# 3 Sub-tidal Reef Invertebrate and Algal Community

Sub-tidal rocky reef habitat is among the most productive habitat in the marine environment characterised by a diverse assemblage of mobile and sessile invertebrates and macroalgae. These reefs are of great value, for their intrinsic worth and because they support important commercial and recreational fisheries (i.e. abalone, lobster). The purpose of this section is to describe the invertebrate and algal community associated with sub-tidal rocky reef habitat within the study area and to assess potential impacts from the Project construction and operational phases.

## 3.1 Background

Ecological impact from the discharge of STP wastewater to sub-tidal reef assemblages has been well studied around the world (Puente and Diaz, 2015) with impact reported as change in community species composition typically detected in close proximity to the discharge or limited to within the wastewater mixing zone (Brown et al. 1990; Fairweather 1990; Smith and Simpson, 1993; Smith, 1996; Roberts et al. 1998; CEE, 2016). Wastewater components that may pose a threat to sub-tidal communities include suspended sediments, particulate organic matter (POM) and the toxic contaminants absorbed to particles. The freshwater discharge, dissolved nutrients and dissolved organic matter are of greater concern to pelagic communities of the water column although can become a concern for sub-tidal communities if they trigger prolific growth of phytoplankton that then deliver additional supply of POM to the benthos.

The objectives of this assessment were to:

- Describe the sub-tidal habitats and communities within the study area to address the Project SEARs; and
- Document species composition and identify community components that may be useful indicators to inform potential ongoing monitoring during post commissioning operational phase of the Project.

The assessment was based upon a review of existing information, a Stage 1 quantitative field survey of shallow sub-tidal reef habitat of Haycock Point (5 m – 15 m depth), and a Stage 2 quantitative field survey of intermediate sub-tidal reef habitat at Hunter Reef, Outer Haycock Point and Long Point (25 m – 35 m depth). This assessment does not include a description of fish associated with sub-tidal reefs as that is provided in **Section 5 – Fish Assemblage**.

## 3.2 Review of Existing Information

### 3.2.1 Regional pattern of sub-tidal reef habitat types

Sub-tidal rocky reef in south-eastern NSW is characterised by a large scale pattern of distinct and definable habitat types, the distribution of which are related to depth, physical disturbance (i.e. wave exposure) and the grazing pressure of herbivorous invertebrates (Underwood et al. 1991, Andrews et al. 1998, Andrew and O'Neill, 2000; DECCW 2010). Five habitats broadly described include fringe habitat, barrens, kelp forest, turf habitat, and sponge gardens (also referred to as sessile filter-feeder community). Examples of some of these habitats (barrens, turf, sessile filter-feeder) in addition to mussel beds were observed on Hunter Reef during broadscale surveys of the study area (refer **Section 2 – Overview of Marine Habitats**) and it was anticipated that most of these habitats would be represented in various proportions on reefs associated with headlands of Haycock Point and Long Point. A broad description of each is provided below.

**Fringe habitat** – occurs from mean low water to approximately 3 m depth and is characterised by a variety of red, green and brown macroalgae. Species composition of fringe habitat is strongly influenced by level of exposure to wave action and ocean swell and herbivorous molluscs such as gastropods, chitons, limpets are also important to the structure of this habitat.

**Barrens habitat** – is a dominant feature of shallow rocky reefs in southern NSW that typically occurs between 3 m to 20 m depth and is characterised by reef devoid of large foliose macroalgae due to the grazing pressure exerted primarily by dense populations of sea urchins (*Centrostephanus rodgersii*) but also other gastropods and herbivorous fishes. The grazing pressure results in bare reef covered in patches of adherent pink, coralline algae. Barrens are characterised by lower species diversity than other sub-tidal habitat types and it has been estimated that up to 50-60% of shallow nearshore reefs in some areas of southern NSW are barrens habitat (Andrews and O'Neill, 2000). The extent and persistence of barrens therefore has important implications for the ecology and biodiversity of shallow reefs including fish, invertebrates and commercially important species such as abalone that rely on dense and luxuriant macroalgal assemblages (Andrew and Underwood, 1992; Ling 2008, Edgar et al. 2009).

**Kelp forest habitat** – is characterised by dense stands of canopy-forming algae typically of *Ecklonia radiata*, with other canopy-forming species including *Phyllospora comosa, Durvillaea, Sargassum* and *Cystophora* dominant in some areas. Kelp forests are among the most productive habitats of rocky reefs that provide structure and micro-habitats for crustacea, worms, bryzoans, ascidians, sponges, and fish.

**Sponge gardens (sessile filter-feeders) habitat** – refers to a diverse community of sessile filter-feeding invertebrates that become the dominant habitat in depths below 20 m and includes sponges, colonial and stalked ascidians (i.e. sea tulip), soft corals (i.e. gorgonian sea fans) and sea whips, bryozoans, and hydroids.

**Turf habitat** – is a distinctive macroalgal habitat that is found at a range of depths and characterised by small, erect and semi-prostrate macroalgae such as coralline red algae and prostrate brown algae like *Lobophora* and *Zonaria*. Turf habitat is common on reefs that are intermittently impacted by sand and is often found at the margins of sub-tidal reef adjacent to sand.

### 3.2.2 Other studies of treated wastewater impacts on sub-tidal reef assemblages

A number of studies in NSW and Tasmania have examined the potential impacts of treated wastewater on sub-tidal reef assemblages. Only those studies of smaller regional and municipal STPs are considered relevant to this Project such as Coffs Harbour (Smith and Simpson, 1993; Smith, 1996; CEE, 2000), Sydney northern beaches and Central coast region (Roberts, 1996; Roberts and Scanes, 1999), Port Stephens (Roberts et al. 1998; Ajani et al. 1999), Milton-Ulladulla (The Ecology Lab, 2008) and Devonport (CEE, 2016).

Studies have used a range of monitoring indicators including:

- Abundance and size population structure of *Ecklonia* kelp (Ajani et al. 1999; Roberts and Scanes, 1999).
- Macroalgal community composition and species cover (Brown *et al.* 1990; Fairweather, 1990; CEE, 2016; CEE, 2000; Robert and Scanes, 1999; The Ecology Lab, 2008).
- Faunal community composition associated with *Ecklonia* kelp (Smith and Simpson, 1993; Smith, 1996; Roberts *et al.* 1998, Roberts and Scanes, 1999; CEE, 2016).

Smith and Simpson (1993) and Smith (1996) were able to attribute changes in the faunal assemblage associated with *Ecklonia* kelp holdfasts to the discharge of wastewater. The study found an increase in the abundance of filter-feeders living in holdfasts adjacent to a wastewater release, but a decrease in brittle stars, worms, and crustaceans. During the eight-year study, impacts attributed to the outfall were restricted to within approximately 300 m of the outfall for most variables (i.e. algal species richness, changes to the structure of invertebrate communities living in kelp holdfasts). However, the ephemeral green alga *Ulva lactuca* had a significantly greater cover for a distance of 500 m from the point of discharge, than at reference sites (Smith, 1996).

Roberts et al. (1998) reported a rapid change in cover and species composition of encrusting macrobenthic assemblages of temperate rocky reefs in the vicinity of the Boulder Bay outfall in Port Stephens. The study recorded data prior to outfall commissioning, immediately post-commissioning and one-year post commissioning. Within three months of the outfall commissioning of the outfall, significant reductions in the cover of crustose and foliose algae were apparent at the outfall location when compared to control (Port Stephens and Tomaree Head) locations. The cover of several species of sponge, including *Cymbastela concentrica*, *Geodinella sp*. and *Spongia sp.*, also underwent marked declines coincident with the commissioning of the extended outfall. However, declines in the cover and number of species of sponges or total fauna did not change significantly. After commissioning of the outfall, the cover of a nondescript matrix comprising silt and microorganisms doubled its representation to almost 60%. The overall community composition adjacent to the outfall changed from one in which macroalgae and sponges were well represented, to an assemblage dominated by silt and ascidians.

The effects of treated wastewater on macroalgae can be variable including shifts in community composition (Brown *et al.* 1990, Fairweather 1990) to loss of species cover (Roberts et al. 1998) or increase in species cover (CEE, 2016) although reported effects have been site and species-specific. Some studies have demonstrated the vulnerability and sensitivity of microscopic life history stages of some canopy forming brown algal taxa belonging to Orders Fucales and Laminariales to wastewater leading to reduced abundances and mortality (Bellgrove et al. 1997; Burridge et al. 1996). In contrast, high levels of dissolved nutrients in wastewater may favour the growth and abundance of some species that include fast-growing opportunistic macroalgal taxa (i.e. *Ulva lactuca* as reported by Smith 1996) including those with potential to form nuisance blooms.

A long-term study of a primary treated wastewater outfall at Devonport, Tasmania identified a positive relationship between *Ecklonia* abundance, with higher abundance and cover reported in proximity to the discharge of primary treated wastewater, concluding the nutrient rich wastewater was favourable for this species (CEE, 2016). The authors identify that the canopy-forming *Ecklonia* can also have a significant interaction with other seabed associated species including other algae, sessile invertebrates, mobile invertebrates and fish. For example, an area of dense *Ecklonia* canopy can provide structural complexity as refuge and cover from some biota but can exclude other species by shading, competing for space or physically abrading the seabed below the canopy and removing small epibiota (CEE, 2016).

For the other studies mentioned above (CEE, 2000; Ajani *et al.* 1999; Roberts and Scanes, 1999; The Ecology Lab, 2008), none detected a negative effect to sub-tidal reef assemblages from wastewater discharge. This suggests that for those studies, wastewater discharge either did not impact the sub-tidal reef communities or that the effects were small or occurred at spatial scales that their experimental designs were not able to detect.

## 3.3 Stage 1 survey of shallow sub-tidal reef habitats

An initial quantitative field survey of shallow sub-tidal reef habitats was undertaken at Haycock Point during Stage 1 marine ecology investigations in 2017. The rationale for surveys at Haycock Point (at that time) was that a location east of Haycock Point was among the potential options being considered for the pipeline and outfall diffuser (**Figure 1-3**) and sub-tidal communities at Haycock Point may be at risk of impact.

### 3.3.1 Approach

Field surveys were conducted using a remote sampling method that utilised a drop camera quadrat frame deployed from the workboat. The quadrat frame was used (as an alternative to divers) to sample and characterise the community composition of sub-tidal habitats along transects at Haycock Point. The quadrat frame was deployed from the workboat lowering it to the seafloor using a mechanical winch. High-resolution video footage from the camera was viewed on the boat simultaneously such that the frame could be lowered and raised to the seafloor using the winch. A high resolution still image of the 1 $m^2$ frame base was recorded every 10 seconds or at approximately 2 m intervals (depending on boat drift) along transect (**Figure 3-1**).

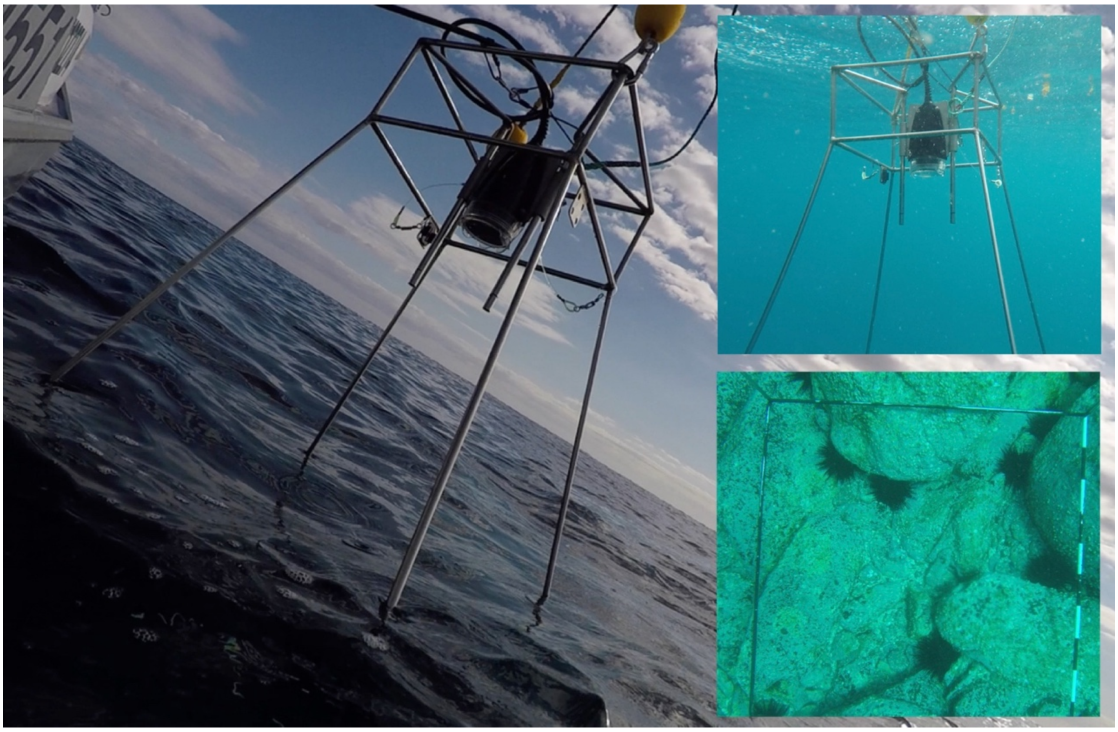


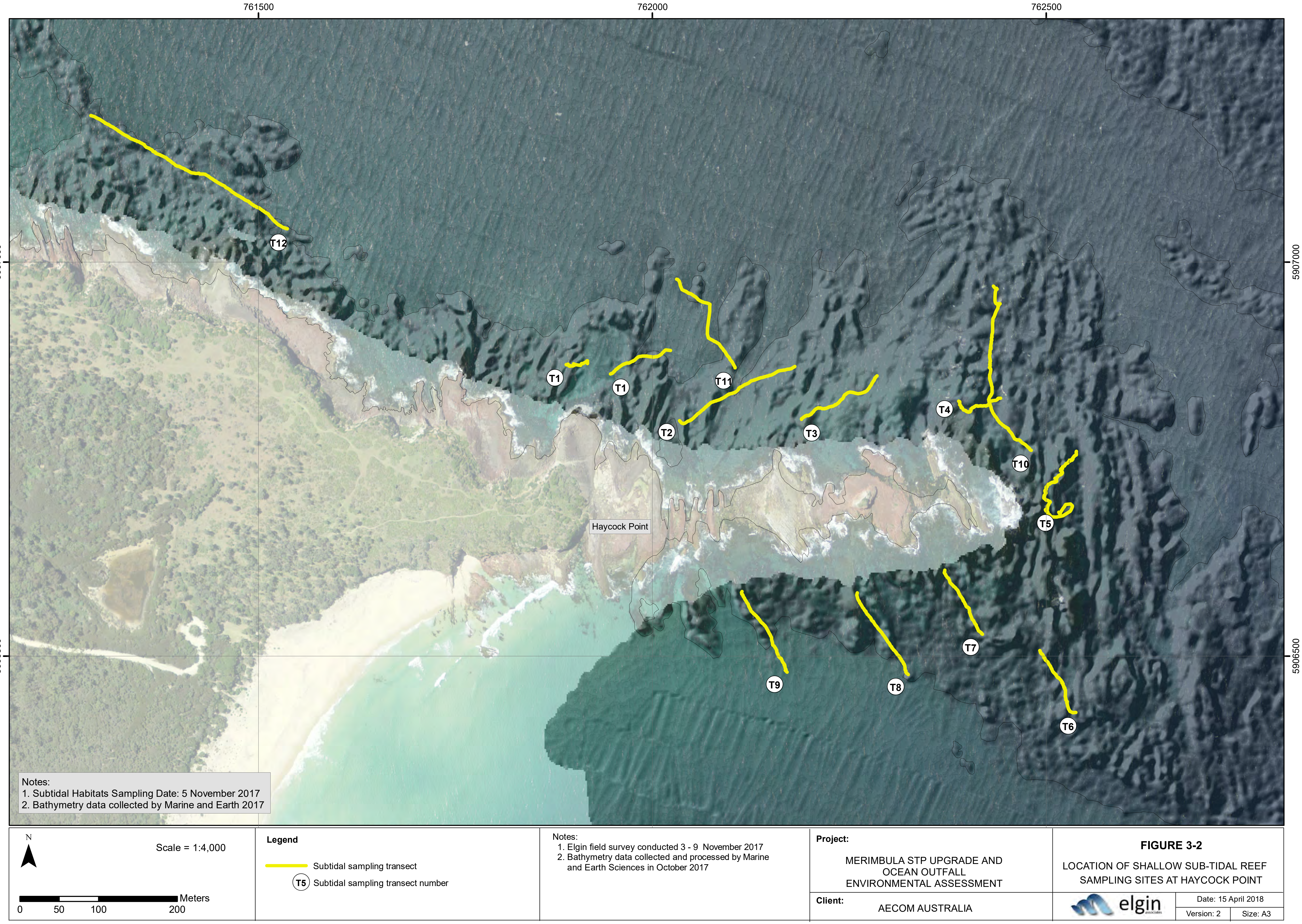
**Figure 3-1 Drop camera frame used to survey sub-tidal reef habitats**

**Sample sites**
Data was collected randomly from along eleven transects distributed around Haycock Point in depths ranging between 4 m to 20 m (**Figure 3-2**). Transects were surveyed according to the drift direction of the workboat. Consequently, transects were oriented perpendicular to the shoreline and as a result, multiple habitat types were encountered along each transect with increasing depth.

**Image analysis**
Images that were not grounded on the seafloor, out of focus, or had greater than 90% sand cover were discarded. A subset of 331 quadrat images from eleven transects were used for analysis of macroalgal cover and abundance of herbivorous invertebrates.

Image analysis was undertaken using software CpCE to record the following variables:

- Percent cover of macroalgae, discriminating taxa to species level where possible, and substrates quantified using a point-intercept (digital grid overlain on photos) method to estimate percentage (%) cover.
- Counts of conspicuous mobile invertebrate fauna (i.e. urchins, abalone, other gastropods such as *Turbo* and *Astralium*).




**Legend**

Subtidal sampling transect

T5 Subtidal sampling transect number

Notes:
1. Elgin field survey conducted 3 - 9 November 2017
2. Bathymetry data collected and processed by Marine and Earth Sciences in October 2017

**Project:**

MERIMBULA STP UPGRADE AND OCEAN OUTFALL ENVIRONMENTAL ASSESSMENT

**Client:**

AECOM AUSTRALIA

**FIGURE 3-2**

LOCATION OF SHALLOW SUB-TIDAL REEF SAMPLING SITES AT HAYCOCK POINT

elgin associates

Date: 15 April 2018

Version: 2

Size: A3

**Data analysis**

Data analysis was undertaken using statistical routines in software PRIMER that included cluster analysis to identify the range of habitat types present at Haycock Point, with further analysis of community composition undertaken using MDS ordinations and SIMPER in PRIMER. Further details are provided in **Appendix A-1**.

### 3.3.2 Results

A total of 46 taxa were identified from shallow sub-tidal habitats at Haycock Point with a list of taxa provided in **Appendix A-2.**

**Habitat Types**

Cluster analysis of 331 randomly collected quadrats resulted in five distinct groups at 40% similarity as shown in the dendrogram in **Appendix A-1**. These five groupings represent the habitats – barrens, *Ecklonia* forest, mixed algae fringe, turf, and deep reef habitat. Barrens, *Ecklonia* and turf habitats were adequately sampled, and reliable descriptions of community composition can be provided. In contrast, mixed algae-fringe, and deep reef habitats were represented by fewer samples (i.e. quadrats) and description of community composition for those habitats provided below is considered indicative.

Other habitats, namely *Pyura* and *Phyllospora* habitat, are also present at Haycock Point although these habitats occur in the shallow sub-tidal zone (1 – 3 m depth) and could not be safely sampled using the drop camera frame method due to swell action and proximity to reef. Images of some of the shallow sub-tidal habitats present at Haycock Point are provided in **Figure 3-3**.

Broad comparison among sub-tidal habitat types is provided below by:

- Mean species richness (**Figure 3-4**);
- Mean percent (%) cover of major community components (**Figure 3-5**); and
- Mean abundance of grazing invertebrates (**Figure 3-6)**

**Species Richness**

*Ecklonia*, turf, and barrens habitat were characterised by the highest number of species (**Table 3-1**) although many species within these habitats were recorded in few samples and considered rare. Consequently, overall mean species richness was relatively low for these habitats. Mixed algae-fringe habitat was characterised by highest mean number of species and deep reef samples the lowest mean number of species (**Figure 3-4**).

**Table 3-1 Summary of species richness (S) for sub-tidal habitats**

| Habitat type | Sample n | Total S | Mean S | SE |
|---|---|---|---|---|
| *Ecklonia* forest | 127 | 18 | 3.49 | 0.11 |
| Barrens | 161 | 13 | 2.60 | 0.09 |
| Turf | 25 | 13 | 2.56 | 0.29 |
| Mixed algae fringe | 4 | 6 | 4.75 | 0.48 |
| Deep reef | 12 | 5 | 1.83 | 0.27 |

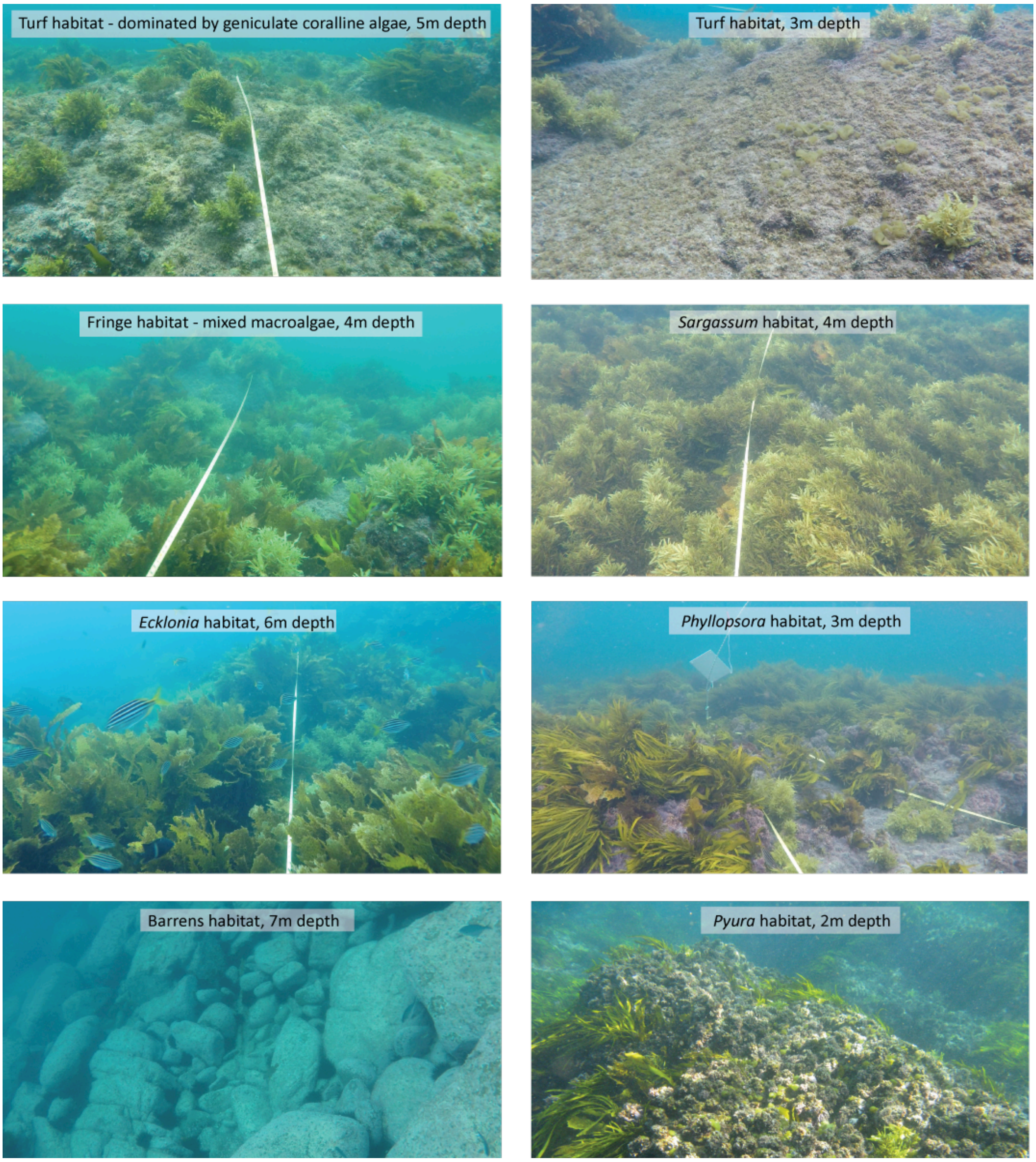



**Figure 3-3**. Example of sub-tidal habitat types observed at Haycock Point. Images were collected during abalone surveys in January 2018 (reproduced from Elgin, 2018).

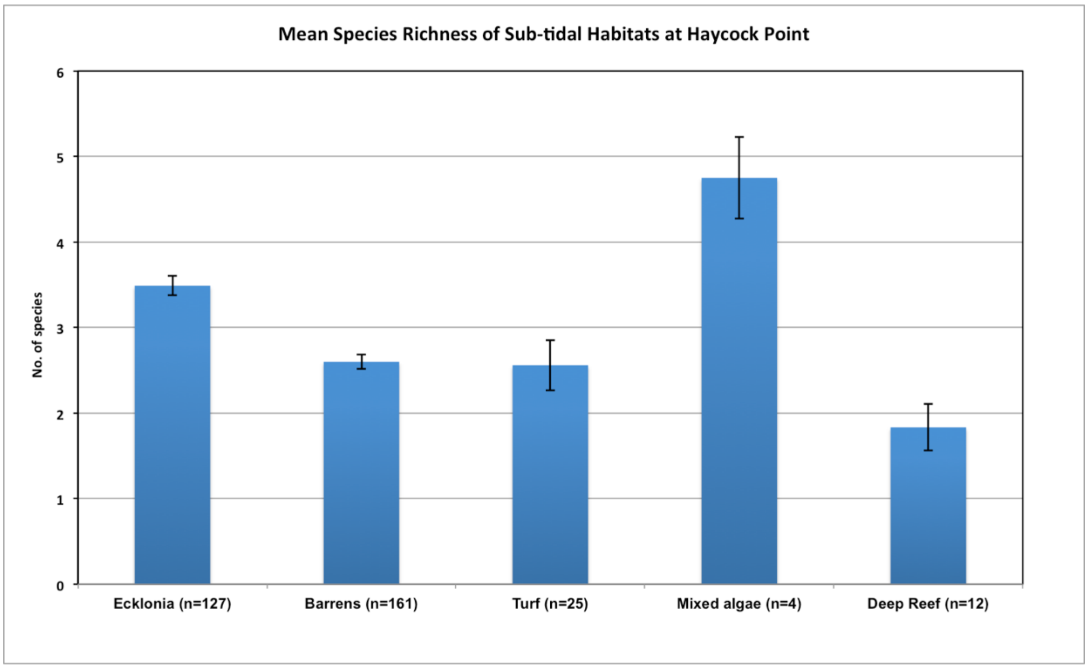



**Figure 3-4 Mean species richness of sub-tidal habitats observed at Haycock Point**

**Percent Cover (%) and major community components**

Proportions of major community components (excluding mobile invertebrates) as percentage cover for each habitat type is provided in **Table 3-2** and shown in **Figure 3-5.** Macroalgae is the dominant community component in mixed algae-fringe, *Ecklonia* and turf habitat. In contrast, barrens habitat is characterised by high proportion of bare rock and low algal cover. Deep reef is characterised by similar proportions of sand and algal cover. In terms of percent cover, sessile invertebrates that include sponges, bryozoans, hydroids and colonial cnidarians are a minor component within all habitat types.

**Table 3-2 Percentage cover (%) of major community components for sub-tidal habitats**

| Habitat type | Sample n | Bare Rock (%) | Sand (%) | Macroalgae (%) | Sessile invertebrates |
|---|---|---|---|---|---|
| *Ecklonia* forest | 127 | 3 | 6 | 87 | 4 |
| Barrens | 161 | 85 | 2 | 8 | 5 |
| Turf | 25 | 12 | 24 | 57 | 7 |
| Mixed algae fringe | 4 | 0 | 0 | 100 | 0 |
| Deep reef | 12 | 0 | 54 | 42 | 4 |

Note – sessile invertebrates includes sponges, bryozoans, hydroids and colonial cnidarians

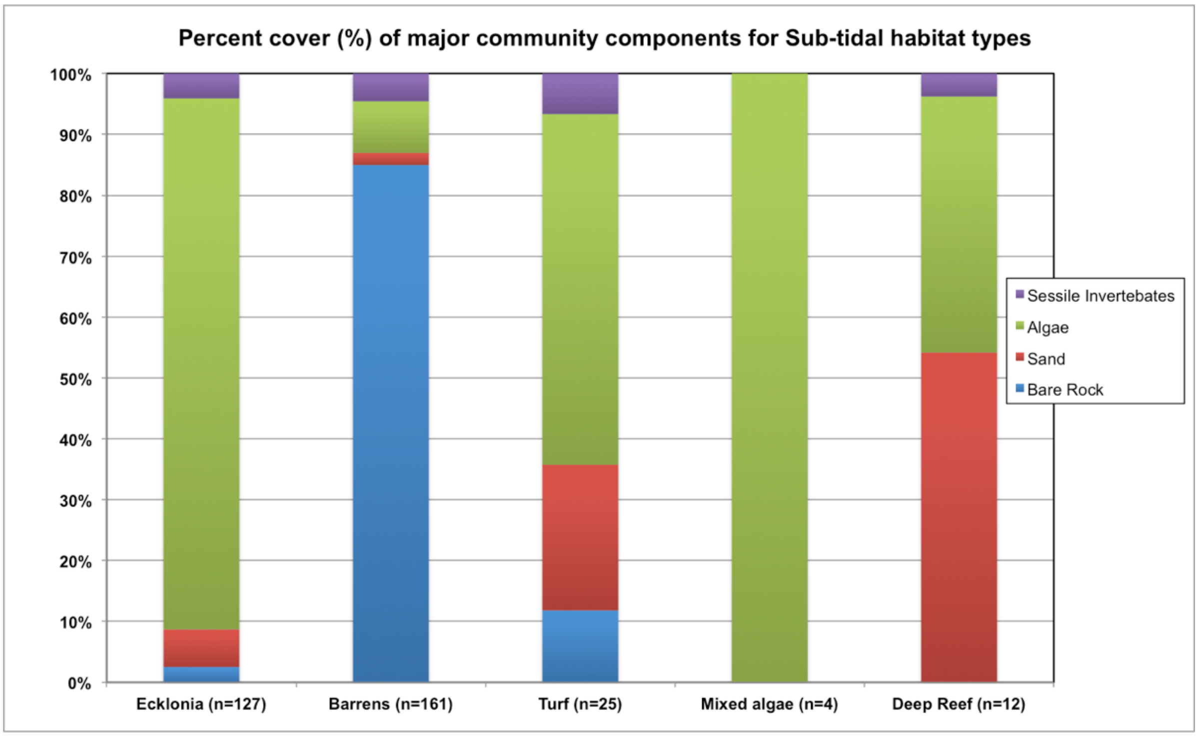



**Figure 3-5 Mean percentage cover (%) of major community components for sub-tidal habitats at Haycock Point**

**Abundance of herbivorous invertebrates**

Abundance of mobile herbivorous invertebrates as mean number of individuals per 1m$^2$ for each habitat type is provided below in **Table 3-3** and **Figure 3-6**. Barrens habitat was characterised by the highest mean abundance of herbivorous invertebrates with low numbers found in *Ecklonia* and turf habitat and zero herbivorous invertebrates found in mixed algae-fringe and deep reef. It is acknowledged that the low abundance and diversity of herbivorous invertebrates recorded in *Ecklonia* and zero invertebrates in mixed algae-fringe habitats, is likely an artefact of the drop camera quadrat method as cryptic species and individuals occurring below the canopy of algae are not visible in quadrat photographs. From visual observations of the algal canopy understorey by SCUBA divers, it is known that other mobile herbivores such as abalone (*Haliotis rubra*), *Dicathais orbita* and turban shell (*Lunella torquata*) are present in varying abundances but would have been obscured by the algal canopy and therefore not visible within the drop camera images. As many as three limpet species (*Cellana tramoserica, Scutellastra chapmani* and *Patelloida insignis*) occur within the habitats described below although due to their small size could only be identified as 'Limpet' from quadrat photographs. Abalone abundance at Haycock Point was subject to its own specific survey by SCUBA, with results presented separately in **Section 4**.

The absence of herbivorous invertebrates from deep reef habitat sampled may be attributed to its proximity to sand with no bare rock recorded within quadrats that is a preferred substrate for grazing invertebrates such as urchins, limpets and gastropods.

**Table 3-3 Mean abundance of herbivorous invertebrates (individuals per m²) for sub-tidal habitats**

| Habitat type | Sample n | Urchins | Gastropods | Limpets |
|---|---|---|---|---|
| *Ecklonia* forest | 127 | 0.64 | 0.03 | 0.09 |
| Barrens | 161 | 6.84 | 0.23 | 14.21 |
| Turf | 25 | 0.12 | 0.24 | 0.08 |
| Mixed algae fringe | 4 | 0 | 0 | 0 |
| Deep reef | 12 | 0 | 0 | 0 |

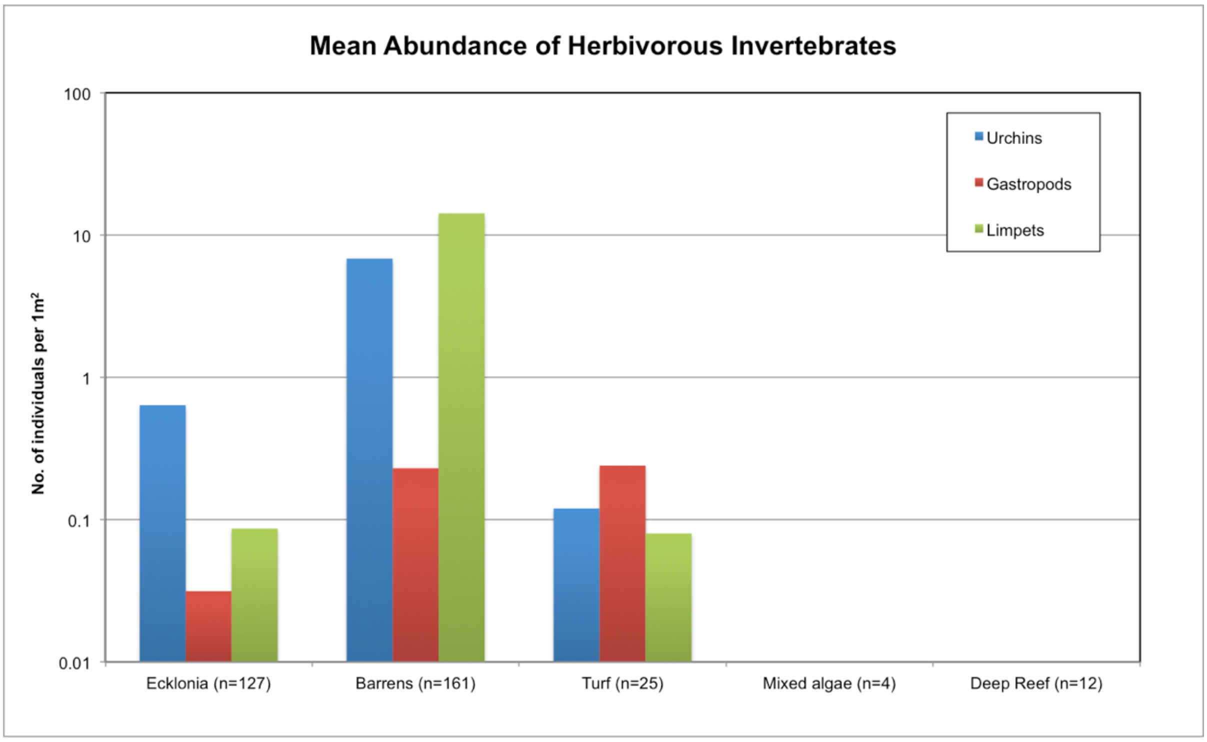



**Figure 3-6 Mean abundance (individuals per m²) of herbivorous invertebrates within sub-tidal habitats at Haycock Point (*note: Y axis displayed as log scale*).**

Further description of each of the sub-tidal habitats observed at Haycock Point is provided in **Appendix A.**

### 3.3.3 Key Findings

Key findings from the Stage 1 survey of shallow sub-tidal reefs at Haycock Point include:

- The sub-tidal reef community of Haycock Point is comprised of at least seven (7) habitat types that have been previously described along the NSW coast (Underwood et al. 1991). Five habitats were sampled including *Ecklonia* forest, barrens, turf, mixed algae-fringe and deep reef habitat. An additional two habitat types, *Pyura* and *Phyllospora* forest habitat, occur in shallower waters but were inaccessible for sampling due to safety reasons of the survey method employed.
- Barrens and *Ecklonia* habitat were recorded in 48% and 38% of total samples respectively and are the most dominant habitats at Haycock Point occurring in depths between 4 to 19 m. The relative

dominance of barrens habitat supports earlier observations by Andrew *et al.* (1998) that found up to 60% of the near shore reef at Haycock Point was comprised of barrens habitat. However, Andrew *et al.* (1998) observed less *Ecklonia* habitat at Haycock Point than was observed during this study.

- *Ecklonia* habitat is the most species-rich comprised of at least 18 species, and with the exception of mixed algae-fringe habitat, also had the highest mean species richness of 3.5. *Ecklonia* habitat is characterised by four dominant algal taxa that includes the canopy forming *Ecklonia* kelp, an understorey of turfing algae (comprising varying proportions of small brown algae such as *Sphacelaria* sp., *Halopteris* sp., *Zonaria* sp., *Lobophora variegata*, and *Padina sp)*, geniculate coralline algae (*Corallina officinalis* and *Amphiroa anceps*) and *Sargassum* spp. Low numbers of conspicuous mobile invertebrates were recorded in *Ecklonia* habitat including the long-spined urchin *Centrostephanus rodegrsii* with mean density of 0.64 urchins per $m^2$.

- A total of 13 species were recorded from barrens habitat with mean species richness of 2.6. Barrens habitat is dominated by bare rock (mean cover 85%) with sessile invertebrates (sponge and bryozoans) and turfing algae the next most dominant community components with mean cover of 4% and 6% respectively. Other minor community components recorded within barrens habitat represented by less than 1% mean cover include foliose algae *Ecklonia radiata*, *Cystophora moniliformis, Sargassum* sp., non-geniculate coralline algae and geniculate coralline algae (*Corallina officinalis* and *Amphiroa anceps*). Encrusting non-geniculate coralline algae (appears as pink rock) is the most common algae in barrens habitat, as it is able to tolerate the intense grazing pressure of urchins and other gastropods. High numbers of conspicuous mobile invertebrates were recorded in barrens habitat including the urchin *C. rodgersii* with mean density of 6.8 urchins per $m^2$. The abundance of *C. rodgersii* observed in barrens habitat in this study is comparable with previous surveys at Haycock Point where over 50 individuals per 10 $m^2$ were recorded (Andrew and O'Neill 2000) equivalent to 5 urchins per 1 $m^2$.

- Species accumulation curves indicate twenty-five (25) 1 $m^2$ quadrats is sufficient sampling effort for describing community structure of sub-tidal reef habitats.

- Five mobile invertebrates were recorded from sub-tidal habitats including the long-spined urchin *C. rodgersii*, carnivorous gastropods *Dicathais orbita* and *Cabestana spengleri*, herbivorous gastropod *Astralium tentoriiforme,* and 'limpets'. As many as three limpet species occur including *Scutellastra chapmani, Cellana tramoserica* and *Patelloida insignis* though due to their small size these were identified collectively as 'limpets'.

- The highest mean abundance of mobile invertebrates was found in barrens habitat with low numbers found in *Ecklonia* and turf habitat and no mobile invertebrates recorded in mixed algae-fringe or deep reef. It is acknowledged that the low abundance and diversity of mobile invertebrates recorded in *Ecklonia* and mixed algae-fringe habitat is likely an artefact of the drop camera quadrat method as cryptic species and individuals occurring below the canopy of algae are not visible in quadrat photographs. From visual observations of the algal understorey by SCUBA divers, it is known that other mobile grazers such as abalone (*Haliotis rubra*) and turban shell (*Lunella torquata*) are present in varying abundances. Abundance of abalone at Haycock Point is discussed in **Section 4**.

- Ongoing monitoring for potential impacts to sub-tidal reef communities by this project needs to consider the naturally variable species assemblages associated within each of the habitats that, collectively, comprise the shallow sub-tidal reef community. Sampling should be stratified and undertaken within the same habitat type at multiple locations. As *Ecklonia* habitat is the most common algal habitat at Haycock Point and likely to be present at reference locations, it was considered a possible indicator for future monitoring should dispersion modelling for the preferred outfall option indicate that sub-tidal habitats at Haycock Point are at risk of potential impact.

## 3.4 Stage 2 surveys of intermediate reef habitat

Further surveys of sub-tidal habitat were initiated in Stage 2 following selection of the North-Short outfall option as the preferred outfall location in 30 m depth in central Merimbula Bay and peer review of the Stage 1 report (Elgin, 2018) that identified sessile filter-feeding assemblages occurring between 25 and 35 m depth as a potential sensitive receptor to wastewater discharge noting that the findings from dispersion modelling were not known at the time.

As has been demonstrated in **Section 2**, depth is an important factor influencing the structure of marine communities and sub-tidal reef habitats can be broadly divided by the three depth zones - shallow reef (0-20 m depth), intermediate reef (20 – 60 m depth), and deep reef (greater than 60 m depth). While Stage 1 provided a description of sub-tidal habitats on shallow reef, the objective of Stage 2 surveys was to describe the community composition of the sessile filter-feeding invertebrate community that is the found on intermediate reefs within the study area.

### 3.4.1 Reconnaisance surveys

Reconnaissance surveys were undertaken in December 2019 to locate appropriate monitoring locations for the Stage 2 survey. Locations inspected included Short Point, Long Point, Hunter Reef, Outer Haycock Point, and Lennards Island.

Three monitoring locations were adopted for survey including Hunter Reef as the location closest to the proposed outfall diffuser, and Long Point, Outer Haycock Point as suitable reference locations based on distance from the Project area and presence of sessile filter-feeding community on intermediate reef.

### 3.4.2 Stage 2 preliminary findings

Stage 2 surveys are ongoing with final data analysis pending at the time of this report. Some preliminary qualitative findings based on data collected to date is provided below.

The sub-tidal community of intermediate reef is considered a single broad category comprising all the sessile filter-feeding invertebrates and is often referred to as ‘sponge gardens’ as sponges are a conspicuous and typically the most species-rich component of the community. However, in addition to sponges, the community also includes ascidians, bryozoans and soft corals. Example photo quadrats of the community recorded from each of the monitoring locations is provided in **Figure 3-7**. A preliminary list of taxa identified from the first two monitoring rounds is included as **Appendix A-3.**

The three monitoring locations support similar species assemblages with the following preliminary observations:

- Hunter Reef is predominately a continuous unbroken reef with small areas of sand, dominated by sponges (particularly encrusting morphologies) and non-geniculate encrusting coralline algae. Hard fenestrate bryozoans (e.g. lace bryozoan, *Triphyllozoon moniliferum*) are common.
- Outer Haycock Reef is a broken reef composed of large boulders and cobbles with areas of sand. Sponges are the dominant with a diverse range of morphologies recorded (encrusting, erect, massive, ball and branching). Cnidarians (e.g. sea-whip, *Primnoella australasiae*, southern sea-fan, *Sphaerokodisis australis*), ascidians (e.g. sea-tulip *Pyura spinifera*, basal sea-squirt, *Cnemidocarpa pedate*) are also abundant.
- Long Point is dominated by sponges (particularly encrusting morphologies) on sections of continuous reef, with cnidarians (e.g. sea-whip, *Primnoella australasiae*), ascidians (e.g. sea-tulip, *Pyura spinifera*) and non-geniculate encrusting coralline algae more commonly observed on the reef margins composed of cobbles and smaller boulders.

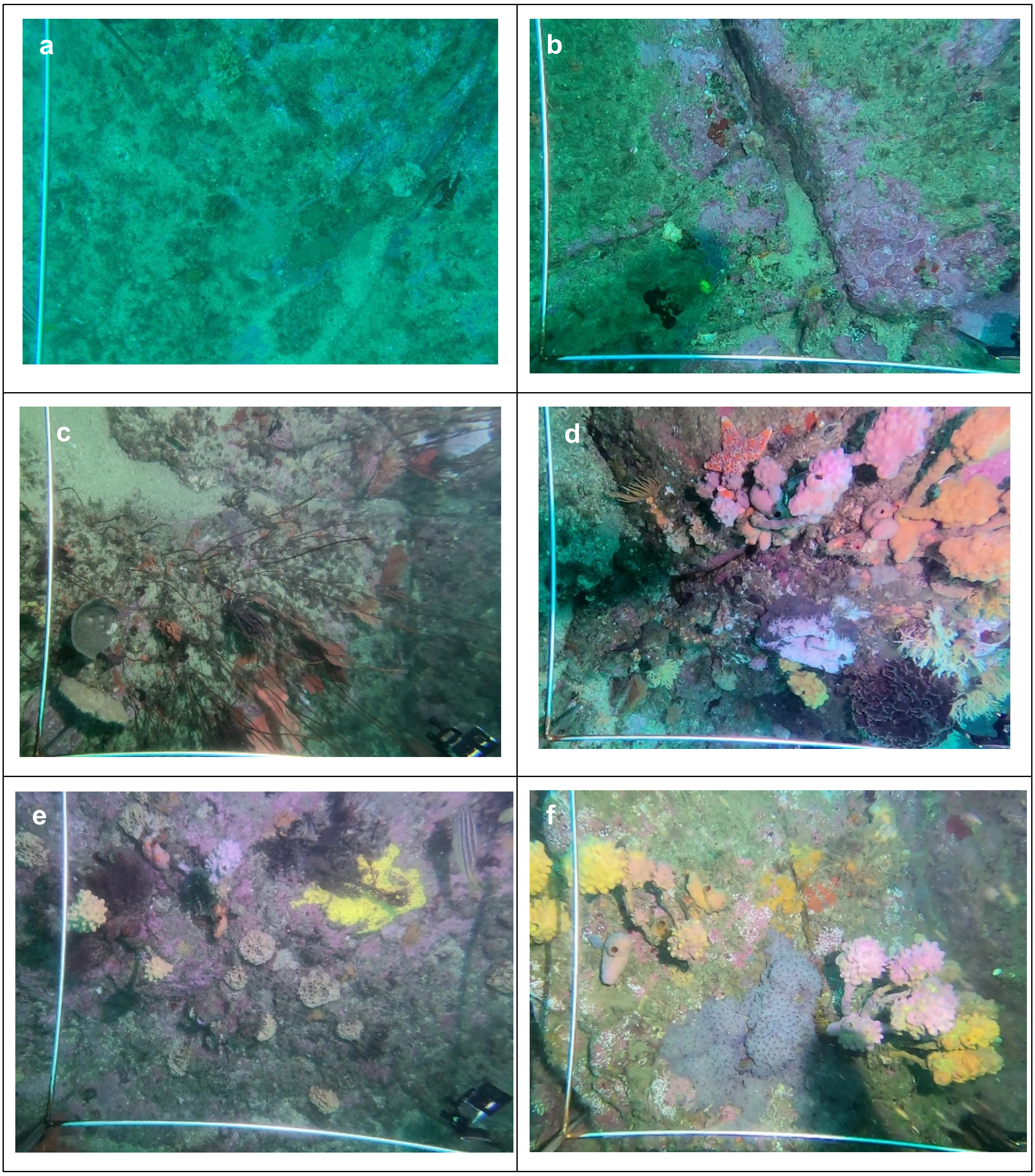



**Figure 3-7 Example quadrat photos collected of sessile filter-feeder community during Stage 2 surveys at Hunter Reef (a-b), Outer Haycock Point (c-d), and Long Point (e-f)**

## 3.5 Construction Phase Impacts

Potential impacts to sub-tidal reef assemblages from construction phase activities include:

- Accidental spill from construction vessels impacting water quality causing stress and or loss of habitat
- Introduction of an invasive marine pest via construction vessels and equipment
- Disturbance of soft sediment habitat causing a turbidity plume

### 3.5.1 Accidental Spill

There is the potential for hazardous substances (*ie.* fuels, oils and other construction vessel related fluids) to accidently enter the water through spills or leaks from construction vessels and/or equipment. Water pollution resulting from vessel accidental spill would typically impact the water surface and have limited direct effect on deeper sub-tidal reef assemblages. This risk can be reduced by implementing a range of control measures to protect water quality during construction. The potential impacts of water pollution on nearby shallow and intermediate reefs is therefore considered a low risk.

### 3.5.2 Introduction of an Invasive Marine Pest (IMP)

Introduction or translocation of an IMP to Merimbula Bay from Twofold Bay during construction phase activities is considered unlikely. Three IMPs reported from the Port of Eden, 30 km to the south (Pollard and Rankin, 2003) that are not yet reported from Merimbula Bay include the dinoflagellate *Alexandrium catenella*, European fan worm (*Sabella spallanzanii*) and the New Zealand Screwshell (*Maoricolpus roseus*). The latter is known to occur on the continental shelf off Merimbula but is not known to be present within the study area of the embayment. Of these IMPs, the European fan worm has ability to establish in high densities and potentially alter the habitats in proximity to the outfall. However, very little work has been done on the impacts of the European fan worm and its effects at an ecosystem level (DPI, 2020).

It is expected that construction vessels would adopt standard environmental management practices and controls as recommended by the *National Marine Pest Plan 2018-2023* to mitigate the risk of IMPs such as European fan worm during construction phase (Refer **Section 15**).

### 3.5.3 Disturbance of soft sediment habitat causing a turbidity plume

Soft sediments along the pipeline alignment are described as medium to coarse grain sands that are likely to settle quickly once disturbed. The disturbance would be short-term and due to the distance from the Project area to the nearest sub-tidal reef assemblages being more than 1,400m away, the risk of impact from a turbidity plume is considered minimal.

## 3.6 Operational Phase Impacts

Potential impacts from the operational phase of the Project considers how the discharge of treated wastewater at the diffuser may affect sub-tidal reef assemblages. Sub-tidal reef communities and species may be directly affected through exposure to treated wastewater, or indirectly affected by changes to the ecology of the seabed and water column (such as by increased turbidity).

Dispersion modelling (AECOM, 2019c) of the treated wastewater discharge indicates that under most conditions and majority of the time, water quality objectives will be met within a 25 m radius mixing zone. Under a worse-case scenario such as wet weather flow that could coincide with stagnant or current conditions, the distance required for the dilute wastewater to meet all MWQOs would extend to 200 m radius buffer from the diffuser.

Exposure to treated wastewater and potential ecological changes to the seabed and water column within the modelled 25 m mixing zone include:

- Habitat disturbance due to altered sediment chemistry.
- Exposure to reduced salinity of water column from freshwater discharge.
- Exposure to discharge of nutrients above MWQOs.
- Exposure to discharge of metals above MWQOs and aquatic ecosystem protection guidelines, including potentially bioaccumulative metals and other bioaccumulative contaminants.

There are no naturally occurring sub-tidal reefs within the mixing zone with the nearest sub-tidal reefs at least 1,400 metres away from the diffuser location. Therefore, the risk of impact to the sub-tidal reef invertebrate and algal communities from the operational phase is considered minimal.

## 3.7 Conclusion

The potential impact of the Project construction phase to sub-tidal reef assemblages has been assessed as low to minimal with mitigation of environmental risks controlled through implementation of a CEMP during construction (**Section 15**).

Given that water quality objectives would typically be achieved within a mixing zone of 25 m extending to 200 m from the diffuser under a worse-case scenario (**Figure 1-5)**, and that no sub-tidal reef habitat is present within the mixing zone, the risk of impact to sub-tidal reef communities from the operational phase of the Project is assessed as minimal.

# 4 Assessment of Abalone

## 4.1 Introduction

A requirement of DPI in its submission to the SEARs (letter dated 30/5/2016) is for the Project to:

- *Provide analysis of potential impacts upon, and risks from both the construction and operational phases to commercial abalone fishery.*

This report section provides an overview of the abalone fishery within the marine ecology study area, provides an independent assessment of the abalone population at Haycock Point, and evaluates the risk of potential impact to the fishery from construction and operational phases of the Project.

## 4.2 Background

Haycock Point has an abalone population that is commercially harvested by a number of fishermen. During client and stakeholder consultation in August 2017, DPI Fisheries recommended that an initial baseline assessment of abalone populations at Haycock Point be undertaken as part of Stage 1 marine ecology investigations. Assessing the abalone population at Haycock Point was based on previous modelling undertaken for the Project (MHL, 2015) that suggested an outfall diffuser directly east of Haycock Point in 30 m depth should be considered among a number of potential outfall options for the Project as the location would provide good dispersion of the treated wastewater plume.

As highlighted in **Section 2 – Overview of Marine Habitats**, two potential pipeline alignments, a northern alignment and southern alignment, and four diffuser location options were considered by the project team and community working group. Location 1, referred to as the 'North-Short' outfall option, was subsequently selected as the preferred option by the project team in October 2019.

The North-Short outfall would be located on sandy habitat in the central region of Merimbula Bay at 30 m depth contour. The nearest rocky reef habitat to the proposed location is approximately 1400 m to the south-east at Hunter Reef, with the shallow rocky reef communities of Haycock Point and Long Point approximately 2000 m to south and 2300 m to north respectively (**Section 2 – Overview of Marine Habitats**).

While Haycock Point and Long Point are known to support abalone populations and are also harvested by commercial abalone fishers, Hunter Reef is not targeted by commercial abalone fishers for reasons that may include – majority of Hunter Reef is below water depths of 20 m, is devoid of foliose macroalgae that are favoured as food by adult abalone, and the high density of sea urchin, *Centrostephanus rodgersii,* that competes with abalone. On this basis, Hunter Reef is considered sub-optimal habitat for abalone and if abalone are present (none were observed during marine ecology investigations, either by remote video or diving), the population is naturally much smaller than the adjacent Haycock Point and Long Point areas.

For the purposes of this assessment, potential impact to the commercial abalone fishery of Merimbula Bay is considered based on available information provided by DPI and a single field survey of the abalone population at Haycock Point undertaken in January 2018.

### 4.2.1 Objective of field survey

The objective of the study was to obtain an independent estimate of the abalone population structure at Haycock Point with an emphasis on describing the abundance and distribution of undersized abalone. Undersize abalone are not subject to commercial harvest and therefore assessing the population structure of this size class is considered a more reliable indicator for identifying potential population change. Legal size abalone can be locally depleted following harvest and this could confound counts in this class and lead to erroneous conclusions about population trends of the commercially harvested size class.

### 4.2.2 Blacklip Abalone – *Haliotis rubra rubra*

The abalone fishery in Australia harvested an annual wild catch of 3,176 tonnes in 2017/18 worth approximately $151.5 million (ABARES, 2020), of which most is exported. The NSW abalone fishery targets a single species, the blacklip abalone, *Haliotis rubra rubra,* and is relatively small (113 t in 2017/18) in terms of total wild catch compared to southern Australian states Victoria (756 t) and Tasmania (1,473 t).

Abalone generally occur in shallow coastal waters and are relatively sedentary once settled on rocky reef habitat. Blacklip abalone live for 20-50 years, reaching a shell length of 150-220 mm, with growth rates highly spatially variable (Doubleday *et al.,* 2011). Reproductive maturity is reached at approximately 5 years of age, at which time shell length may be 80-130mm (Mundy *et al.,* 2018). Following post-larval-stage settlement onto green and/or coralline algae on a reef, the small juveniles graze on microalgae on exposed rock for 4-6 weeks, after which they shift to a more cryptic existence in narrow crevices (Doubleday *et al.*, 2011). Their feeding preference shifts to trapping drift algae and grazing on brown algae and other detritus. At 2-4 years old they shift to inhabiting more exposed locations on the rocky reef, possibly coinciding with sexual maturity (Doubleday *et al.,* 2011).

Blacklip abalone are broadcast spawners with short-duration larval stages (3-12 days) and limited larval dispersal (i.e. planktonic larvae do not settle far from the spawning site). The stocks thus exist as meta-populations of largely discrete local sub-populations with relatively little and infrequent exchange. This characteristic makes abalone especially susceptible to localised depletion and serial over-harvest of entire stocks (TACC Committee, 2019).

### 4.2.3 State of the NSW Abalone Fishery

The NSW commercial abalone fishery was established in the early 1960s and annual production peaked at approximately 1,250 tonnes in 1973. NSW DPI manages the fishery and sets the Total Allowable Commercial Catch (TACC) each year based on advice provided by recommendations outlined in the annual TACC Committee report, tasked with providing an independent assessment of the status of the stock (TACC Committee, 2019).

The status of abalone stock is assessed based on fishery-dependent data from logbooks that includes:

- Catch weight (tonnes)
- Catch Rate as Catch Per Unit Effort (CPUE) – kg of abalone harvested per hour
- Commercial catch size structure
- Mean weight of abalone

Abalone stocks have been managed and reported at a range of spatial scales that include region, zone and sub-zone and more recently using a spatial scheme based on areas and spatial management units (SMU). However, the spatial boundary definitions of the reporting schemes are not necessarily equivalent and comparing stocks between assessment years that have used the different reporting schemes is difficult (TACC Committee, 2019).

The status of the abalone stock and annual production declined steadily to the mid-2000s, to the extent that the TACC was less than 10% of peak production for several years between 2006 and 2012 (TACC Committee, 2019). Data collected by the Abalone Council of NSW (2016) suggested that by 2015 stock had been rebuilding throughout the fishery since 2006 but particularly since 2009, as indicated by increased commercial catch rate (as measured by CPUE) and mean weight of abalone recorded. This was attributed primarily to the reductions in TACC and increases in Legal Minimum Length (LML) of harvestable abalone. The TACC was gradually increased from its 2009 trough of 75 t to levels at or above 10% of peak production (125-130 t; 2014 to 2017), after which declining indicators warranted a TACC decrease in 2018 to the current determination of 100 tonnes (2020/21) (TACC Committee, 2019).

In 2018, the NSW blacklip abalone fishery was classified as '*depleting stock'* in the most recent Status of Key

Australian Fish Stocks (SAFFS) Report (Mundy *et al.,* 2018). This represents a recent decline in formally assessed stock status for the fishery, which was previously classified as '*overfished-recovering*' in the 2016 SAFFS Report and '*sustainable*' in the 2014 SAFFS Report. The 2018 report states: "Since about 2015 biomass has declined but is not yet depleted and recruitment is not yet impaired, however, fishing mortality is too high and moving the stock in the direction of becoming recruitment impaired" (Mundy *et al.,* 2018).

#### 4.2.4 Commercially fished Abalone Populations around Merimbula Bay

There are two rocky reef systems from which blacklip abalone are commercially harvested located in the vicinity of the Project site: Long Point reef, located off the headland bounding Merimbula Bay to the north; and Haycock Point reef, located off the headland bounding the bay to the south). For the purpose of this report, the two reefs are referred to as 'Merimbula Bay reefs' (or 'Merimbula Bay subtidal rocky reef habitat') when being referred to collectively, with the two associated abalone populations collectively referred to as Merimbula Bay abalone populations where appropriate.

Under the various management reporting schemes, both reefs are part of Area 12 – Turingal, which covers an approximate 21 km length of coastline from Turingal Head south to Quondolo Point, and at a broader scale are part of SMU 2 and Region 4 (TEL, 2007; Abalone Council, 2016). At the finest management reporting scale, Long Point is within Sub-zone W3 that extends from the Short Point Beach south to the northern part of Merimbula Bay, while Haycock Point is within Sub-zone X1 that includes majority of Merimbula Bay (excluding Long Point) and extends south to Quondolo Point (TEL, 2007)(**Table 4-1).** Figures showing the boundary definitions of the various management and reporting spatial scales for the NSW abalone fishery are provided in **Appendix B-1** and **Figure 4-1** below.

**Table 4-1 Summary of abalone management and reporting spatial scales that include Long Point and Haycock Point reefs**

| Reef | Area | Area Name | Zone | Sub-Zone | Region | SMU |
|---|---|---|---|---|---|---|
| Long Point | 12 | Turingal | W | W3 | 4 | 2 |
| Haycock Point | 12 | Turingal | X | X1 | 4 | 2 |

In the NSW abalone stock assessment report prepared and submitted in 2016 by the Abalone Council of NSW, it was concluded that abalone stock for Area 12, which includes Long Point and Haycock Point reefs, are currently sustainable and stable, with total catch, catch rate (as CPUE) and mean weight of abalone reported as relatively stable since 2011-12 (Abalone Council, 2016). Estimates of biomass were reported to be increasing by about 10% per year in Area 12 around Merimbula, but declined in 2015-16 (Abalone Council, 2016).

A GPS data logger program was implemented by the NSW abalone fishery to allow reporting of estimates of legal size biomass density at finer spatial scales within sub-zones to assist stock management decisions. Based on this, estimates of catch and catch rate (as CPUE) variation among years would be available for Haycock Point, although these data were not available for this report.

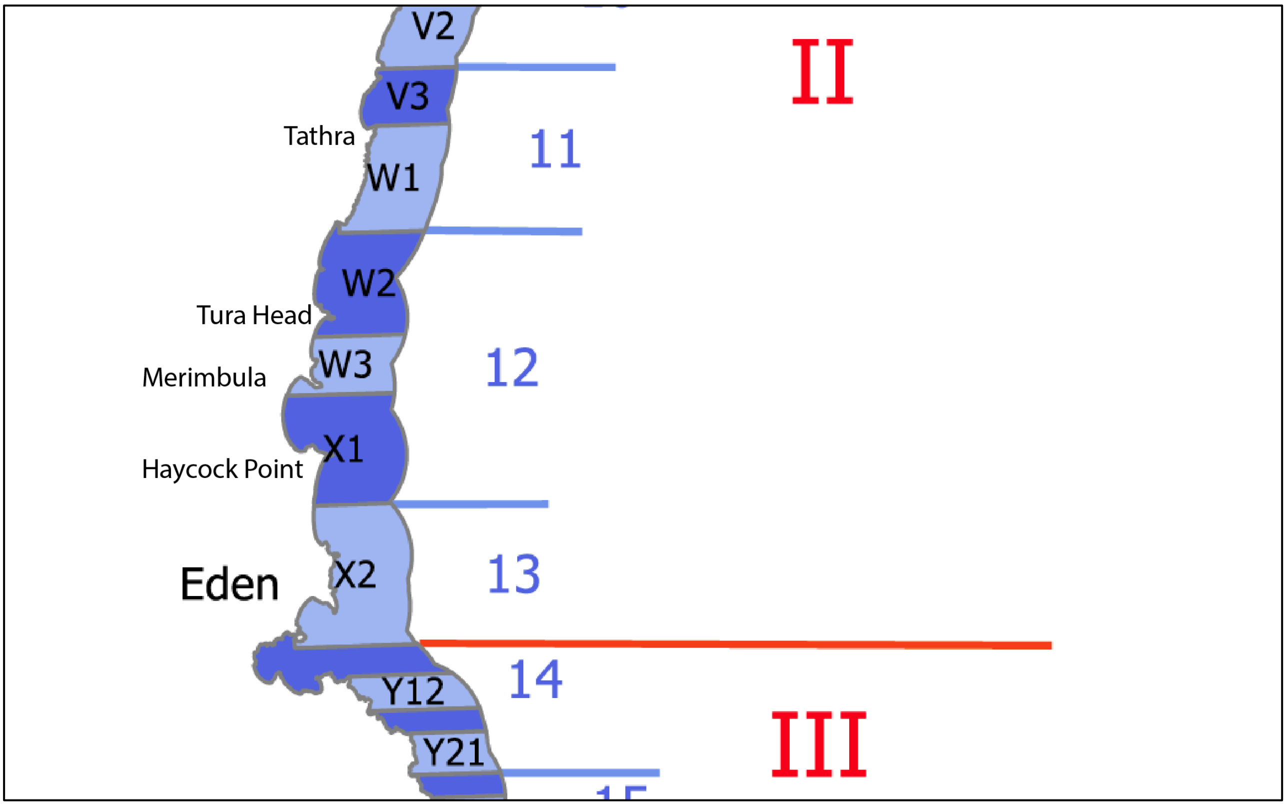



**Figure 4-1 Overview of abalone management reporting scheme at spatial scale of Sub-zone, Area, and SMU**

### 4.2.5 Existing Threats to Abalone Populations in NSW Waters

Blacklip abalone populations in NSW waters have been and are subject to a range of stressors that could possibly facilitate declines in stocks available for commercial and recreational fishing (Doubleday et al., 2011). In addition to depletion via commercial, recreational and illegal fishing pressure, there is an ongoing risk that any given population may be existentially threatened by one or more of a number of 'natural' environmental, pathological or ecological threats. These stressors may range between relatively sudden and potentially short-term (i.e. disease or invasive pest incursion, water quality issues), or gradual and permanent in affect (i.e. ocean warming) and would likely interact to produce a cumulative detrimental impact on stocks and recruitment. Any such impacts may or may not eventually result in recovery of the population to previous levels.

***Climate change – Water quality***

It is predicted that the current trajectory of measured changes in climate associated with the east coast of Australia and Tasman Sea is likely to result in a greater frequency of extreme weather events (Hobday *et al.* 2006). This may mean greater frequency and longer duration of extreme heatwaves, potentially more extensive bushfire events, and more intense east coast lows and precipitation events. The catastrophic bushfire event and subsequent rainfall events experienced throughout the south coast region of NSW during the summer of 2019/20 illuminated the threat of runoff of large volumes of ash-laden water into catchments devastated by fire. While impacts on freshwater ecosystems from such events is understood to a greater degree (McInerney *et al.* 2020), very little research has been done to investigate impacts to coastal marine and estuarine ecosystems posed by runoff of large volumes of ash-laden water into those environments. Nevertheless, it would be reasonable to cautiously expect, however, that exposure to poorer water quality due to mixing of plumes of ash-laden runoff with marine water would be a short-term disturbance to abalone health and habitat, posing no long-term ill effects. In any case, research into this issue is required.

***Climate change – Ocean warming***

While blacklip abalone populations are distributed across a wide latitudinal range in terms of eastern Australian coastal water temperatures (9-11C in winter in Tasmania to 23-25C in summer in northern NSW), populations located off the northern two-thirds of the NSW coastline are considered to be at the maximum thermal limit for the species (Doubleday *et al.,* 2011). Further, it is thought that this species has little physiological ability to cope or adapt to thermal change (Doubleday *et al.,* 2011). Given this, it is reasonable to infer that the expected gradual warming of coastal waters off NSW would most likely result in a southward contraction of the current latitudinal range and exertion of a higher frequency and degree of temperature stress on populations on reefs off the southern third of the state, including reefs off Merimbula. Elevated water temperature is thought to slow and stunt abalone growth and reduce resistance to disease, while temperature is also thought to be an important factor in the timing of spawning, larval development and eventual recruitment to the reef (Doubleday *et al.,* 2011).

Other less direct impacts to NSW abalone populations from rising coastal water temperatures include a range of ecosystem-level impacts that could permanently change the floral and/or faunal profile of the rocky reefs such that abalone populations cannot sustain themselves (Doubleday *et al.* 2011). It should also be noted that ocean acidification facilitated by increasing $CO_2$ levels may also pose an additional threat to abalone, potentially impacting growth, shell development and survival, as has shown to be the case for other $CaCO_3$-producing organisms (Doubleday *et al.,* 2011).

***Abalone pathogens***

During the 1990s and early-2000s, blacklip abalone populations in waters north of Ulladulla were significantly impacted by a steep rise in infections by *Perkinsus olseni* – a protozoan parasite that severely diminishes the condition (slower growth, weakening, poor market condition) of the host, and can result in death (Corbeil and Berthe, 2009; Liggins and Upston, 2010). It is understood that temperature stress predisposes abalone to infection and is likely to increase its prevalence and worsen condition and mortality outcomes in areas that are more at risk of exposure to higher water temperatures (Doubleday et al. 2011). It is also known that *Perkinsus* can be transmitted between adjacent populations via passive means (ocean currents) or via active means (i.e. translocation of infected abalone tissue or *Perkinsus* cells from reef to reef by boat or even faeces of predators), and that the outbreak may be more likely to escalate at higher water/tissue temperatures (>20°C) (Liggins and Upston, 2010; DAWE, 2020). While significant mortalities associated with past outbreaks were generally restricted to rocky reefs north of Ulladulla, it is notable that the parasite was also detected in a small number of individuals sampled from a reef located off the Merimbula area in 2005 (Liggins and Upston, 2010). While there has been no recorded cases of deaths in abalone attributable to *Perkinsus* infection in NSW populations since the mid-2000s (NSW DPI, 2020a), it is thought that the parasite may exist in some mollusc species in a lifelong-carrier state under certain environmental conditions (DAWE, 2020). Given the above, it is reasonable to consider it possible that, given the general pattern of warming of coastal waters off eastern Australia, the risk of *Perkinsus* outbreaks in waters off the south coast of NSW would increase, as would the likelihood of detrimental impacts to populations such outbreaks afflict.

***Reductions in available abalone habitat facilitated by invasive species and native competitors***

In NSW the invasive algal species *Caulerpa taxifolia*, thought to have been originally introduced via specimens discarded from home marine aquariums, is known to readily colonise and consolidate sandy patches in crevices and around rocky reefs, having the potential for smothering exposed or semi-exposed rocky reef habitat important for blacklip abalone (Doubleday *et al.,* 2011; Glasby *et al.,* 2015; DPI 2020b). It is native to tropical regions but can tolerate cooler water. It is able to quickly propagate from fragments of tissue and is known to have been actively translocated between NSW estuaries via inadequately rinsed vessel propellers or drained deck/ballast water (Glasby *et al.,* 2015). While *C. taxifolia* infestations in NSW waters – first recorded in 2000 – have primarily been located within estuaries, one instance was discovered in shallow oceanic waters near Port Hacking (NSW DPI, 2020b). It is notable that there are other, similar looking, less invasive native species of *Caulerpa* in NSW coastal waters that are not considered an invasive threat (DPI, 2020b). In 2011/12, dedicated NSW DPI surveys of abundance in a number of estuaries south of Jervis Bay could not find any *C. taxifolia*, but it is possible that small beds may remain in many south coast estuaries and have the potential to spread if conditions are favourable, such as a warming of coastal waters (Glasby et al. 2015; DPI 2020b).

Other organisms known to colonise and gradually alter rocky reef habitat to the potential detriment of resident abalone include species of mussel (farmed or invasive) and the barren-forming urchin, *Centrostephanus rodgersii* (Doubleday et al., 2011). Mussels have the potential to colonise crevices, overhangs and other hard surfaces of rocky reefs, effectively blanketing hard surfaces with dense mat-like colonies, reducing available habitat for abalone (Doubleday et al., 2011; DPI ,2020c). The blue mussel (*Mytilus galloprovincialis*) is a native but non-endemic species that naturally occurs on hard substrate, usually in sheltered estuarine or less-exposed open marine environments and is farmed in Twofold Bay and Jervis Bay (DPI, 2020d). It would be reasonable to consider blue mussel and other native mussel species unlikely threats to NSW blacklip abalone populations, as there has been no record of local population expansions of these species in natural rocky reef habitats in NSW waters (DPI, 2020c). In contrast, there is real potential for infestations by invasive mussel species such as the black-striped mussel, which has in the past been found and eradicated in Darwin Harbour, NT (DPI, 2020c). NSW DPI acknowledges that the species has the potential to colonise the NSW coastline down to and around Sydney and, if established in NSW waters, these mussels could devastate the shellfish, fishing, tourism and other marine industries" (DPI, 2020c). Mussels have planktonic larval stages, so introduction of larvae to an area via vessel ballast is always a possibilty. In addition, some species of mussel can detach from and reattach to hard surfaces, including the hulls or securing ropes of vessels.

The detrimental impact to the NSW abalone fishery from local population explosions of the long-spined sea urchin, *C rodgersii*, on NSW rocky reef habitats inhabited by blacklip abalone is well documented (Andrew *et al.,* 1998; Doubleday et al., 2011). If urchins become established on a rocky reef on which they were not previously present or abundant, they can overgraze and denude the reef of important foliose brown algae (e.g. *Ecklonia*, *Phyllospora*, *Sargassum*) and invertebrate fauna, completely changing the habitat profile towards a persistent 'barrens' state unsuitable for abalone (Doubleday et al., 2011). On reefs that were observed actively undergoing such a change, the spatial expansion of barrens created retreating front lines of 'fringe' habitat of foliose algae that was found to harbour denser aggregations of abalone than the foliose algae habitat behind it, indicating that the urchin is a superior grazing competitor that can causes retraction in local abalone distribution and abundance (Andrew *et al.,* 1998; Doubleday et al., 2011). It should be noted that there is a small, restricted sea urchin fishery in NSW that harvests urchin for their roe, so there is potential for effective local monitoring and management of abalone and sea urchin populations by local commercial harvesters as a strategy for prevention of urchin overgrazing events (Andrew *et al.,* 1998; Doubleday et al., 2011, DPI, 2020e). Given this, it would be reasonable to expect that an emerging threat from an increasing population of sea urchins to a given rocky reef from which blacklip abalone is harvested would be detected by fishers and appropriately managed.

#### 4.2.6 Previous Studies of Wastewater Impacts to Abalone

The primary concern from industry stakeholders is whether the Project may cause detrimental harm to the abalone populations of Merimbula Bay (i.e. Haycock Point and Long Point) that results in reduced commercial catch. While the biology, ecology and fishery stock dynamics of various abalone species have been well studied to ensure sustainable fishery management, a search of scientific literature found only a few studies have specifically examined potential for, or quantitatively estimated direct and/or indirect impacts of treated sewage wastewater discharge or metal bioaccumulation on abalone, including impacts on commercial abalone fisheries.

In broad terms, the list of potential impacts on any given commercially exploited abalone population that may occur and possibly be determined as attributable to the construction and operation of nearby sub-surface, treated-wastewater diffusion infrastructure can be broadly categorised as:

- direct (e.g. affecting abalone physiology/condition) or indirect (e.g. affecting available habitat) or both;
- lethal (e.g. death of adults/juveniles from physiological stress) or sublethal (e.g. reduced fecundity or successful larval recruitment); and
- either negative (net decrease in abundance through time) or positive (net increase in abundance through time) or neutral, in terms of the abalone population and/or stock available for harvest.

Each potentially causative factor and its associated impact(s) can be theoretically assessed for likelihood and risk according to a systematic consideration of all three of these classes of category. Assessment of the potential for impacts on the abalone population at Haycock Point that could be reasonably attributed to the construction and operation of the proposed ocean outfall pipeline and diffuser installation is presented in **Section 4.7** below.

A sampling program commissioned in the late 1990s to assess the nature and extent of the environmental impact of an existing, long-operating ocean outfall at Boags Rocks, Victoria, surveyed an offshore reef located 600 to 800 m from, and running roughly parallel to the shore (Newell et al., 1999). It was postulated that wastewater discharge was a factor affecting the biological characteristics of the reef to a distance of 1100 m from the line of the outfall, and that a lesser impact may occur out to about 1400 m. Levels of heavy metals or other contaminants in flesh of abalone sampled from the reef in the vicinity of the outfall did not exceed National Food Authority Standards (1996) at all, and exceeded EPA Recommended Water Quality Criteria (1983) for Nickel in the case of only one (of 10) individuals sampled. The study did not, however, provide any detailed data or observations specifically referring to abalone population parameters, and was severely limited by the absence of previously collected survey data.

An ecological monitoring program commissioned in 2005 in association with the installation of an extended offshore release diffuser off Racecourse Beach, Ulladulla (installed as part of the Milton-Ulladulla Sewerage Augmentation project), did not detect any changes in subtidal floral or faunal communities, including the small population of abalone in the vicinity, that could be reasonably attributed to the installation and operation of the new subtidal infrastructure (TEL 2008). It should be noted, however, that in contrast to the currently proposed diffusion infrastructure, this Racecourse Beach diffuser is located in shallower water (~10 m deep) and much closer to shore (~360 m) and abalone aggregations recorded in the vicinity.

Trace metal concentrations in wild abalone have been investigated in several studies in southern Australia, including: to characterise natural background levels (Fabris *et al.,* 2000); assess the impact of the Eastern Treatment Plant in Victoria (CSIRO, 1999); and compare to cultured abalone raised on manufactured feeds (Skinner *et al.,* 2004). One overseas study has examined metals in abalone at sites in the vicinity of a municipal outfall in California (Young *et al.* 1981). For all studies in southern Australia, abalone was found to contain levels of trace metals within acceptable limits recommended by food safety standards (Australia New Zealand Food Authority, 2004). The study in California found that abalone (as *Haliotis cracherodii)* in the vicinity of a municipal outfall contained higher median levels of some metals (Ni, Cr) compared to abalone at control sites, though still within levels considered acceptable for human consumption.

## 4.3 Field Survey Methodology

Surveys of abalone abundance at Haycock Point were undertaken by divers on SCUBA over three days between 7 to 9 January 2018. Ocean conditions were good for diving surveys with clear water and minimal swell. The survey approach was based on the stratified transect methodology recommended by Gorfine *et al.* (1996) and Hart et al. (1997) who compared a number of assessment methods and concluded transect surveys were the preferred method because they accurately reflected absolute abundance, required a relatively small number of sampling days and were reasonably precise.

### 4.3.1 Site Selection

Abalone abundance was assessed at six (6) sites around Haycock Point with site selection based on a range of factors such as habitat preferred by abalone, depth, topography and aspect. Site selection was also based upon information provided by a local abalone fisherman and industry stakeholder, identifying commercially important harvest areas. Given the commercial sensitivity of this information, the location of survey sites is provided in **Appendix B-2** but is not to be distributed with public versions of this report.

### 4.3.2 Transects

At each site, abalone were counted within four belt transects, 50 m long by 1 m wide, radiating out from a fixed central point marked by a buoyed shot-line (**Figure 4-2**). Counts of abalone commenced 5 m away from the fixed point, such that the same area was not counted in multiple transects to avoid sampling bias. The geographic coordinates of each fixed point was recorded on the vessel GPS system, with divers recording the compass bearing and maximum depth for each transect.

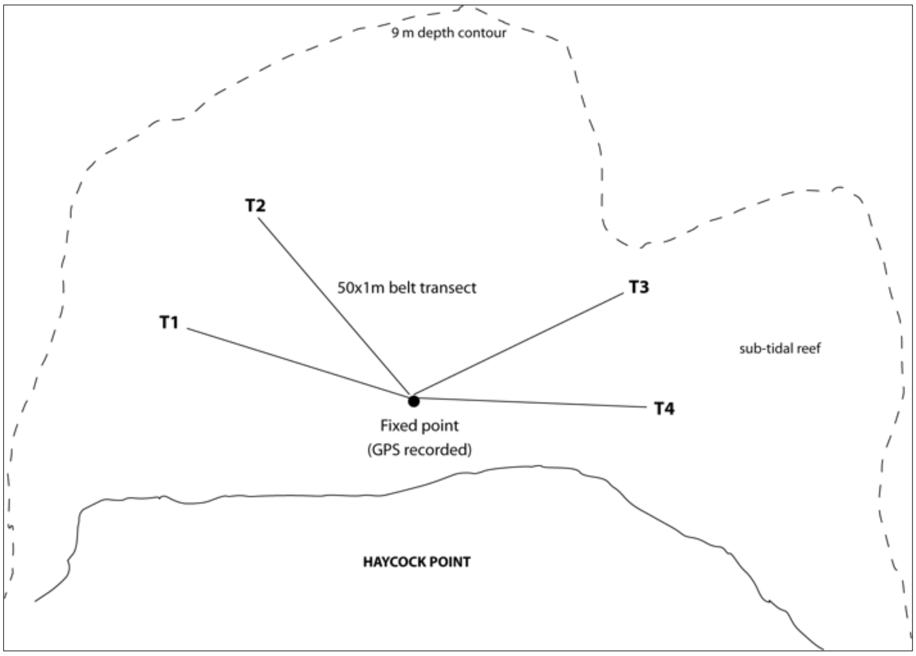



**Figure 4-2 Conceptual diagram of transect sampling at each of six (6) sites**

### 4.3.3 Video Record

The habitats encountered along each transect was recorded using a GoPro underwater camera. This provided archival video footage of each transect for later review as necessary.

### 4.3.4 Abalone Counts

Counts of abalone were recorded within contiguous 10 m intervals along each 50x1 m belt transect. As abalone tend to aggregate in gutters and are typically patchy in their distribution, the total number of individuals observed within a 10 m interval represents the smallest sampling unit from which density can be calculated and provides spatial information regarding their patchiness along each 50m transect. A total of 24 transects (4 transects at each of 6 sites) were surveyed.

Abalone were counted into three size categories: small (0-60mm), medium (60-117mm) and large (>117mm), measuring the widest point of the shell using a pre-marked abalone ruler. The legal minimum length (LML) for abalone in NSW is currently 117mm, meaning that the small and medium classes represent ‘undersize’ abalone, whilst the large class represent ‘legal size’ abalone for commercial (and recreational) harvest. These size classes correspond to those used in abalone stock assessment surveys in NSW. A selection of images recorded during field surveys show transect setup, typical macroalgal habitat surveyed, and abalone within crevices are provided in **Figure 4-3**.

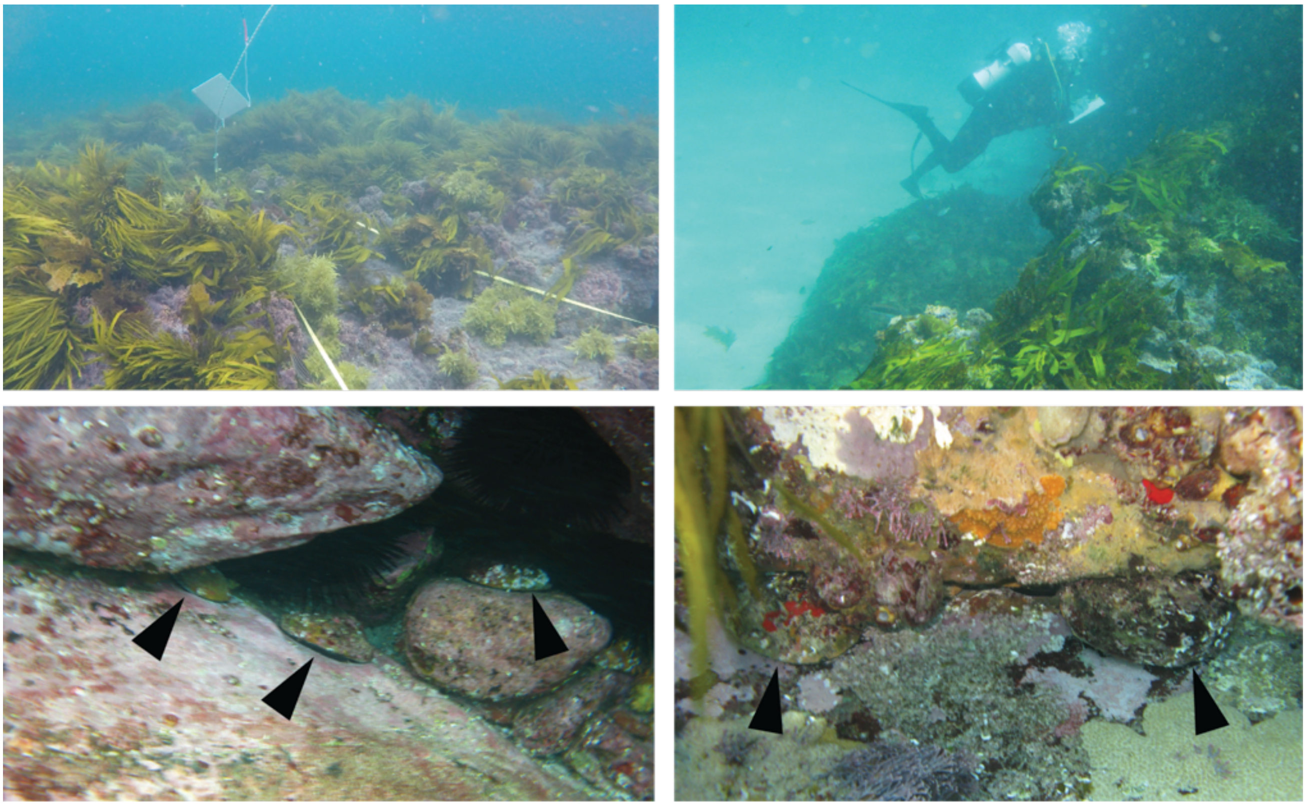


**Figure 4-3 Images recorded during abalone field surveys (arrows indicate abalone)**

## 4.4 Data Processing

Collected data were separately processed for each site to determine:

- Total count of abalone by size class per site;
- Mean count of undersize versus legal sized abalone per transect; and
- Mean density of undersize versus legal sized abalone per 10 $m^2$ interval (representing the smallest sampling unit along each transect).

The raw dataset is contained in **Appendix B-3**.

## 4.5 Results

### 4.5.1 Sampling sites

Three sites were sampled on the southern side and three sites on the northern side of Haycock Point. All transects were 50 m long and positioned at depths between 3 and 9 metres. Transects were paired and generally orientated parallel to the shoreline which reflected the positioning of transects along contour lines. Running transects in parallel also ensured that divers were working within visual proximity to each other.

Substrates sampled along transects included low profile reef, boulder fields and fingers of reef adjacent to sand gutters. Habitat sampled was mostly mixed macroalgae-fringe variously dominated by canopy forming brown algae *Ecklonia radiata*, *Sargassum* sp., *Cystophora monilifera* and *Phyllospora comosa*. Some transects also traversed small areas of barrens habitat though this habitat was generally avoided as it is known to be sub-optimal for abalone.

### 4.5.2 Diver Counts

A total of 715 abalone were counted across all sites with a summary of count data by size class for each site provided in **Table 4-2** and raw count data in **Appendix B-3**. The majority of abalone observed (66%) belonged to the 'undersize' small and medium size classes and below the LML of 117 mm, with 34% of all abalone belonging to the 'legal' large size class (>117 mm) and available for harvest.

The trend in abalone size class abundance was similar across all sites, with medium sized abalone most abundant followed by large abalone, and small abalone the least abundant. The low count of small abalone (*i.e.* juveniles) recorded along survey transects is unlikely to accurately reflect the true abundance of that size class, as small abalone tend to hide behind the medium and large sized abalone and occupy the deepest parts of crevices and other cryptic habitats beyond the vision of divers conducting surveys.

Overall, a higher count of abalone was recorded from sites located on the northern side of Haycock Point compared to the southern side. One reason for this may be that sites 1 and 2 on the southern side of Haycock Point were generally low profile reef with a high proportion of reef impacted by sand with overall lower availability of optimal habitat for abalone. In comparison, the northern side of Haycock Point was comprised of extensive boulder fields and crevices considered optimal for abalone and was not impacted by sand.

**Table 4-2 Summary of abalone count data for each site by size class**

| Site | Aspect | Depth (m) | Transects | Small (0-60 mm) | Medium (60-117 mm) | Large (>117mm) | Total Abalone |
|---|---|---|---|---|---|---|---|
| Site 1 | South | 3 - 6 | 4 | 4 | 45 | 37 | 86 |
| Site 2 | South | 5 - 6 | 4 | 0 | 37 | 17 | 54 |
| Site 3 | North | 6 - 9 | 4 | 5 | 77 | 61 | 143 |
| Site 4 | North | 7 | 4 | 7 | 35 | 25 | 67 |
| Site 5 | North | 5 | 4 | 4 | 155 | 64 | 223 |
| Site 6 | South | 7 | 4 | 5 | 96 | 41 | 142 |
| | | | **TOTAL** | **25** | **445** | **245** | 715 |
| | | | **% Proportion** | **4%** | **62%** | **34%** | |

An estimate of the abalone population size structure at Haycock Point **(Table 4-3)** is presented in **Figure 4-4** that shows the mean count of undersize abalone compared to legal sized abalone recorded within transects at each site. It is clear that undersize abalone are more abundant than legal size abalone, a trend that may be expected at a location that is subject to regular commercial harvest.

Sites 3 and 5 on the northern side of Haycock Point, and site 6 on the southern side of Haycock Point had the highest mean counts of abalone. Abalone counts were highly variable among transects within sites as indicated by large standard error reflecting the patchiness of their distribution within a transect.

**Table 4-3 Mean count of undersize versus legal sized abalone per transect at each site**

| Site | Undersize Abalone (<117 mm) | | Legal size Abalone (>117 mm) | |
|---|---|---|---|---|
| | Mean (n=4) | SE | Mean (n=4) | SE |
| Site 1 | 12.3 | 7.4 | 9.3 | 7.1 |
| Site 2 | 9.3 | 5.1 | 4.3 | 2.3 |
| Site 3 | 20.5 | 4.8 | 15.3 | 6.0 |
| Site 4 | 10.5 | 5.3 | 6.3 | 3.6 |
| Site 5 | 39.8 | 20.6 | 16 | 5.0 |
| Site 6 | 25.3 | 3.6 | 10.3 | 3.1 |

Note

Mean calculated from total abalone count within each of four transects at each site

SE = Standard Error

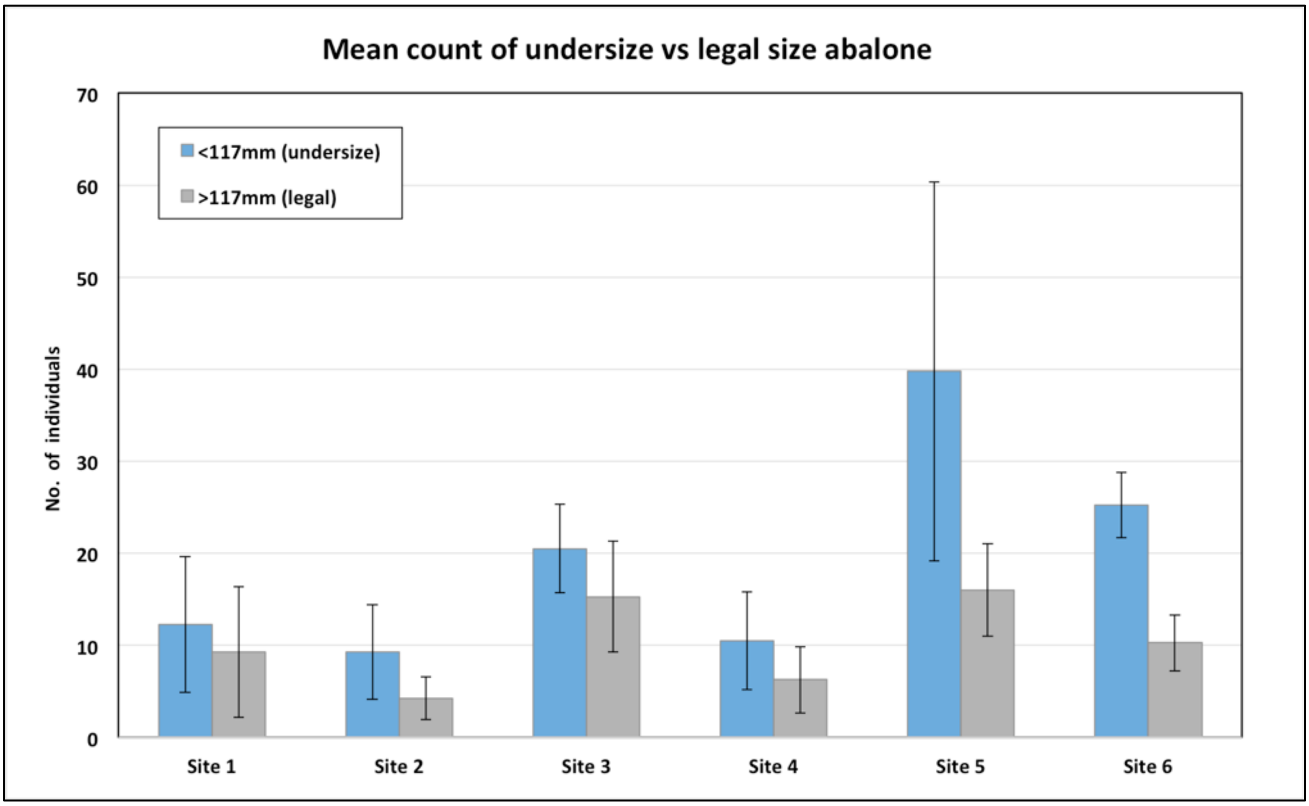



**Figure 4-4 Mean count of undersize versus legal sized abalone per transect at each site (n = 4 transects)**

### 4.5.3 Abalone Density

Abalone abundance as mean density per 10 $m^2$ interval was calculated for undersize and legal-size abalone for each site with overall mean density for Haycock Point based on pooled data (**Table 4-4**) and presented in **Figure 4-5**. Due to the aggregating behaviour of abalone and their naturally patchy distribution, there were many 10 $m^2$ intervals along transects where no abalone were observed and these intervals were excluded from mean density calculations. Sites 1 and 2 had the highest number of intervals where no abalone were recorded suggesting an overall lower availability of optimal habitat (*i.e.* crevices and rock gutters) compared to other sites.

With the exception of site 1, mean density of undersize abalone was greater than legal sized abalone at all sites ranging from 3.8 to 9.9 abalone per 10 $m^2$. The overall mean density of undersize abalone for Haycock Point was 6.8 (n=69) compared to 4.4 for legal sized abalone (n=56) per 10 $m^2$.

Undersize abalone were recorded at the highest densities (9.9 individuals per 10$m^2$) and also over the most intervals at site 5 (n=16) suggesting a greater availability of habitat optimal for abalone present at that site. In contrast, the highest densities of legal sized abalone were recorded at site 1 (7.4 individuals per 10 $m^2$) but were recorded over relatively few intervals (n=5) indicating that optimal habitat was most patchy at that site and that perhaps abalone had not been harvested as recently at site 1 compared to other sites.

Previous studies of abalone abundance in NSW have reported density at the 10 $m^2$ scale (Andrew and Underwood 1992) including locations on the far south coast of NSW at Mowarry Point, Green Cape and Disaster Bay. However, this data was collected in 1989 and is almost 20 years prior to the January 2018 surveys and comparisons between the datasets cannot be made. In order to understand how the population of abalone at Haycock Point currently compares at the local scale, surveys of other nearby locations would need to be undertaken.

**Table 4-4 Mean density of undersize versus legal sized abalone per 10 $m^2$ interval at each site and overall mean density for Haycock Point based on pooled data**

| Site | Undersize Abalone (<117 mm) | | | Legal size Abalone (>117 mm) | | |
|---|---|---|---|---|---|---|
| | *n* intervals | Mean | SE | *n* intervals | Mean | SE |
| Site 1 | 7 | 7.0 | 2.9 | 5 | 7.4 | 2.4 |
| Site 2 | 6 | 6.2 | 2.9 | 4 | 4.3 | 2.1 |
| Site 3 | 13 | 6.3 | 1.6 | 11 | 5.5 | 1.1 |
| Site 4 | 11 | 3.8 | 1.4 | 11 | 2.3 | 0.4 |
| Site 5 | 16 | 9.9 | 2.4 | 14 | 4.6 | 0.9 |
| Site 6 | 16 | 6.3 | 1.2 | 11 | 3.7 | 1.3 |
| **HAYCOCK PT** | **69** | **6.8** | **0.8** | **56** | **4.4** | **0.5** |

Note

Mean calculated from total abalone count within n = 10$m^2$ intervals for each site

SE = Standard Error

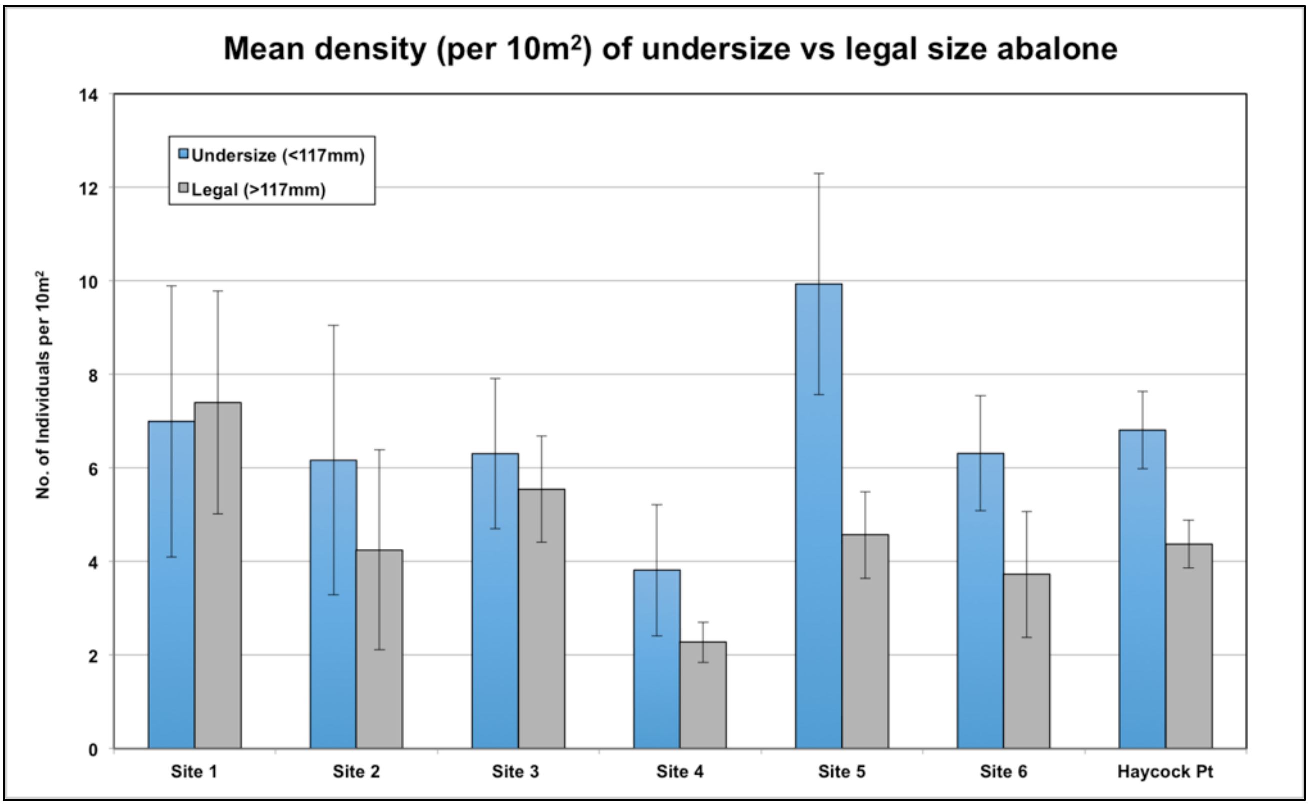



**Figure 4-5 Mean density of undersize versus legal size abalone (per 10 m² interval) at each site and overall mean density (per 10 m² interval) for Haycock Point based on pooled data**

### 4.5.4 Impact of Commercial Harvest on the Haycock Population

Following the population assessment, an industry stakeholder confirmed the survey location had been fished soon after and abalone harvested from sites where surveys had been conducted, thus highlighting the importance of using the undersize component of the local population for assessment purposes.

## 4.6 Key Findings

This stage 1 assessment presents an estimate of abalone population size structure (undersize versus legal size abalone) based on surveys of six sites at Haycock Point in January 2018, at a time when abalone are known to aggregate for spawning. Key findings from the study include:

- A total of 715 abalone were counted across all sites with 66% of abalone observed belonging to the small and medium size classes and below the LML of 117 mm, with 34% of all abalone belonging to the large size class (>117 mm) and available for harvest.
- A higher abalone count was recorded from sites located on the northern side of Haycock Point compared to the southern side. One reason for this may be that sites 1 and 2 on the southern side of Haycock Point were generally low-profile reef with a high proportion of reef impacted by sand with an overall lower availability of optimal habitat for abalone. In comparison, the northern side of Haycock Point was comprised of extensive boulder fields and crevices considered optimal for abalone and was not impacted by sand.
- Undersize abalone were more abundant and generally occurred at higher densities than legal size abalone at all sites, a trend that is not unexpected for a location that is subject to regular commercial harvest.
- The overall mean density of undersize abalone for Haycock Point was 6.8 per 10 m² (n=69) compared to 4.4 per 10 m² for legal sized abalone (n=56).

- The importance of using the undersize component of the abalone population for assessment and potential ongoing monitoring was apparent when the survey location was fished by an industry stakeholder in the days immediately after the assessment concluded, harvesting abalone from areas that had just been surveyed.
- In order to understand how the population of undersize abalone at Haycock Point currently compares at the local scale, surveys of other nearby locations would need to be undertaken.

## 4.7 Assessment of Potential Project Impacts to Merimbula Bay Abalone Populations

A step-wise, qualitative risk analysis methodology was applied to identify and preliminarily assess the potential for impacts on the Haycock Point abalone population that could be reasonably attributed to the construction and operation of the proposed treated-wastewater pipeline and diffuser installation.

The first step in the process required identification of specific environmental threats generated by the construction and operational phases associated with the pipeline and diffuser, with particular attention also given to quantitative and/or spatial considerations such as dispersion modelling of the treated wastewater discharge.

The second step required a search of the available literature concerning the life-cycle biology, physiology and ecology of blacklip abalone, with particular emphasis on the potential Project-generated threats identified as part of the first step.

The final step in the preliminary assessment of potential impacts required the application of a qualitative risk analysis methodology to provide clarity around the specific levels of risk of impacts to abalone populations and/or key abalone habitat posed by the sources of potential threat. A summary of the risk analysis outcomes is provided in **Section 14 – Impact Assessment**. A more detailed discussion of those outcomes is presented below.

### 4.7.1 Construction Phase Impacts

While the construction phase associated with the Project was considered to pose a number of specific threats that could impact negatively on water quality in Merimbula Bay over the short timeframe of pipe and diffuser installation, such as mobilisation of sediments, increased turbidity and the possibility of fuel spill, it was concluded that the collective risk of these threats impacting on the Merimbula Bay abalone populations would be low (refer to **Section 14**). The potential impact of the construction phase and associated construction activities on the Long Point and Haycock Point abalone populations may involve the following negative or potentially positive effects.

**Accidental spill**

There is the potential for hazardous substances (ie. fuels, oils and other construction vessel related fluids) to accidently enter the water through spills or leaks from construction vessels and/or equipment. Water pollution resulting from vessel accidental spill would typically impact the water surface initially, so could potentially spread towards water immediately above the abalone habitat off Long Point or Haycock Point. In the case of a petrochemical slick, there would be changes in its chemical composition over subsequent days such that various components would be mixed into the water column and/or may potentially sink to the substrate.

Pollutants potentially released via vessel accidental spill can be harmful to marine flora and fauna associated with rocky reefs and is considered a medium risk in terms of potential direct impacts to abalone, and indirect impacts via compromising of rocky reef habitat. This risk can be reduced by implementing a range of control measures, including an appropriate risk mitigation and spill response plan, to protect water quality during construction (refer to **Section 15 – Environmental Management**).

**Introduction of invasive marine pest (IMP), pathogen or other organism of concern via construction vessels and equipment**

There is a potential for translocation of invasive marine pests from Twofold Bay to the construction site in Merimbula Bay during construction phase activities that could potentially impact on the Merimbula Bay subtidal rocky reef habitat. One known translocation pathway from one area to another for fragments of *Caulerpa taxifolia* is via vessels or machinery that have not been thoroughly cleaned of such fragments between deployments. Similarly, planktonic larvae of organisms could conceivably be introduced to the waters of Merimbula Bay via ballast water discharge which may also be possible for the *Perkinsus olseni* parasite, which poses a known direct threat to abalone populations. **Section 4.2.5** provides details regarding these potential threats.

Introduction or translocation from Twofold Bay of an IMP or other organism during construction phase activities could potentially impact on the Merimbula Bay reef habitat, but is considered unlikely and a low risk, as is the risk of translocating the *Perkinsus* parasite to the resident abalone population via that transmission pathway. While these risks may be considered low primarily due to the considerable distance (>2,000 m) between the construction site(s) and the subtidal rocky reef habitat off Long Point and Haycock Point, and the water mixing that would occur over this distance, the risks nonetheless must be duly acknowledged and managed. It is expected that construction vessels would adopt standard environmental management practices and controls as recommended by the *National Marine Pest Plan 2018-2023* to mitigate the risk of IMPs and other such vessel-borne threats during construction phase (refer to **Section 15 – Environmental Management**).

**Increased turbidity and sedimentation from pipe and diffuser installation**

Agitation and subsequent mobilisation of sandy sediments during seabed works associated with installation of Section 2 of the pipeline and diffuser has the potential to generate a plume of turbid water. However, there is a considerable distance (>2,000 m) between the construction site(s) and the subtidal rocky reef habitat off Long Point and Haycock Point, and considerable dilution of suspended sediment is likely to occur over this distance. Further, if suspended sediment (turbid water) did sporadically drift over the rocky reefs during the construction phase, it would be unlikely to result in light attenuation to levels that might impact on the health of the algal beds, or in permanent sedimentation to levels that might impact on the health of individual abalone or the rocky reef habitat, to any significant or sustained degree. Should there be settling of fine silt onto the subtidal rocky reefs, it would be minimal and temporary, with the high natural variability in strength of wave action likely to re-suspend and remove fine sediments during swells larger than that at the time of deposition. For these reasons, increases in turbidity and sedimentation from construction activities is considered a minimal risk in terms of the potential to impact on the abalone population inhabiting the Merimbula Bay reefs habitat.

**Other environmental disturbances**

Other potential hazards and immediate threats to the aquatic environment generated by the construction phase of the Project that were identified, such as construction noise, vessel contact damage and hydrodynamic changes associated with establishment of new seabed structures, were not considered to pose any threat of impact to the Long Point and Haycock Point subtidal rocky reefs, nor the resident abalone populations.

*Creation of Type 2 rocky habitat or hard substrate habitat*

It should be noted that the construction phase may provide the first step in generation of a positive outcome in terms of local abalone populations. Construction of the pipeline infrastructure with concrete mattress and/or rock armour protection along its length constitutes a change from sandy seabed habitat to hard substrate habitat, effectively resulting in the creation of an artificial reef. It is conceivable that this artificial reef habitat may eventually transform – over a long timeframe into the operational phase – into algae-covered rocky reef habitat suitable for natural colonisation by blacklip abalone or abalone seeding. This is discussed further in the next section.

### 4.7.2 Operational Phase Impacts

Discharge of treated wastewater at the diffuser during the ongoing operational phase of the Project was considered to pose a suite of specific threats that could potentially detrimentally impact Merimbula Bay abalone populations. These threats are primarily associated with water quality issues that may have potential to affect abalone directly through exposure to treated wastewater, or indirectly via changes to the ecology of the Merimbula Bay rocky reefs they inhabit. Specifically, these potential issues include: deposition of suspended sediments, organic particulates and toxic contaminants from dilute wastewater, and exposure to reduced salinity and increased nutrient levels in dilute wastewater.

Dispersion modelling (AECOM, 2019c) of the treated wastewater discharge indicates that under most conditions and the majority of the time, MWQOs (i.e. equivalent to natural ambient ocean water quality) would be met within a mixing zone of 25 m radius around the diffuser. Under a worst-case scenario such as wet weather flow that could coincide with stagnant current conditions, the distance from the diffuser required for the diluting wastewater to meet all MWQOs may extend to a 200 m mixing zone.

In assessing potential impacts to abalone, it is important to note that sub-tidal rocky reef habitat at Long Point and Haycock Point where abalone are present and commercially fished is at least 2,000 m away from the diffuser location and unlikely to be affected by dilute treated wastewater, even under worse-case conditions. Given this, it was concluded that the collective risk of these water quality threats impacting negatively on the Merimbula Bay abalone populations would be low (refer to **Section 14**), while a potential impact that may be beneficial to local abalone populations was also identified.

**Exposure to reduced salinity of water column from freshwater discharges**

While the mixing zone would be characterised by reduced salinity, the rocky reef habitat off Long Point and Haycock Point is a considerable distance (> 2,000 m) from the expected outer extent of that mixing zone under the worst-case wet weather scenario (i.e. up to 200 m from the diffuser). In any case, blacklip abalone have been shown to be physiologically tolerant of moderately low salinity (i.e. down to 25 ppt) environments (Edwards, 2003). Given the above, decreases in salinity associated with wastewater discharge is considered a minimal risk in terms of the potential to impact on the abalone population inhabiting the Merimbula Bay reefs.

**Exposure to discharge of dissolved nutrients above MWQOs**

Discharge of elevated levels of dissolved nutrients into the mixing zone could potentially have an indirect positive effect on abalone populations associated with Merimbula Bay reefs by stimulating macroalgal growth, thereby increasing food resources. However, such a change to the macroalgal assemblage may also increase the risk of local increases in populations of superior competitors such as the sea urchin *C. rodgersi*, potentially resulting in an indirect negative impact to abalone populations. Nevertheless, the rocky reef habitat off Long Point and Haycock Point greater than 2000 m from the expected outer extent of the mixing zone under the worst-case wet weather scenario (i.e. up to 200 m from the diffuser), so it would be reasonable to expect that any dissolved nutrients reaching the reefs would be highly diluted and therefore would not induce such changes to the reef assemblages. Furthermore, it is expected that the nutrient load would be rapidly assimilated by phytoplankton with only a minor proportion available to other primary producers.

Given the above, increases in dissolved nutrients associated with treated wastewater discharge is considered a low risk in terms of the potential to impact on the abalone population inhabiting the Merimbula Bay reefs habitat.

**Exposure to discharge of metals and potentially bioaccumulative contaminants**

Metals discharged in treated wastewater to the mixing zone at levels above MWQOs include aluminium, arsenic, copper, iron, lead, selenium and zinc. Metals are a physiological stressor for marine organisms with reported effects in blacklip abalone larvae of morphological abnormalities and death following exposure of eggs to a range of concentrations of a range of metals (Gorski and Nugegoda, 2006). Potential impacts of various metals discharged in treated wastewater on Merimbula Bay abalone populations would depend on levels of exposure (i.e. concentration and duration) and tolerance thresholds for each life stage.

Assessment of dispersion modelling results in the *Water Quality Technical Report* (Elgin, 2021) indicates that the discharged treated wastewater would require much less dilution in order to meet the MWQOs for metals than would be the case for nutrients, and that this level of dilution would be expected to occur within 5-25 m of the diffuser location. Further, the minimum concentrations of copper (7 µg/L), mercury (21 µg/L), zinc (35 µg/L), iron (4,102 µg/L), cadmium (4,515 µg/L), and lead (5,111 µg/L) found to be toxic enough to impact on normal morphological development of veliger larvae of blacklip abalone are all well in excess of the corresponding MWQOs for the Project (refer **Table 1.2**) (Gorski and Nugegoda, 2006). In any case, the rocky reef habitat off Long Point and Haycock Point greater than 2000 m from the 25 m mixing zone where detrimental effect of metals to biota may be detected. It would be reasonable to expect that any heavy metals reaching the reefs would be highly diluted so would pose minimal threat to abalone populations or reef assemblages.

Bioaccumulation of metals and other bioaccumulative contaminants in abalone flesh to levels exceeding natural background levels has the potential to impact on the local commercial abalone fishery in the form of reduced consumer confidence in the local product, possibly resulting in economic impact to commercial fishers. **Section 6 – Bioaccumulation Risk to Fish and Shellfish** provides a detailed assessment of the risk of bioaccumulation associated with the Project. However, as outlined above, given the very high degree of dilution of such contaminants that would reasonably be expected to occur before dilute wastewater discharge came in contact with the Merimbula Bay reefs habitat, it is considered unlikely that bioaccumulation of metals and other contaminants would occur to a detectable degree.

Given the above, increases in heavy metal concentrations and potentially bioaccumulative contaminants associated with treated wastewater discharge is considered a low risk in terms of the potential to impact on the abalone population inhabiting the Merimbula Bay reefs habitat and/or the local abalone fishery.

**Potential development of new habitable rocky reef habitat for abalone**

As mentioned in **Section 2.6.2**, installation of the pipeline infrastructure with concrete mattress and/or rock armour protection along its length effectively creates an artificial reef. Over many years this artificial reef habitat may eventually transform into algae-covered rocky reef habitat (i.e. Type 2 fish habitat). Any available hard substrate placed in the marine environment provides habitat opportunity in the short-term for a wide range of colonising sessile invertebrates such as ascidians, bryozoans, sponges, barnacles, oysters and mussels. The pipeline and diffuser are also likely to be colonised by various macroalgae.

Depending on the shape of the structure, the eventual algal community assemblage, permanency of intermittent sand burial or scour, level of immigration of planktonic abalone larvae, and propensity of those larvae to recruit, some sections of the structure may be potentially suitable for natural colonisation by blacklip abalone, or even attempts at abalone seeding. If colonisation of the structure by blacklip abalone is a possibility that is eventually realised to a long-term, sustainable degree, this could potentially produce benefits for the Merimbula Bay reefs habitat and/or the local abalone fishery in the form of increasing the pool of Merimbula Bay reefs contributing to local larval supply.

## 4.8 Environmental Management Measures

There are a range of procedural protocols that would be implemented as part of the construction phase designed to mitigate the risk of: accidental spills; introduction of invasive marine pest (IMP), pathogen or other organism of concern via construction vessels and equipment; and underwater noise. Similarly, mitigation of the risk of discharging unacceptable levels of environmental contaminants during operation has been addressed via improvements to levels of onshore wastewater treatment and can be further addressed in future if required. These mitigation strategies are outlined and discussed in detail in **Section 15 – Environmental Management**.

Given the assessment of potential Project impacts to Merimbula Bay abalone populations has not identified any particular level of risk warranting concern, it is considered that further preventative risk mitigation measures over and above those currently planned as part of the Project would not be required in terms of impact prevention.

It is, however, recommended that an operational phase monitoring strategy be implemented in the form of a two-data-source approach to long-term monitoring to be able to quickly detect potential changes to Merimbula Bay reefs abalone populations, and accurately and reliably assess whether any changes detected could reasonably be attributable to treated wastewater discharge. Source 1 would involve analysis of fishery-dependent CPUE data, while Source 2 would involve analysis of fishery-independent data in the form of field surveys of abalone abundance and size structure conducted during operations. Such a two-pronged approach would mitigate the risk of erroneous conclusions based on observed trends in changes in abalone population health and structure interpreted from just a single source of data lacking independence (i.e. CPUE) (Rotherham *et al.* 2011). Alternatively, a two-stage approach could be implemented, for which Source 2 data (field surveys during operations) would only be required if triggered by Stage 1 (i.e. if the results of CPUE analysis indicated a potential problem with the fisheries at Haycock Point and/or Long Point, or if the CPUE data were assessed to be inadequate).

# 5 Fish Assemblage

## 5.1 Introduction

This section presents a description of the local fish assemblage associated with the variety of habitats present within the study area and its status as a fishery to both recreational and commercial fishers. The assessment is based upon a desktop review of existing information and field surveys.

## 5.2 Background

A range of biological responses in fish have been attributed to sewage pollution including increased mortality, shifts in abundance and diversity, changes to growth and reproduction, increased contaminant levels, alterations to fish parasitic load and infections or behaviour (Tsai 1975; Love et al. 1987; Grigg 1994, 1995; Siddall et al 1994; Lemly 1996). Observations of increased or decreased fish abundance adjacent to outfalls may be site or species specific.

Studies of the Sydney deepwater ocean outfalls attributed decreased abundances of snapper and flathead to the discharge of sewage wastewater, while at the same time observed increased abundances of gurnard and mosaic leatherjackets (Otway, 1995). A study by Smith and Suthers (1999) of ocean outfalls in Sydney and nearby regional towns observed mortalities to the eastern hulafish due to direct exposure to the discharge of sewage wastewater plume.

Recent environmental impact studies on fish assemblages for other STPs in NSW that also have a regional setting similar to Merimbula STP include:

- Coffs Harbour Sewerage Strategy, as detailed in an EIS for the project (CEE, 2000);
- Northern Shoalhaven Reclaimed Water Management Scheme, as detailed in ocean release monitoring for the Penguin Head outfall near Culburra (The Ecology Lab, 2005); and
- Milton-Ulladulla Sewerage Augmentation, with post-extended ocean release monitoring (The Ecology Lab, 2008).

At Coffs Harbour, Smith *et al.* (1995) investigated levels of contaminants (seven metals and 12 organochlorines) in the reef fish Red morwong (*Cheilodactylus fuscus*). Low levels of contaminants were recorded and were well within the guideline limits set by National Health and Medical Research Council (NHMRC) for safe consumption. Furthermore, the origin of these contaminants could not be attributed to the discharge of treated sewage wastewater due to the numerous diffuse sources of contaminants also entering the ocean in catchment runoff and stormwater.

At Penguin Head, near Culburra, The Ecology Lab (2005) monitored the fish assemblage of shallow and deep reef habitats in 1995 and 1997 adjacent to the intertidal release of treated wastewater. Monitoring found that there was little to no evidence of an impact by the release of treated wastewater, as determined by the quantitative comparisons between Penguin Head and control locations at Crookhaven Head and Kinghorn Point.

At Milton-Ulladulla, The Ecology Lab (2008) monitored fish assemblages at a number of locations pre- and post-commissioning of the extended offshore ocean release. While differences in the fish assemblage were detected between pre- and post-commissioning surveys, none of the differences could be unequivocally attributed to the operation of the offshore release. It was considered that any influence of the release on fish assemblages was likely very localised and may have occurred at a spatial scale smaller than what was monitored. Observations by The Ecology Lab (2008) suggested that the diffuser may have actually enhanced fish diversity by providing a structure in which they could shelter.

The local fish assemblage of Merimbula Bay has been exposed to the discharge of treated sewage wastewater since early 1971 when the beach-face outfall was first commissioned. The Project includes STP upgrades to improve wastewater quality (including reduction in nutrients and metals) and would discharge treated wastewater to a deep-water location over sand habitat that is distant from subtidal reef communities.

Potential effects of the Project on fish assemblage of Merimbula Bay can be categorised as:

- direct (e.g. affecting fish physiology/condition) or indirect (e.g. affecting available habitat) or both;
- lethal (e.g. death of adults/juveniles from physiological stress) or sublethal (e.g. reduced fecundity or successful larval recruitment); and
- either negative (net decrease in abundance and diversity through time) or positive (net increase in diversity and abundance through time) or neutral and no detectable change through time

Each potentially causative factor from the Project construction and operation and its associated effect(s) can be theoretically assessed for likelihood and risk according to a systematic consideration of all three of these classes of category. Assessment of the potential effects on the fish assemblage of Merimbula Bay that could be reasonably attributed to the construction and operation of the Project is provided in **Sections 5.9** and **5.10** respectively.

## 5.3 Review of Existing Data

Review of existing data included:

- Records of fish observations reported for survey locations between Batemans Bay (NSW) and Mallacoota (Victoria) by Reef Life Survey (RLS, 2017)
- Correspondence with DPI Fisheries regarding observations of threatened species, commercial catch data and recreational fishing statistics.

### 5.3.1 Reef Life Survey Data

Reef Life Survey global reef fish dataset was reviewed for all records of bony fishes and elasmobranchs observed from RLS sites between Batemans Bay and Mallacoota (RLS,2017). The dataset includes fish species recorded by diver surveys on nearshore reef habitats in 2m to 24m depths. A total of 203 fish species have been recorded over this range and while not all of these species have been observed from RLS sites in the BVSC region, there is high likelihood they may occur (**Appendix C-1)**. Of this list of species, 93 species have been recorded from a subset of 14 RLS sites in the BVSC region between Aragunnu and Green Cape (RLS 2017), approximately 40 km to the north and south of the study area respectively (**Figure 5-1**).

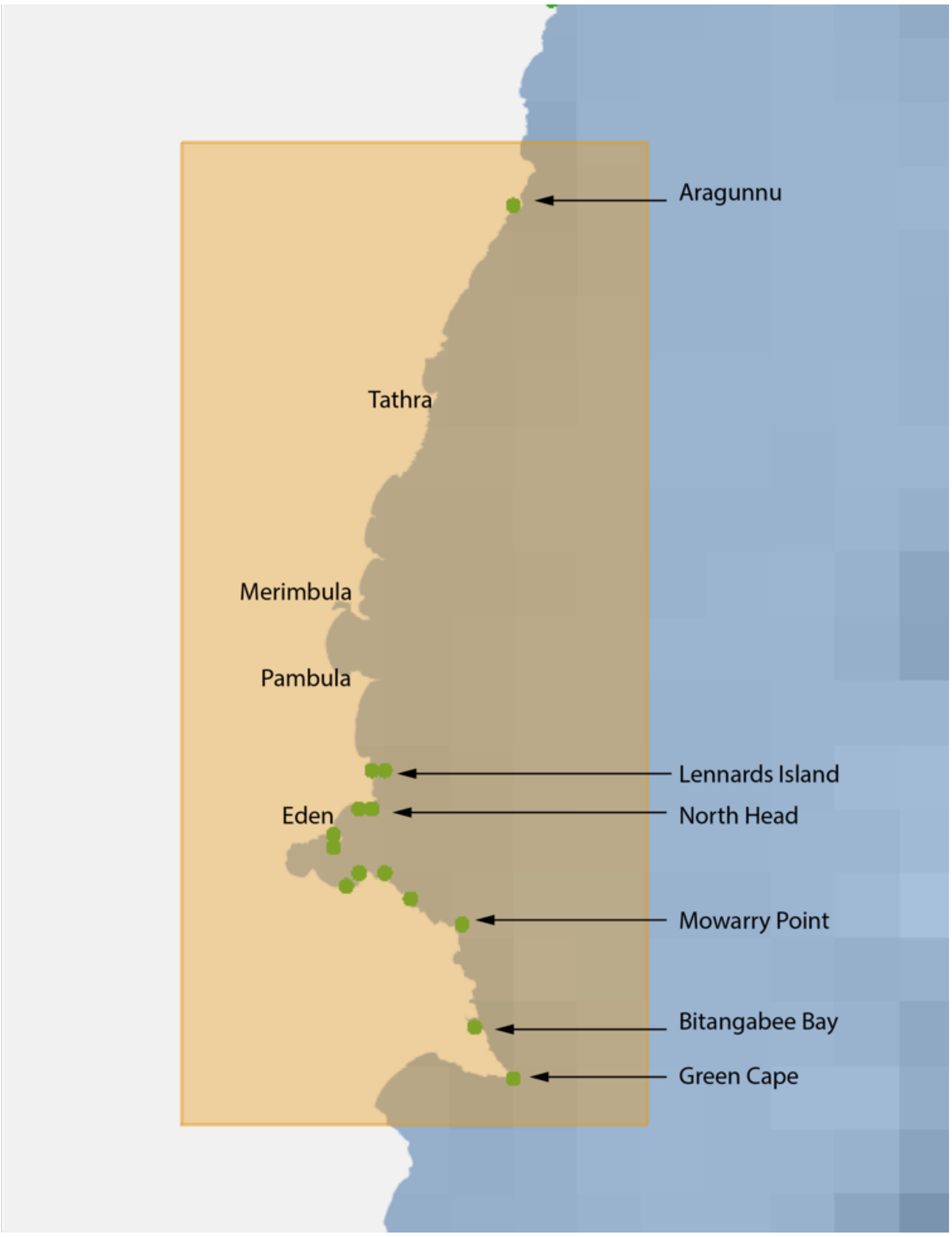



**Figure 5-1 Reef Life Survey (RLS) sites in Bega Valley Shire region (green dots)**

### 5.3.2 Threatened and Protected Fish Species

Five fish taxa listed as threatened or protected under the *FM Act* and or *EPBC Act* are either reported from or considered to have moderate to high likelihood of occurrence within Merimbula Bay. These include the black cod, grey nurse shark, great white shark, southern bluefin tuna and members of the syngnathiformes (seahorses, pipefishes, pipehorses, sea moths).

According to sighting records held by ALA (2020) and RLS (2017), there have been four recorded observations of the black cod in the BVSC region since 1972, none from Merimbula. These include Twofold Bay in 1972, Bitangabee Bay and Green Cape in 1989, and Bermagui in 2005. More regular sightings of black cod are reported from locations in the Eurobodalla shire region such as the Narooma breakwater and Montague Island (RLS 2017). With only 4 confirmed sightings since 1972, the occurrence of a local viable population of black cod in the BVSC region would be considered rare.

The grey nurse shark (GNS) has been reported from at least seven locations in the BVSC region (Refer

Appendix 1 in DoE, 2014) including at Tura Head and Twofold Bay, to the north and south of Merimbula Bay respectively. In addition, GNS sightings have been reported near to the Merimbula wharf on five occasions (2/12/2016, 8/4/2017, 12/10/2017, 21/1/2018, 11/12/2018 as listed by ALCW 2020). No GNS were observed during marine ecology field surveys undertaken for this Project. The nearest known aggregation site of the GNS considered critical to the species survival is Montague Island approximately 80 km north of the study area (DoE, 2014).

White shark individuals are sighted along the BVSC coast each year, typically in spring to summer period with the pattern of sighting records coinciding with the southerly migration of humpback whales. No white sharks were observed during marine ecology field surveys undertaken for this Project.

Southern bluefin tuna are a highly migratory pelagic fish targeted by recreational fishers in BVSC coastal waters each year, typically during July to January period as the species migrates northwards. The species is usually observed in deep offshore waters along the continental shelf and rarely sighted within Merimbula Bay.

Ten species of syngnathiformes are recorded from the BVSC region (ALA, 2020). The majority are typically found in seagrass and algal habitats of estuaries and protected embayments. Two species that may be observed in coastal habitats of Merimbula Bay and Haycock Point include the bigbelly seahorse and the weedy seadragon. A nearest known population of the weedy seadragons is reported from *Posidonia* seagrass and algal habitats in East Boyd Bay (Eden) (Wilson *et al.,* 2016) approximately 30 km to the South of Merimbula Bay.

A detailed assessment of threatened and protected fish species is provided below in **Section 13 – Threatened and Protected Marine Species**.

### 5.3.3 Fish Species Important to Commercial and Recreational Fishing

**Commercial Fisheries**

Commercial fisheries operating in the vicinity of the study area include abalone fishery, lobster fishery, sea urchin and turban shell fishery, ocean haul fishery, ocean trawl fishery, ocean trap and line fishery and estuary general fishery. DPI Fisheries were consulted with regard to reported annual commercial catch data for the Pambula region for the area shown in **Figure 5-2**. A full list of species commercially targeted in the Pambula region ranked from highest to lowest mean annual catch over 8-year period 2009 to 2017 is provided in **Appendix C-2.** The data excludes lobster, sea urchin and turban shell fisheries due to data sensitivities.

Based on fisheries data provided, there are over 130 species targeted within the Pambula region with a combined total mean annual catch of 72.5 tonne. The ocean haul fishery is the most productive for the region, in terms of catch weight, producing the greatest mean annual catch weights for individual species that includes the Australian salmon (*Arripis trutta*) and Australian sardine (*Sardinops sagax*), with mean annual catches of 40 tonne and 34.6 tonne respectively. The abalone fishery is also a significant fishery within the region, producing the third highest mean annual catch of 5.6 tonne (**Appendix C-2**).

The ocean trawl fishery is a diverse fishery within the Pambula region, targeting both prawns and fish, however, it should be noted that the ocean trawl fishery is not permitted to fish the waters of Merimbula Bay west of a line drawn from the eastern extremity of Long Point southerly to the easternmost extremity of Haycock Point (DPI, 2018a). Despite these spatial restrictions, the ocean trawl fishery still produces large catches of a range of species including eastern school whiting, silver trevally and tiger flathead each year from the nearby waters outside of Merimbula Bay. The ocean trap and line fishery is another diverse fishery targeting a wide array of demersal and pelagic fish within NSW coastal waters through multiple methods. Key fish species targeted by this fishery within the Pambula region include snapper, yellowtail kingfish, tuna and silver trevally.

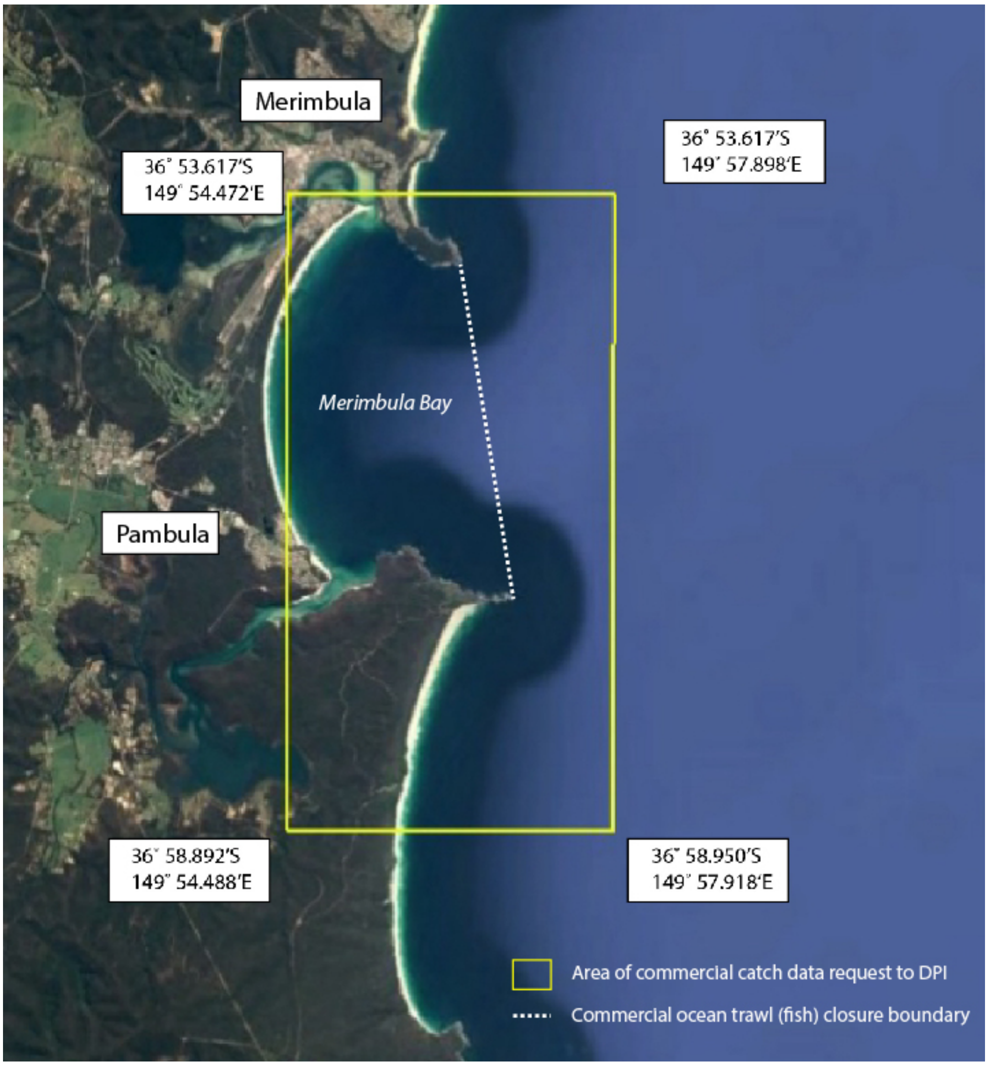



**Figure 5-2 Commercial fisheries reported annual catch data provided for Pambula area (yellow rectangle) for 2009 to 2017**

**Recreational Fishing**

Recreational fishing is popular within the Merimbula region, with a wide variety of areas and fish habitat accessible to anglers, including estuarine, coastal and offshore environments. Although a wide array of species are targeted by recreational fishers throughout the region, a smaller subset of key recreational species account for much of the catch. The key marine recreational fishing species on the south coast of NSW are dusky and sand flathead, yellowfin and black bream, tailor, snapper, sand whiting and Australian salmon (West *et al.,* 2015).

Haycock Point and Hunter reef are both locally renowned for their recreational fishing and a number of offshore fishing charter operators based in Merimbula regularly target these fishing grounds. The deep reefs around Haycock Point and Hunter Reef produce catches of some of the aforementioned key species as well as an array of other highly prized recreational angling species such as gummy shark and kingfish. Catch compositions at these locations and throughout the wider southern NSW coast are seasonally variable. An example of the seasonality would be the higher abundance of Australian salmon in coastal waters during the cooler months of winter and spring, whereas summer and autumn exhibit an increase in warmer water pelagic species such as bonito, tuna and kingfish.

### Merimbula Offshore Artificial Reef (OAR)

The Merimbula offshore artificial reef (OAR) was established in 2018 to enhance the region's recreational fishing opportunities. The OAR is comprised of two pinnacle-like steel structures situated on sandy seabed in 30 m depth south of Long Point (**Figure 5-3**). The Merimbula OAR provides complex vertical habitat situated over an otherwise featureless sandy substrate and is expected to attract a range of demersal and pelagic species (DPI, 2018b). The influence of the Merimbula OAR on surrounding fish communities is as yet unknown, though some inferences may be drawn from observations recorded of fish communities using the Sydney OAR. Similar in construction to the Merimbula OAR, the Sydney OAR was deployed approximately 2 km south-east of South Head off Sydney Harbour in October 2011. The Sydney OAR now provides habitat for over 49 species of fish, including many species important for recreational fisheries. Acoustic telemetry studies have revealed that there is some connectivity between the Sydney OAR and surrounding reefs. Some species using the OAR have been shown to move between the artificial structure and surrounding natural reef habitats (Kellar *et al.,* 2017). Considering the success of the Sydney OAR, there is an expectation the Merimbula OAR would also provide improved recreational fishing opportunity within Merimbula Bay.

The Merimbula OAR is approximately 1000m north of the proposed diffuser location. According to dispersion modelling predictions (AECOM, 2019c), water quality at the Merimbula OAR would not be affected by the treated wastewater discharge. Potential effects on fish assemblage from treated wastewater discharge is discussed in **Section 5.10** below.

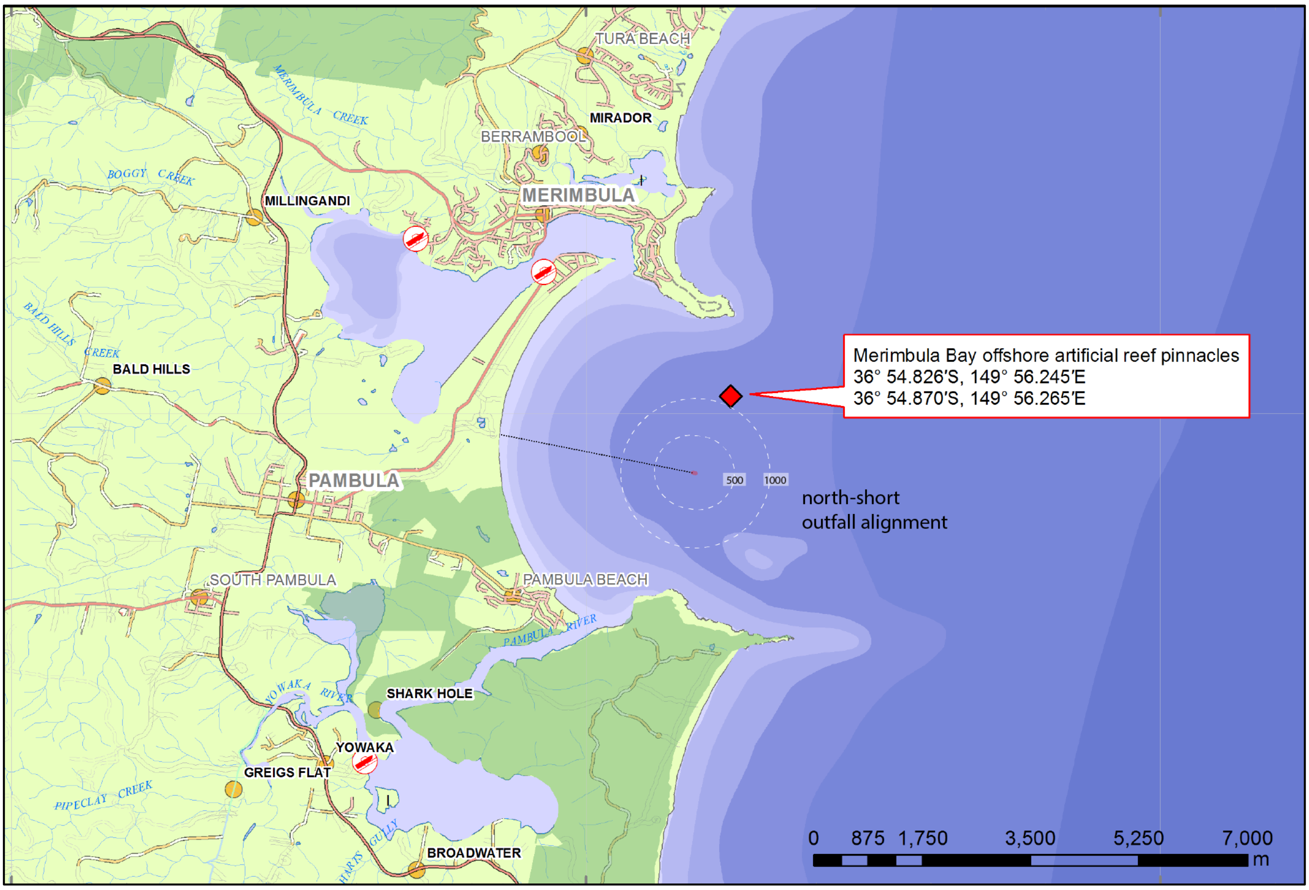



**Figure 5-3.** Location of the Merimbula Offshore Artificial Reef (OAR) in relation to the Project.

**Australian Salmon (*Arripis trutta*)**

Australian salmon is a strongly schooling inshore pelagic fish commonly found adjacent to ocean beaches, reefs and headlands and also at times in large embayments and estuaries up to 30m depth (Gibbs and Jones, 2009). The species distribution range in south-east Australia extends from Tasmania, central and eastern Victoria, throughout NSW to the northern limit of the species range at Brisbane, Queensland. The population is a single well mixed stock that shows a trend in size and age structure with the largest and oldest fish being found further north over the population range. As reproductively mature adult (4 years old), the species makes a one-way migration from southern waters to northern waters. In NSW, spawning of salmon occurs in nearshore coastal waters between October and March, with peak spawning occurring in November (Gibbs and Jones, 2011). The southward flowing EAC current plays an important role in advection of larvae back to cooler southern waters and is considered critical to the species life history.

In terms of broader environmental change, increasing ocean water temperatures associated with strengthening of the EAC current may lead to a southward shift in species distributional range based on evidence that salmon have preference for waters cooler than 23 $^{o}$C. Changing EAC current conditions could have some effect on larval transport patterns and the extent of adult population. However, Gibbs and Jones (2009) consider there is high larval dispersal capacity into the many ideal recruitment areas across south-east Australia, to buffer any observable localised population effects that may be attributable to climate change.

The primary stressor to the species is the commercial and recreational fishing sectors. The commercial catch of Salmon in NSW is currently at the higher end of historical landings and the recreational catch is also substantial estimated at 150-210 tonnes per annum (Henry and Lyle, 2003). The NSW fishery is listed as ‘fully fished’ with commercial landings variable between 500 and 1,500 tonnes per year (Gibbs and Jones, 2011). At a local level, the mean annual commercial catch of salmon from the Pambula region between 2010-2017 was 40 tonne (**Appendix C-2**).

**Australian Sardine (*Sardinops sagax*)**

Australian sardine is a small pelagic fish that is a target species for the commercial ocean haul fishery in south-eastern Australia. The average total annual commercial catch of sardine from the Pambula region between 2010-2017 was approximately 34.5 tonne (**Appendix C-2**), representing the second most important local fishery in terms of gross catch. A small proportion of sardine may also be caught by recreational fishers. Highest catches occur in winter/spring and the product is locally sold for bait, pet food and for human consumption (Scandol *et al*., 2008).

**Mullet (*Mugil cephalus*)**

Sea mullet occur in waters of all Australian states and in NSW, are primarily targeted by the ocean haul fishery operating along beaches. The majority of the catch is taken during the autumn/winter pre-spawning period, when adults are either aggregating in the lower reaches of estuaries or migrating along ocean beaches. Mullet are caught in smaller quantities within estuaries at other times of the year. The average total annual catch of mullet from the Pambula region between 2010-2017 was approximately 407 kg (**Appendix C-2**). Relatively few sea mullet are taken by recreational fishers in NSW.

**Snapper (*Sparus aurata*)**

Snapper are a demersal reef fish and a popular local target species for recreational fishermen and charter operators fishing the offshore waters around Hunter Reef and Haycock Point. The species is also caught by the commercial ocean trap and line fishery with the average total annual catch from the Pambula region between 2010-2017 approximately 270 kg (**Appendix C-2**). When compared to other commercially fished species in the region, the snapper catch itself is small and the species is likely more valuable to the local recreational charter fishing sector.

Snapper are known to migrate from estuaries in eastern Victoria to NSW, but the importance of this crossborder migration to the replenishment of the NSW fishery is not clear (Coutin *et al*., 2003). Local estuarine habitats of Merimbula and Pambula Lake are also important nursery areas to juvenile snapper where they associate with soft sediments, reefs, algae and seagrass. As snapper grow and mature, they move to rocky reef habitats of coastal inshore waters, moving to offshore waters at adult maturity 3-5 years of age (Hutchinson, 2011).

## 5.4 Field Survey Methods

Baited Remote Underwater Video (BRUV) was used to investigate the local fish assemblage associated with demersal habitats within the study area. Demersal habitat types within the study area included sandy seabed and subtidal rocky reef, with reef further categorised as barrens habitat, or macroalgae dominated reef. Stage 1 fieldwork sampled all habitat types between 3-9 November 2017, with Stage 2 fieldwork undertaken on 10 October 2019 to focus specifically on the fish assemblage associated with sand habitats around the preferred alignment option.

BRUVs are widely used in Australia to provide robust estimates of fish assemblages in an independent and non-destructive manner. In NSW, the approach has been adopted and optimised by NSW Marine Parks (Gladstone *et al.* 2012, Harasti *et al.* 2015, Rees *et al.* 2015). The approach employed in this study is based on guidance recommended in recent BRUVs research for deployment times and data analysis.

BRUV units are a truncated pyramidal design (0.6 m x 0.6 m at the base and 1m tall) constructed of stainless steel, comprising a GoPro camera, and a bait container containing 500g of crushed pilchards (**Figure 5-4**). Weights were fixed at the frame base to stabilise the BRUV units on the seabed in strong currents.

In addition to the BRUV survey, observations of fish were also recorded from tow video footage during broadscale surveys of the seabed allowing for additional species not attracted to the bait plume of the BRUV to be recorded as present within the study area.

**Figure 5-4 BRUV unit with GoPro camera used for sampling fish assemblages (*left*), deployment check to ensure correct habitat being sampled (*right)***

### 5.4.1 Sample Sites

A total of 30 BRUV drops were deployed across the study area over Stage 1 and Stage 2 works (**Figure 5-5**). BRUV deployments were distributed across three major habitat types including reef, macroalgae covered reef and sandy substrate (**Table 5-1**). Five BRUV drops were deployed within a hypothetical 500 m buffer radius surrounding the proposed ‘North-Short’ outfall diffuser location, including at the diffuser location (BRUV21) with additional BRUVs deployed at varying distances away from the diffuser.

Sample sites were evaluated and selected through an initial review of seabed bathymetry and substrate type using a drop video to ensure each BRUV unit was deployed in the correct target habitat – reef, macroalgae, or sand.

DPI Fisheries were consulted regarding anecdotal reports and records of threatened and protected fish species (*i.e.* Black cod, Grey nurse shark, Great white shark) in considering the selection of sampling sites. While no specific sampling sites were recommended by DPI with regards to threatened species, location coordinates were provided for an area of subtidal reef in 40 m depth known to be productive for fishing and considered worthy of investigation.

Images of habitat types and depths for each BRUV deployed are provided in **Appendix C-3.**

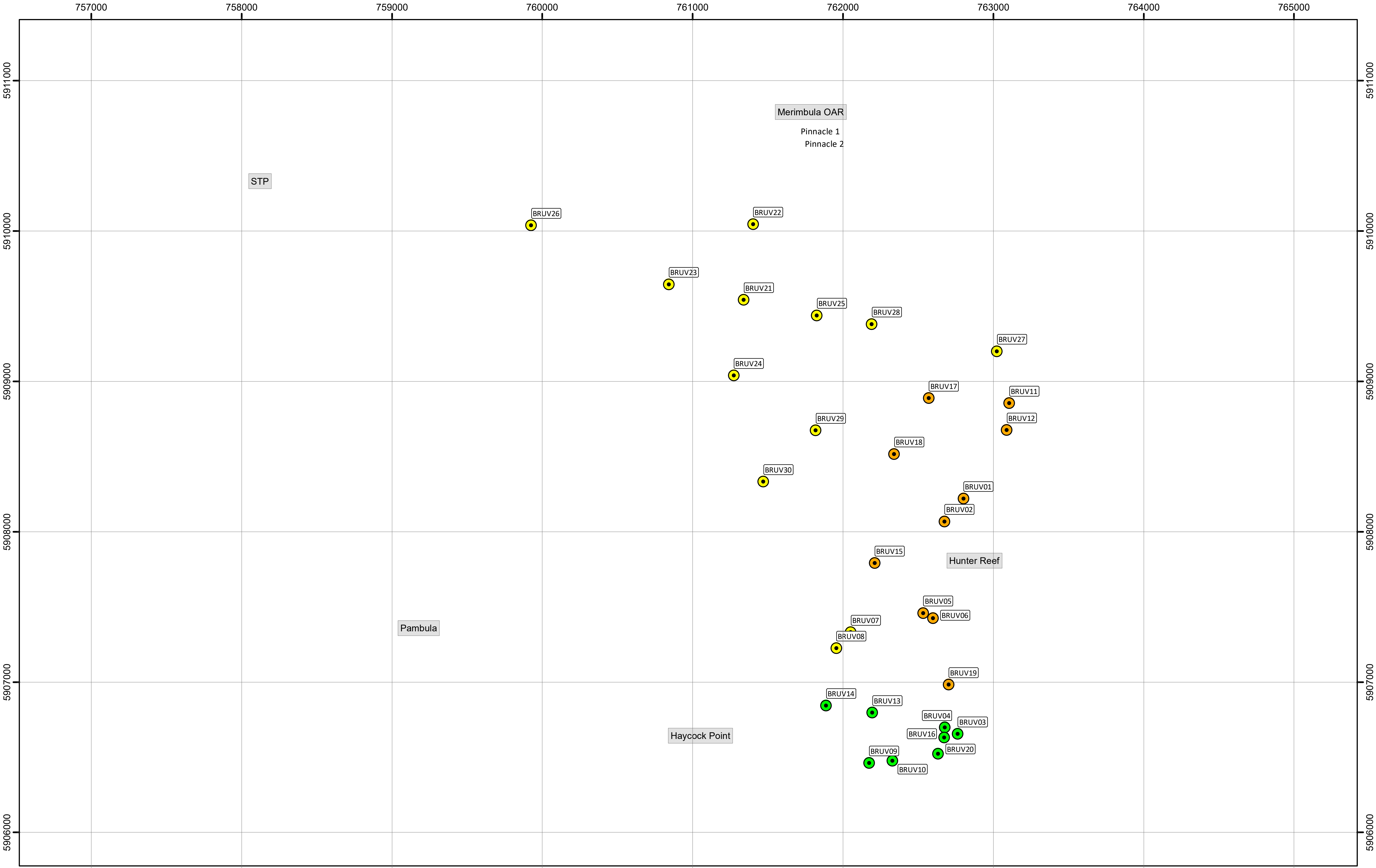
**Table 5-1 Summary of BRUV location coordinates, habitat types and depth sampled**

| BRUV # | Habitat | Depth (m) | Latitude | Longitude | Rationale | Distance to nearest reef (m) |
|---|---|---|---|---|---|---|
| BRUV01 | Reef | 29 | -36.935418 | 149.950744 | Stage 1 - baseline fish assemblage | 0 |
| BRUV02 | Reef | 28 | -36.936824 | 149.949380 | Stage 1 - baseline fish assemblage | 0 |
| BRUV03 | Macroalgae | 20 | -36.949521 | 149.950856 | Stage 1 - baseline fish assemblage | 0 |
| BRUV04 | Sand | 21 | -36.949165 | 149.949876 | Stage 1 - baseline fish assemblage | 20 |
| BRUV05 | Reef | 23 | -36.942348 | 149.948005 | Stage 1 - baseline fish assemblage | 0 |
| BRUV06 | Reef | 20 | -36.942632 | 149.948753 | Stage 1 - baseline fish assemblage | 0 |
| BRUV07 | Sand | 21 | -36.943632 | 149.942645 | Stage 1 - baseline fish assemblage | 160 |
| BRUV08 | Sand | 22 | -36.944611 | 149.941601 | Stage 1 - baseline fish assemblage | 220 |
| BRUV09 | Sand | 10 | -36.951425 | 149.944314 | Stage 1 - baseline fish assemblage | 50 |
| BRUV10 | Sand | 10.5 | -36.951248 | 149.946055 | Stage 1 - baseline fish assemblage | 20 |
| BRUV11 | Sand | 39 | -36.929619 | 149.953941 | Stage 1 - baseline fish assemblage | 50 |
| BRUV12 | Reef | 39 | -36.931230 | 149.953815 | Stage 1 - baseline fish assemblage | 0 |
| BRUV13 | Macroalgae | 5 | -36.948407 | 149.944442 | Stage 1 - baseline fish assemblage | 0 |
| BRUV14 | Macroalgae | 8 | -36.948063 | 149.940974 | Stage 1 - baseline fish assemblage | 0 |
| BRUV15 | Reef | 13 | -36.939451 | 149.944268 | Stage 1 - baseline fish assemblage | 0 |
| BRUV16 | Macroalgae | 16 | -36.949760 | 149.949863 | Stage 1 - baseline fish assemblage | 0 |
| BRUV17 | Reef | 35 | -36.929466 | 149.947926 | Stage 1 - baseline fish assemblage | 0 |
| BRUV18 | Reef | 29 | -36.932879 | 149.945473 | Stage 1 - baseline fish assemblage | 0 |
| BRUV19 | Reef | 24 | -36.946590 | 149.950067 | Stage 1 - baseline fish assemblage | 0 |
| BRUV20 | Sand | 15 | -36.950752 | 149.949449 | Stage 1 - baseline fish assemblage | 50 |

| BRUV # | Habitat | Depth (m) | Latitude | Longitude | Rationale | Distance to nearest reef (m) |
|---|---|---|---|---|---|---|
| BRUV21 | Sand | 30 | -36.923919 | 149.933895 | Stage 2 - @ North Short diffuser (Option 1) | 1220 |
| BRUV22 | Sand | 30 | -36.919374 | 149.934426 | Stage 2- 500m N of North-Short diffuser | 1540 |
| BRUV23 | Sand | 27 | -36.923132 | 149.928277 | Stage 2- 500m W of North-Short diffuser | 1650 |
| BRUV24 | Sand | 28 | -36.928474 | 149.933328 | Stage 2 - 500m S of North-Short diffuser | 970 |
| BRUV25 | Sand | 33 | -36.924717 | 149.939389 | Stage 2 - 500m E of North-Short diffuser | 800 |
| BRUV26 | Sand | 21 | -36.919853 | 149.917877 | Stage 2- 1250m W of North-Short diffuser | 2600 |
| BRUV27 | Sand | 40 | -36.926532 | 149.952903 | Stage 2 - @ North Long diffuser (Option 2) | 275 |
| BRUV28 | Sand | 35.9 | -36.925137 | 149.943499 | Stage 2 - Midway point between North Short and North Long diffuser options | 570 |
| BRUV29 | Sand | 31.3 | -36.931603 | 149.939565 | Stage 2 - 1000m SE of North Short diffuser | 275 |
| BRUV30 | Sand | 27 | -36.93477 | 149.935777 | Stage 2 - 1250m S of North Short diffuser | 570 |

Note:

- Stage 1 survey (3-9 November 2017) to collect baseline data to describe fish assemblage in study area
- Stage 2 survey (10 October 2019) to collect data on sand habitats around the preferred North-Short diffuser option
- BRUVs highlighted yellow sampled the fish assemblage of sand habitats within 500m radius buffer of North-Short diffuser option




N

Scale = 1:22,000

0 250 500 1,000 Meters

- Area of Investigation Boundary
- Pipeline Alignment
- Below Ground
- Diffuser
- SEARs 500m bufffer from diffuser
- Bathymetry (1m intervals)
- BRUVs on Reef
- BRUVs on Sand
- BRUVs on Macroalgae

Notes:
1. Elgin BRUV sampling conducted Stage 1: 3-9 November 2017, Stage 2: 10 October 2019.
2. Southern bathymetry data reported by Marine and Earth Sciences in 2017
3. Northern bathymetry data reported by Southern Divers and Total Hydrographic in 2017

**Project:**
MERIMBULA STP UPGRADE AND OCEAN OUTFALL ENVIRONMENTAL ASSESSMENT

**Client:**
AECOM AUSTRALIA

**FIGURE 5-5**

LOCATION OF BRUV SAMPLE SITES

| Date: 22 September 2020 | |
|---|---|
| Version: 1 | Size: A3 |

Document Path: D:\Elgin Associates Dropbox\Elgin_GIS\New_South_Wales\BVSC GIS Data\Merimbula EIS\1. MXD Mapping Files\Stage2_Reporting\Figure_6-5_BRUV_locations_17Sept2020.mxd

## 5.5 Data analysis

For each BRUV drop, 30 minutes of the video footage was reviewed and fish species observed recorded to establish key metrics species richness and maximum number of fish taxa recorded per video frame (*maxN*) and cumulative maxN over duration of 30 minutes of video footage. All video analysis commenced at 2 minutes elapsed time to allow the bait plume to establish and ensure recorded fish assemblages were comparable with footage from other BRUV drops. All clearly visible fish observed within the field of view were identified and recorded to the lowest possible taxonomic level. Particular attention was given to the possible presence of threatened fish species during video footage analysis. Fish assemblage data was analysed using software PRIMER 7.0 to explore community composition and patterns in fish diversity and abundance between different habitats.

**Diversity Measures**

Species diversity was estimated by total taxa count (*i.e.* taxon richness as *S*) and Shannon diversity (*H'*) index. Shannon diversity takes into account the abundance and evenness of species within a community with index values increasing as both the species richness and evenness of the community increase. Species richness describes the number of different species in an area (more species = greater richness). Species eveness describes the relative abundance of the different species in an area (similar species abundances = more eveness).

A community with high Shannon index can be interpreted as having greater overall diversity with high species richness and high abundance of those species, compared to a community with low Shannon index that may have equally high species richness yet is dominated by high abundances of just a few species. The latter scenario is common in marine communities where habitat patchiness is a major factor influencing the occurrence and distribution of species.

## 5.6 Results

The discussion below characterises the fish assemblage typically associated with each of the three main habitat types – reef, macroalgae and sand, and provides some baseline comparison of the fish assemblage found within the 500 m radius buffer of the proposed diffuser to broader sand habitats of Merimbula Bay.

### 5.6.1 Summary Statistics

A total of 1549 fish were observed comprising 63 species from 33 families (**Appendix C-4**). The majority of the species observed were bony fishes (78% of total), with rays and sharks representing 22% of the fish assemblage.

The three most species-rich groups on reef habitat were the Monacanthidae (leatherjackets, 8 spp.), Labridae (wrasses, 5 spp.) and Serranidae (groupers and sea perches, 4 spp.), while for sand habitat the most species-rich groups were Urolophidae (stingarees, 4 spp), Rhinobatidae (guitarfishes, 2 spp) and Platycephalidae (flathead, 2 spp).

In terms of the most abundant fish taxa observed over all samples, they were Australian mado (*Atypichthys strigatus, maxN=467),* yellowtail horse mackerel (*Trachurus novaezelandiae, maxN=293*), silver sweep (*Scorpis lineolate, maxN=116)*), silver trevally (*Pseudocaranx georgianus*, maxN=147) and eastern sand whiting (*Sillago flindersi, maxN=102*). Australian mado were recorded in the majority of the BRUV deployments and across all the habitats surveyed. Australian mado is primarily considered a reef species that largely forage on zooplankton, however, the species has demonstrated flexibility in its foraging behaviour potentially contributing to the species success (Glasby and Kingsford, 1994).

Rarely encountered fish species, those where only one individual was observed in only one BRUV deployment, included the following ten (10) species - rock cale (*Aplodactylus lophodon*), silver dory (*Cyttus australis*), luderick (*Girella tricuspidata*), zebra fish (*Girella zebra*), bastard trumpeter (*Latridopsis forsteri*), tootbrush leatherjcket (*Acanthaluteres vittiger*), flounder sp., eastern barred grubfish (*Parapercis allporti*), bronze whaler shark (*Carcharhinus brachyurus*), school shark (*Galeorhinus galeus*), Melbourne skate (*Spiniraja whitleyi*), and Kapala stingaree (*Urolophus kapalensis).*

Images of each of the 63 species recorded in BRUV deployments are provided in **Appendix C-5**.

### 5.6.2 Species Richness and Abundance

Mean species richness was highest over reef habitats (15 spp.), followed by macroalgae-covered reef (13 spp.) with the lowest mean number of species (7 spp.) recorded over sand **(Figure 5-6).** The greatest variation was observed between macroalgae-reef samples as indicated by the standard error around the mean.

Mean abundances of fish, as indicated by the maxN metric, by habitat type is presented in **Figure 5-7**. Highest mean abundance of fish was recorded over macroalgae-covered reefs largely due to the high numbers of Australian mado (*Atypichthys strigatus*)**.** Sand and reef habitat showed similar levels of mean abundance, though much lower than macroalgae-covered reefs.

In terms of overall diversity, as indicated by Shannon index, reef habitat is considered the most diverse as it was characterised by the highest number of species and each species has good abundance (*i.e.* the assemblage is comprised of relatively even abundances of each species) (**Figure 5-8**). In contrast, the lower Shannon index of sand and macroalgae habitats indicates those fish assemblages to be characterised by an overall lower number of species with the assemblage dominated by high abundances of a few species (**Figure 5-8**).

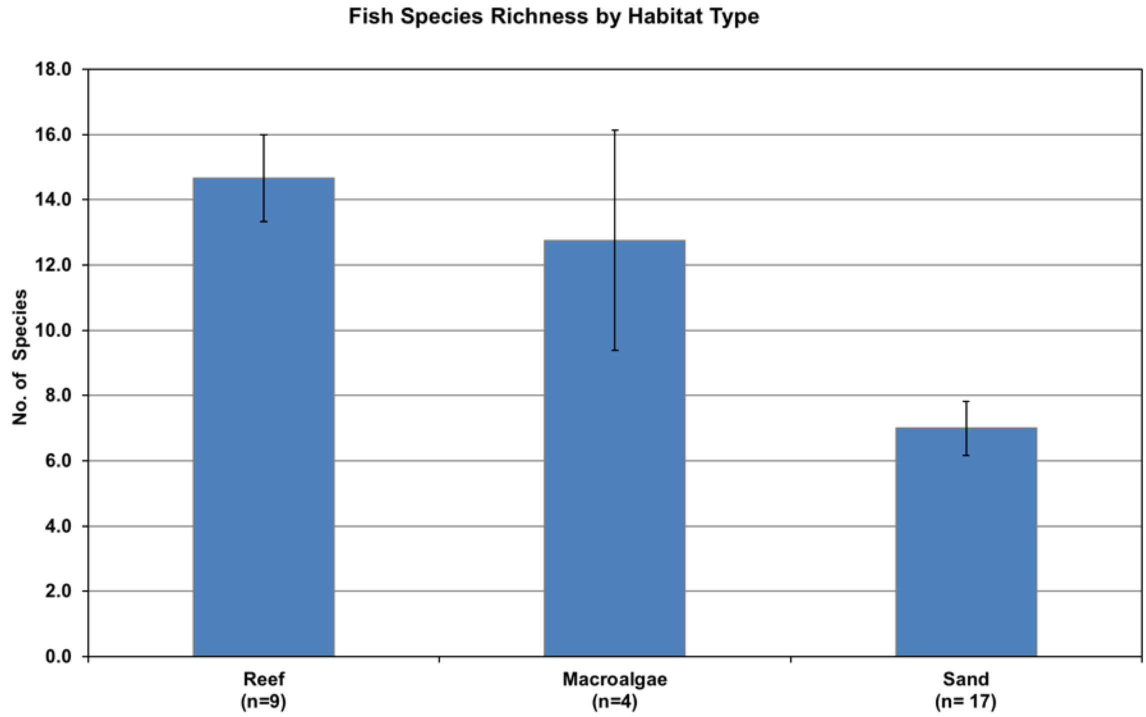



**Figure 5-6 Mean fish species richness by habitat type**

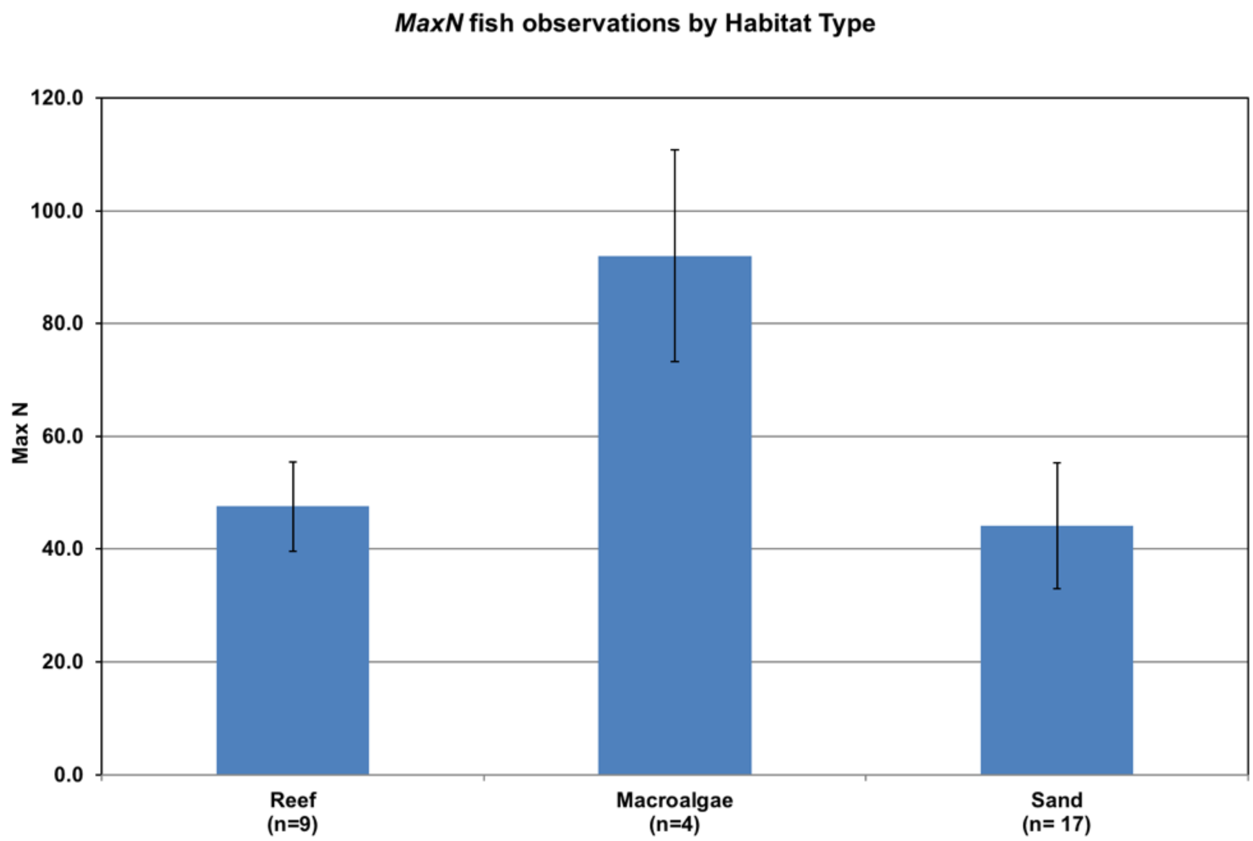



**Figure 5-7 Mean fish abundance (*maxN observations*) by habitat type**

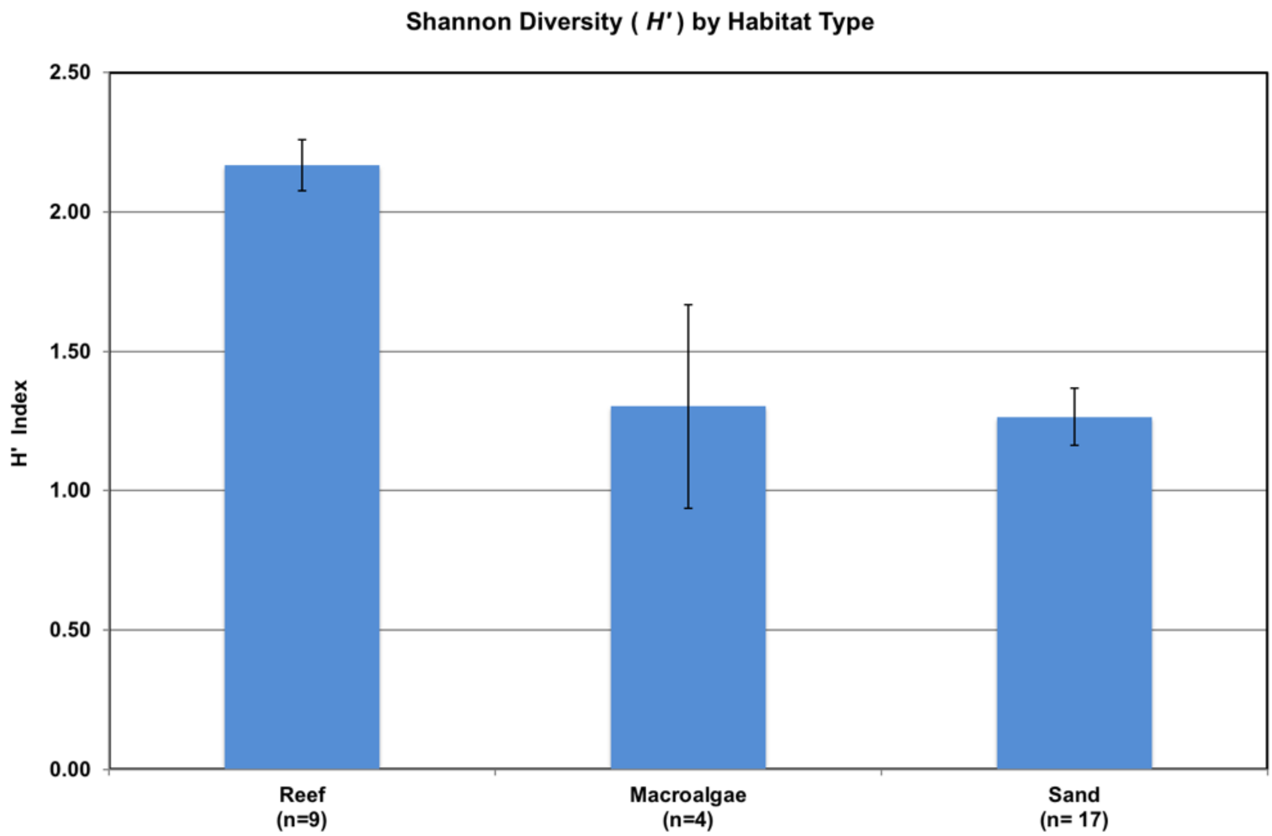



**Figure 5-8 Mean Shannon diversity of fish assemblage by habitat type**

### 5.6.3 Sand habitat

The fish assemblage diversity associated with sand habitat was further examined by considering the mean species richness observed within the 500 m radius buffer for the proposed diffuser (n=5), as well as for sand habitats far (n=10) and close (n=7) to reef areas (**Figure 5-9)**. A distance of 500m was considered far from reef, while less than 500 m was considered close enough to reef that there was potential for typical reef fish species to be attracted and respond to the BRUV bait plume resulting in a higher species richness that is not necessarily typical for sand habitat in Merimbula Bay.

For sand habitats within the 500 m radius buffer of the proposed diffuser, mean species richness was six, although there is evidently a high level of variability across that 500 m buffer area as indicated by the large standard error. Six species was also found to be the typical level of diversity for other sand habitat far from reef areas. A higher level of species diversity was found for sand habitats within 500 m of reef areas (mean spp richness = 8) due to mixed assemblage of sand and reef dwelling species. Some reef dwelling species such as the Port Jackson shark were also observed on sand habitats greater than 500 m from reef areas as shown in **Figure 5-10.**

In terms of abundance, much lower fish abundance was observed on sand habitats within the 500 m radius buffer and far from reef areas (mean maxN = 15), compared to sand habitats near to reef areas (maxN = 70) (**Figure 5-11**). Furthermore, the high abundance for the fish assemblage on sand habitats was attributed to a few species as is reflected in the low shannon index (0.57) (**Figure 5-12**).

For the diffuser site and indicative of the 25 m mixing zone (as indicated by BRUV21), only three species were observed and at very low abundances. These included the Eastern fortesque (*Centropogon australis*), flounder (*Pseudorhombus* sp.) and bluespotted flathead (*Platycephalus caeruleopunctatus*). Similarly, low fish diversity and abundance was observed at BRUV24, 500 m to the south. Further sampling would need to be undertaken of the diffuser site to have more confidence in the fish diversity and abundance trend at that location.

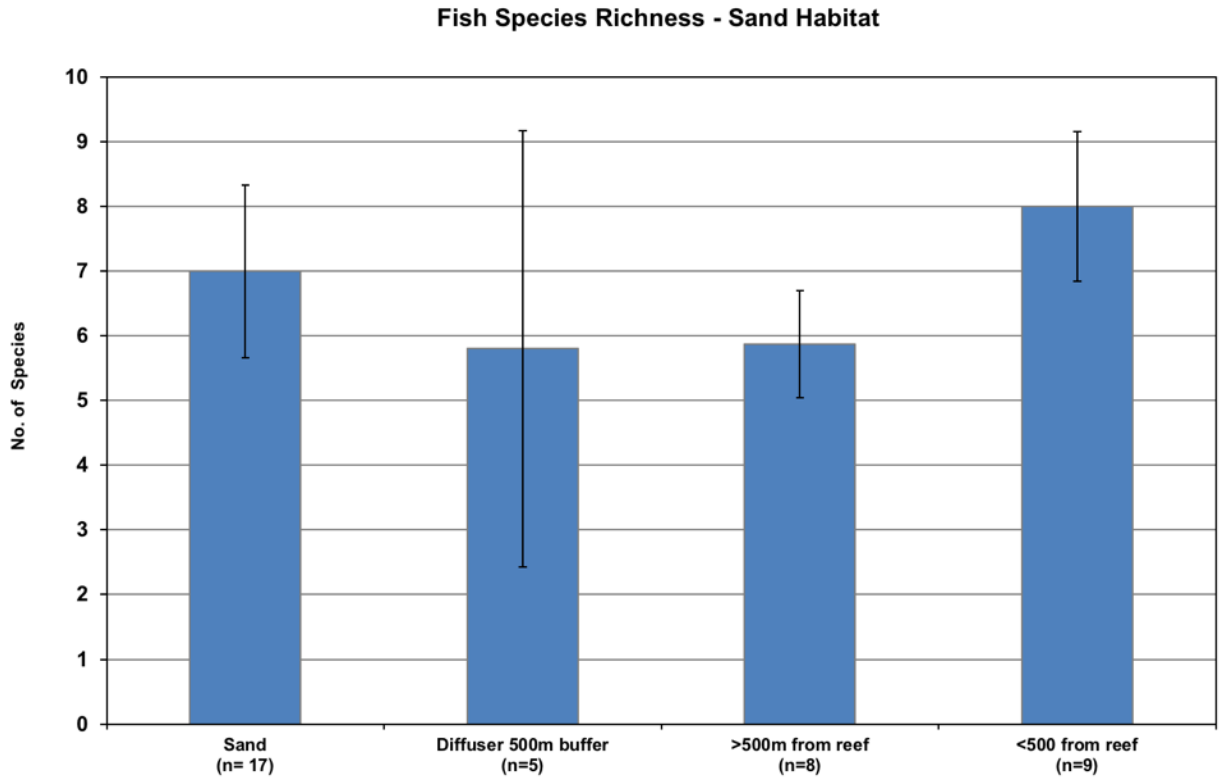



**Figure 5-9** **Mean fish species richness on sand habitat within diffuser 500 m buffer compared to sand habitat more than, and less than 500 m distance from reef areas**

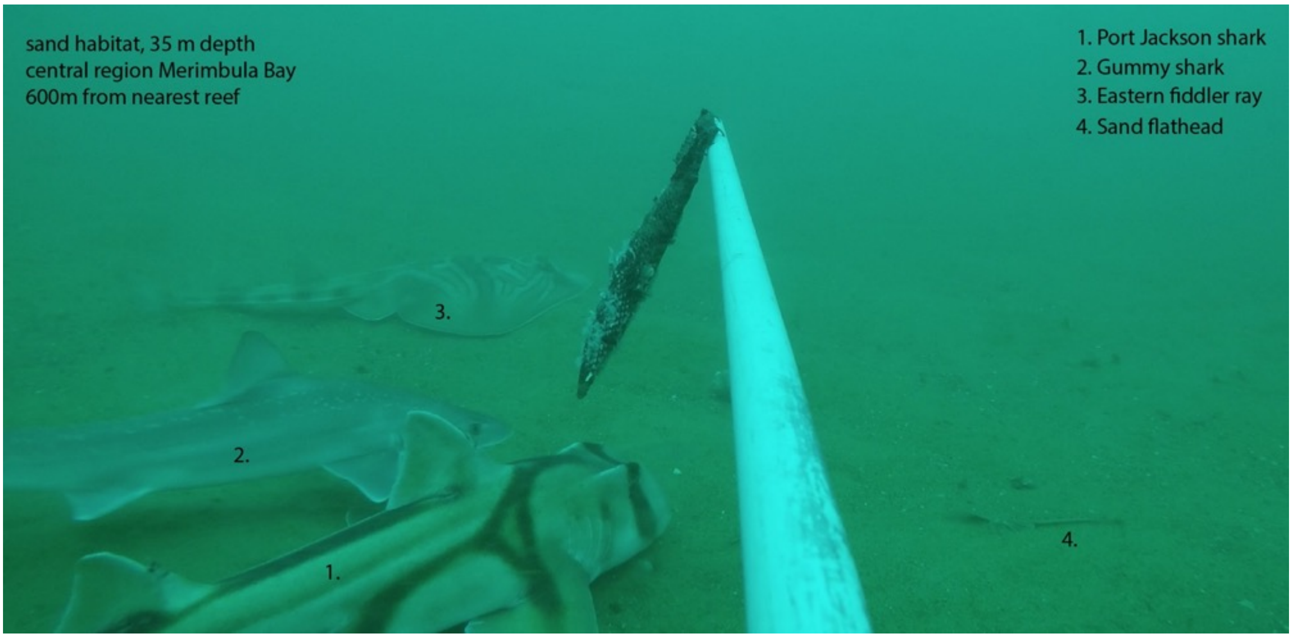



**Figure 5-10** **Mix of sand and reef fish species observed at BRUV28 (sand habitat at 35 m depth)**

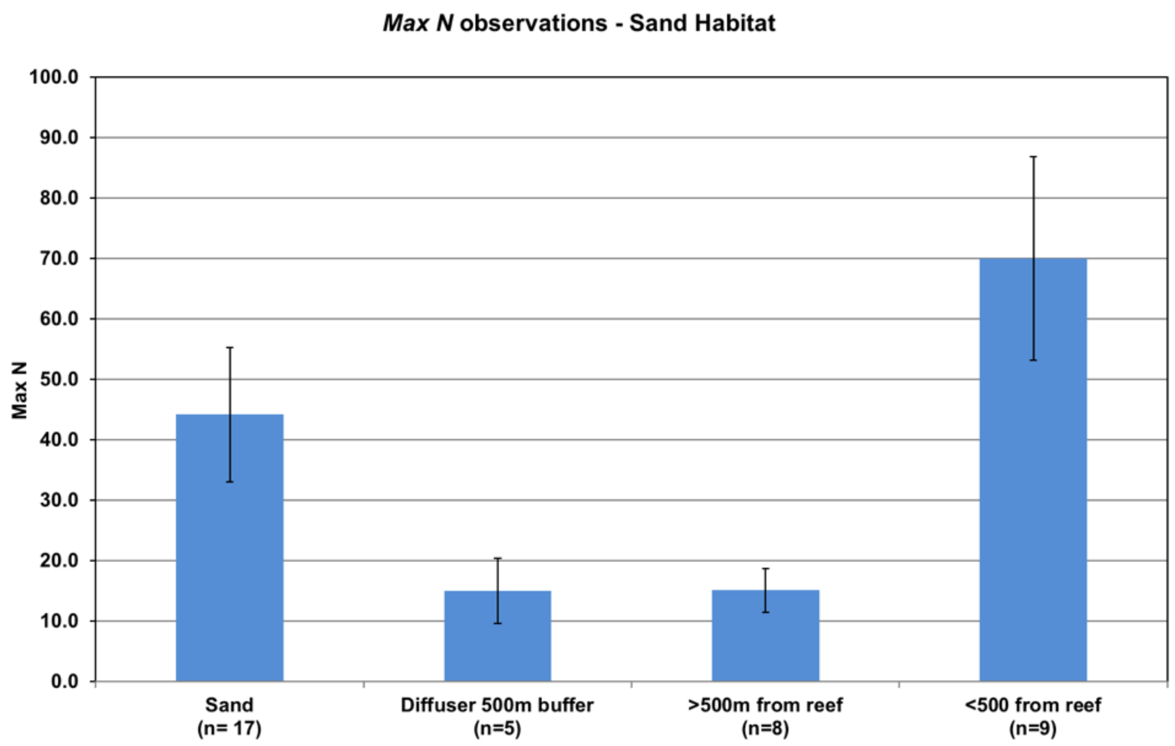



**Figure 5-11 Mean fish abundance (*maxN observations*) for sand habitat within the diffuser 500 m buffer compared to sand habitat more than, or less than 500 m distance from reef areas**

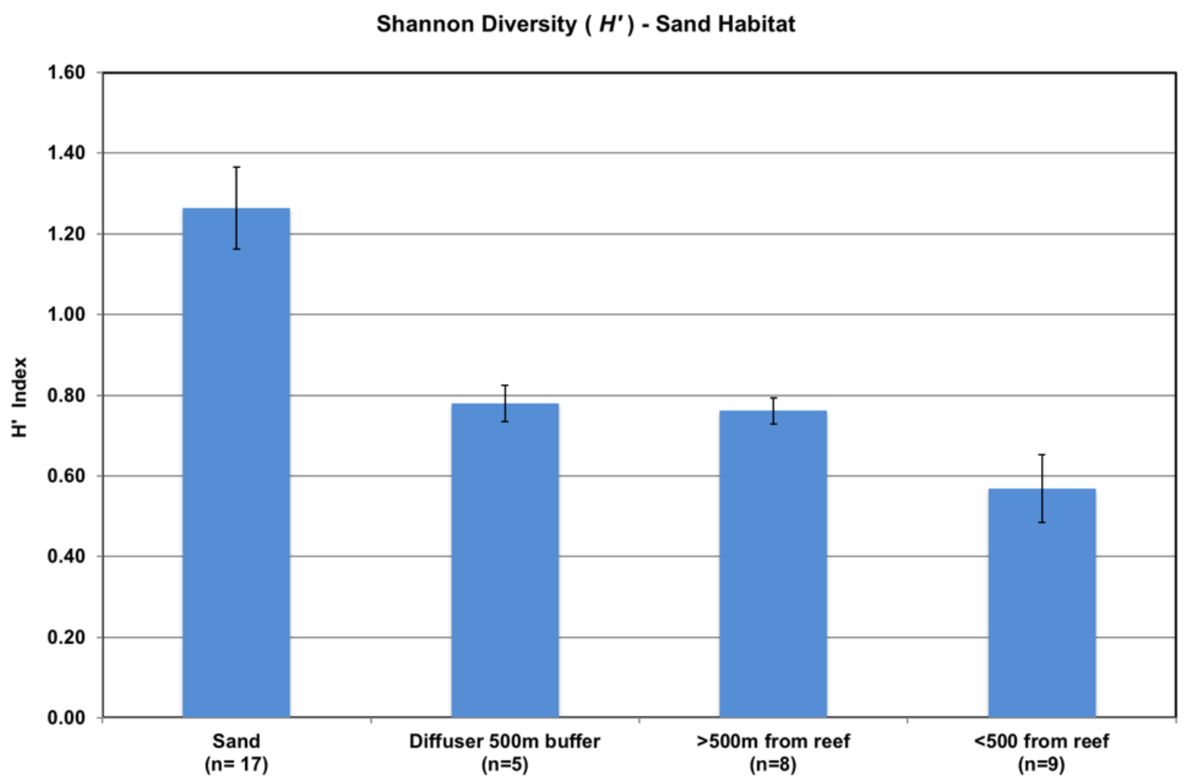



**Figure 5-12 Mean Shannon diversity of fish assemblage on sand habitats for sand habitat within the diffuser 500 m buffer compared to sand habitat more than, or less than 500 m distance from reef areas**

### *5.6.4* Patterns of Diversity

Patterns of fish diversity were explored using software Primer 7.0. The *maxN* raw data (**Appendix C-4**) was square root transformed to down-weight the influence of highly abundant species. Data transformation balances the contributions from common and rarer species in the measure of similarity between two samples. Using the transformed dataset, Bray-Curtis similarities/dissimilarities between samples were calculated and patterns in community structure were evaluated by displaying these results in a two-dimensional ordination using non-metric multidimensional scaling (nMDS), referred to as a MDS plot.

One-way Analysis Of Similarities (ANOSIM) was used to test for differences between groups of samples and examine whether differences were apparent in the fish assemblage based on various *a-priori* defined factors that included location (Haycock Point versus Hunter Reef), depth (shallow to 20 m depth versus deep below 20 m depth) and by habitat type. The output of ANOSIM tests is the *Global R* value that provides an indication of the level of similarity/dissimilarity between groups that may warrant further examination. *R* values typically fall between 0 (no observable difference between groups) and 1 (maximum difference between groups), however, negative values correspond to differences between groups that are greater than those within groups (Clarke and Warwick 2001). Observed *Global R* values of 0.45 or greater were considered to be reliable indicator of differences between groups that would display as separate groups of points in two-dimensional space of an MDS plot.

A summary of ANOSIM test outputs is provided in **Table 5-2.** There was no significant difference in the fish assemblage based on depth or location as indicated by low global R-values. There was a significant difference between the fish assemblage of different habitat types – reef versus macroalgae covered reef versus sand (**Table 5-2**). For the fish assemblage of sand habitats, there was no significant difference between the fish assemblage observed within the 500 m radius buffer of the proposed diffuser compared to other sand areas (Global R = 0.036, p = 0.39).

**Table 5-2 ANOSIM results for tests of differences in fish assemblages between habitat types**

| ANOSIM Test | Global R-value | Significance Level |
|---|---|---|
| Depth | 0.071 | p<0.22 |
| Location | 0.179 | p<0.013 |
| Reef * Sand | 0.668 | p<0.001 ** |
| Reef * Macroalgae | 0.616 | p<0.001 ** |
| Sand * Macroalgae | 0.485 | p<0.003 ** |
| Sand Far from reef * Sand Near to reef | 0.189 | p<0.071 |
| Sand diffuser 500m buffer * Sand Far from reef | 0.036 | p<0.39 |

Note
** significant difference detected at significance level of 0.1% or 0.3%

An MDS ordination of the fish assemblage shows samples generally forming distinct groups based on habitat type (as substrate) at 40% similarity (**Figure 5-13)**. The routine SIMPER (in PRIMER) statistical software was used to identify the most dominant taxa driving the pattern in the MDS plot.

- For reef habitats (including barrens, boulder fields, broken reef), seven (7) fish species contributed to 70% of the similarity between samples including (in order of % contribution): silver sweep (21%), mado (12%), yellow moray eel (11%), maori wrasse (10%), blue morwong (8%), snapper (6%), and butterfly perch (6%)
- For macroalgae habitats (that were typically dominated by a canopy of *Ecklonia*), three (3) fish species contributed to 70% of similarity between samples including (in order of % contribution): mado (50%), yellowtail horse mackerel (12%) and herring cale (10%).

- For sand habitats, four (4) fish species contributed to 70% of similarity between samples including (in order of % contribution): blue-spotted flathead (27%), eastern school whiting (23%), yellowtail horse mackerel (15%) and eastern fiddler ray (14%).

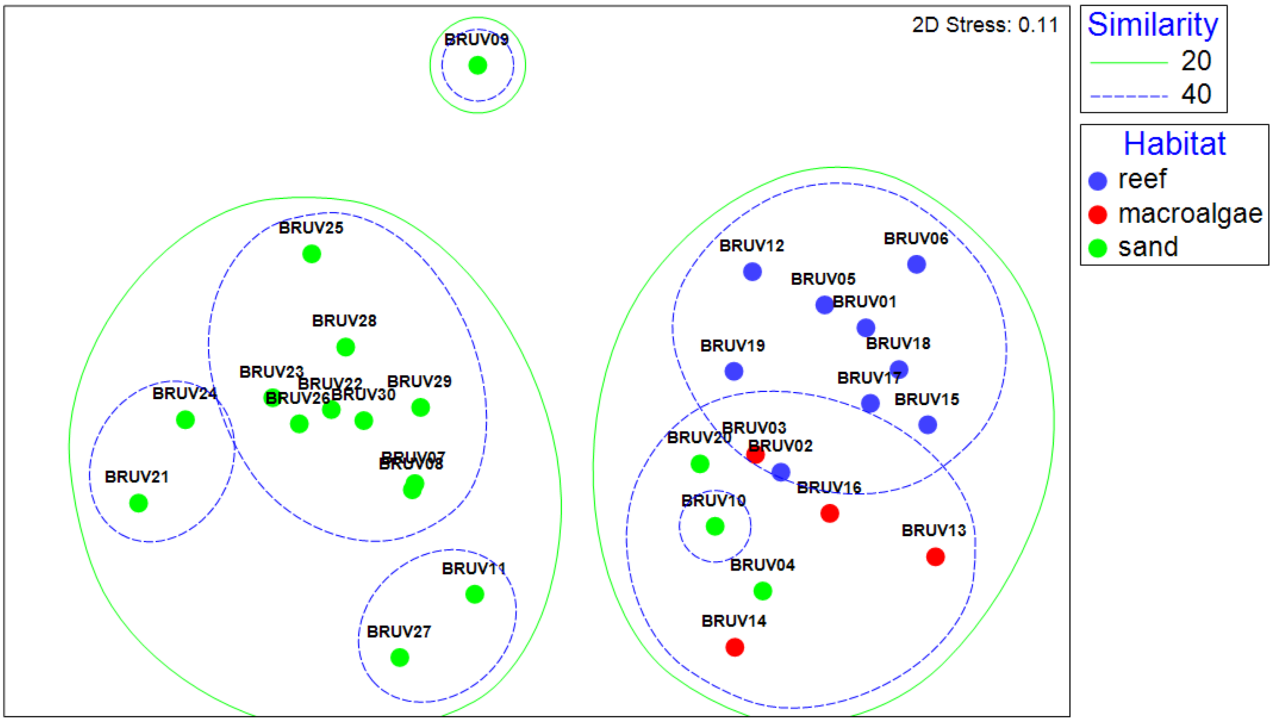



**Figure 5-13 MDS plot of Merimbula Bay fish assemblage by reef, macroalgae and sand habitats**

### 5.6.5 Threatened and Protected Fish Species

No threatened or protected fish species were recorded in the study area using BRUVs, underwater tow video or by diver observation.

### 5.6.6 Other fish species observed

An additional ten (10) fish species were observed using methods other than BRUV. These species were recorded either by tow video during broadscale surveys of the seabed or by diver observation. Species included Australian salmon (*Arripis trutta*), banded morwong (*Cheilodactylus spectabilis*), magpie perch (*Cheilodactylus nigripes*), smallscale bullseye (*Pempheris compressa*), bigscale bullseye (*Pempheris multiradiata*), longsnout boarfish (*Pentaceropsis recurvirostris*), banded seaperch (*Hypoplectrodes nigroruber*), threebar porcupine fish (*Dicotylichthys punctulatus*), black reef leatherjacket (*Eubalichthys bucephalus*) and southern sawshark (*Pristiophorus nudipinnis*). In addition to these fish, octopus, calamari squid and cuttlefish were also observed.

Images of each of the species recorded using other methods are provided in **Appendix C-6**.

## 5.7 Key Findings

Key findings from the assessment of fish assemblages within the study include:

- A total of 73 fish species were observed in the study area with 63 species recorded in BRUV surveys and an additional ten (10) species recorded by other methods. Of the 73 species, 56 were bony fishes (Actinopterygii) and 17 were rays or sharks (Elasmonbranchii).
- The three most species-rich groups on reef habitat were the Monacanthidae (leatherjackets, 8 spp.), Labridae (wrasses, 5 spp.) and Serranidae (groupers and sea perches, 4spp.), while for sand habitat

the most species-rich groups were Urolophidae (stingarees, 4 spp), Rhinobatidae (guitarfishes, 2 spp) and Platycephalidae (flathead, 2 spp).

- The three most abundant species observed were Australian mado (*Atypichthys strigatus),* yellowtail horse mackerel (*Trachurus novaezelandiae*) and silver sweep (*Scorpis lineolata*). Yellowtail horse mackerel was most frequently observed over sand habitats while silver sweep was most often observed over reef habitats.
- The most abundant fish taxa observed over all samples were Australian mado (*Atypichthys strigatus, maxN=467),* yellowtail horse mackerel (*Trachurus novaezelandiae, maxN=293*), silver sweep (*Scorpis lineolate, maxN=116)*), silver trevally (*Pseudocaranx georgianus*, maxN=147) and eastern sand whiting (*Sillago flindersi, maxN=102*). Mado and sweep were most abundant over reef, yellow tail horse mackeral over sand areas close to reef, with trevally and whiting most abundant over sand areas.
- The fish assemblage of reef habitats was significantly different to macroalgae-covered reef and sand habitats. Reef habitat was found to support an overall higher diversity of species that were more evenly represented within that habitat compared to macroalgae-covered reef and sand that had lower species diversity and were dominated by high abundances of fewer species.
- A total of 40 fish species were recorded over sand habitat (inclusive of sites far from and near to reef areas) with an average of seven species observed per sample site. Higher levels of diversity and abundance were observed on sand habitats nearer to reef areas.
- For the diffuser site and indicative of the 25 m mixing zone (as indicated by BRUV21), only three species were observed and at very low abundances. These included the Eastern fortesque (*Centropogon australis*), flounder (*Pseudorhombus* sp.) and bluespotted flathead (*Platycephalus caeruleopunctatus*). Together with BRUV24, on sand 500 m to the south, these samples showed the lowest diversity and abundance in the study. Further sampling would need to be undertaken of the diffuser site to provide more confidence around describing the level of fish diversity and abundance typical at that location.
- All of the species observed during the study are common to south-eastern NSW. Many of the species observed are popular with recreational anglers and targeted by commercial fishermen.
- Five (5) fish taxa listed as threatened or protected under the FM Act and or EPBC Act are either reported from or are predicted to occur in habitats within Merimbula Bay. These include the grey nurse shark, black cod, great white shark, southern bluefin tuna and members of the syngathiformes (seahorses, pipefishes, pipehorses, sea moths). None of these species were observed in the study area during field surveys using the methods employed. The Project is unlikely to have an adverse effect on the aforementioned threatened fish species.

## 5.9 Construction Phase Impacts

Potential impacts to the fish assemblage of Merimbula Bay from construction phase activities include the following:

- Noise impact from construction vessels and equipment
- Accidental spill from construction vessels
- Disturbance and loss of Type 3 sand habitat establishing the pipeline and diffuser infrastructure
- Introduction of an invasive marine pest via construction vessels and equipment

### 5.9.1 Construction noise

Noise disturbance from the Project include vessel and equipment noise during construction activities. These include vessel movements, establishing the pipeline and diffuser infrastructure on the seabed, and anchoring the pipeline with protective concrete mattress or rock armour. Noise from these activities is considered steady-state continuous noise as opposed to impulsive noise such as from pile driving that are detrimental to fish.

Research as shown that fish, and in particular fish with swim bladders, have sensitivities to noise disturbance although the physiological effects from noise are not yet well understood (Hawkins and Popper, 2017; Popper and Hawkins, 2019). Ambient noise levels in Merimbula Bay comprise recreational, charter and commercial vessel noise. It is anticipated that the noise levels from project construction activities would not be significantly higher than current ambient noise levels. Noise effects would be short-term, localised and fish within the vicinity of construction activities would avoid or move away from the area as required. The risk of noise disturbance to fish assemblage of Merimbula Bay is considered minimal (**Section 14 – Impact Assessment**).

### 5.9.2 Accidental spill

There is the potential for hazardous substances (*ie.* fuels, oils and other construction vessel related fluids) to accidently enter the water through spills or leaks from construction vessels and/or equipment. Water pollution resulting from vessel accidental spill would typically impact the water surface and have limited direct effect on fish assemblage. The potential impacts of water pollution on marine fauna can be harmful and is considered a low risk. This risk can be reduced by implementing a range of control measures to protect water quality during construction.

### 5.9.3 Disturbance and loss of Type 3 soft sediment habitat

As discussed in **Section 2.5.1**, construction of the pipeline would result in the loss of 4,320 $m^2$ of Type 3 sand habitat, considered minimally sensitive with regard to fish habitat. Overall this represents a 0.04% loss of Type 3 sand habitat mapped within the study area. The scale of the sand habitat loss (as a result of the Project) is minor and is unlikely to have a negative effect on the fish assemblage that relies on sand habitat within Merimbula Bay, in terms of their diversity and abundance.

### 5.9.4 Introduction of an invasive marine pest (IMP) via construction vessels and equipment

Introduction or translocation of an IMP to Merimbula Bay from Twofold Bay during construction phase activities is considered a medium risk. Three IMPs reported from the Port of Eden, 30 km to the south (Pollard *et al*. 2003) that are not yet reported from Merimbula Bay include the dinoflagellate *Alexandrium catenella*, European fan worm (*Sabella spallanzanii*) and the New Zealand Screwshell (*Maoricolpus roseus*). The latter is known to occur on the continental shelf off Merimbula but is not known to be present within the study area of the embayment. Of these IMPs, the European fan worm with its ability to establish in high densities and potentially alter the community composition of soft sediment habitat could indirectly affect the fish assemblage associated with that habitat through potential change in food resource availability. However, very little work has been done on the impacts of the European fan worm and its effects at an ecosystem level (DPI, 2020).

It is expected that construction vessels would adopt standard environmental management practices and controls as recommended by the *National Marine Pest Plan 2018-2023* to mitigate the risk of IMPs such as European fan worm during construction phase (Refer **Section 15 – Environmental Management**).

## 5.10 Operational Phase Impacts

Potential impacts from the operational phase of the Project considers how the discharge of treated wastewater at the diffuser and the pipeline infrastructure may affect the fish assemblage. Fish may be directly affected through exposure to treated wastewater, or indirectly affected by changes to the ecology of the seabed and water column and creation of new habitat by the pipeline infrastructure.

Dispersion modelling (AECOM, 2020) of the treated wastewater discharge indicates that under most conditions and majority of the time, water quality objectives would be met within a 25 m radius mixing zone. Under a worse-case scenario such as wet weather flow that could coincide with stagnant or low current conditions, the distance required for the dilute wastewater to meet all MWQOs would extend to within 200 m radius buffer from the diffuser.

Exposure to treated wastewater and potential ecological changes to the seabed and water column within the modelled 25 m mixing zone include:

- Habitat disturbance due to altered sediment chemistry from settlement of suspended sediments and organic particulate material
- Exposure to reduced salinity of water column from freshwater discharge
- Exposure to discharge of nutrients above MWQOs
- Exposure to discharge of potentially bioaccumulative metals

Marine communities occurring over sand habitat within the 25 m mixing zone are likely to experience regular exposure to treated wastewater, with those occurring within a 200 m radius from the diffuser exposed to dilute wastewater a minor proportion of time. Fish that utilise sand habitats are typically mobile, demersal species such as flathead, whiting and rays. Those that transit the 25 m mixing zone would be exposed to treated wastewater intermittently, and unlikely to be affected as the treated wastewater, being less dense than seawater, would rise upwards through the water column away from the seabed. The fish most likely to be exposed to the dilute wastewater plume are pelagic or mid-water species such as the yellowtail horse mackerel, or demersal species that may show high fidelity to substrates and habitat within the 25 m mixing zone. A study by Fetterplace *et al.* (2016) showed that flathead in Jervis Bay can show strong site attachment for periods of at least 60 days (duration of the study) and it is this type of behaviour that would increase the likelihood of exposure to treated wastewater if the individuals occur within the 25 m mixing zone.

### 5.10.1 Habitat disturbance due to altered sediment chemistry in mixing zone

Changes to sediment chemistry may occur from the deposition of particulate organic material discharged in treated wastewater. Over the long-term this can cause enrichment of sediments and localised depletion of oxygen. Such changes can have negative impacts for the infauna assemblage (i.e. reduced diversity and abundance) that may impact fish species indirectly through reduction of food availability.

As wastewater quality is characterised by low levels of suspended solids (median concentration 5 mg/L) the potential for impact to infauna communities within the mixing zone from deposition of particulate organic material is considered a moderate risk and the indirect impact to fish through potential reduction of food availablility is considered minimal given the scale of the impact would be limited to the localised area of the mixing zone. The pathway by which sediments may become enriched is if nutrient levels stimulate excessive phytoplankton growth that will deliver additional particulate organic matter to the benthos. Should changes to sediment chemistry occur from the Project, these would likely to be limited to the near-field mixing zone of 25 m radius from the diffuser with some level of change to the benthic infauna community possible. It is then expected that the magnitude and likelihood of potential change would decrease with increasing distance from the outfall and the ability to detect change beyond the mixing zone, if some change has occurred, becomes less likely.

### 5.10.2 Exposure to reduced salinity of water column from freshwater discharge

The treated wastewater mixing zone would be characterised by reduced salinity that may favour fish that can tolerate a broader range of salinity compared to stenohaline species, those that prefer a narrow range of salinity. Fish found within the mixing zone (i.e. sand habitat) were mobile, demersal species such as flathead, whiting and rays. Those fish that transit the mixing zone would be exposed to dilute wastewater intermittently, and unlikely to be affected as the treated wastewater, being less dense than seawater, would rise upwards through the water column away from the seabed. The fish most likely to be exposed to reduced salinity from the dilute wastewater plume are pelagic or mid-water species such as the yellowtail horse mackerel. However, none of the fish observed in the study are considered stenohaline and the zone of reduced salinity would have minimal risk to the fish assemblage.

### 5.10.3 Exposure to discharge of nutrients above MWQOs

Discharge of elevated levels of dissolved nutrients to the mixing zone may have an indirect positive effect on fish assemblage by stimulating local primary productivity leading to increased food resources. In particular, the discharge of treated wastewater may result in localised increase in planktivorous fish such as Mado and baitfish. However, if fish are attracted to the diffuser structure it raises concerns regarding the potential risk of bioaccumulation of metals.

### 5.10.4 Exposure to discharge of potentially bioaccumulative metals

Metals are a physiological stressor for marine organisms with reported effects in some fish that can include changes to growth and reproduction, and abnormal courtship and aggressive behaviours (McCallum *et al.*, 2019). Potential impact of metals to fish would be dependent of levels of exposure and some fish may be more tolerant than others. The pathway by which metals may impact fish is if metals bound to particulate material settles to the sediment, and sediment metal concentrations accumulate and increase over time. In this scenario there is potential for uptake of metals by infauna (i.e. polychaete worms, crustacea and molluscs) and trophic transfer to and bioaccumulation of metals in resident fishes that may feed upon them.

Review of wastewater quality in the *Water Quality Technical Report* (Elgin, 2021) showed that metals detected in treated wastewater with potential to bioaccumulate in biota include cadmium, total chromium, cobalt, mercury, selenium, silver and zinc. Cobalt, selenium and zinc were detected most frequently (at least 90% of samples), whilst cadmium, total chromium and mercury were detected infrequently (4-7% of samples). Dispersion modelling of wastewater discharge (AECOM, 2020) indicates that metals would be rapidly diluted and meet MWQOs within a 5 – 25 m mixing zone around the diffuser. Furthermore, the addition of tertiary filtration if adopted as part of the STP upgrade would enhance the removal of metals from the wastewater stream resulting in lower metal concentrations and the mixing zone required for metals to meet MWQOs would be expected to be further reduced. However, it is noted that tertiary filtration would be expected to only marginally decrease and already low risk and including tertiary filtration for the purposes of enhanced metal removal alone would be unjustified (Elgin, 2021). Nonetheless, if metal concentrations of sediments increase, it is expected that this would be limited to the relatively small area of the mixing zone and the risk of exposure to fish from bioaccumulative metals, confined to those fish showing high attraction and residency within the mixing zone.

Fish that utilise sand habitats in the mixing zone were found to be demersal species such as flathead, whiting and rays. Those fish that transit the 25 m mixing zone would be exposed to treated wastewater intermittently, such that lethal or sublethal effects are unlikely. However, it is not known how fish would respond to the Project. It is possible that some fish species may be attracted to the mixing zone due to localised increases in food availability and structure provided by the pipeline and diffuser. Fish considered at most risk of exposure to metals through trophic transfer are demersal bottom feeding species and those that have tendency to show high site fidelity. A study by Fetterplace *et al.* (2016) shows that flathead in Jervis Bay can show strong site attachment for periods of at least 60 days (duration of the study) and it is this type of fish behaviour that would increase the likelihood of exposure to metals if occurring within the 25 m mixing zone. Under this scenario the risk of trophic transfer of metals to fish foraging upon prey within the mixing zone would be higher, if indeed metal concentrations of sediments increase over time and there is an uptake of those metals by infauna. The fish assemblage associated with sand habitat at the diffuser location and within a 200 m radius buffer was found to be characterised by low diversity and low abundances. While there may be some effect to fish on sand habitats from exposure to metals, the effect may be limited to a few fish species that naturally occur at low abundances within the proposed mixing zone. Thus, the overall scale of the effect would be considered small and impact to fish assemblage of sand habitats minor.

Futher discussion of the potential impacts to fish from the discharge of metals in wastewater is provided in **Section 6 – Bioaccumulation Risk to Fish and Shellfish**.

#### 5.10.5 Creation of Type 2 rocky habitat

Construction of the pipeline infrastructure with concrete mattress and or rock armour protection along its length constitutes a change from sandy seabed habitat to hard substrate habitat, effectively resulting in the creation of an artificial reef. Any available hard substrate placed in the marine environment provides habitat opportunity in the short-term for a wide range of colonising sessile invertebrates such as ascidians, bryozoans, sponges, barnacles, oysters and mussels.

The pipeline and diffuser are also likely to be colonised by various macroalgae. In effect, by laying the pipeline on the seabed rather than trenching and burial, the Project is creating an artificial reef that after some period of colonisation by various invertebrates and algae would be considered Type 2 fish habitat. Over the long-term, or for some periods, the pipeline may become buried by sand. Intermittent sand burial and sand scour of hard substrates is a naturally occurring process in the marine environment that can result in an overall net increase in species diversity due to the intermediate disturbance that provides both early and late successional species an opportunity to establish.

Construction of the pipeline and diffuser infrastructure would result in loss of Type 3 fish habitat but over the long-term result in a gain of Type 2 fish habitat, the latter recognised as being more valuable in terms of fish habitat. Therefore, the Project may result in a net positive effect on the fish assemblage with increased diversity and abundance in the central region of Merimbula Bay. The potential attraction of fish to the pipeline structure may also result in improved recreational fishing opportunities within the vicinity of the pipeline.

#### 5.10.6 Potential impact to fish species important to recreational and commercial fishing

Potential impact of the Project to fish species important to local recreational and commercial fishing including the Australian salmon, sardine, mullet and snapper considered the species life history, migratory patterns and the scale of predicted water quality impact to a 25 m mixing zone over sand habitat.

**Australian Salmon**

Given the highly migratory behaviour of the species, salmon populations moving through Merimbula Bay are most likely to transit along the inshore beach zone at shallow depths more than 1,000 m away from the from the mixing zone. For the minor proportion of the overall population that do transit directly through the mixing zone, they would do so for short periods such that exposure to treated wastewater discharge would have negligible effect on the salmon population and the local ocean haul fishery that targets the species.

**Sardine**

Similar to Australian salmon, sardine are a highly migratory pelagic fish and move through waters of Merimbula Bay. Exposure to dilute treated wastewater may occur intermittently as they transit the mixing zone but overall is considering unlikely to have a negative effect on the individuals or the regional population. Studies of NSW sardine populations have shown a positive relationship to ocean upwellings with sardine growth rates higher in areas downstream of upwellings (Uehara *et al*., 2005). This is attributed to the increased primary productivity of these areas triggered by the higher levels of nutrients provided by the upwelling events. Localised increase in productivity within the vicinity of the mixing zone due to release of dissolved nutrients may have a positive effect for sardines, which in turn may be beneficial to predators of sardine such as larger pelagic fish, seabirds and marine mammals.

**Mullet**

Mullet migrating through Merimbula Bay are most likely to transit along the shallow waters of the inshore beach zone distant from the mixing zone. For the minor proportion of the overall population that do transit directly through the mixing zone, they would do so for short periods such that exposure to treated wastewater discharge would have negligible effect on the mullet population and the local ocean haul fishery that targets the species.

**Snapper**

As the Project is situated on the sandy seabed with the nearest rocky reef habitat preferred by snapper approximately 1,400 m to the south-east, and the wastewater mixing zone typically a 25 m radius majority of time, it is considered unlikely that snapper and the recreational fishing opportunities the species provides would be affected. Snapper may be attracted to the pipeline and diffuser infrastructure and in doing so, the Project may have a positive effect on local snapper population.

### 5.10.7 Conclusion

In discussing the potential impact of the Project to fish assemblage and recreational and commercial fishing, it should be noted that treated sewage wastewater has been discharged to a beach-face outfall at Merimbula Bay since 1971. The Project also includes proposed upgrades to the STP that would result in improved wastewater quality including reduction in metal concentrations via sand filtration. Disposal at the proposed outfall would provide improved dispersion of the treated wastewater compared to the current beach-face outfall.

Overall, it is unlikely that the fish assemblage of Merimbula Bay and those occurring at the Merimbula OAR would be adversely affected by the Project. Based on an evaluation of potential risk factors above, the threat to commercial and recreational fishing within Merimbula Bay from the Project is considered low, including taking into account the potential risk of bioaccumulation assessed in **Section 6 – Bioaccumulation Risk to Fish and Shellfish**.

It is expected that water quality and sediment quality monitoring would be required as part of operational phase environmental management (**Section 15 – Environmental Management).** A proposed water quality monitoring program is outlined in the *Water Quality Technical Report* (Elgin, 2021). Monitoring would be recommended for the mixing zone and at a range of sites situated away from the discharge point such that predictions around the modelled treated wastewater mixing zone extent and dispersion trajectory could be validated including assessing whether sediment metal concentrations change and if so, whether a risk of potential bioaccumulation to fish within the mixing zone exists.

runoff from local banana plantations, and that concentrations of metals were generally comparable to other locations with exception of lead. CEE concluded that the impacts detected in the fish were due to regional conditions and not related to discharge of treated wastewater (CEE 2000).

For the ocean release monitoring at Penguin Head, The Ecology Lab (2005) tested for bioaccumulation in cunjevoi *Pyura stolonifera*, with analysis for metals and organochlorine pesticides. No organochlorines were detected in the *Pyura* collected from Penguin Head. For metals, it was found that there was no difference in concentrations of arsenic, chromium, lead, nickel, selenium and zinc between Penguin Head and control locations. Whilst concentrations of lead, nickel, selenium and zinc were found to be more variable than other metals, it was concluded this was unrelated to the release and indicative of temporal variation in background levels (The Ecology Lab, 2005).

The ocean release monitoring program for the Milton-Ulladulla Sewerage Augmentation included monitoring of wastewater and marine waters, and survey of intertidal and subtidal habitats, fish and abalone (The Ecology Lab 2008). The program included a desktop review of studies of contaminants in fish and invertebrates at the large Sydney Malabar outfall within the context of the much smaller and regional Milton-Ulladulla release.

## 6.3 Stage 1 - Preliminary Assessment

A preliminary assessment of the bioaccumulation risk to fish and shellfish at Merimbula Bay was undertaken in Stage 1 marine ecology investigations (Elgin, 2018). The assessment was based on desktop review of wastewater data from the existing STP discharge between October 2014 to May 2017. Documentation for the Merimbula STP was also reviewed, including its EPA licence, BVSC management plans and trade waste policies and procedures. The review also considered guidance for bioaccumulation provided in the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* (ANZECC 2000), and also drew upon comparative outfall studies that considered the risk of contaminants in recreational and commercially important fish species.

The review of bioaccumulative contaminants included the following:

- Characteristics of the wastewater as a potential source of bioaccumulative contaminants, including existing wastewater quality under the current STP and ambient background water quality in Merimbula Bay, off Haycock Point and south to Quondolo;
- The exposure pathway for the wastewater stream to enter Merimbula Bay marine waters, both under the existing beach face outfall and proposed ocean outfall;
- Fish and shellfish receptors identified in the marine receiving waters of Merimbula Bay, and the potential for exposure to bioaccumulative contaminants if present in the wastewater stream.
- Existing and proposed future risk controls for bioaccumulative contaminants entering the STP

Key references for the evaluation included:

- NSW Water Quality and River Flow Objectives, Towamba and Genoa River (DECC, 2006).
- Marine Water Quality Objectives for NSW Ocean Waters – South Coast (DECC, 2005); and
- Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC, 2000).

The Marine Water Quality Objectives for NSW Ocean Waters – South Coast (DECC, 2005) includes water quality objectives to protect values and uses of south coast marine waters, including in and around Merimbula Bay. For protection of aquatic ecosystem health from bioaccumulative contaminants, the water quality objective is "*bioaccumulation of contaminants – no change from natural conditions*" (DECC, 2005). Background or 'natural' water quality conditions in Merimbula Bay, off Haycock Point and south off Quondolo Point can be assessed from water quality data collected by Elgin Associates in 2014-2017 (Elgin, 2017).

The NSW Water Quality and River Flow Objectives, Towamba and Genoa River (DECC 2006) include water quality objectives for estuaries in the catchment that defer to the toxicant trigger levels in ANZECC (2000). The toxicant trigger levels have different levels of species protection according to the ecosystem being assessed, with three systems defined in ANZECC (2000) that include:

- High conservation/ecological value systems, where 99% species protection is typically applied;
- Slightly to moderately disturbed systems, where 95% species protection for non-bioaccumulative contaminants is typically applied. Because the ANZECC (2000) toxicant guidelines do not take into account bioaccumulation, ANZECC (2000) recommend that the 99% species protection level be adopted for bioaccumulative contaminants for these types of systems; and
- Highly modified systems, where a lower level of species protection (such as 80% or 90%) may be applied.

DECC (2006) states that a '*high level protection of aquatic ecosystems applies to waters in and immediately upstream of national parks, nature reserves, state forests, drinking water catchments and high-conservation-value areas. This reflects their largely unmodified aquatic ecosystems'*. For the marine waters of Merimbula Bay, the following was noted in this context:

- Waters in and around Haycock Point are adjacent to Ben Boyd National Park, and marine ecology survey results presented elsewhere in this report have found high diversity of marine ecological communities including fish assemblages at Haycock Point and Hunter Reef. Based on this, ecosystems in this area are considered to have high conservation/ecological value.
- Waters in other parts of Merimbula Bay, such as to the west towards the beach and north towards Long Point are adjacent to developed land uses (i.e. urban and rural), estuary entrances of Merimbula and Pambula Lakes, and closer to the existing beach face outfall. Based on this setting, these waters are considered to most closely represent a slightly to moderately disturbed system.

For bioaccumulative contaminants, ANZECC (2000) guidance finds that 99% species protection levels would be applicable for both types of systems above. This is because the 99% levels are appropriate for a high conservation/ecological value system, and also appropriate for bioaccumulative contaminants in a slightly-moderately disturbed system. Therefore, the trigger levels adopted for Merimbula Bay marine waters for bioaccumulative contaminants should be the 99% species protection trigger values.

The limitation that the toxicant trigger values do not take into account bioaccumulation is acknowledged by ANZECC (2000), with this limitation often addressed by undertaking direct measurement of contaminant levels in biota tissue to evaluate this risk. This is reflected in impact studies of outfalls elsewhere, including major city and regional outfalls where direct measurement of contaminant levels in aquatic biota is often undertaken. Recent examples of this at regional outfall locations is included in **Section 6.2**.

### 6.3.1 Identifying Potential Bioaccumulative Contaminants

The potential impacts from bioaccumulative contaminants in wastewater is related to both their toxicity and ability to uptake into marine biota such as fish and shellfish and enter the marine food web, which can include angling species caught by fishers for human consumption.

Bioaccumulative contaminants include inorganic and organic compounds, with the potential for bioaccumulation often increasing as water solubility of a chemical decreases. ANZECC (2000) note that indicative properties of a chemical with potential to bioaccumulate includes an octanol-water partition coefficient ($K_{ow}$) between 3 and 7 (*i.e.* a lower solubility in water), and/or a bioconcentration factor (BCF) greater than 10,000. Based on this guidance, the following compounds were flagged with potential to bioaccumulate:

- Metals mercury and selenium, both with BCFs >10,000. In marine systems, cadmium was also considered to have this potential. Other metals chromium, silver, zinc and cobalt also have some ability to bioaccumulate but at BCFs below 10,000 (ANZECC 2000);

- Organochlorine (OC) pesticides and organophosphorus (OP) pesticides. For example, OC pesticide DDT has a $K_{ow}$ of 6.36 and its metabolite DDE a $K_{ow}$ of 6.06, and OP pesticide chlorpyrifos has a $K_{ow}$ of 4.7 (ANZG 2018).

In the subsequent years since publication of the ANZECC (2000) water quality guidelines, other bioaccumulative contaminants have been identified which include:

- Per and Poly Fluoro Alkylated Substances (PFAS), with the PFAS National Environment Management Plan (NEMP) (HEPA 2020) stating that 'PFAS bioaccumulate in aquatic organisms' with high water solubility and protein binding properties that distinguish it from other organic persistent pollutants.
- Microplastics, with NSW EPA (2016) noting that microplastics can enter wastewater treatment systems (such as microbeads in cosmetic products), and that marine species have been shown to uptake microplastics with the potential to transfer up the food chain.

**Metals**

Metals are present in seawater and sediments from both naturally occurring and anthropogenic sources. Naturally occurring sources include mineral deposits in coastal soils and rocks, fluvial inputs, and air/dust deposition, whilst anthropogenic sources can include coastal urban development (runoff, sewage, industry), agriculture and mining. Metals in ambient waters of Merimbula Bay were monitored between 2014 and 2017 to establish ambient water quality (AWQ) values for the Bay and nearby coastal waters and are considered to represent the net sum of influences on marine waters for this part of the NSW coast. AWQ values are included in **Table 6-1**, below.

Wastewater monitoring data collected by BVSC at Merimbula STP has included a period of monthly monitoring of metals between 2014 and 2017. Metals detected in the wastewater included cadmium, total chromium, cobalt, mercury and zinc, with cobalt, selenium and zinc detected most frequently (at least 90% of samples), whilst cadmium, total chromium and mercury were detected infrequently (4-7% of samples).

Screening of metals concentrations against ANZG (2018) ecosystem protection default guideline values (DGVs) found that $90^{th}$ percentile concentrations of cobalt, selenium and zinc exceeded these values. For potentially bioaccumulative metals, the 99% species protection DGVs were adopted following ANZECC (2000) guidance that the level of protection is stepped up (i.e. from 95%) to account for potential bioaccumulative effects. A summary of these DGV exceedances in context of ambient water quality is provided in **Table 6-1**, below.

Based on the review of wastewater quality dataset, along with available metals data in ambient marine waters of Merimbula Bay and north and south of the Bay, further evaluation of metals was triggered which included:

- Predicted dispersion of metals in wastewater to receiving marine waters (Section 6.6); and
- Collection of a baseline dataset for metals in biota in and around Merimbula Bay (Section 6.7).

**Table 6-1 Summary of metals in existing treated wastewater, ambient waters and water quality objectives**

| Parameter | Units | MWQO [1] | Aquatic Ecosystem Protection DGV (95%) [2] | Aquatic Ecosystem Protection DGV (99%) [2] | Ambient Marine WQ (AWQ) [3] | Existing Wastewater Water Quality (ETWWQ) [4] | Dilution Factor to Achieve AWQ or 99% DGV (whichever greater) |
|---|---|---|---|---|---|---|---|
| Cadmium | ug/L | 0.7 | 5.5 [a] | 0.7 [b] | 0.1 | 0.025 | NA - already achieved |
| Chromium (Total) | ug/L | 20 | - | - | 0.25 | 1 | NA - already achieved |
| Chromium (III) | ug/L | 27.4 | 27.4 [a] | 7.7 [b] | - | - | - |
| Cobalt | ug/L | 1 | 1 [a] | 0.005 [b] | **0.025** | **0.5** | 100 |
| Mercury | ug/L | 0.1 | 0.4 [a] | 0.1 [b] | 0.05 | 0.05 | NA - already achieved |
| Selenium (Total) | ug/L | 3 | 3 [c] | 3 [c] | 1 | **7.8** | 8 |
| Silver | ug/L | 1.4 | 1.4 [a] | 0.8 [b] | 0.35 | 0.5 | NA - already achieved |
| Zinc | ug/L | 5 | 15 [b] | 7 [b] | 2.5 | **140.4** | 56 |

Note:

[1] Marine Water Quality Objective (MWQO) trigger values adopted for the Project based on all environmental values - Aquatic Ecosystem Health, Primary and Secondary Contact Recreation, Aquatic Foods (Table 2-2- in *Water Quality Technical Report*, Elgin 2021). In some cases, the adopted MWQO is lower than the Aquatic Ecosystem Health trigger value.

[2] Aquatic Ecosystem Health guideline values (ANZG 2018) relevant to the assessment of marine ecological values, and based on the following:
[a] ANZECC (2000) Toxicant stressor guidelines for marine aquatic ecosystems – 95% protection – Table 3.4.1. Also DGV in ANZG (2018).

[b] ANZECC (2000) Toxicant stressor guidelines for marine aquatic ecosystems – 99% protection – Table 3.4.1. Also DGV in ANZG (2018).

[c] ANZECC (2000) Interim working level, Volume 2.

[3] Ambient marine water quality (from Table 3-3 in *Water Quality Technical Report, Elgin 2021*)

[4] Existing treated wastewater water quality (ETWWQ) used for dispersion modelling - based on 100th percentile EPL discharge limits where applicable, otherwise 90th percentile concentrations adopted as worse-case discharge scenario such as wet-weather flow (Table 5-1- in *Water Quality Technical Report*, Elgin 2021)
Values in bold and shaded indicate existing exceedance of MWQO and or Aquatic Ecosystem Health draft guideline value (DGV)

### OC and OP Pesticides

The risk of OC and OP pesticides being present in STP wastewater has decreased significantly since bans on the use and importation of these pesticides were implemented in the 1980s and 1990s. There remains some residual risk for legacy contamination (such as from illegal discharges) and this is generally managed in clauses in trade waste policies and environmental protection licences, as is the case for Merimbula STP. The BVSC procedure for liquid trade waste has a specific prohibition clause for organochlorines and organophosphorus. EPL 1741 also has specific conditions (O5.1 and O5.2) to prevent the discharge of organochlorine and organophosphorus contaminants into the sewerage system.

These conditions, along with the largely domestic catchment of the Merimbula STP, indicate that the risk of OC and OP pesticides being present in the wastewater is low. Therefore, OC and OP pesticides were not considered further in this study.

**PFAS**

The PFAS NEMP (HEPA 2020) notes that PFAS can enter wastewater treatment plants from sources that can include domestic and industrial discharges. According to the NEMP, as at 2020 further work is to be undertaken in collaboration with the wastewater industry to establish criteria and guidance for PFAS for water authorities and environment regulators. For Merimbula STP, the current licence does not reference PFAS and it is not a parameter in the wastewater monitoring program, although relevant provisions in BVSC trade waste policy and EPL 1741 would apply for PFAS in context of prohibiting substances assessed as unsuitable ischarging into the sewerage system.

### 6.3.2 Microplastics

Environmental impacts from microplastics in marine ecosystems have been increasingly reported in the scientific literature due to their persistence in the environment, their ability to attract toxicants and act as vectors for them, and transfer of microplastics through food webs. Microplastics are small plastic particles with an upper size limit of 5 mm in diameter. A report by NSW EPA (2016) 'Plastic Microbeads in Products and the Environment' noted that microbeads reach and persist in the environment as they are in products designed to be washed or rinsed down the drain and are not captured by most wastewater treatment systems. Once microplastics enter a marine environment, it noted they are too small and widely dispersed for it to be cost-effective to recover. The report also included background information by Sydney Water who confirmed that primary treatment plants can only capture particles to 5 mm diameter and are not effective at removing the majority of microplastics. Tertiary treatment can capture much lower particle sizes, down to 0.001 mm.

The NSW EPA report (2016) noted that a working group has developed responses to the issue, which has included state and federal environment ministers working on a voluntary agreement for industry to phase out microbeads in personal care, cosmetic and cleaning products. This phase out period was to commence following government agreement but no later than 1 July 2018. It noted that steps taken by industry up to 2016 had included voluntary phase outs or commitments not to include microbeads in their products, although this appeared limited by smaller businesses and online retailers. The report also included an overview of actions being undertaken to address the general impacts of plastics on the environment, including plastic recycling initiatives, reduction of plastic shopping bag use and commencement of research into impacts of plastic clothing fibres.

Correspondence with NSW Department of Primary Industries in May 2019 indicated that it is currently working in association with University of Newcastle and University of Sydney to collect and characterise microplastics in the marine environment, including trophic transfer and their potential to act as vectors for other pollutants (Carbery et al., 2018 and Cole *et al*., 2011). The ability of microplastics to act as vectors for other pollutants was discussed by Cole et al (2011), and in the context that highest concentrations of microplastics are often reported in the coastal zone and that ingestion of microplastics has been reported in a range of marine organisms. The uptake of microplastics can also facilitate the trophic transfer of pollutants to biota, such as chemical additives or hydrophobic waterborne pollutants. Carbery et al (2018) discussed how the physical and chemical properties of microplastics can facilitate the sorption of contaminants to the particle surface, serving as a vector of contaminants to organisms following ingestion. A data gap is that bioaccumulation factors for higher trophic organisms and impacts on wider marine food webs is unknown. The factors influencing microplastic ingestion was also discussed, including descriptions of the biological impacts of associated chemical contaminants and evidence for the trophic transfer of microplastics and contaminants within marine food webs.

There is no known data from the Merimbula STP that indicates whether microplastics are present in influent flowing into, or out of the plant and therefore, whether they are likely to pose a risk to the receiving environment. If they are present, a combination of government and community initiatives to minimise the input of microplastics into the sewerage system along with filtration capability in the STP itself are measures that are expected to reduce the risk of microplastics in the treated wastewater and discharge into Merimbula Bay.

#### 6.3.3 Stage 1 Key Findings

The Stage 1 preliminary assessment identified metals in the wastewater stream with potential to bioaccumulate in fish and shellfish resources (including abalone) as receptors to be included in the impact assessment for the Project. Further assessment of the bioaccumulation risk was undertaken during Stage 2 marine monitoring based on the Stage 1 preliminary assessment findings that included:

- The Marine Water Quality Objectives for NSW Ocean Waters – South Coast (DECC, 2005) describes water quality objectives to protect values and uses of south coast marine waters, including in and around Merimbula Bay. For protection of aquatic ecosystem health from bioaccumulative contaminants, the water quality objective is *"bioaccumulation of contaminants – no change from natural conditions"* (DECC, 2005) and is based on the ANZG (2018) water quality guidelines for ecosystem protection.
- Wastewater quality data collected at the STP included nutrients, metals, microbiological and physico-chemical parameters. Of these parameters, metals with potential to bioaccumulate in fish and shellfish included cadmium, total chromium, cobalt, mercury, selenium and zinc. Cobalt, selenium and zinc were detected most frequently in the wastewater (at least 90% of wastewater samples), whilst cadmium, total chromium and mercury were detected infrequently (4-7% of wastewater samples). Zinc, cobalt and selenium were the metals most often reported above the water quality objectives. Other potential bioaccumulative contaminants identified in the desktop review included organochlorine and organophosphorus pesticides and emerging contaminants microplastics and PFAS. There is no wastewater quality data for these contaminants and initial assessment of their potential risks (if included in the scope) was based on desktop review.
- ANZG (2018) water quality guidelines used to evaluate risks from bioaccumulative contaminants have a number of limitations that is acknowledged by ANZG (2018), with a preferable approach to undertake direct measurement of contaminants levels in biota tissue. As concentrations of zinc, cobalt and selenium were reported above these guidelines, it was recommended that direct tissue sampling of shellfish and fish species be undertaken during Stage 2 marine monitoring to assess background levels of metals in biota in the area of investigation and nearby areas to further inform the risk of bioaccumulative contaminants.
- A field survey program was designed to sample mussels, abalone and blue-spotted flathead (*Platycephalus caeruleopunctatus).* Mussels and abalone are present on reefs within Merimbula Bay while blue-spotted flathead is known to be abundant over sand habitats within Merimbula Bay all year round including at the proposed diffuser location. These species represent commercial and or popular recreational catch and were therefore selected as candidate shellfish and fish receptors for direct measurement of contaminant levels in tissue.

### 6.4 Stage 2 – Biota Sampling and Impact Assessment

The Stage 2 objectives were to assess potential impacts and risk posed by bioaccumulative contaminants in the wastewater to fish and shellfish, including the risk to the South Coast water quality objective of *"bioaccumulation of contaminants – no change from natural conditions"* (DECC, 2005). This included obtaining empirical data on background levels of metals in target fish and shellfish under existing conditions.

#### 6.4.1 Scope of Work

The scope of work included the following additional assessment:

- Review wastewater quality data against ambient concentrations of metals in marine receiving waters of Merimbula Bay.

- Review of the preferred option for the STP upgrade and outfall since completion of Stage 1, including outfall location and treated wastewater plume dispersion modelling.
- Field sampling of target fish and shellfish species for analysis of metals in biota. Flathead, blue mussel and black-lip abalone were identified as suitable target species, with sampling at the proposed North-Short outfall location in Merimbula Bay, along with other locations inside and outside the Bay. The biota data were collected to comprise a baseline dataset that would inform on potential risks and also be available for future reference if required.
- Assessment of the biota data in context of species, sampling locations, broader regional NSW data and food health standards where available.

### 6.4.2 Approach and Methodology

**Desktop Review**

Following on from Stage 1, assessment of the risk from bioaccumulative contaminants include the additional desktop review:

- STP licence conditions and policies, existing beach face outfall discharge and the proposed upgrades to the STP and location of proposed outfall 'North-Short' option.
- Ambient water quality in marine receiving waters of Merimbula Bay and dispersion modelling results of the 'North-Short' option.
- Published literature on biota metal concentrations in fish and shellfish in NSW coastal waters, including food standards for metals in fish and shellfish.

Information reviewed included:

- Bega Valley Shire Council Policy 4.07 – Water and Sewerage Services (November 2017);
- Bega Valley Shire Council Procedure 4.06.3 – Liquid Trade Waste (September 2009);
- Environment Protection Licence 1741, issued to Bega Valley Shire Council by NSW EPA (22 May 2020).
- BVSC influent and wastewater data for the STP in spreadsheet file entitled '*Merimbula STP influent and wastewater quality data_kb02112017.xls*.'
- AECOM (2019a). 30% Concept Design Report - Merimbula STP and Deep Ocean Outfall. Prepared for Bega Valley Shire Council, 6 December 2019.
- AECOM (2019c). Merimbula Ocean Outfall – Dispersion Modelling Report. Prepared for Bega Valley Shire Council, 20 December 2019.
- ANZG (2018). Australian & New Zealand for Fresh and Marine Water Quality. 2018 including and as updated from ANZECC (2000).
- Heads of EPAs Australia (2020). PFAS National Environment Management Plan 2.0. January 2020
- NSW EPA (2016). Plastic microbeads in products and the environment. July 2016.
- Papers by:
  - Bebbington *et al.* (1977). *Heavy metals, selenium and arsenic in nine species of Australian commercial fish.* Aust. J. Mar. Freshwater Res., 1977, 28, 277-86.
  - McClintock (2012). *Trace metals in marine food species (Saccosteria commercialis, Metapenaus macleaya, Girella tricuspidata and Platycephalus fuscus) from Lake Illawarra, New South Wales.* Honours Thesis, University of Wollongong, October 2012.

- Birch and Apostolatos (2013). *Use of sedimentary metals to predict metal concentrations in black mussel (Mytilus galloprovincialis) tissue and risk to human health (Sydney estuary, Australia).* Env. Sci. and Pol. Research, 20-8, August 2013.
- Fabris *et al.* (2006). *Trace metal concentrations in edible tissue of snapper, flathead, lobster, and abalone from coastal waters of Victoria, Australia*. Ecotoxicology and Environmental Safety 63 (2006) 286-292.
- McVay *et al.* (2018). *Metal concentrations in waters, sediments and biota of the far south-east coast of New South Wales, Australia, with an emphasis on Sn, Cu and Zn used as marine antifoulant agents.* Environ Geochem Health, 21 November 2018.

### 6.4.3 Fish and Shellfish Sampling and Analysis

Field sampling of target shellfish and fish species included the following:

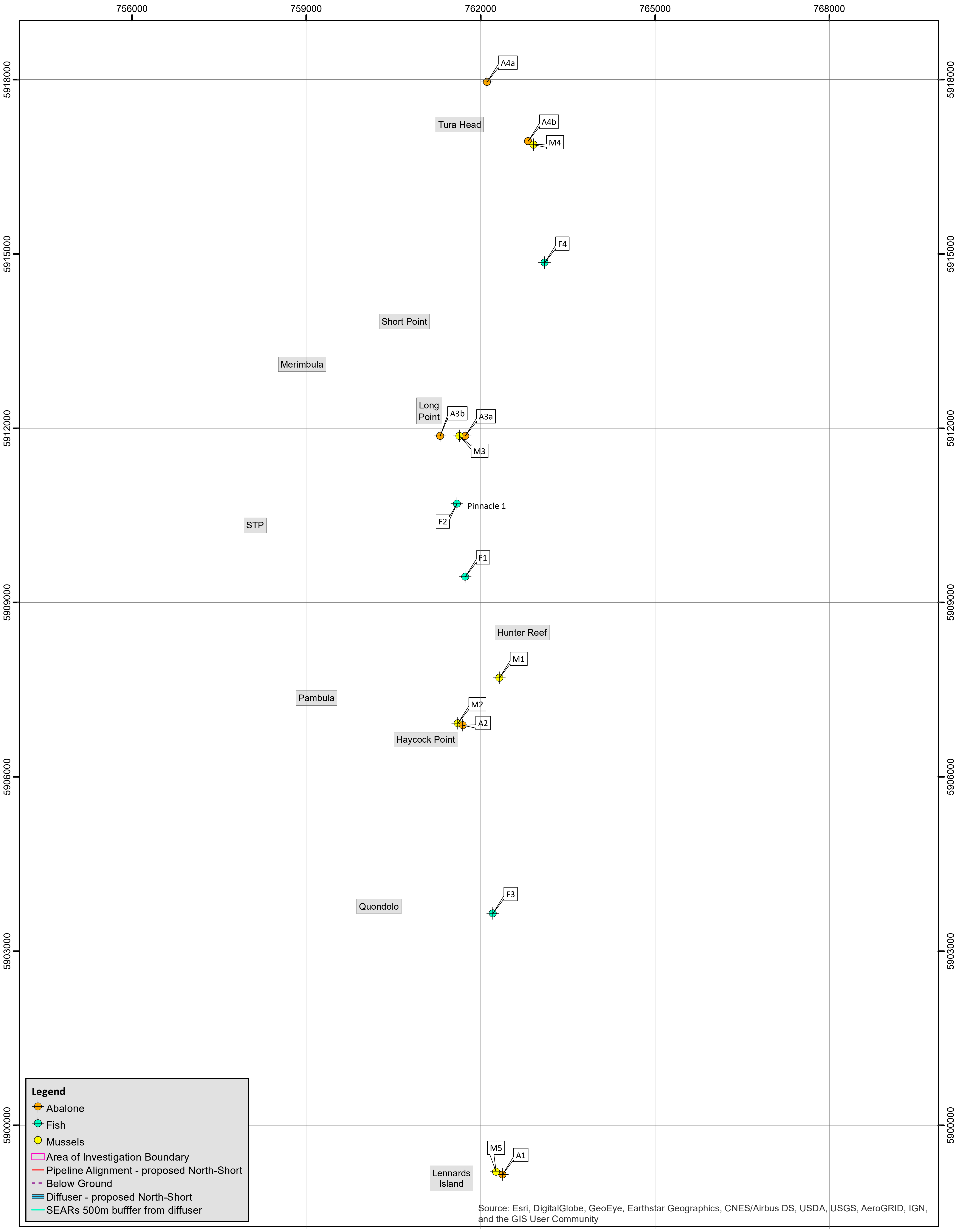
- Collection of blue mussel (*Mytilus galloprovincalis*), black-lip abalone (*Haliotis rubra*) and flathead (*Platycephalus spp.*) from locations within Merimbula Bay and the area of investigation, along with other locations to the north and south of the Bay.
- Analysis of blue mussel, black-lip abalone and flathead biota samples for metals by NATA accredited Australian Laboratory Services (ALS).

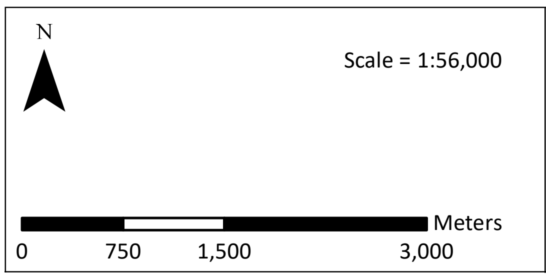
The rationale for the target species included:

- The blue mussel (*Mytilus galloprovincalis*) inhabits reef in the low intertidal to shallow subtidal zone to approximately 12 m depth where it can form extensive beds. Blue mussels were observed at Hunter Reef in 8-10 m depth, on the northern side of Haycock Point in 4-5 m depth and on the southern side of Long Point in 5 m depth. Mussels were collected by hand by diver on SCUBA. Mussels were collected from a total of five locations (three locations within Merimbula Bay and two outside). The mussel collection locations M1-M5 are summarised in **Table 6-2** below and shown on **Figure 6-1**.
- The black-lip abalone (*Haliotis rubra)* is the basis of a commercially important fishery. They occur on shallow rocky reef habitats and are relatively sedentary once settled on rocky reef habitat. The black-lip abalone lives for 20-50 years reaching a size of 150-220 mm shell length. Reproductive maturity is reached at approximately five years of age at which time shell length may be 80-130 mm long. Four legal minimum length (LML) individuals, >117 mm, were collected by hand by diver on SCUBA from a total of four locations. The abalone collection locations A1-A4 are summarised in **Table 6-2** below and shown on **Figure 6-1**.
- Flathead, in particular blue-spotted flathead (*Platycephalus caeruleopunctatus*), observed to be abundant over sand habitats in Merimbula Bay, are a popular angling species and are expected to be present all year round. A recent study by Fetterplace et al. (2016) reported that whilst flathead in Jervis Bay are known for large scale movements, they can also show strong site attachment for periods of at least 60 days (duration of the study). As the outfall diffuser at 'North-Short' is proposed over sand and distant from reef areas, the flathead represented a suitable fish species. Flathead were collected by line fishing methods from a total of four locations (two locations within Merimbula Bay and two outside to the north and south). The fish catch locations F1-F4 are summarised in **Table 6-2** below and shown on **Figure 6-1**.

**Table 6-2 Biota sampling - sample sites and rationale**

| Site Code | Sample Type | Location | Rationale |
|---|---|---|---|
| M1 | Mussels | Hunter Reef | Merimbula Bay south-east of 'North-Short' diffuser location |
| M2 | Mussels | Haycock Point | Southern end of Merimbula Bay |
| M3 | Mussels | Long Point | Northern end of Merimbula Bay |
| M4 | Mussels | Tura Head | Outside of Merimbula Bay, northern reference location |
| M5 | Mussels | Lennards Island | Outside of Merimbula Bay, southern reference location |
| A1 | Abalone | Lennards Island | Outside of Merimbula Bay, southern reference location |
| A2 | Abalone | Haycock Point | Southern end of Merimbula Bay |
| A3a / A3b | Abalone | Long Point | Northern end of Merimbula Bay |
| A4a/ A4b | Abalone | Tura Head | Outside of Merimbula Bay, northern reference location |
| F1 | Fish | Merimbula Bay | Within 500 m radius buffer of 'North-Short' diffuser location, and north of Hunter Reef |
| F2 | Fish | Merimbula Bay | Close to Merimbula Offshore Artificial Reef, 1,000 m north of 'North-Short' diffuser location. |
| F3 | Fish | Quondolo Point | Outside of Merimbula Bay, southern reference location |
| F4 | Fish | Tura Beach | Outside of Merimbula Bay, northern reference location |




Notes:

1. ElginBiota sampling conducted on 24/04/2020 (mussels and abalone), 29/09/2019 & 02/10/2019 (fish).
2. Southern bathymetry data reported by Marine and Earth Sciences in 2017.
3. Northern bathymetry data reported by Southern Divers and Total Hydrographic in 2017.

**Project:** MERIMBULA STP UPGRADE AND OCEAN OUTFALL ENVIRONMENTAL ASSESSMENT

**Client:** AECOM AUSTRALIA

**Title:** BIOTA SAMPLING LOCATIONS

Date: 07 December 2020

Version: 2 | Size: A3

**FIGURE 6-1**

Coordinate System: GDA 1994 MGA Zone 55
Projection: Transverse Mercator
Datum: GDA 1994
False Easting: 500,000.0000
False Northing: 10,000,000.0000
Central Meridian: 147.0000
Scale Factor: 0.9996
Latitude Of Origin: 0.0000
Units: Meter

Document Path: D:\Elgin Associates Dropbox\Elgin_GIS\New_South_Wales\BVSC GIS Data\Merimbula EIS\1. MXD Mapping Files\Stage2_Reporting\Figure_6-1_BIOACCUM_Sites_07Dec2020.mxd

Sample collection was conducted under the Scientific Collection Permit issued to Elgin (P17/0047-1.1) by DPI for the project. Following collection, mussel, abalone and fish samples collected at each site were measured, weighed and photographed, with field sampling sheets and photographs in **Appendix D-1**.

Mussel and abalone were shucked whilst flathead were dissected to sample fillet flesh and livers, with samples despatched in chilled eskies under chain of custody to ALS for the following analysis.

- Mussels – analysis of soft flesh from four selected individual samples from each of the five sampling sites, for total of 20 samples.
- Abalone – analysis of foot flesh from four LML individuals from each of the four sampling sites, for total of 16 samples.
- Flathead – analysis of the edible fish fillet on four selected individual fish from each of the four fish sampling sites, with a total of 16 fillet samples. The liver was also dissected from the four selected fish collected at each site. Due to the anticipated small mass of the liver for analytical requirements, fish livers were composited into a single sample from each site, with a total of four (4) liver composite samples.

The laboratory analysis is summarised in **Table 6-3** below:

**Table 6-3 Summary of biota laboratory analysis**

| Analyte | Tissue | Sample # |
|---|---|---|
| Metals | Mussel flesh | 20 individuals |
| Metals | Abalone | 16 individuals |
| Metals | Flathead fillet flesh | 16 individuals |
| Metals | Flathead liver flesh composite | 4 (based on composite of 4 livers per site) |

The metals suite included cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, silver and zinc.

Laboratory analytical certificates are included in **Appendix D-2**.

## 6.5 Current STP Operation

The operation of Merimbula STP is subject to policies and procedures of BVSC and licencing by NSW EPA which includes provisions relevant to bioaccumulative contaminants. The BVSC policy for water and sewerage services (Policy 4.07) has policy commitments that include:

- Complying with NSW government regulatory requirements, which includes environmental legislation in the *Environmental Planning and Assessment Act 1979* and the Environment Protection Licence for the Merimbula STP; and
- Wastewater of a quality suitable for disposal to the environment.

Trade waste is a potential source of bioaccumulative contaminants into a wastewater system, and the BVSC procedure for liquid trade waste (Procedure 4.06.3) sets out a process for regulating sewage and trade waste inputs into its sewerage system, and includes:

- An objective to protect the environment from the discharge of waste that may have a detrimental effect.
- Prior approval from Council for disposal of liquid trade waste to sewer, noting that disposal without prior approval is an offence under the *Local Government Act 1993*. Council are also not able to grant an approval without concurrent approval from the Director-General of the NSW Office of Water.

- Categorisation of dischargers of liquid trade waste according to the risk profile to the sewerage system, with Categories 1, 2 and 3. Category 1 is considered low risk, whilst Category 2 (low to medium risk commercial activities) and Category 3 (industrial activities and/or large discharge volume) have requirements as a prescribed type of waste which includes pre-treatment and the wastewater well characterised.
- Non-compliance penalty provisions where Council can seek compensation for illegal, prohibited or unapproved liquid trade waste discharged to the sewerage system. This also includes fines under the *Environment Operations Act 1997* for pollution of any waters by a discharger who fails to comply with conditions of approval of liquid trade waste to sewer.
- A set of criteria for approval of liquid waste discharges, which is based on preventative risk management and which includes parameter limits for acceptance and substances prohibited to be discharged into the sewerage system. These include:
    - Acceptance limits for cadmium, selenium, mercury, phenolic compounds (except pentachlorophenol), polyaromatic hydrocarbons, and pesticides in general (except organochlorine and organophosphorus).
    - Specific prohibition of organochlorines, organophosphorus, organic solvents and chromate from cooling towers.
    - Ability to prohibit substances which are assessed as unsuitable for discharge into the sewerage system, which may include bioaccumulative contaminants.
- A formalisation of a discharge of trade waste to sewer in an enforceable trade waste agreement between Council and a discharger. Compliance provisions include monitoring and testing, with medium and high-risk dischargers also requiring due diligence and contingency planning.

The Merimbula STP is licenced under EPL 1741, issued by NSW EPA. The EPL defines pollutant load limits, pollutant concentration limits, and volume discharge limits, including:

- Concentration limits for a list of tabulated pollutants that includes nutrients, oil and grease, biochemical oxygen demand, faecal coliforms, pH and total suspended solids. Limits are not specifically included for bioaccumulative contaminants, but the EPL does include condition L3.3 that it “does not authorise the pollution of waters by any pollutant other than those specified in the table/s”.
- Specific conditions (O5.1 and O5.2) relating to organochlorine and organophosphorus contaminants where it states that “the licencee must not enter into any trade waste agreement to discharge organophosphate pesticides (including chlorpyrifos, diazinon, malathion) into the system and “the licencee must not consent to any discharge of organochlorine pesticides (such as dieldrin, heptachlor and chlordane) into the sewerage system.

## 6.6 STP Upgrade and Wastewater Discharge

The current and proposed operation of the STP (with upgrades) and the wastewater discharge, as discussed in **Section 1 - Introduction** and the *Water Quality Technical rep*ort (Elgin, 2021), was considered in the context of risk from bioaccumulative contaminants to fish and shellfish. Key aspects include:

- Treated wastewater has been discharged to the Merimbula Bay environment from the current beach-face outfall since 1971.
- Current discharge volumes to the beach-face outfall ranges between 280 and 660 ML/year (from 2009-2016 data). This is expected to increase in the future with the Project adopting an upgraded STP capacity of 1,350 ML/year of wastewater.

- BVSC has a long-term strategy to upgrade the reticulated water system that should also result in a reduction of copper and zinc concentrations in the treated wastewater stream.
- Ceasing the beach-face outfall and discharging treated wastewater instead to the ocean outfall would improve wastewater dispersion.

Dispersion modelling of treated waterwater discharge considered both current discharge from the beach-face outfall and the proposed ocean outfall option. For the current beach face outfall:

- Treated wastewater discharged at the beach-face outfall is entrained in the nearshore beach zone and rapidly achieves 1,000 times dilution and gradually disperses to the north and south along the beach zone, and offshore into deeper waters of the central bay region where it dilutes further.
- For the metals listed in **Table 6-1** above, cobalt requires the highest dilution (x100) in order to meet either ambient water quality or the aquatic ecosystem DGV (99% protection). The modelling indicates this is met in the nearshore beach zone a short distance from the beach face outfall discharge point.
- Under the existing conditions, the treated wastewater although highly dilute, typically remains within Merimbula Bay and on occasion may enter Merimbula Lake. The likelihood of the latter scenario is greatest when the EAC and its eddy field is running strongest and close to the coast. Whilst modelling indicates occasional entry into Merimbula Lake, the level of wastewater dilution was modelled between 10,000 to 100,000 times which is unlikely to be detected by laboratory water analysis and would be below estuary WQOs.

For the proposed ocean outfall option:

- The proposed diffuser is located in sandy seabed habitat in approximately 30 m depth. The nearest receptors of subtidal and intertidal reef communities, Merimbula Offshore Artificial Reef (OAR), Pambula and Merimbula Lakes and recreational beaches have the following distances from the proposed diffuser location:
    - Hunter Reef ~1400 m to the south-east.
    - Rocky reef shorelines from Pambula Lake entrance to Haycock Point ~2000 m to the south south-west.
    - Rocky reef shorelines from Merimbula Lake entrance to Long Point ~2300 m to the north.
    - Merimbula Offshore Artificial Reef (OAR) ~1000 m to the north-east.
    - Entrances to Merimbula Lake, Pambula Lake and the between recreational beaches of ~2700 m to 3000 m to the southwest, west and northwest.
- Dispersion modelling of treated wastewater discharge indicates that the treated wastewater is rapidly diluted by factors of at least 1000-10,000 in proximity to the diffuser location under a range of ocean current conditions. This included southward, northward and zero current, in stratified and uniform conditions. For all conditions modelled, the modelling included outputs that a dilution factor of 237 is achieved within 5-25 m of the diffuser, whilst a higher dilution of 2500 is achieved within 200 m of the diffuser, using both median and 90$^{th}$ percentile treated wastewater concentrations. The worse case condition modelled was wet weather flows or wastewater at licence discharge limits coinciding with weak ocean current conditions. The best conditions for dilution was when currents were strongest typically under southward flowing boundary conditions.
- For the metals listed in **Table 6-1** above, the highest dilution (x100) to meet either ambient water quality or the aquatic ecosystem DGV (99% protection) would also be expected to be achieved within a 5-25 m mixing zone around the diffuser.

## 6.7 Fish and Shellfish Biota Sampling

### 6.7.1 Literature Data on Metals in Biota in Marine Waters

A search of the literature was undertaken for data on metals in biota in marine waters of the eastern seaboard, specifically in NSW and Victorian waters as regional areas that include or are nearby to Merimbula. Brief points from each of the papers listed in Section 6.3.1 are included below.

**Bebbington et al (1977)**

Bebbington *et al.* (1977) conducted a survey of metal levels in nine species of fish caught from major fishing ports along the NSW coast, including flathead. Muscle (or flesh) samples from 30 dusky flathead individuals were analysed for metals and reported in parts per million (or mg/kg), with results that included zinc in flathead ranging 2.2-15.0 mg/kg, mean of 6.79 mg/kg.

**McClintock (2012)**

McClintock assessed metal concentrations in marine food species caught from Lake Illawarra on the NSW South Coast, including dusky flathead caught from the estuary entrance and inner estuary. Zinc reported in flathead flesh ranged 6.97-11.83 mg/kg with a median of 8.33 mg/kg, mean of 8.90 mg/kg and standard deviation of 2.00 mg/kg.

**Birch and Apostolatos (2013)**

Birch and Apostolatos (2013) assessed metals in sediments and black mussel in NSW waters in the Sydney estuary, and involved collection of mussels (n=4) at 21 locations in the estuary. Mean zinc concentrations in mussels were reported in µg/g (or mg/kg) and ranged from 6-35 mg/kg. The use of filter feeding bivalves such as mussel was also noted for their extensive use as an indicator for ecosystem condition, including their ability to filter large volumes of seawater and suspended matter and bioaccumulate a range of contaminants. Other factors included their sedentary nature, widespread distribution and ease of sampling.

**Fabris et al (2006)**

Fabris et al (2006) reported the concentrations of heavy metals in edible tissue of commonly fished species, including flathead and abalone in Victorian waters. Results from edible tissue (of flesh) samples from two flathead species and abalone were reported across five zones of the Victorian coast and reported in µg/g (or mg/kg). Zinc was noted as an essential metal with results indicating it was well regulated in all of the examined species. Results for flathead included mean zinc of 3.4 mg/kg and 6.1 mg/kg for the two species with standard deviations of 0.8 mg/kg and 2.1 mg/kg, respectively. Results for abalone included mean zinc of 11.3 mg/kg with standard deviation of 3.0 mg/kg. Screening of the results against the Food Standards Code reported concentrations that were below the Maximum Levels (ML)s and similar or below the Generally Expected Levels (GEL)s.

**McVay et al (2018)**

McVay *et al.* (2018) reported metal concentrations in waters, sediments and three sedentary molluscs in 11 estuaries on the NSW far south coast, including Merimbula estuary and Twofold Bay. The molluscs studied included oyster *Saccostrea glomerata*, rocky intertidal gastropod *Austrocochlea porcata* and sediment dwelling gastropod *Batillaria australis*. Whilst the biota species did not include abalone or mussel, the results still provide useful reference data.

The study referenced background concentrations of metals and had a focus on potential impacts from boating activities and marine antifoulant agents whilst also considering other sources such as coastal urban development (including sewage), agriculture, current and historical mining. Zinc was noted as a potential contaminant from these anthropogenic activities, whilst fertilisers used in agriculture was noted as a potential source of cadmium.

Biota results were presented in boxplots with inference from these boxplots indicating the following:

- For *Saccostrea glomerata*, zinc median at Merimbula was reported at ~2300 mg/kg and at Twofold Bay at ~2500 mg/kg. Cadmium was reported in Merimbula estuary at a median of ~5 mg/kg, with Twofold Bay at ~4 mg/kg.
- For *Austrocochlea porcata*, median zinc concentrations of ~70 mg/kg and median concentrations of cadmium of ~0.5 mg/kg were reported for samples collected in Merimbula. Median concentrations of zinc and cadmium at Twofold bay were higher, with zinc ~90 mg/kg and cadmium ~0.9 mg/kg. The concentration range for all seven estuaries was reported for zinc (42-92 mg/kg) and cadmium (<0.01-0.9 mg/kg).
- For *Batillaria australis,* these molluscs were missing at several of the 11 sites including Merimbula and Twofold Bay. They were present at Bega River estuary to the north, where a median zinc concentration of ~110 mg/kg and median cadmium concentration of ~0.6 mg/kg was reported.

### 6.7.2 Food Standards

Along with data in the literature, the biota results were also considered in context of MLs and (GELs in the Food Standards Code (FSANZ). These were referenced from:

- Standard 1.4.1 - Contaminants and natural toxicants (FSANZ, 2016)
- Schedule 19 - Maximum levels of contaminants and natural toxicants (FSANZ 2017)
- GELs for Metal Contaminants (FSANZ July 2001)

These include:

- For zinc, GELs for fish that include median of 5 mg/kg and 90$^{th}$ percentile of 15 mg/kg. The GEL does not specify species or groups of fish or distinguish between marine or freshwater fish. A GEL for oysters was considered as the most closely related to mussel and abalone, with a median of 130 mg/kg and 90$^{th}$ percentile of 290 mg/kg.
- For cadmium, a ML of 2 mg/kg for molluscs.

### 6.7.3 Fish Biota Sampling Results

The metal analytical results for flathead fish samples are included in laboratory reports in **Appendix D-2.** For the metals analysed, zinc was the only metal that was detected with concentrations above the limit of reporting (LOR) of <5 mg/kg. All other metals (cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium and silver), returned results below the LOR, which ranged between <0.1 mg/kg to <5 mg/kg. The concentrations of zinc in flathead samples by site are summarised in **Table 6-4** below.

The pooled zinc dataset and summary statistics for all four sites between Tura Head and Quondolo Beach, are shown in **Table 6-5**, below with comparison to GELs in the FSANZ Food Standards Code and data from other relevant studies.

**Table 6-4 Flathead fish zinc analytical results (mg/kg wet weight)**

| Location | Flathead tissue (*Platycephalus sp*) | Number of Fish (n), length (cm) and mass (g) | Metal | Min (mg/kg wet weight) | Max (mg/kg wet weight) | Composite Result (mg/kg wet weight) |
|---|---|---|---|---|---|---|
| F1 – Merimbula Bay Proposed North-Short Diffuser Location (within 500m) | Flesh | n=4<br>l: 34-39cm<br>m: 225-324g | Zinc | <5 | 6 | - |
| | Liver (composite) | | Zinc | - | - | 44 |
| F2 – Merimbula Bay OAR | Flesh | n=4<br>l: 27-31cm<br>m: 98-168g | Zinc | 5 | 20 | - |
| | Liver (composite) | | Zinc | - | - | 44 |
| F3 – Quondolo Beach (south of Merimbula Bay) | Flesh | n=4<br>l:31-37cm<br>m:169-311g | Zinc | <5 | 6 | - |
| | Liver (composite) | | Zinc | - | - | 40 |
| F4 – Tura Head (north of Merimbula Bay) | Flesh | n=4<br>l:29-31cm<br>m:154-259g | Zinc | 5 | 17 | - |
| | Liver (composite) | | Zinc | - | - | 36 |

Note: where results reported below LOR, half the LOR was used for calculating the mean.

**Table 6-5 Summary statistics of flathead zinc results (all locations) compared to generally expected levels and previous studies**

| Location | Flathead (*Platycephalus spp*) tissue | Number of Fish (n), length (cm) and mass (g) | Metal | Min (mg/kg wet weight) | Max (mg/kg wet weight) | Mean (mg/kg wet weight) | Median (mg/kg wet weight) | $90^{th}$ Percentile (mg/kg wet weight) | Std Dev (mg/kg wet weight) |
|---|---|---|---|---|---|---|---|---|---|
| All locations (F1, F2, F3, F4) | Flesh | n=16<br>l:27-39cm<br>m:98-324g | Zinc | <5 | 20 | 8.4 | 6.0 | 17.5 | 6.0 |
| *FSANZ (2001) Generally Expected Levels (GEL) for Fish*<br>*n=62, species or freshwater/marine not specified* | | | *Zinc* | *-* | *-* | *-* | *5* | *15* | *-* |
| *Bebbington et al (1977). Flathead metals analysis in fishing ports along NSW coast.*<br>*n=30, length 32-63cm* | | | *Zinc* | *2.2* | *15.0* | *6.79* | *-* | *-* | *-* |
| *McClintock (2012). Flathead metals analysis Lake Illawarra entrance and inner estuary.*<br>*n=6, mass 473-792g* | | | *Zinc* | *6.97* | *11.83* | *8.90* | *8.33* | | *2.00* |
| *Fabris et al (2006). Flathead metals analysis in Victorian Waters*<br>*Two flathead species* | | | *Zinc* | | | *3.4 and 6.1* | | | *0.8 and 2.1* |

Note: where results reported below LOR, half the LOR was used for calculating the mean.

### 6.7.4 Mussel and Abalone Biota Sampling Results

The metal analytical results for the mussel and abalone biota samples are included in laboratory reports in **Appendix D-2.** For the metals analysed, zinc and cadmium were the only metals detected with concentrations above the LOR with the following noted:

- Zinc was reported in at least one individual of abalone and mussel from each sampling site.
- Cadmium was reported in mussels only, and in individuals from the M1 Hunter Reef, M2 Haycock Point, M4 Tura Head and M5 Lennards Island sites.

Other metals (chromium, cobalt, copper, lead, nickel, selenium, mercury and silver) were reported below LOR.

The concentrations of zinc and cadmium in mussel and abalone biota samples by site are summarised in **Table 6-6**, below.

The pooled zinc and cadmium dataset and summary statistics for all sites (five sites for mussels, four sites for abalone) between Tura Head and Lennards Island are shown in **Table 6-7**, below.

**Table 6-6 Abalone and mussel biota zinc and cadmium analytical results**

| Location | Shellfish | Number of individuals (n) mass (g) | Metal | Min (mg/kg wet weight) | Max (mg/kg wet weight) |
|---|---|---|---|---|---|
| M1 – Hunter Reef, Merimbula Bay | Mussel (*Mytilus sp.*) | n=4 x composites of 4 individuals<br>m: 60-73g | Zinc | 11 | 21 |
| | | | Cadmium | <1 | 2 |
| M2/A2 – Haycock Point, Merimbula Bay | Mussel (*Mytilus sp.*) | n=4 x composites of 4 individuals<br>m: 56-72g | Zinc | 9 | 18 |
| | | | Cadmium | <1 | 1 |
| | Abalone (*Haliotis rubra)* | n=4<br>m:103-133g | Zinc | 11 | 14 |
| M3/A3 – Long Point, Merimbula Bay | Mussel (*Mytilus sp.*) | n=4 x composites of 4 individuals<br>m: 62-85g | Zinc | 7 | 32 |
| | Abalone (*Haliotis rubra)* | n=4<br>m:96-118g | Zinc | 12 | 15 |
| M4/A4 – Tura Head, north of Merimbula Bay | Mussel (*Mytilus sp.*) | n=4 x composites of 4 individuals<br>m: 82-95g | Zinc | 5 | 12 |
| | | | Cadmium | <1 | 1 |
| | Abalone (*Haliotis rubra)* | n=4<br>m:112-125g | Zinc | 9 | 17 |
| M5/A1 – Lennards Island, south of Merimbula Bay | Mussel (*Mytilus sp.*) | n=4 x composites of 4 individuals<br>m: 34-50g | Zinc | 13 | 25 |
| | | | Cadmium | <1 | 1 |
| | Abalone (*Haliotis rubra)* | n=4<br>m:112-140g | Zinc | 14 | 17 |

Note: where results reported below LOR, half the LOR was used for calculating the mean.

**Table 6-7 Summary statistics of mussel and abalone biota zinc and cadmium results (all locations) compared to generally expected level and previous studies**

| Location | Shellfish | Number of individuals (n) mass (g) | Metal | Min (mg/kg wet weight) | Max (mg/kg wet weight) | Mean (mg/kg wet weight) | Median (mg/kg wet weight) | 90th Percentile (mg/kg wet weight) | Std Dev (mg/kg wet weight) |
|---|---|---|---|---|---|---|---|---|---|
| All Locations (M1-M5) | Mussel (*Mytilus sp.*) | n=20 x composites of 4 individuals m: 34-95g | Zinc | 5 | 32 | 14.6 | 12.5 | 19.5 | 6.7 |
| | | | Cadmium | <1 | 2 | 0.9 | 0.5 | 2.0 | 0.6 |
| All Locations (A1-A4) | Abalone (*Haliotis rubra*) | n=16 m:96-140g | Zinc | 9 | 17 | 13.0 | 13.5 | 16.0 | 2.4 |
| *FSANZ (2001) Generally Expected Levels (GEL)* | | *Oysters* | *Zinc* | *-* | *-* | *-* | *130* | *290* | *-* |
| *FSANZ (2017) Maximum Level* | | *Molluscs* | *Cadmium* | *-* | *2* | *2* | *2* | *2* | *-* |
| *Fabris et al (2006) metals analysis in Victorian Waters.* | | *Abalone* | *Zinc* | *-* | *-* | *11.3* | *-* | *-* | *3.0* |
| *Birch and Apostolatos (2013) metals analysis in Sydney estuary.* | | *Mussel* | *Zinc* | *-* | *-* | *6-35* | *-* | *-* | *-* |
| *McVay et al (2018) metals analysis in estuaries of South Coast NSW* | | *Molluscs (x3) – Oyster, reef gastropod, sediment gastropod* | *Zinc* | *Merimbula and Twofold Bay medians of ~2300-2500 (oyster), ~70-90 (rocky gastropod), ~110 (sediment gastropod)* | | | | | |
| | | | *Cadmium* | *Merimbula and Twofold Bay medians of ~4-5 (oyster), ~0.5-0.9 (rocky gastropod), ~0.6 (sediment gastropod)* | | | | | |

Note: where results reported below LOR, half the LOR was used for calculating the mean.

### 6.7.5 Discussion of Results

The biota sampling results indicated the following:

- For flathead caught from the four sites, zinc concentrations in flesh varied between the two sites within Merimbula Bay F1 (<5-6 mg/kg) and F2 (<5-20 mg/kg) and was similar to the variation in sites to the north at Tura Head F4 (5-17 mg/kg) and south at Quondolo Beach F3 (<5-6 mg/kg). Zinc in composited flathead livers from each site ranged between 36-44 mg/kg, from Quondolo Beach (36 mg/kg) to Merimbula Bay (44 mg/kg) and Tura Beach (40 mg/kg). The fish with most mass and length (and inferred oldest) were caught from Merimbula Bay site F1 and Quondolo Beach F3, which reported similar zinc in liver at 40-44 mg/kg and <5-6 mg/kg in flesh. Fish from Merimbula Bay F2 and Tura

Beach F4 sites had lower mass and were generally shorter and reported zinc in liver 36-44 mg/kg and a higher range in flesh of 5-20 mg/kg.

- For the pooled flathead dataset for all four sites between Tura Beach, Merimbula Bay and Quondolo Beach, zinc in flathead flesh ranged <5-20 mg/kg with a mean of 8.4 mg/kg, median of 6.0 mg/kg, 90th percentile of 17.5 mg/kg and standard deviation of 6.0 mg/kg. This range and mean was slightly higher when compared to flathead results reported by Bebbington *et al.* (1977) for NSW fishing ports and Fabris *et al.* (2006) for Victorian waters, and slightly lower when compared to flathead mean and median results reported by McClintock (2012) for Lake Illawarra. The median and 90th percentiles were also slightly higher than the FSANZ GELs for fish of 5 mg/kg and 15 mg/kg, respectively.
- For mussels collected from the five sites, zinc concentrations over all sites ranged 5-32 mg/kg, with the greatest range reported at Long Point M3 (7-32 mg/kg) and lowest range at Tura Head M4 (5-12 mg/kg). The other three sites of Haycock Point M2 (9-18 mg/kg), Hunter Reef M1 (11-21 mg/kg) and Lennards Island M5 (13-25 mg/kg) reported zinc within these ranges. Cadmium was reported in mussels collected from Hunter Reef (<1-2 mg/kg), Haycock Point (<1-1 mg/kg), Tura Head (<1-1 mg/kg) and Lennards Island (<1-1 mg/kg) but was not detected in mussels from Long Point.
- For the pooled mussel dataset for all five sites between Tura Beach and Lennards Island:
    - Zinc in mussel flesh ranged 5-32 mg/kg with a mean of 14.6 mg/kg, median of 12.5 mg/kg, 90th percentile of 19.5 mg/kg and standard deviation of 6.7 mg/kg. The median concentration of zinc (12.5 mg/kg) was lower when compared to the median concentrations reported for gastropods (~70-90 mg/kg and ~110 mg/kg) and oysters (~2300-2500 mg/kg) by McVay *et al.* (2018) for Merimbula and Twofold Bay, and at the lower end of the zinc means reported for mussels by Birch and Apostolatos (2013) for the Sydney estuary.
    - Whilst mussel data wasn't available in McVay *et al.* (2018), comparison with its oyster data was included as it represented another filter feeding bivalve along with the gastropod grazers. It is noted the oyster concentrations reported by McVay et al. (2018) are elevated and this is possibly because of their physiological differences but also the estuarine setting of Merimbula Lake and more industrialised port setting of the Twofold Bay embayment that McVay *et al.* (2018) sampled from and it may not be a valid comparison in this instance.
    - Zinc median and 95th percentile concentrations in mussels were lower than FSANZ GELs for oysters, adopted as the closest GEL that could apply (albeit limited) for mussel. The GEL was adopted because they are both bivalve filter feeders notwithstanding their physiological differences.
    - Cadmium in mussel flesh ranged <1-2 mg/kg, with a mean of 0.9 mg/kg, median of 0.5 mg/kg, 90th percentile of 2 mg/kg and standard deviation of 0.6 mg/kg. The cadmium median was at or below the median reported for gastropods by McVay et al (2018) for Merimbula and Twofold Bay. Results were at or below the FSANZ ML for molluscs (2 mg/kg).
- For abalone collected from the four sites, zinc concentrations over all sites were reported in a relatively narrow range of 9-17 mg/kg, and lower than the range reported for mussels. This included Lennards Island A1 (14-17 mg/kg), Haycock Point A2 (11-14 mg/kg), Long Point A3 (12-15 mg/kg) and Tura Head A4 (9-17 mg/kg).
- For the pooled abalone dataset for all four sites between Tura Head and Lennards Island, zinc ranged 9-17 mg/kg with a mean of 13.0 mg/kg, median of 13.5 mg/kg, 90th percentile of 16.0 mg/kg and standard deviation of 2.4 mg/kg. The mean of 13.0 mg/kg was slightly above the mean of 11.3 mg/kg reported for abalone in Victorian waters by Fabris *et al.* (2006). The median of 13.5 mg/kg was well below the median reported for rocky gastropods (70-90 mg/kg) by McVay *et al.* (2018) for Merimbula

and Twofold Bay. Median and 95th percentile results were also lower than FSANZ GELs for oysters, adopted as the closest GEL that could apply (albeit limited given that oysters are a filter feeder) for abalone.

- Overall, the fish and shellfish biota results were comparable to data and standards published in the literature and did not indicate metal bioaccumulation that could be attributed to wastewater discharge from the STP under existing conditions. Metal concentrations were similar at both Merimbula Bay sites and at reference sites to the north at Tura Beach and south at Quondolo Beach/Lennards Island.

## 6.8 Key Findings

Based on the scope of work undertaken, key findings include:

- The Merimbula STP has operational policies and legal controls in its licence that mitigate the potential risk of bioaccumulative contaminants entering the STP treatment system and wastewater stream. This includes specific clauses on prohibited substances, including OC and OP pesticides.
- Potential bioaccumulative contaminants identified from the desktop review include metals, OC and OP pesticides, PFAS and microplastics.
  - Review of metals found they are commonly detected in wastewater. For Merimbula STP under current operating conditions, some metals were reported in the wastewater at concentrations above DGVs for aquatic ecosystem protection, which triggered further evaluation in wastewater dispersion modelling and collection of an aquatic biota dataset for existing 'baseline' conditions (this study).
  - Review of OC and OP pesticides found that the risk of these being present in the wastewater is low, with lines of evidence that include STP operational policies and licence conditions, bans on the use and importation of these pesticides implemented in the 1980s and 1990s and the largely domestic catchment of the Merimbula STP.
  - Review of the PFAS NEMP (HEPA 2020) found that it can be present in wastewater. PFAS is not a parameter in the wastewater monitoring program, although operational policies and the STP licence would apply for PFAS in prohibiting substances assessed as unsuitable to discharge into the sewerage system. The PFAS NEMP notes that further work in collaboration with the wastewater industry is to be undertaken to establish criteria and guidance in relation to PFAS for water authorities and environment regulators.
  - Review of microplastics found there is no known data from the STP that indicates whether they are likely to pose a risk to the receiving environment. If they are present in the sewerage system, a combination of government and community initiatives to minimise the input of microplastics into the sewerage system along with filtration capability in the STP itself are measures that are expected to reduce the risk of microplastics in the treated wastewater being discharged into Merimbula Bay.
- Dispersion modelling results for metals in wastewater discharged from the existing beach face outfall and proposed ocean outfall was considered. For the existing beach face outfall, metals are expected to rapidly dilute in the nearshore beach zone a short distance from the discharge point, with dispersion to the north and south along the beach (towards Merimbula Lake and Pambula Lake) and out into Merimbula Bay.
- For the proposed ocean outfall, dispersion modelling of a range of conditions indicates that the required dilution factor of 100 to achieve all metal DGVs is predicted to occur within a 5-25 m diameter mixing zone around the diffuser, which would be located over soft seabed habitat. When enhanced metals removal is included in the STP upgrade, this mixing zone would be expected to be further reduced. Nonetheless, if

metal concentrations of sediments accumulate and increase over time, there is potential for uptake of metals by infauna (i.e. polychaete worms, crustacea and molluscs) and trophic transfer to and bioaccumulation of metals in resident fishes that may feed upon them. Demersal bottom-feeding fish such as flathead are potentially at risk of bioaccumulation, if indeed metal concentrations of sediments increase over time and there is an uptake of those metals by infauna. Should this occur, it is expected that this would be limited to the relatively small area of the mixing zone and the risk of exposure to bioaccumulative metals to fish, would be confined to those fish that potentially show high attraction and residency within the mixing zone. For most fish that transit through the mixing mixing zone on an intermittent basis, exposure time would be low and risk from bioaccumulative metals discharged from the diffuser low.

- The nearest rocky reef habitats for reef fish, abalone and other shellfish are located at least 1,400 m away (Hunter Reef), fish aggregation ocean artificial reef (Merimbula OAR) at 1,000 m distance and oyster aquaculture 2,700-3,000 m distance (Merimbula and Pambula Lakes). The relatively small mixing zone required for metal attenuation and the relatively large distances to these habitats indicates that the risk of bioaccumulative metals to fish, abalone and other shellfish at these locations is low and that the water quality objective of "*bioaccumulation of contaminants – no change from natural conditions*" would not be precluded by the Project.
- Aquatic biota sampling for metals undertaken for this assessment provides a baseline bioaccumulation dataset for existing conditions at Merimbula Bay compared to reference sites Tura Head and Lennards Island. Target species included flathead, mussel and black-lip abalone.
    - Zinc, as an essential trace metal, was detected in flathead, mussel and abalone at concentrations that were within or slightly above data reported in the literature and slightly above GELs in the food standards.
    - Cadmium, as a non-essential trace metal, was reported in mussels at four of the five sampling sites at concentrations that were within data reported in the literature and at or below maximum levels in the food standards.
    - The biota metals dataset is considered to reflect existing environmental conditions in this part of the NSW coast. There were no standout results for any of the sites, including at the Merimbula Bay sites, and no evidence to suggest a potential impact from bioaccumulative metals discharged in wastewater from the existing STP operation and its beach-face outfall. The dataset provides a line of evidence that the risk of bioaccumulative metals to fish and shellfish is low and that the water quality objective would not be precluded.
- Future monitoring of the risk of bioaccumulative contaminants from an upgraded STP and outfall at North-Short should include, in the first instance, the monitoring of metals in the treated wastewater and in sediments at the proposed water quality monitoring sites in Merimbula Bay. If these results indicate a potential risk of bioaccumulation to biota in the Bay, then follow up biota sampling may be triggered with the results able to be compared to the biota baseline dataset collected as part of this study.

# 7 Soft Sediment Infauna and Epifauna Community

The purpose of this section is to describe the marine fauna associated with the Type 3 soft sediment habitats of the study area that includes infauna and epifauna and assess potential impacts from the Project.

## 7.1 Background

Benthic infauna refers to those macroscopic fauna that live within the interstitial space of sediments and comprise a diverse array of taxa that includes crustaceans (*i.e.* amphipods, isopods, tanaids, cumaceans), worms (*i.e.* polychaetes and nemerteans), molluscs (*i.e.* gastropods and bivalves) and echinoderms (*i.e.* brittle stars and urchins). Epifauna refers to all those sessile and mobile invertebrates living on the sandy seabed that may include anemones, sea pens, sea cucumbers, sea-stars, and a wide array of crustaceans.

Benthic infauna assemblages are known to be sensitive to changes in sediment chemistry and have been routinely used in the assessment of the potential impacts of sewage wastewater (Gray and Elliot, 2009). Impacts on infauna communities that can be measured include changes in species abundance and richness with community composition of disturbed or impacted sites typically dominated by higher abundances of fewer species or taxon groups (Dauvin and Ruellet 2006, Dean 2008).

No information exists regarding the infauna community or the current background sediment chemistry of Merimbula Bay. Therefore, this study was undertaken to address that data gap and present a description of the sandy seabed community limited to:

- Observations of epibenthic fauna recorded in underwater video footage during broadscale scale surveys of the seabed; and
- A quantitative assessment of benthic infauna of Merimbula Bay that included a Stage 1 pilot study and more detailed sampling program in Stage 2 designed around the ocean outfall pipeline and predicted water quality impacts.

## 7.2 Review of Existing Data

There is a paucity of studies on sediment infauna on the south coast of NSW, particularly of deeper exposed open coast sites such as Merimbula Bay. The nearest relevant study relates to the EIS undertaken at the dredge disposal spoil ground for the Eden Harbour – Breakwater Wharf Extension project (AMA, 2016). The EIS included an ecological characterisation of the proposed offshore spoil ground location. The dredge spoil ground is situated approximately 15 km to the south of Merimbula, and 7.2 km offshore from the mouth of Twofold bay in 65 to 70 m depth. The infauna assemblage of the Eden spoil ground was found to be dominated by five taxa including two families of polychaete worms (*Spionidae* and *Orbiniidae*), two crustacea (Gammaridean amphipods and the tanaid family Apseudidae) and Hydrozoa. Owing to the high reproductive capacity of these taxa, the authors inferred the infauna community of the spoil ground to be characterised by opportunistic taxa capable of rapid colonisation (AMA, 2016).

## 7.3 Stage 1 Pilot study

An initial pilot study was undertaken in Stage 1 to collect some preliminary baseline information to describe the benthic infauna community and sediment conditions at two locations that, at the time, were being considered as potential options for the pipeline and outfall diffuser. The locations included 30 m depth at Haycock Point and 30 m depth in central Merimbula Bay (**Appendix E**).

Findings from the pilot study were then used to design an appropriate sampling program to be implemented during Stage 2 following selection of the preferred pipeline and diffuser option.

### 7.3.1 Approach

Sediment samples were collected from a total of four sites at each location spaced at 0 m, 50 m, 200 m and 400 m intervals along a uniform depth contour with similar sediment characteristics. The spacing of sample sites was designed to provide some information regarding the spatial variability among taxon assemblages that may occur over a hypothetical mixing zone of 400 to 500 m that would be relevant to this project. At the time of survey, neither the preferred pipeline alignment option nor dispersion modelling predictions of the wastewater plume were known. A summary of sample site coordinates is provided in **Appendix E.**

Sediment samples were collected using a van veen grab sampler for analysis of taxonomy of sediment chemistry (**Figure 7-1**).

**Figure 7-1 Van Veen grab sampler used to sample marine sediments**

**Sediment Chemistry**

Sediment samples were submitted to ALS under chain of custody (COC) for analysis of:

- Total Organic Carbon (TOC) – important for understanding current status of organic enrichment of sediments.
- Sediment Particle Size Distribution (PSD) sorted to 63 µm by laser diffraction – to characterise/demonstrate differences (if any) in sediment types between sites.
- Total Metals (Sb, As, Be, B, Cd, Cr, Co, Cu, Pb, Mn, Mo, Ni, Se, Ag, Sn, Zn, Hg); and
- Nitrogen and Phosphorus nutrients.

**Infauna Taxonomy and QAQC**

Samples were sorted under dissection stereoscope, separating major taxonomic groups *(i.e.* polychaete worms, crustaceans, gastropods) with taxa identified to lowest practical taxonomic resolution and abundances of each taxon recorded (**Figure 7-2**). Taxon groups were stored in separate vials of 70% ethanol and archived as project vouchers. Taxonomic identification QAQC included sending voucher specimens to the Australian Museum for identification checks. All voucher reference samples have been lodged with the Australian Museum marine invertebrate collection.


**Figure 7-2 Process of sorting infauna samples**

**A.** Initial pick of specimens from tray, **B.** Some samples required multiple sieve sizes to efficiently sort through sample volume, **C.** Major taxon groups sorted into separate petri dishes for identification under stereoscope.

### 7.3.2 Key Findings

Key findings from the Stage 1 study included:

- Haycock Point sediment characteristics were different to Merimbula Bay (**Figure 7-3**). Haycock Point sediments were dominated by fine to medium grain sands (median particle size ranging between 332 μm to 363 μm), compared to the medium to coarse grained sediments of Merimbula Bay sites with median particle size ranging from 642 μm to 1200 μm and higher proportion of shell grit and some pebble. Locations were characterised by similar levels of organic enrichment with total organic carbon (TOC) ranging between 0.05% to 0.07% across all samples.
- A total of 66 infauna taxa were recorded representing 10 phyla and 63 families. Of these, 12% of taxa (8 of 66) were represented by a single specimen.
- Annelid worms made up the greatest proportion of taxa including 21 families. Crustacea were next most diverse with 18 families, followed by mollusca (gastropods and bivalves) with 14 families. All other phyla accounted for the remaining 10 families representing a minor proportion and included pycnogonids (sea spiders), nemerteans, echinoderms (brittle stars, urchins, sea cucumbers), bryozoans, foraminferans, and chordate (lancelets and sand-burrowing fish).
- Merimbula Bay sites were characterised by a more diverse infauna assemblage than Haycock Point, with greater number of species and generally higher abundances (**Figure 7-4**). This difference between locations was attributed to the different physical conditions of the sediment with the larger sediment particle size of Merimbula Bay favouring a more diverse infauna assemblage.

- Four taxa that were important for discriminating differences between locations (*i.e.* abundant at one location but absent from other) included the polychaete worms Maldanidae and Sabellidae at Haycock Point, and Tanaids and Ostrocods at Merimbula Bay.

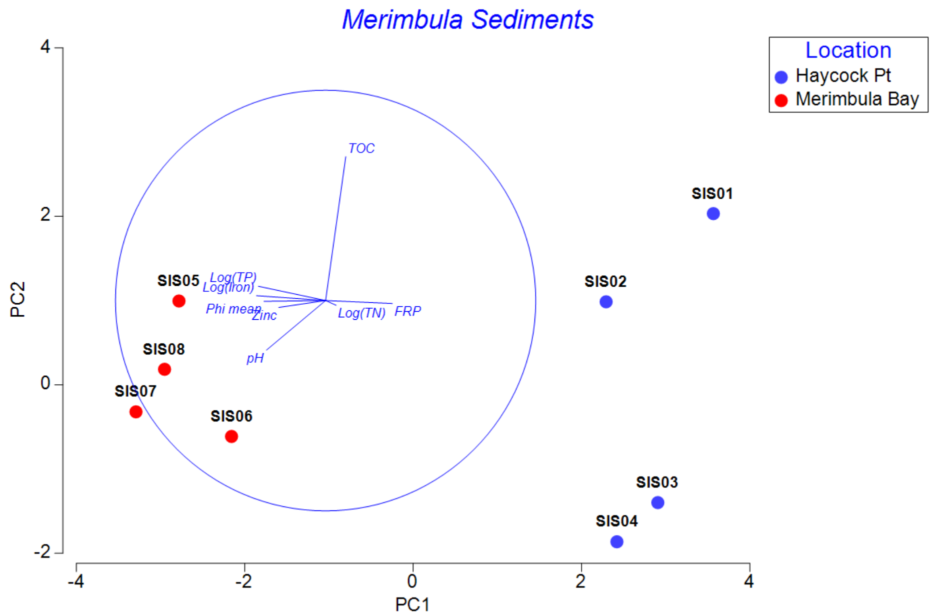



**Figure 7-3 PCA plot of sediment characteristics by location**

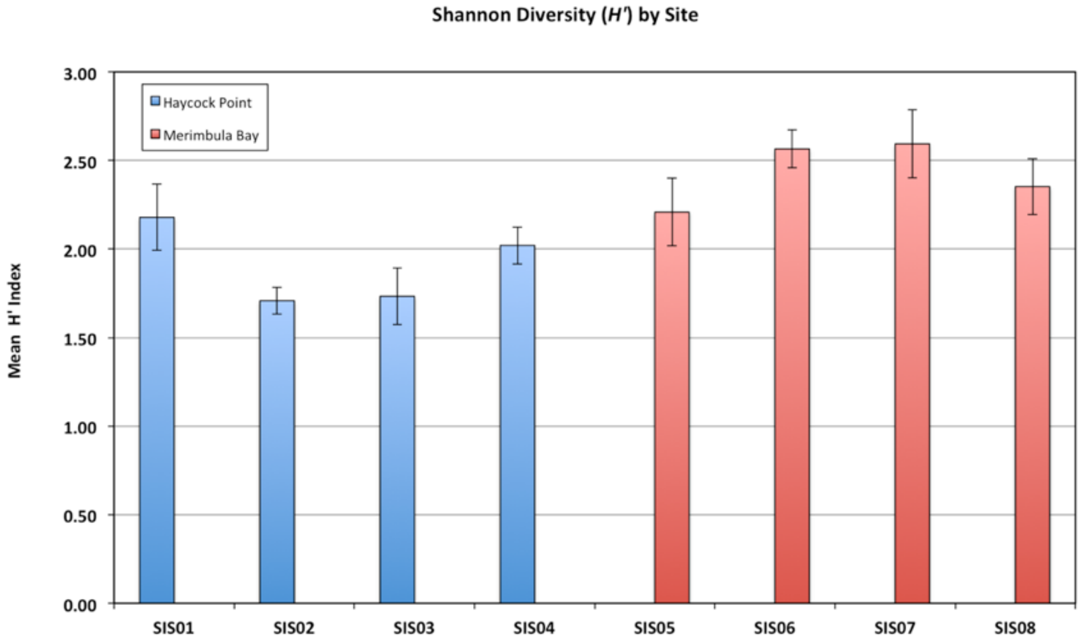



**Figure 7-4 Mean Shannon diversity for infauna by sample site (n = 3 replicates)**

## 7.4 Stage 2 Field Surveys

Further field surveys for benthic infauna were initiated in Stage 2 following selection of the preferred North-Short outfall location in 30 m depth in central Merimbula Bay.

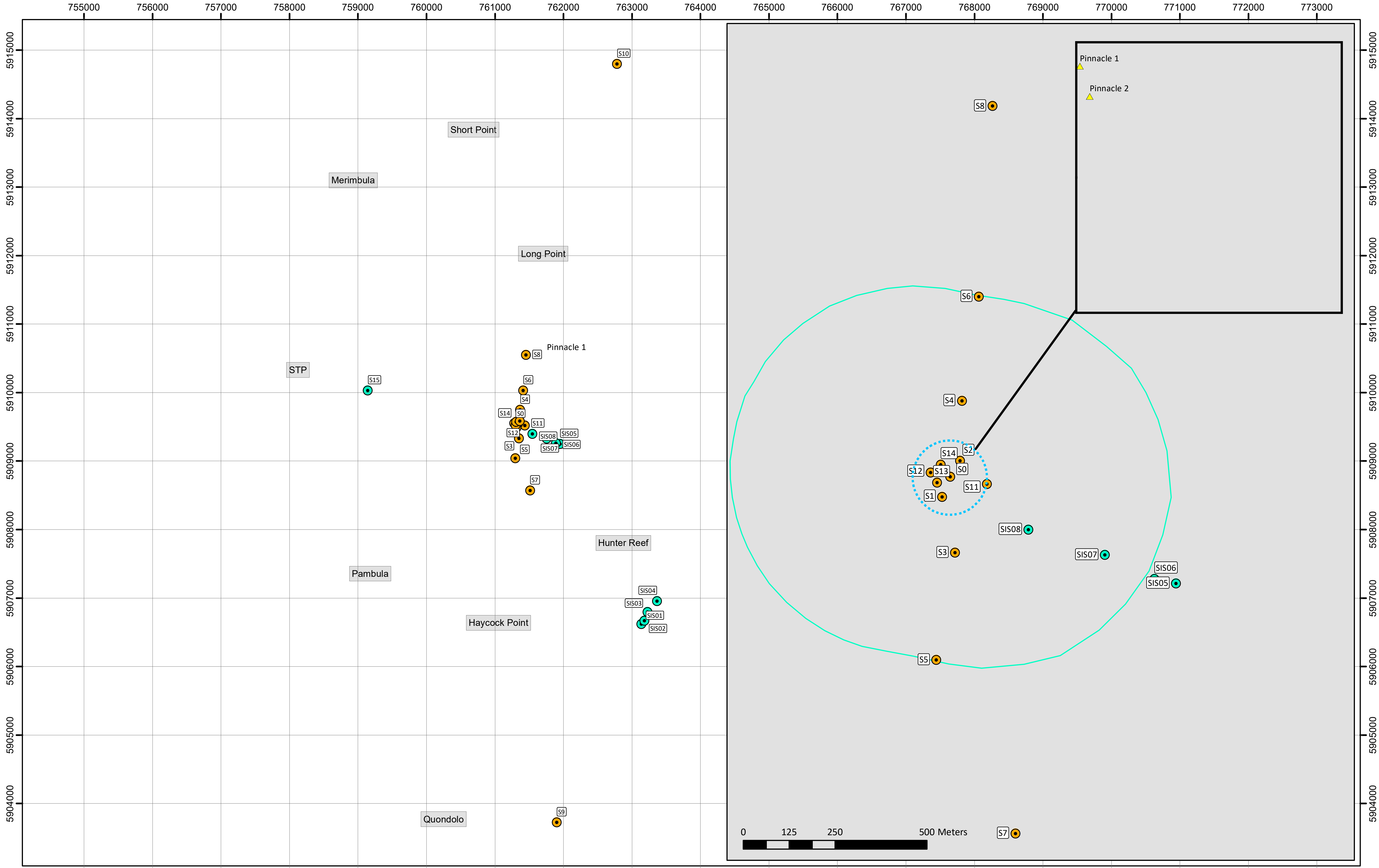
The Stage 2 objective was to assess the natural background variation of the infauna community at the proposed diffuser location prior to operational phase and compare to number of reference locations situated at varying distances from the diffuser location. Further understanding of how the soft sediment infauna community naturally fluctuates over time and over a range of spatial scales is necessary to inform what level of community change (i.e. diversity, abundance, richness) could be considered an effect that may be attributable to the discharge of treated wastewater.

#### 7.4.1 Sampling Design

A total of 15 monitoring sites were established including four sites within the near-field mixing zone (diffuser to 25 m), five sites in the far-field mixing zone (25 m to 200 m), and six sites outside the far-field mixing zone at increasing distances from the diffuser (**Figure 7-5**).

#### 7.4.2 Results

Analysis of the Stage 2 dataset is pending at the time of this report. Conclusions regarding the assessment of potential impacts to this community is unlikely to change.




N

Scale = 1:22,000

0 500 1,000 2,000 Meters

Stage 2 Benthic Infauna Site
Stage 1 Pilot Study
Area of Investigation Boundary
100m zone around Site S0.
Pipeline Alignment
Below Ground
Diffuser
SEARs 500m bufffer from diffuser

Notes:
1. Elgin BRUV sampling conducted Stage 1: 3-9 November 2017, Stage 2: 10 October 2019.
2. Southern bathymetry data reported by Marine and Earth Sciences in 2017
3. Northern bathymetry data reported by Southern Divers and Total Hydrographic in 2017

**Project:**
MERIMBULA STP UPGRADE AND OCEAN OUTFALL ENVIRONMENTAL ASSESSMENT

**Client:**
AECOM AUSTRALIA

**FIGURE 7-5**

LOCATION OF SOFT SEDIMENT INFAUNA SAMPLE SITES

Date: 22 September 2020
Version: 1
Size: A3

Document Path: D:\Elgin Associates Dropbox\Elgin_GIS\New_South_Wales\BVSC GIS Data\Merimbula EIS\1. MXD Mapping Files\Stage2_Reporting\Figure_8-1_Benthic_Infauna_locations_22Sept2020.mxd

## 7.5 Epibenthic fauna

A qualitative assessment of sessile and mobile epifauna inhabiting the sandy seabed was based on observations recorded opportunistically from video surveys of the seabed. Epifauna observed within the study area include hermit crabs (unid.), seastars *Plecaster* sp. and *Astropecten vappa*, and the sessile great sea pen (*Sarcoptilus grandis*).

## 7.6 Construction Phase Impacts

Potential negative effects from construction phase activities to the benthic infauna and epifauna communities include:

- Accidental spill from construction vessels
- Disturbance and loss of Type 3 sand habitat establishing the pipeline and diffuser infrastructure

### 7.6.1 Accidental spill

There is the potential for hazardous substances (*ie.* fuels, oils and other construction vessel related fluids) to accidently enter the water through spills or leaks from construction vessels and/or equipment. Water pollution resulting from vessel accidental spill would typically impact the water surface and have limited direct effect on fish assemblage. The potential impacts of water pollution on marine fauna can be harmful and is considered a low risk. This risk can be reduced to low risk by implementing a range of control measures to protect water quality during construction.

### 7.6.2 Disturbance and loss of Type 3 soft sediment habitat

Disturbance and loss of Type 3 soft sediment habitat is an unavoidable impact. As described in **Section 2 – Overview of Marine Habitats**, it is estimated that construction of Section 2 pipeline and diffuser would result in the direct disturbance and loss of 0.00432 km$^2$ Type 3 unconsolidated sand habitat, considered minimally sensitive with regard to fish habitat. Based on an estimate of 12 km$^2$ of sand habitat within the study area, this represents a 0.04% loss of Type 3 soft sediment habitat mapped within the study area.

Establishing the pipeline infrastructure would result in the smothering of benthic infauna or sessile epifauna directly below the pipeline footprint. This includes the sessile epifauna such as the great sea pen (*Sarcoptilus grandis*) and benthic infauna community. Benthic infauna includes polychaete worms, crustacea, molluscs, and other minor groups. The impact on infauna is expected to be minimal as infauna are highly mobile and can move to adjacent habitat. Sessile epifauna such as the Great sea pen may be present along the pipeline alignment and could be lost due to direct physical damage. Few individuals of the Great sea pen were noted during surveys, conservatively estimated at one per 10,000 m$^2$, and the potential loss of a few individuals is not expected to have an adverse effect on the local population.

Overall, the scale of the sand habitat lost to the Project is minor and is unlikely to have a long-term negative effect on the faunal assemblages that rely on sand habitat within Merimbula Bay in terms of their diversity and abundance.

## 7.7 Operational Phase Impacts

Potential effects from the operational phase of the Project to the soft sediment infauna community could arise from changes to sediment chemistry caused by the deposition of suspended sediments, particulate organic material (POM) and contaminants absorbed to particles discharged in treated wastewater.

Accumulation of deposited POM can cause localised enrichment of sediments and or depletion of oxygen within those sediments. The benthic infauna community is known to be sensitive to such changes and the potential effects of enrichment associated with wastewater discharges has been widely studied both in

Australia (Otway, 1995; Scanes and Philip, 1995) and overseas (Warwick, 1993; Gray and Elliot, 2009; Puente and Diaz, 2015). Typical effects include altered community structure (i.e. change is species richness and abundance), and changes in the proportions of opportunistic-sensitive species or trophic groups.

Soft sediment infauna community is comprised typically of variable proportions of three main taxon groups – polychaete worms, crustacea, and molluscs, as well as other minor groups (nemerteans, sipunculids, cnidarians, echinoderms and fish). Among these, particular polychaete worm taxa have been identified as useful bio-indicators of organic enrichment as they tend to be tolerant of hypoxic conditions and proliferate under organically enriched conditions. One such example, commonly used as an bio-indicator of eutrophic systems, is the polychaete genus *Capitella* (Pearson and Rosenberg, 1978; Macleod and Forbes, 2004; Edgar et al., 2005; Weisberg et al., 2008). Other polychaete families that are considered indicators of pollution include Spionidae, Orbiniidae, Cirratulidae, Neridae, Nephtyidae, Dorvilleidae, Goniadidae, Hesionidae, Lumbrineridae and Phyllodocidae (Pearson and Rosenberg, 1978). Among studies somewhat relevant to Merimbula Bay are the studies associated with monitoring the environmental impact of wastewater discharge from the Sydney deep water outfalls (Otway, 1995; Scanes and Philip, 1995; Leadbitter, 1996; Philip and Pritchard, 1996). Long-term effects on the soft sediment infauna community noted at two of Sydney deep-water outfall sites were increased abundance of crustaceans Anthurid and Paranthurid isopods, with an overall decrease in polychaete worm diversity noted at one site (Otway, 1995; Scanes and Philip, 1995).

While impacts to infauna communities from wastewater discharges are well documented, the scale of the potential impact is related to the hydrodynamic environment, volume and quality of wastewater being discharged, with higher energy environments such as deep water oceanic settings at lower risk of POM accumulation near the outfall. For the three Sydney deep water outfalls, the daily discharge volume ranges between 126 and 498 ML per day (from Table 1 in Puente and Diaz, 2015), that is many magnitudes greater than the current daily average dry weather (ADW) discharge volume at Merimbula of 1.4 ML per day. Furthermore, future average daily disposal volumes are forecast to decrease to 1.1 ML from the current 1.4 ML (**Section 1.3)**. As described in **Section 1.4**, existing treated wastewater at Merimbula is characterised by typically low total suspended sediment load (median TSS = 5 mg/L), with intermittent higher suspended loads discharged during wet weather flows. Historical exceedance of the TSS discharge limit (30 mg/L) occurred six times over a 10-year period and is attributed to microalgae growth within the wastewater storage pond prior to discharge. The discharge of freshwater microalgae in wastewater represents a potential food-source for filter-feeding invertebrates and zooplankton and may provide some benefit in the marine environment. Furthermore, TSS of ambient ocean waters can often be higher than that contained in wastewater discharge, particularly during upwelling events and catchment flood flow discharges. Overall, exisiting wastewater quality is very clear and the risk of POM accumulation to sediments within the mixing zone over long-term is considered low to medium noting that upgrades to the STP would result in further improvements in wastewater quality.

Another pathway by which sediments may become enriched is if the discharge of dissolved nutrients to the water column stimulates excessive phytoplankton growth that could then deliver additional POM to the benthos. The threat of dissolved nutrient load discharged to the mixing zone and its potential effect on the phytoplankton community and risk of increased occurrence of algal blooms was assessed in **Section 9 – Phytoplankton**. It concluded that the discharge of nutrients to the mixing zone would provide a localised stimulus for increased primary productivity where it is expected that majority of this nutrient load would be assimilated by phytoplankton within the 25 m mixing zone. However, the overall effect this may have on the phytoplankton assemblage of Merimbula Bay would be minimal. This finding is based on the nutrient discharge being localised, small in scale compared to episodic nutrient inputs from upwellings and catchment flood events, and an understanding that phytoplankton assemblage dynamics of Merimbula Bay (i.e. change in species composition and abundance) are more likely to be influenced by environmental factors operating at broader bioregional, ocean basin scales.

Should changes to sediment chemistry occur from the Project, these would likely to be limited to the near-field mixing zone of 25 m radius from the diffuser with some level of change to the infauna community possible. It is then expected that the magnitude and likelihood of potential change would decrease with increasing distance from the outfall and the ability to detect change beyond the mixing zone, if some change has occurred, becomes less likely. The risk of potential change to sediment chemistry within the predicted 25 m mixing zone that could cause some level of community response among benthic infauna is considered a medium risk.

Control measures to mitigate risk of potential impact to the infauna community include proposed STP upgrades of PAC dosing for enhanced phosphorous removal, and UV disinfection to remove microbial contaminants. If required, an option to reduce the risk of metals in the treated wastewater would be the addition of tertiary filtration which would improve metal removal from the wastewater stream. However tertiary filtration would be expected to only marginally decrease an already low risk and would not change the extent of the mixing zone required for the necessary dilution of nutrients (NOx and ammonia) to meet MWQOs. Therefore, including tertiary filtration for the purposes of enhanced metal removal would be unjustified.

It is recommended that soft sediment infauna monitoring at sites within and outside the mixing zone form a key element of operational phase environmental monitoring. Combined with water quality monitoring, an appropriately designed benthic infauna monitoring program would provide a useful approach for validating assumptions of dispersion modelling, the extent of the mixing zone and predicted impacts on water quality from the Project.

# 8 Assessment of Intertidal Rocky-Shore Community

## 8.1 Introduction

This section presents a description of the intertidal rocky reef communities located adjacent to the study area based upon a desktop review of existing information and field survey of intertidal rocky reef habitats at Haycock Point.

Intertidal reef communities at Haycock Point were considered to be potentially affected by the Project based on initial advice that a location east of Haycock Point in 30 m depth was suitable in terms of dispersion of the treated wastewater plume (MHL, 2015) and was among the options being considered for position of the outfall diffuser.

Prior to the commencement of fieldwork for this study the final location for the outfall had not been decided. The aim of the assessment was to provide a preliminary description of the intertidal community at Haycock Point, document species composition using a combination of quantitative and qualitative methods and identify the dominant community components that may be useful indicators to inform ongoing monitoring requirements for the EIS, if deemed required.

The potential effects of the Project construction and operational phases on intertidal rocky shore community is evaluated.

### 8.1.1 Description of Intertidal Environment of Haycock Point

The underlying geology of Haycock point is Devonian and displays a large amount of highly sinuous metamorphosed red siltstone. The folds and bends that are characteristic of this geology have promoted weathering that has left the rock surface ribbed and serrated, creating potholes and rock-pools. Differential weathering across the platform has formed a series of medium to high relief gutters and crevices oriented north to south. These formations are characterised by boulders, cobbles and sand that are inundated during high tide and subject to high water movement with waves washing through. These gutters also limit access to the eastern-most part of the point to periods of low tide springs. Alternative access may be achieved by boat in calm conditions.

The easternmost part of the point is characterised by the prominent Haystack Rock surrounded by high flat reef platforms that are exposed during periods of high tide springs with margins that drop-off vertically into the water. Consequently, the majority of the intertidal community here is located on vertical rock faces with only taxa characteristic of the high littoral splash zone found on the reef platforms. This area of the point was not included in the assessment of intertidal communities.

The intertidal reef area that is immediately accessed below Haycock Point is characterised by a variety of substrates (*i.e.* sand deposits, boulder fields, bedrock platforms) ranging from moderately sloped to vertical. The intertidal area can be divided broadly into two environmental aspects with south-facing and north-facing reef exposed to swell from southern and northern quadrants respectively. It is assumed that easterly swells would have a relatively even affect along both shoreline aspects. Environmental aspect and level of wave exposure are important abiotic factors that influence the structure of intertidal communities and these were considered during the selection of sampling sites.

An image of Haycock Point is provided in **Figure 8-1**.

**Figure 8-1 View of Haycock Point intertidal rocky shore**

## 8.2 Field Survey

### 8.2.1 Sample Sites and Sampling Effort

A site walkover to establish sampling sites was undertaken on 7 December 2017. A large NE swell event was washing over the point at the time limiting sampling to the southern side only. The northern side was subsequently sampled on 2 January 2017.

Sampling sites were established on the conducted south and north side of Haycock Point to examine the natural variation of the intertidal communities associated with different exposure aspects as per the approved workplan. Quantitative sampling focused on the low shore zone while the mid and high shore zones were qualitatively assessed.

Two replicate 25 m long transects were sampled within the low shore zone on the south side (site 1) and two on the north side (site 2), with 12 photo-quadrats (size 0.25 $m^2$) sampled per transect at random intervals (using a random number generator). The location of transects is provided in **Figure 8-2**. Sampling of each quadrat used photographic methods to provide an archival record of the assessment and provided an efficient method to capture data for post-fieldwork processing using image analysis.

The low shore zone available for survey was limited to sloping reef characterised by algae and cunjevoi (*Pyura stolonifera-praeputialis*). Vertical rock faces were present in the low shore zone, but these were excluded due to difficult access for assessment. Consequently, replicate transects targeting sloping reef areas were separated over a scale of 50 to 100 m.

A diagrammatic representation of sampling effort at low shore height is provided in **Figure 8-3**.

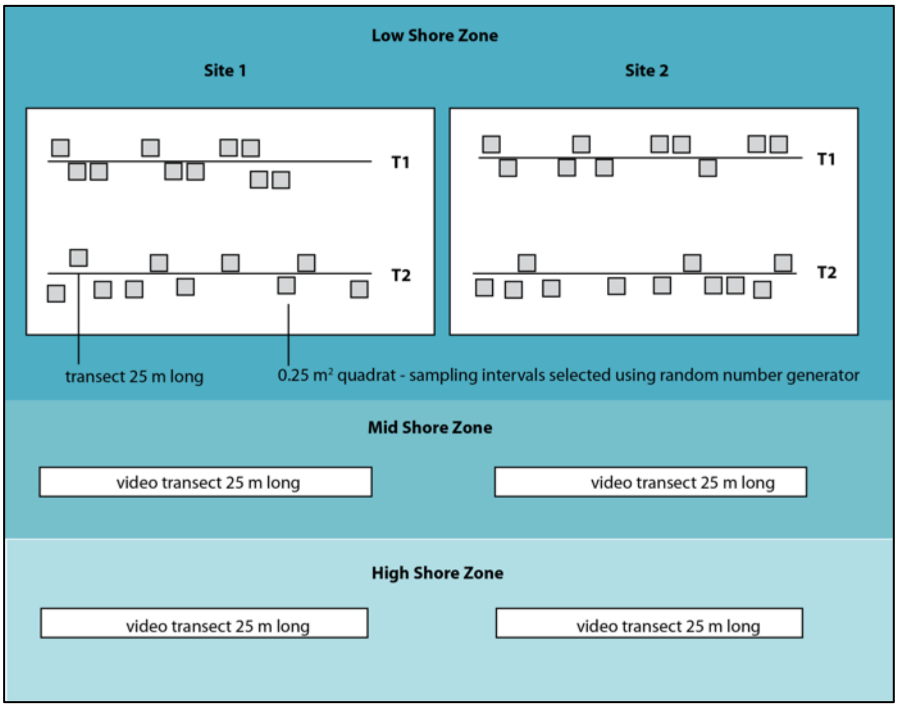



**Figure 8-2 Assessment of intertidal community - sampling effort at each site and shore height**

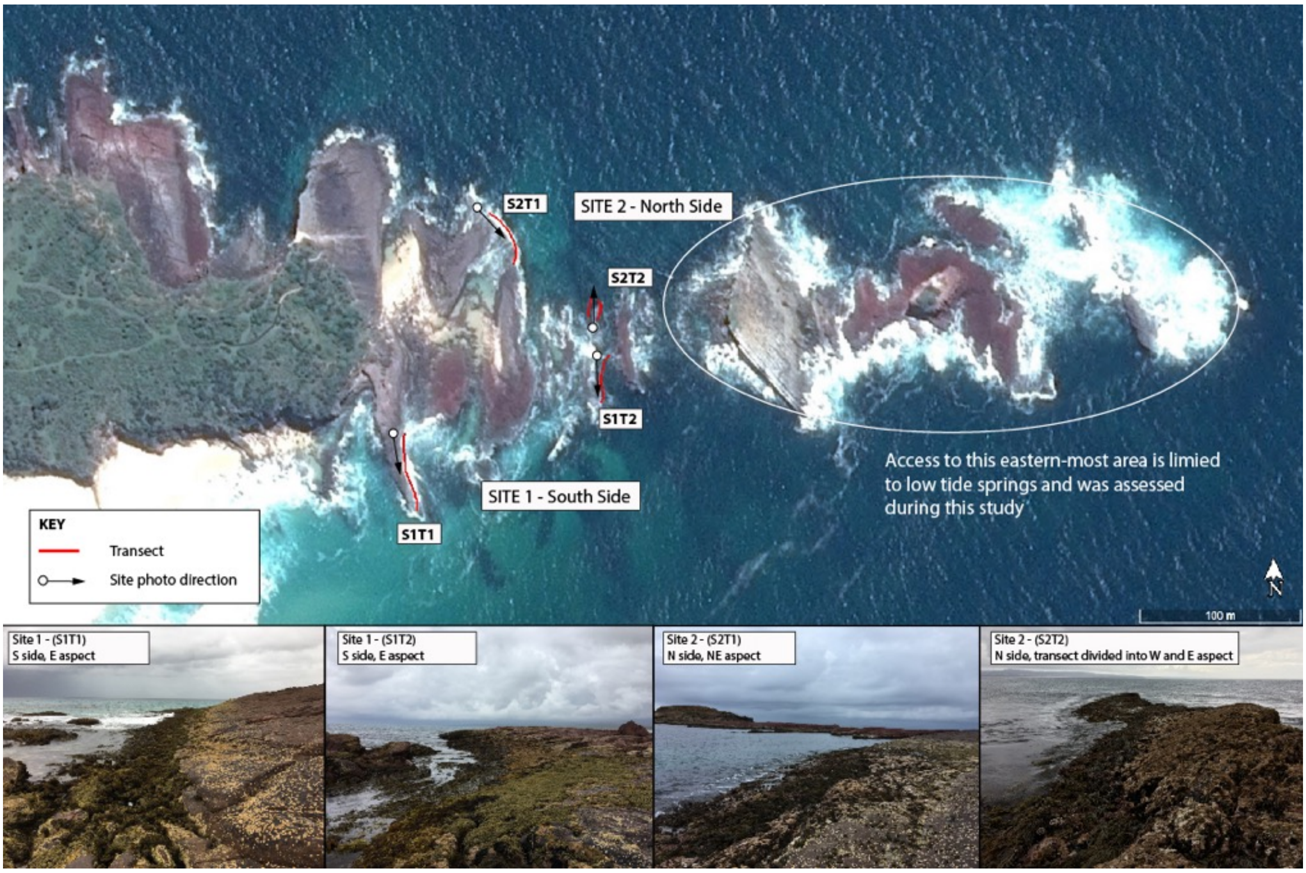



**Figure 8-3 Location of intertidal sampling sites at Haycock Point**

### 8.2.2 Data Analysis

**Data Management**

Species counts and percentage cover estimates were entered into an excel spreadsheet. All raw community data is provided in **Appendix F**.

**Species Diversity Measures**

Species diversity was estimated by total taxa count (*i.e.* taxon richness as *S*) and Shannon diversity (*H'*) index. Shannon diversity takes into account the abundance and evenness of species within a community with index values increasing as both the richness and evenness of the community increase. A community with high Shannon index can be interpreted as having greater overall diversity with high species richness and high abundance of those species, compared to a community with low Shannon index that may have equally high species richness yet is dominated by high abundances of just a few species. Shannon diversity was only calculated for gastropods and algae as they represented the most diverse and abundant taxon groups.

**Community Composition**

Community composition of the low shore was explored using a combination of univariate and multivariate statistical methods in Primer (Clarke 1993). The raw data were square root transformed to down-weight the influence of highly abundant species in describing community structure. Data transformation balances the contributions from common and rarer species in the measure of similarity between two samples. Using the transformed species counts and percent cover data, Bray-Curtis similarities/dissimilarities between samples were calculated and patterns in community structure were evaluated by portraying these results in a two-dimensional ordination using non-metric multidimensional scaling (nMDS). The Bray-Curtis index compares the abundance or percent cover of each taxon between two samples and derives a single value that represents the difference between the samples, expressed as a percentage (Clarke 1993).

The 'stress' value is an indication of how well the high-dimensional relationships among the samples are represented in a two-dimensional MDS plot with values near to zero providing best representation.

## 8.3 Results

### 8.3.1 Sampling Effort - Species Accumulation Curve

A general characteristic of most intertidal communities is their patchiness or heterogeneity, which typically results in an increased number of species observed with increasing area sampled. The species accumulation curve is a way of visualising whether sampling effort employed has adequately captured the species diversity of the community sampled, with the curve beginning to stabilise or reach an asymptote once majority of species diversity has been observed.

Species accumulation curves were plotted for each of the four transects (**Figure 8-4**). With the exception of S2T2 that suggests majority of species diversity has been captured with 12 quadrats, the increasing trend of the curves for the other transects suggest that additional quadrats need to be sampled in order to adequately capture the diversity present due to the presence of relatively rare taxa that are sparsely distributed over the low shore habitats. At least 11 taxa were recorded in the low shore zone that were considered rare and these curves suggest that additional taxa with low abundances may be observed with further sampling.

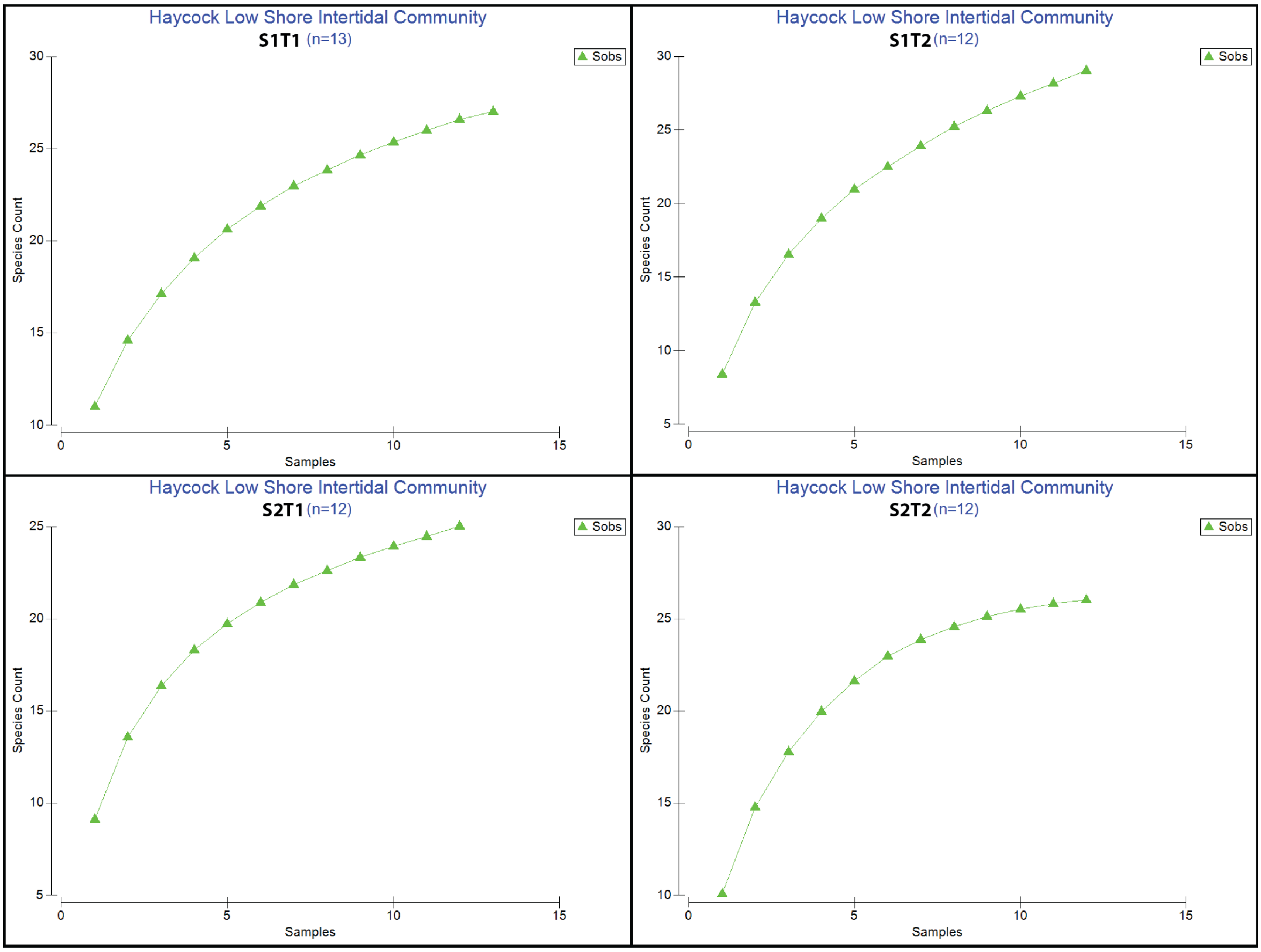



**Figure 8-4 Species accumulation curves for transects sampling the low shore community at Haycock Point**

### 8.3.2 Intertidal Community

A total of 45 taxa were recorded over the intertidal rocky shore at Haycock Point with a summary of taxa and the shore height in which they were observed provided in **Appendix F-1**. All raw community data used for analysis is provided in **Appendix F-2**.

### 8.3.3 Low Shore Zone

A total of 35 taxa were recorded across the low shore zone of Haycock Point including 13 mobile invertebrates (*i.e.* gastropods, limpets, chitons, urchin, seastar), four sessile invertebrates (*i.e.* cunjevoi, barnacles, tubeworm) and 18 algal taxa. A summary of taxon richness is provided in **Table 8-1** that shows the south side of Haycock Point to be characterised by slightly higher taxon richness than the northern side with 34 and 30 taxa recorded respectively (**Figure 8-5**).

A comparison of Shannon diversity (**Figure 8-6**) indicates that the north side of Haycock Point (Site 2), despite having fewer taxa, is characterised by more even taxon abundances (*i.e.* presence of fewer rare taxa) compared to southern side of Haycock Point, where the lower Shannon index suggests there were an increased number of taxa with low abundances recorded and likely to be relatively rare community components.

Rare taxa were considered to be those mobile invertebrates where two or fewer individuals or total percentage cover of sessile invertebrate or algae was less than 5% observed over all samples at each site. Seven taxa observed as rare on the south side of Haycock Point included the gastropod *Austrolittorina unifasciata*, limpet

*Patelloida alticostata*, sea urchin *Heliocidaris erythrogramma*, anemone *Oulactis mucosa*, foliose red alga, *Dictyota dichotoma* and green filamentous alga (*Cladophora/Chaetomorpha* spp.). In comparison, four taxa were observed to be rare on the north side of Haycock Point that included the gastropod *Austrocochlea porcata*, limpet *Patelloida mufria*, alga *Hypnea* sp. and *Hormosira banksii.*

**Table 8-1 Summary of taxon richness and mean Shannon diversity for the low shore community at Haycock Point**

| | Site 1 – South Side | Site 2 – North Side |
|---|---|---|
| Total taxon richness | 34 | 30 |
| Mean Shannon diversity (*H'* index) | 1.60 ± 0.09 | 1.92 ± 0.11 |

Note: Sample n = 24 for Site 1 and Site 2

The different species assemblages of the south (Site 1) and north (Site 2) sides of Haycock Point are confirmed in the MDS plot that shows two clear groups at 40% similarity (**Figure 8-7**). The routine SIMPER (in statistical software PRIMER) was used to identify the dominant variables or taxa driving the pattern of difference between the two sides of the headland:

- For Site 1 on the southern side of Haycock Point, the top five variables contributing to 70% similarity between samples included (in order of % contribution) cunjevoi (23%), the brown alga *Sargassum sp.* (18%), encrusting coralline algae (10%), bare rock (10%) and *Corallina officinalis* (9%).
- For Site 2 on northern side of Haycock Point, the top five variables contributing to 70% of similarity between samples included (in order of % contribution) *Corallina officinalis* (30%), cunjevoi (13%), the brown alga *Phyllospora comosa* (11%), *Amphiroa anceps* (9%) and *Lethesia* sp. (7%).

Additional analyses were undertaken to further explore and characterise the gastropod and algal species composition of the low shore community.

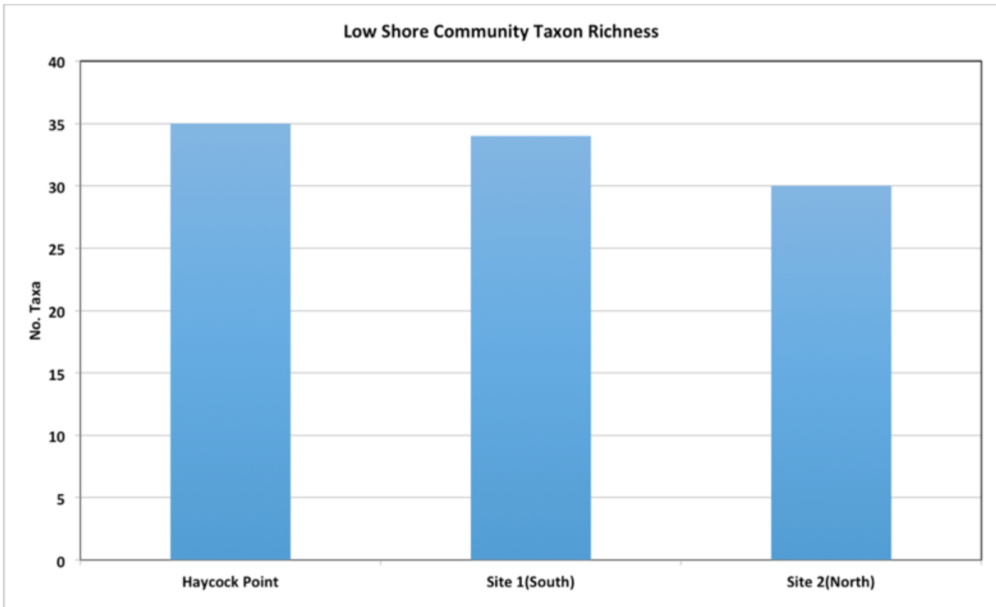



**Figure 8-5 Total taxon richness of low shore community for Haycock Point by site**

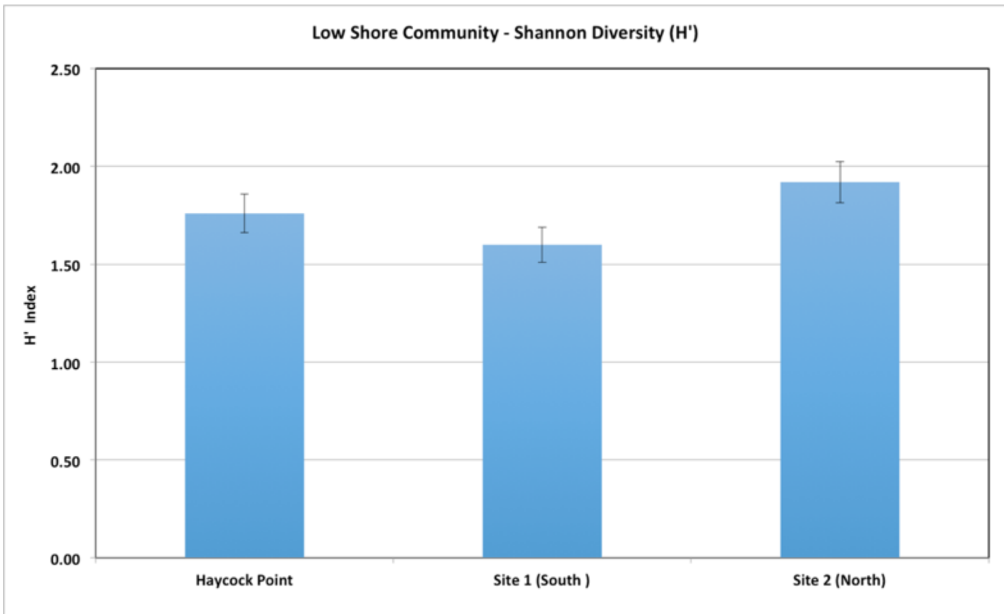



**Figure 8-6 Mean Shannon diversity of low shore community of Haycock Point (n=48) compared to each individual site (n=24)**

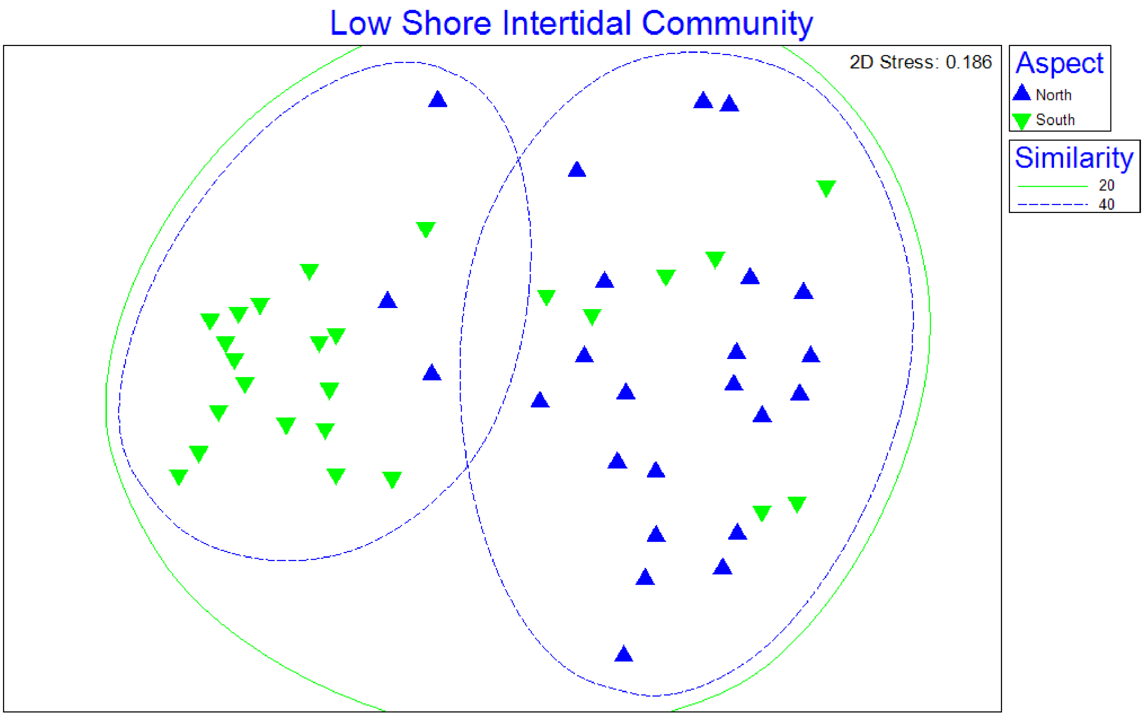



**Figure 8-7 MDS plot of Haycock Point low shore community by shoreline aspect**

**Gastropod Community**

Ten gastropod species were observed in the low shore community at Haycock Point with all 10 species found on the southern side and eight species on the northern side (**Figure 8-8).** Two relatively rare taxa recorded on the southern side but not the northern side included a few individuals of *Patelloida alticostata* and a single individual of *Austrolittorina unifasciata.* The latter is typically found in the high shore zone and is considered anomalous. Overall gastropod richness and abundance was higher on the south side (Site 1) compared to the north side (Site 2) and this is also reflected in the mean Shannon diversity of 0.58 and 0.35 respectively (**Figure 8-9**).

The most abundant gastropods on the low shore at Haycock Point included *Scutellastra peronii*, *Cellana tramoserica*, *Patelloida latistrigata, and Montfortula rugosa* with mean abundance of each species varying according to shoreline aspect (**Figure 8-10**). Abundance was patchy across each shoreline as indicated by the high standard error around the mean. The pattern in gastropod abundance appears related to the dominant substrates present at each site. The northern shoreline is characterised by a higher percent cover of *Corallina officinalis* that is known to be preferred by *Montfortula rugosa,* whereas the increased proportion of bare rock, interstitial space between the higher cover of cunjevoi (*Pyura*) and low cover of *Corallina officinalis* on the southern shoreline appears to favour higher abundances of *S. peronii*, *C. tramoserica* and *P. latistrigata.*

In terms of monitoring for environmental change, the most abundant gastropod taxa are preferred as they provide an increased likelihood of detecting change in abundance and distribution given adequate sampling. Rare taxa are not as useful as monitoring indicators as they require considerably more sampling effort in order to detect change given their sparse distributions. Gastropod abundance on the south side of Haycock Point was higher but also more variable between replicate transects compared to the north side that had lower abundances and less variability between replicate transects.

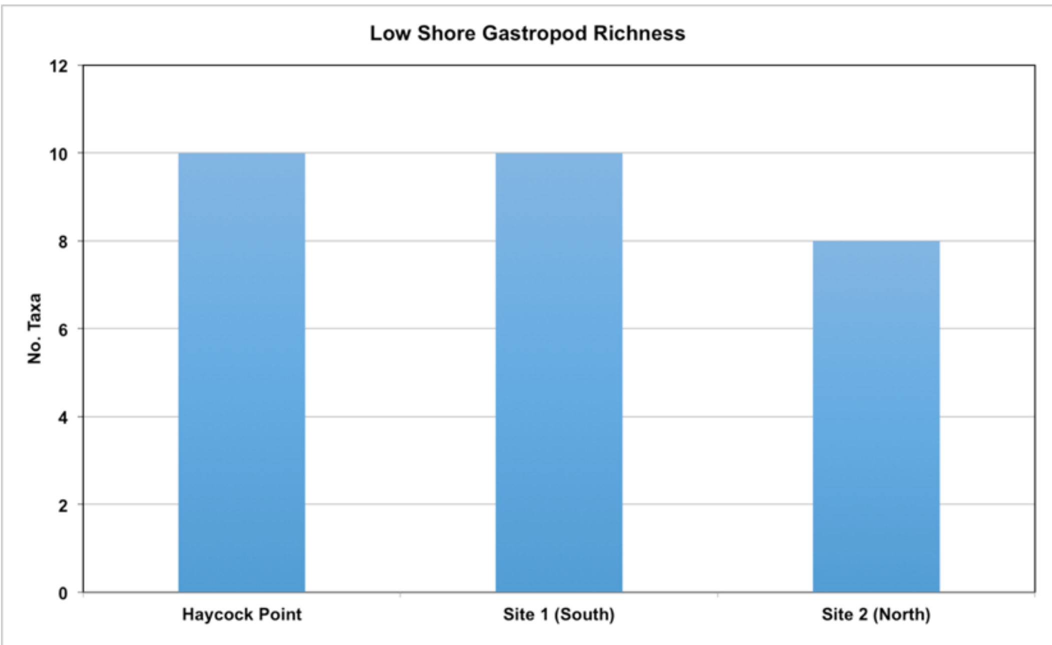



**Figure 8-8 Total gastropod richness of low shore community for Haycock Point and by site**

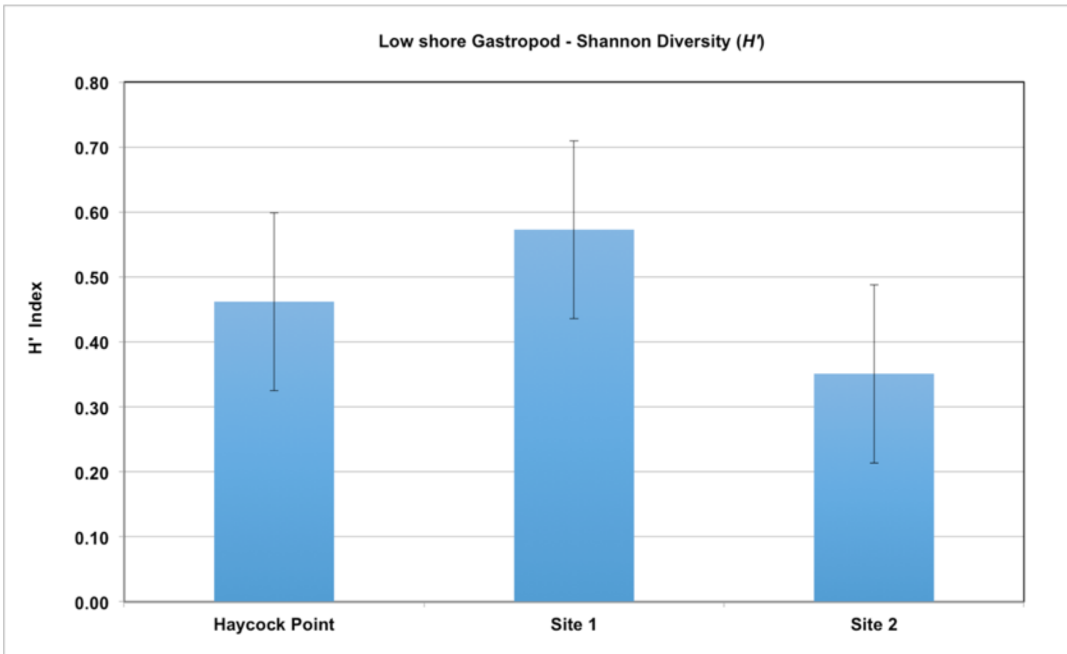



**Figure 8-9 Mean Shannon diversity for gastropod assemblage at Haycock Point (n=48) compared to each individual site (n=24)**

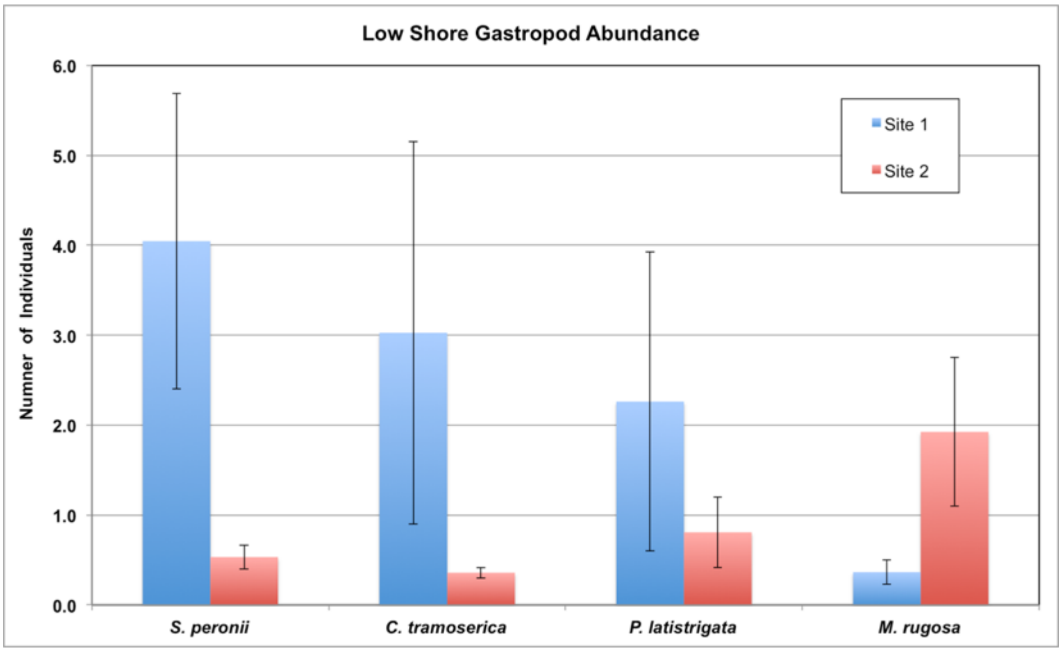



**Figure 8-10 Mean abundance and standard error of dominant gastropods by Site (n=24) at Haycock Point**

### Low Shore Macroalgal Community

The macroalgal community is the dominant substrate of the low shore zone representing approximately 73% of total cover, followed by cunjevoi (22%), bare rock (4%) with other sessile invertebrates such as tubeworms (*Galeolaria caespitosa*), barnacles and anenomes representing less than 2% cover over the low shore zone (**Figure 8-10**).

A total of 18 algal taxa were recorded in the low shore zone represented by eight (8) red algae, seven (7) brown algae and three (3) green algae. Geniculate coralline species *Corallina officinalis* and *Amphiroa anceps,* and encrusting non-geniculate coralline algae (appears as pink rock) accounted for majority of the total algal cover in the low shore zone (**Figure 8-11**). However, the macroalgal assemblage also varied according to shoreline aspect with the south and north sides 67% dissimilar. SIMPER analysis was used to identify those variables contributing to 90% of the within group similarity associated with shoreline aspect.

- Five algal taxa contributed to at least 90% of the similarity between Site 1 (south side) samples including (in order of % contribution): *Sargassum* sp. (40%), encrusting non-geniculate coralline (24%), *Corallina officinalis* (17%), *Ralfsia* sp. (8%) and *Ulva* (4%).
- Seven algal taxa contribute to at least 90% of the similarity between Site 2 (north side) samples including (in order of % contribution): *Corallina officinalis* (38%, *Phyllospora comosa* (15%), *Amphiroa anceps* (12%), *Lethesia* sp. (8%), *Sargassum* sp. (6%), *Chondrophycus* sp. (6%) and *Dictyota dichotoma* (5%).

The main algal taxa contributing to differences between the macroalgal assemblage of the south and north side of Haycock Point include higher abundance of *Corallina officinalis, Phyllospora comosa* and *Amphiroa* on north side compared to the higher prevalence of *Sargassum* sp. and encrusting non-geniculate coralline algae on the south side. It is this variation in macroalgal assemblage that also influences the associated gastropod assemblage.

Of further interest is the relatively low representation of green algae in the low shore zone. The most abundant green alga, in terms of percent cover, was *Codium lucasii* followed by *Ulva.* Filamentous green algae, collectively referring to a mixture of *Cladophora, Rhizonclonium* and *Chaeotomorphra*, were only observed on the south side but in low abundance and would be considered rare in the low shore zone. Green algal species such as *Ulva* and *Cladophora* are known to respond rapidly to increased availability of nutrients and were more prevalent in the mid shore zone though patchy in distribution.

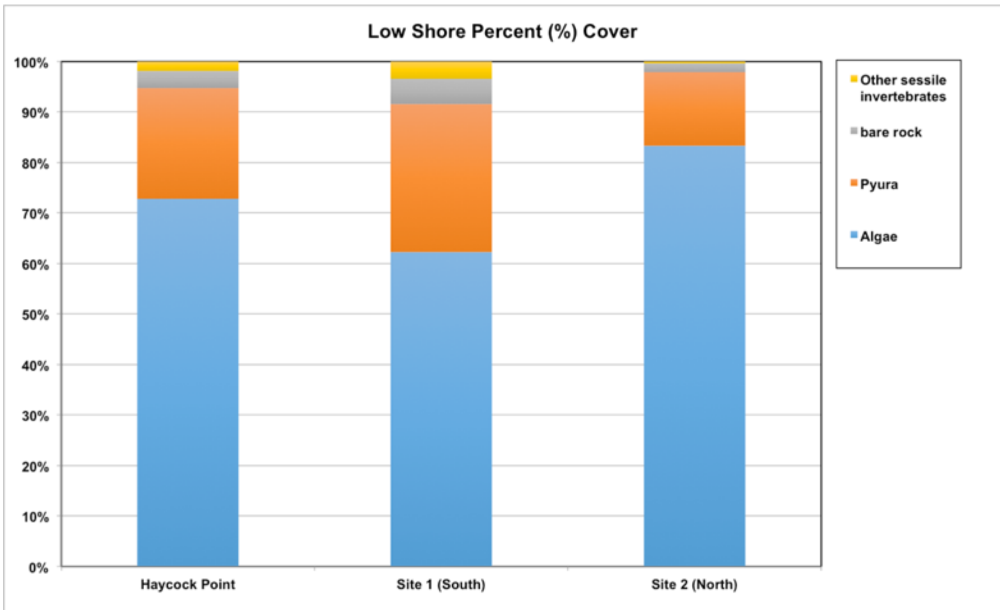



**Figure 8-11 Percentage (%) cover of sessile invertebrates and algae in low shore zone**

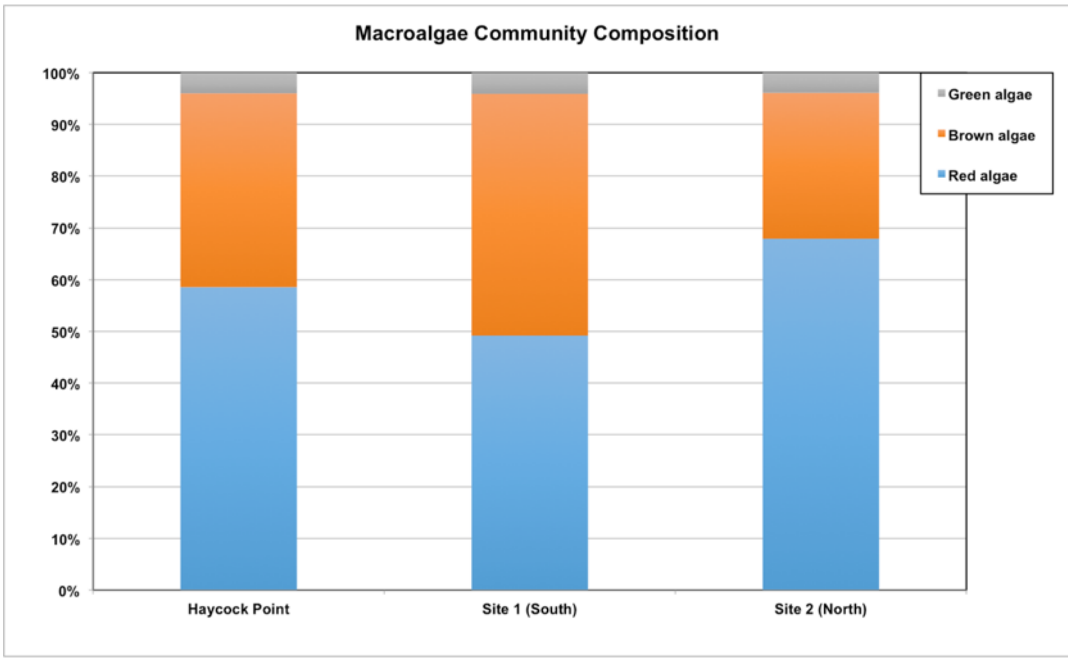



**Figure 8-12 Macroalgal community composition showing relative abundance of major algal groups**

### 8.3.4 Mid and High Shore Zone

Qualitative observations were made of the community composition of the mid and high shore zones at Haycock Point by walking over 50 m long transects and noting species present. These zones are characterised by an increased abundance of barnacles and gastropods with an overall reduced cover of algae represented by fewer algal species. A total of 23 taxa were recorded in the mid shore zone with the high shore typically species poor with only five (5) taxa recorded (**Appendix F-1**).

Five barnacles are observed in the mid to upper shore zone including *Catomerus polymerus, Tetraclitella purpurascens, Chamaesipho tasmanica, Chthamalus antennatus* and *Tesseropora rosea*. Only the latter was also observed in the low shore zone.

The algal community of the mid to high zone is represented by six taxa whose distribution is best described as patchy with bare rock the dominant substrate. The mid shore includes the encrusting brown alga *Ralfsia* sp., *Hormosira banksii, Splachnidium rugosum*, the foliose green alga *Ulva* sp. and filamentous green algae (comprising a mixture of *Cladophora*, *Rhizonclonium* and *Chaetomorpha*). The only alga found on the high shore is *Porphyra lucasii* that is well-adapted to living in highly exposed conditions. These algae are all typical of the mid to high shore zone in south eastern NSW. Of these algae, only *H. banksii, Ulva* sp. and filamentous green algae were also observed in the low shore zone where they represented minor components of that algal community.

The mid to high shore zone had higher gastropod species richness than the low shore with a total of twelve gastropods recorded (**Appendix F-1**). Three taxa present on the mid to high shore that were not observed in the low shore included *Nerita atramentosa, Bembicim nanum*, and *Austrocochlea concamerata.* In general, gastropod abundances appears to be higher in the mid to high shore zone compared to low shore zone although quantitative sampling was not undertaken to evaluate this trend.

## 8.4 Key Findings

Key findings from the assessment include:

- The intertidal community of Haycock Point is comprised of at least 45 taxa with higher diversity observed at lower shore heights (35 taxa) compared mid shore (23 taxa) and high shore (5 taxa). This pattern of diversity is typical for most intertidal shorelines.

- Quantitative sampling of the low shore community indicated that species composition and abundance was different between the southern and northern sides of Haycock Point. The low shore zone on the northern side was characterised by an increased cover of coralline algae with fewer gastropod species and overall lower abundances compared to the southern side that was characterised by less algal cover and an increased cover of cunjevoi and bare rock that provided habitat for a more diverse and abundant gastropod assemblage.
- The dominant gastropods of the low shore zone include *Scutellastra peronii, Cellana tramoserica, Patelloida latistrigata* and *Montfortula rugosa*. These taxa would be suitable indicator species for monitoring changes in community composition at the low shore zone if required.
- Green algal species that are known to respond rapidly to increased availability of nutrients such as the foliose *Ulva*, and filamentous species *Cladophora, Rhizoclonium* were rare in the low shore zone but prevalent in the mid shore zone although patchy in distribution.
- Limitations associated with the photo-quadrat method include counts of mobile invertebrates in highly complex habitats such as cunjevoi. Many gastropod species such as chitons prefer to inhabit the interstitial spaces provided by cunjevoi yet, these individuals are difficult to observe without considerable additional sampling time required. As such, the observed species richness is an estimation of total species richness given the limitations of approach and it is acknowledged that other species not yet observed are likely to exist in the low shore zone.

## 8.5 Potential Impacts of the Project on Intertidal Rocky Shore Community

### 8.5.1 Construction Phase Effects

Construction phase activities that have potential to cause negative effects to intertidal shorelines includes accidental spill of hazardous substances (i.e. fuels, oils and other construction vessel related fluids) from construction vessels and equipment while mobilising to site or during pipeline construction works. Water pollution resulting from vessel accidental spill would typically impact the water surface initially, so could potentially spread towards intertidal shorelines of Haycock Point and Long Point depending on prevailing weather conditions. Lower shore heights would most likely be affected by a spill event. The negative effects of a potential spill event would likely be short-term, and while loss of some taxa may occur, it is expected that it would not influence the overall recovery capacity of the habitats and communities.

It is expected that a Construction Environmental Management Plan (CEMP) for the Project would be developed to cover management of risks and controls for support vessels, including measures that would typically include procedures for storage and use of fuel, oil and hydraulic fluids. Without controls, the initial risk was assessed as medium, with this reduced to a low risk with inclusion of CEMP controls.

### 8.5.2 Operational Phase Effects

Potential effects from the operational phase of the Project considers how the discharge of treated wastewater released from a diffuser at 30 m depth may affect intertidal reef communities of Hayock Point and Long Point that are more than 2,000 m from the Project area. Dispersion modelling indicates water quality objectives would be achieved within a 25 m mixing zone under typical conditions and 200 m under worse-case conditions, with potential effects from exposure to dilute wastewater most likely to be detected in ecological receptors occurring within the 25 m near field mixing zone. As intertidal shorelines of Haycock Point and Long Point are more than 2,000 m from the mixing zone, the likelihood that treated wastewater would reach these areas at concentrations that may cause adverse effects is considered rare and the consequence insignificant. There would be minimal risk to intertidal reef communities by the dispersing treated wastewater plume.

## 8.6 Conclusion

The continuation of monitoring the intertidal rocky shore community in Stage 2 was contingent on the selection of the preferred pipeline alignment and diffuser option. As the Project adopted the North-Short location as the preferred option, and dispersion modelling indicates the treated wastewater would rapidly dilute and meet MWQOs within a 25 m mixing zone under typical conditions and 200 m under worse-case conditions.It was considered rare that intertidal shorelines would be affected by the dispersing treated wastewater plume based on distance from the Project area. Intertidal shorelines of Haycock Point are approximately 2,000 m to the south and Long Point approximately 2,300 m to the north. The risk to intertidal rocky shore communities from the Project operational and construction phases is assessed as minimal to low with no further need to monitor intertidal rocky shore communities.