Effects of long-term irrigation with reclaimed water on soils of the Northern Adelaide Plains, South Australia.

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On a small scale, reclaimed water (RCW) use has been practised on the Northern Adelaide Plains (NAP) horticultural districts for more than 28 years. The RCW has had approximately 1.7 times the salinity and twice the sodium absorption ratio (SAR) of bore water commonly used for irrigation in the district. Recently, a large-scale reclamation scheme has been commissioned which could eventually supply approximately 30 GL of RCW to over 250 growers on the NAP. This study compared historical water quality and time of use data with physico-chemical properties of soil cores taken from sites where reclaimed (RCW-irrigated) or bore water had been used for irrigation, or sites that had not been irrigated (virgin). The aim was to determine if current farming practices irrigating with RCW could, now or in the future, lead to a decrease in yields through detrimental increases in soil salinity, sodicity, and boron (B) concentrations, and to determine if these changes were significantly different from bore-irrigated or virgin sites. Data suggested that changes in soil salinity and B concentration from RCW use would not decrease yields. However, changes in soil SAR had the potential to restrict drainage and consequently increase salinity: although a more functional critical SAR value for the NAP soils needs to be defined to assess this potential. These findings suggest that farming methods, in the 1967-95 period, did not address the physico-chemical changes associated with the use of more sodic RCW. Considering the future scale of RCW use, the SAR of the irrigation water may need to be decreased and/or appropriate farming methods developed and practised with the use of RCW to protect these soils for future horticultural activities.

A low cost soil test, using a simple 1:5 soil:water extract was compared with accepted soil extracts (for assessing detrimental physico-chemical soil changes) and is proposed as a grower management tool to assist in monitoring the physico-chemical changes of the NAP soils.

Additional keywords: SAR, recycled water, electrical conductivity, boron.

Introduction

Current and future generations worldwide need to be more efficient with their use of limited water resources. The reclamation and agricultural reuse of wastewater is one means of achieving this (Oster 1994; Bahri 1999). A small number of agriculturalists of the Northern Adelaide Plains (NAP) have been, within South Australian Department of Human Services restrictions, irrigating with water reclaimed from Adelaide's major sewage treatments work for up to 28 years. These agriculturalists invested in their own infrastructure to pump secondary treated water (Class D; Anon. 1999) to their properties from the Bolivar (Sooth Australia) Sewage Treatment Plant's outfall channel. Recently, a grower and South Australian Government initiative has seen the implementation of a further water treatment step (dissolved air flotation/filtration (DAFF); Bosher et al. 1998) and a reclaimed water (RCW) pipeline has been laid servicing much of the NAP (Huijbregsen et al. 1999). The DAFF-treated water is of much higher quality pathogenically (Class A), and in some cases chemically, than water previously used (Bosher et al. 1998; Anon. 1999). However, the salinity of DAFF-treated RCW can range up to 1500 mg/L of total dissolved salts (TDS; currently averaging approx. 1200) compared with <900 mg/L commonly found in bore water. DAFF-treated RCW also has a higher sodium adsorption ratio (SAR), 8.5 compared with 5.5, commonly found in bore water (tertiary aquifers 1 and 2) currently used (Stevens et al. 2000a, 2000b; D. P. Stevens, unpublished data). Boron concentration in DAFF treated RCW is approximately double than that of the bore water, 0.4 compared with 0.15 mg/L. The chemical water quality of the new Class A RCW were marginally better than Class D RCW used over the last 30 years (Miles 1952; Hodgson 1966; Matheson and Lobban 1974; Anon. 1976).

The poorer quality of the RCW, compared with bore water, could lead to soil degradation through sodicity, salinity, or B toxicity if farm management practices are inappropriate. For example, it is well known that the key to irrigating with poorer quality (more saline) water is maintaining an appropriate leaching fraction to ensure the net movement of salts through the root-zone (Oster 1994; ANZECC and ARMCANZ 2000). Leaching requires maintenance of soil permeability, which is influenced by soil sodicity (measured by SAR or exchangeable sodium percentage) and water salinity (measured by electrical conductivity or TDS) (Rengasamy et al. i984; Sumner et al. 1998; ANZECC and ARMCANZ 2000)

Growers require simple, cost-effective tools to help monitor/manage their soils, as historically many specialised soil tests have required tedious and costly laboratory soil preparation and analysis. An example is the standard soil saturation extract, a tedious, time-consuming procedure often flawed by operator errors (Shaw 1994). However, much of the scientific literature available refers to soil saturation extract data when relating soil salinity to yield reduction in plants (Maas 1986; ANZECC and ARMCANZ 2000)

Our first objective was to assess changes in soil chemistry and the related physical changes from long-term use of RCW (Class D) compared with soils irrigated with bore water or not irrigated (Virgin). This permitted an evaluation of current irrigation and farm management practices with Class D RCW to determine if they are adequate to manage the associated changes in soil physico-chemical properties. This study will help assess whether irrigation using the higher quality Class A RCW on the NAP soils is sustainable in the long-term. Our second objective of the study was to develop and verify a simple cost-effective method for assessing soil degradation (a) to overcome the inherent variability in soil and soil management history by allowing a large number of samples and sites to be analysed, and (b) for future use by growers.

Materials and methods

Site

The NAP are approximately 30 km north of central Adelaide, South Australia. A diverse range of vegetables (grown in glass or shade houses or broad acre), pastures, fruits, and ornamental crops are grown and there are 5 major soils types used for growing these crops (Table 1).

Selection of soil sampling locations

Soil sampling locations were determined using 3 criteria:

(1) Historically, soils were virgin (not used agriculturally for at least the last 20 years), bore-irrigated (i.e. irrigated with bore water for agricultural purposes for at least 10 years), or RCW-irrigated (i.e. irrigated with reclaimed water for agricultural purposes for 10-28 years).

(2) Sampling sites should provide a cross-section of the 5 main soil types of the NAP (Table 1) and within each soil type contain soils that meet criterion 1.

(3) Within each soil type (criterion 2) and irrigation history (criterion 1) agricultural practices were similar.

Not all criteria could always be met due to the large number of possible criteria combinations and the diverse range of horticultural crops and practices on the NAE In total, soil cores were taken from 46 sites from across the NAP.

Soil coring

All soil cores were taken using a 50-mm-diameter steel push tube to 1.0 m deep. In some cases rocks or dense dry clay were encountered preventing sampling to 1.0 m. To assess seasonal variation in soil chemical properties, samples were cored after winter and after summer. For the first sampling (after summer, 1998), paired soil cores were taken 1 m apart at 10 cm depth intervals and bulked for analysis (n = 11 for bore, 19 for RCW, and 16 for virgin). Occasionally, duplicate paired cores were taken 15 in from the first pair, to assess site variability. To reduce sampling and analysis costs and allow some assessment of site variability, for the second sampling (after winter, 1999), 2 soil cores were taken 5 m apart, and sections of 3 soil horizons sampled (A, B, and C), at each location (n = 11 for bore, 14 for RCW, and 2 for virgin). Depth of the A, B, and C horizons varied with respect to soil type and in some soils the true C horizon was not attained, but the 90-100 cm depth sampled. Cores at each location were not bulked, but analysed separately.

Chemical analysis, characterisation, and classification of soil types

Matheson developed an intensive 'agricultural-use' soil map of the area in 1975. To relate soil classification to the agricultural-use definitions of Matheson (1975; Table 1), 7 soil pits were dug, where core sampling indicated soils typical of those described by Matheson (1975). Soil profiles within each soil type were described according to the Australian Soil and Land Survey Handbook (McDonald et al. 1990) and classified according to the Australian Soil Classification (Isbell 1996).

The major soil horizons were sampled and chemically analysed for pH and electrical conductivity (EC) (1:5 soil:water; Rayment and Higginson 1992), exchangeable cations by leaching with 1 M N [H.sub.4]Cl (Rayment and Higginson 1992), and particle-size distribution (percent by weight of sand, silt, clay; Allen 1981).

Measurement of soil core chemical properties

Soil core sections were dried at 40[degrees]C and ground to pass through a 2-mm sieve. Soil extracts were analysed for pH and EC in a 1:5 soil:water extract (E[C.sub.1:5]; Rayment and Higginson 1992). A subsample, diluted with Ba[(N[O.sub.3]).sub.2] to a final concentrations of

0.01 M was used for Cl analysis (automated ferricyanide method; Rayment and Higginson 1992). A 15-mL subsample of the 1:5 soil:water extract was then centrifuged (10000 rpm for 10 min), passed through a 0.45-[micro]m cellulose nitrate filter, acidified with one drop of 10 M HCl, and stored at 4[degrees]C prior to analysis by inductively coupled plasma-atomic emission spectrophotometry (ICP-AES) for Ca, Mg, Na, and B. Concentrations of Ca, Mg, and Na in the 1:5 soil:water extracts determined by ICP-AES were used to calculate the SAR (SA[R.sub.1:5]; Rengasamy et al. 1984).

Comparison of methods for soil chemical and physical analyses

Due to the large number of analyses required, it was impractical to complete commonly used chemical analysis (e.g. saturation extracts) on all samples to allow comparison with literature values. However, the more commonly used soil chemical analytical procedures were completed on several cores (>10, depending on analysis) selected to represent all soil types and soil horizons of the NAP. This allowed correlation of data to literature values.

The more commonly used methods which were correlated with our 1:5 soil:water method on selected soil cores were:

(1) The original SA[R.sub.1:5] method of Rengasamy et al. (1984) using atomic adsorption spectrophotometry (AAS). We used a universal flaming solution (UFS; derived from Varian 1989) to suppress elemental ionisation during the determinaton of Ca, Mg, and Na, although Rengasamy et al. (1984) did not record if they used UFS. [SAR.sub.1:5] values determined by AAS was compared with those determined by ICE This comparison was made as the higher ionisation temperatures of the ICP compared with AAS could potentially measure some colloidal (<0.45 [mu] m) cations (Spiers et al. 1983). All SA[R.sub.1:5] values quoted in this paper were determined using ICP unless stated otherwise.

(2) Hot-water soluble extractable B ([B.sub.hws]; Cartwright et al. 1983) was compared with room temperature 1:5 soil : water B ([B.sub.1:5]), and saturation paste extract B ([B.sub.se]; Rayment and Higginson 1992).

(3) Soil exchangeable sodium percentage (ESP; Rayment and Higginson 1992) was compared with SA[R.sub.1:5].

(4) Saturation paste electrical conductivity (E[C.sub.se]:Rayment and Higginson 1992) was compared with E[C.sub.1:5].

(5) Soil sodicity rankings were determined, using a simple field test sodicity meter (Rengasamy and Bourne 1998). These rankings (non-sodic. sodic, and highly sodic) were then related to E [C.sub.1:5] and SA[R.sub.1:5].

Determination of water chemistry

To compare water quality parameters for reclaimed and bore water used at sites where soil cores were taken, water chemical data were sourced from several publications and databases (Table 2), as well as analysis of current site irrigation water quality (APHA 1998).

Statistical analysis

GENSTAT (Anon. 1998) was used for regression analysis. Because of the unbalanced nature of core sampling, residual maximum likelihood (REML) methods were used for estimation of variance components when comparing results for soil history, type and depth (Anon. 1998). The analyses were performed using REML in GENSTAT, where the cores were considered random and the depths and irrigation types were considered as fixed. The inclusion of the random core term ensured that the testing of the main effects of the irrigation water was against an appropriate error. The normality assumption was investigated using residual plots.

Results and discussion

Development and verification of low cost soil analysis methods

Soil SAR values were not significantly different between AAS and ICP methods (Fig. 1). These data suggest that the filtering and acidification steps in the sample preparation for ICP analysis removed colloidal particles by settling or precipitation and hence these were not sampled, aspirated, and analysed (Spiers et al. 1983).

[FIGURE 1 OMITTED]

Comparison of soil B measurements indicated a significant relationship between [B.sub.hws] and [B.sub.1:5] (Fig. 2). Using this relationship, [B.sub.hws] was converted to mannitol extractable B (Cartwright et al. 1983) and then to saturation extract B ([B.sub.se]; Rayment and Higginson 1992), and the relationship between [B.sub.1:5] and [B.sub.se] modelled (1):

(1) [B.sub.se] = 0.021 + 1.459[B.sub.1:5] + 0.472[B.sub.1:5.sup.2] + 0.033[B.sub.1:5.sup.3]

[FIGURE 2 OMITTED]

The model (1) was verified using samples taken from the 5 main soil types (Table 1) of the NAP at 3 sampling depths (0-10, 40-50, and 90-100 cm, n = 30). There were no significant differences (P > 0.05) between [B.sub.se] and that calculated from [B.sub.1:5] ([r.sup.2] = 0.92, n = 28). These data confirmed that for the soils of the NAP, [B.sub.1:5] can be converted to [B.sub.se] with a high degree of confidence using Eqn 1 above. This conversion allowed a low cost measure of plant available soil B for the NAP that was easily related to critical published values for crops (Maas 1986).

There was a significant correlation of SA[R.sub.1:5] with ESP (Fig. 3). This relationship was similar to that found by Rengasamy et al. (1984) on similar soils using similar methods. However, the data of Rengasamy et al. (1984) were linear ([r.sup.2] = 0.47) and data presented in this paper curvilinear ([r.sup.2] 0.83; Fig. 3). These differences were because SA[R.sub.1:5] data from the present study ranged from approximately 0,100 compared with 0.1-12.4 in the study of Rengasamy et al. (1984). In the SA[R.sub.1:5] range of 0,20 (ESP < 40) differences between the 2 studies were insignificant (Fig. 3). SA[R.sub.1:5], a more cost-effective analysis than ESP, can therelbre be confidently used as a surrogate measurement of ESP on these soils. It should be noted that even though there is commonly a close relationship between SAR and ESP (ANZECC and ARMCANZ 2000), they measure 2 distinct soil parameters. ESP measures the proportion of Na ions on the soil exchange phase (after soil solution cations are removed), and SAR measures the proportions of Na, Ca, and Mg in the soil solution, where the divalent cations would be preferentially adsorbed onto the exchange surfaces of the soil.

[FIGURE 3 OMITTED]

Regression analysis between ESP and SA[R.sub.1:5] indicated that coefficients of determination were not significantly improved if grouped by irrigation water history ([r.sup.2] = 0.86, standard error 0.09). This analysis indicated that there was little effect of soil irrigation history on the relationship between ESP and SAR, and therefore the effects of water quality on soil cation exchange properties were minimal.

The good correlation between E[C.sub.1:5] and E[C.sub.se] for the NAP soils (Fig. 4) suggests that for any soil type on the NAP a soil E[C.sub.1:5] multiplied by 8.2 gives a good estimate of E [C.sub.se], allowing direct comparison with critical values found in the literature (Maas 1986; Anon. 1992). This relationship allows comparison with salinity toxicity E[C.sub.se] data used in the literature using the simpler E[C.sub.1:5] measurement. There was no significant difference in the relationship when data (n = 30) were grouped and regressed in soil textural classes (n = 5). This analysis suggests that the use of different conversion factors for different soil textures, as suggested by Shaw (1988), is not required for the soils of the NAP.

[FIGURE 4 OMITTED]

When we arbitrarily divided data from the field test sodicity meter into 3 divisions (Fig. 5), 64, 67, and 55% of the non-sodic, sodic, and highly sodic rankings (respectively) related directly to E [C.sub.1:5] and SA[R.sub.1:5] data. Much of the variation between these arbitrary divisions was between sodic and highly sodic categories. By combining sodic and highly sodic rankings, the relationship between the field test sodicity ranking and E[C.sub.1:5] and SAR was 91% accurate. These data suggest that for the soils of the NAP, E[C.sub.1:5], and SA[R.sub.1:5] give a good first approximation of soil sodicity (i.e. sodic or non-sodic) and the potential of NAP soils to disperse. However, the field sodicity meter uses rainwater or deionised water in the dispersion test. Low EC water may be inappropriate for assessing dispersion below the soil surface. Under normal conditions these soils would be exposed to water that accumulated salts from the upper soil profile when the wetting front moved through the soil profile. Therefore, interpretation of the field sodicity meter's results at depth, in comparison with what may actually occur in the field, is limited.

[FIGURE 5 OMITTED]

These data verify that a 1:5 soil:water extract, which is currently a routine extraction method for determining soil pH, can offer a low cost method for determining several key soil physico-chemical parameters (as discussed above). These parameters can then be confidently converted to critical soil values reported in the literature, providing a tool for monitoring and managing soils when irrigating with RCW on the NAP.

Properties of the Northern Adelaide Plain soils and irrigation water

Many of the subsoils of the NAP are naturally sodic because of the shallow (1.5 m), saline water table on the western side of the NAP (Matheson and Lobban 1974). There are several main orders of soil types on the NAP: Sodosols, Dermosols, Vertosols, and Kandosols (Table 1). Others have also reported on the heterogeneity of the NAP soils (Matheson 1975).

Some typical water quality properties of bore and RCW used for irrigation at the sites where soil cores were taken are listed in Table 2. Mean TDS, EC, and SAR values of bore water were approximately half that of RCW over the study period (1978-98). Such differences in water quality should be reflected in changes in soil chemical properties given similar farming practices. For example, the ESP of the soil surface would be expected to equilibrate with the SAR of the irrigation water (Oster 1994).

Changes in soil chemical properties with the use of reclaimed water

The calculated means for the after-summer samples, where saline and sodic effects would be expected to be greatest, for irrigation history and soil depth changes (see Figs 6, 8, and 9) were typical of data from both sampling times. For simplicity, data discussed below refer to the after-summer sampling.

[FIGURES 6, 8-9 OMITTED]

Electrical conductivity

Electrical conductivity of irrigated soils (bore and RCW) was significantly different from virgin soils (Fig. 6). The ECs of bore and RCW irrigated soils were not significantly different and were approximately 2.8 dS/m (E[C.sub.se] or 0.34 dS/m E[C.sub.1:5]) in the top 20 cm of soil. On average, the EC of borewater-irrigated soils was lower than RCW-irrigated soils and did not change significantly until depth in both irrigated soils (Figs 4 and 6). In contrast, the EC of the virgin soils was low in the upper soil profile (E[C.sub.se] 1.1 dS/m; Fig. 4 and Fig. 6) and increased significantly (P [less than or equal to] 0.05) down the profile to an E[C.sub.se] of 6.9 dS/m.

Crop salinity responses are influenced by climate, irrigation and agronomic management (Rhoades and Loveday 1990). However, most vegetable crops (except zucchini) suffer a 10% yield reduction at an E[C.sub.se] of 2.7 [+ or -] 0.8 dS/m (ANZECC and ARMCANZ 2000). Topsoil salinities measured in these soils were near these thresholds and careful management of soil salinity is required.

It is generally accepted that a soil with E[C.sub.se] >4 dS/m is a saline soil (Richards 1954; Sumner et al. 1998). However, this arbitrary value of soil salinity should reflect the crops grown. Given the salinity threshold of most vegetable crops, we suggest that a critical E[C.sub.se] >3.0 dS/m would be appropriate in soils where vegetable crops are grown on the NAP. On average, at the end of the growing season (after-summer sampling; Fig. 6) the top 30 cm of the irrigated soils (reclaimed or bore water) and virgin soils did not exceed the critical E[C.sub.se] of 3 dS/m. These data suggest that current and past irrigation scheduling has included a leaching fraction adequate to maintain soil salinity at acceptable levels for most crops grown on the NAP. For example, a leaching fraction of 20-50% would be considered an ideal leaching fraction for RCW use (Rhoades and Loveday 1990). However, below 30 cm, RCW-irrigated soils are nearing the proposed salinity threshold (E [C.sub.se] 3 dS/m), which could have detrimental consequences for deep-rooted crops in the future. The salinity changes in virgin soils are a consequence of rainfall leaching the topsoil, but rainfall has been insufficient to leach the lower depths and prevent upward migration of shallow saline groundwater on the western side of the NAP (Matheson and Lobban 1974)

Soil pH

The pH of the NAP soils were generally greater than 7. Differences in soil pH due to irrigation with RCW and bore water were not significant ($P > 0.05$; Fig. 7). The use of soil amendments (e.g. gypsum and lime) as part of normal horticultural practice, will have a much greater effect on soil pH than the use of the 2 water sources studied here (Stevens et al. 2000b).

[FIGURE 7 OMITTED]

Sodium absorption ratio

There were significant differences in SAR between irrigation history, soil depth, and soil type (P [less than or equal to] 0.05). Irrigation water history had a greater influence on soil SAR1:5 than E [C.sub.1:5] (Figs 8 and 6). Mean SA[R.sub.1:5] values of RCW-irrigated soils were generally 3 times greater than bore water irrigated soils at all depths down the soil profile (Fig. 8). SA [R.sub.1:5] values in virgin soils were low in the topsoil (<0.7) but higher than bore and RCW-irrigated soils at depth (>7.0), this was likely to be a consequence of the upward migration of saline, Na-dominant, groundwater (i.e. SAR = 44).

There is still considerable debate on the functional definition of a sodic soil (reviewed by Sumner et al. 1998). One definition of a sodic soil is a soil with an ESP >6 (Northcote and Skene 1972). For the soils of the NAP this would equate to a SA[R.sub.1:5] of 1.2 (Fig. 3), and all soils except for the 0-20/30 cm depth of the bore/virgin irrigated soils would be considered sodic (Fig. 8). However, if a scheme for prediction of the dispersive behaviour of soils, similar to Rengasamy et al. (1984) is adopted, and SAR and EC values are set to define sodicity in relation to soil dispersivity and sodicity-induced problems (e.g. restrictive drainage, surface crusting, and high soil strength); an SA[R.sub.1:5] of 2 at low E[C.sub.1:5] (<0.5) and 3 at higher E[C.sub.1:5] (0.5-1.0) would give a more practical definition for sodic soils on the NAP (calculated from Fig. 5). Using this practical definition, on average the bore water irrigated soils would not be classified as sodic at any depth (Fig. 8). Below 20 cm, RCW-irrigated and virgin soils would be considered sodic, and the upper 20 cm of the RCW-irrigated soil would be marginally sodic.

Detrimental soil chemical changes, due to the sodic nature of RCW used for irrigation on the NAP for the last 10 to 28 years, can lead directly to changes in soil physical properties. Current farming practices on the NAP do not address these potential detrimental effects, as sodicity has increased in soils with RCW use. lf these sodicity-induced effects are not managed correctly now, soil permeability could be reduced to a level that will not sustain adequate leaching fractions. However, the functional implications of these SAR changes are difficult to interpret, given the discussion above on the functional definitions of sodicity. If leach fraction are decreased due to lower soil permeability, this may lead to increased salinity or the formation of perched water tables necessitating mechanical drainage. Others have reported significant decreases in soil infiltration rates or hydraulic conductivities as the soil ESP or SAR increased (McIntyre 1979; Oster 1994) on a range of soils. Matheson and Lobban (1974) speculated that higher salinity at one site was from a perched water table on the NAP.

As an indication of possible soil saturated hydraulic conductivity changes, data from McIntyre

(1979) and (Fig. 8) were used to estimate decreases in soil permeability. Using SAR values for a soil depth of 0-10 and 90-100 cm (Fig. 8), we calculated that 2.75 and 10 cm/day (respectively) decrease in permeability of the RCW-irrigated soil compared with the bore-irrigated soil could occur. Permeability data collected in 1975 (Anon. 1976) from the NAP suggests, even though many of the NAP soils had high permeability, for some subsoils (below 30 cm depth) a decrease in permeability of 10 cm/day would impact significantly on soil drainage. However, the fact that soil salinity has remained at acceptable levels after 10-28 years of RCW use, suggests that leaching fractions have been maintained and soil permeability has not been sufficiently restricted by SAR changes. Decreases in SAR of bore water-irrigated soils relative to virgin soils suggest that these soils, which are highly sodic at depth (80-100 cm), maintain sufficient permeability to allow leaching when irrigated with good quality water. These observations contradict the permeability calculation above.

This apparent contradiction could be due to: (1) heterogeneity of NAP soils and sodic soils which are not sufficient to develop a continuous barrier to vertical water permeability, (2) permeable surface soils that can allow drainage water to flow horizontally if vertical movement is restricted, (3) the sodicity effects are not yet extensive and severe enough to be physically significant, or (4) the data reported by McIntyre (1979) used to estimated soil permeability was obtained using low salinity water (EC = 0.09 dS/m). Therefore, our permeability calculations would not take into account effects arising from the increase in soil-water salinity, from salts in the soil, as water passes through the soil profile. Increases in soil-solution salinity would suppress soil dispersion and increase permeability (Rengasamy et al. 1984)

Further research is required to quantify the effects of RCW-induced soil sodicity changes, relative to bore water-irrigated and virgin sites (Fig. 8), on the changes in permeability (hydraulic conductivity) of NAP soils. The variability in soils, water and crop management bas made it difficult to determine if changes in SAR are at equilibrium or are on an upward trend. However, there were no significant ($P > 0.05$) correlations between soil SAR and period of RCW irrigation at any depth, suggesting that a new equilibrium may have been reached within a 10-year period of RCW use.

Data presented in this paper and the literature suggest a potential problem with soil SAR in the future if management strategies are not devised and implemented. Further work is required to assess any correlation between SAR values and soil permeability in order to obtain a functionally defined critical SAR value for the soils of the NAE Once defined, a functional SAR value for the soils of the NAP will allow more rational assessment of shifts in soil SAR values from use of RCW.

Boron

The minimum [B.sub.se] concentration found in soil cores taken from the NAP was equivalent to 0.07 mg/kg. Non-limiting concentrations of [B.sub.hws] in soils, from a deficiency perspective, are generally in the 0.15-0.5 mg/kg soil (Bell 1999). This equates to approximately 0.02 0.04 mg [B.sub.se]/L, suggesting B deficiency would not be an issue on these soils.

There were significant (P [less than or equal to] 0.05) interactions for irrigation history and depth (Fig. 9), but not for soil type. Data suggest that long-term irrigation with bore water has led to leaching of some B down the soil profile, whereas long-term RCW use has increased B concentrations in the topsoil (0 20 cm), but maintained similar B concentrations to virgin soils in the subsoils (Fig. 9). Others have found that irrigation of sandy soils with sewage water containing similar concentrations of B (0.4 mg/L) increased B concentrations in the lower soil profile, but after 67 years of irrigation they had not reached toxic levels (El-Hassanin et al. 1993)

[FIGURE 9 OMITTED]

Use of RCW has increased B concentrations in topsoils (0-20 cm) to a level where according to Maas (1986), B-sensitive plants could suffer yield decreases, i.e. [B.sub.1:5] >0.3 mg/L or [B.sub.se] >0.5 mg/L (we assumed that soil water B ~ [B.sub.se]). At lower depths (e.g. 50-100 cm), B concentrations in RCW-irrigated soils are similar to virgin soils and levels are such that root penetration of moderately tolerant plants (Maas 1986) could be restricted, i.e. [B.sub.1:5] >1.2 mg/L or [B.sub.se] 2.0-4.0 mg/L.

The fact that most of the data for critical B toxicity values quoted by Maas (1986) were derived from sand culture experiments, led us to conclude that for most soils and plants grown on the NAP, the B concentrations reported here would not cause yield reductions. However, further soil and crop monitoring studies should be undertaken to determine if B levels increase in the future.

Although RCW use has led to a significant increase in soil B concentrations in the topsoils, these changes should not be detrimental to plant growth. However, caution in the use of fertilisers containing B should be practised as B concentrations for all soils were close to critical toxicity thresholds.

Conclusion

Use of RCW on the soils of the NAP has not led to any detrimental changes in the B concentrations in soils or soil salinity. However, the use of RCW has led to significant changes in soil SA [R.sub.1:5] values. This may lead to the development of more sodic, dispersive soils if RCW SAR remains >9. Changes in soil sodicity may result in decreases in soil permeability, which in the long-term could restrict leaching, leading to increases in soil salinity. If good management is adopted now, i.e. correct irrigation scheduling and application of soluble calcium amendments, these detrimental changes should be manageable. The lower SAR of the Class A water and future decreases in this SAR (due to changes in the chemistry of influent into the water treatment plant) should decrease the likelihood of detrimental sodicity developing. However, a functionally defined critical SAR value for the soils of the NAP needs to be determined to help assess SAR changes rationally.

A low cost soil analysis method for determining several key soil physico-chemical parameters has been verified as a tool fur managing soil when irrigating with RCW on the NAP.

Table 1. Description of soils on the Northern Adelaide Plains used this study and the irrigation history of selected sites

Soil type (A)	Characteristics (A)	Soil Classification (B)
1	More than 35 cm sandy	Lithocalcic,

	topsoil over permeable clay or clay loam	Mesonatric, Red Sodosol
2	Between 25 and 35 cm sandy topsoil over permeable clay or clay loam	Sodic, Eutrophic, Red Kandosol
2	As above	Mesonatric, Hypercalcic, Red Sodosol
3	Between 15 and 25 cm sandy topsoil over permeable clay or clay loam	Calcic, Subnatric, Red Sodosol
4	Between 10 and 15 cm sandy topsoil over impermeable clay or dense nodular calcrete	Sodic, Calcie, Red Dermosol
5	About 50 cm friable clay loam over permeable clay and clay loam	Episodic, Epipedal, Black Vertosol

Soil type (A)	Texture (C) (0-15 cm)	Irrigation history
1	Loamy sand	25 years RCW, some bore water for last 3 years
2	Loamy sand	18 years RCW/bore
2	Loamy sand	[greater than or equal to] 12 years RCW/bore
3	Sandy loam	25 years RCW, some bore water for the last 3 years.
4	Sandy clay loam	[greater than or equal to] 14 years only
5	Medium clay	[greater than or equal to] 15 years bore

(A) Matheson (1975).

(B) Described by McDonald et al. (1990) and classified according Is (1996).

(C) Hand textural analysis.

Table 2. Chemical properties of irrigation water used historically

Property	Units	Mean	Median
	Bore water		
TDS	mg/L	844 (A)	820 (A)
EC	dS/m	1.59 (B)	1.55 (B)
SAR	[([mmol.sub.c]/L).sup.0.5]	6.13 (B)	5.85 (B)
pH		7.73 (C)	7.70 (C)
	Reclaimed water		
TDS (E)	mg/L	1446	--
EC (D)	dS/m	2.6	--
EC (F)	dS/m	2.4	--
SAR (E)	[([mmol.sub.c]/L).sup.0.5]	11.3	--
pH (E)		8.0	--

Property	Range	s.d.
	Bore water	
TDS	570-1127 (A)	187 (A)
EC	1.13-2.06 (B)	0.31 (B)
SAR	5.02-7.93 (B)	1.09 (B)
pH	7.60-8.10 (C)	0.11 (C)
	Reclaimed water	
TDS (E)	1195-1814	--
EC (D)	2.1-3.1	--
EC (F)	1.8-3.1	--
SAR (E)	8.0-12.2	--
pH (E)	6.8-9.3	--

--, Insufficient data provided to determine median and standard deviations from the mean.

(A) Historical data (1978-97) from Primary Industries and Resources South Australia, Groundwater Section (n = 7).

(B) Calculated from (A) using data from Stevens et al. (2000a).

(C) From Stevens et al. (2000a).

(D) Calculated from (E) using data from Anon. (1995).

(E) Historical data (1967-70) from Matheson and Lobban (1974).

(F) Historical data (1967-95) from Anon. (1996).

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