Appendix G-7:

Survey for Juvenile Prawn Recruitment at Mullaquana Station

Fisheries

Northern Water Project: Baseline survey results of juvenile prawn recruitment at Mullaquana Station, South Australia



Hooper, G.E., Carmena-Wood, J., Catalano, S., Heldt, K. and Cipriani, V.

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August 2026

Final Report to the Department for Infrastructure and Transport

Government of South Australia
Department of Primary Industries and Regions

SARDI
SOUTH AUSTRALIAN RESEARCH AND DEVELOPMENT INSTITUTE

Northern Water Project: Baseline survey results of juvenile prawn recruitment at Mullaquana Station, South Australia

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We recognise and own a difficult past, and together we will walk forward.

We acknowledge the many Aboriginal people of this Country as the oldest continuous living culture in the world.

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EXECUTIVE SUMMARY

The Northern Water Project (the Project) involves the construction and operation of a 51 GL/year reverse osmosis seawater desalination plant located at Mullaquana Station in the Upper Spencer Gulf, 20 km south of Whyalla. The environmental impact assessment for the Project requires baseline surveys of the macrofauna species assemblage on sand and mudflats adjacent to the Project site. This report presents the results of surveys to quantify the abundance of post-larval and juvenile prawns and other species of epibenthic macrofauna adjacent the proposed marine infrastructure.

Jet-net sampling using a small vessel was undertaken at three sites in February and April in 2025 and 2026, resulting in a total of 48 benthic samples. Samples were sorted in the laboratory, and all macrofauna identified to the lowest possible taxonomic level. Prawns were identified to species using both morphological and DNA metabarcoding approaches and indicated the potential occurrence of three species: *Penaeus (Melicertus) 1* (M1)*, Penaeus (Melicertus) 2* (M2), *and Metapenaeopsis lindae*.

Spatial patterns in juvenile prawn abundance in this study indicated that the area overlaps with juvenile prawn habitat. Post larval and juvenile prawn abundance showed marked variation among years, months, and sites that was consistent with previous studies. Prawn abundance was higher in 2026 than 2025, particularly for *Melicertus* species, indicating stronger recruitment to survey sites in 2026. Historical comparison of 2025 and 2026 data to other sites sampled in April across Spencer Gulf from 1993 to 2019 indicated that prawn abundance at Mullaquana Station (0.52 prawns/m^2) was comparable to sites previously surveyed.

Other macrofauna identified in surveys included fish (gobies), annelids (worms) and crustaceans (isopods, amphipods). Abundances were generally higher in April than February in both 2025 and 2026, although species composition varied among surveys. No commercially harvested fish species were recorded, but juvenile Blue Swimmer Crabs (*Portunus armatus*) were present.

An ongoing, harmful algal bloom (HAB) of *Karenia* spp. has occurred throughout much of the South Australian gulf systems since March 2025. Water sampling indicated that the HAB was present near Mullaquana Station in December 2025 and January 2026 (Algal Bloom update, 2026). Mortalities of finfish, elasmobranchs, cephalopods and crustaceans recorded elsewhere in Spencer Gulf (iNaturalist, 2026) may indicate regional impacts on benthic intertidal communities that should be considered when interpreting results of this study.

Keywords: Desalination plant, Mullaquana Station, Spencer Gulf, juvenile prawns.

1. INTRODUCTION

1.1. Project background

The Northern Water Project (the Project) aims to provide a secure, climate-resilient water source for industry located in South Australia's Upper Spencer Gulf and Far North (Northern Water, 2026). This will enable the growth of industries and key mining resources crucial for contributing to decarbonisation initiatives, sustainability and unlocking long-term economic prosperity for South Australia.

Delivery of the Project will involve the construction and operation of a 51 GL/year seawater reverse osmosis desalination plant located at Mullaquana Station in the Upper Spencer Gulf, and associated water transfer system (including marine infrastructure, transfer pump stations, and water storages) and electricity infrastructure. Mullaquana Station is located approximately 20 km south of Whyalla.

To support environmental approvals for the Project, a baseline survey was conducted to characterise the assemblage of decapod crustaceans (e.g. prawns) in the area around the proposed site.

In South Australia, there are three commercial prawn fisheries that exclusively target Western King Prawn (*Penaeus (Melicertus) latisulcatus*): 1) Spencer Gulf Prawn Fishery (SGPF), 2) Gulf St Vincent Prawn Fishery (GSVPF) and 3) West Coast Prawn Fishery (WCPF). The SGPF is the largest in terms of area (22,367 km^2), licence holders (39) and annual total catch (~1,800 t) (Heldt *et al*. 2025).

The SGPF and GSVPF are reliant on inshore nursery habitats in the upper regions of the gulfs for successful recruitment and therefore sustainability of the resource (Kangas & Jackson 1998; McLeay *et al.* 2016). Juvenile prawns predominantly occur on intertidal sand and mudflats, among seagrass beds and mangroves (Kangas 1999; Tanner & Deakin 2001). Given the reliance on recruitment from northern nursery habitats, understanding patterns of juvenile abundance and distribution is important when assessing potential environmental pressures on this resource. This is particularly relevant in the context of the proposed desalination plant, where seawater intake and brine discharge may influence early life stages and benthic habitats (Panagopoulos and Haralambous, 2020).

The life cycle of Western King Prawn consists of an offshore phase, where spawned, pelagic larvae drift and grow, and an inshore phase, where settlement of post larvae and growth through to the juvenile stage occurs. Post larvae settle in inshore nursery areas at 2-3 mm carapace length from December (early spawning) through to March (late spawning).

Juvenile prawn (i.e. post larval to juvenile) surveys were periodically conducted in Spencer Gulf between 1992 and 2003, (Carrick 1996; Roberts *et al*. 2005, 2010) and again at selected locations from 2009 to 2019 (Noell 2019). In Gulf St Vincent, surveys were undertaken between 1990 and 1996, and again in 2009 (Kangas 1999; Roberts *et al*. 2009). A modified "jet net", an epibenthic sled fitted with a water pump was towed behind a small vessel within the intertidal zone during high tide. In Spencer Gulf, key nursery sites sampled historically were False Bay, Shoalwater Pt, Plank Pt, Mt Young, South Cowleds, the Spit and 5th Creek (Fig. 1), with peak abundances in April, followed by March, February and May (Roberts *et al.* 2005).

Historical surveys sampled key nursery areas in both gulfs, and through morphological species identification, collected information on juvenile prawn abundance and size at various spatial and temporal scales (Roberts *et al*. 2005). Whilst these surveys showed that juvenile prawn abundance differed between years and key nursery sites, indicating recruitment is highly variable and unpredictable, inconsistent sampling (spatially and temporally) limit interpretation.

This project surveyed sites adjacent to Mullaquana Station that are located within key juvenile prawn nursery habitat in Spencer Gulf (e.g., False Bay and Mt. Young; Roberts *et al*. 2010; McLeay *et al.* 2016) and compared contemporary abundance data with historical records. Sites were selected based on proximity to major infrastructure works and were located central to, north and south of the proposed intake/outflow pipes. Data collected in 2025 and 2026 provide a baseline to inform the Project Environmental Impact Statement (EIS) and for comparison with historical data (i.e. 1992 – 2003 and 2011-2019) from other locations in Spencer Gulf (i.e. False Bay, Plank Pt, Mt Young, the Spit and 5th Creek – Figure 1). Specifically, this report provides baseline information on the abundance of juvenile prawns and other macrofauna collected from sites adjacent to Mullaquana Station over four surveys conducted in February and April 2025 and February and April 2026.



Figure 1. Key Juvenile prawn nursery sites sampled in Spencer Gulf, South Australia.

1.2. Objectives

The objectives of this project were to:

- Estimate the abundance of juvenile prawns adjacent to Mullaquana Station, Spencer Gulf at sites located north, south and central to the proposed marine infrastructure.
- Compare abundance of juvenile prawns collected in 2025 and 2026 near Mullaquana Station to abundance values from other areas in Spencer Gulf sampled between 1993-2009 and 2011-2019.
- To provide a count and species list of other macrofauna collected during sampling.

2. METHODS

2.1. Study area and survey design

Prawn recruitment surveys were conducted on 26/27 February 2025, 27/28 April 2025, 16/17 February 2026 and 15/16 April 2026 adjacent to Mullaquana Station, and near the proposed intake and outfall structures of the proposed desalination plant (Figure 2). Surveys were conducted during historical surveys when higher abundances of juvenile Western King Prawn were typically observed (Carrick 1996; Roberts 2005, 2010).

Sampling at each of the three sites (North, Central, South) comprised four 100-m transects spaced 50 m apart and perpendicular to the shore (Figure 2). Each site was spaced approximately 500 m apart (GPS location in Appendix 6.1).

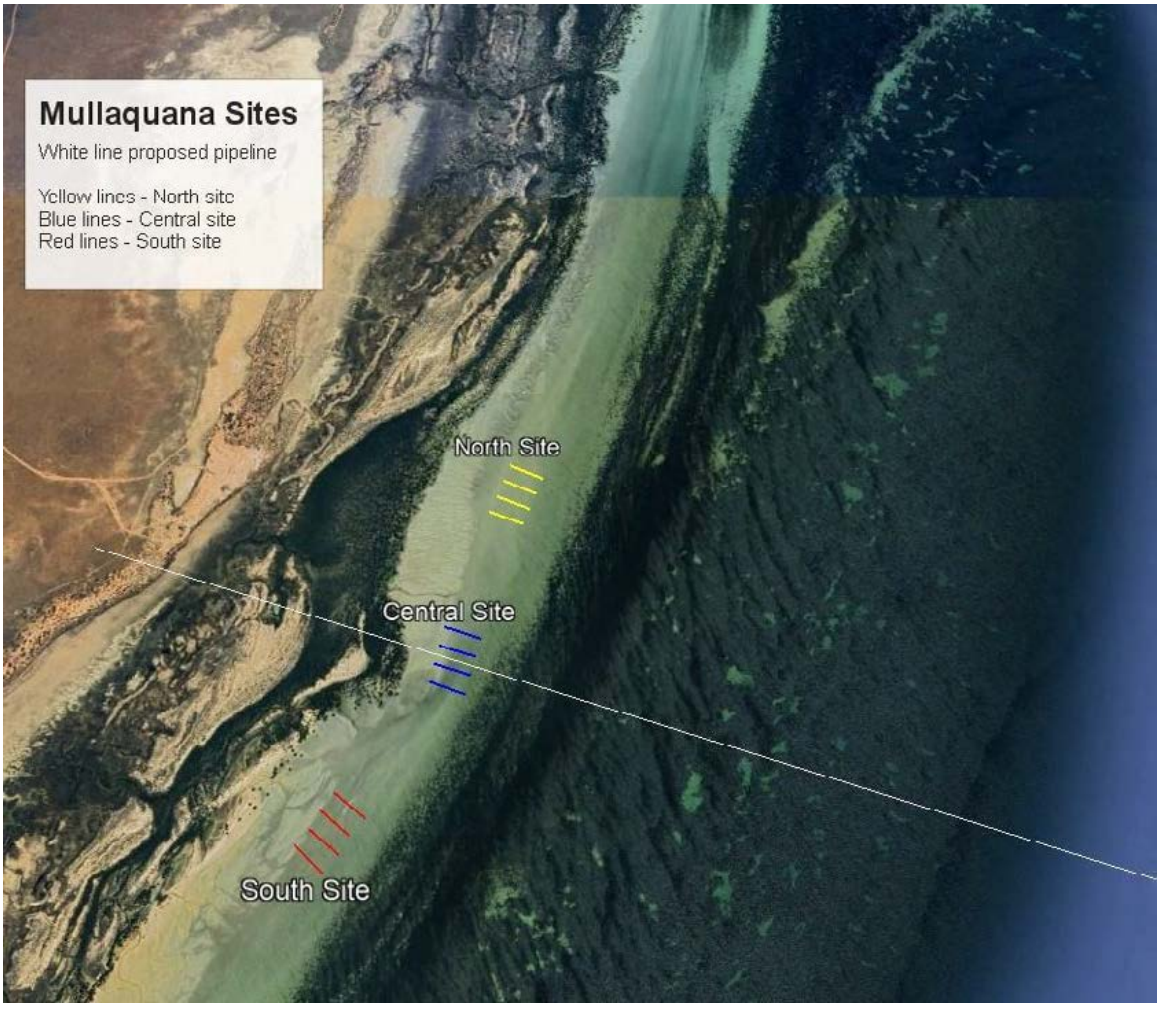


Figure 2. Map of the survey sites located near the planned sub-surface intake and outfall (white line) structure for the Project at Mullaquana Station on the eastern coastline of the Eyre Peninsula in Upper Spencer Gulf.

All surveys were conducted in suitable weather conditions, during daylight hours and at highest monthly tides at depths of 0.9–1.4 m (based on Bom Tide predictions: 2.16 to 2.80 m for Whyalla, Table 1).

Table 1. Tide predictions for Whyalla for February and April 2025 and February and April 2026. (BOM, 2026).

February 2025	**Time**	**Depth (m)**	**April 2025**	**Time**	**Depth (m)**
Wednesday 26th	1.54	1.69	Sunday 27th	1.13	0.63
	7.05	2.56		7.02	2.31
	14.40	0.22		12.45	0.95
	21.11	2.22		18.41	2.87
Thursday 27th	2.13	1.39	Monday 28th	1.46	0.46
	7.44	2.77		7.34	2.16
	15.01	0.19		12.54	1.03
	21.15	2.25		18.56	3.06
February 2026	**Time**	**Depth (m)**	**April 2026**	**Time**	**Depth (m)**
Monday 16th	1.59	1.52	Wednesday 15th	0.32	1.17
	7.29	2.65		6.03	2.34
	14.56	0.31		12.30	0.71
	21.13	2.18		18.36	2.52
Tuesday 17th	2.20	1.27	Thursday 16th	0.57	0.87
	8.00	2.80		6.37	2.42
	15.15	0.30		12.48	0.74
	21.20	2.24		18.46	2.69

2.2. Recruitment Surveys Field Sampling

Juvenile prawn samples were collected at each site using a jet net as previously used in recruitment surveys in the intertidal zone (Figure 3) (Kangas & Jackson 1998; Roberts *et al.* 2005, 2010, Noell 2019). The jet net frame has an internal path width of 0.9 m. The water jet penetrates the sand/mud substrate to a depth of approximately 5 cm which enables displacement and collection of prawn and macrofauna. The mesh net is made from 2 mm nylon (nytal) square mesh fitted to cover the entire frame, with a codend length from the frame measuring 3.2 m long. The codend is tied off with 5 mm rope, approximately 500 mm in length.

Prior to sampling, two surface buoys were deployed 100 m apart, each held down by 10 kg blocks, with a string line attached at each end to accurately measure the towed distance of 100 m (alongside use of GPS navigational equipment). The jet net was deployed from SARDI RV *Rakali* (~6 m vessel, Yamaha 90hp engine) at the start surface buoy. The water pump was

engaged before the jet net arrived at the start buoy, approximately 30 m from the vessel and towing began. The net was towed at approximately 1 m/s at an engine speed of 1200-1300 revs/minute. Once the net reached the end surface buoy, the water pump was turned off and the net was retrieved back onto the vessel. The frame of the net was lifted from the water along with the cod-end, and the contents of the net were washed down with a seawater deck hose into a 22 L bucket. Samples were then briefly sorted to remove and return to water any larger shells, seagrass, and identifiable macrofauna (after being counted). The remaining contents were placed into a 4.5 L jar and preserved in seawater and 40% formalin at a ratio of 10:1. In April 2025, an additional transect was completed at each site, to obtain a sub-sample of juvenile prawns, which were preserved in ethanol for morphological identification and DNA analyses.



Figure 3. Example of the recruitment survey net (jet-net) deployed adjacent to Mullaquana Station. Left = net, frame and cod end. Right = water pump used to transfer water to jet net through 40 m long x 30 mm wide hose.

2.3. Laboratory sample processing and species identification

In the laboratory, formalin was drained from the sample and discarded, and each sample was washed thoroughly with water to remove any residual formalin. Smaller quantities of each sample were then removed and placed into a sorting tray to be carefully sorted. Crustaceans (e.g. prawns, crabs, amphipods, isopods, mysids) were manually separated from other macrofauna species in each sample. For quality assurance each sample was sorted by two personnel before the unwanted contents were discarded. Retained specimens for morphological analysis were preserved in 100% ethanol. Accurate counts of prawn and macrofauna were obtained as follows:

Juvenile prawns:

1) Prawns were separated from other crustaceans. Carapace length (CL) of prawns was visually estimated, and prawns were separated into juvenile (i.e. CL > 3mm) and post-larval (CL < 3mm) categories. Post-larval prawns were retained but not counted or analysed.
2) Each juvenile prawn was identified using key morphological features (Appendix 6.3), grouped into species and placed into a separate vial (i.e. one species per vial). Morphological identification was independently validated by a second person.
3) If a prawn was damaged during sampling, the tail was used for morphological identification, and it was counted as an 'individual'. Heads were used for morphological identification where possible but not included in final counts to avoid counting the same individual twice. A small number of prawns were damaged and not identifiable (<4%).
4) The number of juveniles of each species for each transect was then counted using a handheld tally counter and recorded.

Macrofauna:

1) Remaining macrofauna were sorted to the lowest reliable Operational Taxonomic Unit (OTU) at SARDI using key reference guides. If the individual was not identified to species, the next taxonomic level possible was recorded (e.g. genus, family, order). Although the focus of jet-net sampling was on epibenthic species, infaunal species (e.g. worms) were also counted.
2) The number of each species/taxonomic groups was counted using a handheld tally counter and recorded for each transect.

2.4. DNA-barcoding Techniques

To confirm morphological identification of prawn species, modern DNA-barcoding techniques were utilised. DNA-barcoding techniques are used in a hybrid process with morphological identification to firstly verify the identifications and, where possible, provide a finer level of taxonomic resolution. For instance, an Operational Taxonomic Unit (OTU) identified only to family level by morphology could potentially be resolved to genus or species level using DNA barcoding. In DNA-barcoding, the DNA sequence of an individual (single sample) for a distinguishing gene region (e.g. the hypervariable region of COI metazoans) is identified and taxonomically resolved by matching to sequences deposited in publicly available reference databases, such as the Barcode of Life (BOLD) and National Centre of Biotechnology Information (NCBI).

Prawn samples stored in ethanol were provided to the SARDI Molecular Sciences Laboratory for DNA-barcoding. Prior to commencing DNA extractions, the preparation bench was cleaned using 70% ethanol, and dissecting equipment was sterilised with LookOut® DNA Erase (Sigma), water wash and 70% ethanol. The whole individual prawn was removed from the original sample tube, and a sub-sample was taken from the centre of the individual (~3 mm^2). This sample was blotted dry on paper towel to remove residual ethanol and placed into a new 1.5 mL tube. The remaining part of the individual was returned to the original sample tube.

Dissecting equipment was sterilised in-between the processing of each individual as described above, and extractions were performed using the DNeasy® Blood and Tissue Kit (Qiagen) following the manufacturer's instructions for animal tissue (spin-column protocol). Samples were eluted in 75 µl Buffer AE, which was passed through the spin column twice. All samples were quantified using the NanoDrop One spectrophotometer (ThermoFisher Scientific).

After DNA extractions, polymerase chain reaction (PCR) amplification and gel electrophoresis was completed. DNA extracts were PCR amplified over 35 cycles using cytochrome c oxidase subunit (COI) primers (4x primer mix with M13 tags) which target a ~650 base pair fragment of the COI gene. The PCR reaction included positive (Common galaxias) and negative (no template – water) controls. Samples were visualised via gel electrophoresis, with 14/16 extracts positive, 1/16 extracts weak positive and 1/16 extracts negative for amplification.

Sanger sequencing (barcoding) followed, and PCR amplified products (n = 15) were sent to the Australian Genome Research Facility (AGRF) for sequencing in the forward and reverse direction. Sequences were edited and primers removed using Geneious Prime® 2023.2.1. Out of the 15 amplicons sequenced, consensus sequences were made for 12 (as three amplicons failed to produce readable chromatograms). Western King Prawn sequences deposited in publicly available databases (e.g. Genbank/BOLD) were included in the final alignment (*Penaeidae* sp. n = 4 Egypt/India, *Melicertus* sp. n = 12 Australia), along with *Melicertus* OTUs from the Northern Waters Supply Project (n = 3) and sequences from adult *Metapenaeopsis lindae* (Linda's velvet prawn) individuals (n = 3) that were morphologically identified and barcoded at the South Australian Museum. The final alignment was trimmed to 310 bp to match with reference sequences.

DNA barcoding was not repeated in 2026 given most of the genetic results from 2025 were aligned with morphological identification at the genus level. The 2025 DNA barcoding validated and facilitated development of morphological identification methods and a prawn species identification key.

2.5. Data Analyses

All datasets were entered in Microsoft Excel and checked for consistency and accuracy. Analyses were then carried out in *R* programming environment (version 4.2.2: R core team 2024).

Relative abundance of juvenile prawns was estimated as the number of prawns/m^2, which was calculated from the distance trawled (100m) and width of the jet net (0.9m). Comparisons among sites, months and years were conducted, where appropriate, using analysis of variance (ANOVA). Prior to analysis, relative abundance data were log-transformed ($\log_{10}$ (x + 1)) to improve normality and homogeneity of variance, which were assessed using Shapiro–Wilk and Levene's tests, respectively. Tukey's post-hoc tests were used to determine significant differences at spatial and temporal scales

Historical data used to compare annual changes was sourced from 1) PIRSA Fisheries and Aquaculture Information Services prawn databases and 2) SARDI stock assessment reports. Relative abundance from 2025 and 2026 surveys was compared with historical data, restricting analyses to April to ensure seasonal comparability. In addition, comparisons were made with the nearest sampled sites from 1993–2009 at False Bay, Mt Young, The Spit, 5th Creek and Port Pirie (Figure 3) and from 2009–2019 at False Bay and Port Pirie.

Macrofauna data is presented as a percentage of the total abundance for each sampling period to compare the differences between assemblages expressed in proportional plots.

3. RESULTS

3.1. Environmental conditions

Daily weather conditions (BOM, 2025/2026) for Whyalla during February 2025 (Table 2), April 2025 (Table 3), February 2026 (Table 4) and April 2026 (Table 5) were optimal for towing the juvenile prawn sampling equipment.

Table 2. Daily observations from the Bureau of Meteorology website (February 2025) for Whyalla.

Date	Day	Temperature			Observations at 9 am				
		Min	Max	Rain	Temperature	Relative Humidity	Wind		MSLP
		°C	°C	mm	°C	%	Dir	Speed (km/h)	hPa
26	We	14.7	30.3	0	23.9	56	SSW	15	1018.4
27	Th	20.3	31.5	0	22.1	79	SSW	13	1017.5

Table 3. Daily observations from the Bureau of Meteorology website (April 2025) for Whyalla.

Date	Day	Temperature			Observations at 9 am				
		Min	Max	Rain	Temperature	Relative Humidity	Wind		MSLP
		°C	°C	mm	°C	%	Dir	Speed (km/h)	hPa
27	Su	15.2	20.2	1.6	17.0	66	S	19	1021.3
28	Mo	10.3	20.8	0	17.3	67	SSE	9	1021.7

Table 4. Daily observations from the Bureau of Meteorology website (February 2026) for Whyalla.

Date	Day	Temperature			Observations at 9 am				
		Min	Max	Rain	Temperature	Relative Humidity	Wind		MSLP
		°C	°C	mm	°C	%	Dir	Speed (km/h)	hPa
16	Mo	23.2	37.9	0	29.6	44	NNE	19	1008.2
17	Tu	28.4	32.5	0	31.1	44	NNW	33	1005

Table 5. Daily observations from the Bureau of Meteorology website (April 2026) for Whyalla.

Date	Day	Temperature			Observations at 9 am				
		Min	Max	Rain	Temperature	Relative Humidity	Wind		MSLP
		°C	°C	mm	°C	%	Dir	Speed (km/h)	hPa
15	We	8.5	23.9	0	18.4	72	CALM	-	1022.4
16	Th	10.3	29.7	0	21.5	45	NNW	13	1018.5

3.2. On site observations

Whilst undertaking the sampling at Mullaquana Station, SARDI staff observed some other activities that potentially impacted recruitment in the sampled intertidal sand/mud flats. These included:

- Interactions with commercial net fishers – no fishing gear was deployed during this time, but this intertidal area is known for marine scalefish haul netting.
- Geotechnical Barge - a substrate core drilling barge was onsite during the sampling period near the proposed marine infrastructure, which was directly on top of central site 2 and 3. Drilling occurred in between the survey in February and late March 2026, taking place for 10-12 hours per day for approximately 5 days.

3.3. Summary of samples collected

During the four surveys over the two-year period, a total of 48 samples were collected, with four samples collected at each of the three sites (i.e. north, central, south) each month (February and April) (Table 6).

Overall, 5,546 macrofauna specimens were sorted into the following categories: prawns, crabs, other crustaceans (i.e. isopods, amphipods, and shrimp), fish, annelid and molluscs. Sorting and quantification of juvenile prawn samples resulted in a total of 2,252 prawns (including whole, part and damaged individuals). Of the samples, 96% were whole individuals (and some recognisable tail parts) and 4% were unidentifiable. Prawn heads and unidentified parts were excluded. A complete list of identified macrofauna taxa can be found in Appendix 6.4.

Table 6. Summary of samples collected, total number of prawns, crustaceans, fish, crab and other macrofauna caught during February and April 2025, February and April 2026 surveys at the proposed desalination site off Mullaquana Station.

	Site	Number of samples	Prawns	Other crustaceans	Fish	Crab	Other macrofauna
Feb-25	North	4	61	39	117	15	45
	Central	4	112	34	65	24	32
	South	4	96	1	70	14	120
Apr-25	North	4	104	124	209	9	310
	Central	4	122	112	268	17	179
	South	4	77	102	308	11	192
Feb-26	North	4	94	12	8	8	12
	Central	4	198	11	16	11	5
	South	4	133	19	14	6	4
Apr-26	North	4	395	40	174	4	38
	Central	4	240	65	232	12	19
	South	4	620	2	144	11	3
	Total	**48**	**2252**	**561**	**1625**	**142**	**966**

3.4. DNA-barcoding

DNA-barcoding sequence data identified three possible prawn species – one *Metapenaeopsis lindae* and two *Penaeus (Melicertus)* spp. (noting that Western King Prawn belongs to this genus). From the literature and published research reports (e.g. Heldt *et al.* 2025), the only species of king prawn distributed in SA waters is Western King Prawn, however our preliminary DNA result suggests there are two king prawn groups, populations, or possibly even species, in SA waters. One is likely Western King Prawn, and the second may be a variant, or Eastern King Prawn. To resolve this, further sequence data is needed, including sequencing other gene regions, along with king prawns from across their distribution in Australian waters. By adding in king prawn sequences from WA, where only Western King Prawn is expected to occur, and Tasmania and NSW, where only Eastern King Prawn is expected to occur, the two clades we have observed from SA may be able to be resolved.

Although some researchers have used DNA sequence data to identify king prawn species, these studies have been restricted to king prawns collected from overseas locations (e.g. Abbas *et al.* 2018, Yang *et al.* 2023; Egypt, India and Taiwan) and do not include any Australian samples. Furthermore, the sequences from overseas king prawns, when imported and aligned with our two SA *Penaeus (Melicertus)* spp., are completely distinct and sit within their own clade in a preliminary phylogenetic tree.

For the SA samples tentative *Penaeus (Melicertus)* sp. 1 (e.g. C8, Appendix 6.2) and tentative *Penaeus (Melicertus)* sp. 2 (e.g. N6, Appendix 6.2) differ by 10bp over the 310bp COI fragment

(96.8%), with typically ≥ 99% sequencing similarity used to classify to species level and 95 – 99% sequencing similarity used to classify to genus level. Out of the 15 juvenile prawn individuals that produced positive amplicons, sequence data identified four as *Metapenaeopsis lindae* (Linda's velvet prawn), seven as *Penaeus (Melicertus) sp.* 1, and one as *Penaeus (Melicertus) sp.* 2 (Appendix 6.2). Three samples were unable to be aligned due to poor sequence data.

Importantly, most of the genetic results aligned with, and therefore validated morphological identification at the genus level.

3.5. Juvenile Prawn Abundance

To align with morphological and genetic identification, prawns were categorised as:

1) *Melicertus* 1 (M1),
2) *Melicertus* 2 (M2)
3) *Melicertus* unidentifiable (MU) and
4) *Metapenaeopsis* sp.

The mean ± SE (standard error) of each species per month was determined (Table 7).

Table 7. Monthly average (individuals per transect) of prawns collected at the proposed desalination plant sites during recruitment surveys conducted in February and April 2025 and February and April 2026, off Mullaquana Station, SA.

	February 2025			**April 2025**			**February 2026**			**April 2026**		
Prawn Species	Mean	SE	n	Mean	SE	n	Mean	SE	n	Mean	SE	n
Melicertus 1 (M1)	10.08	1.85	12	12.92	1.45	12	18.42	3.43	12	51.5	8.93	12
Melicertus 2 (M2)	3.42	0.94	12	1.75	0.6	12	13.25	3.05	12	28.58	8.32	12
Melicertus U (MU)	0.67	0.36	12	0.58	0.22	12	0.08	0.08	12	5.5	1.18	12
***Metapenaeopsis* sp.**	6.67	2.33	12	9.17	1.81	12	1.25	0.71	12	16.42	3.93	12

Relative prawn abundance (prawns/ m²) varied among prawn categories and sampling periods (Figure 4).

For *M1*, relative abundance differed significantly between years and months, with a significant Year × Month interaction (ANOVA, $p < 0.05$). Post-hoc comparisons showed significantly higher abundance in 2026 compared to 2025 in both February and April, and higher abundance in April relative to February in 2026 ($p < 0.05$). No significant differences among sites were detected.

For *M2* relative abundance varied significantly with year and site, with significant interactions among Year, Month, and Site ($p < 0.05$). Post-hoc comparisons confirmed that spatial differences varied across sampling periods, with no site differences in April 2025 ($p > 0.05$), higher abundance at the south site in February 2025, and the central site in February 2026; and significant differences among all sites in April 2026, with highest abundance at the south site and lowest at the central site.

For *Metapenaeopsis*, relative abundance differed significantly among sites and months, with a significant Year × Month interaction ($p < 0.001$), but no overall year effect. Post-hoc comparisons showed significantly lower abundance at the south site compared to central and north sites ($p < 0.001$ for both), with no difference between central and north. Abundance was significantly higher in April compared to February in both 2025 ($p = 0.0170$) and 2026 ($p < 0.001$). MU being unidentified prawns were excluded from this analysis.

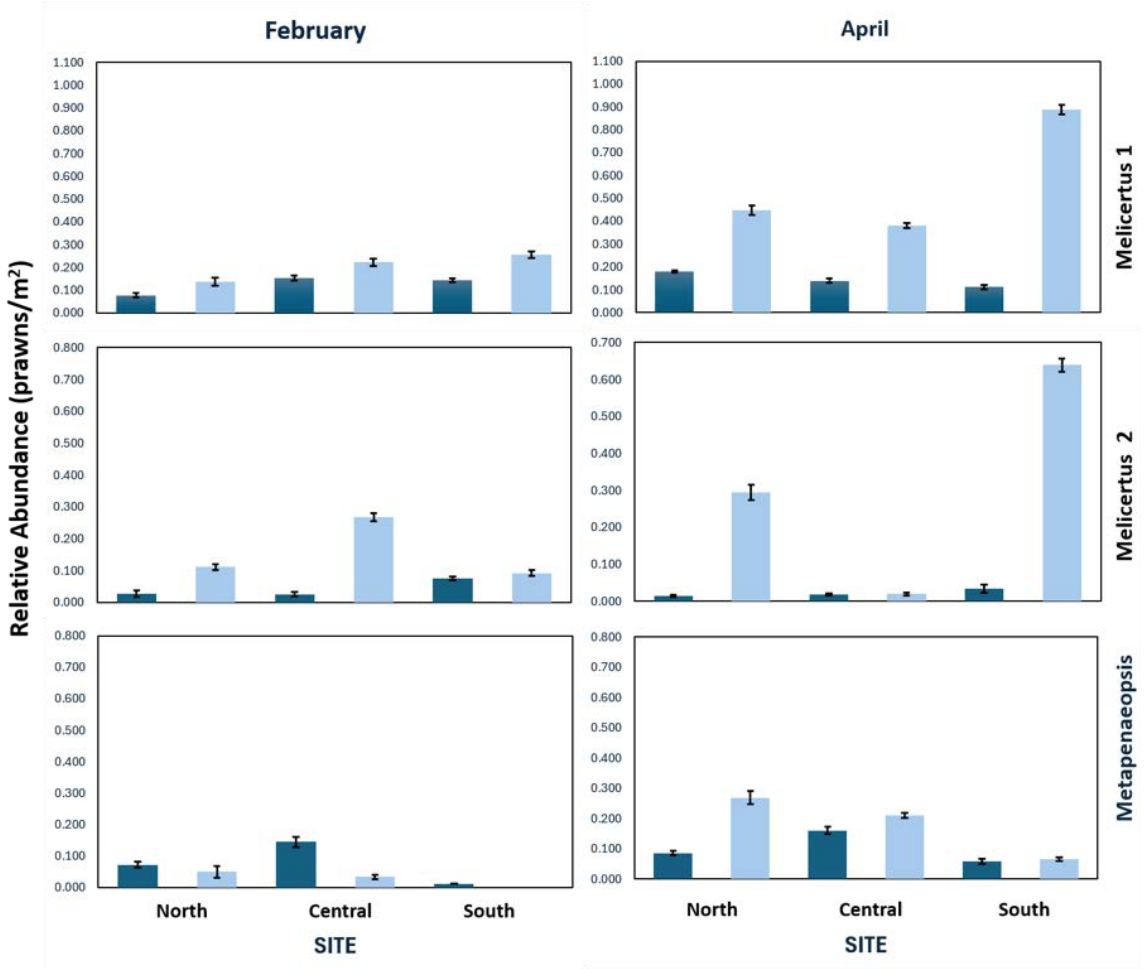


Figure 4. Relative abundance (prawns/m^2) of each prawn grouping and site in February and April 2025 (dark blue) and 2026 (light blue) off Mullaquana Station, SA. Error bars: standard error.

3.6. Temporal comparison of juvenile prawn abundance

Juvenile prawn sampling has been undertaken on an inconsistent basis from 1992 with up to 32 sites sampled in Spencer Gulf form Tumby Bay to Blanche Harbour (western side) and Chainman's Creek to Reef Point at Port Victoria (eastern side). The two sites closest to Mullaquana Station, Mt Young and The Spit (Figure 1), were sampled from 1993 through to 2009. The site at False Bay, (north of Whyalla) was consistently sampled through all sampling periods.

3.6.1 Trends in Juvenile prawn abundance in Upper Spencer Gulf 1993 – 2009

Juvenile prawn sampling was undertaken from 1993 to 2019 at five consistent sites in northern Spencer Gulf (Carrick 1996, 2003, Roberts *et al.* 2009). Relative abundance of juvenile prawns significantly differed among sites and years and found to be highest at False Bay during all years, while the remaining sites recorded inter-annual variations in juvenile abundance in all years. In 2000, the mean annual relative abundance peaked at all sites with 11.79 prawns /m^2 recorded at False Bay, 7.66 prawns/m^2 at Mt Young and 2.34 prawns/m^2 at the Spit, which are the closest sites to Mullaquana Station (Appendix 6.5).

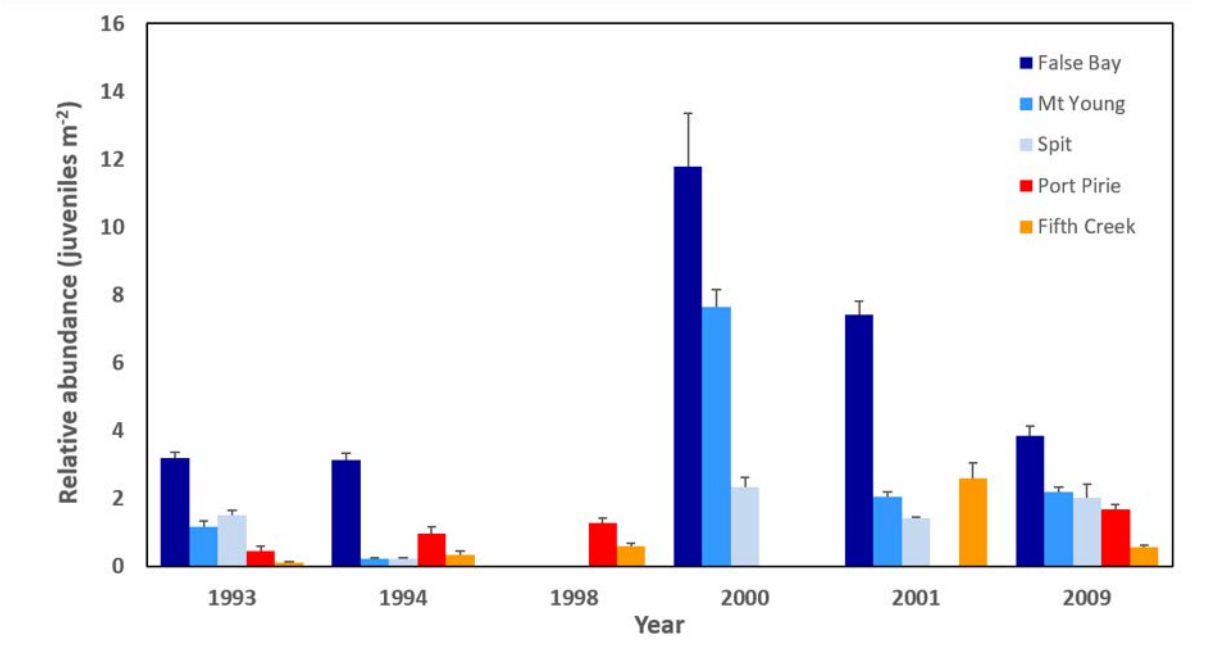


Figure 5. Historical mean (±se) (1993-2009) juvenile prawn abundance (prawns/m^2) in Spencer Gulf (April) for five key nursery sites (Roberts *et al.* 2009).

3.6.2 Trends in Juvenile prawn abundance in Upper Spencer Gulf 2011 – 2019

From 2011, juvenile prawn sampling was reduced to two sites, one on each side of northern Spencer Gulf; False Bay (western side) and Port Pirie (eastern side) (Figure 1). False Bay

recorded relatively higher abundance of juvenile prawns than Port Pirie (average difference = 41%; Figure 6, Appendix 6.5). In 2011, juvenile prawn abundance peaked at 1.44 prawns/m^2 in Port Pirie, and in 2014 False Bay juvenile prawn abundance peaked at 3.47 prawns/m^2. However, these estimates were below the historical peaks recorded at Port Pirie in 2009 (1.69 prawns/m^2) and at False Bay in 2000 (11.79 prawns/m^2), representing declines of approximately 15% and 71%, respectively.

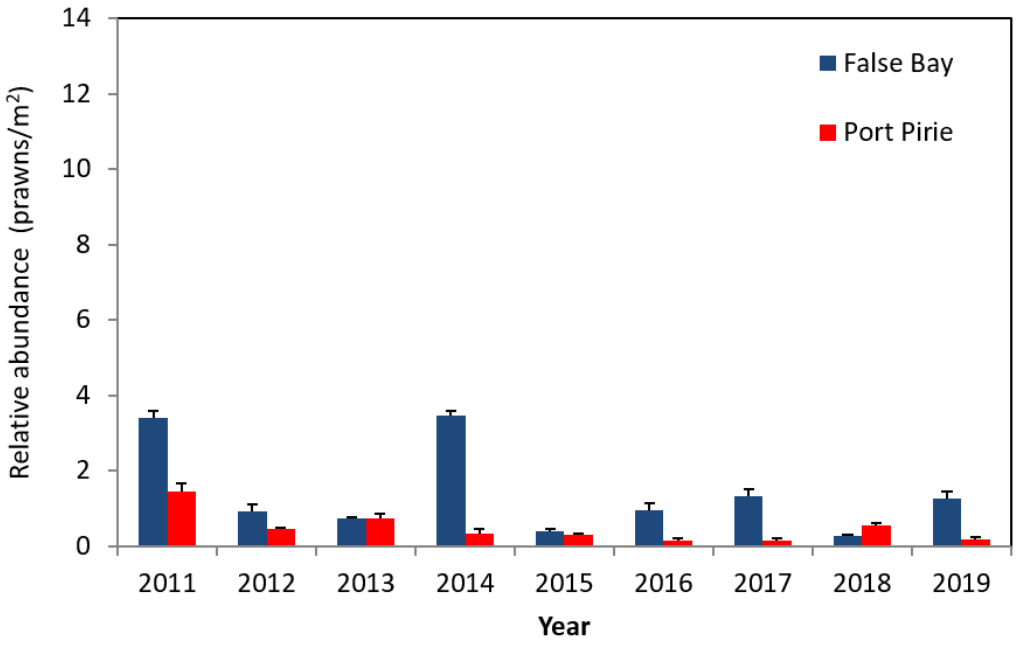


Figure 6. Historical mean (± SE) juvenile prawn abundance (prawns/m^2) in Spencer Gulf for two key nursery sites at False Bay and Port Pirie between 2011-2019. Note only 2013, 2014, 2016, 2017 and 2019 were April surveys, with 2012, 2015, 2018 sampling occurred early May due to weather or tides.

3.6.3 Trends in Juvenile prawn abundance at Mullaquana Station in 2025 and 2026

Relative abundance of juvenile prawns (M1 and M2 combined) at Mullaquana Station differed significantly among years, months, and sites, with all main effects and interactions significant (three-way ANOVA on log-transformed data; Year: $F_{1,36}$ = 68.56, p < 0.001; Month: $F_{1,36}$ = 21.60, p < 0.001; Site: $F_{2,36}$ = 6.83, p = 0.003; Year × Month × Site: $F_{2,36}$ = 9.28, p < 0.001, Appendix 6.6).

Overall, abundance was higher in 2026 than in 2025 and greater in April compared to February (Appendix 6.5). The south site consistently supported higher abundance than central and north sites. A significant increase from February to April was observed in 2026 but not in 2025, with the highest abundance recorded at the south site in April 2026, while the central site showed comparatively lower abundance during this period.

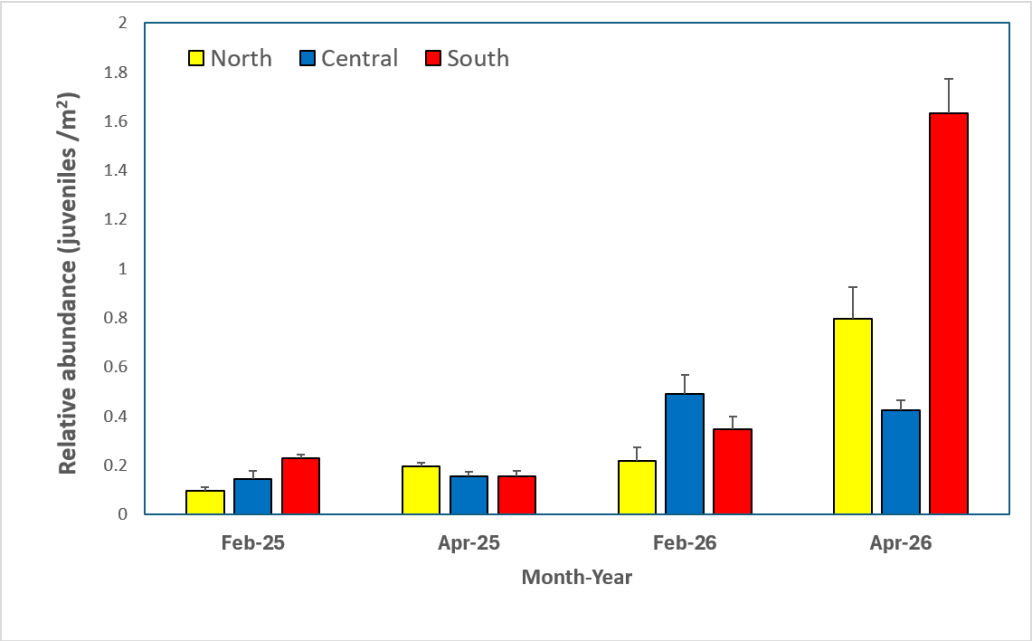


Figure 7. Mean (± SE) juvenile prawn abundance (prawns/m^2) at Mullaquana Station sites in February and April 2025 and 2026.

3.6.4 Trends in Juvenile prawn abundance at Mullaquana Station compared to other Spencer Gulf sites

Relative abundance of juvenile prawns at Mullaquana Station in April 2025–2026 (mean = 0.56 ± 0.12 SE prawns/m²) was below the historical average of 1.69 ± 0.21 prawns/m² recorded across Spencer Gulf (1993-2019), and near the lower end of the historical range (0.49–3.33 prawns/ m², Table 8).

When compared to individual sites, juvenile prawn abundance at Mullaquana Station was lower than that recorded at most historical locations, including 5th Creek (−27%), The Spit (−49%), Mt Young (−77%), and False Bay (−82%). In contrast, abundance at Mullaquana Station was similar to that observed at Port Pirie (approximately 19% higher).

When compared to the most recent sampling period (2013–2019), mean juvenile prawn abundance at Mullaquana Station was intermediate between False Bay (1.6 prawns/m²) and Port Pirie (0.3 prawns/m²), with False Bay consistently supporting the highest abundance of Juvenile prawns of all sites in Spencer Gulf.

Table 8. Relative long-term average of juvenile prawns in the five key nursery areas and Mullaquana Station sampled in April only.

Location	Relative abundance (prawn/m^2)	SE	Transects	Years
5th Creek	0.77	0.24	16	5
False Bay	3.05	0.48	42	10
Mt Young	2.41	0.79	14	5
Port Pirie	0.47	0.07	33	9
Spit	1.09	0.18	21	5
Mullaquana	0.56	0.12	24	2

3.7. Abundance of other Macrofauna

A total of 3,294 macrofauna individuals were captured during the four sampling periods at the Mullaquana Station. Individual species were morphologically identified where possible into five categories, Annelids, Crabs, Fish, Molluscs and Other Crustaceans. Overall macrofauna abundance was highly variable between each sampling month and year across all species groupings, except for a low abundance of fish in February 2026, representing only 30% of the abundance (Table 9, Figure 8). The assemblage was dominated by fish (49%) with the fish component primarily consisting of Southern Longfin Goby (*Favonigobius lateralis*), Southern Blue Spotted flathead (*Platycephalus speculator)*, Small Tooth Flounder *(Pseudorhombus jenynsii)*, and Prickly Toadfish (*Contusus brevicaudus*). Annelids (29%) were mostly unidentified species of worm. Other Crustaceans (17%) comprised isopod, amphipod, and shrimp species. Crabs (4%) were mostly identified as Blue Swimmer Crab (*Portunus armatus),* as well as a small number of Shore crab, Hermit crab and Decorator crab. Molluscs (<1%) comprised Southern Dumpling Squid (*Euprymna tasmanica*) and Chitons. A list of macrofauna species identified in samples from Mullaquana Station is provided in Appendix 6.4.

Table 9. Count and percentage relative abundance (% - in parenthesis) of macrofaunal assemblages by category across sampling months (February and April) in 2025 and 2026 off Mullaquana Station, South Australia. Values are expressed as proportions of total individuals per sampling period. Crustaceans = non prawn species.

Sampling period	Annelids	Crab	Crustaceans	Fish	Molluscs
February 2025	195 (34%)	53 (9%)	74 (13%)	252 (44%)	2 (<1%)
April 2025	664 (36%)	37 (2%)	338 (19%)	785 (43)	20 (1%)
February 2026	21 (17%)	25 (20%)	42 (33%)	38 (30%)	
April 2026	60 (8%)	27 (4%)	107 (14%)	550 (74%)	4(<1%)
Total	940(29%)	142(4%)	561(17%)	1625(49%)	26 (1%)

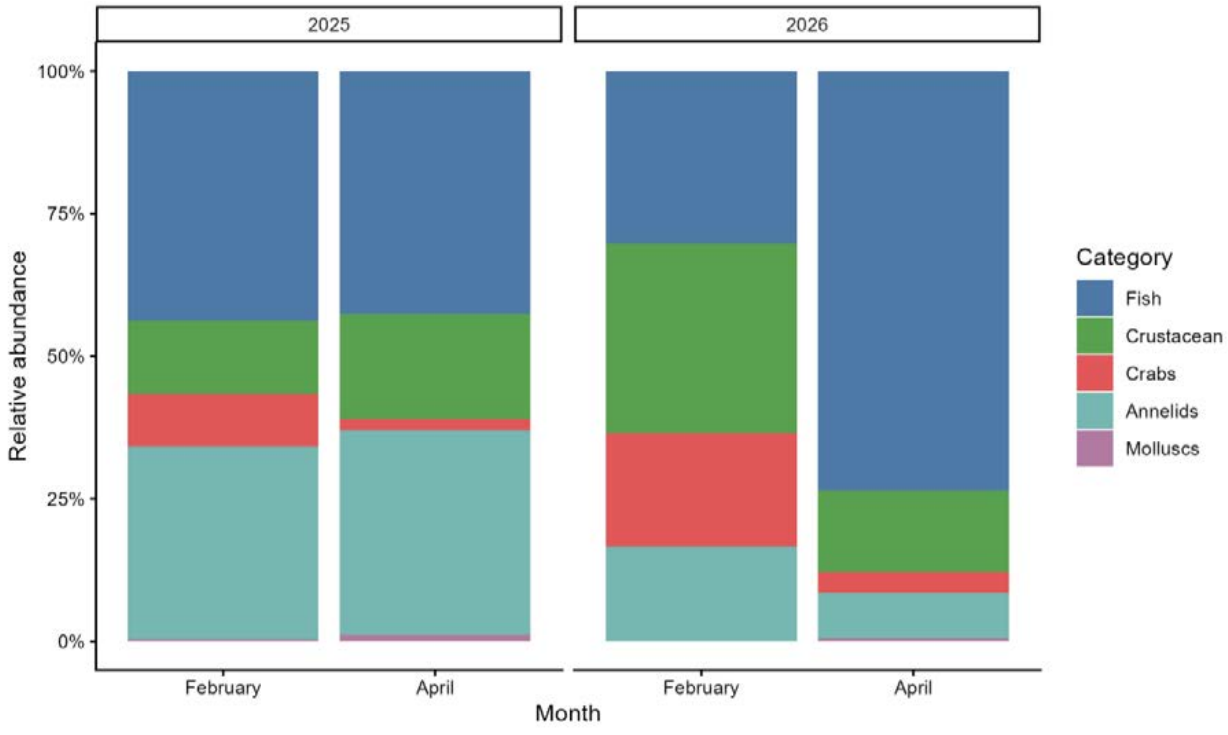


Figure 8. Relative abundance (proportion of total counts) of macrofaunal assemblages across sampling months (February and April) for 2025 and 2026 off Mullaquana Station, SA. See Table 9 for exact values.

4. DISCUSSION

The study provides a contemporary assessment of the distribution and abundance of juvenile prawns and other macrofauna in the vicinity of the proposed Northern Water Project infrastructure at Mullaquana Station. This report presents results from benthic sampling undertaken in February and April 2025 and February and April 2026. Given the reliance of the Spencer Gulf Western King Prawn stock on successful recruitment from northern nursery areas, understanding local patterns of juvenile abundance and distribution is critical for evaluating potential impacts associated with the construction and operation of the proposed plant. In particular, seawater intake processes and the discharge of desalination return water have the potential to affect the survival, growth and distribution of larval and juvenile prawns (e.g., Panagopoulos and Haralambous, 2020) and alter the ecological structure of macrobenthic fauna (Bianchelli *et al.* 2022; de-la-Ossa-Carretero *et al.* 2016; Riera *et al*. 2012).

4.1. Juvenile prawn abundance at Mullaquana Station

Relative abundance of juvenile prawns at Mullaquana Station varied between sampling years, months, and sites, reflecting both temporal dynamics and localised influences on recruitment and distribution. Historically, only Western King Prawn have been recognised in South Australia (King 1977, Heldt *et al.* 2025). However, the genetic analyses conducted in this study suggest the presence of two potential *Melicertus* prawn groups, populations, or possibly even species, supported by clear morphological differences. Interpretation of the results remains constrained by taxonomic uncertainty: this detection is a novel finding but limits our ability to assess whether environmental variables differentially affect the distribution and abundance of these species within inshore nursery habitats.

Temporal patterns in juvenile prawn abundance were apparent. The abundance of juvenile prawns (M1 and M2) was higher in 2026 than 2025, particularly in February and April, potentially indicating a stronger cohort or improved recruitment conditions in 2026. The observed increase from February to April 2026 further suggests seasonal progression in juvenile settlement for both M1 and M2.

Spatial patterns in juvenile prawn (M1 and M2) abundance were variable, particularly for *Melicertus 2* (M2), which showed significant interactions among year, month, and site. This supports the presence of complex spatial and temporal recruitment dynamics, consistent with previous observations for Western King Prawns (Kangas and Jackson 1998, Roberts 2005). Despite this, the fine-scale environmental factors shaping these distributions remain poorly understood.

For *Metapenaeopsis* sp., consistent increases in abundance from February to April in both years indicate a strong seasonal signal, likely associated with recruitment timing or favourable environmental conditions. The lack of an overall year effect suggests relatively stable interannual abundance for this group compared to juvenile *Melicertus* spp.

During this study, a significant harmful algal bloom of *Karenia spp*. identified in the coastal waters around the Fleurieu Peninsula in March 2025, expanded into Investigator Strait, Gulf St Vincent, and southern Spencer Gulf (Algal bloom Update 2026). Water sampling indicated that the algal bloom was present near Mullaquana Station during December 2025 and January 2026 (Algal Bloom Update 2026), which was in between the sampling years. Thus, potential regional effects on benthic intertidal communities near the study area should be considered when interpreting these results.

A decline in juvenile (M1 and M2 combined) prawn abundance of 14% was observed between February and April 2026 at the central site and coincided with nearby geotechnical drilling activities. Crustaceans are known to exhibit changes in behaviour in response to underwater sound (Davies *et al.* 2024). Tidal flats with low sediment dynamics and immobile macrofauna are at a higher risk of declining abundance under intensified sediment disturbances than areas with high sediment dynamics and mobile macrofauna (Zhou *et al*. 2024). This was in contrast to the patterns in juvenile abundance observed at the north and south sites away from the drilling activity, which increased 262% and 370% over the same time period, respectively. The potential contribution of regional ecological impacts associated with the *Karenia* spp. bloom cannot be excluded.

Overall, the study indicated that areas adjacent to the proposed intake and outfall infrastructure overlap with habitats of juvenile prawn recruitment. However, as observed in previous studies (Roberts 2005), recruitment patterns were inherently variable both spatially and temporally, likely reflecting environmental drivers such as hydrodynamics, temperature, and habitat condition. This highlights the importance of undertaking multiple surveys to assess the potential importance of coastal habitats to recruitment processes.

4.2. Historical juvenile prawn comparison with Mullaquana Station

Overall, juvenile prawns were well represented across survey sites, supporting previous findings that the upper Spencer Gulf contains suitable habitat for early life stages. The presence of juveniles across sampling periods is consistent with known seasonal recruitment patterns, where post-larval settlement occurs over summer followed by growth through late summer and autumn (Heldt *et al.* 2025).

The presence of juvenile prawns at multiple survey sites at Mullaquana Station indicates that the area likely provides important habitat for juvenile prawn recruitment and development. However, interpretation of the importance of Mullaquana Station as a nursery site relative to other sites in Spencer Gulf is constrained by historical spatial and temporal differences in sampling. The abundance of juvenile prawns at Mullaquana Station in April 2025–2026 was 0.56 ± 0.12 prawns/m². This was below the historical average (1993-2019) recorded across Spencer Gulf (Mean: 1.69 ± 0.21 prawns/m²; Range: 0.47 - 3.05 prawns/m²). While low relative to other sites sampled, abundance at Mullaquana Station is comparable to that recorded at Port Pirie, a site previously identified as an important nursery area for juvenile prawns (Carrick 1996). Moreover, the historical dataset used for comparison spans multiple decades and incorporates sampling under a wide range of environmental conditions that may result in extremes (highs and lows) in the estimates of juvenile abundance recorded. This contrasts with the Mullaquana Station dataset that currently represents only two years of sampling. Consequently, it is not possible to determine the influence of natural interannual variability on juvenile prawn abundance at Mullaquana Station.

4.3. Macrofauna at Mullaquana Station

There is no direct comparison of macrofauna samples with previous years, as no prior sampling has been conducted at this specific location and there is no historical dataset for these taxa. However, the composition and relative abundance observed in this study are broadly consistent with assemblages typically observed for similar shallow, soft-sediment habitats within Spencer Gulf (Ward and Hutchings 1996, G.H. personal observation). The dominance of fish and annelids, along with the presence of mobile crustaceans such as isopods, amphipods, and brachyuran crabs, reflects a productive and heterogeneous habitat capable of supporting a range of trophic levels.

The presence of juvenile Blue Swimmer Crab (*Portunus armatus*), including individuals as small as 4mm in carapace width, aligns with the known spawning and recruitment period for this species in southern Australia, which occurs primarily over the summer-autumn months (Kumar *et al.* 2003). This observation highlights the ecological importance of the study area as a potential nursery habitat for other species. Additionally, the occurrence of fish species such as gobies, flathead, and flounder further supports the role of the habitat in providing shelter and foraging opportunities during critical developmental stages. Collectively, these findings indicate that the proposed Project site at Mullaquana contributes to local biodiversity and plays a functional role in supporting key life stages of several species within the Spencer Gulf ecosystem.

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6. APPENDICES

Appendix 6.1: GPS Coordinates for Sampling Locations

The below table provides the GPS coordinates of recruitment survey sites at the Project proposed desalination site off Mullaquana Station, SA.

Site	Site type	Latitude	Longitude
North1	sand	-33.2445	137.4643
N2	sand	-33.2487	137.4629
N3	sand	-33.2537	137.4614
N4	sand	-33.2457	137.4695
Central1	sand	-33.2467	137.4747
C2	sand	-33.251	137.4733
C3	sand	-33.2562	137.4715
C4	sand	-33.2775	137.4533
South1	sand	-33.2818	137.4519
S2	sand	-33.286	137.4505
S3	sand	-33.2788	137.4585
S4	sand	-33.2829	137.4572

Appendix 6.2: Summary of PCR Amplification and Sequencing Result for Juvenile Prawns

Original ID	Species ID (barcoding)
C1	Failed sequencing
C5	Failed sequencing
C8	*Melicertus* sp. 1
C9	*Melicertus* sp. 1
N1	*Melicertus* sp. 1
N3	*Metapenaeopsis lindae* (Linda's velvet prawn)
N5	*Metapenaeopsis lindae* (Linda's velvet prawn)
N6	*Melicertus* sp. 2
N7	*Melicertus* sp. 1
N15	Failed sequencing, rePCR
S3	*Melicertus* sp. 1
S4	*Melicertus* sp. 1
S6	*Metapenaeopsis lindae* (Linda's velvet prawn)
S7	*Metapenaeopsis lindae* (Linda's velvet prawn)
S8	*Melicertus* sp. 1
S10	Failed PCR, rePCR

Appendix 6.3: Prawn Species Identification Steps

Prawn species Key:

Note: The colour, ventral spines on rostrum, spots or hairs on the last tail segment should not be used in separating species (Carrick 1996).

Note 2: Where possible, use features describe in step 3 to confirm identification made from step 2.

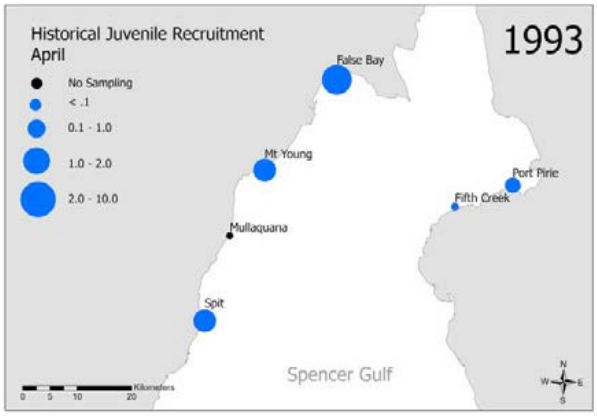
Note 3: Tails were used to further differentiate between *Penaeus (Melicertus)* spp.

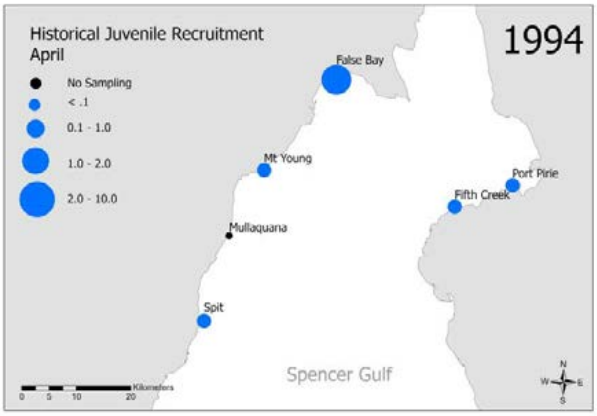
Step	Parameter	Next step	Species grouping
Size	Carapace length smaller than 1.7mm	-	Laval stage
	Carapace length between 1.7mm and 3mm	-	Post Larval
	Carapace length greater than 3mm	1	Juvenile
1	Only tail or whole body examined	2	
	Only head examined	3	
	Only mid-section of body examined or damaged specimen	-	Not identifiable
2	Telson apex cleft and rostrum not extending much further than the ocular papilla (Meland *et al*. 2015)	-	*Mysidacea* spp.
	Telson with four pairs of movable spines	3	*Penaeus (Melicertus)* spp. (M2)
	Telson with three pairs of movable spines		*Penaeus (Melicertus)* spp. (M1)
	Telson trident-shaped; posterior pair of spines fixed, three anterior pairs movable	3	*Metapenaeopsis* sp.
3	No Pterygostominal spine	-	*Penaeus (Melicertus)* spp.
	Pterygostominal spine	-	*Metapenaeopsis* sp.

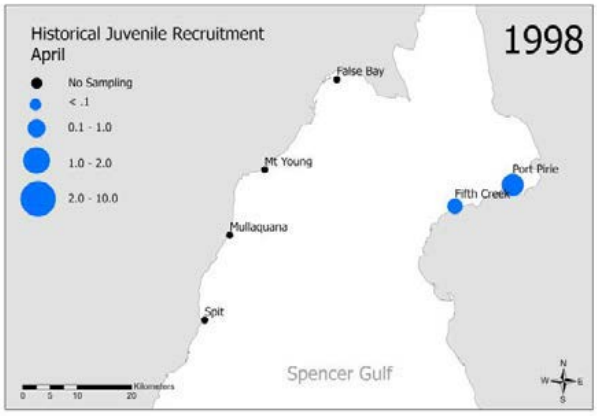
Appendix 6.4: Macrofauna Species List

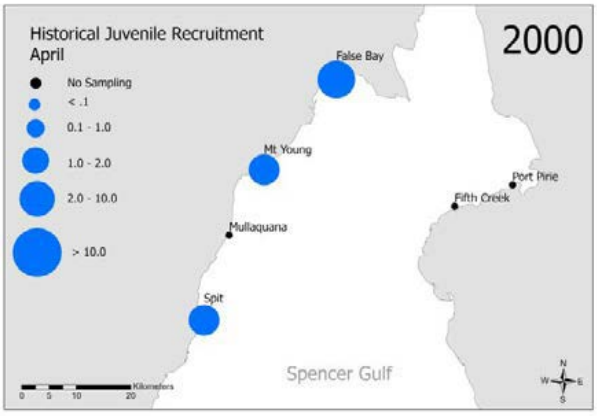
Phyla	Common name	Scientific name (if known)	Count
Annelida	Worms		940
Cephalopod	Southern Pygmy Squid	*Euprymna tasmanica*	23
Crab	Blue Swimmer Crab	*Portunus armatus*	132
	Hermit Crab	*Paguristes frontalis*	5
	Decorator Crab	*unknown*	2
	Purple Mottled Shore Crab	*Cyclograpsus granulosus*	3
Other Crustacean	Isopod		278
	Amphipod		251
	Amphipod 2		1
	Sand Shrimp	*unknown*	12
	Smooth Mantis Shrimp	*Belosquilla laevis*	1
	Unidentified shrimp	*unknown*	
Mollusc	Chiton	*unknown*	3
Fish	Southern Longfin Goby	*Favionigobius lateralis*	1580
	Elongate Flounder	*Ammotretis elongatus*	1
	Smalltooth Flounder	*Pseudorhombus jenynsii*	8
	Southern Blue Spotted Flathead	*Platycephalus speculator*	2
	Prickly Toadfish	*Contusus brevicaudus*	*14*
	Soldier Fish	*Gymnapistes marmoratus*	*2*
	Longsnout pipefish	*Vanacampus poecilolaemus*	*1*
	Unidentified 1	*unknown*	*7*
	Unidentified 2	*unknown*	*6*
	Unidentified 3	*unknown*	*1*
	Unidentified 4	*unknown*	*1*
	Unidentified 5	*unknown*	*1*
	Unidentified Flathead	*unknown*	*1*

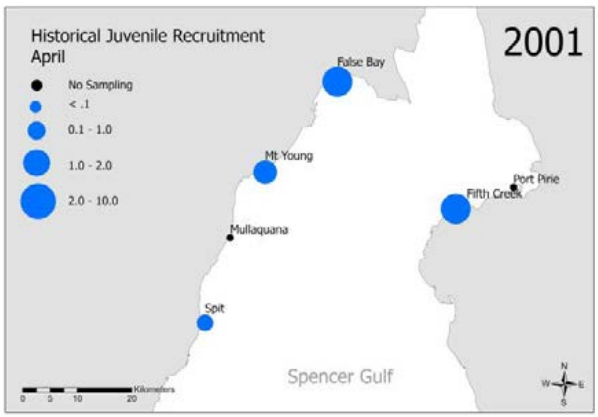
Appendix 6.5: Spencer Gulf Spatial Maps 1993 – 2026

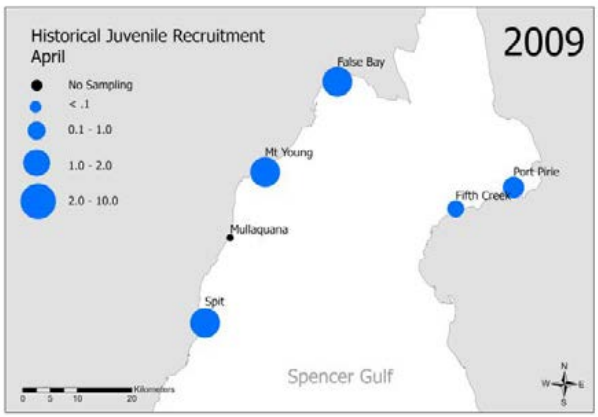


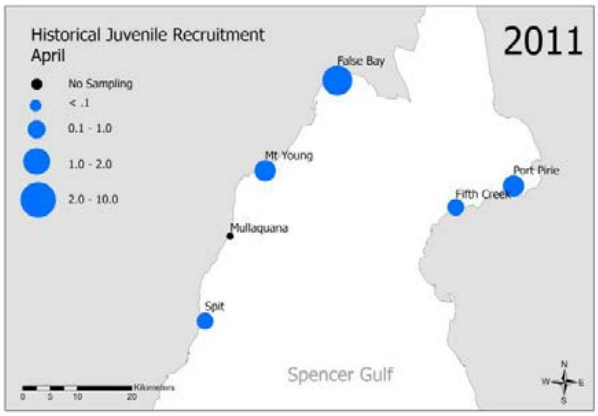


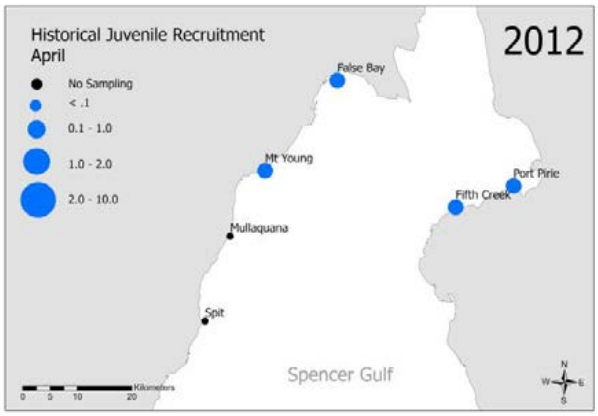


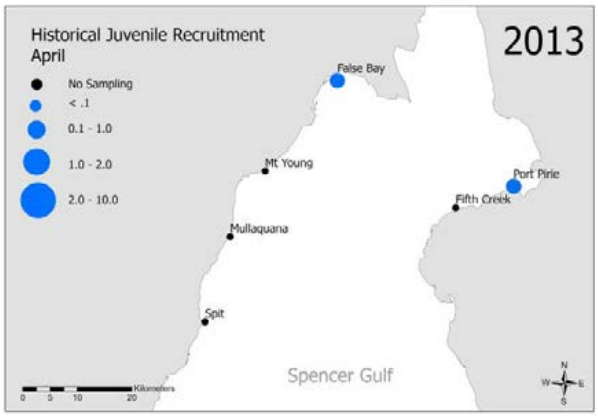


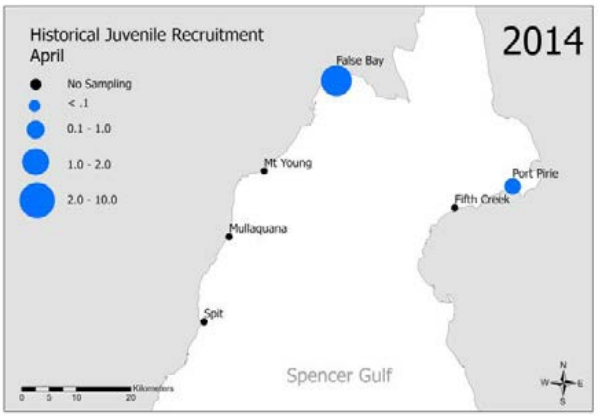


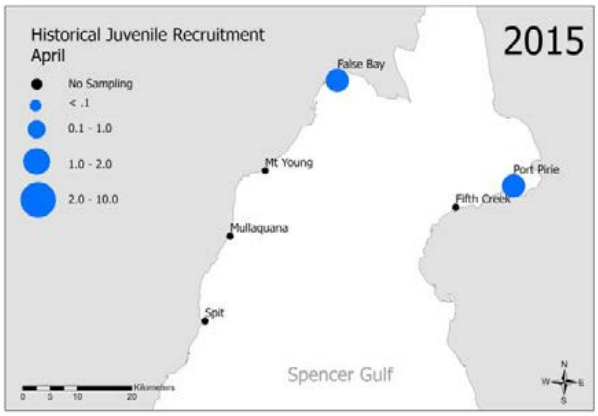


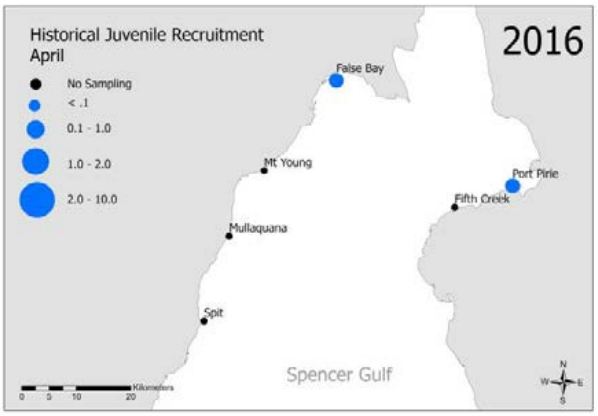


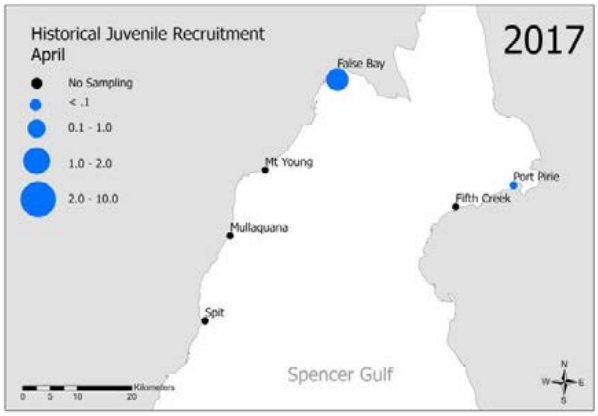


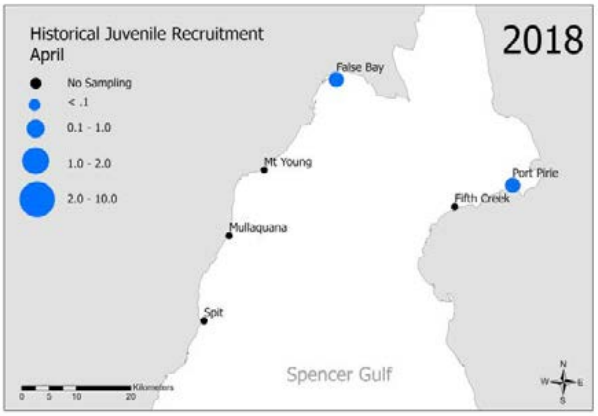

Historical Juvenile Recruitment
April
2013
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf

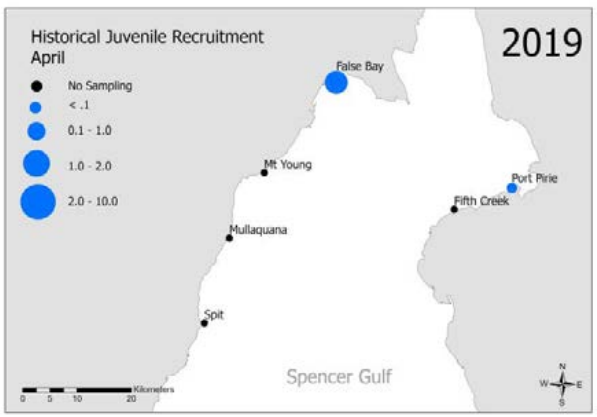

Historical Juvenile Recruitment
April
2014
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf

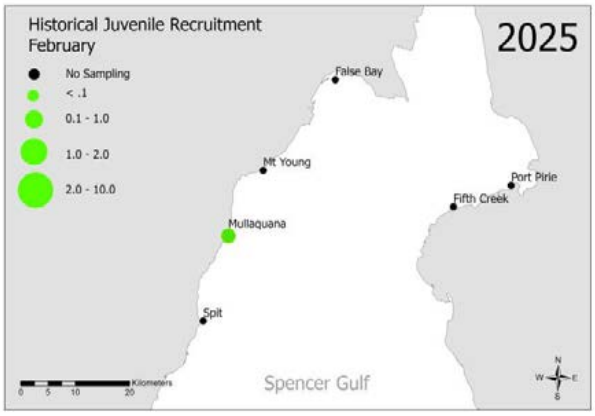

Historical Juvenile Recruitment
April
2015
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf

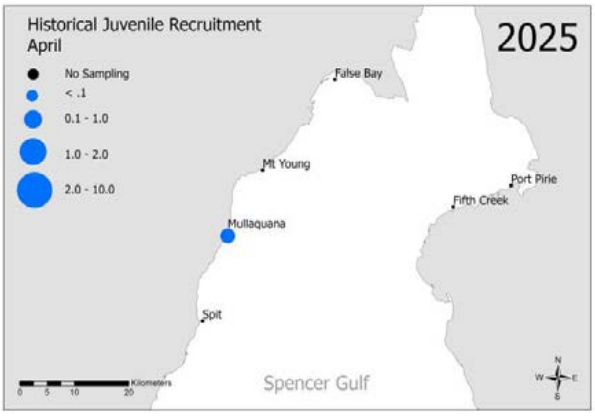

Historical Juvenile Recruitment
April
2016
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf

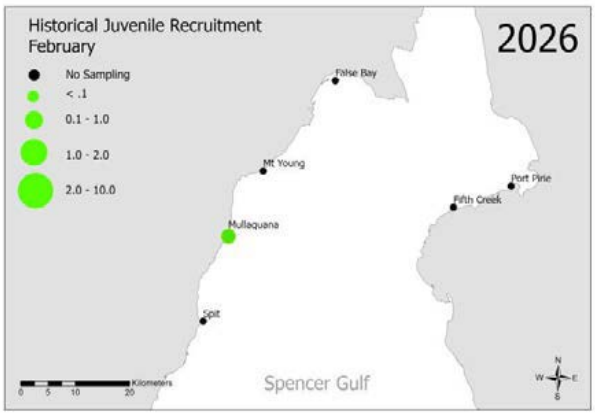

Historical Juvenile Recruitment
April
2017
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf

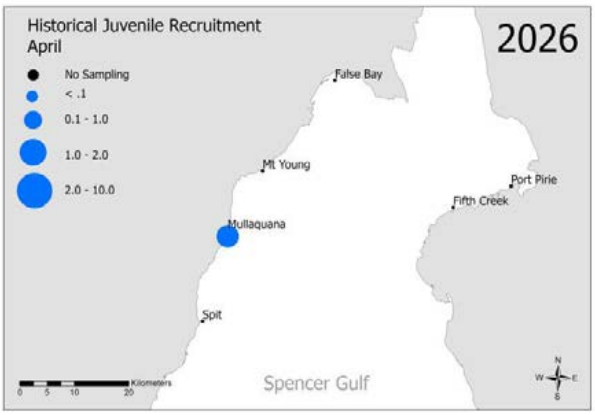

Historical Juvenile Recruitment
April
2018
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf


Historical Juvenile Recruitment
April
2019
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf


Historical Juvenile Recruitment
February
2025
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf
Kilometers
0 5 10 20
N
W E
S


Historical Juvenile Recruitment
April
2025
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf
Kilometers
0 5 10 20
N
W E
S


Historical Juvenile Recruitment
February
2026
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf
Kilometers
0 5 10 20
N
W E
S


Historical Juvenile Recruitment
April
2026
No Sampling
< .1
0.1 - 1.0
1.0 - 2.0
2.0 - 10.0
False Bay
Mt Young
Port Pirie
Fifth Creek
Mullaquana
Spit
Spencer Gulf
Kilometers
0 5 10 20
N
W E
S

Appendix 6.6: Results of statistical testing

Three-way ANOVA results for $\log_{10}$-transformed prawn abundance ($\log_{10}[x + 1]$) examining the effects of Year, Month, Site, and their interactions; and Tukey-adjusted pairwise comparisons from estimated marginal means for $\log_{10}$-transformed prawn abundance. Results are shown for main effects and selected contrasts illustrating the significant Year × Month × Site interaction. Statistically significant effects ($p < 0.05$) are indicated in bold.

ANOVA *Melicertus 1*

Term	Df	Sum Sq	Mean Sq	F value	p-value
Year	1	10.018	10.018	20.534	**<0.001**
Month	1	6.725	6.725	13.784	**0.0007**
Site	2	1.858	0.929	1.905	0.164
Year × Month	1	1.306	1.306	2.677	0.111
Year × Site	2	1.161	0.581	1.190	0.316
Month × Site	2	0.917	0.459	0.940	0.400
Year × Month × Site	2	1.121	0.560	1.149	0.328
Residuals	36	17.564	0.488		

Tukey-adjusted pairwise comparisons

Contrast	Estimate	SE	df	t-ratio	p-value
2025 – 2026	-1.244	0.285	36	-4.361	**0.0006**
April – February	0.964	0.285	36	3.782	**0.003**

ANOVA *Melicertus 2*

Term	Df	Sum Sq	Mean Sq	F value	p-value
Year	1	26.656	26.656	52.933	**<0.001**
Month	1	0.009	0.009	0.018	0.895
Site	2	7.533	3.766	7.479	**0.0019**
Year × Month	1	1.499	1.499	2.976	0.093
Year × Site	2	0.809	0.404	0.803	0.456
Month × Site	2	9.594	4.797	9.525	**<0.001**
Year × Month × Site	2	10.958	5.479	10.880	**<0.001**
Residuals	36	18.129	0.504		

Tukey-adjusted pairwise comparisons

Contrast	Estimate	SE	df	t-ratio	p-value
2025 Apr.S. – 2026 Apr.S.	-2.837	0.502	36	-5.653	**0.001**
2026 Apr.C. – 2026 Apr.S	-3.043	0.502	36	-6.064	**<0.001**
2026 Apr.S. – 2026 Feb.S	1.910	0.502	36	3.807	**0.023**

ANOVA *Metapenaeopsis*					
Term	**Df**	**Sum Sq**	**Mean Sq**	**F value**	**p-value**
Year	1	1.307	1.307	3.246	0.080
Month	1	24.599	24.599	61.106	**<0.001**
Site	2	13.048	6.524	16.206	**<0.001**
Year × Month	1	7.363	7.363	18.290	**<0.001**
Year × Site	2	0.940	0.470	1.168	0.323
Month × Site	2	0.457	0.229	0.568	0.572
Year × Month × Site	2	1.677	0.838	2.083	0.139
Residuals	36	14.493	0.403		

Tukey-adjusted pairwise comparisons					
Contrast	**Estimate**	**SE**	**df**	**t-ratio**	**p-value**
Central – North	0.244	0.224	36	1.089	0.5272
Central – South	1.208	0.224	36	5.384	**<0.0001**
North – South	0.964	0.224	36	4.295	**0.0004**
2025 Apr – 2026 Feb.	2.215	0.259	36	8.552	**<0.001**
2026 Apr. – 2026 Feb.	1.762	0.259	36	6.801	**<0.001**
2026 Apr. – 2025 Feb.	1.102	0.259	36	4.253	**0.0008**

ANOVA Combined *Melicertus*					
Term	**Df**	**Sum Sq**	**Mean Sq**	**F value**	**p-value**
Year	1	0.213	0.213	68.563	**<0.001**
Month	1	0.067	0.067	21.601	**<0.001**
Site	2	0.042	0.021	6.828	**0.003**
Year × Month	1	0.058	0.058	18.704	**<0.001**
Year × Site	2	0.023	0.011	3.687	**0.035**
Month × Site	2	0.040	0.204	6.566	**0.004**
Year × Month × Site	2	0.058	0.029	9.281	**<0.001**
Residuals	36	0.112	0.003		

Tukey-adjusted pairwise comparisons					
Contrast	**Estimate**	**SE**	**df**	**t-ratio**	**p-value**
Central – North	-0.0008	0.019	36	-0.042	0.9990
Central – South	-0.063	0.019	36	-3.221	**0.0075**
North – South	-0.063	0.019	36	-3.179	**0.0083**
April – February	0.075	0.0161	36	4.648	**<0.001**
2025 – 2026	-0.133	0.0161	36	-8.280	**<0.001**
2025 Apr.S. – 2026 Apr.S.	-0.353	0.039	36	-8.963	**<0.001**
2026 Apr.C. – 2026 Apr.S	-0.263	0.039	36	-6.681	**<0.001**
2026 Apr.S. – 2026 Feb.S	0.289	0.039	36	7.329	**<0.001**

Appendix G-8:

Interaction Between the Desalination Plant and Snapper and Whiting

TO: **DEPARTMENT FOR INFRASTRUCTURE AND TRANSPORT (DR LEXI SCHWARZ)**

FROM: **SARDI AQUATIC SCIENCES (DR TROY ROGERS)**

SUBJECT: **POTENTIAL INTERACTION BETWEEN THE PROPOSED DESALINATION PLANT AT MULLAQUANA STATION AND SNAPPER AND KING GEORGE WHITING IN SPENCER GULF**

DATE: **7 AUGUST 2025**

KEY ISSUES

- The Department for Infrastructure and Transport (DIT, Dr Lexi Schwarz) has requested information regarding the potential interaction of Snapper (*Chrysophrys auratus*) and King George Whiting (*Sillaginodes punctatus*) in Spencer Gulf with the proposed desalination plant as part of the Northern Water Project (NWP). The NWP is investigating two sites for the desalination plant. This Advice Note relates to the Mullaquana Station site, located 20 km south of Whyalla.
- This Advice Note provides information about the life history of Snapper and King George Whiting in Spencer Gulf, particularly spawning areas, egg and larval dispersal pathways, nursery areas associated with juvenile recruitment, and adult movement patterns.
- For Snapper, the proposed site is directly adjacent to key spawning and nursery areas in northern Spencer Gulf; consequently, there is the potential for interaction at all life history stages.
- For King George Whiting, the proposed site is adjacent to a key nursery area; consequently, the potential for interaction is highest at the larval, juvenile, and sub-adult life stages.

BACKGROUND

A desalination plant in Spencer Gulf (SG) has been proposed as part of the Northern Water Project. A potential location of the desalination plant site is at Mullaquana Station on north-eastern Eyre Peninsula, approximately 20 km south of Whyalla. The intake structure is proposed to be located approximately 3.9 km offshore at 13 m depth, with the intake screens 3-6 m off the seafloor. The outfall diffusers to release the discharged brine is proposed to run parallel to the intake, with the outfall at 17 m depth and approximately 4.8 km from shore. As such, the area of ecological concern for the project site is the near shore waters of northern SG adjacent to Mullaquana Station (i.e., extending approximately 5 km from the coast).

Fishing industry stakeholders have raised concerns regarding the location of the proposed desalination plant at Mullaquana Station and potential risks to Snapper and King George Whiting in SG. In response, DIT have requested information regarding the potential interaction between these species and the proposed site at Mullaquana Station. This Note provides an overview of the life history of Snapper and King George Whiting in Spencer Gulf including:

- Spawning patterns and locations;
- Egg and larval dispersal;
- Recruitment areas and associated habitats; and
- Adult movement patterns and associated habitats.

RESULTS / DISCUSSION

Snapper

Snapper is a large-bodied, demersal marine fish species that is distributed throughout temperate and subtropical waters of Australia (Kailola et al., 1993). It is long-lived, estimated to live up to 36 years in South Australia (SA), and can reach up to 1.2 m total length (TL) (McGlennon et al., 2000). Snapper is an iconic fish species that supports important commercial and recreational fisheries throughout its distribution. This is particularly true for SA, which has traditionally supported the largest fisheries for Snapper in Australia (Rogers et al., 2024). The Snapper fishery in SA encompasses most of the State's coastal waters, although the highest abundances and fishery catches were generally taken from SG and Gulf St Vincent (GSV) (Drew et al., 2022).

There are three recognised stocks of Snapper throughout SA waters: the Spencer Gulf / West Coast (SG/WC) stock, the Gulf St Vincent (GSV) stock, and the Western Victoria Stock (Fowler, 2016; Fowler et al., 2017). Each stock is dependent on spawning and recruitment (the addition of juvenile fish) into a primary nursery area. The SG/WC stock includes SG and all waters westward to the WA-SA border. The primary nursery area for the SG/WC stock is northern SG, which experiences highly variable inter-annual recruitment (Fowler and Jennings, 2003; Fowler et al., 2010). This means that even though the fish spawn each summer, in most years spawning does not result in recruitment. The occasional strong year class (e.g., once every decade) sustains the fishery through periods that are characterised by poor to average recruitment.

The SG/WC stock experienced a significant decline in biomass during the late 2000s and 2010s that related to an extended period of poor juvenile recruitment and the concurrent exploitation of the remaining population (Fowler, 2025). The decline in biomass and stock status for the SG/WC stock prompted a suite of management actions from 2012 that culminated in the complete closure of the stock to fishing on 1 November 2019. The most recent stock assessment was delivered in November 2022 which indicated that the closures had arrested the decline in biomass, but there was not yet sufficient evidence of recovery (Drew et al., 2022). Consequently, the fishery closure was extended to 30 June 2026.

Spawning areas

For the SG/WC stock, the primary spawning and nursery area for Snapper is in northern SG (Fowler et al., 2010; Fowler, 2023) (Figure 1). Snapper has a protracted spawning period that extends from November to February, and peaks in December and January (McGlennon, 2003; Saunders, 2009). Mature fish form dense aggregations prior to and during the spawning period and can spawn repeatedly over consecutive days for several weeks. This aggregating behaviour can result in the localised concentration of adult fish (Drew et al., 2022). There are numerous sites throughout northern SG that are recognised as adult aggregation sites, which are typically characterised by moderate to high relief structures including large banks, natural hard substrates (e.g., limestone ledges, rock piles), and artificial structures (e.g., shipwrecks). Several of these structures were the sites of spatial spawning closures for Snapper that were in effect from 15 December to 31 January annually between 2013 and 2019 (Fowler and McGarvey, 2014).

Egg and larval dispersal

Snapper eggs are positively buoyant and remain at or near the surface during embryonic development. The eggs hatch after approximately 36 hours at 20°C, releasing a larva of approximately 2 mm body length (BL) (McGlennon, 2003; Fowler, 2023). The pelagic larvae reside in a mixed planktonic community for 20-30 days before they undergo settlement to the demersal environment and become juveniles at 10-12 mm TL (Fowler and Jennings, 2003; Fowler et al., 2010; Fowler, 2023). Snapper larvae progressively develop swimming and sensory abilities during the larval period that can influence their dispersal, including horizontal and vertical movement (Table 1). Predictions of larval dispersal from a biophysical model indicate that larvae settle to nursery areas that are relatively short distances (i.e., tens of kilometres) from where they were spawned (Rogers et al., in prep.). In general, the larvae are dispersed northward by south-easterly winds that prevail during the summer months (Figure 1).

Table 1. Summary of larval development and behaviour for Snapper. BL – body length, dph = days post hatch.

Stage	Egg	Hatching	Pre-flexion	Flexion	Post-flexion	Settlement
Size (mm)	0.8–0.9 (dia.)	2.1–2.6 BL	2.6–5.0 BL	5.0–6.5 BL	6.5–10.0 BL	10.0–13.5 BL
Growth (mm/d)	-	0.25	0.25	0.3–0.6	0.3–0.6	0.3–0.6
Age (dph)	-	0–2	2–10	10–13	13–18	18–30
Duration (d)	1.5	2	10	3	5	10
Behaviour	Passive (drift) Buoyant (float)	Passive (drift) Buoyant (float)	Passive (drift)		Capable	Capable Settle (demersal)
Vertical distribution	Surface (0–3 m)	Surface (0–3 m)	Day – mid Night – random	Day – mid Night – random	Day – mid-lower Night – random	Day – lower Night – random
Horizontal movement				1 cm/s	3 cm/s	5 cm/s

Recruitment areas

Throughout its distribution in Australia and New Zealand, nursery areas for juvenile Snapper (i.e., <1 year of age) are typically located in relatively protected coastal waters of estuaries, bays, and harbours. Within these environments, juvenile Snapper can occupy a diversity of different habitats that range from deep areas of bare sediment through to shallow seagrass beds. For northern SG, the distribution and abundance of juvenile Snapper was strongly associated with relatively small areas characterised by soft, muddy benthic substrate (Fowler and Jennings, 2003; Saunders, 2009; Fowler et al., 2010). These areas are located near Western Shoal, False Bay, Fitzgerald Bay, and Ward Spit (Figure 1). The nursery area near Western Shoal has consistently provided the highest abundances of juvenile Snapper in northern Spencer Gulf (Fowler and Jennings, 2003; Fowler et al., 2010). It has been hypothesised that the preference for muddy benthic substrate as a nursery area may relate to the availability of appropriate prey fauna (Saunders et al., 2012). Juvenile Snapper remain in the nursery areas for several months after settlement and then transition to more complex micro-habitats as they develop.

Adult movement and habitats

The extent of movement for Snapper during the sub-adult and adult life stages is density-dependent and relates to high inter-annual variation in juvenile recruitment (Fowler and Jennings, 2003; Fowler and McGlennon, 2011). As such, the rate of emigration of Snapper from their nursery areas and dispersion to other regions varies between years depending on year class strength and the relative size of the population (Fowler, 2016). In years of low recruitment, juveniles develop in northern SG and remain there as adults. However, in years of high recruitment and when the population is sufficiently large, sub-adult fish (2-3 years old) emigrate southward from northern SG to southern SG as available habitat and resources become limited and remain in these areas for the remainder of their lives (Figure 1) (Fowler, 2016; Fowler et al., 2017). In SG, Snapper can occupy a diversity of habitats extending from shallow, coastal bays to offshore trenches and reefs.

Potential interaction

Northern SG supports the key spawning and nursery areas for the SG/WC stock of Snapper (Fowler et al., 2017). The proposed site at Mullaquana Station is directly adjacent to the primary nursery area near Western Shoal. The recovery of the SG/WC stock is dependent on juvenile recruitment and the subsequent accumulation of biomass (Drew et al., 2022). This area is also utilised by sub-adult and adult Snapper at various times of the year. Consequently, there is the potential for all life stages of Snapper to interact with the proposed site.

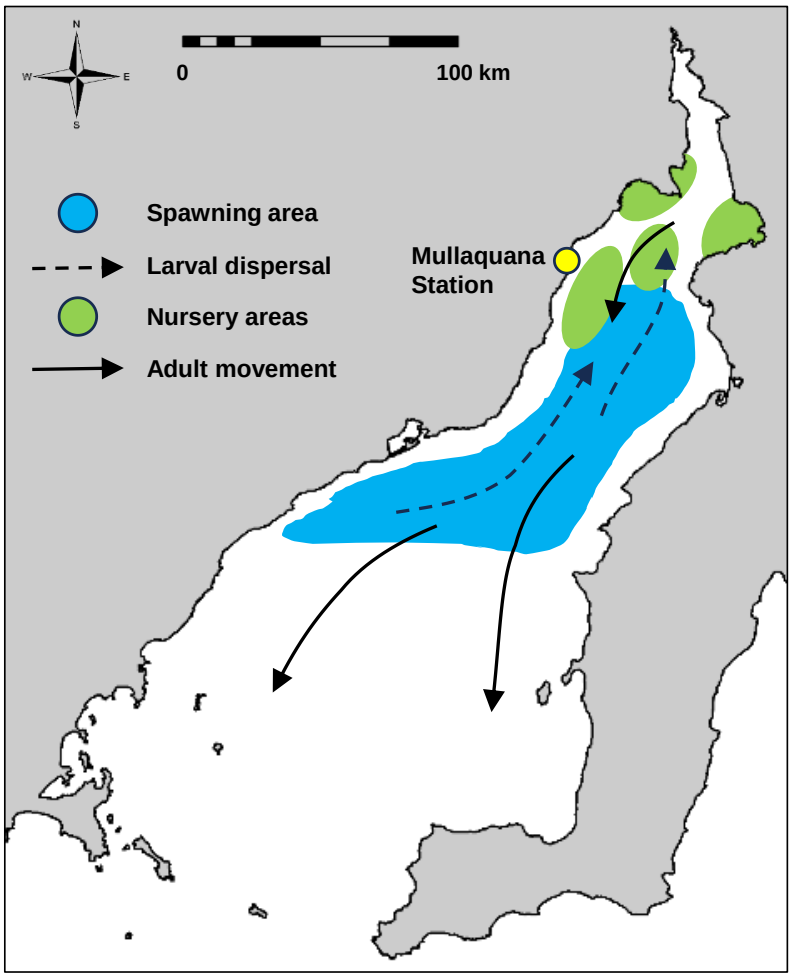


Figure 1. Conceptual model of life history for Snapper in Spencer Gulf showing the location of the proposed desalination plant site at Mullaquana Station (yellow circle).

King George Whiting

King George Whiting is a demersal marine finfish species that is endemic to temperate coastal waters of southern Australia (Kailola et al., 1993). It is estimated to live up to 22 years and grow to 70 cm TL (Fowler et al., 2000a). South Australia is at the geographic centre of this distribution and has historically supported the most productive fisheries (Drew et al., 2024).

There are three recognised stocks of King George Whiting in South Australian waters: the west coast of Eyre Peninsula (WC), SG, and GSV. This conceptual model of stock structure was developed from the connectivity between key spawning grounds and nursery areas in each region (Fowler et al., 2000b; Rogers et al., 2019a; 2021).

King George Whiting is an iconic species that is highly sought by commercial and recreational fishers. In SA, the highest commercial catches have traditionally been taken from SG, with the highest proportion of catch in SG taken from the southern part of the gulf near Port Victoria and the Sir Joesph Banks Group (Dennis et al., 2024).

Spawning areas

King George Whiting has a complex life history that involves ontogenetic changes in habitat use that are linked by movement at different life stages (i.e., larval dispersal and adult movement). In SA, the only recognised spawning area is throughout southern SG and Investigator Strait (Figure 2) (Fowler et al., 1999; 2000a; Drew et al., 2020). Spawning areas are characterised by low-profile reefs and shoals in areas of moderate to high wave energy, generally at depths greater than 20 m. King George Whiting has a protracted spawning period from March to July, and peaks in April and May (Fowler et al., 1999; 2000a).

Egg and larval dispersal

King George Whiting eggs are positively buoyant and remain near the surface during embryonic development. The eggs hatch after approximately two days at 18°C, releasing a larva of approximately 2 mm BL (Ham and Hutchinson, 2003). King George Whiting larvae progressively develop swimming and sensory abilities during the larval period that can influence their dispersal (Table 2) (Rogers et al., 2021). King George Whiting has a prolonged pelagic larval phase during winter and spring that ranges in duration from 80 to 130 days (Fowler and Short, 1996; Rogers et al., 2019a; 2019b). During this time, the developing larvae disperse from the spawning area in southern SG to inshore nursery areas throughout the gulf over distances of tens to hundreds of kilometres (Fowler et al., 2000b; Drew et al., 2020; Rogers et al., 2021). In general, the larvae disperse northward by south-westerly winds that prevail during the winter months (Figure 2).

Table 2. Summary of larval development and behaviour for King George Whiting (from Rogers et al., 2021). BL – body length, DVM – diurnal vertical migration, dph – days post hatch.

Stage	Egg	Hatching	Pre-flexion	Flexion	Post-flexion	Settlement
Size (mm)	0.8 dia.	2.1–3.0 BL	3.0–5.5 BL	5.5–6.5 BL	6.5–15.0 BL	15.0–20.5 BL
Growth (mm/d)	-	0.10	0.09-0.15	0.16	0.18	0.18
Age (dph)	-	0–9	9–29	29–35	35–82	82–112
Duration (d)	2	9	20	6	47	10
Behaviour	Passive (drift) Buoyant (float)	Passive (drift)	Passive (drift)	DVM	DVM	DVM Settle
Vertical distribution	Surface (0–5 m)	Surface (0–5 m)	Surface (0–5 m)			
Vertical movement				1 cm/s	2 cm/s	2 cm/s

Recruitment areas

Following the prolonged pelagic larval phase, King George Whiting larvae of approximately 15-18 mm TL settle to shallow beds of seagrass (*Zostera* spp.) in protected bays along the coast (Fowler and Short, 1996; Steer et al., 2014; Rogers et al., 2019a, 2019b). Numerous nursery areas for King George Whiting have been identified throughout SG (Figure 2). Juveniles develop for two to three years in the vicinity of the nursery areas to which they settled, before they move offshore to replenish the populations of spawning adults (Fowler et al., 2000b; 2002).

Adult movement and habitats

Due to the geography of SG, sub-adult and adult King George Whiting typically move southward as they migrate from inshore coastal waters to the spawning population in southern SG (Figure 2) (Fowler et al., 2002). Once these fish reach the spawning population, they remain in this region for the remainder of their lives. Consequently, the population in southern SG is composed of fish of mixed sizes (30-70 cm TL) and ages (4-20 years) (Fowler et al., 1999, 2000a; Drew et al., 2020). Preferred habitats for adult fish in this region are low-profile reefs and shoals in areas of moderate to high wave energy.

Interaction

Northern SG supports important inshore nursery areas for King George Whiting. In particular, the nursery area at Cowleds Landing is directly adjacent to the proposed site. Consequently, there is the potential for interaction between the larval, juvenile, and sub-adult life stages of King George Whiting and the proposed site at Mullaquana Station.

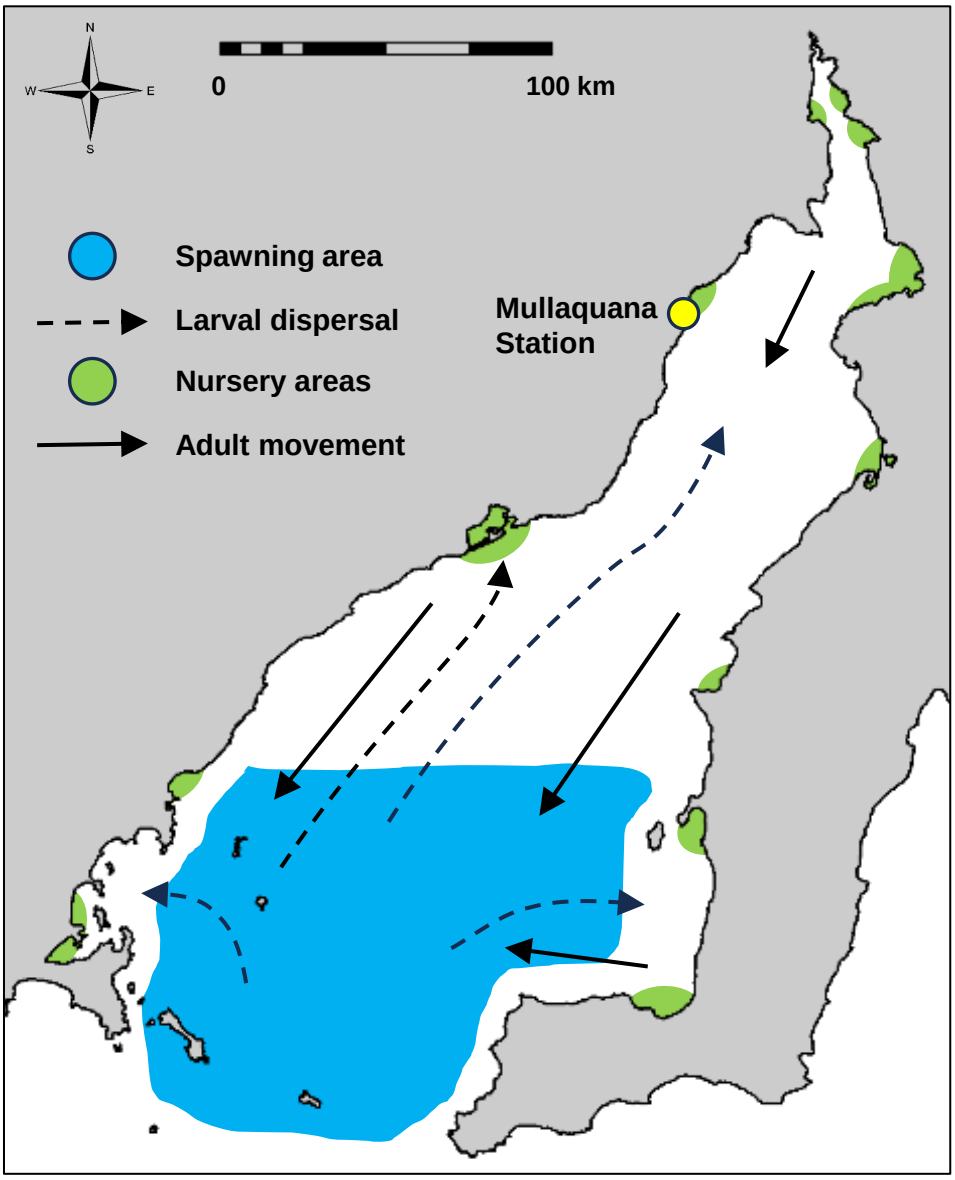


Figure 2. Conceptual model of life history for King George Whiting in Spencer Gulf showing the location of the proposed desalination plant site at Mullaquana Station (yellow circle).

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A/Research Director, SARDI Aquatic and Livestock Sciences

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Appendix G-9:

Underwater Noise and Vibration Assessment

Northern Water Project

Underwater Noise and Vibration Impact Assessment

A220729RP3 Revision B

Monday, 21 September 2026

Document Information

Project	Northern Water Project
Client	JBS&G Pty Ltd
Report title	Underwater Noise and Vibration Impact Assessment
Project Number	A220729

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0	10 November 2025	Draft for comment	Nick Henrys	Darren Jurevicius
A	12 March 2026	First Issue	Nick Henrys	Darren Jurevicius
B	21 September 2026	Response to review comments	Nick Henrys	Darren Jurevicius

Glossary

Airblast overpressure	The sudden increase in air pressure, generated by a shock wave, produced when an explosive is detonated.
Ambient sound	Sound that would be present in the absence of a specified activity. Ambient sound can be anthropogenic (e.g. industrial or shipping) or natural (e.g. wind, biota).
Auditory frequency weighting	The process of band-pass filtering sounds to reduce the importance of inaudible or less-audible frequencies for individual species or groups of species of aquatic mammals. In other terms, a frequency weighting function that compensates for a species' (or functional hearing group's) frequency-specific hearing sensitivity.
Decibel (dB)	Decibel—a unit of measurement used to express sound level. Decibels express the ratio of sound relative (re) to a reference level on a logarithmic scale. For airborne noise the reference level is 20 µPa, while for underwater noise the reference level is typically 1 µPa.
Frequency (Hz)	The number of times a vibrating object oscillates (moves back and forth) in one second. Fast movements produce high frequency sound (high pitch/tone), but slow movements mean the frequency (pitch/tone) is low. 1 Hz is equal to 1 cycle per second.
Hearing group	Category of animal species when classified according to their hearing sensitivity and to the susceptibility to sound. Examples for marine mammals include low-frequency (LF) cetaceans, high-frequency (HF) cetaceans, very high-frequency (VHF) cetaceans, otariid pinnipeds in water (OPW), phocid pinnipeds in water (PPW), sirenians (SI), other marine carnivores in air (OCA), and other marine carnivores in water (OCW). (Southall et al. 2019).
Hearing threshold	The hearing threshold represents the lowest signal level an animal can detect at a particular frequency, usually referred (and measured) as the threshold at which an animal will indicate detection 50% of the time.
Impulsive sound	Transient sound that has extremely short duration and a high peak sound pressure level.
MFO Level 1	Marine Fauna Observer, Level 1. A person with qualifications in ecology, zoology or environmental sciences and demonstrated experience with the identification and management of dolphins or whales, including behaviour, as well as distance estimation.
MFO Level 2	Marine Fauna Observer, Level 2. A person who has sufficient experience in marine fauna identification and distance estimation.
Pa	Pascals – SI unit of sound pressure (note µPa is micro Pascals). Because the hearing range of humans and fauna is very large when expressed in Pa, the logarithmic decibel scale is typically used.
Peak particle velocity	The peak particle velocity (PPV) is the peak level of the particle velocity calculated from the vector formed by the magnitude of the three orthogonal components of the particle velocity over their measured time history.

Permanent threshold shift (PTS)	Permanent threshold shift (PTS) is a permanent reduction in hearing sensitivity caused by irreversible damage to the sensory hair cells of the ear.
re	Relative a reference level. For example dB re 1 µPa is decibels relative to a reference level of 1 µPa
Sound Exposure Level (SEL)	Level of the sound exposure as defined in ISO 18405. In underwater acoustics, the reference value of time-integrated squared sound pressure is 1 μPa^2 s.
$SEL_{24\ hour}$	The cumulative sound exposure level, which includes multiple acoustic pulses from piling or the time duration of dredging within a 24 hour period. It is also assumed for intermittent, repeated exposure that there is no recovery between subsequent exposures.
SEL_{SS}	Single-strike sound exposure level, representing the sound exposure from a impulse, for example during impact piling
Source level	Source level (SL) is the sound pressure level at a distance of 1 m from a hypothetical point source radiating the same amount of sound energy as the actual source. Units: dB re 1 $\mu Pa^2 \cdot m^2$ (sound pressure level), dB re 1 $\mu Pa^2 \cdot m^2$ s (sound exposure level).
Sound pressure level	Sound pressure level (SPL) is the root-mean-square sound pressure expressed in the decibel (dB) scale. Units: dB re 1 µPa (underwater), dB re 20 µPa (air).
Temporary threshold shift	Temporary threshold shift (TTS) is a temporary reduction in hearing sensitivity as a result of exposure to sound. Exposure to high levels of sound over relatively short time periods can cause the same amount of TTS as exposure to lower levels of sound over longer time periods. The duration of TTS varies depending on the nature of the stimulus.

Abbreviations

AEP	Auditory Evoked Potential
AUD INJ	Auditory Injury
BIA	Biologically Important Area
CCSZ	Cuttlefish Coast Sanctuary Zone
CEMP	Construction Environmental Management Plan
CNVMP	Construction Noise and Vibration Management Plan
dB	Decibel
dB_{peak}	Peak Sound Pressure Level
dB_{rms}	Root Mean Square Sound Pressure Level
DCCEEW	Department of Climate Change, Energy, the Environment and Water
DIT	Department for Infrastructure and Transport
DN	Nominal Diameter (pipeline size)
EIS	Environmental Impact Statement
EP Act	Environment Protection Act 1993 (SA)
EPBC Act	Environment Protection and Biodiversity Conservation Act 1999 (Commonwealth)
EPBS	Earth Pressure Balance Shield (tunnel boring machine)
FM Act	Fisheries Management Act 2007 (SA)
GL/y	Gigalitres per year
GRP	Glass Reinforced Plastic
HDPE	High-Density Polyethylene
HF	High-Frequency (cetaceans)
Hz	Hertz
JUB	Jack-Up Barge
kV	Kilovolt
LF	Low-Frequency (cetaceans)
MFO	Marine Fauna Observer
ML/d	Megalitres per day
MNES	Matters of National Environmental Significance
MW	Megawatt
NMFS	National Marine Fisheries Service (USA)
NPW Act	National Parks and Wildlife Act 1972 (SA)
NW	Northern Water

OCA	Other Marine Carnivores in Air
OCW	Other Marine Carnivores in Water
Pa	Pascal (SI unit of pressure)
PCA	Phocid Carnivores in Air
PCW	Phocid Carnivores in Water
PEZ	Potential Effects Zone
PL	Propagation Loss
PMST	Protected Matters Search Tool
PS	Pump Station
PTS	Permanent Threshold Shift
re	Relative to a reference level
RMS	Root Mean Square
RO	Reverse Osmosis
SEL	Sound Exposure Level
$SEL_{24\ hour}$	24-hour Cumulative Sound Exposure Level
SEL_{SS}	Single-Strike Sound Exposure Level
SL	Source Level
SPC	State Planning Commission
SPL	Sound Pressure Level
SWRO	Seawater Reverse Osmosis
TBM	Tunnel Boring Machine
TNVIA	Terrestrial Noise and Vibration Impact Assessment
TTS	Temporary Threshold Shift
UNVIA	Underwater Noise and Vibration Impact Assessment
VHF	Very High-Frequency (cetaceans)
PPV	Peak Particle Velocity
µPa	Micropascal

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1 Introduction

1.1 Northern Water project

The Northern Water Project (the Project) aims to provide a secure climate resilient water source for industry in South Australia's Upper Spencer Gulf and Far North. Delivery of the Project will involve the construction and operation of a 51 GL/y seawater reverse osmosis desalination plant located at Mullaquana Station on the Upper Spencer Gulf, and associated water transfer system (including pipelines, transfer pump stations, and water storages) and electricity infrastructure. The Project comprises the following key elements:

- a 51 GL/y seawater reverse osmosis desalination plant located at Mullaquana Station approximately 25 km south-west of central Whyalla
- marine intake and outlet tunnels extending underneath the seabed and into Spencer Gulf
- diffuser structures for discharge of return water into Spencer Gulf
- a transfer system consisting of:
    - ~360 km main transfer pipeline, largely constructed underground
    - ~46 km lateral branch pipeline to a delivery point at the Carrapateena copper-gold mine
    - associated transfer pump stations and storage tanks
- high voltage electricity supply including powerlines, network and facility substations to provide power to the desalination plant and transfer pump stations
- associated temporary construction-related infrastructure (e.g. construction camps, site offices and amenities, yards, storages and laydown areas)
- access tracks including upgrading existing tracks and establishing new tracks
- enabling roadworks and road upgrades.

Key elements are described in further detail in Section 2 (Table 2).

1.2 Scope of this assessment

This Underwater Noise and Vibration Impact Assessment (UNVIA) addresses potential underwater noise and vibration impacts from the Project, including mitigation strategies to eliminate or minimise impacts and will form part of the Environmental Impact Statement (EIS) for the Project under the *Planning, Development and Infrastructure Act, 2016*.

The Project was referred under the *Environment Protection and Biodiversity Conservation Act 1999* (the EPBC Act; referral 2025/10397) and determined to be a controlled action, with controlling provisions for listed threatened and migratory species. This UNVIA therefore addresses potential impacts on the relevant listed marine fauna.

Note that terrestrial noise and vibration impacts from the Project are addressed separately in the *Terrestrial Noise and Vibration Impact Assessment* (TNVIA) prepared by Resonate (March 2026).

1.2.1 Assessment requirements

This UNVIA has been undertaken in consideration of the State Planning Commission Assessment Requirements (SPC, 2026), which contains the following requirements in relation to underwater noise:

- *Undertake an underwater noise assessment covering both construction and operation of the desalination plant facility, with consideration for potential short and long term impacts to marine fauna.*
- *Demonstrate how the development will meet the General Environmental Duty under the Environmental Protection Act 1993 (EP Act) with respect to underwater noise.*
- *Describe and outline environmental management strategies to mitigate noise and vibration impacts of underwater noise on marine flora and fauna*

While vibration is not referenced in the above underwater noise assessment requirements, the 'general' assessment requirements in relation to noise and vibration include a requirement to consider the impacts of vibration on fauna (including marine fauna). For clarity, potential impacts on benthic marine fauna are addressed in this UNVIA.

The scope of this assessment in response to the above requirements is presented below in Table 1, along with which report section(s) address each requirement.

Table 1: State Planning Commission assessment requirements

Assessment requirement	Scope of this assessment to address requirement	Relevant report section(s)
Undertake an underwater noise assessment covering both construction and operation of the desalination plant facility, with consideration for potential short and long term impacts to marine fauna.	Undertake measurements to quantify the existing (baseline) underwater noise environment in the vicinity of the Project site.	Section 5.1 – Baseline underwater noise measurements
	Identify the significant sources of underwater noise and vibration for construction and operation phases of the Project, and establish the characteristics of these sources, including the source level, frequency spectral information and temporal characteristics.	Section 6.1 – Construction noise sources Section 6.2 – Operational noise sources
	Identify marine species potentially exposed to noise (and vibration if relevant) emitted by the proposed noise-generating activities, and their ecological characteristics.	Section 3.3 – Fauna included in this assessment Section 3.4 – Marine fauna hearing groups
	Assessment is based on the 'design' capacity requirement of 51 GL/year, noting any future expansion beyond this would be subject to further assessment.	Section 7 – Underwater noise modelling
	Both short term (dB_{peak}, dB_{rms}, VPPV) and cumulative exposure ($SEL_{24\ hour}$) assessment metrics are used, consistent with literature and guidance on the impacts of noise and vibration on marine fauna	Section 8 – Marine fauna underwater noise assessment Section 10 – Benthic invertebrates underwater vibration assessment

Assessment requirement	Scope of this assessment to address requirement	Relevant report section(s)
Demonstrate how the development will meet the General Environmental Duty under the EP Act with respect to underwater noise.	Compare the predictions to the underwater noise criteria to determine zones of impact. Assess the likely impacts using a risk-based approach, in accordance with the Underwater Piling and Dredging Noise Guidelines (2023) (the *Guidelines*), developed by the Department for Infrastructure and Transport (the Department). Adoption of an assessment approach consistent with the Guidelines will meet the General Environmental Duty under the EP Act with respect to underwater noise, along with EPBC Act requirements.	Section 4 – Assessment criteria Section 4.3.1 – Underwater Piling and Dredging Noise Guidelines Section 8 – Marine fauna underwater noise assessment Section 10 – Benthic invertebrates underwater vibration assessment
Describe and design environmental management strategies to mitigate noise and vibration impacts of underwater noise on marine flora and fauna.	Identify and propose reasonable and practical management and mitigation measures to minimise or eliminate the adverse underwater noise and vibration effects for the project.	Section 9 – Underwater noise mitigation and management Section 10.4 – Vibration mitigation and management

1.3 Input information

In preparing this UNVIA, Resonate have relied primarily, but not exclusively, on input information contained in the following documents:

- *Reference Design Report: Northern Water Project* (GHD, 3 October 2025)
- *Northern Water Supply Project - Reference Design Constructability Report* – SYS-R-NWSP-006-R3 (Systra, 20 November 2025)

Where other input information has been relied on, this is noted in the following sections.

2 Project description

Key infrastructure listed in Section 1 is further described in Table 2. Note that for assessment of underwater noise and vibration impacts from the Project, only construction and operation of intake and outlet infrastructure is relevant. Description of other Project components is provided here for information and context only and to align with the Terrestrial noise report.

Table 2: Project component descriptions

Project component	Description
Desalination plant	
Location	• Mullaquana Station (Crown lease) located 25 km south-west of central Whyalla • Proposed site located ~14 km east of the Lincoln Highway. Currently accessed by unsealed local roads and private station tracks. • Desalination plant main site is ~1.5 km from the coast of the Upper Spencer Gulf • Possible sensitive receptors are: - Mullaquana Station homestead – 5 km west - Cowleds Landing – 8 km north - Murninnie Beach – 10.5 km south
Access	• Access proposed from Lincoln Highway via Eight Mile Creek Road.
Desalination plant	• Seawater reverse osmosis desalination plant - sized for design conditions of an average annual capacity of 51 GL/y and turn down capacity of 40 ML/day (and a hypothetical highest capacity understood to be 73 GL/y) - one pass reverse osmosis (RO) system operating at 40% overall plant recovery (assumed)
Intake and outlet pipe	• Marine intake and outlet tunnels and pipes constructed under the seabed using tunnel boring machines (TBMs) • TBMs to be launched from a temporary box cut and decline constructed in south-east corner of the construction site • A water intake pump station and pipeline tunnel extending under the seabed to approximately 13 m depth into the Spencer Gulf with associated intake pumps and intake head (overall length of approximately 3.9 km) • An outlet pipeline tunnel extending under the seabed and terminating at a minimum 17 m water depth at the tunnel riser / outlet diffuser (approximate length 4.8 km)[1].
Operating hours	• Desalination plant is expected to be staffed twenty-four hours per day / 7 days per week with most maintenance activities being undertaken during daylight hours or from 7:30 am to 4:00 pm, or as negotiated by the Contractor.
Construction hours	• Standard construction hours will be 7am-7pm Monday to Saturday. Depending on construction scheduling, some work may need to occasionally be undertaken outside these hours in accordance with regulatory requirements.

[1] An alternative construction method may be considered for part of the outfall pipe. This would involve a 'float and sink' method, which involves floating the pipeline into position after tunnelling reaches a suitable depth, then lowering and securing it to the seabed using piling. The outfall pipe landward of the intake location would still be tunnelled under the seabed.

Project component	Description
Construction	• Civil and building works will be sized and constructed for the design capacity of 51GL/y • RO trains will be designed and built for the design capacity of 51GL/y
Total construction footprint	• Approximately 200 ha of land would be required for the purposes of construction (on shore)
Water transfer system	
Water transfer pipeline	• Total length ~406 km • ~360 km main transfer pipeline extending north from the desalination plant to Olympic Dam • ~46 km offtake pipeline to the existing Carrapateena mine provision (i.e. blank interface flanges) for future additional offtake points • ~1050 air valves 250 scour valves • ~140,000 tonne of pipeline (or 7000 semi-trailers) • ~2 million m^3 of excavation • standard width for construction corridor = 40 m which can be constrained in areas of high sensitivity • 500 linear metres /day (spread length ~200m)
Pipeline materials and sizing	• PN35 mild steel cement lined (MSCL) pipeline • Pipe diameters will vary along the length of the pipeline alignment. Nominal size leaving the desalination plant of DN1200, reducing to DN1000 at Olympic Dam (noting some sections of DN1100). • Carrapateena offtake nominal diameter DN450 (DICL) pipeline
Pipeline construction	• Main trunk and branch pipeline are proposed to be buried, constructed using trench and cover fill for the majority of the alignment (note that the Carrapateena branch pipeline may be constructed above or below ground) • Trenchless installation (e.g. micro-tunnelling / auger-boring) will be used for rail and major road / highway crossings and some watercourse crossings. Above ground sections of pipeline (where the pipeline will be constructed on raised structure on footings) may be designed for crossing of environments sensitive to excavation. • Current estimates for trenchless crossings: - 9 x major creeks - 1 x No Go Zone - 5 x rail crossings • Depth of the buried pipeline will depend on the pipe diameter for the various sections of the pipeline, and in line with minimum cover values calculated using AS/NZS 2566.1:1998 Buried Flexible Pipelines. • Minimum depth of cover will also vary with prevailing land use, with a minimum of 600 mm depth of cover provided in pastoral areas. Where pipeline crosses major roads, cover will be minimum of 1000 mm.
Transfer pump stations (PS) and water storage (south to north alignment)	• 4 transfer pump stations (PS) - PS 1 – Mullaquana pump station (desalination plant site) - PS 2 – Lincoln Gap pump station and storage - PS 3 – Kalaya pump station (Kootaberra) and storage - PS 4 – Mangata pump station (Oakden Hills) and storage • Bimba (Pimba) high point storage

Project component	Description
Electricity supply	
Transmission lines	• 4 new transmission lines (connections to desalination plant and pump stations) - Mullaquana 132 kV (approx. 20 km) - Pandurra 275 kV (approx. 3 km) - Bookaloo 132 kV (approx. 4 km) - Oakden Hills 33 kV (approx. 15 km)
Network substations	• 2 new network substations - Mullaquana substation (serves 132kV line and desalination plant and intake pump station) - Kootaberra substation • 2 upgraded network substations - Coraberra Hill substation (ElectraNet) (275kV) - Mount Gunson substation (SAPN) (33 kV)
Pump station substations	• Mullaquana Station desalination plant pump station • Lincoln Gap • Kalaya pump station (Kootaberra) • Mangata pump station (Oakden Hills)
Power supply to storage tanks	• Bimba storage – solar panels array • Carrapateena and Olympic Dam storage will utilise existing power sources

3 Marine fauna and underwater noise

This section provides an overview of underwater noise and the effects that may occur when an animal is exposed to anthropogenic underwater noise.

3.1 What is underwater noise?

Sound is an acoustic pressure wave that travels through a medium, such as water or air, and occurs as an oscillatory motion of the water or air particles driven by a vibrating source. The magnitude of the water or air particle motion determines the intensity of the sound. The rate at which the water or air particles oscillate determines its frequency, given in cycles per second measured in Hertz (Hz).

Sound travels about four-and-a-half times faster in water than in air. The absorption of sound at frequencies where anthropogenic noise generally has the most energy is much smaller in water than in air. As a result, noise is typically audible over much greater ranges underwater than in air. Most sources of underwater noise, including pile driving, and movement of large shipping vessels generate acoustic energy over a broad range of frequencies. Screeching or whistling noises are composed mainly of high frequency sounds while rumbles or booms are composed mainly of low frequency sounds.

Sounds are usually characterised according to whether they are continuous or impulsive in character. Continuous sounds occur without pauses and examples include shipping noise and dredging. Impulsive sounds are of short duration and can occur singularly, irregularly, or as part of a repeating pattern. Blasting represents a single impulsive event whereas the periodic impacts from a pile driving rig results in a patterned impulsive sequence. Impulsive signals typically sound like bangs and generally include a broad range of frequencies.

3.2 Overview of noise effects

The following provides a brief overview of the effects that may occur because of an animal being exposed to underwater noise. Consideration of this information, together with information on the biological importance of the habitat for the considered species, e.g. breeding, calving or resting areas, or confined migratory routes or feeding areas, is used to assess the likely impact of a noise source.

- Risk of fatality – When exposed to significant noise levels, either immediate mortality or tissue and/or physiological damage can result. The injury may be sufficiently severe that death occurs sometime later due to decreased fitness. Mortality can also have a direct effect upon animal populations, especially if it affects individuals close to maturity. Tissue and other physical damage or physiological effects, that are recoverable, but which may place animals at lower levels of fitness, may render them more open to predation, impaired feeding and growth, or lack of breeding success, until recovery takes place.
- Hearing injury – Temporary threshold shift (TTS) or permanent threshold shift (PTS) in hearing sensitivity may, or may not, reduce fitness and survival. Impairment of hearing may affect the ability of animals to capture prey and avoid predators, as well as cause deterioration in communication between individuals. This may affect growth, survival, and reproductive success.
- Masking – The presence of man-made sounds may make it difficult to detect biologically significant sounds against the noise background. Masking of sounds from predators may result in reduced survival. Masking of sounds used for communication, orientation and navigation may affect the ability to find preferred habitats and in the case of fish, spawning areas, affecting recruitment, growth, survival, and reproduction.
- Behavioural responses – Behavioural responses may cause displacement from preferred habitats, which could affect feeding, growth, predation, survival, and reproductive success (if a mammal is displaced from preferred habitat).

Figure 1 acknowledges that the severity of noise effects relates to distance from the noise source, however, note that the ‘zones’ of hearing injury, masking and behavioural response may overlap. Overlap results from comparing cumulative sound exposure threshold metrics with single event peak or behavioural sound level metrics.

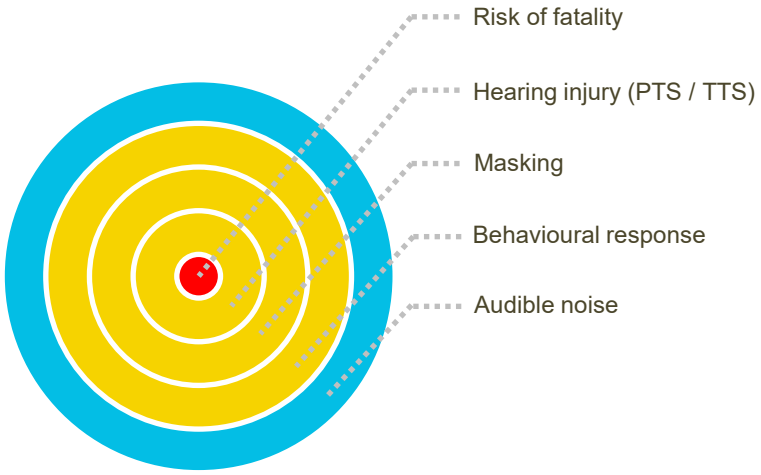


Figure 1: Overview of potential noise effects upon marine fauna (source: DIT)

The threshold of underwater noise that has the potential to result in masking of communication or other biologically significant sound is difficult to quantify in practice because it is highly dependent on the existing ambient noise environment, frequency range of vocalisation of relevant species, and their ability to adapt vocalisation range and energy to overcome a change in the noise environment. This can vary greatly from species to species (even within the same hearing group) and there are limited studies or guidelines which attempt to define sound level thresholds above which masking may occur.

A reasonable approach is to consider that the guideline values for behavioural response presented in Section 4, or the existing ambient noise levels (whichever is higher), represent a conservative threshold for sound levels which may result in masking.

3.3 Fauna included in this assessment

3.3.1 Nationally protected fauna

The following species or communities have been identified using the Department of Climate Change, Energy, the Environment and Water (DCCEEW) Protected Matters Search Tool (PMST) using a 5 km buffer from the Project Area. Please refer to the EPBC referral (2025/10397) for information on category/status under the EPBC Act and *National Parks and Wildlife Act 1972*, and likelihood of occurrence.

Table 3: Protected species and communities

Species, or Community
Threatened Fauna - Reptiles
Caretta caretta (Loggerhead Turtle)
Chelonia mydas (Green Turtle)
Dermochelys coriacea (Leatherback Turtle, Leathery Turtle)
Threatened Fauna - Mammals
Eubalaena australis (Southern Right Whale)
Neophoca cinerea (Australian Sea-lion)

Species, or Community
Threatened Fauna – Fish
Seriolella brama (Blue Warehou) (1)
Threatened Fauna – Shark
Carcharodon carcharias (White Shark, Great White Shark)
Galeorhinus galeus (School Shark, Eastern School Shark, Snapper Shark, Tope, Soupfin Shark) (1)
Threatened Fauna - Birds
Diomedea antipodensis (Antipodean Albatross)
Diomedea epomophora (Southern Royal Albatross)
Diomedea exulans (Wandering Albatross)
Macronectes giganteus (Southern Giant Petrel)
Macronectes halli (Northern Giant Petrel)
Pachyptila turtur subantarctica (Fairy Prion)
Phoebetria fusca (Sooty Albatross)
Thalassarche carteri (Indian Yellow-nosed Albatross)
Thalassarche cauta (Shy Albatross)
Thalassarche impavida (Campbell Albatross, Campbell Black-browed Albatross)
Thalassarche melanophris (Black-browed Albatross)
Thalassarche steadi (White-capped Albatross
Ardenna grisea (Sooty Shearwater)
Migratory Species (not considered as threatened species above)
Apus pacificus (Fork-tailed swift)
Ardenna carneipes (Flesh-footed Shearwater)
Balaenoptera edeni (Bryde's Whale)
Caperea marginata (Pygmy Right Whale)
Lagenorhynchus obscurus (Dusky Dolphin)
Lamna nasus (Porbeagle, Mackerel Shark)
Carcharias taurus, (Grey Nurse Shark)
Megaptera novaeangliae (Humpback Whale)

(1) Conservation dependent, not MNES

3.3.2 Other species

The following species are not listed as threatened or migratory under the EPBC Act, however are known to have biologically or commercially important areas within the vicinity of the Project, and are therefore considered in this assessment.

Giant Australian Cuttlefish

The Giant Australian Cuttlefish (*Sepia*[2] *apama*) is an iconic species that occurs across the southern coastline of Australia. The Upper Spencer Gulf population of cuttlefish represents a distinct population which forms the largest known breeding aggregation in the world, along a small 8 km stretch of subtidal reef around Point Lowly, approximately 34 km northeast of the proposed Northern Water desalination plant intake and outlet structures. This unique aggregation area has recently been declared a National Heritage Place under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) as the Cuttlefish Coast Sanctuary Zone (CCSZ) and is highly valued from an ecological perspective and by the community of South Australia.

The species may be susceptible to changes in environmental conditions at their breeding grounds particularly considering their eggs are fixed to the rocky substrate (and thus, unable to move away from intolerable conditions).

Whilst underwater noise and vibration levels due to the Project are expected to be negligible at 34 km, outside of breeding season cuttlefish disperse throughout the northern Spencer Gulf where they are found close to the seafloor in deeper water up to 30 m deep (TSSC, 2011). A migration 'pathway' for the species is not evident as it appears (predominantly from fisheries bycatch data) that the species aggregates to breeding sites from all areas around northern Spencer Gulf (Gillanders and Payne, 2014).

The species may traverse the Project site at times, and as a precautionary measure, potential underwater noise and vibration impacts on the Giant Australian Cuttlefish have been considered in this assessment.

Other invertebrates

The bare sand and seagrass habitat in the vicinity of the Project is utilised by benthic invertebrates species such as Blue Crab (*Portunus armatus*) and Western King Prawn (*Penaeus (Melicertus) latisulcatus).* Whilst not listed under the under the EPBC Act, these are commercially important fisheries species and have therefore been considered in this assessment.

Foraging seabirds

The proposed desalination plant site is within foraging-related Biologically Important Areas (BIAs) for Short-tailed Shearwater, Black-faced Cormorant and Fairy Tern.

Commercially important fish

The Spencer Gulf, including areas in the vicinity of the Project, is habitat for commercially important fisheries such as Snapper. Although not directly assessed, impacts on Snapper and other commercial fish species are expected to be consistent with those on other fish, e.g. *Seriolella brama* (Blue Warehou).

[2] Genus has recently changed to *Ascarosepion.* Species name is now '*Ascarosepion apama'* (MolluscaBase 2026) but this report uses 'Sepia apama' for consistency with EPBC Act documentation

3.4 Marine fauna hearing groups

For the purpose of assessment of potential underwater noise and vibration impacts, marine fauna are grouped into the following functional hearing groups based on hearing mechanism, sensitivity and frequency range.

3.4.1 Marine mammals

Marine animals live in an environment in which vision is not the primary sense because light does not penetrate far beneath the surface of the ocean. As such, marine mammals are reliant upon sound, instead of sight, as their primary sense for communication and being aware of their surrounding environment. Marine mammal vocalisations have a variety of functions such as intra-sexual selection, mother/calf cohesion, group cohesion, individual recognition, echo-location, predation, and danger avoidance.

Baleen whales produce sounds that are primarily at frequencies below 1kHz and have durations from approximately 0.5 to over 1 second and sometimes much longer (Richardson et al. 2013). Humpback whales and some other species produce sounds with frequencies above 1kHz. Many baleen whale sounds are uncomplicated tonal moans or sounds described as knocks, pulses, ratchets, thumps, and trumpet-like. Blue whales for example produce low frequency moans in the frequency range of 10-15Hz.

Pinnipeds, including hair and eared seals and sea lions, produce underwater vocalisations sounding like barks and clicks with frequencies ranging from below 1kHz to 4kHz (Richardson et al. 2013). Pinnipeds are especially vocal during the breeding season.

In summary, baleen whales produce sounds that are dominant at frequencies that overlap with anthropogenic industrial noise (e.g. drilling). In contrast, the social sounds produced by pinnipeds occur above the low-frequency range where most man-made sounds have their dominant energy (with the exception of sonar).

Marine mammal hearing sensitivity

The hearing sensitivity of marine animals generally varies with frequency. Audiograms are therefore used to represent an animal's sensitivity to sounds of different frequencies. An audiogram of a species relates the absolute threshold of hearing (in dB re 1μPa) of that species to frequency. An animal is most sensitive to sounds at frequencies where its absolute threshold of hearing is lowest. As an example, human beings are most sensitive to sounds between 2-4kHz where the absolute threshold is lowest.

Substantial progress has been made in quantifying marine mammal hearing and the effects of noise on hearing for a range of taxa since the review provided by Southall et al. (2007). Southall et al. (2019), in light of subsequent scientific findings over the past decade, presents estimated audiograms for six species groupings, including all marine mammal species. In-air audiograms are also provided for amphibious species.

The intent of Southall et al. (2019) was to provide the best scientific interpretation and application of the available information within different marine mammal hearing groups while acknowledging data limitations that existed at the time for some hearing groups. These audiograms form the basis for the United States' National Marine Fisheries Service (NMFS) *Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing (Version 3.0),* released in 2024 (NMFS, 2024). The criteria used in this assessment for marine mammals are derived from the NMFS 2024 guidance, as described in Section 4.4.1.

Further recent advances have been made since the release of the Version 3.0 guidelines, including Houser et al. (2024) direct hearing measurements in minke whales showing hearing up to ~45-90 kHz; behavioural audiometry on humpback whales giving a masked hearing curve for the frequency range 0.25-16 kHz (Dunlop et al. 2025); and a predicted humpback whale hearing curve based on modified behavioural observation audiometry data which suggests potentially reduced sensitivity at low frequencies and increased sensitivity at higher frequencies compared to previous predictions based on anatomy (Dunlop et al. 2026).

Houser (2025) also undertook a comprehensive review of marine mammal hearing sensitivity, with findings consistent with the material summarised above.

The following sections briefly summarise the audiograms that relate to specific hearing groups of marine mammals considered in this assessment.

Low frequency cetaceans

The low frequency (LF) cetaceans relevant to this study include:

- Humpback Whale (*Megaptera novaeangliae*)
- Southern Right Whale (*Eubaleana australis*)
- Pygmy Right Whale (*Caperea marginata*)
- Bryde's Whale (*Balaenoptera edeni*).

The Southall et al. (2019) estimated audiogram for this hearing group is presented in Figure 3.

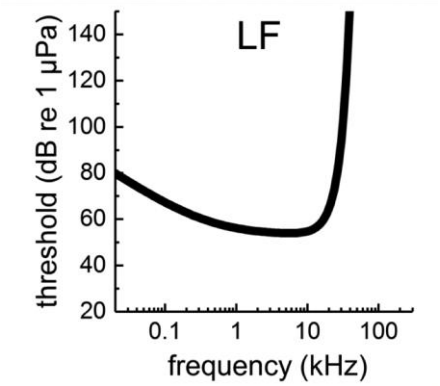


Figure 2: Estimated group audiogram for low-frequency (LF) cetaceans (Southall et al., 2019)

In relation to the audiogram presented in Figure 2, Southall et al. (2019) advise that no direct hearing data (behavioural or electrophysiological) were available at the time, at any frequency for any species. That is, there were no comprehensive, directly measured audiograms for any baleen whale from which an estimate of the LF cetacean group audiogram could be made. The audiogram presented above is constructed from anatomy-based predictions using biomechanical models of the ear, alongside known vocalisation frequency ranges.

Baleen whale vocalisations are low in frequency content for a number of species, and the frequency range of acute hearing presumably includes the frequency range of vocalisations. From behavioural observations, it is apparent that baleen whales are quite sensitive to frequencies below 1kHz, but can hear sounds up to a considerably higher but unknown frequency (Richardson et al. 2013). The audiogram in Figure 2 indicates that the frequencies where the hearing is most sensitive ranges from 100Hz to 10kHz.

As noted above, Houser et al. (2024) since undertook direct measurements in minke whales showing hearing up to ~45-90 kHz, and the behavioural audiometry studies of Dunlop et al. (2025, 2026) indicate increased sensitivity at higher frequencies than previously assumed, alongside potentially reduced sensitivity at low frequencies. Although these recent findings are significant, it remains unclear if they can be generalised to the hearing group as a whole.

In relation to this project, the dominant underwater noise sources are generally low frequency in nature. The above recent findings indicate the audiogram presented in Figure 2 is a conservative basis for this assessment because it possibly overstates sensitivity at low frequencies.

High frequency cetaceans

The high frequency (HF) cetaceans relevant to this study include:

- Dusky Dolphin (*Lagenorhynchus obscurus*)

The Southall et al. (2019) estimated audiogram for this hearing group is presented in Figure 3.

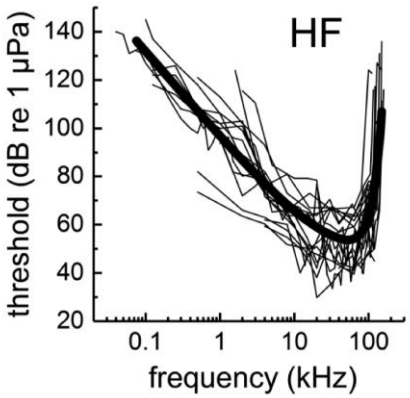


Figure 3: Estimated group audiograms based on original behavioural threshold data for high-frequency (HF) cetaceans (Southall et al., 2019).

The audiogram in Figure 3 indicates that the frequencies where the hearing is most sensitive for this group ranges from 8kHz to 100kHz. Note that noise from shipping, drilling and piling activities occur predominantly in the frequency region where the high frequency cetaceans hearing is the least sensitive. Auditory evoked potential measurements for several previously untested odontocete species (Houser et al. 2022) are consistent with the estimated group audiograms presented above.

Pinnipeds

Pinnipeds, comprise the extant families *Odobenidae* (walrus), *Otariidae* (the eared seals: sea lions and fur seals), and *Phocidae* (the earless seals, or true seals). The relevant species to this study is the Australian Sea Lion (*Neophoca cinerea*) which is in the 'Other Carnivores' group according to Southall et al., 2019.

The audiograms for Phocid Carnivores in water (PCW), Phocid Carnivores in air (PCA) and Other Carnivores in water/air (OCW / OCA) are presented in Figure 4 and Figure 5. There are no protected species in the Phocid Carnivores category in the vicinity of this project based on the Protected Matters Search Tool (PMST), however these audiograms are provided for information purposes.

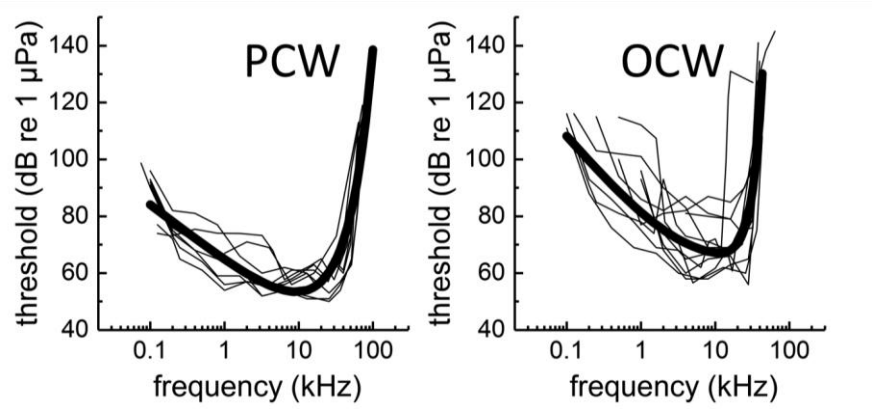


Figure 4: Estimated group audiograms based on original behavioural threshold data for phocid carnivores in water [PCW] and other carnivores in water [OCW] (Southall et al., 2019)

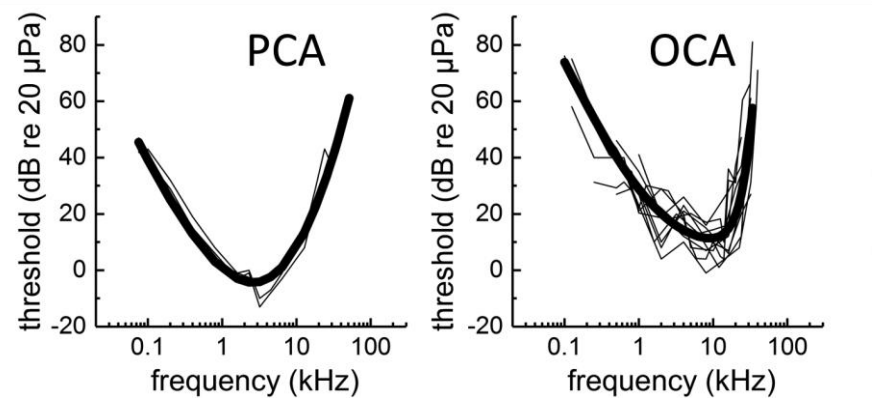


Figure 5: Estimated group audiograms based on original behavioural threshold data for phocid carnivores in air [PCA] and other carnivores in air [OCA] (Southall et al., 2019)

The audiograms indicate that phocid carnivores are generally more sensitive over a wider frequency range than other carnivores in both water and air. Phocid carnivores are also more sensitive to the lower frequencies where piling and shipping noise emissions are mostly dominant. Of note, the maximum hearing sensitivity of phocid carnivores is 10 kHz in water and 3 kHz in air, while other carnivores are similar at around 10 kHz in both water and air. Furthermore, both species have greater hearing sensitivity in water than in air, particularly at lower frequencies.

In comparison to toothed whales (i.e. high frequency cetaceans), pinnipeds generally tend to have maximum hearing sensitivity at lower frequencies and therefore more able to hear man-made noise.

3.4.2 Fish and sharks

All fishes have ears to detect sound and convey sensitivity to gravity and to linear and angular acceleration (Popper et al. 2014). The adaptations that provide fish with additional sensitivity to sound pressure are gas-filled structures near the ear and/or extensions of the swim bladder that functionally affect the ear. The enclosed gas changes volume in response to fluctuating sound pressure, generating particle motion. In fishes where the swim bladder is near the ear (or connected to it mechanically as in the *Otophysi)*, the particle motion radiated from the bladder is sufficiently large to cause the sensory epithelium to move relative to the otolith. Fishes with these adaptations generally have lower sound pressure thresholds and wider frequency ranges of hearing than do the purely particle motion-sensitive species.

Conversely, fish species that lack a gas-filled cavity, including sharks, are not as vulnerable to trauma from extreme sound pressure changes as fish with a gas-filled space. This difference has been demonstrated by comparing the effects of pile driving sounds on fishes with and without a swim bladder (Halvorsen et al. 2012a).

Hearing abilities among sharks have demonstrated highest sensitivity to low frequency sound (40Hz to approximately 800Hz), which is sensed solely through the particle-motion component of an acoustical field. Free-ranging sharks are attracted to sounds possessing specific characteristics: irregularly pulsed, broad-band (most attractive frequencies: below 80Hz), and transmitted without a sudden increase in intensity. Such sounds are reminiscent of those produced by struggling prey. A sound, even an attractive one, can also result in immediate withdrawal by sharks from a source, if its intensity suddenly increases 20 dB or more above a previous transmission (Myrberg 2001).

3.4.3 Marine turtles

Data on hearing of marine turtles is very limited. Electrophysiological studies on hearing have been conducted on juvenile green sea turtles, juvenile Kemp's Ridleys, and on juvenile loggerheads. Ridgway et al. (1969) obtained an auditory evoked potential (AEP) audiogram to aerial and vibrational stimuli that extended from below 100 Hz to 2000 Hz with the lowest threshold at 400 Hz. Other studies using AEPs found similar low-frequency responses to vibrations delivered to the tympanum (the external ear on the surface of the head) for the loggerhead sea turtle, and to underwater sound stimuli for the loggerhead, Kemp's Ridley, and green sea turtles (Bartol et al. 1999)

The limited state of knowledge regarding noise impacts on marine turtles is highlighted in the systematic review of Nelms et al. (2016). While that review relates primarily to seismic surveys, which are not proposed for this Project, its findings regarding turtle sound detection are consistent with the information presented above.

Martin et al. (2012) measured underwater thresholds in the Loggerhead sea turtle (*Caretta caretta*) by both behavioural and AEP methods. Behavioural sensitivity showed the lowest thresholds between 100 and 400 Hz, with thresholds at about 100 dB re 1 µPa. AEP measurements on the same individual were up to 8 dB higher; however, both techniques showed a similar frequency response and a high frequency loss of sensitivity above 400 Hz of about 37 dB per octave. These results are presented in Figure 6.

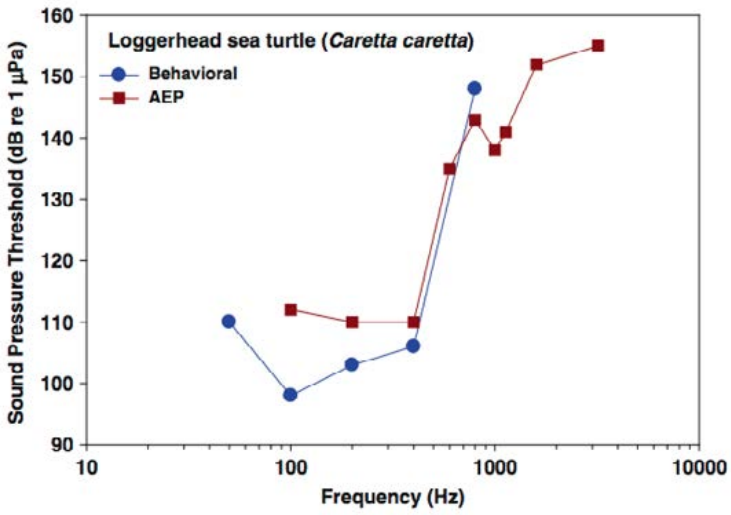


Figure 6: Behavioural and auditory evoked potential thresholds for the Loggerhead sea turtle (Martin et al. 2012)

Morphological examinations of green and loggerhead sea turtles (Ridgway et al. 1969) describe the sea turtle as having a typical reptilian ear with a few underwater modifications, supporting the proposal that fish hearing, rather than mammalian hearing, is the better model to use for sea turtles until there are much more data.

3.4.4 Marine invertebrates

In a sound wave, particles of the medium (e.g. water) oscillate around a point of origin, causing local compressions and expansions that transfer sound energy to neighbouring particles. Sound involves both pressure and particle motion fluctuations. Sound pressure fluctuations are omnidirectional and measured in Pascals (Pa). Sound particle vibrations are directional and described by displacement (m), velocity (m/s), or acceleration (m/s^2).

Cephalopods (cuttlefish, squid, octopus, and their relatives) form a varied group of molluscs that occupy a wide range of habitats, from coastal to pelagic waters and the deep sea. All cephalopods, as well as some bivalves, molluscs (common name mussels), and crustaceans, have a sac-like structure called a statocyst, which includes a mineralised mass (statolith) and associated sensory hairs (Edmonds et al., 2016).

Invertebrate statocysts can be defined as internal receptor systems, analogous to the vertebrate inner ear (otolith organ), that act as equilibrium receptor systems. However, most are thought to be gravity receptor systems only (Anken and Rahmann, 2002). In addition, statocysts of cephalopods and decapod crustacea include angular acceleration detector systems (Budelmann, 1988; Budelmann,1992a). In these groups, the statocyst, as linear accelerometers, can also detect acoustic particle motion (since the whole animal vibrates together with the water column) and are involved in underwater hearing (Budelmann 1992a; Budelmann 1992b). Statocysts present a different range of complexity, from the simplest gravity receptor systems to the more complex organs of cephalopods, which show receptor systems for linear and angular accelerations (Budelmann, 1992b).

Compared to marine mammals, fishes and sea turtles, the scientific knowledge of the biological significance of sound perception and production in marine invertebrates is scarce (Solé et al. 2023). However, several laboratory and field studies have been undertaken in recent years, which provide some insight into the range of sound levels that can potentially cause behavioural impacts or trauma (injury) to invertebrates.

For example, Samson et al. (2014) examined the behavioural responses of adult Common cuttlefish (*Sepia officinalis*) to pure-tone pips, and found that escape responses (inking, jetting) were observed between frequencies of 80 and 300 Hz and at sound levels above 140 dB re 1 µPa RMS and 0.01 ms^2 (0.74 ms^2 for inking responses).

Solé et al. (2022) studied the physiological effects of Common cuttlefish at all life stages in response to simulated impact piling and drilling noise and various levels. Damage was observed in the statocyst sensory epithelia (hair cell extrusion) in adults when they were exposed to maximum pile-driving and drilling sound levels. However, sound levels lower than 163 dB re 1 $µPa^2$ were not found to elicit severe damage.

Crustaceans (i.e. lobsters, crayfish, crabs and prawns) have been studied with respect to a broad range of metrics including catch rates and physical, behavioural, and physiological effects (Edmonds et al., 2016). Crustaceans are sensitive to low frequency acoustic stimuli (Salmon and Horch, 1972; Goodall et al., 1990; Roberts et al., 2016). Mechanical disturbances of water/sediment (associated with sound waves) are detected by a pair of statocysts, chordotonal organs linked to joints of antenna or legs and internal and external sensilla (Popper et al., 2001; Breithaupt, 2002). Crustaceans present sensory hairs covering their bodies, which when stimulated by water or substrate-borne vibrations associated with changes in acceleration, hydrodynamic flow or sound, help animals sense nearby biological movements (Tautz and Sandeman, 1980; Radford et al., 2016).

Animals exposed to low-frequency sounds may suffer physical damage, such as changes in hearing threshold or barotraumatic ruptures. Morphological or histological analysis allows the detection of physical trauma, including internal injuries, damage to sensory cells of statocysts and epidermal sensory cells, as well as neurons, which can lead to death. This trauma can affect structures involved in sound perception (Solé et al. 2023).

Invertebrates can behaviourally respond to sound in various ways, including increased aggressiveness, alarm responses, predator defence, orientation, and habitat selection, which could have consequences for reproduction and survival. Stress bioindicators such as hormones, immune responses, heat shock proteins, cardiac physiology, and overall degraded body condition are the main physiological responses. Metabolic rate, which is the most direct

indicator of stress, can be measured from respiration, oxygen consumption, or feeding rate. In some cases, irreversible DNA damage has been reported (Solé et al. 2023).

Solé et al. 2023, "Marine invertebrates and noise" provides a comprehensive summary of the current scientific knowledge on invertebrate bioacoustics (sound production, reception, sensitivity), and how marine invertebrates are affected by anthropogenic noise. Table 4 provides a summary of the relevant recorded noise impacts on invertebrates to this study.

Table 4: Observed effects of noise on invertebrates relevant to the Project study (Solé et al., 2023)

Species	Common name	Stage	Sound effects	Sound source	Levels	Reference
Sepia officinalis	Common cuttlefish	Adult/ Larva/ Eggs	**Damage to sensory systems** Cellular damage to the statocysts and lateral line system (adult and larva). Decreased larval survival rate. Decreased hatching success.	Pile driving playback Drilling playback	Max. 170 dB re 1 µPa. < 163 dB re 1 µPa no severe damage Max: 167 dB re 1 µPa	(Solé et al., 2022)
			Behaviour Escape responses (inking, jetting)	Pure tone sound playback	140 dB re 1 µPa RMS	Samson et al. (2014)
Callinectes sapidus	Blue crab	Adult	**Mortality** No effects	Underwater explosions Vessel noise	< 1 kHz	(Moriyasu et al., 2004)
			Behaviour Changes competitive behaviour. **Physiology** Increase haemolymph glucose.	Mid-frequency sonar	1-s 1.67 kHz / 2.5 to 4.0 kHz 1-s	(Hudson et al., 2022)
			Behaviour No impact on olfactory-mediated foraging. No cross-modal effects	Natural sounds of predators and soundscape	Atlantic croaker (*Micropogonias undulates*) and red drum (*Sciaenops ocellatus*) and marine background sounds	(Solé et al., 2023)
			Behaviour No impact on olfactory-mediated foraging No cross-modal effects. **Physiology** Righting reflex, damaged sensory statocyst epithelia	Sinusoidal wave sweep	171 dB of 1 µPa; max 180 dB of 1 µPa	

Species	Common name	Stage	Sound effects	Sound source	Levels	Reference
			No damaged antennule or eye sensory epithelia			
Sepioteuthis australis	Southern calamari or Southern reef squid	Adult	**Stress / behaviour** Alarm responses Aggression jetting	Seismic airgun	168-173 dB re 1 µPa	(Fewtrell & McCauley, 2012)
Palaemon serratus	Common prawn	Adult	**Behaviour** Change in locomotor patterns. **Physiology** Change in haemolymph and brain total protein content, DNA fragmentation Change in brain protein (HSP 27, HSP 70) level expression.	Boat noise (Laboratory experiments)	Power spectrum peaks up to 140 dB re 1 µPa RMS in the frequency band 0.1-3 kHz	(Filiciotto et al., 2016)

The acoustic perception of invertebrates is typically low frequency in character, that is less than 1500 Hz, with several species thresholds occurring within 20 to 300 Hz. The Common Prawn (*Palaemon serratus*) has a wider range of acoustic perception 100 – 3000 Hz (Solé et al. 2023).

Table 5 provides the substrate-borne vibration perception of invertebrate species from the available literature. The data available for literature is limited and is an area where further research is ongoing.

Table 5: Substrate-borne vibration perception of invertebrates

Species	Common name	Acoustic perception	Method	Reference
Pagur berhnardus	Hermit crab	5 – 400 Hz at 0.02 – 0.044 m/s^2 (RMS). The study concluded that anthropogenic substrate-borne vibrations have a clear effect on the behaviour of a common marine crustacean.	Behavioural responses to particle motion	(Roberts et al., 2016)
Panopeus sp.	Mud crabs	Predatory fish sounds (or vibrations) 90 – 200 Hz, (vibrations <0.01 m/s^2)	Electrophysiological, auditory evoked potential (AEP)	(Hughes et al., 2014)
Mytilus edulis	Blue mussel	Vibration stimulus (Sinusoidal excitation - tonal signals (5 – 410 Hz). Thresholds 0.06–0.55 m/s^2 (RMS). The study concluded that seabed vibration is likely to impact the overall fitness of both individuals and mussel beds due to disruption of natural valve periodicity.	Behavioural changes (valve closure)	(Roberts et al., 2015)

3.4.5 Birds (while diving)

Compared to marine mammals, there is little information available on the impact of underwater noise on birds (while diving). There is some limited data in the literature regarding hearing injury and mortality impacts to diving birds from proximity to an explosion (Yelverton et al., 1973), indicating that levels above 190 dB re 1 µPa could cause injury or mortality.

Sørensen et al. (2020) established behaviour responses to sound play back trials of stimulus sound levels ranging from 100 to 120 dB re 1 µPa. This study identified clear behavioural response and startle responses at 120 dB re 1 µPa.

4 Assessment criteria

4.1 National legislation

4.1.1 *Environment Protection and Biodiversity Conservation Act 1999*

The *Environment Protection and Biodiversity Conservation Act 1999* (the EPBC Act) is the central piece of environmental legislation relevant to this assessment. It provides the legal framework to protect and manage nationally and internationally important biota, ecological communities, and heritage places, which are defined in the Act as 'matters of National Environmental Significance' (MNES). Under the provisions of the Act, it is an offence for any person to take an action that is likely to have a significant impact on MNES without approval.

4.2 State legislation

4.2.1 *Environment Protection Act 1993*

The general environmental duty in section 25 of the *Environment Protection Act 1993* (EP Act) states that:

> A person must not undertake an activity that pollutes, or might pollute, the environment unless the person takes all reasonable and practicable measures to prevent or minimise any resulting environmental harm.

4.2.2 *National Parks and Wildlife Act 1972*

Section 68 of the *National Parks and Wildlife Act 1972* (NPW Act) states that a person must not interfere with, harass or molest a protected animal, or undertake or continue an act or activity that is, or is likely to be, detrimental to the welfare of a protected animal unless authorised by a permit granted by the South Australian Department for Environment and Water. The marine mammal species listed as 'protected animals' under the Act are also listed under the EPBC Act.

4.2.3 *Fisheries Management Act 2007*

Section 77 of the *Fisheries Management Act 2007* (FM Act) states that a person must not engage in an operation involving or resulting in interference with aquatic animals of any waters forming part of an aquatic reserve, except as authorised by the regulations or a permit issued by the Minister.

4.2.4 *Marine Parks Act 2007*

Section 37 of the *Marine Parks Act 2007* establishes a general duty of care: a person must take all reasonable measures to prevent or minimise harm to a marine park through their actions or activities. In determining required measures, regard must be had to (among other things) the sensitivity of the affected environment and the extent to which an activity may have a cumulative effect on a marine park.

4.2.5 *Biodiversity Act 2025*

The *Biodiversity Act 2025* is not fully operational at the time of writing. commencement is expected in stages, and until then the NPW Act continues to apply. Once commenced, it will replace the wildlife provisions of the NPW Act (including the protected animal provisions referenced above) and establishes a general statutory duty to take reasonable and practicable measures to prevent or minimise harm to biodiversity, consistent in intent with the general environmental duty under s25 of the EP Act. The Act will also extend protections to previously unlisted groups including fish and invertebrates, and provides for critical habitat declarations. The obligations it imposes with respect to underwater noise and vibration overlap considerably with the EPBC Act and State legislation described above.

4.3 Policy, guidelines and standards

4.3.1 Underwater Piling and Dredging Noise Guidelines

The *Underwater Piling and Dredging Noise Guidelines* (DIT 2023; referred to in this report as the **DIT guidelines**) have been developed by the Department for Infrastructure and Transport (the Department). They provide objective noise exposure criteria for marine mammals, fishes, and sea turtles; and guidance to addressing underwater noise for marine maintenance activities or a marine infrastructure project. The Guidelines designate safety zones for marine activities based on the potential for hearing injury in marine mammals.

4.4 Summary of assessment criteria

The noise exposure criteria adopted in this assessment based on the above literature, legislation and guidelines are summarised below.

4.4.1 Marine mammals

The noise exposure criteria for marine mammals in the DIT Guidelines are based on United States' National Marine Fisheries Service (NMFS) *Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing* (NMFS, 2018) and Southall et al. (2019). Since the release of the DIT Guidelines, NMFS have published a peer-reviewed update to their technical guidance (NMFS, 2024). Specifically, the update introduces the term 'auditory injury' (AUD INJ) to replace 'permanent threshold shift' (PTS), in recognition that recent studies have found circumstances where AUD INJ) can occur, and which may or may not result in PTS. Thus, the Updated Technical Guidance acknowledges that AUD INJ can occur, which includes but is not limited to PTS.

The updates also include changes to the auditory weighting and exposure function parameters for marine mammal hearing groups as well as thresholds for the onset of AUD INJ (formerly PTS) and temporary threshold shift (TTS) in marine mammal hearing for non-impulsive and impulsive sound sources.

Table 6 summarises the NMFS (2024) Auditory Injury (AUD INJ) and temporary threshold shift (TTS) onset threshold for marine mammals. The below criteria are based on the best available science (i.e. relevant, applicable, contemporary, peer-reviewed and published by reputable sources) and should therefore be considered to supersede the criteria listed in the DIT Guidelines (2023).

Table 6: Onset of Auditory Injury and Temporary Thresholds Shifts (NMFS 2024) – Marine mammals

Functional hearing group	Impact	Physiological noise exposure onset criteria, dB(1)	
		Impulsive	Non-impulsive
In water / underwater			
Low-frequency cetaceans (LF)	TTS	Peak 216 dB $SEL_{24\ hour}$ 168 dB(LF)	$SEL_{24\ hour}$ 177 dB(LF)
	AUD INJ	Peak 222 dB $SEL_{24\ hour}$ 183 dB(LF)	$SEL_{24\ hour}$ 197 dB(LF)
High-frequency cetaceans (HF)	TTS	Peak 224 dB $SEL_{24\ hour}$ 178 dB(HF)	$SEL_{24\ hour}$ 181 dB(HF)
	AUD INJ	Peak 230 dB $SEL_{24\ hour}$ 193 dB(HF)	$SEL_{24\ hour}$ 201 dB(HF)

Functional hearing group	Impact	Physiological noise exposure onset criteria, dB(1)	
		Impulsive	Non-impulsive
Very high-frequency cetaceans (VHF)	TTS	Peak 196 dB $SEL_{24\ hour}$ 144 dB(VHF)	$SEL_{24\ hour}$ 161 dB(VHF)
	AUD INJ	Peak 202 dB $SEL_{24\ hour}$ 159 dB(VHF)	$SEL_{24\ hour}$ 181 dB(VHF)
Phocid pinnipeds (PW)	TTS	Peak 217 dB $SEL_{24\ hour}$ 168 dB(PW)	$SEL_{24\ hour}$ 175 dB(PW)
	AUD INJ	Peak 223 dB $SEL_{24\ hour}$ 183 dB(PW)	$SEL_{24\ hour}$ 195 dB(PW)
Otariid pinnipeds (OW)	TTS	Peak 224 dB $SEL_{24\ hour}$ 170 dB(OW)	$SEL_{24\ hour}$ 179 dB(OW)
	AUD INJ	Peak 230 dB $SEL_{24\ hour}$ 185 dB(OW)	$SEL_{24\ hour}$ 199 dB(OW)
In Air(2)			
Phocid pinnipeds (PW)	TTS	Peak 156 dB $SEL_{24\ hour}$ 125 dB (PA)	$SEL_{24\ hour}$ 134 dB (PA)
	AUD INJ	Peak 162 dB $SEL_{24\ hour}$ 140 dB (PA)	$SEL_{24\ hour}$ 154 dB (PA)
Otariid pinnipeds (OW)	TTS	Peak 171 dB $SEL_{24\ hour}$ 148 dB(OA)	$SEL_{24\ hour}$ 157 dB (OA)
	AUD INJ	Peak 177 dB $SEL_{24\ hour}$ 163 dB(OCA)	$SEL_{24\ hour}$ 177 dB(OCA)

(1) Units of peak sound pressure level are dB re 1 µPa in water and dB re 20 µPa in air. Cumulative sound exposure levels ($SEL_{24\ hour}$) are dB re 1 μPa^2s in water and dB re 20 μPa^2s in air. Note that peak levels use a 'flat' or unweighted level within the generalised hearing range of marine mammals, while cumulative sound exposure levels use the relevant marine mammal hearing group weighting function (LF, HF, VHF cetaceans and PW, OW, PA and OA pinnipeds).

(2) Criteria for pinnipeds in air expressed in dB relative to a reference pressure of 20 µPa (peak) and 20 μPa^2s ($SEL_{24\ hour}$). All other criteria (in water) use a reference level of 1 µPa or 1 μPa^2s.

4.4.2 Fishes, and marine turtles

Table 7 provides the underwater noise exposure criteria adopted by the DIT Guidelines for physiological impacts on fishes and marine turtles, based on The *Sound Exposure Guidelines for Fishes and Sea Turtles* (Popper et al., 2014). These provide a widely accepted approach to assessment of the potential noise impact upon the relevant species. After consideration of the diversity of fish and sea turtles, The *Sound Exposure Guidelines for Fishes and Sea Turtles* were developed for broad groups of animals, defined by the way they detect sound.

Table 7: Underwater noise exposure criteria for fishes and marine turtles

Functional Hearing Group	Source character	Organ damage / increased risk of fatality	PTS	TTS	Behavioural Response
Fish (no swim bladder) For example: • Great White Shark • Mackeral Shark	Continuous	**N**: Low **I**: Low **F**: Low	**N**: Low **I**: Low **F**: Low	**N**: Moderate **I**: Low **F**: Low	**N**: Moderate **I**: Moderate **F**: Low
	Impulsive	Peak 213 dB $SEL_{24\ hour}$ 219 dB	Peak 213 dB $SEL_{24\ hour}$ 216 dB	$SEL_{24\ hour}$ 186 dB	**N**: High **I**: Moderate **F**: Low
Fish (with swim bladder) For example: • Pipefish • Seahorses • Seadragons	Continuous	**N**: Low **I**: Low **F**: Low	SPL 170 dB for 48 h	SPL 158 dB for 12 h	**N**: High **I**: Moderate **F**: Low
	Impulsive	Peak >207 dB $SEL_{24\ hour}$ 207 dB	Peak >207 dB $SEL_{24\ hour}$ 203 dB	$SEL_{24\ hour}$ 186 dB	**N**: High **I**: High **F**: Moderate
Marine Turtles For example: • Loggerhead Turtle • Green Sea Turtle • Leatherback Turtle • Pacific Ridley Turtle	Continuous	**N**: Low **I**: Low **F**: Low	**N**: Low **I**: Low **F**: Low	**N**: Moderate **I**: Low **F**: Low	**N**: High **I**: Moderate **F**: Low
	Impulsive	Peak 207 dB $SEL_{24\ hour}$ 210 dB	**N**: High **I**: Low **F**: Low	**N**: High **I**: Low **F**: Low	**N**: High **I**: Moderate **F**: Low

(1) Popper et al. (2014) note that where insufficient data exist to make a recommendation for Guidelines, a subjective approach is adopted in which the relative risk of an effect is placed in order of rank at three distances from the source:

- Near (N) = tens of metres from the source
- Intermediate (I) = hundreds of metres from the source
- Far (F) = thousands of metres from the source.

4.4.3 Invertebrates

Underwater noise

Indicative underwater noise exposure criteria for cuttlefish, squid, crabs and prawns are provided below in Table 8 based on Samson et al. (2014), Solé et al. (2022, 2023), (Fewtrell & McCauley, 2012), and (Filiciotto et al., 2016). All criteria are expressed as dB_{rms} relative to a reference level of 1 µPa. Due to the relative scarcity of objective noise exposure thresholds in the literature, no distinction is made here between impulsive and continuous noise sources.

Table 8: Indicative underwater noise exposure criteria for invertebrates

Functional hearing group	Physiological response / damage to organs or statocysts	Behavioural response
Cuttlefish	163 dB_{rms}	140 dB_{rms}
Squid	-	168 dB_{rms}
Crabs	171 dB_{rms}	171 dB_{rms}
Prawns	140 dB_{rms}	140 dB_{rms}

Underwater vibration

Table 9 provides the adopted vibration criteria for invertebrates (crabs and mussels) potentially affected by substrate-borne vibration.

Table 9: Substrate-borne vibration criteria for invertebrates

Functional Hearing Group	Source character	Physiological damage	Behavioural Response
Invertebrates, for example Crabs Mussels	Vibration	-	0.02–0.55 m/s^2 (5 to 400 Hz) 0.01 m/s^2 (90 to 200 Hz)

4.4.4 Birds (while diving)

Indicative criteria for birds (while diving) are based on Yelverton et al. (1973) and Sørensen et al. (2020) and are presented in Table 10.

Table 10: Underwater noise exposure criteria for birds (while diving)

Functional Hearing Group	Source character	Organ damage / increased risk of fatality	PTS	TTS	Behavioural Response
Birds (while diving)	Continuous	-	-	-	120 dB_{rms}
	Impulsive	190 dB_{rms}	190 dB_{rms}	190 dB_{rms}	120 dB_{rms}

5 Existing underwater noise environment

The ocean is filled with sound that is generated by a variety of natural sources, such as rain, breaking waves, marine life, and anthropological sources such as shipping, industrial sources, marine construction, geophysical-surveys and sonar activity. The study area is considered a shallow water coastal environment. In underwater acoustics, shallow water commonly refers to coastal waters extending from the shoreline out to the edge of the continental shelf to a depth of about 200 m. In the study area, the water depth is typically less than 30 m. Ambient noise levels in shallow water vary widely in frequency and level depending on time and location (Wenz, 1962; Cato, 2008).

In comparison to deep water, a wider range of ambient noise levels occurs in shallow water under corresponding wind and wave conditions (Richardson et al. 2013). Above approximately 500 Hz, ambient noise levels in coastal areas are often 5-10 dB higher than in deep water for the same wind speeds. In the absence of shipping and biological noise, however, low-frequency (<300 Hz) ambient noise levels can be lower in shallow water than in deep water. In the absence of anthropogenic noise, ambient noise levels in shallow waters are generally related to wind speed. For wind speeds above 2.5 m/s, the ambient noise level in the frequencies range between 50 Hz and 20 kHz is better predicted by wind speed than by wave height (Richardson et al. 2013).

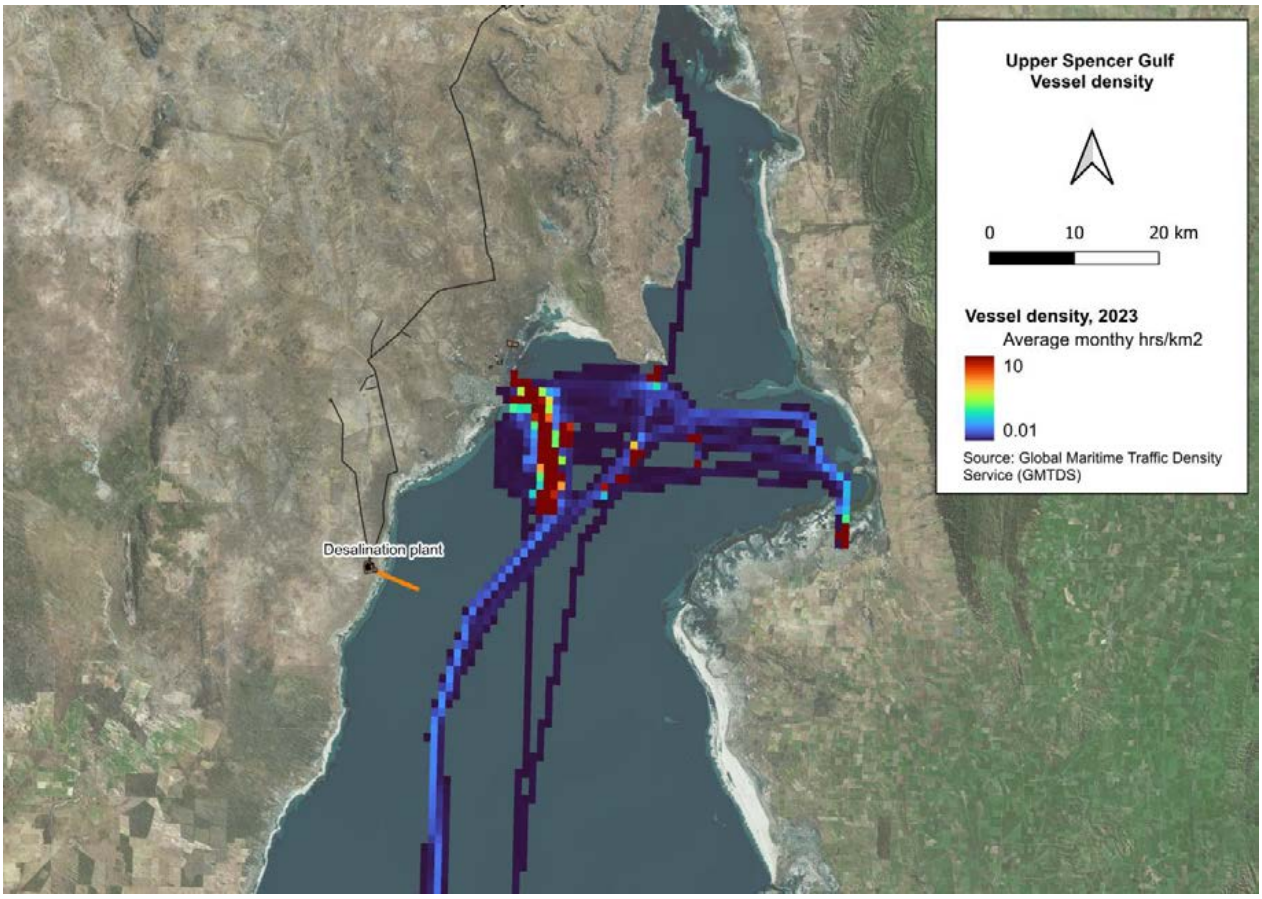


Figure 7: Upper Spencer Gulf vessel density (source: Global Maritime Traffic Density Service)

Shipping density data published by the Global Maritime Traffic Density Service (GMTDS) and presented above in Figure 7 shows a relatively low density of marine traffic, in the order of 2 hours or less per month per km^2 cell within a 10 km radius of the desalination plant intake and outlet structures. This indicates that the existing underwater noise environment is likely to be dominated by natural sources, most of the time.

5.1 Baseline underwater noise measurements

5.1.1 Measurement methodology

Baseline underwater noise measurements were undertaken for this study using hydrophones at two locations near the Mullaquana Station Project site, from 3 July to 17 July 2025.

The measurement locations (UW1 and UW2) in relation to intake and outlet alignments are shown in Figure 8. Table 11 summarises equipment used for underwater noise measurements.

Table 11: Unattended measurement equipment details

Location ID	Equipment	Serial number	Sensitivity, dB re 1V/uPa	Nominal water column depth, m	Nominal Measurement depth, m
UW1	Ocean Instruments SoundTrap ST400	9355	-176.2	14	7
UW2	Ocean Instruments SoundTrap ST400	9419	-176.8	17	8

The hydrophones were deployed from a boat and anchored to the seabed using weights. The instruments were suspended in the water column by a combination of self-buoyancy and supplementary buoys, such that the transducer depth was approximately half the water column depth.

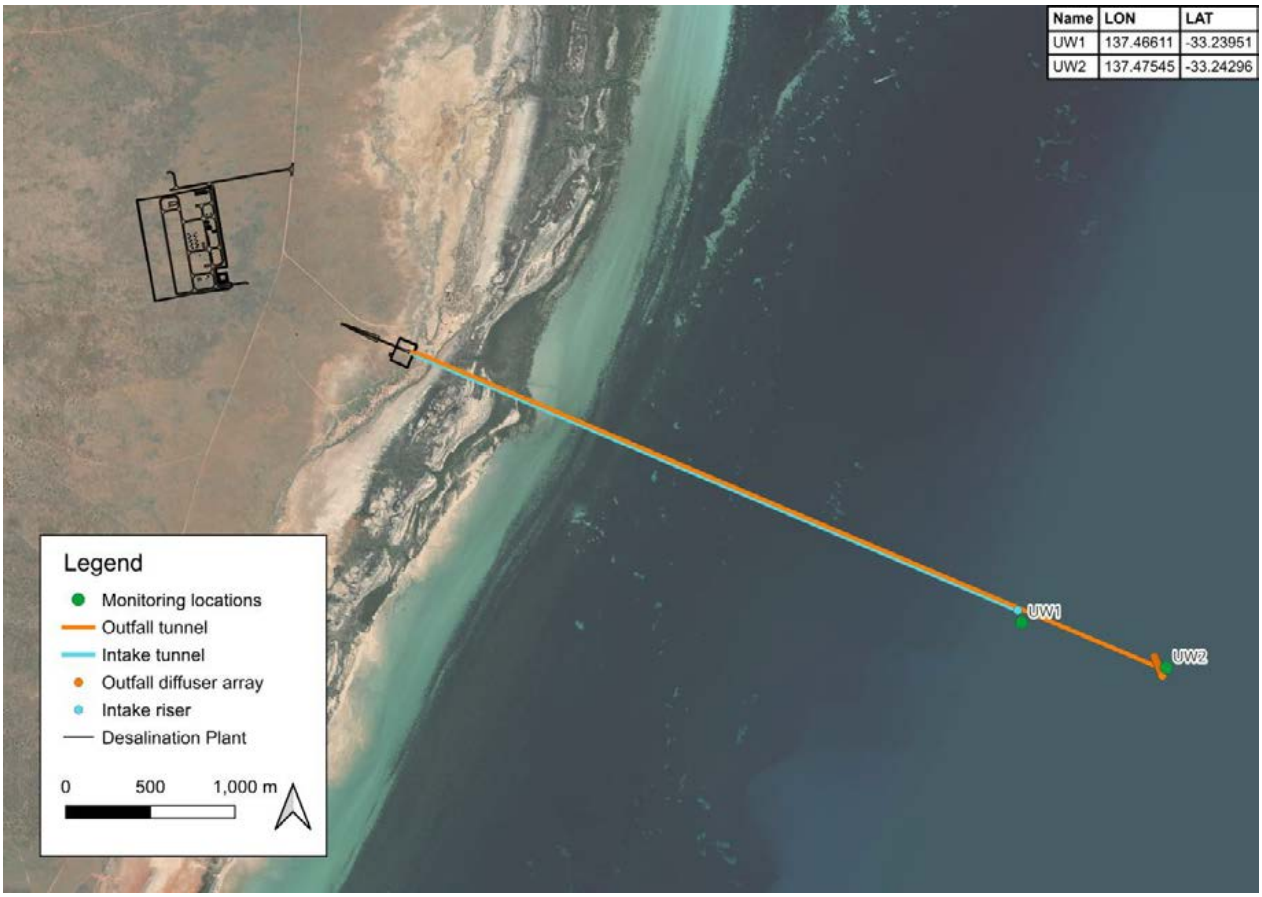


Figure 8: Baseline underwater noise measurement locations

5.1.2 Results

Underwater noise data were processed in intervals of 60 seconds. For each interval, the overall average sound pressure level (dB_{rms}) and spectra were calculated from the raw waveform data. Wind speed data was also obtained from the Bureau of Meteorology for the nearest weather station located at Whyalla.

Based on inspection of spectrograms and samples of audio data, the background level was dominated by snapping shrimp and wind/wave noise. Snapping shrimp produce a loud 'snap' from cavitation of a vapour bubble that is formed from rapid closing of their claws. The sound has sufficient energy to stun prey in close proximity. Colonies of snapping shrimp produce a sound that resembles frying fat or twigs burning (Johnson et al., 1947). Sound from snapping shrimp and waves are common features of coastal ocean underwater noise environments.

Weekly plots of overall sound pressure level (SPL dB re 1 μPa) and wind speed over time are shown in Appendix A. In summary, the results indicate:

- SPLs generally varied between approximately 97 and 120 dB_{RMS} re 1 μPa at both monitoring locations. There was not a significant variation in noise levels between locations UW1 and UW2.
- There is some correlation between wind speed and sound pressure level, which is generally expected in coastal environments where higher wind speeds drive increased wave activity.

6 Underwater noise source characterisation

Underwater noise emissions associated with both the construction and operation of the desalination plant are considered in this report. Construction activities include tunnel boring, offshore drilling (using a pile topped drill rig) for intake and outlet riser construction, placement of marine structures, and the use of support vessels for transporting equipment and workers, and dredging.

An alternative construction methodology would involve a 'float and sink' method, which involves floating the pipeline into position after tunnelling reaches a suitable depth, then lowering and securing it to the seabed using piling. The outlet pipe landward of the intake location would still be tunnelled under the seabed. Compared to the fully bored tunnel option assessed below, noise and vibration impacts from a float and sink method are expected to be similar or reduced, noting that the potential vibration impacts from tunnel boring would be confined to a smaller. Piling would still be needed to secure the pipe to the seabed.

Operational noise sources include the intake and outlet pumps; and maintenance activities such as vessel movements and dredging. For clarity this assessment is based on the design capacity of 51 GL/y. A ‘turn down’ capacity of 40 ML/day is expected to require lower intake and outlet pump duty and therefore result in less underwater noise.

6.1 Construction noise sources

6.1.1 Tunnel boring

It is expected that the tunnels will be typically excavated in clay or clayey soils, using either an earth pressure balance shield (EPBS) tunnel boring machine (TBM) or slurry TBM, with a cut diameter of approximately 4 m or less. Similar machines and methods were used on Victoria and Adelaide desalination projects.

Vibration peak particle velocities (PPV) measured inside larger TBMs are of the order 0.1 mm/s in clay soils, 1 mm/s in sand and gravel, 10 mm/s in rocky soils and up to 20 mm/s in granite (Rallu et al. 2023). Given that the TBM is envisaged to operate at minimum 2 x tunnel diameter below the seabed (i.e. minimum 8 metres), the particle velocities at distances between 10 and 30 m in front of the cutting face are expected to be about 10 times lower than that measured inside the TBM (refer Section 10 for more information). The seabed vibrations result in compressional and shear waves in the ground that transform into sound in the water column at the water-seabed interface. However, with low vibration amplitudes at the seabed and noting that 80% of the vibration energy is expected to be within 0 to 40 Hz (Rallu et al., 2023), the noise emission into the water column is expected to be largely masked by shallow water ambient noise from wind and wave activity. Underwater noise from tunnel boring has not been included quantitively in Section 7.2.2 due to the very low risk of impact. Note that *vibration* from tunnel boring is assessed in Section 10.2.

6.1.2 Intake and outlet riser construction

It is anticipated that the intake riser and outlet diffuser structures will be constructed from a jack up barge (JUB) moored at each riser location. A JUB is a non-powered floating platform with legs extending into the sea floor.

The intake riser comprises a single shaft connecting the intake tunnel to the open sea via a cross connection constructed from within the tunnel. Rock armouring is placed around the shaft collar to prevent scouring, and a precast concrete superstructure is placed onto the shaft collar and supporting piles, pre-drilled prior to riser installation. The riser cap contains the coarse screens and access/velocity hatch. The riser comprises a glass reinforced plastic (GRP) liner, grouted into position within the caisson.

The outlet riser comprises a single riser similar to the intake, connected to the tunnel in the same manner as described for the intake riser in the paragraph above. The riser comprises a GRP liner inserted into the central column and grouted into position. Rock armouring is placed around the shaft collar and the precast riser cap is placed on top

of the shaft collar, with additional support provided by pre-drilled piles. Additional piles are driven into the seabed to support the diffuser branch pipe assembly, with nozzles to release and disperse the return water into the sea.

The current Reference Design requires ground improvement piles (via ground replacement techniques) to improve the ground at the proposed riser and tunnel connection, for both intake and outlet risers.

The construction sequence for both intake and outlet riser construction is expected to be as follows:

- Establish JUB on riser location
- Complete ground replacement block and permanent piling
- Install riser and grout
- Carry out localised dredging and armouring for scour protection
- Install superstructure
- Flood the tunnels.

Based on Systra (2025), underwater noise sources associated with intake and outlet riser construction are likely to include:

- Support vessels
  - supply barge
  - spoil barge
  - tugs
  - other small support/safety vessels.
- Piling (for ground replacement block and permanent piles)
- Drilling (to cut through the concrete of the primary columns)
- Dredging.

Whilst Systra (2025) describes vibratory hammer as the preferred pile driving methodology, Resonate understands that there is a possibility for impact piling to be used. Further details of each potential noise source are given below.

Support vessels

Small to medium tugs, barges and other support vessels may be required at various stages of construction, for example barges to lay anchors for the JUB; transport of materials, workers, and spoil to and from the jack-up barge; undertaking marine environmental measurements, sampling, and the like. Systra (2025) expects 2-3 spoil barge trips during riser construction, and 6-7 trips during dredging assuming a spoil barge capacity of 500 m^3. Up to 3 tug boats may be required to position the JUB and tow/maneuver spoil barges if unable to move under their own power.

Underwater source noise levels from small to medium vessels (5 m to 25 m) of the type expected to be used on this Project are typically 150–170 dB re 1 µPa rms at 1 m (Richardson et al., 2013; Erbe et al. 2019).

Vibratory piling

Noise from vibratory piling (also known as vibro-piling or vibro-driving) is continuous in character and usually of a much lower level than impact piling. Typical source levels range from SPL 160–200 dB re 1 µPa, with most of the sound energy occurring between 100 Hz and 2 kHz. Strong tones at the driving frequency and associated harmonics may occur with the driving frequency typically ranging between 10 and 60 Hz. Sound propagation at such low frequencies is often poor in shallow water environments, such that the tones may not be noticeable at greater distances from the source (Denes et al. 2016).

Impact hammer piling

The need for impact piling during intake and outlet construction, along with specific details about the pile diameter and hammer energy, has yet to be determined.

Typical source levels at 1 m range from SEL 170–225 dB re 1 μPa^2s for a single pulse, and peak level of 190–245 dB re 1 µPa m (DIT, 2023). Most of the sound energy usually occurs at lower frequencies between 100 Hz and 1 kHz. Factors that influence the source level include the size, shape, length and material of the pile, the weight and drop height of the hammer, and the seabed material and depth.

Drilling

McCauley (1998) conducted measurements of the radiated underwater noise from an oil exploration drilling rig located in the Timor Sea. Over the study period, the drilling rig was involved in coring work, maintenance of the drill hole and active drilling. Dominant tones in the 31 and 62 Hz one-third octave bands were measured with the rig actively drilling. The highest noise levels encountered ranged between 115 dB and 117 dB (SPL) at distances of 405m and 125m, respectively. Assuming spherical spreading, the reported noise levels relate to a source level ranging between 160-170 dB 1 µPa rms at 1 m.

CMST (2016) noted that construction drilling (e.g. drilling sockets in rocky substrates / primary ground replacement concrete columns for piles) usually produces more intense noise in the water column than drilling related to exploration or production, as the drill cutting head is typically much closer to the seabed surface during construction drilling. However, measurement data is not available from cited literature to support this.

For additional context, the measured SPL_{rms} of geotechnical investigation drilling noises on the jack-up platforms are 142–145 dB re 1µPa @ 1 m (Erbe and McPherson, 2017) and 120 dB re 1µPa @ 41 m (Todd et al., 2020), which are lower than the value of 155.9 dB re 1µPa rms @ 1 m measured in a recent study (Huang et al., 2023). The difference in the noise levels was noted by Huang et al. to potentially be related to the platforms, where the anchored vessel in the Huang et al. study coupled well with the water with equipment noise on the vessel, while the jack-up platforms are well above the water line.

Dredging

Dredging works may be required for the Project. Based on Systra (2025) it is considered that a grab dredge will be used due to the relatively small quantity of material to be removed, and to allow for more localised and precise control over the extent of dredging. A focused literature review undertaken by McQueen et al. (2020) summarised that underwater dredging sounds are typically low-intensity (i.e. sound pressure levels (SPLs) <190 dB re 1 µPa at 1 m) and non-impulsive, with frequencies below 1,000 kHz. Dredging sound exposure characteristics, in terms of SPLs and frequencies, are similar to sounds emanating from commercial ship traffic (i.e. vessel propulsion). For this assessment, the source level of 175 dB re 1 µPa has been adopted.

6.2 Operational noise sources

6.2.1 Intake and outlet pumps

Operational noise (other than maintenance) is expected to be limited to water-borne noise from intake and outlet pumps propagating through the intake and outlet pipes and riser structures. The intake pump station design is based on up to 5 pumps (4 x duty/assist, and 1 x standby), while the outlet pump station design is up to 4 pumps (3 x duty/assist, and 1 x standby). Each pump is expected to be submersible pumps with a nominal power of 215 kW (GHD, 2025).

Studies undertaken for a comparable Australian facility, the Victorian Desalination Plant (Bassett Acoustics, 2008) indicate a total source noise level of approximately 150 to 160 dB re 1 µPa rms at 1 m, based on eight 625 kW pumps operating simultaneously for a total power of 5 MW. This source level was in turn based on measurements undertaken by Bassett (2008) of two 310 kW seawater intake pumps at AGL's Torrens Island Power Station, which indicated a total source level of up to 153 dB at 1m at the harmonics related to the running speed of the pumps, after adjusting for measurement distance.

A doubling in pump power typically results in an increase in source level of approximately 1 dB at 1m (Bies and Hansen, 2009). Under this assumption the Project intake (4 x 215 kW) and outlet (3 x 215 kW) pumps are expected to generate similar levels to the Torrens Island Power Station intake pumps.

Noise from the intake pumps propagates through the intake tunnel and intake riser openings (screens) out to sea; similarly noise from outlet pumps will propagate via the outlet tunnel and riser and into the sea via diffuser nozzles.

The noise transmission loss that occurs as the pump noise propagates along intake and outfall tunnels is difficult to quantify. However, this transmission loss is expected to be relatively low because sound is 'contained' in the tunnel and the hard inside surface of the tunnel provides little sound absorption. This is the case for both the reference design (concrete lined tunnel), or HDPE or similar pipe that may be used if an alternative 'float and sink' methodology is used for part of the outlet alignment. Absorption of sound by seawater is also not significant at frequencies below 1 kHz (in the order of 0.001dB/km based on Bies and Hansen, 2003). Some losses due to reflection may occur at 90 degree bends, for example where tunnels meet risers.

A conservative (worst-case) allowance of 5 dB has been adopted for overall transmission losses between intake and outlet pumps and the intake riser and outlet diffusers, respectively. This is consistent with the approach adopted for the Victorian desalination plant noting functionally similar noise transmission pathways, albeit the Northern Water desalination plant has significantly longer intake and outlet tunnels.

Based on the above, the intake head source level is expected to be up to 149 dB_{rms} re 1 µPa at 1m, while the outlet diffuser total source level is expected to be up to 148 dB_{rms} re 1 µPa at 1m, at the harmonics related to pump running speeds.

Pipe radiated noise

Note that if an alternative 'float and sink' methodology is used for part of the outlet alignment, noise transmission through the HDPE (or similar) pipe wall is also a potentially relevant consideration. The mechanism for this would be noise from outlet pumps or flow-induced noise from within the pipe causing pipe walls to vibrate, which in turn could cause underwater noise to be radiated into the surrounding water.

There is limited information in the Reference Design Report (GHD, 2025) or Constructability Report (Systra, 2025) to enable a quantitative assessment of this potential noise source. However, it is assumed that a relatively stiff, thick-walled pipe is likely necessary to provide sufficient strength to avoid buckling during the sinking process. This is expected to substantially limit vibration of the pipe and therefore significantly reduce underwater noise radiation into the surrounding water column. The risk of any adverse impacts on marine fauna from outlet pump or flow-induced underwater noise radiated from pipes is therefore considered to be low.

Notwithstanding the above it is recommended that if this methodology is adopted, consideration should be given to design of the pipe and associated components (joints, bends, supports, and the like) to minimise underwater noise and vibration.

6.2.2 Outlet discharge nozzle noise

The reference design report (GHD, 2025) indicates that outlet discharge will have a maximum nozzle velocity of 6.7 m/s based on 140 ML/day and 8 x discharge nozzles. Water jets at this velocity are not expected to generate underwater noise which is detectable above noise from outlet pumps, and is not expected to have any material impact on marine fauna.

6.2.3 Maintenance activities

Maintenance during the operational phase of the project may involve use of vessels, and potentially maintenance dredging. Noise source levels and characteristics for these sources are assumed to be equivalent or similar to those described in Section 6.1 in relation to the construction phase.

6.3 Summary of noise sources

A summary of construction and operation phase noise sources, including type and range of levels in the literature is presented in Table 12 along with notes on frequency and character. Note that range of levels for all sources is generally present in the literature, depending on the size and characteristics. The source levels adopted for this assessment, along with additional underwater acoustic metrics and frequency spectra data are presented in Section 7.2.2, Table 14.

Table 12: Summary of noise sources

Noise source	Source type	Noise source levels (from literature)	Notes on frequency and character
Construction			
Tunnel boring (TBM)	Continuous	N/A	0 to 40 Hz (Rallu et al., 2023), noise emission into the water column is expected to be minimal, largely masked by shallow water ambient noise from wind and wave activity
Vessels	Continuous	150–170 dB_{rms} re 1 µPa at 1 m	Low-mid frequency dominated. Levels are based on small-medium (5 to 25m) vessels
Vibratory piling	Continuous	160–200 dB_{rms} re 1 µPa at 1m	Generally between 100 Hz and 2 kHz with strong tones. Typically lower noise compared to impact piling
Impact hammer piling	Impulsive	SEL_{SS} 170–225 dB re 1 μPa^2s	Most energy between 100 Hz and 1 kHz. Levels dependent on pile and hammer characteristics
Drilling	Continuous	160-170 dB_{rms} 1 µPa at 1 m	Low frequency tones.
Dredging	Continuous	<190 dB_{rms} re 1 µPa at 1 m	Frequencies below 1,000 kHz, character similar to commercial shipping
Operation			
Intake pumps	Continuous	150 to 160 dB re 1 µPa rms at 1 m(1)	Frequencies at pump operating speed harmonics
Outlet pumps	Continuous	150 to 160 dB re 1 µPa rms at 1 m(1)	Frequencies at pump operating speed harmonics
Outlet discharge nozzle flow noise	Continuous	N/A	Not expected to have a material impact based on maximum velocity of 6.7 m/s
Maintenance (vessels and dredging)	Vessel and dredging levels as per construction phase		

(1) Based on Victoria Desalination Plant (5 MW), levels at source (i.e. not adjusted for tunnel transmission losses)

7 Underwater noise modelling

7.1 Basic principles

An underwater noise model can predict the sound propagation loss (PL) between a noise source and a receiver. The source level (SL) of the considered noise source and the predicted PL in a particular direction is used to predict the sound pressure level (SPL) at the receiver location as (received SPL = SL – PL).

Factors that determine the propagation loss are discussed below.

- In the ocean, the speed of sound typically ranges from 1450 m/s to 1570 m/s. It increases by approximately 1.3 m/s for every 1 unit increase in salinity (0.997 g/cm^3), 4.5 m/s for every 1°C increase in temperature, and 1.7 m/s for every 1 dbar (10,000 Pa) increase in pressure.
- *Spherical spreading* – Along the direct path between the source and the receiver, spherical spreading of the sound energy causes the noise level to drop off at 20log$_{10}$ (R) with R the distance from the source.
- *Reflection, absorption, scattering and refraction* – The transmission path is often not only the direct path between the source and receiver. Multiple transmission paths can occur due to reflections from the surface and seafloor. A rough surface or seafloor causes scattering of the source noise, and some of the noise impacting on the seafloor is absorbed. Temperature variations in the water column cause refraction of sound. These transmission loss mechanisms are generally frequency dependent and depend on the seafloor geo-acoustic properties and the surface and seafloor roughness.
- *Substrate-borne sound* – When impact pile driving is the sound source, there is the potential for substrate-borne sound caused when the hammer strikes the pile to be re-radiated back into the water. This can significantly complicate the prediction of sound to any point within the water column and potential degrade the effect of a bubble curtain or coffer dam in shallow water.
- *Total transmission loss* – The combination of the various transmission loss mechanisms gives a total transmission loss that may be smaller than due to spherical spreading alone, especially in shallow water environments. For example, this occurs when surface and seafloor reflected sound waves interfere at the receiver location such that the noise level is increased, i.e. the transmission loss is reduced.

7.2 Modelling methodology

Underwater noise modelling has been undertaken using a hybrid modelling approach whereby the parabolic equation and ray tracing methods implemented in dBSea v3.0.9 software. The parabolic equation (PE) method is well suited to low-frequency noise and ray tracing methods for high frequency noise, with a cross-over frequency of 80 Hz. The above methods are widely used in underwater acoustics and were selected for proven accuracy in shallow water environments representative of the Project site and surrounding areas. These methods are described extensively in Jensen et al., 2011.

The model takes into account the following:

- Noise source levels and frequency spectra
- Noise source locations and depths
- Bathymetry
- Sound speed profile
- Temperature
- Salinity
- Seafloor geology.

Noise sources are assumed to be omni-directional and have been modelled as point sources.

The modelling procedure used in the dBSea software to derive results is outlined below:

- Third-octave band transmission losses are calculated in a radial grid of nominally 2-degree azimuth increments and discrete distance steps around each source location, based on bathymetry and seabed geology along with sound speed, temperature, and salinity profiles.
  - In this assessment the PE algorithm has been used for low frequencies (80 Hz or less) and ray tracing solver for high frequencies as outlined above
- Using the noise source levels frequency spectra along with the calculated transmission losses, the predicted third-octave or octave band sound levels are calculated at each 'receiver' point in the radial grid
- Overall levels are calculated at each radial grid point by summing the calculated frequency band spectral levels and applying the relevant marine fauna hearing group weighting function where applicable.
- The 'weighted distances' (as presented in Section 8) can then be determined in the software by interrogating the radial grid of results. These represent the greatest distances where the predicted noise level is equal to the relevant noise exposure criteria, i.e. the maximum (worst-case) radii of impact.
- The resulting radial grid of overall weighted underwater noise levels also can be exported as noise contours in specified increments (see Appendix B)

7.2.1 Model inputs

Inputs to the model are summarised in Table 13 below. Note that minor changes in temperature and salinity will not significantly affect the modelling results and assessment outcome for this study.

Table 13: Underwater noise modelling inputs

Parameter	Input data	Notes / assumptions
Noise source locations	GIS data provided by JBS&G(1)	• Construction and operational noise sources have been modelled at intake and outlet riser locations
Bathymetry	• Project specific bathymetry data provided by JBS&G(2) • Australian Bathymetry and Topography 250m 2023 (Geoscience Australia)	• Smoothing applied to interface between two datasets where required
Temperature profile	Range of 11-26 °C with average 16°C (GHD, 2025)	• Average temperature of 16° C has been adopted for this assessment • Site specific baseline water quality data shows that temperature doesn't change materially with depth at the Project location and supports the use of a flat temperature profile.
Sound speed profile	1,500 m/s (flat)	• Expected sound speed as a function of water depth is not able to be determined from available information. However, similarly to the above, the assumption of a flat sound speed profile is considered reasonable given the shallow water environment.
Salinity	41,000 mg/L(3)	Salinity figure based on site specific baseline data
Seafloor geology	Stiff or very still clay (GHD, 2025)	

(1) GIS Rev 5, provided by JBS&G, 6 November 2025

(2) Received 22 October 2025

(3) Provided by JBS&G, Feb 2026

7.2.2 Underwater noise source levels

Based on the referenced literature (Section 6), the derived underwater noise source levels for modelling purposes are presented below in Table 14.

Table 14: Underwater noise source levels

Noise source	Source level at 1m, dB re 1 µPa				Octave-band spectra, dB re 1 µPa					
	RMS	$SEL_{24\ hour}$	SEL_{ss}	Peak	31.5	63	125	250	500	1000
Construction										
Drilling (Riser)	165	207(1)	-	-	146	157	163	152	157	145
Vibratory piling	175	217(1)	-	-	164	169	169	167	166	166
Impact Piling	213	237(2)	205	229	185	192	211	213	209	213
Vessels	160	202(1)	-	-	159	153	132	133	127	122
Dredging	175	217(1)	-	-	166	172	166	165	164	158
Operation										
Intake pumps	149	198(3)	-	-	148	142	121	122	116	111
Outfall pumps	148	197(3)	-	-	147	141	120	121	115	110
Maintenance	Source levels for vessels and dredging as presented above									

(1) Based on four hours of operation/sound exposure in a 24 hour period. Note that although some activities may operate for a longer period, sound levels from most construction sources fluctuate and are frequently intermittent. The assumption of four continuous hours at nominally high sound output is therefore expected to equal or exceed the sound exposure accumulated over a longer period of typical operation. Additionally, the SEL criteria assume a static receiver position, while many of the assessed species are mobile and transiting, and unlikely to remain exposed for significant periods; this is a large, well-recognised conservatism acknowledged in the NMFS (2024) guidance.

(2) Based on 1,500 pile strikes in a 24 hour period

(3) Based on 24/7 operation

The adopted noise source levels and operational assumptions are indicative based on reasonable nominal 'worst-case' operation and construction scenarios, derived from currently available reference design (GHD, 2025) and constructability (Systra, 2025) information. Source levels are based on the literature reviewed in Section 6, and Resonate's database of underwater noise sources.

In practice, actual equipment and operation may vary, resulting in potentially different source levels. Construction noise source levels and assumptions will be assessed further during the preparation of a Construction Noise and Vibration Management Plan (CNVMP), forming part of the Project Construction Environmental Management Plan (CEMP). Expected underwater noise levels from intake and outlet pumps may also be assessed further during detailed design.

If necessary, additional underwater noise modelling and monitoring may be undertaken during detailed design and construction stages to confirm noise levels at distance and validate the recommended underwater noise mitigation and management controls.

7.3 Results

Modelling results for construction and operation phases are presented as noise contour figures in Appendix B.

The noise contours show unweighted underwater sound levels expressed in dB_{rms} and $SEL_{24\ hour}$ metrics. For brevity, the weighted noise contours for each marine fauna hearing group are not shown. However, the maximum distances within which predicted noise levels are above the relevant weighted effect thresholds are presented in the following section (Section 8).

Note that the predicted $SEL_{24\ hour}$ levels are generally significantly higher than the RMS levels since $SEL_{24\ hour}$ represents the cumulative sum of noise exposure in a given 24-hour period. Comparison of predicted noise levels with the existing baseline noise environment (described in Section 5) should be made with reference to the RMS predicted levels shown in figures B1, B3, B5 and B8.

The following section presents an assessment of the potential impact of underwater noise associated with the Project for each of the relevant marine fauna hearing groups.

8 Marine fauna underwater noise assessment

Based on the adopted criteria in Section 4.3.1, and modelling results described in the previous section, the 'weighted distance' to the various effects thresholds (physiological effects, TTS, AUD INJ, and behavioural response) have been predicted for each noise source where applicable using the methodology and process described in Section 7.2, and are shown below in Table 15.

The 'weighted distance' represents the greatest distance from the source to where the predicted noise level (when weighted according to the relevant marine fauna hearing group) equals the respective effect threshold/criteria. In other words these distances represent the worst-case potential impact zone radii from each source, noting that actual predicted noise levels are direction dependent to some extent because of bathymetry (as shown in noise contours figures in Appendix B).

That is, there is potential for an effect to occur within the weighted distance; and conversely, beyond the weighted distance the effect is not expected to occur.

Where no numeric values are provided in the table below this is either because:

- The relevant effect threshold is not reached at any distance from noise source, even in close proximity (i.e. there is no potential for the noise source to cause that effect to occur) and is denoted by 'TNR'; or
- There is no quantitative criteria or threshold in the literature, as denoted by 'NQTL'. This may be for a variety of reasons including:
  - There is insufficient data in the literature to support definition of a discrete/quantitative threshold (for example effects of continuous noise on fish).
  - No threshold is defined because the effect is unlikely to occur, even at high noise levels. For example, this is the case for organ damage or physiological impacts on marine mammals (beyond those related to PTS, TTS and behavioural responses).
  - The effect is not relevant to the hearing group. For example, invertebrates do not have auditory systems and therefore cannot experience TTS or PTS in the same manner as other species. However, damage to statocyst or sensory hairs (categorised under organ damage / physiological response) may result in a loss of sensitivity to underwater sound and is therefore analogous to TTS and PTS.

Table 15: Predicted weighted distance to effects thresholds for each hearing group

Hearing Group	Noise source	Weighted distance to effects thresholds (m)			
		Organ damage / physiological response	PTS	TTS	Behavioural response
Low Frequency (LF) Cetaceans	Drilling	NQTL	50	130	300
	Vessels	NQTL	TNR	TNR	150
	Impact piling	NQTL	900	2,750	750(1)
	Vibratory piling	NQTL	120	200	350
	Dredging	NQTL	< 10	70	510
	Intake pumps	NQTL	TNR	TNR	40
High Frequency (HF) Cetaceans	Drilling	NQTL	TNR	40	300
	Vessels	NQTL	TNR	TNR	150
	Impact piling	NQTL	90	750	750

Hearing Group	Noise source	Weighted distance to effects thresholds (m)			
		Organ damage / physiological response	PTS	TTS	Behavioural response
	Vibratory piling	NQTL	25	140	350
	Dredging	NQTL	TNR	TNR	510
	Intake pumps	NQTL	TNR	TNR	40
Pinnipeds – Other Carnivores in Water (OCW)	Drilling	NQTL	TNR	TNR	300
	Vessels	NQTL	TNR	TNR	150
	Impact piling	NQTL	280	1,400	750[(1)]
	Vibratory piling	NQTL	25	140	350
	Dredging	NQTL	TNR	TNR	510
	Intake pumps	NQTL	TNR	TNR	40
Fish (no swim bladder, including sharks)	Drilling	NQTL	NQTL	NQTL	NQTL
	Vessels	NQTL	NQTL	NQTL	NQTL
	Impact piling	TNR	TNR	700	NQTL
	Vibratory piling	NQTL	NQTL	NQTL	NQTL
	Dredging	NQTL	NQTL	NQTL	NQTL
	Intake pumps	NQTL	NQTL	NQTL	NQTL
Fish (with swim bladder)	Drilling	NQTL	TNR	< 10	NQTL
	Vessels	NQTL	TNR	< 2	NQTL
	Impact piling	20	60	700	NQTL
	Vibratory piling	NQTL	TNR	< 10	NQTL
	Dredging	NQTL	TNR	< 10	NQTL
	Intake pumps	NQTL	TNR	TNR	NQTL
Marine Turtles	Drilling	NQTL	NQTL	NQTL	NQTL
	Vessels	NQTL	NQTL	NQTL	NQTL
	Impact piling	<10	NQTL	NQTL	NQTL
	Vibratory piling	NQTL	NQTL	NQTL	NQTL
	Dredging	NQTL	NQTL	NQTL	NQTL
	Intake pumps	NQTL	NQTL	NQTL	NQTL
Cuttlefish	Drilling	20	NQTL	NQTL	125
	Vessels	TNR	NQTL	NQTL	< 10

Hearing Group	Noise source	Weighted distance to effects thresholds (m)			
		Organ damage / physiological response	PTS	TTS	Behavioural response
	Impact piling	330	NQTL	NQTL	3,750
	Vibratory piling	< 10	NQTL	NQTL	130
	Dredging	25	NQTL	NQTL	140
	Intake pumps	TNR	NQTL	NQTL	< 10
Squid	Drilling	NQTL	NQTL	NQTL	TNR
	Vessels	NQTL	NQTL	NQTL	TNR
	Impact piling	NQTL	NQTL	NQTL	190
	Vibratory piling	NQTL	NQTL	NQTL	TNR
	Dredging	NQTL	NQTL	NQTL	TNR
	Intake pumps	NQTL	NQTL	NQTL	TNR
Crabs	Drilling	TNR	NQTL	NQTL	TNR
	Vessels	TNR	NQTL	NQTL	TNR
	Impact piling	115	NQTL	NQTL	115
	Vibratory piling	TNR	NQTL	NQTL	TNR
	Dredging	TNR	NQTL	NQTL	TNR
	Intake pumps	TNR	NQTL	NQTL	TNR
Prawns	Drilling	125	NQTL	NQTL	125
	Vessels	< 10	NQTL	NQTL	< 10
	Impact piling	3,750	NQTL	NQTL	3,750
	Vibratory piling	130	NQTL	NQTL	130
	Dredging	140	NQTL	NQTL	140
	Intake pumps	< 10	NQTL	NQTL	< 10
Birds (while diving)	Drilling	NQTL	NQTL	NQTL	300
	Vessels	NQTL	NQTL	NQTL	150
	Impact piling	<10	<10	<10	750
	Vibratory piling	NQTL	NQTL	NQTL	350
	Dredging	NQTL	NQTL	NQTL	510
	Intake pumps	NQTL	NQTL	NQTL	40

(1) The radius of behavioural impacts is less than that of TTS because the latter is based on a cumulative noise exposure threshold assuming exposure to a high number of pile strikes within a 24 hour period. In practice, any audible noise capable to causing TTS is also likely to cause an indirect behavioural response. Typically, cetaceans and other mobile

animals will respond by moving away from an unfamiliar or disturbing noise source which in turn is expected to reduce the likelihood of TTS occurring.

8.1 Underwater noise impact summary

Overall there is a greater potential for underwater noise impacts during construction phase of the Project (compared to during operation), particularly during pile driving and especially if impact hammer piling methodologies are used. It should be noted that construction phase impacts are temporary.

Any noise impacts on fauna from day-to-day operation (i.e. from intake and outlet pumps) are expected to be contained within 40m of intake and outlet risers. Marine maintenance activities during the operation phase may include vessels and dredging, and are expected to generate equivalent or similar noise levels as the same sources used during the construction phase. However it should also be noted that these activities are expected to be infrequent and short-term in nature.

Underwater noise impacts on each of the relevant marine fauna hearing groups are described in more detail below. Management and mitigation of underwater noise impacts is discussed in Section 9

8.1.1 Low frequency cetaceans

The construction activity with the highest potential impact on low-frequency cetaceans (Southern Right Whales) is impact piling (if undertaken on the project). There is potential for auditory injury to occur within approximately 900 m of impact piling, and for TTS to occur within 2,750 m. These distances are based on exposure to 1,500 pile strikes in a 24-hour period. Southern Right Whales are unlikely to often be present in the Spencer Gulf despite it being mapped (along with the entire southern Australian coastline between Perth and Encounter Bay) as biologically important for migration. Therefore, it is highly unlikely that an animal would occur within this distance for sufficient duration (i.e. 24 hrs continuously) for the cumulative noise levels to reach the relevant TTS and AUD INJ thresholds.

There is also potential for behavioural response to occur within 510 m of dredging, or within 350 m or less from other sources. Note that a typical behavioural response for low frequency cetaceans is to avoid unfamiliar and/or uncomfortable noise sources, which may impact migration patterns however turn this reduces the likelihood of auditory injury or TTS.

8.1.2 High frequency cetaceans

There is a low risk of impact to high frequency cetaceans (including dolphins) because noise from the proposed construction and operation sources occurs predominantly in the frequency region where the high frequency cetaceans hearing is the least sensitive. Impact piling has the potential to result in TTS and/or behavioural response within 750 m respectively, and noise from the Project is not expected to reach levels where auditory injury could occur to high frequency cetaceans.

Vibratory piling, dredging, drilling and vessel activities also have the potential for behavioural effects in high frequency cetaceans, within 510 m or less.

Similarly to low frequency cetaceans, a common behavioural response to high levels of noise is avoidance, reducing the likelihood that an animal would remain in the area for sufficient duration for TTS to occur.

8.1.3 Otariid pinnipeds

Similarly to high frequency cetaceans, there is a low risk of underwater noise impacts on otariid pinnipeds, including Australian Sea Lions, due to reduced hearing sensitivity at low frequencies. There is potential for auditory injury to occur within approximately 280 m from impact piling, and for TTS to occur within 1,400 m.

Vibratory piling, dredging, drilling and vessel activities also have the potential for behavioural effects in high frequency cetaceans, within 510 m or less (consistent with other marine mammals).
There is a low likelihood that otariid pinnipeds would remain in high noise areas for sufficient duration for TTS to occur, during these activities.

8.1.4 Fish, sharks and marine turtles

Generally, the risk of underwater noise impacts on fish (including sharks) and marine turtles is limited to impulsive sources (impact piling).

Continuous noise sources such as drilling, vessels, and pumps are generally not expected to impact fish, sharks, and marine turtles. Although TTS has the potential to occur in fish species with a swim bladder within 10 m of drilling, vibratory piling and dredging.

Fish with swim bladders are the most susceptible to impact from impulsive sources. Auditory injury or TTS may occur within 60 m or 700 m of impact piling respectively. Distances for other species are generally less. It should be noted that auditory injury or TTS would only occur as a result of continuous exposure over a period of hours, therefore the risk is low for mobile species because they will generally move away from high noise areas.

8.1.5 Invertebrates

Cuttlefish

There is the potential risk of physiological impact (cellular damage to the statocysts and lateral line system) within 330 m of impact piling, and for behavioural impacts to occur within 3,750 m.

There is also potential for drilling, vibratory piling and dredging to have physiological and/or behavioural impacts on cuttlefish however these impacts are expected to be limited to distances of 140 m or less. Behavioural impacts due to noise from vessels and pumps are only expected to occur within 10 m.

Squid

There is potential for behavioural impacts to squid within 190 m of impact piling. The risk of impact to squid from all other sources is low.

Crabs

Similarly to squid, there is potential for behavioural impacts to crabs within 115 m of impact piling, with minimal risk of noise impact from other sources.

Prawns

The literature suggests that prawns are potentially more sensitive than other crustaceans (i.e. crabs), with predicted noise levels indicating potential for behavioural and physiological effects from impact hammer piling within 3,750 m. The potential impacts from other sources are limited to distances of 140 m or less, however. It should be noted that the adopted threshold for behavioural and physiological effects is based on a single laboratory study using playback of continuous vessel noise. It has been assumed that this threshold can be generalised to impulsive noise (e.g. from pile driving) however this assumption is subject some uncertainty.

It is also unclear whether the noted behavioural and physiological effects are significant to the extent that they could have a material impact on population and/or catch rates of prawns in the area in the vicinity of the project.

8.1.6 Birds

There is the potential risk of behavioural impact for diving birds within 750 m of piling activities, however the potential for hearing injury or other physiological impacts is limited to the area in the immediate vicinity of piling (less than 10 m).

There is also potential for a behavioural response from other noise sources, up to distances of 510 m or less.

8.2 Cumulative noise impacts

The assessment of operational noise assumes intake and outfall pumps are operating concurrently. The nearest other significant existing or proposed source of anthropogenic underwater noise is Whyalla Port (operated by Simec), which is approximately 25 km from the proposed Project intake and outlet infrastructure. At this distance there is negligible potential for cumulative noise impacts.

Cumulative noise impacts may also arise where construction activities occur concurrently. Typical construction methodology is also sequential and precludes concurrent operation of many of the activities assessed above, although there is potential for concurrent operation of some activities for example dredging and vessel movement for spoil removal.

Underwater noise from separate sources combines logarithmically, such that the loudest source dominates the received level: two equal sources combine to give an increase of 3 dB, while a source 10 dB below the dominant source increases the combined level by less than 0.5 dB. The predicted source levels for impact piling exceed those of all other project activities by significantly more than 10 dB, and therefore concurrent operation of all other activities alongside impact piling would not materially increase the worst-case predicted levels or effect envelope distances presented in Section 8 for impact piling alone.

Construction noise impacts (including the potential for cumulative impacts from concurrent activities) will be assessed further during the preparation of a Construction Noise and Vibration Management Plan (CNVMP), when additional information is expected to be available in relation to the sequencing of construction activities.

8.3 Commissioning and decommissioning noise impacts

Whilst not a specific assessment requirement directly relating to underwater noise, the assessment requirement AEQ2 in relation general noise and vibration requires consideration of noise and vibration that would be emitted during the commissioning and decommissioning (closure) stages of the project. For completeness, underwater noise impacts from these stages are addressed here.

Whilst there is limited information regarding activities that would generate noise during the commissioning phase of the Project, it is assumed that this is likely to include operation of intake and outlet pumps in various settings. The underwater noise levels and associated impacts from commissioning are therefore expected to be similar or less than those associated with operation.

Similarly, there is limited available information on which to base an assessment of decommissioning/closure noise sources and impacts. The extent to which marine structures and intake/outlet pipes would either be removed or remain in place (for example as artificial reef habitat) is also unclear at this stage. In general, some noise sources such as vessels, JUB and dredge are likely to resemble those used in the construction phase and result in similar noise impacts. Other construction phase noise and vibration sources such as TBM and vibratory or impact piling, are unlikely to be used for decommissioning. Noise sources specific to decommissioning may include underwater hydraulic saws, wire cutters and the like, assuming some deconstruction of intake and outlet structures is undertaken. Noise emissions from these sources are temporary and less than those associated with vibratory or impact pile driving.

Overall, underwater noise impacts from the decommissioning stage are expected to be similar or less than those associated with construction, depending on the exact nature of decommissioning works. It is expected that decommissioning would be undertaken in accordance with underwater noise policy and regulation in place at the time.

9 Underwater noise mitigation and management

From the impact assessment undertaken, mitigation and management measures are likely to be required for impact piling if it occurs during construction of the Project.

Management measures (i.e. safety zones) may also be required for vibratory piling and dredging depending on the equipment and methodology used.

The following sections outline the range of practical options available in line with the SA *Underwater Piling and Dredging Noise Guidelines* (DIT, 2023). Mitigation and management measures will be further developed in the CNVMP once additional detail on the construction methodology is available.

9.1 Mitigation options – impact piling

9.1.1 Soft start

Adopting a soft start procedure in which the piling impact energy is gradually increased over a 10 minute time period may alert marine fauna to the presence of the noise source and enable animals to move away to distances where injury is unlikely. The soft start procedure should also be used after long breaks of more than 30 minutes in piling activity.

9.1.2 Whale migration season

To reduce the potential for noise impact on low frequency cetaceans, scheduling of impact piling outside of the Southern Right and Humpback Whale migration season (May to October), will be considered post-approval as part of the CNVMP development.

9.1.3 Bubble curtains

A bubble curtain may be utilised to attenuate impact piling activities, where practical and cost effective. A bubble curtain is a sheet of air bubbles that are produced around the location where piling activity occurs. The bubbles in the bubble curtain create an acoustic impedance mismatch between the water and air trapped in the bubble, which results in sound attenuation across the bubble curtain (i.e. the bubble curtain acts as a reflector, the bubbles also resonate in response to sound and absorb sound energy). Reported noise reductions range from 3 to 20 dB (Reyff, 2009).

However, bubble curtains may not be as effective in shallow water (less than 4 m). This is because sound energy from piling is transmitted through both the substrate (seabed) and water. Sound propagating via the substrate can be re-radiated into water, effectively ‘bypassing’ the bubble curtain. In deeper water (greater than 4 m), bubble curtains have been shown to provide effective noise mitigation in some frequency ranges (Lucke et al., 2011). However multiple rings of bubble curtains may be required in these situations to overcome bubble dispersal over the height of a deep-water column.

Care is required in these situations to not over prescribe expected attenuation performance of bubble curtains. Generally, it is preferable to determine the likely effectiveness and practicality of bubble curtains prior to construction (i.e. when preparing the CEMP / CNVMP) when more details of the construction methodology are expected to be known, including the need for impact piling and the type/size of impact hammer required.

9.2 Management options

9.2.1 Safety Zones

In accordance with the DIT Guidelines, Safety Zones include an *Observation Zone* and *Shut-down Zone* for marine mammals, which are a defined radius around the works with intent to be monitored for the presence of megafauna (in particular) prior to and/or during noisy activities. Safety zones are not applicable to fishes, marine turtles and invertebrates, because of the difficulty in visually identifying and managing these species in comparison to marine mammals.

Safety zones aim to minimise the likelihood of temporary or permanent hearing injury to occur to marine mammals. The zones are not intended to prevent behavioural responses to audible, but non-physical injury noise events. Marine mammals in the vicinity of a noisy activity will likely show an avoidance reaction to the noise, which reduces the likelihood of marine mammals approaching the source close enough to enter the zone of hearing injury (i.e. shut-down zone). The impacts of such a temporary displacement are unlikely to be significant unless it occurs during critical behaviours, such as breeding, feeding and resting.

Observation Zone

In the observation zone, the movement of marine mammals shall be monitored by a marine mammal observer (MMO) to determine whether they are approaching or entering the shut-down zone (see below). When a marine mammal is sighted within or appears to enter the shut-down zone, piling or dredging activities must be stopped as soon as reasonably practical.

Under the DIT Guidelines, the shut-down zone is equivalent to an exclusion zone. The observation zone is sized based on a nominal 250 m distance from the outer edge of the shut-down zone.

Shutdown Zone

The DIT Guidelines recommend that shut-down zones are sized based on the potential for a Temporary Threshold Shift (TTS) in marine mammals.

The shut-down zone allows for the cumulative effect of multiple hammer strikes during impact piling and the time duration of a continuous noise source. This allows some time for the marine mammal to move away from the noise source thereby reducing the likelihood of hearing injury to occur. However, the cumulative sound exposure from other underwater noise sources (i.e. industrial sources of noise, major shipping channels), in addition to the piling activity, may also contribute to the cumulative sound exposure.

9.2.2 Preliminary safety zones

Effective management of the shutdown zone to minimise the potential for temporary hearing impacts (i.e. TTS) requires consideration of both:

- modelled TTS distances versus exposure time; and
- mobility potential of the animals as well as in response to the noise.

Avoiding impact piling during whale migration season is preferable (where reasonable and practicable) for low frequency cetaceans. However, if piling is required during whale season, the mitigation and management measures for low frequency cetaceans must be implemented.

Note that the proposed Safety Zones are preliminary and are based on conservative assumptions, that is the noise source levels and noise propagation characteristics adopted for this assessment are conservative. The preliminary shutdown zones and observation zones for impact piling are presented in Appendix C.

Safety zones will be further developed in the CNVMP when more information and detail is available in relation to noise sources (including make/model, impact hammer energy and the like). This may include safety zones for vibratory piling and dredging if required.

9.2.3 Potential effects zones

Under the DIT Guidelines, the potential effects zones are applicable to the impact assessment upon fishes and marine turtles for impact piling activities (i.e. impulsive sound) only. The potential effects zones are not applicable for vibratory or audible activities (i.e. continuous sound sources).

The potential effects zones for fishes and marine turtles are not to be considered a shut-down zone, but zones to inform the projects risk evaluation process and identification of reasonable and practicable noise mitigation measures where required. Mitigation measures could include the adoption of alternative lower noise methods, design changes (e.g. pile material type, number of piles required) and soft starts to warn fish and marine turtles.

The potential effects zones for fishes (with/without swim bladder) and invertebrates are shown in Figure 9 and Figure 10, respectively.

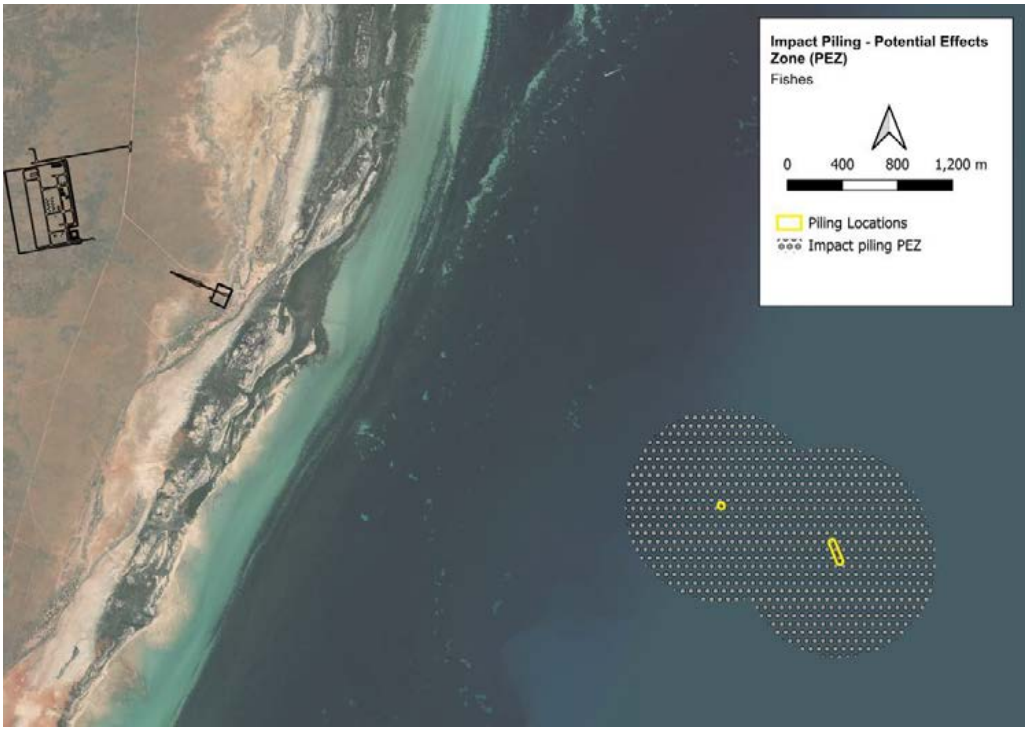


Figure 9: Potential noise effects zone for fishes from impact piling activities

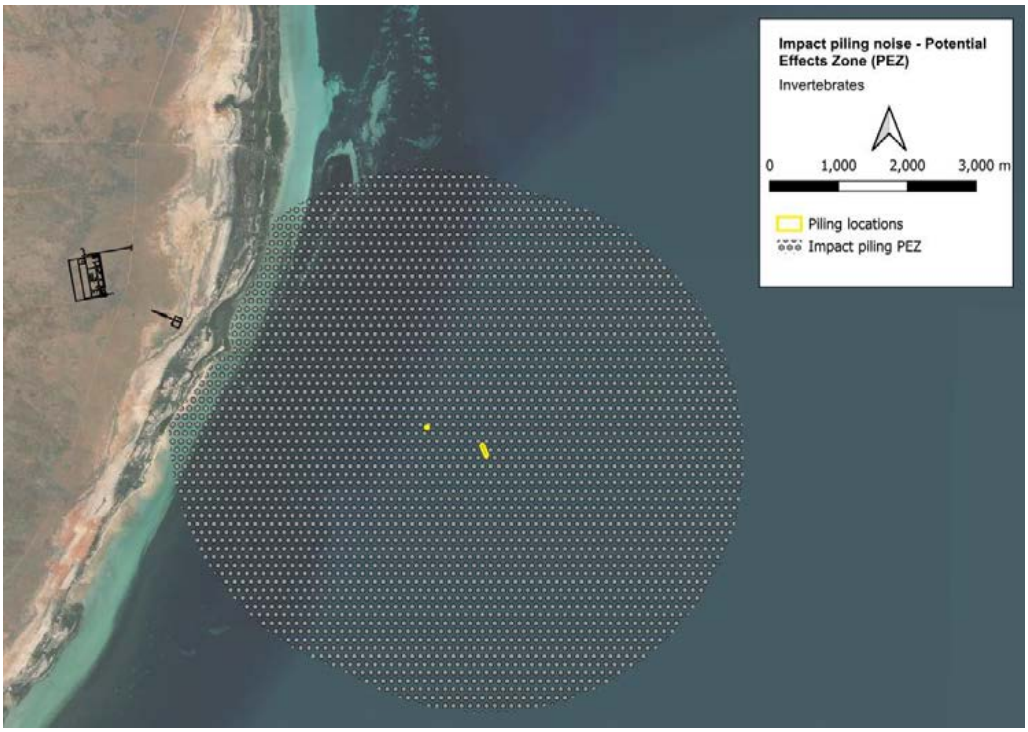


Figure 10: Potential noise effects zone for invertebrates from impact piling activities

9.2.4 Marine Fauna Observers

Under the DIT Guidelines, the requirements for different levels of marine fauna observers are as follows:

- Marine Fauna Observer (MFO) Level 1 – a person who is a suitably qualified marine fauna specialist with experience in marine mammal identification, including behaviour, as well as distance estimation.
- Marine Fauna Observer (MFO) Level 2 – a person who has sufficient experience in marine fauna identification and distance estimation.

The MFO level relates the likelihood for the project activities to be 'significant' under the below definition contained within the *Significant Impact Guidelines 1.1 - Matters of National Environmental Significance* (Commonwealth of Australia, 2013).

An action is likely to have a significant impact on a critically endangered or endangered species if there is a real chance or possibility that it will:

- lead to a long-term decrease in the size of a population
- reduce the area of occupancy of the species
- fragment an existing population into two or more populations
- adversely affect habitat critical to the survival of a species
- disrupt the breeding cycle of a population
- modify, destroy, remove, isolate or decrease the availability or quality of habitat to the extent that the species is likely to decline
- result in invasive species that are harmful to a critically endangered or endangered species becoming established in the endangered or critically endangered species' habitat
- introduce disease that may cause the species to decline, or
- interfere with the recovery of the species.

In cases where a noise impact is expected to be significant based on the above definition, a marine fauna observer (MFO) Level 1 is required to be present for the duration of related works or to train MFO Level 2 with project and site-specific details.

No significant noise impacts on critically endangered or endangered species have been identified in Section 8, therefore MFO Level 2 are expected to be sufficient for monitoring of safety zones on this Project during construction works.

10 Benthic invertebrates vibration assessment

Apart from potential impact piling and, to a lesser extent, TBM activities, construction and operation of the Project are not expected to generate significant seabed vibration. The potential effects of these sources upon benthic invertebrates (crabs and mussels) are assessed below in comparison to the adopted criteria of 0.02–0.55 m/s^2 in the frequency range of 5 to 400 Hz, and 0.01 m/s^2 in the frequency range of 90 to 200 Hz (refer to Section 3.4.4 for details).

10.1 Impact piling

Vibration generated by impact piling is explained as follows:

- The impact of the pile hammer on the pile helmet generates a stress wave that propagates through the pile.
- Dynamic forces develop along the interface between the pile and the surrounding soil, which causes vibration. The magnitude of the vibration generated varies with respect to the dynamic soil resistance and the force applied by the hammer.
- Vibration propagates in the form of different wave types depending on whether the waves are emitted along the pile shaft and/or from the pile toe. At the toe (bottom of the pile), compression waves (P-waves) and shear waves (S-waves) occur, which both extend as spherical waves in all directions. When the waves reach the surface, they are reflected and refracted. The refracted waves are spread as surface waves (R-waves), which propagate with lower attenuation than body waves (e.g. P or S-waves) along the ground surface.
- Vibrations attenuate with increasing distance from the pile, although in some soil layers, they may become amplified due to resonance effects.

10.1.1 Impact piling predictions

Impact piling vibration predictions have been undertaken utilising the Attewell et al. 1992 (quadratic regression model).

Modelling assumptions:

- Impact hammer = IHC S-150 Hydrohammer (150 kJ maximum energy)
- Hammer efficiency = 95% (conservative, typically we would expect the hammer to be less efficient than this)
- Pile material = Steel

Table 16 provides a summary of the predicted impact piling vibration levels (both mm/s PPV and m/s^2) from the pile with distance. It has been Resonate's experience that the predicted half-standard deviation results using the quadratic regression model more closely match measurement results within approximately 2 pile lengths from impact piling.

Table 16: Predicted vibration levels (mm/s PPV) in accordance with Attewell et al. 1992 – impact piling

Distance (m)	Best Fit		Half Standard Deviation		One Standard Deviation	
	PPV (mm/s)	Acceleration (m/s^2) [1]	PPV (mm/s)	Acceleration (m/s^2) [1]	PPV (mm/s)	Acceleration (m/s^2) [1]
10	11.9	1.79	19.9	3.00	33.2	5.01
20	7.3	1.10	12.1	1.82	20.3	3.06
50	3.3	0.50	5.4	0.81	9.1	1.37
100	1.6	0.24	2.6	0.39	4.4	0.66
200	0.7	0.11	1.1	0.17	2.0	0.30
400	0.28	0.04	0.47	0.07	0.78	0.12

Distance (m)	Best Fit		Half Standard Deviation		One Standard Deviation	
	PPV (mm/s)	Acceleration (m/s^2) [1]	PPV (mm/s)	Acceleration (m/s^2) [1]	PPV (mm/s)	Acceleration (m/s^2) [1]
800	0.1	0.02	0.17	0.03	0.28	0.04
950	0.08	0.01	0.13	0.02	0.22	0.03

(1) Acceleration level calculated assuming 24 Hz dominant frequency using equation A (m/s^2) = (PPV x 2π x f(Hz)) / 1000.

The tabulated results indicate that invertebrates potentially affected by substrate-borne vibrations may change their behaviour within approximately 950 m from impact piling activities, based on a criteria of 0.02 m/s^2 at 24 Hz. The potential effects zone (PEZ) is shown in Figure 11.

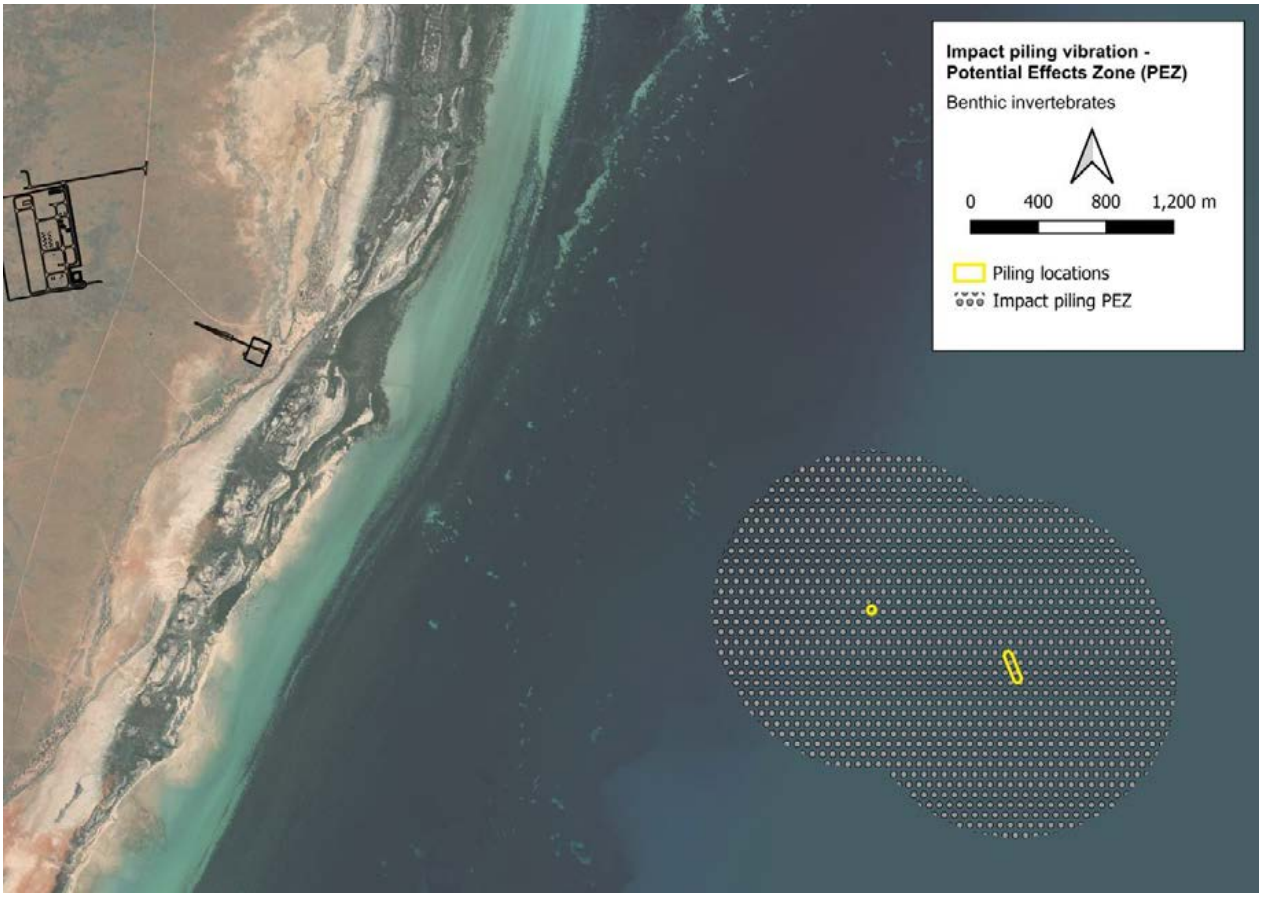


Figure 11: Potential vibration effects zone for benthic invertebrates from impact piling activities

10.2 Tunnel boring

Rallu et al. (2023) provide a comparison of four TBM campaigns in different geotechnical contexts in France, as well as data for different TBM types obtained from various literature for comparison. The comparisons are comprehensive and relevant to determining the range of vibration levels that might be generated under varying scenarios.

For each TBM campaign reviewed by Rallu et al. (2023), a curve fit in the form of a power function (Eq. (4)) was developed for all the sensors installed on the ground surface in front of the TBM. In relation to Eq. (4), the predicted PPV is the peak particle velocity (PPV) at the surface (mm/s), d (m) is the total distance from the TBM face, and α and

β are two coefficients relating to attenuation with distance (α) and the nature of the ground, the diameter of the tunnel excavated, and the operating conditions of the TBM (type of TBM, axial force and torque exerted on the face, consistency of the material present in the working chamber)

$$PPVsurface = \max\left(v_{k;}^{X}\, v_{k;}^{Y}\, ; v_{k;}^{Z}\right) = \frac{\beta}{d^{\alpha}} \qquad ...(4)$$

The obtained regressions with Eq. (4) on different sites are shown in Figure 12. The parameters (α, β) and the determination coefficient (R^2) are summarised in Table 17.

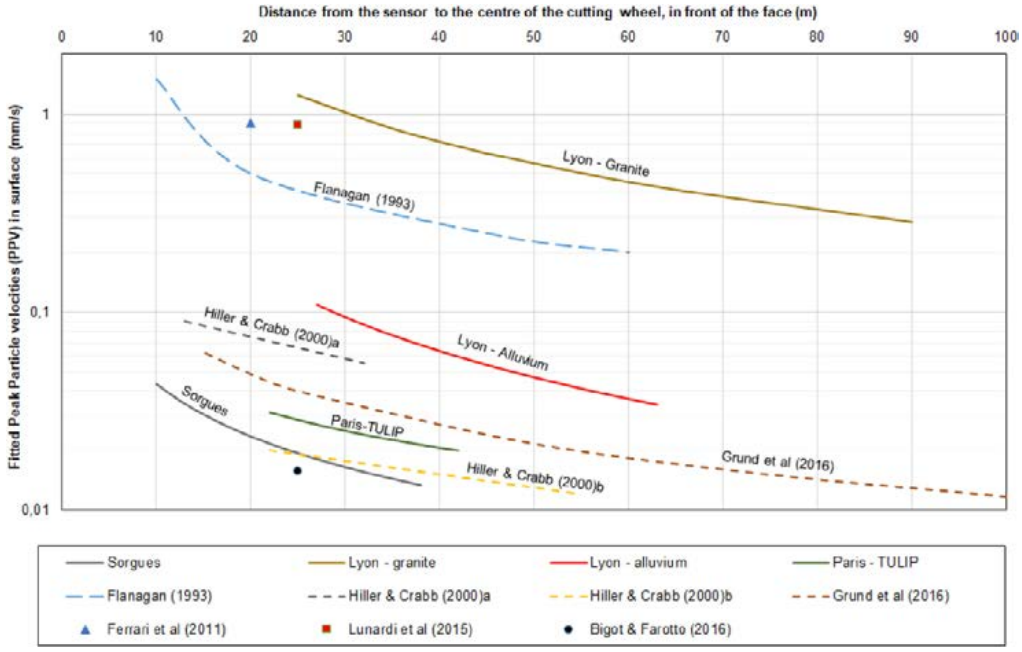


Figure 12: PPV at the ground surface with distance from the tunnel face (Rallu et al. 2023)

Table 17: Summary of the main characteristics of various projects (Rallu et al. 2023)

Source (as cited in Rallu et al. 2023)	Project	TBM	D (m)(1)	C (m) (1)	Dominant excavated ground	α	β	R^2
Flanagan (1993)	Buffalo subway (USA)	Hard rock TBM	5.6	5–10	Limestones and dolostones	1.1	17	–
Hiller and Crabb (2000)	London JLE contract 110 (UK)	EPBS	5.1	10.5	London clay	0.6	0.4	–
Hiller and Crabb (2000)	London JLE contract 107 (UK)	Slurry Shield	5.1	21.2	Sands and clays	0.6	0.1	–
Ferrari et al. (2011)	Lyon subway (France)	Slurry Shield	9.5	15	Granites	–	–	–

Source (as cited in Rallu et al. 2023)	Project	TBM	D (m)(1)	C (m) (1)	Dominant excavated ground	α	β	R^2
Lunardi et al. (2015)	Breschia subway (Italy)	EPBS	8.8	20	Boulders in an alluvial matrix	–	–	–
Bigot and Farotto (2016)	Rennes subway (France)	EPBS	9.5	20	more or less altered shales	–	–	–
Grund et al. (2016)	Karlsruhe subway (Germany)	Slurry Shield	9.3	10	Gravels and sands	0.9	0.7	–
Rallu et al. (2023)	Sorgues (France)	Micro TBM	1.3	10	Gravels and sands	0.9	0.3	0.41
Rallu et al. (2023)	Lyon subway (France)	Slurry shield	9.8	25	Granites	1.1	50	0.47
Rallu et al. (2023)	TULIP (France)	EPBS	9.9	22.5	Clayey sands	0.7	0.3	0.26

(1) Note that D is the tunnel diameter and C is the overburden thickness.

(2) EPBS = Earth Pressure Balanced Shield

In summary, the results indicate that the PPV measured on the surface is typically low, between 0.01 and 1 mm/s for a distance to the cutting wheel between 10 m and 80 m. The Project geology is expected to be generally clay in the marine section with potentially some sandstone / sand lenses, meaning that the most relevant results from Figure 12 are those in clays, sands, alluvium and gravels. These indicate PPV levels of 0.1 mm/s or less at distances of 30 m or more from the centre of the TBM cutting face, noting most of these results indicate that PPV is 0.1 mm/s or less beyond a distance of 10 m.

To directly compare to potential vibration effects criteria for invertebrates, the equivalent acceleration vibration levels for the relevant TBM results presented in Table 17, equate to 0.02 m/s^2 under the assumption of dominant 30 Hz frequency content within 30 m from the centre of the TBM cutting head.

The minimum tunnel overburden (i.e. distance between the seabed top of tunnel) is expected to be minimum two tunnel diameters, meaning the distance from the centre of a nominal 4 m diameter TBM to the seabed is expected to be a minimum of 10 m, and increases in some tunnel sections depending on bathymetry and tunnel vertical alignment.

We have conservatively shown the PEZ for vibration based on TBM located near the surface, i.e. based on horizontal distance only, ignoring additional distance due to depth below seabed. In reality, the width of the potentially impacted area of the seabed will reduce to zero where the depth to the centre of TBM cutting head is 30 m or more.

The vibration PEZ is shown in Figure 13.

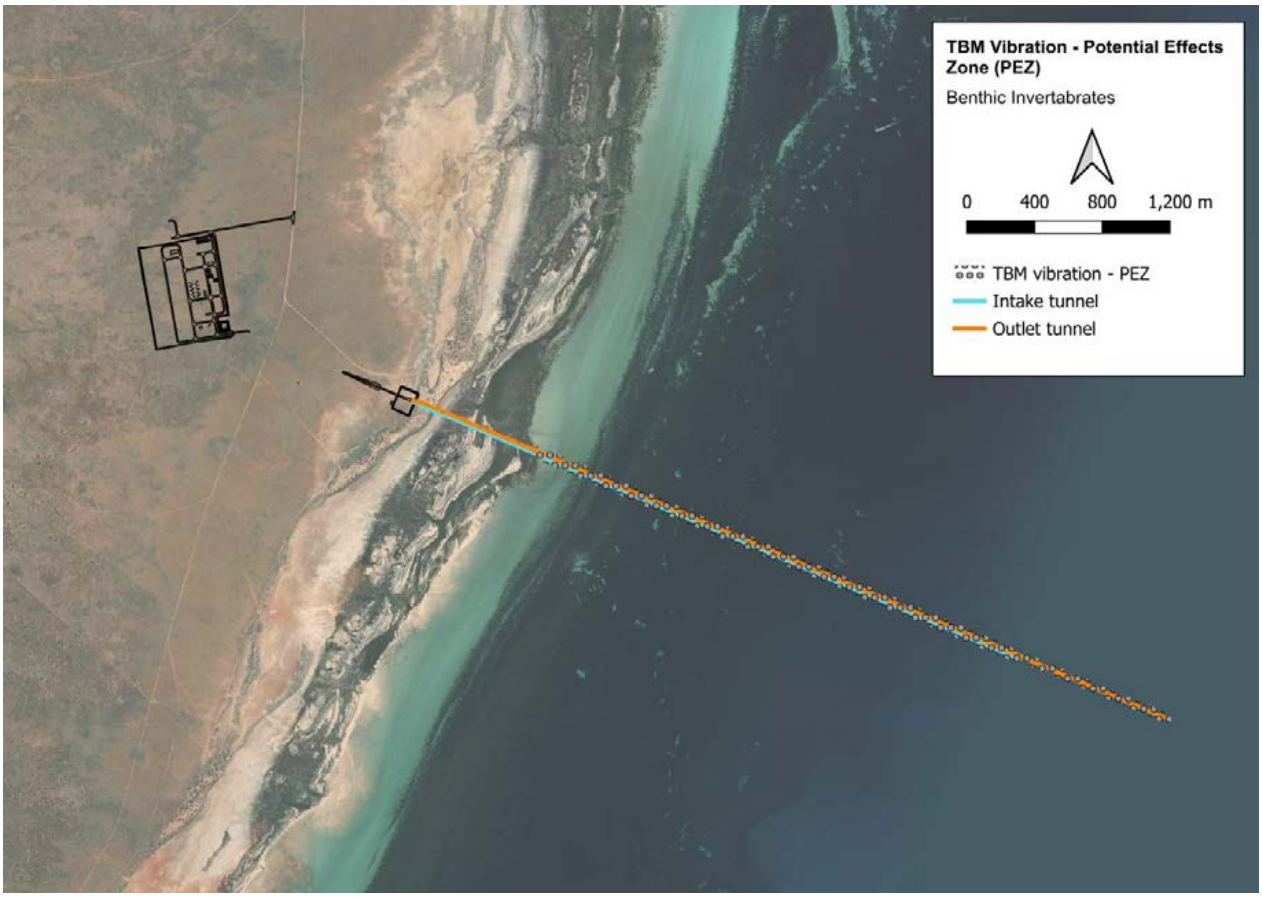


Figure 13: Potential vibration effects zone for benthic invertebrates from the tunnel boring machine (TBM)

It is noted that two TBMs may be deployed to tunnel the intake and outlet tunnels simultaneously. Cumulative vibration impacts are not expected because both TBMs would need to be operating in close proximity (within approximately 50 m of each other) and generating vibration in phase in order to have a summative effect (i.e. to amplify the vibration compared to a single TBM).

10.3 Summary of vibration impacts

This assessment has identified that there is a potential benthic invertebrate behavioural response within 950 m of impact piling, and within 30 m of the centre of the TBM cutting head due to vibration. Vibration impacts from tunnelling are therefore expected to be localised to the area of seabed immediately above the intake and outlet tunnels.

10.4 Vibration mitigation and management

As shown above, any vibration impacts from TBMs are expected to be highly localised to the area of seabed directly above the intake and outlet tunnels, and there is limited opportunity for mitigation and management of vibration from this activity.

Mitigation and management measures for piling activities may include:

- Adoption of soft start measures where reasonable and practicable, as described in Section 9.1.1.
- avoid impact piling where possible, noting that vibratory or drilled piling is expected to result in significantly lower levels of underwater vibration (and noise).

- selection of a smaller pile driving hammer and/or reduced drop height or hammer energy, to reduce the force and propagation of vibration
- scheduling of impact piling will be considered post-approval in the development of the CNVMP, with consideration to avoiding times or seasons when the relevant species may be more sensitive to vibration, for example feeding times; or periods of aggregation, mating and spawning.

11 Construction Noise and Vibration Management Plan

Before the commencement of works, it is recommended that the construction team prepare a Construction Noise and Vibration Management Plan (CNVMP), a sub plan of the Construction Environmental Management Plan (CEMP).

The CNVMP shall consider the adopted marine assessment criteria and sensitive receptors, as well as mitigation and management measures contained within this document (unless superseded by new information or government policy), namely:

- Section 4 Assessment criteria
- Section 9 Underwater noise mitigation and management
- Section 10 Benthic invertebrates vibration assessment.

12 Summary and conclusion

This report addresses potential underwater noise and vibration impacts from the Project, including mitigation strategies to eliminate or minimise impacts; and will form part of the Environmental Impact Statement (EIS) for the Project under the *Planning, Development and Infrastructure Act, 2016*.

The marine fauna considered in this assessment are listed in Section 3.3 along with a review of literature relating to the sensitivity of each species (or grouping of species) to anthropogenic underwater sound and vibration. The adopted assessment criteria, based on literature and policy including the Underwater Piling and Dredging Noise Guidelines (DIT, 2023) and more recent updates including NMFS (2024), are presented in Section 4.

The assessment includes measurements of the existing (baseline) underwater noise environment in the vicinity of the Project site. The measurement methodology and results are described in Section 5. The existing ambient environment is typical of South Australian coastal areas away from significant anthropogenic noise sources, with ambient levels dominated by sound from snapping shrimp and wind/wave noise.

Section 6 provides a description of the underwater noise sources associated with construction and operation of the Project. Construction noise sources considered in this assessment are tunnel boring, offshore drilling (using a pile-topped drill rig), impact hammer and/vibratory piling, dredging and support vessels. Operational noise sources considered in this assessment are limited to intake and outlet pumps, along with marine vessels and dredging that may be required intermittently for maintenance during the operational phase of the Project.

The underwater noise modelling methodology and inputs used in this assessment are described in Section 7. Modelling was undertaken using a combination of the parabolic equation (PE) and ray-tracing propagation methods implemented in dBSea v3.0.9 software, with a cross-over frequency of 80 Hz between PE and ray-tracing methods. These approaches are widely used in contemporary underwater acoustics practice and are consistent with the technical basis of the Guidelines. The dBSea modelling procedure results in predicted noise levels at a radial grid of points, which are then used to derive both noise contour plots and “weighted distances” representing the nominal radii of potential noise effects/impacts for each functional hearing group.

Results of underwater noise modelling are presented as noise contour plots in Appendix B and as a table of weighted distances in Table 15.

Section 8 presents an assessment of potential noise impacts on marine fauna, based on the predicted noise levels. To assist with quantifying the risk of impacts to fauna, the “weighted distance” to the various effects thresholds (physiological effects, auditory injury, TTS and behavioural response) is provided for the relevant hearing groups. The Project activity with the highest risk of adverse impacts on marine fauna is impulsive noise from impact piling (if required), with potential auditory injury to low-frequency (LF) cetaceans within approximately 900 m of the activity, and potential for TTS and/or behavioural response within approximately 2,750 m, based on 1,500 pile strikes within a 24-hour period. For cuttlefish and prawns, there is potential for physiological and/or behavioural impact within approximately 3,750 m of impact piling. The weighted distances for other species groups are lower, i.e. there is a reduced radius of potential impact for other species.

To a lesser extent, there are potential impacts to marine fauna due to noise from continuous or intermittent sources construction such as drilling, dredging support vessels, and vibratory piling. For these sources, the potential impact zones are contained within a radius of 510 m or less from the noise source, depending on species group and effect threshold.

Any underwater noise impacts on fauna from operation (i.e. from the intake and outlet pumps) are expected to be contained within the immediate vicinity of the intake and outlet risers. Section 8.2 also considers cumulative noise impacts and concludes that, given the separation distance to other significant anthropogenic underwater noise sources, there is negligible potential for cumulative underwater noise impacts.

Section 9 presents underwater noise mitigation and management measures applicable to impact piling which could be adopted during Project construction (where reasonable and practicable). These include:

- use of “soft start” procedures
- avoiding impact piling during the whale migration season
- use of bubble curtains where appropriate and practicable
- adoption of safety zones (i.e. shut-down and observation zones) for marine mammals
- adoption of potential effects zones (PEZ) for fishes, marine turtles and invertebrates to help inform the need for other mitigation and management strategies.

Section 10 presents an assessment of the potential impacts of Project-related vibration on marine fauna. The fauna considered in this assessment are limited to benthic invertebrates, since vibration of the substrate does not directly affect pelagic animals. Available scientific literature on the effects of substrate-borne vibration on benthic invertebrates is limited and the threshold levels for potential impacts beyond behavioural responses are not well defined.

Nevertheless, the assessment indicates that there is likely to be a behavioural response to construction vibration-generating sources typically within about 950 m for impact piling, and within approximately 30 m of the TBM cutting face.

Mitigation and management measures for piling activities could include vibratory or drilled piling (as alternative to impact piling) if possible; selection of a smaller pile driving hammer and/or reduced drop height or hammer energy; and scheduling of impact piling to avoid times or seasons when the relevant species may be more sensitive to vibration, for example feeding times; or periods of aggregation, mating and spawning.

Finally, Section 11 recommends the preparation of a Construction Noise and Vibration Management Plan (CNVMP), as a sub-plan of the Construction Environmental Management Plan (CEMP). The CNVMP should consider the adopted marine assessment criteria and sensitive receptors, as well as the mitigation and management measures outlined in this report (unless superseded by new information or government policy).

Overall, based on the adopted assessment criteria, the modelled underwater noise and vibration levels, and the implementation of the recommended mitigation and management measures (including a CNVMP), the risk of significant underwater noise or vibration impacts on marine fauna during construction and operation of the Project is considered low, and potential impacts can be appropriately managed.

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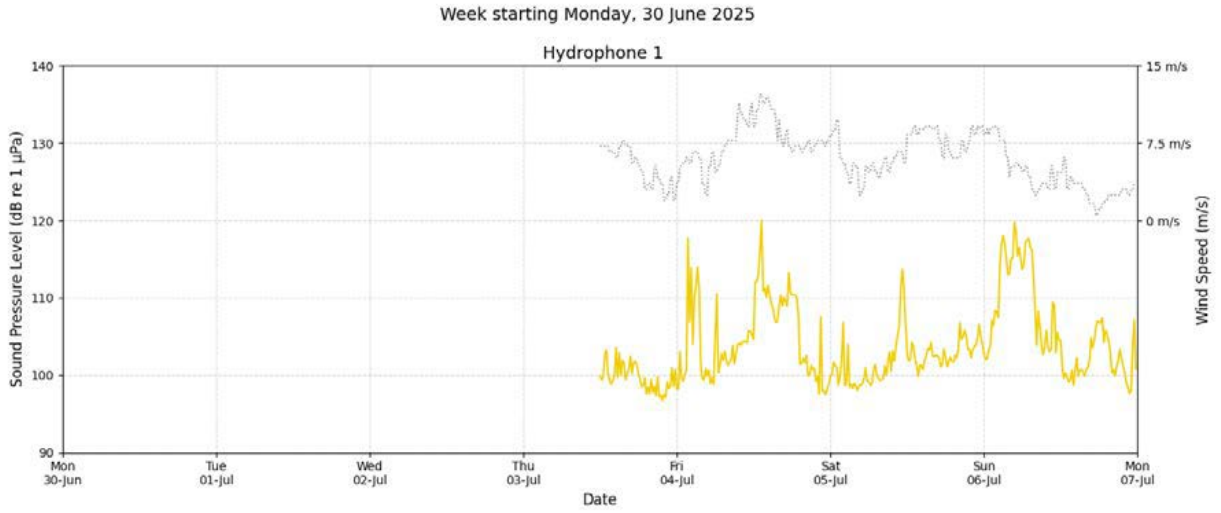
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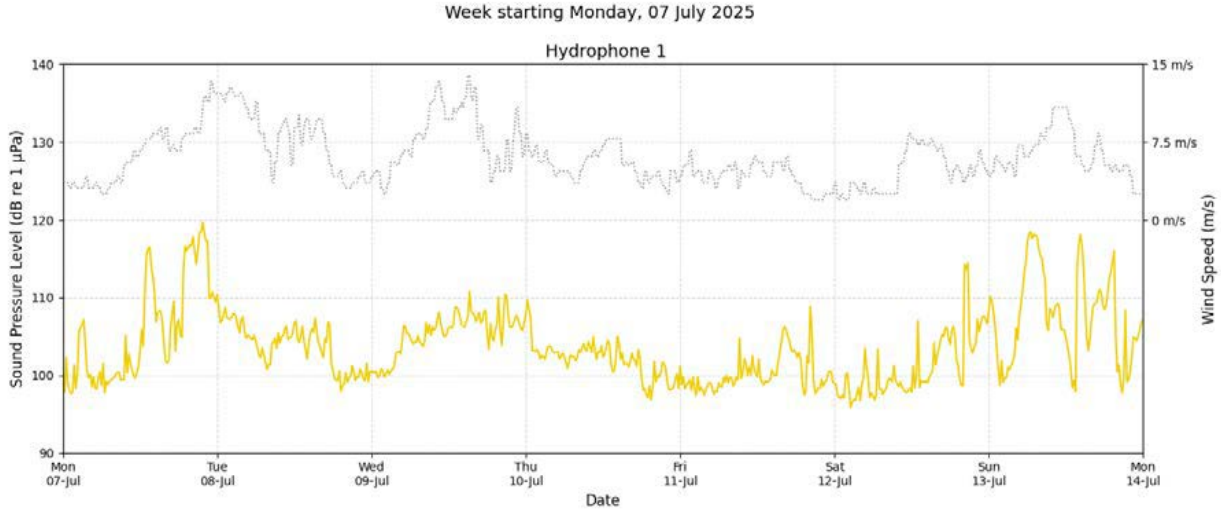
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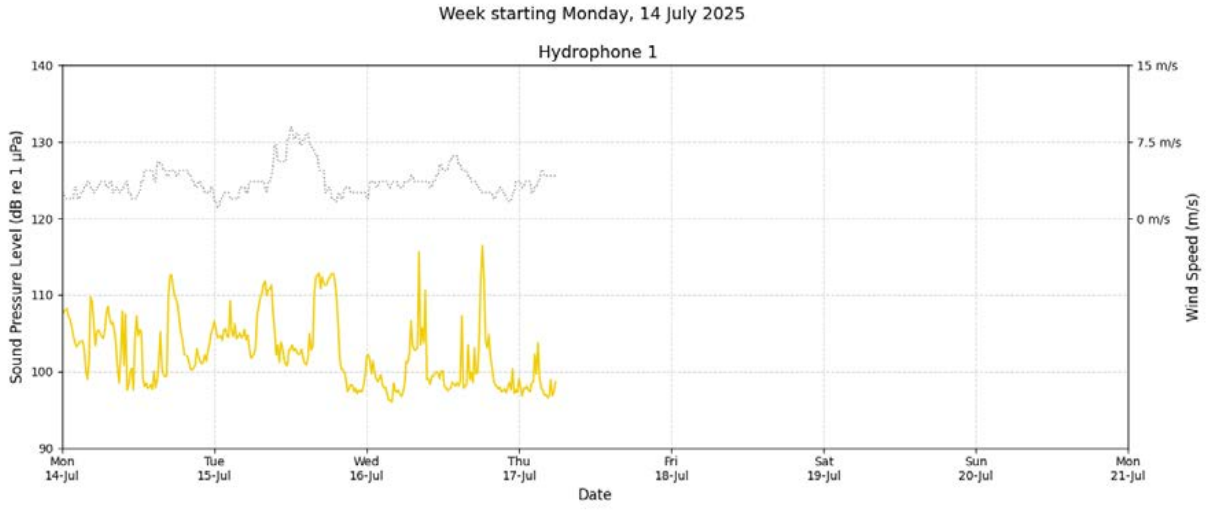
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Appendix A – Baseline underwater noise monitoring results

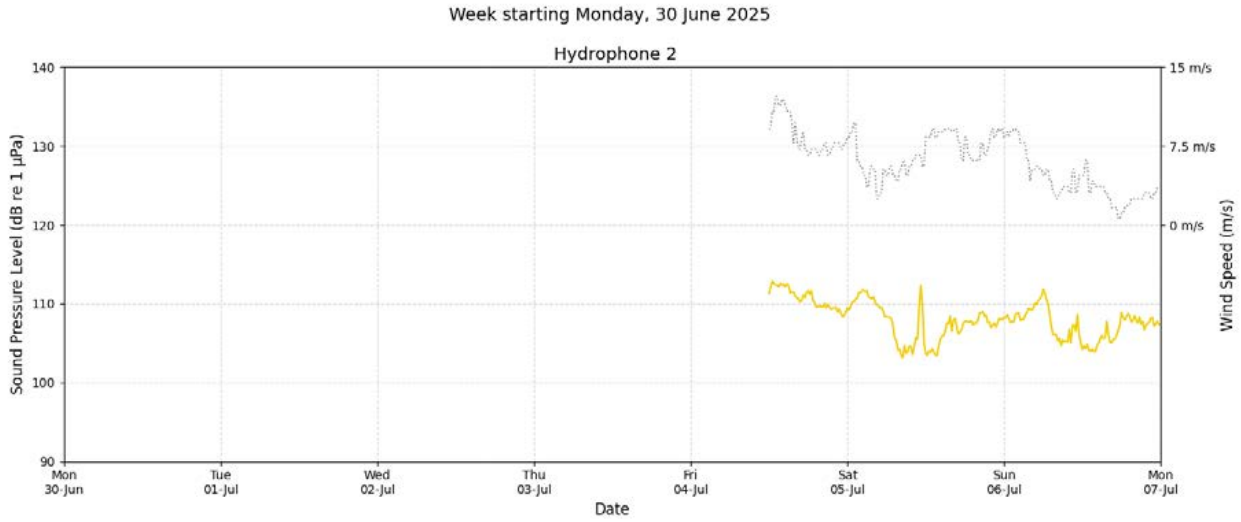
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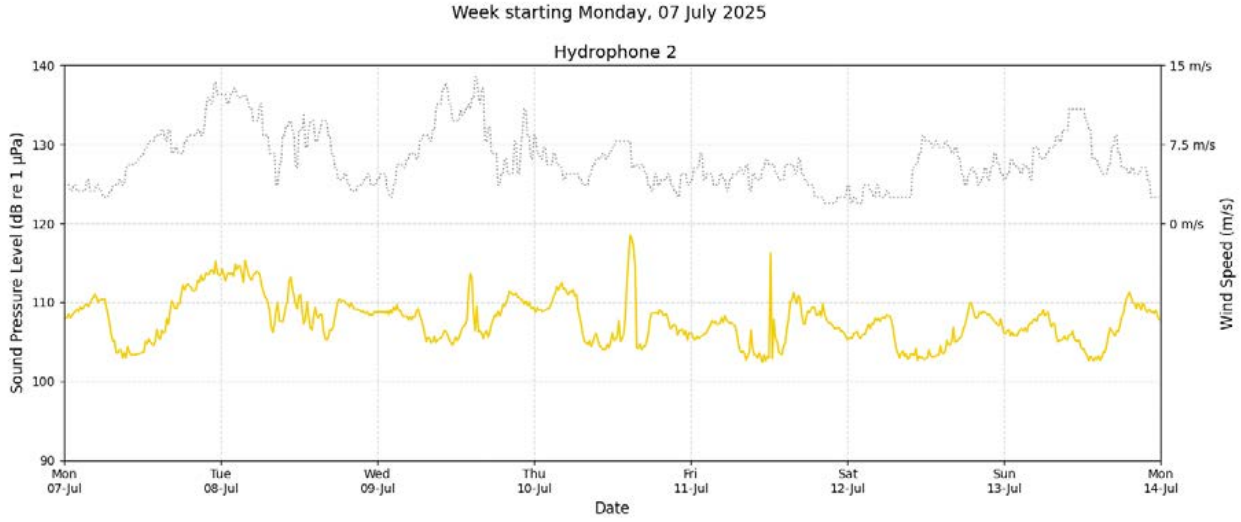


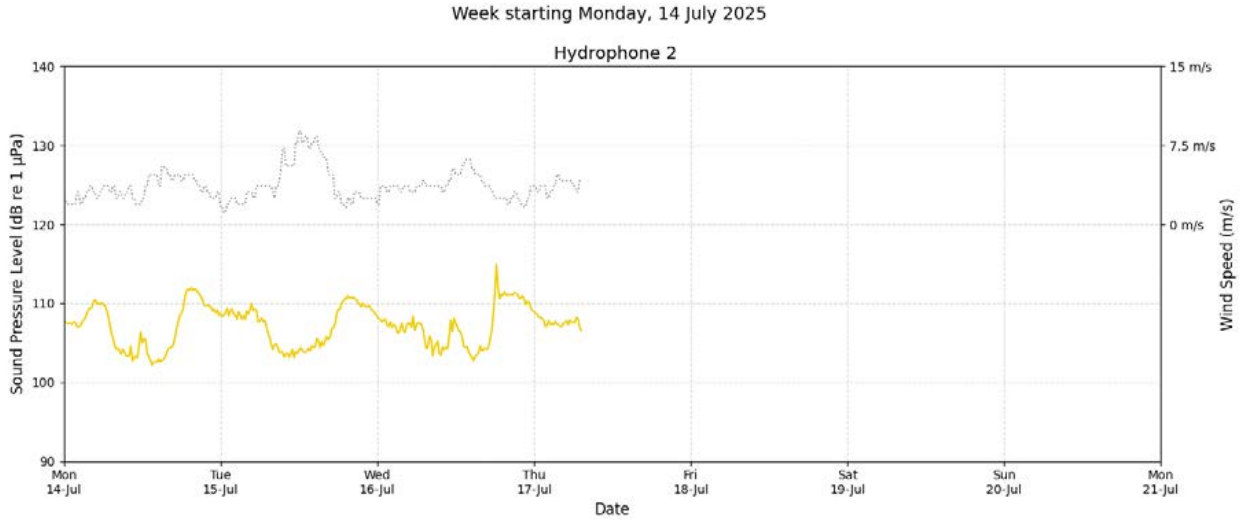




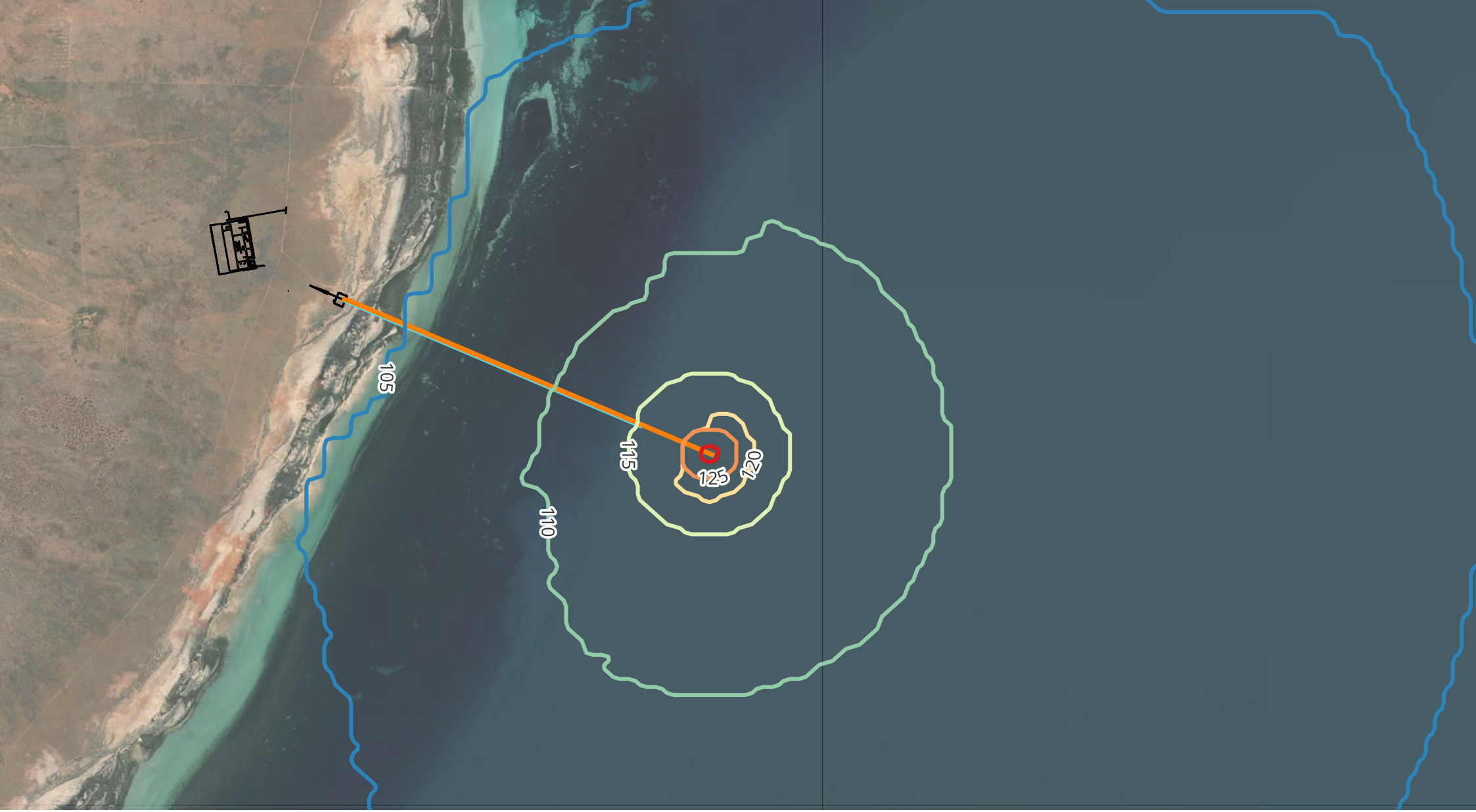
Hydrophone 2 (UW2)







Appendix B – Predicted underwater noise contours



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, dB rms re 1 µPa

- 105
- 110
- 115
- 120
- 125
- 130

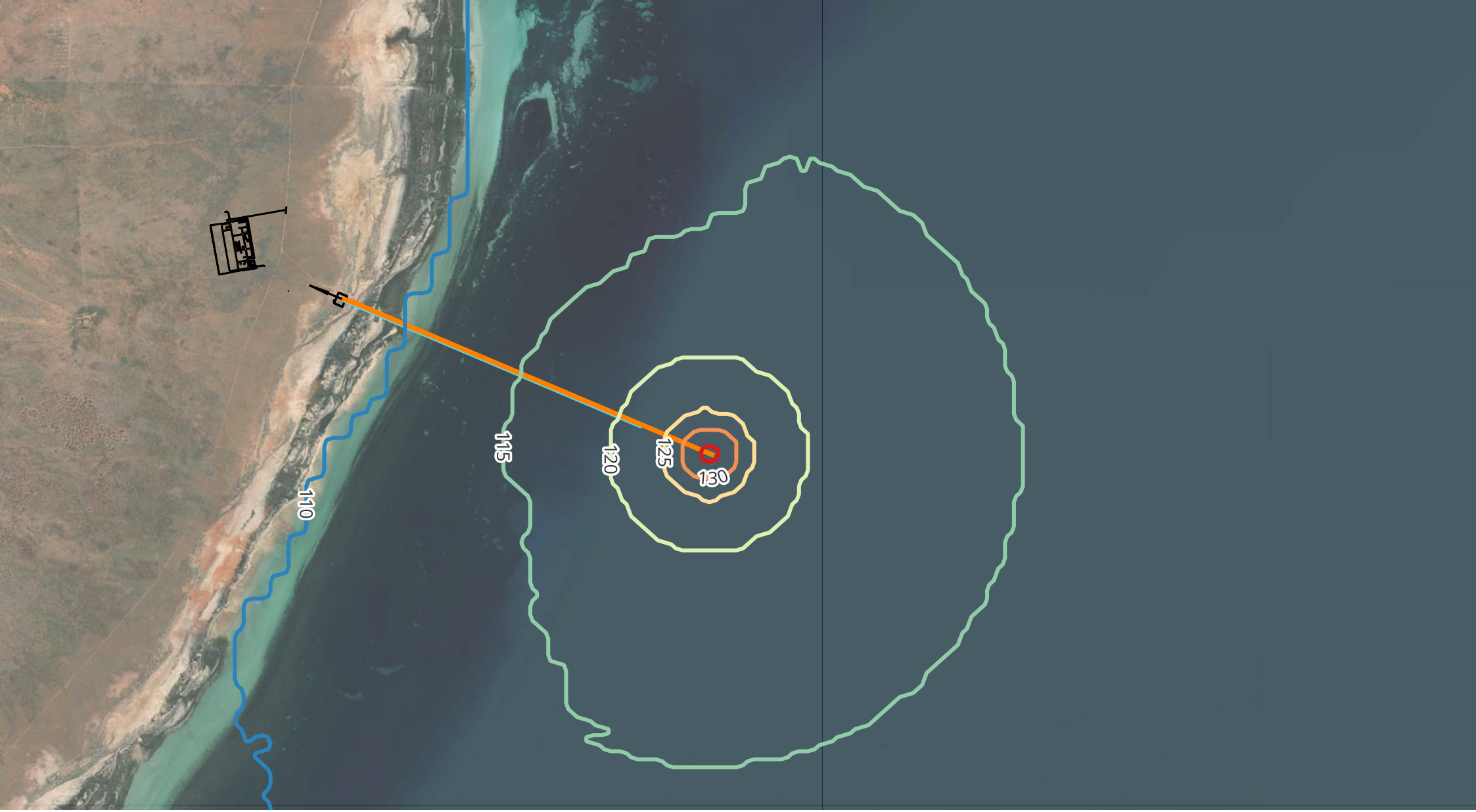
Northern Water Project

Drilling at outlet site
Predicted underwater noise levels (rms)

Drawn by: JH
Checked by: NH
Dated November 2025
Page size: A3

Figure

B1



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, SEL24 hour
dB re 1 µPa2s

- 110
- 115
- 150
- 125
- 130
- 135

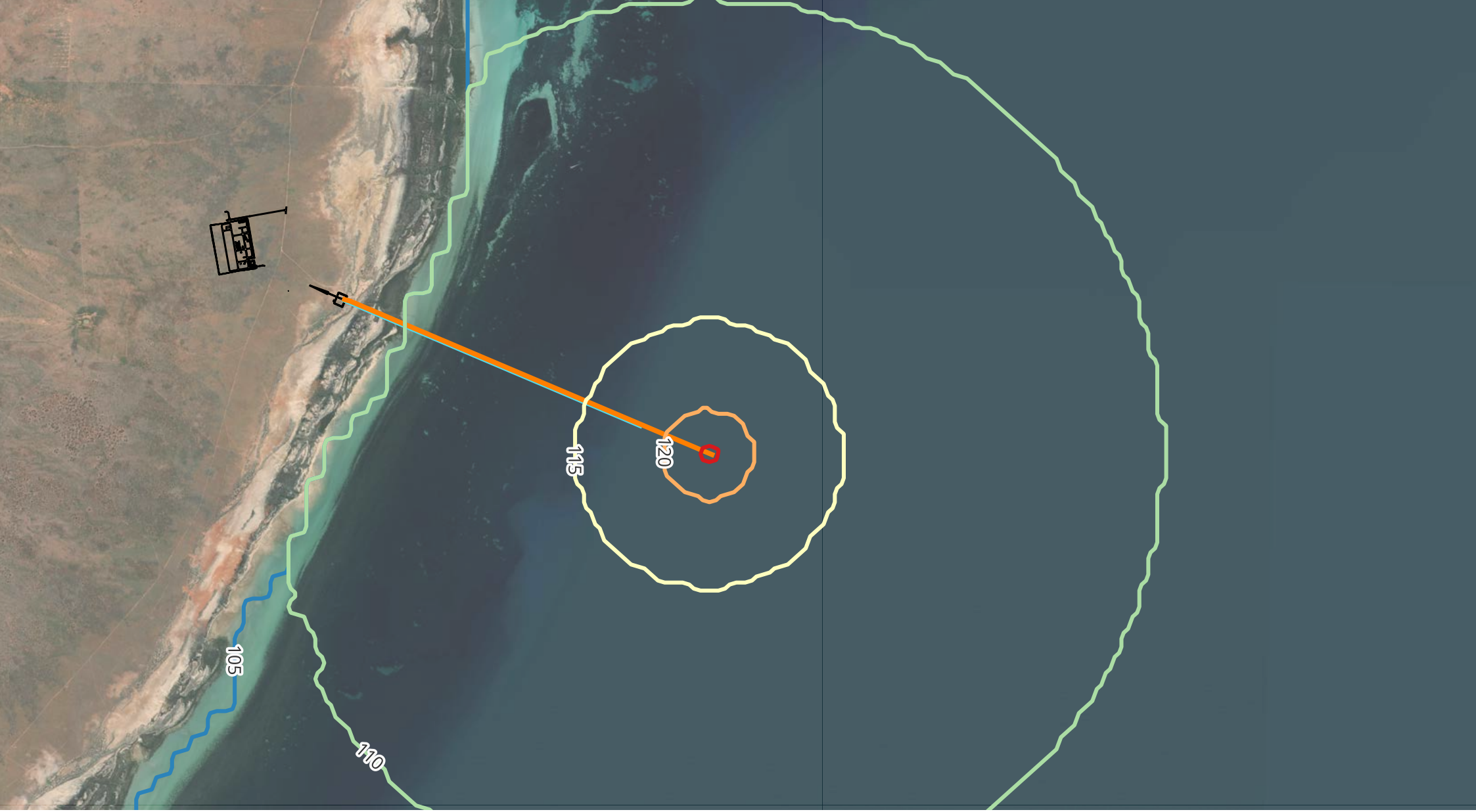
Northern Water Project

Drilling at outlet riser
Predicted underwater noise levels (SEL24 hour)

Drawn by: JH
Checked by: NH
Dated November 2025
Page size: A3

Figure

B2



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, dB rms re 1 µPa

- 105
- 110
- 115
- 120
- 125

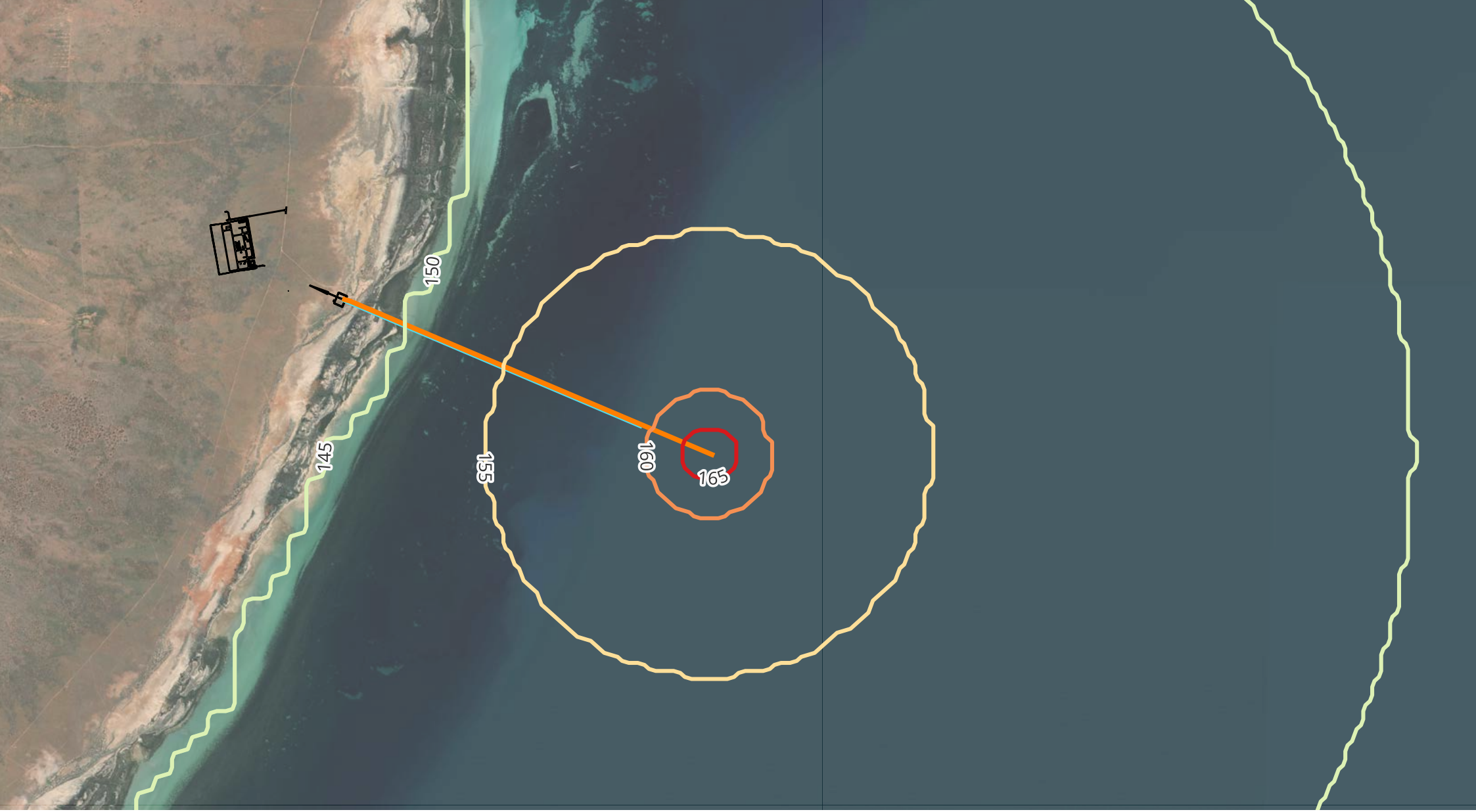
Northern Water Project

Vessel at outlet site
Predicted underwater noise levels (rms)

Drawn by: JH
Checked by: NH
Dated November 2025
Page size: A3

Figure

B3



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, SEL24 hour dB re 1 µPa2s

- 140
- 145
- 150
- 155
- 160
- 165

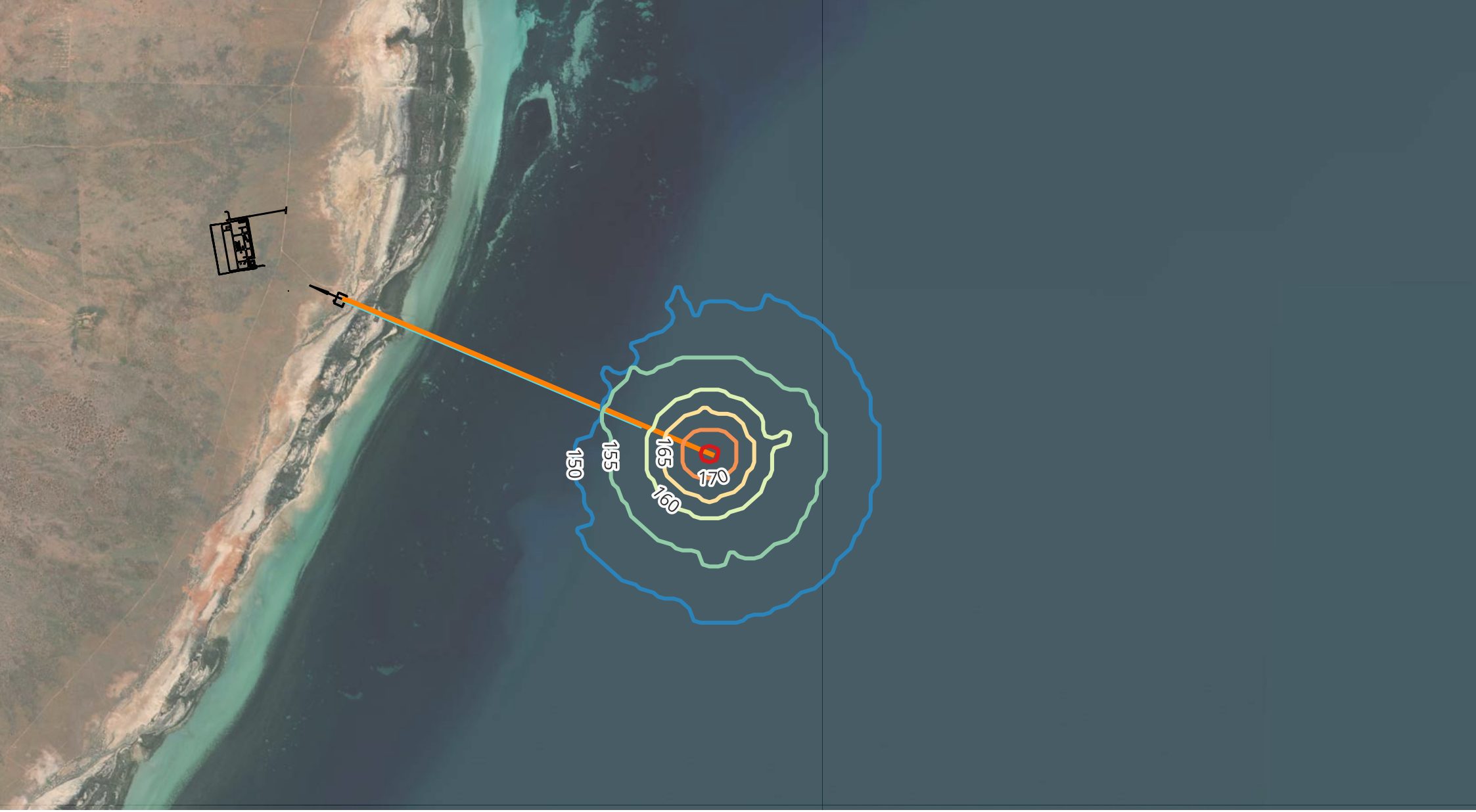
Northern Water Project

Vessel at outlet riser
Predicted underwater noise levels (SEL24 hour)

Drawn by: JH
Checked by: NH
Dated November 2025
Page size: A3

Figure

B4



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, dB rms re 1 µPa

- 150
- 155
- 160
- 165
- 170
- 175

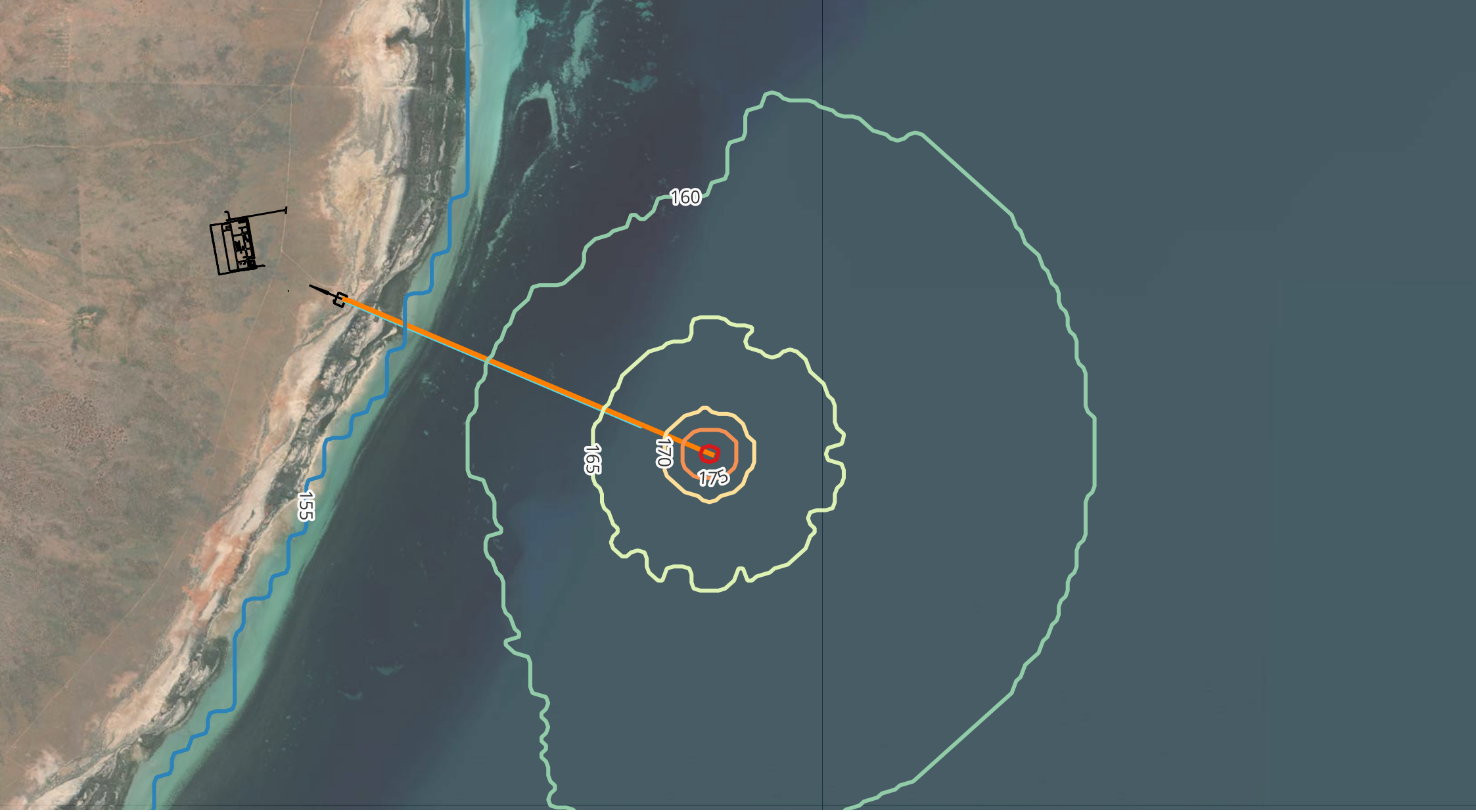
Northern Water Project

Impact piling at outlet site
Predicted underwater noise levels (rms)

Drawn by: JH
Checked by: NH
Dated November 2025
Page size: A3

Figure

B5



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, SEL24 hour
dB re 1 µPa2s

- 155
- 160
- 165
- 170
- 175
- 180

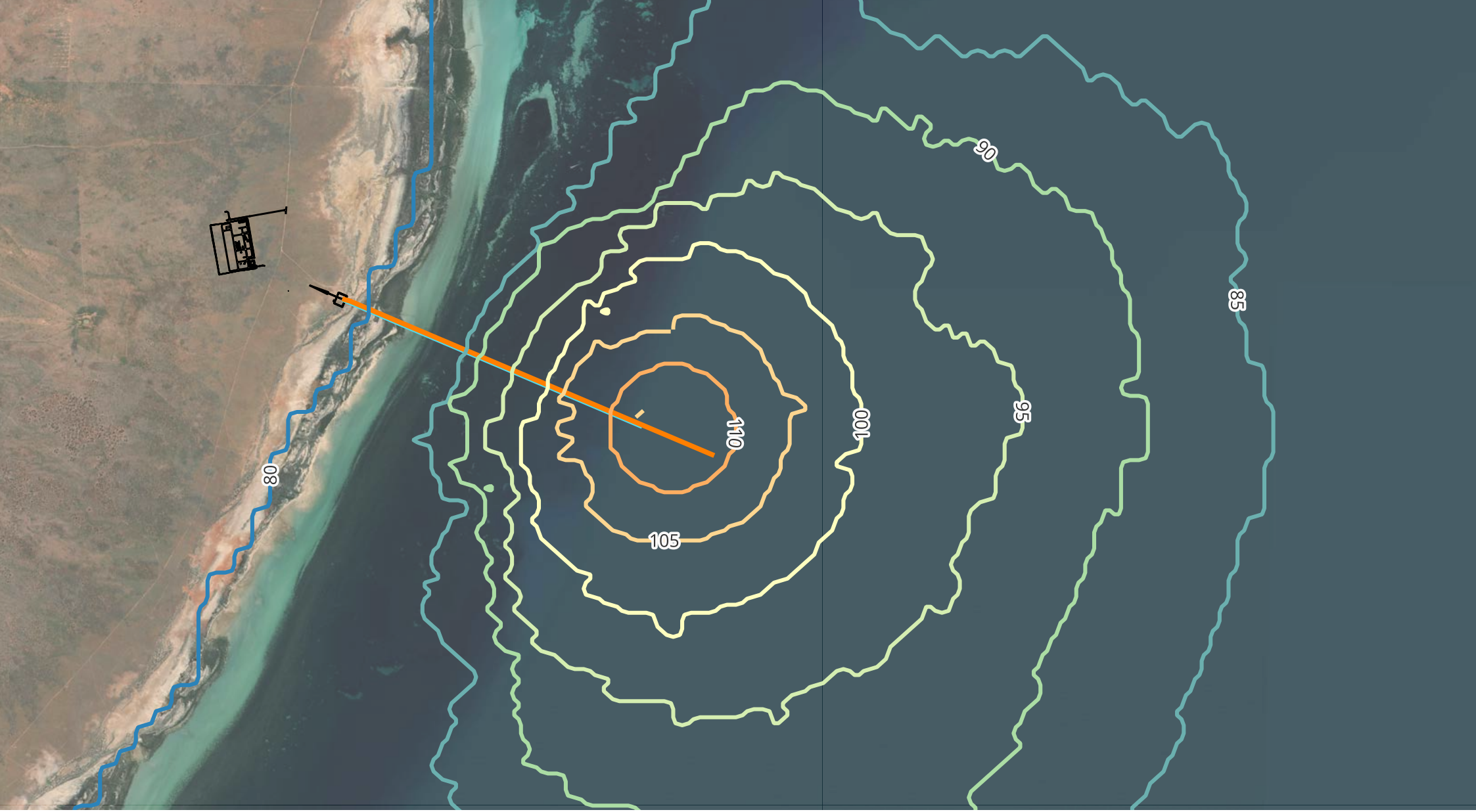
Northern Water Project

Impact piling at outlet riser
Predicted underwater noise levels (SEL24 hour)

Drawn by: JH
Checked by: NH
Dated November 2025
Page size: A3

Figure

B6



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, dB rms re 1 µPa

- 80
- 85
- 90
- 95
- 100
- 105
- 110

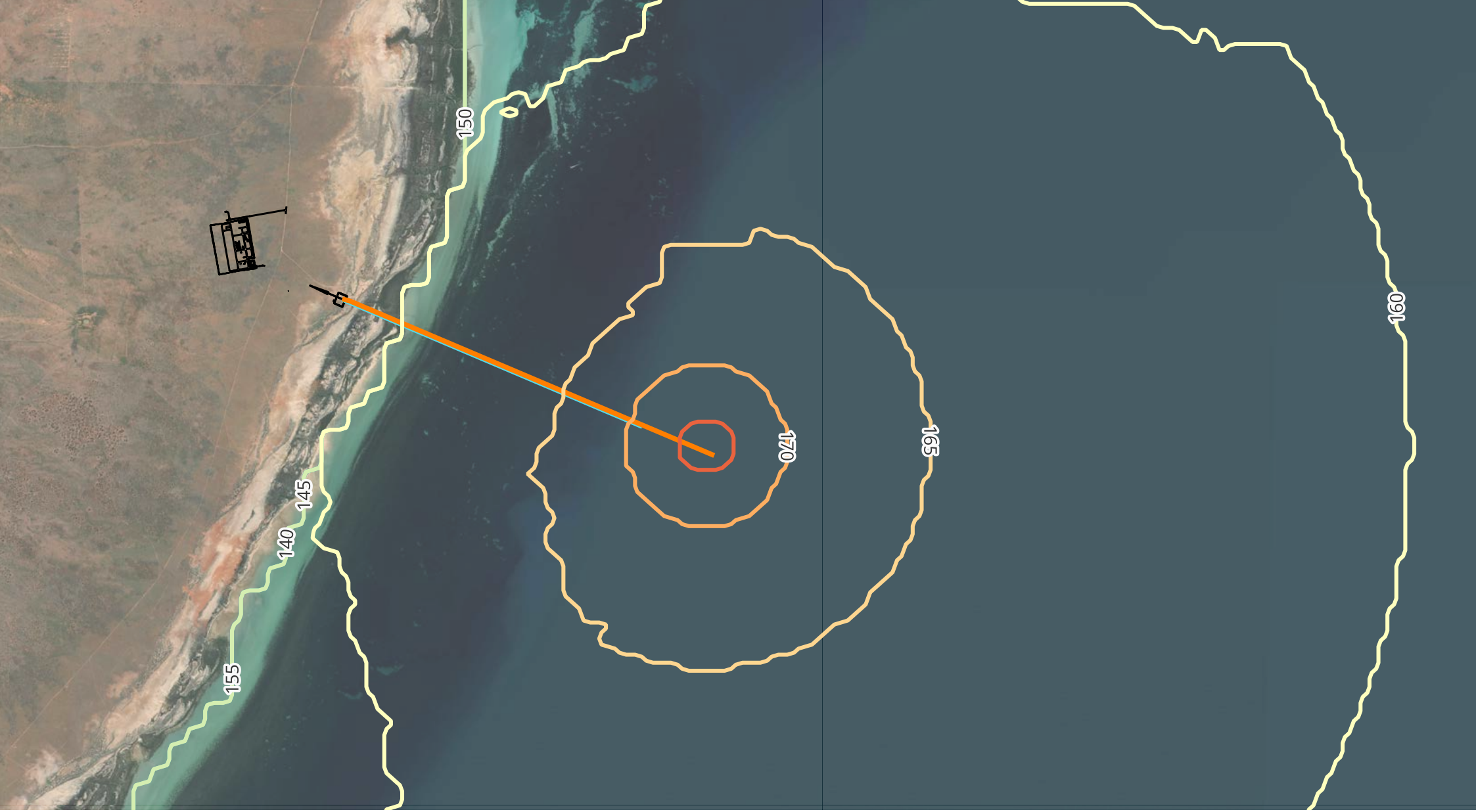
Northern Water Project

Vibratory piling at outlet site
Predicted underwater noise levels (rms)

Drawn by: JH
Checked by: NH
Dated November 2025
Page size: A3

Figure

B7



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, SEL24 hour
dB re 1 µPa2s

- 150
- 155
- 160
- 165
- 170
- 175

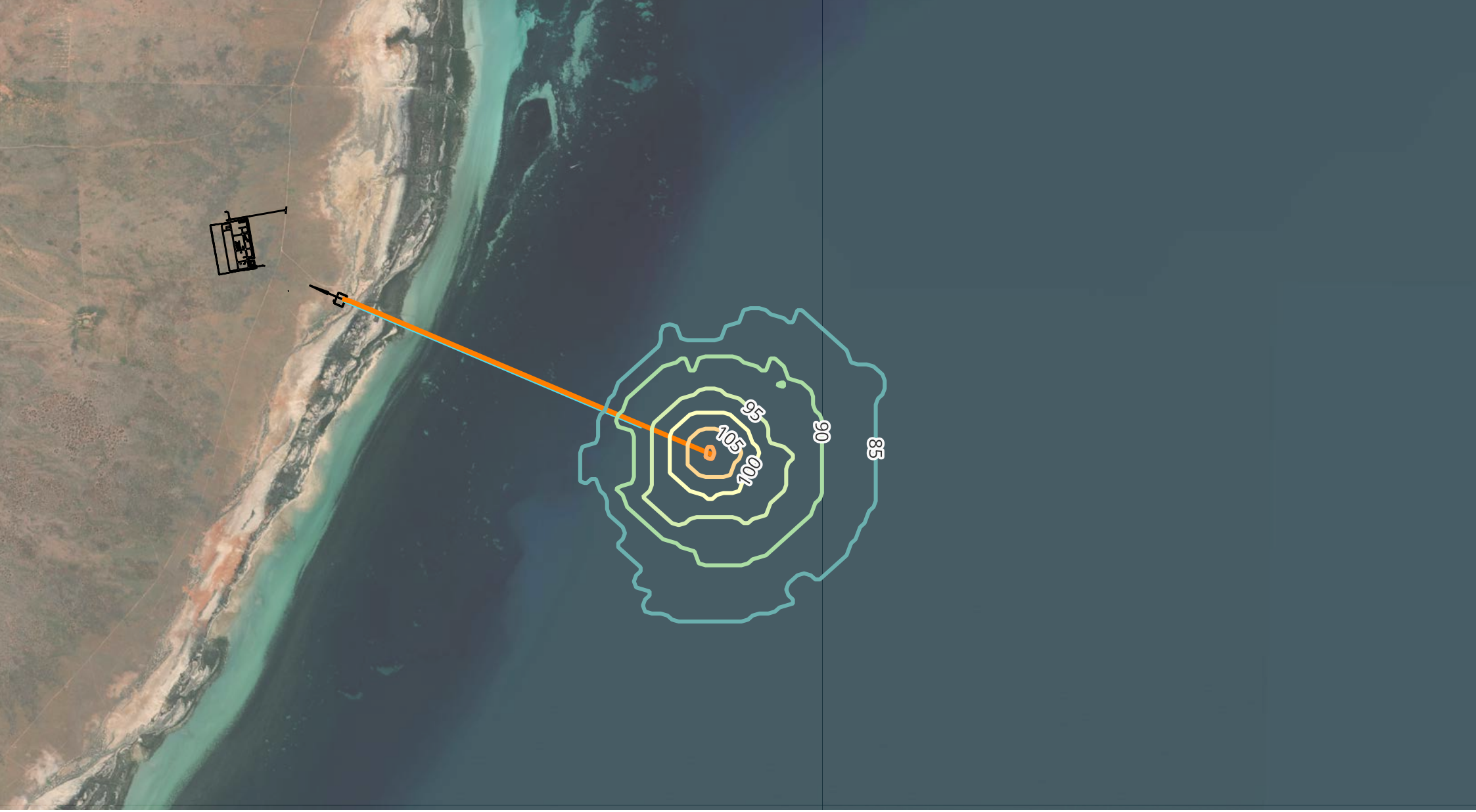
Northern Water Project

Vibratory piling at outlet riser
Predicted underwater noise levels (SEL24 hour)

Drawn by: JH
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Dated November 2025
Page size: A3

Figure

B8



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, dB rms re 1 µPa

- 85
- 90
- 95
- 100
- 105
- 110

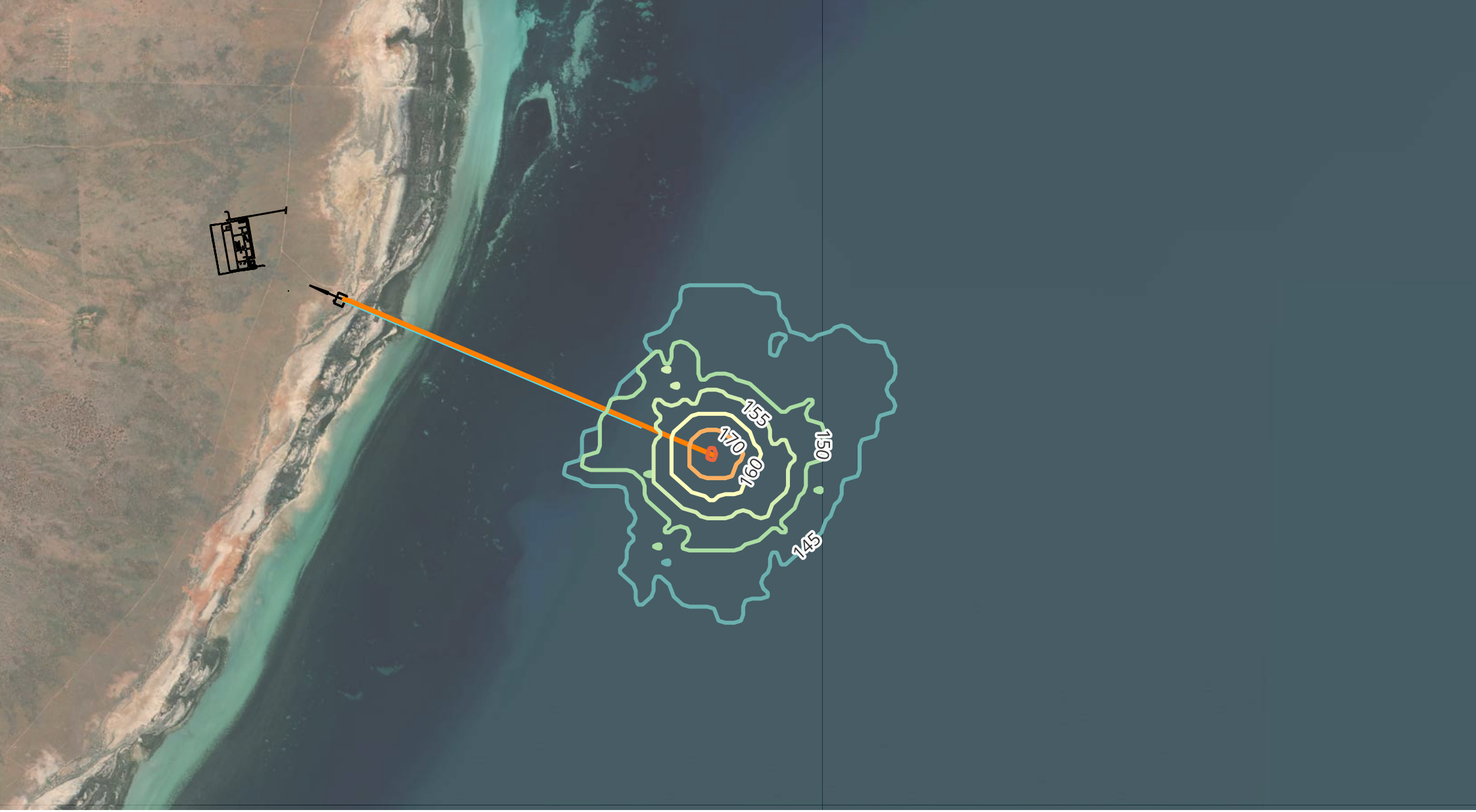
Northern Water Project

Dredging at outlet site
Predicted underwater noise levels (rms)

Drawn by: JH
Checked by: NH
Dated November 2025
Page size: A3

Figure

B9



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, SEL24 hour
dB re 1 µPa2s

- 145
- 150
- 155
- 160
- 165
- 170
- 175

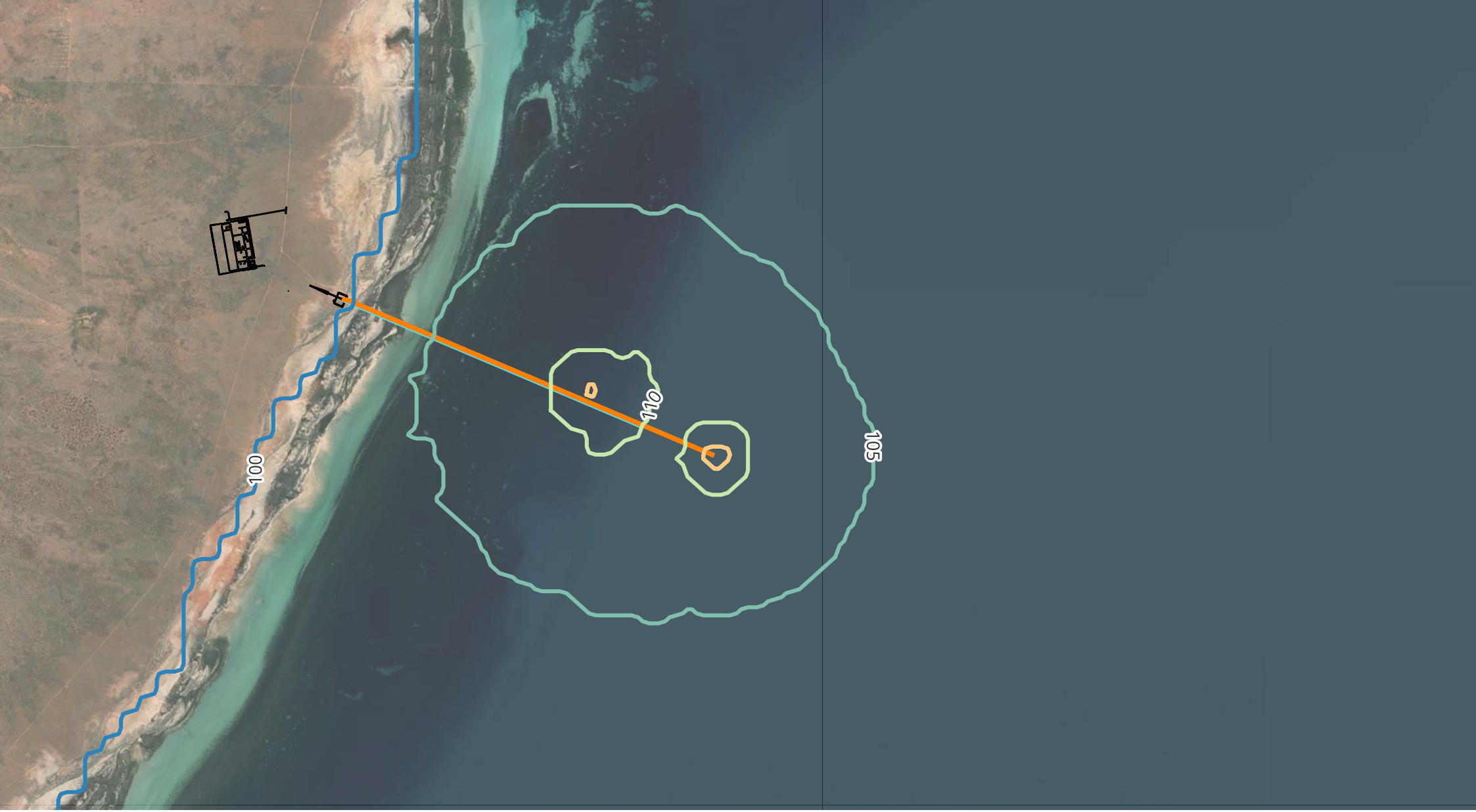
Northern Water Project

Dredging at outlet riser
Predicted underwater noise levels (SEL24 hour)

Drawn by: JH
Checked by: NH
Dated November 2025
Page size: A3

Figure

B10



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, dB rms re 1 µPa

- 100
- 105
- 110
- 115
- 120

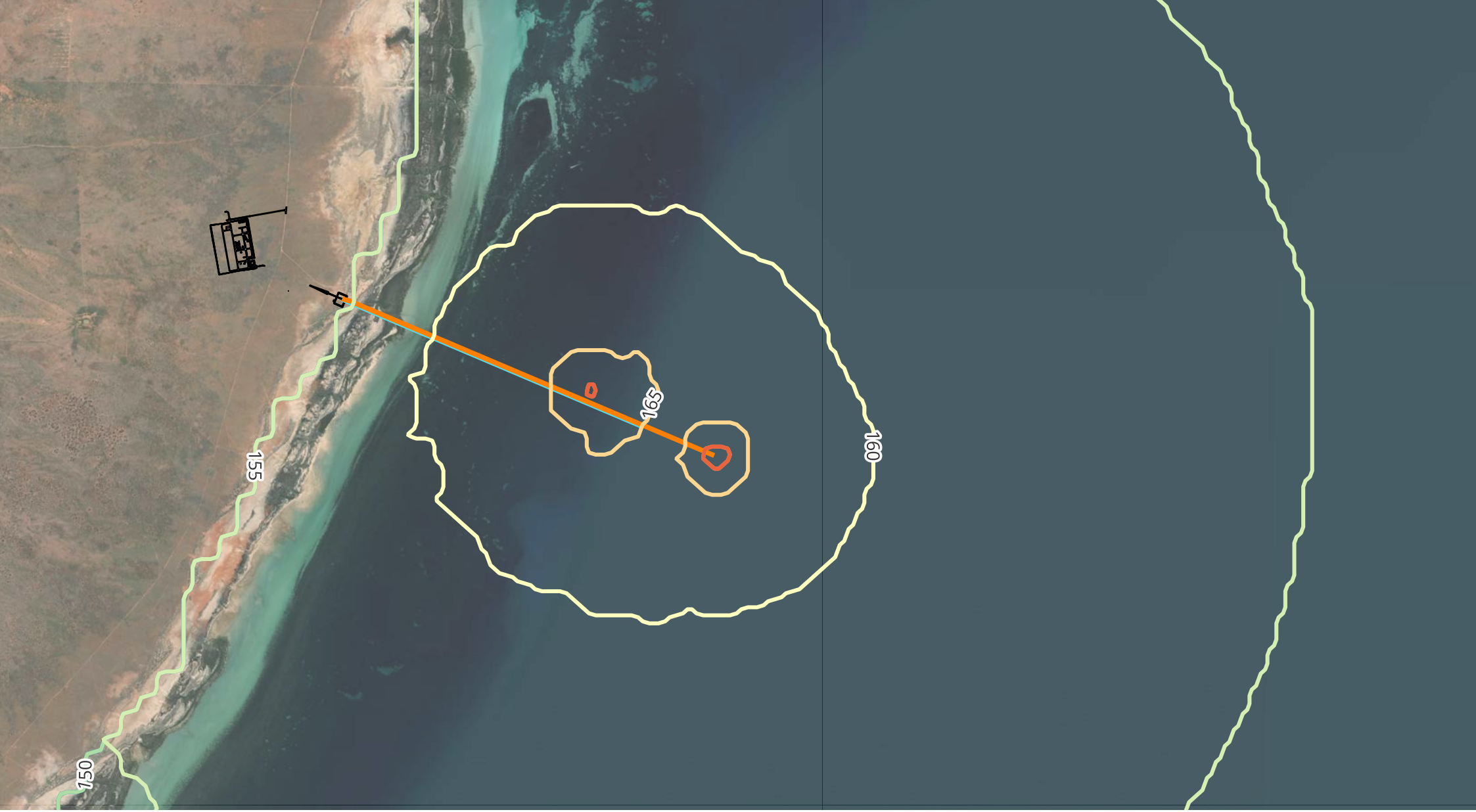
Northern Water Project

Intake and outlet pumps
Predicted underwater noise levels (rms)

Drawn by: JH
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Dated November 2025
Page size: A3

Figure

B11



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Desalination plant
- Intake tunnel
- Outlet tunnel

Predicted noise level, SEL24 hour
dB re 1 µPa2s

- 150
- 155
- 160
- 165
- 170
- 175

Northern Water Project

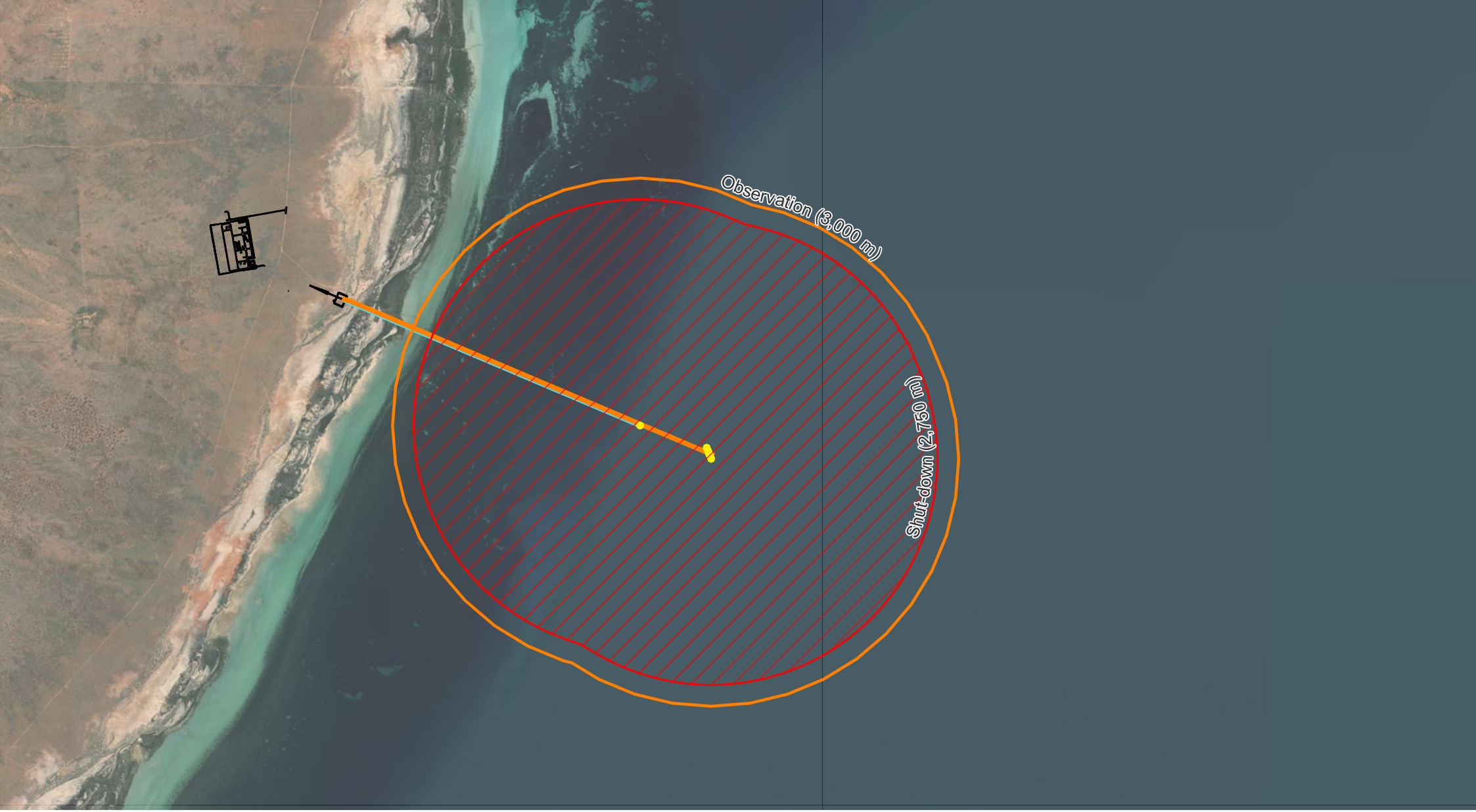
Intake and outlet pumps
Predicted underwater noise levels (SEL24 hour)

Drawn by: JH
Checked by: NH
Dated November 2025
Page size: A3

Figure

B12

Appendix C - Preliminary shutdown zones and observation zones



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Shut-down zone
- Observation zone
- Piling locations

Northern Water Project

Impact Piling
Indicative Safety Zones for Whales

Drawn by: NH
Checked by: DJ2
Dated March 2026
Page size: A3

Figure C1



Datum GDA2020/MGA Zone53

0 1 2 km

Resonate

Legend

- Shut-down zone
- Observation zone
- Piling locations

Northern Water Project

Impact Piling
Indicative Safety Zones for Sea Lions and Dolphins

Drawn by: NH
Checked by: DJ2
Dated March 2026
Page size: A3

Figure

C2

Appendix G-10:

Marine Desalination Plant Ecotoxicology Assessment

Marine desalination plant ecotoxicology assessment for the proposed Northern Water Project – Mullaquana Station option

Prepared for JBS&G Pty Ltd

Dr Rick van Dam

WQadvice

July 2026



Marine desalination plant ecotoxicology assessment for the proposed Northern Water Project – Mullaquana Station option

Prepared for JBS&G Pty Ltd

July 2026

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Key terms / Definitions

Acute toxicity	A lethal or adverse sub-lethal effect that occurs as the result of a short exposure period to a chemical relative to the organism's life span.
Antiscalant	A pre-treatment chemical that is designed to inhibit the formation and precipitation of crystallized mineral salts that form scale in water treatment systems.
Brine	A salt water solution containing a high concentration of dissolved salts, either collected *in situ* (field-collected from ocean or brackish waters) or prepared in a laboratory for experimental (testing) or other purposes.
Clean in place (CIP)	An automated method of cleaning the interior surfaces of pipes, vessels, equipment, filters and associated fittings.
Chronic toxicity	A lethal or sublethal adverse effect that occurs after exposure to a chemical for a period of time that is a substantial portion of the organism's life span or an adverse effect on a sensitive early life stage.
Control water / test diluent	The clean water that is used in a toxicity test for control (i.e. unexposed) treatments and to dissolve/dilute a chemical/wastewater for exposed treatments.
Direct toxicity assessment	The use of toxicity tests to determine the acute and/or chronic toxicity of waste water discharges or total pollutant loads in receiving waters. Analogous to whole effluent toxicity (WET) testing.
Ecotoxicity	Toxicity to non-human biota.
Ecotoxicity testing	The means by which the toxicity of a chemical or mixture of chemicals to an organism is determined. An ecotoxicity test is used to measure the degree of response produced by exposure to a chemical (across multiple concentrations) for a specified duration.
Practical salinity units (PSU)	The standard measure of salinity based on the Practical Salinity Scale (PSS-78). 1 PSU is approximately equivalent to one part per thousand salinity.
Receiving water	The ambient water into which a wastewater is to be discharged.
Return water	The concentrated saline solution produced by the desalination of seawater, that is returned back into the ocean via the outfall pipeline diffusers.
Safe dilution	The dilution rate of a wastewater discharge in the aquatic environment that is predicted to protect the aquatic ecosystem. It can be expressed in terms of the percent of wastewater or a ratio of wastewater volume to receiving water volume.
Toxicity values	Estimates of the toxicity of a chemical to an organism obtained from the results of toxicity tests.

Acronyms and abbreviations

ADP	Adelaide desalination plant
ANZECC/ARMCANZ	Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand
ANZG	Australian and New Zealand governments
BHP	BHP Billiton
CIP	Clean in place
CRED	Criteria for Reporting and Evaluating Ecotoxicity Data
DTA	Direct toxicity assessment
EC10	10% effect concentration
EC50	Median effect concentration (i.e. 50% effect)
EIS	Environmental Impact Statement
EPDP	Eyre Peninsula desalination plant
GL/y	Gigalitres per year
LC50	Median lethal concentration (i.e. 50% lethality)
ML/day	Megalitres per day
NOEC	No observed effect concentration
ODX	Olympic Dam expansion
ppt	Parts per thousand (effectively the same as PSU and grams per Litre)
PSU	Practical salinity units (effectively the same as ppt and grams per Litre)
SEIS	Supplementary Environmental Impact Statement
SGWCPFA	Spencer Gulf & West Coast Prawn Fisherman's Association Inc.
SPTV	Species protection trigger value
SSD	Species sensitivity distribution
SSDP	Southern Seas Desalination Plant
TDS	Total dissolved solids

Executive summary

The Northern Water Project (the Project) aims to provide a secure climate resilient water source for industry in South Australia's Upper Spencer Gulf and Far North. Delivery of the Project will involve the construction and operation of a 51 gigalitre per year (GL/year) seawater reverse osmosis desalination plant located at Mullaquana Station on the Upper Spencer Gulf, and associated water transfer system (including pipelines, transfer pump stations, and product water storages) and electricity infrastructure. An Environmental Impact Statement (EIS) is being prepared for the Project.

One of the environmental issues being investigated is the potential ecotoxicity (i.e. toxicity to non-human biota) of the concentrated saline water solution (return water) that will be generated by the proposed desalination plant and discharged back to the local marine environment.

WQadvice was engaged to undertake a literature review and gap analysis of existing ecotoxicity studies from other desalination plants and to design an ecotoxicity testing program for the Project.

Ecotoxicity study data were reviewed for five Australian desalination plants. Ecotoxicity studies conducted for BHP's proposed Olympic Dam Expansion at Point Lowly in the Upper Spencer Gulf and for the Adelaide Desalination Plant were found to be the most relevant to the Project. Twenty two existing chronic toxicity values for 12 species were found to be relevant to the current assessment, albeit they all had minor limitations. The relevant data were limited primarily by the salinities of the tested desalination brines and the test diluent/control waters and, to a lesser extent, by the local relevance of the species used in the ecotoxicity tests.

Of the 22 relevant toxicity values, 17 were determined to be reliable. This dataset comprised data for 12 species from eight taxonomic groups. From this relevant and reliable dataset, a preliminary ecotoxicity dataset was compiled for the Project that comprised chronic toxicity data for 10 species from seven taxonomic groups. Seven of the species represented in the dataset were considered to be locally- or regionally-relevant at the species or genus level, but none were locally collected from Upper Spencer Gulf. Toxicity values for the remaining two species, *Sepia apama* (Australian Giant Cuttlefish) and *Seriola lalandi* (yellowtail kingfish) were excluded from the preliminary dataset on the basis that the brine and control water/test diluent salinity conditions under which they were tested were extreme and implausible for the proposed site based on existing salinity monitoring data.

Using the preliminary dataset, preliminary safe dilutions were established for the Project's desalination return water of 1.9% (1 in 53), 3.9% (1 in 26) and 5.3% (1 in 19) for protection of 99%, 95% and 90% of species, respectively. These preliminary safe dilutions are expected to be conservative (i.e. protective) for the proposed site at Mullaquana Station, largely due to the conservative relevance criteria that were adopted. Consequently, the preliminary safe dilutions were concluded to be appropriate for the purposes of the EIS, with no further testing required for this phase of the approvals process.

A draft ecotoxicity testing program for the commissioning and operational phases of the Project was developed taking into account the key gaps in the preliminary ecotoxicity dataset, likely minimum requirements for future testing and key stakeholder concerns. The draft program comprised the following two components:

- Pre-commissioning phase ecotoxicity testing to validate and update the preliminary safe dilutions. This should involve assessment of the toxicity of a range of simulated desalination return water samples with and without chemical additives at their expected concentrations to approximately six species, three of which have established test methods and are already represented in the preliminary dataset, and with the remaining three being local species not

currently included in the preliminary dataset (to fill key data gaps, including unrepresented taxonomic groups).

- Operational phase ecotoxicity testing to confirm that the operational desalination plant return water toxicity is not different to what was expected based on previous testing, and that appropriate protection is being achieved. This should involve an initial round of testing using the same six species that were used for the pre-commissioning phase testing program, followed by three subsequent testing rounds (e.g. at three month intervals or as production reaches specific milestones) using the three most sensitive species from the commissioning phase testing, but ensuring that at least one of the existing standard (locally present) species is represented. The samples of desalination return water to be tested should be representative of normal operating conditions.

1. Introduction

1.1 Background

The Northern Water Project (the Project) aims to provide a secure, climate resilient water source for industry in South Australia's Upper Spencer Gulf (USG)1 and Far North. This will enable the growth of industries and key mining resources crucial for contributing to decarbonisation initiatives, sustainability and unlocking long term economic prosperity for South Australia.

Delivery of the Project will involve the construction and operation of a 51 gigalitre per year (GL/year) seawater reverse osmosis desalination plant, and associated water transfer system (including pipelines, transfer pump stations, and product water storages) and electricity infrastructure.

The proposed desalination plant location is on the western shore of the Upper Spencer Gulf at Mullaquana Station, south-west of Whyalla, and the transfer system runs generally north from the desalination plant to delivery points at the Olympic Dam and Carrapateena copper-gold mines.

An Environmental Impact Statement (EIS) is being prepared for the Project. One of the environmental issues being investigated is the potential ecotoxicity (i.e. toxicity to non-human biota) of the desalination return water (i.e. the concentrated saline solution produced by the desalination of seawater, referred to herein as 'return water' for the Project, and as 'brine' in relation to other desalination plants, or laboratory prepared toxicity testing solutions) that will be discharged into Upper Spencer Gulf due to the Project. The ecotoxicity of brines to marine species has been assessed for numerous desalination plants in South Australia and elsewhere in Australia since the mid-2000s (see Section 4.2 for further details) .

In considering the previous assessments in the context of the Project, it was prudent to ask the following questions:

- Are any existing data from previous desalination brine ecotoxicity assessments relevant?
- If so, can they be used for the Project approvals process (i.e. the Environmental Impact Statement)?
- What additional ecotoxicity work would still need to be done for the Project to meet requirements that are consistent with relevant guidance in ANZG (2018, 2023) on direct toxicity assessment (DTA)?

JBS&G engaged WQadvice to address these questions and, in the event that additional ecotoxicity studies are required, to develop a draft scope of works that reflects the needs, is scientifically robust and has taken account of lessons learned from previous studies of similar capacity desalination plants.

1.2 Assumptions

As the Project is still in the design and pre-approval phase, the following assumptions were made to progress the ecotoxicology assessment:

- The salinity of the receiving water (i.e. the ambient water into which a wastewater is to be or is being discharged) at the proposed site is in the range of 38.32 to 40.70 Practical Salinity Units (PSU; the standard measure of salinity based on the Practical Salinity Scale, with one PSU being approximately equivalent to one part per thousand salinity) (Department for Infrastructure and Transport 2025).
- The desalination return water will have the following characteristics:
    - 65.77 to 70.95 PSU salinity (based on 40% overall desalination plant recovery rate from seawater) (Department for Infrastructure and Transport 2025)

  - will include additives such as antiscalants, biocides and cleaning products but, as these are unknown at this stage, they cannot be properly factored into the current assessment.
  - will be largely continuous (except for periods of backwashing or maintenance/repair, etc.).
- The proposed design production capacity of the desalination plant is 51 GL/y with a possible ultimate future proof production capacity of 73 GL/y.

2. Objectives

The objectives of the study were to:

i. Review existing information from other desalination plant ecotoxicity studies and provide advice on the relevance and reliability of, and gaps in, the information, for the Project;

ii. Based on the outcomes of i, above, determine whether preliminary "safe" dilutions for the discharge of desalination return water to the marine environment can be derived for the EIS;

iii. If possible, derive 90%, 95% and 99% species protection safe dilutions for the discharge of desalination return water to the marine environment at Mullaquana Station; and

iv. Based on the outcomes of i to iii, above, develop a fit-for-purpose ecotoxicity testing program and associated scope of works for implementation during the relevant future phases of the Project.

3. Approach

The approach involved the following steps:

i. Review existing literature on desalination brine ecotoxicity;

ii. Assess existing ecotoxicity data for relevance and reliability;

iii. Using only data identified as both relevant and reliable, identify an acceptable preliminary dataset and, if possible, derive preliminary 'safe' dilutions for the Project's desalination return water;

iv. Identify data gaps; and

v. Based on steps i to iv, above, develop a scope of works for a future ecotoxicity testing program for the Project.

3.1 Review of existing desalination brine ecotoxicity studies

3.1.1 Literature review sources

The literature reviewed included a range of published and unpublished reports and scientific papers. Specifically, it included:

- Any relevant documentation provided to JBS&G by the Department for Infrastructure and Transport, including relevant stakeholder submissions;
- Historical ecotoxicity assessment reports and supporting information for other desalination plants, such as BHP's proposed Point Lowly desalination plant, the Adelaide Desalination Plant (ADP), and interstate desalination plants;
- Salinity data for the proposed desalination plant site;
- Information on the biota of the marine environment adjacent to the proposed desalination plant sites;

- Other government reports related to the impacts of desalination brine on the marine environment; and
- Peer reviewed scientific papers, including ecotoxicity/impact assessments and literature reviews.

To ensure transparency of the assessment, all information used in the review needed to be publicly available. The review focused at two levels:

i. Overview of general desalination brine ecotoxicity literature – rather than reviewing many publications focused on individual studies that might not have direct relevance to the Project, recent literature reviews were relied upon. Moreover, only a brief overview of the current understanding was provided; and

ii. Detailed review and assessment of specific Australian desalination brine ecotoxicity studies – this review aimed to identify existing reliable data that were relevant for deriving a preliminary safe dilution (i.e. the dilution rate of a wastewater discharge in the aquatic environment that is predicted to protect the aquatic ecosystem. It can be expressed in terms of the percent of wastewater or a ratio of wastewater volume to receiving water volume) for the discharge of the Project's desalination return water, as described in Sections 3.1.2 and 3.2, below.

3.1.2 Relevance and reliability of ecotoxicity data

A key aspect of the review of the Australian desalination brine ecotoxicity studies was to determine the adequacy of existing toxicity data for deriving preliminary 'safe' dilutions for discharge of the desalination return water for the Project. As the assessment was solely based on the use of existing data most of which were well over 10 years old, it was critical that a formal, transparent and rigorous approach was used to assess the adequacy of the data for the Project. The assessment of the data involved determining their relevance and reliability. Relevance and reliability are further explained in Sections 4.2 and 4.3, respectively. Briefly, data used for an assessment should be *relevant* to the issue that is being assessed (e.g. toxicity data based on short-term pulse exposures may not be relevant to long-term continuous discharges) and be *reliable* to use in the sense that they are of good scientific quality.

The approach for assessing the data, which is explained in more detail in Sections 4.2 and 4.3, was consistent with the nationally approved method for deriving water quality guideline values for toxicants (Warne et al. 2018). However, because Warne et al. (2018) does not formally consider data relevance, the data assessment approach needed to be supplemented with a process for assessing relevance. This was done based on the "Criteria for Reporting and Evaluating ecotoxicity Data" (CRED) approach developed by Moermond et al. (2016), which is formally used in Europe for assessing ecotoxicity data reliability and relevance. The CRED method is considered to be to be more accurate, applicable, consistent, and transparent for evaluating ecotoxicity data than the older but potential alternative Klimisch (Klimisch et al. 1997) method.

Thus, data relevance and reliability were assessed to help identify a preliminary dataset for deriving preliminary 'safe' dilutions, as well as to identify data deficiencies to inform future ecotoxicity testing requirements. For site-specific assessments such as the current assessment, it is more efficient to first screen for relevance and then, assess the reliability of all data deemed to be relevant.

3.2 Derivation of preliminary safe dilutions

A species sensitivity distribution (SSD) was constructed for the preliminary dataset, and preliminary safe dilutions derived using the *shinyssdtools* package (v 0.4.0) (Dalgarno 2018, Thorley et al. 2025). This package uses a model averaging approach for deriving protective concentration (PC) values

(which can be applied as guideline values or safe dilutions) that is considered to be technically superior to the single model approach used by the nationally approved software for deriving guideline values, Burrlioz 2.0 (Fox et al. 2021, 2022). Moreover, the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG 2018) are currently in the process of transitioning from Burrlioz 2.0 to *shinyssdtools* as the nationally-approved software for deriving guideline values (Fox et al. 2023).

The *shinyssdtools* package uses a model averaging approach whereby it fits several distinct distributions (e.g. gamma, log-gumbel, log-logistic, log-normal and weibull) to the toxicity data and offers the ability to derive the PC values based on one or more distributions depending on their relative fits to the data. Where more than one distribution is used for the PC value derivation, the final PC value represents a weighted average of the outputs of those models used, with weightings related to the fits of the models. In contrast, Burrlioz 2.0 fits just a single distribution (log-logistic where $n < 8$ and Burr III where $n \geq 8$) regardless of how adequately the model represents the data.

For the purposes of this assessment desalination return water dilutions predicted to protect 95% and 99% of species were estimated. Typically, 95% species protection values would be applied to slightly to moderately disturbed ecosystems and 99% species protection values would be applied to high conservation/ecological value ecosystems. However, it was outside the scope of the current assessment to make decisions on which level of protection is most applicable to the proposed site at Mullaquana Station. The reliability of the safe dilutions (as distinct to the reliability of the test data) was determined using the approach described by Warne et al. (2018), which takes into account the number and type of data that were used, and the fit of the SSD to the data. If necessary, additional caveats for the safe dilutions were reported.

3.3 Identification of data gaps

Identification of data gaps focused on the results of the relevance and reliability assessment but was also informed by key stakeholder concerns that arose from the assessment of the previously proposed Olympic Dam Expansion (ODX) at Point Lowly in the Upper Spencer Gulf or early stakeholder consultation that has taken place for the Project.

3.4 Recommendations for future ecotoxicity testing program for the Project

Recommendations for a future ecotoxicity testing program were informed by the outcomes of the literature review and data assessment (Section 4), derivation of preliminary safe dilutions (Section 5) and identification of data gaps (Section 6). In addition, minimum requirements for an ecotoxicity testing program according to relevant national guidance were taken into consideration when designing the program.

4. Review of desalination brine ecotoxicity studies

4.1 Existing literature on desalination brine ecotoxicity

4.1.1 Existing reviews

Many published reviews on the impacts of brine from desalination plants identify a series of potential or actual impacts, only some of which are related to toxicity. The likelihood of ecotoxicological impact will be dependent on one or more of numerous factors, including (but not limited to): the salinity of the brine; naturally occurring metals (e.g. copper) and other compounds concentrated in the brine or released as a result of corrosion; types and concentrations of chemicals added to the desalination brine (e.g. antiscalants, corrosion inhibitors, biocides); other physico-chemical characteristics of or influenced by the discharged desalination brine (e.g. temperature, pH, dissolved oxygen); the salinity of the receiving environment; the bathymetry and hydrodynamics of the receiving environment; the nature of the discharge plume dispersion and dilution; and specific sensitivity of organisms in the

receiving environment (Roberts et al. 2010, Missimer & Maliva 2018, Kress 2019, Panagopoulos & Haralambous 2020, Gillanders & Stockbridge 2022). Given these many factors, it is very difficult to directly apply published statements of impact or no impact of desalination brines to a specific individual situation such as the Project. Consequently, general reviews of the literature offer limited value for assessments of specific (actual or proposed) desalination brines and, as such, only a limited review was conducted here, with more effort focused on the review and assessment of potentially relevant ecotoxicity data (Sections 4.2 to 4.5).

The review of general desalination brine ecotoxicity detailed in this section was mostly confined to peer-reviewed, published reviews from the past decade (2016-2026), although a few older reviews and experimental studies were included where they were deemed to be particularly relevant. Summaries of key points from the reviewed publications are provided below. The primary interest was on information related to the direct ecotoxicity of desalination brines, with environmental monitoring studies considered only where they added context or understanding to the ecotoxicity knowledge.

Roberts et al. (2010) reviewed the impacts of desalination brines on the marine environment. Although a little dated, there are some noteworthy observations/findings. Roberts et al. (2010) noted that laboratory ecotoxicity results for desalination concentrates do not necessarily properly reflect salinity tolerances in the field, where organisms can be adapted to higher salinities. Nevertheless, they recommended that ecotoxicity testing of desalination concentrates could be conducted over a range of environmental conditions using local species, to better characterise the interactions of these conditions with the concentrates.

Missimer & Maliva (2018) noted the differential sensitivity of marine species based on their osmoregulatory ability (e.g. marine fish have high osmoregulatory ability) and natural exposure to high salinities (e.g. the high salinity tolerance of some shellfish and reef-building corals that live in naturally high salinity seawater in the Arabian Gulf).

Kress (2019) summarised impact assessments for numerous desalination brines from different regions. Effects were measured more often in laboratory ecotoxicity studies than *in situ* studies, suggesting that laboratory ecotoxicity results should yield conservative (i.e. protective) safe dilutions (assuming the approaches for deriving safe dilutions are appropriate). Overall, however, comparisons are difficult to make due to the relatively small number of published studies, which are conducted under highly variable conditions. Reinforcing the limited value of published studies when assessing a specific discharge, Kress (2019) also concluded that the extent of impacts of desalination brines was highly site-specific and was influenced by various factors including the capacity of the plant, the desalination process, and the mode of desalination brine disposal.

Panagopoulos & Haralambous (2020) similarly noted the differential sensitivity of marine species to desalination brine, with some species able to tolerate 10 ppt increases in salinity while others only able to tolerate 2-3 ppt increases. They also raised the issue of effects on microbial communities, which represent a wide range of taxonomic groups not typically covered by ecotoxicity studies.

Gillanders & Stockbridge (2022) reviewed approximately 100 articles related to the impacts of desalination concentrates. They identified numerous potential impacts related to the types of the factors listed earlier. Accordingly, they noted that, while increased salinity is considered the greatest risk, it is not the only risk. The largest knowledge gap identified was understanding of long-term impacts of the combined effects of a desalination concentrate (i.e. combined effects of all the physico-chemical constituents in a concentrate). Regarding the Project, Gillanders & Stockbridge (2022) noted that, while there are data on the effects of desalination brine on embryo/larval Australian Giant Cuttlefish (*Sepia apama*), there is no knowledge on the effects on juvenile or adult cuttlefish and how desalination brines in Upper Spencer Gulf might affect them as they approach the breeding aggregation site. The proposed site at Mullaquana Station is approximately 34 km from the breeding

site at Point Lowly. Although there is no cuttlefish breeding habitat at the proposed desalination site, juvenile and adult cuttlefish are broadly distributed in Upper Spencer Gulf and are likely to be present at the site. Regardless, due to logistical and, almost certainly, animal ethics constraints, this knowledge gap could only be addressed by post-commissioning field monitoring studies rather than laboratory ecotoxicity studies.

Omerspahic et al. (2022) noted that the ecotoxicity of desalination concentrates can be influenced by the temperature of the mixture, with potentially higher toxicity at higher temperatures. They also reported studies that found that salinity tolerance was species-specific, with fish being more tolerant to salinity than invertebrates (also noted by Gillanders & Stockbridge 2022).

Henrique-Gomes et al. (2023) reviewed the effects of desalination discharges on phytoplankton and zooplankton. They identified hyper-salinity as the main cause of toxicity to plankton, and concluded that phytoplankton are more sensitive than zooplankton. In laboratory studies, EC50 concentrations for phytoplankton exposed to desalination brines ranged from around 42 ppt to 62 ppt, whereas EC50s for zooplankton were >65 ppt. However, the number of studies was limited and the results may not be broadly representative. The main reported effects on phytoplankton included reduced primary productivity, a loss in diversity and a change in the community structure of planktonic populations due to the dominance of saline-tolerant groups. They also noted that temperature and chemical additives, particularly coagulants, can also contribute to toxicity but to a lesser extent than salinity.

Shokri & Sanavi Fard (2023) reviewed the environmental footprints of water desalination and alleviation strategies. Only part of the review was dedicated to ecotoxicity and effects on marine biota. They reported that elevated salinity is the dominant contributor to the ecotoxicity of desalination brines, with temperature increases and chemical additives (such as antiscalants, coagulants, and heavy metals) potentially acting as important secondary stressors. They noted sub-lethal and lethal toxicity to seagrass species at salinities of >39 ppt and 45 ppt, respectively. While the review noted effects on numerous other types of marine organisms, it did not report the associated salinity levels. In the field, the severity of impacts to marine ecosystems is site-specific and depends on brine composition, discharge method, and local hydrodynamics.

Song et al. (2025) compared results of laboratory and field studies on impacts of desalination brines. They reported that, for most species, salinity is the primary cause of toxicity in both laboratory and field settings. Chemical additives (especially antiscalants) can contribute to toxicity, albeit generally to a lesser extent. Laboratory toxicity tests generally appeared to overestimate ecological risks compared to field observations, due to constant exposure and lack of environmental variability in lab settings. This finding is consistent with the comments by Roberts et al. (2010) that laboratory studies may under-estimate the salinity tolerances of some species.

Gupta et al. (2026) reviewed the global literature on the impacts of desalination brine discharges on marine ecosystems. It concluded that taxa exhibit a wide range of sensitivities to brine exposure, from very sensitive to highly tolerant, with benthic and sessile species being the most vulnerable due to their limited mobility. Temperature was noted as a potential stressor albeit not in detail, and no specific conclusions were drawn about its role.

4.1.2 Specific studies

Casas & Leparc (2011) assessed the ecotoxicity of four desalination brine discharges to five species and derived safe dilution factors for each brine. The ambient seawater salinity (used for the diluent/control water) and the brine salinity are shown in Table 1. Two of the four assessments were done using a diluent/control water of 39 ppt, while the other two were done using diluent/control salinities of approximately 36-37 ppt. Brine salinities were also variable, ranging from 51 to 71 ppt. Based on the salinities of the diluent/control water and the brine, it would have been predicted that

Table 1 Details of the ecotoxicity study of four desalination brines, by Casas & Leparc (2011)

Desalination brine	Salinity (PSU)		95% species protection concentration (%) and safe dilution [a]	Salinity (PSU) at safe dilution (salinity difference from ambient)
	Ambient seawater salinity	Brine salinity		
Plant 1	39.2	71	9.4% 1:11	42.2 (+3.0)
Plant 2	39.1	69	5.5% 1:18	40.7 (+1.6)
Plant 3	36.7	61	13.2% 1:7.6	39.9 (+3.2)
Plant 4	35.9	50.6	5.75% 1:17.4	36.7 (+0.8)

[a] Based on ecotoxicity EC10 values.

the brines from plant 1 and 2 (i.e. with higher diluent/control water and brine salinities) would have been the most toxic and resulted in the most conservative safe dilutions with the lowest tolerable increases in salinity from the ambient seawater. However, while the plant 2 brine did result in the most conservative safe dilution (5.5% or 1:18) with a tolerable increase in salinity from ambient salinity of 1.6 ppt, the plant 4 brine, which had the lowest diluent/control water and brine dilution (5.75% or 1:17:4) with an even lower tolerable increase in salinity from ambient salinity (only 0.8 ppt).

Thus, in laboratory ecotoxicity studies, higher ambient water and brine salinities may not always equate to higher toxicity or higher safe dilutions. This can be due to a range of factors including other constituents in the brines that are contributing to toxicity, but also artefacts associated with the quality of the ecotoxicity data and the nature of the SSD models used to derive the safe dilutions. In the case of the Casas & Leparc (2011) study, the use of only five species to assess ecotoxicity might have been insufficient to properly characterise the toxicity of the brines and associated SSDs and safe dilutions. Nowadays, robust ecotoxicity data for at least eight species (from at least four taxonomic groups) would be the minimum required for such an assessment (ANZG 2023). The outcomes of the study also indicate the difficulty in using data from one brine to predict toxicity and safe dilutions for another brine when ambient seawater and brine salinity are similar, let alone different. This is further considered in Sections 4.3.1 and 4.3.2.

Clark et al. (2018) undertook a field study to assess the impacts and subsequent recovery of sessile marine invertebrate recruitment near a desalination outfall near Kurnell in New South Wales. Within the vicinity of the outfall (i.e. ~100 m), polychaetes, bryozoans and sponges were reduced in cover, whereas barnacles were more abundant near the outfall. The effects represented a localised re-distribution of species rather than any large loss of biodiversity. The salinity change across this zone of impact was approximately 1 PSU, which was considered to be relatively minor and disproportionate to observed effects. However, the nature of the community changes were consistent with flow preferences of the various taxonomic groups. Consequently, Clarke et al. (2018) proposed that the measured responses were due to changed hydrodynamic conditions due to the discharge of the desalination brine rather than water quality.

Two studies on the effects of desalination brine discharges on fish assemblages indicated that fish were not affected by the small increases in salinity near the outfall sites (i.e. 1 to 1.3 PSU above ambient salinity) (Kelaher et al. 2020, Whitmarsh et al. 2020). For the ADP outfall site, fish abundance and diversity was high and comparable to the reference sites, and the brine discharge had no discernible detrimental effect (Whitmarsh et al. 2020). For the Sydney desalination plant outfall site,

fish assemblages were enhanced during plant operation and tended to return to reference levels following cessation of the brine discharge (Kelaher et al. 2020). As with the findings by Clark et al. (2018) for sessile invertebrates, Kelaher et al. (2020) hypothesised that the increased turbulence associated with the outfall site may have been the cause of the increased fish abundance (i.e. due to an association of local water column disturbance with a potential for increased food availability).

Cruces et al. (2025) assessed the combined toxicity of salinity, antiscalant and coagulant in the kelp, *Lessonia berteroana*. The sub-lethal EC50 value (based on photosynthesis) for salinity alone was 54 ppt, compared to 47 ppt when 5 mg/L antiscalant (diethylenetriamine pentamethylene phosphonic acid) was added. Thus, the antiscalant resulted in a 13% increase in the toxicity of the hypersaline solution, demonstrating that salinity is the primary stressor although additives can also contribute to toxicity. Addition of 10 mg/L coagulant (ferric chloride) had no effect on the toxicity.

4.1.3 Key messages

Although the peer-reviewed, published literature offers limited information to assist with the current ecotoxicology study, some key messages from the literature are worth reiterating, as follows:

- The high salt concentration of desalination brines is the main cause of toxicity.
- Other factors can contribute to effects of desalination brines, including: metals, chemical additives (e.g. antiscalants, corrosion inhibitors, biocides), temperature, pH, dissolved oxygen, receiving environment salinity and hydrodynamics.
- Different species can have very different sensitivities to desalination brines.
- Laboratory ecotoxicity studies tend to provide conservative (i.e. protective) results, but this cannot always be assumed to be the case.
- There is still a relatively poor understanding of long-term impacts on marine ecosystems, although some Australian studies have documented environmental responses for up to 6 years.

4.2 Australian desalination brine ecotoxicity studies

Since the mid-2000s, ecotoxicity studies have been undertaken for numerous Australian desalination plants as part of their environmental impact assessments. However, not all of these studies are publicly available; those that could be obtained within the timeframe of the current assessment were reviewed (Table 2). It should be noted that all studies followed the accepted ANZECC/ARMCANZ (2000) guidance at the time for conducting DTA (the use of toxicity tests to determine the acute toxicity (a lethal or adverse sub-lethal effect that occurs as the result of a short exposure period to a chemical relative to the organism's life span) and/or chronic toxicity (a lethal or sublethal adverse effect that occurs after exposure to a chemical for a period of time that is a substantial portion of the organism's life span or an adverse effect on a sensitive early life stage) of waste water discharges or total pollutant loads in receiving waters).

4.2.1 BHP Olympic Dam expansion desalination plant ecotoxicity studies

Ecotoxicity data

As part of studies undertaken for BHP Olympic Dam Expansion (ODX) EIS and the Supplementary EIS, laboratory-based DTA studies were used to determine the toxicity of the proposed desalination plant return water (Hydrobiology 2006, Geotechnical Services 2006, 2008a, b, c). The return water tested in these studies was produced by passing seawater from the proposed desalination plant site at Point Lowly in Upper Spencer Gulf through a pilot reverse osmosis (RO) plant, with an antiscalant (Nalco PC 1020T) product also added. Point Lowly is approximately 34 km north-east of the proposed site at Mullaquana Station.

Table 2 Summary of selected ecotoxicity studies undertaken for Australian desalination brines (ordered by year)

Desalination plant	Year	No. species	Salinity of diluent (ppt)	Salinity of brine (ppt)	Reported safe dilution (% brine) [a]		Reference
					95% species protection	99% species protection	
Point Lowly (ODX)	2008 – 2010	14	35 – 45	77 – 84	3.4% (1 in 30) [b]	2.2% (1 in 45) [b]	Hydrobiology (2006); Geotechnical Services (2006, 2008a, b, c); Warne (2010a)
Adelaide (ADP)	2008	6	35 – 36	61 – 62	9.4% (1 in 11)	7.8% (1 in 13)	Hydrobiology (2008)
Adelaide (ADP)	2009	8	36	60 – 69	6.5% (1 in 16)	4.5% (1 in 23)	Hydrobiology (2009)
Victoria	2010	6	35	68 – 70	5.5% (1 in 18) [c]	3.2% (1 in 31)	Warne (2010b)
Cape Riche, WA	2011	6	35	68	2.5% (1 in 40)	1.9% (1 in 53)	Intertek (2011)
Onslow, WA [d]	2021	5	34 – 36	63 – 68	7.8% (1 in 13) [e]	4.4% (1 in 23) [e]	O2 Marine (2021)

[a] Based on samples that represented brine only or brine with antiscalant added, where the antiscalant did not significantly contribute to toxicity.

[b] Based on data for 41 ppt salinity diluent water and 79 ppt salinity brine.

[c] Not available in source reference, so calculated using shinyssdtools (v0.2.0) (Dalgarno 2018).

[d] Brine from Perth's Southern Seas Desalination Plant (SSDP) was used for the Onslow assessment

[e] Based on the 2020 sample, 36 ppt salinity diluent water and 64 ppt salinity brine.

Brief details of the Point Lowly pilot brine ecotoxicity testing program are provided below:

- Sixteen species from nine taxonomic groups were tested across several studies;
- The species list included 12 species known to be locally or regionally present and four species not (or unknown to be) locally or regionally present;
- Tests for two of the local species (sponge, *Aplysina* sp., and blue swimmer crab, *Portunus armatus*) were unsuccessful and the data were not useable; nevertheless, ecotoxicity data for 14 species from eight taxonomic groups were available to potentially inform 'safe' dilution rates;
- Three species were tested on more than one occasion, either using different life stages or to repeat tests using the same life stage;
- Tests included acute and chronic exposures (with chronic exposures including short-term sub-lethal tests on early life stages, as per the accepted definition in Australia (Warne et al. 2018)), depending on the species, with sub-chronic or chronic exposures being used for 12 of the species for which data were available; and
- Depending on the study, the reported salinity of the control/diluent water used for the testing ranged from 35 ppt to 45 ppt, while the reported salinity of the desalination brine ranged from 77 ppt to 84 ppt. Notably, not all of the control/diluent water salinities were relevant to the receiving water salinity at Point Lowly.

- The toxicity of the desalination brine samples ranged from approximately 3% to >100% brine (as EC10s), but were more typically between 3% and 30% brine.

Overall, (i) the proximity of Point Lowly to the proposed site (i.e. both being in Spencer Gulf), and (ii) the apparent local/regional relevance of many of the species used to test the Point Lowly pilot brine toxicity, suggested that these data might be of relevance to the Project.

Gillanders & Stockbridge (2022) also summarised the Point Lowly ecotoxicity studies, tabulating the test species used and associated details for those tests. In relation to the Project being located in Upper Spencer Gulf, they recommended additional ecotoxicity testing using water from the proposed site spiked with relevant additives that are expected to be used, with a focus on species that are found in Upper Spencer Gulf. These recommendations are revisited in Sections 4.5, 5.3 and 6.

Derivation of safe dilution rates

Warne (2010a) screened the Point Lowly pilot brine ecotoxicity data from the EIS and SEIS and determined a "best dataset" and a "second best dataset" with which to derive 'safe' dilutions, referred to as species protection trigger values (SPTVs). The best dataset relied on data from tests for seven species from six taxonomic groups where the diluent/control water salinity was approximately 40 ppt. Tests for species that were performed using lower (i.e. ~27-36 ppt) and higher (i.e. 45 ppt) diluent/control water salinities were not used.

Based on the best dataset for the proposed Point Lowly desalination plant, a dilution factor of 1:45 (2.2% brine) was estimated to be required to protect 99% of species (in line with ANZECC/ARMCANZ (2000) for high conservation value ecosystems) if receiving water salinity was 41 ppt or less. To account for the fact that (i) salinity at Point Lowly can reach 43 ppt, and (ii) the proposed brine discharge site represented important habitat for the Australian Giant Cuttlefish, BHP committed to a more conservative dilution factor of 1:85, theoretically to protect 100% of species. This was a very conservative (i.e. protective) dilution factor compared to those set for other desalination brines in Australia.

4.2.2 Adelaide desalination plant ecotoxicity studies

Ecotoxicity data

The initial ecotoxicity study used to assess the toxicity of the brine for the proposed ADP was conducted in 2008 (Hydrobiology 2008). The return water tested in these studies was produced by passing seawater from the proposed desalination plant site at Port Stanvac in Gulf St Vincent through the Penneshaw desalination plant on Kangaroo Island. A range of brine types were tested, including raw brine, brine following chlorination/dechlorination and brine with one of four antiscalants added (Flocon 295, Flocon 285, Nalco PC1020T and Spectroguard). The brine types adjusted to pH 7.1 (from pH 8.1) were also assessed. Brief details of the initial ADP brine ecotoxicity testing program are provided below:

- Six species from six taxonomic groups were tested, of which five were considered to be locally relevant;
- Tests included acute and chronic exposures, depending on the species; and
- The salinity of the control/diluent water used for the testing was approximately 35 ppt, while the reported salinity of the brine was 61 – 62 ppt.
- The toxicity of the brine samples ranged from 4% to >100% brine (as EC10s), but were more typically between 10% and 50% brine.

- Antiscalants contributed very little additional toxicity to the brine samples, with the exception of one antiscalant to the diatom, *Nitzschia closterium*. Antiscalant toxicity is further considered in Section 4.2.8.

A second ecotoxicity study was undertaken for the ADP using brine produced from a pilot plant at the ADP site (Hydrobiology 2009). A range of brine types were again tested (including pH-unadjusted and pH-adjusted brine), while some longer-term sub-lethal ecotoxicity tests were employed. Brief details of the pilot plant ADP brine ecotoxicity testing program are provided below:

- eight species from seven taxonomic groups were tested, of which six were considered to be locally or regionally relevant;
- Tests included acute and chronic exposures, depending on the species, including some longer than usual exposures for standard chronic tests, of 14 to 16 days for several species; and
- The salinity of the control/diluent water used for the testing was approximately 36 ppt, while the salinity of the brine was 60 – 69 ppt.
- The toxicity of the brine samples ranged from 6% to >100% brine (as EC10s), but were more typically between 10% and 50% brine.
- The diatom, *N. closterium*, was particularly sensitive to most of the brine types tested, with toxicity (as EC10s) <10% brine for most samples.
- An unidentified antiscalant contributed very little to the toxicity of the brine samples. There were small differences in toxicity between brine samples with and without antiscalant added, but these were considered to be negligible given the extrapolations and uncertainties inherent with calculating protective concentrations and safe dilutions from species sensitivity distribution data. Antiscalant toxicity is further considered in Section 4.2.8.

Further evaluation of the ecotoxicity of the ADP brine was undertaken periodically once the plant was operational, between October 2011 and August 2013 (Hydrobiology 2013). The brine toxicity was assessed for two species on five occasions. They found that the levels of toxicity were well below the level of dilution expected to be achieved within the mixing zone around the discharge point.

At least another two ecotoxicity studies were undertaken for the ADP by Hydrobiology; however, the original studies were unable to be obtained for the current review. Some of the results relating to the toxicity of chemical additives were summarised in Adelaide Aqua (2009), and are discussed in Section 4.2.8.

Derivation of safe dilution rates

Hydrobiology (2008)

For the ambient pH brine, a dilution factor of approximately 1 in 11 (~9% brine) was estimated to be required to protect 95% of species. This was the same with for brine with antiscalant added except for one antiscalant, where a dilution factor of 1 in 26 (3.9% brine) was required. The 95% species protection dilution factors for the pH-adjusted samples ranged from 1 in 11 (~9% brine) to 1 in 13 (~8% brine) for all samples, including the antiscalant that appeared to be more toxic at ambient pH (see Section 4.2.8).

Hydrobiology (2009)

For the ambient pH brine, dilution factors of approximately 1 in 11 (~9% brine with antiscalant) and 1 in 15 (~6.5% brine without antiscalant) were estimated to be required to protect 95% of species. The 95% species protection dilution factors for the pH-adjusted samples ranged from 1 in 11 (~9% brine with antiscalant) to 1 in 13 (~8% brine without antiscalant). The results supported the findings from

other studies that antiscalants and other chemicals such as sulfuric acid generally make a negligible contribution to the toxicity of the brine (see Section 4.2.8).

4.2.3 Victoria desalination plant ecotoxicity studies

Ecotoxicity data

The initial ecotoxicity study for the Victoria desalination plant used brine prepared from the Perth seawater desalination plant. The report for this study was difficult to access and, hence was not reviewed. However, the second ecotoxicity study (Warne 2010b), which assessed the toxicity of various brine samples prepared from a pilot desalination plant using local (Wonthaggi) seawater, was obtained and is summarised, below.

Four brine samples were tested that represented a range of plant operating conditions. Samples included brine plus antiscalant, as well as brine with antiscalant and a range of clean in place (CIP) additives (e.g. caustic and acidic agents) and other chemicals (e.g. surfactants, biocides). Brief details of the brine ecotoxicity testing program are provided below:

- Six species from six taxonomic groups were tested, of which all six were considered to be locally relevant (if not locally-collected);
- Tests included acute and chronic exposures, depending on the species; and
- The salinity of the control/diluent water used for the testing was approximately 35 ppt, while the reported salinity of the brine was 68 – 70 ppt.
- The toxicity of the brine samples ranged from approximately 8% to 60% brine (as EC10s).
- Only minor differences in toxicity were observed between the four samples.

Derivation of safe dilution rates

The dilutions required to achieve 99% species protection ranged from 3.2% (1 in 31) for brine with antiscalant to 7.0% (1 in 14) for brine with antiscalant and CIP chemicals. Given the high uncertainty associated with predicting 99% species protection dilutions based on data for only six species, these dilution factors can be considered to be similar.

4.2.4 Cape Riche desalination plant ecotoxicity studies

Ecotoxicity data

The ecotoxicity of a simulated brine for the proposed Cape Riche desalination plant was assessed in 2011 (Intertek 2011). A single brine sample was prepared using seawater from Cape Riche, and included an antiscalant (Nalco Permatreat PC-1020) at expected residual levels. Brief details of the ecotoxicity testing program are provided below:

- Six species from six taxonomic groups were tested, of which all six were considered to be locally relevant (if not locally-collected);
- Tests included acute and chronic exposures, depending on the species; and
- The salinity of the control/diluent water used for the testing was approximately 35 ppt, while the reported salinity of the brine was 68 ppt.
- The toxicity of the brine sample was relatively high compared to most other brine samples reviewed, with EC10s for five of the six species being in the order of 3% to 6% brine. Despite the brine and diluent water salinities being relatively similar to a number of other studies (Hydrobiology 2008, 2009, Warne 2010b, O2 Marine 2021); the median EC10/NOEC for the Intertek (2011) study was 2-4 times lower than that for the other studies (data not shown). There was no obvious reason for the apparent difference in toxicity. Although the brine

sample did contain some antiscalant, this was also the case for brines from other studies where toxicity was lower. The original author of the study, Dr David Strom, and current Intertek Ecotoxicology Laboratory Director, Dr Tristan Stringer, were contacted for any insights on the Intertek (2011) study. Neither was able to provide definitive information on why the toxicity of the sample was higher than observed in the other studies. However, operational desalination brine samples that exhibit atypically high toxicity relative to other brine samples are periodically observed, possibly as a result of differences in brine composition and also within-species variability in sensitivity of the test organisms (Dr Tristan Springer, pers comm, April 2024).

Derivation of safe dilution rates

Safe dilutions for 99% and 95% species protection were calculated to be 1.9% (1 in 53) and 2.5% (1 in 40), respectively. These dilution factors were similar to those derived for the Point Lowly desalination plant, for which brine and control water/test diluent salinities were markedly higher.

4.2.5 Onslow desalination plant ecotoxicity studies

Ecotoxicity data

The ecotoxicity of brine samples from Perth's Southern Seas Desalination Plant (SSDP) was assessed for the proposed Onslow desalination plant (O2 Marine 2021). The samples were tested in 2018 and 2020. For the 2018 assessment, four different samples were assessed, including raw brine (no additives), brine with a single cleaning chemical (Memclean R at either 0.17 g/L or 2 g/L) and brine with CIP chemicals and waste from the backwash sump. Brief details of the 2018 ecotoxicity testing program are provided below:

- Six species from six taxonomic groups were tested, most of which were considered to be locally relevant (if not locally-collected) to Onslow's sub-tropical location;
- Tests included acute and chronic exposures, depending on the species; and
- The salinity of the control/diluent water used for the testing was approximately 34 ppt, while the reported salinity of the brine was 63 – 68 ppt.
- The toxicity of the brine samples ranged from approximately 2% to 90% brine (as EC10s), but were more typically between 10% and 60% brine.
- Only minor differences in toxicity were observed between three of the samples, with the sample with the highest Memclean R concentration (2 g/L) being markedly more toxic.

The 2020 assessment involved the assessment of a single brine sample with no additives, brief details of which are provided below:

- Five species from five taxonomic groups were tested, most which were considered to be locally relevant (if not locally-collected) to Onslow's sub-tropical location;
- Tests included acute and chronic exposures, depending on the species; and
- The salinity of the control/diluent water used for the testing was approximately 36 ppt, while the reported salinity of the brine was 64 ppt.
- The toxicity of the brine samples ranged from approximately 8% to 30% brine (as EC10s).

Derivation of safe dilution rates

For the 2018 assessment, safe dilutions for 95% species protection were around 8% (1 in 13) for three of the samples, and 2.5% (1 in 40) for the more toxic sample with 2 g/L Memclean R. For the 2020 sample, the safe dilution for 95% species protection was 7.5% (1 in 13).

4.2.6 Eyre Peninsula desalination plant ecotoxicity review

The Eyre Peninsula desalination plant (EPDP) is a 24 ML/day desalination plant currently being built by SA Water in the vicinity of Port Lincoln. A desktop ecotoxicological review was undertaken as part of the approvals process for the EPDP (Hydrobiology 2022). This approach was similar to that undertaken in the current report for the Project, whereby existing ecotoxicity data were reviewed and no new ecotoxicity testing was done to inform safe dilutions.

4.2.7 Gold Coast desalination plant ecotoxicity study

An ecotoxicity study for the Gold Coast desalination plant focused on acute toxicity associated with short-term exposure (i.e. < 4 hours) to brine (as summarised in Hydrobiology 2022). Short-term exposures were used because the focus of the assessment was on the potential risk associated with occasional short exposures to poorer quality waste brines as a result of emergency plant shut downs during which low flow desalination effluent would be released through the diffuser. As the focus of the ecotoxicity assessment for the Project is on continuous brine discharge, the data for the Gold Coast desalination plant ecotoxicity study were considered to not be relevant for the current study. Moreover, as the data were not publicly available, they were not suitable for use in the current study.

4.2.8 Toxicity of chemical additives

As noted above, several of the ecotoxicity studies for Australian desalination plants have compared the ecotoxicity of desalination brine with different chemical additives (e.g. antiscalants, cleaning products, biocides) (e.g. Hydrobiology 2006, Geotechnical Services 2008c, Hydrobiology 2008, Warne 2010b, O2 Marine 2021). This has been explored in more detail below, with a summary of results from studies that derived safe dilutions provided in Table 3.

Table 3 Summary of toxicity of desalination brines with and without specific chemical additives for Australian desalination plants.

Desalination plant (study)	**Mixture**	**Type of additive**	**Safe dilution for 95% species protection**	**Difference from brine only safe dilution**
ADP (Hydrobiology 2008)	Brine only	Nil	1 in 11	–
	pH-adjusted brine	Acid	1 in 12	+10%
	Brine + 7 mg/L Flocon 295 (or 14 mg/L Flocon 295 for microalgal test)	Antiscalant	1 in 11	0%
	pH-adjusted brine + 2 mg/L Flocon 295 (or 4 mg/L Flocon 295 for microalgal test)	Antiscalant, acid	1 in 13	+20%
	Brine + 7 mg/L Flocon 285 (or 14 mg/L Flocon 285 for microalgal test)	Antiscalant	1 in 11	0%
	pH-adjusted brine + 2 mg/L Flocon 285 (or 4 mg/L Flocon 295 for microalgal test)	Antiscalant, acid	1 in 12	+10%
	Brine + 2 mg/L Nalco PC1020T (or 4 mg/L Nalco PC1020T for microalgal test)	Antiscalant	1 in 26	+240%
	pH-adjusted brine + 2 mg/L Nalco PC1020T (or 4 mg/L Nalco PC1020T for microalgal test)	Antiscalant, acid	1 in 11	0%

Desalination plant (study)	Mixture	Type of additive	Safe dilution for 95% species protection	Difference from brine only safe dilution
	Brine + 4 mg/L SpectraGuard (or 8 mg/L SpectraGuard for microalgal test)	Antiscalant	1 in 11	0%
	pH-adjusted brine + 4 mg/L SpectraGuard (or 8 mg/L SpectraGuard for microalgal test)	Antiscalant, acid	1 in 12	+10%
	Brine + 5 mg/L chlorine + 8.5 mg/L sodium bisulfite	Disinfectant, chlorine scavenger	1 in 55	+500%
	pH-adjusted brine + 5 mg/L chlorine + 8.5 mg/L sodium bisulfite	Disinfectant, chlorine scavenger, acid	1 in 12	+10%
ADP (Hydrobiology 2009)	Brine only	Nil	1 in 16	–
	pH-adjusted brine	Acid	1 in 12	-25%
	Brine + 5 mg/L unidentified antiscalant	Antiscalant	1 in 13	-20%
	pH-adjusted brine + 2.2 mg/L unidentified antiscalant	Antiscalant, acid	1 in 12	-25%
Victoria (Warne 2010b) [a]	Brine + 1.7 mg/L Nalco PC1020T	Antiscalant	1 in 18	– [b]
	Brine + 1.7 mg/L Nalco PC1020T + 0.23 mg/L Sodium dodecylbenzenesulfonate	Antiscalant, membrane cleaning surfactant	1 in 12	-35%
	Brine + 1.7 mg/L Nalco PC1020T + 0.23 mg/L Sodium dodecylbenzenesulfonate + 0.03 mg/L dibromonitrilopropionamide + 15 mg/L chlorine + unknown concentration sodium bisulfite	Antiscalant, membrane cleaning surfactant, biocide, Disinfectant, chlorine scavenger	1 in 16	-10%
	Brine + 1.7 mg/L Nalco PC1020T + 0.23 mg/L Sodium dodecylbenzenesulfonate + 15 mg/L chlorine + unknown concentration sodium bisulfite	Antiscalant, membrane cleaning surfactant, Disinfectant, chlorine scavenger	1 in 13	-20%
Onslow (O2 Marine 2021)	Brine only	Nil	1 in 12	–
	Brine + 0.17 mg/L Memclean R	Cleaning chemical	1 in 12	0%
	Brine + 2 mg/L Memclean R	Cleaning chemical	1 in 40	+330%
	Brine + CIP chemicals and backwash sumps	Unspecified	1 in 13	+10%

[a] 95% species protection safe dilution rates not available in source reference, so calculated using shinyssdtools (v 0.2.0) (Dalgarno 2018).

[b] This study did not test a brine only sample, so the brine + antiscalant sample was used to make comparisons with toxicity of the other samples.

As part of the Point Lowly ecotoxicity studies, Hydrobiology (2006) demonstrated that the antiscalant, Nalco PC1020T at a concentration of 7 mg/L, increased the toxicity of the brine to the diatom, *N. closterium*, but this was not the case for the other species. Also as part of the Point Lowly studies, Geotechnical Services (2008c) assessed the toxicity of Nalco PC1020T to larval yellowtail kingfish (*Seriola lalandi*). The toxicity of brine with 7 mg/L Nalco PC1020T was no different to that of brine

without the antiscalant following both 2-day pulsed and 7-day continuous exposures. When its toxicity was assessed alone (i.e. in clean seawater), a concentration of 7 mg/L Nalco PC1020T resulted in an 18% reduction in larval growth.

As part of the ADP ecotoxicity studies, Hydrobiology (2008, 2009) assessed the toxicity of brines with various additives, including four specific antiscalants (Flocon 295 up to 7 mg/L, Flocon 285 up to 7 mg/L, Nalco PC1020T at 2 mg/L and SpectraGuard at 4 mg/L for most species and 14 mg/L, 14 mg/L, 4 mg/L and 8 mg/L, respectively, for *Nitzschia closterium*). The resultant safe dilutions for the various mixtures are presented in Table 3. Only two of the samples exhibited toxicity that was markedly different to the brine only sample – i.e. the ambient pH sample that included the antiscalant Nalco PC1020T and the ambient pH chlorinated/dechlorinated sample (Hydrobiology 2008). The increase in the safe dilution of these two samples compared to other samples was mostly due to high toxicity of the samples to the marine diatom, *N. closterium*. The result for Nalco PC1020T was consistent with that previously reported by Hydrobiology (2006). However, the corresponding pH-adjusted brine samples exhibited similar toxicity to the ambient pH brine only sample, making the reliability of the two anomalous results difficult to ascertain. Nevertheless, 14 of the 16 samples tested across these two studies exhibited very similar toxicity, indicating that, at the concentrations tested and with the two exceptions noted, the additives did not significantly contribute to toxicity.

As part of the Victorian desalination plant ecotoxicity studies, Warne (2010b) assessed four brine samples with different chemical additives including an antiscalant (Nalco PC1020T at 1.7 mg/L), membrane cleaning surfactant (Sodium dodecylbenzenesulfonate at 0.23 mg/L) and biocide (dibromonitrilopropionamide at 0.03 mg/L) (Table 3). The minor differences in toxicity and, therefore, safe dilutions that were measured were considered to be negligible. Although a brine only sample was not tested, the toxicity results were consistent with those of the brine only samples from other studies.

As part of the Onslow desalination plant ecotoxicity studies, O2 Marine (2021) assessed the contribution to toxicity of the cleaning chemical, Memclean R. The toxicity of brine with a Memclean R concentration of 0.17 mg/L was no different to that of brine only. However, the toxicity of brine with a Memclean R concentration of 2 mg/L was markedly higher than that of brine only, highlighting the importance of the concentrations of chemical additives that are used. O2 Marine (2021) also assessed the toxicity of brine with unspecified CIP chemicals and backwash water, and found that toxicity was no different to brine only.

Adelaide Aqua (2009) provided summary results from two additional studies by Hydrobiology that assessed the ecotoxicity of specific CIP chemicals as part of the ADP ecotoxicity studies. Unfortunately, the original Hydrobiology reports were unable to be obtained for the current review, so the reliability of the original data could not be assessed or confirmed. Nevertheless, a summary is provided here. Six samples were assessed for toxicity, as follows:

- Desalination brine with no additives
- Desalination brine with sodium hypochlorite (with free chlorine of 15 mg/L), neutralised with sodium metabisulfite
- RO permeate with 1000 mg/L sulfuric acid and 2500 mg/L citric acid, neutralised with caustic soda
- RO permeate with 30 mg/L dibromonitrilopropionamide (DBNPA), neutralised with sodium metabisulfite
- RO permeate with 0.13% w/w sodium hydroxide, 0.05% w/w sodium dodecyl sulfate and 0.35% w/w ethylenediamine tetraacetic acid (EDTA), neutralised with sulfuric acid
- Seawater with 0.1% w/w Polymer LT25 (a polyelectrolyte flocculant).

As with the studies summarised in Table 3, the samples with chemicals added were generally no more toxic than the brine only sample. The estimated safe dilution for the brine sample was 1 in 20, while the estimated safe dilutions for four of the other samples ranged from 1 in 10 to 1 in 21. The only exception to this was the RO permeate with sodium hydroxide, sodium dodecyl sulfate and EDTA, for which the toxicity was apparently much higher and the resultant safe dilution was estimated to be 1 in 2500. This specific type of CIP solution was recommended to be disposed of via trade waste rather than as a component of the final brine discharge. Although it was not possible to verify the reliability of the ecotoxicity testing and associated analyses from the two original studies, the results reported in Adelaide Aqua (2009) are, nevertheless, consistent with those from other similar studies that were accessed and checked as part of the current review.

Hydrobiology (2008, 2022) provided brief reviews on the toxicity of key chemical additives, including those tested for toxicity. A summary of these reviews is provided in Table 4.

Table 4 Summary of review of chemical additive toxicity by Hydrobiology (2008, 2022)

Process	**Typical chemical**	**Comments**	**Typical toxicity range for marine species**
Chlorination	Chlorine (usually as sodium hypochlorite)	Free chlorine in seawater rapidly reacts with bromide and other compounds to form chlorine producing oxidants (CPOs) that can be toxic to marine biota. However, the intention for seawater desalination processes is that all free chlorine is removed prior to the desalination process, and it should not be relevant for the brine discharge.	30 to 500 µg/L CPO [a].
Dechlorination	Sodium bisulfite	As sodium bisulfite is an oxygen scavenger, it can reduce levels of dissolved oxygen in the brine.	Limited toxicity data; toxicity range unclear.
Antiscalant dosing	Sodium polyacrylate Polyphosphates	Antiscalants tested for their contribution to brine toxicity include: Flocon 295, Flocon 285, Nalco PC1020T and SpectraGuard (see Table 3).	Limited toxicity data; but see Table 3 for summary of effect of antiscalants on brine toxicity.
Flocculation	Polyacrylamide	Generally considered to be non-toxic, but can contain small amounts of, and can degrade to, acrylamide, which is a neurotoxin.	Limited toxicity data; toxicity range unclear.
Membrane cleaning	Citric or hydrochloric acid Ethylenediamine tetraacetic acid (EDTA) Surfactants – e.g. sodium dodecylbenzenesulfonate Biocides – e.g. dibromonitrilopropionamide	Membrane cleaning chemicals can have high toxicity. Membrane cleaning is never undertaken when a desalination plant is operational and all material used in this process is typically collected and stored for appropriate disposal.	Acids – 0.1 to 10,000 µg/L. EDTA – 40 to 20,000 µg/L. Sodium dodecylbenzenesulfonate – see Table 3 for information. Dibromonitrilopropionamide – 70 to 14,000 µg/L.

[a] Taken from Batley & Simpson (2020)

Despite some studies showing that some additives can contribute to the toxicity of desalination brines to some species, all the above-cited studies concluded that the toxicity is predominantly caused by the high salt concentrations in the desalination brines. Therefore, the safe dilution rates derived from brine ecotoxicity data (with or without antiscalants) are highly likely to also protect the aquatic receiving environment from toxicity of the various chemical additives. Because there may be some exceptions to this (as noted above), it is important that any commissioning and operational phase ecotoxicity testing programs for the proposed Project desalination plant include testing of brine samples that contain typical process chemicals (see Section 7.2).

4.3 Relevance of existing data

According to Moermond et al. (2016), relevance relates to the extent to which data and tests are appropriate for a particular hazard identification or risk characterization. For example, chronic exposure tests would generally be more relevant than acute exposure tests for continuous discharges, while tests based on local species would be more relevant than tests based on non-local species for a site-specific discharge toxicity assessment.

For the current study, three relevance categories were defined for ecotoxicity tests, as follows (adapted from Moermond et al. 2016):

- *Rv1 – Relevant without restrictions:* the test is relevant for the Project
- *Rv2 – Relevant with restrictions:* the test has limited relevance for the Project
- *Rv3 – Not relevant:* the test is not relevant to the Project and should not be used.

For the Project, the relevance of existing desalination brine ecotoxicity tests was assessed based on four relevance factors, as follows:

i. Salinity of the brine used in a test

ii. Salinity of the diluent/control water used in a text

iii. Test species

iv. Exposure duration (i.e. acute or chronic)

Some aspects related to relevance were *a priori* assumed or known to be met, including:

- The test species were relevant to the compartment under evaluation (i.e. the water column)
- The test endpoints were considered to be ecologically relevant (i.e. effects at the whole organism level that could reasonably be expected to manifest in population level effects – such as mortality, reproduction, development, growth). Notably, organism life stage was not considered when assessing species and endpoint relevance even though in some cases certain life stages may not be present at the proposed site.

The criteria, and associated justifications, to assess relevance for each of the four relevance factors are shown in Table 5. Where possible, tests were screened and excluded at the whole study level, thus, reducing the number of individual tests that needed to be assessed.

Once all the factors were assessed, the overall relevance score for a test was assigned as follows:

- Rv1 (relevant without restrictions) – all four relevance criteria are assigned Rv1
- Rv2 (relevant with restrictions) – at least one relevance criterion is assigned Rv2
- Rv3 (not relevant) – at least one relevance criterion is assigned Rv3.

Once a study or test scored an Rv3 for a factor, it was not further assessed for the remaining factors as at that point it would be considered not relevant regardless of the scores for the rest of the factors. Where professional judgment was required to make a decision on relevance, this was documented.

Table 5 Relevance criteria for ecotoxicity data using the four relevance factors

Relevance factor and category	Criterion	Justification
Salinity of the desalination brine used for the ecotoxicity testing		
Rv1	66 – 71 PSU [a, b]	Represents the predicted salinity for the Project's desalination return water (based on assumed 40% overall desalination plant recovery for local seawater with salinity range of 38 – 41 PSU).
Rv2	72 – 78 PSU	Represents up to 10% above the predicted high end of salinity range (71 PSU) for the Project's desalination return water. Considered to have limited relevance but could be used if insufficient data of higher relevance. Note that a salinity range below the predicted low end of salinity range (66 PSU) was not included in the Rv2 category in order to ensure that the results would not risk underestimating the toxicity of the proposed Project return water.
Rv3	≤65 PSU OR ≥79 PSU	The applicability of toxicity values (i.e. estimates of the toxicity of a chemical to an organism obtained from the results of toxicity tests) estimated from ecotoxicity tests where the brine salinity was ≤65 PSU or ≥79 PSU was considered to be too uncertain to warrant inclusion in the assessment.
Salinity of the control water/test diluent used for the ecotoxicity testing		
Rv1	38 – 41 PSU	Represents the salinity range at the proposed site, as measured over a 6½ month period (October 2022 to May 2023). It would be difficult to discern significant differences in toxicity of desalination brines across this relatively narrow site water salinity range. Therefore, this range was considered to be a reasonable criterion to identify tests that are relevant without restrictions.
Rv2	36 – 37 PSU OR 42–45 PSU	36 – 37 PSU represents up to 5% below the predicted low end of the natural salinity range (36 PSU) at the proposed site. Only 5% deviation below lower ambient salinity was allowed to minimise risk of underestimating the toxicity of the proposed Project return water. 42 – 45 PSU represents up to 10% above the predicted high end of the natural salinity range (41 PSU) at the proposed site. A 10% deviation above upper ambient salinity was allowed as data are likely to be conservative, albeit with some risk that results will overestimate the toxicity of the proposed Project return water. Considered to have limited relevance but could be used if insufficient data of higher relevance.
Rv3	≤35 PSU OR ≥46 PSU	The use of diluent/control waters with salinities below the natural salinity range at the proposed site has the potential to underestimate toxicity. Therefore, tests with diluent/control water salinities of ≤35 PSU were considered not relevant. The applicability of toxicity values estimated from ecotoxicity tests where the diluent/control water salinity was ≥44 PSU for the Project was considered to be too uncertain to warrant inclusion in the assessment, and such tests were considered not relevant.
Test species		
Rv1	The test species is locally present and was collected from or nearby the site	For site-specific ecotoxicity testing, it is preferable that data for local species collected from or nearby the site of interest are used. Such species will be adapted to any unique local environmental conditions such as higher than normal salinity or temperature.

Relevance factor and category	Criterion	Justification
Rv2	(A) The test species (or its genus) is locally present but was not collected from or nearby the site OR (B) The test species is not locally present but represents ecologically important taxonomic group that is under-represented OR (C) The test species is not locally present but was tested under brine and control water/diluent salinities representative of the site of interest	Acknowledging the challenges associated with testing sufficient locally collected species (which may require significant method development with no guarantee of success), toxicity data for other species might be considered acceptable to use under certain circumstances, according to the following hierarchy: A. In some cases, test methods and data exist for species that are locally present but were not locally collected. Or, in some cases the test species might not be locally present, but members of the same genus are locally present. If necessary, data for such test species can be used as surrogate data for local species. B. In some cases, test species might not be locally present but they represent an important taxonomic group (or trophic level) in the ecosystem that is under-represented in the dataset. There is more value in ensuring that an important taxonomic group is represented in the dataset than excluding the data because the species is not locally present. C. A significant determinant of toxicity for a toxicant or mixture is the environmental conditions in which the toxicant or mixture occurs. This is because the interactions that occur between toxicants and the chemical environment in which they are present can have a significant influence on the bioavailability and therefore toxicity of the toxicant. Thus, data for a non-local species tested under environmental conditions relevant to the site/issue of interest can still have sufficient relevance to be of value to an assessment, especially when there are too few preferred data.
Rv3	Does not meet the Rv1 or Rv2 criteria	
Exposure duration (acute / chronic)		
Rv1	Chronic exposure [c]	As the discharge of the Project's desalination return water is expected to be continuous, its toxicity is more appropriately assessed using chronic exposures than acute exposures
Rv2	No criterion for this factor	
Rv3	Acute exposure	

[a] All salinity criteria are reported to two significant figures only.

[b] Salinity values reported here as PSU but also corresponds to salinity values reported as ppt or g/L.

[c] Chronic exposures include long-term tests and short-term sub-lethal tests that are typically accepted as indicators of chronic toxicity, including short-term sub-lethal tests on critical life stages (e.g. 48-h or 72-h larval development tests). See Warne et al. (2018) for generally accepted chronic toxicity classifications for different taxonomic groups.

The results for each of the four relevance factors are presented in Appendix B, with summaries below.

4.3.1 Salinity of desalination brine used in a test

As noted in Warne (2010a), the toxicity of desalination brines is largely a function of (i) the concentrated salinity of the brine and (ii) the natural salinity of the ambient receiving water (excluding any toxicity potentially caused by chemical additives). Both sources of salts contribute to increase the salinity to a point that is toxic to an organism. There are multiple desalination ecotoxicity studies covering a range of brine salinities and diluent/control water salinities (simulating receiving water salinities) (Table 2). A key question that needs to be addressed by the relevance assessment is, which of these studies (or specifically, the individual ecotoxicity tests within the studies) are sufficiently relevant to the desalination brine salinity and receiving water salinity expected for the Project? This largely depends on the extent to which the toxicity of desalination brine differs with different brine salinity and/or test diluent salinity. To inform these questions, an assessment was undertaken using existing data for multiple desalination plants (Appendix A). It found stronger relationships for brine salinity and toxicity compared to test diluent water salinity and brine toxicity. Although relationships were mostly weak, they tended to support a positive relationship, whereby higher salinity brines are more toxic than lower salinity brines. Consequently, brine salinity was selected as the first factor to be assessed for the relevance assessment, followed by control water/test diluent salinity, test species and exposure duration.

The assessment in Appendix A confirmed that for existing ecotoxicity data to be relevant to the Project, the data need to be based on brines with a salinity similar to the predicted salinity of the Project's return water. The predicted salinity of the Project's return water at the point of discharge is 65.77 – 70.95 ppt, based on an assumed 40% fresh water recovery rate and local seawater (receiving water) salinity of 38.32 – 40.70 ppt (Department for Infrastructure and Transport 2025). Thus, these ranges were used as the basis for the relevance criteria. Table 5 lists the relevance criteria and associated justifications for the brine salinity. Due to the apparent positive relationship between brine salinity and toxicity (Appendix A), it was important to not use data from studies that used brines with salinities lower than the Project's predicted return water salinity range, in order to minimise the likelihood that toxicity would be underestimated for the Project. Therefore, the relevance criteria only considered brine salinities within or higher than 65.77 – 70.95 ppt. To avoid decisions on the criteria being made at the 10^{-1} PSU level, which was considered too precise for the purpose of the assessment, the salinity criteria were rounded to two significant figures.

The toxicity tests undertaken for the studies presented in Table 2 were all assessed for relevance. This compilation comprised data for 72 endpoints for 23 species from 10 taxonomic groups (Appendix B). Based on brine salinity alone, 33 endpoints for 13 species were assigned Rv1 (with brine salinities of 66 to 70 ppt), 10 endpoints for seven species were assigned Rv2 (with brine salinities of 77 to 78 ppt), and 29 endpoints for 17 species were assigned Rv3 (with brine salinities of 60 to 64 ppt or 79 to 84 ppt).

4.3.2 Salinity of diluent/control water used in a test

Although the toxicity of a desalination return water is also likely to depend on the salinity of the receiving water (which is simulated in ecotoxicity tests by the diluent/control water), it appears to be a less influential factor than return water salinity (Appendix A). However, the analyses comparing control water/test diluent salinity and return water toxicity were based on small sample sizes and showed relatively high variability for some species (Appendix A), making it difficult to discount control water/test diluent salinity as an important variable. Therefore, it was assumed that to be considered relevant to the Project, existing ecotoxicity data needed to be based on control water/test diluent with a salinity that is similar to the salinity of the receiving water at the proposed site.

Information provided by JBS&G indicated that the seawater salinity range at the proposed site is 38.3 to 40.7 PSU, with a median of 39.4 PSU (Department for Infrastructure and Transport 2025). Thus, the reported salinity range was used as the basis for the relevance criteria shown in Table 5. Because test diluent/control water salinity appears to be less influential than return water salinity for determining toxicity, additional flexibility in the relevance criteria for diluent/control water was afforded, with the ability to consider data from studies that used diluent/control waters with salinities slightly lower than the reported range for Mullaquana Station. To avoid decisions on the criteria being made at the 10^{-1} PSU level, which was considered too precise for the purpose of the assessment, the salinity criteria were rounded to two significant figures. Justifications for each of the relevance criteria are also provided in Table 5.

Based on diluent/control water salinity alone, 16 endpoints for 10 species were assigned Rv1 (with diluent/control water salinities of 38 to 41 ppt), 33 endpoints for 17 species were assigned Rv2 (with diluent/control water salinities of 36 to 37 ppt or 44 to 45 ppt), and 23 endpoints for 12 species were assigned Rv3 (with diluent/control water salinities of 34 to 35 ppt) (Appendix B).

4.3.3 Test species

Ideally, site-specific ecotoxicity studies should focus solely on local species, preferably specimens collected from or nearby the site. However, it is often difficult to obtain data for sufficient locally collected species (e.g. ≥8 species) to generate defensible safe concentrations, and some pragmatism is often needed to ensure that additional data with sufficient relevance can be included. In assessing data for species that were not locally collected, a hierarchy of relevance was applied, as follows:

A. Data for existing test species that are known to be locally present (e.g. the diatom *N. closterium*) or a member of the same genus is known to be locally present (e.g. the sea urchin *Heliocidaris tuberculata*)

B. Data for test species that are not locally present but represent ecologically important taxonomic groups that are otherwise under-represented in the dataset (e.g. zooplankton)

C. Data for test species that are not locally present and for which the taxonomic group is not under-represented in the dataset, but were tested under return water and diluent/control water salinities representative of the site of interest.

Otherwise, tests based on non-local test species are not relevant unless it is deemed appropriate to include them on the basis of increasing sample size and taxonomic diversity. The relevance criteria and associated justifications for test species in Table 5 were informed by the guidance provided in van Dam et al. (2019). The Rv2 category was sub-divided according to the three aspects (a – c) listed above in order to provide greater resolution of limitations of the data in relation to the relevance of the test species, which in turn helped inform the future ecotoxicity testing needs. Species occurrence at or near the proposed site was based on information from previous Environmental Impact Statements from the region (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013) and other available literature (e.g. Wiltshire et al. 2010) and advice from Mr James Brook (J Diversity Pty Ltd).

Based on test species relevance alone, 14 endpoints for three species were assigned Rv1 (species locally present and locally collected), 49 endpoints for 17 species were assigned Rv2 (species or genus locally present but not locally collected), and nine endpoints for five species were assigned Rv3 (species not locally present, not belonging to an under-represented taxonomic group and not tested under relevant conditions) (Appendix B). Those endpoints assigned Rv2 for test species relevance were further categorised, as follows (see Appendix B for details):

- 36 were assigned Rv2(A), representing 12 species
- 10 were assigned Rv2(B), representing three species

- three were assigned Rv2(C), representing two species.

4.3.4 Exposure duration (i.e. acute or chronic)

As the Project's return water discharge will be a continuous discharge, its toxicity is more appropriately assessed using chronic exposures than acute exposures. In some cases, acute exposures might be deemed relevant, or the associated acute toxicity data can be converted to estimated chronic toxicity data using conversion factors. Table 5 lists the relevance criteria and associated justifications for exposure duration. It is important to note that chronic exposures are inclusive of long-term exposures as well as short-term exposures that result in sub-lethal effects to typically sensitive early life stages of organisms. This is consistent with the definition of chronic toxicity adopted for the ANZG (2018) Guidelines (including Warne et al. 2018) as well as for ecotoxicity test methods developed by the United States Environment Protection Agency (e.g. USEPA 2002).

Based on exposure duration alone, 63 endpoints for 22 species were assigned Rv1 (chronic exposure), and nine endpoints for five species were assigned Rv3 (acute exposure) (Appendix B).

4.3.5 Relevance results

A summary of the results of the relevance assessment is provided in Table 6. None of the endpoints attained an overall relevance score of Rv1 (relevant without restrictions), while 22 endpoints were assigned an overall relevance score of Rv2 (relevant with restrictions) (Table 6; Appendix B). Thus, all the relevant data have some limitations associated with them, as summarised in Table 6. That no data were relevant without restrictions is not surprising given that the data were generated for other desalination plants in other locations and with brine or receiving water salinities that generally differed to some degree to those predicted to be the case for the Project's return water. Fifty endpoints were considered to not be relevant (i.e. Rv3). The 22 Rv2 endpoints listed in Appendix B were subsequently assessed for reliability, as detailed in Section 4.4.

At this point it is worth noting the toxicity data for two species, *S. apama* (Australian Giant Cuttlefish – three toxicity values) and *S. lalandi* (yellowtail kingfish – two toxicity values) from Geotechnical Services (2006) and Geotechnical Services (2008c), respectively, were based on a brine salinity (78 ppt) and diluent salinity (44.3 – 45 ppt) that were both at the upper end of their relevance thresholds (see Appendix B). This represents an extreme scenario for the proposed site given that the maximum recorded salinity at the proposed site is 40.7 ppt (Department for Infrastructure and Transport 2025) and the proposed 40% overall desalination plant recovery rate from local seawater would result in a maximum desalination return water salinity of 71 ppt (Department for Infrastructure and Transport 2025). These data are likely to markedly overestimate the toxicity of desalination return water for the proposed site. Although the data for these two species were progressed through to the reliability assessment, their relevance for inclusion in the final dataset was further reduced for reasons other than salinity, as outlined in Section 5.1.

4.4 Reliability of relevant data

According to Moermond et al. (2016), reliability relates to the inherent scientific quality of a conducted ecotoxicity test and its associated data and data analyses, regardless of the purpose for which the test was done or is intended to be used. It provides an indication of the level of confidence in the quality of the data and the study from which they were generated. Reliability is determined by the assessment of the design, documentation, performance and analysis of the experiment, and includes consideration of aspects such as (but not limited to) test control performance, maintenance of water quality, reference toxicity testing and data analysis.

Table 6 Summary of overall relevance results for desalination plant ecotoxicity test endpoints[a]

Relevance score	Number of tests/endpoints	Details
Rv1 – relevant without restrictions	0	• Zero endpoints that scored Rv1 for all four relevance criteria.
Rv2 – relevant with restrictions	22	Key restrictions included: • Nine endpoints were based on brine salinity 6-7 ppt above that predicted for the Project's return water – this could slightly overestimate toxicity for the Project's return water. • All 22 endpoints were based on diluent/control water salinity outside the salinity range measured at the proposed site: 17 endpoints were 1-2 ppt lower than the range, which could slightly underestimate toxicity for the Project's return water, while five endpoints were 3-4 ppt higher than the range, which could slightly overestimate toxicity for the Project's return water. • 10 endpoints were based on six species (or genera) that are locally present but were not locally collected (relative to the proposed site). • Four endpoints were based on two species (or genera) that are not locally present, but which represent an under-represented taxonomic group. • Three endpoints were based on two species that are not locally present nor under-represented, but were tested at relevant brine and diluent salinity conditions.
Rv3 – Not relevant	50	• 29 endpoints scored Rv3 for the brine salinity (i.e. salinity too low for 13 endpoints and too high for 16 endpoints). • 23 endpoints scored Rv3 for the diluent/control water, all for the salinity being too low. • Nine endpoints scored Rv3 for the test species relevance. • Nine endpoints scored Rv3 because they were based on acute exposures.

[a] Refer to Appendix B for details of the relevance assessment results.

For the current study, three reliability categories were defined for the data from the ecotoxicity tests/endpoints that successfully passed through the relevance assessment (i.e. that were assigned relevance categories of Rv1 or Rv2), as follows (adapted from Moermond et al. 2016):

- *Rb1 – Reliable without restrictions:* the data are of acceptable quality with no or only minor limitations
- *Rb2 – Reliable with restrictions:* the data are of acceptable quality but with moderate limitations
- *Rb3 – Not reliable:* the data are of (i) unacceptable quality, or (ii) acceptable quality with major limitations.

For the Project, the reliability of existing desalination brine ecotoxicity tests was assessed based on two reliability factors, as follows:

i. Formal data quality assessment

ii. Expert assessment of data limitations.

The criteria, and associated justifications, to assess reliability for each of the two reliability factors are shown in Table 7. Once both the factors were assessed, the overall reliability score for each toxicity was assigned as follows:

- Rb1 (reliable without restrictions) – both reliability criteria are assigned Rb1
- Rb2 (reliable with restrictions) – expert assessment of data limitations criterion is assigned Rb2
- Rb3 (not reliable) – at least one reliability criterion is assigned Rb3.

Toxicity values that scored an Rv3 for the formal data quality assessment were not further assessed. Where professional judgment was required to make a decision on relevance, this was documented.

Table 7 Reliability criteria for ecotoxicity data using the two reliability factors

Reliability factor and category	Criterion	Justification
Formal quality assessment (from Warne et al. (2018)		
Rb1	Quality score ≥50%	Acceptable quality data, as recommended in the method for deriving toxicant guideline values for the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG 2018).
Rb2	Not applied	
Rb3	Quality score <50%	Unacceptable quality data, as recommended in the method for deriving toxicant guideline values for the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZG 2018).
Expert assessment of limitations		
Rb1	No or only minor limitations	The formal quality assessment system is not designed to detect all limitations/flaws associated with a toxicity value (Warne et al. 2018). Additional assessment of study limitations is required to ensure that a toxicity value is reliable to use and, if not, what its restrictions are. The assessment was primarily focused on the following three aspects: • Results of test acceptability criteria (e.g. control performance, reference toxicity testing) • Nature of the concentration-response relationship • Appropriateness of data analysis methods Professional judgment, with appropriate justification, was used to determine the final reliability category for each toxicity value.
Rb2	Moderate limitations that are unlikely to substantively change the outcomes of the assessment if the toxicity value is used.	
Rb3	Major limitations that render the data unsuitable for use.	

4.4.1 Formal data quality assessment

The data quality assessment was based on the formal data quality assessment system prescribed in Warne et al. (2018) for assessing the quality of toxicity data to be used for deriving water quality guideline values. The assessment is based on a series of questions that informs the overall quality of an ecotoxicity test. For the purposes of this project, the formal scoring assessments for contaminants to marine/estuarine animal (Table A4 in Warne et al. 2018) and algal/plant (Table A5 in Warne et al. 2018) species were used. These have been reproduced in Appendix C. The assessment results in a percentage score that signifies that data are either of acceptable quality (score ≥50%) or unacceptable quality (score <50%). If data were considered to be of unacceptable quality, they were rejected and were not further assessed for reliability.

The quality assessment is typically done for each type of toxicity value (e.g. EC10, NOEC) for every endpoint (e.g. growth, reproduction, development) that is reported from a test. For the present assessment, where multiple types of toxicity values were reported, only the most preferred value

based on the guidance in Warne et al. (2018) was assessed. For example, EC10s are preferred over NOECs, so these were selected in preference to NOECs where they were available, and the NOECs were not assessed. Where EC10s were not available, NOECs were assessed.

Appendix D provides the data quality scores for the data from the tests that passed the relevance assessment. Based on the formal data quality assessment, all 22 toxicity values were determined to be of acceptable quality and, thus, assigned Rb1. Data quality scores ranged from 82.4% to 92.6%.

4.4.2 Expert assessment of data limitations

The acceptable quality data were further scrutinised for specific data limitations. As noted in Warne et al. (2018), professional judgment often needs to be used when assessing the reliability of data, as the formal quality assessment system may not identify or sufficiently penalise all significant flaws in an ecotoxicity test or the associated data analyses. Table 7 lists the three key data aspects that were considered for each toxicity value. The results are provided in Appendix D. Professional judgment, with appropriate justification, was used to determine the final reliability category for each toxicity value.

Ten of the 22 toxicity values were assigned Rb1 on the basis that they had no limitations or only minor limitations that were inconsequential to the final values. Seven of the toxicity values were assigned Rb2 on the basis that they had minor to moderate limitations that might have had minor implications for the analysis outcomes. The final five toxicity values were assigned Rb3 on the basis that they had significant flaws that greatly reduced the confidence in the values.

4.4.3 Reliability results

The overall reliability results are presented in Appendix D. They reflected the results of the data limitations assessment, as follows:

- 11 of the 22 toxicity values were assigned Rb1 and, thus, were considered reliable without restrictions.
- Six toxicity values were assigned Rb2 and, thus, were considered reliable with restrictions.
- Five toxicity values were assigned Rb3 and, thus, were considered not reliable.

Although the formal quality assessment scored all data highly (i.e. all >82%) and assigned them Rb1, closer scrutiny of the data and associated data analyses revealed several limitations, as summarised in Table 8. The four toxicity values assigned Rb3 (not reliable) would not be considered acceptable for use by current quality standards.

Table 8 Summary of overall reliability results for desalination plant ecotoxicity test toxicity values[a]

Reliability score	Number of tests/endpoints	Details
Rb1 – reliable without restrictions	11	• All scored >82% for the formal data quality scoring assessment and were found to have no significant limitations in terms of QA/QC, the concentration-response relationship from which they were estimated, and the statistical analysis.
Rb2 – reliable with restrictions	6	• Limitations with the nature of the dataset and/or data analysis for five of the tests, including calculation of NOECs instead of the more preferred EC10s, for four of the tests. • Lack of test acceptability criteria due to a non-standard test method for one test.
Rb3 – Not reliable	5	• No or poor concentration-response relationships were observed for all four tests; therefore, no reliable negligible effects concentrations (e.g. EC10) could be estimated.

[a] Refer to Appendix D for details of the reliability assessment results.

4.5 Results of relevance and reliability assessment

Table 9 presents the final outcomes of the relevance and reliability assessment. The outcomes can be summarised as follows:

- No data were classified relevant without restrictions / reliable without restrictions
- 11 toxicity values (for nine species) were classified relevant with restrictions /reliable without restrictions
- Six toxicity values were classified relevant with restrictions / reliable with restrictions
- Five toxicity values (for four species) were classified relevant with restrictions / not reliable.

The 17 toxicity values classified as reliable (with or without restrictions; i.e. those entries not shaded grey in Table 9) represent the best possible dataset that can be collated for the Project based on existing data and information. This dataset contained data for 12 species from eight taxonomic groups (one diatom, two macroalgae, one annelid, two echinoderms, two bivalve molluscs, one cephalopod mollusc, one crustacean, two fish). For four of the species, there was greater than one toxicity value. As noted in Section 4.3.5, the data for *S. apama* and *S. lalandi* are based on an extreme return water and receiving water salinity scenario, and their relevance for the final dataset is further considered in Section 5. The following section (Section 5) describes how the data were used to derive preliminary safe dilution factors for the Project.

Table 9 Final outcomes of the relevance and reliability assessment for the ecotoxicity data [a]

Species	Test endpoint	Type of toxicity value	Toxicity (% brine)	Relevance	Reliability	Overall status
Allorchestes compressa	14-d juvenile survival	EC10	28.8%	Rv2 (A) [b]	Rb2	Relevant with restrictions / Reliable with restrictions
Allorchestes compressa	14-d juvenile growth	EC10	28.3%	Rv2 (A)	Rb1	Relevant with restrictions / Reliable
Chrysophrys auratus	7-d larval growth (length)	EC10	13.8%	Rv2 (A)	Rb1	Relevant with restrictions / Reliable
Diopatra dentata	14-d juvenile survival	EC10	19.4%	Rv2 (B)	Rb1	Relevant with restrictions / Reliable
Diopatra dentata	14-d juvenile growth (biomass)	EC10	>50%	Rv2 (B)	Rb3	Relevant with restrictions / Not reliable
Ecklonia radiata	14-d germination	EC10	>100%	Rv2 (C)	Rb3	Relevant with restrictions / Not reliable
Ecklonia radiata	14-d growth	EC10	59.2%	Rv2 (C)	Rb1	Relevant with restrictions / Reliable
Heliocidaris erythrogramma	16-d larval metamorphosis	NOEC	12.5%	Rv2 (A)	Rb2	Relevant with restrictions / Reliable with restrictions
Heliocidaris erythrogramma	1-h fertilisation	EC10	19.6%	Rv2 (A)	Rb1	Relevant with restrictions / Reliable
Heliocidaris tuberculata	72-h larval development	NOEC	4.1%	Rv2 (A)	Rb2	Relevant with restrictions / Reliable with restrictions
Heliocidaris tuberculata	1-h fertilisation	EC10	20.6%	Rv2 (A)	Rb1	Relevant with restrictions / Reliable
Heliocidaris tuberculata	72-h larval development	EC10	13.5%	Rv2 (A)	Rb1	Relevant with restrictions / Reliable
Hormosira banksii	72-h germination	NOEC	16.5%	Rv2 (A)	Rb2	Relevant with restrictions / Reliable with restrictions
Mimachlamys asperrima	48-h larval development	EC10	12.9	Rv2 (A)	Rb1	Relevant with restrictions / Reliable
Nitzschia closterium	72-h growth rate	EC10	12.8%	Rv2 (B)	Rb1	Relevant with restrictions / Reliable
Nitzschia closterium	72-h growth rate	EC10	5.8%	Rv2 (B)	Rb1	Relevant with restrictions / Reliable
Saccostrea commercialis	48-h larval development	NOEC	8.3%	Rv2 (B)	Rb2	Relevant with restrictions / Reliable with restrictions
Sepia apama	5-month embryo/larval development	EC10	2.37%	Rv2	Rb3	Relevant with restrictions / Not reliable
Sepia apama	5-month embryo days to hatch	EC10	3.16%	Rv2	Rb3	Relevant with restrictions / Not reliable
Sepia apama	5-month post-hatch survival	EC10	1.86%	Rv2	Rb2	Relevant with restrictions / Reliable with restrictions
Seriola lalandi	7-d larval growth (Brine without Nalco additive)	EC10	4.4%	Rv2	Rb1	Relevant with restrictions / Reliable
Seriola lalandi	7-d larval growth (Brine with Nalco additive)	EC10	1.5%	Rv2	Rb3	Relevant with restrictions / Not reliable

[a] Entries shaded grey represent data determined to be not reliable and not suitable to use for derivation of preliminary safe dilutions for the Project.

[b] Letters in parentheses denote the different subcategories of relevance for the test species, as described in Table 5.

5. Derivation of preliminary safe dilutions for the Project desalination return water

5.1 Acceptable preliminary dataset

Data that are relevant and reliable without restrictions are the preferred data type for deriving guideline values and safe dilutions. In the case of the current assessment, there were no such data, and there was a need to rely on the data that were classified as relevant with restrictions and reliable without or with restrictions. These data were considered acceptable to use for two main reasons:

- Based on the Warne et al. (2018) derivation method, the "with restrictions" data would simply be classified as being of acceptable quality, and would be acceptable to use for deriving a guideline value.
- Under the European approaches for deriving environmental quality standards (the equivalent of guideline values) or predicted no effect concentrations for risk assessments, both of which formally consider data relevance and reliability, the use of "with restrictions" data is acceptable (Klimisch et al. 1997; Dr Dean Leverett, WCA Environmental UK, pers comm, 10 May 2023).

Warne et al. (2018) recommends the selection of EC10 values over NOEC values, although professional judgement can be used. For the current study, it was decided to retain and consider NOEC values for selection in the preliminary dataset because there were only four NOEC values in the dataset and they were all reasonable estimates of a negligible effect concentration and determined to be reliable.

As the dataset contained multiple toxicity values for four of the 12 species, the dataset needed to be reduced to one toxicity value per species. This was done by selecting the lowest value for a species, as recommended in Warne et al. (2018). Geometric means of multiple toxicity values for the same species, endpoint, statistical estimate and test duration were not calculated because the brine samples from the different desalination plants could not be confirmed to be identical in composition (e.g. some included antiscalants and others did not). This process resulted in the preliminary dataset shown in Table 10, comprising data for 12 species from eight taxonomic groups.

The five desalination brine ecotoxicity studies from which the relevant/reliable data were sourced (Geotechnical Services 2006, 2008c, Hydrobiology 2006, 2009, Intertek 2011) assessed brines across a range of different salinities (66 – 78 ppt), using diluent waters across a range of salinities (36 – 45 ppt). However, as noted earlier, none of the data were based on tests conducted at both the predicted return water salinity for the Project (66 – 71 ppt) as well as the known ambient seawater salinity at Mullaquana Station (38 – 41 ppt). Nevertheless, for most of the data, the brine salinity condition was within the range predicted for the Project's return water, and where the brine salinity was higher, the test diluent salinity was lower than the receiving seawater salinity at Mullaquana Station. In this latter case, the potential for higher toxicity associated with the higher salinity brine would be somewhat offset by the lower salinity of the test diluent. The exceptions to this were the data for *S. apama* and *S. lalandi*, as noted in Section 4.3.5, which were based on an extreme, high toxicity condition of high brine salinity (78 ppt) and high test diluent salinity (44 – 45 ppt). Indeed, these two species were the most sensitive and third most sensitive species in the preliminary dataset (Table 10). The retention of the data based on these extreme conditions through the relevance assessment was because the relevance assessment did not cover for the scenario where both the brine and test diluent salinity were markedly above (or below) the predicted range for the Project (i.e. potentially resulting in a marked overestimation (or underestimation) of toxicity for the proposed site). To ensure transparency in the derivation process, it was decided to retain the data for *S. apama* and *S. lalandi* when considering the acceptable preliminary dataset rather than modifying the relevance criteria to result in the exclusion of these data in the relevance assessment.

Table 10 Final relevant and reliable (with restrictions) chronic toxicity preliminary dataset for deriving preliminary safe dilutions for the Project

Species (taxonomic group)	**Species or genus locally present?**	**Locally collected?**	**Test endpoint**	**Type of toxicity value**	**Toxicity value (% brine)**
Nitzschia closterium (Diatom)	Unknown	N	72-h growth rate	EC10	5.8%
Ecklonia radiata (Brown macroalga)	N	N	72-h germination	EC10	59.2%
Hormosira banksii (Brown macroalga)	Y	N	72-h germination	NOEC	16.5%
Diopatra dentata (Annelid)	Y	N	14-d juvenile survival	EC10	19.4%
Heliocidaris erythrogramma (Echinoderm)	Y	N	16-d larval metamorphosis	EC10	12.5%
Heliocidaris tuberculata (Echinoderm)	Y	N	72-h larval development	NOEC	4.1%
Mimachlamys asperrima (Bivalve mollusc)	Y	N	48-h larval development	EC10	12.9%
Saccostrea commercialis (Bivalve mollusc)	N	N	48-h larval development	NOEC	8.3%
Allorchestes compressa (Crustacean)	Y	N	14-d juvenile growth	EC10	28.3%
Sepia apama (Cephalopod mollusc)	Y	Y	5-mnth post-hatch survival	EC10	1.86% [a]
Seriola lalandi (Fish)	Y	Y	7-d larval development/survival	EC10	4.4% [a]
Chrysophrys auratus (Fish)	Y	N	7-d larval growth (length)	EC10	13.8%

[a] The toxicity values for *S. apama* and *S. lalandi* were not included in the dataset used to derive the preliminary safe dilutions, for reasons detailed in Section 5.1. A secondary derivation is provided in Appendix F that uses the preliminary dataset inclusive of the *S. apama* and *S. lalandi* data.

Nine species, or at least their genera, in the dataset are considered to be locally present at or near the proposed site (i.e. those species assigned Rv2(A) in Table 9), with two of these species (*S. apama* and *S. lalandi*) being collected from Upper Spencer Gulf. One species (*N. closterium*) is unknown to be present in the region, while two other species (*E. radiata* and *S. commercialis*) are known to not be present at or near the proposed site.

Overall, the restrictions associated with the preliminary dataset were mostly centred around the following issues related to relevance:

- Brine salinity above the range of the projected Project return water salinity, for several species, and in particular for *S. apama* and *S. lalandi*, as discussed above.
- Control water/test diluent salinity slightly below or, in the case of *S. apama* and *S. lalandi*, well above the range of ambient seawater salinity at the proposed site.
- Most test species not locally collected, and a few test species not local to the region.

From the perspective of collating a dataset that will be most representative of the potential toxicity of return water from the proposed site, it was considered appropriate to exclude the data for *S. apama* and *S. lalandi* from the preliminary dataset used to derive the preliminary safe dilutions, for the reasons detailed above. Notably, these two species are the only test species in the dataset that were actually collected from Upper Spencer Gulf. However, it is also worth noting that the toxicity data for *S. apama* were derived from exposures to eggs and larvae – life stages that are highly unlikely to occur at the proposed site given the absence of rocky reef habitat required for *S.apama* egg attachment. This supposition/conclusion is reinforced by comprehensive marine ecological surveys conducted across all four seasons in 2025, which did not record a single occurrence of *S.apama* at the site (J Diversity 2026).

5.2 Derivation of preliminary safe dilutions

5.2.1 Preliminary safe dilutions

The SSD based on the preliminary dataset presented in Table 10 (excluding the is *S. apama* and *S. lalandi* data) is shown in Figure 1, with the associated model averaging details presented in Appendix E. Preliminary safe dilutions for both 99%, 95% and 90% species protection, derived from the SSD, are presented in Table 11. From a regulatory perspective, the minimum dilution that will be required to be achieved for the Project will be equivalent to the preliminary safe dilution for 95% species protection (Draft EIS Assessment Requirements – 26 May 2026). A secondary derivation, based on the preliminary dataset inclusive of the *S. apama* and *S. lalandi* data, is presented in Appendix F, for transparency; however, this derivation is considered to be overly conservative due to the inclusion of the *S. apama* and *S. lalandi* data based on a high toxicity scenario (i.e. high salinity brine and high salinity receiving water) that implausible for the Project at Mullaquana Station.

The fit of the SSD to the dataset was deemed by a set of three independent experts (Dr Jenny Stauber, Dr Graeme Batley, Dr John Chapman, pers comm, 2 July 2025) to be good. Based on the guideline value reliability classification in Warne et al. (2018), the preliminary dataset size (n = 10) and type (all chronic toxicity data), and the good SSD model fit (Figure 1), combined to result in the preliminary safe dilutions being assigned a *high reliability* rating.

Based on the projected Project return water salinity range of 66 to 71 ppt and ambient receiving water salinity range of 38 to 41 ppt, the 95% species protection dilution for the return water of 1 in 26 would result in a receiving water salinity range of approximately 39 to 42 ppt (i.e. 1 ppt above ambient salinity).

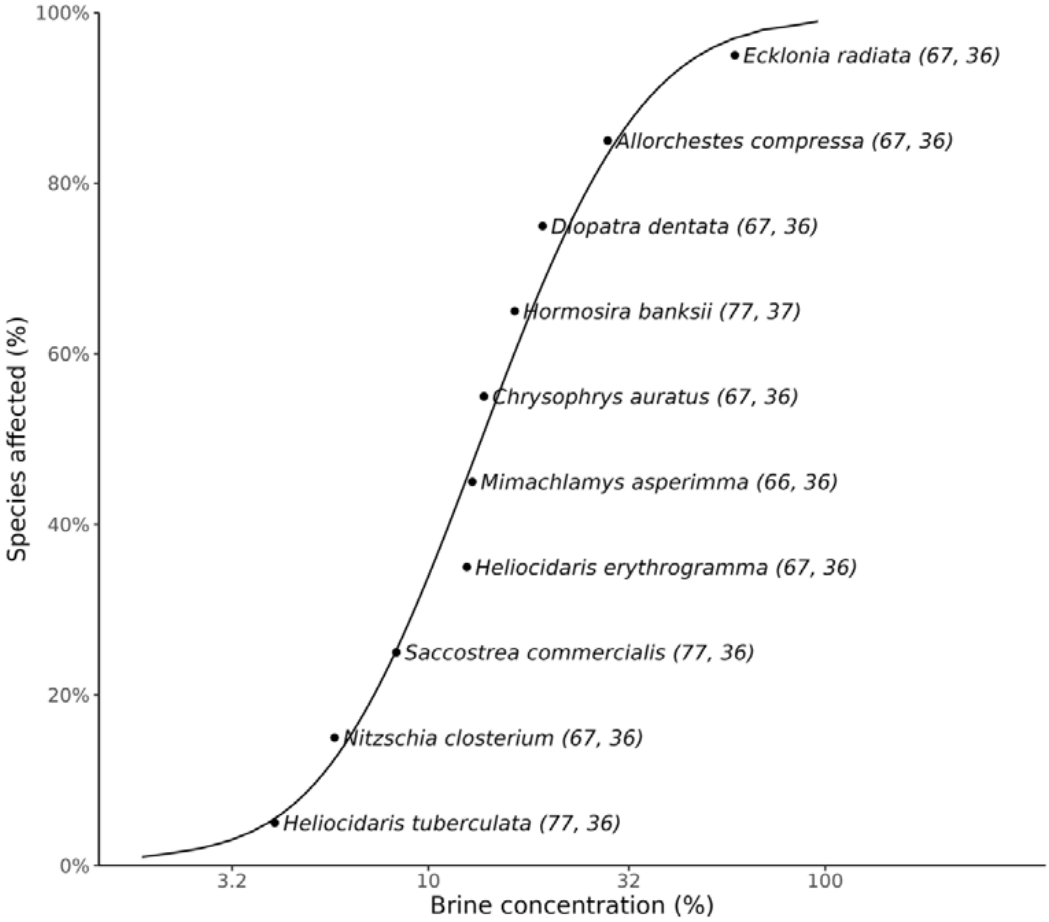


Figure 1 Model averaged species sensitivity distribution for preliminary chronic toxicity dataset

The values in parentheses for each species represent the brine salinity and test diluent salinity, respectively, used in the toxicity tests.

Table 11 Preliminary safe dilutions for desalination return water for the Project

species protection	Preliminary safe dilution	
	% return water	Dilution factor
99%	1.9	1 in 53
95%	3.9	1 in 26
90%	5.3	1 in 19

5.2.2 Comparison with safe dilutions for other desalination plant ecotoxicity studies

The preliminary safe dilutions for the Project are more conservative (i.e. higher required dilution rates) than those for most of the desalination plant ecotoxicity studies described in Section 4.2 and reported in Table 2 (see Figure 2 for comparison of the 95% species protection safe dilutions). Apart from the anomalous Cape Riche study, for which the toxicity of the desalination brine was unexplainably high (see Section 4.2.4 for details), the results are consistent with the salinities of the brines and the test diluents that were used. The preliminary dilution for the Project is higher than those from the Victorian, Adelaide and Onslow desalination plant studies because the toxicity data for the current assessment were selected to reflect higher salinity brine and receiving water conditions than were the case for the other three desalination plants. In contrast, the preliminary dilution for the Project is slightly lower than that for the ODX Point Lowly study because the receiving water and brine salinities for Point Lowly are higher than those at Mullaquana Station[1].

Hydrobiology (2013) noted that the desalination plants in Adelaide, Victoria, Sydney and Perth all had stated achievable dilution rates of ≥1 in 30, which would accommodate the 1 in 26 dilution estimated for the Project return water for 95% species protection. However, ultimately, the preliminary safe dilutions will need to be confirmed based on ecotoxicity testing of the actual Project return water. The remainder of the report focuses on remaining data gaps and associated requirements for future ecotoxicity testing.

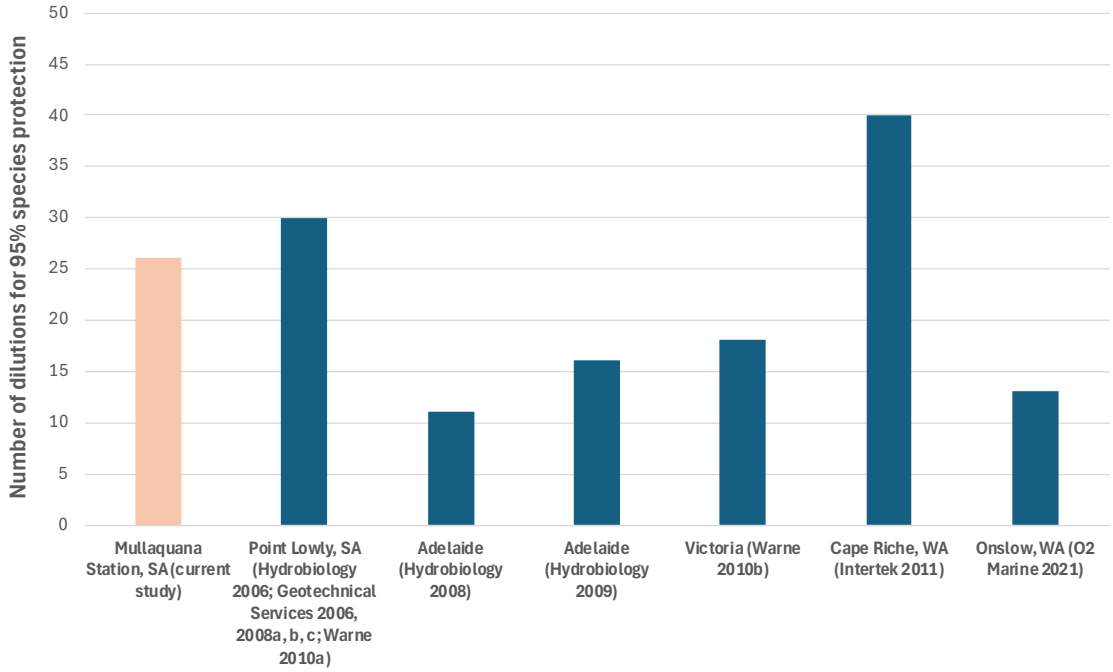


Figure 2 Comparison of 95% species protection safe dilutions from the current study (preliminary dataset) and other desalination plant ecotoxicity studies (as reported in Table 2).

[1] It is worth noting that the 95% species protection safe dilution for the ODX Point Lowly study (Warne 2010a) of 3.4% (or 1 in 30 dilution) was derived using Burrlioz 2.0. If shinyssdtools (V 0.4.0) is used (see Section 3.2 for justification of this), the 95% species protection safe dilution is 2.5% (or 1 in 40 dilution) (analysis not shown), which is probably more reflective of the higher salinity receiving water (39 – 43 ppt) and associated higher salinity brine at this location.

6. Identification of data gaps

6.1 Stakeholder considerations

Key stakeholder concerns that arose either from the assessment of the proposed Point Lowly desalination plant in Upper Spencer Gulf or early stakeholder consultation for the Project have been considered here because they can inform data gaps. Although the proposed locations for the Point Lowly desalination plant and the Project are different, they are both in Upper Spencer Gulf and, therefore, stakeholder concerns for Point Lowly were considered likely to still be relevant.

In summarising the Point Lowly ecotoxicity studies, Gillanders & Stockbridge (2022) noted several stakeholder concerns, including:

- not conducted using consistent, representative conditions for Upper Spencer Gulf
- not using the same chemicals as would be used in a desalination plant
- test species needed to be more relevant to the local area.

Morgan (2011) criticised the Point Lowly ecotoxicity assessment on the basis that there was a lack of reproductive and immunological endpoints, and an over-reliance on growth endpoints. More recently, and as part of the Project's consultation process, concerns were raised by the Spencer Gulf & West Coast Prawn Fisherman's Association about the potential impacts of desalination brines on various life stages of the western king prawn (and other species) (SGWCPFA 2023). This included potential effects of hyper-salinity on dispersal and migration of prawns.

Gillanders & Stockbridge (2022) recommended studies to better understand the long-term impacts of desalination brines, including interactions with climate change and its range of stressors (e.g. acidification, increased temperature), as well as other stressors. However, such knowledge can only be gained from pre- and post- commissioning field monitoring programs, and are separate to ecotoxicity assessments. Table 12 summarises if or how the desktop assessment of existing ecotoxicity data and derivation of preliminary safe dilutions was able to address any of the identified stakeholder concerns.

Table 12 Information on extent to which the current desktop assessment has addressed stakeholder concerns

Stakeholder concern	Addressed by data assessment and preliminary safe dilutions?	Still a gap?
The Point Lowly tests were not conducted using consistent, representative conditions for Upper Spencer Gulf	*Partially resolved.* The relevance assessment filtered the available data to retain only those that were sufficiently representative of, or at least conservative relative to, the range of conditions expected at the proposed site. However, additional testing of the actual Project return water in local receiving water during the commissioning and operational phases of the Project will be needed to validate the results of the current assessment.	Yes. See Section 6.2.1.
The Point Lowly tests did not use the same chemicals as would be used in a desalination plant	*Partially resolved.* The literature is clear that the salts in desalination brine are the major contributor to toxicity and effectively the sole contributor to toxicity for most species. Thus, details of additives are less important when considering toxicity. Nevertheless, a proper ecotoxicity assessment for a desalination plant should also consider chemical additives. While it is too early in the Project's approval process to do this, it needs to be included in any future ecotoxicity testing program.	Yes. See Section 6.2.4
The Point Lowly test species needed to be more relevant to the local area	*Partially resolved.* The current assessment was only able to rely on existing data; thus, no new species could be tested as yet. However, it is worth pointing out that seven of the 10 species represented in the preliminary dataset for the Project are local to Spencer Gulf at either the species or genus level. Moreover, there is merit in including a mix of local and non-local species in a site-specific ecotoxicity assessment, in order to provide greater assurance of data reliability (where non-local species represent well-established test species) and also to increase sample size (van Dam et al. 2019). Potentially, a more important	Yes. See Sections 6.2.2 and 6.2.3.

Stakeholder concern	Addressed by data assessment and preliminary safe dilutions?	Still a gap?
	consideration than species relevance is that all species are tested under locally relevant water quality conditions. Nevertheless, additional data for locally-collected species would benefit future studies and provide additional stakeholder assurance.	
The Point Lowly tests lacked reproductive and immunological endpoints	*Partially resolved.* The current assessment was only able to rely on existing data. Of the acceptable preliminary dataset, two values were based on tests that assessed reproduction in some form, for the macroalgae, *Ecklonia radiata* and *Hormosira banksii* (both of which assessed germination). While it would be useful to include tests assessing reproductive effects on one or more animal species, there are few such established tests for marine species, and it should be noted that other endpoints, especially those related to development of critical early life stages are also sensitive to toxicant exposure and provide an ecologically relevant indication of effects (e.g. the *Heliocidaris erythrogramma* larval metamorphosis test was more sensitive than the fertilisation test for the same species). Finally, it is unclear why immunological endpoints were highlighted, as these are still not routinely accepted as demonstrating sufficient ecological relevance to be acceptable to use for guideline value derivation purposes.	Yes. See Section 6.2.3.
Tests using the cuttlefish *S. apama* were limited due to reliability issues, and need to be repeated.	*Not resolved.* It is correct that the cuttlefish tests undertaken for the Point Lowly desalination plant ecotoxicity assessment experienced quality problems. However, the data from these tests were not used in the derivation of the preliminary safe dilutions because they were done under conditions that were not sufficiently relevant for the proposed site and/or the data were deemed to be not reliable. Importantly, there is no suitable cuttlefish breeding habitat at the proposed site and, moreover, no *S. apama* eggs nor juveniles/adults were detected during the 2025 seasonal marine surveys (J Diversity, 2026). Although future ecotoxicity studies could include this species if an appropriate sub-lethal test method can be developed, probably for juvenile life stages, there are inherent difficulties and risks in doing this that need to be taken into account, including potential animal ethics constraints; it would be a time consuming and expensive process that would not have a high probability of success, and is not recommended given there is no suitable breeding or nursery habitat at the proposed site.	Yes. See Section 6.2.2.
Consideration needs to be given to the potential impacts to western king prawns, including various life stages, and the effects of hyper-salinity on prawn dispersal and migration.	*Not resolved*. It is noteworthy that the two western king prawn ecotoxicity tests from the Point Lowly assessment were considered to be not relevant for the current study and, although the results are not presented here, the tests were also not reliable. Thus, they were excluded from the preliminary dataset. Consequently, this species is not currently represented. Future ecotoxicity studies could include this species if an appropriate sub-lethal test method can be developed, and should be considered; however, it would most likely be a time consuming and expensive process that may not have a high probability of success. Studies on long-term impacts of the desalination brine discharge on prawns, including their dispersal and migration, are beyond the scope of laboratory ecotoxicity studies, and would require long-term monitoring programs both pre- and post- commissioning of the plant.	Yes. See Section 6.2.2.
Needs to be more focus on understanding long-term impacts and multiple stressors, including interactions with climate change.	*Not resolved but not an issue for an ecotoxicity assessment*. This is a valid concern, but typically not one for which laboratory ecotoxicity studies would be employed. Laboratory ecotoxicity studies are best designed to assess either single toxicants or toxicant mixtures, such as desalination brines that also contain additives. Beyond that, specific experiments could be designed to look at multiple stressors (e.g. effect of temperature on brine toxicity), but these would be outside the scope of a standard testing program that aims to derive safe dilutions. They would need to be viewed more as research studies to improve knowledge.	Yes, but not relevant to current assessment.

6.2 Gaps in existing data

There are now many ecotoxicity data available for desalination brines, with those that were able to be obtained within the timeframe of the current assessment summarised in Section 4.2 and considered in the relevance and reliability assessment. It is generally accepted that the toxicity is predominantly caused by the salinity and much less so by any chemical additives in the brines (at least for most species most of the time). Consequently, it stands to reason that existing ecotoxicity data may be relevant to proposed new desalination plants, and that where this is the case, full ecotoxicity assessments for proposed plants may not be required. This was the approach that was adopted for the proposed EPDP, near Port Lincoln, which focused on reviewing existing ecotoxicity data and generated no new ecotoxicity data (Hydrobiology 2022). Existing gaps in data and knowledge, and their implications for future ecotoxicity testing for the Project, are considered below, with a summary of the key gaps, and whether they are considered essential or not essential to address, provided in Table 13. These gaps do not need to be filled as part of the EIS process, but should be addressed in the future, ideally during the commissioning phase of the Project when actual return water is available.

Table 13 Summary of key ecotoxicity data gaps for the Project

Aspect	Data gap	Essential / Not essential
Local salinity conditions	Ecotoxicity data that are fully representative of the expected return water salinity and receiving water salinity for the Project	Essential
Number and types of species	Reliable data for locally collected species Reliable data for taxonomic groups not already represented	Essential Not essential
Types of toxicity tests and endpoints	Reliable data for a reproduction endpoint	Not essential
Toxicity of chemical additives	Site-specific assessment of the Project return water with expected chemical additives	Essential

6.2.1 Local salinity conditions

The relevance of the existing ecotoxicity data to the expected Project return water salinity was based on a projected salinity range of 66–71 ppt that assumed a 40% overall desalination plant recovery rate from seawater with a salinity range of 38–41 ppt. Thirteen species were tested at brine salinities within the projected Project salinity range, while 10 species were tested at diluent/control water salinities within the proposed site receiving water range. However, no species were tested at salinities within both of these ranges.

Ultimately, assessment of the toxicity of brine with a salinity that matches that of the proposed Project return water would be ideal. However, given the likely conservative nature of the current assessment (see below), this could wait until the commissioning phase of the Project.

Notwithstanding the above uncertainties, the preliminary dataset has resulted in preliminary safe dilutions that are likely to be protective of the local ecosystem at Mullaquana Station, largely due to the conservative relevance criteria that were adopted for the brine salinity. However, locally-relevant validation of the preliminary safe dilutions is considered an essential requirement that should be obtained in the future prior to full operation of the Project**,** when ecotoxicity testing using return water from the Project and local receiving water as diluent water can be undertaken.

6.2.2 Number and type of species

The size of the preliminary dataset used to derive the preliminary safe dilutions (10 species from seven taxonomic groups) is considered good for deriving safe dilutions. Such sample sizes are often not achieved for site-specific assessments because it can be difficult to assess that many local species. However, as noted in Table 12, site-specific assessments need not rely solely on local species data, and there are sound reasons to include data for non-local species.

While seven of the 10 species in the preliminary dataset are considered to be locally present at the species or genus level, none were actually collected from Upper Spencer Gulf. The data for the only two locally-collected species (*S. apama* and *S. lalandi*), were not used in the preliminary dataset because the testing conditions were considered too extreme for the proposed site at Mullaquana Station (see Section 4.3.5). Therefore, the lack of relevant and reliable data for locally collected species represents a gap that is essential to fill and should be addressed in the future.

The selection of local species should ensure that the relevant data gaps are filled. Also, and ideally, local species for which established test methods already exist (either for that species or a closely related species) can be selected, as this reduces the amount of method development work that is required. However, there should be a recognition that some method development work might be required. Examples of locally/regionally present species that can readily be used for ecotoxicity testing include:

- Brown macroalga, *Hormosira banksii* (using the standard test methods for this species)
- sea urchin, *Heliocidaris erythrogramma* (using the standard *H. tuberculata* test methods)
- doughboy scallop, *Mimachlamys asperrima* (using the standard test methods for this species)
- amphipod, *Allorchestes compressa* (using the standard acute test methods for this species) – however, it is an introduced species (Dr James Brook, J Biodiversity, pers comm, 23 March 2023) for which there is only an acute test available, although a chronic growth test has been proposed by Ecotox Services Australia (Dr Rick Krassoi, Director, Ecotox Services Australia, pers comm, 8 May 2023)
- local fish species, such as yellowtail kingfish, *S. lalandi*, and snapper, *Chrysophrys auratus* (using the standard fish 7-d test methods) – although the 7-d growth test is typically not a very sensitive test nor an ideal assessment of chronic toxicity to fish, inclusion of a fish test in any suite of tests can provide additional stakeholder assurance.

Notably, most of the above-listed species are already represented in the preliminary dataset, although ecotoxicity data based on locally-collected specimens of these species would provide greater assurance of the relevance of the data. In addition to the above types of species, it may be possible to collect and isolate microalgal species from sites of interest for use in already established test methods. Overall, it would be preferable to assess some local species that are not already represented in the preliminary dataset, as discussed below.

The preliminary dataset had good taxonomic representation, covering seven taxonomic groups that included key trophic levels (e.g. primary producers, primary consumers, secondary consumers). However, data for some additional taxonomic groups not currently represented may be desirable, albeit not essential. This would include groups such as green macroalgae, seagrasses, sponges, bryozoans, ascidians and gastropods, noting that some method development would be required to establish reliable test methods. However, some degree of method development has already been undertaken for some species (or closely related species) from most of these taxonomic groups that are known to be present in Upper Spencer Gulf (Associate Professor Craig Styan, University of South Australia, pers comm, May 2025; Dr James Brook, J Biodiversity, pers comm, August 2025). These

include *Ulva* spp. (green macroalgae), *Bugula* spp. (bryozoans), *Ciona* spp. and *Phallusia obesa* (ascidians) and *Nassarius pauperatus* (gastropod). Some of these species are further considered in Section 7.2.2.

As noted in Table 12, the Australian giant cuttlefish and western king prawn data from the Point Lowly study (Geotechnical Services 2008a) were excluded from the preliminary dataset. Both these species have been of particular interest to stakeholders. Any consideration to develop/adapt appropriate sub-lethal ecotoxicity test methods for these two species would represent significant research and development efforts that would not be guaranteed to succeed. The difficulties in developing and performing reliable and reproducible ecotoxicity test methods for these species was evidenced by the test and data quality problems experienced during the Point Lowly ecotoxicity studies. Further challenges associated with these species are summarised below:

- *Australian giant cuttlefish* – As there is no suitable breeding or nursery habitat at the site, ecotoxicity data based on cuttlefish embryos and larvae can be considered to have limited ecological relevance. Therefore, test development would need to focus on juvenile or adult stages. Testing juvenile or adult life stages in the laboratory would be logistically challenging due to the large volumes of seawater required and the need to maintain good water quality conditions over long durations of exposure (i.e. weeks) that would be required to assess chronic toxicity. Moreover, there may be animal ethics constraints or even general ethical constraints associated with exposing juvenile or adult *S. apama* to potentially harmful chemicals.
- *Western king prawn* – Most of the concerns raised by the prawn fisherman's association cannot be addressed by ecotoxicity studies, but will require long-term monitoring programs pre- and post-commissioning of the Project. Given that data for a crustacean (*A. compressa*) are already included in the preliminary dataset, the inclusion of an ecotoxicity test for the western king prawn is not required to fill an ecological gap.

As a result of the various challenges, efforts to develop/adapt appropriate sub-lethal ecotoxicity test methods for these two species are not currently recommended. In contrast, there may be merit in considering additional development of existing test methods for some of the earlier-mentioned species from the unrepresented taxonomic groups.

6.2.3 Types of ecotoxicity tests and endpoints

Typically, chronic ecotoxicity tests have been employed for desalination brines on the basis that most brine discharges are continuous when in operation. There have been some instances of acute tests being justified. For example, an ecotoxicity study assessed the acute toxicity of brine from the Gold Coast desalination plant specifically to assess short-term pulses associated with certain release scenarios (as summarised in Section 4.2.7 and Hydrobiology 2022). Also, modifications have been made in the past to the 21-d reproduction test for the copepod, *Gladioferens imparipes*, to involve a short 24-h or 48-h pulse exposure followed by assessment of reproduction for the rest of the test duration. This was based on a rationale that zooplankton that drift with the currents would only ever be exposed to desalination brine for short periods of time (Geotechnical Services 2008a). On the basis that (i) chronic ecotoxicity tests will result in more conservative estimates than acute toxicity, and (ii) it would be difficult to confidently characterise exposure profiles for all species, it is recommended that chronic toxicity studies are the focus of future ecotoxicity studies unless specific discharge scenarios arise for which acute testing is more relevant.

As noted in Table 12, there is a lack of reproductive endpoints for aquatic animals associated with the preliminary dataset. Apart from the need to focus on sub-lethal endpoints, there is no minimum requirement for the types of endpoints to include in an ecotoxicity study. Consequently, the inclusion

of a reproductive endpoint is not essential. Marine ecotoxicity reproduction tests are very uncommon and there are very few established tests in Australia. One such test is the copepod (*G. imparipes*) 21-d reproduction test; however, this test exhibits high variability and a high failure rate and is, therefore, unreliable (Dr Tristan Stringer, Principal Ecotoxicologist, Intertek, pers comm, 24 March 2023). Consequently, this test is no longer used by Intertek (formerly Geotechnical Services) and has been replaced by a more reliable 5-7-d naupliar development test. Therefore, it is not recommended to pursue the *G. imparipes* reproduction test. It should also be noted that existing echinoderm and bivalve fertilisation tests and macroalgal germination tests provide an indication of effects on aquatic animal and plant reproductive success. In the case of the echinoderm and bivalve fertilisation tests, toxicity values were not selected for the preliminary dataset because larval development endpoints for the relevant species were more sensitive.

A possible alternative to the copepod reproduction test is a 10-d amphipod (*Melita plumulosa*) reproduction test that has been established (Mann et al. 2010, Spadaro & Simpson 2016) and is offered by Ecotox Services Australasia (ESA). This species is epibenthic, hence, is relevant for water column ecotoxicity testing. The test duration would need to be extended to 14 d to be considered a chronic test for this species; however, this should not represent a significant challenge (Dr Rick Krassoi, Director, ESA, 8 May 2023). This species, and other members of the family Melitidae, are known to occur in Upper Spencer Gulf (Dr James Brook, J Biodiversity, pers comm, October 2025).

Stakeholders also raised the issue of differential sensitivity of different life stages to desalination brine (Gillanders & Stockbridge 2022, SGWCPFA 2023). This would be expected. The general rule in ecotoxicology is that early life stages tend to be more sensitive to toxicant exposure than juvenile or adult life stages, as the various physiological mechanisms for coping with toxicant exposure are less well developed. Moreover, the surface area to volume ratio of early life stages is often higher than for older life stages, thus, increasing their direct exposure to water-borne toxicants. Results from Geotechnical Services (2008a) that indicated that adult western king prawns were more sensitive to desalination brine than juveniles (in terms of growth), and which were the primary source of the stakeholder concerns, should be treated with caution. Both tests had significant limitations and, on closer inspection, the results were deemed to be not reliable. Consequently, the typical approach of targeting early life stages is still considered appropriate for the purposes of ecotoxicity assessments.

Ultimately, for any future ecotoxicity testing, a range of endpoints should be assessed. It should be noted that existing fertilisation tests for echinoderms and bivalves provide an indication of reproductive success, while the range of larval development tests that also exist are known to be sensitive to a wide range of toxicants, including desalination brines. Therefore, there are several additional useful sub-lethal endpoints that can be employed in addition to growth as an endpoint.

6.2.4 Toxicity of chemical additives

It was outside the scope of the current assessment to undertake a detailed desktop assessment of the ecotoxicity of chemical additives in desalination brines. However, as demonstrated in Section 4.2.8, previous studies have consistently shown that the high salt concentrations in desalination brines are the predominant contributor to toxicity, and additives typically only contribute a small amount, if anything, to the toxicity. Nonetheless, some studies have observed significant toxicity associated with additives, so it should not be ignored. For example, Hydrobiology (2006) demonstrated that the antiscalant, Nalco PC1020T, was contributing to brine toxicity to the diatom, *N. closterium*.

Through engagement between the client and the EPA, the client advised that it is their understanding that the approved chemicals list in the ADP environment protection licence should inform selection and use of chemical additives for the proposed Mullaquana Station desalination plant. The current reference design for the Mullaquana Station plant (GHD 2025) includes the identification of chemicals and dose rates, for example:

- Antiscalant (type not specified) – 4 mg/L maximum dosing concentration, 2 – 7 mg/L return water concentration
- Chlorine – 3 mg/L maximum dosing concentration
- Sodium bisulfite – 25 mg/L maximum dosing concentration

The EPA Licence for the ADP lists Flocon 190, Flocon 285, Flocon 295, Genesys MP, Nalco PC1020T and SpectraGuard as approved antiscalants (SA EPA 2020). It is understood that the ADP is currently using Genesys MP, with a concentration in the discharge of approximately 2.5 mg/L. If the antiscalants previously tested as part of the ADP ecotoxicity studies (Hydrobiology 2008, 2009) are used for the proposed Mullaquana Station plant at or below the concentrations used in the ADP ecotoxicity studies (see Section 4.2.8), this should pose a very low risk of toxicity due to antiscalants at the preliminary safe dilution rate of 1 in 26 for 95% species protection (see Table 11). Notably, the toxicity of Genesys MP was not assessed in any of the ADP ecotoxicity reports accessed during the current review. Therefore, if this antiscalant was to be considered for use for the Project, its toxicity would also need to be assessed as part of a commissioning phase ecotoxicity assessment.

Notwithstanding the considerable evidence from the ADP ecotoxicity assessments and elsewhere on chemical additive toxicity, confirmation of the contribution of chemical additives to brine toxicity is considered to be an essential requirement. Therefore, the toxicity of the Project's return water with its typical additive constituents should be assessed. This is best done during the commissioning and operational phases when the additives and their typical dosing rates are known.

7. Recommendations for future ecotoxicity testing programs

7.1 Minimum requirements for site-specific ecotoxicity studies

There is little prescriptive instruction on minimum requirements for site-specific ecotoxicity assessments, as such requirements tend to be issue-specific. National guidance on the use of DTA has recently been updated (ANZG 2023). Below are some aspects covered in the guidance that are of relevance to the ecotoxicology requirements for the Project:

- It refers to the Warne et al. (2018) guideline value derivation method that requires toxicity data for eight or more species from four or more taxonomic groups to be able to derive 'high reliability' guideline values. Notably, the preliminary safe dilutions reported in Section 5.2.1 are based on data for 10 species, thus, meeting this requirement.
- Although data for as few as five species can be used to derive guideline values (or safe dilutions), this will only result in "low reliability" guideline values (Warne et al. 2018). Notably, the use of only five species for such purposes is not considered best practice nowadays.
- It recommends the use of local species, but does not specify how many.
- It allows use of regionally-relevant species (i.e. that are not collected from the site) to supplement local species.
- It recognises use of DTA for commissioning and post-commissioning purposes to validate previously derived safe dilutions, but is not prescriptive on how this should be done apart from noting that some jurisdictions might require periodic early operational phase testing for this purpose.

Thus, determination of ecotoxicity testing requirements for site-specific issues requires expert judgement based on the specifics of the issue, the objectives and the available evidence. Comparison of the results of the gap analysis (Section 6) with the above guidance indicates that some of the requirements for site-specific ecotoxicity assessment have been met (e.g. size of the preliminary

ecotoxicity dataset, use of locally-relevant species), but there are other aspects that are still required (e.g. validation of preliminary safe dilutions, more locally-collected test species). This is addressed in the following section.

7.2 Future ecotoxicity testing requirements

Although a preliminary dataset was compiled and preliminary safe dilutions were derived, the details provided in Sections 6 and 7.1 have demonstrated that there is still a need for future ecotoxicity testing for the Project. The size of the testing program proposed below is considered to be commensurate with the size of the Project relative to previous desalination plant ecotoxicity assessments. However, direct comparisons with other plants are difficult to make. Most other testing programs have involved an upfront full DTA of a simulated brine for the EIS, comprising at least six species, with the associated derivation of safe dilutions, followed by some type of commissioning phase and even post-commissioning phase ecotoxicity testing. The approach for the Project is slightly different, relying instead on a large body of relevant and reliable existing data for the EIS, followed by targeted pre-commissioning phase and operational phase ecotoxicity programs. This approach is considered valid given the existing and large body of knowledge on desalination brine ecotoxicity. Details are provided below.

7.2.1 Ecotoxicity studies for the Project EIS

As already noted, the preliminary safe dilutions derived for the Project are based on an good dataset of relevant and reliable (with restrictions) data from other desalination plant ecotoxicity studies, and which exceeds minimum requirements according to national guidance (Warne et al. 2018, ANZG 2018, ANZG 2023). Notwithstanding the data limitations, the preliminary safe dilutions are considered likely to be protective of the local ecosystem at Mullaquana Station. Consequently, it is recommended that no further ecotoxicity testing is required as part of the EIS to further inform the preliminary safe dilutions.

7.2.2 Pre-commissioning phase ecotoxicity study

The appropriateness of the preliminary safe dilutions for the Project's return water discharge should be validated via an ecotoxicity testing program prior to the commissioning phase of the Project. Details of what this program should entail are listed below, and are designed to be readily adaptable into a scope of works for ecotoxicity testing. The focus is on safe dilution validation and filling the gaps that were previously identified. Some details (e.g. timing, nature of samples to be tested) cannot be fully resolved here, and will need to be determined later between the proponent and regulator.

Objective of the testing program

- to validate and enable an update of the preliminary safe dilutions for the Project's return water using newly-acquired, locally-relevant ecotoxicity data.

Timing

- testing should be carried out prior to commissioning of the Project's plant (i.e. prior to any return water being discharged to the marine environment). The testing will need to be carried out using a relevant simulated return water sample produced using receiving water from the proposed site.

Samples to be tested

- testing needs to be undertaken using relevant samples of simulated return water for the Project as well as actual receiving water as control water/test diluent. The salinity of the simulated return water will need to be the same as that expected to be produced by the Project's plant.

- the program should include testing of several simulated return water samples with and without chemical additives, based on a number of relevant discharge scenarios (e.g. as done for the Victorian and other desalination plants – see Section 4.2).
- Sufficient simulated return water sample concentrations should be tested to enable estimation of EC10 (or appropriate alternative estimate of a negligible effect concentration) and EC50 values (i.e. concentration resulting in a 10% and 50% effect relative to the control response, respectively). This would typically be at least eight concentrations including the control treatment (e.g. 0% (control), 1.6%, 3.2%, 6.3%, 12.5%, 25%, 50% and 100% return water).

Species, endpoints and test types

- testing should be undertaken using a mix of species including (i) those that have established test methods and are already represented in the preliminary dataset used to derive the preliminary safe dilutions, and (ii) local species not currently included in the dataset. A candidate list of species/tests is provided in Table 14. It is acknowledged that there needs to be some flexibility in the selection of the new candidate species to accommodate issues that could arise during method development (e.g. inability to collect stock, husbandry/culture limitations, new information on local presence of species, etc.). Therefore, the candidate list is a starting point and does not preclude other species from being proposed, but the overall selection would need to represent more than one taxonomic group.
- From the species listed in Table 14 (or proposed alternative new candidate species), it is recommended that six species from at least four taxonomic groups are tested for each sample, as described below:
    - Three of the five standard species that are represented in the preliminary dataset.
    - Three of the proposed (or alternative) new candidate species/tests.
- The use of several standard test species that are locally present and were included in the preliminary safe dilution dataset is predominantly for validation purposes. It provides relevant results while also allowing a comparison with previous desalination brine ecotoxicity data for these species. Moreover, as these tests are well-established with well-known performance, they offer an additional degree of rigour and reliability to the overall program.
- The use of several locally present, locally collected species addresses the identified gap on local species knowledge and helps provide assurance that the local ecosystem will be protected. Selection of the new species should also aim to fill the gaps related to unrepresented taxonomic groups and lack of animal reproduction endpoints, although they are not considered to be essential requirements.

Use of the data

- The ecotoxicity data from the pre-commissioning phase ecotoxicity testing will be used for the following purposes:
    (a) The new ecotoxicity data will be compared with the existing data that were identified by the relevance and reliability assessment to assess if toxicity is the same or different.
    (b) The new ecotoxicity data will be added to the preliminary dataset identified by the relevance and reliability assessment in order to update the proposed safe dilutions.
- If the toxicity of the simulated return water is significantly higher than the data used to derive the preliminary safe dilutions, testing of additional species listed in Table 14 may be required in order to derive a new set of safe dilutions. If this is required, it is recommended that an additional four species be tested.

Table 14 Summary of species proposed to be used for the pre-commissioning phase ecotoxicity testing of simulated return water

Species (taxonomic group)	Endpoint	Existing / New species?	Species/genus locally present?	Locally collected?	Justification	Commercial ecotoxicity testing providers
Standard species already represented in the preliminary dataset						
Nitzschia closterium (Diatom)	72-h growth rate	Existing	Y	N	Established commercial test species that is sensitive, and has also been found to be sensitive to antiscalant. Provides a benchmark with which to compare with existing data for this species.	ESA, Intertek
Hormosira banksii (Macroalga)	72-h germination	Existing	Y	Y	Established commercial test species that exhibits mid-range sensitivity. Provides a benchmark with which to compare with existing data for this species. Can be locally collected with careful adherence to collection instructions.	ESA, Intertek
Mimachlamys asperrima (Bivalve mollusc)	48-h larval development	Existing	Y	Y	Established commercial test species that is sensitive. Provides a benchmark with which to compare with existing data for this species. Can be locally collected with careful adherence to collection instructions.	ESA
Heliocidaris tuberculata (Echinoderm)	72-h larval development	Existing	Y	N	Established commercial test species that is sensitive. Provides a benchmark with which to compare with existing data for this species and also for the locally present *Heliocidaris erythrogramma*.	ESA
Chrysophrys auratus (fish)	7-d larval growth	Existing	Y	N	Established commercial test that can be readily adapted to this species (snapper). Not a very sensitive test, but can compare with existing data for this species. Inclusion of this popular recreational and commercial fish species would provide additional stakeholder assurance.	ESA
New candidate species [a, b]						
Ulva sp. (Green macroalga)	72-h germination	New	Y	Y	Locally present. Existing test methods available (albeit not commercial methods) that could be adapted for local population of the same species (Han et al. 2009).	Nil
Bugula neritina (Bryozoan)	48-h or 72-h larval development	New	Y	Y	Locally present, albeit introduced; *Bugula stolonifera* is a locally present native species that could be used as an alternative option Existing test methods available (albeit not commercial methods) (Tania & Keough 2003, Bennett 2006).	Nil

Species (taxonomic group)	Endpoint	Existing / New species?	Species/genus locally present?	Locally collected?	Justification	Commercial ecotoxicity testing providers
Ciona sp. (Ascidian)	48-h or 72-h larval development	New	Y	Y	Locally present, albeit introduced; *Phallusia obesa* is a locally present native species that could be used as an alternative option Existing test methods available (albeit not commercial methods and also not published).	Nil
Nassarius pauperatus (Gastropod mollusc)	7-d to 14-d larval growth	New	Y	Y	Locally present and easy to spawn (but may be seasonal). Existing standard method available for chronic endpoint (Trenfield et al. 2016), planned to be adopted commercially for a temperate species by Intertek. Option to use different species depending on seasonality.	Planned by Intertek
Galeolaria caespitosa (Annelid)	48-h to 96-h larval development	New	Y	Y	Locally present. Not an unrepresented taxonomic group but included because fertile throughout the year and easy to spawn. *Galeolaria hysterix* could be used as an alternative option. Existing test methods available (albeit not commercial methods) (Ross & Bidwell 2001, Lockyer et al. 2019)	Nil
Melita plumulosa (Crustacean)	14-d reproduction	New	Y	Y	Locally present, epibenthic species; therefore, might have additional relevance in the event high salinity plumes form at the seabed. Offers aquatic animal reproduction endpoint that is currently missing from the preliminary safe dilution dataset. 10-d method already developed (Mann et al. 2009, Spadaro & Simpson 2016) and offered commercially, and can be extended to 14 d to represent chronic toxicity. This test would represent a useful reproduction endpoint to include.	ESA
Gladioferens imparipes (Crustacean)	7-d naupliar development	Existing	Uncertain	To be advised	At present, the main candidate species is *G. imparipes*; however, it may not be locally present. A commercial method exists for this species. Represents a true zooplankton.	Intertek

[a] Candidate species list compiled following consultation with: Associate Professor Craig Styan, University of South Australia; Dr Tristan Stringer, Principal Ecotoxicologist, Intertek; Dr Rick Krassoi, Director, Ecotox Services Australia; and Dr James Brook, J Diversity Pty Ltd).

[b] There may need to be some flexibility in the selection of the new candidate species to accommodate issues that could arise during method development (e.g. inability to collect stock, husbandry/culture limitations, new information on local presence of species, etc.). Therefore, other species could be proposed but the overall selection would need to represent more than one taxonomic group.

7.2.3 Operational phase ecotoxicity study

An operational phase ecotoxicity testing program should be undertaken to confirm that the Project's return water toxicity is not different to what was expected based on previous testing, and that appropriate protection is being achieved. Details of what this program should entail are listed below, and are designed to be readily adaptable into a scope of works for ecotoxicity testing. As with the pre-commissioning phase program, some details would need to be finalised later, between the proponent and regulator.

Objective of the testing program

- to confirm that toxicity of the return water discharge from the operational plant is as expected based on previous ecotoxicity testing, and that the updated safe dilutions are still appropriate.

Timing

- testing should be carried out periodically over the first year following the commencement of discharge (e.g. on four occasions approximately three months apart or as plant production is increased) when the desalination plant is operating under normal conditions and discharging to the environment.
- Any ecotoxicity testing beyond the above specification would most likely only be required if the salinity and/or composition of the return water markedly changes, and should be treated as a separate program of works to the operational phase ecotoxicity testing program described here. The current report has not considered this aspect any further.

Samples to be tested

- Each testing round needs to be undertaken using a relevant sample of return water from the Project under normal operating conditions, as well as actual receiving water as control water/test diluent.
- Sufficient return water sample concentrations should be tested to enable estimation of EC10 (or appropriate alternative estimate of a negligible effect concentration) and EC50 values. This would typically be at least eight concentrations including the control treatment (e.g. 0% (control), 1.6%, 3.2%, 6.3%, 12.5%, 25%, 50% and 100% return water).

Species, endpoints and test types

- The first round of testing should be undertaken using the same six species that were used for the pre-commissioning phase testing program.
- The remaining three rounds of testing should be undertaken using the three most sensitive species from the first round of testing. However, at least one of the tests should be for one of the existing standard (locally present) species listed in Table 14, as this will provide additional confidence in the reliability of the data.

Use of the data

- The ecotoxicity data from the operational phase ecotoxicity testing will be used for the following purposes:

    (a) The new ecotoxicity data will be compared with the ecotoxicity data for the same species from the pre-commissioning phase testing program to determine if toxicity is different.

(b) The new ecotoxicity data will be added to the updated ecotoxicity dataset from the pre-commissioning phase testing program to determine if it changes the safe dilutions.

- If the toxicity of the return water is significantly higher than the simulated return water tested during the pre-commissioning phase, testing of additional species listed in Table 14 may be required in order to derive a new set of safe dilutions. If this is required, it is recommended that an additional four species (to the six species tested in the first round of operational phase testing) be tested.

8. Summary and recommendations

Key conclusions associated with the ecotoxicology review and gap analysis are summarised below.

Assessment of relevance and reliability of ecotoxicity data from other desalination plant ecotoxicity studies

- Data from ecotoxicity studies for five Australian desalination plants were reviewed: Point Lowly, Adelaide, Victoria, Cape Riche and Onslow.
- Most of the ecotoxicity data found to be relevant to the current assessment were from the Adelaide desalination plant or the proposed Point Lowly desalination plant ecotoxicity studies.
- Twenty two chronic toxicity values for 12 species were found to be relevant to the current assessment, albeit they all had minor limitations.
- The main limitations of the relevant data were:
    - Nine of the toxicity values were based on brines with salinity 6-7 ppt higher than the projected Project return water salinity (upper range of 71 ppt)
    - All of the toxicity values were based on control water/test diluent with salinity outside the range of ambient receiving water salinity at the proposed site (38 – 41 ppt).
    - Only two of the 12 test species were locally collected from the Upper Spencer Gulf region, although nine of them are locally present at the species or genus level.
- Of the 22 relevant toxicity data, 17 toxicity values were determined to be reliable (with or without restrictions) and five toxicity values were determined to be not reliable. The reliable dataset comprised data for 12 species from eight taxonomic groups.

Derivation of preliminary safe dilutions for the Project

- A preliminary ecotoxicity dataset was compiled for the Project that comprised a single chronic toxicity value for 10 species from seven taxonomic groups. Toxicity values for the remaining two species, *S. apama* (Australian Giant Cuttlefish) and *S. lalandi* (yellowtail kingfish) were excluded from the preliminary dataset on the basis that the brine and control water/test diluent salinity conditions under which they were tested were extreme and implausible for the proposed site based on existing salinity monitoring data.
- The preliminary dataset was used to derive preliminary safe dilutions of 1.9% (1 in 53), 3.9% (1 in 26) and 5.3% (1 in 19) return water for protection of 99%, 95% and 90% of species, respectively.
- The preliminary safe dilutions are likely to be conservative (i.e. protective) for the proposed site.

Key data gaps

The data gaps identified below are considered to be important but not urgent, and it is recommended that they form part of investigations beyond the EIS phase.

- *Local salinity conditions*
    - Despite the preliminary dataset being determined to be relevant for the Project, the data did have limitations, and they did not fully align with the expected return water salinity and ambient receiving water salinity for the Project. Thus, data more relevant to these salinity conditions are required, .
- *Types of species*
    - Although many of the species represented in the preliminary dataset are thought to be locally present at the species of genus level, none were locally collected. More data for locally present and collected species are required, while data for unrepresented taxonomic groups (e.g. green macroalgae, bryozoans, ascidians, gastropods) are desirable.
- *Types of ecotoxicity tests and endpoints*
    - The preliminary dataset is missing reliable reproductive endpoint data for an aquatic animal. While this is not essential, it would be desirable for future ecotoxicity testing to fill this gap.
- *Toxicity of chemical additives*
    - The preliminary safe dilutions are based on ecotoxicity data for brine only or brine with only antiscalant. Future ecotoxicity testing needs to also capture any toxicity associated with other chemical additives, such as CIP chemicals, biocides, etc. However, it is worth noting that most previous studies have found that the toxicity of desalination brines is driven by the high salt concentrations, and chemical additives contribute very little to the toxicity for most species.

Future ecotoxicity testing program for the Project

- *Ecotoxicity testing for the EIS*
    - The preliminary safe dilutions are considered likely to be conservative and protective of the receiving marine environment and are appropriate for the purposes of the EIS. Consequently, no further ecotoxicity testing is recommended for the EIS.
- *Pre-commissioning phase ecotoxicity testing program*
    - The appropriateness of the preliminary safe dilutions should be validated via an ecotoxicity testing program during the pre-commissioning phase of the Project.
    - A full DTA of the Project's return water is not required; rather, tests for a selection of approximately six species should be used to confirm and enable an update of the preliminary safe dilutions.
    - Testing should be carried out on a range of simulated return water samples with and without additives, based on a number of relevant discharge scenarios.
    - Testing should be undertaken using a mix of species including (i) three that have established test methods and are already represented in the preliminary dataset used to derive the preliminary safe dilutions, and (ii) three local species not currently included in the preliminary dataset. The final selection of species should aim to fill the key data gaps (e.g. unrepresented taxonomic groups).

- *Operational phase ecotoxicity testing program*
    - An operational phase ecotoxicity testing program should be undertaken to confirm that the toxicity of the Project's return water during normal operations is not different to what was expected based on previous testing, and that appropriate protection is being achieved.
    - Testing should be carried out periodically (e.g. every three months for a year or as plant production is increasing) on a sample of return water representative of normal operating conditions.
    - The first round of testing should be undertaken using the same six species that were used for the pre-commissioning phase testing program.
    - Subsequent rounds of testing should be undertaken using the three most sensitive species from the first round of testing, but ensuring that at least one of the tests is for one of the existing standard (locally present) species.

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Appendices

Appendix A Assessment of relationships between desalination brine salinity or diluent/control water salinity and brine toxicity.

As noted in Warne (2010a), the toxicity of desalination brines is largely a function of (i) the concentrated salinity of the brine and (ii) the natural salinity of the ambient receiving water (excluding any toxicity potentially caused by chemical additives). Both sources of salts contribute to increase the salinity to a point that is toxic to an organism. Given that desalination brine salinity is often around twice the salinity of the ambient intake/receiving water, higher intake/receiving water salinity will result in higher brine salinity. Therefore, it would be expected that the combination of higher salinity brine and ambient intake/receiving water should result in higher toxicity compared to lower salinity brine and ambient intake/receiving water. While it could be argued that species are adapted to the salinity of their local environment, species living in higher salinity waters will be subjected to higher salinity desalination concentrates, and are also likely to be living closer to their upper salinity tolerance threshold, than species living in lower salinity waters.

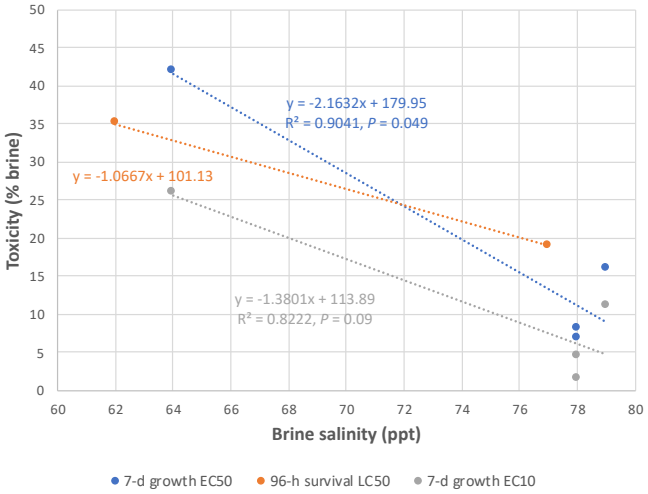
A key question that needs to be addressed by the relevance assessment is, which of the existing desalination brine ecotoxicity studies (or specifically, the individual ecotoxicity tests within the studies) are sufficiently relevant to the return water salinity and receiving water salinity expected for the Project? This largely depends on the extent to which the toxicity of desalination brine differs with different brine salinity and/or control water/test diluent salinity. In reviewing the literature on desalination brine ecotoxicity, no studies were found that had directly investigated the relationship between brine salinity or ambient intake/receiving water salinity and brine toxicity. Such knowledge could help inform the relevance of the substantial amount of ecotoxicity data from previous studies on desalination brines to the Project.

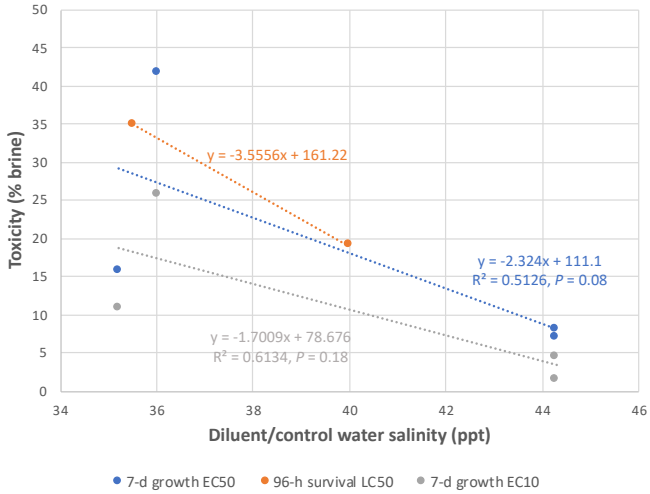
In an attempt to inform this knowledge gap, existing ecotoxicity data based on desalination concentrates and diluent/control water salinities from studies for five proposed or existing desalination plants in Australia (Table 2) were compared. This comparison required an overlap of species and test methods with the same endpoints across at least three studies (i.e. n≥3) for each species that were conducted at a range of diluent/control water salinities. This was the case for five species, as shown in Table A.1. The compiled data represent toxicity data for desalination brine with antiscalant added. Although the type and concentration of antiscalant varied, no attempt was made to correct for this given that several desalination brine ecotoxicity studies have demonstrated that, for most species, the toxicity is predominantly driven by the salinity (e.g. Geotechnical Services 2008c, Hydrobiology 2006, 2008, Warne 2010a). The plots in Figure A.1 are derived from the data in Table A.1 and show relationships between brine toxicity and brine salinity or control water/test diluent salinity for four of the species. The data for the microalga *Isochrysis galbana* were not included because this species did not show sufficient sensitivity to desalination brine to enable an effective assessment. Where possible, toxicity was plotted based on both EC50 and EC10 ecotoxicity data (as well as acute LC50 data for the yellowtail kingfish *S. lalandi*), noting that EC50 data are typically better for characterising and comparing toxicity. Ideally, a multiple regression approach that assessed the combined influence of brine salinity and ambient intake/receiving water salinity on brine toxicity would have been performed; however, the sample sizes for each of the species were too small to enable this.

Table A.1 Summary of desalination brine ecotoxicity data for species tested across multiple control water/test diluent salinities (from different desalination plant ecotoxicity studies).

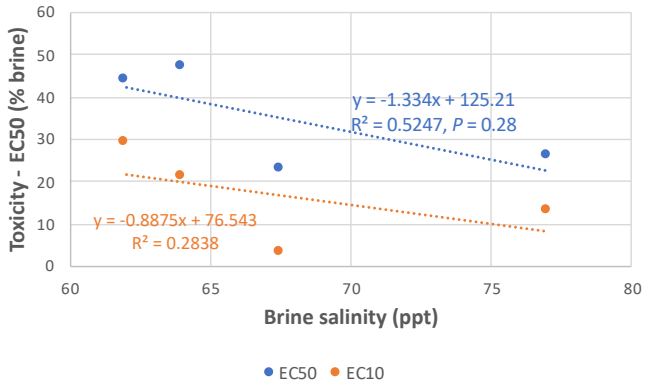
Species	Desal plant	Endpoint	Control water/test diluent salinity (ppt)	Brine salinity (ppt)	Toxicity (% brine)		Reference
					EC10	EC50	
Ecklonia radiata	Adelaide	72-h germination	35.5	62	54	100	Hydrobiology (2008)
Ecklonia radiata	Point Lowly	72-h germination	39.9	79	28	59	Geotechnical Services (2008a)
Ecklonia radiata	Cape Riche, WA	72-h germination	35.3	68	4.4	20	Intertek/Geotechnical Services (2011)
Ecklonia radiata	Perth (SSDP)	72-h germination	36	64	59	87	O2 Marine (2021)
Heliocidaris tuberculata	Point Lowly	72-h larval development	36.3	77	–	12	Hydrobiology (2006)
Heliocidaris tuberculata	Adelaide	72-h larval development	35.5	62	13	18	Hydrobiology (2008)
Heliocidaris tuberculata	Victoria	72-h larval development	35	67.5	–	17.4	Warne (2010)
Isochrysis galbana	Point Lowly	72-h cell yield	39.9	79	–	84	Geotechnical Services (2008a)
Isochrysis galbana	Cape Riche, WA	72-h cell yield	35.3	68	–	100	Intertek/Geotechnical Services (2011)
Isochrysis galbana	Perth (SSDP)	72-h cell yield	36	64	46	100	O2 Marine (2021)
Nitzschia closterium	Point Lowly	72-h cell yield	37	77	13	26	Hydrobiology (2006)
Nitzschia closterium	Adelaide	72-h cell yield	35.5	62	29	44	Hydrobiology (2008)
Nitzschia closterium	Victoria	72-h cell yield	35	67.5	3.1	23	Warne (2010)
Nitzschia closterium	Perth (SSDP)	72-h cell yield	36	64	21	47	O2 Marine (2021)
Seriola lalandi	Adelaide	96-h larval imbalance	35.5	62	26	35	Hydrobiology (2008)
Seriola lalandi	Point Lowly	96-h survival	40	77	–	19	Hydrobiology (2006)
Seriola lalandi	Point Lowly	7-d larval growth	35.2	79	11	16	Geotechnical Services (2008a)
Seriola lalandi	Point Lowly	7-d larval growth	44.3	78	1.5	6.9	Geotechnical Services (2008c)
Seriola lalandi	Point Lowly	7-d larval growth	44.3	78	4.4	8.1	Geotechnical Services (2008c)
Seriola lalandi	Perth (SSDP)	7-d larval development	36	64	26	42	O2 Marine (2021)

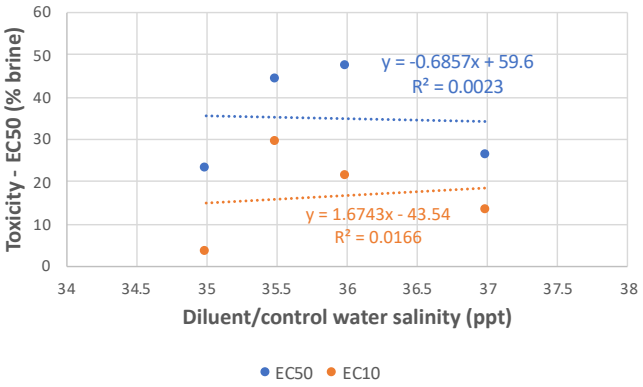
A. *Seriola lalandi*



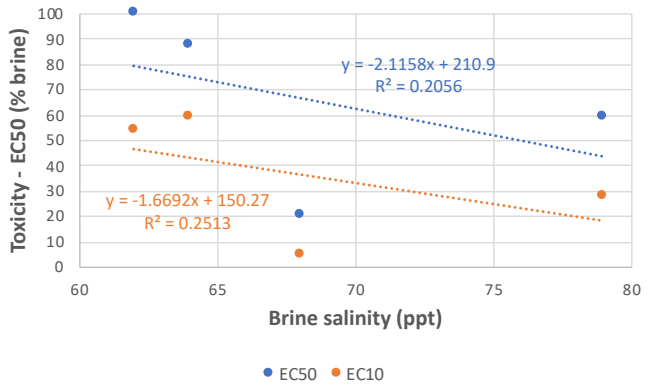


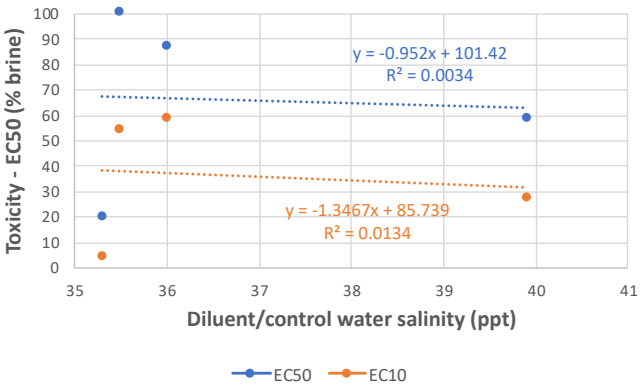
B. *Nitzschia closterium*



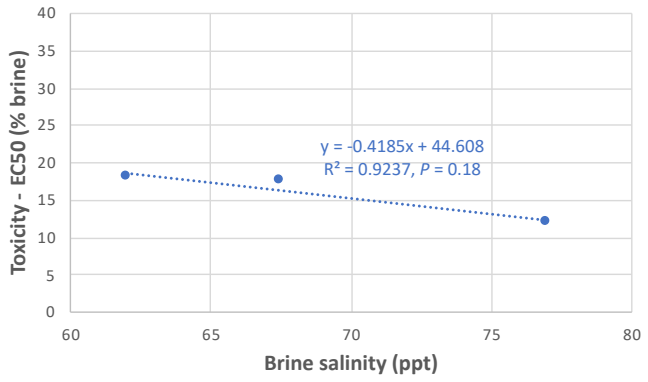


C. *Ecklonia radiata*





D. *Heliocidaris tuberculata*



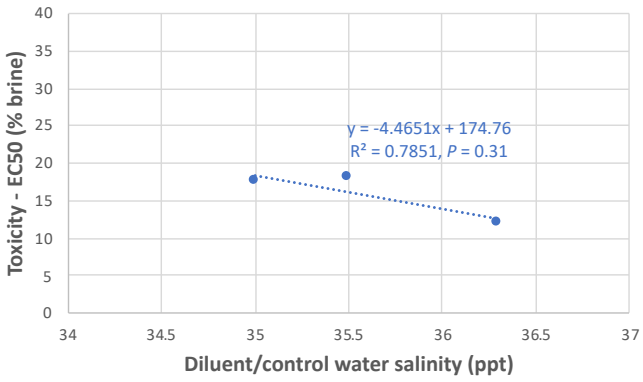


Figure A.1 Relationships between desalination brine salinity and brine ecotoxicity (left panels) and control water/test diluent salinity and brine ecotoxicity (right panels) for different species. *P* values only shown where $r^2 > 0.5$.

There was evidence of positive relationships between brine salinity and brine toxicity for several of the species, especially *S. lalandi*. For *S. lalandi*, there was a significant positive relationship between brine salinity and brine toxicity expressed as the EC50 (at the α=0.05 level). Although not significant or unable to be calculated due to insufficient sample size, the slopes of the relationships for brine toxicity expressed as the EC10 and LC50 were generally similar to the significant relationship based on the EC50.

Relationships between diluent/control water salinity and brine toxicity were either absent or very weak except for *S. lalandi*. For *S. lalandi*, there were positive relationships between diluent/control water salinity and brine toxicity expressed as the EC50, EC10 and LC50, although these were non-significant (at the α=0.05 level).

Although the relationships for most of the species were equivocal, there was enough of an indication of increasing brine toxicity with increasing salinity to not ignore this relationship. This is especially so given that the studies from which the data were sourced were not performed at the same time and in the same place and, thus, the data were also likely affected by other sources of variability (e.g. inter-laboratory variability, inter-test/intra-species variability, sample variability). Additionally, the generally high variability of the data for most of the species confirmed that it is not appropriate to automatically assume that ecotoxicity data from one desalination plant are applicable for the assessment of the ecotoxicity of a brine from another desalination plant. Thus, it is important for both the return water salinity and, to a lesser extent, the diluent/control water salinity for existing data to match those of the Project as closely as possible.

In addition to looking at the ecotoxicity data, it was possible to look at relationships between brine salinity or control water/test diluent salinity and the safe dilution factors that were derived for the five desalination plant ecotoxicity studies listed in Table 2. Figure A.2 shows these relationships for both the 95% and 99% species protection safe concentrations. Similar to the previous analyses, there is an apparent relationship between brine salinity and the safe concentration (Figure A.2A), supporting the notion that existing ecotoxicity data need to be based on brine salinity that closely matches that of the Project's return water. The relationship between the control water/test diluent and the safe dilution factor is less evident (Figure A.2A); however, again, the high variability suggests it is prudent to also ensure that existing ecotoxicity data need to be based on control water/test diluent salinity that closely matches that of the receiving waters for the proposed site.

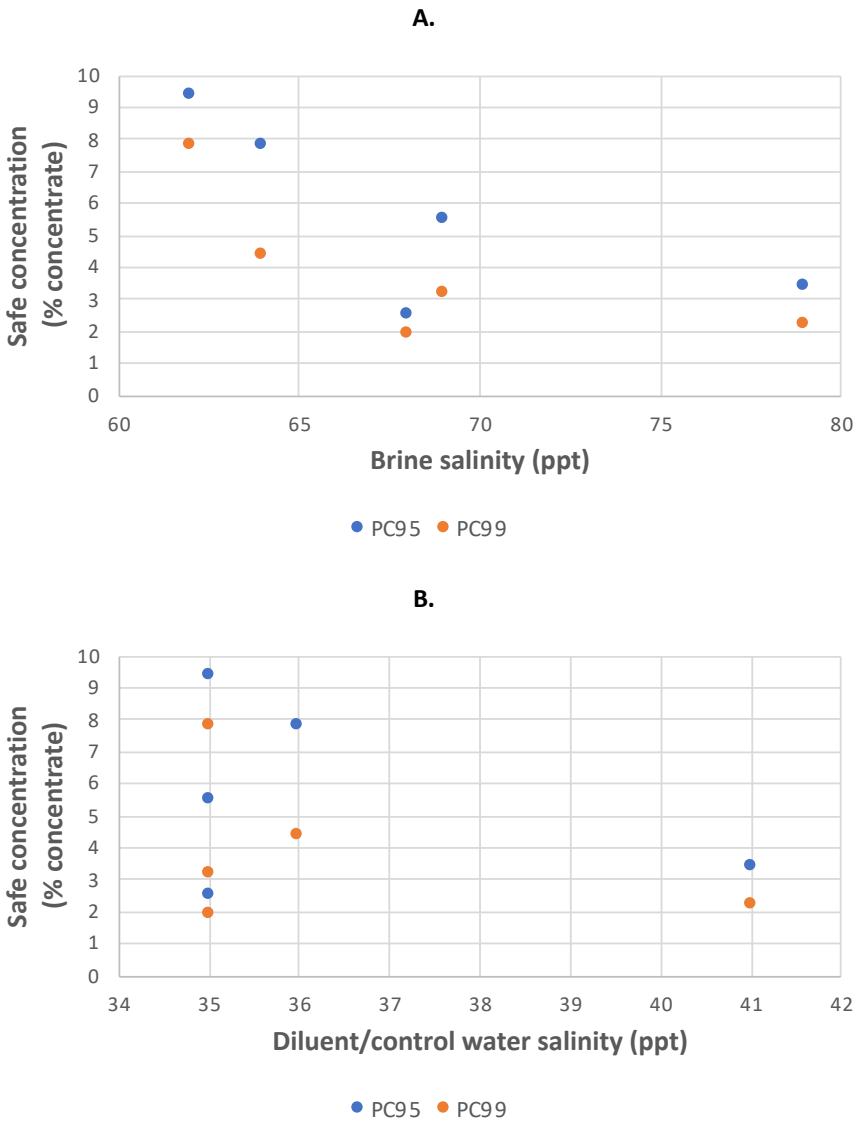


Figure A.2 Relationships between desalination brine salinity and safe concentration (A) and control water/test diluent salinity and safe concentration (B). PC95 refers to 95% species protection value; PC99 refers to 99% species protection value.

Appendix B Details of the relevance assessment

Table B.1 Details of the relevance assessment [a]

Study	Species	Family/Super-family	Taxonomic group (common name)	Test duration	Life stage	Endpoint	Brine salinity (ppt)	Brine salinity relevance	Test diluent salinity (ppt)	Test diluent salinity relevance	Test species locally present and locally collected?	Test species or its genus locally present but not locally collected? (A)	Represents ecologically important taxonomic group that is under-represented? (B)	Species not local but tested at relevant brine and diluent water salinities? (C)	Test species relevance	Test type (acute/ chronic)	Test type relevance	Overall relevance	Comments
O2 Marine (2021)	*Aiptasia pulchella*	Metridioidea	Cnidarian (sea anemone)	8-d	pedal lacerate	development	64	Rv3	36	Rv2	N	N	N	N	Rv3	Chronic	Rv1	Rv3	Not present in Upper Spencer Gulf (J Brook pers comm, April 2024)
Hydrobiology (2009)	*Allorchestes compressa*	Dogielinotidae	Crustacean (amphipod)	96-h	juvenile	survival	66-67	Rv1	36	Rv2	N	Y			Rv2a	Acute	Rv3	Rv3	Introduced species present in western and/or Upper Spencer Gulf (Dittmann et al. 2010, BHP Billiton 2011).
Hydrobiology (2009)	*Allorchestes compressa*	Dogielinotidae	Crustacean (amphipod)	14-d	juvenile	survival	66-67	Rv1	36	Rv2	N	Y			Rv2a	Chronic	Rv1	Rv2	Introduced species present in western and/or Upper Spencer Gulf (Dittmann et al. 2010, BHP Billiton 2011).
Hydrobiology (2009)	*Allorchestes compressa*	Dogielinotidae	Crustacean (amphipod)	14-d	juvenile	growth (biomass)	66-67	Rv1	36	Rv2	N	Y			Rv2a	Chronic	Rv1	Rv2	Introduced species present in western and/or Upper Spencer Gulf (Dittmann et al. 2010, BHP Billiton 2011).
Warne (2010)	*Allorchestes compressa*	Dogielinotidae	Crustacean (amphipod)	96-h	juvenile	survival	69	Rv1	35	Rv3	N	Y			Rv2a	Acute	Rv3	Rv3	Introduced species present in western and/or Spencer Gulf (Dittmann et al. 2010, BHP Billiton 2011).
Hydrobiology (2008)	*Allorchestes compressa*	Dogielinotidae	Crustacean (amphipod)	96-h	juvenile	survival	62	Rv3	38	Rv1	N	Y			Rv2a	Acute	Rv3	Rv3	Introduced species present in western and/or Upper Spencer Gulf (Dittmann et al. 2010, BHP Billiton 2011).
Geotechnical Services (2008a)	*Argyrosomus japonicus*	Sciaenidae	Fish (mulloway)	7-d	larva	growth	79	Rv3	41	Rv1	N	Y			Rv2a	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2011a, Spencer Gulf Port Link 2013)
Hydrobiology (2009)	*Chrysophrys auratus*	Sparidae	Fish (Pink snapper)	96-h	larva	imbalance	66-67	Rv1	36	Rv2	N	Y			Rv2a	Acute	Rv3	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013; Smart et al. 2022).
Intertek (2011)	*Chrysophrys auratus*	Sparidae	Fish (snapper)	7-d	larva	growth	68	Rv1	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013; Smart et al. 2022).
Hydrobiology (2009)	*Chrysophrys auratus*	Sparidae	Fish (Pink snapper)	7-d	larva	growth (length)	66-67	Rv1	36	Rv2	N	Y			Rv2a	Chronic	Rv1	Rv2	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013; Smart et al. 2022).
Geotechnical Services (2008a)	*Chrysophrys auratus*	Sparidae	Fish (snapper)	7-d	larva	growth	79	Rv3	41	Rv1	N	Y			Rv2a	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013; Smart et al. 2022).
Geotechnical Services (2008a)	*Crassostrea gigas*	Ostreoidea	Bivalve (pacific oyster)	48-h	larva	development	79	Rv3	41	Rv1	N	Y			Rv2a	Chronic	Rv1	Rv3	Introduced/aquaculture species present in Upper Spencer Gulf (J. Brook pers comm, May 2025)
Hydrobiology (2009)	*Diopatra dentata*	Onuphidae	Polychaete	14-d	2-month old	survival	66-67	Rv1	36	Rv2	N	N	Y		Rv2b	Chronic	Rv1	Rv2	Present at genus level (at least) at Mullaquana (J Brook pers comm, May 2025).
Hydrobiology (2009)	*Diopatra dentata*	Onuphidae	Polychaete	14-d	2-month old	growth (biomass)	66-67	Rv1	36	Rv2	N	N	Y		Rv2b	Chronic	Rv1	Rv2	Present at genus level (at least) at Mullaquana (J Brook pers comm, May 2025).
O2 Marine (2021)	*Echinometra mathaei*	Echinometridae	Echinoderm (sea urchin)	72-h	larva	development	64	Rv3	36	Rv2	N	N	N	N	Rv3	Chronic	Rv1	Rv3	Not present in Upper Spencer Gulf (J. Brook, pers comm, May 2025)
Hydrobiology (2009)	*Ecklonia radiata*	Lessoniaceae	Brown macroalga	14-d	Gametes	germination	66-67	Rv1	36	Rv2	N	N	N	Y	Rv2c	Chronic	Rv1	Rv2	Not present in Upper Spencer Gulf (BHP Billiton 2011b, J. Brook pers comm, May 2025)
Hydrobiology (2009)	*Ecklonia radiata*	Lessoniaceae	Brown macroalga	14-d	gametophyte	growth	66-67	Rv1	36	Rv2	N	N	N	Y	Rv2c	Chronic	Rv1	Rv2	Not present in Upper Spencer Gulf (BHP Billiton 2011b, J. Brook pers comm, May 2025)
Intertek (2011)	*Ecklonia radiata*	Lessoniaceae	Brown macroalga	72-h	Gametes	germination	68	Rv1	35	Rv3	N	N	N	N	Rv3	Chronic	Rv1	Rv3	Not present in Upper Spencer Gulf (BHP Billiton 2011b, J. Brook pers comm, May 2025)

Study	Species	Family/Super-family	Taxonomic group (common name)	Test duration	Life stage	Endpoint	Brine salinity (ppt)	Brine salinity relevance	Test diluent salinity (ppt)	Test diluent salinity relevance	Test species locally present and locally collected? (A)	Test species or its genus locally present but not locally collected? (A)	Represents ecologically important taxonomic group that is under-represented? (B)	Species not local but tested at relevant brine and diluent water salinities? (C)	Test species relevance	Test type (acute/ chronic)	Test type relevance	Overall relevance	Comments
O2 Marine (2021)	*Ecklonia radiata*	Lessoniaceae	Brown macroalga	72-h	Gametes	germination	68	Rv1	34	Rv3	N	N	N	N	Rv3	Chronic	Rv1	Rv3	Not present in Upper Spencer Gulf (BHP Billiton 2011b, J. Brook pers comm, May 2025)
Geotechnical Services (2008a)	*Ecklonia radiata*	Lessoniaceae	Brown macroalga	72-h	Gametes	germination	79	Rv3	41	Rv1	N	N	N	N	Rv3	Chronic	Rv1	Rv3	Not present in Upper Spencer Gulf (BHP Billiton 2011b, J. Brook pers comm, May 2025)
Hydrobiology (2008)	*Ecklonia radiata*	Lessoniaceae	Brown macroalga	72-h	Gametes	germination	61	Rv3	36	Rv2	N	N	N	N	Rv3	Chronic	Rv1	Rv3	Not present in Upper Spencer Gulf (BHP Billiton 2011b, J. Brook pers comm, May 2025)
Intertek (2011)	*Gladioferens imparipes*	Centropagidae	Crustacean (copepod)	21-d	Adult	Reproduction	68	Rv1	35	Rv3	N	N	Y		Rv2b	Chronic	Rv1	Rv3	Not present in Upper Spencer Gulf (BHP Billiton 2011b, J. Brook, pers comm, May 2025)
O2 Marine (2021)	*Gladioferens imparipes*	Centropagidae	Crustacean (copepod)	5-7-d	larva	development	68	Rv1	34	Rv3	N	N	Y		Rv2b	Chronic	Rv1	Rv3	Not present in Upper Spencer Gulf (BHP Billiton 2011b, J. Brook, pers comm, May 2025)
Geotechnical Services (2008a)	*Gladioferens imparipes*	Centropagidae	Crustacean (copepod)	2-d	Adult	Reproduction (28-d)	79	Rv3	41	Rv1	N	N	Y		Rv2b	Acute	Rv3	Rv3	Not present in Upper Spencer Gulf (BHP Billiton 2011b, J. Brook, pers comm, May 2025)
Intertek (2011)	*Heliocidaris erythrogramma*	Echinometridae	Echinoderm (sea urchin)	72-h	larva	development	68	Rv1	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2011a, b, Spencer Gulf Port Link 2013)
Intertek (2011)	*Heliocidaris erythrogramma*	Echinometridae	Echinoderm (sea urchin)	1-h	sperm/egg	fertilisation	69	Rv1	36	Rv2	N	Y			Rv2a	Chronic	Rv1	Rv2	Present in Upper Spencer Gulf (BHP Billiton 2011a, b, Spencer Gulf Port Link 2013)
Hydrobiology (2009)	*Heliocidaris erythrogramma*	Echinometridae	Echinoderm (sea urchin)	16-d	larva	metamorphosis	66-67	Rv1	36	Rv2	N	Y			Rv2a	Chronic	Rv1	Rv2	Present in Upper Spencer Gulf (BHP Billiton 2011a, b, Spencer Gulf Port Link 2013)
Warne (2010)	*Heliocidaris tuberculata*	Echinometridae	Echinoderm (sea urchin)	72-h	larva	development	68	Rv1	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	*Heliocidaris erythrogramma* present in Upper Spencer Gulf (BHP Billiton 2011a, b, Spencer Gulf Port Link 2013)
Warne (2010)	*Heliocidaris tuberculata*	Echinometridae	Echinoderm (sea urchin)	1-h	sperm/egg	fertilisation	68	Rv1	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	*Heliocidaris erythrogramma* present in Upper Spencer Gulf (BHP Billiton 2011a, b, Spencer Gulf Port Link 2013)
Hydrobiology (2009)	*Heliocidaris tuberculata*	Echinometridae	Echinoderm (sea urchin)	1-h	sperm/egg	fertilisation	66	Rv1	36	Rv2	N	Y			Rv2a	Chronic	Rv1	Rv2	*Heliocidaris erythrogramma* present in Upper Spencer Gulf (BHP Billiton 2011a, b, Spencer Gulf Port Link 2013)
Hydrobiology (2009)	*Heliocidaris tuberculata*	Echinometridae	Echinoderm (sea urchin)	72-h	larva	development	66	Rv1	36	Rv2	N	Y			Rv2a	Chronic	Rv1	Rv2	*Heliocidaris erythrogramma* present in Upper Spencer Gulf (BHP Billiton 2011a, b, Spencer Gulf Port Link 2013)
Hydrobiology (2006)	*Heliocidaris tuberculata*	Echinometridae	Echinoderm (sea urchin)	72-h	larva	development	77	Rv2	36	Rv2	N	Y			Rv2a	Chronic	Rv1	Rv2	*Heliocidaris erythrogramma* present in Upper Spencer Gulf (BHP Billiton 2011a, b, Spencer Gulf Port Link 2013)
Hydrobiology (2008)	*Heliocidaris tuberculata*	Echinometridae	Echinoderm (sea urchin)	1-h	sperm/egg	fertilisation	62	Rv3	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	*Heliocidaris erythrogramma* present in Upper Spencer Gulf (BHP Billiton 2011a, b, Spencer Gulf Port Link 2013)
Hydrobiology (2008)	*Heliocidaris tuberculata*	Echinometridae	Echinoderm (sea urchin)	72-h	larva	development	62	Rv3	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	*Heliocidaris erythrogramma* present in Upper Spencer Gulf (BHP Billiton 2011a, b, Spencer Gulf Port Link 2013)
Warne (2010)	*Hormosira banksii*	Hormosiraceae	Brown macroalga	72-h	Gametes	germination	69	Rv1	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2011a)
Hydrobiology (2006)	*Hormosira banksii*	Hormosiraceae	Brown macroalga	72-h	Gametes	germination	77	Rv2	37	Rv2	N	Y			Rv2a	Chronic	Rv1	Rv2	Present in Upper Spencer Gulf (BHP Billiton 2011a)
Intertek (2011)	*Isochrysis galbana*	Isochrysidaceae	Golden-brown microalga	72-h	log phase	cell yield	68	Rv1	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	Genus found in Upper Spencer Gulf (BHP 2009)
O2 Marine (2021)	*Isochrysis galbana*	Isochrysidaceae	Golden-brown microalga	72-h	log phase	cell yield	68	Rv1	34	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	Genus found in Upper Spencer Gulf (BHP 2009)
Geotechnical Services (2008a)	*Isochrysis galbana*	Isochrysidaceae	Golden-brown microalga	72-h	log phase	growth rate	79	Rv3	41	Rv1	N	Y			Rv2a	Chronic	Rv1	Rv3	Genus found in Upper Spencer Gulf (BHP 2009)

Study	Species	Family/Super-family	Taxonomic group (common name)	Test duration	Life stage	Endpoint	Brine salinity (ppt)	Brine salinity relevance	Test diluent salinity (ppt)	Test diluent salinity relevance	Test species locally present and locally collected?	Test species or its genus locally present but not locally collected? (A)	Represents ecologically important taxonomic group that is under-represented? (B)	Species not local but tested at relevant brine and diluent water salinities? (C)	Test species relevance	Test type (acute/ chronic)	Test type relevance	Overall relevance	Comments
O2 Marine (2021)	*Lates calcarifer*	Latidae	Fish (barramundi)	7-d	larva	imbalance	64	Rv3	36	Rv2	N	N	N	N	Rv3	Chronic	Rv1	Rv3	Not present in region (tropical species).
O2 Marine (2021)	*Lates calcarifer*	Latidae	Fish (barramundi)	7-d	larva	biomass	64	Rv3	36	Rv2	N	N	N	N	Rv3	Chronic	Rv1	Rv3	Not present in region (tropical species).
Warne (2010)	*Mimachlamys asperrima*	Pectinidae	Bivalve (scallop)	72-h	larva	development	68	Rv1	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2011a)
Hydrobiology (2009)	*Mimachlamys asperrima*	Pectinidae	Bivalve (scallop)	48-h	larva	development	66	Rv1	36	Rv2	N	Y			Rv2a	Chronic	Rv1	Rv2	Present in Upper Spencer Gulf (BHP Billiton 2011a)
Hydrobiology (2008)	*Mimachlamys asperrima*	Pectinidae	Bivalve (scallop)	48-h	larva	development	62	Rv3	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2011a)
Intertek (2011)	*Mytilus edulis*	Mytilidae	Bivalve (blue mussel)	48-h	larva	development	68	Rv1	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	Genus (possibly same species) present in Upper Spencer Gulf (BHP Billiton 2011a, Spencer Gulf Port Link 2013, J Diversity 2025)
O2 Marine (2021)	*Mytilus edulis*	Mytilidae	Bivalve (blue mussel)	48-h	larva	development	68	Rv1	34	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	Genus (possibly same species) present in Upper Spencer Gulf (BHP Billiton 2011a, Spencer Gulf Port Link 2013, J Diversity 2025)
Warne (2010)	*Nitzschia closterium*	Bacillariaceae	Diatom	72-h	log phase	growth rate	70	Rv1	35	Rv3	N	Unknown	Y		Rv2b	Chronic	Rv1	Rv3	Genus found in south west Spencer Gulf (Van Ruth et al. 2009); unclear if present in Upper Spencer Gulf (J. Brook, pers comm, May 2025)
Hydrobiology (2009)	*Nitzschia closterium*	Bacillariaceae	Diatom	72-h	log phase	cell yield	66-67	Rv1	36	Rv2	N	Unknown	Y		Rv2b	Chronic	Rv1	Rv2	Genus found in south west Spencer Gulf (Van Ruth et al. 2009); unclear if present in Upper Spencer Gulf (J. Brook, pers comm, May 2025)
Hydrobiology (2006)	*Nitzschia closterium*	Bacillariaceae	Diatom	72-h	log phase	growth rate	77	Rv2	37	Rv2	N	Unknown	Y		Rv2b	Chronic	Rv1	Rv2	Genus found in south west Spencer Gulf (Van Ruth et al. 2009); unclear if present in Upper Spencer Gulf (J. Brook, pers comm, May 2025)
O2 Marine (2021)	*Nitzschia closterium*	Bacillariaceae	Diatom	72-h	log phase	growth rate	64	Rv3	36	Rv2	N	Unknown	Y		Rv2b	Chronic	Rv1	Rv3	Genus found in south west Spencer Gulf (Van Ruth et al. 2009); unclear if present in Upper Spencer Gulf (J. Brook, pers comm, May 2025)
Hydrobiology (2008)	*Nitzschia closterium*	Bacillariaceae	Diatom	72-h	log phase	cell yield	60	Rv3	35	Rv3	N	Unknown	Y		Rv2b	Chronic	Rv1	Rv3	Genus found in south west Spencer Gulf (Van Ruth et al. 2009); unclear if present in Upper Spencer Gulf (J. Brook, pers comm, May 2025)
Geotechnical Services (2008a)	*Penaeus (Melicertus) latisulcatus*	Penaeidae	Crustacean (western king prawn)	21-d	juvenile	growth	79	Rv3	41	Rv1	Y				Rv1	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013, J Diversity 2025)
Geotechnical Services (2008a)	*Penaeus (Melicertus) latisulcatus*	Penaeidae	Crustacean (western king prawn)	28-d	adult	growth	79	Rv3	41	Rv1	Y				Rv1	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013, J Diversity 2025)
Hydrobiology (2006)	*Penaeus monodon*	Penaeidae	Crustacean (tiger prawn)	96-h	15-d post-larvae	survival	77	Rv2	36	Rv2	N	Y			Rv2a	Acute	Rv3	Rv3	Family represented by western king prawn
Hydrobiology (2006)	*Saccostrea commercialis*	Ostreoidea	Bivalve (Sydney rock oyster)	48-h	larva	development	77	Rv2	36	Rv2	N	N	N	Y	Rv2c	Chronic	Rv1	Rv2	Present at family level (Ostroidae) (BHP Billiton 2011a); Species and genus not recorded in Upper Spencer Gulf (J. Brook, pers comm, May 2025)
O2 Marine (2021)	*Saccostrea echinata*	Ostreoidea	Bivalve (milky oyster)	48-h	larva	development	64	Rv3	36	Rv2	N	N	N	N	Rv3	Chronic	Rv1	Rv3	Present at family level (Ostroidae) (BHP Billiton 2011a).
Geotechnical Services (2008a)	*Sepia apama*	Sepiidae	Cephalopod (Australian giant cuttlefish)	4 months	Embryo/larva	development	79	Rv3	41	Rv1	Y				Rv1 [b]	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013).
Geotechnical Services (2008a)	*Sepia apama*	Sepiidae	Cephalopod (Australian giant cuttlefish)	4 months	Embryo/larva	days to hatch	79	Rv3	41	Rv1	Y				Rv1 [b]	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013).
Geotechnical Services (2008a)	*Sepia apama*	Sepiidae	Cephalopod (Australian giant cuttlefish)	4 months	Embryo/larva	length at hatch	79	Rv3	41	Rv1	Y				Rv1 [b]	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013).

Study	Species	Family/Super-family	Taxonomic group (common name)	Test duration	Life stage	Endpoint	Brine salinity (ppt)	Brine salinity relevance	Test diluent salinity (ppt)	Test diluent salinity relevance	Test species locally present and locally collected?	Test species or its genus locally present but not locally collected? (A)	Represents ecologically important taxonomic group that is under-represented? (B)	Species not local but tested at relevant brine and diluent water salinities? (C)	Test species relevance	Test type (acute/ chronic)	Test type relevance	Overall relevance	Comments
Geotechnical Services (2008a)	*Sepia apama*	Sepiidae	Cephalopod (Australian giant cuttlefish)	4 months	Embryo/larva	weight at hatch	79	Rv3	41	Rv1	Y				Rv1 [b]	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013).
Geotechnical Services (2008a)	*Sepia apama*	Sepiidae	Cephalopod (Australian giant cuttlefish)	4 months	Embryo/larva	width at hatch	79	Rv3	41	Rv1	Y				Rv1 [b]	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013).
Geotechnical Services (2008a)	*Sepia apama*	Sepiidae	Cephalopod (Australian giant cuttlefish)	5 months	Embryo/larva	post-hatch survival	79	Rv3	41	Rv1	Y				Rv1 [b]	Chronic	Rv1	Rv3	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013).
Geotechnical Services (2006)	*Sepia apama*	Sepiidae	Cephalopod (Australian giant cuttlefish)	5 months	Embryo/larva	development	78	Rv2	45	Rv2	Y				Rv1 [b]	Chronic	Rv1	Rv2	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013).
Geotechnical Services (2006)	*Sepia apama*	Sepiidae	Cephalopod (Australian giant cuttlefish)	5 months	Embryo/larva	days to hatch	78	Rv2	45	Rv2	Y				Rv1 [b]	Chronic	Rv1	Rv2	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013).
Geotechnical Services (2006)	*Sepia apama*	Sepiidae	Cephalopod (Australian giant cuttlefish)	5 months	Embryo/larva	post-hatch survival	78	Rv2	45	Rv2	Y				Rv1 [b]	Chronic	Rv1	Rv2	Present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b, Spencer Gulf Port Link 2013).
O2 Marine (2021)	*Seriola lalandi*	Carangidae	Fish (yellowtail kingfish)	7-d	larva	survival/ development	68	Rv1	34	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	aquaculture species present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b).
Warne (2010)	*Seriola lalandi*	Carangidae	Fish (yellowtail kingfish)	96-h	larva	imbalance	70	Rv1	35	Rv3	N	Y			Rv2a	Acute	Rv3	Rv3	aquaculture species present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b).
Hydrobiology (2006)	*Seriola lalandi*	Carangidae	Fish (yellowtail kingfish)	96-h	fingerling	survival	84	Rv3	40	Rv1	Y				Rv1	Acute	Rv3	Rv3	aquaculture species present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b).
Hydrobiology (2008)	*Seriola lalandi*	Carangidae	Fish (yellowtail kingfish)	96-h	larva	imbalance	61	Rv3	36	Rv2	N	Y			Rv2a	Acute	Rv3	Rv3	aquaculture species present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b).
Geotechnical Services (2008c)	*Seriola lalandi*	Carangidae	Fish (yellowtail kingfish)	7-d	larva	growth	78	Rv2	44.3	Rv2	Y				Rv1	Chronic	Rv1	Rv2	aquaculture species present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b).
Geotechnical Services (2008c)	*Seriola lalandi*	Carangidae	Fish (yellowtail kingfish)	7-d	larva	growth	78	Rv2	44.3	Rv2	Y				Rv1	Chronic	Rv1	Rv2	aquaculture species present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b).
Geotechnical Services (2008a)	*Seriola lalandi*	Carangidae	Fish (yellowtail kingfish)	7-d	larva	growth	79	Rv3	35	Rv3	N	Y			Rv2a	Chronic	Rv1	Rv3	aquaculture species present in Upper Spencer Gulf (BHP Billiton 2009, 2011a, b).

[a] See Table 5 for details of relevance assessment criteria.

[b] Note that toxicity tests for *Sepia apama* were done using embryos and larvae – life stages not likely to be present at the proposed site due to the lack of appropriate breeding/nursery habitat.

Appendix C Data quality assessment systems from Warne et al. (2018)

The Tables below are reproduced from Warne et al. (2018) and represent the quality assessment scoring systems for marine/estuarine animals (Table C.1) and plants/algae (Table C.2). They were used to assess the quality of the relevant ecotoxicity data for the Project.

Table C.1 Scoring system for assessing the quality of toxicity data for contaminants to marine/estuarine non-plant species to be used in the derivation of guideline values for toxicants

QUESTION		MARK
1	Was the duration of the exposure stated (for example 48 or 96 h)?	Yes (10), No (0)
2	Was the biological endpoint (for example immobilisation or population growth) stated and defined?	Yes (10), Stated only (5), Neither (0)
3	Was the biological effect stated (for example LC or NOEC)?	Yes (5), No (0)
4	Was the biological effect quantified (for example 50% effect, 25% effect)? Note: The effect for NOEC and LOEC data must be quantified.	Yes (5), No (0)
5	Were appropriate controls (for example a no-toxicant control and/or solvent control) used?	Yes (5), No (0)
6	Was each control and chemical concentration at least duplicated?	Yes (5), No (0)
7	Were test acceptability criteria stated (for example mortality in controls must not exceed a certain percentage) or were test acceptability criteria inferred (for example test methods used were USEPA or OECD)? Note: Data that fail the acceptability criteria are automatically deemed to be of unacceptable quality and must not be used.	Stated (5), Inferred (2), Neither (0)
8	Were the characteristics of the test organism (for example length, mass, age) stated?	Yes (5), No (0)
9	Was the type of test media used stated?	Yes (5), No (0)
10	Was the type of exposure (for example static, flow-through) stated?	Yes (4), No (0)
11	Were the contaminant concentrations measured at the beginning and end of the exposure? Note: Normally, toxicity data calculated using nominal concentration data would not be used to derive GVs; however, professional judgement can be used to include such data, provided a justification for their use is provided.	Yes (4), Measured once (2), Not measured or stated (0)
12	Were parallel reference toxicant toxicity tests conducted?	Yes (4), No (0)
13	Was there a concentration-response relationship either observable or stated?	Yes (4), No (0)
14	Was an appropriate statistical method or model used to determine the toxicity? Note: They should be accepted by a recognised national or international regulatory body (for example USEPA, OECD and ASTM)	Yes (4), No (0)
15	For LC/EC/NEC/BEC data, was an estimate of variability provided? OR For NOEC/LOEC/MDEC/MATC data, was the significance level 0.05 or less?	Yes (4), No (0)
16	Were the following parameters measured and stated?	
16.1	Conductivity/Salinity	Measured and stated (3), Measured only (1), Neither (0)
16.2	Dissolved oxygen	Measured and stated (3), Measured only (1), Neither (0)

QUESTION		MARK
16.3	Conductivity	Measured and stated (3), Measured only (1), Neither (0)
16.4	Dissolved organic carbon	Measured and stated (3), Measured only (1), Neither (0)
16.5	pH - pH should be measured at least at the beginning and end of the toxicity test	Measured at the beginning and end of the test and stated (3), Measured once (1), Not measured or stated (0)
17	Was the temperature measured and stated?	Measured and stated (3), Measured but not stated or temperature of the room or chamber was stated (1), Neither (0)
18	Were test solutions, blanks and/or controls tested for contamination or were analytical reagent grade chemicals or the highest possible purity chemicals used for the experiment?	Yes (3), No (0)
	Total score **Total possible score for Marine and estuarine/contaminants/non-plant data = 100** **Quality score: [Total score/Total possible score] x 100** **Quality class:** **high quality = when quality score ≥ 80%** **acceptable quality = when quality score ≥50–79%** **unacceptable quality = when quality score <50%,**	

Source: modified from Zhang et al. 2015 (note the modifications only affect the appearance of the table)

Table C.2 Scoring system for assessing the quality of toxicity data for contaminants to marine/estuarine plant species to be used in the derivation of guideline values for toxicants

QUESTION		MARK
1	Was the duration of the exposure stated (for example 48 or 96 h)?	Yes (10), No (0)
2	Was the biological endpoint (for example immobilisation or population growth) stated and defined?	Yes (10), Stated only (5), Neither (0)
3	Was the biological effect stated (for example LC or NOEC)?	Yes (5), No (0)
4	Was the biological effect quantified (for example 50% effect, 25% effect)? Note: The effect for NOEC and LOEC data must be quantified.	Yes (5), No (0)
5	Were appropriate controls (for example a no-toxicant control and/or solvent control) used?	Yes (5), No (0)
6	Was each control and chemical concentration at least duplicated?	Yes (5), No (0)
7	Were test acceptability criteria stated (for example mortality in controls must not exceed a certain percentage) or were test acceptability criteria inferred (for example test methods used were USEPA or OECD)? Note: Data that fail the acceptability criteria are automatically deemed to be of unacceptable quality and must not be used.	Stated (5), Inferred (2), Neither (0)

QUESTION		MARK
8	Were the characteristics of the test organism (for example length, mass, age) stated?	Yes (5), No (0)
9	Was the type of test media used stated?	Yes (5), No (0)
10	Was the type of exposure (for example static, flow-through) stated?	Yes (4), No (0)
11	Were the contaminant concentrations measured at the beginning and end of the exposure? Note: Normally, toxicity data calculated using nominal concentration data would not be used to derive GVs; however, professional judgement can be used to include such data, provided a justification for their use is provided.	Yes (4), Measured once (2), Not measured or stated (0)
12	Were parallel reference toxicant toxicity tests conducted?	Yes (4), No (0)
13	Was there a concentration-response relationship either observable or stated?	Yes (4), No (0)
14	Was an appropriate statistical method or model used to determine the toxicity? Note: They should be accepted by a recognised national or international regulatory body (for example USEPA, OECD and ASTM)	Yes (4), No (0)
15	For LC/EC/NEC/BEC data, was an estimate of variability provided? OR For NOEC/LOEC/MDEC/MATC data, was the significance level 0.05 or less?	Yes (4), No (0)
16	Were the following parameters measured and stated?	
16.1	Conductivity/Salinity	Measured and stated (3), Measured only (1), Neither (0)
16.2	pH	Measured and stated (3), Measured only (1), Neither (0)
16.3	Dissolved organic carbon	Measured and stated (3), Measured only (1), Neither (0)
17	Was the temperature measured and stated?	Measured and stated (3), Measured but not stated or temperature of the room or chamber was stated (1), Neither (0)
18	Were test solutions, blanks and/or controls tested for contamination or were analytical reagent grade chemicals or the highest possible purity chemicals used for the experiment?	Yes (3), No (0)
	Total score **Total possible score for Marine and estuarine/contaminants/plant data = 94** **Quality score: [Total score/Total possible score] x 100** **Quality class:** **high quality = when quality score ≥ 80%** **acceptable quality = when quality score ≥50–79%** **unacceptable quality = when quality score <50%**	

Source: modified from Zhang et al. 2015 (note the modifications only affect the appearance of the table)

Appendix D Details of the reliability assessment

Table D.1 Results of reliability assessment for data from tests that passed the relevance assessment.

Species	Test endpoint	Type of toxicity value	Reliability factors [a]				Overall reliability	Reference
			Quality assessment		Data limitations			
			Quality score (%)	Reliability	Details	Reliability		
Allorchestes compressa	14-d juvenile survival	EC10	88.3	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, with zero, partial and full effects. • Data analysis (linear interpolation) probably not appropriate for survival (quantal) data (e.g. probit analysis more appropriate), which may have had an effect on the EC10.	Rb2	**Rb2**	Hydrobiology (2009)
Allorchestes compressa	14-d juvenile growth	EC10	88.3	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, with zero, partial and full effects, but with some non-monotonicity at low concentrations. • Data analysis (linear interpolation) appropriate.	Rb1	**Rb1**	Hydrobiology (2009)
Chrysophrys auratus	7-d larval growth (length)	EC10	84.0	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, but only zero and full effects; no partial effects. • Data analysis (linear interpolation) appropriate.	Rb1	**Rb1**	Hydrobiology (2009)
Diopatra dentata	14-d juvenile survival	EC10	84.0	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, with zero, partial and full effects. • Data analysis (maximum likelihood probit) appropriate.	Rb1	**Rb1**	Hydrobiology (2009)
Diopatra dentata	14-d juvenile growth (biomass)	EC10	84.0	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship not observed; High variability in growth both between and within treatments. • Data analysis type not applicable; No EC10 could be estimated, other than >50%.	Rb3	**Rb3**	Hydrobiology (2009)
Ecklonia radiata	14-d germination	EC10	82.4	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship not observed. • Data analysis type not applicable; No EC10 could be estimated, other than >100%.	Rb3	**Rb3**	Hydrobiology (2009)
Ecklonia radiata	14-d growth	EC10	82.4	Rb1	• Test acceptability criteria met. • Partial concentration-response (CR) relationship observed; ~50% response at highest concentration. • Data analysis (linear interpolation) appropriate.	Rb1	**Rb1**	Hydrobiology (2009)
Heliocidaris erythrogramma	16-d larval metamorphosis	NOEC	92.6	Rb1	• Test acceptability criterion met. • Concentration-response (CR) relationship observed, but only with zero and full effects; no partial effects. • Data analysis (Dunnett's test) appropriate for hypothesis test, although concentration-response modelling would have been better (i.e. EC10s preferred over NOECs); note that % effect at the NOEC was +1%.	Rb2	**Rb2**	Hydrobiology (2009)
Heliocidaris erythrogramma	1-h fertilisation	EC10	92.6	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, with zero, low, medium and high partial effects. • Data analysis (maximum likelihood probit) appropriate.	Rb1	**Rb1**	Intertek (2011)

Species	Test endpoint	Type of toxicity value	Reliability factors [a]				Overall reliability	Reference
			Quality assessment		Data limitations			
			Quality score (%)	Reliability	Details	Reliability		
Heliocidaris tuberculata	72-h larval development	NOEC	91.5	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, but only zero and full effects; no partial effects. • Data analysis (Dunnett's test) appropriate for hypothesis test, although concentration-response modelling would have been better (i.e. EC10s preferred over NOECs); note that % effect at the NOEC was <5%.	Rb2	**Rb2**	Hydrobiology (2006)
Heliocidaris tuberculata	1-h fertilisation	EC10	92.6	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, with zero, low partial and full effects. • Data analysis (log-logit interpolation) appropriate.	Rb1	**Rb1**	Hydrobiology (2009)
Heliocidaris tuberculata	72-h larval development	EC10	91.5	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, but only with zero and full effects. • Data analysis (log-logit interpolation) appropriate.	Rb1	**Rb1**	Hydrobiology (2009)
Hormosira banksii	72-h germination	NOEC	89.0	Rb1	• Test acceptability criteria met. • Poor concentration-response (CR) relationship observed, with mostly zero and only one partial effect. • Data analysis (Dunnett's test) appropriate for hypothesis test, although concentration-response modelling would have been better (i.e. EC10s preferred over NOECs); note that % effect at the NOEC was <1%.	Rb2	**Rb2**	Hydrobiology (2006)
Mimachlamys asperrima	48-h larval development	EC10	91.5	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, but only with zero and full effects. • Data analysis (log logit interpolation) appropriate.	Rb1	**Rb1**	Hydrobiology (2009)
Nitzschia closterium	72-h growth rate	EC10	91.2	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, with zero, low partial and full effects. • Data analysis (linear interpolation) appropriate.	Rb1	**Rb1**	Hydrobiology (2006)
Nitzschia closterium	72-h growth rate	EC10	82.4	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, with two low partial, two mid partial and one full effect; no zero effects except control; effect at lowest concentration was 13%. • Data analysis (linear interpolation) appropriate.	Rb1	**Rb1**	Hydrobiology (2009)
Saccostrea commercialis	48-h larval development	NOEC	91.5	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, but only zero and full effects; no partial effects. • Data analysis (Dunnett's test) appropriate for hypothesis test, although concentration-response modelling would have been better (i.e. EC10s preferred over NOECs); note that % effect at the NOEC was <2%.	Rb2	**Rb2**	Hydrobiology (2006)
Sepia apama	5-mnth embryo/larval development	EC10	87.2	Rb1	• No test acceptability criteria as non-standard test. • Poor concentration-response (CR) relationship observed, with three low partial effects and four full effects; no zero effects except control. • Data not suitable for concentration-response modelling – high uncertainty in the EC10 estimate (note that NOEC was <1.5%).	Rb3	**Rb3**	Geotechnical Services (2006)
Sepia apama	5-mnth embryo days to hatch	EC10	83.0	Rb1	• No test acceptability criteria as non-standard test.	Rb3	**Rb3**	Geotechnical Services (2006)

Species	Test endpoint	Type of toxicity value	Reliability factors [a]				Overall reliability	Reference
			Quality assessment		Data limitations			
			Quality score (%)	Reliability	Details	Reliability		
					• Poor concentration-response (CR) relationship observed, with three partial effects and one full effects; no zero effects except control; effect at lowest concentration was 30%. • Data not suitable for concentration-response modelling – high uncertainty in the EC10 estimate (note that NOEC was <1.5%).			
Sepia apama	5-mnth post-hatch survival	EC10	87.2	Rb1	• No test acceptability criteria as non-standard test; however, control survival was 80%, which is typically deemed to be acceptable. • Concentration-response (CR) relationship observed, with zero, mid partial and full effects. • Data analysis (maximum likelihood probit) appropriate, albeit still high uncertainty in the EC10 estimate.	Rb2	**Rb2**	Geotechnical Services (2006)
Seriola lalandi	7-d larval growth (Brine without Nalco additive)	EC10	93.6	Rb1	• Test acceptability criteria met. • Concentration-response (CR) relationship observed, with zero, partial and full effects. • Data analysis (maximum likelihood probit) appropriate.	Rb1	**Rb1**	Geotechnical Services (2008c)
Seriola lalandi	7-d larval growth (Brine with Nalco additive)	EC10	89.4	Rb1	• Test acceptability criteria met. • Poor concentration-response (CR) relationship observed, with four partial effects and one full effect; no zero effects except control; effect at lowest concentration was 14%. • Data analysis (maximum likelihood probit) appropriate, but dataset not very amenable to concentration-response modelling	Rb3	**Rb3**	Geotechnical Services (2008c)

[a] Refer to Table 7 for details of reliability categories for each factor.

Appendix E Details of shinyssdtools SSD fitting process for the preliminary ecotoxicity dataset

The SSD software package shinyssdtools (V 0.4.0) uses a model averaging approach to generating a SSD (as described by Fox et al. 2021). Figure E.1 shows the full set of default distributions fitted to the preliminary ecotoxicity dataset (excluding the *S. apama* and *S. lalandi* data), while Table E.1 provides the relevant goodness of fit and weighting estimates for each of the distributions. The final, model-averaged SSD is shown in Figure 1 of the main report.

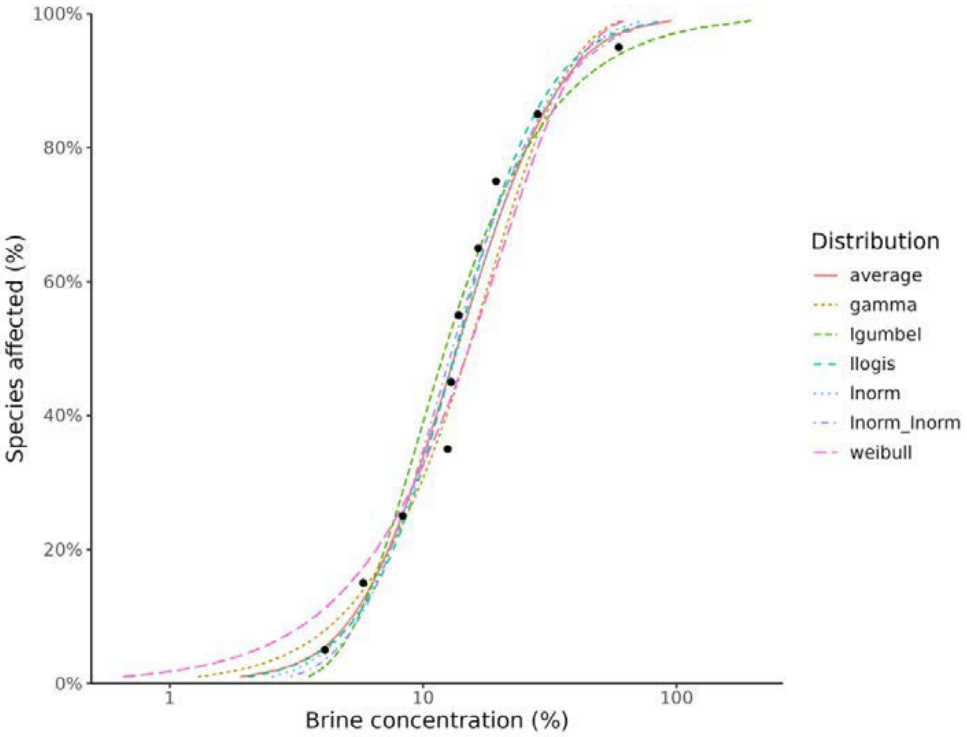


Figure E.1 SSDs for the set of six default distributions used to model the preliminary ecotoxicity dataset (excluding the *S. apama* and *S. lalandi* data)

Table E.1 Goodness of fit and weighting estimates for distributions used in model averaging for the preliminary ecotoxicity dataset

Distribution	Akaike's Information Criterion (AIC)	Corrected Akaike's Information Criterion (AICc)	Delta	Weight
Log normal	78.4	80.1	0	0.278
Log logistic	78.5	80.2	0.162	0.256
Log gumbel	78.8	80.5	0.407	0.227
Gamma	79.7	81.4	1.33	0.143
Weibull	80.5	82.2	2.1	0.097
Log normal - Log normal	84.2	99.2	19.1	0

Appendix F Species sensitivity distribution and associated safe dilutions using the preliminary dataset inclusive of the *Sepia apama* and *Seriola lalandi* data

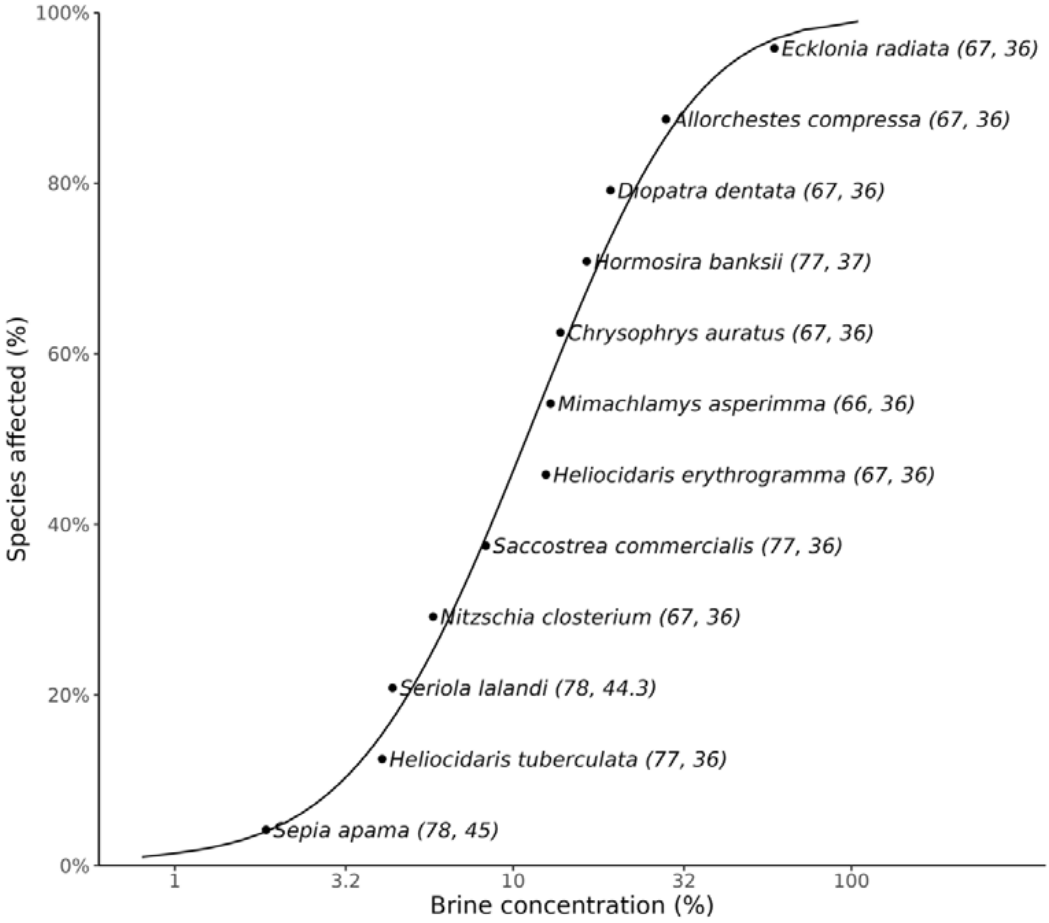


Figure F.1 Model averaged species sensitivity distribution for preliminary chronic toxicity dataset inclusive of the *S. apama* and *S. lalandi* data.

In Figure F.1 - values in parentheses for each species represent the brine salinity and test diluent salinity, respectively) used in the toxicity tests.

Table F.1 Secondary preliminary safe dilutions for desalination return water for the Project

species protection	Preliminary safe dilution	
	% return water	Dilution factor
99%	0.8	1 in 125
95%	2.1	1 in 48
90%	3.1	1 in 32

Appendix G-11:

Desalination Plant Discharge Modelling – Nearfield Modelling

WATER TECHNOLOGY
WATER, COASTAL & ENVIRONMENTAL CONSULTANTS

Report

Northern Water Desalination Plant Discharge Modelling – Nearfield Modelling

JBS&G/Department for Infrastructure and Transport

23 September 2026



Document Status

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Department for Infrastructure and Transport

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ACKNOWLEDGEMENT OF COUNTRY

The Board and employees of Water Technology acknowledge and respect the Aboriginal and Torres Strait Islander Peoples as the Traditional Custodians of Country throughout Australia. We specifically acknowledge the Traditional Custodians of the land on which our offices reside and where we undertake our work.

We respect the knowledge, skills and lived experiences of Aboriginal and Torres Strait Islander Peoples, who we continue to learn from and collaborate with. We also extend our respect to all First Nations Peoples, their cultures and to their Elders, past and present.

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ABBREVIATIONS

DN – Diameter Nominal

LAT – Lowest Astronomical Tide

PAE – Projected-Area Entrainment

SPTV – Species Protection Trigger Value

TEOS-10 – Thermodynamic Equation of Sea Water 2010

TSS – Total Suspended Solids

1 INTRODUCTION

Northern Water (the Project) aims to provide a new, climate resilient water source for industry in South Australia's Far North and Upper Spencer Gulf. This will enable the growth of industries and key mining resources crucial for contributing to decarbonisation initiatives, sustainability and unlocking long term economic prosperity for South Australia.

The Project involves construction and operation of a large-scale seawater desalination plant in Spencer Gulf, connected to northern South Australia via several hundred kilometres of pipeline, on the traditional lands of the Barngarla and Kokatha peoples.

The desalination plant and main water supply pipeline will run north from the desalination plant to Olympic Dam passing Whyalla, Port Augusta, Woomera and Roxby Downs, and a lateral branch pipeline to a delivery point at the Carrapateena copper-gold mine.

Water for the Project will be sourced from a seawater desalination plant to be installed near Mullaquana Station in Spencer Gulf, located approximately 25 km south-west from the centre of Whyalla (Figure 1-1). The delivery of the Project will involve the construction and operation of a 51 gigalitre per year (GL/y) seawater reverse osmosis desalination plant including marine infrastructure to extract raw seawater and discharge return water to the marine environment. As part of Project environmental approval studies, marine modelling is required to assess dispersion of the return water and the effects of the plant water intake on the environment and other receptors.

Modelling of return water dispersion was undertaken using a three-tiered modelling approach, including:

- A **farfield** model focusing on Spencer Gulf-wide effects of the desalination plant (from 10's to 100's of km);
- A **midfield** model focused on dispersion of the return water in the local vicinity of the point of discharge (from 50 m to 10 km); and
- A **nearfield** model focused on effects of the discharge in the immediate vicinity of the point of discharge (from 1 to 100 m from the outfall).

In this report, details of nearfield modelling are presented, with mid and farfield modelling described in separate reports (Doubell and James; 2026 and Water Technology; 2026, respectively). The modelling was based on the diffuser engineering design provided by the Department for Infrastructure and Transport (DIT) and provides details of return water dispersion in the immediate vicinity (1-100 m) of the diffuser for a range of ambient flow velocities within the receiving environment. Details of the outfall design and modelling configuration are provided below.

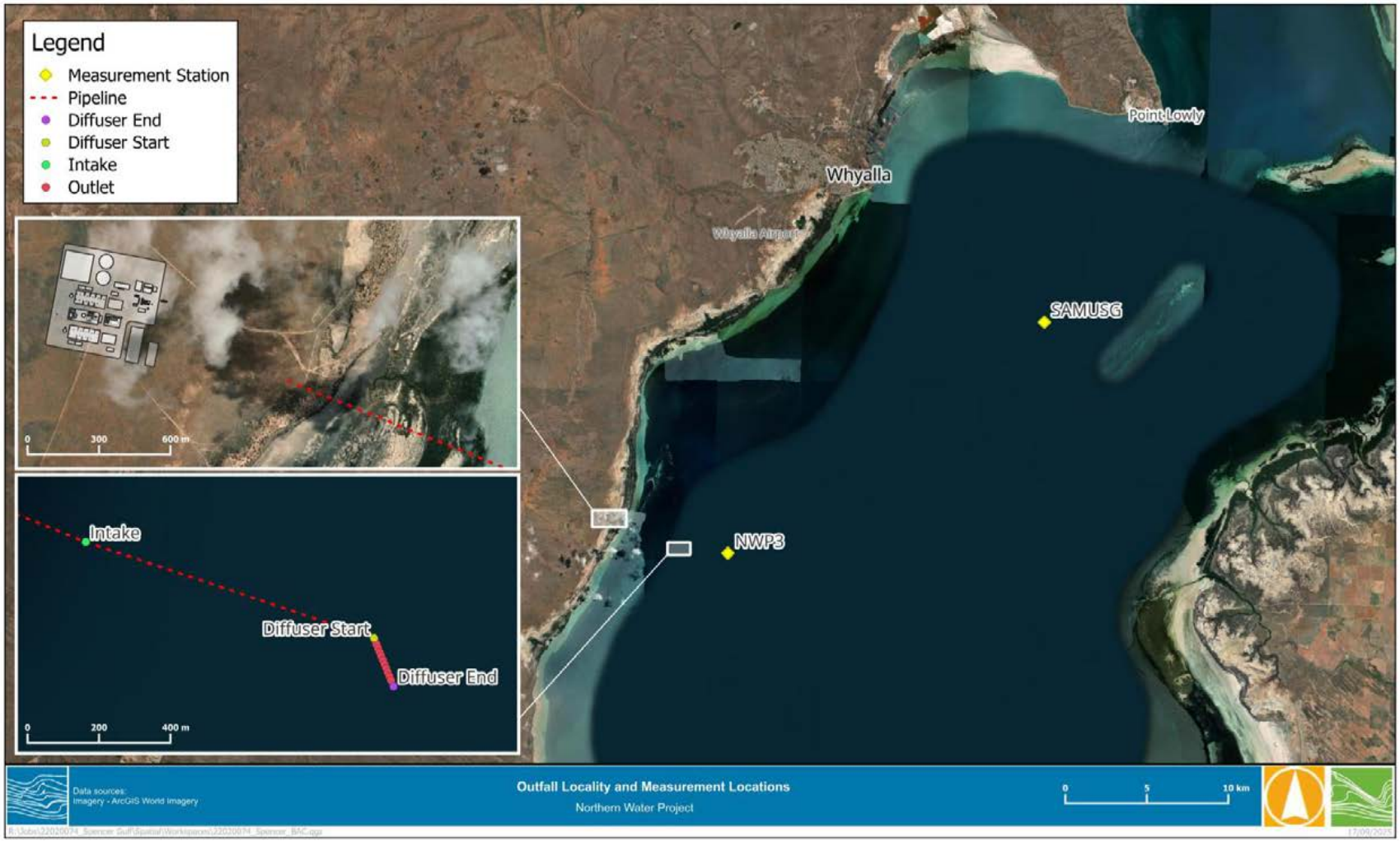


Figure 1-1 Outfall locality and measurement locations

2 DIFFUSER DESIGN

Proposed infrastructure will include:

- Separate (but running in parallel) intake and outfall tunnels;
- A single intake structure; and
- A linear alternating-ports diffuser.

The linear diffuser will have provision for up to 12 ports, with the aim of servicing proposed flow rates at different stages of the Project. Locations of the proposed intake and outfall alignments are shown in Figure 1-1. A schematic of the proposed outfall configuration is shown in Figure 2-1, while conceptual engineering drawings are provided in Appendix A.

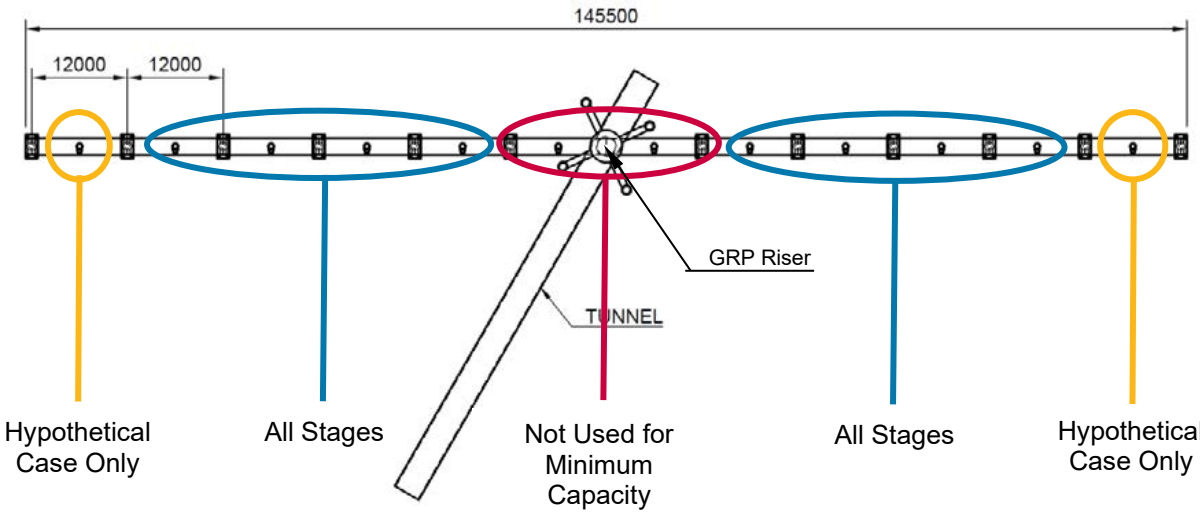


Figure 2-1 Alternating riser configuration for the Mullaquana Station Desalination Plant outfall diffuser (from GHD 2025)

Main components of the outfall design include:

- An outfall tunnel, ending in approximately 17 m water depth;
- A Glass-Reinforced-Polymer (GRP) riser through the seabed at the deep end of the tunnel; and
- A diffuser manifold comprising 12 alternating ports (250 mm DN) installed on the seabed.

A list of outfall design criteria relevant to the present assessment is provided in Table 2-1. Of note, characteristics for minimum capacity (40 ML/d treated water production, also referred to as 'turn down'), 51 GL/year production capacity (160 ML/d treated water production), and 73 GL/year hipthetical high-capacity scenario (200 ML/d treated water) are presented. For turn down conditions, a 35 ML/d bypass flow between the intake and outfall is adopted to reduce discharge salinity and improve jet mixing.

Table 2-1 Outfall reference design characteristics

Item	Criteria/Characteristic			
Production Capacity	51 GL/year (160 ML/d) – Design capacity		73 GL/year (200 ML/d) – Hypothetical high-capacity scenario	
Species protection trigger level	Assessment against 90%, 95% and 99% protection (noting that assessment requirements stipulate 95%)			
Species Protection Trigger Values (SPTV) and associated dilution	Point of return dilution of 30 to 40:1 Ecotoxicology assessment recommends following dilutions:			
	SPTV	90%	95%	99%
	Dilution	19	26	53
Dispersion	Avoid any visible plume			
Diffuser spacing	Minimum of 500 m between outfall diffusers and intake structure			
51 GL/year production capacity - Daily production, intake and outfall flows	160 ML/d; 389 ML/d (intake); 229 ML/d (outfall)			
73 GL/year hypothetical high-capacity scenario - Daily production, intake and outfall flows	200 ML/d; 486 ML/d (intake); 286 ML/d (outfall)			
Minimum production capacity - Daily production, intake and outfall flows	40 ML/d (instantaneous); 135 ML/d (intake); 92 ML/d (outfall); 35 ML/d (bypass flow)			
Temperature range	12 °C to 25 °C (ambient) 15 °C to 28 °C (outfall)			
Salinity range	38.0 to 41.0 (ambient) 65.8 to 71.0 (discharge, 51 and 73 GL/year capacity) 55.8 to 59.6 (discharge turn-down)			
Total suspended solids (TSS) minimum and peak times conditions	Estimate	Ambient	At outfall	
			Minimum	51 and 73 GL/year
	Lower	7 mg/L	12 mg/L	15 mg/L
	Higher	17 mg/L	28 mg/L	35 mg/L
Local depth	Minimum 17 m at point of discharge			
Number of diffuser ports (nozzles) for different capacities	Minimum		51 GL/year	73 GL/year
	6		10	12
Port inclination	45° with the vertical			
Port direction bearings	31 °N, 211 °N			
Clearance of nozzle from seabed	3.0 m			
Nozzle exit diameter	250 mm			
Nozzle exit flow and velocity	Minimum		51 GL/year	73 GL/year
	0.133 m^3/s 2.71 m/s		0.265 m^3/s 5.39 m/s	0.275 m^3/s 5.61 m/s

3 SCALING ANALYSIS

Abessi and Roberts (2015) addressed the nearfield characteristics of a negatively buoyant discharge assuming a single port diffuser at different discharge inclination angles in a stagnant/stationary environment. A schematic diagram of the nearfield characteristics provided by the scaling is shown in Figure 3-1.

The momentum to buoyancy length scale (L_M) and the port densimetric Froude number (F) respectively provide scaling relationships for plume geometrical characteristics and dilution at different distances from the port. These parameters are given by (Abessi and Roberts, 2015):

$$L_M = \frac{M^{3/4}}{B^{1/2}}$$

$$F = \frac{U}{\left(dg\frac{\rho-\rho_o}{\rho_o}\right)^{1/2}}$$

where U is the port exit velocity, g is the acceleration due to gravity, ρ is the return water density, and ρ_o is the ambient density. Momentum flux (M) and buoyancy flux (B) are given by:

$$M = U^2\frac{\pi d^2}{4}$$

$$B = U\frac{\pi d^2}{4}g\frac{\rho-\rho_o}{\rho_o}$$

Thus, geometrical characteristics and dilution are calculated as follows:

$$\chi = L_M f(\theta) \text{ and } S = Ff(\theta)$$

with respective constants for a port angle $\theta = 45^o$:

- Maximum terminal plume height above port $y_t = 2.0L_M$;
- Distance to the point of return ($x_i = 3.8L_M$) (i.e., when the descending plume reaches the same level as the port);
- Dilution at the point of return ($S_i = 1.6F$);
- Distance to the end of the nearfield ($x_n = 11.9L_M$) (i.e., where large scale instabilities leading to mixing significantly reduce); and
- Dilution at the end of the nearfield ($S_n = 2.4F$).

This scaling analysis was used for initial checking of the diffuser design and nearfield modelling under stagnant conditions, as presented in the following sections.

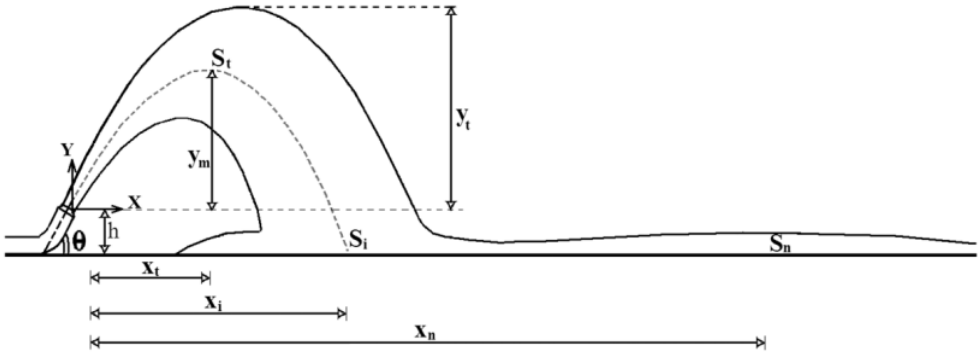


Figure 3-1 Characteristics of a single inclined dense jet in stagnant water (from Abessi and Roberts 2015)

4 NEARFIELD MODELLING

4.1 Software

Nearfield modelling was undertaken with the software VISJET 2.5. VISJET is a general interactive computer modelling system that predicts the impact of an effluent discharge into the water environment. The model provides 3D flow visualization of the predicted path and mixing of an arbitrarily inclined buoyant plume in moving receiving water, which may be density stratified.

VISJET was selected for this application because it can simulate groups of inclined buoyant jets in three-dimensional space. It therefore lends itself to application to the proposed Project outfall design to support environmental impact assessments and risk analyses of the return water discharge.

The computational engine in VISJET adopts the Lagrangian model JETLAG (Lee and Cheung 1990) to predict the mixing of buoyant jets with three-dimensional trajectories. The model simulates the key physical processes expressed by fluid motion, heat and mass transport governing equations. The unknown jet trajectory is viewed as a series of non-interfering “plume-elements”, which increase in mass as a result of shear-induced entrainment (due to jet discharge) and vortex-entrainment (due to crossflow); while rising or falling by buoyant acceleration. The model tracks evolution of the average properties of a plume element at each step by conservation of horizontal and vertical momentum, conservation of mass accounting for entrainment, and associated conservation of tracer mass and heat.

The vortex entrainment is determined by a heuristic Projected-Area Entrainment (PAE) hypothesis originally proposed by Frick (1984) for buoyant jets with 2D trajectories, while pressure drag is ignored. A treatment of the transition from the jet/plume-dominated to the ambient current-dominated regime is included, with specific formulations for shear entrainment, providing better representation of the plume trajectory and dilution in both co- and cross-flow conditions (Lee et al., n.d.).

Further information on model formulation and experimental validation is provided in Lee and Cheung (1990) Lee et al. (n.d.) and Lai et al. (2011).

4.2 Model Set-up

4.2.1 Diffuser Configuration

Set-up in the model comprised an outfall line for each diffuser port. These lines were set up to be oriented perpendicular to the tidal flow direction so diffuser ports could be accordingly aligned to the flow direction. 1.0 m radius risers with 3.0 m height were then set up to lodge each of the alternating ports. The distance between manifolds was adjusted in such a way to ensure the 12.0 m spacing between ports was preserved. Each of the ports had an inclination of 45° from the vertical. The overall set-up is illustrated in Figure 4-1.

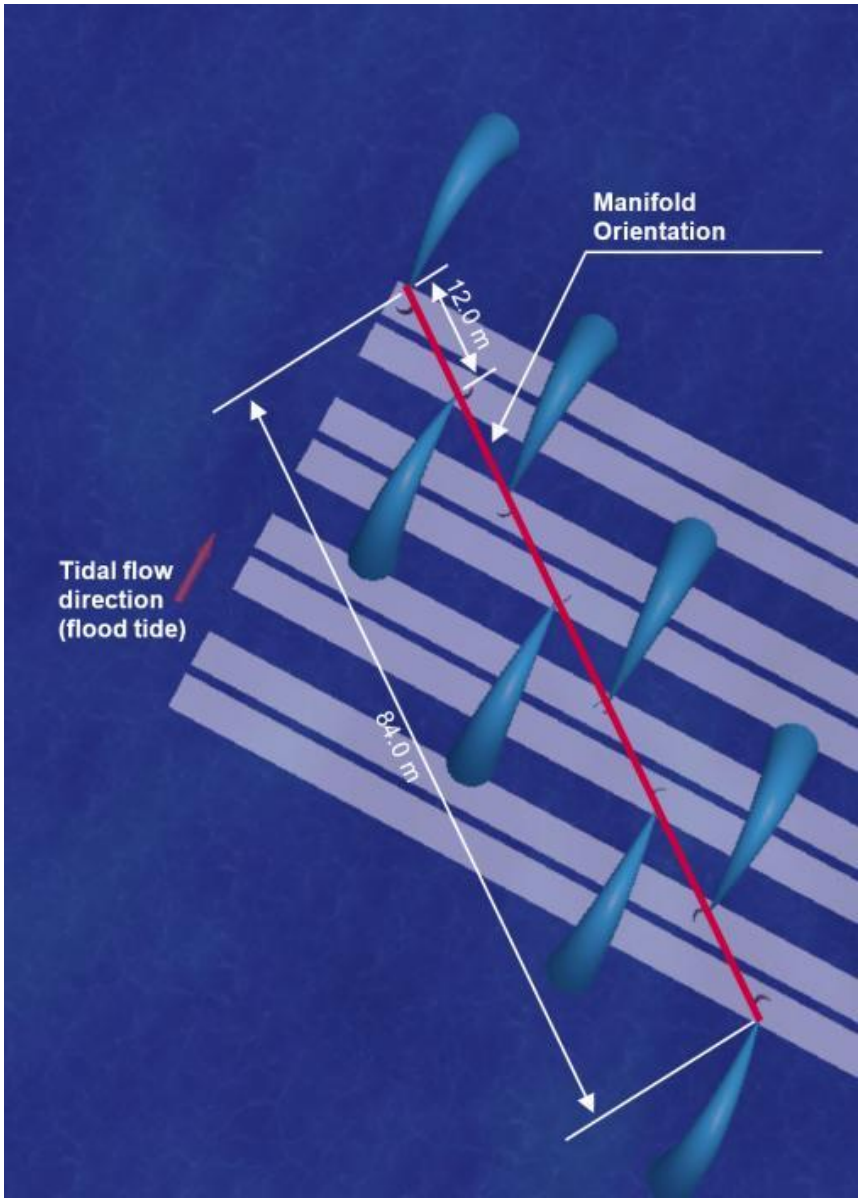


Figure 4-1 Diffuser configuration set-up (plan view)

4.2.2 Ambient Conditions

Ambient data to derive receiving marine environment conditions was collected in the region of the outfall at NWP3 station (~2.7 km offshore of the outfall) and SAMUSG (approximately 31.5 km NE of the outfall). The station locations are shown in Figure 1-1. Available data includes:

- Acoustic Doppler Current Profiler data at NWP3 (Figure 4-2);
- Continuous salinity and temperature data at approximately 1.0 m from the seabed at both NWP3 and SAMUSG (Figure 4-3 and Figure 4-4); and
- Nearbed *In-situ* TSS data at the proposed outfall site collected as part of the marine water quality baseline data collection program (Table 4-1).

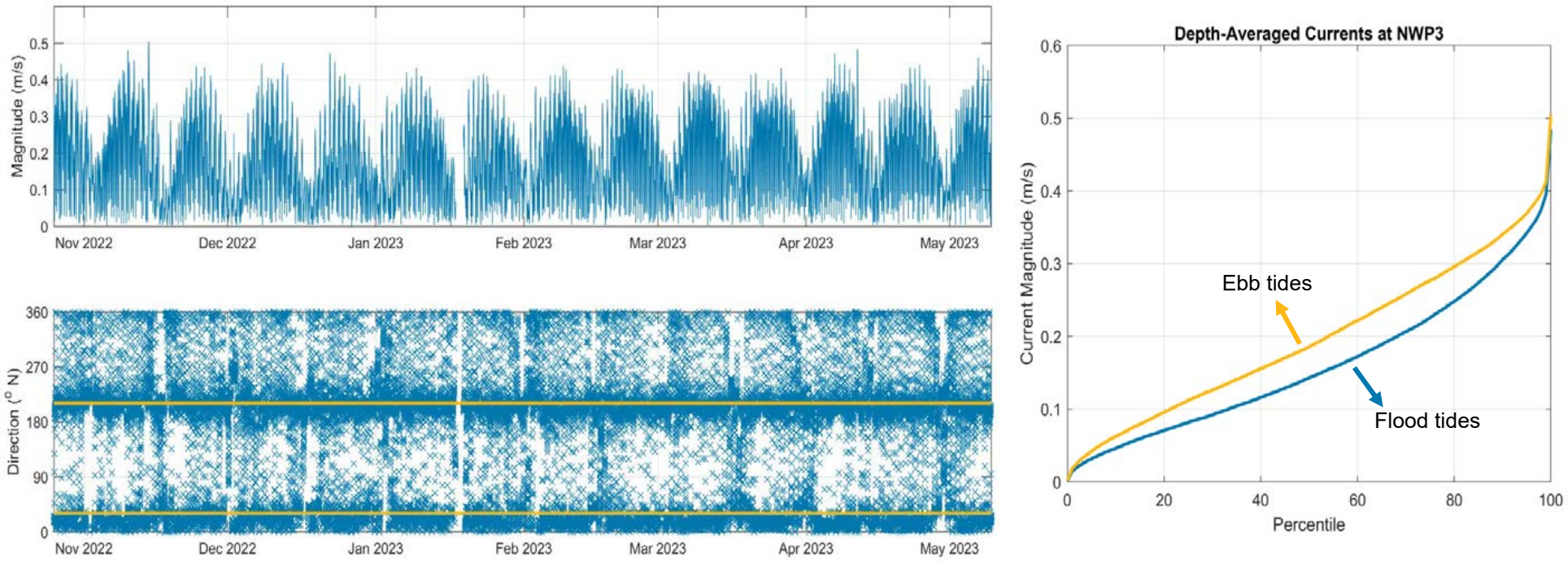


Figure 4-2 Measured depth-averaged ADCP currents at NWP3 station (2022-2023 data)
Left panels: Current magnitude and direction timeseries. Yellow lines indicate the average directions adopted in nearfield modelling. Right panel: Current magnitude distribution for different tidal conditions

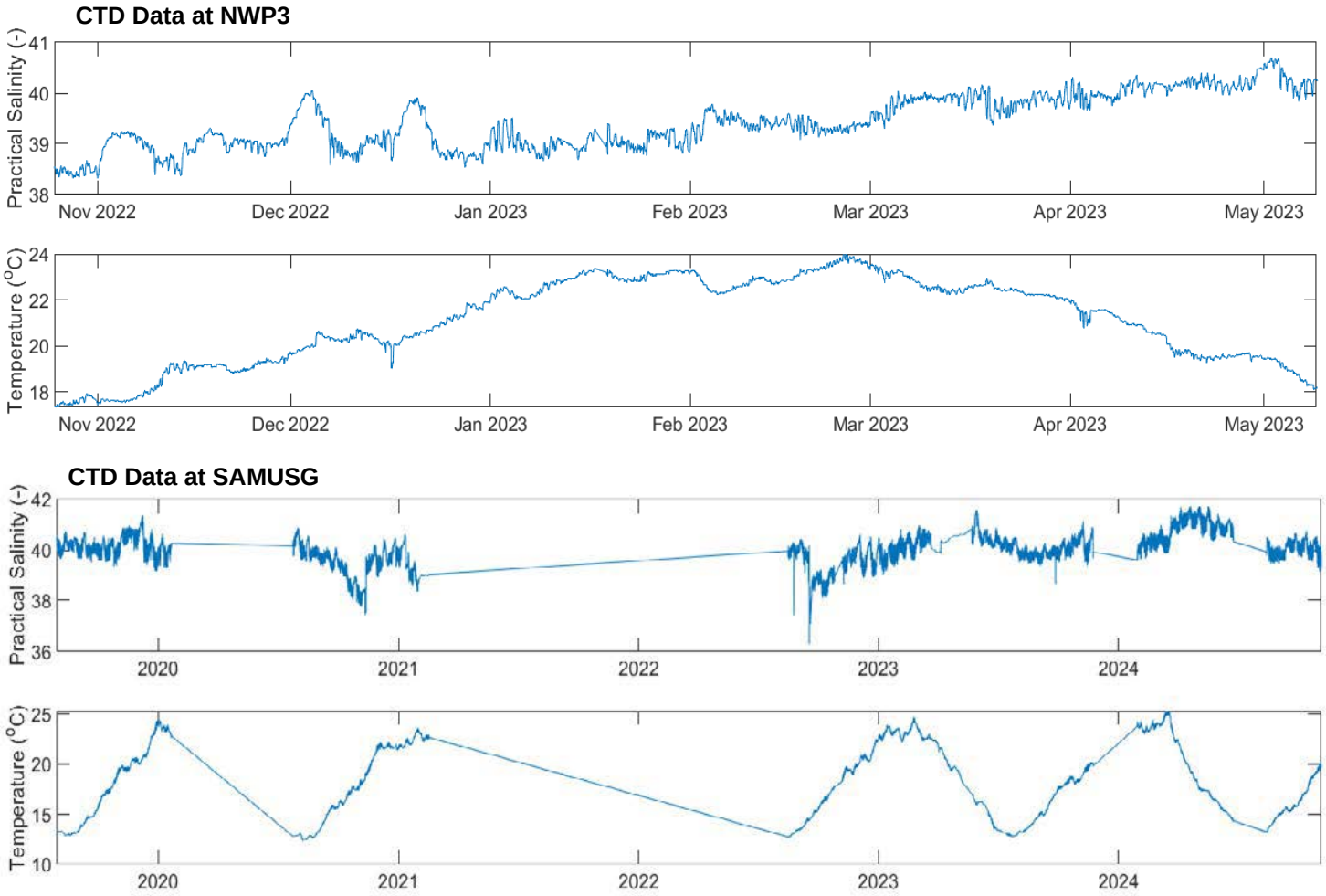


Figure 4-3 Continuous salinity and temperature data at NWP3 (2022-2023 data) and SAMUSG (2019-2021; 2023-2024)

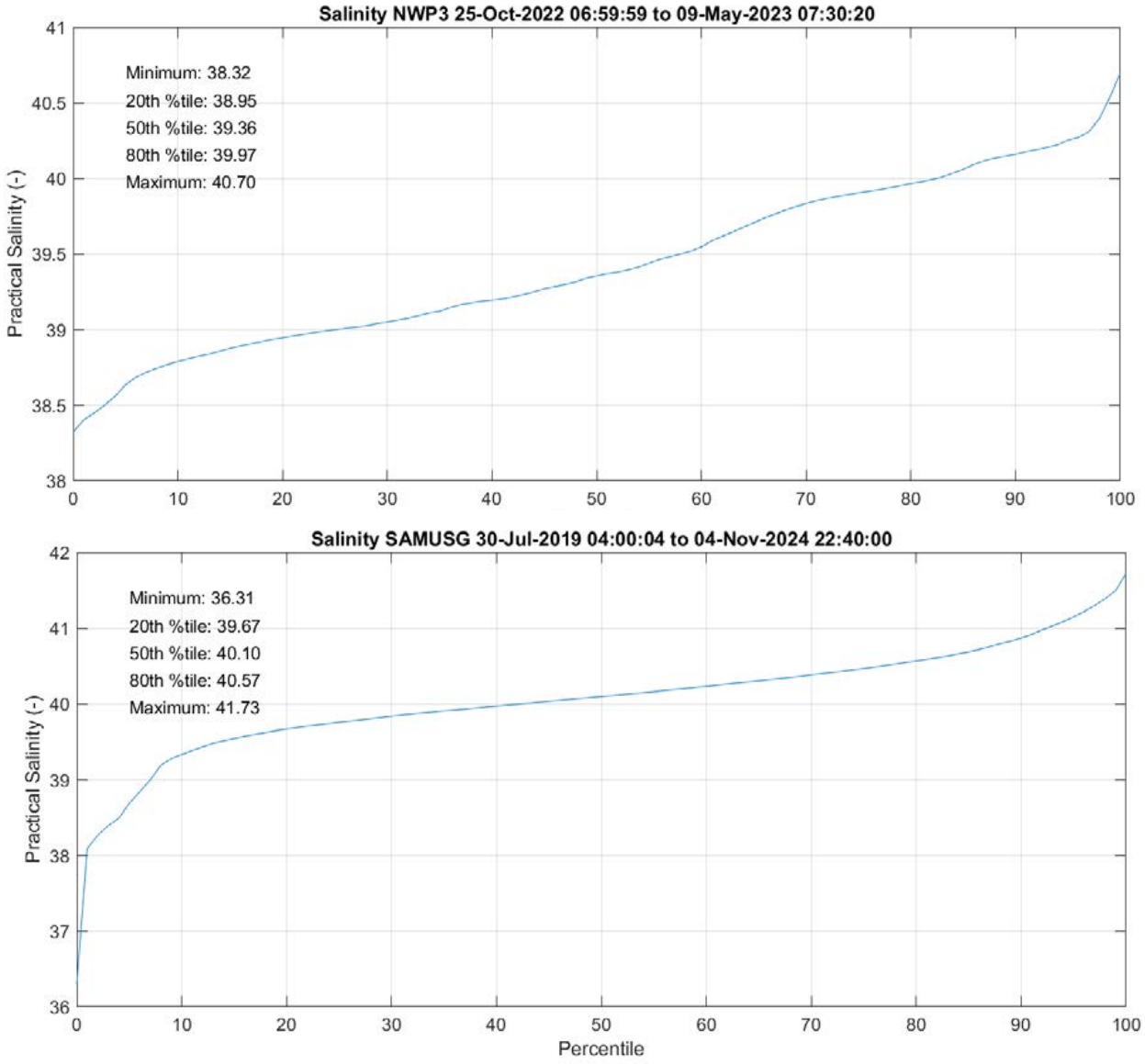


Figure 4-4 Cumulative distribution of salinity data at NWP3 (2022-2023 data) and SAMUSG (2019-2021; 2023-2024)

Statistics of the *in-situ* TSS data are provided in Table 4-1.

Data presented in this section were used to derive ambient conditions for the project area and adopted in the present assessment as summarised in Table 4-2.

In the model, uniform conditions were assumed for the water column, which is consistent with recently collected data. The model was set-up based on densities in preference to the use of temperature and salinity. In this case, the maximum density was selected, as it offered the largest differential between effluent and ambient conditions. As a result (due to the direct relationship between Froude number and dilution), the simulated cases reproduce maximum salinity effects and lowest mixing potential from the diffuser. The model therefore provided conservative or worse case assumptions for impact assessment purposes.

Table 4-1 Statistics of *in-situ* baseline TSS data

TSS Data Statistics	Values at Outfall (mg/L)	
	Bottom Water	All Data
Minimum	3.6	2.4
Maximum	19	26
Mean	11	7.3
Standard Deviation	5.7	5.2
95% Upper Confidence Level	4.4	1.5
50% Percentile	12	4.9
90th Percentile	17	14

Table 4-2 Ambient conditions assumed in nearfield modelling

Parameter	Range	Note
Current magnitude	0.00 to 0.50 m/s	For both ebb and flood tides
Predominant current directions	211 °N (ebb tides) 31 °N (flood tides)	From 2023-2024 data
Temperature	26 °C	Upper estimate for NWP3 and SAMUSG data
Salinity	41	Upper estimate for NWP3
Density	1027.6 kg/m^3	Calculated from TEOS-10[1] for temperature and salinity provided above
Total suspended solids	7 mg/L 17 mg/L	Mean (all data) and 90th (bottom water) percentile of marine water quality baseline data

4.2.3 Simulation Scenarios

Simulation scenarios were constructed for three different discharge conditions, as summarised in Table 4-3. Ambient density data in Table 4-2 was adopted.

Table 4-3 Conditions in simulated scenarios

Scenario	N. of Diffuser Ports	Ambient Density (kg/m^3)	Effluent Density (kg/m^3)	Total Outfall Flow (MLD)	Port Flow (m^3/s)	Ambient Velocity (m/s)
73 GL/year Capacity	12	1027.6	1049.1	229	0.265	0.00, 0.05, 0.10, 0.15, 0.20, 0.25, 0.30, 0.35, 0.40, 0.45 and 0.50
51 GL/year Capacity	10	1027.6	1049.1	286	0.289	
Minimum Capacity	6	1027.6	1040.4	92	0.178	

[1] Thermodynamic Equation of Sea Water 2010

4.3 Environmental Criteria

Applicable dilution criteria were derived from ecotoxicological tests for the protection of 90%, 95% and 99% of species (Van Dam, 2025). These are referred to as 90%, 95% and 99% Species Protection Trigger Values (SPTVs). The SPTVs are to be achieved at an appropriate distance from the diffuser to ensure environmental protection. Dilution SPTVs are provided below:

- 90% SPTV: 19 to 1
- 95% SPTV: 26 to 1
- 99% SPTV: 53 to 1

For return water discharges, primary toxicity is due to the effects of salinity on aquatic organisms. However, additional toxicants (e.g., biocides, anti-scalants) are added to the effluent stream to condition the influent and avoid accumulation of microorganisms in the membranes and pipelines. For salinity, derived dilutions should be compared with median concentrations (over time) (ANZG 2018). For the combined effects of salinity and additional constituents in the return water, no guideline is available - such that criteria are derived from ecotoxicological tests (undertaken by others). For toxicants, as in the case here, the 95th percentile of concentrations over time (or 5th dilution percentile) are used for comparison against the SPTVs (ANZG 2018). A distance of 100 m is often used as a criterion for nearfield dilutions to be achieved.

4.4 TSS Analysis

TSS changes associated with operation of the desalination plant were estimated based on modelled dilutions, as follows:

$$\Delta TSS = \frac{TSS_o - TSS_a}{S}$$

where ΔTSS is the TSS change resulting from the desalination plant discharge, TSS_o is the discharge TSS, TSS_a is the ambient TSS and S is the dilution at the point of contact with the seabed calculated by VISJET. TSS_o and TSS_a adopted the values for both lower and upper ranges of TSS presented in Table 2-1, Table 4-1 and Table 4-2.

This assessment neglects any settling of the solids and conditions prior to contact with the bed. The TSS concentrations represent conditions within the nearfield and are therefore conservative from a water column impact perspective. The assessment does not quantify TSS deposition and/or resuspension.

5 SIMULATION RESULTS

5.1 General Behaviour

Model results showing 10:1, 30:1 and 50:1 iso-surface dilutions for a range of simulated ambient conditions are presented in Appendix B. The iso-surface dilution presentation style was adopted to reveal the shape of the plume in three dimensions. VISJET predictions show the effluent plume tended to deflect towards the direction of the ambient flow within the receiving environment (i.e., towards the northeast or southwest, depending on ambient conditions). For all scenarios, separate plumes from ports adjacent to each other did not interact.

For the minimum production capacity (turn down) scenario, the size of the plumes was smaller than the other high flow scenarios and were more readily influenced by ambient currents. As ambient velocities become stronger, the plumes bent in the direction of the ambient flow. The folding of the plumes resulted in less efficient mixing, which however was counteracted by shorter plume excursions. As ambient currents reached 0.15 m/s to 0.20 m/s, depending on scenario, ports pointing in the opposite direction to the flow collapsed on themselves. The lesser degree of mixing was manifested by thicker plumes (i.e., these plumes undergo a slower rate of dilution) as shown in Appendix B. While the degree of mixing reduced, plume excursions were relatively short. As currents increased, mixing intensified and the plumes experienced longer excursions away from the diffuser.

5.2 Stagnant Conditions

Results of the zero-ambient velocity case were compared against scaling derived from Abessi and Roberts (2014, 2015) laboratory measurements (Table 5-1). Note that S_i in VISJET was obtained from the average dilution at the point of impact with the seabed, while the scaling provides the result at the level of the ports (i.e., 3.0m above the seabed). The scaling analysis for the stationary case provides first estimates of the minimum degree of mixing obtained in areas proximate to the regulatory mixing zone.

5.2.1 End of Nearfield

While dilutions at the end of the nearfield are of relevance to this assessment, VISJET did not provide results at that location, as the computations are interrupted at the location the plume intersects with the seabed before it reaches the end of nearfield location, as provided by the laboratory scaling. Further, the end of the nearfield is possible to obtain from laboratory measurements, because it relates to the location where turbulent eddies significantly reduce their intensity (Abessi and Roberts 2014). On the other hand, the turbulent fluctuations are not directly calculated by VISJET or other industry-standard nearfield models, as such its location from nearfield simulations cannot be obtained. However, the comparisons shown in Table 5-1 for areas within the nearfield (i.e., closer to the diffuser) show that VISJET generally calculated lower values for the plume terminal height, and higher values for both the return point and dilution.

For the turn down scenario, the end of the nearfield was considerably shorter (33 m) and while dilution (29 to 1) was lower compared to the other scenarios, it did achieve the 95% SPTV environmental criteria under stagnant conditions. The 51 GL/year and 73 GL/year production capacity scenarios achieved dilutions higher than the 99% SPTV at distances less than 100 m from the diffuser. Again, VISJET did not provide results at the end of the nearfield. The scaling (SCA) results indicate the plumes will meet the 99% SPTV at the end of the nearfield for both 51 GL/year and 73 GL/year production capacity scenarios but not for the turn down scenario. It is likely that the 99% SPTV will also be achieved for this scenario, noting the following:

- Outfall discharges are turbulent in nature, so the exact dilution number will not necessarily occur all the time. Abessi and Roberts (2014) reported accuracy of the scaling to be 20%, which indicates a dilution range of 22~34 to 1, the lower range being slightly below the 95% SPTV;

- The spreading layer thickness for the turn down scenario, once the discharge contacted the seabed, was ~2.1 m (following scaling in Abessi and Roberts, 2015), thus the current design favours mixing conditions; and
- The terminal rise (y_t) of the plume for all scenarios was below 13 m (both scaling and VISJET), which is less than the 17.0 m LAT (low astronomic tide) depth. According to the modelling results, surface breaching will not occur, as most ebb tides present a surface elevation above LAT. As the water is unlikely to be completely still for long periods, folding of the plumes by currents (see below) should provide further clearance from the surface.

As demonstrated by the high dilutions achieved under the effects of currents (see Section 5.3), sufficient additional mixing is expected to occur most of the time. Further, the return point location is at the level of the port exit, so there was further room for mixing before the discharge reached the seabed (Table 5-1), although there will be less clearance from the top boundary of the plume.

Table 5-1 Comparison of model results with Abessi and Roberts (2015) scaling. Values in green indicate dilution above the 95% SPTV Level

Production Capacity Scenario	y_t (m)		x_i (m)		Dilution S_i (-)		x_n (m)	Dilution S_n (-)
	SCA	VIS	SCA	VIS	SCA	VIS	SCA Only	SCA Only
73 GL/year	12	7.5	22	18	40	41	69	59
51 GL/year	11	7.2	21	17	38	39	67	57
Minimum (Turn Down)	5.6	6.1	11	14	19	35	33	29

For turn down conditions, there was considerable distance between the end of the nearfield (33m) and the end of nearfield in the other scenarios (65~70 m). As such, for the turn down scenario and from the scaling analysis above alone, it could not be ascertained whether the 99% SPTV dilution criteria would be achieved. Midfield modelling results (presented in a separate report) provided further information to determine this distance (Water Technology, 2026). It is also noted that Froude's number of the turn down conditions (F = 12) was lower than those adopted for the experimental conditions from which the scaling was derived (Froude numbers were above 15 in Abessi and Roberts 2015). As such, for the purposes of this report, scaling (SCA) results need to be seen as an upper limit for dilutions under stagnant conditions.

The reduced discharge load under the turn-down scenario, however, would more likely produce a reduced "footprint" in comparison to the other scenario loads. As such, turn down scenario discharges are expected to be compliant with the 99% SPTV dilution criteria. Midfield modelling confirmed that both the 95% and 99th SPTV criterion will be achieved (Water Technology, 2026).

5.2.2 Point of Return

Dilutions at the point of return obtained from both VISJET and the scaling analysis were equal to or higher than 26 to 1 (the 95% SPTV) for most scenarios (except the scaling for the turn-down scenario). These results, particularly for the 51 GL/year and 73 GL/year production scenarios, are in line with the design intent to achieve dilutions between 30 and 40 at the seabed (Table 5-1). For the turn down scenario, dilution at the point of return was equal to 19 to 1. While the basis of design did not define how often that dilution needed to be achieved, the stagnant case is conservative from an impact assessment perspective, noting the effects of currents increase the level of dilution of the plume. Further, the plume has further opportunity to mix as it falls below the point of return (i.e. the port is at 3.0 m from the seabed).

5.2.3 TSS Concentrations

Predicted TSS concentrations are presented in Table 5-2. By the time the plume reached the seabed, estimated changes in TSS ranged from about 0.1 mg/L (lower estimate) to 0.5 mg/L (upper estimate). These variations are relatively small in comparison to the range of background TSS values obtained from *in-situ* observations (Table 4-1).

Table 5-2 Distance and dilutions when the plume reaches the seabed under stagnant conditions. Values in green are above the 95% SPTV

Production Capacity Scenario	Plume centreline distance from port (m)	Plume average dilution (-)	TSS Concentration change (mg/L)	
			Low Estimate	High Estimate
73 GL/year	18	41	0.20	0.44
51 GL/year	17	39	0.20	0.46
Minimum (turn down)	14	35	0.14	0.32

5.3 Effects of Currents

The VISJET modelling illustrates the effects of currents on plume dilution. As the current magnitude increased, the plumes were thinner for the same iso-dilution surface (Appendix B). Alignment of the ports with tidal flow directions should avoid the level of interaction between plumes.

Average plume dilutions (obtained from the cross-section of its normal plane) were extracted at the locations the plume reached the seabed (Table 5-3 to Table 5-5). In all cases, this distance was less than the 100 m mixing zone defined for the nearfield modelling. As expected, the level of mixing and the distance the plume reached the seabed for the alternating ports were not the same.

As a result of the larger flow rates, dilution (and distance of port to the point of return) was higher (and greater) compared to the stagnant conditions. Only for ports in the opposite direction of flow and currents lower than 0.10 m/s dilutions were below 53 to 1 (the 99% SPTV criterion). The 95% SPTV criterion (dilution of 26 to 1) was achieved for all scenarios as shown in Table 5-3 to Table 5-5. For flow rates above 0.20 m/s, dilutions were well above 100 to 1.

Note that for currents of 0.30 m/s for the 51 GL/year and 73 GL/year production capacity scenarios, VISJET interrupted simulations with plume centrelines above approximately 5.0 m from the port, and as such the dilutions at the final VISJET calculation were reported. Increasing the number of calculation steps for the scenarios did not change the results. While the reasons why simulations were interrupted well before the plume reached the bed are not clear, dilutions are likely to be much larger than those reported for a 0.30 m/s ambient current in Table 5-4 and Table 5-5. For context, results for ambient currents of 0.29 m/s and 0.32 m/s are also presented in Table 5-4 and Table 5-5, providing a dilution of approximately 200 to 1. It follows that the lack of convergence around 0.30 m/s is immaterial given that, in reality, the degree of mixing is not static as assumed in the present analysis and would evolve with the unsteady nature of the local currents (Figure 4-2). Hence, dilutions for the analysed scenarios are expected to achieve the 99% SPTV criteria within 100 m of the diffuser.

For the turn down scenario (Table 5-3), the dilutions were lower compared to the other scenarios but still above the 95% SPTV criterion for 0.10 m/s velocities (ebb tides). Again, the location where the plumes reached the seabed were significantly shorter than 100 m from the diffuser. Given the reduced effluent load, the discharge at the reduced flow rates in the turn down scenarios are expected to comply with regulatory requirements, provided the plume does not accumulate locally. Midfield modelling confirmed that both the 95% and 99% SPTV criterion will be achieved for turn down conditions (Water Technology, 2026).

5.4 TSS Concentrations

TSS concentration changes under the effects of currents were generally significantly lower than for stationary ambient conditions (c.f., Table 5-2 and Table 5-3 to Table 5-5). The range of estimated TSS changes under the effects of currents varied between 0.02 to 0.38 mg/L across all scenarios (assuming the 0.29 m/s results *in lieu* of the 0.30 m/s for ports in the opposite direction of the flow – see above). Again, these variations are relatively small in comparison to typical *in-situ* observations (Table 4-1) and generally reduced as the current magnitudes increase.

Table 5-3 Distance and dilutions when the plume reaches the seabed (minimum turn down scenario). Values in green are above the 95% SPTV

Current (m/s)	Port	Plume centreline distance from port (m)	Plume average dilution (-)	TSS Concentration change (mg/L)	
				Low Estimate	Upper Estimate
0.10	In the flow direction	22	65	0.08	0.17
	Opposite to flow direction	9	55	0.09	0.20
0.20	In the flow direction	33	153	0.03	0.07
	Opposite to flow direction	8	165	0.03	0.07
0.30	In the flow direction	46	218	0.02	0.04
	Opposite to flow direction	29	226	0.02	0.04
0.40	In the flow direction	60	264	0.02	0.04
	Opposite to flow direction	44	264	0.02	0.04
0.50	In the flow direction	57	257	0.08	0.17
	Opposite to flow direction	42	258	0.09	0.20

Table 5-4 Distance and dilutions when the plume reaches the seabed (51 GL/year production capacity scenario). Values in green are above the 95% SPTV

Current (m/s)	Port	Plume centreline distance from port (m)	Plume average dilution (-)	TSS Concentration change (mg/L)	
				Low Estimate	Upper Estimate
0.10	In the flow direction	23	59	0.14	0.31
	Opposite to flow direction	13	47	0.17	0.38
0.20	In the flow direction	31	125	0.06	0.14
	Opposite to flow direction	0.79	142	0.06	0.13
0.30 (0.29; 0.32)	In the flow direction	42	195	0.03	0.07
	Opposite to flow direction	5.0[2] (12; 18)	18 (199; 205)	0.45 (0.03; 0.03)	1.01 (0.07; 0.07)
0.40	In the flow direction	66	246	0.03	0.06
	Opposite to flow direction	47	254	0.03	0.06
0.50	In the flow direction	23	284	0.14	0.31
	Opposite to flow direction	13	283	0.17	0.38

Table 5-5 Distance and dilutions when the plume reaches the seabed (73 GL/year hypothetical high-capacity scenario). Values in green are above the 95% SPTV

Current (m/s)	Port	Plume centreline distance from port (m)	Plume average dilution (-)	TSS Concentration change (mg/L)	
				Low Estimate	Upper Estimate
0.10	In the flow direction	14	61	0.13	0.30
	Opposite to flow direction	13	48	0.17	0.37
0.20	In the flow direction	32	128	0.06	0.14
	Opposite to flow direction	0.59	142	0.06	0.13
0.30 (0.29; 0.32)	In the flow direction	55	201	0.03	0.07
	Opposite to flow direction	4.9[3] (13; 19)	17 (204; 209)	0.46 (0.03; 0.03)	1.03 (0.07; 0.07)
0.40	In the flow direction	68	251	0.03	0.06
	Opposite to flow direction	48	259	0.03	0.06
0.50	In the flow direction	14	290	0.13	0.30
	Opposite to flow direction	13	289	0.17	0.37

[2] VISJET interrupted simulation with plume at 5.5 m from the seabed. Results reported in parenthesis refer to 0.29 m/s and 0.32 m/s ambient current simulations.
[3] VISJET interrupted simulation with plume at 5.2 m from the seabed. Results reported in parenthesis refer to 0.29 m/s and 0.32 m/s ambient current simulations.

5.5 Effects of Waves

The effects of waves on near-field dilution were evaluated based on the laboratory and computational analyses of Antenucci et al. (2025), who proposed a scaling relationship to adjust dilution results obtained under stagnant conditions. Their results are reproduced in Figure 5-1 and were used to undertake a sensitivity assessment of the dilution results presented above in relation to wave conditions experienced at Mullaquana Station. The scaling accounts for the ratio between the wave orbital velocity at the discharge port elevation (u_{wave}) and the port exit velocity (u_{jet}). u_{wave} can be computed as follows:

$$u_{wave} = \frac{\pi H}{T}\frac{\cosh(\kappa h_o)}{\sinh(\kappa D)}$$

where H is the wave height, T is the wave period, κ is the wave number (in rad/m), h_o is the height of the port above the bed, and D is the water depth.

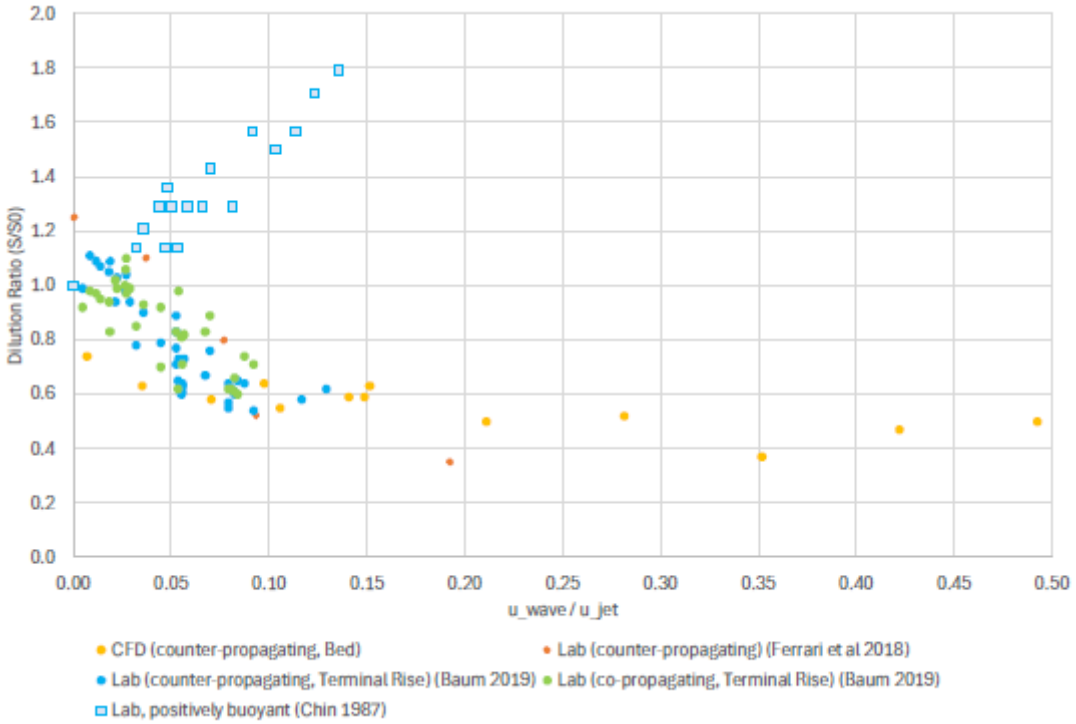


Figure 5-1 Normalised dilution measured at the bed from CFD and laboratory experiments (after Antenucci et al. 2025)

Measurements of significant wave height (H_s), associated peak wave period (T_p) and mean wave direction (D_m) collected at NWP3 (Figure 5-2) between 8 September 2025 and 11 June 2026 were used to obtain wave conditions at the site. The wave number was obtained from the linear wave dispersion relationship:

$$\omega^2 = g\kappa \tanh(\kappa D)$$

where ω is the wave angular frequency and g is the acceleration due to gravity. The wave conditions considered in this assessment are summarised in Table 5-6, assuming a water depth of 17 m.

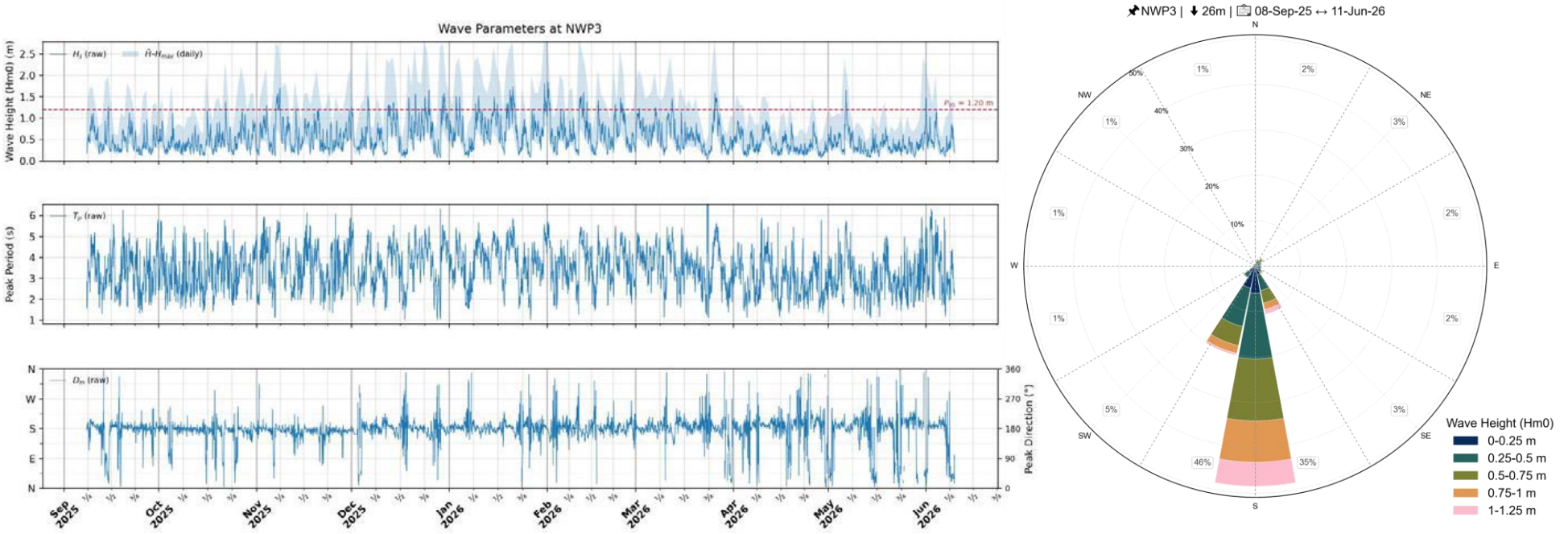


Figure 5-2 Wave measurements at NWP3

Table 5-6 Significant wave conditions considered in this assessment

Wave Condition	H_s (m)	T_p (s)	κ (rad/m)	ω (rad/s)	u_{wave} (m/s)	u_{wave}/u_{jet}
Median H_s	0.47	3.4	0.35	1.85	0.0036	$6e^{-4}$ to $1e^{-3}$
Representative	1.00	4.1	0.24	1.53	0.0329	0.006 to 0.012
Maximum H_s	1.84	5.7	0.13	1.10	0.2429	0.04 to 0.09

Using $\boldsymbol{u_{wave}/u_{jet}}$ values presented in Table 5-6 and the graph shown in Figure 5-1, both the median and representative significant wave conditions are expected to result in dilution performance similar to that predicted for the stagnant case described in Section 5.2. This outcome is expected because Mullaquana Station, located within Spencer Gulf, is sheltered from the larger oceanic swell systems that occur along more exposed coastlines.

Under maximum H_s, representing storm events, effective dilution could potentially decrease to approximately 50-60% of the dilution predicted for the stagnant case However, truly stagnant ambient conditions are unlikely to occur during such events, and the scaling relationship of Antenucci et al. (2025) was not developed for the combined influence of strong waves and elevated currents. Consequently, this estimate should be regarded as a conservative sensitivity test rather than a realistic prediction of storm-time performance. Even if a similar reduction in dilution were to occur, the predicted dilutions at the point of return under the lowest-current scenarios would remain greater than the 95% SPTV criterion.

6 SUMMARY

Analysis of mixing in the nearfield was performed for the outfall proposed for the Northern Water Project. Scaling analysis and nearfield modelling showed that the current location and proposed outfall design will create sufficient mixing to achieve compliance with prescribed environmental criteria. Dilutions were generally above the 99% SPTV criterion, even for some of the low current cases and lowest flow rates. Dilutions below the criterion occurred at distances considerably less than the edge of the nearfield mixing zone (100 m from the outfall). The effects of waves under representative conditions at Mullaquana Station were considered negligible with respect to the dilution outcomes presented in this study. As a result, the proposed discharge is expected to comply with environmental criteria, indicating an acceptable outfall design for ambient conditions at Mullaquana Station. Assessment of the effects of the discharge on TSS were shown to be relatively small in comparison to the range in existing *in-situ* observations. Integration of the nearfield modelling within the midfield modelling of the same discharge scenarios confirmed that this would be the case (Water Technology, 2026).

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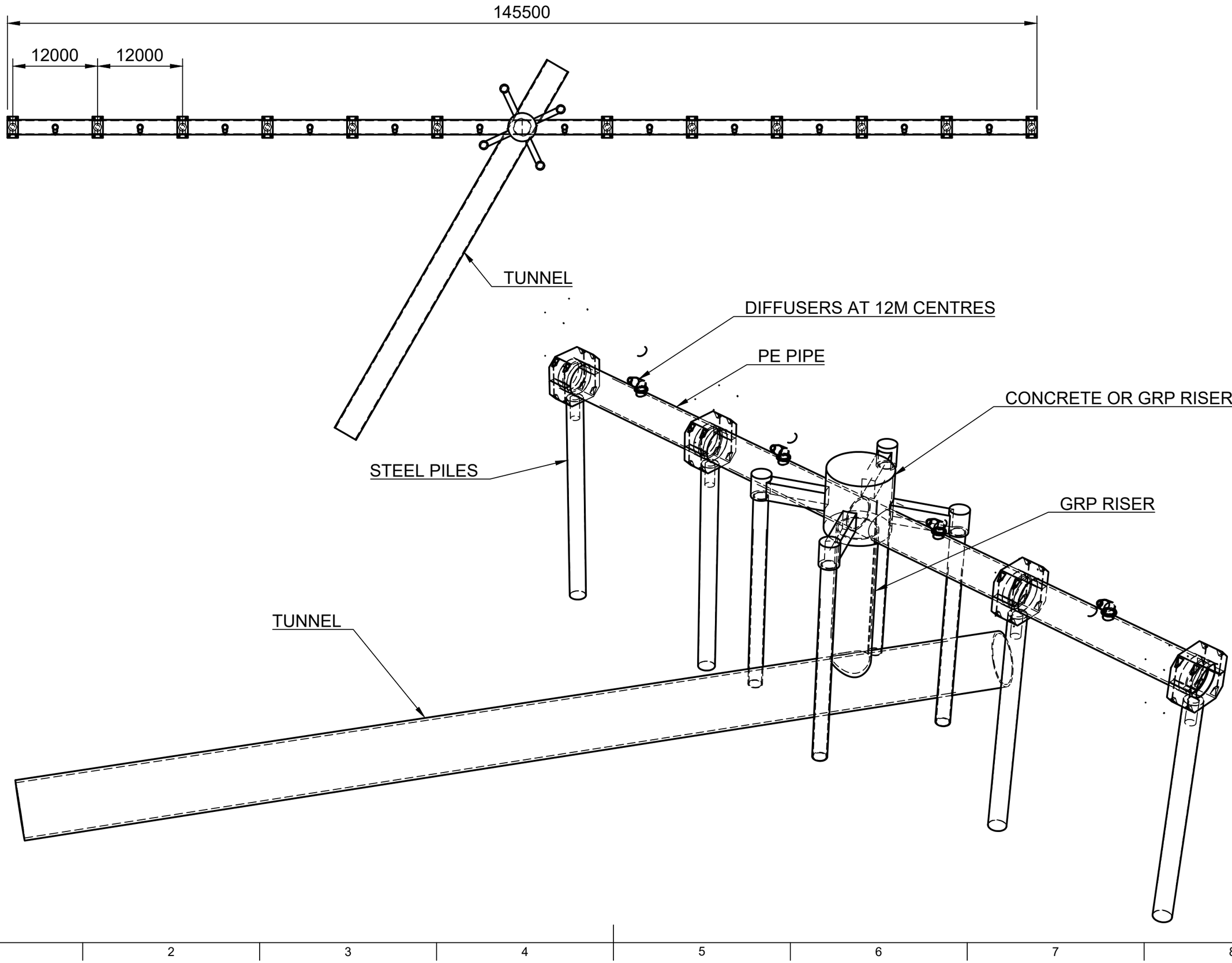
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WATER TECHNOLOGY
WATER, COASTAL & ENVIRONMENTAL CONSULTANTS

APPENDIX A
OUTFALL DESIGN DRAWING


145500
12000
12000
TUNNEL
DIFFUSERS AT 12M CENTRES
PE PIPE
CONCRETE OR GRP RISER CAP
STEEL PILES
GRP RISER
TUNNEL

WATER TECHNOLOGY
WATER, COASTAL & ENVIRONMENTAL CONSULTANTS

APPENDIX B
ISO-SURFACE DILUTIONS



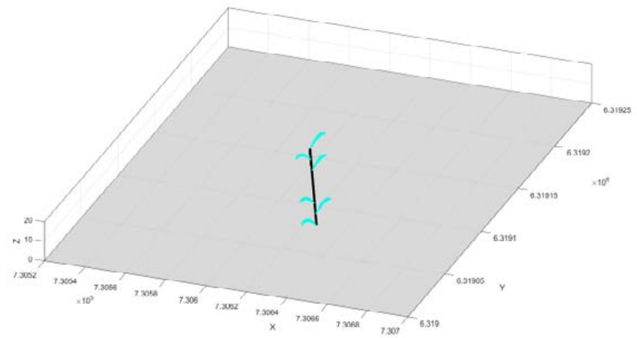
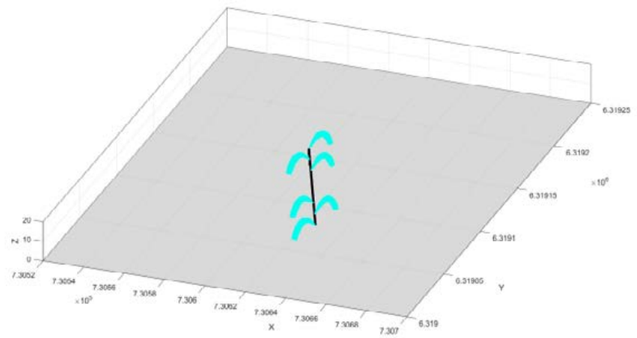
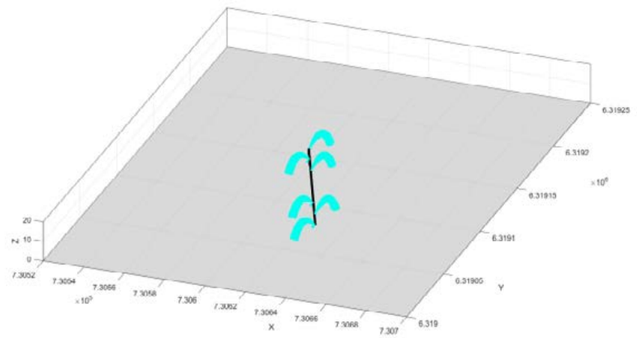
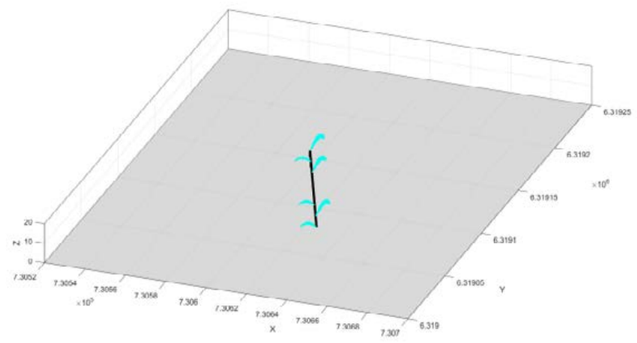
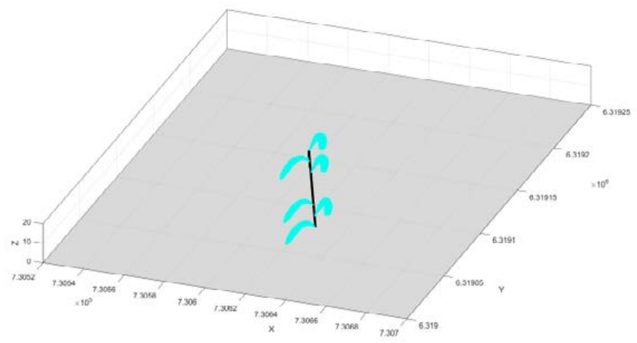
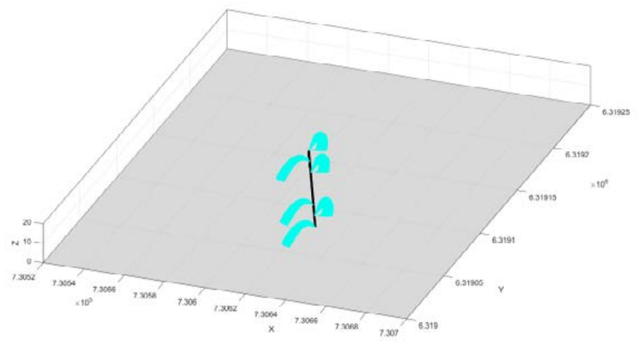
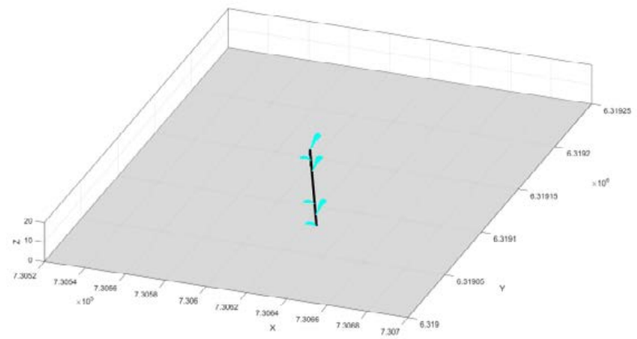
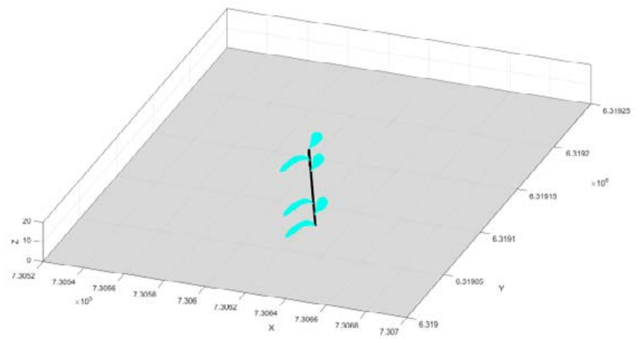
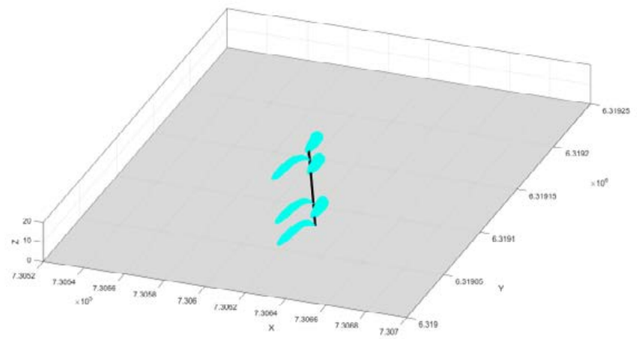
Ambient Velocity	Dilution Iso-Surface (Turn Down)		
	10	30	50
0.0 m/s			
0.1 m/s			
0.2 m/s			

Figure 7-1 Iso-surface Dilutions for Turn Down Scenario (Lower Ambient Velocity Range during Ebb Tides)

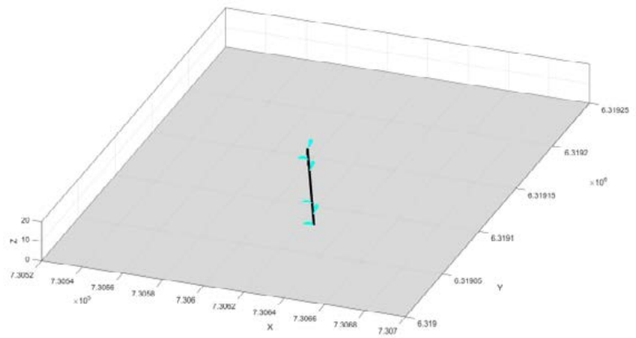
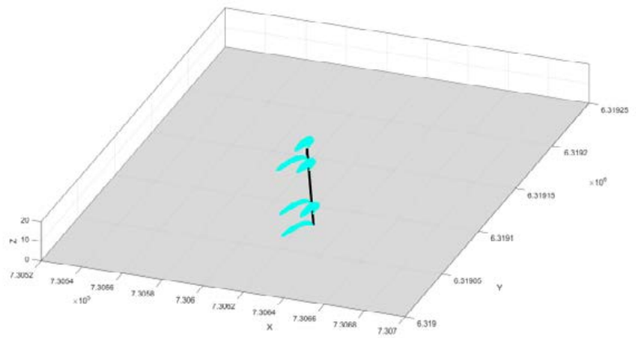
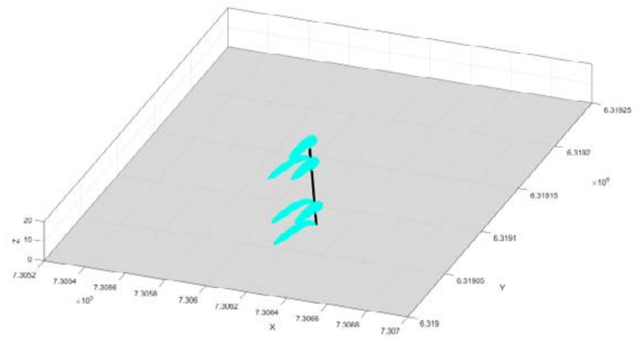
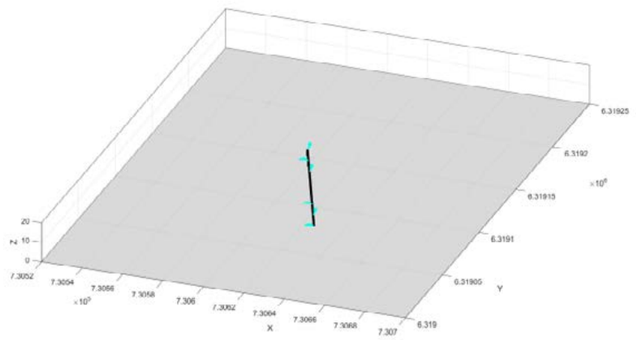
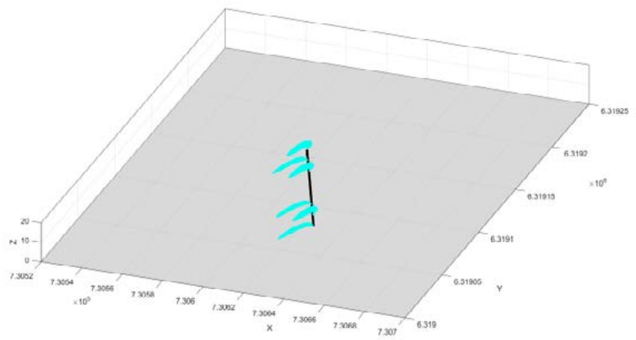
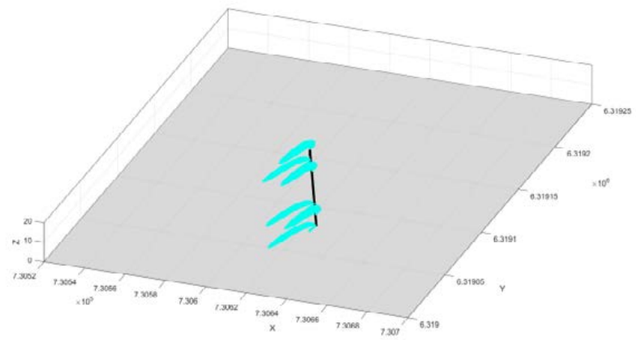
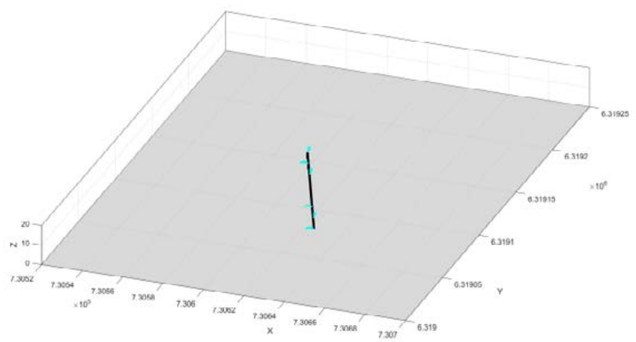
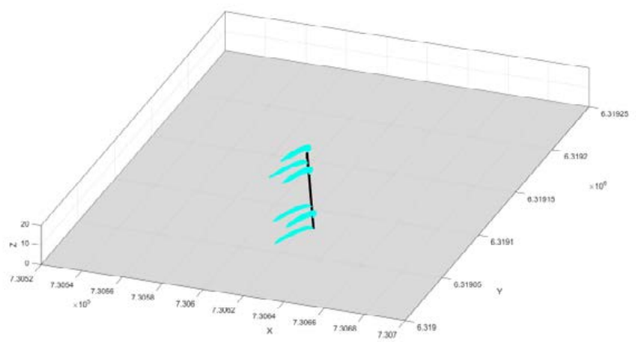
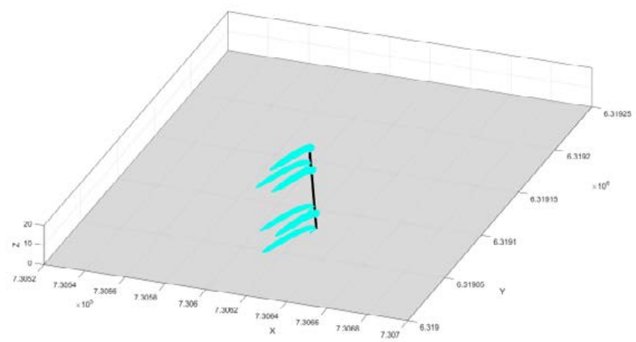
Ambient Velocity	Dilution Iso-Surface (Turn Down)		
	10	30	50
0.3 m/s			
0.4 m/s			
0.5 m/s			

Figure 7-2 Iso-surface Dilutions for Turn Down Scenario (Upper Ambient Velocity Range during Ebb Tides)

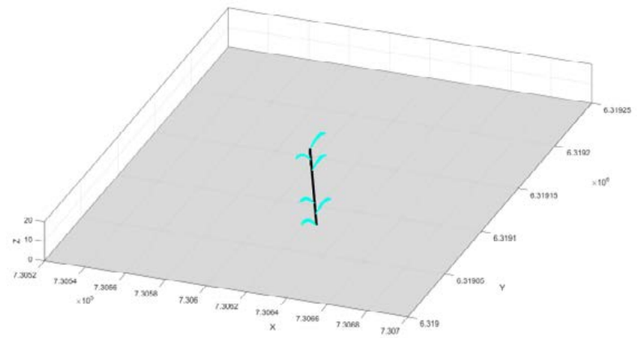
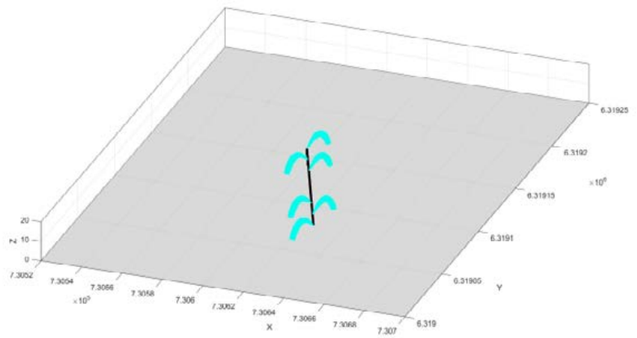
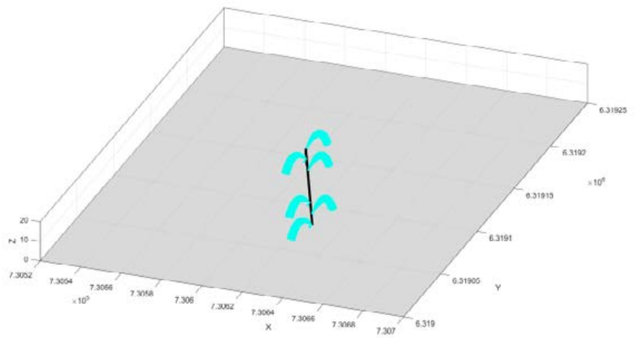
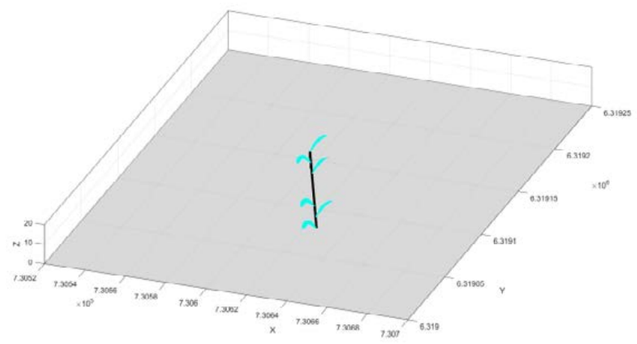
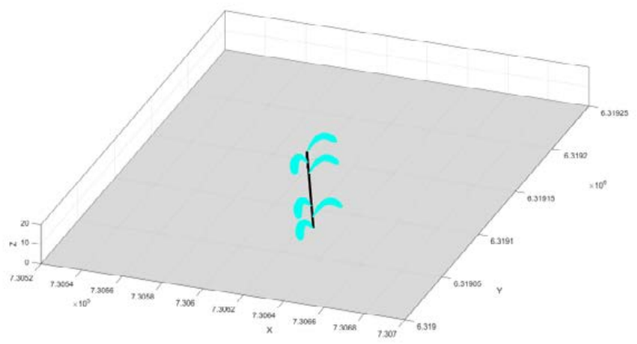
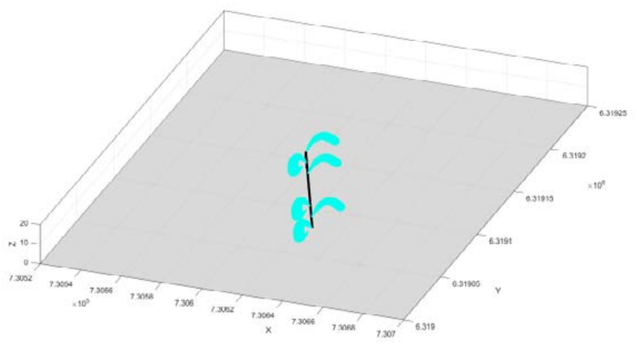
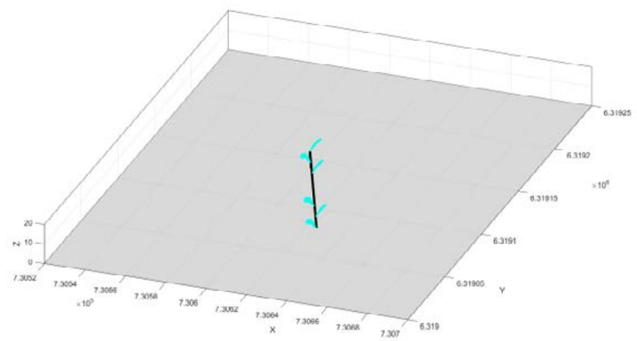
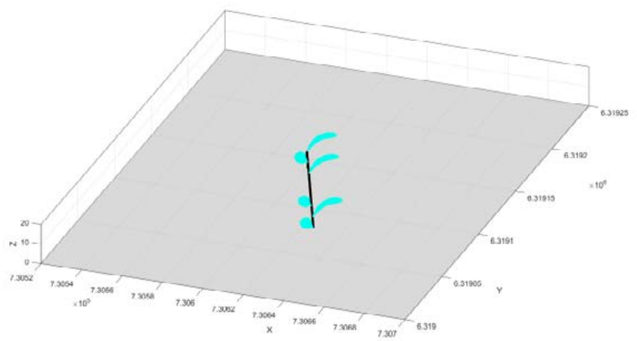
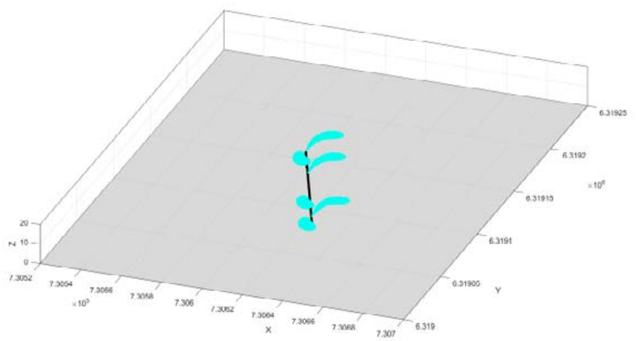
Ambient Velocity	Dilution Iso-Surface (Turn Down)		
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0.0 m/s			
0.1 m/s			
0.2 m/s			

Figure 7-3 Iso-surface Dilutions for Turn Down Scenario (Lower Ambient Velocity Range during Flood Tides)

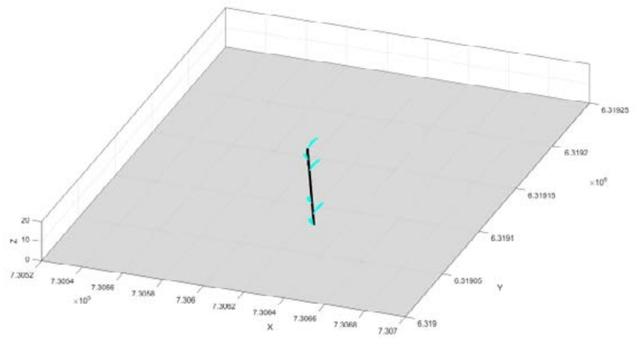
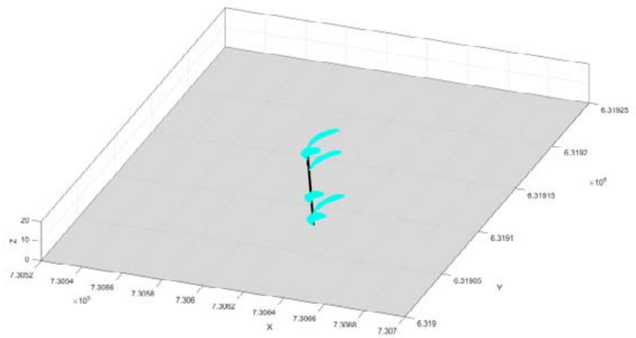
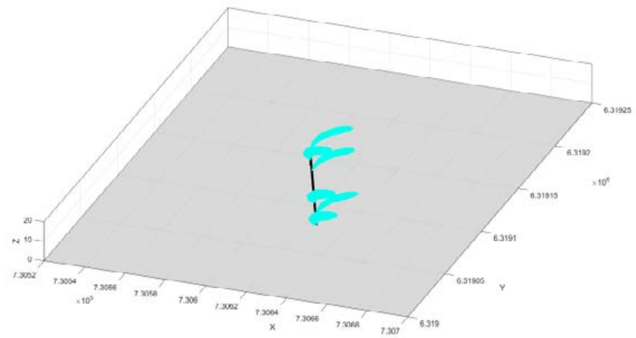
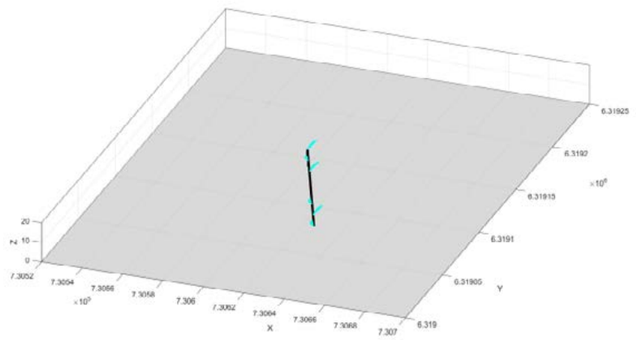
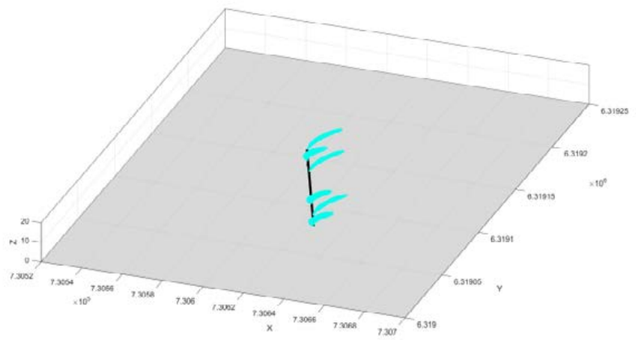
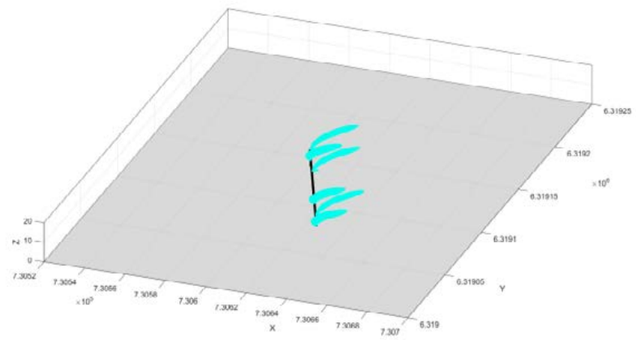
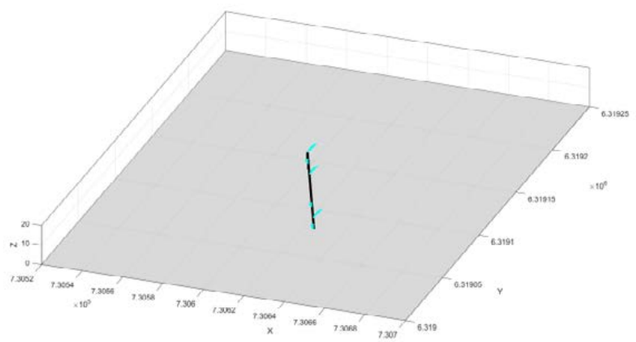
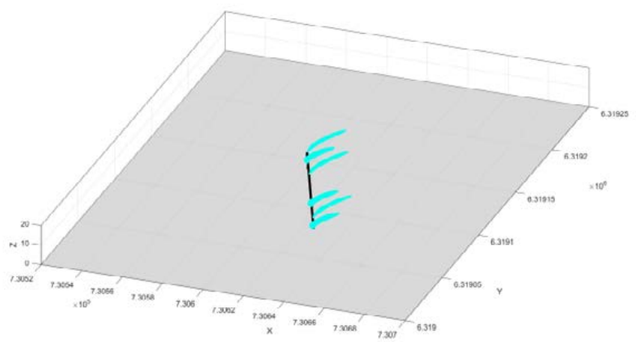
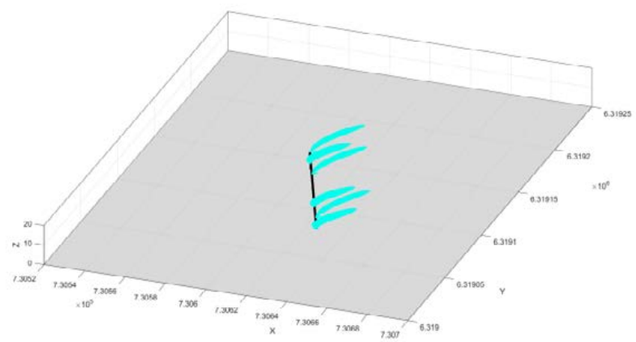
Ambient Velocity	Dilution Iso-Surface (Turn Down)		
	10	30	50
0.3 m/s			
0.4 m/s			
0.5 m/s			

Figure 7-4 Iso-surface Dilutions for Turn Down Scenario (Upper Ambient Velocity Range during Flood Tides)

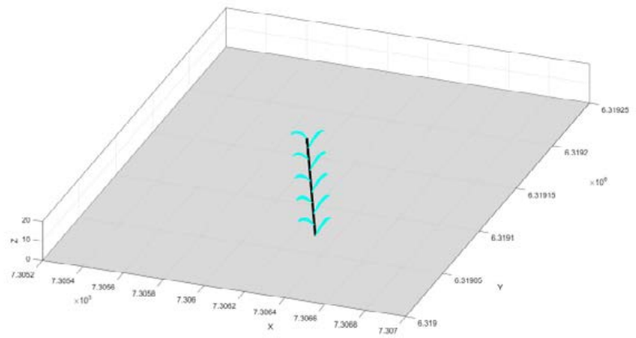
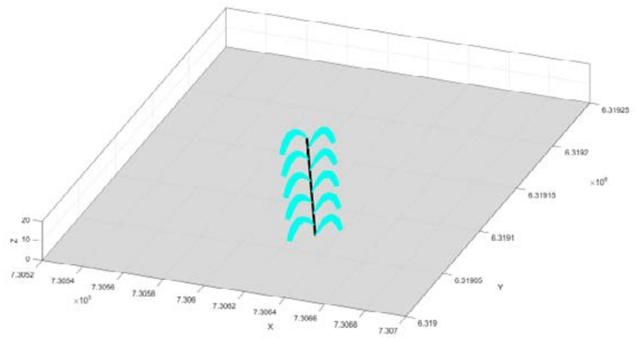
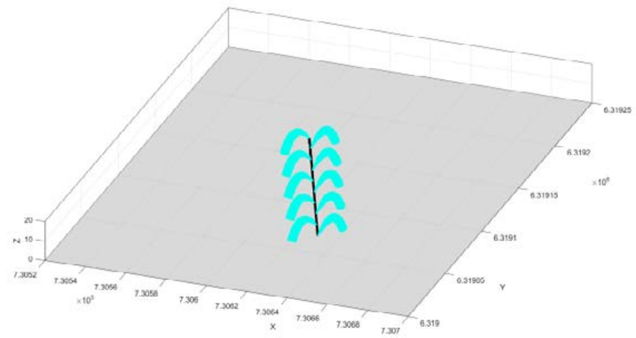
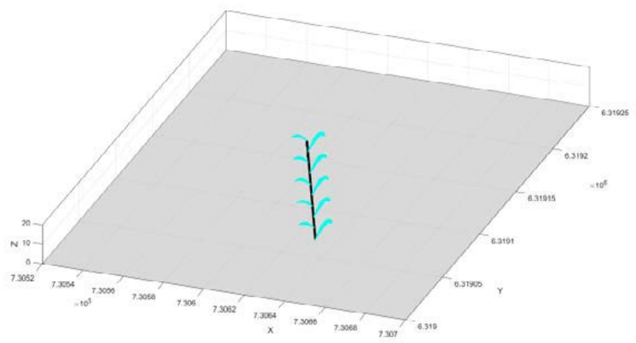
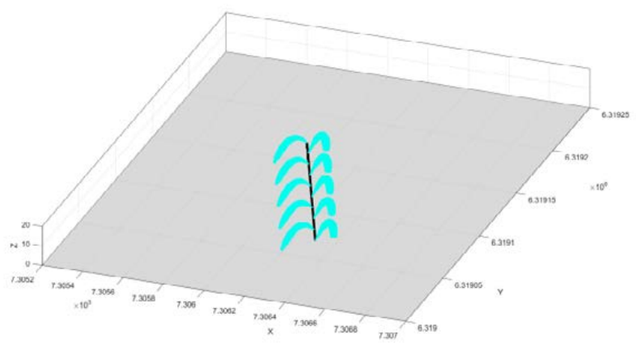
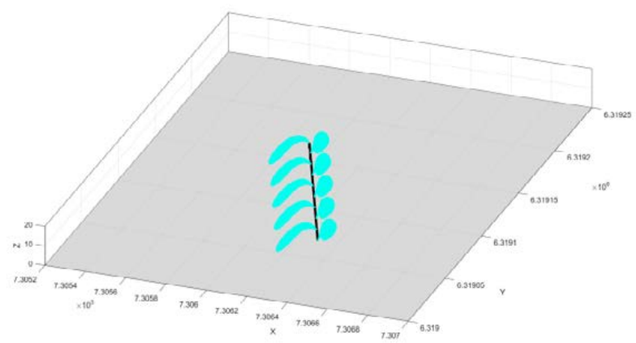
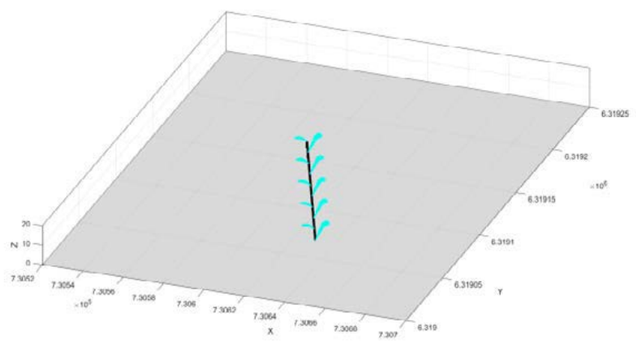
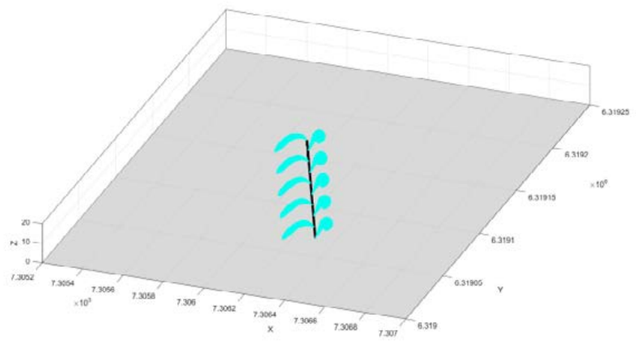
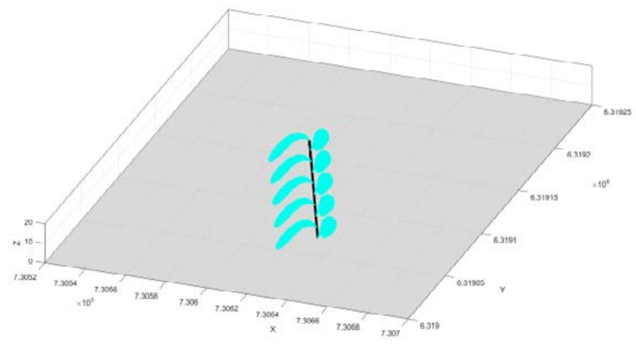
Ambient Velocity	Dilution Iso-Surface (51 GL/year Production Capacity)		
	10	30	50
0.0 m/s			
0.1 m/s			
0.2 m/s			

Figure 7-5 Iso-surface Dilutions for 51 GL/year Production Capacity Scenario (Lower Ambient Velocity Range during Ebb Tides)

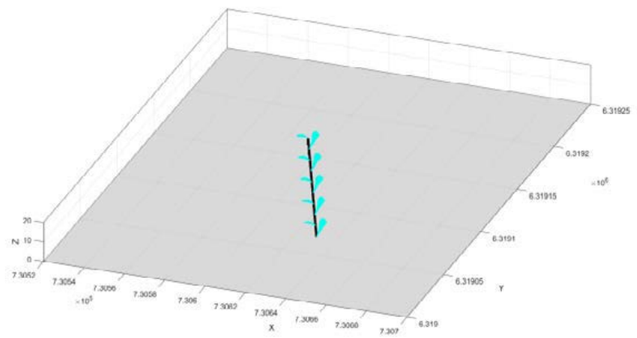
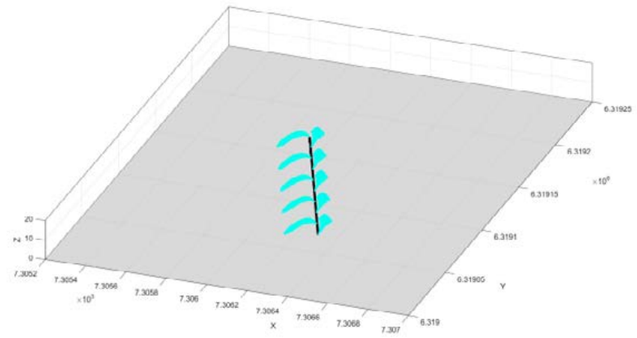
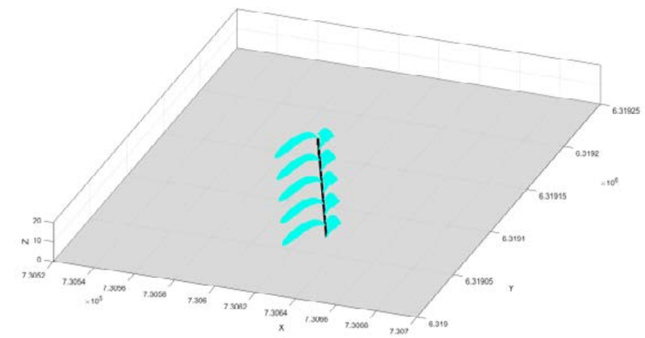
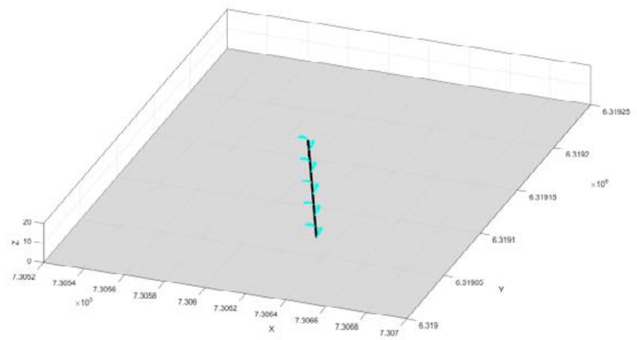
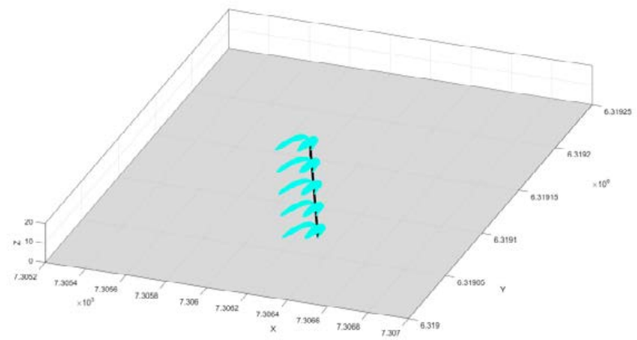
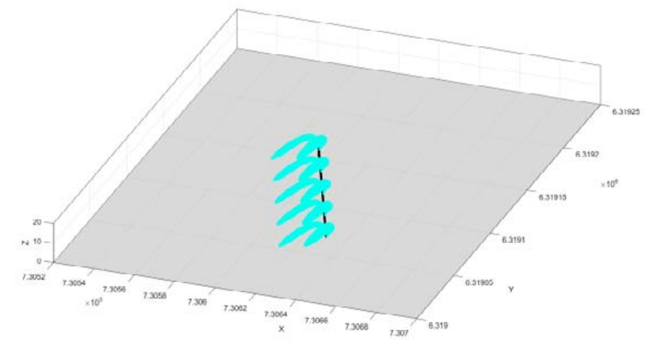
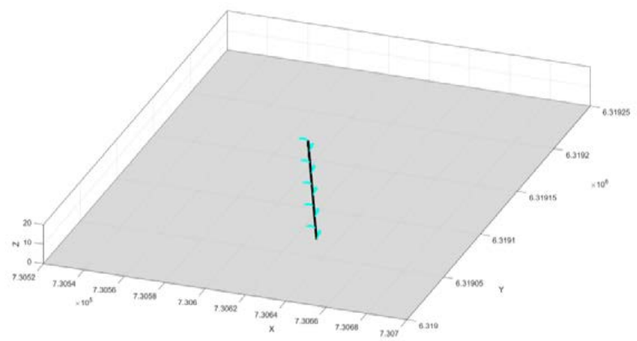
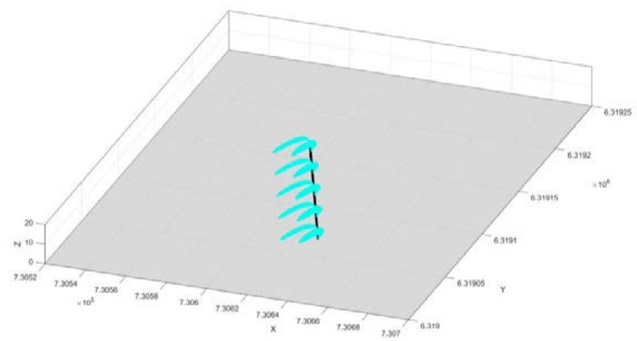
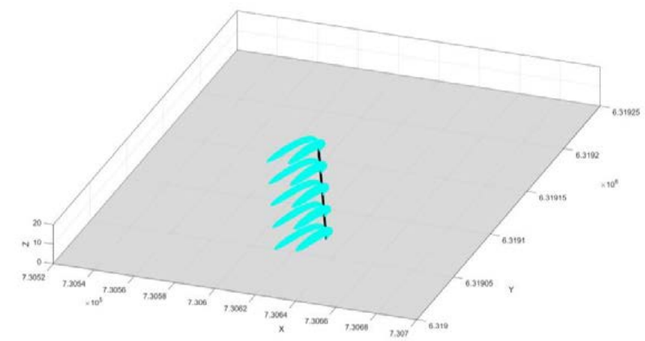
Ambient Velocity	Dilution Iso-Surface (51 GL/year Production Capacity)		
	10	30	50
0.3 m/s			
0.4 m/s			
0.5 m/s			

Figure 7-6 Iso-surface Dilutions for 51 GL/year Production Capacity Scenario (Upper Ambient Velocity Range during Ebb Tides)

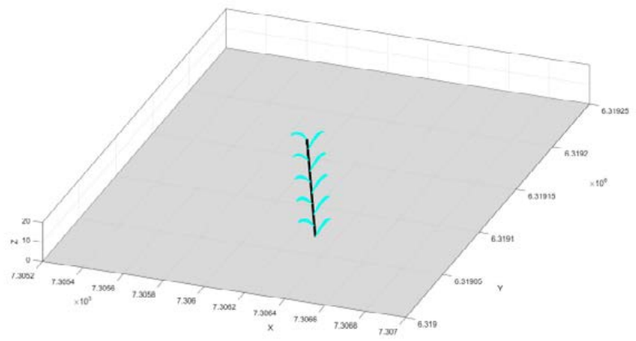
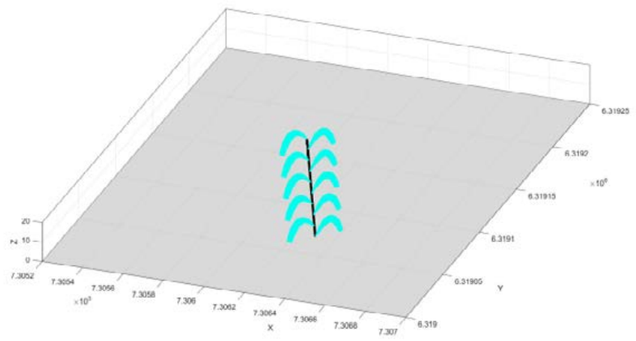
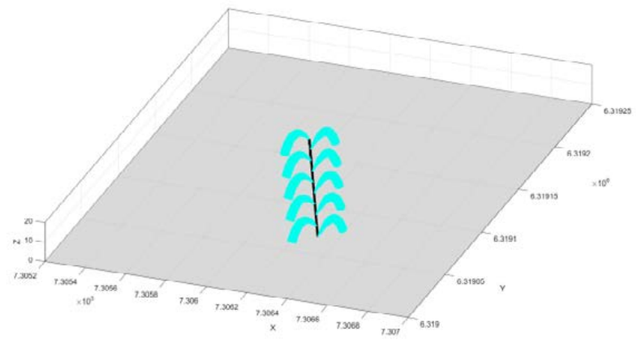
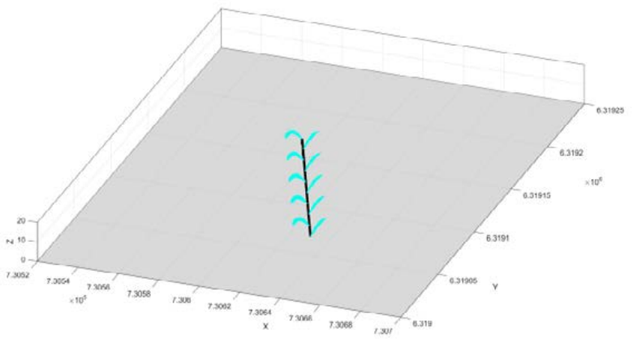
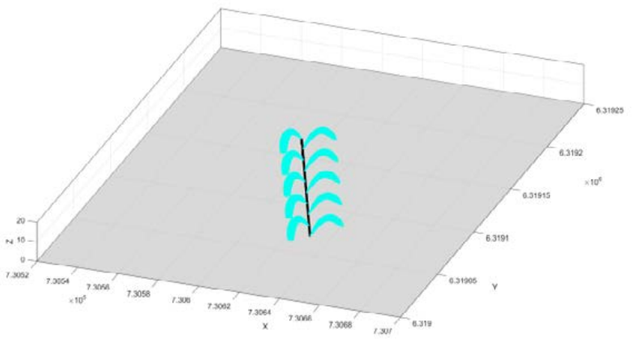
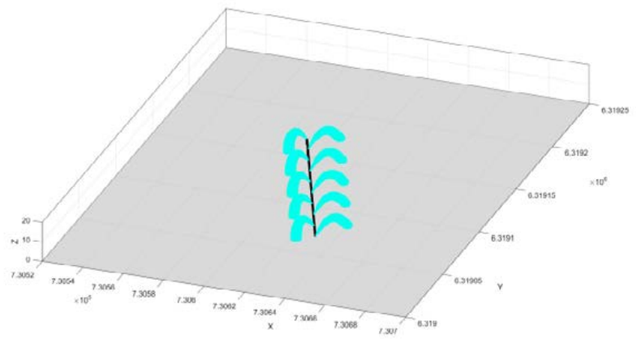
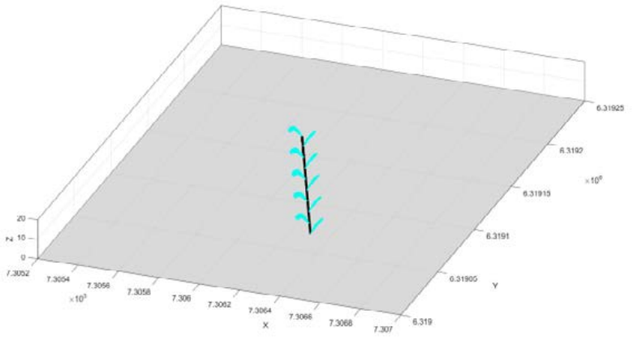
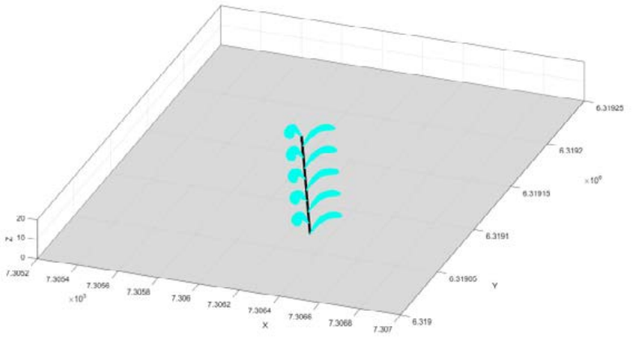
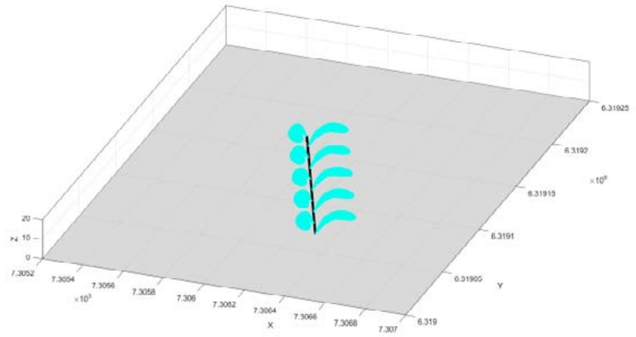
Ambient Velocity	Dilution Iso-Surface (51 GL/year Production Capacity)		
	10	30	50
0.0 m/s			
0.1 m/s			
0.2 m/s			

Figure 7-7 Iso-surface Dilutions for 51 GL/year Production Capacity Scenario (Lower Ambient Velocity Range during Flood Tides)

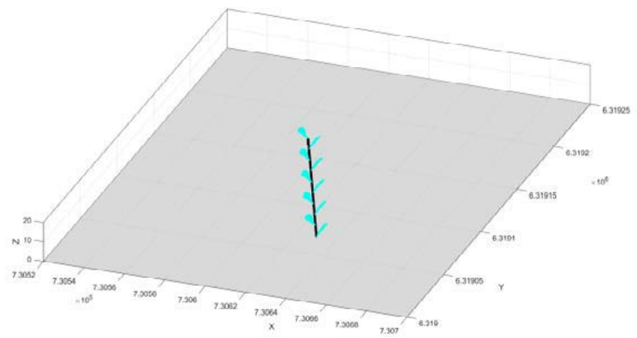
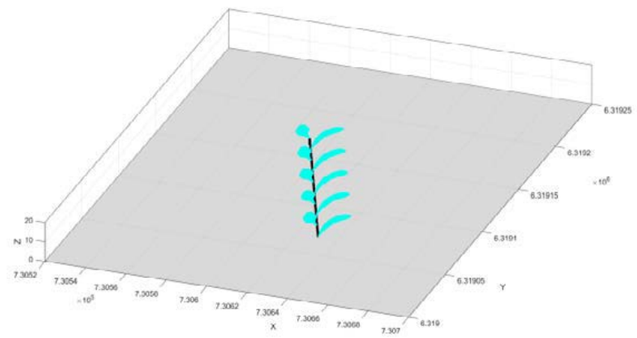
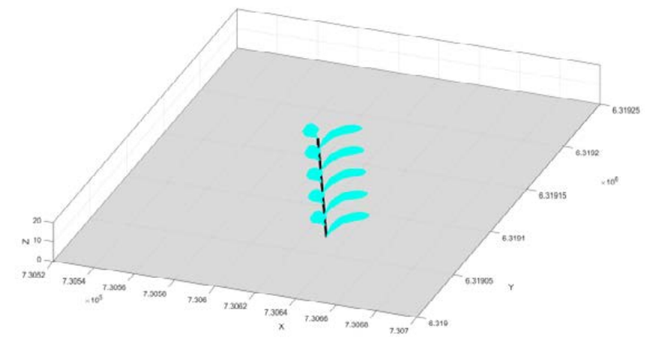
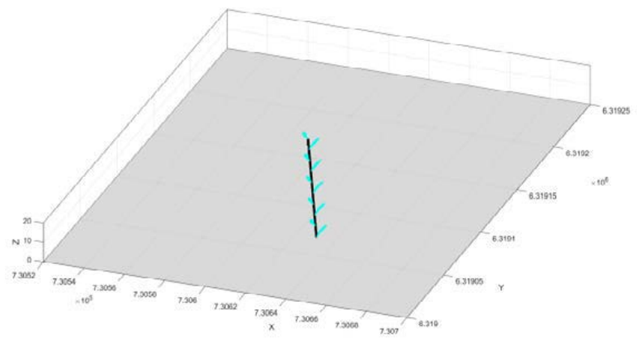
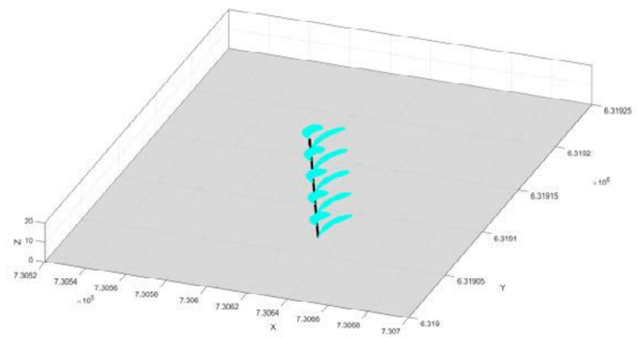
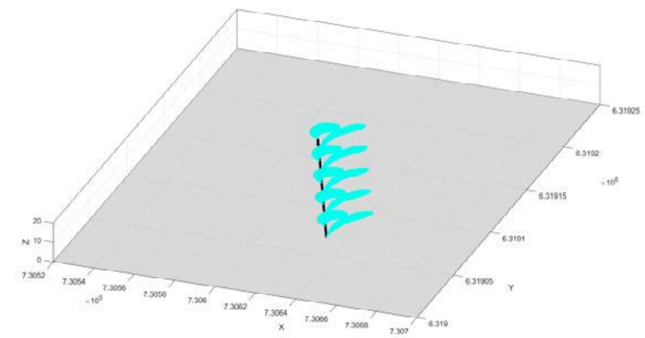
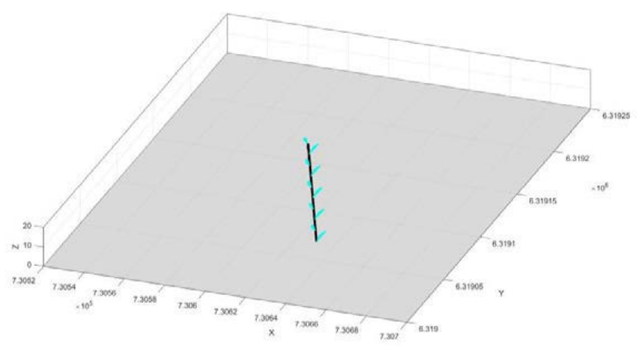
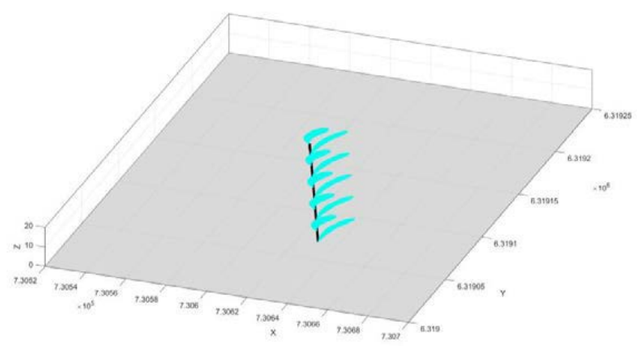
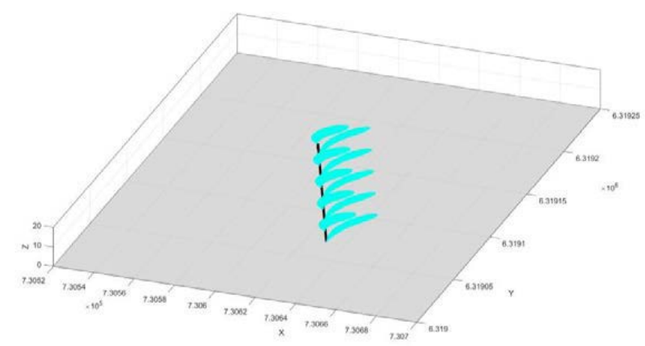
Ambient Velocity	Dilution Iso-Surface (51 GL/year Production Capacity)		
	10	30	50
0.3 m/s			
0.4 m/s			
0.5 m/s			

Figure 7-8 Iso-surface Dilutions for 51 GL/year Production Scenario (Upper Ambient Velocity Range during Flood Tides)

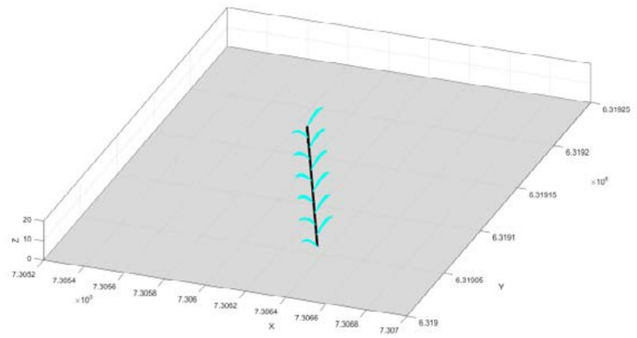
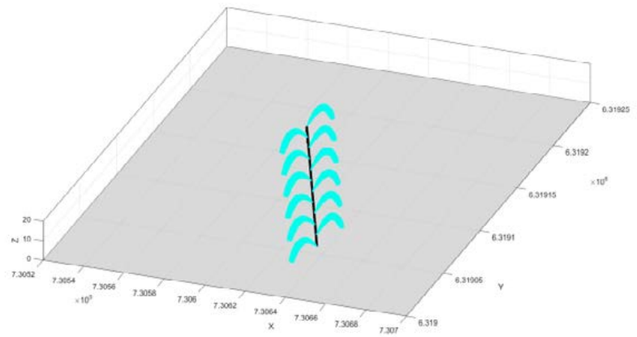
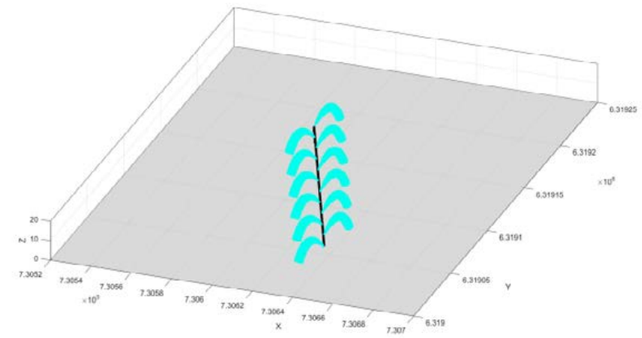
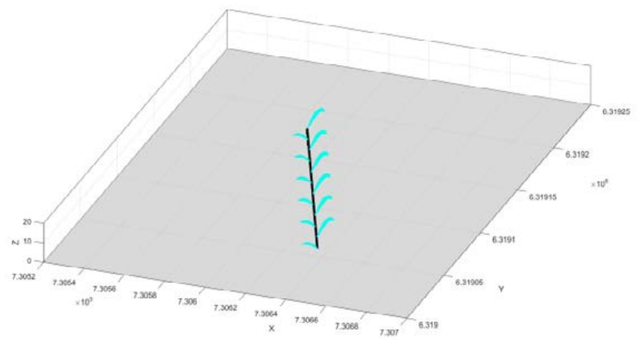
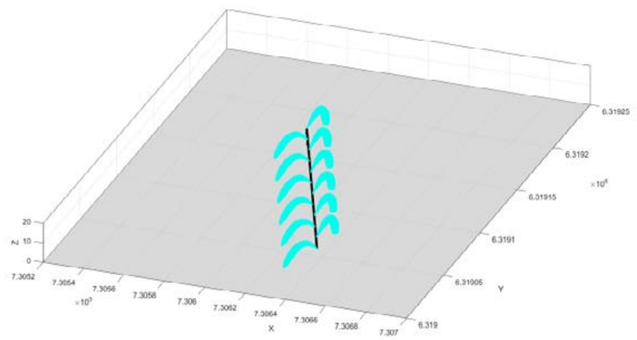
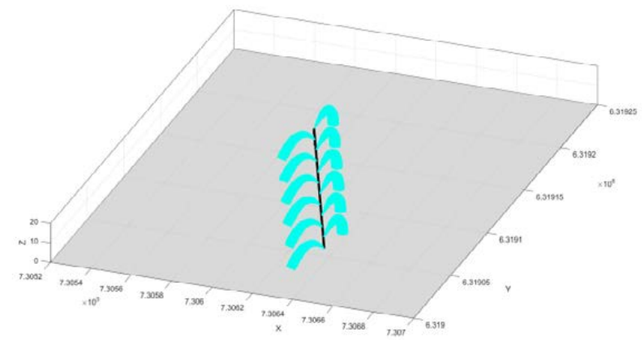
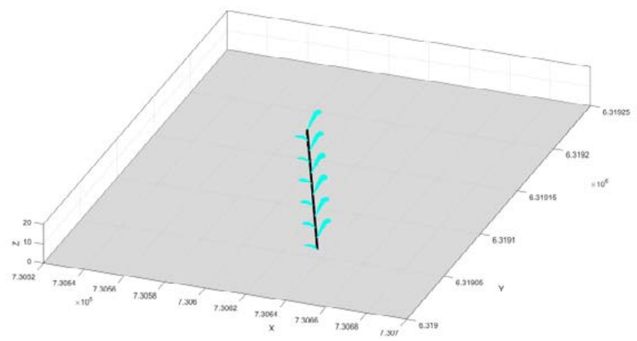
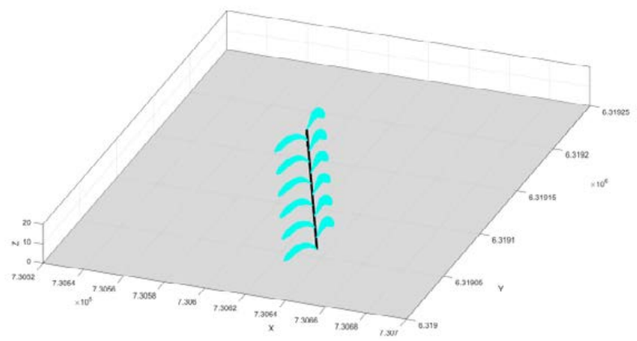
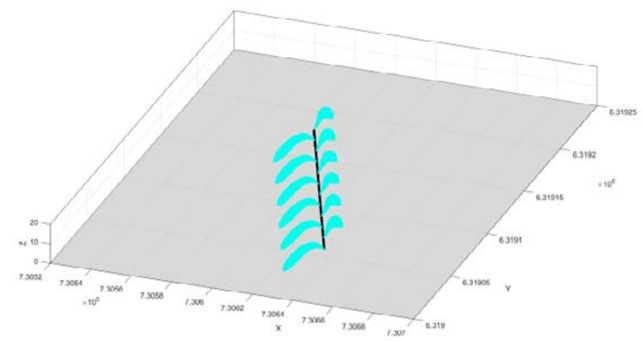
Ambient Velocity	Dilution Iso-Surface (73 GL/year Production Capacity)		
	10	30	50
0.0 m/s			
0.1 m/s			
0.2 m/s			

Figure 7-9 Iso-surface Dilutions for 73 GL/year Hypothetical High-Capacity Scenario (Lower Ambient Velocity Range during Ebb Tides)

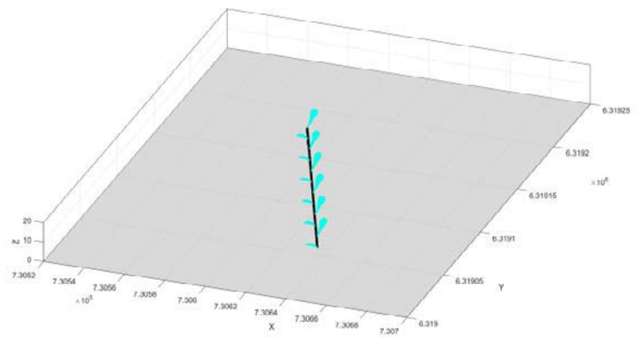
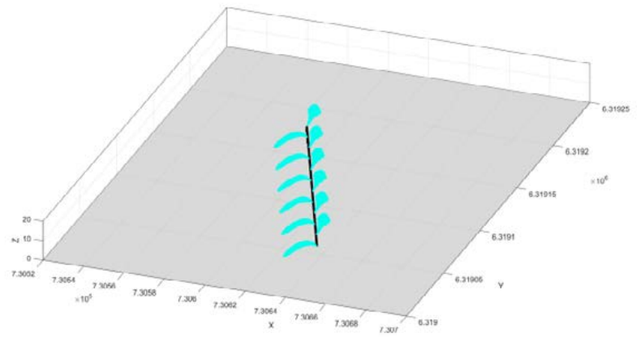
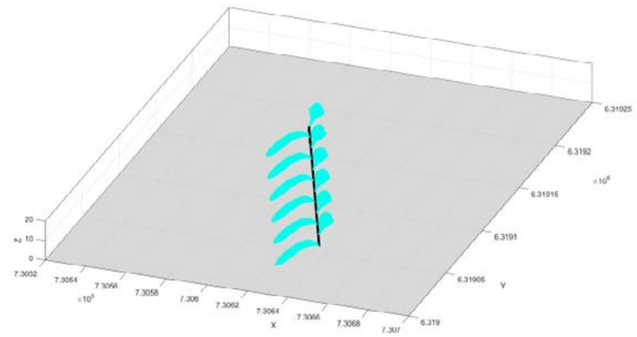
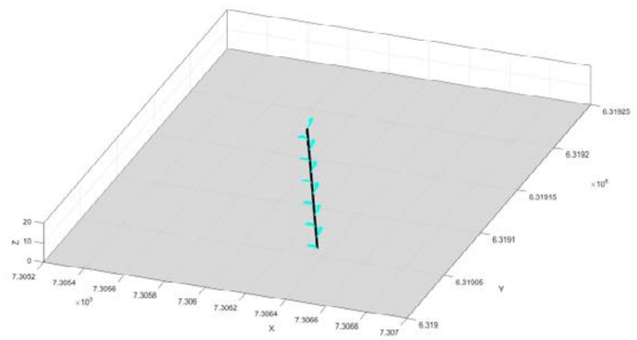
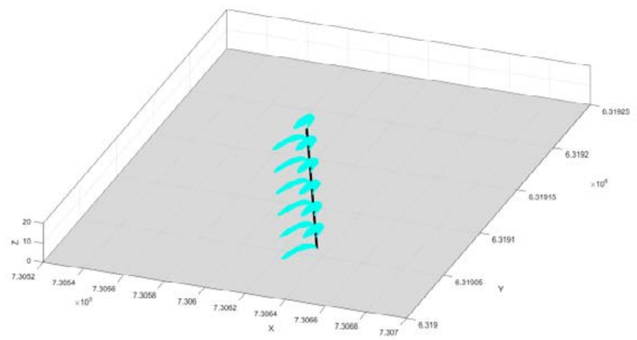
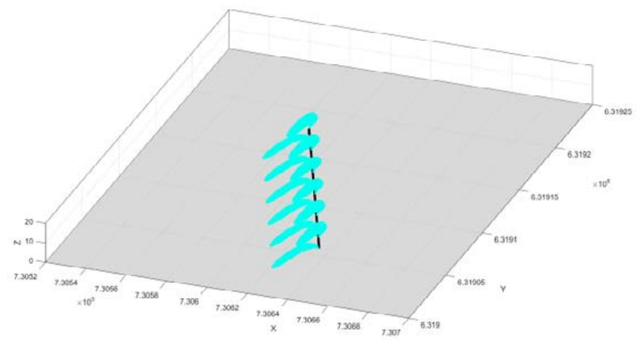
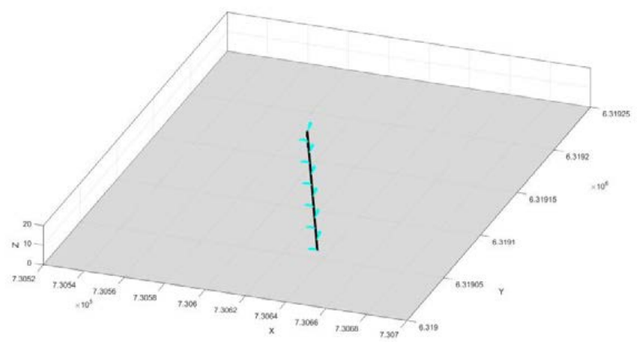
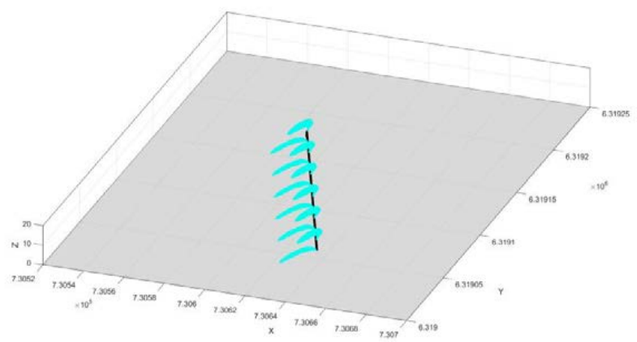
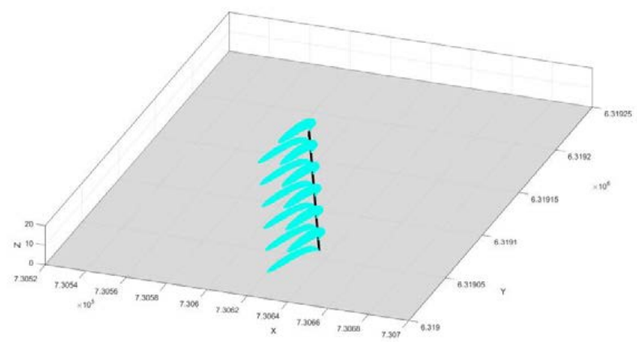
Ambient Velocity	Dilution Iso-Surface (73 GL/year Production Capacity)		
	10	30	50
0.3 m/s			
0.4 m/s			
0.5 m/s			

Figure 7-10 Iso-surface Dilutions for 73 GL/year Hypothetical High-Capacity Scenario (Upper Ambient Velocity Range during Ebb Tides)

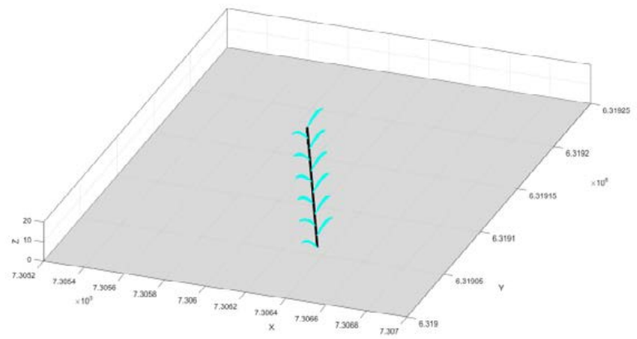
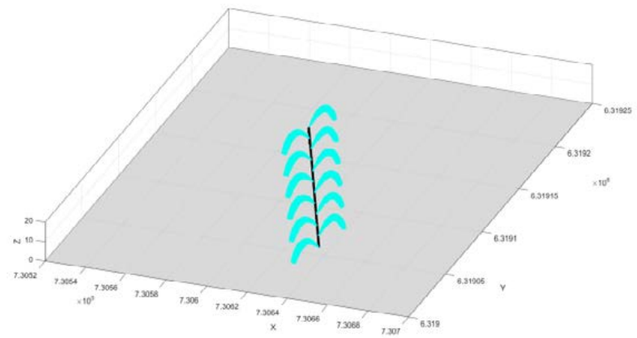
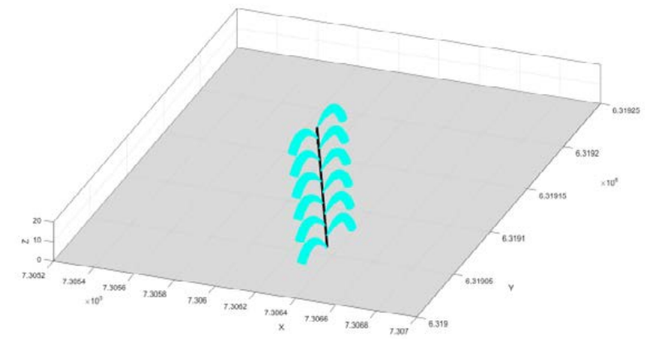
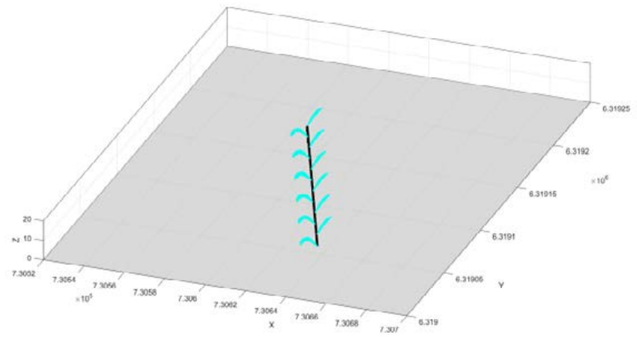
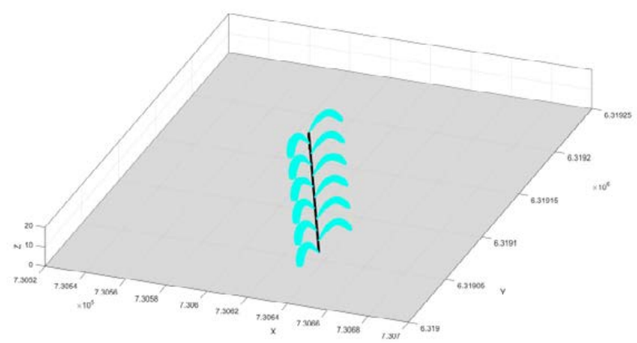
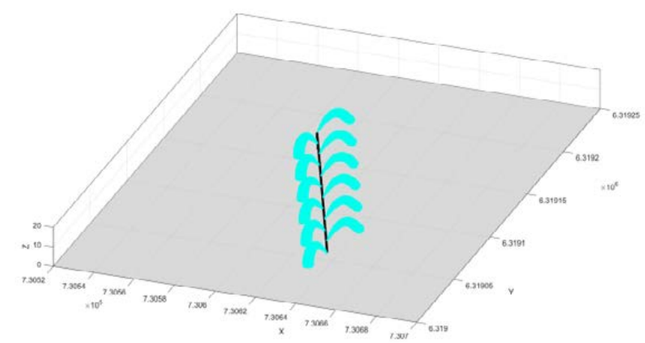
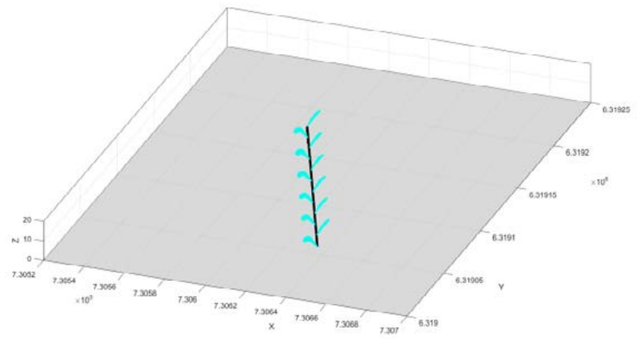
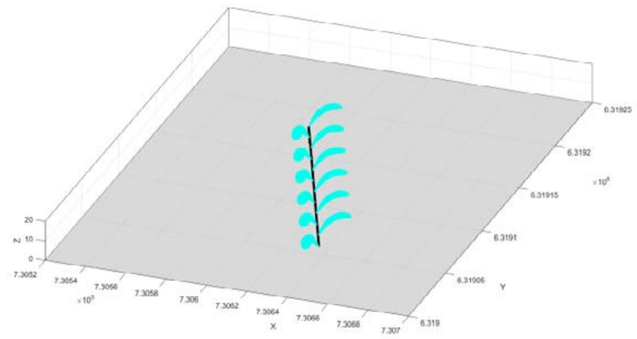
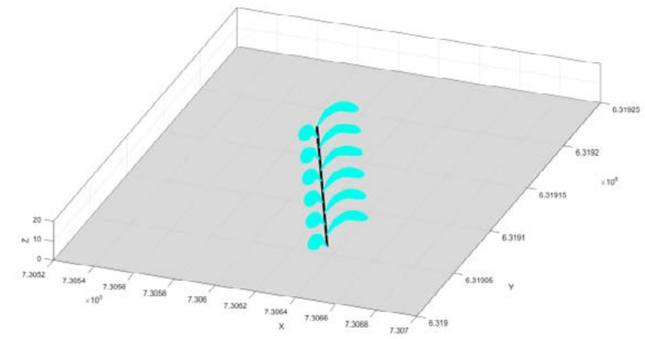
Ambient Velocity	Dilution Iso-Surface (73 GL/year Production Capacity)		
	10	30	50
0.0 m/s			
0.1 m/s			
0.2 m/s			

Figure 7-11 Iso-surface Dilutions 73 GL/year Hypothetical High-Capacity Scenario (Lower Ambient Velocity Range during Flood Tides)

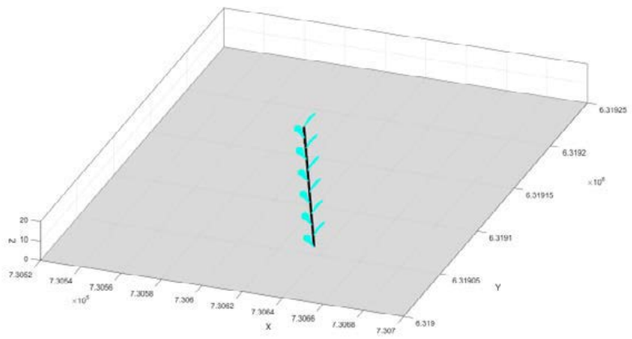
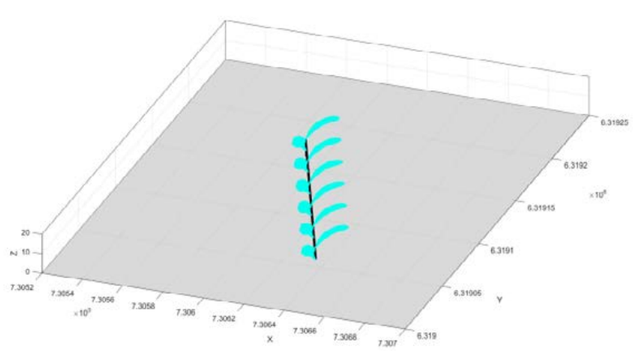
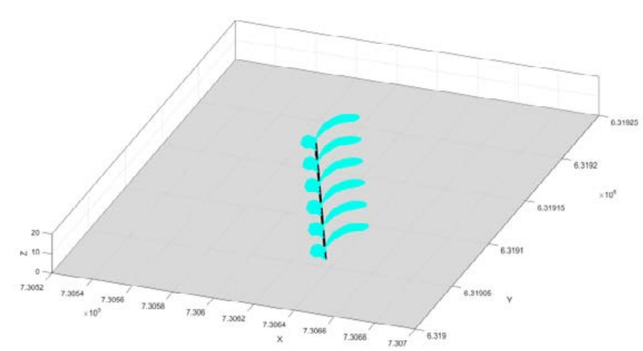
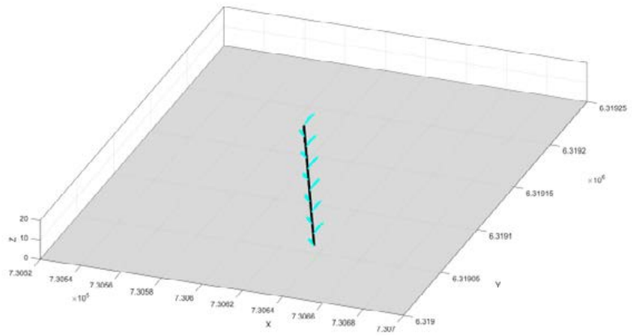
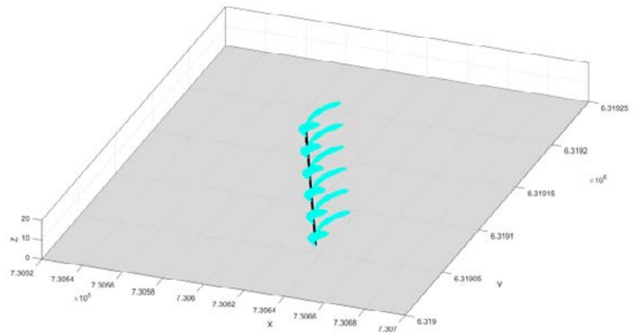
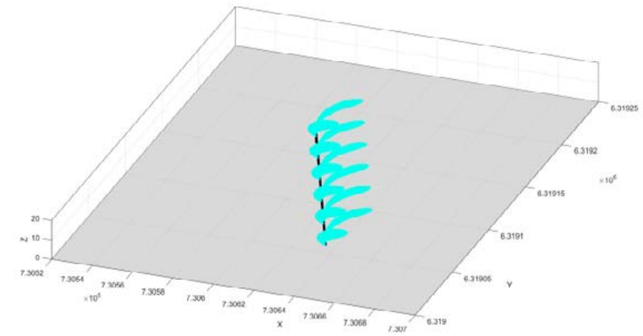
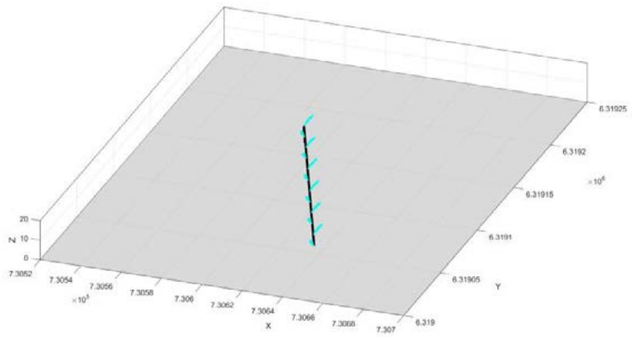
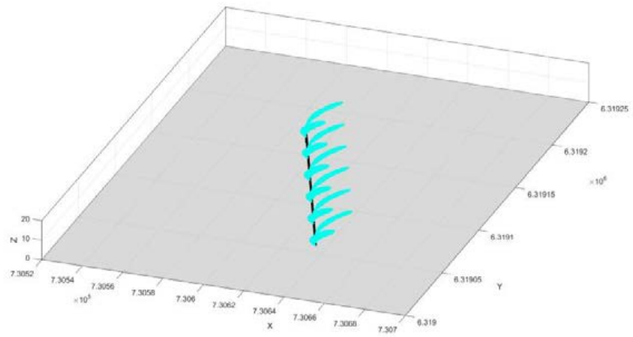
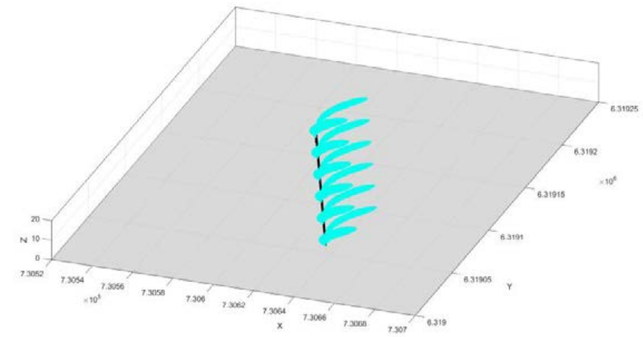
Ambient Velocity	Dilution Iso-Surface (73 GL/year Production Capacity)		
	10	30	50
0.3 m/s			
0.4 m/s			
0.5 m/s			

Figure 7-12 Iso-surface Dilutions for 73 GL/year Hypothetical High-Capacity Scenario (Upper Ambient Velocity Range during Flood Tides)

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Appendix G-12:

Desalination Plant Discharge Modelling - Midfield Modelling

WATER TECHNOLOGY
WATER, COASTAL & ENVIRONMENTAL CONSULTANTS

Report

Northern Water Desalination Plant Discharge Modelling - Midfield Modelling

JBS&G/Department for Infrastructure and Transport

23 September 2026



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Document Number	22020074_R04v02_midfield_modelling_MQ

Government of South Australia
Department for Infrastructure and Transport

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ACKNOWLEDGEMENT OF COUNTRY

The Board and employees of Water Technology acknowledge and respect the Aboriginal and Torres Strait Islander Peoples as the Traditional Custodians of Country throughout Australia. We specifically acknowledge the Traditional Custodians of the land on which our offices reside and where we undertake our work.

We respect the knowledge, skills and lived experiences of Aboriginal and Torres Strait Islander Peoples, who we continue to learn from and collaborate with. We also extend our respect to all First Nations Peoples, their cultures and to their Elders, past and present.

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1 INTRODUCTION

Northern Water (the Project) aims to provide a new, climate resilient water source for industry in South Australia's Far North and Upper Spencer Gulf. This will enable the growth of industries and key mining resources crucial for contributing to decarbonisation initiatives, sustainability and unlocking long term economic prosperity for South Australia.

The Project involves construction and operation of a large-scale seawater desalination plant in Spencer Gulf, connected to northern South Australia via several hundred kilometres of pipeline, on the traditional lands of the Barngarla and Kokatha peoples.

The desalination plant and main water supply pipeline will run north from the desalination plant to Olympic Dam passing Whyalla, Port Augusta, Woomera and Roxby Downs, and a lateral branch pipeline to a delivery point at the Carrapateena copper-gold mine.

Water for the Project will be sourced from a seawater desalination plant to be installed near Mullaquana Station in Spencer Gulf, located approximately 25 km south-west from the centre of Whyalla (Figure 1-1). The delivery of the Project will involve the construction and operation of a 51 gigalitre per year (GL/y) seawater reverse osmosis desalination plant including marine infrastructure to extract raw seawater and discharge return water to the marine environment. As part of Project environmental approval studies, marine modelling is required to assess dispersion of the return water and the effects of the plant water intake on the environment and other receptors.

Marine modelling was undertaken using a three-tiered modelling approach, including:

- A **farfield** model focused on Spencer Gulf-wide effects of the desalination plant (from 10 to 100's of km);
- A **midfield** model focused on the dispersion of the return water in the local vicinity of the point of discharge and potential for fish and larvae entrainment into intake structures (50 m to 10 km); and
- A **nearfield** model focused on the effects of the discharge in the immediate vicinity of the point of discharge (1 to 100 m from the outfall).

In this report, details of the midfield modelling are presented, with near and farfield modelling described in separate reports. The nearfield modelling is described in Water Technology (2026), while the midfield model includes forcing from the farfield model covering the entire Spencer Gulf (Doubell and James, 2026) and the discharge conditions defined by the Department for Infrastructure and Transport (DIT). Details of the desalination plant location and modelling are described below.



Figure 1-1 Intake, Outfall and Measurement Locations

2 DESALINATION PLANT

2.1 Locality

The proposed desalination plant is to be located near Mullaquana Station on the west coast of Spencer Gulf, South Australia, approximately 25 km south-west from the centre of Whyalla (Figure 1-1). The area encompassing the desalination plant precinct is approximately 200 ha and set back from the shore. The proposed intake and outfall tunnels that form the basis of this modelling report extend approximately 4,000 and 4,800 m offshore, respectively.

2.2 Spencer Gulf

Nested between the Eyre and Yorke Peninsulas in South Australia, Spencer Gulf is a large semi-enclosed waterbody, extending approximately 320 km north to south and 130 km east to west at its maximum width. The Gulf is relatively shallow, with mean and maximum depths of approximately 24 m and 59 m, respectively. The Upper Spencer Gulf Marine Park (with a boundary defined by a line between Cowleds Landing and Port Davis) is significantly narrower (~36 km) and shallower (maximum depths of ~26 m), while the remainder of the Gulf presents increasing depths towards the south. The Gulf's mouth is approximately 80 km wide and populated by a number of islands, which restrict and direct flow exchange with the Southern Ocean. Outward flow is predominantly directed to the east between Wedge Island and the tip of Yorke Peninsula, while incoming flows occur to the west between Wedge and Thistle Islands.

Hydrodynamic circulation is characterised by an inverse estuarine circulation, influenced by strong seasonal heating and cooling gradients. Due to the arid climate setting, evaporation exceeds precipitation on an annual basis and leads to a large salinity increase (exceeding 50 g/L) in the shallower and narrower northern areas of Spencer Gulf over spring and summer months (Nunes Vaz et al. 1990, Botelho et al. 2011). As a result, a negative north to south density gradient develops, and subsequently relaxes through autumn to winter. Given its width, the relaxation of this gradient is affected by the Earth's rotation, creating a dense outflow along the eastern side of the Gulf, with a slow return of the water along its western side. As a result, a clockwise circulation pattern forms over winter periods (Nunes Vaz et al. 1990; Middleton et al. 2013).

Flushing of Spencer Gulf is influenced by its size and the oceanic conditions at its mouth acting to suppress water exchange. Modelling assessments indicate flushing times in excess of 300 days in the Upper Spencer Gulf and in excess of 280 days at the Mullaquana Station location (BMT WBM 2011). As such, it is crucial to understand how the discharge and ambient physical processes interact to disperse desalination plant return water.

In addition to the seasonal circulation and large-scale circulation pattern described above, the propagation of the tides is also a dominant feature of the hydrodynamics in Spencer Gulf. At both near and midfield scales, it is expected that their influence will largely dictate plume movement and turbulent intensity leading to local return water mixing.

Tides are generally semi-diurnal with an amphidromic point for the semi-diurnal components near Wallaroo. Amplification of the tidal range follows the reduction of the Gulf's volume to the north. The maximum tide range (given by the Highest Astronomic Tide (HAT) level) at Whyalla is 3.3 m, which is significantly lower than the 3.9 m at the head of the Gulf (Port Augusta). Contrasting to the highest tidal amplitudes, the Gulf also experiences periods of low tidal variation, known as 'dodge' tides'. These tides relate to periods when dominant diurnal (O1 and K1) and semi-diurnal (M2 and S2) tidal constituents approximately cancel each other, resulting in periods of limited water level variation between May and June and November and December every year. Specific conditions at the NW desalination plant site are described below.

2.3 *In-Situ* Measurements

In-situ ambient data currently being collected by South Australia Research and Development Institute (SARDI) for the Project as part of SARDI's ongoing monitoring of Spencer Gulf were adopted for model construction, calibration and validation. The station locations are shown in Figure 1-1. The most relevant location for model comparisons was NWP3 (~2.7 km offshore of the outfall) due to its proximity to the Project desalination plant marine infrastructure. Data from SAMUSG (approximately 31.5 km NE of the outfall) was used to complement periods not covered by the NWP3 measurements. Available data include:

- Acoustic Doppler Current Profiler (ADCP) data (water levels and currents);
- Continuous salinity and temperature at approximately 1.0 m from seabed;
- Temperature and salinity profiles at the time of instrument servicing; and
- Nearbed *in-situ* TSS data at the proposed outfall site collected as part of the marine water quality baseline data collection program.

2.3.1 ADCP Data

ADCP data collected are summarised in Table 2-1 and subsequently discussed below.

Table 2-1 Summary of ADCP measurements

Characteristic	NWP3	SAMUSG
Number of Deployments	2	6
Measurement period	25 October 2022 to 09 May 2023	5 May 2022 to 23 November 2023
Nominal coordinates	137.503883E 33.242583S	137.7083E 33.1112S
Nominal depth (m)	26.5	19.3
Number of valid measurement bins	15	15
Measurement bin thickness (m)	1.0	1.0
Lowest bin level (m above seabed)	1.10	1.10
Sampling interval (min)	10	10

2.3.1.1 Water Levels

Water levels at the NWP3 and SAMUSG stations are shown in Figure 2-1. Similar plots at both stations across different seasons are shown in Appendix A.

Given the stations' proximity, full water level signals, tide-only signals and residuals were closely related with minimal deviations in both phase and amplitude. SAMUSG had amplitudes approximately 0.1 to 0.3 m higher than NWP3 due to tidal amplification moving north into the Gulf. Tide-only ranges over the measurement period were respectively 2.7 m and 3.0 m at NWP3 and SAMUSG, respectively. Maximum residuals (order of ~0.5 m) occurred throughout the year[1].

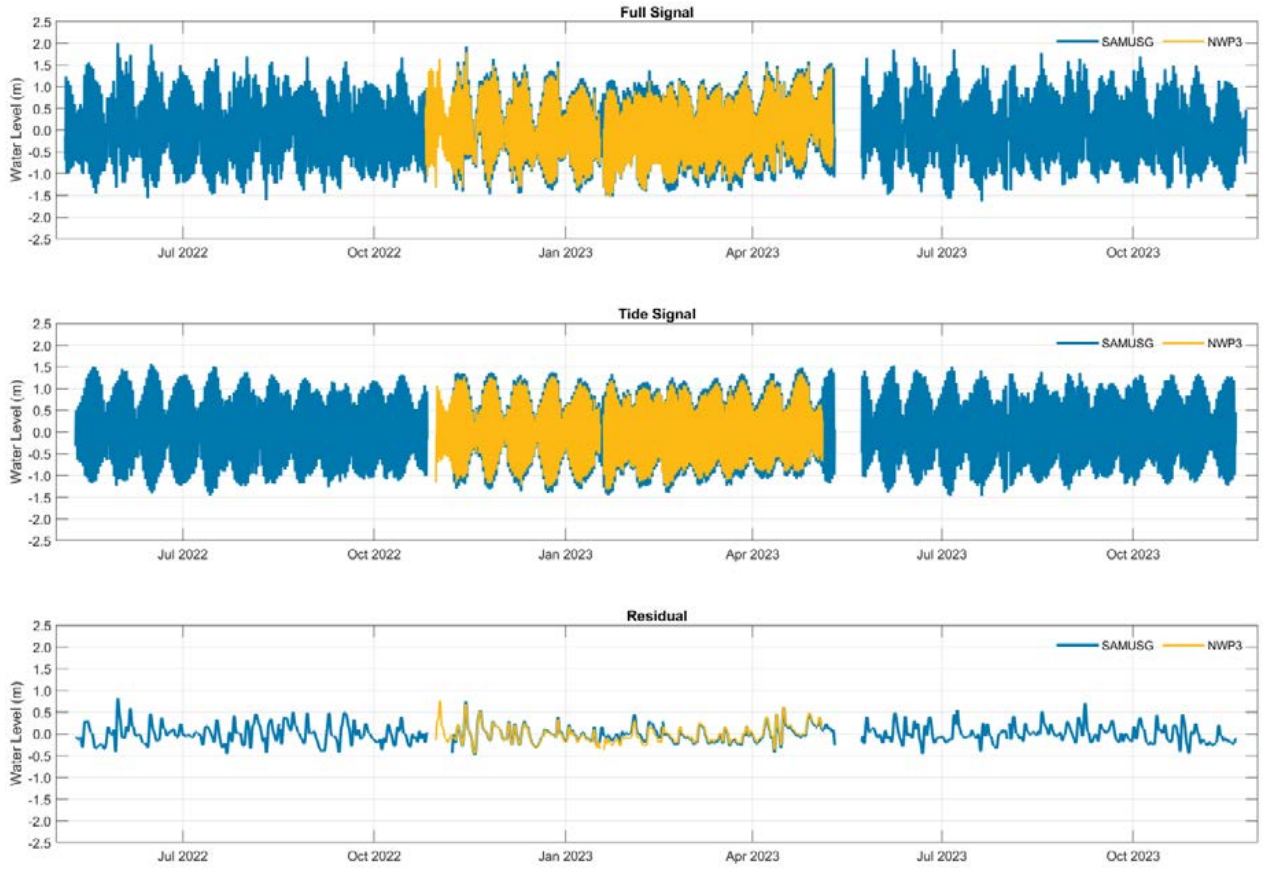


Figure 2-1 Water Level Data Measured by ADCPs at NWP3 and SAMUSG

2.3.1.2 Currents

A subset of measured currents from the ADCP stations are shown in Figure 2-2 to Figure 2-3, in which measurements at the stations present the signature of dodge tides. The full set of measurements is provided in Appendix A. Note that NWP3 had low reliability data in the upper portion of the water column for the deployment period between 19 January and 11 March 2023 (Appendix A).

The data was then filtered using a tide killer filter (Thompson 1983), decomposed into a principal tidal component, from which residual currents were obtained. Depth-averaged tidal and residual signals at the

[1] Note that lower residuals and discrepancies between NWP3 and SAMUSG occurred as an artefact of the data gaps when stations were being maintained and need to be disregarded.

ADCP stations are shown in Figure 2-4. Similar plots at the ADCP stations across different seasons are provided in Appendix A.

The principal tidal component axis difference between stations was ~13°, influenced by local bathymetry (in this analysis, for both stations, only the deployment period common to both stations was considered). The phase of both tides and residuals was similar across the stations, with SAMUSG tending to present slightly higher magnitudes throughout the period. Residual magnitudes were generally below 0.1 m/s. As evidenced by the current roses, currents presented a relatively symmetrical direction distribution aligned with the principal tidal axis components, particularly for NWP3 (Figure 2-5). While currents during flood tides were generally similar to ebb tides at NWP3, they were stronger during floods at SAMUSG (Figure 2-6). This shows the propensity of currents flowing north in the region between the stations over the analysed period.

Cumulative distributions of currents indicated their magnitude at NWP3 was significantly lower than at SAMUSG. Median and maximum current magnitudes during flood tides at NWP3 were 0.15 m/s and 0.50 m/s, respectively, and during ebb tides 0.17 m/s and 0.50 m/s, respectively. At SAMUSG, median and maximum current magnitudes during flood tides were 0.28 m/s and 0.72 m/s, respectively and during ebb tides 0.23 m/s and 0.67 m/s, respectively.

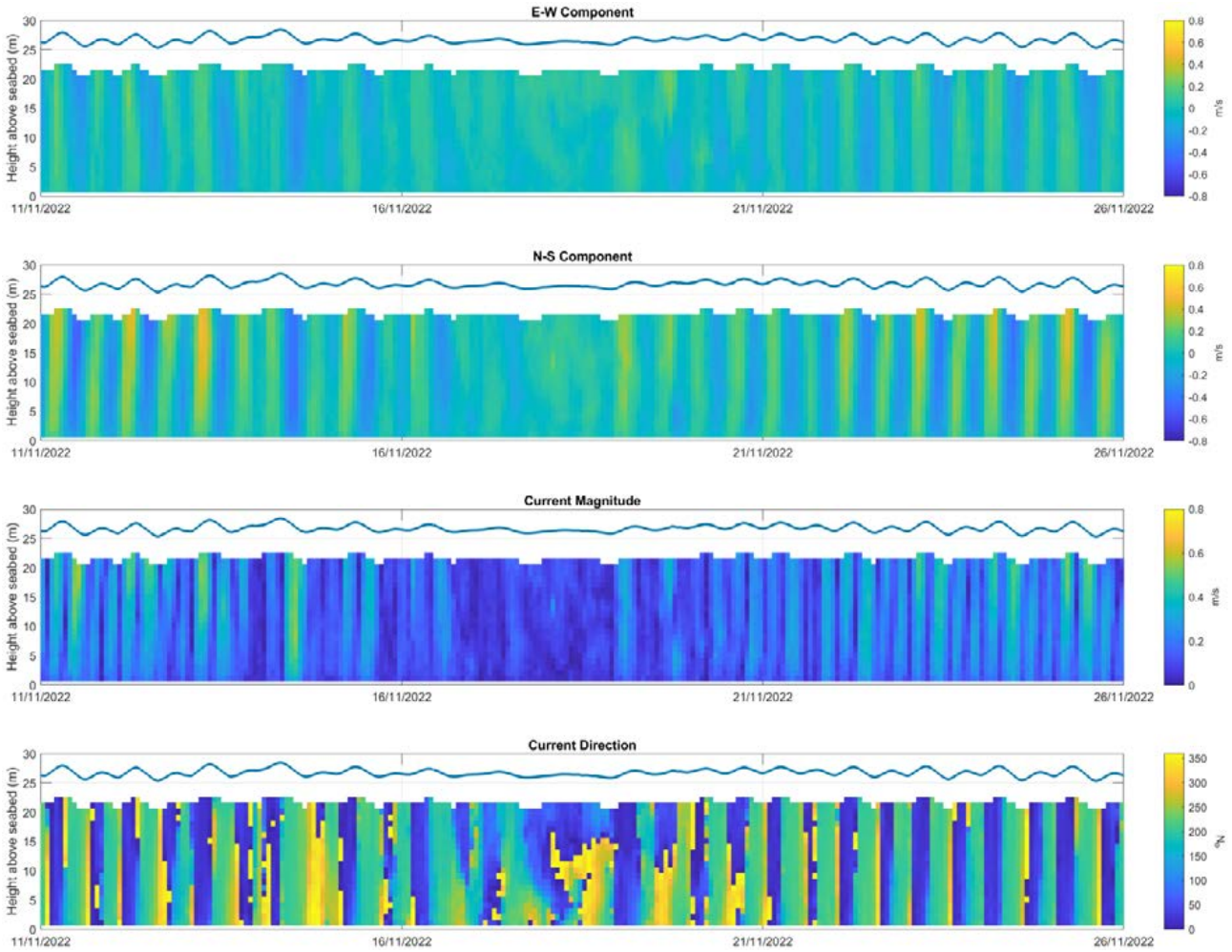


Figure 2-2 Currents Measured by ADCP at NWP3 (11 to 26 November 2022)

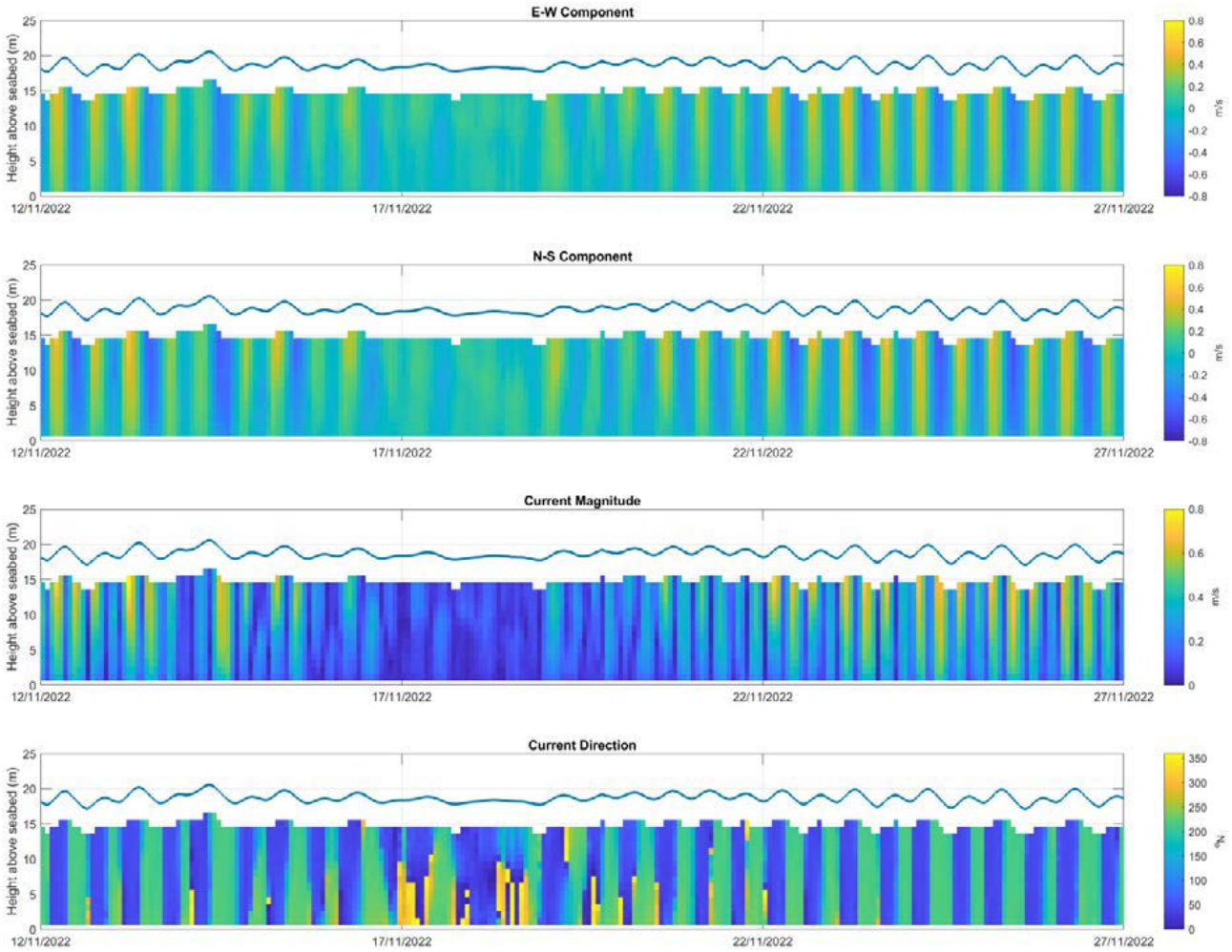


Figure 2-3 Currents Measured by ADCP at SAMUSG (12 to 27 November 2022)

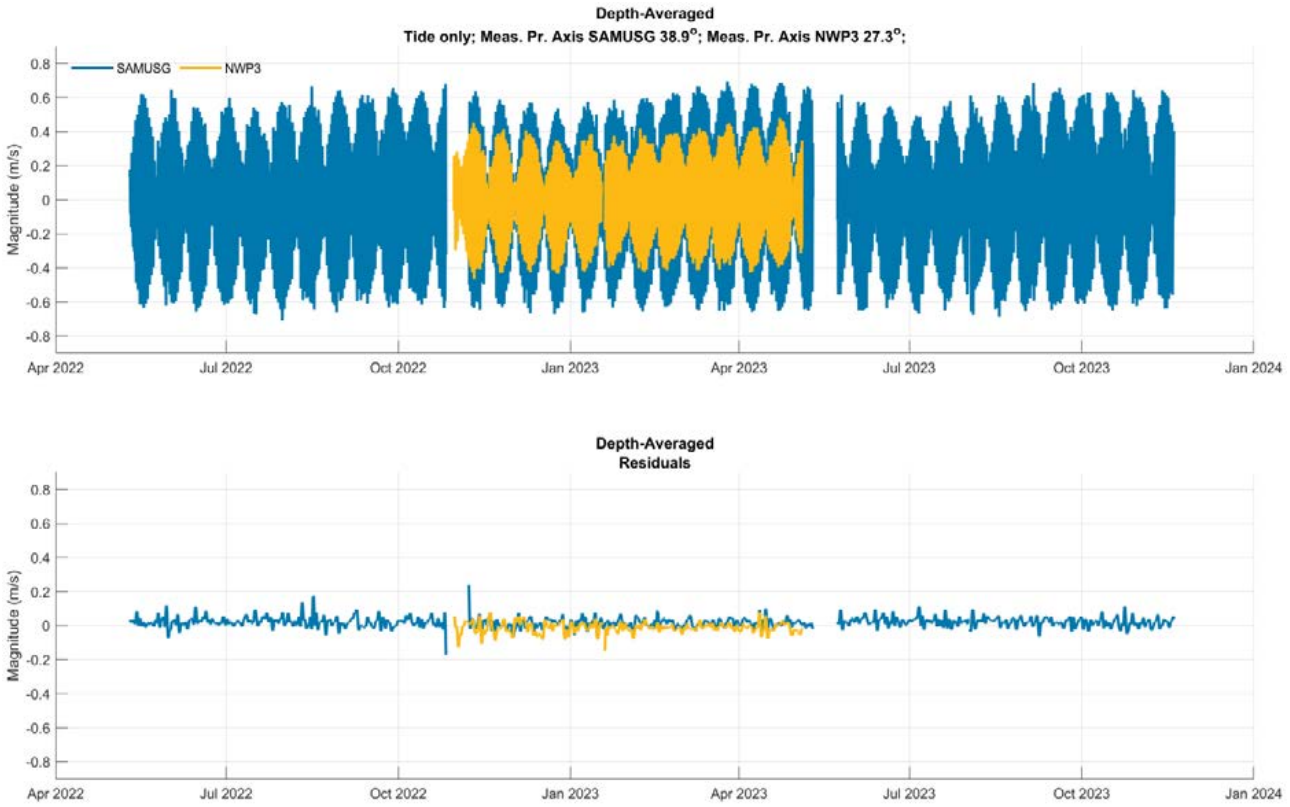


Figure 2-4 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG Stations

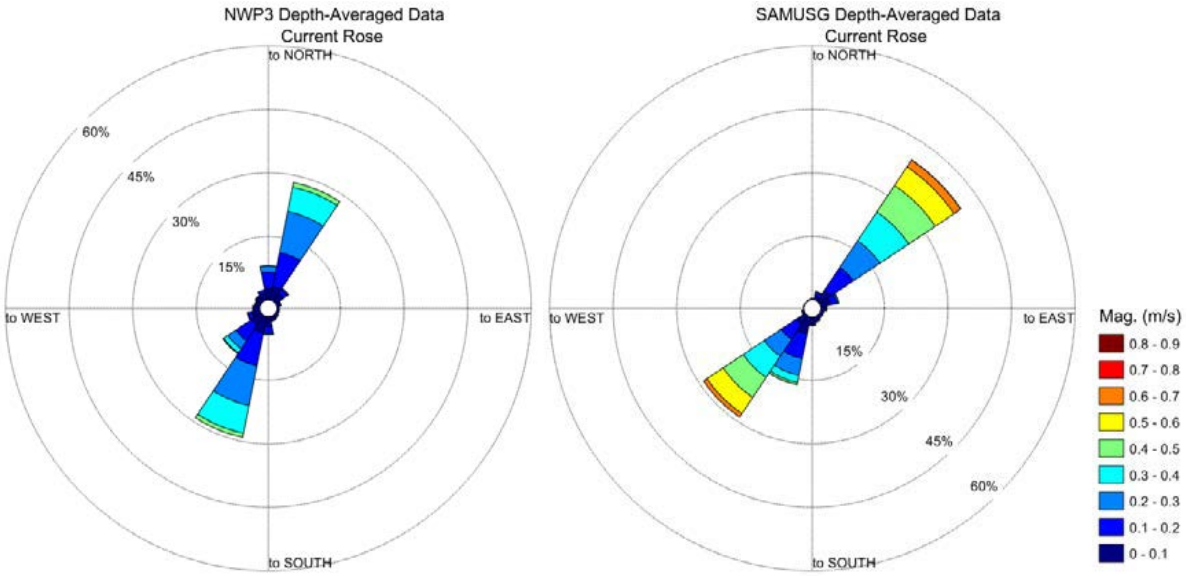


Figure 2-5 Current Roses at NWP3 and SAMUSG Stations from 08 November 2022 to 8 May 2023

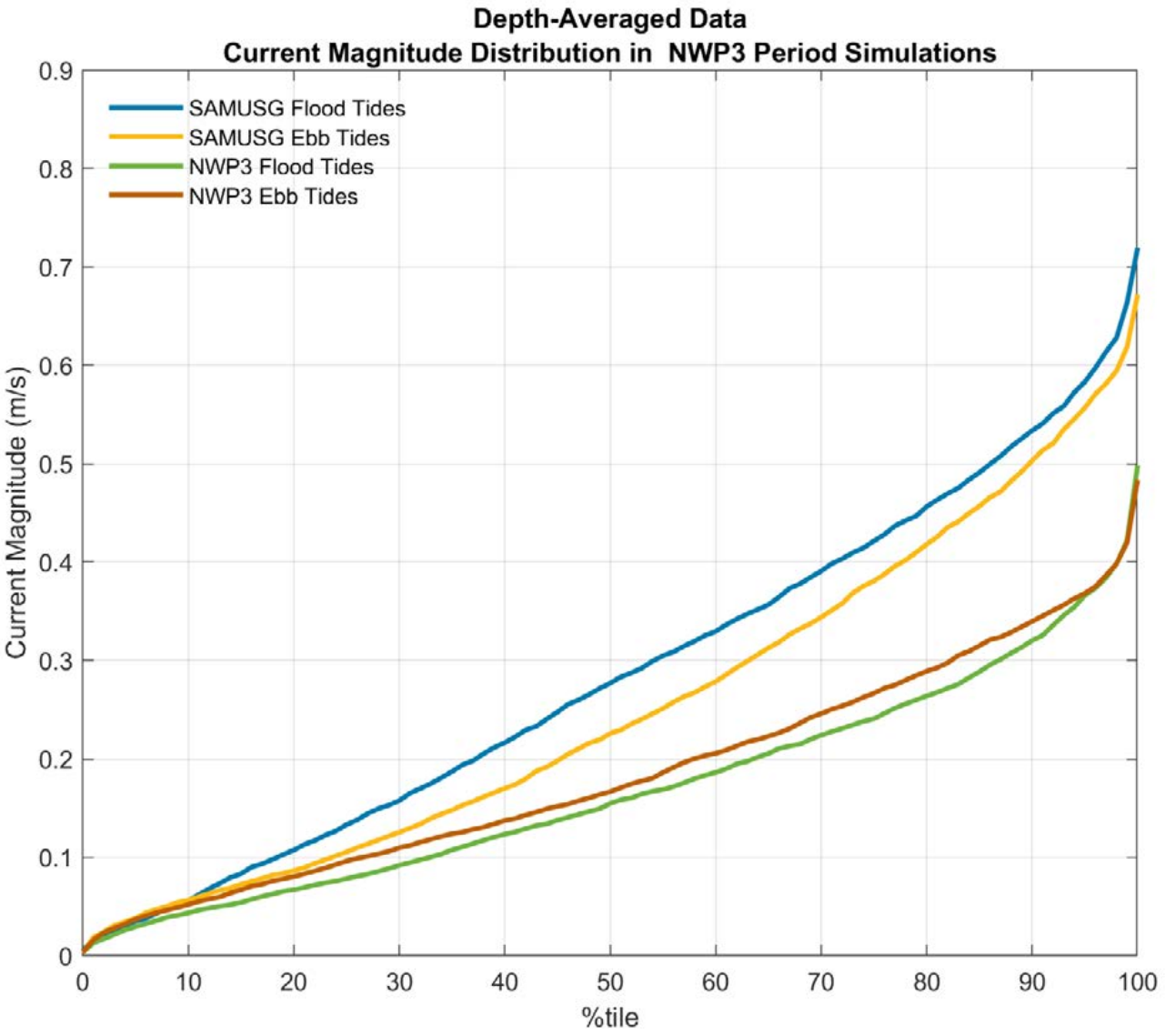


Figure 2-6 Current Magnitude Distributions During Ebb and Flood Tides at NW Stations from 7 November 2023 to 3 March 2024.

2.3.2 CTD Data

Salinity and temperature data collected at NWP3 and SAMUSG are summarised in Table 2-2 and subsequently discussed below.

Table 2-2 Summary of Salinity and Temperature Measurements

Characteristic	NWP3	SAMUSG
Measurement period	25 October 2022 to 09 May 2023	16 August 2022 to 23 November 2023
Nominal coordinates	137.503883E; 33.242583S	137.7083; 33.1112S
Nominal depth (m)	26.5	19.3

Salinity data collected near the bed at the NW stations was converted to g/L following TEOS-10 equations (IOC, SCOR and IAPSO, 2010). As its series was longer, SAMUSG presented a larger range of salinity over

the measurement periods. SAMUSG displayed a salinity range between 38.3 g/L and 41.8 g/L, while NWP3 was between 38.5 and 40.9 g/L (Figure 2-7). Although SAMUSG had a larger range, for the common measurement period between stations, salinity was generally between 0.3 g/L and 1.0 g/L lower at NWP3, which reflected the nature of Spencer Gulf as an inverse estuary. Short-term salinity variations at the sub and daily scales (i.e., in response to tidal movement) were also higher at SAMUSG largely associated with the higher tidal velocities and excursions as presented by ADCP measurements.

Across both stations, salinity was lowest in spring (October and November), with a progressive increase over summer before peaking in autumn around May. Profile measurements during instrument maintenance show good agreement between moored and profiling instruments. Records from both stations showed more drastic changes from late November to January in comparison to other periods, primarily at NWP3. These changes are associated with the density relaxation during neap conditions as salinity builds up in the Upper Spencer Gulf (Nunes Vaz et al. 1990, Botelho et al. 2011).

Temperature was quite similar between stations, with a slight lower bias for NWP3 (Figure 2-7). Moored measurements were also in agreement with the spot profiles.

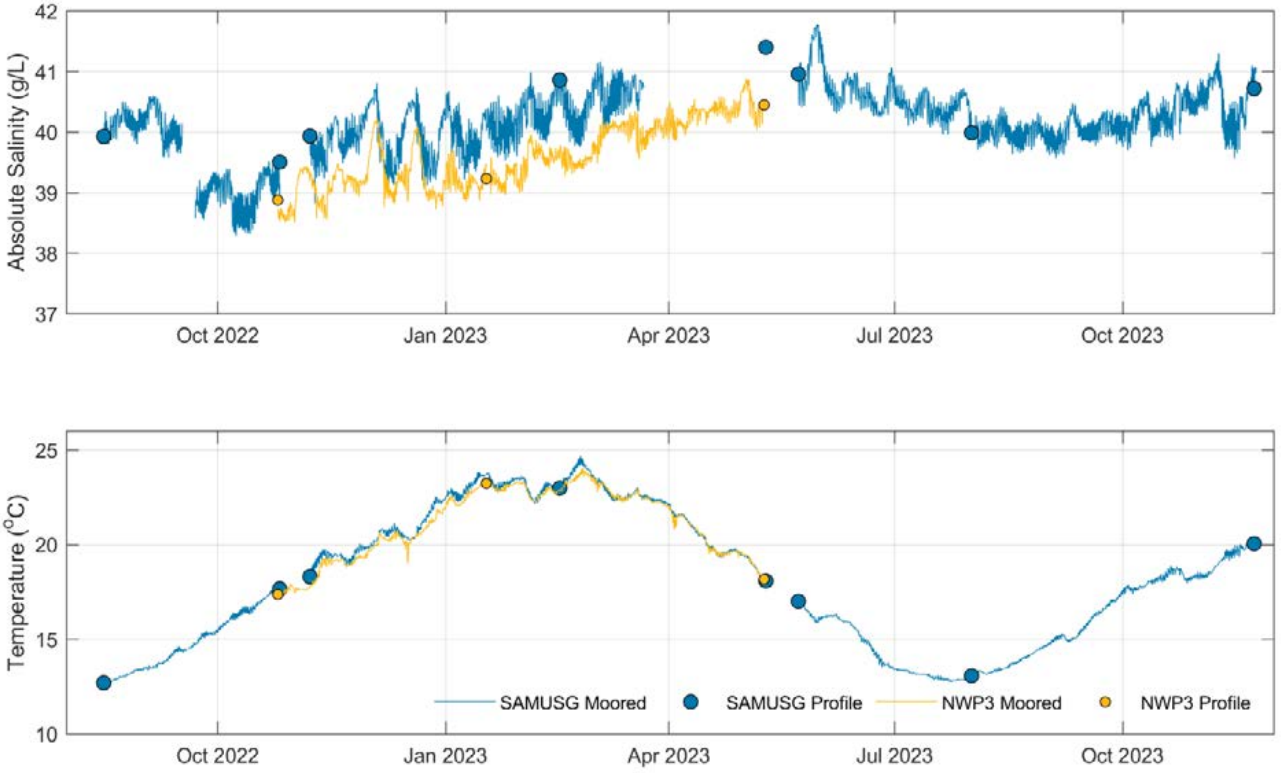


Figure 2-7 Measured Salinity and Temperature at NWP3 and SAMUSG Stations

2.4 Total Suspended Solids

Statistics of *in-situ* TSS data collected (and laboratory-analysed) for the Project are provided in Table 2-3. These include data collected in May and June 2025 and a total of 45 samples near the proposed outfall location.

Table 2-3 Statistics of In-Situ Baseline TSS data

TSS Data Statistics	Values at Outfall (mg/L)	
	Bottom Water	All Data
Minimum	3.6	2.4
Maximum	19	26
Mean	11	7.3
Standard Deviation	5.7	5.2
95% Upper Confidence Level	4.4	1.5
50% Percentile	12	4.9
90th Percentile	17	14

2.5 Proposed Outfall and Intake Designs

Proposed marine infrastructure will include:

- A combined intake and outfall tunnel;
- A single intake structure; and
- A linear alternating-ports diffuser.

A list of design criteria relevant to the present assessment is provided in Table 2-4. Of note, characteristics for minimum capacity (40 ML/d treated water production, also referred to as 'turn down'), 51 GL/year (160 ML/d) treated water design production capacity, and 73 GL/year (200 ML/d) treated water hypothetical high-capacity scenario are presented. For turn down conditions, a 35 ML/d bypass flow between the intake and outfall is adopted to reduce discharge salinity and improve mixing of return water.

2.5.1 Outfall

Main components of outfall design include:

- An outfall tunnel, ending below water approximately 20 m deep;
- A Glass-Reinforced-Polymer (GRP) riser through the seabed at the deep end of the tunnel; and
- A diffuser manifold comprising 12 alternating ports (250 mm DN) installed on the seabed.

The linear diffuser will have provision for up to 12 ports, with the aim of servicing proposed flow rates as required from the Project. Locations of proposed intake and outfall alignments are shown in Figure 1-1. A schematic of the proposed outfall configuration is shown in Figure 2-8, (including the hypothetical high-capacity scenario), while conceptual engineering drawings are provided in Appendix B.

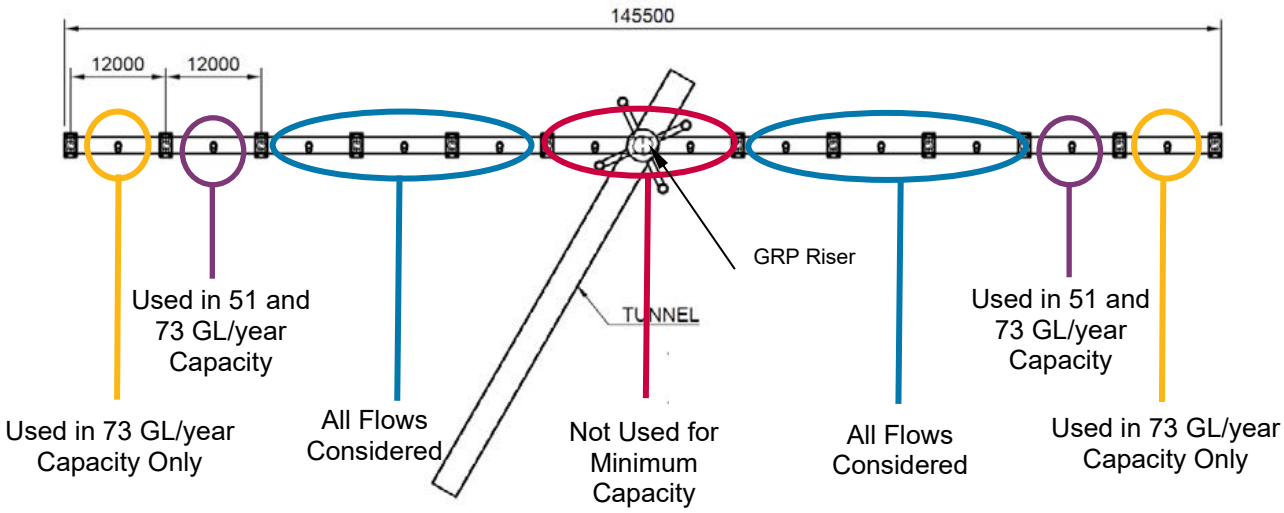


Figure 2-8 Alternating riser configuration for the Mullaquana Station Desalination Plant outfall diffuser (adapted from GHD 2025)

2.5.2 Intake

Main components of the intake design include:

- A 3,900 m concrete-lined intake tunnel (3200 mm DN);
- A 2500 DN riser ending just above the seabed into the intake structure chamber in approximately 15 m water depth;
- An intake chamber of 11.3 to 11.5 m diameter by 2.5 m height; and
- A 2.8 m thick concrete grill panel covering the upper part of the chamber circumference.

Table 2-4 Outfall Reference Design Characteristics

Item	Criteria/Characteristic			
Production Capacity	51 GL/year (160 ML/d) – Design capacity		73 GL/year (200 ML/d) – Hypothetical high-capacity scenario	
Species protection trigger level	Assessment against 90%, 95% and 99% protection (noting that assessment requirements stipulate 95%)			
Species Protection Trigger Values (SPTV) and associated dilution	Point of return dilution of 30 to 40:1 Ecotoxicology assessment recommends following dilutions:			
	SPTV	90%	95%	99%
	Dilution	19	26	53
Dispersion	Avoid any visible plume			
Diffuser spacing	Minimum of 500 m between outfall diffusers and intake structure			
51 GL/year production capacity Daily production, intake and outfall flows	160 ML/d; 389 ML/d (intake); 229 ML/d (outfall)			
73 GL/year hypothetical high-capacity scenario Daily production, intake and outfall flows	200 ML/d; 486 ML/d (intake); 286 ML/d (outfall)			
Minimum production capacity Daily production, intake and outfall flows	40 ML/d (instantaneous); 135 ML/d (intake); 92 ML/d (outfall) 35 ML/d (bypass flow)			
Temperature range	12 to 25 °C (ambient) 15 to 28 °C (outfall)			
Salinity range	38.0 to 41.0 (ambient) 65.8 to 71.0 (discharge, 51 and 73 GL/year capacity) 55.8 to 59.6 (discharge turn-down)			
Total suspended solids (TSS) minimum and peak times conditions	Estimate	Ambient	At outfall	
			Minimum	51 and 73 GL/year
	Lower	7 mg/L	12 mg/L	15 mg/L
	Higher	17 mg/L	28 mg/L	35 mg/L
Local depth	Minimum 20 m at point of discharge			
Number of diffuser ports (nozzles) for different capacities	Minimum		51 GL/year	73 GL/year
	6		10	12
Port inclination	45° with the vertical			
Port direction bearings	31 °N, 211 °N			
Clearance of nozzle from seabed	3.0 m			
Nozzle exit diameter	250 mm			
Nozzle exit flow and velocity	Minimum		51 GL/year	73 GL/year
	0.178 m^3/s 3.62 m/s		0.265 m^3/s 5.39 m/s	0.275 m^3/s 5.61 m/s

2.6 Environmental Criteria

Applicable dilution criteria were derived from ecotoxicological tests for the protection of 90%, 95% and 99% of species (Van Dam, 2025). These are referred to as 90%, 95% and 99% Species Protection Trigger Values (SPTVs). The SPTVs are to be achieved at an appropriate distance from the diffuser to ensure environmental protection. Dilution SPTVs are provided below:

- 90% SPTV: 19 to 1
- 95% SPTV: 26 to 1
- 99% SPTV: 53 to 1

For return water discharges, primary toxicity is due to the effects of salinity on aquatic organisms. However, additional toxicants (e.g., biocides, anti-scalants) are added to the effluent stream to condition the influent and avoid accumulation of microorganisms in the membranes and pipelines. For salinity, derived dilutions should be compared with median concentrations (over time) (ANZG 2018). For the combined effects of salinity and additional constituents in the return water, no guideline is available - such that criteria are derived from ecotoxicological tests. For toxicants, as in the case here, the 95th percentile of concentrations over time (or 5th dilution percentile) are used for comparison against the SPTVs (ANZG 2018). A distance of 100 m is often used as a criterion for nearfield dilutions to be achieved.

In addition to the ANZG (2018) criteria, a method based on licence condition 1.1 currently adopted for the Adelaide desalination plant (ADP) (SA EPA 2012)[2] was selected to better understand the impacts in terms of salinity near the seabed. In this case, the higher salinity range in Mullaquana Station compared to Port Stanvac (location of the ADP) was taken into consideration for the proposed criteria. Hence, for the salinity impact assessments, a reference salinity site representing ambient conditions is required and 24-hour averages of salinity changes with respect to these ambient conditions are calculated. For the assessments presented in this report, the following salinity changes from ambient were investigated:

- 1.0 g/L and 1.3 g/L (based on the ADP criteria); and
- 1.5 g/L; and1.8 g/L (scaled considering the salinity range at Mullaquana Station).

For a desalination plant, expected worst case conditions in the water column are near the seabed where the water accumulates after some degree of mixing in the near field. Once the return water reaches the seabed, the level of mixing reduces considerably and is largely controlled by local currents. For the ADP, no specific directions are provided in relation to what level or depth in the water column measurements are required. For the purposes of the present assessment, near bed salinity changes were used - noting that they represent the worst case in the water column.

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[2] Condition 1.1 in SA EPA (2012) stipulates the following:
"If the salinity, averaged over 24 hours, measured at 100 metres from the diffuser structure, exceeds 1.3 PPT above ambient salinity the Licensee must:

1.1.1 undertake an investigation to determine the cause of the exceedance
1.1.2 provide to the satisfaction of the EPA and within 5 working days of the determination of the exceedence, a program to develop and submit a final investigation report.
1.1.3 in accordance with the program required by 2., provide a final investigation report to the satisfaction of the EPA which includes:
a) the outcomes of the investigation;
b) actions to be undertaken to ensure that a maximum 1.3ppt differential averaged over 24 hours is maintained at all times; and
c) timeframes for the actions to be completed.
1.1.4 implement the actions as required by 1.1.3.b within the approved timeframes upon approval in writing by the EPA".

3 MIDFIELD MODEL

3.1 Modelling Objectives

The main modelling objective was to simulate the dispersion of the return water in the environment so that the following could be assessed:

- Extent of the return water impact footprint near the outfall as defined by the dilution environmental criteria (Section 2.6);
- Extent of the return water impact footprint near the outfall as defined by salinity change environmental criteria (Section 2.6);
- Associated TSS change near the outfall;
- Any potential recirculation between return water and the seawater intake; and
- Potential for entrainment of prawn (mainly *Penaeus (Melicertus) latisulcatus)* larvae, King George whiting (*Sillaginodes punctatus*) and snapper (*Chysophrys auratus*) juveniles in the seawater intake.

Results of these assessments were used to inform and support ecological and water quality impact assessments as part of the broader environmental studies.

3.2 Modelling Software

The DHI MIKE modelling package was used for this project. MIKE is an industry standard software package regularly used for hydrodynamic simulations. The DHI MIKE 3 FM Hydrodynamic model (HD) was used in conjunction with the Advection-Dispersion module (AD) (DHI 2017a) for prediction of return water dilution in the receiving waters. The model is a general modelling system for simulation of unsteady three-dimensional water flows driven by density variations, bathymetry as well as external forcing from meteorology, tide and ocean currents. The model also includes capability to link nearfield dynamics into the HD and AD solutions, making it an ideal system for simulation of the effects of the return water discharge in Spencer Gulf.

For the prawn larvae, King George whiting and snapper juveniles entrainment assessments, the hydrodynamic MIKE model was coupled with the MIKE ECO Lab module (DHI 2017b). This module integrates seamlessly with the existing hydrodynamic modelling package and employs Lagrangian particle tracking methods to trace the trajectories of individual particles influenced by fluid dynamics, including currents, turbulence, and dispersion. The Lagrangian approach tracks particles as they move through the fluid domain, capturing their paths based on localized flow conditions. In the case of neutrally buoyant particles, which have the same density as the surrounding fluid, the model ensures that the particles' motion is governed primarily by the ambient flow velocity and turbulence. This modelling approach enabled analysis of dispersion patterns and entrainment related to the seawater intake.

3.3 Model Mesh and Bathymetry

The model mesh was prepared to simulate the effects of large-scale meteorologic and oceanic forcing for the prediction of water levels, currents, salinity and temperature at the location of the intake and discharge with sufficient resolution to adequately determine the discharge footprint and recirculation between outfall and intake. The model domain, mesh and bathymetry adopted for the present assessments are shown in Figure 3-1. The southern model boundary extends from approximately 23 km northeast of Franklin Harbor entrance to Point Riley (also known as Cap Condorcet), just north of Wallaroo.

These extents were used to derive oceanic boundary conditions from the Spencer Gulf farfield (SGFF) model (Doubell and James, 2026), as described in Section 3.4.2.

Land boundaries were adopted at the northern, western and eastern margins of the Gulf, as well as around islands sufficiently large to be resolved within the model resolution.

Horizontal resolution was set to approximately 1 km at ocean boundaries, progressively reducing in size towards the diffuser area where the resolution was set to 25 m. In the vertical, a mixed sigma-vertical (i.e., thickness varying with the surface elevation) and z-vertical grid was adopted with varied resolution, as follows:

- Four sigma layers above -2.0 mAHD;
- Three 2.0 m layers down to -8.0 mAHD;
- Twenty-six 1.0 m layers down to -28.0 mAHD;
- Two 2.0 m layers down to -32.0 mAHD;

This vertical discretisation allowed adequate resolution at levels influenced by the return water discharge, whilst covering the entire depth range of the model domain.

Model bathymetry was sourced from a combination of the following sources:

- Geoscience Australia bathymetric grid of Australia (Whiteway 2009);
- Australian Hydrographic Office (AHO) for near coastal areas (i.e., < 30 m depth); and
- BathyDHI MIKE Zero software was used to perform point inspections and derive the model bathymetry shown in Figure 3-1.

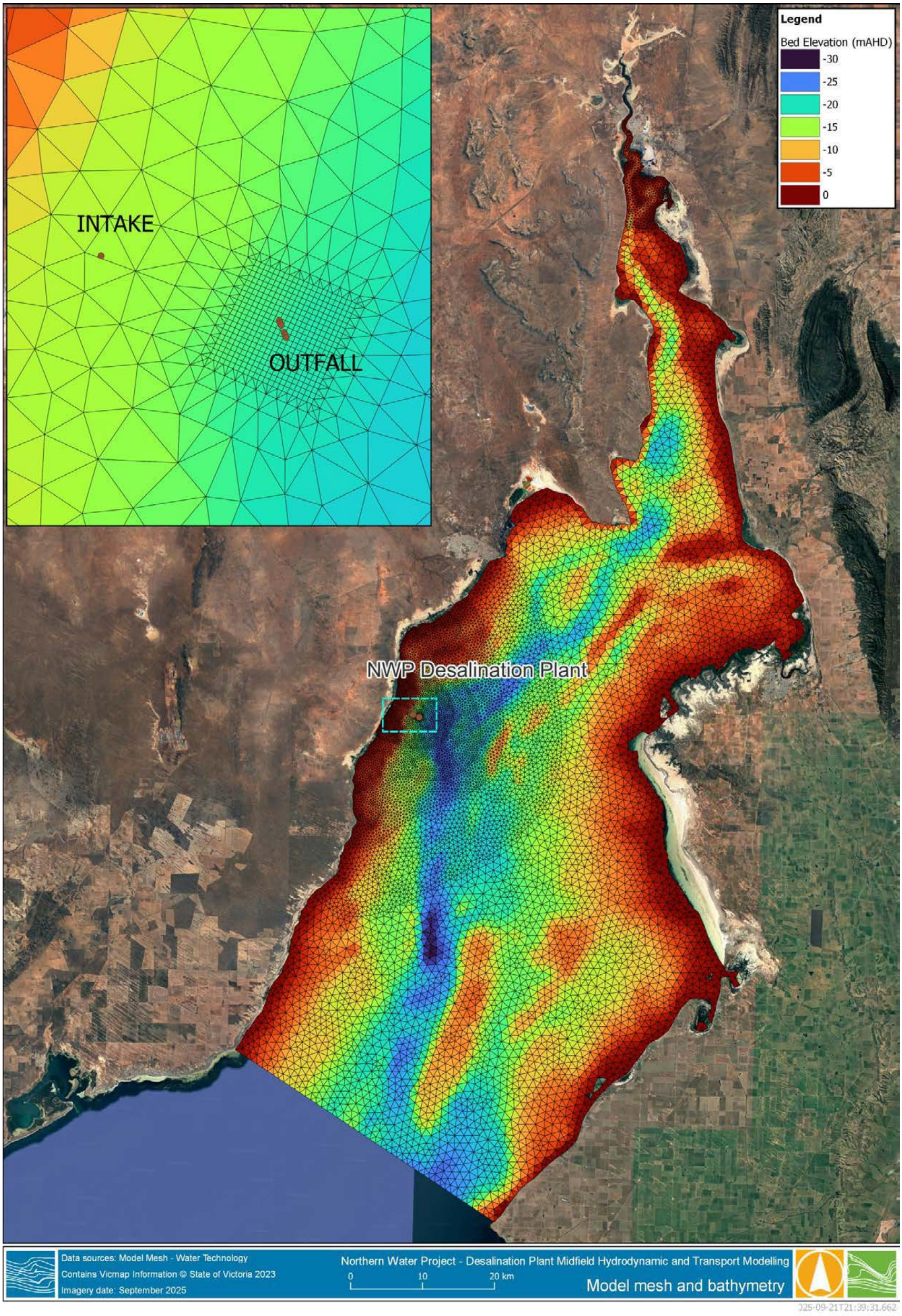


Figure 3-1 Model Domain, Mesh and Bathymetry

3.4 Model Calibration

Model calibration was performed against water levels, currents, salinity and temperature available at NWP3 and SAMUSG.

3.4.1 Simulation Period

The following period was simulated:

- 1 November 2022 to 1 December 2023

This period covers a full year and provides time for model spin up as well as coverage of most of the NWP3 measurement period (until 9 May 2023). The period also allowed further comparisons with SAMUSG measurements for the remainder of the simulation.

3.4.2 Boundary Conditions

3.4.2.1 Ocean Forcing

Ocean forcing comprised prescription of the following variables at the southern model boundaries:

- Water levels; and
- Temperature and salinity vertical planes along the boundary extents.

Water levels at the end points of the south boundary and simulation period are shown in Figure 3-2 (2022-2023 simulation).

For salinity and temperature, surface and near bed timeseries for these parameters at the south boundary are shown in Figure 3-3 and Figure 3-4.

A Flather boundary condition (where velocities are also prescribed) was also attempted but did not result in material model improvements. Hence, velocities obtained from the SGFF model were not adopted at the boundaries.

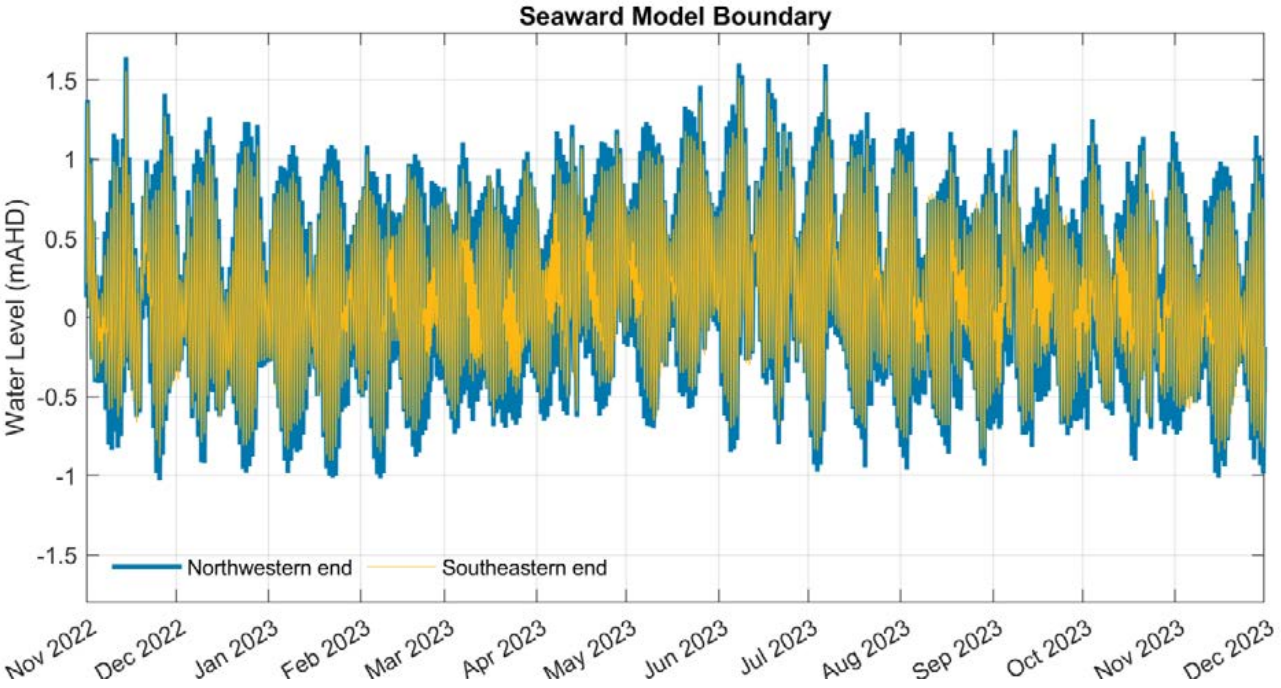


Figure 3-2 Water Levels at the South Boundary for the 2022-2023 Simulation

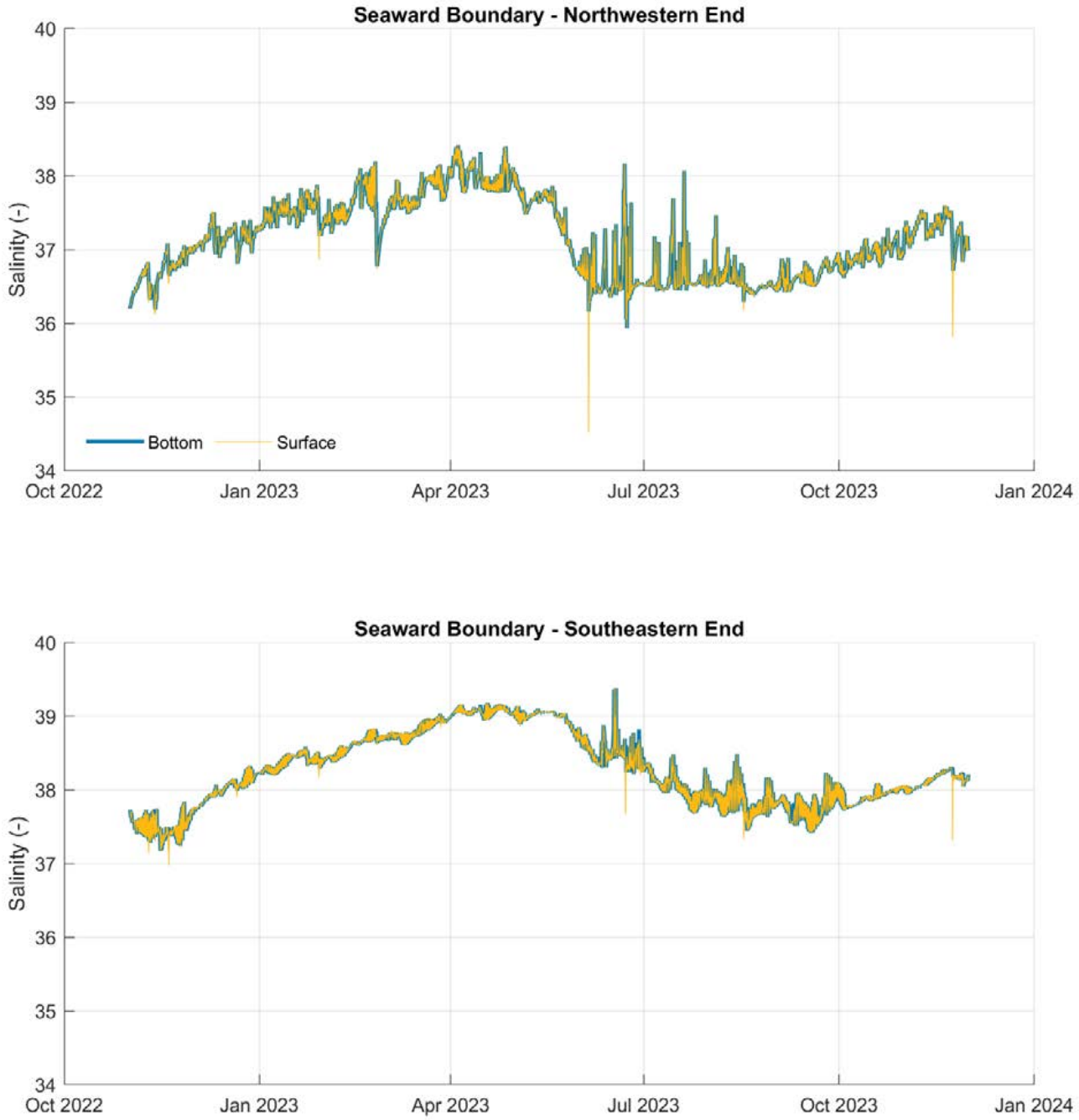
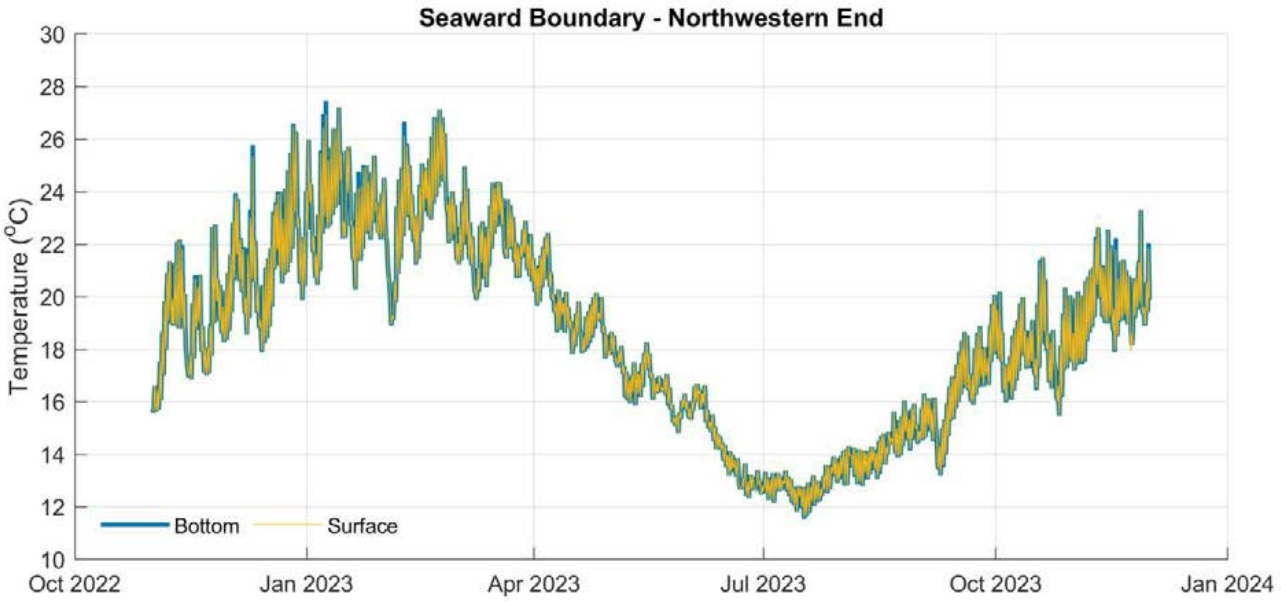


Figure 3-3 Salinity at the South Boundary for the 2022-2023 Simulation



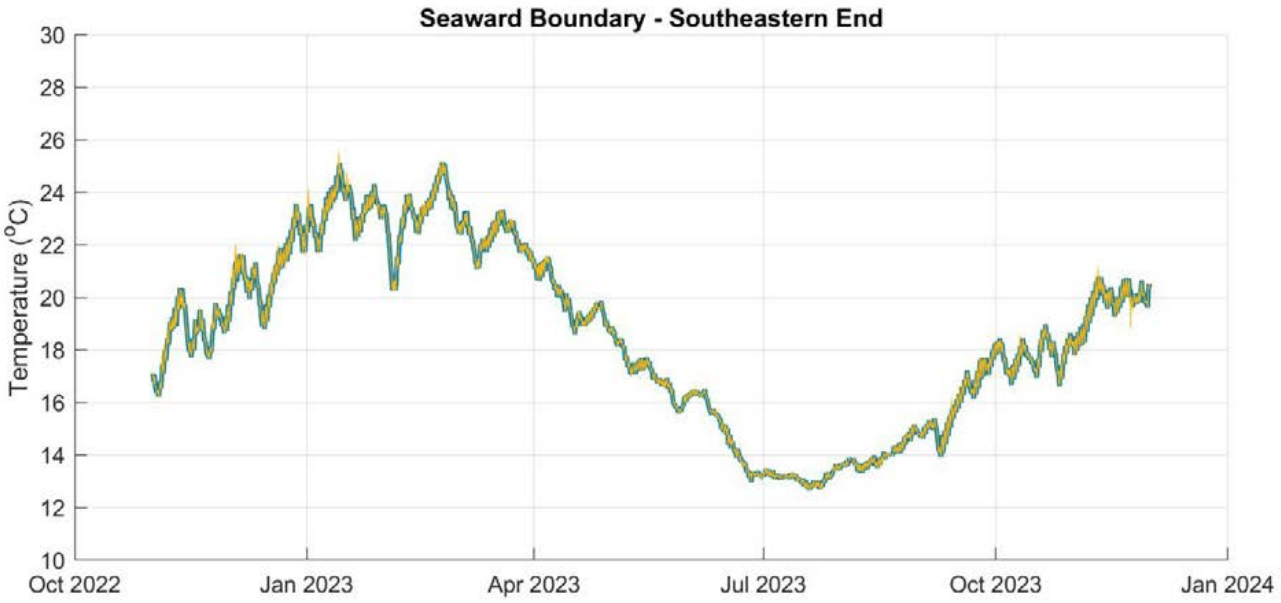


Figure 3-4 Water temperature at the South Boundary for the 2022-2023 Simulation

3.4.2.2 Meteorological Forcing

Meteorological forcing was applied at the model free surface to quantify wind shear stress effects and heat fluxes with the atmosphere. Data were sourced from the NOAA-NCEP Climate Forecast System Reanalysis v2 (CFS2, Saha et al. 2014) for consistency with the SGFF model. Applied data include the following:

- Wind velocity components (at 10 m above mean sea level);
- Mean sea level pressure;
- Air temperature (at 2 m above mean sea level);
- Downward short wave radiation (at mean sea level);
- Downward long wave radiation (at mean sea level); and
- Relative humidity (at 2 m above mean sea level).

The spatial resolution of the NOAA-NCEP data set is presented in Figure 3-5. The meteorological data at the nearest seawater location to the Northern Water Desalination Plant is shown in Figure 3-6.

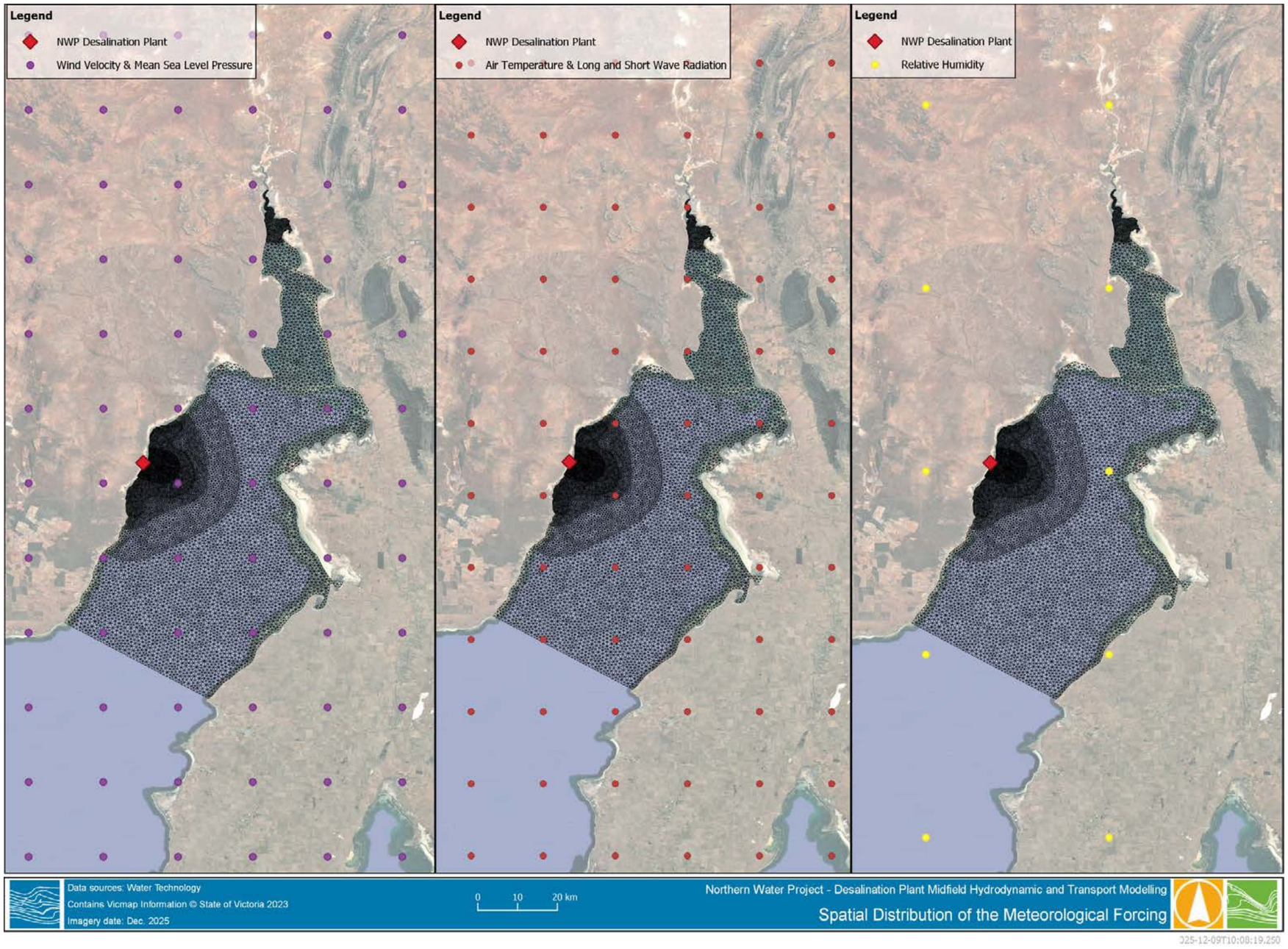


Figure 3-5 Spatial Distribution of the NOAA-NCEP Meteorological Forcing

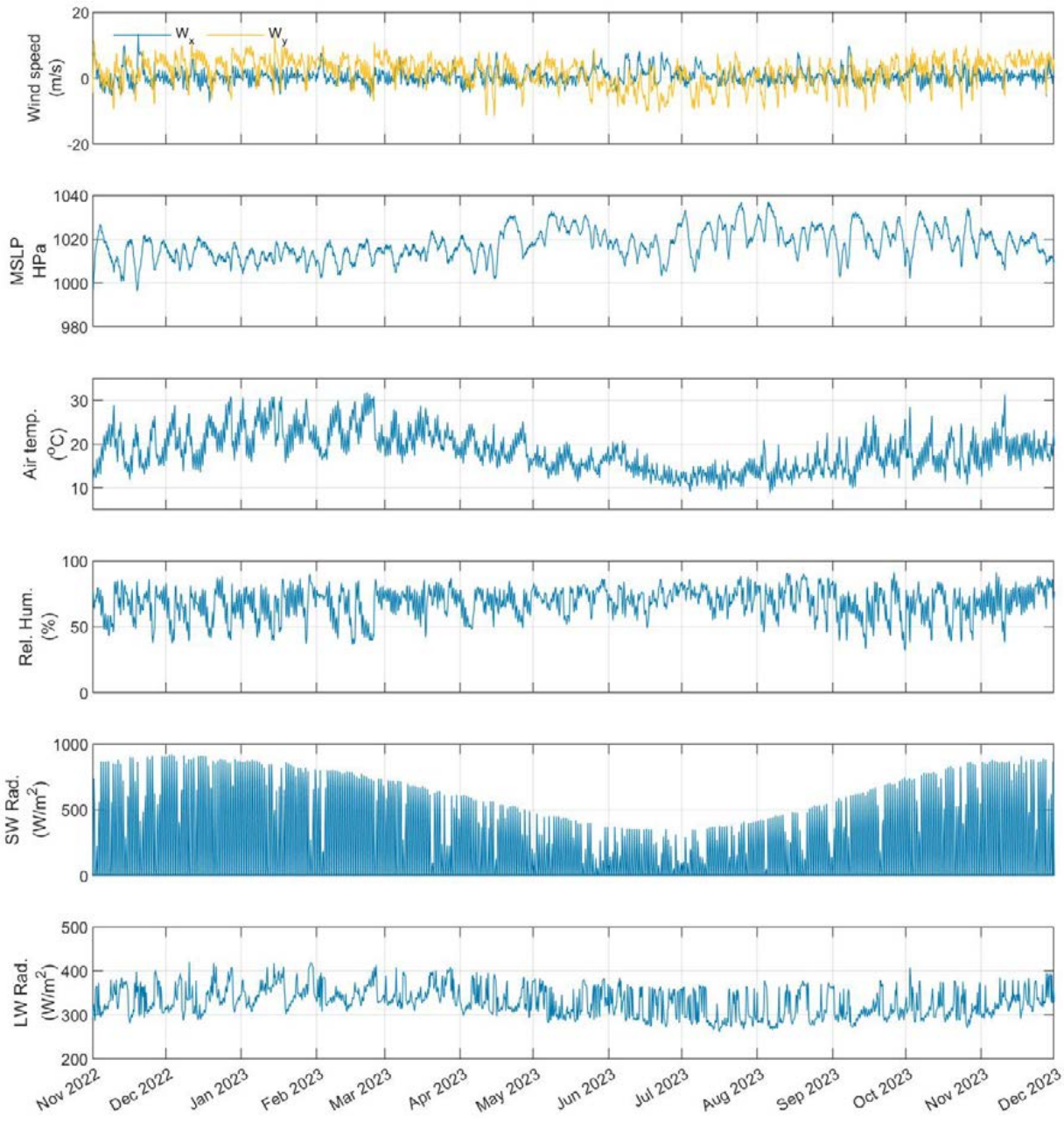


Figure 3-6 Meteorological Data at Seawater Location Nearest to the Northern Water Desalination Plant (2022-2023 Simulation)

3.4.3 Initial Conditions

Initial conditions for water levels and temperature were derived from three-dimensional linear interpolation of SGFF model results into the MIKE model domain. A similar process was adopted for salinity however, as model calibration progressed, salinity at NWP3 was underpredicted over the simulated period. It was identified that one of the reasons for this underprediction was initial salinity conditions in the Upper Spencer Gulf (i.e., north of Point Lowly), noting salinity results from the SGFF were lower than those of Nunes and Lennon (1986). On subsequent modelling, SGFF initial condition model data were adjusted to increase salinity at NWP3 to levels similar to field measurements. Additionally, Upper Spencer Gulf salinity north of Point Lowly was modified to be in the range observed in Nunes and Lennon (1986) between November 1982 and January 1983 (Figure 3-7).

The initial conditions obtained from the SGFF model, and the initial salinity condition modified based on Figure 3-7, are presented in Figure 3-8.

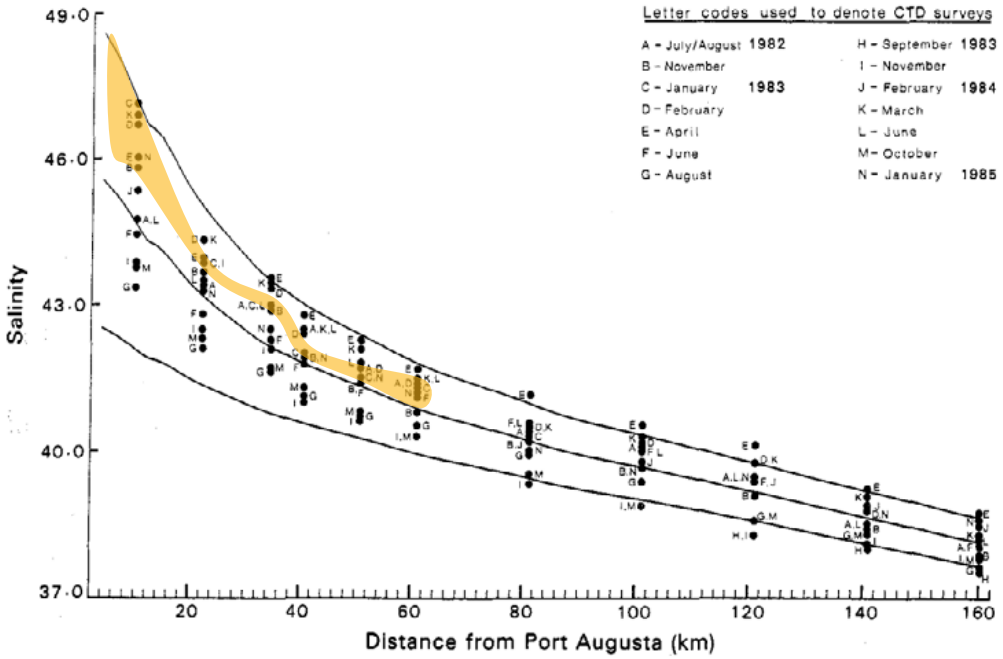


Figure 3-7 Reproduction of Figure 6 of Nunes and Lennon (1986): Mean cross-sectional salinities against distance from the head of the gulf in lettered sequence from the CTD surveys. Included are the mean longitudinal distribution and the envelope of salinity variation. Relevant values for adjusting initial conditions were from 0 to 60 km from Port Augusta as depicted in the yellow zone.

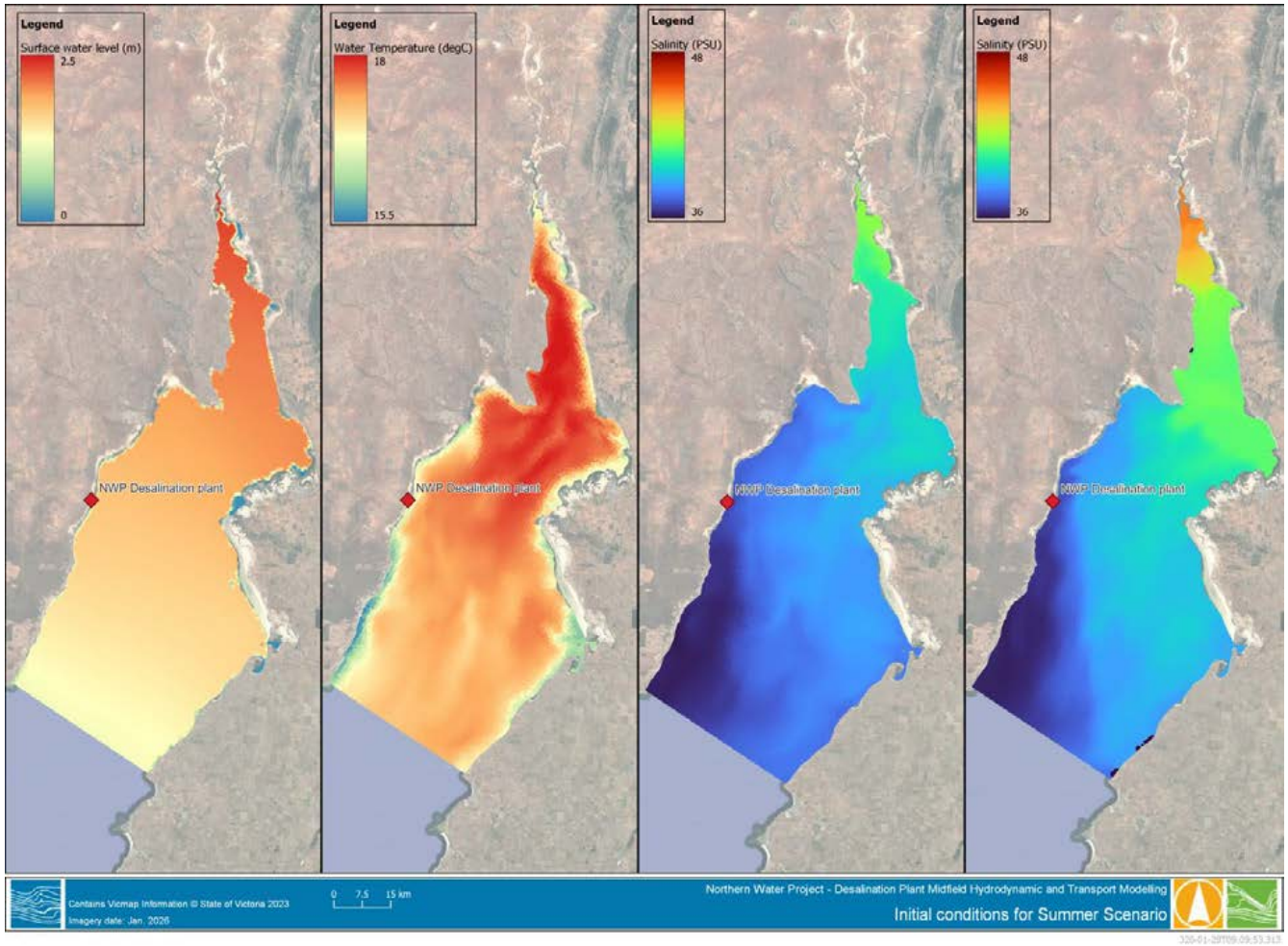


Figure 3-8 Initial Conditions for Summer Scenario (from left to right: surface water level, water temperature and salinity obtained from SGFF model, and salinity modified based on study of Nunes and Lennon)

3.4.4 Model Performance Metrics

Metrics commonly adopted to verify hydrodynamic model performance were used to assess model calibration. Adopted metrics are summarised in Table 3-1.

Table 3-1 Model Performance Metrics

Metric	Calculation
Mean Absolute Error (MAE)	$MAE = \frac{\sum_{i=1}^{n}(\lvert P_i - O_i \rvert)}{n}$
Root Mean Square Error (RMSE)	$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(P_i - O_i)^2}{n}}$
Bias	$BIAS = \frac{\sum_{i=1}^{n}(P_i - O_i)}{n}$
r^2 (coefficient of determination)	$r^2 = 1 - \frac{\sum_{i=1}^{n}(P_i - O_i)^2}{\sum_{i=1}^{n}(P_i - \bar{O})^2}$
IOA (Willmot Index of Agreement)	$IOA = 1 - \frac{\sum_{i=1}^{n}(P_i - O_i)^2}{\sum_{i=1}^{n}(\lvert(P_i - \bar{O})\rvert + \lvert(O_i - \bar{O})\rvert)^2}$

MAE, RMSE and BIAS quantify average dimensional errors and provide an indication of the model error magnitude. Being squared, RMSE provides more weight to larger discrepancies (i.e., outliers) and is most often larger than MAE. BIAS provides an indication whether a given variable is under (negative bias values) or overestimated (positive values).

R^2 and IOA and are non-dimensional performance metrics that vary between 0 and 1. A r^2 equals 1 indicates a perfect linear correlation between model and observations but can be overly sensitive in cases where there is little variance in the observed dataset, despite small residuals between predictions and observations. It is quite often used to infer the percentage of the variability that is explained by the model. In tidal datasets, changes in phase of a variable may lead to a poor model fit in terms of R^2. IOA attempts to mitigate this effect by including the observation variance in the denominator. In oceanographic application IOA > 0.5 is generally considered acceptable.

Combined, the above metrics provide a way to assess model performance and suitability of the model for the assessment.

3.4.5 Model Parameters

Hydrodynamic and advection-dispersion model parameters are summarised in Table 3-2 and MIKE ECO Lab parameters in Table 3-3.

Table 3-2 Model Parameters (MIKE HD and AD Modules)

Parameter	Value
Minimum time step	0.01 s
Maximum time step	300 s
Critical CFL condition	0.8
Drying depth	0.05 m
Wetting depth	0.10 m
Density calculation	UNESCO Equation
Horizontal eddy viscosity	Smagorinsky with constant equals 0.28
Vertical eddy viscosity	Two-equation κ-ε model with default parameters
Coriolis effects	Included
Horizontal dispersion coefficient	Scaled on horizontal eddy viscosity by 1.4
Vertical dispersion coefficient	Scaled on vertical eddy viscosity by 1.4
Bed resistance model	Bed roughness height
Bed resistance parameter	0.05 m
Wind drag model	Constant
Wind drag coefficient	0.001255
Short wave radiation albedo	0.6
Light extinction coefficient	0.5 m^{-1}
Sensible heat transfer coefficient	0.0011
Dalton law's constant (latent heat transfer)	0.5
Dalton law's wind coefficient (latent heat transfer)	0.9

Table 3-3 Particle Tracking Model Parameters (MIKE ECO Lab Module)

Parameter	Value
Model update frequency	600 s
Integration method	Euler
Particle density	Neutrally buoyant
Particle settling velocity	0 m/s
Horizontal dispersion	Eddy viscosity formulation
Vertical dispersion	Eddy viscosity formulation
Drift profile	Raw hydrodynamics

3.4.6 Calibration Results

Calibration was performed in terms of visual inspection and calculation of performance metrics for comparisons between measured and modelled water levels and currents (tidal and residual). Comparisons were also performed for salinity and temperature timeseries. Additional plots with increased temporal resolution are provided in Appendix C (water levels) and Appendix D (water currents). Presentation of horizontal model results (i.e., sheets) were also explored to demonstrate modelled hydrodynamic features.

3.4.6.1 Water Levels

Water levels and associated performance metrics at NWP3 and SAMUSG are shown in Figure 3-9 and Figure 3-10, respectively. Performance metrics demonstrated agreement between model and measurements, with MAE and RMSE being respectively ~4% and ~7% (or less) of the measured level range. Insignificant biases were obtained and an excellent correlation (r^2 and IOA) for full, tidal and residual water level signals was achieved, indicating phase correspondence between model and measurements in the tide-dominated signal. Correlation (r^2) between measured and modelled residual signals were, as expected, lower than the full and tidal signals but still displayed good correlation between model and measurements ($r^2 = 0.80$).

3.4.6.2 Currents

Comparisons between simulated and measured currents at the NWP3 site are shown in Figure 3-11 (depth-averaged) and Figure 3-12 (lower 1/3 of the water column). Similarly to water levels, performance metrics for currents at NWP3 demonstrated agreement between model and measurements, with MAE and RMSE being respectively <7% and <9% of the measured current magnitude range. Model bias was similar between the full signal and the residuals, while insignificant for the tidal signal. This result indicates the model resolved the tidal components well (with principal component axis less than 3° for both depth-averaged and lower 1/3 of the water column signals). Excellent correlation (r^2 and IOA > 0.8) for full and tidal signals was achieved, indicating phase correspondence between model and measurements in the tide-dominated signal. Correlation (r^2) between measured and modelled residual signals was low; however with small bias values and a small range, the simulated and measured values are consistent and unlikely to have a significant influence on the model. This result was largely influenced by the much smaller variance in comparison to the full signals and temporal resolution of the meteorological forcing (6 hours), which also affected far field model results in a similar fashion (Doubell and James, 2026).

Respective comparisons at SAMUSG (Figure 3-13 and Figure 3-14) show similar performance metrics for currents with demonstrated agreement between model and measurements. MAE and RMSE were respectively <8% and <10% of the measured current magnitude range. Model bias followed the same trend as the comparisons at NWP3 with principal component axis less than 5° for both depth-averaged and lower 1/3 of the water column signals. Excellent correlation (r^2 and IOA > 0.9) for full and tidal signals was also achieved, indicating phase correspondence between model and measurements in the tide-dominated signal. Again, correlation (r^2) between measured and modelled residual signals improved in comparison to NWP3 (r^2 >0.6 and IOA ≥ 0.7) but were relatively low in comparison to the full and tidal signals (same reasons as for NWP3 comparisons).

It is noted that despite the agreement, the model showed a degree of underprediction at SAMUSG in comparison to measurements at NWP3. This may be explained by the coarser mesh in the area of the SAMUSG station. Additionally, small deviations in deployment locations over the 6 different deployment periods may contribute to small scale spatial variations not depicted in the model (e.g., clear differences in range between December 2022 to February 2023 period and February to May 2023 period are present in the SAMUSG data).

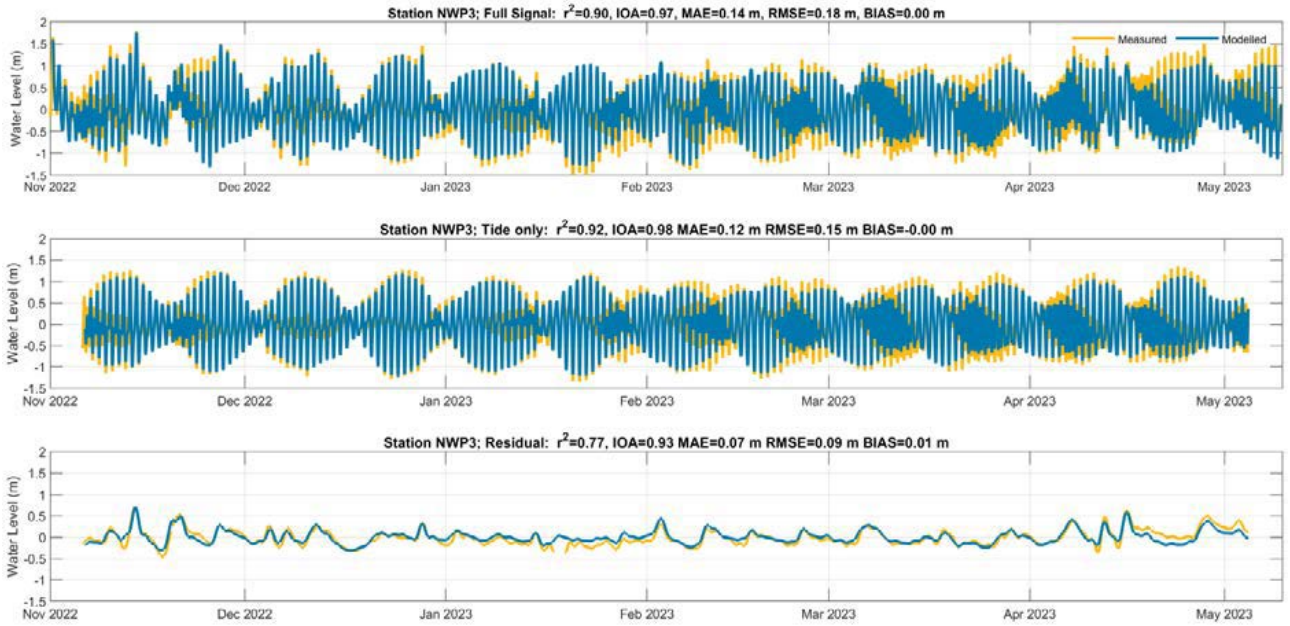


Figure 3-9 Surface Water Level Calibration at NWP3 Station

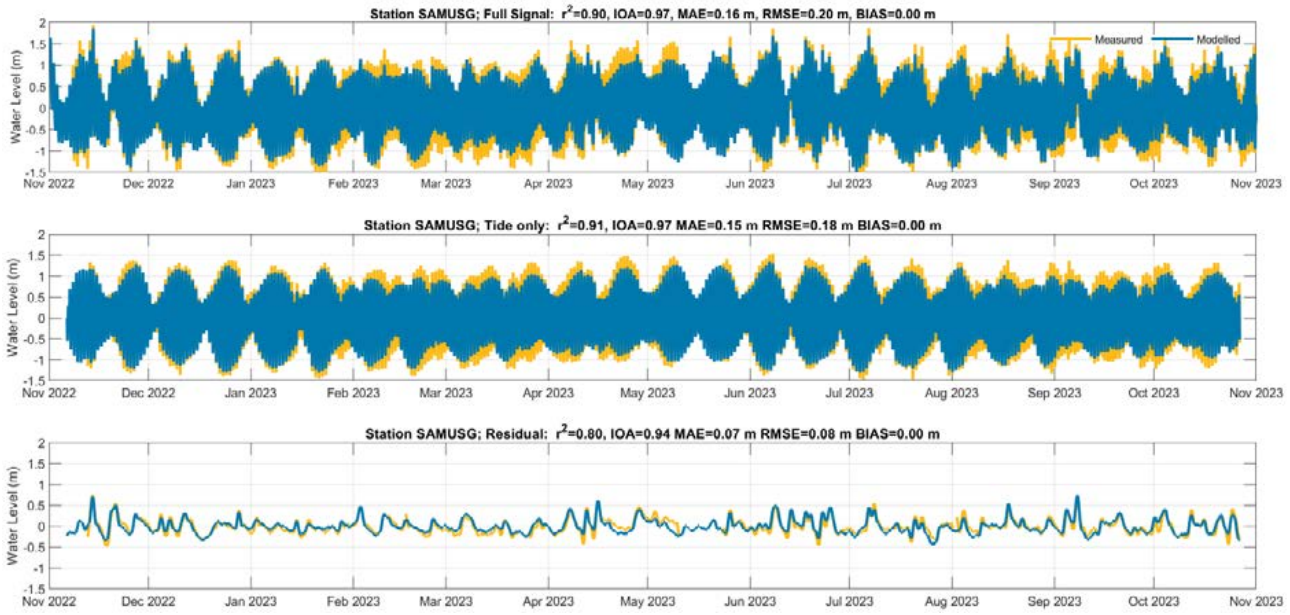


Figure 3-10 Surface Water Level Calibration at SAMUSG Station

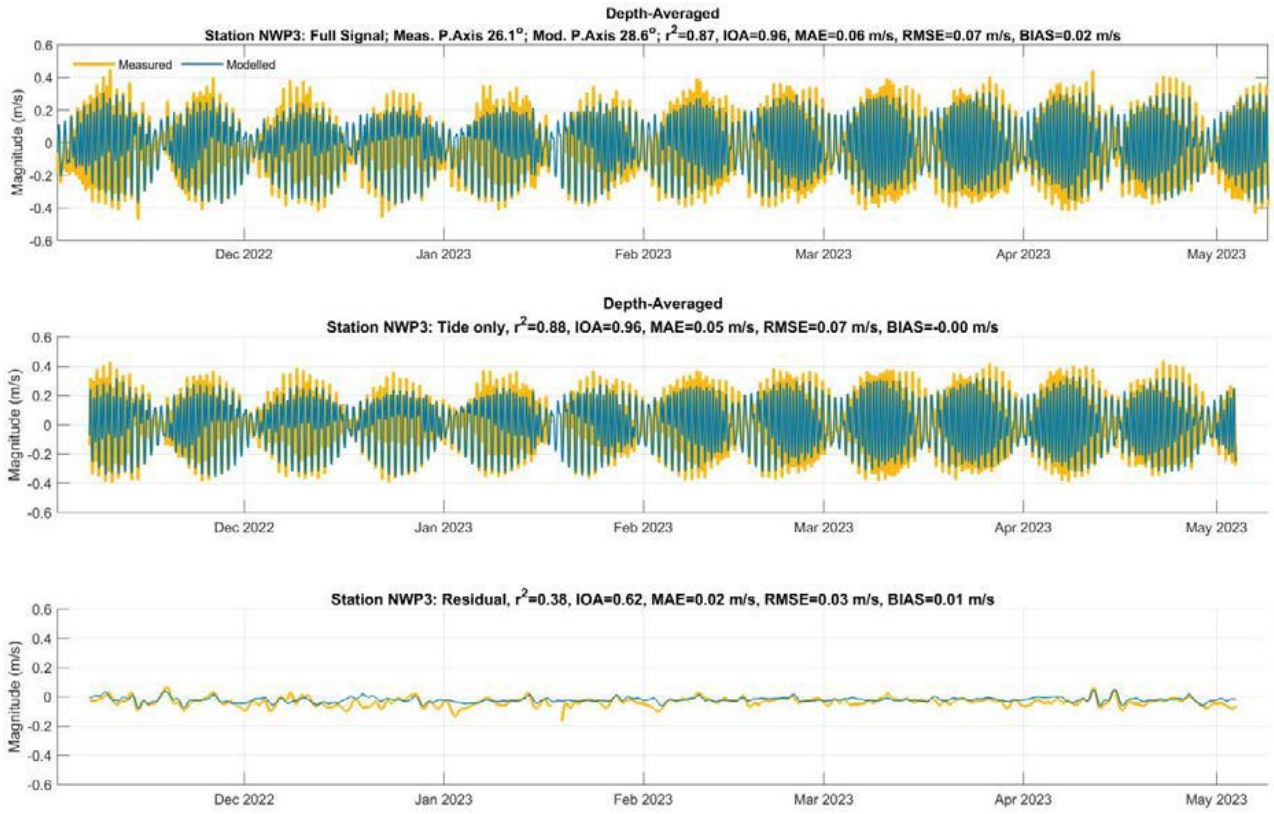


Figure 3-11 Depth-Averaged Currents at NWP3 Station

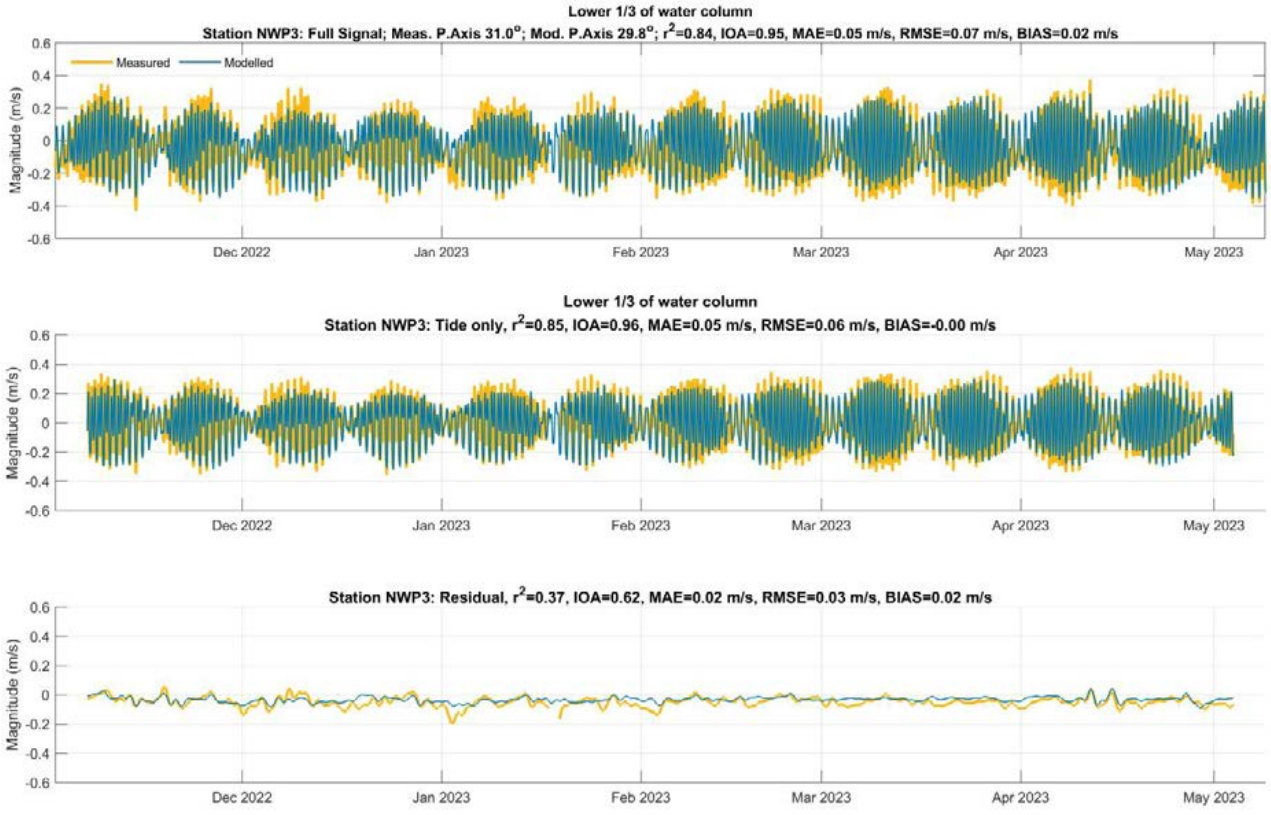


Figure 3-12 Lower 1/3 of the Water Column Currents at NWP3 Station

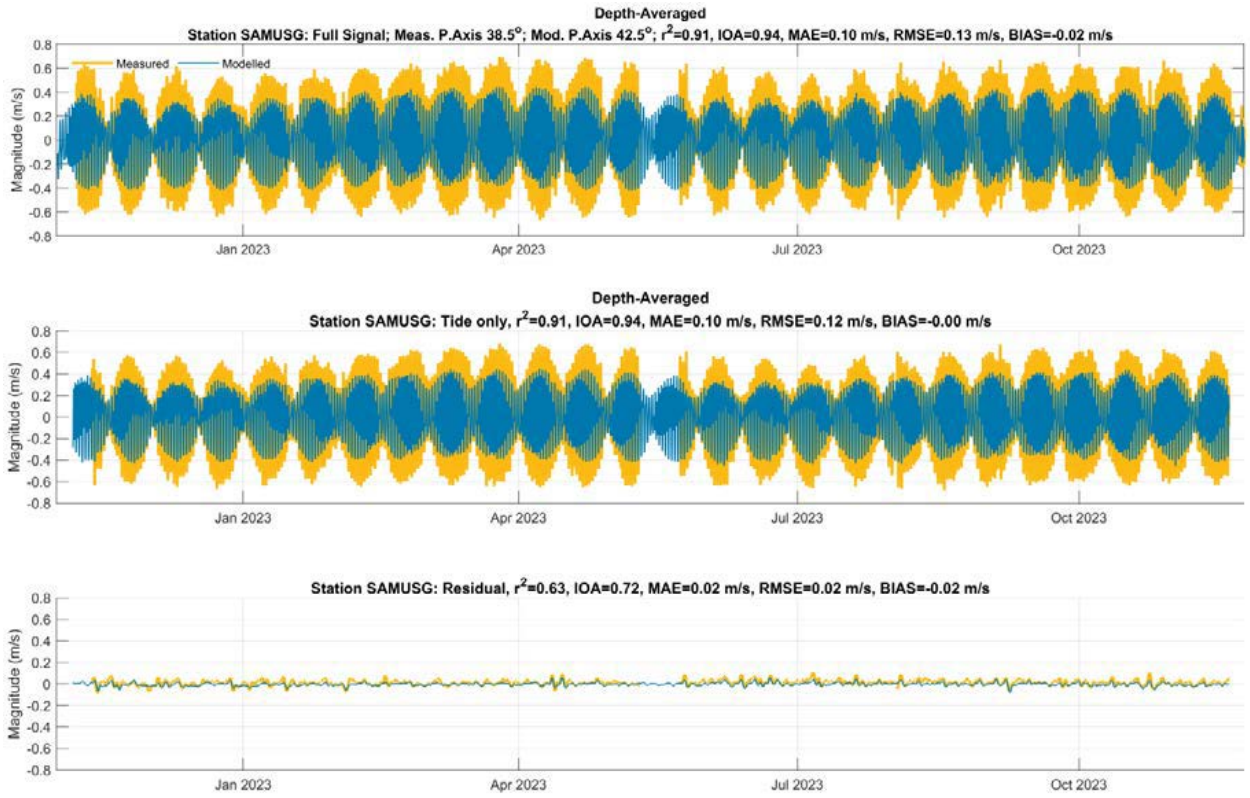


Figure 3-13 Depth-Averaged Currents at SAMUSG Station

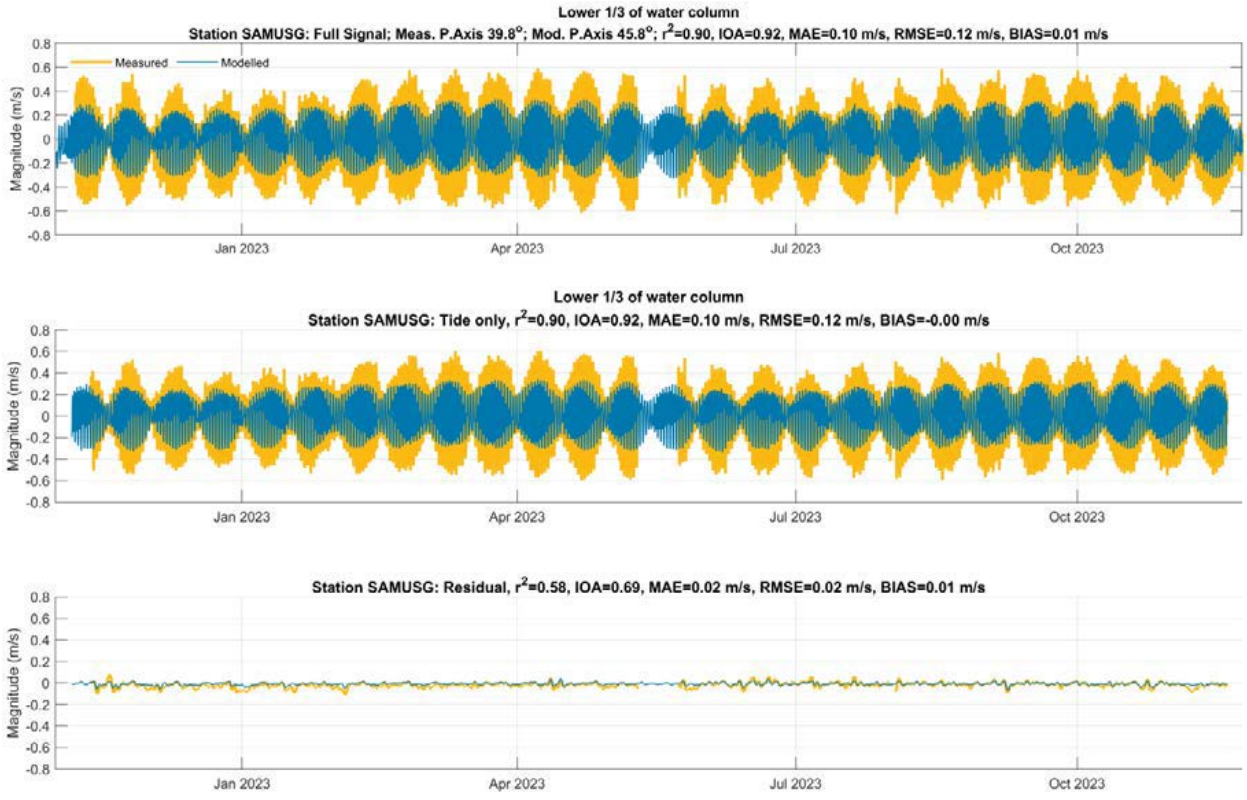


Figure 3-14 Lower 1/3 of the Water Column Currents at SAMUSG Station

3.4.6.3 Salinity and Temperature

Comparisons of salinity and temperature at the NWP3 and SAMUSG sites are shown in Figure 3-15 and Figure 3-16, respectively. Performance metrics for temperature show excellent agreement between the data with both r^2 and IOA ≥ 0.9, small bias and mean errors (≤ 1°C). The model displays the cooling trend in winter and progressive heating throughout spring into summer. Biases across both stations reflect slight model overprediction in the summer months.

The model reproduced the increasing salinity trend from summer into autumn, however it did not display the same performance metrics as temperature. This was not entirely unexpected noting the relatively small range (variance) of the measured salinity dataset in comparison to temperature.

At NWP3, r^2 was equal 0.80 and IOA equals 0.93, which indicated a good level of correlation between model and measurements. Mean salinity errors were small and ≤ 5% of the measurement range. Compared with the expected salinity change to be imposed by the desalination plant discharge, the level of model error is small and expected to have little bearing in the model's ability to simulate the extent of impact areas.

Measurements at SAMUSG also reproduced the seasonal trends with increasing salinity over summer and autumn and a decreasing trend over winter. The rate of decrease of the model salinity in winter, however, was lower than the measurements. As a result, correlation (r^2 = 0.34) and index of agreement (IOA = 0.45) were relatively low. The reasons for the lesser correlation at SAMUSG were in part related to uncertainties in the definition of initial conditions in the Upper Spencer Gulf (see Section 3.4.3), which due to the proximity of the project location (Figure 1-1) had an increased influence at the local model performance. Notwithstanding the lower performance, the character of temporal fine scale features of salinity (i.e., semi-diurnal, and spring-neap tidal oscillations) at SAMUSG from June 2023 onwards improved in comparison to the prior period (noting the lower seasonal rate of salinity decrease).

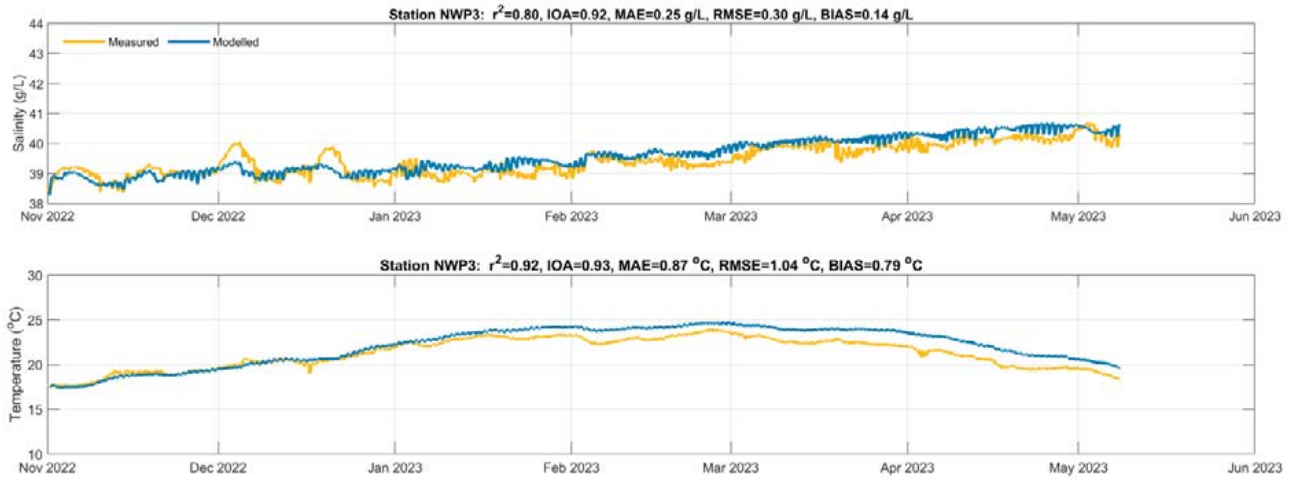


Figure 3-15 Salinity and Temperature Calibration at NWP3 Station

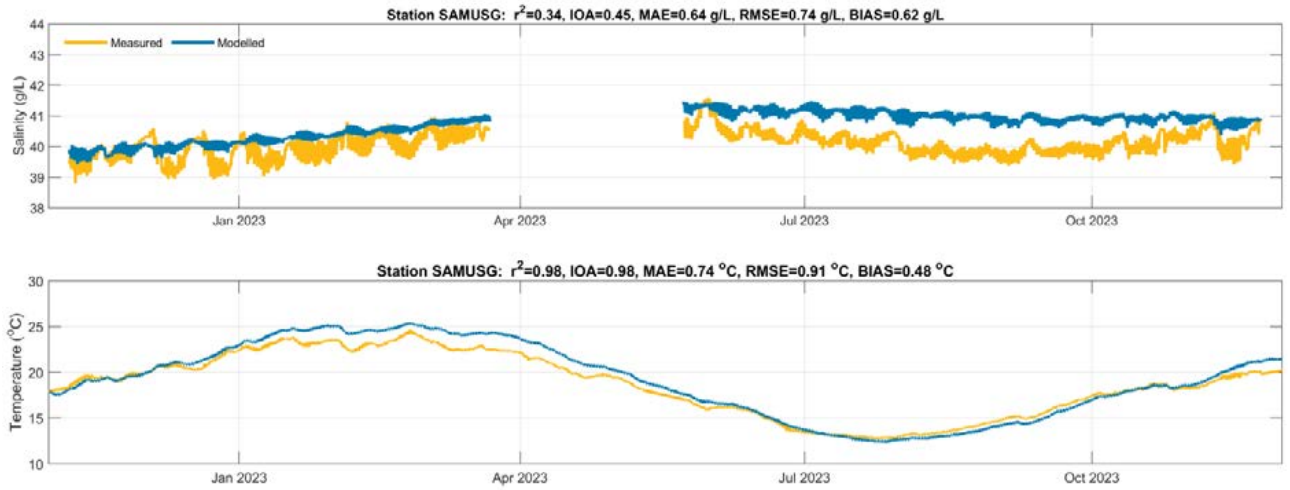


Figure 3-16 Salinity and Temperature Calibration at SAMUSG Station

3.4.6.3.1 Salinity Pulses

The comparisons at NWP3 indicate the model replicated measurements well early in November with salinity increasing during neaps. Whilst also showing high salinity increases through December, the salinity increase in the model was not as pronounced as observations. It is therefore important to discuss how these pulses originate, and how underestimation of these salinity pulses may influence the prediction of the desalination plant mixing zone and associated environmental effects.

Nunes Vaz et al. (1990) observed the salinity pulsing features and postulated it characterises the salt ejection mechanism from the Upper Spencer Gulf, briefly described as follows. As summer approaches and temperatures rise, dry conditions towards the head of Spencer Gulf intensify evaporation rates. The tidal amplitudes during spring tides can generate enough turbulence and mixing in the water column to suppress vertical salinity (and associated density) stratification. However, during neap tides, and provided low wind conditions support it, turbulence is supressed, and salinity increases towards nearshore shallow areas. With time, differential heating builds density gradients towards the shore, which eventually relax moving saline waters to deeper waters offshore. Due to higher evaporation rates at the head of the Gulf, this process intensifies towards the north in such a way the salinity gradients also occur latitudinally, forming a density current moving south. Figure 2-7 shows the salinity increase at both SAMUSG and NWP3, with salinity peaks occurring with up to a couple of days delay at NWP3 in comparison to SAMUSG.

As tides intensify towards springs, vertical salinity gradients reduce with improved mixing conditions and, as a result, the gravitational adjustment driving the density currents subside. Salinity then reduces from the peak salinity during neap tides (as evidenced near the bed at NWP3 and SAMUSG, Figure 2-7) and salinity oscillations (not to the same extent as the peaks) then recover a semi-diurnal pattern with the more vigorous tidal excursion (Figure 3-17). This process modulated by the tidal cycles leads to the highest salinities observed at the end of autumn shown in Figure 2-7. Influenced by Earth's rotation, the high salinity waters travel south and move preferentially on the eastern side of the Gulf before outflowing into the Southern Ocean at the Gulf's mouth. This cycle ensure salinity does not keep increasing year after year in the Gulf, despite observed inter-annual variability.

Salinity build up during neaps near the discharge location is shown to occur over approximately 4 days, with reduction to a background level approximately 0.8~1.0 g/L lower than the peak at NWP3 (Figure 3-17). Over the initial building up period, characteristic of low amplitude neap tides, the numerical model predicted well the tidal currents movement but less so their residuals, which were 0.10~0.15 m/s magnitude and dominated the measured current signal (Figure 3-17). As the tidal amplitude increased and the tides gained amplitude, the residual (and its contribution to flow) reduced.

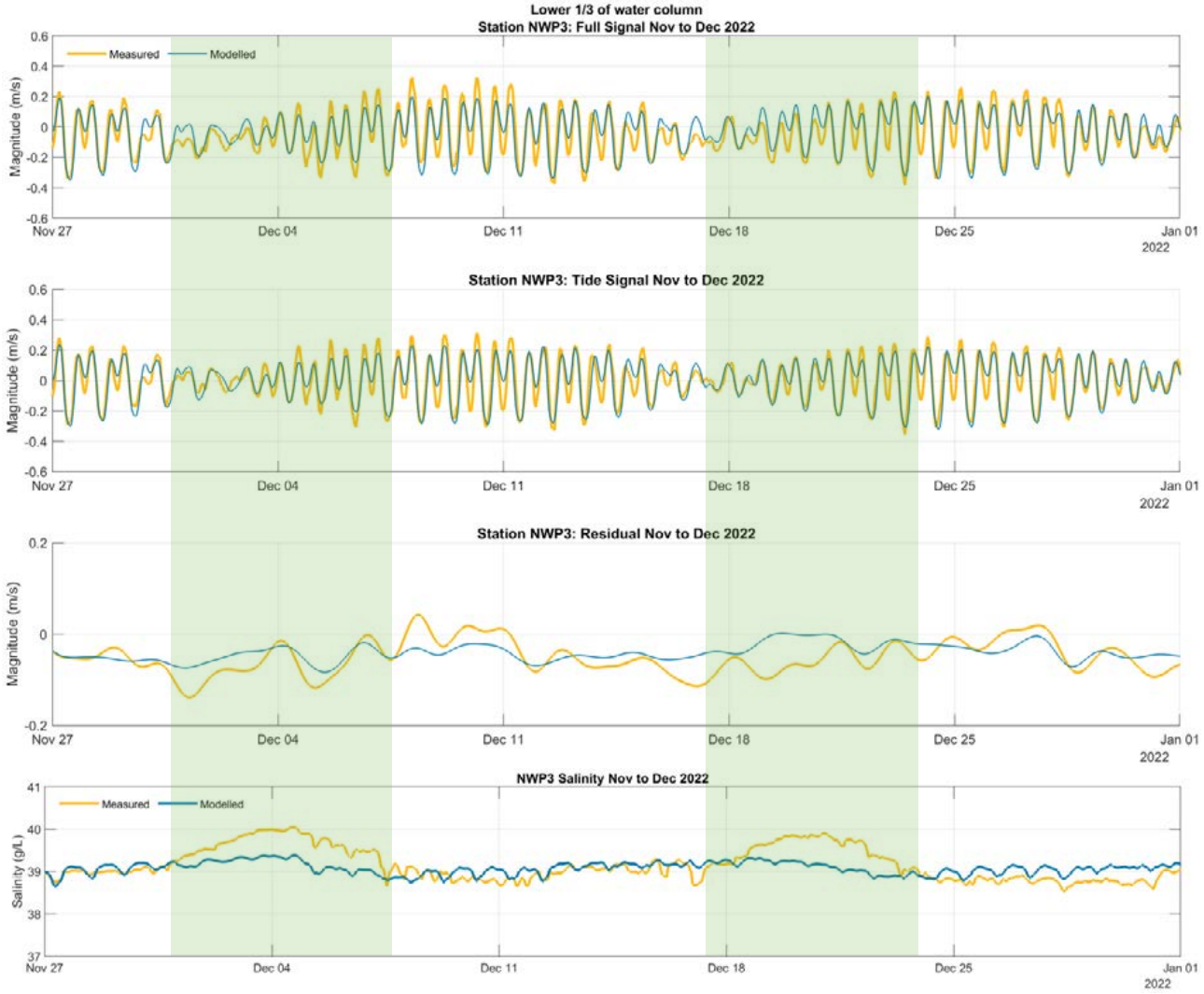


Figure 3-17 Measurement and Model Comparisons at NWP3 over the Salinity Pulses Period. Green shaded area highlights the events.

Noting the effects of the return water are assessed in terms of:

1. Salinity changes (increases) in relation to background; and
2. Level of dilution defining species protection trigger levels (Section 2.6),

the following can be expected from predictions of the desalination plant discharge plume. Modelled ambient currents have a lesser magnitude in comparison to the field measurements. As a result, the expected degree of mixing in the nearfield will be less significant in the predictions due to less energy available for mixing (see nearfield modelling described in Water Technology 2026), with significant reduced mixing under quiescent conditions even when compared to low current speeds (i.e., currents lower than 0.20 m/s).

In terms of prediction of salinity effects, return water salinity in the model was assumed to be equivalent to the one from drawing seawater at the highest recorded ambient salinity (~41.7 g/L, Table 2-4). While the model did not predict the same extent of ambient salinity increase over these salinity pulse events, the comparisons against background levels were based on a higher discharge salinity than would occur in reality. It is therefore anticipated that model predictions in terms of both dilutions and salinity changes were conservative, as they would be respectively lower (dilution) and higher (salinity change) than what would be expected for the measured conditions.

Notwithstanding the above, model predictions would likely produce a plume with reduced movement during ebb tides. As a result, effects may be overestimated to the north and underestimated to the south of the desalination plant outfall. Noting the conservative nature of the predictions, the model was considered adequate for prediction of the effects of the discharge. The quantification of these effects is provided in Section 4.

3.5 Current Patterns

Results of the calibrated (2022-2023) model were used to derive maps of currents under different tidal conditions. Of relevance are different tidal periods (floods vs. ebbs, low vs. high water, springs vs. neaps) and 'dodge' tide periods in which water currents remain low for longer durations. Current maps largely reflected descriptions provided above, however also offered an opportunity to verify spatial current patterns in the model.

3.5.1 Spring Tides

Example currents over a spring tide period are shown in Figure 3-18 to Figure 3-21, taken approximately 4 hours apart.

Starting at high tide (Figure 3-18), in areas adjacent to the intake and outfall pipeline alignment, currents were characterised by low magnitudes with returning currents forming to the north of the pipeline alignment. The currents veered to the northwest direction indicating the turning of the tide. At the outfall and intake locations, currents were approximately perpendicular to the outfall and intake pipeline (moving northeast) and of the order of 0.12 m/s and 0.08 m/s, respectively. Currents offshore were slightly higher, up to 0.15 m/s.

At the running ebb tide (Figure 3-19), currents approached the intake and outfall areas (including pipeline) from the north and northeast with the highest velocities between -10 and -20 mAHD isobaths. Velocities increased offshore and towards the south; and were lowest inshore. At the intake and outfall sites, currents were considerably higher than at the high tide with values of 0.18 m/s to 0.22 m/s.

At low tide (Figure 3-20), currents approached from the south veering from a north direction nearshore to northeast as it approached the -5 m isobath. In contrast, currents offshore at the −15 mAHD isobath flowed southwest and veered towards the west between the −5 and −15 mAHD isobaths within the area near the intake and outfall pipeline alignments. Returning currents flowing north occurred inshore between the −5 and −15 mAHD isobaths south of the pipeline alignments. At the outfall and intake sites, currents were directed towards the southwest, with magnitudes ranging from 0.03 m/s to 0.14 m/s.

At the running flood tide (Figure 3-21), currents tended to be low and without a clear direction pattern in the nearshore areas surrounding the pipeline alignments (shallower than the -5m AHD isobath). Flows offshore of the -20m AHD isobath were comparatively stronger towards the northeast. At the outfall and intake sites, currents were approximately perpendicular to the tunnel alignments, directed towards the northeast, with magnitudes around 0.20 m/s.

3.5.2 Neap Tides

Example of currents over a neap tide period are shown in Figure 3-22 to Figure 3-25, taken approximately 5 hours apart.

Starting at high tide (Figure 3-22), currents nearshore (inshore of the -5m AHD isobath) flowed towards the southwest. Transitional currents occurred in the areas between the −5 and −15 mAHD isobaths, while currents offshore of the −15 mAHD isobath were directed northeast (i.e., flooding direction). At the intake and outfall sites, currents were approximately northward with magnitudes in the range of 0.07 m/s to 0.10 m/s.

At the running ebb tide (Figure 3-23), some areas near the coastline exhibited returning currents (i.e. flowing northwards), while currents across other areas generally flowed southward and southwest-ward. At the intake and outfall sites, current velocities were still within the same range of 0.07 to 0.10 m/s.

At the low tide (Figure 3-24), currents were generally of low intensity (< 0.05 m/s) transitioning into flooding behaviour, with general northward flow nearshore and towards the shore at both outfall and intake locations with intensities below 0.025 m/s. Flow patterns were less discernible in the areas along the pipeline alignment.

At the running flood tide (Figure 3-25), currents were generally stronger than previous descriptions, with clear preferential flow north and northeast. South of the pipeline alignments within areas inshore of the −5 mAHD isobath, currents patterns were less discernible. At the outfall and intake sites, currents were approximately perpendicular to the pipeline alignment, with velocities ranging from 0.14 m/s to 0.18 m/s.

3.5.3 'Dodge' Tides

'Dodge' tide periods (Figure 3-26 to Figure 3-29) can be thought of as a severe neap tide, however flow characteristics were dissimilar to the ones described above. In the cases described below, figures were from results taken 2 to 3 hours apart. Starting at low tide (Figure 3-26), currents generally approached from the east-northeast and veered southward south of the pipeline alignment, with low intensity currents (<0.05 m/s) nearshore. The strongest currents occurred offshore between the −10 and −15 mAHD isobaths, reaching approximately 0.1 m/s near the intake location. At the outfall location currents were weaker with westerly to southwesterly flows below <0.03 m/s.

At the flood tide (Figure 3-27), currents flowed in the opposite direction compared to low tide, generally towards the north or northeast. Low intensity currents occurred nearshore, with a lack of discernible pattern south of the pipeline alignments. At the intake and outfall sites, currents were approximately 0.05 m/s generally perpendicular to the pipeline alignments in a northerly to northeasterly direction.

At the high tide (Figure 3-28), current directions were generally consistent with those during the flood tide, albeit at a lower intensity. At the intake and outfall locations, currents were approximately perpendicular to the tunnel alignments, with velocities around 0.02 m/s.

At the ebb tide (Figure 3-29), returning currents began to shift southward with a clear pattern offshore compared to less discernible pattern nearshore of the −5 mAHD isobath, particularly south of the pipeline alignments. At the intake and outfall sites, currents remained approximately perpendicular to the pipeline alignments, with velocities below 0.02 m/s.

The situation described above indicates a low energy environment with reduced capacity for plume dispersion, which could last for a couple of days. Prior and ensuing periods, however, include the highest spring tide ranges over the annual cycle, which in turn help the redistribution of any accumulated return water plume. The characteristics of the plume for different operational scenarios are explored in the next section.

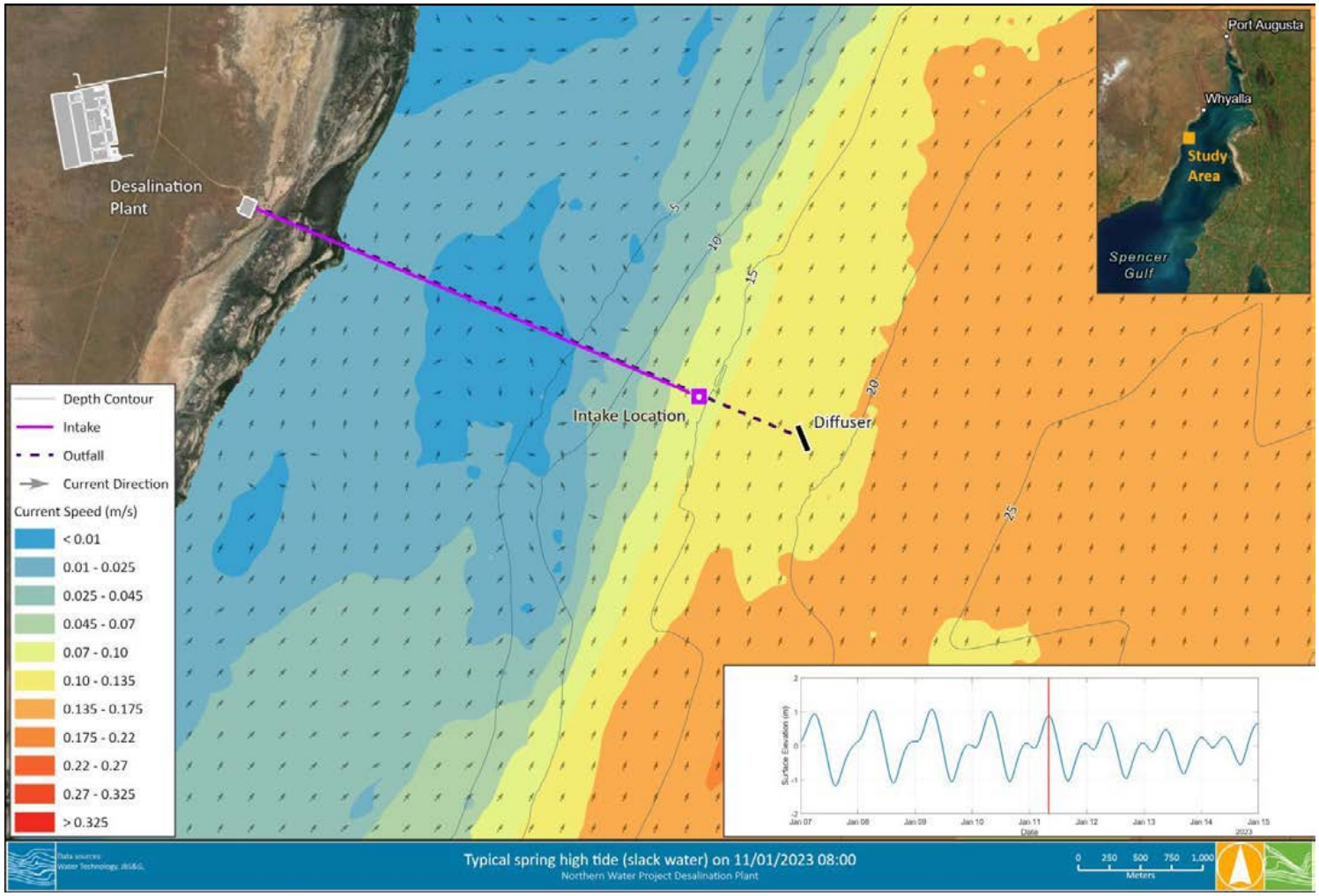


Figure 3-18 Example of Currents on a Spring High Tide

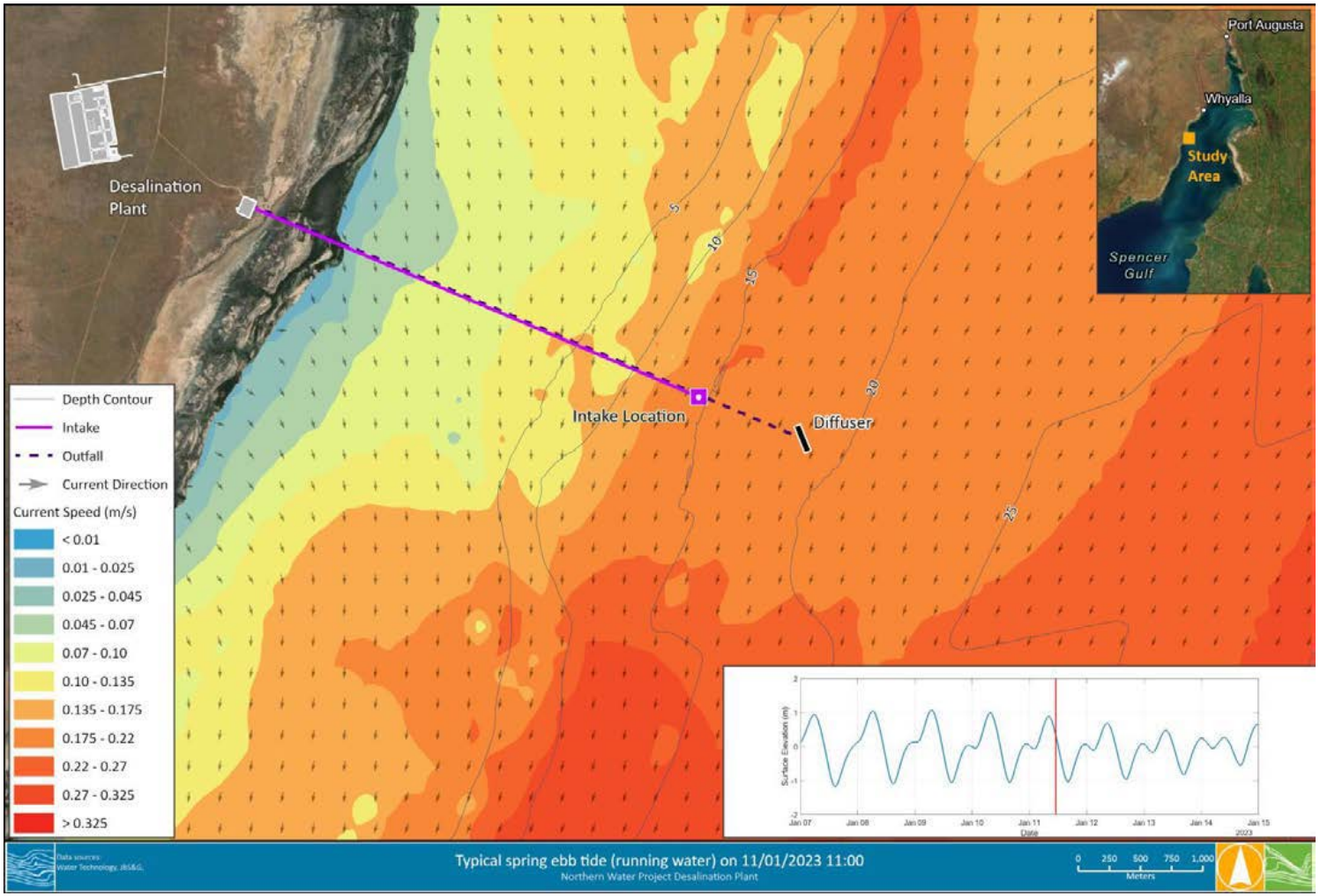


Figure 3-19 Example of Currents on a Spring Ebb Tide

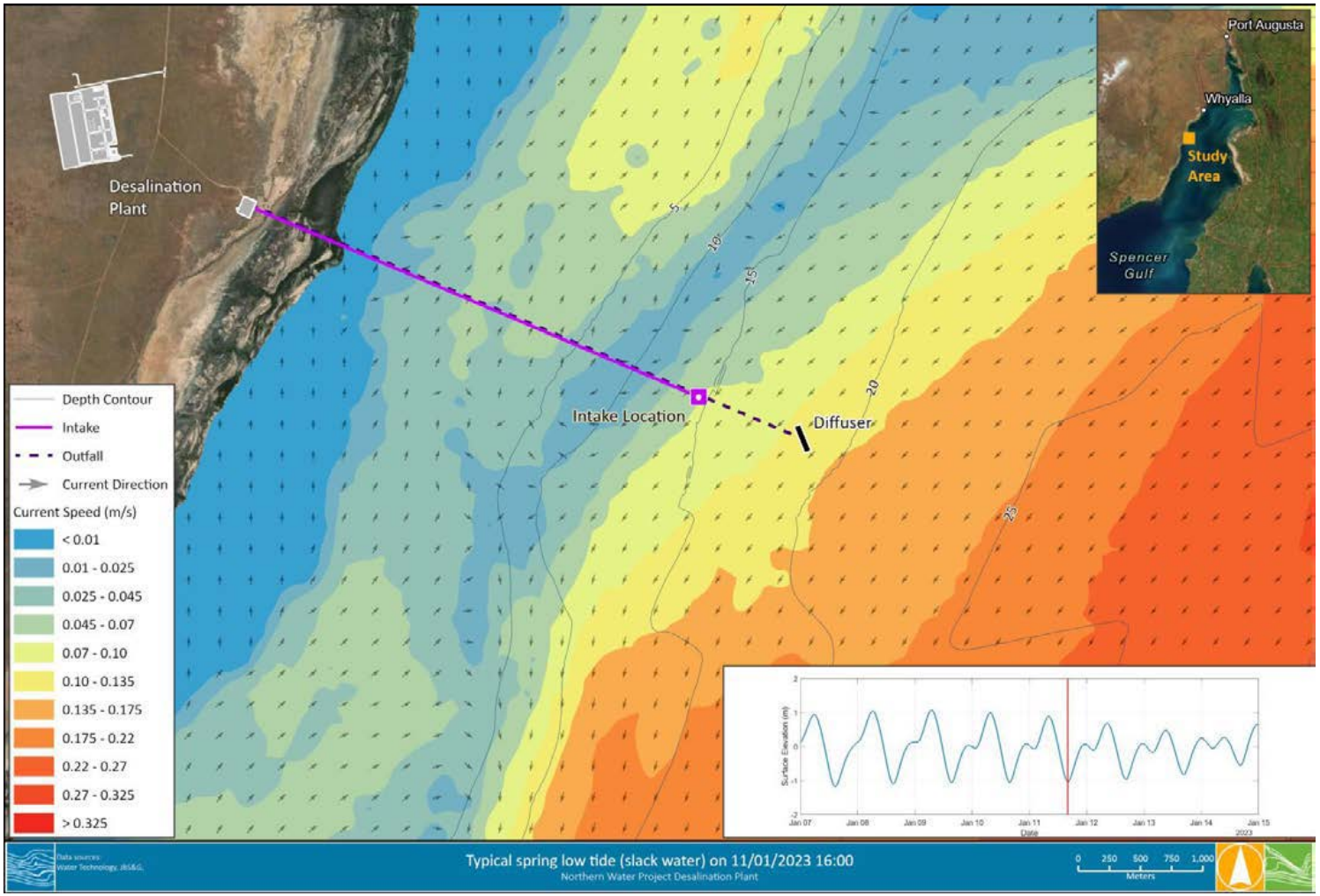


Figure 3-20 Example of Currents on a Spring Low Tide

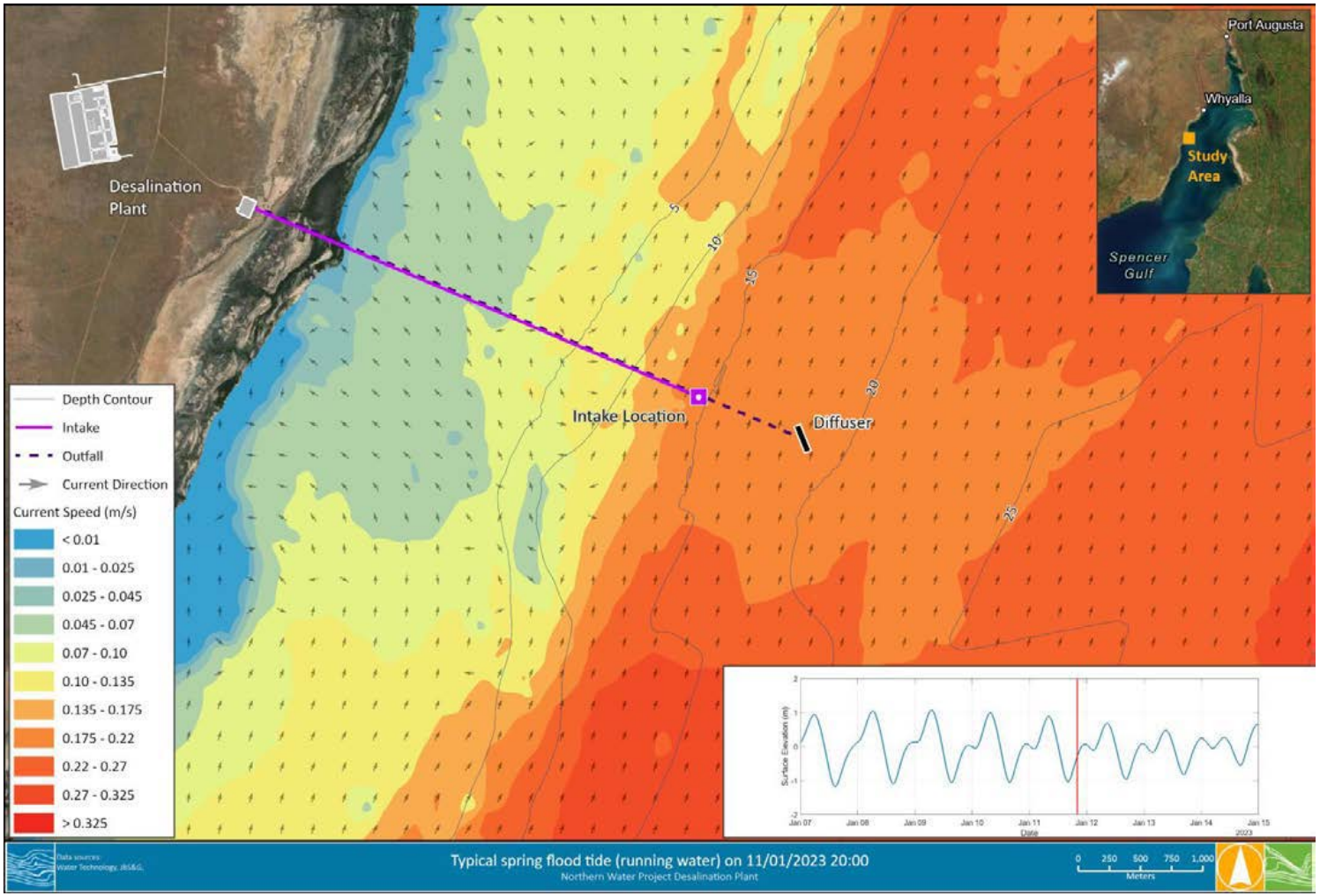


Figure 3-21 Example of Currents on a Spring Flood Tide

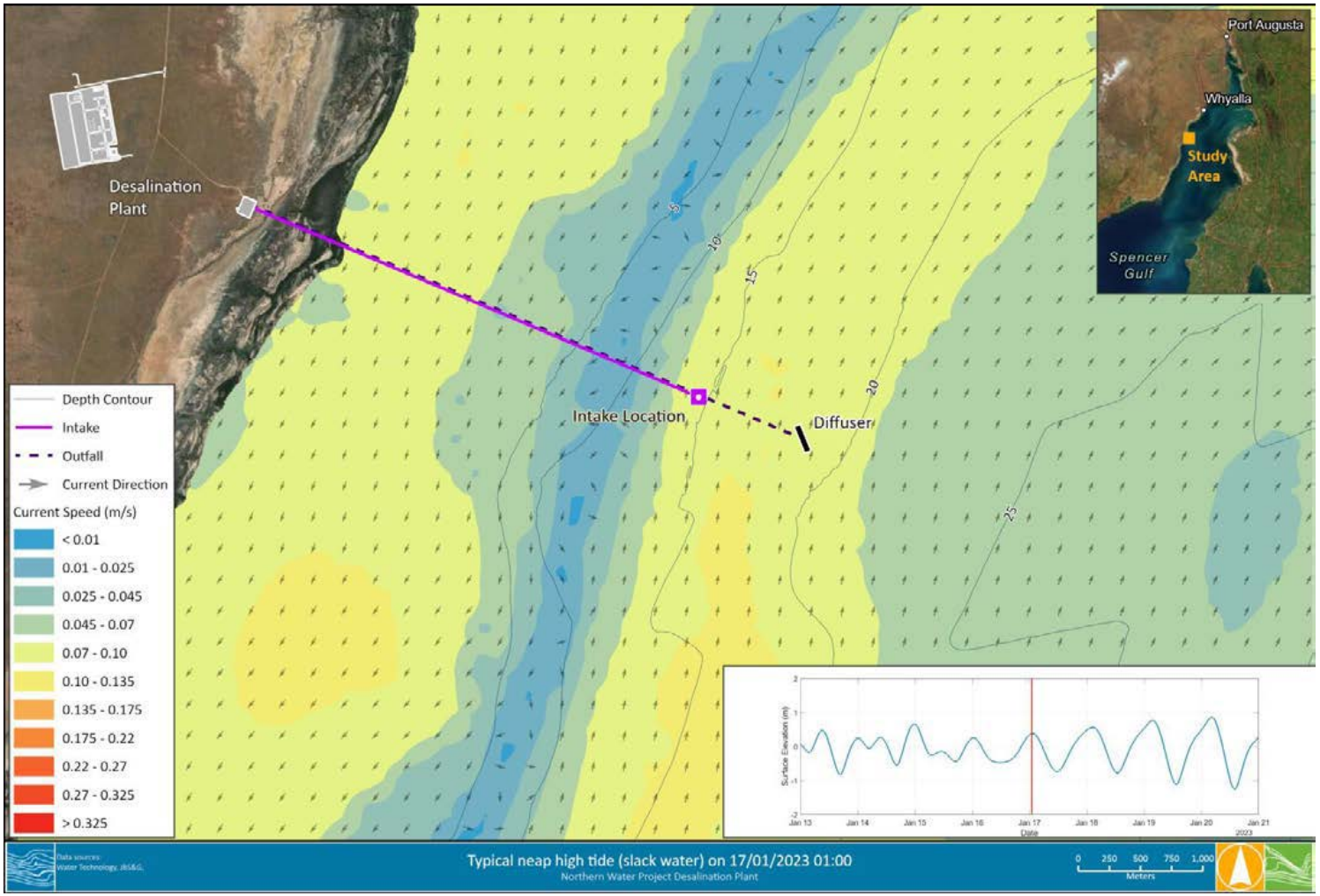


Figure 3-22 Example of Currents on a Neap High Tide

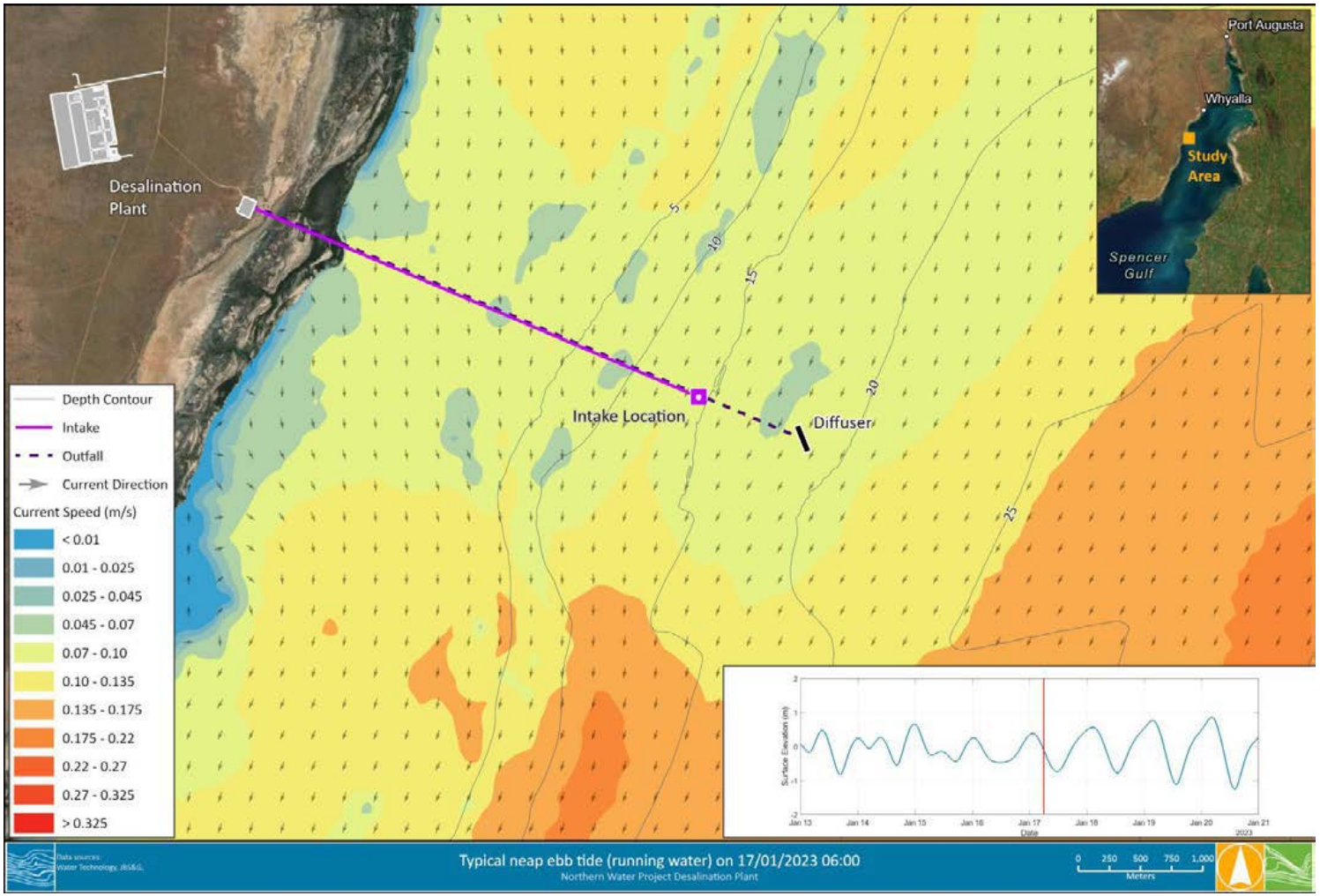


Figure 3-23 Example of Currents on a Neap Ebb Tide

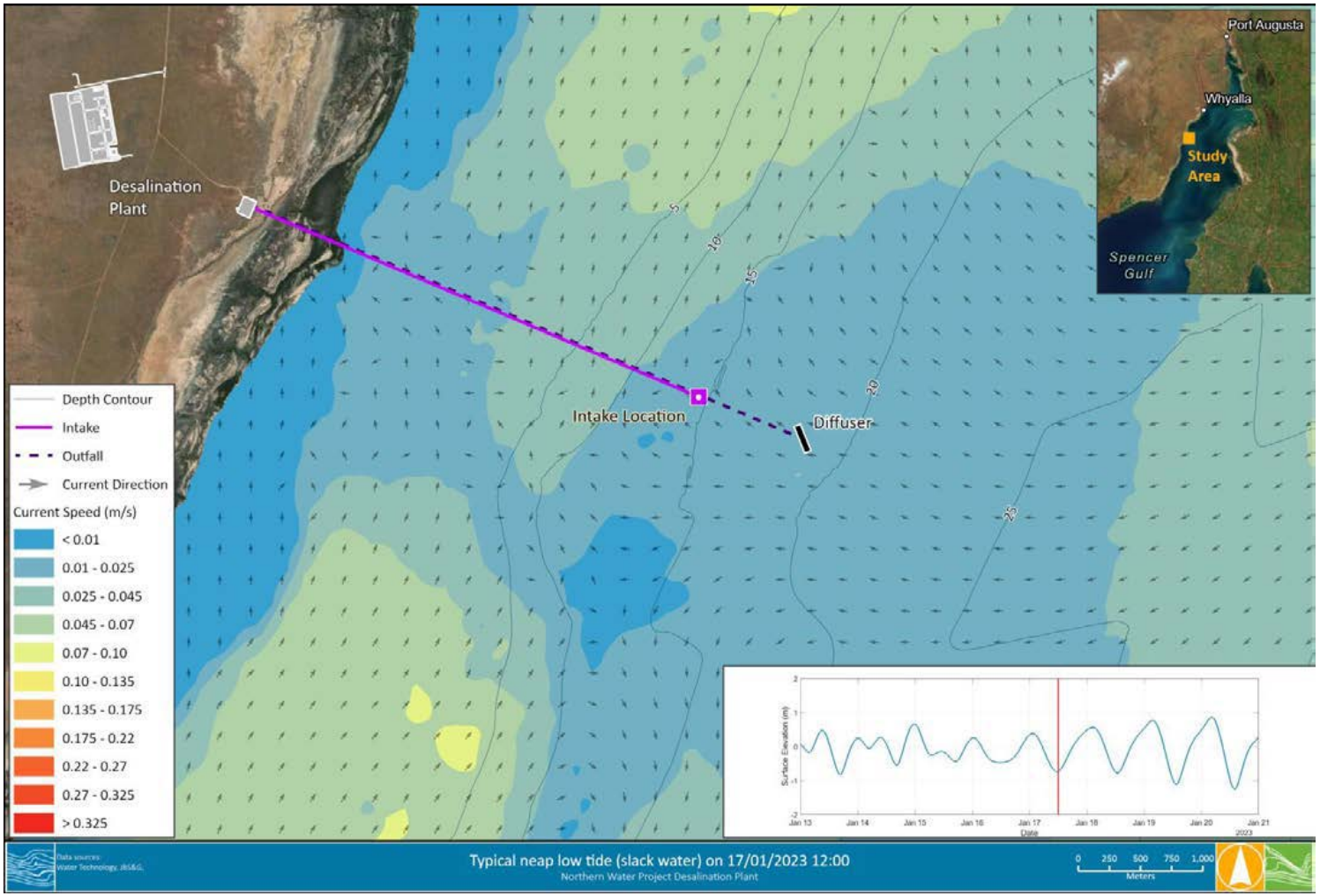


Figure 3-24 Example of Currents on a Neap Low Tide

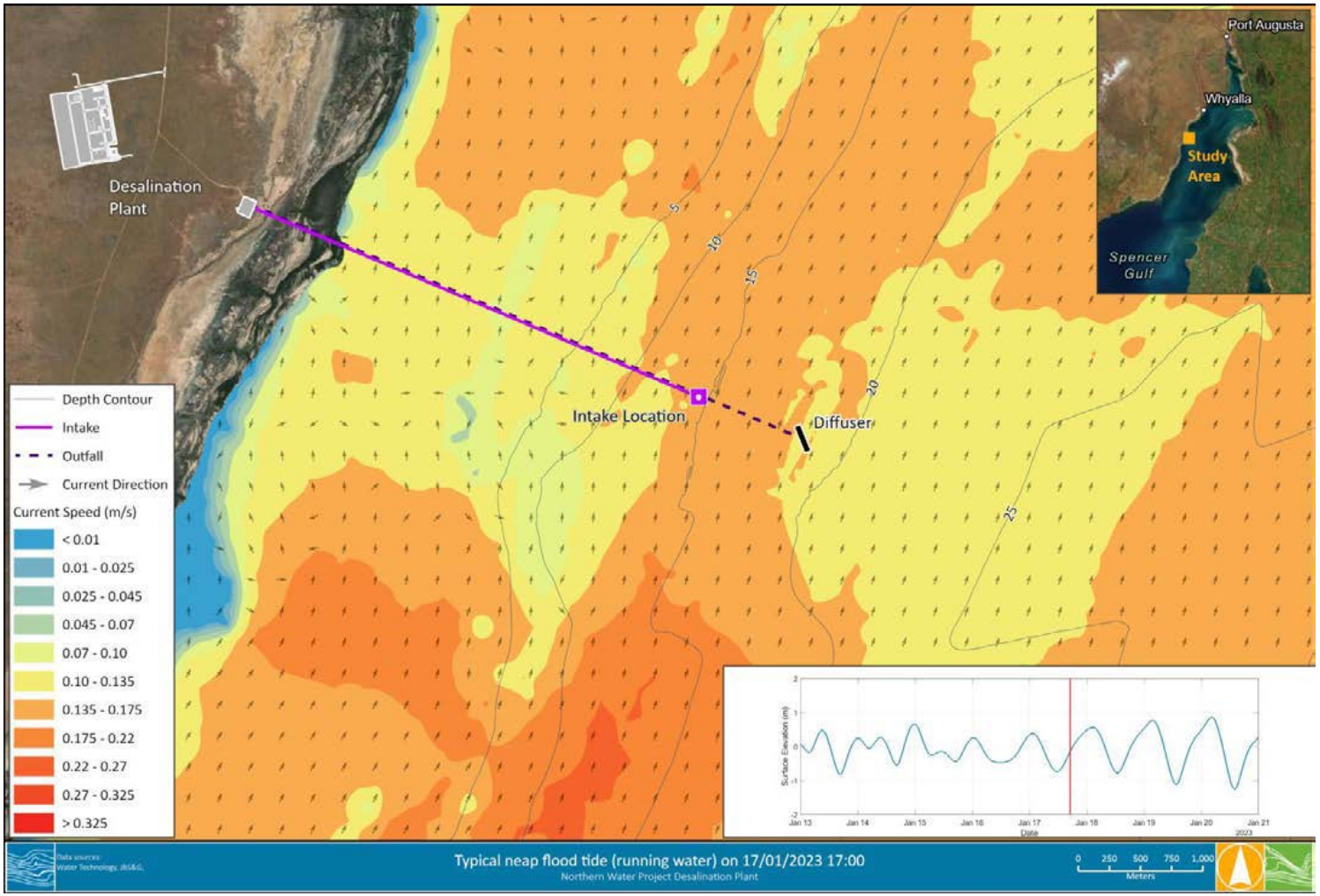


Figure 3-25 Example of Currents on a Neap Flood Tide

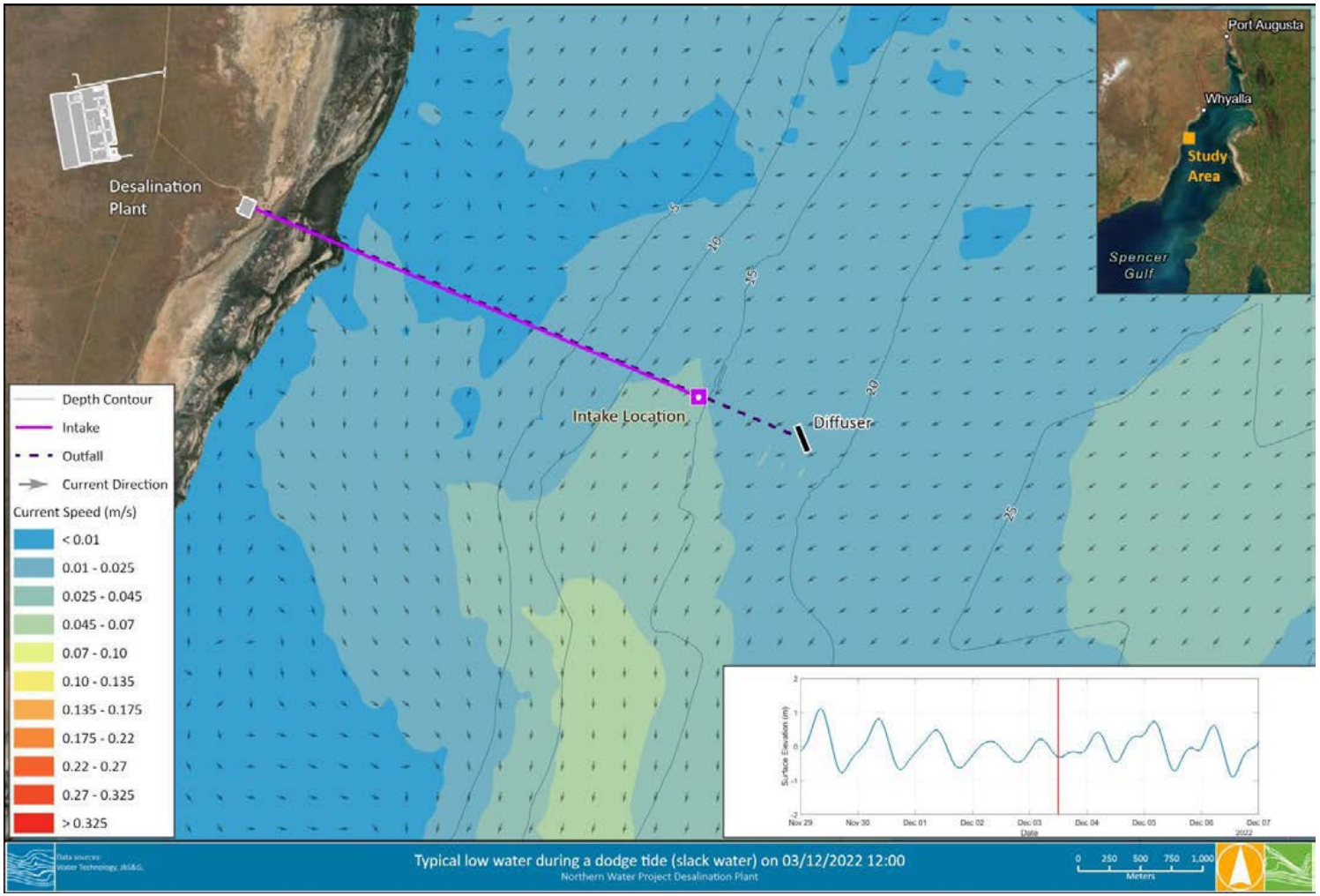


Figure 3-26 Example of Currents on a Low Water During a 'Dodge' Tide

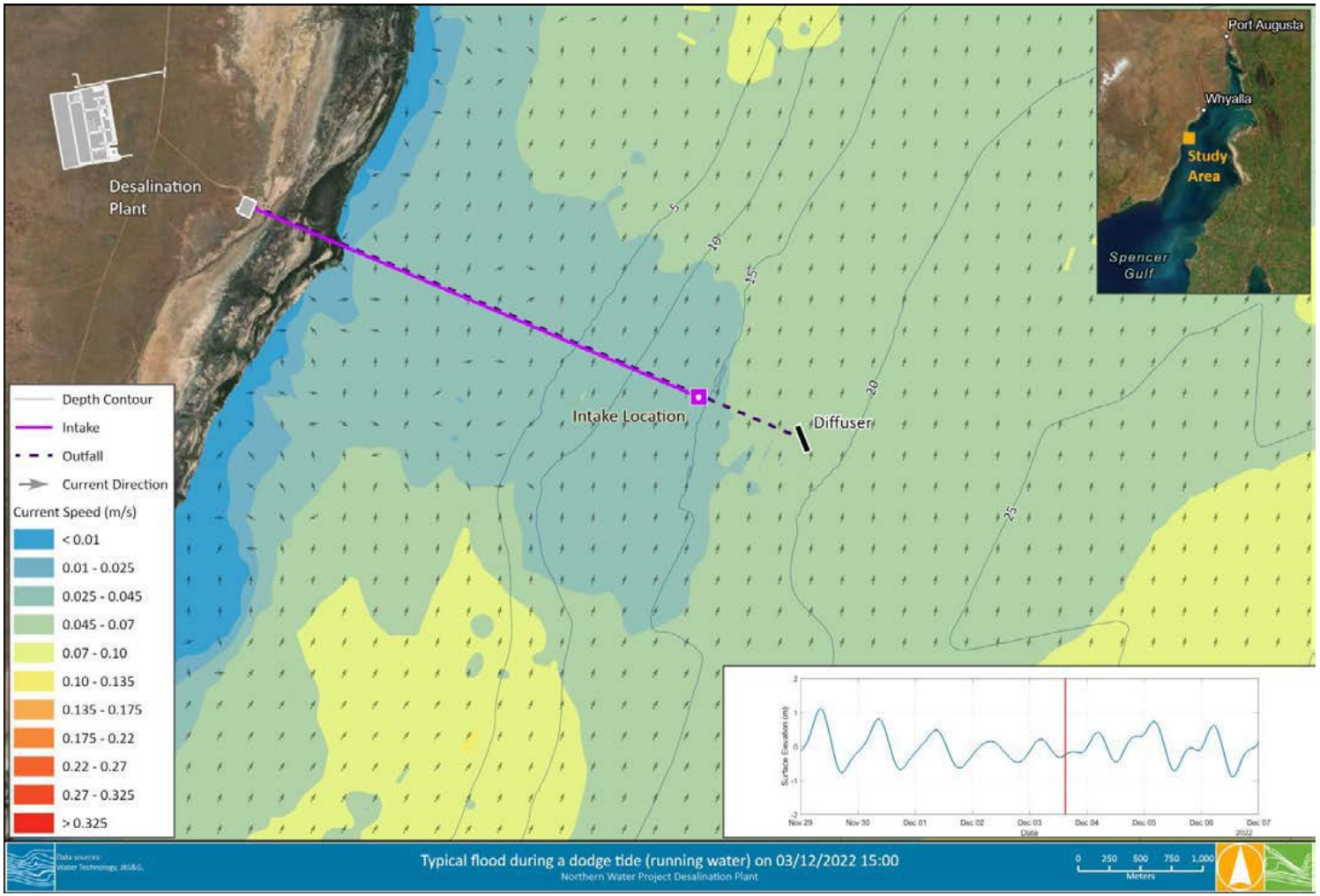


Figure 3-27 Example of Currents on a Flood During a 'Dodge' Tide

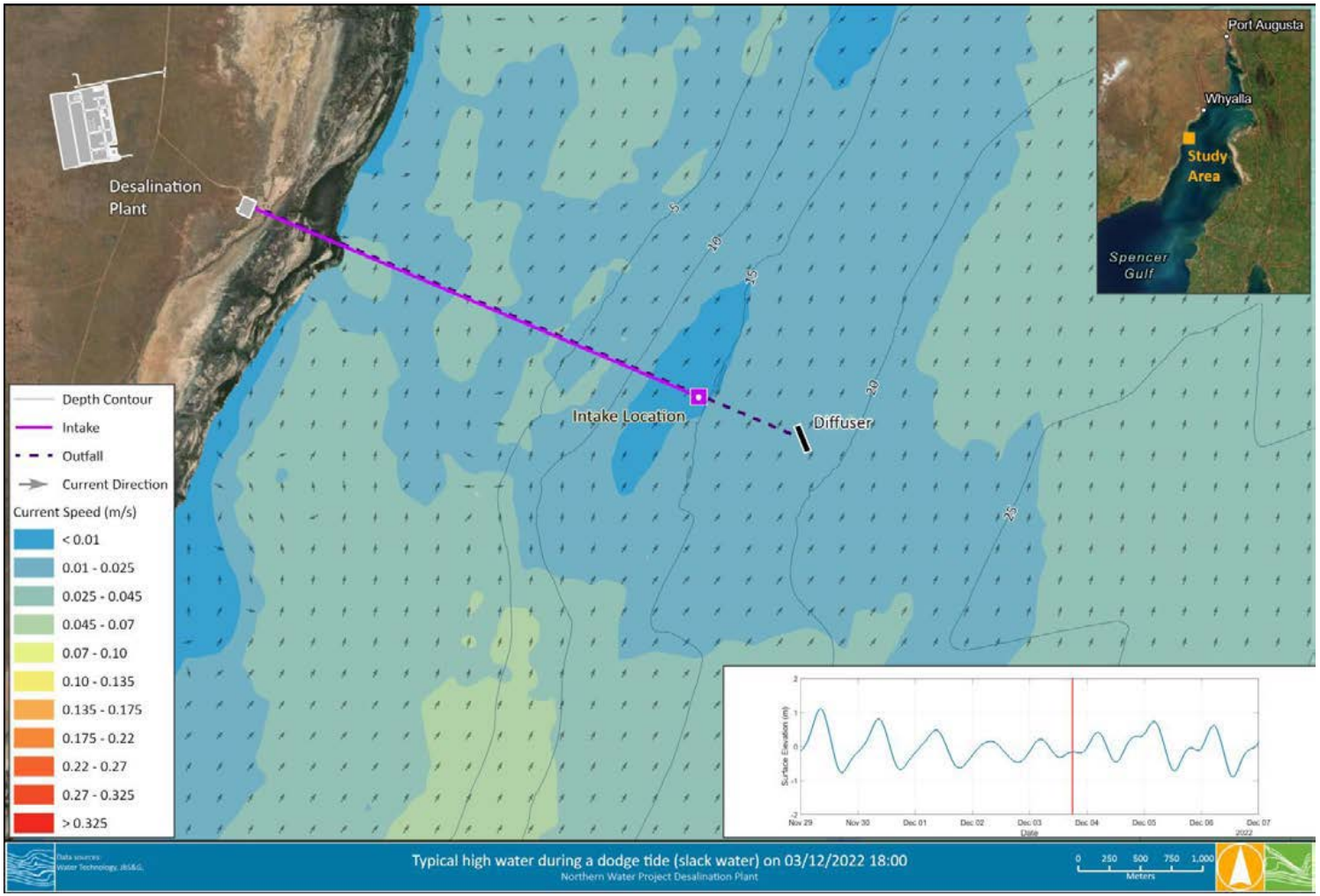


Figure 3-28 Example of Currents on a High Water During a 'Dodge' Tide

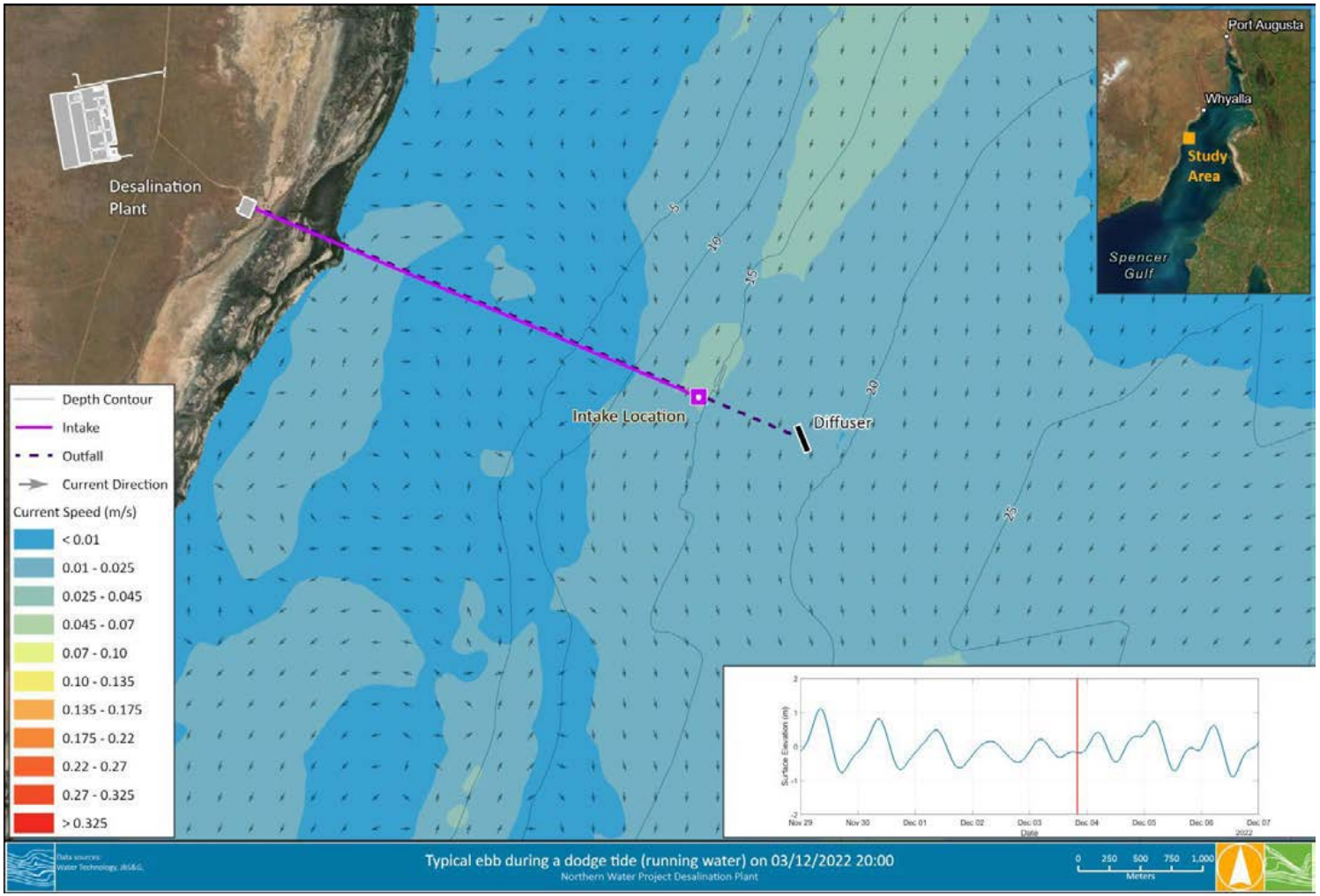


Figure 3-29 Example of Currents on an Ebb During a 'Dodge' Tide

4 DESALINATION MODELLING SCENARIOS

Simulations including the effects of the intake and outfall were executed to inform EIS water quality and ecological assessments. The scenarios comprised the following:

73 GL/year Hypothetical High-Capacity Scenario: The hypothetical high-capacity scenario is 73 GL/ year with associated flow rates for both outfall and intake. Outfall assumes 12 operational ports.

51 GL/year Design Scenario: The annual plant design capacity is 51 GL/year with associated flow rates for both outfall and intake. Outfall assumes 10 operational ports.

Minimum Capacity/Turn Down Scenario: The plant capacity is 40 ML/day with associated flow rates for bypass, outfall and intake. Outfall assumes 6 operational ports.

4.1 Simulation Periods

All scenarios were run for a full year and were analysed for different seasons as follows:

- Summer 2022 to 2023 – simulations were run between 01 November 2022 and 01 March 2023 (results processed for December 2022 to February 2023, November was excluded to allow model 'warm up');
- Autumn 2023 – simulations were run and processed between 01 March and 01 June 2023;
- Winter 2023 – simulations were run and processed between 01 June and 01 September 2023;
- Spring 2023 – simulations were run and processed between 01 September and 01 December 2023;

These periods were selected to provide a year-round picture of the midfield effects of desalination plant operation near the site and to coincide with periods where measurements were available.

4.2 Model Set-up

Model parameters, initial conditions, ocean boundary conditions and meteorological boundary conditions were the same as described previously in this report.

4.3 Discharge and Intake Configurations

The discharge was configured in the MIKE model as a "jet" type discharge for each individual port. This procedure uses a DESA (Distributed Entrainment and Sink Approach) approach to better represent the sub-grid scale near field effects into the mid field model. The procedure is based on the combination of an integrated jet model (Jirka 2004) with the inclusion of jet momentum, mass and heat as proposed by Choi and Lee (2007). The intake was represented as a point sink located between 3.0 m and 5.0 m above the seabed.

Baseline simulations did not include intake or discharge effects. Different desalination scenario configurations for intake and port discharges are presented in Table 4-1.

Table 4-1 Intake and Discharge Configuration for Different Scenarios

Scenario	Intake Flow Rate (ML/D)	Total Outfall Flow Rate (ML/D)	Number of ports	Port Flow Rate (m^3/s)	Port Salinity (PSU) and Temperature (°C)
73 GL/year	486	286	12	0.2756	70.95 and water temperature at intake + 3 °C (all ports)
51 GL/year	389	229	10	0.2646	70.95 and water temperature at intake + 3 °C (all ports)
Minimum Capacity / Turn Down	135	92	6	0.1774	59.6 and water temperature at intake + 3 °C (all ports)

In addition to temperature and salinity, a passive tracer was added in each of the discharge ports assuming a concentration of 1.0 (no units), while all other boundaries were applied a null tracer concentration. Dilution was then calculated as the inverse of the tracer concentration.

4.4 Discharge Scenario Results

4.4.1 Results Presentation

The following results are presented for each simulation:

- **Instantaneous dilution maps** – instantaneous dilution calculated by the near-bottom cells in the midfield model was used to show plume mixing and movement across different stages of the tide. The maps show dilution contours associated with the 95% and 99% SPTV obtained from WET results tests (Van Dam 2025) superimposed on benthic community habitat maps.
- **Dilution percentile contour maps** – a range of percentile (10th, 20th and 50th) dilutions over each simulation period was calculated for each bottom-most model cell. A map superimposing different benthic community habitats was then produced with dilution contours associated with both 95% and 99% SPTV obtained from WET results tests (Van Dam 2025). This type of result was presented to assess the areas with dilution levels lower and higher than the provided SPTVs.
- **Instantaneous near-bed salinity-change maps** – the salinity change in comparison to a reference location (chosen to be sufficiently near to the discharge site to have representative salinities yet sufficiently far from it to not be affected by the discharge) was calculated for each bottom-most cell (see reference location in Figure 4-1). A map superimposing different benthic community habitats was then produced with 1.1 g/L, 1.3 g/L, 1.5 g/L and 1.8 g/L salinity change contours superimposed.
- **Maximum 24-hour average near-bed salinity-change maps** – the salinity change in comparison to a reference location was calculated for each bottom-most cell (see reference location in Figure 4-1). The 24-hour moving average of this change was calculated over the entire simulation period and the maximum of the 24-hour moving average was extracted. A map superimposing different benthic community habitats was then produced with the 1.1 g/L, 1.3 g/L, 1.5 g/L and 1.8 g/L maximum 24-hour average salinity change contours. This type of result was presented to assess conditions in a similar manner to as stipulated in the EPA License for the ADP.
- **Salinity-change curtains** – in addition to the maximum 24-hour average near-bed salinity change maps, 24-hour maximum average salinity change curtains were also produced across a range of segments parallel to the diffuser arrangement (at the manifold alignment, 100 m north and south, 300 m north and south, 500 m north and south and 700 m north and south of the manifold alignment, Figure 4-1). These segments extended from 500 m inshore to 1.0 km offshore of the diffuser (first riser centreline). This type of result was presented to show salinity stratification near the discharge.
- **Sensitive Receptors Timeseries** – timeseries of dilution and near-bed salinity changes were obtained for each scenario. In this case, salinity change was calculated with reference to a baseline simulation without the desalination plant. In addition to dilution and near-bed salinity changes, TSS change was calculated based on dilutions, and assuming ambient and discharge TSS as per Table 2-4.
- **Temperature change maps** – temperature changes in comparison to baseline were calculated for each surface and bottom-most cell (see reference location in Figure 4-1). A map superimposing different benthic community habitats was then produced with temperature change contours superimposed.

While the simulations were performed using salinity expressed as Practical Salinity (PSU), consistent with the source boundary-condition datasets, modelled salinity changes are reported in g/L for consistency with environmental impact assessment requirements. Given the relatively small differences between Practical Salinity and salinity expressed as a concentration (g/L) under typical marine conditions, this reporting approach is considered appropriate and is not expected to materially affect the assessment outcomes.

4.4.2 Dilution

4.4.2.1 Instantaneous dilution

Given the tidal influence in local hydrodynamics and transport at the proposed location, it is informative to illustrate the shape of the discharge under different tidal regimes. Instantaneous dilution maps are shown for the summer season. The contours also approximately indicate salinity and TSS changes assuming ambient and discharge salinities and TSS as per Table 2-4. TSS changes were provided in terms of a range based on measurements shown in Table 2-3.

4.4.2.1.1 'Dodge' Tides

Of all instantaneous plots explored, the largest plume sizes occurred during 'dodge' tides as dictated by weak mixing conditions (Figure 4-2 and Figure 4-3), with limited movement over the transition from flooding to ebbing conditions (see Section 3.5.3). Over the period, the plumes showed some level of accumulation on the southern side of the diffuser. Notwithstanding this accumulation to the south, tidal flow was predominantly (albeit weak) northward during the flooding stage, transporting return water toward the north. By the ebb tide instant (Figure 4-3, lower panels), the plume was more evenly distributed around the diffuser.

The size of the plumes under these weak ambient mixing conditions, as expected, was dictated by the return water flow rate. Plume extent inferred by distance from a riser to the edge of the 95% SPTV contour was relatively short for all scenarios, with the largest distances occurring during ebbing tide of <670 m (73 GL/year scenario), <560 m (51 GL/year scenario) and <300 m (Turn Down scenario). Movement of the 99% SPTV contour, however, showed more significant differences between scenarios. For example, the largest distance from a riser to the edge of the 99% SPTV contour was <1,100 m (73 GL/year scenario), <960 m (51 GL/year scenario) and <440 m (Turn Down scenario), almost double the distances to the edges of the 95% SPTV contour.

4.4.2.1.2 Spring Tides

Contrasting to 'dodge' tide periods, spring tides presented significant improvements in mixing conditions and associated reductions in plume sizes (Figure 4-4 and Figure 4-5). For example, in the low and ebbing tide conditions shown, none of the scenarios displayed the 99% SPTV contour. This was a response to the plume movement with the tendency of the return water to move more slowly during tide transitions (low and high water), showing some level of accumulation near the diffuser. The maximum distances from the risers to the edge of the 95% SPTV contours during the low tide were much shorter (<390 m and <330 m for 73 GL/year and 51 GL/year, respectively) than during 'dodge' tides.

The largest plume sizes (given by the 99% SPTV contours) occurred during flooding tides. In this instance, the dynamic effect of advection in the 73 GL/year and 51 GL/year scenarios were apparent due to the 95% SPTV contour extending away from the diffuser. The maximum distances from the diffuser to the edge of 95% SPTV contour was ~280 m, while the distance from the riser to the 99% SPTV contour edge was larger ~460 m, albeit still shorter in comparison to 'dodge' tides.

From the above, although the return water plumes displayed little movement under weak flow and mixing conditions, high returning water concentrations did not accumulate over large periods of time as the tides transitioned to a more energetic period.

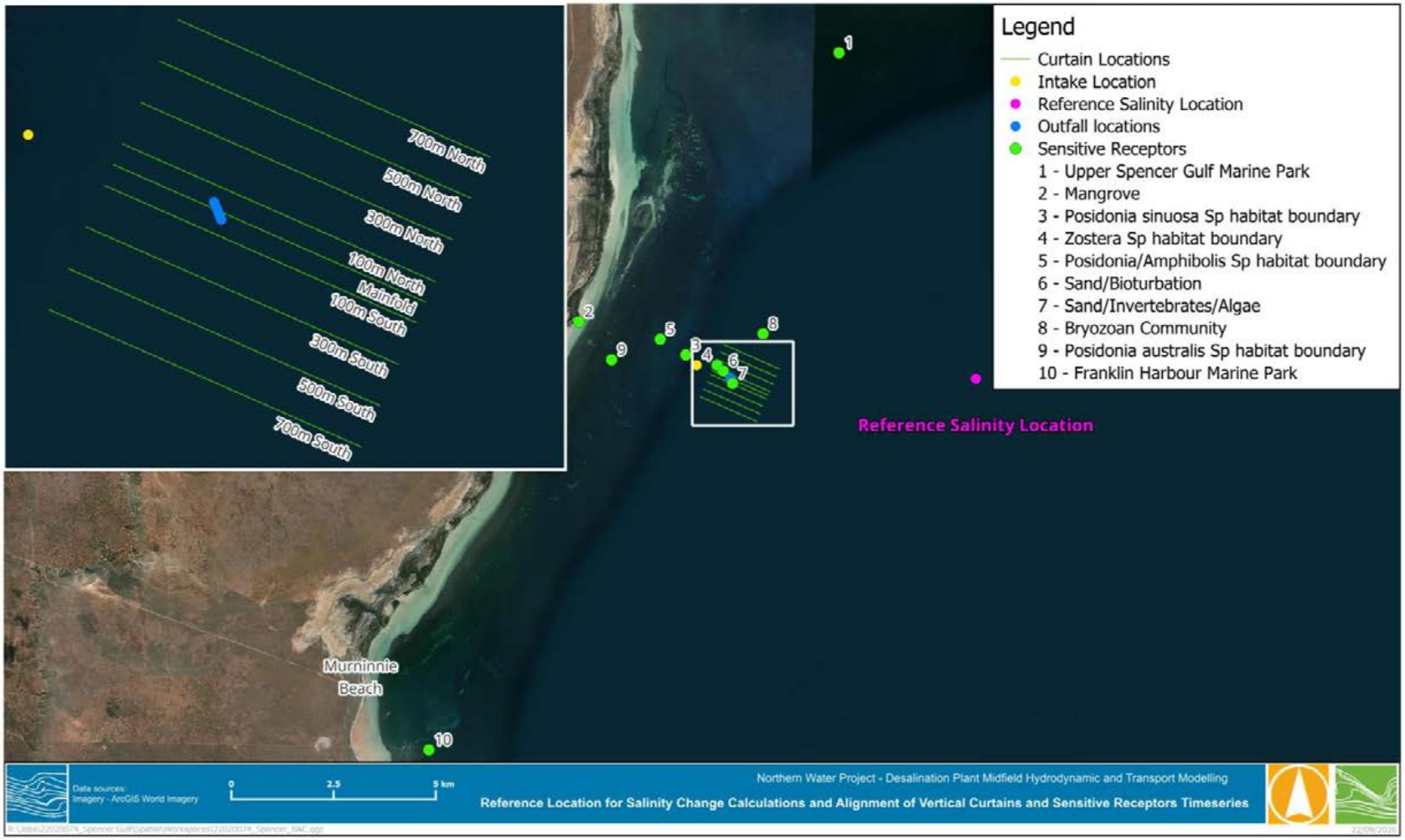


Figure 4-1 Reference Location for Salinity Change Calculations, Alignment of Vertical Curtains and Sensitive Receptor Timeseries

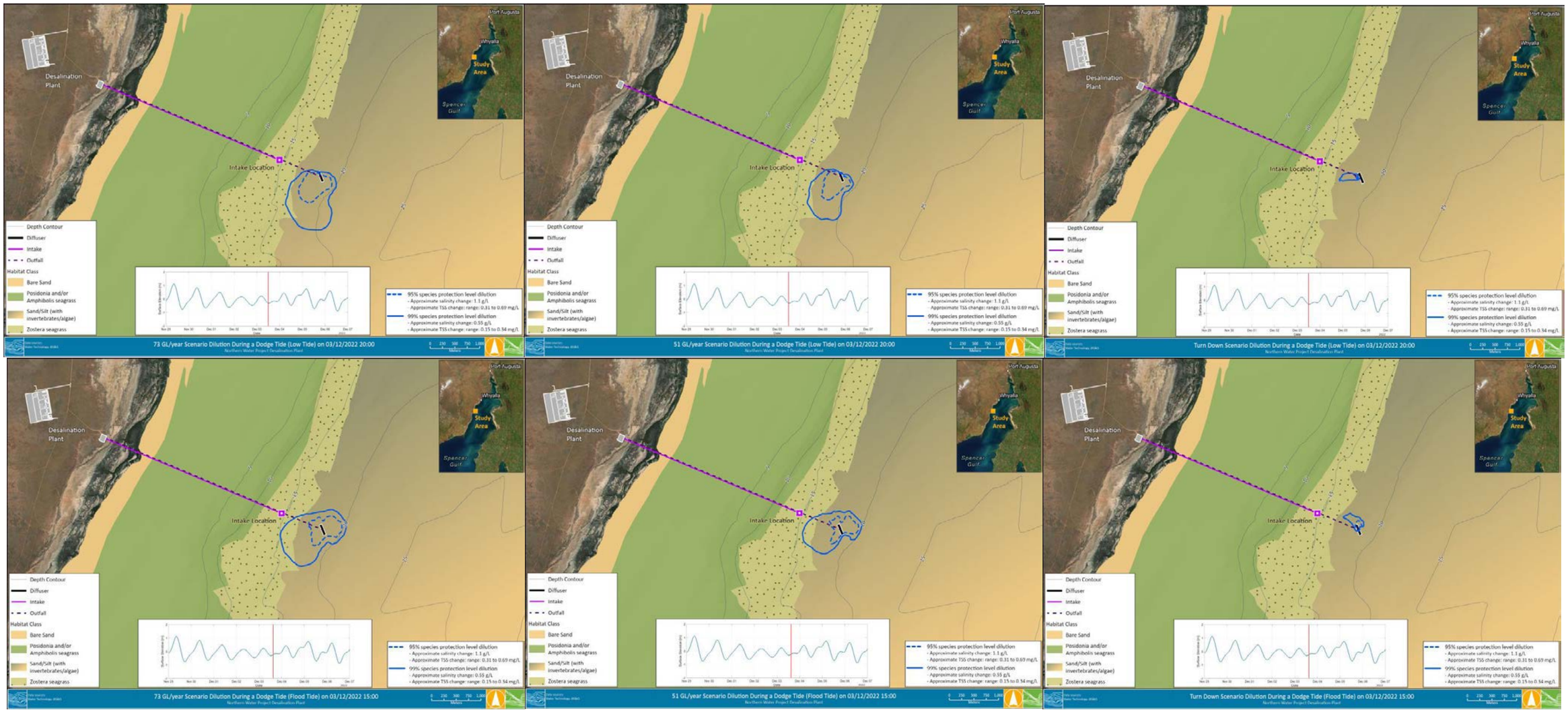


Figure 4-2 Instantaneous Plume Dilution During Low (Upper Panels) and Flooding (Lower Panels) tides of a 'Dodge' Tide. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

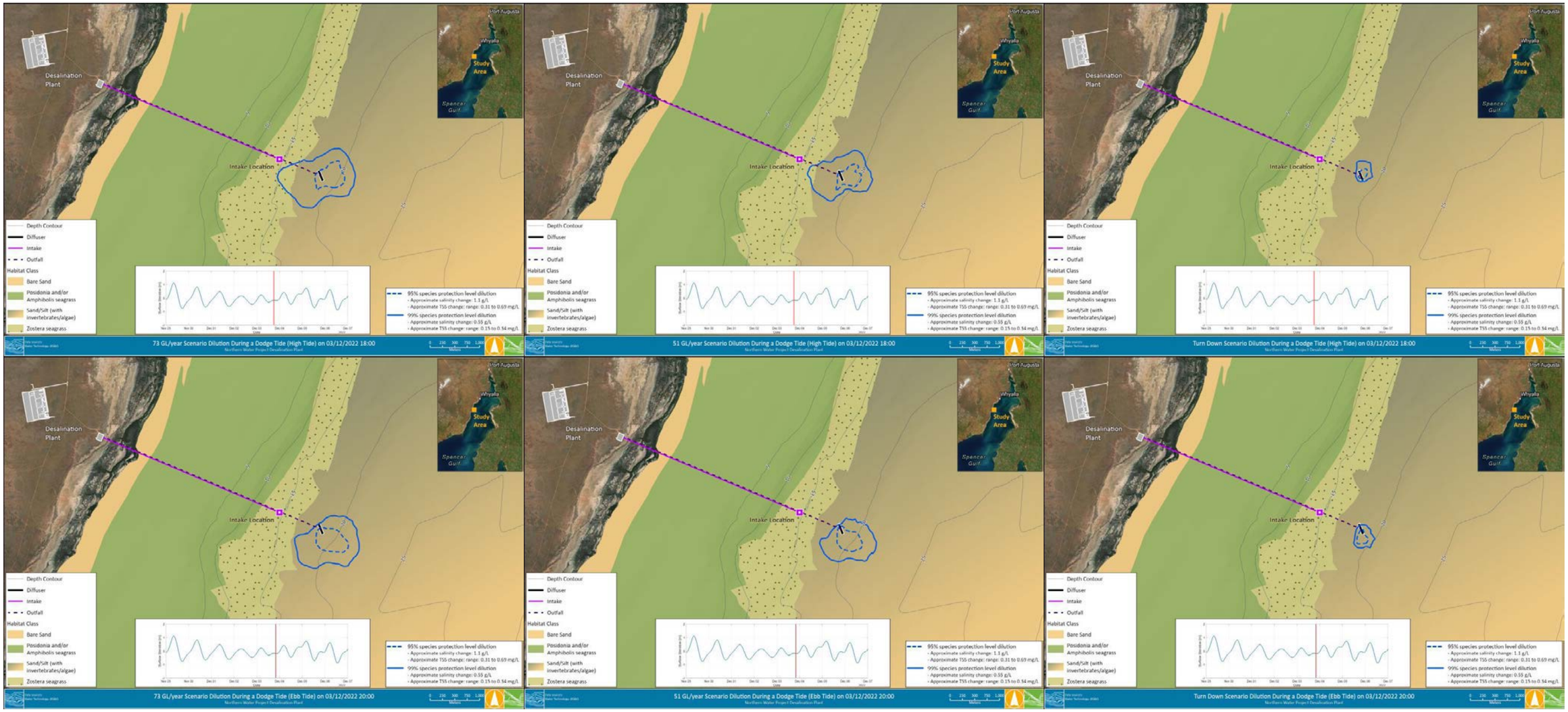


Figure 4-3 Instantaneous Plume Dilution During High (Upper Panels) and Ebbing (Lower Panels) tides of a 'Dodge' Tide. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

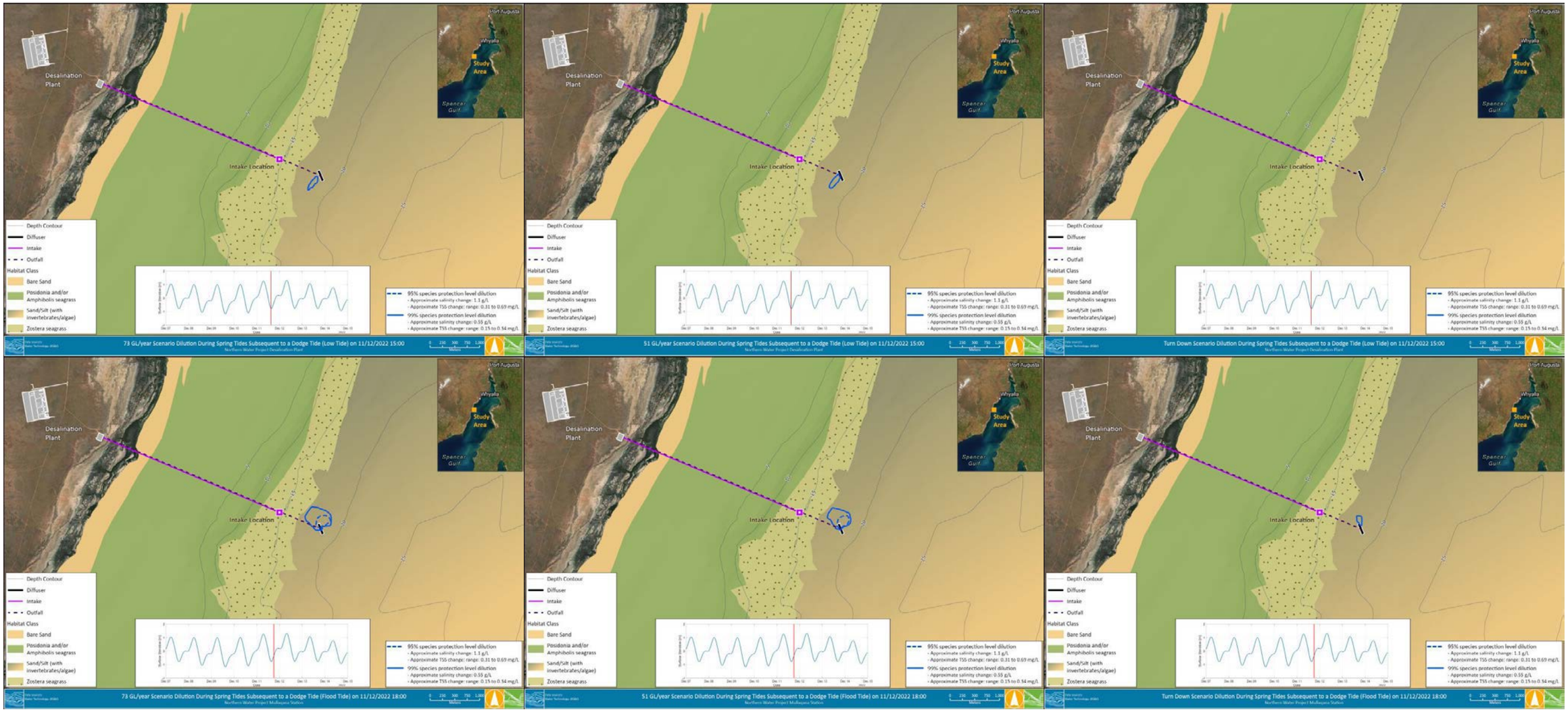


Figure 4-4 Instantaneous Plume Dilution During Low (Upper Panels) and Flooding (Lower Panels) Stages of a Spring Tide. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

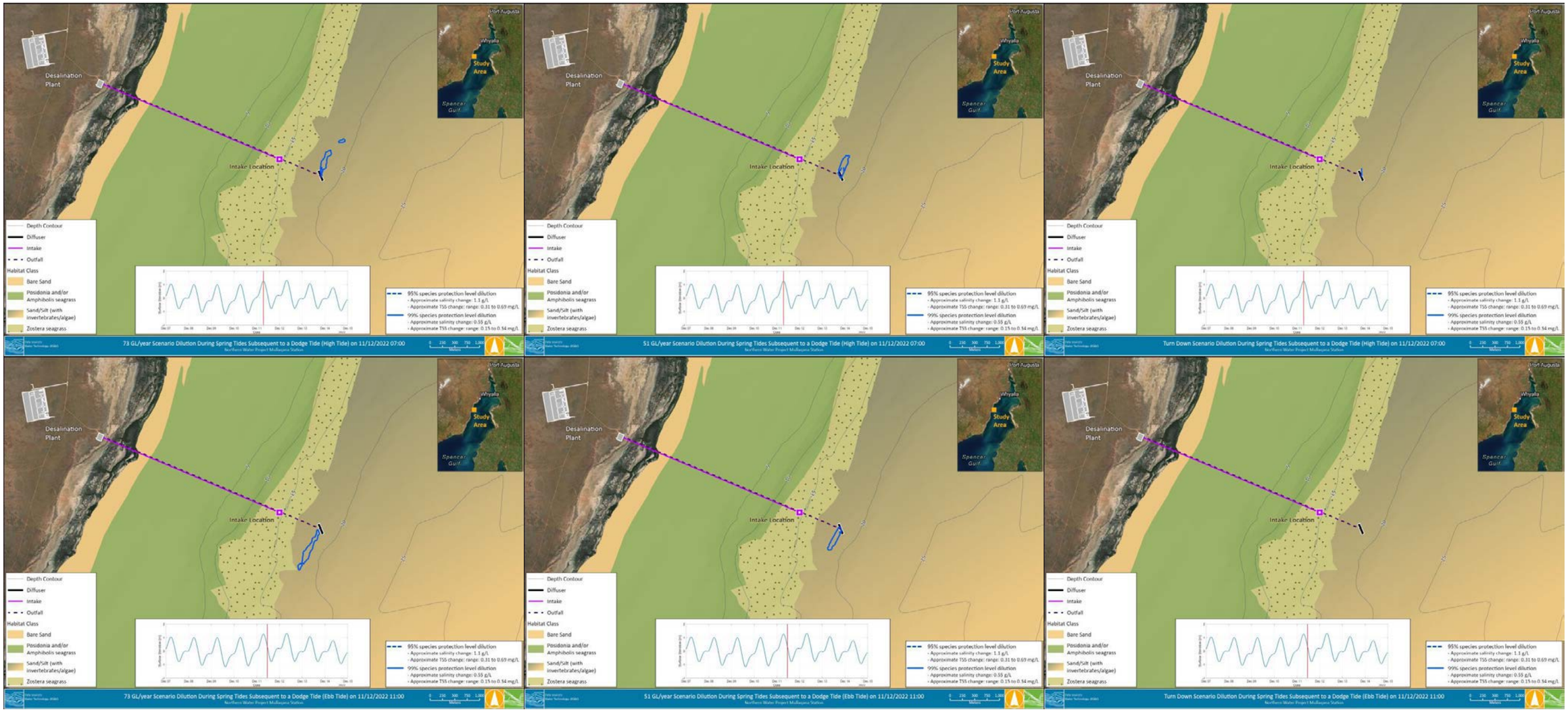


Figure 4-5 Instantaneous Plume Dilution During High (Upper Panels) and Ebbing (Lower Panels) Stages of a Spring Tide. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

4.4.2.2 Dilution Percentiles

Information obtained from the instantaneous conditions (i.e., such as the ones presented in the previous section maps) was adopted to produce temporal statistics in each near bed model cell and to then form the basis of assessment for the quantification of impacted areas. 10th percentile dilution maps displaying 95% and 99% SPTV contours for all scenarios across each season are shown in Figure 4-6 to Figure 4-9. Here, it is noted that the interpretation of a 10th percentile dilution is equivalent to a model location experiencing dilution levels lower than the given SPTV for at most 10% of the time over the simulation period (for a 90-day simulation period, this is equivalent to 9 days). Note that the period is not continuous, as illustrated by the movement of the plumes in Section 4.4.2.1. Inside the contours, SPTVs are not achieved at least 10% of the time over the analysis period. In contrast, outside the contour areas, the SPTVs are achieved more than 90% of the analysed period.

Other percentile dilutions maps (20th and 50th) are presented in Appendix E.

As expected, the 73 GL/year scenario displayed the largest footprint among all scenarios. In general terms, the footprints:

- Extended further offshore than inshore, with gravitational movement downslope;
- Had a larger longshore extent compared to cross-shore extent due to preferential tidal movements; and
- Varied in preferential alongshore transport extent depending on season.

For example, the maximum distance between diffuser and the 95% SPTV contour line for 73 GL/year and 51 GL/year scenarios in winter 2023 were ~360 m and ~300m, respectively. The dilution contours were further analysed to determine the maximum and average distances to the outfall risers across all seasons (Table 4-2). Maximum distances varied between ~360 m and ~430 m (73 GL/year), ~300 m and ~370 m (51 GL/year), and ~55 m and ~90 m (Turn Down). Average distances varied between ~250 m and ~280 m (73 GL/year), ~190 m and ~230 m (51 GL/year), and ~30.0 m and ~65 m (Turn Down)

Given the non-uniform shape of the footprints, comparisons between different scenarios were better expressed in terms of areas within the environmental criteria contours (Table 4-3). The 99% SPTV, again as expected, was wider that the 95% SPTV contour (i.e., the SPTV is based on higher dilutions). For example, in the same winter 2023 scenario, the plume size of the 99% SPTV contour were approximately 5.5 and 6.6 times larger for 73 GL/year and 51 GL/year than the 95% SPTV. For the turn down, although areas were smaller, the 99% SPTV areas were between 8.3 and 29 times larger than the 95% SPTV area. These large ratios reflect the small size of the 95% SPTV area for the turn down scenario.

There were notable differences in the SPTV areas between the 73 GL/year and 51 GL/year scenarios for a same season, particularly for the 99% SPTV, which extended into lower resolution areas of the domain. These differences partly resulted from the numerical artefacts related to changes in the mesh grid away of the outfall location but also indicated the wider size of the 99% SPTV areas. The 95% SPTV areas associated with the 73 GL/year scenarios were 61% to 70% (depending on season) larger than the 51 GL/year scenario. Contrastingly, the 99% SPTV areas associated with the 73 GL/year scenarios were 37% to 49% (depending on season) larger than the 51 GL/year scenario. Turn down areas were significantly smaller, varying between 1.2% to 6.9% (95% SPTV areas, depending on season) and 6.3% to 10% (99% SPTV areas, depending on season) of the 73 GL/year scenario areas.

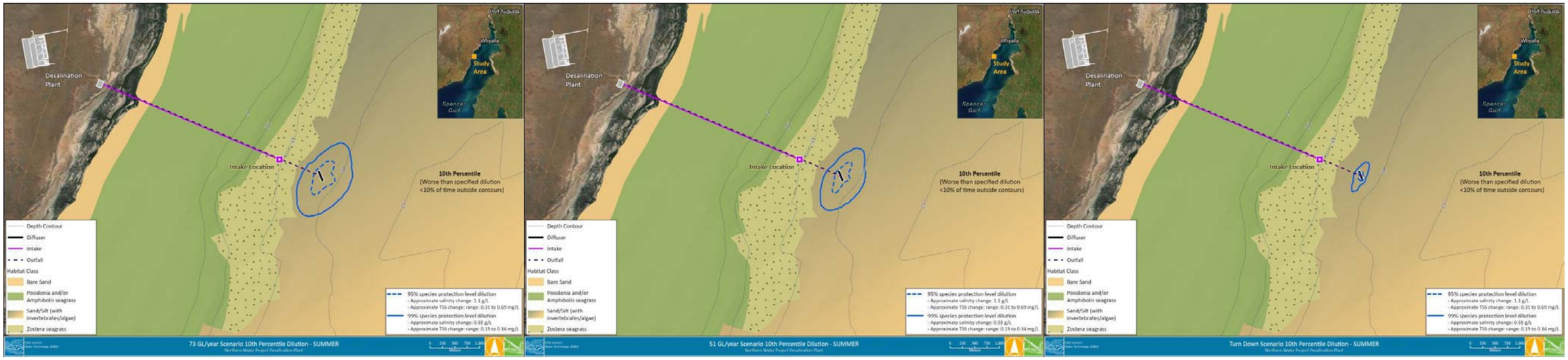


Figure 4-6 10th Percentile Dilution – Summer 2022-2023. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

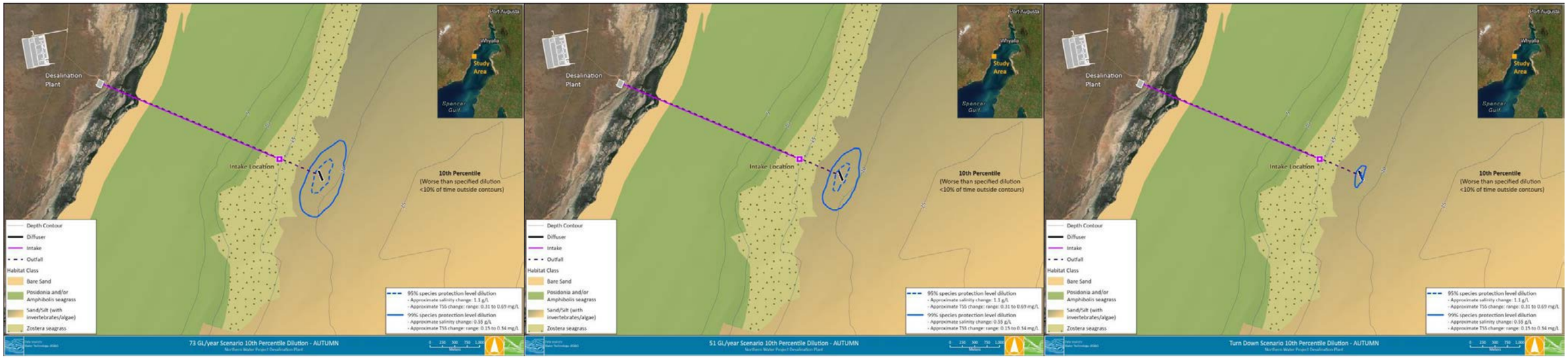


Figure 4-7 10th Percentile Dilution – Autumn 2023. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

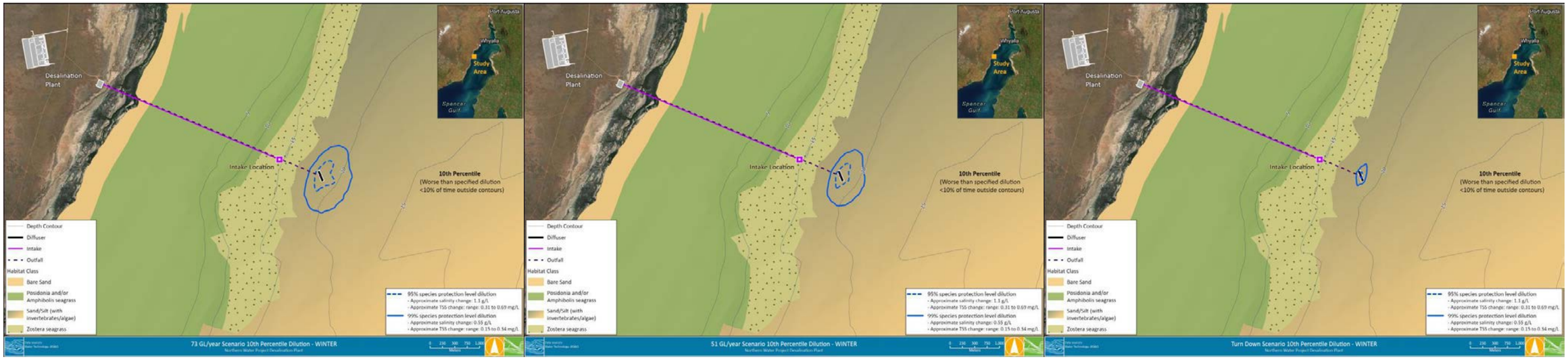


Figure 4-8 10th Percentile Dilution – Winter 2023. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

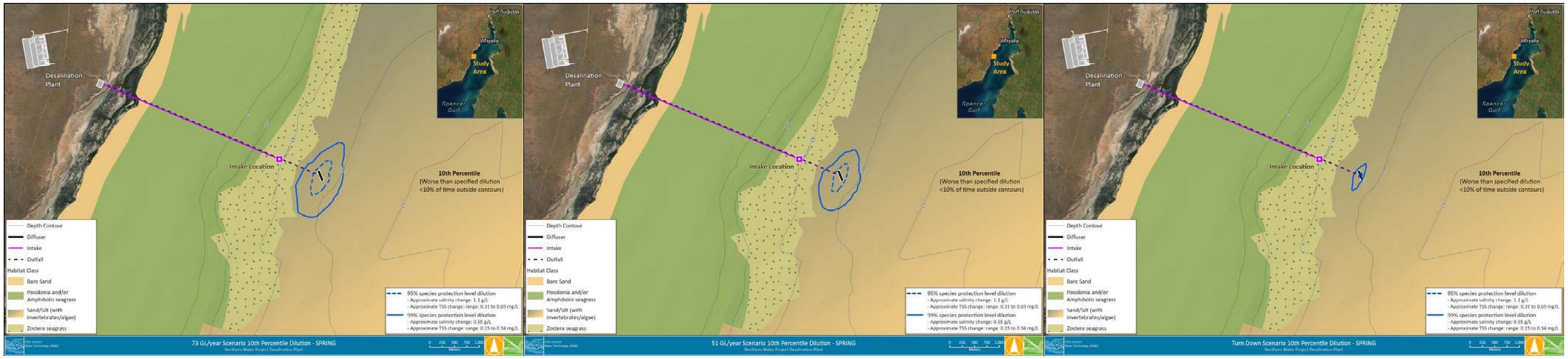


Figure 4-9 10th Percentile Dilution – Spring 2023. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

Table 4-2 Distances from the 95% SPTV 10th percentile Dilution Contours to the Outfall

Scenario	Period	Maximum Distance (m)	Average Distance (m)
73 GL/year scenario	Summer 2022-2023	395.1	284.5
	Autumn 2023	394.5	253.6
	Winter 2023	360.8	253.6
	Spring 2023	427.4	265.5
51 GL/year scenario	Summer 2022-2023	369.4	226.1
	Autumn 2023	289.7	161.0
	Winter 2023	299.4	189.9
	Spring 2023	350.2	214.2
Minimum Capacity (Turn Down) scenario	Summer 2022-2023	91.7	66.8
	Autumn 2023	55.6	32.0
	Winter 2023	68.2	39.9
	Spring 2023	57.2	36.6

Table 4-3 Areas within SPTV Contours

Scenario	Season	10th Percentile Dilution Area (Ha)	
		95% SPTV	99% SPTV
73 GL/year scenario	Summer 2022-2023	20.574	114.067
	Autumn 2023	16.155	89.638
	Winter 2023	16.233	92.058
	Spring 2023	17.919	97.747
51 GL/year scenario	Summer 2022-2023	12.805	78.930
	Autumn 2023	5.876	51.163
	Winter 2023	9.560	61.752
	Spring 2023	10.775	71.333
Minimum Capacity (Turn Down) scenario	Summer 2022-2023	1.413	11.735
	Autumn 2023	0.130	4.088
	Winter 2023	0.386	6.486
	Spring 2023	0.457	7.847

4.4.2.3 Sensitive Receptors

Dilution time series and cumulative distributions at sensitive receptor locations shown in Figure 4-1 are presented in Figure 4-10 to Figure 4-13. Due to the ranges, a logarithmic scale was adopted for the plots, and the scale was limited to a 1:10,000 dilution. Also, the distributions were limited between the minimum and 50th percentiles.

Across all locations, dilutions for all scenarios varied a few orders of magnitude in response to oscillatory tidal movements, as previously inferred by the instantaneous maps. Due to closer proximity of the diffuser, the Sand/Invertebrates/Algae location experienced the lowest dilutions of all sensitive receptor locations. At this location, the 10th percentile dilutions for 73 GL/year scenario and 51 GL/year scenario were similar, at 1:18 for all four seasons. 50th percentile dilutions were considerably higher, varying between 1:102 and 1:110 for 73 GL/year scenario, and between 1:110 and 1:115 for 51 GL/year scenario. For Turn Down scenario, 10th percentile dilutions at the same location were between 1:50 (Summer and Winter) and 1:80 (Autumn and Spring).

The sand/bioturbation location is also located near the diffuser. The 10th percentile dilutions for the 73 GL/year and 51 GL/year scenarios were approximately 1:140. The 50th percentile dilutions ranged from 1:205 to 1:215 for the 73 GL/year and the 51 GL/year scenarios, respectively. For the Turn Down scenario, the 10th percentile dilutions ranged between 1:105 and 1:110.

At locations further from the diffuser, dilutions increased substantially with increasing distance between the receptor locations and the diffuser. For locations far from the diffuser, the 10th percentile dilution of 73 GL/year and 51 GL/year scenarios was above 1:200. The 10th percentile dilution of the Turn Down scenario was above 1:950.

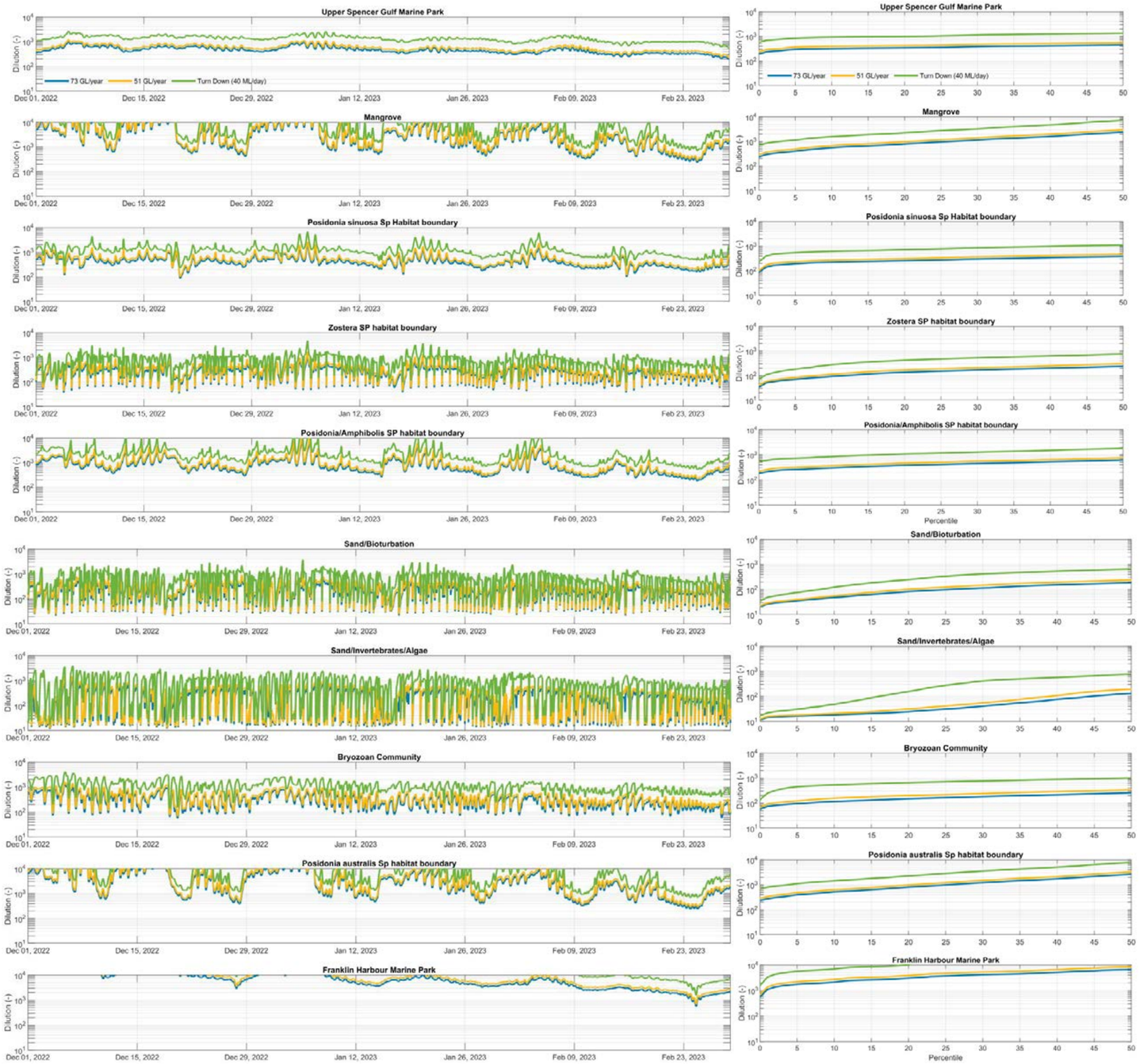


Figure 4-10 Dilution Timeseries (Left Panel) and Cumulative Distribution (Right Panel) at Sensitive Receptors (Summer 2022 to 2023)

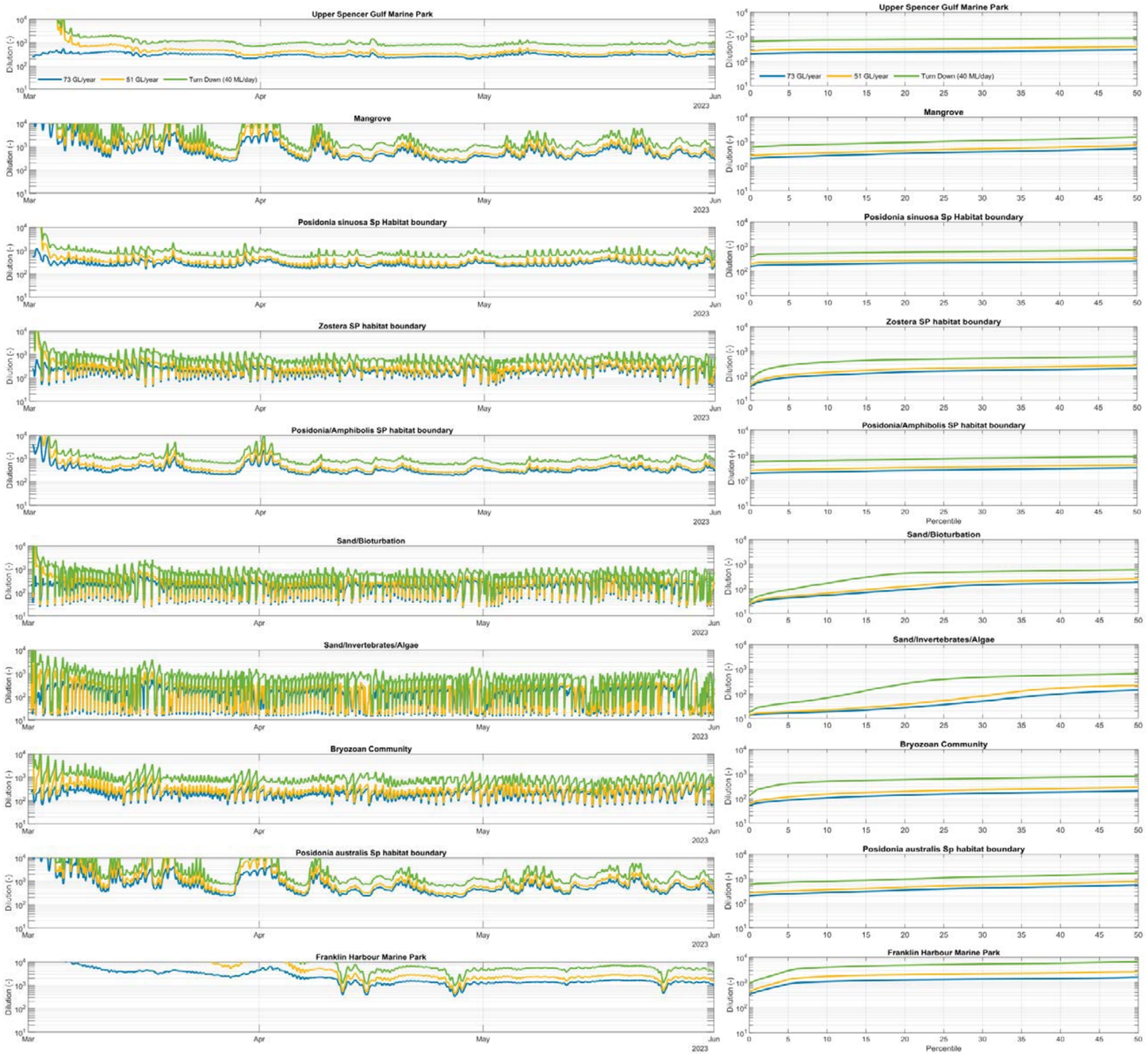


Figure 4-11 Dilution Timeseries (Left Panel) and Cumulative Distribution (Right Panel) at Sensitive Receptors (Autumn 2023)

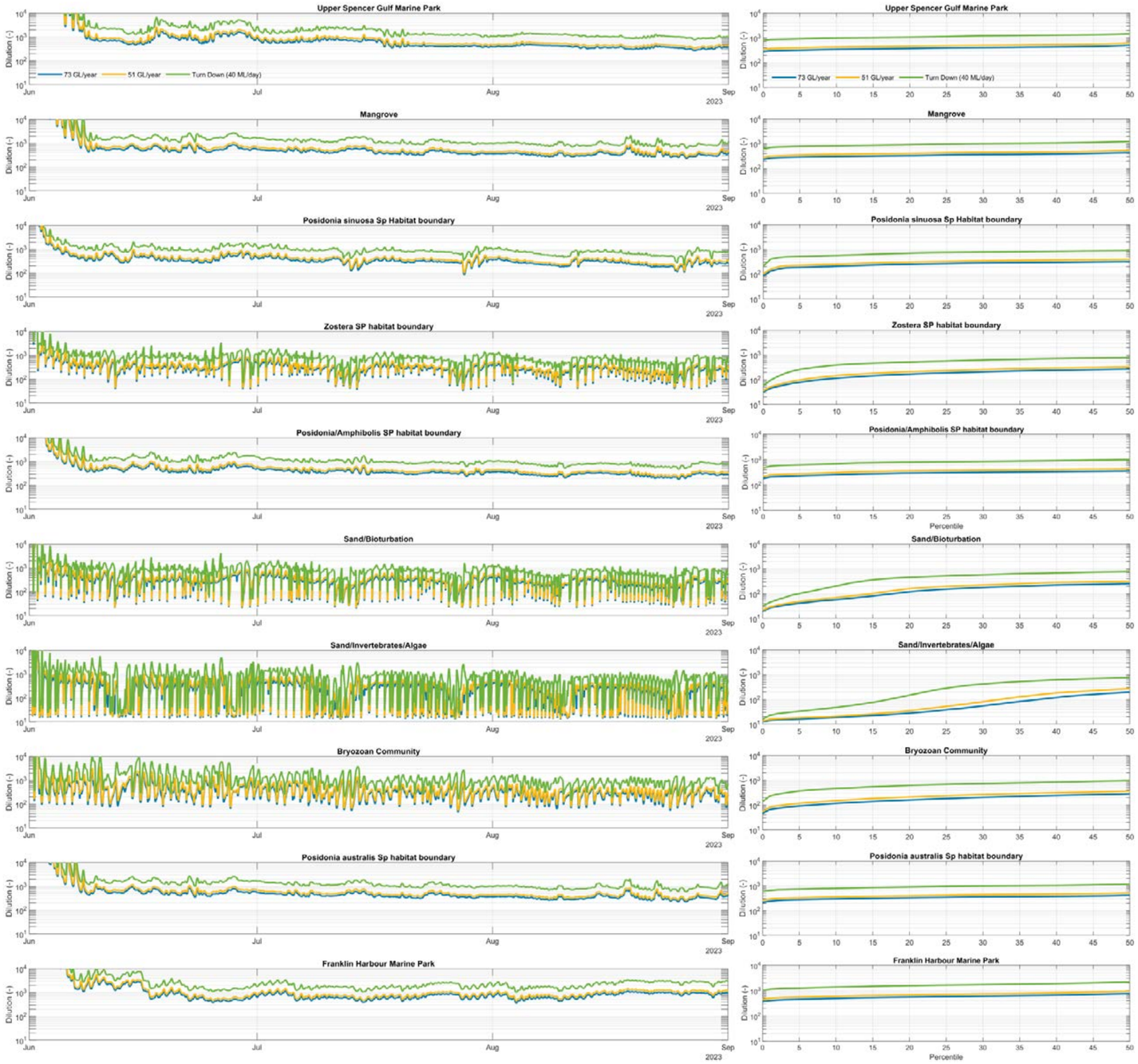


Figure 4-12 Dilution Timeseries (Left Panel) and Cumulative Distribution (Right Panel) at Sensitive Receptors (Winter 2023)

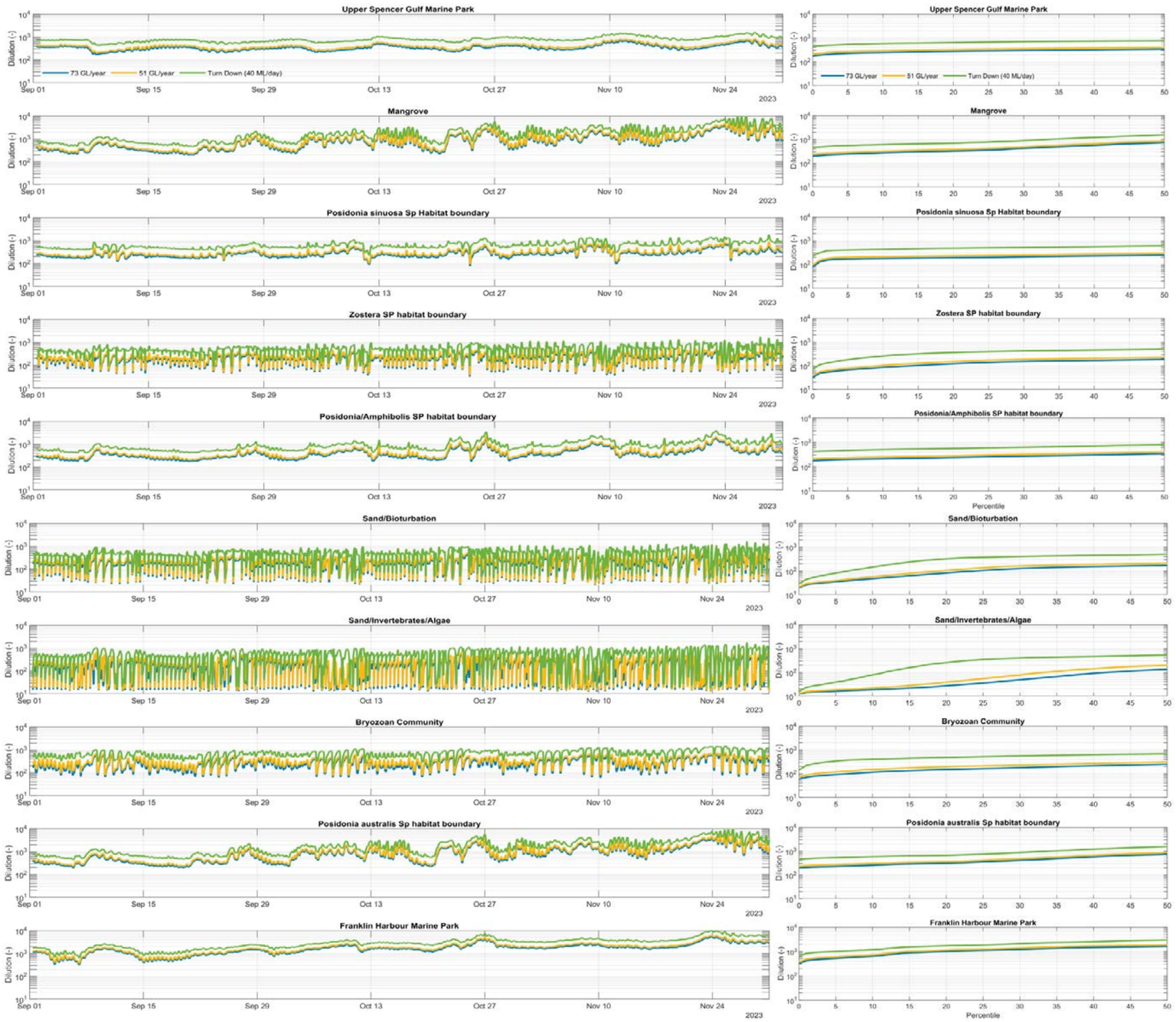


Figure 4-13 Dilution Timeseries (Left Panel) and Cumulative Distribution (Right Panel) at Sensitive Receptors (Spring 2023)



4.4.3 Salinity Change

4.4.3.1 Instantaneous Maps

Similarly to dilution, instantaneous salinity change maps are shown for 'dodge' (i.e., lowest amplitude neaps) and subsequent spring (i.e., highest amplitude) tides for 73 GL/year, 51 GL/year and Turn Down scenarios. Salinity-change contours show the instantaneous difference in relation to the reference salinity location depicted in Figure 4-1. The reference salinity site representing ambient conditions was chosen in such a way that it was located sufficiently close to the desalination plant outfall and yet sufficiently far so that effects of the return water were negligible at the location. Adoption of this reference site allows verification of outfall performance under similar conditions to be undertaken for an operational outfall.

4.4.3.1.1 'Dodge' Tides

Similarly to dilutions, the largest instantaneous condition plume sizes analysed (within provided contours) occurred during the 'dodge' tides due to weak mixing conditions (Figure 4-14 and Figure 4-15). In contrast to dilution maps, the salinity-change maps covered a smaller area. For both 73 GL/year and 51 GL/year scenarios, distances from diffusers to the edge of the plume (here given by 1.3 g/L salinity change contours) were largest during flooding tides (~220 m and ~190 m, respectively) and lowest during low water (~130 m and ~100m, respectively).

As per instantaneous dilutions, locations of the contours did not correlate with the expected tidal movement. At the high tide, the extent of the plumes to the edge of the contours were ~130 m and ~160 m for 73 GL/year and 51 GL/year scenarios, respectively. At the ebbing tide, the extent of the plumes to the edge of the contours were ~195 m and ~180 m for 73 GL/year and 51 GL/year scenarios, respectively.

The 1.3 g/L salinity change contours for the Turn-Down scenario, when present, were within 50m of the outfall.

4.4.3.1.2 Spring Tides

As indicated in the dilution maps, spring tides presented significant improvement in mixing conditions and associated reductions in plume sizes within the provided salinity change contours (Figure 4-16 and Figure 4-17). Except for the flooding tide period, all scenarios did not display any of the provided contours, indicating effective mixing occurred even at small distances from the outfall. The 1.3 g/L salinity-change contours, when present, were approximately parallel to the shoreline and extended northeast of the diffuser ports to distances ~170 m and ~90 m for 73 GL/year and 51 GL/year, respectively.

4.4.3.2 Maximum 24-Hour Averages

4.4.3.2.1 Near Bed Salinity Changes

Juxtaposing the instantaneous maps, results from the simulations were analysed for comparison against the 24-hour average salinity increase criteria. To that end, in every model cell the difference of its salinity to the reference site in Figure 4-1 was obtained and a moving 24-hour average of that change was calculated. The results across the model domain were then mapped.

The areas within the specified salinity change contours (Table 4-4) were considerably smaller than the areas defined by the SPTV contours (Table 4-3), which is a direct consequence of the different mapping methods applied for dilutions and salinity changes. Note also that, for the purposes of presentation of these results, the same 1.3 g/L salinity-change contours used as criterion for the ADP were adopted. Given the higher salinity range at Mullaquana Station in comparison to Port Stanvac, a larger salinity change (i.e., 1.5 g/L or 1.8 g/L) could be accommodated for equivalency to the ADP criterion. These areas can therefore be considered conservative in terms of salinity changes impacts associated with the Project return water.

The areas defined by the 1.3 g/L salinity change contours varied with season (and return water flow rates) and were between 1.8 Ha (Winter) and 3.6 Ha (Autumn) for the 73 GL/year scenario. For the 51 GL/year scenario these areas were smaller (between 21% and 54% of the areas obtained with the 73 GL/year scenarios) and varied between 1.5 Ha (winter and Spring) and 3.0 Ha (autumn). Turn Down scenarios presented significantly smaller areas (maximum of 0.044 Ha in Autumn). Areas inside the other salinity-change contours are also presented in Table 4-4.

As the criterion applies to distances rather the areas, the 24-hour average salinity change contours in Figure 4-18 to Figure 4-21 were further analysed to determine their maximum and average distances to the outfall risers for each season (Table 4-5). Maximum distances from the diffuser ports to contours associated with 1.3 g/L salinity change for the 73 GL/year scenario were between ~95 m (Winter) and ~165 m (Summer), and for the 51 GL/year scenario varied between ~80 m (Spring) and ~140 m (Autumn).

The average distances from the diffuser ports to contours associated with 1.3 g/L salinity change for the 73 GL/year scenario were between ~70 m (Spring) and ~110 m (Summer and Autumn), and for the 51 GL/year scenario varied between ~65 m (Winter and Spring) and ~95 m (Autumn).

Turn Down scenarios presented shorter distances (maximum and average distances were <40 m and <35m, respectively). Further, in some instances (e.g., Winter and Spring), the Turn Down scenario did not display salinity changes above 1.3 g/L.

4.4.3.2.2 Vertical Curtains

While the maps presented above illustrated return water dispersion near the bed, salinity change along vertical curtains parallel to the diffuser alignment were also produced to display the level of salinity stratification resulting from the discharge (Figure 4-22 to Figure 4-25). Similarly to the near bed maps, these curtains display maximum 24-hour average salinity average in relation to the reference site in Figure 4-1. As such, they are not instantaneous and exacerbate the impacts of the discharge by creating a composite of worst 24-hour average simulated conditions across different model cells.

Notwithstanding the above, as expected, 73 GL/year scenarios displayed the strongest salinity stratification, with top to bottom salinity differences of ~0.4 g/L at 300 m to ~0.8 g/L at 100 m from the diffuser. The differences between the 73 GL/year and 51 GL/year scenarios in these curtains were not significant (the 73 GL/year showed higher salinities further up in the water column, and extending further offshore as the plumes moved downslope), and no clear differences were observed between seasons. Along the diffuser manifold, these differences were up to 0.9 g/L with highest salinities over ports occupying approximately one-half of the water column. The curtains also provided an estimate of the thickness of the stratified layers along the different curtains, being the thinnest the furthest away from the diffuser (again noting the composite plots were not instantaneous). As a rough guide, the near-bed saline layers away from the diffuser were of the order of 3 to 7 m from the bed. A weak degree of stratification occurred beyond 500 m from the diffuser, salinity was elevated across the entire water column, with salinities in the plume barely discernible at around 0.1 g/L or less than background.

The level of salinity change and associated salinity stratification was significantly reduced for Turn Down scenarios. For example, the summer Turn Down scenario showed the dark shades of blue across generally all curtains (i.e., insignificant changes with respect to the reference site), and no evidence of the plume beyond 300 m from the diffuser. At the manifold location, the stratification was confined near the seabed with bottom-to-top salinity differences of ~0.2 g/L.

The stratified layer thickness (3 to 7 m) and time scale of salinity changes in the curtain plots (24 hours) were used to provide a high-level estimate of the potential for DO depletion in the stratified layer. Sediment oxygen uptake rates measurements undertaken in Spencer Gulf (Lauer; 2005, Chapter 5) nearest the site (Wallaroo and the Gutter) varied between approximately 0.7 g/m^2/day and 1 g/m^2/day. DO levels in the return water have been assumed at 80% saturation for the purpose of the present analysis, and a value of 5.0 mg/L is an

appropriate low estimate, noting that BHP estimated it to be 5.5 mg/L (BHP 2009). For the provided stratified layer thickness, and assuming a completely isolated layer from either surface reaeration or advected DO, the time scale for the dissolved oxygen concentrations to reduce to a 3.0 mg/L threshold (below which severe biological conditions may occur, Diaz and Rosenberg 1995) would be between 6 to 20 days, depending on the provided range of layer thickness and oxygen uptake rates. Over a 24-hour period, DO levels would therefore reduce to between 4.7 and 4.9 mg/L. As the plume moves, advection, dispersion and vertical mixing of higher concentration DO would therefore supplement any local oxygen deficit caused by the plume-induced stratification.

The level of oxygen and oxygen demand in the return water is to be confirmed by DIT as part of detailed design and noting that the Northern Water desalination plant site sediment oxygen uptake rate is uncertain, the above calculations have been undertaken based on appropriate assumptions (BHP 2009). The modelling results indicate DO depletion is unlikely to happen as a result of the return water discharge.

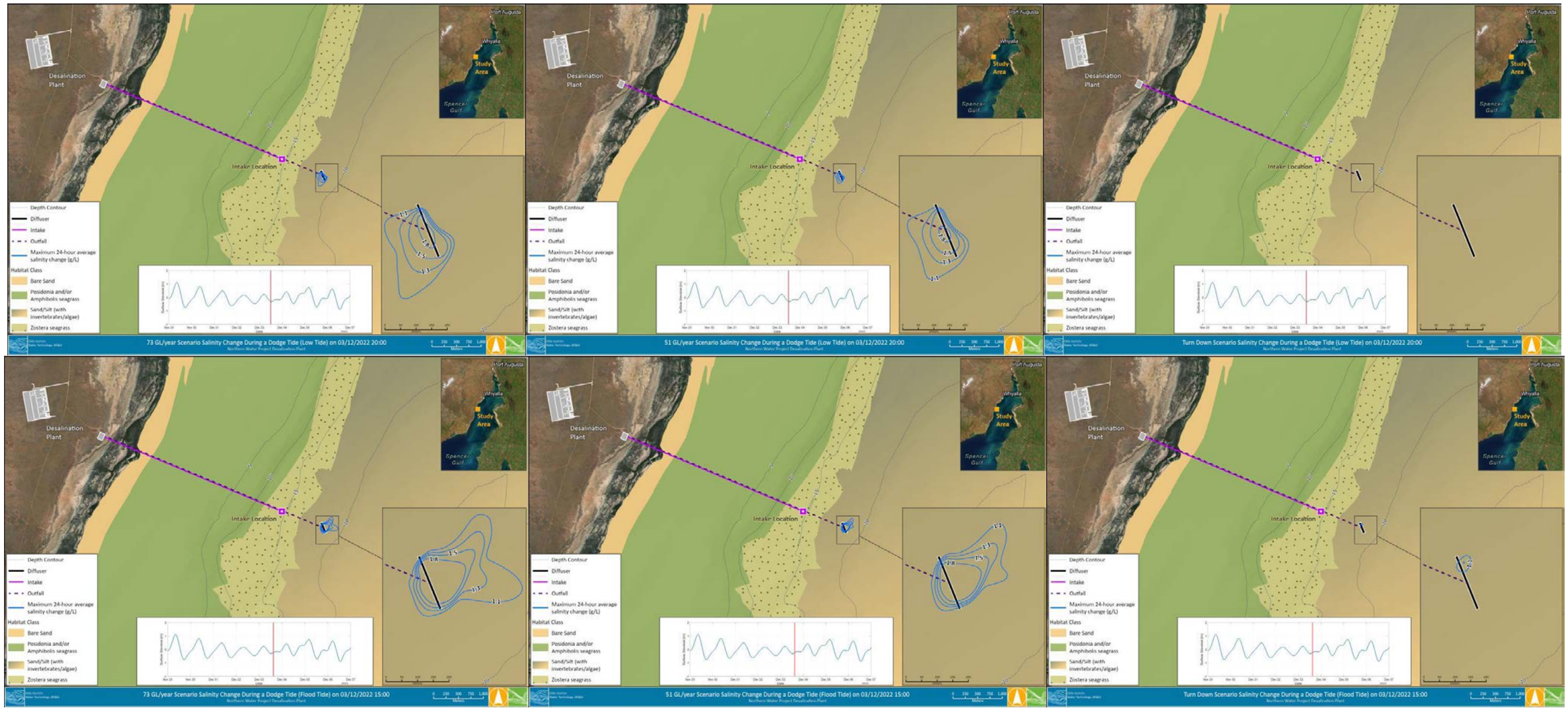


Figure 4-14 Salinity Change During Low (Upper Panels) and Flooding (Lower Panels) Stages of a 'Dodge' Tide. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

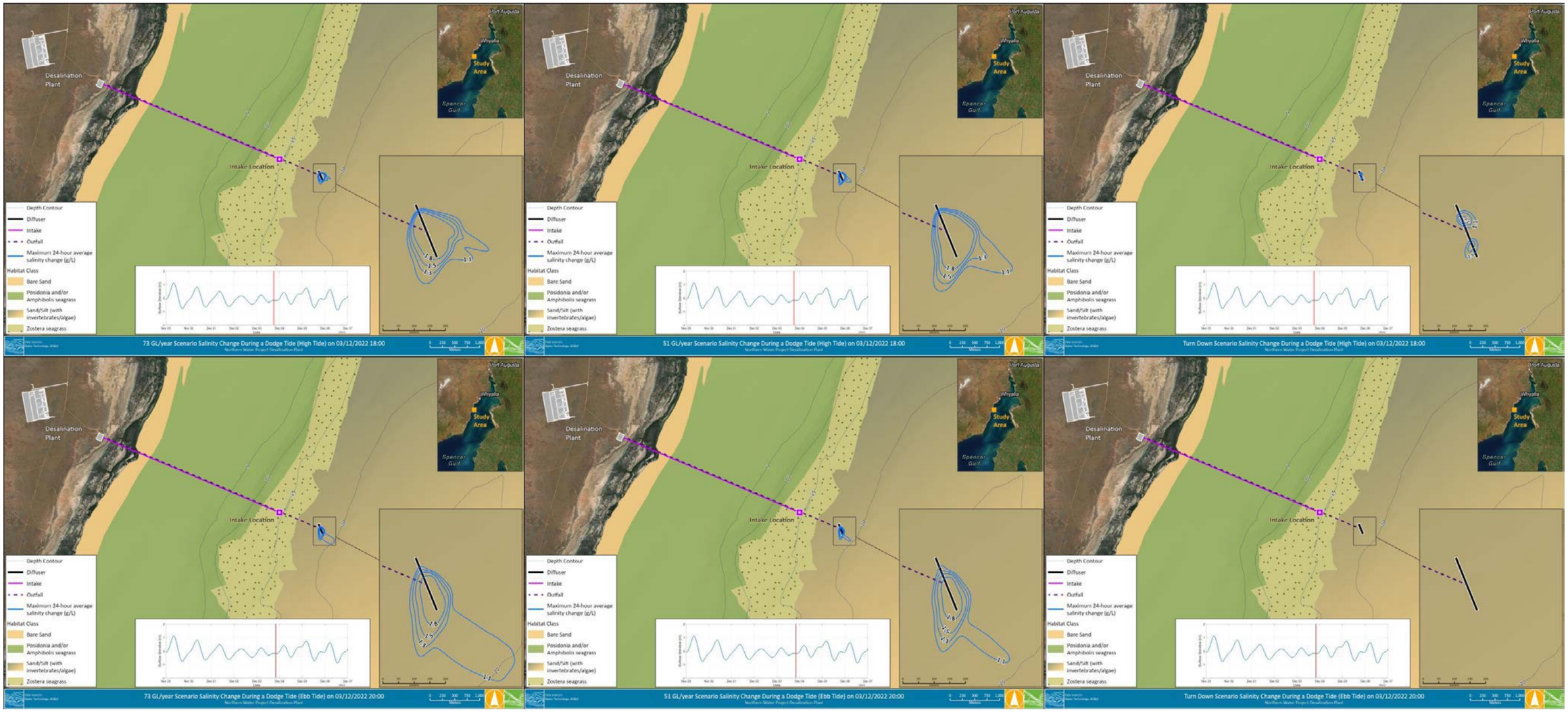


Figure 4-15 Salinity Change During High (Upper Panels) and Ebbing (Lower Panels) Stages of a 'Dodge' Tide. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

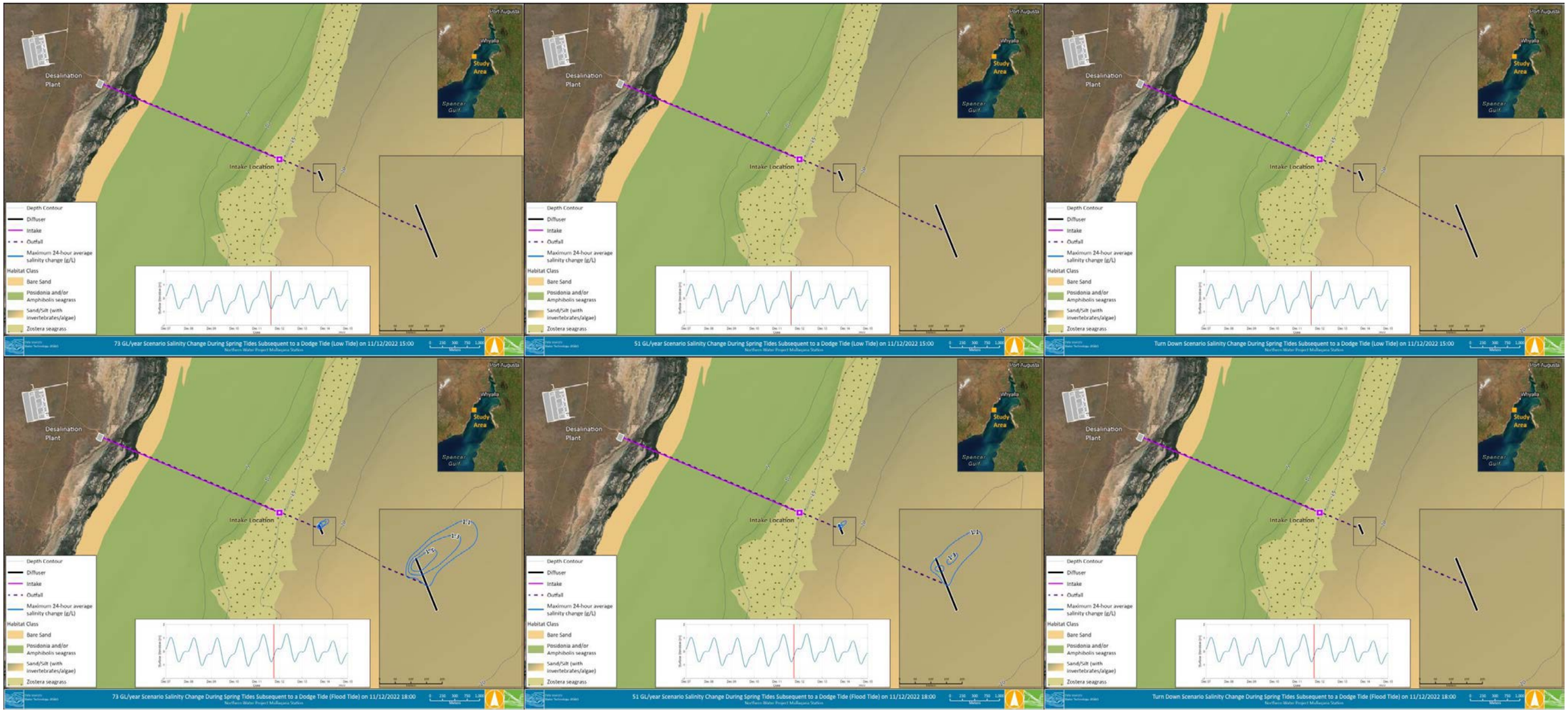


Figure 4-16 Salinity Change During Low (Upper Panels) and Flooding (Lower Panels) Stages of a Spring Tide. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

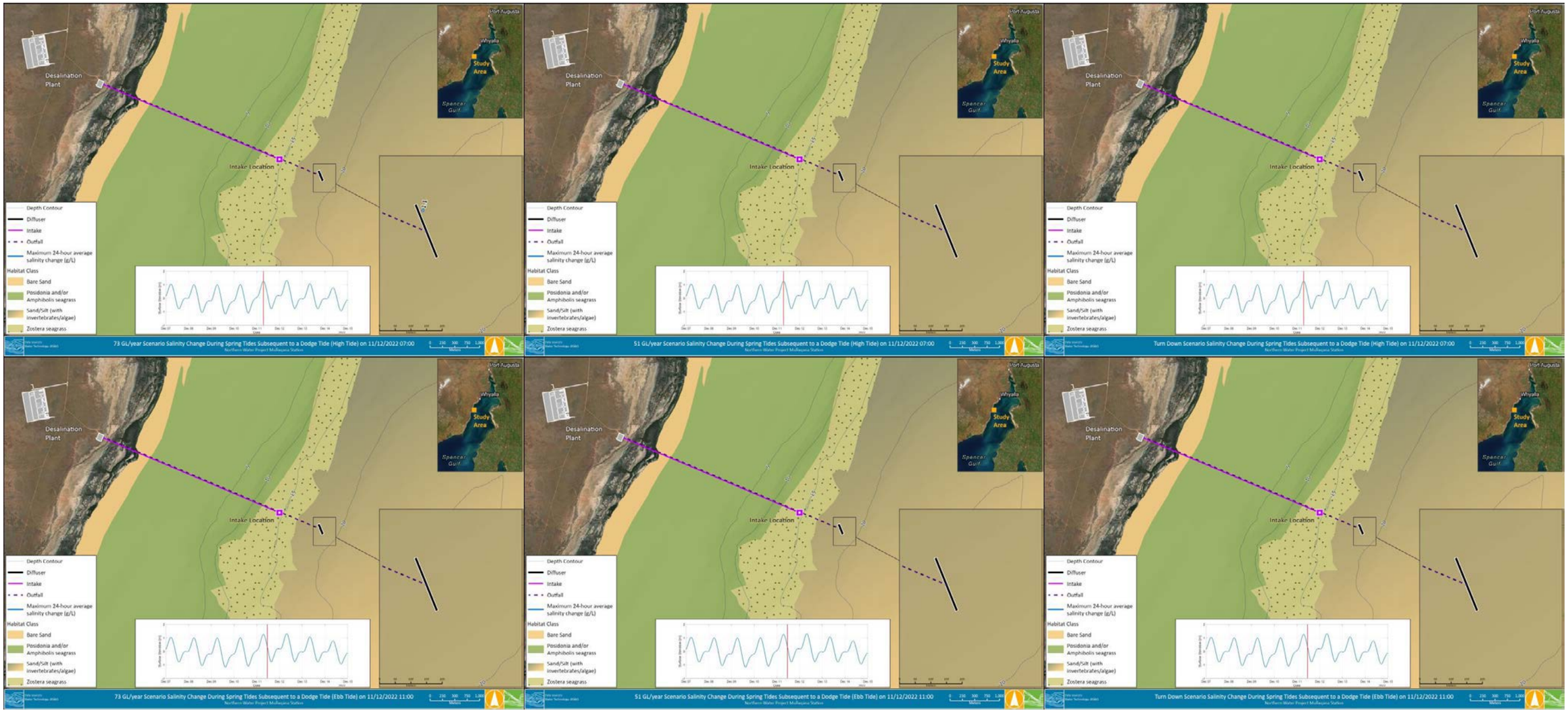


Figure 4-17 Salinity Change During High (Upper Panels) and Ebbing (Lower Panels) Stages of a Spring Tide. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

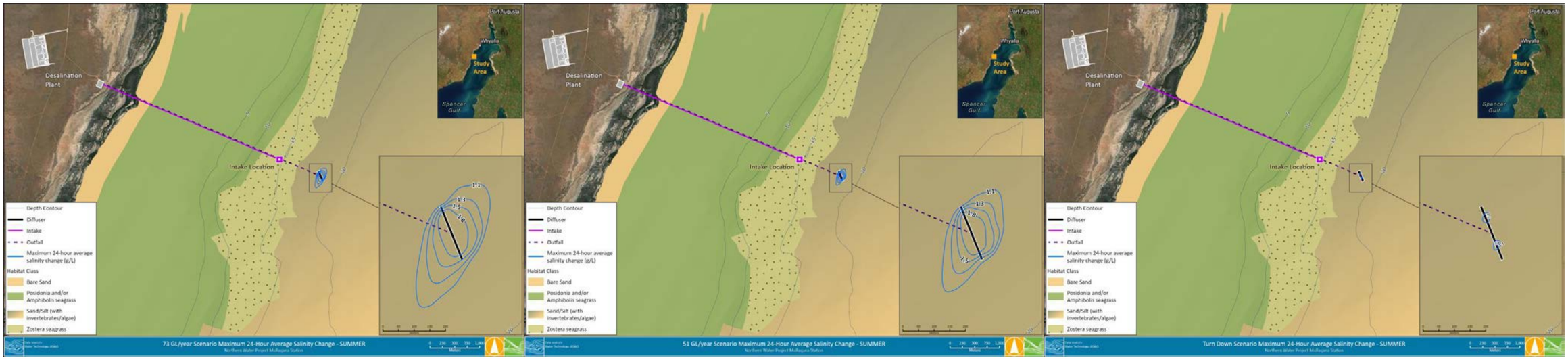


Figure 4-18 24-Hour Average Salinity Change – Summer 2022-2023 (Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario)

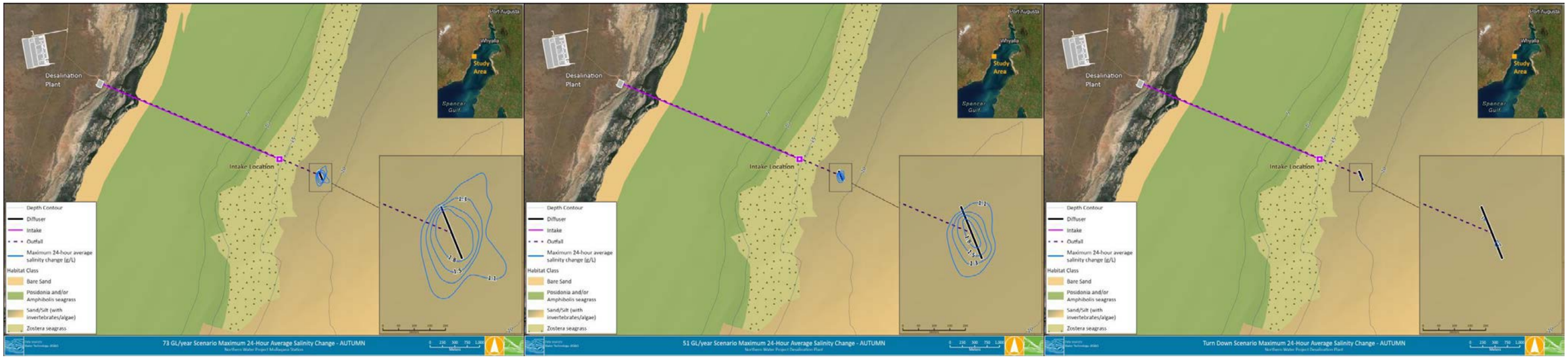


Figure 4-19 24-Hour Average Salinity Change – Autumn 2023 (Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario)

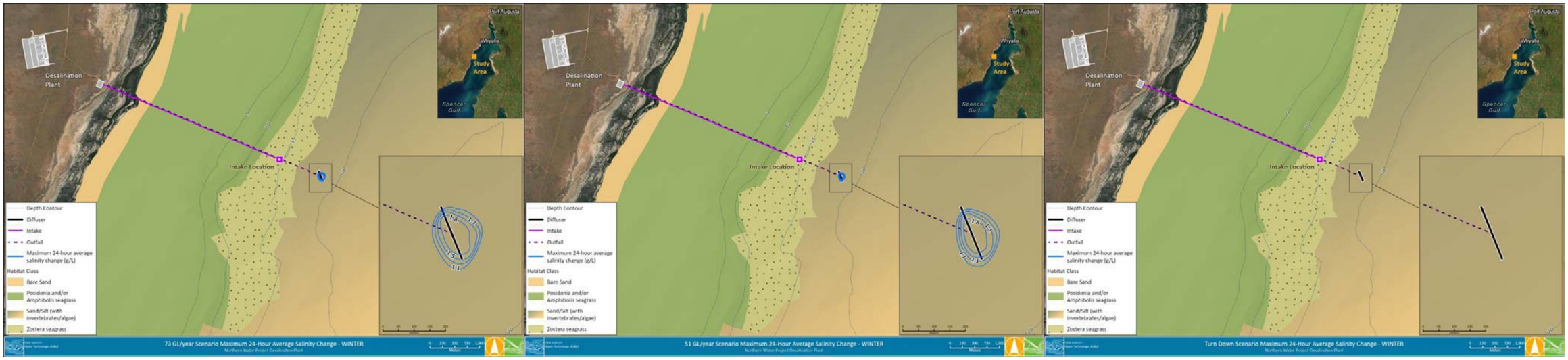


Figure 4-20 24-Hour Average Salinity Change – Winter 2023 (Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario)

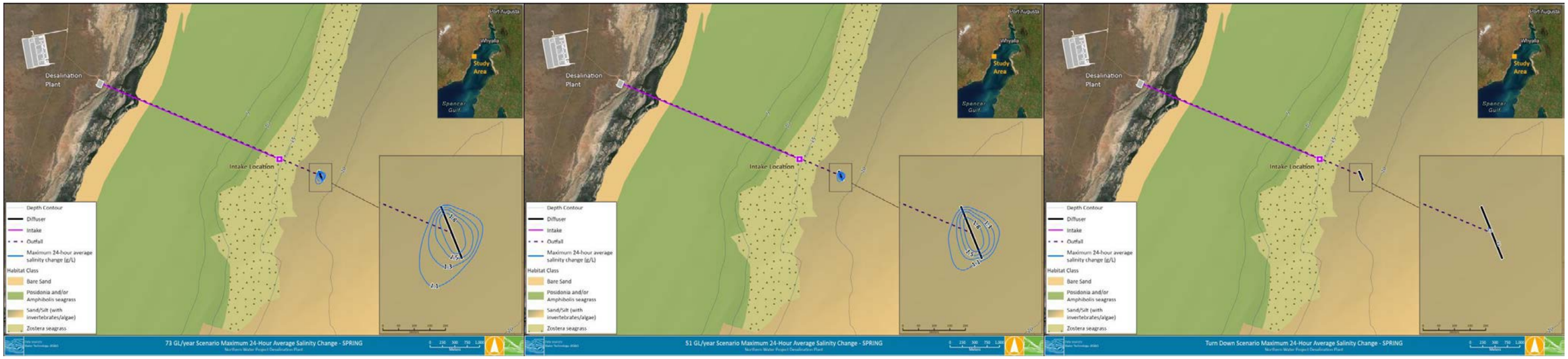


Figure 4-21 24-Hour Average Salinity Change – Spring 2023 (Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario)

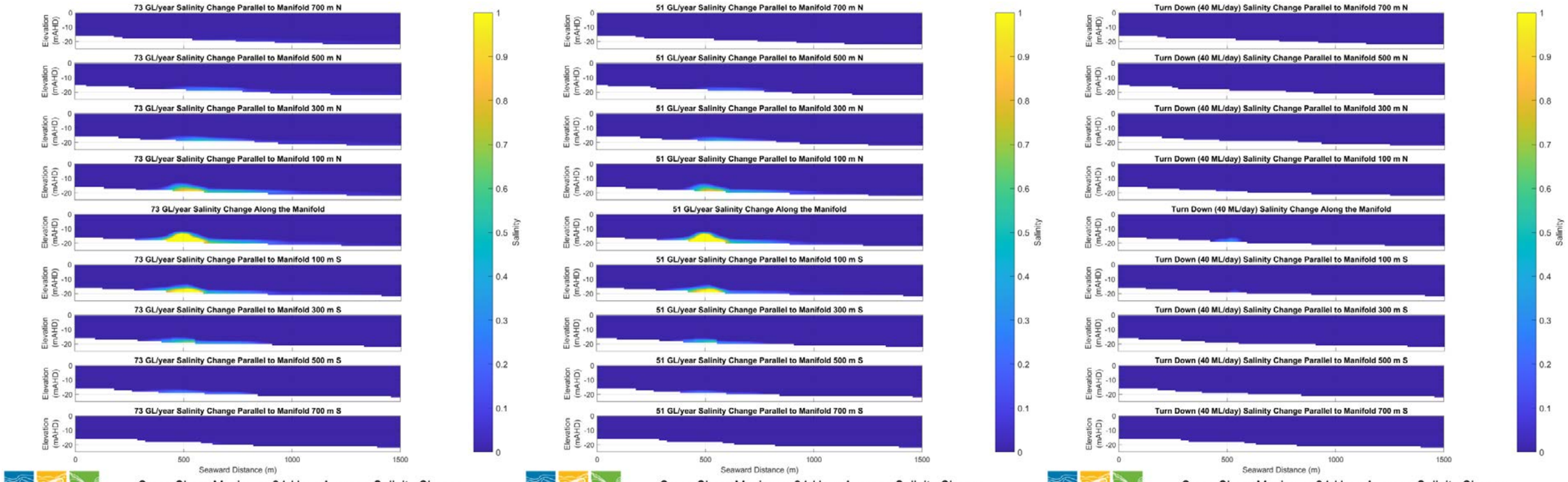


Figure 4-22 24-Hour Average Salinity Change Curtains –Summer 2022-23. The seaward distance is the distance from the most onshore point in Figure 4-1.

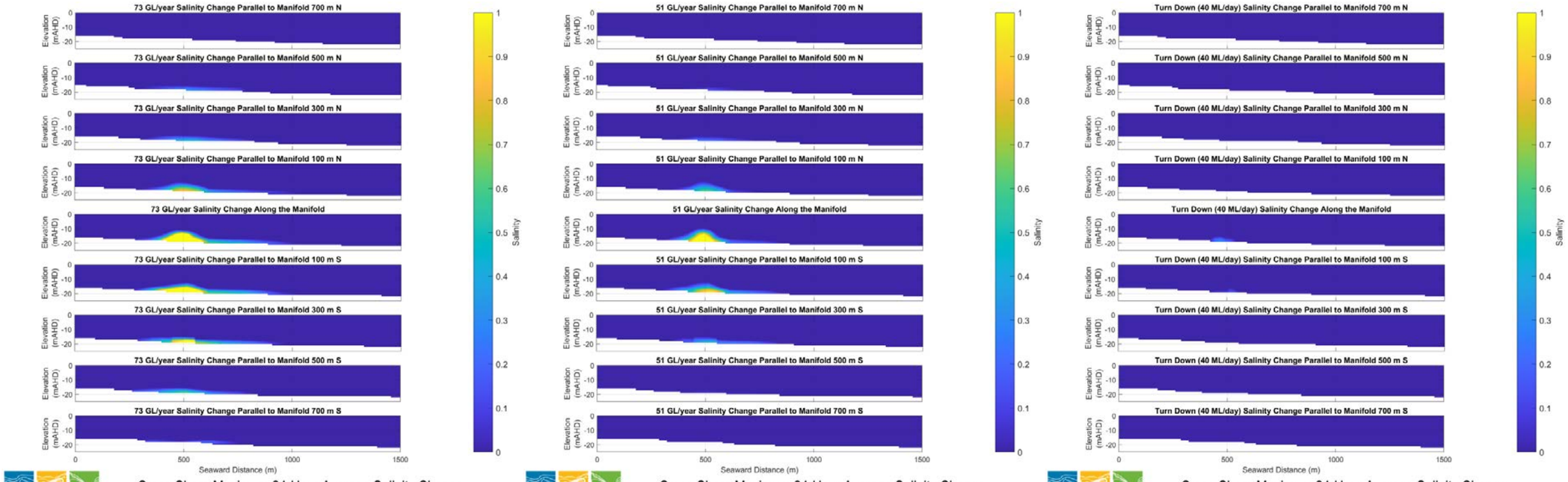


Figure 4-23 24-Hour Average Salinity Change Curtains – Autumn 2023. The seaward distance is the distance from the most onshore point in Figure 4-1.

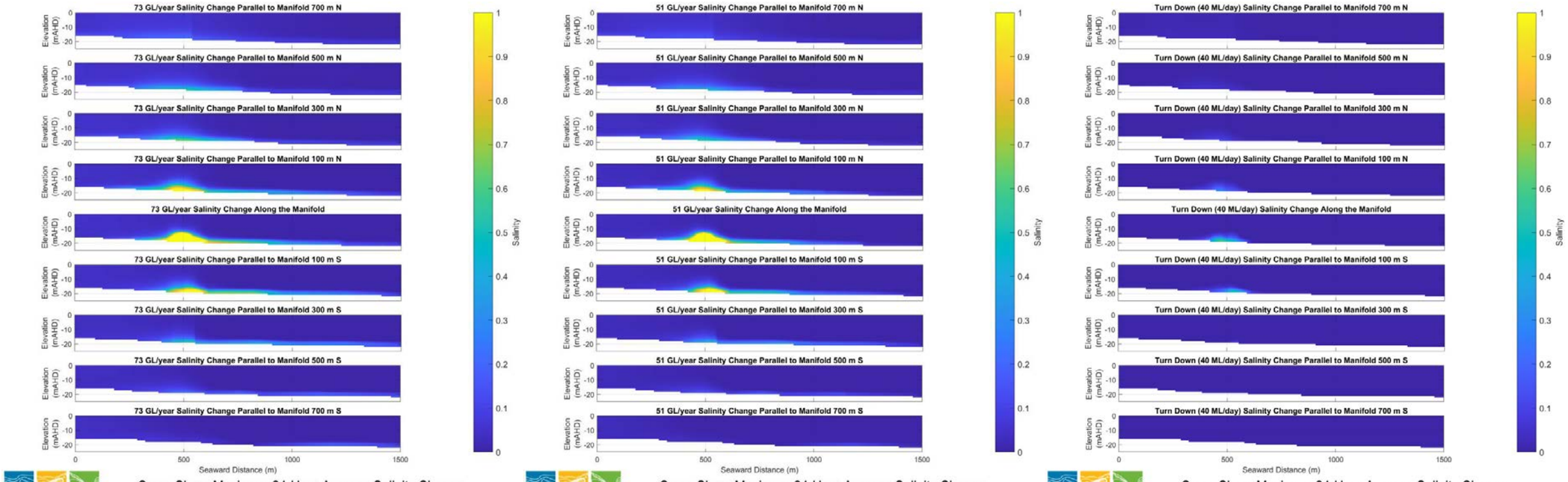


Figure 4-24 24-Hour Average Salinity Change Curtains – Winter 2023. The seaward distance is the distance from the most onshore point in Figure 4-1.

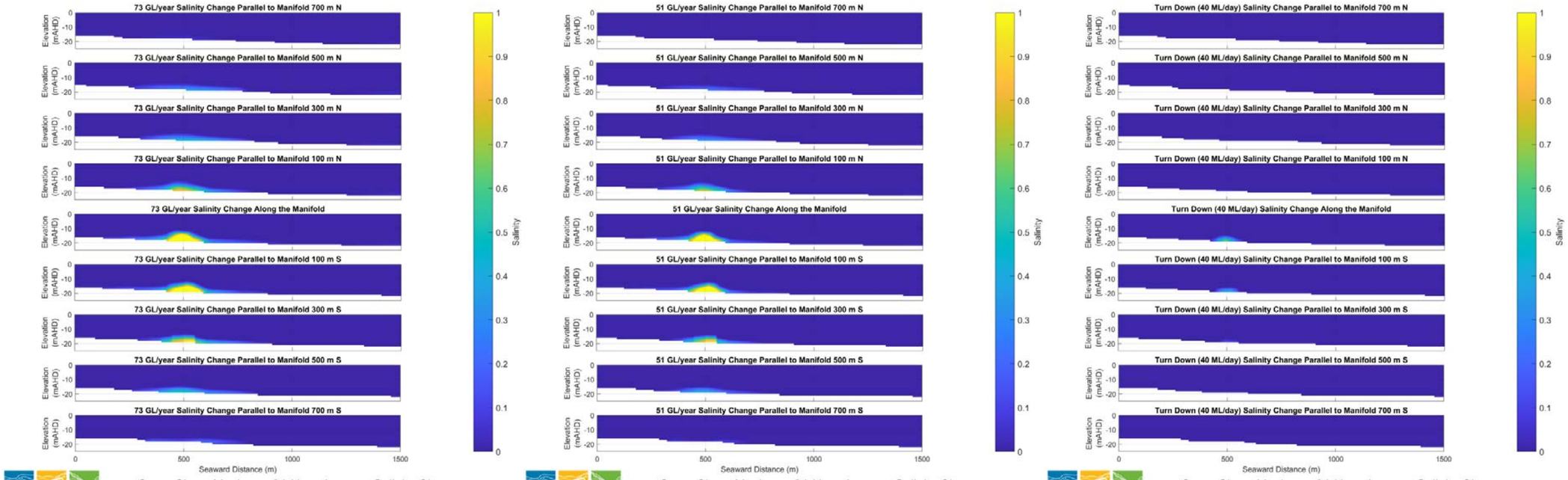


Figure 4-25 24-Hour Average Salinity Change Curtains – Spring 2023. The seaward distance is the distance from the most onshore point in Figure 4-1.

Table 4-4 Areas within Different Salinity Change Contours

Scenario	Season	Salinity Change Area (Ha)			
		1.1 g/L	1.3 g/L	1.5 g/L	1.8 g/L
73 GL/year scenario	Summer 2022-23	5.683	3.428	2.120	1.075
	Autumn 2023	6.672	3.596	2.402	1.458
	Winter 2023	2.333	1.810	1.394	0.851
	Spring 2023	3.710	2.243	1.355	0.691
51 GL/year scenario	Summer 2022-23	4.165	2.552	1.615	0.863
	Autumn 2023	2.774	1.518	0.832	0.292
	Winter 2023	1.922	1.475	1.107	0.632
	Spring 2023	2.382	1.457	0.946	0.483
Minimum Capacity (Turn Down) scenario	Summer 2022-23	0.103	0.005	0.000	0.000
	Autumn 2023	0.000	0.000	0.000	0.000
	Winter 2023	0.000	0.000	0.000	0.000
	Spring 2023	0.008	0.000	0.000	0.000

Table 4-5 Distances from the 24-Hour Average 1.3, 1.5 and 1.8 g/L Contours to the Outfall

Scenario	Period	1.3 g/L		1.5 g/L		1.8 g/L	
		Maximum Distance (m)	Average Distance (m)	Maximum Distance (m)	Average Distance (m)	Maximum Distance (m)	Average Distance (m)
73 GL/year	Summer 2022-23	163.8	105.7	105.0	78.0	70.0	54.7
	Autumn 2023	155.7	106.5	122.0	84.8	89.5	64.5
	Winter 2023	97.6	71.6	87.9	62.8	72.3	49.9
	Spring 2023	107.9	80.4	79.9	61.1	62.9	45.3
51 GL/year	Summer 2022-23	132.1	90.3	88.1	68.2	64.3	49.0
	Autumn 2023	96.0	66.6	71.4	48.5	45.4	29.5
	Winter 2023	89.0	65.5	78.3	56.6	61.3	41.9
	Spring 2023	80.2	63.9	68.3	51.3	56.0	36.3
Minimum Capacity / Turn Down	Summer 2022-23	37.1	32.9	0.0	0.0	0.0	0.0
	Autumn 2023	0.0	0.0	0.0	0.0	0.0	0.0
	Winter 2023	0.0	0.0	0.0	0.0	0.0	0.0
	Spring 2023	0.0	0.0	0.0	0.0	0.0	0.0

4.4.3.3 Sensitive Receptors

Salinity change timeseries and cumulative distributions at sensitive receptor locations shown in Figure 4-1 are presented from Figure 4-26 and Figure 4-33.

As discussed earlier, salinity change for all scenarios varied in response to tidal movements. Results followed dilution timeseries showing that the Sand/Invertebrates/Algae location experienced the highest salinity changes of all sensitive receptor locations. This is not surprising given the location is next to the diffuser. As expected, 24-hour average peaks were lower than peak salinity changes, with highest 24-hour average peaks approximately 25% lower, again providing further indication that water did not accumulate over extended periods of time.

At the Sand/Invertebrates/Algae location, 73 GL/year showed median salinity changes from 0.1 g/L (Winter) to 0.2 g/L (Summer, Autumn and Spring). 90th percentile and maximum salinity changes were around 1.7 g/L and 2.3 g/L, respectively. Median, 90th percentile and maximum 24-hour salinity changes for 73 GL/year scenario were around 0.4 g/L, 0.8 g/L and 1.4 g/L, respectively.

In comparison to the 73 GL/year scenario, salinity changes were lower for the 51 GL/year scenario at the Sand/Invertebrates/Algae location, with median salinity changes around 0.07 g/L for all seasons. 90th percentile and maximum salinities were about 1.5 g/L and 2.1 g/L, respectively. In contrast, median, 90th percentile and maximum 24-hour salinity changes for the 51 GL/year scenario were from 0.2 g/L (Autumn) to 0.3 g/L (other seasons), around 0.7 g/L and between 0.8 g/L (Spring) and 1.3 g/L (Winter), respectively.

Around late June to early July, both the salinity change and the 24-hour average salinity change under the Turn Down scenario were slightly higher than those under the 73 GL/year and 51 GL/year scenarios, which was not expected. The reason for these effects is unclear and likely associated with specific hydrodynamic conditions affecting behaviour of the plume. Further analysis of the period was not pursued given the short duration of the event and general low impact of the Turn Down scenario.

At receptors located further away from the diffuser (i.e. all receptor locations except the *Zostera* SP habitat boundary, Sand/Bioturbation, and Sand/Invertebrates/Algae), all simulations exhibited maximum and 24-hour average maximum salinity changes below 0.4 g/L and 0.2 g/L, respectively.

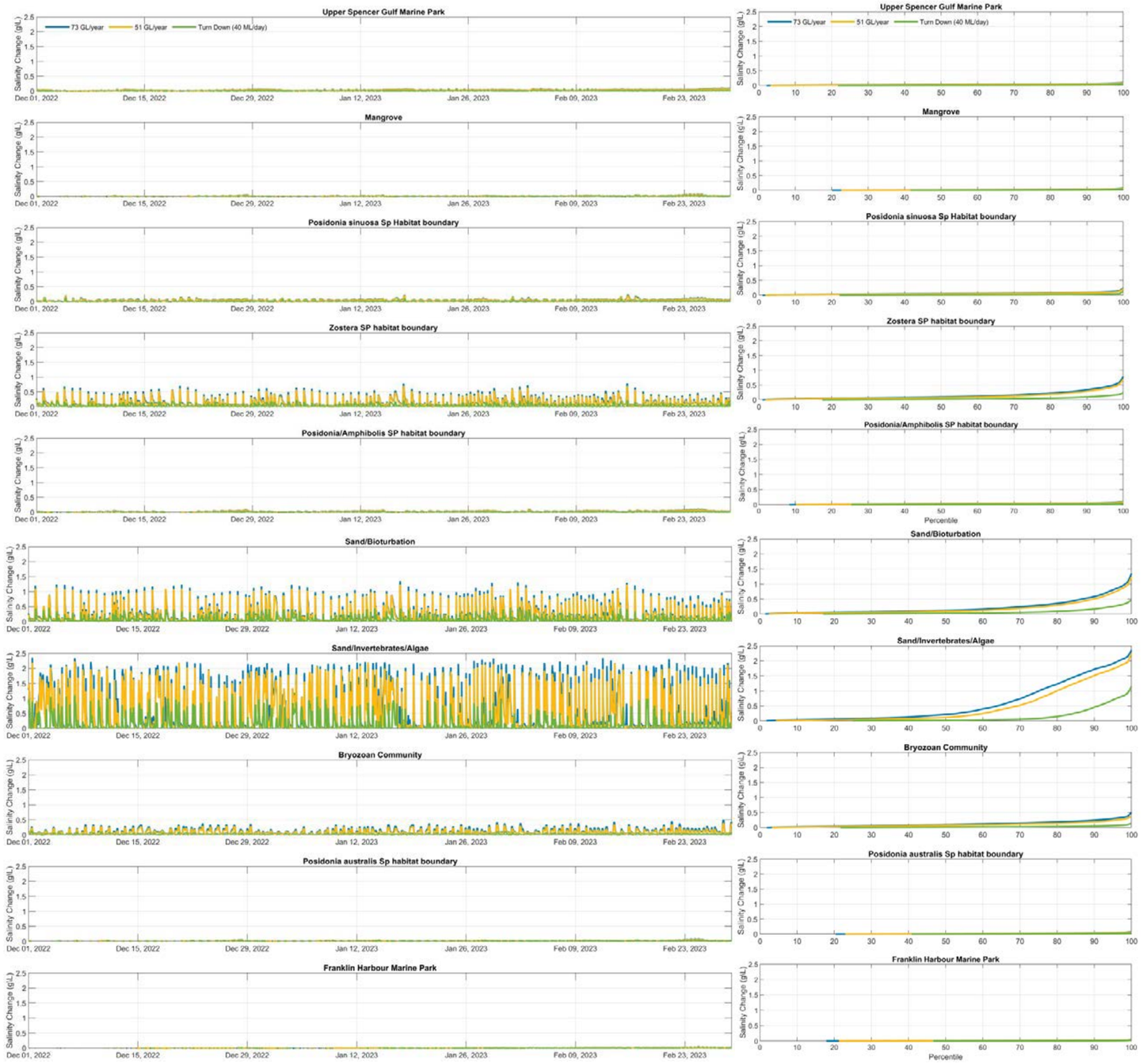


Figure 4-26 Salinity-Change Timeseries (Left Panel) Timeseries and Cumulative Distribution (Right Panel) at Sensitive Receptors (Summer 2022-2023)



Figure 4-27 24-Hour Moving Average Salinity Change Timeseries (Left Panel) and Cumulative Distribution (Right Panel) at Sensitive Receptors (Summer 2022-2023)

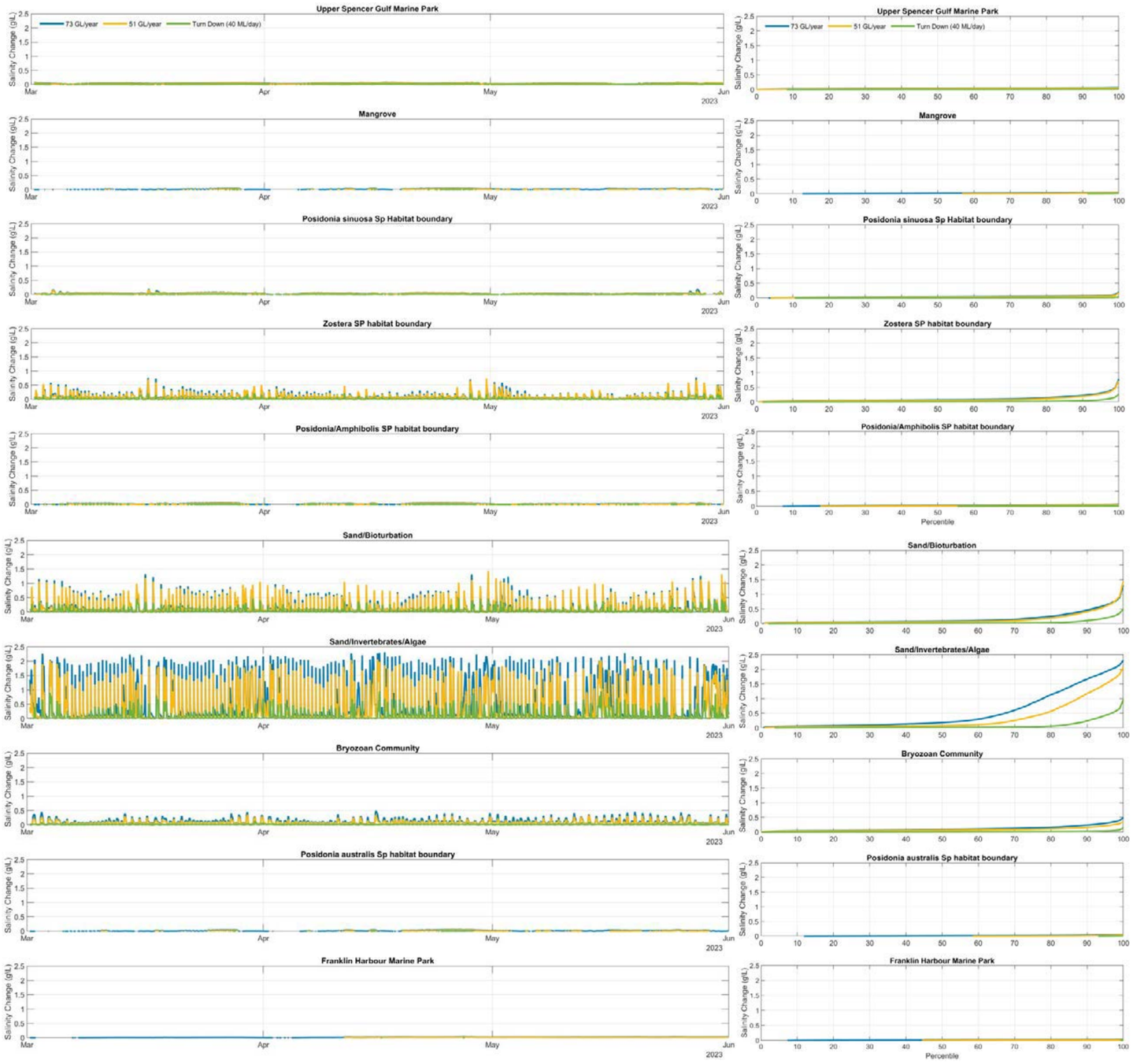


Figure 4-28 Salinity-Change Timeseries (Left Panel) Timeseries and Cumulative Distribution (Right Panel) at Sensitive Receptors (Autumn 2023)



Figure 4-29 24-Hour Moving Average Salinity Change Timeseries (Left Panel) and Cumulative Distribution (Right Panel) at Sensitive Receptors (Autumn 2023)



Figure 4-30 Salinity-Change Timeseries (Left Panel) Timeseries and Cumulative Distribution (Right Panel) at Sensitive Receptors (Winter 2023)

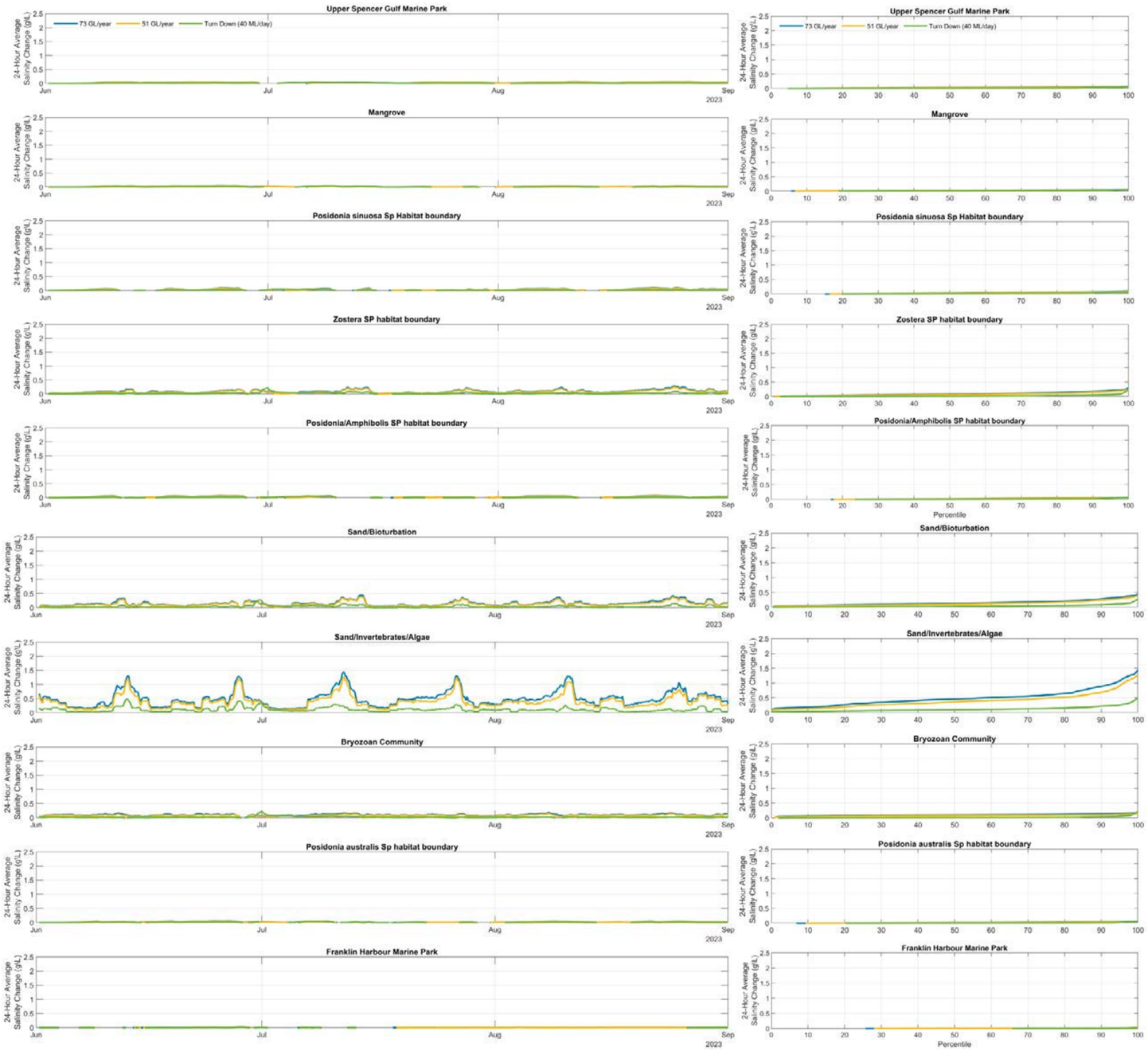


Figure 4-31 24-Hour Moving Average Salinity Change Timeseries (Left Panel) and Cumulative Distribution (Right Panel) at Sensitive Receptors (Winter 2023)



Figure 4-32 Salinity Timeseries (Left Panel) and Cumulative Distribution (Right Panel) at Sensitive Receptors (Spring 2023)

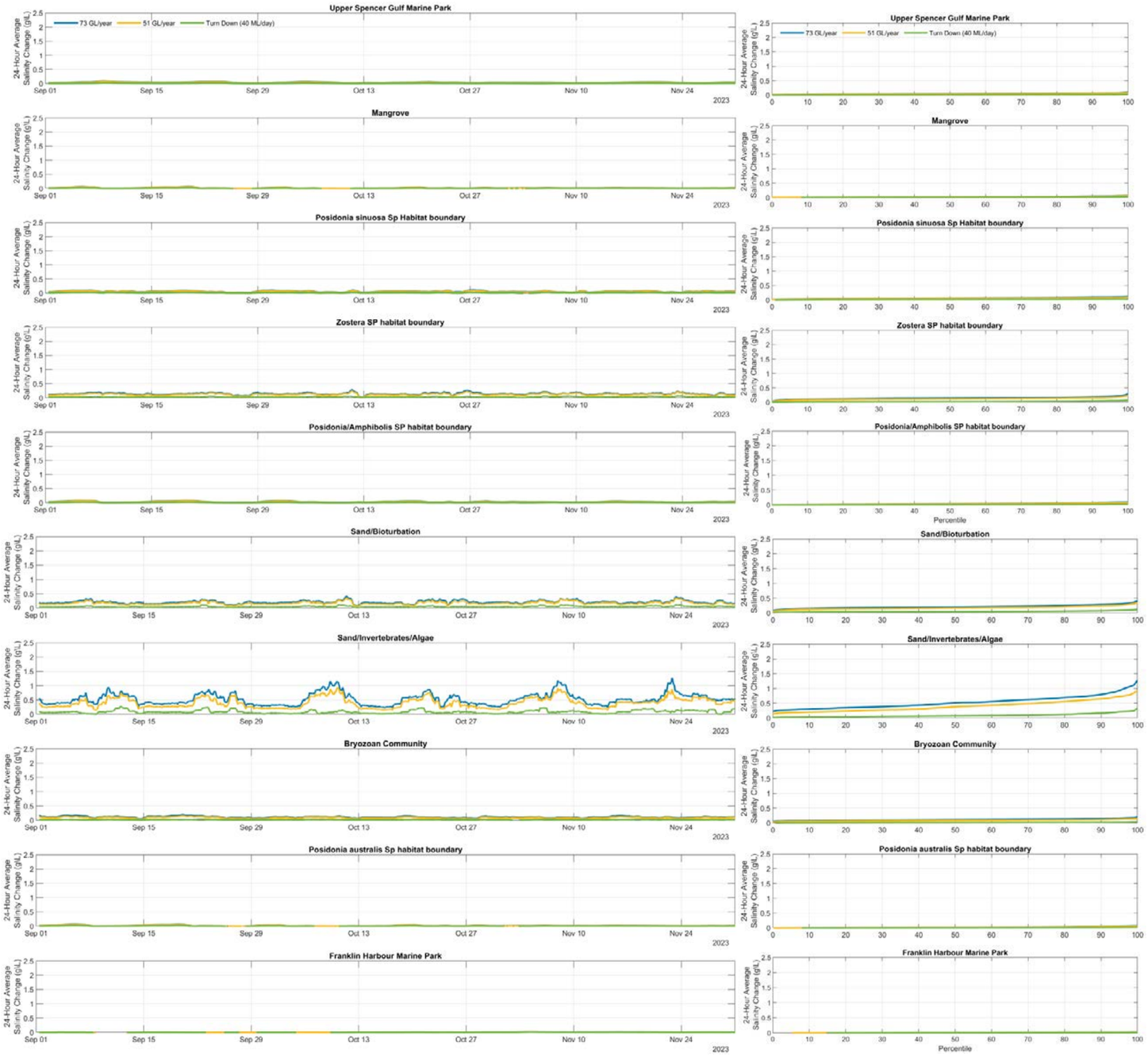


Figure 4-33 24-Hour Moving Average Salinity Change Timeseries (Left Panel) and Cumulative Distribution (Right Panel) at Sensitive Receptors (Spring 2023)

4.4.4 Temperature change

Temperature differences between scenarios including and excluding the desalination plant discharge are presented in this section. These include, for each season, maps showing differences between:

- Median temperature from return water scenarios and the 80th percentile temperature in the baseline scenario (Figure 4-34 to Figure 4-37); and
- Maximum temperature between return water and baseline scenarios (Figure 4-38 to Figure 4-41).

The first set of maps (50th percentile of discharge minus 80th percentile baseline) followed the similar process adopted to derive guideline values using reference-site data for slightly to moderately disturbed ecosystems (ANZG 2018). In this case, if median results of impacted sites do not exceed 80th percentile values of unimpacted sites, impacts may be considered acceptable. The difference between maxima (second set of maps) were produced to demonstrate effects of temperature associated with the desalination plant discharge were minimal.

The difference between the median temperature under the scenarios and the 80th percentile temperature under the baseline ranged from −0.1 to −2.8 °C across different scenarios and seasons. At the surface, the differences were the order of -0.8 and -0.6 °C (Summer); -2.6 to -2.8 °C (Autumn), -1.2 to -1.8 °C (Winter) and -2.3 to -1.8 °C (Spring). Noting that the contours were practically identical across scenarios, this was an indication that the differences were the same to be expected taking the difference between median and 80th temperature percentiles of the baseline.

For the near bed maps, differences were closer to zero near the outfall, albeit still displaying negative values (>-0.1 °C). For near bed contours >-0.2 °C, differences between scenarios could not be readily discerned, indicating the effect of the desalination plant was minimal.

The differences in maximum temperature between the scenarios and the baseline were small for all scenarios and seasons, generally less than 0.15 °C (being even lower near the surface), indicating that the impact on water temperature was minimal. Therefore, no further detailed analysis was undertaken.

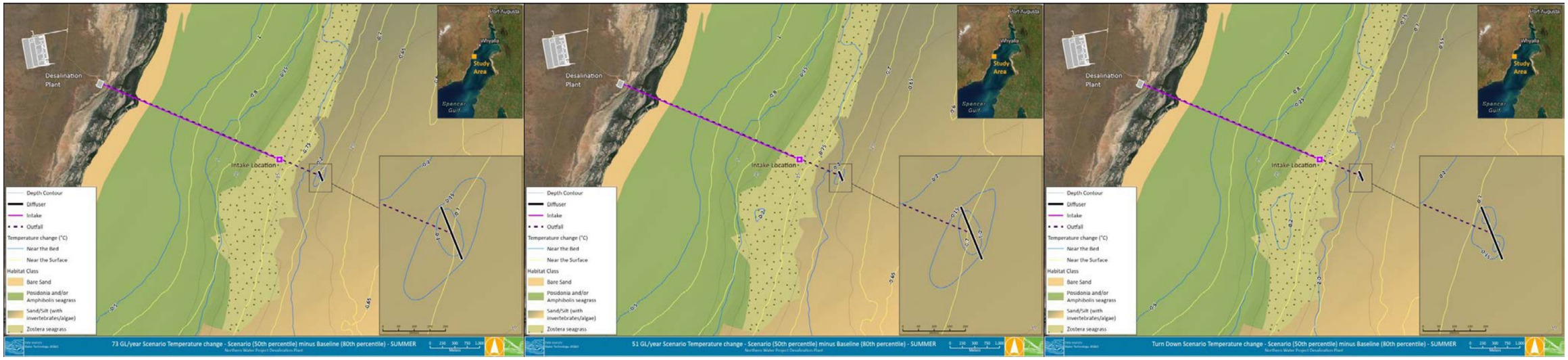


Figure 4-34 Summer Temperature change between scenarios and baseline (median scenario minus 80th percentile baseline). Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

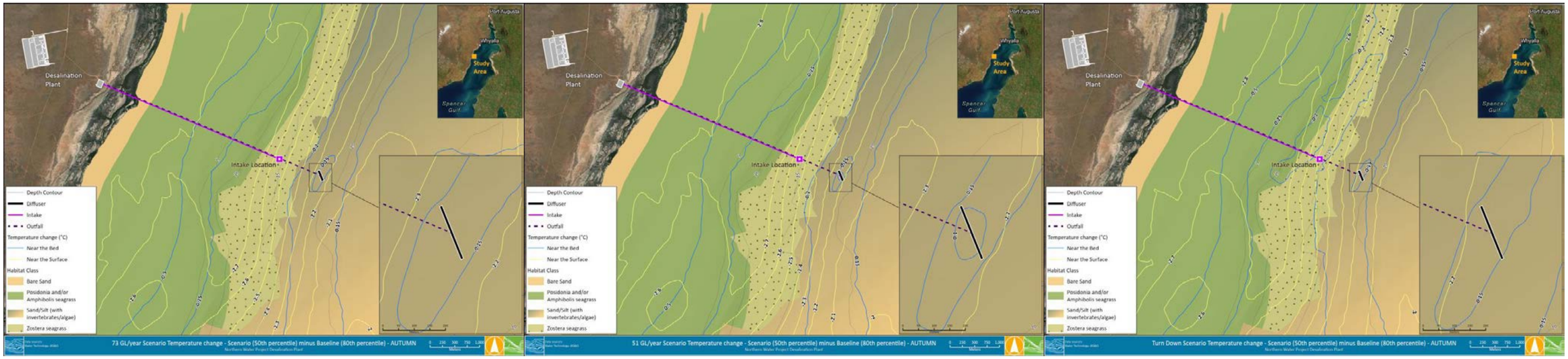


Figure 4-35 Autumn Temperature change between scenarios and baseline (median scenario minus 80th percentile baseline). Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

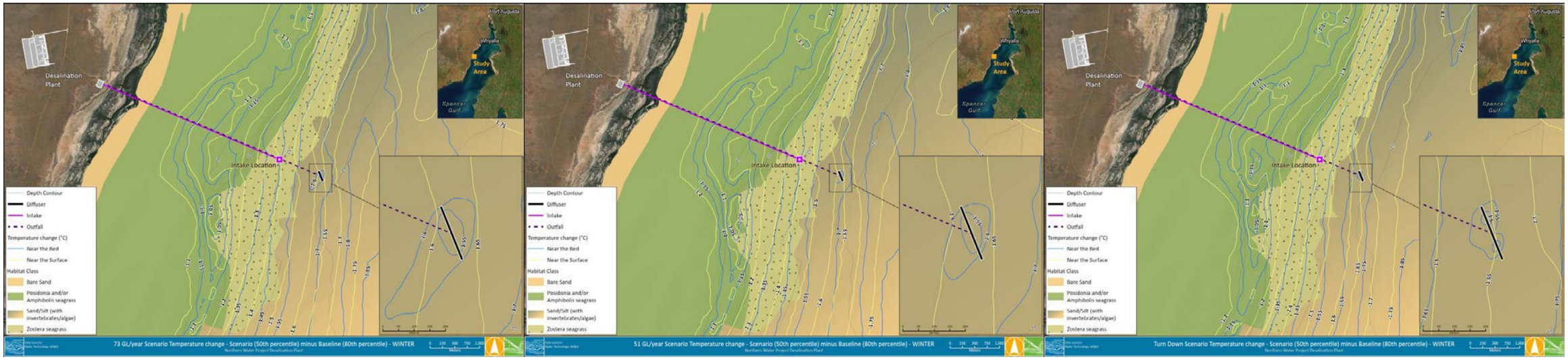


Figure 4-36 Winter Temperature change between scenarios and baseline (median scenario minus 80th percentile baseline). Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

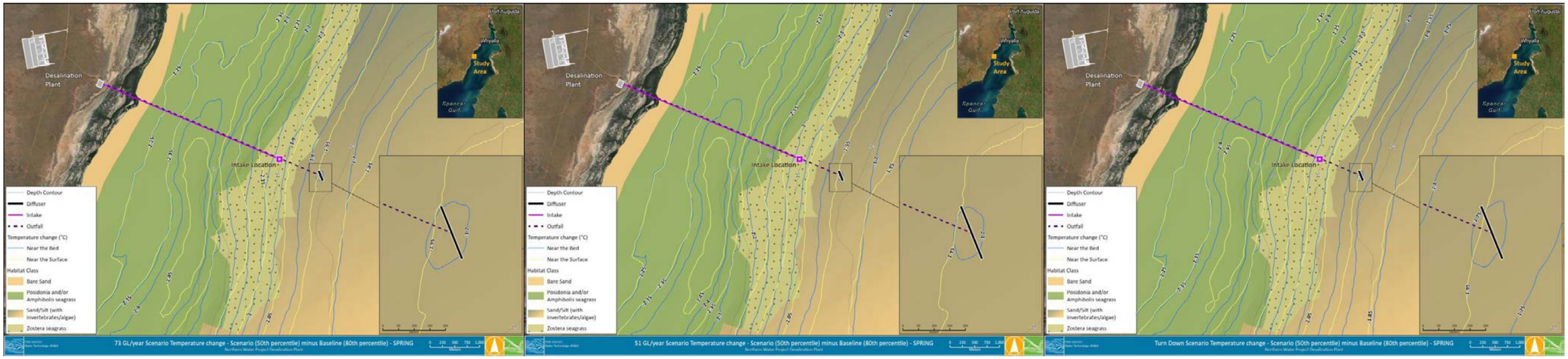


Figure 4-37 Spring Temperature change between scenarios and baseline (median scenario minus 80th percentile baseline). Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

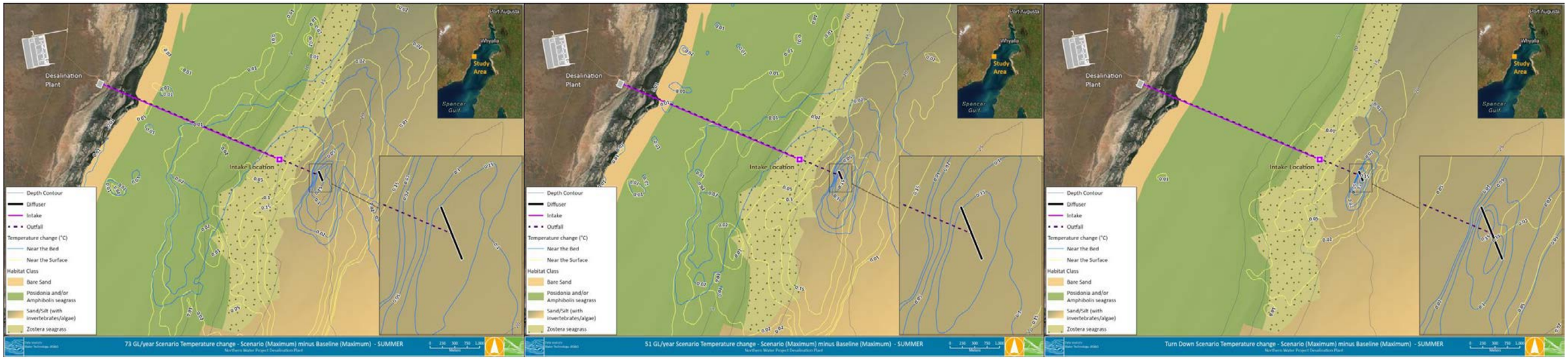


Figure 4-38 Summer Temperature change between scenarios and baseline (maximum scenario minus maximum baseline). Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

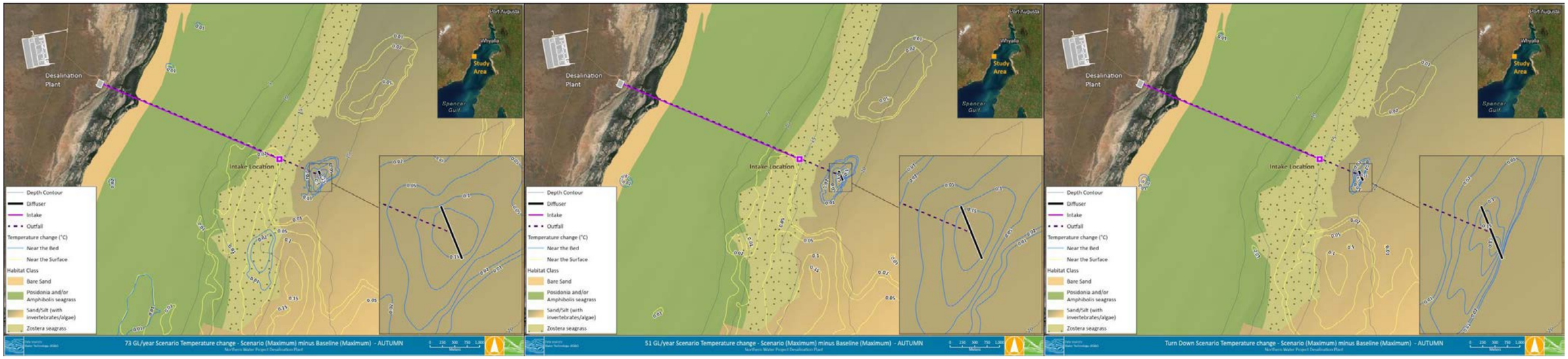


Figure 4-39 Autumn Temperature change between scenarios and baseline (maximum scenario minus maximum baseline). Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario



Figure 4-40 Winter Temperature change between scenarios and baseline (maximum scenario minus maximum baseline). Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

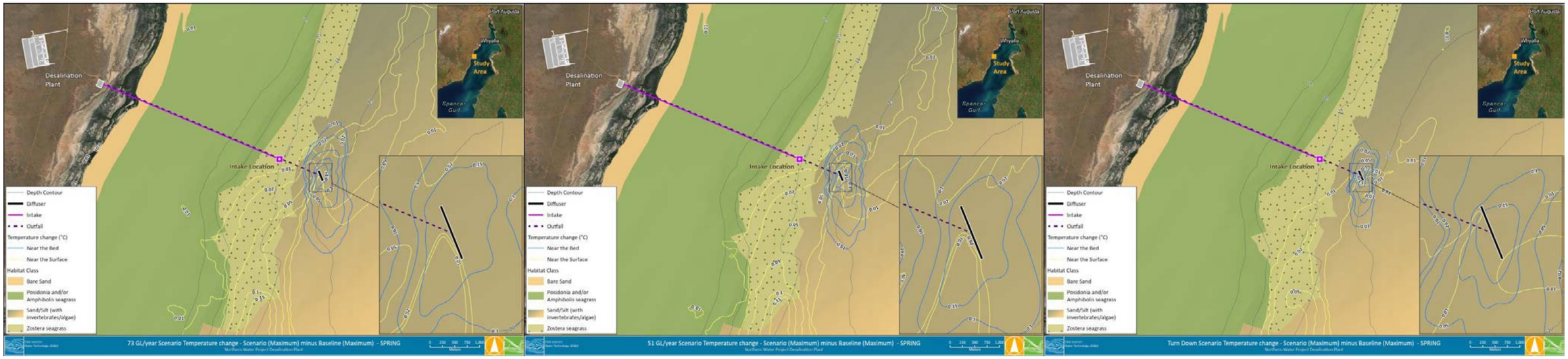


Figure 4-41 Spring Temperature change between scenarios and baseline (maximum scenario minus maximum baseline). Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

4.5 Intake Effects

Two considerations were made to assess potential intake issues associated with the Northern Water desalination plant operation, as follows:

- Short-circuiting between the desalination plant discharge and the seawater intake; and
- Entrainment of relevant fisheries larvae into the seawater intake structures.

Short circuiting between the outfall and intake may affect performance of the desalination plant as higher salinity water would be fed into the desalination process. Entrainment on the other hand refers to the process of small marine fauna that are unable to resist (via swimming or other avoidance strategies) being removed from its habitat and entering the plant water intake, likely affecting their survival in the process.

4.5.1 Short Circuiting

Short-circuiting was assessed by comparing salinity differences between simulations with and without the desalination plant at the intake location. Timeseries and cumulative distributions for each of the simulations are shown (Figure 4-42 and Figure 4-43). Differences to the baseline were generally small with largest differences associated with the transition from neaps to springs during ebb tides. The maximum change to the baseline of 73 GL/year and 51 GL/year scenarios was around 0.25 g/L. Median changes were less substantial, less than 0.1 g/L for 73 GL/year and 51 GL/year scenarios. The maximum and median changes of the Turn Down scenario were below 0.2 g/L and below 0.05 g/L, respectively. These changes are based on the lowest level of the intake and can be further improved with minor adjustments to the design (e.g., elevating the extraction levels, moving relatively short distances inshore, etc.).

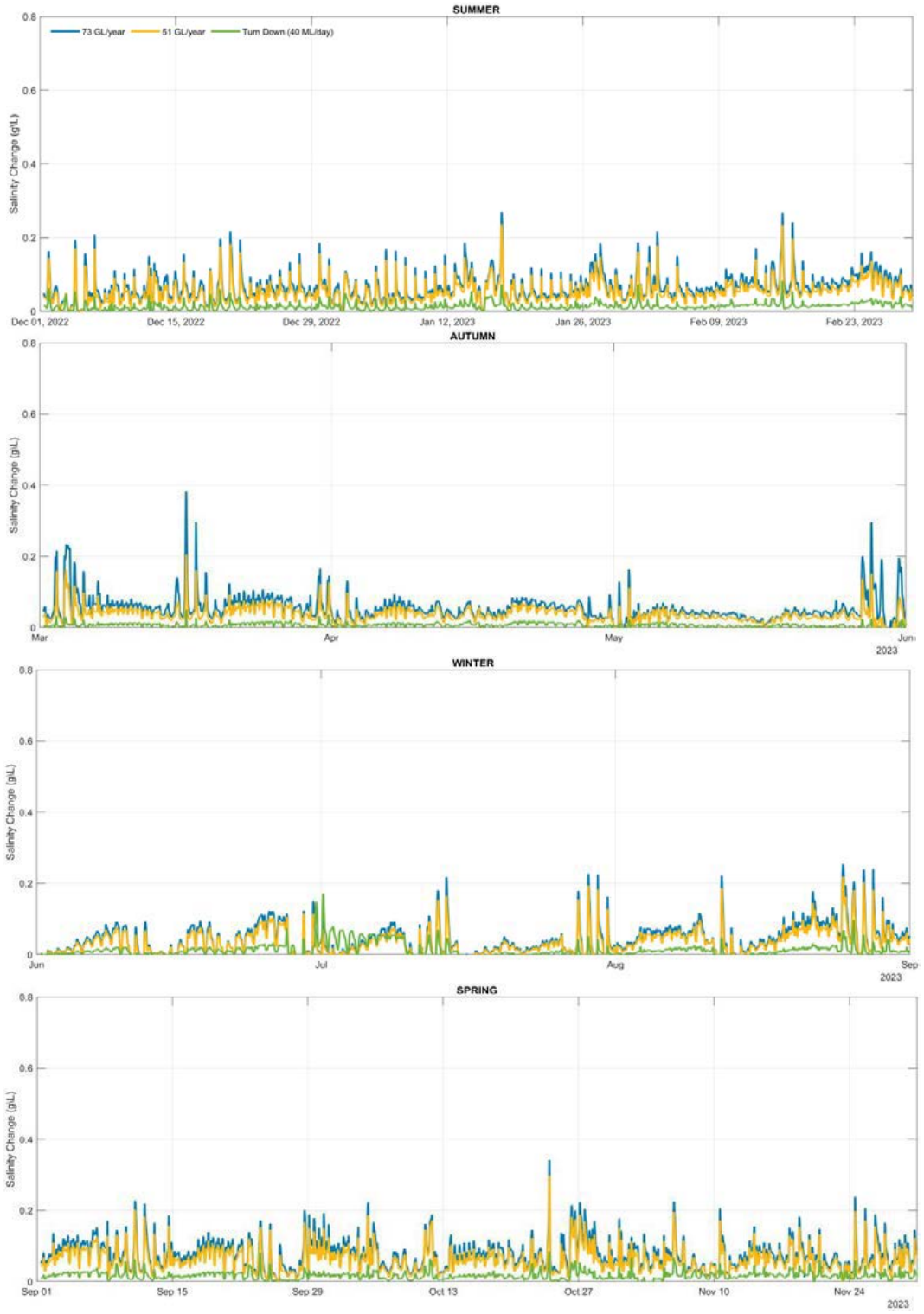


Figure 4-42 Salinity Differences Timeseries at the Desalination Plant Intake Location

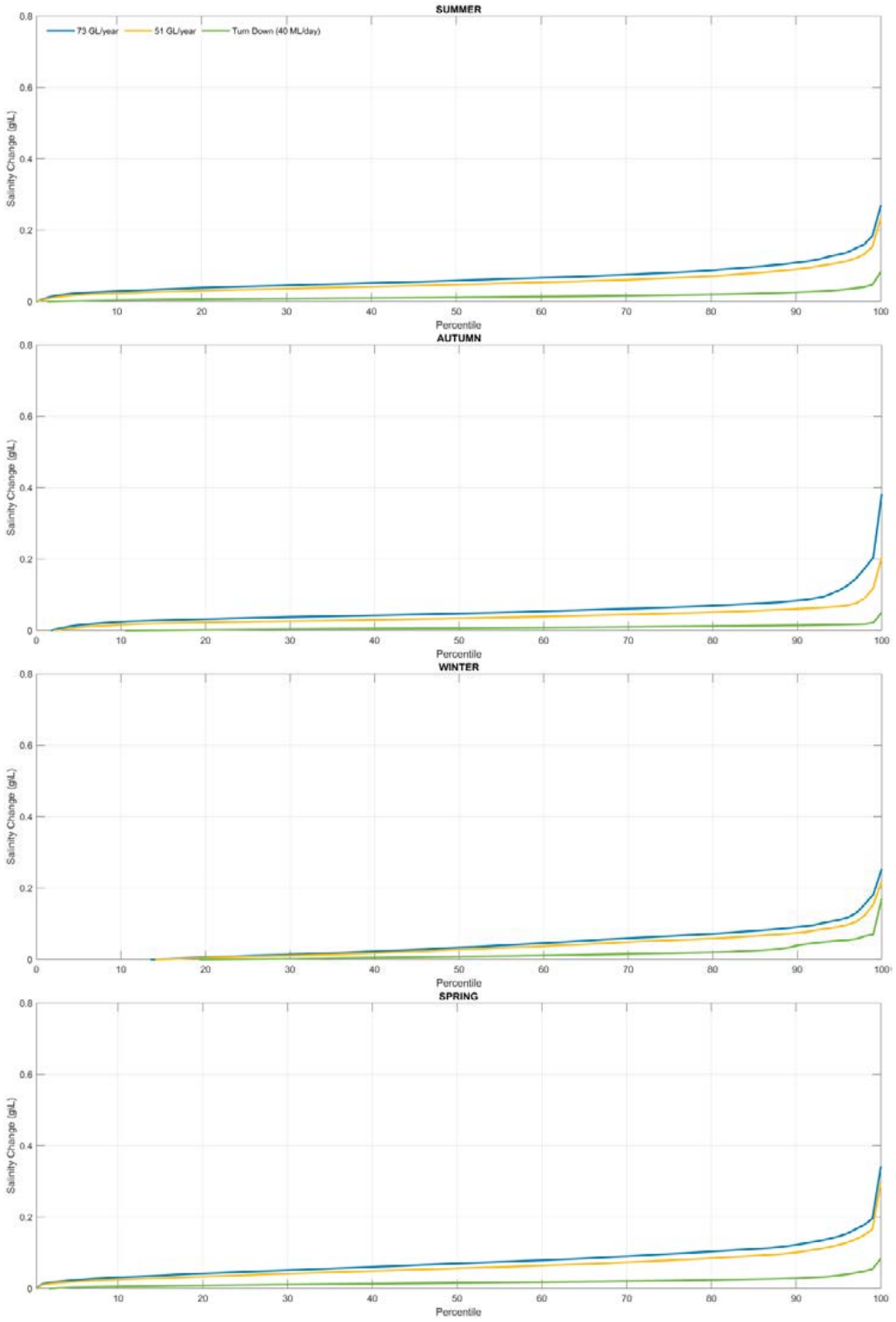


Figure 4-43 Salinity Differences to Baseline Cumulative Distribution at the Desalination Plant Intake Location

4.5.2 Entrainment

4.5.2.1 Assessment Objectives

The purpose of this modelling was to predict potential entrainment of prawn larvae (PL) and King George whiting (KGW) and snapper (SNP) juveniles in the seawater intake. For the purposes of this assessment, both larvae and juveniles are referred to as 'particles', given they were simulated under the assumption that they behaved like neutrally buoyant particles (see below). SNP juveniles were also simulated with the inclusion of specific behaviour (referred to as '*with behaviour*') as part of scenarios analysed (see below). This further inclusion was adopted for consistency with the far field modelling undertaken by others (Doubell and James, 2026), and to verify the sensitivity of entrainment to these behaviours. The modelling and post-processing was designed to provide calculations of the proportion of particles that arrive in the proximity of the intake at a given time and estimate their probability of entrainment into the intake. As such, this assessment did not consider the PL, KGW and SNP Gulf-wide population sizes and seeding areas, both of which were considered separately as part of the far field modelling activities (Doubell and James, 2026).

4.5.2.2 Simulation Periods

Modelling was undertaken for two distinct seasonal periods related to the species of interest, as follows:

- PL and SNP: from 1 November 2022 to 5 March 2023; and
- KGW: from 23 April 2023 to 30 August 2023;

While simulations were designed to coincide with spawning and subsequent dispersion of the *passive* particle scenarios, the absence of behaviour specifics in their modelling indicates the results can be used for assessments of other generalised planktonic species.

4.5.2.3 Model Scenarios and simulation periods

For each simulated period, only outfall and intake flow rates for the 73 GL/year scenario were considered in the modelling. This assumption can be considered conservative for other flow rates considered in this report (Turn Down and 51 GL/year capacity scenarios).

Modelling was undertaken for two distinct seasonal periods related to the species of interest, as shown in Table 4-6.

Table 4-6 Modelled Scenarios

Species	Behaviour type	Simulation period
Snapper juveniles	With behaviour	1 November 2022 to 5 March 2023
Snapper juveniles	Without behaviour (passive)	1 November 2022 to 5 March 2023
Prawn larvae	Without behaviour (passive)	1 November 2022 to 5 March 2023
King George Whiting juveniles	Without behaviour (passive)	23 April 2023 to 30 August 2023

4.5.2.4 Particle Release Conditions

PL and KGW scenarios had the same particle release area. This release area was defined as a 1-km sided square that did not intersect with land (Figure 4-44). The 1-km scale was defined for consistency with the farfield model scale. For both SNP scenarios, a defined 2-km diameter release area was adopted (Figure 4-44), in line with settlement areas considered in Doubell and James (2026).

For PL and KGW, 120 particles were released every 3 hours within the release areas, over approximately 18 weeks. For the SNP scenarios, 10 particles were released every 15 minutes (amounting to 120 every 3 hours)

over approximately 18 weeks. This number of particles was selected so that a representative number of particles were released whilst making sure feasible model computation and post-processing times were achievable.

4.5.2.5 Model Settings and Assumptions

The following settings and assumptions were applied within the neutrally-buoyant models (adopted for the three species):

- Particles in the model were assumed to be passive and neutrally buoyant;
- Particles were randomly released in the top 20 m of the water column;
- Particles did not become immobile if they interacted with the seabed; and
- Particles did not have an associated length or size; therefore, intake screening was not considered. Consequently, the assessment assumed that all particles could be potentially entrained upon entering the radius of influence of the intake.

For SNP, an additional scenario was executed, with inclusion of motility behaviour based on age of the SNP juveniles (Doubell and James, 2026). The parameters adopted corresponded to:

- Vertical position in the water column depending on daily cycles of sunrise and sunset, and
- Random swimming behaviour intensity affecting the larval dispersal in the horizontal direction.

The biophysical parameters considered for the different SNP juvenile stages are summarised in Table 4-7.

Table 4-7 Model Biophysical Parameters Adopted for Different SNP Juvenile Stages

Stage	Age (days)	Vertical Distribution	Horizontal Movement
Eggs hatching	0-3.5	0 to 3 m depth (randomly distributed)	Passive drift
Pre-flexion	3.5-11.5	Day: upper half of water column (normally distributed) Night: entire water column (randomly distributed)	Passive drift
Flexion	11.5-14.5	Day: lower half of water column linearly increasing in depth with age (normally distributed) Night: entire water column (randomly distributed)	1 cm/s random direction swimming
Post-flexion	14.5-19.5	Day: lower half of water column linearly increasing in depth with age (normally distributed) Night: entire water column (randomly distributed)	3 cm/s random direction swimming
Settlement	>19.5-29.5	Day: Lower 80% of water column (normally distributed) Night: entire water column (randomly distributed)	5 cm/s random direction swimming

Once particles settled (> 19.5-29.5 days), they were assumed to competently swim away from the intake and removed from entrainment calculations.

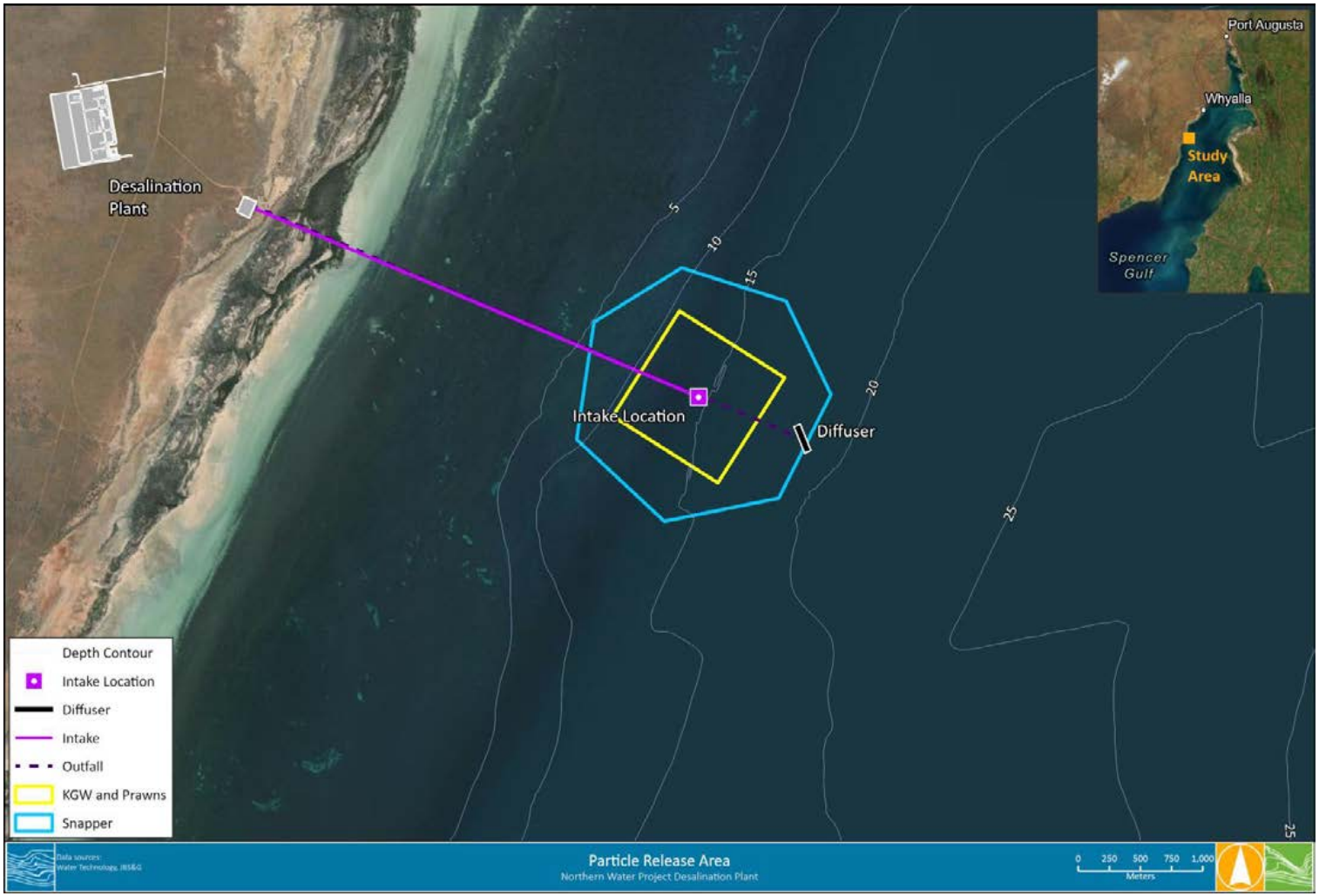


Figure 4-44 Particle Release Area for SNP (blue), and PL and KGW (yellow)

4.5.2.6 Entrainment Calculations

The seawater intake characteristics are described in Table 2-3. The flow rate for the 73 GL/year scenario was considered to estimate the zone of intake influence (Table 4-8). The approach to predicting entrainment followed the methodology by Largier et al. (2007). A radius of influence around the intake (r_0) was calculated as part of model post-processing, adopting the following formula:

$$r_0 = \sqrt{\frac{3\Delta t Q_0}{\pi b h}}$$

Where:

r_0 = distance at which $P(r) = 0$ (m)

Δt = model timestep

Q_0 = intake flow rate

h = opening of the intake

b = probability of entrainment at intake, assumed to be 1.

The entrainment radius of influence parameters are presented in Table 4-8.

Table 4-8 Radius of Influence formula constants

Entrainment Radius of Influence Parameters	73 GL/year Capacity Scenario
Model timestep (minutes)	10
Intake flow rate (m^3/s)	5.625
Riser Height (m)	3
Riser Radius (m)	4.35
Radius of influence (m)	**32.8**

The radius of influence was adopted in the calculations of a particle-entrainment probability based on distance from the intake *r:*

$$P(r) = \max\left[b\left(1 - \frac{r}{r_0}\right), 0\right]$$

Where:

$P(r)$ = Probability of entrainment

b = probability of entrainment at intake, assumed to be 1.

r = distance of particle from intake.

r_0 = radius of influence corresponding to the distance at which $P(r) = 0$ (m)

The radius of influence was 32.8 m (see Table 4-8 and Figure 4-45). In addition to the probabilities based on the radius of influence described above, assuming particles entrained across the entire water column would create an overly conservative entrainment estimate, given the intake is designed to avoid entrainment. As such, the assessment assumed particles would only be entrained if they were within the bottom half of the water column and within the horizontal radius of influence described above. In the absence of specific intake vertical influence, the vertical bounds for entrainment were based on a reasonable and yet conservative assumption in such a way to not unduly overestimate entrainment of particles located well above the intake upper edge.

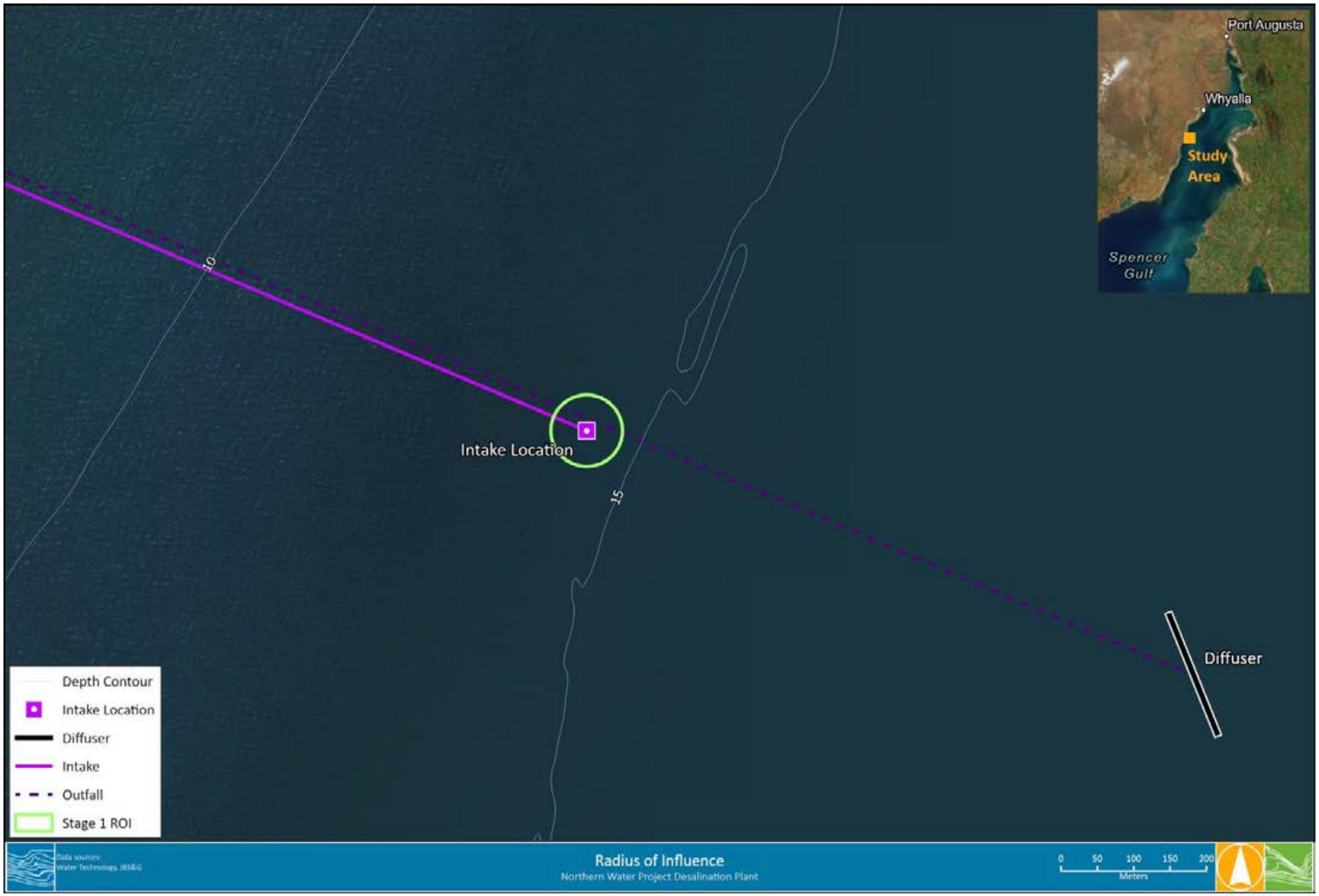


Figure 4-45 Radius of influence

4.5.2.7 Results Summary

A summary of the results is presented in Table 4-9. Key findings are summarised here and expanded in the following sections.

- SNP *with behaviour* had the lowest percentage of entrainment;
- SNP *passive* had the second lowest percentage of entrainment;
- PN *passive* had the second highest percentage of entrainment; and
- KGW *passive* had the highest percentage of entrainment.

Table 4-9 Scenario Entrainment Summary

Scenario	Entrainment (%)
SNP - *with behaviour*	0.3
SNP – *passive*	3.2
PN – *passive*	4.7
KGW - *passive*	6.3

Model results were analysed to obtain the proportion of entrained particles of the simulation period (Figure 4-46 to Figure 4-48). These proportions were calculated based on the time of the particle release. For example, a particle that was released on say 10 November 2022 14:00 and entrained by the intake on 11 November 2022 01:00 became part of the proportion of entrained particles that were released on 10 November 2022 14:00. These results were intended to be combined with wider farfield population dynamics models to assess entrainment relative to population size.

4.5.2.8 Spring-Neap Tidal Interactions

The influence of the tidal conditions on the fate of the particles was investigated. Figure 4-49 and Figure 4-50 respectively illustrate a particle modelling scenario excerpt during spring and neap tide conditions. In these maps, a single release of 120 SNP *passive* particles was tracked for 21 hours in 3-hour increments. Evident in the figures is the initial concentration of particles around the intake and subsequent dispersion with the tide motion. For the illustrated spring tide period, particles dispersed collectively to the north on the flood tide, then reversed toward the south during the ebb tides, passing the intake and dispersing further south of the intake (Figure 4-49).

In contrast, the neap tide period displayed a lesser degree of reversal to the south, whilst maintaining a higher degree of spreading and net migration toward the north (Figure 4-50). Furthermore, the increased dispersion observed during neap tides is likely driven by small scale eddies that are able to develop under lower flow conditions. These eddies arise from the interaction between reduced neap tidal currents and the site's complex bathymetry, resulting in greater outward spreading as eddies shed flow in multiple directions. In contrast, tidal flows during spring tides are more linear in direction and higher in velocity. Under these conditions, the flow is less influenced by local bathymetry, and eddy formation is either suppressed or significantly reduced compared to during neap tides. As a result, there is less outward spreading of particles during spring tides, with transport occurring predominantly in the longitudinal direction aligned with the tidal flow to the north and to the south of the intake.

4.5.2.9 Snapper Juvenile Entrainment

The number of entrained particles for each release timestep varied between 0 and 3 and between 0 and 4 for the *with behaviour* and the *passive* scenarios, respectively (Figure 4-46). The February 2023 period displayed the largest entrainment numbers for the *passive* scenario. The peaks in simulated entrainment were likely due

to spring tide conditions. During spring tides, advection due to the tidal movement caused particles in the water to oscillate back and forth and disperse along the area of influence of the intake, as explained above, increasing the likelihood of their paths intersecting with the intake's radius of influence. This is displayed in Figure 4-49, as an example, showing that this heightened movement enhanced the probability that particles were drawn into the intake structure. In contrast, during neap and dodge tides, the tidal motion was weaker, leading to more opportunity for particle advection driven by secondary flows (e.g., eddies) operating under dynamic directions and speeds. With an increased degree of variation of current directions, particles were less likely to pass back and forward across the intake. This effect resulted in a lower probability of the particles to enter the intake radius of influence (Figure 4-50).

Over the full simulation, the total percentage of entrainment for the *passive* scenario was ~3%, while *with behaviour* scenario was 0.26%. The difference is likely due to the *with behaviour* actively moving up through the water column and away from the lower half of the water column and further from the intake. Of noteworthy interest is on only one occasion the *with behaviour* scenario experienced an entrainment number that exceeded the *passive* scenario. For the remainder of the simulated period, the entrainment obtained in the *with behaviour* scenario was predominantly equal to or less than the *passive* scenario.

The difference in entrainment percentage between the *with behaviour* and *passive* can be attributed to the movement of the particles in the water column for the *with behaviour* scenario. In the earlier days (up to 11.5 days, see Table 4-7) the particles preference the upper half of the water column. Therefore, the particles inherently had a higher chance to move away from the intake area of influence (situated in the lower half of water column) as the simulation progressed. It is not until later in the larvae development (>11.5 days) that the particles preference the lower half of the water column. However, by this age the particles would mostly have been advected away from the general seeding area, with a lower likelihood of being entrainment into the intake.

4.5.2.10 Prawn Larvae Entrainment

The PN entrainment over time is presented in Figure 4-47 with entrainment values ranging from 0 to 22 particles per timestep. The overall percentage entrainment for PN is greater than SNP passive. This is attributed to the seeding area of the PN being smaller in area than the seeding area for the SNP passive. The larger seeding area in the SNP seeding essentially spread the particles out, leading to less particle concentration around the intake and consequently leading to less entrainment. Total number of entrained particles were ~50% higher for PN than SNP. Numbers were however low (< 5%), noting the seeding area adopted in the model represents a considerably small area compared to the Gulf-wide population seeding area.

4.5.2.11 King George Whiting Juvenile Entrainment

The KGW entrainment results are shown in Figure 4-48. Entrainment proportions for KGW were generally higher compared to the PL entrainment, with entrainment values ranging from 0 to 25 particles. The overall percentage entrainment for KGW was 6.3% compared to 4.7% for PN.

The reason for the higher KGW entrainment proportion frequency could be, at least partially, explained by the hydrodynamic conditions. The subset of current roses from the KGW winter period (April and August 2023), shown in Figure 4-51 were different from the current roses for the summer period (November 2022 to March 2023) periods, shown in Figure 4-52. The flooding and ebbing currents were more symmetrical over the KGW autumn period (i.e., more even northeast-southwest current direction axis and more even magnitude). The associated residual flows to the northeast consequentially were lower. This is related to broader seasonal trends in Spencer Gulf. This pattern resulted in a larger proportion of entrained particles as they were advected back and forth without veering away from the intake towards the northeast. Similar to the descriptions of results for SNP. KGW entrainment also increased during spring tides in comparison to neap tides.

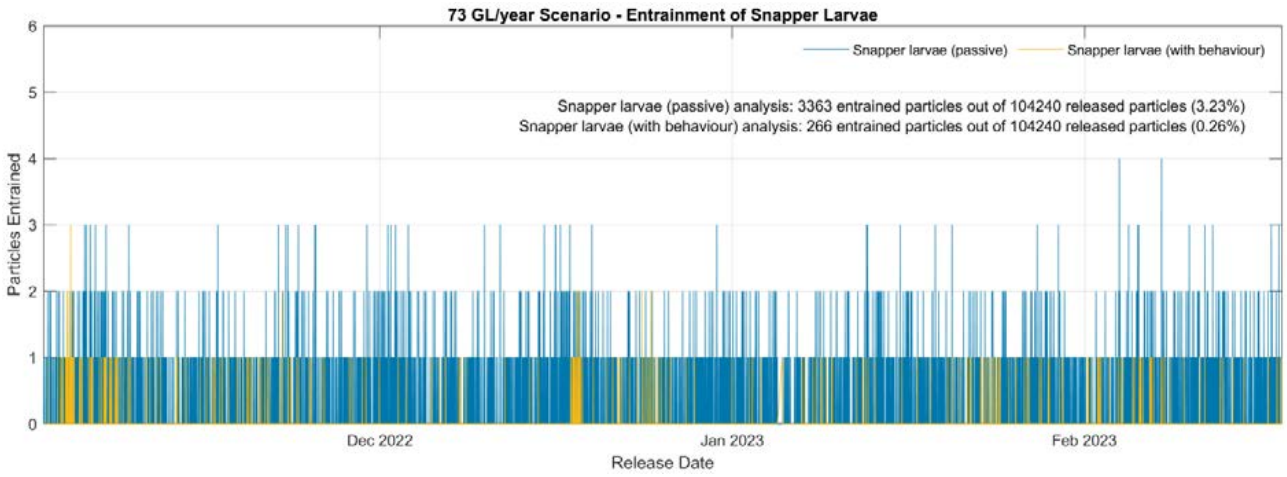


Figure 4-46 Snapper Juvenile Entrainment 2022/2023

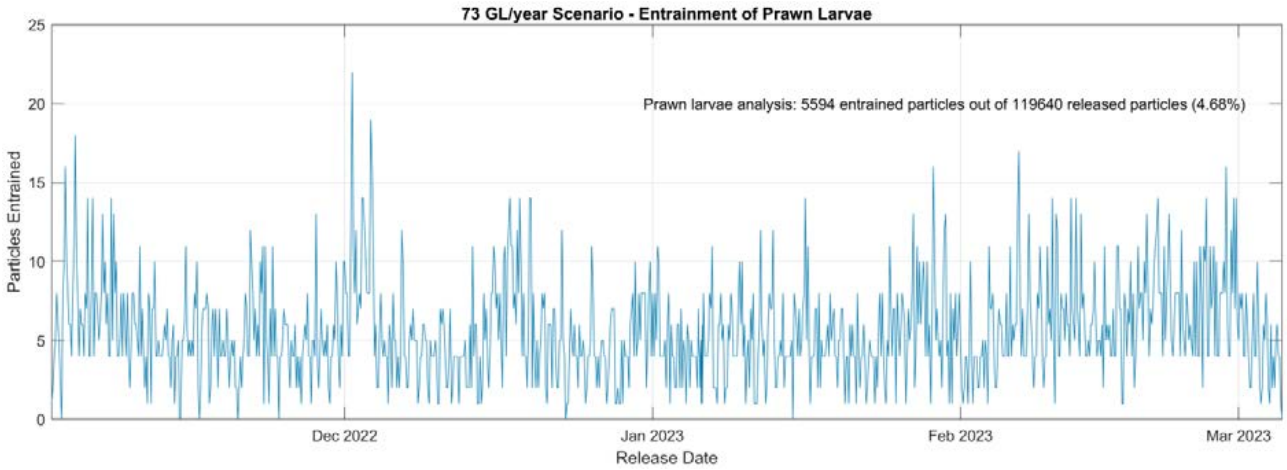


Figure 4-47 Prawn Larvae Entrainment 2022/2023

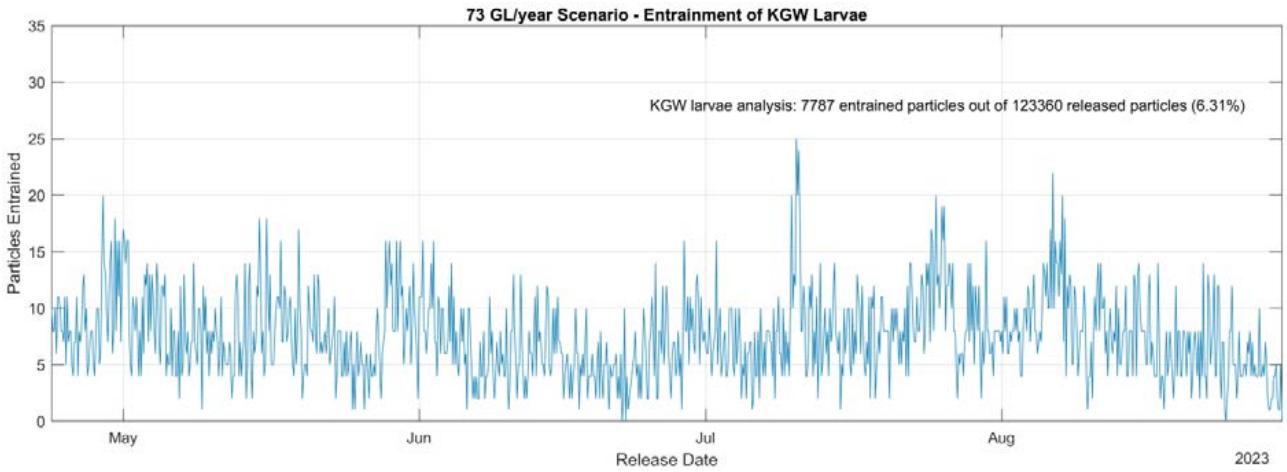


Figure 4-48 King George Whiting Juvenile Entrainment 2023



Figure 4-49 Particle Movement Every 3 hours during a spring tide

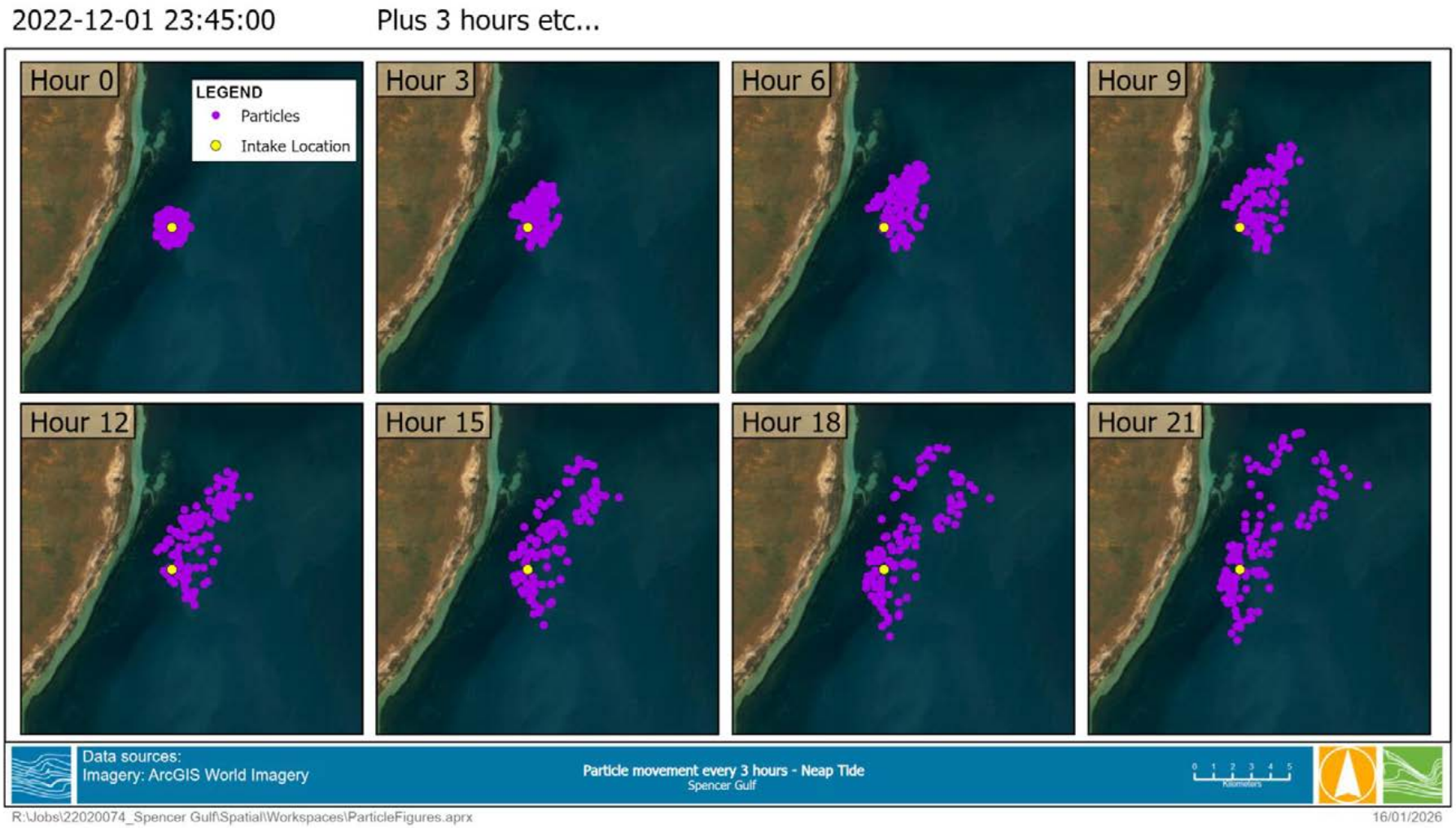


Figure 4-50 Particle Movement Every 3 hours During a Neap Tide

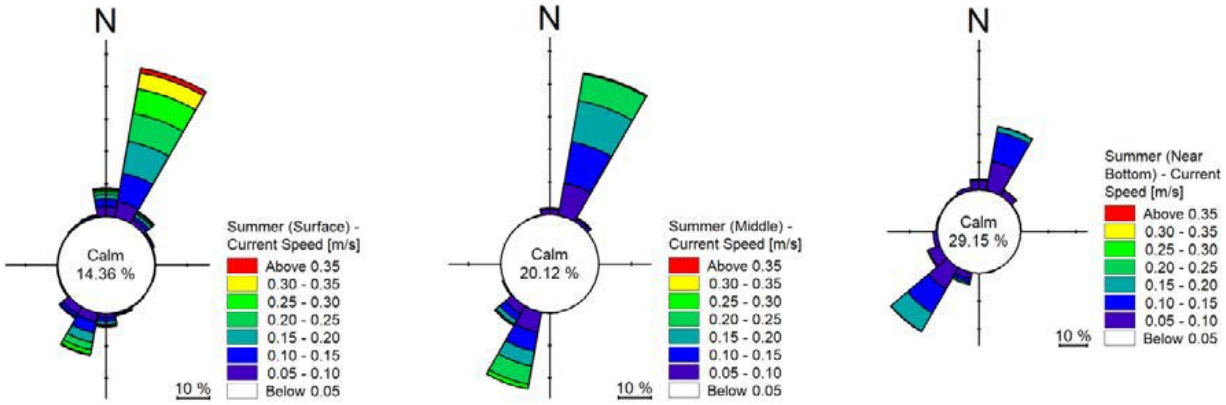


Figure 4-51 Current Roses at the Surface, Mid-Column and Near-Bottom at the Intake Location during the Summer Scenario (Nov 2022 to Mar 2023)

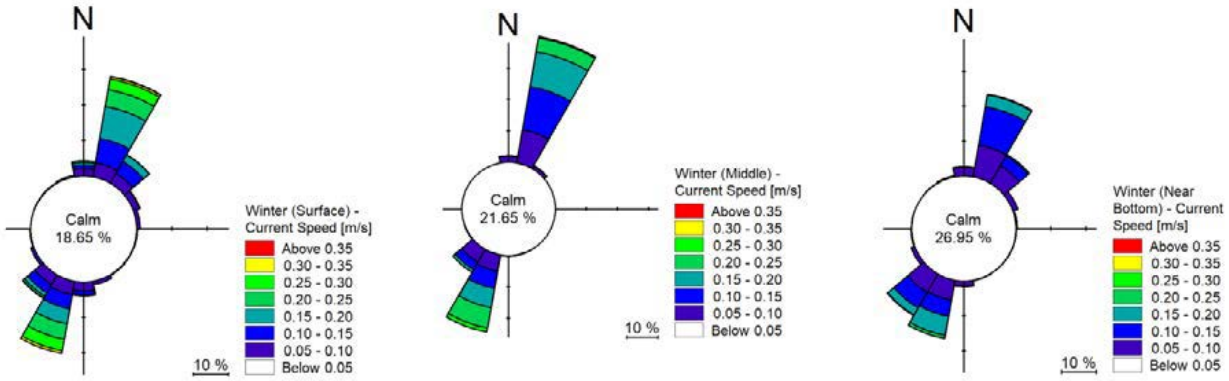


Figure 4-52 Current Roses at the Surface, Mid-Column and Near-Bottom at the Intake Location during the Winter Scenario (Apr 2023 to Aug 2023)

5 SUMMARY

This report presents three-dimensional hydrodynamic and transport simulations to inform and support ecological and water quality impact assessments as part of the broader NW EIS. The developed model simulated the midfield dispersion of the return water in the environment so that the following could be assessed:

- Extents of the return water impact footprint near the outfall as defined by the dilution environmental criteria;
- Extents of the return water impact footprint near the outfall as defined by the salinity change environmental criteria;
- Associated TSS change near the outfall;
- Any potential recirculation between return water and the seawater intake; and
- Potential for entrainment of snapper juveniles (passive and with behaviour), prawn larvae and King George Whiting juveniles in the seawater intake.

The model was calibrated and validated against site-specific data demonstrating its ability to represent local hydrodynamic conditions in terms of water levels, currents, temperature and salinity.

The validated model was subsequently set up to include the effects of the proposed intake and outfall for scenarios based on existing design conditions. These included the following scenarios:

- Hypothetical high-capacity scenario of 73 GL/year;
- Design Production of 51 GL/year; and
- Turn Down (production of 40 ML/day).

Scenarios varied not only in terms of flow rates but also configuration of the diffuser, with 73 GL/year operating 12 outfall ports, 51 GL/year operating 10 outfall ports and Turn-Down operating 6 outfall ports. Simulations were undertaken for different seasonal periods from November 2022 to December 2023.

Simulated plumes were shown to vary significantly with oceanographic conditions, displaying periods of intense mixing over spring tides, and some degree of (short-lived) accumulation during neap and dodge tides.

Simulations also quantified the extent of the return water discharge footprint in the receiving environment, with emphasis on effects near the seabed. The extents included:

- Representation of mixing areas with dilutions below 95% and 99% SPTVs defined via ecotoxicological tests; and
- Representation of mixing areas with salinity changes above specified thresholds (1.1 g/L, 1.3 g/L, 1.5 g/L and 1.8 g/L).

Results were influenced by discharge flow rates and to a lesser extent by season. For 51 GL/year, areas based on 10th percentile dilutions below the 95% SPTV were between 10 and 13 ha, depending on season. For 73 GL/year, the areas varied between 16 and 21 ha, and for Turn Down scenarios, areas varied between 0.2 and 1.4 ha. Areas with 10th percentile dilutions below the 99% SPTV were more extensive, varying between 60 and 79 ha (51 GL/year), 90 and 115 ha (73 GL/year), and 5.7 and 12 ha (Turn Down). Average distances from the edge of the 95% SPTV areas to the outfall (10th percentile dilution) varied between ~190 and ~230 m (51 GL/year), ~250 and 290 m (73 GL/year), and ~30 and ~70 m (Turn Down).

Areas based on salinity changes were significantly smaller, with areas with 24-hour maximum salinity averages above 1.3 g/L varying between 1.5 and 3.0 ha (51 GL/year), 1.8 and 3.6 ha (73 GL/year), and 0.0 and 0.044 ha (Turn Down). It is important to note that these areas were based on a composite of model result across the simulation periods and were, in general, significantly larger than the highly variable instantaneous conditions. Also, the adoption of a 1.3 g/L contour was derived from ADP licence criteria. As such this salinity change

represents conservative conditions for the Mullaquana Station site, as it experiences a comparatively higher salinity range. Average distances from the edge of these areas to the outfall varied between ~65 and ~95 m (51 GL/year), ~70 and ~110 m (73 GL/year), and 0 and ~35 m (Turn Down).

At identified sensitive receptors, a couple of locations in close proximity to the outfall showed 10th percentile dilution results lower than the 99% and 95% SPTV criterion, while all other locations had dilutions above the criteria. With the exception of sensitive receptor located nearest to the outfall, all simulations presented maximum 24-hour average salinity changes at the sensitive receptor locations lower than 0.2 g/L. Maximum instantaneous salinity changes were always below 0.4 g/L.

The return water discharge was shown to influence stratification up to 500m from the outfall for brief periods of time. A desktop analysis indicated that the duration and extent of the stratified layer was deemed insufficient to impact near bed dissolved oxygen concentrations.

Assessment of salinity changes at the intake location indicated a small increase in the feed water salinity. The increase under the 73 GL/year and 51 GL/year scenarios (median and maximum change < 0.1 g/L and <0.25 g/L, respectively) could be mitigated with minor adjustments to the intake design (i.e., moving it inshore and at a higher elevation in the water column).

Snapper juvenile, Prawn larvae and King George whiting juvenile entrainment was assessed with a particle tracking model and determination of a radius of influence around the desalination plant intake. For the assessment, regular particle releases in a 1-km-side square (for prawns and KGW) and a 2-km diameter heptagon (for snapper) around the intake were used to obtain the proportion of released particles likely to be entrained into the seawater intake structures. Model determined that entrainment proportions were lowest for snapper when considering their motility behaviour (0.3 % over the summer season assessment period) and highest for King George Whiting (6.3 % over the winter assessment period). Entrainment proportions for prawns were 4.7%. Lower entrainment proportions were observed for snapper with behaviour than for snapper without behaviour (i.e. passive). The difference is largely attributed to the tendency of the juvenile snapper to preference the upper water column in the earlier days of development (<11.5 days), that keeps the juvenile snapper away from the intake which is situated in the lower half of the water column.

It is noted that the particle seeding, and associated proportion of entrainment particles, did not correspond to the broader Spencer Gulf spawning areas and were in fact, assumed over a significantly smaller area with scales relatively similar to the farfield model scales. Hence, the results presented have to be assessed in conjunction with the assessment undertaken as part of the farfield modelling simulations (Doubell and James, 2026).

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WATER TECHNOLOGY
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APPENDIX A
IN-SITU DATA



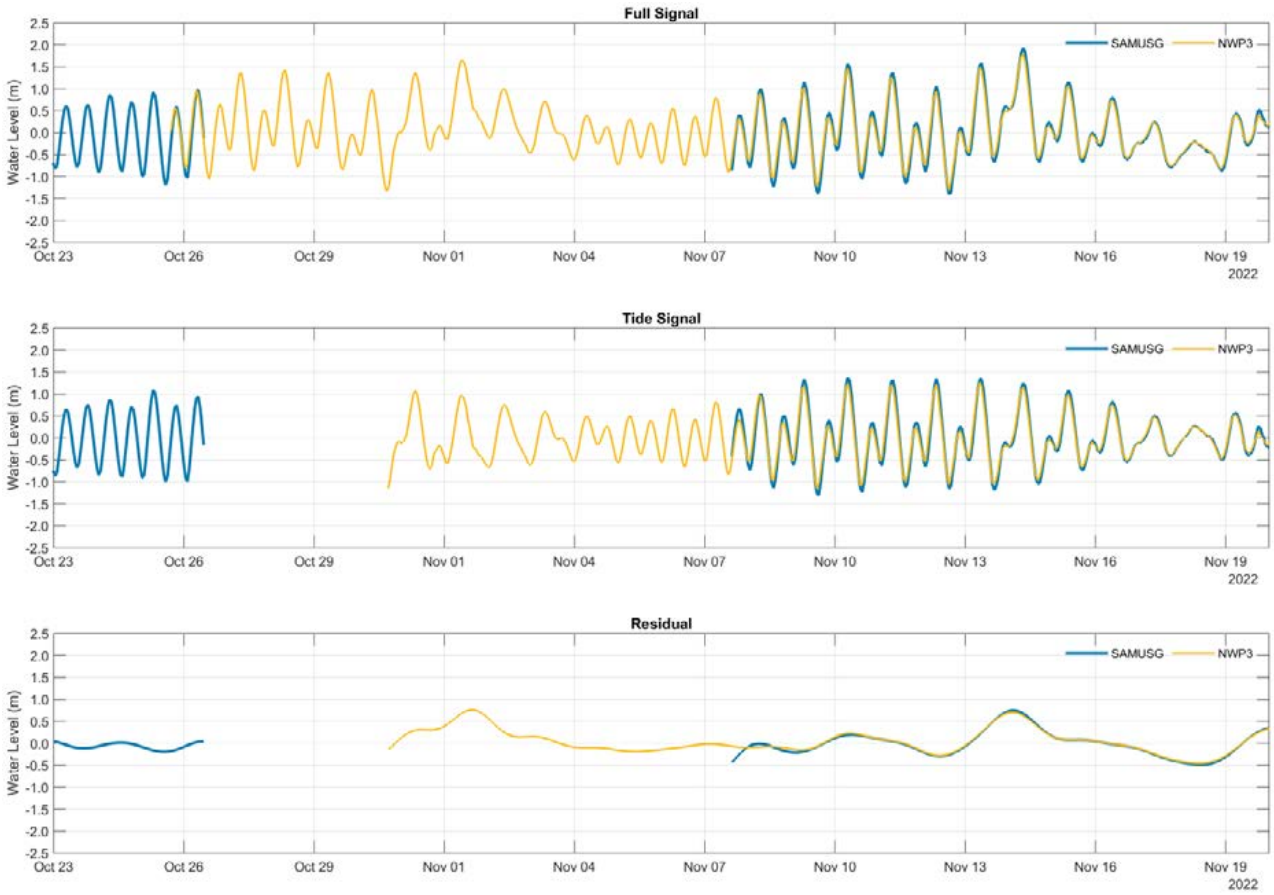


Figure A-1 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 23/10 to 20/11/2022

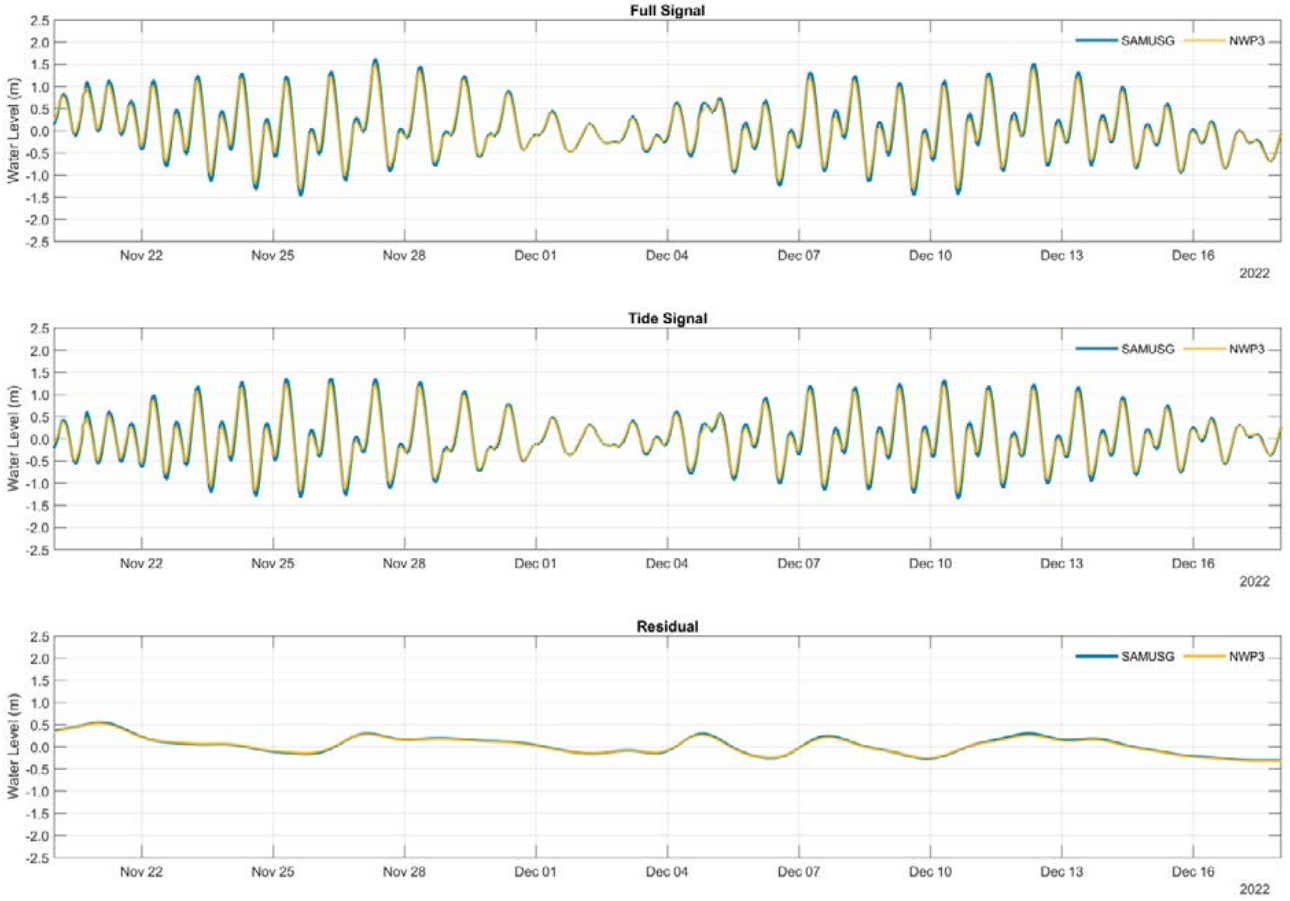


Figure A-2 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 20/11 to 18/12/2022

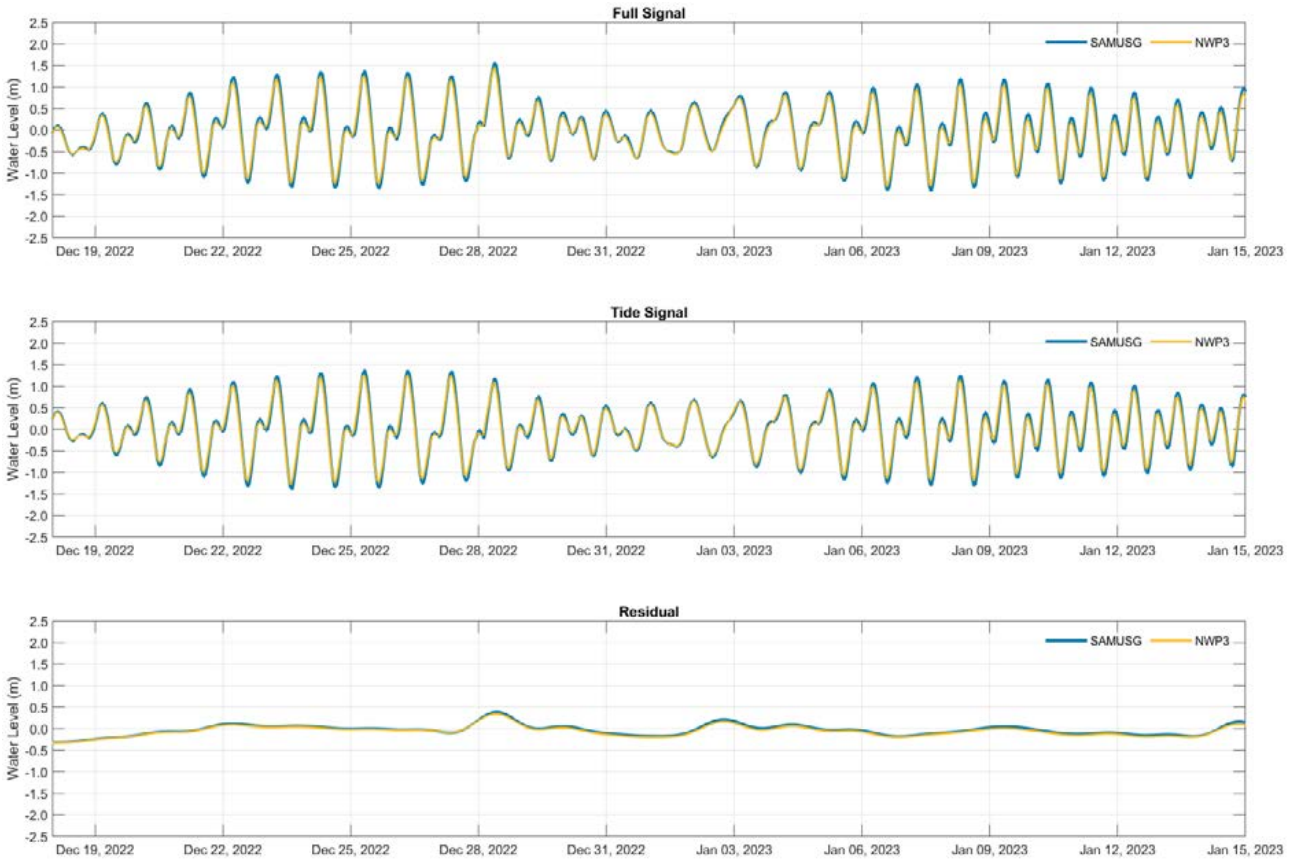


Figure A-3 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations - 18/12/2022 to 15/01/2023

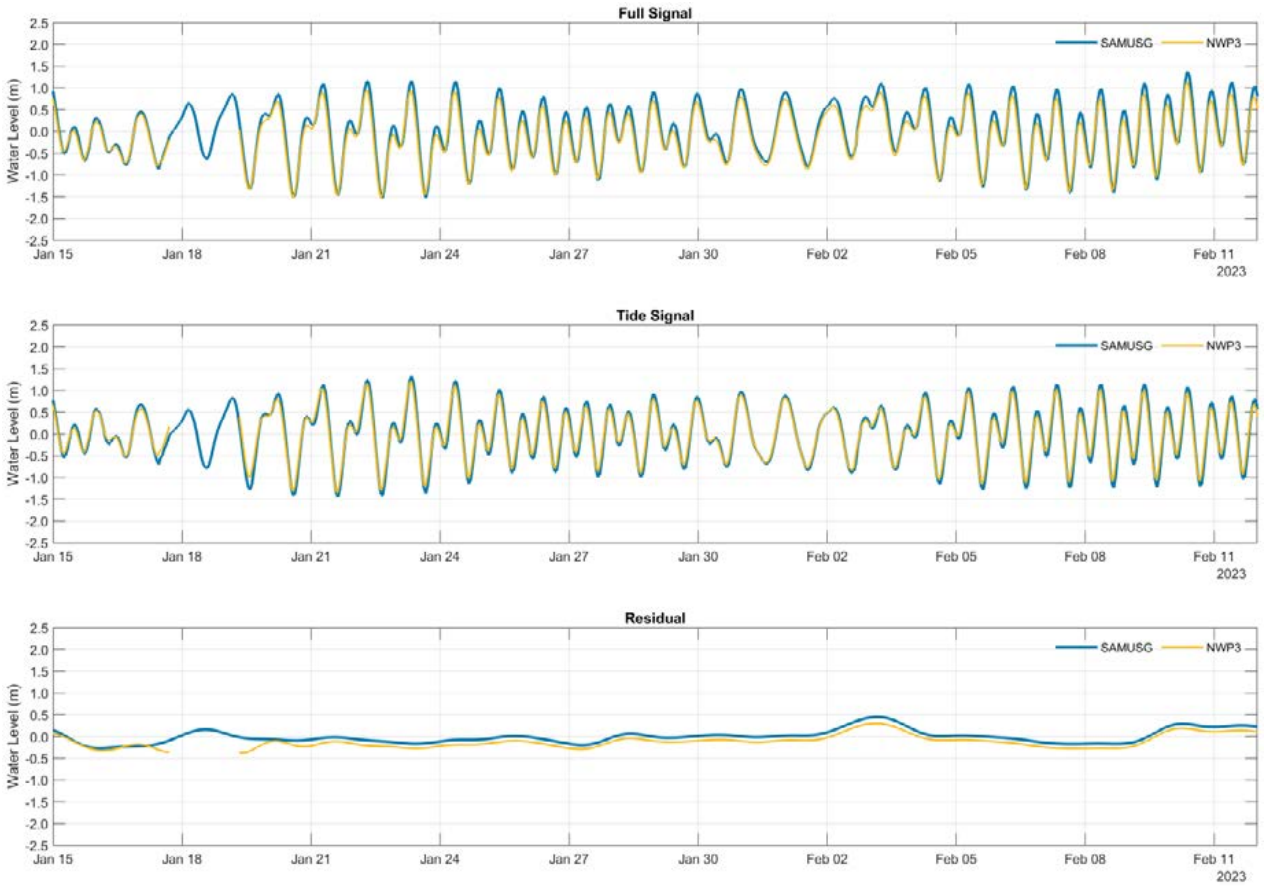


Figure A-4 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 15/01/ to 12/02/2023

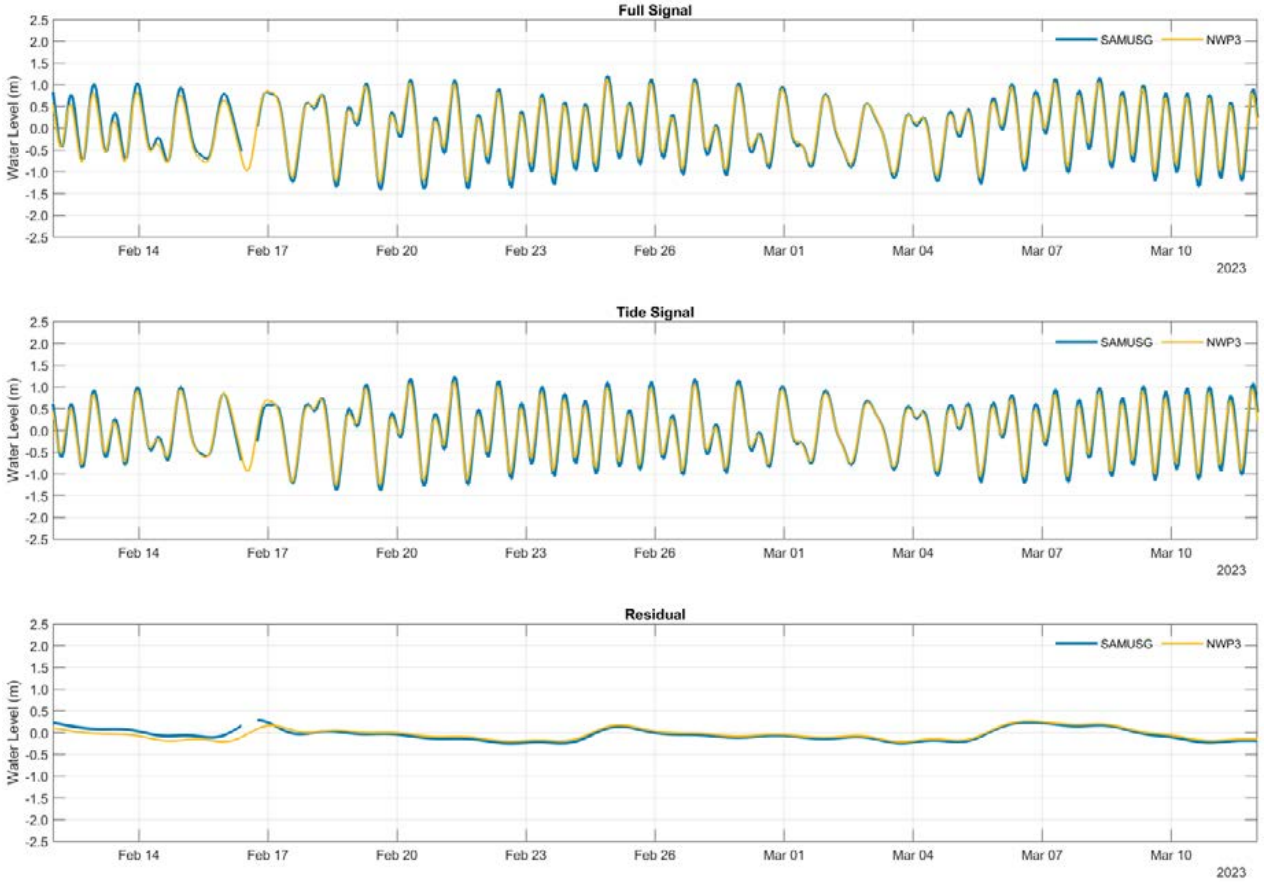


Figure A-5 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 12/02 to 12/03/2023

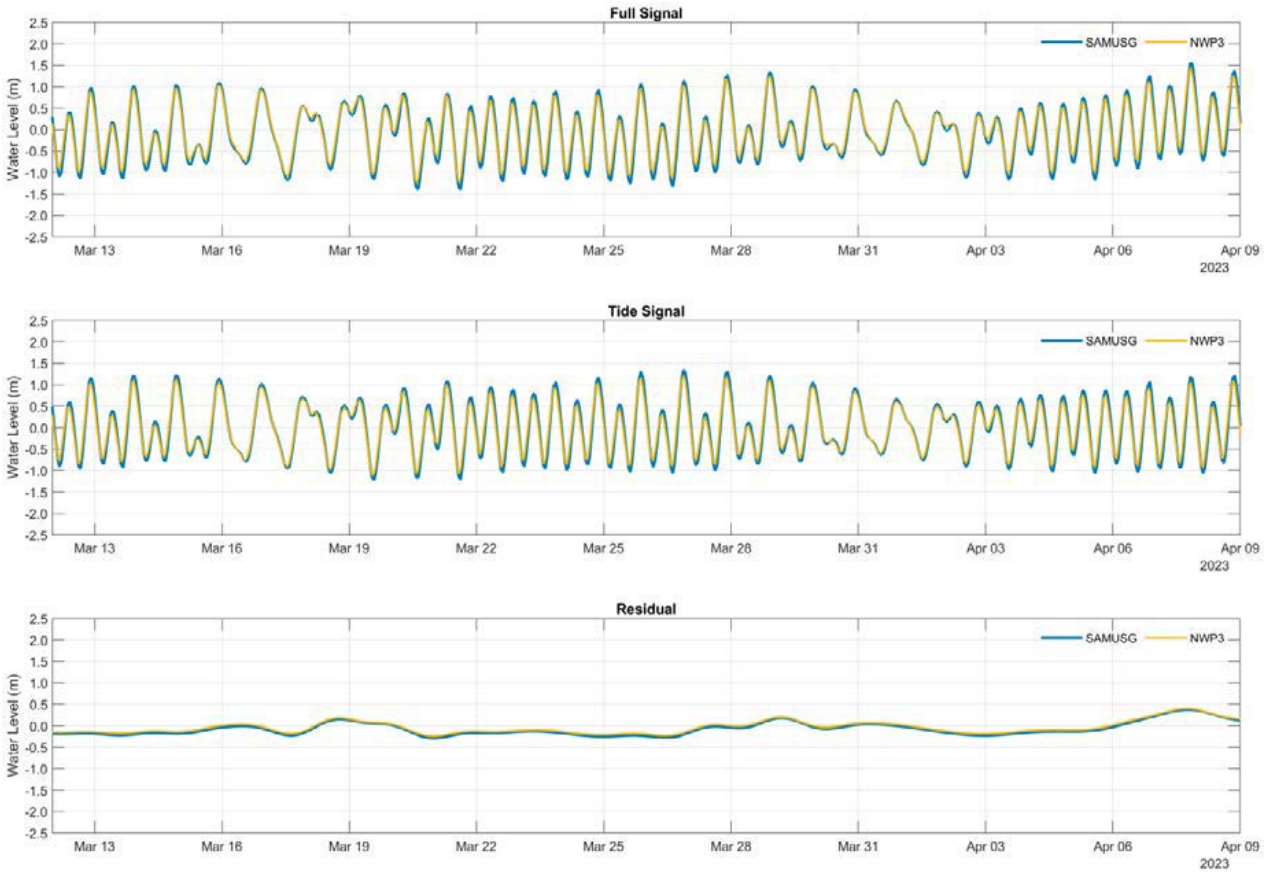


Figure A-6 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 12/03 to 09/04/2023

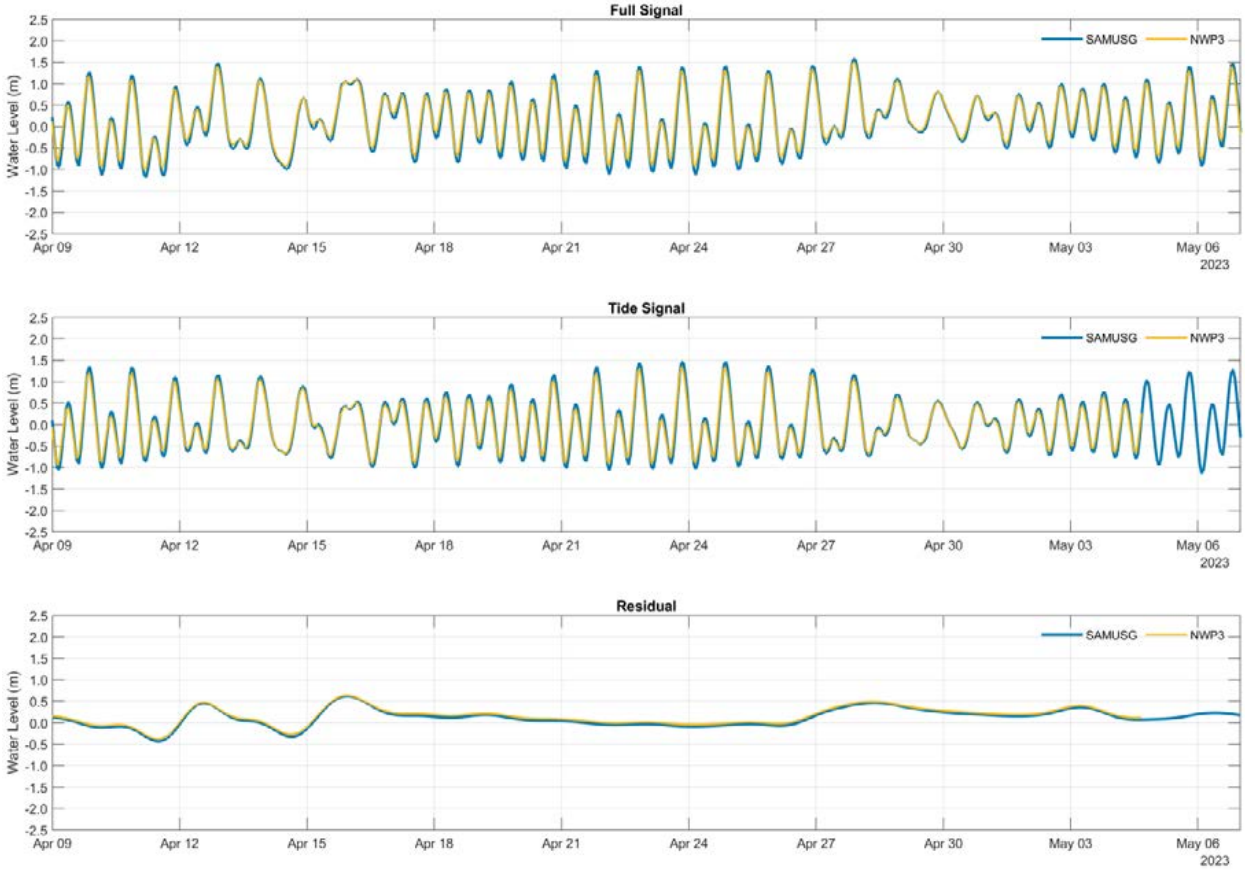


Figure A-7 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations - 09/04 to 07/05/2023

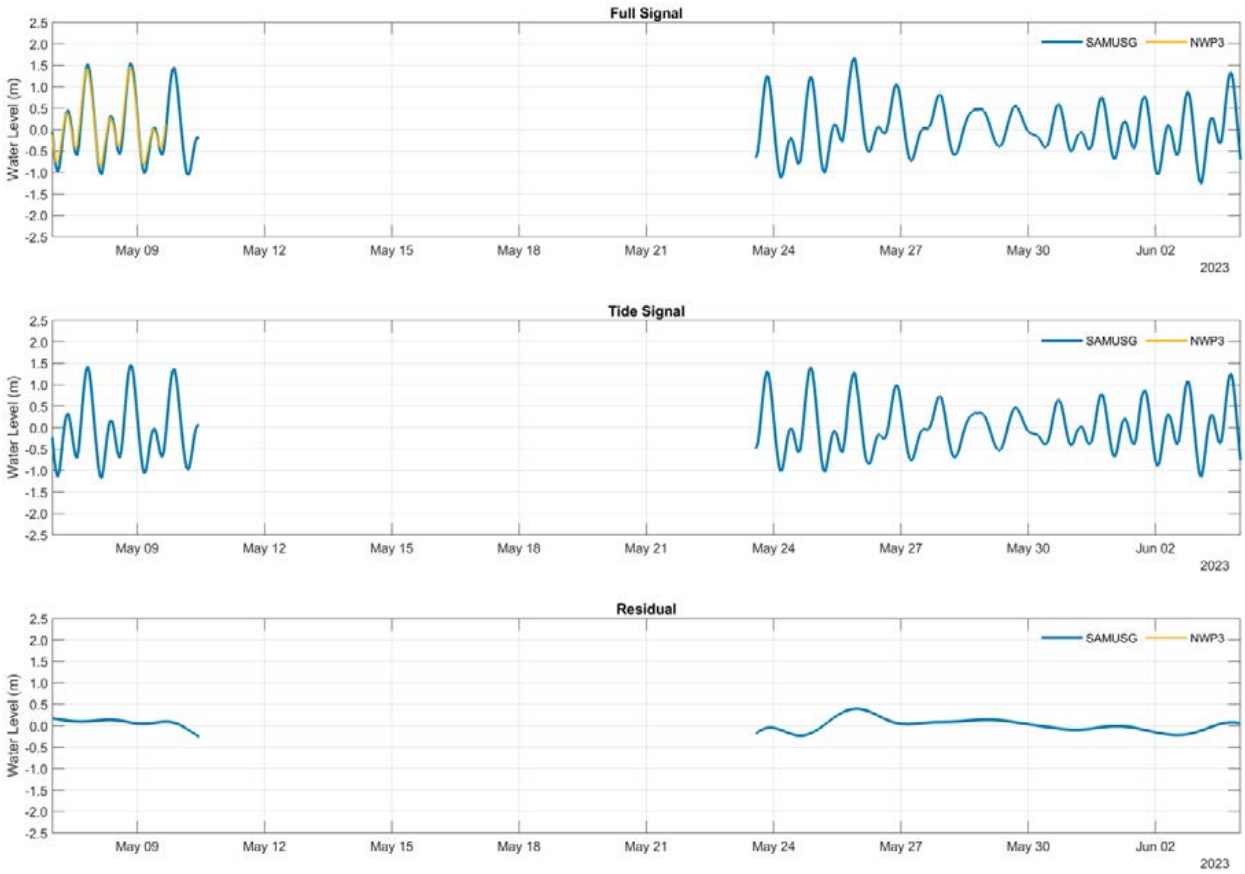


Figure A-8 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 07/05 to 03/06/2023

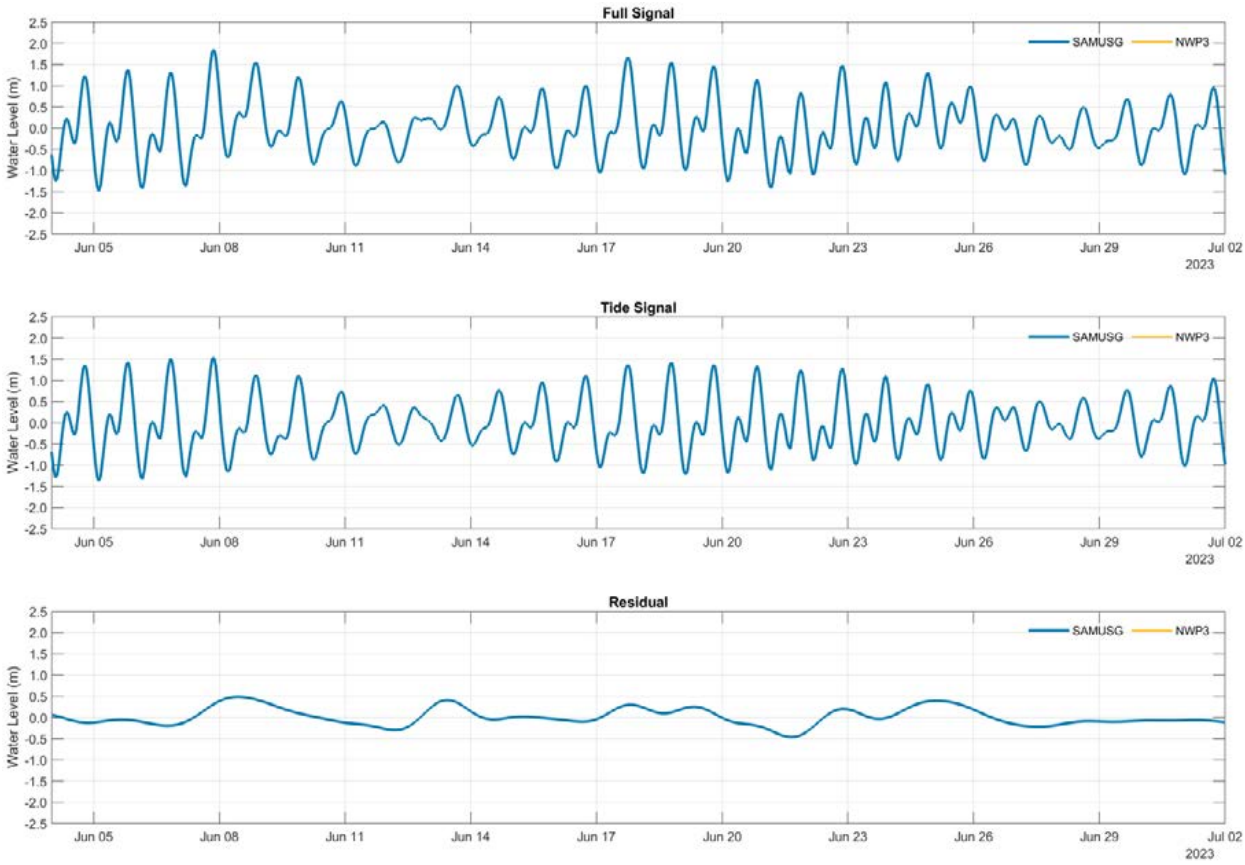


Figure A-9 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 03/06 to 02/07/2023

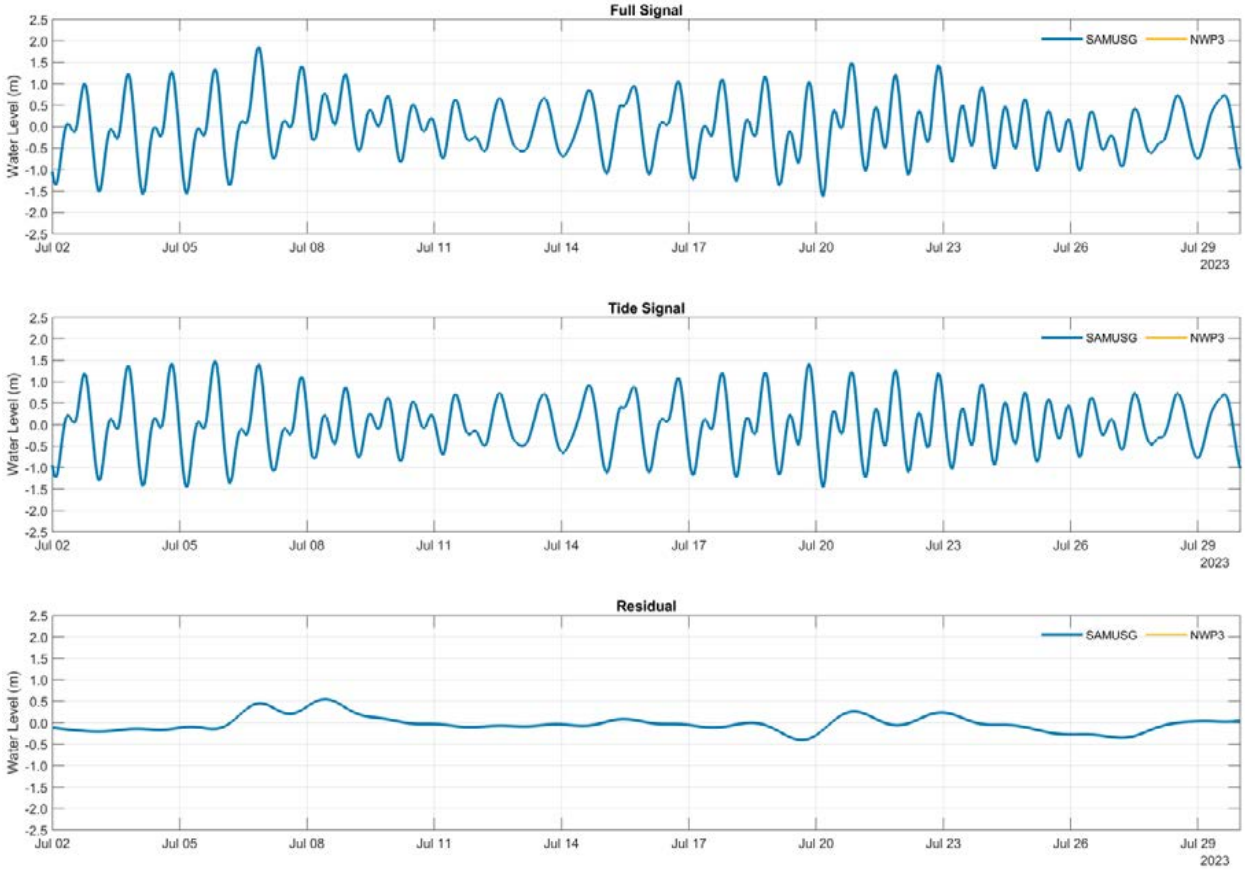


Figure A-10 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 02/07 to 30/07/2023

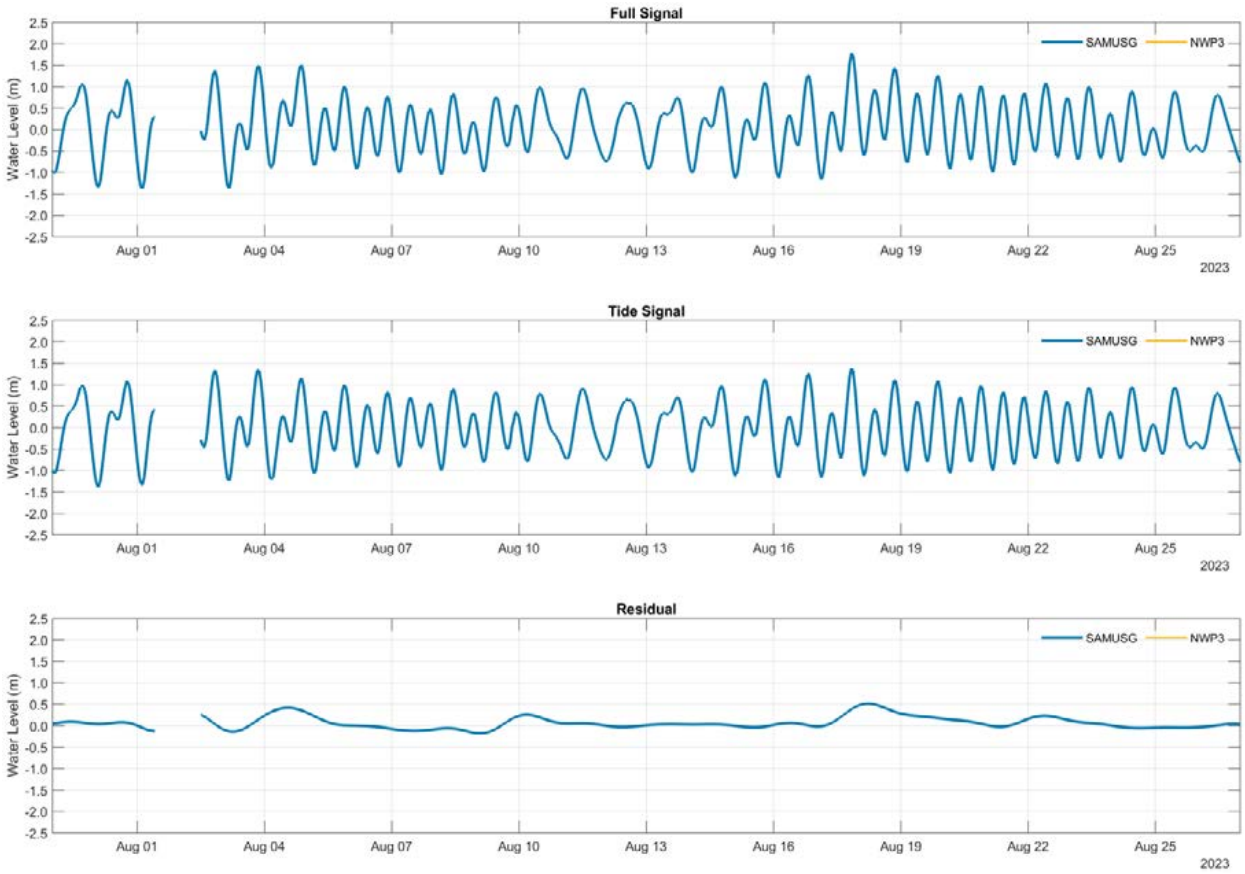


Figure A-11 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 30/07 to 27/08/2023

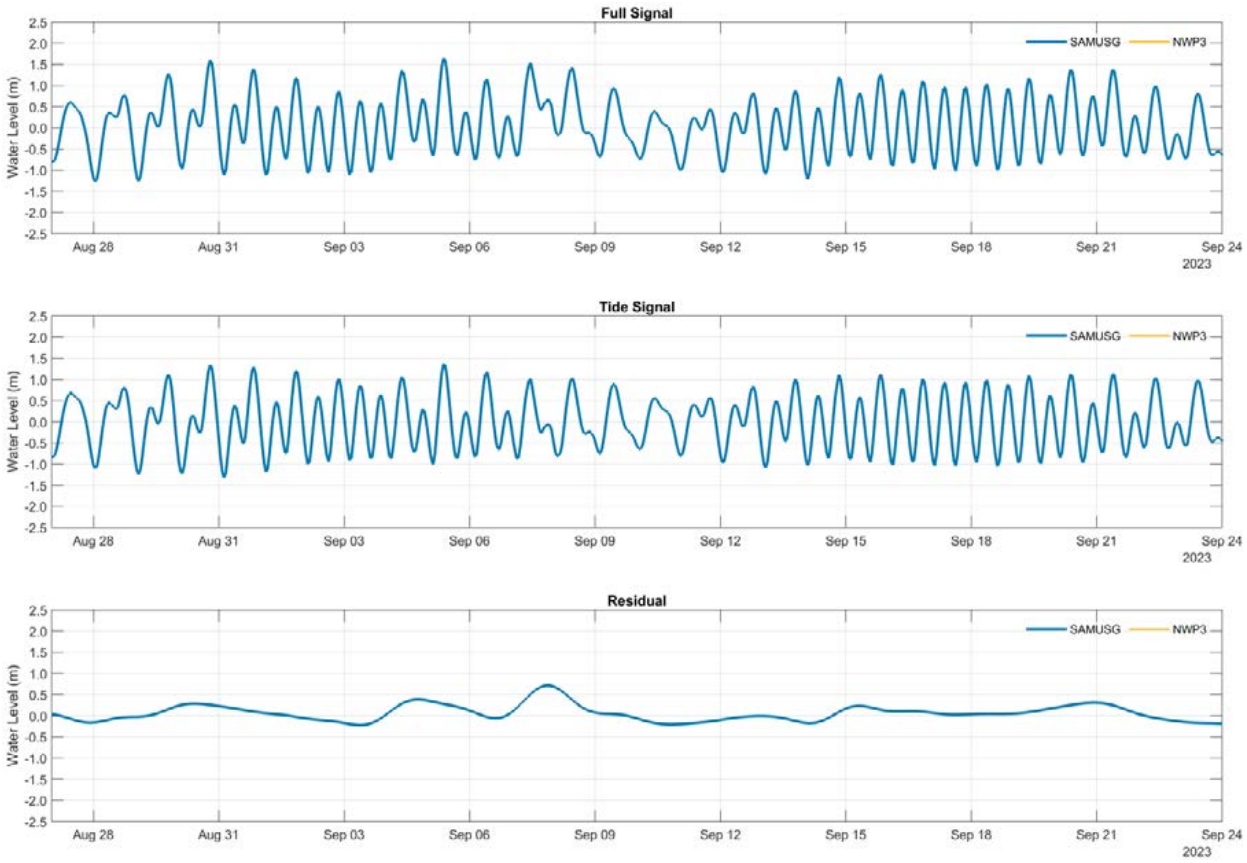


Figure A-12 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 27/08 to 24/09/2023

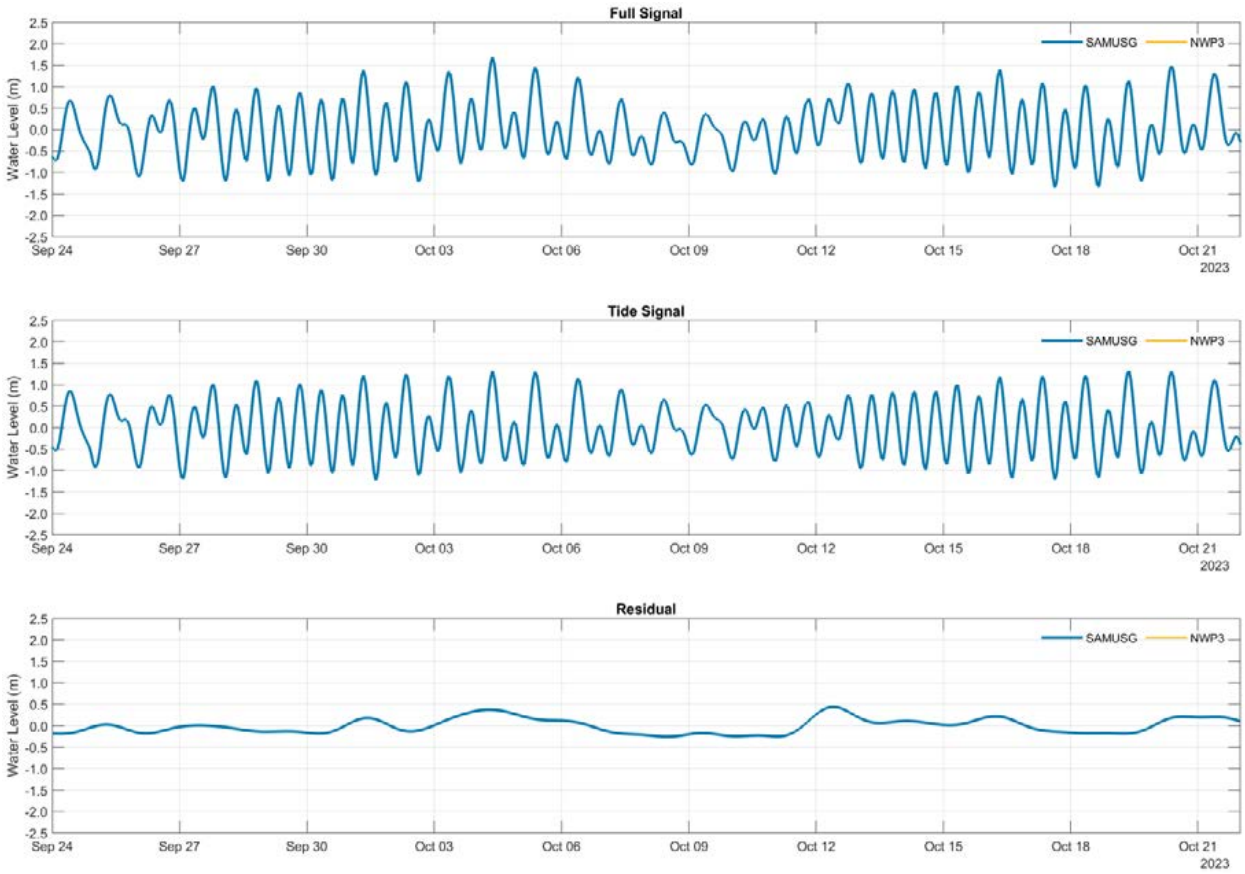


Figure A-13 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 24/09 to 22/10/2023

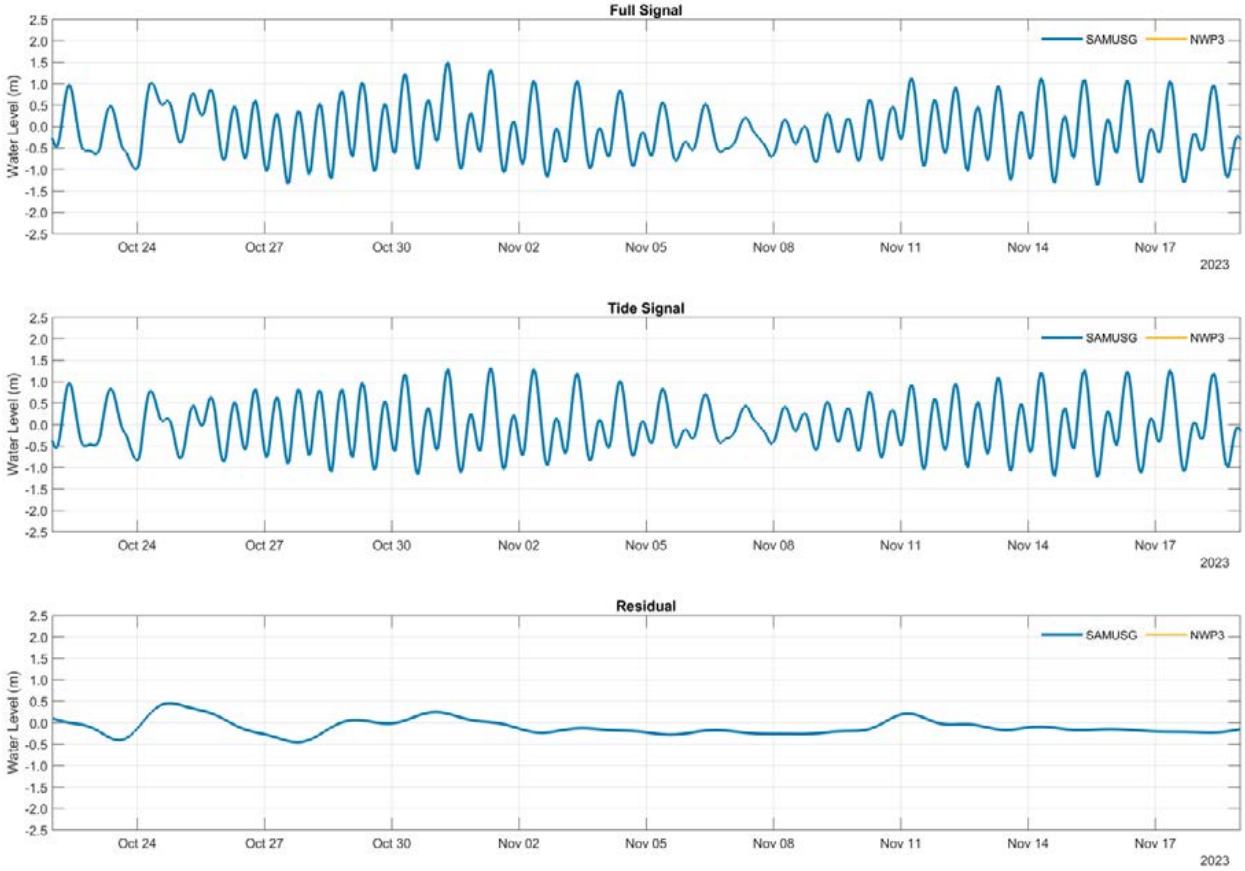


Figure A-14 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations - 22/10 to 19/11/2023

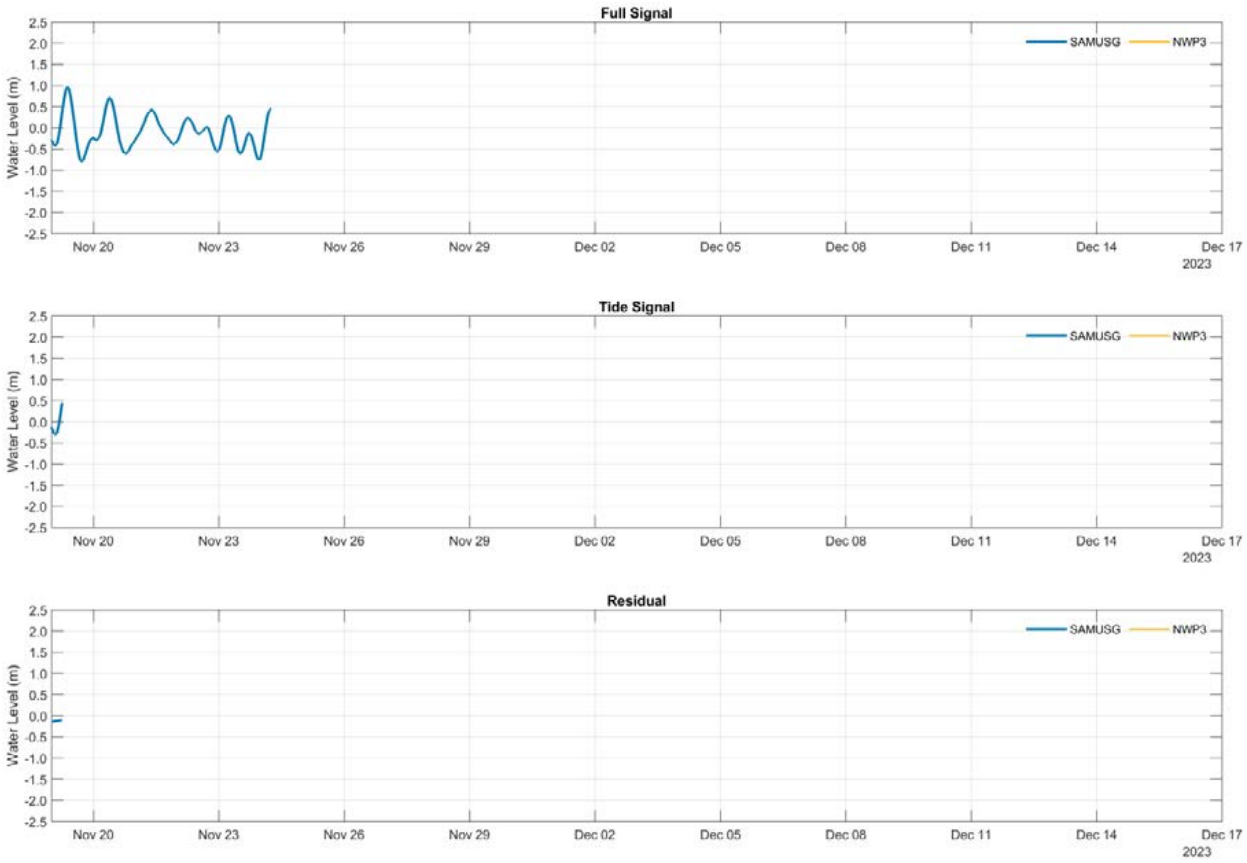


Figure A-15 Water Level Measured by the ADCPs at NWP3 and SAMUSG stations – 19/11 to 17/12/2023

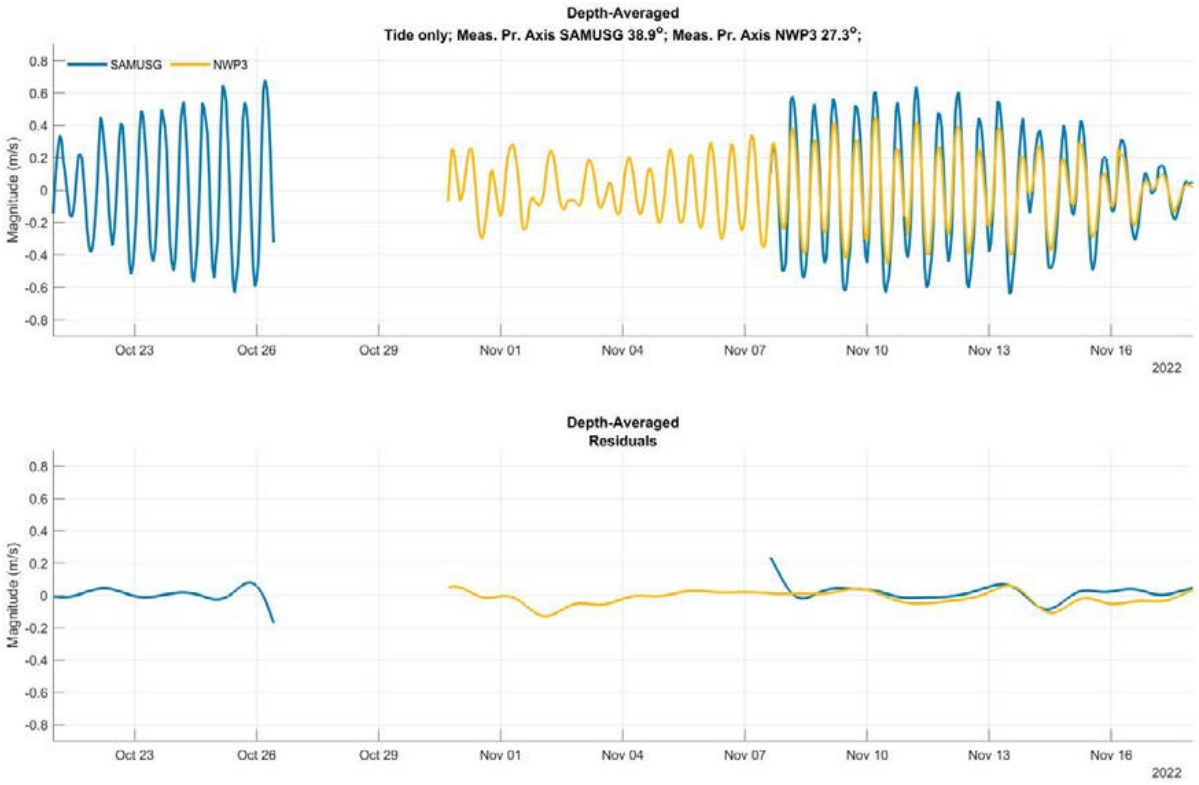


Figure A-16 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG– 21/10 to 18/11/2022

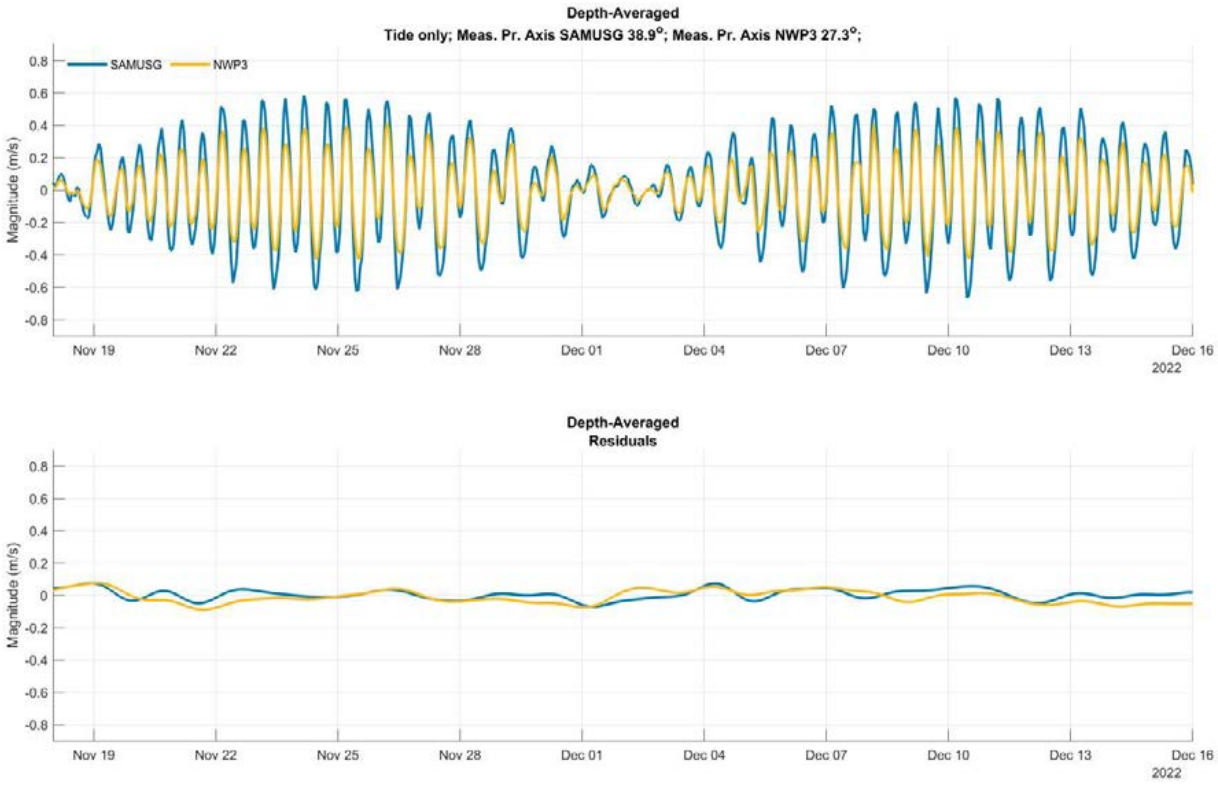


Figure A-17 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 18/11 to 16/12/2022

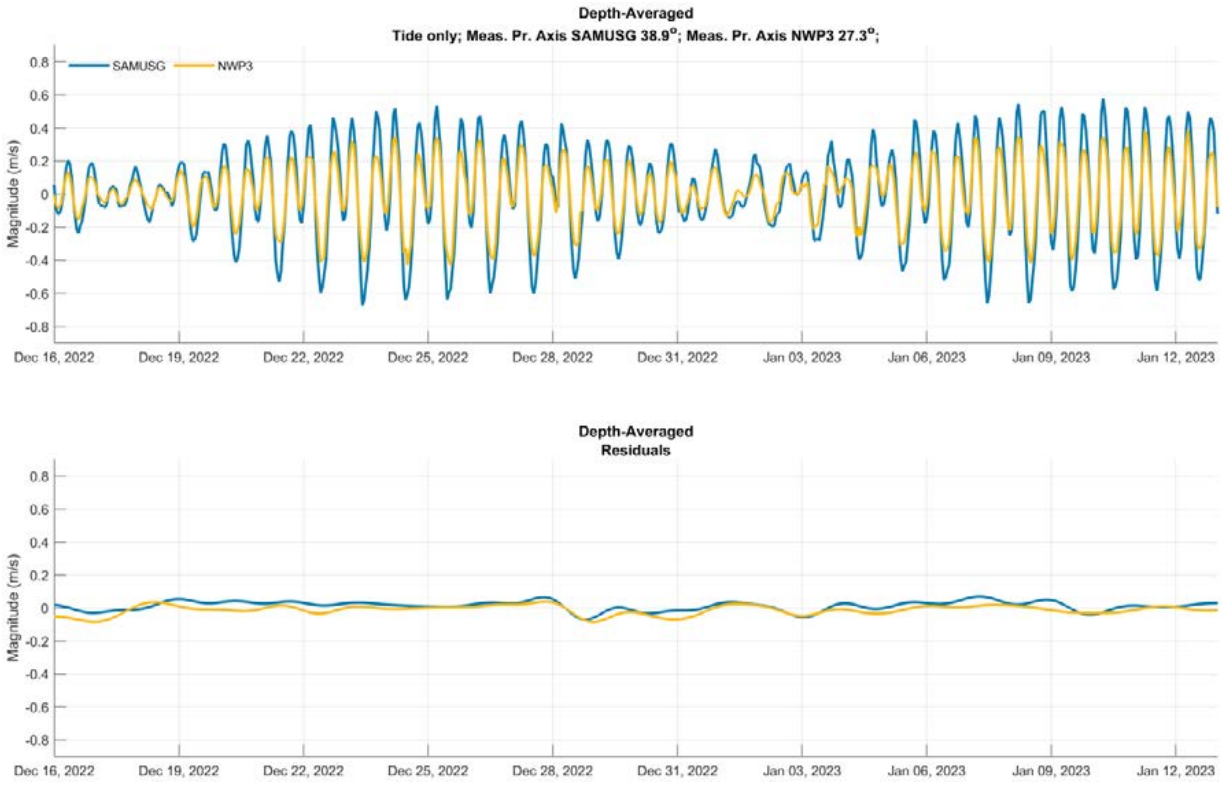


Figure A-18 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 16/12/2022 to 13/01/2023

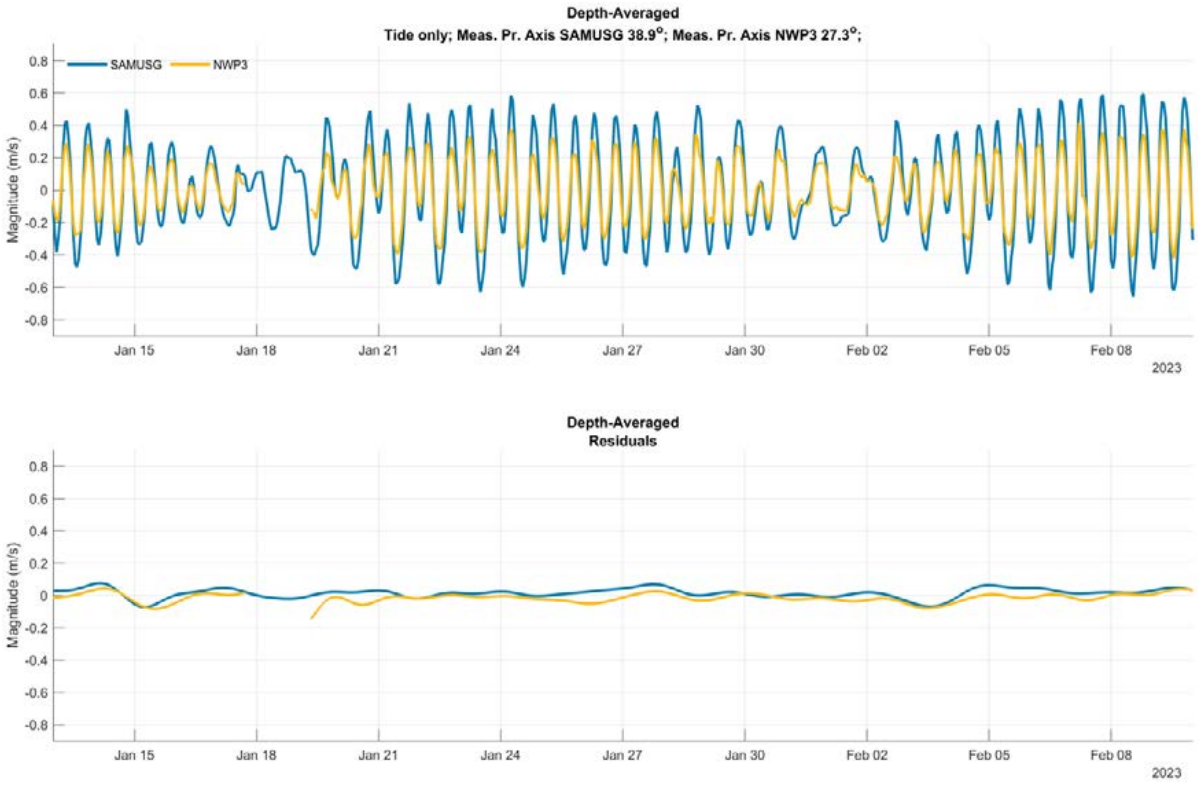


Figure A-19 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 13/01 to 10/02/2023

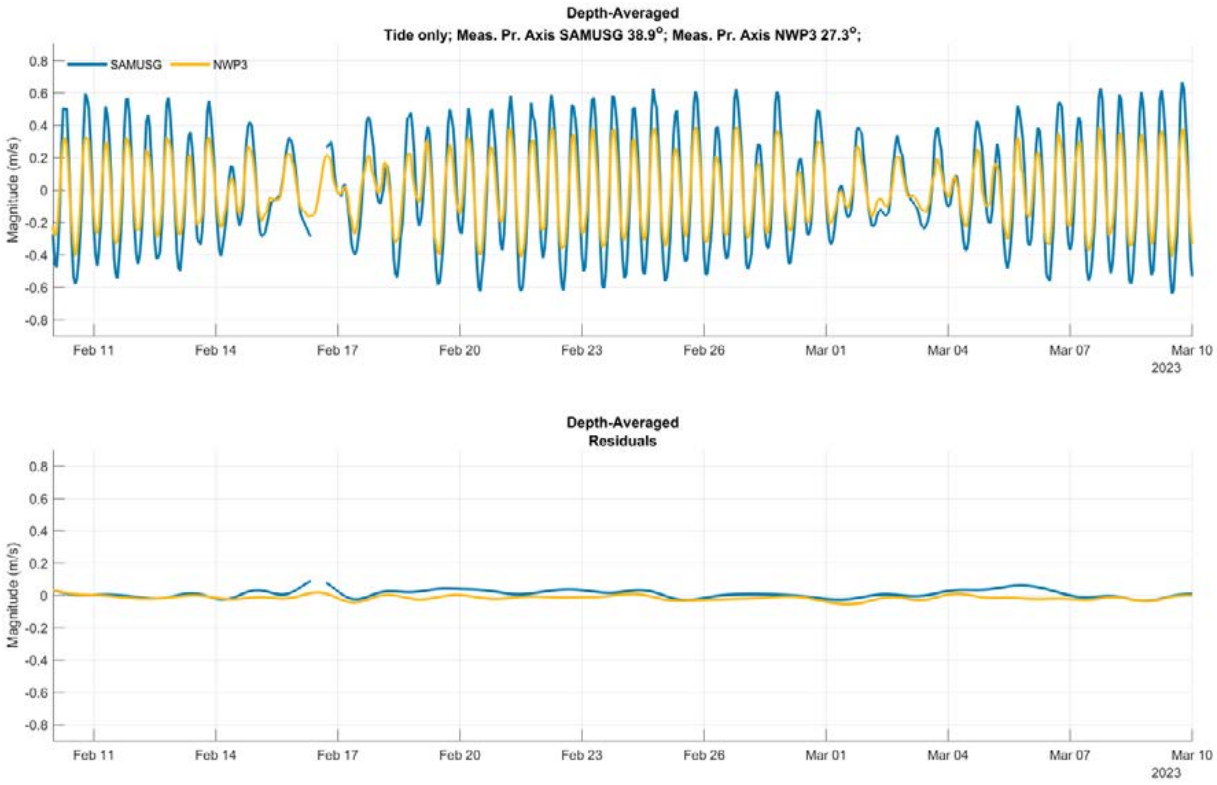


Figure A-20 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 10/02 to 10/03/2023

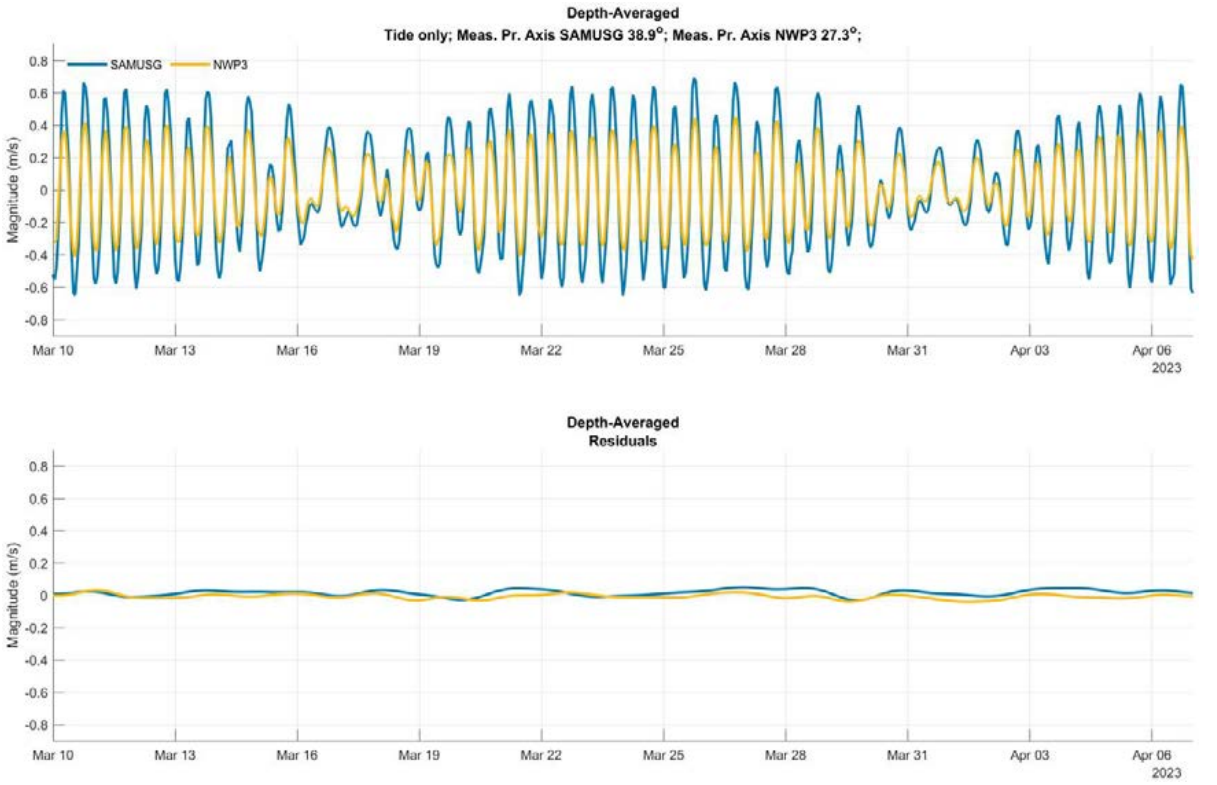


Figure A-21 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 10/03 to 07/04/2023

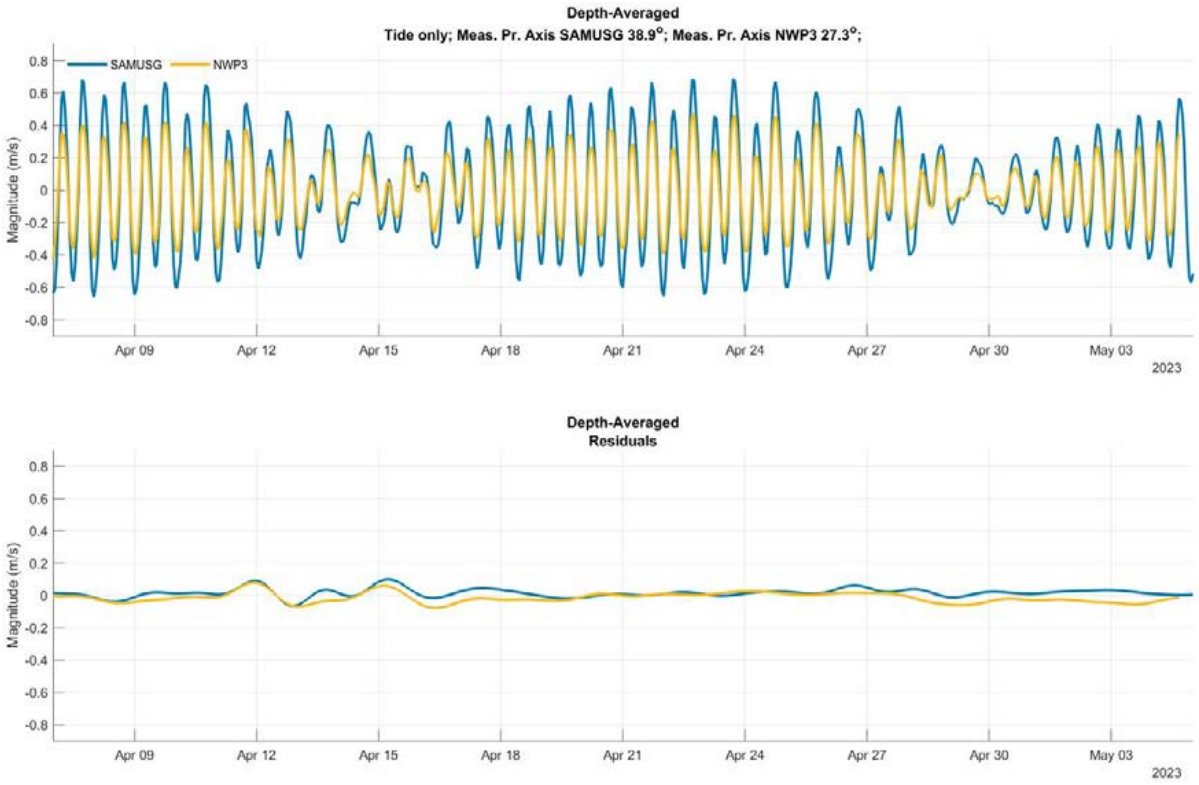


Figure A-22 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 08/04 to 05/05/2023

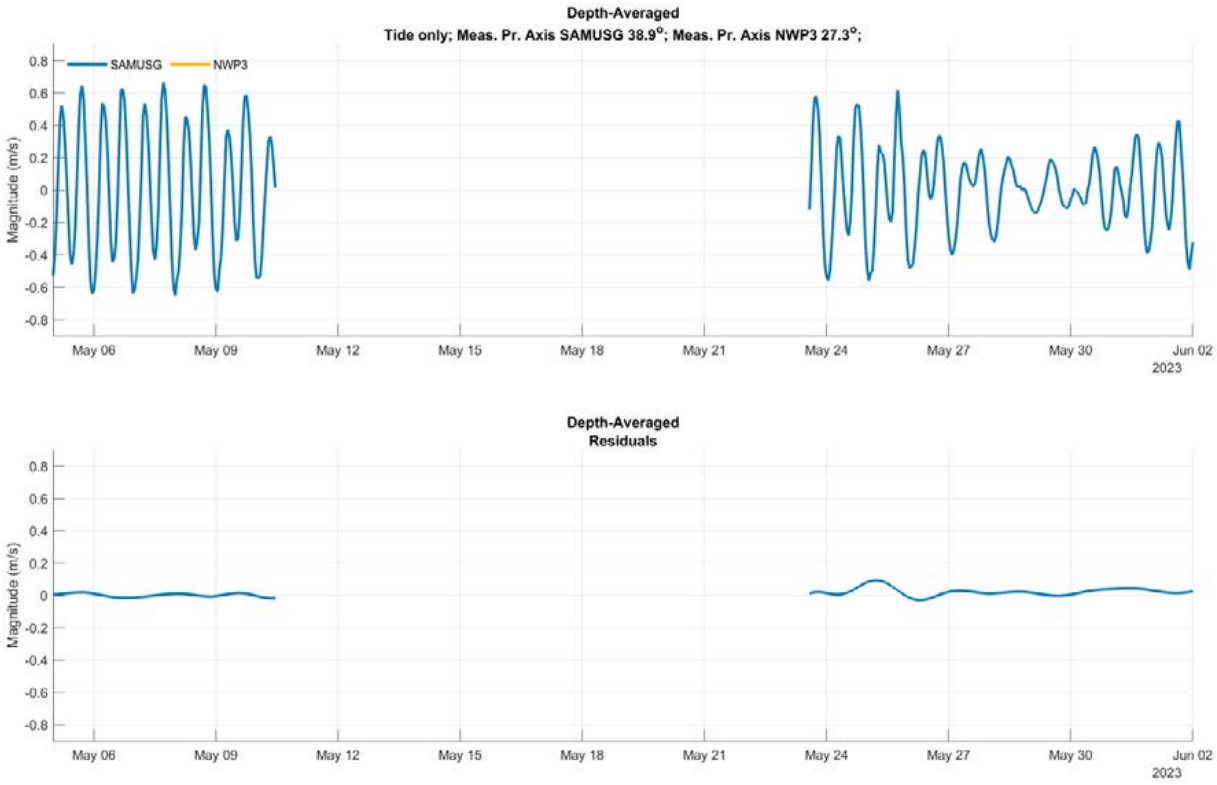


Figure A-23 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 05/05 to 02/06/2023

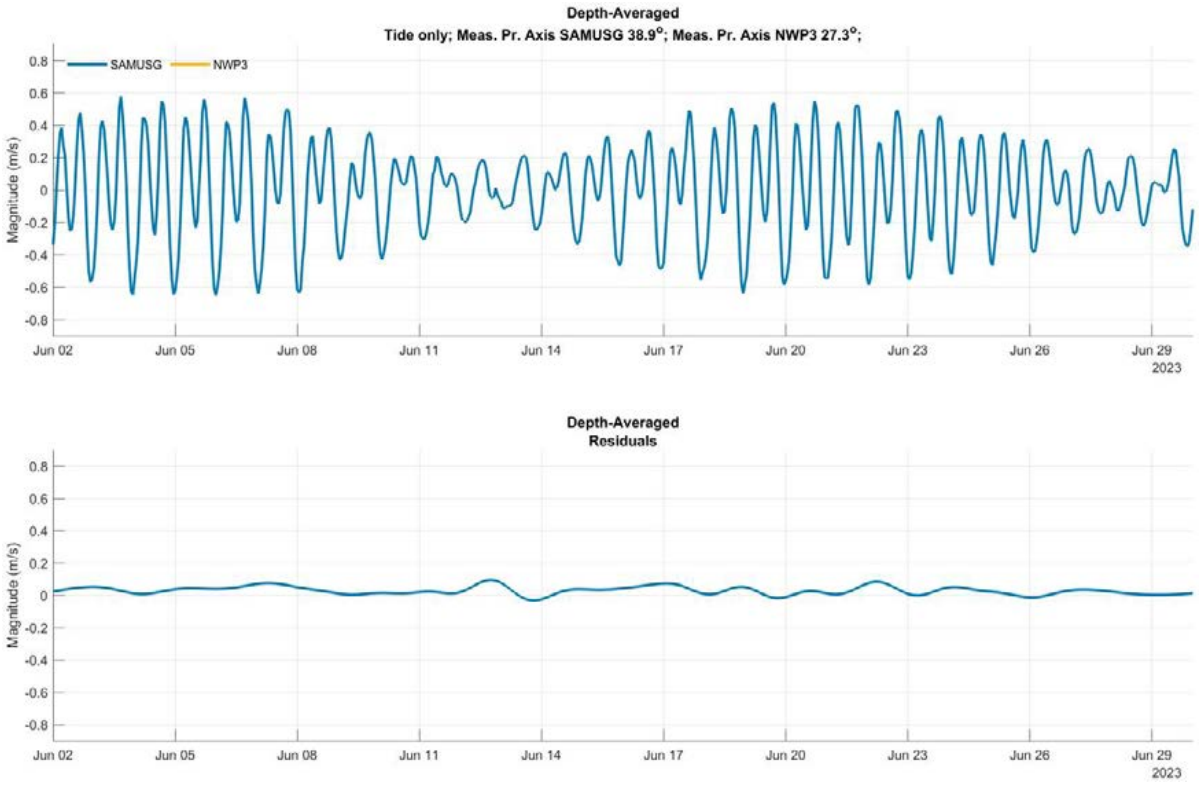


Figure A-24 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 02/06 to 30/06/2023

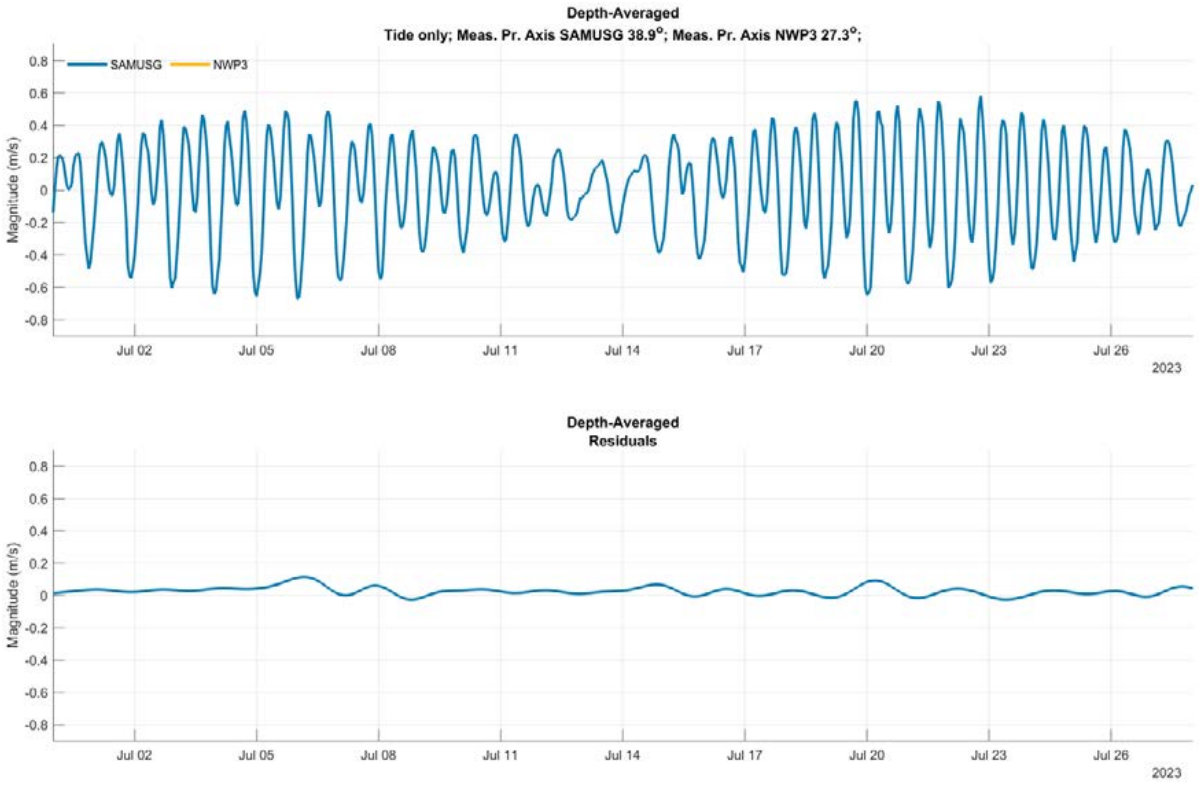


Figure A-25 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 30/06 to 28/07/2023

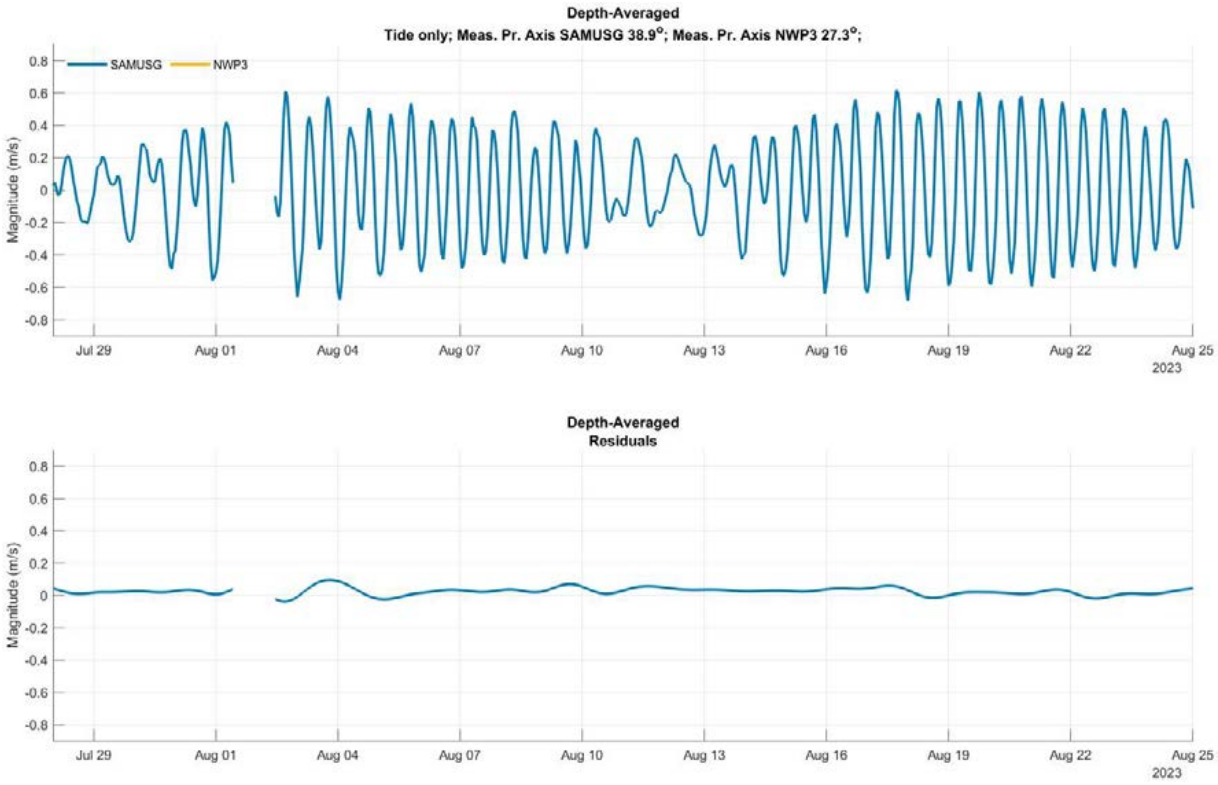


Figure A-26 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 28/07 to 25/08/2023

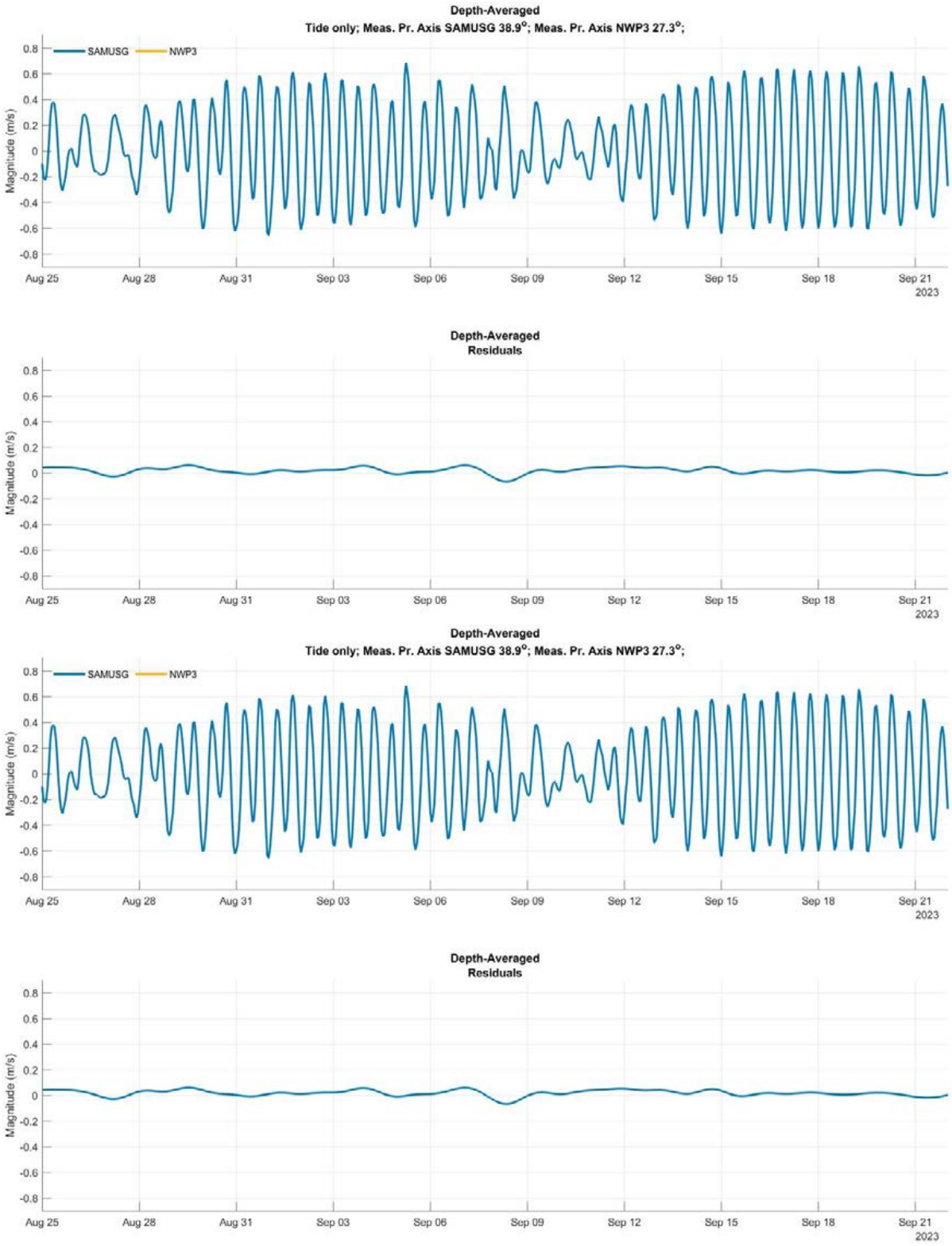


Figure A-27 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 25/08 to 22/09/2023

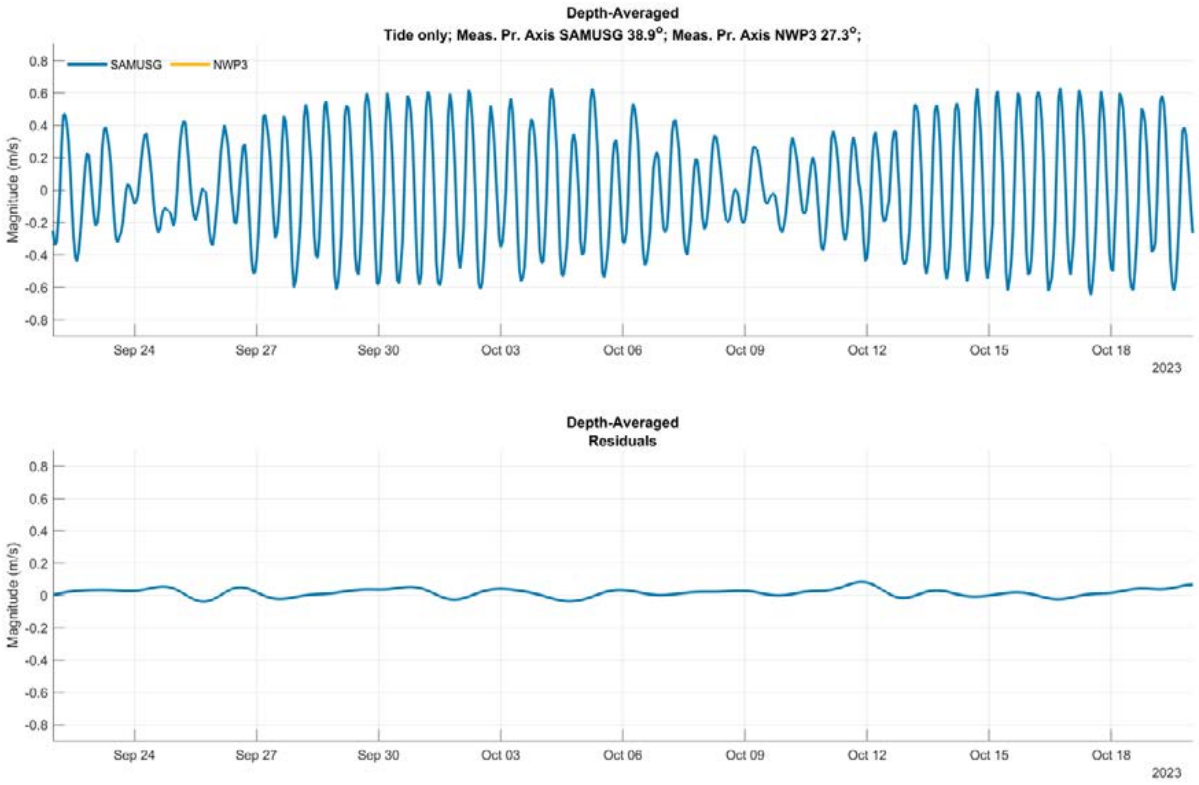


Figure A-28 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 22/09 to 20/10/2023

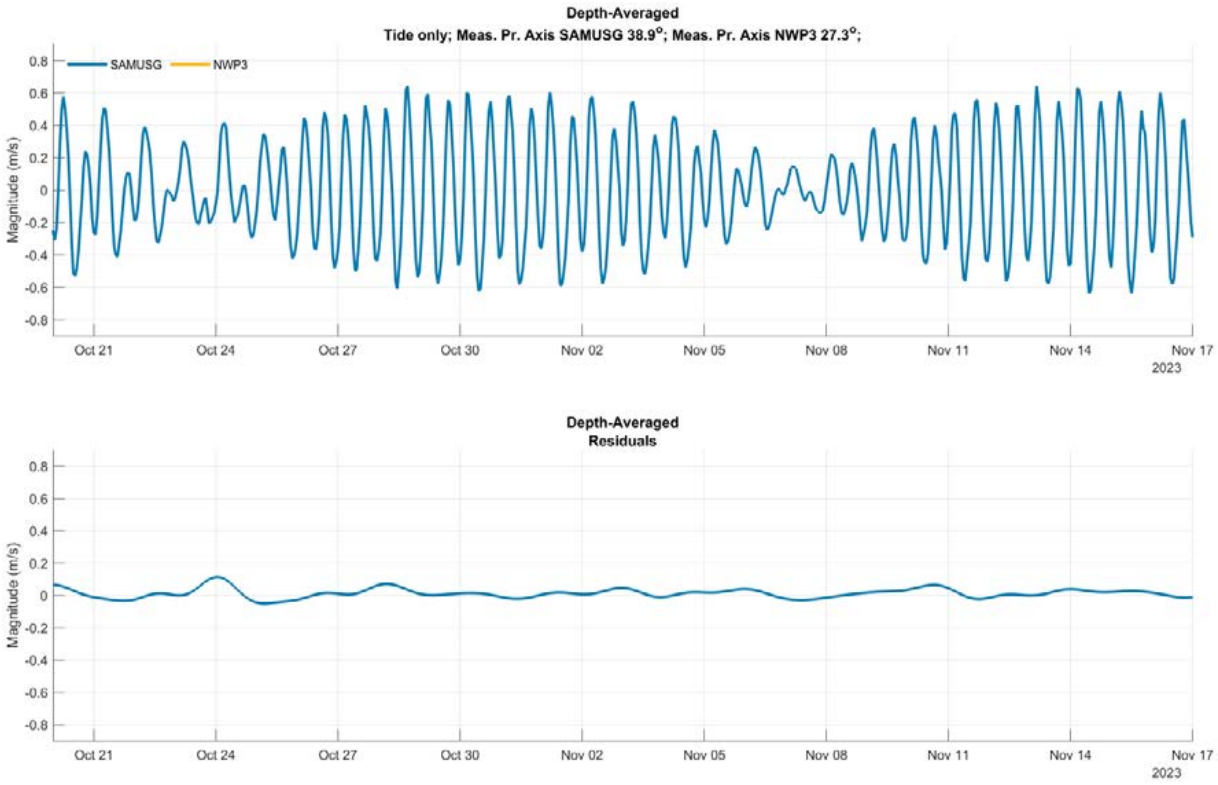


Figure A-29 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 20/10 to 17/11/2023

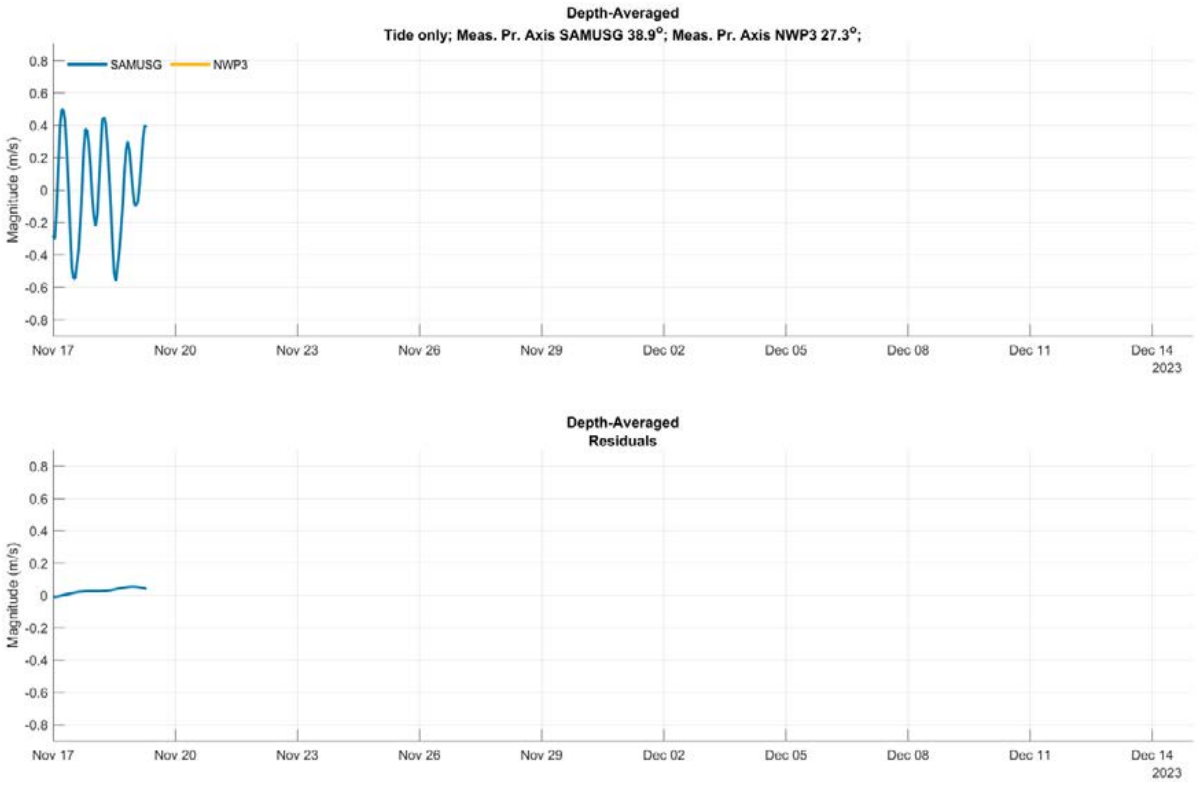


Figure A-30 Depth-Averaged Tidal and Associated Residual Currents at NWP3 and SAMUSG – 17/11 to 15/12/2023

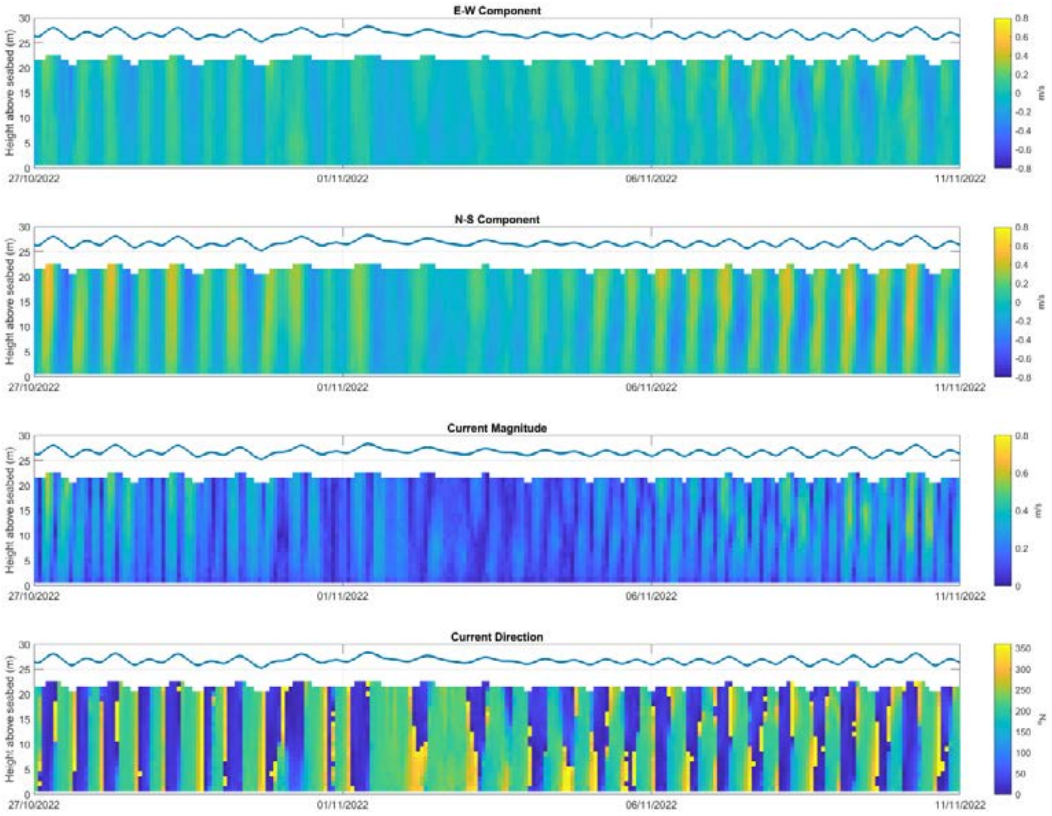


Figure A-31 Currents measured by the ADCP at NWP3 – 27/10 to 11/11/2022

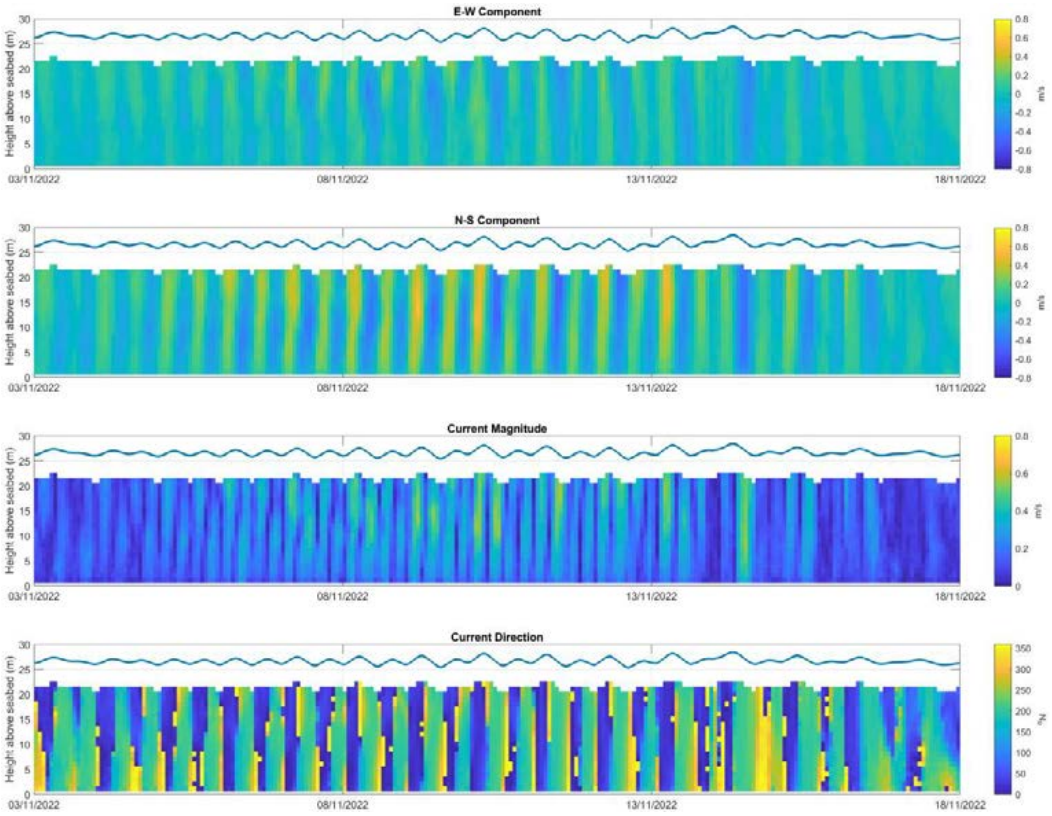


Figure A-32 Currents measured by the ADCP at NWP3 – 03/11 to 18/11/2022

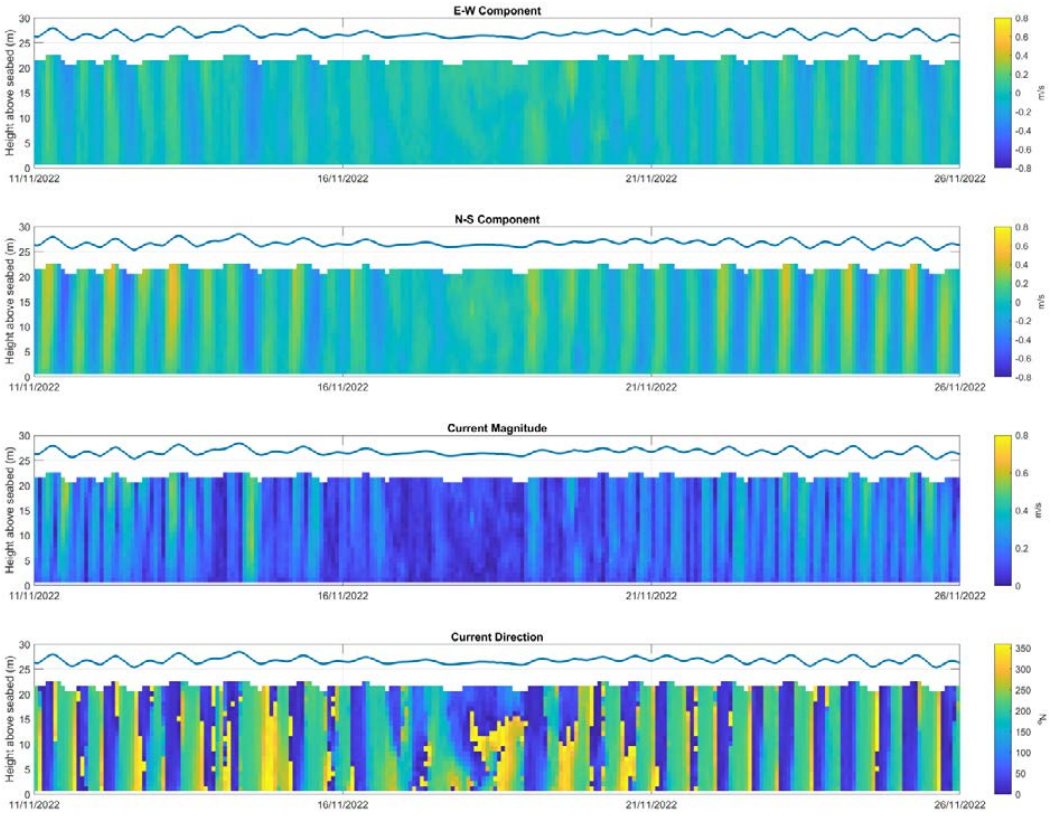


Figure A-33 Currents measured by the ADCP at NWP3 – 11/11 to 26/11/2022

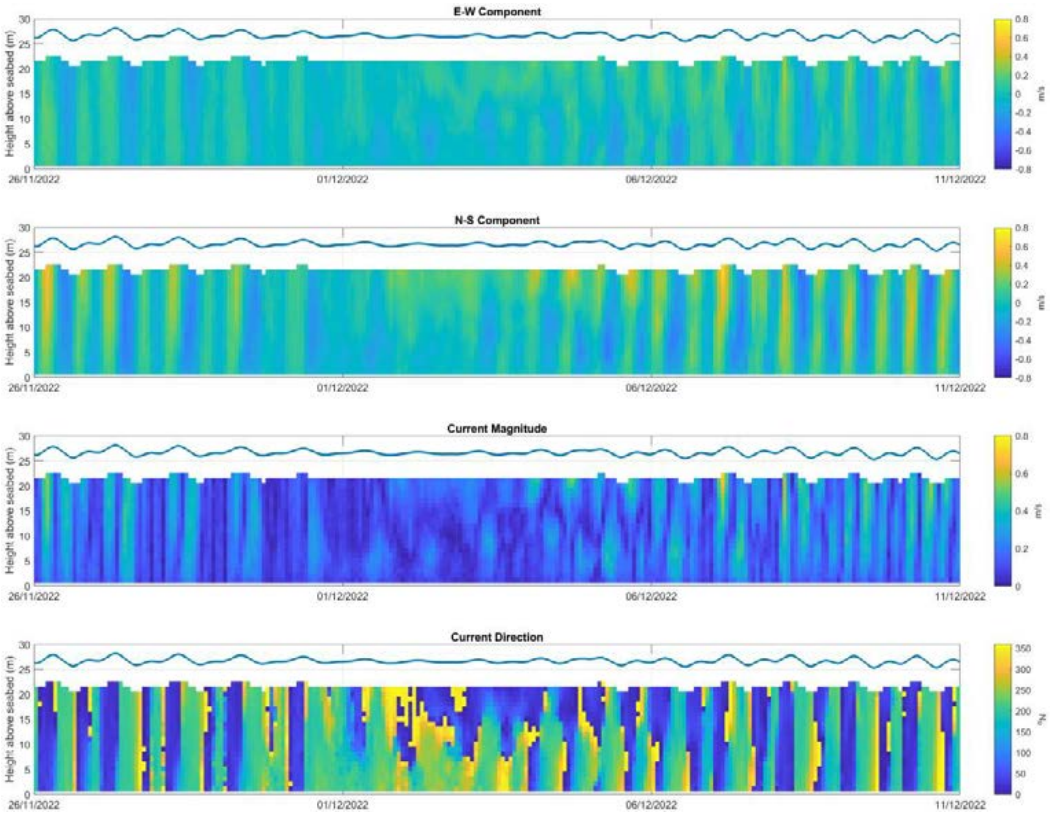


Figure A-34 Currents measured by the ADCP at NWP3 – 26/11 to 11/12/2022

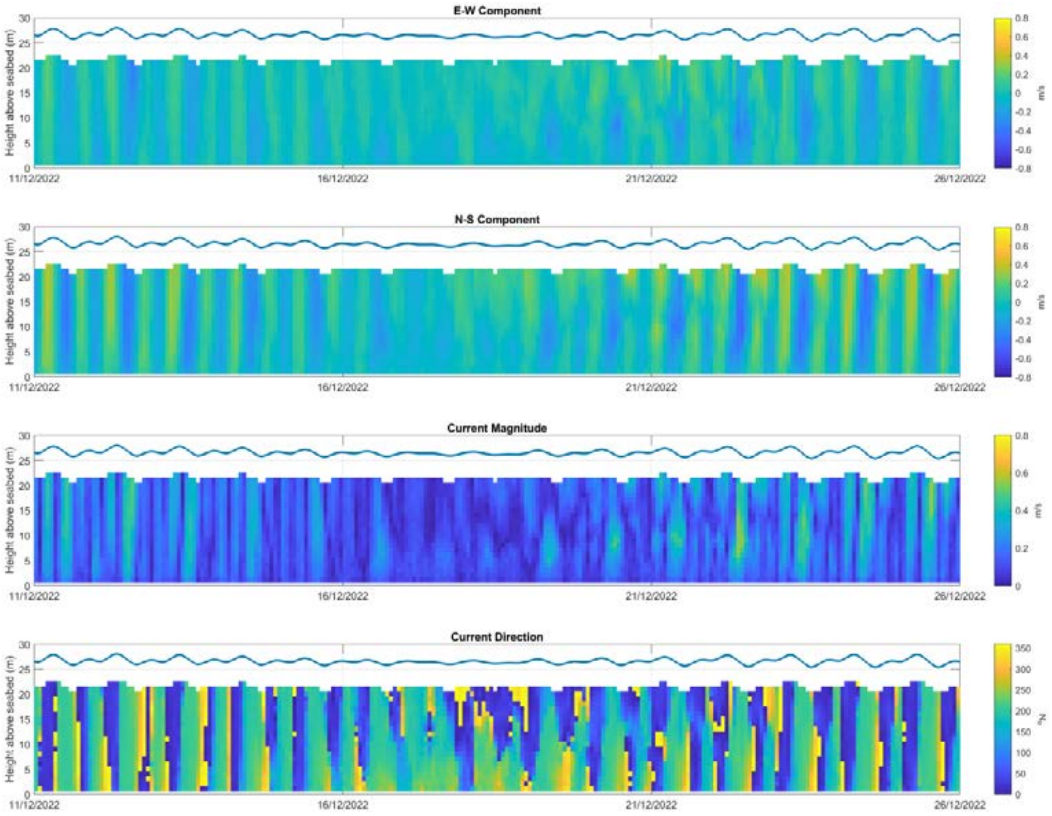


Figure A-35 Currents measured by the ADCP at NWP3 – 11/12 to 26/12/2022

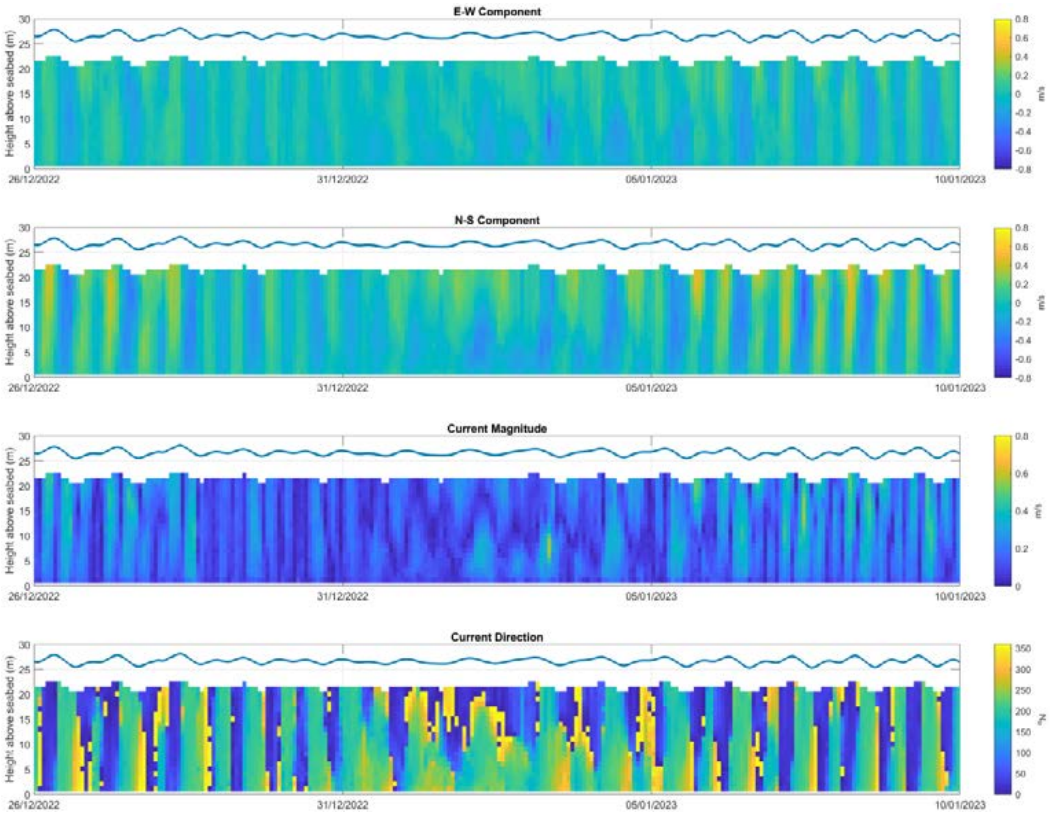


Figure A-36 Currents measured by the ADCP at NWP3 – 26/12/2022 to 10/01/2023

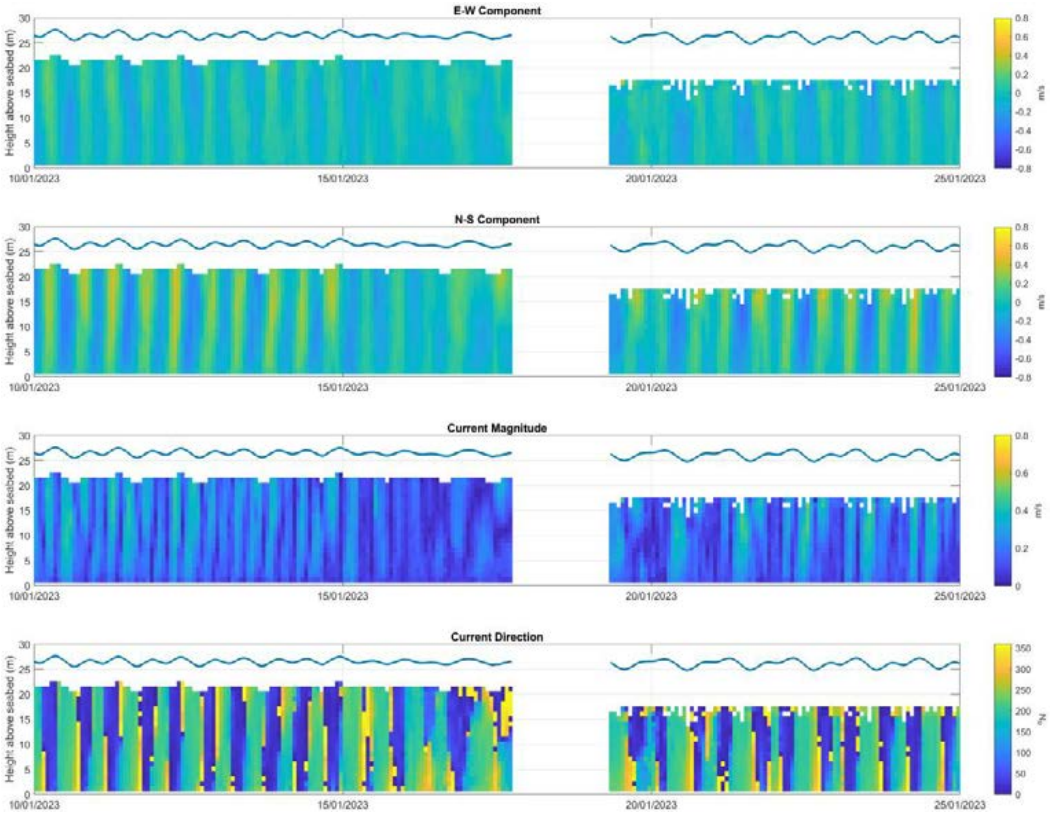


Figure A-37 Currents measured by the ADCP at NWP3 – 10/01 to 25/01/2023

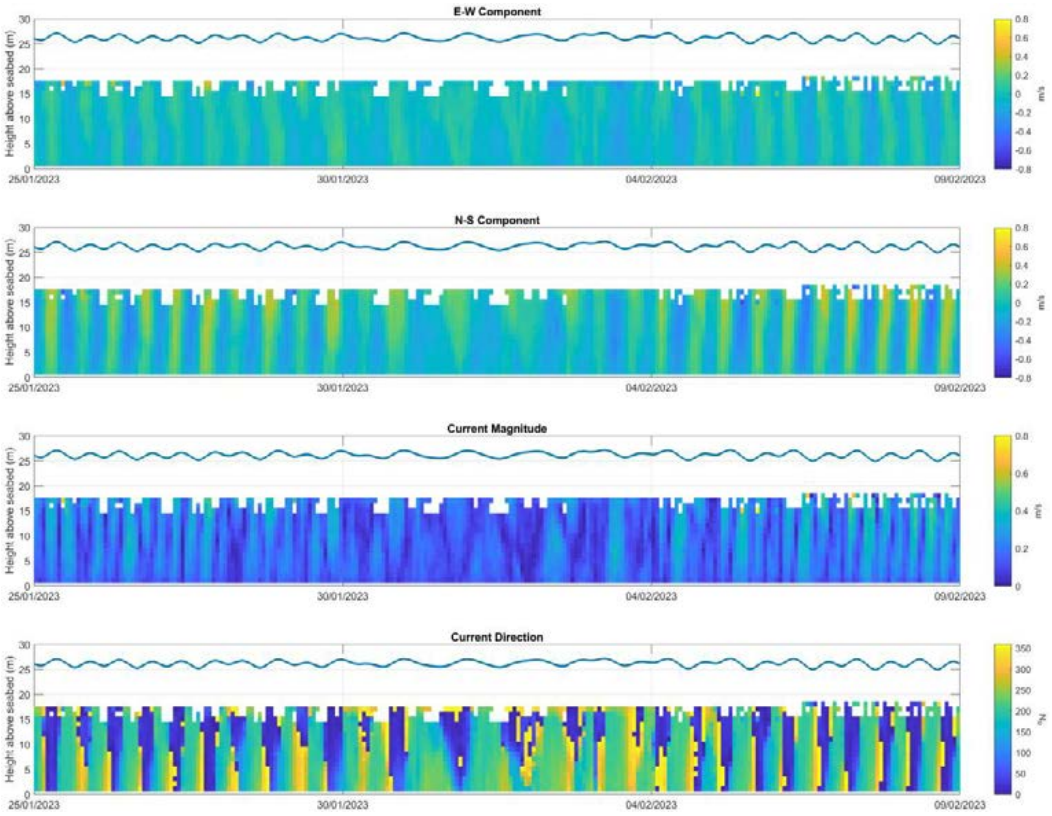


Figure A-38 Currents measured by the ADCP at NWP3 – 25/01 to 09/02/2023

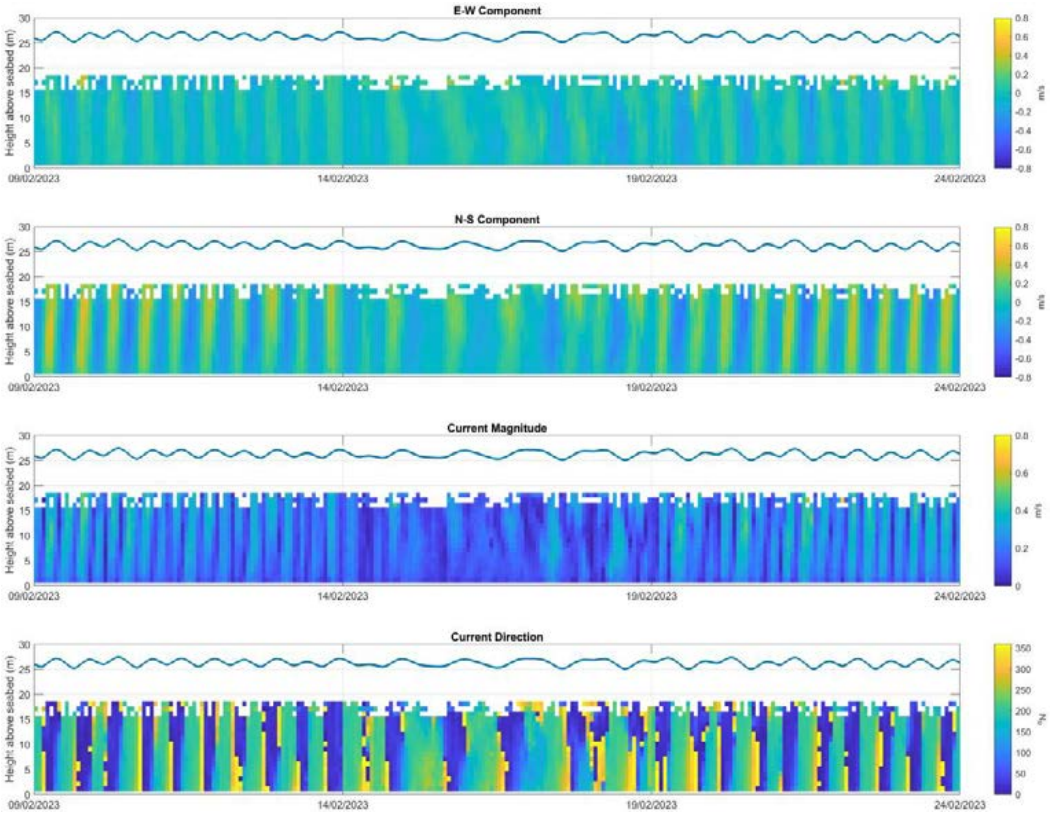


Figure A-39 Currents measured by the ADCP at NWP3 – 09/02 to 24/02/2023

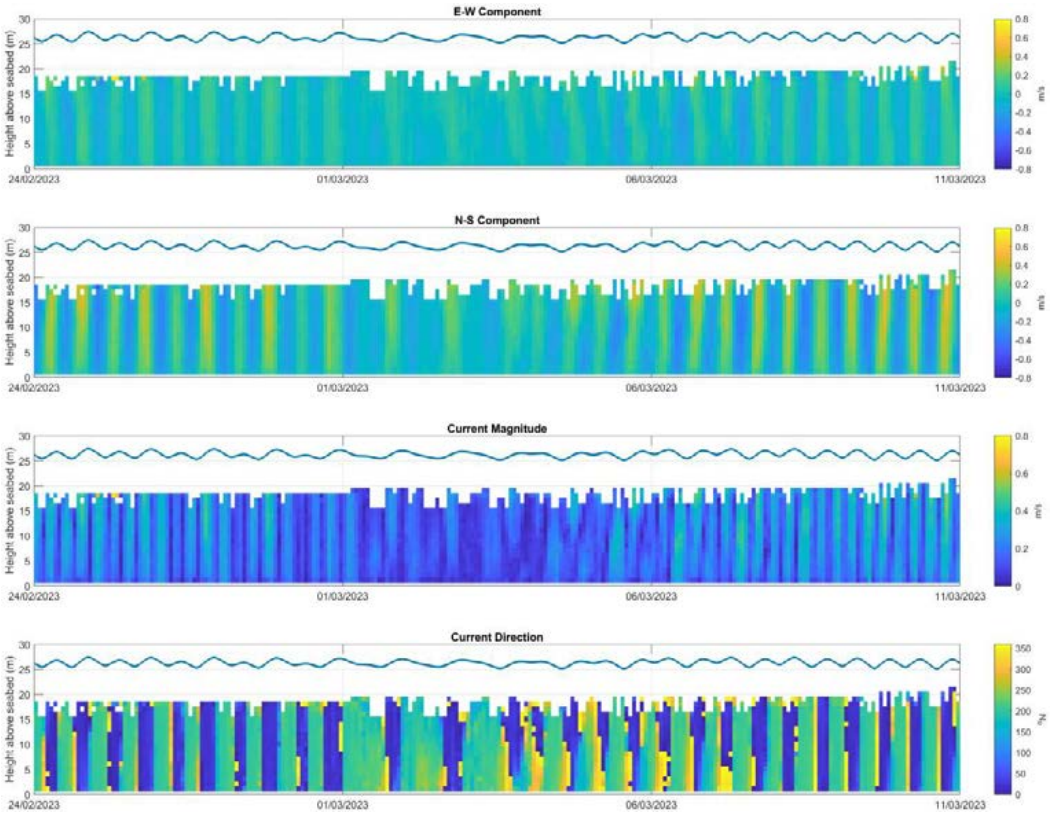


Figure A-40 Currents measured by the ADCP at NWP3 – 24/02 to 11/03/2023

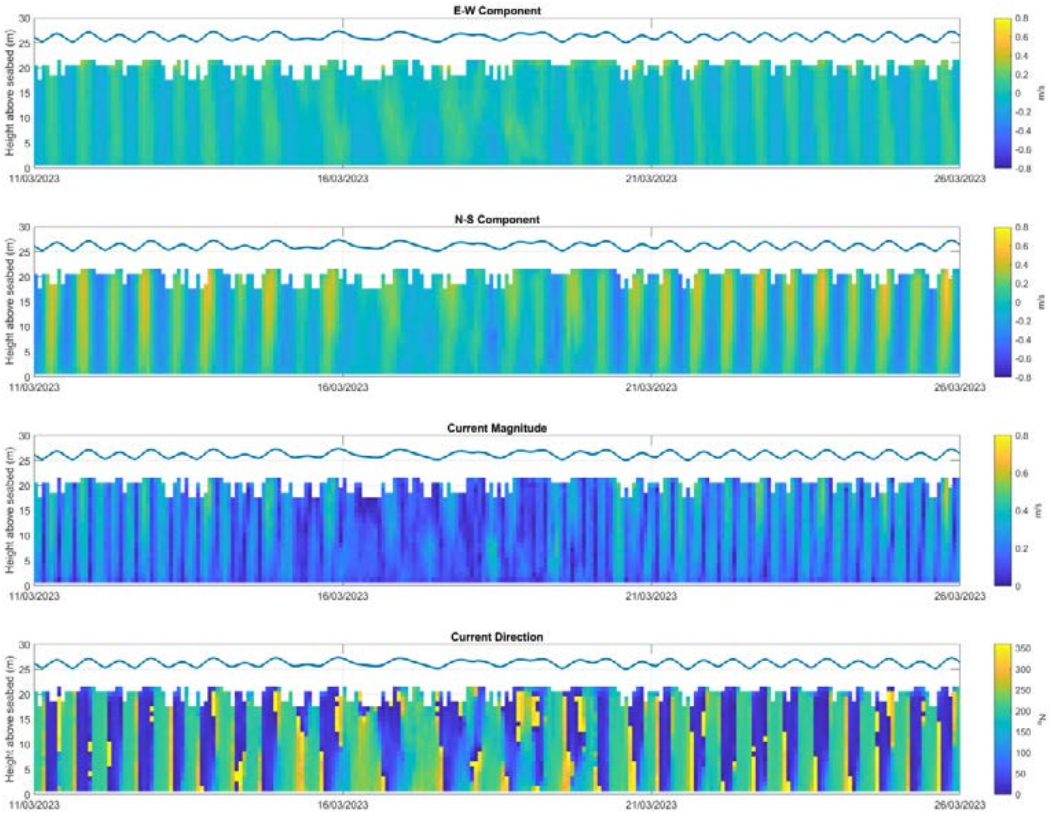


Figure A-41 Currents measured by the ADCP at NWP3 – 11/03 to 26/03/2023

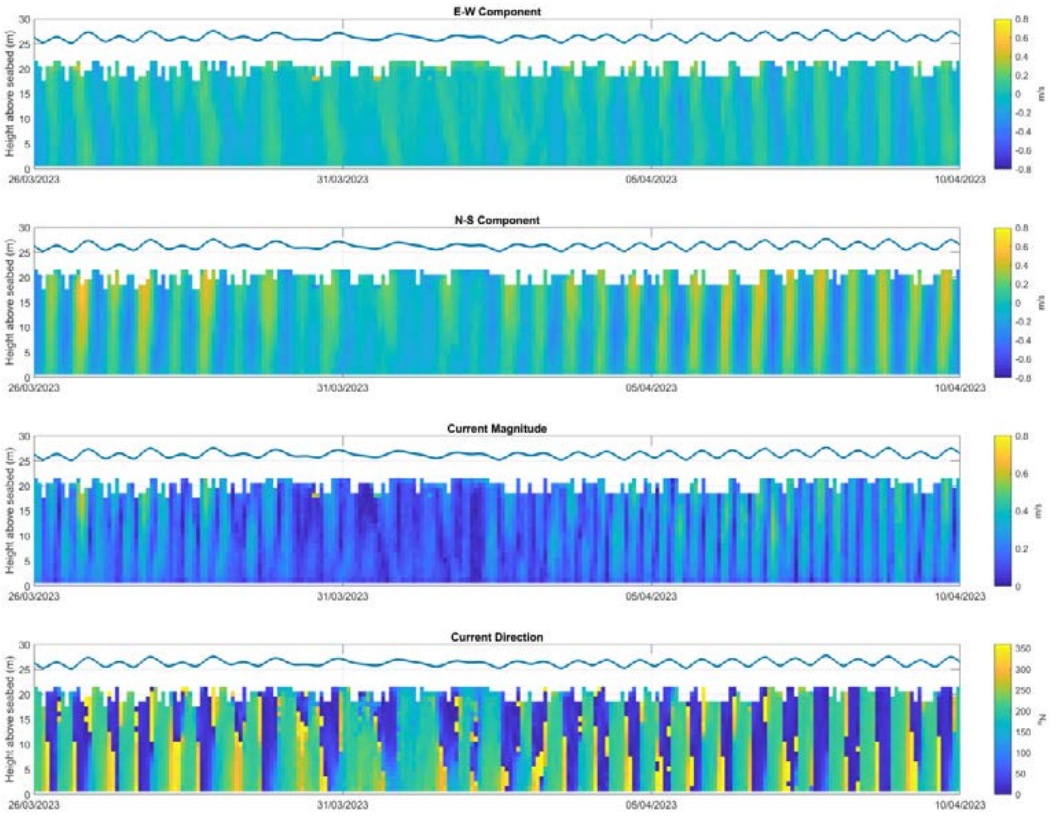


Figure A-42 Currents measured by the ADCP at NWP3 – 26/03 to 10/04/2023

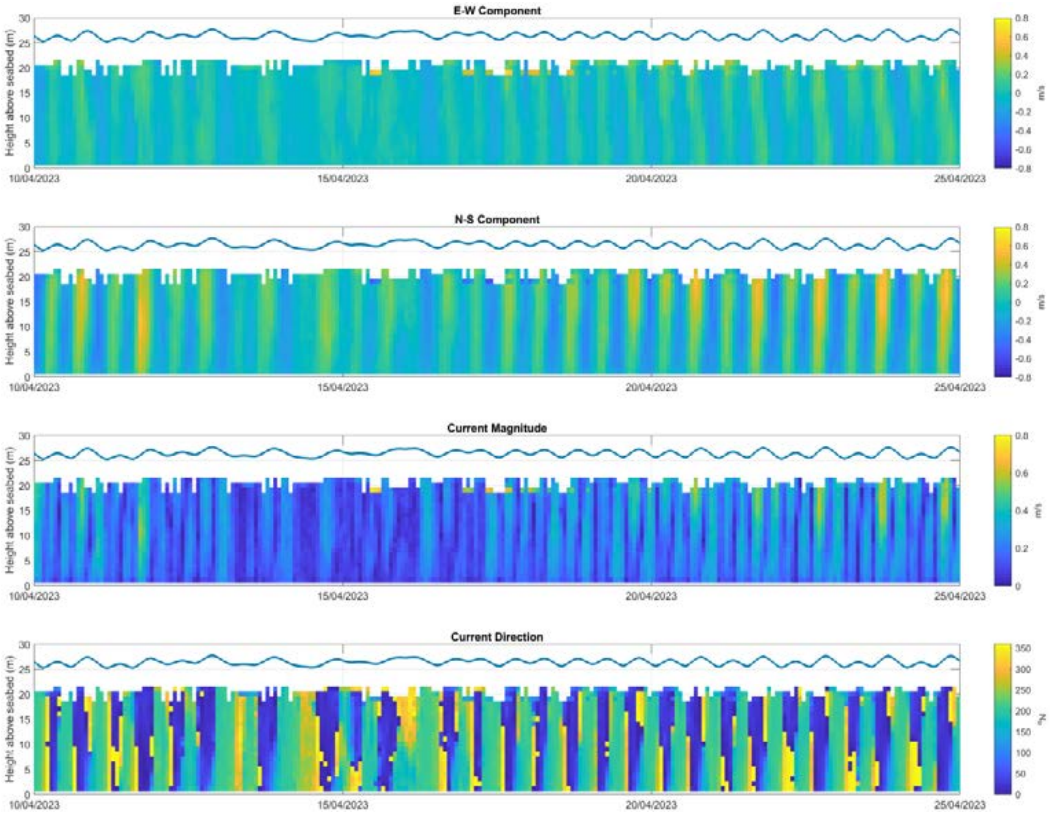


Figure A-43 Currents measured by the ADCP at NWP3 – 10/04 to 25/04/2023

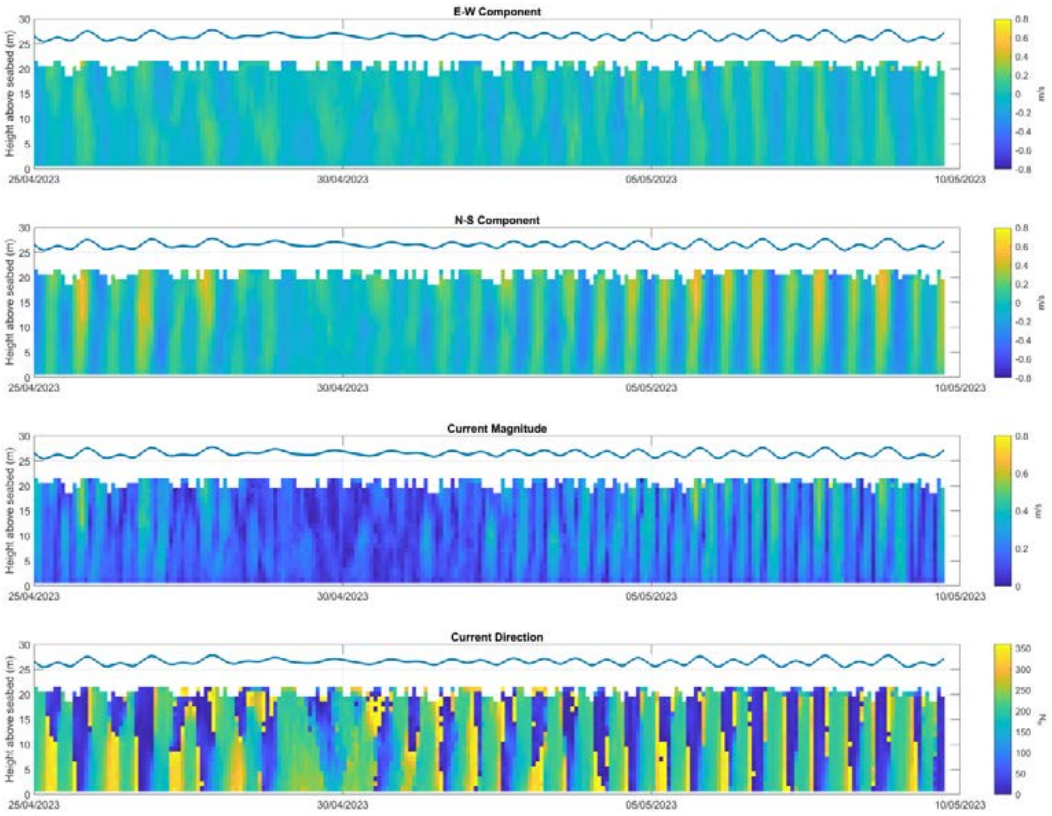


Figure A-44 Currents measured by the ADCP at NWP3 – 25/04 to 10/05/2023

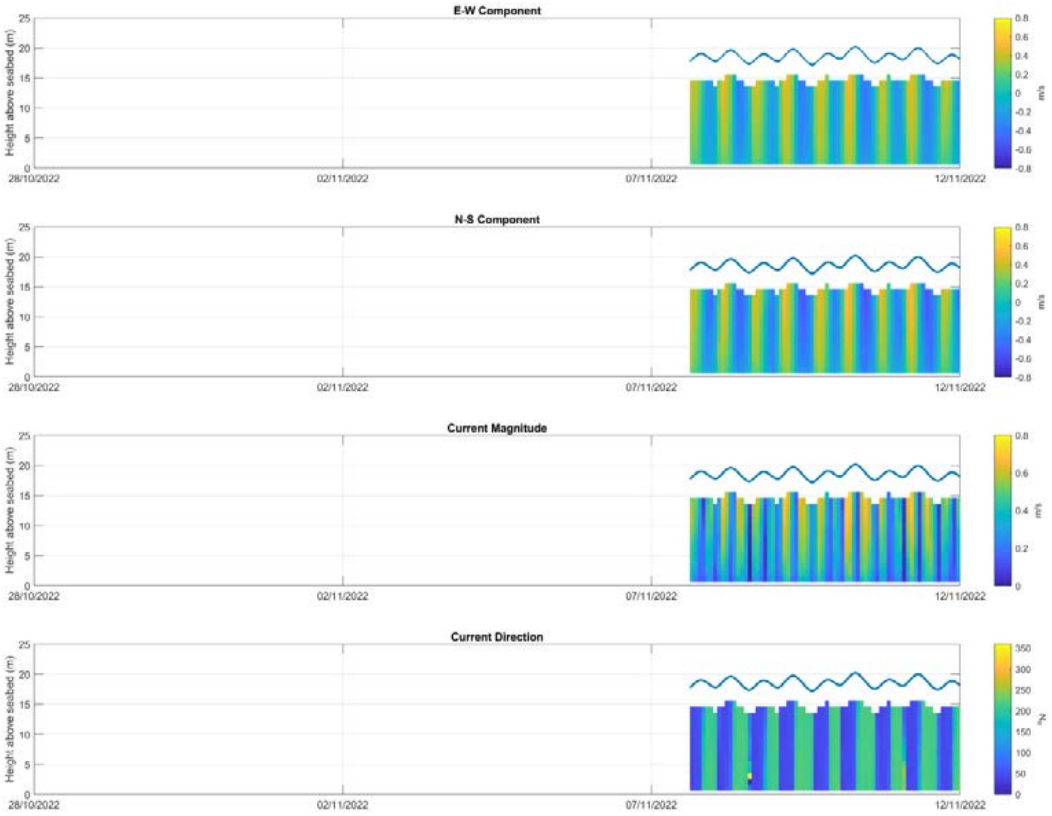


Figure A-45 Currents measured by the ADCP at SAMUSG – 28/10 to 12/11/2022

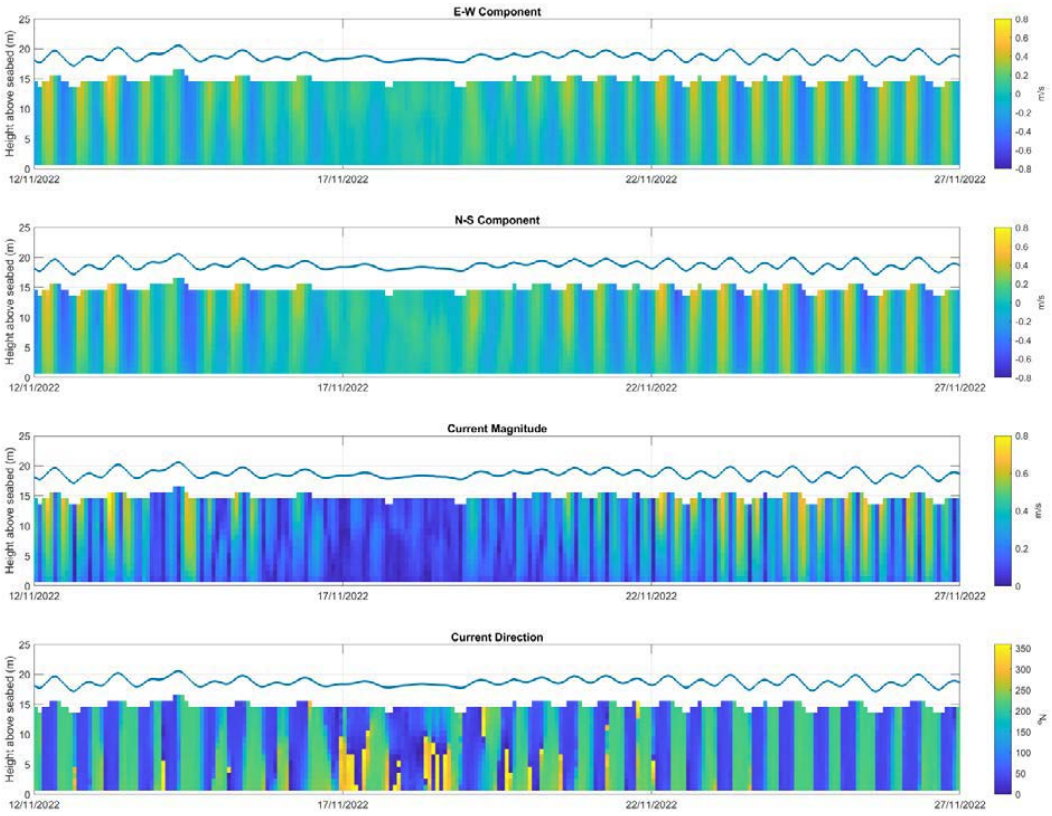


Figure A-46 Currents measured by the ADCP at SAMUSG – 12/11 to 27/11/2022

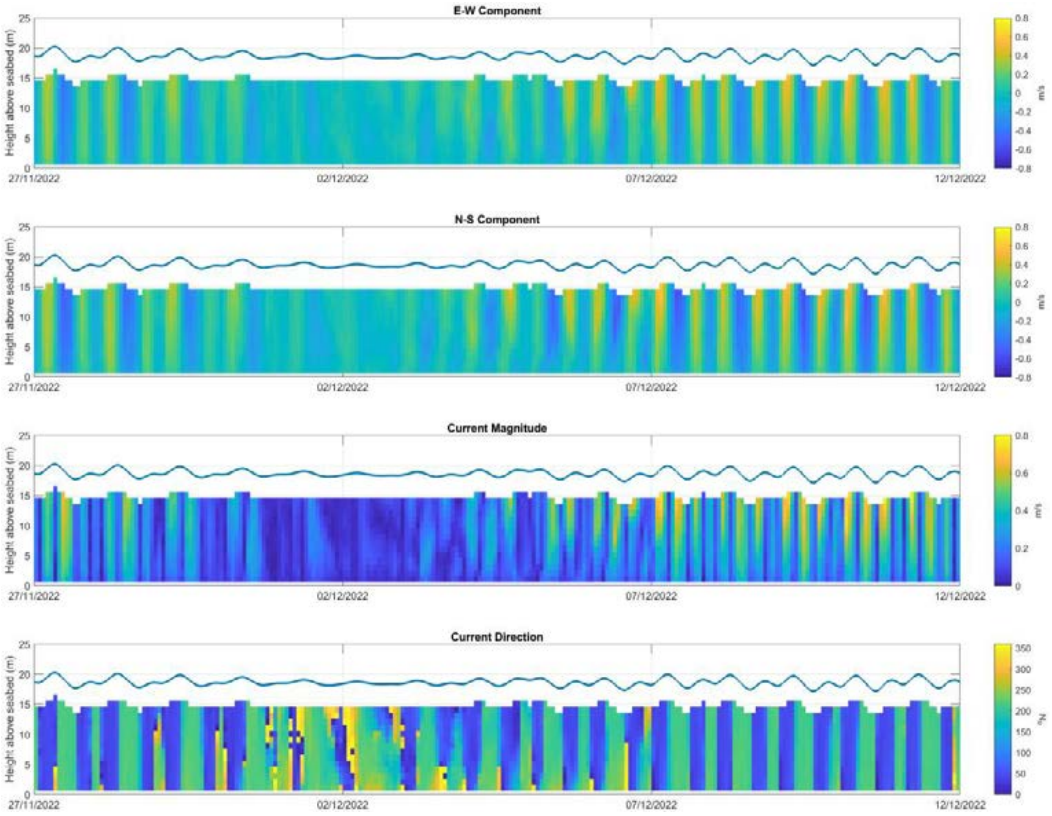


Figure A-47 Currents measured by the ADCP at SAMUSG – 27/11 to 12/12/2022

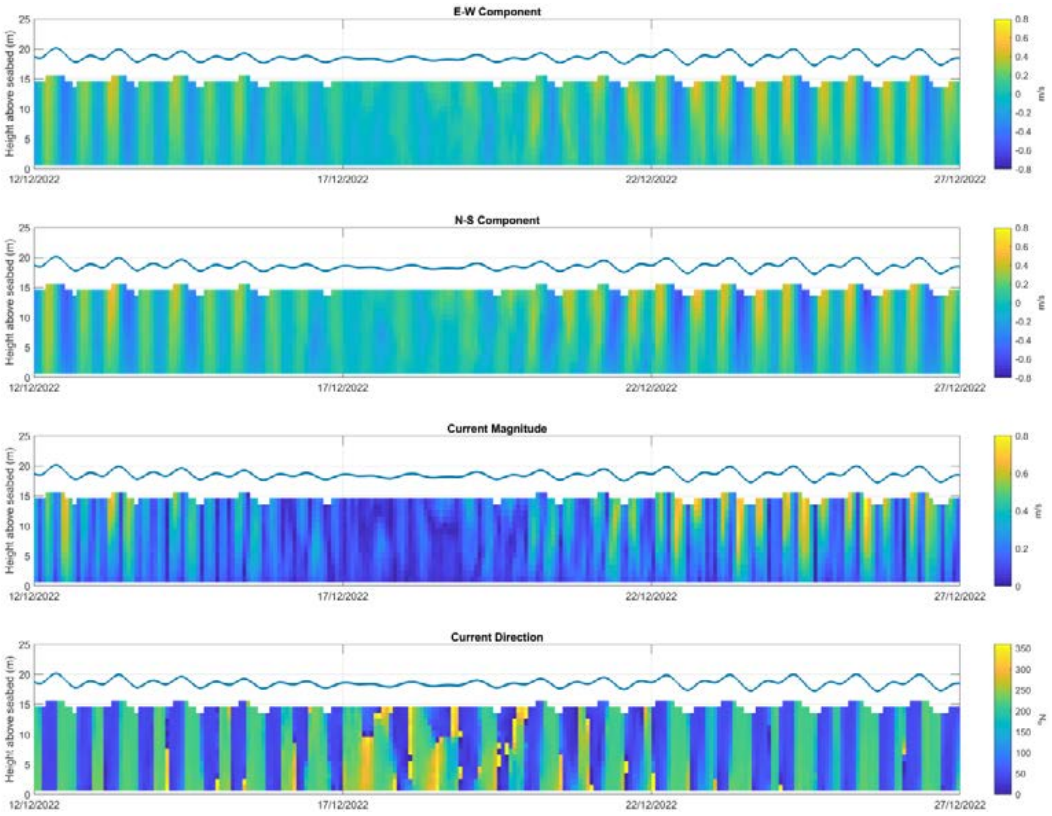


Figure A-48 Currents measured by the ADCP at SAMUSG – 12/12 to 27/12/2022

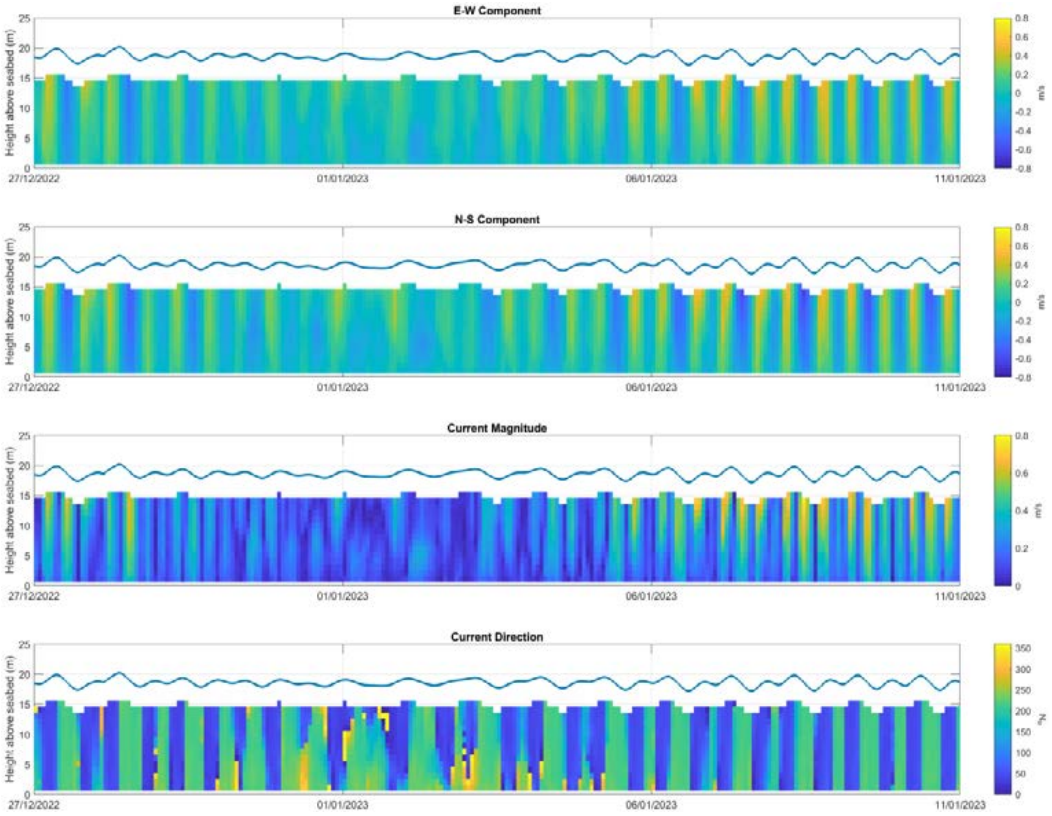


Figure A-49 Currents measured by the ADCP at SAMUSG – 27/12/2022 to 11/01/2023

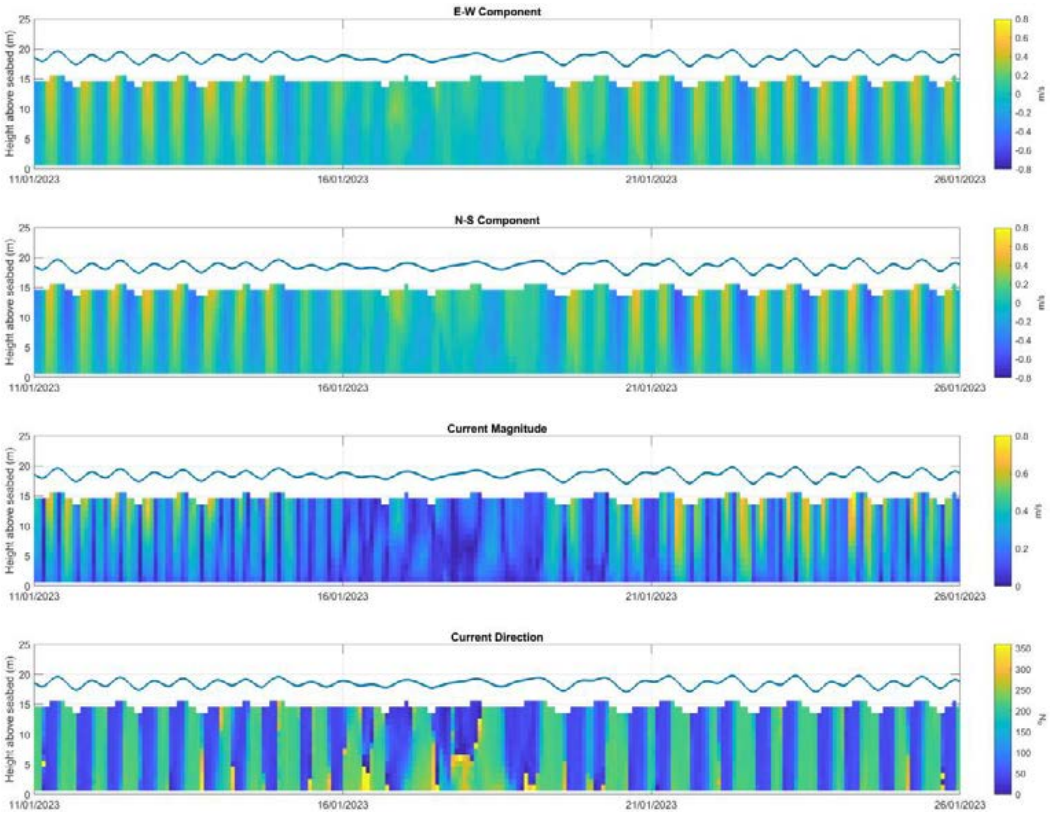


Figure A-50 Currents measured by the ADCP at SAMUSG – 11/01 to 26/01/2023

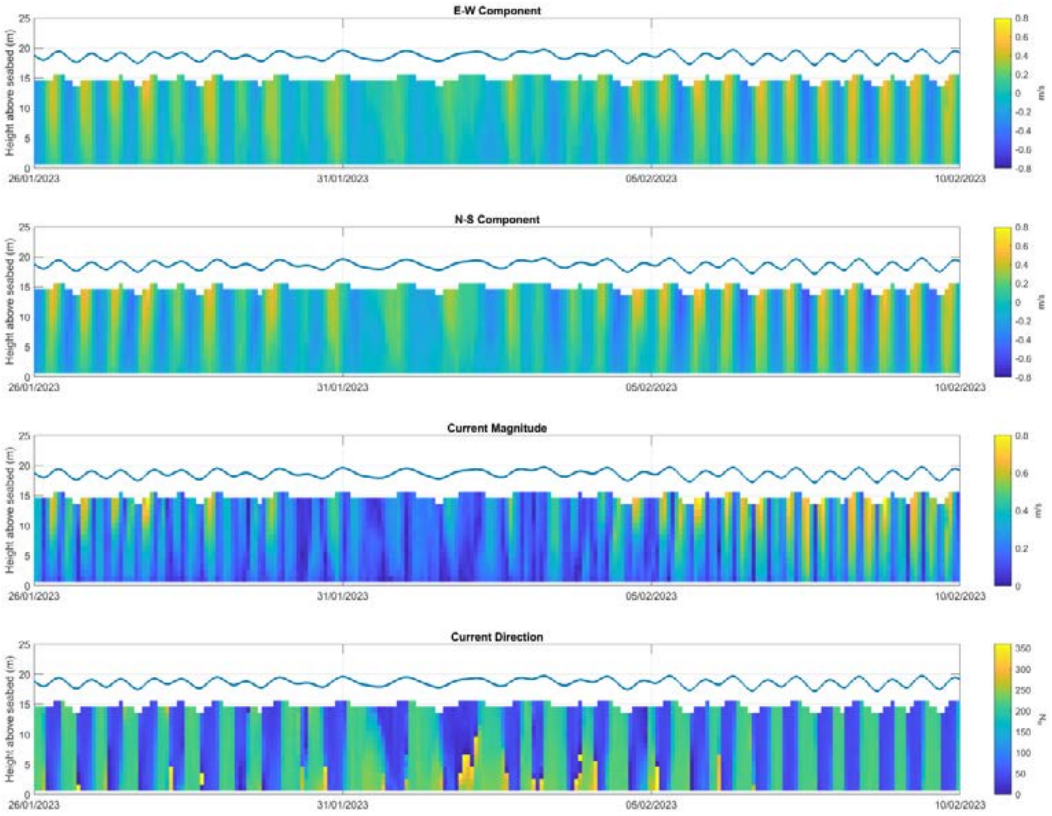


Figure A-51 Currents measured by the ADCP at SAMUSG – 26/01 to 10/02/2023

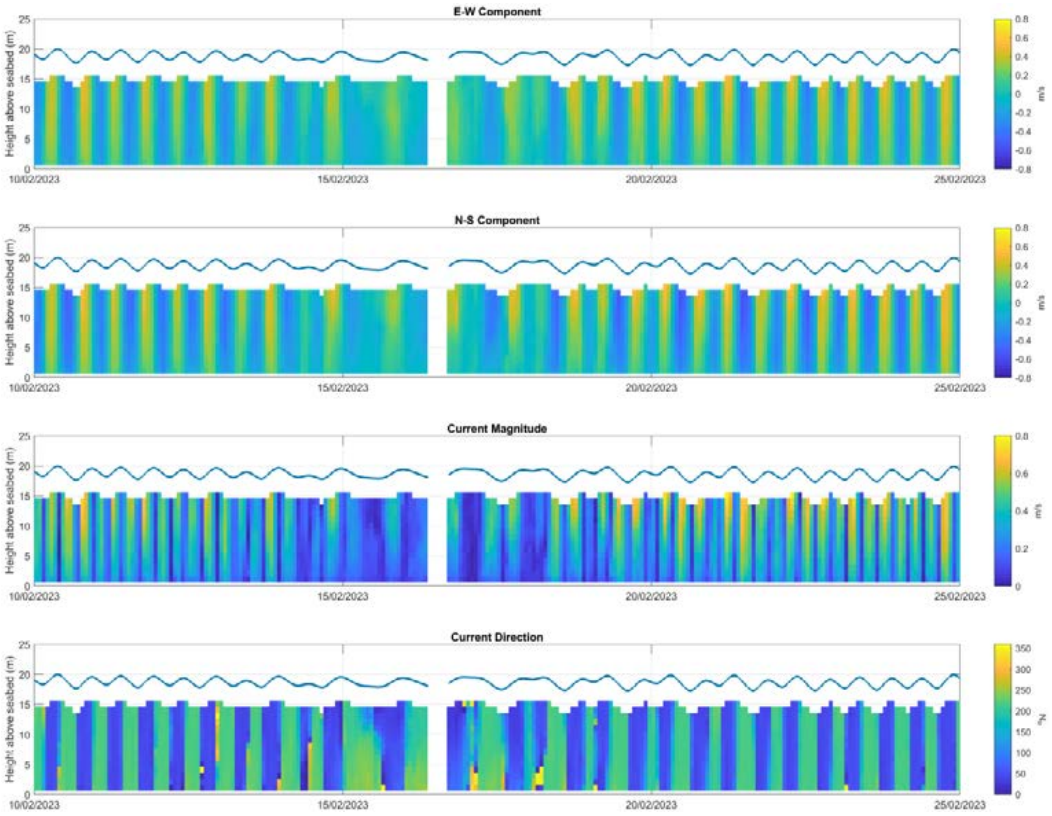


Figure A-52 Currents measured by the ADCP at SAMUSG – 10/02 to 25/02/2023

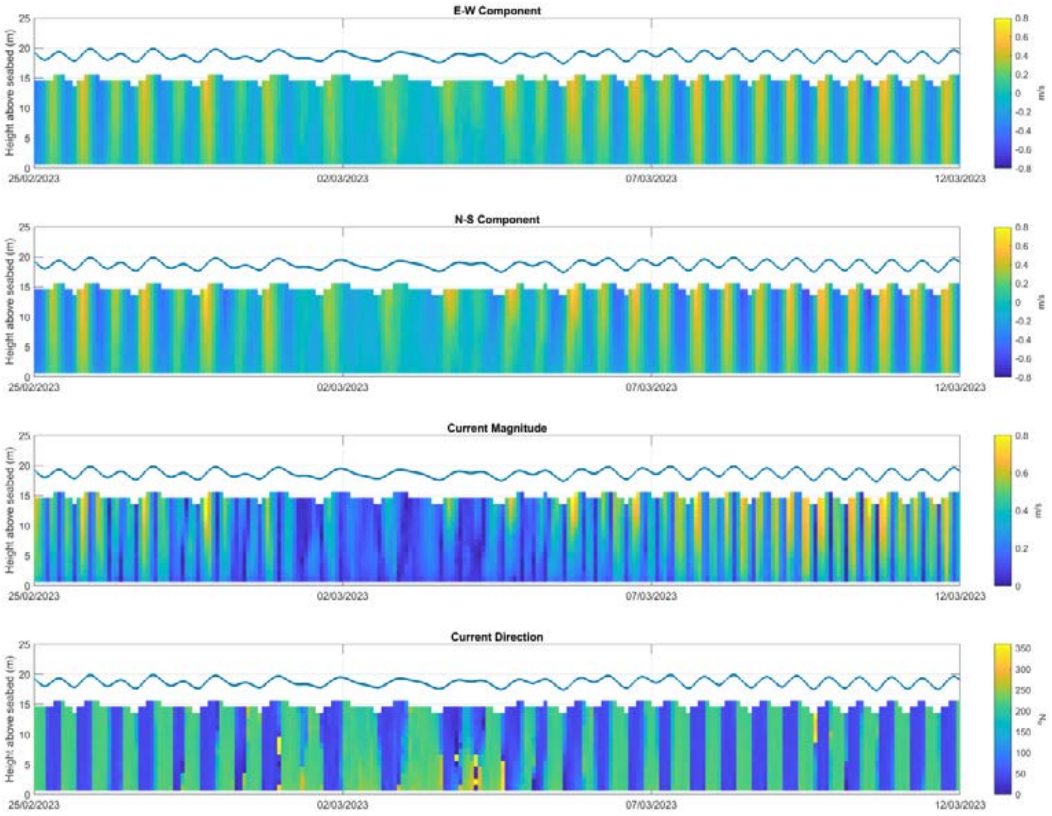


Figure A-53 Currents measured by the ADCP at SAMUSG – 25/02 to 12/03/2023

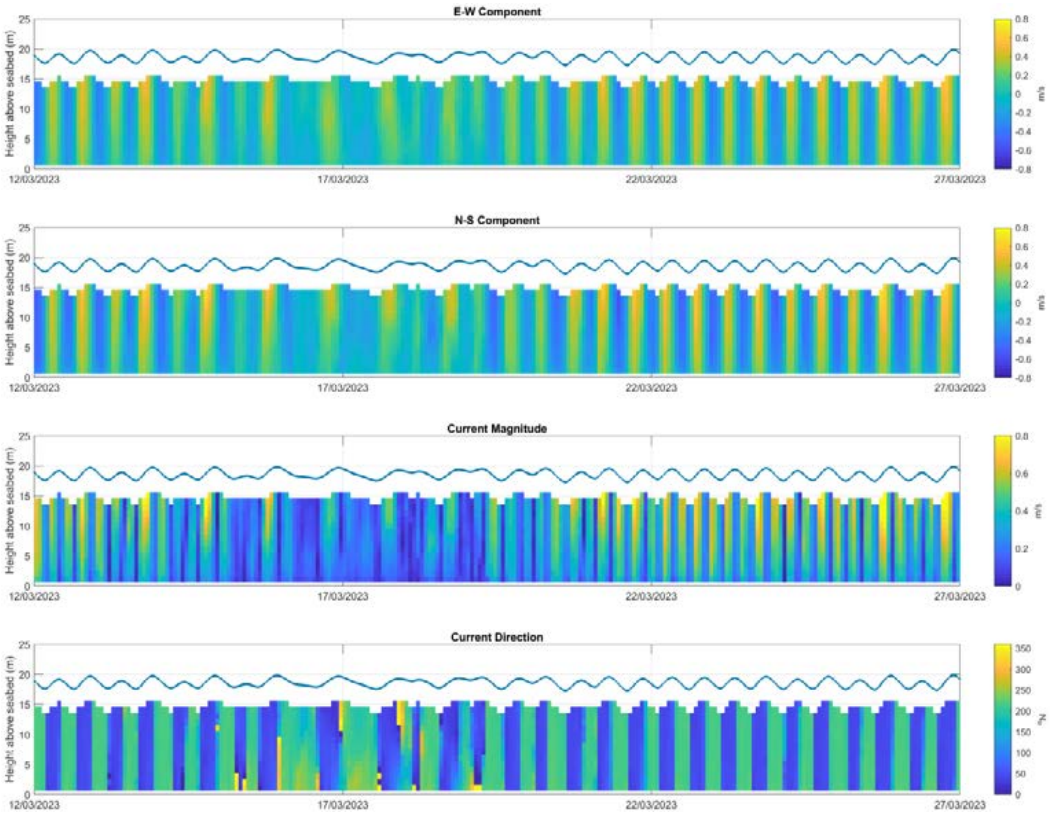


Figure A-54 Currents measured by the ADCP at SAMUSG – 12/03 to 27/03/2023

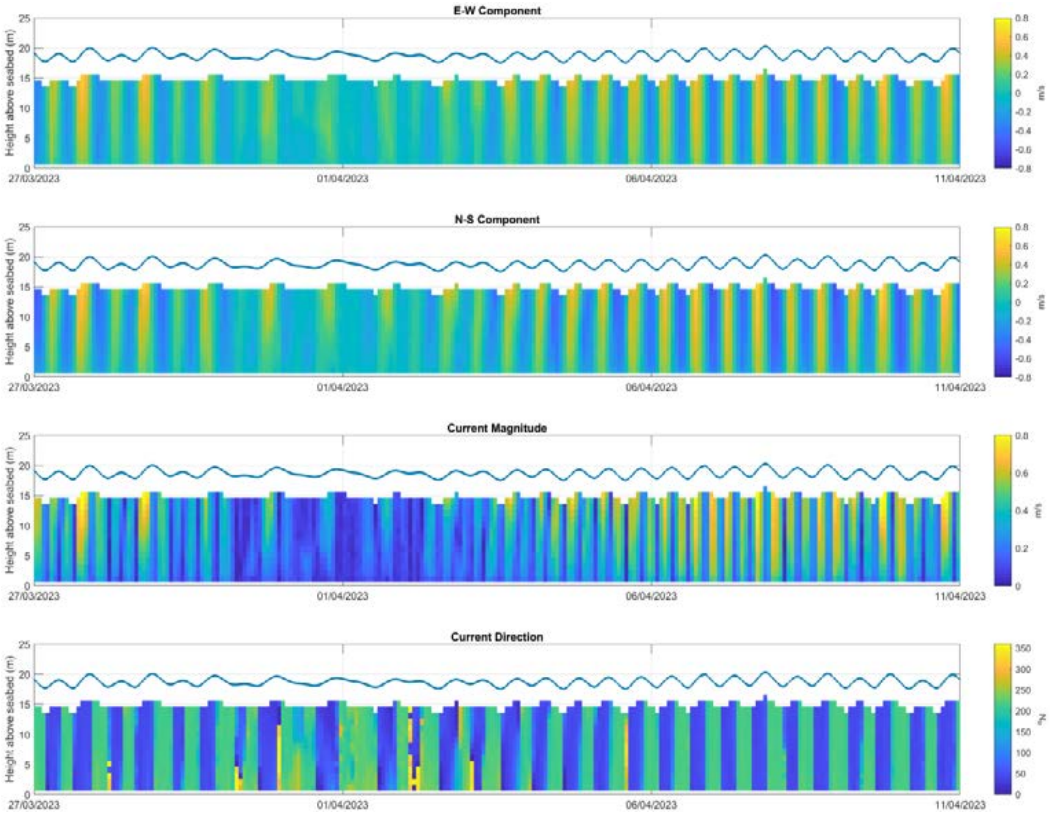


Figure A-55 Currents measured by the ADCP at SAMUSG – 27/03 to 11/04/2023

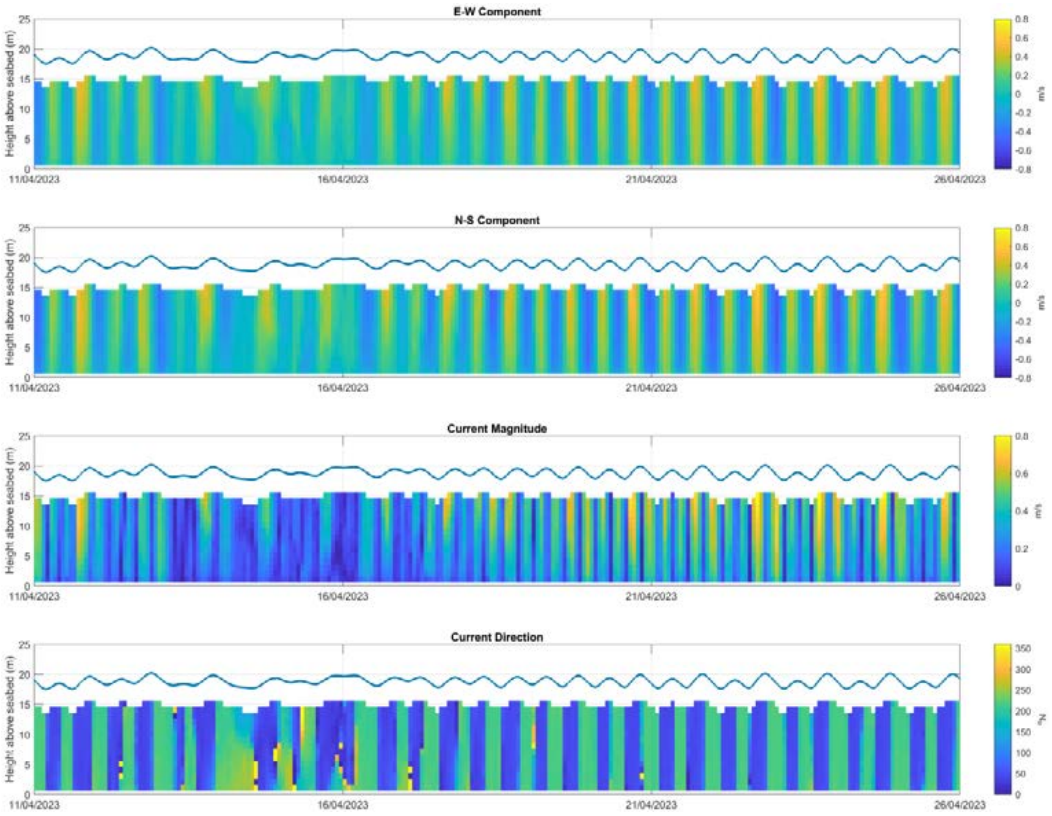


Figure A-56 Currents measured by the ADCP at SAMUSG – 11/04 to 26/04/2023

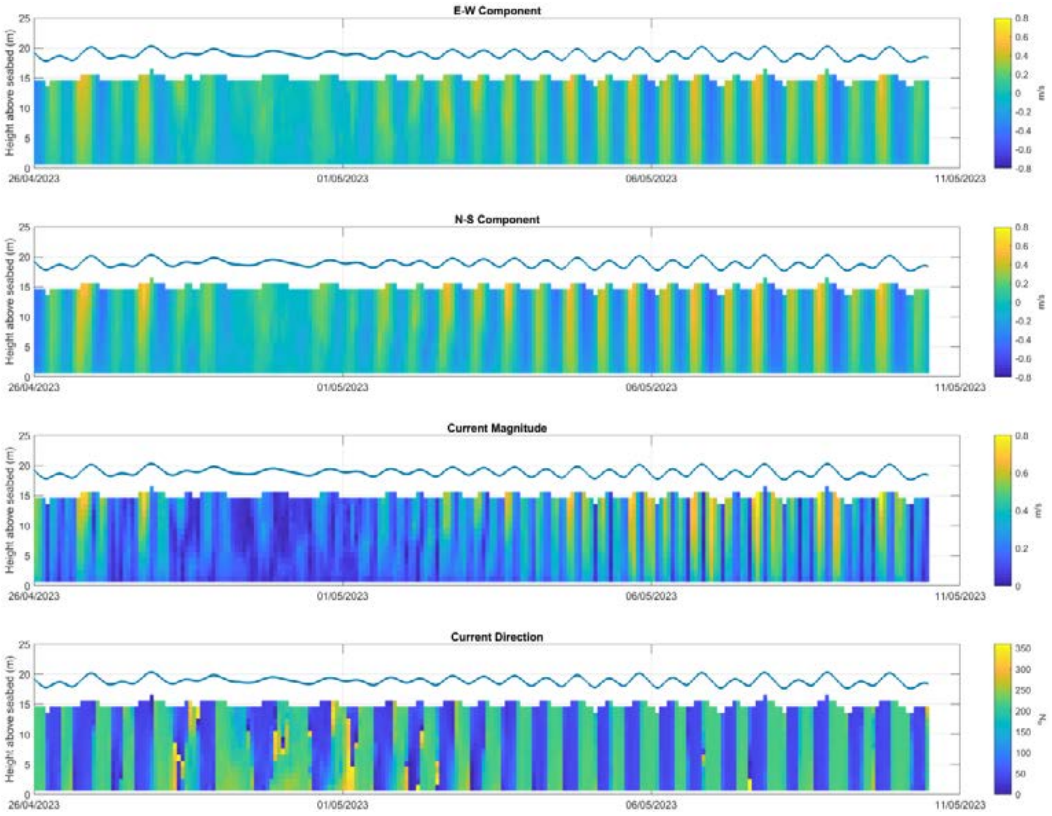


Figure A-57 Currents measured by the ADCP at SAMUSG – 26/04 to 11/05/2023

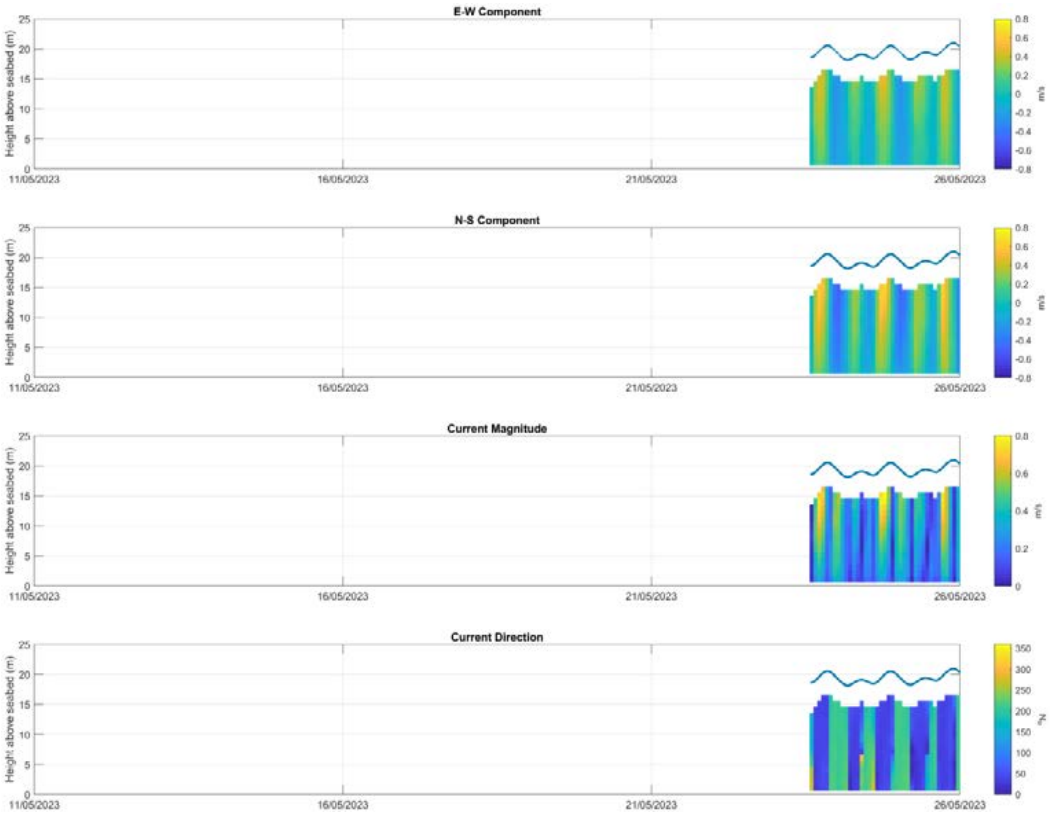


Figure A-58 Currents measured by the ADCP at SAMUSG – 11/05 to 26/05/2023

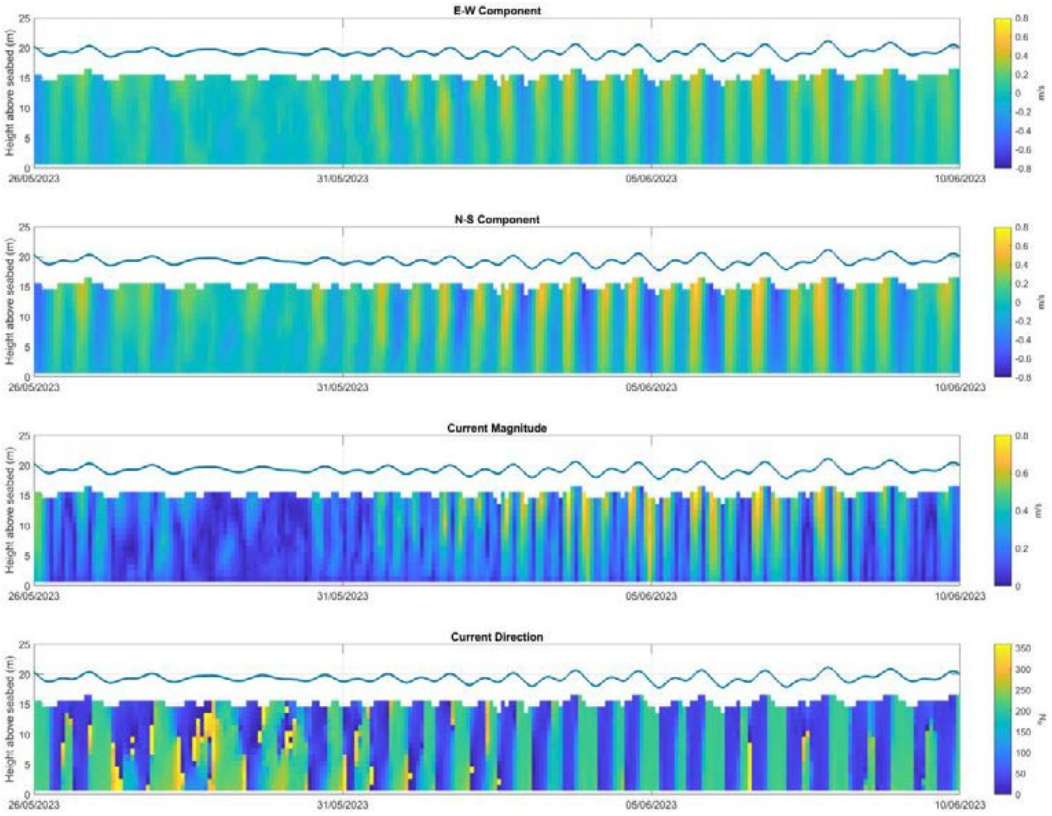


Figure A-59 Currents measured by the ADCP at SAMUSG – 26/05 to 10/06/2023

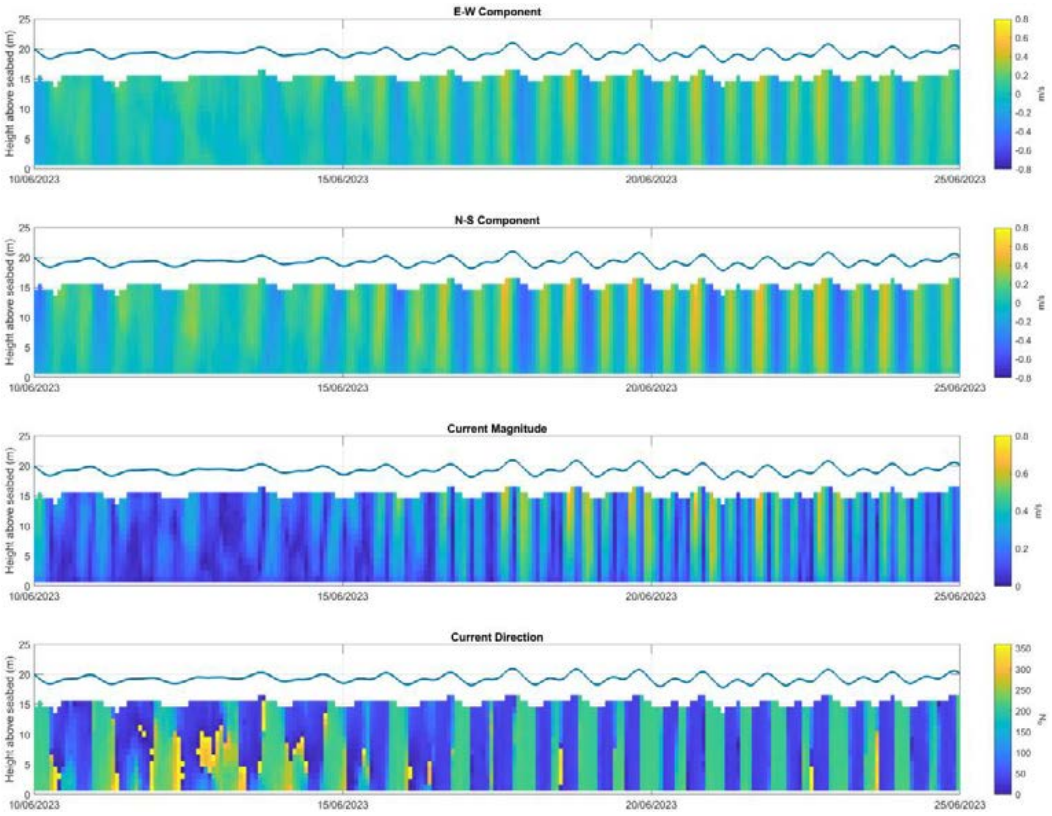


Figure A-60 Currents measured by the ADCP at SAMUSG – 10/06 to 25/06/2023

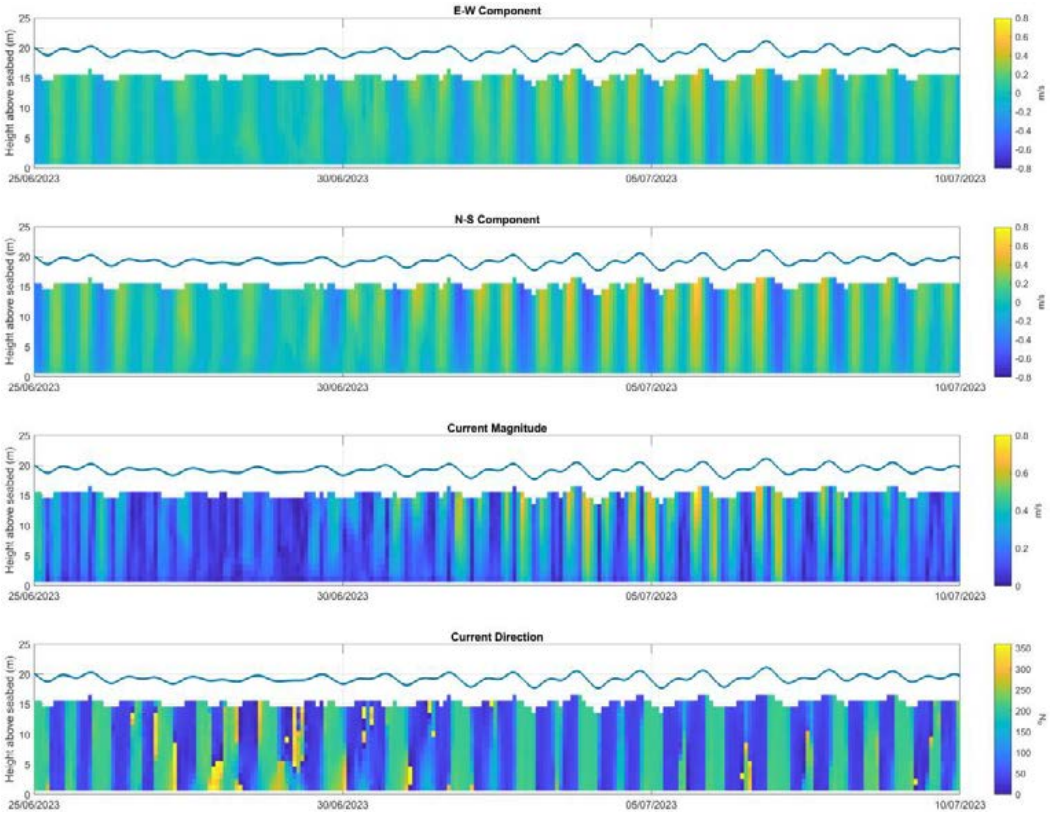


Figure A-61 Currents measured by the ADCP at SAMUSG – 25/06 to 10/07/2023

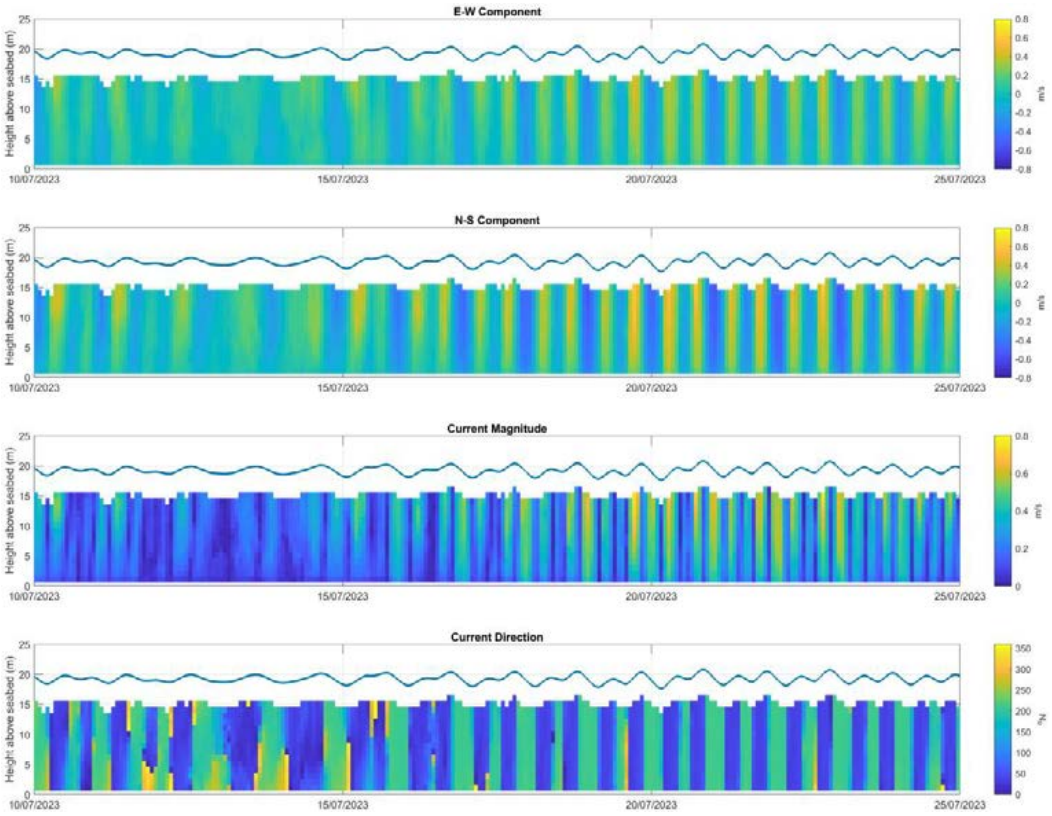


Figure A-62 Currents measured by the ADCP at SAMUSG – 10/07 to 25/07/2023

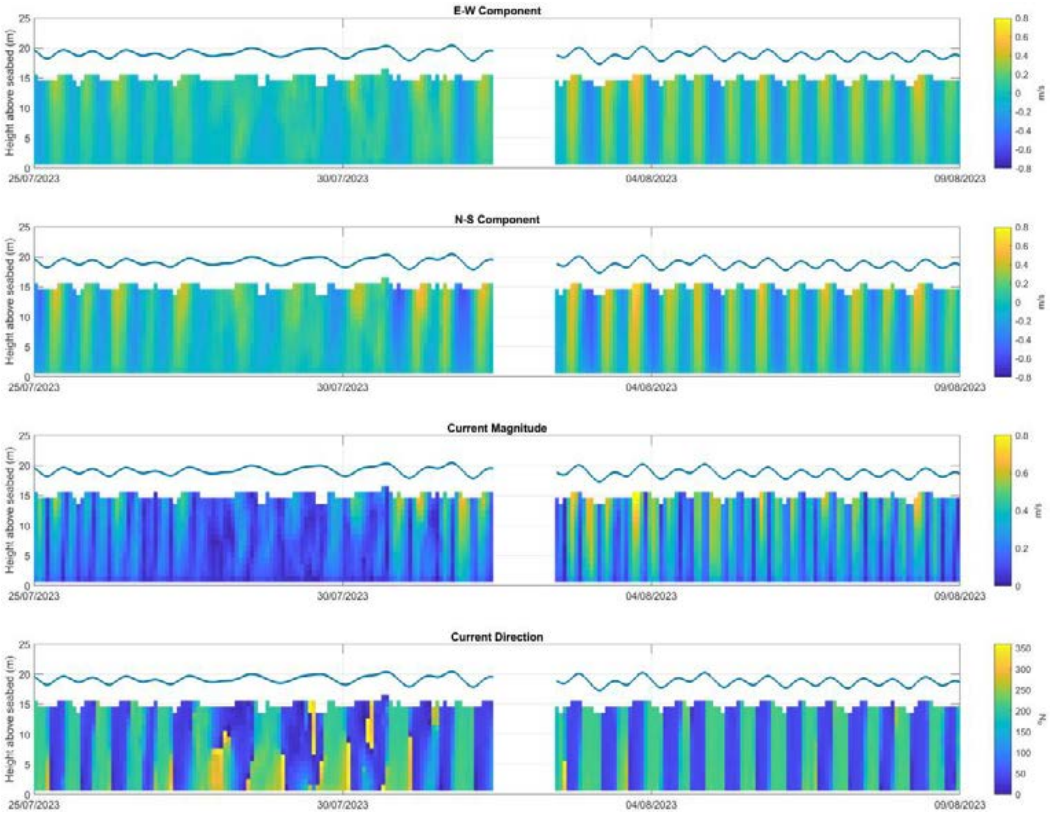


Figure A-63 Currents measured by the ADCP at SAMUSG – 25/07 to 09/08/2023

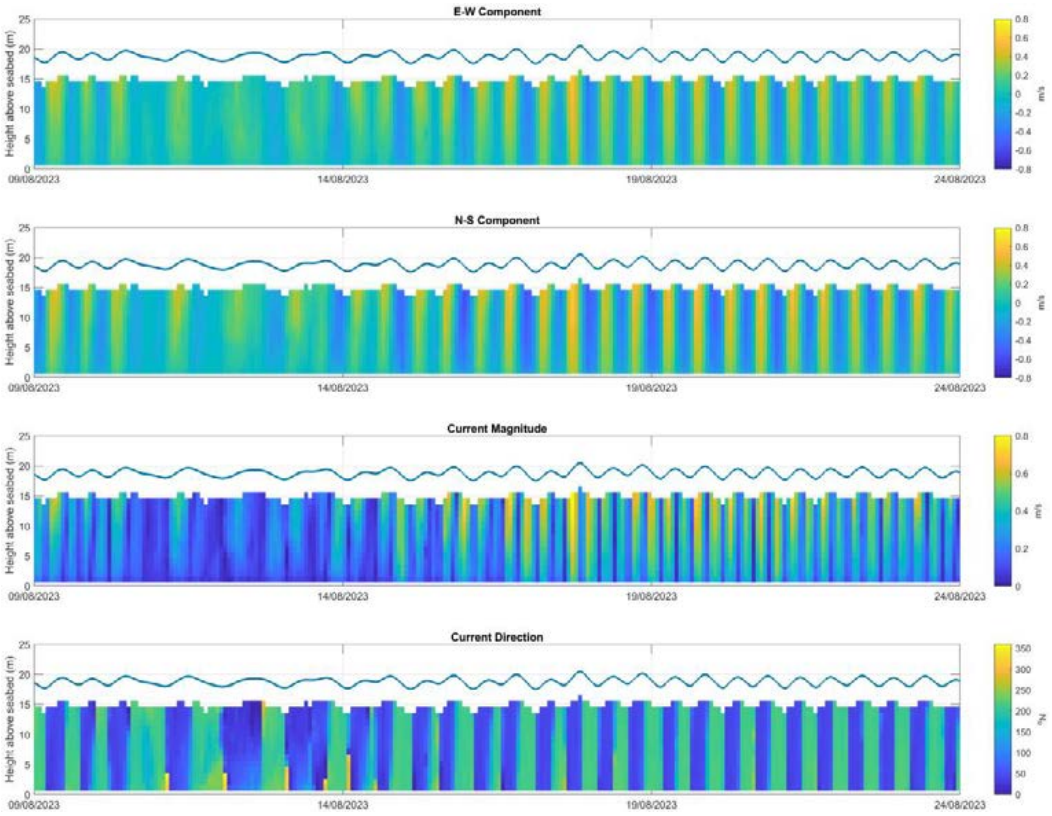


Figure A-64 Currents measured by the ADCP at SAMUSG – 09/08 to 24/08/2023

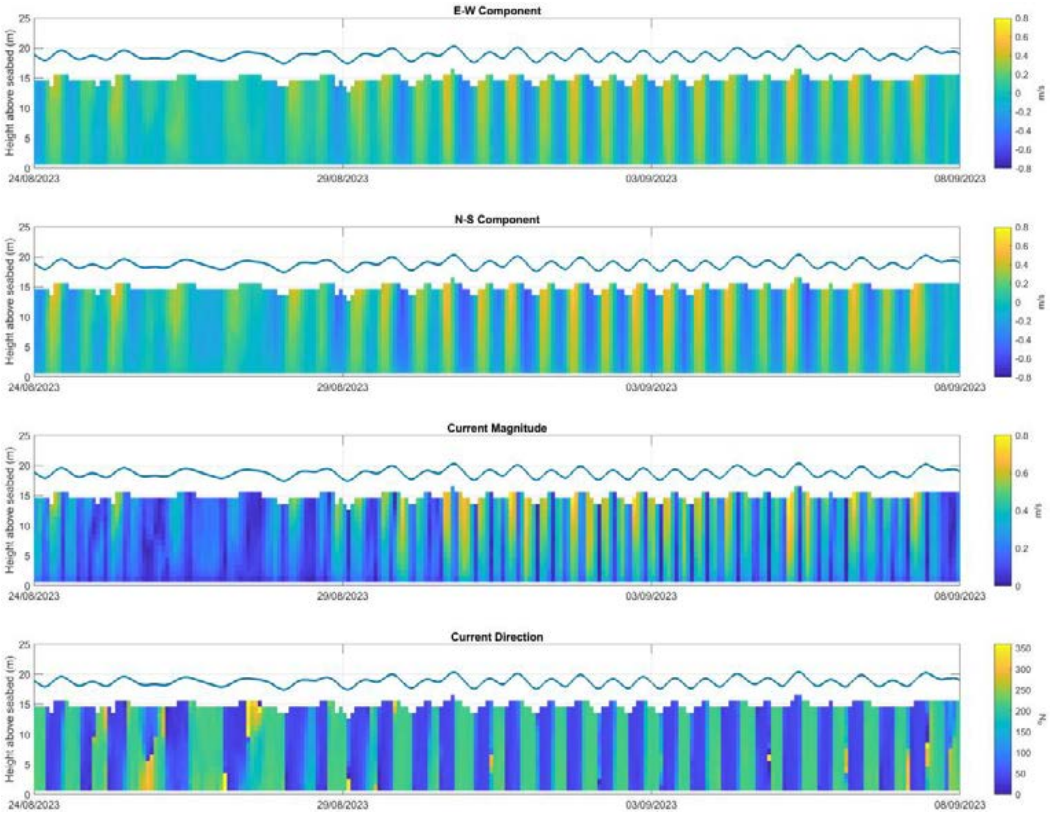


Figure A-65 Currents measured by the ADCP at SAMUSG – 24/08 to 08/09/2023

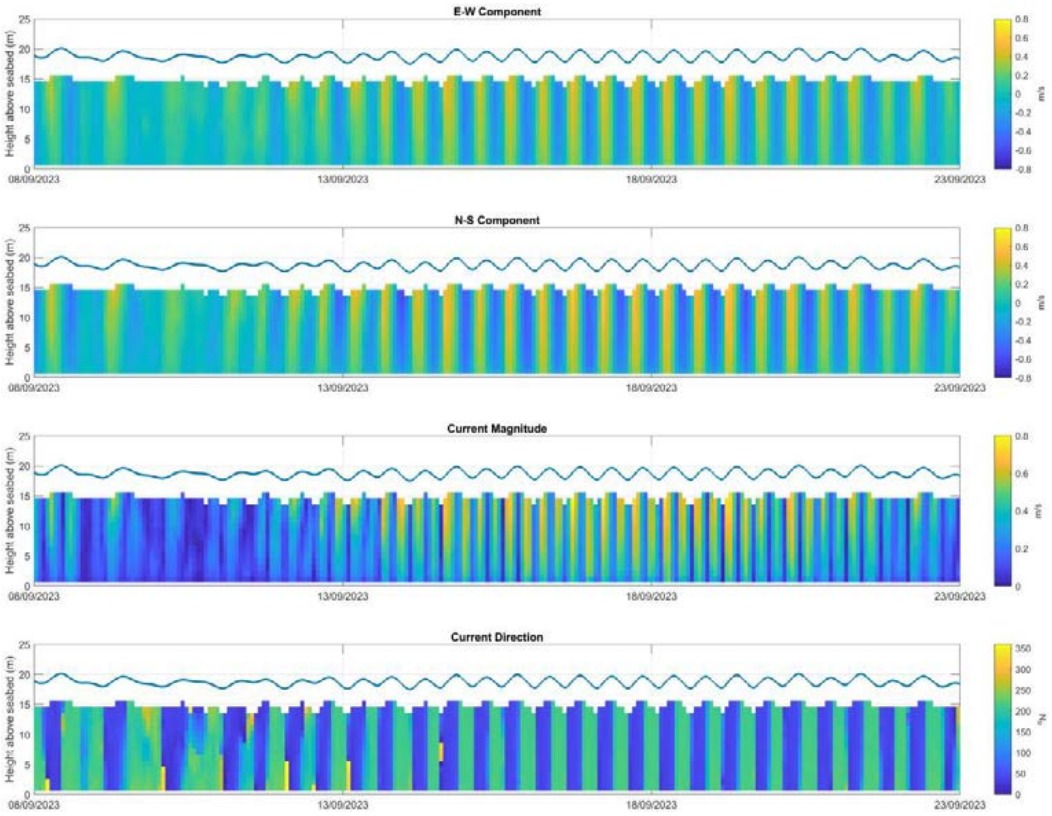


Figure A-66 Currents measured by the ADCP at SAMUSG – 08/09 to 23/09/2023

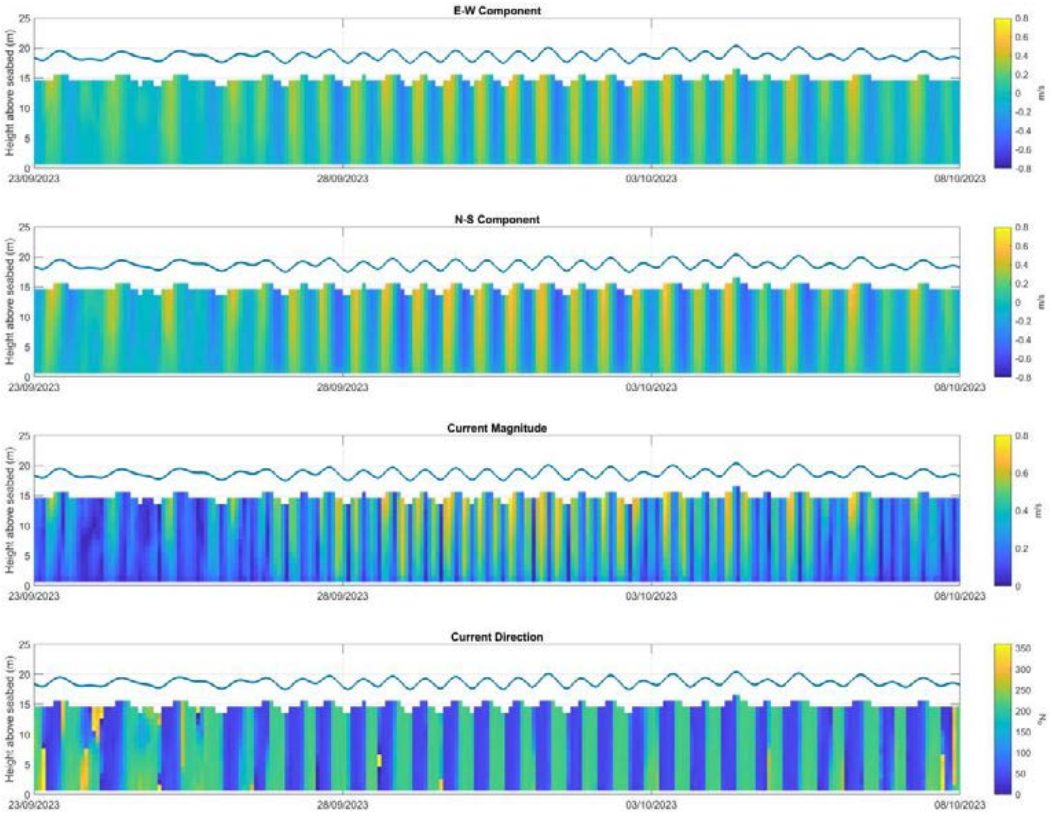


Figure A-67 Currents measured by the ADCP at SAMUSG – 23/09 to 08/10/2023

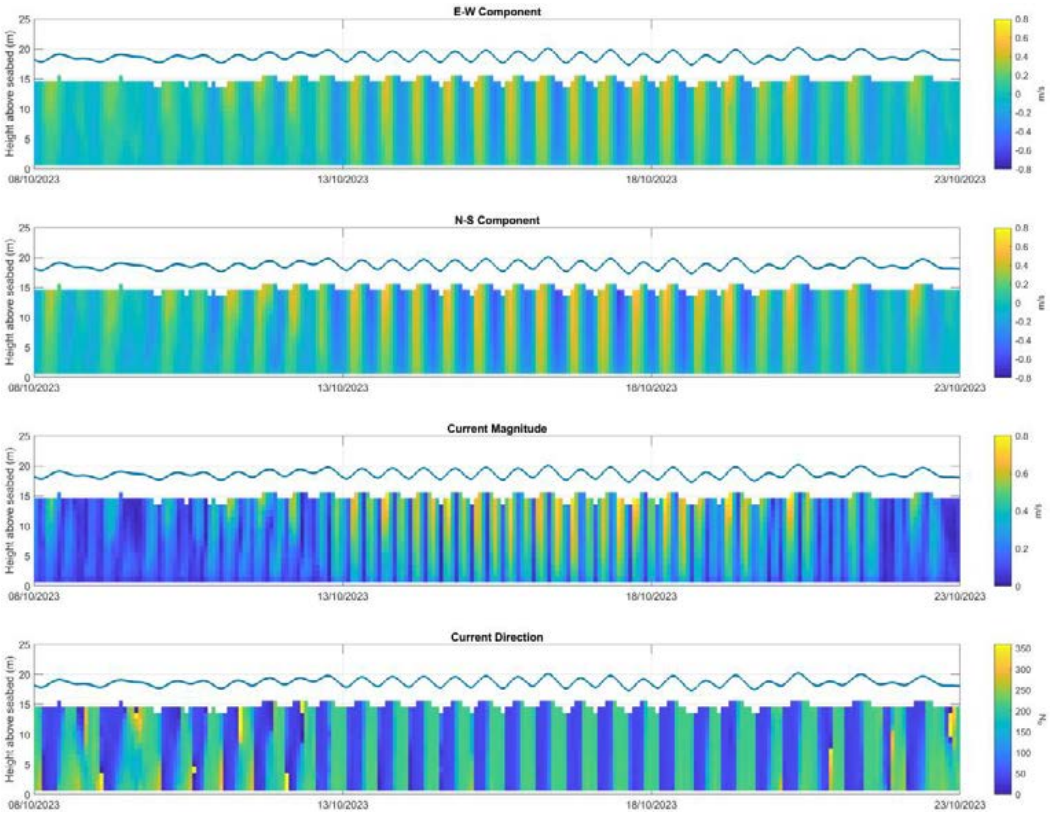


Figure A-68 Currents measured by the ADCP at SAMUSG – 08/10 to 23/10/2023

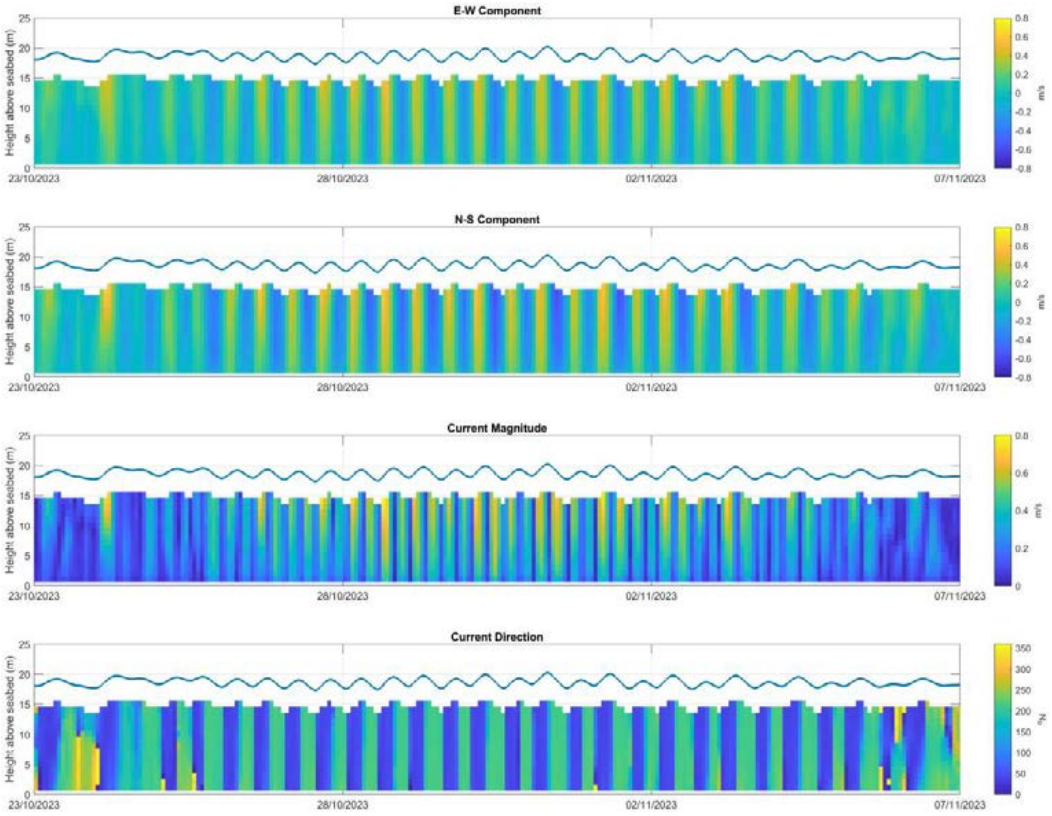


Figure A-69 Currents measured by the ADCP at SAMUSG – 23/10 to 07/11/2023

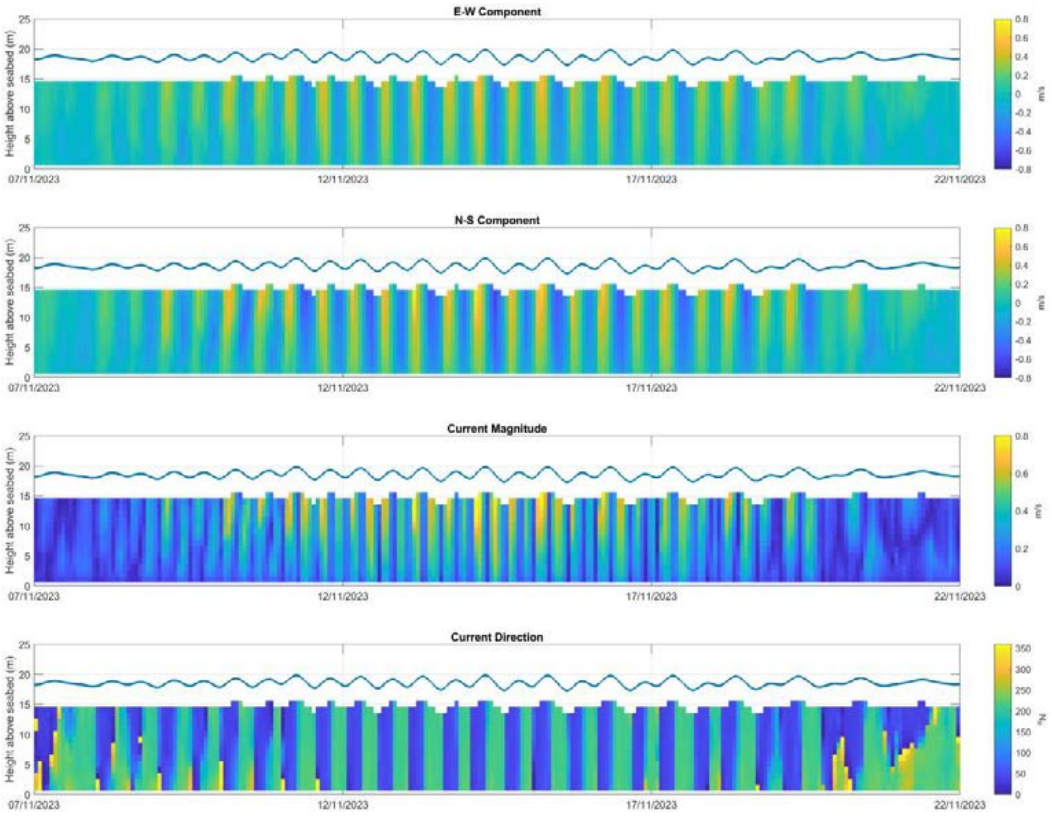


Figure A-70 Currents measured by the ADCP at SAMUSG – 07/11 to 22/11/2023

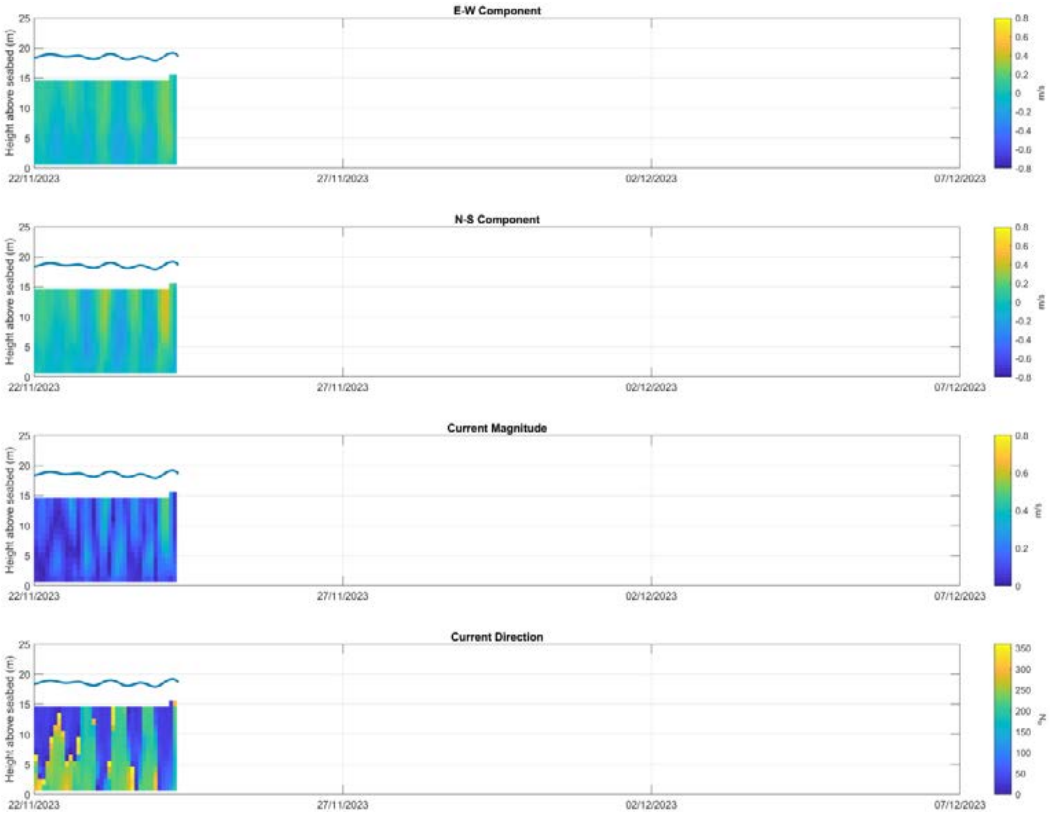


Figure A-71 Currents measured by the ADCP at SAMUSG – 22/11 to 07/12/2023

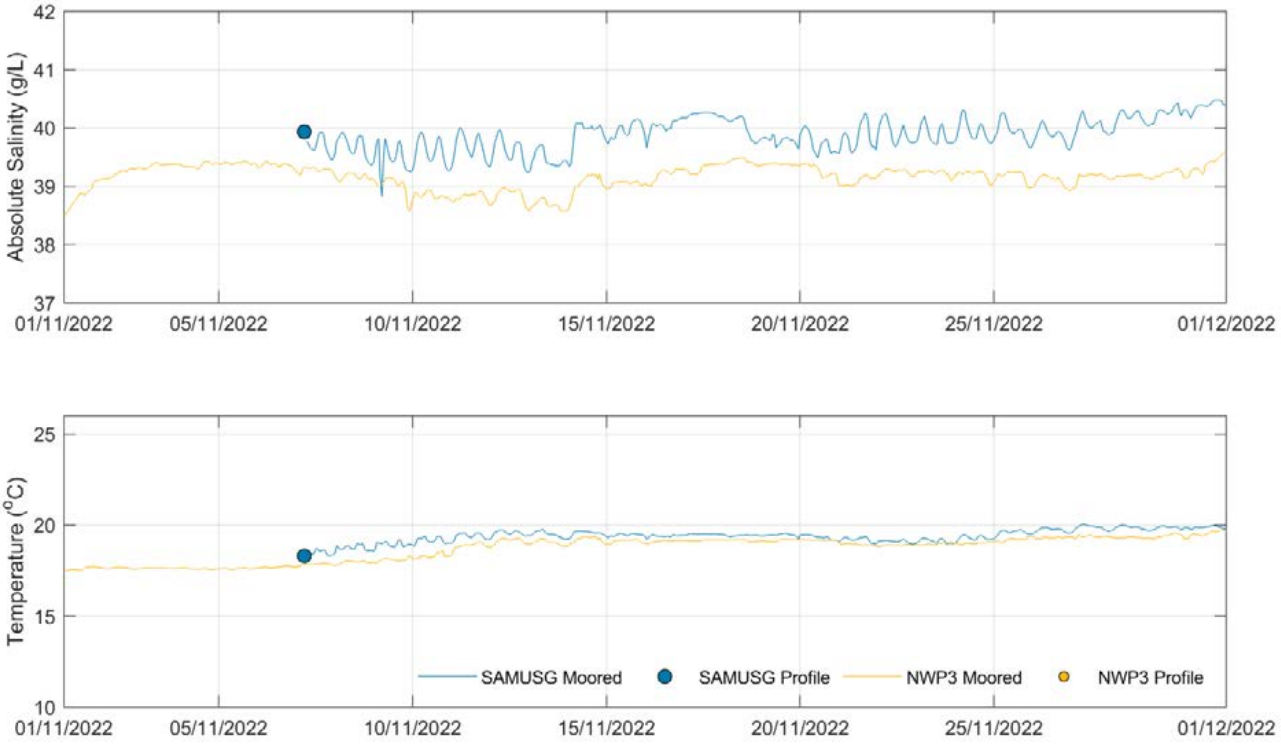


Figure A-72 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 11/2022

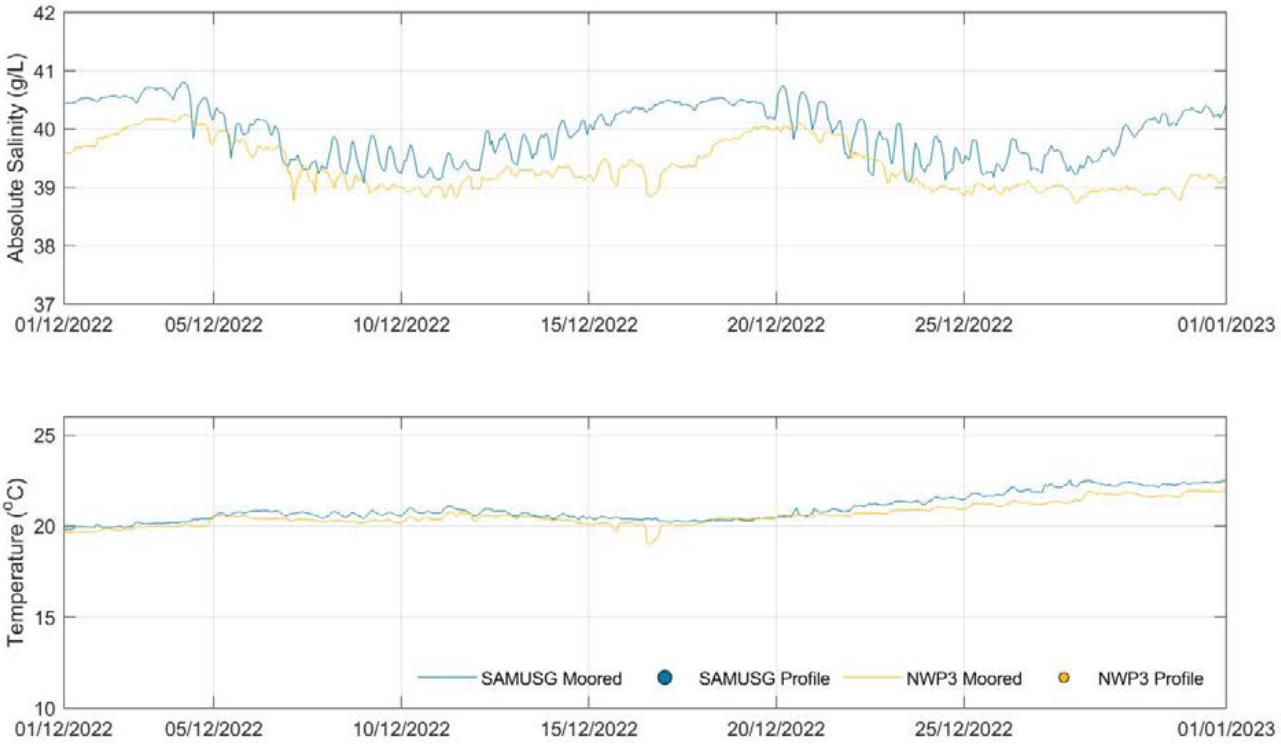


Figure A-73 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 12/2022

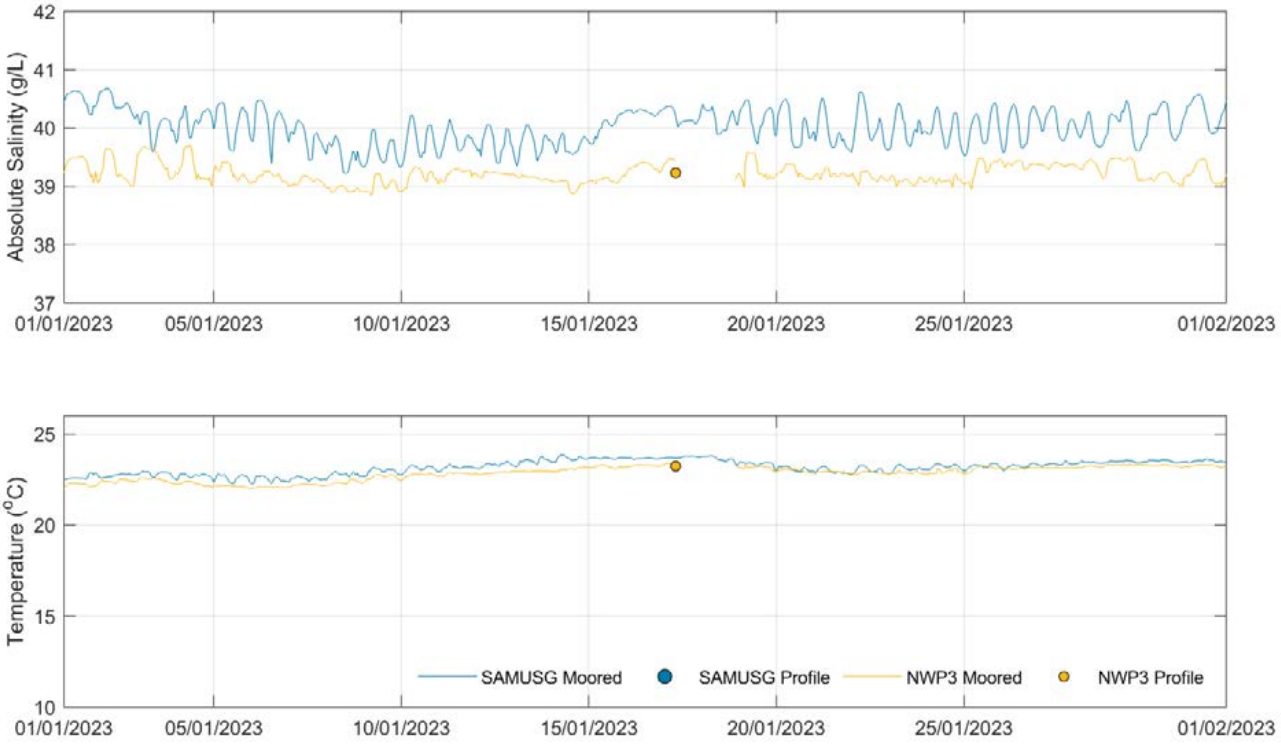


Figure A-74 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 01/2023

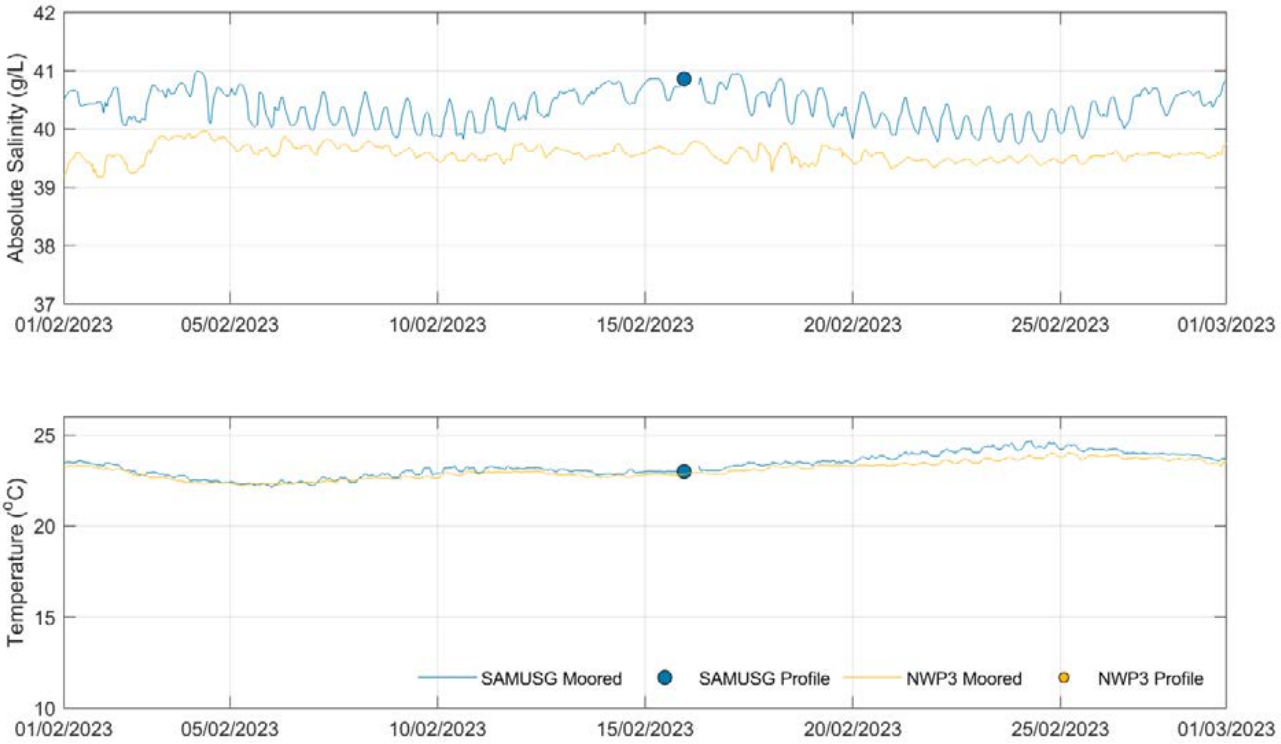


Figure A-75 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 02/2023

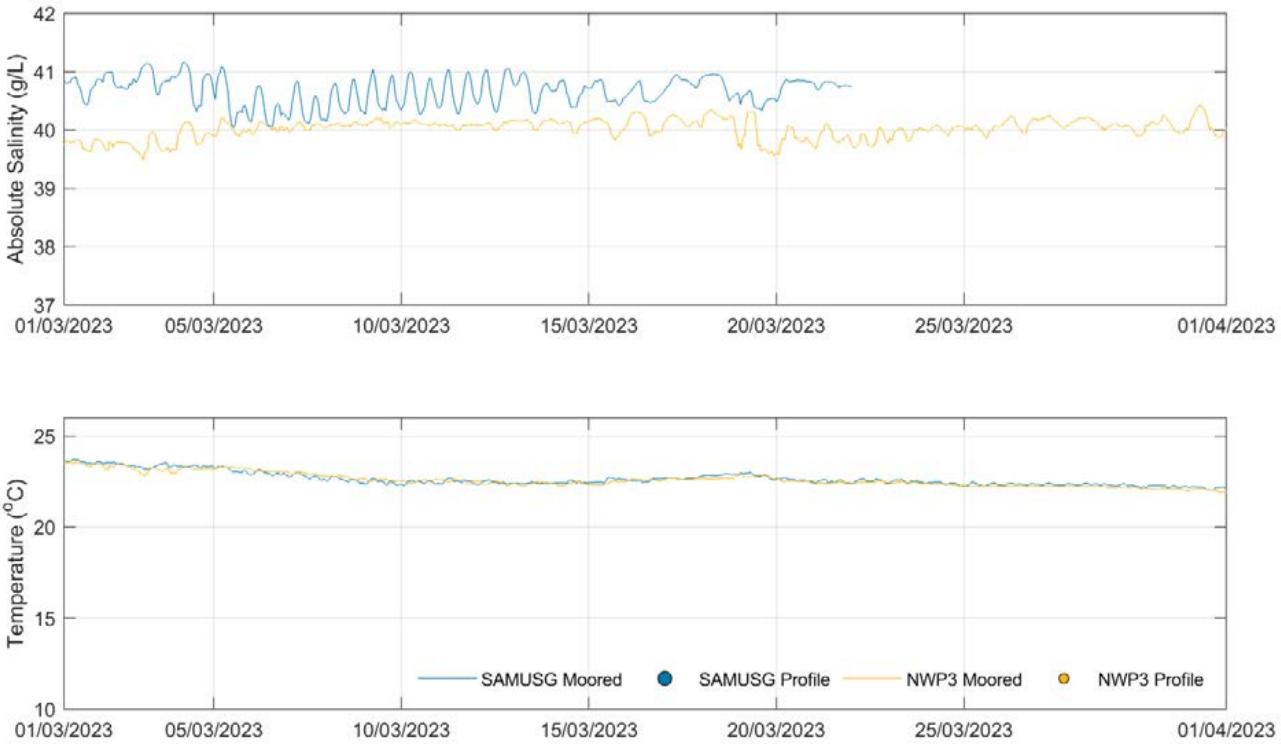


Figure A-76 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 03/2023

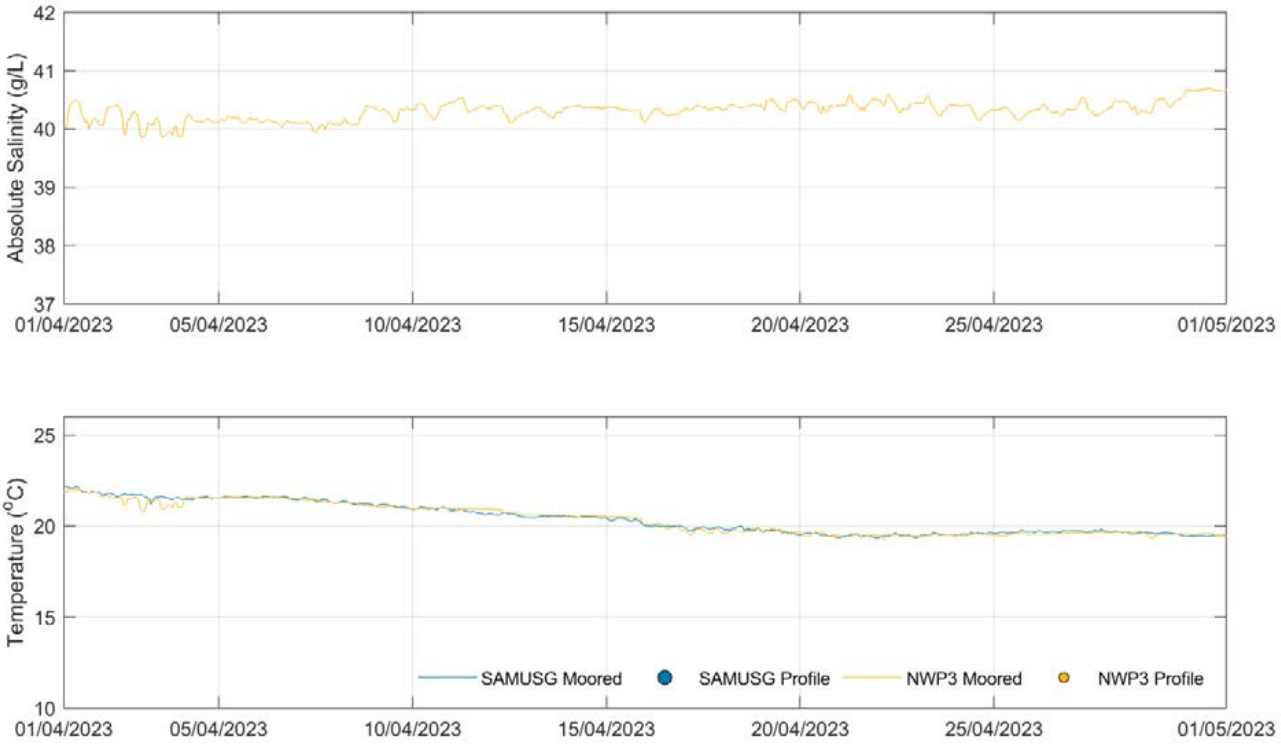


Figure A-77 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 04/2023

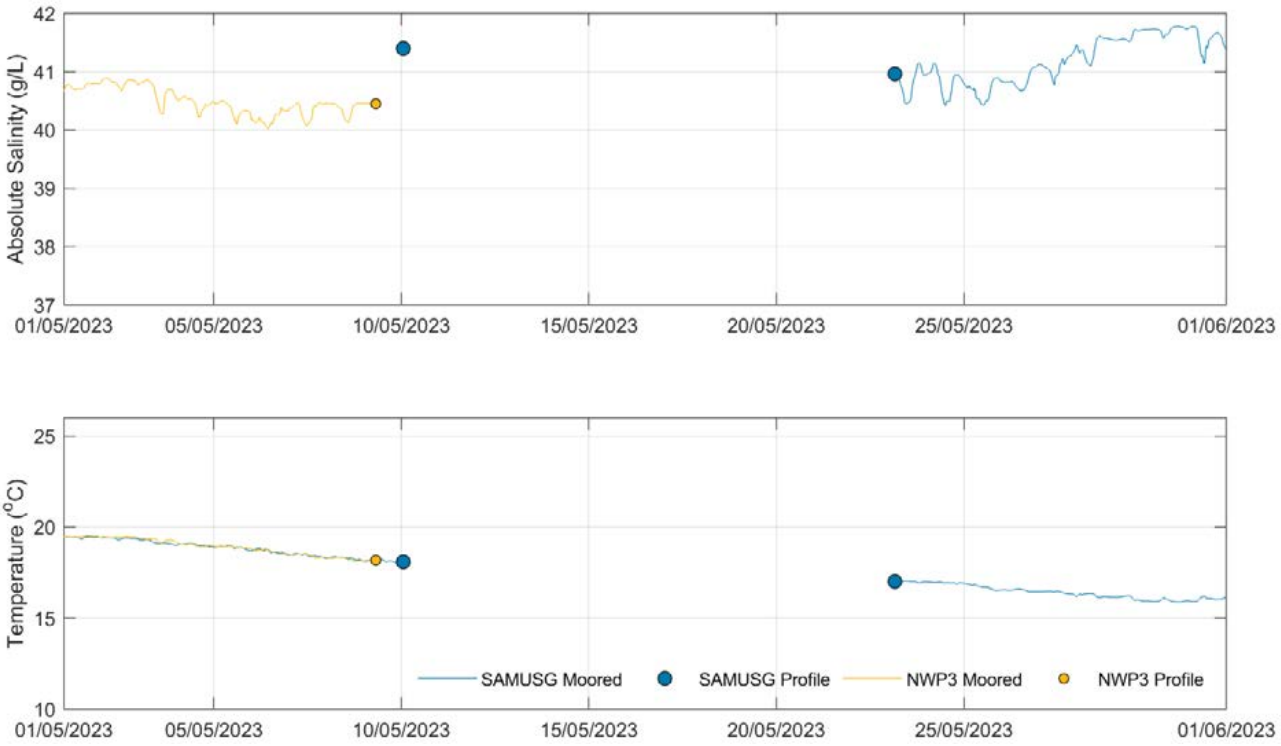


Figure A-78 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 05/2023

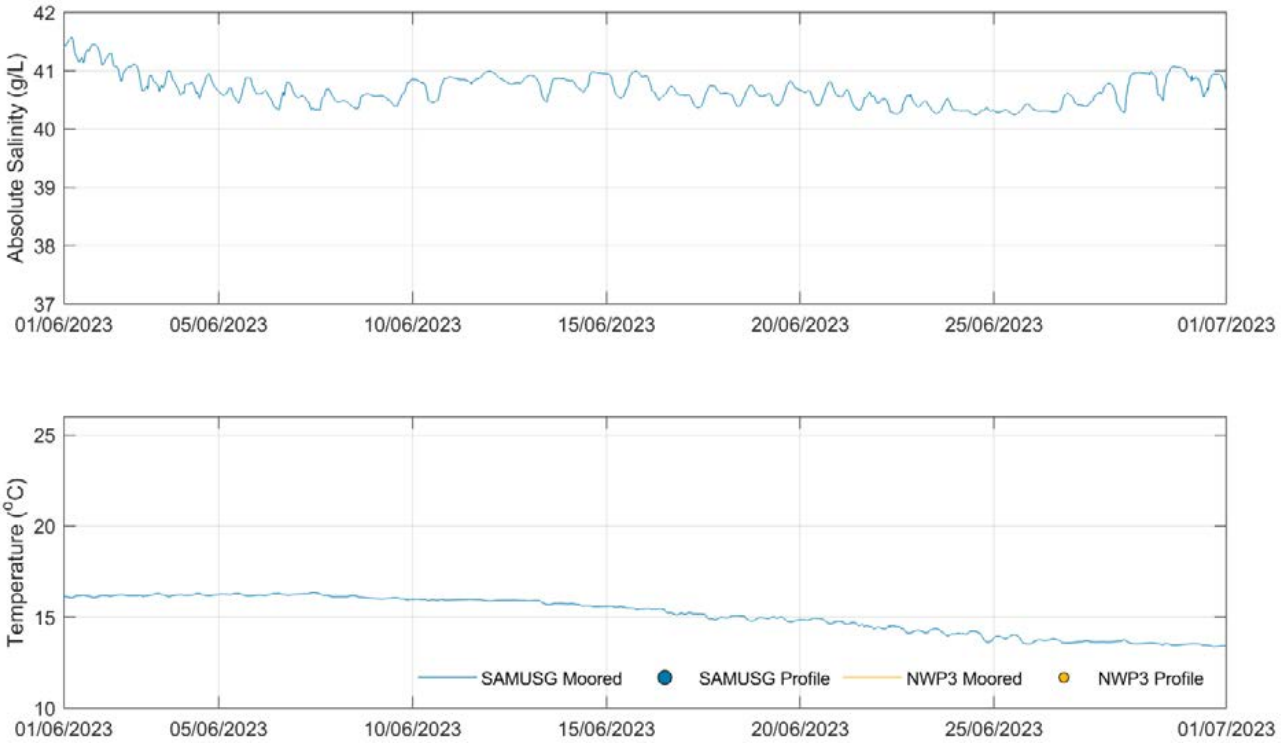


Figure A-79 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 06/2023

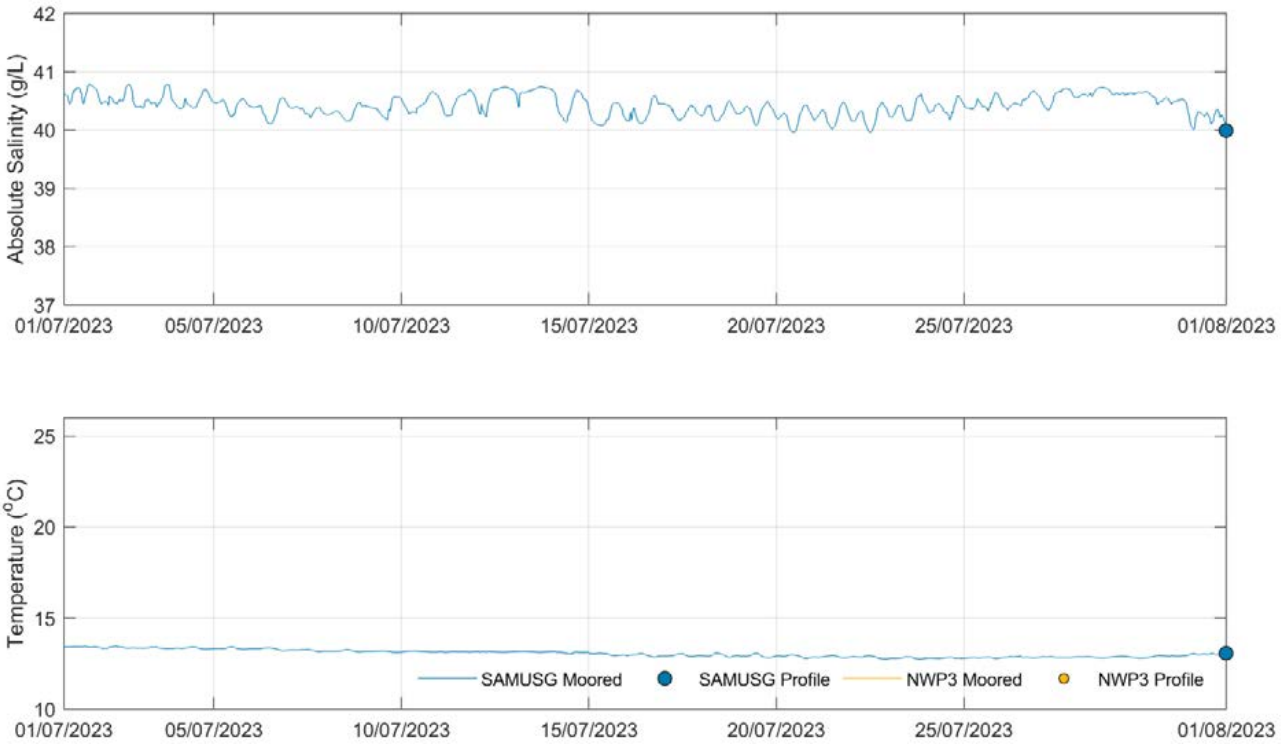


Figure A-80 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 07/2023

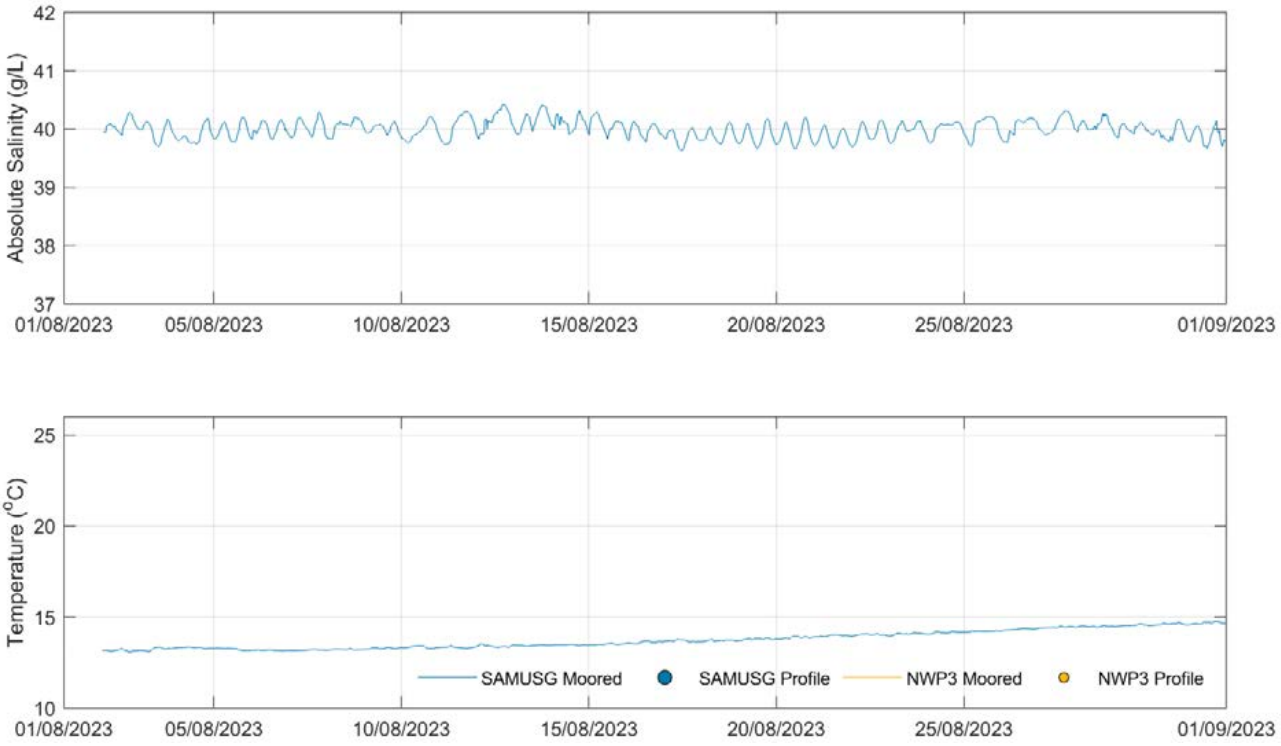


Figure A-81 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 08/2023

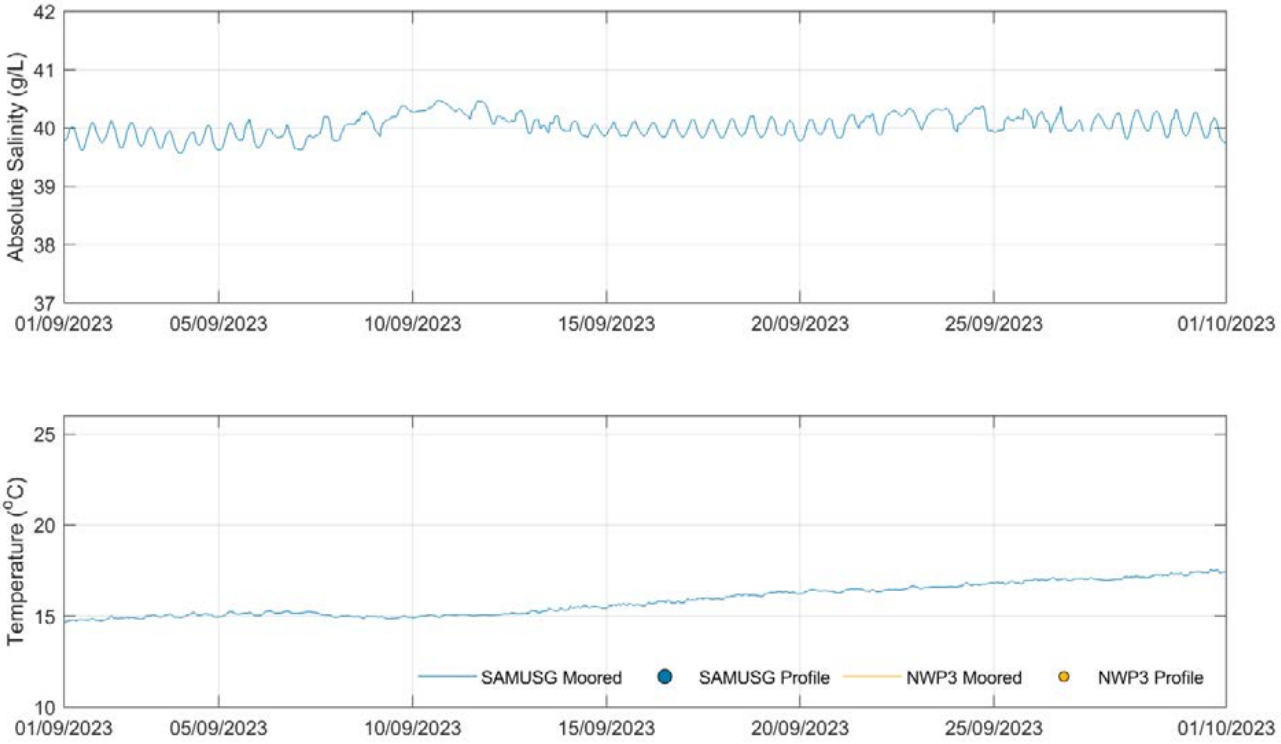


Figure A-82 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 09/2023

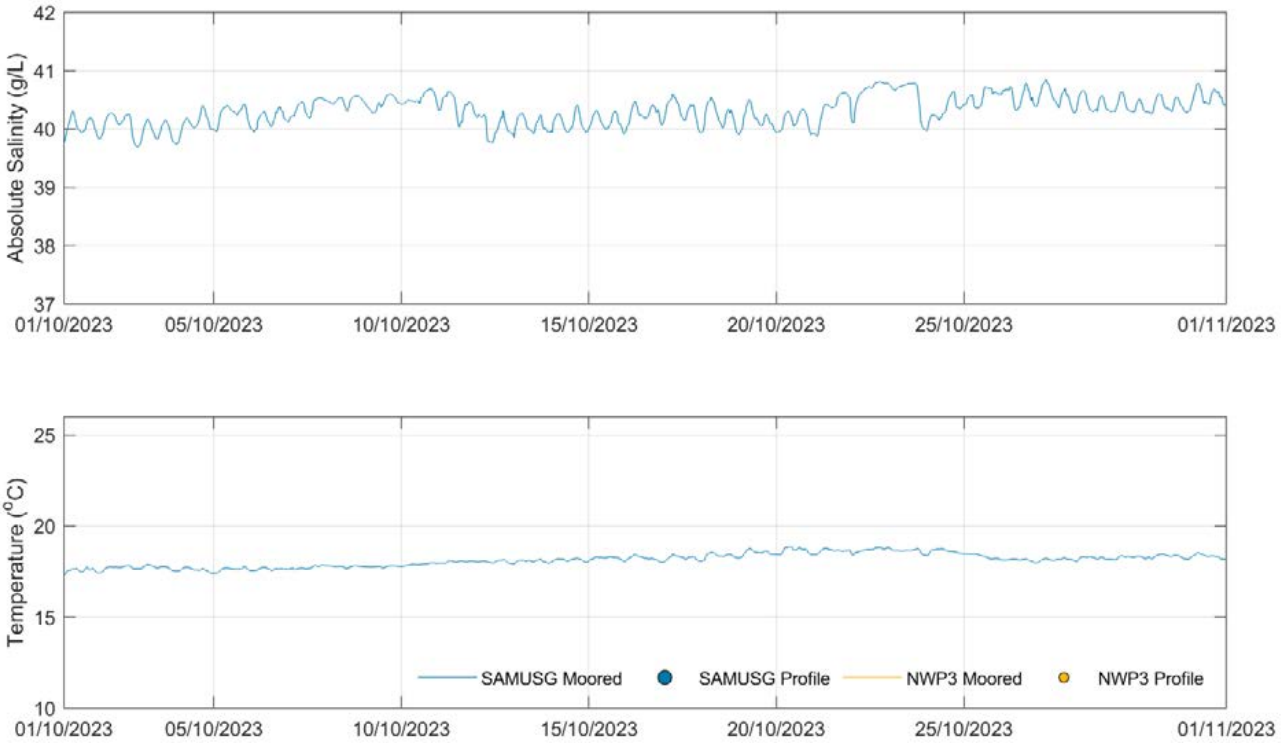


Figure A-83 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 10/2023

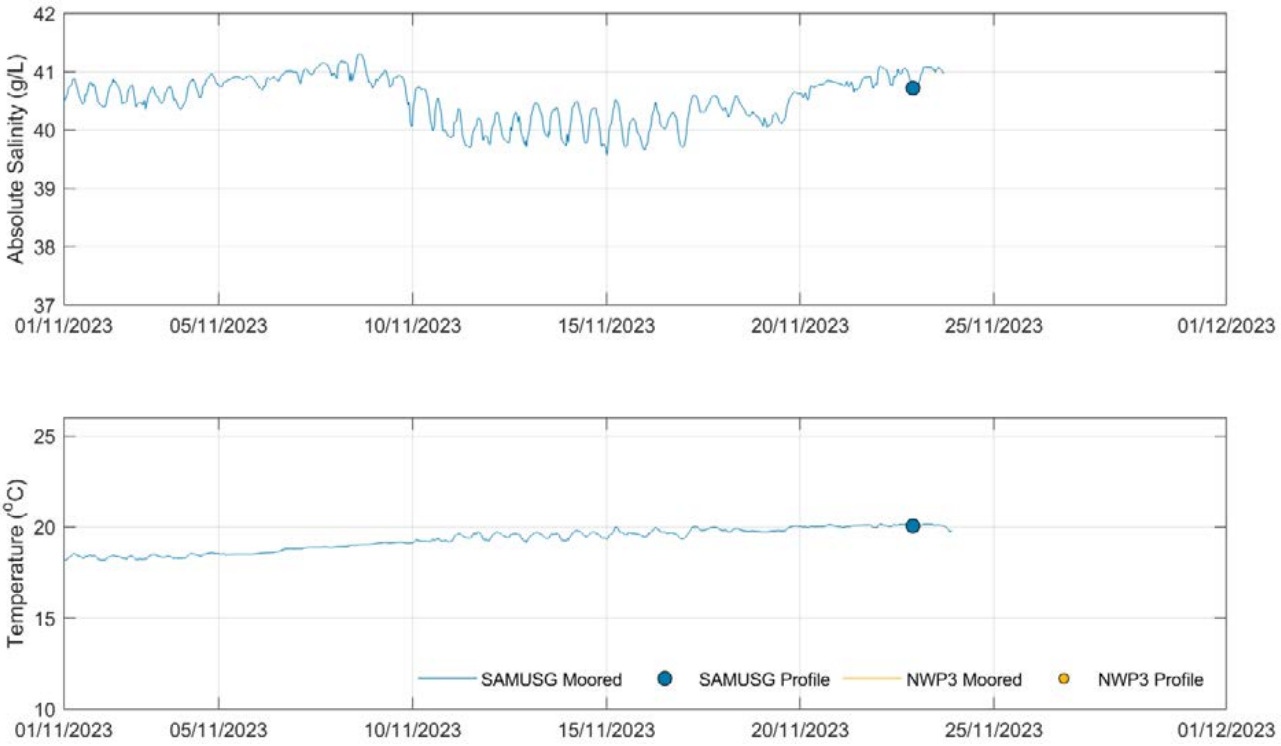
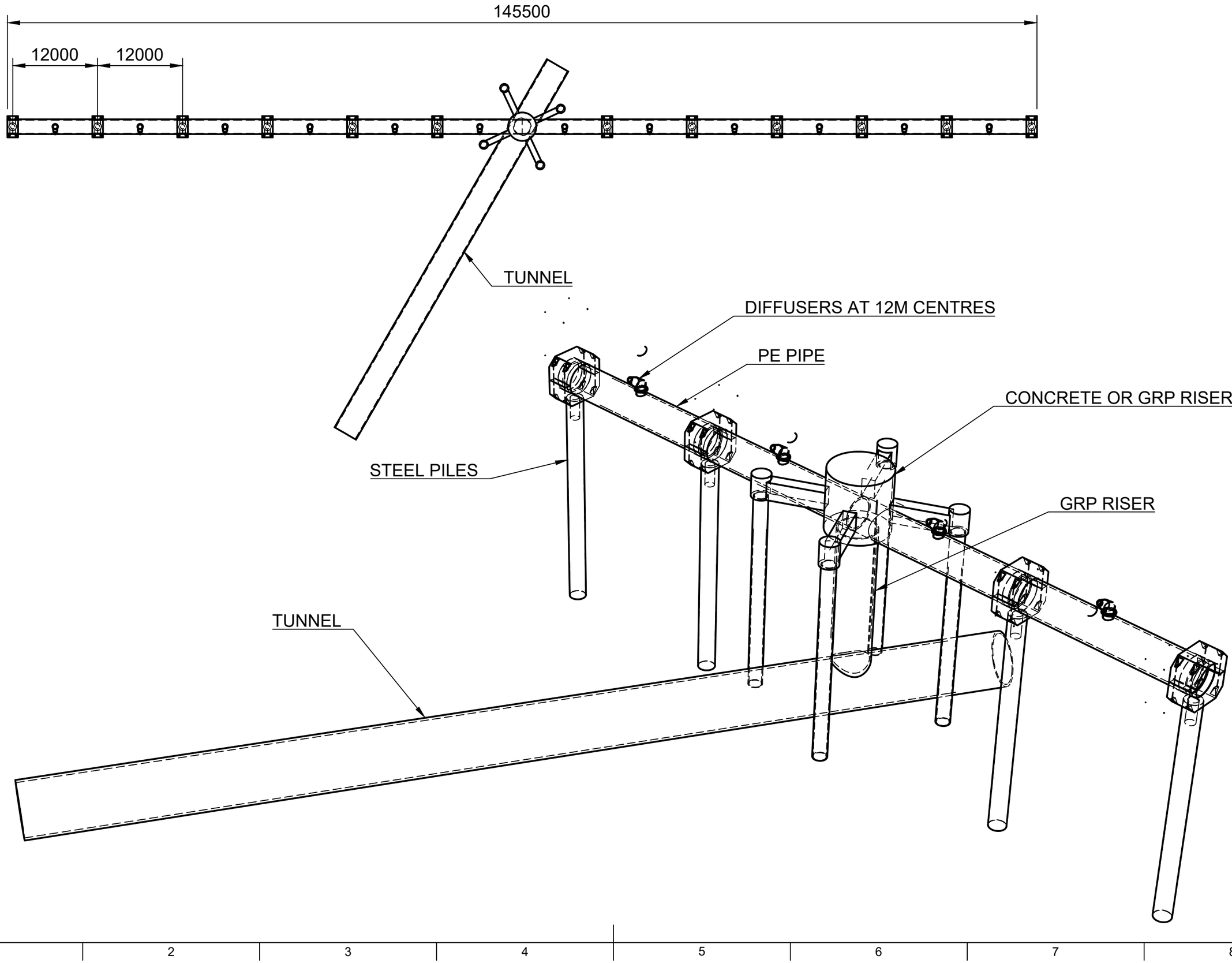


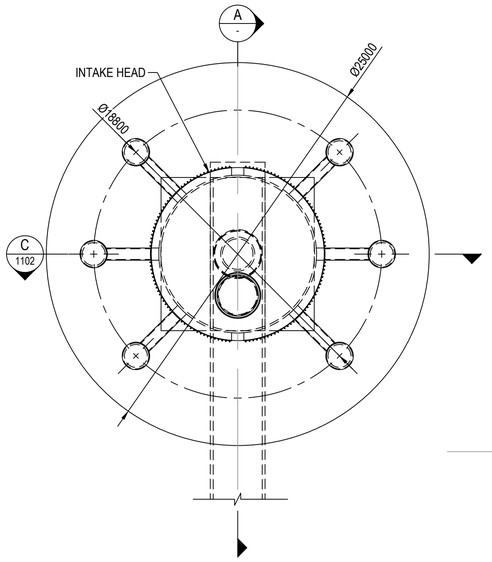
Figure A-84 Measured Salinity and Temperature at NWP3 and SAMUSG Stations – 11/2023

WATER TECHNOLOGY
WATER, COASTAL & ENVIRONMENTAL CONSULTANTS

APPENDIX B
OUTFALL AND INTAKE DRAWINGS

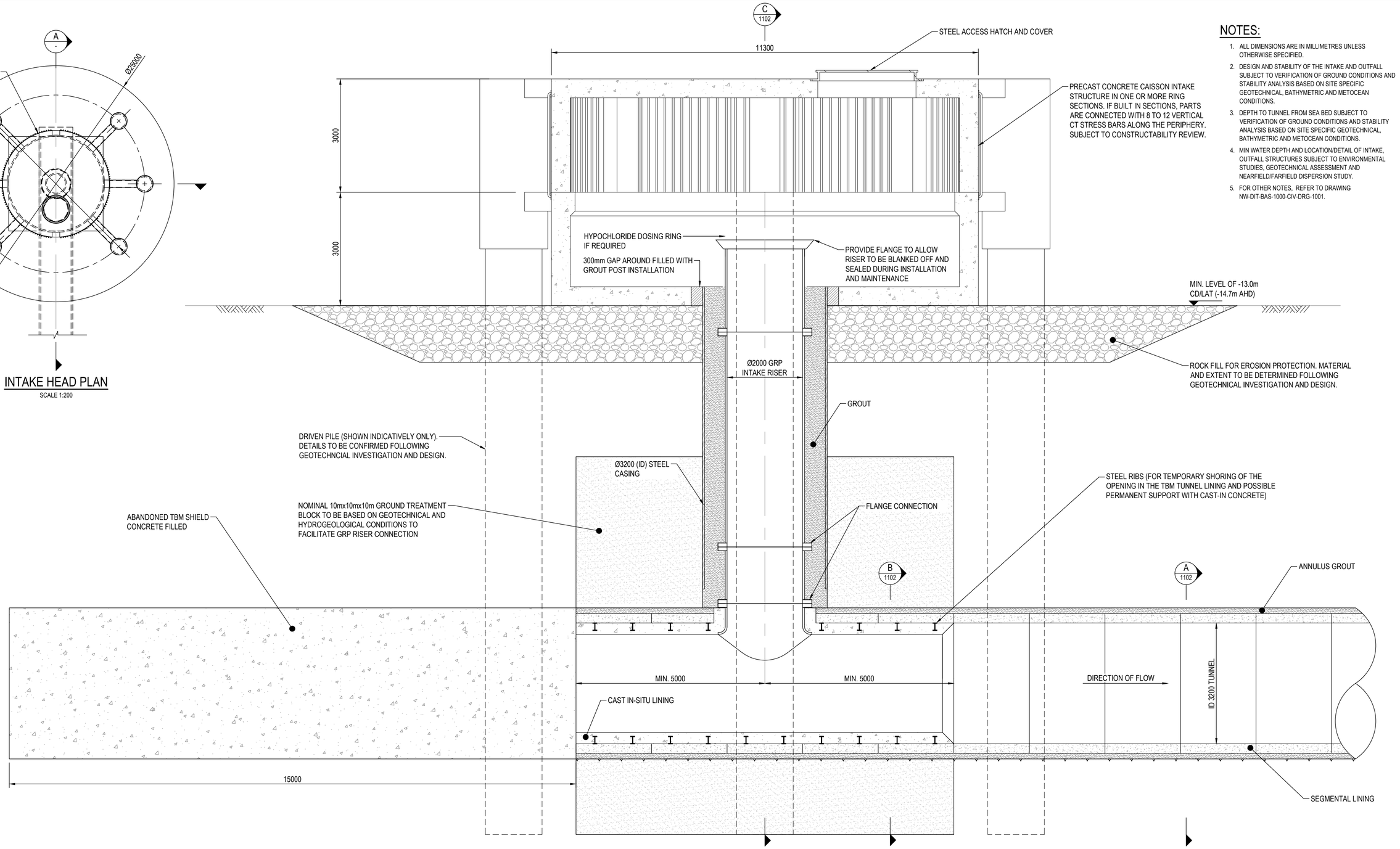



145500
12000
12000
TUNNEL
DIFFUSERS AT 12M CENTRES
PE PIPE
CONCRETE OR GRP RISER CAP
STEEL PILES
GRP RISER
TUNNEL



INTAKE HEAD PLAN

SCALE 1:200



A SECTION

SCALE 1:50

INTAKE RISER AND HEAD DETAILS

NOTES:

1. ALL DIMENSIONS ARE IN MILLIMETRES UNLESS OTHERWISE SPECIFIED.
2. DESIGN AND STABILITY OF THE INTAKE AND OUTFALL SUBJECT TO VERIFICATION OF GROUND CONDITIONS AND STABILITY ANALYSIS BASED ON SITE SPECIFIC GEOTECHNICAL, BATHYMETRIC AND METOCEAN CONDITIONS.
3. DEPTH TO TUNNEL FROM SEA BED SUBJECT TO VERIFICATION OF GROUND CONDITIONS AND STABILITY ANALYSIS BASED ON SITE SPECIFIC GEOTECHNICAL, BATHYMETRIC AND METOCEAN CONDITIONS.
4. MIN WATER DEPTH AND LOCATION/DETAIL OF INTAKE, OUTFALL STRUCTURES SUBJECT TO ENVIRONMENTAL STUDIES, GEOTECHNICAL ASSESSMENT AND NEARFIELD/FARFIELD DISPERSION STUDY.
5. FOR OTHER NOTES, REFER TO DRAWING NW-DIT-BAS-1000-CIV-DRG-1001.

FOR INFORMATION ONLY

NOT FOR CONSTRUCTION

SCALES: 1 0 0.5 1 1.5 2 4 0 2 4 6 8

No.	AMENDMENT DESCRIPTION	BY	DATE
B	FOR INFORMATION ONLY	AC	30.06.25
A	DRAFT FOR COMMENT	NA	16.05.25

Government of South Australia

Department for Infrastructure and Transport

REFERENCE DESIGN

NORTHERN WATER

MARINE WORKS

INTAKE RISER DETAILS

DESIGNED	DRAFTED	CHECKED	PATCHS No	SKETCH No.	AMEND No.:
IL/DO	NA	BH/SH	24C347	NW-DIT-BAS-1000-CIV-DRG-1101	B

ALL DIMENSIONS ARE IN METRES UNLESS SHOWN OTHERWISE ◄ 100 MILLIMETRES ON ORIGINAL DRAWING ► UNCONTROLLED COPY WHEN PRINTED

CAD FILE NAME: NW-DIT-BAS-1000-CIV-DRG-1101.dwg

WATER TECHNOLOGY
WATER, COASTAL & ENVIRONMENTAL CONSULTANTS

APPENDIX C
WATER LEVEL TIMESERIES COMPARISONS



C-1 Comparisons at NWP3

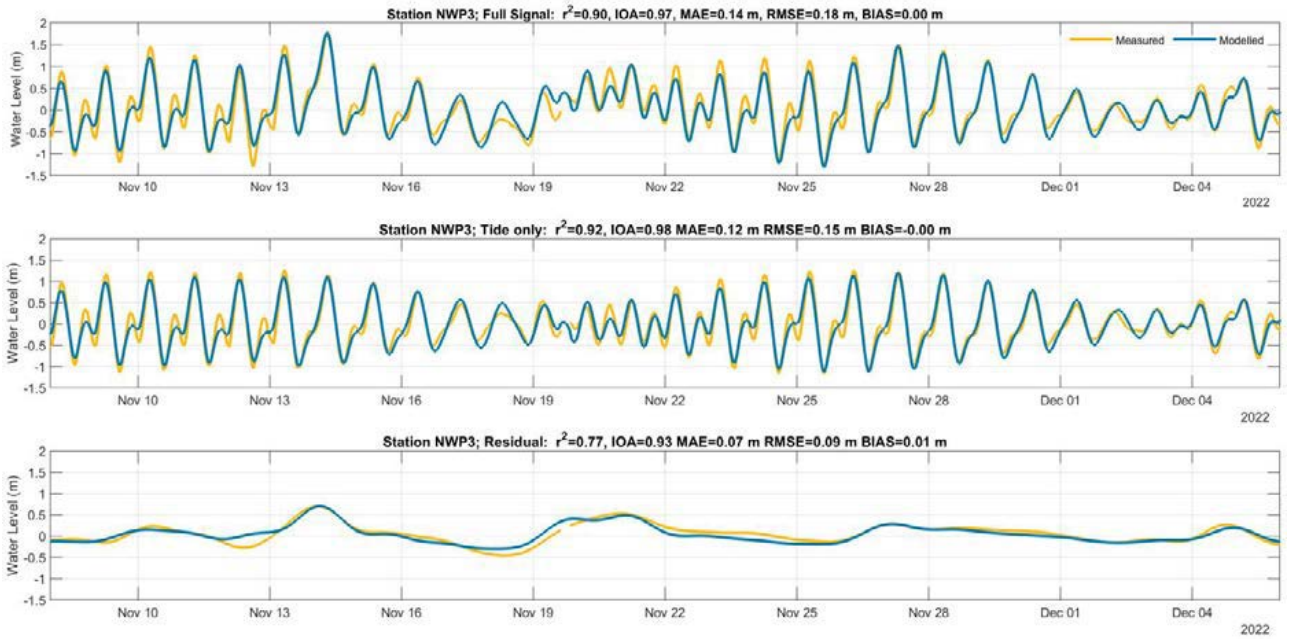


Figure C-1 Water Levels at NWP3 – 08/11 to 05/12/2022

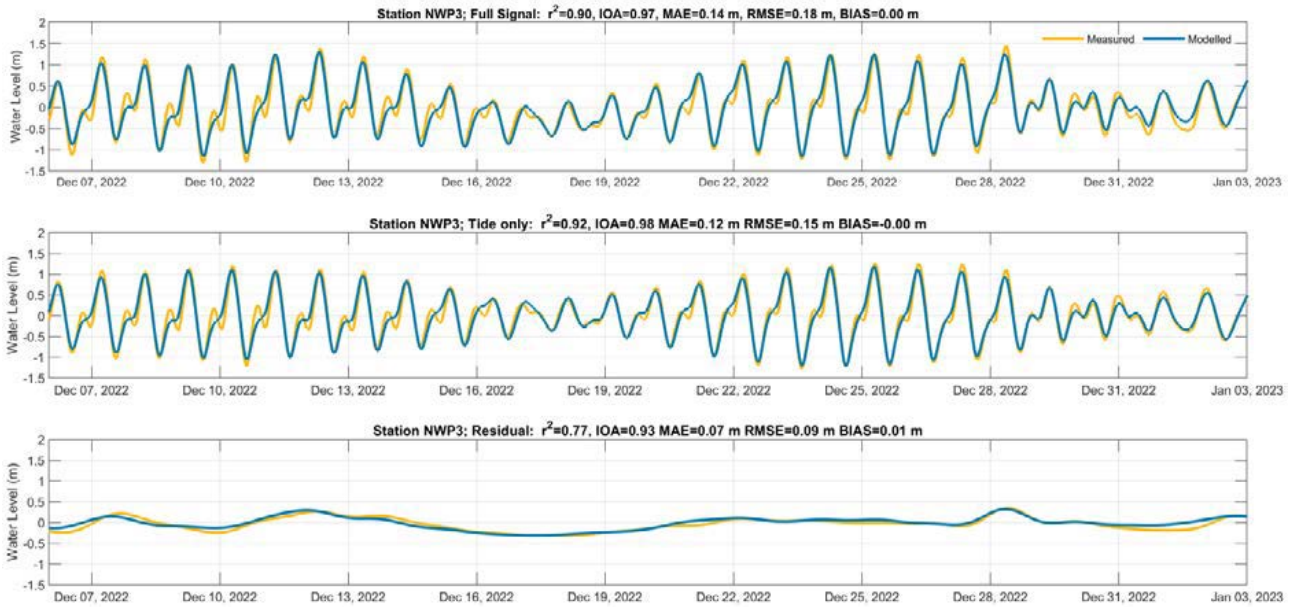


Figure C-2 Water Levels at NWP3 – 08/12/2022 to 03/01/2023

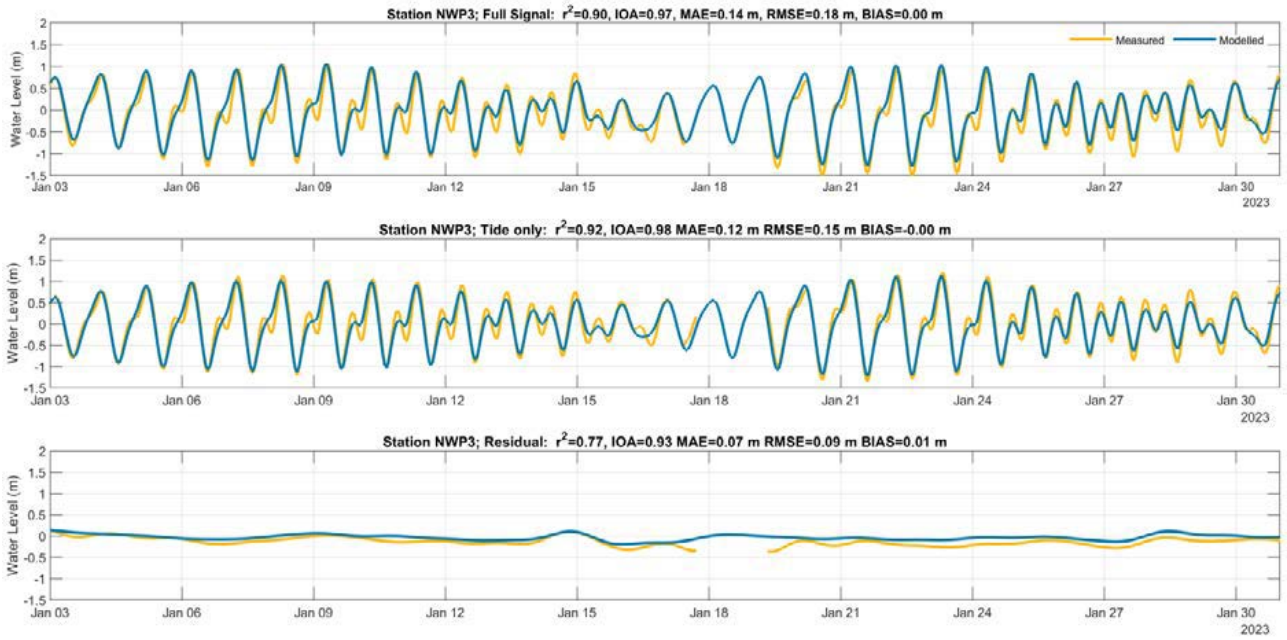


Figure C-3 Water Levels at NWP3 – 03/01 to 31/01/2023

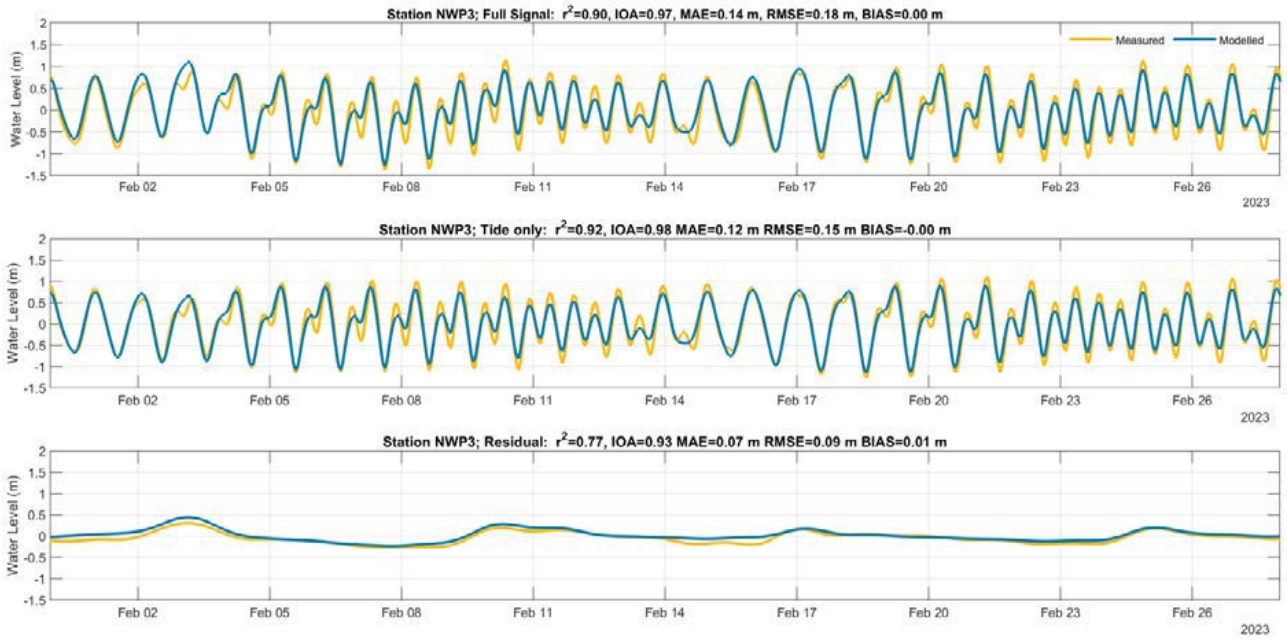


Figure C-4 Water Levels at NWP3 – 31/01 to 28/02/2023

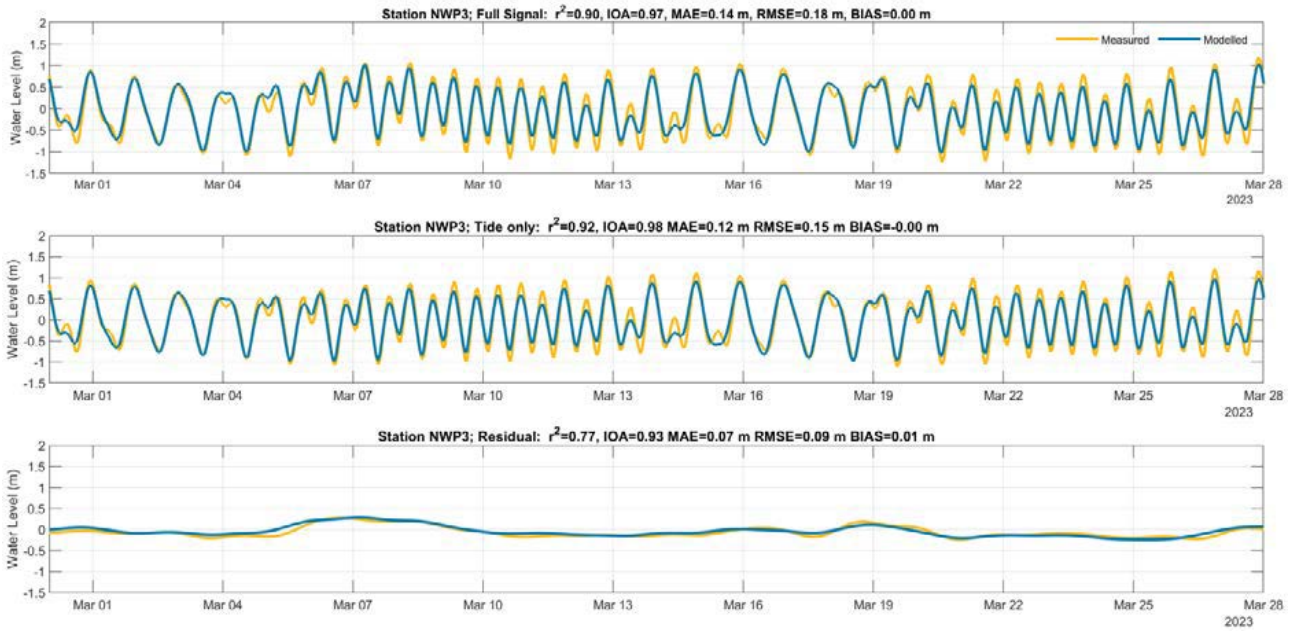


Figure C-5 Water Levels at NWP3 – 28/02 to 28/03/2023

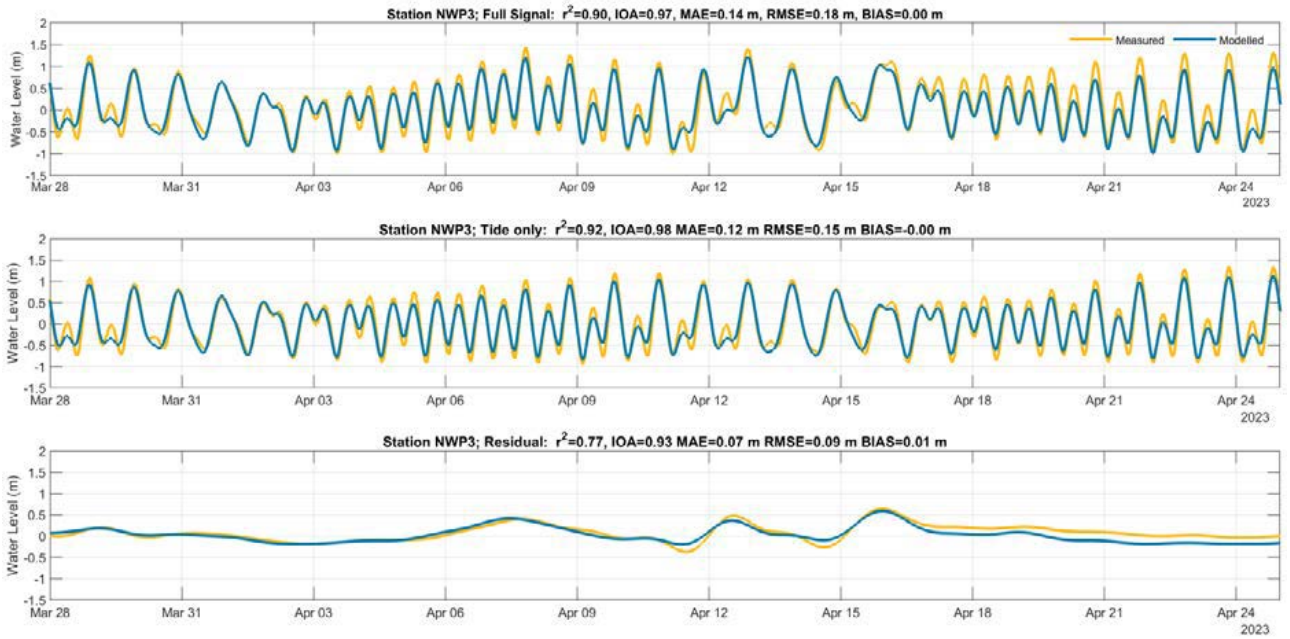


Figure C-6 Water Levels at NWP3 – 28/03 to 25/04/2023

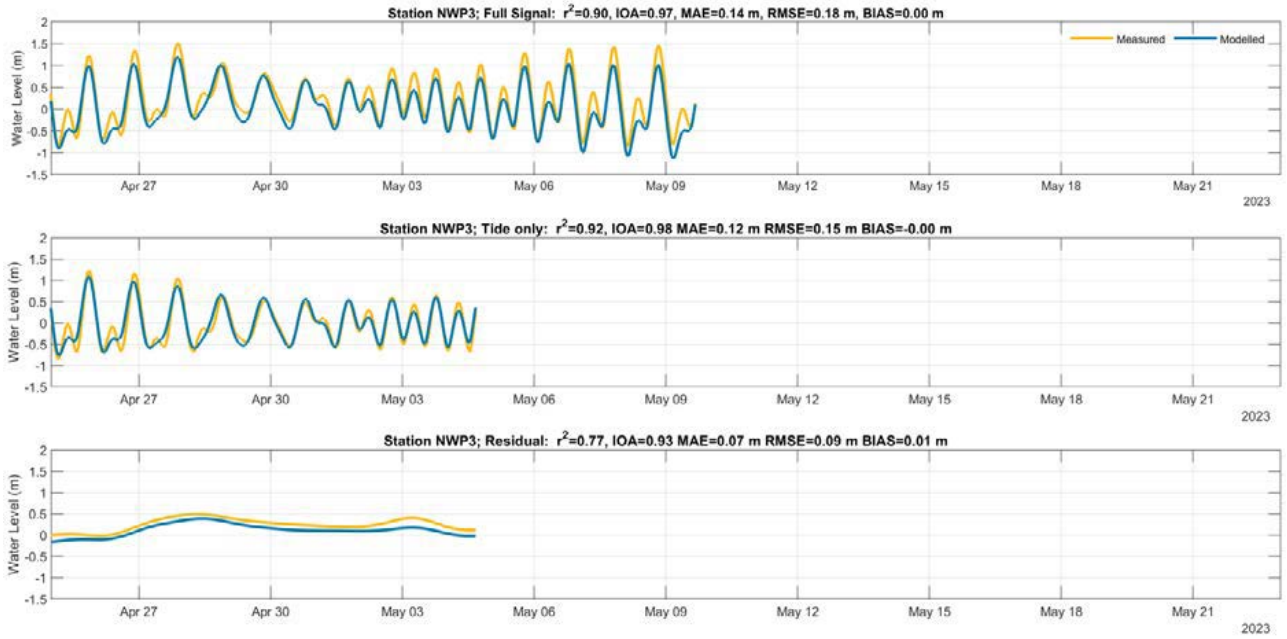


Figure C-7 Water Levels at NWP3 – 25/04 to 09/05/2023

C-2 Comparisons at SAMUSG

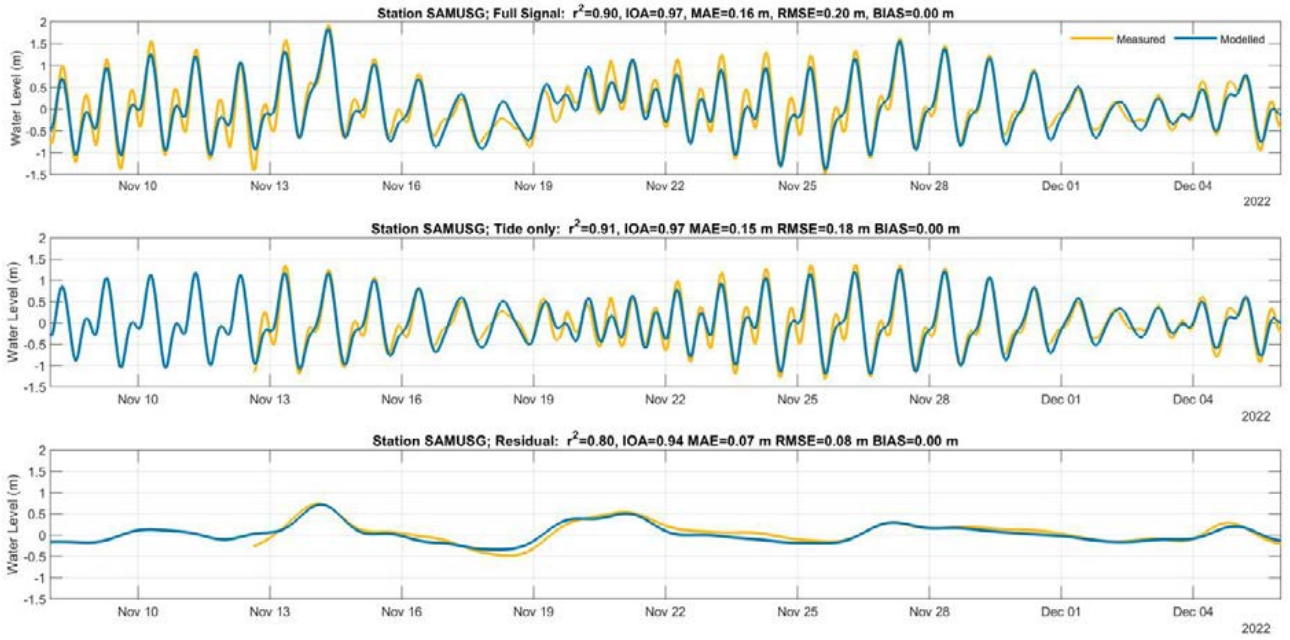


Figure C-8 Water Levels at SAMUSG – 08/11 to 06/12/2022

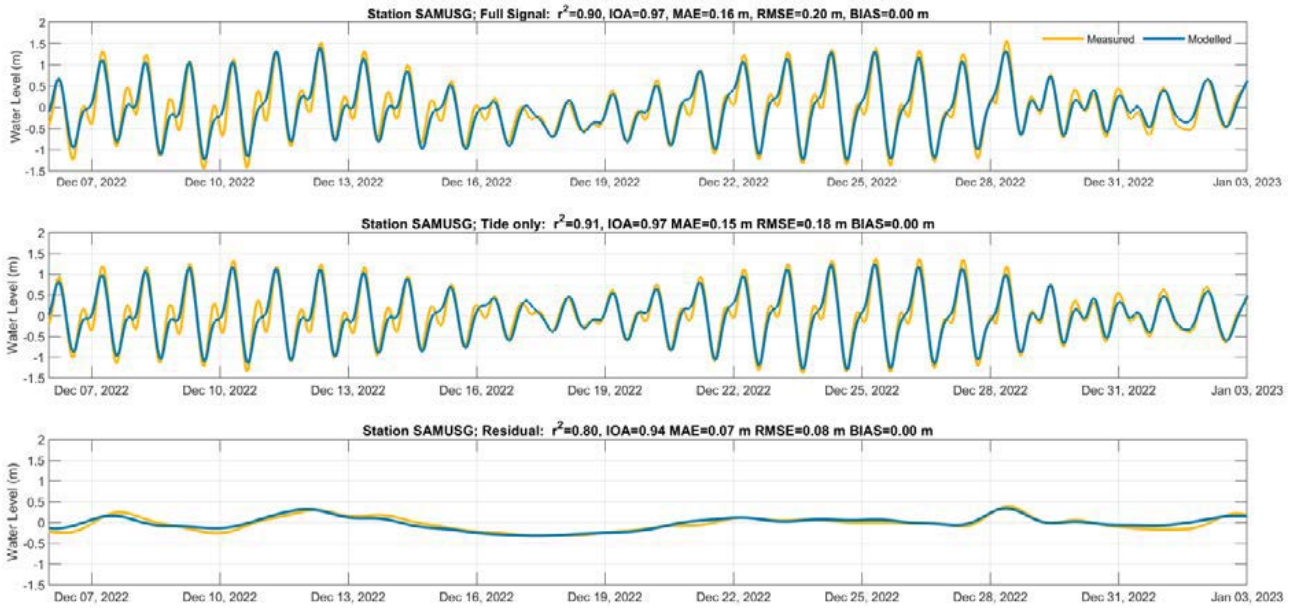


Figure C-9 Water Levels at SAMUSG – 06/12/2022 to 03/01/2023

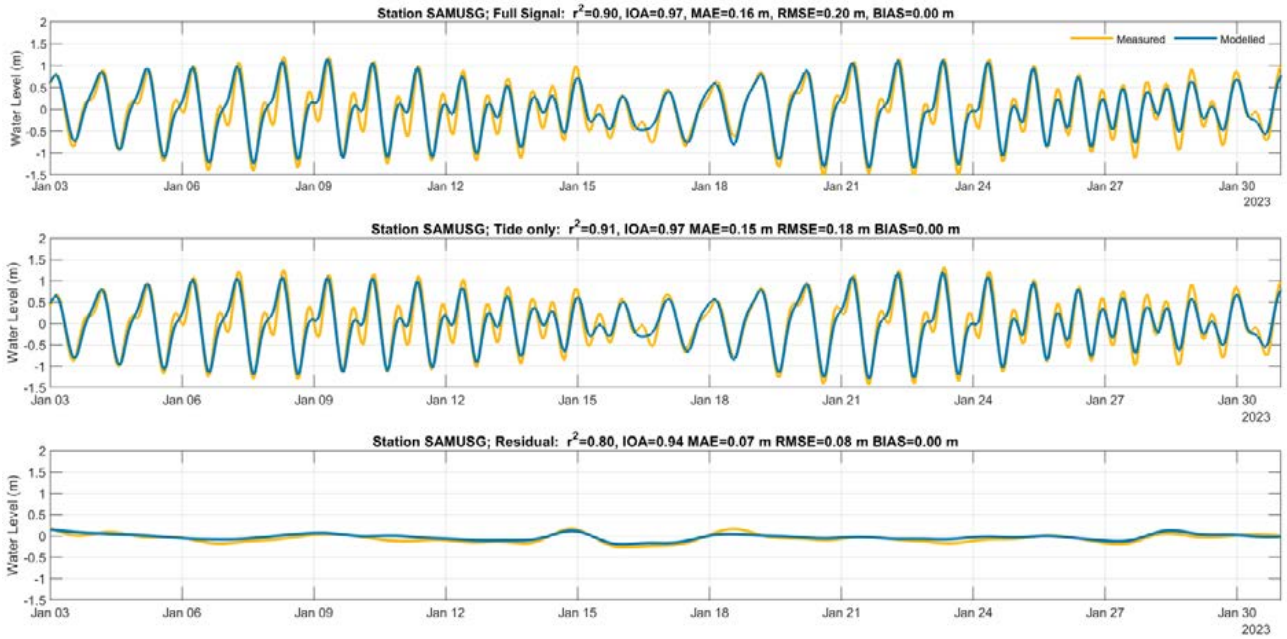


Figure C-10 Water Levels at SAMUSG – 03/01 to 31/01/2023

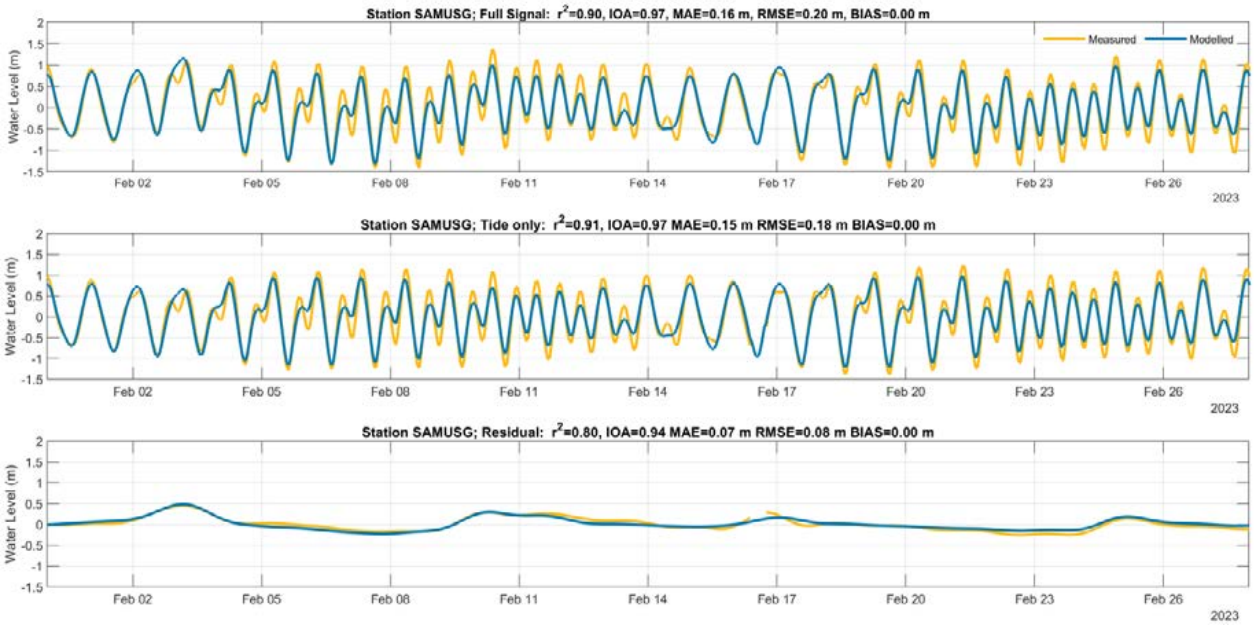


Figure C-11 Water Levels at SAMUSG – 31/01 to 28/02/2023

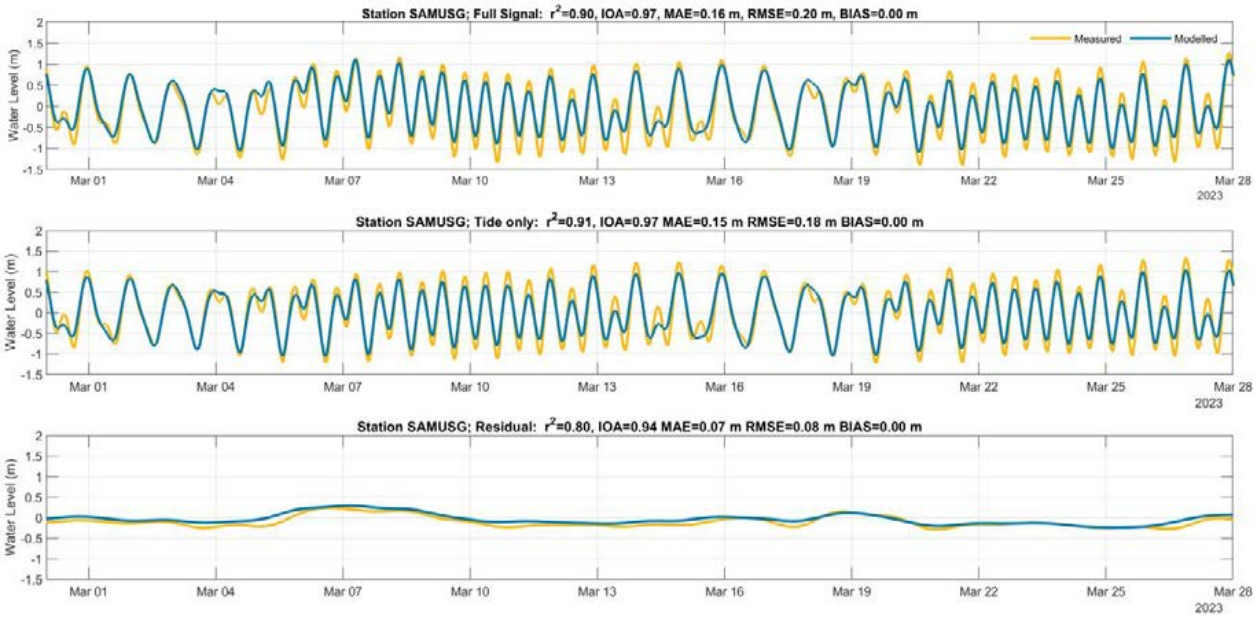


Figure C-12 Water Levels at SAMUSG – 28/02 to 28/03/2023

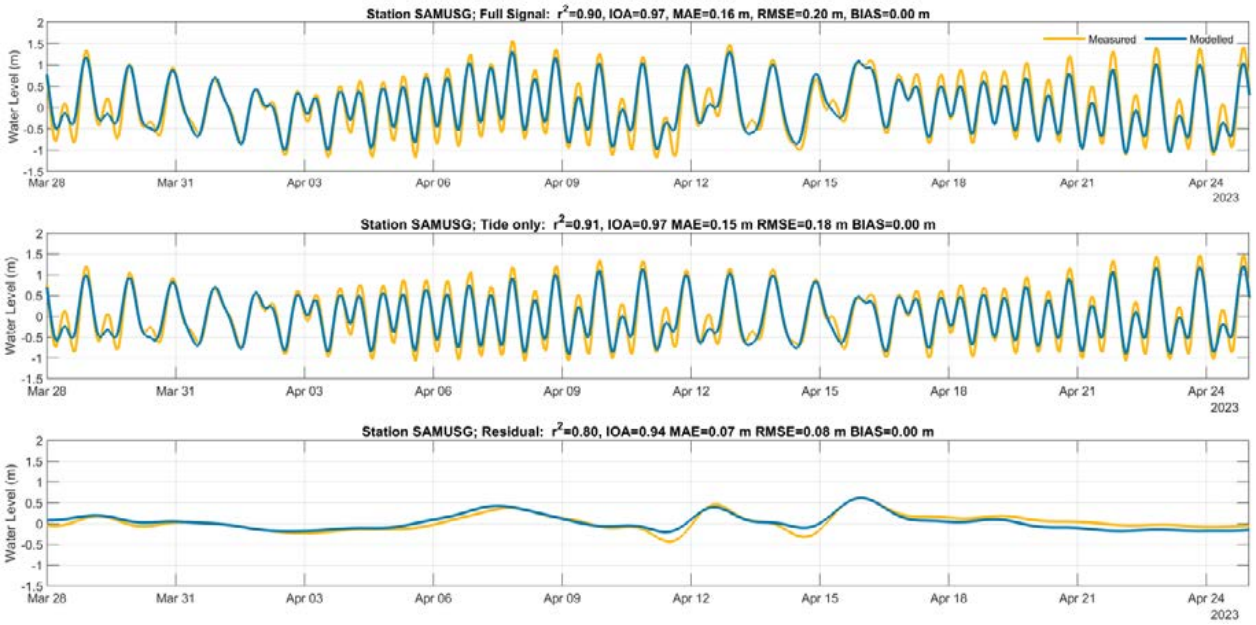


Figure C-13 Water Levels at SAMUSG – 28/03 to 25/04/2024

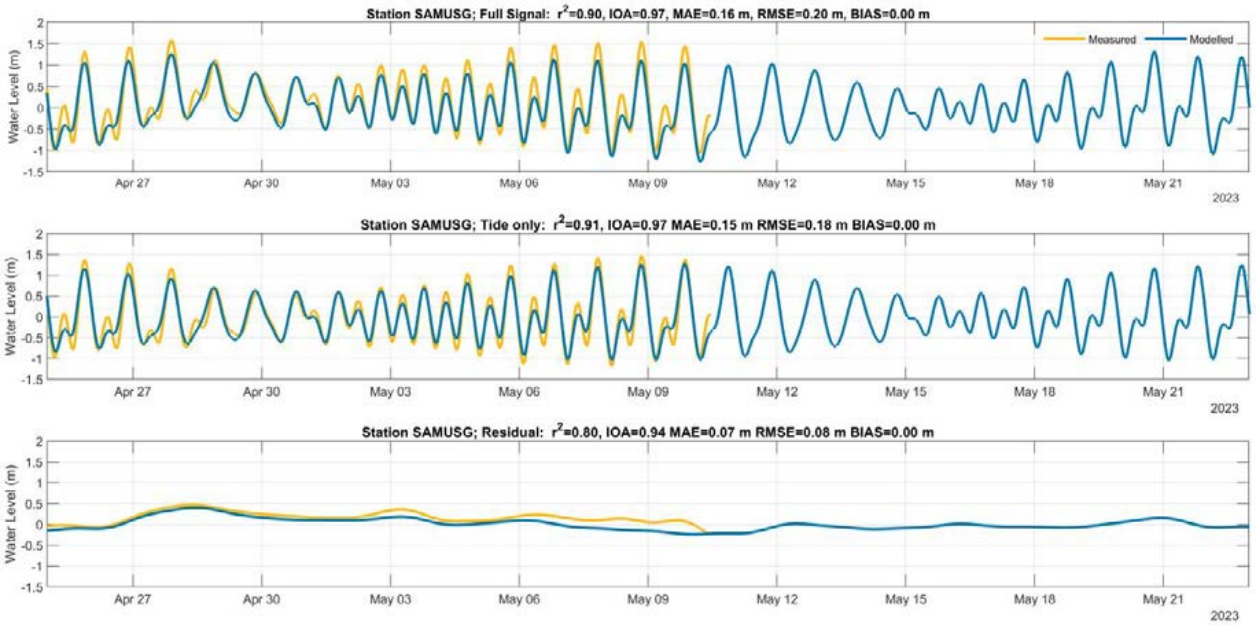


Figure C-14 Water Levels at SAMUSG – 25/04 to 23/05/2023

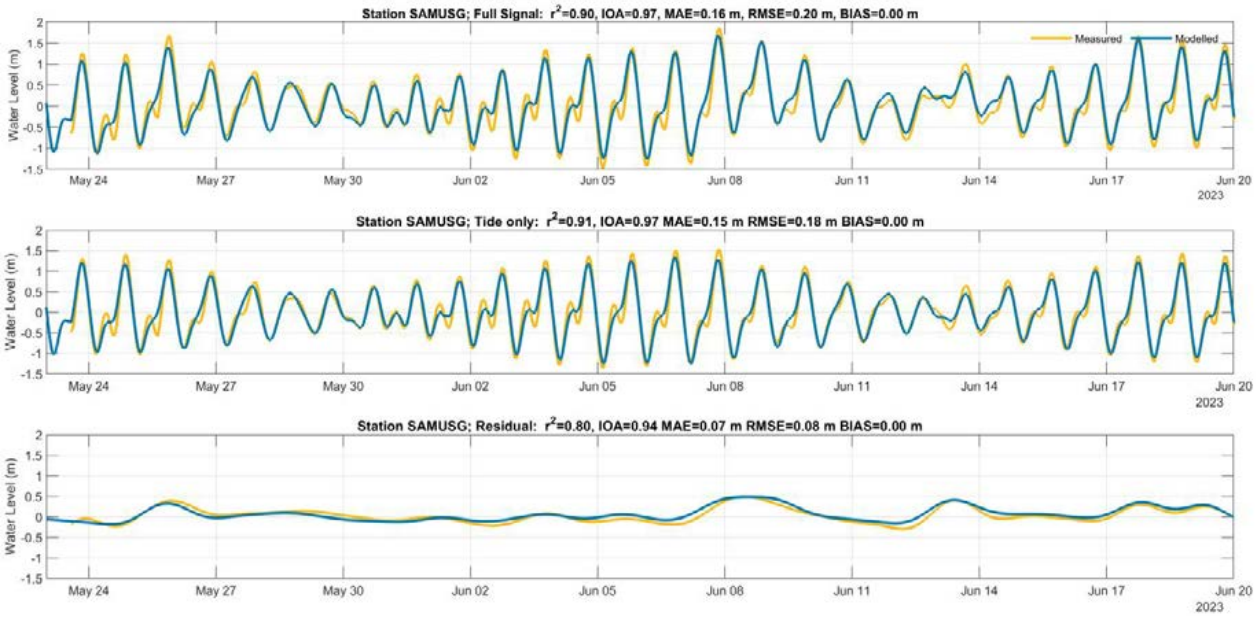


Figure C-15 Water Levels at SAMUSG – 23/04 to 20/06/2023

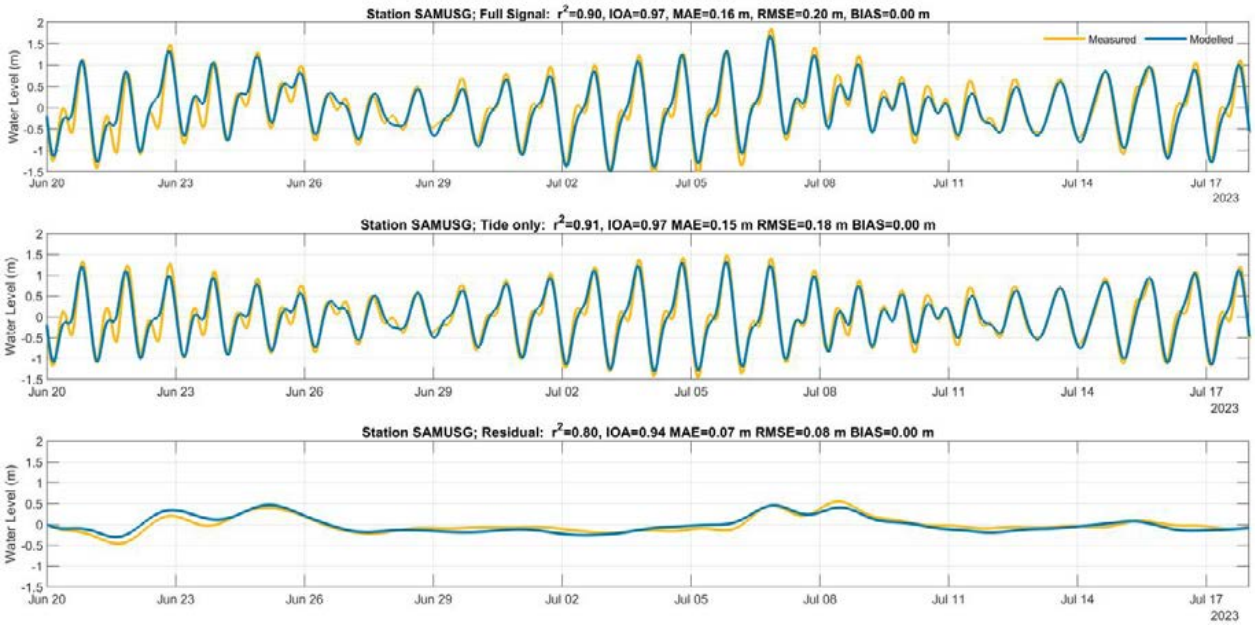


Figure C-16 Water Levels at SAMUSG – 20/06 to 18/07/2023

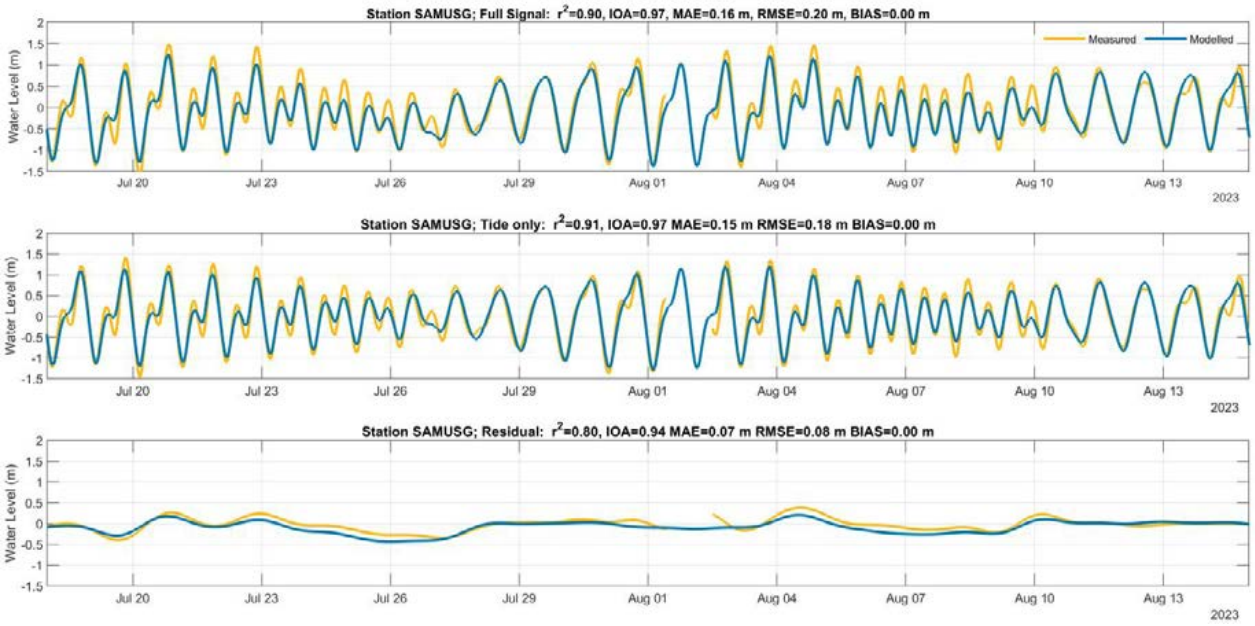


Figure C-17 Water Levels at SAMUSG – 18/07 to 15/08/2023

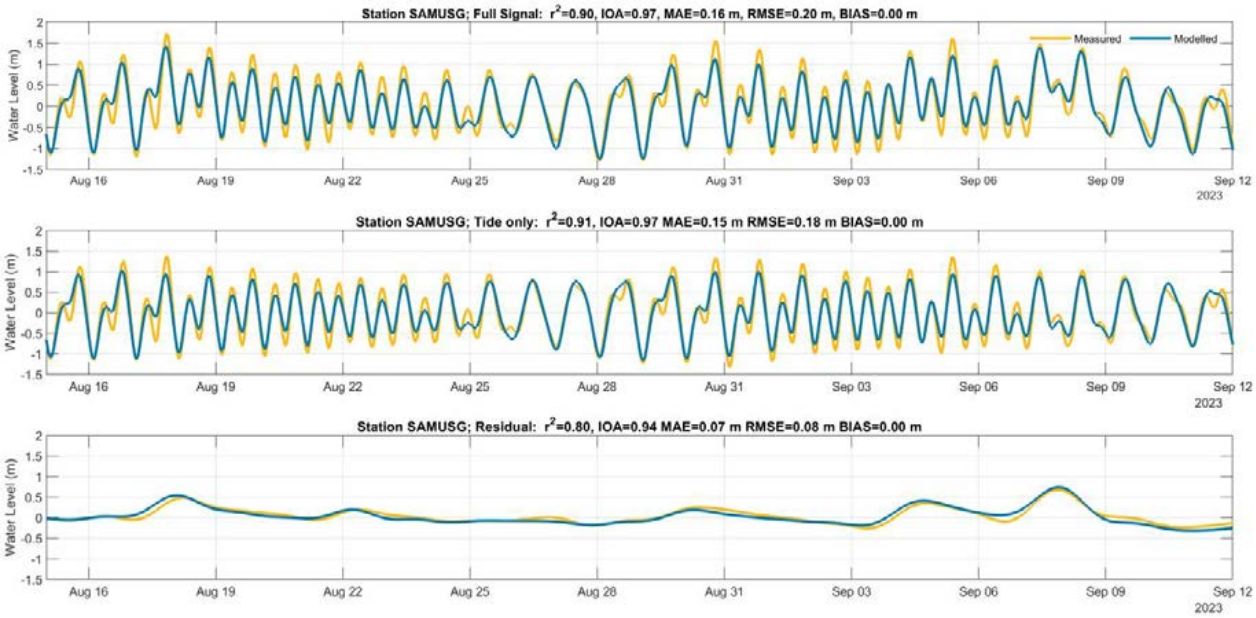


Figure C-18 Water Levels at SAMUSG – 15/08 to 12/09/2023

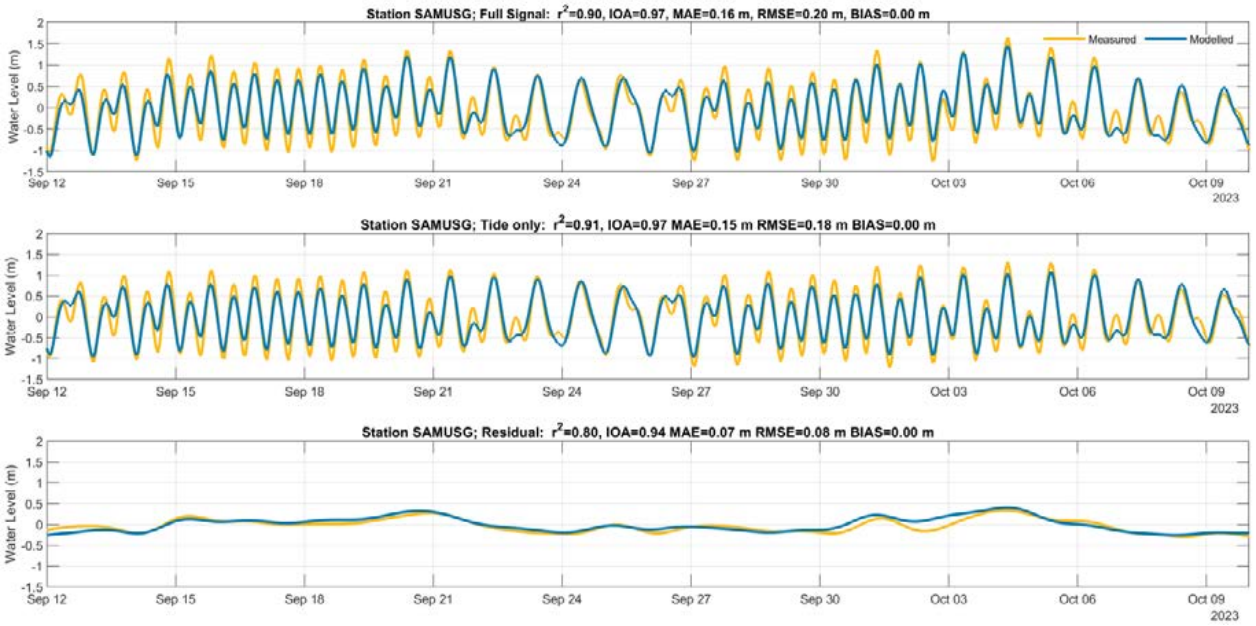


Figure C-19 Water Levels at SAMUSG – 12/09 to 10/10/2023

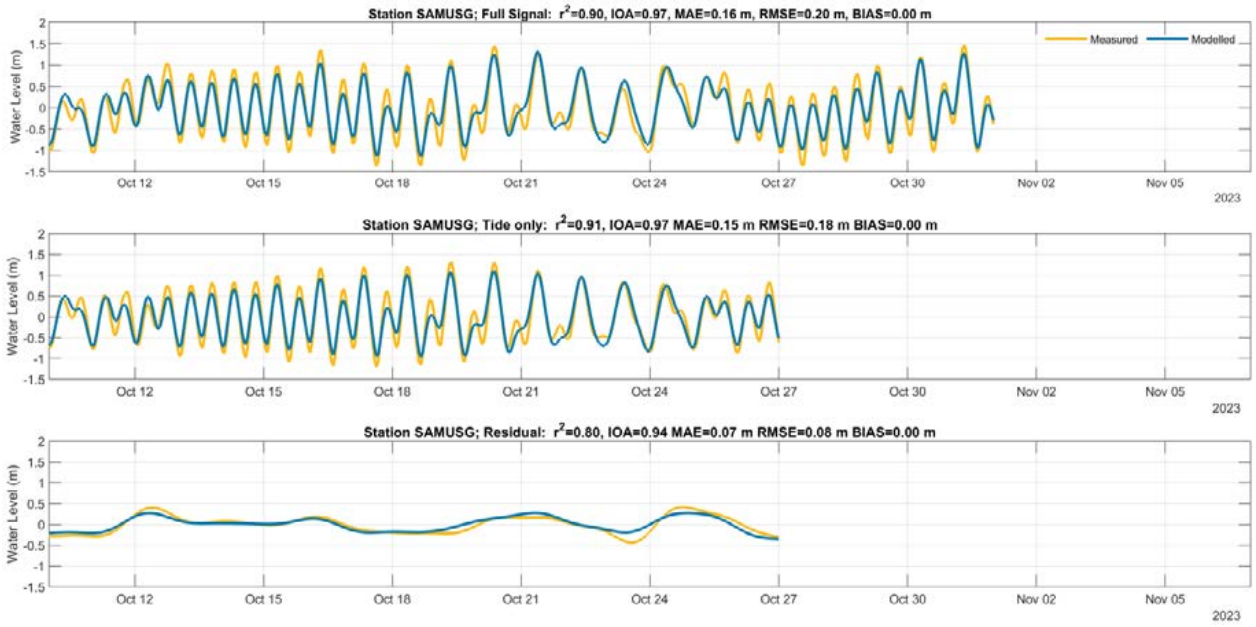


Figure C-20 Water Levels at SAMUSG – 10/10 to 01/11/2023

WATER TECHNOLOGY
WATER, COASTAL & ENVIRONMENTAL CONSULTANTS

APPENDIX D
WATER CURRENTS TIMESERIES COMPARISONS



D-1 Comparisons at NWP3

D-1-1 Depth-Averaged Currents

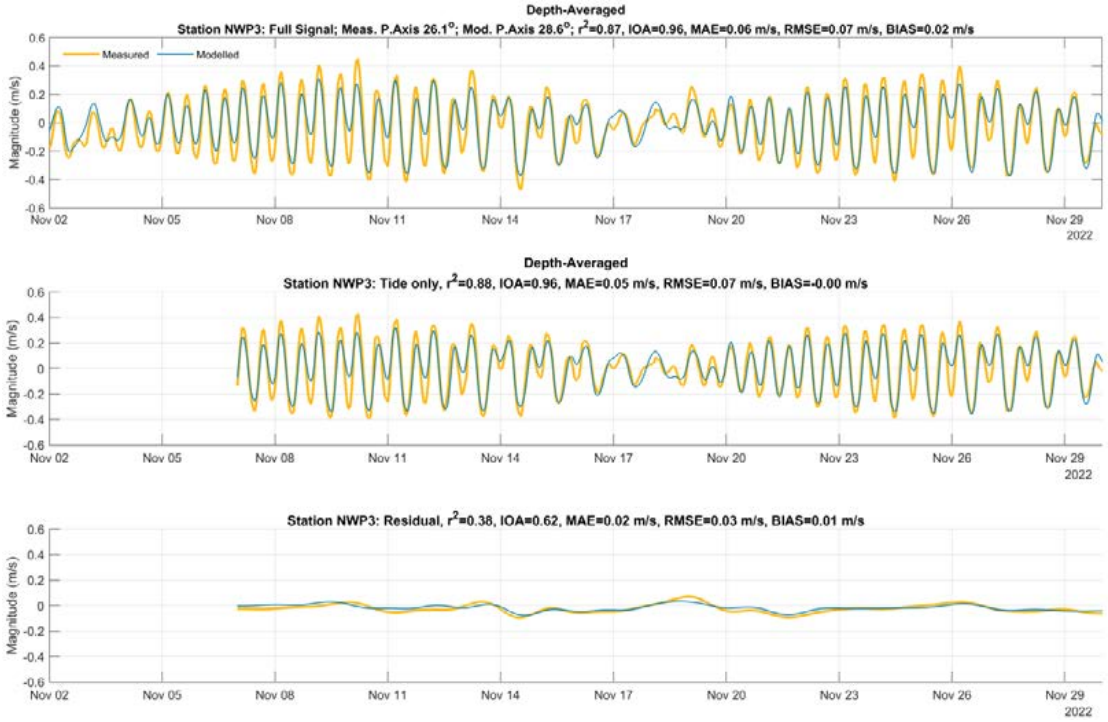


Figure D-1 Depth-Averaged Currents at NWP3 – 01/11 to 30/11/2022

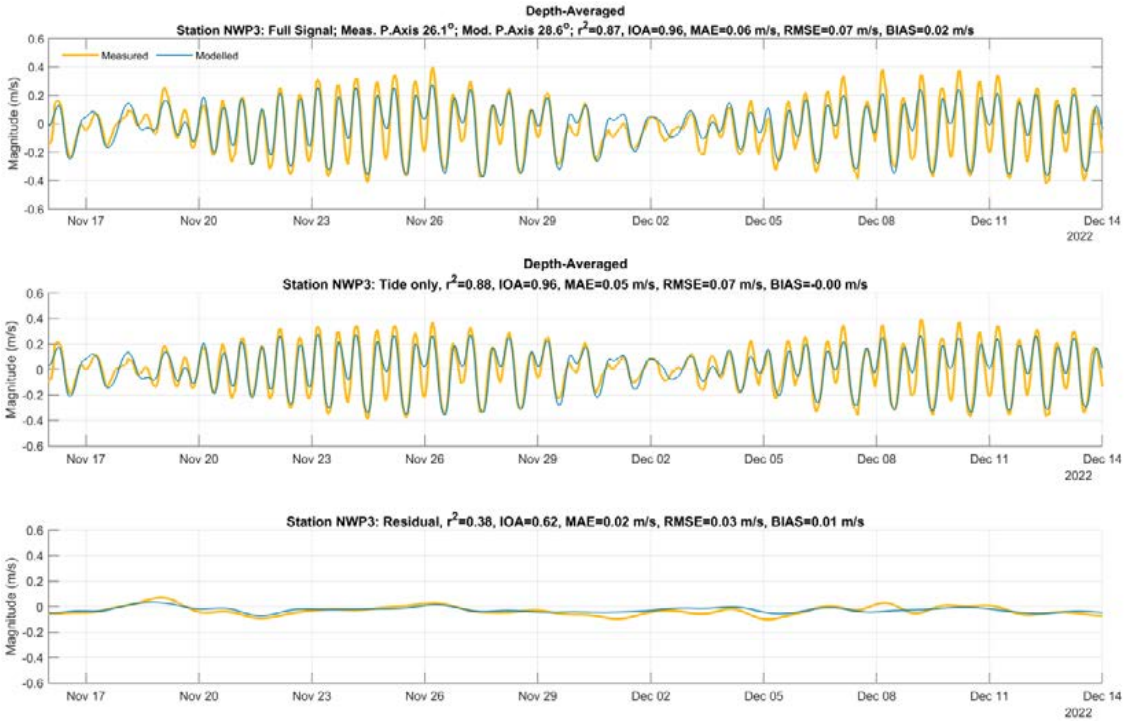


Figure D-2 Depth-Averaged Currents at NWP3 – 16/11 to 14/12/2022

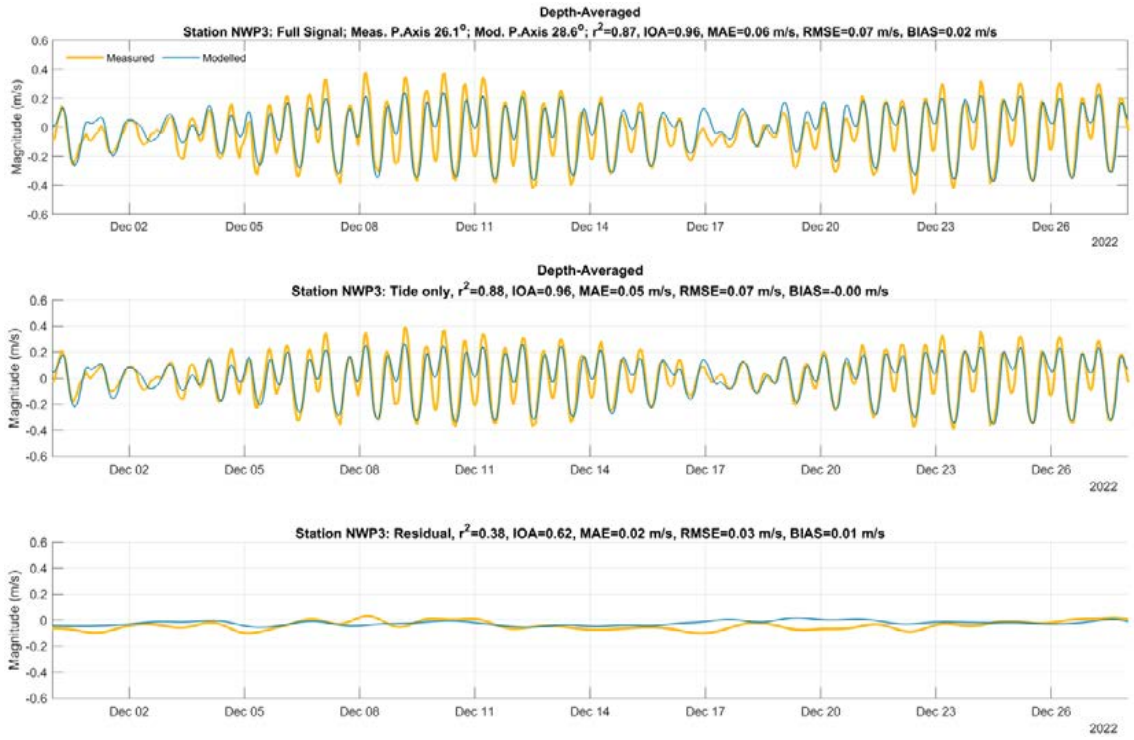


Figure D-3 Depth-Averaged Currents at NWP3 – 01/12 to 28/12/2022

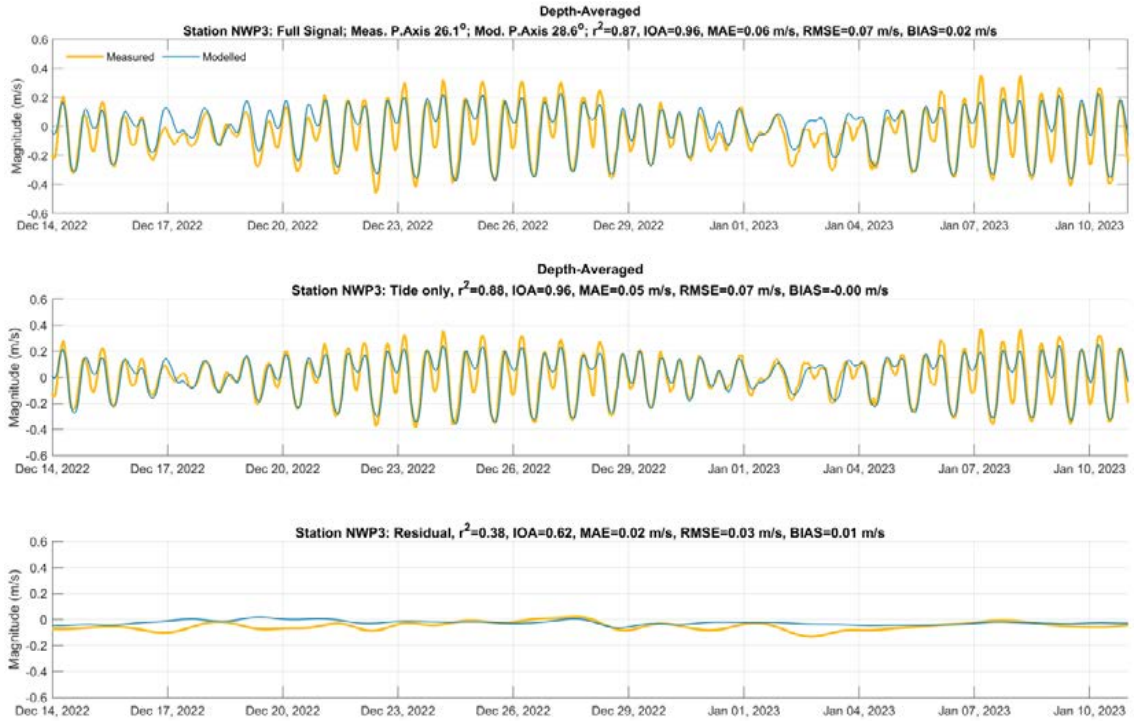


Figure D-4 Depth-Averaged Currents at NWP3 – 14/12/2022 to 11/01/2023

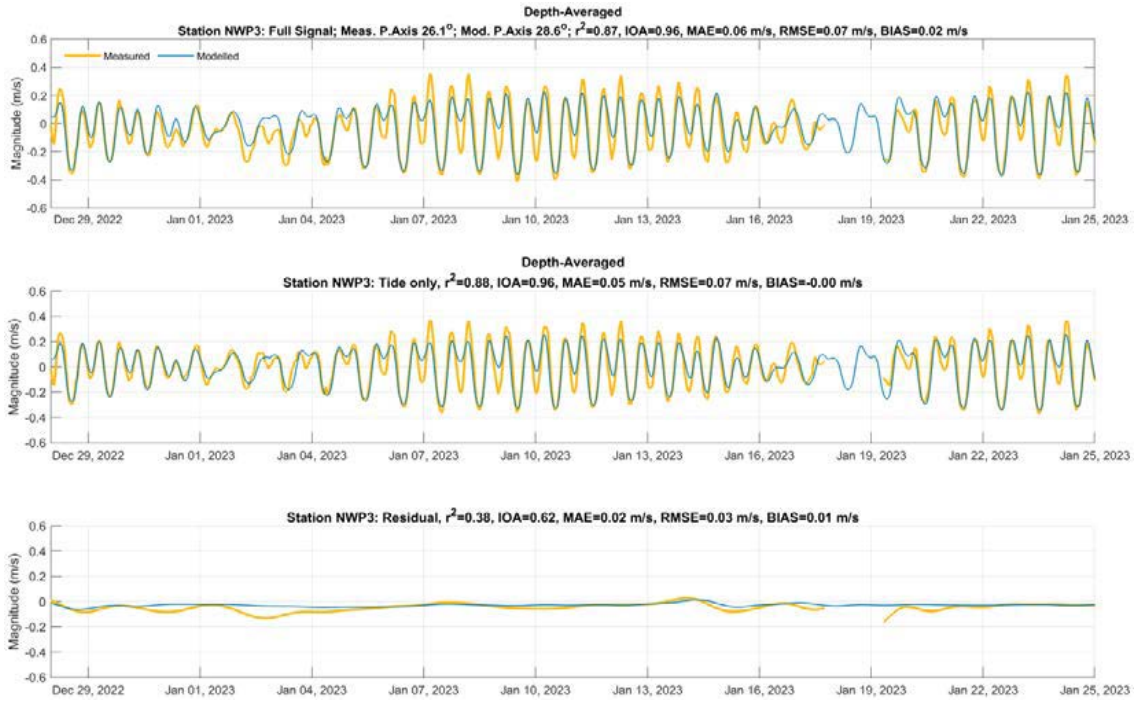


Figure D-5 Depth-Averaged Currents at NWP3 – 28/12/2022 to 25/01/2023

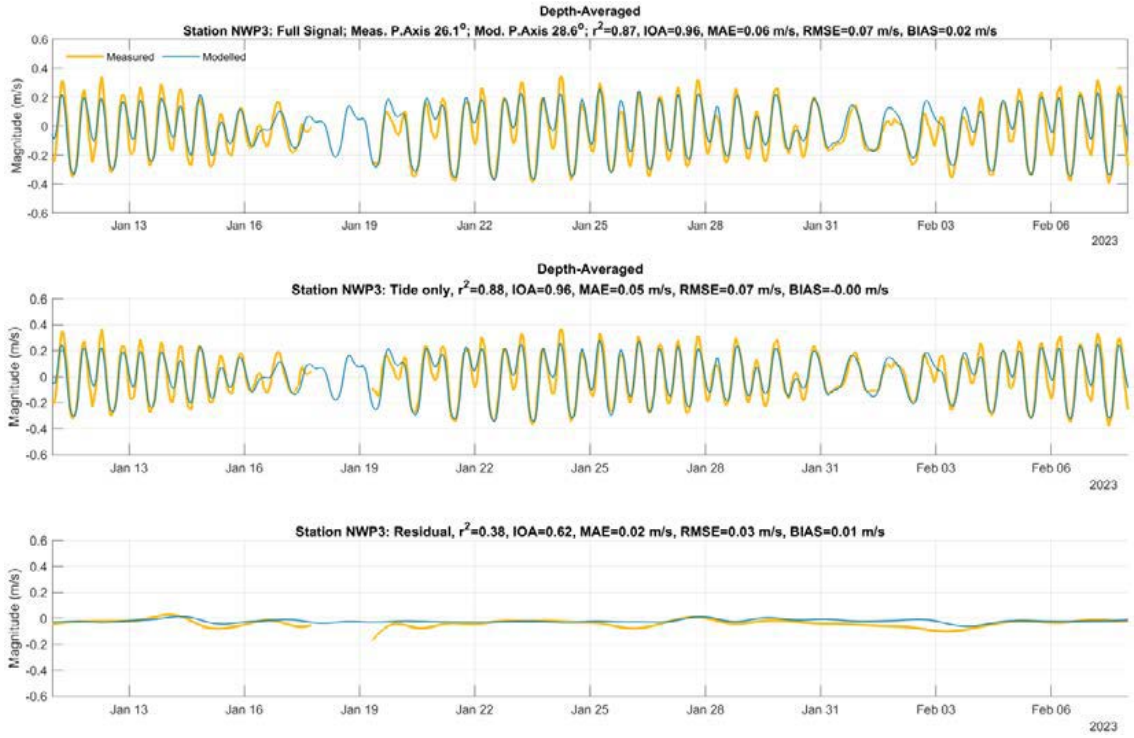


Figure D-6 Depth-Averaged Currents at NWP3 – 12/01 to 07/02/2023

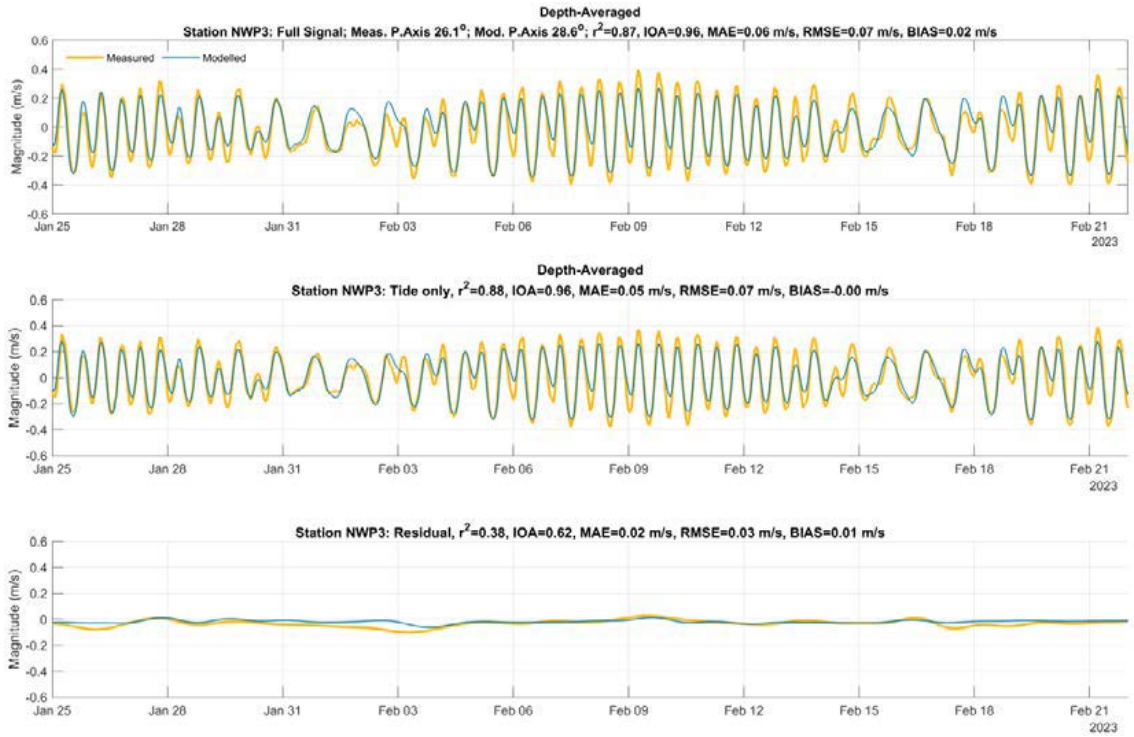


Figure D-7 Depth-Averaged Currents at NWP3 – 25/01 to 22/02/2023

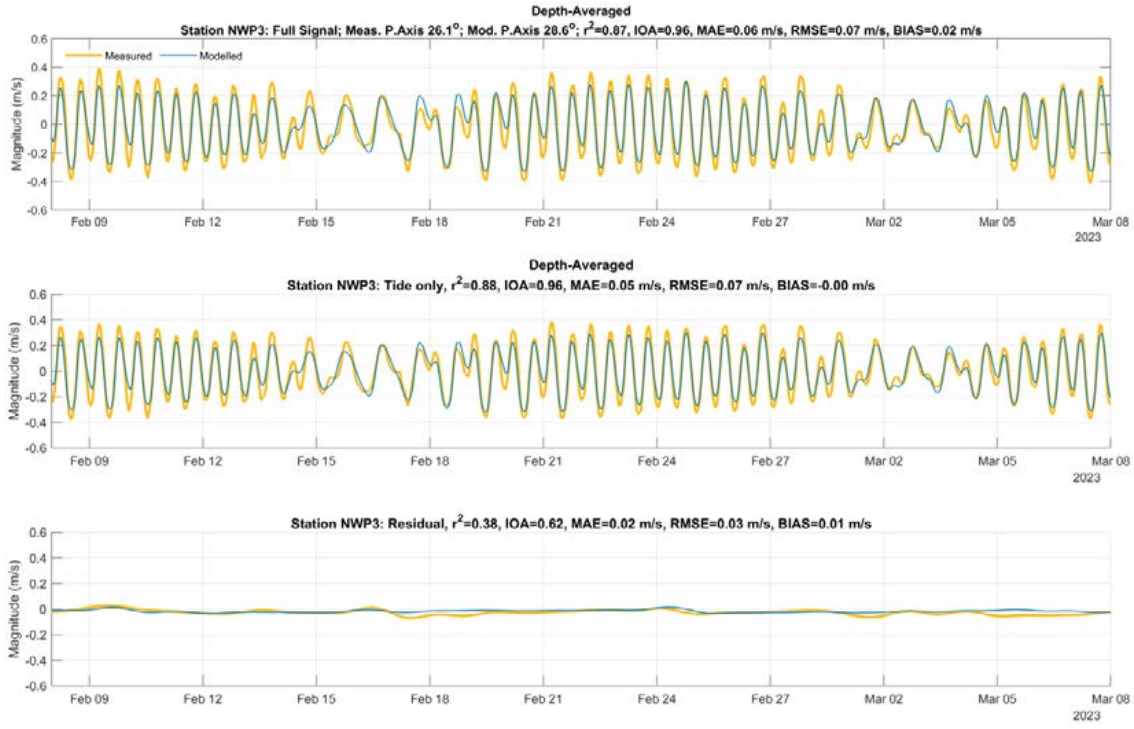


Figure D-8 Depth-Averaged Currents at NWP3 – 08/02 to 06/03/2023

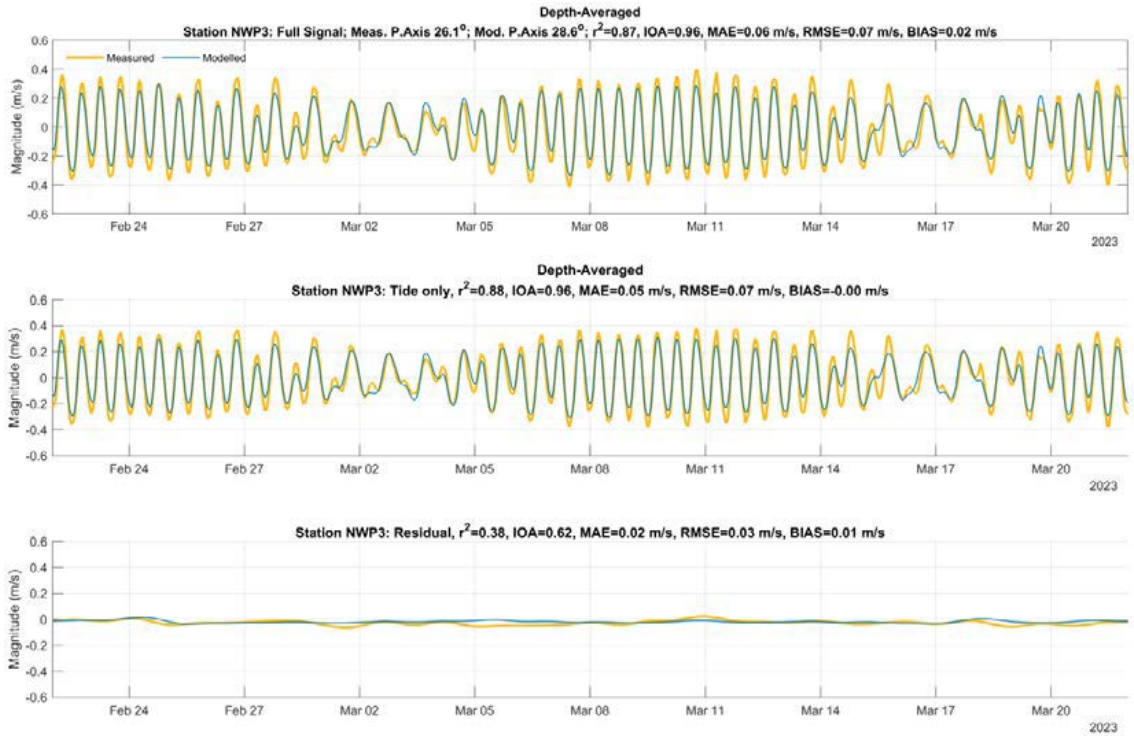


Figure D-9 Depth-Averaged Currents at NWP3 – 23/02 to 21/03/2023

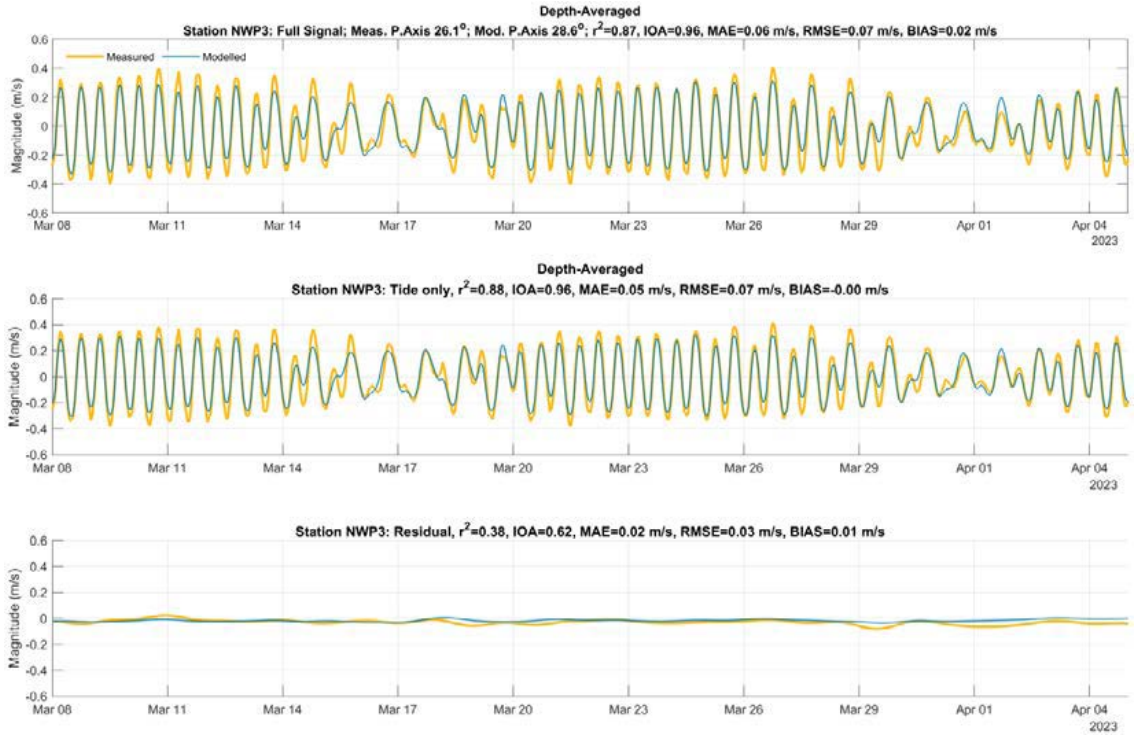


Figure D-10 Depth-Averaged Currents at NWP3 – 08/03 to 05/04/2023

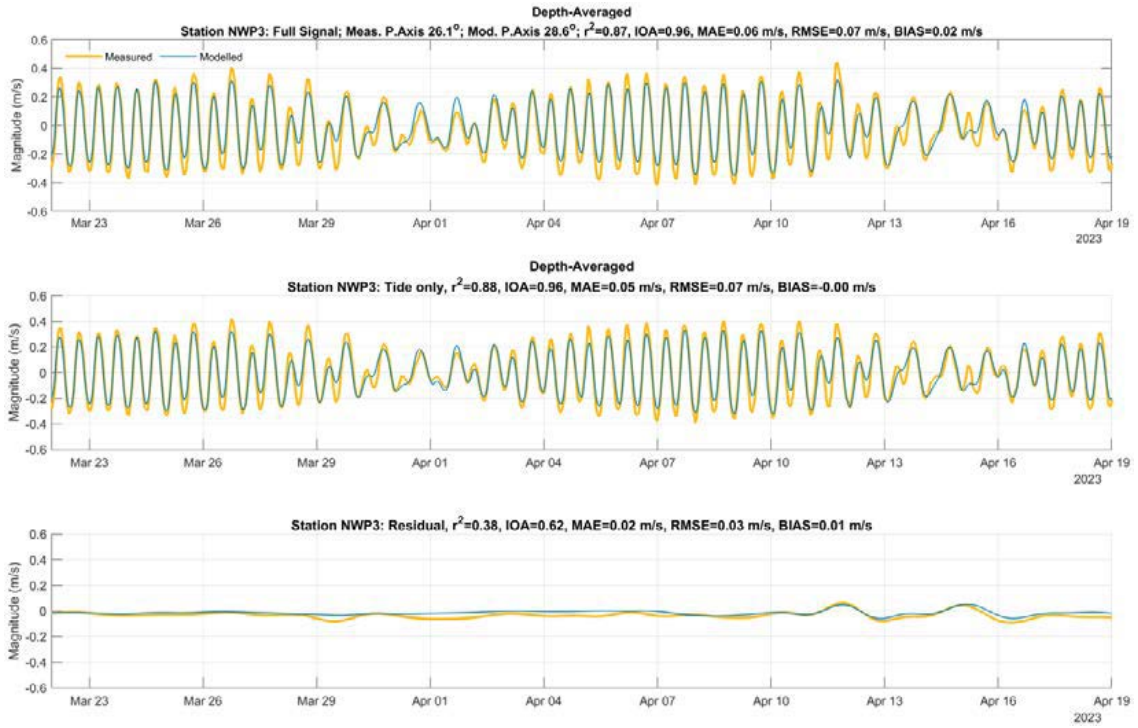


Figure D-11 Depth-Averaged Currents at NWP3 – 22/03 to 19/04/2023

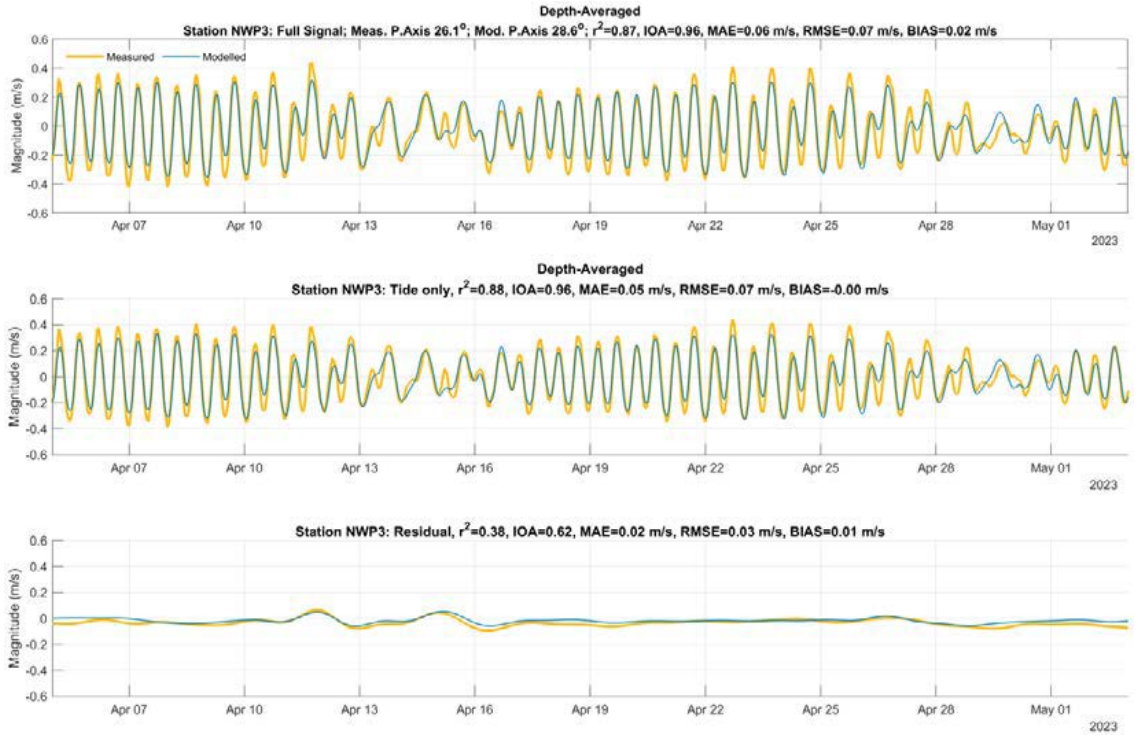


Figure D-12 Depth-Averaged Currents at NWP3 – 08/04 to 02/05/2023

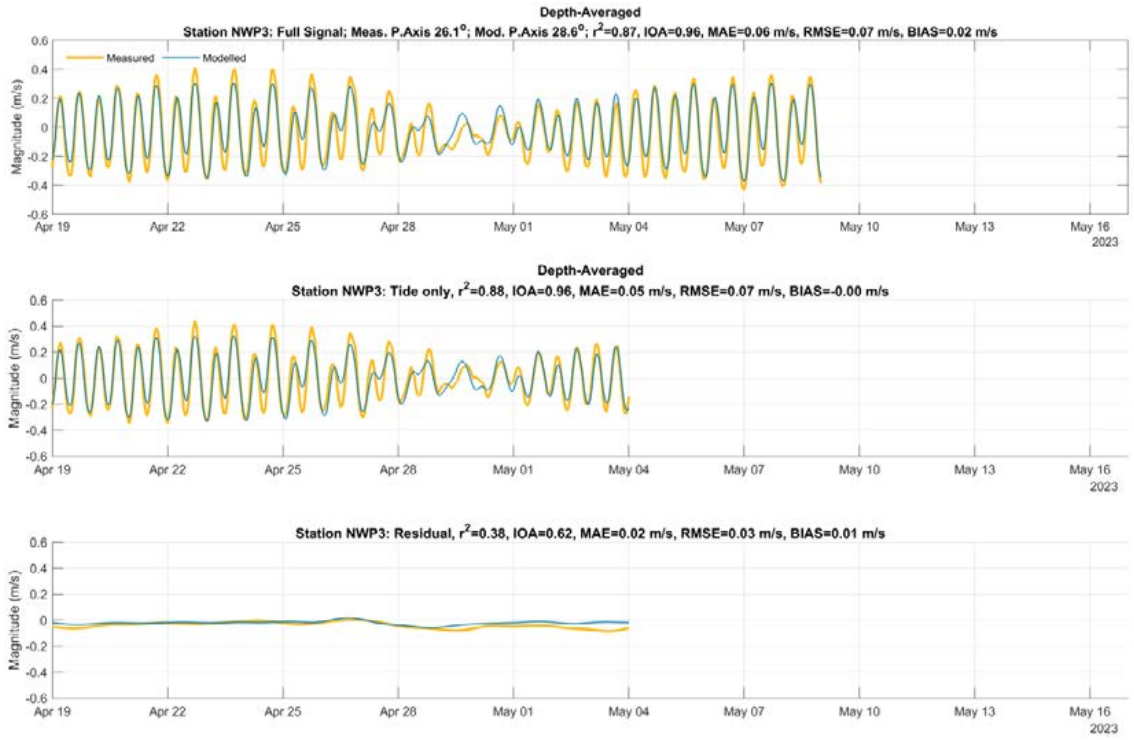


Figure D-13 Depth-Averaged Currents at NWP3 – 19/04 to 09/05/2023

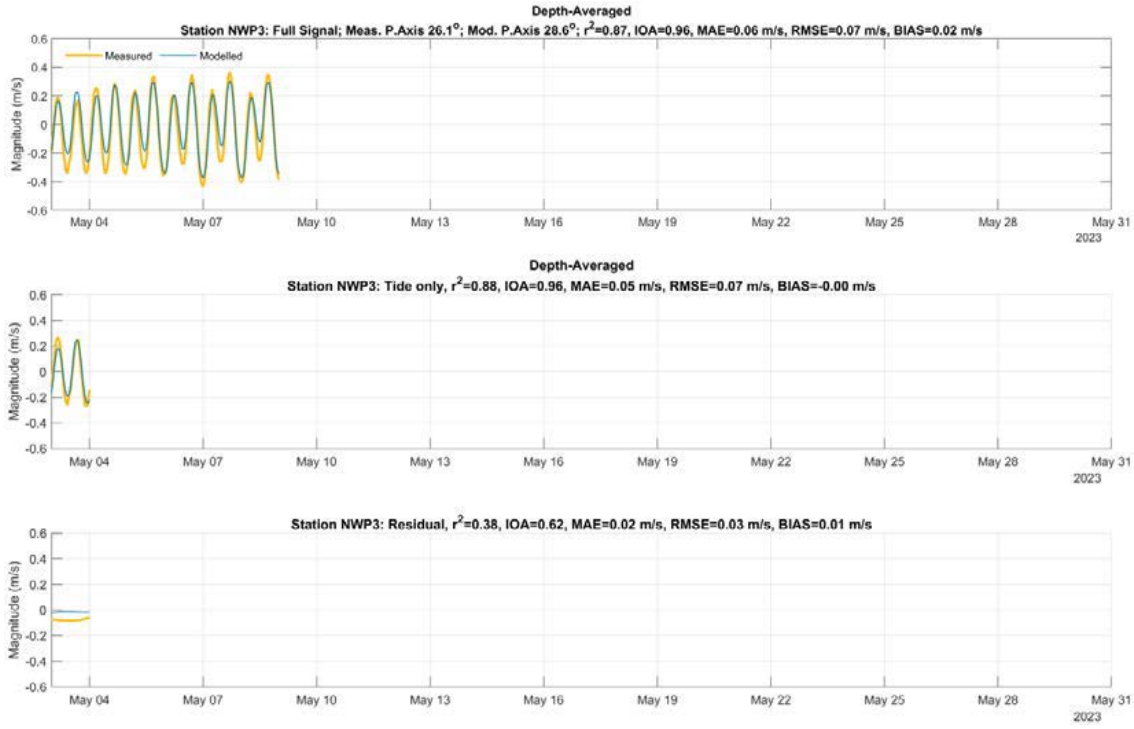


Figure D-14 Depth-Averaged Currents at NWP3 – 03/05 to 09/05/2023

D-1-2 Lower 1/3 of the Water Column Currents

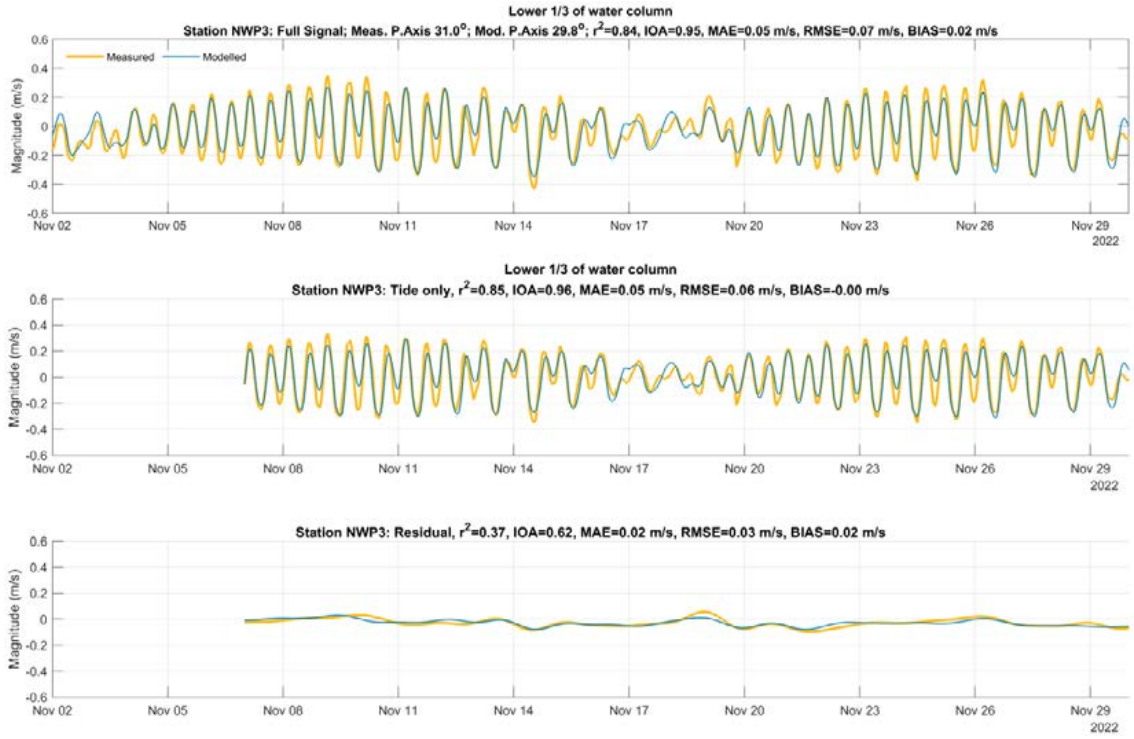


Figure D-15 Lower 1/3 of the Water Column Currents at NWP3 – 02/11 to 30/11/2022

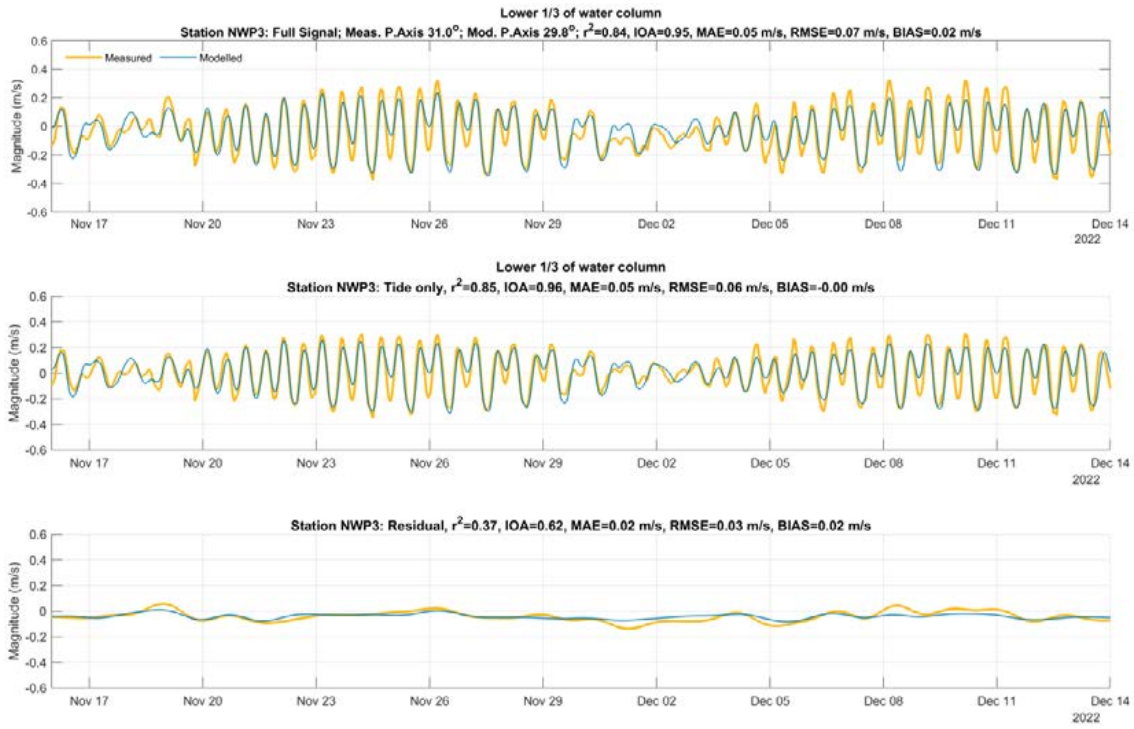


Figure D-16 Lower 1/3 of the Water Column Currents at NWP3 – 13/11 to 14/12/2022

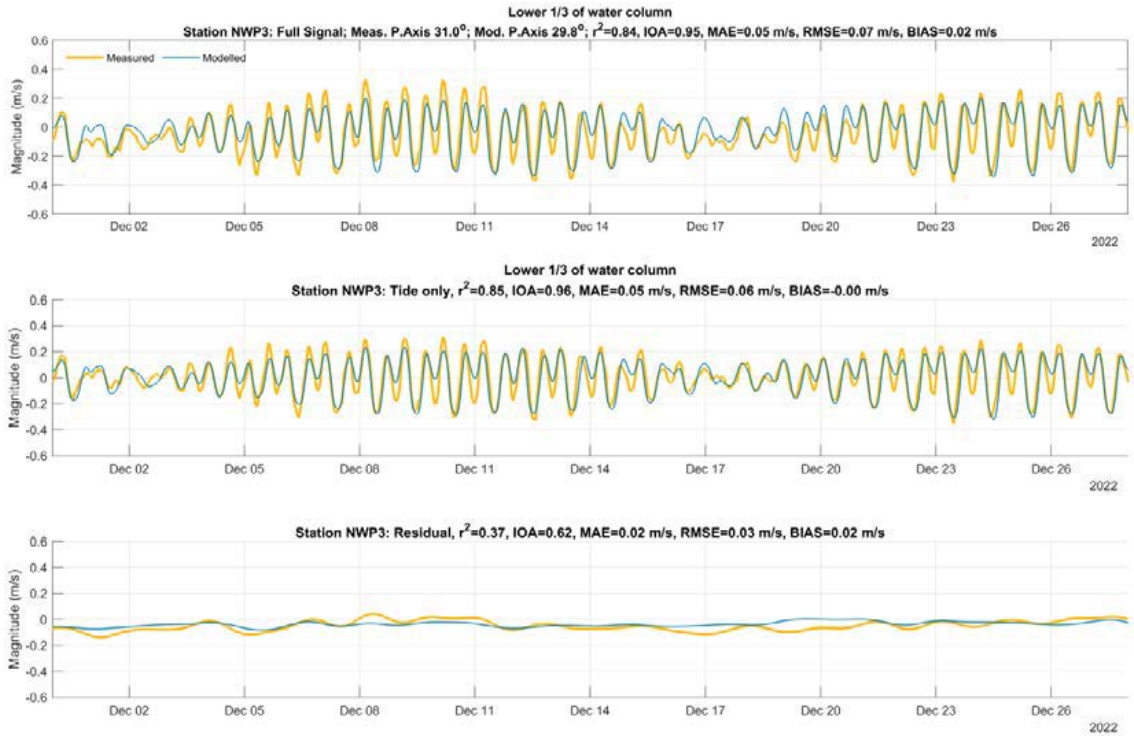


Figure D-17 Lower 1/3 of the Water Column Currents at NWP3 – 01/12 to 28/12/2022

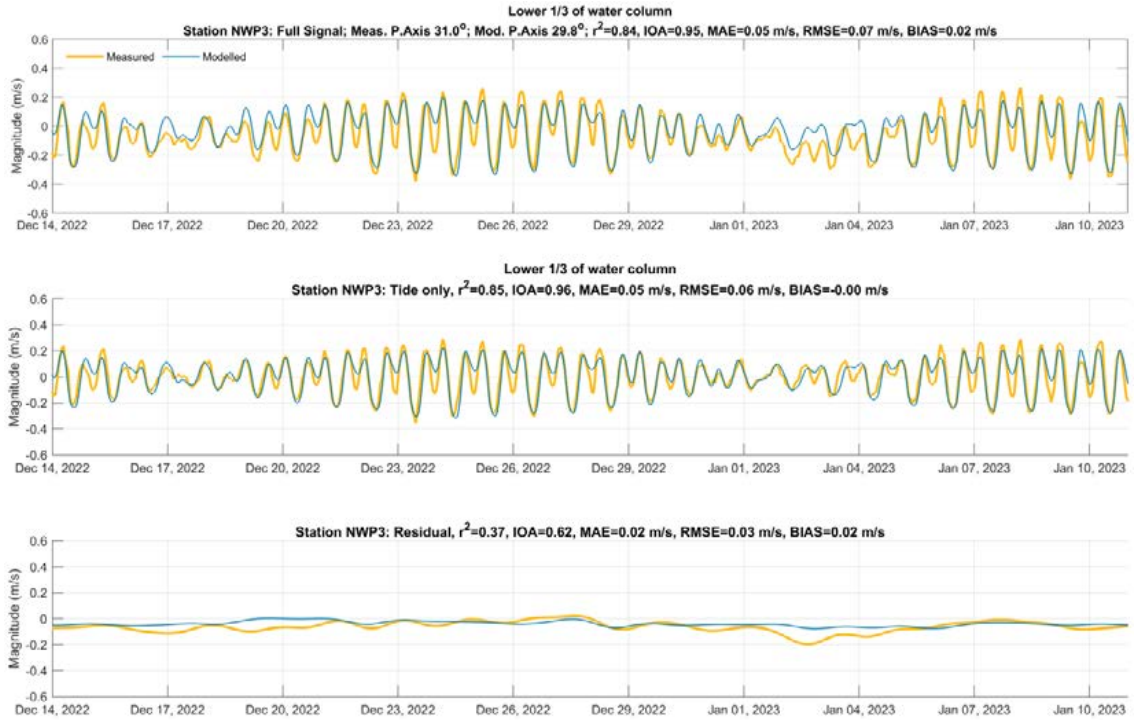


Figure D-18 Lower 1/3 of the Water Column Currents at NWP3 – 14/12/2022 to 11/01/2023

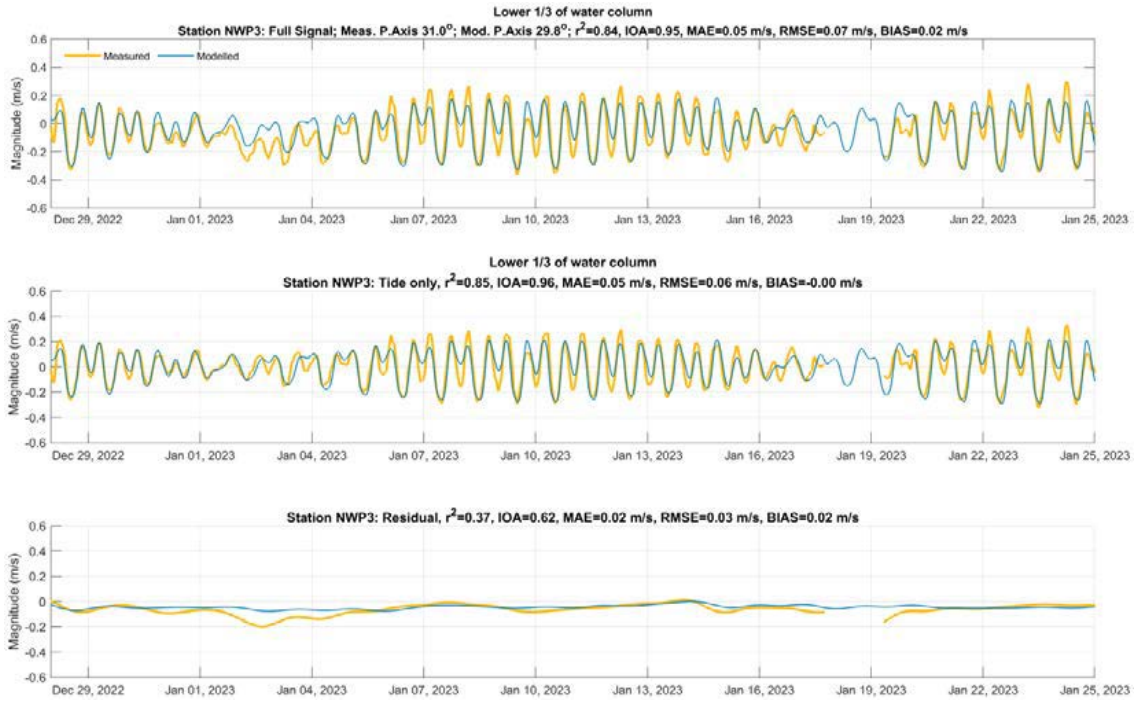


Figure D-19 Lower 1/3 of the Water Column Currents at NWP3 – 28/12/2022 to 25/01/2023

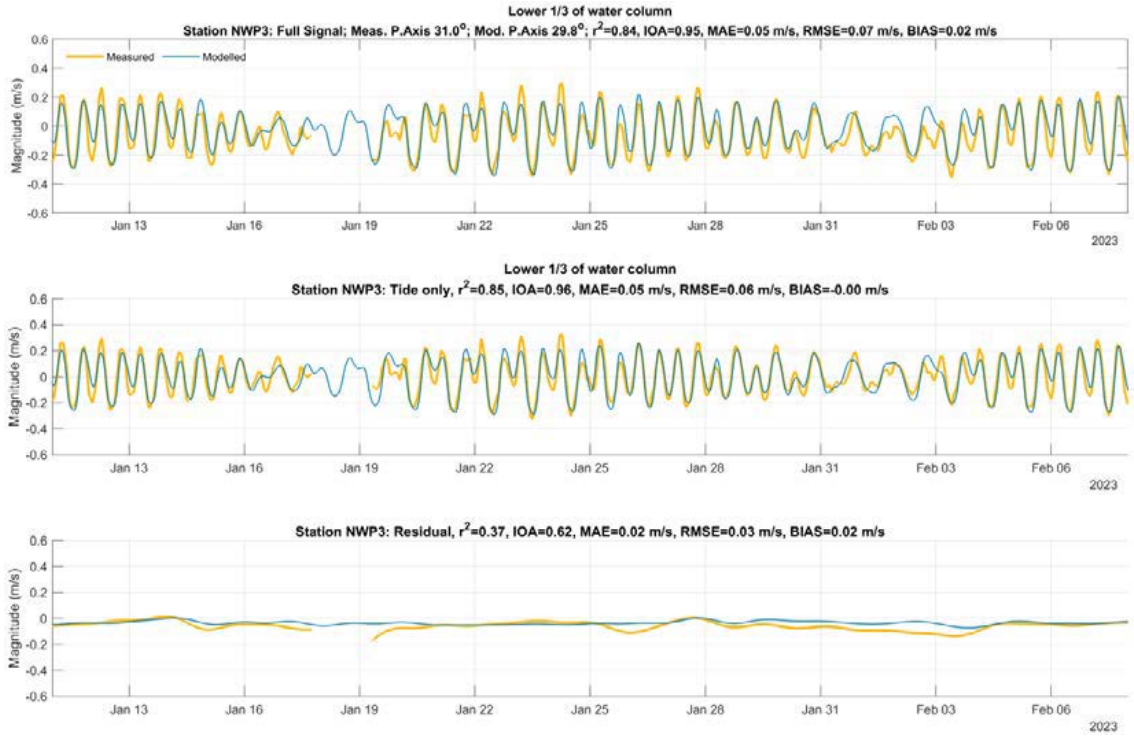


Figure D-20 Lower 1/3 of the Water Column Currents at NWP3 – 12/01 to 07/02/2023

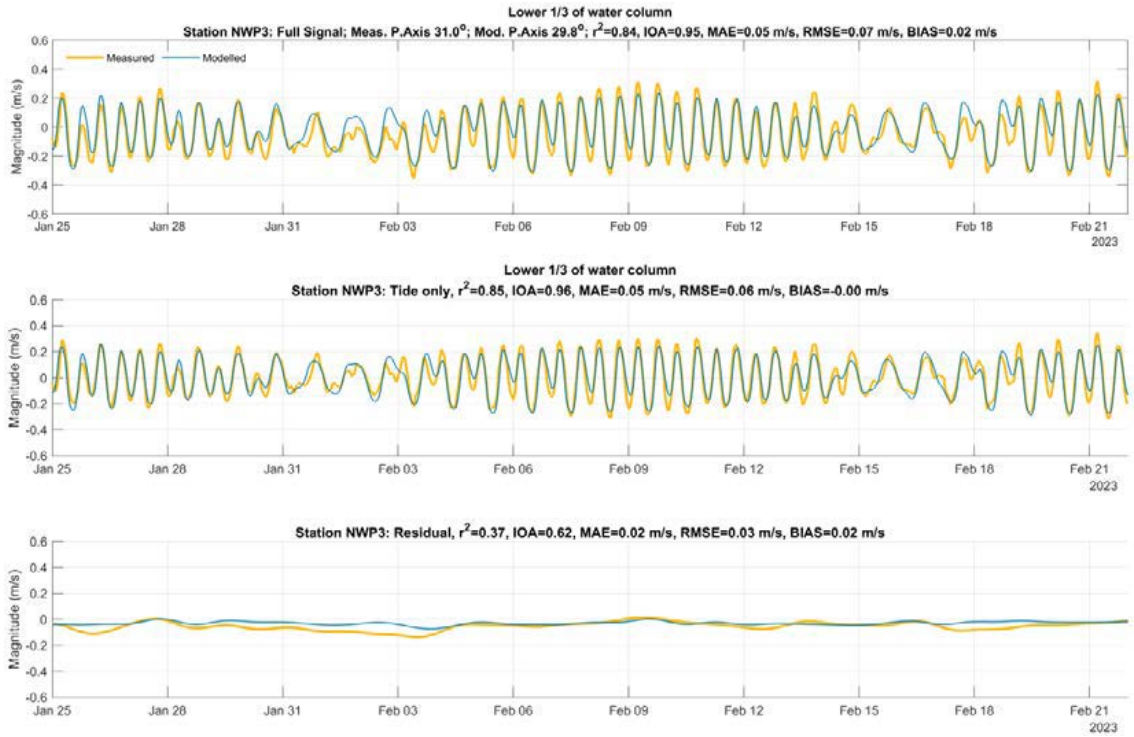


Figure D-21 Lower 1/3 of the Water Column Currents at NWP3 – 25/01 to 22/02/2023

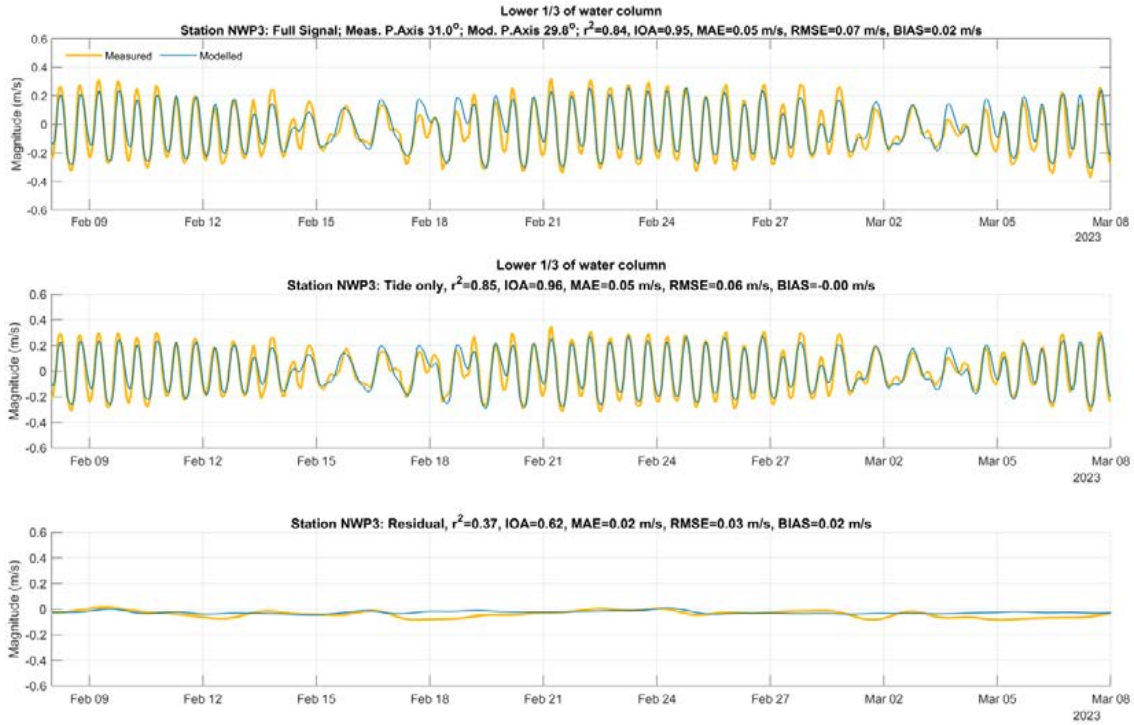


Figure D-22 Lower 1/3 of the Water Column Currents at NWP3 – 08/02 to 08/03/2023

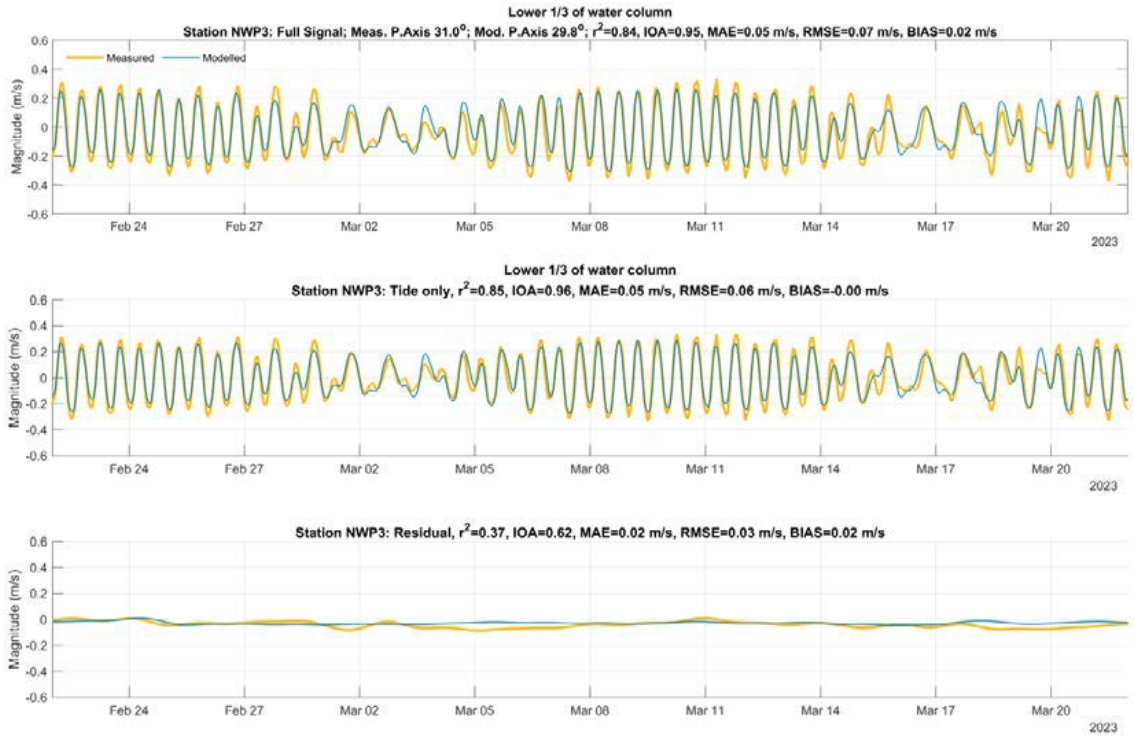


Figure D-23 Lower 1/3 of the Water Column Currents at NWP3 – 22/02 to 21/03/2023

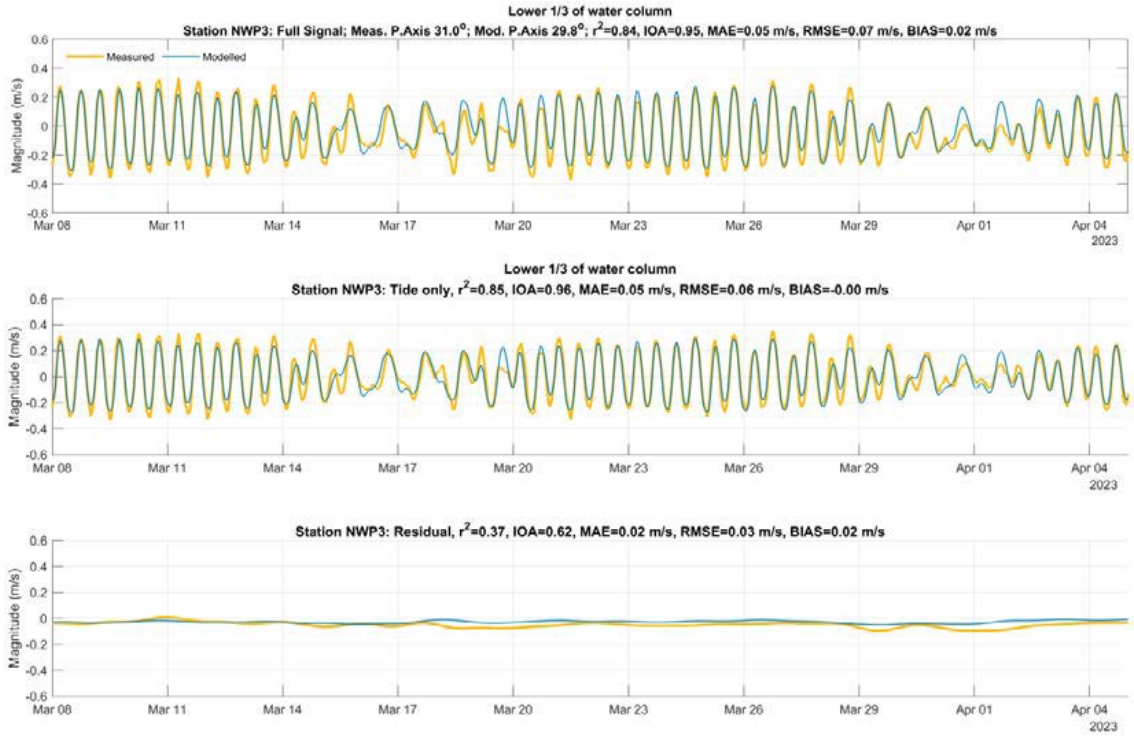


Figure D-24 Lower 1/3 of the Water Column Currents at NWP3 – 08/03 to 05/04/2023

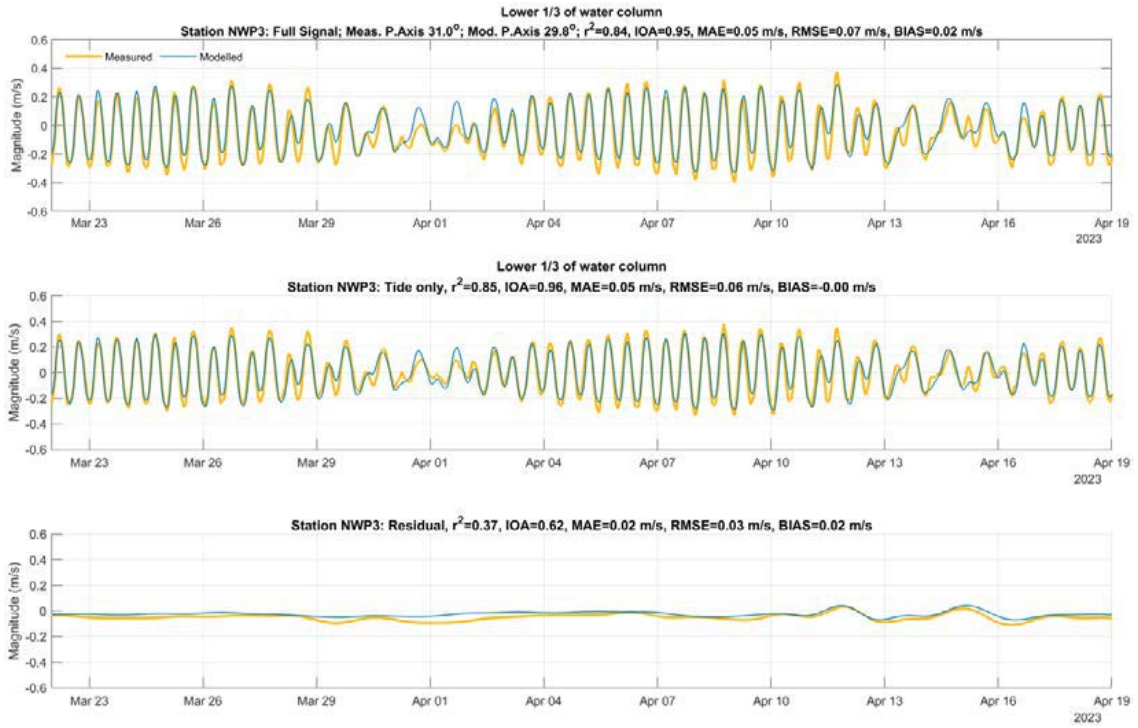


Figure D-25 Lower 1/3 of the Water Column Currents at NWP3 – 22/05 to 19/04/2023

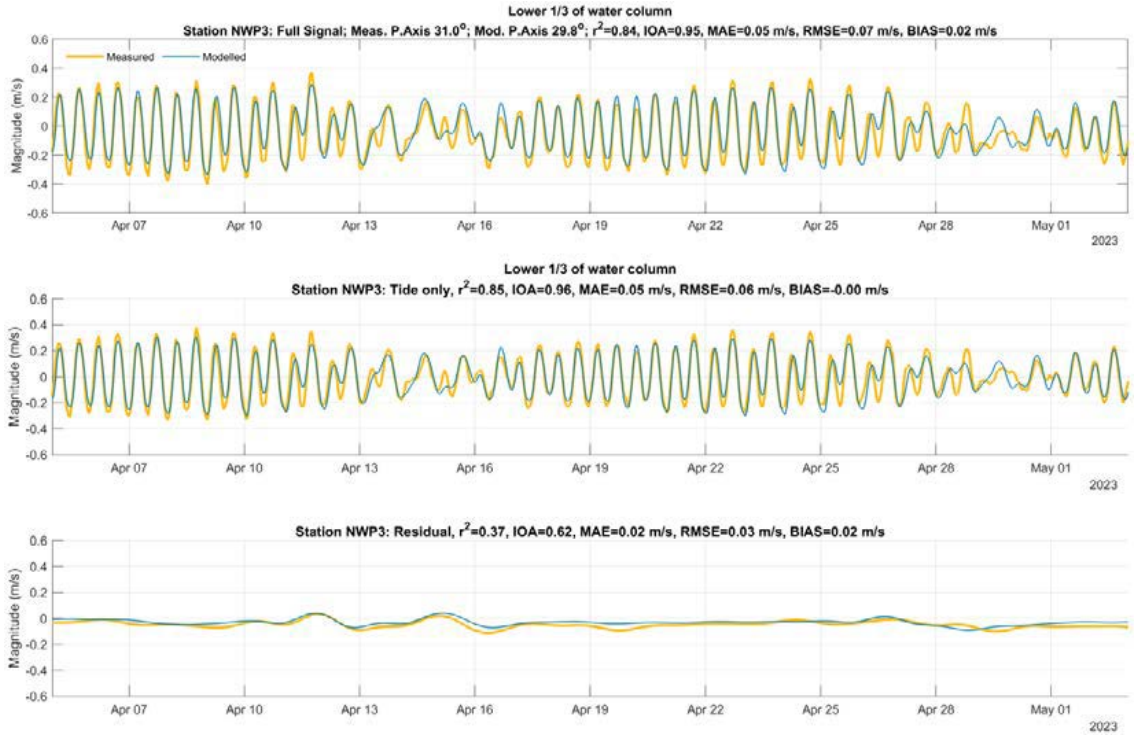


Figure D-26 Lower 1/3 of the Water Column Currents at NWP3 – 06/04 to 02/05/2023

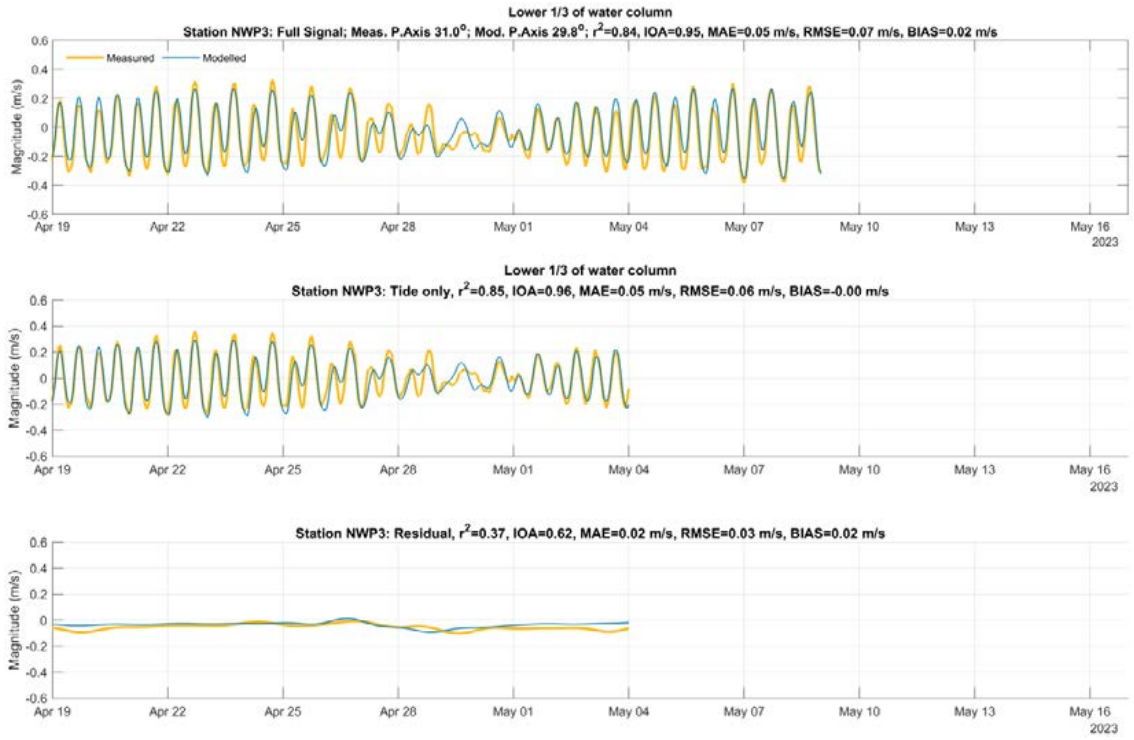


Figure D-27 Lower 1/3 of the Water Column Currents at NWP3 – 19/04 to 09/05/2023

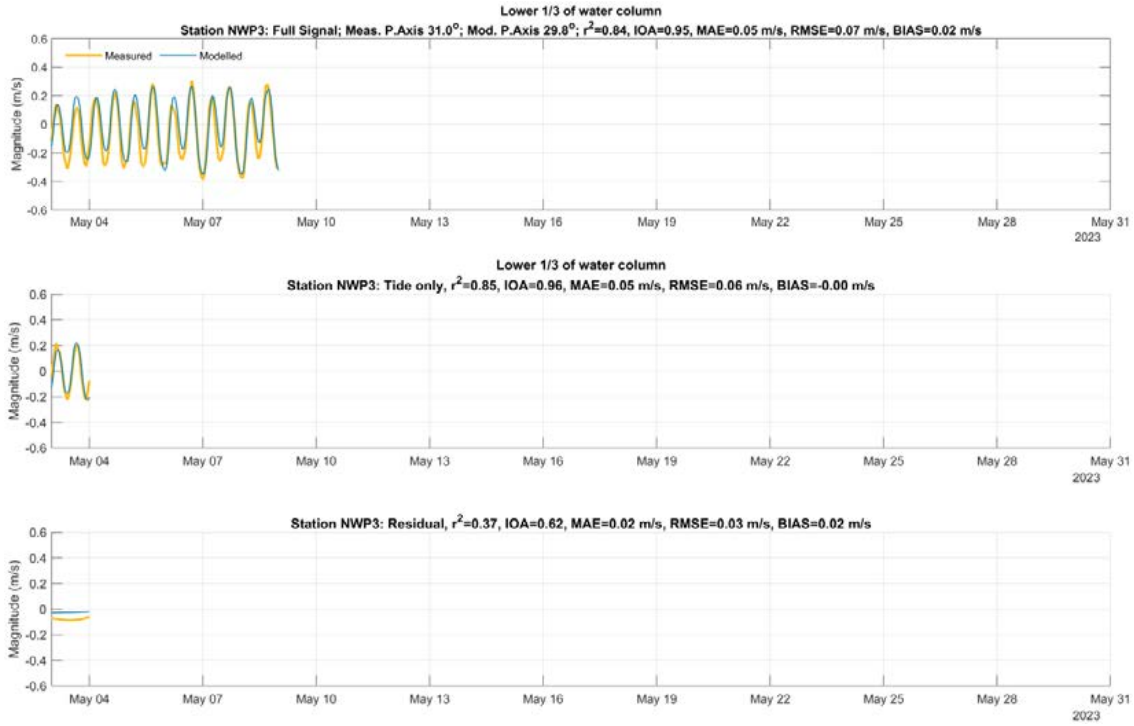


Figure D-28 Lower 1/3 of the Water Column Currents at NWP3 – 03/05 to 09/05/2023

D-2 Comparisons at SAMUSG

D-2-1 Depth-Averaged Currents

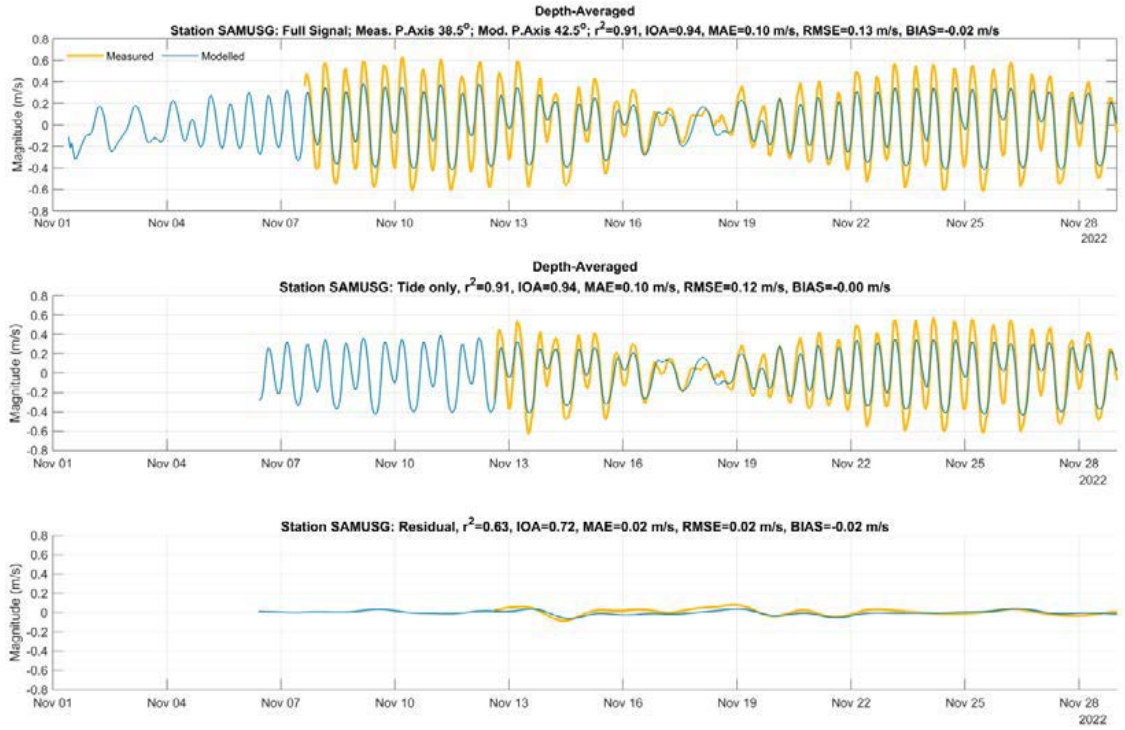


Figure D-29 Depth-Averaged Currents at SAMUSG – 01/11 to 29/11/2022

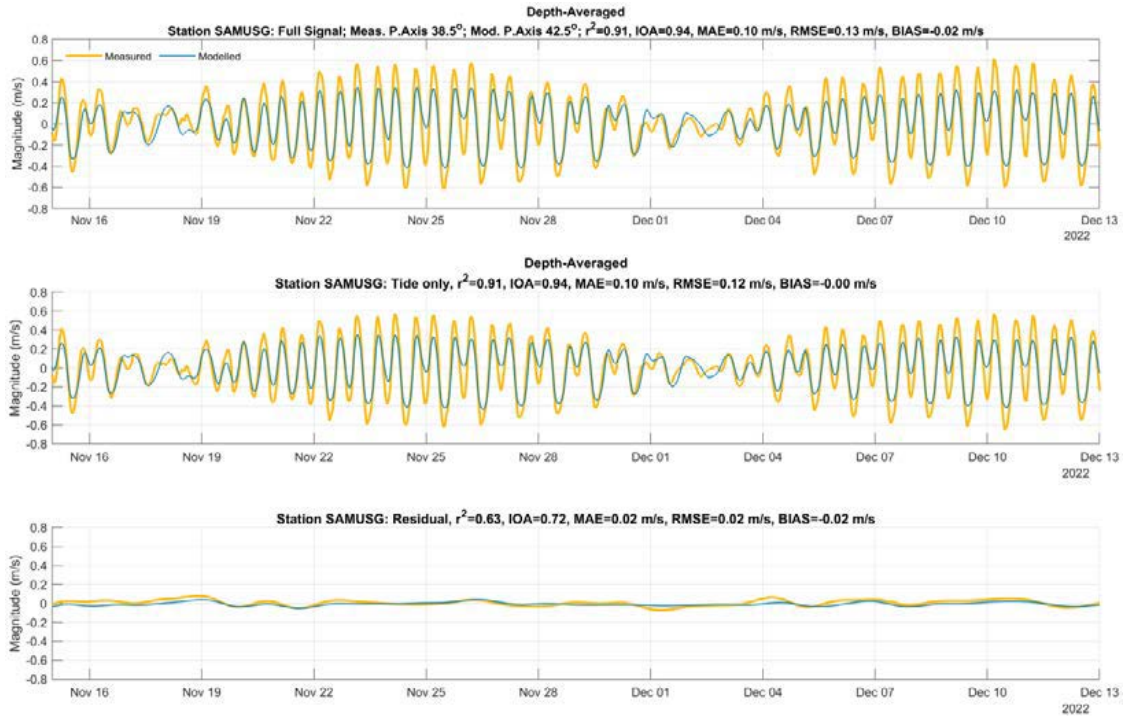


Figure D-30 Depth-Averaged Currents at SAMUSG – 15/11 to 13/12/2022

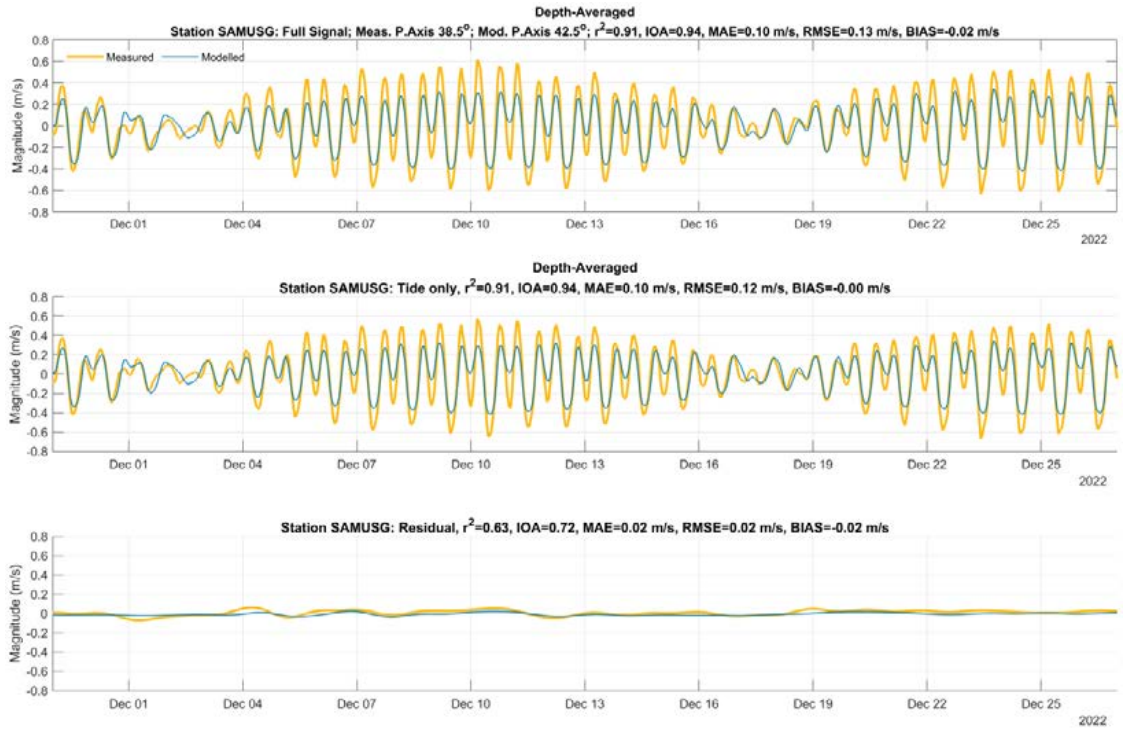


Figure D-31 Depth-Averaged Currents at SAMUSG – 30/11 to 26/12/2022

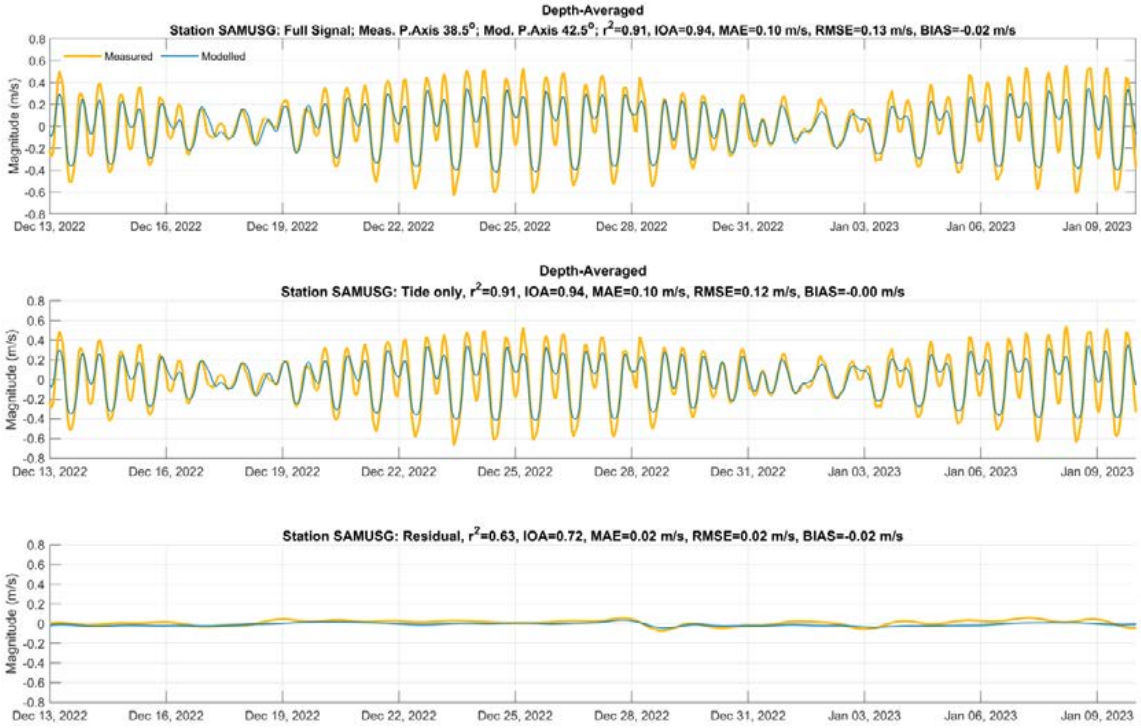


Figure D-32 Depth-Averaged Currents at SAMUSG – 13/12/2022 to 10/01/2023

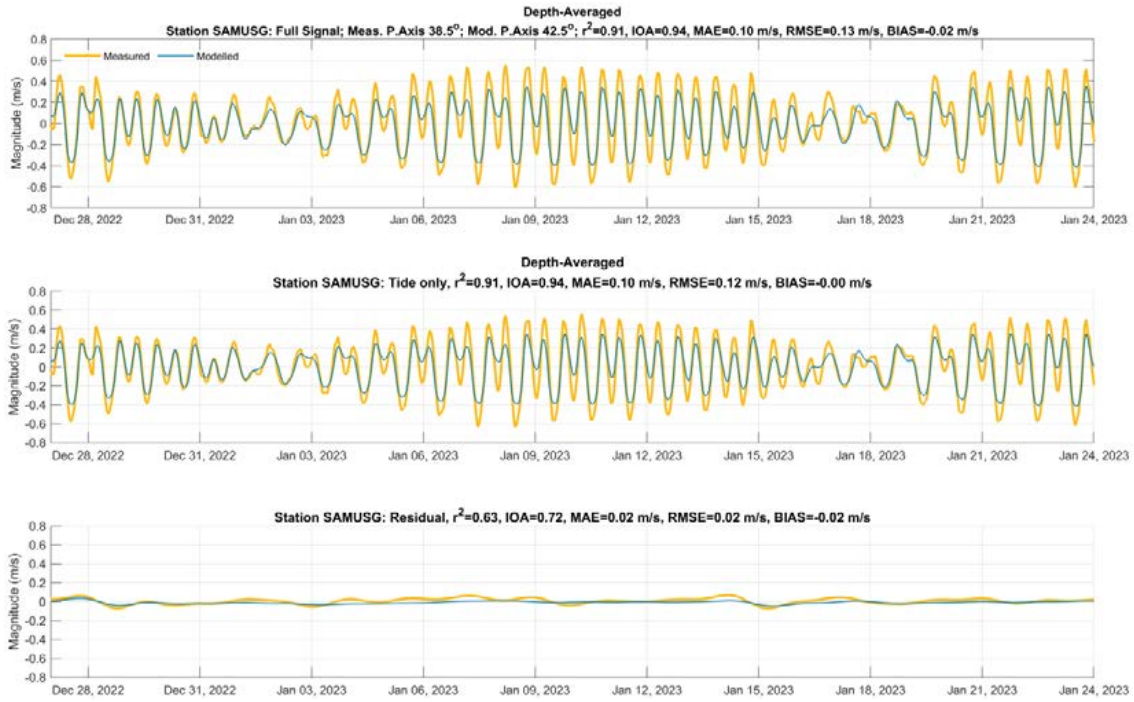


Figure D-33 Depth-Averaged Currents at SAMUSG – 25/12/2022 to 24/01/2023

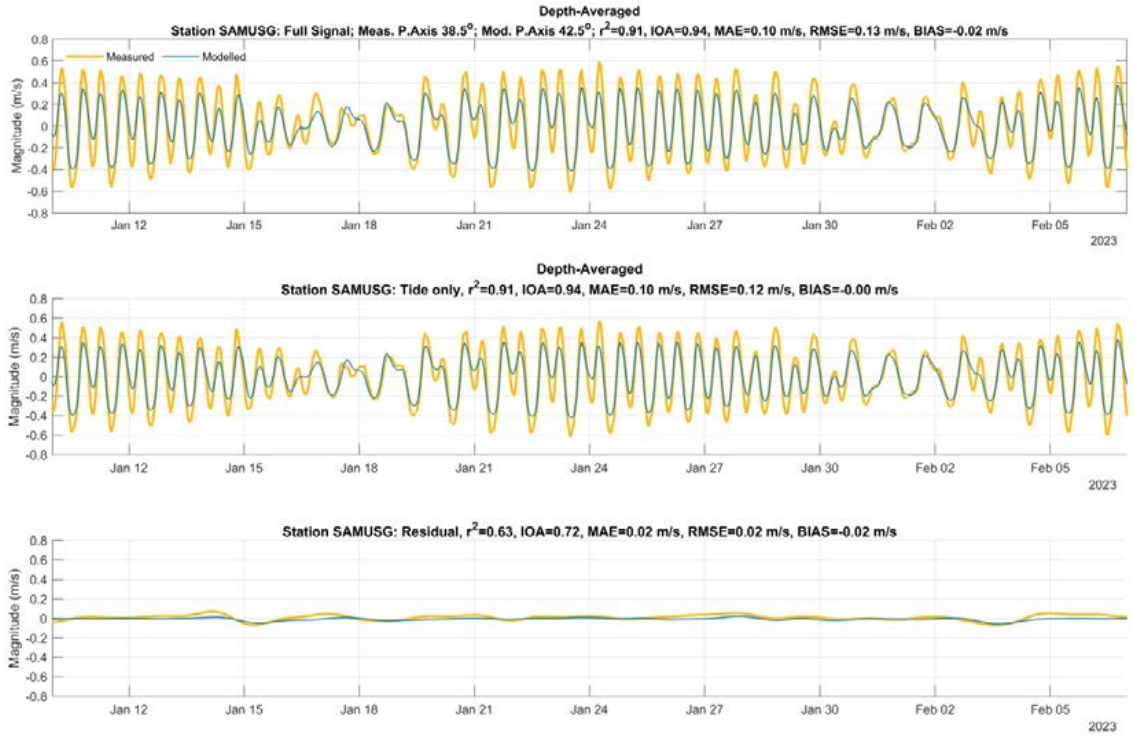


Figure D-34 Depth-Averaged Currents at SAMUSG – 11/01 to 06/02/2023

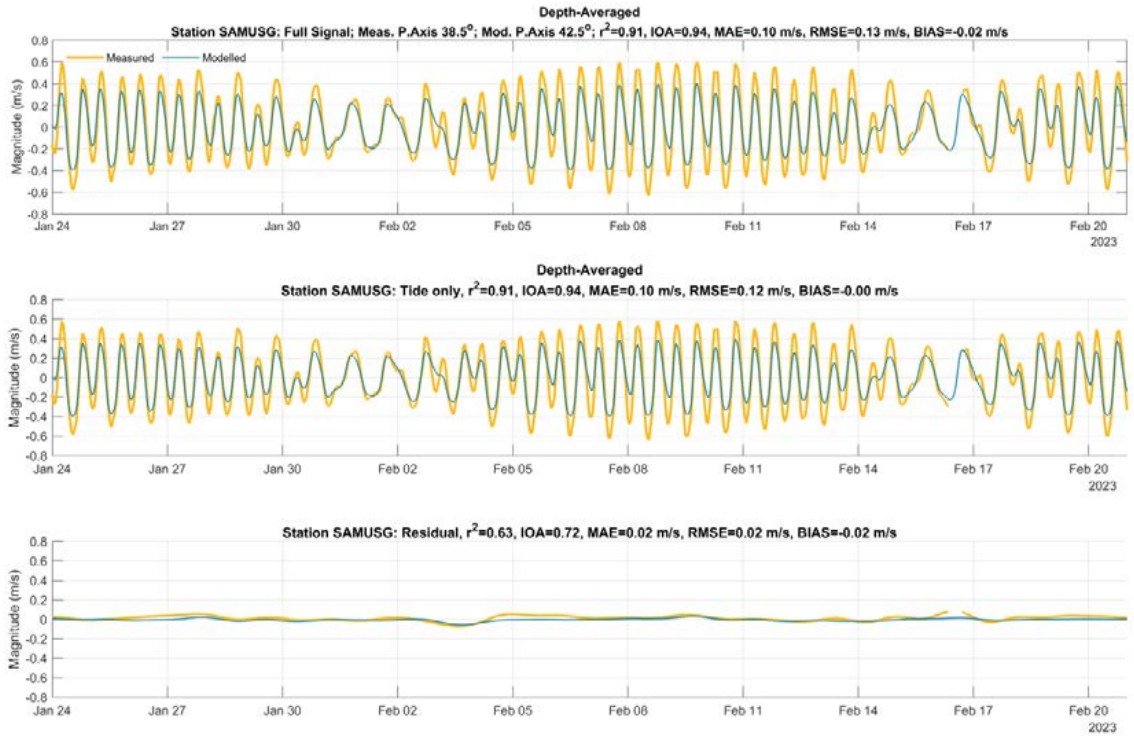


Figure D-35 Depth-Averaged Currents at SAMUSG – 24/01 to 21/02/2023

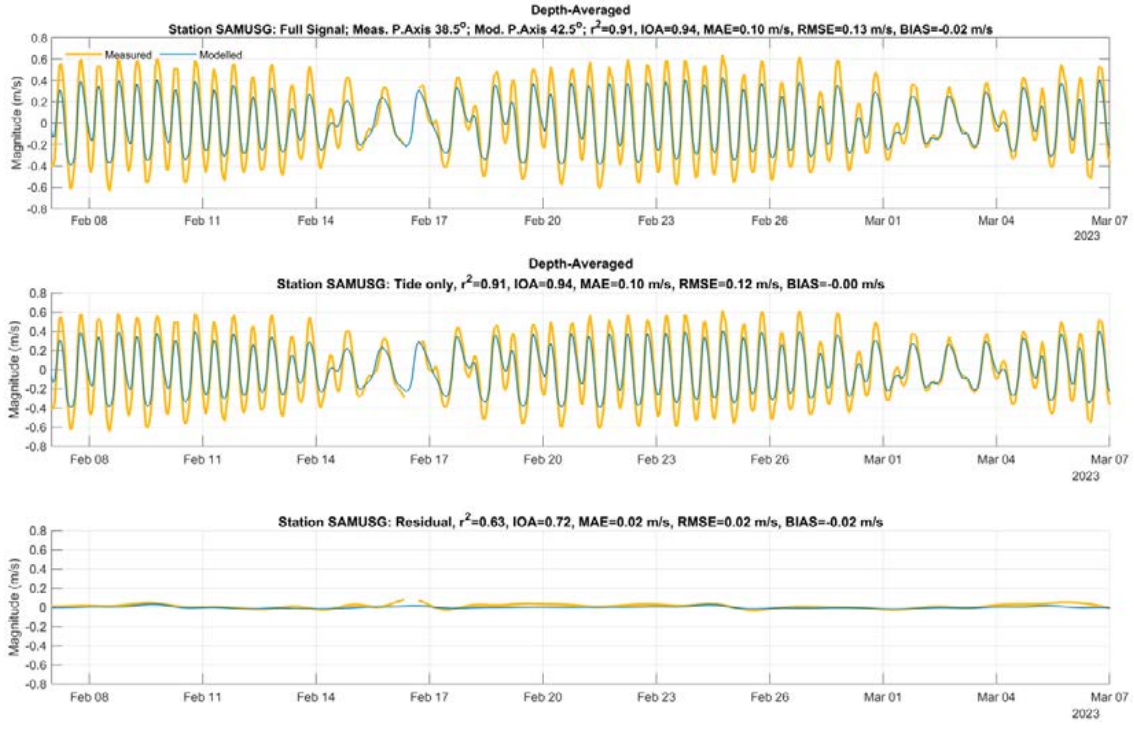


Figure D-36 Depth-Averaged Currents at SAMUSG – 07/02 to 07/03/2023

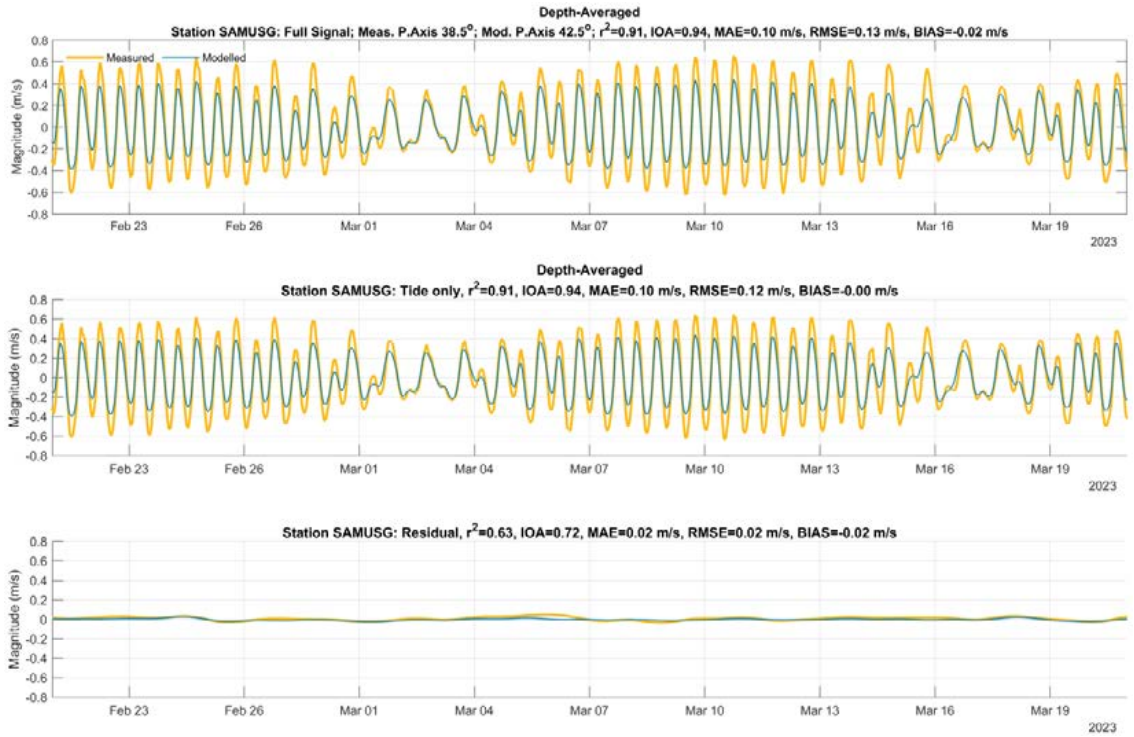


Figure D-37 Depth-Averaged Currents at SAMUSG – 22/02 to 20/03/2023

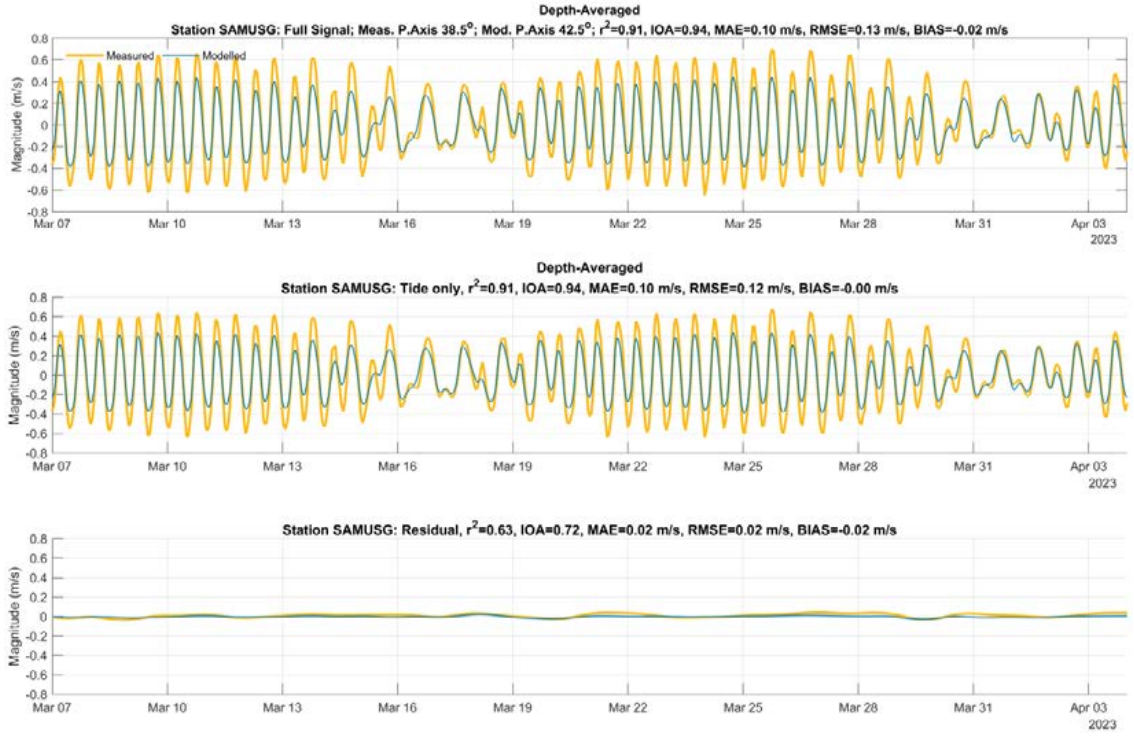


Figure D-38 Depth-Averaged Currents at SAMUSG – 07/03 to 04/04/2023

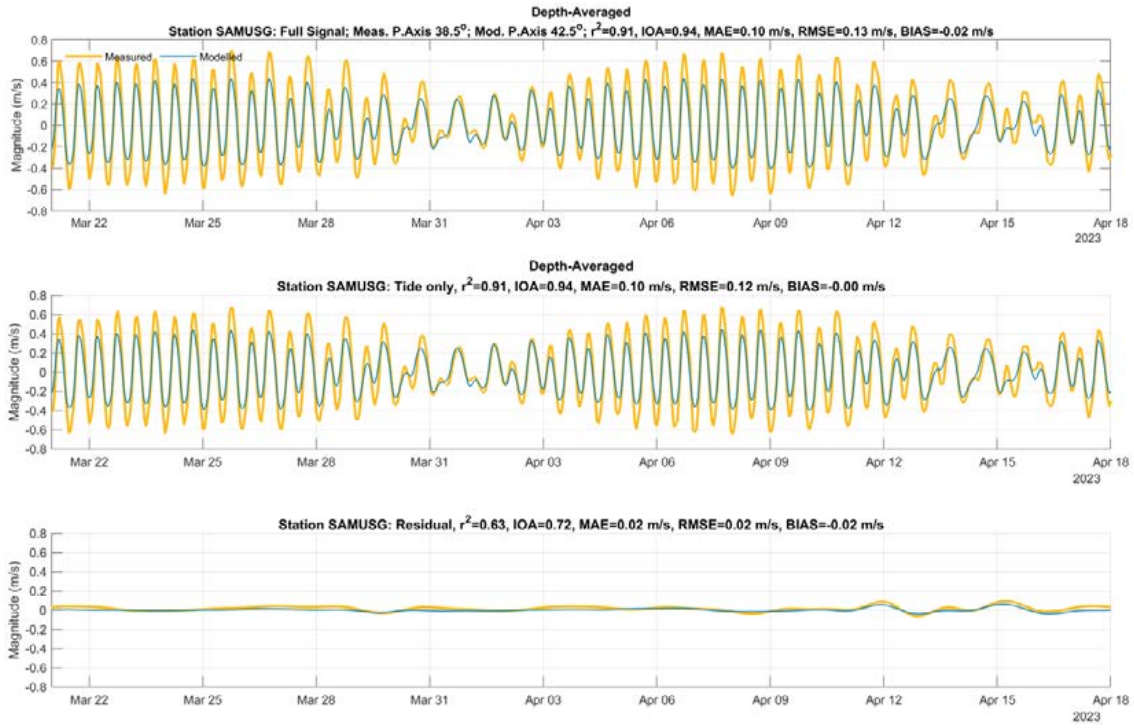


Figure D-39 Depth-Averaged Currents at SAMUSG – 21/03 to 18/04/2023

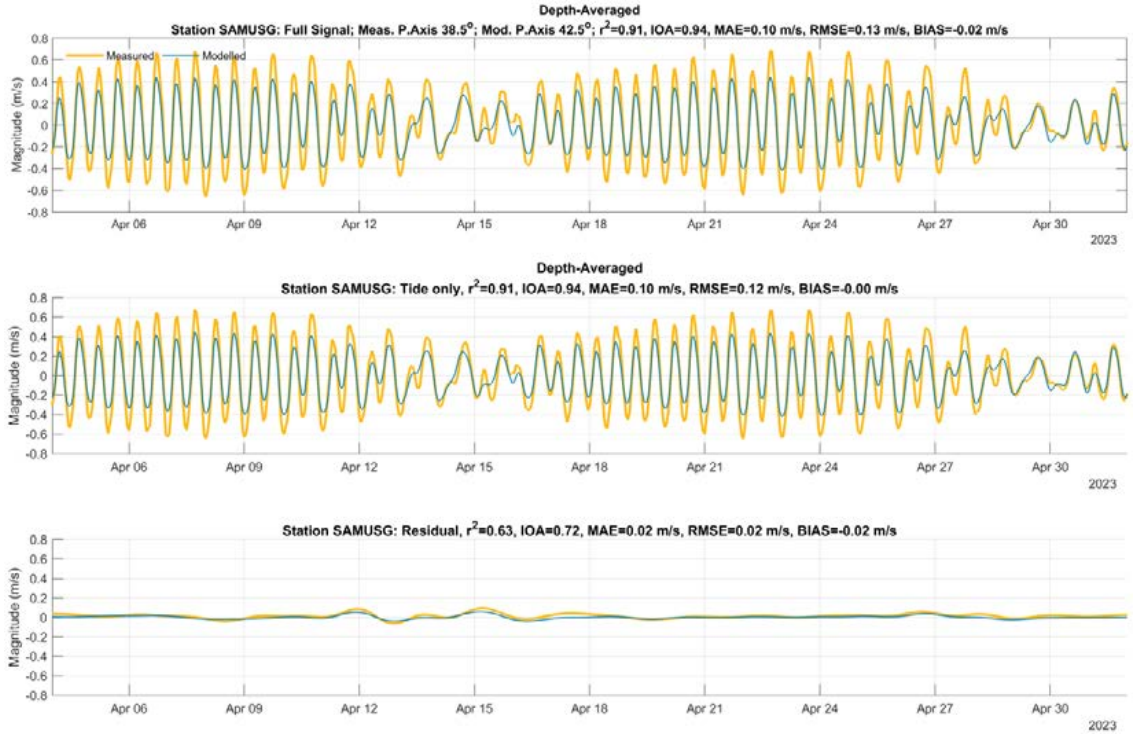


Figure D-40 Depth-Averaged Currents at SAMUSG – 05/04 to 1/5/2023

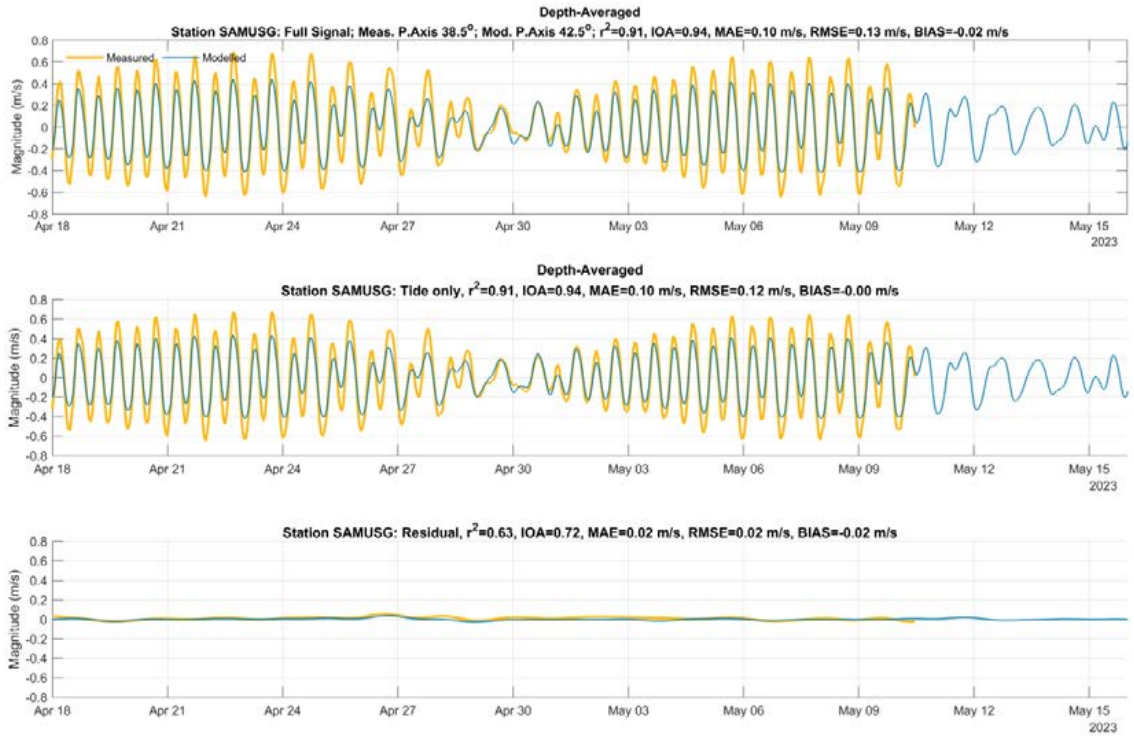


Figure D-41 Depth-Averaged Currents at SAMUSG – 18/04 to 16/05/2023

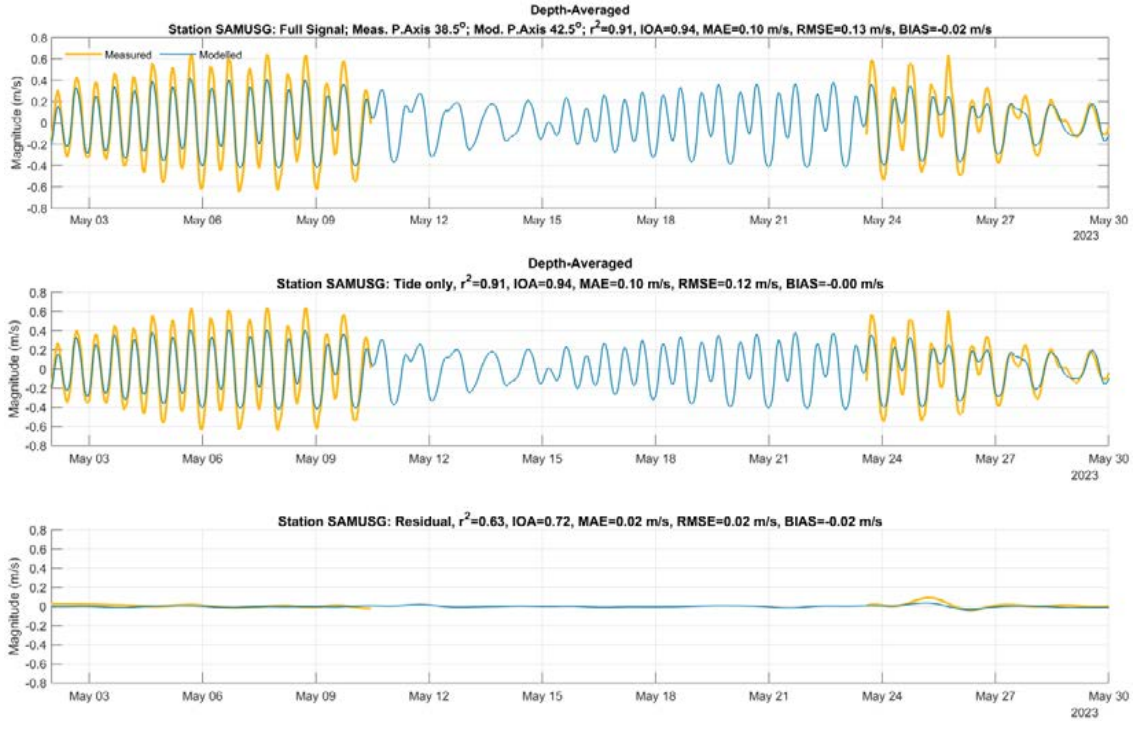


Figure D-42 Depth-Averaged Currents at SAMUSG – 02/05 to 30/05/2023

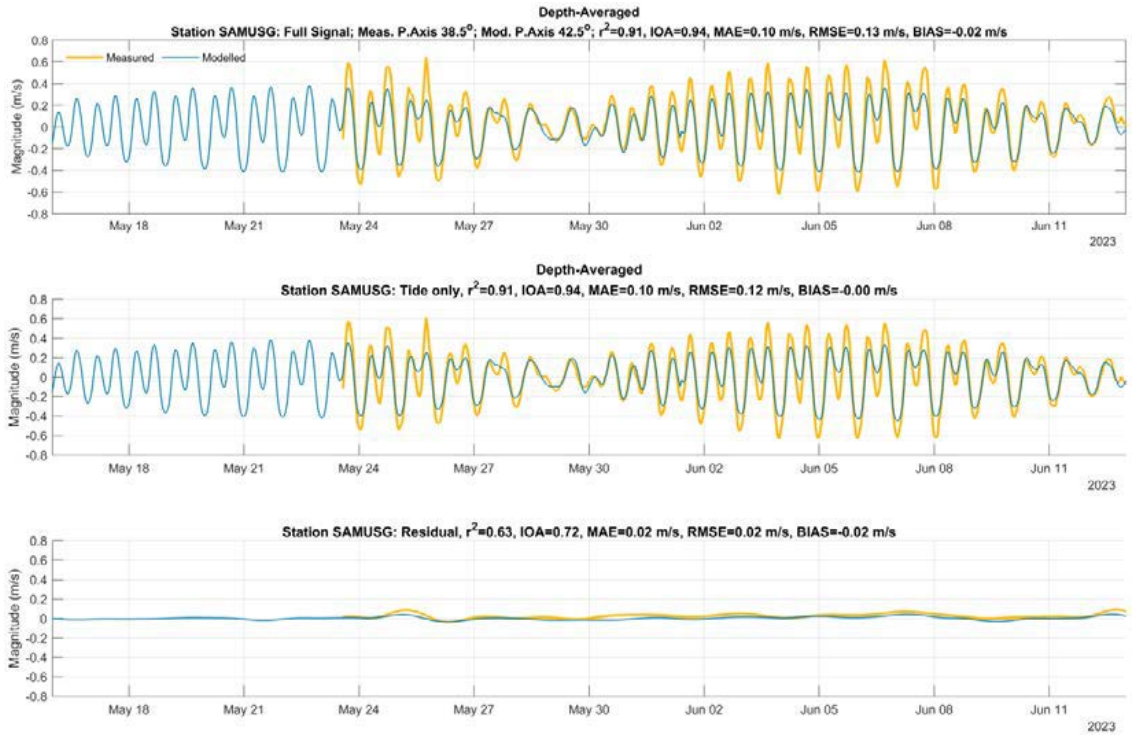


Figure D-43 Depth-Averaged Currents at SAMUSG – 17/05 to 12/06/2023

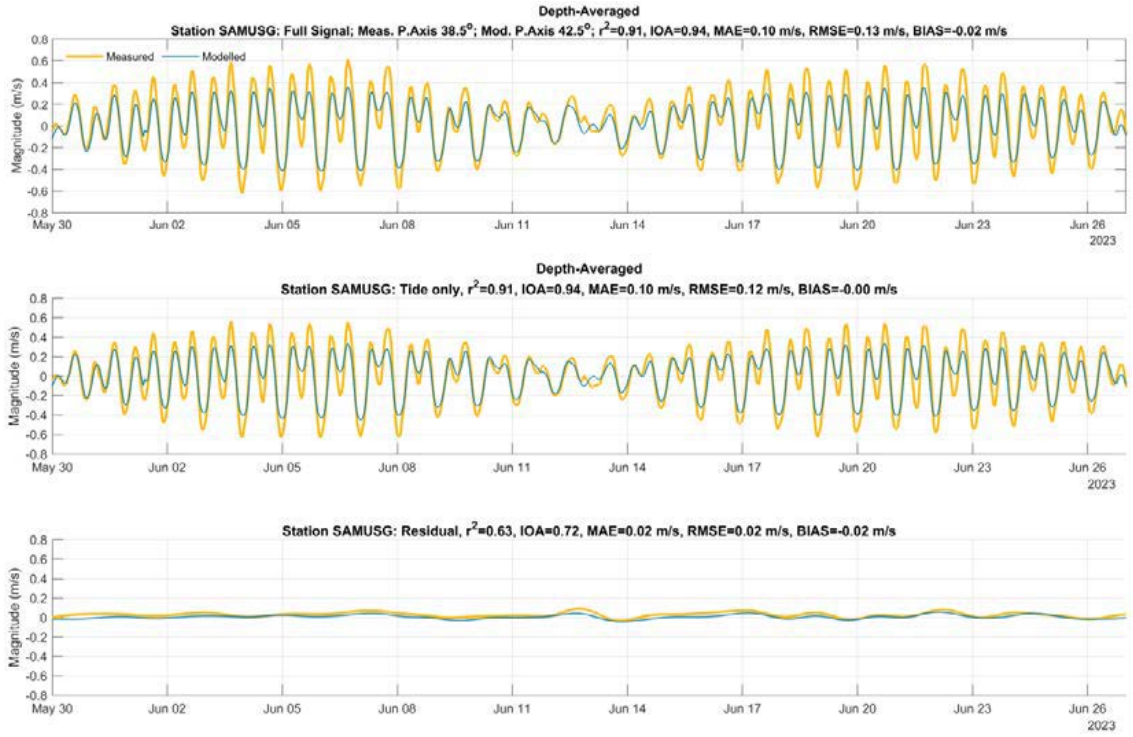


Figure D-44 Depth-Averaged Currents at SAMUSG – 30/05 to 27/06/2023

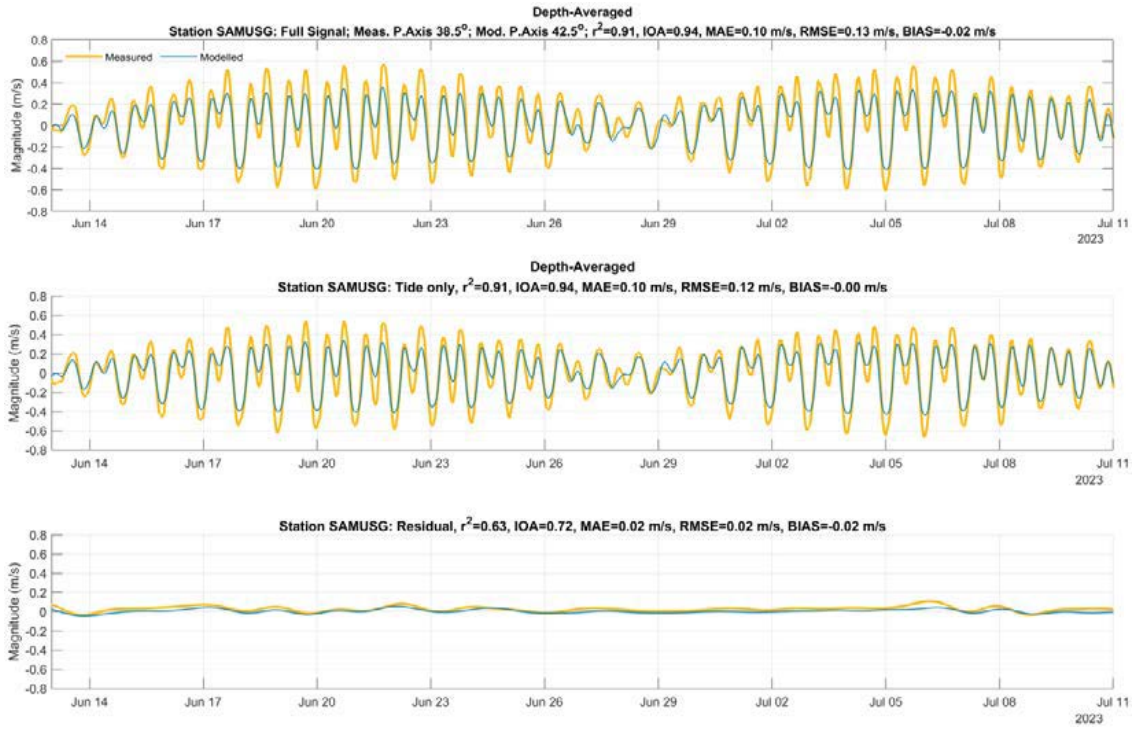


Figure D-45 Depth-Averaged Currents at SAMUSG – 13/06 to 11/07/2023

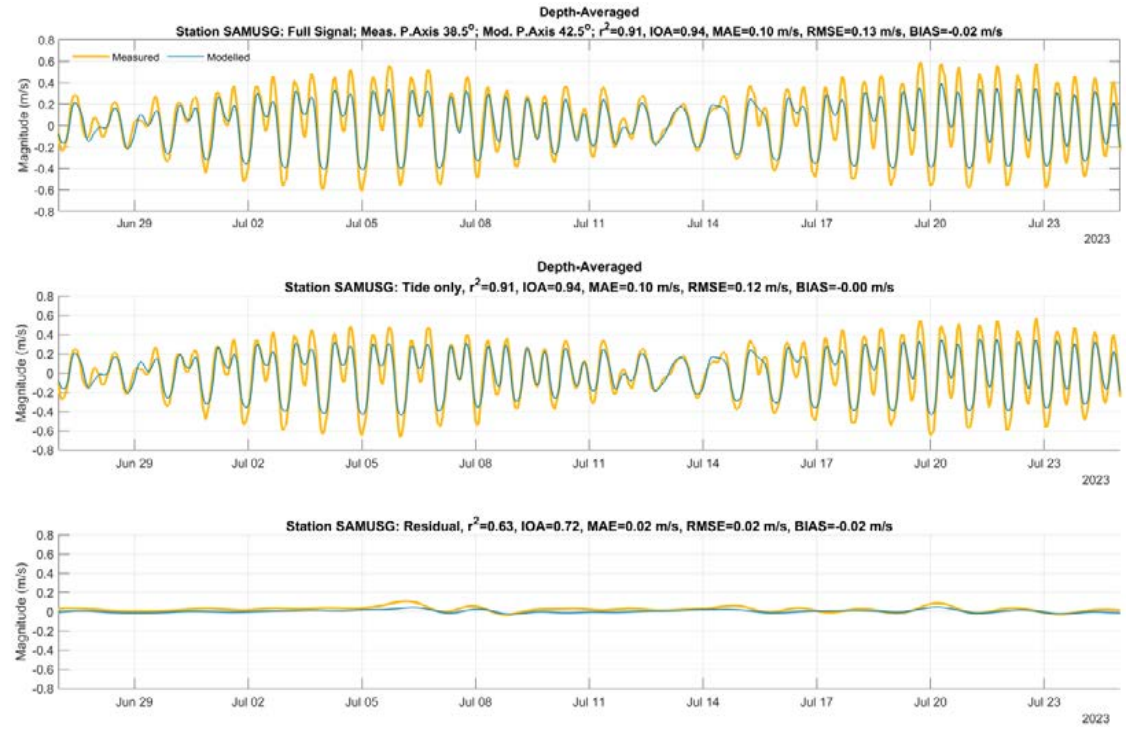


Figure D-46 Depth-Averaged Currents at SAMUSG – 28/06 to 24/07/2023

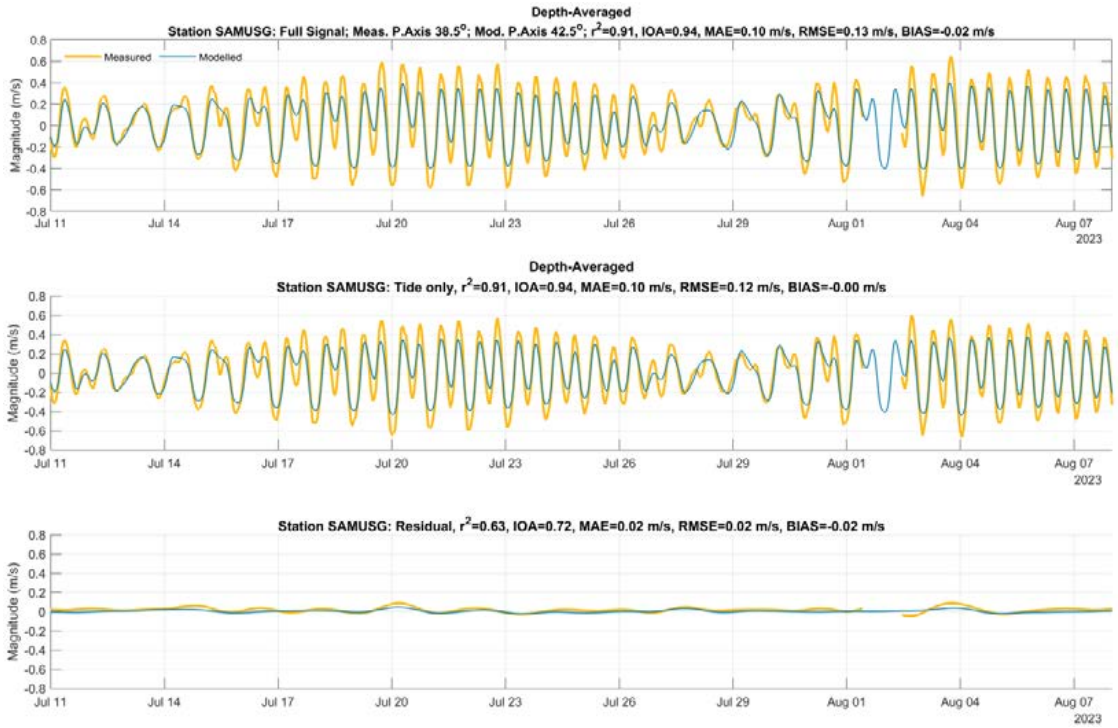


Figure D-47 Depth-Averaged Currents at SAMUSG – 11/07 to 08/08/2023

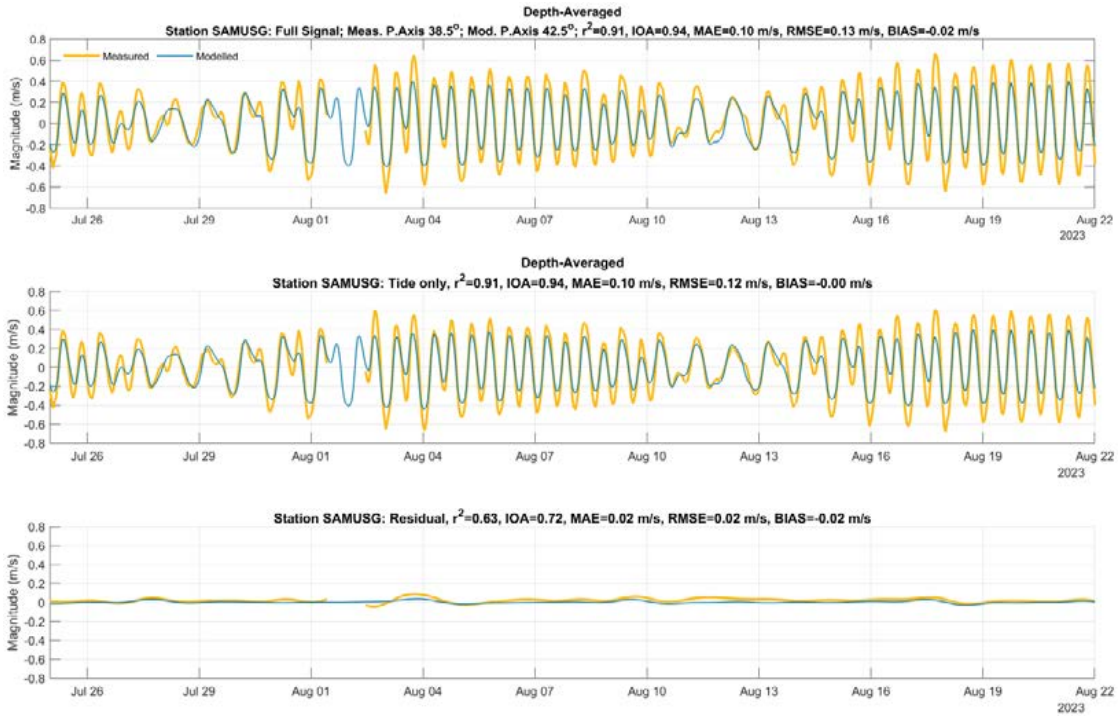


Figure D-48 Depth-Averaged Currents at SAMUSG – 25/07 to 22/08/2023

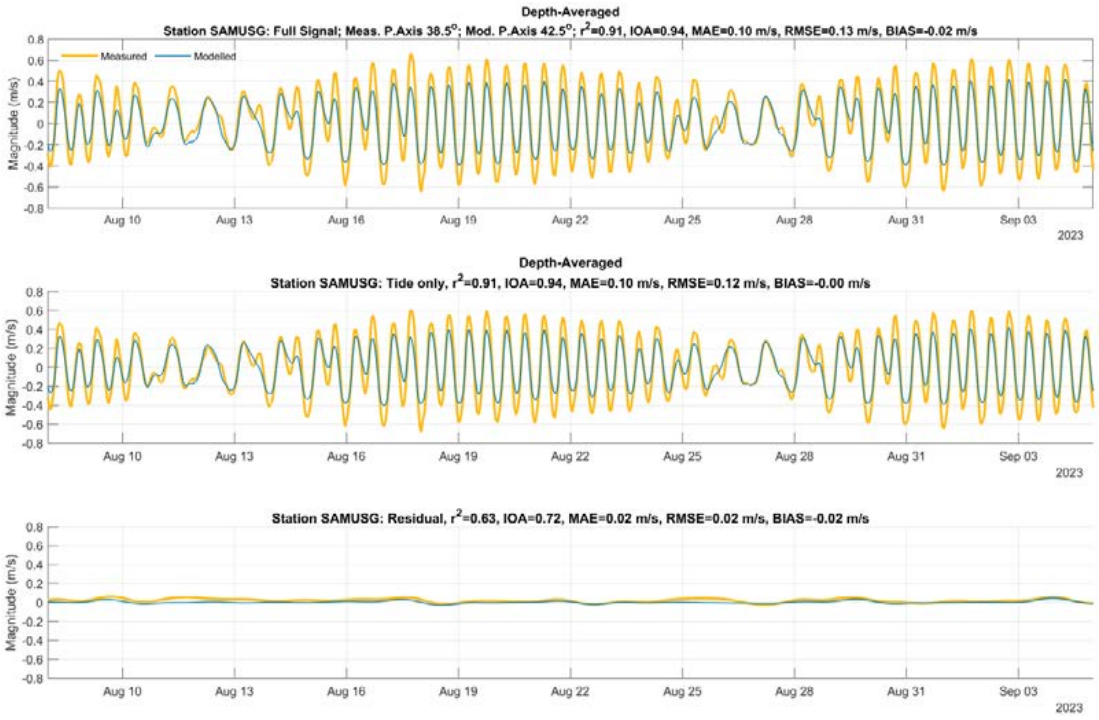


Figure D-49 Depth-Averaged Currents at SAMUSG – 09/08 to 04/09/2023

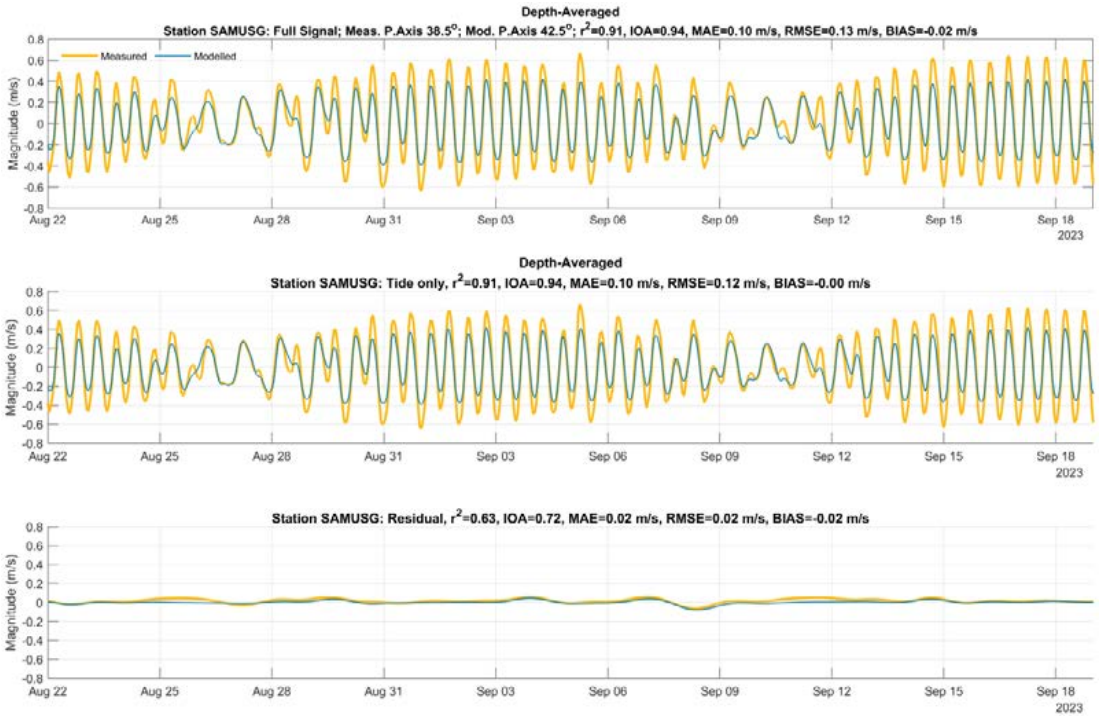


Figure D-50 Depth-Averaged Currents at SAMUSG – 22/08 to 19/09/2023

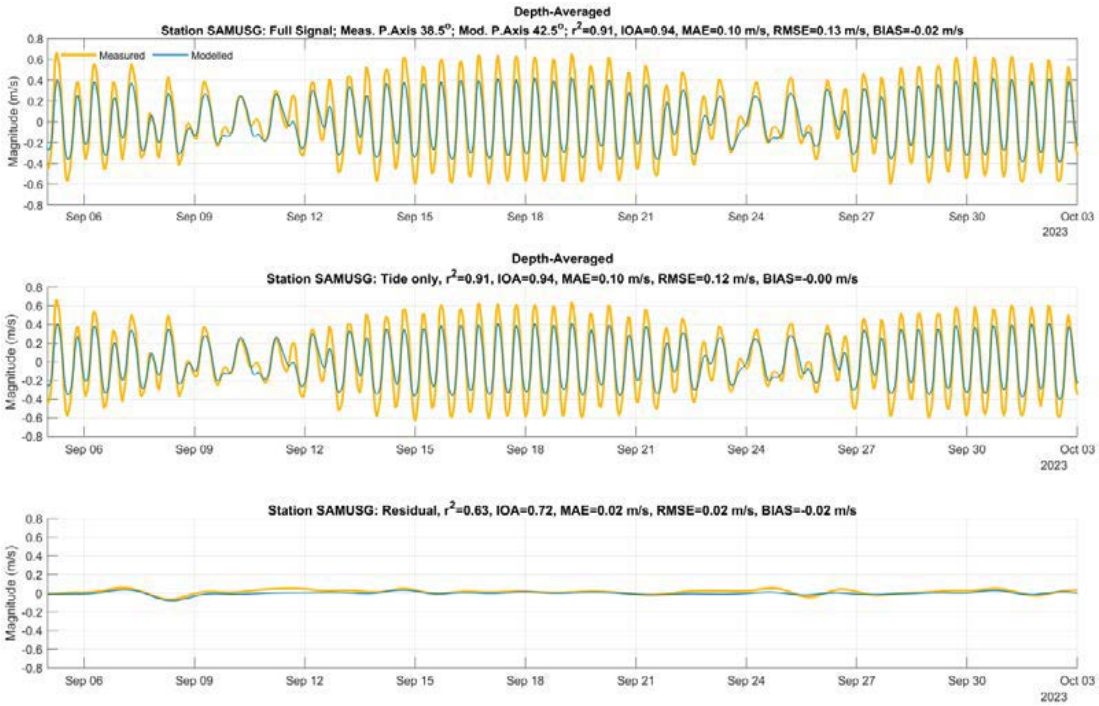


Figure D-51 Depth-Averaged Currents at SAMUSG – 05/09 to 03/10/2023

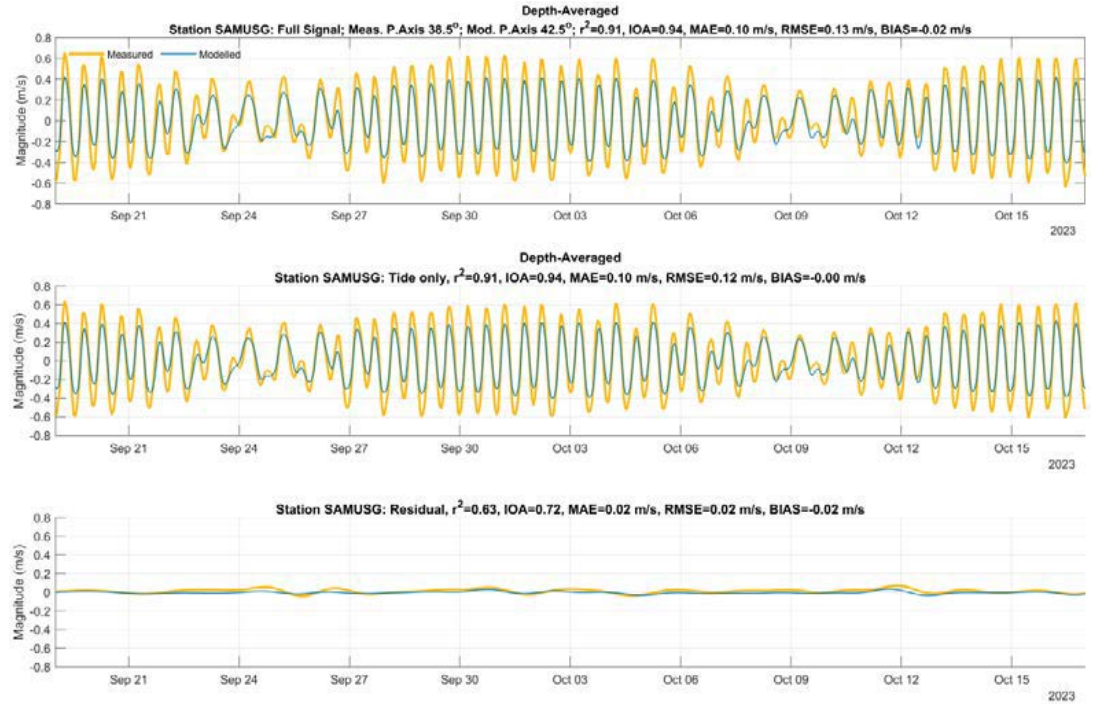


Figure D-52 Depth-Averaged Currents at SAMUSG – 20/09 to 16/10/2023

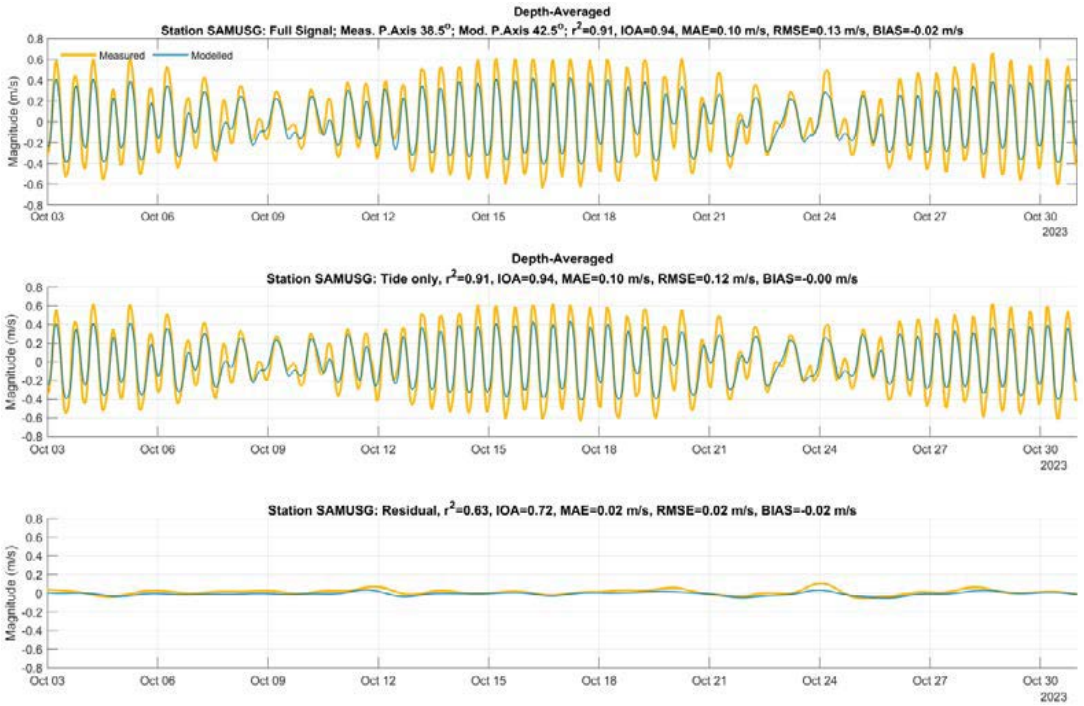


Figure D-53 Depth-Averaged Currents at SAMUSG – 03/10 to 31/10/2023

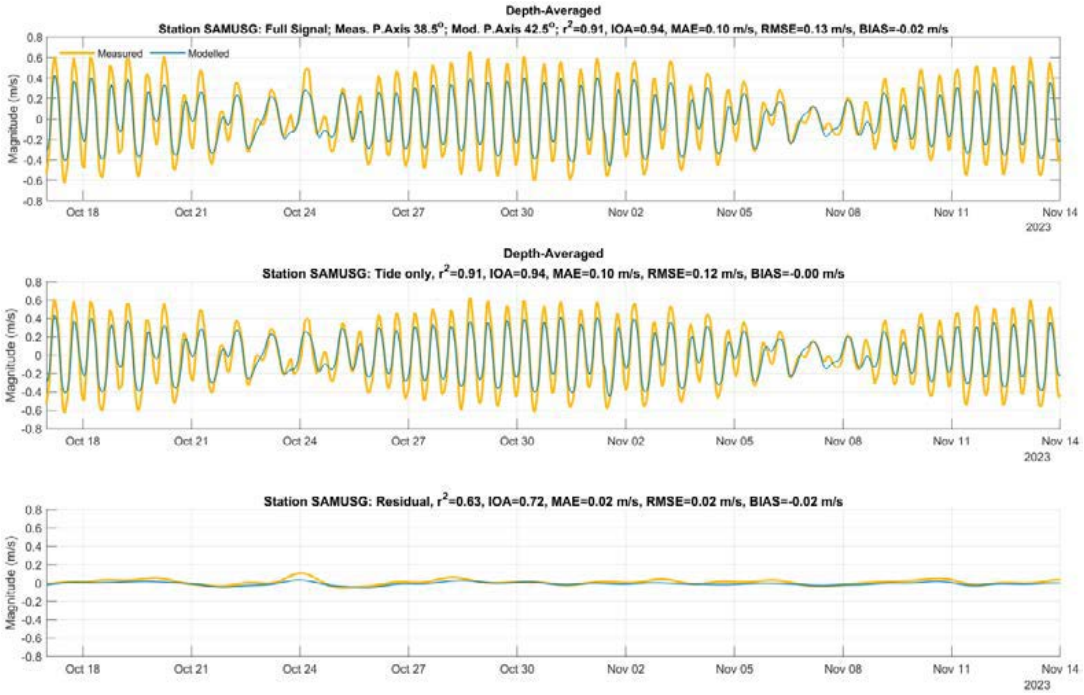


Figure D-54 Depth-Averaged Currents at SAMUSG – 17/10 to 14/11/2023

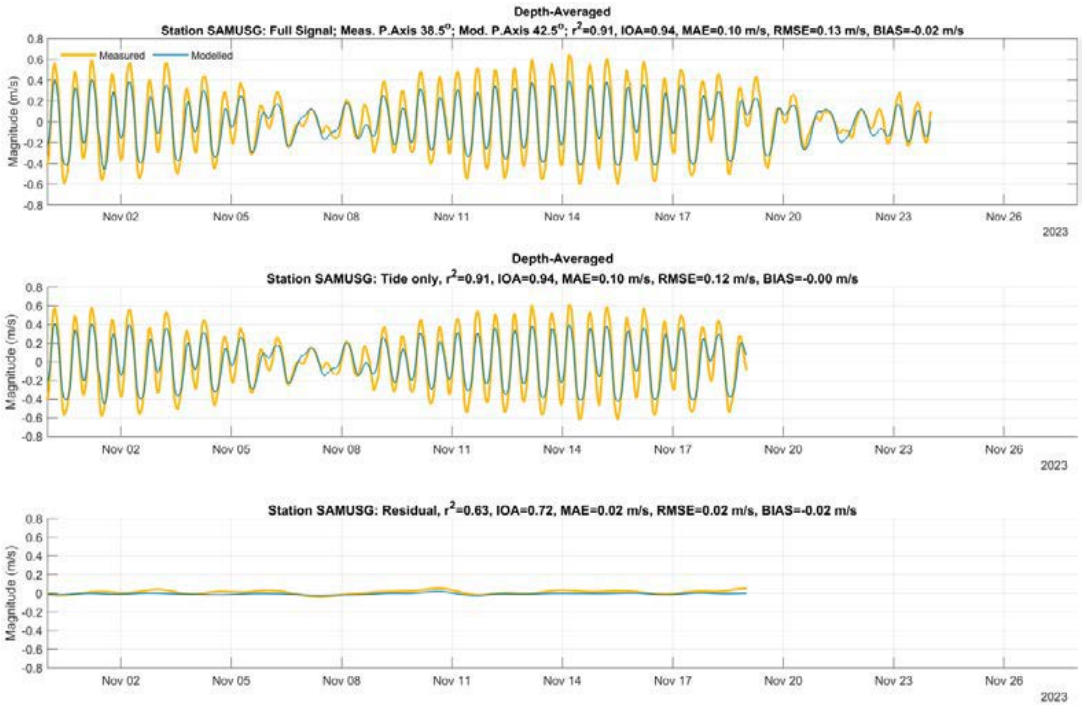


Figure D-55 Depth-Averaged Currents at SAMUSG – 01/11 to 25/11/2023

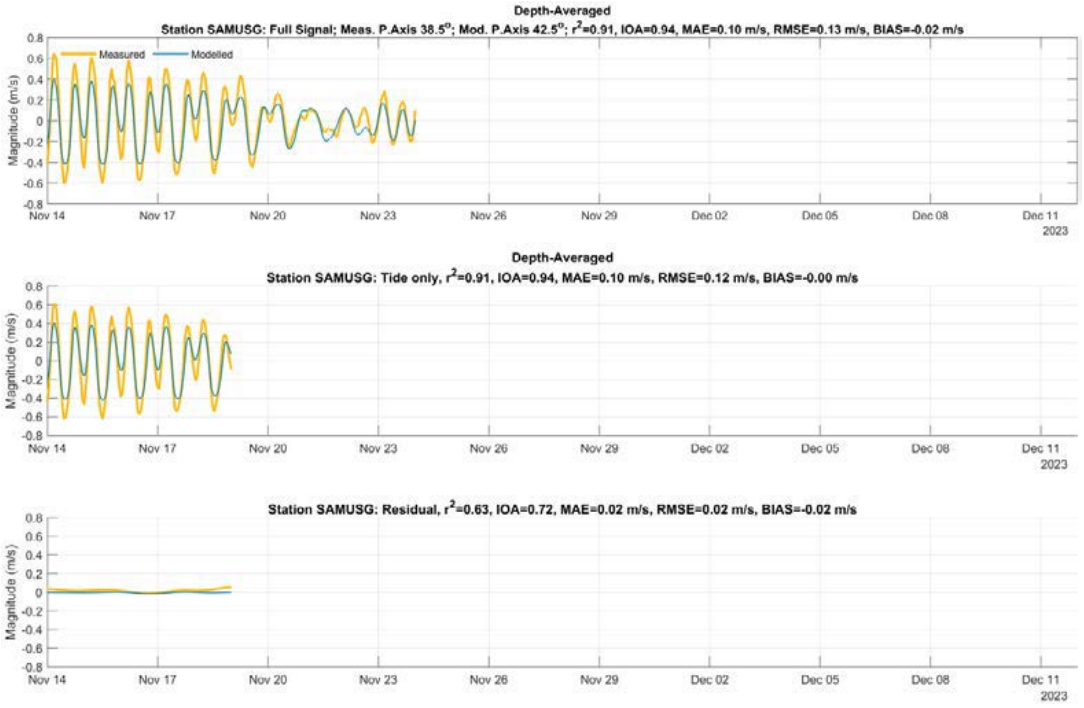


Figure D-56 Depth-Averaged Currents at SAMUSG – 14/11 to 25/11/2023

D-2-2 Lower 1/3 of the Water Column Currents

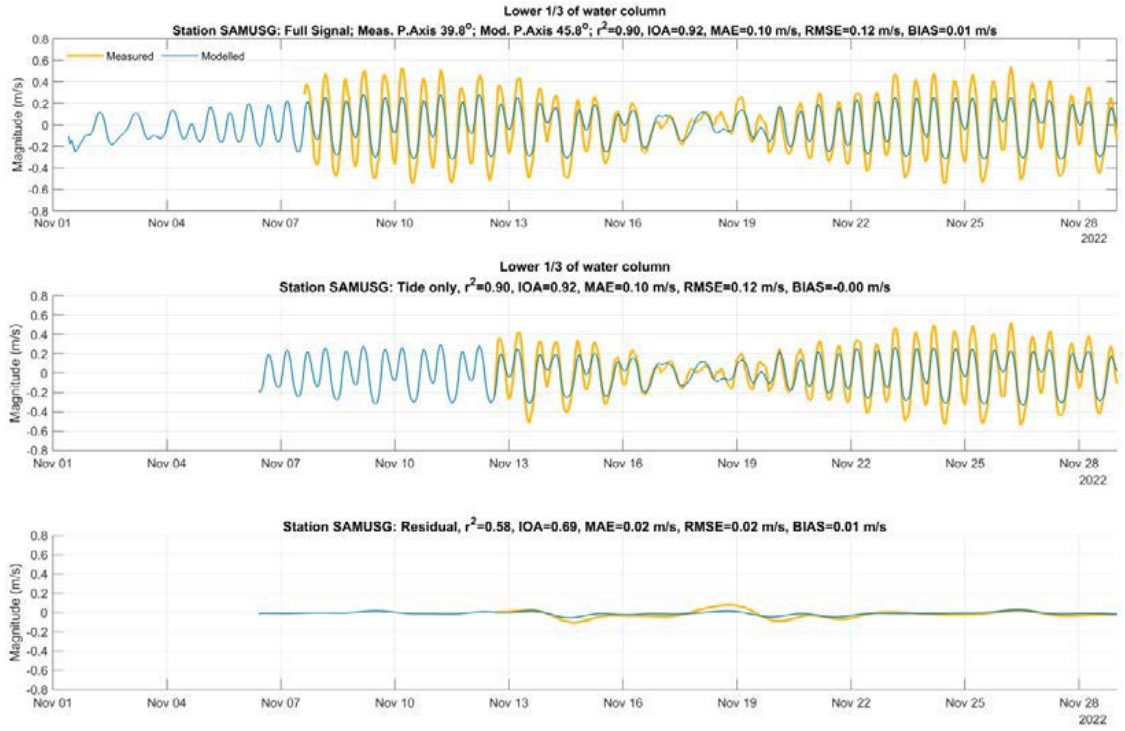


Figure D-57 Lower 1/3 of the Water Column Currents at SAMUSG – 01/11 to 29/11/2022

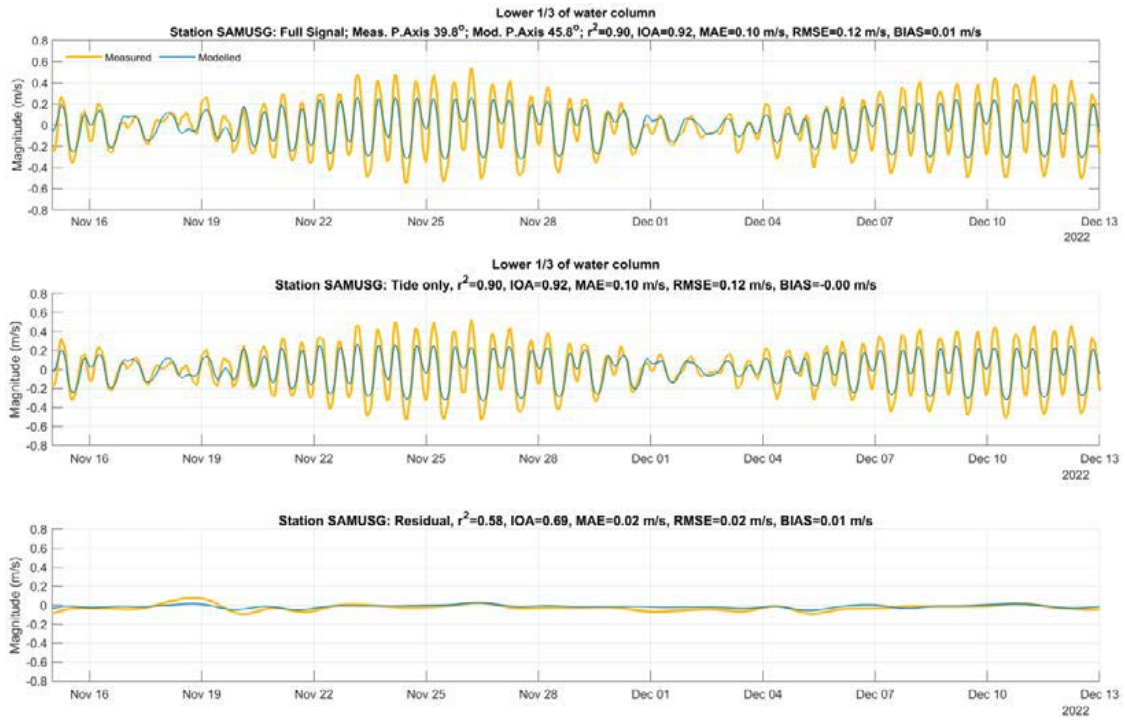


Figure D-58 Lower 1/3 of the Water Column Currents at SAMUSG – 15/11 to 13/12/2022

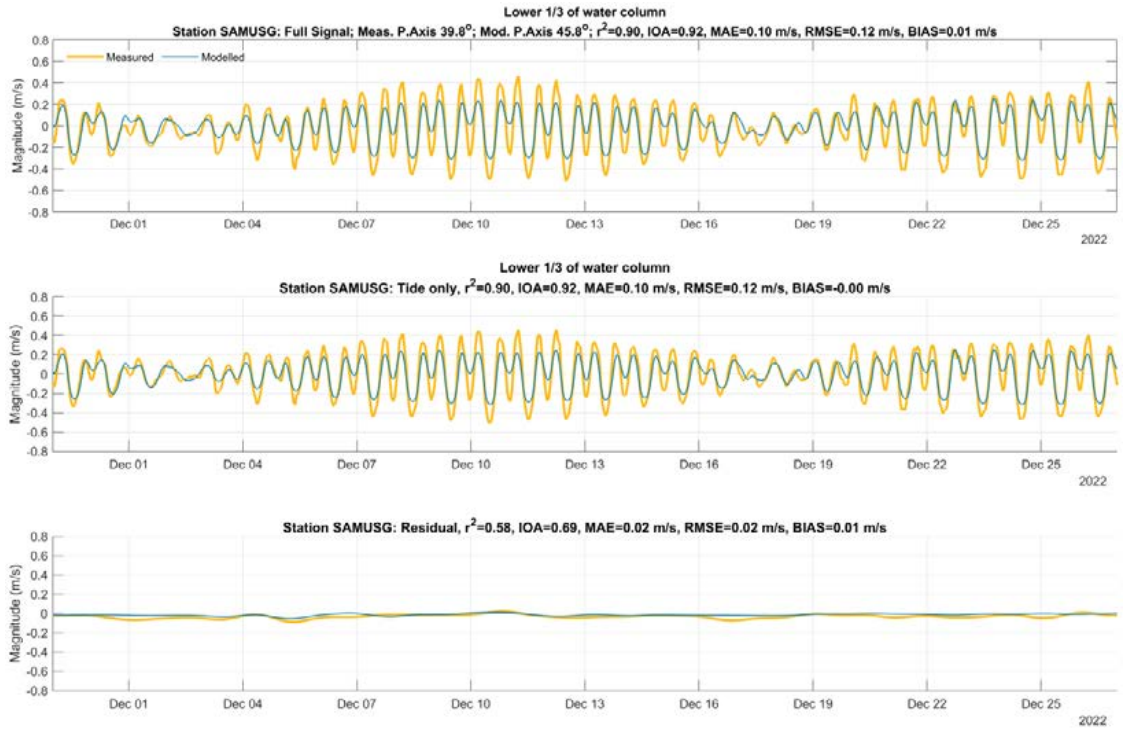


Figure D-59 Lower 1/3 of the Water Column Currents at SAMUSG – 30/11 to 26/12/2022

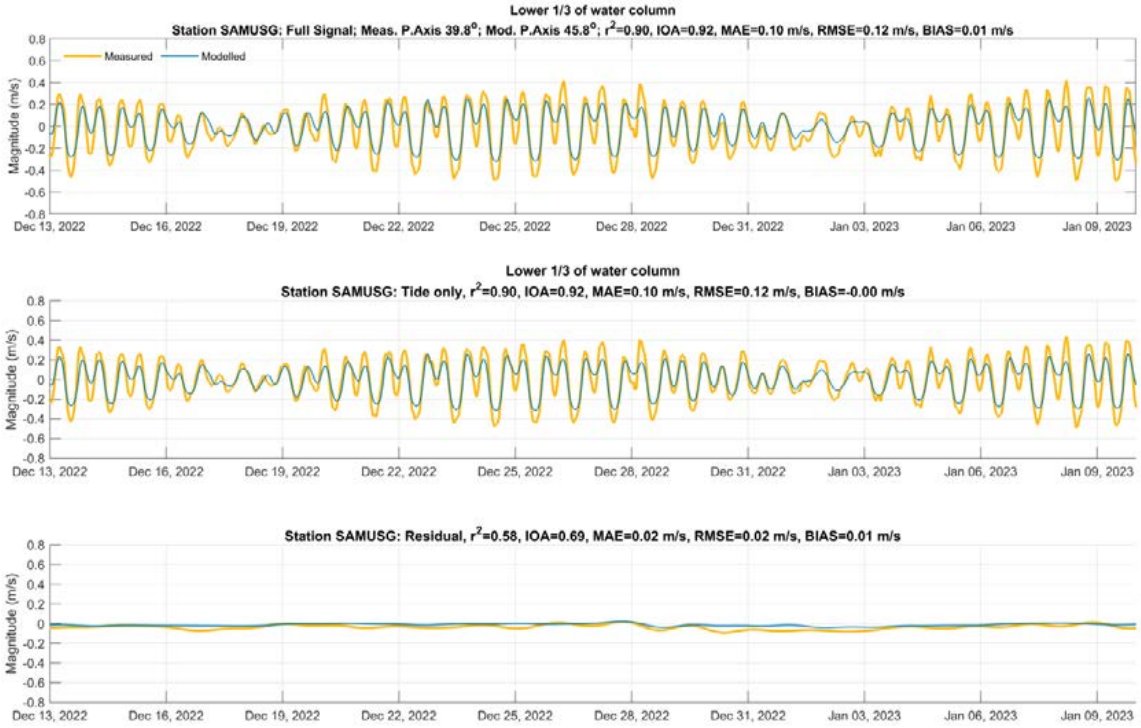


Figure D-60 Lower 1/3 of the Water Column Currents at SAMUSG – 13/12/2022 to 09/01/2023

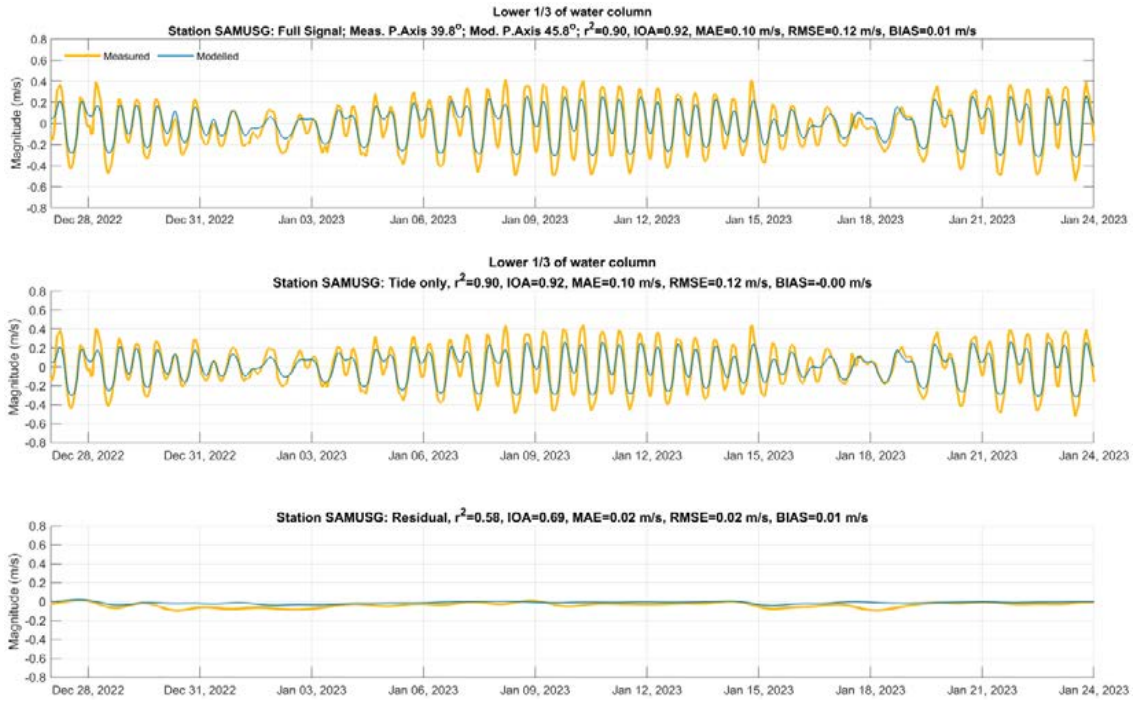


Figure D-61 Lower 1/3 of the Water Column Currents at SAMUSG – 27/12/2022 to 24/01/2023

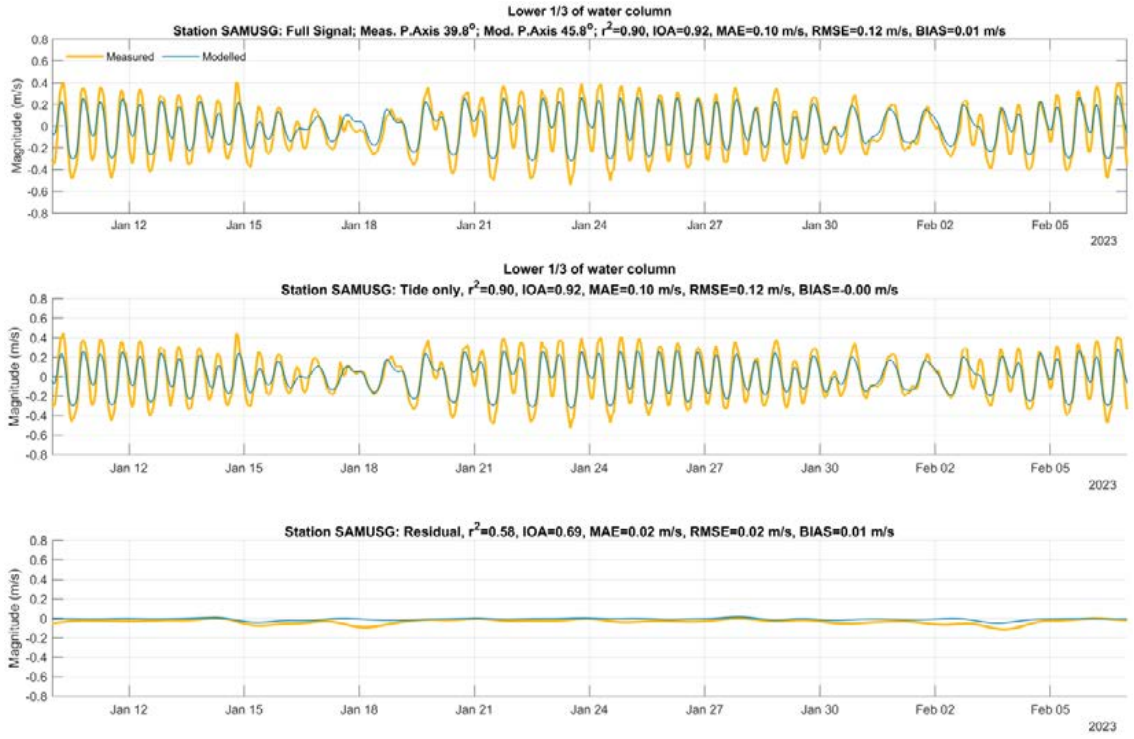


Figure D-62 Lower 1/3 of the Water Column Currents at SAMUSG – 10/01 to 06/02/2023

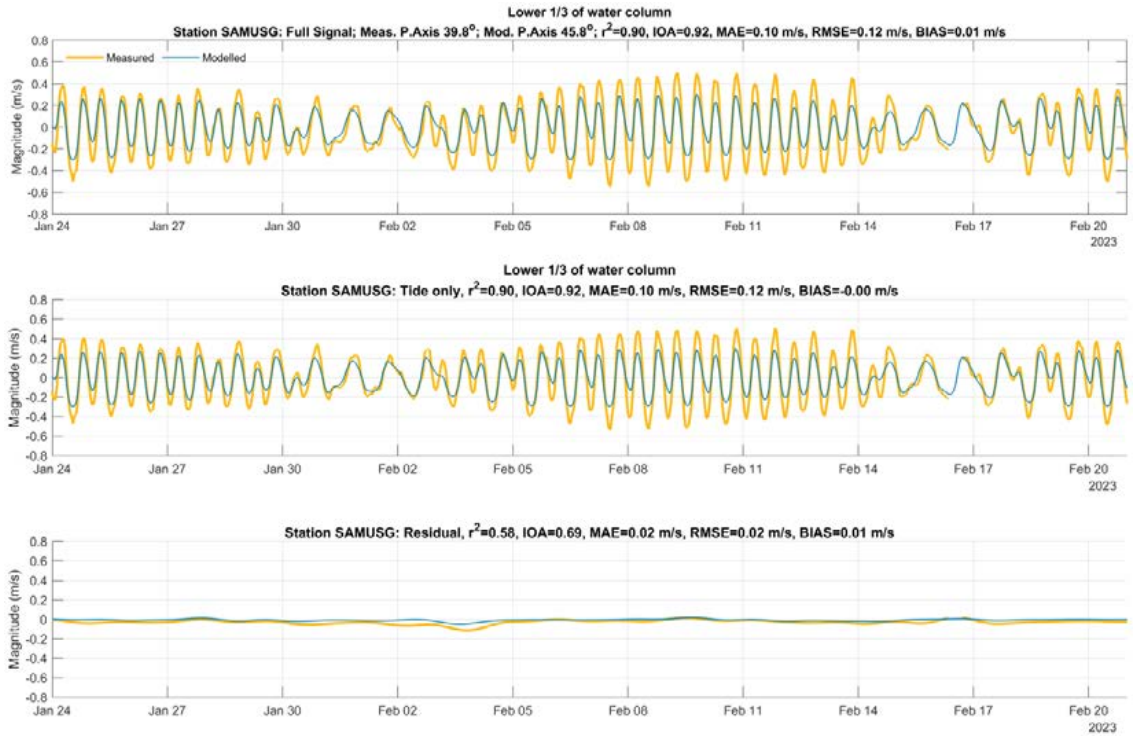


Figure D-63 Lower 1/3 of the Water Column Currents at SAMUSG – 24/01 to 21/02/2023

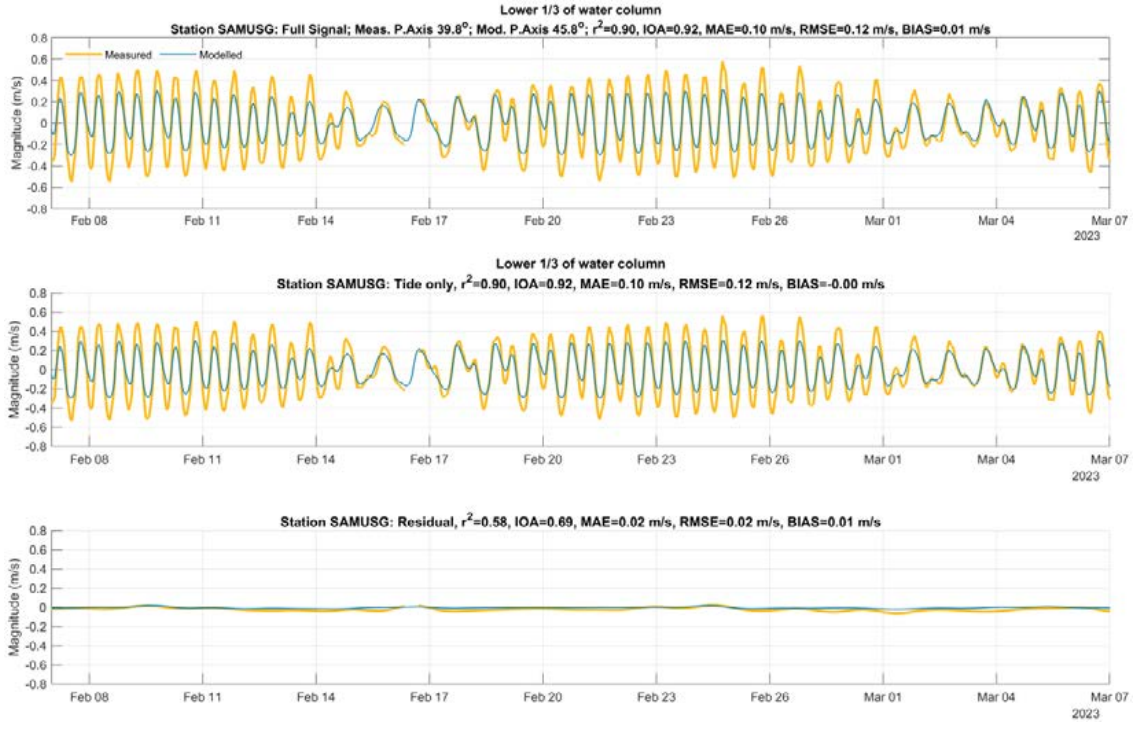


Figure D-64 Lower 1/3 of the Water Column Currents at SAMUSG – 07/02 to 07/03/2023

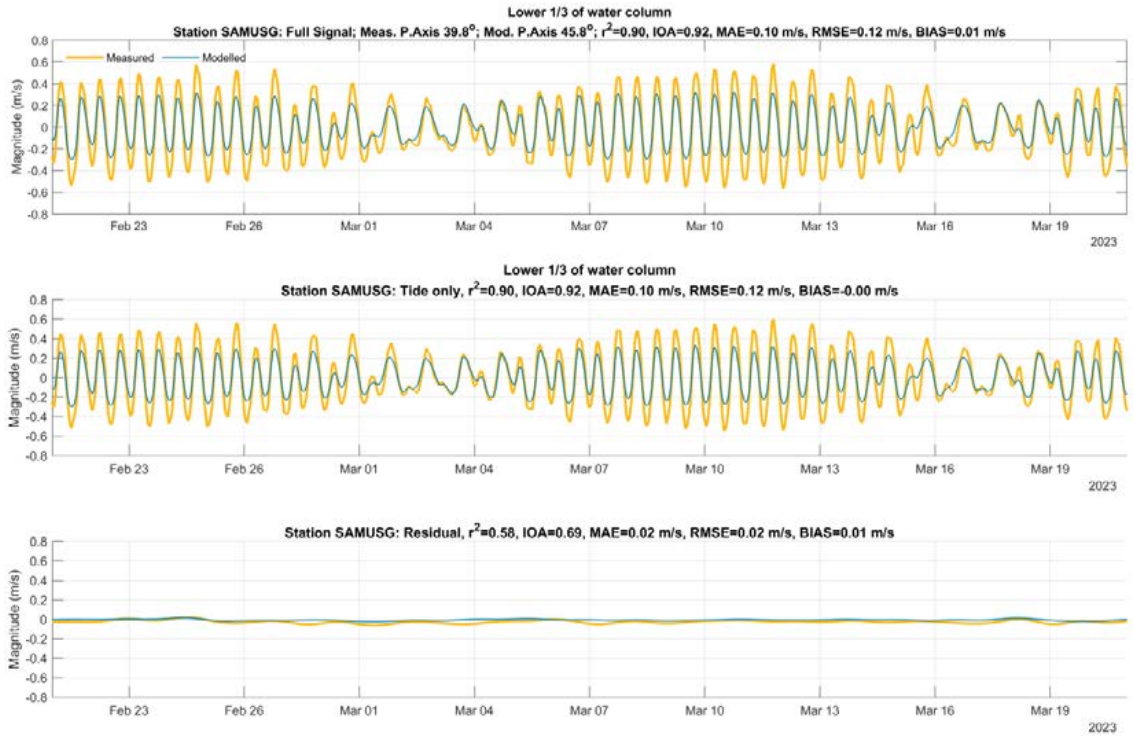


Figure D-65 Lower 1/3 of the Water Column Currents at SAMUSG – 21/02 to 21/03/2023

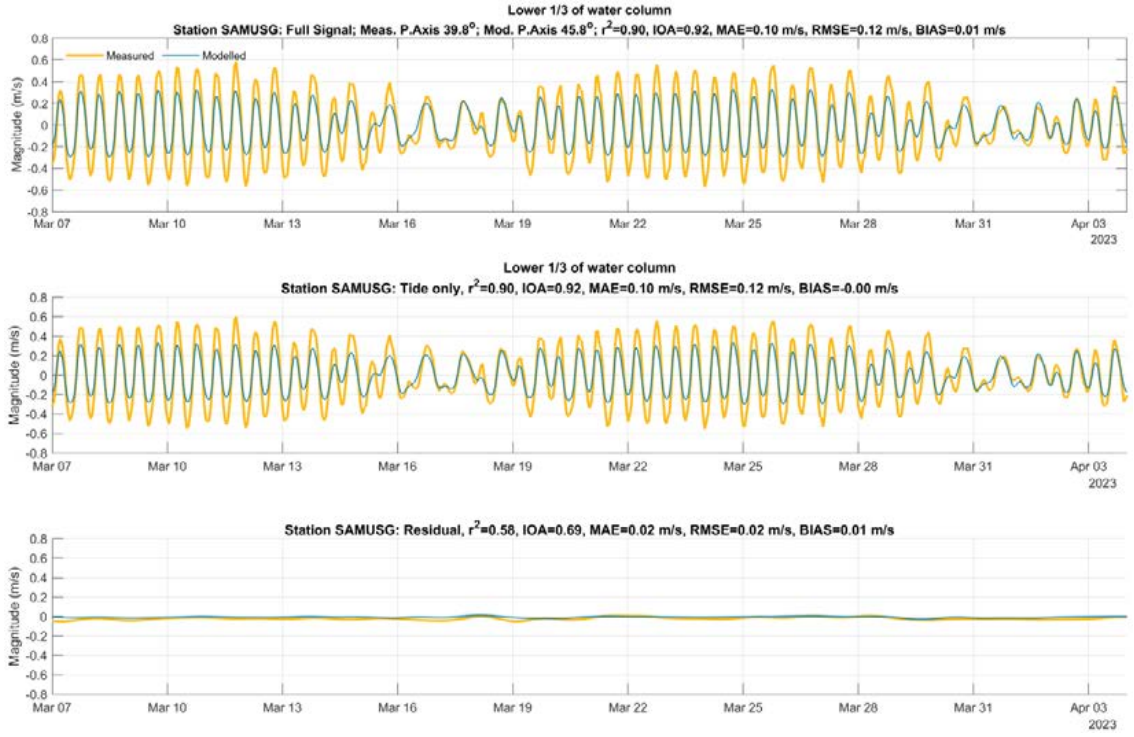


Figure D-66 Lower 1/3 of the Water Column Currents at SAMUSG – 07/03 to 04/04/2023

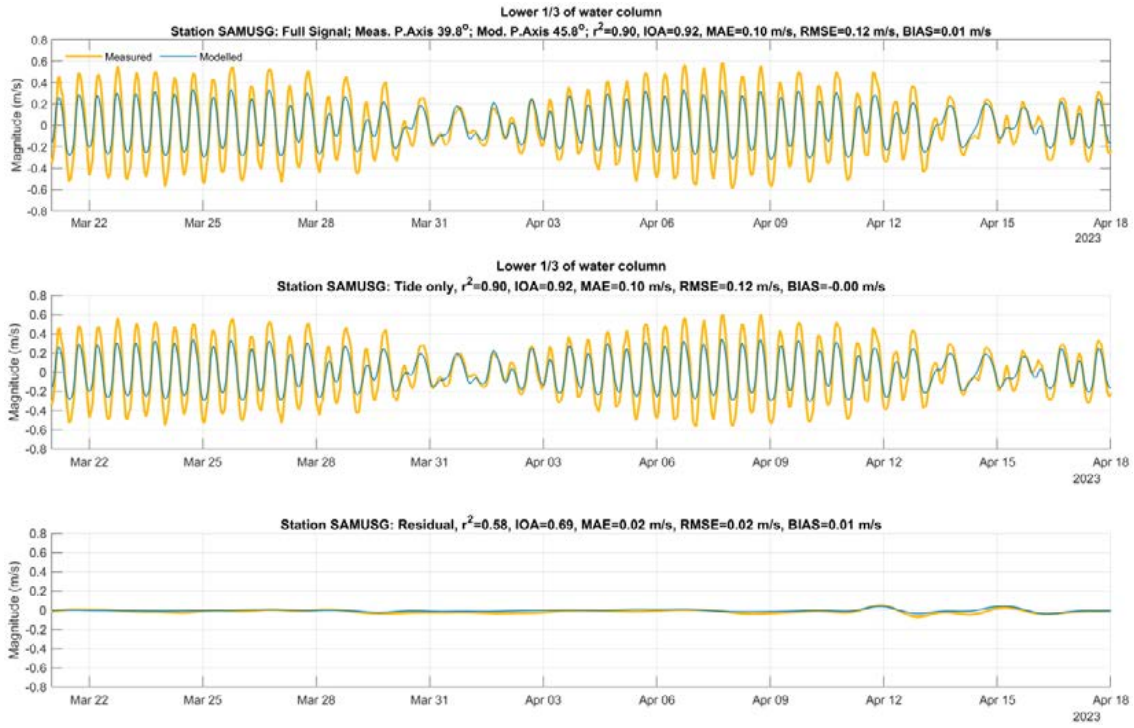


Figure D-67 Lower 1/3 of the Water Column Currents at SAMUSG – 21/03 to 18/04/2023

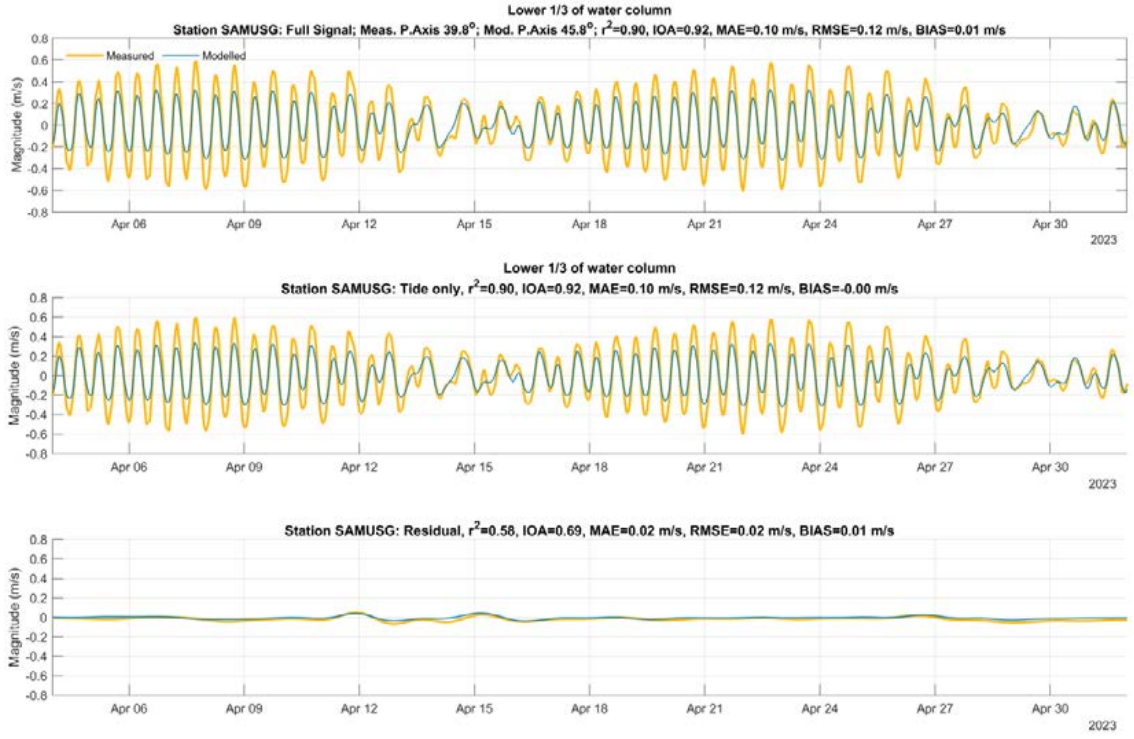


Figure D-68 Lower 1/3 of the Water Column Currents at SAMUSG – 04/04 to 1/5/2023

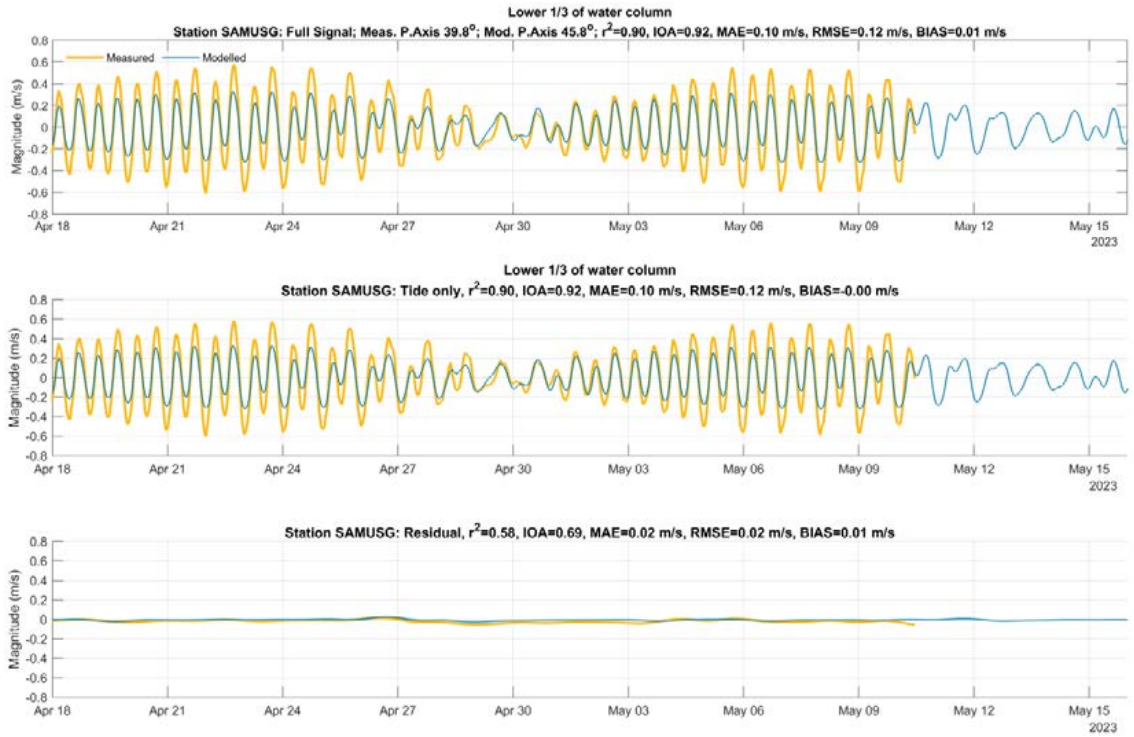


Figure D-69 Lower 1/3 of the Water Column Currents at SAMUSG – 18/04 to 16/05/2023

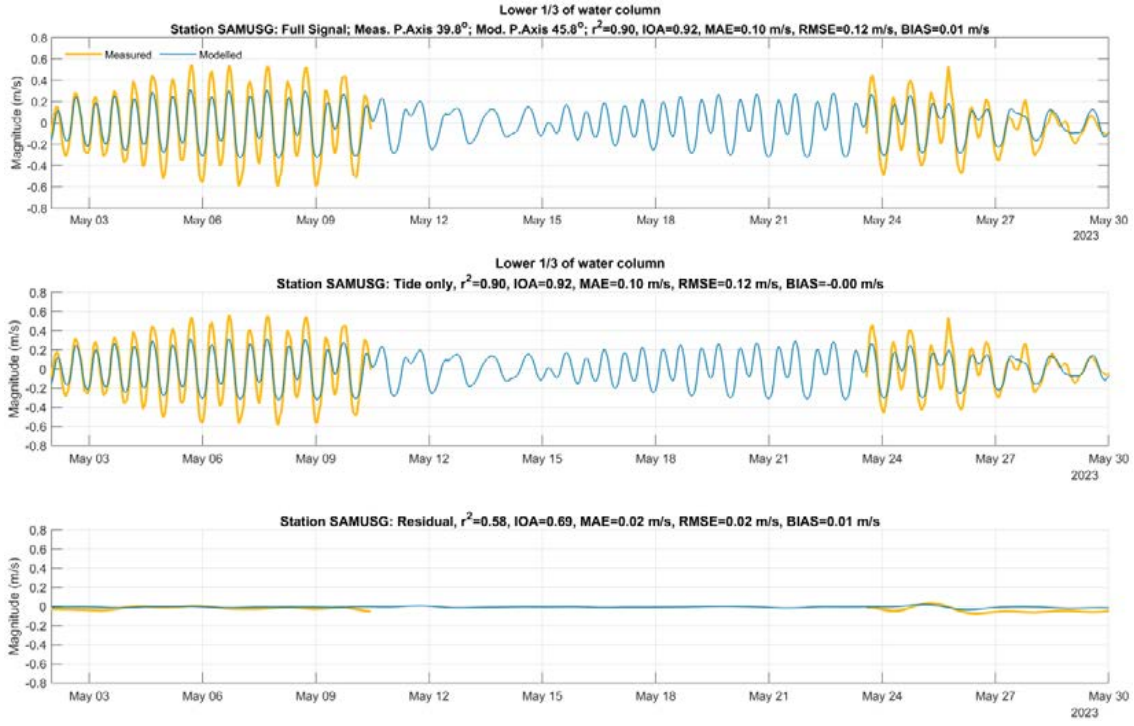


Figure D-70 Lower 1/3 of the Water Column Currents at SAMUSG – 02/05 to 30/05/2023

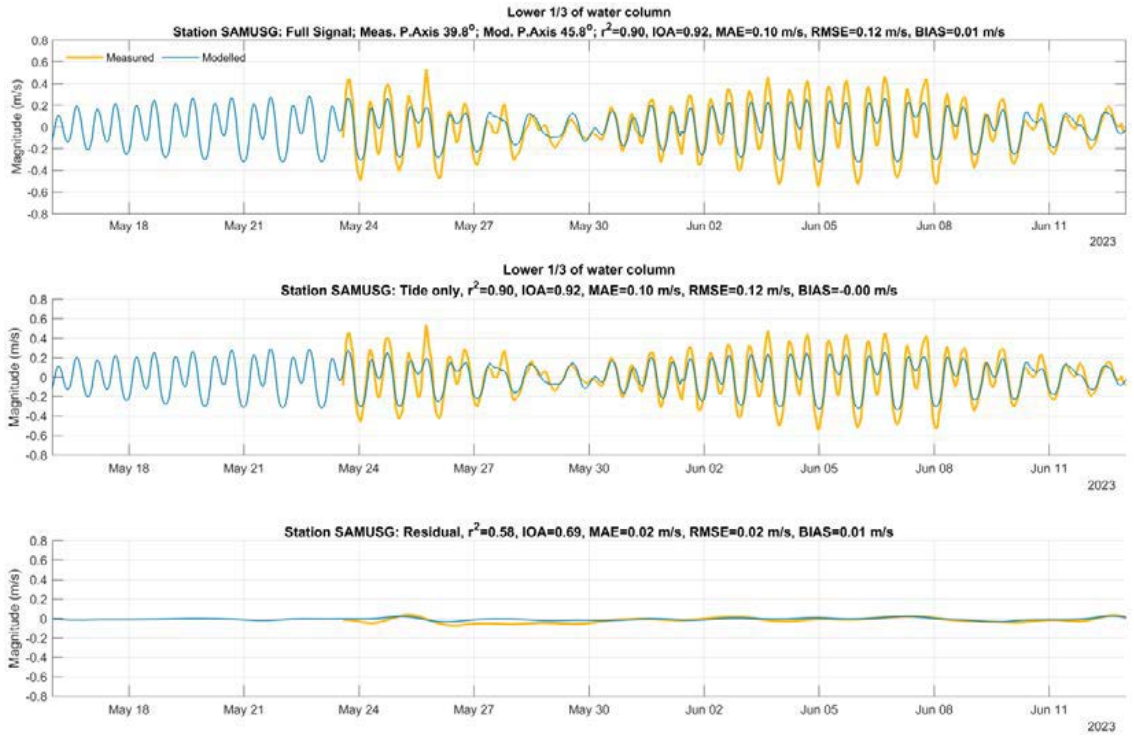


Figure D-71 Lower 1/3 of the Water Column Currents at SAMUSG – 17/05 to 12/06/2023

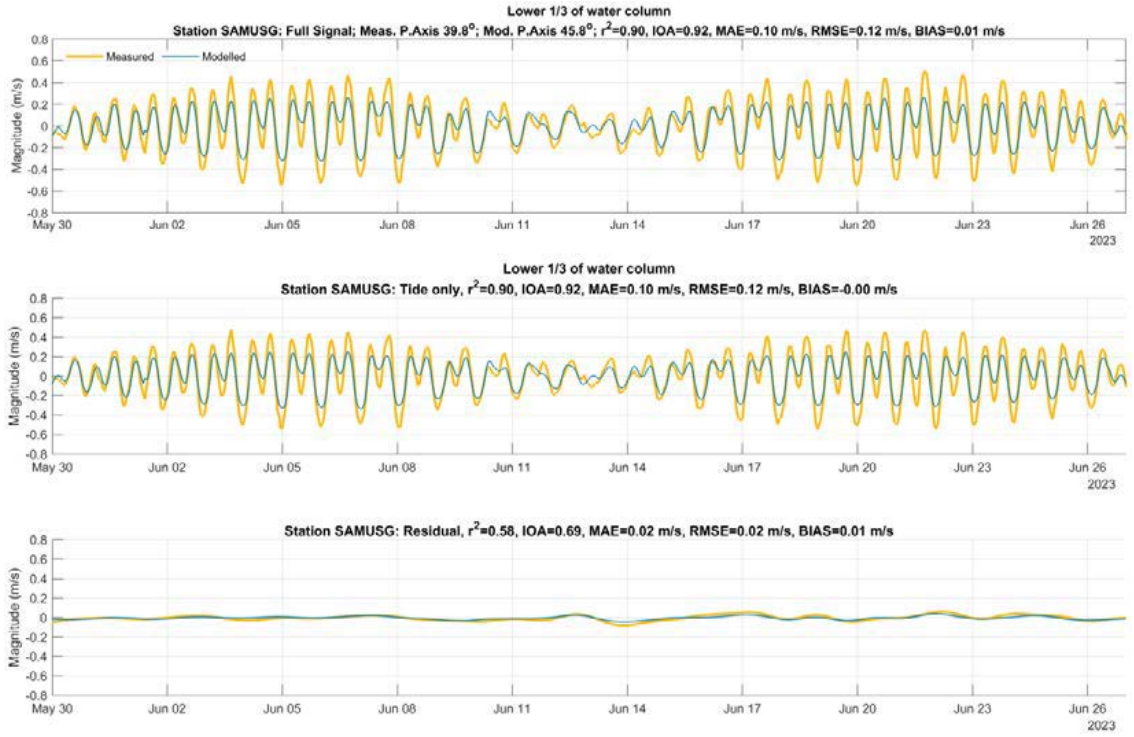


Figure D-72 Lower 1/3 of the Water Column Currents at SAMUSG – 30/05 to 27/06/2023

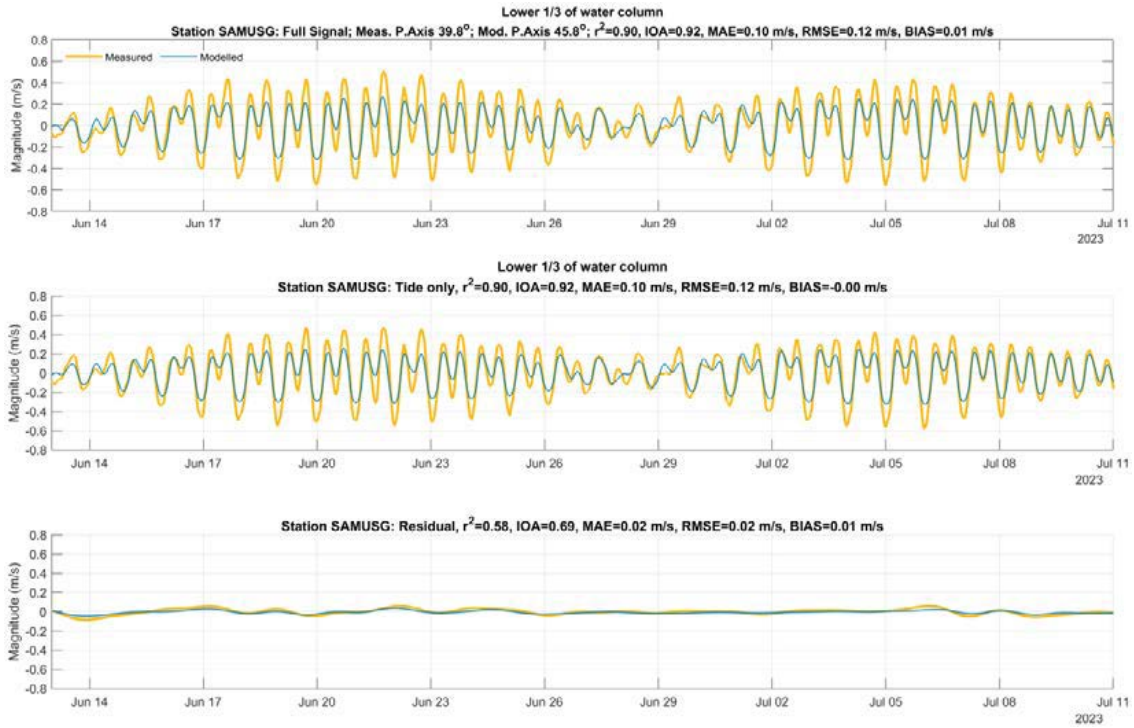


Figure D-73 Lower 1/3 of the Water Column Currents at SAMUSG – 13/06 to 11/07/2023

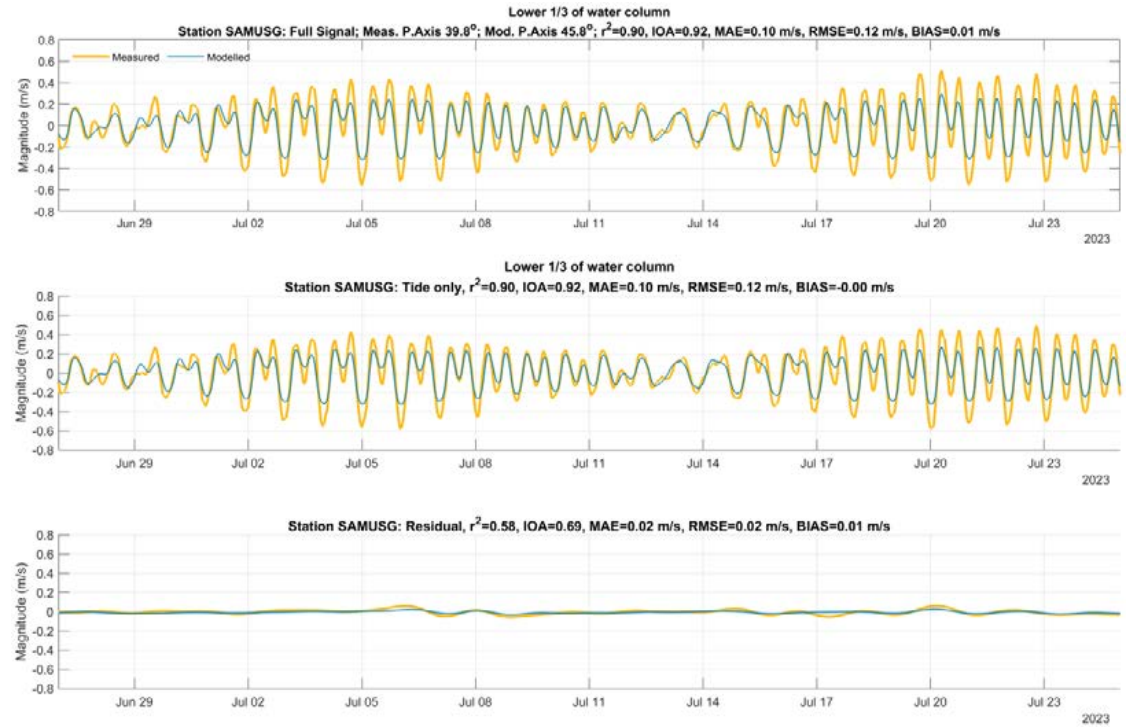


Figure D-74 Lower 1/3 of the Water Column Currents at SAMUSG – 08/06 to 25/07/2023

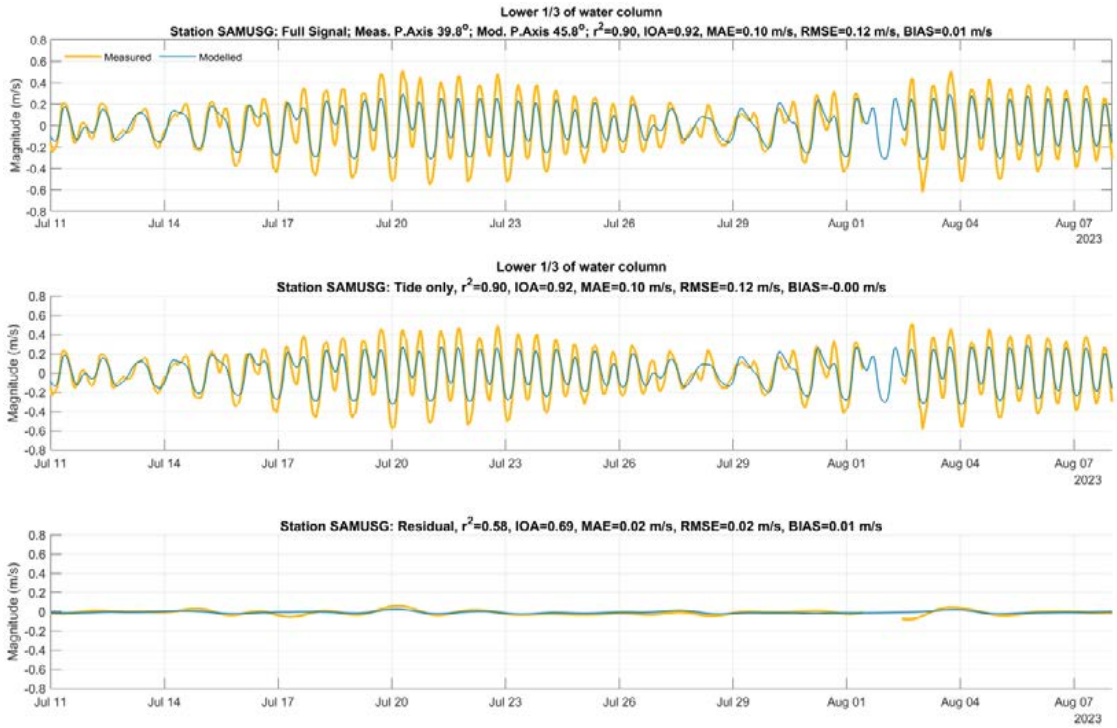


Figure D-75 Lower 1/3 of the Water Column Currents at SAMUSG – 11/07 to 08/08/2023

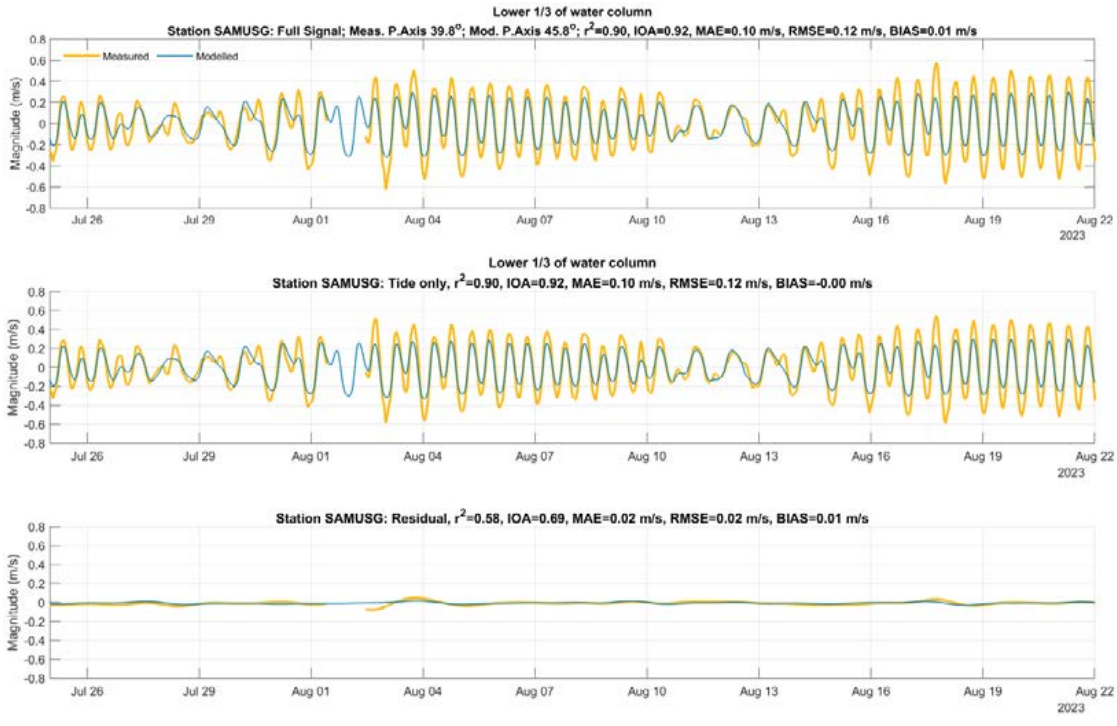


Figure D-76 Lower 1/3 of the Water Column Currents at SAMUSG – 25/07 to 22/08/2023

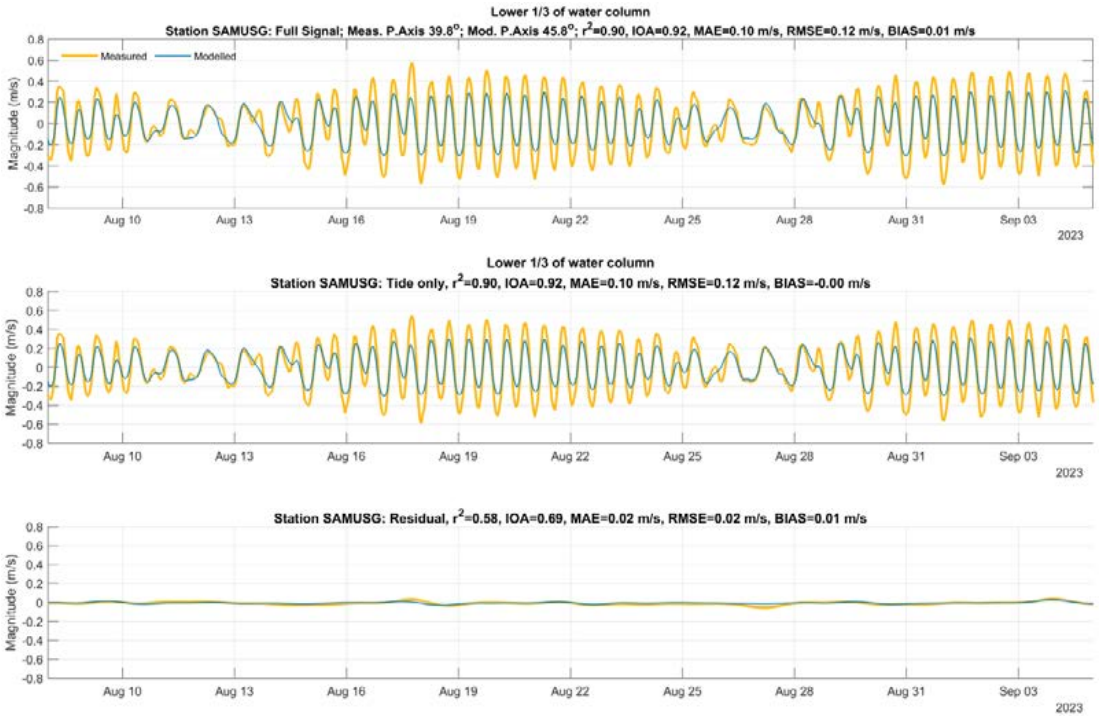


Figure D-77 Lower 1/3 of the Water Column Currents at SAMUSG – 08/08 to 04/09/2023

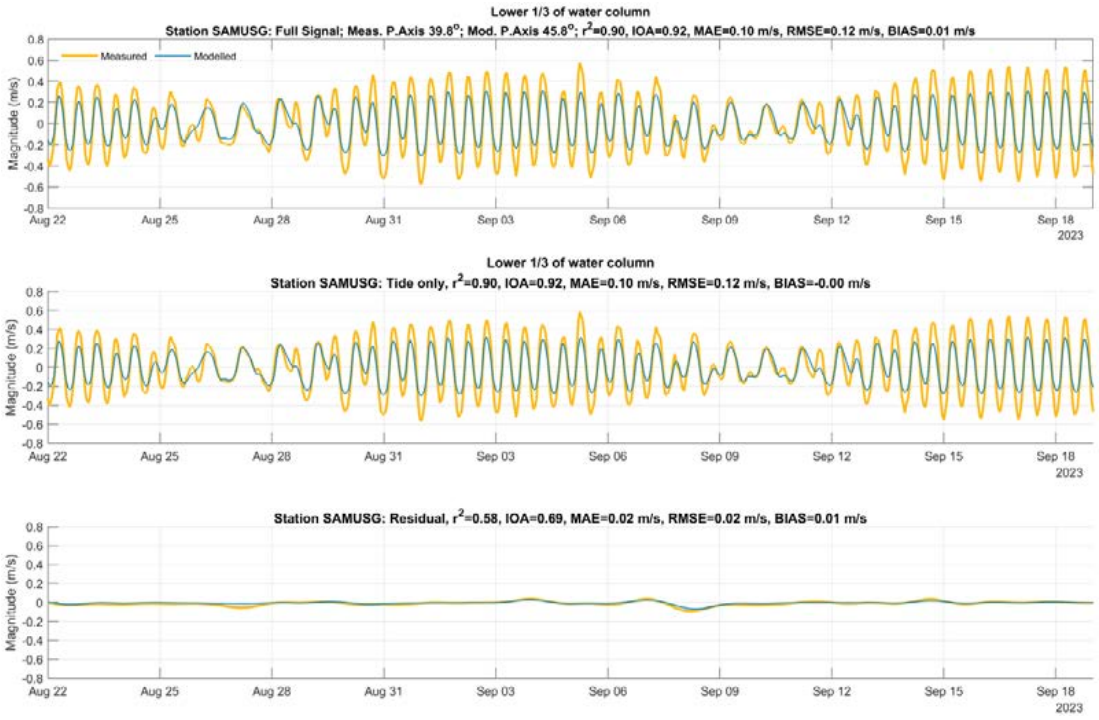


Figure D-78 Lower 1/3 of the Water Column Currents at SAMUSG – 22/08 to 19/09/2023

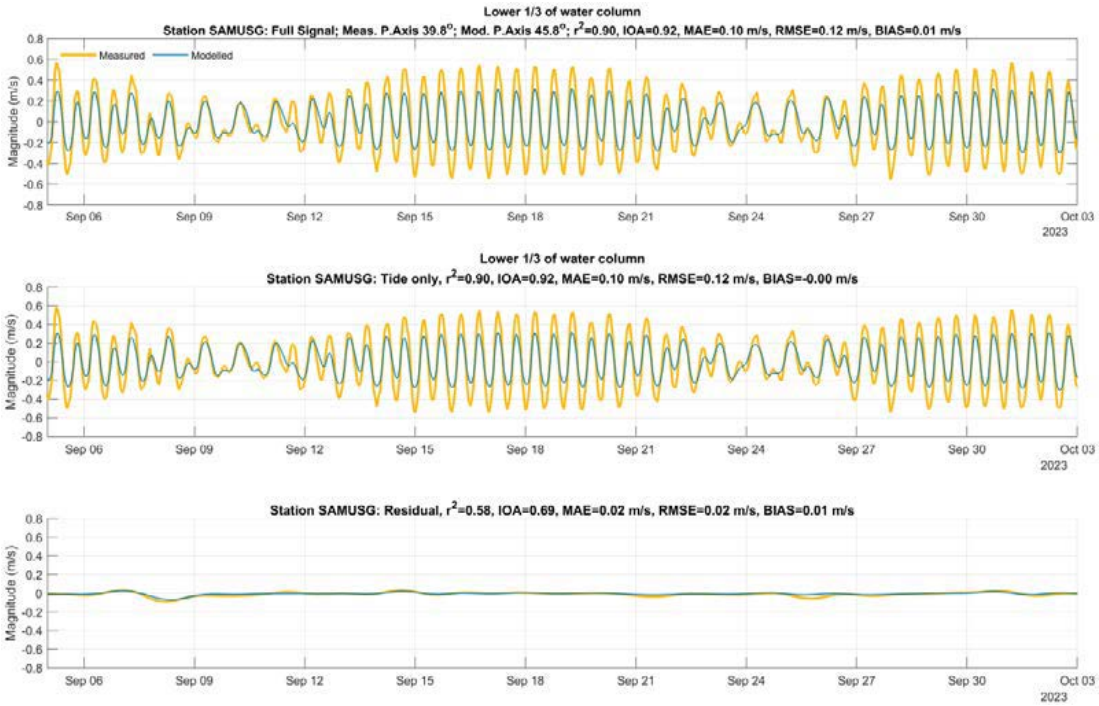


Figure D-79 Lower 1/3 of the Water Column Currents at SAMUSG – 05/09 to 03/10/2023

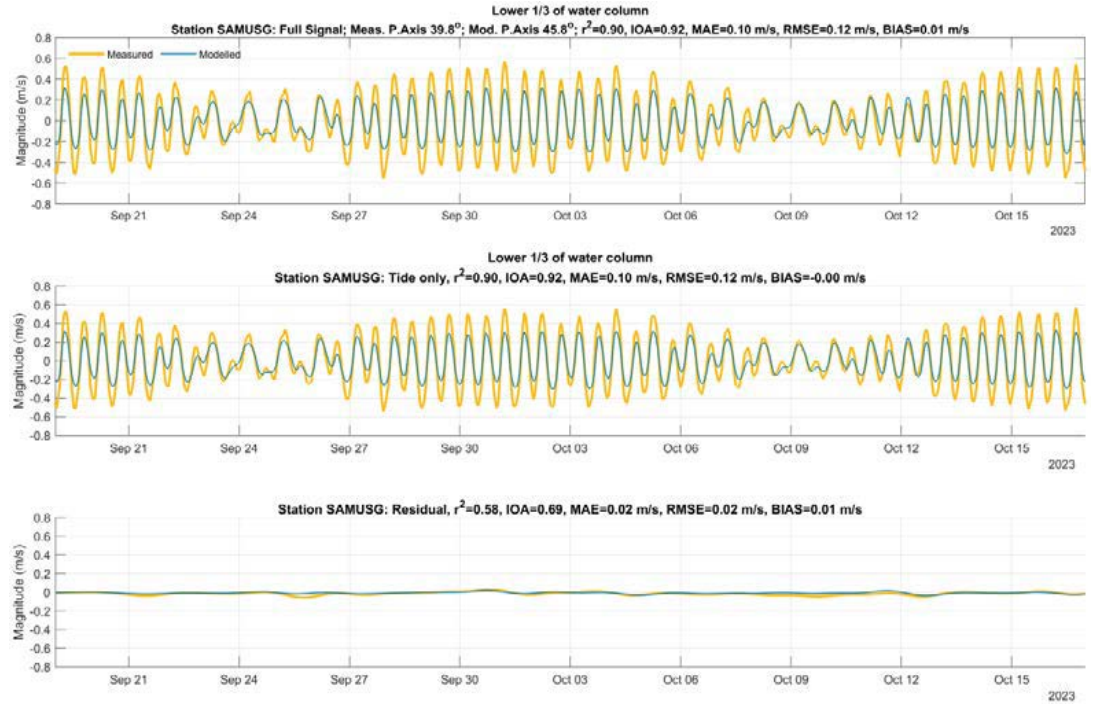


Figure D-80 Lower 1/3 of the Water Column Currents at SAMUSG – 20/09 to 16/10/2023

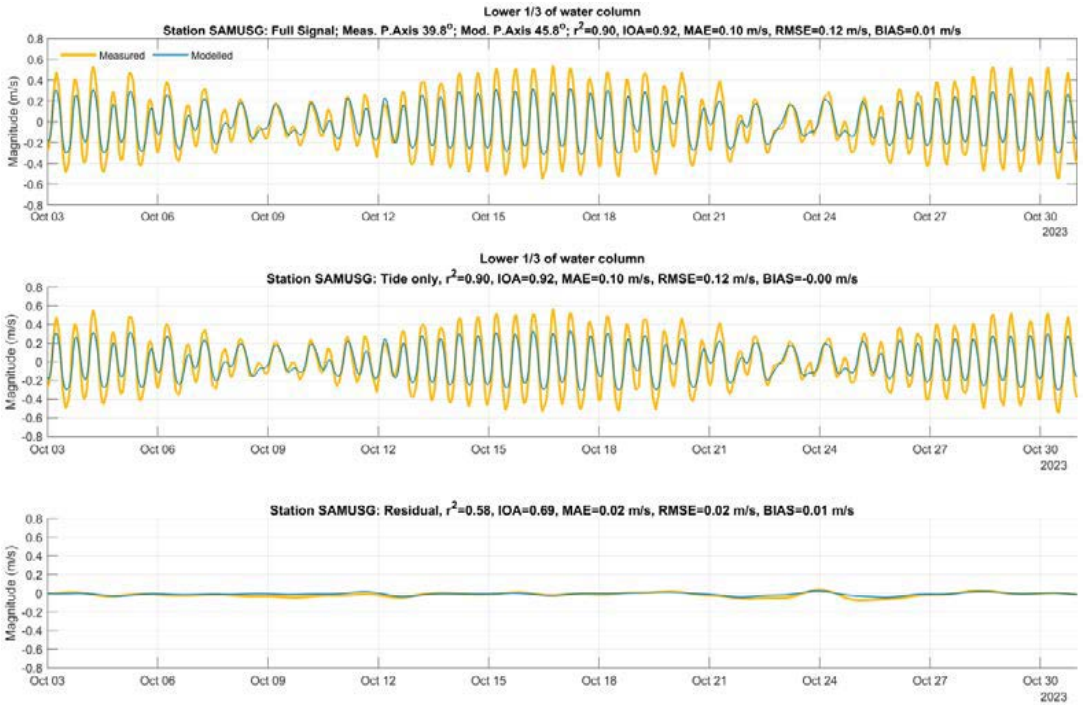


Figure D-81 Lower 1/3 of the Water Column Currents at SAMUSG – 03/10 to 31/10/2023

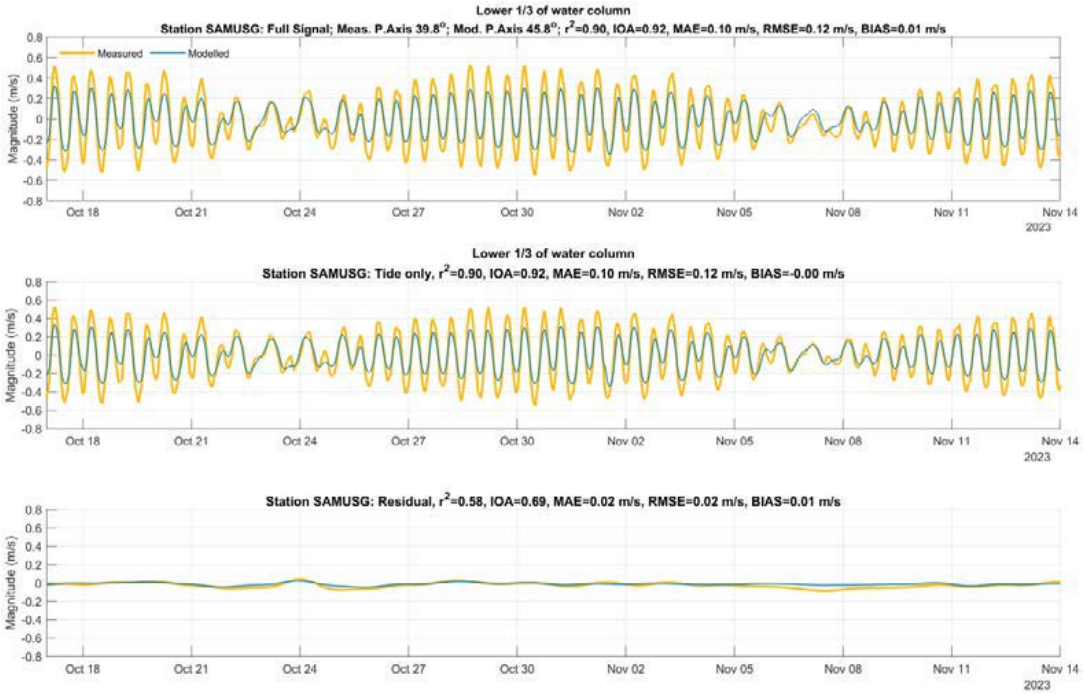


Figure D-82 Lower 1/3 of the Water Column Currents at SAMUSG – 17/10 to 14/11/2023

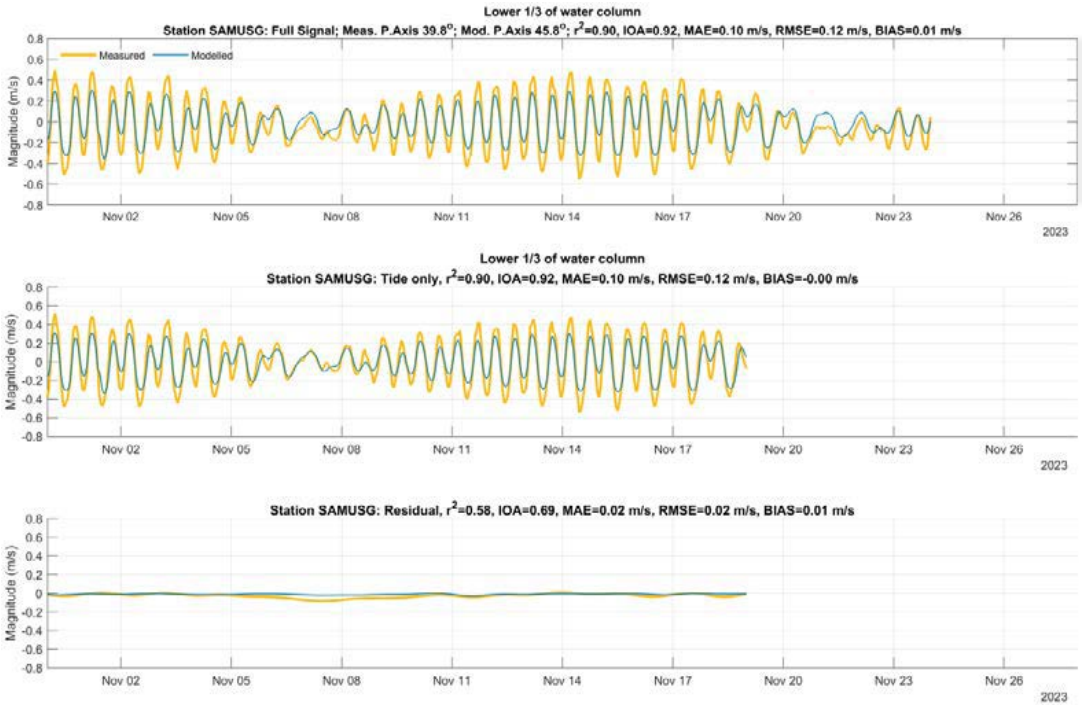


Figure D-83 Lower 1/3 of the Water Column Currents at SAMUSG – 01/11 to 25/11/2023

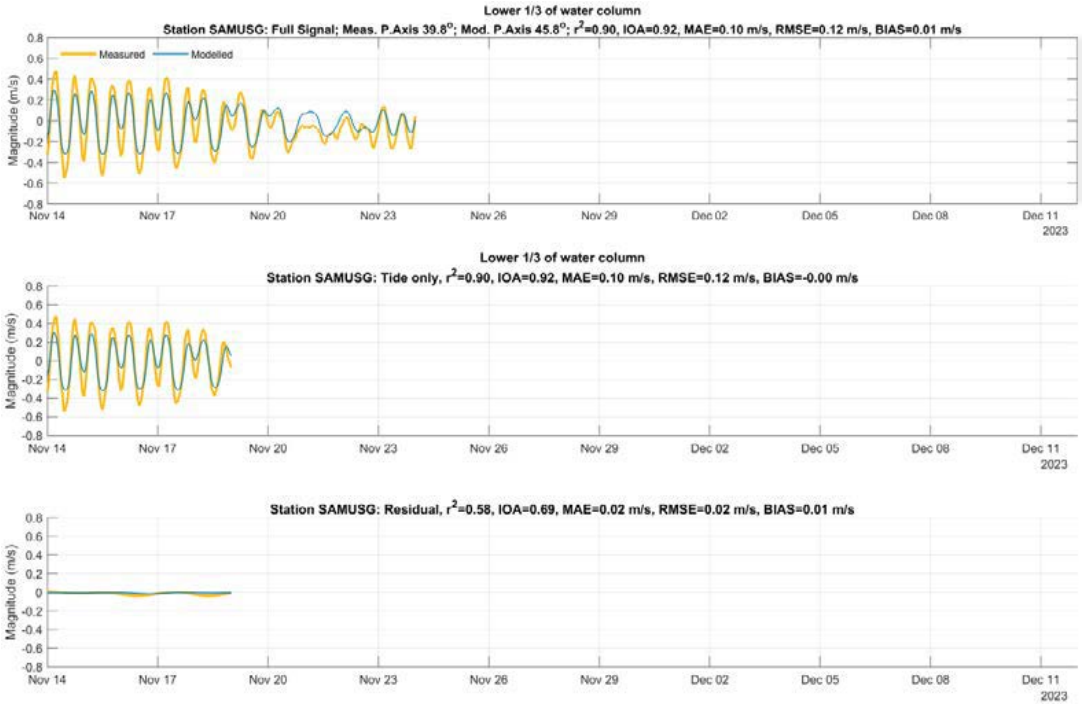


Figure D-84 Lower 1/3 of the Water Column Currents at SAMUSG – 14/11 to 25/11/2023

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APPENDIX E
MEDIAN AND 20TH PERCENTILE DILUTION MAPS



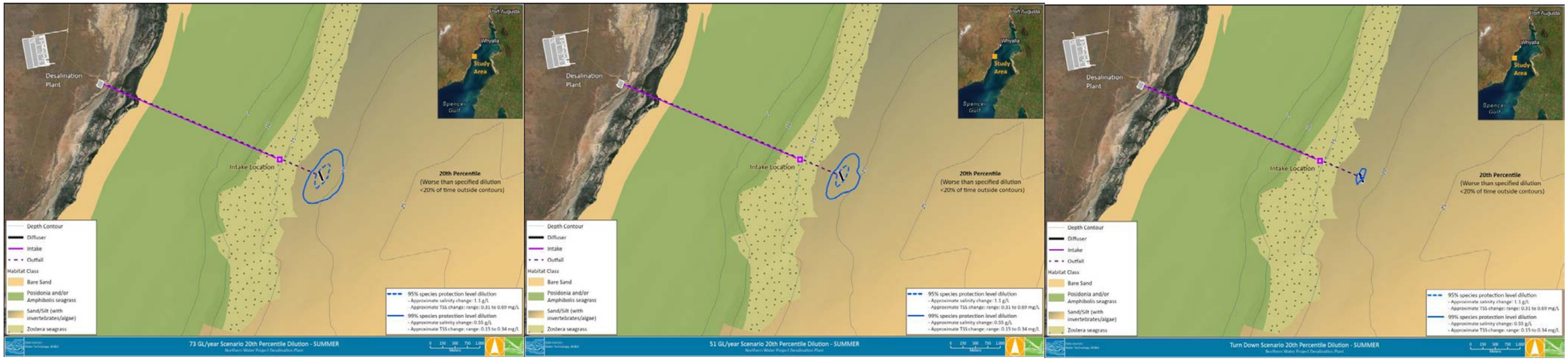


Figure E-1 20th Percentile Dilution – Summer 2022-2023. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

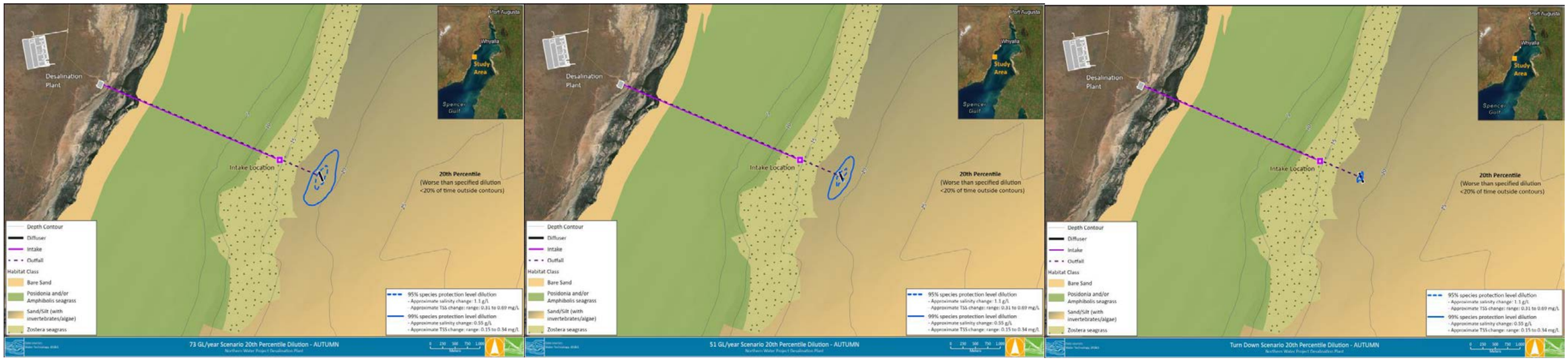


Figure E-2 20th Percentile Dilution – Autumn 2023. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

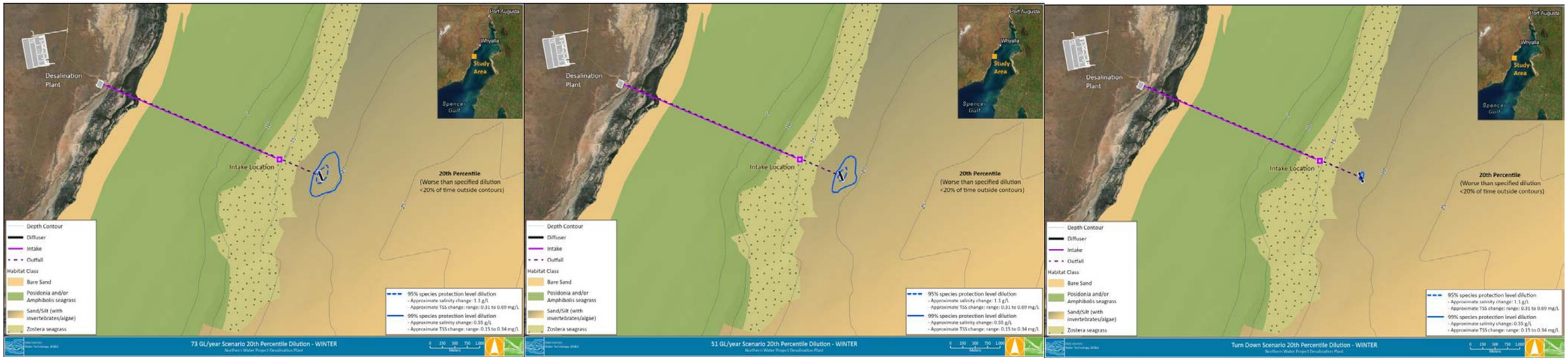


Figure E-3 20th Percentile Dilution – Winter 2023. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

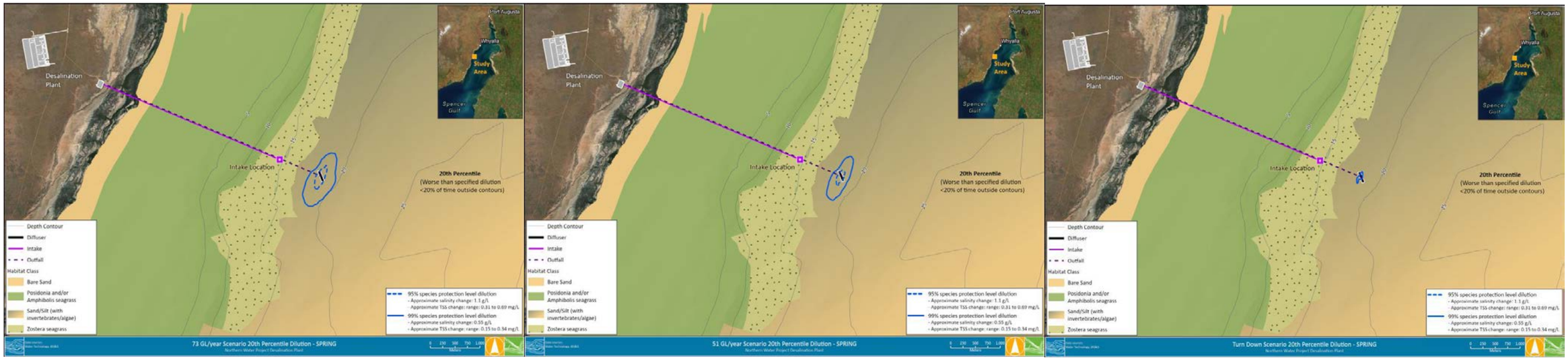


Figure E-4 20th Percentile Dilution – Spring 2023. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

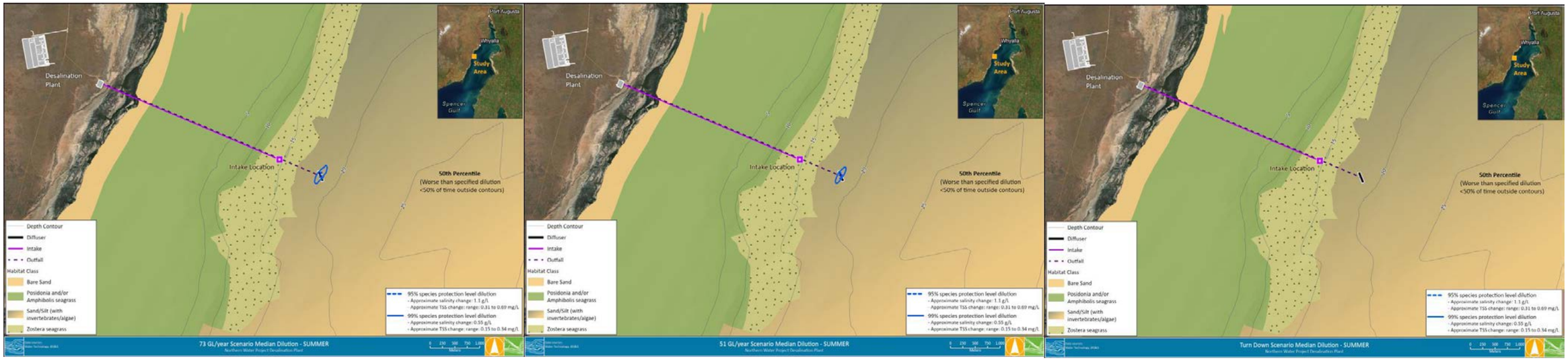


Figure E-5 Median Dilution – Summer 2022-2023. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

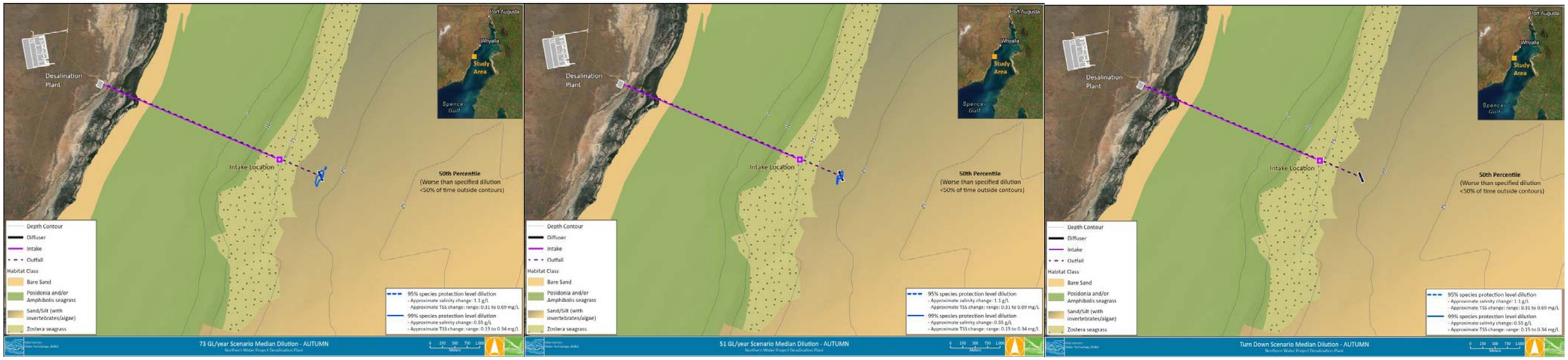


Figure E-6 Median Dilution – Autumn 2023. Left Panels: 73 GL/year Scenario, Middle Panels: 51 GL/year Scenario, Right Panels: Minimum Capacity Scenario

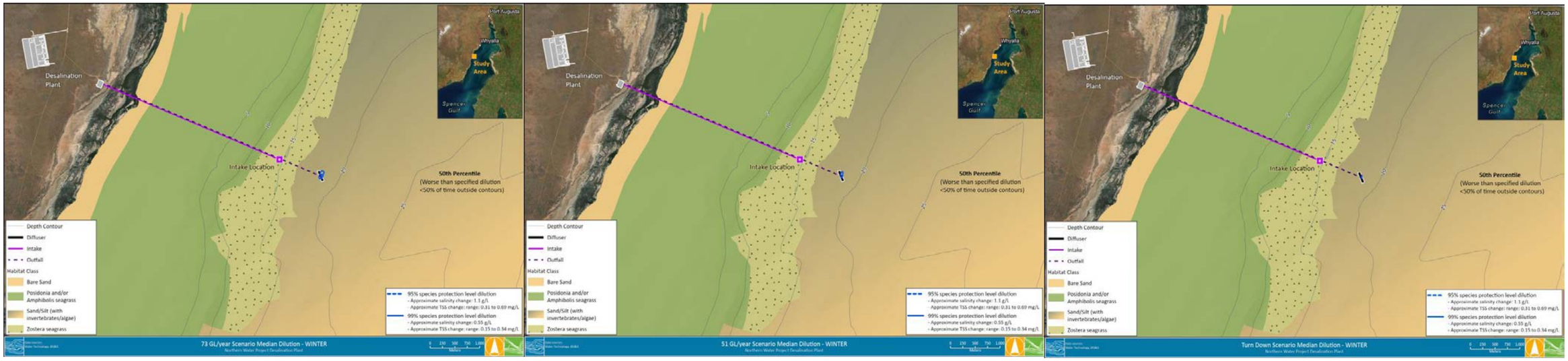


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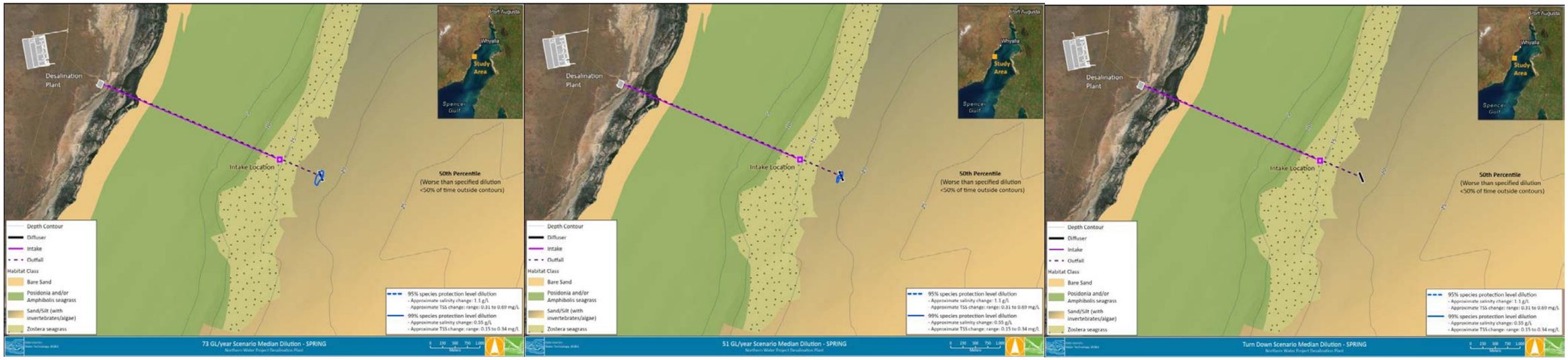


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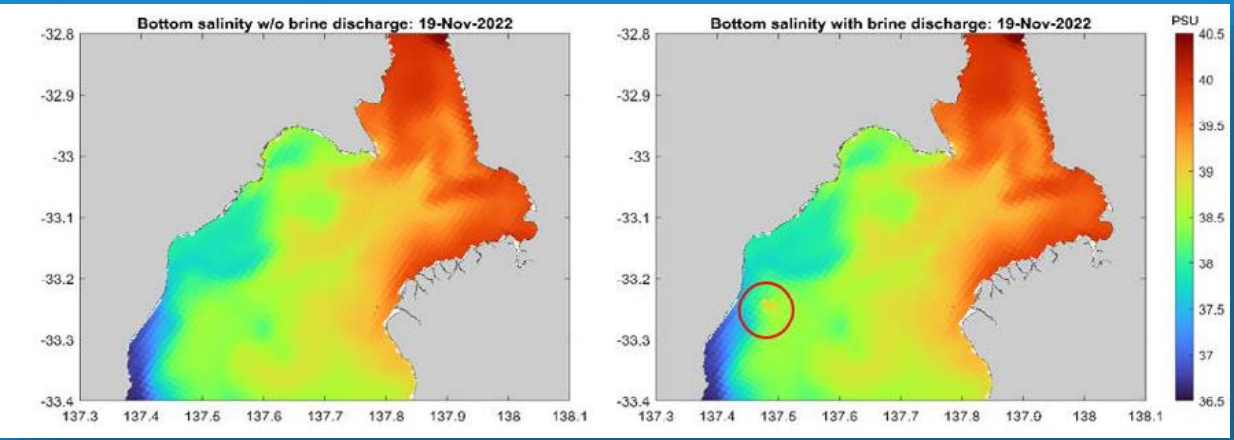


Appendix G-13:

Oceanographic Farfield Modelling of Return Water Discharge and Larval Connectivity

Marine Ecosystems

Oceanographic far-field modelling of return water discharge and larval connectivity to inform Northern Water, Spencer Gulf



Mark Doubell and Charles James

SARDI Publication No. F2026/000323-1
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Government of South Australia
Department of Primary Industries and Regions

SARDI
SOUTH AUSTRALIAN RESEARCH AND DEVELOPMENT INSTITUTE

Oceanographic far-field modelling of return water discharge and larval connectivity to inform Northern Water, Spencer Gulf

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September 2026

The Department of Primary Industries and Regions cares for the Country and communities of South Australia.

We ensure the land, sea, water and sky are healthy for future generations and strive to better understand their spiritual and cultural significance for Aboriginal people across the state.

In the spirit of reconciliation, our commitment is to build progressive and trusting relationships, to share knowledge and learn from each other.

We recognise and own a difficult past, and together we will walk forward.

We acknowledge the many Aboriginal people of this Country as the oldest continuous living culture in the world.

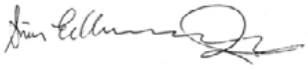
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EXECUTIVE SUMMARY

The South Australian Government's Northern Water project proposes a reverse-osmosis seawater desalination plant at Mullaquana Station in Upper Spencer Gulf. The reference design capacity for the proposed development is 51 GL y^{-1} (140 ML d^{-1}). A hypothetical higher-capacity scenario of 73 GL y^{-1} (200 ML d^{-1}) was modelled as a conservative sensitivity case; it does not represent the reference design capacity for which approval is being sought. Given the ecological, fisheries and socio-economic importance of Spencer Gulf, environmental assessment of the project requires consideration of the far-field fate of hypersaline return water and the potential connectivity of planktonic larvae with proposed desalination infrastructure and sensitive habitats.

This report presents far-field return water-dispersion and larval-connectivity studies undertaken using an extensively validated, three-dimensional hydrodynamic model of Spencer Gulf. Comparison of the model predictions with long-term Integrated Marine Observing System (IMOS) and Northern Water project observations provides confidence that the model reproduces the principal sea-level, current, temperature and salinity dynamics relevant to regional circulation, mixing and transport. These physical processes provide the basis for assessing far-field salinity changes and regional larval transport. Initial diffuser dilution, plume behaviour at sub-kilometre scales, and larval entrainment at the intake are assessed separately through near-field and mid-field studies and are not resolved here.

Across the 12-year hindcast simulations, return-water discharge under the 73 GL y^{-1} higher-capacity scenario produced seasonally averaged salinity anomalies of at least 0.05 PSU within approximately 5 km of the outfall. These anomalies represent <0.15 % of typical ambient salinity. In spring, smaller anomalies of 0.02 PSU or less, equivalent to less than approximately 0.05% of ambient salinity, extended northwards to Point Lowly, approximately 40 km from the outfall. During autumn and winter, transport was directed more strongly southwards. At monitoring locations more than several kilometres from the outfall, modelled return water discharge produced 30-day filtered and long-term mean anomalies of 0.01 PSU, or less than 0.03% of ambient salinity. These predicted anomalies were small relative to observed daily-to-annual salinity variability across Spencer Gulf, which was approximately 1–2 PSU. Under the simulated conditions, the model predicted no progressive salt accumulation over the 12-year hindcast period.

The absence of predicted regional brine accumulation does not imply an absence of potential local ecological impacts. Global evidence from operating desalination facilities indicates that the greatest physical and biological responses associated with desalination discharges are commonly concentrated within tens to hundreds of metres of an outfall. Potential effects

depend on multiple factors including diffuser design, site-specific hydrodynamics and bathymetry discharge chemistry, the magnitude, duration and frequency of exposure, and the sensitivity and ecology of affected species and habitats. At distances of several kilometres or more from the outfall, the far-field salinity increases predicted in this study were below the 1 PSU incremental-salinity benchmark commonly referenced in the desalination literature. This benchmark should not be interpreted as a universal ecological tolerance limit, as biological responses vary among species, life stages, habitats and exposure conditions. Salinity changes at sub-kilometre scales are assessed separately through near- and midfield modelling studies.

Larval-connectivity simulations showed differences in the connectivity of particles representative of different key species with the proposed intake regions over multiple years. While each species was modelled with different spawning times, locations and biological behaviours, connectivity was estimated for all species as the proportion of modelled larvae entering a 1-km radius around the proposed outfall. Estimated connectivity differed among species and across simulated years and was less than 0.003% for King George Whiting, 0.5-2.7% for Western King Prawn and 6.1-7.4% for Snapper. These values describe modelled far-field connectivity only, they are not estimates of intake entrainment, larval mortality, recruitment loss or population-level effect. Although the same broad connectivity metric was applied, the species were modelled using different spawning periods, release locations, larval durations, behaviours and settlement assumptions. The percentages should therefore not be interpreted as directly comparable measures of relative species risk.

Upper Spencer Gulf is a key spawning area and nursery area for several ecologically, commercially and recreationally important species including Snapper, Western King Prawn, King George Whiting and the iconic Giant Australian Cuttlefish. The impact of cumulative, overlapping ecological stresses, including those associated with the 2025/26 *Karenia* species Harmful Algal Bloom event, remain uncertain. Continued monitoring and adaptive modelling would help test predictions and account for changing environmental conditions.

Keywords: Desalination, far-field impacts, return water, salinity, larval connectivity, Spencer Gulf, Mullaquana Station, regional circulation, hydrodynamics.

1. INTRODUCTION

1.1. Background

The South Australian Government through the Northern Water project is investigating construction and operation of a reverse osmosis seawater desalination plant located at Mullaquana Station in the Upper Spencer Gulf, approximately 25 km south of Whyalla (Figure 1). The plant would have a reference design capacity of 51 GL y^{-1} (140 ML d^{-1}). The project aims to provide a secure, climate resilient water source for industry in South Australia's Upper Spencer Gulf and Far North.

Assessment of the proposed development requires consideration of its potential impacts on the surrounding marine ecosystem. Potential impacts associated with seawater desalination include discharge of hypersaline return water and thermal loads, residual treatment chemicals and corrosion products, and entrainment of planktonic organisms at seawater intakes (Lattemann and Höpner, 2008; Panagopoulos and Haralambous, 2020; Tanner and Drabsch, 2021; Omerspahic et al., 2022). The potential ecological effects of return water discharge are strongly influenced by the hydrodynamic characteristics of the receiving environment. Rapid dilution and dispersion can reduce exposure to elevated salinity and other constituents, while poorly flushed environments may be more susceptible to cumulative effects (Roberts et al., 2010). Entrainment of fish eggs, larvae and plankton remains among the least understood environmental risks associated with seawater desalination. Accordingly, environmental assessment should therefore combine hydrodynamic and biophysical modelling with field observations to characterise risk and inform intake design and operation (Nielsen et al., 2024).

These considerations are particularly relevant to Spencer Gulf, South Australia. In Spencer Gulf the long-term residual circulation and short-term tidal mixing operate over markedly different spatial and temporal scales. Spencer Gulf is a shallow, semi-enclosed inverse estuary extending more than 300 km northward from an entrance approximately 80 km wide. It covers approximately 21,700 km² and has a mean depth of 24 m. The Gulf's semi-arid climate results in evaporation substantially exceeding freshwater inflows, producing a persistent northward salinity gradient from approximately 36 PSU (practical salinity unit) at the entrance to more than 45 PSU near Port Augusta (Nunes and Lennon, 1986; Nunes Vaz et al., 1990). During winter, cooling increases the density of hypersaline water in the upper Gulf. This dense water flows southward along the eastern side of the Gulf, while lower-density shelf water enters the Gulf along the western side, producing a density-driven inverse estuarine circulation (Lennon et al., 1987; Kämpf et al., 2009; Middleton et al., 2013; Kämpf, 2026).

During summer, surface warming weakens the density gradient and reduces exchange with adjacent shelf waters (Petrusevics et al., 2011; Kämpf, 2026).

This inverse-estuarine circulation is characterised by a weak, residual, clockwise flow with mean currents typically less than 0.05 ms^{-1} (Middleton et al., 2014). Consequently, gulf-scale water renewal is relatively slow, with modelling studies estimating residence times in Upper Spencer Gulf of approximately one to one and a half years (Kämpf et al., 2009; Kämpf, 2026). These long residence times have raised concerns that dissolved salts from continuous desalination discharge could accumulate over time. In contrast, local and short-term dispersion in Upper Spencer Gulf is dominated by semi-diurnal tides (Easton, 1978; Middleton et al., 2014). Tidal range increases from approximately 1 m at the Gulf entrance to more than 3 m in Upper Spencer Gulf, where tidal currents reach at least 0.5 m s^{-1} (Middleton et al., 2014). Except during fortnightly neap or 'dodge tide' conditions, tidal mixing promotes rapid dilution and dispersion of point-source discharges (Middleton et al., 2014). Upper Spencer Gulf therefore has the capacity for substantial local dilution while also exhibiting comparatively slow whole-of-gulf renewal (Nunes Vaz, 2012; Kämpf et al., 2009; Kämpf, 2026). Far-field modelling is consequently important for assessing potentially longer-term, gulf-scale effects of desalination return water discharge.

Given the ecological, commercial, recreational and socio-economic importance of Spencer Gulf (Gillanders et al., 2015; Tanner et al., 2019), an integrated three-tier modelling program has been undertaken to support the Northern Water environmental assessment. The program comprises: (i) far-field modelling, at approximately 1–300 km scales, to assess potential salt accumulation from return-water discharge in Upper Spencer Gulf and larval connectivity between source areas across the Gulf and the proposed intake area; (ii) mid-field hydrodynamic modelling, at approximately 50 m–10 km scales, to assess local dispersion and dilution of hypersaline return water and quantify larval entrainment at seawater intakes (Water Technology, 2026b); and (iii) near-field hydrodynamic modelling, at approximately 1–100 m scales, to assess the effects of diffuser arrays in the immediate vicinity of the outfall (Water Technology, 2026a).

A suite of hydrodynamic models has previously been used to investigate regional circulation, flushing, larval and microplastic connectivity, and environmental processes in South Australia's shelf, gulf and coastal waters (Kämpf et al., 2009; Middleton et al., 2013, 2014; Doubell et al., 2015; McLeay et al., 2016; Tanner et al., 2020; Rogers et al., 2021; Doubell and James, 2022; Doubell and James, 2023; Kämpf, 2026; Rogers and Fowler, 2026). Building on this capability, a new high-resolution Spencer Gulf hydrodynamic model, with a horizontal resolution of 1 km, was developed to inform the Northern Water environmental

assessment. Model performance was evaluated against an extensive set of oceanographic observations, including long-term measurements from the Integrated Marine Observing System (IMOS) National Mooring Network and Northern Water project-specific moorings deployed in Spencer Gulf.

The environmental impact assessment considers a reference design capacity of 51 GLy^{-1}. However, project infrastructure may accommodate a larger production capacity. On the basis of the current reference design, this upper capacity has been estimated at approximately 73 GL y^{-1}. The far-field modelling therefore adopted 73 GL y^{-1} as a conservative higher capacity scenario; it does not represent the reference design capacity being assessed.

This report presents the far-field component of the modelling program. It integrates three-dimensional hydrodynamic with biophysical particle-tracking simulations to assess gulf-scale dispersion of return water and connectivity of planktonic early life stages of three commercially and recreationally important species: Western King Prawn (*Penaeus (Melicertus) latisulcatus*), King George Whiting (*Sillaginodes punctatus*) and Snapper (*Chrysophrys auratus*). The results inform separate near-filed and mid-field assessment of initial dilution of dense return water from diffusers and larval entrainment by seawater intakes. Together, the regional, mid-field and near-field studies provide an integrated assessment of the potential hydrodynamic and biophysical effects of the proposed Northern Water desalination development in Upper Spencer Gulf.

1.2. Objectives

1. Using updated bathymetry and oceanographic observations, develop and validate a 1 km resolution regional ocean model for Spencer Gulf.

2. Use the Spencer Gulf Model to perform long-term hindcasts (>10 years) to understand the far-field (1 - >300 km) transport and fate of return water outfalls associated with a desalination plant proposed for development near Mullaquana Station in Upper Spencer Gulf.

3. Apply the Spencer Gulf Model predictions to previously developed larval transport models to assess the far-field connectivity of three key fishery species to within 1 km of the proposed location.

4. Provide hydrodynamic and larval connectivity modelling results to support separate near- and mid-field modelling assessments of desalination discharges and larval entrainment (Water Technology, 2026a; Water Technology, 2026b).

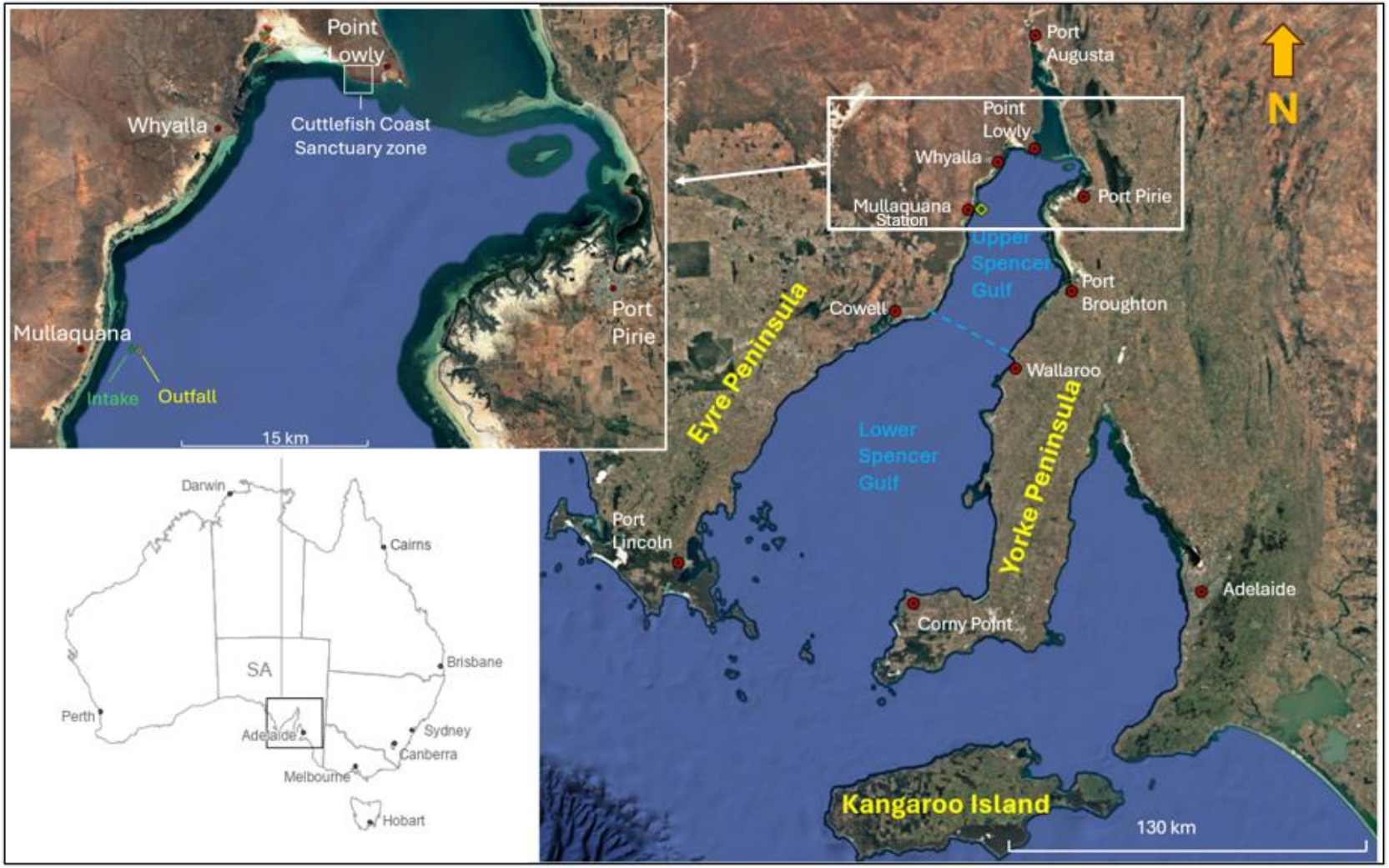


Figure 1. Map showing the location of the proposed desalination plant intake and outfall near Mullaquana Station on the northwestern side of Spencer Gulf, South Australia. The plant is located approximately 25 km south of Whyalla and 35 km south of the heritage listed Giant Australian Cuttlefish (*Sepia apama*) Sanctuary Zone.

2. METHODS

This chapter describes the field observations, hydrodynamic models, return water discharge modelling, larval connectivity simulations and model validation metrics used to assess the potential far-field environmental effects of the proposed desalination plant.

2.1. Oceanographic Monitoring

Hydrodynamic model validation used observations collected from multiple mooring locations and vertical CTD profiles collected throughout Spencer Gulf by the Integrated Marine Observing System (IMOS) and Northern Water projects (Table 1, Figure 2).

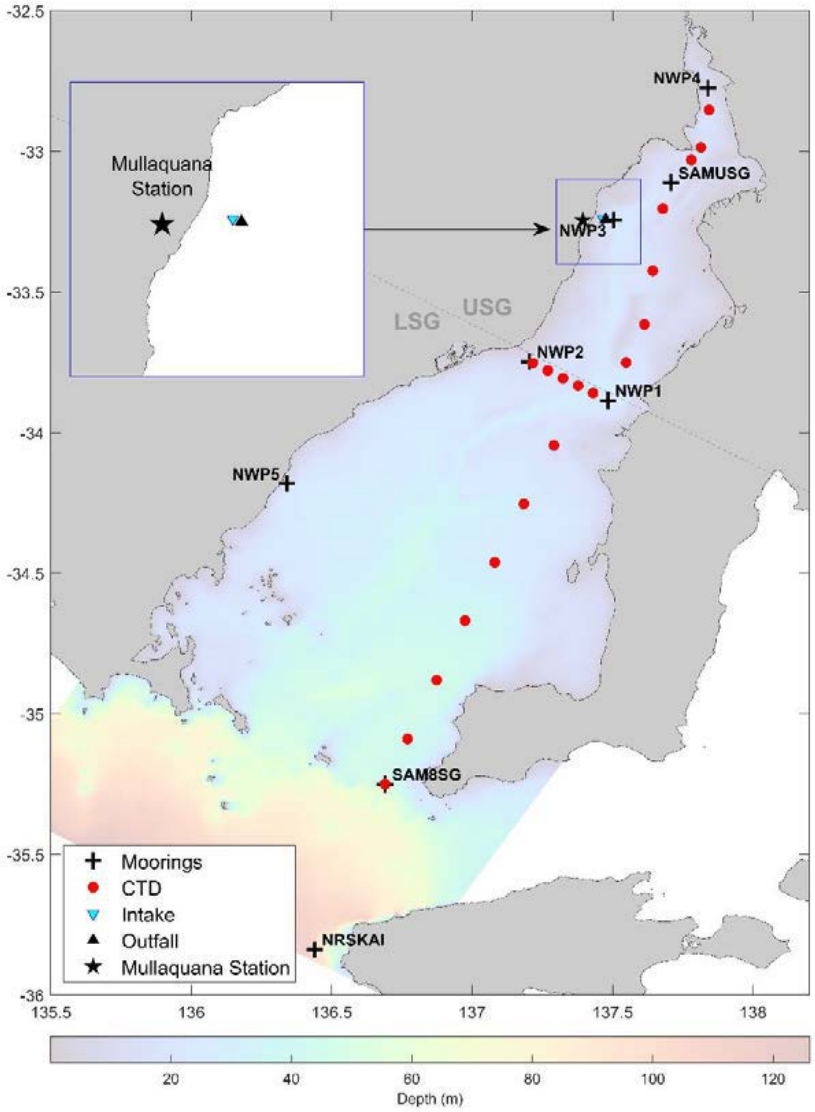


Figure 2. Oceanographic monitoring locations including moorings and CTD profiling sites. (Inset) The location of the proposed desalination intake (blue triangle) and outfall (black triangle) at Mullaquana Station is shown in Upper Spencer Gulf (USG).

Table 1. Name, location and temporal availability of data collected by moorings deployed in Spencer Gulf.

Mooring	Lat (°S)	Long (° E)	Depth (m)	Observation Period
NRSKAI	136.4475	35.8384	100	2012- October 2025
SAM8SG	35.2372	136.6934	50	2012- October 2023
SAMUSG	33.1112	137.7083	18	2019-November 2023
NWP1	33.8861	137.4838	24	October 2022-April 2023
NWP2	33.7470	137.2042	11	October 2022-October 2023
NWP3	33.2432	137.5039	25	October 2022-May 2023
NWP4	32.7738	137.8397	22	October 2022-August 2024
NWP5	34.1824	136.3325	18	June 2023-June 2024

Moorings were equipped with “Seabird 16” or “Seabird 37” CTDs for the measurement of salinity, temperature, and pressure (i.e., sea-level height). Vertical temperature and salinity profiles were measured using a “Seabird 19 plus” CTD. At several locations, moorings included Nortek Signature or RDI Acoustic Doppler Current Profilers (ADCP) to measure current speed and direction throughout the water column (Figure 3). Additional water quality sensors for the measurement of dissolved oxygen, chlorophyll and turbidity concentrations were also included on NWP moorings deployed at Cape Hardy (NWP5) and Mullaquana Station (NWP3) and on the vertical CTD profiler. The NWP3 mooring site was located adjacent to the proposed desalination outfall (Figure 2, Table 2).

Moored sensors sampled every 15–30 minutes and were serviced approximately every three months. All raw observations were converted to quality-controlled data using the IMOS MATLAB toolbox (https://github.com/aodn/imos-toolbox/wiki). These observations provided the long-term baseline required to evaluate seasonal and interannual variability throughout Spencer Gulf. Quality controlled observations from the IMOS sites (NRSKAI, SAM8SG and SAMUSG) are available from the Australian Ocean Data Network (https://portal.aodn.org.au/).

Table 2. Location of proposed desalination plant intake and outfall investigated in this study.

Site Name	Intake		Outfall (mean position of riser array)	
	Longitude (°E)	Latitude (°S)	Mean Longitude (°E)	Mean Latitude (°S)
Mullaquana	137.466	33.240	137.475	34.242

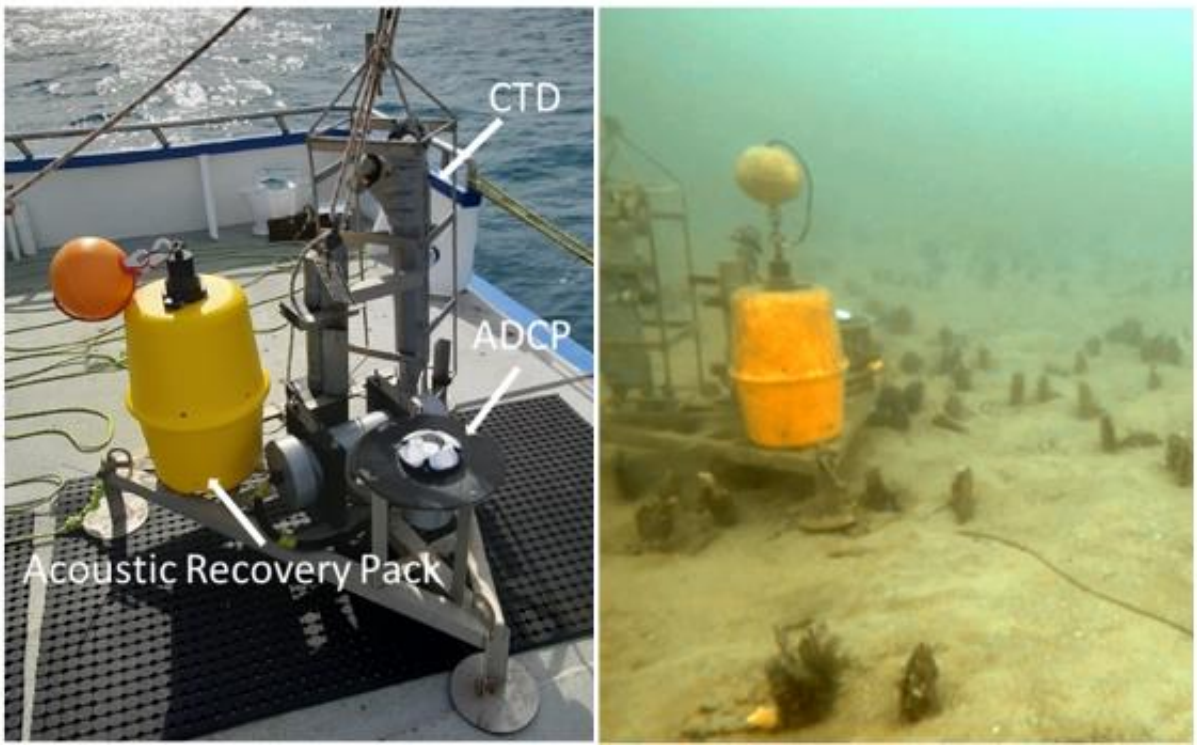


Figure 3. (Left) Picture showing the coastal ocean mooring configuration for water depths less than 30 m. (Right) Image of the coastal mooring deployed at the NWP5 monitoring site.

2.2. Spencer Gulf Hydrodynamic Model (SGM1000)

The open-source Regional Ocean Modelling System (ROMS, https://www.myroms.org/) was used to simulate the circulation and mixing in Spencer Gulf. ROMS is a three-dimensional, free-surface hydrodynamic model that uses topography-following (sigma) coordinates in the vertical direction, and orthogonal curvilinear coordinates in the horizontal direction (Shchepetkin & McWilliams, 2005; Song & Haidvogel, 1994).

The updated Spencer Gulf Model (SGM1000) uses a 1 km horizontal grid with 20 terrain-following sigma layers and allows for wetting and drying along the coastal boundaries (Warner et al., 2013). The model grid was based on the Geoscience Australia 250m resolution Australian Bathymetry and Topography 2023 model (Beaman, 2023).

The model is configured to use the bulk-fluxes formulation to compute atmospheric forcing (Fairall et al., 1996) using air pressure, temperature, wind, humidity, heat-fluxes, and precipitation from the NCEP Climate Forecast System Reanalysis v.2 (CFSv2, Saha et al., 2014). Tidal forcing was provided by the global TPXO8 model (Egbert & Erofeeva, 2002). Lateral oceanic boundary conditions and initial fields for temperature, salinity, currents, and sea level were provided by the latest release (BRAN2023) of the 10 km resolution CSIRO Blue Link Reanalysis (BRAN; Chamberlain et al., 2021). The Smagorinsky scheme (Smagorinsky,

1963) was used to calculate the horizontal eddy viscosity, and a constant horizontal tracer diffusion of 0.5 $m^2 s^{-1}$ was used for temperature and salt. The LMD turbulence closure scheme (Large et al. 1994) was used for vertical mixing. A quadratic bottom stress formulation was assumed with a bottom roughness length of 2 cm.

Improvement to the modelled sea surface temperature (SST) was achieved by adjusting the heat-fluxes using remote sensed SST provided by the Level 4 Multi-scale Ultra-high Resolution (MUR) SST Analyses (Chin et al. 2017). Evaporation rates are often underestimated around the gulfs by the CFSv2 model as it only computes relative humidity to 0.5° resolution. To improve estimates of evaporation and salinities, climatological estimates of relative humidity from Bureau of Meteorology stations at Kadina, Port Augusta, Port Lincoln, Port Pirie, and Whyalla were used. In addition, using the Nunes-Lennon (hereafter N-L) salinity climatology (Nunes and Lennon, 1986; Nunes Vaz et al., 1990), a relaxation to salinities was applied at latitudes above 34°S and longitudes east of 137.5°E. A detailed description of salinity corrections applied to the model is provided in Appendix A.

The hindcast simulation period extended from 1-January 2012 up until 1-January 2024. A twelve-month model spin-up was run from 1-January 2011 to 1-January 2012 and preceded the 2012–2024 hindcast to provide artefact free initial conditions for the hindcast simulations.

2.3. Model Validation

Model validation provides the basis for assessing the capability of the SGM1000 for predicting far-field transport of desalination return water and larval connectivity. The validation compares model predictions with oceanographic observations which capture variations due to tides, weather-band forcing and seasonal to interannual changes.

The principal validation data comprise the long-term observations from the IMOS National Mooring Facility sites including the National Reference Station Kangaroo Island (NRSKAI), the SAM8SG bottom mooring near the mouth of Spencer Gulf and the SAMUSG bottom mooring in Upper Spencer Gulf. Additional Northern Water project mooring observations (NWP1–NWP5) broadened the model validation across the hydrographic regimes of Spencer Gulf, including desalination plant site-specific observations at NWP3 (Figure 2), in the later period of the hindcast. Model assessment included comparisons of sea surface height, depth-averaged tidal and residual currents, temperature and salinity observations made over the 2012–2024 hindcast (Table 1).

Model performance was quantified using several standard metrics including;

(i) Bias $= \frac{1}{n}\sum_{i=1}^{n}(M_i - O_i)$, where M_i is the model prediction, O_i the in-situ observation and n is the total number of data points.

(ii) Root-Mean-Square Error, $RMS = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(O_i - M_i)^2}$

(iii) the coefficient of determination, $R^2 = 1 - \frac{SS_{res}}{SS_{tot}}$ where SS_{res} is the residual sum of squares (the sum of squared differences between actual observations and model predictions) and SS_{tot} is the total sum of squares (the sum of squared differences between observations and their mean).

These metrics evaluate systematic error, overall predictive accuracy and the proportion of observed variance reproduced by the model. Bias is the difference between the mean value predicted by the model and the observed mean and measures systematic over- or underprediction. The RMSE is the sum of the root-mean-square of differences between the model and the measurement and quantifies the overall magnitude of differences between modelled and observed values. Lower bias and RMSE values indicate more accurate predictive performance. R^2 is a measure of the proportion of variability (i.e., variance) explained by the model. Higher R^2 values indicate the model more accurately explains the observed variability. These metrics should be considered together because a model may reproduce mean conditions with low bias and RMSE while explaining relatively little variance when the observed signal has a narrow range.

2.4. Return Water Discharge Modelling

Return water discharge from the proposed Mullaquana Station desalination plant was injected into the model as a point source flux at the seafloor (Kämpf et al., 2009; Doubell and James, 2022) such that:

$$\Delta S = \frac{(Sb - S)q_b}{V}\Delta t$$

where ΔS is the salinity increment for each time step (Δt) within the model, S is the salinity concentration (PSU) in the model grid cell corresponding to the outfall location, Sb is the salinity of the return water discharge, q_b is the return water discharge rate, and V is the volume of the bottom model grid cell.

Return water discharge rates were provided by the Department for Infrastructure and Transport and assumed a hypothetical higher capacity scenario with a production of 200 ML day^{-1} (8,333 m^3 hr^{-1}) of fresh water. The projected outfall flow rate was 286 ML day^{-1} (11,917

m^3 hr^{-1}) which implied an intake rate of 486 ML day^{-1} (20,250 m^3 hr^{-1}) with an overall plant recovery efficiency (E) of approximately 40%. A conservative approach was taken by discharging return water directly into the bottom sigma layer of the model grid cell containing the outfall location, thereby always maximising the bottom anomaly. Due to the use of a terrain following sigma-coordinate model, the thickness of this bottom layer in the model is 1.6 m at the outfall location. Sb is calculated as a function of the salinity at the intake and temperature is set to the background water temperature at the outfall. The resulting salinity of the discharged return water (Sb) is (1-E)$^{-1}$, or approximately 1.7 times the intake value.

Two 12-year model hindcast simulations were undertaken: a baseline simulation without desalination discharge and a simulation including continuous return water discharge. Salinity differences (ΔS) between simulations define the far-field salinity anomalies presented in the Results section.

2.5. Larval Connectivity Modelling

The transport and dispersal of pelagic early life stages (i.e., eggs and larvae) of marine species are heavily influenced by interactions among ocean hydrodynamics (e.g., currents, tides and mixing) and the biological characteristics of the species (e.g., larval development, behaviour and spawning distribution) (Robins et al., 2013). Larval connectivity was quantified using the Lagrangian Larval Transport model (LTRANS), with outputs from regional hydrodynamic models previously applied to understand connectivity and recruitment in South Australia's gulfs for commercially important taxa including Western King Prawn (McLeay et al., 2015, 2016), Blue Swimmer Crab (McLeay et al., 2015), Southern Garfish (Fowler, 2019), Blue Mussel (Doubell and James, 2023), King George Whiting (Drew et al., 2020; Rogers et al., 2021) and Snapper (Rogers and Fowler, 2026).

In this study, the open-source larval transport particle-tracking model LTRANS version v2b (North et al., 2006, 2008; https://northweb.hpl.umces.edu/LTRANS.htm) was used to quantify the potential far-field connectivity between spawning habitats and the proposed intake location for three ecologically and economically important species: Western King Prawns (*Penaeus (Melicertus) latisulcatus*), King George Whiting (*Sillaginodes punctatus*) and Snapper (*Chrysophrys auratus*).

LTRANS was forced by hourly current, temperature and sea level height predictions from the SGM1000 hydrodynamic model. LTRANS interpolated the hydrodynamic model predictions using an internal time-step of 120 seconds. The transport of particles representing eggs and

larvae is simulated using a 4th order Runge-Kutta scheme for advection and a random displacement model to account for sub-grid scale turbulence on particle motion (Laurent et al., 2020). Horizontal diffusivity was set to a constant 1 m^2 s^{-1} and vertical diffusivity equivalent to the shear in the water column (Visser,1997). A logarithmic reduction in current velocities was implemented to simulate the influence of bottom friction on currents. Reflective boundary conditions prevent artificial particle loss at the coastlines, seafloor and sea surface.

To produce realistic connectivity pathways, the biophysical model incorporated species-specific biological sub-models to represent the different behavioural characteristics of the early life history stages (i.e. eggs and larvae) and followed those previously applied to particle tracking studies for Western King Prawns (WKP) (McLeay et al. 2015; 2016), King George Whiting (KGW) (Drew et al. 2020; Rogers et al. 2021) and Snapper (SNP) (Rogers and Fowler, 2026) in South Australia's Gulfs. Parameterisations account for spawning season duration, changes in vertical and horizontal movement and mortality (for KGW and SNP). Particle release locations were based on knowledge of peak spawning periods and spawning distributions and were aligned with the hydrodynamic model grid cells (Figure 4). Simulations were undertaken over multiple years (2021, 2022, and 2023) to capture seasonal and interannual variability in circulation and temperature.

Descriptions of the behavioural models adopted for each species are summarised below.

For KGW, 1,188,000 particles were released at midnight for 15 consecutive days beginning at midnight on 24-April of each year. Particles were seeded across the model domain throughout southern Spencer Gulf and into Investigator Strait (Figure 4a) and were released at the sea surface as KGW eggs are known to be positively buoyant. The pelagic larval duration for KGW ranges from 80 to 130 days, and subsequently particles were tracked for up to 130 days (i.e., until 30-August). Behavioural characteristics changed during larval development. For the first 29 days, larvae were modelled as passive particles that were confined to the surface layer (0-5 m depth). From day 29 to day 35 and day 35 to 82, larvae were modelled as having a diurnal vertical migration (DVM) behaviour with vertical swimming speeds of 1 and 2 cms^{-1}, respectively. From day 82 onwards larvae maintained DVM behaviour with a vertical swimming speed of 2 cms^{-1} in water depths >10 m. In shallower regions <10 m, larvae switched to a reverse DVM behaviour triggered by an estimate of irradiance at the surface at sunrise, whereby they remained near the surface during the day and moved deeper in the water column during the night. Larvae were subjected to a conservative mortality rate of 0.0275 day^{-1} (Rogers et al. 2020) which gradually removed particles throughout the simulation period. Any particles older than 130 days were also considered mortalities and were removed from the simulation.

Tracking of WKP coincided with the October/November spawning season. Each year on 1-November, 100 particles were released 0.3 m above the seafloor at each model grid cell greater than 15 m depth across the entire gulf (Figure 4b), a total of 2,534,400 particles were tracked for 46 days to 17-December. For the first two days, the larvae were passive. Between day two and day 22 larvae were modelled as having a DVM behaviour in which larvae were located near the bottom from dawn to dusk and migrated into the water column between dusk and dawn. Vertical swimming speeds increased from 0.4 cms^{-1} on day two to 2 cms^{-1} on day 22. From day 22, if located in waters <15 m deep, larvae switched to a tidal-stream transport behaviour, whereby they remained on the seabed during the day and at night during ebb tides, and swim vertically into the water column with a speed of 2 cms^{-1} during flood tides between dusk and dawn.

For Snapper (SNP), knowledge of the spatial extent of spawning areas was less clearly defined (Rogers and Fowler, 2026). For this reason, 100 particles were released each day at midnight across the entire model domain (Figure 4c) for 91 days starting on 1-November of each simulation year. Due to the assumed broader spawning range and large number of release dates, only every 4th cell on the model grid was designated as a release site (1,966 sites total). Each particle was deemed settled or dead after 29.5 days. The SNP behaviour model commenced as passive particles that remain in the surface layer (0–3 m) during the first 3.5 days before being uniformly scattered from surface to bottom at night and slowly descending with a mean vertical velocity of 0.0014 mm s^{-1} during the day. The horizontal movement of particles gradually increased with time to a maximum of 50 mm s^{-1}, resulting in a uniform distribution throughout the water column during the night and a gaussian distribution during the day, with distributions becoming progressively deeper in the water column as particles aged. After 18.5 days, a percentage of particles located in settlement polygons (see Figure 4c for locations) could settle, increasing from 0% at 18.5 days to 100% at 23.5 days. For this study 17,890,600 particles were released for each year.

The far-field connectivity modelling was limited to assessing the proportion of all released particles that came within a 1km radius of the proposed intake pipe location while alive. For WKP and KGW, evaluating connectivity with the offshore intake involved excluding larval settlement to keep particles active within the water column. This approach minimized the modelling uncertainties associated with settlement dynamics, such as pelagic larval duration and spatial variations in nursery habitat distribution or suitability (McLeay et al. 2016; Rogers et al. 2021). This provided a conservative assessment by maximizing the number of particles capable of reaching within 1 km of the intake.

Conversely, a more conservative definition of connectivity was used for SNP because hypothetical release sites covered the entire model domain, reflecting uncertainty around exact spawning locations, and the intake sits directly within a larger primary nursery area (Rogers & Fowler 2026). Connectivity estimates excluded particles from release sites where they did not eventually settle. Larval settlement was included to simulate successful recruitment of larvae prior to coming within 1 km of the intake.

For each species, particles entering within 1km of the intake were removed from tracking to prevent repeat counting. Connectivity (C_{ij}) with the intake was estimated as a percentage,

$$C_{ij} = \left(\frac{N_{ij}}{N_i}\right) x\ 100$$

where N_{ij} represents the number of active particles released from grid cell i that entered the 1km radius zone centred on the desalination outfall j. For WKP and KGW, N_i represents the total number of particles released. For SNP, N_i represents only the total number of particles that settled which generally results in much higher estimates of connectivity. Connectivity estimates should not be confused with entrainment. Estimates of entrainment are based on mid-field simulations that quantify how many of the particles within 1 km of the intake would be expected to enter the intake (Water Technology, 2026b). This fraction is then multiplied by the regional connectivity values provided in this report to estimate the overall percentage of particles which may be potentially entrained by the desalination intake.

Particle tracks for all species are accumulated over the duration of the simulation to generate the density heat maps indicating where particles spent most of their time

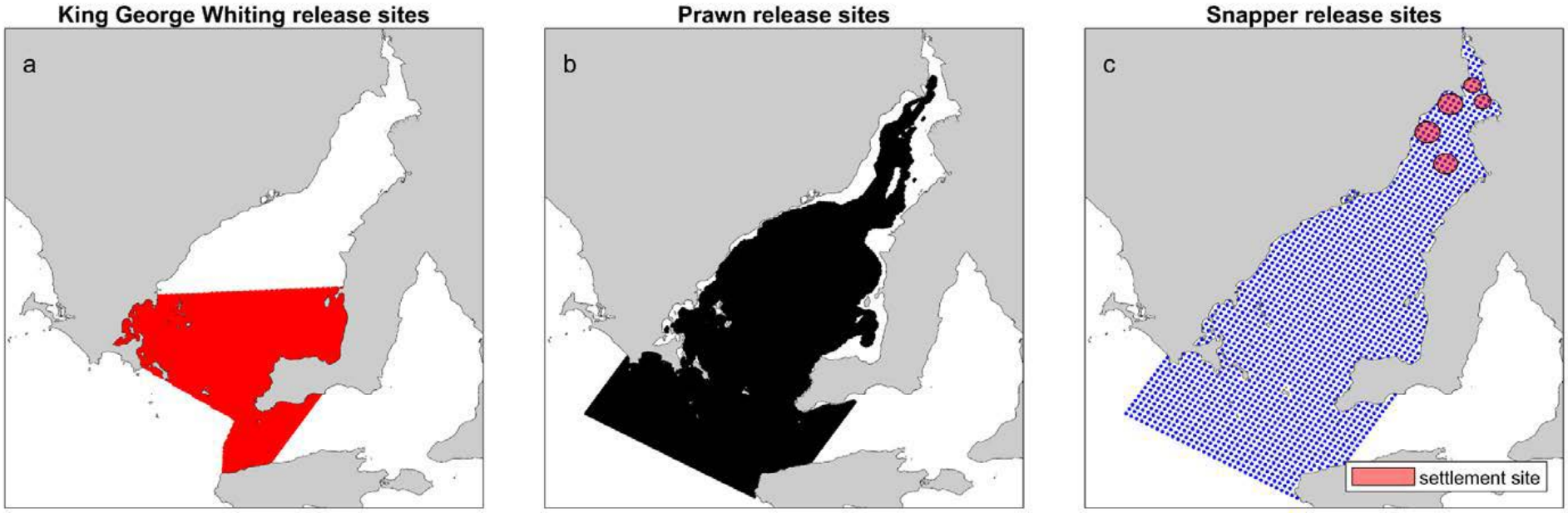


Figure 4. Particle release locations and settlement sites (Snapper only) used for the larval tracking studies: (a) King George Whiting (b) Western King Prawn and (c) Snapper.

3. RESULTS: MODEL VALIDATION

3.1. Mooring observation comparisons

Model validation was undertaken to assess the capability of SGM1000 to reproduce the circulation and hydrographic variability relevant to far-field return water dispersion and larval-connectivity modelling. Model predictions were compared with observations spanning tidal, weather-band, seasonal and interannual timescales.

The principal validation datasets were obtained from the long-term IMOS moorings located at the Kangaroo Island National Reference Station (NRSKAI), the SAM8SG mooring near the mouth of Spencer Gulf and the SAMUSG mooring in Upper Spencer Gulf. Additional observations from the Northern Water Project moorings (NWP1–NWP5) extended the assessment across the major hydrographic regions of Spencer Gulf and included site-specific observations at the proposed desalination-plant location near the NWP3 mooring site (Figure 2). The assessment included sea-surface height, depth-averaged tidal and residual currents, and temperature and salinity observations collected during the 2012–2024 hindcast period (Table 1).

3.2. Comparison with long-term IMOS observations

The NRSKAI is located in approximately 110 m of water. Sea-Bird CTDs were deployed at approximately 40 and 100 m depth, with additional temperature observations obtained from Aqualogger thermistors at approximately 50 and 90 m. Modelled temperature agreed well with observations throughout the water column (Figure 5). Biases were ≤0.68 °C, RMSE values ≤1.20 °C and R^2 values ≥0.63, except at 50 m, where the R^2 was 0.54. The 50 m temperature sensor was located near the seasonally varying base of the surface mixed layer, where relatively small vertical displacements of the thermocline can produce large changes in temperature. Consequently, differences in the timing or vertical position of the modelled thermocline likely contributed to the reduced validation at this depth.

Salinity biases were ≤0.15 PSU in magnitude and RMSE values ≤0.34 PSU, indicated that mean salinity was reproduced with relatively small errors. However, the R^2

decreased from 0.53 at 100 m to 0.12 at 40 m. The model captured the dominant seasonal and interannual signals, including high-salinity outflows from Spencer Gulf and lower-salinity water associated with summertime upwelling. The low R^2 at 40 m should be considered alongside the small bias and RMSE because the limited observed variability at this depth reduces the variance available to be explained. Differences in timing, unresolved vertical structure, measurement uncertainty and possible conductivity-cell biofouling may also have contributed to the reduced variance explained by the model at this depth.

Tidal currents and sea-surface height were reproduced well, with negligible bias, with RMSE values of 0.05 ms^{-1} and 0.06 m and R^2 values of 0.86 and 0.90, respectively. The modelled principal tidal-current axis was 74.9° compared with an observed orientation of 80.2°, a difference of 5.3°. Model performance was reduced for the non-tidal residual signals. Residual-current and residual-SSH biases were both negligible, with RMSE values of 0.15 m s^{-1} and 0.11 m, respectively. The model explained 44% of the observed residual-current variance and 64% of the residual-SSH variance and reproduced the lower-frequency interannual and seasonal variability well but was limited in its ability to reproduce weather-band fluctuations.

SAM8SG is located in southeastern Spencer Gulf in approximately 50 m of water, with temperature and salinity sensors positioned near the seabed (Figure 6). The model reproduced near-bottom temperature with a bias of 0.41 °C, an RMSE of 1.15 °C and an R^2 of 0.72. These results indicate that it captured most of the observed temperature variability, with a modest warm bias. Salinity performance was reduced, with a bias of 0.39 PSU, an RMSE of 0.81 PSU and an R^2 of 0.14. Early observations were affected by conductivity-sensor problems, and these sensors were replaced with more reliable Sea-Bird CTDs in 2019. Consequently, the salinity statistics incorporate some observational uncertainty. Nevertheless, the model reproduced the characteristic sawtooth-shaped seasonal cycle, comprising relatively rapid salinity accumulation during summer followed by a more gradual decline after the onset of autumn–winter outflow.

Tidal currents and sea-surface height were reproduced well, with R^2 values of 0.86 and 0.94, respectively. The tidal-current RMSE was 0.06 ms^{-1}, equivalent to approximately 10% of the observed maximum current speed of 0.5 ms^{-1}, while the

SSH RMSE was 0.06 m. The modelled principal tidal-current axis was 93.0°, compared with an observed value of 80.9°, a difference of 12.1°. For the lower-frequency signals, residual-current and residual-SSH bias values were negligible, and RMSE values were 0.17 ms^{-1} and 0.06 m, respectively. Residual SSH was reproduced well (R^2=0.86), whereas residual-current correlation was considerably lower (R^2=0.16). The model therefore reproduced the magnitude of non-tidal currents but captured relatively little of their temporal variability.

SAMUSG is located in approximately 20 m of water in Upper Spencer Gulf, with sensors positioned approximately 0.5–1.0 m above the seabed (Figure 7). Near-bottom temperature was reproduced very well, with a bias of 0.47 °C, an RMSE of 0.55 °C and an R^2 approaching 1.00. The model captured almost all observed temperature variability, although it retained a modest warm bias. Salinity performance was weaker, with a bias of 0.55 PSU, an RMSE of 2.81 PSU and an R^2 of 0.12. The relatively large RMSE and low explained variance were partly associated with biofouling of the conductivity sensors. Despite this observational limitation, the model reproduced the broad seasonal cycle and tidally driven salinity variability.

Tidal currents and sea-surface height were reproduced strongly, with R^2 values of 0.93 and 0.94, respectively. The tidal-current RMSE was 0.10 ms^{-1} and the SSH RMSE was 0.36 m. The modelled principal tidal-current axis was 51.3°, compared with an observed orientation of 53.3°, a difference of only 2.0°. Residual SSH was also reproduced well, with a bias of -0.01 m, RMSE of 0.07 m and an R^2 of 0.92. In contrast, residual-current performance was reduced, with a bias of 0.01 m s^{-1}, RMSE of 0.36 m s^{-1} and an R^2 of 0.01. The model therefore reproduced the dominant tidal circulation at SAMUSG and the magnitude of the residual currents but did not capture the temporal variability of residual currents.

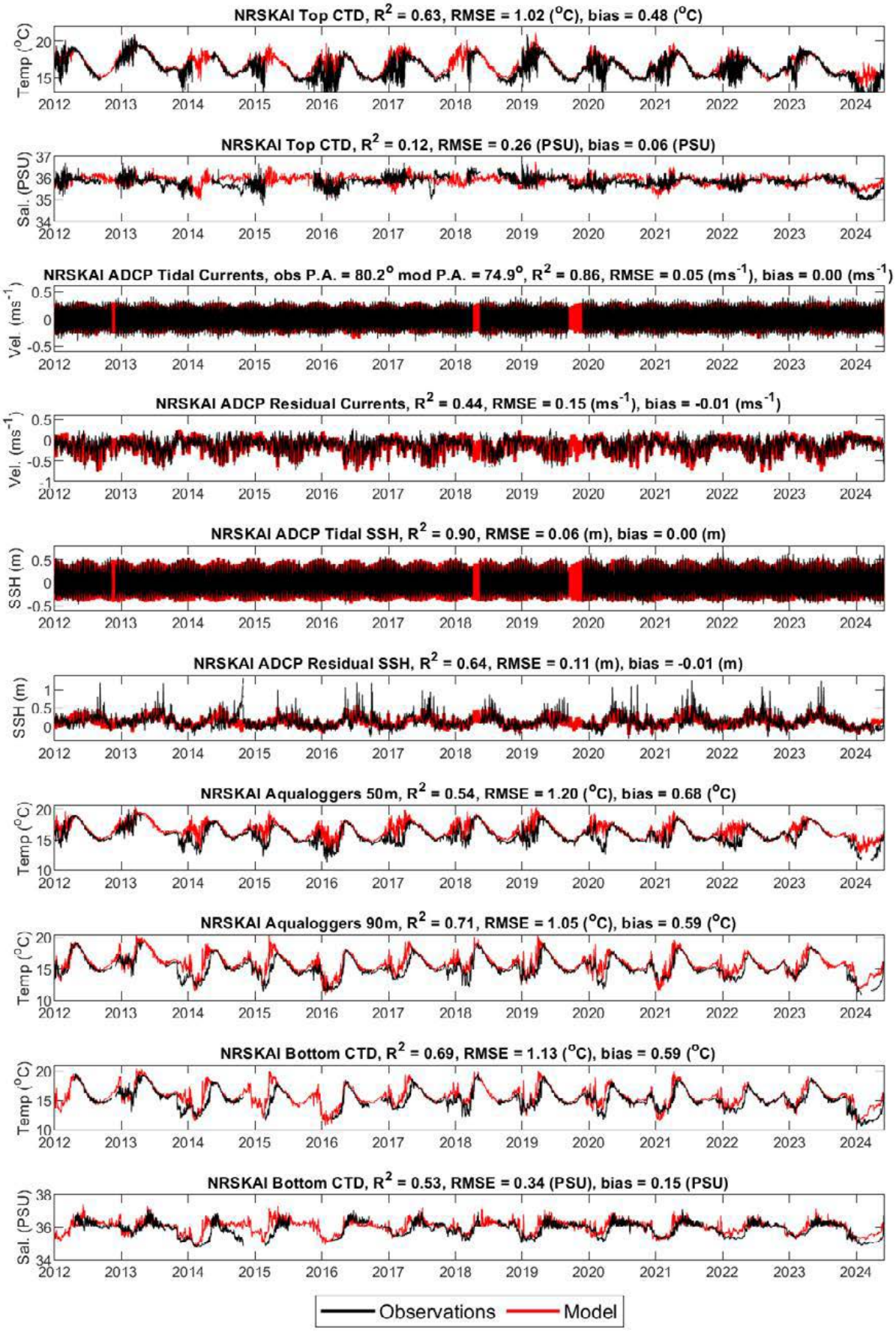


Figure 5. Validation results for currents (tidal and residual), temperature (°C), salinity (PSU), and sea-surface height (SSH; tidal and residual) at the IMOS NRSKAI location.

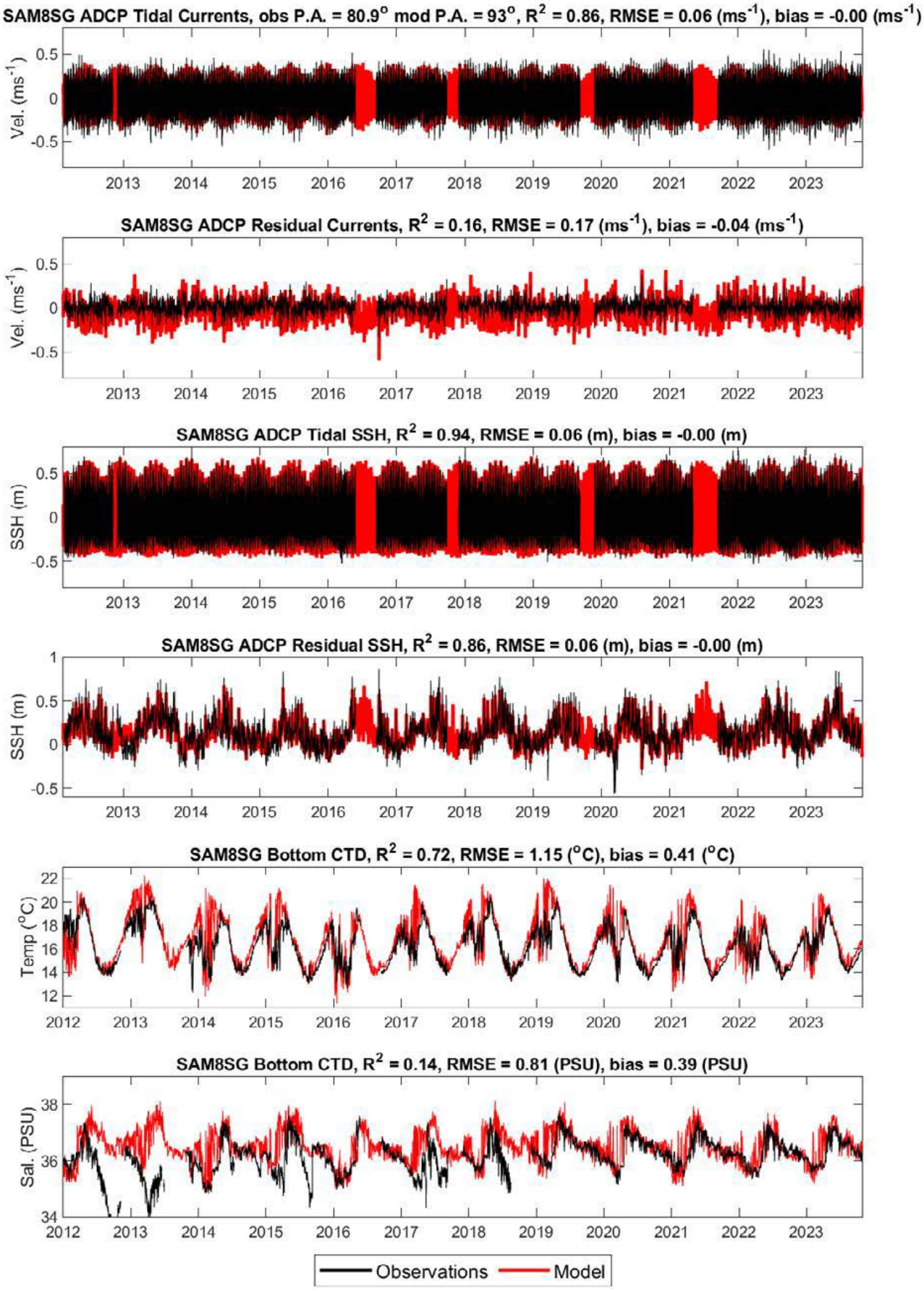


Figure 6. Validation results for currents (tidal and residual), temperature (°C), salinity (PSU), and sea-surface height (SSH; tidal and residual) at the IMOS SAM8SG location.

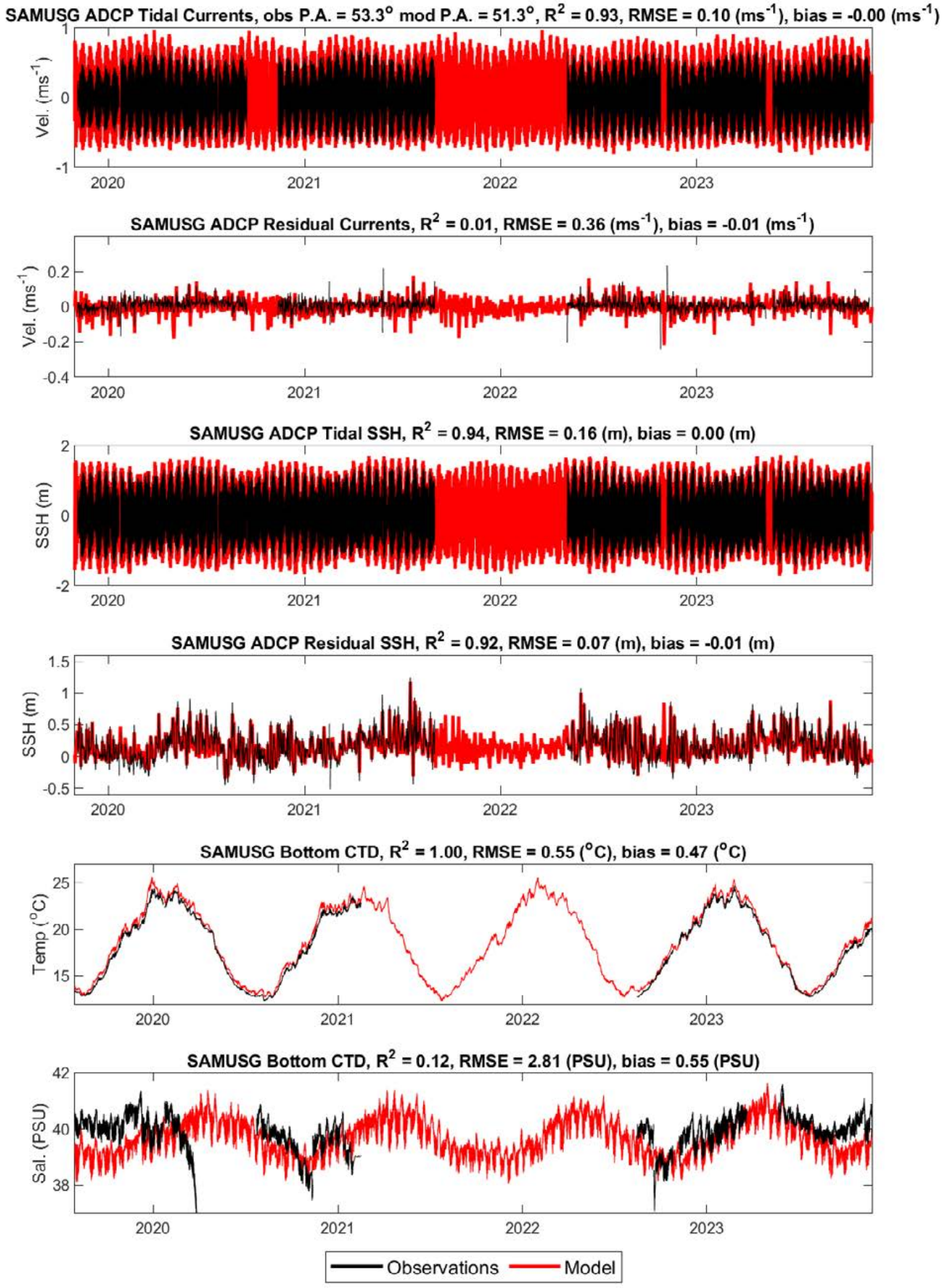


Figure 7. Validation results for currents (tidal and residual), temperature (°C), salinity (PSU), and sea-surface height (SSH; tidal and residual) at the IMOS SAMUSG location.

3.3. Comparison with Northern Water Observations

Comparisons with the observations collected from Northern Water moorings (NWP1–NWP5) are presented in Figures 8–12. These moorings broaden the assessment of model performance across the different hydrographic regimes of Spencer Gulf. All sites included measurements of temperature and salinity, and additional current observations were collected at sites NWP1, 3 and 5. Overall performance was consistent with that obtained at the long-term IMOS sites, although model validation varied among parameters and locations.

Temperature was reproduced very well across the NWP sites (Figure 8 toFigure 12) Absolute biases were ≤0.46 °C, RMSE values ≤0.54 °C and R^2 values ≥ 0.98. These results indicate that the model accurately reproduced both mean temperature and its temporal variability over the validation period. Salinity biases ranged from −0.48 PSU at NWP4 to 0.03 PSU at NWP2, while RMSE values ranged from 0.17 PSU at NWP5 to 0.68 PSU at NWP4. Across all sites, the R^2 value was ≥0.68, demonstrating that the model reproduced most of the observed salinity variability. Salinity remained slightly underestimated in the northern Gulf, most likely because the coarse spatial resolution of the atmospheric forcing underestimates evaporation over this relatively small and shallow region. This sensitivity is less pronounced in the wider and deeper lower Gulf. Incorporating station-based humidity climatology's and relaxation toward the Nunes–Lennon salinity climatology (Appendix A) substantially improved the simulation, reducing salinity-bias magnitudes from as much as 2 PSU to less than 0.5 PSU at all NWP sites.

At NWP1, NWP3 and NWP5, tidal currents and sea-surface height agreed strongly with observations (Figure 8 toFigure 12). Tidal-current bias values were negligible, RMSE values ≤0.08 ms^{-1} and R^2 values ≥0.90. SSH bias values were negligible, RMSE values ≤0.14 m, with R^2 values ≥0.94. Differences between modelled and observed principal tidal-current orientations were no greater than 5.5°. Residual SSH was also reproduced well, with RMSE values ≤0.06 m and R^2 values ≥0.89. Residual-current biases were negligible, with RMSE values ≤0.30 ms^{-1} and R^2 values ≤0.03. The model therefore represented the general magnitude of the residual currents but did not replicate fine-scale temporal variability. The low correlation is attributed to

limitations in the spatial and temporal resolution of the wind forcing, as well as the statistical limitations of comparing datasets characterized by very low variability.

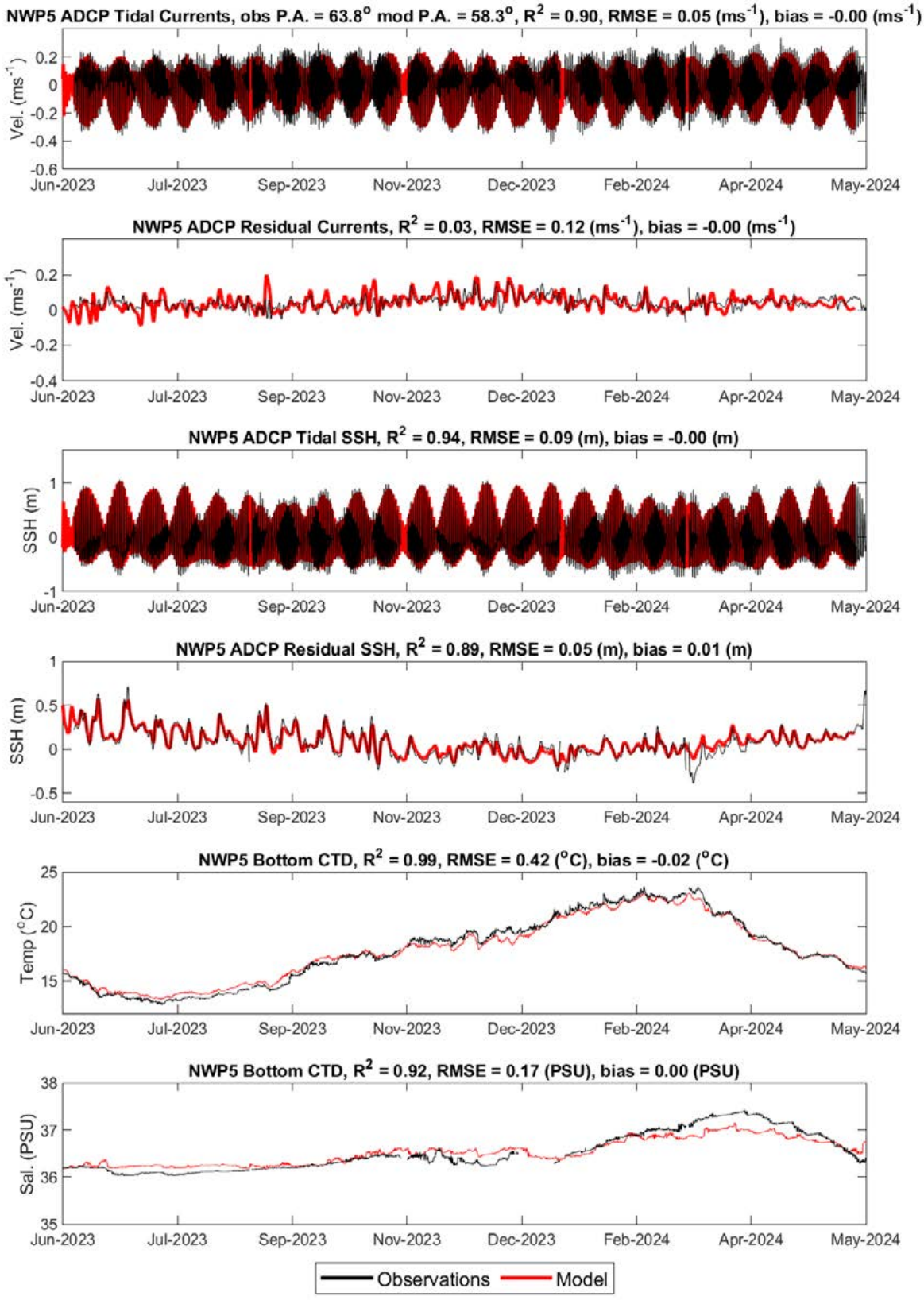


Figure 8. Validation results for currents (tidal and residual), temperature (°C), salinity (PSU), and sea-surface height (SSH; tidal and residual) at the NWP5 location.

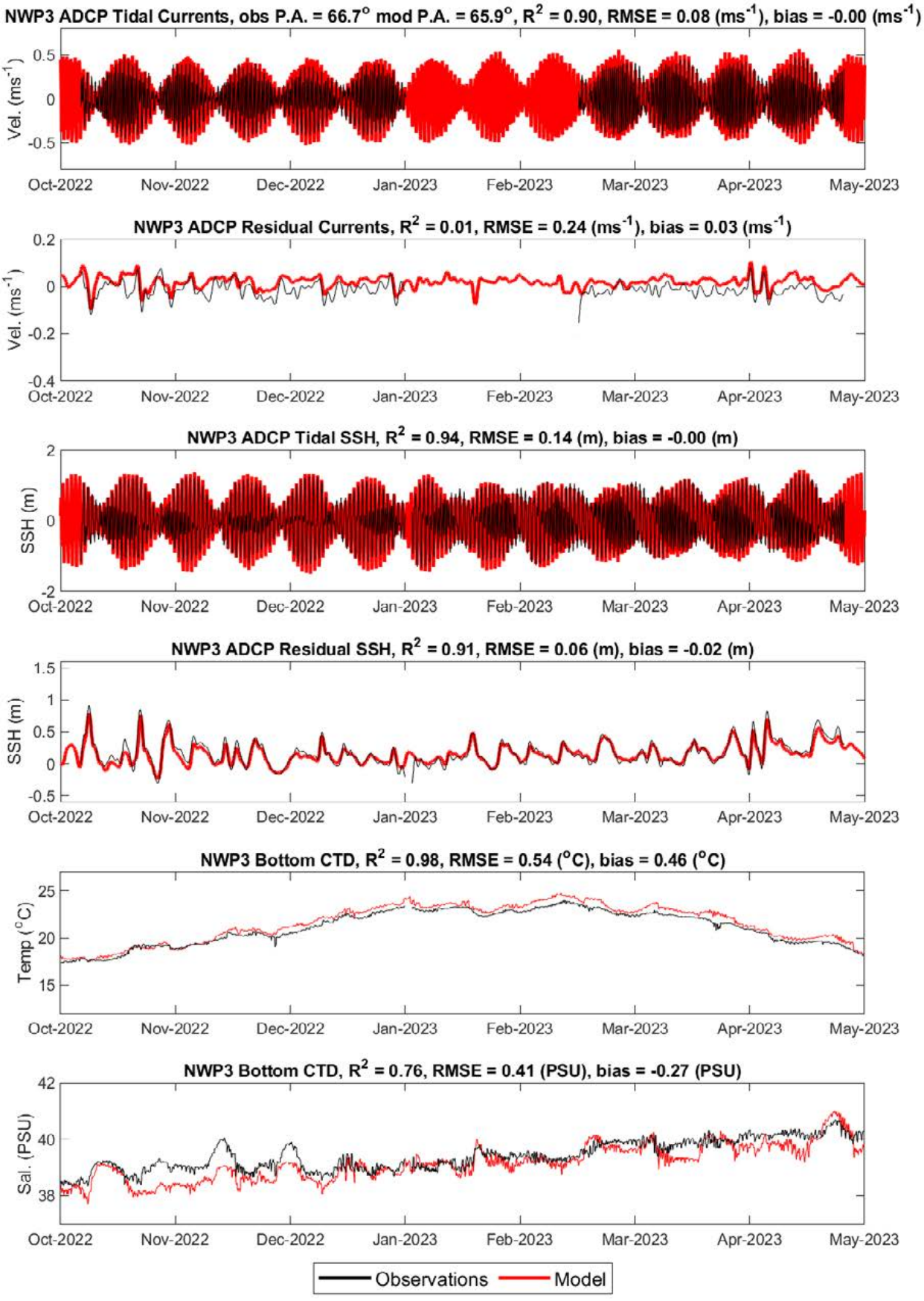


Figure 9. Validation results for currents (tidal and residual), temperature (°C), salinity (PSU), and sea-surface height (SSH; tidal and residual) at the NWP3 location.

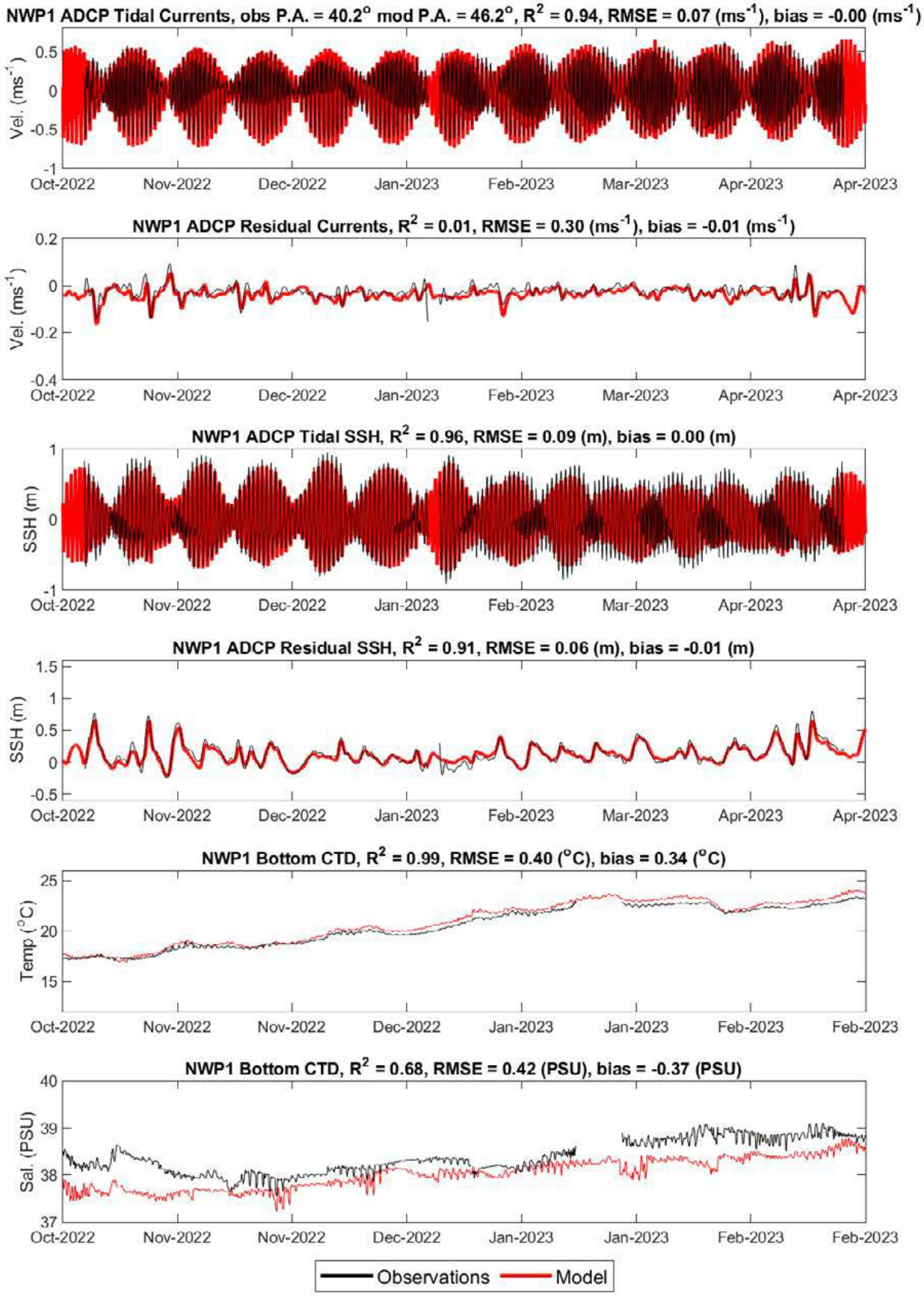


Figure 10. Validation results for currents (tidal and residual), temperature (°C), salinity (PSU), and sea-surface height (SSH; tidal and residual) at the NWP1 location

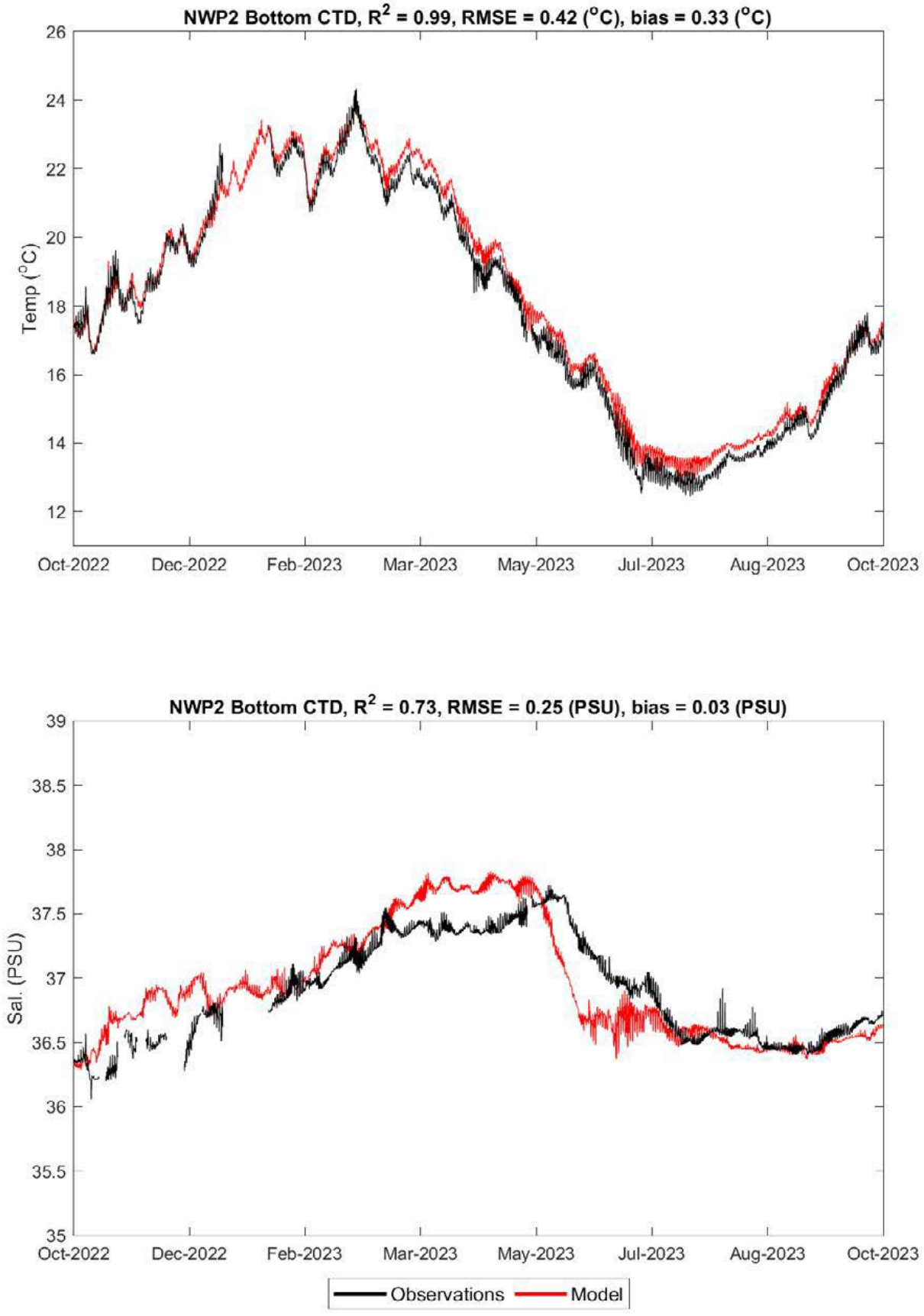


Figure 11. Validation results for temperature (°C) and salinity (PSU) at the NWP2 location.

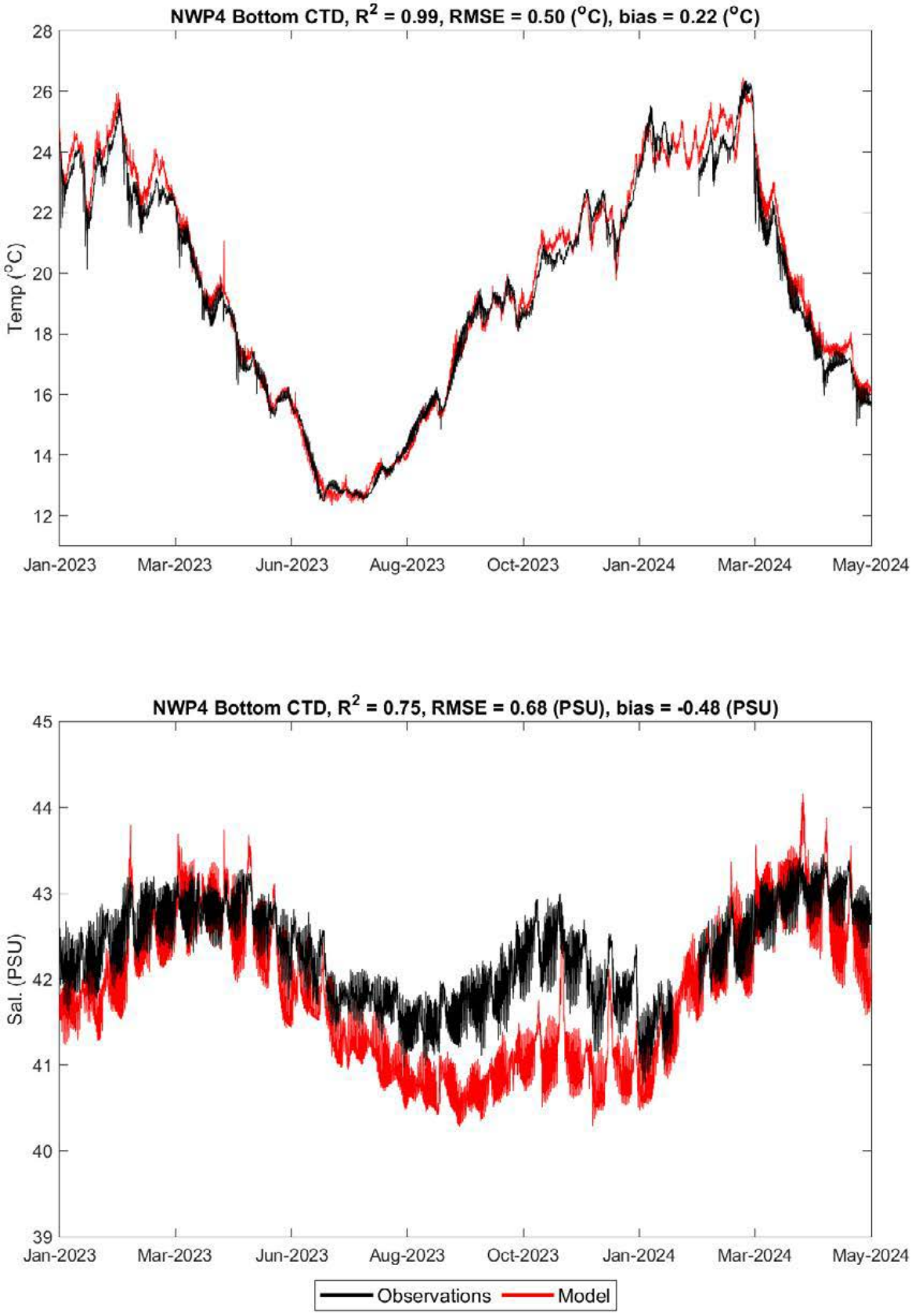


Figure 12. Validation results for temperature (°C) and salinity (PSU) at the NWP4 location.

Comparisons of vertical profiles of temperature and salinity made along- (north–south) and across- (west–east) the gulf are presented in Figure 13 to Figure 16.

In January 2023, the model reproduced the observed along-gulf distributions, including the north–south temperature and salinity gradients, summer upwelling near the southern entrance to Spencer Gulf, and the density minimum near the gulf mouth at approximately 34.5°S (Figure 13). Observations along the across-gulf transect in January were limited to the western side of the gulf (Figure 14). Within this observational constraint, the model reproduced the measured temperature, salinity and density structure, including the relatively lower-salinity water on the western side and the sloping isopycnals associated with weak westward transport of denser water from the eastern side of the gulf.

In May 2023, the model slightly underestimated salinity in Upper Spencer Gulf north of 33.0°S (Figure 15). Nevertheless, it reproduced the observed along-gulf distributions of temperature, salinity and density, including the bottom outflow of high-density water, which was most pronounced between approximately 33.4°S and 33.8°S. The across-gulf comparison (Figure 16) also showed that the model reproduced the characteristic density-driven circulation pattern: an inflow of slightly cooler, lower-salinity water along the western side of the gulf and an outflow of relatively warmer, higher-salinity water along the eastern side.

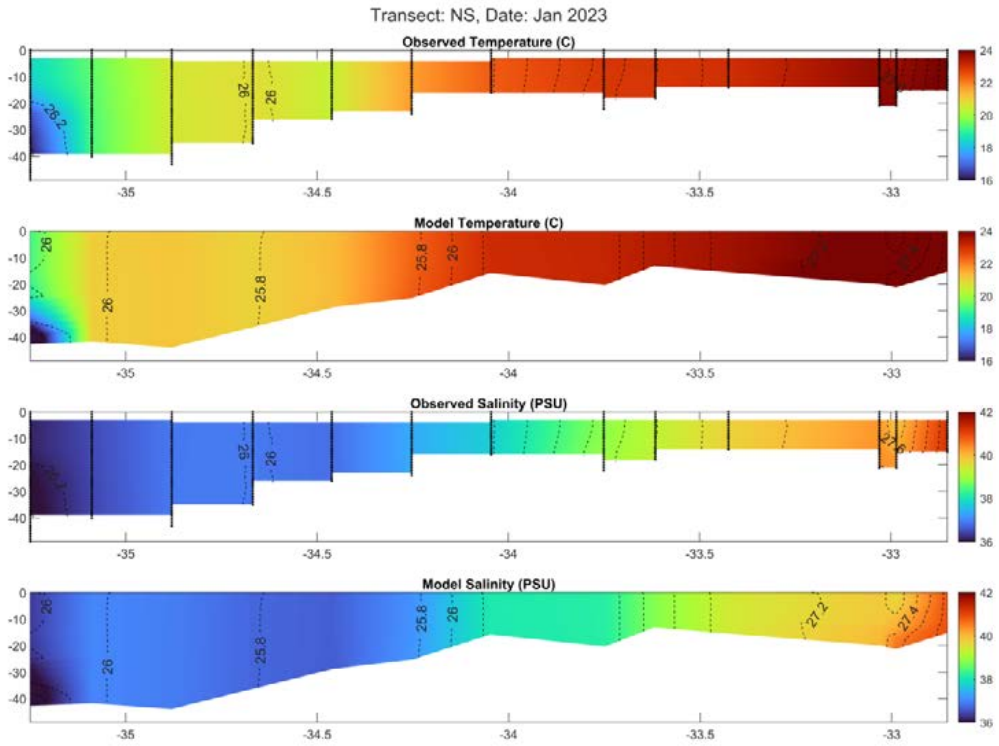


Figure 13. Along gulf (north-south) comparison of observed and modelled temperature (ºC) and salinity (PSU) signals with density (σ, kg m^{-3}) contours in January 2023.

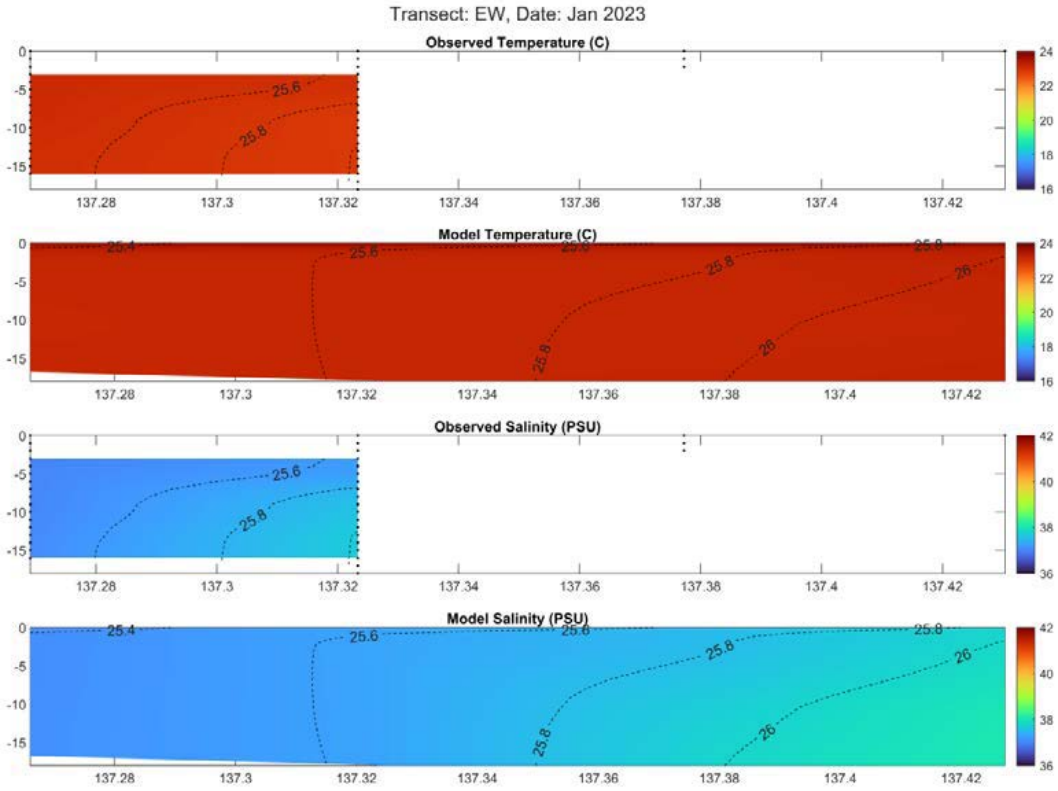


Figure 14. Across gulf (west-east) comparison of observed and modelled temperature (ºC) and salinity (PSU) signals with density (σ, k gm^{-3}) contours in January 2023.

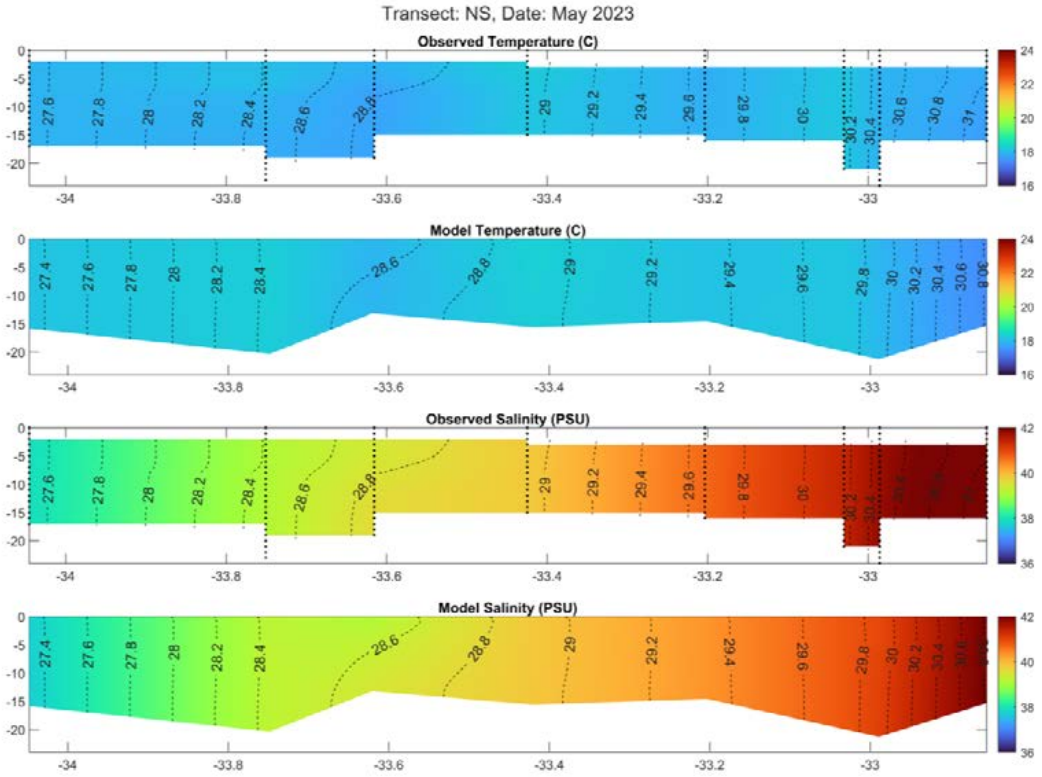


Figure 15. Along gulf (north-south) comparison of observed and modelled temperature (°C) and salinity (PSU) signals with density (σ, k gm^{-3}) contours in the upper half of Spencer Gulf in May 2023.

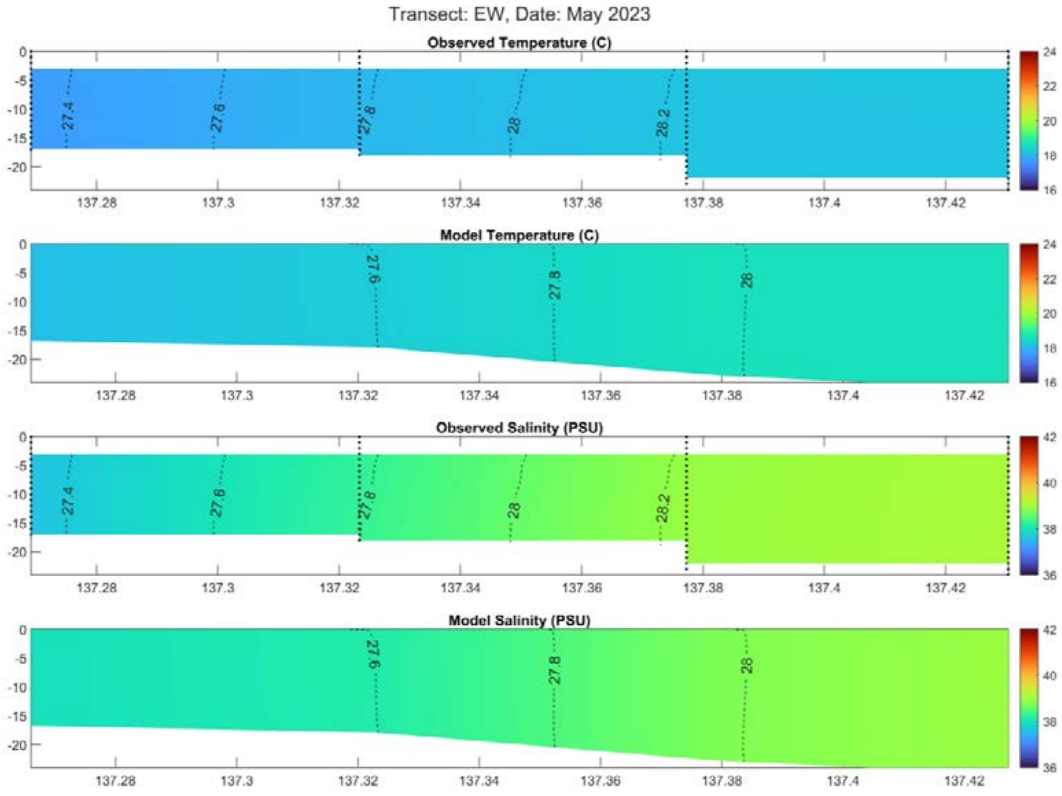


Figure 16. Across gulf (west-east) comparison of observed and modelled temperature (°C) and salinity (PSU) signals with density (σ, k gm^{-3}) contours in the upper half of Spencer Gulf in May 2023.

3.4. Overall Model Performance

The Spencer Gulf Hydrodynamic Model (SGM1000) was validated against an extensive set of measurements including long-term IMOS (2012–2024) and targeted Northern Water observations spanning Spencer Gulf, including the proposed desalination site, as detailed above. The model performance was assessed against measures of sea surface height, tidal and residual currents, temperature, salinity and CTD transects and demonstrated that the model provides a reliable representation of Spencer Gulf hydrodynamics over tidal, weather-band, seasonal and interannual timescales.

The model reproduced the dominant tidal dynamics exceptionally well throughout Spencer Gulf. Tidal current predictions explained 86–94% of the observed variance (R^2 = 0.86–0.94), with tidal current RMSE values of only 0.05–0.10 ms^{-1}, representing a mean error of approximately 10% of the observed tidal current speeds. The orientation of the principal tidal-current axis differed from observations by only 2–12°. Tidal variations in sea-surface height were similarly reproduced with high accuracy (R^2 = 0.90–0.96) and negligible bias. While residual sea level variability was reproduced well (R^2 up to 0.92), the model explained relatively little of the short-term temporal variability in residual currents (R^2 ≤0.16). This is attributed to limitations in the model bathymetry and the coarse spatiotemporal resolution of atmospheric wind forcing, which reduced the temporal correlation. Importantly, the magnitude of residual currents remained well represented.

Temperature was also strongly reproduced by the model. Across the long-term IMOS sites, temperature biases were generally <0.7 °C, with RMSE values <1.2 °C and R^2 values >0.6. Model performance was improved at several Northern Water mooring locations, where absolute biases were ≤0.46 °C, RMSE values ≤0.54 °C, and R^2 values were ≥0.98, demonstrating that the model reproduced both seasonal and interannual temperature variability.

Salinity represents one of the most demanding validation variables because of limitations in global atmospheric forcing relative to the shallow and narrow geometry of Upper Spencer Gulf, and difficulties in maintaining the long-term accuracy of moored CTD measurements due to biofouling and sensor drift. Salinity predictions showed good agreement with observations but exhibited greater uncertainty than temperature. Comparisons against long-term IMOS observations showed salinity biases ≤0.55 PSU, although explained variance varied considerably (R^2 = 0.12–0.53). Corrections to local humidity measurements and relaxation towards the Nunes–Lennon salinity climatology substantially improved performance in Upper Spencer Gulf, reducing salinity bias from approximately 2 PSU to less than 0.5 PSU.

Validation against independent CTD transects further demonstrated that SGM1000 reproduced the large-scale three-dimensional hydrographical structure of Spencer Gulf. The model successfully reproduced the observed north–south temperature salinity and density gradients and summer upwelling near the gulf entrance. It also reproduced the observed cross-gulf density structure associated with the Gulf's characteristic inverse estuarine circulation.

Overall, the SGM1000 validation provides a comprehensive assessment of model performance across long-term observations, a broad range of environmental conditions and multiple locations spanning Spencer Gulf using quantitative performance metrics (bias, RMSE and R^2). The metrics demonstrate that the model can reliably reproduce the dominant physical processes necessary for predicting far-field desalination return water dispersion, cumulative salinity impacts and larval connectivity. The metrics also indicate an improvement in model predictive capacity relative to previous whole-of-gulf models that were based on reduced spatial and vertical resolution (Teixeira, 2010), simplified climatological atmospheric forcing (Kämpf et al., 2009; Kämpf, 2026) or limited simulation run-times (Middleton et al., 2013; 2014).

4. RESULTS: RETURN WATER DISPERSION MODELLING

4.1. Regional Circulation and salinity anomalies at mooring sites

This section presents the predicted fate of desalination return water released from the proposed Mullaquana Station outfall in Upper Spencer Gulf.

Initial dilution, transport and long-term flushing of return water discharge are controlled by a hierarchy of currents operating across overlapping temporal and spatial scales that ultimately determine the circulation within Spencer Gulf. These currents include high-velocity, oscillatory, semi-diurnal tidal currents reaching up to 0.5 ms^{-1}, wind driven currents of less than 0.15 ms^{-1} operating on weekly timescales and seasonal-scale density driven currents with velocities typically less than 0.05 ms^{-1} (Figure 9).

Figure 17 shows the mean time and depth-averaged seasonal circulation patterns derived from the 12-year model simulation without desalination. Temporal averaging removed sub-daily transient and periodic flows due to wind and tidal currents and provides a climatology of the long-term circulation of the region. Model results show a clockwise circulation was maintained year-round with predicted mean current velocities less than 0.03 ms^{-1} consistent with the moored observations (Figure 5 to Figure 12). Seasonal differences in the circulation were most pronounced between summer and winter and are described below. The circulation in spring and autumn showed a transitional phase between the summer and winter patterns, respectively.

In summer, residual current speeds were weaker throughout the Gulf. The net flow along the western side of the Gulf is northwards, with stronger currents occurring along the coast, particularly in Upper Spencer Gulf. There is a weak southward return flow of water along the central axis of Upper Spencer Gulf and the eastern side of lower Spencer Gulf. Exchange between the shelf and the Gulf is limited during summer due to weaker along-gulf density gradients and the development of a density minimum at the entrance to the Gulf (Figure 13).

In contrast, during winter the residual circulation increases. This is due to the additive effect of local wind forcing and seasonal cooling that increases the along-gulf density gradient and the resulting density driven circulation (Figure 15). Exchange between the shelf and the Gulf increases and is characterised by the outflow of dense, high-salinity water on the eastern side of the Gulf, which is compensated for by an increased entrainment of water from the shelf in the central-western region of Spencer Gulf (Figure 16).

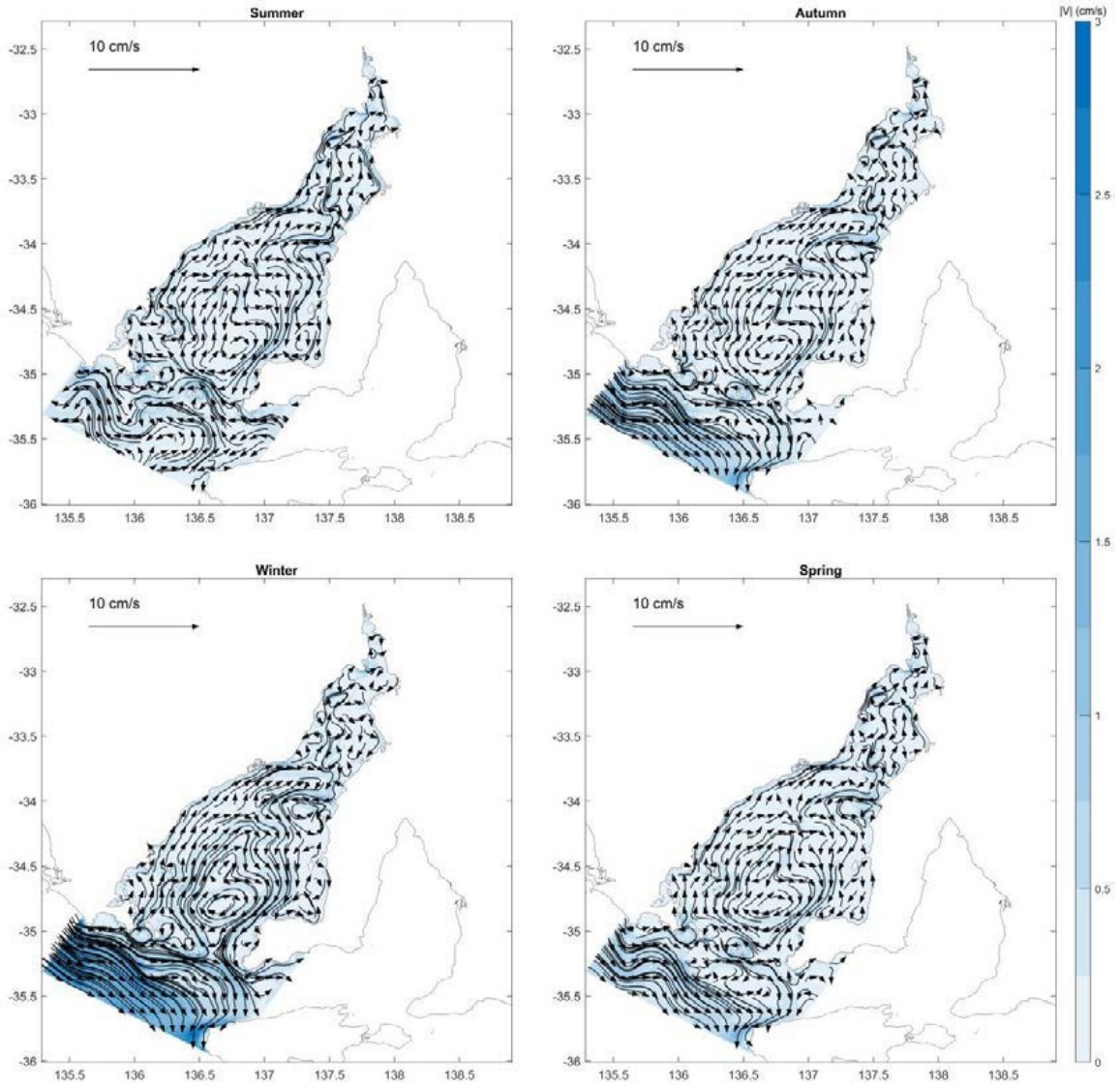


Figure 17. Modelled 12-year and depth-averaged seasonal circulation patterns within Spencer Gulf.

Figure 18. Modelled 12-year and depth-averaged seasonal circulation patterns within Upper Spencer Gulf (close-up from Figure 17). shows a close-up of the mean depth-averaged seasonal circulation in Upper Spencer Gulf. Circulation in Upper Spencer Gulf is characterised by northward flows along both sides of the gulf and southward flow in the centre in all seasons except for winter, when there is a southward flow of water along the eastern side of the gulf. A persistent weak cyclonic gyre is centred slightly to the east of the proposed desalination intake and outfall. At the desalination site, northward flows associated with this gyre intensify slightly during summer and spring relative to winter and autumn, likely in

response to changes in the seasonal wind forcing. Other small-scale persistent gyre-like features are predicted either side of Point Lowly.

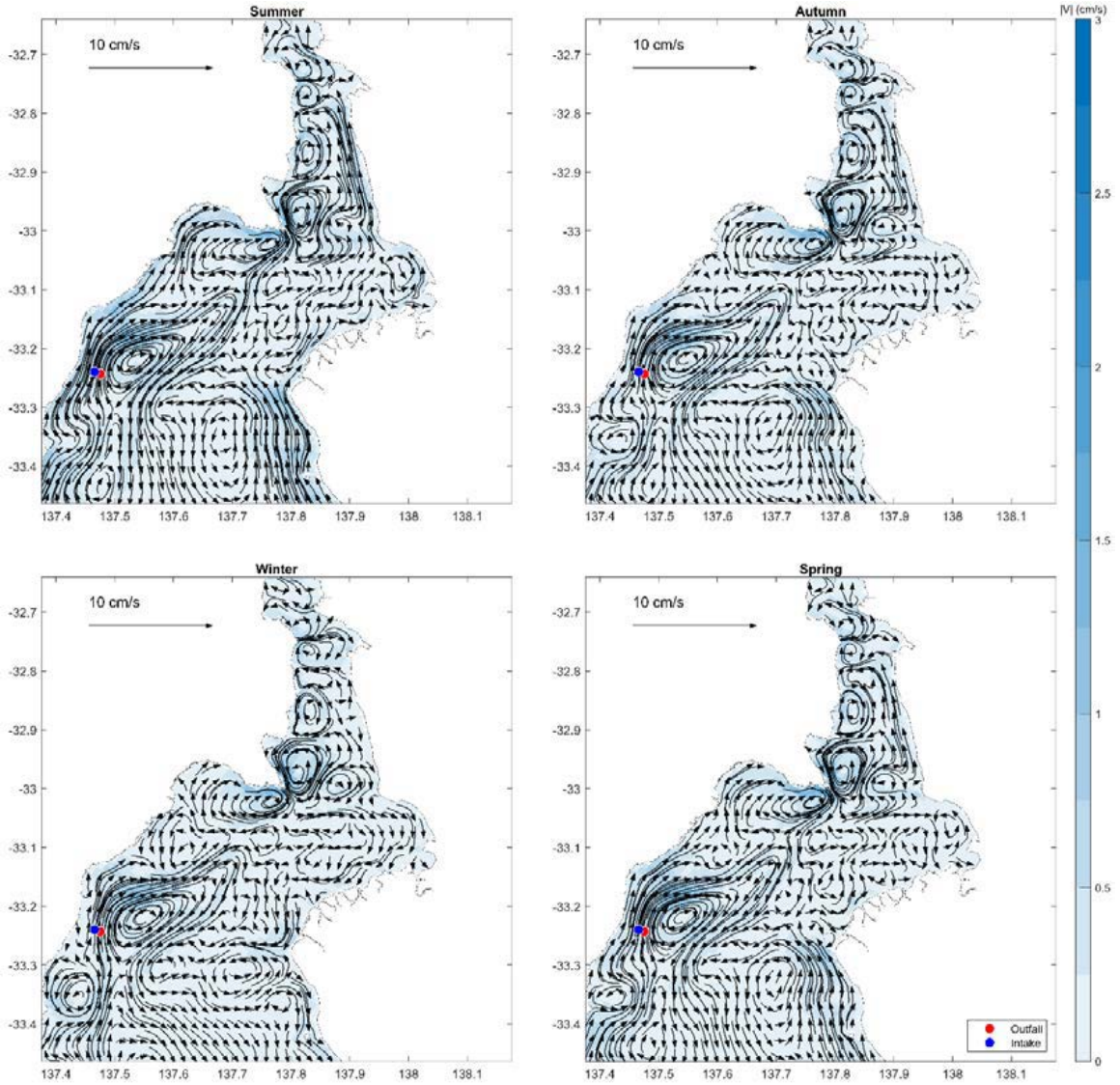


Figure 18. Modelled 12-year and depth-averaged seasonal circulation patterns within Upper Spencer Gulf (close-up from Figure 17).

Salinity concentrations are largest on the bottom where desalination return water discharge is injected into the model. A snapshot of the instantaneous bottom salinity across Upper Spencer Gulf for model scenarios with and without return water discharge is shown in Figure 19. Return water discharged from the Mullaquana Station desalination plant is predicted to initially flow south-eastward down slope (i.e. cross isobaths) due to its increased density relative to the surrounding ambient water.

Comparison of hourly model predictions of bottom salinity with and without desalination at the proposed desalination intake, outfall and mooring locations used for model validation Figure 20) were used to estimate the maximum expected salinity anomalies (∆S) associated with tidal variability, including dodge tide periods (Figure 21). In addition, a low-pass filter (30-day running-mean) was applied to the hourly anomaly time series to remove tidal variations and assess potential salt accumulation over larger spatial and temporal scales. The long-term mean difference ($\overline{\Delta S}$) and rate of change (i.e. trend) given by linear regression over the 12-year simulation were estimated to assess how much extra salt is in the water at each location on average and to understand if salt accumulation was occurring on inter-annual timescales, respectively.

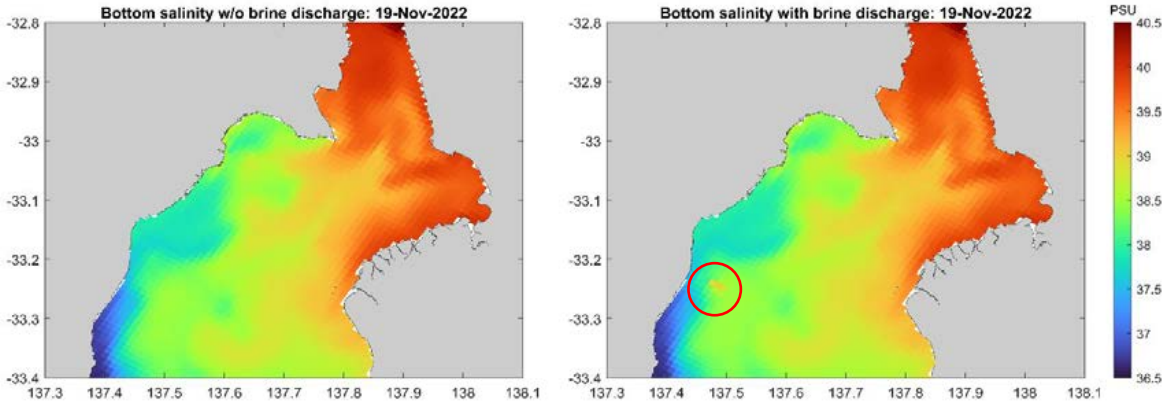


Figure 19. Snapshot of modelled bottom salinities in Spencer Gulf (left) without return water discharge and (right) with return water discharge (highlighted in red circle) at Mullaquana Station on 19 Nov 2022.

Predicted salinity differences between simulations with and without return water discharges were small. Maximum hourly salinity anomalies decreased rapidly from ≤1.7 PSU in the bottom sigma layer corresponding with the outfall location to ≤0.7 PSU at the intake located approximately 1 km away and ≤0.4 PSU at the IMOS SAMUSG site located approximately 25 km to the northeast. Because the return water discharge is an active tracer that modifies the density field in the model, non-linear oceanographic features like eddies that are sensitive to small perturbations will not be exactly aligned between the simulations with and without return water discharges leading to positive and negative anomalies. These anomalies are greatest at monitoring sites located near the entrance to southern Spencer Gulf (NRSKAI and SAM8SG) where passing eddies transport salt from Upper Spencer Gulf to the open shelf.

Over longer timescales, 30-day low-pass filtered anomalies and mean increases across the 12-year hindcast at all sites located greater than several kilometres from the outfall location in Upper Spencer Gulf (NWP1, NWP2, NWP4, SAMUSG) were very small (≤0.01PSU). These

increases are much smaller than the daily to annual natural variability, which is of the order of 1-2 PSU (Figure 5 to Figure 12) and below the detection resolution of CTD instrumentation used in monitoring programs. Linear regression was applied to the bottom salinity over the 12-year simulation, and the trends demonstrated no accumulation of salt within Spencer Gulf over the 12-year hindcast.

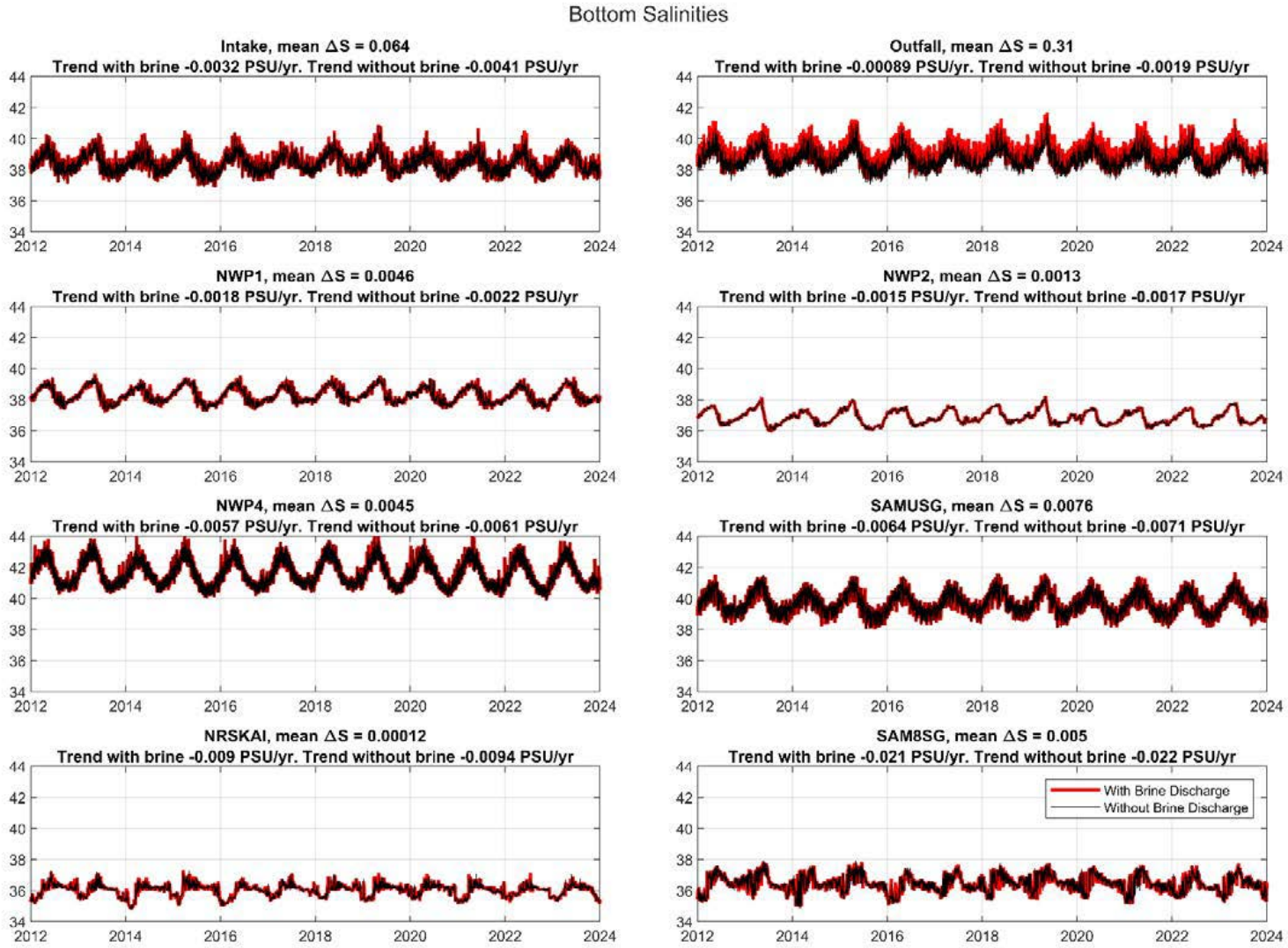


Figure 20. Hourly time series of simulated bottom salinities for model scenarios with (red line) and without (black line) return water discharge at the proposed desalination intake, outfall and mooring locations.

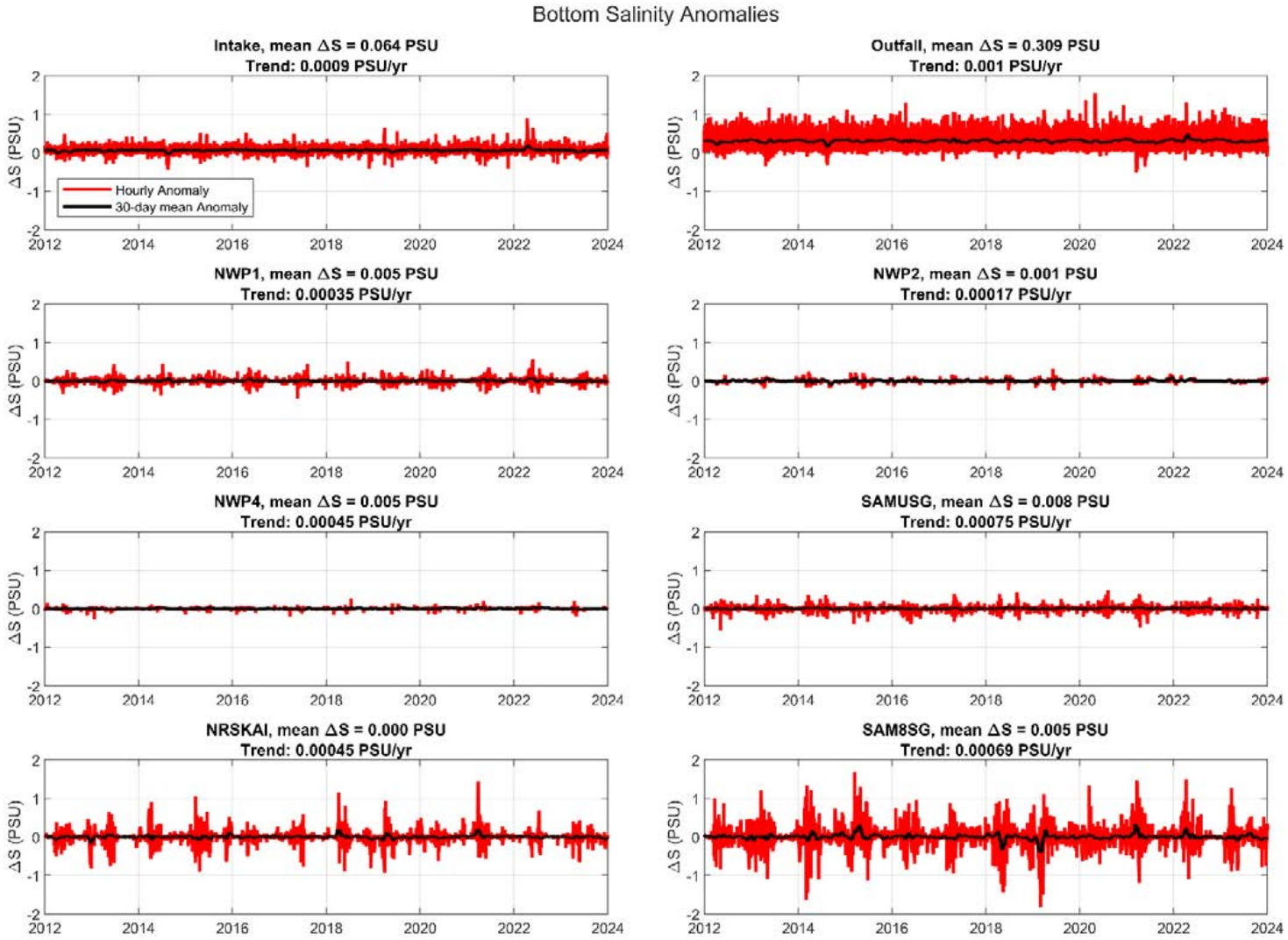


Figure 21. Hourly time series of simulated bottom salinity anomalies for model scenarios with and without return water discharge showing anomaly based on raw output (red line) and on 30-day averaged output (black line).

4.2. Seasonal Salinity Anomalies

The predicted seasonally averaged spatial distribution of maximum daily salinity anomalies provides an understanding of the extent and changes in spatial footprint of salinity changes resulting from return water discharge (Figure 22 and Figure 23). Predicted seasonal anomalies between 0.05 and 0.1 PSU (0.15 – 0.25% change in the ambient salinity) are limited to within approximately 5 km of the outfall and align with the principal axis of the local tidal current ellipse. Predicting salinity changes at spatial scales <5 km is limited by the model's 1 km grid resolution. For more accurate, small-scale predictions, readers should refer to near and mid-field modelling results (Water Technology, 2026a, Water Technology, b). At larger distances from the outfall the seasonal anomalies are predicted to be small (≤ 0.05 PSU) (<0.15 % change in the ambient salinity) and primarily advected north-eastwards along the coast towards the top end of Spencer Gulf, consistent with the clockwise circulation pattern shown in Figure 19. North-eastward transport is amplified in spring and summer due to the prevailing south-easterly wind transport with small anomalies of ≤0.02 PSU (<0.05 % change in the ambient salinity) predicted to extend up to 40 km away from the outfall at Point Lowly in spring. In autumn and winter, when winds are typically more from the north and west and the density driven exchange between the gulf and open shelf is enhanced, the northern extent of the return water footprint is reduced and transport is directed south. Small negative anomalies associated with modification to the density field due to simulated addition of return water results in small seasonal anomalies with an absolute value less than 0.1 PSU (<0.25 % change in the ambient salinity) predicted at distances >20 km from the outfall (Figure 22). The amplitude of the negative anomalies is predicted to be greatest in southern Spencer Gulf during winter and is the result of changes in the eddy field distribution associated with the density driven flushing of Spencer Gulf, which is greatest at this time of the year.

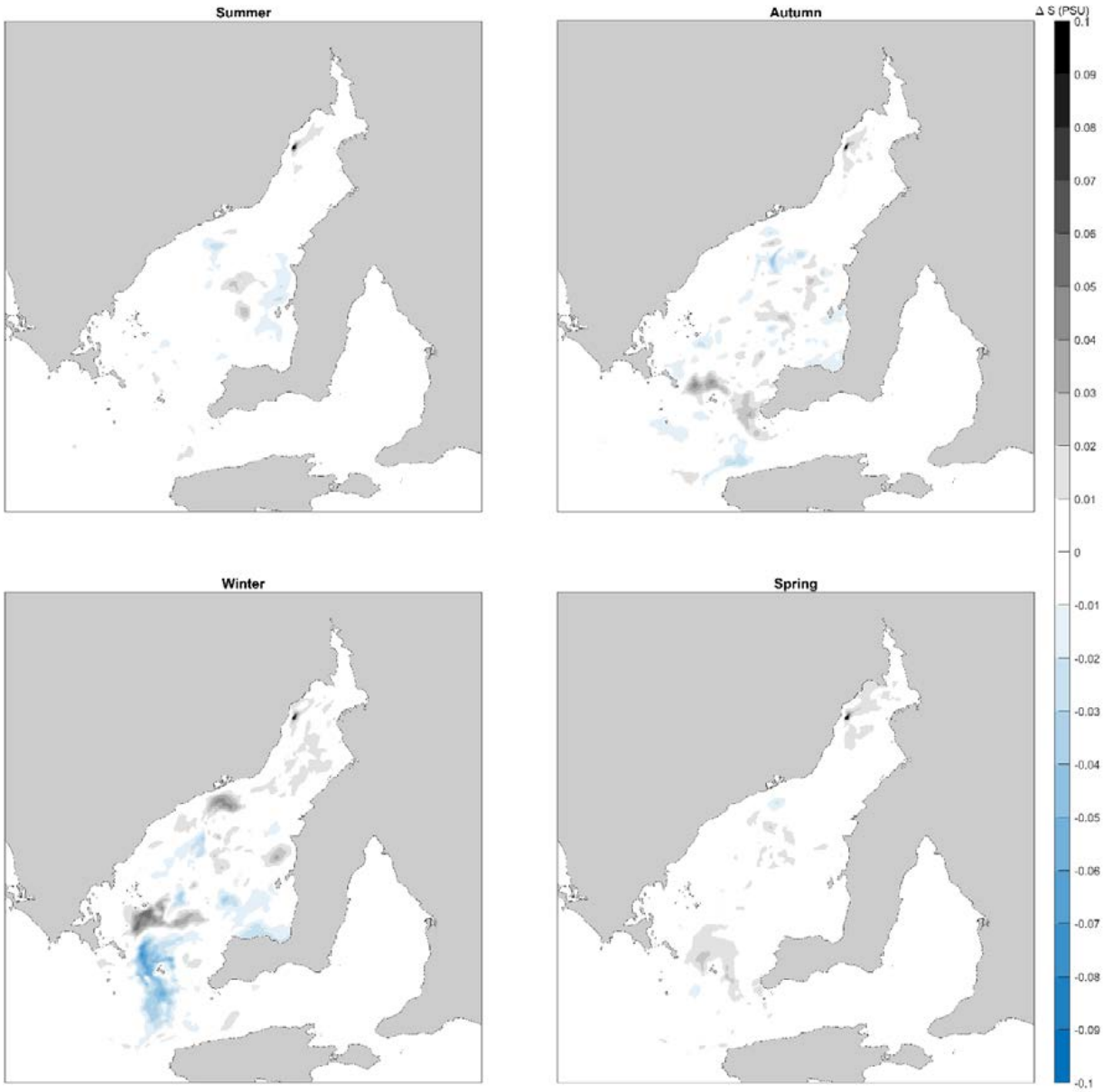


Figure 22. Seasonal-mean salinity anomalies based on maximum differences between model with return water discharge and model without return water discharge.

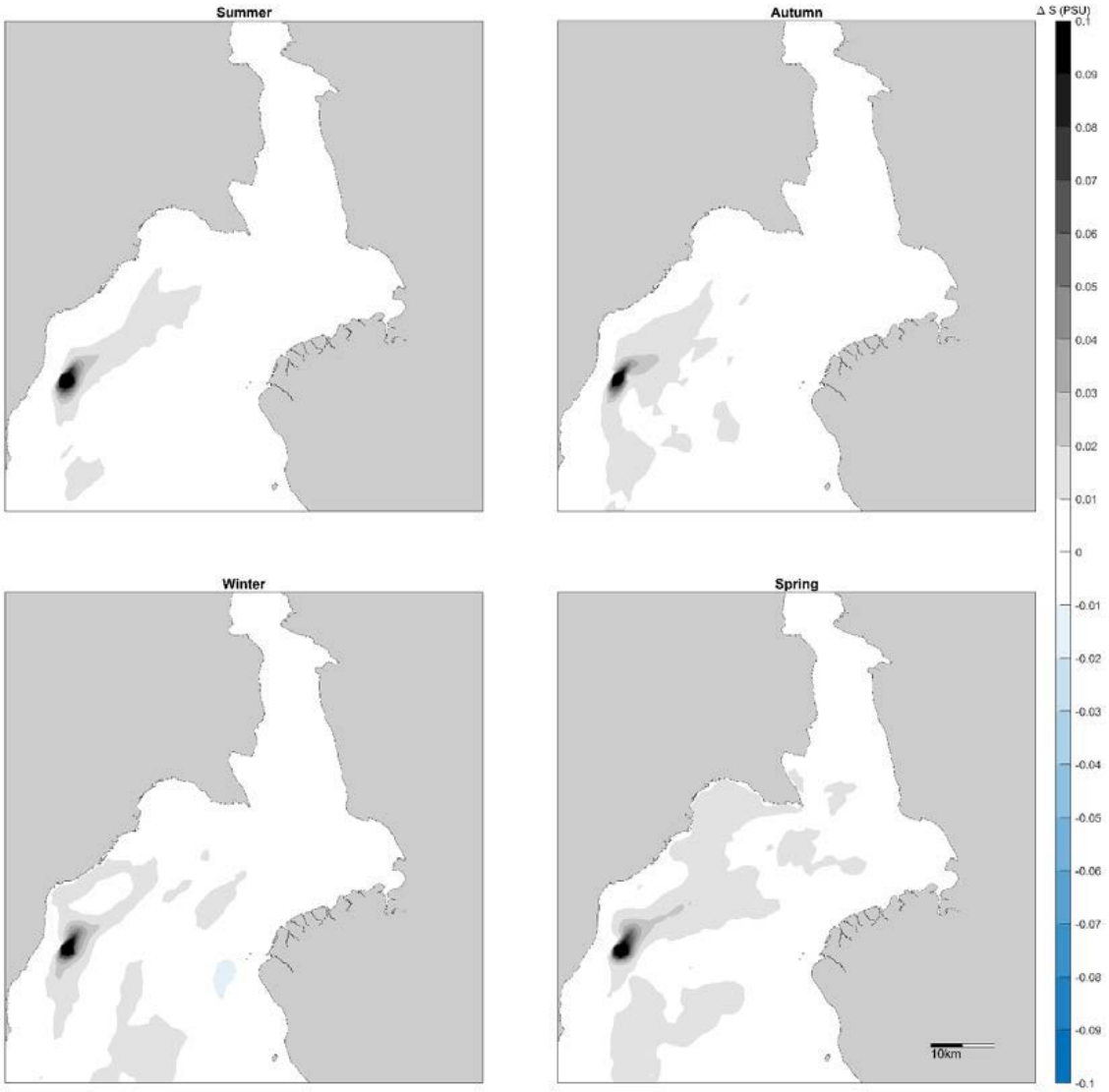


Figure 23. Close up of Upper Spencer Gulf from Figure 22. Seasonal-mean salinity anomalies based on maximum differences between model with return water discharge and model without return water discharge.

5. LARVAL BIOPHYSICAL MODELLING

5.1. Larval Connectivity Estimates

Particle density distribution maps for KGW (Figure 25), WKP (Figure 24), and SNP (Figure 26) show the final accumulated density heat maps of particles at the end of each simulation. Changes in the predicted final distributions shown across different spawning seasons (i.e. years) for each species reflect differences in the simulated ocean circulation and its interaction with the assumed modelled spawning distributions and larval behaviours.

For KGW, larvae were generally transported north towards mid-Spencer Gulf along the western boundary associated with the gulf's inverse estuarine circulation. Some larvae were lost from the model domain, particularly along the southeastern boundary where dense water outflows exit Spencer Gulf. The estimated connectivity with the 1 km boundary around the intake was <0.003% during their simulated lifespan and remained consistently low across all three years.

The WKP larval behaviour model is more complex than KGW and the tidal-stream transport applied when larvae are in waters <15 m deep, 22 days after their release resulted in a pronounced northward transport of prawn larvae to Upper Spencer Gulf. The estimated connectivity with the 1 km boundary around the intake varied during their simulated lifespan from 0.5% in 2022 to 2.6 % in 2020 and 2.7 % in 2021. The relatively lower connectivity predicted in 2022 was related to changes in the circulation that resulted in less particles being in water shallow enough to engage the tidal-stream transport behaviour.

The estimated connectivity with the 1 km boundary around the intake varied during their simulated lifespan between 6.1% in 2020 and 7.4 % in 2022. The dense accumulation of particles at the five settlement sites located in Upper Spencer Gulf, which includes the proposed desalination intake location, indicates high localised connectivity between release sites in Upper Spencer Gulf and the settlement areas.

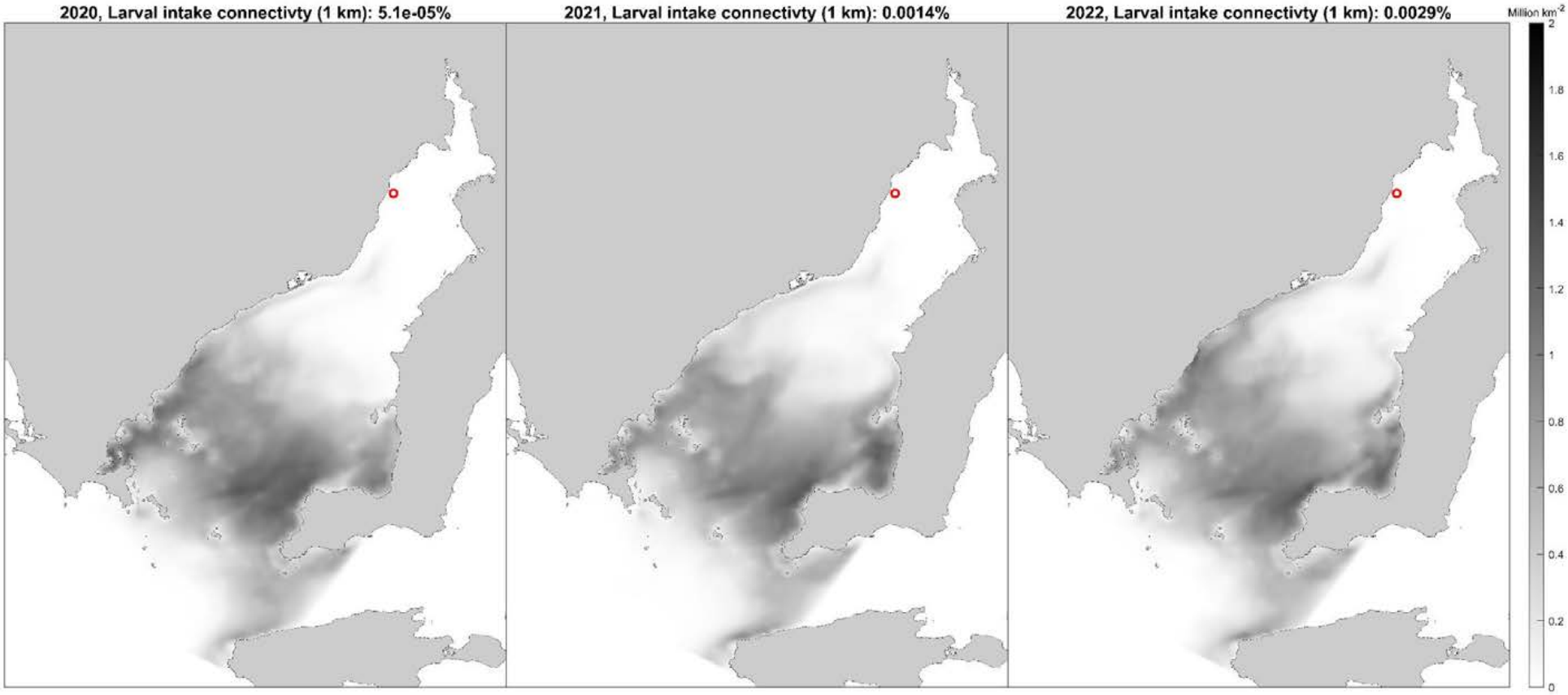


Figure 24. Heat Map of particle track densities (million km^{-2}) representing KGW larvae, released daily from April 23 to May 7 for three consecutive years (2020-2022), and tracked for up to 130 days. The red circle indicates location of desalination plant intake.

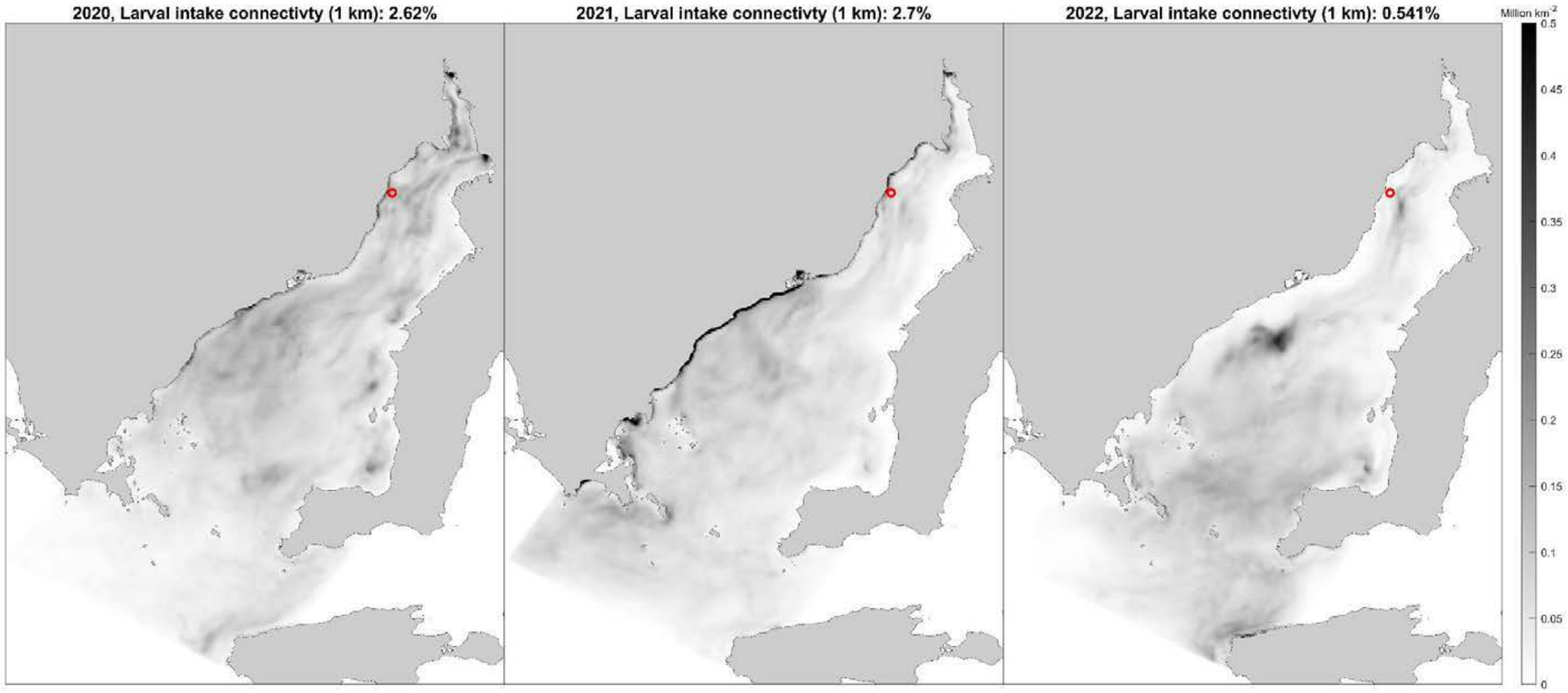


Figure 25. Heat Map of particle track densities (million km^{-2}) representing prawn larvae released on November 1, 2020-2022, and tracked for 46 days. The red circle indicates location of desalination plant intake

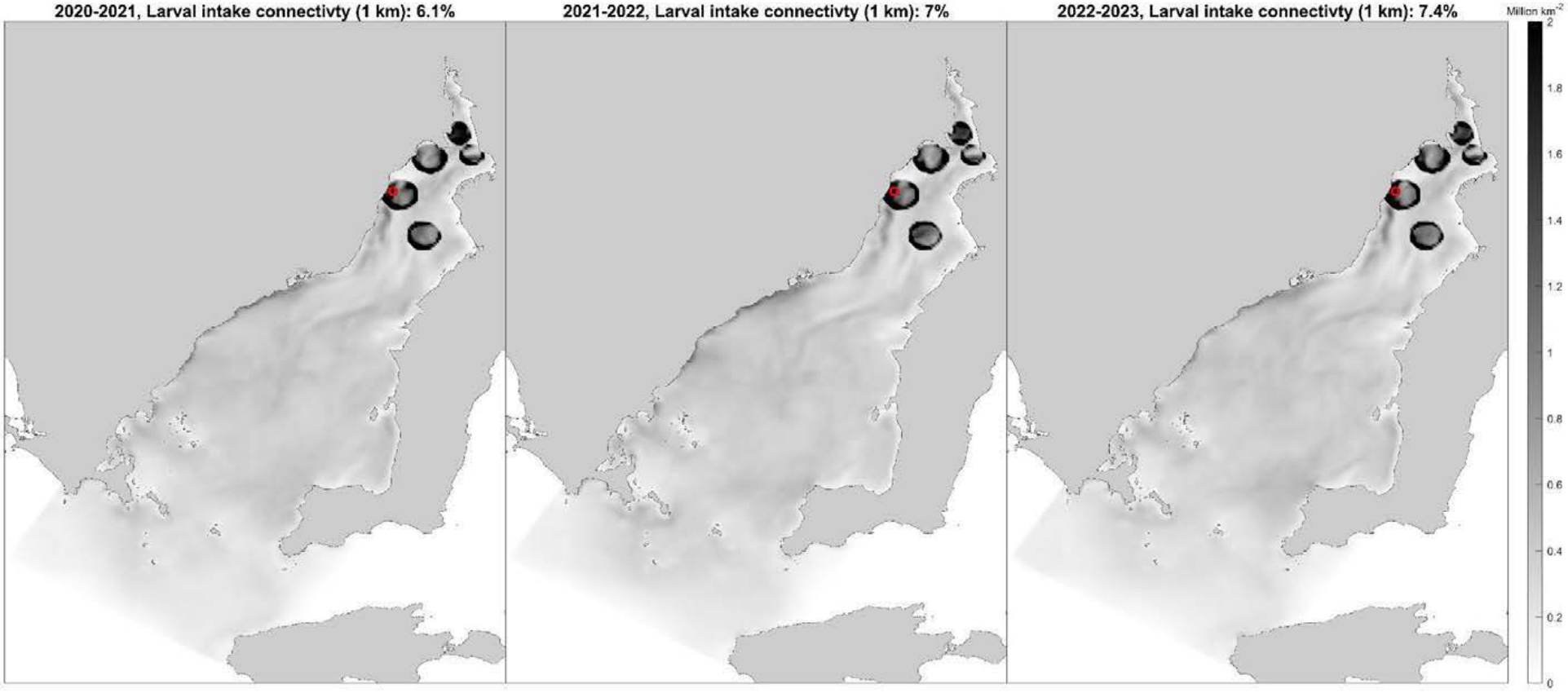


Figure 26. Heat Map of particle track densities (million km^{-2}) representing snapper larvae released daily from November to February over three consecutive years (2020-2022) and then tracked for 29.5 days. The red circle indicates location of desalination plant intake.

6. SUMMARY AND DISCUSSION

This study used the 1-km-resolution, three-dimensional Spencer Gulf Model to assess the far-field dispersal of desalination return water and regional larval connectivity with a 1 km assessment area surrounding the proposed Mullaquana Station intake. Comparison with long-term IMOS records and Northern Water project observations provided confidence that the model reproduces the principal sea-level, current, temperature and salinity dynamics relevant to regional far-field transport and dispersion. Larval connectivity estimates were derived by coupling the validated hydrodynamic model with species-specific particle-tracking assumptions. Initial diffuser dilution and entrainment at the intake occur at sub-kilometre scales and are assessed separately in the near-field and mid-field studies (see Water Technology, 2026a; Water Technology, 2026b).

The findings are interpreted within the established understanding of the physical oceanography and ecology of Spencer Gulf: seasonal hypersaline outflow and inverse-estuary structure (Nunes and Lennon, 1986; Nunes Vaz et al., 1990), salt-balance and flushing dynamics (Kämpf et al., 2009; Nunes Vaz, 2012; Middleton et al., 2013, 2014), biophysical modelling of fisheries connectivity (McLeay et al., 2015, 2016; Drew et al., 2020; Rogers et al., 2021; Rogers and Fowler, 2026), and documented importance of Upper Spencer Gulf as nursery grounds for multiple species and a region of unique and important ecological relevance (Shepherd et al., 2014).

6.1. Return water dispersion and Gulf-scale salt balance

The modelled response to the desalination return water reflects the interaction of energetic tidal mixing with slower wind- and density-driven circulation. Seasonally averaged anomalies of 0.05 – 0.1 PSU, equivalent to approximately 0.15–0.25% of ambient salinity, were largely confined to within approximately 5 km from the outfall. In spring, anomalies of 0.02 PSU (0.05% change in the ambient salinity) or less extended to Point Lowly. At monitoring locations more than several kilometres from the outfall, hourly anomalies were below 0.4 PSU (~1% change in the ambient salinity), while 30-day filtered and long-term mean anomalies were 0.05 PSU (0.15% change in the ambient salinity) or less. These predicted anomalies were substantially smaller than the observed natural daily-to-annual salinity variability of approximately 1-2 PSU (~2-5% changes in ambient salinity). The hourly anomalies reported at monitoring locations several kilometres from the outfall should not be interpreted as estimates of short-duration peak salinities immediately adjacent to the outfall or intake. These local conditions occur at sub-kilometre spatial scales that are not resolved by the far-field

model and are assessed separately using near-field and mid-field modelling (see Water Technology, 2026a; Water Technology, 2026b).

The spatial and temporal patterns of return water dispersion and transport are consistent with established understanding of Spencer Gulf. Previous modelling shows that enhanced diffusion is a particularly important characteristic of the flushing properties in Upper Spencer Gulf (Middleton et al., 2013, 2014), whereas density-driven exchange contributes to renewal over timescales of 6 to 12 months (Kämpf et al., 2009; Nunes Vaz, 2012; Kämpf, 2026). The 12-year simulations extend this understanding to the Mullaquana Station discharge scenario. They indicate that return water is rapidly diluted by tidal mixing and subsequently transported by wind and density-driven circulation. Under the simulated 73 GL y^{-1} higher-capacity scenario and 12-year hindcast conditions, the model predicted no progressive accumulation of salt within Spencer Gulf.

At distances greater than approximately 5 km from the outfall, predicted far-field salinity increases were small relative to observed daily-to-annual salinity variability of 1-2 PSU observed across Spencer. They were also below the environmental and ecological tolerance limits of 1 PSU for flora and fauna commonly reported in the desalination literature (Lattemann and Höpner, 2008; Panagopoulos and Haralambous, 2020; Tanner and Drabsch, 2021; Omerspahic et al., 2022). This benchmark should be interpreted as a screening criterion rather than a universal ecological tolerance threshold. Global evidence from operating desalination plants shows that whilst the main salinity footprint is often concentrated within tens to hundreds of metres of the discharge, consistent with the results of the midfield modelling studies for Northern Water (Water Technology, 2026b), biological sensitivity to salinity elevation varies among species, life stages, habitats, discharge chemistry, exposure durations and the ecology of the receiving environment (Sisma-Ventura et al., 2022; Sola et al., 2024; Sirota et al., 2024; Grammatiki et al., 2025).

6.2. Larval connectivity with the outfall assessment area

The coupled hydrodynamic-particle-tracking simulations indicated clear interspecific differences in the estimated connectivity with the 1-km intake assessment area. Across the simulated years, connectivity was less than 0.003% for King George Whiting, 0.5-2.7% for Western King Prawn and 6.1-7.4% for Snapper. These values describe modelled connectivity under species-specific particle-tracking assumptions; they are not directly comparable estimates of entrainment, mortality or population-level loss. These species contrasts are biologically plausible because larval pathways emerge from complex interactions between

hydrodynamic circulation, assumed spawning habitats, pelagic larval durations, larval behaviours, mortality, and settlement parameterisations.

For Western King Prawn and King George Whiting, coastal nursery settlement areas lay outside the 1-km intake assessment area. Connectivity was therefore calculated using all released particles, rather than only particles predicted to settle. This avoided additional uncertainty associated with settlement rates and the spatial and temporal variability of habitat suitability, mortality and egg production (McLeay et al., 2016; Rogers et al., 2021). Including particles that may not ultimately settle provides a deliberately broad estimate of potential intersection with the intake assessment area. Accordingly, these results should be interpreted as conservative estimates of regional connectivity, rather than estimates of connectivity among successfully recruited individuals.

A different approach was required for Snapper because the proposed intake is located within its broader nursery area, and hypothetical release sites covered the full model domain to represent uncertainty in spawning locations (Rogers and Fowler, 2026). For Snapper, connectivity was calculated from particles predicted to settle successfully. This restricted the analysis to particles originating from locations capable of contributing to settlement, thereby avoiding an artificially low connectivity estimate arising from the inclusion of source areas that could not contribute to recruitment. To represent potential exposure before recruitment, particles were required to settle after, rather than before, entering the 1-km intake assessment area

Because the species-specific analyses estimate different conditional probabilities, the resulting connectivity percentages should not be compared directly as equivalent measures of exposure, entrainment risk or population-level risk. Furthermore, the sensitivity of connectivity to behavioural parameterisations has been clearly demonstrated by the northward transport predicted for WKP after the activation of tidal-stream transport (McLeay et al., 2015; 2016). Comparable biophysical studies have similarly demonstrated that vertical behaviour and life-history assumptions can alter predicted dispersal and settlement patterns (North et al., 2008; McLeay et al., 2016; Rogers et al., 2021).

6.3. Conclusion

Validation against multiple observational datasets provides confidence that the Spencer Gulf model reproduces the principal physical processes relevant to far-field transport and dispersion of desalination return. Under the simulated 73 GL y^{-1} higher-capacity scenario,

which is conservative relative to the 51 GL y^{-1} reference design capacity, the model predicted small far-field salinity anomalies and no progressive salt accumulation over the 12-year hindcast period. At distances beyond the immediate outfall area, predicted anomalies were small relative to observed daily-to-annual salinity variability in Spencer Gulf. They were also below the 1 PSU incremental salinity benchmark commonly applied in desalination assessments, noting that ecological sensitivity depends on species, life stage, habitat and exposure conditions. Conditions at sub-kilometre scales, including initial diffuser dilution and local salinity peaks, were outside the resolution of this far-field model and are assessed in separate near-field and mid-field studies (see Water Technology, 2026a; Water Technology, 2026b)

The larval simulations provided modelled estimates of regional connectivity with the 1-km intake assessment area for Western King Prawn, King George Whiting and Snapper. These estimates are not direct measures of entrainment, larval mortality, recruitment loss or population-level effect. They are sensitive to assumptions about spawning distributions, larval duration and behaviour, mortality, settlement and habitat suitability. The analytical approaches also differed among species; consequently, the reported percentages should not be compared as equivalent estimates of population-level risk. Intake entrainment is addressed separately through complementary mid-field modelling (Water Technology, 2026b).

Upper Spencer Gulf supports ecologically, commercially and recreationally important habitats and species (Shepard et al., 2014). The present study establishes baseline, scenario-specific predictions for far-field return-water dispersion and larval connectivity. It does not assess the combined effects of additional or cumulative environmental pressures e.g., (Stockbridge et al., 2025), including harmful algal blooms (Murray et al., 2026), or climate-driven change in Gulf oceanography (Bastos de Oliveira and Doubell, 2026). Continued integrated environmental monitoring and periodic model updating would therefore be valuable to test predictions, characterise changing baseline conditions and support adaptive environmental management.

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APPENDIX A: MODEL SALINITY CORRECTIONS

The northern part of Upper Spencer Gulf is a hyper-saline region with salinities regularly exceeding 45 PSU. This region is very narrow and shallow, which creates a challenging environment to simulate with a hydrodynamic model. Atmospheric forcing for the whole of Spencer Gulf is from the NCEP Climate Forecast System Reanalysis v.2. Surface winds and surface heat fluxes are provided at 6 hourly intervals on a 0.2-degree resolution grid, which equates to approximately 22 km in the north-south direction and between 16 and 20 km in the east-west direction. While this is sufficient resolution to capture the surface forcing in most of the Gulf, at 34°S the Gulf narrows to about 60 km and at 33°S. At approximately the latitude of Point Lowly, the Gulf is only around 28 km across. Efforts to model the salinity of Upper Spencer Gulf usually underestimate the observed values by several PSU and it is speculated that poor resolution of the surface fluxes and the complex sea breezes in this region contribute to underestimating the surface heating and evaporation rates.

Early efforts to improve the salinities by using humidity and air temperature from Bureau of Meteorology Automated Weather Stations (AWSs) situated around the Gulf helped but still failed to reproduce observed salinities approaching 50 PSU at Port Augusta. Reducing the humidity further to enhance evaporation helped but led to unexpected errors in the surface temperature due to latent heat loss.

Another approach often used to improve model performance is to relax surface tracers to known values. This approach was implemented to relax sea surface temperature to satellite observations and provided excellent validation against observations at the mooring sites (Figure 5 to Figure 12). One common climatology is the CSIRO Atlas of Regional Seas (CARS), which is based on historic profiles of temperature and salinity, however it proved to be unusable because the CARS salinities were still too low at the top of the Gulf and the model was already providing better salinities in the southern parts of the Gulf than CARS. One of the few sets of observations that reflect the hyper-saline environment of the Upper Spencer Gulf are the observations documented in Nunes and Lennon (1986). Figure 27 summarizes the T-S relationships at four sites in Upper Spencer Gulf and one site on the shelf south of the Gulf.

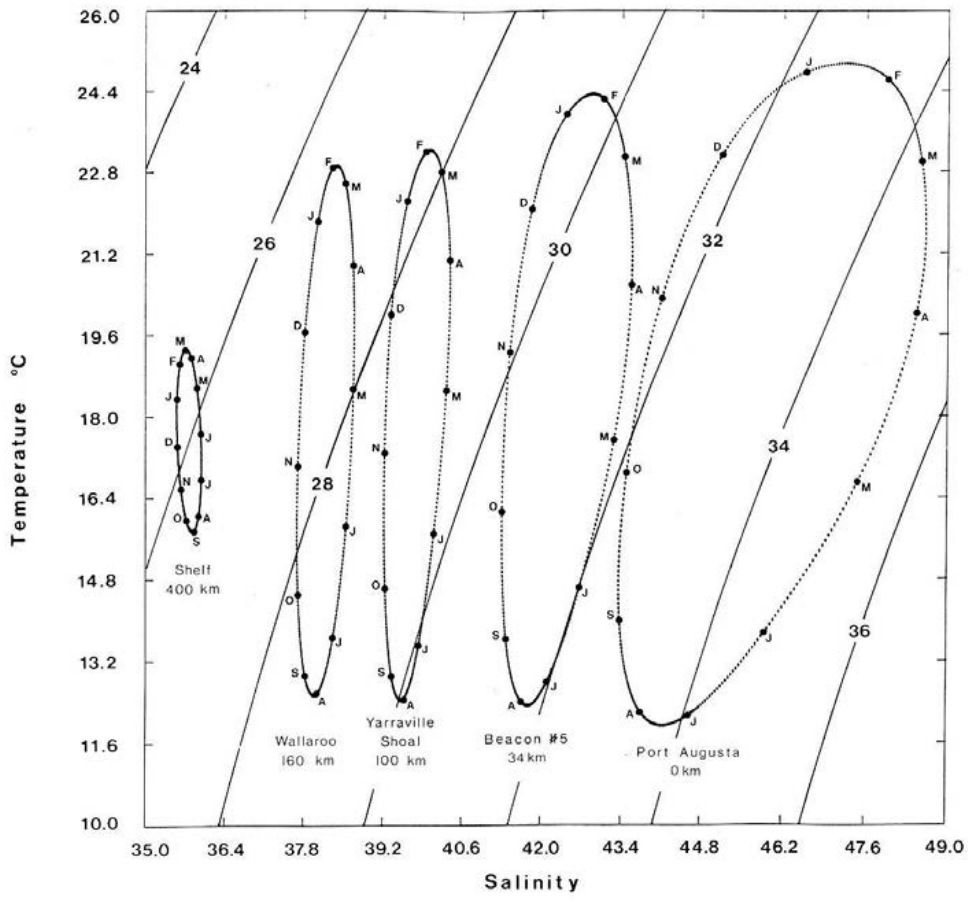


Figure 27. Climatological estimates of temperature and salinity from Nunes and Lennon (1986, Fig. 11) at the top of Spencer Gulf taken from 30 months of observations between 1983 and 1985.

Since the Nunes and Lennon observations only apply at four sites along the eastern side of the upper Gulf it was important to only apply the corrections where they were needed and not to degrade model performance in regions where the model was in good agreement with observations. The lowest salinities of the four sites occurred at Wallaroo where the minimum salinity was approximately 38 PSU. Looking at an annual climatology formed from the previous model run, the cutoff for this isohaline was found north of 34°S and ran approximately along 137.5°E (Figure 28). The fresher water on the western side of the Gulf at these latitudes reflects the general thermohaline circulation of the Gulf, which is especially strong during the winter outflow when an influx of shelf water from the south is advected northward along the coast. The salinities from the Nunes and Lennon observations were used to generate a monthly varying climatology within this north-east box and used to correct the salinities in this hypersaline region.

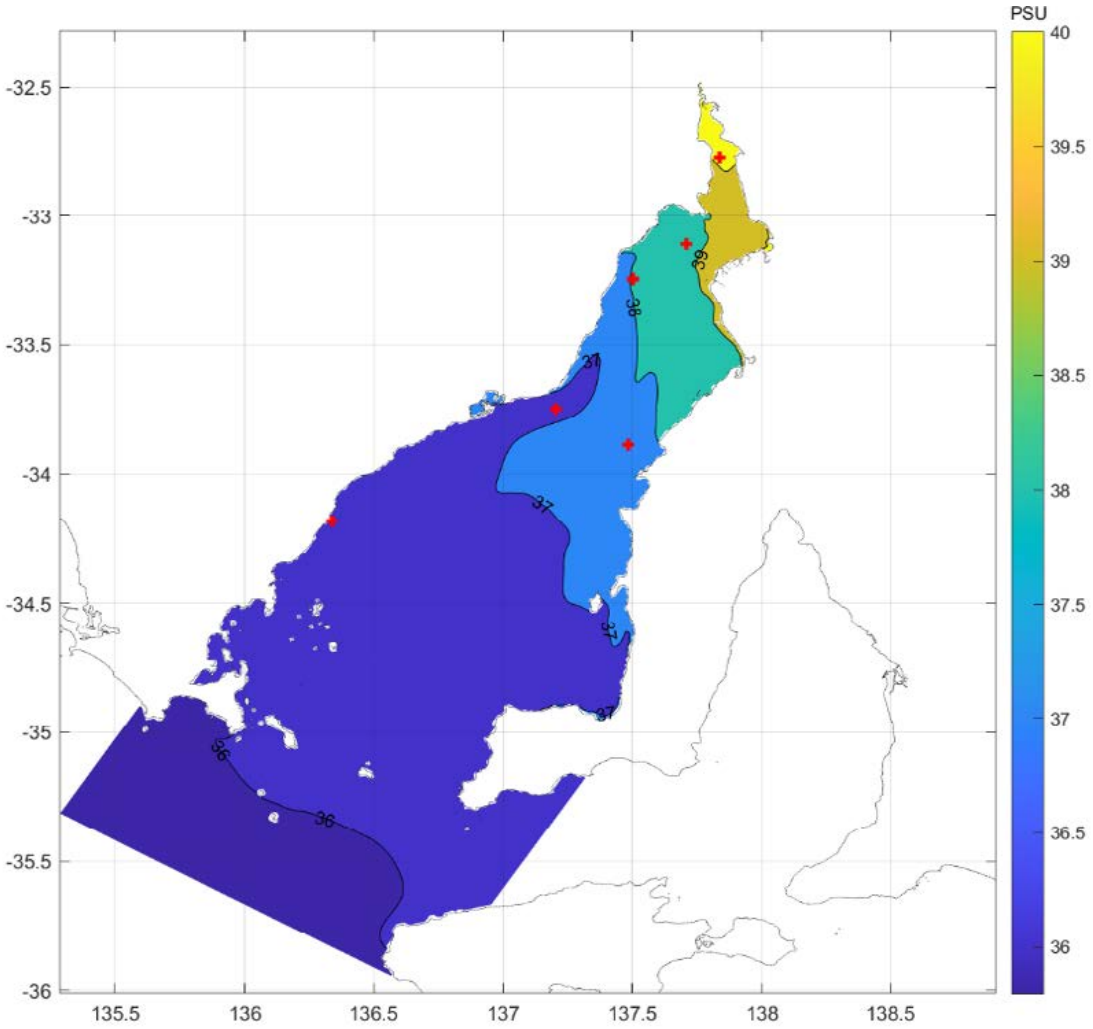


Figure 28. Annual average sea-surface salinity (SSS) from the 12-year preliminary model run indicating the 38 PSU contour and the location of the moorings used for validation (red crosses).

The improvements in model performance from the application of this Nunes-Lennon correction are shown in Figure 29 for the main mooring sites and the reduction in salinity bias is significant for the four moorings within the northeastern region. Notably, there is little change at the two sites along the western side of the Gulf where the tongue of fresher water from the shelf inflow is found.

The results of the Nunes-Lennon (N-L) correction suggest that a study from nearly 40 years ago is still relevant to Gulf processes today. A reasonable question to ask is “how has climate change impacted these T-S relationships over this time scale?”. Unfortunately, there are no long-term salinity observations in northern Spencer Gulf that we can use for comparison, but we can use the SST measurements from the MUR SST analyses over the last 12 years to see how temperatures have changed and compare this with the seasonal temperature variations used in the N-L corrections. The results of this comparison are shown in Figure 30 and

demonstrate that the temperature variations have remained fairly stable, in fact the summer temperatures at the top of the Gulf have dropped slightly over the modelling period. This may be due to enhanced upwelling bringing cooler waters on to the shelf which are then entrained into the Gulf, but further investigation is beyond the scope of this study.

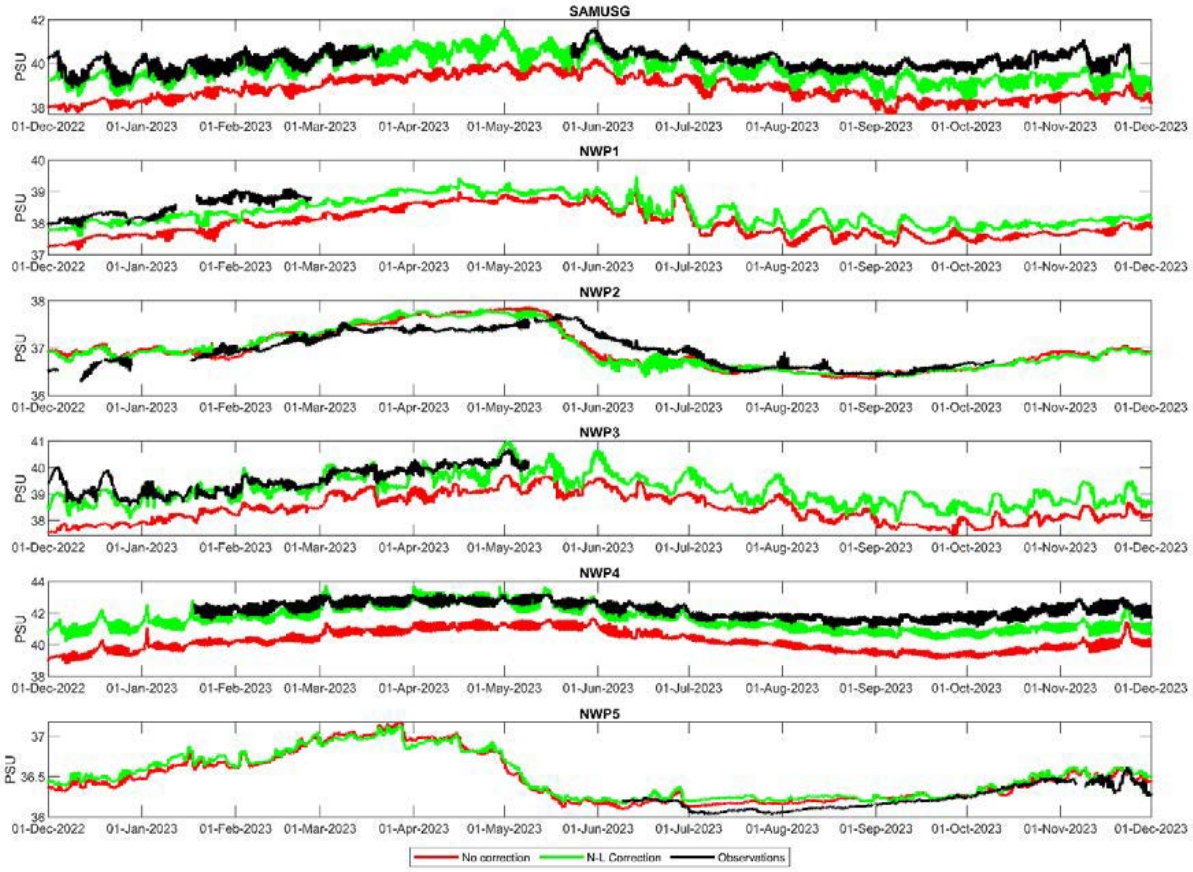


Figure 29. Validation improvement for model runs with and without the Nunes-Lennon (N-L) correction at the mooring sites in Upper Spencer Gulf.

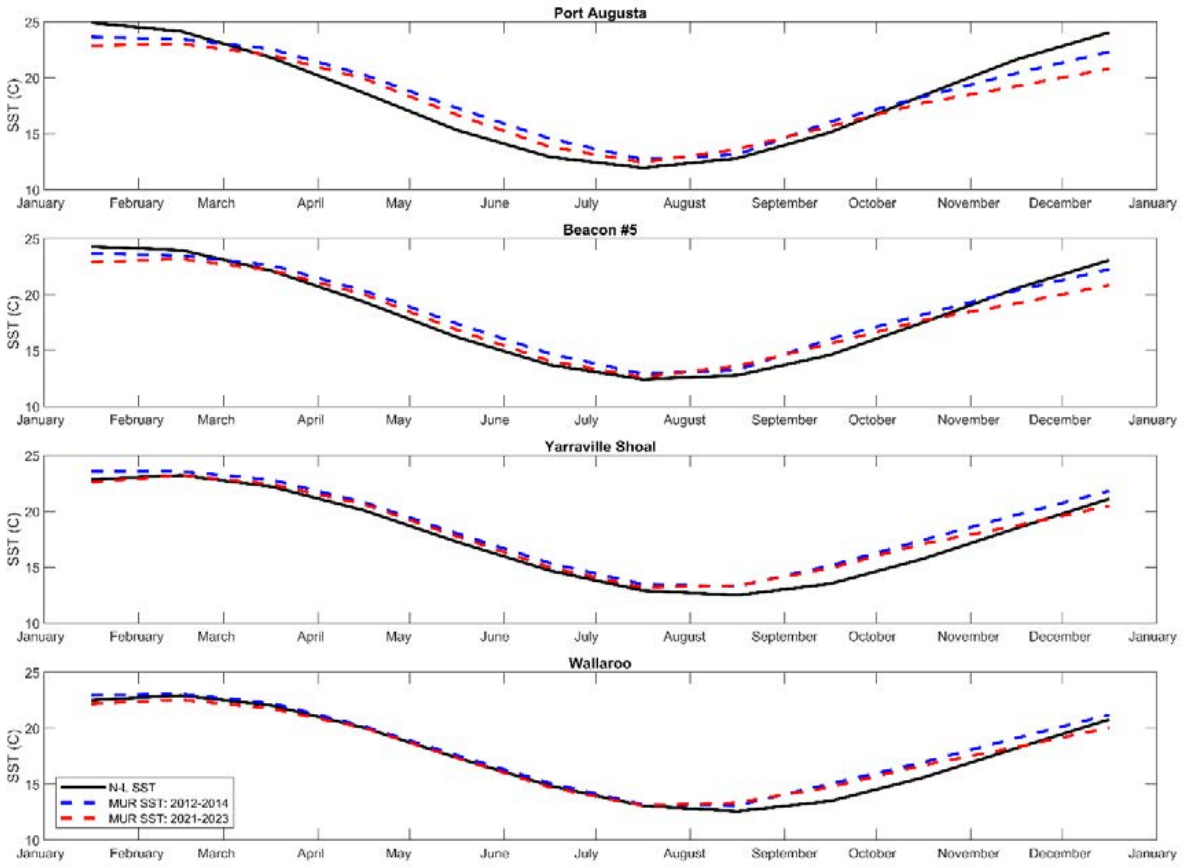
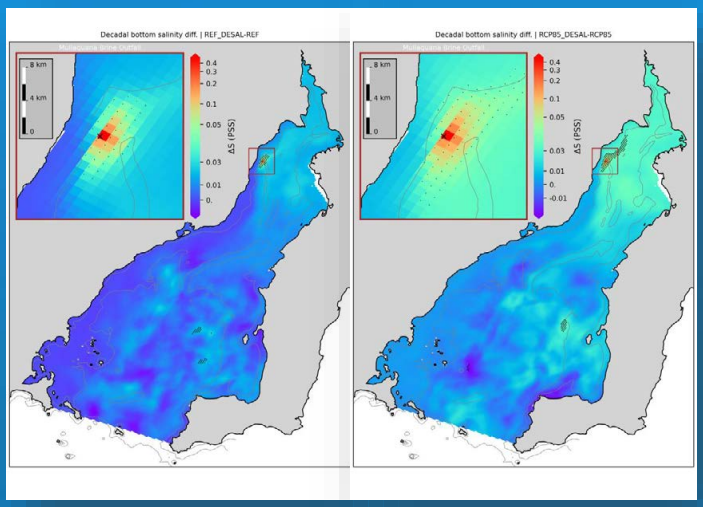


Figure 30. Nunes-Lennon (N-L) temperature climatology at the four N-L sites and from the MUR SST measurements for the first and last 6 years simulated in this model study.

Appendix G-14:
Oceanographic Farfield Modelling of Return Water Discharge Under a Future Climate Scenario

Marine Ecosystems

Northern Water: Oceanographic far-field modelling of return water discharge under a future climate scenario



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Northern Water: Oceanographic far-field modelling of return water discharges under a future climate scenario

Hugo Bastos de Oliveira and Mark Doubell

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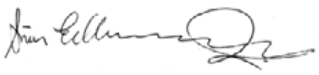
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EXECUTIVE SUMMARY

The South Australian Government's Northern Water project proposes a reverse-osmosis seawater desalination plant at Mullaquana Station in Upper Spencer Gulf. The reference design capacity for the proposed development is 51 GL y^{-1} (140 ML d^{-1}). Given the ecological, fisheries and socio-economic importance of Spencer Gulf and the potential operating life of the plant, environmental assessment of the project requires consideration of the far-field fate of hypersaline return water under present-day and projected future climate conditions.

This report applies projected changes from the high-emissions Representative Concentration Pathway 8.5 (RCP8.5) scenario to a validated 1 km resolution model of Spencer Gulf. Seasonal climate-change perturbations were derived from differences between a future 30-year period (2066-2095, centred on 2080) and a reference 30-year period (2006-2035, centred on 2021), then applied to the regional model. The modelling uses a hypothetical higher-capacity scenario of 73 GL y^{-1}, above the proposed 51 GL y^{-1} reference design capacity for which approval is being sought. The modelling results are best interpreted as a plausible upper bound of the potential impacts of desalination and climate change together. This assessment covers the far-field dispersion processes only; near-field changes are not assessed. Four simulations compare present-day and future-climate conditions, each with and without desalination return-water discharge.

Under the modelled climate-change scenario alone, mean water temperatures rise by 1-2 °C across Spencer Gulf. Mean bottom salinity increases by approximately 0.3 PSU in the southern gulf (~0.8%) above present ambient conditions and by more than 0.8 PSU (~2%) in the Upper Spencer Gulf. The Mullaquana Station outfall lies within the region of the largest projected increases, where bottom salinity is predicted to increase by 0.4-0.6 PSU (~1.5%) across seasons (highest in summer) and bottom temperature by 1.4-2.2 °C (highest in winter).

Under the future climate scenario, the modelled salinity anomaly attributable to return-water discharge is less than 0.2 PSU (~0.5% increase) over the future ambient conditions within 2-8 km from the outfall, and below 0.03 PSU (~0.1% increase) at greater distances from the outfall. The plume dispersion is broadly similar to that modelled under present-day conditions, although it extends slightly farther to the north-east.

Near the outfall, the modelled salinity increases attributable to climate change and return-water discharge are additive, with climate change contributing the larger part under the scenario examined. Their simulated combined salinity increase remains below 3% of present-day ambient salinity levels at the site. Far-field increases attributable to return water remain below 1% of the ambient salinity. Under a lower-emissions scenario, return water could

potentially contribute a larger proportion of the combined change, although this was not modelled.

The Gulf's clockwise, inverse-estuarine circulation persists in all simulations and strengthens slightly under the future-climate scenario. Dense water continues to exit along the eastern side of the Gulf, with a compensating inflow of shelf water along the western side. Overall, the modelling predicts a small far-field salinity impact from the proposed return water discharge under present-day and the future RCP8.5 climate scenario. To our knowledge, this is the first application of dynamical ocean downscaling used to inform an Australian desalination proposal.

Keywords: Far-field impacts, desalination, Spencer Gulf, climate downscaling, return water, salinity, RCP8.5 climate scenario, Mullaquana Station, regional circulation, hydrodynamics.

1. INTRODUCTION

1.1. Background

The South Australian Government through the Northern Water project is investigating construction and operation of a reverse osmosis seawater desalination plant located at Mullaquana Station in the Upper Spencer Gulf, approximately 25 km south of Whyalla (Figure 1-1). The plant's reference design capacity is 51 GL y^{-1}. The project aims to provide a secure, climate resilient water source for industry in South Australia's Upper Spencer Gulf and Far North. This study models a hypothetical higher-capacity desalination scenario of 73 GL y^{-1} as a conservative case for assessing potential far-field oceanographic effects under current and projected future climate conditions.

Spencer Gulf has an inverse-estuarine circulation (Nunes Vaz et al., 1990; Middleton and Bye, 2007; Kämpf, 2026) with reduced flushing compared to higher-energy open ocean coastlines (Kämpf et al., 2010; Middleton et al., 2013; 2014). The gulf supports marine ecosystems of ecological, commercial and recreational importance (Gillanders et al., 2015; Tanner et al., 2019; 2020). These conditions make it necessary to assess the gulf's capacity to receive return water from the proposed desalination plant under present day and possible future climate conditions.

Seawater desalination produces a hypersaline return-water stream (brine) (Elsaid et al., 2020) that may affect the marine environment if it is not adequately diluted and dispersed (Gacia et al., 2007; Lattemann and Höpner, 2008; Roberts et al., 2010; Alameddine and El-Fadel, 2007; Frank et al., 2019). The primary environmental risks associated with desalination return-water relate to its thermal and hypersaline characteristics, alongside the residual treatment chemicals it carries (Omerspahic et al., 2022). Within the Spencer Gulf, the entrainment of planktonic organisms at the intake pipe represents an additional ecological risk (e.g., see Doubell and James, 2023; Doubell and James, 2026).

To understand the potential effects of desalination return water its dispersion should be assessed across a range of spatial and temporal scales. Dispersion modelling begins at the near-field, or mixing zone, where plume behaviour is governed by the physical properties of the return water and the diffuser design. The discharge momentum and excess density of the return water strongly influence initial plume behaviour and dilution, while ambient circulation provides an additional influence (Doneker and Jirka, 2001). The near field is also of regulatory interest because near-field studies inform part of the plant approval and licensing processes, including assessment against applicable dilution thresholds defined within its boundary (ANZECC/ARMCANZ, 2000; Kämpf and Clarke, 2013). Such studies

assess whether the return water plume is well mixed, surface trapped, or bottom attached (e.g., Alameddine and El-Fadel, 2007).

At larger spatial scales, typically beyond the immediate mixing zone, transport and dispersion of the return water are increasingly governed by regional oceanographic processes. Far-field modelling studies are used to assess this transport and dispersion, and potential for longer-term or gulf scale changes in salinity across multi-kilometre scales. Modelling the fate of the return water plume across representative oceanographic and atmospheric conditions helps assess the broader environmental implications of desalination proposals and identifying potential ecological impacts (e.g., Kämpf et al., 2009; Roberts et al., 2010; Ibrahim et al., 2020; Doubell and James, 2023).

While modern desalination plants can operate for more than 30 years (Sommariva et al., 2001), few far-field studies evaluate potential effects over comparable multi-decadal periods or against the changing environmental baselines projected under climate change. Recent work indicates that assessing desalination under future climate scenarios can reduce uncertain ty regarding how its environmental impacts might evolve in a changing environment (Paparella et al. 2022). However, global climate models are too coarse to resolve the bathymetry and oceanographic processes of Spencer Gulf. Consequently, assessing desalination under projected future climate conditions requires climate information to be downscaled to a higher-resolution regional hydrodynamic model using simulations of sufficient duration to capture natural variability.

This report builds on current period far-field simulations of desalination return water in Spencer Gulf (Doubell and James, 2026) and evaluates the same return water configuration at Mullaquana Station (Figure 1-1, inset) under projected climate conditions. Although the reference design capacity is 51 GL y^{-1}, this study adopted a hypothetical higher-capacity scenario of 73 GL y^{-1} to provide a conservative assessment of potential far-field salinity effects. The future climate scenario follows the delta downscaling approach of Shin and Alexander (2020), which is equivalent to the pseudo-global warming method of Schär et al. (1996). In this approach, projected climate changes are imposed on present-day forcing while retaining present-day tidal and synoptic variability.

To represent a high-emission future scenario, the climate-change signal was derived from the Representative Concentration Pathway (RCP) 8.5 scenario of the Geophysical Fluid Dynamics Laboratory Earth System Model (GFDL-ESM2M; Dunne et al., 2012, 2013) for scenarios with and without desalination return water in Upper Spencer Gulf (USG). The combination of a high-emission climate scenario and a return water discharge with an above-

reference capacity provides a conservative scenario for assessing potential combined far-field salinity changes in Spencer Gulf.

Given the potential plant operational life of several decades, the ambient conditions in which the return water is discharged may differ appreciably from the present-day baseline considered by Doubell and James (2026). This report therefore examines how the spatial extent, concentration, and persistence of desalination return water may differ under a warmed climate, and whether conclusions drawn from present-day conditions remain applicable under the simulated future scenario.

The report is structured as follows. Section 1.2 sets out the study objectives. Section 2 describes the model configuration, climate-downscaling approach and experimental scenarios. Section 3 presents projected regional climate changes and downscaled results for Spencer Gulf. Section 4 discusses the modelling findings and their broader implications.

1.2. Objectives

The objectives of this report were to:

1. Downscale a global high-emissions RCP8.5 climate projection scenario to Spencer Gulf using the validated high-resolution (1 km) SARDI regional ocean model developed (Doubell and James, 2026) for the Northern Water project.
2. Assess mean oceanographic conditions in Spencer Gulf, with and without a hypersaline return-water from an idealised, higher-capacity case (73 GL yr^{-1}) desalination scenario at Mullaquana Station.
3. Evaluate whether the return water dispersion, outfall conditions and the plume characteristics under the future climate scenario differ from those simulated under present day conditions.
4. Separate the effects of return water and climate change at the outfall and assess whether their combined salinity effect is additive.

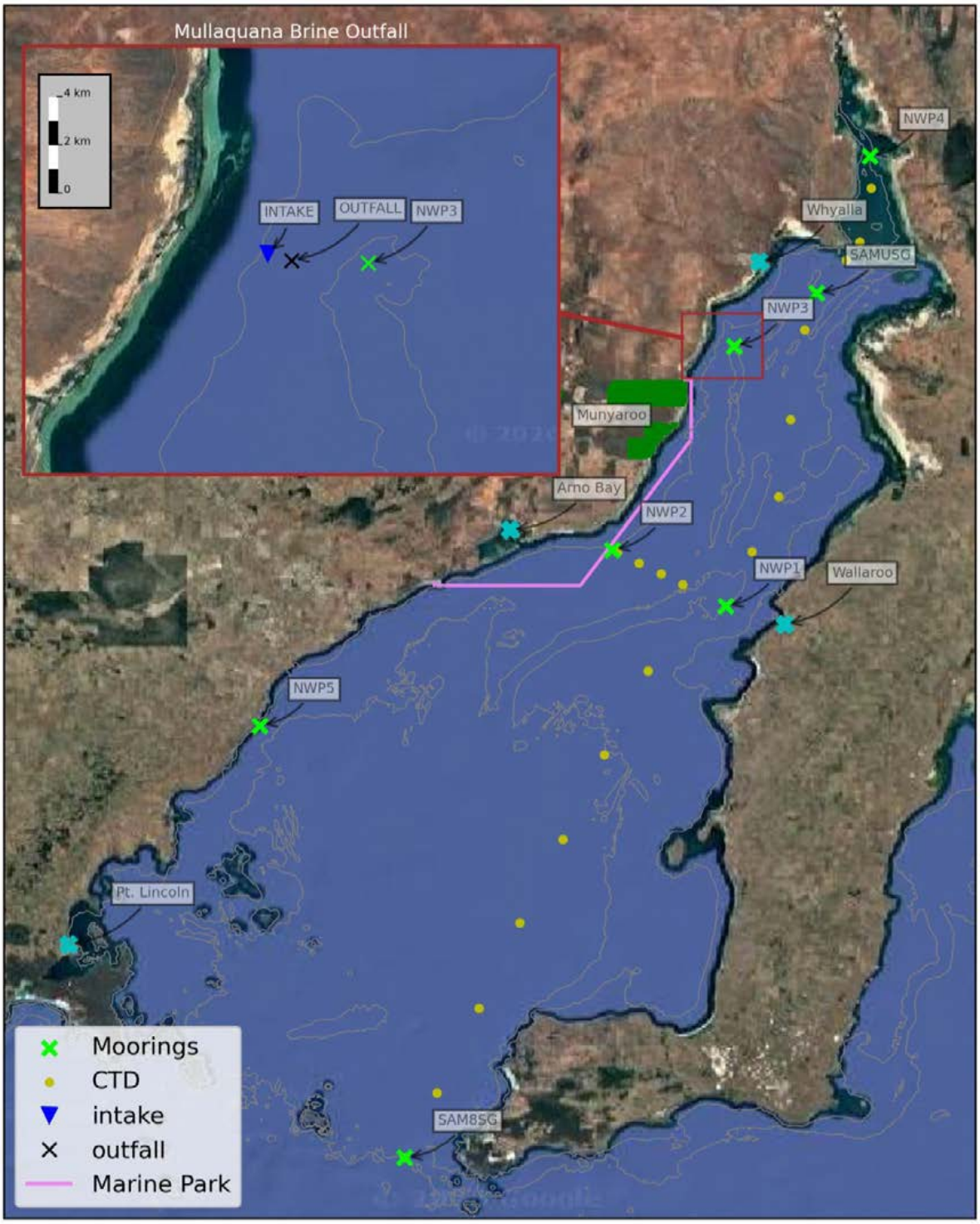


Figure 1-1. Map of Spencer Gulf showing the Mullaquana Station outfall region (inset), with the proposed intake (blue triangle) and the outfall site (black cross) positions. CTD (yellow dots) and moorings observation (green crosses) locations used to validate the base model are shown. The light grey lines represent the 10, 25, and 50m isobaths derived from the Geoscience Australia (2023) AusBathyTopo 250m grid. The magenta polygon represents the Franklin Harbour Marine Park and green polygons the Munyaroo Conservation Park.

2. METHODS

2.1. The Global Climate Model

The Representative Concentration Pathway 8.5 (RCP8.5) is one of several scenarios in the IPCC (Intergovernmental Panel on Climate Change) framework that projects future greenhouse gas emissions and their impact on Earth. The RCP8.5 scenario represents a high-emission pathway in which global temperatures and greenhouse gases continue to rise significantly throughout the 21st century without significant emissions mitigations. Under this scenario, atmospheric carbon dioxide levels are projected to more than double current concentrations reaching 940 ppm by the end of the century (van Vuuren et al., 2011), with an annual mean atmospheric forcing warming target of 8.5 Wm^{-2} above pre-industrial levels. The use of the RCP8.5 scenario is this report is aligned with the assessment requirements for Environmental Impact Statement for the Northern Water project (DIT, 2026).

The global climate model chosen for downscaling future projections of the oceanographic conditions in Spencer Gulf under RCP8.5 is the GFDL-ESM2M model (Dunne et al., 2012; 2013; 2020) and is one of several from the Coupled Model Intercomparison Project Phase 5 (CMIP5). The choice of GFDL-ESM2M over its CMIP6 successor, GFDL-ESM4.1, was a pragmatic decision at the project design stage. This choice was based on the model's maturity and well-established validated use in regional downscaling's (Shin and Alexander, 2020; Pilcher et al., 2022). Furthermore, the GFDL-ESM2M has been evaluated for its and it simulated climate response and feedback mechanisms (Dunne et al., 2012; 2020) and representation of atmospheric and oceanic states and variability (e.g., Grose et al., 2020). CMIP5 has been instrumental to IPCC assessments, and GFDL-ESM2M is one of a set of eight models recommended for downscaling studies in Australian scenarios (CSIRO and Bureau of Meteorology, 2015). Hence, downscaling from a single, well evaluated model that is representative of the CMIP5 ensemble was kept as the core technical strategy.

The climate model has been used to drive recent climate downscaling projections in the ocean, particularly due it's coupled atmospheric, ocean, land and ice components, which resolve feedback processes that govern climate dynamics (Figure 2-1). The model's atmospheric component has a resolution of 2° latitude × 2.5° longitude and 24 vertical levels, while the ocean component has a nominal 1° resolution and 50 vertical levels. Readers are directed to Dunne et al. (2012) for a complete description of the model and to Lyu et al. (2020) and Grose et al. (2020) for recent model evaluations of this and other climate models. The baseline reference simulations and future projections outputs of GFDL-ESM2M are openly accessible and were retrieved obtained from Australia's National Computing Infrastructure

(NCI). Several global sea level components are omitted by design from the CMIP5 scenarios. Consequently, total sea level rise is not included and falls outside the focus of this report (e.g., see Zhang et al., 2017 for details). Table 2-1 provides a list of the monthly atmospheric and oceanic variables used in this report.

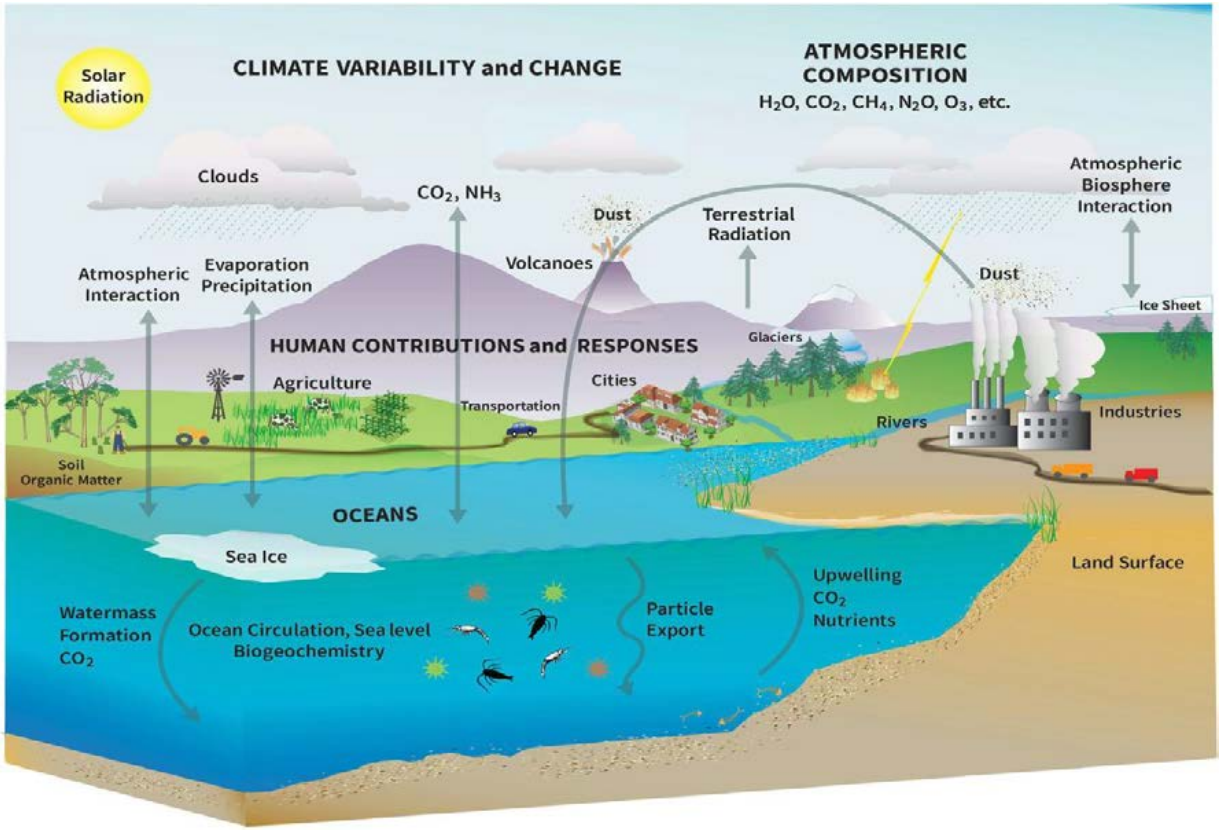


Figure 2-1. A schematic illustration of natural and human-induced factors influencing climate, as represented in Earth System Model components from GFDL-ESM2M (image reproduced from Dunne et al., 2020).

Table 2-1. GFDL-ESM2M extracted monthly atmospheric (Amon) and oceanographic (Omon) variables used for downscaling in Spencer Gulf.

Variable	CMOR Table	Long Name
evspsbl	Amon	Water Evaporation Rate
hfss	Amon	Surface Upward Sensible Heat Flux
hfls	Amon	Surface Upward Latent Heat Flux
hurs	Amon	Near-Surface Relative Humidity
huss	Amon	Near-Surface Specific Humidity
pr	Amon	Precipitation
psl	Amon	Sea Level Pressure
rlds	Amon	Surface Downwelling Longwave Radiation
rlus	Amon	Surface Upwelling Longwave Radiation
rsds	Amon	Surface Downwelling Shortwave Radiation
rsus	Amon	Surface Upwelling Shortwave Radiation
sfcWind	Amon	Near-Surface Wind Speed
tas	Amon	Near-Surface Air Temperature
tauu	Amon	Eastward Surface Stress
tauv	Amon	Northward Surface Stress
uas	Amon	Eastward Near-Surface Wind
vas	Amon	Northward Near-Surface Wind
evs	Omon	Water Evaporation Rate (Ocean)
mlotst	Omon	Ocean Mixed Layer Depth
sos	Omon	Sea Surface Salinity
zos	Omon	Sea Surface Height Above Geoid
tos	Omon	Sea Surface Temperature
so	Omon	Sea Water Salinity
thetao	Omon	Sea Water Potential Temperature
uo	Omon	Sea water X velocity
vo	Omon	Sea water Y velocity

2.2. Derivation of climate deltas

Using a climate projection to derive properties in shallow seas such as Spencer Gulf is challenging due to the resolution limitations of global climate models. Coastal processes are typically poorly resolved or omitted entirely, including; bathymetric steering, tidal mixing, coastal buoyancy-driven exchange and near-shore wind-driven transport. For example, the Lincoln Shelf from Ceduna to Kangaroo Island, hosts at least three distinct evaporative water masses generated by nearshore processes (Richardson et al., 2018). Furthermore, within Spencer Gulf, the local hydrography reflects a balance between evaporation, tidal dissipation and turbulent transport, with wind-driven and density driven important for shelf exchange (Middleton and Bye, 2007). Because current climate models are unable to resolve these local processes, downscaling using a high-resolution regional model is required. Importantly, the

global climate model resolution limitations do not undermine their capability to project broad future changes. Global models remain the best source for capturing large-scale signals, such as ocean warming, changes in surface fluxes and shifts in regional winds. The main objective of the downscaling approach implemented here is to systematically extract that large-scale climate signal and project it onto a regional model that accurately resolves the local, regional dynamics of Spencer Gulf.

A transient downscaling, spanning the recent past to the end of the century, remains computationally prohibitive at the resolution required for this study. To overcome this constraint, the climate delta method (Shin and Alexander, 2020), or pseudo-global warming method (Schär et al., 1996; Hall et al., 2024; Brogli et al., 2023) was chosen. This approach reduces computational cost while allowing the downscaled simulation to retain a realistic sequence of synoptic and tidal variability.

The climate deltas represent the long-term mean change between recent (or past) and future conditions in the climate projection. In other words, they represent a climatological mean annual cycle change. Besides its simplicity, the delta method offers a further advantage over forcing a regional model directly with a climate projection output. Because the delta is computed as the difference between two time periods of the same climate simulation, any systematic bias that remains constant between the periods cancels out, while the delta retains the climate change signal, including any further change in the bias itself. Consequently, a mean state that is too warm or too fresh does not propagate into the regional hydrodynamic solution, the model is forced instead with present-day conditions plus the projected seasonal change (see also Hall et al. 2024; van Hooidonk et al. 2015).

Following Shin and Alexander (2020), the climate deltas were computed as the difference between 30-year mean monthly conditions over a reference baseline period and a future projection period. A 30-year climate normal period was used to reduce the influence of low-frequency internal variability on the delta estimates, such as Pacific Decadal Oscillation and El-Niño-Southern Oscillation. The baseline reference period was defined as 2006-2035, while the future climate period was selected for the period 2066-2095, with both taken from the canonical RCP8.5 scenario of GFDL-ESM2M. Deriving both intervals from the same scenario branch avoids discontinuity in forcing that occurs between the historical (pre-2005) and future (post-2006) scenarios within the CMIP5 ensemble. The baseline reference period aligns with the present-day interval simulated by Doubell and James (2026), ensuring a consistent baseline foundation. The late-century future projection interval was selected in preference to a mid-century alternative (e.g. 2036–2065) because the climate change signal under the RCP8.5 scenario continues to intensify throughout the century (not shown). Furthermore, the

selected late century interval makes the climate response easier to distinguish from internal variability, ensuring that the downscaled climate change signal is large relative to model noise. This is important for the delta method application; with a substantial mean climate change imposed, any error incurred by omitting changes in future interannual variability can become negligible by comparison. Although the 8.5 W m^{-2} warming pathway has not been carried forward into the next CMIP7 scenario framework (Van Vuuren et al., 2026), the RCP8.5 scenario via GFDL-ESM2M remains informative for this study and aligns directly with regulatory impact assessment requirements. The resulting simulations presented here could then be interpreted as a conservative, upper-bound scenario rather than a predictive, most-likely projection.

To implement the downscaling experiments, the regional oceanographic model is perturbed using conservative interpolation from the global climate model components. Seven atmospheric variables were perturbed: air pressure, relative humidity, air temperature, surface winds, longwave and shortwave radiation, and precipitation. The perturbed ocean variables are temperature, salinity, sea surface height, and vertically integrated velocities.

2.3. Regional Downscaling and Desalination Scenarios

The future oceanographic conditions in Spencer Gulf and the gulf's response to the proposed Mullaquana Station desalination plant are simulated using the Regional Ocean Modelling System (ROMS) version 3.9 (Shchepetkin and McWilliams, 2005). Four experiments form a two-by-two design: present-day conditions (REF, REF_DESAL) and downscaled high-emissions future climate (RCP85, RCP85_DESAL), each with and without desalination return water. Comparing the future climate scenarios against present-day references isolates the forced climate change signal from local environmental return water effects in a framework analogous to a Before-After, Control-Impact (BACI) design.

For the downscaling, the REF and REF_DESAL simulations were adapted from the validated Spencer Gulf Model (SGM1000x), described in Doubell and James (2026). The domain covers part of the Lincoln Shelf and Spencer Gulf at 1km horizontal resolution (Figure 2-2A) and simulations were run for the period 2011 to 2024. Bathymetry was provided by Geoscience Australia (2023) and was filtered for numerical stability (Figure 2-2B). Surface atmospheric forcing was sourced sub-daily from the NCEP Climate Forecast System Reanalysis v.2 (CFSRv2, Saha et al., 2014), with relative humidity adjusted. Lateral oceanic boundary conditions and initial fields were taken from CSIRO's BlueLink Reanalysis version 2023 (BRAN2023) (Oke et al., 2013, Chamberlain et al., 2021) at 10km resolution, with tides imposed from the global TPXO8 model (Egbert and Erofeeva, 2002). The RCP85 downscaling experiments were produced by running the SGM1000x configuration over a present-day

period reference period (2011-2024), applying monthly atmospheric and oceanic climate deltas to boundary and surface forcing. Because the control and perturbed simulations span an identical interval, the REF and RCP85 runs share the same sequence of synoptic and interannual variability and differ only in the imposed mean annual cycle. All sub-daily CFSRv2 surface forcing variables were perturbed by their respective monthly deltas, including mean sea level pressure. At the lateral boundaries the projections perturbed sea surface height (SSH), temperature, salinity, and vertically integrated momentum flux from BRAN2023. Spatial extrapolation was applied along the model boundaries where the oceanic climate deltas were incomplete due to their limited resolution, particularly along the western and eastern boundaries of the SGM1000x domain. The RCP85 and RCP85_DESAL scenarios are the respective future analogues to the baseline REF and REF_DESAL experiments presented in Doubell and James (2026).

The first three years of simulation were discarded as model spin-up, allowing the model to adjust from its initial conditions and equilibrate under the imposed deltas. Consequently, the analysis presented here is restricted to the following ten years, from April 2014 to May 2024. To quantify internal variability, a three-member ensemble of the reference desalination scenario was generated using perturbed initial-conditions. This ensemble established a noise floor against which the return water results were evaluated. These experiments differ from the methodology used in Doubell and James (2026) in one key aspect: no relaxation to sea surface temperature (SST) observations or sea surface salinity (SSS) climatology was applied. Because relaxation continuously nudges the modelled fields towards prescribed historical values, this nudging mechanism would counteract the imposed climate deltas and artificially damp the projected future climate response.

Following Doubell and James (2026), the higher salinity return water was simulated conservatively in the REF_DESAL and RCP85_DESAL scenarios. The outfall lies approximately 25 km south of Whyalla, 3 km west of the NWP3 mooring site and 13 km north of the northern limit of the Munyaroo Conservation Park (green hatch, Figure 1-1, inset). The modelled return water represents a hypothetical, conservative, higher-capacity ~73 GL yr^{-1} plant and was imposed as a salinity perturbation alone, without momentum flux, in a single idealised, fixed-in-time, grid cell at 18.6 m depth. The original plant design assumed a plant efficiency of ~40%, and as such, a return water flow with salinity increases equivalent to ~1.7 times the intake value. The salinity perturbation was introduced as a constant input confined to the bottom sigma layer of the model (which represented <10% of total water column depth). The simulations here are limited to the far-field oceanographic response to projected climate

conditions and the return-water from the desalination plant. Near-field mixing zone processes and diffuser hydraulics and performance were not considered in this study.

Results are presented at two scales: the large-scale climate-model seasonal projections and climate deltas across the Great Australian Bight, and the localised, regional downscaling and desalination statistics within Spencer Gulf. To resolve shifts in regional dynamics, along and cross-shore transects are evaluated alongside variability at the outfall location (Figure 2-2b). Throughout this report, salinity is reported on the Practical Salinity Scale 1978 (PSS-78) and is technically dimensionless but is referred to in the text as PSU for clarity and consistency with historical observations and the companion report of Doubell and James (2026).

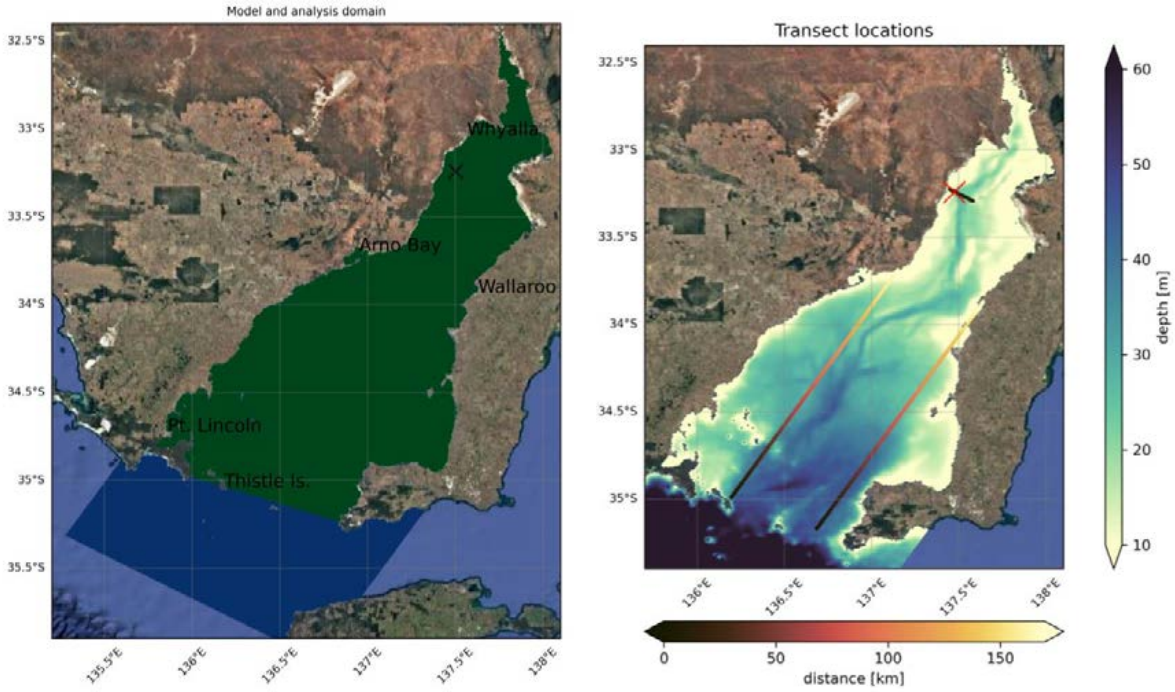


Figure 2-2. (A-Left) The high-resolution Spencer Gulf model domain showing the adjacent shelf region (blue shaded area) and the Spencer Gulf interior region included in the analysis (green shaded area). (B-Right) The model bathymetry (m) showing the along- and across-gulf transects used in the analysis. The colour shading of the along (cross) gulf transect(s) indicates the distance (km) from the gulf mouth or coast for the transect located in Upper Spencer Gulf near Mullaquana Station. The red cross marks the outfall location.

3. RESULTS

3.1. Future projections along the Great Australian Bight and the Lincoln Shelf

This section summarises projected changes in key ocean variables along the GAB, including sea surface height (SSH), temperature and salinity derived from the global GFDL-ESM2M model under RCP85. Results are presented to: (i) place the projected climate changes for the Lincoln Shelf and the South Australian gulfs region (SAG) in a broader geographical context, (ii) describe the broad-scale features resolved by the climate model alongside its limitations in coastal and the gulf region; and (iii) demonstrate why dynamical downscaling is necessary to resolve the Spencer Gulf environment. The analysis focuses on austral summer and winter climatologies because these seasons represent the dominant contrasting hydrographic and circulation conditions, including summertime seasonal upwelling events. Additionally, area-averaged climatologies are provided as complementary diagnostics to show the timing and overall seasonal trajectory of the projected changes.

Figure 3-1 shows summer and winter climatologies of the Sea Surface Height (SSH) and mean transport in the upper 200m for the baseline reference period (2006-2035) and the corresponding climate delta (bottom). The reference-period SSH and mean flow reproduce broad-scale circulation features of the GAB, including alongshore SSH gradient that decreases from Western Australia to Tasmania. Marked seasonal differences occur in the western GAB, characterized by a westward slope flow in summer and an eastward flow in winter. These patterns are consistent with the Flinders current, and the winter eastward flow matches the South Australia Current (SAC) reported in observational (Ridgway and Condie, 2004; Atlas et al., 2011; Ridgway and Dunn, 2007; Duran et al., 2020) and modelling (Middleton and Cirano, 2002; Oke et al., 2018; 2021) studies. The SSH field is also consistent with north-westward geostrophic flow and northward Sverdrup transport feeding the Flinders Current (Bye, 1978; Cirano and Middleton, 2004; Tomczak et al., 2004), so the summertime and wintertime circulations are upwelling and downwelling-favourable (Middleton and Platov, 2003). The upper 200 m circulation is broadly geostrophic. However, the model's resolution is a clear limitation over the shelf, as indicated by the coastal masking, and is insufficient to resolve Spencer Gulf.

Under the RCP8.5 scenario, dynamic sea-level is projected to decrease by approximately 4 cm near Spencer Gulf as part of broader large-scale pattern across the GAB (Figure 3-1). This reduction is relatively uniform over the Lincoln Shelf and is broadly consistent with CMIP5 ensemble for the region (Zhang et al., 2017; Grose et al., 2020; Lyu et al., 2020). Notably, the

change is sea surface height (ΔSSH) represents the ocean dynamic sea-level component only, reflecting regional circulation-driven changes, and excludes contributions from global thermosteric sea-level rise, icesheet melt and other processes (e.g. Yin, 2012; Zhang et al., 2017). These are also not the focus of this report. Readers seeking a component-based analysis of projected sea-level change around Australia are referred to Zhang et al. (2017).

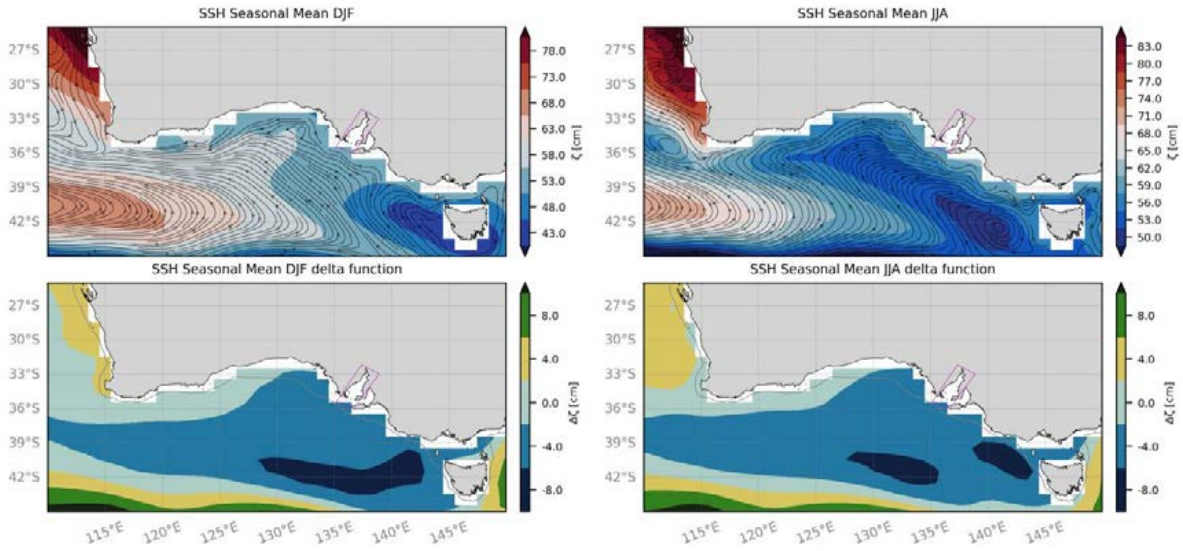


Figure 3-1. Results of the GFDL-ESM2M RCP8.5 scenario showing mean transport in the upper 200m (streamlines) and sea surface height (SSH, cm) in summer (left) and winter (right). Top: 30-year mean summertime SSH for the baseline reference period (REF – 2006-2035). Bottom: projected change in SSH (ΔSSH) between the future period 2066-2095 (RCP85) and the baseline reference period (RCP85 − REF). The SGM1000x model domain is represented by the magenta polygon, and the light grey line is the 200m isobath.

The summer and winter climatology of Sea Surface Temperature (SST) and its projected future change are shown in Figure 3-2. The baseline reference-period SST reproduces the typical meridional SST gradient, although regional shelf features, such as upwelling fronts, are limited by the model's spatial resolution (Foster et al., 2014). For the summer season, cooling signals along the slope off Spencer Gulf and the Bonney Coast are captured. In winter, the export of warmer water along the eastern GAB is also captured. However, the narrow South Australian current (SAC) front appears substantially broader compared to that predicted in reanalysis products (Oke et al. 2018), likely due to the model's limited resolution.

Under the RCP8.5 scenario, projected coastal SST increases of 1-1.3 ˚C along the coast are broadly consistent with positive trends of +0.2 ˚C decade^{-1} observed in SA waters between 1990-2016 (McLeay et al., 2019). The summer deltas in the GAB are flanked by much higher anomalies in the west and southeast, while in winter, this regional attenuation is absent (Figure 3-2). The lower projected SST increases in summer appear to coincide with an incursion of

colder water associated with the Flinders Current, whereas the larger winter deltas may reflect warmer waters associated with the Leeuwin Current. These results suggest that surface waters off Spencer Gulf may experience their largest projected warming during winter.

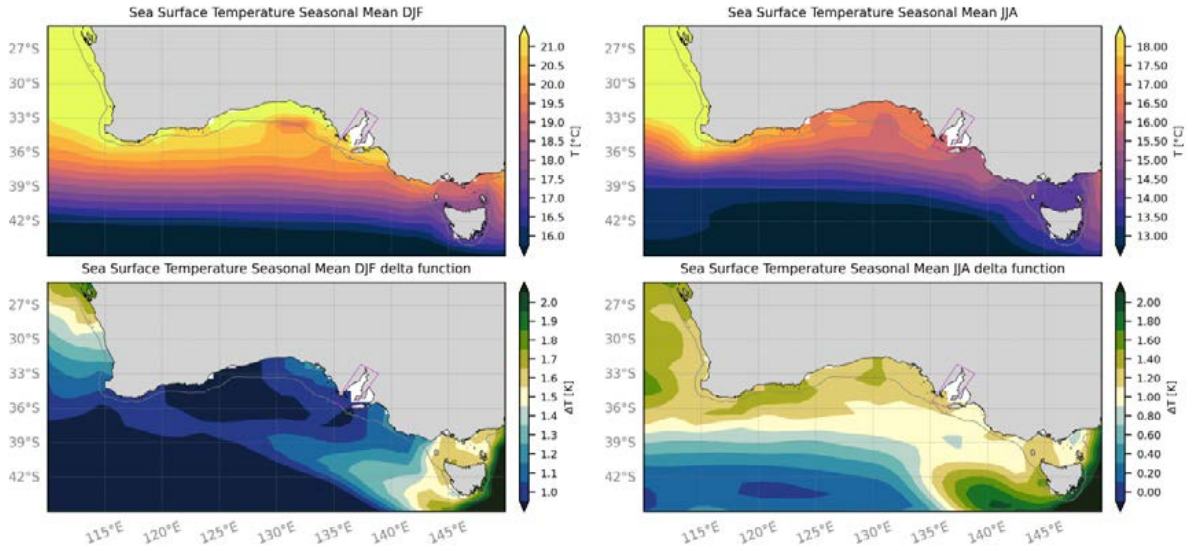


Figure 3-2. Results of the GFDL-ESM2M RCP8.5 model for sea surface temperature (SST, °C) in summer (left) and winter (right). Top: 30-year mean summertime SST for the baseline reference period (REF – 2006-2035). Bottom: projected change in SST (ΔSST) between the future period (2066-2095; RCP85) and the baseline reference period (RCP85 - REF). The SGM1000x model domain is represented by the magenta polygon, and the light grey line is the 200m isobath.

Figure 3-3 shows the summer climatology of sea-surface salinity (SSS) and is highlighted by high-salinity band between 33˚S and 36˚S that freshens markedly towards the south-east along the eastern GAB coastline. Offshore of Spencer Gulf, the baseline mean oceanic salinity is below 36 PSU. Under the RCP8.5, salinity is projected to increase by approximately 0.3 PSU as part of a broader along-slope pattern across the GAB. This pattern appears to have a predominantly oceanic origin and is consistent with the transport of high-salinity anomalies (exceeding 0.46 PSU) associated with the Tasman Outflow (Ridgway and Condie, 2004). Notably, the summer distribution is consistent with the winter season (not shown).

The projected increases in SST (1–1.3 °C) and SSS (approximately 0.3 PSU) are consistent with the mean ensemble projections from the CSIRO and the Bureau of Meteorology (2015). While the GFDL-ESM2M model reproduces the broad regional seasonal pattern its spatial resolution limitations are partly addressed by using BRAN2023 reanalysis data to provide higher-resolution, interannual and sub-monthly boundary variability.

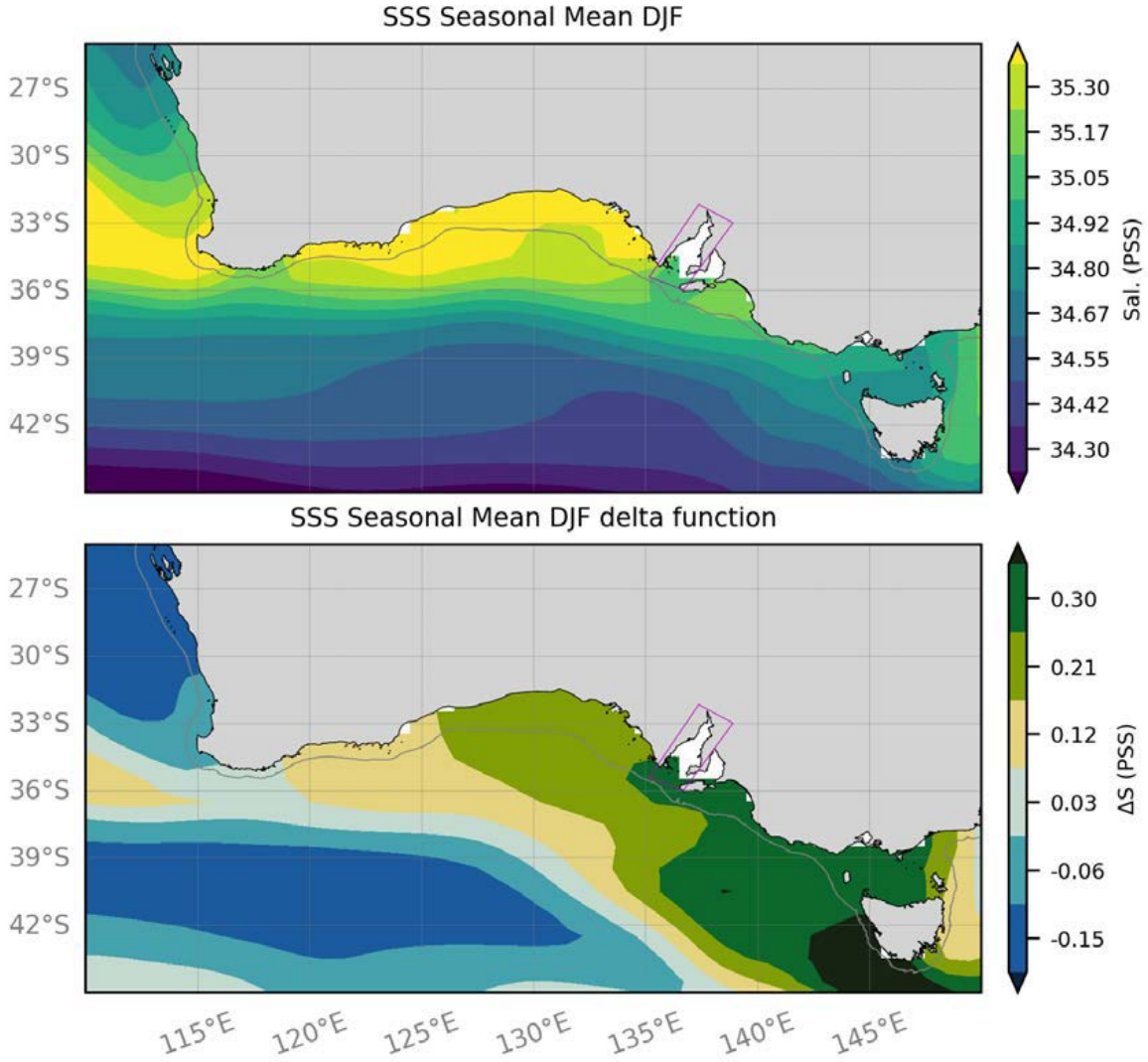


Figure 3-3. Results of the GFDL-ESM2M RCP8.5 model for sea surface salinity (SSS) in summer (December, January, February). (Top) The 30-year mean summertime SSS for the historical reference period (REF– 2006-2035). (Bottom) The future change in summertime conditions (ΔSSS) under the RCP8.5 emissions scenario relative to 2066-2095 (RCP85 – REF). The SGM1000x model domain is represented by the magenta polygon, while the light grey line is the 200m isobath. The first nearshore estimates masked as land were extrapolated from nearest neighbours (see Figure 3-1) Salinity values are derived using the practical salinity scale (PSS-78) equation. These dimensionless values are presented here as equivalent to Practical Salinity Units.

To examine the climate-delta perturbations vertically across the water column, the summertime reference conditions from BRAN2023 (Figure 3-4, top) are compared with projected changes from GFDL-ESM2M (bottom) along the southern boundary of the SGM1000x domain. The reanalysis shows a warm, saline, well-mixed surface layer approximately 20 m deep and a weak Kangaroo Island Pool (KIP): a cold, relatively fresh tongue of water, with temperature below 14 °C, extending westward along the seabed near Kangaroo Island (McClatchie et al., 2006). Salinity in BRAN is also markedly higher nearshore (Figure 3-4, top right) while progressively colder, fresher bottom waters between 40 and 120km offshore are consistent with an upwelling signal.

The projected changes are coarse in the horizontal, as expected, but show pronounced vertical structure (Figure 3-4, bottom). Projected warming is approximately +1°C in the surface mixed layer and near the seabed. At the surface mixed layer interface (25-50 m), the temperature increases are weaker, reducing from 0.7°C offshore to ~0.5°C onshore. This pattern may reflect increased upwelling or enhanced advection of cooler southern waters onto the Lincoln Shelf. Projected salinity increases strengthen towards the surface, reaching approximately 0.3, and towards the coast (Figure 3-3). The smallest increases occur in the bottom layer (<0.2), a pattern consistent with a greater influence of upwelled waters at depth.

Surface waters in BRAN are approximately 1 °C cooler and 0.33 PSU fresher than in the climate model, although the averaging periods differ (see Figure 3-2 and Figure 3-3). These differences are not examined further because the products differ in resolution, but they are expected, since BRAN2023 assimilates a wide range of observations, uses atmospheric reanalysis, and has a much higher horizontal resolution. All regional-model scenarios use BRAN2023 as the baseline for oceanic boundary and initial conditions, with RCP85 and RCP85_DESAL scenarios modified by monthly climate-delta perturbations. The implications of differences between BRAN2023 and GFDL-ESM2M in offshore sea-surface height, temperature and salinity are not examined further here. Nonetheless, given the restricted exchange with the adjacent shelf and the shallow bathymetry of Spencer Gulf, atmospheric fluxes and their seasonality cycle play an important role in gulf dynamics. The following section examines projections for the atmospheric surface boundary conditions within the region.

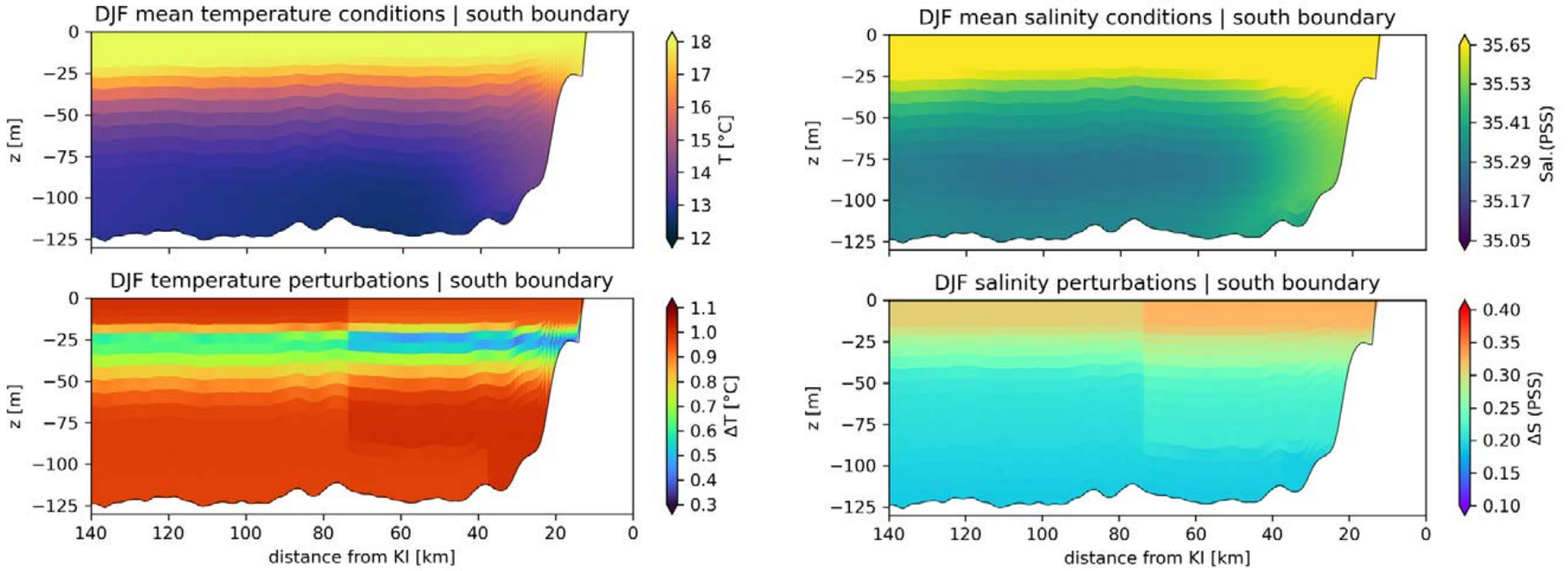


Figure 3-4. Mean (2013-2023) baseline summertime reference conditions for (left) temperature (°C) and (right) salinity (PSU) across the water column derived from the BRAN2023 reanalysis model used to force the higher-resolution Spencer Gulf (top) and the climate delta perturbations from GFDL-ESM2M RCP8.5 scenario model (bottom). The transects correspond with southern boundary of the Spencer Gulf model (SGM1000x) used for dynamical downscaling shown in (Figure 2-1), with the x-axis showing the distance (km) from the eastern extent of the model boundary near Kangaroo Island.

3.2. Atmospheric delta functions

This section presents projected seasonal atmospheric conditions and associated climate deltas derived from the GFDL-ESM2M model under the RCP8.5 scenario (RCP85).

Evaporation and precipitation determine the net freshwater flux and strongly influence the salinity balance and distribution in Spencer Gulf (Nunes Vaz et al., 1990). Figure 3-5 compares the evaporation and precipitation rates for the baseline reference and future climate scenarios over the SAG region (magenta square, shown in Figure 3-6). The SAG polygon is used because it defines the GFDL-ESM2M atmospheric region contributing to the downscaling. Comparison of monthly rates between the baseline reference and future climate periods indicates an overall increase in evaporation. Mean oceanic evaporation rates increase by 0.1 - 0.5 mm day^{-1} in most months, except during autumn. January remains the month with the highest evaporation, whereas the largest projected mean increase occurs in July and August. In most months, the upper quartile rises (75th percentile, indicated by orange upper dashed line), as do the January and February minima. By the end of the century, January and February evaporation rates are not projected fall below 3.5 mm day^{-1}.

The projected increase in evaporation contrasts with a decrease in precipitation (Figure 3-5). Precipitation remains lower than evaporation, indicating that net evaporation continues to dominate the surface freshwater balance in the region. This is consistent with the limited influence land run-off on Spencer Gulf. The predicted seasonal cycle also indicates an approximately 33% reduction in spring rainfall. Together, the projected changes in evaporation and precipitation, indicate an overall reduction in net freshwater input under the RCP8.5 climate scenario. This is consistent with the SSS described in the previous section. Biases in the absolute magnitude of evaporation and precipitation may remain, particularly for winter precipitation. However, the delta-downscaling approach applies the difference between future and reference periods, so the influence of systematic biases is reduced where those biases remain stable between periods.

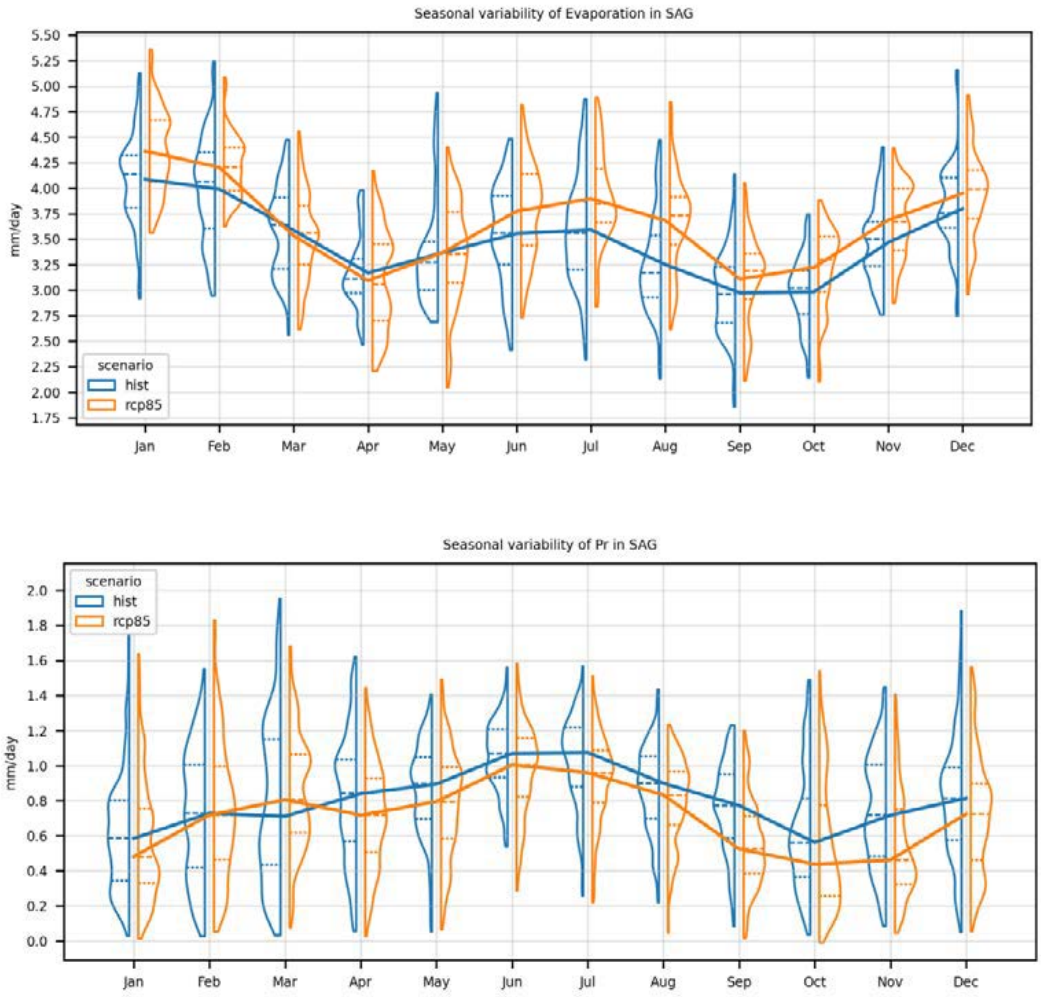


Figure 3-5. The seasonal cycle of monthly mean evaporation (top) and median precipitation (bottom) in mm day^{-1} for the historical reference period (REF) (blue) and future RCP8.5 climatic condition (RCP85) (orange) in the SAG region. For each month, the corresponding probability distribution is determined over the 30-year reference (2006-2035) and future period (2066-2095). Evaporation estimates exclude land contributions.

Figure 3-6 shows mean summertime climate deltas for near-surface air temperature, relative humidity, latent heat flux and sensible heat fluxes, mean sea-level pressure, and surface wind speed. Projected air temperature increases show a strong land to ocean gradient across southern Australia, with summer increases of 1-2.5°C predicted over the SAG region (panel A). Relative humidity decreases by more than 4% over land, but by less than ±1 % over the SAG region (panel B). Latent heat flux increases by 2 - 8 Wm^{-2} over the shelf (panel C), while changes in sensible heat flux are smaller (panel D), indicating enhanced evaporation-driven heat transfer to the atmosphere.

During summer, the Great Australian Bight lies beneath a persistent high-pressure system while a thermal low develops to the north of Australia. The resulting cross-shore pressure gradient promotes southeasterly winds that are favourable for coastal upwelling. The summer

deltas reinforce this configuration, with mean sea-level pressure increasing over the ocean and decreasing over land (panel F). The strengthened pressure gradient is consistent with the increase in surface wind speed shown in panel E. Projected increases in wind speed are larger offshore than within the gulfs, increasing from approximately 0.05 m s^{-1} within the gulfs to 0.3 ms^{-1} over the shelf. This could imply an increase of approximately 10% in the Ekman Transport assuming upwelling winds of 6 ms^{-1}. The resulting enhancement in upwelling is consistent with the reduced projected SST increases in summer (Figure 3-2). Latent heat flux changes also peak at SAG in summer (-4 Wm^{-2}), and with air-temperature increases persisting year-round (not shown) complete a consistent picture of higher evaporation throughout the year (Figure 3-5).

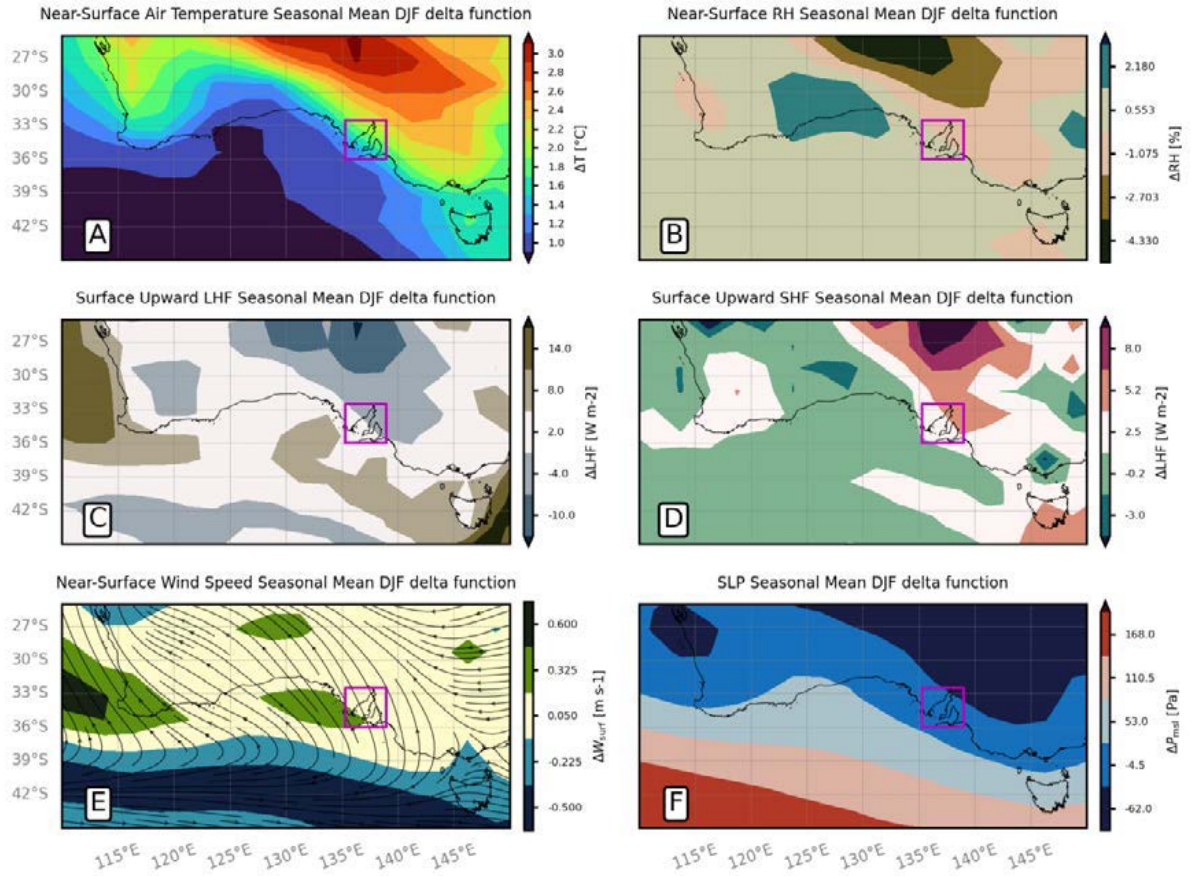


Figure 3-6. Maps of the predicted summertime climate delta (Δ) determined between a future climate change period and the baseline reference period (i.e., RCP85 - REF) for: A. surface air temperature (°C), B. relative humidity (%), C. surface latent heat flux, D. sensible heat flux (Wm^{-2}), E. surface wind strength (ms^{-1}) and F. mean sea-level pressure (Pa). Wind direction streamlines are from the reference period. The SAG region is enclosed by the magenta square and includes all atmospheric grid elements which contributed to the dynamical downscaling.

Figure 3-7 compares the seasonal cycle of meridional surface wind stress over the SAG region for the baseline reference and future climates periods. Projected meridional wind stress are predicted to increase throughout the year and peak in summer at approximately 0.01 Pa. The projected changes suggest that summer winds may strengthen and acquire a slightly more westward component, given the small increment in the zonal wind stress (not shown). This is consistent with enhanced coastal upwelling and potentially greater transport of cooler subsurface waters towards the mouth of SG in summer at depth (Petrusevics,1993; Petrusevics et al., 2011). The distribution of the monthly wind stresses also shows an increasing frequency of high-wind months, particularly during summer and autumn, indicated by an upward shift of the predicted minima.

As for the ocean-boundary forcing described above, monthly atmospheric flux perturbations were added to the sub-daily CFSR forcing for the 2011-2024 simulation period. Heat and freshwater fluxes were applied jointly within the 1km resolution gulf model, with the atmospheric conditions imposed via bulk-flux parametrisations (Fairall et al., 2003).

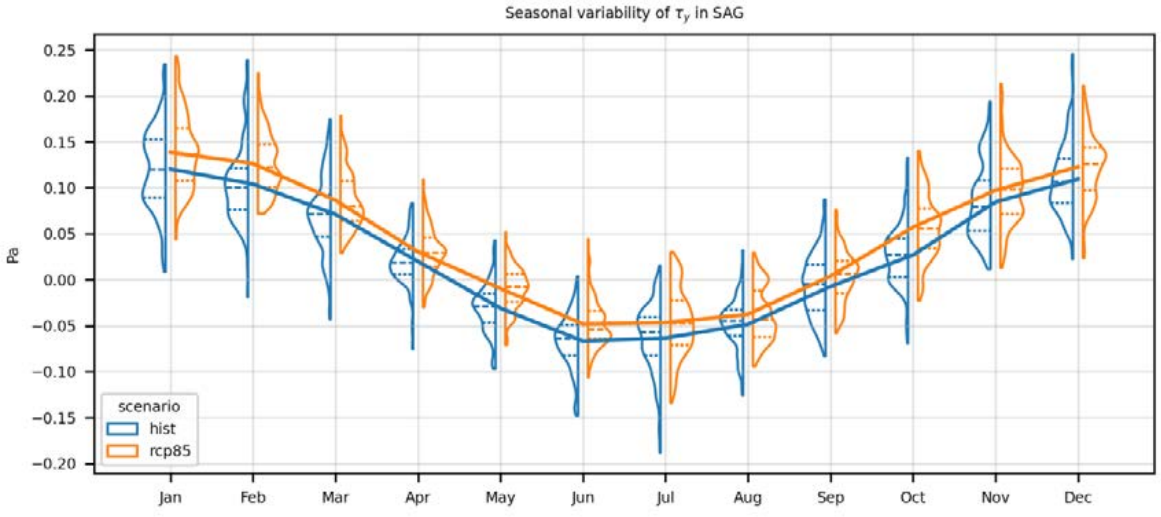


Figure 3-7. The seasonal cycle of surface meridional wind-stress (Pa) for the historical reference period (REF) (blue) and future RCP8.5 climatic conditions (RCP85) (orange) in the SAG region. For each month, the corresponding violin plot shows the probability distribution of the monthly values determined over the 30-year reference and future periods, respectively.

3.3. Downscaling of future climate conditions

This section presents downscaling results obtained by applying the climate-delta functions to the Spencer Gulf model (SGM1000x). Figure 3-8 shows 10-year mean bottom salinity and circulation in the REF experiment (left), together with the corresponding salinity anomaly (ΔS, unitless) under the downscaled future RCP85 scenario (right). The bottom circulation shows higher salinity water from Upper Spencer Gulf exiting on the eastern side of the Gulf. This is balanced by an inflow of lower salinity water on the shallower western side. A clockwise circulation is most prominent in the southern Gulf and is broadly aligned with the bathymetry, although cross-isobath flow is evident off Port Lincoln, within Boston Bay and Franklin Harbour, and south of Wallaroo (see Figure 1-1 for regional location references).

In the narrower Upper Spencer Gulf region, the mean circulation reveals several consistent recirculation cells. Near the outfall, the dominant flow is north-eastward over the shallower 10 m isobath, whereas higher salinity water flows westward between the 10 and 25 m isobaths (Figure 3-8, inset). The salinity distribution is consistent with the Gulf's expected inverse-estuarine circulation, with salinities exceeding 38.5 PSU in the west and at depth along the deeper channel (25 m). A sharp east-west salinity gradient occurs at the narrowing point north of Franklin Harbour, where a relatively fresher water mass, with salinity below 37 PSU, extends towards the outfall region in water shallower than 25 m depth.

Under RCP85, mean bottom salinities increase throughout Spencer Gulf (Figure 3-8, right panel). The largest increases occur in the west and north and broadly correspond with the areas of higher salinity shown for the baseline reference-period. Within Upper Spencer Gulf, long-term mean salinity increases exceed 0.6 and reach more than 1.0 PSU in areas north and east of Whyalla. Within the desalination outfall region, bottom salinities are expected to increase between 0.5 and 0.65 PSU. The hatching identifies cells where the daily mean salinity differs significantly between simulations according to a median-centred Levene's test at a $p \leq 0.05$ level (Brown and Forsythe, 1974). Bottom salinity variability differs significantly over wide areas, as expected. Within the outfall region and along the 10 m isobath, however, salinity variability does not differ significantly between scenarios, potentially reflecting strong tidal stirring and mixing in this region under both conditions. The depth-average salinity pattern (not shown) mirrors the presented bottom salinity estimates.

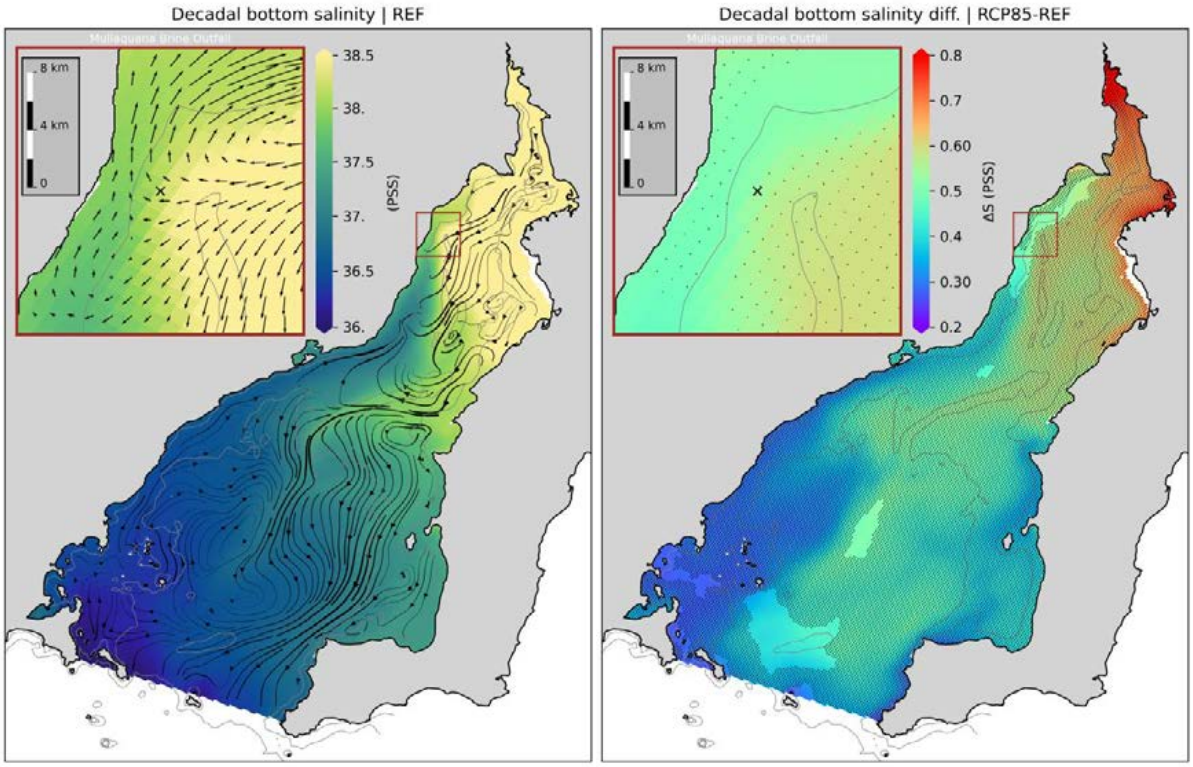


Figure 3-8. Predicted changes in the 10-year mean salinity distribution in Spencer Gulf under future climate conditions. (Left). The 10-year mean circulation and near bottom salinity from the reference simulation (REF). (Right) Future changes in salinity (ΔS, unitless) predicted under the future climate change scenario (i.e., RCP85 – REF). The hatching represents regions of statistically significant ($p<0.05$) variances between the scenarios). The light grey lines represent the 10,25 and 50m isobaths.

Future salinity increases are accompanied by marked increases in depth-averaged temperature throughout the Gulf (Figure 3-9). The largest changes in mean water temperatures occur in Upper Spencer Gulf with predicted increases of 1-2˚C. The increases also follow a north-south pattern along the eastern coastline and are broadly consistent with the pathway of high-salinity outflow from the gulf. The smallest increases in temperature are predicted in the south-eastern and western regions of Spencer Gulf. In the Mullaquana Station region, the depth-averaged temperature is projected to increase by more than 1.6˚C. The change in variability between baseline reference and future climate scenarios is stronger in depth-averaged temperature than in bottom salinity and affects most of the gulf, except for an evident irregular belt spanning Port Lincoln to Yorke Peninsula across the deeper water (>50m) regions of the southern Spencer Gulf.

While warming and salinification increase in some regions and cancel in others, the net density change (Figure 3-9 – right) resembles that predicted for the bottom salinity anomaly (Figure 3-8) rather than the temperature anomaly. This indicates that salinity make the larger

contribution to the density response, consistent with the expectation that salinity dominates the density response when the temperature change is less than four times the salinity change (IOC, SCOR, and IAPSO, 2010). Density anomalies are mostly within ±0.15 kg m^{-3}, with the largest positive anomalies occurring along the eastern coast and in USG, where salinity changes are maximum. Near-zero or negative density anomalies are confined to shallow areas along the western coastline, within the northward flow around Franklin Harbour, where salt anomalies are less than 0.3 PSU but higher temperatures are expected (~1.3˚C). The change in density variability (hatching) reflects this as most areas of small density changes are not marked as significant. In the Mullaquana Station region, density decreases slightly nearshore and changes little offshore (inset), leaving the outfall in slightly lighter waters in the future.

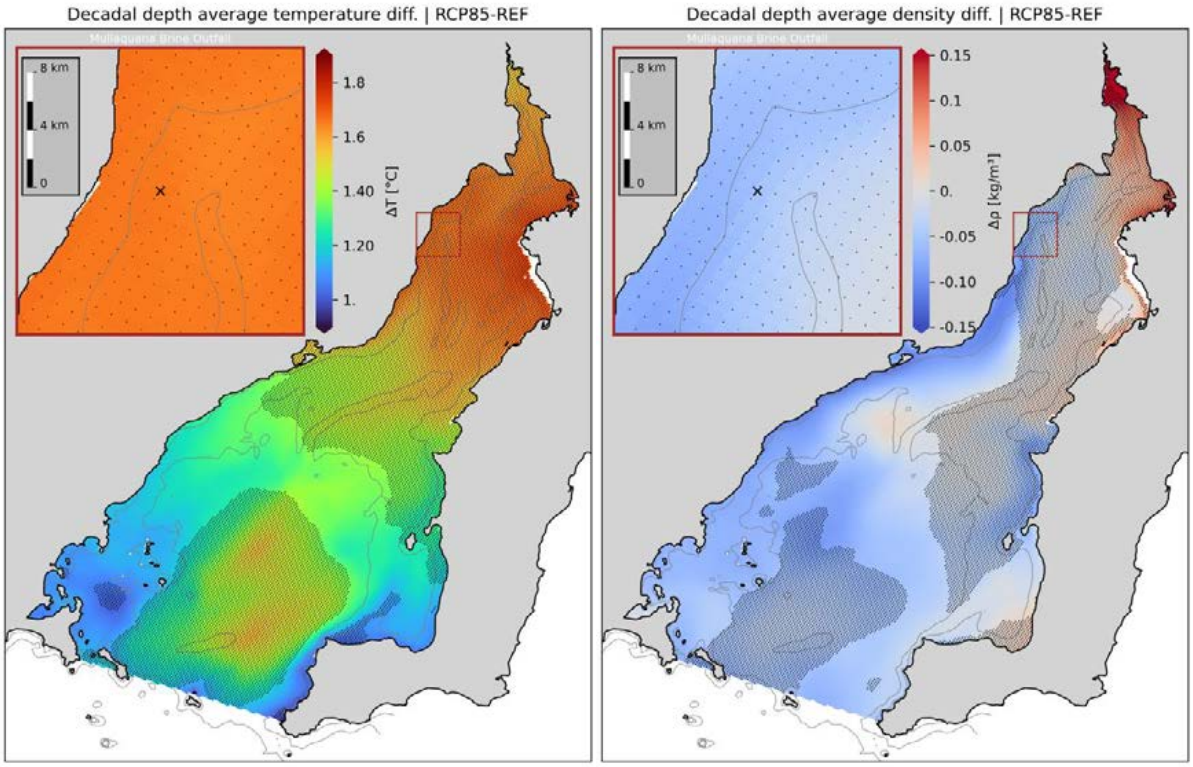


Figure 3-9. The 10-year mean changes in depth-averaged (left) temperature (°C) and (right) density (kgm^{-3}) between the future climate change scenario and the historical reference scenario (i.e., RCP85 - REF). The hatching represents regions of statistically significant ($p<0.05$) variance between the scenarios). The black cross (inset) represents the planned location of the Mullaquana Station desalination plant return water outlet (outfall).

Figure 3-10 compares mean vertical distributions of salinity and density for the baseline reference (REF) and future-climate (RCP85) scenarios along the southern transects on the western and eastern sides of Spencer Gulf shown in Figure 2-2. In the west (top-left panel),

horizontal gradients of salinity are smaller than in the east (top-right panel), as is vertical stratification. The eastern sector of SG holds much denser, saltier water than the west, particularly within the first 100 km from the mouth. This dense, saline water is consistent with an origin in the northern sector with predicted densities above 26.7 kg m^{-3}.

The climate future scenario along both transects (Figure 3-10, bottom) predict reductions in density, with a strong southward shift of isohalines. In the western transect, for example, the 36.6 isohaline shifts more than 80km south relative to baseline scenario (bottom left). Surface density decreases most in the west, increasing the upper stratification and horizontal density gradients, for example between 100 and 120 km. In the eastern transect, the high-salinity bottom waters between 0-40km from the mouth also become less dense, because its source water beyond 100km is itself lightened and more stratified (37.8 PSU isohaline). Salinity perturbations over most of the transects under the future climate scenario are close to 0.4 PSU, consistent with the bottom estimates in Figure 3-8. Thus, at the northern limit of the eastern transect, the southward shift of isohalines and the reduction in density are maintained throughout the water column. The wider spacing of near-bed isohalines at both sites is consistent with enhanced near-bottom transport.

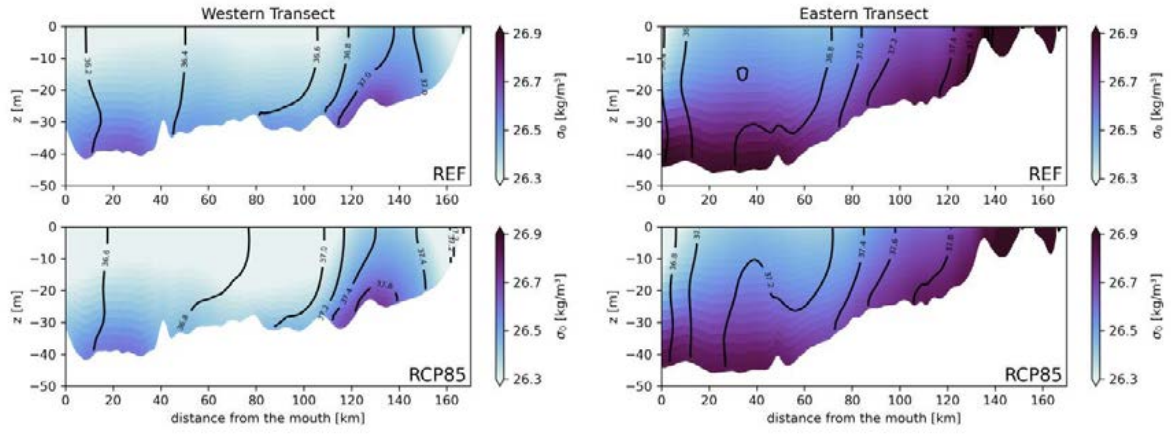


Figure 3-10**.** The 10-year mean vertical distribution of salinity (contours) and density (shading - kgm^{-3}) over the along gulf transects on the eastern and western side of Spencer Gulf predicted under the (top) historical reference scenario (REF) and (bottom) future climate change (RCP8.5) scenarios. See Figure 2-2 for the transect locations.

Ten-year mean depth-average circulation and sea-surface height (SSH) in the REF and RCP85 simulations (Figure 3-11) indicate that the Gulf's general circulation pattern remains largely unchanged. The clockwise circulation near the mouth persists, with only marginal reorganisation of the barotropic gyre. The main pathway of relatively fresher water in the south-west is also retained: the mean current crosses isobaths south-east and north-east of Boston Bay, feeding deeper, fresher water to the west coast that is then advected north. A

strong outflow of high-salinity water veers topographically along the channel, west and south-west of Wallaroo, with this flow predominantly leaving through the eastern side of the mouth.

In the Upper Spencer Gulf, the depth average circulation also crosses isobaths, with net northeast advection at the Mullaquana Station region. This contrast with the westward near-bottom flow suggests a vertically sheared profile, or a very compressed boundary layer in the return water region, since the bottom flow was predominantly to the west (Figure 3-8, inset). Changes under RCP85 scenario are small and not readily discernible in the depth-averaged circulations at this location, which is not surprising given the strong tidal dominance characteristic observed in the region (e.g., see Figure 9 in Doubell and James, 2026).

On decadal scales, and particularly in the deeper regions, the stream function and SSH nearly coincide, leading to a strong geostrophic footprint in the south and near the mouth. Consistent with the climate-delta boundary conditions, the dynamical sea level offshore falls by about 4 cm on average, but this is partly transmitted inward, leaving a smaller reduction (~1cm) at the head of the gulf. The along-gulf pressure gradient between the mouth and the head therefore approximately twice that in the reference scenario. This is expected, as increased evaporation and salinity outweigh the warming at the head, the USG region becomes a stronger source of dense water and density-driven flow. A stronger bottom flow towards the mouth would increase frictional effects, force new pressure balance to offset greater mass loss, and reorganise the circulation, including a stronger return flow (surface intensified). At Mullaquana Station, there are no significant differences in the barotropic flow structure in the long-term mean (Figure 3-11).

To examine circulation changes through the water column, Figure 3-12 shows the mean currents along the western and eastern transects of lower Spencer Gulf. In the west, under REF, the predominant alongshore flow is north-eastward and broadly uniform to approximately 100km from the mouth. Near the channel edge, however, the water column has a weak vertical sheared, with a south-westward flow near the bottom (blue shading, dash lines). Under the future scenario (bottom panel), this flow strengthens by approximately 0.03m/s^{-1}) across much of the transect, particularly near the surface, and vertical shear increases in both along and cross-shore directions. The upper inflow and limited bottom outflow in this sector are therefore enhanced.

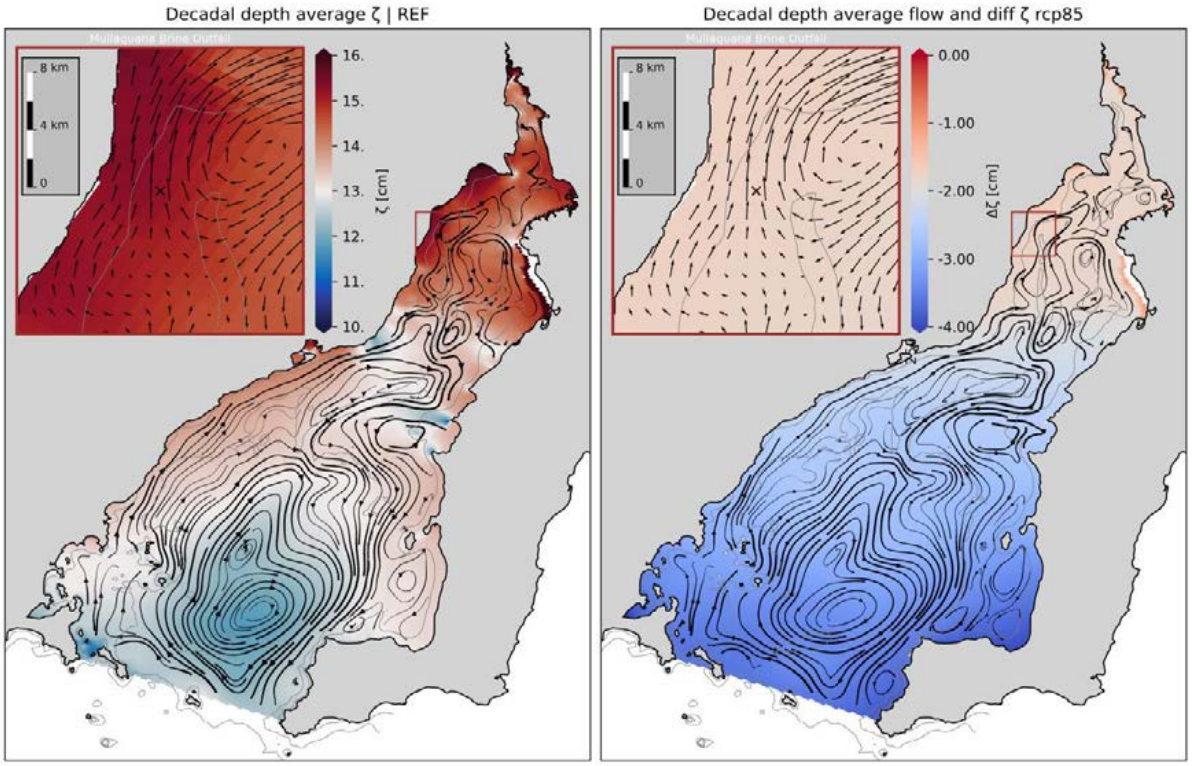


Figure 3-11. Comparison of the 10-year mean sea-surface height (SSH) and depth-averaged circulation simulated under (left) the historical reference (REF) and (right) future climate change (RCP85) scenarios. For the right panel, the shading is the differences in SSH. Streamlines are computed with the same parameters and linewidth scale with depth-average flow magnitude.

The eastern sector shows a markedly different structure. A dominant south-westward flow extends through most of the transect and is bottom intensified, with marked vertical shear near the mouth (Figure 3-12). Surface inflow and bottom outflow occur concurrently, an important difference from the depth-averaged circulation shown in Figure 3-11. Additional areas of vertical shear occur approximately 20 and 100 km away from the mouth. A bottom-intensified, along-transect flow directed south-westward appears 60-80km from the mouth, the signature of dense upper-gulf water cascading seaward along the floor. In the future, the three-dimensional circulation is again enhanced (Figure 3-12, bottom right panel): the northward flow intensifies, most noticeably where vertical shear is strong, and the flow at the mouth becomes more asymmetric, with stronger inflow and outflow in the decadal average. Near the bed, velocities are amplified, such as 80km and 120km in the western sector and at the mouth in the eastern one. The reductions in barotropic stream function in Figure 3-11 may reflect the combined imposed perturbation, particularly to the thermohaline changes, given the long timescales examined and the nature of the climate-delta forcing.

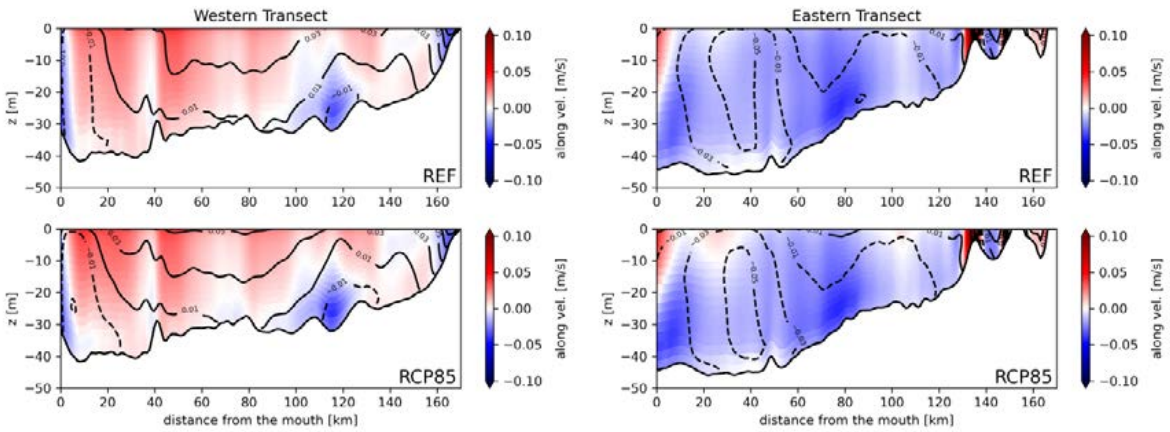


Figure 3-12. The 10-year mean vertical distribution of velocity (ms^{-1}) for the along gulf transects on the eastern and western side of Spencer Gulf predicted under the (top) reference (REF) and (bottom) future climate change (RCP85) scenarios. The colour bar represents the magnitude of the along transect currents to the northeast (red) and southwest (blue). Contour lines represent the magnitude of the cross transect currents, positive (negative) to the east (west). The transect bearing is 122 ^{o}T. See Figure 2-2 for transect locations.

3.4. Downscaling of future climate conditions with desalination return water

The desalination and climate signals were separated by differencing the four simulations in pairs, each pair isolating one effect. The return water effect is the change caused by the desalination return water alone, obtained by differencing the runs with and without desalination under a fixed climate: REF_DESAL - REF in the present day, and RCP85_DESAL - RCP85 under a high emission future climate. The climate effect is the change caused by the projected climate alone, without desalination (RCP85 - REF). The total change combines both, comparing the future climate with desalination against present-day conditions (RCP85_DESAL - REF).

Figure 3-13 (left) shows bottom-salinity anomalies attributed to return water discharge under present-day conditions (REF_DESAL) and the future climate scenario (RCP85_DESAL). Consistent with Doubell and James (2026), the largest salinity anomaly is confined to the vicinity of the outfall. The mean plume anomaly lies north-east and offshore of the outfall, where its amplitude is less than 0.4 salinity units within 8 km, falling sharply within 2km. Hatching in the figure delineates the plume's statistically detectable area of influence: cells where the salinity distributions differ significantly between scenarios according to a Mann-Whitney test ($p \leq 0.05$). Across most of Spencer Gulf, the impact is likely negligible, as the differences away from the plume are similar in magnitude to the model's estimated internal

variability (maximum ~0.01 PSU in the south, not shown). Beyond the Mullaquana Station region, the plume-related salinity anomaly is generally below 0.03 PSU.

In the future climate scenario (RCP85_DESAL – RCP85), return water salinity-anomalies are comparable in magnitude to those under present-day conditions but are shifted further north, with the statistically detectable plume footprint extending beyond Mullaquana Station and the 10 m isobath to the northeast (Figure 3.13). While salinity increases remain similar within a 2–4 km outfall radius, the surrounding regional increases and future far-field salinity anomalies are slightly more elevated (~0.04 PSU) and extend beyond the stippled boundary, indicating a broader indirect influence beyond the outfall region driven by local turbulent mixing and advective transport. Overall, the modelled desalination plume is projected to extend slightly further to the north-east under the future climate scenario.

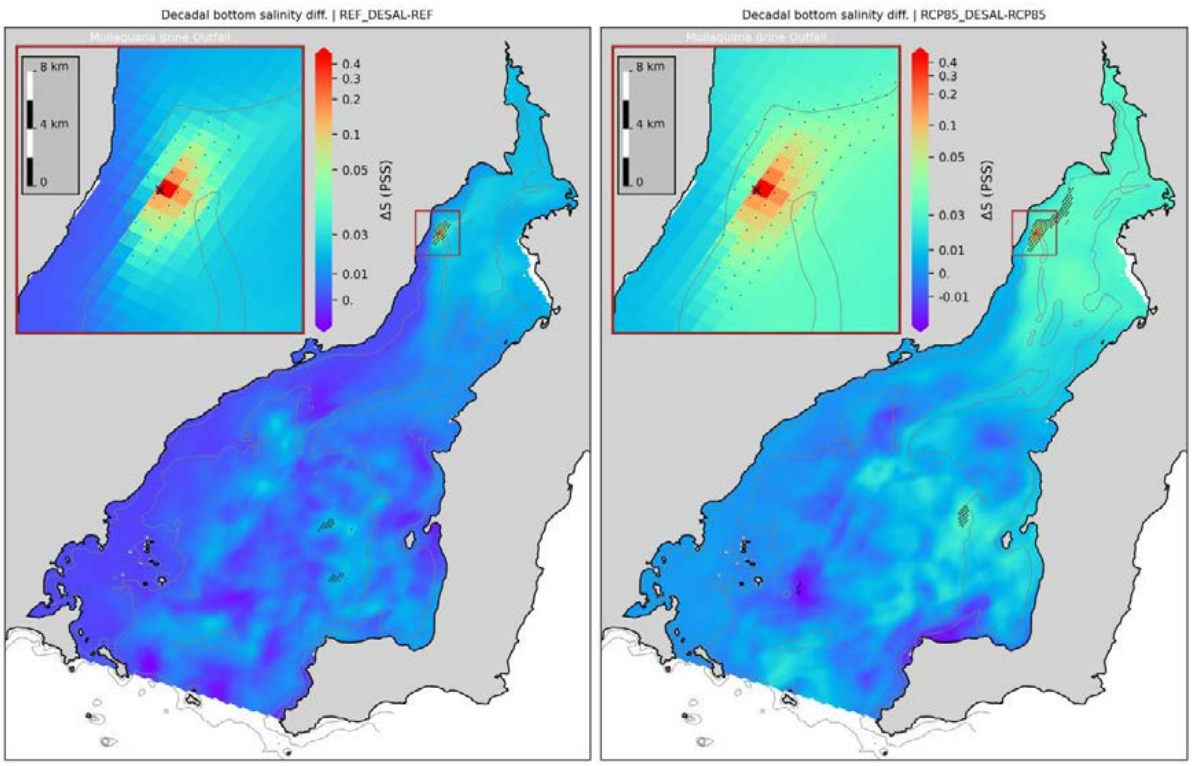


Figure 3-13. The 10-year mean differences in bottom salinity (ΔS, PSU) between the desalination impact scenarios. Left – The difference between the baseline reference scenario with and without desalination return water (REF_DESAL - REF). Right – The difference between the future climate scenario with and without desalination return water (RCP85_DESAL - RCP85). The black hatching represents significant differences ($p<0.05$) in the salinity between scenarios with and without desalination.

Figure 3-14 shows the mean distribution of salinity anomalies (ΔS) along the cross-shore transect through the outfall, illustrating the plume's vertical extent under each impact scenario (see Figure 2-2 for the transect location). In the reference period (panel A), the desalination

return water anomaly is mostly confined to the model's bottom layers, with a maximum salinity increase of less than 0.5 at 2 km from the coast, at the return water site. Concentrations decrease towards the deeper region (~25 m) and fall rapidly within the first 2 km, with the highest values confined to the lower 5 m of the water column. Considerable vertical and turbulent mixing occurs 2-5 km offshore of the site, reducing the anomaly to less than 0.03 with no considerable gradients.

The scenario with only the climate deltas (panel B) raises salinity through the whole water column by about 0.5. The combined effect (panel C) indicates that the return-water salinity anomaly is approximately additive and remains localised. On average, its amplitude is slightly smaller than the near-uniform climate-induced change. The future return water effect (panel D) shows a stronger spread of the return water towards the surface and west, with marginal increases predicted across the water column consistent with Figure 3-13 (right panel), while retaining a vertical profile and near-bottom decay broadly similar to the present-day return water scenario (panel A).

Finally, Figure 3-15 explores the climatology of bottom temperature and salinity at the outfall site for all scenarios. The climatology is computed as the daily calendar mean (and interquartile range) of the hourly time series during the 10-year analysis period. Under RCP8.5, mean bottom temperature exceeds the reference simulation by at least 1.4 ˚C on every day of the year (orange and green lines, top). The increase is greatest in winter (up to 2 ˚C) and smallest in summer, potentially because stronger summer advection of cooler southern waters moderates warming as part of the Gulf's seasonal clockwise circulation cycle (not shown). This estimate agrees closely with the long-term depth average values (e.g. 1.6 ˚C average increase for the whole period, Figure 3-9). Temperature changes associated with return-water discharge and local mixing alone are much smaller (<0.1 ˚C, red/blue lines).

Salinity increases attributable to return water alone exceed 0.125 PSU under both the reference (blue) and future-climate (red) scenarios at Mullaquana Station. They are most commonly between 0.25 and 0.37 PSU (~1% increase) and rarely exceed 0.5 PSU throughout the year (Figure 3-15, bottom). Variability is small throughout the year, with maxima occurring in January and July. Under RCP85 without desalination (orange line), climate-driven salinity anomalies are substantially larger than those attributable to return-water discharge, peaking slightly above 0.625 PSU (~1.7% increase) in summer and autumn. The salinity differences reduce considerably in winter. The future climate projections alone raise salinity above that due to the return water year-round.

Under the future climate scenario with desalination (RCP85_DESAL – REF), salinity anomalies are approximately additive at monthly time scales (green line), since the combined climatology closely matches the sum of the different effects. For all seasons, this scenario shows increases above 0.75 PSU (~2% increase) relative to present-day conditions without desalination across all seasons, and frequently exceeding 1 PSU in summer, spring and autumn. Predicted increases in salinity were consistently lower in winter. This again suggests the gulf will produce more dense water, since the largest salinity increases coincide with the smallest temperature increases.

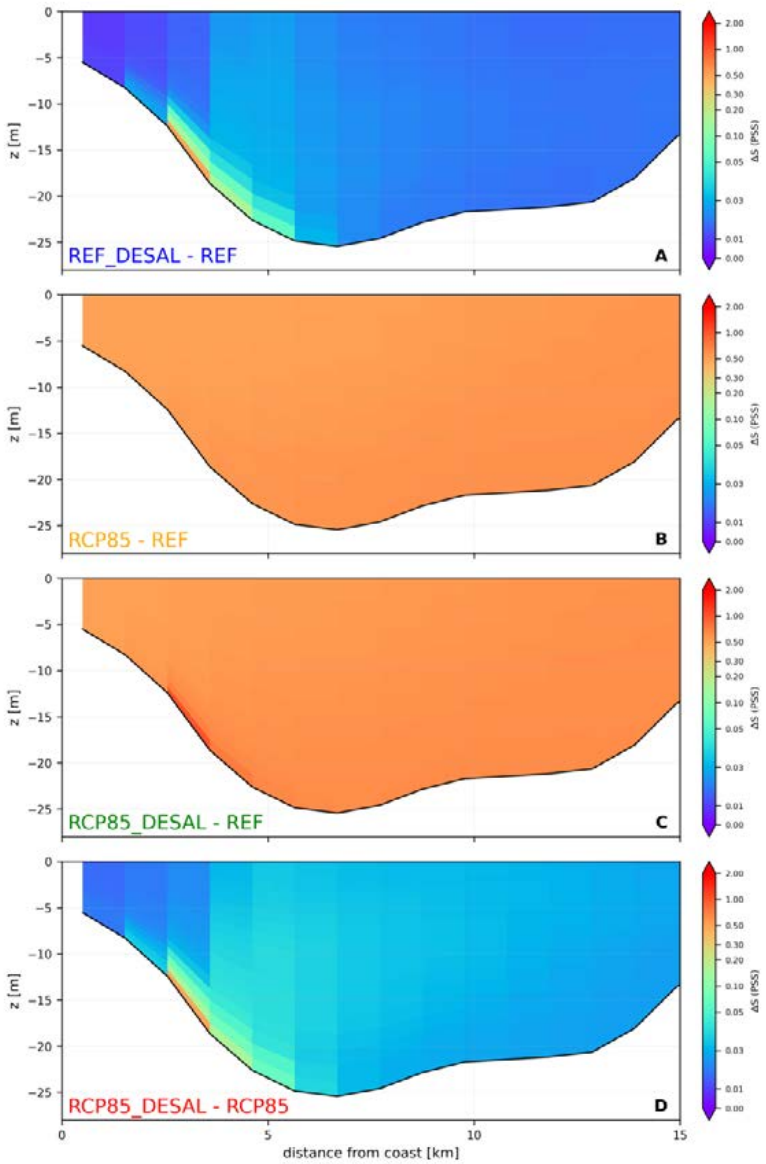


Figure 3-14. The vertical distribution of changes in salinity (ΔS, unitless) along a cross-shore transect passing through the Mullaquana Station outfall. (A) Return water effect under the present-day baseline scenario (REF_DESAL - REF). (B) The future climate effect (RCP85 – REF). (C) Total change, future climate with return water versus present day (RCP85_DESAL - REF). (D) Return water effect under future climate scenario (RCP85_DESAL – RCP85). The x-axis shows the distance (km) from the coast. See Figure 2-2 for the transect location.

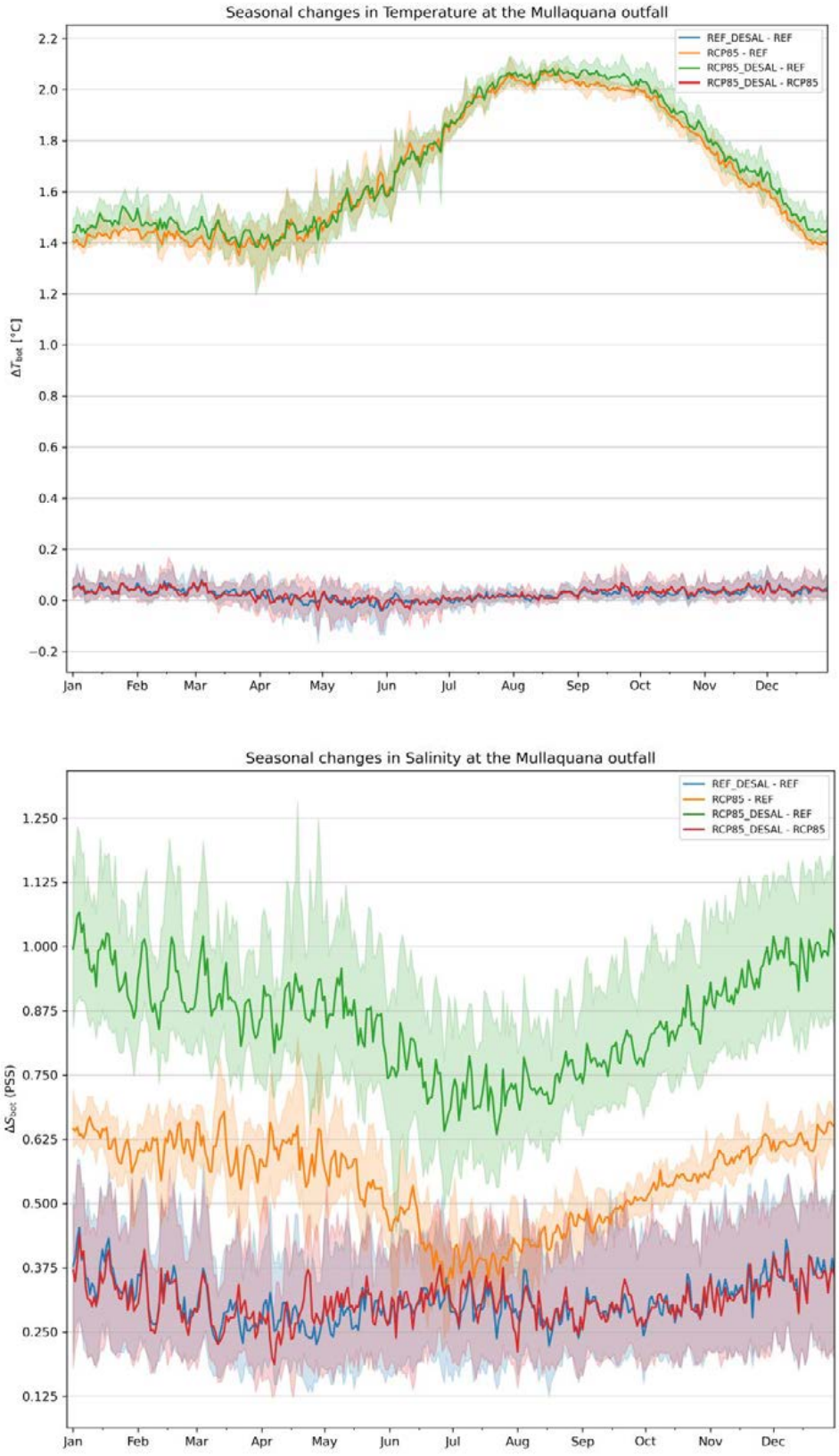


Figure 3-15. Time-series showing the 10-year seasonal mean daily changes (Δ) at the Mullaquana Station outfall location for temperature (top) and salinity (bottom) between all four modelled scenarios. The estimates are based on near bottom predictions at the desalination outfall location. The solid line represents the mean difference for each scenario, and the shading is the interquartile range (25th and 75th percentile limits for each day of the year).

4. DISCUSSION AND CONCLUSION

This report presents a dynamic downscaling study of future oceanographic conditions in Spencer Gulf. A high-resolution regional ocean model was forced by atmospheric and ocean reanalysis and perturbed in two ways: (a) a high-emissions climate change scenario (RCP8.5), and (b) an idealised, higher-capacity desalination-discharge scenario of 73 GL yr^{-1} desalination plant at Mullaquana Station, in Upper Spencer Gulf. The two perturbations were applied separately and together, allowing their modelled contributions to changes in oceanographic conditions to be distinguished.

Climate change was imposed using the delta method, as 30-year mean differences in atmospheric and ocean state between a future period (2066–2095) and a reference period (2006–2035), both taken from the GFDL-ESM2M global climate model. These seasonal perturbations were applied to the 2012–2024 baseline of Doubell and James (2026), building directly upon their prior work involving model validation, return water far-field dispersal, and larval tracking.

The resulting climate deltas (Figure 3-1 to Figure 3-7) agree with the literature and with recent evaluations of multiple global climate projections (CSIRO and Bureau of Meteorology, 2015; Grose et al., 2020; Goyal et al., 2021a, b). Increases in evaporation, sea surface temperature and salinity are projected year-round over the Lincoln Shelf towards the end of the century. These trends represent the large-scale forcing transferred to the gulf by the dynamical downscaling.

The Spencer Gulf model reproduces the principal features of long-term, regional circulation described in the literature: a clockwise barotropic circulation; the classical inverse-estuarine exchange, with inflow of relatively fresh water along the western shore and outflow of dense, saline water along the eastern; and a stratified, vertically sheared circulation near the mouth, giving way to a well-mixed column in the north. The projected changes under RCP8.5 scenario are not spatially uniform but follow a north–south pattern set by the forcing and circulation, with the largest changes in the north, where the desalination plant is proposed. The Gulf's regional seasonality was not examined in detail, because this study focussed on long-term mean changes and return water effects in Upper Spencer Gulf.

Modelled climate-driven increases in bottom salinity range from approximately 0.3 PSU (0.8% increase) over the current ambient salinity in the southern Spencer Gulf to more than 0.8 PSU (2% increase) in Upper Spencer Gulf, while temperature rises by 1-2 °C. At the Mullaquana Station outfall, the seasonal climatology of bottom salinity increases ranged from 0.37 to 0.62 PSU (1-1.7% increase) over the present-day ambient salinity and increases were greatest in

summer. Bottom temperature was predicted to increases by 1.4–2.2 °C with the opposite seasonality to salinity (lowest in summer, highest in winter) and an annual mean of 1.8 °C.

Due to this non-uniform warming and salinification, water density rises in the upper eastern gulf and falls in the south-west. The gulf's natural clockwise, inverse-estuarine exchange is maintained but enhanced, where vertical shear increases and both the mean annual surface inflow and bottom outflow at the mouth become stronger. At Mullaquana Station, the barotropic flow keeps its direction; however, with the along-gulf pressure gradient increasing, inflow is likely to intensify slightly along the western side to compensate for mass loss in the east. North of Franklin Harbour, tides drive mixing and northward advection, resulting in a vertically homogeneous response (Figure 3-9 to Figure 3-11).

Under a future climate scenario with the desalination (RCP85_DESAL), the predicted bottom salinity increase from return water is less than 0.5 PSU (<1.3 % change over the future ambient salinity) and is predominantly confined within 2–8 km of the outfall (Figure 3-13,Figure 3-14). Consistent with present-day plume modelling (Doubell and James 2026), the plume moves along the bottom towards the north-east and is diluted by advection and mixing to less than 0.03 PSU (<0.1% change over future ambient conditions) within approximately 5 km from the outfall.

Projected future regional circulation changes suggest the plume will be advected further north, marginally raising the background salinity in that direction due to downstream mixing. At the outfall location, the total increase from the two processes is linear and additive. Net temperature and salinity increases exceed 1.4 °C and 1 PSU relative to current conditions, with the RCP85 climate scenario contributing up to 60% more to the total change than the modelled desalination scenario itself (Figure 3-15). The combined bottom salinity increases are strongly seasonal, peaking in summer and autumn, while dropping to minimums in winter.

The climate-downscaling results show that temperature and salinity increases, and associated changes density and circulation, vary across Spencer Gulf. Under the upper-bound RCP8.5 scenario, the largest increases in the gulf are in the Upper Spencer Gulf region, including the Mullaquana Station area. The scenario for the desalination plume further increased mean salinity in the region, but by less than 0.5 PSU near the outfall and with substantially smaller increases at distances further away from the outfall. In the combined future-climate and desalination scenario, the mean increase at the outfall remains below 1.2 PSU in bottom salinity (<3% of present-day ambient salinity) and with a 2.1°C rise in temperature. At the Mullaquana station, modelled temperature increases are greatest in winter, whereas salinity increases are greatest in late-spring and summer. If the future climate tracks closely to high-

emissions pathways, the modelled regional salinity increases from climate change are expected to exceed the local far-field increment attributable to return water discharge based on the examined scenario.

These conclusions are bound by the study's objectives and experimental design. The downscaled climate projection is based on a single global model (GFDL-ESM2M) and a single high-emissions scenario (RCP8.5) and is best interpreted as a high-emissions sensitivity scenario rather than a most-likely projection. Because current emission trajectories make this scenario unlikely, the 8.5 W m^{-2} pathway was removed from the CMIP7 scenario framework in 2026 (Van Vuuren et al. 2026). Consequently, the climate signal applied in this study represents a strict upper- bound, and it is important to note that under a lower-forcing pathway, return water could account for a larger share of the combined salinity change at the outfall, although this was not assessed here. A different climate model or emissions scenario may change the magnitude of the response. The GFDL-ESM2M (CMIP5) model provided a reliable tool for simulating the Earth's climate with projections consistent with newer models from CMIP6 (Grose et al., 2020). Hence, the results here can be considered a first step towards more robust modelling efforts in the region, and the first evaluation of a desalination return water in Spencer Gulf under projected climate conditions.

By design, the climate-delta method retains present-day sequence of interannual and synoptic variability while imposing projected changes in the mean seasonal cycle. Consequently, future changes in the forcing variability are not represented. Furthermore, the desalination return water was also simulated conservatively, fixed in time, and with a higher capacity than planned. The predicted far-field footprint is also therefore a conservative upper bound of the return water impact. Although surface relaxation to the observed temperature and salinity was deliberately omitted to avoid damping the projected climate signal, developing bias-correction methods that preserve that signal could improve future downscaling efforts.

In summary, based on the oceanographic modelling of Spencer Gulf undertaken and the stated conservative assumptions, the modelled climate response indicated the Upper Spencer Gulf as the region most sensitive to future climate driven environmental shifts, with increases expected in both salinity and temperature. Under the conservative high-emissions RCP8.5 downscaled future climate scenario, the overall salinity impact of the desalination return water remained small and equivalent to approximately 1% increase in ambient salinity under both present-day and the future climate scenario conditions. At the outfall, the salinity increases from climate change and from the desalination return water were additive, with climate change contributing the dominant share of the total change. The simulated combined effect predicted salinity increases below 3% above the present-day ambient salinity at the site. Under a lower

emissions climate scenario, it is likely that the desalination return water would contribute proportionally more. Importantly, Spencer Gulf's characteristic clockwise circulation persists and strengthens slightly under the future climate scenario, consistent with the inverse estuarine exchange, where denser water exits along the eastern side, while relatively fresher water enters along the western side.

In conclusion, the presented dynamical downscaling modelling predicted that the far-field salinity increases from conservative return water inputs remained small under the present-day and future RCP8.5 climate scenarios. To our knowledge, this is the first application of dynamical ocean downscaling used to inform an Australian desalination proposal.

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Appendix G-15:
Invasive Marine Species

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Invasive Marine Species Risk Assessment for a Desalination Plant at Mullaquana Station, South Australia



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PURPOSE

This Report is a non-exhaustive summary of Risk Assessment criteria towards the minimization of risk of a marine pest being detected or establishing at Mullaquana Station SA during a construction phase or operating phase of a desalination plant. Using and applying some or all of the information contained within this report does not guarantee a pest will not colonise the Mullaquana Station location or any other location. The risk profile generated by this report is subject to change and is not perpetual.

Whilst all due care and diligence has been exercised in the collection of data for and the preparation of this report, Aquatic Biosecurity Pty Ltd purports to provide a collection service only for this report, based on the opinion and experience of the individual consultant responsible for its compilation. Aquatic Biosecurity Pty Ltd issues this report in good faith and without prejudice. Anyone wishing to rely on such opinion should first satisfy themselves as to its accuracy and feasibility. Aquatic Biosecurity Pty Ltd shall not be liable for any loss (including indirect and consequential loss), damage, delay, and loss of market, costs, and expenses of whatsoever nature or kind and however sustained or occasioned with respect to the information in this report.

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On the cover: Asian Greenlip mussels *Perna viridis* (PIRSA Marine Biosecurity).

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1. Summary

The Northern Water Project (the Project) aims to provide a secure climate resilient water source for industry in South Australia's Upper Spencer Gulf and Far North. Delivery of the Project will involve the construction and operation of a 51 gigalitre per year (GL/year) seawater reverse osmosis desalination plant located at Mullaquana Station on the Upper Spencer Gulf, and associated water transfer system (including pipelines, transfer pump stations, and product water storages) and electricity infrastructure.

Minimisation of risk, including the risk posed by Invasive Marine Species (IMS) is a commitment of the Project.

Aquatic Biosecurity Pty Ltd was engaged by JBS&G to conduct a desktop Risk Assessment of current and future IMS risks to the Project site at Mullaquana Station, Spencer Gulf, South Australia with the objective to establish baseline IMS risk assessment data for the Project.

This document discusses the vector categories, the vector translocations, where potential risks exist and criteria for minimizing risk. Background information is discussed using the following topics:

- The Australian Priority Marine Pests List (APMPL)
- Existing Marine pest and disease records for Mullaquana Station
- Potential connection of infected ports and nodes to the Project site
- Potential vectors from the construction phase of the Project
- Potential vectors from the operational phase of the Project, and
- Potential risk to local aquatic-based stakeholders.

The Risk Assessment then adapts the five-step process undertaken by Hewitt et al. (2011) S*pecies Biofouling Risk Assessment. The National System for the Prevention and Management of Marine Pest Incursions*, whilst including new developments in IMS pest management, research and elements of the National System for the Prevention and Management of Marine Pest Incursions (2011).

The Hewitt et al. (2011) risk assessment process:

1. discusses vectors
2. identifies endpoints
3. identifies hazards
4. determines consequences
5. determines likelihood
6. calculates risk.

As the purpose of this Risk Assessment is focused on the construction and ongoing operations of a desalination plant at Mullaquana Station, risk was limited to and evaluated across two end points; IMS detection and IMS establishment at Mullaquana Station. The vector's consequence and

likelihood were described where they relate to vessel types, vessel characteristics and donor locations. Hazards were identified as species, although risks and consequences for most species are effectively unpredictable except for those which have already spread across South Australia. Colonisation of marine species has historically been difficult to predict, as different habitats, population dynamics and stakeholders exist (Hayes et al. 2019). As such species-specific risks and consequences are not considered as part of this risk assessment.

Based on Hayes et al. (2019), all vessels pose a risk for translocating IMS through biofouling which is included as a principle of the current Risk Assessment. International voyages to Australia were discussed for the risk they pose in relation to detection on a vessel and establishment, as both factors represent a risk to the Project. They are reasoned to be given a lower risk score than close domestic journeys of IMS due to decreased propensity for IMS survival, stress from high vessel speed, less propensity for commonality of habitat and lower regulatory compliance for domestic journeys versus international journeys. Absence of local predators for an IMS is considered consistent whether an IMS is coming from overseas or from one Australian location to another.

The highest risk vessels that would operate as part of the Project are reasoned to be large, slow, in water berthed vessels, on short domestic journeys including dumb barges, jack-up barges, lighters, dredges, mobile drilling rigs and seismic survey ships possibly coming from interstate or overseas. Trailerable vessels that are cleaned regularly with high pressure freshwater, detergent or descalant then left in the sun for UV exposure are reasoned to represent the lowest risk to the Project.

It is recommended that all vessels require an individual risk assessment before a voyage to Mullaquana Station to minimise Project risks. Current risks are determined to be low where vessel types are lower risk types (e.g. trailerable vessels), journeys are low risk (moving from areas where IMS populations are low or nonexistent) although the local environment at Mullaquana Station has been disturbed by industry, which aids to establishment.

Mitigation techniques to reduce risk involve but are not limited to:

- individual vessel biofouling risk assessments according to the appropriate set of National Biofouling Guidelines;
- vessel inspections by qualified biofouling inspectors before vessels leave donor locations to ensure clean hulls;
- journey tracing to minimise idle time in donor locations for in-water vessels; and
- setting up quarantine zones for checks on vessels before entry to the Mullaquana Station location.

It is a recommendation of this study that all vessels should be confirmed to have clean hulls, appropriate biosecurity hygiene maintenance applied and regular checks of ballast water and biofouling management plans where appropriate based on vessel type.

2. Introduction

2.1 Northern Water Project

The Northern Water Project (the Project) aims to provide a secure climate resilient water source for industry in South Australia's Upper Spencer Gulf and Far North (Figure 1). Delivery of the Project will involve the construction and operation of a 51 gigalitre per year (GL/year) seawater reverse osmosis desalination plant located at Mullaquana Station on the Upper Spencer Gulf, and associated water transfer system (including pipelines, transfer pump stations, and product water storages) and electricity infrastructure.

The Project comprises the following key elements:

- a 51 GL/year seawater reverse osmosis desalination plant located at Mullaquana Station approximately 25 km south-west of Whyalla
- marine intake and outfall tunnels extending underneath the seabed and into Spencer Gulf
- diffuser structures for discharge of return water into Spencer Gulf
- a transfer system consisting of:

o a ~360 km main transfer pipeline, largely constructed underground terminating at the delivery point at the Olympic Dam copper-gold mine

o a ~46 km lateral branch to a delivery point at the Carrapateena copper-gold mine (which may be constructed above or below ground)

o associated transfer pump stations and product water storage

o high voltage electricity supply including new transmission and distribution lines and new network substations to provide power to the desalination plant and transfer pump stations

o associated temporary construction-related infrastructure (e.g. construction camps, site offices and amenities, yards, storages and laydown areas)

- access tracks including establishing new tracks and upgrading existing tracks
- enabling roadworks and road upgrades.

The Minister for Planning directed that an Environmental Impact Statement (EIS) was to be prepared for the Project on 23 December 2025. The proponent's commitment is for the minimisation of biosecurity risk which includes the risk posed by Invasive Marine Species (IMS) during marine construction work and operation of the proposed desalination plant.

2.2 Scope and Objectives

Aquatic Biosecurity Pty Ltd was engaged by JBS&G on behalf of the Department for Infrastructure and Transport (DIT) to conduct a desktop IMS Risk Assessment for the Project. The objective of the Risk Assessment is to identify potential IMS risks from the Project on the marine environment and on the Project itself.

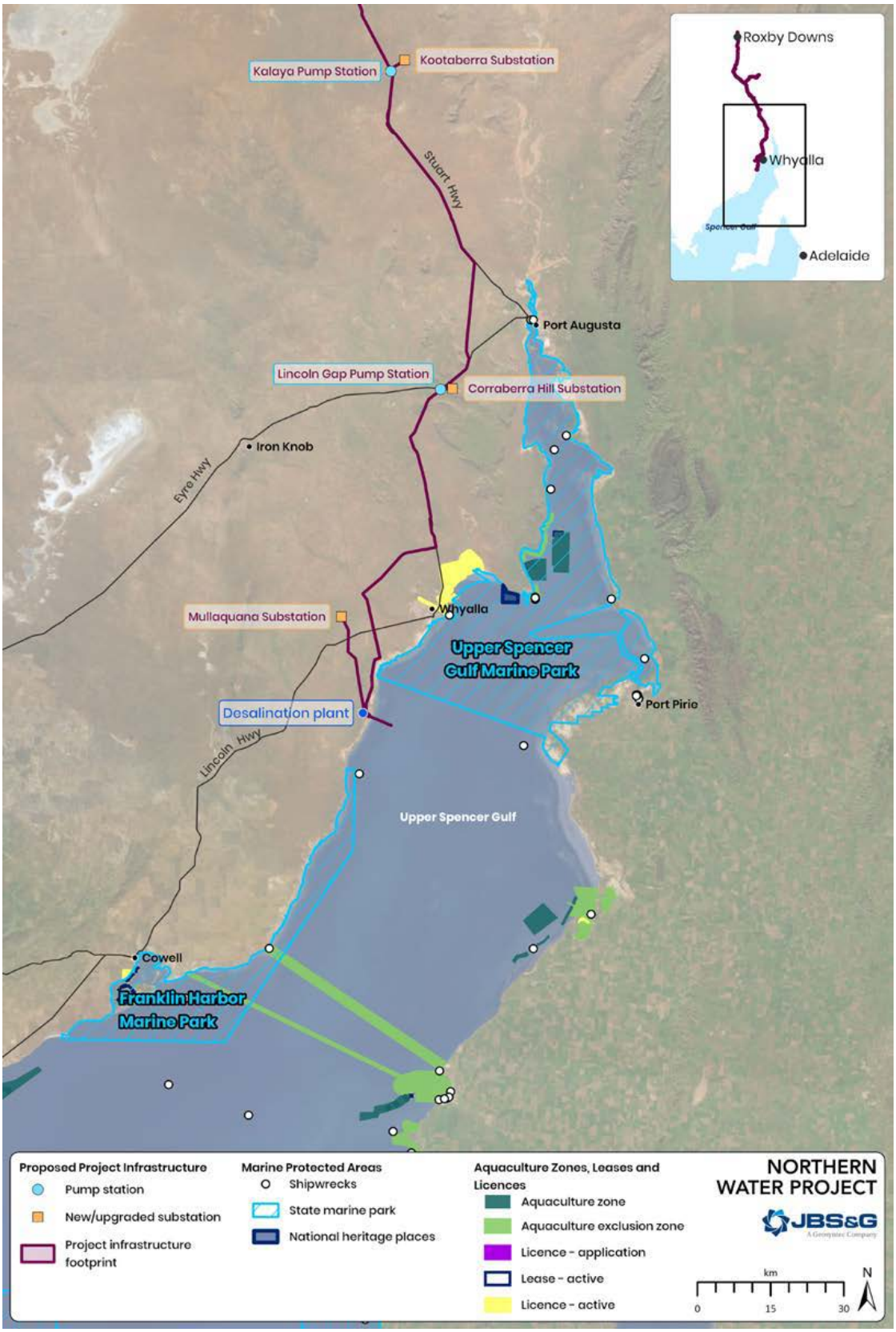


Figure 1: Location of the desalination plant in Spencer Gulf, South Australia (Source: JBS&G)

3. Invasive Marine Species Background

3.1 Invasive Marine Species

IMS are defined as plants or animals that are not native to Australia but have been introduced to Australian waters by human activities including cargo shipping, cruise ship tourism, oil rig operations and recreational yachting. IMS can outcompete existing native communities and smother benthic habitat. They can cause macrophyte loss resulting in sediment destabilization, sediment loss and subsequent bathymetry changes (James et al. 2020). IMS pose a significant risk to any marine capital development as they can cause severe environmental, economic and social damage. It is important to control IMS risks during the construction and operation phase of any project to ensure progress is on schedule and to prevent any effects on stakeholders. It is equally important to prevent IMS entering the site in the future where large-scale effects can occur.

Disturbed habitats, including marine construction sites, are particularly suitable settling habitat for IMS (Altman and Whitlatch, 2007). Areas where marine pests have successfully established may become an ongoing pest risk and are sources for domestic translocation to other locations within Australia and overseas. There are numerous documented examples of IMS colonisation from overseas. An example is the introduction of the Atlantic comb jelly *Mnemiopsis leidyi* into the Black Sea, which caused the collapse of the local Kilka anchovy (*Engraulis encrasicolus)* fishery and had widespread impacts on other fisheries in the region (Shiganova, 1998) and factored in the overall collapse of the Black Sea ecosystem. Anthropogenic threats analysis has identified IMS as a key threat to marine fauna in South Australia's Spencer Gulf (Robbins et al. 2017). Therefore, it is important to control IMS risks during all phases of the Project to prevent impacts to the environment, stakeholders, to the Project development and operation.

3.2 IMS vectors

IMS are introduced to a new environment via vectors. Vessels represent the major IMS vectors, whereby they can be transported through the vessel's ballast water and via biofouling on vessel surfaces and in niche areas. These main pathways of IMS introduction and spread are detailed below.

3.2.1 Ballast Water

Ballast water is carried in tanks built into the hulls of ships to improve stability, balance, and structural integrity. Ships draw ballast water from one location and discharge it at another (Figure 2). This occurs across different locations, regions or countries. The process is essential for the safe and efficient operation of vessels.

Ballast water poses several significant risks including invasive species and diseases. Ballast water may contain a variety of marine organisms, including bacteria, plants, algae, animals and the diseases they carry; e.g. Pacific Oyster Mortality Syndrome (POMS), Abalone Viral Ganglioneuritis (AVG), and human diseases (e.g. Cholera) (Karim and Karim, 2015). When discharged into a new environment, IMS can be introduced, outcompeting native species, disrupting local ecosystems and causing ecological damage.

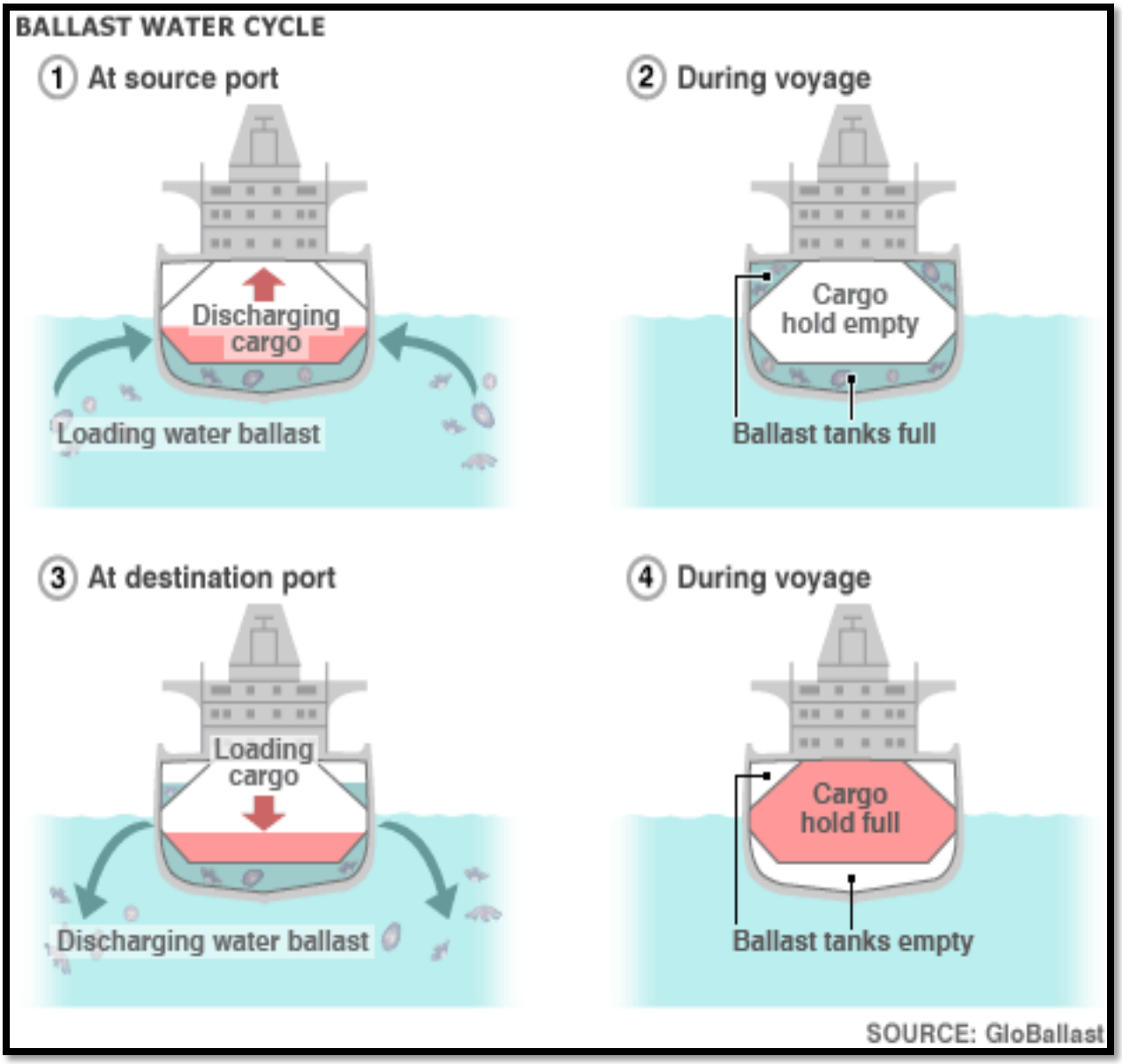


Figure 2: Ballast Water diagram from GloBallast (2024)

3.2.2 Biofouling

Marine biofouling is the accumulation of microorganisms, plants, algae, or small animals on submerged surfaces, such as the hulls of ships, underwater structures, pipelines, and aquaculture equipment (Figure 3). Typically, any substrate placed in natural waters is quickly colonized by micro-organisms through the creation of a biofilm (or slime layer). This is followed by a succession of diverse sessile or sedentary micro- and macro-organisms (Flemming, 2002; Aldred and Clare, 2008; Amara et al. 2018).

Vessel biofouling is acknowledged as a major, and perhaps the most important, pathway for the introduction, establishment, and subsequent spread of marine non-indigenous macro-organisms (Drake and Lodge, 2007; Hewitt and Campbell, 2010; Bell et al. 2011; Ruiz et al. 2015; Bailey et al.

2020; Georgiades et al. 2021). In addition to potential impacts to the marine environment, biofouling can also promote microbial-induced corrosion, which can weaken metal structures and pose safety risks (Knisz et al. 2023).

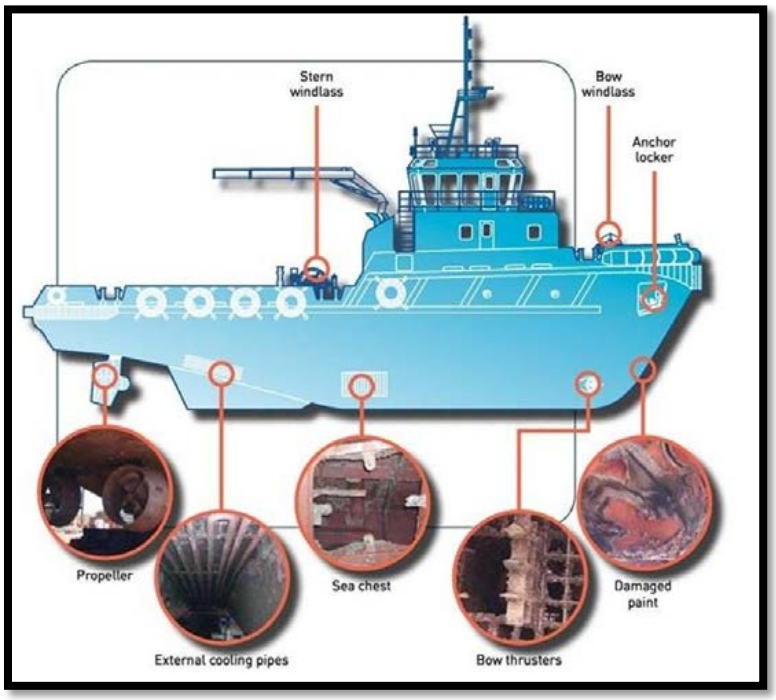


Figure 3: Niche areas where biofouling can accumulate on a non-trading vessel (MPSC 2018)

3.3 IMS nodes

Nodes are defined as the locations where IMS have the potential to occur. Nodes can be where an IMS may establish, or from where it may be transported. IMS commonly colonise disturbed artificial habitats more readily than native species due to their wider environmental tolerances, higher growth and / or fecundity rates and lack of local predators. Artificial marine habitats can be considered as nodes where IMS translocation has the potential to occur. They can be broad areas such as ports and marinas, or localised marine structure such as wharves, jetties, submerged equipment, underwater pipes and moorings.

3.4 Management of IMS in Australia

Management of an IMS in Australia is dependent on its state of invasion, and ranges from prevention to eradication and containment. Risk assessment is a key tool for the prevention of IMS incursions. Australian Governments have collectively agreed that prevention is the preferred option to manage pests and diseases for both socio-economic and environmental reasons as illustrated in Figure 4 (below).

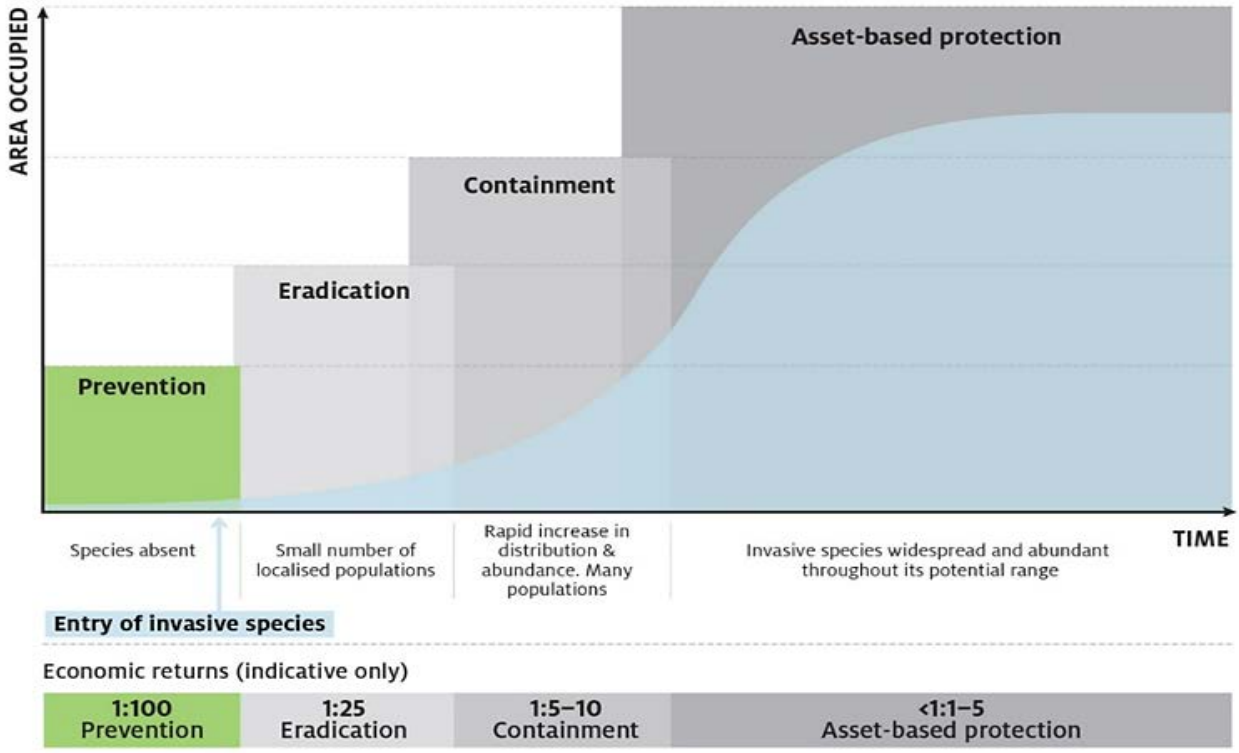


Figure 4: Marine Pest Invasion Curve (MPSC 2019)

3.4.1 Prevention

Australia's international arrangements to manage IMS are coordinated by the Australian Quarantine Inspection Service (AQIS) using the Maritime Arrivals Reporting System (MARS). The compliance regime coordinated by AQIS, designed to ensure that Australia is exposed to minimal IMS risk. Ballast water and biofouling compliance arrangements are pre-approved for arrival to Australia accordingly with existing international guidelines and conventions (refer Section 3.5.1).

The ballast water and biofouling risks of internationally operating commercial vessels are managed by AQIS using the Maritime Arrivals Reporting System (MARS) portal. The risks of international and domestic ballast water are managed by vessel operators by maintaining a ballast water management plan and ballast water record book. The vessel operators will either operate a ballast water treatment system, empty and refill ballast tanks in oceanic waters, or three times volume discharge at defined distances from nearest land. Ballast water exchange should be conducted in at least 200 nautical miles (NM) from nearest land and in waters 200 m deep (DAWE, 2020). For voyages that cannot practically meet these requirements, ballast water exchange must occur at least 12 NM from the nearest land and in water at least 50 m deep (DAWE, 2020). These standards were nationally accepted by Australian Governments to represent a standard that propagules will likely not reach the Australian coast to colonise habitats.

Domestic arrangements for ballast water are managed through MARS, while domestic biofouling is managed on a voluntary best practice basis using biofouling guidelines relevant to the vessel type and industry.

3.4.2 Eradication and Containment

Australian examples of eradication of an established marine pest are few, expensive, and usually in close-able systems. Eradication of the Black striped mussel *Mytilopsis sallei* occurred in Cullen Bay Marina (Darwin) in 1999 and cost AUD $2M (Wells, 2019). Over $5M was spent on the eradication of the alga *Caulerpa taxifolia* in West Lakes (South Australia) and associated research operations (Cheshire et al. 2002). Some eradications in more open systems have been successful but are considered rare, and include the eradication of *Caulerpa taxifolia* at North Haven Boat Ramp (South Australia) in 2008 (Fletcher, 2008), and the eradication of the New Zealand Greenlip Mussels *Perna canaliculus* in the Port River (South Australia) in 1996 (Wiltshire et al. 2010).

Any significant IMS detection, whether it is detected on a visiting vessel or established in a new location, can result in an emergency quarantine zone being set up around that location for delimitation survey inspections and vessel tracing. Depending on the species found, and its relationship to Australia as a pest or not, vessels can be seized and nodes locked down in quarantine for extended periods.

3.4.3 Quarantine Emergencies and Consequences

In an IMS emergency, time costly delimitation inspections on all vessels, nodes and local assemblages of the affected area take precedence over any vessels moving in or out of a quarantine zone. This can also result in critical time delays for non-affected vessel arrivals and or departures.

Concessions are made by the Australian Government for IMS affected vessels to leave Australian waters to at least 12 NM from the territorial sea baseline (DAFF 2023). These arrangements generally remain until vessels have been inspected and risk assessed as a minimum level of management, or remedial treatment actions have been taken. These delay aspects have caused cost over-runs of tens of millions of dollars in a previous Australian marine project e.g. Barrow Island Asian Greenlip Mussels (AGM) *Perna viridis* quarantine (Mundi, 2020) and large numbers of vessels have been quarantined, e.g. AGM in Trinity Inlet QLD (NIMPCG, 2015); Black Striped Mussel in Cullen Bay Marina, Darwin (Wells 2019).

Delay costs are significant when IMS detections occur, however they are relatively low compared to the ongoing costs of fouling aquatic pests that could threaten the operation of a desalination plant. For example, between 1993 and 1999, freshwater mussel infestations (quagga *Dreissena bugensis* and zebra *D. polymorpha*) in the Great Lakes region cost the power sector US $3.1 billion (Power, 2019) and introduced golden mussels *Limnoperna fortunei* in the Amazon basin are estimated to be costing $US120M per year (Moutinho, 2021). The Asian Green Lipped mussel and several other bivalves of invasive risk to Australia can similarly clog marine cooling pipes and intake systems (Government, 2024).

3.5 Laws, Guidelines and Policies

International, Commonwealth, State and Territory regulations and guidelines can be used in federal and state waters to manage biosecurity issues. These include:

3.5.1 International

- International Convention for the Control and Management of Ship's Ballast Water and Sediments (2004) (Ballast Water Convention - BWM)
- International Convention on the Control of Harmful Antifouling Systems on Ships (IMO, 2001)

- Guidelines for the control and management of ships' biofouling to minimize the transfer of aquatic species (IMO Biofouling Guidelines, 2023)

3.5.2 Commonwealth of Australia

- *Biosecurity Act (2015)* and *Biosecurity Regulations* (2016)
- *Environment Protection and Biodiversity Conservation Act 1999*
- Australian Ballast Water Management requirements (2020)
- Marine Pest Plan, Department of Agriculture, Water and the Environment (DAWE) (2018 – 2023)
- Australian biofouling management requirements, Department of Agriculture, Fisheries and Forestry (DAFF)(2023)
- Anti-Fouling and In-Water Cleaning Guidelines for Australia and New Zealand (2015)
- National biofouling management guidelines for the aquaculture industry (2009)
- National biofouling management guidelines for commercial vessels (2009)
- National biofouling management guidelines for commercial fishing vessels (2009)
- National biofouling management guidelines for non-trading vessels (2009)
- National biofouling management guidelines for marinas, slipways, boat maintenance and recreational boating facilities (2021)

3.5.3 South Australia

- *Biosecurity Act (2025)*
- *Fisheries Management Act (2007)*
- *Environment Protection Act (1993)*
- *Native Vegetation Act 1999 - Biodiversity Act (2025)*

4 Existing environment

4.1 Project site

The desalination plant at Mullaquana Station is located east of the Lincoln Highway approximately 25 km south-west of Whyalla. The land at Mullaquana Station is currently utilised for sheep grazing. The Upper Spencer Gulf Marine Park is the closest marine protected area to the proposed desalination plant marine infrastructure, located approximately 8 – 10 km north-east. The Franklin Harbour Marine Park is located approximately 11.5 km south of the proposed desalination plant marine infrastructure. The Cowleds Landing Sanctuary Zone and an adjoining Habitat Protection Zone are located approximately 8 – 10 km from the desalination plant outfall diffuser structure (refer to Figure 1).

4.2 Physical environment

The Project site is in the shallow, sheltered waters of the upper north-western reach of Spencer Gulf. The site is relatively sheltered from Southern Ocean swells due to the southerly headlands of Mitchelville to the east and the tip of Yorke Peninsula to the west whereby it experiences low wave energy (EPA-SA 2018). Salinity ranges from 39 PSU to 41 PSU at Mullaquana Station. Seawater temperature follows a seasonal trend ranging from around 13°C in late winter to around 24°C in later summer/early autumn (JBS&G 2026).

4.3 Benthic communities and habitats

The aquatic area is characterised by sandy beaches, mangroves, salt marshes and sand flat habitats at the shore including macro algae and seagrass in the waters beyond (e.g. *Posidonia* sp, *Amphibolis*, *Halophila*, *Zostera*). Recent surveys of seagrass and water quality around the Mullaquana Station area (the Yonga Nearshore Marine Biounit) undertaken by the EPA SA (2018) show the marine environment to be declining from good in 2012 to 'poor to fair' condition in 2018.

4.4 Marine pest records for Mullaquana Station and nearby locations

Desktop searches from Google, Researchgate, Google Scholar and Flinders University Library suggest there are no records of marine pests at Mullaquana Station however a recent study by Brooks (2025) found European fanworms *Sabella spallanzanii*. A study undertaken by the South Australian Research and Development Institute (SARDI) found no records of IMS at Mullaquana Station up to 2010 (Wiltshire et al. 2010) although a 2025 study also found *S.spallanzanii*. The European Fanworm *Sabella spallanzanii* is a resident of adjacent Whyalla waters (Wiltshire 2010) as is the sea vase *Ciona Intestinalis* (Wiltshire et al 2017). The worm species *Myxicola infundibulum* has been recorded at Whyalla and Port Bonython based on unpublished BHP Billiton data (Wiltshire 2010, Arup and ENSR 2008). *Magallana (Crassostrea)) gigas* (Pacific oyster) adults and DNA of *Corbula gibba* (European clam) (detected by qPCR) have also been recorded at Whyalla (Wiltshire et al 2017).

As part of the current Project EIS, a molecular DNA study by SARDI, analysing plankton tows, targeting 20 IMS species (Table 1) Testing for IMS was conducted on 60 samples from the February 2025 and July 2025 surveys using qPCR assays targeting 20 priority species. A single detection of the European Fan Worm, *Sabella spallanzanii* was recorded in a February night sample from the proposed desalination site, with no other IMS detected.

Molecular DNA survey plankton tows have also been conducted nearby in Spencer Gulf by Wiltshire et al. (2017) and the results have also been used to inform this study in presence and absence of those pest species.

S.spallanzanii is the only IMS identified at Mullaquana Station however there are several at the site of Whyalla (Wiltshire *et al* 2010). Marine pests are known at nearby locations and elsewhere in South Australia (Table 1). The Pacific oyster *Magallana gigas*,Sea Vase *Ciona intestinalis* and European Fanworm *Sabella spallanzanii* were detected in Tumby Bay about 222.2 km from Mullaquana Station (Wiltshire et al. 2010). The Asian Date Mussel *Arcuata (Musculista) senhousia* was recorded at Port Spencer (Lipson) on the Eyre Peninsula 92.6 km from Mullaquana Station during an underwater survey (Golder 2012). The European fanworm *Sabella spallanzanii* have also been found in Port Lincoln at the Jetty and Naval Shipworm *Toredo navalis* are resident in the Port Lincoln marina (PIRSA Biosecurity SA response operations undertaken by author in 2009).

Table 1: IMS Targeted Species in SARDI's Molecular Survey of Mullaquana Station[1]

Species	Common name	Present	Assay development reference
Arcuatula senhousia	Asian Bag Mussel	Aus	Wiltshire et al. in review
Perna canaliculus	NZ greenlip mussel	-	(Bott, 2011)
Perna perna	Brown mussel	-	(Dias et al., 2013, Simpson et al., 2018)
Perna viridis	Asian green mussel	-	(Dias et al., 2013, Simpson et al., 2018)
Mytella strigata	Charru mussel	-	(Wiltshire et al., 2021)
Mytilopsis sallei	Black-striped false mussel	-	(Bott.N et al., 2012)
Varicorbula gibba	European basket shell	Aus	Wiltshire et al. in prep
Magallana gigas	Pacific Oyster	SA	(Bott and D, 2012)
Mya japonica	Japanese soft-shell clam	Aus	(Giblot Ducray, 2022)
Maoricolpus roseus	New Zealand screw shell	Aus	(Giblot Ducray, 2022)
Asterias amurensis	Northern Pacific Sea star	Aus	(Bax N., 2006)
Carcinus maenas	European shore crab	SA	(Bott et al., 2010)
Charybdis japonica	Asian paddle crab	Aus	(Simpson et al., 2018)
Rhitropanopeus harrisii	Harris mud crab	-	(Simpson et al., 2018)
Hemigrapsus sanguineus	Asian shore crab	Aus	(Wiltshire, 2021)
Eriocheir sinensis	Mitten crab	-	(Andersen et al., 2018)
Sabella spallanzanii	European fanworm	SA	(Ophel-Keller et al., 2007)
Ciona intestinalis	Vase tunicate	SA	(Bott and Giblot-Ducray, 2011)
Didemnum vexillum	Carpet sea squirt	Aus	(Simpson et al., 2018)
Undaria pinnatifida	Japanese kelp	SA	(Bott et al., 2010)

4.5 Marine Disease Records

There are no marine disease records for Mullaquana Station in the available literature, however there are diseases records from nearby locations. The parasite *Perkinsus olensi* severely depleted the Abalone *Haliotis laevigata* fishery in 1985 to 1986 along the eastern coast of Spencer Gulf (Goggin and Lester, 1995; Lester, 1981). Various other fish kills have occurred in Spencer Gulf and Gulf St Vincent, South Australia. In 1995 and 1998 a herpes virus caused mass fish kills in the Spencer Gulf sardine fishery, with the virus likely to have been introduced from imported untreated sardines

[1] *Aus* indicated species known to occur in Australia; *SA* indicates species known to occur in South Australia; – indicates species exotic to Australia.

Sardinopsis sagax used as aquaculture feed (Ward et al., 2001; Whittington et al. 2008). Infections of abalone with Vibrio spp. have also been implicated in causing mass mortalities in cultured abalone in Australia (Bower, 2003, Dang and Miller, 2016).

Other marine diseases of current concern to Mullaquana Station include Pacific Oyster Mortality Syndrome (POMS) and Abalone Viral Ganglioneuritis (AVG), which are both established in South Australia. AVG has the potential to severely impact aquaculture stocks and the fishery in Spencer Gulf reef systems. Spencer gulf also has aquaculture for pacific oysters at Cowell, a POMS infection could spread further to intensive oyster farming nearby at Coffin Bay.

4.5.1 Pacific Oyster Mortality Syndrome

POMS is a disease affecting Pacific oysters *Magallana (Crassostrea) gigas*. POMS is caused by the Ostreid herpesvirus-1 (OsHV-1), a highly contagious virus that primarily affects juvenile and spat (young oysters). POMS causes high mortality rates in affected oyster populations, leading to severe economic losses for the aquaculture industry. POMS was first identified in France in the 1990s and has since spread to oyster farming regions worldwide, including Australia, New Zealand, and the United States. In Australia, POMS was first recorded in New South Wales in 2010, and later spread to Tasmania and South Australia. In South Australia, POMS is contained to the Port River in Port Adelaide and has not been recorded in the Spencer Gulf (PIRSA, 2024).

POMS outbreaks typically occur seasonally, with disease prevalence linked with elevated water temperatures. The virus can spread rapidly within oyster farming areas through the movement of infected stock, equipment, or via waterborne transmission. Efforts to manage POMS typically involve implementing biosecurity measures to prevent the introduction and spread of the virus, such as strict hygiene protocols, quarantine procedures, and surveillance programs.

4.5.2 Abalone Viral Ganglioneuritis

AVG is a highly contagious viral disease that affects abalone. AVG is caused by the abalone herpesvirus (AbHV) (Dang and Miller, 2016). The virus primarily affects Blacklip Abalone *Haliotis rubra* and Greenlip Abalone *Haliotis laevigata*, although other species may also be susceptible (Dang and Miller, 2016).

The symptoms of AVG can vary depending on the severity of the infection, but typically include loss of coordination, abnormal swimming behaviour and, in advanced stages of the disease, neurologic signs such as paralysis. Mortality rates can be high in infected populations, especially in abalone farms where the virus can spread rapidly among densely stocked animals (Dang and Miller, 2016).

AVG poses significant challenges to the abalone aquaculture industry, as outbreaks can result in substantial economic losses due to mortality and reduced production. AVG can also impact wild abalone populations, leading to concerns about its potential effects on marine ecosystems and commercial fisheries.

AVG was first reported in wild stocks in Victoria, Australia, in 2005, and has since been detected in other regions where abalone aquaculture or wild abalone populations exist (Dang and Miller, 2016). AVG has been detected in Portland in 2021 and there are current management measures in place aiming to decrease the risk of further spread. AVG was detected in Port MacDonnell in 2024 and has caused considerable damage resulting in a closure of the Southern Zone Abalone Fishery (PIRSA 2025)

5 Risk Assessment Procedure

The consequence and likelihood of IMS translocation is dependent on the existing vulnerability of the receiving environment and the potential for the proposed marine activities to facilitate IMS translocation. These factors are identified and used to inform the risk assessment in this section.

5.1 Resources at risk

5.1.1 Environmental and socio-economic resources

Mullaquana Station is not located in a regulated State or Australian Marine Park but it is adjacent to several. The Upper Spencer Gulf Marine Park is the closest marine protected area to the proposed desalination plant marine infrastructure, located approximately 8 – 10 km north-east. The Franklin Harbour Marine Park is located approximately 11.5 km south of the proposed desalination plant marine infrastructure. The Cowleds Landing Sanctuary Zone and an adjoining Habitat Protection Zone are located approximately 8 – 10 km from the desalination plant outfall diffuser structure (refer to Figure 1).

Other protected areas in the region include the Franklin Harbor Marine Park ~ 120.4 km to the south west.

Several threatened migratory marine species have previously been recorded in the wider Spencer Gulf, including the Nationally Endangered Southern right whale *Eubalaena australis*, the Nationally Vulnerable White shark *Carcharodon carcharias* and the South Australian Rare Bryde's whale *Balaenoptera brydei* (DEW, 2024). Bottle nose dolphins, migratory shorebirds and resident waterbirds such as pelicans, cormorants and terns use the local habitat for food, shelter and breeding.

Spencer Gulf provides resources to recreational and commercial fisheries and aquaculture sectors. Stakeholders in the region include:

- Abalone Fishery
- Blue crab Fishery
- Charter boat Fishery
- Commonwealth Southern Eastern Scalefish
- Marine scalefish Fishery
- Mussel Aquaculture
- Oyster Aquaculture
- Prawn Fishery (including Calamari by-catch)
- Recreational Fishery
- Rock Lobster Fishery
- Sardine Fishery
- Shark Fishery
- Southern bluefin tuna Aquaculture

- Yellowtail kingfish Aquaculture.

5.1.2 Project infrastructure and operation

Detection of IMS at Mullaquana Station has the potential to cause significant delays during construction and operation. IMS detection during construction may trigger timely and costly inspections and quarantine of Project infrastructure and vessels, and potentially, the implementation of eradication programs. Delays by these management measures are likely to result in significant costs.

In addition to costly inspections and quarantine measures, an establishment of IMS at Mullaquana Station also has the potential to impact on the Project operation through biofouling of Project's pipework, intake and diffusers. Whilst desalination plants possess internal treatment systems to reduce fouling in pipework, it is more difficult to prevent fouling obstructions at the intake screens and the diffusers. Maintenance cleaning requirements are commonly required in the case of marine bivalve pests due to their high growth rates and fecundity.

Preventing marine pest incursions is essential to ensure public access to marine waters and marine facilities continue to operate (MPSC, 2021).

5.2 IMS of concern to Mullaquana Station

Ten marine pests have been deemed national priorities to Australia by the Marine Pest Sectoral Committee and constitute the Australian Priority Marine Pest List (APMPL) (Table 2). The APMPL was established to facilitate national improvements in marine pest communication, surveillance, preparedness, response and management. Characterisation of these species as priority pests was based on their potential or documented impact on the environment, human health and socio-economic and cultural values (PSC, 2018).

Three of the ten APMPL species are already established in Australia, and therefore have a higher likelihood of being introduced to Mullaquana Station. The European shore crab *Carcinus maenas* and the Japanese kelp *Undaria pinafitida* have been recorded within South Australia, while closest records of the Northern Pacific seastar *Asterias amurensis* are in Victoria and Tasmania (Byrne et al. 1997; Boisvert, 2023) (Table 2). These three species can be entrained and released via shorter voyages than the other APMPL species, survival and establishment at the Project site is more likely.

APMPL species that have not yet been introduced to Australia are also of particular concern, as their detection near the Project site would automatically generate an emergency quarantine response from the government with consequent costs and delays to the Project development.

Other IMS of concern to Mullaquana Station are species that have the potential to establish at the Project site based on their environmental tolerance ranges. Table 2 also lists the IMS that are not listed under the APMPL, but which have previously been identified as target species for a marine pest survey in Port Adelaide (Wiltshire and Deveney, 2011).

IMS listed in Table 2 that commonly occur in South Australia, such as the Sea Vase *Ciona intestinalis*, the European fanworm *Sabella spallanzanii* (already present at Mullaquana Station) or Pacific Oyster *Magallana gigas*, would not usually constitute a critical marine pest emergency as they are widespread in South Australia. Nevertheless, their potential introduction and impact to the local environment could affect business reputation for future projects.

Planktonic toxic dinoflagellates that could colonise the area, have been excluded from Table 2 below as they have been accepted to be effectively unregulatable by the former governing committee, the National Introduced Marine Pests Coordination Group (NIMPCG) when Australia ratified the Convention for the Control and Management of Ballast Water and sediments (BWM) convention in 2016.

Table 2: Pest Species of Perceived Risk to Mullaquana Station (non-exhaustive) (from Wiltshire et al. (2010), Wiltshire and Deveney (2011), NIMPIS (2024) and iNaturalist (2025)

Common Name	Species Name	Closest established locations in Australia[2]
APMPL species already established in Australia		
European shore crab	*Carcinus maenas*	Gulf St Vincent (SA)
Japanese kelp	*Undaria pinafitida*	Cape Jaffa (SA) and Robe (SA)
Northern Pacific seastar	*Asterias amurensis*	Port Phillip Bay (VIC) and Hobart (Tas)
APMPL species not known to be established in Australia		
Asian green mussel	*Perna viridis*	Not established in Australia
Black striped false mussel	*Mytilopsis sallei*	Not established in Australia
Brown mussel	*Perna perna*	Not established in Australia
Charru mussel	*Mytella strigata*	Not established in Australia
Chinese mitten crab	*Eriocheir sinensis*	Not established in Australia
Harris' mud crab	*Rhithropanopeus harrisii*	Not established in Australia
New Zealand green-lipped mussel	*Perna calaniculus*	Not established in Australia
Non APMPL-listed marine pests[3]		
American slipper limpet	*Crepidula fornicata*	Not established in Australia
Asian basket clam	*Corbula amurensis*	Not established in Australia
Asian shore crab	*Hemigrapsus sanguineus*	Port Phillip Bay (VIC)
Asian paddle crab	*Charybdis japonica*	Detected but not established in Fremantle (WA), St Kilda and Outer Harbor (SA)
Rapa or veined whelk	*Rapana venosa*	Not established in Australia
Sea vase	*Ciona intestinalis*	Tumby Bay (SA) Most South Australian ports and marinas including Port Lincoln (SA) and Whyalla (SA)
Dead mans fingers	*Codium fragile spp. tsomentoides*	North Haven (SA) and West Lakes System (SA)
Eastern Atlantic Clam	*Varicorbula gibba*	Detected in Port Adelaide in 2008 (Wiltshire 2008) but not established. Detected in Whyalla (2010) but not established.
European fanworm	*Sabella spallanzanii*	Most South Australian ports and marinas including Whyalla (SA). Port Lincoln (SA) and Port Augusta (SA)

[2] Data from NIMPIS (2024)

[3] Listed on www.marinepests.gov.au/pests/map; Listed on I-Naturalist for Whyalla (2025)

Common Name	Species Name	Closest established locations in Australia[2]
Asian date mussel	*Arcuata senhousia*	Detected in Port Adelaide (SA) but not established. Established at Sheep Hill (Port Spencer), Lipson (SA).
Red macroalga turfing algae	*Polysiphonia brodiei*	Established in Hobart (TAS)
Lace Coral Encrusting bryozoan	*Watersiporia spp. arcuata, W.subtoquata*	Most South Australian ports and marinas Including Port Adelaide (SA) and Port Lincoln (SA)
Pacific oyster (for disease and fouling risk)	*Magallana (Crassostrea) gigas*	Established in Port Lincoln (SA) and Cowell (SA) Industry
Atlantic Jacknife Clam	*Ensis directus*	Not established in Australia
Soft Shell Clam	*Mya arenaria*	Not established in Australia
Polychaete tubeworm	*Hydroides dianthus*	Not established in Australia
Colonial ascidian	*Didemnum spp.*	Established at Garden Island (WA) and Henderson (WA) and Sydney (NSW)
Comb jelly	*Beroe ovata*	Not established in Australia
Sea Walnut Comb jelly	*Mnemiopsis leidyi*	Not established in Australia
Calanoid Copepod	*Acartia tonsa*	Not established in Australia
Brush-clawed shore crab	*Hemigrapsus takanoi/penicillatus*	Not established in Australia
Encrusting Bryozoan	*Schizoporella errata*	Established in Whyalla
Marbled spinefoot fish	*Siganus rivulatus spp*	Not established in Australia
Solitary ascidian	*Styella plicata*	Established in Whyalla
Caulerpa racemosa	*Caulerpa racemosa var. cylindracea*	Established in West Lakes and Port River (SA)
Caulerpa taxifolia	*Caulerpa taxifolia*	Established in West Lakes (eradicated 2003), North Haven (eradicated 2008) and Port River (SA) resident
Red alga	*Bonnemaisonia hamifera*	Not established in Australia
Devils Tongue Weed	*Grateloupia turuturu*	Established in St Marys (Tas), Port Philip Bay (VIC) and Botany Bay (NSW)
Red alga	*Womersleyella setacea*	Not established in Australia
New Zealand Screwshell	*Maoricolpus roseus*	Tasmania, New South Wales, Victoria
Pleated sea squirt	*Styella plicata*	Port Adelaide

5.3 Risk Factors

Existing and proposed vessel traffic, vessel types, proximity of the Project site to already infected nodes, and nature of proposed marine activities and infrastructure are all risk factors influencing the successful translocation of an IMS to Mullaquana Station during Project activities. These risk factors are described below.

Details regarding the procurement of equipment and materials will form part of Project detailed design, including confirmation if any equipment and materials will be imported, and from which location/marine ports those shipments would come from (if any). Additionally, as part of detailed design, the source location/port of ships/vessels that are to be used during the Project construction will be confirmed. In both instances, post-approval management plans will address the specific risks

and mitigation measures related to pests that exist in those marine port locations in relation to the Project's site-specific environment.

5.3.1 Vessel movements and types

Vessels movements act as a primary vector for IMS translocation to a new environment via vessel biofouling and ballast water release and or exchange. The likelihood of a successful IMS translocation associated with a vessel movement is dependent on both the vessel and travel characteristics.

Vessel movements

Vessels travelling at high speed are at lower risk of transporting IMS. The transport process of a vessel can create significant physical and physiological stresses on species, impacting their ability to survive a voyage. Several studies have demonstrated the relationship between speed and shear stress on species survival (e.g. Coutts 1999; Davidson et al. 2009; Coutts et al. 2010). Vessel speed influences IMS stress and, consequently, influences the success of IMS establishment to a new environment. The faster the vessel travels, the less likelihood of IMS translocation exists.

The probability for an IMS to survive a voyage decreases with the distance between donor and receiving locations. Hewitt et al. (2011) previously showed that longer voyages are relatively less likely to support mature biofouling species than shorter voyages. The shorter the voyage from an infected node to a new environment is, the more likely the potential for successful IMS translocation.

Similarly, longer vessel visit durations at a new location increase the risk of IMS translocation to the environment. The longer a potentially infected vessel remains in a new location, the higher the risk that an IMS could colonise to that new environment based on reaching suitable local eco-physical tolerances and providing organisms time to condition and spawn or recruit via other means.

Vessel type

Trailerable vessels are at lower risk of translocating marine species than vessels with long residence times in the water. Trailerable vessels are regularly dried, experience UV exposure to their wetted areas and are more often cleaned with freshwater, which reduces the risks of biofouling. Biofouling on trailers is commonly dislodged during driving on the road. In comparison, vessels that are rarely removed from the water have an increased risk of being colonised by IMS through biofouling, and therefore represent a greater risk of entraining and translocating marine pests when they are moved (Hewitt et al. 2011; Hayes et al. 2019).

The extent and distribution of area available for colonisation by biofouling species differs across vessel types, influencing the risk of transporting IMS. Moser et al. (2017) previously characterised biofouling niche areas on different commercial ships and showed that passenger vessels had the highest extent of biofouling niche areas relative to their total wetted surface area, while bulk carriers and tankers had the lowest extent of biofouling niche areas. The magnitude of biofouling niche areas is accepted to increase the likelihood of pest entrainment on a vessel (Moser et al. 2017).

5.3.2 Connection with infected nodes

The closer the distance between the Project site and an infected node increases the likelihood for IMS translocation. IMS already established at an 'infected' node may contaminate a visiting vessel via ballast water and/or biofouling. This vessel then has the potential to act as an IMS vector and carry the IMS to its next destination where it may become established.

5.4. Existing IMS risks at Mullaquana Station

5.4.1. Vessel movements and types at Mullaquana Station

Mullaquana Station is a relatively sheltered site shallow sand-flat, appearing to be rarely visited by vessels with the exception of occasional visits by commercial research vessels, recreational line and pot fishermen. Current levels of sea traffic would be expected to be negligible.

Most of the vessels visiting Mullaquana Station are trailerable vessels, which represent a lower risk of translocating IMS. This includes most current research and service vessels which enter the Project site from Whyalla (Figure 5).

Potential remains for infected vessels to reside in the vicinity of Mullaquana Station on their way to Whyalla. This includes commercial vessels, fisheries patrol, fishing, aquaculture, and recreational yachts which may occasionally pass through the area coming from distances of more than 10NM away. Their residence time at Mullaquana Station would be expected to be short though, as it is not a suitable harbor or anchorage. The majority of traffic to the site would be expected to be due to the current project.

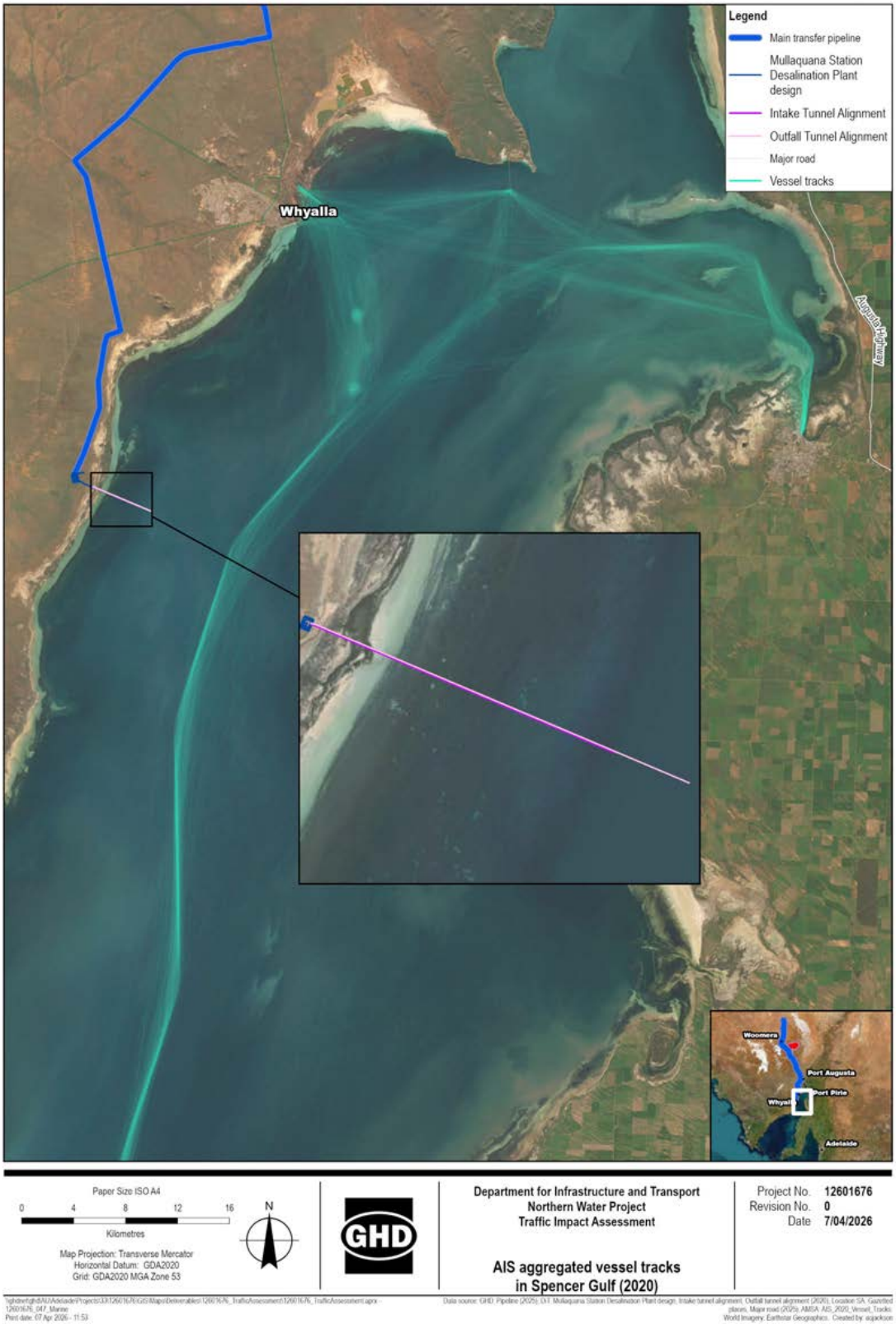


Figure 5: Automatic Identification System aggregated vessel tracks in Spence Gulf (2020) (Source: GHD 2026)

5.4.2. Existing infected nodes

South Australian nodes

Table 3 and Table 4 summarise the South Australian marinas and ports that receive international and domestic vessel traffic, and where marine pest species have previously been recorded[4] (Wiltshire et al. 2010). The risk of IMS translocation associated with each marina and port was determined based on pest presence or absence as illustrated by Wiltshire et al. (2010). Marinas with no IMS records were considered to have a low risk of IMS translocation; marinas where one IMS has previously been recorded were considered to have a moderate risk, and marinas where two or more IMS have previously been recorded were considered to have a high risk (Table 3).

Table 3: South Australian Marinas and Risk ratings

SA Marinas	**Voyage distance to Mullaquana Station(km)**	**Recorded established introduced marine species (Wiltshire et al. 2010 or local report)**	**Risk rating**
Cape Jaffa	670.4	*Undaria pinfitida* (2023)	Low
Penneshaw	388.9	Nil results	Low
Wallaroo	98	*Sabella spallanzanii, Ciona intestinalis*	High
North Haven (CYCSA & Crown)	437.1	*Codium fragile, Sabella spallanzanii, Polycera hedgpethi, Carcinus maenas, Musculista senhousia, Caulerpa racemosa var. cylindracea, Pseudopolydora paucibranchiata, Coriohium acherusicum, Ciona intestinalis*	High
Garden Island	444.5	*Sabella spallanzanii*	Low
Holdfast Quays	453.7	Nil results	Low
Lincoln Cove (Port Lincoln)	240.8	*Magallana gigas, Watersiporia arcuata, Watersiporia subtorquata*	High
Marina Adelaide	444.5	Nil results (Magallana gigas)	Moderate
Outer Harbor (RSAYS)	435.2	*Sabella spallanzanii, Carcinus maenas, Watersiporia arcuata, Ciona intestinalis*	High
Port Vincent	416.7	*Sabella spallanzanii, Carcinus maenas*	High
St Kilda	463.0	*Sabella spallanzanii, Carcinus maenas, Caulerpa taxifolia, Caulerpa racemosa, Styella plicata, Bugula nerita*	High
Robe	611.2	*Gymnogongrus crenulatus, Undaria pinafitda*	High
Wirrina	416.7	*Sabella spallanzanii*	Moderate
Glenelg / Patawalonga	435.2	*Sabella spallanzanii*	Moderate
Whyalla	5.6	*Sabella spallanzanii, Parasesarma erythodactyla*	High
Tumby Bay	218.5	*Magallana gigas, Ciona intestinalis,*	High

[4] Cryptogenic species and toxic dinoflagellates have been omitted from this table.

Table 4: Active South Australian Saltwater Ports

SA Shipping Ports and Key Nodes	Details	Voyage distance to Mullaquana Station (km)	Recorded established introduced marine species (Wiltshire et al. 2010 or local report)	Risk rating
Port Neill	Small town and boat ramp located on the eastern side of the Eyre Peninsula in South Australia, Australia. It is primarily a commercial and recreational line fishing vessels port (dominantly small vessels<7.5m)	185.2	Nil results	Low
Port Adelaide	South Australia's largest trading port handling all types of vessels.	438.9	*Caulerpa taxifolia, Sabella spallanzanii, Ciona intestinalis, Caulerpa racemosa, Carcinus maenas, Charybdis japonica, Magallana gigas, Watersiporia arcuata, Watersiporia subtorquata*	High
Port Lincoln	Major commercial port. It serves as a hub for shipping aquaculture and fishing products, as well as general cargo vessels.	235.2	*Magallana gigas, Watersiporia arcuate, Ciona intestinalis, Caulerpa racemose, Caulerpa taxifolia, Styela plicata, Coryne eximia*	High
Port Augusta	It is a multi-purpose port handling various cargo	453.7	Nil results	Low
Port Pirie	Trading port handling mineral concentrates and general cargo	46.3	Nil results	Low
Port Giles	Bulk grain port	231.5	Nil results	Low
Wallaroo	Handles general cargo vessels	92.6	*Sabella spallanzanii, Ciona intestinalis*	Moderate
Thevenard (Ceduna)	Trading port handling bulk commodities	703.8	*Magallana gigas*	Moderate
Port Bonython	Trading port handling petroleum products	20.4	Nil results	Low
Whyalla	Steel production facilities and serves as a key port for general cargo vessels	5.6	*Sabella spallanzanii, Megabalanus tintinnabulum*	Moderate
Klein Point	Limestone port	240.8	*Sabella spallanzanii*	Moderate
Ardrossan	Deepwater multi use port	300.0	Nil results	Low
Beachport	Fishing port	520.4	Nil results	Low
Kangaroo Island	Multiuser port	231.5	Nil Results (*Sabella spallanzanii*)	Low
Cape Jervis	Ferry port	253.7	Nil Results	Low
Tumby Bay	It primarily serves the local agricultural and fishing industries	29.6	*Magallana gigas*	Low

Australian nodes

Outside of South Australia, national marinas and ports with the highest risk of marine pest translocation to Mullaquana Station are those that experience similar temperature, nutrient and salinity regimes (Hewitt et al. 2011). These are located in the southern temperate bioregions and include the ports of Fremantle (Western Australia- WA), Hobart (Tasmania - TAS), Port Phillip Bay (Victoria - VIC), Portland (VIC), Sydney (New South Wales - NSW), Port Kembla (NSW), Newcastle (NSW) and Botany Bay (NSW). They represent the highest connectivity for interstate journeys to Mullaquana Station.

Table 5 summarises the Australian temperate ports outside South Australia where marine pest species have previously been recorded[5]. The risk of IMS translocation associated with each marina and port was determined based on pest presence or absence as per Wiltshire et al. (2010). Ports with no IMS records were considered to have a low risk of IMS translocation; ports where one IMS has previously been recorded were considered to have a moderate risk, and ports where two or more IMS have previously been recorded were considered to have a high risk.

Table 5: Interstate ports in southern temperate bioregions

Interstate Temperate Ports[6]	**Distance to Mullaquana Station (approx. Km)**	**Recorded marine pest**	**Risk rating**
Fremantle (WA)	1852	*Arcuata senhousia, Sabella spallanzanii, Didemnum vexillum*	High
Hobart (TAS)	1389	*Carcinus maenas, pinafitida, Asterias amurensis, Maoricolpus roseus*	High
Port Phillip Bay (Vic)	1111.2	*Carcinus maenas, Arcuata senhousia, Sabella spallanzanii, Hemigrapsus sanguineus, Undaria pinafitida, Asterias amurensis*	High
Portland (Vict)	740.8	*Arcuata senhousia, Sabella spallanzanii, Undaria Pinafitida, Maoricolpus roseus.*	High
Sydney (NSW)	1481.6	*Caulerpa taxifolia, Didemnum vexillum, Carcinus maenas, Maoricolpus roseus, Sabella spallanzanii*	High
Port Kembla (NSW)	1333.4	*Caulerpa taxifolia, Maoricolpus roseus, Carcinus maenas*	High
Newcastle (NSW)	1574.2	*Nil results*	Low
Botany Bay	1463.1	*Maoricolpus roseus, Caulerpa taxifolia*	High

International nodes

This study assumes that some materials and equipment that will be used for the construction of the desalinisation plant could be sourced from international locations and transported directly to Mullaquana Station, whilst the remainder will arrive from domestic locations by sea or by road freight. International ports of high risk to Mullaquana Station are located on trade routes with our major trading partners, being China, Japan, South Korea, India, United States, Taiwan, New Zealand, Vietnam, Indonesia, & Singapore (Workman, 2024). International commercial shipping maintenance

[5] Cryptogenic species and toxic dinoflagellates have been omitted from this table.
[6] From Marinepests.gov.au/pests/map

hubs which include China, Singapore, India, Indonesia, Taiwan and Korea have traditionally become donor ports for marine pests where large numbers of ships have been cleaned and maintained over the last century.

Source ports and risks from international vessels are managed by the Commonwealth Government's MARS but minimisation of risk is still undertaken by accredited biofouling inspectors overseas and in Australia. International tugs and barges are often dry docked, cleaned and antifouled in smaller shipping yards before picking up cargo due for Australia. These present a high risk where the heavily disturbed habitats are often rich in pests and the strict requirements for Australia's biofouling standards are not well understood in these countries. Alternatively, Australia has an internationally respected reputation in the area of marine biosecurity management (Piola et al 2018). This represents a disparity between the donor country and Australian destinations management compliance expectations which can present difficulties.

5.5. Potential IMS vectors from the Project

Potential IMS vectors during construction and operation of the Project are described below.

5.5.1. Potential vectors from the construction phase

Domestic and international commercial vessels will be required for the Project construction activities and may be a risk of IMS introduction to Mullaquana Station. Part of the materials and equipment that will be used for the Project construction may be sourced from international locations and transported directly to Mullaquana Station. All equipment requires biofouling inspection prior to arrival at site. Commercial vessels may depart from or visit an infected port or marina for refuelling, maintenance or to pick up supplies, and later transport an IMS to Mullaquana Station through biofouling and/or ballast water.

The key waste streams from construction of the intake and outfall are derived from the launch shaft excavations, TBM slurry, and spoil returned from the JUB to land from the excavation of the risers and other marine infrastructure. Anticipated volumes for each are ~110,000 m^3 from the TBM and 2,150 m^3 from the JUB. Spoil material from the TBM will be transported to surface. Spoil from the riser and other marine infrastructure excavations will be removed by a supply vessel to the supporting infrastructure at Whyalla port.

Table 6: Expected vessel type and numbers during project from project operations and local traffic from GHD (2026)

Vessel type	Number	Likely Donor Port
Jack Up Barge	1	State
Floating support barge with mobile crane	1	State
Tug Boat	3	Local (Whyalla)
Crew Transfer Vessel	2	Local (Whyalla)
Materials stockpile and transport barge	2	State
Offshore Support Vessel (OSV) with hydrographic survey capabilities	1	State

Vessel type	Number	Likely Donor Port
Rock Dump Vessel	1	State
Dive Support Vessel	1	Local
Cargo ship	26	International or Interstate
Fishing	3	Local
Other-All	3	Local, State, Interstate or International
Port tender, pilot vessels and tugs	13	Local
Sailing	27	Local, State. Interstate or International
Search and Rescue	3	Local
Tankers	4	International

The construction phase of the Project will involve the use of temporary submersible equipment and infrastructure. Submersible equipment may be contaminated by IMS or disease from a previous location and will require biofouling inspection. If not cleaned properly before departing that location, the equipment may act as vector for IMS translocation. Submersible equipment for the Project construction includes, but is not limited to, a tunnel boring machine, float and sink, shaft and drill, blast and bulk excavation equipment. Where the tunnel boring occurs from land, the equipment will be low risk from IMS.

5.5.2. Potential vectors from the operational phase

Operation of the Project will involve permanent underwater infrastructure that may provide settlement habitat for potential IMS. Permanent underwater infrastructure will include:

- Intake and outfall tunnels (to be located underneath the seabed and not located in sea water);
- intake structure;
- outfall diffuser structure; and
- Associated riser connections.

The EIS assessment is broadly based on the engineering Reference Design for the Project. Regarding marine structures, the Reference Design includes:

Reference Design: two tunnels for the intake and outfall, both constructed using a Tunnel Boring Machine (TBM). The outfall diffuser structure has a series of nozzles (manifold) attached to the end of the outfall pipe which is placed in the marine environment using a float and sink methodology.

For marine structures, the following option for marine structures also forms part of the EIS assessment:

Float and Sink Option: A single intake tunnel extending to the intake location and continuing offshore via approximately 1,000 m of additional outlet conduit placed on the seafloor to the outfall

diffuser manifold using a float and sink method. The outfall diffuser manifold would also be placed using a float and sink method.

The Reference Design intake structure has the following configuration:

- located in a minimum approximate water depth of 13 m from LAT (approximately -15m AHD)
- intake head diameter = approximately 11.3 to 11.5 m
- vertical extent of intake screens = 2.5 m
- intake structural cap = 0.5 m
- clearance of bottom of screen to seabed = 3.0 m

The Reference Design diffuser design selected has the following configuration

- a diffuser manifold array with 8 nozzles at 45-degree angles, 12 m spacing between nozzles and 12 m spacing between driven support piles.
- outfall diffuser located at a minimum 17 m depth at LAT (-19m AHD) with 12 m of separation between the nozzles.
- the foundation of the diffuser structure will comprise rockfill placed into a 1.5 m deep trench.

Marine traffic and use of temporary submersible equipment will be limited during the Project operation activities. Vessels to be used during the operation phase would typically include maintenance, monitoring and research vessels. Temporary submersible equipment may occasionally be used during maintenance activities.

5.6. Risk Assessment

5.6.1. Methodology rationale

Various approaches have been proposed for IMS Risk Assessments over the last 35 years, each presenting different challenges. These include species-specific and environmental matching assessments (Hayes et al. 2019). Species-specific assessments have the potential to overestimate risks and can be challenging to undertake due to the lack of available data on IMS tolerance range and distribution (Hayes et al. 2019). Environmental matching assessments, which considers environmental similarities and dissimilarities between donor and receiving locations, can be biased by the choice of environmental variables (Hayes et al. 2019).

As described in Sections 3.2 and 5.3, vessels and donor ports are the main factors at the source of IMS translocation. As such, while risk species have been listed in this report (refer Table 2), the vessel and donor port risk vectors have been reasoned to be the quantifiable risk assessed component where mitigation effort can be applied.

5.6.2. Methodology

The methodology used in this Risk Assessment adapts the five-step process undertaken by Hewitt et al. (2011) and detailed below.

Step 1 – Identification of endpoints

This report evaluates the risks associated with two endpoints: the inoculation and the establishment of an IMS at Mullaquana Station as a result of the Project activities.

Step 2 - Identification of hazards

Hazards are the IMS species of concern to Mullaquana Station. These species were identified in Section 5.2.

Step 3 - Determination of consequences

Consequences of potential IMS inoculation and establishment at Mullaquana Station were considered at the environmental, economic, social/cultural and human health level following the methodology of Hewitt et al. (2011). In this assessment all IMS detections and establishments are considered to be of Major Consequence for the construction or operation of a desalinisation plant. The criteria used in this Risk Assessment is provided below in Table 7.

Step 4 – Determine the increase or decrease in consequences based on donor ports

Where vessels are later determined to be from Moderate risk donor ports from Table(s) 3, 4 and 5 their consequence rating can be kept at Major consequence (shown).

Where vessels are later determined to be from high-risk donor ports from Table(s) 3, 4 and 5 their consequence rating can be increased from Major to Critical and applied to Table 12.

Where vessels are later determined to be from low-risk donor ports from Table(s) 3, 4 and 5, their consequence rating can be decreased from Major to Minor and applied to Table 12.

Where locations are several days voyage away, (<1000NM) their consequence rating can be decreased to a lower risk rating by one criterion and applied to Table 12.

Table 7: Consequence criteria

Category	Description
Critical	Results in the permanent establishment of an APMPL species, or other species from Table 2 at Mullaquana Station which results in: • the construction phase being quarantined and/or • ongoing maintenance costs during operation of the desalination plant and/or • affects one or more stakeholders requiring repatriation costs and/or • measured effects on local species. National media attention and stakeholder relations damaged. Government mediated response actions taken.
Major	Results in the establishment of a APMPL or other species from Table 2 inside a designated vessel quarantine zone which results in the construction phase being delayed and/or alerts several stakeholders restricting operations, state media attention and local unrest. Government mediated response actions taken.
Moderate	Results in the detection of a APMPL or other species from Table 2 inside a designated vessel quarantine zone which results in the construction phase being delayed and/or alerts one or more stakeholders restricting operations and picked up by local media.
Minor	Results in the detection of a less important IMS species in Table 2 (e.g. *Sabella spallanzanii*) inside a designated vessel quarantine zone outside Mullaquana Station, which does not restrict operations but is picked up by local media.
Insignificant	Does not affect any stakeholders or environmental values

Step 4 - Determination of likelihood

The likelihood of IMS inoculation and establishment is dependent on a range of risk factors described in Section 5.3. The likelihood of successful IMS translocation across vessel types was ranked based on the risk factors described in Section 5.3.1. This was undertaken using the likelihood criteria presented in Table 8. The higher likelihood factor (*Almost Certain*) was chosen towards the precautionary principal where reasonable. Almost certain assumes that a biofouling organism has attached in the donor port and that the receiving environment is suitable.

A likelihood of Rare was not considered to be a viable initial likelihood, given that all vessel journeys have an inherit IMS risk whilst also adopting the precautionary principle (Hayes et al. 2019). The residence time of a vessel at a potential infected port/marina and at a receiving site cannot be predicted, and therefore this was not considered in the likelihood criteria (Table 9). It could be considered on an individual vessel risk assessment in the future.

Table 8: Definition of likelihood using the precautionary principle

Likelihood	Description
Almost Certain	Almost certain to occur in most circumstances.
Likely	Is likely to occur in most circumstances.
Possible	Possible to occur in most circumstances.
Unlikely	Unlikely to occur in most circumstances.
Rare	May occur but only in rare and exceptional circumstances.

Table 9: Likelihood criteria for successful IMS translocation based on vessel speed

	Likelihood			
Criteria	**Unlikely**	**Possible**	**Likely**	**Almost Certain**
Vessel Speed (Knots)	30+	20-29.9	10-19.9	<10
Number of niche areas (From Moser et al. 2017) x 10^3 (m^2) or estimated via Author's inspection experience	None	>5	6-10	11+
Trailerable or vessel predominately stays in the water	Trailerable	Trailerable	In water	In water
Distance between donor and receiving location[7]	International	National	State	Adjacent

Step 5 – Calculation of Risk

Risk of IMS inoculation and establishment of IMS at Mullaquana Station as a result of the Project was calculated as the product of the likelihood and consequence as per the risk matrix provided in Table 10 and Table 11.

[7] Assumes higher level of AQIS scrutiny for international arrivals

Table 10: Risk matrix

		Consequences				
Likelihood		Insignificant	Minor	Moderate	Major	Critical
	Almost Certain	Moderate	Moderate	High	Extreme	Extreme
	Likely	Moderate	Moderate	High	High	Extreme
	Possible	Low	Moderate	Moderate	High	High
	Unlikely	Low	Low	Moderate	Moderate	High
	Rare	Low	Low	Low	Moderate	High

Table 11: Priority for Action

Risk Level	Priority for Action Guide[8]
Extreme	**Act immediately:** Cease vessel translocation into Mullaquana Station area until risk is eliminated/reduced from the extreme risk level.
High	**Act today**: The situation must be controlled as a matter of priority. Vessel should be independently checked before entry to Mullaquana Station area. Implement immediate temporary controls. Further permanent controls must be managed.
Moderate	**Act soon**: The situation is important. Immediate/permanent controls must be managed.
Low	The situation may be managed using routine procedures. Be sure that this level does not have the potential to escalate to a higher level.

[8] *Reduce the level of risk to as low as reasonably practicable*

5.6.3. Results

The likelihood of successful IMS translocation relative to different vessel types is given in Table 12, and was used for the Risk Assessment as summarised in Table 13.

Table 12: Likelihood of potential for successful IMS translocation across vessel types (untreated)

Vessel type	Generalised Average Vessel speed (Knots)	Number of biofouling niche areas. International vessels data are provided by Moser et al. (2017). Other vessels data not in Moser et al (2017) are estimated.	Trailerable or in-water	Voyage distance	Likelihood
International Bulk Carriers	15	26	In water	International	Possible
International Barges and lighters	5	15	In water	International	Possible
Domestic Berthed Charter boats	30	8	In water	State	Almost certain
Domestic Trailerable Charter boats	30	5	Trailerable	State	Unlikely
Domestic Coastguard and Water Police patrol boats	30	5	Trailerable	State	Unlikely
International Dredges	10	10	In water	International	Likely
Domestic Dredges	10	10	In water	National	Almost certain
Domestic Mobile drilling rigs	5	10	In water	National	Almost certain
Offshore support vessels	25	10	In water	National	Possible
Domestic Tugboats	15	10	In water	State	Almost certain
Small powered craft (cabin cruisers, dive boats and tinnies)	25	5	Trailerable	Adjacent	Unlikely
Research vessels (Berthed)	25	5	In water	State	Likely
Seismic survey ships (other ships)	5	18	In water	International	Likely

5.6.4. Project Risk mitigation discussion

A key mitigation measure for the Project will be to avoid vessel movements between the Project site and known infected nodes with high-risk species (Refer Section 5.4.2). Where an infected node is to be used, timing of operations to avoid spawning seasons and increased application of vessel hygiene controls can be applied. A range of additional risk mitigation best practice techniques can be undertaken to minimise risk of IMS. These include:

- Track record: Use vessel operators with a successful track record of ballast water and biofouling management.
- Vessel Tracing and Journey Assessment: for all vessels during construction and operational phases.
- Biofouling guidelines: Checking every vessel is being managed according to the appropriate set of National Biofouling Guidelines for each vessel type.
- Antifouling Coatings: Application of coatings on ship hulls and submerged surfaces to prevent or reduce biofouling. Ensure that the antifouling used matches the manufacturer's recommendations for vessel type and speed.
- Regular Cleaning: Scheduled maintenance and cleaning of submerged surfaces to remove biofouling organisms.
- Design Improvements: Developing structures and surfaces that are less prone to biofouling through design and material selection.
- Establishing long journey vessels arrive at virtual quarantine zone(s) before entry to the site to ensure that vessels are clear of pests.
- Every vessel involved in the Mullaquana Station location should be assessed according to the Generic Approach to Biofouling Risk Assessment (MPSC, 2009) for each journey into the site.
- Regular monitoring of the area during construction phase for early identification of IMS and diseases and initiation of early response (if required).
- Pre-checks on any international vessels: For ballast water and biofouling with AQIS and other Australian Ports to ensure standards are being met.
- Ensure all vessels entering the area have been either slipped, cleaned, antifouled and inspected immediately before entering the area with an Under Water Inspection In Lieu of Dry Docking (UWILD) to comply with the Australian Biofouling Management Requirements.
- Ensure all vessels entering the area are inspected immediately before entering the local quarantine zone by a Suitably Qualified Invasive Marine Pest Expert
- Ensure all trailerable vessels are hygiene inspected and vector traced by the operator before entering the area.
- Ensure a vessels' residence time at the donor port and Mullaquana Station is minimised as it will influence the likelihood of successful IMS translocation. Idle residence periods at either destination should be minimised in time as much as reasonably possible when undertaking marine activities.

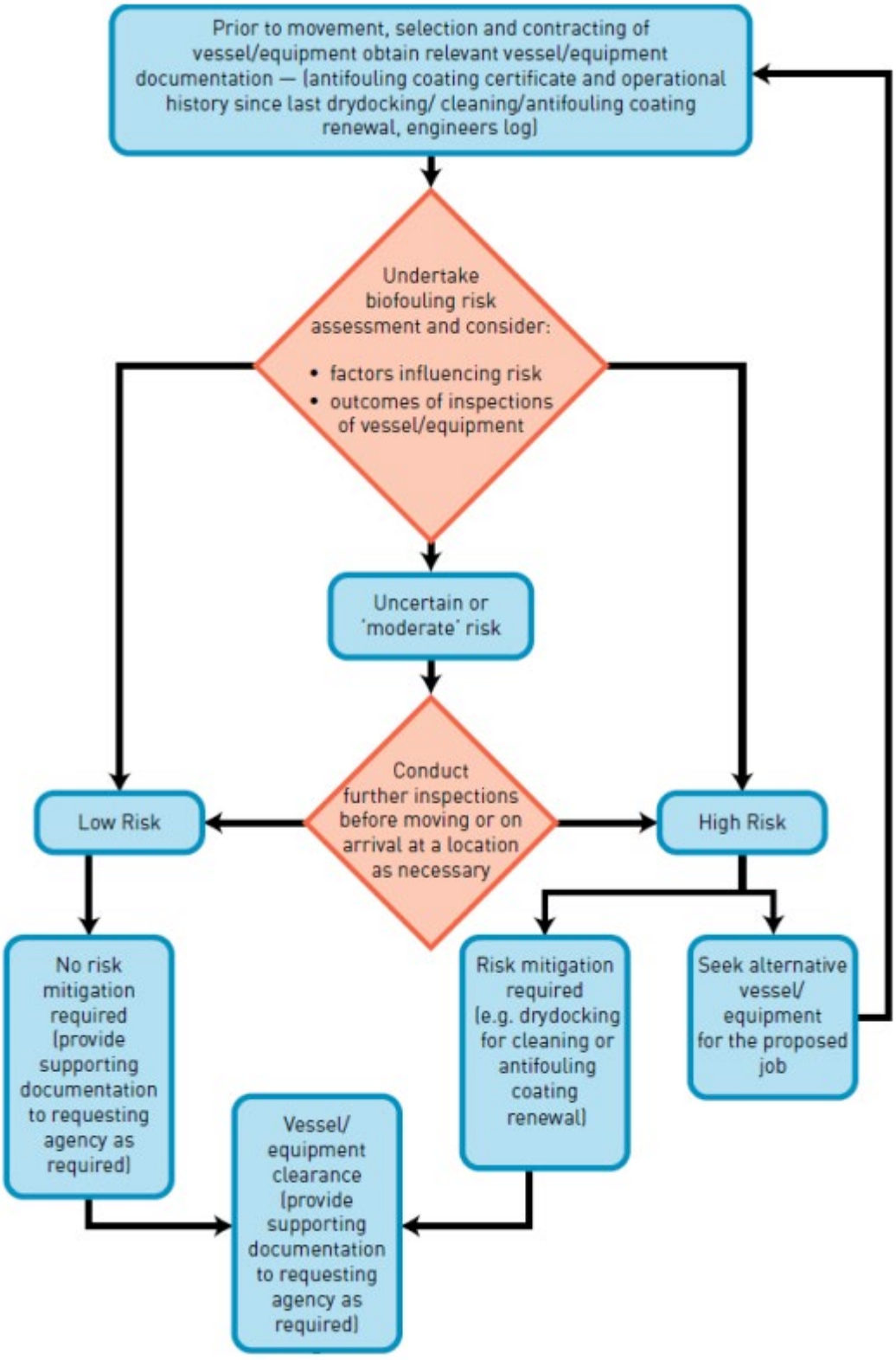


Figure 6: Generic approach to biofouling risk assessment (source: MPSC, 2009)

Table 13: Summary of Risk Assessment

Control measures set out below (including management plans / communications plans) are to be confirmed post approval and as part of detailed design.

Vessel type (Risk vector)	Hazard	Risk	Inherent Risk Rating			Recommended Control Measures	Residual Risk Rating		
			Consequence	Likelihood	Risk Level		Consequence	Likelihood	Risk Level
International Bulk Carriers	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Possible	High	Ballast water and Biofouling managed by AQIS / MARS. Communications plan. Regular checks of adherence to ballast water management plan, record books and biofouling management plan. antifouling certificate.	Major	Rare	Moderate
International Barges and lighters	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Possible	High	Ballast water and Biofouling managed by AQIS / MARS. Communications plan. Regular checks of adherence to ballast water management plan, record books and biofouling management plan. Regular checks of antifouling certificate.	Major	Rare	Moderate

Vessel type (Risk vector)	Hazard	Risk	Inherent Risk Rating			Recommended Control Measures	Residual Risk Rating		
			Consequence	Likelihood	Risk Level		Consequence	Likelihood	Risk Level
Domestic Berthed Charter boats	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Almost certain	Extreme	Communications plan. Vessel tracing. Donor port assessed. Vessel slipped and cleaned within reasonable time of disembarking for Mullaquana Station. Targeted risk assessment and inspection by qualified biofouling inspector before journey to site.	Major	Rare	Moderate
Domestic Trailerable Charter boats	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Unlikely	Moderate	Communications plan. Vessel tracing. Donor port assessed. Targeted risk assessment and inspection by qualified biofouling inspector before journey to site.	Major	Rare	Moderate
Domestic Coastguard and Water Police patrol boats	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Unlikely	Moderate	Communications plan. Vessel tracing. Donor port assessed. Targeted risk assessment and inspection by qualified biofouling inspector before journey to site.	Major	Rare	Moderate

Vessel type (Risk vector)	Hazard	Risk	Inherent Risk Rating			Recommended Control Measures	Residual Risk Rating		
			Consequence	Likelihood	Risk Level		Consequence	Likelihood	Risk Level
International Dredges	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Likely	Extreme	Ballast water and Biofouling managed by AQIS / MARS. Communications plan. Regular checks of adherence to ballast water management plan, record books and biofouling management plan. Regular checks of antifouling certificate. Targeted risk assessment and inspection by qualified biofouling inspector before journey to site.	Major	Rare	Moderate

Vessel type (Risk vector)	Hazard	Risk	Inherent Risk Rating			Recommended Control Measures	Residual Risk Rating		
			Consequence	Likelihood	Risk Level		Consequence	Likelihood	Risk Level
Domestic Dredges	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Almost certain	Extreme	Communications plan. Vessel tracing. Donor port assessed. Vessel slipped and cleaned within reasonable time of disembarking for Mullaquana Station. Targeted risk assessment and inspection by qualified biofouling inspector before journey to site.	Major	Rare	Moderate
Domestic Mobile drilling rigs	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Almost certain	Extreme	Communications plan. Vessel tracing. Donor port assessed. Vessel slipped and cleaned within reasonable time of disembarking for Mullaquana Station. Targeted risk assessment and inspection by qualified biofouling inspector before journey to site.	Major	Rare	Moderate

Vessel type (Risk vector)	Hazard	Risk	Inherent Risk Rating			Recommended Control Measures	Residual Risk Rating		
			Consequence	Likelihood	Risk Level		Consequence	Likelihood	Risk Level
Domestic Offshore support vessels	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Possible	High	Communications plan. Vessel tracing. Donor port assessed. Vessel slipped and cleaned within reasonable time of disembarking for Mullaquana Station. Targeted risk assessment and inspection by qualified biofouling inspector before journey to site.	Major	Rare	Moderate
Domestic Tugboats	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Almost certain	Extreme	Communications plan. Vessel tracing. Donor port assessed. Vessel slipped and cleaned within reasonable time of disembarking for Mullaquana Station. Targeted risk assessment and inspection by qualified biofouling inspector before journey to site.	Major	Rare	Moderate

Vessel type (Risk vector)	Hazard	Risk	Inherent Risk Rating			Recommended Control Measures	Residual Risk Rating		
			Consequence	Likelihood	Risk Level		Consequence	Likelihood	Risk Level
Small powered craft (cabin cruisers, dive boats and tinnies)	IMS (Table 2 and international species not listed)	Detection or establishment of IMS as per Table 7	Major	Unlikely	Moderate	Communications plan. Vessel tracing. Donor port assessed. Targeted risk assessment and inspection by qualified biofouling inspector before journey to site.	Major	Rare	Moderate
Research vessels (Berthed)	IMS (Table 2 and international species not listed)	Detection or establishment of IMS	Major	Likely	Extreme	Communications plan. Vessel tracing. Donor port assessed. Vessel slipped and cleaned within reasonable time of disembarking for Mullaquana Station. Targeted risk assessment and inspection by qualified biofouling inspector before journey to site.	Major	Rare	Moderate

Vessel type (Risk vector)	Hazard	Risk	Inherent Risk Rating			Recommended Control Measures	Residual Risk Rating		
			Consequence	Likelihood	Risk Level		Consequence	Likelihood	Risk Level
International Seismic survey ships (other ships)	IMS (Table 2 and international species not listed)	Detection or establishment of IMS	Major	Likely	Extreme	Ballast water and Biofouling managed by AQIS / MARS. Communications checks of ballast water management plan, record books and biofouling management plan, antifouling certificate	Major	Rare	Moderate
International Bulk Carriers	IMS (Table 2 and international species not listed)	Detection or establishment of IMS	Major	Likely	Extreme	Ballast water and Biofouling managed by AQIS / MARS. Communications checks of ballast water management plan, record books and biofouling management plan, antifouling certificate	Major	Rare	Moderate

5.6.5 Project control measures to prevent IMS colonisation during the construction phase

A range of optional measures can be applied directly to the site to guard against IMS colonisation of Project infrastructure. Preparation to prevent an incursion is necessary so the construction phase is not suspended in the case of a detection or colonisation of IMS. These measures may include:

1. Preconditioning submerged infrastructure with local species fouling to limit the provision of colonisation by visiting vessels.
2. Installing biofouling settlement plates as an early warning system at the Project site. These settlement plates may also act as primary settling zones for any pests that reach the site, which can then be removed easily.
3. Having a marine pest protector sleeve or other smothering quarantine blankets or wrapping available to smother pests on submerged infrastructure.
4. Using suitable antifouling paints on submerged infrastructure at Mullaquana Station.

Following and applying any or all of the recommendations in this report does not guarantee an IMS will not enter the Project site or colonize the local area. The proposed control measures will ensure that the risk of IMS translocation is maintained to as low as reasonably practicable.

Water entry for vessels can be via the Whyalla boat ramp and the nearest major shipping port is also Whyalla. Port Lincoln and Port Pirie are also close ports that could be involved in major logistics operations for the construction and maintenance of the desalination plant. Focusing on the risks posed by these locations as well as Port Adelaide will be imperative to prevent IMS introduction and establishment from the Project construction activities. Infected nodes that cannot be avoided will require increased vessel hygiene controls and inspections. This can include more regular in water cleans, inspections and logistics timing to avoid species spawning seasons.

If a new IMS is detected, where possible the vessel should immediately move outside 12 NM from the territorial sea baseline and PIRSA Biosecurity SA should be contacted directly or through PIRSA Fishwatch 1800 065 522 where the Department will either manage the quarantine response or defer to DAFF for quarantine instructions.

5.6.6 Project control measures to prevent IMS colonisation during the operational phase

The operational phase control measures are largely the same as those of the construction phase (Section 5.6.5 above), relying on inspection and treatment for control. Additional control measures that can be taken during the operational phase can include:

- Regular chemical descaling / flushing and diver/Remotely Operated Vehicle ROV maintenance of diffusers and intake screens.
- Use of type approved antifouling paints on submerged infrastructure and vessels.
- Continuing with Mullaquana Station area based quarantine site controls for vessel access.
- Regular biofouling treatments applied to submerged infrastructure (e.g. smothering covers, Use and removal of denso-tape or other infrastructure wraps).

- Having several sets of removable submerged infrastructure (e.g. diffusers) that can be removed and retrofitted whilst any biofouling / cleaning is underway so the desaliniaation plant can continue to operate.
- Regular sealing of intake pipe and diffuser pipe for time period to generate anoxic/toxic conditions in the pipes suitable to kill off all fouling species.

This document requires regular 6 monthly revision (or another appropriate time frame) during any monitoring or capital works of the proposed Project.

6. Glossary and Acronyms

Term	Definition
AbHV	Abalone herpesvirus
AGM	Asian Green Mussel *Perna viridis*
APMPL	Australian Priority Marine Pest List
AQIS	Australian Quarantine Inspection Service
AVG	Abalone Viral Ganglioneuritis
CCIMPE	Consultative Committee for Introduced Marine Pest Emergencies
CCSZ	Cuttlefish Coast Sanctuary Zone
DAFF	Department of Agriculture Forestry and Fisheries (Commonwealth)
EIS	Environmental Impact Statement (Assessment)
IMS	Introduced Marine Species
Km	Kilometres
MARS	Maritime Arrivals Reporting System
MIC	Microbial-induced corrosion
NBC	National Biosecurity Committee
NIMPCG	National Introduced Marine Pests Coordination Group
Nm	Nautical Mile
Node	Location where IMS have the potential to occur
ONWD	Office of Northern Water Delivery
OsHV-1	Ostreid herpesvirus-1
OSV	Offshore Support Vessel
PDI Act	*Planning Development and Infrastructure Act 2016*
POMS	Pacific Oyster Mortality Syndrome
PSU	Practical Salinity Unit
SA	South Australia

SARDI	South Australian Research and Development Institute
The Project	The Northern Water project
UWILD	Under Water Inspection In Lieu of Dry Docking

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Appendix G-16:

Giant Australian Cuttlefish Assessment

Northern Water Project

Department for Infrastructure and Transport

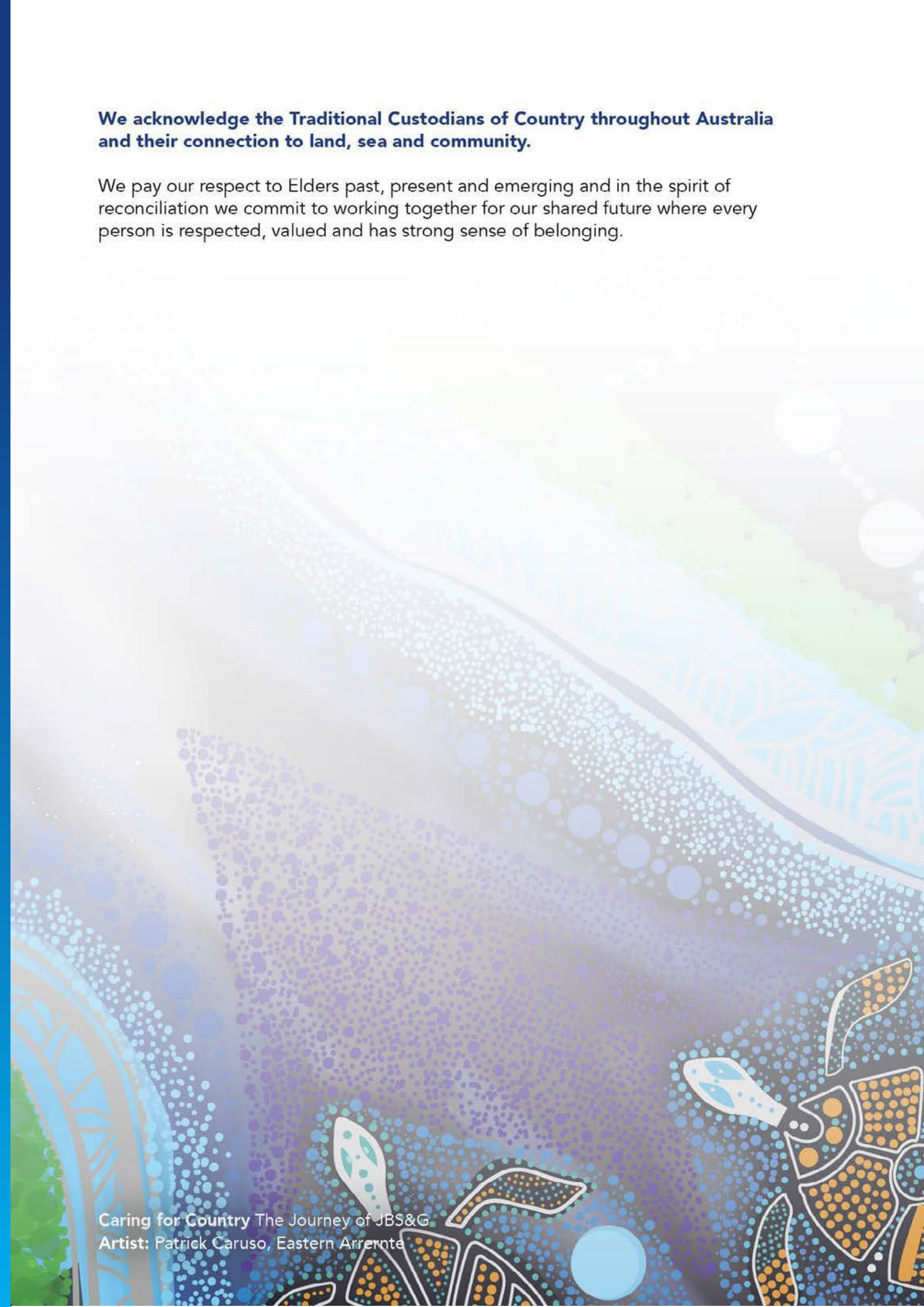
Giant Australian Cuttlefish Assessment

63753 | 172,522 (Rev 1)

22 September 2026

We acknowledge the Traditional Custodians of Country throughout Australia and their connection to land, sea and community.

We pay our respect to Elders past, present and emerging and in the spirit of reconciliation we commit to working together for our shared future where every person is respected, valued and has strong sense of belonging.



Caring for Country The Journey of JBS&G
Artist: Patrick Caruso, Eastern Arrernte

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Appendices

Appendix A **Assessment Methodology for Giant Australian Cuttlefish Impact Assessment**

Appendix B **Giant Australian Cuttlefish Assessment – Impact Tables**

Abbreviations & Terms

Term	Definition
CCSZ	Cuttlefish Coast Sanctuary Zone
CIP	Clean-in-place
DCCEEW	Department of Climate Change, Energy, the Environment and Water
DoE	Department of the Environment (now DCCEEW)
EIS	Environmental Impact Statement
EP Act	*Environment Protection Act (South Australia)*
EPA	Environmental Protection Authority
EPBC Act	*Environment Protection and Biodiversity Conservation Act 1999 (Cwth)*
GL/y	Gigalitres per year
GSV	Gulf Saint Vincent
JUB	Jack-up Barge
LAT	Lowest Astronomical Tide
LSG	Lower Spencer Gulf
MNES	Matters of National Environmental Significance
PDI Act	*Planning Development and Infrastructure Act 2016 (South Australia)*
PIRSA	Primary Industries and Regions South Australia
ppt	Parts per thousand (measure of salinity)
Project Area	The Project Area covers temporary construction activities as well as the predicted area of potential ecological effect during operation
PSU	Practical salinity unit
RO	Reverse Osmosis
SARDI	South Australian Research and Development Institute
SEDMP	Soil erosion and drainage management plan
TBM	Tunnel boring machine
TSSC	Threatened Species Scientific Committee
the Project	Northern Water Project
USG	Upper Spencer Gulf
USG cuttlefish	*Ascarosepion apama* Upper Spencer Gulf (northern genetic cluster of cuttlefish).

Executive Summary

The Northern Water Project (the Project) proposes a secure, climate-resilient water source for industry located in South Australia's Upper Spencer Gulf and Far North. Central to the project is a seawater reverse osmosis desalination plant at Mullaquana Station, approximately 25 km south-west from the centre of Whyalla, delivering up to 51 GL/year via a pipeline network extending north to Olympic Dam, with connections to key mining operations.

An important environmental consideration is the Giant Australian Cuttlefish, particularly the unique breeding aggregation located 34 km northeast of the proposed plant at Point Lowly. This area, now designated as the Cuttlefish Coast Sanctuary Zone (CCSZ) under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), holds National Heritage status and is of significant ecological and community value.

Through site selection investigations and associated stakeholder engagement for the Project has identified public interest in potential impacts on the Giant Australian Cuttlefish from the Project, especially from return water discharge. Although the project footprint is geographically distinct from the CCSZ, an assessment was undertaken to evaluate any potential impacts to the National Heritage place or to the population more broadly.

This report supports the EPBC Act Referral and the Environmental Impact Statement for the Project and seeks to provide a comprehensive understanding of potential impacts to the Giant Australian Cuttlefish as a result of the construction and operation of the Project, and whether there is risk of significant impact to the National Heritage values associated with the Cuttlefish Coast Sanctuary Zone. The assessment considers the Project's proposed mitigation measures and their adequacy in protecting the Giant Australian Cuttlefish and their breeding grounds.

Key findings of the Giant Australian Cuttlefish assessment:

- The CCSZ lies 34 km from the project site; no direct interaction is expected, and EPBC Act National Heritage values will remain unaffected.
- Project specific marine ecology surveys conducted over a 12 month period during 2025 recorded no Giant Australian Cuttlefish individuals at the Project site, which is not a known high-use area for the species.
- While foraging or resting individuals have the potential to occur within the Project Area occasionally, construction impacts are expected to be short-term and localised.
- Operational impacts are limited to a localised area around the outfall, with negligible risk to individual cuttlefish or the broader population. The discharge of return water is expected to have negligible impact on the Giant Australian Cuttlefish population because:
    - salinity beyond the immediate vicinity of the diffuser would remain within the range of natural seasonal variability in the marine Project Area for most of the year.
    - the Giant Australian Cuttlefish has a tolerance for higher salinity than would be experienced as a result of return water discharge, as inferred from its distribution in Spencer Gulf.
    - Review of peer reviewed ecotoxicity studies indicated that within close proximity to the diffusers, Giant Australian Cuttlefish would be resilient to salinities likely to be experienced as a result of return water discharge (see Environmental Impact Statement for further detail).
    - Return water discharge is not anticipated to materially affect Giant Australian Cuttlefish movement.

- Return water discharge is not anticipated to have any impact on Giant Australian Cuttlefish breeding, as there is no breeding habitat present in the Project Area and the breeding aggregation location is 34 km away.

The construction and operation of the Northern Water desalination plant are expected to have no direct or indirect impacts on the Giant Australian Cuttlefish population or the EPBC Act National Heritage listed CCSZ.

1. Introduction

1.1 Overview

The Northern Water Project (the Project) aims to provide a secure, climate resilient water source for industry located in South Australia's Upper Spencer Gulf and Far North. This would enable the growth of industries and key mining resources crucial for contributing to decarbonisation initiatives, sustainability and unlocking long term economic prosperity for South Australia.

The delivery of the Project will involve the construction and operation of a 51 gigalitre per year (GL/y) seawater reverse osmosis desalination plant at Mullaquana Station in the Upper Spencer Gulf and its connection to the Far North via approximately 400 km of transfer system (including pipeline, transfer pump stations and water storages) and electricity infrastructure. The proposed desalination site at Mullaquana Station is located approximately 25 km south-west from the centre of Whyalla.

The Giant Australian Cuttlefish is an iconic species that occurs across the southern coastline of Australia. The Upper Spencer Gulf population of cuttlefish represents a distinct population which forms the largest known breeding aggregation in the world, along a small 8 km stretch of subtidal reef around Point Lowly, ~34 km from the proposed Northern Water desalination plant. This unique aggregation area has recently been declared a National Heritage Place under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) as the Cuttlefish Coast Sanctuary Zone (CCSZ) and is highly valued from an ecological perspective and by the community of South Australia.

Northern Water has undergone a comprehensive site selection process to achieve the Project goal to identify the location that best balances the environmental, economic, social and engineering inputs and constraints to create a Project which is able to support current and future needs.

Feedback received over 3 years of Project engagement and consultation with stakeholders and community members highlighted public interest in the Giant Australian Cuttlefish. In particular, results of engagement identified interest in the potential impact of return water and the impact on cuttlefish migrating to Point Lowly to breed.

Although the Project and its area of influence is ~34 km south and spatially distinct from the breeding grounds of the Giant Australian Cuttlefish, given the iconic status and EPBC Act National Heritage listing of the breeding aggregation, and its importance to the local, national and international community, the potential for impact on the National Heritage listed CCSZ, and the Giant Australian Cuttlefish more broadly has been assessed and is presented in this report.

1.2 Purpose of this report

This report has been prepared to support the Environmental Impact Statement (EIS) for the Project and present the outcomes of the desktop Giant Australian Cuttlefish Assessment. The purpose of the Giant Australian Cuttlefish Assessment is to evaluate whether Project activities will impact on the key values of the CCSZ, or the Giant Australian Cuttlefish more broadly.

This report has been prepared with consideration to the EPBC Act and *Planning Development and Infrastructure Act 2016* (PDI Act) and associated requirements and guidelines.

1.3 Objectives

This assessment seeks to provide a comprehensive understanding of potential impacts to Giant Australian Cuttlefish as a result of the construction and operation of the Project, and whether there is risk of significant impact to the National Heritage values associated with the CCSZ. The assessment considers the Project's

proposed mitigation measures and their adequacy in protecting the Giant Australian Cuttlefish and their breeding grounds.

The objectives of the Giant Australian Cuttlefish Assessment are to:

- Gain an understanding of the known distribution, biology and ecology of Giant Australian Cuttlefish through a desktop review of relevant literature.
- Interrogate Project activities and proposed mitigation measures and the potential for Project interaction with the CCSZ and Giant Australian Cuttlefish based on Project detail provided in the Northern Water Reference Design (GHD 2025) and Constructability Report (Systra 2025), as well as Northern Water hydrodynamic modelling of desalination plant operation by both SARDI and Water Technology.
- Assess the significance of potential impacts on the key values of the National Heritage listed CCSZ, based on guidance criteria provided in the Department of Climate Change, Energy, the Environment and Water (DCCEEW) Significant Impact Guidelines (DoE 2013).
- Assess the impact on the Giant Australian Cuttlefish population more broadly, using the Northern Water EIS impact assessment methodology provided in Appendix A.

1.4 Scope

The scope of this report is to cover potential impacts to the CCSZ and Giant Australian Cuttlefish during construction and operation of the Project. There are two assessments with different scopes:

1) The CCSZ Significant impact Assessment assesses the potential impacts of the Project construction and operation on the CCSZ, and the significance of those impacts. As per the Significant Impact Guidelines (DoE 2013), to have a significant impact on the National Heritage values of a National Heritage place there needs to be a real chance or possibility that the Project activities will cause one or more of the National Heritage values to be lost, degraded or damaged, or notably altered, modified, obscured or diminished.
2) The Giant Australian Cuttlefish Impact Assessment considers potential impacts of Project construction and operation on the cuttlefish within the Project Area[1]. The Project Area covers temporary construction activities as well as the predicted area of potential ecological effect during operation (as a result of Project dispersion modelling, see Section 5). The assessment covers site vessel movements to and from Whyalla, but focuses on potential impacts within the Project Area itself.

Figure 1-1 shows the Upper Spencer Gulf highlighting the location of the Northern Water Project Area in relation to the CCSZ.

[1] The Project Area is defined as a 2.5 km buffer of the alignment of linear infrastructure (i.e. a 5 km wide corridor) with isolated areas identified outside of this buffer for potential laydown locations.

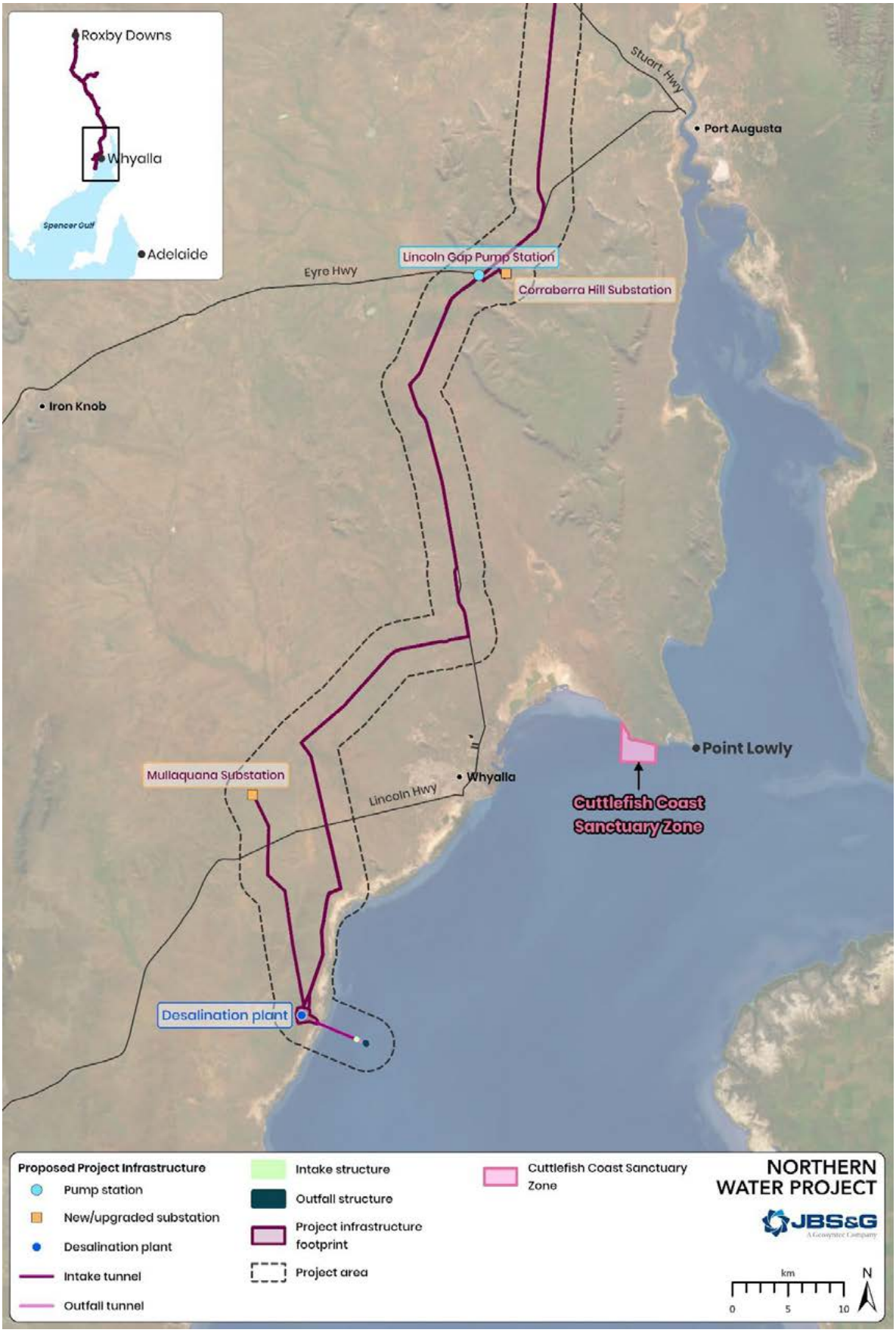


Figure 1-1: Location of the Project Area within the Upper Spencer Gulf and in relation to the Cuttlefish Coast Sanctuary Zone

1.5 Key values to be protected

The unique Giant Australian Cuttlefish aggregation area is a listed National Heritage Place under the EPBC Act as the Cuttlefish Coast Sanctuary Zone (CCSZ) and is part of the Upper Spencer Gulf Marine Park, South Australia. On 24 February 2023, the CCSZ breeding ground was declared a National Heritage Place under the EPBC Act (Place ID 106353). The CCSZ is highly valued from an ecological perspective and by the community of South Australia.

The relevant National Heritage List criteria and key values of the CCSZ as listed in the declaration in the Government Gazette (Commonwealth of Australia 2023) are:

- **Criterion (b) Possession of uncommon, rare or endangered aspects of Australia's natural or cultural history** – "*because each winter, a breeding aggregation of an iconic population of the Giant Australian Cuttlefish, involving tens of thousands of individuals occurs there. This is the largest and densest spawning aggregation of Giant Australian Cuttlefish in the world. During the event, intense competition between male cuttlefish within the aggregation leads to complex behaviours and spectacular massed displays of colour and shapeshifting. A shallow, fringing, rocky reef system within the place may offer a specific, spatially rare resource which benefits spawning success in the Giant Australian Cuttlefish*".
- **Criterion (c) Potential to yield information that will contribute to an understanding of Australia's natural or cultural history – "***because the place, and the annual aggregation of Giant Australian Cuttlefish it supports, is likely to yield further information on this iconic marine species, its habitat, and its role in broader marine ecology. The Giant Australian Cuttlefish exhibits complex and deceptive behaviour within the aggregation, using rapid camouflaging and signaling, which have provided the biological context for potential new bioinspired materials*."

2. Methodology

2.1 Review of relevant background literature

A review of existing literature was conducted to better understand Giant Australian Cuttlefish water quality requirements, breeding ecology and distribution and habitat use in northern Spencer Gulf. Consulted literature included:

- Peered-reviewed scientific articles
- Advice Note on Giant Australian Cuttlefish population estimates from SARDI (SARDI 2025a, 2026a)
- PIRSA Fishery Assessment Reports
- Olympic Dam Expansion Draft Environmental Impact Statement (BHP 2009).

2.2 Assessment methodology

There are two components to the assessment:

- Assess the impact that Project activities may have on the CCSZ, and the significance of any such impact.
- Assess impacts of Project activities on the Giant Australian Cuttlefish more broadly.

The assessment of the operation of the proposed desalination plant uses hydrodynamic modelling undertaken for the Project by both SARDI and Water Technology Pty Ltd as the basis to establish and describe the zone of influence from the desalination return water as described in Section 5.1.3.

The zone of influence from the mid-field hydrodynamic modelling, relied on the safe dilutions established in the Ecotoxicology Assessment (WQ Advice 2025) that would protect 95% and 99% of species, as per the EIS Assessment Requirements (which stipulates a requirement to assess against a 95% species protection level).

2.2.1 CCSZ Significant Impact Assessment

The assessment of the likelihood of significant impacts from the Project on the CCSZ follows the process outlined in the MNES Significant Impact Guidelines (DoE 2013).

As per the criteria provided in the guidelines (DoE 2013), the assessment interrogates whether there is a real chance or possibility that activities associated with the Project's construction or operation will cause:

- one or more of the National Heritage values to be lost
- one or more of the National Heritage values to be degraded or damaged, or
- one or more of the National Heritage values to be notably altered, modified, obscured or diminished.

All values associated with *Biological and ecological values* and *Wilderness, aesthetic, or other rare or unique environment values* and the relevant values associated with *geology or landscapes* as defined in the significant impact guidelines have been included in this assessment.

2.2.2 Giant Australian Cuttlefish Impact Assessment

The Giant Australian Cuttlefish impact assessment process for the Project construction and operation activities followed the Northern Water EIS Impact Assessment Methodology as detailed in Appendix A.

The process involved reviewing all phases of the Project to identify the ways in which the Project could potentially impact Giant Australian Cuttlefish ("potential impact events"). Duration, magnitude, and extent of each impact event were assessed assuming standard mitigation measures.

3. Review of Relevant Literature - Giant Australian Cuttlefish

3.1 Taxonomy

The Giant Australian Cuttlefish (*Ascarosepion*[2] *apama*) is classified within the cephalopod family Sepiidae. The species is endemic to waters off southern Australia between Western Australia to New South Wales.

Within their broad Australian distribution, genetic studies found that there were distinct populations in New South Wales, South Australia/Victoria and Western Australia (Hall & Fowler 2003). Further genetic studies found that cuttlefish from northern Spencer Gulf were a separate population from those in southern Spencer Gulf, and appeared to be the only population represented in the spawning aggregation at Point Lowly (Gillanders *et al.* 2016). The northern genetic cluster, or *Ascarosepion apama* Upper Spencer Gulf (USG), occurs above -33.828° (i.e. north of Arno Bay/Wallaroo) shown in Figure 3-1. The southern genetic cluster, or *Ascarosepion apama* Lower Spencer Gulf (LSG), occurs below -33.502° latitude in Spencer Gulf and Gulf St Vincent (GSV) with no breeding aggregation known for this population (Figure 3-1).

Although their distribution overlaps, research has confirmed that no gene flow is occurring between the two Spencer Gulf populations, meaning they could be considered as two separate species (Gillanders *et al.* 2016).

Although the USG cuttlefish population has not been formally described as a distinct species and is not considered to be a distinct species for the purposes of the EPBC Act (TSSC 2011), this report focuses on

[2] Species name has recently changed from '*Sepia apama*' to '*Ascarosepion apama'* (MolluscaBase 2025a). '*Sepia apama*' used in EPBC Act documentation at the time of writing.

Ascarosepion apama Upper Spencer Gulf (USG cuttlefish), which is the only population to aggregate at the EPBC Act listed Cuttlefish Coast Sanctuary Zone.

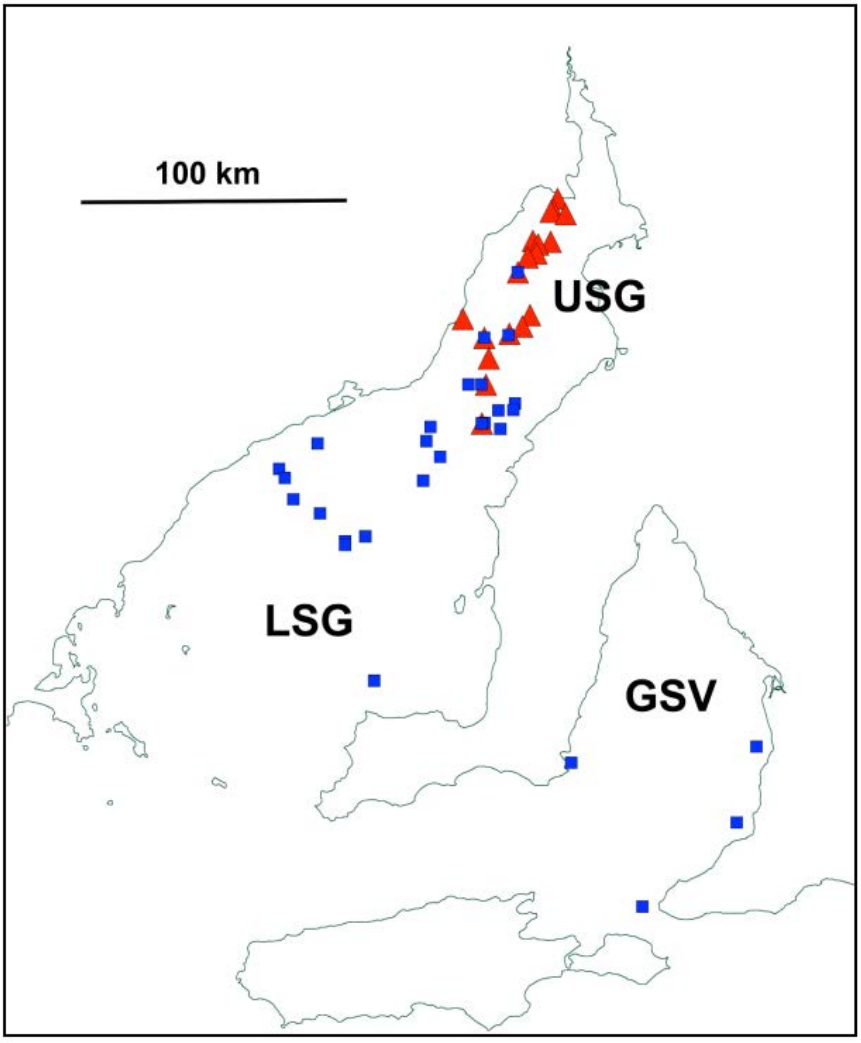


Figure 3-1: Distribution of the Giant Australian Cuttlefish USG population (red triangles) and Giant Australian Cuttlefish LSG population (blue squares)

(Source: Gillanders et al. 2016)

3.2 Description

The Giant Australian Cuttlefish is the largest species of cuttlefish in the world, with their mantle reaching up to 52 cm in length and weighing up to 10.5 kilograms (TSSC 2011). The size of Giant Australian Cuttlefish is highly variable and is generally smaller in individuals from South Australian populations compared to individuals from eastern coast populations (Gillanders and Payne 2014).

The Giant Australian Cuttlefish has a broad, flattened mantle with a forward-projecting dorsal margin that remains separate from the head. Its lateral fins run the full length of the mantle and are pale with narrow white edges. The species has ten appendages, eight arms and two longer tentacles, with distinct sucker arrangements. The tentacle clubs feature crescent-shaped rows of unevenly sized suckers, while the arms have uniform suckers arranged in multiple rows and are marked with pale bars and spots. Internally, it possesses a straight, dorsal cuttlebone that matches the mantle length and aids in buoyancy.

Cuttlefish are remarkable at camouflage and have the ability to change both colour and texture to blend into their surroundings. Specialised skin cells called chromatophores contain different pigments, and through contracting and expanding these cells, can change their colour instantaneously.

Cuttlefish have one pair of gills and a beak for feeding. Difference in beak morphology was identified between the USG and the LSG populations (Gillanders et al. 2016). Gillanders et al. (2016) suggested that this morphological variation may be related to feeding ecology and/or sexual selection.



Plate 3-1: Giant Australian Cuttlefish
(Source: www.environment.sa.gov.au/)

3.3 Distribution and abundance

3.3.1 Distribution

At the species level, the Giant Australian Cuttlefish can be found in the southern coastal waters of Australia in New South Wales, Victoria, South Australia, Tasmania and Western Australia. The distribution of the USG cuttlefish population, however, is restricted to an area of approximately 6,500 km^2 in Spencer Gulf (South Australia) north of Wallaroo and Arno Bay, above -33.828° in latitude (Gillanders et. al. 2016).

During the breeding season between May and August, mature individuals aggregate to the shallow water of Point Lowly and Backy Point (TSSC 2011). The aggregation site at Point Lowly is spatially limited to a subtidal reef along 8 km of coastline, extending 70-130 m offshore and with a 2-8 m depth. When not breeding, the cuttlefish disperse throughout northern Spencer Gulf where they are found close to the seafloor in deeper water up to 30 m deep (TSSC 2011). A migration 'pathway' for the species is not evident as it appears (predominantly from fisheries bycatch data) that the species aggregates to breeding sites from all areas around northern Spencer Gulf (Gillanders and Payne 2014).

3.3.2 Abundance

Since monitoring of the cuttlefish breeding aggregation began in 1998 cuttlefish abundance in Spencer Gulf has experienced significant fluctuations and, prior to 2026, ranged between an estimated 13,452 individuals at the aggregation site in 2013 and a peak of 247,146 individuals in 2020 (SARDI 2025). Annual surveys at the breeding aggregation site in Point Lowly and analysis of by-catch records from long-term trawl surveys have highlighted a broad-scale dramatic decline in cuttlefish abundance in Spencer Gulf from 2010 to 2013 (Steer et al. 2013; Prowse et al. 2015; SARDI 2025). The Spencer Gulf cuttlefish fishery was closed in April 2013. Figure 3-2 shows the annual peak estimates of total abundance and biomass of Giant Australian Cuttlefish, and this data shows that cuttlefish abundance then significantly increased with annual estimates consistently exceeding 100,000 cuttlefish from 2015 to 2023 (note: this statement does not imply causation between the closure and an increase in abundance). However, the abundance estimate dropped to just 63,734 individuals in 2025 (the fifth lowest values on record since 1998 (SARDI 2025)). While the abundance decreased over 2023, 2024 and 2025, it remained above the low levels that were observed in 2013. However, the 2026 population abundance estimate (SARDI 2026a) indicated there were less than 2000 Giant Australian Cuttlefish individuals in the aggregation. The harmful algal bloom that was identified in South Australian waters in 2025 is likely to have been a significant contributing factor in the observed decline in Giant Australian Cuttlefish abundance between 2025 and 2026 (SARDI 2026a).

Note that there were no records of Giant Australian Cuttlefish in marine ecology surveys (J Diversity 2026) of the Project Area (Figure 1-1). The surveys were undertaken in summer, autumn, winter and spring of 2025 and were completed prior to the algal bloom passing through the area and represent a pre-bloom condition as the bloom had not yet reached the area (J Diversity 2026).

3.4 Biology and ecology

The cuttlefish is a demersal species spending most of its time close to the sea floor, where it feeds on molluscs, crustaceans, smaller cephalopods, and fish. Gillanders *et al.* (2016) identified variations in feeding ecology between the USG and LSG populations. Diet of the LSG population was found to have a lower proportion of fish and higher proportion of crustaceans when compared to the USG population. This difference was linked to variation in prey availabilities within Spencer Gulf, with fish availability being lower in the LSG due to higher trawling pressure (Gillanders *et al.* 2016).

Every winter the USG cuttlefish population aggregates to the waters of Point Lowly to breed. This philopatric behaviour has only been observed in the USG population and could be explained by the limited extent of breeding habitat (rocky substrate) for this species in the Upper Spencer Gulf (Gillanders et al. 2016). Cuttlefish are semelparous, meaning they only breed once, generally near the end of their lifecycle.

Cuttlefish in the CCSZ display a complex mating system, which involves males transferring sperm to specific receptacles and/or to the mouth region of females. Male-male competition for female is high, with male exhibiting a range of behaviour for successful fertilisation, such as female guarding, open or hidden stealth and female mimicry (Hall and Hanlon 2002). After fertilisation, the female cuttlefish deposits its eggs underneath rocks or in rocky crevices.

Embryonic development of the cuttlefish takes between 3 and 5 months, with hatching of juvenile starting in early spring (Hall and Hanlon 2002). Juvenile and adult cuttlefish remain solitary during the non-breeding season. This species displays two alternative life cycles in both males and females (Hall et al. 2007). The first is characterised by rapid development, with cuttlefish reaching sexual maturity within 7 to 8 months and spawning in their first year. The second involves a slower growth phase, with individuals spawning in their second year. Adult cuttlefish in the second life cycle typically exhibit a greater body size compared to those in the first life cycle.

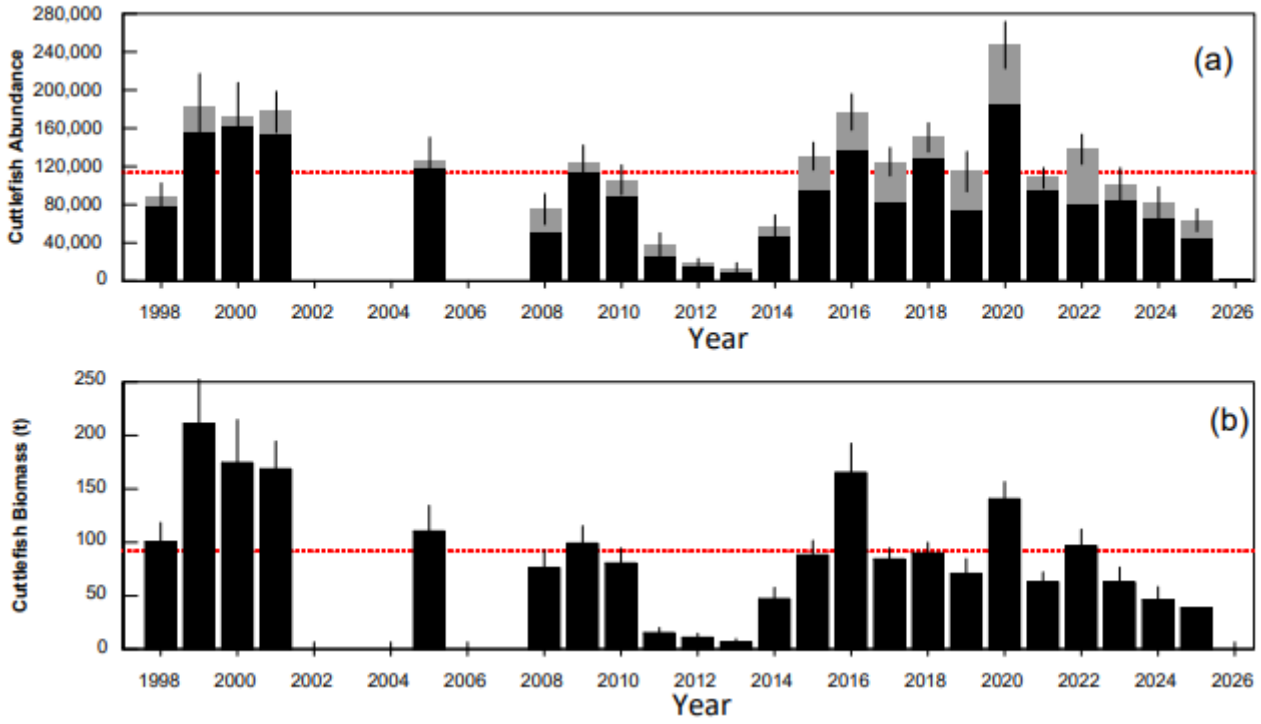


Figure 3-2: Annual peak estimates of total (a) abundance and (b) biomass of Giant Australian Cuttlefish at Point Lowly during peak spawning

(Source: SARDI 2026)[3].

3.5 Habitat

The subtidal rocky reef between Black Point and Point Lowly offers optimal spawning habitat for USG cuttlefish, with rock crevices and dens providing stable structures for egg attachment and refuge for hatchlings. A study of the rocky reef extending from Black Point to False Bay found that dens used by the cuttlefish were on average 42.5 cm deep, with entrances predominantly oriented towards the south-eastern quadrant (Steer 2015). Other areas of sub-tidal rocky reef were identified elsewhere in northern Spencer Gulf, however those were not associated with cuttlefish breeding or spawning activities (Steer 2015). Factors such as high energy wave action may affect the suitability of some of these sites as spawning habitats (Steer 2015). Higher density of the fast-growing macroalgae *Hincksia sordida* was also suggested to affect spawning habitat suitability through preventing cuttlefish from spawning and increasing fouling of cuttlefish eggs, however further investigation is required to corroborate this hypothesis (BHP 2009; Steer 2013; Steer 2015).

Artificial structures in the vicinity of the cuttlefish mass aggregation site were also observed to provide suitable spawning substrate for the cuttlefish and could be used to promote spawning activity. The OneSteel sea wall in Whyalla has been found to support high densities of cuttlefish during the breeding season (Hall and Fowler 2003). Similarly, cuttlefish eggs were found under concrete pavers deployed by BHP in Fitzgerald Bay (Plate 3-2). However, artificial reefs deployed at five sites elsewhere in northern Spencer Gulf were unsuccessful in supporting cuttlefish spawning (Steer 2015).

Outside of the breeding season, USG Cuttlefish can be found in various benthic habitats such as seagrass, rocky reefs, and sandy substrates throughout northern Spencer Gulf (Hall and Fowler 2003, Miller et al. 2014). Being a demersal species, it spends most of its time close to the sea floor.

[3] In (a), black and grey abundance bars represent shallow and deep transects respectively. The red dashed lines represent the overall average between 1998 and 2026 (SARDI 2026).

The marine Project Area off the coast of Mullaquana Station does not provide important, unique or breeding habitat for USG cuttlefish and the species is not known to regularly utilise the area (based on data from seasonal surveys undertaken for the Project by J Diversity (2026)).



Plate 3-2: Giant Australian Cuttlefish eggs under concrete pavers deployed in Fitzgerald Bay
(Source: BHP 2009)

3.6 Water quality requirements

Cuttlefish are unable to control salt concentrations in their body fluids (i.e. cannot osmoregulate), which makes them vulnerable to variations in water salinity (Gillanders and Payne 2014). Low water salinity has been linked with elevated mortality rate and cellular damage in the Cuttlefish *Acanthosepion pharaonis*[4] (Xin *et al*. 2022), and high salinity was shown to negatively impact cuttlefish embryo development and egg hatching (Gillanders *et al*. 2016). Tolerance levels to water salinity in cuttlefish appears to be species and population specific, with different species showing different optimal ranges.

The USG cuttlefish population is adapted to the higher levels of salinity found in Upper Spencer Gulf, compared with the lower Gulf (Steer *et al.* 2013). With minimal freshwater inputs and high evaporation, Spencer Gulf behaves as an inverse estuary where salinity increases along a gradient from the south to north, with salinities typical of oceanic water (~36 g/L) in the south and salinities in the north reaching ~48 g/L (Nunes Vas *et al*. 1990, Heldt and Hooper 2023). The salinity gradient in Spencer Gulf has been proposed as a possible mechanism for the maintenance of population differentiation between the Upper and Lower Spencer Gulf populations (Gillanders et al. 2016).

Embryos of the USG cuttlefish population may be adversely affected by elevated salinity levels. While mobile adult and juvenile cuttlefish may move away from an area with unsuitable conditions, embryos are attached to the substrate and therefore are particularly vulnerable to variation in water chemistry. Temperature and salinity experiments conducted on cuttlefish eggs collected from the breeding aggregation site in False Bay found that embryo survival began to decline at salinities above 44 parts per thousand (ppt), with no eggs surviving to hatching at salinities above 48 ppt (Figure 3-3) (Gillanders et al. 2016). The negative effects of high salinity on embryo development were further intensified by elevated water temperatures, with decreased survival evident at 24°C. Results of temperature spike experiments on embryos in tanks at an ambient temperature of 18°C indicated that spikes more than 2°C above ambient temperature may negatively impact development of embryos (Gillanders *et al*. 2016).

Research on the effects of desalination brine on USG Cuttlefish also found that elevated salinities of 45 ppt and above negatively impacted cuttlefish embryos. In experimental tanks containing varying levels of locally sourced desalination brine – 39 ppt, 40 ppt, 45 ppt, 50 ppt and 55 ppt - cuttlefish embryo survival rates

[4] Species name recently changed from '*Sepia pharaonis*' (MolluscaBase 2025b)

decreased markedly with increasing salinity (Dupavillon and Gillanders 2009). At the highest concentrations tested (50 ppt and 55ppt), no hatchlings survived (Figure 3-3). Additionally, smaller mantle lengths and reduced weights of hatched juveniles were observed in tanks with higher salinities. This study also suggested that chemicals and contaminants (e.g. anti-fouling agents, chlorine and acid) present in typical desalination brine may further impact the cuttlefish. Previous studies have found that the toxicity of desalination brines is driven by the high salt concentrations, and chemical additives contribute very little to the toxicity for most species (WQ Advice 2025).

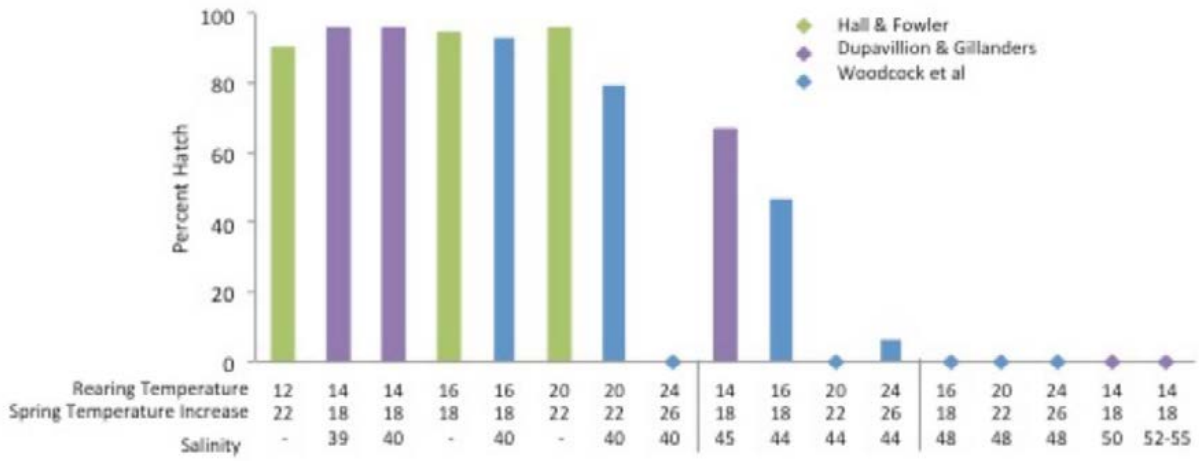


Figure 3-3: Percent of embryos successfully hatching under different temperature and salinity treatments
(Source: Gillanders et al. 2016)

Gillanders & Stockbridge (2022) noted that, while there are data on the effects of desalination brine on embryo/larval cuttlefish, there is no knowledge on the effects on juvenile or adult cuttlefish and how they may be affected by desalination brine in the USG as they move up the gulf to the breeding aggregation site. The fact that adults and juveniles are mobile, as opposed to the sessile embryos, suggests they are less vulnerable to localised increases in salinity as they have the ability to swim away.

The likelihood of ecotoxicological impact can depend on various factors such as the salinity of the return water, physico-chemical properties, the salinity of the receiving environment, the bathymetry and hydrodynamics of the receiving environment, the nature of the discharge and its dispersion, and the specific sensitivity of organisms in the receiving environment (WQAdvice 2025, Panagopoulos & Haralambous 2020, Gillanders & Stockbridge 2022).

Elevated turbidity is not known to have an adverse effect on Giant Australian Cuttlefish embryos, thought to be partly due to their orientation (Plate 3-2) being less susceptible to accumulation of sediment (Woodcock *et al* 2014). Woodcock *et al.* (2014) found no effect of turbidity on the embryonic development and hatching success of cuttlefish embryos exposed to tank turbidity of 70 NTU (using bentonite clay), which is well above the average background turbidity recorded at the Project site of approximately 1 NTU (from the results of monthly water quality monitoring between June 2025 and May 2026). Although cuttlefish may rely on visual cues for prey and predator detection, navigation and reproductive behaviours, studies from Europe have shown that cuttlefish are able to respond to predators in turbid waters (Cartron *et al.* 2013a and Cartron *et al*. 2013b, cited in Woodcock *et al.* 2014).

4. Desalination Plant Overview

The proposed desalination plant and associated marine infrastructure location is on the western shore of the USG at Mullaquana Station, ~25 km south-west from the centre of Whyalla. The site was identified through a comprehensive site selection process as the most appropriate for the desalination plant to successfully support regional industrial opportunities, and best balances social, environment and economic opportunities and risks. A key consideration in the site identification process was that the location was an adequate distance from the CCSZ to ensure the desalination plant return water did not interact with this National Heritage Place.

4.1 Subject site

The subject site for the desalination plant is at Mullaquana Station in the Upper Spencer Gulf. The Upper Spencer Gulf is an inverse estuary (where evaporation exceeds freshwater inputs) with elevated salinities in the upper reaches (Nunes Vas *et al.* 1990).

The USG supports commercial fisheries, recreational boating and fishing, aquaculture, shipping, industry and tourism. There are also areas managed for conservation purposes under the *Marine Parks Act 2007*. The USG Marine Park is located approximately 8 km north and the Franklin Harbour Marine Park is located approximately 11 km south of the proposed desalination plant marine intake and outfall infrastructure.

The CCSZ National Heritage Place is located adjacent to Port Bonython, within the USG Marine Park approximately 34 km north-east from the proposed location of the desalination plant marine intake and outfall infrastructure.

The benthic habitats at the site are generally sandy and flat with extensive seagrass beds and intertidal mangroves. Habitat mapping has been undertaken for the project and is presented in Figure 4-1. Habitats within the marine Project Area vary along a depth gradient and were generally consistent within each depth range. Mangroves are present in the intertidal zone with areas of bare sand, *Posidonia* seagrasses dominate the subtidal area to 13 m where the seagrass then transitions to *Zostera* sp. out to approximately 15 - 16 m depth whereafter bare sand dominates. Bryozoans, algae and other invertebrates are present within sandy areas.

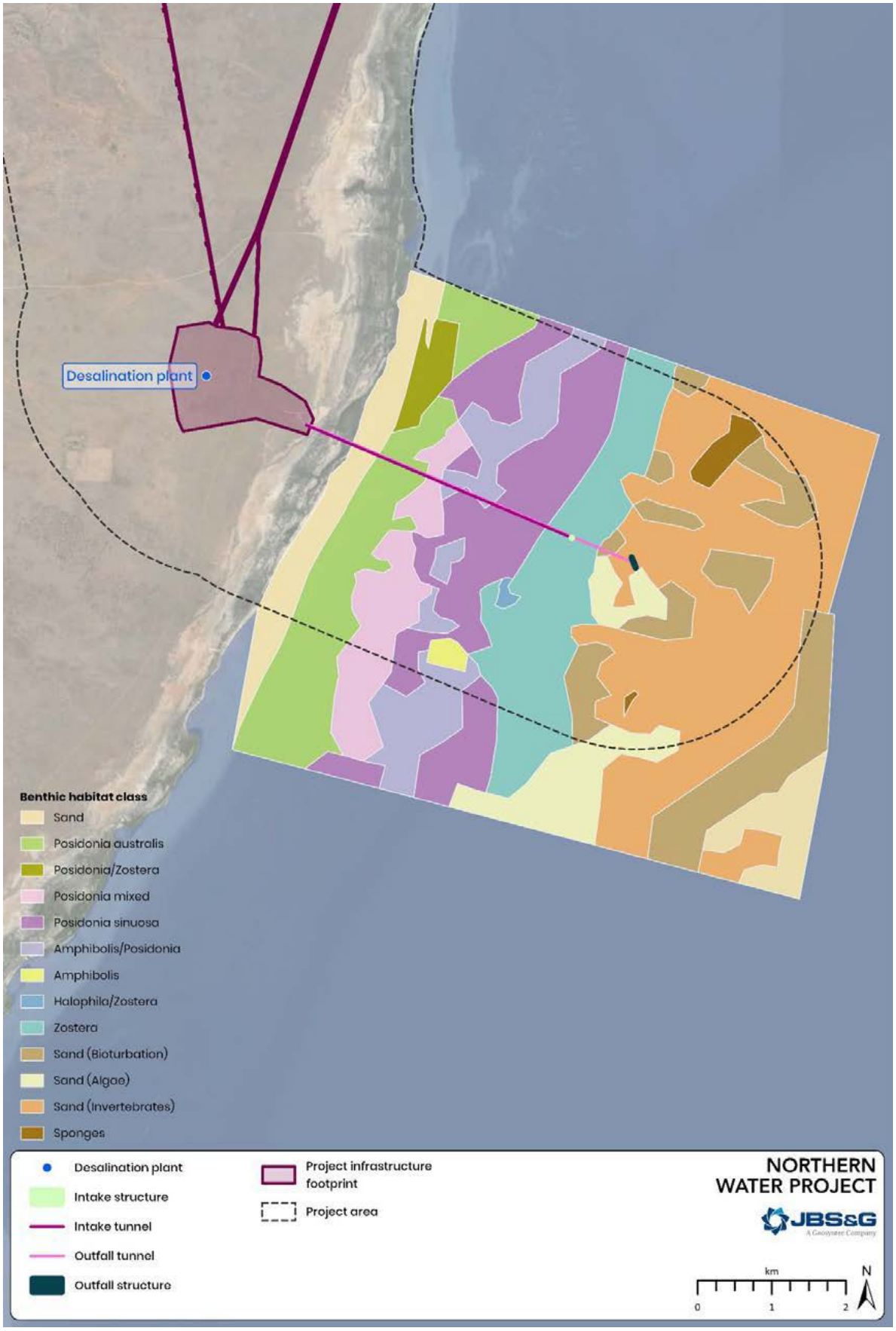


Figure 4-1: Marine benthic habitats surveyed in the Project Area

4.2 Desalination plant and marine infrastructure

The desalination plant includes an intake pump station, the desalination treatment process (pre-treatment and reverse osmosis membranes), product water storage tank(s) and/or pond(s) and ancillary equipment, facilities and power infrastructure at the plant. One of the five transfer system pump stations is also located at the desalination plant site.

Based on the Reference Design (GHD 2025), the Northern Water desalination infrastructure comprises the following key elements:

- a 51 GL/year seawater reverse osmosis desalination plant located at Mullaquana Station approximately 25 km south-west of Whyalla
- marine intake and outfall tunnel/s extending under the seabed no less than 3 km (intake) and no more than 5 km (outfall) from the intake pump station
- a water intake head located at a minimum 13 m water depth from the Lowest Astronomical Tide (LAT)
- an outfall riser with diffusers located at minimum 17 m water depth from LAT
- diffuser structures designed to achieve adequate mixing at the outfall for effective dispersal of return water into Spencer Gulf.

4.3 Intake and outfall pipe

Marine infrastructure, including tunnels, intake and outfall structures, may extend as far as 5 km from shore, with no infrastructure to be constructed on the seabed within 3 km from the intake pump station. The marine intake and outfall pipes will be constructed under the seabed using tunnel boring machines (TBMs). The marine infrastructure on the seabed would comprise the intake head and the outfall riser with diffuser array.

The exact length of tunnels and location of the intake and outfall structures will be refined with inputs from hydrodynamic modelling and detailed design, but as a minimum the intake structure is expected to be no less than 3 km from the intake pump station and the outfall structure is expected to be no more than 5 km from the intake pump station. The outfall and intake structures will also be sufficient distance apart to avoid recirculation of seawater concentrate back into the plant.

Alternate marine intake and outfall construction methodologies including construction of a single tunnel, rather than two tunnels, and the use of float and sink construction seaward of the intake may also be considered.

4.3.1 Intake structure

The intake structure facilitates the extraction of seawater for treatment within the desalination plant. Intakes are typically sited to target optimal substrate, benthic, and seawater conditions, and minimise environmental impacts. Intake screens are generally elevated above the seabed to minimise entrainment of sediments and demersal biota. With an intake screen velocity of 0.15 m/s the design minimises entrainment and allows mobile organisms to swim away.

The intake is estimated to be 11.5 m diameter and with the intake grill elevated 3 m from seabed, the total height of the structure is set at 6 m from the seabed (Figure 4-2). The intake location will be a minimum water depth of 13 m (LAT).



Figure 4-2. Visual representation of intake structure

4.3.2 Outfall and diffuser structure

The desalination plant will discharge the return water back into Spencer Gulf through an outfall tunnel containing a pipe connected to a diffuser array. The diffuser array will be connected to the outfall tunnel and pipe via a riser. A riser cap would be fitted to the riser, and the riser array pipe connected into the riser via a tee-arrangement housing eight diffuser nozzles. The diffuser array would be secured to the seabed using piles, rather than being excavated into the seabed.

The outfall structure and diffuser array (Figure 4-3) has been designed to achieve effective dispersal of return water. Section 5 describes the results of preliminary modelling of the return water.



Figure 4-3: Visual representation of outfall diffuser

5. Return Water Characteristics and Dispersion

5.1.1 Key Project operational assumptions

The key assumptions related to Project operation upon which the assessment has been based include:

- Desalination plant design capacity: 51 GL/y
- The system is designed as a one pass reverse osmosis (RO) system operating at 40% overall plant recovery
- Intake velocity: 0.15 metres per second (m/s)

- Mullaquana Station ambient salinity ranges from 38-41 practical salinity units (PSU) (<41 PSU 95% of the time)
- Outfall salinity: 65.77 - 70.95 PSU (modelling uses upper limit)
- Outfall return water is largely continuous and contains antiscalants, biocides and cleaning products.

5.1.2 Safe Dilutions

A desktop ecotoxicity assessment was conducted to assess the potential impact of the return water on marine biota in the marine Project Area or beyond. The assessment involved a comprehensive review of existing desalination plant ecotoxicity studies and compilation of relevant toxicity datasets for the Northern Water Project Area.

Return water 'safe dilutions' that would protect 99% and 95% of species were determined using the toxicity values for the most sensitive species in the final dataset (i.e. the most conservative data was applied).

The preliminary safe dilutions established are 1.9% (1 in 53) and 3.9% (1 in 26) for protection of 99% and 95% of species, respectively. These preliminary safe dilutions are expected to be conservative (i.e. protective) for the proposed site at Mullaquana Station (WQ Advice 2025).

5.1.3 Results of Hydrodynamic Modelling

Modelling of return water dispersion involved a three-tiered modelling approach, including:

- A nearfield model focused on effects of the discharge in the immediate vicinity of the point of discharge (from 1 to 100 m from the outfall) (Water Technology 2025)
- A midfield model focused on dispersion of the return water in the local vicinity of the point of discharge (from 50 m to 10 km) (Water Technology 2026)
- A farfield model focusing on Spencer Gulf-wide effects of the desalination plant (from 10's to 100's of km) (SARDI 2026b).

Dispersion modelling was conservatively based on upper end of ambient range at 41 PSU, and upper end of return water salinity (70.95 PSU).

Key modelling outputs:

Near-field (Water Technology 2025): The proposed outfall design will create sufficient mixing to achieve compliance with prescribed environmental criteria. The nearfield modelling showed that salinity beyond the immediate vicinity of the marine discharge diffusers would remain within the range of natural seasonal variability. Dilutions were generally above 99% species protection trigger value criterion, even under low current and low flow conditions. Dilutions below the criterion occurred at distances considerably less than the edge of the nearfield mixing zone (100 m from the outfall).

Mid-field (Water Technology 2026):

Marine hydrodynamic modelling undertaken for the Project demonstrates that return water dilution would be sufficiently high to protect the marine environment/species and achieve the prescribed environmental criteria of 95% species protection (as per EIS Assessment Requirements). Using the recommended dilutions from the ecotoxicology assessment, key preliminary findings from the mid-field modelling, and relevant to this cuttlefish assessment, are as follows:

- 95% of species would be protected from impacts associated with the return water in close proximity to the diffusers (see EIS for further detail)
- Salinity beyond the immediate vicinity of the diffusers would remain within the range of natural seasonal variability

- In addition to the achieving 95% species protection levels in close proximity to the return water discharge, and that salinity changes are within natural seasonal variability, fish, cuttlefish and other mobile marine fauna generally have the ability to move away from the diffuser or osmoregulate in response to elevated salinity
- Total suspended solids modelling results conclude that discharging desalination plant backwash into the outfall will have no impact on the receiving environment as modelled changes in total suspended solids are small and within natural variation
- The predicted impacts to marine biota from dilutions of return water during dodge tides are considered negligible.

Figure 5-1, Figure 5-2 and Figure 5-3 show key preliminary outputs from the mid-field hydrodynamic model that support and justify the key preliminary finding above.

Far-field (SARDI 2026b): Gulf wide hydrodynamic modelling concluded that discharges do not accumulate over time and that far-field salinity changes are within the range of natural salinity variability within the gulf. The tides at Mullaquana Station are expected to be effective in dispersing and diluting desalination discharges in the far-field (> 3-5km).

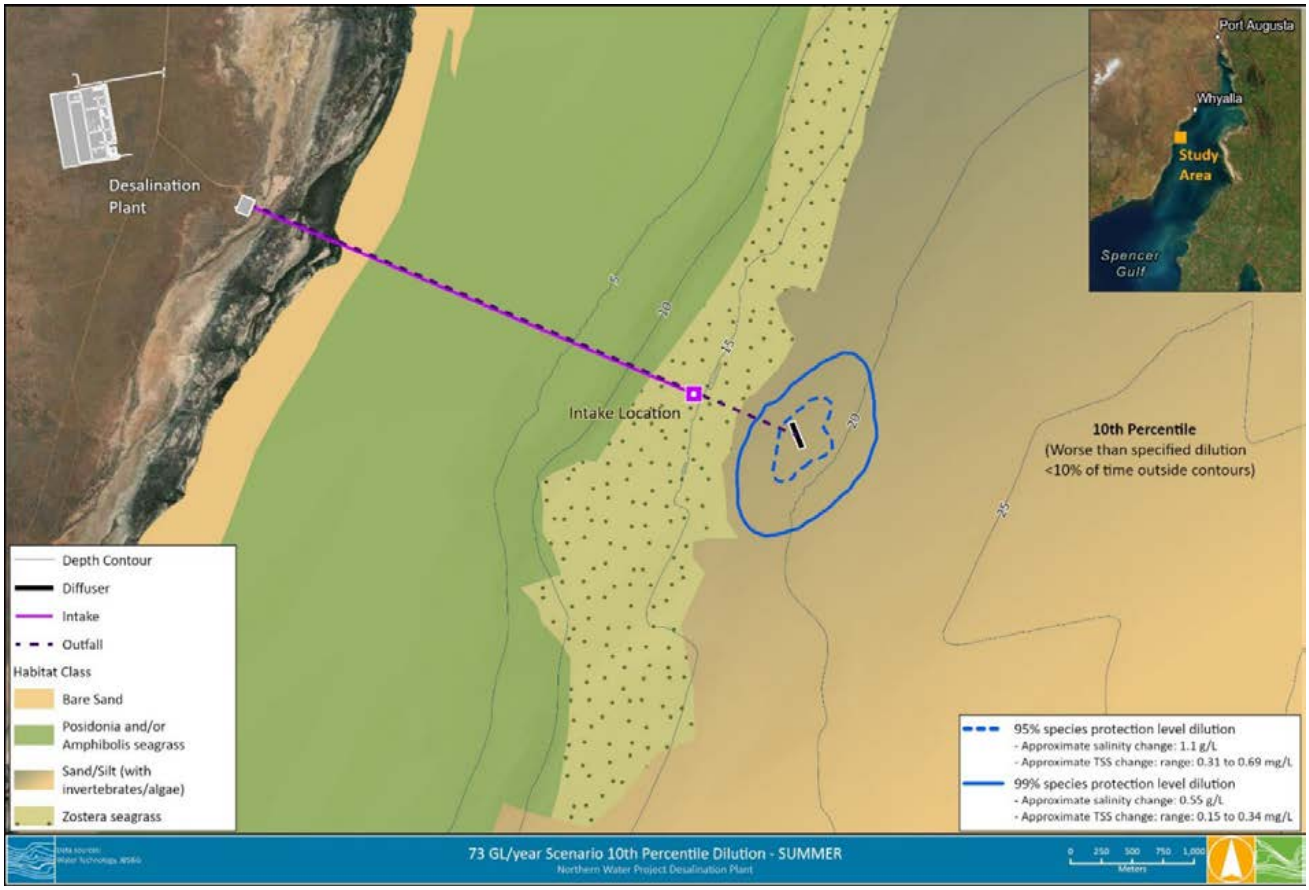


Figure 5-1: 73GL/year dilution modelling output for Summer (10th percentile i.e. worse than specified dilution less than 10% of the time outside contours) and TSS change contours

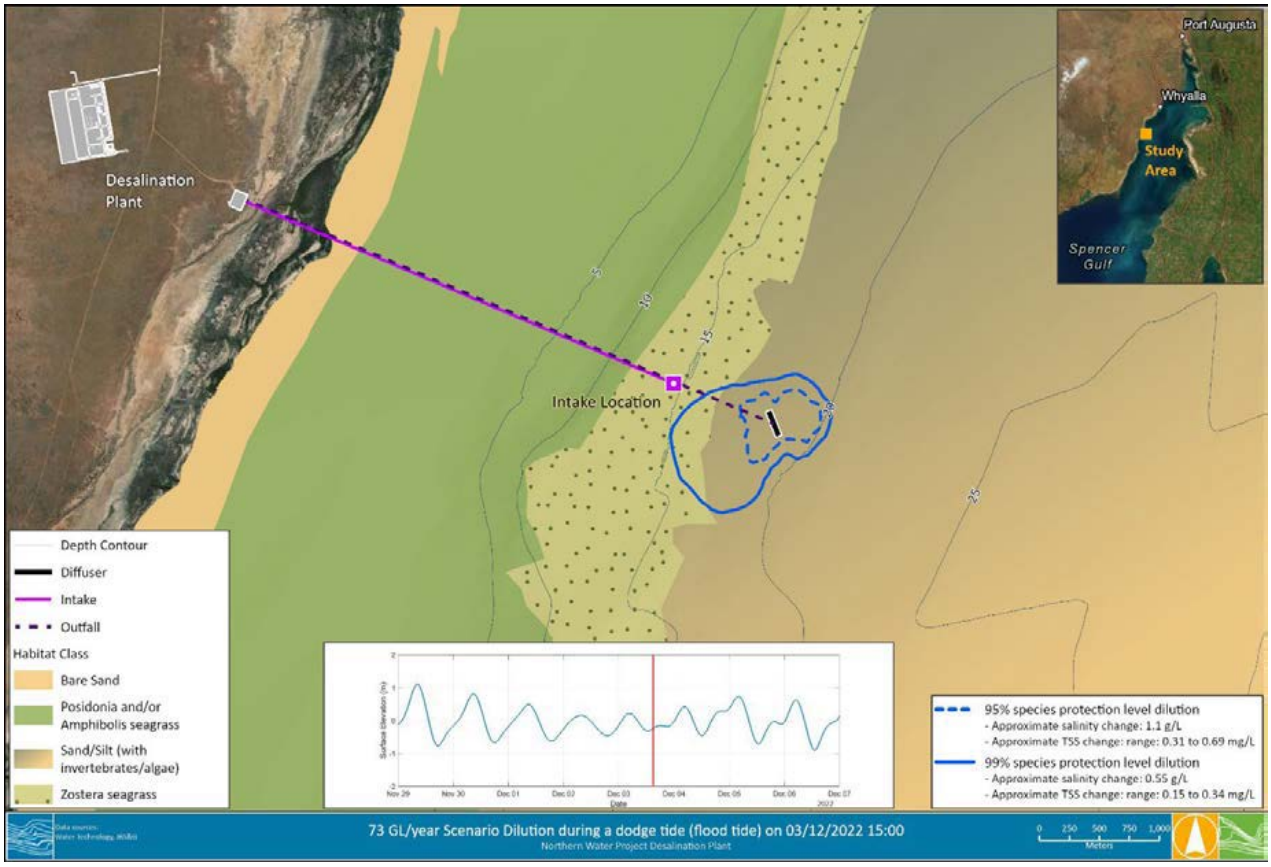


Figure 5-2: Instantaneous dilution during a dodge tide (flood tide) and TSS change contours

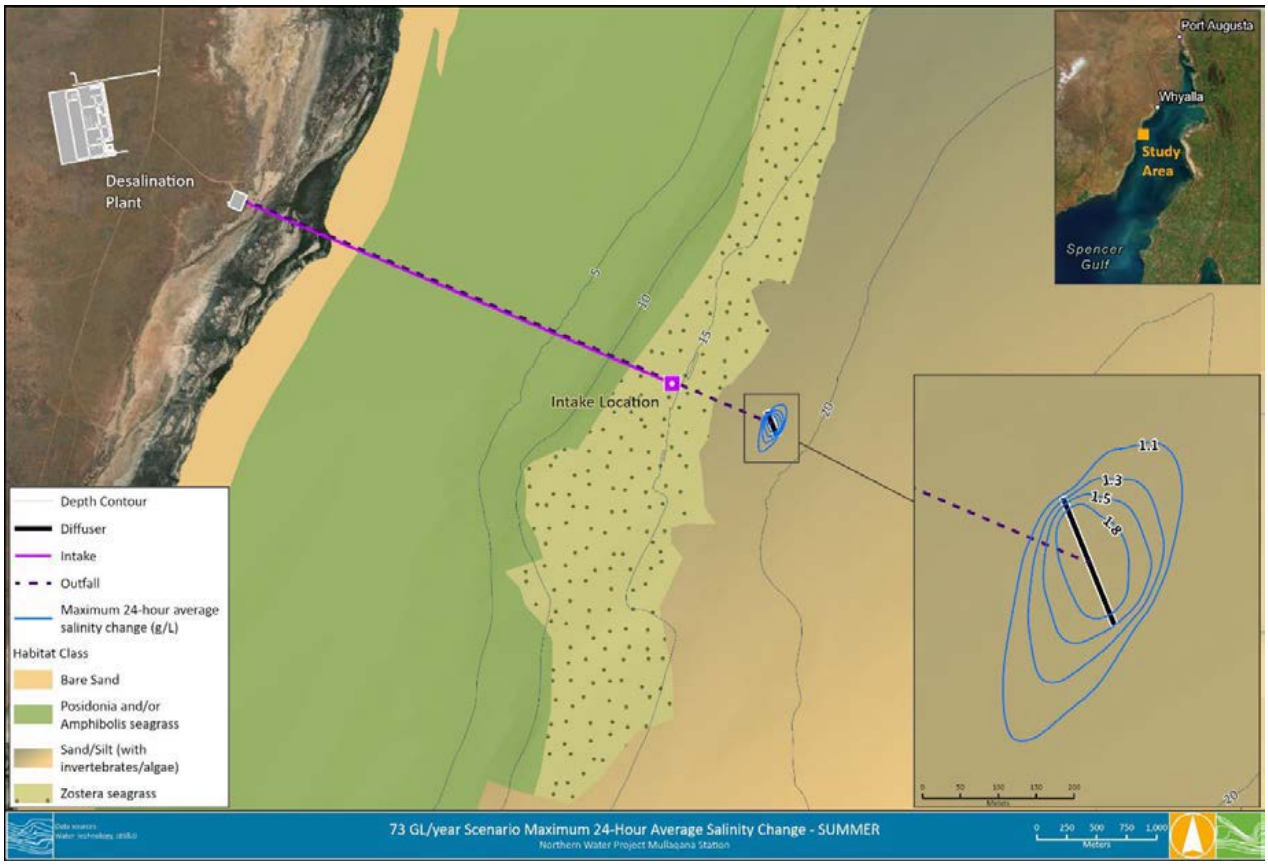


Figure 5-3: 73GL/year – Maximum 24-hour Average Salinity Change (Summer)

6. CCSZ Assessment

The CCSZ assessment follows the EPBC guidelines for assessment of National Heritage places (DoE 2013).

6.1 Identified sources of potential impact and key findings

The assessment of potential impacts to the National Heritage values of the CCSZ provided in Section 6.2 identified the following potential sources for impact and key findings:

- **Desalination plant return water during operation potentially impacting water quality in the CCSZ.** The distance between the proposed desalination plant and the CCSZ means there is no real chance or possibility that return water from operational activities could adversely affect the CCSZ. This was confirmed with hydrodynamic modelling of the return water (see Figure 5-1, Figure 5-2 and Figure 5-3).
- **Noise from construction or operational activities potentially impacting cuttlefish within the CCSZ**. The distance between construction activities and the CCSZ means there is no real chance or possibility that noise from construction or operational activities could adversely affect the CCSZ.
- **Increased shipping from the desalination site to Whyalla during construction potentially impacting cuttlefish within the CCSZ**. This was assessed as negligible due to the anticipated low traffic volumes relative to existing traffic near the CCSZ, the temporary nature of the increased shipping activity, and the implementation of appropriate control measures.
- **Spills of fuel or chemicals in the Project Area potentially impacting the CCSZ**. This risk is regarded as low given the large distance to the CCSZ and the low quantities of fuel and chemicals used.
- **Disturbance of sediments or erosion on the terrestrial pipeline alignment during construction and operation potentially impacting water quality within the CCSZ through sediment transport**. The distance between the location of the Project Area and the CCSZ mean there is no real chance or possibility that sedimentation from the Project Area could adversely affect the CCSZ.

There are no direct impacts from the Project on the CCSZ. At its closest point, the CCSZ is ~34 km from the proposed desalination plant and more than 15 km from the nearest proposed terrestrial infrastructure (water transfer system and transmission line).

6.2 CCSZ impact assessment

6.2.1 Return water during desalination plant operation

There is no real chance or possibility that return water from operational activities could adversely affect the CCSZ.

The operation of the desalination plant has the potential to locally change water quality through elevated salinity and water temperature of the return water discharge, and inclusion of low levels of chemical contaminants including antiscalants, biocides and cleaning agents from plant processing.

The values of the CCSZ would not be affected by the desalination return water, as the desalination plant is ~34 km from the CCSZ. This is well outside the zone of any direct or indirect impact to the marine environment from the desalination plant.

Preliminary mid-field modelling shows salinity at the desalination plant marine discharge returns to within the normal range of local ambient conditions within the immediate vicinity of the outfall. The mid-field hydrodynamic modelling indicates that 95% species protection would be achieved in close proximity to the diffusers.

Discharge of cleaning chemicals or antiscalants in the return water is not expected to affect marine biota, as these additives generally make a negligible contribution to the toxicity of return water. The cleaning chemicals and antiscalants proposed to be used for the Project are in line with those approved by the South Australian Environment Protection Authority (the EPA) for use at the Adelaide desalination plant. Previous studies have found that the toxicity of desalination plant return water is driven by the high salt concentrations, and chemical additives contribute very little to the toxicity for most species (WQ Advice 2025). Therefore, the safe dilution rates derived for salinity ecotoxicity are likely to also be protective for antiscalant and clean-in-place (CIP) chemical toxicity (WQ Advice 2025).

The EPA has strict compliance limits and monitoring requirements as part of the environmental licence under the South Australian Environment Protection Act (EP Act) to operate a desalination plant. The Project would implement a program for operational compliance and water quality monitoring to meet the requirements of the EPA and to reduce environmental risk from the operation of the desalination plant and associated return water.

A key consideration in the Project planning phase was to ensure the desalination plant return water did not interact with this National Heritage Place.

Far-field modelling shows the desalination return water from the outfall at Mullaquana Station has no impact on salinity in the vicinity of the CCSZ or other areas in the Upper Spencer Gulf. Impact of desalination return water to the cuttlefish within the CCSZ is therefore considered unlikely.

In addition, no Cuttlefish were recorded in surveys of the Project Area off Mullaquana Station. The site does not provide suitable habitat for Cuttlefish egg attachment, there is no suggestion of a migration pathway and no evidence of Cuttlefish aggregation at the site.

6.2.2 Noise from construction and operational activities

There is no real chance or possibility that noise from construction and operational activities could adversely affect cuttlefish within the CCSZ.

Construction of the Project will generate both terrestrial and underwater noise which is not expected to be audible at the CCSZ. Earthmoving equipment, vehicles delivering pipeline and fill material, cranes and other machinery will be the main terrestrial noise generating sources. There is no risk that terrestrial construction noise could adversely impact cuttlefish underwater in the CCSZ given the separation distance from the Project Area of at least 15 km to the nearest terrestrial pipeline location.

Underwater construction activities will be undertaken at the desalination plant site and will involve tunnel boring, drilling, support vessels and impact piling. An underwater noise and vibration assessment was undertaken for the Project, which conservatively identified the potential for physiological and/or behavioural effect on cephalopods (cuttlefish) within a 4,750 m radius of impact piling activities (Resonate 2025). Impact piling will be of limited duration during construction, and undertaken 34 km from the CCSZ and is therefore not expected to impact cuttlefish in the CCSZ, or developing cuttlefish embryos. Noise associated with increased vessel movements is covered in 6.2.3.

The main terrestrial operational noise sources include the operation of the desalination plant and the pump stations. Water-borne operational noise will derive from the intake pump. The Project infrastructure is located over 15 kilometres from the CCSZ at its nearest point (terrestrial). As such, it is considered unlikely that terrestrial operational noise would be audible at the CCSZ. Operational underwater noise from the operation of the intake pump was assessed as unlikely to impact cephalopods (Resonate 2026).

Generation of underwater noise during operation is limited to operation of the intake pumps. These are ~34 km from the CCSZ and there is no real chance or possibility of impacts on the CCSZ.

6.2.3 Increased vessel movements during construction

Increased marine traffic is not expected to adversely affect cuttlefish within the CCSZ during construction.

Project-related marine traffic from the desalination plant site to and from Whyalla Port will increase temporarily during construction of the desalination plant. Construction of the intake and outfall structures will require a jack-up barge (JUB) stationed at the site, plus support vessels (supply barge, up to 3 tugs, line boat, safety boat and spoil barge). Construction to install the risers is expected to take approximately 21 months. Approximately 3,000 m^3 of spoil removed for construction of the risers will be transported from the desalination site to Whyalla via a barge, which would generate an estimated 6 to 7 trips in total. The expected daily increase in vessel movements during marine construction is considered negligible in relation to the existing/current volume of shipping movements in the USG (see assessment of marine traffic in the Project EIS). The fact the USG cuttlefish aggregate adjacent to the Port Bonython jetty, an existing hydrocarbon import/export terminal that has been operating for over 40 years, suggests the USG cuttlefish are tolerant of marine traffic.

No impact to the CCSZ from Project construction vessels is anticipated. Whyalla Port is over 10 km from the CCSZ. The CCSZ is north of Whyalla Port and as such the transport route will not be within 10 km of the CCSZ. The temporary increase in the number of vessels may increase the possibility of introducing invasive marine species that could negatively impact cuttlefish and their breeding habitat in the CCSZ. However, the risk of introducing or contributing to the spread of invasive marine species in the CCSZ will be adequately controlled and managed as per the measures detailed in the Invasive Marine Species Risk Assessment undertaken for the Project (Aquatic Biosecurity 2025). The existing/current volume of marine traffic in the vicinity of the USG cuttlefish aggregation area suggests that the population will not be impacted by a small increase more than 10 km from the CCSZ.

There is no real chance or possibility that Project related marine traffic could adversely impact the CCSZ.

6.2.4 Risk of spills in the Project Area

The risk that potential spills of fuel or chemicals in the Project Area reach the CCSZ and adversely impact cuttlefish is low.

Construction of the Project will involve relatively small quantities of chemicals and the risks to surface and marine waters associated with minor spills is generally low. Potential pathways of spills of fuel or chemicals into the environment include:

- chemicals and hydrocarbons used during construction, such as fuels, lubricants, and cleaning agents, could be accidentally released into the marine environment from terrestrial or marine-based construction activities
- accidental discharges from temporary facilities or construction runoff.

Management measures will be implemented to minimise the risk of spills to as low as reasonably practicable, and will include:

- avoid terrestrial vehicle refuelling in close proximity to watercourses
- implement terrestrial and marine measures for fuel, oil and chemical management, spill prevention, response and clean-up, trench/dredge dewatering, hydrotest water disposal and management of contaminated water (e.g. leachate from acid sulfate soils)
- support vessels and safety boats onsite during construction prepared to deploy countermeasures in the event of an accidental oil/fuel spill.
- conduct regular inspections and maintenance of equipment to prevent failures during terrestrial and marine-based construction activities
- undertake field screening to determine presence of acid sulfate soils if excavating in a risk area and apply procedures as needed.

The risk of any spill of fuel or chemicals impacting the CCSZ is further mitigated by the distance of the Project Area from the CCSZ. In addition, the Project terrestrial infrastructure would not disturb any watercourse that discharges directly into Spencer Gulf.

Given the above, the risk to the CCSZ from spills of chemicals or fuels is considered to be low.

6.2.5 Sedimentation and water quality within the CCSZ

There is no real chance or possibility that sedimentation from the Project Area could adversely impact water quality within the CCSZ

Sedimentation from the Project construction is unlikely to impact the CCSZ due to the distance of the Project Area from the CCSZ, the sediment and erosion controls in place during construction, and the limited duration of construction activities.

Sedimentation in the marine environment is not anticipated as a result of terrestrial construction of the desalination plant. Detention basins will be installed and a site-specific Soil Erosion and Drainage Management Plan (SEDMP) developed to contain and manage stormwater onsite and minimise the risk of sedimentation at the desalination plant site to as low as reasonably practicable.

The proposed terrestrial pipeline and transmission line are located more than 15 km from the CCSZ and are not crossed by any watercourses that discharge directly into the CCSZ. There is no pathway for terrestrial elements of the Project to result in surface water run off to the marine environment at or near the CCSZ. Notwithstanding that, during Project construction, appropriate sediment and erosion controls (such as berms on slopes, hay bales and geotextile fencing) will be in place to control erosion in accordance with the Stormwater Management Plan (GHD, 2025).

Seabed disturbance during construction of the intake and outfall risers may locally increase turbidity and suspended sediment in the Project Area (~34 km from the CCSZ), however seabed disturbance has been minimised through constructing the tunnel/s below the seabed via tunnel boring machine (TBM). For operation, preliminary results from the mid-field hydrodynamic model indicate that modelled changes in total suspended solids are small and within natural variation within close proximity of the outfall (see Figure 5-1 and Figure 5-2).

Turbidity management measures will be incorporated into a Marine and Coastal Construction Environment Management Plan, Marine Water Quality and Monitoring Management Plan and Dredge Management Plan. Implementation of control measures will minimise potential impacts of turbidity and silt deposition arising from installation of the intake and outfall risers.

There is no real chance or possibility that sedimentation from the Project Area could adversely affect the CCSZ.

6.3 Cumulative impacts

The CCSZ Assessment in Section 6 concludes that the Northern Water desalination plant and pipeline would not impact the CCSZ, hence, there are no cumulative impacts that arise from the Project.

The USG has a strong industrial history with existing iron and steel making facilities and port infrastructure. Heavy industry and resources (mining, steel, energy) support the coastal cities of Whyalla, Port Pirie and Port Augusta, alongside significant commercial and recreational fishing (prawns, crabs, snapper, whiting), as well as tourism operations. With the heavy focus on USG as an industrial hub, there are several proposed developments in the area. The proposed Port Bonython Hydrogen Hub is proposed adjacent the CCSZ, and would involve an export facility and large-scale clean hydrogen production precinct.

Cuttlefish have co-existed with industry in the USG, particularly near Whyalla and Port Bonython where the following activities occur, shipping and ports, ecotourism, commercial fishing (with restrictions) and the Whyalla Steelworks. The CCSZ breeding aggregation area is immediately adjacent an existing hydrocarbon import/export terminal that has been operating for over 40 years.

The Whyalla Steelworks is also set to be upgraded, with a large investment from the South Australian Government.

Also of note, Alligator Energy is currently undertaking Field Recovery Trial approximately 1.4 km north-west of the desalination plant site at Mullaquana Station for their Samphire Uranium Project (S Uranium Pty Ltd, RL 137).

Projects in the vicinity of the CCSZ that have been referred or approved under the EPBC Act but are yet to be developed include:

- Nonowie Wind Farm (Tilt Renewables DevCo Pty Ltd, EPBC 2025/10164 & REILA 25)
- Middleback Ranges MEP 2 Project (GFG / SIMEC, EPBC 2024/09878)
- Hydrogen Jobs Plan (including all approved ancillary infrastructure) (DEM, EPBC 2023/09759)
- Whyalla Hydrogen Pipeline (EPIC Energy, EPBC 2024/09873)
- Cultana Solar Farm (Cultana Solar Project Company Pty Ltd, EPBC 2023/09658)
- Yoorndoo Ilga Solar Project (Yoorndoo Ilga Solar, Cultana | SA: Yoorndoo Ilga Solar)
- Eyre Peninsula Transmission Line upgrade (ElectraNet, Eyre Peninsula Upgrade | ElectraNet)

Activities associated with Project's marine construction and operations at Mullaquana Station are ~34 km away from the CCSZ and not expected to impact the CCSZ. The Project desalination plant operation would not add to any cumulative effect of existing and proposed industry.

6.4 Assessment of impact significance and risk

Table 6-1 provides an assessment of the significance of potential impacts to the CCSZ against the guidance criteria for National Heritage places provided in DoE (2013).

Table 6-1: Assessment against significant impact criteria for EPBC Act National Heritage places

Criteria	Assessment of potential impact
An action is likely to have a significant impact on the National Heritage values of a National Heritage place if there is a real chance or possibility that it will:	
General	
• cause one or more of the National Heritage values to be lost	There is no real chance or possibility that the Project could cause one or more of the National Heritage values to be lost.
• cause one or more of the National Heritage values to be degraded or damaged, or	There is no real chance or possibility that the Project could cause one or more of the National Heritage values to be degraded or damaged.
• cause one or more of the National Heritage values to be notably altered, modified, obscured or diminished.	There is no real chance or possibility that the Project could cause one or more of the National Heritage values to be notably altered, modified, obscured or diminished.
Criteria for National heritage places with natural heritage values*	
• substantially increase concentrations of suspended sediment, nutrients, heavy metals, hydrocarbons, or other pollutants or substances in a river, wetland or water body in a National Heritage place	**No significant impact expected.** The values of the CCSZ would not be affected by operational discharge, potential sedimentation or spills during the Project construction. The desalination plant is located ~34 km from the CCSZ and the nearest terrestrial works are 15 km from the CCSZ. Risk of spill and sedimentation into the environment during the Project construction

Criteria	Assessment of potential impact
	will be adequately managed. Refer to Sections 6.2.1, 6.2.4,and 6.2.5.
• modify or inhibit ecological processes in a National Heritage place	**No significant impact expected.** The Project is unlikely to impact cuttlefish breeding and spawning in the CCSZ (refer to Section 6.2). Considering there is no known migration 'pathway' for the cuttlefish, with individuals likely to aggregate from all areas around northern Spencer Gulf (Gillanders and Payne 2014), it is considered unlikely the Project would prevent the cuttlefish from aggregating to the CCSZ.
• reduce the diversity or modify the composition of plant and animal species in a National Heritage place	**No significant impact expected.** The Project is unlikely to impact the cuttlefish within the CCSZ (refer to Section 6.2). The Project is unlikely to impact the cuttlefish outside the CCSZ and unlikely to prevent this species from aggregating in the CCSZ during the breeding season (refer to Section 7).
• fragment or damage habitat important for the conservation of biological diversity in a National Heritage place	**No significant impact expected.** The proposed desalination plant and pipeline are located ~34 km and 15 km from the CCSZ respectively. Construction and operation of the project is unlikely to fragment or deteriorate habitat in the CCSZ (refer to Section 6.2).
• cause a long-term reduction in rare, endemic or unique plant or animal populations or species in a National Heritage place	**No significant impact expected.** The Project is unlikely to impact the USG cuttlefish population (refer to Section 7).
• fragment, isolate or substantially damage habitat for rare, endemic or unique animal populations or species in a National Heritage place.	**No significant impact expected.** The proposed desalination plant and pipeline are located ~34 km and 15 km from the CCSZ respectively. Construction and operation of the project is unlikely to affect breeding habitat for the cuttlefish within the CCSZ (refer to Section 6.2).
• involve construction of buildings, roads or other structures, vegetation clearance, or other actions with substantial and/or long-term impacts on relevant values	**No significant impact expected.** The proposed desalination plant and pipeline are located ~34 km and 15 km from the CCSZ respectively. No Project related actions are expected to lead to substantial or long term impacts to the relevant values.
• introduce noise, odours, pollutants or other intrusive elements with substantial and/or long-term impacts on relevant values	**No significant impact expected.** As discussed in Section 6.2.2 there is no real chance or possibility that noise from construction and operational activities could adversely affect cuttlefish within the CCSZ.

* All criteria for *Biological and ecological values and Wilderness, aesthetic, or other rare or unique environment values* and the relevant criteria for *Values associated with geology or landscapes* have been included.

7. Giant Australian Cuttlefish Assessment

This assessment uses the Northern Water EIS impact assessment methodology provided in Appendix A to assess the impact of Project construction and operation on the USG cuttlefish population.

7.1 Identified sources of potential impact and key findings

The assessment of potential impacts to the USG cuttlefish population presented in Section 7.2 identified the following potential sources for impact and key findings:

- **Desalination return water during operation affecting water quality in the Project Area, and potentially impacting USG cuttlefish population.** Hydrodynamic plume modelling suggests small highly localised zone of potential ecological effect which is not anticipated to have a negative impact on the highly mobile, transient cuttlefish individuals that may occur in the Project Area. The site is not a known high usage area with no individuals recorded during marine ecology surveys of the site (J Diversity 2026).
- **Noise from construction or operational activities in the Project Area potentially impacting USG cuttlefish population**. Localised, short term construction noise is not anticipated to have a negative impact on the highly mobile, transient cuttlefish individuals that may occur in the Project Area. No USG cuttlefish were recorded in site surveys (J Diversity 2026).
- **Disturbance during construction potentially impacting water quality within the Project Area through elevated suspended sediments**. The silty sediments are likely to be easily suspended and lead to localised turbidity impacts. This may have a minor effect on individual cuttlefish if present in the immediate vicinity, though no individuals were recorded in site surveys (J Diversity 2026). Negligible impact to population of USG cuttlefish.
- **Increased vessel movements within the Project Area during construction potentially impacting the USG cuttlefish population**. This risk was assessed as negligible due to the anticipated low traffic volumes expected during construction, the temporary nature of the construction, and the implementation of appropriate control measures.
- **Spills of fuel or chemicals in the Project Area potentially impacting the USG cuttlefish population**. USG cuttlefish are not known to frequent the Project Area (J Diversity 2026)), and only small quantities of fuel and chemicals are to be used- the risk of spills is low.

7.2 Impact assessment

7.2.1 Return water during desalination plant operation

Highly localised zone of potential ecological effect from return water would have negligible impact on USG cuttlefish population

Preliminary mid-field modelling shows salinity returns to within the normal range of local ambient conditions within the immediate vicinity of the outfall. The mid-field hydrodynamic modelling indicates that 95% of species would be protected from impacts associated with the return water within a short distance from the diffusers. As discussed in section 6.2.1 any effects of the desalination plant are localised and geographically distinct from the breeding grounds and developing cuttlefish eggs.

There are no known migratory pathways for the species and the Mullaquana desalination site is not known to be high usage area for USG cuttlefish based on bycatch data from snapper trawl surveys (Gillanders and Payne 2014). This is supported by a lack of cuttlefish records during the Marine Ecology Surveys undertaken for the Project (J Diversity 2025).

Any potential for impact to cuttlefish relates to solitary individual cuttlefish as transient visitors near the proposed desalination intake and outfall structures.

Impacts of elevated salinity on USG cuttlefish adults and juveniles are not well understood. There have been limited ecotoxicology studies all of which have involved testing the salinity tolerances of cuttlefish eggs and/or hatchlings only and not the more mobile adults and juveniles.

Ambient salinities at Point Lowly where the cuttlefish aggregate for breeding are much higher than at Mullaquana Station, suggesting the USG population is tolerant of salinities much higher than the ambient salinity at Mullaquana Station. Additionally, mid-field hydrodynamic modelling indicates that salinity beyond the immediate vicinity of the diffusers would remain within the range of natural seasonal variability (see Figure 5-1, Figure 5-2 and Figure 5-3), also suggesting that the USG population would be tolerant to salinity conditions resulting from the return water discharge.

Discharge of cleaning chemicals or antiscalants in the return water is not expected to affect marine biota, as these additives generally make a negligible contribution to the toxicity of return water. The cleaning chemicals and antiscalants proposed to be used for the Project are in line with those approved by the EPA for use at the Adelaide desalination plant. Previous studies have found that the toxicity of desalination plant return water is driven by the high salt concentrations, and chemical additives contribute very little to the toxicity for most species (WQ Advice 2025). Therefore, the safe dilution rates derived for salinity ecotoxicity are likely to also be protective for antiscalant and CIP chemical toxicity (WQ Advice 2025).

As mentioned in Section 6.2.1, EPA compliance limits and monitoring requirements as part of the environmental licence to operate the desalination plant would be achieved through implementation of a program for operational compliance and water quality monitoring. This would reduce environmental risk from the operation of the desalination plant and associated return water.

Localised salinity increases in the vicinity of the outfall are not anticipated to pose a significant risk to adult or juvenile USG cuttlefish populations. Adult or juvenile cuttlefish individuals may be present in the area as they disperse around the northern Spencer Gulf when not at their breeding grounds. The location is not a known or likely migration pathway. At most, the plume may interact with a very small number of individuals. Potential interaction would be transient as the mobile individuals have the ability to swim away from unfavourable conditions and avoid prolonged exposure.

The predicted impact to cuttlefish from the return water is considered negligible (Appendix B). Uncertainty in the predicted impact has been evaluated and although there is a paucity of data on impacts of elevated salinity on adult/juvenile cuttlefish, the level of certainty in the assessment outcomes is high (Appendix B).

7.2.2 Underwater noise during construction

Underwater noise during construction will not affect the USG cuttlefish population

During the proposed desalination plant's construction phase, underwater noise levels are not expected to be significant due to the nature and scale of operations involved. With most construction via TBM under the seabed, the primary source of noise in the marine environment would be impact piling and to a lesser extent drilling, support vessels, and vibratory piling. Construction noise associated with the marine structures will be temporary and of limited duration.

Little research has been conducted regarding the specific impact of underwater noise on Giant Australian Cuttlefish. SARDI (2022) noted that high sound levels can impact cuttlefish by causing direct damage to statocysts and that acoustic trauma can be triggered at approximately 140dB.

An underwater noise and vibration assessment was undertaken for the Project, which conservatively identified a potential risk of physiological and/or behavioural impacts on cephalopods (cuttlefish) within a 4,750 m radius of impact piling activities, 1200 m of vibratory piling, 300 m radius of drilling, and 150 m radius of vessels (Resonate 2025). However, individual cuttlefish are capable of swimming away from short-term exposure noise which provides a degree of resilience against potential disturbances during construction activities.

Mitigation and management measures for underwater noise from construction will be compiled in an Underwater Noise and Vibration Management Plan and include "soft start" procedures during drilling and piling for construction of the risers, to encourage fauna to swim away before the noise increases. These measures will help to ensure noisy activities do not adversely affect marine fauna, including USG cuttlefish. The anticipated short term and localised nature of any potential noise during marine construction suggests that it will not significantly impact cuttlefish individuals.

Given that USG cuttlefish are not known to inhabit the Project Area, it is expected that any short-term underwater noise generated during construction will not significantly impact the population. While there is limited data on the effects of underwater noise specifically for cuttlefish populations, existing evidence indicates that individual cuttlefish can effectively avoid exposure to noise if required.

The predicted impact to cuttlefish from underwater noise during construction is considered to be minor to negligible (Appendix B). The uncertainty has been assessed and although there is limited noise-related data specific to cuttlefish, the certainty in assessment outcomes is high (Appendix B).

7.2.3 Sediment disturbance and turbidity during construction

Short term localised increase in turbidity and sedimentation will not impact the USG cuttlefish population

Marine construction activities may cause a temporary increase in suspended sediment and or turbidity during installation of the risers for the intake and outfall, the diffuser array, and any pipe collars required if pull and sink methodology is employed for a section of the outfall pipe.

Cuttlefish are not known to be particularly sensitive to elevated turbidity (Woodcock *et al.* 2014, see section 3.6). Cuttlefish are capable of swimming through such conditions for limited durations. Preliminary results from the mid-field hydrodynamic model indicate that modelled changes in total suspended solids are small and within natural variation within close proximity of the outfall (see Figure 5-1 and Figure 5-2).

No cuttlefish individuals have been recorded during the Northern Water marine ecology surveys (J Diversity 2026), however it is acknowledged that individuals may be present from time to time.

Turbidity and sedimentation management measures will be included in the Marine and Coastal Construction Environment Management Plan, Marine Water Quality and Monitoring Management Plan and Dredge Management Plans.

Temporary increases in suspended sediment and / or turbidity are expected to be contained, and have a negligible impact to potential resting / foraging habitat. If USG cuttlefish were present during construction, they are highly mobile and able to swim away from minor disturbance. There is no breeding habitat at the site, and therefore it is anticipated that impacts to the USG cuttlefish population due to sedimentation and turbidity will be negligible (Appendix B). The certainty in the assessment has been assessed as high (Appendix B).

7.2.4 Increased vessel movements during construction

Increase in vessel movements during construction is not expected to impact on the USG cuttlefish population

There will be a temporary increase in vessel movements during construction of the desalination plant as described in section 6.2.3. The increase in vessels during Northern Water marine construction will be negligible compared to existing marine traffic in Upper Spencer Gulf. The USG cuttlefish is not known to utilise the Project Area, though it is acknowledged individuals may occur as they move around USG. Marine vessels have the potential to interact with cuttlefish including via vessel noise, risk of introduction of invasive marine species and increased risk of spills.

Cephalopods may be sensitive to low frequency noise such as that of vessels (Resonate 2025; Woodcock *et al*. 2014) though impacts are unlikely as discussed in underwater noise section 7.2.2.

As mentioned in section 6.2.4, the risk from vessel-borne invasive marine species will be adequately controlled and managed as per the measures detailed in the Invasive Marine Species Risk Assessment undertaken for the Project (Aquatic Biosecurity 2025).

While the potential for accidental spills from vessels during marine construction exists, effective vessel management is expected to reduce the likelihood of spills. With vessel management and spill mitigation in place the project can minimise the potential for ecological impacts during the construction phase. Accidental spills are covered further in Section 7.2.5.

Due to the anticipated negligible increase in vessel numbers compared to existing levels, the temporary nature of the construction, the highly mobile nature of cuttlefish (if present), and the implementation of appropriate control measures, expected impacts on cuttlefish from construction vessel movements is considered negligible (Appendix B). The level of certainty in the assessment has been assessed as high (Appendix B).

7.2.5 Accidental spills

Accidental spills in the Project Area are unlikely to occur or impact USG cuttlefish population.

Terrestrial construction activities for the desalination plant at Mullaquana Station will involve relatively small quantities of chemicals. During operation there are design measures in place to prevent unplanned discharges to the marine environment. Potential sources and management measures are described in section 6.2.4. While the potential for accidental spills during the construction of the desalination plant exists, effective management strategies can significantly reduce the likelihood and potential severity of a spill.

Marine construction activities including drilling of the risers, installing and connecting diffuser array will involve the use of fuel and oil. There will be support vessels and safety boats onsite during construction ready to deploy countermeasures in the event of an accidental oil/fuel spill.

The Project Area does not provide important or unique habitat for USG cuttlefish and the species is not known to regularly utilise the area (based on data from seasonal surveys undertaken for the Project by J Diversity (2026)). Cuttlefish are considered unlikely to be present in the Project Area, but in the case of an accidental spill there is the potential for a minor effect on individual cuttlefish if present in the immediate vicinity. This would have a negligible impact on the USG cuttlefish population. The level of risk associated with spills is medium considering a worst-case consequence (Appendix B), but this risk is considered as low as reasonably practicable.

8. Conclusion

The construction and operation of the proposed desalination plant and associated intake and outfall structures at Mullaquana Station is expected to have negligible impact on the USG cuttlefish population. The National Heritage-listed CCSZ will not be affected by the Project.

The anticipated temporary and localised effects during construction would not impact the population of USG cuttlefish as they are not known to rely on the area for aggregation and were not recorded during seasonal site surveys (J Diversity 2026). The site does not provide suitable habitat for Cuttlefish egg attachment and there is no suggestion of a migration pathway. Notwithstanding that, it is possible for individuals to transit through the area, and if they were to be present, they are highly mobile and able to swim away from the area near the return water discharge.

Additionally, mid-field hydrodynamic modelling indicates that salinity beyond the immediate vicinity of the diffusers would remain within the range of natural seasonal variability (see Figure 5-1, Figure 5-2 and Figure 5-3). Ambient salinities at Point Lowly where the cuttlefish aggregate for breeding are much higher than at Mullaquana Station, suggesting that the USG population are physiologically adapted to high salinities (Steer et al 2013), and therefore likely to be tolerant to salinity conditions in the area near the return water discharge.

It is not anticipated that the desalination plant return water would have any material impact on USG cuttlefish moving through the gulf. The outfall location, distance away from breeding aggregation sites, USG cuttlefish tolerance to existing ambient salinity gradients within the Gulf, temporal nature of construction and the Project's operational parameters suggest that any potential effects would be localised to the immediate area surrounding the outfall and the risk of impact to the USG cuttlefish population or breeding habitat would be negligible.

During operation, the Project would adhere to strict compliance limits and monitoring requirements as part of the EPA environmental licence to operate the desalination plant. These measures would be set out in the Project Operational Environmental Management Plan and would seek to reduce environmental risk from the operation of the desalination plant, and associated return water. Mitigation and management measures to avoid or minimise environmental impacts during construction would be incorporated into the Project's Construction Environment Management Plan.

The construction and operation of the Northern Water desalination plant is not expected to have a significant impact on the EPBC Act National Heritage listed CCSZ or its values, and the USG cuttlefish population would remain unaffected.

9. Limitations

Scope of services

This report ("the report") has been prepared by JBS&G in accordance with the scope of services set out in the contract, or as otherwise agreed, between the Client and JBS&G. In some circumstances, a range of factors such as time, budget, access and/or site disturbance constraints may have limited the scope of services. This report is strictly limited to the matters stated in it and is not to be read as extending, by implication, to any other matter in connection with the matters addressed in it.

Reliance on data

In preparing the report, JBS&G has relied upon data and other information provided by the Client and other individuals and organisations, most of which are referred to in the report ("the data"). Except as otherwise expressly stated in the report, JBS&G has not verified the accuracy or completeness of the data. To the extent that the statements, opinions, facts, information, conclusions and/or recommendations in the report ("conclusions") are based in whole or part on the data, those conclusions are contingent upon the accuracy and completeness of the data. JBS&G has also not attempted to determine whether any material matter has been omitted from the data. JBS&G will not be liable in relation to incorrect conclusions should any data, information or condition be incorrect or have been concealed, withheld, misrepresented or otherwise not fully disclosed to JBS&G. The making of any assumption does not imply that JBS&G has made any enquiry to verify the correctness of that assumption.

The report is based on conditions encountered and information received at the time of preparation of this report or the time that site investigations were carried out. JBS&G disclaims responsibility for any changes that may have occurred after this time. This report and any legal issues arising from it are governed by and construed in accordance with the law as at the date of this report.

Environmental conclusions

Within the limitations imposed by the scope of services, the preparation of this report has been undertaken and performed in a professional manner, in accordance with generally accepted environmental consulting practices. No other warranty, whether express or implied, is made, including to any third parties, and no liability will be accepted for use or interpretation of this report by any third party.

The advice herein relates only to this project and all results conclusions and recommendations made should be reviewed by a competent person with experience in environmental investigations, before being used for any other purpose.

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Appendix A Assessment Methodology for Giant Australian Cuttlefish Impact Assessment

Assessment Methodology

The process for the Giant Australian Cuttlefish Impact Assessment was based on the Impact Assessment used for the Northern Water Environmental Impact Assessment, and involved the following:

- Determining the relevant environmental values that could be impacted by the Project and the condition and significance of these values.
- Reviewing the proposed Project design and activities across construction and operation phases, to determine the ways in which the Project could potentially impact the identified values. These were called the 'potential impact events'.
- For each potential impact event, assessing the duration, magnitude and extent of the impacts from the Project, initially assuming industry-standard mitigation measures are applied.
- Identifying any additional management and mitigation measures where relevant for each potential impact event needed to eliminate or otherwise reduce any potential environmental harm to as low as reasonably achievable.
- If relevant, determining the impacts from the project that are expected to remain after any additional proposed control and mitigation measures have been applied (i.e. post controls). These are the impacts that are planned for, and a necessary consequence of carrying out the Project. These are described in conjunction with control, mitigation and management measures.
- Use of risk assessment tools to evaluate the uncertainty in the assessment of expected impacts, where appropriate.

Avoidance and mitigation measures are used to avoid, reduce and / or mitigate the likelihood, magnitude, extent and / or duration of potential adverse impacts. The measures can include physical controls (design measures) and procedural (management) controls.

The impact assessments considered the scale, intensity, duration and frequency of impacts and the sensitivity of each environmental value. Where these could not be predicted with certainty, realistic worst-case assumptions were made to provide a conservative assessment.

Impacts were assessed against the categories in Table 10-1 taking into account any proposed avoidance or mitigation measures. Impacts were considered acceptable if they were in the 'Negligible' category, while impacts in the 'Minor' or 'Moderate' category were generally considered acceptable if they were as low as reasonably practicable following the application of mitigation measures. Major and catastrophic impacts were considered unacceptable necessitating Project redesign.

Table 10-1: Categorisation of impact level– biophysical environment

Category	Flora and fauna (not listed species)	Habitat	Contamination	Water (groundwater, surface water and marine)
Negligible	Local short-term (< 12 months) decrease in abundance of some species without reduction in local community viability	Not measurable	Not measurable	Minimal change with no measurable loss of water quality or quantity
Minor	Local long-term decrease in abundance of some species resulting in little or no change to community structure	Disturbance of well-represented landforms / habitats	Local contamination that can be immediately remediated	Local short-term (< 6 months) reduction in water quality or quantity
Moderate	Regional long-term decrease in abundance of some species and / or local loss of some species diversity resulting in some change to the community structure	Local loss of well-represented landforms / habitats.	Local contamination that can be remediated in less than 12 months.	Local long term or widespread short-term reduction in water quality or quantity
Major	Regional long-term decrease in abundance of numerous species and / or some loss of species diversity resulting in significant changes to community structure	Local loss of a unique or critical landform / habitat.	Widespread contamination that can be remediated in less than 12 months	Widespread (regional) short-term reduction in water quality or quantity
Critical	Regional long-term loss of numerous species resulting in the dominance of only a few species	Regional loss of unique or critical landforms / habitats	Widespread contamination that requires a regional incident response and more than 12 months to remediate	Widespread (regional) long-term reduction in water quality or quantity

Risk assessment tools were used to evaluate the uncertainty in the assessment of expected impact.

Sources of uncertainty create a risk that the impacts on environmental values may be greater than expected. Uncertainties in the impact assessment can derive from:

- uncertainty in the nature and scale of the Project infrastructure and activities
- quality of site-specific baseline and / receptor data
- the reliability of any predictive modelling undertaken in the impact assessment
- control measures not being as effective as expected.

Table 10-2 describes the factors that can affect the level of certainty for each of these sources.

Table 10-2: Factors affecting level of certainty

Level of certainty	Quality of data	Extent to which modelling has been validated	Effectiveness of design measures	Effectiveness of management measures
High	Comprehensive data. Further studies are unlikely to generate additional information that would change the conclusions reached in the impact assessment.	Excellent baseline data available. Model has been run and provides accurate predictions over different seasons. Model has been extensively used and is regarded by discipline experts as leading practice and/or the impact assessment does not rely to any significant extent on the use of a model.	Widely used and demonstrated to be effective at a range of infrastructure sites including sites with similar topographical / climatic conditions. Requires minimal checking and failure risk has been shown to be low.	Management measures are considered routine and used effectively throughout industry. Reduction in the level of impact from an unmitigated level does not rely primarily on the management measures.
Medium	Some site-specific information available to provide ground-truthing of regional desktop information. Further studies could change some of the conclusions reached in the impact assessment.	Some baseline data available. Model shows a reasonable approximation of real conditions but relies on a number of assumptions and sufficient data not available to demonstrate the model accurately portrays seasonal conditions.	Has been used at sites with similar conditions but requires regular checking or maintenance to ensure performance. Has only been used at limited sites. OR Effectiveness has not been established in the long term or at sites similar to the Project site.	Management measures have been effectively used at a limited number of sites and have not been demonstrated at similar sites or in the long term and / or reduction in the level of impact from an unmitigated level relies primarily on the management measures.
Low	Minimal site-specific data available. Reliance on regional desktop studies that may not accurately reflect site conditions. Low level of confidence in the impact assessment.	Minimal baseline data. Model is unable to be validated with current data.	Measures are novel and have not been demonstrated in the field.	Management measures are novel and / or heavily reliant on specialised technical expertise.

The level of certainty for each potential impact event was classified using the factors outlined in Table 10-2. 'High' certainty potential impact events were considered to be sufficiently certain to require no further analysis. A risk assessment was used to evaluate impact events with a 'Low' or 'Medium' level of certainty. The risk assessment considered:

- the credible worst-case consequence (as defined using the impact levels in Table 10-1) that could occur if assumptions made in the impact assessment were found to be incorrect (for example, if management measures were found to be not as effective as expected)
- the likelihood of such a consequence occurring (as described in Table 10-3).

The level of risk was then categorised using the matrix in Table 10-4.

Table 10-3: Description of likelihood

Descriptor	General description	Probability per year
Almost certain	This consequence is expected to occur in most circumstances.	> 80%
	Expected to occur at least once each year.	
Likely	This consequence may occur in some given circumstances.	50 – 80%
	May occur during any given year.	
Possible	This consequence might occur at some time.	10 – 50%
	Not likely to occur in any given year but is possible	
Unlikely	This consequence could occur at some time.	0.5 – 10%
	Very unlikely to occur in any given year.	
Rare	This consequence may only occur in very exceptional circumstances.	< 0.5%
	Examples of this have occurred historically but are not anticipated.	

Table 10-4: Risk rating matrix

	Consequence				
Likelihood	Negligible	Minor	Moderate	Major	Critical
Almost Certain	Low	Medium	High	Extreme	Extreme
Likely	Low	Medium	High	Extreme	Extreme
Possible	Low	Low	Medium	High	Extreme
Unlikely	Low	Low	Medium	High	High
Rare	Low	Low	Low	Medium	High

Appendix B Giant Australian Cuttlefish Assessment – Impact Tables

Key Finding	Potential Impact Event	Expected Level of Impact	Level of certainty in assessment of expected impact	Factors contributing to level of certainty	Evaluation of uncertainty		
					Consequence (worst case)	Likelihood	Risk rating
Highly localised zone of potential ecological effect from return water would have negligible impact on the USG cuttlefish population	Increased salinity from return water impacting upon developing cuttlefish eggs or solitary individuals.	Negligible	High	While there is a paucity of data on the impacts of elevated salinity on adult/cuttlefish there is High level of certainty that the preliminary modelling of the return water plume can reasonably predict the effect of the desalination plant discharge at a distance of 34 km away at the CCSZ (i.e. model predictions at this significant distance from the discharge point have a very low sensitivity to model assumptions and input parameters). Additionally, the model has been run and provides accurate predictions over all four seasons and instantaneous scenarios (including dodge tides). Model has been extensively used and is regarded by discipline experts as leading practice through independent peer reviewers.	N/A		
Underwater noise during construction will not affect the USG cuttlefish population	Noise from construction of desalination plant results in damage to cuttlefish or avoidance of area.	Minor – Negligible	High	There is limited data available on how noise impacts cuttlefish. Underwater noise baseline monitoring was conducted and used to inform noise modelling. A number of baseline marine surveys have results in no detection of cuttlefish in the Project Area and there are no historical records. Noise during marine construction is well understood and cuttlefish are a highly mobile species with the ability to avoid minor disturbance.	N/A		

Key Finding	Potential Impact Event	Expected Level of Impact	Level of certainty in assessment of expected impact	Factors contributing to level of certainty	Evaluation of uncertainty		
					Consequence (worst case)	Likelihood	Risk rating
Short term localised increase in turbidity and sedimentation will not impact the USG cuttlefish population	Increase in suspended sediment and/or turbidity during marine construction period.	Negligible	High	Management of suspended sediment during marine construction is well understood and cuttlefish are a highly mobile species with the ability to avoid minor disturbance.	N/A		
Increase in vessel movements during construction is not expected to impact on the USG cuttlefish population	Impact of marine traffic (including vessel noise, invasive marine species and increased risk of spills).	Negligible	High	Routine management measures are in place and used effectively including at the Whyalla and Port Bonython shipping ports which include significant vessel traffic in close proximity to breeding grounds.	N/A		
Accidental spills in the Project Area could adversely impact USG cuttlefish population	Accidental spill of chemicals during construction results in impacts on cuttlefish.	Minor - Moderate	Medium	There are routine management measures to reduce the likelihood and severity of potential spills.	Moderate	Possible	Medium

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Appendix G-17:

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- Professor Bronwyn Gillanders: School of Biological Sciences, College of Science, Adelaide University
- Dr Alistair Hobday: Chief Research Scientist, CSIRO Environment
- Brett Miller: Principal Engineer, Water Research Laboratory, School of Civil and Environmental Engineering, University of New South Wales

Independent Peer Review

Professor Bronwyn Gillanders, Adelaide University

Adelaide University

Dr Ana Glavinic
Executive Director, Northern Water Project Delivery Office
Department for Infrastructure and Transport
83 Pirie Street, Adelaide SA 5000

30 August 2026

Dear Dr Glavinic

Re: Review of marine documents associated with the Northern Water project

I was engaged by the Department for Infrastructure and Transport to undertake review of 12 technical reports and two chapters associated with the Northern Water Project in South Australia's Upper Spencer Gulf. The technical reports and chapters focused predominantly on marine and coastal environments including potential impacts from construction and operation of a seawater reverse osmosis desalination plant located at Mullaquana Station in Upper Spencer Gulf. The technical reports and chapters formed part of the Environmental Impact Statement for the project.

The following documents were reviewed:

Technical Reports	Author	Report Revision Date
Marine Benthic Habitat Mapping Report	J Diversity	29/05/2025
Marine Ecology Baseline and Impact Assessment (Rev C)	J Diversity	1/06/2026
Underwater Noise and Vibration Impact Assessment	Resonate	12/03/2026
Mullaquana Station Mangrove Baseline Report (v1.4)	COOE	19/02/2026
Baseline Survey for Larval Fish and Larval Decapod Crustaceans	SARDI	27/04/2026
Survey for Juvenile Prawn Recruitment at Mullaquana Station	SARDI	Draft Report June 2026
Marine Desalination Plant Ecotoxicology Assessment Report	Rick Van Dam	1/10/2025
Invasive Marine Species Risk Assessment Report (v0)	Aquatic Biosecurity	25/09/2025
Marine Sediments Summary Baseline Report	JBSG / Chris Bolch	24/02/2026
Giant Australian Cuttlefish Assessment	JBSG	12/01/2026

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Potential Interaction Between the Desalination Plant and Snapper and King George Whiting- Advice Note	SARDI	7/08/2025
Baseline Water Quality	JBSG	21/07/2026
For information only. Northern Water Desalination Plant Discharge Modelling – Nearfield Modelling	Water Technology	23/09/2026
For information only. Northern Water Project - Desalination Plant Midfield Hydrodynamic and Transport Modelling	Water Technology	30/01/2026
For information only. Farfield - Oceanographic far-field modelling of brine discharge and larval connectivity to inform Northern Water, Spencer Gulf	SARDI	Draft Report 20 August 2026
For information only. Far Field Modelling Climate Scenario Report	SARDI	Draft Report 20 August 2026
Coastal Processes Chapter 10	JBSG	28/05/2026
Marine Environment Chapter 11	JBSG	1/07/2026

Chapters 6 (Project description) and Chapter 8 (Impact assessment methodology) were also provided for information only.

Generic comments along with more specific minor comments were provided to DIT for each technical report and chapter. In general, most technical reports provided a comprehensive assessment of potential impacts as a result of construction and operation of the project. A number of the reports were based on a desktop assessment (e.g. Invasive Marine Species) or information from review of literature (e.g. Ecotoxicology assessment). Several had undertaken field sampling across multiple sites and time periods (e.g. Mangrove baseline, larval fish and decapod baseline) usually of up to 1 year. It is important to recognise that some of the sampling occurred during a Harmful Algal Bloom in South Australian waters. Overall, reports used appropriate approaches and provided good summaries of the key components they set out to address.

In relation to the ecotoxicology report several comments were made regarding choice of organisms for additional testing during commissioning and operational phases. It was suggested that a vertebrate species (such as snapper, *Chrysophrys auratus*) be included noting that some tests are not very sensitive. The text of this technical report was revised to suggest a fish species be included in the candidate species list for additional testing during commissioning and operational phases.

Further, for the underwater noise and vibration impact assessment an updated literature search was required as much of the recent literature was missing. This comment was taken on board in the revised technical report. The summary and conclusion did however provide clear evidence that the risk of significant underwater noise or vibration impacts on marine fauna during construction and operation of the project was likely to be low, and potential impacts could be appropriately managed.

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The marine ecology baseline and impact assessment focused predominantly, but not exclusively, on seagrass and silty sand habitats, which were the dominant habitat types at Mullaquana. Good summaries of field results, priority species, fisheries and aquaculture species were provided. Assessment of return water impacts being negligible was well supported and where impacts were suggested they were likely to be minor and highly localised.

Both chapters (Chapter 10 Coastal Processes and Chapter 11 Marine Environment) of the Environmental Impact Statement (EIS) adequately summarised the various technical reports providing good overviews of the potential impacts of construction and operation of the project. The key comment was to ensure consistent terminology among chapters especially in relation to dodge tides versus neap tides of limited or no tidal movement for approximately 1-2 days. Other comments were minor in nature and will be addressed in the final EIS.

Following review of the technical reports and chapters, I also reviewed the response to comments and how they had been addressed. Overall, all comments were appropriately addressed in the revised documents.

Sincerely,

BM Gillanders

Professor Bronwyn Gillanders, FTSE
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Independent Peer Review

Dr Alistair Hobday, CSIRO Environment

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25 September 2026

Dr Ana Glavinic
Executive Director, Northern Water Project Delivery Office
Department for Infrastructure and Transport
83 Pirie Street, Adelaide SA 5000

Dear Dr Glavinic,

Re Independent Expert Review: Northern Water Desalination Plant

I have undertaken an independent expert review of the modelling reports prepared for the proposed Northern Water Desalination Plant for the Government of South Australia, Department for Infrastructure and Transport, Northern Water Project Delivery Office.

I have experience in reviews for such projects and understand the rigor that is required. I am a suitable reviewer based on my experience and expertise as a primary scientist in marine climate change, marine ecology, and hydrodynamic modelling, and thus suited to the climate, farfield and midfield reports. I read the nearfield modelling report for information and background but made no detailed comments on this report.

I was also involved in expert review meetings over the past 18 months, which allowed me to understand the data collection that supported the modelling validation efforts, the modelling approach, and to make additional suggestions to the project team. I confirm that my suggestions have influenced the research delivery and investigation, and as described in the final reports.

In undertaking this final review, I was provided the following reports in September 2026:

1. **Nearfield modelling report**
2. **Midfield modelling report**
3. **Farfield modelling report**
4. **Climate Change modelling report**

These reports are deliverables against the three scales of modelling needed to assess possible impacts.

- A farfield model focusing on Spencer Gulf-wide effects of the desalination plant (from 10's to 100's of km) (undertaken by SARDI);
- A midfield model focused on dispersion of the return water in the local vicinity of the point of discharge (from 50 m to 10 km) (undertaken by Water Technology Pty Ltd);
- A nearfield model focused on effects of the discharge in the immediate vicinity of the point of discharge (from 1 to 100 m from the outfall) (undertaken by Water Technology Pty Ltd).

The package of materials was a comprehensive examination of the hydrodynamic aspects relevant to the ocean water exchange for this potential desalination plant. I made comments on each report and provided additional written comments that were discussed in a meeting with the report authors (Sept 23, 2026). Following revision of the reports, I also reviewed the summary of the response to

comments and how they had been addressed, and the revised reports. The remaining questions and associated uncertainties have been addressed.

With this letter, I conclude my role as Independent Expert Reviewer

Yours sincerely,



Dr. Alistair Hobday, ATSE
Chief Research Scientist
CSIRO Environment
Hobart, Tasmania, Australia

Independent Peer Review

Brett Miller, University of NSW

29 September 2026

WRL Ref: WRL2025064 L20260926 BMM

Dr Lexi Schwarz
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Northern Water Project Delivery Office
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UNSW
Water Research
Laboratory

Dear Dr Schwarz,

Independent Expert Review:
Diffuser Design and Near Field Mixing – Northern Water Project

I (Brett Miller) provided an independent expert review of the outfall design and near field mixing for the proposed Northern Water Desalination Plant for the Government of South Australia, Department for Infrastructure and Transport, Northern Water Project Delivery Office. This review was summarised in my letter of the 21st November 2025.

My review made comments and observations. In summary I agreed that the near field dilutions required to meet environmental objectives would be met with the proposed outfall configuration.

I received a document from you on the 24th September 2026 summarising in a table how each of my comments and observations had been addressed in the final version of the reports. I consider that each of my earlier comments and observations have been considered and addressed. I have no further concerns.

Yours sincerely,

Brett Miller
Principal Engineer