EFFECTS OF A DESALINATION PLANT DISCHARGE ON THE MARINE ENVIRONMENT OF BARROW ISLAND

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1.0 BACKGROUND

Chevron Australia Pty Ltd (Chevron Australia) proposes to construct and operate a desalination plant on the east coast of Barrow Island, Western Australia. The desalination plant will produce potable and service water from sea water using reverse osmosis (RO) technology. It is proposed that reject brine effluent will be discharged to the marine environment via an ocean outfall located in the vicinity of the east coast marine facilities.

Given the variable water requirements of the Gorgon Gas Development, a staged water supply strategy will be implemented. In order to meet the water demand from initial mobilisation to Barrow Island and start of construction, a temporary RO package situated temporarily within the gas treatment plant has been proposed as the first water treatment facility. As construction progresses, the temporary RO package will be supplemented with a bridging RO plant located adjacent to the General Utilities area within the gas treatment plant site. A permanent RO plant will operate during the operations phase of the Gorgon Gas Development, and will be capable of producing a maximum of 2100 m^3 day^{-1} of product (i.e. fresh) water. Based on an assumed RO unit recovery of 40%, the normal reject brine flow rate will be 2100 m^3 day^{-1}, whilst the maximum flow rate will be 3150 m^3 day^{-1}.

In the past, most desalination plants have used distillation techniques, but more recently RO technology has become a preferred option (Water Consultants International 2006; Tularam & Ilahee 2007). Improvements in technology have increased the efficiency of modern RO plants, making them more energy efficient, reliable and cheaper to run than distillation plants (Water Consultants International 2006; Tularam & Ilahee 2007).

Literature reviews have investigated the potential environmental effects of marine discharges from both distillation and RO desalination plants (Water Consultants International 2006; Tularam & Ilahee 2007; Latterman & Hopner 2008). The reviews detail the differences between physical and chemical properties of desalination discharges and sea water. Reviews have also been completed summarising available information on the potential impact of desalination discharges on cetaceans and sea turtles (Pendoley 2007; URS 2008).

This literature review paid particular attention to quantified impacts of desalination discharges on the physical, chemical or biological characteristics of the marine environment. Despite the number of desalination plants that have been constructed around the world there is little data available on the impacts of their discharges (Water Consultants International 2006; Tularam & Ilahee 2007, Latterman & Hopner 2008). Most quantitative research has focused on *a priori* modelling of the potential impacts of brine discharge, but rarely have these predictions then been tested after plants are constructed. Four recent research papers, however, have measured impacts of discharges from operational RO desalination plants at the Canary Islands and North-west Mediterranean Spain (Talavera & Ruiz 2001; Fernandez-Torquemada et al. 2005;

Raventos et al. 2006; Del-Pilar-Ruso et al. 2008). These four key studies provide the best information relevant to the proposed desalination plant at Barrow Island.

This report draws from past literature reviews, the four key studies and additional literature to inform the development of an Ecological Risk Assessment for the proposed Barrow Island reverse osmosis plant.

2.0 PHYSICAL AND CHEMICAL IMPACTS OF WASTE BRINE

The literature includes many model-based predictions about dilution rates of brine released into the sea. There have been, however, limited field-based measurements of the dilution of brine effluent. The four key research papers (Talavera & Ruiz 2001; Raventos et al. 2006; Fernandez-Torquemada et al. 2005; Del-Pilar-Ruso et al. 2008) provided some quantitative field measurements of dilution rates of waste brine and its impact on the physical and chemical properties of the receiving sea water.

2.1 Assessing Physical and Chemical Changes

The procedures for assessing the impact of desalination waste brine on the marine environment are presented in the 'Australia and New Zealand Guidelines for Fresh and Marine Water Quality' (ANZECC 2000), and 'Background Quality for Coastal Marine Waters of the North West Shelf, Western Australia' (Wenziker et al. 2006) The Australian water quality guidelines state that where adequate historical data is available, or sufficient resources are available to collect the necessary information, locally suitable trigger concentrations should be developed (ANZECC 2000). While useful for reference, the default tropical Australia triggers presented in ANZECC (2000) should only be used where site-specific triggers do not exist, or cannot be developed. The development of site-specific trigger values for a comprehensive range of stressors is important in tropical ecosystems, largely due to the high natural variability in these systems (ANZECC 2000).

Locally suitable site-specific trigger values exist for locations near Barrow Island. A comprehensive set of background metal and organic chemical measurements were completed for waters of the North West Shelf during 2003 (Wenziker et al. 2006). Concentrations of analytes in the samples were generally very low, indicating that background metal and organic chemical concentrations in the coastal waters of the North West Shelf approach oceanic levels. Wenziker et al. (2006) combined the North West Shelf measurements, with the recommended guidelines and approaches in ANZECC (2000), to provide a set of environmental quality criteria for the North West Shelf. It is necessary that these criteria be taken into consideration when assessing the impacts of desalination discharges on the marine environment of Barrow Island.

Marine sediments surrounding Barrow Island are generally considered pristine, however there are some areas of localised disturbance.

2.1.1 Temperature

Desalination processes can increase the temperature of waste brine above the temperature of the ambient seawater. However, literature on temperature effects of desalination is often unclear about whether assessment of temperature ranges was based on model predictions or field measurements.

Distillation plants heat the feedwater and this heat is largely retained in the waste brine. RO plants do not heat feedwater but rely on the use of membrane filters to separate the freshwater from saline brine. The heating of feedwater in distillation plants elevates waste brine temperatures by 5–15 °C above ambient, while RO waste brine is generally near ambient temperature (Water Consultants International 2006; Tularam & Ilahee 2007; Latterman & Hopner 2008). The Gold Coast and Sydney RO desalination plants are predicted to produce waste brine typically 1–2 °C above ambient levels (Gold Coast Desalination Alliance ERA; The Ecology Lab 2005). A model predicted the Burrup Peninsula RO waste brine temperature would be less than 2 °C above ambient at 7 m from the outfall, 0.25 °C above ambient at 110 m from the outfall and 0.1 °C a few hundred metres from the discharge (EPA 2001). Field measurements of waste brine temperature at an operating RO plant in the Canary Islands were 2 °C above ambient at release. Other studies suggest some RO plants increase temperatures by between 3–9 °C above ambient (Talavera & Ruiz 2001; Hashim & Hajjaj 2005).

The Barrow Island RO plant has been predicted to produce brine at, or around, ambient sea water temperature on discharge (Preston et al. 2007).

2.1.2 Salinity

Reverse osmosis produces waste brine with salinity up to twice that of sea water (Tularam & Ilahee 2007). This increase in salinity consequently increases the density of the brine and, if undiluted, the brine will stratify below the less dense sea water (Hashim & Hajjaj 2005; Younos 2005; Water Consultants International 2006; Tularam & Ilahee 2007; Latterman & Hopner 2008). At Barrow Island there is no evidence of significant salinity stratification naturally, due to the high degree of mixing through the water column (RPS Bowman Bishaw Gorham 2007).

Dilution of waste discharge has been measured at desalination plants in the Canary Islands, and north-west Mediterranean Spain (Talavera & Ruiz 2001; Fernandez-Torquemada et al. 2005; Raventos et al. 2006). The Canary Islands facility produces waste brine with salinity twice that of the ambient sea water level and similar to the predicted salinity of the Barrow Island facility (Talavera & Ruiz 2001; Preston et al. 2007). Notably, the Canary Islands desalination plant produces over five times the volume of waste brine (17,000 m^3 day^{-1}) than the proposed Barrow Island permanent facility (peak of approximately 3150 m^3 day^{-1}). Samples taken 20 m from the Canary Islands outfall show a rapid near-field dilution of waste brine to near ambient levels. A similar investigation of dilution rates was completed at a RO plant in the north-west Mediterranean (Raventos et al. 2006). The study also found rapid near-field dilution of brine, with ambient salinity achieved at 10 m from the outlets. Measurements of salinity at a larger RO plant in Spain, which produces twenty five times more waste brine (75000 m^3 day^{-1}) than the proposed Barrow Island facility, also found salinity initially declined rapidly close to the outfall (Fernandez-Torquemada et al. 2005, Del-Pilar-Ruso et al. 2008). The measurements from the three facilities suggest that, in general, elevated salinity declines rapidly close to the outfall for RO plants, even where fairly simple diffusers are employed (e.g. Fernandez-Torquemada et al. 2005).

2.1.3 pH

The de-chlorination process at the Barrow Island desalination plant may marginally reduce the pH of the waste brine compared to intake seawater. The literature supports this view by stating that the pH of RO plant waste brine is usually not changed or marginally lower than the feed-water (Hashim & Hajjaj 2005; Latterman & Hopner 2008). ANZECC (2000) presents the default pH trigger of 8.2 for slightly disturbed tropical Australia ecosystems. Considering the small volume of waste brine planned for production at the Barrow Island facility, the high degree of mixing and the buffering capacity of sea water, pH will most likely rapidly revert to that of ambient sea water within meters of the brine outfall.

2.1.4 Dissolved Oxygen

A reduction in dissolved oxygen has the potential to impact marine life (Water Consultants International 2006; Tularam & Ilahee 2007; Latterman & Hopner 2008). Distillation plants often produce brine with reduced levels of dissolved oxygen (Hashim & Hajjaj 2005; Morton et al. 1996). Significant reduction of dissolved oxygen due to heating is unlikely to occur in RO plants, as temperatures are only marginally increased. ANZECC (2000) presents a default dissolved oxygen trigger, for slightly disturbed tropical Australian ecosystems, of 90 % daytime saturation.

The addition of oxygen consuming chemicals can reduce dissolved oxygen levels in RO plants (Latterman & Hopner 2008). The oxygen consuming chemical, sodium bisulphite, is often added to RO feed water to inhibit corrosion and remove residual chlorine (Gold Coast Desalination Alliance ERA; Water Consultants International 2006). The proposed Barrow Island plant is expected to use and discharge sodium bisulphite, which could reduce dissolved oxygen in the waste brine. Rapid dilution is likely to maintain dissolved oxygen levels close to ambient levels at the Barrow Island facility.

A review of field-based measurements found that the effluent from the Perth desalination plant is highly diluted and has not had a measurable impact on dissolved oxygen concentrations in the deeper waters of Cockburn Sound (Oceanica 2007; Okely et al. 2007). Given the small volume of discharge to be released at the proposed Barrow Island facility, it seems unlikely that the ANZECC (2000) default dissolved oxygen trigger would be reached.

2.2 Chemicals Added During Reverse Osmosis

A range of chemicals, such as pre-treatment, post-treatment and cleaning chemicals are required during either distillation and RO desalination. Often these chemicals are discharged to the ocean as part of the waste brine (Campbell & Jones 2005; Preston et al. 2007; Tularam & Ilahee 2007), as will be the case for the proposed Barrow Island RO plant. The range of chemicals that are used in desalination can be separated into pre- and post-treatment chemicals as well as cleaning chemicals. The use of pre- and post-treatment chemicals is necessary to avoid bio-fouling, suspended solid accumulation and scale deposits (Latterman & Hopner 2008). Pre-treatment of the feed water is required to remove and control substances that could impact the efficiency of desalination process (Hashim & Hajjaj 2005). Cleaning chemicals are required to clean the RO membranes on an intermittent basis (Sadhwani et al. 2005; Latterman & Hopner 2008).

2.2.1 Pre- and Post-treatment Chemicals

The mixing of treatment and cleaning chemicals into the waste brine prior to discharge through an ocean outfall is common. Water Consultants International (2006) surveyed seventeen RO plants and found fifteen of the plants discharged some, or all of the chemicals with the waste brine. The Burrup Peninsula RO plant plans to release the pre- and post-treatment chemicals, including anti-scalant and low concentrations of the biocide chlorine, as part of the waste brine (EPA 2001). The Barrow Island desalination plant also plans to dispose of pre- and post-treatment chemicals as part of the waste brine via an ocean outfall. Pre- and post-treatment chemicals can be separated into anti-scalants, coagulants and biocides.

2.2.1.1 Anti-scalants

Desalination plants often use anti-scalants, such as sulphuric acid, polyacrylic acid and polymeric acid, to prevent fouling through scale formation in equipment and RO membranes (Morton et al. 1996; Hashim & Hajjaj 2005; Sadhwani et al. 2005; Water Consultants International 2006; Tularam & Ilahee 2007). The anti-scalant chemicals often work by maintaining a particular pH that inhibits carbonate scale formation (Hashim & Hajjaj 2005). The toxicity of anti-scalants chemicals to aquatic life is generally low (Tularam & Ilahee 2007; Latterman & Hopner 2008). At a typical dose of 2 ppm in feed water, the environmental risk from the use and subsequent discharge of anti-scalants within the brine stream is viewed as relatively low (EPA 2001; Hoepner & Lattemann 2002).

2.2.1.2 Coagulants

Coagulants reduce suspended particulate matter during pre-treatment, through coagulation and flocculation, into larger particles that can be more easily removed via backwashing (Hashim & Hajjaj 2005; Sadhwani et al. 2005). The backwashing products are normally released with the waste brine and often contain coagulants, such as ferric chloride, aluminium sulfate and coagulant aids, such as polyacrylamide (Hashim & Hajjaj 2005; Sadhwani et al. 2005; Water Consultants International 2006; Latterman & Hopner 2008).

The Barrow Island desalination plant may use ferric chloride as a coagulant that would be disposed with the waste brine. The use of the coagulant ferric chloride can colour waste brine, reducing light penetration, which can reduce primary productivity (Water Consultants International 2006; Latterman & Hopner 2008). The clarity of the inshore waters of Barrow Island naturally varies from clear to highly turbid (RPS Bowman Bishaw Gorham 2007). As a result, the release of the coagulant ferric chloride at the proposed Barrow Island desalination plant is unlikely to significantly increase turbidity beyond the broad natural range.

2.2.1.3 Biocides

Feed water contains living organisms that can foul desalination plant components. Biocides are thus used to mitigate potential fouling. Tularam and Ilahee (2007) viewed chlorine as one of the major pollutants in the RO process. Chlorine is a strong oxidant and an effective biocide that can be toxic to marine life even in diluted concentrations (Hashim & Hajjaj 2005; Water Consultants International 2006; Tularam & Ilahee 2007). Chlorine also reacts with organic compounds in sea water to form other compounds, such as halogenated organic by-products, which are harmful to marine life (Grebenyuk et al. 1996; Hoepner & Lattemann 2002; Water Consultants International 2006; Tularam & Ilahee 2007). The proposed Barrow Island facility is expected to use chlorine as a biocide, but will neutralise this before disposal as part of the waste brine. In modern RO plants, the chlorine is typically neutralised with sodium metabisulphite before the water enters the membrane, resulting in low concentrations in waste brine (Morton et al. 1996; Hoepner & Lattemann 2002; Hashim & Hajjaj 2005; Sadhwani et al. 2005; Water Consultants International 2006; Latterman & Hopner 2008).

2.2.1.4 Cleaning Chemicals

RO membranes require regular cleaning to maintain efficiency (Hashim & Hajjaj 2005). Typical RO cleaning solutions are predominantly alkaline (pH 11–12) or acidic solutions (pH 2–3), and include detergents, such as dodecylsulfate and complexing agents (e.g. EDTA). Biocides, such as sodium perborate and formaldehyde, are used to kill bacteria (Hashim & Hajjaj 2005; Water Consultants International 2006; Tularam and Ilahee 2007; Latterman and Hopner 2008). The cleaning chemicals required vary depending on the type of membrane (Younos 2005). Cleaning chemicals can be separated from waste brine and disposed on land, as is proposed for the Burrup Peninsula facility (EPA 2001), or released with brine, as is proposed for the Barrow Island facility. Cleaning solutions are typically used every three to six months and are potentially toxic to aquatic life, if discharged without neutralisation to moderate the extreme pH (Sadhwani et al. 2005; Latterman & Hopner 2008).

2.2.2 Heavy Metals and Nutrients

2.2.2.1 Heavy Metals

Both RO and distillation plants are known to produce waste brine with heavy metals in relatively low concentrations (Latterman & Hopner 2008). Increases in heavy metals are associated with distillation techniques where high temperature aids corrosion of metals constructed with copper, nickel, chromium, molybdenum, iron and zinc alloys (Water Consultants International 2006; Tularam & Ilahee 2007; Latterman & Hopner 2008). RO facilities are less likely to release heavy metals as they are usually constructed largely of corrosion resistant stainless steel (Oldfield & Todd 1996; Tularam & Ilahee 2007). If low quality stainless steel is used corrosion could slightly increase levels of iron, chromium, nickel and molybdenum in waste brine (Oldfield & Todd 1996; Water Consultants

International 2006; Latterman & Hopner 2008). The RO process also adds treatment and cleaning chemicals that can include metals such as iron (Hashim & Hajjaj 2005).

Any heavy metals that naturally occur in sea water, are leached through corrosion, or added in the pre-treatment process, will be concentrated during the RO process after the freshwater is removed (Hashim & Hajjaj 2005). Thus, at the proposed Barrow Island facility it is possible that the concentrations of some heavy metals will be slightly elevated in waste brine. The small plant size, however, will likely provide for rapid dilution to near ambient sea water levels within metres of the brine outfall.

2.2.2.2 Nutrients

The addition of nitrogen and iron during the RO process has potential to affect primary productivity, as they are potentially limiting elements in marine systems. Given the low volume of RO discharge for the Barrow Island facility and high degree of mixing in the receiving waters, eutrophic effects are considered highly unlikely. Nutrient load samples in a north-west Mediterranean plant rapidly diluted to ambient concentrations at a distance of 10 m from the diffuser outlets (Raventos et al. 2006).

2.2.3 Summary of Potential Physical and Chemical Impacts

The proposed Barrow Island facility will produce a relatively small volume of waste brine compared to the desalination plants in the Canary Islands and north-west Mediterranean Spain (Talavera & Ruiz 2001; Fernandez-Torquemada et al. 2005; Raventos et al. 2006; Del-Pilar-Ruso et al. 2008). Rapid dilution of salinity, temperature, pH and chemicals to near ambient levels were recorded in the near-field, at 10–20 m from the outfalls. The smaller volume of waste brine proposed for release at the Barrow Island facility suggests that near-ambient sea water levels of salinity, temperature, pH and chemicals are also likely to be achieved less than 10–20 m from the outfall.

2.3 Entrainment and Impingement

RO desalination plants require substantial intake of sea water, which can result in impingement (i.e. collision with screens) and entrainment (i.e. drawn into the plant) of marine organisms (Latterman & Hopner 2008). Screens of roughly 5 mm mesh width are normally used to avoid marine debris and large marine animals from entering the water intake (Morton et al. 1996). A study conducted for the Sydney desalination plant found that flow rates of 0.1m s^{-1} do not impinge larger fish and animals, but smaller animals including phytoplankton, zooplankton and ichthyoplankton, are likely to be impinged or entrained (The Ecology Lab 2005). Green turtle hatchlings, for example, swim at an average speed of 0.4 m s^{-1} (Lohmann and Lohmann 1992) and so should be able to swim away from an intake flow rate of 0.1 m s^{-1} or less. Implementation of screens in combination with low flow rates will mitigate many impingement and entrainment issues.

2.4 Construction

The construction of any desalination plant will have a direct impact on the surrounding habitat. Impacts from construction include direct habitat loss, smothering through re-suspension of sediments from dredging and damage caused by increased vessel movements (Younos 2005; Latterman & Hopner 2008). The Gold Coast and Sydney desalination plants anticipated very localised disturbance of sediments, which would result in the localised mortality of benthic fauna (Gold Coast Desalination Alliance ERA; The Ecology Lab 2005). The Burrup Peninsula desalination plant was predicted to disturb an area of one hectare where no coral communities or extensive seagrass exists (EPA 2001). In the case of the Barrow Island facility, additional direct impacts from construction of the RO plant and discharge pipeline will be negligible, since it sits within the high impact zone associated with the construction of the Materials Offloading Facility.

3.0 TOXIC EFFECTS

There is limited systematic information on the direct toxicity of desalination waste brine on marine species (Hopner & Windelberg 1996; The Ecology Lab 2005; Raventos et al. 2006; Bleninger & Jirka 2008). There is potential for both short and long-term toxic effects of RO waste brine on marine ecosystems, but any such effects remain unquantified. In addition, although whole effluent testing (WET) ecotoxicology research is being conducted in association with the construction of a number of desalination plants across Australia, at present little reliable WET test information is available publicly.

The Ecological Risk Assessment of the Gold Coast desalination plant anticipated that waste brine released from the outfall would be diluted 40:1 through diffusion, prior to the brine reaching the seabed – at which point, the lower water column would contain less than 0.2 mg L^{-1} of the anti-scalant, *Hypersperse*. Based on toxicology data, it was predicted that the anti-scalant concentrations near the planned Gold Coast desalination plant outfall would be well below concentrations that cause any biological effects (Gold Coast Desalination Alliance ERA).

WET testing of brine collected from the Perth desalination plant found apparently very different levels of toxicity of waste brine between the two years of testing (Woodworth 2008). Principally, there was a great disparity between the data collected 2006 and 2007 for toxicity tests involving copepods and macroalgal fertilization.

The differences in toxicity data between years from the Perth desalination plant and the consequent difficulty this posed for interpreting apparently very different results (Woodworth 2008) illustrate some key inadequacies in the ways WET tests are often reported and interpreted. Almost invariably, single (median) estimates of predicted safety concentrations such as PC99 are reported from WET testing, usually as the output of a Burrlioz analysis (Campbell et al. 2000) or similar, but often with no indication of the precision with which estimates have been made. Additionally, bioassays are rarely repeated so there is usually no indication of how different estimates of measured endpoints like EC10 might have been if they had been done at a different time. There are simple solutions to these issues which involve replicating testing appropriately and reporting the precision with which estimates are made (such as with 95% confidence intervals); these are illustrated in the Barrow Island WET testing report being produced in concert with this literature review and assessment of effects.

4.0 CASE STUDIES

4.1 Direct Measures Environmental Impacts of Reverse Osmosis Plants

The available literature contained four research papers that provide the best direct information on the impact of RO waste brine on marine ecosystems. The studies investigated the impact of desalination plants built in the Canary Islands, north-west Mediterranean and Spain (Talavera & Ruiz 2001; Fernandez-Torquemada et al. 2005; Raventos et al. 2006). All three facilities extract fresh water through RO and discharge the brine at ocean outfalls.

4.1.1 Canary Islands

Benthic habitat near a Canary Islands RO plant and wastewater discharge was sampled several times after brine discharge commenced (Talavera & Ruiz 2001). Visual inspection of the habitat, which comprised primarily *Cymodocea nodosa* and *Caulerpa prolifera,* found that areas near the wastewater treatment plant appeared substantially less healthy than areas influenced by increased salinity near the desalination plant (Talavera & Ruiz 2001). In summary, the Canary Islands RO study did not find evidence the desalination plant had any detrimental impact on seagrass or algae.

4.1.2 North-west Mediterranean Sea, Alicante, Spain

The marine life in the waters surrounding a relatively large Spanish plant was sampled before and several times after desalination brine discharge commenced (Fernandez-Torquemada et al. 2005). Two reference locations of seagrass habitat were also sampled, two kilometres to the north and south of the outfall. Health of the seagrass *Posidonia oceanica* was higher at the site not impacted by increased salinity (Fernandez-Torquemada et al. 2005).

Many echinoderms are not able to tolerate even small salinity or pH changes (Roller & Stickle 1993; Havenhand et al. 2008). In the Spanish desalination plant study echinoderms disappeared from the outfall location and also from a reference site, two kilometres down current from the outfall (Fernandez-Torquemada et al. 2005). This site was unexpectedly impacted by slightly increased salinity in contrast to the other reference site located two kilometres up-current from the outfall which was not. Echinoderm numbers did not decline at the up-current site and there was some evidence to suggest echinoderms relocated from the impacted sites to the reference site (Fernandez-Torquemada et al. 2005).

The effects of the Alicante plant brine discharge on the soft bottom Polychaeta assemblage was also examined at nine sites over a two year period (Del-Pilar-Ruso et al. 2008). This study found that different polychaete families had different sensitivities to brine discharge. Families showed one of four main patterns: (1) decreased considerably in the first months of the discharge but then stabilised (2) decreased or disappeared completely (3) showed resistance a the beginning of the discharge but their abundance decreased with time (4) not affected by the brine discharge. The end results, however, were significant decreases in the abundance, richness and diversity of assemblages that were located in the vicinity of the outfall, and also at the next closest site located 400 m away, over time. Average salinities at the closest site and the next closest site were 40 and 38.9 psu respectively, compared to the 37.9 psu average recorded for all other sites.

4.1.3 North-west Mediterranean Sea, Blanes, Spain

A rigorous assessment on the effect of brine discharge from a desalination plant was completed in the north-west Mediterranean Sea (Raventos et al. 2006). This assessment was based on a multiple-before-after-control-impact design, and consisted of two reference and one impacted sites. All sites were sampled monthly, for a year before and after the plant started operation; each time with sixteen (replicate) 2 x 50 m visual transects. No significant variation was attributable to the brine discharges from the desalination plant. The habitat at the brine outflow was dominated by sand, in which biota typically exhibit great natural variability in abundance (Raventos et al. 2006). Failure to detect a significant impact in the north-west Mediterranean experiment might be explained by the high underlying natural variability. Unfortunately, there were no estimates of the statistical power of these tests to assess this explanation.

5.0 ASSESSMENT OF ECOSYSTEM VULNERABILITY

The vulnerability of marine ecosystems is likely to be influenced by the ecosystem's resilience to change (Einav & Lokiec 2003). In an analysis of ecosystem sensitivity, coastal algal and coral reef ecosystems were viewed as very sensitive compared with high-energy open coast ecosystems (Hopner & Windelberg 1996). Coral, macroalgae and seagrass occur at Barrow Island and in the vicinity of the proposed waste brine outfall. This report therefore considered the potential impact of waste brine on these ecosystem types.

Many marine ecosystems and species can tolerate or recover from short-term changes in temperature, salinity and chemicals beyond normal levels, however, permanent changes are likely to result in mobile species relocating (Latterman & Hopner 2008). Attached biota, such as seagrass, coral and macroalgae, are unable to escape the impact and will either tolerate, decline in health or die. Unless well mixed, dense saline waste brine is likely to sink to the ocean floor where benthic species such as seagrass and low-lying coral are affected (Latterman & Hopner 2008).

The marine and coastal environments of the Barrow Island region comprise a unique combination of offshore island, intertidal and subtidal coral reefs, mangroves, macro algal communities and sheltered lagoons (Chevron 2006). Highly diverse coral communities in good condition occur in the waters surrounding Barrow Island (Chevron 2006). Significant coral reefs occur around the island as intertidal reefs, subtidal reefs and individual coral bombora. Macroalgal meadows are the most extensive benthic habitat of the Barrow Island reserve and make the major contribution to primary production (Chevron 2006).

5.1 Coral Reef Ecosystems

5.1.1 Temperature

Campbell & Jones (2005) stated that 'the current consensus among environmental regulators is that a 1 °C rise in temperature from ambient sea water is a safe limit to place on the effluent stream for most open ocean environments'. Although a 1 °C rise above ambient may appear minor, the impact on species already surviving near the top of their thermal tolerance may be substantial. Experimental studies suggest that reef-building corals currently exist close to their thermal tolerance range and results of climate change modelling suggest that all coral reefs will bleach annually with less than 1 °C increase in temperature (Hoegh-Guldberg 1999).

Studies conducted in the Gulf of Panama on *Pocillopora damicornis* found health declined significantly at slightly elevated temperatures of 30–32 °C (3–4 °C above optimum) and the coral died after five weeks at 32 °C (Glynn & D'Croz 1990). Hawaiian colonies of *Montipora verrucosa* were found to have thermal tolerance of 32–33 °C in the short-term, and 31–32 °C over the long-term (Coles & Jokiel 1978).

5.1.2 Salinity

In general, little is known about the tolerance of marine species or ecosystems to long-term increases in salinity, as would result from a brine outfall. At small scales, however, there are a number of experimental studies that examine responses of scleractinian corals to short-term (days to weeks) exposures to increased salinity (Coles and Jokiel 1978, Marcus and Thourag 1981, Muthiga and Szmant 1987, Coles 1992, Ferrier-Pages et al. 1999, Porter et al. 1999, Manzello and Lirman 2003, Lirman and Manzello 2009). At the other extreme, on an evolutionary scale, a number of coral species are known to withstand salinities elevated well above average (e.g. 50 ppt) in locations such as the Arabian Gulf, Red Sea and in lagoons of oceanic atolls (Kinsman 1964, Coles 1992, Shepperd 1998). Appraisal of both types of information suggests that salinity tolerances in coral species depend on a number of factors including: the speed, magnitude and duration of the salinity increase, ambient salinities before the change, individual species tolerance levels, acclimatisation abilities and whether salinity changes are occurring simultaneously to other stressors such as temperature or turbidity fluctuations.

Information about the distribution of coral species across places with varying salinity can provide some insight into potential salinity tolerances. World-wide reef corals are reported to survive salinities from 25 to 50 ppt (Coles 1992). A total of 21 species from five families have been documented to survive where ambient salinities are in excess of 40 ppt (Table 1). Twelve of these species are present at between 2 and 11 of 12 sites surveyed by rapid visual assessment (Kospartov et al. 2006), and thus form part of the coral assemblage at Barrow Island (Chevron Australia 2009). Of the remaining 9 species, members of the same genera are present at Barrow Island for 8 species, and contribute significantly to the percentage cover of corals at most sites e.g. *Porites* spp. and *Acropora* spp.. In fact, the coral assemblages at many of the sites in the zone of high impact are dominated by *Porites* spp., which is the most highly tolerant genus to saline conditions (Table 1, Kinsman 1964, Coles 1992).

Experimental exposure to high salinity has been carried out only for corals from Florida, Hawaii, the Red Sea and Arabian Gulf; there are no central Indo-Pacific examples (Table 2). For corals from more typical oceanic environments (≈ 35 ppt) it has been suggested that the upper limit for exposures longer than a week will be from 40 to 45 ppt (Edmonson 1928, Marcus and Thorhaug 1981). This is generally confirmed by Table 2. However, tolerances to increased salinity in experimental settings vary widely depending on the species, the ambient salinity from which they were collected, and the exposure period.

Table 1: Natural Upper Salinity Tolerances of Coral Species

Coral Family	Coral Species	Upper Salinity Limit (ppt)	Location	Reference	Number of sites where species present at Barrow Is. (n = 12)	For species not present at Barrow Is., the number of species in the genus that are found at Barrow Is. (range of the number of sites each species is found at)
Acroporidae	*Acropora sp*	48	Arabian Gulf	Sheppard 1988		47 (1 to 11)
Acroporidae	*Acropora sp*	51-52	Christmas Island	In Coles 1992		48 (1 to 11)
Dendrophyllidae	*Turbinaria sp*	42-45	Arabian Gulf	Kinsman 1964		5 (2 to 8)
Faviidae	*Cyphastrea micropthalma*	50	Arabian Gulf	Sheppard 1988	10	
Faviidae	*Cyphastrea seralia*	46	Arabian Gulf	Sheppard 1988	2	
Faviidae	*Favia favus*	42-45	Arabian Gulf	Kinsman 1964	2	
Faviidae	*Favia pallida*	44-45	Arabian Gulf	Sheppard 1988	9	
Faviidae	*Favia speciosa*	48	Arabian Gulf	Sheppard 1988	7	
Faviidae	*Favites chinensis*	48	Arabian Gulf	Sheppard 1988	2	
Faviidae	*Favites pentagona*	44-45	Arabian Gulf	Sheppard 1988	4	
Faviidae	*Leptastrea purpurea*	48	Arabian Gulf	Sheppard 1988	5	
Faviidae	*Platygyra daedalea*	48	Arabian Gulf	Sheppard 1988	9	
Faviidae	*Platygyra lamellina*	42-45	Arabian Gulf	Kinsman 1964	6	
Faviidae	*Platygyra sinensis*	44-45	Arabian Gulf	Sheppard 1988	11	
Faviidae	*Plesiastrea sp*	42-45	Arabian Gulf	Kinsman 1964		1 (3)
Pocilloporidae	*Stylophora pistillata*	42-45	Arabian Gulf	Kinsman 1964	6	
Poritidae	*Porites compressa*	46	Arabian Gulf	Sheppard 1988		8 (2 to 12)
Poritidae	*Porites nodifera*	50	Arabian Gulf	Sheppard 1988		8 (2 to 12)
Siderastreidae	*Coscinarea monile*	44-45	Arabian Gulf	Sheppard 1988		1 (4)
Siderastreidae	*Psammocora sp*	42-45	Arabian Gulf	Kinsman 1964		5 (1 to 8)
Siderastreidae	*Siderastrea savingyana*	50	Arabian Gulf	Sheppard 1988		0

Table 2: Experimental Upper Salinity Tolerances of Corals

Family	Species	Ambient salinity, Location	Maximum salinity exposure	Response	Reference
Acroporidae	*Montipora verrucosa *(3)*	32 - 35, Hawaii	45	20 day exposure; at 40 ppt 50 % normal 50 % pale, at 45 ppt 100 % died	Coles 1992
Acroporidae	*Montipora verrucosa * (3)*	~ 35, Hawaii	40	30 day exposure; 100 % survival unless temperature was above 32° C.	Coles and Jokiel 1978
Faviidae	*Montastrea annularis ^ (3 spp; 1, 2 and 8 sites)*	35, Florida	40	30 hour exposure; reduced photosynthesis, respiration and photosynthesis/respiration (p/r) ratio but no mortality.	Porter et al. 1999
Pocilloporidae	*Pocillopora damicornis * (12)*	32 - 35, Hawaii	45	20 day exposure; at 40 ppt 80 % bleached 10 % pale 10 % normal, at 45 ppt 100 % died	Coles 1992
Pocilloporidae	*Stylophora pistillata * (6)*	38, Red Sea	40	30 hour exposure; reductions in photosythesis and p/r ratio, after 3 weeks 100 % died	Ferrier-Pages et al. 1999
Pocilloporidae	*Stylophora pistillata * (6)*	40 - 43, Gulf	51	20 day exposure; at 43 ppt 100 % normal, at 49 ppt and 51 ppt 100 % died	Coles 1992
Poritidae	*Porites compressa ^ (8 spp; 2 to 12 sites)*	~ 33, Hawaii	45	3 day exposure; at 37 ppt 100 % normal, at 40 ppt bleaching, at 45 ppt mortality	Marcus and Thorhaug 1981
Poritidae	*Porites compressa ^ (8 spp; 2 to 12 sites)*	40 - 43, Gulf	51	20 day exposure; at 45 ppt 100 % normal, at 49 ppt 50 % dead 40 % pale 10 % bleached, at 51 ppt 100 % died	Coles 1992
Poritidae	*Porites compressa ^ (8 spp; 2 to 12 sites)*	32 - 35, Hawaii	45	20 day exposure; at 40 ppt 70 % pale 30 % dead, at 45 ppt 100 % died	Coles 1992
Poritidae	*Porites furcata ^ (8 spp; 2 to 12 sites)*	35, Florida	45	2 hour exposure; significant drop in net photosynthesis, 4 hour exposure; no change in photosynthesis, 10 hour exposure; no change in photosynthesis. No tissue mortality after 24 hours, recovery complete within 1 week. Effects at 45 ppt less severe compared with hyposalinity, but recovery from hypersalinity more	Manzello and Lirman 2003
Poritidae	*Porites porites ^ (8 spp; 2 to 12 sites)*	~ 33, Florida	45	3 day exposure; at 37 ppt 100 % normal, at 40 ppt bleached, at 45 ppt mortality.	Marcus and Thorhaug 1981
Siderastreidae	*Siderastrea radians*	35, Florida	45	21 day exposure; decrease in photosynthesis linearly related to exposure time, few colonies had partial mortality, colonies exposed to hypersalinity less able to clear sediments from tissue, effects more severe at hyposalinity but recovery took longer at hypersalinity.	Lirman and Manzello 2009
Siderastreidae	*Siderastrea siderea*	28 - 30, Florida	42	30 day exposure to gradual increase from 32 to 42 ppt; no change in respiration, but a 25 % decrease in photosynthesis, increase from 28 to 42 ppt caused a decrease in both photosynthesis and respiration but no mortality.	Muthiga and Szmant 1987

* denotes species occurs at Barrow Island and parentheses indicate number of sites present from n = 14
^ denotes genus occurs at Barrow Island and parentheses indicate number of species in the genus that are present at Barrow Is. and the range of sites at which each species is present.

In some cases corals were able to withstand gradual increases in salinity over longer time periods (weeks), e.g. no colony mortality was observed for colonies from Florida exposed to elevated salinities, such as *Siderastrea siderea* exposed to 42 ppt, *Siderastrea radians* exposed to 45 ppt and *Porites furcata* exposed to 45 ppt, and for colonies from Hawaii such as *Montastrea annularis* exposed to 40 ppt and *Montipora verrucosa* exposed to 40 ppt. For species collected from the Arabian Gulf, no mortality occurred in *Stylophora pistillata* colonies after exposures to 43 ppt or in *Porites compressa* colonies exposed to 45 ppt. In contrast, salinites of only 40 ppt induced bleaching or mortality in some colonies of *Montipora verrucosa*, *Pocillopora damicornis*, *Porites compressa* from Hawaii, *Stylophora pistillata* from the Red Sea and *Porites porites* from Florida. Clearly more work is required to determine critical salinity levels that induce mortality for the vast proportion of species, as well as to determine at what salinity levels sub-lethal effects may occur. Experimental evidence suggests that elevated salinities of up to 40 ppt will be tolerated by many species; however, few coral species would be expected to withstand salinity levels in excess of this.

The salinity of the brine discharged from the RO plant at Barrow Island is predicted to be at around 61 ppt, however, it is expected that there will be very rapid dilution due to the small volume of output (3150 m^3d^{-1} maximum) and the use of diffusers (Preston et al. 2007). The dispersion model of brine discharge for the Barrow Island RO plant has shown the brine will dilute to within 2% of background salinity within 50 m from the discharge pipe. This water represents a salinity of less than one ppt above ambient, a level that most species would be expected to tolerate. Subsequently, it is expected that only corals within a very small distance of the discharge diffuser will be affected by the discharge.

5.1.3 Metals

Elevated concentrations of trace metals such as copper, lead, zinc, cadmium and nickel can also reduce the fertilisation success of scleractinian corals, although corals do not appear to be any more sensitive to these metals than other marine invertebrates (Reichelt-Brushett & Harrison 2005). Laboratory toxicity tests found copper was the most toxic metal, followed by lead, zinc, cadmium and nickel (Reichelt-Brushett & Harrison 2005).

5.2 Seagrass and Algal Ecosystems

Macroalgae and seagrasses are sensitive to potential impacts from RO waste brine (Hopner & Windelberg 1996). Numerous field and laboratory based experiments have investigated the impact of temperature and salinity changes on seagrass. In general, most species of seagrass are tolerant to moderate increases in salinity and temperature.

5.2.1 Salinity

A study that investigated tolerance levels of *Thalassia testudinum, Halophila engelmanni, Halodule wrightii, Ruppia maritime* and *Syringodium filiforme* exposed these aquatic angiosperms to salinities that increased by 0.75 ppt day^{-1} (McMillan & Moseley 1967). The order of decreasing hyper-saline tolerance found was *H. wrightii, T. testudinum, R. maritime* and *S. filiforme*. The tolerance of *H. engelmanni* was uncertain but viewed as higher than *S. filiforme* although lower than *H. wrightii* (McMillan & Moseley 1967). Some of these plants were able to tolerate more than 60 ppt (McMillan & Moseley 1967). Optimum growth of the seagrass, *T. testudinum* in Florida, was achieved at salinities of between 30 to 40 ppt (Kahn & Durako 2006) but a significant decrease in survival was observed at salinities above 50 ppt and 100 % mortality occurred at 70 ppt salinity (Kahn & Durako 2006). A laboratory based study of temperature and salinity tolerances of *Halophila johnsonii* found growth and survival was optimised at 30 ppt with reduced growth and increased mortality at 40 ppt and above (Fernandez et al. 2005). A study of the seagrass, *Zostera marina,* found a high tolerance to increased salinity (Biebl & McRoy 1971). The photosynthetic rate of the Australian seagrass, *Zostera muelleri,* was heavily reduced at lower salinities, but was less influenced by salinities two to four times more concentrated than sea water (Kerr & Strother 1985).

One of the main species of seagrass at Barrow Island is *Halophila ovalis*. This species has been found relatively tolerant to short-term increases in salinity (Ralph 1998). Increases of up to 150 % of seawater (i.e. ~ 52 ppt) did not lead to short-term decreases in photosynthetic rate, though increases in salinity greater than that (to 70 ppt) had immediate effects on photosynthetic systems, from which recovery was not possible.

Sargassum spp. are common macroalgal species at Barrow Island. Studies on overseas (tropical) species found that *Sargassum filipendula* and *S. pteropleuron* had quite broad salinity and temperature tolerances (Dawes 1989). Similarly, in laboratory experiments, temperate marine calcareous red algae were found to tolerate salinity of 40 ppt with no change to photosynthesis (Wilson et al. 2004).

In contrast to the above mentioned species of marine macrophytes, field and laboratory assessments found the temperate seagrass, *Posidonia oceanica,* was very sensitive to increased salinity. A significant decline in seagrass structure and health occurred at salinities of 39.1 ppt in the laboratory and 38.4 ppt in the field (Sanchez-Lizaso et al. 2008). Considering the mean ambient sea water at the reference site was at 37.7 ppt, the results suggest this seagrass was negatively impacted by a rise of only 0.71–1.52 ppt (Sanchez-Lizaso et al. 2008). Additional laboratory based experiments on *P. oceanica* also showed little tolerance to increased salinity with reduced growth at 38 ppt, 50 % mortality at 45 ppt and 100 % mortality at 50 ppt (Latoree 2005). Combined, the evidence suggests that *P. oceanica* is sensitive to minor elevations of salinity.

The literature indicates that, although growth and survival is probably optimised at natural salinity, the majority of seagrass and algal species studied will tolerate moderate increases in salinity. The potentially more sensitive *Posidonia* spp. do not occur at Barrow Island. Consequently, seagrass and algal species are unlikely to be affected by any small changes in salinity which might extend out past the small mixing zone associated with the brine outfall at Barrow Island.

5.2.2 Temperature

The temperature tolerance of seagrass has also been investigated for a range of species. A study of *Zostera marina* found a high tolerance to increased temperatures (Biebl & McRoy 1971). The seagrass was able to tolerate temperatures of 30 °C for the subtidal form and 35 °C for the tidal pool form (Biebl & McRoy 1971). These temperatures were high considering the cold climate in which the study was done. The Australian seagrass, *Zostera muelleri,* was found to exhibit an increase in photosynthetic rate up to 30 °C and a sharp decline at 42 °C (Kerr & Strother 1985). Although published research is limited, the information available suggests that seagrasses can tolerate minor temperature increases above background levels. Consequently, any slight increases in temperature associated with brine discharge at Barrow Island are unlikely to have significant widespread effects on seagrass health.

5.2.3 Metals

Exposure to heavy metals resulted in a significant reduction of photosynthesis in a species of calcareous red algae, *Phymatolithon calcareum* (Wilson et al. 2004). After one week, however, macroalgae appeared to have assimilated the contaminants and recovered (Wilson et al. 2004). Considering the predicted low concentrations of heavy metals in the Barrow Island waste brine and the rapid dilution of the plume, the impact of heavy metals is expected to be minimal.

5.3 Fish and Fishery Productivity

At least 456 species of finfish have been identified in the vicinity of Barrow Island (Chevron Australia 2009). Populations are considered stable, apart from some localized impacts on selected (site-specific) species due to fishing. The Montebello/Barrow Island Marine Management Area is used by both recreational and commercial fishermen targeting a variety of fish. There are presently no active aquaculture leases in the vicinity of Barrow Island, although pearling leases exist within the Montebello Islands and at the Lowendal Islands (Chevron Australia 2009). The potential impacts of a desalination plant outfall on fishery productivity include changes caused by habitat creation or destruction, barriers to movement, impingement and entrainment, and changes in nutrient loads.

After construction, the desalination plant intake and outflow pipes can act as artificial reefs (Latterman & Hopner 2008). It is likely structures at Barrow Island will be colonised by reef fish as the discharge pipe is proposed for construction near existing reefs. The potential for waste brine to create a barrier to movement will be limited due to the small spatial area of the brine plume, and the local geography. The plume modelling undertaken for the Gold Coast desalination plant indicated that elevated salinity will affect only a small section of the water column, predominately over the seabed, and would be unlikely to obstruct fish movement (Gold Coast Desalination Alliance ERA).

The volume of intake water and the flow rate is likely to influence the level of impingement and entrainment of fish and fish larvae. Monthly sampling of intake screens at a Florida power plant found 73 species, consisting of mostly juveniles or slow swimming fish were impinged (Grimes 1975). The plant which had a flow velocity of 0.82 m s^{-1} had a large seasonal variation in the numbers caught on filter screens, from almost zero up to over 1000 day^{-1} (Grimes 1975). Studies completed for the Sydney desalination plant concluded that 1.8 million fish larvae day^{-1} could be entrained or impinged (The Ecology Lab 2005). Although this may appear a large number, most marine organisms are highly fecund and suffer great rates of natural mortality as larvae. Consequently, the potential impacts of losses of larvae were expected to be a localised impact at most, with little effect on a regional scale. Other studies in Europe have recorded substantial mortality of adults and up to 100,000 juvenile fish per 100 ML of intake water (Hadderingh 1979). In contrast, one study in a United States river, and a second study in a United States marine ecosystem, found that power stations were not adversely affecting the fish community (Lewis & Seegert 2000; Lorda et al. 2000). Options to mitigate impingement and entrainment at desalination intakes include installing mesh screens, reducing flow velocity and ensuring the source water is drawn from an area of relatively low fish productivity (Latterman & Hopner 2008). Considering the relatively small volume of intake water proposed for the Barrow Island facility, it is unlikely that fish communities will be significantly affected by impingement and entrainment, particularly if flow speeds are low and mesh screens are installed.

The outflow of waste brine has the potential to alter planktonic productivity by altering nutrient loads and temperature. A comparison of planktonic life near a Middle East desalination plant outfall and the open ocean, found a slight increase in production of chlorophyll in winter and a decline in summer (Abdul-Azis et al. 2003). The total population of zooplankton was up to 55 % higher than the open ocean with increased abundance of Protozoa, Annelida, Arthropoda, Chordata, Appendicularia and fish eggs (Abdul-Azis et al. 2003), although aspects of the design of this study means these patterns are not necessarily attributable to effects from the desalination plant.

5.4 Threatened, Endangered and Protected Species

5.4.1 Turtles

Green, flatback, hawksbill, loggerhead and leatherback turtles may occur in the vicinity of Barrow Island (RPS 2008). Nesting flatback turtles tend to prefer the sheltered sandy beaches of the east coast of Barrow Island, whereas green turtles tend to prefer to nest on the exposed beaches on the western side of the islands (RPS 2008). The waters around the island provide a feeding ground for green turtles and possibly also provide habitat for juvenile flatback turtles. The other species are rare.

Ocean intakes located in areas of high turtle abundance pose a risk of entrainment to adults, hatchlings and juveniles (Pendoley 2007). A review of the effects of the petroleum industry on sea turtles concluded that there have been no studies into the impact of increased salinity, reduced dissolved oxygen or increased turbidity on sea turtles in (Pendoley 2007). While the risks were viewed as low, a potential link between agricultural, urban and industrial discharges that reduce water quality (e.g. temperature, salinity, contaminants) and the disease fibropapilloma in sea turtles was noted (Pendoley 2007). The review found no data on the physiological effects of heavy metals or organochlorines on sea turtles (Pendoley 2007).

Sea turtles regularly undertake long-range migrations and can navigate back to a specific location (Avens et al. 2003). It has been hypothesised that turtles use a range of navigational cues including: magnetic fields, wind-born odours visual landmarks, aquatic chemical cues, solar and celestial navigation. It has also been suggested that these cues may function sequentially over different spatial scales to assist navigation (Lohmann et al. 2008; Pendoley 2007). While turtles can detect chemicals in water, it is not known whether turtles use chemical cues to navigate (Manton et al. 1972; Avens & Lohmann 2003). The scientific literature does not currently include specific experiments into the impact of waste brine effluent on the health or navigation of sea turtles.

Some turtle migration paths are almost a direct line between two points over many hundreds of kilometres, in currents that appear unsuitable for water based chemoreception-based navigation (Papi et al. 1995; Luschi et al. 1996). Research at Ascension Island raised the possibility that direct routes were possibly determined by airborne chemoreception, as most of the direct migratory paths were from down-wind of the island and not down-current (Luschi et al. 2001). In laboratory experiments without chemical cues it was found that turtles can maintain direction when either their vision or magnetic senses are available, but when both are impacted the turtles' orientation changes (Avens & Lohmann 2003). In further laboratory experiments translocated turtles were found to orient themselves toward their capture location (Avens & Lohmann 2004; Lohmann et al. 2004). Similar results have been found in experiments on hatchlings (Lohmann & Lohmann 1996). The orientation without chemical or visual cues suggests turtles are capable of navigation with alternate cues, so even if waterborne cues are used by turtles, discharge from a RO plant is unlikely to prevent successful navigation at Barrow Island.

5.4.2 Marine Mammals

The Barrow Island area supports both resident and migratory marine mammals (RPS Bowman Bishaw Gorham 2007). At least ten species of cetaceans have been recorded in the waters surrounding Barrow Island. The main migration route for humpback whales off the west coast of the island is the deeper water. Dugongs are found in the shallow warm waters around Barrow Island, albeit in relatively low abundance. Any impacts of the RO plant on marine mammals are more likely to be from the construction of pipelines rather than the relatively small discharge plume.

The Gold Coast desalination plant Ecological Risk Assessment suggested that marine mammals are unlikely to be impacted by the construction of the desalination plant because they are large, highly mobile species able to avoid areas of disturbance (Gold Coast Desalination Alliance ERA). Similarly, the marine noise associated with the construction of the Sydney desalination plant was expected to cause cetaceans to temporarily avoid the area of construction, but their migration was not expected to be affected due to their wide migratory paths (The Ecology Lab 2005). Dugongs are also known to make long distance movements and are therefore able to avoid areas of disturbance (Hobbs et al. 2007). Accordingly, the construction of a relatively small desalination plant proposed for Barrow Island is unlikely to have a measurable impact on marine mammal populations.

A literature review on the effects of desalination plant brine upon cetaceans also concluded there is currently no information to suggest brine discharge will have a negative effect on cetacean health, but noted effects have not been studied specifically at existing RO desalinisation plants (URS 2008).

6.0 CONCLUSION

Research on the impact of desalination plant construction and discharge on the marine environment is limited. Nevertheless, studies have been completed which provide some background and insight into the potential impacts on marine ecosystems. In addition, several Australian research projects are underway, which when completed will greatly increase the knowledge base to assess the impacts of desalination plant discharges on the physical, chemical and biological properties of marine ecosystems.

The proposed Barrow Island facility will produce a relatively small volume of waste brine compared to many desalination plants in operation (Talavera & Ruiz 2001; Fernandez-Torquemada et al. 2005; Raventos et al. 2006). The smaller volume of waste brine proposed for release in conjunction with the predicted rapid dilution at the Barrow Island facility supports the view that near-background levels of salinity, temperature, pH and chemicals are likely to be achieved very close to the outfall and that it is unlikely that the Barrow Island brine outfall will have a significant impact on the water quality, benthic habitats, fish, sea turtles or marine mammal populations of Barrow Island. The installation of screens, combined with low flow rates will mitigate many impingement and entrainment issues.

Although the relatively small capacity and careful environmental management of the facility is likely to limit the impacts, a monitoring program is recommended; to test the impact predictions and to provide a basis for future decision making.

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