Hunter Catchment Salinity Assessment

Final Report

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Cover photo: Goulburn River at Rosemount – Upstream of the Hunter River Junction. (Photo: M. Krogh.)

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Abbreviations

ANZECC	Australian and New Zealand Environment and Conservation Council
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
AUSRIVAS	Australian River Assessment System
EC	electrical conductivity
ENSO	El Niño–Southern Oscillation
EPA	NSW Environment Protection Authority
GAM	generalised additive model
HCR CMA	Hunter–Central River Catchment Management Authority
IPO	Interdecadal Pacific Oscillation
JI	Jaccard's Index
NOW	NSW Office of Water
OEH	Office of Environment and Heritage
PC	protective concentration
PSS	pooled sample sets
RFR	relative family retention
SIGNAL	Stream Invertebrate Grade Number Average Level
SPEAR	SPEcies At Risk
SSD	species sensitivity distribution
SWC	State Water Corporation
TAD	total allowable discharge
WSP	water sharing plan

Summary

The Hunter River Salinity Trading Scheme (the Scheme) operates to minimise the impact of saline water discharges from industry on the Hunter River. It achieves this by allowing discharge of saline water only at times of high or flood flow in the Hunter River and uses a system of salinity credits to limit the amount of salt that can be discharged at any one time. The Scheme commenced as a pilot in 1995 and was formalised in 2002 when the Protection of the Environment Operations (Hunter River Salinity Trading Scheme) Regulation 2002 (the Regulation) commenced.

In anticipation of the ten-year review of the Regulation, the NSW Environment Protection Authority (EPA) commissioned the Office of Environment and Heritage (OEH) to conduct a desktop study to evaluate the effectiveness of the Scheme based on available water quality and ecological health data.

There are a variety of potential sources of salinity in the Hunter River catchment including rainfall, atmospheric deposition, run-off and infiltration, weathering of geological strata, groundwater and a range of anthropogenic sources including the Scheme. The Hunter River valley is generally considered to be saline due to the marine origin of some of its Permian sediments. However, recent land-use activities in the catchment may have contributed to rising groundwater levels in some areas and an increase in the salinity load reaching many streams. Overlaid on the natural cycling of salts in the Hunter River catchment are anthropogenic sources – particularly mining, power generation and agriculture.

The Scheme restricts saline discharges from mining and power generation to times of high or flood flow. The Scheme's salinity targets apply only in the Hunter River between Glenbawn Dam and Singleton, and not within any of the tributaries.

The key findings of the current salinity assessment are:

- There was little evidence that groundwater levels or the electrical conductivity (EC) of groundwater have been rising in recent times. However, this conclusion is affected to some degree by limited temporal sampling and a bias to current monitoring bores being located in alluvial areas often well away from the areas of major mining operations.
- If future trends in groundwater level and conductivity are to be undertaken and related back to the impact of the Scheme (or mining and power generation), then a more comprehensive and representative groundwater monitoring program is required for the catchment.
- The major impact of the Regulation on EC levels is likely to have been the continued restriction of saline water discharges to periods of high and flood flows when the potential for dilution is at its greatest (as opposed to continuous or intermittent discharges regardless of flow conditions).
- The assessment of the overall effectiveness of the Scheme on surface water quality suggests that the Scheme has:
  - had little effect on flows and electrical conductivity levels in the Hunter River upstream of Denman
  - reduced electrical conductivity levels at (and immediately upstream of) Singleton and Greta, and
  - potentially reduced electrical conductivity levels at monitoring stations between Denman and Singleton.
- The available data suggests that throughout the catchment macroinvertebrate 'health' is on average good, but there are some areas where this is quite poor.

- Although salinity is one of several factors affecting stream macroinvertebrate communities in the Hunter River catchment, salinity appears to be a relatively important factor.
- The weight of scientific evidence suggests that current Scheme salinity targets should not be raised. Further scientific analysis and modelling would be required to support altering the Scheme salinity targets in the future, in order to better understand existing salinity impacts on ecosystem health in the Hunter River and its tributaries.
- On average over the life of the Scheme, participant discharges contributed approximately 10 per cent of the entire salt load of the Hunter River at Singleton. However, recent averages are in the order of 13–20 per cent of total annual salt load.
- On average over the life of the Scheme, participants have utilised approximately 25 per cent of the given opportunities to discharge [the 'total allowable discharge' (TAD)]. However, recent averages are in the order of 40–50 per cent of the TAD.
- Experimental studies are recommended in order to fully understand the environmental effects of the different components of saline water discharged to the Hunter River catchment (e.g. ionic composition, metals/metalloid contamination, etc.).
- The increasing discharge demand, salt load and TAD usage under the Scheme will need careful ongoing monitoring and assessment in order to assess the potential for future trends or changes to impact aquatic ecosystems and environmental values.
- Hunter River salt loads can also be affected by the major tributaries such as the Goulburn River and Wollombi Brook.
- The Goulburn River subcatchment contributes relatively high salinity water to the Hunter River and is not currently captured by the Scheme upstream of Kerrabee. Further strategic real-time monitoring of flow and salinity in the subcatchment is recommended, considering the likely expansion of mining and development of coal seam gas extraction. This monitoring is currently limited.
- The at times high EC levels in the Wollombi Brook at Warkworth in the mid to late 2000s (not related to flow) warrant further investigation.

1. Introduction

The Scheme is implemented under the Protection of the Environment Operations (Hunter River Salinity Trading Scheme) Regulation 2002 (the Regulation). The central idea of the Scheme is to discharge salty water only when there is lots of low-salt, fresh water in the river (DEC 2006). This is when the river can best handle salt discharges because:

- large amounts of fresh water dilute the saltier discharge so the impact on the river is not as great, and
- through careful control, the mixture of river and discharge water can be kept fresh to meet water quality standards.

Monitoring points along the river are used to measure whether the river is in *low* flow, *high* flow or *flood* flow. When the river is in *low* flow, no discharges are allowed. When the river is in *high* flow, limited discharge is allowed, controlled by a system of salt credits. The amount of discharge allowed depends on the ambient salinity in the river which can change daily.

River salinity targets are established for three reference points in each of three River sectors (upper, middle and lower). Denman is the reference point for the upper sector; upstream of the Glennies Creek confluence for the middle sector; and Singleton for the lower sector. The total allowable discharge is calculated so that the salt concentration does not go above 900 EC in the middle and lower sectors of the river, or above 600 EC in the upper sector. When the river is in *flood*, unlimited discharges are allowed as long as the salt concentration does not go above 900 microsiemens per centimetre (µS/cm). Members of the scheme coordinate their discharges so this goal is achieved (DEC 2006). It is important to recognise that the salinity targets only apply to the Hunter River between Glenbawn Dam and Singleton and not within any of the tributaries. The targets also apply only during high or flood flow periods. As a result, the Scheme may actually have little influence over stream salinity levels for the majority of the time.

In anticipation of the review of the Regulation, the EPA commissioned the Office of Environment and Heritage (OEH) to conduct a desktop study to evaluate the effectiveness of the Scheme based on available water quality and ecological health data.

Three key questions were specifically asked to be addressed:

1. Has the Regulation impacted on aquatic ecosystems and associated environmental values since it commenced in 2002?
2. Does the Regulation have the potential to impact on aquatic ecosystems and associated environmental values in the future?
3. What other sources of salinity in the Hunter catchment could influence the operation of the scheme in the future?

This involved:

- collating research on the environmental impact of salt on Australian aquatic biota
- comparing pre-Scheme electrical conductivity and ecological health with post-Scheme conductivity and ecological health upstream of Singleton (and, if insufficient data, comparing existing information on the ecological impacts of salt with recorded conductivity levels)
- determining whether there is any trend in long-term background conductivity when the Scheme is not operating

- assessing existing trends and possible emerging sources and modelling their impact on the flow and conductivity levels in relation to Scheme thresholds and aquatic ecosystem values in the Hunter.

This aim of this report is to address these key questions as far as possible, based on available water quality, quantity and ecosystem health data for the Hunter River catchment.

2. Catchment overview

The Hunter River catchment drains an area of approximately 22,000 square kilometres on the central NSW coast. The valley comprises rugged mountain ranges in the north, undulating farmland in the central and western regions, and widespread fluvial/estuarine flatland coastal areas (PPK 1994). The river and some of its main tributaries have their source in the uplifted, deeply dissected and predominantly Tertiary basalt Mount Royal Ranges to the north of the valley. A large portion of the Hunter River flows are also contributed by its western tributary, the Goulburn River, which drains more than one third of the valley. Another major tributary, Wollombi Brook, joins the Hunter River from the south near Warkworth. The valley's central and southern regions consist of gently undulating topography associated with the more easily weathered Permian sediments. Areas immediately adjacent to the Hunter River are predominantly alluvial.

The Hunter River valley's stream sediments are strongly controlled by its underlying geology. Features such as major fault lines separate the Carboniferous rocks exposed along the northern areas of the valley from the central Permian-age coal measures and the Triassic sandstones in the south and south-east. Extensive folding and faulting of the Carboniferous rocks have resulted in the formation of steep country leading up to the Barrington Tops which is underlain by basalt. The Permian rocks have eroded to form the main corridor of the broad valley. Due to marine transgressions during their formation, some of these rocks are high in salt content, which has resulted in naturally high salinity levels in many of the central valley streams and drainages. Additionally, the valley is often prone to dryland salinity due to extensive clearing of the native vegetation and elevated or intersected saline groundwater tables.

The Hunter River and its tributaries, like most Australian coastal streams, have a highly variable flow. This variability is mainly influenced by the climatic regime; however anthropogenic factors have also altered the frequency, volume and seasonality of stream flows. The Hunter River's average annual discharge is approximately 180 gigalitres, including contributions from the Goulburn, Paterson and Williams rivers and Wollombi Brook, and the upper Hunter tributaries including the Pages and Isis rivers, and Middle, Dart, Stewarts, Moonan and Omadale brooks. The flows within the Hunter catchment are regulated through three major storages: Glenbawn, Glennies Creek and Lostock dams. The largest of these is Glenbawn Dam (completed 1958, enlarged 1987). It can hold 750 gigalitres with a reserve capacity of 120 gigalitres for flood mitigation and can release in excess of 7.5 gigalitres a day (plus whatever spills). Glennies Creek Dam (completed 1983) can hold 283 gigalitres and can release in excess of 4.6 gigalitres a day. Although these dams were constructed for water conservation and flood mitigation, they also affect the natural regime of smaller floods, but have limited impact on the larger floods. The dams' geomorphic effect of trapping coarse sediment causes 'sediment starvation' over an extended distance downstream and can result in bed lowering and channel expansion. The construction of Glenbawn and Glennies Creek dams has enabled significant increases in farming (i.e. wine grape, dairy, horse breeding) activities and the development of an important power generation industry (DLWC, 1996).

Water allocations in the Hunter River catchment are determined by water sharing plans (WSPs) which, following the introduction of the *Water Management Act 2000,* are being progressively developed for rivers and groundwater systems across NSW. These plans protect the health of the rivers, while also providing water users with greater certainty over future access to water and increased trading opportunities. The WSP for the Hunter area (DIPNR 2004) covers 39 water sources and 58 management zones (see Figure 1). The key rules in the WSP specify when licence holders can access water and how water can be traded.

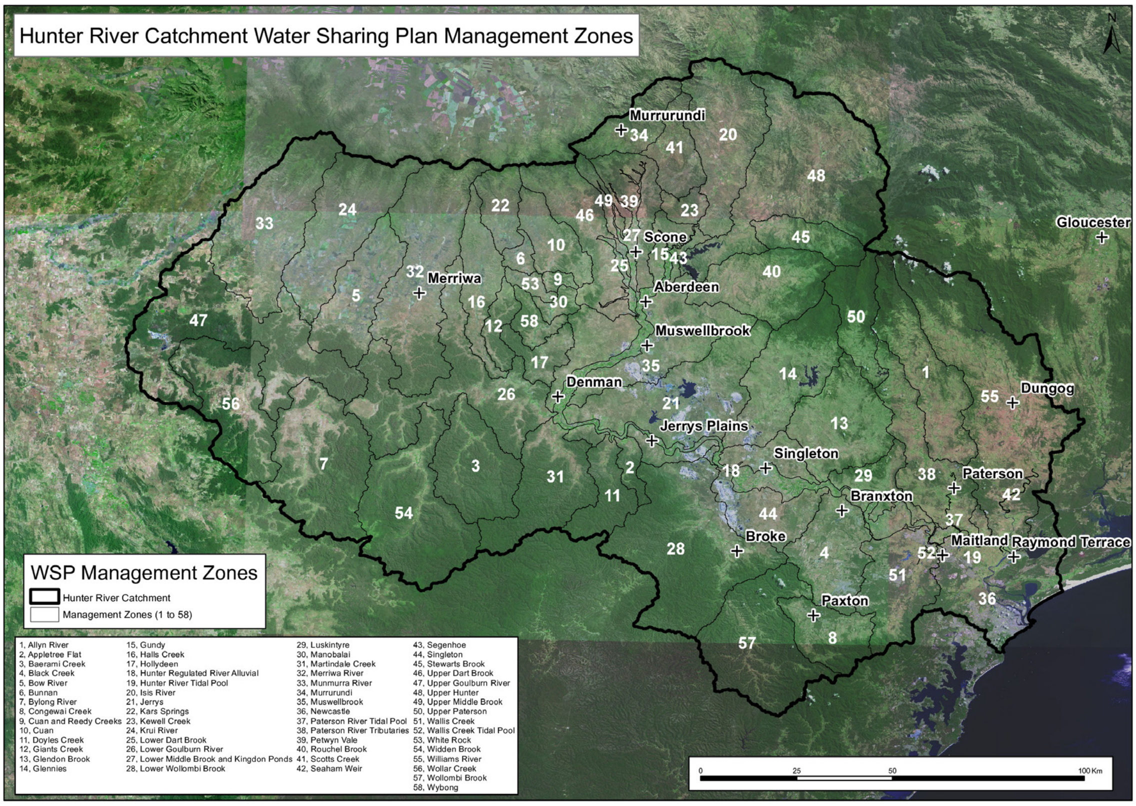

Figure 1: Water sharing plan management zones for the Hunter River catchment

Source: NSW Office of Water

Over the past decade, demand on water use in the Hunter Valley has shifted from predominantly agricultural to predominantly industrial activities. Today mines make up a relatively larger component of the WSP water allocations (over 30 per cent). Some mines use water collected from catchment run-off or dewatering on site for coal washing or dust suppression, and discharge anything unused due to either lack of infrastructure or lack of storage space. Water from catchment run-off is usually only fully utilised on site during dry years.

Since its earliest European settlement in the early 19th century, the Hunter River valley has provided a wide range of often competing land uses, exerting pressure on its natural resources. The Hunter River and its tributaries continue to support important activities including power generation, coal mining, heavy industry, agriculture and associated businesses, infrastructure and fisheries. The Hunter River valley remains the largest coal-producing region in NSW.

3. Sources of salinity in the Hunter catchment

There is a variety of potential sources of salinity in the Hunter River catchment including:

- rainfall
- atmospheric deposition
- run-off and infiltration
- weathering of geological strata
- groundwater, and
- anthropogenic sources (such as mining, power generation, agriculture, urban and peri-urban development, sewage treatment plants, etc.).

Evaporation/evapotranspiration can lead to concentration of salts within the catchment, particularly where widespread land clearing has occurred and where these areas are associated with either existing groundwater discharge zones or where groundwater levels may be rising due to land clearing and increased infiltration and recharge. Kellet *et al.* (1989) noted that during the severe drought conditions of August 1982, banks and low terraces in the lower reaches of almost every minor stream draining the lower Wittingham and Greta coal measures and marine sediments of the Maitland and Dalwood groups were coated by surface salt encrustations (*efflorescences*).

In a study of the salt inputs into the Hunter Valley catchments, Creelman (1994) suggested that rainfall, ions released by rock weathering and mining were the major contributors to salinity in the Hunter Valley catchments. Kellet *et al.* (1989) found that input of groundwater from the Wittingham Coal Measures was also of special significance in terms of salinity contribution to the Hunter River catchment. Kellet *et al.* (1989) concluded that, of all the potential salt sources, geology was the dominant control in the chemistry of upper Hunter River valley groundwater and that high background salinity in groundwater of the Central Lowlands (Jerrys Plains and surrounds) was a natural phenomenon that would persist for the foreseeable future.

Overlaid on the natural cycling of salts in the Hunter River catchment, however, are anthropogenic sources; particularly mining, power generation and agriculture. These activities can either remove salts from the river system (e.g. via water extractions) or add them into the system (via licensed discharges and/or overland run-off). The multiplicity of salt sources and the highly variable spatial and temporal interaction of natural and anthropogenic sources make management of salinity in the Hunter River catchment a very complicated issue. This issue receives even greater focus when the catchment is affected by drought and when competition for sufficient water of suitable quality can become an area of conflict.

Rainfall and atmospheric deposition

The Hunter Valley has a varied climate, depending on elevation and proximity to the ocean. Coastal areas and the area around Barrington Tops receive the highest rainfall: over 1600 millimetres a year at Barrington Tops; and 1140 millimetres a year at Newcastle on the coast. Rainfall decreases with distance inland, with rainfall at Cassilis around 620 millimetres a year. The wettest months away from the coast tend to be December to January. Annual evaporations (> 1300 millimetres) also increase with rising temperatures, generally exceeding the rainfall rates in most parts of the valley. The highly variable nature of the climate in the Hunter Valley has caused both serious droughts and extensive floods (DWE 2009).

Rainfall water chemistry is controlled by a complex interaction of oceanic, mineralogical, geographical, biological and meteorological influences. A number of studies of rainwater chemistry have been undertaken in the Hunter River catchment (e.g. Avery 1984, Rothwell *et al.* 1987, Bridgman *et al.* 1988), although most earlier studies were focused more on acidic deposition and the potential contribution of air pollution sources rather than the salinity (and ionic constituents) of rainfall. The EPA (1994) published rainfall chemistry results for the Hunter River catchment over the period December 1988 to June 1991. These results indicated that the average salinity of rainfall falling in the Hunter catchment during this period was 16.3 microsiemens per centimetre (µS/cm), ranging from a minimum of 2.7 µS/cm at Singleton to a maximum of 89.2 µS/cm at Pokolbin. The major ionic constituents (by concentration) in the rainfall samples were sodium, chloride and sulphate.

Creelman (1994) calculated salt loads for rainfall in the Hunter River catchment based on salt levels [in milligrams per litre (mg/L)] and average annual rainfall. Creelman (1994) suggested that the average salt yield from rainfall was:

- 22 tonnes/km^2/year (range 12–30 tonnes/km^2/year) in coastal areas
- 14 tonnes/km^2/year (range 9–19 tonnes/km^2/year) in the lower Hunter Valley
- 8 tonnes/km^2/year (range 4–14 tonnes/km^2/year) in the mid Hunter Valley, and
- 6 tonnes/km^2/year (range 3–10 tonnes/km^2/year) in the upper Hunter Valley.

Collectively these results indicate there can be significant spatial variability in atmospheric deposition of salts across the Hunter River catchment, with salt loads generally decreasing with increasing distance from the coast.

Run-off and infiltration

Day (1986) provided a good general description of run-off and water transport processes in the Hunter River catchment. Once rainfall hits the Earth's surface it can infiltrate into the soil profile, flow off as overland flow (run-off) and/or be intercepted by soil or vegetation. Soil moisture taken up by vegetation can subsequently be lost as evapotranspiration, and rainfall intercepted by vegetation can be lost by evaporation. Water may flow through soils as unsaturated throughflow and re-emerge in tributary channels or collect in depressions in the landscape. It may then further infiltrate into bedrock fissures and flow downslope to the main stream channel responding to hydraulic gradients. Unsaturated soil may become saturated as precipitation increases and saturated areas may extend and contract along and beside tributary channels (Day 1986). Saturated throughflow moves under gravity within the lower valley sides and some of this water may move over the land surface and return to the stream channel. Water may also move from the saturated soil profile into groundwater aquifers which can themselves move into the drainage lines as baseflow drainage from saturated bedrock and soil (Day 1986). At each step in this process the water originating as rainfall can accumulate additional salts depending on the nature and characteristics of the soil, rock, aquifer, vegetation and stream channel involved.

European land-use changes within the Hunter River catchment date from the early 1800s when considerable vegetation was cleared along the major rivers (Day 1986). Later clearing for grazing and agriculture, particularly on the more fertile soils, and mining in areas where Permian coal seams outcrop or are close to the surface, have led to the current patchwork of land uses and vegetation cover across the Hunter River catchment. A map of current land use in the Hunter River catchment is included as

Figure 2[1]. The vegetation and land-use changes that have occurred since European settlement have in turn caused many changes to the natural run-off and infiltration processes in the catchment. Kellet *et al.* (1989) noted that some point sources of natural salt contamination existed prior to European settlement in the Central Lowlands or were contemporaneous with it, as evidenced by early geographic names with salinity connotations – such as Saltwater Creek. Kellet *et al.* (1989) also suggested that forest clearing had exacerbated degradation of the land by promoting salting under conditions of increased run-off, erosion and rising water tables in the Central Lowlands.

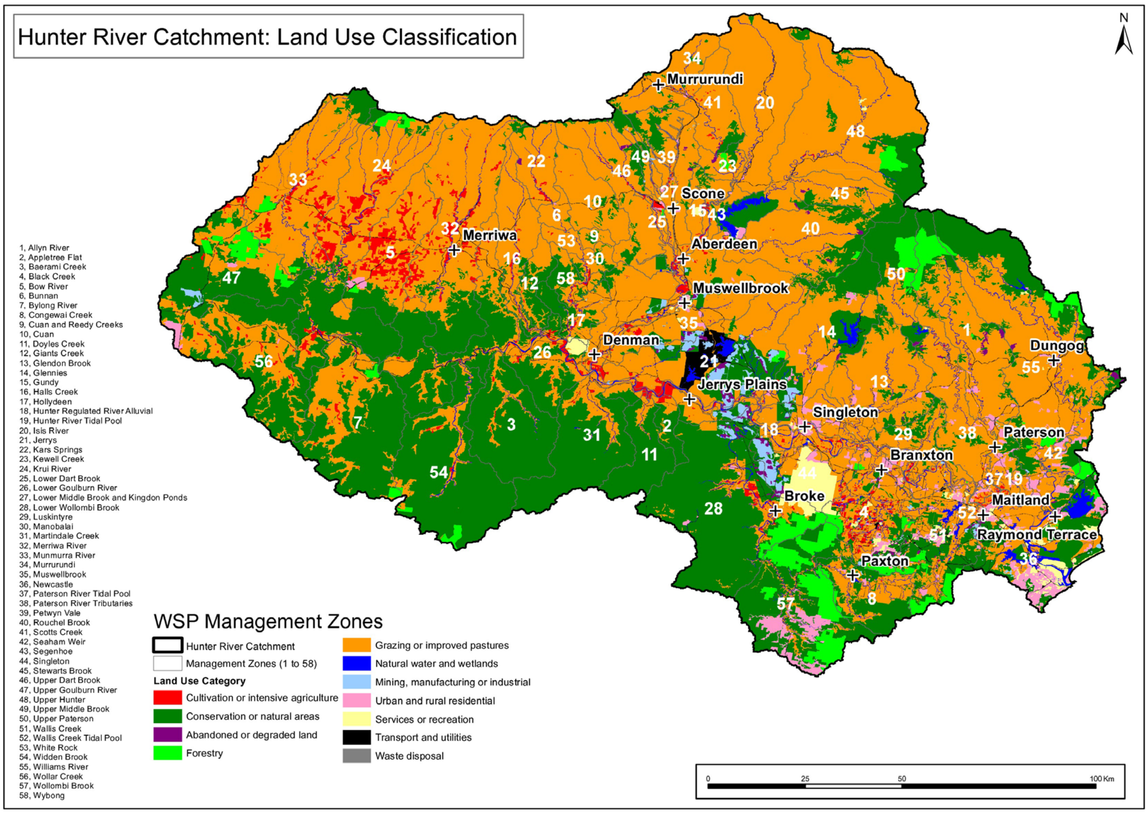

Figure 2: Land use categorised for the Hunter River catchment

Source: NSW Spatial Data Catalogue

Higher rainfall since the late 1940s has also been suggested as a cause for altering the geomorphology of the Hunter River and some of its major tributaries, with additional impacts on erosion, sedimentation and flow pathways (Day 1986, Erskine and Bell 1982, Erskine 1994). Significant changes in riverbed morphology continue to occur where the Hunter River has deepened in some areas, and changes to vegetation on river banks and within the riverbed have contributed to decreased flow velocities in other areas (NOW 2012). Rating table shifts at the gauging stations can also occur where the control is relatively unstable (e.g. sand) and there is a slow gradual shifting of that control. Floods in the 2011–12 period, mainly due to the flood in the Goulburn River in March 2012 (which affected the middle and lower sectors) and the flood in the Upper Hunter in November 2011, had an effect on rating tables (NOW 2012). Large changes were also noted at Singleton and downstream sites after flood flows in June

[1] Summary statistics for the percentage of land use categories in each water sharing plan subcatchment management zone are included in Table A1, Appendix A.

2011 (NOW 2012). There is a need to recognise that the Hunter River is itself a dynamic system.

Weathering of geological strata

The Hunter River valley occupies part of four major geological provinces of eastern Australia: the New England Geosyncline in the north-east; the Sydney Basin in the centre and south; the Great Artesian Basin in the north-west; and the East Australian Tertiary Volcanic Province in the north and west. (Galloway 1963). A map of the underlying geology of the Hunter River catchment is included as Figure 3[2].

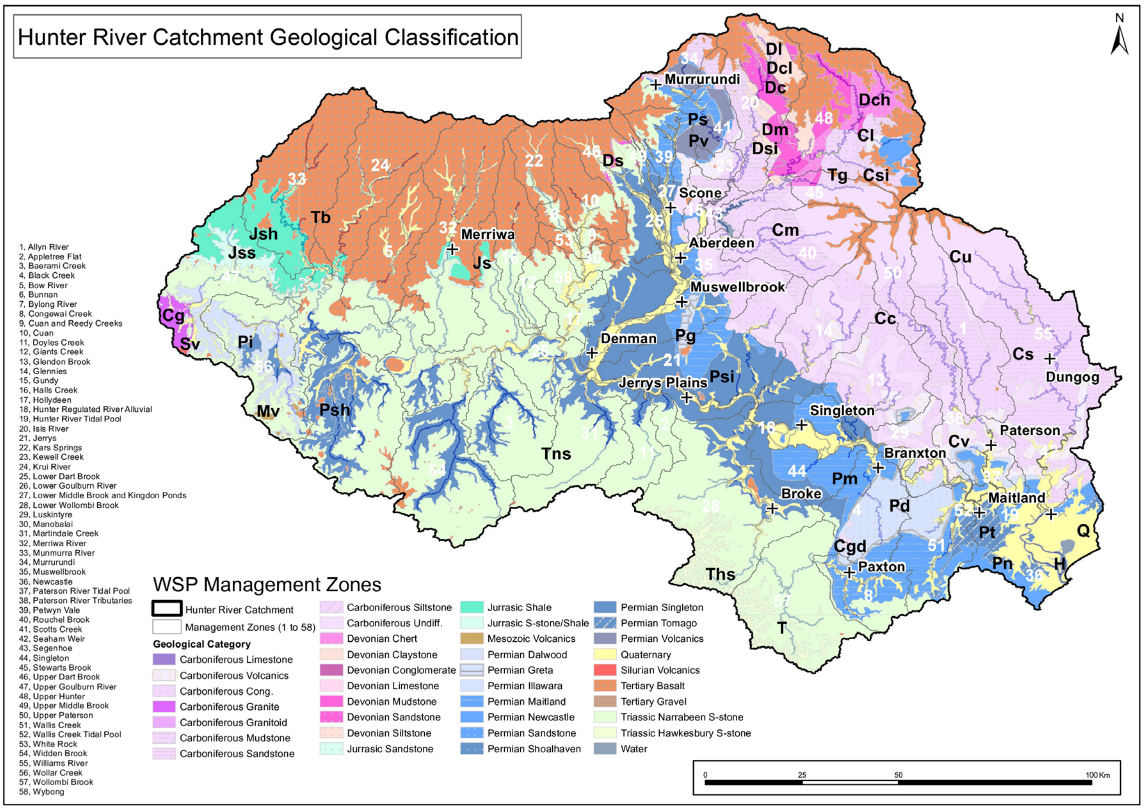

Figure 3: Geological classification of the Hunter River valley

Source: Geological Survey of NSW 1:250,000 Geological Maps

Kellet *et al.* (1989) described the geology of the Hunter River catchment as follows:

> *The Southern Mountains region consists of rugged Triassic sandstone mountains up to 1000 m and deeply incised valleys; soils are generally shallow because the Triassic sandstones are resistant to chemical weathering. At the junction of the Southern Mountains with the Central Lowlands, discontinuous sheets of quartzose sand fan out from the foothills for 1–2 km.*
>
> *To the west, the Central Goulburn valley region is similar to the Southern Mountains; it is underlain by Triassic sandstones and shales, and consists of irregular steep-sided hills and plateaus, and deeply incised rivers.*

[2] Summary statistics for the percentage of various geological categories in each water sharing plan subcatchment management zone are included in Table A2, Appendix A.

The Merriwa Plateau region in the north-west consists of rolling to hilly terrain, developed as a planation surface on extensive Tertiary basalt flows. Lavas have partially filled pre-existing valleys, and post-volcanic streams have incised on either side of the flows, forming sub-parallel valleys that reflect the pre-basalt drainage. The degree of incision has been controlled by uplift, and by marked variations in relative sea level during the Cainozoic.

Resistant folded Devonian and Carboniferous lavas and sedimentary rocks form the Northeastern Mountains — rugged, dissected terrain, which diminishes in relief towards the Central Lowlands. This region is fringed to the north-east by the Liverpool and Mount Royal Ranges, and by the Barrington Tops, which comprise rugged basaltic terrain of small plateaus interspersed with narrow, steep-sided crests and valleys.

In contrast, the Central Lowlands have a gently undulating terrain developed on easily eroded Permian sedimentary rocks, on which deep soils have developed. The region is, therefore, important agriculturally, but is also important economically for its mineable coal deposits. Permian rocks occupy approximately one-fifth of the Hunter Valley and extend in a central belt from Newcastle to Murrurundi. Outlying occurrences are also found in the west where southern tributaries of the Goulburn River have stripped off overlying Triassic sandstone. (Galloway 1963)

The Hunter River itself and many of its tributaries are bounded by alluvial flats from 1–6 km wide. Cainozoic sediments occur as unconsolidated alluvial deposits of the Hunter River floodplain and, to a lesser extent, the alluvial terraces of the major tributary streams. These deposits are generally composed of basal gravels and boulders overlain by an upward-fining sequence of sands, silts, and clays with sporadic shoestring gravels. Secondary pedogenetic pore-filling of the Cainozoic sediments reduces porosity and becomes significant on the oldest (and highest) terraces.

Weathering of geological strata can liberate not only salts within the rock itself but also salt associated with old marine transgressions (often referred to as *connate* salt) that have remained stored within the geological profile. Some of the Permian coal measures are especially important in this context, in particular the Greta and Wittingham coal measures. These geological strata originally formed as peat swamps on alluvial fans close to the sea (Kellet *et al.* 1989). Within the Wittingham Coal Measures two brief marine transgressions are recorded by laminites of the Bulga and Denman formations (Kellet *et al.* 1989) and these are considered to be major sources of connate salts in the associated groundwater. The Wollombi Coal Measures show the least marine influence of the Permian deposits (Kellet *et al.* 1989).

Creelman (1994) provided a first estimate of salt release from the various rock units in the Hunter Valley (see Table 1). Salt releases from the Triassic rocks averaged 5 tonnes/km^2/year; the Carboniferous metavolcanics and glacial sediments averaged 4–5 tonnes/km^2/year; the Wollombi Coal Measures averaged 4–5 tonnes/km^2/year; the Greta Coal Measures averaged 30 tonnes/km^2/year; and the Wittingham Coal Measures averaged 40 tonnes/km^2/year.

Table 1: Salt release (tonnes/km^2/year) from the various rock units in the Hunter Valley (Figures in brackets indicate possible ranges)

Unit	Description	Included units	Salt release by erosion (tonnes/km^2/year)
CARB	Carboniferous volcanoclastic and glaciene sediments of the southern New England Fold Belt.	Carboniferous Rouchel and Gresford blocks.	5 (4–8)
GM	Interbedded coal seams and continental sediments of the Greta Coal Measures, marine sediments of the Maitland and Dalwood groups.	Greta Coal Measures, Mulbring Siltstone, Muree Sandstone, Branxton Formation, Gyarran Volcanics.	30 (25–80)
WI1	Upper Wittingham Coal Measures west of the Muswellbrook Anticline in the north, and near the 'Triassic' escarpment in the south.	Denman Formation, Jerrys Plains Subgroup.	40 (15–40)
WI2	Lower Wittingham Coal Measures east of the Muswellbrook Anticline in the north, and near the 'Triassic' escarpment in the south.	Archerfield Sandstone, Vane Subgroup, Saltwater Creek Formation.	40 (30–60)
WO	Interbedded coal seams and continental sediments of the Wollombi Coal Measures.	Wollombi Coal Measures.	5 (8–12)
TRIAS1	Lower Triassic conglomerate, sandstone, and shale of the Narrabeen Group.	Triassic Narrabeen Group. Mainly in the south of the Hunter and Goulburn valleys.	4 (8–10)
TRIAS2	Narrabeen Group overlain in places by Tertiary basalt in the north of the Goulburn/Hunter River confluence.	Triassic Narrabeen Group with Tertiary basalt flows, mainly to the north of the Goulburn Valley.	4 (8–10)
HFP1	Alluvium upstream of the Goulburn/Hunter confluence.	Recent alluvium, mainly coarser sediments.	Taken as 0
HFP2	Alluvium downstream of the Goulburn/Hunter confluence.	Recent alluvium, mainly finer sediments, but with coarser units at depth.	Taken as 0
Basalt	Tertiary basalts within the Hunter Valley.	All basalts, but excluding the plateau basalts of the Goulburn Valley.	2

Source: Creelman (1994).

Groundwater

Williamson (1958), Griffin (1960), Ringis (1964) and Kellet *et al.* (1989) have all discussed groundwater resources in various parts of the Hunter River catchment. Kellet *et al.* (1989) considered the Hunter River floodplain to be a regional groundwater sink for the Permo–Triassic fractured-rock aquifers; with bed underflow of the Hunter River representing a dividing streamline for groundwater flow, apart from a few important mixing zones. Contrasts in permeability and porosity between the alluvial and fractured-rock aquifers indicated that most groundwater in the upper Hunter River valley was stored in and transmitted through the floodplain sediments.

Several surface reservoirs in the catchment act as sources for groundwater recharge; although they can also impede lateral throughflow from upgradient groundwater stores, and in some areas create springs and artesian conditions (Kellet *et al.* 1989). The largest reservoirs are Glenbawn Dam, Glennies Creek Dam and Lake Liddell (148,000 megalitre capacity). Lake Liddell is filled with a mixture of water pumped from the

Hunter River, run-off, interflow, and groundwater from the 75 square kilometre upper catchment of Bayswater Creek. The upper Bayswater Creek catchment drains Maitland Group rocks on the eastern limb of the Muswellbrook Anticline (Kellet *et al.* 1989).

On a regional scale, groundwater of the fractured-rock aquifers constitutes only a minor proportion of storage and transmission, but the reserves are most important during times of low flow of the Hunter River and its tributaries (Kellet *et al.* 1989). Streams with identified groundwater interactions were often found to have very high salt loads. For example, the Saltwater Creek catchment was found to be releasing approximately 230 tonnes/km^2/year of salt, potentially as a result of groundwater–surface water connections due to the saltwater thrust which traverses Saltwater Creek (Creelman 1994). Groundwater in the Permian fractured-rock aquifers is also very important to the coal mining industry since it can form a large component of the mine water being discharged as part of the Scheme.

Alluvial aquifers are also important in the Hunter River catchment. Basal gravel and overlying sand of the floodplain alluvium of the Hunter River and its major tributaries are by far the most permeable aquifers in the study area (Kellet *et al.* 1989). In many places groundwater quality and yields are sufficient to permit intensive crop irrigation.

Kellet *et al.* (1989) divided groundwater of the upper Hunter River valley into eight hydrochemical provinces, characterised by groundwater of distinctive chemical composition, stored in and transmitted through particular rock and/or soil associations. This yielded information on the major ionic constituents of groundwater derived from differing geological and soil units (see Table 2). A more recent treatment of hydrogeology of the Hunter River catchment is given by Mackie (2009). Mackie (2009) gathered additional data on groundwater chemistry and noted considerable overlap in groundwater ionic composition for the hydrochemical provinces identified by Kellet *et al.* (1989). This led Mackie (2009) to develop a different generalised characterisation of groundwater in the Upper Hunter region (see Table 3). While differences exist in the general characterisations of groundwater hydrochemistry between Kellet *et al.* (1989) and Mackie (2009), they both indicate the importance to the Hunter River catchment of groundwater contributions to salinity of differing ionic composition.

Table 2: Groundwater hydro-chemical provinces of Kellet *et al.* 1989

Province name	Description	Dominant hydrochemical species
HFP1	Alluvium of the Hunter River floodplain between Glenbawn Dam (north east of Scone) and the Goulburn River confluence near Denman	Mg, Na, HCO_3 , Cl
HFP2	Alluvium of the Hunter River floodplain between the Goulburn River confluence and Singleton	Na, Mg, HCO_3 , Cl
TRIAS	Lower Triassic conglomerate, sandstone and shale of the Narrabeen Group, overlain in places by Tertiary basalt in the west and south-west of the area	Na, Mg, Cl
WO	Interbedded coal seams and continental sediments of the Wollombi Coal Measures	Na, Mg, Cl, HCO_3
WI1	Upper Wittingham Coal Measures west of the Muswellbrook Anticline (in northern areas), and near the Triassic escarpment (in southern areas)	Na, Cl, HCO_3
WI2	Lower Wittingham Coal Measures east of the Muswellbrook Anticline (in northern areas), and proximal to the Maitland Group rocks (in southern areas)	Na, Cl
GM	Interbedded coal seams and continental sediments of the Greta Coal Measures, and marine strata of the Maitland and Dalwood Groups	Na, Mg, Cl, SO_4
CARB	Carboniferous volcaniclastics and glacigene sediments of the New England Fold Belt in the north and north-east of the area	Na, Mg, Cl, HCO_3

Source: Mackie (2009)

Table 3: Groundwater hydro-chemical provinces of Mackie (2009)

Shallow alluvium associated with the Hunter River	Ca>Na>>Mg and HCO_3>Cl>>SO_4	Becoming increasingly Na-Cl type waters downstream from Muswellbrook
Shallow alluvium associated with Goulburn River	Na>Ca > Mg and Cl >HCO_3>SO_4	Primary salinity dominates near the confluence with Hunter River as a result of drainage across a large catchment.
Shallow alluvium associated with minor drainages	Ca-Na>Mg and HCO_3>Cl>>SO_4	Generally reflects localised conditions - HCO_3 contributions may derive from volcanics in some catchments
Wollombi Coal Measures	mixed dominance Ca-Na-Mg and HCO_3-Cl-SO_4	Older measures in the Bulga-Broke area may exhibit increased salinity with no dominant species.
Wittingham Coal Measures	mixed dominance Ca-Na-Mg and HCO_3-Cl-SO_4	Older measures may exhibit increased Cl-SO_4 salinity due to proximity to Maitland Group (marine and lower deltaic conditions).
Greta Coal Measures	Ca>Mg>Na and Cl>SO_4>HCO_3	Generally higher SO_4 and Cl due to proximity to Maitland Group (marine and lower deltaic conditions).
coal seam variation	Ca>Mg>Na and Cl>HCO_3>SO_4	Localised influence of cleat secondary mineralogy (eg. calcite, siderite) and CO_2-CH_4 leakage to observation bores
leakage from structural features (faults, bedding flexures etc.)	Ca>Na>Mg and Cl>HCO_3> SO_4	Influenced by secondary mineralisation within leakage pathways

Source: Mackie (2009)

Salt-affected areas

Kellet *et al.* (1989) provided the following summary of salinity affected areas and salt 'efflorescences' in the Hunter River valley:

> *Most salt efflorescences occur in the Wollombi Brook valley between Broke and Singleton. The Mulbring Siltstone, in particular, generates a large number of salt scalds and salt-affected streams. Salinisation of soils, streams and groundwater in the Central Lowlands is closely related to rock type, and the intensity of halite salting is greatest in provinces where groundwater has the strongest connate-marine signature. Some point sources of natural salt contamination existed before European settlement in the Central Lowlands or were contemporaneous with it, as evidenced by early geographic names with salinity connotations – such as Saltwater Creek. Forest clearing has undoubtedly exacerbated degradation of the land by promoting salting under conditions of increased run-off, erosion and rising water tables in the Central Lowlands.*
>
> *The second category of salt efflorescence in the upper Hunter River valley appears to be controlled in part by geological and geomorphological features. The efflorescences consist of small-scale salt scalds extending downstream from ephemeral springs along the nick point separating the upper and lower pediments on the north-eastern and southern sides of the Hunter River valley. Salt crusts in these areas comprise patchy impure films, 1 mm to 2 mm thick, which drape tunnelled dispersive clays. Most springs at the head of the salt scalds on the north-eastern side of the Hunter River valley – between Bowmans Creek and the headwaters of Bettys Creek – are roughly coincident with the Hunter Thrust Fault and probably represent saline water upwelling from the underlying Wittingham Coal Measures. However, similar springs on the western footslopes of Mount Surprise, and in the hills above Muscle and Grasstree Creeks occur in Carboniferous rocks at least 3 km from the Hunter Thrust Fault. Most salt scalds at the change of slope on the southern side of the valley between Alcheringa and Bulga seem to emanate from intermittent saline springs at the contact between the Wollombi and Wittingham Coal Measures. In August 1982, despite the drought, one of these springs was still flowing from the base of the Watts Sandstone (lowest member of the Wollombi Coal Measures) above Appletree Creek. Other salt scalds in this area appear to be related to springs and seepages at the contact between the basal fanglomerate of the Narrabeen Group and the underlying Wollombi Coal Measures.*

A current map of salinity affected areas in the Hunter River catchment is included in Figure 4[3], with black areas indicating known areas of land affected by dryland salinity. Areas of high salinity risk (dark orange/red areas in Figure 4) usually coincide with areas underlain by the Permian Coal Measures. Approximately 80 per cent of salinity in the Hunter catchment is attributed to diffuse sources (EPA 2001, ACARP 2004). A salinity audit completed in 2000 (DLWC 2000) predicted the salt load for the Hunter River at Singleton gauge to exceed 150,000 tonnes per year by 2010, almost 50 per cent of which was contributed by the upper sector.

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[3] Summary statistics for the percentage of salt-affected land and salinity risk in each water sharing plan subcatchment management zone are included in Table A3, Appendix A.

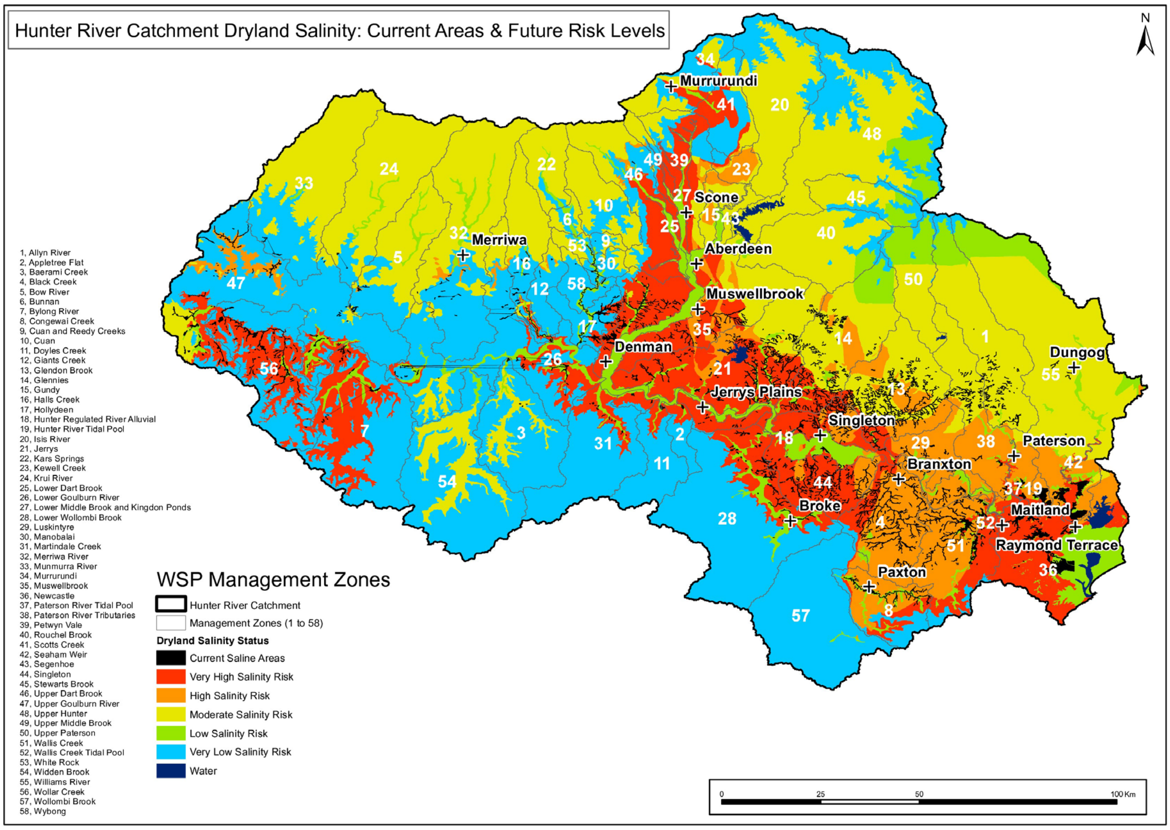


Figure 4: Areas of dryland salinity and salinity risk within the Hunter River catchment

Source: Hunter–Central Rivers CMA Salinity Hazard Map and OEH Known Salinity Areas Data Layer

Anthropogenic sources

A range of anthropogenic actions have the potential to influence salt concentrations and loads in the Hunter River catchment. The Hunter River valley is generally considered to be saline due to the marine origin of some of its Permian sediments, where ground water EC concentrations can reach ~7000 µS/cm. However, recent land-use activities in the catchment may have contributed to rising groundwater levels in some areas and an increase in the salinity load reaching streams (DLWC 2000 salinity audit). Diffuse sources from past land usage and known saline areas have been discussed above. While rises in river salinity in some locations may largely be attributable to natural processes (>75 per cent of current lower Hunter River salt levels have been attributed to natural processes; ACARP 2004), the remainder have been attributed to anthropogenic activities including 10 per cent current/former mining operations (ACARP 2004). Licensed discharges are a major potential source of salt in the catchment. These are distributed throughout the catchment and include licensed discharges from mines and power generators; licensed discharges from council and Hunter Water sewage treatment plants; and licensed discharges from manufacturing industries. It is important to note that not all of these discharges are captured under the Scheme, as the Scheme only applies to salt discharges in the Hunter River between Glenbawn Dam and Singleton. Discharges not covered by the Scheme are managed by individual environment protection licences. These latter discharges have not been considered in great detail for the current assessment, but they do have the potential to affect salinity levels in some areas of the catchment (e.g. the Goulburn River).

4. Hunter River Salinity Trading Scheme operation and salt loads

As identified earlier, the Scheme is implemented under the Regulation. There are a total of 1000 salt discharge credits in the scheme with different licence holders having different numbers of credits. Licence holders can only discharge salt into a river block in proportion to the credits they hold – 1 credit allows a discharge of 0.1 per cent of the total allowed. Discharge credits can also be traded. Credit trading gives each licence holder the flexibility to increase or decrease their allowable discharge from time to time while limiting the combined amount of salt discharged across the valley (DEC 2006).

Participants in the scheme are licensed by the Environment Protection Authority (EPA). The environment protection licence defines the discharge points and the monitoring and reporting requirements. Any licence holder discharging outside the limits of the Scheme is violating their licence conditions, and penalties apply. The Regulation contains the Scheme rules and additional safety measures, such as discounting the value of credits if too many are traded into the one river sector.

The New South Wales Office of Water (NOW) currently operates monitoring, telemetry and modelling components of the Scheme under a service agreement with the EPA. NOW currently monitors flow and electrical conductivity (EC) in the Hunter River and its major tributaries through gauging stations at 21 locations across the catchment. Information collected from the gauging stations is transmitted through a telemetry system and used to model flow and conductivity at the three Scheme reference sites along the Hunter River. The EPA provides an online credit trading facility.

A predictive model run by the State Water Corporation (SWC) estimates the total allowable discharge (TAD) of salt to enable conductivity levels in the Hunter River to remain below set limits of 600 µS/cm in the upper, and 900 µS/cm in the middle and lower reference sectors of the river. The SWC, under an agreement with NOW, models river flow and salinity to determine saline water discharge opportunities and notify Scheme participants. When an approaching flow event is identified, the river operator models river flow and conductivity to predict TAD, timing of saline water discharges into the Hunter River and its tributaries. A River Register is then published as an authorising document to notify licence holders of the amount and timing of saline water discharges allowed, whereby each participant can calculate their share of the TAD and discharge accordingly. An important component of the scheme is its transparency; real-time flow and EC data are available on the NOW website for public viewing and scrutiny.

Salt loads

During the Pilot Salinity Trading Scheme, which operated from 1 January 1995 to 30 November 2002, the average annual discharge of salt by Scheme participants was 18,233 tonnes a year. The NSW Coastal Rivers Salinity Audit, in December 2000, estimated the average annual salt load for the Hunter River at Singleton as 149,500 tonnes a year (DLWC 2000). Saline wastewater discharges therefore contributed on average approximately 12 per cent of the entire annual salt load of the Hunter River at Singleton during the Pilot Scheme period.

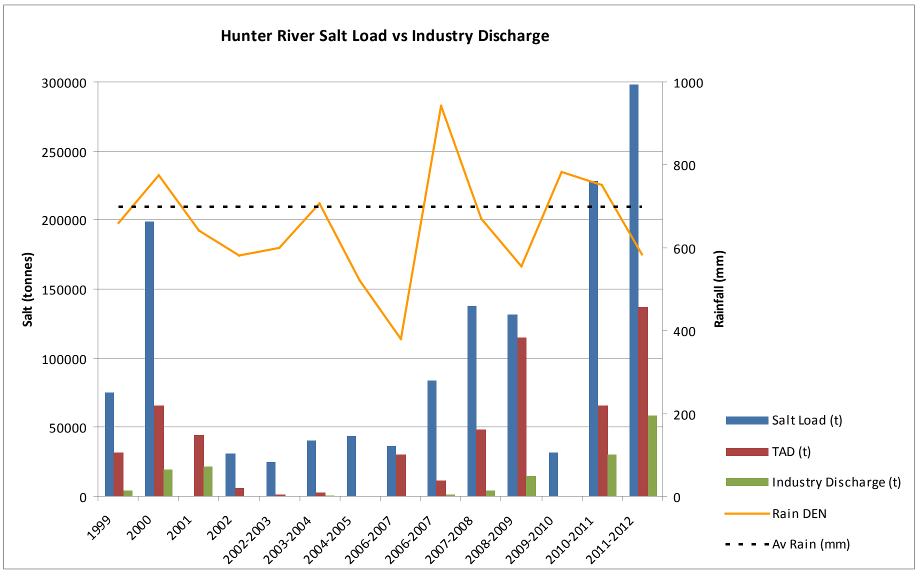


Figure 5: Total salt load carried by the Hunter River past Singleton gauge and allowable discharge opportunities and loads under various climatic conditions

Table 4: Salt discharge utilisation versus total allowable discharge(TAD) for the Hunter River at Singleton gauge (210001/210129)

Source: NSW Office of Water**Year**	**River salt load (tonnes)**	**TAD (tonnes)**	**Industry salt discharge (tonnes)**	**Rainfall at Denman (mm)**	**TAD utilised**
1999	75707	32152	4689	658.4	15%
2000	199652	66239	19693	773.9	30%
2001		44561	22337	640.4	50%
2002	31518	6204	217	580.9	3%
2002–2003	25404	1678	335	598.8	20%
2003–2004	40628	3412	891	706.4	26%
2004–2005	44149	351	170	520.6	48%
2005–2006	37186	30653	0	378.6	0%
2006–2007	83807	12027	1219	943	10%
2007–2008	137892	48585	4884	670.2	10%
2008–2009	131584	115669	14790	556	13%
2009–2010	31958	0	0	781.8	
2010–2011	228713	65940	30987	751.2	47%
2011–2012	298502	137543	59035	583.9	43%

Source: NSW Office of Water

Since the implementation of the Regulation in 2002, there has been a slight increase in the number of participants, and, on average, industry participants have utilised 25 per cent of the given opportunities (see Table 4). Over the past decade, the annual rainfall across the catchment ranged between 350 and 900 millimetres, providing Scheme participants with almost 250 allowable discharge events. During this time participants discharged approximately 112,500 tonnes of salt out of a total allowable discharge of 422,000 tonnes (26.7 per cent utilisation). Overall, a total of approximately 1.1 million tonnes of salt was carried by the Hunter River past the Singleton gauge (so the

Scheme contributed on average approximately 10 per cent of the total salt load). It is worth noting that this period also included one of the most significant drought periods on record, from 2002 to 2007 (See Figure 5). An increased frequency of rainfall events in recent times has led to increased utilisation rates – close to 50 per cent of the published TAD (Table 4).

Summary

The Scheme was an important response to catchment and river salinity levels. Under the Scheme, participants only discharge at higher flows to enable a greater dilution of the saline discharges to occur. On average, the Scheme contributed approximately 10 per cent of the total salt load at Singleton since 2002. However, in recent times significant salt loads (approximately 30,987 to 59,035 tonnes or 13.5 to 19.8 per cent of total annual salt load in the Hunter River at Singleton) have been discharged to the Hunter River and the value of salt credits is increasing. At the same time the utilization of the TAD has also increased (to 40–50 per cent). Additional demand for saline discharge is also coming from new or expanded mining operations. For example, as part of GlencoreXstrata's (Mangoola Coal Mine) bid to increase extraction from 10.5 million to 13.5 million tonnes of coal a year, the mine is applying to discharge 50 megalitres a day over set periods under the Scheme (EMM 2013; Newcastle Herald May 30, 2013). A number of other mine expansions with increased daily discharge volumes are also being proposed for various parts of the Hunter River catchment. It therefore appears there is currently an increasing demand. Although capped by the Regulation salinity targets and total credit allocation, this demand should continue to be monitored and assessed.

5. Groundwater and surface water – state and trend

Groundwater state and trend

The extensive dependence of users on the groundwater resources of the Hunter River catchment requires careful management. This includes ensuring equitable sharing of the resource between water users and protection of groundwater bores from intrusion of saline water from the underlying and enveloping hard rock aquifers associated with the Permian Coal Seams (DIPNR 2003).

Bore levels

Conaghan (1948), Williamson (1958), AGC (1967), Ainsworth (1994), DLWC (2001) and DIPNR (2003) have all reported on the groundwater resources within the Hunter River catchment. DIPNR (2003) analysed water table relationships with rainfall and stream flow finding a significant relationship between the upper catchment rainfall and the resulting bore groundwater levels. During drought, water table declines of around 5.5 metres have been recorded, which is similar to the declines reported by Williamson (1958) during the 1935–47 drought. This drop in water table had the effect of halving the aquifer throughflow and discharge volumes. Williamson (1958; see Table 5) summarised the groundwater resources in the alluvial sections of the Hunter River valley. This study was conducted during the Glenbawn Dam construction phase.

Table 5: Groundwater depth and water quality across the upper–middle Hunter catchment

Site/Reach name	Well depth (feet)	Well depth alluvium (feet)	Water quality
Hunter River at Glenbawn to Pages River confluence	20–48 [6.1–14.6 m]	60 [18.3 m]	Generally good with marginal poor-quality areas
Downstream of Glenbawn Dam to Aberdeen	36 [11 m]	20–60 [6.1–18.3 m]	Water becomes more suitable for irrigation further downstream of the dam
Kingdon Ponds and Dartbrook Creek systems	18–42 [5.5–12.8 m]	49 [14.9 m]	Generally satisfactory for irrigation with few marginal poor-quality areas
Kingdon Ponds and Dartbrook Creek confluence to Hunter River	10–40 [3–12.2 m]	50 [15.2 m]	Extensive areas of poor quality groundwater
Hunter–Dartbrook confluence to Muswellbrook	30–40 [9.1–12.2 m]	25–30 [7.6–9.1 m]	Groundwater salinity is above levels recommended for irrigation
Muswellbrook to U/S Denman	40–55 [12.2–16.8 m]	50–60 [15.2–18.3 m]	Poor groundwater quality for irrigation
Upstream of Denman to Goulburn River confluence	30–50 [9.1–15.2 m]	80 [24.4 m]	Groundwater salinity is above levels recommended for irrigation

Source: Adapted from Williamson (1958)

In the current assessment, historic groundwater levels from a number of reports (and bore surveys within the catchment) were collated and compared to recent data. Most bores with longer term data were found to be in the alluvial parts of the catchment and their levels generally fluctuated in concert with seasonal variability. Downstream of Glenbawn Dam, however, bores appear to have lost their strong connection with seasonal drivers and levels often remain relatively unchanged as they are currently being recharged by the regulated Hunter River flows. Figure 6 shows the long-term variation in groundwater levels in several bores located across the Hunter River catchment. Bore levels declined between 2003 and 2007 during the drought, but recovered during the 2007 catchment-wide rainfall events. This was followed by the 2010–2011 events when further recharge of groundwater aquifers occurred.

Williamson (1958) found groundwater levels around the tributaries fluctuated with seasonal conditions, while alluvial levels were sometimes reported to be lower than groundwater levels in areas further from the river. This latter result may be an effect of the base of the alluvium not being a flat surface, with groundwater being ‘trapped’ or ‘backed-up’ by bedrock highs leading to the impression of higher water tables away from the river. There is also the possibility that the river loses water in some sections (losing stream section), while in others it gains water (gaining stream section), particularly where there is active groundwater discharge into the alluvium. The detailed interaction between groundwater and surface water in many parts of the Hunter River catchment still requires further research.

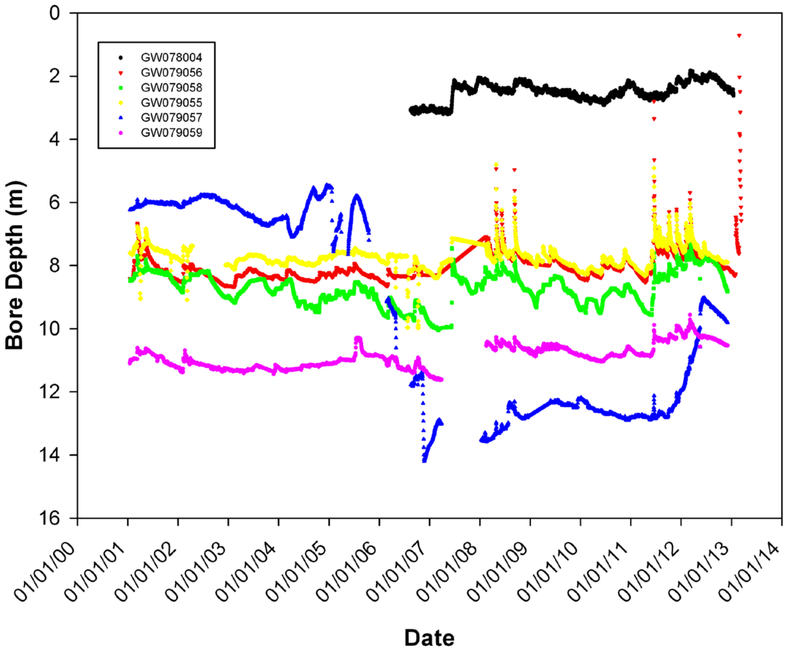


Figure 6: NOW groundwater monitoring bores across the Hunter catchment

Source: NSW Office of Water

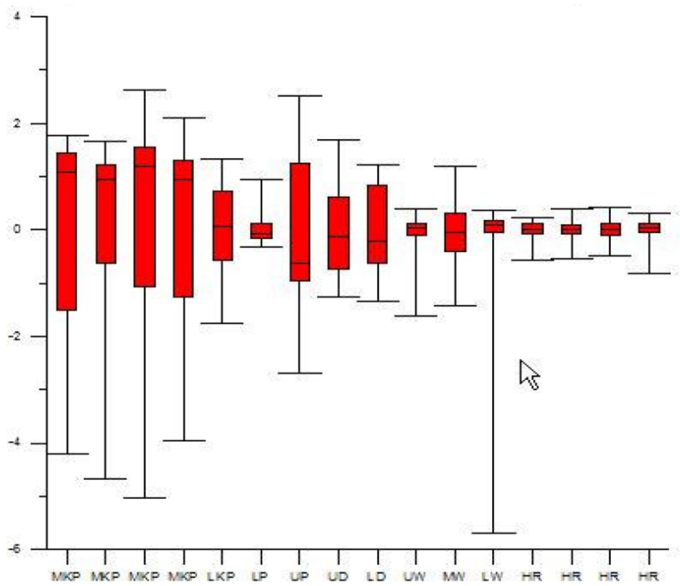


Figure 7: Water level fluctuations (in m) from the Hunter Valley alluvial monitoring bores in the Hunter River (HR); Upper Dart Brook (UD); Lower Dart Brook (LD); Mid Kingdon Ponds (MKP); Lower Kingdon Ponds (LKP); Lower Pages River (LP) Upper Pages River (UP); Mid Wybong Creek (MW); and Lower Wybong Creek (LW)

Source: J. Williams, NSW Office of Water

The impact of river regulation in the alluvial sections of the Hunter River (HR) can be seen in Figure 7, where most bore levels show relatively little variation as a result of the regular supply of river water for recharge. It also illustrates the higher variability in groundwater levels in bores across the unregulated streams such as the Upper and

Lower Dartbrook (UD, LD) and Kingdon Ponds (MKP) areas over the period of record. Earlier studies (e.g. DIPNR 2003) found bores in some Hunter River tributary catchments to have a stronger relationship with upper catchment rainfall rather than rainfall in surrounding areas. AGC (1967) actually suggested effective recharge of the complete system required a period of three wet years. The potential presence of severed hydraulic connections between creeks and water tables as a result of mining in some areas, however, could effectively require even longer periods of wet years for complete recharge to occur. Overall, when recent groundwater level changes are compared to those of historic records, similar patterns are evident. This is largely attributed to bores being either connected to the alluvial sections of the streams and recharging through regular stream flow or recharging during significant storm events when upper catchment surface run-off is greatest.

Groundwater salinity levels

A map of the historic groundwater monitoring network across the Hunter catchment is illustrated in Figure 8. Conaghan (1948) sampled over 300 bores for conductivity and other chemical parameters around the upper parts of the Hunter River, lower Dartbrook and sections of the Pages River (e.g. Figure 9). Samples taken near the river flats often had lower conductivity levels than those taken from the slopes of the valley. These findings were supported by the studies of Ainsworth (1994) and DIPNR (2003). AGC (1967) and Williamson (1958) observed poor quality groundwater along marginal zones during and immediately after the onset of wet periods. This was suggested to be caused by groundwater storage and flow into the alluvium from the long piedmont slopes and was related to the presence of bicarbonate-rich, sulphate-poor and hard waters sourced from Tertiary basalts. In contrast, Ainsworth (1994) reported good quality water in bores tapping the alluvium where flows from the piedmont were negligible, resulting in lower EC levels. As identified earlier, EC in hard rock aquifers associated with the Hunter Coal Seams can at times be high, ranging between 4000–8000 µS/cm, but occasionally rises to over 26,000 µS/cm in some coal mines (DIPNR 2003). This is of particular importance for areas where these groundwater sources interact with surface streams.

NOW alluvial monitoring bores were analysed for changes in EC levels over time. At present only a handful of monitoring bores are operational, providing real-time or instantaneous data. For the current assessment, historic groundwater EC levels recorded across Kingdon Ponds and Dartbrook Creek systems were compared to those of recent bore EC results. Summary statistics for EC levels in various bores are displayed in Table 6 and the locations of bores are illustrated in Figures 9 and 10. Recent EC levels are suggestive of similar patterns and levels compared to those of historic (pre-Scheme) periods for bores near the confluence of Kingdon Ponds and Dartbrook Creek systems (Figure 11). EC levels appear to be correlated with river flow and rainfall, suggesting that alluvial bores in these areas are well-connected to river flows. Some caution needs to be exercised in this interpretation though, since some of these differences are based on single EC readings at different times. In addition, mining activities in and around this part of the catchment are limited and these results may not represent the parts of the catchment more heavily affected by mining.

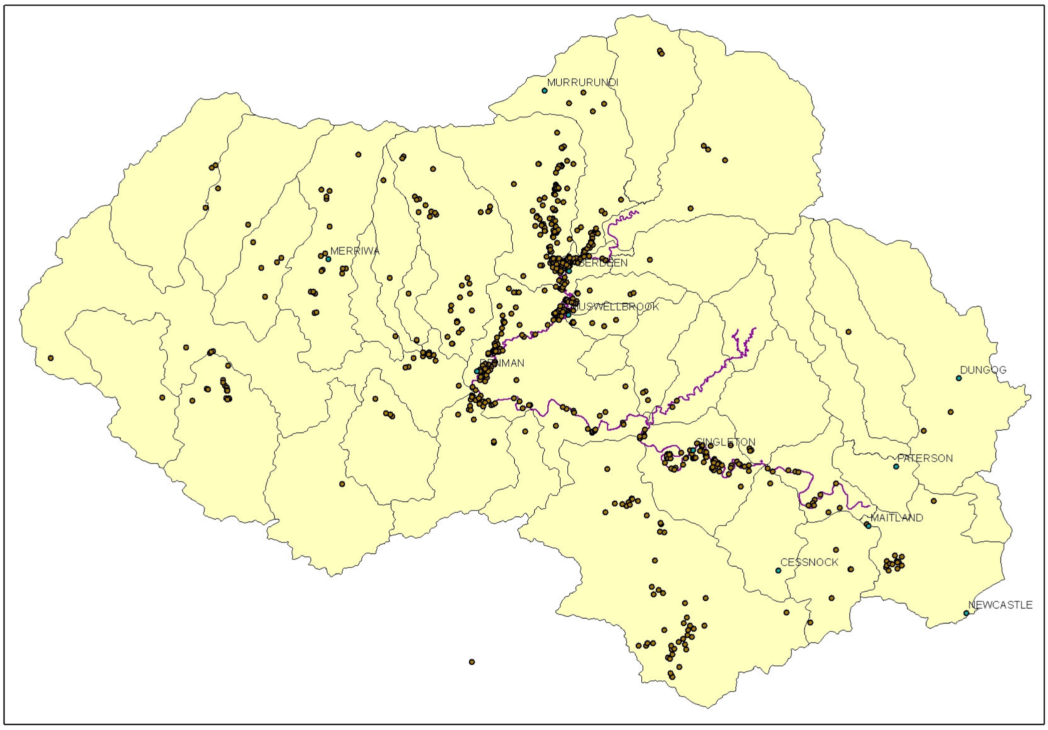


Figure 8: Historic groundwater monitoring network across the Hunter catchment

Source: NSW Office of Water

NOW monitoring bores were also analysed for areas around Singleton (end of Scheme sectors) as shown in Figure 10. Available data between the mid 1970s and 2004 were compared (see Figure 12) and the data again suggested patterns similar to monitoring bores in the upstream catchments, with EC fluctuations potentially exhibiting seasonal and river flow responses. EC levels showed a slight increase during drier conditions; however, again most of these assessments are based on limited temporal sampling. Bore EC levels in the alluvium around Singleton also appear to be affected by their connectivity to the river.

Distance from the river may also play a role in contributing to elevated EC levels in some areas. For example bores GW016053 and GW016054 sampled 6 and 1 times respectively between 1995 and 2005, produced EC readings twice as high as bores closer to the Hunter River. On the other hand, a single reading conducted in early 1995 for bore GW078357 (the furthest site in this reach) had EC similar to that of the alluvial section (see Figure 10). Further detailed assessment is required to fully understand the spatial variation in groundwater conductivity levels in these areas.

Groundwater summary

Since rising groundwater levels and/or EC can affect stream water quality (e.g. stream EC), any increase in groundwater level or EC could potentially have a confounding effect on the interpretation of Scheme effectiveness. Although there is access to a relatively good collection of historic groundwater EC data (e.g. Conaghan 1948, Williamson 1958, AGC 1967, Ainsworth 1994), recent monitoring is much more scattered and limited. There is also a bias towards recent monitoring bores being located in the alluvial areas of the catchment. This produces challenges in identifying trends over time and space, since many bores either have limited temporal replication of samples or are located in less-impacted areas of the catchment. Access to monitoring bores tapping into aquifers in close proximity to major development activities would be very useful, especially around open-cut mine pits with the potential to alter aquifer flow and recharge characteristics. From the groundwater monitoring data available for the current assessment, neither groundwater levels nor EC appeared to be rising in the Hunter River catchment in recent times (except perhaps in some very localised areas). While no obvious trends in groundwater level or EC were identified in the current assessment, if future trends in groundwater level and conductivity are to be undertaken and related back to the impact of the Scheme (or mining and power generation), then a more comprehensive and representative groundwater monitoring program is required for the catchment.

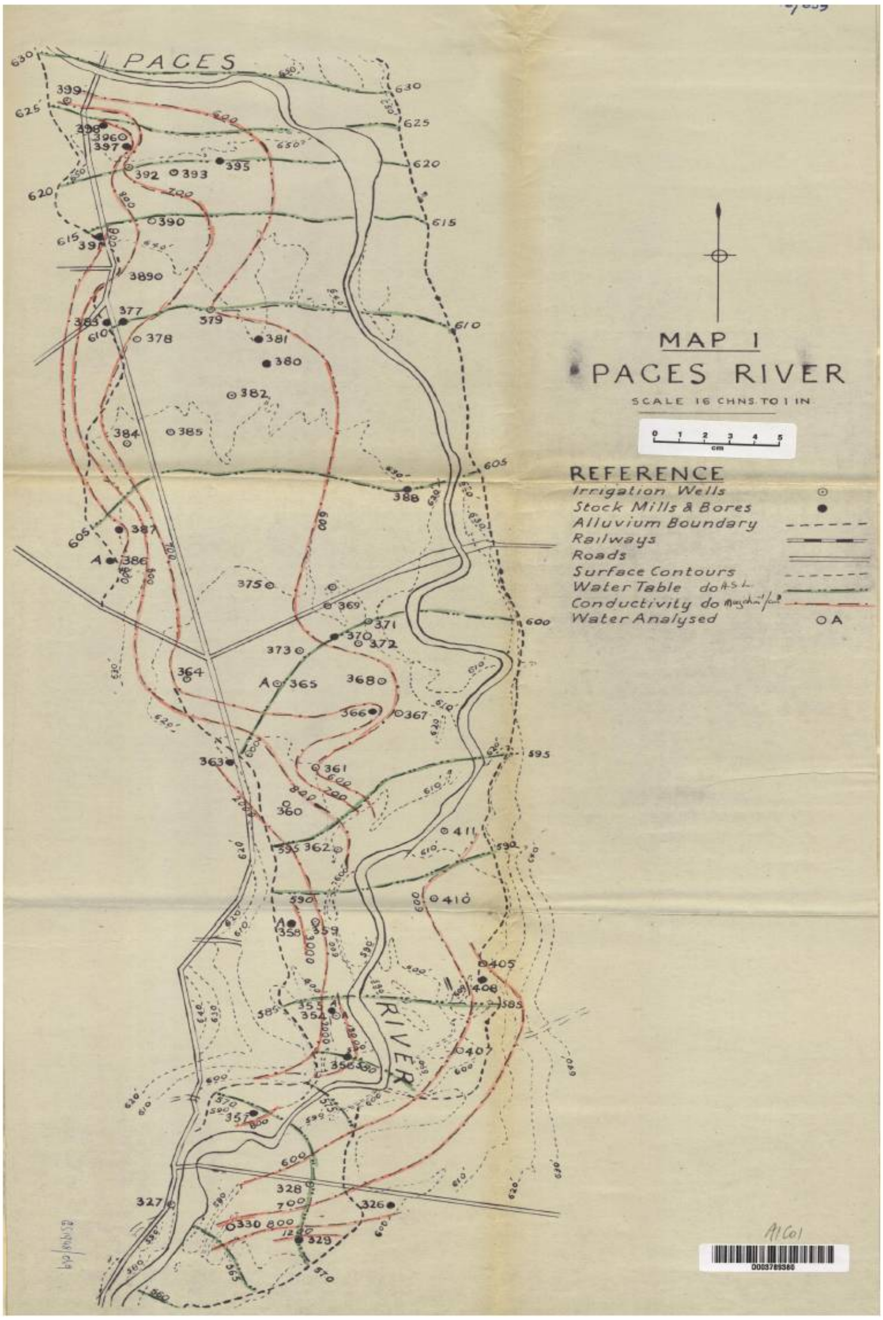


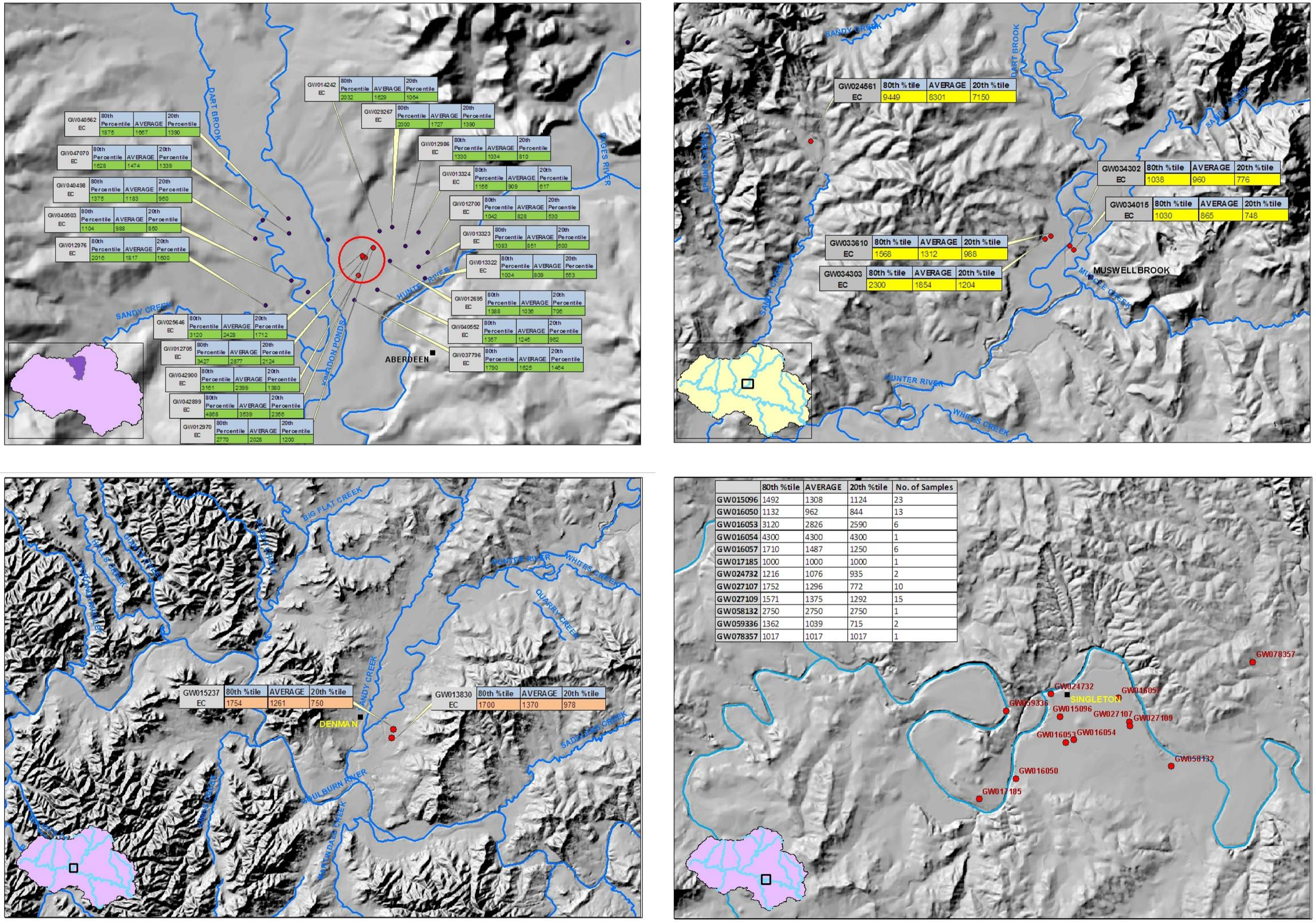
Figure 9: Upper Hunter groundwater conductivity level contours

Source: Conaghan (1948)

Table 6: Groundwater EC data recorded across the upper–middle Hunter catchment

Station No.	GW012695	GW012700	GW012705	GW012970	GW012976	GW012986	GW013322	GW013323	GW013324	GW013830
AVERAGE	1036	828	2877	2028	1817	1034	809	851	909	1370
80th percentile	1388	1042	3427	2770	2016	1330	1004	1083	1156	1700
20th percentile	706	530	2124	1200	1600	810	553	600	617	978
No. of samples	74	70	24	36	69	76	54	67	70	39
Sample year	1953-1988	1953-2004	1953-2008	1961-1976	1953-1986	1903-2004	1961-2004	1961-2004	1953-2000	1961-1975
Station No.	**GW014242**	**GW015096**	**GW015237**	**GW016050**	**GW018523**	**GW022309**	**GW024561**	**GW025646**	**GW025789**	**GW026200**
AVERAGE	1629	1287	1261	962	2165	2874	8301	2428	1133	2484
80th percentile	2032	1478	1754	1132	2588	3601	9449	3120	1272	2847
20th percentile	1064	1108	750	844	2019	2200	7150	1712	1039	2070
No. of samples	50	24	62	13	5	11	12	19	12	11
Sample year	1953-1981	1976-1986	1961-2004	1976-1999	1977-1985	2001-2004	2000-2007	1966-2008	2002-2008	2001-2008
Station No.	**GW026956**	**GW027107**	**GW027109**	**GW029267**	**GW033610**	**GW034015**	**GW034302**	**GW034303**	**GW037733**	**GW037796**
AVERAGE	909	1181	1368	1727	1312	865	960	1854	895	1625
80th percentile	1000	1750	1554	2000	1568	1030	1038	2300	998	1790
20th percentile	785	700	1184	1390	988	748	776	1204	788	1464
No. of samples	76	11	19	16	85	46	62	53	72	102
Sample year	1953-1988	1976-2003	1976-2002	1953-1973	1961-2002	1961-1988	1953-1988	1961-1988	1953-1988	1961-1986
Station No.	**GW038740**	**GW040498**	**GW040503**	**GW040552**	**GW040562**	**GW042899**	**GW042900**	**GW047070**	**GW049660**	**GW078396**
AVERAGE	3270	1183	988	1245	1667	3539	2399	1474	1907	2237
80th percentile	3800	1375	1104	1357	1875	4868	3161	1628	2055	2420
20th percentile	2480	960	850	962	1390	2356	1380	1339	1690	2130
No. of samples	11	46	59	58	61	24	35	23	10	11
Sample year	1993-2004	1964-1981	1953-1988	1961-1988	1961-1988	1977-1988	1977-2008	1978-1988	1979-2008	2001-2007
Station No.	**GW080941**	**GW080944**								
AVERAGE	3724	10630								
80th percentile	4274	11548								
20th percentile	3415	9951								
No. of samples	11	13								
Sample year	2005-2011	2005-2011								

Source: NSW Office of Water



	80th %tile	AVERAGE	20th %tile	No. of Samples
GW015096	1492	1308	1124	23
GW016050	1132	962	844	13
GW016053	3120	2826	2590	6
GW016054	4300	4300	4300	1
GW016057	1710	1487	1250	6
GW017185	1000	1000	1000	1
GW024732	1216	1076	935	2
GW027107	1752	1296	772	10
GW027109	1571	1375	1292	15
GW058132	2750	2750	2750	1
GW059336	1362	1039	715	2
GW078357	1017	1017	1017	1

Figure 10. Location of bores and groundwater EC levels in the Hunter River catchment: near Aberdeen (top left); near Muswellbrook (top right); at the Goulburn River junction (bottom left) and near Singleton (bottom right)

Source: NSW Office of Water

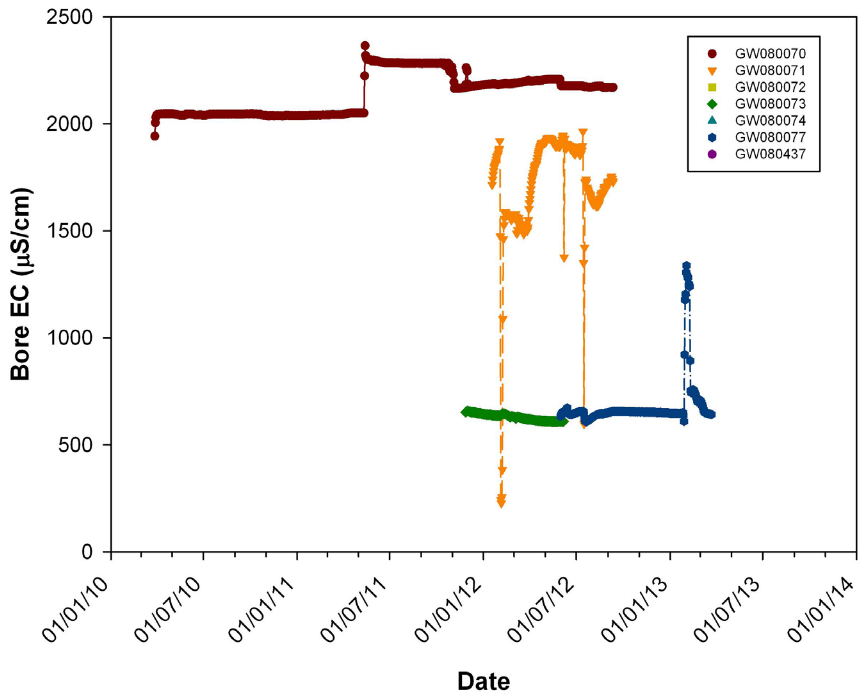


Figure 11: Recent continuous EC levels in bores in the Kingdon Ponds and Dartbrook catchments

Source: NSW Office of Water

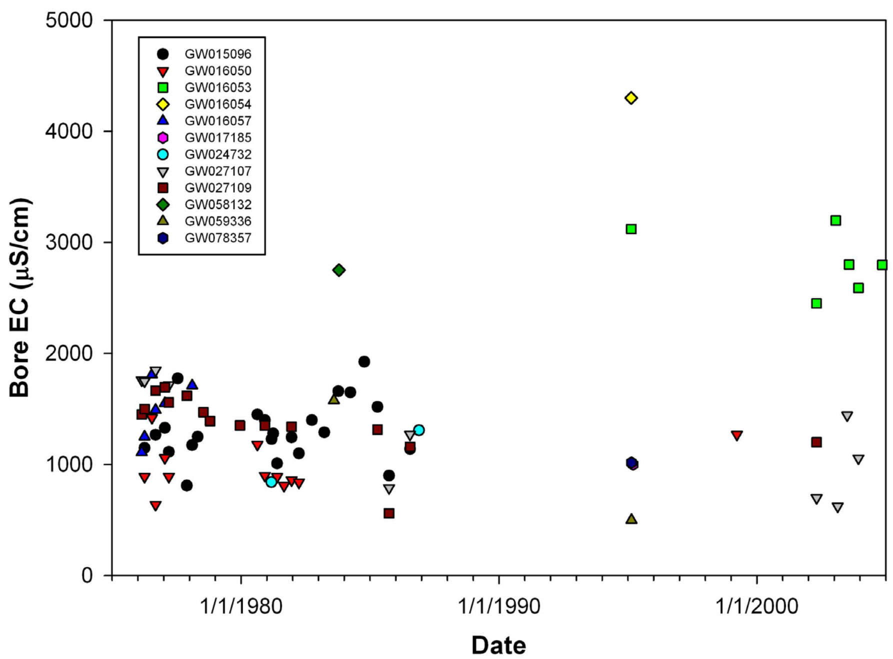


Figure 12: Historic and recent EC levels from groundwater bores near Singleton

Source: NSW Office of Water

Surface water state and trend

Water quality data were obtained from a wide variety of sources, including government agencies (e.g. NOW databases, OEH databases, Hunter Central Rivers Catchment Management Authority databases, local councils, Minerals Department reports, etc.); industry sources[4] (e.g. mining and manufacturers routine monitoring programs and/or environmental assessments) and a variety of research theses (e.g. Mackie 2009, Pritchard 2005, Jasonsmith 2010). These data were collated and used to calculate median conductivity data at monitoring locations throughout the catchment. Median water quality at individual sites is summarised in Table A4, Appendix A and illustrated in Figure 13. Colour coding has been based on the general criteria for the salinity of irrigation water in the Hunter Valley (Creelman 1994, Croft & Associates 1983; see Figure 13 for EC ranges) where: blue represents low salinity, green medium salinity, yellow high salinity, orange very high salinity; and red extreme salinity. Some caution needs to be applied in the interpretation of results where only a single grab sample is available at a site; these are identified in Table A4.

While this analysis provides some general indications of suitability of surface water for irrigation uses throughout the catchment, further site-specific assessment may need to be undertaken since the data come from varying time periods and do not necessarily capture the most recent changes which may have occurred in a local catchment.

Longitudinal variation – Hunter River

The range and variability of conductivity levels in the main stem of the Hunter River can be seen in a longitudinal boxplot of EC levels – from above Glenbawn Dam to downstream of Singleton (Figure 14). Increases in EC levels and variability are particularly noticeable in the Hunter River section between Denman and Glennies Creek. Flows from Glennies Creek appear to lead to both a decrease in the median conductivity levels and a decrease in the variation of EC levels. Higher variability in EC levels is seen again downstream of the Wollombi Brook junction.

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[4] For the current assessment, the EPA wrote to all companies and councils with an environment protection licence to discharge to the Hunter River catchment, requesting any ambient water quality and ecosystem health monitoring data. The contribution of the respondents to the current review is acknowledged and greatly appreciated.

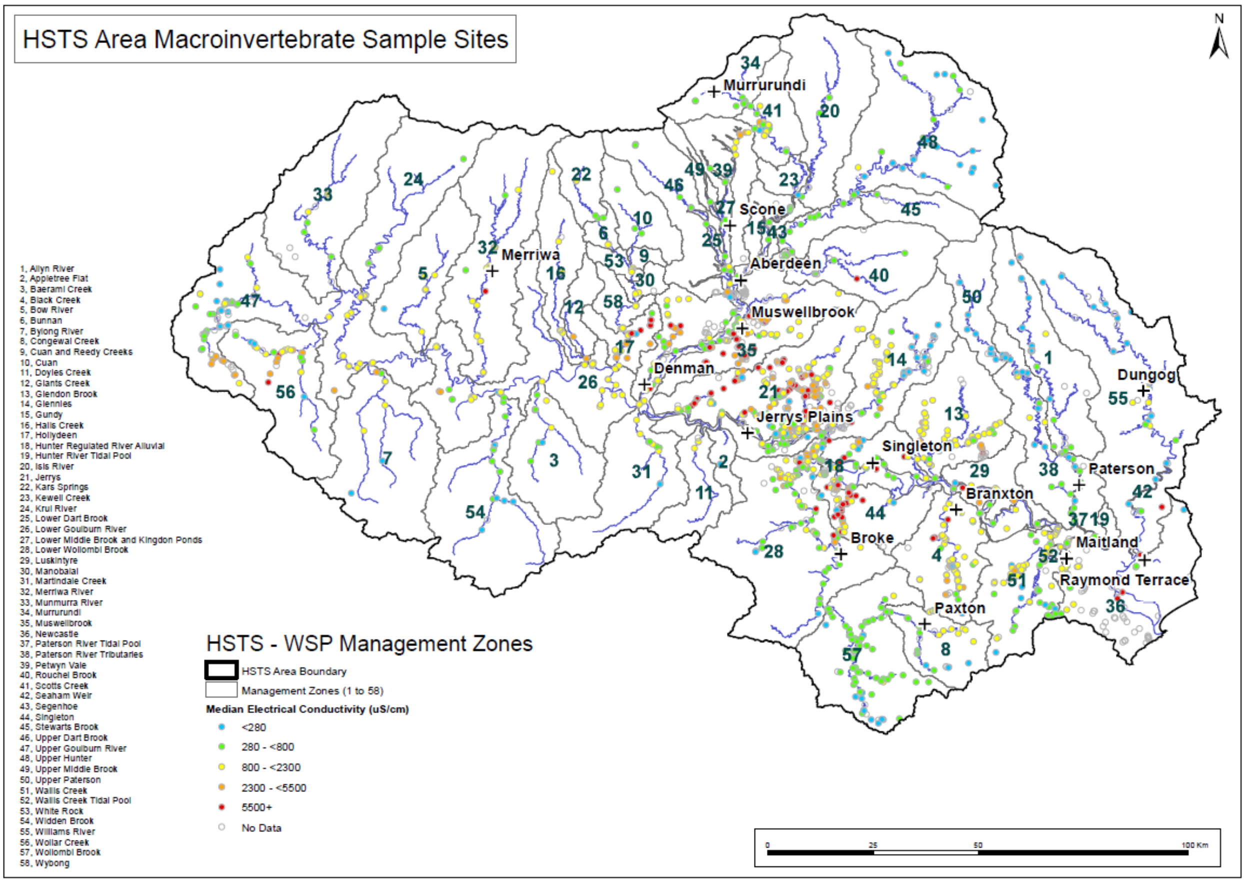

Figure 13. Surface water EC levels in the Hunter River catchment; colour coding based on Creelman (1994) and Croft & Associates (1983)

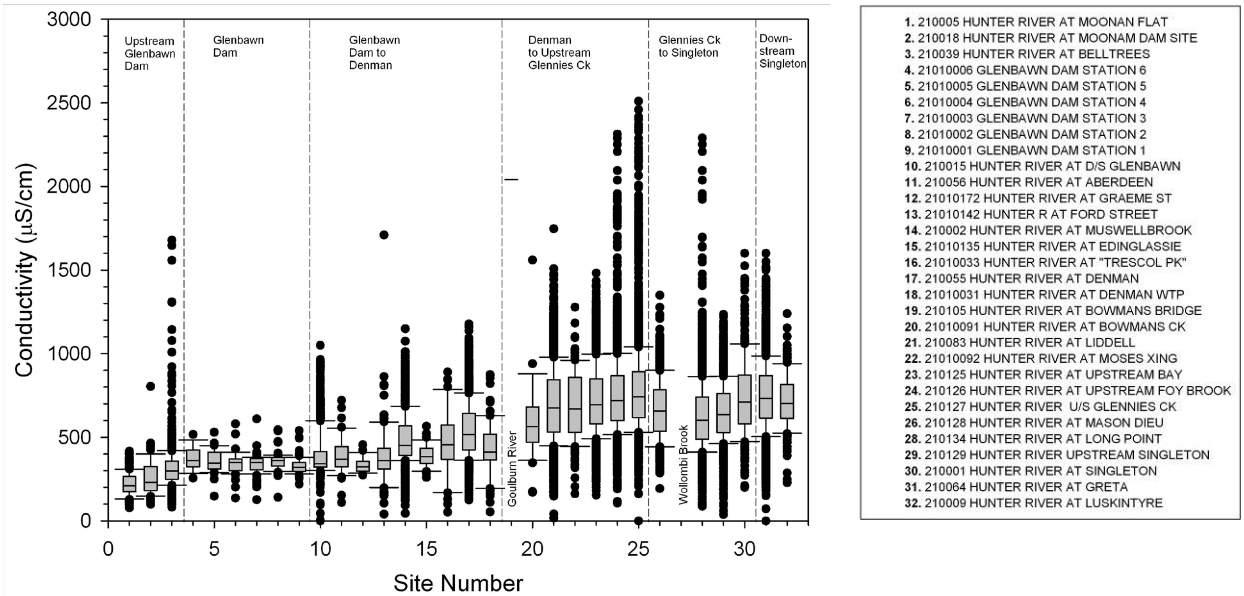

Figure 14. Boxplots of conductivity levels in the main stem of the Hunter River from above Glenbawn Dam to downstream of Singleton.

Single outliers at 210001, 210002, 210083 and 21010092 not illustrated.
Distance between sites not to scale.

Long-term trends in flow and electrical conductivity

Trends in hydrology and water quality in the Hunter River need to be interpreted in terms of both short-term climatic variations (e.g. significant rainfall events), longer term cycles [e.g. the El Niño–Southern Oscillation (ENSO) and Interdecadal Pacific Oscillation (IPO)] and human-induced changes (including the potential for climate change to impact in the future). Although the international scientific community has reached a consensus that global warming is unequivocal (IPCC 2007), the exact implications this has for rainfall and hydrology are far more uncertain, particularly at a regional scale in NSW. There have clearly been cyclic periods of higher and lower rainfall and flow within the Hunter River catchment and such trends are likely to continue even under a global warming scenario. In other catchments (e.g. Hawkesbury–Nepean River), some of these cyclical trends have been related directly to large-scale climatic patterns such as ENSO and IPO (for example, see DECC 2009). Since EC levels can be significantly affected by flow, assessments of changes and/or trends in EC usually also need to consider variation in flow.

Long-term data for the analysis of flow and EC were primarily taken from the NOW gauging station database (Hydstra) and the NOW water quality database (KWiQM). Few other sites have the same length time series of continuous (or near continuous) flow and EC readings at an individual site. However, most continuous (or near-continuous) EC records exist only since about 1993 when the EC meters were progressively installed at the gauging stations. Where appropriate surface water quality data from grab samples existed at the gauging station site, pre-dating the installation of continuous EC meters, these were added to the continuous (or near-continuous) EC records. In doing so it is assumed that there are no systematic differences in EC measurements based on grab samples or EC meters[5]. Where there were multiple EC readings on the same day, the median for these records was calculated and used in subsequent analyses.

Since the establishment of the Scheme extended over approximately a decade before the formal gazettal of the Scheme, the data were split into three time periods as surrogates for *before* Scheme operation[6] (1970s and 1980s), *during the initial stages* of the Scheme (1990s) and *after commencement* of the Scheme (2000s to present). These periods were then compared using empirical distribution functions[7] (analogous to Flow and EC exceedance curves) to see if there were any clear differences between periods in terms of the distribution of flows or EC levels. Most sites show a clear relationship between flow and EC, with EC generally decreasing as flows increased. This is presented graphically for important monitoring sites in Appendix B. These EC–flow relationships were used to underpin the management of Scheme discharges so they occurred during periods of high river flow when the opportunity for dilution was at its greatest. Summary statistics for long-term sites in the Hunter River main stem and other important stream/river sites are also included in Appendix B. In addition, empirical distribution functions for EC and flow at these sites are presented graphically in Appendix B. To some extent, the conclusions from the analysis of flow and EC may be affected by the varying length and consistency of data records at individual sites in the different periods, and where limited data occurs in any period this is identified below.

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[5] A number of 0 or negative values for conductivity were identified in the continuous EC data which may indicate a problem with the meter at the time of record.

[6] Increased water quality monitoring appears to coincide with the establishment of the *Clean Waters Act 1970*, and few water quality records for the Hunter River catchment extend further back than the early 1970s.

[7] The ecdf function in R Version 2.8.0 (see The R Foundation for Statistical Computing 2013) was used to compare empirical distribution functions over time.

Hunter River monitoring stations

Results of comparisons for the Hunter River monitoring stations suggest the following:

- Limited sampling at Belltrees (210039), upstream of Glenbawn Dam, during the 1970s to 1990s makes comparisons between periods difficult, but at this point there appears to be little difference in the distribution of conductivity levels over time at this site.
- Temporal variability in flow and EC levels downstream of Glenbawn Dam (210015) are noticeable, with flows in the 2000s generally being higher than in the 1970s & 1980s or 1990s. However, the distribution of EC levels does not appear to have changed markedly between the 1990s and 2000s.
- Limited EC data is available for the Hunter River at Aberdeen (210056). Flows appear to be relatively similar between the 1990s and 2000s, although some higher flows were recorded during the 1990s and increased medium flows were recorded in the 2000s, potentially as a result of river regulation. Continuous EC records were only available from March 1998, but the distribution of EC records suggests higher EC for the period monitored in the 1990s compared to those recorded in the 2000s.
- The distribution of flow and EC records for the Hunter River at Muswellbrook Bridge (210002) over the various time periods show relatively little change in either flow or EC. Median EC over the period 1970 to 2013 was 451.1 µS/cm.
- The distribution of flow and EC records for the Hunter River at Denman (210055) over the various time periods show relatively little change. Slightly higher flows were recorded in the 1970s, however the distribution of EC levels was similar for all periods. Median EC over the period 1970 to 2013 was 515.5 µS/cm.
- The distribution of flow and EC records for the Hunter River at Liddell (210083) suggests higher flows in the 1970s & 1980s compared to the 1990s and 2000s. The distribution of EC levels was similar for most periods; however, there appeared to be some higher EC levels in the 1970s & 1980s and EC levels in the 2000s were generally lower than in the 1990s. At times EC levels exceeded the 900 µS/cm level for longer periods than at Denman. These higher EC levels were usually associated with lower flow in the river. Median EC over the period 1970 to 2013 was 675.7 µS/cm.
- The distribution of flow and EC records for the Hunter River upstream of Bayswater Creek (210125) suggests similar flows in the 1990s and 2000s. No flow or EC data were available at this site for the 1970s & 1980s. The distribution of EC levels was similar, but EC levels in the 2000s were generally lower than in the 1990s. At times EC levels exceeded the 900 µS/cm level. Median EC over the period 1990 to 2013 was 698 µS/cm.
- The distribution of flow and EC records for the Hunter River upstream of Foy Brook (210126) suggests similar flows in the 1990s and 2000s. No flow or EC data were available at this site for the 1970s & 1980s. The distribution of EC levels was also similar but EC levels in the 2000s were generally lower than in the 1990s. At times EC levels exceeded the 900 µS/cm level. Median EC over the period 1990 to 2013 was 719 µS/cm.
- The distribution of flow and EC records for the Hunter River upstream of Glennies Creek (210127) suggests similar flows in the 1990s and 2000s, but comparatively more high flows in the 2000s. No flow or EC data were available at this site for the 1970s & 1980s. The distribution of EC levels was also similar but EC levels in the 2000s were generally lower than in the 1990s. At times EC levels exceeded the 900 µS/cm level. Median EC over the period 1990 to 2013 was 741.4 µS/cm.

- The distribution of flow and EC records for the Hunter River at Maison Dieu (210128) suggests similar flows in the 1990s and 2000s. No flow or EC data were available at this site for the 1970s & 1980s. The distribution of EC levels was also similar, but EC levels were only recorded between July 1993 and November 2000, making inter-decadal comparisons less meaningful. At times EC levels exceeded the 900 µS/cm level.
- The distribution of flow and EC records for the Hunter River at Long Point (210134) suggests higher flows in the 2000s compared to the 1990s. No flow or EC data were available at this site for the 1970s & 1980s. In contrast, EC levels in the 2000s were generally lower than in the 1990s. Fewer EC levels exceeded the 900 µS/cm level than at sites further upstream, potentially as a result of diluting flows from Glennies Creek. Median EC over the period 1990 to 2013 was 719 µS/cm.
- The distribution of flow and EC records for the Hunter River upstream of Singleton (210129) suggests higher small to medium flows in the 1990s compared to the 2000s, but more high flows in the 2000s period. If data from the Singleton gauge (210001) are included, then flows were even higher (median = 371 megalitres a day) in the 1970s & 1980s. EC levels in the 2000s were generally lower than in the 1990s and much lower than EC levels measured at Singleton (210001) in the 1970s & 1980s. Fewer EC levels exceeded 900 µS/cm over the period 1990 to 2013. Median EC over the period 1990 to 2013 was 639.9 µS/cm, much lower than the median EC level of 831 µS/cm recorded at Singleton (210001) in the 1970s & 1980s.
- The distribution of flow and EC records for the Hunter River at Greta (210064) suggests higher flows in the 1970s & 1980s but similar flows in the 1990s and 2000s. EC levels in the 2000s were generally lower than in the 1990s and much lower than EC levels in the 1970s & 1980s. EC levels exceeded the 900 µS/cm more frequently over the period 1990 to 2013 at Greta than at Singleton. Median EC over the period 1990 to 2013 was 731.9 µS/cm, much lower than the median EC level of 979 µS/cm recorded during the 1970s & 1980s.

Goulburn River monitoring stations

Results of comparisons for the Goulburn River monitoring stations suggest the following:

- Limited sampling of the Goulburn River at Coggan (210006) during the 1970s to 2000s makes comparisons between periods difficult. At this point there appears to be no trend in conductivity levels at this site but further analysis is warranted as more EC data are collected over time. Median EC level over the period of record was 1007 µS/cm.
- The distribution of flow and EC records for the Goulburn River at Kerrabee (210016) suggests higher flows in the 1990s compared to other periods. Very few EC records were available for the 1990s, but the EC levels in the 2000s were similar to EC levels in the 1970s & 1980s, but with some higher EC records overall in the 2000s. EC levels frequently exceeded 1000 µS/cm but there appears to be a declining trend since the mid-2000s. Cyclical patterns were also evident in the data and these require further assessment. Median EC over the period 1970 to 2013 was 1070.4 µS/cm.
- The distribution of flow and EC records for the Goulburn River at Sandy Hollow (210031) suggests higher flows in the 1970s & 1980s compared to more recent periods. EC records for the 1970s & 1980s and 1990s were higher than EC records in the 2000s. EC levels frequently exceeded 1000 µS/cm and again there appears to be a declining trend since the 1990s. Some relatively high EC levels (2500 to 3000 µS/cm) have been recorded in recent times. Cyclical patterns were also

evident in the data for the Goulburn River at Sandy Hollow, but not as pronounced as at Kerrabee. These patterns require further assessment. Median EC over the period 1970 to 2013 was 837.5 μS/cm.

Wollombi Brook monitoring stations

Results of comparisons for the Wollombi Brook monitoring stations suggest the following:

- Flow and EC records for Wollombi Brook at Bulga (210028) suggest higher flows in the 1970s & 1980s compared to the 2000s. Limited flow data were available for the 1990s. EC records for the 1970s & 1980s and 1990s were slightly lower than EC levels in the 2000s but this may be affected to some degree by sample size differences. EC levels exceeded 1000 μS/cm on some occasions and there appears to be a declining trend since the early 2000s. The median EC level over the period 1970 to 2013 was 674 μS/cm.
- Flow and EC records for Wollombi Brook at Warkworth (210004) suggest higher flows in the 1970s & 1980s compared to the 1990s and 2000s. EC records for the 1970s & 1980s and 1990s were obviously lower than EC levels in the 2000s. EC levels exceeded 1000 μS/cm for most of the 2000s with some very high EC levels (approaching 10,000 μS/cm) recorded. The EC–flow relationship demonstrates that EC concentrations were often not well-correlated with flow. This is clearly different to the patterns of EC and flow upstream at Bulga. Overall, the EC data implies impacts either from saline groundwater moving into Wollombi Brook or from mining. Further assessment is necessary to fully understand the underlying mechanisms yielding high EC levels in Wollombi Brook at Warkworth. Median EC over the period 1970 to 2013 was 740.5 μS/cm, however, the median EC level during the 2000s was 891.1 μS/cm.

Glennies Creek monitoring stations

Results of comparisons for the Glennies Creek monitoring stations suggest the following:

- Flow and EC records for Carrow Brook at Carrowbrook (210114), upstream of Glennies Creek Dam (constructed in 1983), suggest flows were similar in all periods. Limited EC data were available for the 1970s & 1980s. EC records for the 1990s and 2000s were similar. EC levels are low and have not exceeded 600 μS/cm. Median EC over the period 1970 to 2013 was 175.8 μS/cm.
- Glennies Creek at the Rocks No. 2 (210044) is downstream of Glennies Creek Dam. Flow and EC records for 210044 suggest higher flows in the 1990s and 2000s compared to the 1970s & 1980s. EC records for the 1970s & 1980s were limited but indicate higher EC levels (median = 427.5 μS/cm) than EC levels in the 1990s and 2000s (median = 263–265 μS/cm). EC levels now rarely exceed 600 μS/cm.
- Flow and EC records for Glennies Creek at Middle Falbrook (210044) suggest higher flows in the 1990s and 2000s compared to the 1970s & 1980s. EC records for the 1970s & 1980s were limited but appear to have been much higher than EC levels in either the 1990s or 2000s. Higher EC levels occurred in the 2000s compared to the 1990s, but EC levels rarely exceed 900 μS/cm. Median EC over the period 1970 to 2013 was 361.4 μS/cm.

Other monitoring stations

Results of comparisons for other monitoring stations suggest the following:

- Flow and EC records for Wybong Creek at Wybong (210040) suggest flows were similar in all periods. Limited EC data were available for the 1970s & 1980s but it appears that EC levels in the 2000s have been significantly higher (median = 1728.1 µS/cm) than in either the 1990s or 1970s & 1980s. There was a clear increasing trend in EC levels from the early 2000s to about 2007 which coincides with drought conditions. Further assessment of EC levels are required for Wybong Creek. Median EC over the period 1970 to 2013 was 1578.8 µS/cm.
- Flow and EC records for Merriwa River upstream of Vallances Creek (210066) suggest flows were much lower in the 2000s compared to either the 1970s & 1980s or the 1990s. Limited EC data were available for the 1970s & 1980s or 1990s, but it appears that EC levels in the 2000s have been significantly higher (median = 1598.2 µS/cm) than in earlier periods. There was a clear increasing trend in EC levels from the early 2000s to about 2007 which coincides with drought conditions. However, since that time EC levels have declined significantly. Further assessment of EC levels is required for the Merriwa River. Median EC over the period 1970 to 2013 was 1590 µS/cm.
- Flow and EC records for Foy Brook downstream of Bowmans Creek Bridge (210130) suggest flows were similar in the 1990s and 2000s. No flow data were available for the 1970s & 1980s. Limited EC data were available for the 1970s & 1980s, but EC levels were higher in the 1990s compared to the 2000s. There was a clear increasing trend in EC levels from the early 2000s to about 2007 which coincides with drought conditions. However, since that time EC levels have declined significantly, although there is a clear outlier (~ 6000 µS/cm) and a gap in the EC record. Further assessment of EC levels is required for Foy Brook. Median EC over the period 1970 to 2013 was 1297.3 µS/cm.
- Flow and EC records for Bayswater Creek (210110) suggest flows were similar in the 1990s and 2000s. No flow data were available for the 1970s & 1980s. Overall flows are low (median = 0.24 megalitres a day). No EC data were available for the 1970s & 1980s, but EC levels were higher in the 2000s compared to the 1990s. EC levels have remained relatively consistent over the past two decades (median = 3118.9 µS/cm), however maximum levels can at times be high (approaching 5000 µS/cm). While a flow concentration relationship exists for Bayswater Creek it appears also to be influenced by discharges at relatively higher flow rates. Further more detailed assessment of EC levels is required for Bayswater Creek.
- The distribution of flow and EC records for Black Creek at Rothbury (210089) suggests higher flows in the 2000s compared to the 1970s & 1980s and 1990s. EC records for the 1970s & 1980s were limited but appear to have been much higher than EC levels in either the 1990s or 2000s. Higher EC levels occurred in the 2000s compared to the 1990s and EC levels often exceed 900 µS/cm. Median EC over the period 1970 to 2013 was 1360.5 µS/cm.

Generalised additive modelling of electrical conductivity

As stated earlier, assessment of trends in water quality need to take into account changes in rainfall, flow and other important environmental variables. A generalised additive model (GAM) was developed to model water quality in the Hunter River using the water quantity and quality data taken from the NOW databases. The modelling approach taken is similar to that used by DECC for the Hawkesbury–Nepean River (DECC 2009). The predictor variables used in these models were flow at the gauging station/water quality sampling site, flow on the day before sampling, seasonal terms and time[8] (consecutive number of days since 1/1/1970). In these analyses, the stochastic effects of rainfall were assumed to be captured through their effects on flow and were not modelled directly.

GAMs were fitted to the data using the *mgcv* package (Wood 2006) in R Version-2.8.0 (see The R Foundation for Statistical Computing 2013). GAMs:

- can provide flexibility in statistical modelling
- do not assume linearity of dependent variables (unless you define them to be linear)
- provide a less subjective choice of appropriate form of relationship between predictor and independent variables, and
- can be implemented in several ways in the R statistical package (DECC 2009).

Due to time constraints, GAM modelling of EC levels was only undertaken for the Hunter River at Muswellbrook (Station 210002) and Hunter River upstream of Singleton (Station 2100129; with the addition of EC levels from Station 210001 prior to 1993 when the continuous EC meter was installed at 210129). Flows, lagged flows and seasonal components were all found to be significant (see Appendix C). The resulting non-linear time trend in EC levels for the Hunter River at Muswellbrook and Singleton is illustrated in Figure 15.

The GAM non-linear time trend analysis supports the conclusions from the assessment of the distribution of flow and EC levels above:

- There is evidence of cyclical temporal trends in EC levels in the Hunter River at Muswellbrook, but EC levels do not appear to be either increasing or decreasing over time.
- There is evidence of cyclical temporal trends in EC levels in the Hunter River at Singleton, and EC levels appear to have declined over the more recent time periods.

To some extent these trend conclusions have been affected in the past by periods when EC levels were not recorded (e.g. the mid to late 1980s) or where sampling was inconsistent, yielding higher levels of uncertainty at these times. It is expected that as further continuous (or near-continuous) flow and EC levels are measured, these models can be further refined to confirm if future trends or changes in EC levels occur.

Surface water summary

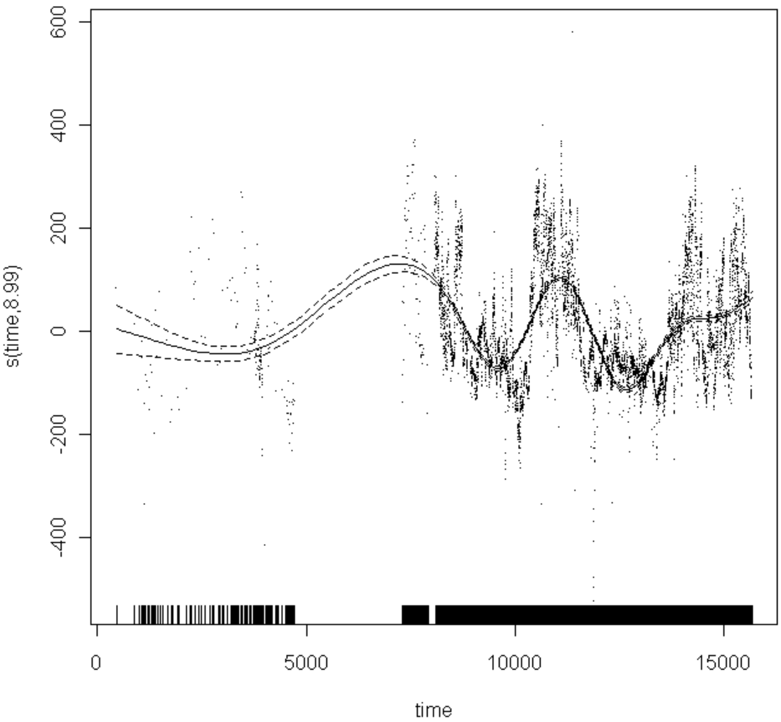
Given that a specific experimental design for testing the impact of the Scheme was not developed at the time, assessment of the effectiveness of the Scheme has relied upon the partitioning of available data into three major time periods: *before* Scheme operation (1970s and 1980s), *during the initial stages* of the Scheme (1990s) and *after formal commencement* of the Scheme (2000s to present). Provided these periods are adequate surrogates for the various stages of the Scheme operation and that natural EC sources have remained relatively constant between periods, then the data suggests that the Scheme has:

- had little effect on flows and EC levels in the Hunter River upstream of Denman over the three time periods
- improved EC levels at (and immediately upstream of) Singleton and Greta, particularly when comparing EC levels in the 2000s with EC levels in the 1970s and 1980s. This is despite lower flows occurring in the 1990s and 2000s, and

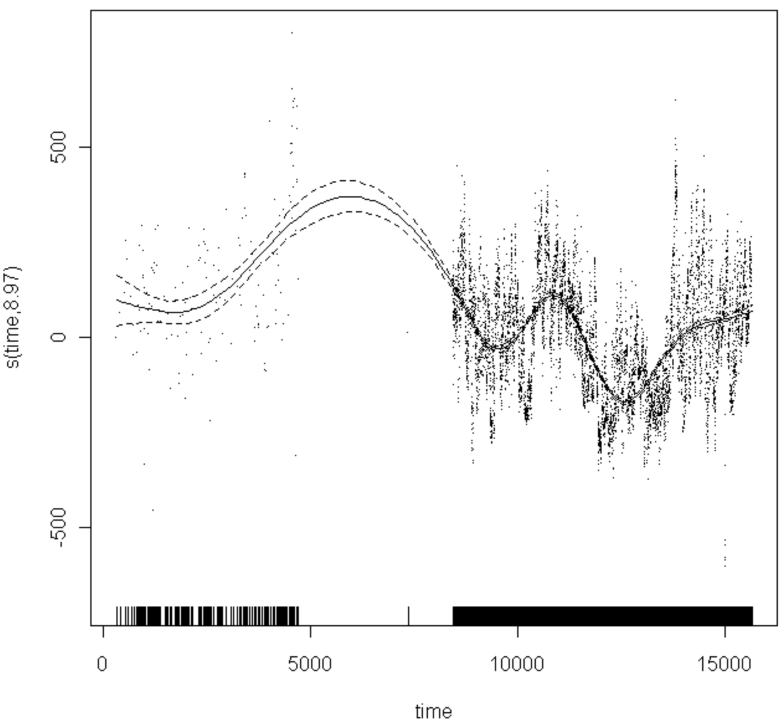
[8] Time was taken to be an increasing series from 1 on a start date of 1/1/1970 up to a maximum on the latest record for that site (e.g. 15,758 for 21/2/2013).

- potentially improved EC levels at monitoring stations between Denman and Singleton, although limited if any EC data were available at these latter stations during the 1970s & 1980s to clarify pre-existing EC levels prior to the Scheme commencing (in ‘pilot’ or ‘full’ implementation). The 1990s appear to have similar flows to the 2000s at these monitoring stations, but the EC levels are usually lower in the 2000s (see Appendix B).

These putative trends are supported by the GAM assessment which found evidence of cyclical temporal trends in EC levels in the Hunter River at both Muswellbrook and Singleton but little evidence that EC levels had either increased or decreased at Muswellbrook, and that EC levels had declined at Singleton over the most recent time period.



210002 Hunter River at Muswellbrook



210001 Hunter River at Singleton

Figure 15: Non-linear generalised additive model (GAM) time trend in EC levels in the Hunter River at Muswellbrook (above) and Singleton (below)

Values on the y-axis represent partial residuals (see Wood 2006);
Time = 1 on the x-axis corresponds to 1/1/1970; Time = 15,000 is 25/1/2011.
(EC data from Stations 210001 and 2100129 were combined and medians calculated where more than one record occurred on the same day.)

Since the Scheme only applies to discharges at times of high or flood flow, if discharges under the Scheme were increasing conductivity levels in the Hunter River at these times, then such an increase would be reflected in the EC-flow relationships, with higher flows leading to higher EC levels. Such a pattern was not evident in the Hunter River monitoring sites (see Appendix B).

While salinity targets do not apply to the Hunter River tributaries, high salinity waters in the tributaries can still affect Scheme discharge opportunites within the Hunter River. Most tributary monitoring sites also showed a decreasing relationship between EC and flow (see Appendix B). An exception to this last generalisation was Wollombi Brook at Warkworth, where the EC–flow relationship demonstrates that EC concentrations were often not well-correlated with flow (as would normally be expected and appeared to be the case for most other monitoring stations, including the upstream Wollombi Brook site at Bulga). Overall, the flow and EC data at Wollombi Brook at Warkworth implies impacts either from saline groundwater and/or mining. Further assessment is necessary to fully understand the underlying mechanisms which yielded the high EC levels at Warkworth, but these relatively high levels have the potential to reduce the opportunities of the Scheme by increasing the EC contributed by Wollombi Brook waters where they join the Hunter River. Fortunately the very high EC levels of the mid to late 2000s have now declined, but still need ongoing monitoring.

Most other monitoring stations throughout the catchment also showed little evidence of increasing EC levels over time, except potentially during the 2000 to 2007 drought. The interaction of rainfall, flow and groundwater contribution can often be complex and requires further assessment in these areas to fully understand the effects of drought on surface water EC levels in these areas of the Hunter River catchment.

While not exhibiting major trends in EC at the stations investigated (apart from during drought conditions), the Goulburn River can also contribute relatively high salinity water (median EC levels often greater than 800–1000 µS/cm) to the Hunter River. Goulburn River salt loads are highly variable and dependant on subcatchment source. Natural salt inputs from the Wollar, Wybong and Merriwa subcatchments are significant and the total salt load from the Goulburn River can at times be greater than the salt load measured in the Hunter River at Denman (Table 7). While the Goulburn River upstream of Kerrabee is not captured by the Scheme, it can exert an influence on Scheme discharge capacity and opportunity downstream of its confluence with the Hunter River. At present, salinity levels in the Goulburn River downstream of Sandy Hollow cannot be regularly assessed until Jerrys Plain, which is approximately 60 river kilometers (half-day travel time) from Sandy Hollow. In addition, three mines (Ulan, Wilpinjong and Moolarben) currently have discharge licences in the Upper Goulburn River catchment and further mining is proposed for the Bylong Valley. There is currently limited monitoring in the upper Goulburn River catchment[9]. With the likely expansion of mining and coal seam gas extraction in the Upper Goulburn River catchment and the lack of real-time monitoring in the both the upper and lower sections of the Goulburn River catchment, strategic real–time monitoring of flow and salinity in other areas of the Goulburn River catchment is recommended.

[9] The NOW Ulan gauge was discontinued some time ago; however monitoring is a requirement for the Ulan, Wilpinjong and Moolarben discharges.

Table 7. Total salt load from the Goulburn River at Sandy Hollow (210031) and the Hunter River at Denman (210055)

Year	Salt load – Goulburn River at Sandy Hollow (tonnes)	Salt load – Hunter River at Denman (tonnes)	Ratio of salt load Goulburn/Hunter
2007–08	34,100	65,900	52%
2008–09	24,200	72,400	34%
2009–10	11,800	24,100	49%
2010–11	93,200	71, 000	130%

Source: NSW Office of Water

Lastly, the above assessment has focused primarily on EC levels but it is known that the ionic composition of saline groundwater and mine/power generation water can often be very different to what naturally occurs in the surface waters of the catchment. Insufficient time was available to fully analyse the differences in ionic composition of surface water, groundwater and mine water throughout the Hunter River catchment. However, such analysis is warranted given the recent literature on ecotoxic effects of some mine waters and the implication of the potential role of differing ionic and metal/metalloid constituents (e.g. Farag and Harper 2012, OEH 2012, Cardno Ecology Lab Pty Ltd 2010). This aspect is discussed further in the Ecosystem Health Section below.

6. Ecosystem health in the Hunter catchment

Aquatic ecology in the Hunter catchment is affected by natural flows, flow regulation and modification, water quality, changes due to catchment disturbance and run-off, the discharge of treated (or untreated) effluent and land use. The most well-developed and widespread of the available biological indicators of stream 'health' in NSW are macroinvertebrates collected by the methods of either Chessman (1995) or Turak *et al.* (2000, 2004). Macroinvertebrates are commonly used throughout the world to assess the environmental health of a river, stream, creek or wetland because they are sensitive to changes in water quality and flow regimes and allow detection of environmental impacts for some time after the event has occurred. They are easily collected, abundant, diverse, readily seen with the naked eye and the knowledge of taxonomy is advanced and well-documented.

The widely accepted and supported AUSRIVAS (Australian River Assessment System) methodology utilises site-specific predictions of the macroinvertebrate assemblage expected to be present at a site in the absence of environmental stressors. The expected assemblages of macroinvertebrates from sites with similar physical and chemical characteristics (characteristics that are not influenced by human activities, e.g. altitude) are compared to the macroinvertebrate assemblage observed during sampling. The ratio of observed to expected macroinvertebrates can vary from zero, when none of the expected macroinvertebrates are collected at a site, to one or greater, when all or more of the expected macroinvertebrates are collected. The observed over expected ratios (scores) are placed in bands thus permitting an assessment of the environmental health of the river for that site. Computer models calculate a band for each site based on the physical and chemical properties of the site, the time of collection (spring or autumn), the habitat (edge or riffle) and the macroinvertebrate families collected (Table 8 and Table 9).

Since the purpose of the Scheme is to control industrial discharges of saline water and ensure they are only released at times of high or flood flows when there is adequate dilution, the Scheme itself provides limited ability to control the more general impacts of high salinity waters on aquatic health. Nevertheless, an assessment of the impacts of saline waters on macroinvertebrates is necessary to provide the context for addressing questions of whether the Regulation has impacted aquatic ecosystems and associated environmental values since it commenced, or its potential to impact on aquatic ecosystems and associated environmental values in the future.

Macroinvertebrate assessment

Chessman (1997a,b) conducted the first extensive survey of macroinvertebrates in the Hunter River catchment. He found most of the Hunter River valley sites were rated as poor or very poor. Several sites were in a fair category and only a few sites were rated as good or excellent, and these were mostly upstream sites around the edges of the catchment (Chessman 1997a).

Macroinvertebrate monitoring is also a major component of the NSW Monitoring, Evaluation and Reporting (MER) Strategy and OEH has collected data on macroinvertebrates in the Hunter River catchment as part of MER (or earlier programs) for the past two decades. Data from Chessman (1997a) and OEH were combined with macroinvertebrate data from the Hunter–Central Rivers CMA (HCR CMA) to assess the current state of macroinvertebrates in the Hunter River catchment. Chessman calculated Stream Invertebrate Grade Number Average Level (SIGNAL) scores for 48 sites and OEH has calculated AUSRIVAS scores for 316 samples in the Hunter River catchment. The majority of HCR CMA samples (9 sites) come from the Goorangoola Creek catchment and AUSRIVAS scores were calculated for these sites as well for the current assessment. It is highly likely that there are other macroinvertebrate monitoring sites which have not been captured in this summary and further work is required to provide a comprehensive summary of all macroinvertebrate sampling that has taken place in the Hunter River catchment.

Of the 173 unique sites identified as having been sampled for macroinvertebrates over the past decade, over half (56.6 per cent) were found to be in *similar to reference* (band A) or *richer than reference* (band X) condition (see Table 10). Forty seven sites (27.2 per cent) were found to be in a significantly impaired (band B) condition. Eight sites (4.6 per cent) were found to be in a *severely impaired* (band C) condition and one site (0.5 per cent) in an *extremely impaired* (band D) condition. Nineteen sites were outside the experience of the model (OEM) or had insufficient data to calculate an AUSRIVAS score. A further 42 sites only had a SIGNAL score calculated (see Chessman 1997a, b). This indicates that the macroinvertebrate 'health' throughout the catchment is on average good, but there are some areas that are poor in terms of macroinvertebrate health. A relatively high number of samples (n = 9) in the Hunter Regulated River Alluvial Zone were found to be in a significantly impaired (band B) condition.

Table 8: AUSRIVAS bands for spring – edge habitat

Band Label	SIGNAL O/E50 (upper limit)	AUSRIVAS O/E50 (upper limit)	Band Name	Band Description
Band X	Infinity	Infinity	More biologically diverse than reference sites.	More taxa found than expected. Potential biodiversity hot-spot. Possible mild organic enrichment.
Band A	1.09	1.16	Reference condition.	Most/all of the expected families found. Water quality and/or habitat condition roughly equivalent to reference sites. Impact on water quality and habitat condition does not result in a loss of macroinvertebrate diversity.
Band B	0.87	0.83	Significantly impaired.	Fewer families than expected. Potential impact either on water quality or habitat quality or both resulting in loss of taxa.
Band C	0.65	0.51	Severely impaired.	Many fewer families than expected. Loss of macroinvertebrate biodiversity due to substantial impacts on water and/or habitat quality.
Band D	0.43	0.19	Extremely impaired.	Few of the expected families remain. Extremely poor water and/or habitat quality. Highly degraded.

Source: DECC (2009)

Table 9: AUSRIVAS bands for spring – riffle habitat

Band Label	SIGNAL O/E50 (upper limit)	AUSRIVAS O/E50 (upper limit)	Band Name	Band Description
Band X	Infinity	Infinity	More biologically diverse than reference sites.	More taxa found than expected. Potential biodiversity hot-spot. Possible mild organic enrichment.
Band A	1.06	1.18	Reference condition.	Most/all of the expected families found. Water quality and/or habitat condition roughly equivalent to reference sites. Impact on water quality and habitat condition does not result in a loss of macroinvertebrate diversity.
Band B	0.93	0.8	Significantly impaired.	Fewer families than expected. Potential impact either on water quality or habitat quality or both, resulting in loss of taxa.
Band C	0.80	0.43	Severely impaired.	Many fewer families than expected. Loss of macroinvertebrate biodiversity due to substantial impacts on water and/or habitat quality.
Band D	0.67	0.06	Extremely impaired.	Few of the expected families remain. Extremely poor water and/or habitat quality. Highly degraded.

Source: DECC (2009)

The spatial distributions of monitoring sites within a subcatchment are illustrated in Figure 16 and further details for individual sites are available in Appendix D. Colour coding has been applied to Figure 16 to identify sites considered to be in good to very good condition (blue and green), fair condition or disturbed condition (yellow) and poor to very poor or severely to extremely impaired condition (red and pink). Sampling sites are often clustered rather than being distributed evenly throughout the WSP catchments and some WSP catchments have no record of macroinvertebrate sampling. Since the majority of sites have not been selected randomly, and the sample size for some subcatchments is relatively small, inference from the percentages above to the entire Hunter River catchment still need to be treated with caution.

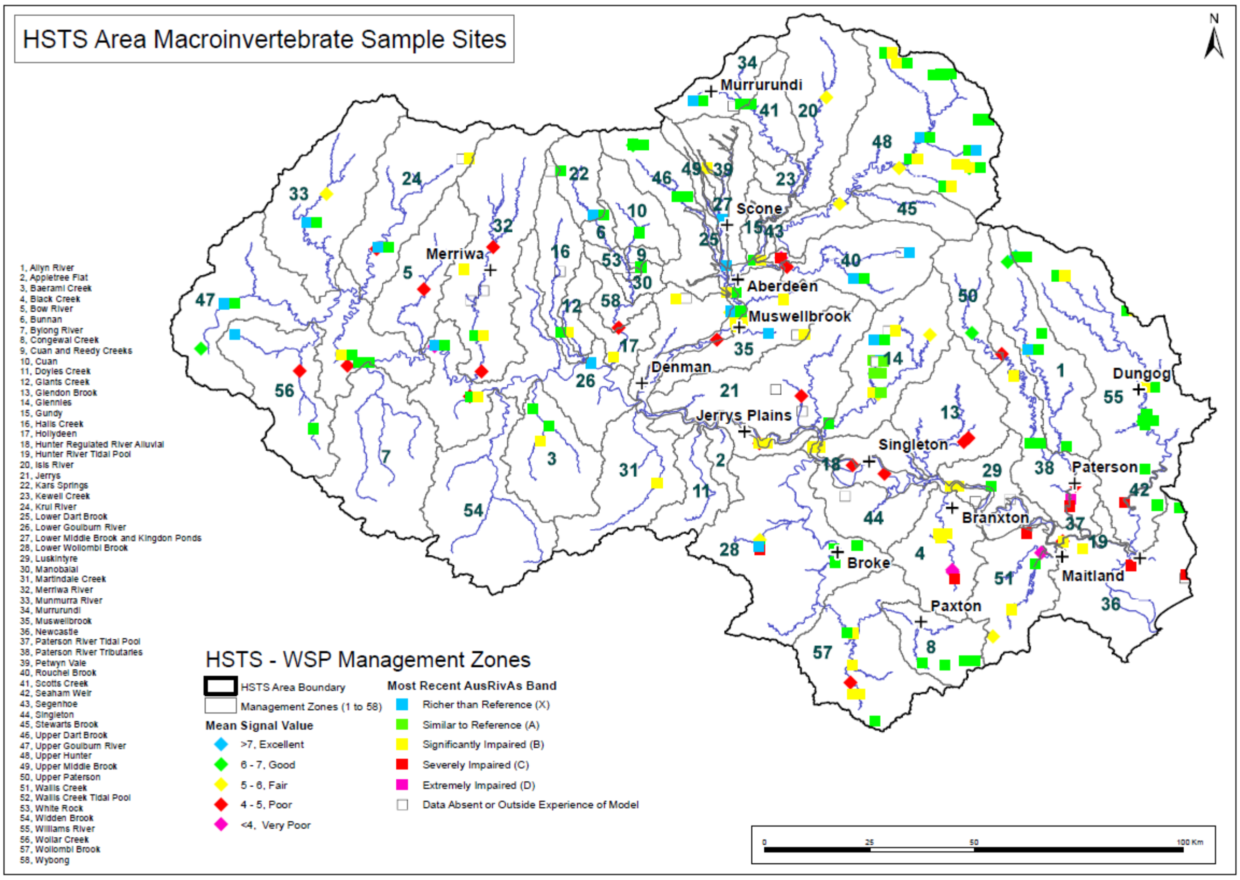


Figure 16: AUSRIVAS and SIGNAL scores for macroinvertebrates collected in the Hunter River catchment

Table 10: Site distribution and AUSRIVAS score for macroinvertebrates in the Hunter River catchment

Water sharing plan management zone	AUSRIVAS Band						SIGNAL	Total
	X	A	B	C	D	OEM		
Allyn River	1	3					2	6
Baerami Creek		2	1					3
Black Creek			3	1			2	6
Bow River							1	1
Congewai Creek		4						4
Cuan		1						1
Cuan and Reedy creeks		1				1		2
Glendon Brook							2	2
Glennies	2	7	2			2	2	15
Halls Creek		2	1				2	5
Hollydeen			1					1
Hunter Regulated River Alluvial	1	2	9			1	5	18
Isis River							1	1
Jerrys						2	1	3
Kars Springs	1	1						2
Krui River	1	1	1			1	1	5
Lower Dart Brook		1	1					2
Lower Goulburn River	2	2	1				2	7
Lower Middle Brook and Kingdon Ponds	2		1					3
Lower Wollombi Brook	1	4		1			1	7
Luskintyre		1	1			1		3
Martindale Creek			1					1
Merriwa River		1	2			2	2	7
Munmurra River	1	1					1	3
Murrurundi	1	3				1		5
Muswellbrook	1		3	1		2	1	8
Newcastle		1	2	2		1	1	7
Paterson River Tidal Pool							1	1
Paterson River Tributaries		2	1	1	1	1		6
Rouchel Brook	2	1				1	1	5
Singleton						1		1
Upper Dart Brook		4					1	5
Upper Goulburn River	2	4	1				2	9
Upper Hunter	2	13	7				3	25
Upper Paterson							2	2
Wallis Creek		1	1	1			2	5
Williams River		11	3	1			2	17
Wollar Creek	1	1					1	3
Wollombi Brook		2	4				1	7
Wybong							1	1
Totals	21	77	47	8	1	19	42	215

River regulation has been shown previously to have a significant impact on macroinvertebrate communities (e.g. Growns and Growns 2001, Marchant and Hehir 2002, DECC 2009). Salinity has also been suggested as a major contributor to impacts on macroinvertebrate communities (e.g. Kefford *et al.* 2005; Kefford *et al.* 2010; Cardno Ecology Lab Pty Ltd 2010 Dunlop *et al.* 2008, 2011; Cañedo Argüelles *et al.* 2013). In areas where flow regulation, land clearing, riparian degradation and saline waters all interact, it is often difficult to tease out the relative contributions of these confounding sources to altered macroinvertebrate communities. Further assessment of the community-level structure and its relationships with salinity has therefore been undertaken, specifically focusing on the effects of salinity on macroinvertebrates (Appendix E).

Impacts of saline water on Australian aquatic biota

Muschal (2006) assessed the the risk of elevated salinity to aquatic biota from the Hunter River. She found the aquatic biota of tributaries had a greater risk of impairment from high salinity than that of the Hunter River. High salinities in the tributaries were attributed to the combined factors of naturally saline geologies, increased liberation of salts due to modification of the landscape, and reduced dilution by flushing flows. There are also a number of other recent scientific publications that suggest increased levels of salinity can affect aquatic communities (e.g. Kefford *et al.* 2005; Kefford *et al.* 2010; Cardno Ecology Lab Pty Ltd 2010 ; Dunlop *et al*. 2008, 2011; Cañedo Argüelles *et al.* 2013; etc.). This prompted a specific analysis of macroinvertebrate data from the Hunter River catchment as part of the current salinity assessment. The analysis was led by Dr Ben Kefford of the University of Technology, Sydney (UTS) and was based on macroinvertebrate samples collected by OEH (and its predecessor organisations) over the past two decades as part of the Monitoring, Evaluation and Reporting (MER) and earlier Monitoring River Health Initiative (MRHI) Programs. The results of this analysis (Kefford *et al.* 2013) are included as Appendix E.

Although the Kefford *et al.* (2013) study was correlative and thus could not prove causality, it made the interim conclusion that salinity changes were likely (at least partly) to be causing changes in macroinvertebrate community structure in the Hunter River catchment. Large-scale changes in macroinvertebrate community structure were observed with relatively small changes in EC, including changes below 600 µS/cm and 900 µS/cm, the current targets for salinity levels in the upper and mid/lower Hunter River, respectively. Changes in community structure associated with similarly low salinity levels have been observed in Victoria and South Australia (Kefford *et al.* 2005, Kefford *et al.* 2010), the Appalachia Mountains, USA (Pond 2010, USEPA 2011, Passmore *et al.* 2012) and France (Piscart *et al.* 2005a, Piscart *et al.* 2005b, Piscart *et al.* 2006). If confounding factors were actually the cause of observed community changes (as opposed to the salinity itself), they would need to be invoked in a number of geographically distant locations with different causes of increased salinity. Furthermore changes in salinity below 600 or 900 µS/cm have been shown experimentally to affect the growth of stream macroinvertebrates (Kefford and Nugegoda 2005; Hassell *et al.* 2006; Kefford *et al.* 2006a,b; Kefford *et al.* 2007b), microinvertebrates (Kefford *et al.* 2007a) and freshwater fish (Boeuf and Payan 2001).

Important support for salinity impacts can also be found in the ACARP study on the effects of mine water salinity on freshwater biota (Cardno Ecology Lab Pty Ltd 2010). Cardno Ecology Lab Pty Ltd (2010) found that discharge waters from mines in the Hunter and Illawarra/Macarthur regions induced deleterious responses in a range of aquatic biota. Arthropods were the most sensitive organisms tested, with the mayfly *Atalophlebia* spp. being the most sensitive of these. The salinity levels at which effects occurred were below those reported in the literature for sodium chloride (NaCl) based solutions and highlighted the need for site-specific toxicity information that takes into account the variable composition of saline mine waters, including the consideration of other constituents (Cardno Ecology Lab Pty Ltd 2010).

Cardno Ecology Lab Pty Ltd (2010) used species sensitivity distribution (SSD) curves for single-species toxicity information to develop protective concentration (PC) values that protect a large proportion of the aquatic species present in the receiving waters. In Brennans Creek/Georges River (where West Cliff Colliery discharges) the SSD curve suggested a conductivity of 585 µS/cm to protect 95 per cent of species and a value of 921 µS/cm to protect 80 per cent of species. For Tea Tree Hollow (where Tahmoor Colliery discharges) the respective values were 1000 µS/cm and 1146 µS/cm; and for Bowmans Creek (where Ravensworth Colliery discharges) the respective values were 876 µS/cm and 1992 µS/cm. According to Cardno Ecology Lab Pty Ltd (2010), the small PC 95 per cent values determined for Brennans Creek/Georges River were strongly influenced by the sensitivity of the mayfly *Atalophlebia* sp. in the laboratory tests. No cladocerans, leptophlebiids or atyid shrimps were collected from Bowmans Creek and consequently, no field toxicity estimates were able to be derived for these taxa in Bowmans Creek (Cardno Ecology Lab Pty Ltd 2010).

Up until recently, the focus on salinity has primarily been associated with total dissolved solids (measured in milligrams per litre) or electrical conductivity [measured in microsiemens per centimetre (µS/cm)]. What is clear from Kellet *et al.* (1989) and many others is that surface waters, groundwaters and coal mine discharges often have very different ionic compositions. Different ions (sodium [Na^+], calcium [Ca^{2+}], magnesium [Mg^{2+}], potassium [K^+], chloride [Cl^-], bicarbonate [HCO_3^-], sulfate [SO_4^{2-}] and the salts they form) can induce varying degrees of toxicity to aquatic life (for example, Young 1923; Mossier 1971; Nelson 1968; Held and Peterka 1974; Rawson and Moore 1944; Farag and Harper 2012). Farag and Harper (2012) recently reviewed the potential effects of sodium bicarbonate ($NaHCO_3$), a major by-product of coalbed natural gas production, on aquatic life. They cited Mount *et al.* (1997) who completed more than a thousand acute experiments and developed a multiple regression model that described the toxicity of common ions in various combinations to zooplankton and fathead minnows (*Pimephales promelas*). One of the major findings of Mount *et al.* (1997) was that all major ions have a lethal concentration, and the toxicity of a mixture of salts is generally equivalent to the additive toxicity of the individual salts (Farag and Harper 2012).

Farag and Harper (2012) constructed a database of toxicity evaluations of sodium bicarbonate ($NaHCO_3$) on aquatic life and used these data to establish acute and chronic criteria for the protection of aquatic life. Chronic toxicity was observed at concentrations that ranged from 450 to 800 milligrams $NaHCO_3$ per litre (also defined as 430 to 657 milligrams HCO_3- per litre or total alkalinity expressed as 354 to 539 milligrams $CaCO_3$ per litre) and the specific concentration depended on the sensitivity of the four species of invertebrates and fish exposed. Acute and chronic criteria of 459 and 381 milligrams $NaHCO_3$ per litre, respectively, were calculated to protect 95 per cent of the most sensitive species (Farag and Harper 2012). More recently, OEH (2012) also found toxic effects of West Cliff mine water, citing bicarbonate as an important potential contributor to the toxic effects. Other potential toxicants found in the mine water at levels exceeding the ANZECC/ARMCANZ (2000) guidelines were aluminium, nickel, zinc, cobalt and copper (OEH 2012).

If protection of 95 per cent of species was used to identify a suitable target for ecosystem protection in the Hunter River catchment[10], then the ACARP results suggest

[10] In most cases the 95 per cent protection level trigger values should apply to ecosystems that could be classified as slightly to moderately disturbed, although a higher protection level could be applied to slightly disturbed ecosystems where the management goal is no change in biodiversity (ANZECC/ARMCANZ 2000).

that 900 µS/cm may potentially be an appropriate upper level target for EC levels overall (albeit slightly higher than the 876 µS/cm 95 per cent protection level calculated by Cardno Ecology Lab Pty Ltd 2010 for Bowmans Creek). It needs to be recognised that the Scheme salinity targets themselves only apply to the main stem of the Hunter River between Glenbawn Dam and Singleton and not in the tributaries. Nevertheless, based on the scientific evidence available and adopting a precautionary approach, the upper salinity target for the Scheme (currently set at 900 µS/cm in the lower sector) should not be raised without further justification and experimentation. The 600 µS/cm target currently set for salinity levels in the upper sections of the Hunter River may actually provide a more conservative level of protection from salinity impacts in these areas than those further downstream where the 900 µS/cm target applies. However, caution needs to be exercised in these conclusions since such a focus purely on EC may mask the effects of different ionic compositions and any additional effects of other constituents in the mine and power generation water discharges. The results of Farag and Harper (2012) and OEH (2012) suggest greater caution needs to be exercised with mine water high in sodium bicarbonate ($NaHCO_3$). There may also be issues associated with metal/metalloid pollution (e.g. aluminium, nickel, zinc, cobalt and copper) since levels for some of these pollutants have been found in mine waters at levels exceeding ANZECC/ARMCANZ (2000) guidelines for the protection of aquatic ecosystems (e.g. OEH 2012).

It is clear that further experimental studies are required to fully understand the environmental effects of the highly variable saline mine water compositions discharged to the Hunter River catchment. Kefford *et al.* (2013) make a number of recommendations for the specific types of experimental studies required in this context, including experimental mesocosm studies; field studies at targeted sites; and long-term laboratory experiments to determine the chronic and sublethal salinity sensitivity of macroinvertebrate taxa. A program investigating the whole of effluent toxicity of the various mine waters prior to discharge is also required to see:

1. whether any toxicity exists, and
2. the degree of dilution (if any) required to mitigate any potential toxic effects.

Ecosystem health summary

Limited information is available on the macroinvertebrate community structure of the Hunter River prior to the implementation of the Scheme. It is therefore impossible to make any before and after Scheme comparisons in terms of macroinvertebrate health. There is also insufficient understanding of macroinvertebrates in 'naturally' saline areas (e.g. Saltwater Creek) prior to the extensive land clearing and development in the catchment. Overall, macroinvertebrate 'health' throughout the catchment is good on average based on the available data, but there are some areas that are quite poor in terms of macroinvertebrate health. A relatively high number of samples (n = 9) in the Hunter Regulated River Alluvial Zone were found to be in a significantly impaired (AUSRIVAS band B) condition.

A specific analysis of macroinvertebrate data from the Hunter River catchment was undertaken as part of the current salinity assessment (Kefford *et al.* 2013). This and other scientific research suggests that saline discharges can potentially have impacts on macroinvertebrate communities at conductivity levels similar to or below those currently being discharged by Scheme participants. In addition, simply focusing on total dissolved solids or EC does not necessarily allow for the effects of discharges of differing ionic composition or other contaminants (e.g. metals/metalloids) that may be in the mine and power generation water discharges. High levels of bicarbonate, in particular, have been shown to have toxic effects in some areas (e.g. Farag and Harper 2012, OEH 2012). Further experimental studies are required to fully understand the environmental effects of the highly variable saline mine and power generation water compositions currently being discharged to the Hunter River catchment. Kefford *et al.* (2013) make a number of recommendations for the specific types of experimental studies required in this context.

Given that the Scheme salinity targets only apply to the Hunter River between Glenbawn Dam and Singleton and not within any of the tributaries, and the targets themselves only apply during high or flood flow periods, the Regulation itself provides limited ability to control the more general impacts of high salinity waters on aquatic health. The weight of scientific evidence suggests that the salinity targets for the scheme should not be raised, but further work is required to better understand existing salinity impacts on ecosystem health in the Hunter River and its catchments. The major impact of the Regulation on ecosystem health is likely to have been the continued restriction of discharges to high and flood flows when the potential for dilution is at its greatest (as opposed to continuous or intermittent discharges regardless of flow conditions). As the Regulation does not regulate salinity levels in the tributaries, any impacts of high salinity discharges on ecosystem health in the Hunter River tributaries may be more appropriately managed through other means (e.g. through environment protection licence conditions).

7. Conclusions

There are various potential sources of salinity in the Hunter River catchment including rainfall, atmospheric deposition, run-off and infiltration, weathering of geological strata, groundwater and a variety of anthropogenic sources including the Scheme. Significant spatial variability in atmospheric deposition of salts occurs across the Hunter River catchment with salt loads generally decreasing with increasing distance from the coast. Some point sources of natural salt contamination existed prior to European settlement in the Central Lowlands (Jerrys Plains and surrounds) or were contemporaneous with it, as evidenced by early geographic names with salinity connotations – such as Saltwater Creek (Kellet *et al.* 1989). Widespread land clearing has probably exacerbated degradation of the land and promoted dryland salinity in some areas due to increased run-off, erosion and rising water tables, particulalrly in the Central Lowlands.

Since rising groundwater levels and/or EC can affect stream water quality (e.g. stream EC), any increase in groundwater level or EC could potentially have a confounding effect on the interpretation of Scheme effectiveness.There was little evidence that groundwater levels or the EC of groundwater have been rising in recent times (except perhaps in some very localised areas). This conclusion is affected to some degree by limited temporal sampling and a bias to current monitoring bores being located in alluvial areas often well away from the areas of major mining operations.

If future trends in groundwater level and conductivity are to be undertaken and related back to the impact of the Scheme (or mining and power generation), then a more comprehensive and representative groundwater monitoring program is required for the catchment.

In terms of addressing the three specific questions posed by the EPA (see the Introduction to this report), the following conclusions are made:

1. Has the Regulation impacted on aquatic ecosystems and associated environmental values since it commenced in 2002?

The major impact of the Regulation on ecosystem health is likely to have been the continued restriction of discharges to high and flood flows when the potential for dilution is at its greatest (as opposed to continuous or intermittent discharges regardless of flow conditions).

The assessment of the overall effectiveness of the Scheme on surface water quality relied upon the partitioning of available data into three major time periods: *before* Scheme operation (1970s and 1980s), *during the initial stages* of the Scheme (1990s) and *after formal commencement* of the Scheme (2000s to present). Provided these periods are adequate surrogates for the various stages of Scheme operation, and that natural EC sources have remained relatively constant between time periods, then the data suggests that the Scheme has:

- had little effect on flows and EC levels in the Hunter River upstream of Denman
- improved EC levels at (and immediately upstream of) Singleton and Greta, and
- potentially improved EC levels at monitoring stations between Denman and Singleton, but limited if any EC data were available at these latter stations during the 1970s & 1980s to clarify pre-existing EC levels prior to the Scheme commencing (in 'pilot' or 'full' implementation).

The generalised additive modelling (GAM) and non-linear time trend analysis support the conclusions from the assessment of the distribution of flow and EC levels above, that is:

- There is evidence of cyclical temporal trends in EC levels in the Hunter River at Muswellbrook, but EC levels do not appear to be either increasing or decreasing over time.
- There is evidence of cyclical temporal trends in EC levels in the Hunter River at Singleton, and EC levels appear to have declined over the most recent time periods.

If discharges under the Scheme were increasing conductivity levels in the Hunter River at times of high or flood flows, then such an increase would be reflected in the EC–flow relationships in recent times, with higher flows leading to higher EC levels. Such a pattern was not evident in the data from the Hunter River monitoring sites.

Limited information was available on the macroinvertebrate community structure of the Hunter River prior to the implementation of the Scheme. It is therefore impossible to make any before and after Scheme comparisons in terms of macroinvertebrate health. There is also insufficient understanding of macroinvertebrates in 'naturally' saline areas (e.g. Saltwater Creek) prior to the extensive land clearing and development that occurred in the catchment.

Based on the available data, macroinvertebrate 'health' throughout the Hunter catchment is on average good, but there are some areas that are quite poor in terms of macroinvertebrate health. A relatively high number of samples (n = 9) in the Hunter Regulated River Alluvial Zone were found to be in a significantly impaired (band B) condition.

Although salinity is one of several factors affecting stream macroinvertebrate communities in the Hunter River catchment, salinity appears to be a relatively important factor. A number of scientific studies suggest that saline discharges can potentially have impacts on macroinvertebrate communities at conductivity levels similar to or well below those currently being discharged by Scheme participants.

Assessment of the effectiveness of the Scheme is largely regulated and assessed on the basis of salinity expressed as either total dissolved solids [milligrams per litre (mg/L)] or EC [microsiemens per centimetre (µS/cm)]. Simply focusing on total dissolved solids or electrical conductivity does not necessarily allow for the effects of discharges of differing ionic composition or other contaminants (e.g. metals/metalloids) that may be in the mine water discharges. High levels of bicarbonate, in particular, have recently been shown to have potentially toxic effects in some areas (e.g. Farag and Harper 2012, OEH 2012).

Since the Scheme restricts discharges to periods of high and flood flows, the level of dilution achieved by the Scheme is very important in determining whether impacts to ecosystem health occur. The weight of scientific evidence currently suggests that the salinity targets for the Scheme should not be raised, but further work would be required to better understand existing salinity impacts on ecosystem health in the Hunter River and its tributaries. The Regulation itself does not address potential saline impacts on ecosystem health in the Hunter River tributaries.

Saline wastewater discharges under the Scheme contributed on average approximately 10 per cent of the entire salt load of the Hunter River at Singleton. Since the formal implementation of the Scheme in 2002, there has been a slight increase in the number of participants and on average industry participants have utilised 25 per cent of the given opportunities. However, in recent times significant salt loads (approximately 13.5 to 19.8 per cent of total annual salt load in the Hunter River) have been discharged to the Hunter River and the value of salt credits is increasing, while at the same time the utilization of the TAD has also increased (to 40–50 per cent).

2. Does the Regulation have the potential to impact on aquatic ecosystems and associated environmental values in the future?

If future discharges occur in a similar manner, frequency and EC concentration to those over the past two decades, then similar effects could be expected to those described above. However it is noted that over the past few years there has been an increasing demand (as indicated by the value of salt credits), salt load and TAD usage under the Scheme. Additional demand for saline discharge under the Scheme is also coming from new or expanded mining operations. While impacts on Hunter River EC levels over the past decade (as monitored at the three reference sites) appear limited at this stage, further assessment is required if the salt load and TAD utilisation continues to increase.

Ongoing monitoring of discharge demand, salt loads and TAD usage under the Scheme is required to assess the potential for future trends or changes to impact aquatic ecosystems and environmental values.

As mentioned above, the weight of scientific evidence suggests that current Scheme salinity targets should not be raised. Further scientific analysis and modelling would be required to support altering the Scheme salinity targets in the future.

As identified above, assessment of the effectiveness of the Scheme is largely regulated and assessed on the basis of salinity expressed as either total dissolved solids (mg/L) or EC (µS/cm). The potential effects of discharges of differing ionic composition or other contaminants (e.g. metals/metalloids) that may be in the mine water discharges still requires further investigation. Further experimental studies are recommended in order to fully understand the environmental effects of these discharges.

3. What other sources of salinity in the Hunter catchment could influence the operation of the Scheme in the future?

Hunter River salt loads can also be affected by the major tributaries such as the Goulburn River and Wollombi Brook.

While the Scheme itself does not apply to the Goulburn River upstream of Kerrabee, high salinity water from tributary sources can affect EC levels and discharge opportunities in the Hunter River downstream of their confluences. Goulburn River salt loads are highly variable and dependant on subcatchment source, but can at times be greater than the salt load measured in the Hunter River at Denman. Three mines (Ulan, Wilpinjong and Moolarben) currently have discharge licences in the Upper Goulburn River catchment, and further mining and CSG exploration is proposed for this area. With this likely expansion of mining and coal seam gas extraction, and the lack of real-time monitoring in the both the upper and lower sections of the Goulburn River catchment, strategic real-time monitoring of flow and salinity in other areas of the Goulburn River catchment is recommended.

Further assessment is necessary to fully understand the underlying mechanisms which yielded the high EC levels in Wollombi Brook at Warkworth, but these relatively high levels also have the potential to reduce the opportunities of the Scheme by increasing the EC contributed by Wollombi Brook waters where they join the Hunter River. Fortunately the very high EC levels measured in the mid to late 2000s have now declined, but still need ongoing monitoring.

Most other monitoring stations throughout the catchment showed little evidence of increasing EC levels, except potentially during the 2000 to 2007 drought. The interaction of rainfall, flow and groundwater contribution needs further assessment in these areas to fully understand the effects of drought on surface water EC levels in the Hunter River catchment.

A return to drought conditions in the Hunter River catchment could lead to reduced flow and increases in EC levels in the Hunter River and its tributaries and decrease the opportunities for saline discharges under the Scheme.

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