Independent Expert Scientific Committee on Coal Seam Gas and Large Coal Mining Development

Australian Government

Department of the Environment

Background review

Co-produced water - risks to aquatic ecosystems

This background review was commissioned by the Department of the Environment on the advice of the Interim Independent Expert Scientific Committee on Coal Seam Gas and Coal Mining. The review was prepared by Sinclair Knight Merz Pty Ltd and revised by the Department of the Environment following peer review.

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Acknowledgements

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Addendum

Changes to state government departments have occurred since the finalisation of this report by the authors. The Queensland, New South Wales and South Australian Government agencies were contacted and updated information provided in September 2013; however, no guarantees can be made as to the completeness of these updates. Up-to-date information should be sourced from the relevant department.

On 1 January 2013, the Queensland Water Commission (QWC) ceased operations. The Office of Groundwater Impact Assessment (OGIA) retains the same powers as the former QWC under Chapter 3 of the *Water Act 2000* (Qld).

Sinclair Knight Merz Pty Ltd is now Jacobs SKM.

Contents

Summary

This report provides an overview of Australian and international experiences of coal seam gas and coal mining co-produced water and risks to aquatic ecosystems.

Key points

- Co-produced water is that generated by the extraction of coal seam gas and dewatering of coal mines.
- Volumes of co-produced water vary significantly between coal seam gas sites in Australia (190 megalitres (ML) per petajule of gas in the Surat Basin compared with 1.2 ML per petajule of gas in the Sydney Basin).
- In 2010, the co-produced water volume across Australia was estimated to be 33 gigalitres (GL) per year, of which 40 per cent was from coal seam gas extraction. Over the next 25 to 35 years, co-produced water volumes are estimated to be larger, driven by projected development of coal seam gas sources in Queensland and New South Wales.
- The quality of water also varies but typically contains elevated levels of salts. Other variables that may require management attention include: temperature, pH, sodium, chloride, fluoride, boron, some heavy metals, ammonia, and phosphorus.
- Management options include: re-using the water for agriculture or other industries; injection; and discharge to surface water systems.
- The impacts of these options in the environment must be assessed on a case-by-case basis – a risk-based, quantitative approach that takes into account cumulative impacts is advocated.
- Key risks to be considered when discharging co-produced water to surface water systems include salinity, toxicity and changes in flow regime, especially for streams that are weakly perennial or ephemeral.

Co-produced water

Co-produced water is also known as ‘associated water’ and is generated through the extraction of coal seam gas and the dewatering of coal mines. For coal seam gas, the quality of water extracted varies depending on the characteristics of the coal seam and whether hydraulic fracturing is used. For coal mining, it depends on the characteristics of the coal seams and overlying strata. Where hydraulic fracturing occurs in Australia, a volume of fluid (including groundwater), generally equivalent to 110 to 150 per cent or more of the volume of injected fluid, is pumped from the well soon after the hydraulic fracturing has occurred. This water is often referred to as ‘flowback water’. Often, flowback water containing elevated levels of hydraulic fracturing chemicals is managed separately from other co-produced water, but any chemicals not recovered in flowback water may be present in other co-produced water. For the purposes of this review, co-produced water does not include flowback, but does include water produced from coal mining, which is commonly referred to as ‘mine affected water’.

The volumes of co-produced water will vary significantly between coal seam gas sites. In the Surat Basin around 190 ML of co-produced water is generated per petajoule (PJ) of gas produced, compared with 1.2 ML/PJ in the Sydney Basin. It has been estimated that in 2010, the co-produced water volume across Australia was 33 GL/year with 40 per cent from coal seam gas and 60 per cent from conventional gas and oil. Over the next 25 to 35 years it is estimated that coal seam gas co-produced water will increase, driven by projected development of coal seam gas sources in Queensland and New South Wales. The volumes of co-produced water generated from coal mining are also significant and vary seasonally according to rainfall, local groundwater recharge and on-site operational water demand.

Co-produced water typically contains elevated levels of salts that are toxic in high concentrations to freshwater plants and animals. A number of water quality variables such as temperature, pH, sodium, chloride, fluoride, boron, some heavy metals, ammonia, and phosphorus may require management attention. Specific constituents and their concentrations can vary widely between wells within a production area and across regions, so impacts can only be assessed on a case-by-case basis.

Co-produced water management options

Because of the volumes and quality of water involved, co-produced water must be managed strategically. The main management options include:

- direct beneficial reuse in agriculture, mining and other industry operations
- injection into depleted aquifers for recharge purposes
- discharge to surface water systems.

A preferred option for the management of co-produced water in coal seam gas producing areas in Queensland is to use it for a purpose that is beneficial to the environment, other water users, or water-dependent industries. If co-produced water cannot be reused on-site or provided to an acceptable end-user, under certain conditions regulations may allow it to be discharged to water systems, including rivers, streams, reservoirs, aquifers and wetlands. When discharged to the environment, the co-produced water may need to be treated to remove salts and other contaminants and be released in a way that firstly avoids, and then minimises and mitigates, impacts on environmental values.

Water quality impacts and aquatic ecosystems

Treatment of co-produced water can introduce a range of other issues. It can make the water too low in salts or turbidity for reuse. Pre-release treatment is often required to re-mineralise water to make it suitable for specific beneficial uses, especially for use in irrigation and for release into aquatic ecosystems.

Although water is treated to meet specified conditions there is still a risk that some water quality constituents may interact with each other to modify toxic impacts. It is often not possible to accurately predict the toxicological effects of multiple toxicants on aquatic biota by simply applying an additive approach. This is largely due to unpredictable synergistic and antagonistic effects that toxicants can exhibit and is a particular risk where multiple discharges occur in a catchment. Site-specific investigations are needed to understand factors controlling contaminant bioavailability and toxicity and determine thresholds for significant impacts. Direct toxicity testing should be considered as part of any assessment, especially since water quality guidelines are not available for all contaminants.

Limiting factors in relation to water quality assessment and management include:

- variability in water quality: water quality varies widely between wells/mines and across regions and there is limited data to enable regional-scale characterisation or to permit water-type assessment
- limitations to water quality guidelines: current water quality guidelines do not cover all stressors and toxicants potentially present in co-produced water; guidelines mostly address water quality in perennial streams and may not be suitable for use in ephemeral systems, which are common in major coal seam gas and coal mining regions
- understanding of cumulative impacts: cumulative impacts at a landscape scale are not well understood; even when individual discharges meet relevant guidelines, the cumulative effects associated with increasing load contributions such as salts, nutrients or heavy metals and increased flow may have downstream impacts
- regulatory implementation: licence conditions for treated waste water discharge may be generic; assessment is needed to match the discharge quality and quantity to the specific requirements of the receiving waterway.

Water quantity impacts and aquatic ecosystems

The changes in flow regime from release of large volumes of co-produced water represent a risk to aquatic ecosystems. The level of risk depends on the timing and volume of the release and on how significantly the co-produced water will change the water regime in the receiving environment. Discharging a small volume of co-produced water may pose less significant risk for streams that are strongly perennial and carry large flow volumes. However, for streams that are weakly perennial or ephemeral, an increase in flow can pose a significant risk. It can change the entire flow regime and result in ephemeral streams becoming perennial and seasonal wetlands becoming permanently inundated. This can lead to increased nuisance plant and algal growth, colonisation by pest species and loss of native species that require a dry phase to complete their life cycles.

Most streams in coal seam gas producing and large coal mining areas are weakly perennial or ephemeral – the stream types most at risk from increased flow.

Environmental risk assessment

Most environmental risk assessments (ERA) relating to coal seam gas and coal mining are based largely on the Standards Australia and Standards New Zealand risk assessment guidelines. Current 'best practice' is to use risk-based approaches to assess the risks of new coal seam gas and coal mining developments to key environmental assets. The outcome of such an assessment is a risk management strategy to minimise impact.

This report reviews available frameworks to assess the risks related to the disposal or use of co-produced water from coal seam gas and coal mining. It suggests that quantitative ERAs, involving the development and use of quantitative models such as Bayesian networks, are preferable to qualitative risk assessments, but will depend upon the level of data available. The development of conceptual models identifying the relationships between key stressors and water-dependent assets and receptors to be protected is an essential pre-condition for undertaking a quantitative ERA process.

ERAs should address the cumulative risks associated with expansion of coal seam gas and coal mining developments. Quantification of cumulative effects of multiple developments will require the development of regional-scale models, which could build on existing hydrological-water quality models, such as those based on the Integrated Quantity Quality Model (IQQM).

They should also include groundwater and surface water interactions and be able to link with ecosystem response models that relate flow to ecosystem response.

Regarding the discharge of co-produced water to waterways, assessments of ecological risks should, at a minimum:

- identify key threats from (a) increased salinity, (b) increased toxicity and (c) changes in the flow regime, particularly in ephemeral streams
- assess risks to key ecological indicators (assets) including: (a) threatened species and communities , (b) fish communities, (c) macroinvertebrate communities and (d) riparian vegetation
- assess possible cumulative risks due to other existing/planned coal seam gas or coal mining developments
- use appropriate modelling techniques, particularly those that quantify the relationships between key threats and key ecological indicators
- refer to local data in addition to comparison against water quality guidelines.

Abbreviations

General abbreviations	Description
ABARE	Australian Bureau of Agricultural and Research Economics
ANZECC	Australia and New Zealand Environment Conservation Council
ARD	Acid rock drainage
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
AVIRA	Aquatic value identification and risk assessment
BN	Bayesian Network
BTEX	Benzene, toluene, ethylbenzene and xylene compounds
C_2H_6	Chemical formula for ethane
CBM	Coal bed methane
CH_4	Chemical formula for methane
CIAT	Cumulative impacts assessment tool
cm	Centimetre
CO_2	Chemical formula for carbon dioxide
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CSG	Coal seam gas
CWMP	Coal seam gas water management plan
DO	Dissolved oxygen
DVWSS	Dawson Valley Water Supply Scheme
EC	Electrical conductivity
ED	Electrodialysis
EDR	Electrodialysis reversal
EIS	Environmental impact statement
EPA	Environment Protection Authority
EPBC Act	*Environment Protection and Biodiversity Conservation Act 1999*
ERA	Ecological risk assessment
ERASC	Ecological Risk Assessment Support Centre
FRP	Filterable reactive phosphorus
GA	Geoscience Australia
GAB	Great Artesian Basin
GAC	Granular activated carbon
GDP	Gross domestic product
GL	Gigalitre (1000 million litres)

General abbreviations	Description
GRIDD	Groundwater and Resource Information for Development Database
H_2O	Chemical formula for water
IESC	Independent Expert Scientific Committee on Coal Seam Gas and Large Coal Mining Development
IQQM	Integrated quantity quality model
LNG	Liquefied Natural Gas
L/s	Litre per second
m	Metre
MDB	Murray-Darling Basin
mg	Milligram
ML	Megalitre (1 million litres)
mm	Millimetre
MNES	Matters of National Environmental Significance
NH_3	Un-ionised ammonia
nm	Nanometre
NO_2	Nitrite
NO_3	Nitrate
NSW	New South Wales
NTU	Nephelometric turbidity units
NVDI	Normalised Vegetation Difference Index
NWQMS	National Water Quality Management Strategy
OWS	Office of Water Science
PCU	Platinum-cobalt units
PJ	Petajoules
PRA	Probabilistic risk assessment
RO	Reverse osmosis
RQ	Risk quotient
SAR	Sodium adsorption ratio
SS	Suspended Solids
TDS	Total dissolved solids (a measure of salinity)
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphorus
TWS	Town water supply
US EPA	United States Environmental Protection Agency
USA	United States of America

General abbreviations	Description
WCM	Walloon Coal Measures
WRP	Water resource plan
WTP	Water treatment plant

Glossary

Term	Description
Acidic	Having a high hydrogen ion concentration (low pH).
Adsorption	The reversible binding of molecules to a particle surface. This process can bind methane and carbon dioxide, for example, to coal particles.
Algae	Comparatively simple chlorophyll-bearing plants, most of which are aquatic and microscopic in size
Alkalinity	The quantitative capacity of aqueous media to react with hydroxyl ions. The equivalent sum of the bases that are titratable with strong acid. Alkalinity is a capacity factor that represents the acid-neutralising capacity of an aqueous system.
Anoxia	No or very low dissolved oxygen. Levels are typically less than 2 mg/L.
Aquatic ecosystem	Any watery environment from small to large, from pond to ocean, in which plants and animals interact with the chemical and physical features of the environment.
Benthic	Refers to organisms living in or on the sediments of aquatic habitats (e.g. lakes, rivers, ponds).
Bioaccumulation	General term describing a process by which chemical substances are accumulated by aquatic organisms from water, either directly or through consumption of food containing the chemicals.
Bioavailable	The fraction of the total of a chemical in the surrounding environment that can be taken up by organisms. The environment may include water, sediment, soil, suspended particles, and food items.
Biofilm	A living layer of microorganisms that exist in a mucilaginous, polysaccharide coating on submerged substrates in streams.
Chemical transformations	Chemical transformations of substances can occur through acid-base reactions or redox reactions. For example, ammonia and metal toxicity and/or availability changes under varying temperatures, pH and oxygen levels.
Carbon cycling/ decomposition of organic matter	Carbon cycling involves the oxidation of complex organic matter into simpler forms (e.g. carbon dioxide, phosphate, ammonia). It is one of the key steps in the decomposition of organic matter. This provides bacteria, protozoa and fungi (at the base of the foodweb) with the energy for cellular metabolism and growth. This process consumes oxygen.
Contaminant	Biological (e.g. bacterial and viral pathogens) and chemical (see ‘Toxicants’) introductions capable of producing an adverse response (effect) in a biological system, seriously injuring structure or function or producing death.
Diatom	A member of the algal group Bacillariophyta, characterized by a cell wall of two siliceous valves.
Dilution	Dilution is the process of making a substance less concentrated by adding water. This can lower the concentrations of ions, toxins and other substances.
Ecological services	Ecological services include provisioning services such as food and water;

Term	Description
	regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services such as nutrient cycling that maintain the conditions for life on Earth.
Environmental values	Environmental Values, as defined in the NWQMS [Doc 4, Chapter 2.1.3, page 2-6], include: aquatic ecosystems; primary industries; recreation and aesthetics; drinking water; industrial water; and cultural and spiritual values.
Ephemeral streams	Ephemeral streams receive water only for a short period very occasionally and highly unpredictably.
Episodic streams	Episodic streams fill occasionally and may last months or years during unpredictable rainfall.
Evapo-concentration	Evapo-concentration is the process by which water is evaporated and the substances present, particularly salts, concentrate.
Flocculation	Flocculation is the aggregation of colloidal (very fine) particles into larger particles that then settle. This occurs in high salinity environments (e.g. estuaries).
Flowback	The fluid that flows back, or is pumped back, to the surface following hydraulic fracturing but prior to gas production.
Heterogeneity	Composition from dissimilar parts.
Hydraulic fracturing	The injection of fluids, gas, proppant, and other additives under high pressure into a geological formation to create a network of small fractures radiating outwards from the well through which the gas, and any associated water, can flow. Also known as 'fracking', 'fraccing', or 'fracture stimulation'.
Hypoxia	Deficiency of oxygen (typically less than 5 mg/L).
Intermittent streams	Intermittent streams receive water quite frequently, either predictably or unpredictably.
Littoral vegetation	Vegetation that exists in the edge or shore region where water is shallow enough for continuous mixing.
Macroinvertebrate	Animals with no backbone and which are visible to the naked eye.
Nutrient cycling	The nutrient cycle describes how nutrients move from the physical environment into living organisms, and are then recycled back to the physical environment. This movement of nutrients, sometimes referred to as nutrient spiralling is essential for life, and is a vital function of the ecology of aquatic ecosystems. There are four biological processes that participate in the cycling of nitrogen. They are Nitrogen fixation $N_2 \rightarrow NH^{4+,}$ Decay: $Organic\ N \rightarrow NH^{4+,}$ Nitrification: $NH^{4+} \rightarrow NO^{2-} \rightarrow NO^{3-}$ and De-nitrification: $NO^{3-} \rightarrow N_2 + N_2O$.
Oviposition	The deposition or laying of eggs.
Osmoregulation	Maintenance of an optimal, constant osmotic pressure in the body of a living organism. Osmotic pressure is essential for the intake of water by plant cells.
Oxidation	The combination of oxygen with a substance, or the removal of hydrogen from it or, more generally, any reaction in which an atom loses electrons.
Perennial streams	Streams that flow year round.

Term	Description
Permeate	To spread or flow throughout.
Photosynthesis	The conversion of carbon dioxide to carbohydrates in the presence of chlorophyll using light energy.
Precipitation of minerals	The formation of a solid when a dissolved substance settles out of the water.
Primary production	Primary production refers to the creation of new organic matter by photosynthesis. Oxygen is produced during this process. Primary production occurs in aquatic systems and includes algae and macrophyte growth.
Re-aeration	The transfer of oxygen from the atmosphere to a body of water at the air/water interface. It affects the dissolved oxygen levels.
Remnant pool	A body of water that persists after a stream has ceased to flow.
River reach	Any length of river between two points.
Sedimentation	The settling of suspended particles. Rates of sedimentation are determined by the size of the particles, the velocity of the water and the ionic environment. Sedimentation affects the water clarity.
Sodium Adsorption Ratio	The ratio of sodium to calcium and magnesium in water. Water with high SAR causes dispersion of soil particles, loss of the ability of the soil to form stable aggregates and a reduction in infiltration and permeability with consequences for crop production. Water with high SAR requires treatment if it is to be suitable for irrigation.
Sorption	A physical and chemical process by which one substance becomes attached to another.
Stratification	The formation of density layers (either temperature or salinity derived) in a water body through lack of mixing. It can create favourable conditions for algal blooms and can lower dissolved oxygen levels in the bottom layers with the associated release of nutrients, metals and other substances.
Temporary stream	Temporary is a general term for non-permanent aquatic systems.
Thermocline	A steep temperature gradient in a body of water such as a lake, marked by a layer above and below which the water is at different temperatures.
Toxicant	A chemical capable of producing an adverse response (effect) in a biological system at concentrations that might be encountered in the environment, seriously injuring structure or function or producing death. Examples include pesticides and heavy metals.
Unconsolidated sediments	Sediments that are not bound or hardened by mineral cement, pressure or thermal alteration.
Water quality	The physical, chemical and biological attributes of water that affect its ability to sustain environmental values.
Water quantity	Water quantity describes the mass of water and/or discharge and can also include aspects of the flow regime, such as timing, frequency and duration.

1 Introduction

This background review is one of a number commissioned by the Department of the Environment on the advice of the Interim Independent Expert Scientific Committee on Coal Seam Gas and Coal Mining. These reviews aim to capture the state of knowledge on the water-related impacts of coal seam gas extraction and large coal mining, but do not aim to provide detailed analysis and evaluation of methods for identifying and managing impacts or to develop such methods.

The subject of this report is co-produced water, including issues associated with quantity, quality, timing and potential risks to aquatic ecosystems and their environmental values. The report focuses primarily on co-produced water in Australia, but also includes some international context. It was prepared from information available in the public domain, including:

- scientific journals
- conference proceedings
- scientific text books
- government standards, guidelines and technical reports
- industry/consultant development assessment reports.

The review was commissioned to provide a critique of:

- impacts of changed surface water flow regimes and quality due to discharge of treated co-produced water
- existing knowledge, including documentation on environmental impact assessments
- examples of releasing co-produced water into natural flow regimes with a high seasonal variability
- risk management frameworks and their applicability to coal seam gas and coal mining activities
- industry practice in managing co-produced water.

The report commences with a discussion of co-produced water from both coal seam gas extraction and coal mining and provides background to aquatic ecosystems and co-produced water management options. It then describes issues associated with the quality and quantity of co-produced water, including primary toxicants, water treatment options, importance of natural flow regimes and ecological risks associated with changing the natural regimes. The report then describes risk management frameworks and the current situation regarding publically available impact assessments and water management options being proposed for future co-produced water management.

2 Co-produced water

Co-produced water is also known as 'associated water'. It is generated through industrial processes and energy production from shale gas, conventional natural gas and coal seam gas (Alley et al. 2011). For the purposes of this review, co-produced water also refers to water produced from large coal mining activities, although the term co-produced water is more commonly associated with coal seam gas. The quality and quantity of co-produced water varies between areas and specific operations, but is typically poor quality with moderate to high total dissolved solids (salinity) and minerals, and may also contain drilling and hydraulic fracturing fluids (SKM 2011; Shaw 2010). Where hydraulic fracturing occurs, the industry typically refers to 'flowback' as the fluid, including groundwater, produced within a few days of the fracturing operation, and to 'co-produced water' as the water that is produced after that (Batley & Kookana 2012), including during the entire coal seam gas production phase.

2.1 Co-produced water from coal seam gas

Coal seam gas is trapped on the surfaces and in the fractures and cleats of a coal seam by groundwater hydrostatic pressure (RPS 2011). Dewatering and depressurisation decreases the hydrostatic pressure and releases the gas. Water production is typically highest in the initial stages and decreases over time (Figure 1) (RPS 2011). Significant quantities of groundwater may be extracted as a by-product of coal seam gas production.

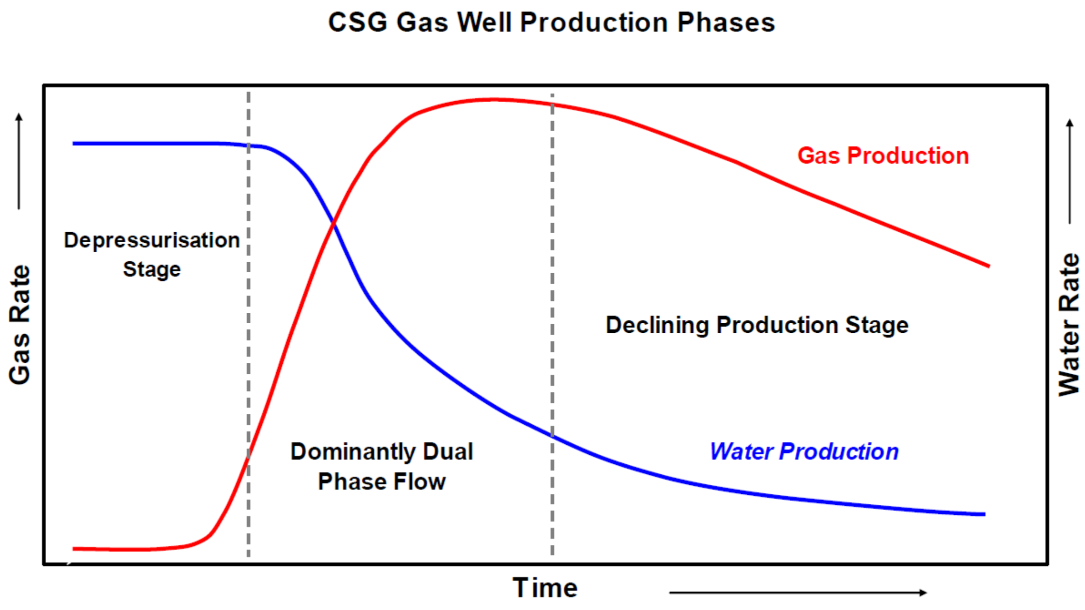

Figure 1 A typical gas and water flow in coal seam gas production (© Copyright, QWC 2012).

It may take about three to five years to lower the hydrostatic pressure to the target level for gas production (QWC 2012). The quality of the co-produced water varies significantly from near potable to highly saline and it can also contain a number of other water quality constituents that may require management attention (Nghiem et al. 2011; SKM 2011;

Shaw 2010). Managing large volumes of co-produced water is an environmental issue for the coal seam gas industry, local communities and regulators (RPS 2011).

Coal seam gas production has been occurring in Queensland and parts of New South Wales from the early 1990s (Geosciences Australia & BREE 2012). Most coal seam gas occurs in the Surat and Bowen Basins, with lesser volumes in the Sydney, Gloucester, Gunnedah and Clarence-Moreton Basins (Figure 2). The co-produced water volume across Australia in 2010 was estimated to be 33 GL/year with 40 per cent from coal seam gas and 60 per cent from conventional gas and oil (RPS 2011). Over the next 25 to 35 years it is estimated that coal seam gas co-produced water production will increase, driven by projected development of coal seam gas sources in Queensland and New South Wales (Table 1 and Figure 2). These estimates are subject to change as more information becomes available (NWC 2012). In the Surat Basin alone, QWC (2012) estimates that water production will be around 125 GL/year over the next three years, reducing to around 95 GL/year for the next 50 years.

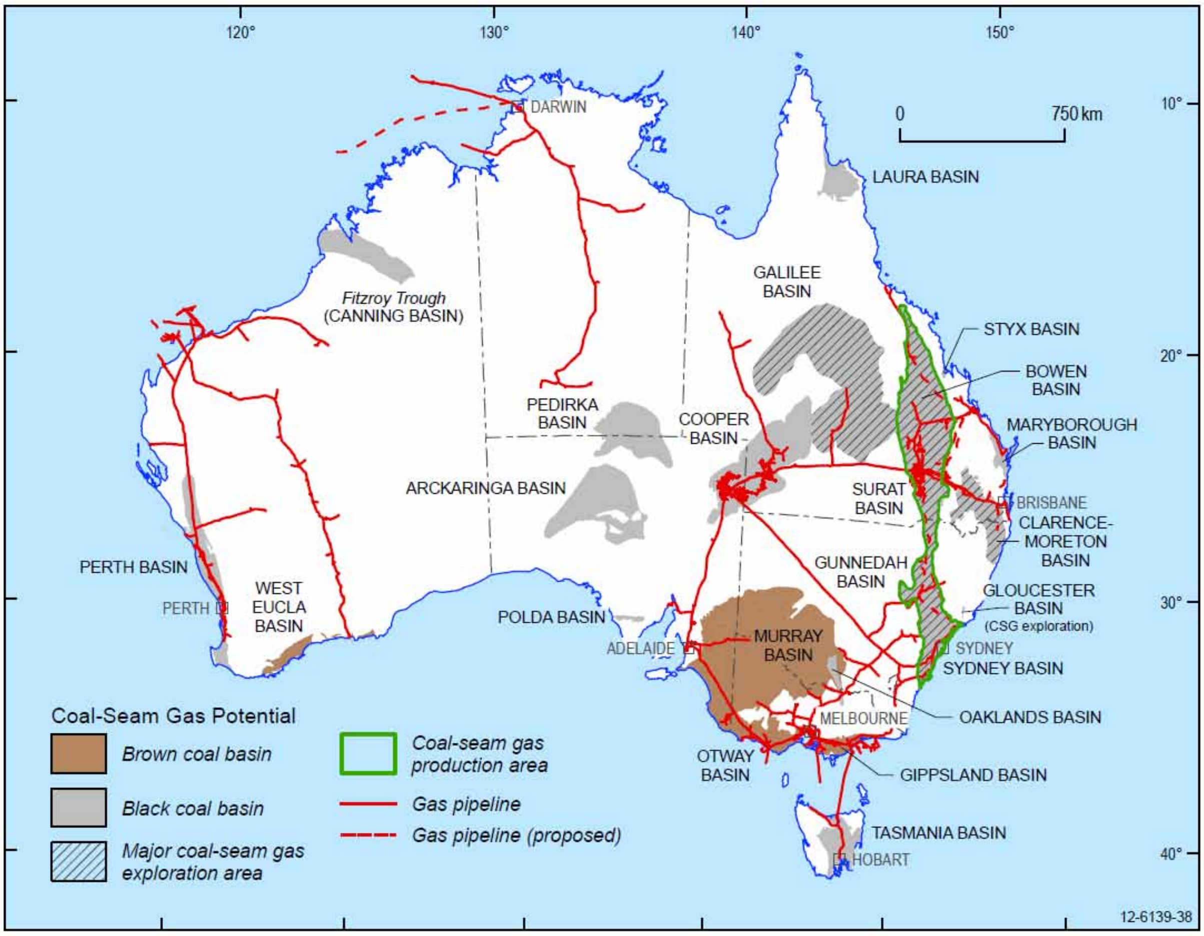

Figure 2 Coal and coal seam gas production and exploration areas in Australia (© Copyright, Geosciences Australia & BREE 2012).

The daily rate of co-produced water production from a typical production area varies according to a range of factors, including geology, the number of wells under production, the production phase and treatment processes. For example, in the Surat Basin around 190 ML of co-produced water is generated per Petajoule (PJ) of gas produced, compared with 1.2 ML/PJ in the Sydney Basin (RPS 2011). AECOM (2010) attribute this to differences in underlying geology. Indeed, the Camden Gas Project in the Sydney Basin reports no discharge of co-produced water to surface waterways in annual independent audits (URS 2009b), whereas 'considerable discharges' of co-produced water have been released to intermittent gullies and streams in the Baffle Creek and Hutton Creek in the upper Dawson River, Queensland since the early 1990s from the Fairview Field in the Bowen Basin (URS 2009a).

Table 1 Estimated potential co-produced water production from coal seam gas over the next 25 years (© Copyright, RPS 2011).

Source	Potential future water production (GL) total over the next 25 years
Bowen Basin (Qld)	2360
Surat Basin (Qld & northern NSW)	5290
Rest of NSW (Gunnedah, Clarence-Moreton, Gloucester & Sydney Basins)	6-47
Total	7697 (average 308 GL/year over 25 years)

Estimated volumes of co-produced water for proposed coal seam gas projects range from 4.5 ML/d for Santos's Pony Hill water treatment plant (URS 2009a), 120 ML/d for Arrow Energy's Surat gas project (Arrow Energy 2012b) and 160 ML/d for Queensland Gas Company's (QGC) Queensland Curtis LNG project (QGC Limited 2009). While daily volumes are small to moderate, the cumulative volume is large, especially if more than one operation is discharging to the same waterway (Moran & Vink 2010).

The quality of water extracted from the coal seam varies depending on its specific characteristics and whether chemicals are used (SKM 2011). Co-produced water typically contains variable but often elevated concentrations of salts as part of the total dissolved solids (TDS). Salinity levels in the Walloon Coal Measures in the Surat Basin range from 250 to 16 000 mg/L with a median concentration of around 1463 mg/L (WorleyParsons 2010). This is generally elevated compared to the overlying and underlying aquifers. A similarly wide range in TDS has been found in other coal seams such as the Bowen Basin (WorleyParsons 2010).

A number of additional water quality variables may also require management attention such as temperature, pH, sodium, chloride, fluoride, boron, some heavy metals, ammonia and phosphorus. Specific constituents and their concentrations can vary widely within a production area and across regions (URS 2009a), so potential impacts can only be assessed on a case-by-case basis. Contaminants may be introduced to the co-produced water as part of the extraction process - for example, residual hydrocarbons from well material or chemicals used in hydraulic fracturing processes (SKM 2012; SKM 2011). The raw water quality and the intended use for the co-produced water determine the type of water treatment required.

2.2 Co-produced water from coal mining operations

Co-produced water from coal mining is derived from direct infiltration of rainfall or floodwaters and/or seepage of groundwater into the excavation, which is commonly referred to by the industry as 'mine affected water'. The water may contain high salinity levels and elevated concentrations of heavy metals, a range of anions and cations and oils and grease (Wright 2012; DEHP 2009b; Singh et al. 1998). Co-produced water from coal mining may be recycled to meet on-site water demands with limited off-site beneficial uses. It is only when the volume of water extracted through mine dewatering activities exceeds the on-site reuse demands that there is the need for planned discharge to local waterways.

Floodwaters have entered mine pits in Queensland (Hart 2008) and in Victoria through the collapse of river diversions during high river flows (ABC News 2012). Flooding has significant implications for mine operations and water management where emergency discharge to surface waters of large volumes of potentially contaminated water (Hart 2008) is required. Under these circumstances, coal mining operations can release far greater volumes of water into waterways than any other mining and therefore, can impact more substantially on water quality in a region (DEHP 2009b).

Cumulative effects can result from concurrent waste discharges, including dewatering from a large number of mines in a catchment and also from the discharge of floodwater from mines after heavy rainfall (DEHP 2009b; Hart 2008). Hence the potential threats to aquatic ecosystems from coal mine water may be greater than from treated coal seam gas co-produced water (DEHP 2009b; Hart 2008).

2.3 Aquatic ecosystems

The aquatic ecosystems that can be influenced by co-produced water include perennial and intermittent rivers and streams, shallow aquifers, wetlands and storages such as lakes, reservoirs, weirs and dams. There are also threats and benefits to downstream aquatic ecosystems, such as estuaries, salt lakes and near-shore marine environments because of their connectivity to the upstream aquatic ecosystems. Where regulations allow, co-produced water may be discharged to surface water systems either routinely or under emergency discharge situations. There may be impacts or benefits to aquatic ecosystems and their associated environmental values, depending on the quality, quantity and timing of the co-produced water discharge and the level of treatment available.

The environmental values associated with aquatic ecosystems are important for a healthy ecosystem or for public benefit, welfare, safety or health, and require protection from the effects of pollution and waste discharges (ANZECC/ARMCANZ 2000). Environmental Values, as defined in NWQMS (1998), are:

- aquatic ecosystems
- primary industries such as irrigation and stock and domestic water, aquaculture and human consumption of aquatic foods
- recreation and aesthetics
- drinking water
- industrial water
- cultural and spiritual values.

Community consultation is an important process to identify the environmental values for particular aquatic ecosystems (FBA 2010). All aquatic ecosystems will have at least one identified environmental value and in many cases will have all six. Each value will have associated water quality and/or water quantity and timing requirements.

ANZECC/ARMCANZ (2000) has water quality trigger levels for aquatic ecosystems and primary industries. Human consumption guidelines are presented in NRMMC/ NHMRC (2011) and NRMMC/NHMRC (2008), which provide water quality requirements for drinking water and recreation and aesthetics respectively. Some Australian states, and particular catchments such as the Dawson River, Queensland, also have specific water quality objectives, often developed using the risk framework specified in the national guidelines (DEHP 2011b; ANZECC/ARMCANZ 2000).

Environmental flow or watering recommendations have been set for many river systems and wetlands throughout Australia, and particularly within the Murray-Darling Basin. These flow and watering recommendations may specify the magnitude, timing, frequency and duration of flows to achieve ecological objectives such as maintaining or improving the health of biological communities and ecosystem functions. Where two or more agreed environmental values are identified for an aquatic ecosystem then the more conservative of the associated water quality and quantity requirements should prevail and become the agreed objectives (ANZECC/ARMCANZ 2000).

2.3.1 Potential threats to environmental values from co-produced water

Water of adequate quality and quantity is central to the health and integrity of the environment (ANZECC/ARMCANZ 2000). The presence or absence of water and its quality, largely determines the species richness and diversity of a particular region (e.g. Hart et al. 1990). It can also be a trigger to breeding and recruitment behaviours for some species (Humphries et al. 1999; Marsh et al. 2012). Changes in the quality or quantity of water may result in immediate change in the structure and function of ecosystems including the numbers and types of organisms that can survive in the altered environment (Boulton & Brock 1999). Furthermore, variations in water quality may change suitability for a range of beneficial uses, such as irrigation.

2.3.2 Potential benefits of co-produced water

Aquatic ecosystems in Australia are frequently under stress from drought and reduced water availability due to other consumptive uses. Climate change may also threaten the surety of water resources in some areas. Provided water quality is fit for purpose, co-produced water, particularly the quantities generated by coal seam gas operations, could be beneficially used to supply a range of possible end uses (RPS 2011). The co-produced water may substitute or supplement natural water resources and, therefore, provide benefits to aquatic ecosystems. Possible beneficial end-uses for co-produced water are described in Table 2.

The supply of co-produced water is not permanent from any coal seam gas or coal mining activity. Coal seam gas reserves are predicted to have a 5 to 25 year life per well (RPS 2011; CWiMI 2008). Areas with multiple wells may only expect production life-cycles of between 25 and 35 years. Co-produced water production also declines over the life of a well. Economic conditions and legislative changes could also influence the water management arrangements and the reliability of the water supply (RPS 2011). Therefore, it is important that any beneficial end-users do not become dependent on the supply of co-produced water.

2.3.3 Management options

Management options for coal seam gas co-produced water include direct beneficial reuse in agriculture, mining and other industry operations, injection into depleted aquifers for recharge purposes and discharge to surface water environments.

Table 2 Possible beneficial uses options for co-produced water that may benefit aquatic ecosystems and their environmental values (© Copyright, RPS 2011).

Environmental Value	Description
Aquatic ecosystems	Co-produced water may be integrated into river flow regimes or wetland watering regimes to improve or maintain ecological values, particularly during droughts or in over-allocated systems.
Irrigation	Co-produced water may be used in preference to natural river water and/or supplement irrigation supplies for agriculture or forestry.
Stock watering	Co-produced water may be used for stock watering, reducing the extractions required from river systems and wetlands.
Drinking water	Potable drinking water supplies may be supplemented with co-produced water, reducing the reliance on river water.
Industrial water	Co-produced water may be preferentially used instead of natural river water to provide for on-site requirements or other industrial uses (e.g. dust control, fire protection, cooling water, plant and vehicle washing, coal washing, irrigation of rehabilitated areas).
Recreation	Co-produced water may supplement (or create new) storage volumes making them suitable for fisheries, recreation and aesthetics. It may also be used to water recreation ovals, community gardens or school grounds.
Aquifer recharge	Aquifers may be recharged using co-produced water.

Although opportunities exist for beneficial use of co-produced water, demand patterns, available volumes, timing of supply and delivery infrastructure constraints mean it is not always feasible (RPS 2011). When co-produced water cannot be reused on-site and without an environmentally acceptable end-user, regulations may allow it to be discharged to surface water systems (Moran & Vink 2010). These aquatic ecosystems include rivers, streams, reservoirs, aquifers and wetlands. Best water management practices require that this water is stored, treated and/or used in a manner that protects the environment from harm and maximises the opportunities for beneficial uses (DEHP 2010a).

In Queensland (DEHP 2012), the preferred hierarchy of disposal of co-produced water in order of priority is:

- Priority 1 – coal seam gas water is used for a purpose that is beneficial to one or more of the following: the environment, existing or new water users and existing or new water-dependent industries.
- Priority 2 – after feasible beneficial use options have been considered, treating and disposing coal seam gas water in a way that firstly avoids and then minimises and mitigates impacts on environmental values.

A portion of co-produced water may be used to meet beneficial use demand at some times of the year, while at other times disposal to waterways may occur. Any option for disposal to a waterway requires an assessment of risks and appropriate treatment to meet receiving water quality and environmental flow regime requirements.

3 Water quality impacts on receiving environments

3.1 Importance of water quality for aquatic ecosystems and associated environmental values

Water quality refers to the physical, chemical and biological attributes of water that affect its ability to sustain environmental values or beneficial uses (ANZECC/ARMCANZ 2000). Good water quality is not only important to support healthy ecological communities, it is equally important for human water users. The availability of adequate supplies of clean water is one of the most important building blocks for economic and social structures of society. It determines the viability of a region to support industries such as agriculture, fishing, irrigation, manufacturing and mining. Changes in the quality or quantity of water may result in immediate change in the structure and function of ecosystems including the numbers and types of organisms that can survive in the altered environment (ANZECC/ARMCANZ 2000).

Guidelines for acceptable water quality to meet various beneficial uses and protect agreed values have been developed at national, state and regional levels. The national guidelines like NRMMC/NHMRC (2011), NRMMC/NHMRC (2008) and ANZECC/ARMCANZ (2000) are not mandatory and application is a state or territory responsibility. Some states have developed their own state-based (e.g. DEHP 2009a; Victorian EPA 2003) or regional guidelines (DEHP 2011b; Rogers et al. 2011), often using the process described in ANZECC/ARMCANZ (2000) for deriving regional trigger levels, or defaulting to ANZECC guidelines for some stressors and/or toxicants (e.g. in New South Wales).

Most guidelines are targeted at permanent rivers and lakes (e.g. ANZECC/ARMCANZ 2000) and may not be appropriate for intermittent streams and wetlands. These systems experience highly variable water quality (see Boulton & Brock 1999) and specific guidelines have generally not been developed due to a lack of reference condition data (e.g. DEHP 2009a). The lack of water quality guidelines for intermittent systems is a critical knowledge gap in the context of managing co-produced water and discharges. The Queensland Government Healthy HeadWaters program partly addresses this issue in the Queensland Murray-Darling Basin through the development of regional salinity guidelines for disposal of co-produced water (Rogers et al. 2011). Furthermore, guidelines are not available for all potential stressors and toxicants, often due to a lack of data to enable an assessment of toxicity (ANZECC/ARMCANZ 2000).

3.2 Water quality threats from co-produced water

Impacts on aquatic ecosystems and other values can arise through direct toxicity associated with one, or a number of, primary stressors and from complex mixtures of various toxicants (Takahashi et al. 2011a). For these reasons, characterising impacts to environmental values can be difficult.

The following sections discuss the primary stressors and toxicants in co-produced water and their environmental impacts, and discuss the issue of mixtures of multiple toxicants and importance of toxicity testing to confirm potential impacts.

3.2.1 Primary stressors and toxicants

In a major review conducted for the Healthy HeadWaters program, Shaw (2010) documented water quality in untreated coal seam gas co-produced water using data provided by the Queensland coal seam gas industry and compared this with relevant Australian guidelines and data from overseas. In other work, SKM (2011) reviewed data from the New South Wales coal seam gas industry, and DEHP (2009b) provides water quality data collected for a number of large coal mines in the Fitzroy catchment. The quality of untreated co-produced water is highly variable, depending on the source of the water in the coal seam, the depth of the coal seam and the surrounding geology.

Table **3** summarises the typical water quality variables and their observed ranges in untreated co-produced water in Australia (SKM 2011).

Table 3 Typical coal seam gas co-produced water quality for data collected from across Australia and typical guideline values ((© Copyright, SKM 2011; ANZECC/ARMCANZ 2000).

Water quality variable	Unit	Min	Max	Guideline trigger value range	Guideline description from ANZECC 2000
TDS	mg/L	200	10000	1000	Recreation
SAR	mg/L	16	567	2-102	Primary industries (irrigation)
Temperature	C	22	32	20^{th}-80^{th} percentile	Aquatic ecosystems
pH	pH	7	9.1	6.5-9.0^{+}	Aquatic ecosystems
EC	µS/cm	200	16000	30-5000^{+}	Aquatic ecosystems
SS	mg/L	9	2669	<40	Primary industries (aquaculture)
Colour (Apparent)	PCU	125	340		No guideline recommended
Colour (True)	PCU	5	14.5		No guideline recommended
UV Transmission @ 254nm	%	99.7	99.98		No guideline recommended
Turbidity	NTU	230	935	0.5-200^{+}	Aquatic ecosystems
Total Hardness as $CaCO_3$	mg/L	39	185	500	Recreation
Hydroxide Alkalinity as $CaCO_3$	mg/L	0	1		No guideline recommended
Carbonate Alkalinity as $CaCO_3$	mg/L	36.5	600		No guideline recommended
Bicarbonate Alkalinity as $CaCO_3$	mg/L	580	8200		No guideline recommended
Total Alkalinity as $CaCO_3$	mg/L	899.5	1460		No guideline recommended
Sodium	mg/L	35	4500	3000	Recreation

Water quality variable	**Unit**	**Min**	**Max**	**Guideline trigger value range**	**Guideline description from ANZECC 2000**
Calcium	mg/L	0.5	49	1000	Primary industries (stock watering)
Magnesium	mg/L	0.7	16	2000	Primary industries (stock watering)
Iron	mg/L	1	25	0.2-10	Primary industries (irrigation)
Barium	mg/L	1	10	1	Recreation
Chloride	mg/L	150	2500	400	Recreation
Sulphate	mg/L	1	10	400	Recreation
Silicon	mg/L	7	20		No guideline recommended
Potassium	mg/L	1	300		No guideline recommended
Boron	mg/L	0.05	3.1	0.37	Aquatic ecosystems
Aluminium	mg/L	0.01	0.3	0.055[#]	Aquatic ecosystems
Arsenic	mg/L	0.001	0.0065	0.013	Aquatic ecosystems
Beryllium	mg/L	0.001	0.001	0.1-0.5	Primary industries (irrigation)
Cadmium	mg/L	0.0001	0.0002	0.0002	Aquatic ecosystems
Chromium	mg/L	0.005	0.3	0.001	Aquatic ecosystems
Copper	mg/L	0.001	0.2	0.0014	Aquatic ecosystems
Lead	mg/L	0.001	0.2	0.0034	Aquatic ecosystems
Manganese	mg/L	0.004	0.3	1.9	Aquatic ecosystems
Nickel	mg/L	0.0001	0.003	0.011	Aquatic ecosystems
Selenium	mg/L	0.001	0.01	0.011	Aquatic ecosystems
Zinc	mg/L	0.005	0.15	0.008	Aquatic ecosystems
Bromine	mg/L	1	12		No guideline recommended
Mercury	mg/L	0.0001	0.001	0.0006	Aquatic ecosystems
Silica	mg/L	15.6	20		No guideline recommended
Fluoride	mg/L	0.4	5.9	1-2	Primary industries (irrigation)
Nitrite and Nitrate as N	mg/L	0.01	0.01	0.005-0.2	Aquatic ecosystems
Sulphide	mg/L	0.1	0.1	0.05	Recreation
TOC	µg/L	2000	3900		No guideline recommended
C6-C9 Fraction	µg/L	20	20		No guideline recommended

Water quality variable	Unit	Min	Max	Guideline trigger value range	Guideline description from ANZECC 2000
C10-C14 Fraction	µg/L	50	50		No guideline recommended
C15-C28 Fraction	µg/L	100	100		No guideline recommended
C29-C36 Fraction	µg/L	50	113		No guideline recommended
1,2-Dichloroethane-D4	µg/L	118	120	ID	Aquatic ecosystems
Toluene-D8	µg/L	94.6	98.22	ID	Aquatic ecosystems
4-Bromofluorobenzene	µg/L	99.2	102.9		No guideline recommended

+ - specific guideline depends on geography (southeast Australia, tropical Australia, southwest Australia, south central Australia), receiving environment (upland river, lowland river, freshwater lakes and reservoirs, wetlands) or beneficial use.

ID - insufficient data to determine guidelines.

- dependent on pH.

Untreated co-produced water from coal seam gas operations is typically moderately to highly saline, dominated by bicarbonate content and therefore a high pH, and has a high Sodium Adsorption Ratio (SAR) (Nghiem et al. 2011; SKM 2011; Shaw 2010). Co-produced water can also have a moderately high initial temperature, upward of 30^{o}C and sometimes as high as 40^{o}C (URS 2009a), and elevated concentrations of lead, copper, zinc and aluminium (Nghiem et al. 2011; URS 2009a). Concentrations of nutrients, comprising ammonia and phosphorous, may also be elevated (URS 2009a). Co-produced water from large coal mines can also be moderately to highly saline and exhibit a range of elevated ions and low pH, depending on the characteristics of the underlying geology and salinity of groundwater in any aquifers that are intersected by the mine voids (Wright 2012; DEHP 2009b; NSW Department of Planning 2005; Singh et al. 1998).

The total dissolved solids (TDS) of untreated coal seam gas co-produced water has been recorded as high as 170 000 mg/L in the Western US. However, in Australia salinities are more typically in the range 1000 to 6000 mg/L, which would classify it as brackish water (Nghiem et al. 2011). Many aquatic organisms are sensitive to changes in salinity, both directly, by impacts on osmoregulation, and indirectly by modifications to the structure and composition of the surrounding ecosystem (ANZECC/ARMCANZ 2000). Both highly elevated and very low TDS can have negative impacts on biota (Takahashi et al. 2011a; Griffith & Biddulph 2010).

In New South Wales, the SAR in untreated co-produced water ranged from around 15 to greater than 500 (SKM 2011). When used for irrigation, water with a high SAR can result in replacement of the calcium and magnesium in the soil by sodium in the water, which causes dispersion of soil particles, loss of the ability of the soil to form stable aggregates and a reduction in infiltration and permeability with negative consequences for crop production (Shalvet 1994). ANZECC/ARMCANZ (2000) provides advice on SAR values for irrigation. Extremely sensitive crops such as avocado, fruits and nuts can tolerate SAR values in the range 2-8, sensitive crops such as legumes can tolerate SAR values up to 18 and more tolerant crops such as wheat, cotton and barley can tolerate SAR values up to around 100. The high SAR in co-produced water means treatment is needed if water is to be made suitable for irrigation.

Most surface waters have a pH of between 6.5 and 8.0 (ANZECC/ARMCANZ 2000). The pH of untreated co-produced water is variable in Australia, although for coal seam gas it is generally alkaline with a pH greater than 7 (SKM 2011; Shaw 2010). This is in contrast to the US, where co-produced water is generally more acidic (Fakhru'l-Razi et al. 2009). High pH can disrupt the physiological functioning of biota with impacts to enzyme actions and membrane permeability (ANZECC/ARMCANZ 2000). Furthermore, changes in pH can alter the toxicity of certain contaminants - for example, an increase in pH and temperature increases the toxicity of ammonia by increasing the proportion of toxic unionised ammonia (ANZECC/ARMCANZ 2000; USEPA 1999). Low pH can also alter the toxicity of certain contaminants, particularly through an increase in the solubility, and hence bioavailability, of some heavy metals (ANZECC/ARMCANZ 2000).

Co-produced water may contain elevated concentrations of trace elements and heavy metals, including (but not limited to) boron, cadmium, lead, copper, manganese, zinc and aluminium (Nghiem et al. 2011; SKM 2011; Shaw 2010). These elements can have a range of impacts on beneficial uses as outlined in detail in ANZECC/ARMCANZ (2000). Heavy metals such as lead, copper and zinc can be toxic to aquatic fauna at high concentrations and bio-accumulate in the food chain. Boron, for example, can be elevated in co-produced water and is poorly removed during standard treatment processes. It is an essential nutrient for plants but there is a small range between deficiency and toxicity (Parkes & Edwards 2005). Boron plays a role in carbohydrate metabolism, pollen germination and normal growth and functioning and is considered to be an essential trace element for plants. However, certain plants such as citrus fruit, stone fruit and some nut trees are sensitive to the toxic effects of boron if irrigated with water with concentrations higher than about 0.5 mg/L (Lazarova et al. 2005).

A range of organic compounds (including hydrocarbons) and radionuclides may also be present in co-produced water (Volk et al. 2011; Shaw 2010). Co-produced water can contain naturally occurring Polycyclic Aromatic Hydrocarbons (PAHs), Total Petroleum Hydrocarbons (TPHs), oxygen-bearing aromatic compounds such as phenols, aldehydes, ketones and various carboxyl-, hydroxyl- and methoxy- bearing compounds, monoaromatic hydrocarbons such as benzene, toluene, ethylbenzene and xylenes (BTEX) and various radioisotopes of uranium, radon, thallium and potassium. These compounds are generally only detected in very low concentrations, if at all, and although they may be naturally occurring, they can still pose risks to environmental and human health in high concentrations (Lloyd-Smith & Senjen 2011; Shaw 2010; ANZECC/ARMCANZ 2000).

Co-produced water has initial elevated temperatures compared to ambient temperatures in most Australian rivers (Nghiem et al. 2011). When temperature is outside the ‘normal’ range it can influence the physiology of biota, alter ecosystem functioning and exacerbate susceptibility to chemical stress (ANZECC/ARMCANZ 2000). For example some toxicants such as heavy metals, ammonia, pesticides and PAHs are more toxic at elevated temperatures than low temperatures (ANZECC/ARMCANZ 2000).

Untreated co-produced water can also contain a range contaminants associated with bore construction and operation, such as greases and oil. Furthermore, in approximately 10 per cent of bores, hydraulic fracturing fluids are also injected to aid in gas extraction (SKM 2011). Hydraulic fracturing fluid contains around 99 per cent water and sand (or other proppant to hold the coal seams open), but also a number of chemicals that aid the fracturing process, such as surfactants, clay breakers, biocides and viscosifiers. While the precise chemical composition of hydraulic fracturing fluids is generally a trade secret (Lloyd-Smith & Senjen 2011), there is increasing disclosure of the chemicals used (SKM 2012). Table 4 lists some of the chemicals which may be used in hydraulic fracturing (APPEA 2012; Lloyd-Smith & Senjen 2011). The addition of toxic BTEX to fracturing fluids has been banned in Queensland, New South Wales and Victoria (SKM 2012).

Table 4 Chemicals which may be used in the hydraulic fracturing process (© Copyright, Lloyd-Smith & Senjen 2011; Australian Petroleum Production and Exploration Association (www.appea.com.au)).

Chemical name	Purpose	Percentage of fracturing fluid
Water	Proppant suspension	98.5-99.6%
Nitrogen	Proppant suspension	
Crystalline silica (sand and quartz)	Proppant	
Glycerine	Additive	<1.5% in total
Methyl-isothiazol	Eliminates bacteria in the water that produces corrosive by-products	
Hydrochloric acid	Helps dissolve minerals and initiate cracks in rock	
Glutaraldehyde	Eliminates bacteria in the water that produces corrosive by-products	
Quaternary Ammonium Chloride	Eliminates bacteria in the water that produces corrosive by-products	
Phosphonium Sulfate	Eliminates bacteria in the water that produces corrosive by-products	
Ammonium Persulfate	Allows a delayed break down of the gel	
Sodium Chloride	Product stabiliser	
Magnesium Peroxide	Allows a delayed break down of the gel	
Magnesium Oxide	Product stabiliser	
Calcium Chloride	Prevents clays from swelling or shifting	
Choline Chloride	Prevents clays from swelling or shifting	
Tetramethyl ammonium chloride	Prevents clays from swelling or shifting	
Sodium Chloride	Prevents clays from swelling or shifting	
Isopropanol	Product stabilizer and/or wintering agent	
Methanol	Product stabilizer and/or wintering agent	
Formic Acid	Prevents corrosion of the pipe	
Acetaldehyde	Prevents corrosion of the pipe	
Petroleum Distillate	Carrier fluid for borate or zicornate crosslinker	
Hydrotreated Light Petroleum Distllate	Carrier fluid for borate or zicornate crosslinker	
Potassium Metaborate	Maintains fluid viscosity as temperature increases	

Chemical name	Purpose	Percentage of fracturing fluid
Triethanolamie Zirconate	Maintains fluid viscosity as temperature increases	
Sodium Tetraborate	Maintains fluid viscosity as temperature increases	
Boric Acid	Maintains fluid viscosity as temperature increases	
Zirconium Complex	Maintains fluid viscosity as temperature increases	
Borate Salts	Maintains fluid viscosity as temperature increases	
Ethylene Glycol	Product stabilizer and/or wintering agent	
Methanol	Product stabilizer and/or wintering agent	
Polyacrylamide	'Slicks' the water to minimise friction	
Petroleum Distillate	Carrier fluid for polyacrylamide friction reducer	
Hydrotreated Light petroleum distillate	Carrier fluid for polyacrylamide friction reducer	
Methanol	Product stabilizer and/or wintering agent	
Ethylene Glycol	Product stabilizer and/or wintering agent	
Guar Gum	Thickens the water in order to suspend the sand	
Petroleum Distillate	Carrier fluid for guar gum in liquid cells	
Hydrotreated Light petroleum distillate	Carrier fluid for guar gum in liquid cells	
Methanol	Product stabilizer and/or wintering agent	
Polysaccharide blend	Thickens the water in order to suspend the sand	
Ethylene Glycol	Product stabilizer and/or wintering agent	
Citric Acid	Prevents precipitation of metal oxides	
Acetic Acid	Prevents precipitation of metal oxides	
Thioglycolic Acid	Prevents precipitation of metal oxides	
Sodium Erythorbate	Prevents precipitation of metal oxides	
Lauryl Sulfate	Prevents formation of emulsions in fracture fluid	
Isopropanol	Product stabilizer and/or wintering agent	
Ethylene Glycol	Product stabilizer and/or wintering agent	
Sodium Hydroxide	Adjusts the pH of fluid	
Potassium Hydroxide	Adjusts the pH of fluid	
Acetic Acid	Adjusts the pH of fluid	
Sodium Carbonate	Adjusts the pH of fluid	
Potassium Carbonate	Adjusts the pH of fluid	
Copolymer of Acrylamide, Sodium Acrylate	Prevents scale deposits in the pipe	
Sodium Polycarboxylate	Prevents scale deposits in the pipe	

Chemical name	Purpose	Percentage of fracturing fluid
Phosphonic Acid Salt	Prevents scale deposits in the pipe	
Lauryl Salt	Used to increase the viscosity of the fracture fluid	
Ethanol	Product stabilizer and/or wintering agent	
Napthalene	Carrier fluid for the active surfactant ingredients	
Methanol	Product stabilizer and/or wintering agent	
Isopropyl Alcohol	Product stabilizer and/or wintering agent	
2-Butoxyethanol	Product stabilizer	

Lloyd-Smith and Senjen (2011) found that even though they might be in low concentrations, the effect of the complex mixture of chemicals on the environment was not well understood and there were no water quality guidelines for many of the compounds. Many of the chemical compounds have demonstrated human health effects - for example, skin exposure to sodium persulfate can lead to sensitisation, ethylene glycol is a respiratory toxicant, naphthalene is a potential human carcinogen and isopropanol is a reproductive toxicant (Lloyd-Smith & Senjen 2011).

3.2.2 Combined effect of chemical mixtures on the environment

Water quality assessments of co-produced coal seam gas water planned for discharge to surface water environments need to follow the appropriate state or territory guidelines and/or regulations, and when these are not available, the National Guidelines (ANZECC/ARMCANZ 2000). However, in some cases there may be interactions between constituents that can result in toxicity greater than the individual toxicities even when the individual components are below guideline levels (Takahashi et al. 2011a). There can be additive effects or in some rare cases a synergistic effect may take place where the presence of one chemical increases the biological activity of others.

ANZECC/ARMCANZ (2000) describes the steps required to test for such interactions through direct toxicity assessment. Toxicity assessment is necessary because these interactions cannot be predicted. The biological response can be measured irrespective of the composition of the water by exposing a range of biota to test water. If no toxicity is found in biological testing then low risk to the receiving environment is indicated, provided that the appropriate number of tests is conducted and that tests include sensitive species.

Takahashi et al. (2011a) completed a direct toxicity assessment comparison of untreated and reverse osmosis treated coal seam gas co-produced water. Macroinvertebrates, fish, microalgae and macrophytes were tested and found sensitive to untreated co-produced water. Only macroinvertebrates were sensitive to reverse osmosis water due to low electrical conductivity. They concluded that the guidelines established for individual contaminants of concern are likely to provide appropriate protection of aquatic ecosystems. However, the quality of treated water varies and it is difficult to provide industry-scale characterisation of toxicity. Hence site-specific bioassays using treated water are recommended that take into account the specific values and conditions of the receiving waterway (Takahashi et al., 2011a).

3.3 Water treatment

3.3.1 Treatment options

There are a number of techniques available for the treatment of co-produced water from coal mine and coal seam gas activities. Water treatment techniques can be categorised into three groups:

- Those that reduce broad ranges of contaminants. These treatment processes include membrane desalination reverse osmosis (RO), electrodialysis (ED), electrodialysis reversal (EDR) and thermal desalination.
- Those that target a specific group of contaminants, including Granular Activated Carbon (GAC), ion exchange, wetland and advanced oxidation. These may supplement any removal deficiencies of techniques in Group 1.
- Those that support Group 1 techniques to improve operational performance of the overall system, including clarification, filtration, ion exchange and chemical addition.

Selection of an appropriate treatment process is heavily dependent on the characteristics of the co-produced water, the primary contaminants present and the water quality requirement for its end use (Table 5).

Table 5 Available water treatment technologies.

Process Technology	Primary Contaminants Removed				
	Suspended Solids or Turbidity	Dissolved Solids (Salts)	Heavy Metals	Organic Contaminants	SAR (Sodium Adsorption Ratio)
Membrane Desalination (Reverse Osmosis – RO)	No	Yes	Yes[1]	Yes[2]	Yes[3]
Electrodialysis & Electrodialysis Reversal	No	Yes	Yes	No	Yes[3]
Thermal Desalination	Yes	Yes	Yes	Yes	Yes[3]
Ion Exchange	No	Limited	Yes	No	No
Advanced Oxidation	No	No	No	Yes	No
Granular Activated Carbon	No	No	Yes	Yes	No
Sedimentation	Yes	No	No	Limited[4]	No
Filtration	Yes	No	No	Limited[4]	No
Wetland	Limited	No	Yes	Yes	No

[1] Removal of metals by RO depends on the chemical form of the metal.

[2] Some organics (e.g. methanol, ethanol, phenols, and ethylene glycol) may be poorly removed through RO depending on pH, temperature, and operating pressure. Some smaller organic contaminants may also pass through RO membranes.

[3] Chemical additions required to adjust SAR.

[4] Where chemical is added.

In most cases a combination of techniques will be needed to meet the water quality requirements for a variety of beneficial reuse or environmental discharge applications. Table 6 summarises the treatment options available and their applicability for co-produced water treatment. In general, the primary water quality variable that requires removal is salt, hence treatments such as reverse osmosis are most suitable (Nghiem et al. 2011). However, most treatment methods do not necessarily address all water quality issues due to an inability to reduce all constituents of concern (Takahashi et al. 2011a). Treatment may introduce new risks, by making the water too clean, such as having low turbidity (which could potentially allow greater light penetration into the receiving water column) and low mineralisation, or through the production of a waste stream such as brine and sludge or the formation of treatment by-products.

Table 6 Summary of treatment options available for coal seam gas co-produced water (© Copyright, SKM 2011).

Treatment option	**Description**	**Contaminant removal**	**Suitability/Application**	**Limitations**	**Additional process requirements**	**Waste**
Membrane Desalination (RO)	Water subject to high pressure through a semi-permeable membrane for separation of contaminants	Broad constituents including salt, heavy metals and trace organic matters	Well-suited for broad ranges of dissolved contaminants, including TDS, heavy metals and organics	Stringent pre-treatment required. High energy demand. May not remove all contaminants to minimum levels. May make water too clean for receiving environment.	Multi-media or membrane filtration. Post-treatment, including chemical adjustment (for SAR). Ion exchange for improving recovery	High concentration brine
Electrodialysis (ED) and Electrodialysis Reversal (EDR)	An electrochemical process in which ions migrate through ion-selective semi-permeable membranes as a result of their attraction to two electrically-charged electrodes	Broad constituents, including salt and heavy metals	Well-suited for removal of TDS, mainly for charged contaminants, including salts and heavy metals	No removal of uncharged forms of contaminants. Not economical for feed TDS > 4000 mg/L.	Filtration – for SS removal. GAC/Advanced oxidation/wetland for the removal of uncharged forms of contaminants	High concentration brine
Thermal Desalination	Water subjected to a phase change for purification	Broad constituents, including salt, heavy metals and trace organic matters	Well-suited for broad ranges of dissolved contaminants, including TDS, heavy metals and organics	High energy demand unless heat source is already available.	Post-treatment, including chemical adjustment	High concentration brine
Ion Exchange	Charged resin replaces conductive salts with replacement ions (e.g. H^+ and OH^-)	Targeted contaminant removal	Not practical for gross TDS removal. May be suited for reduction of targeted contaminants and/or pre-treatment for the desalination processes	No reduction in salinity. Irreversible organic matter adsorption.	None	Regeneration fluid with high salt content

Treatment option	Description	Contaminant removal	Suitability/Application	Limitations	Additional process requirements	Waste
Advanced Oxidation	Generation and use of hydroxyl radical for destruction of trace organic contaminants and micro-organisms	Microorganisms and trace organics	Well-suited for post-treatment to target specific trace organic contaminants as necessary	No reduction in salinity. Effectiveness highly influenced by contaminant levels.		None
Granular Activated Carbon	Adsorption of dissolved organics onto activated carbon	Trace organics and heavy metals	Well-suited for post-treatment as necessary	No reduction in salinity. Blockage of media by solids. Ineffective at high organic concentrations.	Filtration – reduction of suspended solids required	Spent GAC
Clarification	Removal of suspended solids via gravity/floatation	Suspended solids and algae	Well-suited for pre-treatment for subsequent filtration process and provides ad hoc removal of excessive suspended solids	No reduction in salinity		Captured solids and coagulant residuals
Filtration	Suspended solids removed by filtering media or membrane	Suspended solids	Well-suited as a pre-treatment to remove suspended solids for desalination	No reduction in salinity	Screening required for gross solids	Waste backwash water with high suspended solids content
Wetland	Biological removal and plant uptake of nutrients and minerals. Sorption by the sediments in the wetland	Nutrients, organic matters and heavy metals	Well-suited for product polishing to remove trace elements/nutrients prior to discharge	Unfeasible for salinity reduction. Large footprint.	Up-front primary treatment	Vegetation clippings/ periodical removal of sediments

3.3.2 Matching co-produced water to receiving environment quality

Leading practice for the discharge of waste water from mines requires discharge to comply with state and territory specified water quality and quantity condition (DRET 2008). These are often based on generic guidelines, which may not consider the actual water quality in receiving waterways (Hamstead & Fermio 2012). Consequently, it can be difficult to benchmark stressors and toxicants in receiving waterways and match waste water discharges to the quality of the specific receiving water environment.

In a review of water management in the mining sector, Hamstead and Fermio (2012) identified two different approaches to regulation of mine water discharge to waterways:

- Discharge constrained by state-wide standard water quality limits that apply to either the water discharged as end-of-pipe limits or in the receiving water downstream of a specified mixing zone.
- Discharge constrained by the total load of the parameter being managed with the relevant water system, often referred to as load licensing systems.

A concern with end-of-pipe limits is that they are usually designed to cover all scenarios across an entire jurisdiction and may not reflect or respond to local natural water quality conditions (Hamstead & Fermio 2012). Furthermore, in a review of discharge licence limits for coal mines in Queensland's Fitzroy River Basin there were inconsistencies in the application of guideline limits and limits for some constituents of environmental concern were not adequately addressed in many licences (DEHP 2009b). For example, of the 39 mine permits in the catchment, the review found that 34 had an electrical conductivity (EC) limit set for the receiving environment and two were not specified. Of the 34 with a receiving water limit, 18 also had an EC limit specified for the discharge itself and four only had a limit set for the discharge and not the receiving waterway. Some of the limits referenced background condition and others did not and covered a wide range from 500 to 4500 µS/cm (DEHP 2009b). With respect to discharge quantity, there was also inconsistency in licence conditions with some mines required to discharge only at certain times of the year, or during periods when flow in the receiving waterway exceeds a specified threshold. Other mines appear to have no limit on the quantity or timing of discharge (DEHP 2009b).

Efforts are now underway to improve this situation by using an integrated water quality and hydrology model for the Fitzroy River Basin that would enable more realistic limits to be set that take into account the varied quality of receiving waters and the timing of river flows, dilution potential and load transport capacity (Hamstead & Fermio 2012). Similar models are likely to be needed in other catchments where there is significant potential for cumulative impacts from a large number of potential discharges.

Discharge licences for coal mines in New South Wales typically include limits for oil and grease, biological oxygen demand, non-filterable residue, pH, suspended solids and some heavy metals (Singh et al. 1998). However, they don't always include limits for salinity or ionic concentration (Wright 2012). In some situations in the Blue Mountains region, elevated salinity and high ionic concentrations in water discharged from coal mines impact on river health by increasing salinity downstream of discharge points, yet these constituents are not necessarily considered in the treatment process and may not be part of any discharge monitoring program (Wright 2012).

An alternative to the end-of-pipe approach has been developed in the Hunter River Salinity Trading Scheme. This uses the diluting capacity of higher flows to enable discharge of poorer quality excess mine water, within set limits (Hamstead & Fermio 2012). Under the scheme, discharge rights can be traded between mine operators, providing them with the flexibility to increase or decrease their discharge in response to changing circumstances but all within a framework that caps the combined salt load to the river (Hamstead & Fermio 2012). However, such schemes do not necessarily mean the quality of discharge is matched to the quality of the specific receiving environment. A whole of catchment cap on load is set but the specific concentrations and loads discharged to any particular waterway may vary widely from the background quality at that specific location.

There is also inconsistency in setting limits for other beneficial uses. A review of waste water discharge licence conditions and state and national guidelines for reclaimed water reuse for a variety of different beneficial uses such as irrigation, industrial water and potable water supply noted that many types of water quality criteria are similar for discharge and reuse (Higgins et al. undated). However, electrical conductivity and turbidity are often mentioned in reuse guidelines but are not included in discharge licences. When they are included, the recommended levels often differ substantially. More effort should be placed on aligning specific discharge conditions with criteria for the relevant beneficial use.

There is currently a relatively small volume of coal seam gas co-produced water being released to waterways. However, the growth of the industry means there is potential for much larger volumes to be released, depending on the ability to identify beneficial uses. In early stages, there was little attempt to match the quality or the quantity of coal seam gas co-produced water discharge to waterways (for example, from the Fairview operation in central Queensland) to the specific requirements of the receiving waterway (URS 2009a). In this location, monitoring has shown there is a significant increase in salinity and some ions downstream of co-produced water discharge points. There is also some evidence that this has impacted the health of downstream macroinvertebrates (URS 2009a). However, current operations in this area include microfiltration and RO treatment with post-treatment mineralisation to match the receiving waters.

RO is very effective at removing salt and most other contaminants from water. However, post-treatment conditioning is often needed to ensure the treated water is suitable for the intended beneficial use - for example, the preferential removal of some ions means SAR remains elevated and the treated water may be unsuitable for some irrigation beneficial uses. Boron may also remain elevated. Under these circumstances magnesium and calcium may need to be added to make the water suitable for use in irrigation (Nghiem et al. 2011). A very low ionic concentration may have implications for a range of aquatic biota if the treated water is discharged to a waterway in a large volume (Rogers et al. 2011). Low ionic concentration can impact the physiology of some fish and macroinvertebrates through changes in the ability of these organisms to osmoregulate, and this was confirmed in toxicity testing of RO permeate by Takahashi et al. (2011a). Lower limits on salinity and ionic concentration need to be established for receiving waterways to avoid risks to aquatic ecosystems from the treated water and post-treatment conditioning, which is undertaken to ensure that water quality matches criteria to protect aquatic ecosystems (Rogers et al. 2011; Takahashi et al. 2011b).

In addition to low ionic concentration, the treated water is very low in turbidity. Most streams in Australia have a moderate to high level of natural turbidity and hence low light penetration (Takahashi et al. 2011b; Oliver 1990; Kirk 1979). The low light penetration is considered a factor in controlling excessive algal growth in inland waterways, especially given that nutrients are often elevated (Oliver et al. 1999; Donnelly et al. 1997). The discharge of large volumes of very clear water to turbid waterways can increase light penetration and promote excessive macrophyte or algal growth. This is a particular issue associated with the release of treated co-produced water to turbid waterways (Figure 3).

For each treatment process there will also be a number of by-product waste streams, such as brine from EDR, RO and thermal desalination (see Table 6). Often there are only limited options available for further treatment and disposal of waste streams, particularly brine in inland areas.

3.4 Monitoring requirements

Monitoring the discharge of treated waste water to waterways is often limited to the constituents specified in the discharge licence and at limited sites (Higgins et al. undated). It is important that monitoring of both discharge and the receiving environment be conducted and the data used to support adaptive management. Monitoring programs need to consider the identified stressors from co-produced water and an understanding of the likely ecological response (Takahashi et al. 2011b). Reference needs to be made to the specific characteristics of the discharge, such as whether it is treated or untreated, and needs to be compared with the water quality in the receiving environment or relevant criteria for the intended beneficial use. Conceptual models of ecosystem response should be developed and hypotheses proposed for testing by the monitoring program. Any monitoring program needs to be informed by careful design that outlines appropriate monitoring objectives and sites with a temporal scale that enables response to be distinguished from background variability.

3.5 Knowledge gaps and recommendations

The quality of co-produced water is understood at a general level and there is a good understanding of the range of water quality variables that are likely to be of issue. However, the actual quality varies widely between wells and various mine operations across regions and there are limited data to enable regional-scale characterisation or to permit a typology of waters (SKM 2011). There is also no single repository for data or a simple way of identifying and accessing what data are available. These issues could be addressed by more comprehensive data gathering and interpretation and collating existing and new data into a form that makes its identification and access easier.

The general impacts on environmental values and beneficial uses of various stressors and toxicants are well documented and guidelines have been developed at the national level that cover a wide range of potential stressors and toxicants (ANZECC/ARMCANZ 2000). Guidelines are also available at the state level and for some regions (for example, DEHP 2011b). Most guidelines have been developed for perennial streams and may not be applicable for ephemeral waterways, which is the dominant waterway type in many areas where coal seam gas and large coal mine production occurs. Furthermore, there are several stressors and toxicants for which there are no guidelines, or where data are insufficient to determine a guideline.

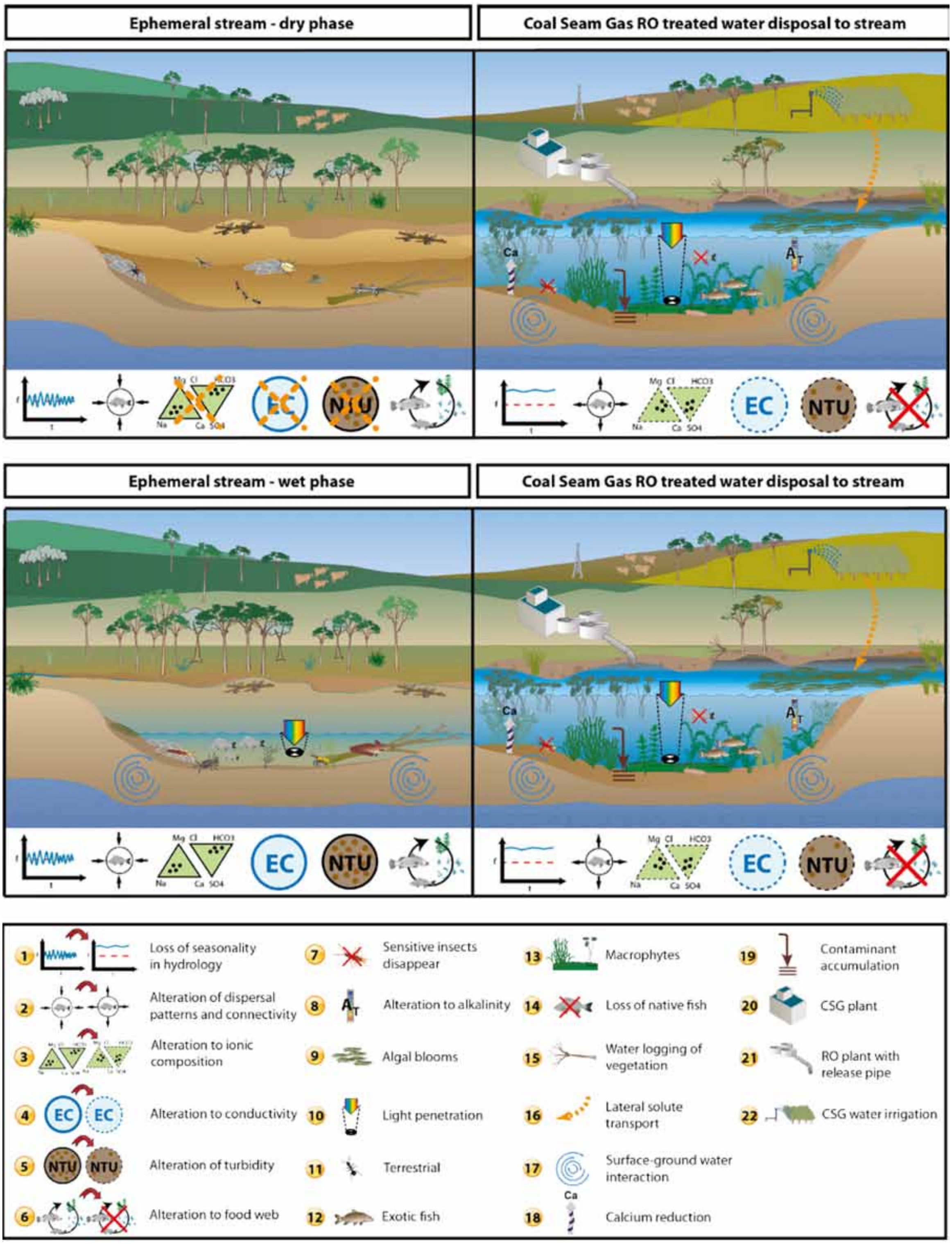

Figure 3: Conceptual model of the ecological effects of treated co-produced water entering an ephemeral stream (© Copyright, Takahashi et al. 2011b).

It is recommended that any review of existing licences and any future licence applications need to consider more carefully the background condition of the receiving waterway or the critical requirements of the intended beneficial use. This should be undertaken using a comprehensive risk assessment framework. Direct toxicity testing should be used as a tool for assessing the potential effect of the complex mixture of chemicals that may occur in receiving waterways, even when individual constituents meet relevant guidelines (Takahashi et al. 2011a) and particularly where compounds are present for which there are no guidelines. Water quality guidelines should also be developed for compounds commonly found in co-produced water for which there are currently no guidelines.

Cumulative impacts on water quality at a landscape scale are not well understood. Even when individual discharges meet relevant guidelines the cumulative impact associated with increasing load contributions such as salts, nutrients or heavy metals may have significant downstream impacts. There are significant knowledge gaps in the understanding of how cumulative water quality impacts develop in river systems, whether systems are able to assimilate multiple impacts and what critical thresholds apply for capping cumulative loads. Even when load limit conditions are applied through the development and application of discharge trading schemes, site-specific limits should be identified to protect values at the site scale, in addition to load limits at the catchment scale. Catchment scale modelling of cumulative water quality impacts is important to identify load limits, and site-specific assessments that consider background water quality are needed at the site scale.

3.6 Summary of critical water quality issues

The critical water quality issues associated with the discharge of co-produced water from coal seam gas and large coal activities are:

- The quality of untreated co-produced water is understood at a general level and there is a good understanding of the types of water quality variables that are likely to be of issue. However, the actual water quality varies widely between wells/mines and across regions and there are limited data to enable regional scale characterisation or to permit a typology of waters.
- The understanding of the ecological impacts of various toxicants is generally good and guidelines for many stressors and toxicants that aim to protect a variety of beneficial uses are well established at national, state and some regional levels. However, guidelines have mostly been established using data from perennial streams and may not be suitable for use in ephemeral systems, where most discharge is likely to occur. Furthermore, guidelines do not cover all compounds potentially present in co-produced water or hydraulic fracturing fluids.
- Treatment options to remove primary toxicants are well established. However, post-treatment is often required to re-mineralise water to make it suitable for specific beneficial uses, especially for use in irrigation and for release to surface waterways.
- Although toxicity of individual chemicals is understood, the effect of complex mixtures of chemicals within co-produced water on the environment is not so well understood, especially with respect to cumulative impacts and interactions between different discharges where multiple discharges may be occurring in a catchment. More investigation is needed to better understand risks around combined effects of chemicals on the environment and thresholds for significant impacts. Direct toxicity testing should

be undertaken and water quality guidelines developed for compounds commonly found in co-produced water and hydraulic fracturing fluids for which guidelines do not currently exist.

- Cumulative impacts on water quality at a landscape scale are not well understood and there are significant knowledge gaps in the understanding of how cumulative water quality impacts develop in river systems, whether systems are able to assimilate multiple impacts and what the critical thresholds should be for capping cumulative loads. At the catchment scale, modelling of cumulative water quality impacts is important to identify load limits. Site-specific assessments that consider background water quality in the receiving environment or for the intended beneficial use are also necessary.

4 Water quantity change impacts on aquatic ecosystems

4.1 Importance of flow regimes for aquatic ecosystems and associated environmental values

The magnitude, duration, frequency and timing of flows are key aspects of a natural flow regime that maintains channel form, influences water quality and drives most of the physical and ecological processes in rivers and wetlands (Nilsson & Renofait 2008; Nielsen et al. 2000; Puckridge et al. 2000; Puckridge et al. 1998; Poff et al. 1997; Richter et al. 1997; Sheldon & Walker 1997). For practical reasons, the natural flow regime is often characterised by different flow components, including cease-to-flow, low flow, freshes, high flows, bankfull flows and overbank flows (DNRE 2002). Both episodic and seasonal changes to the magnitude or timing of particular flow components can have significant environmental impacts (Poff & Zimmerman, 2010; Poff et al. 1997; Richter et al. 1997) and much work has been done in Australia to determine the critical water requirements for rivers and wetlands. Significant investments are being made by state and territory governments and the Commonwealth Government to secure and deliver environmental water to meet those requirements (CEWH et al. 2012).

Discharge of co-produced water increases flow in receiving waterways. This may reduce the number and duration of cease-to-flow events and increase the magnitude of low flow components (McGregor et al. 2011). Therefore, discharge of co-produced water is most likely to affect physical or ecological processes that rely on cease-to-flow or low flow components. Discharge volumes are likely to be relatively small compared to the volume of natural high flow events and, therefore, discharge of co-produced water is likely to have a less significant effect on those higher flow components and the ecological processes that rely on those flows.

Cessation of flow results in partial or complete drying of a river channel. Any water that remains in a river during cease-to-flow periods will occur in remnant pools that are characterized by poor water quality and, therefore, cease-to-flow events generally represent an environmental stress. However, biota that inhabit ephemeral streams have behavioural and/or physiological responses enabling survival in cease-to-flow events, and periodic drying can play an important role in carbon and nutrient cycling (Baldwin & Mitchell 2000; Nielsen & Chick 1997). Periodic drying may also be important in controlling the abundance and distribution of alien pest species that are less adapted to cease-to-flow conditions. For these reasons, cease-to-flow periods are necessary to support the life cycles of species that inhabit ephemeral river systems.

Low flows provide a continuous flow throughout the river channel and are important for sustaining aquatic habitats, maintaining water quality and linking aquatic habitats. Summer low flows are important for preventing or relieving the stress associated with cease-to-flow events, but low flows also provide important relief from higher flows. For example, stable low flows are likely to be important for recruitment of some native fish (King et al. 2011; Humphries et al. 1999) and for maintaining shallow riffle and run habitats and the range of biota that rely on those habitats (Arthington et al. 2000). Large differences between the magnitude of low summer and winter flows and shorter duration high flow and bankfull flow events also influence vegetation zonation within the river channel (Christie & Clarke 1999).

The ‘millennium drought’ and the need to reduce environmental stresses associated with water harvesting from natural systems have increased the focus on the role of low flows in Australian rivers (Rolls et al. 2012). A general conclusion is that actions that reduce natural flows are more harmful to ecological systems than actions that increase flows above natural levels (Marsh et al. 2012). However, depending on their magnitude, timing and frequency, artificial flow increases may still represent a significant environmental threat, particularly if they transform naturally ephemeral streams into permanently flowing waterways.

4.2 What hydrological changes are expected as a result of discharge of co-produced water?

Coal seam gas development in 2010 was estimated to have yielded approximately 13.2 GL (40% of 33 GL) of co-produced water across Australia. Over the next 25 to 35 years, the volume of co-produced water is predicted to increase, driven by projected development of coal seam gas sources in Queensland and New South Wales (RPS 2011). Peak discharge from individual production areas may be up to 200 ML/day, but the peak for most production areas has been reported as most likely to be less than 160 ML/day (QGC Limited 2009). Moreover, production rates will vary below capacity most of the time and, therefore, average yields of co-produced water from most production areas are expected to be less than 100 ML/day and less than 5 ML/day in smaller production areas (e.g. Pony Hill Water Treatment Plant (URS 2009a)).

While discharge from individual treatment plants may be relatively low, the cumulative impact of multiple production areas within the same water catchment could be large. The cumulative effect of multiple discharges should be modelled for all affected catchments. That modelling should specifically compare the effect that future discharge scenarios have on natural flow regimes and consider the four low flow metrics suggested by Mackay et al. (2012):

- average number of zero-flow days per year
- baseflow index
- average annual minimum flow adjusted for catchment area
- flows exceeded 90 per cent of the time.

The risk of co-produced water increasing flow in receiving waterways to such an extent that it could cause environmental harm depends on the magnitude of discharge compared to the natural flow in the receiving waterway. Poff and Zimmerman (2010) reported that the risk of ecological change increases with increasing magnitude of flow alteration. Any discharge that turns a naturally ephemeral stream into a perennial stream would represent a very high risk to environmental values (National Research Council 2010). Similarly, a discharge of 100 ML/day of co-produced water into a receiving waterway at a time when it would naturally have a flow of less than 20 ML/day would fundamentally change the receiving environment and present a very high environmental risk. However, if the discharge is relatively small compared to the flow in the receiving waterway, then the change in flow will be small and the environmental risk less significant.

There is little quantitative information about specific flow changes in receiving waterways as a result of existing discharge of co-produced water. McGregor et al. (2011) modelled the likely hydrological changes to sections of the Dawson River in the Bowen Basin and

Condamine River in the Surat Basin as a result of aggregated coal seam gas industry discharge scenarios. The models were based on a worst case scenario that assumed the maximum discharge volumes and constant, rather than timed, release patterns. The work indicated that discharging co-produced water would significantly reduce or eliminate cease-to-flow periods and increase the magnitude of low flows (McGregor et al. 2011). The modelled hydrological changes were considerable and would not meet the environmental flow objectives for each stream.

The coal seam gas industry in Queensland is not expected to reach peak production until the year 2018 (Shaw 2010). As the industry grows, the total volume of co-produced water that needs to be disposed of will also increase and discharge volumes may peak in approximately 2018. Peak production rates are expected to be maintained for around 10 years and the volume of co-produced water should decline after about 2027 (McGregor et al. 2011). These changes in production volume could significantly increase the risk to receiving environments, because they will need to respond to a wetter flow regime and then adjust back to a more normal flow regime (McGregor et al. 2011). If the initial change to a wetter flow regime is large, then it is unlikely that the system will be able to return to its current state and condition when discharge of co-produced water ceases.

4.2.1 Ecological risks associated with discharging co-produced water into ephemeral streams

Flow modifications that result in the loss of either extreme high flow events or extreme low flow events, including ‘cease-to-flow’ events, commonly alter ecological assemblages. Specific changes can include reduced biological diversity, a change in the most dominant species and an increase in non-native species (Poff & Zimmerman 2010). Relatively small discharges into an ephemeral system during the dry season may increase moisture content in the substrate, top up remnant pools and even provide some flow for a small length of stream. However, the greatest risks occur where the discharge creates perennial flow throughout an entire stream or significant river reach.

Periodic wetting and drying events play an important role in maintaining channel morphology and nutrient cycling in ephemeral systems. In the same way that periodic wetting and drying in wetlands helps to maintain depressions on the floodplain, water held in remnant pools as rivers dry contributes to overall channel complexity (Reich et al. 2010). A loss of periodic cease-to-flow events may limit sediment consolidation and the breakdown of organic matter buried in the streambed, which may lead to an accumulation of unconsolidated sediments (Ryder et al. 2006). Moreover, increased flows without drying may lead to increased bed and bank erosion, which will have detrimental effects on biota that rely on in-stream habitats (National Research Council 2010; Bjornsson et al. 2003). A lack of wetting and drying events can also adversely affect biofilm production and nutrient dynamics (Ryder et al. 2006; Boulton & Brock 1999). Biofilms and nutrients are critically important to riverine food webs and disruptions to one or both of these will have cascading effects on higher order biota such as macroinvertebrates and fish.

Fish are particularly sensitive to increases and decreases in flow magnitude and most studies investigating the ecological effects of modified flow regimes have reported changes in fish communities (Poff & Zimmerman 2010). A shift from an ephemeral to a perennial system may allow larger-bodied, flow-dependent fish species to colonise and out-compete or prey upon smaller endemic species that are adapted to ephemeral systems and normally survive cease-to-flow periods by retreating to refuge pools (Bond et al. 2010; Reich et al. 2010). Moreover, a shift from an ephemeral to a perennial system is likely to favour exotic or non-endemic native fish species (Bond et al. 2010; Reich et al. 2010; USEPA 2010). Several studies in the US have reported different fish community composition in streams that receive co-produced water compared to streams that do not receive any co-produced water (National Research Council 2010; USEPA 2010).

Macroinvertebrate assemblages in ephemeral streams are often dominated by highly mobile taxa and with desiccation-resistant eggs capable of rapidly colonizing habitats when water is present (Reich et al. 2010). A shift to more permanent flow is likely to increase macroinvertebrate abundance and diversity as conditions become more suitable for flow-dependent taxa, such as filter feeding caddisflies, Simuliids and Baetid mayflies, and other species that cannot tolerate periodic drying or the poor water quality that characterizes refuge pools (Reich et al. 2010). Increases in the abundance and diversity of macroinvertebrates may be considered beneficial; however, any shift away from natural endemic conditions can affect other ecological processes and should be treated with caution. For example, an increase in more permanent flow may assist the spread of exotic species or insects that are vectors for diseases.

A shift from ephemeral to perennial flow is also likely to result in significant changes to the abundance and composition of littoral vegetation and aquatic macrophytes. Reich et al. (2010) reported that streams that had artificial perennial flow had more diverse and more extensive macrophyte assemblages than nearby streams that were ephemeral and unregulated. Any loss of cease-to-flow events can cause the local extinction of plant species that need dry conditions at certain times of the year and result in general shift away from more terrestrial plant species (National Research Council 2010). As described for fish and macroinvertebrates, increases in vegetation abundance and diversity may be considered beneficial; however, there is a risk that more stable flows will have other effects on vegetation – for example, enhance the spread of weeds.

Any assessment of the risks of increasing flow in naturally ephemeral systems should also consider whether cumulative effects on different communities will affect heterogeneity at a landscape level. Ephemeral streams often have lower biological diversity and lower abundance than perennial streams, but they also provide important niche habitats for some endemic species. Ecological variability at a landscape scale is critically important and any action that reduces that variability is a risk.

4.2.2 Ecological risks associated with increasing the magnitude of low flows

The risks associated with increasing the magnitude of low flows are similar to those described in regulated systems, where natural river channels are used to carry irrigation water. The main risks are reduced fish recruitment, disturbance to riffle dwelling macroinvertebrates, reduced availability of shallow, slow-flowing habitats, and changes to riparian vegetation zones.

4.2.2.1 Disruption to native fish recruitment

Some native fish, such as Macquarie perch, deposit eggs in gravel substrates in shallow riffle or run habitats in late spring and early summer. Moderate flow through these habitats helps to keep the eggs aerated by passing well-oxygenated water over them and by preventing smothering by fine sediment. However, excessive flow during the spawning and egg development season may wash the eggs out of these habitats, leading to poor recruitment (King et al. 2011).

The larvae and juveniles of other native fish such as Crimson-spotted rainbowfish, Australian smelt and Carp gudgeon rely on shallow backwater and slackwater habitats for food and protection (King 2004). These backwaters are characterized by warm temperatures, abundant food sources, such as zooplankton and small macroinvertebrates, and are a refuge from large-bodied predators that are unable to access these shallow habitats (King 2004; Humphries et al. 1999). The importance of slackwater and backwater habitats for developing fish larvae and juveniles has been widely reported and proposed for Australian and overseas rivers (Humphries et al. 2006; Pease et al. 2006; King 2004; Humphries et al. 2002; Freeman et al. 2001; Humphries & Lake 2000; Humphries et al. 1999; Junk et al. 1989; Schiemer et al. 1989). The distribution and abundance of slackwater, and backwater habitats and their suitability as fish nurseries are strongly linked to water level and flow volume (Humphries et al. 2006; Humphries et al. 2002; Humphries & Lake 2000; Humphries et al. 1999; Schiemer et al. 1989). In waterways with low sinuosity and relatively vertical banks, artificial increases in the magnitude of low flows that effectively fill the bottom of the channel are likely to reduce the abundance and distribution of backwater habitats and could adversely affect the recruitment of native fish that rely on those habitats (Nielsen et al. 2005; Schiemer et al. 1989).

Changes in water temperature can also reduce fish recruitment. Many native fish species spawn in response to increases in water temperature (Astles et al. 2003; Ryan et al. 2002; Schiller & Harris 2001). Water temperature is usually inversely related to flow, and rapid increases in water temperature are most likely to occur when flow is very low. Depending on treatment, co-produced water may be cooler than river water during summer, but even if the temperature difference is not great, simply increasing the magnitude of flow slows the rate at which water temperatures can rise. If discharge of co-produced water prevents or delays temperature-related spawning cues, then fish recruitment is likely to be reduced. Reduced summer temperatures may also have adverse effects on growth and survival rates of native fish and macroinvertebrates (Chessman & Royal 2004; Astles et al. 2003). Conversely, during winter, co-produced water is likely to be warmer than ambient water temperature. The discharge of warm water to a stream may induce spawning in some species but result in poor larval and juvenile survival if they are washed downstream into cooler water.

4.2.2.2 Physical disturbance to in-stream habitat and macroinvertebrates

Increased flows may dislodge macroinvertebrates from the substrate or cause them to voluntarily abandon benthic substrates and drift downstream (Borchardt 1993). Drifting is an important dispersal mechanism for many benthic macroinvertebrates and is often triggered by increased flow, seasonal or temperature cues or an encounter with a competitor or predator. However, excessive or very frequent drift can deplete macroinvertebrate communities in habitats where drift is initiated. Moreover, high mortality rates among drifting macroinvertebrates may reduce populations over a wider area. URS (2009a) recently completed a literature review to assess the potential for the release of co-produced water to increase macroinvertebrate drift in receiving waterways. It concluded that macroinvertebrate drift was unlikely to increase if co-produced water was discharged to receiving waterways at a relatively consistent rate, but the number of drifting animals may increase if it was discharged in discrete pulses and if those pulses caused large and rapid changes in flow magnitude (URS 2009a).

In more erosive rivers, significant increases in flow or sustained increases in flow may erode the streambed and banks (National Research Council 2010). This is likely to affect biota in two ways. First, the plants and animals that rely on the eroded habitats will be displaced. Second, biota in downstream environments may be smothered or have reduced fitness due to higher loads of suspended material in the water column or deposition of that eroded material onto the streambed.

4.2.2.3 Reduced availability of shallow, slow flowing habitats

Increased flows are more likely to affect macroinvertebrates by modifying in-channel habitats, especially shallow riffles and runs. Under relatively low flows, these shallow habitats provide a wide range of hydraulic conditions that allow a variety of different types of macroinvertebrates to co-exist in the same stream section (Williams & Smith 1996). Emergent rocks and logs within these fast flowing environments provide important egg laying sites for many macroinvertebrate species, and the presence of these features can influence which species are likely to occur at any given site (Reich & Downes 2003). Shallow riffle and run habitats are also important sites for biofilm and diatom growth, and for fish spawning (King et al. 2011).

Small increases in the minimum flow or short duration higher flow events represent a slight disturbance to these habitats and their resident biota, but do not represent a significant risk to environmental values. In contrast, substantial increases in the minimum flow can drown riffle and run habitats and either reduce the range of hydraulic environments or create a prolonged disturbance that flushes away or scours resident biota and organic material (McGregor et al. 2011). For macroinvertebrates, these hydraulic changes may lead to increased rates of drift and displacement to less suitable habitats as well as the loss of potential oviposition sites, which may also reduce subsequent re-colonisation.

4.2.2.4 Changes to riparian vegetation zones

Increasing the magnitude of low flows could effectively reduce the difference in elevation between the low flow level and the top of the bank. Wetting and drying throughout this vertical range determines the diversity and condition of riparian vegetation communities (Stromberg et al. 2007). If the discharge of co-produced water significantly increases the height of low flows, then freshes and high flows will be less pronounced and the vertical range available to riparian plants with different wetting and drying requirements may be compressed. Some species may be completely lost from where they most commonly occur on low benches that are exposed during low flow and cease-to-flow conditions, and submerged during freshes and all higher flows. More generally, responses of riparian vegetation to reduced variability in flow levels may include loss of habitat variability and, hence, reduced riparian diversity and shifts in annual and ephemeral plant communities to perennial dominated communities along the stream edge (James & Barnes 2012).

In addition, potential impacts of flow supplementation on riparian vegetation may include shifts in the zonation of vegetation communities with an increase in mesic (i.e. favouring moist conditions) species, promotion of vigour and height of the vegetation fringing the streams, increased opportunities for establishment of vegetation on exposed in-stream benches and bars, and weed encroachment of exotic species preferring more moist conditions (James & Barnes 2012).

4.3 What types of stream are likely to be affected by discharge of co-produced water?

Any continuous discharge, depending on the volume and timing of the discharge, may represent a significant ecological risk to all watercourses.

However, Mackay et al. (2012) suggested that small changes in discharge would have the greatest effect on streams that were weakly ephemeral or weakly perennial. They developed a simple four-level system to classify streams according to their susceptibility to a changed low flow regime (Table 7) – for example, a small increase in discharge could cause a weakly ephemeral stream to become perennial and a small decrease in discharge could cause a weakly perennial stream to become ephemeral. Mackay et al. (2012) stated that these shifts between ephemeral and perennial states would have more profound ecological effects than changes in degree that did not result in a complete change of state, such as a shift from strongly ephemeral to weakly ephemeral. Using that logic, it may be argued that the ecological risks associated with the discharge of co-produced water will be highest in weakly ephemeral or weakly perennial streams, because relatively small increases in discharge will cause a shift from one flow state to another. This does not mean that risks are low for highly ephemeral streams, just that the flow increase needs to be greater to cause a shift to a more perennial state. Even at moderate to small rates, continuous discharge, which turns an ephemeral system into a perennial system, represents a high ecological risk.

Table 7 Simple four level low flow classification system for streams and the ecological risk associated with small changes in discharge (© Copyright, Mackay et al. 2012).

Class	Description	Ecological risk associated with small change in discharge
1	Highly ephemeral	Significant but lower risk than Class 3 because greater volume of continuous discharge needed to change to perennial system
2	Ephemeral	Significant but lower risk than Class 3 because greater volume of continuous discharge needed to change to perennial system
3	Weakly ephemeral to weakly perennial	High risk of ecological change as a result of a small change in discharge
4	Strongly perennial	Lowest risk because increase in discharge will not change the flow state, although a proportionally large increase in flow magnitude may still represent an ecological risk

Mackay et al. (2012) used the Normalised Vegetation Difference Index (NVDI) to spatially extrapolate the simple classification system across Australia. It showed that the streams most likely to be affected by small changes in discharge are on the western foothills of the Great Dividing Range (see Figure 4). There is some overlap between these high risk areas and the basins identified for coal seam gas exploration (see Figure 4). Moreover, most of the streams within the coal seam gas exploration areas are strongly to weakly ephemeral. If enough co-produced water is continually discharged to these streams to make them perennial, then the ecological risks may be very significant.

4.4 Quantifying risks to individual waterways

McGregor et al. (2011) proposed a risk assessment approach to managing discharge of co-produced water to natural waterways. That approach involved five steps:

1. Identify the hazards associated with the disposal of co-produced water. This requires an understanding of the hydrological characteristics of the receiving environment.
2. Select ecological assets that will be used as indicators of hydrological change. This

selection requires an understanding of the flow requirements, particularly the flow Threshold of Concern of different assets.

3. Develop coal seam gas water disposal hydrology scenarios that are based on the likely volume and timing of discharge.
4. Analyse the potential risks associated with the disposal of coal seam gas water. This task considers how the modelled hydrological changes for each scenario are likely to affect the selected ecological assets or indicators.
5. Characterise the risks and development of a management framework that adjusts the volume and/or timing of discharges to minimise risks to particular assets.

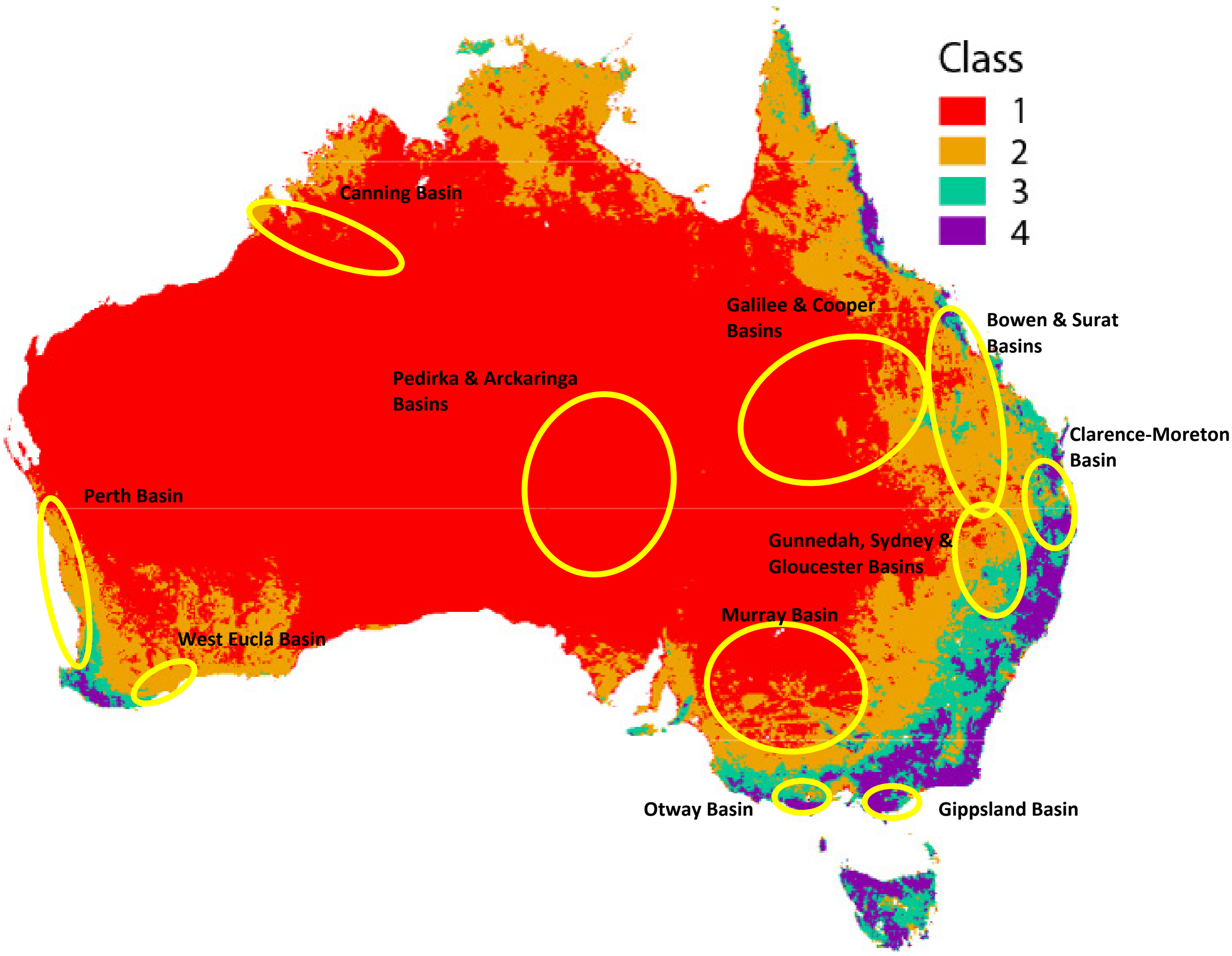

Figure 4 Predicted low flow classification based on simplified flow classification extrapolated using long-term NDVI (figure reproduced from Mackay et al., 2012). Yellow rings approximately represent major areas for coal seam gas and coal mining basins (© Copyright, Geoscience Australia & BREE 2012).

A fundamental tenet of the approach outlined by McGregor et al. (2011) is that the risks of disposing of co-produced water will vary for each waterway depending on the existing water regime, the volume and timing of the proposed discharge and the ecological assets associated with the waterway. In each case, the discharge should be delivered in such a way that it complies with the environmental flow objectives that are specified in the Water Resource Plan (WRP) for the receiving waterway.

4.5 Reducing risks associated with discharging co-produced water

The Queensland Department of Environment and Heritage Protection (DEHP) has the expectation that, where possible, co-produced water will be beneficially used for the environment, existing or new water users, and existing or new water-dependent industries (DEHP 2012). Co-produced water will only be discharged to natural waterways in a manner that improves local environmental values or if there are no viable alternative management options, and discharging to watercourses will not adversely affect environmental value (DEHP 2012; McGregor et al. 2011). Where such needs arise it will be treated to an appropriate standard to minimize the water quality risk to ecological values. It should be noted that discharges of co-produced water to natural waterways are unlikely to be used as environmental flows because the 10 to 20 year timeframe for their discharge means that they cannot be relied on to meet environmental flow objectives in the future (RPS 2011).

In most existing projects, co-produced water is discharged to natural waterways at the same rate that it is extracted from the aquifer (e.g. the Fairview Field - URS 2009a). However, to release the water at a time of least environmental effect it will be necessary to either hold it in storages or dispose through other means when flow in the receiving waterway would naturally be low or zero. On-site storage could be incorporated into the water treatment process and is likely to be an important feature of any co-produced water management strategy. If operated properly it should reduce the risk of significant changes to the flow regime in the receiving waterway. However, if the storages catastrophically fail they could create a very large flow in the receiving waterway at a time that would have severe consequences for ecological values. In summary, the risks associated with discharging co-produced water may be reduced by holding the water in storages when flow in the receiving waterway is low and releasing it during periods of naturally high flow. These storages are different to evaporation ponds, which may be used to dispose of brine.

4.6 Examples of co-produced water being released into waterways with high seasonal variability

Most of the coal seam gas developments in Australia have discharged relatively small quantities of co-produced water to natural waterways and at the time of preparing this report there were no known examples where it has been stored on-site with discharge timed to coincide with naturally higher flows in the receiving environment. The Fairview Field project has been discharging small volumes of approximately 4.5 ML/day of co-produced water to naturally ephemeral streamlines and larger streams in the Dawson River catchment in the Bowen Basin since 1993 (URS 2009a). No ecological condition monitoring was conducted in the receiving waterways prior to discharge commencing and, therefore, it is difficult to quantify any environmental effects of the discharge. However, it was reported that there were only slight differences in ecological condition between the streams that received co-produced water and physically similar streams in the same catchment that did not receive any co-produced water (URS 2009a). The construction of a desalination plant to reduce salinity levels in the co-produced water prior to discharge has improved water quality in the receiving waterway and URS (2009a) suggested that the higher flows as a result of the discharge may actually improve the condition of the macroinvertebrate community; however, the environmental assessment report did not specifically consider potential negative effects associated with reducing the number or duration of cease-to-flow events.

Several other coal seam gas projects have started discharging to receiving waterways. The former Eastern Star Gas operation (now owned and run by Santos) could discharge up to 1 ML/day of reverse osmosis treated co-produced water from its Gunnedah Basin operations to Bohena Creek (RPS 2011). Australia Pacific LNG also discharges up to 20 ML/day of treated co-produced water from the Talinga Water Treatment Facility into the Condamine River (APLNG 2011). Water is harvested from the Condamine River for irrigation and stock and domestic purposes and, therefore, the discharge of 20 ML/day does not increase total annual flows above the estimated pre-development level. Further development of the Australia Pacific LNG project could see up to 140 ML/day of co-produced water discharged to the Condamine River (Conics 2010). The effects of different discharge scenarios, including no discharge at times when the river would naturally have little or no flow, have been modelled and the results of that modelling may influence how co-produced water is discharged in the Condamine Catchment in the future (Conics 2010). At the time of writing, none of the coal seam gas development projects in Australia were known to have methodically varied rates of discharge to receiving waterways to match natural flow patterns.

Approximately 45 per cent of co-produced water from coal-bed methane operations in the US is discharged directly into surface waters (USEPA 2010). This equates to approximately 83 GL/year and is mostly confined to the Black Warrior Basin in Alabama and Mississippi (11 per cent of total water discharged into surface waters), the Powder River Basin in Montana and Wyoming (72 per cent) and the Raton Basin in Colorado and New Mexico (11 per cent) (USEPA 2010). Most of that water is untreated and while some studies have investigated the water quality impacts associated with the discharge, very little work has been done to quantify the hydrological changes in the receiving waterways and ecological effects of those altered flow regimes (USEPA 2010). Moreover, while the risks associated with increasing flows are acknowledged (National Research Council 2010; USEPA 2010), there do not appear to be any instances where the timing of releases is adjusted to match seasonal variations in high and low flows.

The Powder River project is relevant to Australia because the volume of co-produced water discharged to natural waterways is large and because those waterways have seasonally variable flow regimes. The management of co-produced water from the Powder River project varies between the Wyoming and Montana State jurisdictions. In Wyoming, 64 per cent of co-produced water is stored in evaporation basins and 20 per cent is discharged directly to waterways with or without treatment depending on quality (National Research Council 2010). In the Montana part of the Basin, 61 to 65 per cent of co-produced water is discharged to natural waterways and 26 to 30 per cent of the remainder is used for surface irrigation (National Research Council 2010). All of the co-produced water in Montana is treated prior to discharge or beneficial use.

Co-produced water from the Powder River Basin is discharged at multiple points, including into some large, permanent rivers and some smaller ephemeral tributaries. The impact of discharges is likely to be greater in the ephemeral streams than the larger rivers, but there does not seem to be any deliberate attempt to adjust discharge volumes in different seasons. Although not mentioned, it is possible that the use of different disposal methods varies throughout the year with more water disposed of via evaporation ponds in the dry season when flow in the river would be naturally low, which may create a more favourable release pattern for receiving waterways.

The need to dispose unwanted water from mining operations is not limited to coal seam gas operations. Similar problems occur with coal mine operations that intercept aquifers. Many coal mines in Australia have been operating for much longer than coal seam gas operations and some have been producing and discharging large volumes of unwanted water for a considerable time. However, information about the management of that water is limited. DEHP (2009b) reviewed the cumulative impact of water discharges from coal mines in the Fitzroy River Basin and reported that while 25 out of the 39 mines considered in the study were only allowed to discharge when there was at least some natural flow in the receiving waterway, only seven of the environmental authorisations prepared for those mines specified a minimum magnitude for that flow. The report did not specify how water was stored at each mine to manage the timing of the discharge, although most mines tended to have enlarged on-site storage capacity to cope with variable climate conditions (Hamstead & Fermio 2012).

Environmental risks from the discharge of co-produced water into waterways are broadly acknowledged in Australia and overseas, but very little monitoring has been conducted to quantify these risks and no projects are known to vary release volumes in different seasons to minimise those risks. The Queensland Healthy HeadWaters coal seam gas water feasibility study developed specific guidelines for managing flow regimes (McGregor et al. 2011). Those guidelines can be used to assess the specific risk of co-produced water discharge and develop appropriate release plans for each project. Moreover, the release plans should be accompanied by robust monitoring programs to assess their effectiveness at reducing environmental risk and to allow adaptive management to further reduce environmental risks and improve the guidelines.

4.7 Knowledge gaps

In assessing the flow-related risks to individual waterways, it will be necessary to identify:

- the type of flow regime present
- the existing values of that waterway
- likely pest species that may colonise or become more abundant if flows increased
- any other stressors on the system that may exacerbate or mitigate the effects of increased flow.

Many streams in areas where coal seam gas and large coal mining are occurring are weakly perennial or ephemeral – that is, the stream types most at risk from increased flow. Site-specific studies are needed to determine appropriate environmental flow requirements to protect environmental values in these streams, including wetting and drying regimes for wetlands. If discharge of co-produced water to waterways is required, it should not compromise the ability to meet the environmental flow needs.

Methods for determining environmental flow requirements are generally well established and environmental flow studies have been undertaken for many rivers and wetlands, but they are mostly restricted to larger perennial rivers and ecologically important wetlands. Site-specific studies are needed to determine flow characteristics and environmental flow requirements in streams as part of the assessment and approval process for new mines, and proponents should demonstrate that proposed discharge options will not negatively impact environmental values.

Impacts at the local site-scale may be small - for example, where the potential discharge volume is only a small proportion of the mean annual stream flow (Conics 2010), or discharge will only be made during emergency situations as a result of floods, or when supply exceeds other beneficial use demands (Hamstead & Fermio 2012; RPS 2011). However, the cumulative impact of multiple small-volume discharges is not well documented. Catchment scale models are needed to model surface flow scenarios associated with the range of projected discharge volumes and decisions can then be made at a catchment scale of the acceptable discharge volumes and timing of releases to minimise impacts on natural flow regimes. Modelling could also be used to assess whether discharge of co-produced water can help augment environmental flows where excessive extractions have reduced the natural flow.

4.8 Summary of critical water quantity issues

The ecological risks associated with discharging co-produced water to natural waterways will be largely determined by the proportional change in the flow regime and whether the discharge results in a change from an ephemeral to a perennial system. Small increases in flow that do not result in a change in flow state will represent a relatively low risk. However, continuous discharge of large volumes of water that turn ephemeral systems into perennial streams or that significantly increase the magnitude of the minimum flow will represent a high ecological risk.

Specific changes as a result of increased co-produced water discharge may include:

- a change in the composition of the fish community with large-bodied flow-dependent species becoming more abundant and potentially displacing endemic species that are adapted to surviving in remnant pools during dry periods
- an increase in the abundance and diversity of macroinvertebrate taxa in naturally ephemeral streams, but a potential loss of riffle dwelling species from perennial streams if those habitats are drowned out for extended periods
- an increase in the abundance and extent of aquatic macrophytes and littoral vegetation, but a loss of diversity among riparian vegetation because the vertical distance between the minimum flow level and maximum flow level, and hence the proportion of the bank with different watering frequency, will be reduced
- disruption to breeding cycles that rely on specific flow conditions, or flow and temperature cues
- an increase in the abundance of exotic and pest fish, macroinvertebrates and plant species.

Most streams in areas where coal seam gas and large coal mining are occurring are weakly perennial or ephemeral, which are the stream types most at risk from increased flow. Site-specific studies are needed in potentially impacted streams, as part of the assessment and approval process for new mines and proposed discharges, to demonstrate that discharge options will not negatively impact environmental values or significantly alter flow regimes. Furthermore, catchment scale models are needed to model flow scenarios associated with the range of projected discharge volumes.

In summary:

- The total quantity of co-produced water is large, but is variable at a site level.

- If co-produced water is released to waterways the level of risk depends on the timing and volume of the release and on how significantly the co-produced water will change the water regime or channel geometry.
- For streams that are strongly perennial and carry large flow volumes, discharge of small volume of co-produced water represents a relatively low risk. However, for streams that are weakly perennial or ephemeral, the release of co-produced water represents a significant risk, because it can result in a shift in the flow regime. Specifically, a constant discharge of even low to moderate volumes of co-produced water can result in ephemeral streams becoming perennial and seasonal wetlands becoming permanently inundated, and increased flows may lead to increased bed and bank erosion, which will have detrimental effects on biota that rely on in-stream habitats.
- There are some regional scale data that identifies the general areas where ephemeral and weakly perennial streams exist in Australia. Most coal seam gas and large coal mine production that occurs in Queensland and northern New South Wales is in areas where at-risk streams (i.e. weakly perennial or ephemeral streams) are dominant. Other areas like the Clarence-Moreton, Sydney and Gippsland Basins tend to be characterised by perennial streams.
- Protecting ephemeral streams from impacts will require co-produced water to be managed in a way that minimises the likelihood of discharge to waterways and wetlands during periods when they would normally be dry (i.e. summer in temperate areas and the winter dry season in northern areas) and releasing the water at a rate that mimics pre-development flows.
- At a site scale, potential impacts are generally manageable with appropriate treatment and control of discharge patterns; however, cumulative impacts are not well understood. Catchment modelling of stream flow is needed to assess cumulative impacts

5 Risk management frameworks for the assessment of environmental impacts

Given the number and extent of coal seam gas and coal mining developments likely to be established across Australia, a scientifically rigorous, consistent and credible process for assessing environmental effects is essential.

New coal seam gas and large coal mining developments are required to prepare an environmental impact statement (EIS) as part of their environmental approvals process. Approval from the Australian Government Environment Minister may also be required if the action has, will have, or is likely to have a significant impact on a water resource[1]. Current 'best practice' is to use risk-based approaches to assess the risks coal seam gas and large coal mining developments have on key environmental assets (e.g. AS/NZS 2009; URS 2009a). The outcome is a risk management strategy that aims to minimise impacts of these operations to the environment.

This Chapter reviews available risk assessment frameworks that may be appropriate for assessing the risks related to the disposal or use of co-produced water from coal seam gas and coal mining. The discussion is focused on risks to surface water resources as the key environmental assets. In particular, the review focuses on frameworks to assess risks to aquatic environments, but also considers the risks to other beneficial uses or environmental values of water resources, such as domestic use, agricultural use, use for recreation and aesthetics, and industrial use. The beneficial uses or environmental values of a water resource are those considered in the National Water Quality Management Strategy (NWQMS 1998) and the Australian and New Zealand Guidelines for Freshwater and Marine Water Quality (ANZECC/ARMCANZ 2000).

Cultural and spiritual values of the key surface water resource assets may also need to be considered in detail.

5.1 Ecological risk assessment

5.1.1 General background

Risk is defined as the chance or likelihood, within a particular timeframe, of an adverse event with specific consequences occurring (Burgman 2005). The Australian Standards define risk as the effect of uncertainty on objectives (AS/NZS 2009).

Risk assessment is a process used to collect, organise, integrate and analyse information for use in a planning environment, where the outcome is the analysis and prioritisation of risks to a stated objective.

Risk management involves the development of strategies to minimise, monitor and control the probability and/or impact of adverse events.

[1] Required under the *Environment Protection and Biodiversity Conservation Act 1999* (Cwlth) (EPBC Act 1999).

More specifically, an Ecological Risk Assessment (ERA) involves evaluating the likelihood that adverse ecological effects may occur, or are occurring, as a result of exposure to one or more hazards, and the consequences of such an exposure. The objective of an ERA is to provide a robust process that incorporates a transparent, scientific, precautionary and ecologically sustainable approach to the assessment and management of environmental risks.

The essential outcomes of an ERA are:

- a well-articulated statement of the problem
- an increased understanding of the significance of risk
- a determination of where and how to implement measures to minimise risks, supported by use of the ‘best available’ evidence-base for risk analysis.

The risk of adverse effects due to hazards (or stressors) is generally defined as the product of the likelihood or probability of the effect occurring and the consequences of the effect if it occurs. Thus:

ecological risk = likelihood of effect occurring x consequence of that effect

A stressor is any physical, chemical or biological entity that can induce an adverse response in an ecosystem. A ‘threat’ or ‘hazard’ is human-induced factor that directly or indirectly causes a change in an ecosystem.

The risks may be:

- biological - including predation and invasive species
- physical - including drought, flood, and loss of habitat
- chemical - for example, toxicants.

An ERA may also need to consider social, political or economic issues, which may be important in influencing ecological outcomes.

There are several frameworks for risk assessment and risk management available for different settings and disciplines, as well as some specifically aimed at ecological risk (e.g. Peters et al. 2009; Burgman 2005; Hart et al. 2005). These frameworks have many features in common, in that they outline a structured iterative process for the identification of threats, hazards or stressors, the analysis of risks to valued assets, the management of these risks, and the monitoring of outcomes to ensure the management plan is working. The steps involved in conducting an ERA are discussed in some detail in the section below.

The assessment of risks to humans and the environment from waste discharges and contaminated sites are now required in most developed countries and, as a consequence, there are a number of frameworks and guidelines available for conducting ERAs. The US EPA was probably the first to produce extensive guidelines for conducting ERAs (USEPA 1998). These are very prescriptive, but contain considerable guidance[2]. Other useful overseas guidelines include Ohio EPA (2008) and Ashton et al. (2008).

2 The US EPA also runs an Ecological Risk Assessment Support Center (ERASC), which contains useful information on conducting ERAs, although they largely focus on Superfund sites (i.e. severely chemically contaminated sites (see http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=154348)).

Guidelines for risk assessment have been produced by Standards Australia and Standards New Zealand (AS/NZS 2009). These specify that the best available information must be sought when gathering consequence and likelihood data. They also provide practical advice on sourcing data, such as structured interviews with experts in the area of interest, use of a multi-disciplinary group of experts, individual evaluations using questionnaires, use of computer and other modelling and use of fault trees and event trees.

The Australian Standard recognises three levels of risk assessment depending on the information and data available (AS/NZS 2009):

- Initial screening level analysis, where qualitative information is sufficient.
- Quantitative analysis when descriptive scales (e.g. unlikely, possible, likely, highly likely) are available.
- Quantitative analysis when numerical values are available for both consequences and likelihood.

Given that risk assessments for new coal seam gas or coal mining developments are likely to be subject to public scrutiny, it will be important to demonstrate 'best practice' and show that, as a minimum, the Standards Australia guidelines have been followed.

The Victorian Department of Environment and Primary Industries (DEPI) has developed an ERA methodology (called AVIRA – the Aquatic Value Identification and Risk Assessment) to assist in the management of the State's rivers (Peters et al. 2009). Additionally, the Victorian EPA has published risk assessment guidelines for waste water discharges to waterways (Victorian EPA 2009; Victorian EPA 2004).

Many ERAs are qualitative or semi-quantitative. There are criticisms of these methods; one being the use of descriptive scales, which are rarely transparent, often are subjective and based on limited expert opinion, and can be very difficult to validate (Burgman 2005; Burgman 2001). For this reason, quantitative methods are often preferred, if possible. Some guidance is provided below on quantitative modelling approaches that are now being used in ERAs, including in situations where there are significant uncertainties and knowledge gaps. In assessing ecological risks to natural resources like rivers, wetlands and estuaries, there is generally a need to consider a wide range of threats and hazards (multiple stressors) and a wide range of ecological effects.

ERA frameworks that are catchment-based (to address large spatial scales) and assess risks to multiple ecological assets from multiple stressors or hazards, are now being developed (Landis et al. 2012; Hart et al. 2007; Hart et al. 2006; Landis 2005; Serveiss et al. 2004; Serveiss 2002). Some examples of these catchment-based ERAs are provided below.

5.1.2 The ERA process

The ERA process involves undertaking a logical sequence of key steps. This is shown diagrammatically in Figure 5.

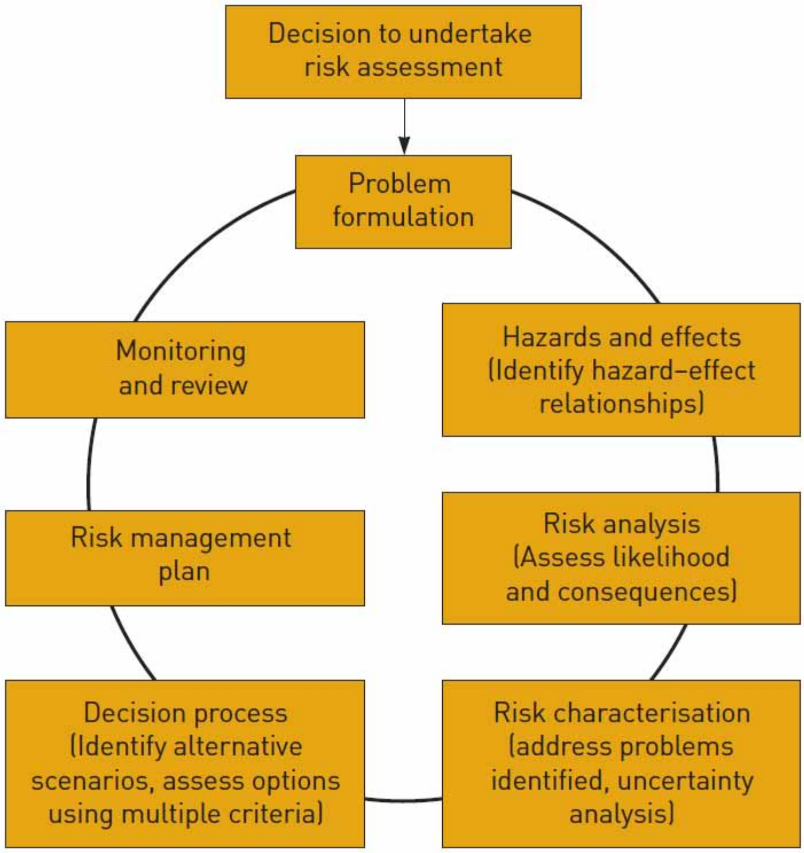

Figure 5 Risk assessment and management framework (© Copyright, Hart et al. 2007).

5.1.2.1 Defining the problem

The problem formulation phase is a planning and scoping process that establishes the goals, breadth and focus of the risk assessment. The end products of the problem formulation phase are:

- an outline of the assessment process, to provide confidence that the process will be transparent and credible
- identification of the important ecological assets (or values) and the key stressors, hazards or threats to these assets. Hazards are prioritised by evaluating their effects on valued elements of ecosystems and ecosystem services
- identification of the appropriate spatial and temporal scales for evaluating the risks
- a conceptual model for each of the ecological issues, where the key stressors are linked to the ecological effect. These conceptual models form the basis for more quantitative ecological models in systems where there is both sufficient knowledge about the linkages and sufficient data to quantify them
- identification of the assessment end-point(s)
- a detailed, documented indication of how the assessment process will be undertaken

and what information (likelihood and consequence data) will be needed, what data are available and the knowledge gaps.

5.1.2.2 Scope of the assessment

The spatial and temporal scope of the ERA needs to be well defined. The scope can be defined by:

- Spatial scope – establishing the areal extent of the assessment. For coal seam gas and coal mining operations, this will include an assessment of risk close to the discharge point (near-field) and further downstream (far-field).
- Temporal scope – determining the timeframe over which the risks will be assessed. For large-scale developments, four timeframes are generally considered:
  - pre-development phase to establish some baseline of the condition of the key assets to be protected prior to the operation commencing
  - construction phase to assess the risk to the key assets during construction
  - operational phase to assess the risk to the key assets during operations
  - post-operation phase to assess the risk to the key assets after operations have ceased and closure of the operation has occurred.

5.1.2.3 Assessment endpoints

A set of assessment end-points are defined to provide a basis for deciding whether an ecological effect has occurred and on what part of the ecosystem. A wide range of biological indicators can be used as assessment end-points in ERAs. These include (MDBA 2011):

- health indicators – biological species or communities used to measure the impact of stressors or other disturbances (e.g. use of macroinvertebrates species-richness to assess the impact of pesticides)
- population indicators – species used to assess trends in the populations of other species (e.g. use of a particular fish species (e.g. Murray Cod) as a surrogate for the condition of the entire fish community)
- biodiversity indicators – the number of species from well-known taxonomic groups used as a surrogate for the number of species that occupy the same range, but are poorly known
- umbrella species – taxa whose presence indicates the size or type of habitat that should be protected (e.g. use of river red gum condition as a measure of floodplain forest ecosystem condition.

Additional to these biological indicators, habitat availability and quality are often used as an indicator in ERAs (Pollino et al. 2004). All the above biological indicators measure changes in the structure of the ecosystem. While it would be useful to also have assessment end-points related to ecosystem processes (e.g. primary production, metabolism), the general lack of both data and a specific understanding of ecosystem response to stressors often precludes the use of these ecosystem process indicators at this stage.

5.1.2.4 Conceptual models

Conceptual models are best described as a representation of the present understanding of a system. They are used in ERAs to show explicitly the links and pathways between stressors and assessment end-points. These are sometimes called cause-effect diagrams. An example is shown in Figure 6.

Conceptual models are particularly useful when defining the essential features of the problem, designing the structure and variables of quantitative ecological models, communicating the workings of a quantitative model to others, or when eliciting knowledge of local ecosystems during stakeholder meetings. Conceptual models benefit from having input from a range of stakeholders and experts.

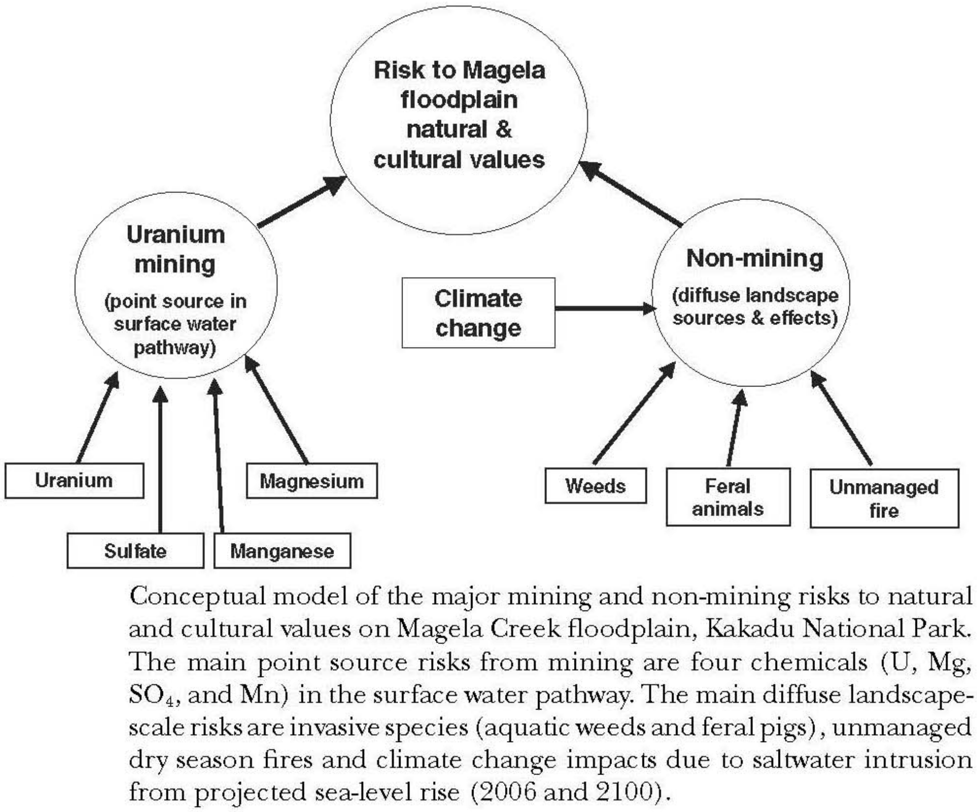

Figure 6 Example of a conceptual model describing the risks to the Magela floodplain from mining and non-mining threats (© Copyright, Bayliss et al. 2012).

5.1.2.5 Uncertainty

It generally becomes apparent early in most ERAs that the ecosystems being investigated are complex and that there are many knowledge gaps. This is often the case for ERAs that are catchment-based with multiple-stressors and multiple-endpoints and when quantifying the cause-effect relationships. These issues are often defined under the broad heading of 'uncertainties'.

Regan et al. (2002) and Burgman (2005) suggested that risk assessments should deal with five types of uncertainty:

1. systematic uncertainty – results from a lack of data or knowledge about a system
2. parameter uncertainty – from measurement error or natural variation. This is the type of

uncertainty most commonly considered. Science addresses this kind of uncertainty by using confidence intervals in statements (e.g. 'the mean size of the change is 54 with the 95 per cent confidence limits of 23')

3. structural uncertainty – where an inappropriate model for the system being studied is used. This highlights the need for well-considered conceptual models
4. shape uncertainty – uncertainty about the distribution of the data being considered
5. dependency – relates to possible correlations between parameters.

Semantic uncertainties, including ambiguous statements and vague definitions of concepts that permit borderline cases to occur, must also be kept to a minimum. An example of a linguistic ambiguity is 'there is a 70 per cent chance of rain' – this could be interpreted as rain during 70 per cent of the day or over 70 per cent of the area or a 70 per cent chance that it will rain at a particular point.

5.1.2.6 Stakeholder involvement

Most quantitative ERAs benefit from involving stakeholders in the problem formulation step and throughout the entire risk assessment process. The initial problem formulation step is particularly difficult in situations where there are multiple stressors and multiple ecological effects.

Contributions from key stakeholders can be critical to the identification of the key ecological assets likely to be at risk. Stakeholder engagement can also improve the likelihood of achieving broad agreement on the consequence criteria. It has been found that not taking into account the motivations of these important groups or the knowledge that they possess can undermine the acceptance of the assessment process (Cain 2001).

Adequate thought, effort and expertise will be needed when deciding which stakeholders are to be involved in the ERA. The knowledge and expertise that they can bring to the process and how that knowledge can be elicited should be considered. There are many well-tried and tested methods available for eliciting information from stakeholder and community groups (see Edelenbos & Klijn 2006; Burgman 2005).

Some of the challenges observed in stakeholder involvement include:

- taking notice of legitimate issues brought forward by stakeholders
- accounting for long-standing disagreements between 'competing' stakeholder groups
- including the necessary effort into informing and running the stakeholder workshops to ensure key groups are adequately engaged and not alienated or disillusioned with the process
- allowing the stakeholders to drive the identification of the issues or hazards that need to be considered.

5.1.2.7 Analysing the risks to the ecological values

The risk analysis step involves bringing together the likelihood and consequences of each adverse effect. The inherent frailties in subjective estimation of probabilities and consequences are well established (Suter 2007; Burgman 2005). All risk assessments should strive to be as quantitative as possible and address the inevitable limitations in ecological knowledge to improve the rigour of the ERA process (Pollino et al. 2012; Hart & Pollino 2008).

The purpose of the likelihood characterisation is to predict or measure the spatial and temporal distribution of the stressor(s) and the co-occurrence or contact with the ecological components of concern. In more quantitative risk assessments, some modelling or estimation of the stressor concentration distributions or extent of physical disturbance will be needed, particularly for future scenarios.

The purpose of the ecological effects characterisation is to identify and quantify the effects caused by the stressor(s) and, to the extent possible, evaluate cause-effect relationships. For toxic stressors, such as salinity, cyanide, heavy metals, pesticides and other toxic organic compounds, the availability of quantitative acute and chronic effect distributions is considerably more advanced (Ostrom & Wilhelmsen 2012; ANZECC/ARMCANZ 2000) than for other stressors or hazards, such as changes to flow regimes or physical changes to habitats.

A method for combining this likelihood and consequence information must also be chosen. This is undertaken via qualitative and quantitative methods, as described below.

Qualitative methods

A large number of ERAs use a qualitative or semi-qualitative risk matrix to combine the data (for example, BHP-Billiton 2009; Peters et al. 2009). Table 8 shows a typical qualitative risk matrix. Five levels of likelihood and consequence criteria are presented and then combined to provide an assessment of the risk. Qualitative ERAs can be useful for making a ‘first cut’ to separate the various risks into broad categories - high, moderate, low – and then identify the risks needing further analysis.

Table 8 Typical qualitative risk matrix (© Copyright, Hart et al. 2005).

Likelihood		**Consequence**				
		Insignificant (1)	Minor (2)	Moderate (3)	Major (4)	Catastrophic (5)
Almost Certain	(5)	5	10	15	20	25
Likely	(4)	4	8	12	16	20
Moderately Likely	(3)	3	6	9	12	15
Unlikely	(2)	2	4	6	8	10
Rare	(1)	1	2	3	4	5

Quantitative methods

There are many quantitative or semi-quantitative methods that have been used in ERAs (Suter 2007; Burgman 2005). Some of the commonly used quantitative methods are described below.

- **Risk quotients**

A risk quotient (RQ) is typically calculated by dividing an environmental exposure value, such as an expected maximum concentration of toxicant in the environment, by a toxicity end-point value, such as a well-known acute or chronic toxicity value or a water quality guideline value (Peterson 2006; ANZECC/ARMCANZ 2000). Risk quotients are generally regarded as highly conservative measures of the ratio of exposure to effect, and can be used by decision-makers to assess whether or not the values exceed some pre-determined level of concern. For example, if the RQ is greater than one, this suggests a high likelihood that toxic effects will occur. Risk quotients are often used to screen toxicants in situations where a large number of toxic stressors are likely to exist (Peterson 2006).

- **Statistical probabilistic risk assessment**

This approach is suited to assessing the risks to large numbers of taxa and translates directly to an expectation of biodiversity effect (Ostrom & Wilhelmsen 2012; Solomon et al. 2000). This approach firstly involves identifying the toxicant or stressor for which the risk assessment is to be conducted. Then sensitivity data for the stressor are collated for the species of interest to achieve a sensitivity distribution curve. Statistical probabilistic risk assessment information is available in a number of compilations (OECD 2012; USEPA 2012; ANZECC/ARMCANZ 2000). Data on the environmental levels of the stressor are then obtained, either from environmental monitoring or from modelling, to arrive at an occurrence distribution. These two data sets (i.e. sensitivity data for the stressor and data on the environmental levels of the stressor) can be viewed as the sensitivity effects and likelihood of occurrence data, respectively.

These two data distribution sets are generally converted to cumulative distributions, and risk is calculated as the proportion of taxa that will have their sensitivity value exceeded by a given percentage of environmental levels (or vice versa, i.e. the proportion of environmental levels that exceed the sensitivity value for a given percentage of taxa). The procedure is outlined diagrammatically in Figure 7.

The derivation of the trigger values for toxicants listed in the ANZECC/ARMCANZ (2000) guidelines is based on this method, with various levels of protection (99 per cent, 95 per cent, 90 per cent) obtained from the available species sensitivity distribution curves of chronic no-observed effects toxicity data. For example, a 95 per cent level of protection is protective of 95 per cent of species with 50 per cent confidence. There are now many examples in the literature where probabilistic risk assessment has been applied (Ostrom & Wilhelmsen 2012), including Australian examples for salinity (Webb & Hart 2004) and tebuthiuron (van Dam et al. 2004).

- **Process-based models**

The application of quantitative models in ERA is increasing, particularly for providing information on likelihood or exposure data for a range of possible scenarios. Although the deficiencies of qualitative risk analysis have been well documented, many of the tools to make this process more quantitative are poorly developed, inappropriate or full of hidden assumptions (Burgman 2005; Burgman 2001). This is the case when attempting to assess the ecological risks associated with contaminants generated from multiple sources within a catchment.

An increasing number of process-based biophysical models are now available that are reasonably capable in predicting the generation and transport of contaminants, such as salt, sediment, and nutrients. Examples include the eWater Source Model (eWater 2012a), SedNet (eWater 2012b), salt transport models and eutrophication models. However, these models still have three major deficiencies, in that they:

- are still not able to address multiple stressors in any systematic way
- rarely treat uncertainty explicitly
- rarely couple the contaminant with its ecological effect, particularly in downstream waterways, wetlands and estuaries.

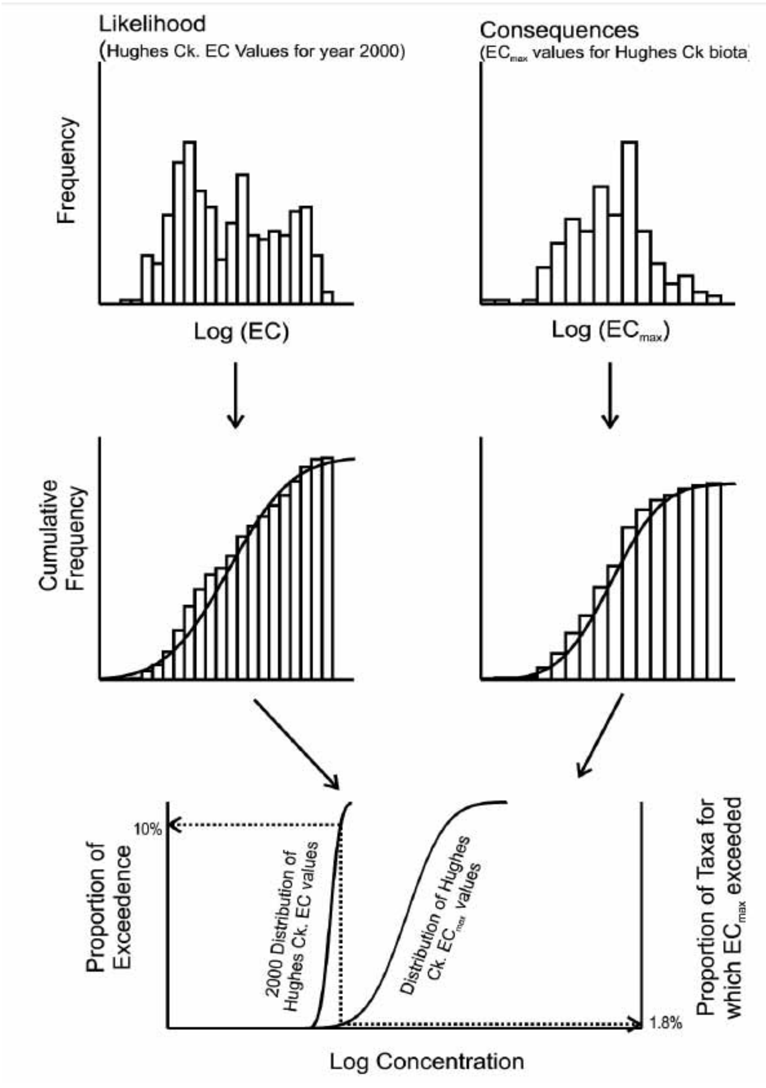

Figure 7 Example of the probabilistic risk assessment (PRA) method (© Copyright, Hart et al. 2003).

Note: The top panel shows frequency distribution for modelled EC (salinity) values for a single year at a single site and EC_{max} values for local aquatic flora and fauna. The middle panel shows the distributions converted to cumulative distributions and fitted with cumulative normal distribution functions. The bottom panel shows the two distribution functions superimposed on the same set of axes. The dotted line indicates that 10 per cent of EC estimates exceed the EC_{max} estimates for about 1.8 per cent of taxa for which an estimate of EC_{max} was available.

- **Causal probabilistic models**

Process-based modelling is likely to improve over time to better address deficiencies associated with multiple stressors and uncertainty. However, given the inherent complexity and lack of knowledge about many of the basic processes and relationships between stressors and biota, other types of models may offer more promise. Perhaps the most promising of them being alternative modelling approaches using Bayesian techniques (Pollino et al. 2012; Hart 2008; Reckhow 2003).

Bayesian models can assist with multiple stressor problems because they are able to incorporate information with high uncertainty, including poor or incomplete understanding of the system and cause-effect relationships. They can also include data outputs from other models, observational data and expert opinion. Prior probabilities can be updated as more information becomes available. Over the past 10 years, Bayesian Network (BN) models have been successfully used for a wide range of risk applications.

Uusitalo (2007) noted that BN models have the following attributes:

- an ability to handle missing data
- an excellent tool for expert elicitation
- allow combination of different forms of knowledge, from expert opinion and intuition to quantitative data
- facilitate learning about causal relationships between variables
- show good prediction accuracy even with rather small sample sizes
- can be easily combined with decision analytic tools to aid management.

Hart et al. (2007) noted that BN models also have some limitations, including:

- conditional probabilities are expressed as discrete forms so that models can be solved analytically; in contract to Bayesian hierarchical models, which use Monte Carlo methods, where distributions are estimated by simulation
- inability to easily incorporate feedback loops or temporal effects in models
- the difficulties associated with eliciting expert knowledge and evaluating models built largely on expert opinion.

5.1.2.8 Characterising the risks and decision-making

The technical details of the risk analyses need to be accessible to decision-makers and broader stakeholders. Uncertainties and assumptions require careful and transparent documentation. Here the likelihood and effects profiles are integrated to provide an estimate of the level of risk. In many applications, it is possible only to make qualitative ratings of these two components of risk. However, assessments should be as quantitative as possible (Hart et al. 2006; Burgman 2005), including those for coal seam gas and coal mining.

ERAs for proposed coal seam gas or large coal mining should also include an assessment of the risk for a number of scenarios that are aimed at minimising risks to aquatic ecosystems. The scenarios may include different mitigation strategies, along with other major factors that influence outcomes, such as climate variability. The best management strategy will be the one that results in the most effective minimisation of ecological risks, is cost-effective and acceptable to stakeholders. Multi-criteria assessment and Bayesian modelling may assist with this process (Ostrom & Wilhelmsen 2012; Suter 2007; Burgman 2005).

The final assessment of the level of risk should also include an estimate of the uncertainty in the predictions. For example, the risk assessment might predict four blue-green algal blooms over the coming summer. However, this may be treated differently if it was known that the upper and lower bounds to this prediction were three and six blooms, or one and ten blooms.

The final ERA should contain a prioritisation of the ecological risks, a summary of the assumptions used, the uncertainties, the strengths and weaknesses of the analyses and the mitigation strategies for priority ecological risks. Before completion, the key findings of the risk assessment should be communicated back to stakeholders for review and comment.

5.1.2.9 Managing the risks

Ultimately, the ERA will inform the development of a risk management plan to provide recommendations on managing or mitigating all high or unacceptable risks. The risk management plan should include a robust program to monitor progress, to ensure that the strategies are appropriate, via a review and feedback process. The plan should be iterative and adaptive so that it can incorporate new information; it is recognised that uncertainties are inherent in any ecological risk assessment and a capacity to modify plans as information becomes available through monitoring and/or research is necessary (Eberhard et al. 2009; Gregory et al. 2006).

5.1.3 Application of ERA frameworks in Australia

5.1.3.1 Example natural resource management risk assessments

Quinn et al. (2013) developed a BN to guide decisions on water abstraction and irrigation-driven land use intensification in the Hurunui River catchment, New Zealand. This model examined the combined effects of different irrigation water sources and four land development scenarios with and without a suite of on-farm mitigation measures. They covered on-ground and surface water quality, key socio-economic values of farm earnings, employment and contribution to regional gross domestic production, and aquatic values of salmon, birds, waterscape, contact recreation, periphyton and invertebrates. The model predicted high farm earnings, jobs and regional GDP with 150 per cent increase in irrigated area, but a range of positive and negative aquatic environmental outcomes depending on the location of storage dams and application of on-farm mitigations. This BN of a complex system enhanced the ability to include aquatic values alongside economic and social values in land use and water resource planning and decision-making.

Chan et al. (2012) reported the development and application of two BN models to assist decision-making on the environmental flows required to maintain the ecological health of the Daly River, Northern Territory. The abundances of barramundi (*Lates calcarifer*) and sooty grunter (*Hephaestus fuliginosus*) were chosen as the ecological end-point for the models, which linked dry season flows to key aspects of the biology of each species. The models showed that if current extraction entitlements were fully used there would be significant impacts on the populations of these two fish species.

Bayliss et al. (2012) reported a quantitative ERA for the Magela Creek floodplain in the Northern Territory that combined both point source mining risks (contamination) from the Ranger uranium mine and diffuse non-mining landscape-scale risks. A high level of protection for the biodiversity of aquatic ecosystems was the end-point. The potential for mine operations to contaminate surface waterways was assessed for the four key mine-associated solutes of uranium, manganese, magnesium and sulphate. The non-mining landscape-scale risks were assessed for weeds, feral pig damage, unmanaged dry season fire and saltwater intrusion from potential sea-level rise due to climate change. The results reported the non-mining landscape-scale risks to be several orders of magnitude greater than the risks from mine water contaminants.

Pollino and Hart (2008) and Pollino et al. (2008) developed a BN to predict the effects of the Ok Tedi mining operation in Papua New Guinea on the fish populations in local rivers. The combined effects of heavy metal toxicity due to Acid Rock Drainage (ARD) and habitat smothering from waste rock disposal in the rivers were modelled. A series of management scenarios to reduce the sulphur content of the tailings added to the rivers, with a subsequent reduction in ARD production, were also modelled.

van Dam et al. (2004) reported an assessment of the ecological risks of the herbicide tebuthiuron to freshwater fauna and flora of northern Australia's tropical wetlands using a quantitative approach.

5.1.3.2 Example coal seam gas and large coal mine risk assessments

Most EISs for new and expanding coal seam gas and large coal mine developments include an assessment of the potential impacts on environmental values, but the level of risk assessment varies. More specifically, the extent of risk assessments related to the release of co-produced water to the environment vary depending on the expected volume of co-produced water likely to be generated and water management options. For example, projects centred on the Surat and Bowen Basins, where large volumes of co-produced water may be generated, tend to have a more comprehensive approach to risk assessment (e.g. APLNG 2010b; URS 2009a) compared to projects in areas where lower volumes of co-produced water are generated, for example the Camden expansion project in the Sydney Basin (AECOM 2010). More specific details on the various approaches adopted by the above studies are provided below.

URS (2009a) reported an assessment of the ecological risks associated with the discharge to surface waters of co-produced water from coal seam gas operations in the Surat and Bowen Basins in Queensland for Santos' Gladstone Liquefied Natural Gas (GLNG) Project. They followed a risk assessment approach based on that described by Webb and Hart (2004) and consistent with the preferred approach described in the section 5.1.2. They identified salinity, temperature and changed flow regime as key stressors and developed a number of conceptual models to help identify specific end-points for more detailed assessment. End-points included macroinvertebrate community composition, fish community composition, diatom assemblages, rates of ecosystem primary production and respiration, and riparian vegetation responses. Through the development of conceptual models and assessment of available data, macroinvertebrates assemblage composition and response was chosen as the critical end-point most likely to respond to the identified stressors. A combination of qualitative and quantitative approaches were considered. Quantitative assessments were made of the salinity impacts on aquatic biota and critical thresholds were established to allow an assessment of potential impact. The risk assessment concluded that untreated co-produced water represented a risk to macroinvertebrates, due mostly to elevated salinity, and that treatment was required to reduce the risk.

The Australian Pacific LNG project EIS (APLNG 2010b) associated with the proposed coal seam gas development in the Condamine-Balonne and Dawson River catchments, Queensland, reported a qualitative assessment of the risks to aquatic habitat, flora and fauna, water quality and fluvial geomorphology. They used a combination of likelihood and consequence to identify risks similar to the example provided in Table 8 above. Conceptual models were not developed to help clarify the relationship between stressors and ecological end-points; rather, risk was based on descriptive consequence criteria or standard water quality guidelines for particular water quality variables and it was assumed that if relevant guidelines were met then ecological values would be protected. However, modelling of surface water flows in the Condamine River was undertaken to assess the changes in river flow that could be expected under a range of co-produced water discharge scenarios. This modelling was used to help develop a proposed discharge scenario to minimise changes to the natural pattern of stream flow.

AECOM (2010) reported a qualitative assessment of the risks to surface waters and flora and fauna associated with the proposed northern expansion of the Camden Gas Project. The estimated volume of co-produced water generated in this proposal was very small and no discharge to waterways was anticipated.

Most assessments of risk of co-produced water to environmental values assume that if relevant water quality guidelines are met then risks to environmental values will be low. As discussed in Chapter 0, this approach may not acknowledge the risks associated with effect of the complex mixture of chemicals on the environment and may not characterise risks where guidelines are not available for some water quality variables.

5.1.3.3 Lessons learned

A number of lessons emerge from these quantitative ERA applications, including:

- Good initial planning is essential and ideally would involve key stakeholders.
- The development of conceptual models to identify the main cause-effect relationships between the key stressors and the key components of the ecological assets to be protected is an essential element of the process. This is best done with the help of key stakeholders.
- Identification of key stressors or hazards is generally relatively straight forward. Much is known about the stressors likely to be present in co-produced water from coal seam gas and coal mining (RPS. 2011; Takahashi et al., 2011a).
- The key ecological assets and environmental values associated with the systems likely to receive co-produced water discharges, such as streams and wetlands, are broadly well-known but the species at highest risk may vary (Takahashi et al., 2011a).
- Most ecological risk assessments related to natural resource management activities, including coal seam gas and coal mining, are qualitative and based largely on the Standards Australia and Standards New Zealand risk assessment guidelines (AS/NZS, 2009). Quantitative assessments generally relate to an assessment of water quality impacts against standard guidelines (e.g. ANZECC/ARMCANZ, 2000) rather than a specific assessment based on local data.
- The potential effect of the complex mixture of chemicals on the environment is rarely considered in ERAs, with little toxicity testing undertaken. An exception to this is the URS (2009a) risk assessment of the Gladstone LNG project, which provides a comprehensive example for a coal seam gas-related project. It followed the ERA framework recommended in this review and includes quantitative components using

relevant local data.

- Quantitative ERAs involving the development and use of quantitative models, such as BN, to link stressors to ecological outcomes are definitely preferable to qualitative risk assessments, but obviously will depend upon the level of available knowledge and budget. However, qualitative assessments are often accepted when a quantitative assessment could not be produced.
- Given the high profile of new coal seam gas and coal mining developments, it is recommend that quantitative ERAs be used to assess the risks from the discharge and use of co-produced water.

5.1.4 Summary

In summary, the ERA process seeks to:

- identify the key ecological issues and key stressors
- identify the linkages between the key stressor drivers and each ecological consequence as a conceptual or quantitative ecological model, and from this provide information on which drivers are most sensitive to management or mitigation
- assess the risks associated with each issue as quantitatively as possible and show the likelihood of occurrence and the consequences
- assess the risks for a number of different management or mitigation strategies
- provide quantitative information for making robust and credible decisions on the risks to downstream aquatic ecosystems
- provide the necessary information to underpin the development of an adaptive risk management plan, including a performance monitoring and evaluation program
- identify and where possible quantify all major uncertainties so the decision-maker knows what confidence can be placed on the information.

5.2 Cumulative risk assessment frameworks

An ERA for addressing a single project will not necessarily assess the impact of a number of such projects in sufficient proximity to cause a cumulative impact. The potential for cumulative environmental effects to be associated with the large-scale extraction of coal seam gas is now being recognised (NWC 2012; Geoscience Australia & Habermehl 2010). Cumulative risk assessment is quite common in the human toxicology field (Boobisa et al. 2008), but there are few examples where cumulative ERAs have been undertaken in Australia. Howe (2011) developed a mining risk framework for the National Water Commission (NWC), which offers a risk-based approach to managing the cumulative groundwater-affecting activities of mine operators. The framework is supported by tools comprising a Groundwater and Resource Information for Development Database (GRIDD), Multi-Mine Water Accounts Tool and Cumulative Impacts Assessment Tool (CIAT). The USEPA (2003) has also developed a framework for cumulative risk assessments.

A semi-quantitative risk assessment of the cumulative impact on water quality of coal mining activities in the Fitzroy River Basin was undertaken by DEHP (2009b). At the time of this study there were 38 operating coal mines in the Fitzroy River Basin. The DEHP study focused on the risk to aquatic organisms, irrigated crops and drinking water from the increased salinity levels resulting from the discharge of mine water. Their analysis identified that six mines were the highest contributors to potential cumulative impacts, with five of these located in the northern Isaac-Connors sub-catchment and the other in the Nogoa sub-catchment.

Geoscience Australia and Habermehl (2010) examined three proposed coal seam gas extraction activities in the Surat and Bowen Basins, Queensland. They identified the potential for cumulative impacts on a number of surface and groundwater characteristics. They also reported that access to commercial (non-public) data needed to feed into the cumulative impact assessment was at time difficult to obtain. They recommended that a regional-scale, multi-state and multi-layer model of the cumulative effects of multiple developments be developed. They also recommended that this be accompanied by a regional-scale monitoring and mitigation approach to assess and manage the impacts. While this recommendation was focused on groundwater effects, the principle of developing regional-scale models for assessing the cumulative effects of multiple developments would also apply to surface water resources.

Current ‘best practice’ ERA frameworks could accommodate a cumulative risks assessment of coal seam gas and coal mining expansions. However, it would be important to consider regional-scale cumulative effects during the problem formulation phase, and include potential stressors and hazards from the other developments in the risk assessment. Quantification of cumulative effects of multiple developments will require the development of regional-scale models. Initially, these regional-scale models could build on existing hydrological-water quality models, such as Integrated Quantity Quality Model (IQQM). Preferably, they would also include groundwater and surface water interactions and be able to link with ecosystem response models that relate flow to ecosystem response. If conducted properly, existing ‘best practice’ ERA processes are robust enough to be able to capture the cumulative risks to surface waters from multiple developments.

5.3 Risk assessment frameworks for other endpoints

After the end-points and main stressors such as salinity, sodium and toxicants are identified, the main task will be to determine, by estimation or modelling, the distribution of concentrations of the key stressors and compare them against the levels at which adverse effect are likely to occur. It will be important to capture possible spikes in the stressor concentrations. This assessment is often done using a simpler version of the probabilistic risk assessment method previously outlined in this review. The distribution of stressor concentrations, or exposure, is compared with a single trigger value or ‘safe’ level, rather than a distribution of effects on different organisms. The trigger values are normally obtained from water quality guidelines such as ANZECC/ARMCANZ (2000). In summary, assessment of the possible risks to other environmental values or beneficial uses, in addition to aquatic ecosystems, can be easily accommodated within the current ‘best practice’ ERA process.

5.4 Suitability of the ERA process for co-produced water

Given the likely scale and number of unknowns associated with new coal seam gas and coal mining projects in Australia, assessments of the ecological risks associated with the discharge of co-produced water from these operations to waterways should:

- address as a minimum the key threats from (a) increased salinity, (b) increased toxicity, and (c) changes in the flow regime, particularly in ephemeral streams
- address as a minimum the risks to key ecological assets including: (a) biodiversity, (b)

fish communities, (c) macroinvertebrate communities and (d) riparian vegetation

- address possible cumulative risks due to other existing or planned coal seam gas or coal mining developments
- use or develop appropriate modelling techniques, particularly those that quantify the relationships between key threats and key ecological indicators, to ensure the risk assessments are as quantitative as possible.

6 Industry practice

Coal seam gas generated more than 14.2 GL of co-produced water in Australia during the 2008 to 2009 financial year (RPS 2011). Almost all was generated in the Bowen and Surat Basins with less than one per cent produced in the Sydney Basin (RPS 2011). There are several significant coal seam gas and large coal mining developments being planned, predominantly in Queensland and New South Wales, and exploration is occurring in other areas such as the Gippsland Basin in Victoria (see Figure 2 for a map of production areas).

This chapter provides a review of the situation, drawing on impact assessment documentation that was publicly available at the time of writing and the water management options that the various proponents proposed for future co-produced water management.

6.1 Environmental impact assessments

Over the past five years, a number of EISs have been prepared for projects that incorporate coal seam gas extraction or large coal mine development. Projects include the Gladstone LNG Project (Santos Ltd 2009), the Australia Pacific LNG Project (APLNG 2010b), the Queensland Curtis LNG Project (QGC Limited 2009) and the Surat Gas project (Arrow Energy 2012b). An EIS is usually carried out at the Feasibility Stage of a project, where the detailed technical information related to the project may not yet be finalised.

An EIS is usually carried out at the feasibility stage of a project, where the detailed technical information related to the project may not yet be finalised. An EIS is intended to describe the potential impacts of the project on the environment, along with any management and mitigation measures to reduce impacts to an acceptable level. The technical information presented is usually limited and often represents a high-level assessment of the project risks and impacts. In some cases, limitations of the technical assessments may be due to a lack of information on the long-term impacts of the proposed management and mitigation measures. In other cases, there is significant uncertainty about the volumes and quality of water being produced.

The technical assessments are likely to focus on construction risks and site-specific operational risks, such as stormwater management, erosion control or flood management rather than the long-term risks to watercourses and aquatic ecosystems posed by the management of co-produced water.

Assessments of water quality impacts usually involve comparison of the treated co-produced water quality against the ANZECC/ARMCANZ (2000) guidelines (APLNG 2010a; QGC Limited 2009), and in some cases against the *Murray Darling Basin salinity management strategy* (URS 2009a). A commitment may be made to develop suitable permeate discharge concentrations, based on 90 to 95 per cent species protection levels, in accordance with ANZECC/ARMCANZ (2000) guidelines and in consultation with relevant regulators (APLNG 2010a). However, this approach may not adequately protect the receiving environment. Background condition of waterways is highly variable across regions and most guidelines have been developed for perennial waterways and may not be suitable for ephemeral waterways, which dominate in regions where most proposed coal seam gas operations occur. Furthermore, issues regarding the potential effects of a complex mixture of contaminants on the receiving environment are not generally considered through the application of toxicity testing.

While some risks have been acknowledged, such as the risk to microcrustacean populations from low calcium concentrations in treated water, many are dismissed with little or no mitigation measures proposed and a comment that it will be monitored (APLNG 2010a). Monitoring itself is not a mitigation measure. Monitoring provides information on whether an impact has occurred, or if actual mitigation measures have been successful, and if monitoring reveals that an impact has occurred could be too late to undertake mitigation.

Projects have generally proposed a range of water management options that may or may not operate at the same time. This is due to the large volumes of water expected and uncertainty of these options. For example, irrigation demand will vary on a seasonal basis, requiring a supplementary beneficial use or disposal option to manage the water that is surplus to the irrigation requirement.

The EIS reports that were reviewed identified a range of constraints to the management of co-produced water. While many of the constraints to beneficial use or disposal are due to environmental concerns, the importance of geography, water quality and water quantity may be less clearly identified (RPS 2011). Table 9 summarises key constraints to the management of co-produced water in Queensland; however, it is clear that the constraints are similar across Australia, although the volumes of water being produced vary significantly on a regional basis.

Table 9 Key constraints to coal seam gas water management in Queensland (© Copyright, RPS 2011).

Key Constraint	Description
Regulatory framework	Restrictions to management options imposed by legislation.
Geography	Production areas are often remote, hence the distance to a beneficial user or disposal point may determine feasibility.
Water quality	Due to the poor quality of co-produced water, treatment is usually required, which introduces potential economic, technology and environmental challenges.
Water quantity	Includes the quantity of water that can be taken for beneficial uses, the stability in demand and the level of uncertainty around projected quantities, which may affect a producer’s ability to guarantee supply.
Economic	Costs associated with management options vary and may influence the feasibility of particular management techniques.
Environment	Includes the natural, social and economic environments and the potential short and long term effects associated with different management options.
Technology	Refers to the proven capability of water management and treatment technologies.

The management of co-produced water is changing rapidly, influenced by legislation and by financial, environmental and practical considerations. Many proposals have undergone changes to their proposed water management during or after the preparation and review of the EIS, with many now testing aquifer injection as one of the primary water management options in preference to a discharge to waterways (Table 10).

Table 10 Recent coal seam gas projects and their proposed disposal options.

Project name	Proponent	Date EIS published	Management options proposed in the EIS	Changes to the project since the EIS
Surat Gas project	Arrow Energy	2012	Treated water for agricultural purposes, potable supply or industrial use. Disposal of water to watercourses or ocean outfall (less-preferred).	Aquifer reinjection currently being tested.
Australia Pacific LNG Project (APLNG Project)	Origin Energy and ConocoPhillips	2010	Treated water for agricultural purposes or discharge to a major watercourse.	Undertaking further investigations into beneficial uses, including aquifer reinjection.
Gladstone LNG Project (GLNG Project)	Santos Limited and Petronas	2009	Dependent on location and water quality: Roma Field - potable, industrial re-use and treated water for irrigation. Fairview Field - treated and untreated water for irrigation. Arcadia Valley Field – treated water for irrigation.	Aquifer reinjection currently being tested.
Queensland Curtis LNG Project (QCLNG Project)	QGC Limited	2009	Treated water for agricultural purposes, potable supply or industrial/mining use. Disposal of water to evaporation ponds (short to medium term solution). Further investigations into forestry, agriculture, reinjection, and industrial and community use underway.	No longer considering forestry. QGC has entered into an agreement with SunWater to supply treated water for agricultural, industrial and community use. SunWater will manage the sale and distribution of this water.

6.1.1 *Water management options being proposed*

Water management options include beneficial use and disposal options, with beneficial use emerging as the preferred choice from a regulatory viewpoint (e.g. DEHP 2012; DEHP 2010a). Beneficial use includes aquifer injection, agricultural use, industrial use, urban use and uses beneficial to the environment. Disposal options include discharge to watercourses, ocean outfall and evaporation ponds (although generally these are no longer approved, except where no other option is available). A summary of water management options is presented in Table 11, with an analysis of their advantages, disadvantages and suitability.

Given that the quality of co-produced water may vary from nearly potable to highly saline and may include a range of heavy metals, its use or disposal is limited unless it is treated to an appropriate level. The current trend in treatment methodology is moving towards reverse osmosis (Nghiem et al. 2011). It is assumed that the treated water would require 'dosing' after treatment, as reverse osmosis can produce water that is too clean compared to natural waters. However, detailed information on post-treatment dosing is difficult to find, as is any information about the risks. Instead, a view has been taken that once the co-produced water is treated it no longer poses any risk to the environment.

6.1.1.1 Queensland

Co-produced water and brine in Queensland are classified as waste under the *Environmental Protection Act 1994* and must be disposed of in accordance with the specifications of the regulatory authority, or beneficially used. The preferred management option in Queensland for coal seam gas operations is that the co-produced water is beneficially used. Options include injecting the water into depleted aquifers for recharge purposes, substitution for an existing water entitlement, supplementary water for existing irrigation schemes, new irrigation use with a focus on sustainable irrigation projects, livestock watering, urban and industrial water supplies, coal washing and dust suppression, and release to the environment in a manner that improves local environmental values (DEHP 2012; Swayne 2012). The intent is that disposal to watercourses will only be considered for residual portions of coal seam gas water where there is no feasible beneficial use and where the disposal option will not adversely affect environmental values.

Evaporation dams are being phased out as an approved water disposal method in coal seam gas extraction operations. They will only be considered for a new coal seam gas operation where the operator can demonstrate that the evaporation dam is for the purpose of water produced during coal seam gas exploration or production testing or, based on an assessment of best practice environmental management, that there is no feasible alternative to the evaporation dam for managing the co-produced water. The hierarchy for brine and solid salts management is (DEHP 2012):

- brine or salt residues must be treated to create useable products wherever feasible
- the disposal of brine and salt must only be considered after a feasibility assessment has determined that that there are no reasonable options to minimise the volume of waste for disposal
- the options for the disposal of salt include injecting brine underground and disposing to a regulated waste facility.

Table 11 Summary of water management options in Australia.

Option	Advantages	Disadvantages	Suitability	Current practice
Underground injection				
Into coal seam	Repressurises the coal seam	May impact gas flow if reinjected into producing coal seams. Risk of reinjected water moving to surrounding aquifers.	More suited to coal seams that are no longer producing. Dependent on ability to accept water, as dewatered coal seams may experience a compaction event.	Feasibility studies have been carried out and trials are now underway.
Into aquifer	Recharges depleted aquifers	Poor quality water could contaminate aquifer. Costs of treating water may make option unviable.	Appropriate only if water quality is matched to the quality of the receiving aquifer. If treatment is required this may limit the economic suitability.	Feasibility studies have been carried out and trials are now underway.
Storage				
Containment ponds	Short-term option	Relatively large land area required. Risk of overtopping onto land or water. Risk of seeping into land or water. No beneficial use.	Suitable for short term requirements or part of a wider water management plan. Lining of ponds required to prevent seepage. Sufficient freeboard needed to prevent overtopping.	In use, as part of overall site water management.
Evaporation ponds	Short-term option	Substantial land area required. Risk of overtopping onto land or water. Risk of seeping into land or water. No beneficial use.	Limited due to large surface areas required. Disposal of brine and other waste material required.	Currently in use, but no longer being approved in New South Wales and Queensland unless the proponent demonstrates that no other disposal option is suitable.

Option	Advantages	Disadvantages	Suitability	Current practice
Infiltration basins	Allows infiltration of co-produced water to near surface aquifers	No beneficial use. Poor quality water could contaminate aquifer.	Suitable for water with a low hydrocarbon content (10 mg/L).	In use. For example, Cooper-Eromanga Basin (South Australia) and in the Amadeus Basin (Northern Territory) (RPS 2011).
Off-site disposal facility	Risk partially transfers to licensed operator of disposal facility	Transportation costs may be large. No beneficial use, unless the water is treated and sold on.	Suitable depending on proximity to production areas and economics.	In use. For example, AGL Camden Gas Project (AGL 2012).
Agricultural use				
Livestock watering	Beneficial use for regional industry	May affect animal health and production if water quality is not suitable.	Suitable for certain livestock depending on water quality and proximity to production areas.	In use. For example, Arrow Energy provides co-produced water to local beef cattle feedlots (Arrow Energy 2012a; RPS 2011).
Irrigation	Beneficial use for regional industry	May affect soil structure and crop yield.	Dependent on water quality. Treatment costs may make the water too expensive for farmers to use.	In use (RPS 2011).
Industrial use				
Coal mine use	Beneficial use (e.g. dust suppression, truck washing, haul and pit road water) Transport costs may be shared	Water transportation costs to coal operations.	Suitable depending on proximity to production areas and economics. Opportunity to share costs with coal operator.	In use. For example, Arrow Energy provides untreated co-produced water to the Wilkie Creek coal mine for its coal washing plant (Arrow Energy 2012a; RPS 2011).
Water cooling tower	Beneficial use for regional industry Transport costs may be shared	May require water treatment or capital expenditure for infrastructure conversion to accommodate lower water quality.	Suitable depending on proximity to production areas and economics.	In use. For example, at the QGC Condamine Power station co-produced water is used for cooling and steam production (RPS 2011).

Option	Advantages	Disadvantages	Suitability	Current practice
Fire protection	Beneficial use for regional communities	Requires storage facilities close to regional townships.	Suitable depending on proximity to coal seam gas production areas and economics.	
Urban use (town water supply)	Beneficial use for local communities	Treatment required. Limited longevity of supply (20 years) may offset economic viability of investment.	Suitable depending on proximity to coal seam gas production areas and economics.	Proposed. For example, Chinchilla Weir and Dalby TWS.
Surface waters	Provides increased flows for waterways suffering depleting water flows	Potential erosion of banks. Difficult to match to natural flow rates. May contaminate soil, water course and ecology if not treated to an appropriate level.	Limited due to sensitive nature of surface water systems.	In use. For example, the Fairview operation in the Bowen Basin (RPS 2011; URS 2009a). Historical approvals have allowed a constant discharge to water courses. More recent assessments have considered a variable discharge, in order to maintain natural flow variability.
Ocean outfall	Almost unlimited capacity to receive water (depending on the quality)	Poor quality water could contaminate the marine ecosystem. Costs of treating water may make option unviable. Transportation costs may be large. No beneficial use.	Suitable depending on proximity to coastline and economics. Dependent on water quality.	Under consideration as an emergency or alternative disposal option. Evaluation of this method of disposal is currently very limited.

Development of coal seam gas is occurring primarily in the Bowen and Surat Basins, where volumes of co-produced water are high relative to the level of gas production (RPS 2011). Large coal mines are also located in this area. A water demand analysis undertaken in 2010 in south-west Queensland noted that non-urban demand for water equalled or exceeded supply throughout the region, with the exception the Maranoa-Balonne subregion (Psi-Delta & MWH 2010). Urban water demand was also high. However, demand for co-produced water is expected to be limited due to issues of quality, cost and timing. Generally, the quality of co-produced water is unsuitable for direct use and agricultural industries may not be able to afford the high treatment and transport costs. There is also limited capacity for many industries to take significant volumes within the various project timeframes, although there are exceptions.

Co-produced water could provide a transitional water supply to irrigators, but its reliability over the longer term is a drawback to adoption. Psi-Delta and MWH (2010) noted that the demand for co-produced water could be increased through the development of commercial arrangements and distribution schemes, used to facilitate use at the local and regional scale. This is being developed through arrangements such as the Woleebee Creek to Glebe Weir pipeline, which is discussed in more detail below.

6.1.1.2 New South Wales

Approved disposal methods in New South Wales include treating the co-produced water to a high standard and allowing beneficial use for agriculture or drinking water supplies; injecting treated water into aquifers; or beneficial re-use of the water ‘as is’, for example, water for livestock (NSW Department of Planning 2012). Disposal through evaporation in ponds has been banned in New South Wales since July 2011; however, in some cases temporary holding ponds or dams may be required for treatment processes.

Volumes of produced water are expected to be lower in New South Wales than in Queensland and disposal is therefore seen as less problematic. There are three operating coal seam gas facilities in New South Wales (two near Narrabri and one in Camden) with five further projects proposed (City of Sydney 2011). The largest is the AGL facility in Camden and co-produced water at this site has been managed through beneficial use for industry, after treatment to a suitable quality. The volume is up to 2 ML per year (AGL 2012). However, there are many sites awaiting approvals and operations in New South Wales are expected to rapidly increase.

6.1.1.3 Other States

Coal seam gas projects in other states are largely in the exploration phase, although there are several operational facilities. RPS (2011) reports that in the Cooper-Eromanga Basin (South Australia) and the Amadeus Basin (Northern Territory) the disposal of co-produced water is predominantly via evaporation ponds, although infiltration basins are used when necessary. Consideration is also being given to reuse options in the Cooper-Eromanga Basin.

6.1.2 Associated projects

Some recent projects are opting to transfer co-produced water to an intermediary, who will then manage the sale and distribution of the water directly to users. This is seen as a logical move for energy producers, who do not consider water supply and distribution as core business. Projects include the Woleebee Creek to Glebe Weir pipeline project, the Kenya to Chinchilla Weir pipeline and the Camden Gas Project water management scheme (AGL 2012; SunWater 2012; SunWater 2011).

The Woleebee Creek to Glebe Weir pipeline project proposes to transport treated coal seam gas water from the QGC LNG project site to Glebe Weir. The water is intended for beneficial use by mining and irrigation customers along the pipeline route and within the Dawson Valley Water Supply Scheme (DVWSS). The project has been proposed by SunWater Pty Ltd, Queensland's largest regional bulk water supplier, who will be responsible for the construction and operation of the pipeline, as well as the management of the permeate discharge and its sale to end users. The 120 km pipeline, once operational, will transport treated coal seam gas water from the QGC water treatment site at Woleebee Creek to Glebe Weir, the headwater weir of the DVWSS.

Similarly, the Kenya to Chinchilla pipeline project involves the transportation of treated coal seam gas water from the QGC water treatment site at Kenya to Chinchilla Weir. The water is intended for beneficial use by agricultural customers and as a supplement to the Chinchilla town water supply. The project is also proposed by SunWater Pty Ltd, who will be responsible for the construction and operation of the pipeline, as well as the management of the permeate discharge and its sale to end users.

Both of these projects involve the discharge of treated permeate into a regulated watercourse. The quality and quantity of discharged treated coal seam gas water will be managed by a 'Site-specific' Environmental Authority under the *Environmental Protection Act 1994* (Qld) and an accompanying Coal Seam Gas Water Management Plan (CWMP) (DEHP 2011a). Documents are not yet publicly available detailing the type of environmental investigations undertaken to achieve the Environmental Authority or the content of the CWMP; however, it is generally understood that a "detailed assessment of the environmental impacts and sustainability of the proposed use in the receiving environment" would be required (DEHP 2010b). Information is also not available regarding the impacts they may have or the risks to ecosystems.

Co-produced water is collected by Worth Recycling at the Camden Gas Project and taken to a water treatment and recycling facility in South Windsor, where it is mixed with other waste water and treated via membrane filtration and microbial systems (AGL 2012). The resulting water is used for a variety of industrial purposes. Unlike the two previous examples, AGL does not have to treat the water prior to its collection. This disposal option is feasible due to the low volume of co-produced water generated and the project being located near a large metropolitan region with a high demand for industrial water. This option would not be available to the majority of coal seam gas projects located in regional areas.

6.2 Overseas studies

Coal seam gas is often referred to as Coal Bed Methane (CBM) in overseas literature. The largest resources are in the former Soviet Union, Canada, China and the US. These resource areas are relatively undeveloped at the moment, with the highest levels of development occurring in the US and Canada (RPS 2011).

The ratio of co-produced water to energy produced varies between areas, as shown in Table 12. Production ratios vary significantly between Australian Basins (1 to 192 ML/PJ). In the Canadian Alberta Plains production ratios are relatively low (0 to 30 ML/PJ); in the US Powder River Basin production ratios are relatively high (245 ML/PJ). These ratios, as well as the geographic, industrial and environmental concerns of the regions, influence the way that co-produced water is managed.

Table 12 Co-produced water-energy ratio by Basin for coal seam gas production (© Copyright, RPS 2011).

Basin	Co-produced water-energy ratio (ML/PJ)
Bowen Basin (Qld)	50
Surat Basin (Qld)	192
Sydney Basin (NSW)	1
Gunnedah Basin (NSW)	n/a
Powder River Basin (Wyoming, US)	245
Alberta Plains (Alberta, Canada)	0-30

Note: Water production data available for the Sydney Basin refers to water removed from site for disposal and does not include the volume of water reused on site.

Insufficient data were available for the Gunnedah Basin.

Powder River Basin data are for the 2003 calendar year.

Alberta Plains data are for the 2008-9 financial year.

6.2.1 United States

Management of co-produced water within the US can be broadly categorised into either storage and disposal options, or beneficial use options (National Research Council 2010). Factors which determine the management of co-produced water include:

- volumes available and reliability of supply over time
- water quality and cost of treatment
- location of water in relation to beneficial use locations
- legal and regulatory considerations, including concerns over liability
- existing infrastructure for storage, disposal or transportation
- financial considerations of the various options.

Management of co-produced water varies considerably and is driven by economic factors (National Research Council 2010). The majority of co-produced water is managed via storage and disposal with very little treated for beneficial use. Storage and disposal options include aquifer reinjection, discharge to watercourses, surface impoundments for evaporation or infiltration and land-applied disposal through water spreading. Beneficial uses included surface irrigation, subsurface drip irrigation, livestock and wildlife water supplies, in-stream flow and wetland augmentation, industrial and municipal use opportunities with a small amount used for dust control (National Research Council 2010).

In the six western US states considered in the National Research Council report (National Research Council 2010), less than five per cent of co-produced water is being used for beneficial irrigation. Irrigation use depends on the quality of water, soil suitability and plant tolerance to salinity. It is generally undertaken as ‘managed irrigation’, which includes the application of soil amendments to avoid the deterioration of soil structure.

Storage and direct discharge to surface water are the main methods of disposal in the Powder River Basin, which produces large volumes of low salinity water. The San Juan, Uinta, New Mexico portion of the Raton and the Picceance Basins produce small volumes of highly saline water. Nearly all is disposed through deep well reinjection with chlorination treatment of the water required prior to reinjection to address bacterial contamination.

Surface water disposal is common in some areas of the US and has previously been poorly managed (USEPA 2010). This has led to altered ecosystems, destruction of salt-intolerant vegetation and organisms, and extensive erosion (Duncanson 2010). There are also concerns that the co-produced water should be reused within the region it originates. For example, in the summer of 2002 the States of Montana, Colorado and Wyoming recorded their fifth consecutive season of drought; at the same time coal seam gas operators were releasing large volumes of co-produced water into watercourses. Although some of this water was accessed by users pumping directly from the rivers, the majority flowed out of the original regions and provided little benefit to the wider community (Duncanson 2010).

In the western US, co-produced water in predominantly managed as a waste product rather than a resource. This is due to impediments such as the cost of treatment and cost of gaining permits for water that may decline as coal seam gas wells cease production.

6.2.2 Canada

Major coal seam gas-producing areas in Canada are located in Alberta and British Columbia. The oil and gas industry as a whole has been dealing with the disposal of saline water for several decades and existing regulations are strict (CAPP 2006). Operations in Alberta have predominantly occurred in areas with relatively ‘dry’ coal beds, with ‘wetter’ areas being developed since 2005. This change may be occurring as the more easily accessed coal seams are depleted and operations are moving into areas requiring higher levels of dewatering. Higher levels of co-produced water may change the way that the industry and regulators view water management.

There is little regulatory emphasis in Alberta on beneficial re-use, even though the Province is already faced with water shortages and over-allocation in some of the major river basins. Co-produced water has been identified as a potential source for improving water supply in the short and long term, given appropriate treatment (Hum et al. 2006). While beneficial reuse is favoured, it is limited by constraints such as treatment and transport costs (CAPP 2006). The cost of treating coal seam gas water in 2006 to a drinking water standard was two to three times the cost of existing drinking water in Edmonton and Calgary, making it uneconomic to pursue in many districts (Hum et al. 2006).

The most common disposal methods are surface disposal evaporation ponds and subsurface disposal. Surface disposal is the cheapest method of disposal, although it requires stringent monitoring to avoid environmental damage. There are also concerns over the loss of opportunity of beneficial use of the water within the regions it is produced (Duncanson 2010). Evaporation ponds are the least popular disposal option, due to large areas required and potential release of toxic organic chemicals.

Subsurface injection is the default disposal method for co-produced water. Current regulation leads to water that could be put to beneficial use being wasted. It also may not protect aquifers from contamination from cross aquifer seepage (Duncanson 2010). Subsurface injection requires reinjecting the co-produced water into the formation it was originally drawn from, or into a deeper, lower quality aquifer. There are conflicting views on whether aquifer reinjection is a desirable disposal option. This option does contain a risk that connecting aquifers may be contaminated, although it is difficult to quantify the level of risk due to limited information on aquifer connectivity. There is also a concern that the reinjected water will not be available for use at a later date, thereby ‘wasting’ a useful water supply. However, others feel that this is a safe disposal method, with the water being stored for future access. Of the three disposal methods described above, subsurface injection is currently the preferred disposal method. It is considered as a lower environmental threat with greater opportunity for storage and later access (Duncanson 2010).

6.3 Recent changes to legislation in Australia

The disposal options considered for the APLNG project (2010) and the GLNG project (2009) were developed under the Queensland coal seam gas water management policy of 2008, when these projects started. A Queensland coal seam gas water management policy was introduced in December 2012 (DEHP 2012) and the hierarchy of preferred coal seam gas water management options is now:

1. beneficially use for one or more of the following: the environment, existing or new water users and existing or new water-dependent industries
2. disposing to watercourses
3. disposal of coal seam gas water to evaporation dams.

Of these options, beneficial use is considered the priority, where it is feasible. Disposal to watercourses will only be considered for residual portions of coal seam gas water where there is no feasible beneficial use, and disposal options will not adversely affect environmental values. Evaporation dams will only be considered for managing coal seam gas water produced during exploration or production testing or where there is no feasible alternative (DEHP 2012).

It is possible that some earlier projects may not have been approved under their current configurations had the new policy been in place at the time of assessment, and that projects with disposal to surface waters may have been required to investigate beneficial uses in more detail.

6.4 Mitigation and management options

Co-produced water presents threats as well as benefits to aquatic ecosystems and their associated environmental values, depending on the quality, quantity and timing of the discharges. The main threats are contamination of surface water and groundwater resources, and changed flow regimes. Where co-produced water is appropriately treated and discharge patterns are managed, there is a potential benefit if it can be used to supplement the water supply to aquatic ecosystems.

Current ‘best practice’ is to use risk-based approaches to assess the risks of new coal seam gas and coal mining developments to key environmental assets (e.g. AS/NZS 2009; URS 2009a). The outcome is a risk management strategy that aims to minimise impacts of these operations to the environment.

At the planning or EIS stage there is often a poor understanding or limited certainty of site-specific data, particularly for water quality, both before and after treatment, as well as for the timing of water production. Projects are usually developing wells over a large area and water quality may vary significantly between these areas. Sampling at the planning stage may not be comprehensive enough to adequately characterise water quality across the project. Project planning is also likely to alter as the project progresses and this will affect the timing of wells coming into production. In response to the uncertainty posed by limited data and project uncertainty, the management and mitigation measures proposed at the planning phase may be quite generic. However, the proponent will generally make a commitment to ongoing monitoring and further investigations to reduce uncertainty as part of the management strategy.

Critical hydrological risks include altering the natural wetting and drying cycles and changing an ephemeral stream to a perennial stream. The risk of co-produced water increasing flow in receiving waterways to the point of causing environmental harm depends on the magnitude of discharge compared to the natural flow in the receiving waterway with the risk of ecological change increasing with increasing magnitude of flow alteration (Poff & Zimmerman 2010 and see Chapter 3). In recent EIS assessments, some proponents have proposed variable discharge regimes to minimise this risk and maintain natural flow variability. However, additional water storages are then required, in order to manage the co-produced water when conditions are not suitable for discharge. This may create further management risks.

Variable discharge is not always proposed, for example the APLNG Talinga site is licensed for a constant discharge of 20 ML per day (APLNG 2011). The reason is often that the discharge represents a very small volume in comparison to the natural flow regime, although this often neglects to consider timing of flows.

An additional risk associated with the large volumes of co-produced water and the emphasis on beneficial reuse is finding a strategy that can use or absorb the required volume. For example, agricultural use may have a seasonal demand profile and any strategy that incorporates beneficial reuse for agriculture may need another beneficial reuse or disposal option to manage it if there is surplus to requirements. Proponents need a combination of beneficial reuse strategies, with disposal options as a backup.

Water quality risks are addressed through water treatment techniques and dosing to match the discharge water quality to licence conditions. This does not always protect values in the receiving environment due to the generic nature of water quality guidelines and a general lack of tailoring license discharge conditions to specific receiving water quality. A combination of techniques is required to meet the water quality requirements for a variety of beneficial reuse or environmental discharge applications.

7 Synthesis

The use and disposal of co-produced water from coal seam gas and large coal mine operations are subject to increasingly rigorous requirements to ensure that impacts on the environment values or beneficial uses are adequately managed or mitigated. As coal seam gas and large coal mine production increases, it is accompanied by an increasing volume of co-produced water that requires management (RPS 2011). For coal seam gas, the volume of co-produced water generated depends on the number of wells in production and the hydrogeology of the coal seam. For large coal mines, the volume of co-produced water depends on the depth of excavation relative to local groundwater levels, the amount of run-off generated from the local site and the amount reused within the mine operation. In Australia, the largest volumes are associated with operations in Queensland, particularly in the Fitzroy River catchment and the Queensland Murray-Darling Basin (RPS 2011).

A preferred option of regulators for the management of co-produced water is via supply to beneficial use, such as agricultural, industrial or town water supply uses, or injection into depleted aquifers for recharge purposes, after treatment (E.g. DEHP 2012; DEHP 2010a). Although opportunities exist for beneficial use of co-produced water, demand patterns, available volumes, timing of supply and treatment and delivery infrastructure constraints mean that many potential uses may not always be feasible. Under these circumstances, direct disposal to waterways or a combination of disposal options need to be considered (RPS 2011).

7.1 Water quality

The quality of raw co-produced water is understood at a general level and there is also a general understanding of the range of water quality constituents that are likely to cause problems. Constituents of most concern are salinity and a range of cations (SKM 2011; Shaw 2010; DEHP 2009a). The quality of co-produced water varies widely between wells and across regions (WorleyParsons 2010) and there are limited data at the site scale (SKM 2011). Risk assessments are therefore needed to evaluate the impacts on environmental values.

There is also a general understanding of the ecological impacts of various individual toxicants. The ANZECC/ARMCANZ (2000) guidelines provide an extensive review of the impacts of various stressors and toxicants on ecological values, although much of this assessment is based on overseas studies and there is limited assessment of toxicity to Australian biota (Shaw 2010). There has been significant research on the impacts of salinity on Australian biota and documented in a number of literature reviews (Rogers et al. 2011; Boon 2008; Clunie et al. 2002; O'Brien 1995; Hart et al. 1991).

Water quality guidelines to protect beneficial uses, including ecosystem protection, have been established at national, state and some regional levels. Most are for permanent or perennial rivers and lakes (ANZECC/ARMCANZ 2000) and may not be appropriate for ephemeral streams and wetlands. Ephemeral wetlands and streams experience highly variable water quality especially during the drying phase (Boulton & Brock 1999) and specific guidelines have generally not been developed due to a lack of reference condition data (DEHP 2009a). The lack of water quality guidelines for intermittent water bodies is a critical knowledge gap in the context of managing co-produced water and discharges. The Healthy HeadWaters Coal Seam Gas Water Feasibility Study partly addresses this issue in the

Queensland Murray-Darling Basin through the development of regional salinity guidelines for disposal of co-produced water (Rogers et al. 2011). Guidelines are not available for all possible stressors and toxicants in co-produced water.

Given that untreated co-produced water generally exceeds water quality guidelines for some stressors and toxicants, especially salt, treatment is necessary to enable beneficial use or release back to the environment (Nghiem et al. 2011). Treatment options are available to remove primary toxicants. However, the treatment processes can introduce a range of other issues, for example, by making the water too clean (Takahashi et al. 2011a). Under these circumstances, post-treatment is often required to re-mineralise water to make it suitable for specific beneficial uses, especially for use in irrigation and for release to surface waters. However, discharge licenses are often set based on reference to generic guidelines with little consideration of the actual water quality in receiving waterways (E.g. Higgins et al. undated; Wright 2012), or of the cumulative loads to a catchment (Hamstead & Fermio 2012). This is changing with increased awareness of water quality impacts, the development of regional water quality guidelines (e.g. DEHP 2011b; Rogers et al. 2011) and the introduction of discharge trading schemes (e.g. the Hunter Valley Discharge Trading Scheme – Hamstead & Fermio 2012).

Although single-constituent toxicity and effective treatment may be known, there is still a risk that the complex mixture of stressors and toxicants may interact to modify contaminant bioavailability or have synergistic toxic impact. This is the case where cumulative impacts and interactions between different discharges may be occurring in a catchment (Takahashi et al. 2011a). Toxicity testing has shown that untreated co-produced water results in higher toxicity compared with water treated through RO. Even with RO treatment, some toxicity impacts are still detectable being typically related to the very low ionic concentrations (Takahashi et al. 2011a). More investigations are needed to better understand risks around mixtures of chemical contaminants and thresholds for significant impacts. Direct toxicity assessment should be considered as part of any assessment of future discharges, especially since water quality guidelines are not available for all potential contaminants*.* Of particular concern is the need to assess the toxicity of the co-produced water after dilution in a receiving system.

There are significant knowledge gaps in the understanding of how cumulative water quality impacts develop in river systems, whether systems are able to cope with multiple impacts and what the critical thresholds should be for capping cumulative loads. Furthermore, even when load limit conditions are applied at the catchment scale through development and application of discharge trading schemes, site-specific limits are also required to protect values at the site scale. At the catchment scale, modelling of cumulative water quality impacts is required to identify load limits. Site-specific assessments are required at the site scale to consider background water quality in the receiving environment or for the intended beneficial use. An appropriate regulatory framework is required to allow for catchment-based load limits. These are critical knowledge gaps that need to be addressed to avoid potentially irreversible impacts*.*

7.2 Water quantity

The total quantity of co-produced water generation is highly variable across sites. This means that management options also vary and need to be flexible across regions. For example, where co-produced water volumes are low and there is a significant local demand, discharge to waterways is very rarely necessary (e.g. AGL's Camden Gas Project – AGL 2012). In other locations, the discharge of coal seam gas co-produced water to waterways has been occurring for some period of time (e.g. at Santos' Fairview Field since the early

1990s – URS 2009a). Discharge of co-produced water from large coal mines is also occurring across a wide range of sites (e.g. Wright 2012; DEHP 2009b).

If co-produced water is released to a waterway or wetland, the level of risk depends on the timing and volume of the release and on how significantly it will change the water regime (McGregor et al. 2011). For streams that are strongly perennial and carry large flow volumes, co-produced water represents little risk. Where streams are weakly perennial or ephemeral, an increase in flow represents a significant risk because it can change the water regime (Mackay et al. 2012). A constant discharge of even low to moderate volumes of co-produced water can result in ephemeral streams becoming perennial and seasonal wetlands becoming permanently inundated. This can result in an increase in nuisance plant and algal growth, colonisation by pest species and loss of native species with breeding triggered by cease-to-flow periods (Takahashi et al. 2011b). Unfortunately, most discharge licenses do not include or only vaguely acknowledge requirements to manage the volume of water discharged relative to flow in the receiving waterway (DEHP 2009b).

The Healthy HeadWaters Coal Seam Gas Water Feasibility Study developed specific guidelines for managing flow regimes (McGregor et al. 2011) and a decision support system was developed to guide assessments (Takahashi et al. 2011b). Such guidelines should be used to assess the specific environmental risks from increased flow identified for each project and to subsequently protect environmental values. Moreover, management plans should be accompanied by robust monitoring programs to assess their effectiveness and to allow adaptive management.

The cumulative impact of multiple small-volume discharges is not well documented. As with water quality impacts, catchment scale models are required to model surface flow scenarios. These should consider a range of projected discharge volumes so that decisions can be made at a catchment scale of the acceptable discharge volumes and timing of releases to minimise impacts on natural flow regimes. Modelling could also be used to assess the potential for discharge of co-produced water to augment environmental flows where over extraction has reduced the natural flow.

7.3 Risk assessment and water management

As our knowledge has increased over the past few years of the potential impacts of co-produced water to water quality and quantity, the management requirements for co-produced water are changing rapidly. In response, most current proposals for coal seam gas expansion projects have changed their proposed water management strategy during or post the preparation and review of EIS, with many now testing aquifer reinjection as one of the primary water management options in preference to discharge to waterways. Some energy producers are also looking at alternatives to the direct supply of co-produced water to end users, such as transfer of co-produced water to an intermediary who then manages the sale and distribution of the water. This is a logical move for energy producers, as water supply and distribution is not their core business.

Current 'best practice' is to use risk-based approaches to assess the risks of new coal seam gas and coal mining developments to key environmental assets (e.g. AS/NZS 2009; URS 2009a). The outcome is a risk management strategy to minimise impacts on the environment. A lack of data and cumulative impact models mean current approaches are generally qualitative. Criticisms of these methods include the use of descriptive scales that are subjective, rarely transparent, based on limited expert opinion and very difficult to validate (Burgman 2005; Burgman 2001). Future assessments could make better use of the quantitative risk assessment frameworks that are available, which should:

- address as a minimum the key threats from (a) increased salinity, (b) increased toxicity, and (c) changes in the flow regime, particularly in ephemeral streams
- address as a minimum the risks to key environmental indicators (assets), including (a) biodiversity, (b) fish communities, (c) macroinvertebrate communities, (d) riparian vegetation and (e) specific beneficial use criteria
- address possible cumulative risks due to other existing or planned coal seam gas or coal mining developments
- use or develop appropriate modelling techniques, particularly those that quantify the relationships between key threats and key ecological indicators, to ensure the risk assessment is as quantitative as possible.

Within the EIS process, cumulative impact assessments are generally carried out at a high level and in a simplistic manner. This is primarily because of the lack of information that is publicly available to the proponent, as well as uncertainty over whether other projects will be approved, and if so, in what form. This uncertainty makes it difficult to assess what should realistically be considered under a cumulative impact assessment and is an area that would benefit from management or better guidance from regulators. As part of cumulative impact assessments, thresholds for significant change need to be identified and incorporated into regional guidelines for both catchment scale load limits and site scale concentration and discharge limits.

7.4 Critical knowledge gaps

There is generally a good understanding of the major water quality constituents in untreated co-produced water and their environmental impact. Treatment processes are available but there are limited attempts to specifically match the quality of treated water to the characteristics of the receiving environment or particular uses. Rather, water is treated to meet discharge licence conditions that may not adequately match receiving water condition or beneficial use requirements depending on the level of knowledge of the system. Furthermore, in areas where multiple impacts are possible, there is limited understanding of critical load thresholds to catchments for various water quality constituents such as salt, nutrients, heavy metals and suspended solids. There is limited understanding of the potential for interactions between constituents to contribute to impacts, even when guidelines for individual water quality constituents are met.

The ecological consequences of altering the flow regime of ephemeral waterways by increasing discharge are broadly known. However, there is a general lack of data on the volumes of water that may be discharged to streams and even less understanding of how much additional water a stream could receive before fundamental and irreversible changes to stream ecology occur.

Impacts from the discharge of co-produced water at a site scale may be relatively small and contained to a small area in some cases. However, in some locations, current and potential future discharge volumes could be high. In these locations the cumulative impacts on water quality and quantity are unknown and modelling needs to be undertaken to identify the critical load and flow thresholds. Toxicity testing is also required as part of a comprehensive quantitative risk assessment process, especially since water quality guidelines are not available for all potential contaminants. There is a lack of publicly available data to inform cumulative impact assessments.

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