with sodium (Qadir et al. 2000, 2005; Rasouli et al. 2013; Walker et al. 2013). The second one is plant uptake of dissolved salts in general and/or sodium in particular (Rabhi et al. 2009; Shelef et al. 2012; Walker et al. 2013; Manousaki and Kalogerakis 2011a). The relative importance of each of these two mechanisms is still a question of debate in the literature (Qadir et al. 2006; Rabhi et al. 2009). However, with the plant biomass obtained in the process, added value opportunities might be available such as their use as bioenergy crops or for celullose production (Abideen et al. 2011; Suer and Andersson-Sköld 2011; Wang et al. 2011; Wicke et al. 2011; Glenn et al. 2013).

Many research articles (Minhas et al. 2007; Shekhawat et al. 2006; Gharaibeh et al. 2011) have extensively suggested that plant salt uptake is small in comparison with salt input or salt content in a salinized soil and therefore $CaCO_3$ dissolution would be the main mechanism of remediation. In particular, Qadir et al. (2000) concluded that even in the best possible scenario (high yield and high quality irrigation water), the plant in question (*Leptochloa fusca*) could only remove 90 % of the salt added through the water used in the remediation process, and thus soil salinity would in fact increase, rather than decrease. However, other authors (Rabhi et al. 2010; Ammari et al. 2011; Shelef et al. 2012) demonstrate the potential for salt and, more specifically, sodium uptake. Rabhi et al. (2009) argue that Qadir et al. have neglected salt accumulation in plant shoots. In fact, in greenhouse experiments and under non-leaching conditions, Rabhi et al. (2009) found a significant decrease in sodium levels (up to 70 %) and in overall salinity of contaminated soils.

Although both salt removal mechanisms are valid, the different perspectives discussed above may result from varying experimental conditions, namely leaching (e.g., Qadir et al. 2000) or non-leaching conditions (e.g., Rabhi et al. 2009), which influence plant uptake and accumulation in the shoots or even result from utilization of different plant species. For practical purposes, however, it is crucial to clarify if plant uptake is or is not a significant mechanism of salt removal, since this may limit the phytoremediation approach to calcareous soils, as well as to situations in which water for leaching is available.

Plants referred in soil salt phytoremediation studies are either salt tolerant or halophytes. Within halophytes, salt uptake is highly dependent on plant species (Tipirdamaz et al. 2006). Yensen and Biel (2006) suggested a new classification system for halophytes, which partly accounts for their different behavior in salt remediation processes and divides them into three groups: excluder, accumulator, and conductor plants. Excluder and accumulator types of plants are well-known and applied classifications (Ammari et al. 2008; Gamalero et al. 2009; Shelef et al. 2012; Guittonny-Philippe et al. 2014); however, conductor plant is a relatively novel classification. Excluders prevent salts from entering their tissues as a salinity tolerance mechanism; accumulators uptake and accumulate salts in their tissues and the third type, called conductor plants, absorb salts and excrete them by salt glands, conducting the salts from the soil into the air. This classification and the mechanisms behind phytoremediation actions are the main factors for plant species selection and associated remediation efficiency. Therefore, if $CaCO_3$ dissolution is the main mechanism for saline soil remediation, the most adequate plant characteristics for treatment would be plants with a higher capacity to increase pCO_2 and stronger and larger root systems. The type of salinity tolerance mechanism is, in this case, irrelevant, as long as plants are able to withstand high salinity. If plant uptake is a key element for successful remediation, excluder type of plants are obviously not recommended and accumulator plants would be more appropriate, assuming they possess high total salts uptake (and more specifically sodium) and have high aerial biomass productivity. Perennial plants would also allow for a more extended active period of remediation throughout the year. For conductor type plants, screening could be done by identifying plants with salt glands or bladders or in situ visualization of excreted salt. Sufficient dispersion of salts by wind to avoid soil recontamination is unlikely, although more research is needed to verify this hypothesis (Yensen and Biel 2006).

An improved clarification of plant contribution to the remediation process is needed, in particular, the role of salt uptake in aerial biomass. If this is significant, it can extend the use of this technology to non-calcareous soils and/or under non-leaching conditions (Rabhi et al. 2009). Non-calcareous soils, which are also salt affected, still represent a significant problem. For instance, non-calcareous salt-affected soils represent 30 % (75,000 ha) and 23.1 % (294,000 ha) of all salt-affected soils in France and Hungary, respectively (Van-Camp et al. 2004). Therefore, the development of remediation techniques that are effective in these types of soils is of paramount importance.

Performance comparison and affecting parameters

A comparison of the available studies in the literature regarding the efficiency of phytoremediation for salt-affected soils is challenging due to different testing conditions. Moreover, remediation techniques have a strong case-specific component that cannot be fully accounted for in a review. Therefore, two different basic comparisons of salt phytoremediation information were made.

The first focused on comparing articles referring ECe and SAR reductions from phytoremediation and chemical amendments tests (set no. 1), and the second focused on plant salt uptake capacity (set no. 2). It would be useful to classify the studied plants into one of the previously discussed categories (accumulator, excluder, or conductor), but due to the relative novelty of this categorization, sufficient data has yet to be compiled. Furthermore, it will be seen in both sets that very few plant species were tested. As in other phytoremediation applications, authors prefer the use of plants that have been tested elsewhere to enable comparisons of other relevant or novel parameters tested.

The first set of data compares articles with appropriate levels of information for ECe and SAR reduction capacity in the first 15 or 30 cm of soil (Table 1). ECe and SAR reductions are based on treatment reduction minus control values, when such differences were not already taken into account. The first two studies analyzed can be considered a direct comparison between chemical remediation and phytoremediation, while

Table 1 Soil ECe and SAR reduction through phytoremediation and chemical amendments using different plants (*i* initial, *f* final)

Amendment or plant species	ECe_i (dS m^{-1})	ECe_f (dS m^{-1})	ECe reduction (%)	SAR_i	SAR_f	SAR reduction (%)	Source
Sesbania aculeata	7.5	5.5	27	55.6	43.5	22	Qadir et al. (1997) (1st year)
Leptochloa fusca	7.4	5.3	28	57.9	44.7	23	
Sorghum bicolor	7.8	6.4	18	62.3	55.1	12	
Gypsum	9.0	7.2	20	73.0	53.3	27	
Sesbania aculeata	5.5	4.4	20	43.5	30.1	31	Qadir et al. (1997) (2nd year)
Leptochloa fusca	5.3	4.9	8	44.7	32.5	27	
Sorghum bicolor	6.4	6.0	6	55.1	40.0	27	
Gypsum	7.2	6.8	6	53.3	24.7	54	
Sesbania bispinosa	11.1	4.6	58	35.0	7.9	77	Qadir et al. (2002)
Leptochloa fusca	11.1	4.2	62	35.0	9.0	74	
Gypsum	11.1	4.0	64	35.0	15.0	57	
Sesbania bispinosa	10.3	6.8	34	65.9	35.0	47	
Leptochloa fusca	10.3	7.8	24	65.9	37.0	44	
Gypsum	10.3	7.8	24	65.9	30.0	54	
Sesbania bispinosa	8.4	6.7	20	68.9	40.0	42	
Leptochloa fusca	8.4	5.8	31	68.9	45.0	35	
Gypsum	8.4	7.0	17	68.9	50.0	27	
Suaeda maritima	4.9	1.3	72	15.6	2.81	82	Ravindran et al. (2007)
Sesuvium portulacastrum	4.9	2.5	50	15.7	3.94	75	
Clerodendron inerme	4.8	2.6	45	15.5	4.50	71	
Ipomoea pes-caprae	4.7	3.1	35	15.6	5.13	67	
Heliotropium curassavicum	4.8	3.6	26	15.3	7.65	50	
Gypsum	6.3	4.78	24	14.9	4.9	67	Abd Elrahman et al. (2012)
Citric acid		5.08	19		9.0	40	
Farm manure		4.88	23		7.4	50	
Compost		5.02	20		8.1	46	
Sesuvium portulacastrum	19	9.1	52	2.1 mg g^{-1} Na^+	0.63	70	Rabhi et al. (2009)
Arthrocnemum indicum		10.1	47		0.76	64	
Suaeda fruticosa		12.0	37		0.94	56	
Sesuvium portulacastrum	14.4	9.1	37	59	39	34	Rabhi et al. (2010)
Lotus corniculatus	5.27	2.4	54	20.5	15.8	20	Aydemir and Sünger (2011)
	8.37	2.8	67	24.2	19.1	17	

the comparison between examples 3 and 4 is based on similar initial ECe and SAR, but from different studies, introducing further variability.

By analyzing data from Table 1, some trends are visible. For instance, for plants referred in more than one study, the higher the initial ECe value, the higher the difference between initial and final ECe values. This can be seen for *Leptochloa fusca, Sesbania aculeata,* and *Sesuvium portulacastrum* studies from different sources. Furthermore, the final three tests were conducted in non-leaching conditions, further indicating the possibility of plant uptake as the most significant driving force for remediation. In the case of the study presented by Ravindran et al. (2007), ECe and SAR values decreased from the above-recommended values for soil (ECe>4 dS m^{-1} and SAR>13) to values that may be considered non-saline or sodic. Yet, the initial values were significantly lower compared with those of other studies, and in the other cases analyzed, phytoremediation must be maintained for a prolonged period of time for total remediation.

A direct comparison of phytoremediation with chemical amendments (Table 1), namely gypsum, shows that ECe reduction does not appear to be markedly different between different treatments, regardless of remediation time. Both treatment types experienced a significant reduction of treatment rates for this parameter by the second year (Qadir et al. 1997) and with lower initial ECe (Qadir et al. 2002), indicating that treatment efficiency is dependent, once again, on initial contaminant values. This is potentially due to salt dilution in the leaching water and, as a result, every leaching event, in terms of mass balance, removes less and less salts from the soil, thereby decelerating remediation rates.

The reverse effect is visible on SAR reduction with a slight decrease with increasing SAR values, as well as an improvement over time, possibly reflecting improved hydraulic characteristics of the soil for leaching. For this parameter, there are observable, yet contradictory, differences between the two treatment types: while in Qadir et al. (1997), gypsum (compared with phytoremediation) had a significantly superior SAR removal (in particular in the second year), in Qadir et al. (2002), phytoremediation almost always showed superior SAR reduction over gypsum (including at higher depths). In the indirect comparison between sources 3 and 4 in Table 1, phytoremediation with different plants (with one exception) revealed higher ECe and SAR removal than four different amendment types.

Regarding plant behavior, in Qadir et al. (2002), *S. aculeata* had the highest yield, while *L. fusca* had the lowest yield for the same soil type, despite the fact that both plants showed similar effects on SAR reduction. This may reflect different responses to salt stress for these two plant species, which implies that, in cases where phytoremediation is mainly due to enhanced soil structure for leaching, above ground biomass is not an appropriate indicator of the potential of a plant for salt remediation.

The application of non-leaching conditions provides further information on salt uptake capacity of plants in soils. Rabhi et al. (2009) reported that in the field, *Suaeda fruticosa* contributed to desalination of the surrounding rhizosphere mostly by improved leaching due to enhancement of soil structure, while the contribution of *Arthrocnemum indicum* was by salt uptake. When both plants were tested in non-leaching conditions, the maximum salt uptake of *S. fructicosa* was in fact higher. It is possible, therefore, that *S. fructicosa* improves the structure of the soil in a more efficient way than *A. indicum*, possibly due to different root systems, and in such a way that leaching occurs too quickly to enable significant amounts of salt uptake.

On the other hand, Aydemir and Sünger (2011) showed a reduction of calcite levels in the saline-sodic soils tested but not in the non-saline soil while planted. The authors attributed this difference to lower levels of calcite in the non-saline soil, but it is also possible that this difference was due to the initial high pH of the saline-sodic soils and subsequent reduction of pH after phytoremediation. The mobilization of calcium ions by the dissolution of calcite may have enhanced sodium desorption but, without leaching, the exchanged sodium would recapture its place during cation exchange and recontaminate the soil (Qadir et al. 2001). Therefore, salt uptake by the plants was likely to be the most important removal mechanism, aided by the dissolution of calcite which made sodium more available for plant uptake.

The second set of data acquired on phytoremediation performance regards phytoextraction and includes eight different articles in which salt uptake by different plant species was analyzed (Table 2). Values for salt removal in kilograms per hectare per year were extrapolated when needed by dividing salt uptake data by the duration of the experiments and multiplying by the number of days of the growing season for each plant species (365 days in the case of perennial plants). Therefore, for the purpose of this analysis, it was assumed that plant productivity and salt uptake did not change significantly over the course of the growing season. Although a clear simplification, this extrapolation was required to account for the varied duration of the different studies and therefore allowed a direct comparison of plant salt uptake data.

According to Table 2, values of salt uptake can vary from 91 kg ha^{-1} $year^{-1}$ for *Lotus corniculatus* to up to 5376 kg ha^{-1} $year^{-1}$ for *S. portulacastrum*, which are both halophytic plants. However, non-halophytes (or salt tolerant) can have significant salt uptake capacity as, for example, *Typha angustifolia* removed 1200 kg ha^{-1} $year^{-1}$, while others have the potential for salt uptake but were never actually tested in the conditions necessary to be referenced in the present comparison. Initial ECe plays a significant role in total salt uptake, as seen in Aydemir and Sünger (2011), where the higher the ECe value, the larger the salt uptake capacity. On the other

Table 2 Salt uptake in milligrams per gram dry weight (DW) and kilograms per hectare per year of different plant species in saline soil remediation applications

Plant species	ECe$_i$ (dS m^{-1})	SARi	Salt uptake (mg g^{-1} DW)	Salt uptake in kg ha^{-1} year^{-1}	Source
Suaeda salsa	42	–	155 (Na^+)	2300 (Na^++Cl^-)	Zhao et al. (2005)
Kalidium folium	42	–	168 (Na^+)	2800 (Na^++Cl^-)	
Tetragonia tetragonioides	21	–	–	4760 (Na+Cl)	Neves et al. (2007)
Sesuvium portulacastrum	9.1	–	163 (Na^+)	5376 (Na^+)	Rabhi et al. (2009)
Arthrocnemum indicum	10.1	–	113 (Na^+)	1527 (Na^+)	
Suaeda fruticosa	12.0	–	176 (Na^+)	1726 (Na^+)	
S. portulacastrum	14.4	59	273 (Na^+)	1931 (Na^+)	Rabhi et al. (2010)
Suaeda maritima	4.9	15.6	184 (TDS)	1512 (TDS) [a]	Ravindran et al. (2007)
Sesuvium portulacastrum	4.9	15.7	147 (TDS)	1422 (TDS)[a]	
Clerodendron inerme	4.8	15.5	94 (TDS)	1189 (TDS)[a]	
Ipomoea pes-caprae	4.7	15.6	81 (TDS)	1079 (TDS)[a]	
Heliotropium curassavicum	4.8	15.3	71 (TDS)	976 (TDS)[a]	
Lotus corniculatus	5.27	20.5	–	91 (TDS)	Aydemir and Sünger (2011)
	8.37	24.2	–	200 (TDS)	
Atriplex halimus	65.3	26.4	288 (Na^+)	2419 (Na^+)	Gharaibeh et al. (2011)
Atriplex halimus[b]	65.3	26.4	304 (Na^+)	3192 (Na^+)	
Typha angustifolia	18.8	–	370 (TDS)	1200 (TDS)[c]	Boonsaner and Hawker (2012)
Acanthus ebracteatus		–	620 (TDS)	2400 (TDS)[c]	

[a] Values estimated by mass balance

[b] With gypsum added to the soil

[c] Assuming approximate productivity of 10 ton ha^{-1} year^{-1}

hand, simultaneous gypsum addition may also increase the salt uptake capacity by 132 % (Gharaibeh et al. 2011).

A compromise between halophytic (or salt tolerant) crop yield and remediation goals may be required in harsh phytoremediation conditions, e.g., highly salt-affected soils. Depending on whether or not the aerial biomass obtained is to be used, remediation options may differ. For instance, Boonsaner and Hawker (2012), proposed an initial crop of *Glycine max* for salt remediation, alleging that the high salt uptake per dry weight and the small price of plant seeds would make the process viable.

The salt accumulation data shown in Table 2 needs to be analyzed in the proper context. Assuming a hypothetical case study of a medium textured soil with a water saturation percentage of 35 % and bulk density of 1300 ton m^{-3}, ECe of 20 dS m^{-1}, and a remediation goal of lowering ECe to 4 dS m^{-1}, then the mass of total dissolved salts that needs to be removed is 71.5 ton ha^{-1} at a soil depth of 1 m. To achieve this by plant remediation, even with the best performing plant listed in Table 2 (S. *portulacastrum*), it would take approximately 13 years, not considering further salt inputs that might occur during that period. Considerations like these may have led many researchers to deem that phytoextraction alone is not a viable remediation option. However, most studies consider remediation of only the first 15 to 30 cm of soil as this is the most important depth for most agricultural crops. At a soil depth of 15 cm, the value of salts to be extracted in this scenario would be significantly lower, circa 10.725 ton ha^{-1}, reducing the remediation time needed to only 2 years.

With the goal of making a clear assessment of the potential of plant species to remediate salt-affected soils, a bioconcentration factor (BCF), similar to that applied to heavy metals phytoremediation, could be used. However, the obtained BCF could be, as seen before for the plant performance in Table 1, dependent on initial salt concentration, as well as on productivity (Liang et al. 2009). Furthermore, a differentiation between BCF for total dissolved solids (TDS) and for sodium would be required to define salt hyperaccumulating plants. In addition, the perfect plant for remediation should have a high sodium uptake, but a low uptake of calcium and magnesium since they are stabilizing agents for the soil, and therefore contribute more quickly to SAR reduction.

The distribution of sodium within plant tissues is also a relevant aspect (Table 3) to assess salts translocation capacity to aboveground biomass. This approach is needed not only to assess overall salt removal but more specifically to calculate the ratio between sodium to calcium and magnesium. Potassium is also a relevant ion, since a high K^+/Na^+ ratio may indicate that the plant needs potassium to tolerate sodium toxicity. This could increase potential nutritional needs or indicate that the plant is highly selective to this cation over sodium. As such, sodium uptake would be smaller in the presence of high levels of potassium.

Table 3 Ion distribution (mmol L^{-1}) in different plants (Tipirdamaz et al. 2006) and plant tissues (Rabhi et al. 2010)

	Na^+	K^+	Ca^{2+}	Mg^{2+}	K^+/Na^+	Cl^-
Halocnemum strobilaceum	3.30	0.25	0.20	0.13	0.08	2.04
Chenopodium album	1.54	0.78	0.03	0.34	0.51	1.70
Atriplex tatarica L.	1.42	0.29	0.01	0.37	0.20	2.24
Petrosimonia brachiata	1.42	0.46	0.05	1.49	0.32	3.14
Plantago maritima	1.33	0.26	0.46	0.24	0.20	0.78
Reaumuria alternifolia	1.44	0.30	0.28	0.57	0.21	1.79
Salicornia europaea	4.49	0.40	0.21	0.39	0.09	5.57
Sesuvium portulacastrum						
Leaves	6.52	0.40	0.58	0.19	0.06	–
Stems	3.81	0.78	1.12	0.21	0.21	–
Roots	1.63	0.45	0.54	0.20	0.28	–

For example, comparing sodium uptake content alone, *Petrosimonia brachiata* and *Atriplex tatarica* L. could be classified as similar in their potential for SAR reduction in a contaminated soil (based only on data in Table 3). However, *P. brachiata* also showed a significantly larger concentration of calcium, magnesium and potassium when compared with *A. tatarica* L. Therefore, *A. tatarica* L. seems to be a more appropriate choice for saline soil phytoremediation.

As previously mentioned, plant productivity is extremely relevant for overall salt phytoremediation. Even when phytoextraction values are extremely high, the impact of plants on soil remediation can be low due to low plant growth. This can be better understood using, for instance, the work of Goulet et al. (2005) on aluminum phytoremediation, were it was reported that in mesocosm tests, *Lemna minor* had an aluminum concentration capacity close to five times that of *Typha latifolia*, although the latter was responsible for 99 % of the aluminum removed. In the situation under analysis, while aboveground productivity is considered in the results expressed in Table 2, this value is dependent on a variety of conditions and further studies should be developed to assess productivity in saline environments closer to actual plant exposure in a salt-affected soil.

Furthermore, productivity is dependent on plant density, which is yet another parameter that is far from being optimized in this context. So far, data shows that increasing density can result in decreased productivity, but it may also increase salt accumulation in plant tissues (Hansi et al. 2014), possibly determining an overall increase in salt uptake.

Opportunities for enhancing salt phytoremediation

In order to enhance salt phytoremediation, there is a need to improve the two main mechanisms by which plants can remediate a salt-affected soil: phytoextraction or leaching enhanced by plant roots. How these goals can be approached may differ: either by increasing salt uptake per unit of mass (through various methods, mostly biological) or by increasing tolerance to salinity stress and therefore increasing yield, which can create more leaching through larger and stronger roots and/or increased overall salt uptake. Also, different management techniques can contribute to an increase of the efficiency of the process. An analysis of the potential applicability of different enhancement techniques for phytoremediation of salt-affected soils, which are summarized in Fig. 2, may provide information of future research trends and co-treatment possibilities to enable the management of complex soil contaminations.

Many of the techniques that will be described have yet to be implemented and, in some cases, were not even tested in soil salinization processes or with halophytic plants. In some instances, therefore, the techniques presented were applied in other experimental contexts but can provide relevant information for the enhancement of salt-affected soils phytoremediation.

Combination of remediation techniques and multi-contaminated soils

To increase plant salt remediation efficiency, there are opportunities for synergetic combinations between different treatment types for salt-affected soils. For instance, Gharaibeh et al. (2011) described that phytoremediation and gypsum addition could increase plant phytoextraction capabilities and productivity as well as increasing overall salt removal. Theoretically, the plants without gypsum may have had a calcium deficiency (Zia et al. 2007; Ahmad et al. 2011) that was supplemented by gypsum addition, which resulted in increased productivity. However, it is more likely that gypsum increased the bioavailability of sodium ions by supplying calcium, which removed adsorbed sodium from soil particles. Additionally, plants may have decreased pH (Ghafoor et al.

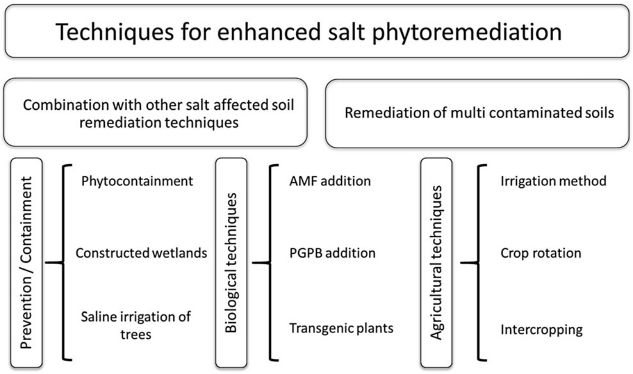

Fig. 2 Techniques for enhanced salt phytoremediation grouped by type

2012) and therefore increased gypsum dissolution rates. These hypotheses, however, require further studies to be confirmed.

Nevertheless, other studies showed that the combination of plant remediation and amendments were not always beneficial. In fact, gypsum and H_2SO_4 decreased the productivity of Kallar grass (*L. fusca* L.) and Berseem (*Trifolium alexandrinum* L.) when compared with the control conditions (Zia et al. 2007). Reduced plant productivity was also obtained for the combination of *Sesbania bispinosa* with H_2SO_4, especially in a high SAR environment, due to chemical burns of plant roots (Ahmad et al. 2011).

Therefore, doubts remain on the existence and importance of synergetic combinations of salt remediation treatments.

Phytoremediation can be applied for combined treatment of salt-affected soils also contaminated with other pollutants. Several contaminant combinations have already been considered, for instance, the combined treatment of saline soil with organic degradation of polycyclic aromatic hydrocarbons (PAHs) or total petroleum hydrocarbons (TPHs) (Hue et al. 2002). The uptake of multiple heavy metals by a single plant species is possible (Satpathy and Reddy 2013) and therefore simultaneous uptake of heavy metals and sodium is likely to be possible as well since some researchers have found a link between the root zone ionic strength and composition (i.e. saline levels and ion distribution) with the type of excreted salts from salt glands (Manousaki et al. 2008). Although, salinity does not seem to affect heavy metal phytoextraction in the same way for all metals (Manousaki and Kalogerakis 2009).

Salinization prevention or containment

Controlling or limiting the salt flow is of the utmost importance to prevent further soil degradation through salinization processes. Hydraulic control of catchment areas is an important part of an integrated management policy to control salt flows. Phytohydraulic containment using salt-tolerant trees has already been applied in the control of saline seepage, salt mobilization, and capillary rise of salts from contaminated groundwater. By consuming large amounts of water in their growth, trees enable the reduction of the water table, decrease runoff and upflow of the groundwater, and can even phytoextract significant quantities of salt without deleterious effects (Crosbie et al. 2008; Rodríguez-Suárez et al. 2011).

To prevent further expansion of salt-affected soils, intensifying and/or improving saline wastewater discharge control can have a significant impact. Several industries produce saline wastewater, ranging from fish farming activities to oil and gas extraction. Although phytoremediation has been successfully used to treat saline wastewaters (1 to 3.5 % salinity levels) from a variety of industries such as aquaculture (Laudicina et al. 2009), tannery (Calheiros et al. 2012), or olive mill (Herouvim et al. 2011), the existing effluent excess salts were not considered a problem to be treated. Some studies with constructed wetlands have, however, demonstrated the ability of salt co-treatment with other pollutants (Lymbery et al. 2006; Jesus et al. 2014), while in other studies, constructed wetlands were designed for the sole purpose of salt phytoextraction, with promising results (Shelef et al. 2012). Constructed wetlands may be a good option for intercepting non-point contaminants (from agriculture and greenhouse leachates, for instance) in catchments areas, avoiding salt discharge in the soils.

The reuse of several saline wastewater sources in phytoirrigation of trees is also under investigation which can prevent salt accumulation and/or leaching through the soil due to uptake by trees (Jordahl et al. 2004; Zalesny and Bauer 2007; Smesrud et al. 2011).

Biological techniques

The application of arbuscular mycorrhizal fungi (AMF) and plant growth-promoting bacteria (PGPB) has been extensively proposed in order to increase plant salt tolerance and promote the growth of plants in saline soils (Gamalero et al. 2009). These applications include not only agricultural crops but also salt marsh halophytes, potentially to increase their phytoextraction efficiency (de-Bashan et al. 2012). The subjects of plant salt tolerance and the impact of these applications have been extensively and adequately analyzed elsewhere (Evelin et al. 2009; Gamalero et al. 2009; Dodd and Pérez-Alfocea 2012; Porcel et al. 2012). Therefore, in this review, only the potential of AMF and PGPB addition in the enhancement of salt-affected soils phytoremediation (particularly in phytoextraction) will be explored since this was not yet addressed.

It is known that the addition of AMF affects plant accumulation of Na^+ and K^+, which may be relevant to phytoremediation of salt-affected soils (Evelin et al. 2009; Gamalero et al. 2009; Cartmill et al. 2012; Ruiz-Lozano et al. 2012). There are reports of increased uptake of sodium and chloride with AMF, which can be accompanied by increased nutrient uptake and productivity (Evelin et al. 2009). In some other studies, however, there is reduced uptake of sodium (Abdel Latef and Chaoxing 2011), which may be regarded as undesirable for phytoremediation goals. However, these results refer to glycophytic plants, as studies with AMF inoculation of halophytes with emphasis on salt phytoextraction are very rare. A recent example can be found in the work of Zhang et al., (2014) where AMF added to *Ricinus communis* lead to reduced ECe and sodium values in the soil possibly (as reported by the authors) due to increased salt phytoextraction through the roots. The existence of limited studies on AMF addition to halophytes can be due to multiple reasons: there is a higher interest in applying AMF in crop plants to increase their yield, as halophytes have limited commercial uses; salinity limits the richness of naturally occurring AMF species (Krishnamoorthy et al. 2014), and many halophytic plants are considered to be non-mycorrhizal (Caravaca et al. 2005). Nevertheless, there are some clues to the potential of AMF addition for enhanced phytoremediation. In Zhang et al. (2011), for instance, despite the fact that AMF reduced sodium uptake in the aboveground biomass of the halophyte *Leymus chinensis* by 29 %, the concurrent 222 % increase in biomass lead to an overall increase of accumulated sodium by 130 %. Furthermore, AMF addition can lead to an improvement of soil structure and therefore leaching of salts and soil remediation (Caravaca et al. 2005; Qin et al. 2015).

Regarding PGPB, there are several studies in which its utilization enhanced salt tolerance and therefore plant productivity under saline conditions. An extensive list of examples can be found in de-Bashan et al. (2012), and to illustrate this possibility two examples are referred in the present work: in Goswami et al. (2014), *Arachis hypogaea* treated with *Bacillus lincheniformis A2 s*howed an increase of 31 % in plant length and 43 % in fresh biomass at 50 mM NaCl while in Siddikee et al. (2011), several types of PGPB were tested and all led to improved root length, accumulation of dry matter in roots and reduction of ethylene stress levels, leading to increased salt tolerance in *Capsicum annum* L. Yet, in this study, sodium uptake by the plants decreased. As previously noted for AMF, studies with PGPB application in halophytes are less common. However, Rueda-Puente et al.(2007) reported higher plant height, length of the root system and fresh biomass of the halophyte *Salicornia bigelovii* with the addition of PGPB in saline conditions, concluding that PGPB could be used reliably to promote the growth of halophytic plants. Although PGPB have been extensively studied for the enhancement of several different types of phytoremediation of contaminated soils (de-Bashan et al. 2012), in the bibliographic search undertaken in this work no studies were found that showed increased salt phytoextraction on a per mass basis in any plant as a result of PGPB addition. Nevertheless, similarly to what is observed with AMF, PGPB may increase overall salt phytoextraction by enhancing plant yield (Chang et al 2013).

The development and utilization of transgenic plants can also be a potential way to further increase plant salt tolerance. In Bhavanath et al. (2013) transgenic plants of *Jatropha curca*, in which the SbNHX1 gene was cloned from *Salicornia brachiata*, had a higher yield compared with wild types as well as a 43 % increase in sodium uptake. Similar results are also reported in others studies (Rajagopal et al. 2007; Jha et al. 2011). Most studies with transgenic plants are focused on increasing salt tolerance, not necessarily on phytoextraction. However, by focusing on genes that regulate the plasma membrane-bound or vacuolar Na^+/H^+ antiporters, improved salt compartmentation in the vacuoles was observed (Saqib et al. 2005; Apse and Blumwald 2007; Jha et al. 2011; Hasegawa 2013), and therefore, increased phytoextraction is a by-product of these enhancements. Curiously, Ruan et al. (2010) report the existence of 100 claims of plant transformation by genetic engineering aimed at increasing salt tolerance but, yet again, few of them focus on halophytes, as these plants are much more frequently used as the source of the gene rather than the tested plant.

Agricultural techniques

The way irrigation water is applied to a salt-affected soil can also be optimized to enhance salt removal from both the soil and irrigation water. For instance, it has been reported that water logging tends to increase salt phytoextraction; therefore, ponding could be used to enhance salt uptake by some halophytes (Barrett-Lennard and Shabala 2013), as the foliar concentration of sodium may double (Carter et al. 2006). On the other hand, directly exposing plant leaf surface to the saline solution can increase salt foliar absorption (Qadir et al. 2000; Sultana et al. 2001; Chondraki et al. 2012).

A scheme for annual crops could be developed by crop rotation. This method has already been used in salt-affected soil remediation and prevention of secondary salinization (Kaur et al. 2007; Zia et al. 2007; Mandare et al. 2008; Ahmad et al. 2011; Al Khamisi et al. 2013). Crop rotation in this context can take many forms, but mostly involves the use of either a halophytic plant or rice crop, followed by an economic crop (Ahmad et al. 2011): the first crop is salt tolerant and is used to leach the salts, particularly in the case of rice, and is intended to create more adequate conditions for the growth of the second, more economically valuable crop. This possibility is well explored elsewhere (Qadir et al. 2008).

Intercropping salt removing plants with existing crops could also be an option to prevent or remediate salt-affected soils (Qureshi et al. 2003; Kan et al. 2008; Al Khamisi et al. 2013). An intercropping agroforestry scheme could provide simultaneous salt removal in a preventive approach (Kiliç et al. 2008). However, there seems to be conflicting results in the literature: Inal and Gunes (2008) concluded that interspecific root interactions might be helpful for mineral nutrition and salt tolerance in mixed crops, while Kurdali et al. (2003) found no substantial differences with the use of intercropping in a saline soil, and Patra et al. (2002) reported no changes in productivity with intercropping, but a significant sodium accumulation in one of the plants used (*Matricaria chamomila*), which increased with gypsum addition. Further studies are required to clarify the potential of intercropping on salt-affected soils remediation.

Economically, crop rotations and intercropping may provide further income and cost less in fertilization and water use, since the leaching requirement is decreased in summer months.

Climate change: effects on soil salinization and adaptation measures

Climate change has repercussions in soil salinization expansion and prevalence, as well as in the remediation techniques that can be applied. Prevalent climatic conditions affect the choice of remediation technique in any given location and contamination scenario. However, regional predictions of future climatic changes should also be taken into account in the choice of a soil remediation technique, given the fact that remediation efforts may span a considerable amount of time and climate change may cause significant and ever evolving differences in contaminant concentration and biochemical parameters (Bradford et al. 2010; Van den Berge et al. 2011).

The impact of climate change on soil salinization expansion is difficult to assess (Schofield and Kirkby 2003). In the case of Europe, current levels of soil salinization are estimated at 50 million ha. However, Szabolcs (1974) estimated an increase, due to direct and indirect impacts of climate change, of at least 26.7 million ha by the year 2050, a 53.4 % increase in Europe alone. The expected increase would be due to expansion of arid and semi-arid environments, sea level rise, and irrigation (Van-Camp et al. 2004; Tóth et al. 2008). In Australia, one of the most affected countries by both salinity and climate change, beyond the confirmed 1.047 million ha of salt-affected soils, there are an additional 1.7 million ha estimated cases of salinization or in risk of salinization (Jardine et al. 2007).

Therefore, a flexible remediation technique that is both adaptable to climatic changes and environmentally sustainable, is required (Hou and Al-Tabbaa 2014; Hou et al. 2014). Climate change impacts on remediation efficiency are already being evaluated for other types of contaminants and remedial techniques, such as bioaugmentation, organic amendments, and acid mine drainage remediation (Al-Tabbaa et al. 2008; Anawar 2013). Also, new management options are being considered for field remediation scenarios such as accelerating the initiation of the restoration process to prevent further deterioration from climate change or the application of compensatory restoration, when necessary (Harris et al. 2006; Rohr et al. 2013).

Climate change can have unpredictable and even contradictory impacts on phytoextraction techniques. Rajkumar et al. (2013) review several possible implications of climate change on metal phytoextraction capabilities of various plants, with examples in which climate change lead to increasing or decreasing metal uptake, depending on plant type, and tolerance to metal ions. Also, ion competition for cation exchange sites between salts and heavy metals was shown by Hamzenejad Taghlidabad et al. (2014). For salt phytoextraction, however, there is little information on the impact of climate change. Therefore, further studies are necessary to increase the robustness of all remediation techniques to climate change and of soil salinization remediation techniques in particular. In 1996, Intergovernmental Panel on Climate Change (IPCC), addressed this issue within the broader future scenarios of “warm and dry” and “warm and wet” climates. The impact of the two scenarios set up by the IPCC on irrigated soils, in existing salt-affected soils, and in their expansion, is explored in Table 4.

By comparing the two analyzed scenarios, it becomes clear that the “warm and dry scenario” is the most negative and influential one on soil salinization, as expected. However, the warm and wet scenario also presents significant, although rather unexpected, negative impacts. In particular, in situations where the water table is already high, the excess rainfall may create water logging and further agricultural damages. On the other hand, intensive rainfall will leach out the

Table 4 Climate change effects on multiple parameters, including salt-affected soils, within two future scenarios: warm and dry and warm and wet conditions

Effects on	Scenario warm and dry	Scenario warm and wet
Rainfall	Decreased (IPCC 1996)	Increased (IPCC 1996)
Water table	Decreased (IPCC 1996)	Increased (IPCC 1996)
Irrigation	Increased (Szabolcs 1990)	Normal or decreased
Drainage[a]	Increased (Van-Camp et al. 2004)	Decreased (Ritzema et al. 2008)
Irrigated non-salt-affected soils	Salt buildup by evaporation, use of brackish water or saline groundwater (Szabolcs 1990; Van-Camp et al. 2004)	More leaching, less irrigation needed, more drainage necessary in some cases (IPCC 1996; Ritzema et al. 2008)
Existing salt-affected soils	Further salt buildup, less leaching and vegetation cover, more wind erosion and transport (Szabolcs 1990; Van-Camp et al. 2004)	More leaching, reducing dissolved salts but decreasing soil structure (IPCC 1996; Horneck et al. 2007)
Expansion of salt-affected soils	Expansion of affected area to adjacent soils following the trend of aridity (IPCC 1996; Szabolcs 1990; Van-Camp et al. 2004)	Leaching salts to groundwater or, if low drainage, lateral spread of salts and water logging (Daily 2005)

[a] In case of shallow aquifer occurrence

dissolved salts, further destabilizing the soil, and resulting in clay loss, macropore clogging, and reduced permeability (Diamantis and Voudrias 2008; Sahin et al. 2011). This will effectively transform a manageable saline-sodic soil into a hard set sodic soil (Horneck et al. 2007). This excess water can also create lateral transport of salts, propagating soil salinity to neighboring, potentially not yet affected soils, or forming saline seepage and waterlogging (Daily 2005; Dragovich and Dominis 2008).

It is certain, however, that soil salinization is severely affected by climate change. Yet, the reverse effect, the impact of soil salinization on climate change, must also be considered, since the expansion of soil salinization translates into less CO_2 uptake due to its impact on plant yield (Setia et al. 2013). However, it might also negatively affect decomposition rates, thereby lowering CO_2 emissions. Overall, it has been found that soil salinization has a total negative effect on climate change, with an estimated past contribution of 2 Pg of CO_2 (Setia et al. 2013).

The “warm and dry scenario” is likely to present more challenges for phytoremediation not only due to increased salt concentrations but also due to the action of other simultaneous stress inducers, such as heat and drough. Therefore, there is more work developed to study the negative impacts of this scenario. Drought and heat stress in plants entails a very similar response to salt stress, since salt stress may also cause both osmotic imbalances and water absorption deficiencies. Heat and salinity stresses also have similar effects in the sense that both cause oxidative stress. As a result, foliar application of compatible solutes, such as proline and glycinebetaine, has been suggested to increase plant tolerance to heat and salt (Wahid et al. 2007). Accumulation of proline and glycinebetaine has been reported in salt accumulating plants, since they are used to maintain osmotic balance in the cell, which is disrupted by the presence of ions stored in the vacuoles (Abdel Latef and Chaoxing 2011; Manousaki and Kalogerakis 2011b). Therefore, these organic solutes do not necessarily preclude salt uptake and may be used to enhance it by increasing salt tolerance, although further studies are necessary to confirm or deny these hyptheses. Nevertheless, even if having neither a positive nor a negative effect on salt uptake, these solutes would increase tolerance to heat and salinity stress, enabling improved plant acclimation to heat waves. Hansi et al. (2014) indicated that the combination of heat and salt stress had less negative effects compared with each stress acting individually in tomato plants, possibly due to the increased glycinebetaine accumulation stimulated by both stress inducers.

Heat acclimation and foliar application of calcium may also be used to improve heat tolerance and again, in the case of calcium application, may simultaneously alleviate salt stress (Wahid et al. 2007).

Climate change will also increase the occcurrence of extreme events, namely floods, specifically in the “warm and wet scenario”. Plants have varying resistance to flooding, which are mainly limited by oxygen deprivation. Physiologically, plants adapt to this situation through the growth of adventitious roots containing aerenchyma (Carter et al. 2006).

In temperate climates, in which dryland salinity is not a severe problem, halophytes can be found mostly in wetlands and are likely to be acclimated to both salinity and flood conditions. In arid or semi-arid regions, however, continuous flood conditions are unlikely, and acclimation to this type of stress is rarer, since it is exacerbated by uneven distribution of precipitation, such as in monsoon areas (Akhter et al. 2004).

Final considerations and perspectives

Salt-affected soils threaten global agricultural productivity. Although several remediation techniques have been successfully developed and implemented, there are still various situations for which there is a lack of appropriate technical solutions. Under certain circumstances, phytoremediation can be the best option from both technical and economical perspectives.

In this work, we intended to clarify the efficiency of the phytoremediation approach for the salt remediation of soils. A thorough review of the available literature, and in particular of the research directly comparing phytoremediation with other approaches, reveals removal efficiency similar to other techniques. However, debate on the main mechanism behind salt phytoremediation has yet to be settled and requires a more focused research effort to assess the contribution of phytoextraction to the remedial process. Furthermore, recent research in the field of phytoremediation, and particularly phytoextraction, hints at several new possibilities to increase the efficiency and quality of the treatment of salt-affected soils (combination of treatment types, mixed plant cultures, biostimulation, etc.) or expand to new applications such as cotreatment and salt flow control measures. Nevertheless, these novel applications are still in their infancy and further development is essential.

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