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Changes in run-off and groundwater under saltbush grazing systems: preliminary results of a paired catchment study

D L. Bennett

Richard J. George Dr

Richard Silberstein
CSIRO

GOVERNMENT OF WESTERN AUSTRALIA

Department of **Agriculture** and **Food**

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Changes in run-off and groundwater conditions under saltbush grazing systems: preliminary results of a paired catchment study

Don Bennett, Richard George and Richard Silberstein[1]

1. CSIRO Land and Water, Floreat

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Cover picture: Saltbush catchment and run-off measurement weir and equipment (Don Bennett, 2011).

Copies of this document are available in alternative formats upon request.

3 Baron-Hay Court, South Perth WA 6151
Tel: (08) 9368 3333
Email: enquiries@agric.wa.gov.au
www.agric.wa.gov.au

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This study could not have proceeded without the interest, advice and help from Chris, Michael and Dianne Walton on whose property (Condering Hills) the work was undertaken.

The SGSL project (funded by Australian Wool Innovation, Land and Water Australia [Land, Water and Wool project] and the Future Farm Industries CRC) funded the design, planting, instrumentation and monitoring of the study site from 2002 to 2007. From 2007 to 2011, the Future Farm Industries provided funding for continued monitoring. The Department of Agriculture and Food and the CSIRO provided research, technical and analytical services in addition to the provision of capital costs during the nine years of the project.

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Summary

In 2003 a site near Yealering in Western Australia was selected for a paired catchment study to monitor and compare surface water run-off and salt export from unimproved saltland and saltbush systems. This study reports the results of the investigations, undertaken between 2003 and 2010, of changes in hydrology under an improved saltbush system that may lead to both on-site and off-site environmental benefits.

Over the seven years of observation, this study has shown that managing saltland by establishing saltbush, together with an improved annual pasture component, has dramatically reduced the amount of salt, nutrient and sediment discharged in surface run-off. Both the concentration and mass of these elements in run-off were consistently much lower than from an adjacent unimproved area of saltland. Only one-tenth of the salt was discharged from the improved catchment once the saltbush system was established.

The groundwater head relationships observed at the site indicate that head-driven discharge of salts to the ground surface is likely to be small and that evaporation is probably the major mechanism responsible for salt concentration near the soil surface. The study provided direct evidence that watertables were reduced seasonally under the saltbush system. Although the reduction is not of a large magnitude, it occurs predominantly in the mid-summer to mid-winter period, creating greater potential for early winter leaching of salts that accumulate at the soil surface over the summer in salt-affected environments. Reducing the depth to watertable and surface soil salinity in the early winter period creates a better environment for the growth of annual pastures during their critical germination and establishment phases.

Overall, these observations suggest that the saltbush-based system could be used to stabilise a saline area (by providing ground cover, reducing the severity of salinity and minimising soil erosion), and reduce salt, nutrient and sediment run-off into streams over the long term. Further, the system allows the salinity in the topsoil to be reduced through the slowing of capillary rise (due to a lower watertable and dryer soil) coupled with an improved leaching regime. Because the saltland valley systems in the Western Australian wheatbelt are essentially hydrologically one-dimensional, in situ treatments such as establishing saltbush-based systems provide perhaps the best option for saltland management in the short and medium term. Such in situ treatments are also likely to be more attractive to farmers to implement, as they can see the benefit of intervention on their own land. They also see an improved aesthetic value and possible biodiversity benefits, although these latter are not quantified.

The mechanisms responsible for the observed changes in salt run-off are likely to be due to the saltbush inducing changes within the near-surface zone at a much smaller scale than was the scope of observations made during this study. It is recommended that further appropriately scaled studies be undertaken to detect and characterise any changes within the near-surface zone, in order to better determine the mechanism responsible for changes observed. It is also recommended that a saltbush system be established on the untreated catchment and that changes continue to be observed in relevant groundwater and surface water characteristics over time, together with the aforementioned, more detailed observations.

1. Introduction

Dryland salinity is caused by infiltration of rainfall below the root zone (recharge) of annual crops and pastures that are sown on land that was once covered by deep-rooted native perennial vegetation. Increased recharge leads to rising watertables, which in turn dissolve salts in the soil and bring them to the soil surface (discharge), causing secondary salinity.

Dryland salinity continues to be a major threat to agriculture in southern Australia. In the Western Australian (WA) wheatbelt an estimated 0.82 million ha of land is severely salinised while the future equilibrium salinity hazard may be between 2.8 and 4.4 million ha (Caccetta et al. 2009). These forecasts take account of recent groundwater trend data and observations that showed a continued expansion of salinity between 1996 and 2005, even though this was a period of reduced rainfall (George et al. 2008). The forecasts do not cover future climate impacts nor the degree to which management systems can impact on the extent and expansion of saltland.

In the wheatbelt, methods to contain or recover salinity vary from methods that attempt to prevent recharge, or that seek to manage land already affected by shallow watertables. Attempts to manage recharge by partial reforestation or agronomic manipulation have largely failed because of the hydrogeological characteristics of wheatbelt catchments. The inherent low gradient and low hydraulic conductivity of the aquifers mean that the rate of groundwater outflow from wheatbelt catchments is very low. Consequently, a significant amount (over 70 per cent) of the additional recharge must be removed before useful reductions in depth to groundwater and land salinity will result at the catchment scale (Bennett et al. 2011; Bennett & George 2008; George et al. 2004; George 1992). This is beyond the scope of current crop and tree options and is impracticable because of the required impact on farm management and hence its financial viability.

There is therefore a hydrological case for directly targeting current and future saltland with in situ recharge management systems. In the absence of viable salinity recovery or containment there is also an economic case for establishing financially viable salinity management options on saltland (Rogers et al. 2005). Without management, the expected continued expansion of saltland will also increase downstream environmental degradation caused by increasing discharges of salt, water, sediments and nutrients.

Saltland perennial plant systems have been developed that persist and produce fodder, as well as stabilise and improve the visual amenity of saline land (Barrett-Lennard et al. 2005). For example, Anon (2008) showed that improved saltbush systems contained up to 61 per cent more edible dry matter and provided up to about 4.6 times more grazing days per hectare than unimproved saltland, resulting in significant additional wool production.

Revegetation with halophytic chenopods such as saltbushes (*Atriplex* species) has occurred on an estimated 60 000 ha of saline and waterlogged soils in south-western Australia (Hardy & Ryder in press), predominantly in the low to medium rainfall (300–500 mm annual rainfall) mixed crop and livestock zone. Being perennial and active in the summer and autumn, saltbush also has the potential to directly reduce the recharge to watertables in the discharge zones and access saline groundwater.

Despite the potential benefits of saltbush systems, in 2001 it was unclear whether saltbushes produced enough feed of sufficient quality to justify the costs of establishment for livestock systems. There was also little scientific data on their hydrological impacts, in particular their impact on watertables, salt storage, water run-off and salt, nutrient and sediment discharge from saltland.

As a result, the Sustainable Grazing on Saline Lands (SGSL) project was initiated to develop profitable options for livestock production from saline land that reduce the environmental and social impacts of salinity. The SGSL project focused on systems incorporating halophytic shrubs in the low to medium rainfall zone of Western Australia. Two SGSL research sites, at Tammin and Yealering, were selected for study. Research at both sites focused on the animal production aspects as well as comparative studies of water use and groundwater response between saltbush systems and unimproved saltland (Anon. 2008).

In 2003, the Yealering site was also selected for a paired catchment study to monitor and compare surface water run-off and salt export from unimproved saltland and saltbush systems. This study reports the results of investigations undertaken between 2003 and 2010 of changes in hydrology under an improved saltbush system that may lead to both on-site and off-site environmental benefits.

2. Methodology

2.1 Site and treatment details

The site is located about 15 km west of Yealering on Chris, Dianne and Michael Walton's property (117.5^o S, 32.6^o E). The Waltons purchased the property in 1974 and run it as a mixed cropping and sheep grazing enterprise. In 1974 there was no evidence of salinity on the property. However, by the late 1970s, moist areas and later salt emerged on the clayey valley floor country. In the mid 1980s the Waltons initiated a tree planting program and installed several deep drains. Neither initiative was particularly effective, and during the early 1990s experimental plantings of saltbush seedlings, puccinellia, balansa clover and Persian clover were undertaken. Few of these plants persisted.

Soils at the site consist of sandy loam over clay at less than 50 cm. They are referred to as Salmon Gum soils and are widespread within valleys of the wheatbelt region of WA. Verboom & Galloway (2004) mapped the site as being within the Aldersyde 3 Soil Subsystem, which they describe as comprised of mostly salinised broad valley flats with sandy duplex, saline and wet soils, that are usually either bare or have a salt tolerant vegetation cover, being Sea Barley Grass (*Hordeum marinum*) at the site. The surface-soil texture becomes progressively finer towards the northern end of the site.

In 2002, 56 ha of the unimproved valley floor were selected as a SGSL research site (Anon 2008). The site was characterised during 2002 and 2003, prior to planting the saltbush and annual pasture understorey in 2004. Characterisation included assessing the soil conductivity and surface water and groundwater conditions (see Section 2.2). Figure 1 shows the layout of the site, which was divided into two catchments, each about 25 ha, and isolated from each other and from surrounding land by a bund wall constructed large enough to contain surface flow from a 1 in 100 year rainfall event.

Prior to treatment, both catchments were grazed with sheep and on 14 July 2004 the northern catchment (SBP catchment) was sown with a mixed annual legume understorey. The understorey was planted using a 30 run combine with knife-edge points at a rate of 9 kg of seed per ha. Legume species sown included balansa, Persian, gland and subterranean clovers (*Trifolium michelanium, T. resupinatum, T. glanduliferum* and *T. subterraneum*), plus burr and barrel medics (*Medicago polymorpha* and *M. truncatula*). Single superphosphate (containing about 9 per cent phosphorus by weight) was applied with the pasture seed at planting, at a rate of 100 kg/ha. The southern untreated catchment (UT catchment) was not fertilised at this time, nor were both catchments subsequently fertilised.

River Saltbush (*Atriplex amnicola*) and Old Man Saltbush (*A. nummularia*) seedlings, in about equal proportions, were also planted in July 2004, set out in mostly a double row configuration, spaced 10m apart, at a planting density of 660 stems/ha. In 2006, each of the catchments was also divided into four equal paddocks to facilitate grazing measurement as reported in Anon (2008). The plots were not grazed during the establishment or regeneration years. To maximise regeneration of the legumes after establishment, the dry summer pasture residue was lightly scarified to put the legume seeds on the soil surface (where breakdown of seed dormancy was likely to be maximised). Regeneration in 2005 was successful and by 2010 the understorey within the SBP catchment still contained a substantial legume component.

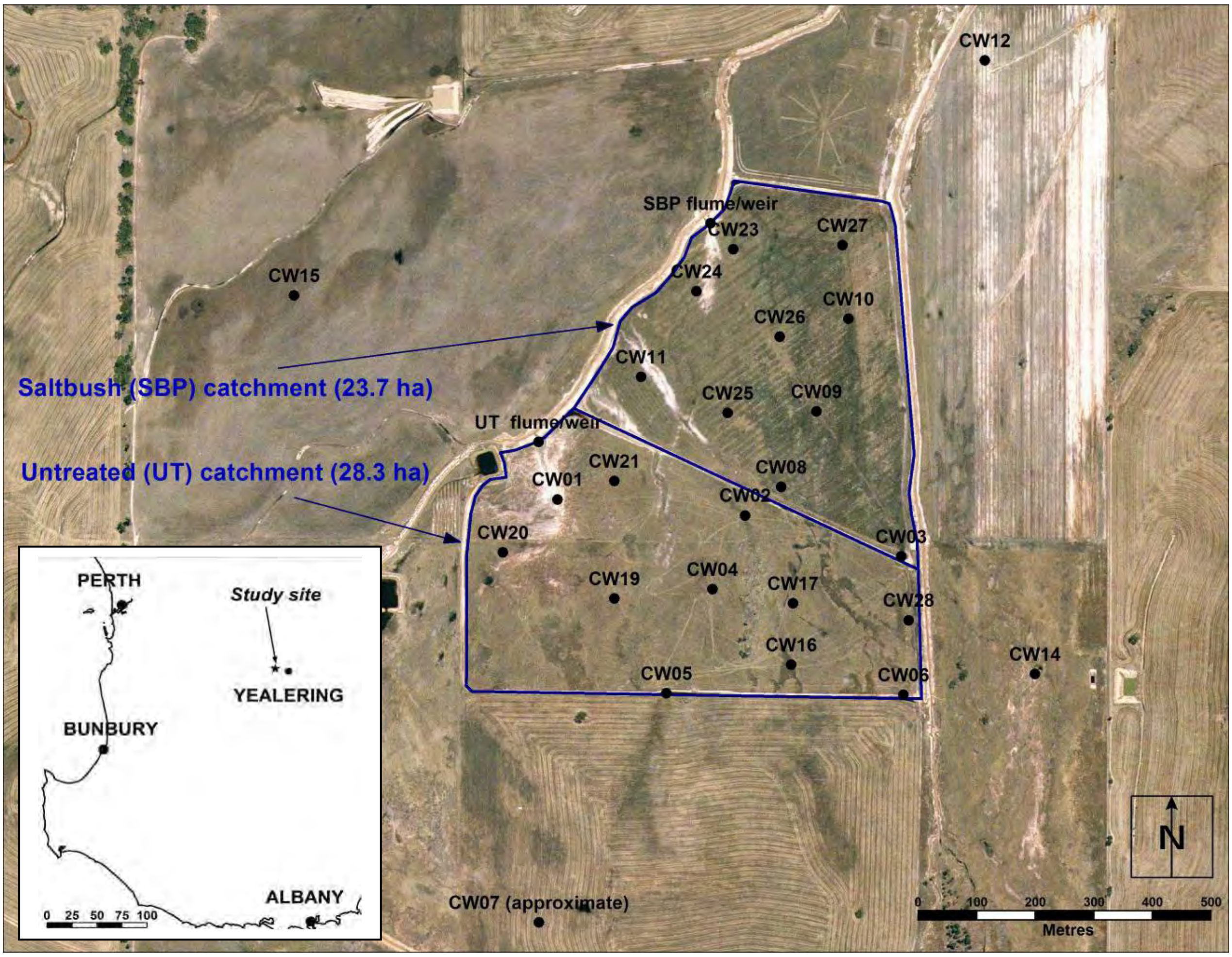

Figure 1 **Location of the study site and the site details, including catchment boundaries, weir outlets and bore locations**

2.2 Site characterisation

Electromagnetic surveys were undertaken across the site in December 2002 using Geonics EM31 (vertical orientation) and EM38 (horizontal orientation) instruments mounted on (EM31v) and towed behind (EM38h), a DGPS-equipped quad bike. Survey lines were 25 m apart, with EM data recorded every 3–5 m along each line. The results of these surveys are shown in Appendix A (Figures A1 and A2). EM survey data shows that the two catchments are generally well matched in terms of the total electromagnetic conductivity. Numerical analysis of the EM38 data (which measures soil electrical conductivity to about 1 m depth) showed similar levels of 'salinity'—as defined by electrical conductivity—in both catchments, with the mean and median EM38h being within two per cent (Table 1). Frequency distribution analysis of the EM38h data also indicates that the catchments have a similar distribution of salinity (Appendix A, Figure A3).

EM38 levels above 50 mS/m, particularly when combined with waterlogging, have been shown to adversely impact on growth rates of salt-sensitive pasture and trees (e.g. Bennett & George 1995). Figure A2 indicates that much of the entire area of both catchments has EM38 levels exceeding this level by a factor of more than four times, indicating a highly salinised landscape. The EM31 measures soil salinity to about 6 m and can be a useful indication of salinity risk. Figure A1 shows that the EM31 signature generally matches the EM38 result but shows a slightly larger area of high conductivity, as would be expected from greater depth. The similarity of the two figures indicates that the site is inherently saline, and that the surface pattern is a close reflection of the underlying salinity.

Ten intact soil cores to 3 m depth were obtained across the site (five in each plot) prior to saltbush and pasture establishment, and were analysed for texture, total salt storage and all major ions. The results indicated that there was about 20 per cent more total salt stored within the top 1.5 m of the soil in the SBP catchment than the UT catchment (Table 1), which is in apparent contrast to the EM38 survey. This difference may be an artefact of the much reduced intensity of soil sampling data relative to the high intensity of EM data acquisition.

During 2003 rotary air blast drilling was undertaken to determine the depth to basement rock and to enable piezometers to be installed. Basement rock was reached at a maximum depth of around 12 m at the site. A total of 16 deep (8 m) and 26 shallow (3 m) piezometers were installed and from these groundwater depth was recorded and groundwater chemistry samples collected and analysed. The initial (June 2003) groundwater results are also shown in Table 1. On average, the SBP catchment had slightly (about 8 per cent) lower deep and shallow groundwater salinity and Electrical Conductivity (EC) than the UT catchment. The initial measurements also showed that the average watertable was slightly (0.07 m) deeper across the SBP catchment.

Table1 **Summary of initial site characteristics of the two catchments in 2003**

	UT catchment	SBP catchment	SBP: UT ratio (%)
Catchment area (ha)	28.3	23.7	83
Mean EM38h	214	213	99
Median EM38h	199	196	98
Area of bare saltpan (ha)	3.2	1.5	47
Sampled salt store in top 1.5 m (kg/m^2)	109	89	82
Depth to watertable (m)	0.67	0.75	112
Deep groundwater salinity (mg/L)	17 100	15 900	93
Shallow groundwater salinity (mg/L)	17 400	15 900	91
Deep groundwater EC (mS/m)	2670	2490	93
Shallow groundwater EC (mS/m)	2720	2480	92
pH	4	4	100
Shallow K_{sat} (m/d)	0.03	0.01	300
Deep K_{sat} (m/d)	0.18	0.09	200

For groundwater there is a highly statistically significant relationship between EC and Total Dissolved Solids (TDS); EC and NaCl; and TDS and NaCl (see Appendix B, Figures B1–B3). Sodium chloride is the major constituent of the salts in the groundwater, comprising about 85 per cent of the TDS. The relationship between groundwater EC and TDS was consistently defined by the relationship:

$$\text{TDS (mg/L)} = 6.36 \times \text{EC (mS/m)},\ R^2 = 0.97$$

The area of semi-bare and eroded saltpan in each catchment was determined using GIS analysis of aerial photography; it showed that the UT catchment has over twice the area (3.2 ha compared to 1.5 ha, Table 1). The saltpan areas are visible in Figure 1 as light grey/white areas, mainly located in the western part of each catchment. They are also confined to the areas of highest EM31v and EM38h response (Appendix A, Figures A1 and A2).

The hydraulic conductivity of the deep (basement) and shallow aquifer materials underlying the catchments is low (about 0.14 and 0.02 m/d respectively).

2.3 Groundwater monitoring

Depth to groundwater was continuously monitored with Odyssey™ capacitance probe dataloggers, which were installed in selected bores. Manual depth measurements were recorded whenever loggers were downloaded and also approximately annually when groundwater samples were collected for EC determination.

2.4 Surface run-off monitoring

To monitor surface run-off of water and transported salt, in 2003 a trapezoidal Repolge Bos Clemmens (RBC) flume was installed in each plot, with bund walls installed to isolate each plot from external surface flows and to guide all surface flow from the paddocks through the stainless steel flumes. Depth of water flow through the flumes was monitored by a temperature-compensated pressure transducer (Monitor Sensors Pty Ltd), and a water sampler (Gamet Pty Ltd) programmed to take a single sample each day for subsequent EC determination. The rate of flow through the flume is determined by a 'rating curve' provided by the supplier of the flume (Geo and Hydro Services Ltd) and tested using the computer program WinFlume™. In 2004 the autosamplers were removed and replaced with Odyssey™ continuous electrical conductivity sensors to monitor the EC of the water travelling through each flume. Due to an untimely computer theft, much of the run-off data from 2008 is not available.

To improve the resolution of flow measurements, in early 2009 the RBC flumes and associated logging equipment were replaced with identical rated 90 degree 'V-notch' weirs. The flow measurement structure of each catchment was equipped with a Sigma® brand (model 900MAX) auto sampler/logger fitted with integrated water depth, velocity and EC sensors. Programming these instruments with the appropriate run-off depth : flow-rate relationship (Boss 1989) allowed logging of hourly flow rate and automatic flow-volume proportional sampling. The sampling interval used was one 0.7 L sample for every one millimetre of run-off (catchment average run-off, based on area). Samples were collected from the autosamplers after each run-off event. Collected samples were then frozen while stored, awaiting chemical analysis by the Chemistry Centre of WA. An additional Ceradiver® combined depth/EC datalogger was installed at each weir as back-up to the Sigma® dataloggers.

Run-off samples were analysed for EC, TDS, Total Suspended Solids (TSS), pH, Chloride (Cl), Total Nitrogen (TN) and Total Phosphorus (TP). Appendix C shows the relationships between EC, TDS and Cl. There is a highly significant statistical relationship between the EC recorded by the dataloggers and the laboratory analysis, with no difference between catchments (Figure C1). Datalogger EC values were subsequently used (with flow rate data) to calculate the discharge salt loads, based on the relationship between EC and TDS (Figure B2) during 2009 and 2010:

$$\text{TDS (mg/L)} = 5.87 \times \text{EC (mS/m)},\ R^2 = 0.97$$

There is also a highly significant statistical relationship between laboratory determined EC and Cl concentration (Figure C3), indicating that Cl comprises about half of the TDS.

Sample nutrient, Cl and TSS data were integrated with continuous flow data to determine the loads of these constituents during 2009 and 2010.

3. Results

3.1 Surface run-off comparison

3.1.1 Run-off rate comparison

Figure 2 shows the comparison between run-off for the catchment planted with saltbush (SBP) and the untreated catchment (UT) for the period July 2003 to March 2011. The graph shows the highly statistically significant relationship between the run-off from the two catchments, with the SBP catchment consistently having about half of the daily run-off rate of the UT catchment over the period.

During the initial period from 2003 to 2005 (corresponding to 108 mm of cumulative run-off from the UT catchment, Figure 1), prior to and until just after the SBP treatment was established—and therefore not expected to cause much impact on run-off rate—the relationship was only slightly higher than that of the entire period of observation.

The slope of the relationship was 0.5873 (coefficient of determination [R^2] = 0.945) in 2003, 0.5845 (R^2 = 0.983) during 2003–2004, and 0.5555 (R^2 = 0.995) during 2003–2005, slightly higher than the slope of 0.4907 (R^2 = 0.994) for the entire period from 2003 to 2010. While there is a trend of reducing slope over the period when the saltbush is becoming more established and which may indicate a small effect on run-off rate, the consistency of the relationship over time suggests that the UT catchment has inherently higher run-off generation characteristics than the SBP catchment.

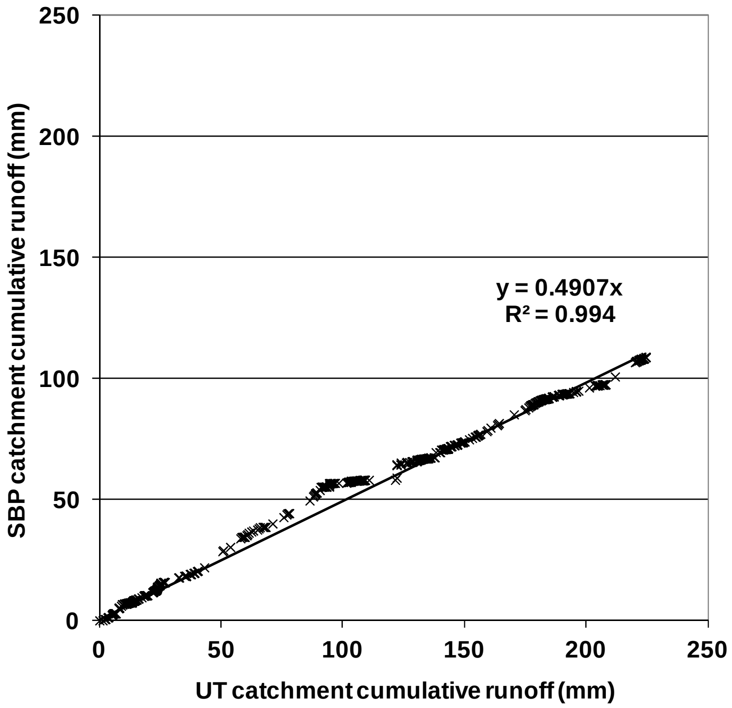

Figure 2 **Double mass curve showing the relationship between cumulative daily run-off from the untreated (UT) and treated (SBP) catchments, 2003–2010**

Average annual rainfall at the site between 2003 and 2011 was about 50 mm less than the long-term average recorded at the Yealering rain gauge (http://srig-web/climate/framesetup.asp, accessed May 2011). Annual run-off generated from the catchments ranged between 2 and 10 per cent (SBP) and 5 and 20 per cent (UT) of annual rainfall (Figure 3).

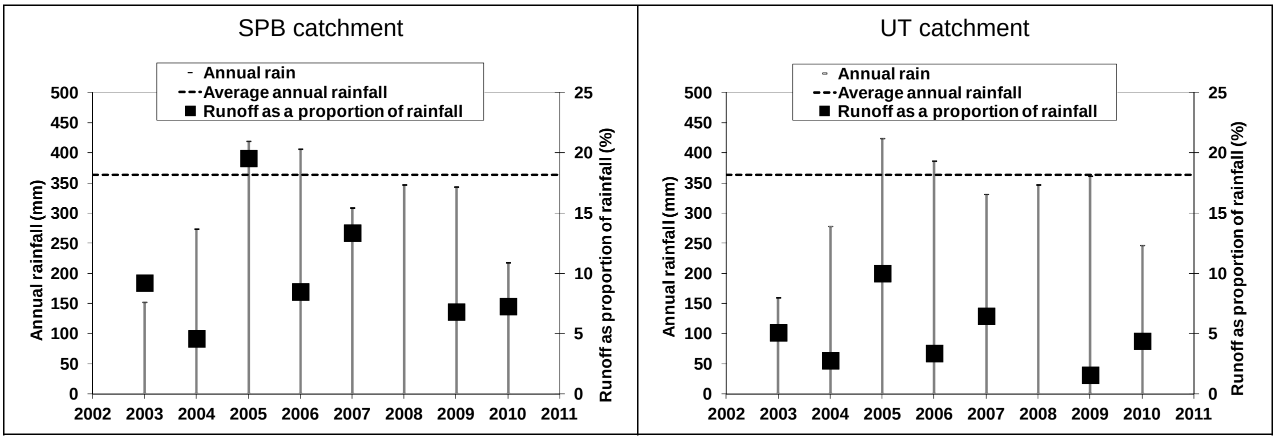

Figure 3 **Annual run-off as a proportion of annual rainfall for the saltbush (SBP) and untreated (UT) catchments, compared to the observed and long-term average rainfall**

3.1.2 Comparison of salt load in discharge

In contrast to the run-off comparison, the slope of the salt load (as TDS) double mass curve is generally much flatter and has a declining slope with time (Figure 4). A slope of unity would indicate the same rate of salt discharged from both catchments. The slope of the relationship between the daily salt load discharged from each catchment in each year (together with their R^2 values) is also reported in Figure 4.

The trace has a relatively steep slope during 2003–2005, the period during which the saltbush was becoming established. The average of the annual slope for the 2003–2005 period is 1.37, which indicates that the SBP catchment was exporting 37 per cent more salt than the UT catchment, even though it had about half of the run-off.

The slope reduces in subsequent years until 2009 when it has a value of 0.07, indicating that the mass of salt exported from the SBP catchment was just seven per cent of that exported from the UT catchment. There is a subsequent slight increase (to 0.28) in 2010, largely resulting from an unseasonal heavy rainfall event (of about 90 mm) that occurred over two days in March. This brought significant flooding and run-off in each catchment and caused a higher rate of salt to be released, from the SBP catchment in particular, than in more normal winter rainfall/run-off events.

Between 2003 and 2010, the total amount of salts discharged from the SBP catchment was 2380 kg/ha, which is 43 per cent of the 5470 kg/ha discharged from the UT catchment. Normalisation of the loads for the inherent lower run-off rate of the SBP catchment was achieved by dividing the SBP salt load by the slope of the run-off double mass curve (from Figure 2). Even after normalising the salt discharge rates to account for the higher run-off from the UT catchment (Section 3.1.1), the SBP catchment still discharged 88 per cent less salts than the UT catchment over the entire period. Moreover, this period includes pre-treatment and establishment phases during which the SBP catchment actually discharged more salts per ha. After 2006, when the saltbush was more mature, just 650 kg/ha (130 kg/ha/yr) of salt was discharged, just 12 per cent (or 24 per cent when normalised for inherent differences in the rates of water run-off) of the 5560 kg/ha (1110 kg/ha/yr on average) discharged from the UT catchment.

It could be argued that the difference in the rate of salts export is, as for the run-off rate, an inherent difference between the two catchments. However, because the rate of salt export was actually higher until 2005, similar in 2005, and then reduced markedly as the saltbush treatment matured, the more plausible explanation is that the effect is due to the established saltbush system.

There are highly statistically significant relationships between Cl and EC, and Cl and TDS, with Cl comprising about half of the soluble solids in the run-off water (Appendix C, Figures C3 and C4).

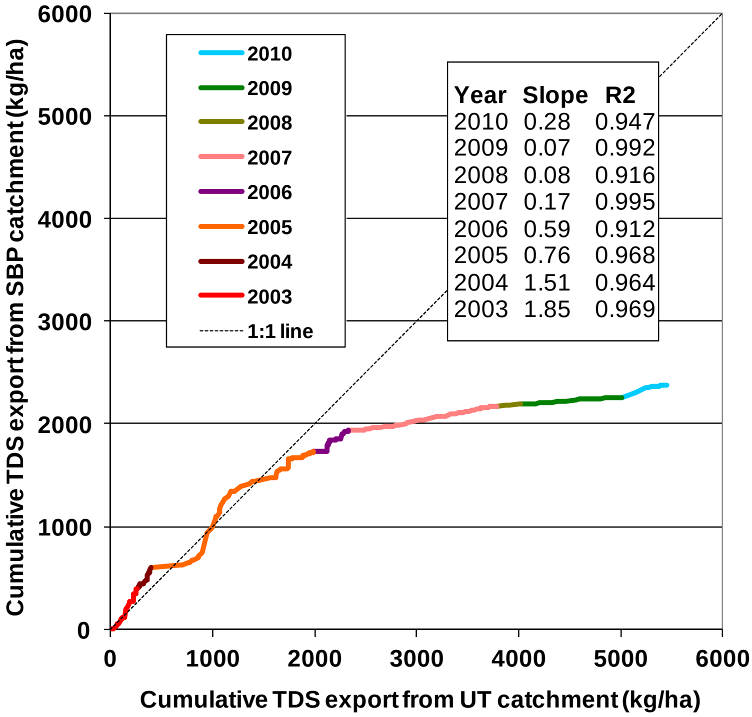

Figure 4 **Relationship between the cumulative daily amount of salts (as Total Dissolved Solids, TDS) exported from the untreated (UT) and saltbush (SBP) catchments, 2003–2010**

3.1.3 Nutrient, Cl and TSS load comparison between catchments

Table 2 shows the annual loads of Total Nitrogen (TN) Total Phosphorus (TP), Chloride (Cl) and Total Suspended Solids (TSS) discharged in the run-off water from the UT and SBP catchments in 2009 and 2010. The SBP catchment had much lower loads of all of the measured constituents than did the UT catchment in both years. The mean loads of TN, TP, Cl and TSS exported from the SBP catchment were about 46, 38, 16 and 14 per cent respectively of the loads exported from the UT catchment over the two years of observation.

When the loads are normalised for the lower run-off rate of the SBP catchment, the amount of TN, TP, Cl and TSS exported from the SBP catchment remains substantially lower (Table 2). The mean ratio of Cl export between the SBP and UT is of similar magnitude to the ratio of TDS exported from the two catchments during 2009 and 2010 (Figure 4).

Table 2 **Annual loads of TN, TP, Cl and TSS from the two catchments in 2009 and 2010 (reported per hectare)**

Catchment	Year	TN (kg/ha)	TP (kg/ha)	Cl (kg/ha)	TSS (kg/ha)
UT	2009	0.82	0.13	368	50
	2010	0.95	0.19	166	106
	Mean	0.89	0.16	267	78
SBP	2009	0.10	0.03	21	6
	2010	0.30	0.09	63	15
	Mean	0.20	0.06	42	11
Normalised SBP#	2009	0.21	0.07	43	11
	2010	0.61	0.18	128	31
	Mean	0.41	0.12	85	21
Ratio*		**0.46**	**0.75**	**0.31**	**0.27**

\# SBP catchment TN, TP, Cl and TSS annual loads divided by the slope of the run-off relationship between the SBP and UT catchment (from Figure 2).

* Ratios of the normalised mean SBP catchment loads of TN, TP, Cl and TSS to the respective mean UT catchment loads.

3.2 Groundwater electrical conductivity (EC)

Figure 5 shows the means of the groundwater EC observed in the shallow and deep bores in the SBP and UT catchments from 2003 to 2011. The catchments have a similar mean shallow groundwater EC of about 2500 mS/m, which has not changed significantly over time. The UT catchment has a consistently higher mean EC in the deep bores, of about 3000 mS/m compared to about 2500 mS/m in the SBP catchment. As with the shallow groundwater, there is no trend in deeper groundwater EC over time.

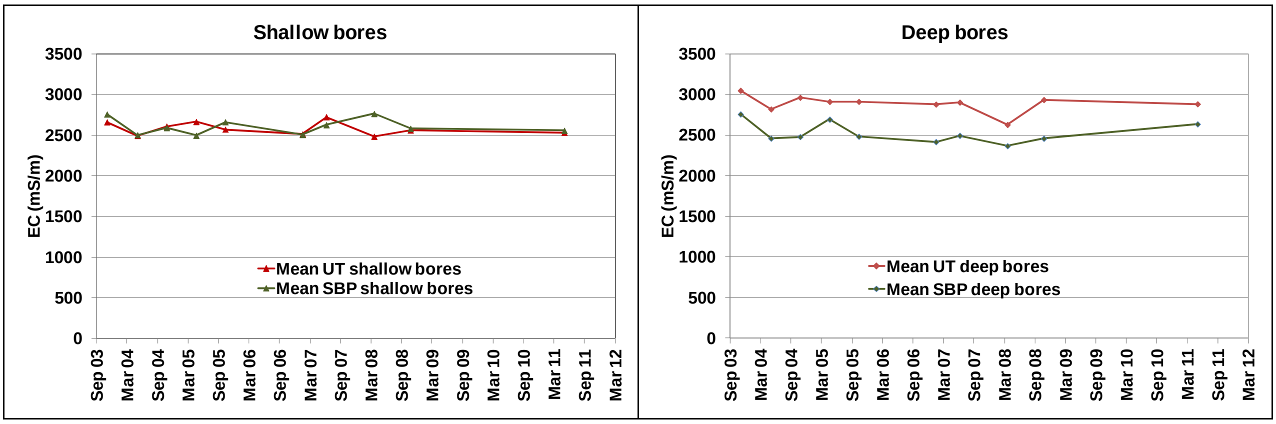

Figure 5 **Mean electrical conductivity of the shallow and deep groundwater beneath the saltbush (SBP) and untreated (UT) catchments**

There is considerable site variability in groundwater EC beneath both catchments. The shallow groundwater varies from about 1000 to 4000 mS/m under each catchment (Appendix D, Figures D1 and D2). The deeper groundwater EC beneath the SBP catchment varies between 2000 and 3000 mS/m, while the UT catchment has higher deep groundwater EC variability, from about 1000 to 3500 mS/m (Appendix D, Figures D3 and D4). There is a general spatial pattern of higher to lower deep and shallow groundwater EC from east to west in both catchments. Also, in general, the spatial pattern of groundwater EC aligns with the spatial pattern of soil apparent EC, as measured at the commencement of the study using EM31 and EM38 instruments (Appendix A, Figures A1 and A2).

3.3 Groundwater depth

The depth to groundwater, as measured using dataloggers installed in all deep and shallow bores in the UT and SBP catchments during 2003–2011, is shown in Appendix E (Figures E1 and E2). Groundwater heads mostly remain within 1.5 m of the ground surface in the deep and shallow bores.

Figure 6 shows the groundwater response in deep and shallow pairs of bores located at the approximate centre in each of the SBP (bore pair CW09D and S) and UT (bore pair CW04D and S) catchments (Figure 1). At each location the deep and shallow bore hydrographs are similar, with generally only slight differences between groundwater heads observed in the different depth bores, indicating a small potential for an upward flux. There are some relatively brief periods when there is slight downward head potential.

Groundwater heads dynamically respond to seasonal rainfall and recharge. Prior to 2006 the response is almost identical in both catchments. Subsequently however, the dynamics of the groundwater within the SBP catchment bore pair changes—with the peaks in groundwater level becoming more subdued and the troughs becoming more pronounced, compared to both the earlier period and the entire record for the UT catchment.

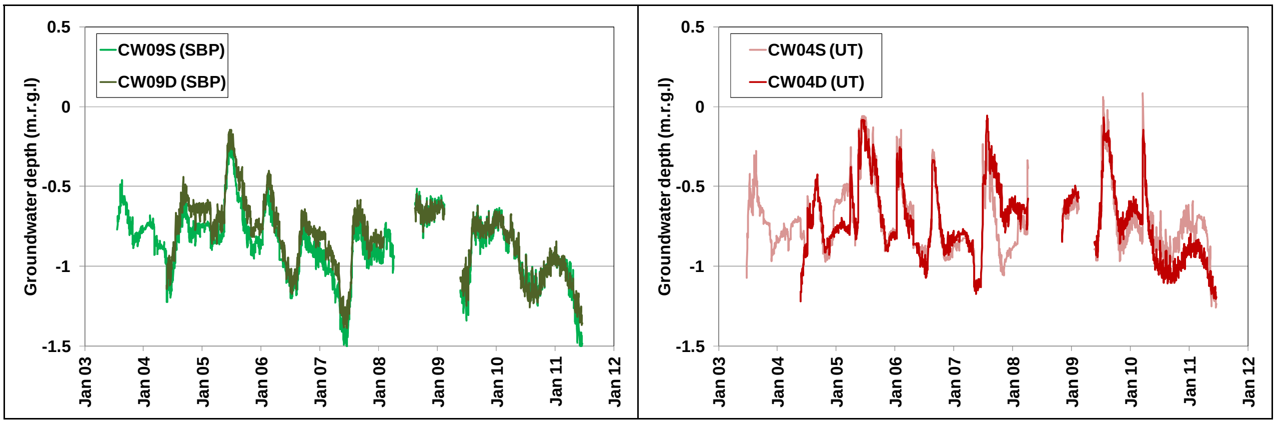

Figure 6 **Groundwater heads in deep and shallow bore pairs located beneath the saltbush (SBP) and untreated (UT) catchments**

The variation between the two catchments is clearer in Figure 7, which compares the difference between the depth to groundwater in a shallow bore located in the approximate centre of each catchment (bores CS04S and CW09S) and bores located near the outlet of the catchments (bores CW24s and CW20S).

Bores at the central location during 2003–2005 indicate only a small difference, that varies seasonally, in groundwater depth (less than 0.5 m). The largest difference occurs each year over this period and, subsequently, during the initial winter period when groundwater levels in the SBP catchment have a delayed response to rainfall-induced recharge, compared to the UT catchment. After 2005, these peaks in the difference graphs become larger, so that by 2007 there are substantial intervals when the SBP catchment bores have significantly lower groundwater levels. Well below average rainfall in 2010 and 2011 resulted in much lower winter recharge, as evidenced by the subdued response in groundwater levels at all bore sites. Over this period, the groundwater level in the SBP bores fell more than in the UT catchment, resulting in a steady increase in the difference graph. This trend may be due to the saltbush continuing to transpire from the groundwater under the low rainfall conditions.

An exception to the general trend during the 2010–2011 period occurred in March 2010 when about 90 mm of rain fell on the site over two days. The effect of this can be seen as spikes in the groundwater level at both mid and lower catchment locations (Figure 7). Bore CW04S (mid UT catchment) responded to this rainfall event whereas the mid-catchment SBP bore did not. Both UT and SBP bores at the lower catchment location respond similarly, however groundwater levels in the SBP bore fell more rapidly afterwards.

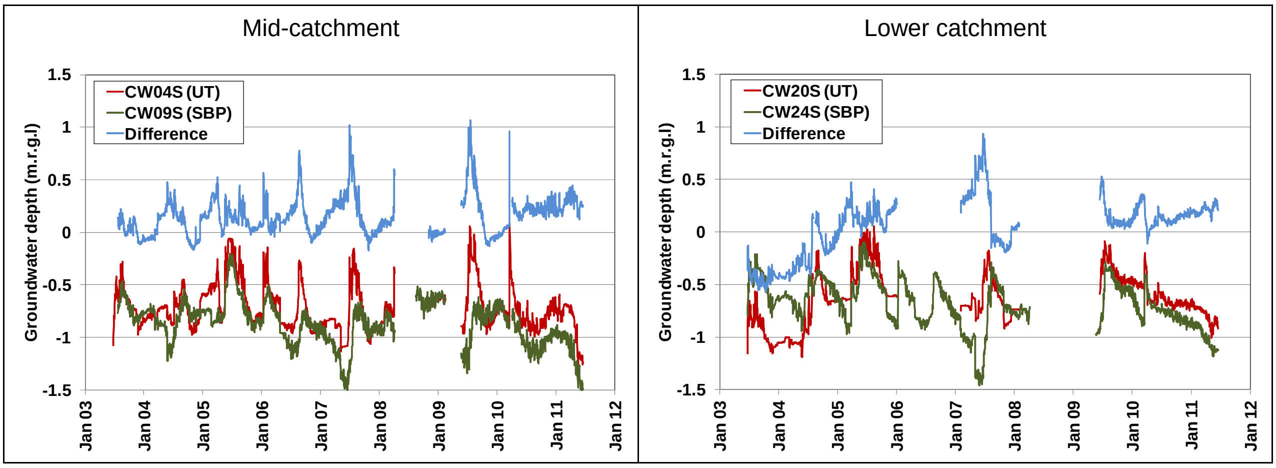

Figure 7 **Difference between shallow bore groundwater head in the saltbush (SBP) and untreated (UT) catchments at mid and lower catchment locations**

Figure 8 compares the mean depth to groundwater across the SBP and UT catchments, calculated from all bores that contain loggers, and shows patterns similar to those of the individual bores shown in Figure 7. Again the catchments behave similarly in 2003 and 2004, with the SBP catchment displaying lower groundwater levels, particularly during the autumn and early winter period as a result of the characteristic delay in response to early winter recharge events. However, the early winter differences (manifested as peaks in the difference graph) intensify each year from 2005, up to a maximum in 2009, when the difference is greater than 0.5 m. After 2009 there are, as for the individual bores, no pronounced peaks, as a result of below average rainfall and recharge. However the groundwater levels in the SBP catchment continue to fall relative to the UT catchment, so that by June 2011 the average groundwater level is about 0.4 m deeper than in the UT catchment.

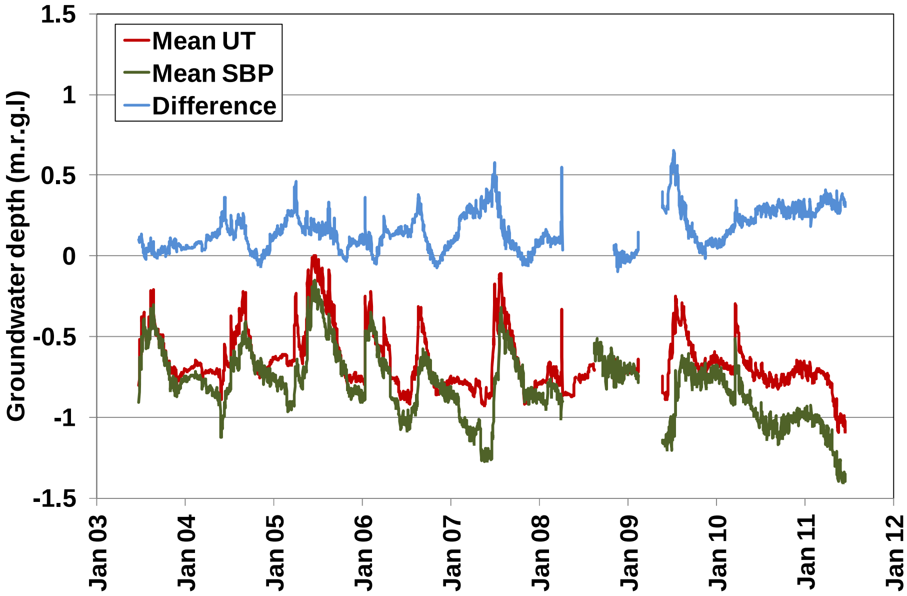

Figure 8 **Difference between the mean groundwater head from all shallow bores in the saltbush (SBP) and untreated (UT) catchments**

4. Discussion

The higher run-off generated by the UT catchment throughout the entire monitoring period is likely due to it having a lower run-off threshold than the SBP catchment. Several factors may contribute to this difference of threshold, however the major factor is likely to be the large area of semi-bare and eroded saltpan within the UT catchment. The saltpan areas have almost no vegetation on them to slow run-off and have lost most of their original sandy soil surface horizon, leaving exposed clay over much of their area. They are also located at the lowest elevation in the catchments and have the shallowest watertables. In addition, the saltpan area within the UT catchment is immediately adjacent to the catchment outlet, meaning that any run-off retention and transmission losses from it would be negligible and certainly less than in the case of the SBP catchment with its more remote and dispersed saltpan areas.

The run-off generation process is likely a combination of saturation excess (whereby the soil profile is completely saturated and cannot accept any more rainfall) and infiltration excess (whereby the rainfall rate exceeds the soil infiltration rate) processes. It is unclear from the data whether either process is dominant, making it difficult to hypothesise the effect of the treatment on run-off. The treatment's substantial vegetation cover could be expected to reduce run-off rates in the long term by stabilising the soil surface and reducing erosion, as well as slowing run-off to allow more time for infiltration. However, a treatment effect on run-off has not been pronounced at the site, which may be due to the dominance of the saltpan areas in generating the run-off, masking more subtle changes from the other areas.

The rates of run-off observed during this study are higher than rates reported from other catchments in the WA wheatbelt (e.g. Dogramaci et al. 2004, Mayer et al. 2005). However, no other run-off studies have been undertaken in isolation on small saline areas where salt affected areas contribute a disproportionally large amount of a catchment's total run-off. Given that bare and eroded sections of saltland with shallow watertables contribute a high proportion of run-off, there is a strong argument to target saltland with profitable vegetation management systems that maximise ground cover and minimise erosion to reduce water, salt and nutrient run-off.

The similarity between deep and shallow bore heads indicates generally only a weak potential for upward flux of deeper groundwater, even though the sites are located in a valley, down-gradient of surrounding hillslope aquifers, where upward heads may be expected. This lack of significant potential for upward flux of groundwater is likely a result of a combination of modest catchment-scale groundwater gradient plus the low in situ hydraulic conductivity of the regolith in the valley.

The EM31 (depth of penetration 6 m) survey image is very similar to the EM38 (penetration to about 0.6 m) image (Appendix A), also indicating that the hydrology of the site is predominantly one-dimensional, without large lateral groundwater fluxes. This is typical of saline areas in wheatbelt valleys in WA, and emphasises that salinity management in these situations needs to directly target the affected areas. These one-dimensional systems are amenable to treatment as farmers can see the benefit of intervention on their own land (Barrett-Lennard et al. 2005). The similarly high shallow and deep salt levels also suggest that unless a management technique is found to either dramatically enhance salt export from the site or minimise concentration at the soil surface, the underlying salt will continue to rise into the shallow soil and maintain high levels of soil salinity well into the future. There is so much water and salt on the site studied that techniques to enhance salt export may take hundreds of years to have a major impact. Given the salt storage in the top 1.5 m of soil (890,000–1,090,000 kg/ha, from Table 1) and the rate of salt removal in run-off (130–1110 kg/ha/yr, see Section 3.12), an estimated 1000 and 8000 years (UT and SBP catchments respectively) would be required to remove all the salt from this zone, if there were no additional salt deposition from groundwater discharge. Substantial salt discharge enhancement would thus be needed to have much effect within normal planning timeframes. Techniques that minimise salt accumulation within the root zone are therefore a more practical technique for managing saltland.

The results show that the saltbush system has dramatically reduced salts, nutrient and sediment discharged in surface run-off, both sequentially over time and in comparison with the untreated catchment. Most of the reduction in salts run-off is likely a result of reduction in surface accumulation of salts due to reduced evapo-concentration caused by the saltbush system drying the near surface profile and lowering watertables.

The lower TP (and TSS to some extent) run-off may be a direct result of the increased level of vegetative cover on the SBP catchment, which then reduces the amount of sediment loosened and removed by erosion and soil flocculate loss. The lower TN loss from the SBP catchment was not expected; given it has higher productivity, grazing intensity and proportion of N-fixing clovers (Anon 2008), conditions that would typically result in higher nitrogen runoff. However, the likely higher levels of soil organic matter in the SBP catchment could greatly increase the potential for de-nitrification processes to occur (Whitehead 1995), plus there is increased potential for N uptake from the soil by the perennial saltbush plants. A combination of both of these processes may explain the reduced N export from the SBP catchment.

The groundwater observations indicate that the saltland pastures are increasing the depth to groundwater in the SBP catchment relative to the UT catchment, with the greatest differences occurring in the summer to mid winter period. This is not surprising given that the saltbush is capable of transpiring soil water and drying the profile during this period.

These results are consistent with those presented by Anon. (2008), who report measurements of transpiration (using the Bowen Ratio and small ventilated chamber techniques) obtained in the SBP and UT catchments during the SGSL study. Their data shows that prior to the saltbush system being established, the total evaporation from the two catchments was very similar, but by 2005 there was a significant increase in evapotranspiration in the SBP catchment. While the saltbush plants covered only about 15 per cent of their catchment, they accounted for about half of the total evapotranspiration. Of the few studies of water use by saltbush, there is evidence that they will use groundwater with salinities of 80 to 100 per cent of seawater (Barrett-Lennard & Malcolm 1999).

Slavich et al. (1999) showed that transpiration rates of regularly grazed saltbushes are low, being less than 0.3 mm/d on average over the year, and concluded that saltbush has negligible hydrological impact. This conclusion is valid for the magnitude of rainfall, recharge and soil evaporation rates in catchment scaled water balances. However at much smaller scales, such as within the near-surface root zones of saltbush and annual pasture, the extra transpiration may provide a significant benefit during certain critical times of the year. For example, relatively small reductions in moisture content and reduced watertables caused by the saltbush, particularly during summer, autumn and early winter, may reduce evaporative salt concentration effects, enhance salt leaching from the soil surface, and reduce waterlogging. Slavich et al.’s (1999) data showed that the transpiration rate varied seasonally and was higher during the autumn period when the saltbush was most active in transpiring shallow groundwater. Relatively small reductions in soil water content that may also manifest as temporary reductions in groundwater level during the autumn/early winter period, as indicated by our data, while not regarded as hydrologically significant at the catchment scale, likely provide improved near-surface soil salinity conditions for plants, and so are significant in that respect. Improved conditions will manifest as better germination and growth of the annual pasture component of the system.

If, as the results suggest, saltbush systems are able to keep the near surface soil profile drier and the watertable deeper than annual pastures, improved growing conditions for more productive pasture species should follow. This will allow greater productivity, and if suitably managed, also protect the erosion-prone saltland from further degradation.

5. Summary and recommendations

Over the seven years of observation, this study has shown that managing saltland by establishing saltbush, together with an improved annual pasture component, has dramatically reduced salt, nutrient and sediment discharged in surface run-off. Both the concentration and mass of these elements in run-off were consistently much lower than from an adjacent unimproved area of saltland. By comparison, one tenth of the salt was discharged from the improved catchment once the saltbush system was established.

The groundwater head relationships observed at the site indicate that head-driven discharge of salts to the ground surface is likely to be small and that evaporation is probably the major mechanism responsible for salt concentration near the soil surface. The study provided direct evidence that watertables were reduced seasonally under the saltbush system. Although the reduction is not of large magnitude, it occurs predominantly in the mid-summer to mid-winter period, creating greater potential for early winter leaching of salts that accumulate at the soil surface over the summer in salt-affected environments. Lowering the watertable and reducing surface soil salinity in the early winter period creates a better environment for the growth of annual pastures during their critical germination and establishment phases.

Overall, these observations suggest that the saltbush-based system could be used to stabilise a saline area (by providing ground cover, reducing the severity of salinity and minimising soil erosion), reduce salt, nutrient and sediment run-off into streams over the long term, and allow the salinity in the topsoil to be reduced through the slowing of capillary rise (due to a lower watertable and dryer soil) coupled with an improved leaching regime.

Importantly, in parallel with the above environmental benefits, pasture and animal production on saltland has been shown to increase as a result of the saltbush system, as shown by a complementary study undertaken at the same site (Anon. 2008).

Because the saltland valley systems in the WA wheatbelt are essentially hydrologically one-dimensional, in situ treatments such as establishing saltbush-based systems provide perhaps the best option for saltland management in the short and medium term. Such in situ treatments are also likely to be attractive to farmers to implement, as they can see the benefit of intervention on their own land (Barrett-Lennard et al. 2005).

However, the results provide an apparent conundrum in the hydrologic mechanism that is responsible for the changes. On the one hand, there were reductions in salt discharge and reductions in watertable depth, while on the other hand there were no changes in either run-off volume or groundwater salinity—as could have been expected. The answer may lie in the hydrologic changes that may be induced by the saltbush at the soil surface and within the near-surface unsaturated zone. It may be that small changes within the near-surface zone are enough to produce the observed results, yet are at a much smaller scale than are able to be detected by the coarser-scaled observations of groundwater condition made during the study.

It is therefore recommended that further appropriately scaled studies be undertaken to detect and characterise any changes within the near-surface zone, in order to better understand the mechanism responsible for the changes observed. This research should also be designed to measure the changes to the salt concentration within the near-surface soils over the longer term, to determine whether there is a risk of detrimental salt accumulation (Barrett-Lennard & Malcolm 1999) in the near-surface under saltbush based systems.

Although the data show clear indications of the effects of establishing saltbush systems, some questions remain about the methodology. Firstly, seven years is a fairly short interval of observation from which to have confidence that trends will persist and be meaningful in the long term. Secondly, there was a very short calibration period in which baseline conditions of the saltbush treated catchment were established, so some questions remain about the observed changes that are regarded as an effect of the treatment. While the paired catchment methodology used is a way of reducing this uncertainty, a more certain test would be to establish a similar saltbush system on the untreated catchment and continue to observe changes in relevant groundwater and surface water characteristics over time. This approach is recommended, together with the aforementioned, more detailed observations of salt and water changes in the near-surface unsaturated zone.

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Appendix A: EM38 and EM31 surveys

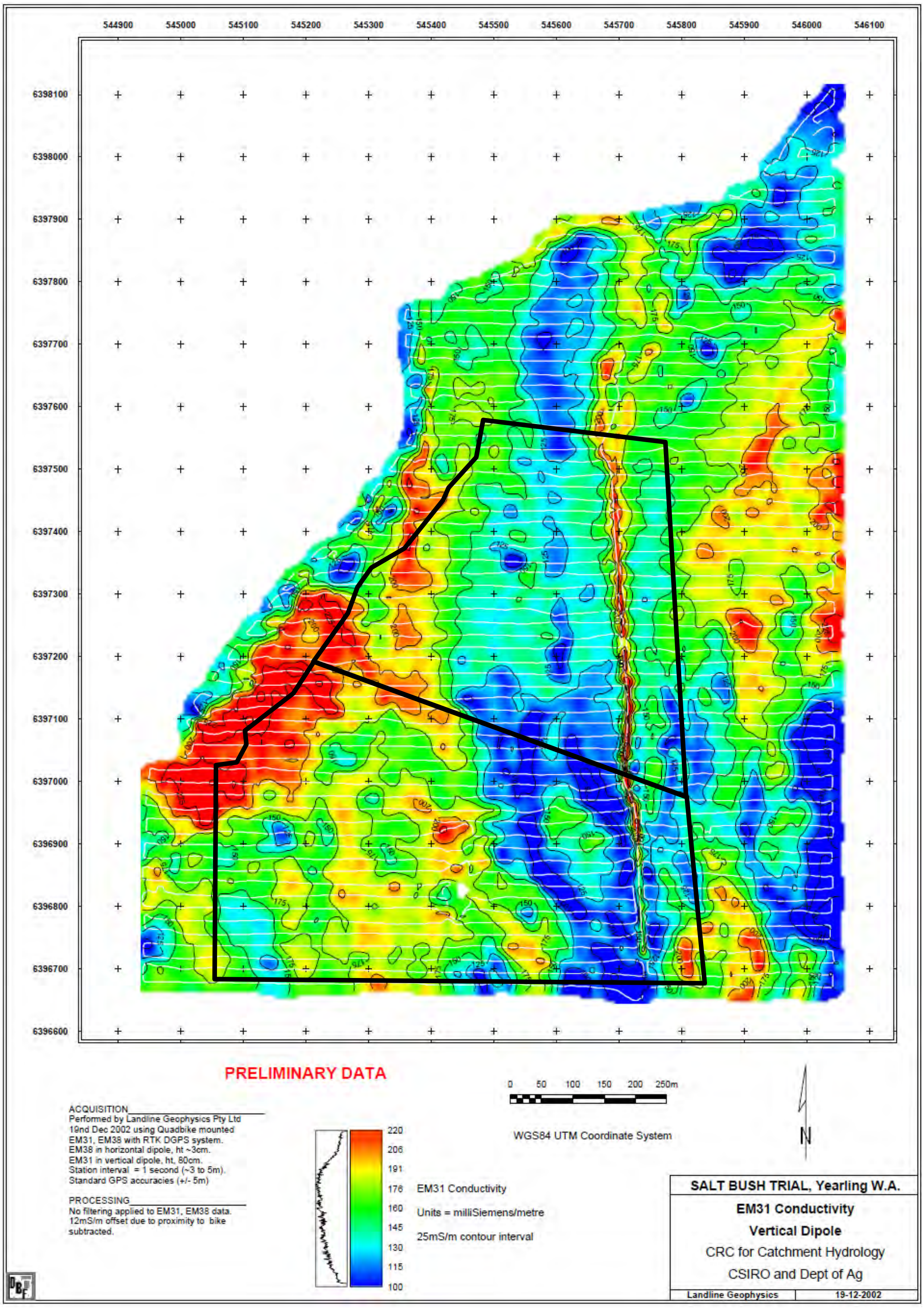

Figure A1 **Apparent electrical conductivity of the catchments' soils determined by the EM31 in vertical mode**

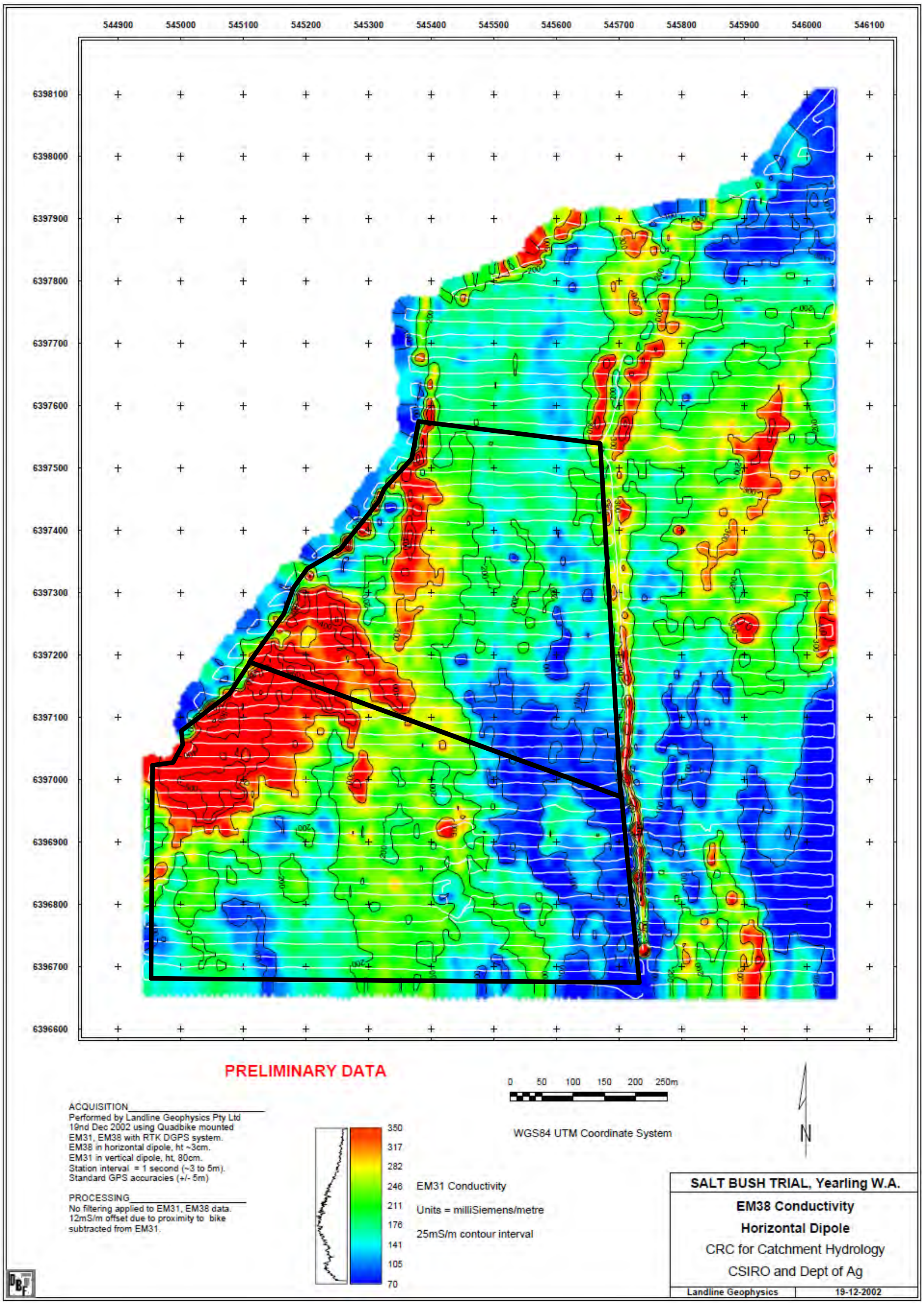

Figure A2 **Apparent electrical conductivity of the catchments' soils determined by the EM38 in horizontal mode**

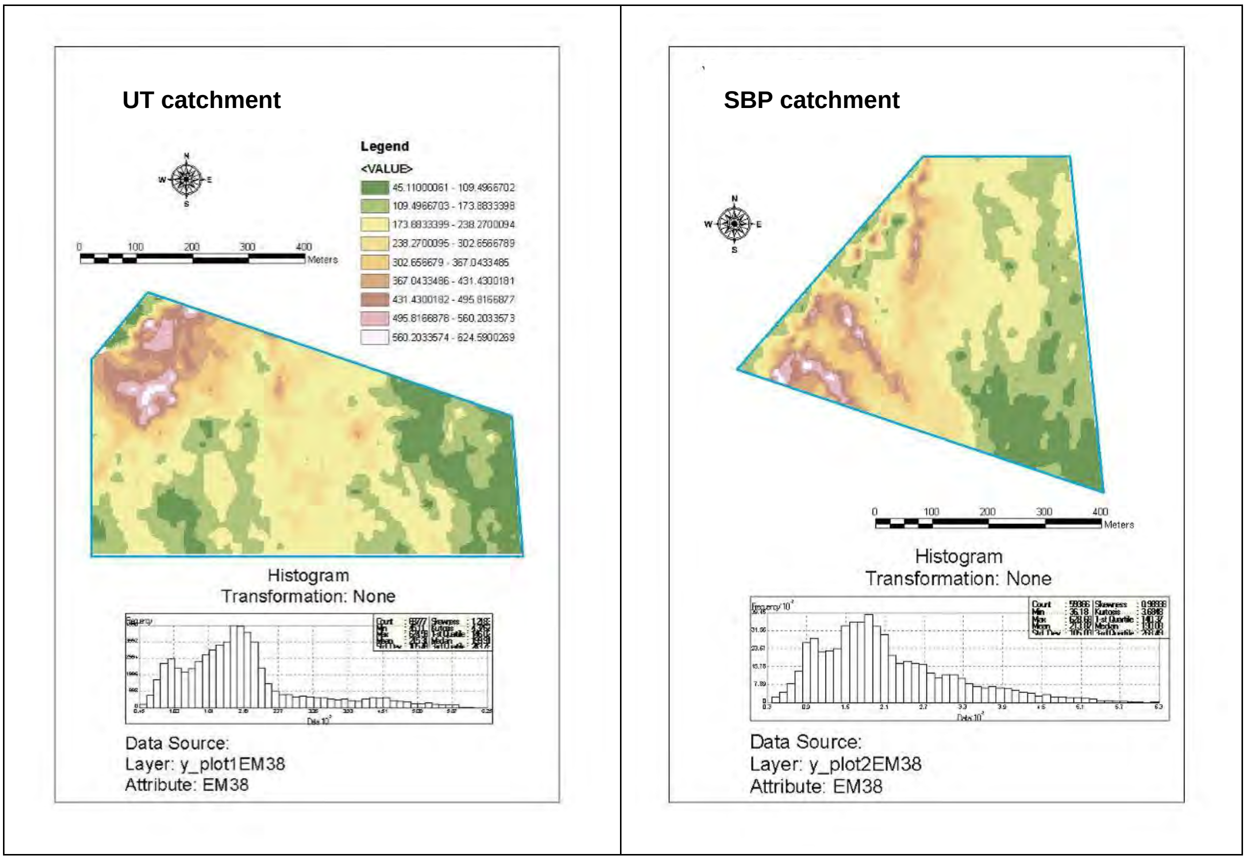

Figure A3 **Frequency distribution plots of EM38h data for the untreated (UT) and saltbush (SBP) catchments**

Appendix B: Groundwater EC, TDS and NaCl

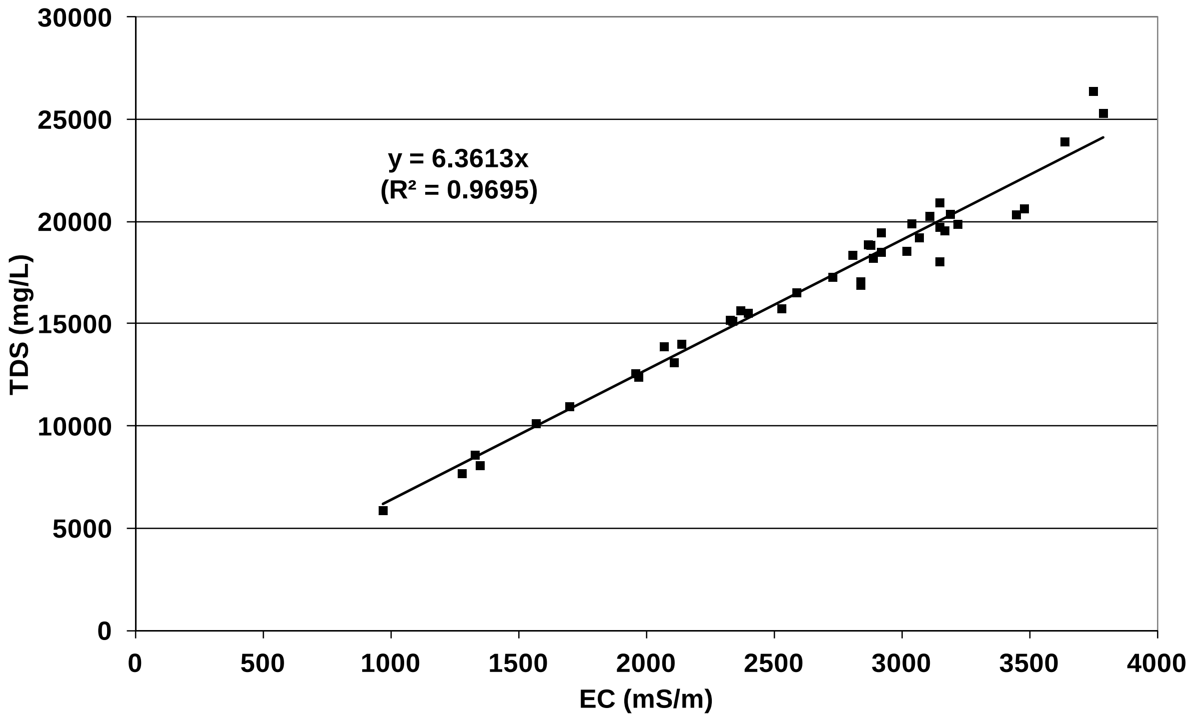

Figure B1 **Relationship between Electrical Conductivity (EC) and Total Dissolved Solids (TDS)**

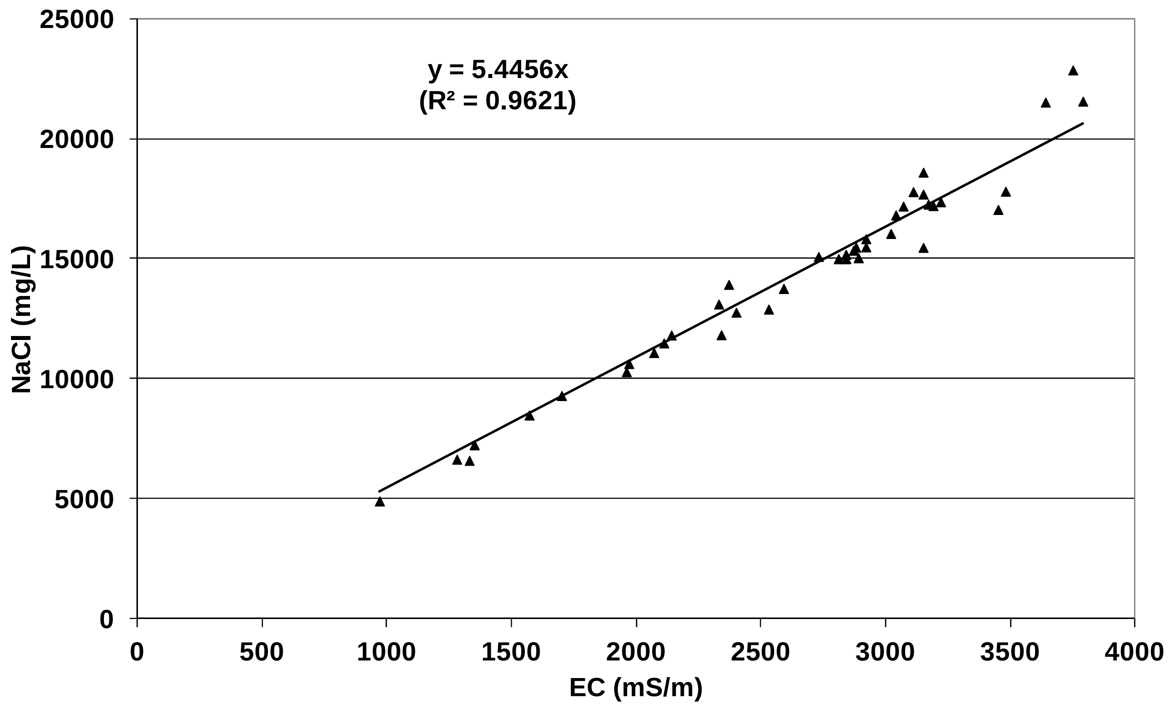

Figure B2 **Relationship between Electrical Conductivity (EC) and NaCl**

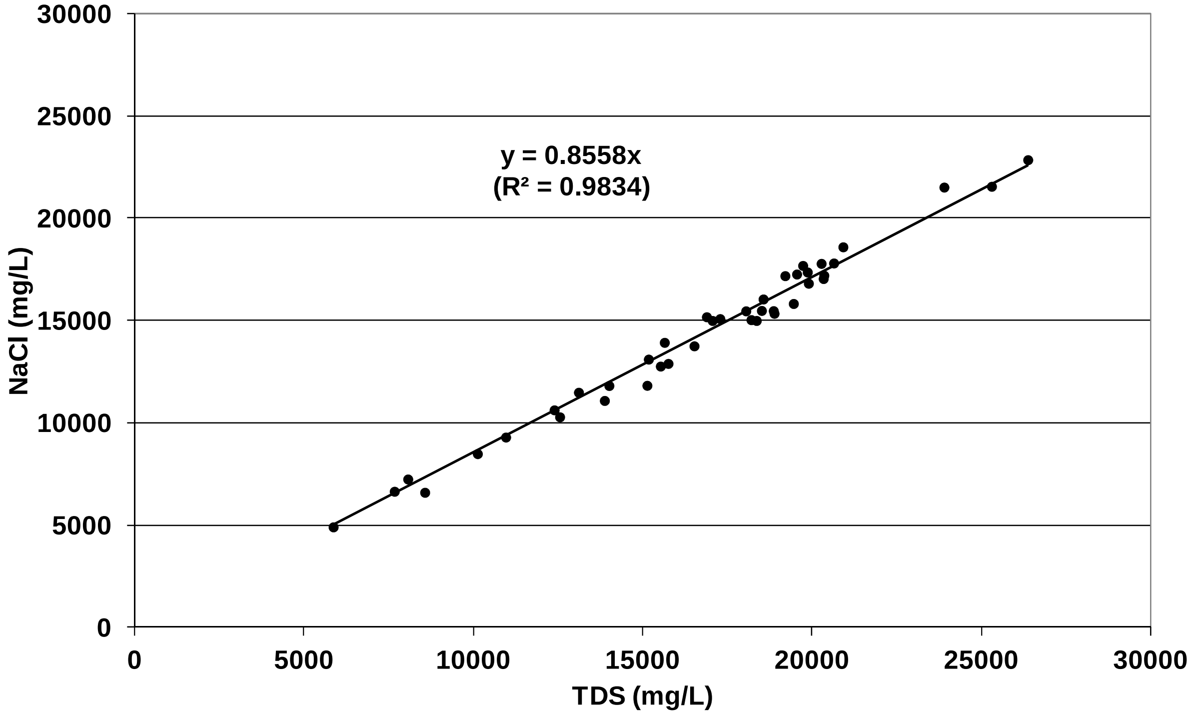

Figure B3 **Relationship between Total Dissolved Solids (TDS) and NaCl**

Appendix C: Surface water EC, TDS and Cl relationships

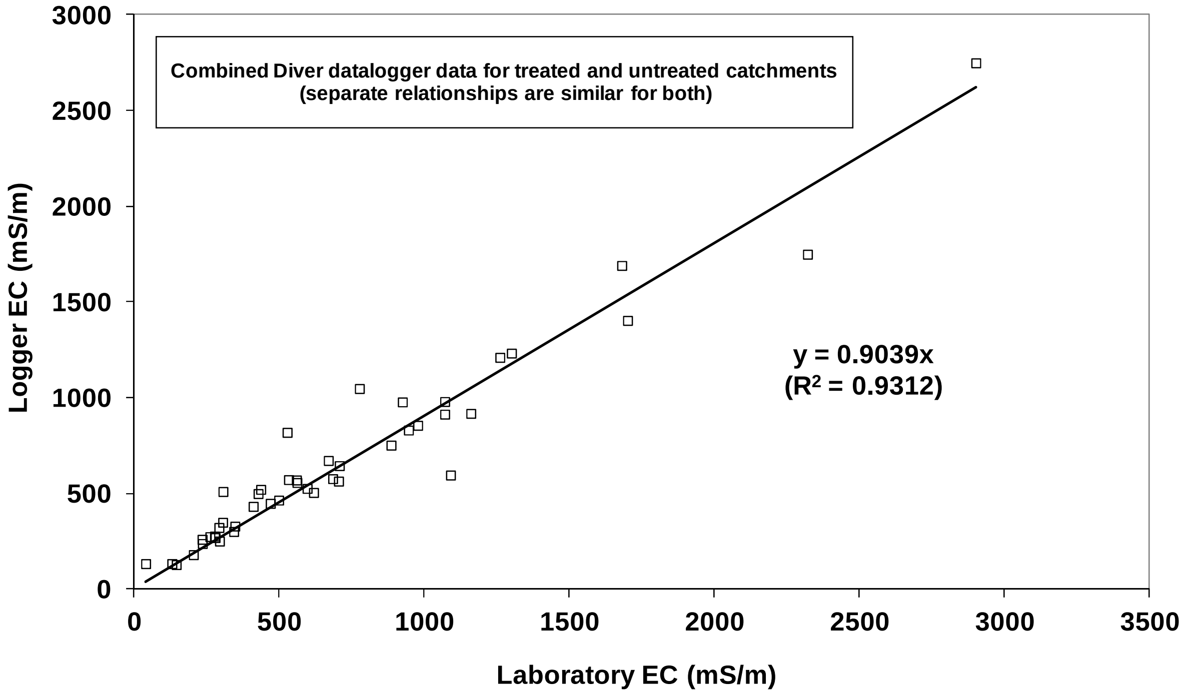

Figure C1 **Relationship between laboratory and logger determined Electrical Conductivity (EC) of surface water**

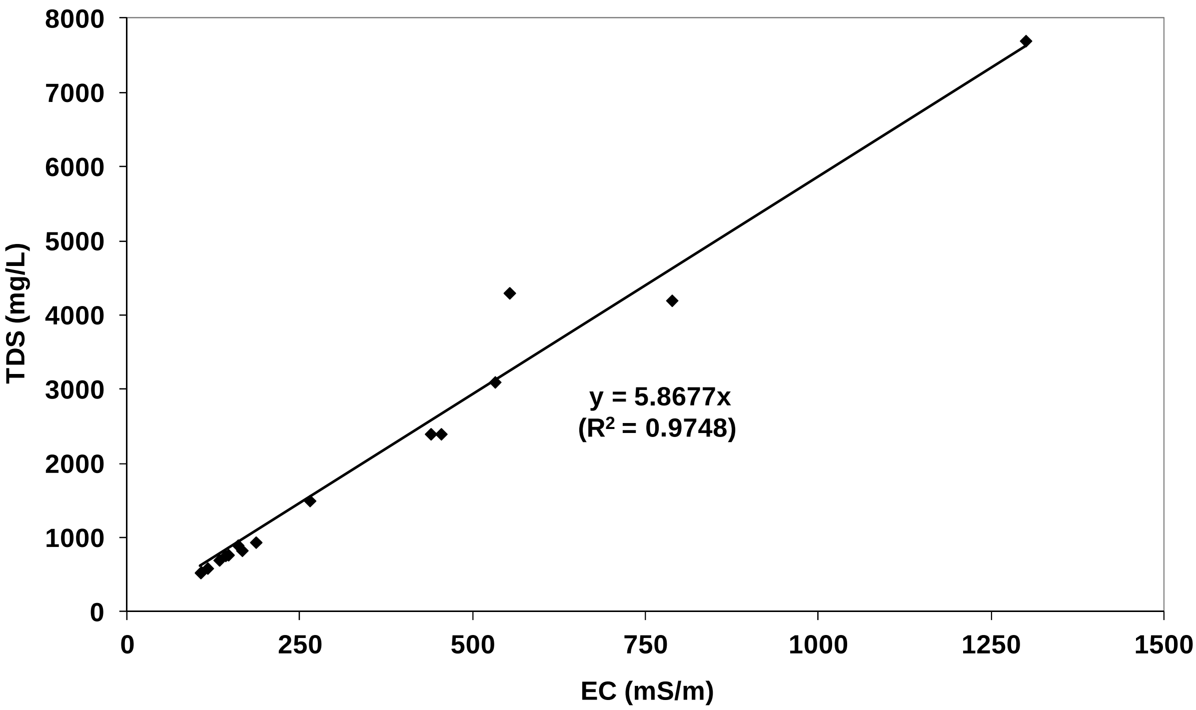

Figure C2 **Relationship between Electrical Conductivity (EC) and Total Dissolved Solids (TDS)**

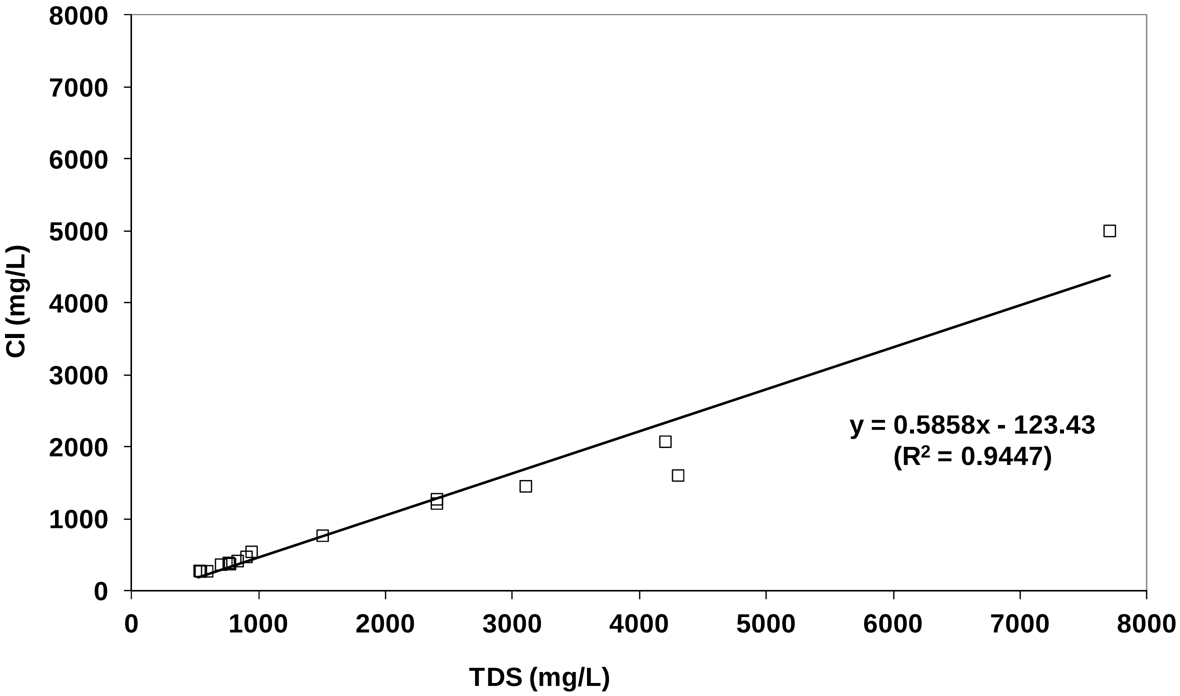

Figure C3 **Relationship between Total Dissolved Solids (TDS) and Chloride (Cl) of surface water**

Appendix D: Groundwater EC time series

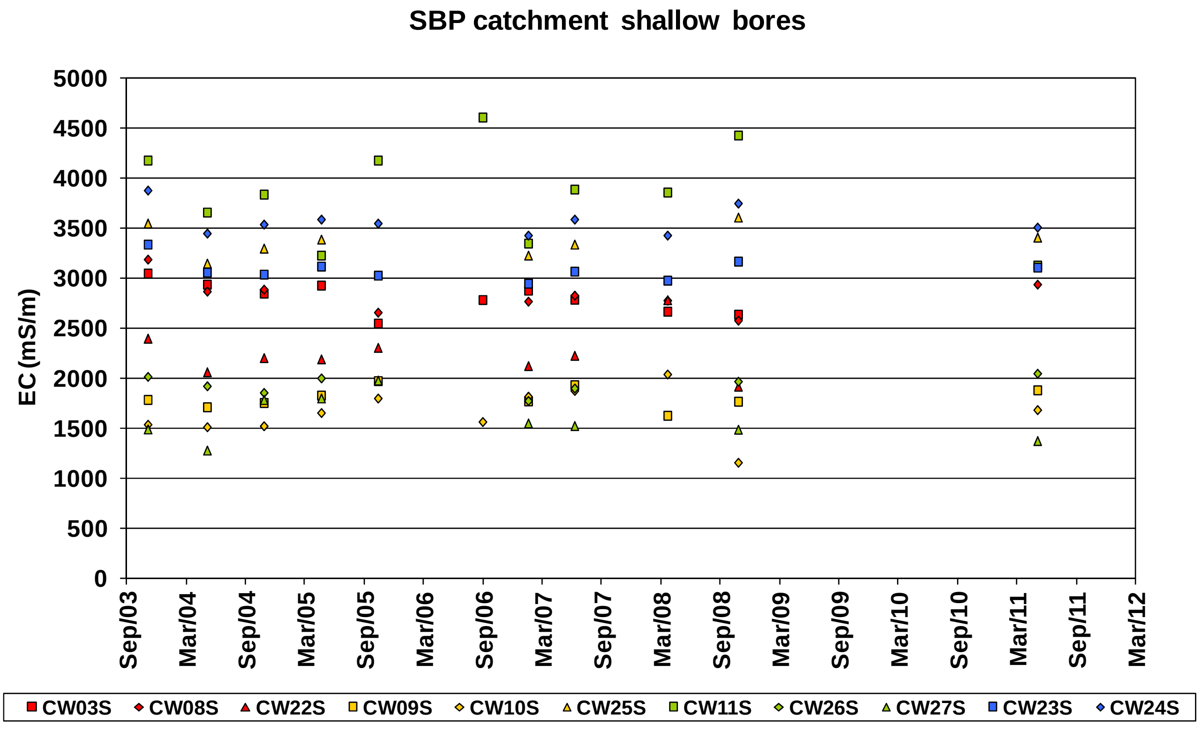

Figure D1 **EC of groundwater sampled from shallow bores located in the saltbush (SBP) catchment**

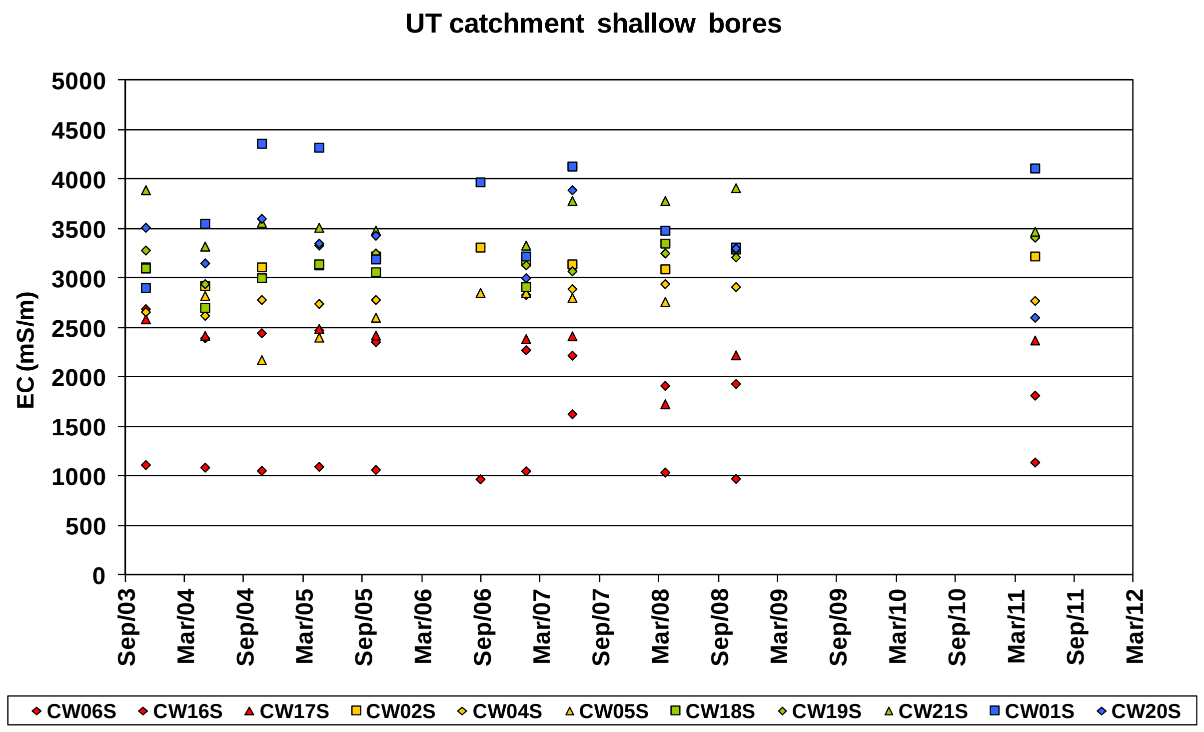

Figure D2 **EC of groundwater sampled from shallow bores located in the untreated (UT) catchment**

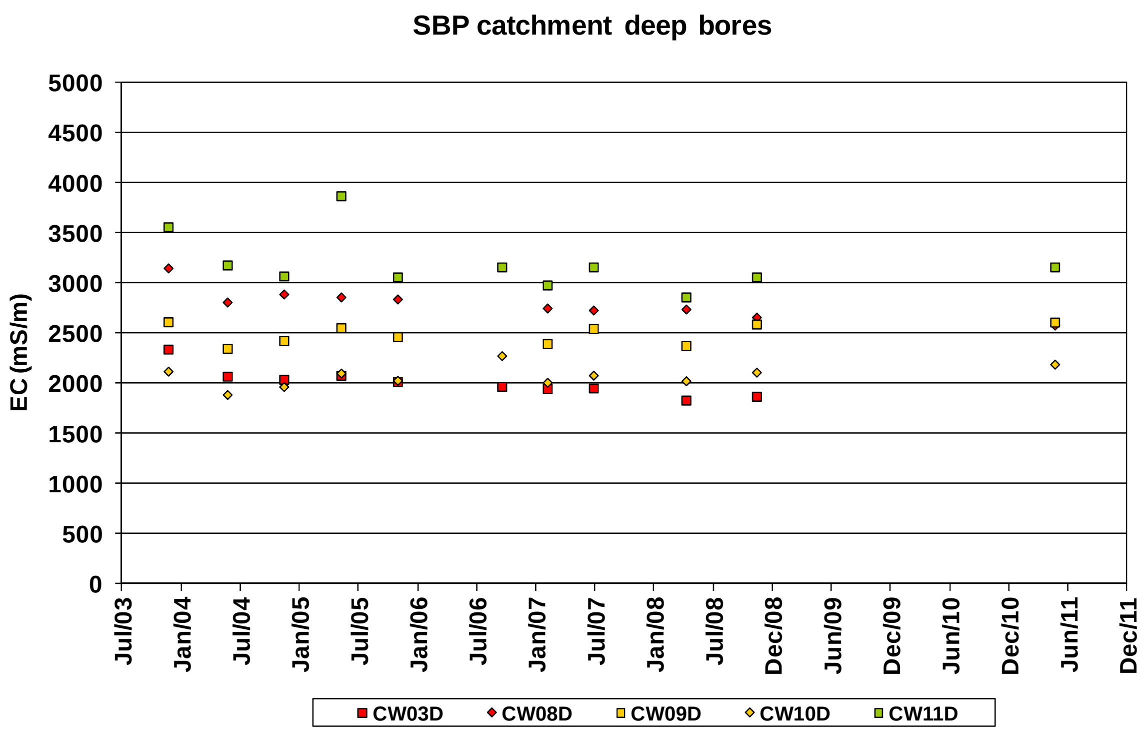

Figure D3 **EC of groundwater sampled from shallow bores located in the saltbush (SBP) catchment**

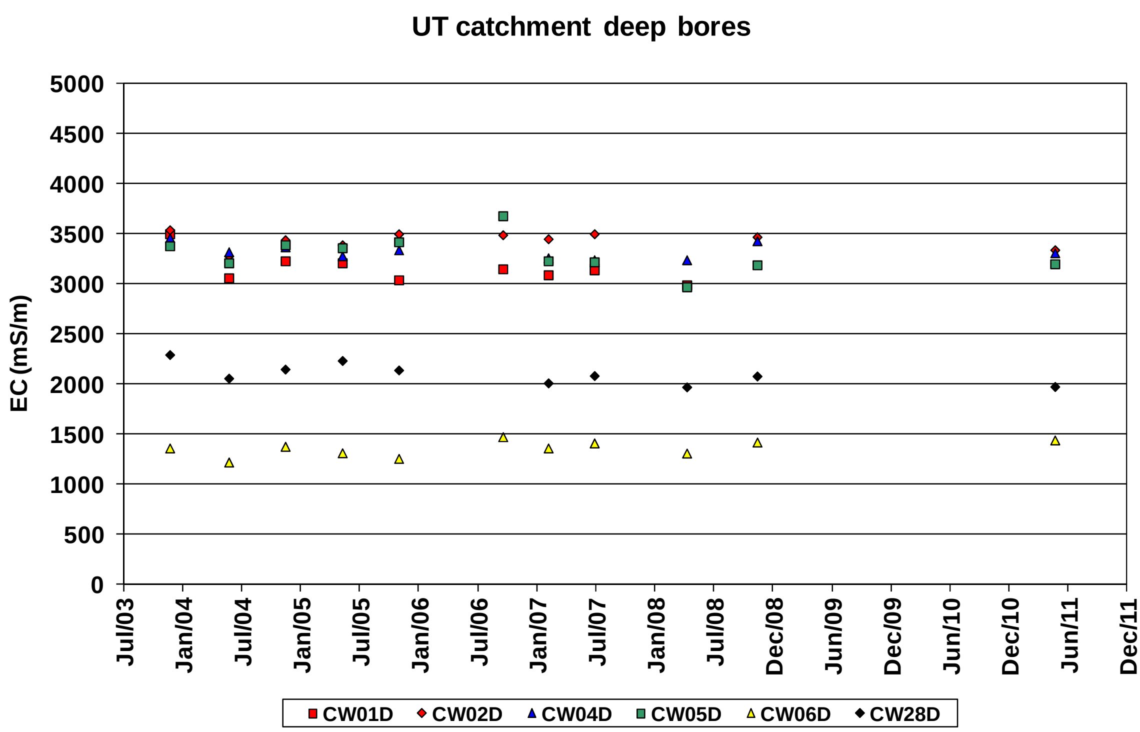

Figure D4 **EC of groundwater sampled from shallow bores located in the untreated (UT) catchment**

Appendix E: Groundwater levels

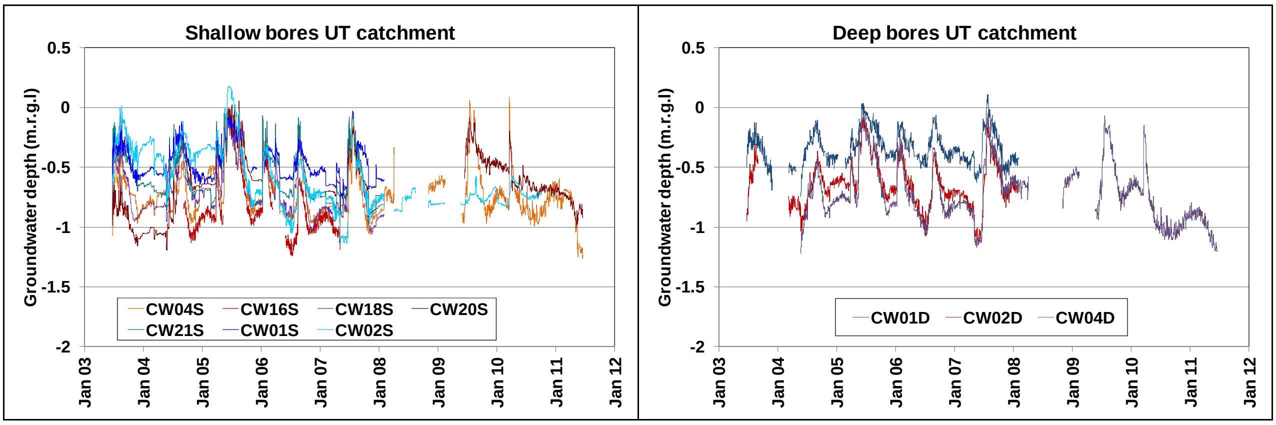

Figure E1 **Untreated (UT) catchment groundwater levels in deep and shallow bores recorded using loggers, relative to ground level (r.g.l.)**

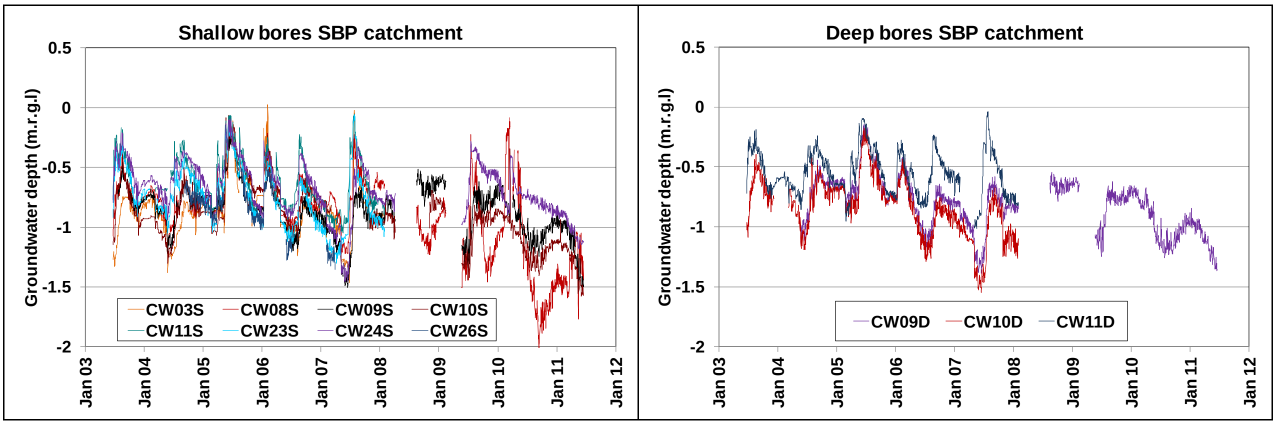

Figure E2 **Saltbush (SBP) catchment groundwater levels in deep and shallow bores recorded using loggers, relative to ground level (r.g.l.)**