5 Groundwater level response to land use change and the implications for salinity management in the West Moorabool River catchment, Victoria, Australia.

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5.1 Abstract

The connection between the removal of native vegetation, rising water tables and increasing stream salinity has been established for many catchments across Australia. However, the West Moorabool River in south west Victoria is an example of a catchment where there has been little discernable effect on groundwater levels following land clearing. Over the past 150 years a significant portion of the catchment has been cleared of dense forest for agricultural development. Historic standing water-level records from 1870-1871 and 1881 are compared with contemporary measurements (1970s to 2007) recorded in the government bore databases. The data show that the earliest recorded groundwater levels are well within the seasonal range of values observed today. By integrating geology and hydrogeology with historical observations of groundwater levels, climate data and land use, the contemporary field observations of stream salinity are linked to the changed water use and shift in rainfall. In contrast to the normally accepted axiom, reafforestation as a management strategy to mitigate the rising salinity in the West Moorabool River catchment would seem inappropriate.

Keywords: salinization, land use, environmental history, Australia.

5.2 Introduction

The salinisation of land and water resources is often related to land clearing and replacement of native trees with shallow-rooted crops (Ghassemi et al. 1995). This widespread change in land use is associated with increased recharge and raised water tables, resulting in salt being mobilised to the soil surface or transported laterally to watercourses. From the late 1980s, management of the increasing stream salinity in Australia has focussed on revegetation to lower the groundwater levels (Schofield et al. 1991). There are some examples where this strategy has been successful, such as Mundaring in Western Australia (Hatton 2003) and Pine Creek in Victoria (Zhang et al. 2007), although there is a growing awareness of the impact of reafforestation on reducing the surface water runoff component of stream flow, potentially resulting in increased salinity where saline groundwater baseflow occurs (Walker et al. 2007). The salinisation problem in Australia, has resulted in the Federal and State Governments investing around

AUD$1.4 billion in the management of salinity through the National Action Plan for Salinity and Water Quality (2000 – 2007) (CoAG 2000). Much of this investment in salinity management is based on replacing the native vegetation to reduce groundwater recharge, and lower the shallow water tables.

In Australian studies of stream salinity, the link between the cause (land clearing) and the effect (development of salinity) was initially derived from observations in Western Australian landscapes (Wood 1924). The quantification of this link was subsequently shown in a study of five catchments in the Collie River Basin, where the catchments that were cleared experienced substantial rises in water tables compared to those that were not cleared (Peck and Williamson 1987). This rise in groundwater and onset of salinity is related to the change in the output to input ratio of water and salt before and after clearing (Williamson et al. 1987; Jolly et al. 2001). The streams in the cleared catchments experience increases in salinity, although the degree to which this occurs depends on the annual rainfall, salt storage, groundwater hydrology, proportion of the catchment cleared and the environmental history (Schofield and Ruprecht 1989). The hydrologic response model assumes that following the clearing of the native vegetation, the rising groundwater levels increase discharge into streams, which in turn increases salt output. As the salt is leached and exported from the catchment, a new equilibrium is ultimately reached (Jolly et al. 2001).

This paper describes an example to the contrary, by examining the link between land clearing, groundwater levels and increasing salinity in the West Moorabool River catchment, Victoria. In Australian hydrogeological studies, historical data of groundwater levels from more than a century past are rare and therefore not often considered in the modern observations relating causes to the effects. In this paper, such historic data are used to argue that despite the dramatic changes in land use over the past 150 years, there is little discernable effect on the groundwater levels in this catchment. It is argued therefore that contemporary increases in river salinity are the result of causes other than rising water tables related to land-use change.

5.3 Background

The West Moorabool River is a major tributary of the Moorabool River, which is regarded as one of the most stressed rivers in Victoria. The Index of Stream Condition for the reaches of the West Moorabool River system rates it as being in very poor and poor condition (DSE 2005). The poor condition of the river and the water quality can be related to the diminution of flow as water is harvested and exported to other river basins.

The West Moorabool River catchment occupies 33,760 hectares (ha) of the Moorabool River basin. The northern boundary is the Great Dividing Range, which forms the watershed between the Murray-Darling Drainage Division and the South East Coast Drainage Division. The eastern boundary forms the drainage divide with the East Moorabool River and the west boundary with the Yarrowee River. The end of valley is the confluence of the west and east branches of the Moorabool River just to the north of Morrisons (Figure 5.1). Elevations range from around 740 m AHD (Australian Height Datum) on the highest volcanic cones and northern dividing range, to 300 m AHD at the end of valley. The northern third of the catchment generally comprises gently dissected volcanic plains and eruption points that constitute the most valuable agricultural land. Gently undulating landscapes formed on granites and sedimentary rocks of Palaeozoic age, surrounded by Quaternary volcanic plains and eruption points, make up the central third of the catchment. The southern third of the catchment comprises more dissected landscapes on Palaeozoic sedimentary and metamorphic rocks.

Climate is temperate, with cool winters and hot summers, with the greatest rainfall in August and September. The mean annual rainfall at Scotsburn, measured since 1857, is 777 mm, with a maximum of 1123 mm and a minimum of 426 mm (station #87046; BoM 2009c). The average monthly rainfall exceeds the average monthly pan evaporation from April to October, measured at the Moorabool Reservoir (station #87045; BoM 2009c), where the mean annual rainfall measured since 1913 is 940 mm (maximum 1415mm; minimum 498 mm). Mean annual aerial actual evapotranspiration for the catchment is approximately 605 mm, and the mean annual aerial potential evapotranspiration is 1014 mm (BoM 2002).

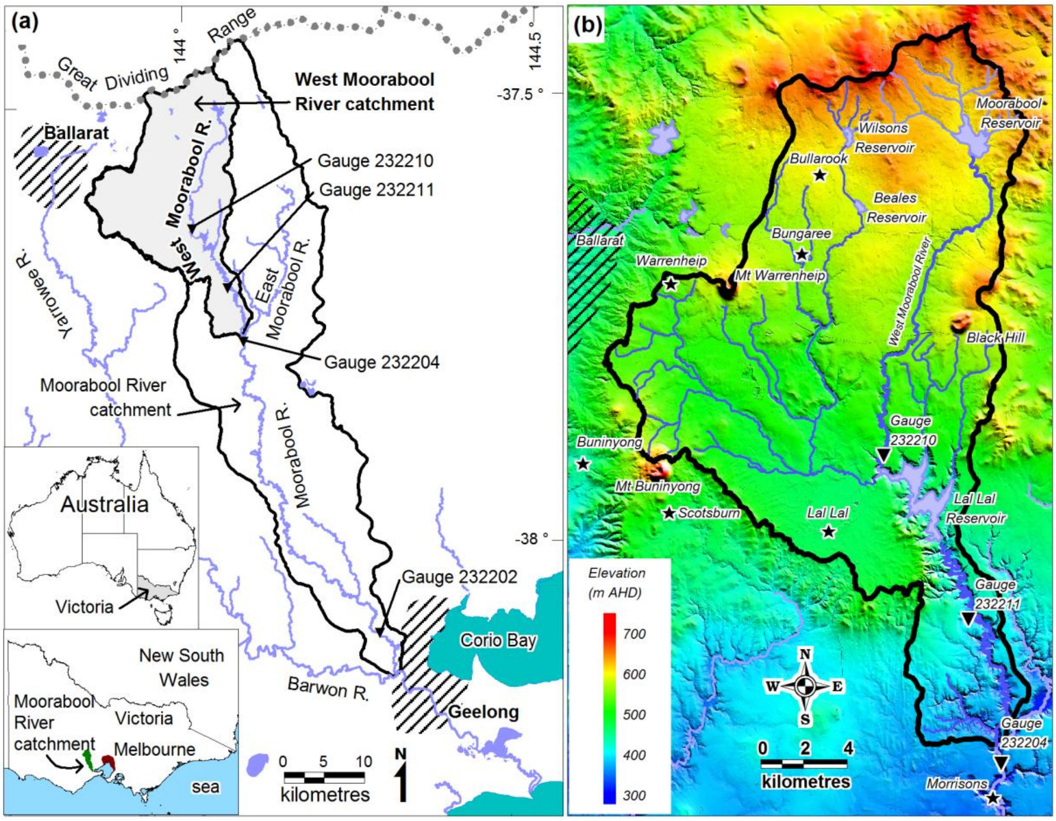


Figure 5.1 (a) Location and (b) physiography of the West Moorabool River catchment. Gauge locations are shown by the inverted triangles, town locations are indicated by the stars and the black lines show the extent of the urban areas of the cities.

5.3.1 *Hydrogeological setting*

The geology and hydrogeology of the West Moorabool River catchment has been described by Evans (2006) who collated the existing geological mapping, geophysical surveys (gravity field data and airborne magnetic data) and the drilling records of approximately 1850 bores (985 with lithological information). This work identified four basic components of the hydrogeological framework: the basement aquifer, the Tertiary sediment aquifers, the deep lead aquifers, and the Newer Volcanic aquifers (Figure 5.2).

The basement aquifer comprises Palaeozoic rocks consisting of Lower Ordovician sedimentary rocks that have been folded, faulted, and regionally metamorphosed; and Late Devonian granitic intrusives that underlie the central and northern portion of the catchment. The groundwater in the fractured rocks of the basement aquifer ranges from 600 – 3000 mg/L total dissolved solids (TDS), with groundwaters sampled from metasediments having a higher average TDS (1300 mg/L) than groundwaters sampled from the granite (900 mg/L).

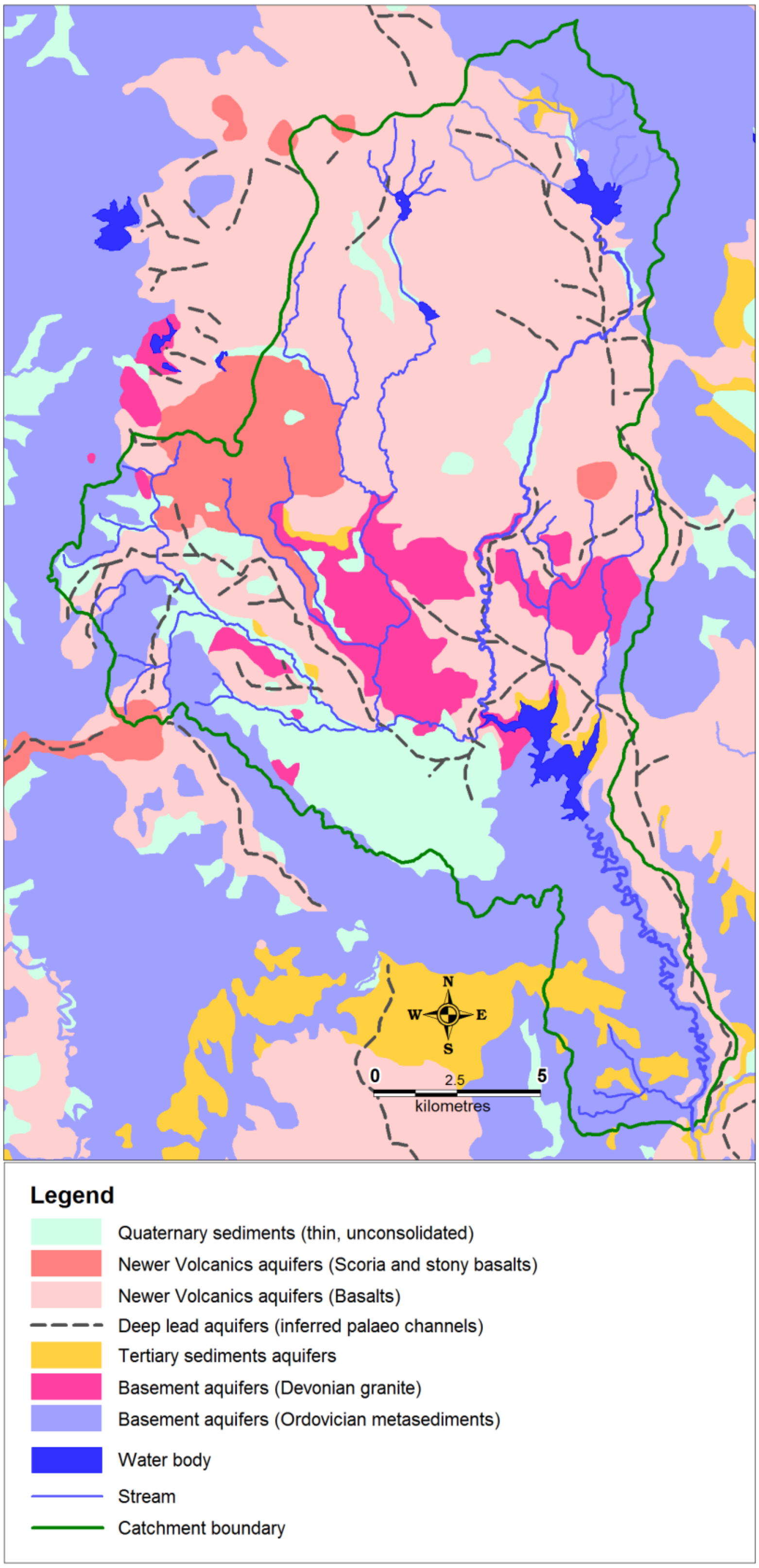


Figure 5.2 Hydrogeology framework of the West Moorabool River catchment. Based on Evans (2006) and GSV (2008).

The properties of Tertiary sediment aquifers are relatively poorly known as they have not been exploited as a water resource. At depth, sediments up to 220 m thick were locally deposited in two graben-like basins bounded by north-west trending faults in the basement rocks. At the surface, remnants of Palaeogene sediments cap some ridges in the catchment as sheet-like deposits of silicified and ferruginised quartz gravels. These gravels form sporadic, thin, unconfined to confined aquifers with local flow systems and water quality varying from 1000 to 10,000 mg/L TDS.

The deep leads represent a palaeo-drainage system filled with a basal layer of sands and gravels overlain by clays, silts and sands (Evans 2006) that developed during the late Eocene through to early Pliocene (Taylor and Gentle 2002; Holdgate et al. 2006). These channels have been subsequently covered by the volcanic rocks to form confined to semi-confined aquifers that have been locally developed primarily for agricultural use, with TDS generally in the range 200 – 3000 mg/L. Based on the evidence from previous investigations, Evans (2006) speculated that recharge occurs via leakage from the overlying volcanics, especially in the vicinity of the scoria cones.

The Newer Volcanic aquifers comprise fractured basalts, scoria and pyroclastic sediments emplaced by volcanic eruptions during the Pliocene to the late Pleistocene (the Newer Volcanic Formation). The eruptive history has been previously documented (Yates 1954; Taylor et al. 1996; Evans 2006) and can be broadly delineated into two earlier basalt units that filled the palaeo-valleys and formed the basalt plains, and a later series that formed the eruption points and stony basalt landscapes. Evans (2006) distinguished the groundwater system in the deeper basalts (>40 m depth) that are confined by the pre-volcanic palaeo-valleys from that in the overlying extensive planar basalts. Water quality in these unconfined aquifers is generally less than 1500 mg/L TDS, but can be up to 3500 mg/L and they are widely exploited as an irrigation and urban water supply in the northern portion of the West Moorabool River catchment (the Bungaree Water Supply Protection Area). The Newer Volcanic aquifers are the main focus of this paper.

5.3.2 *Land-use changes*

Although Aboriginal occupation of the West Moorabool River catchment probably extends back to the late Pleistocene (McNiven et al. 1999) the earliest historical accounts of the indigenous landscape are from the time of the first European settlers (Clark 1990; Nathan 2007). The pastoral settlers arrived at Geelong in 1835 and explored the Moorabool River valley, establishing settlements in the West Moorabool River catchment by 1838 (Learmonth 1853). At that time, the northern third of the catchment was comprised of the heavily timbered Bullarook Forest; the central third was more moderately timbered; and the southern portion was a mixed eucalypt forest (Nathan 2004).

Tree clearance by the first settlers would not have been significant, but this changed following the discovery of gold at Buninyong in 1851 when the population of the catchment and nearby gold fields increased significantly. Ballarat, a city which emerged with the gold rush, supported around 1,000 miners in late 1851, expanding to over 30,000 miners by 1854 (Withers 1887). The majority of the mining occurred within 10 km of the western boundary of West Moorabool River catchment, which was heavily exploited for its timber and water, particularly from the late 1850s onwards when the mines proceeded deeper underground.

The transformation of the northern West Moorabool River catchment from largely forested to cleared land occurred primarily over a thirty year period. By 1880 much of Bullarook Forest had been milled for timber and the land was then made available as small agricultural holdings (Nathan 2004; 2007). With reliable rainfall and fertile soils, this agricultural landscape has been used for potato cropping, with some dairying and sheep and cattle grazing, to the present day. The central portion of the catchment remained essentially intact as a lightly-treed pastoral unit and is still used for grazing and light cropping. The southern portion of the catchment ─ the Lal Lal Forest ─ was much thinned once mining activity declined around the time of the First World War (1914-1918). Most of this area remains forested, with some cleared for grazing and peri-urban development.

Four urban water supply reservoirs have been constructed in the catchment - Beales (415 ML; 1864); Wilsons (1013 ML; 1890); Moorabool (6737 ML; 1914);

and Lal Lal (59549 ML; 1972). The associated water reserves comprise about 1200 ha of treed and commercially forested land adjacent to the waterways in the north part of the catchment. The vast majority of the stored water is diverted to urban communities outside of the catchment (CHW 2007).

5.4 Methods

5.4.1 Surface water salinity trends

The single asset identified as threatened by salinity in the West Moorabool River catchment is the main urban water storage – Lal Lal Reservoir (Nicholson et al. 2006). As a surrogate for salinity, electrical conductivity (EC) has been measured on grab samples collected monthly at the outlet tower in the reservoir by the urban water authority (Central Highlands Water) over the past two decades.

Stream flow and EC have been measured at four gauging stations in the whole of the Moorabool River catchment, which are part of the Victorian Water Quality Monitoring Network overseen by the Department of Sustainability and Environment. Two stations are located on the West Moorabool River (Figure 5.1, Table 5.1), the Lal Lal gauge (#232210) approximately 1km upstream of the reservoir; and the Mount Doran gauge (#232211) approximately 4km downstream of the reservoir. The remaining two are located on the main trunk of the Moorabool River: the Morrisons gauge (#232204) approximately 800m downstream of the junction of the West Moorabool River and the East Moorabool River; and the Batesford gauge (#232202) approximately 7km upstream of the confluence of the Moorabool and Barwon rivers.

A semi-parametric statistical method (Morton 1997) based on the Generalised Additive Model (GAM) approach (Hastie and Tibshirani 1990) was used to obtain stream salinity trends from the gauging station data. Corrections for flow volume and seasonal effects are implicit in the method, although the long-term climatic variations remain evident (Smitt et al. 2005). The additive regression technique calculates log EC (log µS/cm) using the variables: time (months), log flow (log ML/day) and sinusoidal seasonal terms. The analysis provides both linear trends for the entire period of data analysed with their standard errors (95% confidence level) adjusted for the degree of autocorrelation in the GAM analysis (Morton

1997) and non-linear trends plotted as the cubic spline-smoothed mean EC for each month. The autocorrelation of the residuals in the model used either ordinary least squares regression or first-order autoregressive time series modelling, to get the best model fit. The trends were calculated using all of the available EC data for each gauging station. High residual values identified any obvious erroneous flow or EC values which were removed (maximum of 5 per gauge) before recalculating the final trends.

5.4.2 Groundwater chemistry

Over 570 groundwater chemistry analyses from bores drilled for supply or monitoring and investigation programs are available in State Government databases and unpublished research reports and theses. These data, which date from the early 1970s onwards, include basic EC and pH measurements, and some analyses of major ions and other elements were collated into one database.

The chemistry records were matched to the aquifers in the basic hydrogeological framework of Evans (2006) using the available bore lithology and bore construction records. In cases where the screened intervals of the bores were not recorded or the bores were recorded as open hole below the upper cased interval (i.e. not screened), the aquifer was assigned on the basis of the total depth of the hole, the lithological log and interpolated thickness of the aquifer unit at the bore location.

Accuracy of some older analyses is unknown as the sampling methods, analytical techniques, and quality assurance were generally not recorded. Within these limitations, the accuracy of the analyses was checked by their ionic balance, and 198 samples with a ionic balance of ± 5% were deemed acceptable.

5.4.3 Early historical records of groundwater levels

The history of exploitation of the water resources of the West Moorabool River from the European settlement to the present situation of being an over-allocated resource has been documented by Nathan (2004; 2007). These historical treatises also document that groundwater springs were utilised by the European explorers and pastoral settlers and many are shown on the earliest maps and surveys, especially around the scoria cones such as Mt Buninyong and Mt Warrenheip. As documented by Nathan (2004) these earliest historic accounts also make reference to fern-tree gullies, ti-tree, swamps, marshes, etc. indicating the likely presence of groundwater discharge in the natural landscape.

Records of hydrogeological studies in the West Moorabool River catchment date to 1870 and early 1871, when F.M. Krause produced four geological maps, covering approximately 7,800 ha of the catchments of tributaries to the West Moorabool River, which were proposed as a water supply area for Ballarat. The maps were printed at a scale of 1:15,840 (four inches to the mile) and record 21 wells, 16 springs and one shaft in which groundwater was encountered (Krause 1870a; 1870b; 1871a; 1871b).

In annotations on the mapsheets and in notes in the map margins, Krause records observations about the groundwater discharge at the time. On Sheet No.5 SE, geologically surveyed on the 4th of March 1871, he notes numerous springs with permanent, abundant and pure spring water that could be utilised as a valuable and inexhaustible urban supply. Similarly, on Sheet No.3 SE, surveyed May13th 1870, Krause records an extensive annotation in the area just north of Bungaree, which states that the series of spring-fed swampy depressions are separated by low saddles with abundant ironstone pisoliths and ridges of ferricrete, the formation of which he attributes to the constant groundwater discharge.

A second survey of wells and springs was undertaken on November 9th 1881 by Ballarat Water Commissioner, Thompson, and associates (Nathan 2004) who recorded 17 springs, and the standing water level depths (SWL) and total depths of 70 wells in and adjacent to the upper West Moorabool catchment. The water-table depths in the wells varied from 0.9 m to 12.2 m, with an average of 3.2 m. Thompson's list records the property owner and occupier at the time of the survey, but does not map the exact location of the wells or springs. As with the first survey by Krause (1871a), the 1881 survey also noted the presence of numerous springs in the area south of Beales Reservoir, and suggests that they could be connected by cutting trenches to enhance Ballarat's water supply.

A geological report on the Ballarat water supply by Dunn (1888) also noted that the wells sunk in the basalt in and around the upper West Moorabool water reserve yielded a supply of good quality water, with seasonally fluctuating water tables generally less than 10 m deep. Dunn's report discusses the sources and potential sources of water supply and notes the permanent shallow springs and constant discharge of iron-rich groundwater from basalts. However, he notes that the yield of the spring discharge has diminished since the clearing of the forest from the water reserve and adjacent lands, and speculates that there is little potential for deeper sources of water supply, based on the observed absence of springs in the underlying Palaeozoic rocks.

All the historical wells and springs are located in the Newer Volcanics aquifers and some assumptions have been made about this data. The 1870-1871 data recorded 21 wells but only four SWL measurements. For wells without an SWL recorded it is assumed that all intersected the groundwater (and Krause's map annotations suggested that they did), and therefore the total depth of the well was taken as the SWL. This assumption would overestimate the depth to water table by the depth of water in the well (estimated as < 3 m). For both the 1870-71 data and the 1881 data it is not recorded if the depth to water table was measured from the ground surface or the lip of the well structure, but as the height of the well structure was not recorded, it is assumed as depth below the ground surface. However, if the assumption is incorrect, this will overestimate the depth to groundwater by the height of the well structure (estimated < 1.5 m).

5.4.4 *Contemporary records of groundwater levels*

Prior to the introduction of the Groundwater Act in 1969, the only groundwater data held in public records are those from sporadic government drilling programs for geological investigations. However, from 1970 onwards, the SWL of bores was recorded (at the time of bore construction) on the government bore databases. With the introduction of groundwater management in the area from 1994 onwards, 28 State Government observation bores were constructed in the West Moorabool River catchment and are regularly monitored for resource management by government appointed monitors. In addition, seven monitoring bores were constructed in 2004 for a fast-rail corridor. In all cases the bores are monitored at 3-monthly intervals, with some at monthly intervals.

To compare the historical and contemporary groundwater levels, the records of groundwater-level data measured in bores constructed in the volcanic aquifer in the same area (as the historic data) were chosen for the period from the early 1970s to 2007. The number of bores constructed during each decade varied, with a peak during the 1980s, when irrigated agriculture intensified. Since 2000 there have been fewer bores constructed in the area, but vastly more SWL data collected due to the regular monitoring of the government observation bores.

For the comparison, the relative elevation of the groundwater level (RWL) was used in preference to the standing depth to water table (SWL). This reduces the variations in SWL due to local topographic undulations of spatially varied data points for each time period. The RWL was calculated by subtracting the SWL from the ground elevation for each data point, taken from a 2007 digital elevation model accurate to ±0.5 m that was acquired using airborne Light Detection and Ranging (LIDAR) technology. The LIDAR data have been calibrated to the Australian Height Datum and quality assured by comparison to ground survey points (DSE 2008a). The contemporary RWL data were aggregated and averaged over a decade to smooth out the range of seasonal and climatic fluctuations of spatially disparate single point measurements taken at different times.

5.5 Results

5.5.1 *Surface water salinity trends*

The EC measurements of the water sampled at the outlet tower in the Lal Lal Reservoir show that salinity has risen from 390 μS/cm in July 1998 to over 850 μS/cm by October 2007 (Figure 5.3). The EC of the reservoir water has exceeded the upper limit of 'good water' under the Australian Drinking Water Guidelines (ADWG 2004) since June 2007. A linear fit of the data shows EC is increasing on average by 33 μS/cm/yr.

The GAM analysis of the EC measured at the four gauging stations along the Moorabool also show generally rising trends in salinity (Table 5.1). Of greatest interest is the Lal Lal gauge (#232210) immediately upstream of the reservoir, which has a calculated linear trend of 1.9±1.0 μS/cm/yr for the period from December 1976 to February 2005. The non-linear trend analysis (Figure 5.4) shows a significant rise in the past ten years. The salinity non-exceedence curve (Figure 5.5) shows that the EC values are mostly within the range of 100 to 700 µS/cm, with the EC exceeding these values on relatively few occasions (4% of the time). However, EC greater than the 90% exceedence value of 600 µS/cm have been recorded 28% of the time in the most recent years (February 1999 to February 2005). This includes five occasions when the values exceeded 1000 µS/cm, and the three highest values ever recorded: 1600 µS/cm (April and May 2003) & 1500 µS/cm (May 2004).

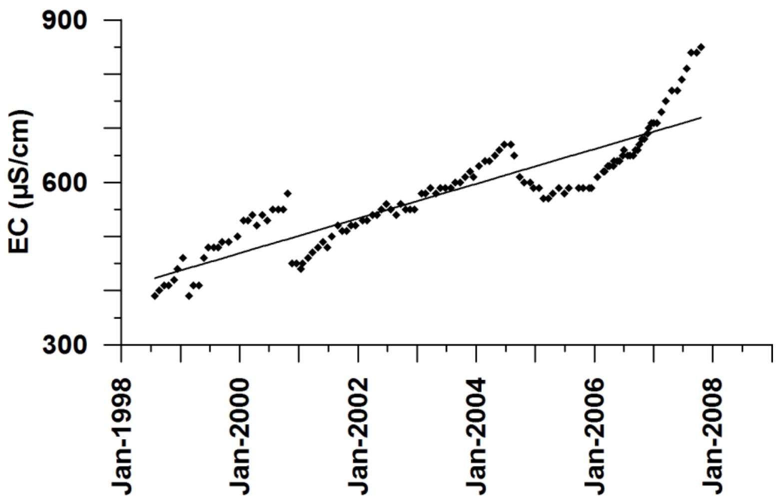


Figure 5.3 Salinity (EC) trends in Lal Lal Reservoir, 1998 - 2007

Table 5.1 EC trends from Generalised Additive Model (GAM) analysis at four gauging stations.

Gauge Number	Station name	EC Records		Mean salt load	Mean EC	Linear EC trend
		Start	End	tonnes/ day	µS/cm	µS/cm/yr
232210	Moorabool River West Branch @ Lal Lal	December 1976	February 2005	5	421	1.9 ± 1.0
232211	Moorabool River West Branch @ Mount Doran	November 1976	July 1990	21	473	10.9 ± 2.7
232204	Moorabool River @ Morrisons	November 1976	June 2001	44	643	-0.8 ± 2.6
232202	Moorabool River @ Batesford	November 1976	February 2005	81	1521	23.7 ± 6..0

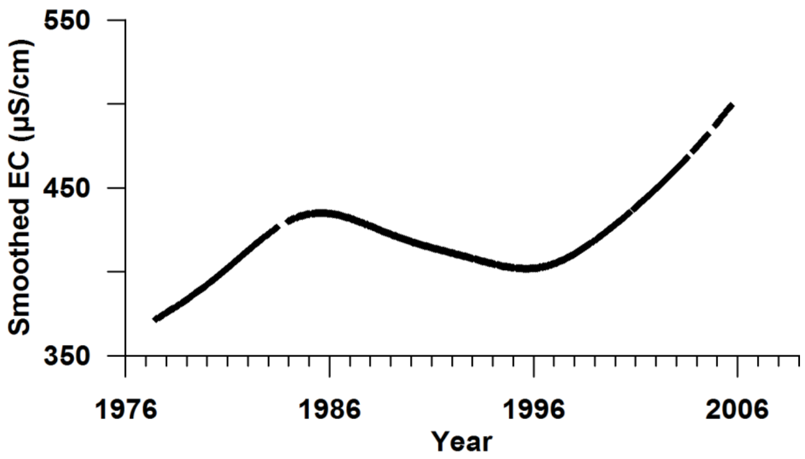


Figure 5.4 Smoothed salinity (EC) trends in the West Moorabool River, 1976 – 2005, adjusted for flow and season.

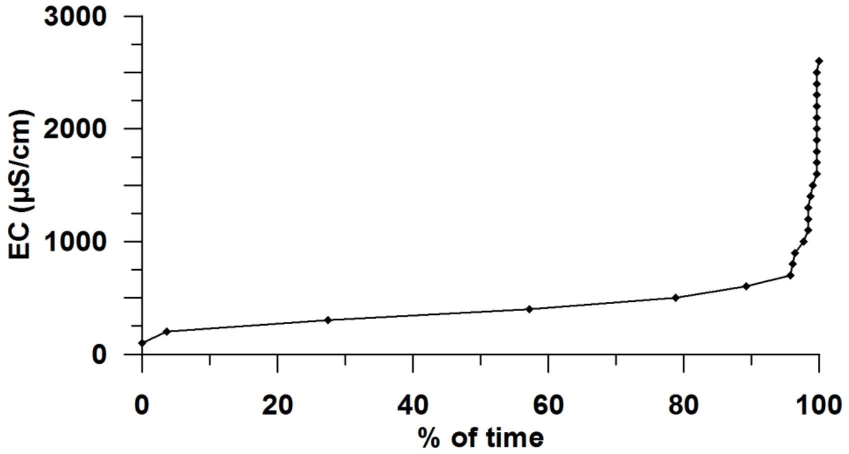


Figure 5.5 Salinity (EC) non-exceedence curve for the West Moorabool River, 1976-2005.

The EC data at the Mount Doran (#232211) gauge downstream of the reservoir has only been recorded for the period from November 1976 to July 1990, over which time the calculated linear trend is 10.9±2.7 µS/cm/yr. Further downstream below the junction of the east and west branches of the river, the trend at the Morrisons gauge (#232204) over the period from November 1976 to June 2001 appears to be falling slightly, although the 95% confidence value indicates that it is not statistically significant. The EC recorded at the Batesford gauge (#232202) a few kilometres from the end of the Moorabool River has a calculated linear trend of 23.7±6.0 µS/cm/yr for the period from November 1976 to February 2005, over which time the spline smoothed mean EC value has risen from 1218 to 2123 µS/cm.

5.5.2 Hydrogeology

The synthesis of previous investigations (Evans 2006) and the contemporary bore data indicate that the Newer Volcanic aquifers dominate the groundwater systems in the West Moorabool River catchment upstream of Lal Lal Reservoir, whereas the basement aquifer is more influential downstream of the reservoir. In the Newer Volcanic aquifers, porosity can be primary or secondary in nature, depending on the eruption and cooling history of the volcanic rocks. Hydraulic conductivity varies by orders of magnitude and may be heterogeneous and anisotropic at the local scale. Aquifer conditions range from unconfined to semi-confined with variable thicknesses of clay-rich regolith, interflow sediments, palaeosols and dense, massive basalts acting as sporadically distributed local-scale confining layers.

Based on the contours of RWL, flows in the upper Newer Volcanics aquifer appear to be controlled by topography (Figure 5.6). The overall direction of flow is from north to south and the divides in groundwater flow generally correspond to the surface water divides. Two flow fields are apparent in the upper aquifer within the boundaries of West Moorabool River catchment: the western field which generally corresponds to the Lal Lal Creek subcatchment; and the eastern field which generally conforms to the boundaries of the West Moorabool River subcatchment.

Representative bore hydrographs for the upper aquifer flow fields and the deeper (>40m) aquifer of the groundwater flow system in the Newer Volcanics are shown in Figure 5.7. The hydrographs show that the SWL is generally within 15 m of the ground surface and has a strong seasonal pattern. The amplitude of the fluctuations is typical of fractured rock aquifers and the pattern reflects groundwater recharge over winter - spring as well as the extraction of groundwater for the irrigation of summer crops. The drier than average climate over the past decade is reflected in the declining water levels, especially evident in the lower aquifer and the eastern flow field of the upper aquifer, but less obvious in the western flow field.

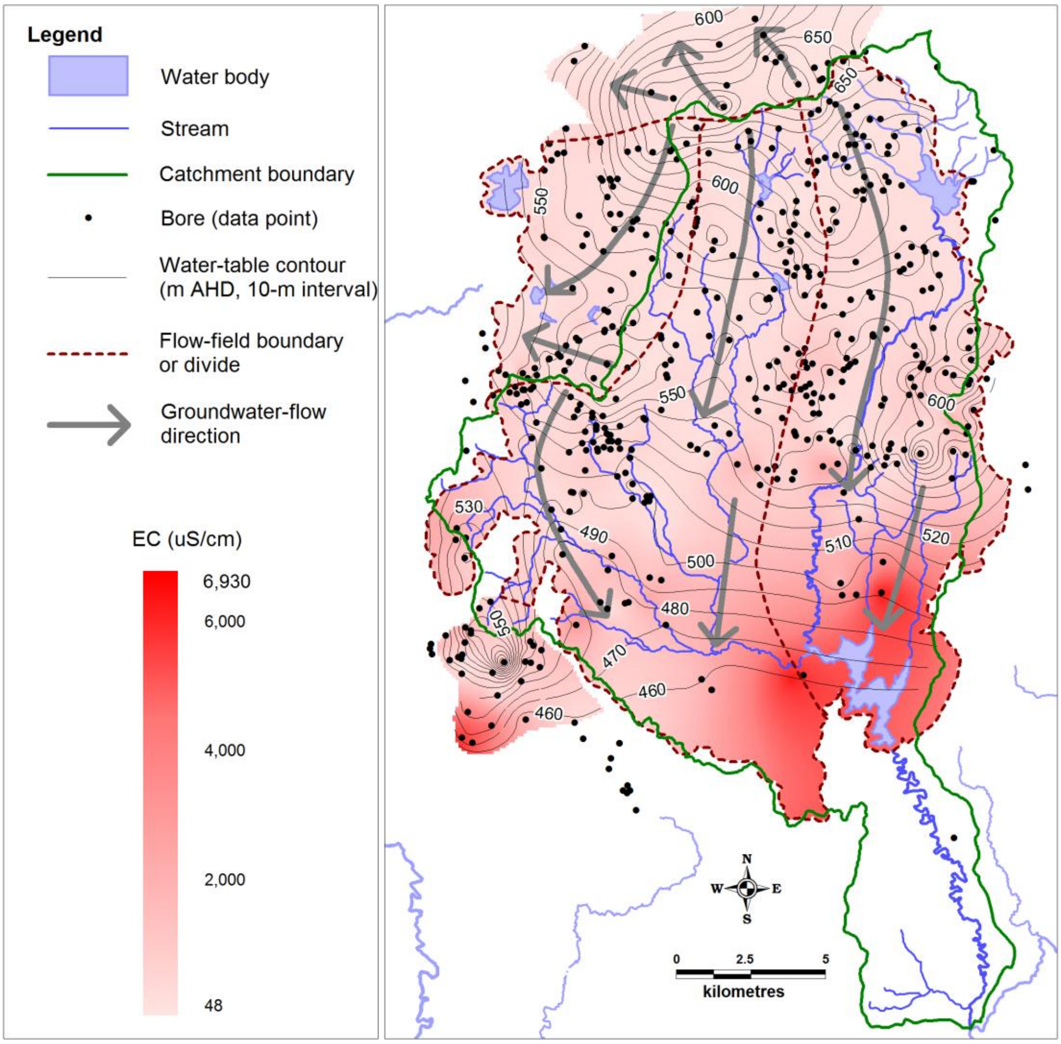


Figure 5.6 Groundwater flow and salinity of the upper Newer Volcanic aquifer (West Moorabool River catchment).

The hydrochemistry data of the four main aquifers are presented as a Piper Plot (Figure 5.8). Although available data are sparse in some areas, the major ion chemistry and the relatively similar ranges of pH and EC suggest connectivity between the deeper volcanic (>40 m) and shallow volcanic (<40 m) aquifers, with both dominated by low salinity Mg-Na-Ca-HCO_3-Cl type water. The relatively low salinity and presence of bicarbonate in the volcanic aquifers indicates a short groundwater residence time. Partial confirmation of this is provided in a study by Cox et al. (2007) which obtained modern groundwater dates (i.e. <1000 years old) using ^{14}C isotopes for five bores in volcanic aquifers in the West Moorabool catchment. Their study included the analysis of chlorofluorocarbons (CFCs) in two bores, both of which returned apparent ages of less than four decades. Based on the stable isotopic composition of the groundwater Cox et al. (2007) also concluded that rainfall is the main source of recharge to the groundwater in the volcanic aquifers of the wider region.

When considered with the bore hydrograph data, the hydrochemistry confirms that the volcanic aquifers of the West Moorabool River catchment are rapidly recharged by local rainfall in the wetter months when precipitation generally exceeds evaporation and irrigation is at a minimum.

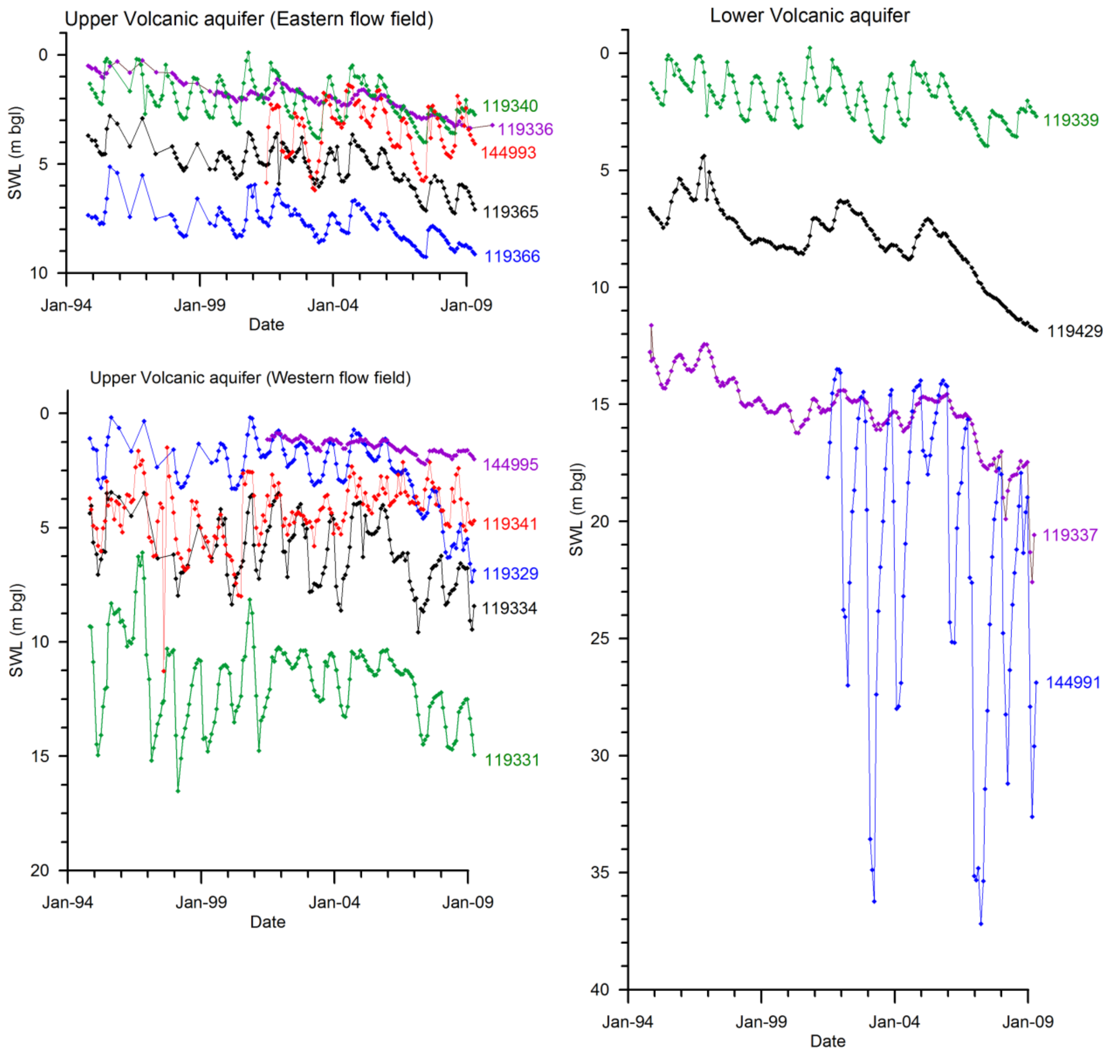


Figure 5.7 Representative monitoring bore hydrographs. SWL = standing water level depth, bgl = below ground level

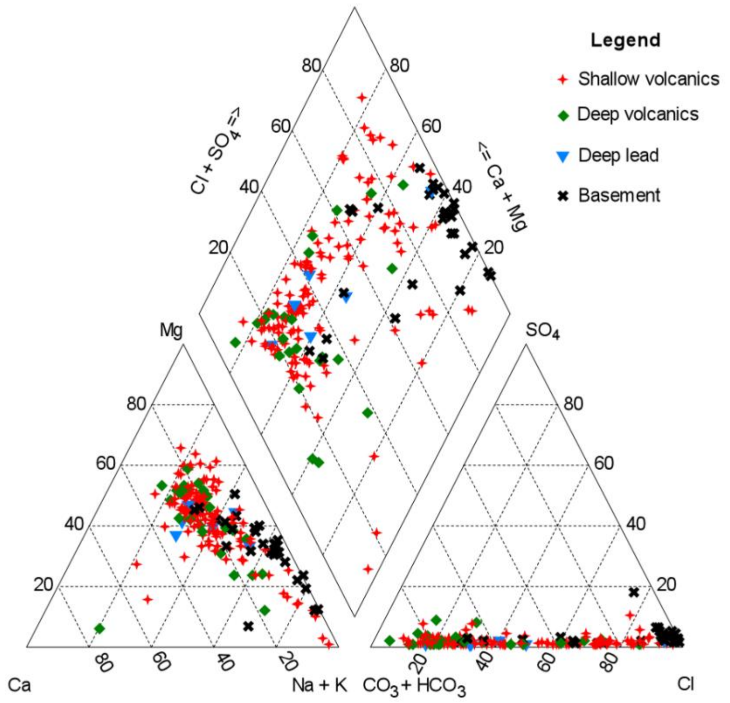


Figure 5.8 Piper Plot of groundwater chemistry

5.5.3 *Groundwater levels*

The locations of the historic wells and springs are shown in relation to the contemporary groundwater monitoring bores in Figure 5.9. An estimate has been made of where 70 of the 87 features listed in 1881 were likely to be located by reference to the Parish Plans of the time, in addition to consulting local knowledge, topography and geology. The comparison of RWL for the historic and contemporary water levels is presented in Table 5.2 and Figure 5.10.

The comparison has some inevitable limitations. The number and locations of the SWL measurements varies considerably for each time period, and the aggregation of the contemporary data includes the variations due to the seasons and pumping. However, within these limitations, the data show no substantial rise in groundwater levels from the earliest observations to the present day. The slightly lower RWL for the earliest records has the greatest uncertainty in the method of measurement and may be underestimated by up to 4.5 m, which would place it in the same range as those of 1881 and the 1980s. Alternatively, it may indicate a rapid groundwater response to the removal of vegetation, with a new equilibrium reached within a decade.

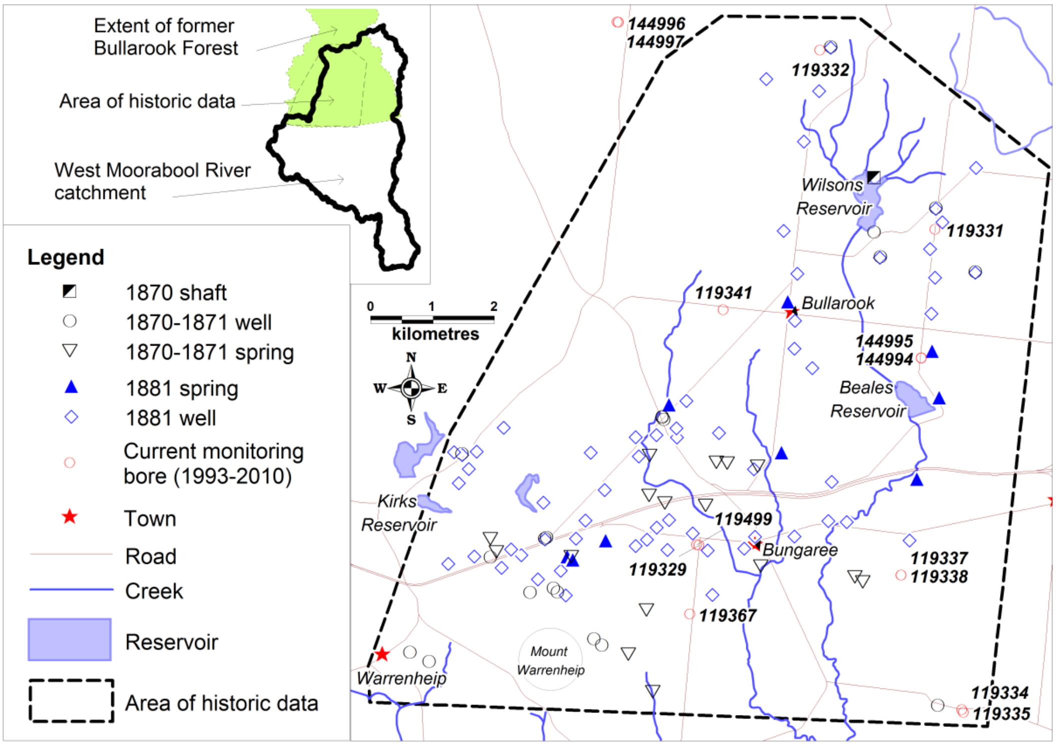


Figure 5.9 Location of historic and recent water-level data.

Table 5.2 Comparative statistics of historical and contemporary relative water level (RWL) data.

Date	No. of sites	No. of records	Median	Maximum	Minimum	75th percentile	25th percentile
1870 – 71	38	37	564.3	653.4	531.5	577.9	558.6
1881	87	70	568.8	678.8	535.1	591.9	559.1
1970-79	56	56	575.7	687.0	531.6	618.0	561.0
1980-89	85	85	568.3	635.2	511.7	584.5	554.2
1990-99	49	462	574.1	646.3	525.2	613.0	563.5
2000-09	18	1189	574.2	646.8	546.2	592.7	561.8

To investigate the variations in more detail, the contemporary hydrographs of the monitoring bores were compared to the nearest historic observations. Two monitoring bores are situated within a radius of less than 2 km from three or more historic springs or wells - bores 119329 and 119331. The RWL of the historic water table was interpolated from 1870-71 data and the 1881 data for the location of the monitoring bore using a minimum curvature algorithm, which applies a two-dimensional cubic spline function to fit a smooth surface to the historic RWL values. The results, which are graphically illustrated in Figure 5.11, show no consistency in the modern water table having either risen or fallen in relation to the historic 'snapshot' of groundwater levels.

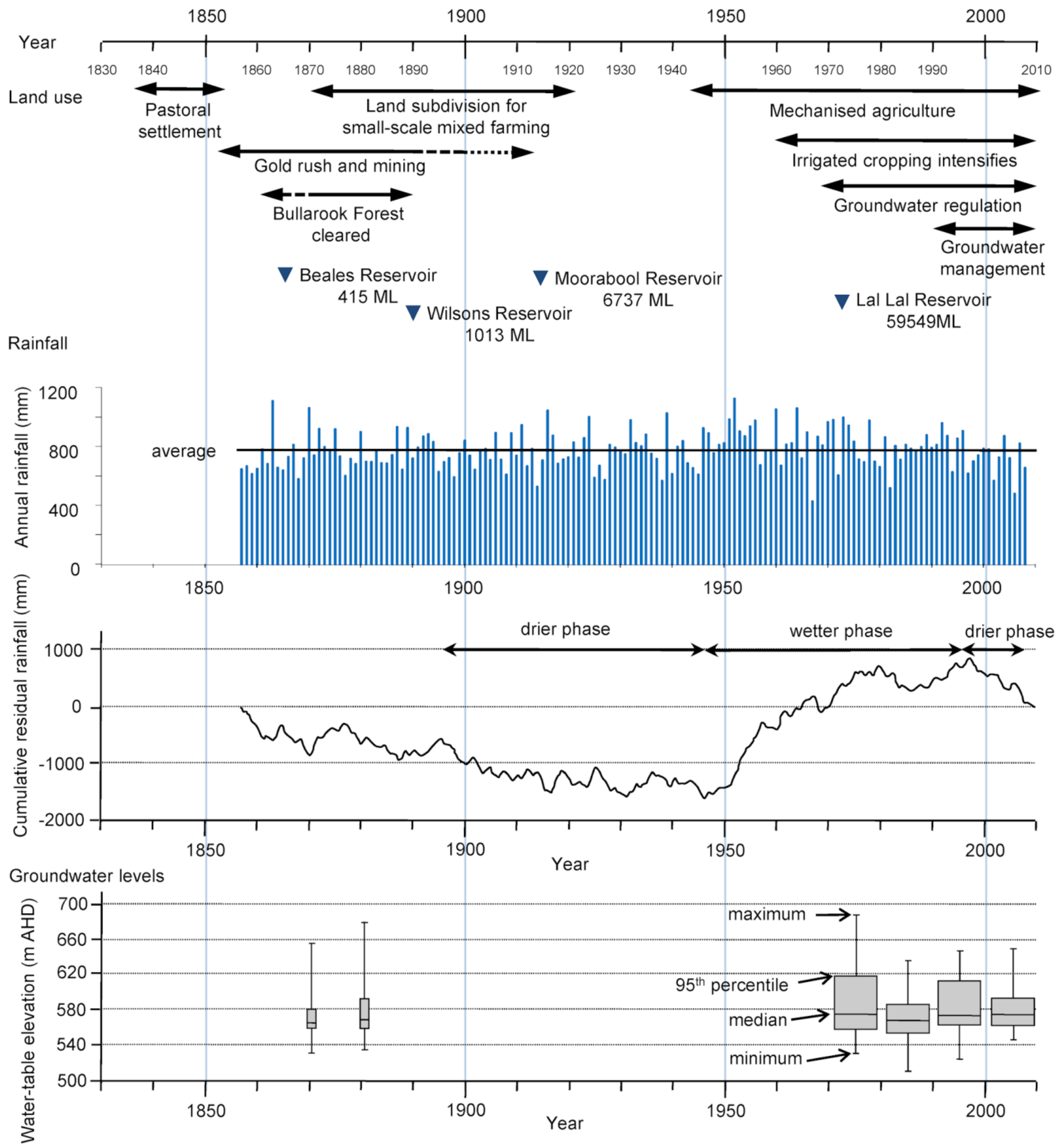
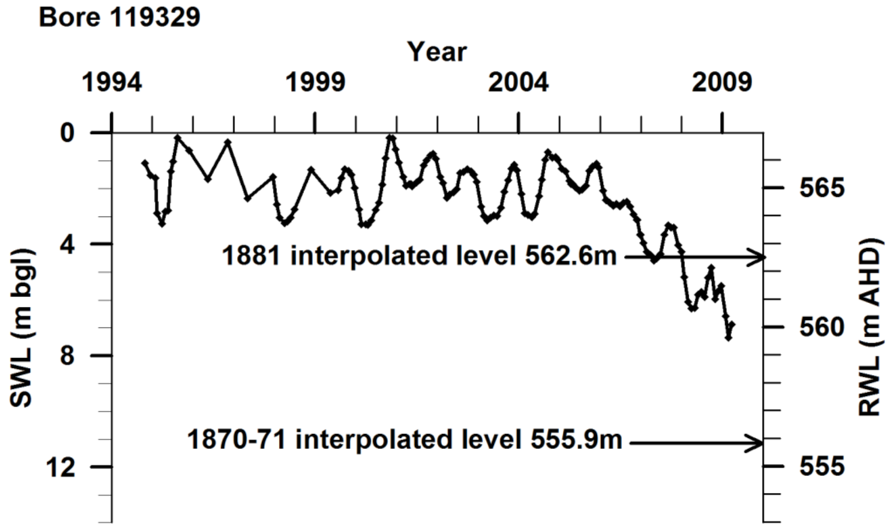


Figure 5.10 Timeline of land-use change, climate and groundwater levels.



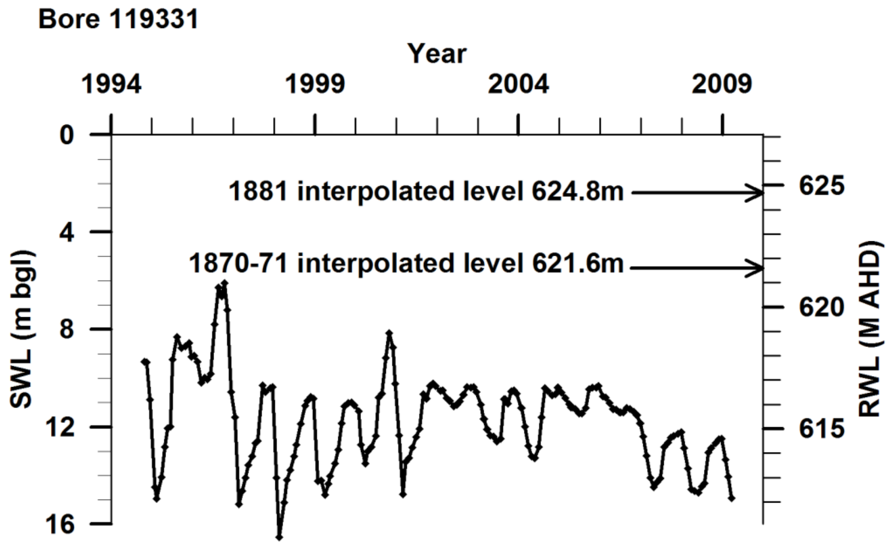


Figure 5.11 Interpolated historic water levels at two current monitoring sites. SWL = standing water level depth, RWL = relative water level (elevation)

5.6 Discussion

The rising salinity of Lal Lal Reservoir is an urgent issue for the asset manager since higher salinity water is a problem for industry and domestic users alike. Many industries, especially those utilising boilers, experience a significant rise in maintenance costs as the salinity increases.

The rising trend in salinity in the water-supply reservoir is partly due to the current prevalent drought conditions that since 1996 have reduced inflows by around 60% (compared to the period from 1974 to 1996), while increasing evaporation and water usage. This has concentrated the salts in the low water volume within the reservoir. However the rise is also reflected in salinity trend in the inflows from the West Moorabool River above the reservoir (i.e. the Lal Lal gauge; Figure 5.4) and the rising salinity trends along the length of the river.

According to the contemporary salinity management theory (e.g. Ghassemi et al. 1995; Jolly et al. 2001), the rise in salinity suggests that hydrologic equilibrium following land-use changes is yet to be achieved. One solution would be to reafforest the catchment, thereby lowering water tables and reducing the river salinity. However, this paper argues that when examined in the context of the historic data, there is no evidence that reafforestation would result in the desired effect.

The Bullarook Forest which occupied the northern third of the West Moorabool River catchment has undergone the greatest change in tree cover (Nathan 2004; 2007), summarised as follows:

- Aboriginal occupation (pre-European settlement): Dense forest cover of large trees with mosaic burning of undergrowth; perhaps more regular burning around the ti-treed swamps and springs.
- 1838 – 1855: Early pastoral land use with minimal tree clearance as there was ample acreage for stock to freely range from homesteads and outstations situated near freshwater, including the headwater wetlands such as Beales Swamp. Cultivated paddocks rarely exceeded 8 ha [20 acres].
- 1855 – 1880: Forest cover decreases substantially as portable timber mills move progressively from west to east to supply gold-mining demands; mining water races are constructed; subsistence farming commences (Residence and Cultivation Licences granted); land selection commences from 1865; Beales Reservoir constructed; swamps are drained; and a greater reliance on springs for domestic water supply.
- 1880 – 1920: Forest cover becomes light as the greater security of land tenure leads to more mechanised farming, initially with the crop combination of potatoes, oats, hay and wheat and moving to more mixed farming from the

1890s, with dairying and livestock rearing; farmers become more self-reliant in regards to water supply; Wilsons and Moorabool Reservoirs are constructed.

- 1920 – 1970: Farmlands dominate with the advent of tractors; increased pasture improvement; irrigated cultivation of potatoes; farm amalgamations; construction of winterfill dams; and increasing groundwater use.
- 1970 - present: Construction of Lal Lal Reservoir (1972); intensification of potato cropping; mechanisation and corporatisation of agriculture; larger farms; move from fresh market to contracts for food processing corporations causing increase in irrigation requirements; increase in number of bores and winterfill dams throughout the 1980s; declaration of groundwater management areas and urban water allocation supply entitlements during 1990s; and prolonged drought increases tension over groundwater and surface water allocations in 2000s.

By coincidence, the earliest historic records of depth to groundwater are in the northern portion of the West Moorabool catchment, where much of the Bullarook Forest was eventually cleared. The first records are those of Krause in the summer of 1870-71 at a time when the Bullarook Forest was more treed than cleared, with portable sawmills and some land selection creating a more open landscape (Nathan 2004). The 1881 record was taken when these land clearance processes had further progressed, although the reservoir reservation itself remained more treed than the surrounding forest remnants.

A comparison of groundwater levels against the timeline of land use change, water use change and climate, shows that the aquifers are remarkably robust. The change in rainfall over the past 150 years is provided by the cumulative residual rainfall (Figure 5.10) of the data from Scotsburn (station #87046; BoM 2009c). The period from 1896 to 1946 represents a drier phase, followed by a wetter phase from 1947 to 1996, when the current drier phase began. This cycle accords with those observed elsewhere in Australia, which have been implicated in an increase in recharge and salinity during the wetter phase (Rancic and Acworth 2008). However in the West Moorabool River catchment, the opposite provides a more logical explanation.

The historic data describe the presence of springs in the Bullarook Forest (Figure 5.9), some of which are shown on the earliest maps surveyed before those of Krause (Nathan 2004). It appears that the combination of high recharge rates and excess water available in winter months has been sufficient to maintain high water tables, even under forest cover. In addition, the observed hydrological response to the clearing of the forest was that the springs ceased to flow (Dunn 1888), suggesting a slight lowering of the groundwater levels. With the progressive development of the catchment (Figure 5.10), water use increased as the urban population grew and resulted in the increased harvesting and diversion of surface water out of the catchment (Nathan 2007). The shift to mechanised and irrigated agriculture also increased surface water harvesting in winter-fill farm dams (SKM 2004) and groundwater use for irrigating summer crops. The cumulative result of these water-use changes was that progressively less low-salinity water was available to sustain the stream flows, especially during the drier months.

Although the majority of these water-use changes occurred during the period from 1970 to 2000, their effect on stream salinity was less obvious during the wetter phase of increased rainfall (Figure 5.4; Figure 5.10). However as the rainfall shifted into a drier phase, the proportionally greater harvesting and export of low salinity surface water and groundwater from the catchment has become obvious in the declining stream flows, declining groundwater levels, and the increased stream salinity.

5.7 Conclusions

The connection between the removal of native vegetation, rising water tables and increasing stream salinity is well established for many catchments across Australia. However, the West Moorabool River in south west Victoria is an example of a catchment where widespread land clearing and changed land use has not had any discernable effect on water tables.

A logical explanation for the increasing stream salinity is that the combination of increased surface water harvesting, the export of both surface water and low-salinity groundwater from the catchment, and the shift to a drier climate has reduced the stream flows and proportionally increased the amount of saline baseflow.

Based on the historic evidence presented in this paper, reafforestation as a management strategy to mitigate the rising salinity in the West Moorabool River catchment would seem inappropriate.

5.8 Acknowledgements

The authors thank Chris Smitt for processing the stream gauging data; June and Roy Huggins, and Ron Trigg for assistance with locating the historical data; Central Highlands Water for the Lal Lal Reservoir EC data, and Southern Rural Water for additional bore monitoring data. The Corangamite Catchment Management Authority provided funding for the research. William Milne-Home and two anonymous reviewers are thanked for their contributions to improving the manuscript.

6 Accentuate the positive: a standard framework for salinity risk management

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6.1 Abstract

Although salinity is widely regarded as a significant geohazard within Australia, there is no nationally consistent approach to salinity risk management. Salinity risk assessment, prediction or management, is limited by the variety of meanings of "risk" in its popular usage. Salinity is viewed as a threat and rarely recognised as an asset in the conservation of biological diversity and the security of water for environmental purposes. This paper outlines a process for salinity risk assessment that has been developed for the Corangamite region of south west Victoria, Australia, one of the priority regions in the National Action Plan for Salinity and Water Quality. In keeping with international and national risk management standards, both the negative and positive impacts of salinity risk are considered in the broader context. The risk assessment considers both salinity as a threat to assets, and salinity as sustaining the region's most valuable environmental assets. This systematic, disciplined and rigorous approach to salinity risk management has been applied in statutory planning regulations developed in collaboration between catchment managers and local governments to specifically consider environmental values. The standard provides a logical and defendable framework for the assessment of salinity risk that can assist in statutory planning decisions to protect all classes of assets which are threatened by changes to salinity processes, even those where the salinity itself is the asset. In a time of hydrological and climate change, the adoption of a standard risk management framework based is both logical and timely.

Key words: salinity, risk, geohazards, urban planning, risk management, Corangamite.

6.2 Introduction

Land and water salinisation is a well-documented problem in Australia, North and South America, Africa and Asia (Abrol et al. 1988; Ghassemi et al. 1995). In Australia salinity is widely regarded as a significant geohazard, as shown by the federal and state government investment of AUD$1.4 billion in the National Action Plan for Salinity and Water Quality 2000 - 2007 (CoAG 2000). Its impact on all classes of assets: agricultural productivity, urban infrastructure, natural ecosystems, and water resource quality has been documented in a national Audit (NLWRA 2001) that estimated that 5.7 million hectares were considered at risk or affected by dryland salinity and predicted that it could rise to 17 million hectares by 2050.

Although salinity is a recognised geohazard in Australia, there is no nationally consistent approach to salinity risk assessment or risk management. This limitation may be partially due to the variety of meanings of "risk" in its popular usage. Although risk management terminology is defined by the International Organisation for Standardisation (ISO 2002) and the Australian/New Zealand Standard on Risk Management - ASNZS 4360:2004 (Standards Australia 2004a) the standard definitions are not strictly used in the scientific literature and government policies. In particular, 'risk', the standard definition of which is "*the chance of something happening that will have an impact upon objectives*", is often interchanged with 'hazard', which is "*a source of potential harm or a situation with a potential to cause loss*" (Standards Australia 2004a).

As an example, in the national Audit, salinity risk is defined as "*estimation of the expected amount of harm that will occur to the asset when a condition occurs*" and hazard as "*anything that can cause harm to an asset*" (NLWRA 2001). The Audit notes that the two are often used as equivalent terms in common language and that 'risk areas' are those where dryland salinity impacts from shallow groundwater are known or expected to occur. Similarly, in the State of Victoria's Salinity Management Framework (DNRE 2000), salinity risk is illustrated as a series of maps and tabulated as areas of land in which watertables of various depths are predicted. Confusingly the framework also contains maps of 'hazard land at risk of salinity' showing generally non-forested agricultural areas regarded as higher recharge landscapes.

The variety of methods used in Australia to assess salinity risk have been reviewed by Gilfedder and Walker (2001) who categorized these into: composite index methods, which assign weights (usually based on empirical judgment) to various data layers that are summed to produce a risk map; strongly inverse methods, which use the distribution of salinised areas to build decision trees or expert methods to assess the risk of further salinisation; and trend based methods, which use historical trends in land, stream or groundwater salinities to forecast future development of salinity in a region. They concluded that a suitable approach to risk assessment should include: landscape disaggregation to provide consistency in methods; salinity risk assessment based on a range of approaches to target key processes within a landscape element; and temporal change to allow predictions which can relate to the lead-time required for management intervention.

In the scientific literature, salinity risk is generally referred to as areas predicted to become saline (e.g. Bui 2000; Evans and Caccetta 2000; Petheram and Walker 2001; Peck and Hatton 2003; Grundy et al. 2007; Biggs et al. 2008; Smith 2008). Salinity hazard has been defined as low-energy long-duration chronic salinisation (Haw et al. 2000) and areas with a high salt store in the soil which are underlain by shallow watertables (e.g. Bui et al. 1996; Tickell 1997; Williams et al. 1997). The distinction between risk and hazard is generally given as: salinity hazard being the predisposition of a spatial location to salinisation; and salinity risk being the likelihood that a hazard will eventuate and impact on an asset at some defined place and time in the future (e.g. Bui 2000; Robins 2004; Lawrie 2005; Spies and Woodgate 2005). A consistent theme in the assessment of both salinity risk and hazard is their relationship to shallow groundwater tables.

This relationship of salinity to shallow watertables dates from the earliest scientific studies of salinity in Australia (e.g. Wood 1924; Teakle 1938; Holmes et al. 1939; Downes 1949). These studies observed that salt was stored in the landscape and that salinity had been induced by hydrologic imbalance following land clearing for agriculture. Henceforth scientific studies of salinity focused on the hydrologic balance and salt balance in various landscapes (e.g. Peck 1978; Dyson 1983; Peck et al. 1983; Peck and Williamson 1987; Williamson et al. 1987; Macumber 1991), with the majority of research focused on the Murray Darling Basin and the wheatbelt of south west Western Australia (Ghassemi et al. 1995). The cause (rising watertables) and effect (salinity) has become an axiom, with the National Natural Resource Management Monitoring and Evaluation Framework nominating the area of land threatened by shallow or rising water tables as the single indicator for land salinity (NRM Ministerial Council, 2003). Depth to watertable and time-series trends in groundwater levels have therefore been adopted as the indicators of salinity risk (Peterson et al. 2003; Peterson and Barnett 2004a; 2004b; Spies and Woodgate 2005).

Since the mid 1990s scepticism has emerged that rising water tables are the single cause of salinity in many of Australia's landscapes (e.g. Dahlhaus and MacEwan 1997; Nathan 1998; 1999; Dahlhaus et al. 2000; Acworth and Jankowski 2001; Jones 2001; Rengasamy 2002; Fawcett 2004; Wagner 2005; Bann and Field 2006; 2007; Dahlhaus et al. 2008a). Importantly, the basis for this scepticism arose from disciplines outside of hydrogeology, such as pedology (MacEwan and Dahlhaus 1996; MacEwan et al. 1996), environmental history (Nathan 2000); agricultural policy (Tunstall 2005; Wagner 2005), agronomy (Kreeb et al. 1995; Jones 2001), and biology (Bann and Field 2005). The evidence presented in these studies undermines the use of rising watertables as a universal indicator of salinity risk or hazard and suggests that other factors such as soil hydrology, environmental history and agronomic management may be more important in some landscapes.

The second and perhaps more important flaw in the current approach to salinity risk assessment is the assumption that salinity is always a threat. The international and national standards define risk as the combination of the probability of an event and its consequences, and note that risk may have a positive or negative impact (ISO 2002; Standards Australia 2004a). The potential for events and consequences to constitute not only threats to success but also opportunities for benefit is increasingly recognised by risk managers (IRM et al. 2002). However there are no known salinity risk assessments that consider salinity as having a positive impact, despite the fact that salinity processes sustain some of Australia's most highly rated environmental assets, such as wetlands listed under the Ramsar convention and international migratory bird treaties (Clifton and Evans 2001; EA 2001; Dahlhaus et al. 2008a).

The focus of salinity risk on the negative impacts has been reflected in the investment priorities for salinity management in Australia. The stated aims of the National Action Plan for Salinity and Water Quality 2000 - 2007 were to: 1) prevent, stabilise and reverse trends in dryland salinity affecting the sustainability of production, the conservation of biological diversity and the viability of our infrastructure; and 2) improve water quality and secure reliable allocations for human uses, industry and the environment (CoAG 2000). Most of the State salinity strategies assumed that the trends would have a negative impact and therefore salinity would threaten the conservation of biological diversity and the security of water for environmental purposes (e.g. DNRE 2000; Government of WA 2002), rather than enhance it. As a result much of the management has been aimed at lowering groundwater tables to mitigate the salinity risk, although the complex behaviour of different landscapes is now recognised (Williams 2008).

However, lowering groundwater tables and eradicating salinity can have negative impacts on high value assets. In this period of changing climate, south eastern Australia has entered into a prolonged drought (BoM 2006; 2008) which has dried many of the saline wetlands and increased the demand for groundwater resources. The long-term impact on the groundwater and salinity processes in south east Australia remains uncertain, due to the predicted changes in rainfall frequency and intensity, anthropogenic impacts on land use and the resulting impacts on recharge (Cartwright and Simmons 2008). In these circumstances maintaining saline groundwater discharge to wetlands, streams and other groundwater-dependent ecosystems is critical to their survival, and in some landscapes may conflict with salinity management aimed at lowering saline watertables.

In this context, the adoption of a risk management framework for salinity, which is based on the international and national standards, is an appropriate method to ascertain both the positive and negative risks associated with salinity management. An example is provided by the salinity risk management framework based on the AS/NZS 4360:2004 standard (Standards Australia 2004a) that has been developed in the Corangamite region of south eastern Australia, one of the priority regions in the National Action Plan for Salinity and Water Quality 2000 - 2007. In this region, the salinity risk management has been applied to developments regulated under the municipal planning schemes.

6.3 The Australian Risk Management Standard

The main elements (Figure 6.1) of the risk management process are:

- Communicate and consult with internal and external stakeholders at each stage of the risk management process.
- Establish the context in which the risk management will take place, including the criteria against which the risk will be evaluated.
- Identify the risks which could prevent, degrade, delay or enhance the objectives.
- Analyse the risks by determining the likelihood and consequences of the possible events which impact on the risk.
- Evaluate the risks by comparison against the pre-established criteria and balance the potential benefits against the adverse outcomes.
- Treat the risks if required to increase the potential for positive outcomes.
- Monitor and review the risk management process at all stages.

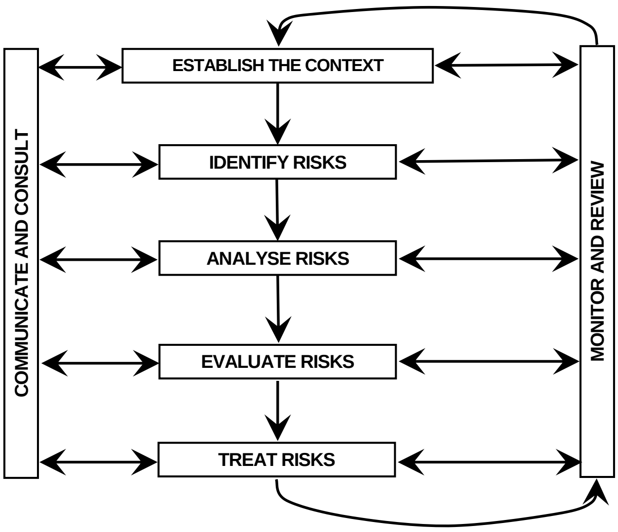

Figure 6.1 AS/NZS 4360:2004 Risk Management Process

An accompanying document, HB 436:2004 Risk Management Guidelines - Companion to AS/NZS 4360:2004 (Standards Australia 2004b) provides considerable guidance on the process. The standard also notes that recording the risk management process is an important aspect of good corporate governance, and is generally required for legal and business needs. In an increasing litigious society, the need for a defendable approach to salinity risk management cannot be overstated.

Within Australia, this standard has been adopted across a broad range of industries and government sectors. It includes the management of environmental hazards such as landslides (Australian Geomechanics Society 2007); weeds (Standards Australia 2006b); effluent (Standards Australia 2008); and generic environmental management (Gough 2001; Standards Australia 2006a).

6.4 Salinity Risk Management: a case history from the Corangamite Region

The Corangamite region in south west Victoria, Australia (Figure 6.2) comprises around 13,340 square kilometres of diverse natural landscapes (coast, mountains, plains, lakes), covers all or parts of nine municipalities supporting a population of around 400,000 inhabitants growing at 5.2 % per year, with manufacturing, tourism, agriculture and forestry as major industries (CCMA 2003). Dryland salinity impacts on the water quality, agricultural land, environmental assets, urban and rural infrastructure, and cultural heritage assets of the region. The most urgent threats are to the urban water supplies of the region's two major provincial cities – Ballarat and Geelong, and to wetlands of international significance listed under the Ramsar Convention and habitats for migratory birds subject to international treaties. Over 17,000 hectares of salt-affected land have been mapped and the area continues to expand (Nicholson et al. 2006).

In the national Audit (NLWRA 2001) the predictions for the Corangamite region were dire. The worst-case scenario suggested that 48.5 % of agricultural land will be at risk from shallow water tables by 2050, costing the region AUD$29 million per year; in addition to 16 towns, 4000 kilometres of roads and over 40 % of the region's wetlands threatened by 2050 (SKM 2000). Based on these predictions, the Corangamite region was nominated as one of the priority regions in Australia, under the National Action Plan for Salinity and Water Quality (CoAG 2000). Salinity risk management in the region is a shared responsibility between the asset managers and government authorities, with the Corangamite Catchment Management Authority (CMA) at the vanguard.

6.5 Establishing the context

The Corangamite CMA has developed the Corangamite Regional Catchment Strategy (RCS) 2003 - 2008 (CCMA 2003) as required by the *Catchment and*

Land Protection Act 1994 and using guidelines provided by the federal and state governments. The RCS provides long-term goals, targets for resource condition and management actions, and the investment framework for natural resource management in the region. For salinity management, the Corangamite Salinity Action Plan (SAP) identifies priority investment areas and management actions which are appropriate to mitigate the risk of salinity impacting on assets (Nicholson et al. 2006). This includes a variety of methods such as traditional groundwater recharge control using trees to protect agricultural land and the installation of surface and subsurface drainage to alleviate the impacts of groundwater discharge. As one of the targets in the RCS aims to protect 4,000 urban dwelling allotments from secondary salinity by 2020, the SAP identified a need to collaborate with local governments in the region to introduce statutory planning regulations aimed at achieving this goal.

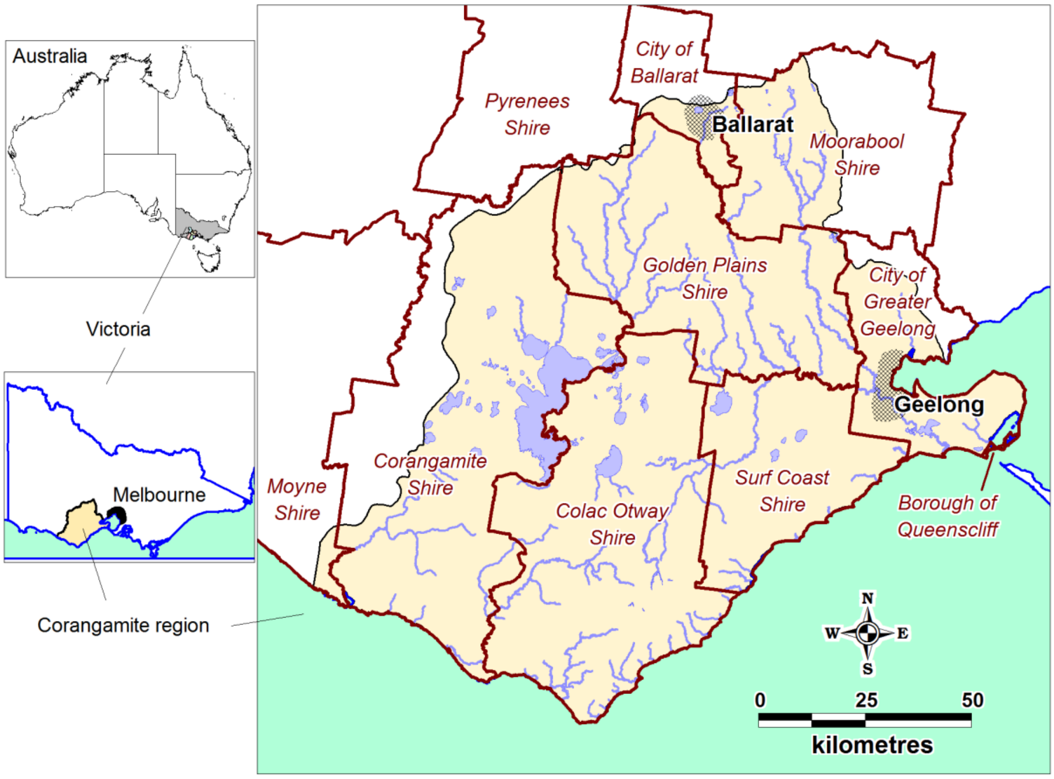

Figure 6.2 Location of the Corangamite region and municipalities

Statutory planning in Victoria uses a hierarchical structure to address the wide range of planning issues in an ordered manner. These include (from highest to lowest) the State Planning Policy Framework; the Local Planning Policy Framework incorporating a Municipal Strategic Statement (MSS); Zones;

Overlays; and Particular Provisions. Zones and Overlays apply to subdivisions, buildings and development works and generally concern environmental, landscape, land and site management issues. They are incorporated into the municipal planning schemes in the form of maps and policies that control the use and development of land. While Zones dictate land uses applicable to areas, Overlays regulate development and can define the conditions under which development can be undertaken.

To address the goals of the RCS and SAP, a project to implement Salinity Management Overlays for each municipality in the Corangamite region was undertaken. A Salinity Management Overlay (SMO) is primarily used to control development in saline areas and mitigate damage to the infrastructure but can also be used protect saline environmental assets from inappropriate development. In this context the project supports the view of the Corangamite CMA and state and federal governments that salinity is a significant issue within the region and that it needs to be better managed through the planning framework. With regards to municipal planning there are two possible salinity risks resulting from planning applications for development or works: 1) the potential impact of salinity on the development at a site; and 2) the potential impact of the development at a site on the environment elsewhere in the catchment.

A main driving force for local government interest is management of their legal risk. Without an SMO there is the potential for municipalities that approve development in known saline areas, or areas with a known potential for salinisation, to be subject to litigation from property developers or purchasers of such areas. In addition, the SMO and associated policy will maintain social amenity by ensuring that new urban areas will be zoned, subdivided and developed to mitigate salinity damage to public infrastructure, protect natural environmental values, sustain the value and productivity of soils and maintain the economic and environmental sustainability of rural and urban areas within their jurisdiction.

6.6 Identifying the risks

Identifying the salinity risks is the equivalent of a hazard assessment, in that it requires an analysis of what can happen, where in the landscape it is likely, and when it could occur. Once it is established what, where and when salinisation may occur, the causes and scenarios also need to be considered.

In regard to what might happen, the impacts of salinity have been previously documented (Ghassemi et al. 1995; NLWRA 2001) as threatening agricultural productivity, environmental values, water quality, rural and urban infrastructure, and cultural and heritage assets. Within the Corangamite region, many of the impacts on land, rivers, wetlands, urban water supplies and biodiversity have been documented in the SAP (Nicholson et al. 2006). In urban areas it may affect the establishment and growth of gardens, damage building structures and associated pipes and conduits, corrode hot water services and reduce the longevity of on-site sewerage treatment systems. The longevity of roads, bridges, culverts, and electricity, telecommunication, gas and water services can also be reduced at a considerable cost to the asset managers (Buckland and McGhie 2005; Kelliher et al. 2005; Austroads 2007; Nicholson et al. 2008).

In regard to where and when the impacts of salinity might occur, five categories were considered in the development of the SMO:

1) Areas currently affected by salinity regarded as a threat
2) Areas of primary salinity regarded as an asset
3) Areas not currently affected by salinity, but with a stated likelihood of experiencing secondary saline groundwater discharge within a given time-frame
4) Areas where inappropriate land-use or development may adversely impact on primary salinity assets
5) Areas where development or inappropriate land-use may ultimately initiate or exacerbate secondary salinity elsewhere in the landscape.

Categories 1 and 2 have been identified mainly on the basis of salinity indicator vegetation (Bozon and Matters 1989; DPI 2006; DPI 2008b). By observing the structure and variety of halophytic plants, an inference can be made about the likelihood of the site being naturally saline. A site is more likely to be a primary site if it has a high diversity of native salt tolerant plants that have low capacities for colonising newly saline sites (Allen 2007). Where possible these sites were confirmed using historic information, especially historic aerial photographs and maps. By comparison a site where salinity has been induced by anthropogenic land use change since European settlement has abundant halophytic species considered efficient colonisers, or ruderal species. It is acknowledged that degraded primary sites may be misidentified as secondary, especially where disturbance has decreased the diversity of non-ruderal species; and increased the diversity of ruderal species. However, at these ambiguous sites it can be argued that the difference is irrelevant as the environmental value is diminished and the salinity has the same potential to impact on built infrastructure.

In addition to primary salinisation of land, the Category 2 sites also include saline wetlands categorised by Corrick (1982) as either 'permanent saline' or 'semi-permanent saline'. Wetlands are classified as saline if the average annual salinity of the water exceeds 3,000 milligrams per litre total dissolved solids (Corrick and Norman 1980). A permanent saline wetland includes coastal and intertidal wetlands, as well as inland salt lakes. A semi-permanent saline wetland may be inundated to a depth of 2 metres for up to 8 months of the year and include commercial salt harvesting operations. The Corangamite region includes significant proportions of Victoria's semi-permanent saline (13 %), permanent saline (58 %) and salt works (82 %) wetlands (Sheldon 2005).

Category 3 has less certainty. These areas were identified using depth to watertable maps interpolated from groundwater bores in the region (Peterson and Barnett 2004a; 2004b) and validated using the time-series hydrographs from approximately 500 monitoring bores installed under the SAP (Dahlhaus et al. 2004b; 2008b). A depth to saline watertable of less than 2 metres is generally regarded as a salinity hazard (DNRE 2000; NLWRA 2001). In these areas, a development such as a building could be affected by salinity if the foundations or service pipes interact with the watertable or its capillary fringe. Similarly, if the shallow watertables rise, a greater area of land salinity could be expected.

Category 4 areas are those surrounding the high value saline assets where development has the potential to impact on the integrity of the asset. As examples, inappropriate land-use may include excessive irrigation, such as results from the unrestricted use of recycled wastewater delivered to residential suburbs via 'third-pipe' schemes. Modifications to natural drainage and stormwater discharge from developments could also impact on assets such as saline wetlands by reducing the salinity and consequently threatening the ecology. These areas are generally delineated on the basis of the surface water and groundwater catchments surrounding the asset.

Category 5 areas are those with responsive groundwater flow systems which are implicated in salinity processes (Dahlhaus 2004). These areas are those in which broad-scale land-use changes, especially those that increase groundwater recharge, can ultimately exacerbate existing salinity or initiate secondary salinity at a local or more distant location. Such land-uses may include broad-scale irrigation schemes, the construction of large lakes, irrigated recreational areas such as golf-courses and sporting fields, diversion of waterways, or the widespread permanent clearing of native vegetation or tree plantations.

6.7 Risk assessment

The analysis of salinity risk requires the estimation of both the likelihood of salinity impacting on an asset and the consequence of that impact on the asset. The likelihood may be analysed in terms of increasing or decreasing salinity. For example, increasing salinity may negatively impact on biodiversity (Hart et al. 2003) or the integrity of a road (Austroads 2004), whereas decreasing salinity may impact on the ecology of a saline wetland or estuary (Gillanders and Kingsford 2002), or the stability of a sodic soil (Sumner 1993). In both cases salinity risk analysis should estimate both the likelihood of the change in salinity impacting on a development and the likelihood of a development impacting on salinity processes. The standard indicates that this estimation can be qualitative, semi-quantitative, or quantitative (Standards Australia 2004a).

Typically, the estimation of likelihood is established by a site investigation that would include a regional assessment of the hydrogeology and salinity processes. As a minimum, the investigation should establish the salinity of the soils, depth to groundwater, the salinity of the groundwater, the response of the groundwater system to hydrologic change, and the geomorphological and hydrological setting including soil hydrology. The historical trends of the groundwater levels require analysis and predictions made as to whether the groundwater is likely to rise or fall during the life of the development (taking into account the impact of regional land-use on the hydrology). The aim of the investigation is to deduce a conceptual model of the salinity processes at the site which can be used to predict the post-development scenarios.

For most geohazard risk assessments, likelihood is stated in terms of the frequency of an event, such as the indicative annual probability of a landslide (Australian Geomechanics Society 2007). However, salinity is not an event-based geohazard and a qualitative estimation of likelihood can only be stated in terms of probability (Table 6.1).

Table 6.1 Example likelihood (probability) scale

Descriptor	Description
Certain	Salinity exists at the site and is likely to persist for the life of the project; or a change to the existing salinity (increase or decrease) is certain during the life of the project
Probable	Either the onset of salinisation or a decline in existing salinity can be expected to occur during the life of the project
Possible	Either the onset of salinisation or a decline in existing salinity are not expected to occur during the life of the project
Improbable	Either the onset of salinisation or a decline in existing salinity is conceivable, but highly unlikely to occur during the life of the project

(based on: Standards Australia 2004b)

Alternatively, a semi-quantitative or quantitative approach may be used where the appropriate site measurements may be combined to calculate a value for likelihood. As an example: Likelihood = *function* (soil salt store, predicted change to soil hydrology parameters, rate and volume of groundwater discharge or baseflow, development design life, rate of change of land-use, depth to groundwater, rate of rise or fall of the watertable, salinity of the groundwater, the proportion of a site altered in relation to the groundwater flow system, the responsiveness of the groundwater flow system, etc.).

However the cost and time required to conceptualise the salinity process model, quantify the parameters and develop a predictive numerical model for each site can only be justified for the larger developments. Most planning applications are for domestic or small scale commercial developments or works (ABS 2009b) where a qualitative approach is favoured. Designing a generic qualitative likelihood assessment is not possible, since the likelihood of increasing or decreasing salinity at each site depends on both the salinity processes operating in that particular landscape and the specific development proposed, and therefore the factors that need to be considered are unique to each site.

Similarly, the consequence of the predicted changes in salinity on the proposed development and the consequence of the proposed development on the salinity processes both need to be considered. This would typically involve an examination of the design elements such as the type and purpose of a building (e.g. domestic residence, agricultural outbuilding, factory, etc.), construction materials and design elements (e.g. architectural, engineering and building detail), landscaping proposals (e.g. proposed garden plants, watering systems, area of impervious surfaces), drainage and stormwater design, and the proposed occupation rate and use of the development.

Consequence may be quantified in terms of triple bottom line (i.e. economic, environmental and social) cost or benefit. It may include the site value, number of species lost or gained, salt content of potable water, or other measures as appropriate. However the quantification of the damage or improvement to an asset through changes in salinity has many uncertainties (Pannell 2001; Buckland and McGhie 2005), and therefore qualitative estimations of consequence are usually adopted (Table 6.2) in most planning applications. The uncertainties arise from the insidious nature of salinity damage or improvement to an asset and the long period over which it occurs. Salt attack on building and construction materials often involves chemical corrosion and erosion of materials by prolonged wetting and drying cycles (Bucea and Sirivivatnanon 2003). Similarly, the degradation of saline ecologies by the introduction of fresh water (Gillanders and Kingsford 2002) or by increasing the salinity (Williams 2001) may also be a relatively slow and cyclical process.

Table 6.2 Example consequence scale

	Descriptor	Description
Negative	Severe cost	Irreversible damage, huge cost (e.g. species extinction, loss of urban water supply, loss of Ramsar wetland, urban centre abandoned, agricultural land abandoned, etc.)
	Major cost	Extensive damage, major cost (e.g. loss of habitat, degradation of potable water supply, destruction of built infrastructure, etc.)
	Moderate cost	Some damage, high cost (e.g. invasion of exotic species, damage to built infrastructure, loss of recreational amenity, lower quality irrigation water, etc.)
	Minor cost	Little damage, low cost (e.g. episodic changes to water salinity, low level salinisation of urban gardens, etc.)
	Negligible cost or benefit	Negligible impact, no measurable cost or benefit
Positive	Minor benefit	Little improvement, low benefit (e.g. episodic changes to water salinity, low level improvement in agricultural productivity, etc.)
	Moderate benefit	Some improvement, high benefit (e.g. colonisation by native species, longer periods of baseflow to saline wetlands, etc.)
	Major benefit	Measurable and noticeable improvement, major benefit (e.g. reclamation of native habitat, improved potable water supply, increased biodiversity, etc.)
	Outstanding benefit	Significant and sustained improvement, huge benefit (e.g. reclamation of agricultural productivity, return of migratory species, etc.)

(based on Standards Australia 2004b)

The risk is then described as a function of the likelihood and the consequence. In a qualitative example, these may simply be combined in a matrix:

Table 6.3 Example matrix for determining the level of risk

Threat				
Likelihood	Negative Consequence			
	Severe	Major	Moderate	Minor
Certain	Very high	Very high	High	Moderate
Probable	Very high	High	Moderate	Low
Possible	High	Moderate	Low	Low
Improbable	Moderate	Low	Very Low	Very Low
Opportunity				
Likelihood	Positive Consequence			
	Minor	Moderate	Major	Outstanding
Certain	Moderate	High	Very high	Very high
Probable	Low	Moderate	High	Very high
Possible	Very low	Moderate	High	High
Improbable	Very low	Low	Moderate	High

The use of qualitative risk assessments in environmental management has been criticised for their subjective approach and linguistic uncertainty (Burgman 2001; 2005). For the same reason many geohazard assessments require quantitative estimations for the most serious risks, such as loss of life (Australian Geomechanics Society 2007). As an alternative, the level of risk may be calculated as a probabilistic equation. As an example (based on that in: Australian Geomechanics Society 2007), the salinity damage to a building might be calculated as:

$$R_{(Prop)} = P_{(H)} \cdot P_{(S:H)} \cdot P_{(T:S)} \cdot V_{(Prop:S)} \cdot E$$

Where:

$R_{(Prop)}$ = the risk (e.g. annual loss of building or property value)
$P_{(H)}$ = the annual probability of the hazard (e.g. shallow saline watertables existing at the site, taking into account the longer-term rate of rise/fall)
$P_{(S:H)}$ = the probability of the spatial impact by the hazard (e.g. the saline groundwater coming into contact with the building foundations, taking into account the building design elements)
$P_{(T:S)}$ = the temporal spatial probability of the hazard (e.g. the amplitude and frequency of the seasonal fluctuations in the groundwater levels and salinity)
$V_{(Prop:S)}$ = the vulnerability of the property to the spatial impact (e.g. the proportion of building or associated infrastructure damaged by the salt)
E = the element at risk (e.g. the value or net present value of the property).

Although salinity processes and their effects are deterministic, the empirical observations from a particular site may indicate that the commonly accepted models do not apply (e.g. Bann and Field 2007; Dahlhaus et al. 2008a). Hence probabilistic methods are more reliable than deterministic methods and qualitative risk assessment is the preferred approach in planning applications for residential and small-scale commercial developments in areas covered by the SMO. Basic information such as the mapped salinity, the depth to groundwater, salinity of the groundwater and time-series groundwater monitoring hydrographs have been made available on the internet (UB Spatial 2009), providing assessors with the data on which to base their conceptual model and subjective judgement. Quantitative salinity risk assessment is more applicable for larger and expensive developments where a developer may need to argue a case using data accumulated for the development of the application.

6.8 Risk evaluation

The objective of risk evaluation is to decide on whether the risk is acceptable, whether risk treatment is required, and to set priorities. The standard and guidelines (Standards Australia 2004a; 2004b) note that risk generally is categorised into three levels, *viz:* acceptable, tolerable and intolerable. An acceptable risk is one which fits with the specified criteria and does not need further treatment. A tolerable risk is one which is too high to be acceptable, but can be tolerated under certain conditions, such as where treatment measures are undertaken or liability is transferred. Intolerable risks are those which are unacceptable or too costly to treat. Risk evaluation often includes a consideration of issues such as cost of treatment, business or public confidence, public reaction, politics, availability of alternatives, environmental impact, availability of insurance, and fear of litigation (Burgman 2005).

For some risks, the evaluation criteria are established by the standards (e.g. building codes, materials standards, environmental standards), the regulators (e.g. government agencies), the asset managers (e.g. government, municipalities, corporations), the proposed developers, or the owners. Because there are no formal standards for salinity in the Building Code of Australia (ABCB 2009) at present, specific information was developed in consultation with the Victorian and National building legislation bodies for the introduction of the SMO in the Corangamite region (EnPlan-DBA 2006). Another example of an established risk evaluation standard is the State Environment Protection Policy that stipulates the salinity values for various uses of surface water and groundwater (EPA 1997; 2003).

Where the risk is assessed as an opportunity (i.e. high benefit), detailed strategic planning may be required to maximise the full benefit. For example, a substantial improvement to the ecology of a saline lake my provide opportunities for passive recreation such as bird watching, which in turn could increase ecotourism and bring economic benefits if the appropriate infrastructure was in place.

6.9 Risk treatment

In the municipal planning schemes, the treatment of salinity risk makes use of different hierarchical components as appropriate. For Category 1, 2 and 3 hazards, the SMO is the appropriate statutory tool. For Category 2 and 4 areas, an Environment Significance Overlay (ESO) is also used to protect a saline environmental asset of high value. The ESO is a versatile planning overlay that is often applied to significant wetlands or areas with designated ecological values that warrant specific planning attention. For Category 5 areas, the MSS is the preferred planning tool, as it contains the strategic planning, land use and development objectives of the planning authority and the strategies for achieving those objectives. Since the delineation of the responsive groundwater flow systems and the land-uses which may initiate a hydrological response are both broad-scale, a strategic planning tool is more appropriate than an Overlay.

Where a risk is unacceptable, a decision can be made to treat the risk to bring it within a tolerable range. Typical options would include (Standards Australia 2004a):

- Accept the risk (e.g. No further treatment required)
- Avoid the risk (e.g. Utilise the planning scheme to prevent development)
- Mitigate the risk (e.g. Restrict the development options)
- Reduce the likelihood (e.g. Implement site conditions or salinity management)
- Reduce the consequences (e.g. Select appropriate building materials)
- Share the risk (e.g. Insurance)
- Retain the risk (e.g. Develop strategies for future salinisation)
- Physically separate (e.g. Install moisture or salt barriers)
- Duplicate resources (e.g. Relocate species for preservation)
- Transform the risk (e.g. Protect assets using groundwater pumping)
- Postpone the risk (e.g. Extend investigation, or install warning systems)

The selection of risk treatment should be considered in the context of the 'knock-on' effects elsewhere. For example, groundwater pumping is only acceptable where disposal options are available. The potential benefits of the treatment options, their effectiveness at reducing losses, their cost of implementation and their impact on other stakeholder objectives (including the introduction of new risks) need to be considered. Where the salinity risk assessment presents a benefit, it is unlikely that any risk treatment would be required.

6.10 Essential components

Throughout the risk management process, three elements are essential (Standards Australia 2004a):

Consult and communicate: Burgman (2005) suggests that those bearing the risks need to be involved from the outset in all stages, recognising that human perceptions and values affect experts and analysts as strongly as other stakeholders. At the very least, consultation and communication is essential to clarify the roles and responsibilities of the government, catchment management and municipal planning authorities. Throughout the risk management process, regular communication is required to inform and educate stakeholders of the risk and risk treatment options.

Monitor and review: monitoring and reviewing the risk assessment may be in the form of third-party audits as quality assurance. Regular monitoring of the effectiveness of the treatment is essential and should result in a regular re-evaluation of the risks. This is particularly important in landscapes undergoing continuous land-use change which results in continuous hydrologic change.

Recording the process: recording the risk management process is usually required as part of the legal and business requirements of an organization and should include a risk register and incident database.

6.11 Conclusions

This framework based on the Australian standard provides a systematic, disciplined and rigorous approach to salinity risk for municipal planning and catchment management in the Corangamite region. In particular, it provides the authorities with unambiguous, logical and defendable processes and practices for the assessment of salinity risk. It can inform the development of strategies and decision making to protect all classes of assets which are threatened by changes to salinity processes, even those where the salinity itself is the asset (e.g. a saline wetland or estuary).

In the most recent Federal and State salinity frameworks, the emphasis has been placed on the protection of community and catchment assets. In this context, the adoption of a national risk management framework based on national and international standards is both logical and timely. Climate change will result in hydrologic uncertainty (Cartwright and Simmons 2008) and this framework can assess both the negative and positive salinity risks to all assets.

6.12 Acknowledgements

The contributions of Chris Harty (Chris Harty Planning and Environmental Management), Darrel Brewin (Darrel Brewin and Associates Pty Ltd), Graeme David (EnPlan Australia Pty Ltd), and Tony Miner (A.S. Miner Geotechnical) are acknowledged in the preparation of this manuscript. The work was funded through the Corangamite Catchment Management Authority and the assistance of Peter Codd, Angela Vary and Lucas Oram is gratefully acknowledged.

7 Conclusions

This chapter summarises the research presented in the four preceding chapters (i.e. those written as papers) in relation to the three research questions that were stated at the conclusion of Chapter 2.

The first question asked whether the assumptions about the causes of salinity were valid for the Corangamite region. These assumptions are based on the vast body of scientific work in Australia that has resulted in the adoption of a paradigm, or common way of thinking held by the majority of members of a scientific community. The paradigm is that dryland salinity is caused by rising groundwater tables as a result of broad-scale clearing of native vegetation. This validity of this paradigm had previously not been tested in the Corangamite region.

The variation in the biophysical landscape - geology, geomorphology, climate, biogeography, surface water hydrology, hydrogeology and salinity occurrences - provides an extraordinary diverse milieu in which to test the paradigm. Adopting a systems-thinking 'big picture' approach to complement traditional scientific methods has brought new insights from other disciplines. In particular the historical evidence of environmental condition and change has added observations and data to test the paradigm over a much longer timeframe than has been previously used in salinity studies in Australia.

The research presented in Chapter 3 (paper 1) shows that salinity is a permanent and inseparable attribute of the Corangamite region. The geomorphic evidence in the lunettes, the palaeolimnological evidence of salinity and the anthropological record of aboriginal settlement all support the fact that primary salinity, supported by shallow saline groundwater tables, has episodically existed throughout the Quaternary. The earliest written historical records, years before the widespread clearing of native vegetation had commenced, noted shallow watertables, some of which were saline. The early scientists noted hydrological changes and attributed them to anthropogenic causes, but these were contrary to the current axiom in that they recorded that the widespread clearing and grazing increased runoff. From these historical records there is no evidence found which supports significant rises in groundwater following widespread land-use change.

The hypothesis that the current paradigm does not hold in the Corangamite region is further strengthened by the research presented in Chapter 5 (paper 3), which also uses historical records to complement the scientific method in investigating the causes of increasing salinity in the West Moorabool River catchment. In this region, the geological maps of 1870-71 note abundant discharge from at least 16 springs in a thickly timbered forest, and record 21 wells intersecting shallow watertables. The historical records of 1881 list 17 springs and 70 wells with similarly shallow watertable depths, following the widespread clearing of the native vegetation in the same area. The data of the past four decades include 1792 monitoring records of groundwater levels, which are, on average, within the range of those a century ago. From this research it is evident that the current paradigm does not hold in the West Moorabool River catchment, since there is no discernable rise in groundwater levels following the clearing of the forest over a century ago.

Although it is proven that the current paradigm does not hold in the areas of the Corangamite region investigated in this research, it is not entirely clear why that is the case. It is speculated (in Chapters 3 & 5) that in areas with permeable or thin soils where the average rainfall exceeds the average evaporation for several months of the year, the average annual recharge may be in excess of discharge. These discharge-driven groundwater systems may have been relatively ‘full’ for centuries, regardless of the changes to vegetative cover.

The second research question asked if the assumed processes for secondary salinisation were valid for the Corangamite region. If the causes of secondary salinity are not due to rising watertables mobilising salt to the surface, then the management actions aimed at lowering watertables are also invalid. The current paradigm - established by scientific studies of salinity elsewhere in Australia - has prevailed in social policies such as the regional salinity management plans implemented from the late 1980s to the present. To mitigate the expansion of dryland salinity and protect the catchment assets under threat, the establishment of tree plantations has been promoted to control recharge and restore hydrogeologic equilibrium.

This research question was tested by investigating the impact of 20 years of salinity management in the Pittong target area (Chapter 4, paper 2). Using salinity mapping data from the past three decades, it was shown that the area of salinised land has continued to expand and new saline discharge areas have emerged. This observation is also verified by the records from the nominated State Government salinity monitoring site. However, a statistical analysis of the monitoring records, using a published method, showed a general trend of falling watertables. In the Pittong area, the research reveals an apparent contradiction to the widely accepted theory. The salinity has continued to expand, despite the efforts of active salinity management, a decade of below average rainfall and generally falling watertables.

Based on the observed behaviour and locations of the discharge seeps, the groundwater salinities, the watertable time-trends and the available regional groundwater bores, a model in which the salinity is caused by regional saline groundwater brought to the surface along geological structures is conceptualised. Although this model has yet to be proven, it provides a logical explanation for the expansion of salinity while the local groundwater tables fall. If this model is correct, then the risk of salinity damage to agricultural land and waterways in this landscape is better managed by through controlling the saline discharge from springs (for example, using vegetation, drainage, pumps or evaporation basins), than lowering local watertables using tree plantations.

This second research question is also tested by the research in Chapter 5 (paper 3), where it is apparent that rising watertables cannot be the cause of the increasing stream salinity in the West Moorabool River catchment. Although the statistical analyses and observations show a rise in stream salinity over the past 30 years, the groundwater tables have remained relatively unchanged for more than a century. A logical (although unproven) explanation for the increasing stream salinity is that the combination of increased surface water harvesting, the export of both surface water and low-salinity groundwater from the catchment, and the shift to a drier climate has reduced the stream flows and proportionally increased the amount of saline baseflow from deeper aquifers.

Although both these case histories leave open questions about the true causes of salinity in their respective landscapes, they both question the paradigm that secondary salinity is caused by rising groundwater tables. Holistic thinking using a wide range of evidence reveals that the causes of both land and stream salinity are more complex than initially assumed and probably unique to each landscape. Rising water tables are not the single cause of salinity in many of Australia's landscapes, a fact which is recognised more widely than in this study alone (e.g. Dahlhaus et al. 2000; Acworth and Jankowski 2001; Fawcett 2004; Wagner 2005; Bann and Field 2006).

The final research question asked if the salinity threat and risk to assets was valid. This questions a second paradigm in contemporary Australian salinity risk assessment studies: that salinity is universally regarded as a threat to assets.

The research presented in Chapter 3 (paper 1) shows that in many areas of the Corangamite landscape salinity is an inherent asset that sustains environments, among which are wetlands of international importance that are subject to international migratory bird treaties. Published biological research shows that maintaining appropriate salinity levels is vital for their ecological health (Williams 1995; 2002). In these environs, managing salinity by lowering the groundwater levels may indeed threaten environmental assets. It is clear that salinity risk assessment and management needs to consider both negative and positive impacts.

In Chapter 6 (paper 4), a systematic, disciplined and rigorous framework for salinity risk management has been developed to protect all classes of assets which are threatened by changes to salinity processes, even those where the salinity itself is the asset. This brings a totally new perspective to salinity risk management in Australia. Salinity and salinisation may be assessed as a threat or an opportunity.

The framework developed through this research has recently been integrated into municipal planning schemes in collaboration with catchment managers and local governments, as a logical and defendable basis for strategic and statutory planning decisions. Since it is based on the generic international and national risk management standards it can be more broadly adopted, and should be included as the basis for all salinity management policies.