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Impact of Agricultural Practices on Groundwater Salinity

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ABSTRACT

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The impact of agricultural practices on water quality has been examined predominantly with an emphasis on surface water. Impacts on groundwater, as compared with surface waters, are much more difficult to quantify. This is due to larger travel times to and in groundwaters as compared with surface waters and difficulty in sampling groundwaters properly. Despite these difficulties in quantification, the impacts on ground- and surface waters are equally important. In non-irrigated areas agriculture often leads to increased recharge, sometimes resulting in the leaching of salts from the unsaturated zone into groundwater. In irrigated areas groundwater salinization can result from irrigation with saline water, salt water intrusion owing to pumping of groundwater, downward movement of salts in the unsaturated zone or dissolution of saline minerals, and from the unavoidable concentration of salts owing to plant water uptake.

The interrelationship of surface and groundwaters must involve water quality as well as quantity. Optimization of water resources entails consideration of conjunctive use, which in turn requires consideration of water quality in all parts of the system. In this paper examples are given showing how improvements made to reduce river salinity can cause groundwater salinization, which may not represent the optimum management strategy.

INTRODUCTION

Extensive research has been conducted on the management of agricultural lands in order to minimize the impact of salinity on crop productivity. Research and management programs have often been instituted to study the effect of agriculture on irrigation return flows and the subsequent effects on surface water salinity. In contrast, very little research has been focused on the impact of agriculture on groundwater salinity. This is a serious deficiency in that ground- and surface waters are inevitably linked and need to be considered together in the overall management of our resources. This concept, in the form of conjunctive use, has long been recognized for quantitative water utilization, but less so for water quality.

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In unirrigated areas, agricultural practices do not usually add salts to the environment, but rather cause a redistribution of salts. Shallow-rooted crops such as wheat consume less water than native grasses or trees, and result in increased subsurface recharge. Recharge is especially increased when these crops are grown in a fallow–crop rotation. This practice reduces evapotranspiration and thus should lower salinity concentrations. Unfortunately, in arid lands recharge can result in lateral movement of subsurface salts and subsequent formation of saline seeps.

An equally important problem is the salinization of subsurface waters by the displacement of salts from the unsaturated zone below the root zone into groundwater. These salts often accumulate over extensive time periods with minimal leaching. Groundwater contamination occurs because these waters are often not in equilibrium with soil water in the overlying unsaturated zone. These groundwaters are often recharged in upland areas of higher rainfall or may represent "fossil water" from earlier and wetter times.

NON-IRRIGATED AGRICULTURE

Ferguson and Bateridge (1982) examined the salinity from soil cores taken beneath cultivated (unirrigated) and native grassland from glacial till soils of north-central Montana. After about 50 years of crop–fallow farming, electrical conductivities (EC) of the saturation extracts from the upper 1 m were reduced to below 1.0 dS m^{-1} for almost all samples. The samples from the native vegetation sites typically had EC values of 1–15 dS m^{-1}. Ferguson and Bateridge (1982) estimated that 90 t ha^{-1} had been moved toward the groundwater in this region as a result of crop–fallow farming practices.

Transport of salts to groundwater is also a problem in Australia. The salt load stored in the unsaturated zone of western Australia is in the order of 1.7×10^5 to 9.5×10^5 kg ha^{-1} (Dimmock et al., 1974). The effects of increased recharge on groundwater salinity can be determined by examining the soil profiles of uncleared land. Soil profiles in south-western Australia often have high salt concentrations in the unsaturated zone above less saline groundwater (Watson, 1982). Farming practices have been attributed as the cause of serious groundwater salinization (Bahls and Miller, 1973; Ferguson and Bateridge, 1982). Also, Peck and Hurle (1973) measured groundwater Cl concentrations of 70–400 mg l^{-1} beneath forested catchments, and 160–3000 mg l^{-1} beneath catchments with substantial areas of cleared farmland. Although the major emphasis of these studies was to relate farming to increased river salinity, farming also contributed to groundwater salinization.

The replacement of native vegetation by dryland farming frequently results in movement of salts to the groundwater, rising water tables and the formation of saline seep areas in various parts of the world. For example, the south-western region of Australia (Peck, 1978), as well as the Northern Great Plains in the U.S. have been particularly affected by saline seeps. Groundwaters are already high in salinity in many of these regions and the agricultural impacts are usually a rise in water table, further salinization and lateral movement of the saline groundwater into nearby soils and surface waters.

The major emphasis of recent research studies and salinity control measures has been on minimizing or reversing the increase in surface water salinity caused by agriculture. The likely reasons for this emphasis include the fact that surface water quality is easier to characterize than groundwater, and that the impact of surface water deterioration is felt by users outside the immediate area that has generated the problem. Groundwater deterioration is usually a slower process but one which deserves equal attention.

IRRIGATED AGRICULTURE

The impact of irrigated agriculture on groundwater salinization can be divided into three processes: (1) concentration of salts as a result of plant water uptake: (2) movement of salts already in the unsaturated zone down into groundwater as a result of leaching or subsurface mixing of saline water with better quality groundwaters: (3) intrusion of saline water into high quality groundwaters as a result of groundwater pumping for irrigation. The first process is unavoidable in that water leaving the root zone is always of higher salinity than rain or irrigation water received at the soil surface. Plants preferentially take up water and leave most of the salts behind in the remaining water. This salinity increase, in the absence of mineral dissolution or precipitation, is equal to the ratio of applied to drained water. Additional salts may also be displaced or dissolved by the irrigation water.

Previous studies on irrigation with saline water have rarely considered the effects on groundwater quality. While interest in use of saline waste waters for irrigation is increasing (Hanks et al., 1984), its effects on groundwater salinity need to be examined. Hanks et al. (1984) investigated the use of 5 dS m^{-1} water for irrigation on fields above groundwater with an EC of 5–25 dS m^{-1}. No adverse effect was observed at this and another site where the groundwater was of higher quality; however, observable effects are not expected for quite a few years. This delay is due to the long travel times of salt to the groundwater as it moves through the unsaturated zone. Even if groundwater is of higher salinity than the recharging water, long-term adverse effects may still occur. For example, recharge of a saline aquifer can result in subsequent migration of that water into either surface water or to connecting, less saline groundwaters. Agricultural use of saline waters often meets legal regulatory requirements for disposal, since it constitutes what is classified as a beneficial use of that water. However, disposal by agricultural use may in fact have the same (or worse) result on water quality as direct discharge.

Irrigation in arid regions often leads initially to a deterioration of the quality of the shallow groundwater. Accumulated salts stored in the unsaturated zone move to the groundwater after irrigation. Doneen (1967) observed that subsurface salinity was greater in deep soil profiles of irrigated fields in the San Joaquin Valley than in the areas covered by native vegetation. Extensive areas had large quantities of gypsum-bearing sediments (Doneen, 1967) which result in long-term additions of salt to the drainage water. As shown by Jury et al. (1978), travel times of salt fronts are dependent on leaching fraction (fraction of total water applied that moves below the root zone). For example with a 0.05 leaching fraction and an evapotranspiration rate of 0.5 cm day^{-1}, it could take up to 5 years to reach steady-state salinity at a depth of 1.5 m below the surface. Clearly the long-term impact of current agricultural practices on groundwater salinity cannot be directly measured. It follows that the groundwater contamination potential of proposed practices cannot be observed with short-term field experiments. Nonetheless, preliminary effects of irrigation on groundwater quality can be readily assessed in many areas. Maricopa County in Arizona has experienced 30–80 years of groundwater pumpage for irrigation. Subsequent percolation of drainage waters has resulted in a 3–5-fold increase in salinity in perched zones above the falling water table in several regions of that county. This salinity increase is consistent with the estimate based on irrigation efficiency in these regions (Schmidt, 1984). The perched water is transferred to the major aquifers via wells which are perforated above the water table.

Another example pertains to the unconfined aquifer of the San Luis Valley of the Upper Rio Grande, which contains an estimated 2×10^{12} m^3 of groundwater (Emery et al., 1971). The aquifer is presently recharged by irrigation drainage water and conveyance system losses. Although the southern portion of the valley drains to New Mexico via the Rio Grande, its northern half drains internally. Salinity levels have built up to 14 000 mg l^{-1} in the shallow groundwater beneath the northern half of the valley (Clark, 1972) as a result of evapotranspiration. Increased salinity in the Rio Grande, and its tributary the Pecos River, is caused by irrigation. As a result groundwater associated with the river alluvium has also become more saline. This groundwater is also extensively utilized for irrigation, and increased salinity has already restricted the crops that can be grown without yield reduction.

In the Wadi Dhuleil Catchment in Jordan, extensive irrigation using local groundwater over a period of 25 years has resulted in decreasing water levels and rapid salinization of the groundwater. The average salinity in the wells of the central area has increased from 350 to 2750 mg l^{-1} between 1971 and 1982 (Bichara, 1986). Increasing salinity has already greatly reduced the productivity of the area and Bichara (1986) suggests that the irrigated area must be reduced by half in order to maintain long-term agriculture.

PREDICTIONS

Projections of salinity based on solute concentrations and travel times in the unsaturated zone are generally not feasible because of the high spatial variability of salinity in the unsaturated zone. Salt loadings to the groundwater are often based on estimates of salt inputs at the soil surface and computer simulation models. Simulations of subsurface water quality as a result of irrigation in the San Joaquin Valley predict deterioration below the root zone as a result of gypsum dissolution (Tanji et al., 1967). Steady-state models have been made and tested in lysimeters to predict salt concentrations and fluxes below the rootzone (Rhoades et al., 1974; Oster and Rhoades, 1975). These simulations and lysimeter studies indicate that reductions in leaching fraction lead to higher soil salinity but reduced mass emissions below the root zone.

Groundwater quality may be either improved or degraded as a result of increasing irrigation efficiency. Prediction of the impact on groundwater quality depends on a number of site-specific conditions, such as source of water, depth to the water table, and the type and quality of the irrigation and groundwater. In the absence of additional recharge sources (other than drainage) reduced leaching increases groundwater salinity, as the salinity of the groundwater will eventually approach the salinity of drainage water. Irrigation can add anywhere from 0.3 to 32 tons salt h^{-1} $year^{-1}$ (Rhoades and Suarez, 1977). Salt loading to the groundwater can of course be higher since these numbers do not take into account salt dissolution or displacement of saline water in the vadose zone.

If all salinity stems from the irrigation water, the impact on groundwater salinity will largely be determined by the leaching fraction and, to a lesser degree, by the precipitation of calcite and gypsum. Rhoades and Suarez (1977) classified waters into several types, including those that are calcite saturated and those that precipitate gypsum. Several different situations can occur. If groundwater is exclusively used for irrigation in an arid region, the salt pulse of the drainage may never reach the declining water table. In this instance, water extraction is comparable to mining practices, and the optimum management strategy is to minimize the volume of water lost to leaching.

In another example, moderately saline water is brought into a basin and used for irrigation in a region with less saline groundwater. If leaching fractions are decreased and less saline water is imported one might expect less groundwater degradation. The initial response of the system does indicate less groundwater degradation with low leaching, but this effect is reversed with time (Fig. 1; Rhoades and Suarez, 1977). The answer is of course site specific, in that it depends on the salinity and composition of the irrigation and groundwater as well as the leaching fractions selected.

Suarez and van Genuchten (1981) simulated the effect of irrigation on salinity in several non-steady-state groundwater systems. They assumed that ir-

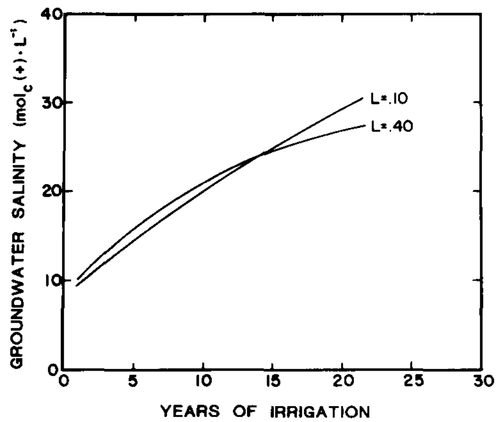

Fig. 1. Changes in groundwater salinity with time for a basin irrigated with imported surface water at two leaching fractions (L) (after Rhoades and Suarez, 1977).

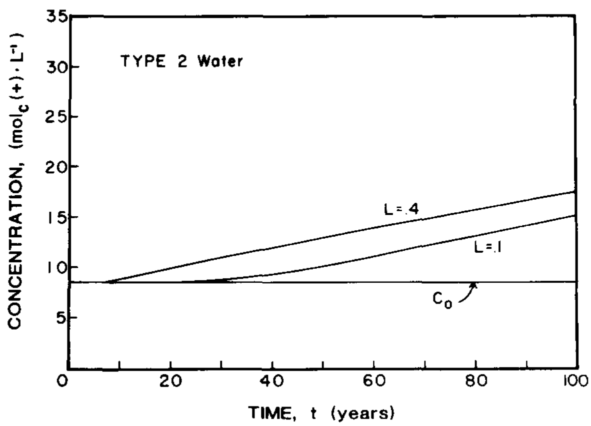

Fig. 2. Average groundwater concentration with time for a closed basin for two leaching fractions (L). Irrigation is a combined imported water and groundwater system with $CaHCO_3$-type water (after Suarez and van Genuchten, 1981).

rigation consisted of local groundwater plus sufficient imported water to offset evapotranspiration and hence to maintain a stable water table. The simulation considered a water table at 20 m and an impermeable layer at 50 m, no mass flow out of the closed basin, and a net evapotranspiration minus rainfall value of 1.0 m year^{-1}. Groundwater was pumped from wells that drew water uniformly from the saturated zone. For calcite-saturated waters, groundwaters degraded slowly with relatively small differences between high and low leaching management regimes (Fig. 2, after Suarez and van Genuchten, 1981). The salts travelled more rapidly through the unsaturated zone under high leaching; thus increasing groundwater salinity occurred earlier than in the case for low

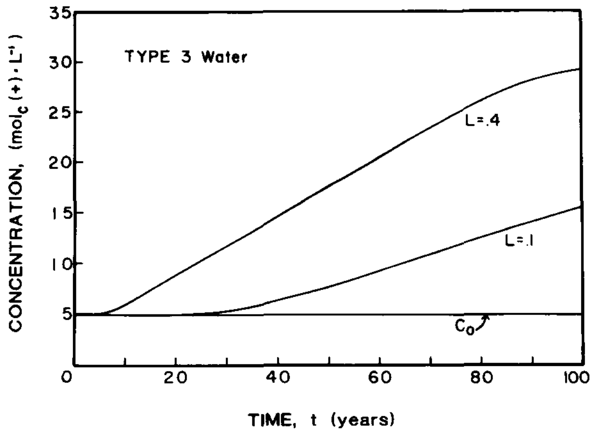

Fig. 3. Average groundwater concentrations with time for a closed basin for two leaching fractions (L). Irrigation is a combined imported water and groundwater system with $CaSO_4$-type water (after Suarez and van Genuchten, 1981).

leaching. Low leaching thus serves to increase the storage of salts in the unsaturated zone. The differences in groundwater salinity as a function of leaching fraction are considerable for gypsum-precipitating waters, as shown in Fig. 3 (after Suarez and van Genuchten, 1981). Low leaching in this case results in the precipitation of large quantities of gypsum in the unsaturated zone. The earlier rise in salinity with the higher leaching regime is again attributable to lower residence times of the drainage water under high leaching. These examples illustrate the hazards of relying on direct but short-term field measurements to evaluate the long-term impacts of irrigation on groundwater salinity.

MANAGEMENT OPTIONS

In a study of the San Luis Rey River basin in southern California, Labadie and Khan (1979) considered modifying the irrigation sources within the basin in order to optimize groundwater quality. The basin consists of a series of interconnected sub-basins. The proposed management is to irrigate each sub-basin with water from the adjacent higher sub-basin rather than the present system of utilizing local groundwater and allowing for natural flows between the sub-basins. The strategy was shown to decrease the rate of salinization in the sub-basins. Their modeling study represents the type of analysis that is needed to examine our options for improving or maintaining groundwater quality. It should be noted that the salinity improvements occurred with an equivalent increase in salt discharge to the lowest sub-basin and downstream waters. The San Luis Rey River system discharges to the ocean; thus there are no presumed drawbacks to increased salt discharge. This optimization is applicable to groundwater basins that discharge either to the ocean or into a terminal saline sink, but may not be applicable to basins which discharge into usable water supplies.

Problems associated with overpumping of groundwater are often related to agricultural water use. For example, overpumping for irrigation purposes in the Koo-wee-rup basin in Victoria, Australia threatens to degrade the groundwater quality as a result of seawater intrusion (Longley et al., 1978). Extensive groundwater pumping, primarily for irrigation, also resulted in seawater intrusion in the coastal aquifer in the Pajaro valley in central California (Muir, 1974). Bond and Bredehoeft (1987), using a two-dimensional transport model, projected that with present levels of water use in the latter example, seawater intrusion will result in a threefold increase in areas with greater than 500 mg l^{-1} Cl by the year 2000, relative to 1981 levels. The only apparent choices are either to reduce irrigation pumpage or to recharge sufficient quantities of water to maintain water levels.

The use of amendments either for soil reclamation or for maintaining adequate infiltration rates when irrigating with sodic waters inevitably results in the addition of large amounts of salt. For economic reasons, it is desirable to reduce the time between the initiation of reclamation and the production of a crop. Irrigation with saline waters (with high sodicity but moderate sodium adsorption ratio (SAR[1]) has been proposed as a potentially inexpensive and rapid method of reclaiming soils high in sodium (Reeve and Bower, 1960; Reeve and Doering, 1966). This method, however, results in a much larger discharge of salt than conventional methods and is not suitable in areas that have usable groundwater supplies or in areas that recharge groundwaters of usable quality.

The impact of reclamation on groundwater salinity is not important if the shallow groundwaters are already saline. In many regions, however, the local groundwater is utilized for domestic or agricultural purposes. Nadler and Magaritz (1986) examined the effect of gypsum amendments, applied with sodic water irrigation. After 11 years of amendment application of up to 8×10^3 kg ha^{-1} $year^{-1}$, the sulfate front had extended beyond the 4.0-m sampling depth. Sulfate concentrations in the irrigated fields were more than 50-fold greater than concentrations in adjacent uncultivated fields. The combination of $NaHCO_3$ water and gypsum dissolution resulted in calcite precipitation and discharge of $NaSO_4$ water. Nadler and Magaritz (1986) considered that the continued use of this water results in deterioration of the groundwater and that the excessive use of gypsum serves only to displace additional Na from the clay exchange sites in the unsaturated zone. The amount of gypsum applied in such areas should be limited to the minimum necessary to provide adequate infiltration. Excessive use of gypsum not only increases the salinity of the ground-

[1]Where $SAR = \frac{Na}{(Ca+Mg)^{0.5}}$ with concentration expressed in mmol l^{-1}.

water but also increases the SO_4 content. As the SO_4 content of the irrigation water increases, the reduced solubility of gypsum makes its use increasingly ineffective. Reduced gypsum dissolution will prevent the reduction in exchangeable sodium in the surface horizons where the major barriers to infiltration usually exist.

Improved irrigation efficiency has been advocated as a way of reducing salt loads in drainage return flows (van Schilfgaarde et al., 1974). Reduced leaching in citrus groves in the Wellton Mohawk Irrigation and Drainage District was initially predicted to reduce the salt load from the valley and into the Colorado River by 130 000 tons annually (Hoffman et al., 1978). However, using a similar calculation technique as that described in Rhoades and Suarez (1977), one can show that there are negligible long-term benefits to river quality from improved irrigation efficiency. Long-term benefits from improving irrigation efficiency clearly result if the drainage waters are not returned to the Colorado River. Unfortunately, institutional constraints often prevent implementation of improved management concepts. Furthermore, the above analyses on the benefits of improved irrigation efficiency do not take into account the degradation of groundwater that accompanies improved leaching. Recommendations regarding irrigation management might be different if we consider the potential benefits of conjunctive use of surface and groundwater.

Water quality problems in the lower Colorado River occurred as a result of reduced flows due to the filling of Lake Powell and the initiation of extensive pumping of saline drainage water from the Wellton Mohawk Irrigation District into the lower part of the river. The optimum strategy for the lower Colorado River basins may not be improvement of irrigation efficiency as is currently proposed, but rather a strategy that keeps the groundwater as non-saline as possible. During years of short water supply (and thus higher than normal salinity in the lower Colorado River) the Wellton Mohawk District could be irrigated with groundwater. Approximately 25 900 ha were irrigated in 1973 and the district has an entitlement to use 3.70×10^8 m^3 of Colorado River water annually or a water depth of about 1.4 m (Advisory Committee on Irrigation Efficiency, 1974). A rough estimate of the groundwater storage can be obtained by examining the reduction in salinity of the pumped groundwater with time. The drainage water composition decreased from 6000 mg l^{-1} TDS in 1962 to 3700 mg l^{-1} in 1973 (Advisory Committee on Irrigation Efficiency, 1974). Assuming a recharge of about 2.47×10^8 m^3 $year^{-1}$ owing to deep percolation, a 1-year lag time for the water to reach the water table, and subsequent pumping of equal volumes from all depths, at least 1.23×10^9 m^3 of water is in storage. With a consumption of 1.2 m $year^{-1}$ (3.1×10^8 m^3) and a 0.20 leaching fraction, sufficient groundwater will be available for at least 3 years of irrigation. McDonald and Loeltz (1976) calculated that increased recharge from irrigation caused an increase in groundwater storage of 1.6×10^9 m^3 below Yuma Mesa alone (adjacent to Wellton Mohawk). If the added groundwater is used only once for irrigation and not recycled, it represents sufficient water for at least 7 years of irrigation. During years of high water flow in the river, additional water could be diverted into the basin to replenish the groundwater, and saline groundwater could be discharged to the Gulf of Mexico via the bypass drain. This alternative will not increase the salinity of water delivered to Mexico.

The Palo Verde Irrigation District in California is another example of a river–groundwater system that should be considered as an integrated unit. Approximately 35 000 ha are irrigated with Colorado River water in an alluvial basin of the river. Water (1.13×10^9 m^3) is diverted upstream by a gravity system, and the return flows are estimated at 5.55×10^8 m^3. The relatively low irrigation efficiency of about 50% does not appear to result in any additional costs since drainage consists of large channels that intercept the water table and drain by gravity back into the river. The composition and salinity of the groundwater (D.L. Suarez, unpublished data, 1977) is only slightly higher than expected from 50% irrigation efficiency, thus the valley is close to a steady-state situation.

Proposed changes to increase irrigation efficiency, such as canal and lateral lining, on-farm improvements and water management programs, have been suggested under the salinity control program authorized by the U.S. Congress. The short-term effect would be to decrease the volume of drainage water and presumably to divert less irrigation water. The short-term reductions in salinity of the lower Colorado River will be made at the expense of an increase in salinity of the groundwater in the basin. The overall salinity effect on the river will be negligible once the system reaches a new steady state (i.e. the basin stops accumulating salts). Based on the projected drainage of 6.16×10^7 m^3, and 90% irrigation efficiency, steady state at the depth of the drains would be reached after about 10–15 years. The time to steady state would be longer if drainage is to the existing open drains, because the travel times of water between the drains is greater than that of the water closer to the drains.

Groundwater quality will rapidly decrease in Palo Verde Valley if the irrigation efficiency is increased from 50 to 90%. At steady state the groundwater will increase from twice as saline as Colorado River water to almost 10 times as saline. At the projected salinity levels of 8000 mg l^{-1}, the drainage water is virtually unusable as an irrigation water source. If the groundwater is less saline this water may be used for irrigation during water-short years, thus effectively increasing the storage capacity of the river system. The U.S. Bureau of Reclamation (unpublished data, 1978) estimated that there are approximately 3.0×10^9 m^3 of groundwater in a high-porosity aquifer in the Palo Verde subarea alone (5000 ha). This volume of water is equivalent to 4% of the total amount of water stored in the Colorado River system. Undoubtedly the total groundwater in the lower Colorado River basins constitutes a sizeable storage volume. Assuming that recoverable water in the alluvium represents only 30%

of the volume (Hely, 1969) and that consumptive use is 1.2 m year^{-1} then the basins could be irrigated for 3 years with a 12-m drop in water levels. An additional benefit of this management strategy is that the salinity of the Colorado River water delivered to Imperial, Coachella and Mexicali Valleys would be improved during water-short years when salinity is highest. This improvement would result because there would be no diversion of water into Palo Verde Valley or release of saline drainage water to the river when groundwater is used. During periods of higher water flow groundwater could be pumped out of the Palo Verde basin. Since river salinity during these periods is low, no salt damage would occur due to discharge of saline groundwaters. Shortly after accelerated drainage practices, recharge (in addition to that occurring from irrigation with the river water) could be attempted by blocking the drain exits and filling the drains with Colorado River water. This would reduce the groundwater salinity back to suitable levels. This strategy should be considered, although institutional or economic constraints may render it unfeasible.

The Parker Valley, just upstream from Palo Verde, is another example of an alluvial valley in the lower Colorado River whose groundwater quality could be managed jointly with that of the surface water. Alluvial valleys along the river typically contain 50 m of alluvium. If the available water is 30% by volume, 15 m of water could be extracted. In some of these areas salinities may be too high at present to enable them to be used, but the ultimate salinity will be determined by irrigation management and Colorado River salinity. Thus groundwater should be considered conjunctively with surface water in any salinity control program.

CONCLUSIONS

The above examples demonstrate the many ways in which groundwater salinity can be impacted by agriculture. Agricultural water users should not be in conflict with those concerned with groundwater salinity, as their long-term viability depends on these water resources. Salinity control strategies must go beyond maximizing surface water quality and consider conjunctive use and quality of river–groundwater systems.

In many arid land irrigated areas long-term viability of irrigation is not possible with present management practices. Limitations include not only the evident problem of unsustainable water supplies at the present rate of water use but also salinization of the water supply due to return flows of drainage water. In many areas no provisions exist for natural drainage from agricultural basins. Without outlets to oceans or a potential for drainage to inland seas, groundwater salinization will threaten irrigation projects which have sustainable groundwater supplies. Collection of the saline drainage waters and removal from the basin may be the only way to assure the long-term future of these projects.

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