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PART FOUR: DIAGNOSIS OF SALT PROBLEMS

CHAPTER 11

IRRIGATION WATER QUALITY ASSESSMENTS

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INTRODUCTION

This chapter discusses the effects of inorganic elements in irrigation water on the long-term sustainability of the agricultural soil-water system. It focuses on salinity, sodicity, and the effects of sodicity on soil permeability, major cations and anions, and trace elements. Taking into account interactions of irrigation water, soils, and crops, it describes the net change in salinity resulting from irrigation at various leaching fractions and identifies generally acceptable levels of trace elements in irrigation supplies.

A meaningful assessment of the quality of water used for irrigation should consider such local factors as the chemical reactivity of constituents dissolved in the water, the soil's chemical and physical properties, climate, and irrigation management practices. It should also consider the effects of irrigation on the quality of agricultural drainage, effects on humans and animals of chemicals concentrated in harvested plant products, and economic conditions that determine how much salinity-induced reduction in yield or quality can be tolerated.

To avoid the long-term accumulation of toxic amounts of waterborne substances in the rootzone of irrigated lands, the input of those substances to the soil from irrigation and other sources must not exceed the sum of losses from the soil and conversions to unavailable forms. Losses include removal in harvested crops, transport by subsurface drainage, erosion by wind and water, and, for some elements, volatilization of gaseous compounds. Relatively immobile elements, such as arsenic (As) and copper (Cu), often are converted in the rootzone to less available forms (such as adsorbed or precipitated solid phases). These forms are sometimes considered unavailable; however, availability depends on the subsequent chemical conditions. Conversion processes are primarily due to precipitation or changes in redox status, and these conditions could change again in the future. For example, As in forms that may be considered unavailable under aerobic conditions could be remobilized under anaerobic conditions, or in response to changes in pH. In contrast, selenium (Se) is highly mobile under aerobic conditions and less mobile under anaerobic conditions. Changes in oxidation status, pH, or other chemical conditions could be related to changes in cropping patterns (such as conversion to rice cultivation) or changes in land use.

The most mobile ion of importance, chloride (Cl^-) is relatively nonreactive, as most of its salts are highly soluble, and the ion undergoes little adsorption or exchange. Other mobile ions, such as nitrate (NO_3^-), also undergo little adsorption or exchange but are subject to redox transformations, such as NO_3^- to ammonium (NH_4^+), which may be retained by the exchange sites, volatilized as ammonia (NH_3), or incorporated into organic matter. Other elements, such as sodium (Na) and magnesium (Mg), exist in cationic form (Na^+ and Mg^{2+}), are readily exchangeable, and are thus less mobile when going into soil exchange sites. Elements, such as boron (B), are adsorbed and less mobile, followed by elements, such as As, that are highly adsorbed.

Most soluble constituents, being relatively mobile, can be removed by leaching. Thus, leaching often can be used to adjust the concentrations of soil chemical constituents to accommodate crop production. If the element of interest is immobile under existing soil conditions and if leaching losses are insignificant, then the elemental inputs not removed by plants or converted in the soil to unavailable forms will accumulate as soluble and labile (adsorbed) forms. These forms are related as follows:

$$\text{Soluble} \leftrightarrow \text{Labile} \leftrightarrow \text{Residual}$$

The soluble element adsorbs or desorbs into the labile form as the amount in solution increases or decreases. The labile element is transformed to or from the residual (relatively unavailable) form. Only the soluble form is immediately available to the plant. As the soluble element is removed by plant roots, desorption from the labile pool replenishes the soluble pool. Although the residual pool may not impact current agricultural production, it may nonetheless be of environmental concern, both in terms of potential mobility under different chemical conditions and as potential transport as dust to other sensitive environments, such as wetlands.

The level of toxicity depends directly on the amount of the toxic constituent in solution, and indirectly on the capacity of the labile pool. The hazard posed by elements that exist in soluble and labile forms in the soil is that once toxic levels are attained, eliminating or reducing these levels involves removal in harvested crops and conversion to residual forms, both of which are processes that can take decades, even if inputs of the element cease. These situations can be avoided by ensuring that inputs of potentially toxic elements remain below the levels that are tolerable by the most sensitive crop to be grown and by avoiding crops that will bioaccumulate the elements of concern.

SALINITY

Salinity in water is defined as the total sum of dissolved inorganic ions and molecules. The major components of salinity are the cations Ca^{2+}, Mg^{2+}, and Na^{+}, and the anions Cl^{-}, sulfate (SO_4^{2-}), and bicarbonate (HCO_3^{-}) and NO_3^{-}. The potassium (K^{+}) and carbonate (CO_3^{2-}) ions are usually minor components of the salinity. The effects of these and other minor dissolved constituents, such as B, are generally neglected in assessing the salinity of irrigation waters but nonetheless are important when assessing the suitability of waters for irrigation.

Salinity reduces crop growth by reducing the ability of plant roots to absorb water, by accumulation of toxic concentrations of salts in plant tissue, specific ion toxicity, and ion imbalances. The soluble ions and molecules reduce the availability of water to a plant, a phenomenon known as the osmotic pressure effect. The osmotic pressure effect is especially important at high salinity. Water availability in the soil relates to the combined (but not the simple sum) of the matric and osmotic potential stresses.

As a first approximation, we can consider that the combined effects of osmotic and matric stress can be represented by multiplying the relative yield response of the individual stresses. For example, if the calculated salinity level is such that we predict a 70% relative yield and the matric stress is such that we predict a 50% relative yield, then the combined effect gives a predicted relative yield of 35%. This calculation must be based on actual measurements or modeling that accounts for the effect of salinity on matric stress and the effect of matric stress on salinity (as both reduce water uptake). The multiplication of yield response from multiple stresses has been utilized by several investigators (Suarez and Šimůnek 1997; Shani et al. 2007). Shani et al. (2007) present an extensive review of available data related to plant response to multiple stresses. The resultant user-friendly *SWS* model (see Chapter 27 of this manual) derived from *UNSATCHEM* maintains these features. The dynamic models predict water consumption based on the actual stress rather than the evapotranspiration (ET) multiplied by crop coefficient information. In the above example, if the osmotic stress produced a 70% relative yield independent of matric stress, the water consumption is reduced 30% from the crop optimal ET and the soil salinity and matric stress is reduced; thus, the predicted yield from a dynamic model is greater than the 35% value given. More detail is provided in an example in Chapter 27.

As the water content of the soil decreases, the matric and osmotic potential decreases (i.e., it becomes more negative). Evaporation and transpiration by plants remove almost pure water, leaving behind soluble salts in the soil. Depending on the water composition, salinity, plant species, and climatic conditions, about 5% to 10% of the salts are taken up by plants and the remainder is either left in the soil or leached with the drainage water.

Electrical Conductivity

Specific ion effects on plant yield are most evident in salt-sensitive species, such as rice, lettuce, strawberries, and stone fruits. Toxicity can be related to either the Na^+ cation or Cl^- anion, and is related to the ability of the individual plant species and cultivar to restrict uptake and movement of these ions.

Salinity is most easily and conveniently measured by determining the electrical conductivity (EC) of the solution (see Chapter 10 of this manual for more detail). The term specific electrical conductance (SpC) is sometimes used as well. The U.S. Salinity Laboratory (USSL 1954) showed that the EC in soil extracts was highly correlated with total salts when the data were expressed in $mmol_c/L$. The osmotic potential (OP) can be approximately related to EC by the equation $OP = -36 \times EC$, where OP is expressed in kPa and EC in dS/m at 25 °C. While useful, these approximations should not be used in research experiments where more accurate calculations are warranted. More accurate estimations of OP can be made by consideration of the ion composition of the water, such as presented in the *Extract Chem* model (Suarez and Taber 2007).

Soil-Water Extracts

The EC is used as an expression of salinity in the irrigation water (EC_{iw}), salinity in the soil saturation extract (EC_e), and salinity in the soil solution (EC_{SS}). The U.S. Salinity Laboratory researchers (1954) developed the saturation paste-saturation extract technique, a way to estimate soil salinity that uses a reference water content. The saturation paste is defined as a mixture of demineralized water added to a soil sample until the mixture (soil paste) glistens and slightly flows when the container is tipped. The soil paste is then typically left overnight to equilibrate and is filtered under suction the next day. The solution obtained is analyzed for

EC_e and soluble constituents. This extract, while not ideal, is nonetheless the most recommended for standardized representation of the soil-solution composition.

Direct determination of the soil-solution composition is difficult due to the extraction, especially when the soil is not near saturation. Also, direct determination makes spatial and temporal comparisons difficult as the composition depends on water content at time of sampling. Extracts are convenient and rapid, providing data at reference water contents. Other extracts used include 1:1, 1:2, and 1:5 soil/water ratios. Clearly, the larger the dilution, the greater the deviation from the soil-water composition in situ and the more uncertain the interpretation of the data due to dissolution, exchange, and desorption. The saturation extract has the advantage of minimizing salt dissolution, relative to other dilution-extraction methods, since less water is added, but has the disadvantage of being the most time consuming.

The water content of the saturated paste is roughly 1.5 to 2 times that of field capacity, but the exact value is quite variable depending on soil texture and mineralogy. The EC_e is thus approximately one-half the EC_{SS} at field capacity. These are relatively rough approximations suitable for field evaluation but not for reporting of salt tolerance data, as the errors can be in the range of 10% to 30%. These approximations do not consider the unique water content relation of each soil (saturated paste vs. field capacity), the nonlinearity between EC and salt content, or the reactivity of the soil, especially dissolution of gypsum if present during the addition of water and extraction.

Recently Suarez and Taber (2007) developed the *Extract Chem* program. The program allows for conversion of the inorganic chemical composition of soil water from one water content to another, considering cation exchange, precipitation/dissolution of calcite and gypsum if specified, and adsorption/desorption of B. The model calculates EC using the routines developed by McNeal et al. (1970), based on solution composition. Comparison of the model to analyzed extracts reveals some of the problems associated with extracts, such as incomplete equilibration after reaction overnight (gypsum soils), and variability in CO_2 and thus calcium depending on soil biological activity and experimental conditions.

The EC_e provides a way to assess the salinity of field samples. The relationships among EC_{iw}, EC_e, and EC_{ss} are critical, as a large amount of data on salinity tolerances of crops is based either on EC_e or EC_{iw}, whereas plant response is related to the EC_{ss}. The salinity of irrigation waters can be assessed by relating EC_{iw}, the leaching fraction (LF), the EC_{ss} at field capacity, and the salt tolerance of crops of interest. Unfortunately, there are various recommendations for calculation of the soil salinity relevant to crop response, and they provide significantly different results. See Chapter 10 for a more detailed discussion of the variability in soil salinity tests.

Plant Response to Soil Salinity

The most common way to represent the soil-solution EC relevant to plant response has been to use the average soil EC_e (Ayers and Westcot 1985). This method simply averages the calculated or measured EC_e of several depths. If the EC_e data are not available, it has been suggested to calculate average EC_e using the EC_{iw} and the concentration factor F_c, which equals 1/LF at the bottom of the rootzone, and an assumed distribution of water uptake (Rhoades 1984; Ayers and Westcot 1985). Using this method, it is assumed that water is removed by ET in proportions of 0.40, 0.30, 0.20, and 0.10, from the rootzone's first, second, third, and fourth quarters, respectively. Alternatively, an exponential water uptake function can be used; however, the concentration factors (F_c values) would not greatly change.

Since the EC_e is about one-half of the EC_{SS}, the F_c values to convert from EC_{iw} to EC_e are 2.79, 1.88, 1.29, 1.03, 0.87, and 0.77 for LF of 0.05, 0.10, 0.20, 0.30, 0.40, and 0.50, respectively. These F_c values have been used to calculate the EC_e values expected in the rootzone as a function of overall LF. These in turn have been used to calculate average rootzone soil EC_e as related to LF and EC_{iw}.

The use of the average rootzone EC_e to predict salinity effects on crop yield is widely accepted but questionable on several grounds. First, plant water uptake is not uniform throughout the rootzone. If we use the same water uptake functions that were used to generate the EC soil profiles, multiply the soil salinity at each depth by these factors, and sum the product for the rootzone, then we generate EC values that correspond to the average EC of the water that the plant has taken up. These uptake-corrected EC values are considerably lower than the average EC_e values, and the differences increase with decreasing LF, as shown in Table 11-1. For example, at an LF 0.05, the mean soil EC is 55% greater than the uptake-weighted EC, whereas at an LF of 0.5 it is only 10% greater. It is recommended to use these uptake-weighted factors and not the average salinity to calculate plant response to soil salinity. As long as we use the same function or distribution for water uptake as we used to calculate the soil salinity depth profile from LF and irrigation water EC, then we will have a reasonable estimate of the salinity experienced by the plant. For instance, if the water uptake pattern is different from that assumed here, we still get the same uptake-weighted salinity concentration factors as the water uptake drives the salinity distribution. We need only ensure that we have divided the soil into sufficient compartments (four compartments appears satisfactory in most instances).

TABLE 11-1. Relative Solute Concentrations of Soil Water (Field Capacity Basis, F_c) Compared to That of Irrigation Water Related to Depth in the Rootzone and Leaching Fraction[a]

		F_c at Leaching Fraction Values Of:					
Rootzone in Quarters (1)	V_{cu}[b] (2)	0.05 (3)	0.10 (4)	0.20 (5)	0.30 (6)	0.40 (7)	0.50 (8)
0	0	1.0	1.0	1.0	1.0	1.0	1.0
1	40	1.61	1.56	1.47	1.39	1.32	1.25
2	70	3.03	2.70	2.27	1.96	1.72	1.54
3	90	7.14	5.26	3.57	2.70	2.17	1.82
4	100	20.00	10.0	5.00	3.33	2.50	2.00
Mean F_c[c]							
Uptake-weighted F_c[d]		5.58	3.76	2.58	2.06	1.74	1.53
		3.6	2.71	2.07	1.75	1.54	1.40

[a] Assuming a water uptake of 0.4, 0.3, 0.2, and 0.1, respectively, from the first through fourth quarters of the root zone

[b] Cumulative percentage of consumptive use above each indicated depth in the rootzone

[c] The average for the rootzone obtained by the sum of quarter of the root zone divided by 4

[d] The water uptake-weighted mean for the rootzone

The water uptake-weighted salinity, while more realistic than the mean rootzone salinity in representing plant salt stress, is nonetheless still a simplification. It does not consider the following factors:

1. In the short term, plants can compensate for reduced water uptake in some areas of the rootzone by increased uptake in other regions. However, in the longer term, this redistribution of water uptake causes a redistribution of roots and redistribution of the salinity profile, with the water uptake reverting back to the previous concentration factors. For example, if plants consume 90% of the water applied, then over time they must extract water up to the salinity level corresponding to this concentration factor, and the water uptake-weighted salinity goes back to the steady-state concentration factors listed here and in Table 11-1.
2. The concentration factors do not consider the changes in EC due to chemical processes, mostly calcite and gypsum precipitation and dissolution; these can easily change the concentration factors by ±10% to 30% or more, depending on the specific conditions. In most instances this results in lower salinity than calculated by the concentration factors. The important exception, where salinity in the soil is greater than

that calculated by the concentration factors, is when a gypsiferous soil is irrigated with a water containing small concentrations of calcium and sulfate.

3. The steady-state factors do not consider the dynamics of wetting and drying cycles. As the soil dries out, the resultant in situ soil salinity and EC increase. Infrequent irrigation results in increased soil salinity averaged over time, in addition to possible matric stress. This is not an issue in the case of high-frequency irrigation.
4. When the LF is calculated, the actual ET—not the potential ET—must be considered as increased salinity results in decreased plant water uptake. This requires a feedback loop from the salt stress response to the calculation of ET. The *UNSATCHEM* model (Suarez and Šimůnek 1997), and the user-friendly *SWS* version (see Chapter 27) uses a water uptake response function (separate osmotic and matric functions) at each point in the rootzone. Thus, the LF fraction calculated by the model is not solely defined from $ET_{o,}$ crop coefficients and water inputs.

If plant response is to osmotic stress, then osmotic stress needs to be calculated rather than estimated from EC, as there is a significant difference in the relationship of osmotic pressure and EC for chloride salts compared to sulfate salts. The *SWS* model also calculates osmotic pressure and EC after consideration of chemical processes. The salinity threshold values, meaning the salinity at which plant yields start to decline, are derived from the following relationship between yield and EC_e:

$$\text{Yield} = 100 - B\,(EC_e - A) \tag{11-1}$$

where A = the salinity concentration at which growth depression (threshold) starts, and B = the percent of yield decrease per unit EC_e above the threshold level (Maas and Hoffman 1977).

Figure 11-1 shows the relationships between EC_{ss} and EC_{iw} for various LF based on calculations as described for Table 11-1. In the previous edition of this manual (1990), Fig. 11-1 was used for high-frequency irrigation systems only and the average rootzone salinity was used for furrow and other nonfrequent irrigation systems. This special consideration has been dropped because, despite theoretical expectations, there is no clear evidence that frequent irrigation reduces salt damage (Shalhavet 1994). Conversion of these EC_{ss} data to EC_e should consider the specific soil properties and water composition; in the absence of such information, the user would have to use the approximate conversion $EC_e \approx 0.5\ EC_{ss}$.

To use Fig. 11-1 for evaluation of potential yield loss due to salinity damage, determine the EC_{iw} and then estimate the range in LF that can be obtained for the soil with the available irrigation management system. Next, compare the resultant EC_{ss} values with the EC_{ss} values from the salt

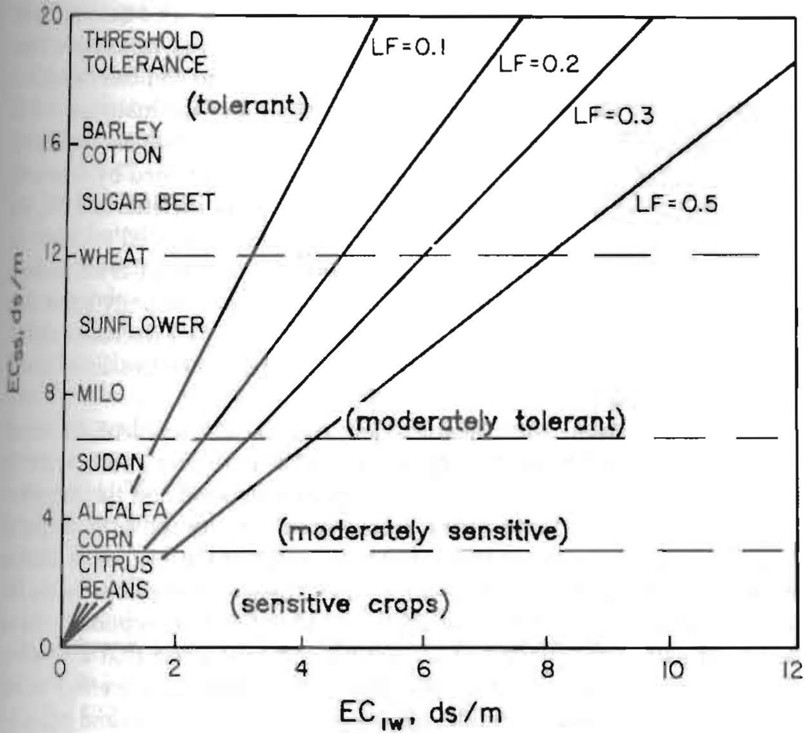


FIGURE 11-1. *Relationship between average rootzone salinity (field capacity basis), EC of irrigation water, and LF required to avoid yield loss. Modified from Rhoades (1982).*

tolerance tables. This will indicate crops that can be grown successfully without decreases in yield from salinity. For example, if EC$_{iw}$ is 4.0 and an LF of 0.20 is expected, only salt-tolerant plants can be grown without yield loss. If LFs of 0.5 or greater are possible, moderately salt-tolerant plants can be grown. If the nature of the soil hydraulic properties or water availability is such that only very small LFs are possible, then in this instance (where EC$_{iw}$ = 4.0) the water will reduce yields in even the most salt-tolerant crops. Thus, assessing the effects of salinity as a parameter of water quality depends on the soil, crops, amount of water available, reference crop ET of the site (ET$_0$), irrigation system, irrigator's expertise in achieving the needed leaching, and decrease in yield that can be tolerated. In short, from the standpoint of salinity, the suitability of a given irrigation water supply requires an evaluation of how the applied water will interact with the soils, the resultant LF (dependent on ET$_0$ and salt stress), and the net change on soil salinity.

The method of assessing the water salinity as described can be adapted to different sites. The suitability of the water supply can be assessed based on such local conditions as the ease with which the soil can be leached, salt tolerance of the crops, irrigation system, skill of the manager, and climate. Perhaps the weakest link in this system is the estimation of the LF, which is seldom measured directly, but often determined by measuring water application and estimating ET from crop coefficients and ET_0. The difficulty is that not all applied water infiltrates (we need to correct for surface runoff, often called tail water), and that actual ET is not an input but a response, depending on crop stress. If there is salinity stress, then for a fixed application of water, as salinity increases, ET decreases and the LF increases, with LF determined by the crop response to salinity as well as by the water application.

In irrigation waters that are sprinkled, there is also a potential for direct injury to the plant from absorption of salts in the irrigation water by foliage. The foliar injury from salts on plants depends on the concentrations of the individual ions in the water, sensitivity of the crop, frequency of sprinkling, presence of sunlight, and environmental factors (such as temperature, relative humidity, and water stress of the plants before irrigation). Maas et al. (1982) reported that rates of salt absorption by leaves increased as the frequency of irrigation increased but that a threefold increase in the duration of sprinkling had no measurable effect on salt absorption. Night-time sprinkling reduces foliar absorption and injury.

Foliar absorption by Na^+ or $Cl-$ ions at concentrations of less than 5 mmol/L damages some fruit trees. Other crops can tolerate Na^+ and Cl^- ion concentrations of greater than 20 mmol/L. Thus, no concentration limits can be recommended, although an increase in Na or Cl in the water reduces its suitability for sprinkler systems by reducing the types of crops that can be grown without foliar injury. Also, the degree of injury depends on the crop, the irrigation system, and how it is operated. For example, Suarez et al. (2003) observed almost a doubling of the Se shoot concentration of Brassica species under sprinkler rather than flood irrigation, but the relative increase in Se uptake was crop-dependent. Foliar uptake can be expected to be related to shoot morphology, as well as leaf structural characteristics. Consequently, limits or guidelines for sprinkler irrigation at current levels of knowledge are too arbitrary to be useful.

SODICITY

Sodium hazards of irrigation and soil waters can negatively affect crop production due to both specific ion toxicity (as discussed) and the adverse effect of Na on soil physical properties, especially water infiltration. The growth of plants is, thus, affected by either an unavailability of soil water or poor aeration due to reduced water movement and subsequent waterlogging.

The reduction in water infiltration caused by Na can usually be attributed to surface crusting, dispersion and migration of clay into the soil pores, and swelling of expandable clays. All of these phenomena relate to the distance of charge neutralization for soil particles, predominantly clay, but also oxides in more weathered landscapes.

The hydrated exchangeable cations neutralize the net negative charge on clays. The distance of charge neutralization (the double-layer thickness) depends on the cation valence, hydration energy, and ion concentration in solution. Divalent cations, such as Ca^{2+} and Mg^{2+}, neutralize the surface charge in relatively short distances, even at low concentrations. Particles are repulsed when the charge is neutralized too far from the surface and the electrostatic repulsion between particles exceeds the attractive (van der Waals) forces. In contrast to Ca^{2+} and Mg^{2+} ions, the exchangeable Na^+ ion neutralizes the surface charge at a longer distance (much larger, double-layer thickness) and requires high concentrations in solution before particle aggregation and swelling are reduced. Consider Ca^{2+} as a stabilizing ion, Mg^{2+} less so (Dontosova and Norton 2002), and Na^+ as a destabilizing ion in regard to the soil structure.

The sodicity of a soil is given by the exchangeable sodium percentage, ESP, which is the percentage of the exchangeable charge neutralized by Na^+. The ESP of a soil can be estimated from the sodium adsorption ration (SAR) of the water, in other words, $ESP = 1.475\ SAR/(1 + 0.0147\ SAR)$, based on a set of data from soils in the western United States (U.S. Salinity Laboratory 1954; also see Chapter 3 of this manual). The ESP value alone is insufficient for predicting soil stability. Soil structure depends on many other factors, including soil salinity, tillage, mineralogy, organic matter, and pH.

Sodicity Hazard Guidelines

The sodic-hazard potential of water is often evaluated from the SAR and salinity. At the same SAR, the dispersion potential of dilute water exceeds that of a more saline water. Various investigators have developed stability lines related to concentration and SAR. Perhaps the most widely used is that presented by Ayers and Westcot (1985). Figure 11-2 shows the guidelines of Rhoades (1982) and Quirk and Schofield (1955) represented as solid and dashed lines, respectively. Rhoades based his guidelines primarily on experience and data from arid soils in California. Quirk and Schofield (1955) based their guidelines on a noncalcareous soil in England. In each instance, the region below the line represents unstable soil structure and permeability loss, and the region above it represents stable permeability.

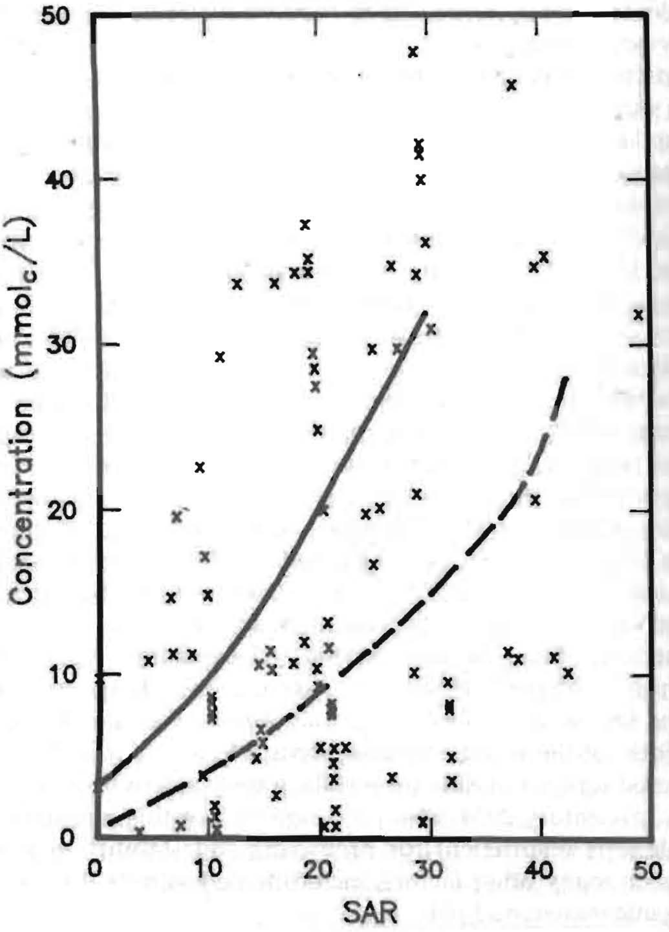


FIGURE 11-2. Relationship between SAR and solute concentration (in mmol/L) at which a 25% reduction in soil hydraulic conductivity was observed. The data were obtained from laboratory studies of packed soil columns containing arid-land soils. The dashed and solid lines are guideline values recommended by Quirk and Schofield (1955) and Rhoades (1982), respectively.

Figure 11-2 also shows the concentration and SAR values at which a 25% reduction in saturated hydraulic conductivity took place in packed laboratory soil columns, from available published data from arid soils. A general relationship cannot be predicted because soils greatly differ, but a good SAR versus concentration relationship for a set of soils from a region or locality is always possible. For all arid soils examined, decreasing salinity, or increasing sodicity, or both, decreases soil stability.

Differences among soils shown in Fig. 11-2 are at least partly due to experimental procedures used by different researchers, such as different column packing, flow rates, and saturation methods. However, variations in clay mineralogy, clay content, organic matter, and oxide content likely account for most of the variation. Almost all of the soils have been examined under relatively low pH (<7.0).

For specific groups of soils, several researchers have demonstrated that soil stability correlates well with organic matter, or oxide content, or both. Soils with a very large amount of oxides, such as some tropical soils, show little or no loss of hydraulic conductivity, even when saturated with sodium and equilibrated to minimal levels of salinity. Organic soils may also be highly stable at low salinity, as long as the pH is not elevated. Texture and initial hydraulic characteristics have also not received sufficient attention. Sandy soils with high infiltration rates can remain productive with 25% losses in infiltration rates, but this is not true for clay soils where infiltration may be barely sufficient to supply crop water needs during high ET conditions, even without sodicity effects.

Since the effects of variables other than salinity, SAR, and their interactions have not been quantified, EC-SAR suitability figures offer only an approximate guideline. A representation of the stability of arid zone soils as related to irrigation water quality in the absence of rain is shown in Fig. 11-3, based primarily on research at the U.S. Salinity Laboratory. This guideline differs from the other guidelines in that it includes the effect of pH and is based to a considerable extent on longer-term infiltration experiments. The relationships at low SAR and EC are primarily based on the experiments of Suarez et al. (2006, 2008) and D. L. Suarez and A. Gonzalez Rubio (unpublished data). The slopes of the solid lines (EC vs. SAR) are similar to those used by others (Ayers and Westcot 1985; Rhoades 1982). The area between the lines represents a region of little to 25% reduction in infiltration. If we were to select a line where all soils had 25% or less reduction in infiltration, it would be a line almost on the x-axis, as shown by the data in Fig. 11-2. Similarly, the upper left line in Fig. 11-3 is for a typical arid land soil; below that line less stable soils may already have severe reductions in infiltration.

Effect of pH on Infiltration

Increasing pH is known to increase the salt concentration necessary for flocculation of soil clays (Suarez et al. 1984; Goldberg and Forster 1990). An increase in pH also has an adverse effect on saturated hydraulic conductivity (Suarez et al. 1984). Also shown in Fig. 11-3 are estimates of the impact of irrigation water pH on water infiltration. Increased pH has an adverse effect on infiltration (D. L. Suarez and A. Gonzalez Rubio, unpublished data), as well as saturated hydraulic conductivity. The relative

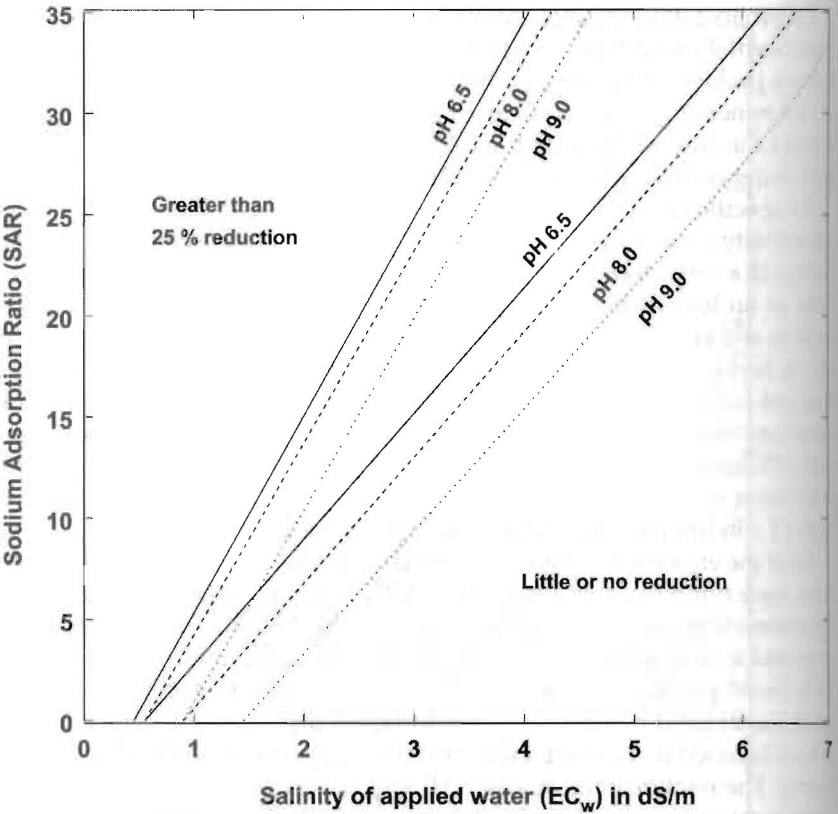


FIGURE 11-3 Relationship between SAR and solute concentration (in mmol/L), at which little or no reduction or greater than 25% reduction in infiltration is expected, based on data from arid and semiarid land soils, in the absence of rain. The solid lines represent the effects for waters at pH 6.5, the dashed lines represent the effects for waters at pH 8, and the dotted lines are for waters at pH 9.

impact of pH on infiltration may also be expected to depend on texture and clay type. The information on the effect of pH on hydraulic conductivity or infiltration is limited to only a few soils.

Figure 11-4 shows a representation of the impact of irrigation water quality in the presence of substantial rain. It represents the results of different experiments conducted in Riverside, California, with various soils of different texture and geographic origin (Suarez et. al. 2006; Suarez et al. 2008; D. L. Suarez and A. Gonzalez Rubio, unpublished data). In this representation, no irrigation waters with an SAR above 5 can be considered safe in the presence of rain. The adverse results of the rain results not only from physical impact of the drops but also from the chemical changes at

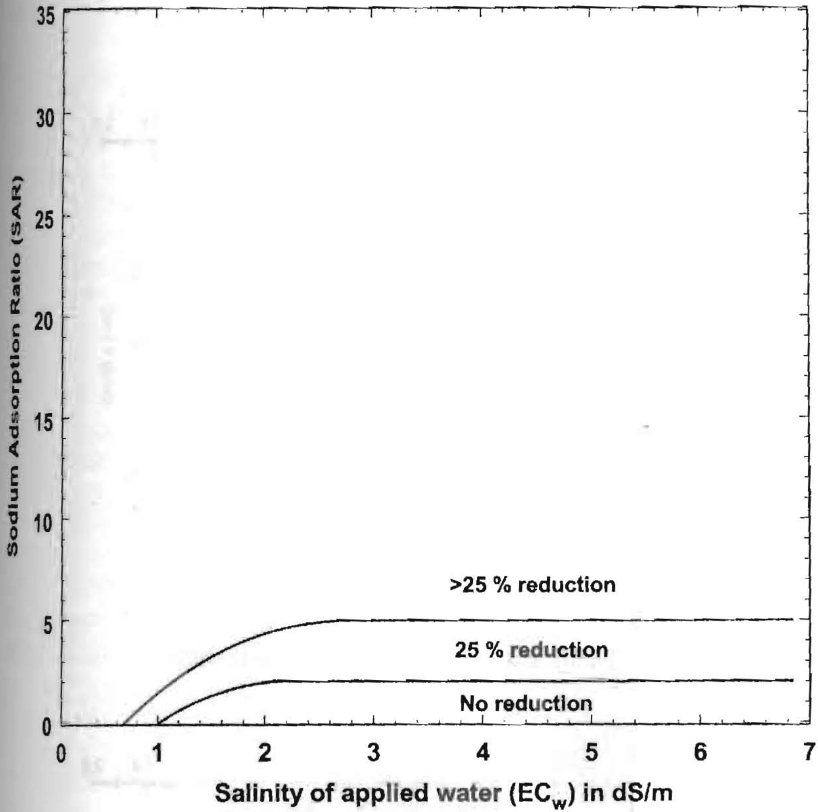


FIGURE 11-4. Relationship between SAR and solute concentration (in mmol/L), at which a 25% reduction in infiltration is expected, based on data from arid and semiarid land soils, in the presence of rain.

the soil surface. Rain results in a rapid decrease in EC as the water infiltrates, as shown in Fig. 11-5, for simulations of two calcareous soils of differing texture (Suarez et al. 2006).

As shown in Fig. 11-6, with infiltration of rain, there is a much slower change in SAR than EC (Fig. 11-5) and the change depends on the cation exchange content of the soil, with higher cation exchange soils having a greater resistance to changes in SAR. Noncalcareous soils would have a considerably slower change in SAR than shown in Fig. 11-6, thus increased sensitivity to rain on a sodic soil.

It is recommended that the effects of an irrigation water be tested directly on the soil of interest with column leaching studies, tests of aggregate stability, or tests of flocculation after the soil has been dispersed in a

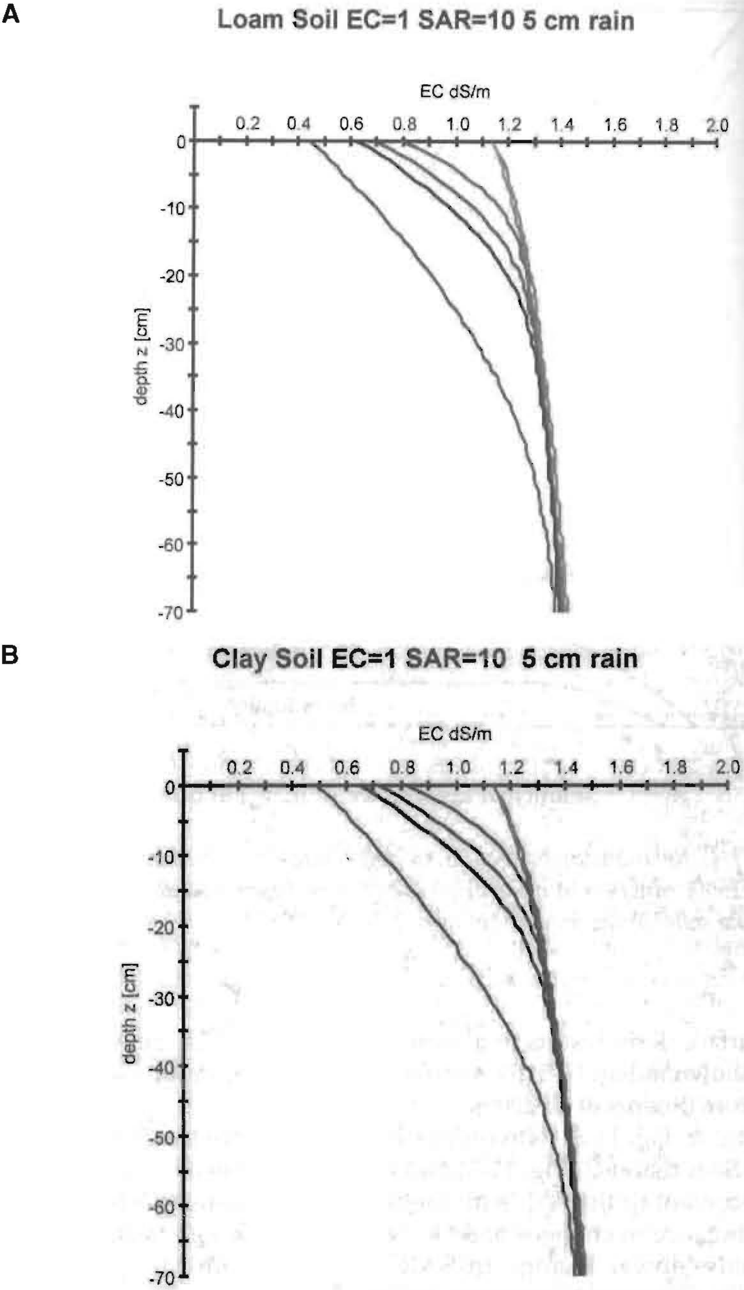
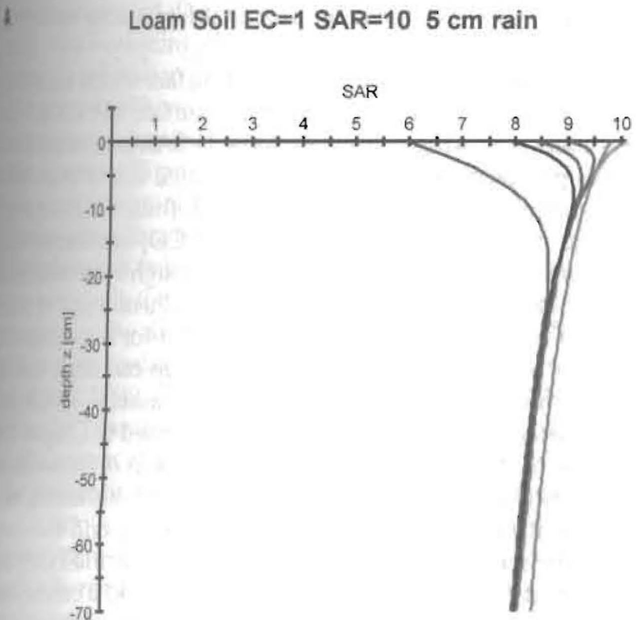


FIGURE 11-5. Predicted relationship of EC with depth and quantity of rain infiltrated for (a) loam soil, and (b) clay soil. The initial condition was EC = 1.0 ds/m and SAR 10. Each curve represents the addition of 1 cm of rain.



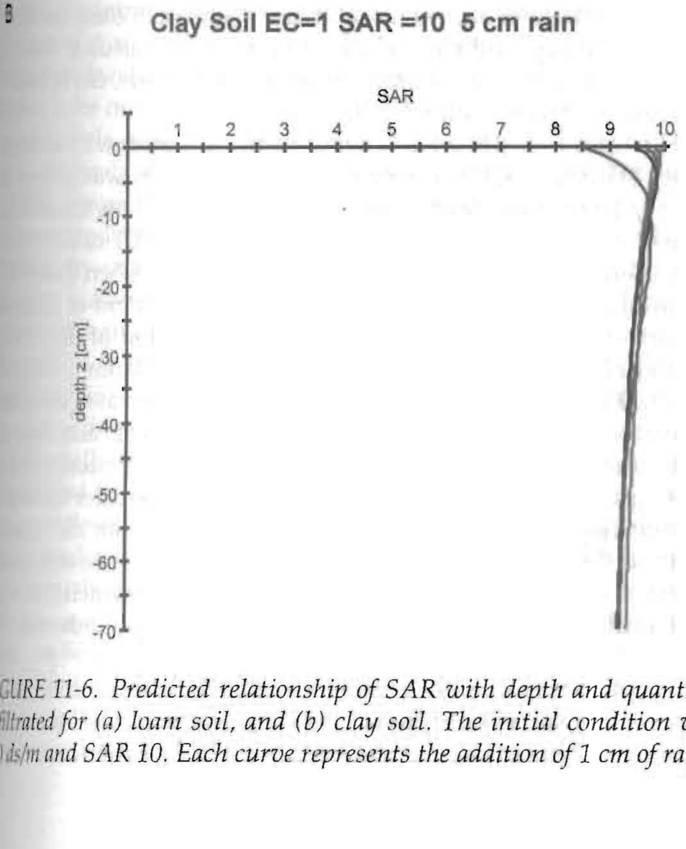


FIGURE 11-6. *Predicted relationship of SAR with depth and quantity of rain infiltrated for (a) loam soil, and (b) clay soil. The initial condition was EC = 1.0 ds/m and SAR 10. Each curve represents the addition of 1 cm of rain.*

test tube. This need arises because of the variability among soils in their response to Na.

The SAR value calculated from analyses of surface waters usually represents the SAR of the irrigation water on the surface of the soil. In this instance there is little justification for making an SAR adjustment. This is not the case for groundwaters that are equilibrated with a much higher partial pressure of carbon dioxide (PCO_2) and, thus, are lower in pH. Exposure of the groundwater to atmospheric CO_2 conditions, which occurs with sprinkler irrigation or conveyance through open canals, raises the pH and may cause calcite precipitation. The adjusted SAR of the irrigation water is a correction of the SAR to account for the change in Ca concentrations as related to changes in the calcium carbonate solubility (see Chapter 3 of this manual). In such instances, assume a PCO_2 of 0.1 kPa at the soil surface and adjust the SAR as described in Chapter 3. The adjusted SAR can also be used to estimate the SAR in or below the rootzone by correcting for mineral precipitation and assuming no ion exchange. The concentration factor (1/LF), the PCO_2, and the chemical composition of the irrigation water are needed. If specific PCO_2 data in the rootzone are unavailable, the values of 1 kPa and 5 kPa can be used for sandy and clay soils, respectively. Since the publication of the earlier edition of this manual, there has been a dramatic increase in computer availability, user capability, and ease of use of software to calculate chemical equilibria. The *Extract Chem* software, among many others, can be readily used to calculate a precise, adjusted SAR value.

High pH values (i.e., pH >8.5) always indicate waters with an excess of alkalinity ($HCO_3^- + CO_3^{2-}$) over Ca. These high-pH waters pose an extra sodicity hazard for several reasons. When alkalinity exceeds Ca, the increased concentration of salts in the soil due to ET causes calcite precipitation and a decrease in the Ca concentration (when Ca = alkalinity in $mmol_c/L$), the Ca concentration remains constant or increases slightly during plant water extraction or evaporation of the water). Waters with pH values below 8.5 can also have high alkalinity, depending on PCO_2. For groundwaters, samples should be aerated or shaken until the water is equilibrated with ambient CO_2 levels and then the pH remeasured. If pH >8.5 after aeration, then the concentration of alkalinity is greater than Ca. The higher the pH, the greater the imbalance. Imbalances in alkalinity and Ca concentrations can also exist in waters with a pH <8.5 if they are dilute waters that are undersaturated with respect to calcite, such as surface waters from snowmelt. The very low EC of rain or snowmelt water (0.1 dS/m) compounds their Na hazard.

High pH (pH >9.0) directly and adversely affects infiltration as discussed above, as well as limiting Ca concentrations and increasing the SAR. The sodicity hazard of an irrigation water also depends on the management system used. Dispersing the soil at the surface requires inputs of energy and a weakening of chemical bonding. Irrigation by sprinkler will increase the infiltration problems associated with irrigation waters. Due to the greater likelihood of surface crusting, sprinkler irrigation may be unsuitable for waters that tend to cause dispersion or swelling. Drip or surface systems will produce less physical disruption with such waters. This is a very important consideration when using waters with a potential dispersion or swelling hazard.

High pH values in irrigation waters may cause nutritional and infiltration problems and, thus, need to be amended to reduce the alkalinity. The composition of the divalent ion component slightly affects the stability of soils at a particular ESP value, with Ca slightly more stable than Mg (McNeal et al. 1968). The greater selectivity of most soils for Ca^{2+} as compared to Mg^{2+} means that the Mg-Na system has a higher ESP than does the Ca-Na system at the same SAR values. This, combined with the high pH that frequently occurs in low Ca systems, also accounts for why Mg seems deleterious to infiltration as compared to Ca under field conditions.

Weathering of Ca containing minerals, primarily gypsum, calcite, and dolomite, decreases SAR and increases electrolyte concentration. The hydraulic conductivity response of sodic soil in arid areas to rain or waters of minimum salinity seems related to its weathering potential (Shainberg et al. 1981). The more stable soils appear to maintain higher electrolyte levels than do unstable soils. Tropical Hawaiian soils also appear to be much more stable than arid soils at comparable SAR and salinity levels, possibly due to their high oxide content (McNeal et al. 1968). Moderate amounts of organic matter also increase the stability of a soil (Kemper and Koch 1966; Dong et al. 1983).

The potential hazard of reduced water infiltration is partly related to the intensity and timing of rainfall in a region. Rainfall, generally <0.06 dS/m, is relatively pure water. When it infiltrates the soil, the salinity of surface soil can decrease rapidly but the soil may remain at almost the same ESP. As a result, the potential for dispersion by rainfall is especially high if the ESP of the soil is high. Rainfall, as with sprinkler irrigation, contributes dispersive energy.

Surface (flood, furrow, or drip) irrigation also can cause particles to migrate and result in sealing of the soil surface. However, the inputs of energy are less than with sprinkler irrigation, and lower-quality water can be better tolerated. In areas with little rainfall, such as California's Imperial Valley, these phenomena are generally neglected. In areas with appreciable seasonal rainfall (>200 mm), surface amendments, such as gypsum, can be applied to maintain the electrolyte concentrations above values for dispersion and swelling to allow the water to infiltrate. If rainfall occurs throughout the year, tillage or repeated gypsum applications may be needed.

IONIC BALANCES

Calcium (Ca), Mg, K, S, N, and P are the major elements needed for plant nutrition. Minor requirements include Fe, Mn, Zn, Cu, Mo, and Ni (Marschner 1995). Plants generally tolerate widely varying concentrations of the major cations, including Na, which is not required for plant growth. The Ca requirement of a plant is generally low, i.e., 0.7 mmol/L to 1.5 mmol/L; however it appears to depend on the presence of other ions. The Ca requirement may be related to ion competition and, thus, is better expressed in terms of ion ratios. High Mg/Ca ratios in solution may result in Ca deficiencies in plants, despite high absolute Ca concentrations. Carter et al. (1979) observed reduced growth in barley, starting at Mg/Ca ratios of 1.0, independent of salinity or absolute Ca concentrations. Calcium requirements are also greater at low pH than at high pH (Marschner 1995). Guidelines for specific cation ratios cannot be developed at this time due to insufficient information; cultivars respond in widely varied ways to cation composition, and this has not been sufficiently researched.

Concentrations of Mg and K generally are high enough in irrigation waters to prevent deficiency symptoms in plants. The micronutrient cations of Fe, Mn, Cu, and Zn are virtually absent from most irrigation waters, but the soil generally supplies these nutrients. The irrigation water limits the availability of micronutrients if the water causes the soil pH to increase. Specific cation toxicity takes place with excess Na, predominantly in citrus and stone fruits.

For anions, specific toxicities occur, rather than ionic imbalances. Although most plants tolerate high Cl concentrations, woody species and some grape rootstocks do not. High levels of nitrate, often associated with saline waters, may narrow the selection of crops suitable for irrigation. Nitrogen is one of the essential elements for the growth of plants. Optimal growth requires 2% to 5% N on a dry weight basis, depending on the species, developmental stage, organ to be optimized, and ultimate use of the plant or parts (Marschner 1995). High levels of nitrate during early growth enhance shoot elongation which, in cereals, increases susceptibility to lodging. High substrate levels of nitrate increase total N in ryegrass but decrease carbohydrates and increase cellulose content. Such high levels of NO_3, that is, $>1\%$ to 2% by dry weight, can be toxic to grazing animals. High nitrate concentrations can cause excessive vegetative growth and reduce production of fruits and other harvested products. Excess N reduces the production of fruits in some varieties of tomatoes and reduces the sugar content and increases impurities in sugar beets. Due to such problems, Ayers and Westcot (1985) indicate that the restriction on the use of water increases as the NO_3-N concentration increases from 5 mg/L to 30 mg/L.

BORON

Boron is essential for and potentially toxic to plants. Boron deficiencies take place in the (g/L concentration range in soil solutions. Boron toxicities take place at concentrations above a few mg/L for most plants. In the toxicity range, plants respond to B in the soil solution (B_s) rather than to B adsorbed on soil particles. Hence, solution and sand-culture data are used to evaluate the response of plants to B.

Bingham et al. (1985) and Francois (1984) demonstrated that yield decreases related to B toxicity can be fitted to the two-parameter model used to describe salt tolerance (Maas and Hoffman 1977). The expression for this model is

$$Y = 100 - m\,(x - A) \tag{11-2}$$

where Y = relative yield, m = the decrease in yield per unit increase in B concentration, A = the maximum concentration of B that does not reduce yield (threshold); and x = the B concentration in the nutrient, sand culture, or soil solution.

Early recommendations and ranking of B tolerance of plants were largely based on visual symptoms. Francois (1984) showed that visual symptoms of B toxicity do not generally correlate with the yield of marketable product.

Decreases in yield from B toxicity depend on the tolerance of the crop to B and on the B_s, which depends on the concentration of B in the irrigation water (B_{iw}), the LF, and the departure from a steady-state relationship between adsorbed B and B_s. At steady-state input and output of B from the rootzone, the mean B_s is related to B_{iw} and the LF, in the same manner as for salinity (Table 11-1), as shown in Fig. 11-7.

Since B is adsorbed onto and released from the surfaces of soil particles, soil solutions are buffered against rapid changes in B concentration. If the B in irrigation water is increased, B is adsorbed, resulting in a smaller increase in the solute B concentration than the increase to irrigation water. The time required to reach a steady-state concentration of B depends on the increased B concentration, the amount of water used, the LF, and the sorption capacity of the soil volume of the rootzone. Jame et al. (1982) reported that the time ranged from 3 to 150 years. Three years was adequate for a sandy soil that can adsorb small quantities of B and has been treated with a B solution of 10 mg/L, and 150 years was required for a clay loam soil that could adsorb large quantities of B and was treated with a solution of 0.1 mg/L B. If B in the irrigation water is decreased, the soil releases B and time is needed to reduce the B_s. Also, as a rough approximation, the volume of low-B water needed to reduce the B_s from toxic to nontoxic levels is two to three times

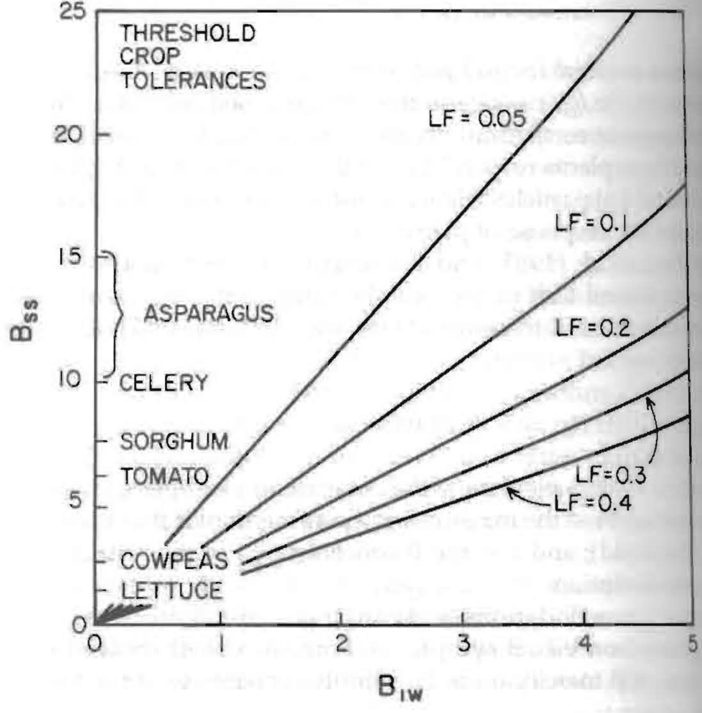


FIGURE 11-7. Relationship between average rootzone boron (field capacity basis), boron in irrigation water, and LF required to avoid yield loss. Modified from Rhoades (1982).

greater than is needed for a comparable reduction in Cl when the pH is below 7.5.

The ratio of concentration in the soil solution at field capacity to the concentration in the saturation extract is approximately 2 for anions not adsorbed or precipitated, such as Cl. However, the ratio is <2 for B because the adsorption of B on the soil surface depends on the concentration. When ET decreases the water content and concentrates the soil solution, B is adsorbed. The concentration factor decreases as the adsorption or buffer capacity of the soil increases. Jame et al. (1982) reported that this ratio ranged from 1.0 to 1.8, depending on the concentration of B and the adsorption capacity of the soil. Consequently, the B concentration in the saturation extract does not adequately represent or indicate B toxicity under field conditions. It is suggested that the *ExtractChem* model be used to convert B concentrations from one water content to another.

TRACE ELEMENTS

Trace elements are those that occur in waters and soil solutions at concentrations of less than a few mg/L, with most concentrations in the µg/L range. Some are essential for plants and animals, but all can become toxic to plants and animals at elevated concentrations or doses.

Virtually no experiments have yet been conducted to determine the criteria for quality for trace elements in irrigation water. Hence, guidelines have been based on results from sand, solution, and pot cultures, field trials with applications of chemicals, laboratory studies of chemical reactions, and animal feeding and grazing trials.

Table 11-2 presents the recommended maximum concentrations of 3 trace elements. Shown for comparison are the U.S. Environmental Protection Agency drinking water standards (US EPA 1985, 2008). In most (but not all) instances, the drinking standards are lower than the recommended maximum concentrations for irrigation. The irrigation standards are designed to protect the most sensitive crops and animals that consume those crops from toxicities when the most vulnerable soils are irrigated. These concentrations should be considered as guidelines but not as criteria for water quality.

If sufficient knowledge becomes available to show that these concentrations can be exceeded without adversely affecting soils, crops, and animals, then new guidelines can be established. For example, for the irrigated lands of the west side of California's San Joaquin Valley, Pratt et al. (1988) recommended that the guideline for Se in the selenate form be increased to 0.10 mg/L and the guideline for Mo be increased to 0.05 mg/L. The conditions included alkaline, fine-textured soils; saline drainage waters, which need high LFs to prevent reduced yields; and drainage waters dominated by SO_4 anions, which inhibit the absorption of Se and Mo by plants. These guidelines do not consider the long-term consequences on soil loading or the impact on discharge of drainage water to surface or subsurface water supplies. In most instances these environmental considerations are the limiting factor when using waters elevated in trace elements. Sprinkler irrigation may also result in increased trace element uptake.

Other water-quality guidelines list the elements Al, Fe, Sn, Ti, and W (NAS 1973; Ayers and Westcot 1985), but limits for these elements have little meaning. If certain soil conditions develop, such as low pH for Al and highly reduced, waterlogged conditions for Fe, these elements can become toxic to plants due to the dissolution of Al or Fe from soil solids. Aerated soils with pH values above 5.5 will precipitate the Al and Fe in irrigation waters. When pH values are >7, the solubility of most trace metals is greatly reduced. Guidelines for tin (Sn) titanium (Ti), and tungsten (W) cannot be made due to insufficient information.

TABLE 11-2. Recommended Maximum Concentrations of 15 Trace Elements in Irrigation Waters for Long-Term Protection of Plants and Animals

Element (1)	Recommended Maximum Concentration[a] (mg/L) (2)	US EPA MCl[b] Drinking Water (3)	Comments (4)
Arsenic	0.10	0.01	This guideline will protect sensitive crops grown on sandy soils. Higher concentrations can be tolerated by some crops for short periods when grown in fine-textured soils.
Beryllium	0.10	0.004	Toxicities to plants have been reported at concentrations as low as 0.5 mg/L in nutrient solutions and at levels in the soil greater than 4% of the cation-exchange capacity.
Cadmium	0.01	0.005	Concentrations ≤0.01 mg/L will require 50 years or more to exceed the recommended maximum Cd loading rate. Removal in crops and by leaching will partially compensate and perhaps allow use of the water indefinitely.
Chromium	0.10	0.10	Toxicity in nutrient solutions has been observed at a concentration of 0.50 mg/L and in soil cultures at a rate of 120 kg/ha. Toxicity depends on the form of Cr existing in the water and soil and on soil reactions.
Cobalt	0.05	N.S.[e]	A concentration of 0.10 mg/L is near the toxic threshold for many plants grown in nutrient solution. Toxicity varies, depending on type of crop and soil chemistry.
Copper	0.20	1.3	Concentrations of 0.1 mg/L to 1.0 mg/l in nutrient solutions have been found to be toxic to plants, but soil reactions usually precipitate or adsorb Cu so that soluble Cu does not readily accumulate.
Fluoride	1.0	4.0	This concentration is designed to protect crops grown in acid soils. Neutral and alkaline soils usually inactivate F, so higher concentrations can be tolerated.

Lead	5.0	0.015	Plants are relatively tolerant to Pb, and soils effectively sorb or precipitate it. Toxicity to animals typically is caused not by Pb absorption from soils but by aerial deposition of lead on foliage of pasture and forage plants.
Lithium	2.5[c]	N.S.[e]	Most crops are tolerant to Li up to 5 mg/L in nutrient solutions. Citrus, however, is highly sensitive to Li. Lithium is a highly mobile cation that will leach from soils over an extended period of time.
Manganese	0.20	0.05[d]	Some crops show Mn toxicities at a fraction of a mg/L in nutrient solution, but typical soil pH and oxidation-reduction potentials necessary for plant growth control Mn in the soil solution so that the Mn concentration of irrigation water is relatively unimportant.
Molybdenum	0.01	N.S.[e]	This concentration is below phytoxic level but is recommended to protect animals from molybdosis because of excess Mo in forages.
Nickel	0.20		Many plants show toxicity at Ni concentrations of 0.5 mg/L to 1.0 mg/l. Toxicity of this element decreases with increase in pH, so acid soils are the most sensitive.
Selenium	0.02	0.05	This guideline will protect livestock from selenosis because of Se in forage. Selenium absorption by plants is greatly inhibited by SO_4, so the guideline for this element can be increased for gypsiferous soils and waters.
Vanadium	0.10	N.S.[e]	Toxicity to some plants has been recorded at V concentrations above 0.5 mg/L.
Zinc	0.50	5[d]	A number of plants show Zn toxicity at concentration of 1 mg/L in nutrient solution, but soils have a large capacity to precipitate this element. This guideline is designed to provide protection for acid sandy soils. Neutral and alkaline soils can accept much larger concentrations without developing toxicities.

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[a] Loading rates in kg/ha-yr can be calculated from the relationship that 1 mg/L in the water gives 10 kg/ha-yr when water is used at a rate of 10,000 m^3/ha-yr

[b] EPA maximum contaminant level, legal standards for public water systems (US EPA 2008)

[c] For citrus, the maximum recommended concentration is 0.075 mg/L

[d] EPA secondary maximum contaminant levels, voluntary standards for nonhealth-threatening elements

[e] No EPA standard

SUMMARY

To avoid the long-term accumulation of toxic amounts of waterborne substances in the rootzone of irrigated lands, the input of those substances to the soil from irrigation and other sources must not exceed the sum of losses from the soil and conversions to unavailable forms. Losses from the soil include plant uptake (5% to 10%) and leaching. In addition, there is an ongoing and reversible conversion of soluble, labile, and insoluble forms of minerals, which is affected by variables such as the oxygen content and pH of the soil water. Assessing the effects of irrigation water salinity and trace element concentrations on the suitability of a water supply for a given crop thus depends on the soil, crops, amount of water available, reference crop ET of the site (ET_0), irrigation system, irrigator's expertise in achieving the needed leaching, and decrease in yield that can be tolerated. It is recommended that the effects of irrigation water be tested directly on the soil of interest with column leaching studies, tests of aggregate stability, or tests of flocculation after the soil has been dispersed in a test tube.

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NOTATION

A = salinity concentration at which growth depression (threshold) starts, or maximum concentration of boron that does not reduce yield
B = percent of yield decrease per unit EC_e above the threshold level
B_{iw} = concentration of boron in irrigation water
B_{ss} = boron in soil solution
EC = electrical conductivity
EC_{aw} = average EC_{SS} at field capacity
EC_e = electrical conductivity of soil saturation extract
EC_{iw} = electrical conductivity of irrigation water
EC_{ss} = electrical conductivity of soil solution
ESP = exchangeable sodium percentage
F_c = concentration factor
LF = leaching fraction
m = decrease in yield per unit increase in boron concentration
OP = osmotic potential
x = boron concentration in the nutrient, sand culture, or soil solution
Y = relative yield

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