

Identifying water quality and ecosystem health threats to the Torres Strait and Far Northern GBR arising from runoff of the Fly River

Jane Waterhouse, Caroline Petus, Scott Bainbridge, Simone C. Birrer, Jon Brodie, Anthony C. Chariton, Katherine A. Dafforn, Johanna E. Johnson, Emma L. Johnston, Yanfang Li, Janice Lough, Flavio Martins, Dominique O'Brien, Dieter Tracey, Eric Wolanski



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Supported by the Australian Government's
National Environmental Science Program

Project 2.2.1 Identifying the water quality and ecosystem health threats to the high diversity Torres Strait and Far Northern GBR from runoff from the Fly River

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National Library of Australia Cataloguing-in-Publication entry:
978-1-925514-27-8

This report should be cited as:

Waterhouse, J., Petus, C., Brodie, J., Bainbridge, S., Wolanski, E., Dafforn, K.A., Birrer, S.C., Lough, J., Tracey, D., Johnson, J.E., Chariton, A.C., Johnston, E.L., Li, Y., Martins, F., O'Brien, D. (2018) *Identifying water quality and ecosystem health threats to the Torres Strait and Far Northern GBR from runoff of the Fly River*. Report to the National Environmental Science Program. Reef and Rainforest Research Centre Limited, Cairns (162pp.).

Published by the Reef and Rainforest Research Centre on behalf of the Australian Government's National Environmental Science Program (NESP) Tropical Water Quality (TWQ) Hub.

The Tropical Water Quality Hub is part of the Australian Government's National Environmental Science Program and is administered by the Reef and Rainforest Research Centre Limited (RRRC). The NESP TWQ Hub addresses water quality and coastal management in the World Heritage listed Great Barrier Reef, its catchments and other tropical waters, through the generation and transfer of world-class research and shared knowledge.

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Cover photographs: Jane Waterhouse

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CONTENTS

Contents.....	i
List of Tables.....	iii
List of Figures.....	iv
Appendices: List of Tables	viii
Appendices: List of Figures	ix
Acronyms and Abbreviations.....	xi
Acknowledgements	xiii
Executive Summary	xiv
1. Introduction	1
1.1 Background.....	1
1.2 Project objectives	2
2. The Torres Strait region.....	4
2.1 Regional overview.....	4
2.2 The Fly River and its catchments	5
2.3 Ecological features of the region	6
2.4 Current status, threats and pressures to ecosystems	9
2.4.1 Status and trends of Torres Strait ecosystems potentially influenced by the Fly River	9
2.4.2 Pressures and threats: current and future	15
3. Existing evidence of the extent of influence of Fly River discharges in the Torres Strait region	17
3.1 Water quality monitoring in the Torres Strait.....	17
3.2 The 2014 <i>in-situ</i> monitoring of dissolved metals.....	19
3.3 Assessment of oysters for the bioaccumulation of metals	23
3.4 Potential impacts of increased sediment and trace metal inputs.....	23
4. Results from this study	27
4.1 Coral cores.....	27
4.1.1 Overview of methods and results	27
4.1.2 Recommendations	28
4.2 Hydrodynamic modelling	29
4.2.1 Overview of methods and results	30
4.2.2 Recommendations	35
4.3 Remote sensing analysis	35
4.3.1 Overview of methods and results	37

4.3.2	Recommendations	41
4.4	<i>In-situ</i> continuous loggers	42
4.4.1	Overview of methods and results	43
4.4.2	Recommendations	50
4.5	Salinity monitoring.....	51
4.5.1	Overview of methods and results	51
4.5.2	Recommendations	55
4.6	Water quality exposure assessment.....	56
4.6.1	Overview of methods and results	57
4.7	Testing correlations between water quality monitoring datasets	71
4.7.1	Overview of methods and results	71
4.8	Gene sequencing in sediment samples.....	77
4.8.1	Overview of methods and results	78
4.8.2	Recommendations	100
4.9	Metal analysis in seagrass leaves	101
4.9.1	Overview of methods and results	101
4.9.2	Recommendations	106
4.10	Other analysis	107
4.10.1	Bioaccumulation of metals – oysters and DGTs	107
5.	Discussion.....	108
5.1	Water quality characteristics and exposure	108
5.2	Biological indicators.....	112
5.2.1	Sediment community surveys.....	112
5.2.2	Metals in seagrass leaves	113
5.3	Integration of data sources	114
6.	Future work	115
7.	References.....	118
8.	Appendix 1: Salinity monitoring training manual	133
9.	Appendix 2: Additional details of the exposure analysis	136
10.	Appendix 3: Gene sequencing in sediment sampling and seagrass metal surveys	157

LIST OF TABLES

Table 3-1:	Average, minimum and maximum metal concentrations ($\mu\text{g/L}$) at each monitoring site compared to concentrations at other North Queensland marine sites.....	22
Table 4-1:	Freshwater river discharge used in the MOHID model.....	30
Table 4-2:	Summary statistics of the corrected salinity time series.	55
Table 4-3:	Habitat (coral reef and seagrass) sites selected for the risk assessment (and corresponding monitoring codes where relevant) and potential zone of influence: Zone 1: North, 2: North East, 3: North West, 4: South, 5: Central and 6: East. Note: SG = seagrass site; RF = reef site. The site names correspond to the TSRA coral reef monitoring locations.....	60
Table 4-4:	Frequency of cloud cover in the daily images for each of the assessment sites (average % of total number of days in 2008–2018).....	63
Table 4-5:	Exposure (area km^2 and % of habitat area) of coral reef, intertidal and subtidal seagrass habitats to Primary and Secondary (CC1–5) waters between 2008 and 2018.	64
Table 4-6:	Summary results from the Torres Strait coral reef monitoring surveys 2015–2016 (TSRA 2016).....	67
Table 4-7:	Correlations between the annual NINO3.4 and the annual frequency of exposure to Primary (CC1–4) and Secondary (CC5) and Tertiary (CC6) water types.....	75
Table 4-8:	Sampling location details including locations (Figure 4-32), coordinates, depth and sampling date.	79
Table 4-9:	Proportion of sediment from each grain size class at each location.	82
Table 4-10:	Carbon and nitrogen results for each location.....	84
Table 4-11:	Ten most abundant orders in each bacterial biological assemblage.	86
Table 4-12:	Ten most abundant orders in each eukaryotic biological assemblage.....	88
Table 4-13:	Bacterial (16S) and eukaryotic (18S) richness and diversity (mean $\pm$ sd, $n=5$) in the sediment samples at each location.	90
Table 4-14:	Results from the PERMANOVAs, showing which environmental variables are related to the bacterial community composition (p -value ≤ 0.05) and to what extent (R^2).	98
Table 4-15:	Sampling location details for metal analysis in seagrass leaves including location, coordinates, depth and sampling date.	102

LIST OF FIGURES

Figure 2-1:	Map of the Torres Strait region showing the major rivers, islands, habitats and the boundary of the Torres Strait Protected Zone.	5
Figure 2-2:	Map of the Fly River catchment, PNG.....	7
Figure 2-3:	Mean currents (m/s) and general circulation model of Torres Strait in (top) the entire Torres Strait and (bottom) a zoomed-in view in the north western Torres Strait.	8
Figure 2-4:	Spatial distribution of coral reefs (grey) in the Torres Strait region.	10
Figure 2-5:	A spatial comparison of the number of species in each scleractinian coral family (bottom axis) in the Central and Eastern reefs of Torres Strait (Osborne et al. 2013).	11
Figure 2-6:	Spatial distribution of seagrasses in the Torres Strait region showing the mapped extent of intertidal and subtidal seagrass.	13
Figure 2-7:	Relative density of the dugong population in the Torres Strait (left) and the bathymetry of the region showing the concentration of dugongs around shallow areas as well as their use of deeper non-reef areas (right).	14
Figure 4-1:	Map showing the locations of two coral coring sites—Bramble Cay (BRM) and Erub (Darnley) Island (DNL).	28
Figure 4-2:	(Left) MOHID-predicted surface salinity distribution during strong SE winds. (Right) MOHID-predicted vertical distribution of the salinity along the transects shown by the yellow lines on the left.	31
Figure 4-3:	MOHID-predicted surface salinity distribution in the Gulf of Papua and Torres Strait during (left) the SE trade wind season and (right) the monsoon season.	31
Figure 4-4:	MOHID-predicted surface salinity distribution during the SE trade wind season for two different values of the Δ MSL across the Torres Strait.....	31
Figure 4-5:	Comparison between the observed and predicted surface salinity distribution in the Torres Strait in early August 1994.....	32
Figure 4-6:	An example of the variable mesh of the SLIM model.	33
Figure 4-7:	The open boundaries of the MOHID and SLIM models.....	33
Figure 4-8:	Predictions of the salinity plume at (left to right) 10, 60 and 950 hours during the SE trade wind season for (from the top downward) Δ MSL of 0.2, 0.4 and 0.6 metres and (bottom) for a variable Δ MSL as measured during September–October 2013 (when the wind often reversed direction).	34
Figure 4-9:	Focusing on the area between Badu and Saibai Islands, the bathymetry used by (left) the MOHID model and (right) the SLIM model.	35
Figure 4-10:	Triangular colour plot showing the characteristic colour signatures of the Great Barrier Reef river plume waters in the Red-Green-Blue (or true colour) space..	37
Figure 4-11:	Summary figure illustrating the mean long-term (2008–2016) and large-scale patterns in surface water colour in the Torres Strait and Gulf of Papua regions (the study area).	38
Figure 4-12:	Mean long-term seasonal areas of the Southwest Fly District (km ²) exposed to the six distinct colour classes (CC1–6).	40

Figure 4-13:	Relationships between NINO3.4 and areas of the Southwest Fly District (including the highest risk area) exposed to (a) all coloured waters (CC1–6) and (b) only Primary waters (in km ²).	41
Figure 4-14:	Location of the Temperature Loggers deployed in the Torres Strait (map Google Earth).	43
Figure 4-15:	Temperature logger deployed on a small concrete block with a float for visibility.	44
Figure 4-16:	Land based component of the Bramble Cay weather station.	44
Figure 4-17:	Average daily ocean temperatures at Thursday Island (red line), Masig Island (orange line) and Bramble Cay (blue line) showing ocean temperatures during the 2015–2016 coral bleaching event, the dashed grey line is the regional empirical bleaching threshold.	45
Figure 4-18:	Relationship between wind speed (blue line) and average daily ocean temperature (red line) at Thursday Island for the 2015–2016 coral bleaching event.	46
Figure 4-19:	Turbidity (green line) measured at Russell Island off Babinda, North Queensland. Extracted from Figure 2.4 in Schaffelke et al. (2009).	47
Figure 4-20:	Turbidity (NTU, red line) and wind speed (kph, blue line) at Bramble Cay.	48
Figure 4-21:	Turbidity (NTU, red line) and rainfall (mm, blue line) at Bramble Cay.	49
Figure 4-22:	Map of available sediment data near Bramble Cay. Orange box shows an enlargement of the main map and shows mud levels around Bramble Cay would be expected to be in the range of 10-20%. Source: Harris et al. (2002).	50
Figure 4-23:	Location of islands where weekly salinity measurements have been taken since 2016–2017 by TSRA ranger teams.	53
Figure 4-24:	Time series of salinity measured at Saibai, Boigu (insufficient data), Erub (insufficient data), Masig, Poruma and Warraber Islands: Corrected salinity values.	54
Figure 4-25:	Example of MODIS true colour images and CC maps collected on 21 February 2018 and 13 April 2018 (orange crosses on Figure 4-24, Saibai Island).	56
Figure 4-26:	Location of the Torres Strait coral reefs, seagrass beds and habitat sites selected for the risk assessment (white dots) (between 142.015°E to 144.295°E and 8.860°S to 10.354°S).	59
Figure 4-27:	Maps showing the multiannual frequency (2009–2017, excluding 2011 and 2012) of A) Primary water type (CC1–4), B) Secondary water type (CC5) and C) Tertiary water type (CC6) as well as D) cloud frequency occurrence; where the highest frequency is shown in orange and the lowest frequency is shown in blue.	61
Figure 4-28:	Exposure (area km ²) of coral reef, intertidal and subtidal seagrass habitats to Primary and Secondary (CC1–5) waters between 2008 and 2018 (excluding 2011 and 2012).	64
Figure 4-29:	Long-term frequency (decadal mean, 2008–2018) of exposure to different CCs (CC1 to CC6) at the selected coral reef and seagrass monitoring sites in the northern Torres Strait (Table 4-3).	65
Figure 4-30:	Spatial analysis of the mean frequency of exposure of habitat sites (1 km radius) to Primary (CC1–4), Secondary (CC5), Tertiary (CC6) and marine waters (CC7) into different zones of influence as defined in Table 4-3.	68
Figure 4-31:	Mean adjusted salinity and rainfall measured at Masig Island.	72

Figure 4-32:	Monthly MODIS CC composite maps: median CC category per pixel over a) July 2016, b) December 2016, c) July 2017 and d) December 2017.	73
Figure 4-33:	Examples of daily MODIS (a, b) true colour and (c, d) CC maps recorded around a turbidity spike measured at Bramble Cay and (e) turbidity and wind data recorded at Bramble Cay (e).....	74
Figure 4-34:	Time series (a, c) and correlation coefficients (b, d) between annual NINO3.4 and the annual frequency of exposure to Secondary waters (CC5) at (a, b) Bramble Cay and (c, d) Masig Island as between annual NINO3.4 and the annual frequency of exposure to Primary waters (CC5) at (e, f) Saibai Islands.	76
Figure 4-35:	Map of sediment sampling locations in the Torres Strait (Table 4-8).....	80
Figure 4-36:	(Upper) Spatial distribution of the different sediment grain sizes.....	83
Figure 4-37:	Dendrogram of all locations based on bacterial community composition (median of all replicates). The Bray-Curtis dissimilarity index was used in combination with the unweighted pair-group method with arithmetic mean (UPGMA). Based on a cut-off of 0.2 (red dotted line), the locations were grouped into nine biological assemblages.....	85
Figure 4-38:	Map depicting the different biological assemblages based on bacterial community composition.	85
Figure 4-39:	Dendrogram of all locations based on eukaryotic community composition (median of all replicates).....	87
Figure 4-40:	Map depicting the different biological assemblages based on eukaryotic community composition.	87
Figure 4-41:	Rarefaction curve displaying the recoveries from the extraction and PCR process.....	89
Figure 4-42:	Bacterial (a) diversity and (b) richness in the sediment samples at each location.	91
Figure 4-43:	Bacterial community structure in all sampling locations.	92
Figure 4-44:	Rarefaction curve displaying the recoveries from the extraction and PCR process.....	93
Figure 4-45:	Eukaryotic (a) diversity and (b) richness in the sediment samples at each location.	94
Figure 4-46:	Eukaryotic community structure in all sampling locations.....	96
Figure 4-47:	Relative abundance of eukaryotic taxa of interest in all sampling locations....	97
Figure 4-48:	CCA plot of the (a) bacterial community composition and (b) eukaryotic community composition based on Bray-Curtis distances.	99
Figure 4-49:	Map of the sampling locations for the collection of seagrass leaves (Site G, E and SG only).....	102
Figure 4-50:	Comparison of metal concentrations within the seagrass leaves and sediment samples at northern Warrior Reef (E) and Warrior Reef (G).	103
Figure 4-51:	Comparison of a suite of metal concentrations (Cd, Cr, Cu, Ni, Pb and Zn) at the northern Warrior Reef (E), Warrior Reef (G) and Saibai Island (SG) sites, shown with values from polluted and unpolluted sites in a global meta-analysis (Govers et al. 2014).	104
Figure 4-52:	Selected metal concentrations in seagrass leaves sampled from Saibai Island (SG) and Warrior Reefs (E and G).	105
Figure 5-1:	Summary of the highlights of the NESP Project 2.2.1 for each component, within the different zones of influence as defined in Table 4-3.	109

Figure 5-2:	Conceptual scheme showing the importance of integrating different data sources to provide comprehensive water quality and environmental assessments.....	114
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APPENDICES: LIST OF TABLES

Table A3-1:	Actual and measured metal concentrations in certified reference materials (DOLT-4; fish liver) used to calculate recoveries and the limits of detection (LOD) for the analysis. Values shown as mg/kg dry weight or % (recoveries)...	158
Table A3-2:	Metal concentrations measured in each seagrass sample. All values are shown as mg/kg dry weight.....	158
Table A3-3:	ANOVA comparing differences in seagrass metal concentrations among locations. Significant p-values are highlighted in bold.	161
Table A3-4:	Tukey's Honest Significance Difference test results showing differences in seagrass metal concentrations among locations. Significant p-values are highlighted in bold.....	162

APPENDICES: LIST OF FIGURES

Figure A2-1:	Cloud frequency (with clear skies shown in blue) at selected habitat sites in the NE Torres Strait from 2008 to 2018 during July–August (within the SE trade wind season), February–March (within the monsoon season) and annual. Site codes in brackets after each site name (TSRA 2016).	138
Figure A2-2:	Cloud frequency (with clear skies shown in blue) at selected habitat sites in the NE Torres Strait from 2008 to 2018 during July–August (within the SE trade wind season), February–March (within the monsoon season) and annual. Site codes in brackets after each site name (TSRA 2016).	139
Figure A2-3:	Cloud frequency (with clear skies shown in blue) at selected habitat sites in the NE Torres Strait from 2008 to 2018 during July–August (within the SE trade wind season), February–March (within the monsoon season) and annual. Site codes in brackets after each site name (TSRA 2016).	140
Figure A2-4:	Areas (percent of pixels) of (first panel) coral reefs exposed to Primary (top), Secondary (centre) and Primary + Secondary (bottom) water types in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual.....	142
Figure A2-5:	Areas (percent of pixels) of intertidal seagrass exposed to Primary (top), Secondary (centre) and Primary + Secondary (bottom) water types in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual.....	143
Figure A2-6:	Areas (percent of pixels) of subtidal seagrass exposed to Primary (top), Secondary (centre) and Primary + Secondary (bottom) water types in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual.....	144
Figure A2-7:	Frequency of exposure to different CCs at habitat sites at Saibai and Boigu Islands and Warrior Reefs in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).....	147
Figure A2-8:	Frequency of exposure to different CCs at habitat sites at Bramble Cay, and Ugar, Erub and Masig Islands in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).	148
Figure A2-9:	Frequency of exposure to different CCs at habitat sites at Poruma Island and Mer in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).....	149
Figure A2-10:	Frequency of exposure to different CCs in Zone 1 (North) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).	151
Figure A2-11:	Frequency of exposure to different CCs in Zone 2 (North East) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).....	152
Figure A2-12:	Frequency of exposure to different CCs in Zone 3 (North West) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).....	153

Figure A2-13: Frequency of exposure to different CCs in Zone 4 (South) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).	154
Figure A2-14: Frequency of exposure to different CCs in Zone 5 (Central) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).	155
Figure A2-15: Frequency of exposure to different CCs in Zone 6 (East) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).	156

ACRONYMS AND ABBREVIATIONS

AIMS	Australian Institute of Marine Science
ANOVA	Analysis of variance
BoM	[Australian] Bureau of Meteorology
bp	Base pair
BDL	Below detection limit
CCA	Canonical correspondence analysis
CDOM	Coloured dissolved organic matter
Chl-a	Chlorophyll-a
C/N	Carbon/Nitrogen
COTS	Crown of thorns starfish
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DoEE	[Australian] Department of the Environment and Energy
DNA	Deoxyribonucleic acid
ENSO	El Niño–Southern Oscillation
EODIS	Earth Observing System Data and Information System (A NASA utility)
GBR	Great Barrier Reef
GPS	Geographical positioning system
GPTS	Gulf of Papua-Torres Strait
IRMS	Isotope-ratio mass spectrometry
JCU	James Cook University
MANOVA	Multivariate analysis of variance
MODIS	Moderate resolution imaging spectroradiometer
MOHID	Modelo Hidrodinamico ('hydrodynamic' in Portuguese)
Mt/yr	Million tonnes per year
MSL	Mean sea level
NERP	National Environment Research Program
NESP	National Environmental Science Program
NIST	National Institute of Standards and Technology
NOAA	National Oceanic and Atmospheric Administration (USA)
NE	Northeast
NW	Northwest
OTU	Operational taxonomic unit
PCR	Polymerase chain reaction
PERMANOVA ...	Permutational multivariate analysis of variance
PNG	Papua New Guinea
rRNA	Ribosomal ribonucleic acid
RRT	Rivers to Reef to Turtles Project (WWF)
Sd	Standard deviation

SLIM Second-generation Louvain-la-Neuve Ice-ocean Model
SE Southeast
TSRA Torres Strait Regional Authority
TSS Total suspended solids
UNSW University of New South Wales
UPGMA Unweighted pair group method with arithmetic mean
ΔMSL Difference in mean sea level (MSL) across the Strait. ΔMSL is > 0 when the MSL in the Gulf of Papua is greater than the MSL in the Gulf of Carpentaria

Units of measure

h hour
km kilometre
m metre
Mt/yr Million tonnes per year
ng nanograms
NTU Nephelometric turbidity unit
µg micrograms
µL microlitre
µm micrometre

Island names

Island	Other names
Maizab Kaur	Bramble Cay
Erub	Darnley
Masig	Yorke
Iama	Yam
Ugar	Stephens
Warraber	Sue
Mer	Murray
Poruma	Coconut
Saibai	Saibai
Boigu	Talbot
Dauan	Mt Cornwallis
Thursday	Waiben
Horn	Ngurupai

ACKNOWLEDGEMENTS

The authors would like to thank the National Environmental Science Program Tropical Water Quality Hub and Reef and Rainforest Research Centre Ltd for funding this project and to Dr Julie Carmody for overall program management. We would also like to thank the Torres Strait Regional Authority for their engagement in project activities and, in particular, the local rangers for providing field support and undertaking local salinity monitoring. The work of scientists involved in previous research programs, such as the National Environmental Research Program, is acknowledged in providing baseline and background data for this project.

Finally, the authors of this report are grateful for the peer review comments provided by Dr David Haynes, Mr Barry Butler and Professor Damien Burrows whose input has provided valuable improvements to the final report. Thank you to Samantha Talbot from Tropical North Editing for the final editing of this report.

EXECUTIVE SUMMARY

Torres Strait Islanders depend on their marine resources for food, livelihoods and cultural activities. Regular plumes of freshwater from the Fly River in Papua New Guinea (PNG) have been observed in satellite images entering northern Torres Strait waters and can potentially threaten the quality of marine resources. However, the extent and scale of this threat is unknown, particularly as future climate projections are for more rainfall extremes and, potentially, larger river discharges.

Under certain climatic conditions, dilute Fly River plume waters have been detected across the northern Torres Strait, east of the Warrior Reefs, as far west as Saibai Island and south to Masig Island. It is important to note that this is a natural process that has been occurring for several thousand years. However, since 1984, when the Ok Tedi copper mine opened in the headwaters of the Fly River and the Porgera gold mine opened in 1990 in the headwaters of the Strickland River (a major tributary of the Fly River) in PNG, the plume waters are potentially delivering mine-derived contaminants including trace metals and sediments.

The implications of these changes to water quality in the Torres Strait are not well known, but potential impacts can be deduced from knowledge in other marine ecosystems including the Great Barrier Reef. For example, increased sediment inputs can result in increased turbidity and reduced water clarity in receiving waters, which in turn reduces light availability for marine ecosystems. Coral reef and seagrass ecosystems have minimum light requirements for healthy growth and reproduction, and in cases of prolonged exposure to reduced light conditions, can result in changes in community composition, increased incidence of disease and in extreme cases, fatality. Reef fishes can be impacted by reduced water clarity and sedimentation indirectly by changing their coral and seagrass habitats, or directly, as increased sediment loading can have direct behavioural, sub-lethal, and lethal impacts on fish. Increases in trace metal concentrations above National water quality guidelines could also have direct toxicological impacts on coral reef and seagrass meadow health as well as direct effects on particular species such as corals, turtles, dugongs and fish. These impacts on seagrass can also have flow-on effects for other species such as seagrass-feeding animals (e.g. green turtle and dugong) which have an important dietary role in some communities as well as being iconic and vulnerable species in the region. To enhance our understanding and predictive ability of the spatial and temporal extent of the Fly River plume and identify its potential impacts on Torres Strait ecosystems and dependent communities, this project was undertaken via the National Environmental Science Program (NESP), building on previous work, to determine the following: (i) spatial extent, temporal patterns and constituent contaminants of Fly River discharge in the Torres Strait region and (ii) presence of ecosystems in the Torres Strait exposed to Fly River discharge (through existing data). It was outside the scope of this project to fully assess the impact of Fly River discharges on Torres Strait ecosystems, rather, this phase focused on determining when and where the influence from this discharge is likely to occur.

The project has delivered further understanding of river flows via modelling, driven by real time marine observations and identification of areas of potential exposure to turbid waters through remote sensing. A partner project conducted by the Commonwealth Scientific and Industrial Research Organisation (NESP Project 2.2.2) has undertaken further analysis of metals in sediment and the water column at selected northern Torres Strait locations. Using these

multiple lines of evidence, we have made an assessment of the potential influence of the Fly River discharge in the Torres Strait region. This information is important for ensuring the protection of these regions given their importance to Torres Strait communities and turtle and dugong populations, and their connectivity with the Great Barrier Reef Marine Park.

These multiple lines of evidence are summarised below:

- Long-term (1781–1993) annual massive coral growth and luminescence records contained in coral cores from Erub Island (one core) and Bramble Cay (two cores) in the northern Torres Strait have been found to exhibit annual coral growth rates typical of massive *Porites* corals living in average (tropical) sea surface temperatures. There is no evidence of growth hiatuses in any of the cores that might be associated with environmental stress events. Luminescent lines (indicative of freshwater flood plumes) are evident in both corals from Bramble Cay but not the coral from Erub Island. At Bramble Cay, there are usually several pulses of freshwater each year, likely associated with atmospheric or oceanic processes pushing the freshwater onto the reef. These freshwater pulses are evident back to the beginning of the core in 1781 and are likely to be linked to the Fly River discharge.
- Hydrodynamic modelling results show that Fly River plumes commonly move into the northern Torres Strait and along the PNG coast, particularly affecting the northern and northwestern islands and reefs during the six months of the southeast (SE) trade wind season (approximately May to November). The plume extends to the northwest (NW) of the Torres Strait in the SE trade wind period, but to the SE of the Fly River mouth during the NW monsoon period (approximately December to April).
- Analysis of remote sensing imagery collected from 2008 to 2016 confirmed the regular intrusion of turbid waters (the Fly River plume and/or resuspension) from the Gulf of Papua into the northeast (NE) corner of the Torres Strait region, including Daru Island, the north of Warrior Reef and the north and south of Saibai Island. Turbid waters were also mapped around Boigu Island and may be linked to sediments discharged from the Wassi Kussa and Mai Kussa Rivers and potentially transported from the Fly River, as well as to resuspension. Spatial analyses of the southwest Fly District identified (i) larger turbid areas during the trade wind period that may reflect both the presence of suspended sediment in the Fly River plume and seasonal benthic sediment resuspension during the trade winds, and (ii) smaller turbid areas during El Niño conditions that may indicate an impact of El Niño–Southern Oscillation on sediment transport.
- Regular salinity monitoring in the northern Torres Strait including weekly sampling at Masig, Erub, Warraber, Poruma, Iama and Saibai Islands has identified periods of lower salinity, particularly at Saibai Island, which may be associated with river discharge (freshwater inputs).
- At Masig Island, occasional reductions in salinity were measured during the trade wind seasons in 2016 and 2017. Comparison with the local rainfall data from the continuous loggers and satellite imagery, in addition to hydrodynamic modelling results, indicate that this may be influenced by the Fly River discharge. A longer-term dataset is required to substantiate these findings.
- Analysis of the frequency of exposure of coral reef and seagrass habitats to waters with higher turbidity using remote sensing imagery has identified that sites around Saibai and Boigu Islands are most frequently exposed to turbid waters, compared to

islands in the central areas of the Torres Strait. Habitats around the northern Warrior Reefs, Erub and Mer are also exposed to turbid waters, although less frequently. Importantly, these are sites that have healthy coral reefs with high coral cover and low macroalgae cover, making them at high risk of ecological impact. Further analysis of these results is required, including additional *in-situ* sampling to validate the results.

- Deployment of continuous turbidity loggers at Bramble Cay (2017 to 2018) showed higher than expected levels of turbidity (average of 10.7 NTU) for a remote offshore reef (as compared to offshore reefs in the Great Barrier Reef (GBR) that average 0–3 NTU and near-shore reefs that average 3–5 NTU in the dry season and a maximum of 10–15 NTU in the wet season). The observed patterns of turbidity at Bramble Cay were correlated with wind speed and most likely represent local wind-driven resuspension of *in-situ* material and not episodic movement of material from the (Fly River) north. The exception is a peak in turbidity observed in June 2017 that seems to be unrelated to local environmental conditions. Analysis of satellite imagery in this period shows evidence of Fly River discharge entering the region. The predominance of fine terrestrial material at Bramble Cay indicates that there is long-term transport of material from PNG as the nearest source. However, this may be via more complex long-term transport mechanisms rather than short-term episodic events, which in turn will have implications for the potential transport of contaminants into the area.
- A clear signature of the Fly River discharge in the Torres Strait sediment communities was not detected. Changes in microbial community composition were related to several potential environmental drivers, including sediment fines content and metals. However, the amount of variation explained by each variable was low. Furthermore, metal concentrations were generally low suggesting that the correlations are unlikely to be causative, though the results might also suggest that the microbial assemblage is particularly sensitive. Alternatively, several of the metals found in this study (e.g. Cu, Zn, Fe) are essential micronutrients at low levels and are potentially associated with other nutrients such as sulphur, CO₃ and organic ligands, which could be selectively utilised by some microbes. Therefore, observed relationships could be a consequence of nutritional preferences rather than toxicological sensitivity. Spatial changes in microbial communities at most locations appeared to correlate best with a decreasing gradient of fines content from east to west. Metal concentrations best explained the community composition at the three sampling locations closest to the PNG mainland. Here, community change may be related to terrestrial run-off.
- A pilot study assessing the concentrations of metals contained in seagrass leaves from three sites near Saibai Island and the northern and central Warrior Reefs indicated that the three sites were unpolluted by metals, although the samples collected adjacent to PNG (site near Saibai Island) had higher metal concentrations than samples collected near the northern Warrior Reefs. Metal concentrations in samples from the Saibai Island site were comparable to samples collected from coastal sites in the GBR. In contrast, metal concentrations in seagrass collected at the two sites from the Warrior Reefs were similar to metal concentrations in seagrass collected from an offshore GBR site in the Howick Island group.
- The results from the partner project, NESP Project 2.2.2, also support the evidence obtained from this project. Results on the concentrations of mine-derived contaminants in waters and sediments at 21 sites across the Torres Strait showed that metals were detected in highest concentrations in the northern Torres Strait, including around Saibai Island and Bramble Cay. Correlations between suspended sediment and metal

concentrations indicate that the most likely source of metals are inputs of sediments and waters from the PNG mainland.

Overall, using multiple lines of evidence from the project has provided greater certainty of the extent and frequency of the influence of runoff from the Fly River in PNG on important marine ecosystems in the Torres Strait. It is now feasible to conclude that the influence of the Fly River and associated contaminants is focused in the northern Torres Strait and along the PNG coast, i.e. northern and NW Torres Strait. This varies from previous understanding, which indicated that the area of the Fly River influence in the Torres Strait extended to Bramble Cay and south towards Erub and Masig Islands, with some intrusion to the NW. The available evidence now supports that habitats located in the NE corner of the Torres Strait Protection Zone including Bramble Cay, north of Masig Island and NW as far as Boigu Island, are located in a higher potential risk area of exposure to brackish and turbid waters from or derived from the Fly River, as well as from/or derived from local PNG river discharges. While this movement of water from the Fly River is a historical pattern, the estimated 40% increase in sediment discharge associated with the operation of the Ok Tedi mine is likely to have changed the characteristics of sediment and contaminant concentrations in this region. Further work is required to assess the sources of influence and identify temporal links to Fly River discharge and gain a better understanding of the spatial extent of the influence of this discharge.

Analysis of the likely ecological impact of these influences is a subsequent step of investigation which was outside the scope of the current study. A number of further studies are recommended to assist in this assessment in the future. In particular, combined interpretation of the findings including a full risk assessment to ecosystems and communities would be required to assess the overall implications of the results of the exposure of sediments and trace metals to ecosystems in the northern Torres Strait and the probable sources. This would include a qualitative assessment of the potential implications of the findings for the local community. To support this work, local coral and seagrass monitoring efforts need to continue (and be reported) to support analysis of the potential correlations between the results and ecosystem health.

1. INTRODUCTION

1.1 Background

The Fly River in southern Papua New Guinea discharges into the Gulf of Papua (Figure 2-1) to the northeast (NE) of the Torres Strait. It is a large river by world standards, with an estimated flow volume of $6,000 \text{ m}^3 \text{ sec}^{-1}$ and it is the 17th largest river in the world in terms of sediment discharge (Galloway 1975; Milliman and Syvitski 1992).

Torres Strait Islanders depend on their marine resources for food, livelihoods and cultural activities. Fly River plumes have been observed in satellite images and oceanographic studies (e.g. this report; Petus 2013; Wolanski et al. 1999) regularly entering northern Torres Strait waters and can potentially threaten the quality of marine resources. However, the extent and scale of this threat is unknown, particularly as future climate projections are for more rainfall extremes and, potentially, larger river discharges.

Mining at the headwaters of the Fly River (the Ok Tedi gold and copper mine) in Papua New Guinea (PNG) started in 1981 with the mine becoming largely a copper mine (the gold having been exhausted) after 1985 and in the Strickland River, which is the largest tributary of the Fly River (the Porgera gold mine), in 1990. Estimates suggest that mining operations have increased sediment discharge from the whole of the Fly River by 40% (Wolanski et al. 1995b). The tailings and waste rock discharged from the mine are elevated in a suite of particulate and, to a lesser extent, dissolved metals including copper, lead, zinc, cadmium, arsenic and iron (Bolton et al. 2009). Given the close proximity of the Torres Strait to the mouth of the Fly River, concerns have also been raised since the start of mine operations that trans-boundary contamination may occur.

Under certain conditions, Fly River plume waters have been detected across the northern Torres Strait, east of the Warrior Reefs, as far west as Saibai Island and south to Masig Island (Wolanski et al. 2013; Martins and Wolanski 2015). These areas contain complex and important seagrass and reef communities that are potentially threatened by changes in water quality (Carter et al. 2014).

Exploration of model and remote sensing scenarios will help identify which environmental and oceanic conditions are drivers of Fly River plumes reaching the Torres Strait. To enhance the prediction of the Fly River plume behaviour and its impacts on Torres Strait ecosystems and dependent communities, this project was undertaken through the National Environment Science Program (NESP), building on previous work, to determine the following: (i) spatial extent, temporal patterns and constituent contaminants of Fly River discharge in the Torres Strait region and (ii) presence of ecosystems in the Torres Strait exposed to Fly River discharge (via the examination of pre-existing data). These studies were led by James Cook University (JCU; including TropWATER and the Centre of Excellence for Coral Reef Studies) in partnership with the Australian Institute of Marine Science (AIMS), University of New South Wales (UNSW), Macquarie University, University of Algarve, the Yantai Institute of Coastal Zone Research China and C2O Consulting. A partner project conducted by the Commonwealth Scientific and Industrial Research Organisation (CSIRO; NESP Project 2.2.2, Apte et al. 2018) has undertaken further analysis of metals in sediment and the water column at selected northern Torres Strait locations.

Using these multiple lines of evidence, it is possible to make an assessment of the potential influence of the Fly River discharge in the Torres Strait region. This information is important to describe the pressures on these regions of the northern Great Barrier Reef (GBR) given their importance to Torres Strait communities and turtle and dugong populations, and their connectivity with the Great Barrier Reef Marine Park (Johnson et al. 2018).

1.2 Project objectives

The primary **objectives** of this project were to:

1. Summarise the characteristics of Fly River discharges including hydrodynamics, contaminant sources and material transport—particularly trace metals.
2. Define the temporal patterns (frequency and duration) of plume suspended sediment delivery and transport to the Gulf of Papua and northern Torres Strait. This included historic analysis through coral coring techniques.
3. Undertake a preliminary desktop analysis to estimate the spatial and temporal extent of exposure of coral reefs and seagrasses in the Torres Strait to Fly River discharges.
4. Provide the results in a format that can be spatially and temporally delivered via the Torres Strait eAtlas to inform future environmental decision making.

The intended **outcomes** of the project were to obtain:

1. Greater certainty of the extent and frequency of the influence of runoff from the Fly River on important marine ecosystems in the Torres Strait.
2. A synopsis of the estimated spatial extent of Fly River discharge intrusion into the Torres Strait, the temporal patterns of discharge and the plume constituents. This will be informed by *in-situ* water quality monitoring, analysis of satellite imagery and hydrodynamic modelling. This relies on input from CSIRO for the analysis of toxic metals in sediments and documentation of anecdotal evidence of plume intrusions from local communities.
3. Analysis of the location and frequency of exposure of Torres Strait marine ecosystems to Fly River discharges, including a preliminary desk top assessment of the influence that exposure may have on the current status of vulnerable ecosystems in the study area.

This report presents the results of this project, supported by a synthesis of previous work in the region. We provide an overview of the Torres Strait region including its ecological features and provide a summary of the understanding of current status, pressures and threats to the region. The results are presented in Section 4 representing the major components of the project:

- Analysis of coral cores for freshwater influence;
- Marine hydrodynamic modelling;
- Remote sensing of water quality conditions;
- *In-situ* continuous monitoring of salinity and turbidity;
- Weekly salinity monitoring conducted by TSRA rangers;
- Water quality exposure assessment using the remote sensing outputs;
- Assessment of correlations between water quality datasets;
- Scoping of gene analysis in sediment samples to provide bacteria and infauna indices in response to environmental conditions; and
- Pilot study for the analysis of trace metals in seagrass leaves to assess potential influence of river discharge on these levels.

In collaboration with NESP Project 2.2.2, these lines of evidence provide a basis for a combined assessment of the hazards presented to vulnerable ecosystems and communities of the Torres Strait by Fly River contaminants of concern.

2. THE TORRES STRAIT REGION

2.1 Regional overview

The Torres Strait region covers an area of 48,000 km², of which only 2.6% is land mass, 6.2% is tidally inundated reef flats, and 91.2% is open seas, most of which are relatively shallow (20–60 m). The region is located on the border between Australia (northern Queensland) and PNG and stretches 200 km from the tip of the Cape York Peninsula to the southwest coast of PNG (Figure 2-1). There are more than 247 islands and cays in the Straits, 18 of which are inhabited and support an estimated 7,000 people (Torres Strait Regional Authority, 2018).

The Torres Strait Protected Zone is an area of the Torres Strait recognised by Australia and PNG that provides arrangements for Torres Strait Islanders and the coastal people of PNG to engage in their traditional way of life between international boundaries. For example, traditional people from both countries may move freely (without passports or visas) for traditional activities in the Protected Zone. The Protected Zone is defined in the Torres Strait Treaty (signed in December 1978 and entered into force in February 1985) as the border between Australia and PNG and provides a framework for the management of the common border area. The formation of the Protected Zone has also helped to preserve and protect the land, sea and air of the Torres Strait, including the native plant and animal life. For example, subsidiary management arrangements for commercial fisheries in the Zone have been put in place under the Treaty.

The Torres Strait has the largest continuous area of seagrass meadows in the world, significant areas of high diversity coral reefs, extensive areas of coastal mangroves and productive fisheries (Figure 2-1). It is rich in biodiversity and cultural significance, with its ecosystems being amongst the most pristine in the world and retaining a high degree of natural and wilderness values. It is protected from swell by the northern GBR and has strong tidal currents and irregular bathymetry with a narrow continental shelf. The region's variety of habitats support highly diverse Indo-Pacific marine flora and fauna, including dugongs and marine turtles (Sobtzick et al. 2014; Hamann et al. 2015a, 2015b). The strategic location of the Torres Strait places it at risk from the impacts of shipping, downstream impacts of mining and land clearing in PNG. Other major risks include climate change and local resource over-exploitation.



Figure 2-1: Map of the Torres Strait region showing the major rivers, islands, habitats and the boundary of the Torres Strait Protected Zone. Map prepared by D. Tracey, TropWATER JCU using datasets from Great Barrier Reef Marine Park Authority (GBRMPA), Carter et al. (2014), Hayward et al. (2008) and Carter and Rasheed (2016).

2.2 The Fly River and its catchments

The catchment area of rivers draining into the Gulf of Papua is $1.42 \times 10^5 \text{ km}^2$, and its rivers deliver approximately 350 million ML of freshwater to the Gulf annually, containing about 200 million tonnes per year of suspended sediments. The catchments of these rivers are largely undisturbed, except for the pre-existing small scale human settlements throughout the catchments, likely dating back thousands of years, and in the last 50 years two large mines Ok Tedi Mining Ltd in the headwaters of the Fly River and Porgera in the headwaters of the Strickland River, a major tributary of the Fly River. However, these catchments are subject to higher levels of natural disturbance than, for example, most Australian sites due to their seismicity and less stable geomorphology. For example, a landslide in 1989 contributed 125 Mt of sediment to the Fly River and in 8,800 BP a larger scale event is estimated to have contributed 7 km^3 of soil to the system (Bolton et al. 2009).

The Fly River discharges into the Gulf of Papua (Figure 2-2) to the NE of the Torres Strait. It is a large river by world standards, by flow volume (the 50th largest) at $6,000 \text{ m}^3 \text{ sec}^{-1}$. Despite the relatively small area of the catchment ($75,000 \text{ km}^2$) (International Union for Conservation

of Nature 1995; Figure 2-2), the Fly River was the 17th largest river in the world in terms of sediment discharge (Galloway 1975; Millman and Syvitski 1992) even before the construction of the Ok Tedi mine (Pickup and Marshall 2009). Before the construction of the Ok Tedi mine, the mean annual sediment discharge of the Ok Tedi/middle Fly River system was about 7 Mt/yr at Everill Junction (the junction point of the Strickland and the middle Fly Rivers), and the mean annual load carried by the Strickland River at Everill Junction was approximately 80 Mt/yr for a total load towards the mouth of the Fly of about 87 million tonnes (Dietrich et al. 1999) making the sediment discharge the 17th largest in the world (Walsh and Ridd 2009). In fact, the Fly River carries more sediment to the sea than all the rivers draining Australia combined (estimated to be about 60 million tonnes year⁻¹ by Millman and Meade, 1983). Copper and gold mining at the headwaters of the Ok Tedi River (Ok Tedi mine) started in 1981 when the main product was gold. Later after the gold was exhausted the mine became a copper mine. The Porgera gold mine commenced operations on the headwaters of the Strickland River in 1990. The mines are estimated to have caused a 40% increase in the sediment discharge by the Fly River of which they are tributaries (Wolanski et al. 1995b), i.e. to approximately 120 Mt/yr (Canestrelli et al. 2010) which would elevate the discharge to be the 12th largest in the world (compared to data from Harris et al. 1993 and Millman and Meade 1983). The Fly River and its tributaries are wet tropical rivers characterised by only small variations in freshwater discharge across and within years, even though significant discharge reductions have been observed during exceptionally strong drought periods associated with El Niño (Wolanski et al. 1997; Dietrich et al., 1999). The average freshwater river discharge at the delta is estimated to be approximately 6,000 m³/s, with typically $\pm 25\%$ seasonal variations (Wolanski et al. 1997). El Niño–Southern Oscillation (ENSO) has some impact on the delivery of Fly River fine sediment to the offshore environment by creating a reduced flow from the river and a concomitant reduction to the relatively constant sediment discharge. However, the Fly River is notably different than, for example, Australian rivers which have very variable flows both within years and between years. The flow (and hence sediment discharge) of the Fly is relatively constant across the year and with limited variability across years.

The discharge of copper from the Fly River, both in particulate and dissolved phases, has increased in recent decades associated with mining operations indicating a significant increase in the copper content of waters and sediments in the Fly River estuary (Angel et al. 2010a, 2014). The ecological impacts of increased copper loads in the lower Fly River itself and the delta (see also Waterhouse et al. 2013) has not been studied extensively, although Stauber et al. (2009) examined the likely speciation, bioavailability and toxicity of copper in the system. The presence of mine-derived sediments has also been noted in the delta by Walsh and Ridd (2009) based on copper concentrations in the sediment, which have increased since mining commenced. Their conclusions were based on the results of Chappell (1992) and Baker (1998) who report that a pronounced post-mine copper signal is preserved in sediment from discrete areas in the delta.

2.3 Ecological features of the region

The Torres Strait region includes 247 islands and cays, large areas of coral reefs, seagrass meadows and mangroves. The massive freshwater and sediment input from nearby coastal rivers in PNG influence this unique marine ecosystem (Gehrke et al. 2011). The influence of increased runoff of sediment associated with mining in PNG delivered by these large river

flows has been identified as a major threat to seagrass in the Torres Strait by increasing turbidity, which may lead to reduced light availability and potential sedimentation (Coles et al. 2012). This effect of reduced light may also affect reefs in the NE. However, 'natural' large sediment discharges have been occurring for thousands of years. Water currents in the Torres Strait are driven by wind, tides and adjacent water circulation patterns in the Coral Sea, the northern GBR continental shelf, the Gulf of Papua and the Gulf of Carpentaria. The net east-west water flow through Torres Strait is small, and areas of shallow waters or those that are densely covered with reefs and islands are poorly flushed. Only reef passages and reef-free open waters are relatively well flushed as demonstrated by current movements (Figure 2-3).

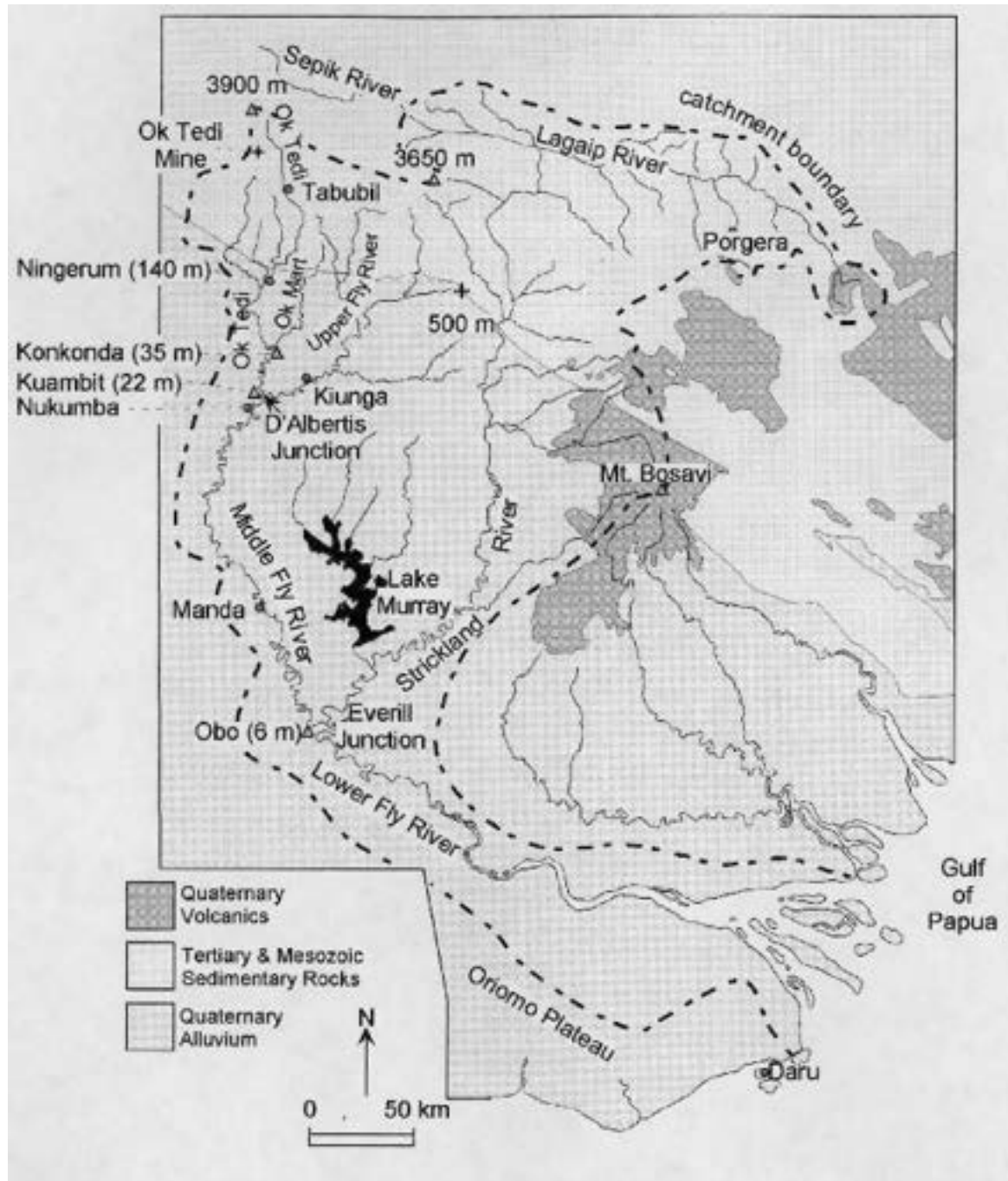


Figure 2-2. Map of the Fly River catchment, PNG. Reproduced from Dietrich et al. (1999).

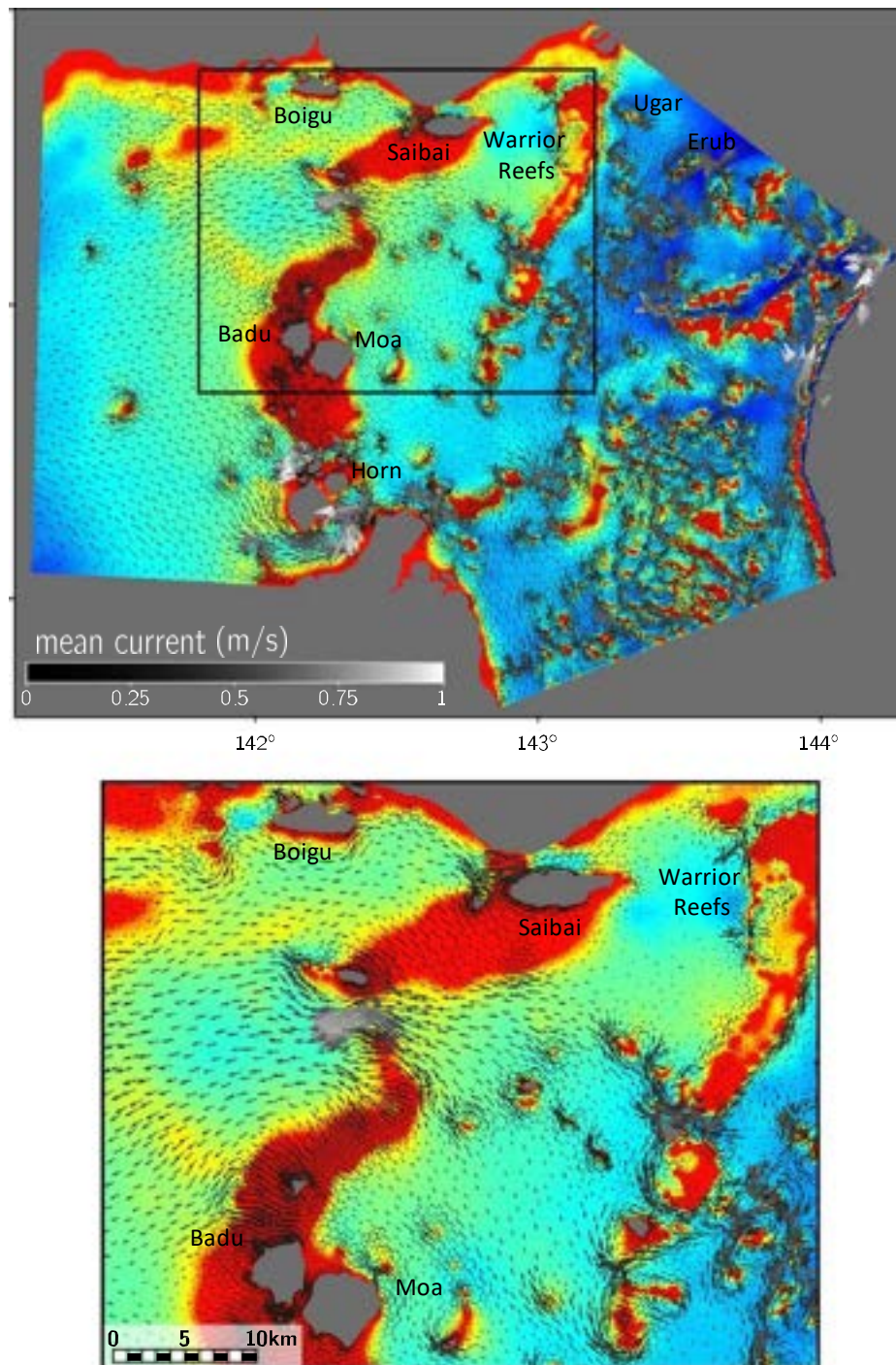


Figure 2-3: Mean currents (m/s) and general circulation model of Torres Strait in (top) the entire Torres Strait and (bottom) a zoomed-in view in the north western Torres Strait. The mean velocity of the currents is represented by the intensity in the grey scale of the arrows (where the lightest arrows represent the highest velocities – top figure). Colours represent depth from red (< 20 m) to blue (maximum depth 200 m). Source: Wolanski et al. (2013).

Coral reefs of the Torres Strait are part of the northern section of the GBR and provide ecological functions that influence the broader GBR. As elsewhere, climate change, potential crown-of-thorns starfish (COTS, *Acanthaster cf. solaris*) outbreaks, coral diseases, potential increased frequency of severe storms, and pollution from river runoff and shipping are threatening the ecological integrity of Torres Strait reefs. The Torres Strait Regional Authority (TSRA) coordinates a monitoring programme to measure key indicators of coral health and to

document major changes to coral reefs, especially as a result of climate change, such as coral bleaching events (Bainbridge et al. 2015). The Torres Strait has some of the most extensive seagrass meadows in northern Australia and possibly the world, with 13 species of tropical seagrasses, present mainly in sub-tidal waters. The wetlands—mangroves, salt marsh and freshwaters—on the islands and cays in the Torres Strait region are ecologically important and support a diversity of biota. Whilst freshwater wetlands are rare, most Torres Strait islands have extensive mangrove areas and several islands (e.g. Saibai and Boigu Islands) are predominantly made up of intertidal wetlands. Many Torres Strait islands are low lying with large wetland areas expected to be vulnerable to the effects of climate change, e.g. sea level rise and increased storm surge frequency (Duke et al. 2015).

Torres Strait marine habitats support significant populations of dugongs (Sobtzick et al. 2014); hawksbill, green and flatback turtles (Hamann et al. 2015a, 2015b)—species of conservation concern—as well as sharks, fish and invertebrate species, many of which are important for local fisheries, such as tropical rock lobster and sea cucumbers (Welch and Johnson 2013). In Queensland, dugongs (*Dugong dugon*) are listed as vulnerable under the *Nature Conservation Act 1992*. Green (*Chelonia mydas*) and flatback (*Natator depressus*) turtles are listed as vulnerable under the Australian *Environment Protection and Biodiversity Conservation Act 2000* and the Queensland *Nature Conservation Act 1992*.

2.4 Current status, threats and pressures to ecosystems

2.4.1 Status and trends of Torres Strait ecosystems potentially influenced by the Fly River

Located on one of the world's most extensive continental shelves, the Torres Strait has been recognised for its ecological complexity and biodiversity. The region has significant tropical marine ecosystems and populations of important and vulnerable marine species, including the most important dugong habitat in the world (Sobtzick et al. 2014).

Coral reefs

Strong physical drivers in the Torres Strait—large tidal ranges, strong currents and turbidity—have influenced the formation and character of the ~1,200 coral reefs, with an east-west elongation of reefs. Coral reefs dominate in the clear warm waters on the eastern shelf and form the northern extent of the GBR system (Figure 2-4). Until recently, Torres Strait reefs were in good condition with high coral cover, presence of major taxonomic and functional groups, and minimal incidence of coral disease (Sweatman et al. 2015, TSRA 2016); however, following mass bleaching and coral mortality in 2016 (Hughes et al. 2017, 2018a, 2018b) their status has declined greatly. The 2016 bleaching event has been linked to a pool of warm water moving from the western Torres Strait across to the eastern Torres Strait and down into the northern GBR (Wolanski et al. 2017). Final coral mortality data for the Torres Strait following the 2016 bleaching event have yet to be released. Corals in the Torres Strait were not recorded as being affected by the 2017 GBR mass bleaching event, where damage was most severe to the central GBR.

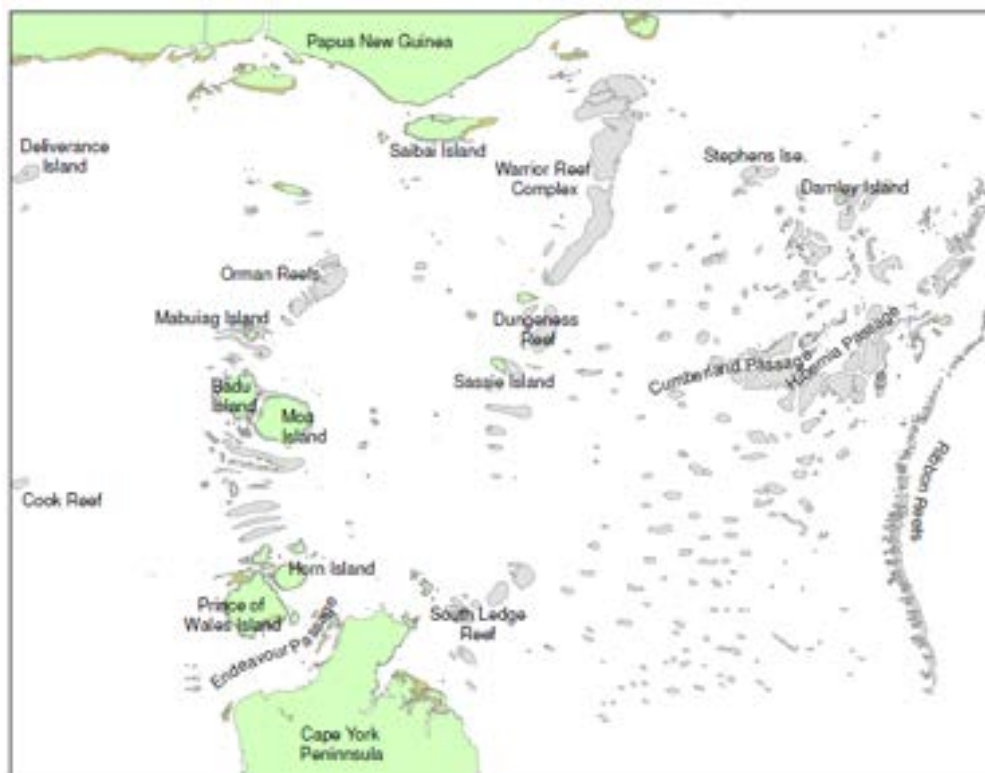


Figure 2-4: Spatial distribution of coral reefs (grey) in the Torres Strait region. Source: AUSLIG; Haywood et al. (2008).

The large sediment discharges from the Fly River (and possibly from other large rivers flowing into the Gulf of Papua, e.g. the Purari River) (Figure 2-1) influence reefs in the northern Torres Strait (Wolanski et al. 2013). This results in reefs with muddy carbonate sediments in the northern areas, while sediments in the southern and western areas of the Torres Strait are mainly reef-derived with > 80% carbonates (Sweatman and Berkelmans 2012). The geological structures of reefs, particularly in the more turbid waters of the western Torres Strait, include extensive reef tops and reef flats covered in soft sediments and seagrasses as well as growing reef edges and slopes that have more hard substrates and corals (Sweatman and Berkelmans 2012).

Reef surveys in the central and eastern reefs in 2014 documented 246 scleractinian (hard) coral species of which 77 are new records for the Torres Strait and six are potentially new records for Australia. This includes five species that were new records for the GBR and Torres Strait region: *Acropora spicifera*, *Cantharellus jebbi*, *Herpolitha weberi*, *Montipora palawanensis* and *Pavona bipartite* (Sweatman et al. 2015).

Prior to 2015, mean live coral cover was generally < 30% and was highest on the eastern Ribbon Reefs (up to 47%) and very low on the western reefs (Haywood et al. 2007; Sweatman et al. 2015). Surveys in 2013 recorded 279 species of coral belonging to 60 genera, with branching and digitate coral growth forms the most common in the region, and massive corals locally dominant on the Warrior Reefs (Osborne et al. 2013; Figure 2-5). Species richness varied along an east-west gradient, with the Acroporidae and Pocilloporidae coral families having higher richness on the eastern reefs and Poritidae, Fungiidae and Mussidae having higher richness on central reefs (Osborne et al. 2013).

These results indicate that the Torres Strait may be a coral reef biodiversity hotspot. Coral communities from central sites differed from those in eastern sites, reflecting the gradient in turbidity and wave exposure (Osborne et al. 2013; Figure 2-5). The locations fell into two groups according to a cluster analysis based on species in common. The reefs of the central Torres Strait islands of Poruma, Aureed (in the Bourke Isles) and Masig formed one group (hereafter Central) and those of the NE islands, Erub and Mer formed the second group (hereafter Eastern). The similarity between reefs reflects the physical location of the reefs along the environmental gradient from ESE to WNW. With the exception of Aureed Reef, the total number of species was fairly consistent across the gradient, implying that species replacement, rather than species loss or gain, was occurring with the changing environmental gradient. Differences in species richness between Central and Eastern reefs were observed for five coral families. Species richness in the Acroporidae and Pocilloporidae was higher on the Eastern reefs than that on the Central reefs. Conversely, richness of Poritidae, Fungiidae and Mussidae species were higher on the Central reefs than that on the Eastern Reefs (Sweatman et al. 2015).

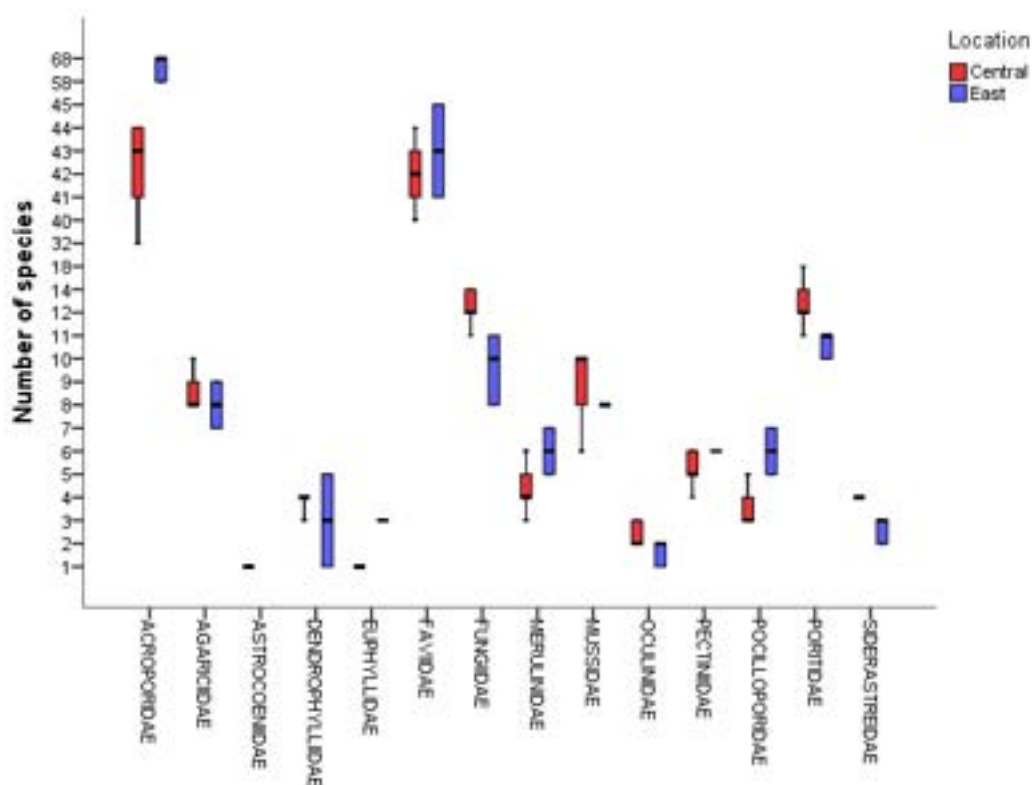


Figure 2-5: A spatial comparison of the number of species in each scleractinian coral family (bottom axis) in the Central and Eastern reefs of Torres Strait (Osborne et al. 2013).

Baseline surveys conducted by the TSRA in 2015 at 20 sites in five locations using standard Reef Health and Impact Surveys plus manta tow documented the current condition of reefs in the region (TSRA 2016). The survey results showed that hard coral cover was moderate to high based on standard conditions in the GBR World Heritage Area. The average cover of macroalgae was low at all sites, except for Ugar Reef, which also recorded the lowest average percentage of hard coral cover. An increase in macroalgae cover can be a sign of reef degradation, with potential causes including reduced numbers of herbivorous fish or poor water

quality due to excess nutrients and sediments. The presence of COTS was recorded on all surveyed reefs; however, the density of COTS was recorded as No Outbreak (GBR standards) at every site except for Erub and Masig. Successive surveys at Erub identified a low incidence of COTS that was therefore assessed as being a low threat. However, surveys at Masig identified a constant presence of COTS that reached outbreak densities during one survey and may indicate some underlying issue (TSRA 2016).

A network of *in-situ* ocean monitoring stations measures current environmental conditions and potential stressors, such as unusually high sea temperatures. Coral bleaching was reported in western Torres Strait in 2010 and again in 2016, coinciding with high water temperatures. While the data show no long-term ocean warming, a small temperature rise and calm conditions in 2010 saw widespread coral bleaching for the first time in the region (Bainbridge and Berkelmans 2014), which, as noted above, was repeated in 2016 with the most severe and extensive bleaching event ever recorded (Hughes et al. 2017). To understand this threat, temperature loggers and ocean monitoring stations have been deployed and satellite images have been used to understand how patterns of warming in the Pacific Ocean impact the Torres Strait. Data show that conditions for 2011–2014 were cooler than normal, consistent with regional climate conditions, with a corresponding low risk of bleaching. The satellite data, however, show significant warming in the Pacific, which was the driver of the El Niño conditions during the 2015–2016 summer (Bainbridge et al. 2015) that caused severe coral bleaching in the Torres Strait in early 2016. This bleaching was associated with a pool of very warm water (Wolanski et al. 2017) leading to widespread coral mortality (Hughes et al. 2017, 2018a) and dramatic shifts in coral species assemblages (Hughes et al. 2018b).

Seagrass and species of conservation concern

Seagrasses cover an estimated 13,425 km² (Coles et al. 2002) to 17,500 km² (Poiner and Peterkin 1996) in the Torres Strait region and seagrass are found predominantly in the western and central (Warrior Reefs) areas (Figure 2-6). In contrast to coral reef extent, seagrass habitats dominate in the more turbid and sediment-laden conditions in the west that are likely to be influenced by a number of small coastal rivers flowing into the northern and northwestern Torres Strait (Haywood et al. 2007) and potentially also the Fly River under certain weather and oceanic conditions (Wolanski et al. 2013). The northern Warrior Reefs have a greater diversity of seagrasses (*Halodule* spp., *Thalassia* spp., *Thalassodendron* spp. and *Cymodocea* spp.), while the south tends to be dominated by a single species of *Thalassia*. In the west, seagrass communities are more diverse to the north (*Halodule* spp., *Thalassia* spp. and *Syringodium* spp.), around the Orman Reefs (Haywood et al. 2007). Seagrass monitoring from 2002 to 2014 documented high seagrass biomass in the Warrior Reefs, the eastern edge of the Dugong Sanctuary, and reef top meadows and surrounding islands between Prince of Wales Island and Orman Reefs (Carter et al. 2014). Seagrass diversity hotspots were identified in the central Torres Strait between Horn, Wednesday and Hammond islands, between Badu and Moa islands, and the eastern edge of the Dugong Sanctuary (Carter et al. 2014). The region's diverse seagrass communities contain 13 species recorded in estuaries, reef flats and subtidal areas adjacent to continental islands (McKenzie et al. 2010).

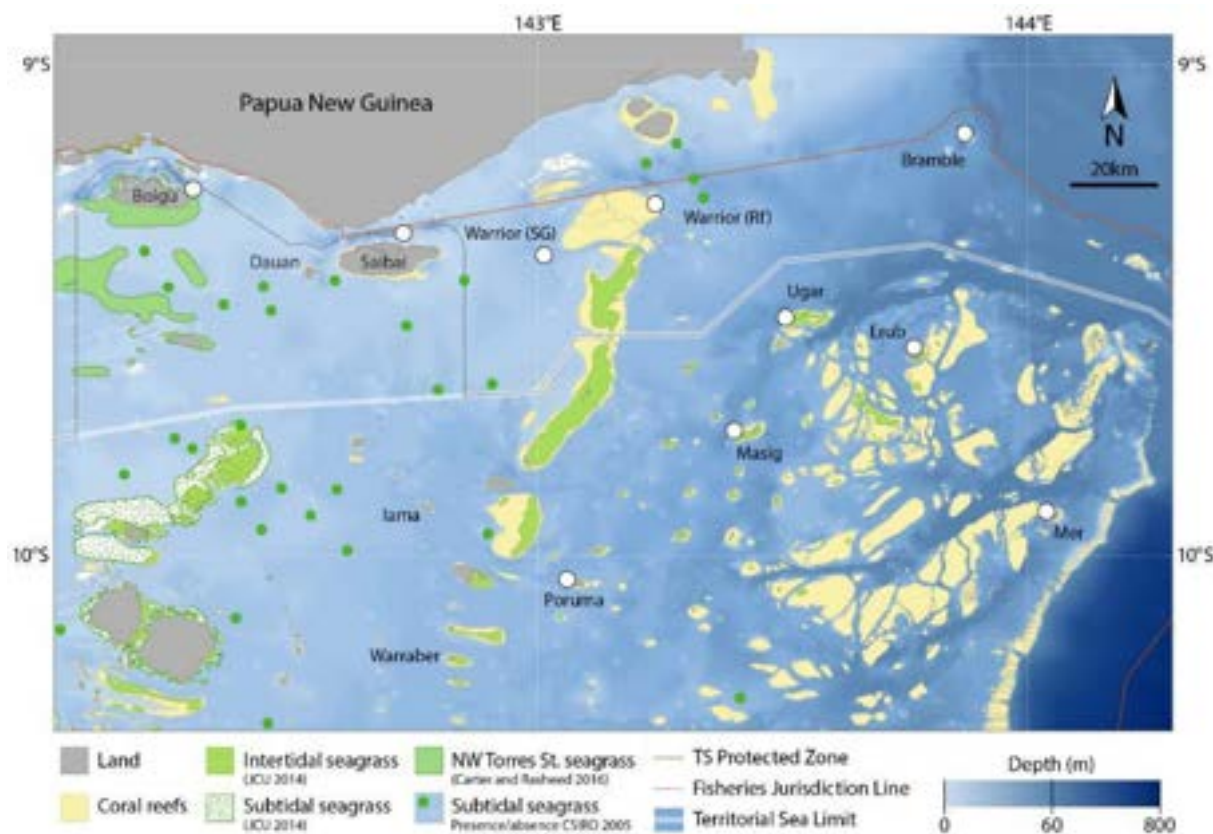


Figure 2-6: Spatial distribution of seagrasses in the Torres Strait region showing the mapped extent of intertidal and subtidal seagrass. Map prepared by D. Tracey, TropWATER JCU using datasets from GBRMPA, Carter et al. (2014), Hayward et al. (2008) and Carter and Rasheed (2016).

Aerial surveys in 2014 estimated the population of dugongs in the central and western Torres Strait at 103,000 animals (Hagihara et al. 2016; Appendix 2). This is the largest aggregation of dugongs globally, with dugongs moving between reef and non-reef areas in both Australian and PNG waters (Figure 2-7). Aerial surveys since 1987 suggest that the population has not changed significantly over time (Sobtzick et al. 2014). These aerial surveys also found a significant population of turtles (most likely greens; Fuentes et al. 2015) in the survey region, including within the Dugong Sanctuary in the western Torres Strait.

Monitoring of nesting turtles since 2006 has identified the islands in the Mer group (Mer, Dauar and Waier) and Bramble Cay as the most significant green turtle rookeries in the Torres Strait and the main nesting sites for the Torres Strait/northern GBR genetic population. Nesting success (percent of female turtles that emerge each night to lay eggs) at Dauar Island and Bramble Cay was 33% in 2006, 62% in 2007 and 50–60% in 2008–2009. Data suggest that rookeries receive around 100 to 3,000 green turtles per year, but the size of nesting female green turtles is declining. The largest green turtle population is in the western and central Torres Strait with estimates of ~600,000 large juvenile and adult animals (Hamann et al. 2015a).

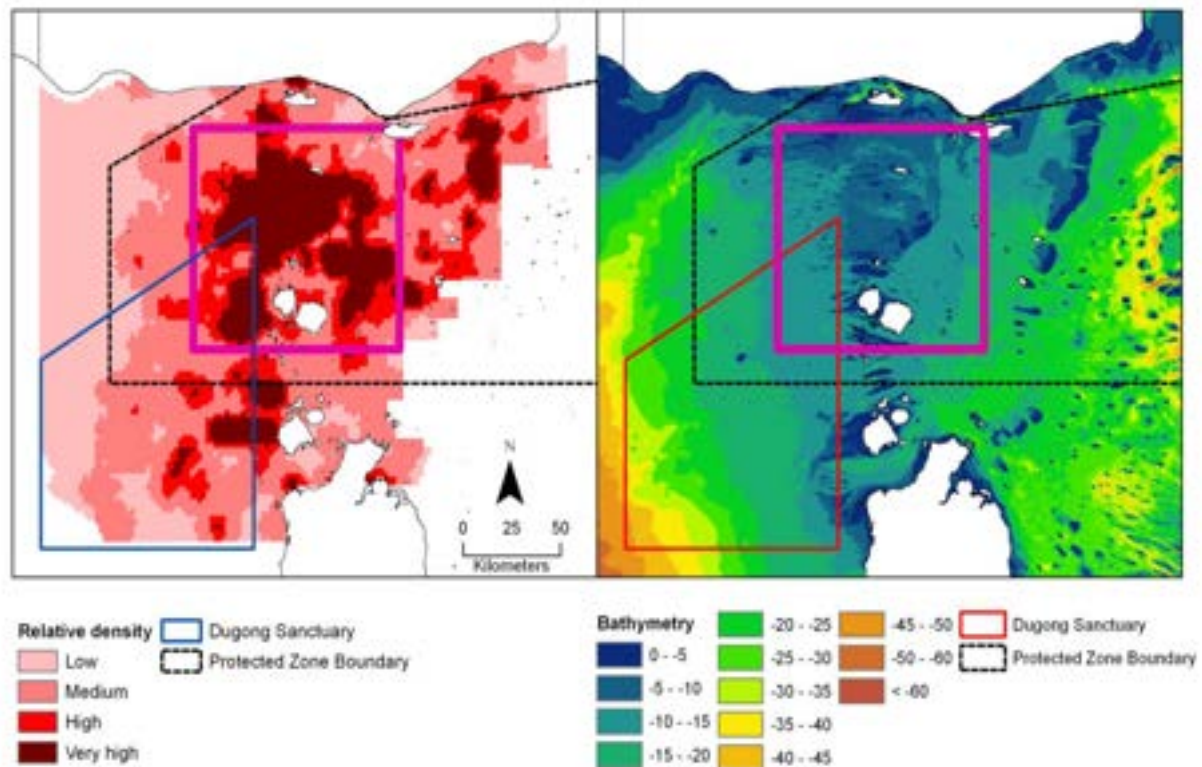


Figure 2-7: Relative density of the dugong population in the Torres Strait (left) and the bathymetry of the region showing the concentration of dugongs around shallow areas as well as their use of deeper non-reef areas (right). Source: Soltzick et al. (2014).

The islands of western and southern Torres Strait, plus the adjacent mainland beaches are important for flatback turtle nesting. Warul Kawa (Deliverance Island) is an important flatback turtle rookery and it receives around 100 to 200 turtles per year. It is likely that the rookeries form part of the same genetic population as Crab Island and the Western Cape in the Gulf of Carpentaria. Tracking of flatback turtles demonstrates that they undertake migrations of between 100 and 2,000 km towards foraging areas in the west as far as the Kimberley coast of Western Australia and to the north into the waters of Indonesia and PNG (Hamann et al. 2015b). The foraging areas of Torres Strait turtles are typically large (516–4,324 km²), and are larger than those of green and loggerhead turtles in eastern Australia.

Marine fish biodiversity

Coral cover is higher in the eastern Torres Strait, supporting a high diversity and abundance of reef fish and sea cucumber (Haywood et al. 2007). A total of 323 fish species have been recorded in the region (Haywood et al. 2007; Osborne et al. 2013) and fish abundance is greatest on the eastern reefs. Lutjanids (snappers), Serranids (groupers) and Zanclids (Moorish idol) dominate on the Ribbon Reefs, and Caesonids (fusiliers) dominate in the Central Warrior Reefs (Haywood et al. 2007). For both corals and reef fishes, the communities from the Central reefs differ from those in the Eastern reefs, reflecting a gradient in turbidity and wave exposure (Osborne et al. 2013).

Marine fish biodiversity assessed at five reefs (Aureed Island, Masig Island, Poruma Island, Erub Island and Mer Island reefs) in 2013 documented 301 fish species. Of these, 143 were new to the list of known species for the region (Haywood et al. 2007), increasing the species

list of reef-associated fish in the Torres Strait to 326. Fish communities from the Central sites differ from those in the Eastern sites, reflecting the gradient in turbidity and wave exposure (Osborne et al. 2013). Fish communities include species from the northern GBR and to a lesser extent species normally associated with reefs further north in PNG. There is no data available to assess any influence of longer-term changes in turbidity on the local and regional fish populations that might be associated with increased sediment discharges from the Fly River.

2.4.2 Pressures and threats: current and future

The Torres Strait is under increasing pressure from a range of regional factors—PNG population growth and resource demand, downstream mining and development impacts, increased shipping, and climate change—as well as local factors—resource exploitation, emerging tropical diseases and inadequate waste management. External drivers affect the entire region, namely climate change, mining discharge and shipping. Internal or local drivers show some differences between the Torres Strait Islands and the Western Province region and Treaty Villages in PNG partially due to differences in social factors (Butler et al. 2015). Natural resource extraction is escalating in Western Province, and related infrastructure is placing additional pressures on coastal and marine resources in southern PNG and the Torres Strait.

A study in 2013–2014 of potential water quality issues in the Torres Strait region identified regional pollution—discharge of metal (and other) contaminants from the Fly River associated with mining in the upper catchment, the port at Daru, other mines in PNG, land clearing and shipping—as well as local pollution—sewage and stormwater discharge—as key risks to the region, now and in the future (Waterhouse et al. 2013). A 2D second-generation Louvain-la-Neuve ice-ocean model (SLIM) hydrodynamic model of the transport of water-borne material to determine the delivery and fate of contaminants revealed the large-scale flow dynamics in the Torres Strait, highlighting that some areas are flushed relatively quickly while water (and associated contaminants) tends to stagnate in some shallow areas. The model also revealed the prevalence of high energy small-scale flow dynamics near shoals, reefs, islands and passages (Wolanski et al. 2013).

Preliminary analysis of moderate resolution imaging spectroradiometer (MODIS) satellite images suggested that high turbidity levels along the PNG coast were constrained to the coast, and that potential intrusions of Fly River plumes into the Torres Strait were limited to the northern islands (Saibai and Dauan) and possibly northwestern islands (Boigu) and reefs. High turbidity levels recorded along the southwestern PNG coast suggested a combined influence of turbid outflows from the several rivers draining the southern PNG margin, enhanced by bottom resuspension in the shallow coastal zone (Petus 2013). Information on the current (and potential future) status of contaminant sources in the region identified three main sources of water quality contaminants in the Torres Strait (Waterhouse et al. 2013):

1. Local island waste management, including sewage and waste disposal,
2. Shipping, commercial vessels and marine infrastructure, and
3. Large-scale developments in adjacent areas (such as PNG).

Of these, the large-scale developments were considered to have the potential to pose the most consistent threat at the largest scale, although there was limited direct evidence to support this. Hence, the need to gain a better understanding of the likely patterns of influence of the

Fly River discharge and mine-derived contaminants was identified, with the current study designed to deliver this information in the first instance, and Project 2.2.2 aiming to collect further evidence of current water quality conditions potentially linked to this influence in the Torres Strait.

As described in the Introduction, the prominent water quality issues likely to be associated with the Fly River discharge are increased sediment loads, and elevated trace metal concentrations associated with mining waste discharge in the upper Fly River catchments.

3. EXISTING EVIDENCE OF THE EXTENT OF INFLUENCE OF FLY RIVER DISCHARGES IN THE TORRES STRAIT REGION

3.1 Water quality monitoring in the Torres Strait

Monitoring and reporting on the status of water quality in the Torres Strait over the past two decades has focused on the assessment of the impact of water quality; particularly with regard to the influence of expanding/continuing large scale development in PNG, on the health of the local communities and ecosystems. Previous monitoring programmes (reviewed in Waterhouse et al. 2013) have investigated water quality through the assessment of water, sediment and biota. Studies include:

1. The Torres Strait Baseline Study (1990):
 - Torres Strait Baseline Study Pilot study 1991–1992: assessment of trace metal concentrations in selected marine organisms, sediment and seagrass (Dight and Gladstone, 1993).
 - Torres Strait Baseline Study Comprehensive study 1992–1993 (Gladstone, 1996).
 - Torres Strait Baseline Study Assessment of heavy metals in commercial prawn and crayfish species and marine sediments in the early 1990s (Evans-Illidge, 1997).
2. Sediment sampling (1998–2000): assessment of heavy metal concentrations in traditional seafood and the viability of undertaking long-term monitoring of temporal changes (Haynes and Kwan 2002).
3. Monitoring mangrove cockle (1998–2001) (Haynes and Kwan 2001; Haynes and Kwan 2002): assessed arsenic, cadmium, chromium, copper, mercury, nickel, lead, selenium and zinc.
4. Pesticides in sediments (2000) showed no detectable concentrations of herbicides or insecticides in subtidal sediments (Haynes and Johnson 2000).
5. Seagrass and coral reef ecosystem assessment: ongoing since the late 1990s. Studies have identified turbidity, often the result of sediment resuspension (tides and storms), as the greatest influence on sporadic seagrass die back events (Livingston et al. 1998; Longstaff and Dennison 1999; Haynes and Kwan 2001; Saint-Cast and Condie 2006; Marsh and Kwan, 2008; Bainbridge et al. 2015).
6. The 2003 assessment of the decline in coral reef ecosystems rated the Torres Strait reefs relatively well (i.e. in relatively good condition) compared to other global systems (Pandolfi et al. 2003).
7. *In-situ* water quality monitoring using data loggers (2014–2017). AIMS have been monitoring temperature at a number of sites throughout the Torres Strait and have established three real-time ocean monitoring sites (two funded as part of the National Environmental Research Program; NERP; and a third funded from other sources) that monitor temperature, salinity and total suspended solids (TSS) to provide an indicator of current status and variability in water quality conditions throughout the Torres Strait (Bainbridge et al. 2015).
8. A desktop hazard assessment of water quality threats to Torres Strait marine waters and ecosystems (NERP Project 4.4, Waterhouse et al. 2013) including 2D hydrodynamics modelling (Wolanski et al. 2013).

9. Monitoring and assessment of water quality threats using remote sensing imagery to assess turbidity and current movement, including assessment of Fly River plume incursion into the Torres Strait, and the quantification of bioavailable dissolved metals using diffusion gradient in thin-films (DGTs) and oysters as bio-monitors (O'Brien et al. 2015).

Conclusions drawn from these studies:

- Metal concentrations in mangrove cockle tissue:
 - Only showed significant variation in nickel between sampling years (concentrations in organisms collected in 1998 were significantly higher than those sampled in 1999).
 - A majority of metals exhibited significant variation between sampling regions. Differences in the distribution of metals were more easily distinguished in mangrove cockles than in sediments. In particular, concentrations of copper are higher in both northern and central Torres Strait waters compared with concentrations present in southern Torres Strait waters. Cockles however tended to be difficult to locate at many sampling sites, which often resulted in significantly reduced sampling numbers, making the species unsuitable for routine monitoring in the region.
- The influence of the Fly River on trace metal concentrations in sediments and selected indicator organisms of Torres Strait, including the burrowing clam (*Tridacna crocea*), mangrove cockles (*Polymesoda erosa*) and several seagrass species was largely limited to northeastern Torres Strait and trace metal concentrations in marine sediment from the north-central region of Torres Strait were determined to be primarily influenced by smaller coastal rivers flowing from PNG (Gladstone 1996). This result was supported by other studies that concluded that the impact of Fly River discharge on local trace metal concentrations was generally restricted to the northern Torres Strait (Baker et al. 1990; Alongi et al. 1991; Baker 1991).
- Results indicated high cadmium bioavailability in the Torres Strait; however, the high concentrations are not due to pollution (Evans-Illidge 1997). Cadmium is a naturally occurring element in the marine carbonate sediments that are prevalent throughout the Torres Strait and it is not a metal associated with Fly River runoff. However, it is known that indigenous people of the Torres Strait have greater potential for cadmium exposure and renal damage than other Australians due to high cadmium in some traditional seafood and a high prevalence of Type 2 diabetes, hypertension, smoking, and obesity (Haswell-Elkins et al. 2008).
- Assessment of metal concentrations in commercial species (crayfish and prawns) found general agreement in the concentrations of most metals found in comparable tissues (Evans-Illidge 1997). Exceptions included arsenic and copper that occurred in higher concentrations in crayfish and cadmium that was found in much lower concentrations in crayfish than that in prawns. These results were supported by further monitoring undertaken by Haynes and Kwan (2001). Final recommendations from these studies was to advise individuals to avoid the consumption of the hepatopancreas and head tissues of these organisms.
- Temperature from the established ocean monitoring stations has been reported by Bainbridge et al. (2015) and Bainbridge (2016) and is described as an important proxy for water movement and mixing that distributes temperature and other water attributes within the Torres Strait.

- Understanding the hydrodynamics of the Torres Strait region is required for confident predictions of contaminant transport. Through NERP Project 4.4 the existing dataset on the oceanography of Torres Strait (separating Australia and PNG) was reviewed and used to demonstrate that the water circulation in the Torres Strait is driven by the wind, the tides and circulation in the Coral Sea, the northern GBR continental shelf, the Gulf of Papua and the Gulf of Carpentaria (Wolanski et al. 2013). These data were used to set the open boundary conditions for a high resolution, finite element, depth-integrated (2D) model of Torres Strait. The model showed that the water circulation in the Torres Strait is characterised by events lasting a few days to three weeks and that this explains the observations of a very small net circulation in the Torres Strait. The model predicted that net east-west flow through the Torres Strait is small (in agreement with field data) and reveals that areas of shallow waters and areas that are densely populated with reefs and islands are poorly flushed. Only reef passages and reef-free open waters are relatively well flushed.
- Remote sensing as part of the O'Brien et al. (2015) study concluded that the MOHID1 water modelling system was successful at predicting the spread of river plumes when applied to the Gulf of Papua and that the mechanism explaining the episodic intrusion of fresh water in the Great NE Channel and Torres Strait seems to be a delicate equilibrium between the eastward action of fresh water input and Coriolis force, and the westward action of the trade winds and coastal anticlockwise current (Martins and Wolanski 2015).
- Monitoring of dissolved metal using the DGTs facilitated the measurement of 11 bioavailable metals in the water column at nearshore marine deployment sites at eight locations (O'Brien et al. 2015).
- Deployment and retrieval of exposed oysters was successful at the Horn Island deployment site (a control site for the Fly River) but not at the other two test sites at Masig and Saibai Islands (potentially influenced by the Fly River). Oysters were retrieved after 3- and 6-months deployment off Horn Island. There was an increase in the tissue concentration of six metals (aluminium, arsenic, manganese, vanadium, chromium and lead) of the exposed oysters deployed, which indicated that the use of oysters as bio-indicators has monitoring potential. However, the losses of the oysters at the other sites indicated that site selection and security are important considerations that need to be addressed for successful long-term deployments of this type.

3.2 The 2014 *in-situ* monitoring of dissolved metals

The O'Brien et al. (2015) study was the first monitoring programme to measure dissolved metal concentrations in marine waters in the Torres Strait. The DGTs are considered “artificial bivalves” that facilitate the measurement of an average concentration of dissolved metals for the period (between 3 and 5 days) for which the devices have been exposed (Zhang and Davison 1999). DGT techniques are based on a simple device that accumulates solutes on a binding agent after passage through a hydrogel that acts as a well-defined diffusion layer. The use of this technique reduces some of the problems inherent in the analysis of water, sediment and biota samples for metals (i.e. contamination prevention and preservation requirement).

¹ Modelo Hidrodinamico; 'hydrodynamic' in Portuguese

The technique was first proposed for routine use in the Torres Strait by Haynes and Kwan in 2001, and the DGTs were subsequently shown to be a suitable tool for TSRA rangers to employ in the assessment of bioavailable dissolved metals given training and administrative support for the local staff members involved was provided. The metal concentrations recorded indicated that there is a possible northern river influence on the concentrations of copper and to a lesser extent arsenic at the most northern sites monitored (Saibai Island and Bramble Cay) while urban/industrial influence on the concentrations of chromium, lead and arsenic were observed at Thursday, Horn and Wednesday Islands.

In the study by O'Brien et al. (2015) deployment of the DGT samplers was undertaken by local ranger staff across nine islands (Thursday, Horn, Wednesday, Badu, Masig, Warraber, Erub and Saibai Islands and Bramble Cay). However, not all units were retrieved and analysed successfully.

The concentrations of the metals detected in the study were generally low compared to the concentrations reported in other north Queensland studies including sites at Cleveland Bay, Pioneer Bay, Nelly Bay, Townsville Harbour, Ticklebelly Bay, Lizard Island, Orpheus Island and Heron Island for cadmium, copper, lead, nickel and zinc (Denton and Burdon-Jones 1986a, 1986b, 1986c; Esslemont 2000). Metal concentrations (average, minimum and maximum) detected across all sites are presented in Table 3-1. In summary:

- Copper concentrations were highest in samples collected at Saibai Island and Bramble Cay with concentrations ranging from 0.06 to 0.6 µg/L compared to 0.004 – 0.2 µg/L across all other sites. While two samples were ≥ 0.3 µg/L (the 99% species protection guideline), all copper concentrations detected were comparable to the concentrations measured at islands in the GBR (0.13–0.18 µg/L) (Denton and Burdon-Jones 1986c; Esslemont 2000) and low compared to the concentrations of 0.38–1.42 µg/L measured at Cleveland Bay/Townsville Harbour (Denton and Burdon-Jones 1986c; Esslemont 2000) and a mean of 0.53 (+ 0.8) µg/L at Port Curtis (Angel et al. 2010b).
- The concentrations of arsenic observed ranged between 0.27 and 2.46 µg/L, comparable to the concentration of dissolved arsenic (0.39–1.35 µg/L) measured in north Australian coastal and estuarine waters (Munksgaard and Parry 2001). The highest concentrations of arsenic recorded were measured at the southern islands, Thursday and Wednesday Islands, in samples collected in December 2014. However, arsenic concentrations were below the analytical detection limit in samples collected during other sampling periods. Arguably the arsenic concentrations across the other sites showed a possible slight northern river influence (given the more frequent detection of arsenic), and while there was no significant difference between the concentrations there was some increase in concentration as distance from the Fly River decreased.
- Concentrations of cadmium and vanadium were low compared with the range detected in waters at other north Queensland marine systems (Burdon-Jones et al. 1982; Burdon-Jones and Denton 1984a, 1984b; Esslemont 2000).
- Metals that are normally linked to urban and industrial sources include chromium, cobalt, lead, manganese and zinc. Chromium, specifically Cr(VI), is widely used in industrial applications (brake linings, motor vehicle exhaust, some inks, paints, paper, rubber and some timber preservation products; Australian Government 2014) and has been identified as an important urban solid-waste contaminant parameter (Borai et al. 2002). Chromium was only detected at the southern sites indicating a likely urban

influence in these areas. Cobalt, lead and manganese were detected at all sites and concentrations showed limited urban influence (i.e. concentrations somewhat higher at southern island sites); however, higher concentrations of these metals were also observed in samples collected at Saibai Island in June 2014.

- For most metals, background concentrations were not an issue; however, zinc was detected at higher than normal concentrations in the field blanks that lead to higher than normal detection limits (up to 1.6 µg/L). This was attributed to potential contamination issues (particularly with regard to zinc as it is present in many sunscreens) and sampling staff were instructed in the use of sampling methods that would minimise the potential risk of contamination. Thus, no conclusion was made on the trends in zinc concentrations within the Torres Strait or potential sources.
- Nickel and aluminium are ubiquitous in the environment, which are reflected in the results obtained that showed no trend in concentrations across the sampling sites.

Table 3-1: Average, minimum and maximum metal concentrations (µg/L) at each monitoring site compared to concentrations at other North Queensland marine sites. Shaded cells indicate values that exceed the Australia and New Zealand Water Quality Guideline (2000) values. Chromium analysis did not differentiate between Cr(III) and Cr(VI)). Note: ECL = Environmental Concern Level; 99% SP = 99% species protection guideline; ND = not detected.

			Aluminium	Arsenic	Cadmium	Chromium	Cobalt	Copper	Lead	Manganese	Nickel	Vanadium	Zinc
Site	Distance to Fly River mouth (km)	Guideline value	ECL 0.5	ECL 3.2	99% SP 0.7	Cr(VI) 0.14 Cr(III) 7.7	99% SP 0.005	99% SP 0.3	99% SP 2.2	Trigger 80	99% SP 7	99% SP 50	99% SP 7
Thursday	260	Average	4.497	1.953	0.005	0.411	0.017	0.054	0.035	1.639	0.178	2.230	ND
		Min	4.007	1.810	0.003	0.260	0.003	0.008	0.005	0.642	0.099	1.988	
		Max	4.986	2.133	0.006	0.491	0.041	0.086	0.084	3.806	0.291	2.484	
Horn	255	Average	8.930	0.990	0.008	0.260	0.011	0.068	0.029	0.983	0.179	1.629	2.786
		Min	5.426	0.869	0.007	0.173	0.000	0.039	0.012	0.201	0.118	1.543	1.158
		Max	14.405	1.127	0.010	0.339	0.025	0.093	0.051	2.014	0.262	1.757	4.913
Wednesday	250	Average	9.793	1.869	0.002	0.184	0.009	0.076	0.022	0.900	0.154	2.406	3.308
		Min	2.374	1.326	0.001	0.141	0.002	0.032	0.005	0.371	0.098	1.810	3.202
		Max	17.211	2.457	0.003	0.242	0.019	0.238	0.046	1.694	0.242	3.356	3.413
Badu	225	Average	10.793	0.996	0.007	0.194	0.009	0.074	0.027	0.733	0.178	1.708	1.179
		Min	4.185	0.935	0.004	0.150	0.003	0.053	0.013	0.441	0.115	1.650	0.556
		Max	17.401	1.056	0.011	0.261	0.017	0.100	0.059	1.099	0.241	1.766	1.504
Warraber	190	Average	ND	0.636	0.002	ND	0.003	0.012	0.003	0.246	0.065	1.724	0.016
		Min		0.384	0.001		0.002	0.004	0.001	0.182	0.051	1.279	0.016
		Max		0.823	0.003		0.005	0.018	0.006	0.336	0.085	2.222	0.016
Erub	90	Average	0.790	0.837	0.005	ND	0.003	0.055	0.008	0.359	0.092	2.330	0.935
		Min	0.178	0.272	0.002		0.000	0.027	0.000	0.196	0.071	1.512	0.100
		Max	2.184	1.120	0.008		0.013	0.089	0.015	0.546	0.117	3.089	2.399
Saibai	135	Average	6.306	1.245	0.011	ND	0.019	0.251	0.021	1.574	0.211	2.575	0.938
		Min	0.437	0.875	0.007		0.004	0.129	0.007	0.749	0.153	1.688	0.215
		Max	24.228	1.570	0.017		0.096	0.608	0.082	6.315	0.396	3.671	3.554
Bramble Cay	50	Average	1.276	0.962	0.006	ND	0.004	0.133	0.008	0.518	0.103	2.073	0.591
		Min	0.273	0.692	0.004		0.002	0.063	0.003	0.352	0.089	1.758	0.189
		Max	2.871	1.200	0.009		0.007	0.249	0.016	0.767	0.124	3.103	1.249
All sites	--	Average	24.228	2.457	0.017	0.491	0.096	0.608	0.084	6.315	0.396	3.671	4.913
		Min	0.178	0.272	0.001	0.141	0.000	0.004	0.000	0.182	0.051	1.279	0.016
		Max	4.823	1.071	0.006	0.262	0.009	0.105	0.018	0.863	0.143	2.137	1.327
Other Nth Qld sites*	> 300	Average											
		Min			0.05			0.06	0.13		0.06		0.3
		Max			1.34			2.56	1.84		0.54		15

*Denton and Burdon-Jones (1986a, 1986b, 1986c); Esslemont (2000).

3.3 Assessment of oysters for the bioaccumulation of metals

Accumulation of metals from the water column by bivalve shellfish has been shown to occur rapidly and to reflect ambient exposure concentrations for many metals. The mangrove cockle has been shown to be a suitable indicator organism for the Torres Strait based on its metal accumulation characteristics and relatively widespread distribution (Haynes and Kwan 2001). However, collection of the mangrove cockle is difficult and resulted in inconsistencies in sample numbers collected (Haynes and Kwan 2001). O'Brien et al. (2015) encountered difficulties when undertaking deployment of oysters that were native to the area (mostly *Pinctada albina*) sourced from the Friday Island Pearl Farm. Thirty oysters were exposed at three sites: Horn, Masig and Saibai Islands. The loss of all oysters from 2 of the 3 deployment sites (retained at Horn Island only) limited the amount of data available to assess the efficacy of oysters as a monitoring method when applied within the Torres Strait region.

While the results from only one deployment site did not facilitate the assessment of oysters as bioindicators of bioavailable metal concentrations the results obtained from the Horn Island deployment indicated that the change in metal tissue content was consistent over the full six months of deployment for some metals and variable for arsenic, cobalt and vanadium. This is consistent with other studies that have observed that the rate at which metals are accumulated/released by the oysters is dependent on both biological processes and local water metal concentrations (Forsberg et al. 2006; Menegário et al. 2017). The results obtained may indicate that the concentration of arsenic, cobalt and vanadium within the oyster tissue are more reactive to changes in local dissolved water concentrations while the concentration of other metals accumulated in the tissues is more dependent on biological uptake/release processes.

3.4 Potential impacts of increased sediment and trace metal inputs

As demonstrated by the evidence above, the detailed status of water quality as a result of increased sediment inputs and trace metal concentrations in the Torres Strait region is largely unknown, let alone having an understanding of the ecological implications of these inputs. However, to provide some indication of the potential implications of the exposure of Torres Strait marine ecosystems to the Fly River discharge, the potential impacts can be deduced from knowledge in other marine ecosystems including the GBR. Key points from the published literature are summarised below.

Increased sediment loads

While evidence of the impacts of increased sediment loads in the GBR may be transferrable to the Torres Strait to some degree, there is one very important difference. The discharge patterns of the Fly River are much more consistent (and larger) than those in the GBR and hence, the influence of the sustained elevated sediment concentrations and resulting increases in turbidity from the Fly River may not be directly comparable to ecosystem responses in the GBR. In the GBR, major seagrass losses have occurred (Collier et al. 2012; Petus et al. 2014) following a series of large discharge events with large loads of fine sediment followed by prolonged resuspension of the fines in waters less than 15m deep and extended periods of reduced light (Fabricius et al. 2014, 2016). However, periods of reduced river

discharge of fine sediment and hence disturbance have allowed for seagrass recovery (for example, Wells and Rasheed 2017). The Fly River discharge is relatively consistent and therefore, ecosystems are likely to be exposed to both sediment plumes and resuspension events every year and throughout the year. Despite these differences, the ecosystem responses to increased sediment in the receiving environment of the Fly River discharge are expected to be similar to many inshore areas of the GBR but have occurred with different patterns of exposure. In addition, limited monitoring data exists to substantiate these potential impacts in the Torres Strait, or to compare current conditions to conditions prior to the commencement of Ok Tedi mine and the increased sediment discharge, highlighting that this information should be used as a guide only. This scenario is evident in the Torres Strait as identified in Section 4.3 and 4.6. This re-mobilisation of material can also release contaminants that were previously stored in the bottom sediment.

The potential impacts of increased fine sediment inputs to reef ecosystems are summarised below.

- Increased suspended particulate matter can cause several stressors including light reduction, disturbance by suspended particles and sedimentation (Jones et al. 2015; Duckworth et al. 2017). The impacts depend on the intensity and duration of exposure (e.g. Collier et al. 2016; Ferguson et al. 2017; O'Brien et al. 2018; Statton et al. 2018).
- Light limitation is a primary driver of seagrass production in the GBR (Collier and Waycott 2009) and reductions in light availability have been directly linked to seagrass loss (Collier et al. 2012; Petus et al. 2014). Under low light conditions photosynthetic carbon fixation reduces in seagrass meadows which requires biochemical, physiological and morphological changes to minimise respiratory loss (O'Brien et al. 2018) and to enhance light capture (e.g. Collier et al. 2009; McMahon et al. 2013; Hedley et al. 2014; Schliep et al. 2015). Mortality occurs if light levels drop below minimum light requirements (e.g. Longstaff and Dennison 1999). When benthic light is very low or negligible, the time to mortality ranges from two weeks for small 'colonising' species (*sensu* Kilminster et al. 2015) to two years for 'persistent' species that can resist mortality by drawing on storage reserves in their large rhizomes (Collier et al. 2009; O'Brien et al. 2018).
- Light reduction, elevated suspended sediments and sediment deposition negatively affect the reproductive cycle and early life histories of corals (Jones et al. 2015). Some coral reef species are highly sensitive to reduced water clarity, largely from the loss of light for photosynthesis (Erftemeijer et al. 2012, Bessell-Brown et al. 2017) while others can feed on the particulate matter gaining energetic advantage over others as long as light is not limiting (Anthony 2000; Anthony and Fabricius 2000). Sensitivity depends on life stage, with young corals being particularly susceptible to damage of suspended particulate matter on its own (i.e. without sedimentation or light reduction). Fertilisation and subsequent settlement, and the growth and/or survival of coral juveniles can also be negatively affected (Humanes et al. 2017a, b).
- Ecosystem-wide shifts can occur from species responses to increased turbidity. For example, studies of spatial gradients in coral reef communities have shown that in some locations the abundance, biomass and species diversity of corals, invertebrates and fish can be lower on turbid reefs than non-turbid reefs or reefs adjacent to healthier river basins (Fabricius et al. 2005, Mallela et al. 2007; Rodgers et al. 2012; de Bakker et al. 2017). Shifts from corals to macroalgae (De'ath and Fabricius 2010; Begin et al.

2016; de Bakker et al. 2017), and at more severe conditions, shifts from macroalgae to heterotrophic filter feeders have been observed (Birkeland 1988).

- Reef fishes can be impacted by reduced water clarity and sedimentation indirectly, by changing their coral and seagrass habitats, or directly, as increased sediment loading can have direct behavioural, sub-lethal, and lethal impacts on fish (Wenger et al. 2017).

Elevated trace metal concentrations

Changes in trace metal concentrations in Torres Strait waters could have a direct impact on coral reef and seagrass meadow health as well as being a public health concern through contaminated seafood. Physiological responses, such as reduced photosynthesis in zooxanthellae (Siddiqui and Bielmyer-Fraser 2015; Patel and Bielmyer-Fraser 2015) and oxidative stress in both host and symbiont, have been reported consequences of metal exposure in corals and other cnidarians (e.g. Brock and Bielmyer 2013; Patel and Bielmyer-Fraser 2015; Siddiqui and Bielmyer-Fraser 2015; Bielmyer et al. 2010; Schwarz et al. 2013). Specific stress responses of corals and reef organisms to elevated levels of toxic metals (and specifically possibly to copper) were recently described for the GBR in Pratchett and Hoogenboom (2019) and are likely to be relevant in the Torres Strait. Examples include:

- inhibition of coral fertilisation and reduced reproductive success (Reichelt-Brushett and Harrison 2005; Richmond et al. 2018; Gissi et al. 2017);
- decreased settlement and survival of coral larvae (Reichelt-Brushett and Harrison 2000; Woods et al. 2016);
- changes in the population and growth of zooxanthellae (endosymbiotic algae) (e.g. Rodriguez et al. 2016);
- changes in the rate of photosynthesis resulting in a decrease in coral calcification and growth rates during the juvenile polyp stage (e.g. Biscéré et al. 2017; Ferrier-Pages et al. 2001);
- increased coral bleaching (e.g. Reichelt-Brushett and Hudspeth 2016; Sabdono 2009);
- reduced thermal tolerance in some corals specifically associated with copper enrichment (Banc-Prandi and Fine 2019);
- enhanced coral mortality especially at the juvenile stage (e.g. Sabdono 2009); and
- outright mortality in invertebrates and fishes (Peters et al. 1997; Wenger et al. 2015).

For fish effects include:

- immuno-suppression, leading to increased susceptibility to disease;
- impaired olfaction (essentially sense of smell);
- disruption of critical behaviours such as predator avoidance, social interactions and reproduction;
- disruption of the metabolism of sex hormones; and
- adverse effects on embryonic development in the egg by changing development rates and causing malformations, and to reduce hatching success.

Seagrass is generally reported as being in good condition in the Torres Strait (Carter and Rasheed 2018; Carter et al. 2018).

The main stress responses of seagrasses to elevated levels of heavy metals reported in the literature has not been intensively summarised. However, Bonanno and Orlando-Bonaca (2017, 2018) note the following evidence:

- Seagrasses and macroalgae accumulate similar levels of trace elements;
- Seagrasses and macroalgae show similar tolerance to high levels of trace elements;

- Seagrasses can accumulate high concentrations of trace elements;
- Trace element accumulation in seagrasses is mainly element- and organ-specific;
- For most trace elements, phytotoxic levels in seagrasses are unknown;
- Seagrasses and macroalgae act as bioindicators of trace element pollution; and
- Validated monitoring protocols that combine seagrasses and macroalgae are scarce.

Overall seagrasses are not particularly susceptible to metal pollution in comparison to animals and hence a more immediate threat is when metals are biomagnified up the food chain from seagrasses into seagrass grazing organisms such as dugongs and turtles.

Globally Govers et al. (2014) and Vonk et al. (2018) summarised concentrations of metals in seagrass and the importance of metals to seagrass functioning. Govers et al. (2014) concluded that trace metal concentrations in seagrass leaves, regardless of the species, can vary over a 100-1000-fold range, and are related to the level of anthropogenic pressure, making seagrasses highly valuable indicators. The analysis of seagrass leaf element content (Vonk et al. 2018) showed correlations with macronutrients (N and P), indicating that productivity also depends on the presence of other elements. While there is limited data on trace metals in seagrass leaves in the Torres Strait or the GBR, Haynes (2001) and Thomas et al. (2018) do provide useful comparative data relevant to the current study (see Section 4.93.4). Further work is required both locally and globally to understand the importance of various metals to seagrass health and productivity.

Potential impacts on seagrass meadows can also have flow-on effects for other species such as seagrass-feeding animals (e.g. green turtle and dugong) which have an important dietary role in the Torres Strait communities (Gladstone 1996) and where dietary intake may cause human health issues (Nunez-Nogueira et al. 2019).

4. RESULTS FROM THIS STUDY

Runoff from the Fly River in PNG influences water quality conditions in the Torres Strait region; however, the extent and frequency of this influence, and the potential ecological impacts, are not well understood. Further investigation is required to understand the prevalence and frequency of the extension of the Fly River plume into the Torres Strait, and the characteristics of plume constituents, particularly metals. This project builds on previous efforts to determine the spatial extent, temporal patterns and constituent contaminants of Fly River discharge, and to a lesser extent, assess the vulnerability of ecosystems in the Torres Strait exposed to the discharge.

4.1 Coral cores

Purpose: Historical analysis of coral cores (from Erub Island and Bramble Cay) to identify freshwater influence. **The detailed results of this component are reported separately in Lough (2016).**

Highlights:

- Historical analysis of existing *Porites* coral cores from Erub Island and Bramble Cay (longest period 1781–1993) was used as a method to identify freshwater influence in these locations over many years.
- Annual coral growth rates for all three cores are typical of massive *Porites* corals living in the average sea surface temperatures of the northern Torres Strait.
- There is no evidence of growth hiatuses in any of the cores that might be associated with environmental stress events.
- Luminescent lines are evident in both coral cores from Bramble Cay but not the coral core from Erub Island (during 1831–1994). At Bramble Cay, there are usually several pulses of freshwater each year, likely associated with atmospheric or oceanic processes pushing freshwater onto the reef. These pulses of freshwater are evident back to 1781.
- La Niña events tend to be associated with higher rainfall totals and El Niño events with lower rainfall totals in the Fly River catchment (and concomitantly influence Fly River flows).
- These results are correlated with the hydrodynamic model outputs (see Section 4.2).
- It should be noted that the Ok Tedi mine commenced production in 1984. Any recent impacts (1993 onwards) are not captured in this study.

4.1.1 Overview of methods and results

The backbone of tropical coral reef ecosystems is sustained production of calcium carbonate (CaCO_3) skeletons (calcification), which withstand the natural forces of physical and biological erosion and provide the structural complexity to support the many thousands of reef-associated organisms characteristic of a healthy reef. Certain massive corals, such as *Porites* spp., contain annual variations in the density of CaCO_3 skeleton revealed in X-rays of slices of the coral (Knutson et al. 1972). These annual density bands consist of a high-density band, usually formed in summer and a low-density band, usually formed in winter. As some corals continuously form skeleton over many years, these natural, dateable archives can provide

insights into coral growth and environmental conditions (through various tracers incorporated into the skeleton) over tens to hundreds of years (Lough 2010).

Skeletal luminescence (as a proxy for freshwater/river flow; Lough et al. 2002, 2015) and skeletal density (to derive annual growth characteristics; Lough and Cooper 2011) were measured in slices from three *Porites* coral cores from the AIMS Coral Core Archive. Two cores were from Bramble Cay at the northeastern edge of the Torres Strait region (~55 km southeast (SE) of the mouth of the Fly River) and one core from Erub Island (Darnley), ~46 km southwest of Bramble Cay in the eastern section of Torres Strait (Figure 4-1). Detailed methods are described in Lough (2016).

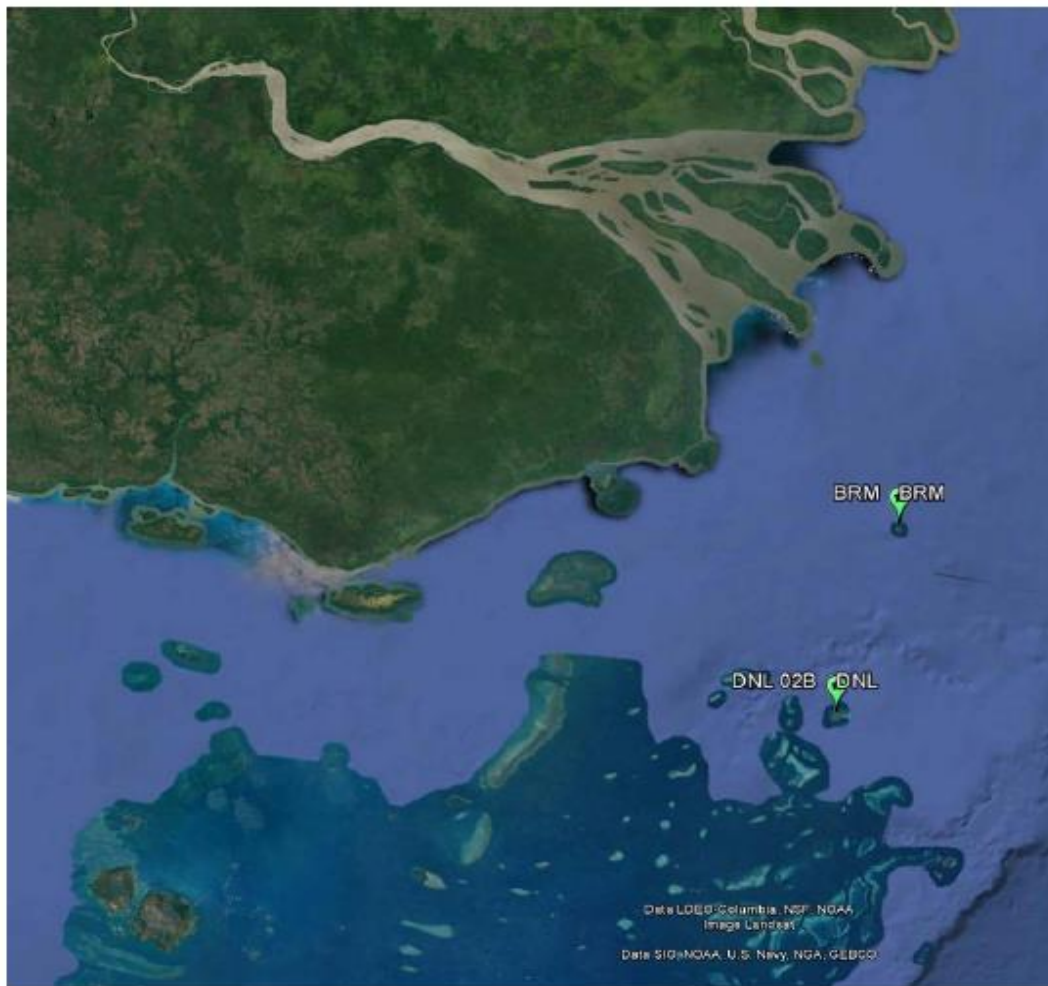


Figure 4-1: Map showing the locations of two coral coring sites—Bramble Cay (BRM) and Erub (Darnley) Island (DNL).

4.1.2 Recommendations

This study demonstrated the common occurrence of the Fly River plume at Bramble Cay, in the far northeastern region of the Torres Strait. Further analysis of coral cores would provide additional evidence of long-term influences and could include:

- Collection and analysis of cores from other northern locations in the Torres Strait region, e.g. Warrior Reefs, Saibai, Duaun or Boigu Islands.

- Analyses for mine-associated trace metal in coral cores that show evidence of Fly River exposure, e.g. from Bramble Cay.

4.2 Hydrodynamic modelling

Purpose: Evaluate wind and current intensities that result in freshwater intrusions into the Torres Strait. Use MOHID 3D coastal model of Gulf of Papua with 2D-SLIM high-resolution model for the Torres Strait²; calibrated and validated using observational data and tracing studies. **The detailed results of this component are reported separately in Li et al. (2017).**

Highlights:

- The modelling suggests that the plume of potentially polluted river water from the Fly River is probably mostly of concern in the Torres Strait region during the SE trade wind season. During that season the plume is permanent, although variable in characteristics following fluctuations of the wind and differences in mean sea level (ΔMSL ³) across the Torres Strait.
- The study also reviewed the historical field data from ship-borne observations and oceanographic moorings, which showed:
 - a permanent intrusion into the northern top third of the Torres Strait of brackish, river plume waters from the Gulf of Papua;
 - lower salinity in the east than in the west; and
 - strong seasonal variation with the largest and most intense intrusions during the SE trade wind season, and the smallest and least intense intrusions during the monsoon season.
- Analysis of the field data describing the physical oceanographic processes that control this intrusion showed that the key influencing factors are freshwater discharge in the Gulf of Papua; the wind; the MSL in the Gulf of Carpentaria, the Gulf of Papua and the northern Coral Sea; and tidal mixing.
- Field data was used to drive oceanographic models to describe the behaviour of the fresh water plumes over the Gulf of Papua and their intrusion in the Torres Strait using the 3D MOHID oceanographic model. The model predictions were successfully verified against field data. The river plume intrusion was largely limited to the northern third of the Torres Strait and was more intense in the east than in the west. The plume was vertically well-mixed throughout the Torres Strait. Tidal influences have a lower influence than other physical oceanographic influences.
- The MOHID model results were used at the open boundaries to drive the high resolution, 2D-SLIM model. The SLIM model demonstrated the high variability of the extent and dilution of the Fly River plume in the Torres Strait.
- This study was limited to understanding the intrusion of the Fly River plume in the Torres Strait. It did not address the intrusion of Fly River fine sediment and its particle-bound contaminants.

² Wolanski et al. (2013) and Martins and Wolanski (2015).

³ Difference in mean sea level (MSL) across the Strait. ΔMSL is > 0 when the MSL in the Gulf of Papua is greater than the MSL in the Gulf of Carpentaria.

The large-scale patterns identified in the modelling are consistent with the key findings from the remote sensing analysis (Section 4.3); however, the modelling of salinity and remote sensing of turbidity are not directly comparable due to alternative sources of sediment and turbidity in the region including the dominant processes of resuspension.

4.2.1 Overview of methods and results

We used the MOHID 3D model for the large-scale modelling of Torres Strait and the Gulf of Papua. The model was forced by imposing sea level (tide + MSL taken from altimetry) at the open oceanic boundaries, the fresh water discharge at its land-sea boundaries and the wind at Coconut Island. The sea level was imposed at the open boundary of the 2D external model as a wave synthesised from 14 tidal constituents, varying along the length of the boundary in addition to the MSL. The tidal constituents were extracted from the Finite Element Solution 2004 Global Tide data product (AVISO+) for a series of 30 points bordering the 2D model boundary and then interpolated by linear triangulation to the boundary cells. Table 4-1 summarises the river discharges used in the model.

Table 4-1: Freshwater river discharge used in the MOHID model.

River	Discharge ($\text{m}^3 \text{s}^{-1}$)
Fly	6,000
Aramia	2,000
Kikori	4,000
Era	2,000
Purari	4,000

For open boundaries corresponding to the SE trade wind season, the MOHID model predicted a river plume that covered much of the coastal waters of the Gulf of Papua (Figure 4-2). The plume was highly stratified in salinity in the Gulf of Papua and vertically well-mixed in the Torres Strait, in agreement with past observations. MOHID also predicted that the Fly River plume intruded in the Torres Strait the most during the SE trade wind season, i.e. when the ΔMSL across the Strait was > 0 (i.e. the MSL in the Gulf of Papua was greater than the MSL in the Gulf of Carpentaria), and that the plume intruded the least in the Torres Strait during the monsoon season when $\Delta\text{MSL} < 0$ (Figure 4-3 and Figure 4-4). During the SE trade wind season the plume width inside the Torres Strait and the salinity within the Torres Strait depended on the ΔMSL across the Torres Strait (Figure 4-4). This prediction is also in general agreement with past observations described above. Most importantly, the MOHID predictions of the salinity within the Torres Strait during the SE trade wind season agree well with the observations of September 1994, as illustrated in Figure 4-5.

MOHID also suggested an explanation for the observation of patchiness of the Fly River plume in August 1994, indicating that the temporal shutting down of the SE trade wind resulted in increasing the MSL in the Gulf of Carpentaria; this resulted in reversing the net currents in the Torres Strait that pushed back the established Fly River plume eastward towards the Gulf of Papua, generating a low salinity patch in the northern Torres Strait (Figure 4-5).

MOHID was successful in reproducing the salinity plume in the Torres Strait in August and September 1994, which were the only detailed salinity data for the Torres Strait available during the SE trade wind season.

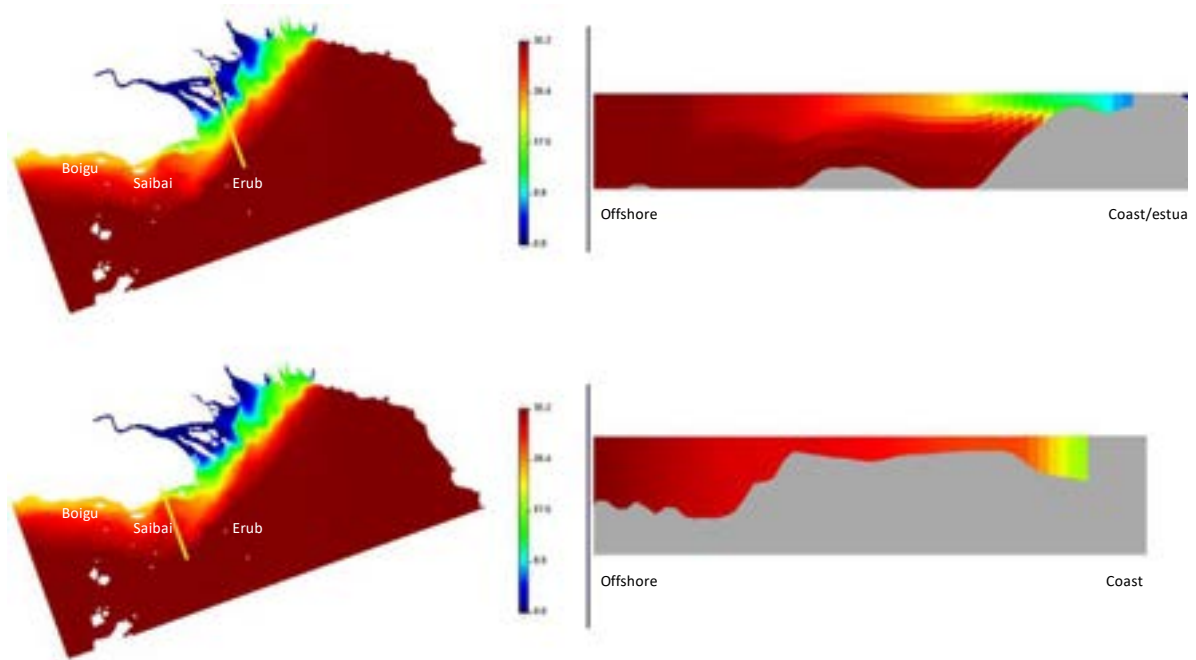


Figure 4-2: (Left) MOHID-predicted surface salinity distribution during strong SE winds. (Right) MOHID-predicted vertical distribution of the salinity along the transects shown by the yellow lines on the left.

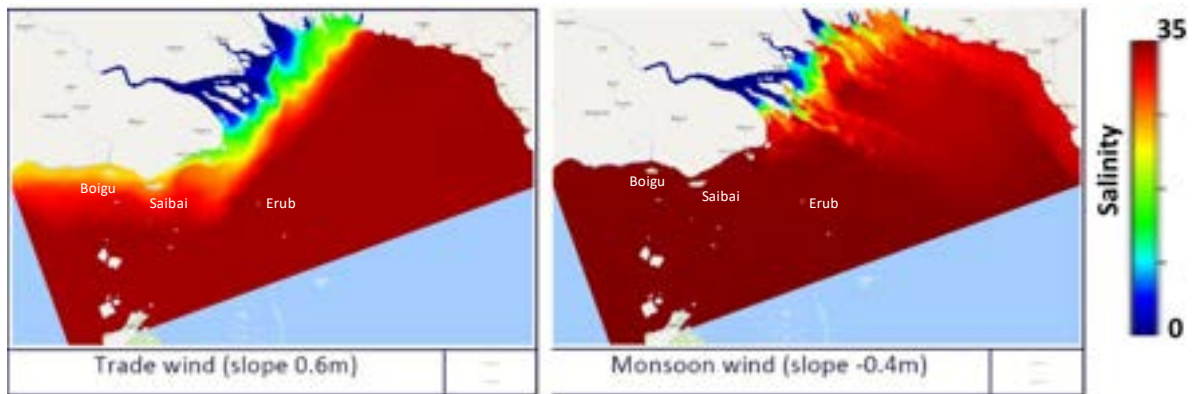


Figure 4-3: MOHID-predicted surface salinity distribution in the Gulf of Papua and Torres Strait during (left) the SE trade wind season and (right) the monsoon season.

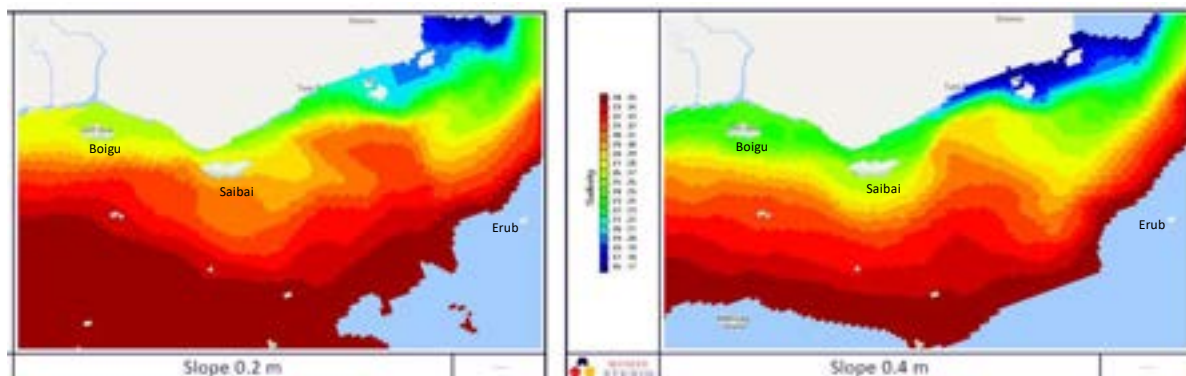


Figure 4-4: MOHID-predicted surface salinity distribution during the SE trade wind season for two different values of the Δ MSL across the Torres Strait.

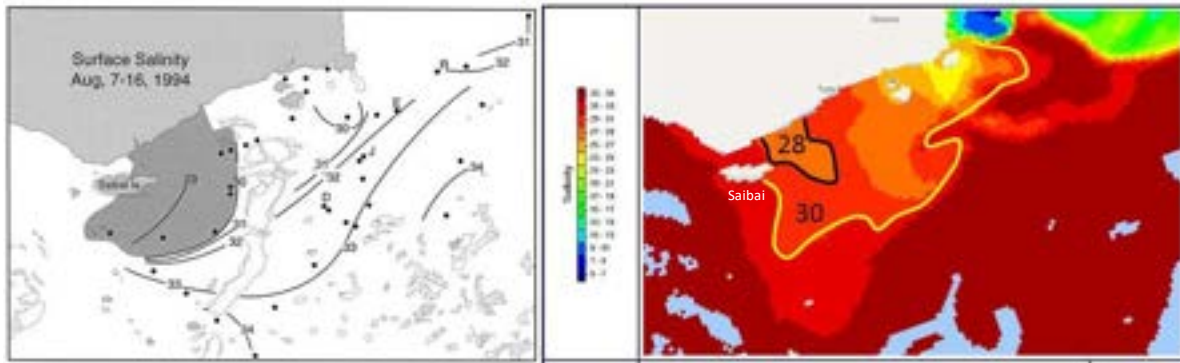


Figure 4-5: Comparison between the observed and predicted surface salinity distribution in the Torres Strait in early August 1994.

In summary, the MOHID model appeared reliable to investigate the mechanisms controlling river plume movements in the Gulf of Papua and the conditions favouring an intrusion of the plume into the Torres Strait. It suggests that the plume of potentially polluted river water is probably mostly of concern during the SE trade wind season. During that season the plume is permanent, although variable in time following fluctuations of the wind and Δ MSL across the Torres Strait. The MOHID-predicted plume appears limited to the northern third of the Torres Strait, and has lower salinity in the east than in the west.

To investigate further details of the Fly River plume intrusion in the Torres Strait, the SLIM model was nested in the MOHID model and used to refine the MOHID model predictions. The SLIM model can do that because its non-structured mesh allows for a high spatial resolution near the coast, islands and reefs (Figure 4-6). To predict the hydrodynamics, the open boundary forcing of the SLIM Torres Strait model of Wolanski et al. (2013) was modified to incorporate the new findings described above regarding the wind and Δ MSL across the region, in the same manner that these were included in the MOHID model.

To predict the river plume intrusion, an Eulerian advection-dispersion model was then used, taking from the MOHID model predictions of the open boundary conditions of salinity along the open boundary transect line NE-E (Figure 4-7); thus, nesting the SLIM model into the MOHID model.

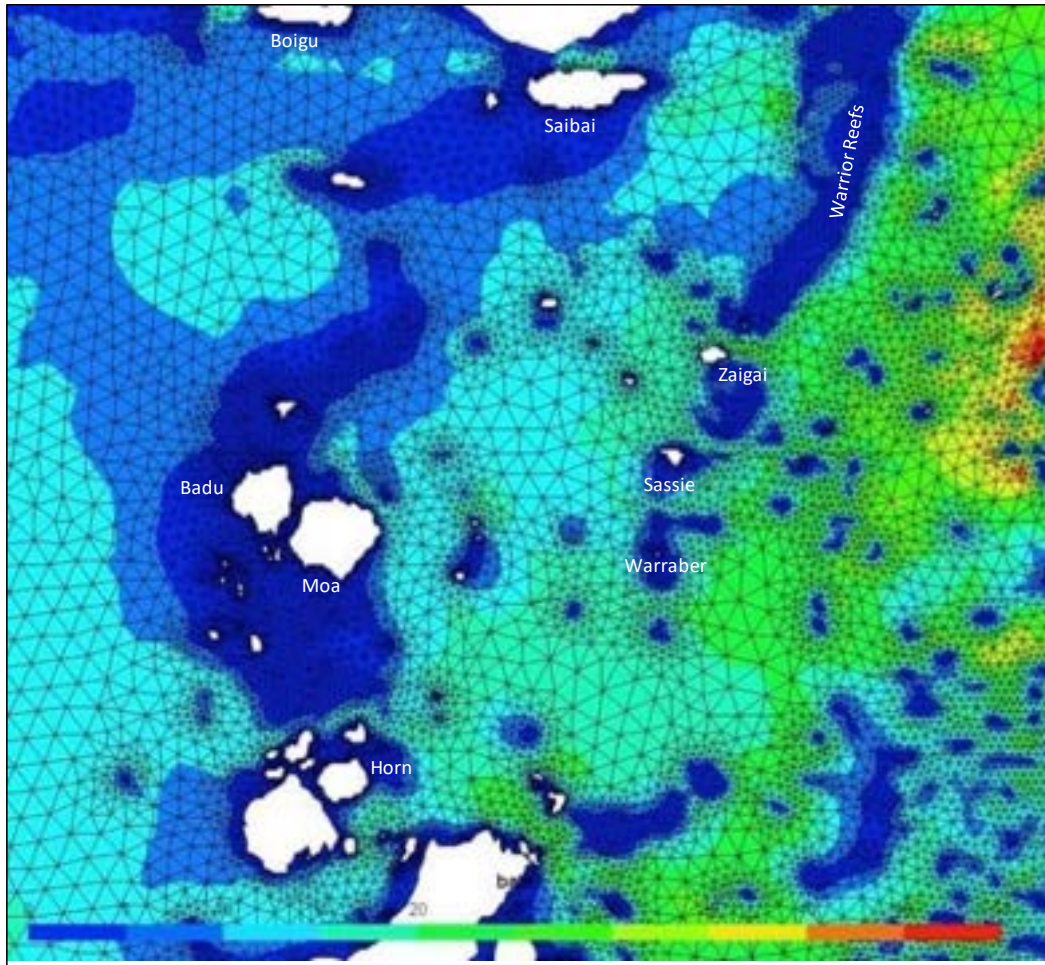


Figure 4-6: An example of the variable mesh of the SLIM model. The colour bar shows the depth in metres.

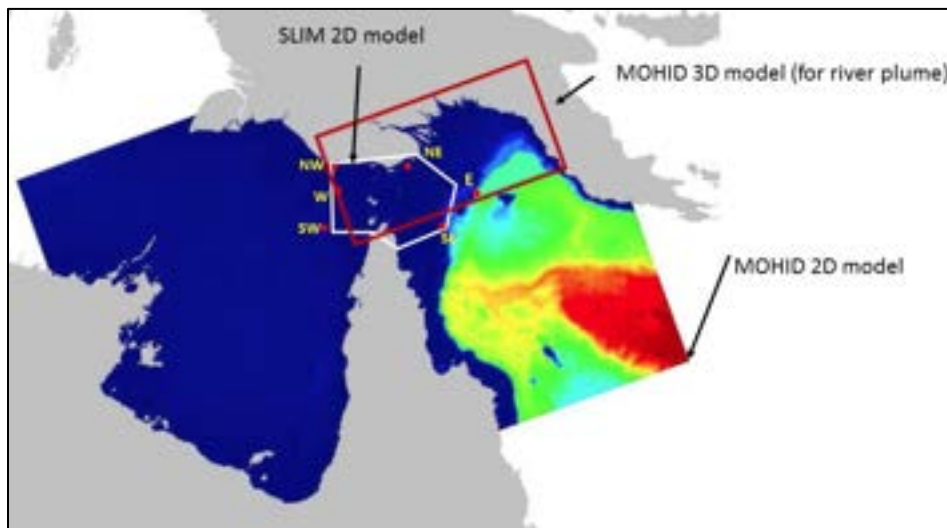


Figure 4-7: The open boundaries of the MOHID and SLIM models.

Examples of the SLIM-predicted river plume intrusion in the Torres Strait are shown in Figure 4-8 as a function of time, for scenarios during the SE trade wind season. The SLIM model predictions agree with those of the MOHID model as follows: (1) the plume intrusion in the

Torres Strait is of concern mostly during the SE trade wind season; (2) during that season the plume is permanently present in the Torres Strait; (3) plumes fluctuate in time and extent following fluctuations of the wind and Δ MSL across the Torres Strait; and (4) plumes are limited to the northern third of the Torres Strait, with lower salinity in the east than in the west.

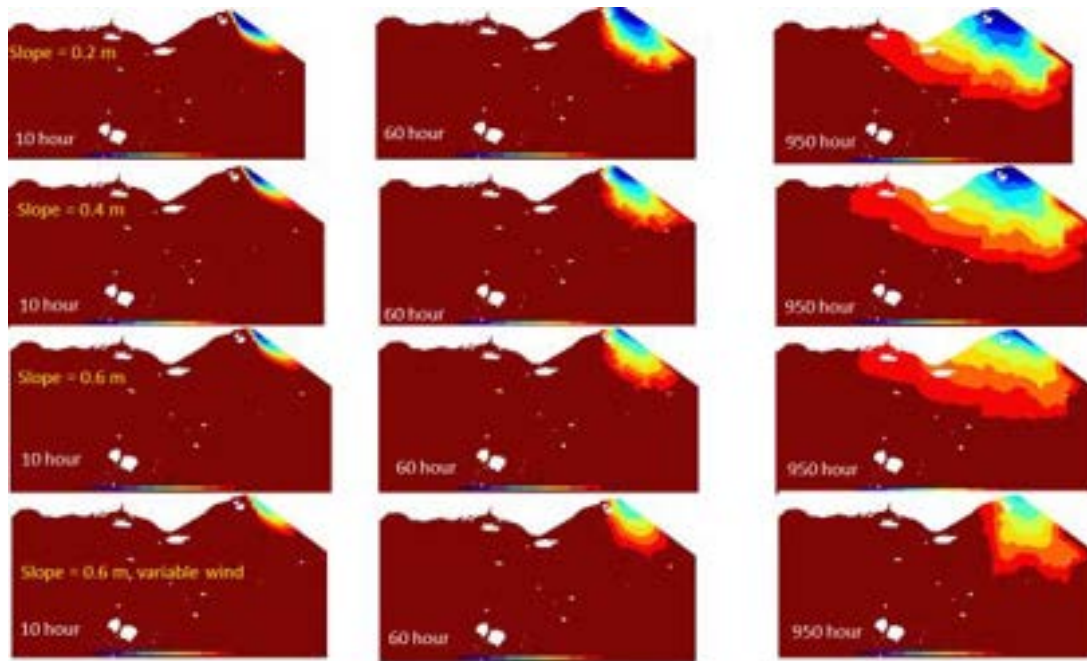


Figure 4-8: Predictions of the salinity plume at (left to right) 10, 60 and 950 hours during the SE trade wind season for (from the top downward) Δ MSL of 0.2, 0.4 and 0.6 metres and (bottom) for a variable Δ MSL as measured during September–October 2013 (when the wind often reversed direction).

There is one major difference, however, between the MOHID and SLIM model predictions of the salinity plume intrusion in the Torres Strait, and that is in the western region. Comparing Figure 4-4 and Figure 4-8, MOHID predicts a stronger intrusion in the western Torres Strait than SLIM. We believe that this discrepancy is due to the different grid resolution and different bathymetry data used by the models (Figure 4-9). The lower resolution in MOHID does not allow incorporation of the “sticky water” effect of small islands and shoals, resulting in a higher permeability across the east-west direction with a consequent spreading of the plume to the west. That effect is not present in the observations or in the SLIM results. Also, MOHID used the bathymetry from the British Oceanographic Data Centre whereas SLIM used bathymetry initially based on the GeoScience Australia dataset. Those data were missing six islands in the Torres Strait (including some inhabited islands), as well as tens of reefs on the GBR continental shelf. These missing islands were included in the bathymetry by using the digitised global map of islands and the missing reefs were included in the bathymetry by using the bathymetry data ‘gbr100’ (project led by Dr Robin Beaman from JCU in partnership with AIMS., Geoscience Australia, CSIRO and the Queensland Department of Agriculture and Fisheries; <https://www.deeppreef.org/projects/48-depth-model-gbr.html>).

The difference between the two bathymetric maps is small on the east side of the Torres Strait, i.e. the entry point for the Fly River plume intrusion—hence the MOHID and SLIM models agree quite well on the eastern side. The difference is quite large north of Badu Island. There, the British bathymetry data show deep waters (hence allowing the river plume to readily travel through) while the GeoScience Australia data show shallow waters (thus inhibiting the plume

from travelling further west). In any case, the salinity is much higher in the east side of the Torres Strait than in the west side; hence, the discrepancy between the SLIM and MOHID models in the western side of Torres Strait should not be of concern.

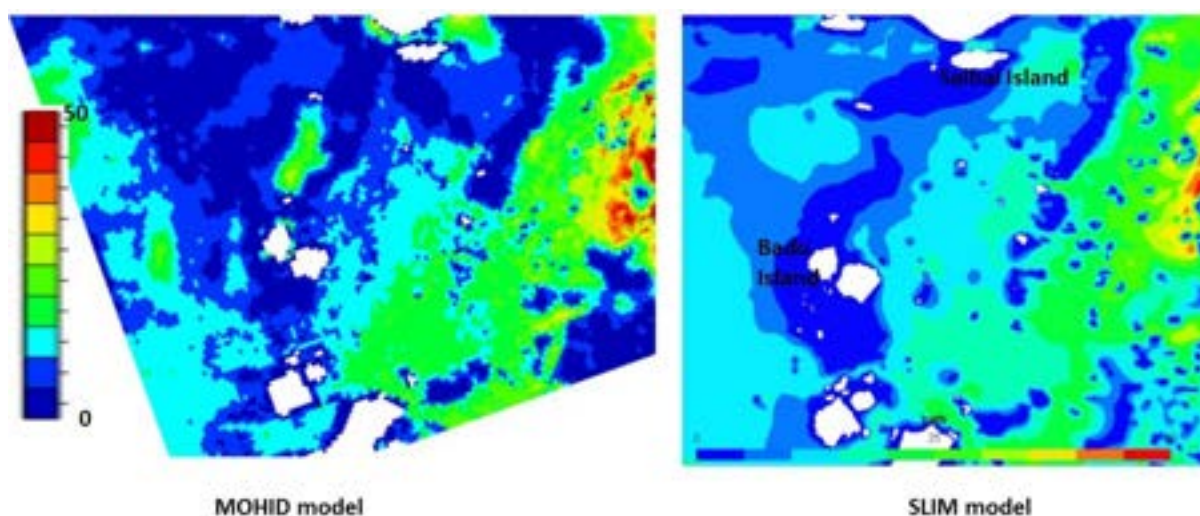


Figure 4-9: Focusing on the area between Badu and Saibai Islands, the bathymetry used by (left) the MOHID model and (right) the SLIM model.

4.2.2 Recommendations

This study demonstrated the common occurrence of the Fly River plume in the northern region of the Torres Strait, and particularly in the NE corner. That area is data poor and the project team encourages the TSRA to continue a weekly salinity monitoring programme around the islands in the Torres Strait, for which handheld salinity probes and training were provided to rangers in December 2016 (see Section 4.5), so that the river plume could be tracked and its potential threat evaluated in real time.

This study was limited to understanding the intrusion of the Fly River plume in the Torres Strait. It did not address modelling another potential and poorly understood threat—the intrusion of Fly River fine sediment and particle-bound contaminants. The ship-borne data to model that are insufficient; also, turbidity data derived from satellite images can be limited in application for this specific purpose because the bottom sediment is eroded and resuspended by tides and waves, which makes it difficult to distinguish turbidity that is driven by the input of Fly River plumes and local resuspended sediment (e.g. driven by wind and tides).

It is recommended that further effort be undertaken to understand the intrusion of Fly River fine sediment into the Torres Strait and its particle-bound contaminants via further targeted marine monitoring and sediment modelling with a focus on the northern Torres Strait.

4.3 Remote sensing analysis

Purpose: Acquisition and analysis of daily true-colour satellite imagery to identify instances of likely Fly River plume intrusion, linking to coincident oceanographic and meteorological conditions to refine modelling. **The detailed results of this component are reported in Petus et al. (2018).**

Highlights:

- This study used medium resolution satellite data to: (i) provide a large-scale baseline of the composition of coastal waters around the Gulf of Papua–Torres Strait (GPTS) region, (ii) investigate the spatial and temporal variability of the Fly River turbid plume, and (iii) identify instances and areas with likely plume intrusion into the Torres Strait Protected Zone.
- Nine years of daily MODIS true colour imagery (2008–2016) were processed using a semi-automated colour classification method developed for the GBR. This method classifies the coastal waters into six different colour classes (CCs). It is particularly useful in remote areas where *in-situ* water quality data are scarce or missing, such as the GPTS region, as it relies only on the apparent colour of the ocean.
- The brownish to brownish-green CCs (1–4) represent very turbid, sediment dominated waters with high coloured dissolved organic matter (CDOM) concentrations and are classified as ‘Primary’ waters; the greenish to greenish-blue CC (5) represents turbid waters with dominant to moderate concentrations of Chlorophyll-a (Chl-a, phytoplankton algae) and finer sediments and are classified as ‘Secondary’ waters; and the blueish-green CC (6) represents waters with slightly above ambient turbidity and Chl-a concentrations and are classified as ‘Tertiary’ waters.

Long-term water quality conditions in the study area

- In the estuarine area of the Fly River, brownish-green (Primary) waters were mapped inside the Fly River delta (< 5 m depth) with the most brownish, turbid waters located inside the very shallow delta plains. This suggested that a large proportion of sediment is generally retained in the Fly River estuary through flocculation, in agreement with previous studies.
- This study showed the regular intrusion of greenish-blue (Secondary) waters from the Gulf of Papua in the NE corner of the Torres Strait Protected Zone (including Daru), the north of Warrior Reefs and north and south of Sabai Island. The presence of these turbid waters is likely the result of the complex physico-chemical transformation occurring across the Fly River plume, local resuspension of sediments and the water circulation in this area.
- Brownish-green (Primary) and greenish-blue (Secondary) waters were also mapped around Boigu Island. The presence of these turbid waters is likely the result of the local resuspension of sediments and the water circulation in this area, as well as sediments discharged by the Wassi Kussa and Mai Kussa Rivers and potentially transported from the Fly River.

Annual and seasonal dynamics in the southwest Fly District

- The Southwest Fly District was defined in this study as the area extending from the central Fly River mouth and as far west and south as the northern Warrior Reefs. The spatial extent of the Primary and Secondary waters in this area varied seasonally and larger turbid areas (km²) were mapped during the SE trade wind season.
- The spatial extent was also correlated with ENSO events and smaller areas (km²) were mapped during El Niño years. El Niño creates a large negative perturbation (i.e. low flow) to the relatively constant sediment discharge and limits the transport of sediment from the tributaries to the nearshore zone.

- The results are consistent with the key findings from the hydrodynamic modelling (Section 4.2).
- Additional *in-situ* sampling is required to validate these results and the semi-automated colour classification method.

4.3.1 Overview of methods and results

This study used daily MODIS satellite true colour imagery and a semi-automated colour classification technique developed for the GBR (Álvarez-Romero et al. 2013; Devlin et al. 2015). Nine years of daily CC maps of the GPTS study region, 2008–2016, were produced using MODIS true colour satellite imagery with a spatial resolution of 500 × 500 m. The MODIS true colour data were downloaded from the NASA Earth Observing System Data and Information System (EOSDIS) worldview website (<https://worldview.earthdata.nasa.gov/>), spectrally enhanced and clustered into ‘cloud’, ‘ambient water’ and six distinct CCs through a supervised classification using typical apparent surface colour signatures of flood waters in the GBR (Álvarez-Romero et al. 2013 and Figure 4-10). The determination of CCs was based on the GBR flood plume typology as applied in the Reef Plan Marine Monitoring Program (described most recently in Waterhouse et al. 2018). The classification was calibrated and validated with satellite and *in-situ* water quality data in the GBR, respectively (Álvarez-Romero et al. 2013; Devlin et al. 2013, 2015; Petus et al. 2016).

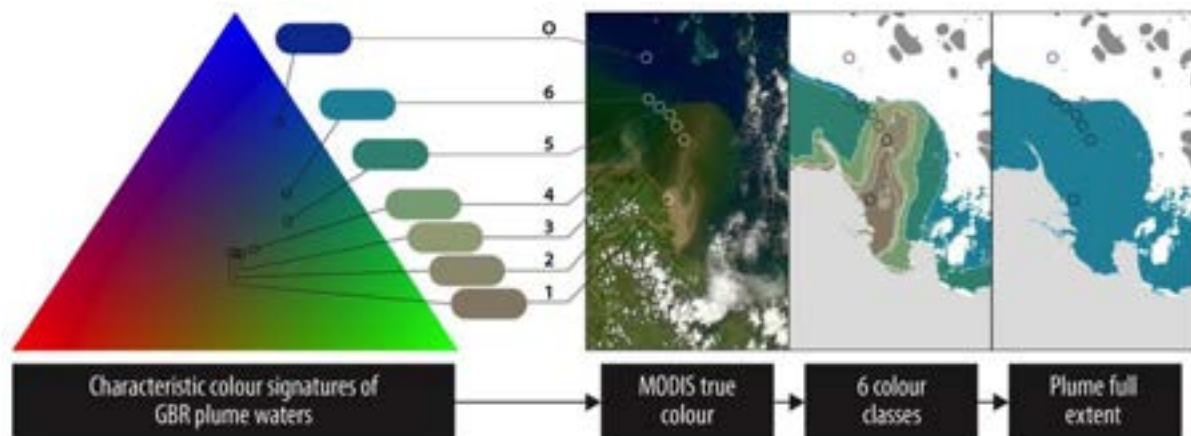


Figure 4-10: Triangular colour plot showing the characteristic colour signatures of the Great Barrier Reef river plume waters in the Red-Green-Blue (or true colour) space. Álvarez-Romero et al. (2013) developed a method to map these characteristic coastal water masses in the GBR using a supervised classification of MODIS true colour data (modified from Devlin et al. 2015).

In the GBR, the brownish to brownish-green CCs (1–4) represent turbid, sediment dominated waters with high CDOM and Chl-a concentrations and are classified as ‘Primary’ waters. They are typically found in inshore regions of GBR river plumes or nearshore marine areas with high concentrations of resuspended sediments during the wet season (Figure 4-10). The greenish to greenish-blue CC represent turbid waters with dominant to moderate concentrations of Chl-a (algae) and finer sediments. This CC is found in the GBR open coastal waters as well as in the mid-region of river plumes and are classified as ‘Secondary’ waters. The blueish-green CC (6) represent waters with slightly above ambient turbidity and Chl-a concentrations and are classified as ‘Tertiary’ waters. In the GBR, this colouration is typical for areas towards the open sea or offshore regions of river flood plumes. Due to a lack of ground-truthing information at

the time of this study, the same water quality characteristics were assumed for both the GBR and the CCs in the study area. This assumption is thought to be valid for the present study as the apparent colour of the ocean is controlled by the concentrations and mutual proportions of optically active water quality components (TSS, Chl-a and CDOM) in the water.

Despite the lack of *in-situ* water quality data, analyses of the MODIS CC maps allowed description of the large-scale and long-term trends in water quality conditions and sediment distribution in the study area. The imagery revealed large-scale general patterns along and across the PNG shoreline, summarised in Figure 4-11, and allowed characterisation of seasonal and inter-annual variations in long-term water quality conditions in the area defined as the 'Southwest Fly District', which extends from the central Fly River mouth and as far west and south as the northern Warrior Reefs (highlighted in Figure 4-11).

Long-term water quality conditions in the study area

The assessment area for this section is shown by the full extent of the map in Figure 4-11. Coastal waters along the southwest PNG coast were influenced by local river discharges entering the Gulf of Papua and were characterised by brownish to brownish-green coloured waters (Primary waters, CCs 1–4).



Figure 4-11: Summary figure illustrating the mean long-term (2008–2016) and large-scale patterns in surface water colour in the Torres Strait and Gulf of Papua regions (the study area). The boundary of the Southwest Fly District is outlined in red. CC1 to CC6 show the different colour classes.

In the estuarine area of the Fly River, CC1 to 4 (Primary waters) were mapped inside the Fly River delta (< 5 m depth), and the brownest waters (CC1 and 2) were mapped inside the shallow delta plains (the upper areas of the delta). Sediment concentrations at the mouth of the Fly River have been estimated to be up to 70 mg L⁻¹ (Robertson et al. 1993; Ayukai and

Wolanski 1997) and rapidly reduced to a few milligrams per litre in the area where salinity was about 26 ppt (Ayukai and Wolanski 1997). Colour patterns observed in Figure 4-11 seem to confirm that a large proportion of sediment in the Fly River flocculate and are generally retained in the Fly estuary (Wolanski and Gibbs 1995; Wolanski et al. 1995a).

Locations of the surface turbidity fronts inside the Fly River delta were variable, as illustrated by the high standard deviation observed on the delta front (Figure 4-11, orange lines within the Gulf of Papua). It has been shown that the coupling of river discharge and tidal processes influence the location of estuarine processes and control the sediment flux from the Fly River delta to the marine environment (Ogston et al. 2008). The temporal resolution of the MODIS data (daily) did not allow description of sediment processes for short-term (semi-diurnal) tidal scale; however, it did provide observations over seasonal and inter-annual time-frames.

Greenish to greenish-blue water masses (CC5 or Secondary waters) were mapped in the Torres Strait and covered a coastal band toward the open seas extending from the western Torres Strait to the northeastern Gulf of Papua (Figure 4-11). Blueish-green waters (CC6 or Tertiary waters) were mapped further offshore at the transition between coastal and marine ambient conditions (pure blue waters). In the Gulf of Papua, the greatest variability in long-term average water quality conditions was observed at the boundaries of CC5 and 6 and the marine waters (Figure 4-11, orange lines).

A frequent intrusion of greenish-blue waters (CC5) from the Gulf of Papua east was mapped in the NE corner of the Torres Strait Protected Zone (including Daru), the north of Warrior Reefs and north and south of Saibai Island. This CC likely indicates the presence of turbid waters, with fine sediment concentrations and moderate to high Chl-a concentrations. It is likely to be the result of the complex water circulation in this area, local resuspension of sediment and the influence of the Fly River plume. Brownish-green (Primary) and Greenish-blue (Secondary) waters were also mapped around Boigu Island. The presence of these turbid waters is likely the result of the local resuspension of sediments and the water circulation in this area, as well as sediments discharged by the Wassi Kussa and Mai Kussa Rivers and potentially transported from the Fly River.

Further analysis below focuses on the area shown as the southwest Fly District and the maximum risk area, which includes Daru and the northern Warrior Reefs, as well as Bramble Cay (Figure 4-11, red box).

Annual and seasonal dynamics in the Southwest Fly District

The spatial extent of the six CCs in the Southwest Fly District varied seasonally, with exposure to larger areas of coloured and brownish turbid waters from the Gulf of Papua and Fly River during the trade wind season than during the northwest (NW) monsoon (Figure 4-12).

In-situ sediment core data results have suggested possible seasonal sediment storage in the inner shelf of the Fly River delta (15 m water depth) during the NW monsoon winds with out of phase storm-wave transport to deeper water depths during SE trade wind conditions (Walsh et al. 2004). The larger (plume) areas mapped during the trade wind season could reflect the resuspension and transport of sediment to deeper water depths due to greater wave stress at this time of year.

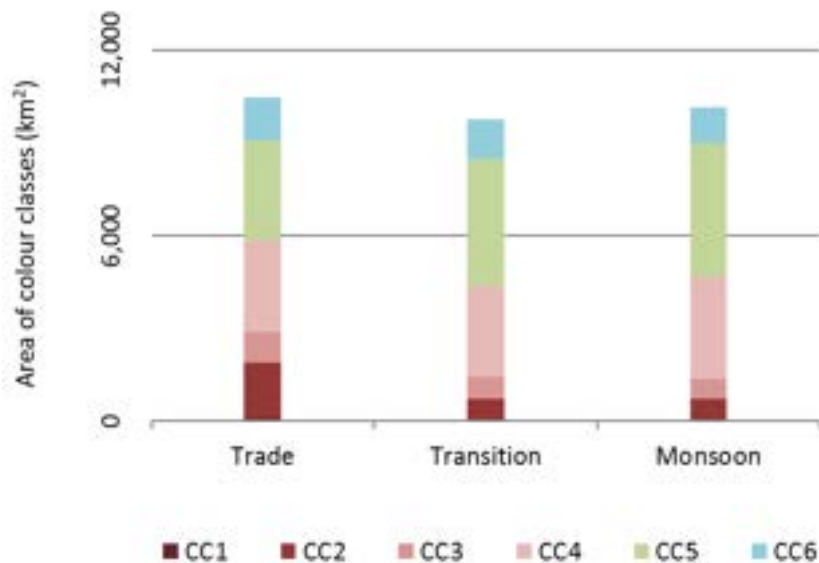


Figure 4-12: Mean long-term seasonal areas of the Southwest Fly District (km²) exposed to the six distinct colour classes (CC1–6).

A frequency analysis of the occurrence of clouds in the Torres Strait region was undertaken in the water quality exposure assessment (Section 4.6). It showed that the region is covered, on average (decadal), more than 50% of the year by clouds and that cloud frequency is, on average, greater or similar in July–August (within trade wind season) than in February–March (within the monsoon season) in a selection of sites across Torres Strait (Section 4.6, Figure 4-27d and Table 4-4, and Appendix 2A). More analysis is needed to confirm these seasonal trends and characterise how much cloud presence may alter the results presented in this study. However, the spatial and temporal trends in water composition described in this study were consistent with the key findings from the hydrodynamic modelling, indicating that this influence is not significant enough to alter the primary data interpretation.

El Niño conditions create a large negative perturbation (i.e. low flow) to the relatively constant sediment discharge and limit the transport of sediment from the Fly River tributaries to the nearshore zone of sediment temporary storage (Ogston et al. 2008). These conditions were experienced within the study period during 2014–2016 (Figure 4-13). The effect was measured on the maps by a reduction of the total area of the Southwest Fly District exposed to brownish to brownish-green (CC1–4 or Primary waters) and coloured waters (CC1–6) during El Niño years (measured as warmer (positive) Niño34 SST). This result may indicate an impact of ENSO on the transport of sediment by the Fly River plume to the Torres Strait region. However more analysis is needed to confirm the impact of ENSO on local sediment transport, and whether sediment delivery is still strongly influenced by increased inputs from mining operations in the Fly River catchment during these years.

Finally, several daily MODIS true colour and water type maps showed the CC6 water masses reaching Bramble Cay (in agreement with Section 4.1, Coral Core results). However, the frequency of occurrence and instances of likely plume exposure at Bramble Cay has not been resolved in this study.

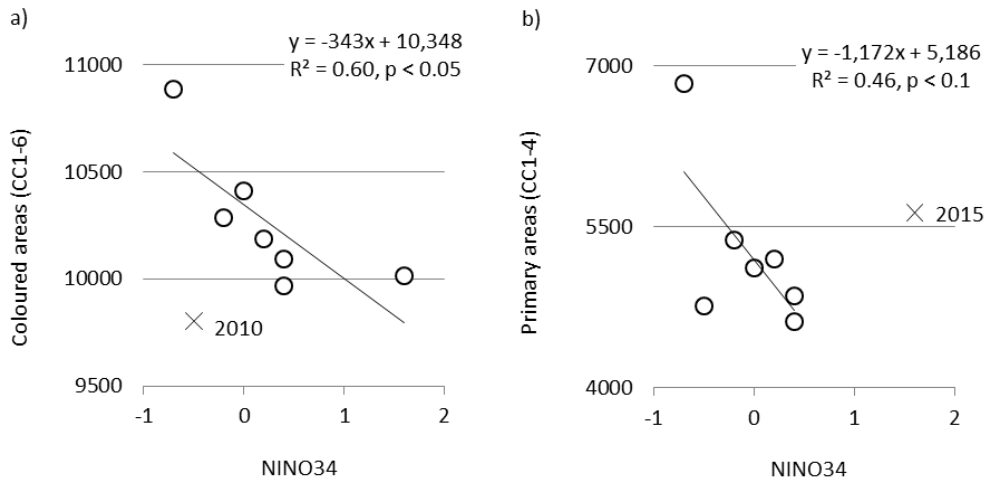


Figure 4-13: Relationships between NINO3.4⁴ and areas of the Southwest Fly District (including the highest risk area) exposed to (a) all coloured waters (CC1–6) and (b) only Primary waters (in km²). Outliers are symbolised with a cross symbol. Relationship statistics are calculated without considering the outliers.

4.3.2 Recommendations

The colour of the ocean derived from satellite imagery provides simple water quality indicators for inclusion into water quality monitoring frameworks. The results obtained in this study reiterated the potential of using satellite data and simple colour scale classification methods for the mapping of turbid waters and river plumes in nearshore marine environments of the Gulf of Papua and Torres Strait, and worldwide. The large-scale patterns defined in this study will need to be refined and validated through ongoing monitoring, mapping and assessment and additional work is required to resolve the primary drivers of elevated turbidity in the study area. It is also important to note that this study described patterns in water turbidity but did not evaluate patterns and threats linked to particle-bound or dissolved land-sourced contaminants. Future work should focus on collecting *in-situ* water quality measurements across the six CCs to ground-truth and validate the preliminary results described in this study. A preliminary validation exercise has been conducted using water quality data collected in October 2016 and showed promising but inconclusive results (Petus et al. 2018). The project team recommends that more fieldwork should be conducted, especially around the Southwest Fly District and Boigu Island to further characterise the water quality concentrations across colour gradients in the study area. These analyses should also be extended to characterise the potential threat and spatial influence of other contaminant sources, such as the Wassi Kussa and Mai Kussa Rivers (southern PNG coast). Further investigation of the influence of the Fly River plume around Bramble Cay is also recommended.

The MODIS satellites have been in orbit for more than 15 years and there is a substantial risk that the satellites may be decommissioned in the coming years. If the remote sensing component is to be continued in the future, we recommend testing the feasibility of using the

⁴ In this study, we used the annual NINO3.4 sea surface temperature anomaly (2008–17, hereafter NINO3.4) obtained from NOAA as a measure of the strength of ENSO events (NOAA 2013). This index is defined as the average of SST anomalies over the region 5°N to 5°S and 170° W to 120°W and has been recently used to study ENSO-related rainfall changes over the New Guinea region (Smith et al. 2013).

new Sentinel-3 Ocean Land Colour Instrument imagery and the Forel-Ule scale colour classification toolbox developed through the European project 'Citclops' for the monitoring of coastal waters and river plumes in the Gulf of Papua and Torres Strait. This toolbox includes a smartphone application (<http://www.eyeonwater.org>) that could be used in future field campaigns to record the colour of the coastal waters in the study area independently of the cloud cover, which can be a major limitation at different times of the year (see Appendix 2A).

This study has focused on describing mean and median water quality patterns at seasonal, annual and multi-annual scales. Daily MODIS CC maps have been processed since 2008 and this database is regularly updated. It is recommended that the daily satellite dataset (and/or Sentinel-3 CC maps) be used in conjunction with the salinity monitoring programme currently undertaken by local rangers around the Torres Strait islands to assess patterns and influences at a smaller site-specific scale. Examples of this are provided in Section 4.5. This will provide a synoptic water quality context to the salinity measurements and track river plumes from the Fly River and other local rivers in near real time.

Finally, satellite CC maps can be used in the future to validate plume and risk scenarios from the modelling component. The maps are used in the water quality exposure assessment (Section 4.6) to assess the frequency of exposure to the CCs, and periods of time over the last 10 years when Torres Strait seagrasses and corals were exposed to higher turbidity levels. The periods of elevated turbidity could be linked to the Fly River discharge and/or processes of resuspension. Further sediment modelling will assist in making this distinction. This type of desktop analysis linking remote sensing data and ecosystem data should be continued in the future and should also include more extensive field measurements.

4.4 *In-situ* continuous loggers

Purpose: Maintain real-time weather (wind, precipitation, humidity, barometric pressure, air temperature), water temperature, salinity and turbidity measurements from monitoring stations at Masig Island and Bramble Cay to assess local patterns. This data can also be used to support model parameterisation and validation.

Highlights:

- Bramble Cay had higher than expected levels of turbidity (average of 10.7 NTU) for a remote offshore reef (as compared to offshore reefs in the GBR that average 0–3 NTU and near-shore reefs that average 3-5 NTU in the dry season and a maximum of 10-15 NTU in the wet season) reflecting its location in a shallow system located directly south of the Fly River.
- The observed patterns of turbidity at Bramble Cay were most strongly correlated with wind speed and most likely represent local wind driven resuspension of *in-situ* material and not episodic movement of material from the north. The exception was a peak observed in June 2017 that seemed to be unrelated to local environmental conditions.
- The presence of fine terrestrial material at Bramble Cay, which seems to make up much of the resuspension, indicated that there is transport of material from PNG as the nearest source. However, this may be via more complex long-term transport mechanisms rather than short-term episodic events, which in turn will have implications for the potential transport of contaminants into the area.

- Logger data and weather stations recorded anomalously high sea temperatures over the 2015–2016 summer related to a period of high air temperatures and low wind speed. The station at Thursday Island recorded a new record maximum above the empirical bleaching threshold and subsequently coral bleaching was observed in the region.
- The logger data confirmed previous data and modelling work that showed that the eastern part of the Torres Strait is cooled by off-shelf upwelling making this part of the Torres Strait more resilient to coral bleaching, as seen during the 2015–2016 and 2016–2017 bleaching events where the most severe bleaching was in the southern and western reefs.

4.4.1 Overview of methods and results

Weather stations and temperature loggers

Temperature loggers were deployed at 15 stations across the Torres Strait as a follow on from previous work conducted under NERP funding (Bainbridge et al. 2015) (Error! Reference source not found.). The loggers are small self-contained temperature recorders that are deployed on reefs located off the main islands in the region. Figure 4-15 shows a logger deployed using a cement block to weigh it down and a small float to aid recovery. Loggers were deployed in January 2016 and recovered in December 2017. Weather stations (see Figure 4-16) were installed and maintained at Thursday Island, Masig Island and Bramble Cay (Bainbridge et al. 2015).



Figure 4-14: Location of the Temperature Loggers deployed in the Torres Strait (map Google Earth).



Figure 4-15: Temperature logger deployed on a small concrete block with a float for visibility.



Figure 4-16: Land based component of the Bramble Cay weather station.

The 2015–2016 summer saw widespread coral bleaching along the GBR (Hughes et al. 2017) with ocean temperatures exceeding empirical bleaching thresholds at a number of sites (Bainbridge 2017). In the Torres Strait, the main warm areas were in the south and southwest; a plot of data from Thursday Island (southern), Masig Island (central) and Bramble Cay (northern) (Figure 4-17) shows that only Thursday Island exceeded the empirical bleaching threshold and this was reflected in the observed patterns of coral bleaching that were mostly in the south and southwest of the Torres Strait (Hughes et al. 2018a).

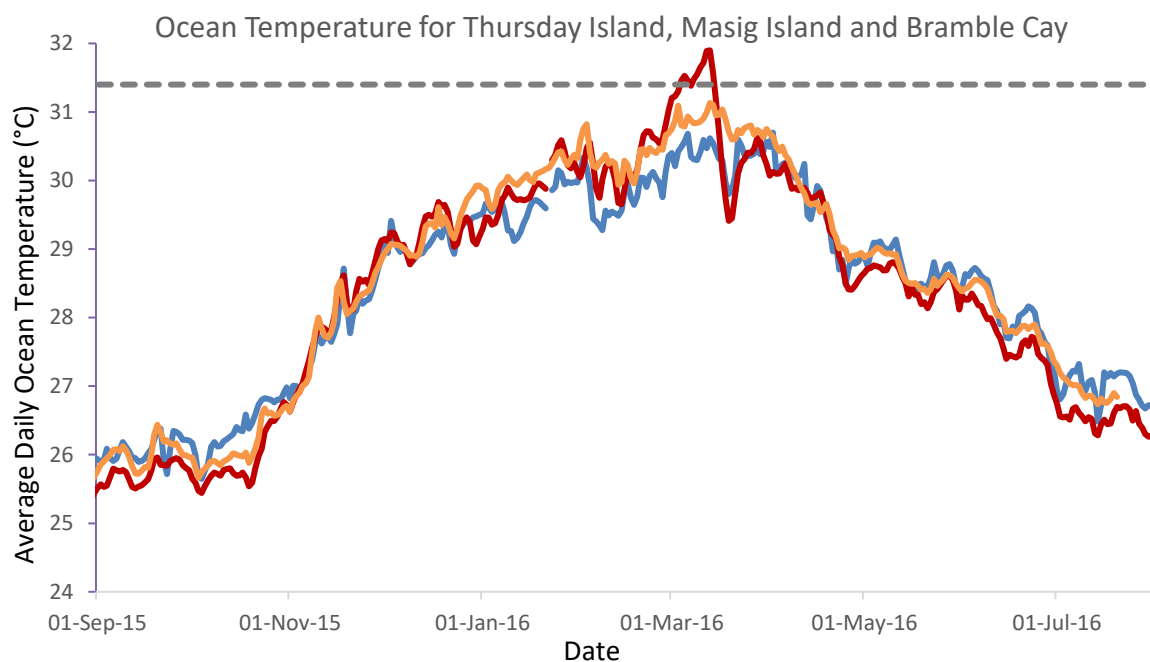


Figure 4-17: Average daily ocean temperatures at Thursday Island (red line), Masig Island (orange line) and Bramble Cay (blue line) showing ocean temperatures during the 2015–2016 coral bleaching event, the dashed grey line is the regional empirical bleaching threshold.

The analysis of the Thursday Island data (Figure 4-18) shows that the bleaching occurred during a two-week period of rapid ocean heating caused by high air temperatures and low wind speeds (effectively doldrums conditions). A change in meteorological conditions at the end of this period resulted in increased wind speeds, which cooled the ocean temperatures but not before widespread coral bleaching had occurred (Hughes et al. 2018a).

The logger data confirmed previous data that shows the presence of a cooler area in the eastern Torres Strait around Erub and Mer islands (Bainbridge et al. 2015) that represents an upwelling signature of cooler off-shelf waters being drawn onto the shelf around Mer Island (Benthuyssen et al. 2018). These cooler waters provide a degree of protection for these reefs to coral bleaching stress and, along with the southern GBR reefs, may represent an area that has a degree of protection from coral bleaching events.

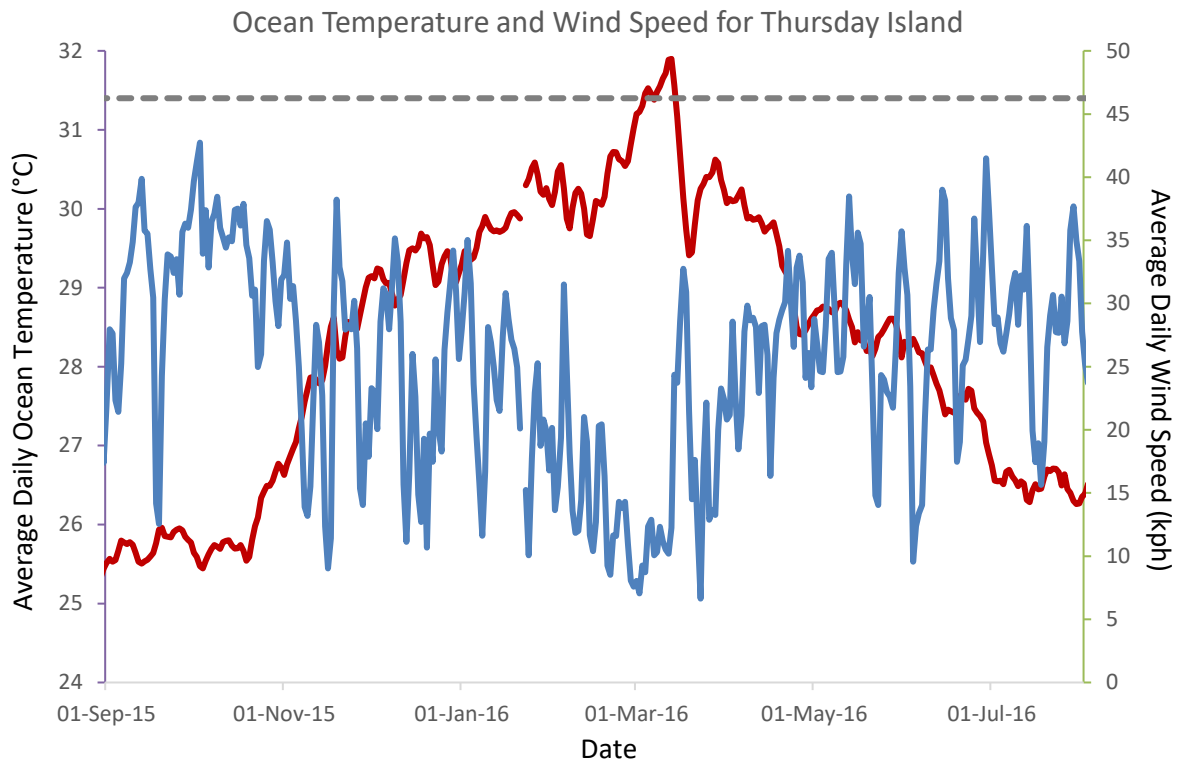


Figure 4-18: Relationship between wind speed (blue line) and average daily ocean temperature (red line) at Thursday Island for the 2015–2016 coral bleaching event.

Salinity loggers

Salinity loggers (SeaBird SBE-37 instruments attached to Campbell Scientific CR1000 loggers) were deployed at Masig Island to record environmental salinity levels. However, the salinity loggers failed with only a small amount of data being collected. The data that was collected (May 2018) showed that salinity levels for the 2017 winter periods were stable around oceanic levels (34.5–35 PSU) with no obvious strong signal from freshwater run-off outside the immediate area.

Turbidity loggers

Turbidity loggers (Wet-Labs NTUS loggers) were located at Bramble Cay to detect potential turbidity from run-off material coming from PNG. These were co-located with the weather station (Figure 4-16) to allow for the collection of meteorological and in-water data (Bainbridge et al. 2015). A single logger was deployed on the eastern side of the cay approximately 50m from the shore in shallow water (around 2m at HAT) on a star-picket so to be 50cm off the bottom. Note that all work was done on snorkel off the cay which restricted where loggers could be placed (depth and distance from the cay) and limited any ability to do turbidity profiles or separately measure bottom turbidity, which may reflect local re-suspension, and surface turbidity that may reflect transported material.

Daily average turbidity was 10.7 NTU while the maximum value recorded was close to 50 NTU with sustained values above 20 NTU. These values are in line with those reported by Hemer et al. (2004) who recorded values of suspended sediment concentration (SSC, where approximately 1 mg/L is equivalent to 1 NTU) of 9.14 mg/L in the NW monsoon, a mean SSC of 5.4 mg/L during the SE trades and up to 20–30 mg/L during spring tides.

These values are higher than off-shore reefs along the GBR that are typically around 0–3 NTU with average dry-season levels often around 1 NTU (Fabricius et al. 2013). For inshore GBR reefs, values are typically below 10 NTU (Schaffelke et al, 2009). For example, values for Russell Island, which sits only 15km off shore in the wet tropics, are below 5 NTU in the winter rising to 10-15 in the summer or wet season (see green line, Figure 4.19 extracted from Figure 2.4 of Schaffelke et al 2009) and for Geoffrey Bay, located just off Townsville, values are again typically below 10 NTU (AIMS un-published data).

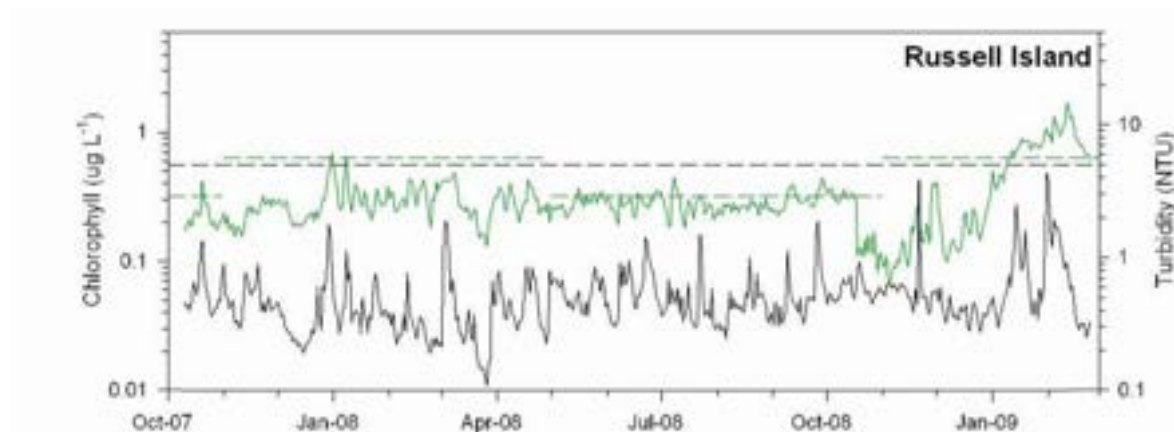


Figure 4-19: Turbidity (green line) measured at Russell Island off Babinda, North Queensland. Extracted from Figure 2.4 in Schaffelke et al. (2009).

This highlights that Bramble Cay has a turbidity profile atypical of GBR reefs, where reefs distant to the coast have relatively low turbidity levels and even those close to the coast near large seasonal rivers have maximum values that are at or below the average values seen at Bramble Cay and confirmed from other studies (Hemer et al. 2004; Harris 2001). While the Torres Strait is a different system to the GBR, for example much of the GBR contains granite based catchments that produce differing forms of sediment, river flows tend to be seasonal and the oceanography is very different, Bramble Cay sees turbidity levels that are higher than what is seen in the GBR.

The turbidity values indicate that there is the potential for Bramble Cay to be influenced by other sources of material, such as from PNG, especially as the area around Bramble Cay is relatively shallow (typically 17–20 m deep versus 30–40 m for the GBR lagoon; Li et al. 2017) and so benthic sediments are more available to be re-suspended back into surface waters.

There are three possible sources of material that result in local turbidity. The first is re-suspension of existing *in-situ* fine material normally driven by localised water movement caused by wind and tides. The second is from localised run-off from the cay that would normally be linked to rain events that may wash cay material into the ocean. The third, and the one that the equipment was deployed to detect, is from remote river runoff, in this case from riverine material from PNG coming down from the Fly River as episodic weather or current driven events.

To separate these three, which can co-occur, the turbidity was compared with the wind and rainfall data to see whether the turbidity patterns matched. Figure 4-20 shows the turbidity data

plotted against the wind speed data, with the data showing a good correlation with wind speed ($r = 0.52$, $n = 1,886$ days, $p < 0.01$) with the exception of the spike in June 2017.

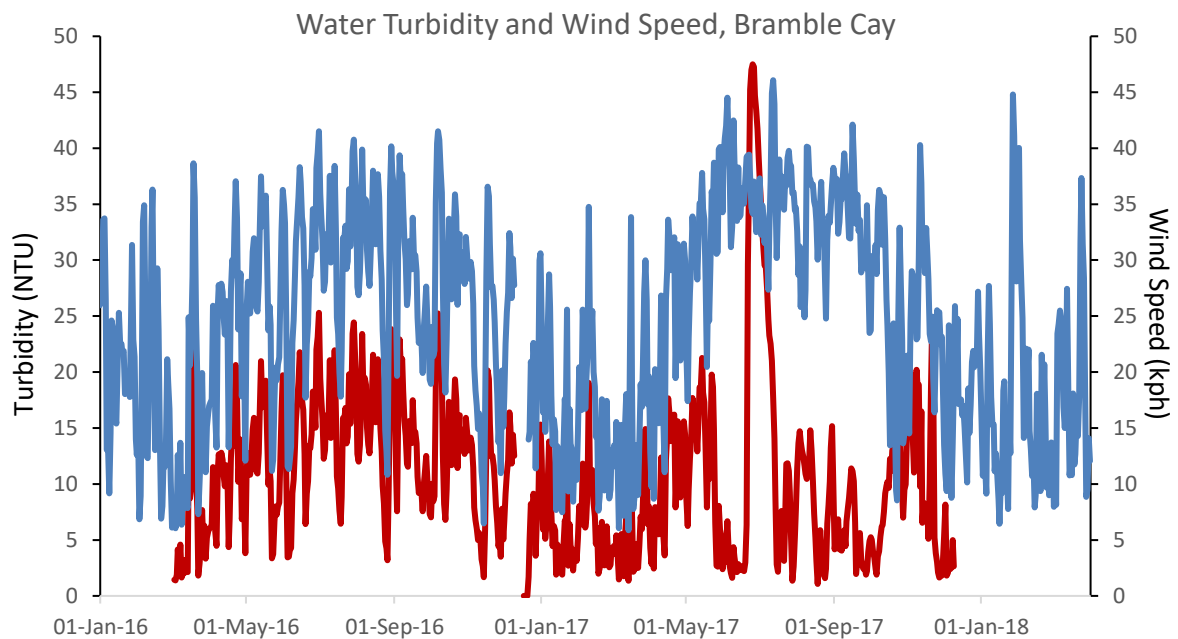


Figure 4-20: Turbidity (NTU, red line) and wind speed (kph, blue line) at Bramble Cay.

The plot of turbidity with rainfall (Figure 4-21) shows no correlation between the two ($r = 0.04$, $n = 1,886$ days), which is not surprising as Bramble Cay is a low sand cay with little terrestrial type material (such as soil) and almost no topography to promote high-energy run-off. As such, the impact of localised run-off from the cay is expected to be minimal and not a major source of local turbidity.

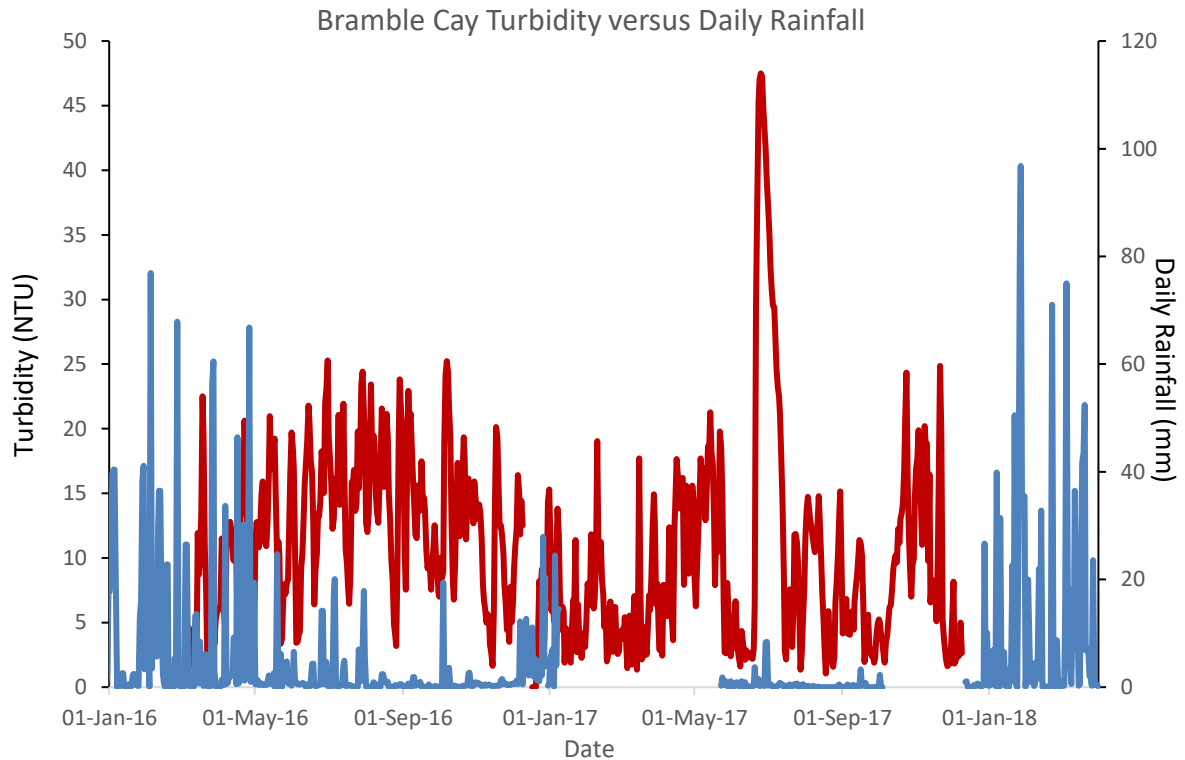


Figure 4-21: Turbidity (NTU, red line) and rainfall (mm, blue line) at Bramble Cay.

To determine whether the observed turbidity is related to remote sediments moving through the Bramble Cay area requires correlation with the satellite images. In particular, the turbidity spike seen around June 2017 cannot be explained by the local environmental data alone, and thus may represent an event outside the immediate area of Bramble Cay.

The most likely source of the material that is causing higher than expected average turbidity is local fine sands/silts that are re-suspended through the action of local winds causing increased wave energy and subsequently bringing fine shallow benthic material back into the water column. The question then becomes what is the source of the fine sediments observed around Bramble Cay?

For most remote reefs the surrounding sand is created from the break-down of coral and other calcareous organisms and, therefore, tends to be coarse calcium carbonate-based sands. These are different to terrestrial-sourced materials that are typically much finer with more organic material such as mud and silts (Larcombe et al. 2001). The fine benthic material seen around Bramble Cay appears to be partially terrestrial in origin as it has a predominance of fine material rather than the expected courser calcium carbonate material from simple *in-situ* sources (although this material has not been formally analysed, work under NESP Project 2.2.2 may provide more information on sediment types). Sediment analysis around Bramble Cay (Harris et al. 2002) was undertaken in the areas to the west of the cay but showed that typical values would be 10-20% mud and 50-60 % calcium carbonate. Figure 4-22 shows the available data overlaid onto a map, the orange box shows an enlarged version which, while the sampling missed Bramble, shows mud levels should be around 10-20% for the cay.

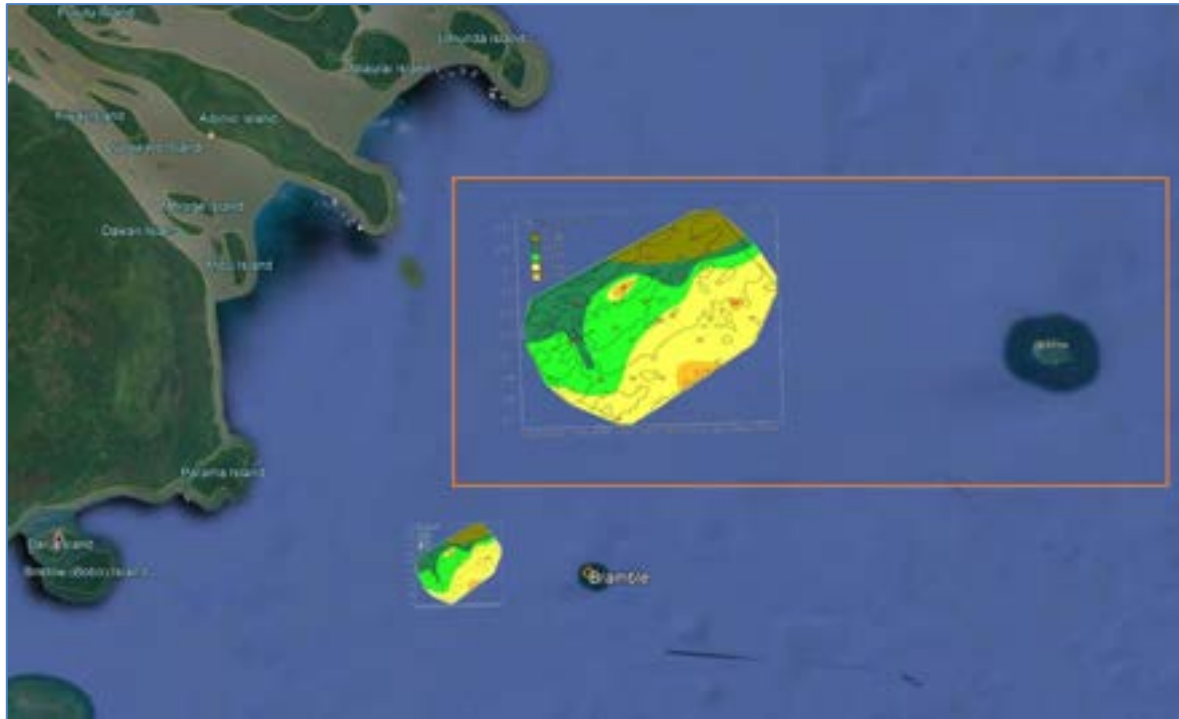


Figure 4-22: Map of available sediment data near Bramble Cay. Orange box shows an enlargement of the main map and shows mud levels around Bramble Cay would be expected to be in the range of 10-20%. Source: Harris et al. (2002).

There is a high probability that the source of the fine material around Bramble Cay is from PNG given that Bramble Cay is 60 km from the mouth of the Fly River but some 200 km from any mainland Australian rivers. In addition, analysis of satellite imagery during this period shows evidence of Fly River discharge entering the region, although this is not confirmed by field observations.

The conclusion is that Bramble Cay is receiving, or has received, terrestrial material from PNG but that this may be due to more complex long-term transport rather than short-term episodic events. This may imply a differing pathway and probability for the transport of potential contaminants from PNG to Bramble Cay. As such, while the long-term source of material is most likely from PNG, the source of short-term episodic turbidity events is from wind driven re-suspension of localised *in-situ* material.

4.4.2 Recommendations

The *in-situ* logger data provides a highly valuable continuous dataset of environmental conditions relevant to the assessment of the Fly River influence in the region. It is recommended that the deployment of salinity, turbidity and temperature loggers be continued at the existing Bramble Cay and Masig Island stations. Installation of instrumentation at additional sites also likely to be exposed to periodic influence from Fly River discharge, including the northern Warrior Reefs, Ugar and Saibai Island, is also recommended.

4.5 Salinity monitoring

Purpose: Provide further real-time data as another line of evidence to assess the potential influence of the Fly River into the Torres Strait (in addition to the remote sensing and hydrodynamic modelling data) and engage the local communities in making this assessment.

Highlights:

- Rangers from the TSRA were trained through the project to undertake a weekly salinity monitoring program using salinity and temperature meters. Data was collected at Boigu, Saibai, Erub, Masig, Iama, Poruma and Warraber Islands from February 2017.
- Results showed that the northern sites were exposed to lower salinity waters and were exposed to greater salinity ranges than those located in the mid and southern region of the Torres Strait.
- The salinity measurements were divided into three zones:
 - i. The islands NW of Masig Island—Saibai and Boigu: mean salinity of 29 and 26 PSU, respectively, with the highest salinity range of 10.7 and 8.4 PSU, respectively; however, only four samples were collected at Boigu Island at the time of reporting;
 - ii. The islands south of Masig Island—Iama, Poruma and Warraber: mean salinity around 35 PSU and salinity range < 5 PSU; and
 - iii. Masig Island: transitional area with a mean salinity of 33.3 PSU and a salinity range of 6.3 PSU.
- The salinity monitoring provided useful *in-situ* data that could ultimately be used to support further hydrodynamic modelling and validate the remote sensing analysis.
- Satellite images can provide a synoptic context to the salinity measurements.

4.5.1 Overview of methods and results

Local rangers from TSRA were trained to undertake a weekly salinity monitoring programme around different islands in the Torres Strait. Salinity data was collected at Boigu, Saibai, Erub, Masig, Iama, Poruma and Warraber Islands from February 2017 (Figure 4-23:). This involved:

- July 2016—Introduction to the use of the handheld salinity and temperature meters to 'validate' the salinity modelling for the dispersion of the Fly River plume into the Torres Strait.
- December 2016—Delivery of the meters, instruction sheets and standards to the rangers. Attempted visits to Warraber, Masig and Erub Islands were unsuccessful due to weather conditions but provided direct training to the Ranger in Charge (Troy Stow) on Thursday Island and all the equipment and training manuals were handed over. Conducted training at Saibai and Boigu Islands.
- Regular (mostly weekly) sampling at Warraber, Masig, Poruma and Iama Islands. Collections at Saibai and Boigu Islands commenced in April 2017, although this was not regular.
- Data was entered directly to the TSRA automated Fulcrum system, with notification of the sample collection to the project team. This enabled all entered data to be downloaded in a collated spreadsheet of the sampling template.

The materials in the training manuals including the methods are provided in Appendix 1.

The rangers were provided with two handheld salinity-temperature meters (AZ Instrument Corp. Water Quality Meter–Salinity meter, Model #8371). The meters were calibrated in the TropWATER JCU laboratory prior to dispatch.

The monitoring sites were located in at least 1 metre of water, and were either located at the end of a jetty or off the beach. The rangers were advised not to locate the sites close to any land based freshwater influence such as a stream or discharge area, and to avoid periods of heavy rainfall. The position of the site was recorded using a GPS.

Once a week, as much as possible on the same day of the week, the rangers collected seawater using a bucket at the selected site. Additional information including the date, time, estimate of the tide and the wind, and the collector's name and contact details was captured on the sample recording sheet, which was eventually entered directly into the TSRA Fulcrum automated data entry system.

The rangers followed the steps below to record salinity:

1. Remove the cap from the meter, turn on the meter using the top On/Off button.
2. Fill a small jar with the salinity standard from the 1 L bottle; dip the probe into the water to the top of the probe and measure the salinity once the reading is stable. Write that value in the sample recording sheet. The water from the small container is thrown away every second time it is used. If unsure, use a new standard.
3. Submerge the probe in the seawater sample in the bucket to the top of the probe. Read the temperature and the salinity result on the meter, and write down these values in the sample recording sheet. Then turn off the meter, rinse it with freshwater, dry it with a tissue and put the cap back on for storage.

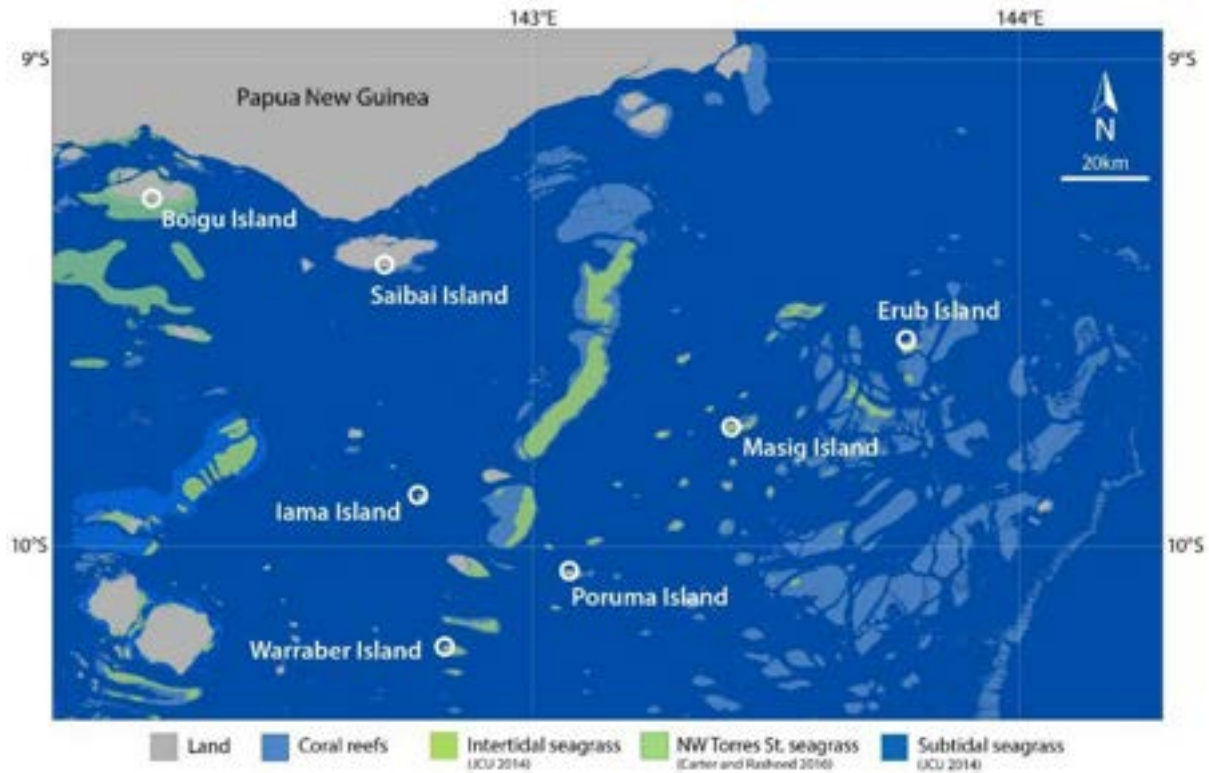


Figure 4-23: Location of islands where weekly salinity measurements have been taken since 2016–2017 by TSRA ranger teams.

A value of corrected salinity was calculated by dividing the expected TropWATER salinity standard value by the local salinity reading of the TropWATER salinity standard (recorded by the rangers prior to sampling) to generate a correction factor. This factor was then applied to the measured salinity ready to calculate the correction salinity value:

$$\text{Corrected salinity} = \text{bucket salinity reading} \times \text{correction factor} \quad \text{Eq. 1}$$

$$\text{with correction factor} = \text{TropWATER Salinity Standard} / \text{Salinity Standard reading} \quad \text{Eq. 2}$$

Outlier measurements were defined as the standard difference (Eq. 3) and values greater than $Q3 + 1.5IQ$ or lower than $Q1 - 1.5IQ$ (with $Q1$ being the first quartile, $Q3$ the third quartile and IQ the interquartile range $Q3 - Q1$):

$$\text{with standard diff} = \text{TropWATER Salinity Standard} - \text{Salinity Standard reading} \quad \text{Eq. 3}$$

Outliers ($N = 7$, Table 4-2) were removed from the salinity database and the time series of the adjusted salinity (hereafter salinity) were plotted (Figure 4-24) and summarised (Table 4-2).

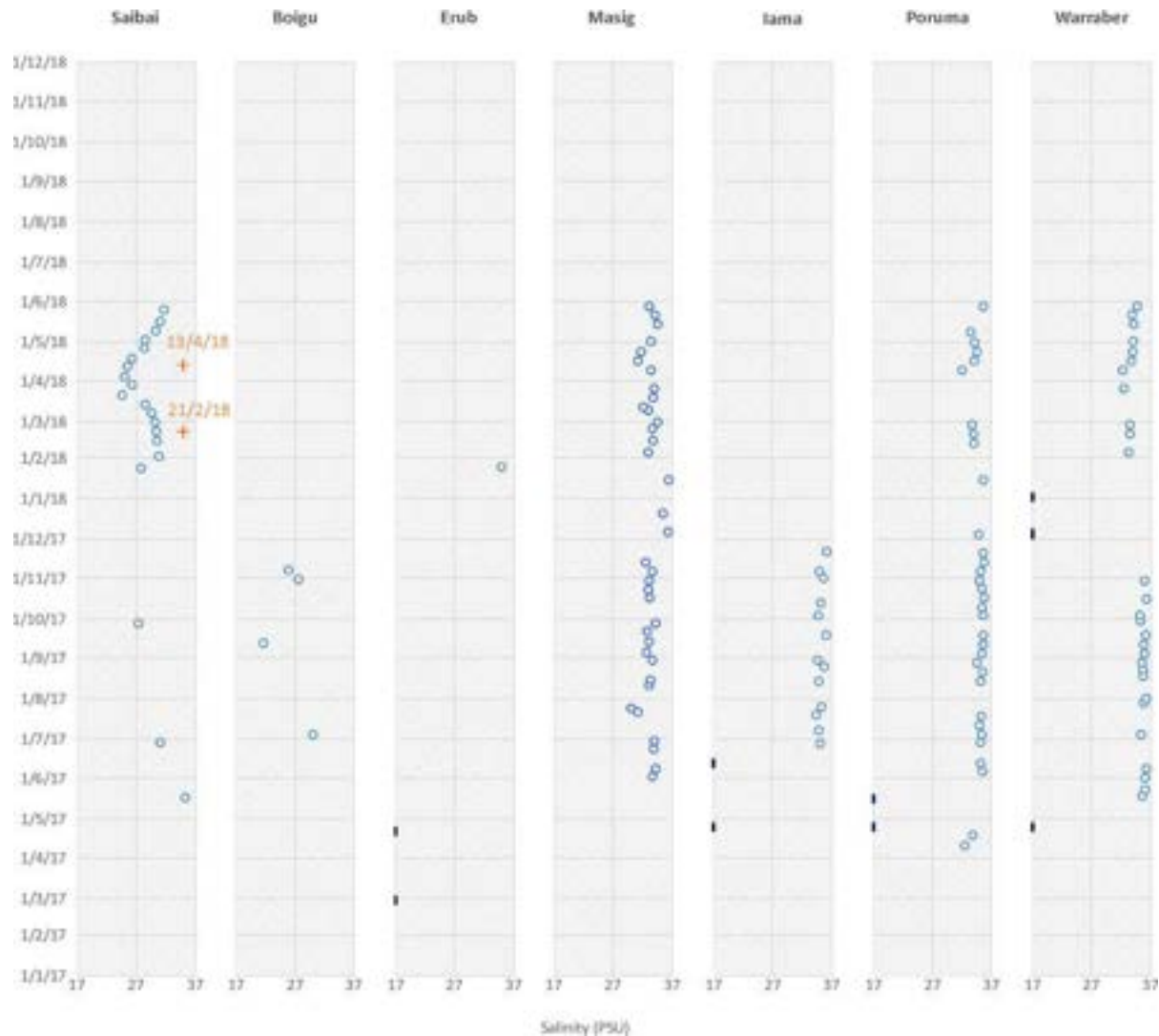


Figure 4-24: Time series of salinity measured at Saibai, Boigu (insufficient data), Erub (insufficient data), Masig, Poruma and Warraber Islands: Corrected salinity values. Dark blue dashes indicate days when the salinity sampling was cancelled due to rain. Examples of the CC maps of 21/02/2018 and 13/04/2018 (orange crosses) are presented in Figure 4-25.

As shown in Table 4-2, the number of salinity measurements was greater at Masig, Warraber, Poruma and Saibai Islands (≥ 20 measurements) than at Lama Island (14). Insufficient data was collected at Boigu and Erub Islands at the time of reporting (< 5) to allow any meaningful analysis. The lower mean salinities were measured at the most northern site, Saibai Island (29.0 ± 2.6 PSU), and the highest mean salinities were measured at the more southern sites, Lama (35.3 ± 0.7 PSU), Warraber (35.0 ± 1.2 PSU) and Poruma Islands (34.8 ± 0.9 PSU). The mean salinity measurement at Masig Island was lower than expected for an open ocean site with 33.3 ± 1.3 PSU, but the range was relatively high (6.3 PSU) in the 39 samples collected between June 2017 and May 2018. The greatest salinity range was measured at Saibai Island (10.7 PSU), with the lowest ranges at the southern sites, Lama: (2.6 PSU), Poruma (3.7 PSU) and Warraber Islands (4.7 PSU). This illustrates that Saibai Island had more variable salinity levels (potentially due to riverine influence) than southern sites where salinity always exceeded 32 PSU.

Table 4-2: Summary statistics of the corrected salinity time series. SD: standard deviation, range: maximum – minimum, Q1/Q3: first/third quartile, N outliers: number of outliers removed. Statistics are calculated without the outliers. Results for Boigu and Erub Islands are shaded in grey due to recording of less than five samples.

Location	Saibai	Boigu	Erub	Masig	Iama	Poruma	Warraber
Count	20.0	4.0	1.0	39.0	14.0	34.0	29.0
Mean	29.0	26.3	35.0	33.3	35.3	34.8	35.0
Median	29.1	26.8	35.0	33.4	35.1	35.1	35.5
SD	2.6	3.1	0.0	1.3	0.7	0.9	1.2
Min	24.7	21.7	35.0	30.2	34.4	32.1	32.3
Max	35.3	30.1	35.0	36.4	37.1	35.7	37.0
Range	10.7	8.4	0.0	6.3	2.6	3.7	4.7
Q1	27.1	24.8	35.0	32.9	34.8	34.2	34.0
Q3	30.6	28.2	35.0	34.0	35.7	35.5	35.9
N outliers	0	4	3	0	0	0	0

Seasonal patterns of salinity were observed at Masig Island, with lower salinity measured around July 2017 and higher values recorded around January 2018 (Figure 4-24). As noted in Section 4.4, the salinity data collected with the continuous loggers was not reliable for comparison.

At Saibai Island, higher salinities were measured around May 2017, February 2018 and June 2018, and lower salinities around April 2018. However, the limited number of measurements did not allow further analysis of the periodicity of this time series. Examples of MODIS true colour images and CC maps collected on 21 February 2018 and 13 April 2018 were selected to illustrate changes in water compositions that can exist around Saibai Island between the periods of lower and higher water salinity (Figure 4-25). The image for 13 April 2018, when lower salinity measurements were recorded, indicates the existence of turbid waters in the vicinity of Saibai Island, potentially extending from the Fly River plume. However, these images are only examples and do not describe typical/systematic trends in water composition associated with low/high salinity values at Saibai Island.

4.5.2 Recommendations

These results demonstrated that northern areas in the Torres Strait region were exposed to lower salinity waters and greater salinity ranges than those localised in the mid and southern region of the Torres Strait. Measurements allowed a preliminary description of seasonal patterns of salinity at Masig and Saibai Islands.

The project team encourages the TSRA to continue the weekly salinity monitoring programme around the northern islands in the Torres Strait. When this programme has been implemented for a long enough period of time and when representative (several seasons) time series of salinity data are available, it will be possible to define statistically significant trends and seasonal patterns in salinity. It will also be possible to define ‘alert’ salinity thresholds at each island under which satellite images can be downloaded and reviewed. This will support more accurate tracking of the Fly River plume and the evaluation of its potential threat in real time.

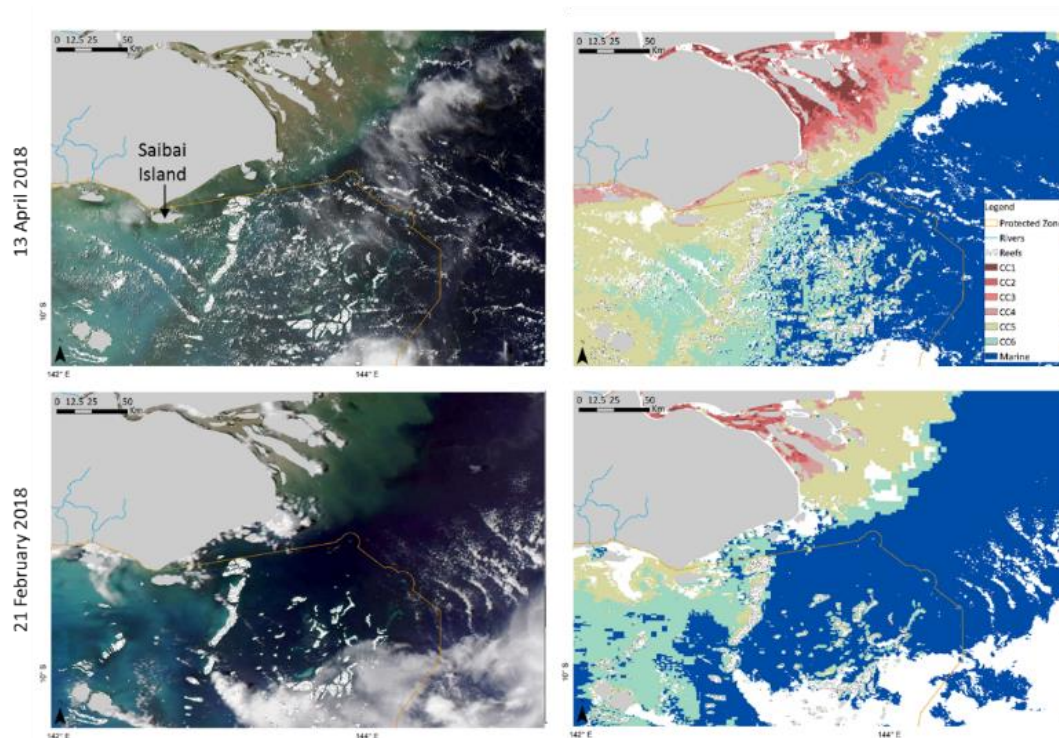


Figure 4-25: Example of MODIS true colour images and CC maps collected on 21 February 2018 and 13 April 2018 (orange crosses on Figure 4-24, Saibai Island).

4.6 Water quality exposure assessment

Purpose: Map the frequency of exposure of reef and seagrass habitats to turbid waters using the classification of remote sensing true colour images into water CCs (water types) and develop a potential exposure map for the region. Using evidence from other project components, undertake a qualitative assessment of the potential contribution of the Fly River to the areas of greatest exposure to turbid waters.

Highlights:

- The frequency of exposure of waters, coral reefs and seagrass habitats to turbid waters in the Torres Strait was assessed using the daily CC maps (six distinct colours) produced via the remote sensing analysis for 2008 to 2018. **Turbid conditions can be driven by resuspension and/or the influence of Fly River discharge under certain conditions and/or the influence of the southern PNG rivers (e.g. Wassi Kussa and Mai Kussa rivers at certain sites around Boigu and Saibai islands).**
- Multi-annual (2008 to 2018) frequency of exposure maps illustrated an inshore to offshore spatial pattern, with the highest frequency of the Primary water type (i.e., more turbid conditions) in the coastal areas, and offshore areas most frequently exposed only to the Tertiary water type (see explanation of water types in Section 4.3). These results are in agreement with the satellite analysis of long-term water quality conditions (Section 4.3).
- Assessment of the frequency of exposure of coral reefs and seagrass habitats (including subtidal, intertidal and NW Torres Strait seagrasses) to turbid waters (2008–2018) showed that the majority of the coral reef and seagrass habitat areas across the Torres Strait are exposed to turbid Primary and Secondary water types (CC1–5) less

than 50% of the time, except for the NW Torres Strait seagrass area. In this area, approximately 71% (273 km²) of the habitat area was exposed frequently (50 to 90% of the time) and 29% (110 km²) was exposed to turbid waters very frequently (90–100%).

- Spatial analysis of the frequency of exposure of 10 selected coral reef and seagrass habitat sites to in the northern and central Torres Strait Primary and Secondary waters revealed that habitats at Boigu and Saibai Islands experience the greatest exposure to turbid waters. Habitats around the northern Warrior Reefs, Erub and Mer are also exposed to turbid waters, although less frequently. Importantly, these are sites that have healthy coral reefs with high coral cover and low macroalgae cover, making them at high risk of ecological impacts, however, the evidence to date does not indicate residual impacts from previous (potential) exposures to Fly River plumes.
- Further analysis of these results is required **to identify the source of turbid waters**, including additional *in-situ* sampling and habitat surveys to validate the results.

4.6.1 Overview of methods and results

This study used daily MODIS satellite true colour imagery and a semi-automated colour classification technique developed for the GBR (Álvarez-Romero et al. 2013; Devlin et al. 2015) and fully described in Section 4.3. In this technique, MODIS true colour imagery are enhanced (Red-Green-Blue, RGB to Intensity-Hue-Saturation, IHS) and clustered into ‘cloud’, ‘ambient water’ and six wet season CCs (turbid water masses) through a supervised classification system using typical apparent surface colour signatures of (i) flood waters (IHS values) and (ii) dense clouds and sun glint (RGB values) in the GBR.

This approach, based on the automated classification of true-colour satellite imagery, offers a way of mapping river plumes under moderate cloud cover and with sun glint, as well as in highly turbid waters where the performance of other remote sensing methods is often limited (Álvarez-Romero et al. 2013). Due to a lack of validated remote sensing algorithms for the Torres Strait region at the time of this study, this approach was considered to be adequate for assessing the composition and frequency of turbid coastal waters as well as cloud frequency in the study area.

Seven complete years (2008 to 2017⁵ but excluding 2011 and 2012 due to NASA data issues) of daily CC maps of the Torres Strait region were produced to generate water type maps (used here and in Section 4.3). Water type mapping is a tool used to characterise large areas in terms of predicted water quality conditions. The colour classification captures colour gradients occurring in coastal and marine waters and aims to summarise typical changing water quality along- and cross-shore through the classification into Primary (turbid, sediment-dominated), Secondary (less turbid, dominant to moderate concentrations of Chl-a and finer sediments) and Tertiary (slightly above ambient turbidity and Chl-a concentrations) water types. A full description of the Primary, Secondary and Tertiary water type characteristics is shown in Section 4.3. Water type maps were used here to assess the frequency of occurrence of each

⁵ For all annual and seasonal frequency maps and figures, only complete years/seasons were used, i.e. 2009–2017 (excluding 2011 and 2012) for annual, 2008–2017 for Jul–Aug (excluding 2011) and 2009–2018 (excluding 2011 and 2012) for Feb–Mar.

water type at different temporal and geographical scales and to evaluate the frequency of exposure of coral reefs and seagrass habitats to turbid waters in the Torres Strait.

Annual and multi-annual frequency maps were generated from the daily CC maps for each of the CC1 to 6, as well as 1–4 combined (Primary water type), 1–5 (Primary plus Secondary water types) and 1–6 combined. The water type frequency was defined as the total number of days per year exposed to a given water type divided by the number of data days (non-cloud) recorded per year, resulting in a normalised frequency on a scale from 0 to 1.

An annual 'cloud frequency' raster was also produced, calculated as the number of cloud-masked days per pixel divided by the total number of daily rasters available for the year (Appendix 2A).

A buffer was applied around all features (banks, islands, mainland, reefs, other) in the AIMS "Complete Great Barrier Reef Island and Reef Feature boundaries including Torres Strait Version 1b" to a width of 1 pixel (px; 0.002197 decimal degrees). A buffer was also applied around the reef and seagrass layers to a width of 1 km. The first buffer was subtracted from the second buffer, then the polygon converted to raster (assigned by pixel centre and snapped to the existing raster grid). This raster mask was used to mask all of the daily CC maps to capture the surrounding waters for each habitat.

The annual and multi-annual frequency maps were then analysed following the steps below:

1. Seagrass and coral reef exposure (whole Torres Strait scale): extraction of pixel counts of percent frequency of exposure for primary (CC1–4), secondary waters (CC5) in 10% frequency increments—multi-annually (decadal average) as well as for July–August (within the SE trade wind season), February–March (within the monsoon season) and annually (Appendix 2B). As previously described, these data are for surrounding waters greater than 1 px from benthic features and within 1 km.
2. Exposure at different habitat sites across the Torres Strait: extraction of the frequency of exposure to the six CCs at different sites in the northern Torres Strait—multi-annually (decadal average) as well as for July–August, February–March and annually (Appendix 2C). The data were analysed at 10 sites where the SLIM hydrodynamic modelling, remote sensing or other studies indicated potential influence from Fly River plume waters, and/or where coral reef and seagrass data were available (Figure 4-26 and Table 4-3).
3. Exposure at different potential zones of influence: the sites were finally grouped into six different potential zones of influence based on their geographical location and their likely exposure to Fly River waters, defined as exposure to Fly River plume waters and subsequent resuspension (Table 4.3). The mean exposure (1 km radius) to Primary, Secondary, Tertiary and marine waters was extracted into the different zones of influence. Analyses were performed at different temporal scales: multi-annually (decadal average) as well as for July–August (within the SE trade wind season), February–March (within the monsoon season) and annually (Appendix 2C).

The assessment area is shown by the full extent of the map in Figure 4-26, encompassing the area west to Kawa Island, south to Beresford Shoals, north to Poruma Island and east of Mer (between 142.015°E to 144.295°E and 8.860°S to 10.354°S). This area includes 2,952km² of mapped coral reefs and 1,337km² of mapped seagrass meadows. The habitat data included in the assessment was:

- Coral reefs: mapped coral reefs (AIMS, Lawrey and Stewart 2016); and
- Seagrass: intertidal seagrass (Carter et al. 2014), subtidal seagrass (Carter et al. 2014), NW Torres Strait seagrass (Carter and Rasheed 2016) and subtidal seagrass presence/absence (Hayward et al. 2008).

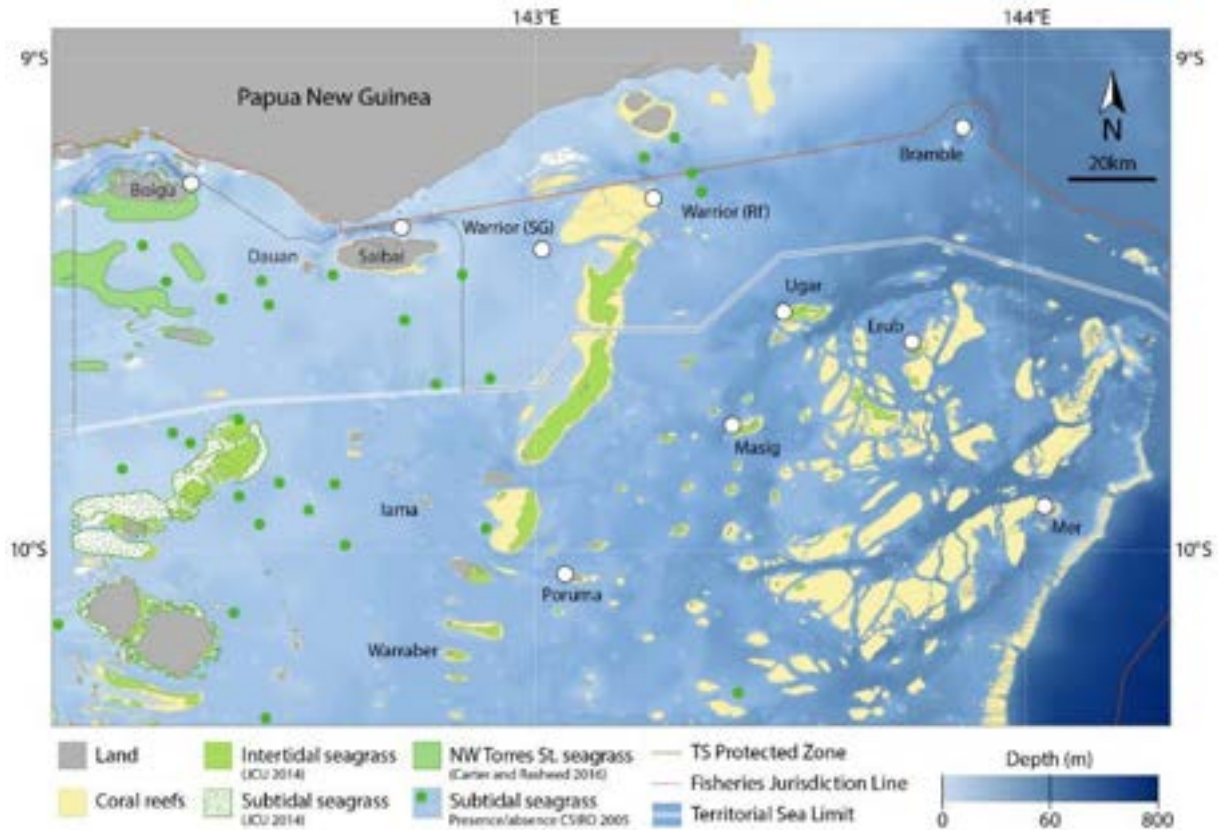


Figure 4-26: Location of the Torres Strait coral reefs, seagrass beds and habitat sites selected for the risk assessment (white dots) (between 142.015°E to 144.295°E and 8.860°S to 10.354°S). Map prepared by D. Tracey, TropWATER JCU using datasets from GBRMPA, Carter et al. (2014), Hayward et al. (2008) and Carter and Rasheed (2016).

Table 4-3: Habitat (coral reef and seagrass) sites selected for the risk assessment (and corresponding monitoring codes where relevant) and potential zone of influence: Zone 1: North, 2: North East, 3: North West, 4: South, 5: Central and 6: East. Note: SG = seagrass site; RF = reef site. The site names correspond to the TSRA coral reef monitoring locations.

Site	Lat	Long	Habitat type	Zone	Description and evidence from other studies: Coral cores (CC), <i>In-situ</i> loggers (L), Remote Sensing (RS), Modelling (M) and Bibliography (B)
Warrior Reef (SG)	9.42	143.05	seagrass	1	Very likely influence of Fly River waters (B, M, RS)
Warrior Reef (RF)	9.28	143.24	reef	1	Very likely influence of Fly River waters (B, M, RS)
Bramble Cay	9.14	143.87	reef	2	Several pulses of freshwater each year, likely from the Fly River (CC, L, RS)
Boigu	9.28	142.3	seagrass	3	Potential influence of Fly River (B, M, RS) and very likely influence of southern PNG rivers (RS)
Saibai	9.36	142.77	reef	3	Likely influence of Fly River (B, M, RS) and likely influence of southern PNG rivers (RS)
Poruma (P1R1)	10.05	143.06	reef	4	Low risk of influence from Fly River (B, M, RS)
Ugar (U1R1)	9.51	143.51	reef	5	Low risk of influence from Fly River (B, M, RS)
Masig (Y1R1)	9.75	143.4	reef	5	Low risk of influence from Fly River (B, M, RS)
Erub (E2R1)	9.58	143.77	reef	6	Low risk of influence from Fly River (B, M, RS, CC)
Mer (M3R1)	9.91	144.04	reef	6	Low risk of influence from Fly River (B, M, RS)

Long-term patterns

Multi-annual frequency maps

Multi-annual frequency maps (2009–2017, excluding 2011 and 2012) combine the outputs from annual assessments to generate a longer-term assessment of water quality conditions in the region. These maps illustrated an inshore to offshore spatial pattern, with the highest frequency of the Primary water type (i.e., more turbid conditions) in the coastal areas and offshore areas most frequently exposed only to the Tertiary water type (Figure 4-27).

To assist with interpretation of the data, the frequency of exposure to turbid waters has been grouped into categories based on the decadal average: Infrequently (0–10% of the time), occasionally (10–50% of the time), frequently (50–90% of the time) and very frequently (90–100% of the time).

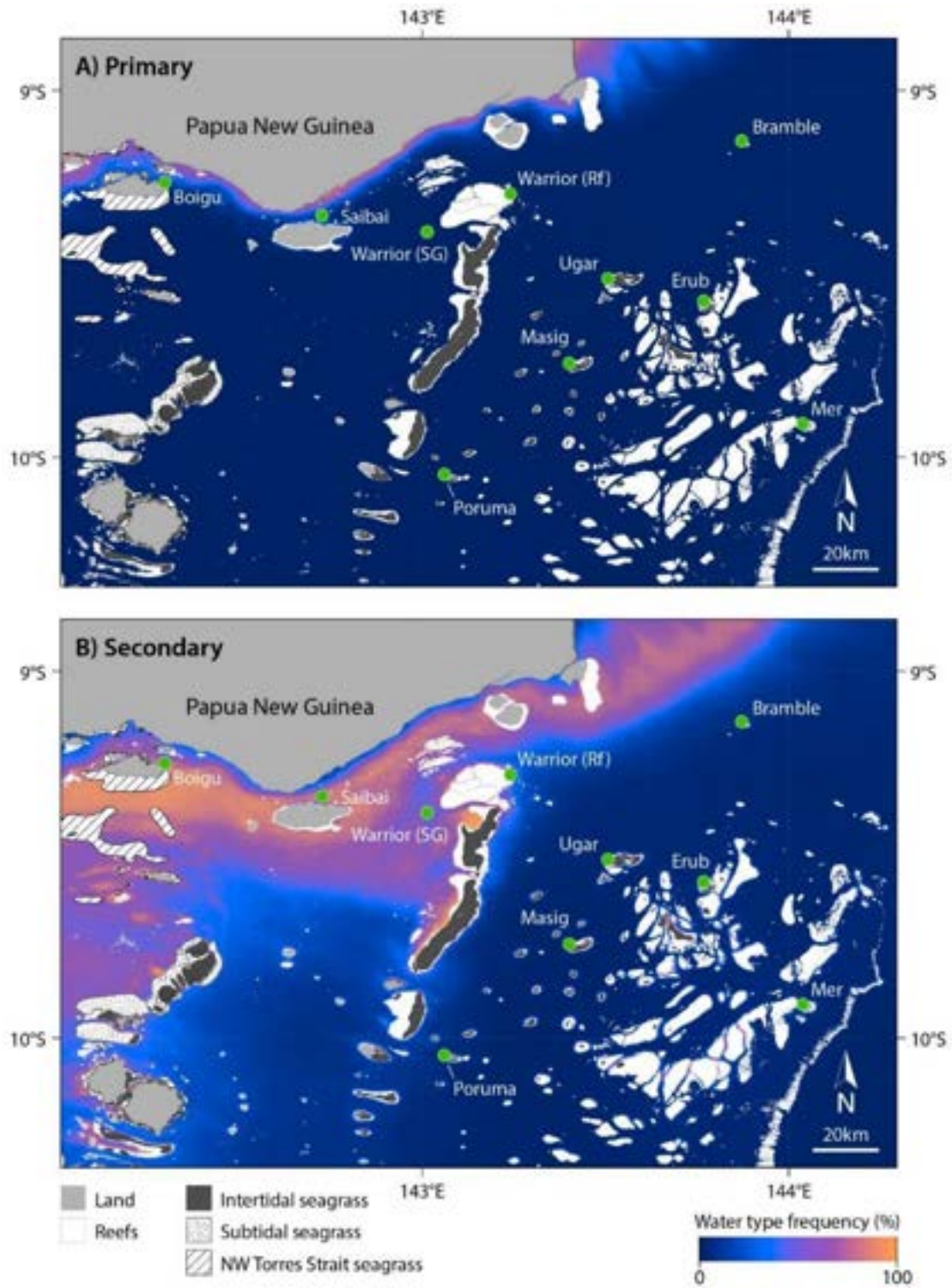


Figure continued below

Figure 4-27: Maps showing the multiannual frequency (2009–2017, excluding 2011 and 2012) of A) Primary water type (CC1–4), B) Secondary water type (CC5) and C) Tertiary water type (CC6) as well as D) cloud frequency occurrence; where the highest frequency is shown in orange and the lowest frequency is shown in blue. Maps prepared by D. Tracey, TropWATER JCU using datasets from GBRMPA, Carter et al. (2014), Hayward et al. (2008) and Carter and Rasheed (2016).

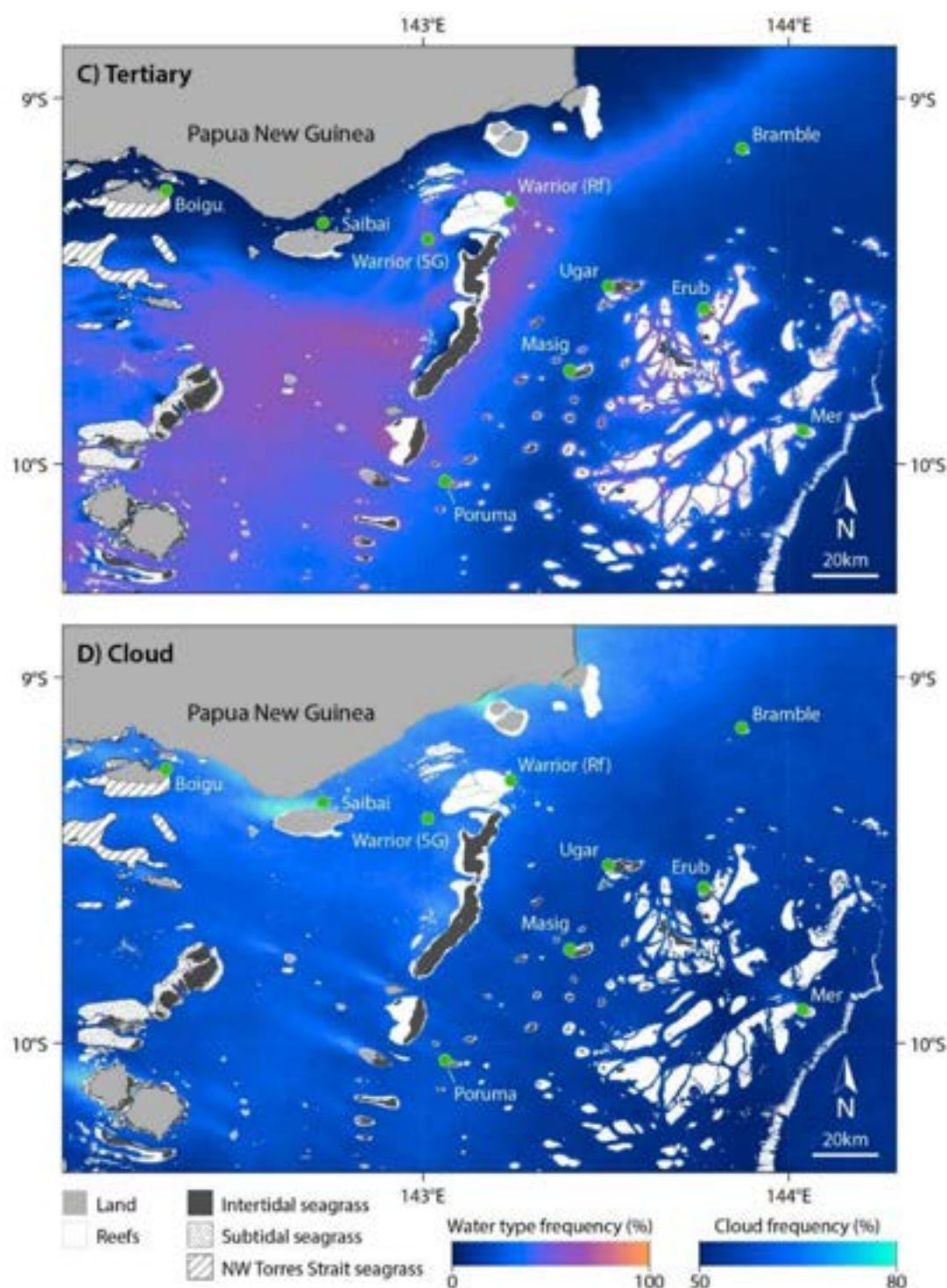


Figure 4-27: Maps showing the multiannual frequency (2009–2017, excluding 2011 and 2012) of A) Primary water type (CC1–4), B) Secondary water type (CC5) and C) Tertiary water type (CC6) where the highest frequency is shown in orange and the lowest frequency is shown in blue; and D) cloud frequency occurrence where the highest frequency is shown in aqua and the lowest frequency is shown in blue. Maps prepared by D. Tracey, TropWATER JCU using datasets from GBRMPA, Carter et al. (2014), Hayward et al. (2008) and Carter and Rasheed (2016).

The multi-annual frequency analysis of the occurrence of clouds showed the highest cloud frequency at the Saibai and Boigu Islands and Warrior Reef sites (Figure 4-27 and Table 4-4) and typically greater or similar cloud frequency during July–August (within the SE trade wind season) and February–March (within the monsoon season) in most locations (based on multi-annual average conditions—this varies between years, see detailed graphs in Appendix 2A).

Table 4-4: Frequency of cloud cover in the daily images for each of the assessment sites (average % of total number of days in 2008–2018).

	Boigu	Bramble Cay	Erub (E2R1)	Masig (Y1R1)	Mer (M3R1)	Poruma (P1R1)	Saibai	Ugar (U1R1)	Warrior Reef RF	Warrior Reef SG
Annual	64.7	60.3	61.0	59.8	61.7	63.4	71.3	59.9	66.1	64.6
July–August	70.3	66.7	60.4	62.1	64.4	65.6	80.9	65.6	73.1	72.2
February–March	71.7	58.0	61.4	61.9	64.5	68.1	74.0	61.6	61.5	63.1

Coral reef and seagrass exposure (whole Torres Strait scale)

The assessment of the areas of coral reef and seagrass habitats (shown in Figure 4-26) exposed to Primary and Secondary water types combined (CC1–5) between 2009 and 2017 is shown in Table 4-5 and Figure 4-28. The assessments for July–August (within the SE trade wind season), February–March (within the monsoon season) and annual periods are shown in Appendix 2B.

In summary, 53% (or 1,573 km²) of coral reef area was exposed to Primary and Secondary waters infrequently (0–10% of the time, decadal average) and only approximately 3% (or 89 km²) of coral reef area was exposed to turbid waters very frequently (90–100% of the time) (Table 4-5).

Intertidal seagrass habitats show greater frequency of exposure to turbid waters, with 11.9% (or 67 km²) of habitat area being exposed to Primary and Secondary waters infrequently (0–10% of the time), and 63% (or 354 km²) of area exposed occasionally to turbid waters (10–50% of the time). However, only approximately 2.4% (or 14 km²) of intertidal seagrass area is exposed to turbid waters very frequently (90–100%).

The subtidal seagrass also showed greater frequency of exposure to turbid waters, with 0.2% (or 1 km²) of habitat area being exposed to Primary and Secondary waters infrequently (0–10% of the time), and 54% (or 212 km²) of area being exposed occasionally (10–50% of the time) to turbid waters. However, only approximately 0.9% (or 3 km²) of subtidal seagrass area was exposed to turbid waters very frequently (90–100%).

These patterns are explored further in relation to seasonal and inter-annual differences in Appendix 2B.

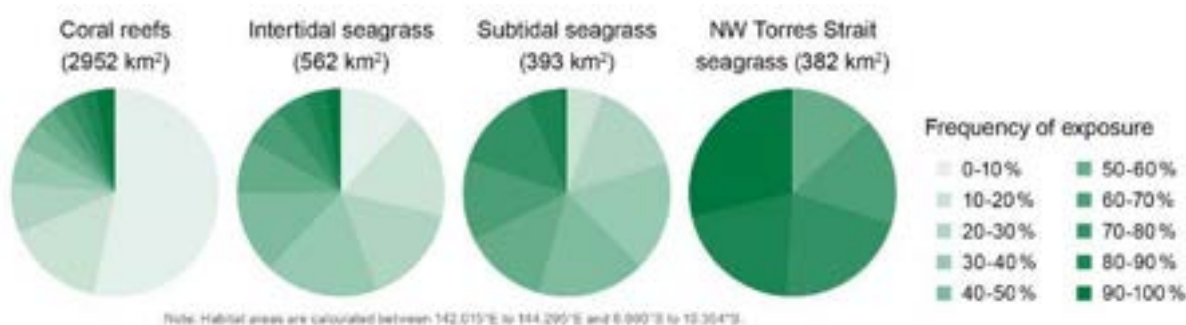
Finally, the NW Torres Strait seagrasses showed the greatest frequency of exposure of all habitats. None of the habitat area was exposed to Primary and Secondary waters less than 50% of the time. Approximately 71% (273 km²) of habitat area was exposed frequently (50 to 90% of the time) and 29% (110 km²) of the NW Torres Strait seagrass area was very frequently (90–100%) exposed to turbid waters.

Table 4-5: Exposure (area km² and % of habitat area) of coral reef, intertidal and subtidal seagrass habitats to Primary and Secondary (CC1–5) waters between 2008 and 2018.

Frequency of exposure:	0-10%	10-20%	30-30%	30-40%	40-50%	50-60%	60-70%	70-80%	80-90%	90-100%
Habitat	Area of habitat exposed to primary or secondary waters (km ²)									
Coral reefs	1573	456	227	177	123	88	80	67	72	89
Intertidal seagrass	67	94	89	99	72	47	35	27	18	14
Subtidal seagrass	1	21	58	67	66	52	49	53	22	3
NW Torres St. seagrass	0	0	0	0	0	50	64	81	78	110

Frequency of exposure:	0-10%	10-20%	30-30%	30-40%	40-50%	50-60%	60-70%	70-80%	80-90%	90-100%
Habitat	Percentage of habitat area exposed to primary or secondary waters (%)									
Coral reefs	53.3	15.4	7.7	6.0	4.2	3.0	2.7	2.3	2.4	3.0
Intertidal seagrass	11.9	16.7	15.9	17.7	12.8	8.3	6.2	4.8	3.2	2.4
Subtidal seagrass	0.2	5.4	14.8	17.1	16.9	13.2	12.4	13.6	5.6	0.9
NW Torres St. seagrass	0.0	0.0	0.0	0.0	0.0	13.0	16.8	21.1	20.3	28.8

■ >50% of habitat area
■ 10-50% of habitat area
■ <10% of habitat area

**Figure 4-28: Exposure (area km²) of coral reef, intertidal and subtidal seagrass habitats to Primary and Secondary (CC1–5) waters between 2008 and 2018 (excluding 2011 and 2012).**

Exposure at different habitat sites

Figure 4-29 shows the average long-term (2008 to 2018) frequency of exposure to each CC for each of the habitat sites. The average (decadal) exposure to the most turbid CCs (CC1–4 or Primary water type) decreased from the most northern to the sites further south. The greatest frequency of exposure to Primary waters was measured at Saibai and Boigu Islands, the northern sites (Figure 4-29, top graph); exposure at the more southern sites (Figure 4-29, bottom graph) was greatest at Erub Island. The Warrior Reef sites, and Saibai, Boigu, Erub and Mer Island sites were predominantly exposed to Secondary waters (CC5, $\geq 50\%$ of the time), with the greatest frequency of exposure to Secondary waters also measured at Saibai and Boigu Islands (84% of the time). The Ugar, Masig and Poruma Island sites were predominantly exposed to Tertiary waters (CC6, $\geq 50\%$ of the time). Bramble Island was exposed 26%, 38% and 34% of the time to Secondary, Tertiary and marine waters, respectively.

These patterns are explored further in relation to seasonal and interannual differences in Appendix 2C.

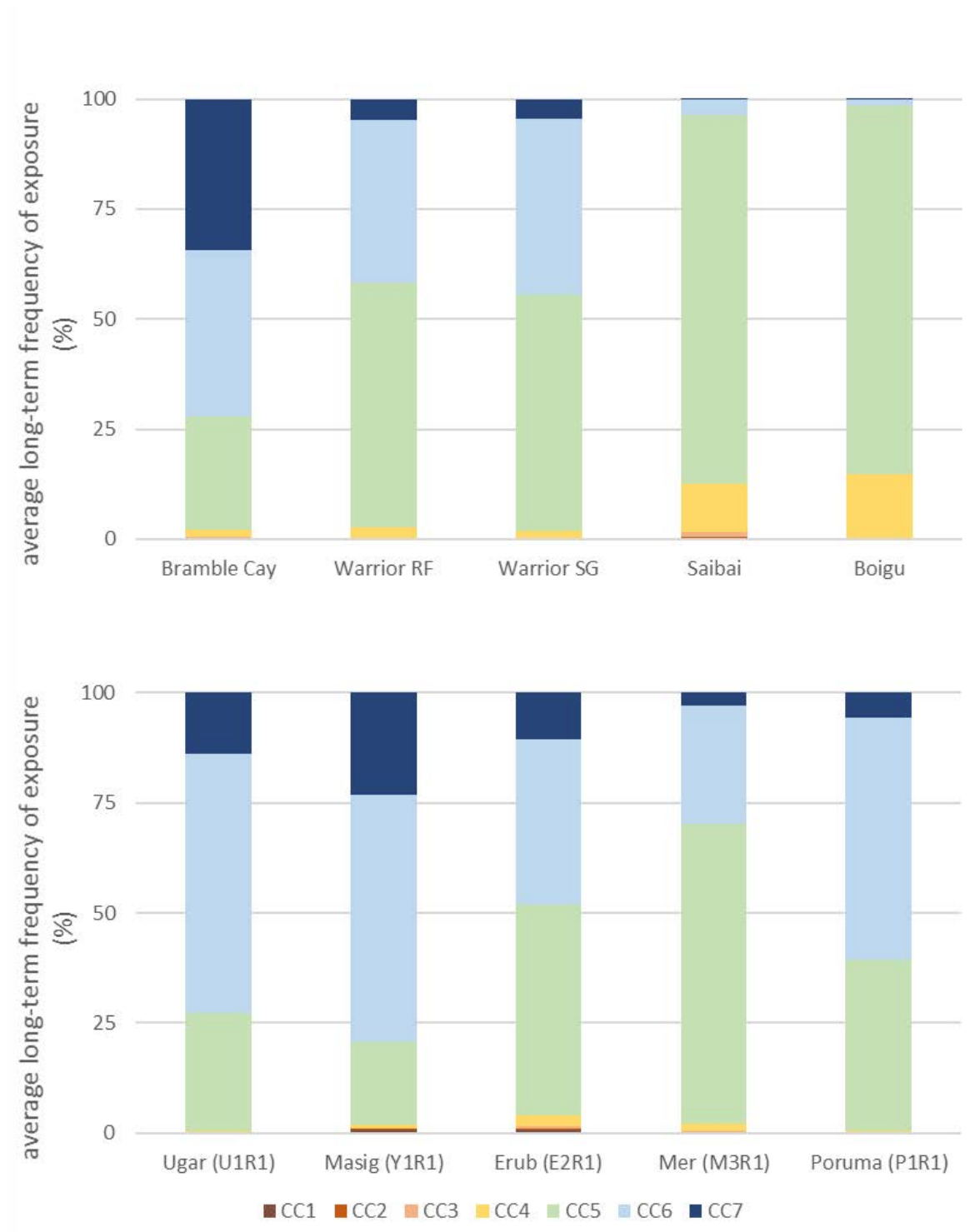


Figure 4-29: Long-term frequency (decadal mean, 2008–2018) of exposure to different CCs (CC1 to CC6) at the selected coral reef and seagrass monitoring sites in the northern Torres Strait (Table 4-3). Site codes in brackets after each site name (TSRA 2016): Top graph: Northern sites, bottom graph: Central sites. The water type frequency was defined as the total number of days per year exposed to a given water type divided by the number of data days (non-cloud) recorded per year, resulting in a normalised frequency on a scale from 0 to 1. Locations of the sites are presented in Figure 4-26.

Exposure to different potential zones of influence

Six potential zones of influence for the assessment area have been defined based on the results of the potential influence of the Fly River discharge in other sections of this report. The results for these zones are shown in Figure 4-308. The zonal results support the findings at the habitat site level presented above. The greatest frequency of exposure to Primary waters (CC1–4) was measured in the NW zone (3) and the greatest frequency of exposure to Secondary waters (CC5) was in the NW and North zones (3 and 1). The North and Central zones (1 and 5) had the greatest exposure to Tertiary waters (CC6) and the eastern zones (2 and 6) were the most often classified as marine waters (CC7).

This supports the premise that habitats in the northern Torres Strait are exposed more frequently to turbid waters than habitats in the south (Figure 4-30, Zones 1 and 3). The source cannot, however, be directly attributed to the Fly River and may include other coastal rivers (e.g. Wassi Kussa River at Boigu Island) and resuspension of sediment via currents, wind and wave action (e.g. at Erub, Masig and Mer Islands).

Potential links to ecosystem health

As noted above, the habitat sites for this analysis were selected from locations that are regularly monitored either by TSRA as part of their Coral Reef Monitoring Program or by TropWATER as part of ongoing seagrass health assessments (Carter and Rasheed 2016; TSRA 2016). However, it is difficult to make any causal links between the exposure results and ecosystem health assessments.

While the monitoring results from the 2015–2016 surveys conducted by the TSRA coral reef program (TSRA 2016) provide some indication of reef condition when compared with GBR long-term standards (Table 4-6), additional survey data are required to make any assessment of reef condition against the frequency of exposure to Primary and Secondary waters. ***The exposure results are only included here as an example of how this data could be considered in the future with longer term datasets.*** The lowest hard coral cover was monitored at Ugar Island (between 18.7% and 24.7%) and also the highest macroalgae cover (17.2%). Ugar Island had among the lowest frequency of exposure to Primary waters (CC1–4, Figure 4-29), indicating that turbidity is not likely to be the main driver of low coral cover in this location.

All other sites surveyed had moderate to high coral cover and low macroalgae cover (Table 4-6), which is consistent with previous surveys that recorded generally high cover (< 30%), particularly on the eastern Warrior Reefs (up to 47%) (Haywood et al. 2007; Sweatman et al. 2015; note that this is the most recent published data available). Other indicators generally associated with exposure to turbidity—such as coral disease—were also low and within a range generally considered healthy (see Thompson et al. 2017).

Table 4-6: Summary results from the Torres Strait coral reef monitoring surveys 2015–2016 (TSRA 2016).
Colour shading: Green = high; Orange = moderate.

Location	Long term exposure results – average frequency		Hard coral cover (average %)	Comparison with GBR standards	Macroalgae cover (average %)	Comparison with GBR standards	Coral disease (average affected %)
	Primary	Secondary					
Poruma	0.50	38.82	27.3 – 50.9		7.5		n.d.
Masig	1.95	18.85	44.2 – 44.3		3.3		8.1
Ugar	0.49	26.79	18.7 – 24.7		17.2		0.2
Erub	4.14	47.78	27.7 – 34.2		2.5		1.3
Mer	2.13	68.05	41.8 – 48.1		3.1		0.3

Surveys in 2013 recorded 279 species of coral with massive corals locally dominant on the Warrior Reefs (Haywood et al. 2007; Osborne et al. 2013) and species richness varying along an east-west gradient, with the Acroporidae and Pocilloporidae coral families having higher richness on the eastern reefs (Osborne et al. 2013). These results indicate that the species richness of coral communities in the Torres Strait may reflect an east-west gradient in turbidity and wave exposure (Osborne et al. 2013).

While the exposure results provide a preliminary estimate of the spatial extent of exposure of habitats in the Torres Strait to turbid waters, **no direct link between exposure to river plume waters and reef condition can be made with these data at this stage.**

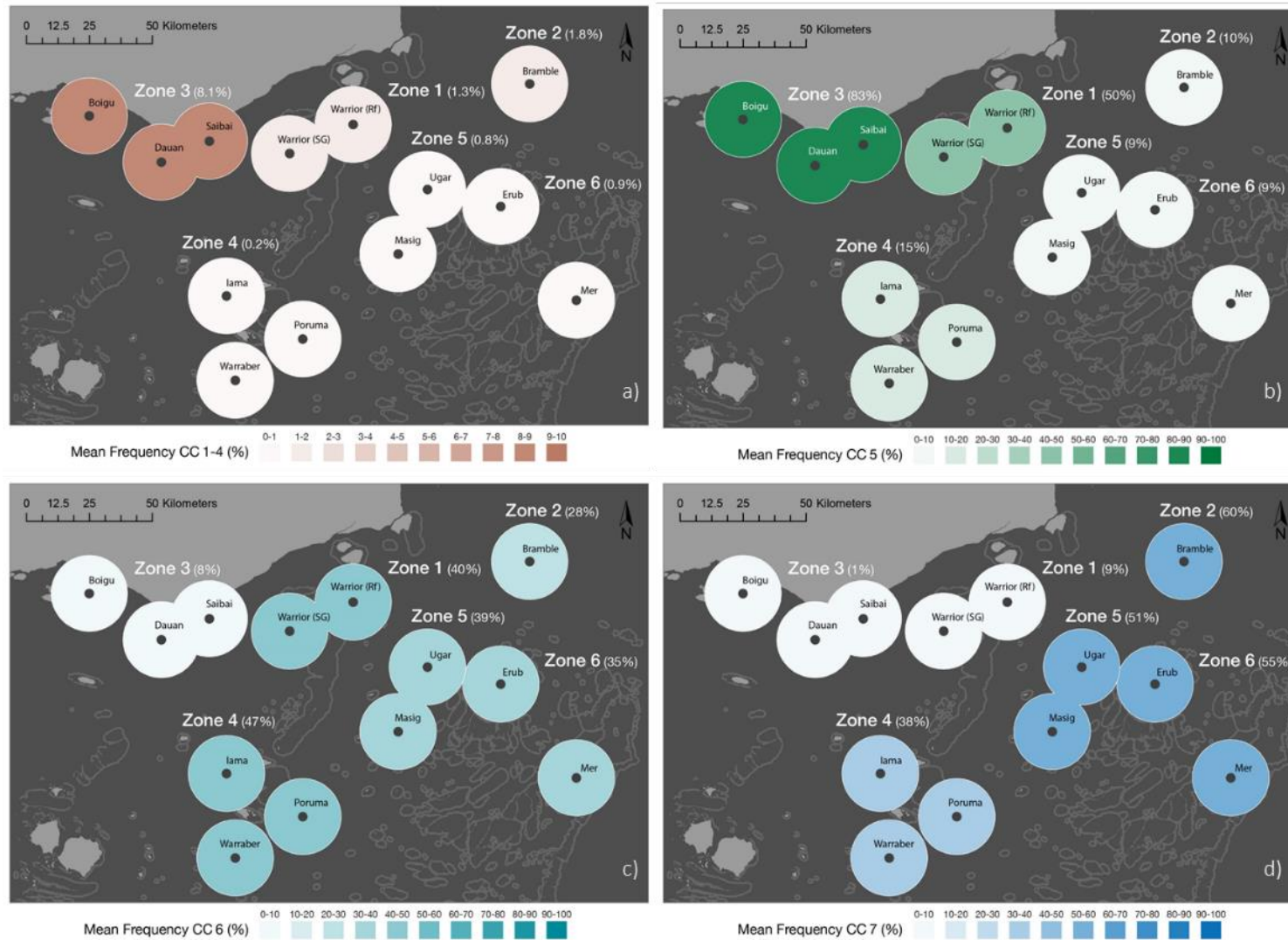


Figure 4-30: Spatial analysis of the mean frequency of exposure of habitat sites (1 km radius) to Primary (CC1–4), Secondary (CC5), Tertiary (CC6) and marine waters (CC7) into different zones of influence as defined in Table 4-3. Zone 1: North; Zone 2: North East; Zone 3: North West; Zone 4: South; Zone 5: Central and Zone 6: Eastern. Maps prepared by D. Tracey, TropWATER JCU.

Annual and seasonal patterns

In specific years and seasons, the Torres Strait reefs and seagrasses had a higher frequency of exposure to turbid waters, as shown by the exposure assessments for July–August (within SE trade wind season), February–March (within monsoon season) and annual periods at the (i) habitats (whole of Torres Strait scale), (ii) habitat sites and (iii) zonal geographical scales in the Appendices (Appendices 2B, 2C and 2D, respectively). These variations are expected **given the seasonal nature of turbidity, in part driven by trade winds and resuspension** of the muddy carbonate sediments found around northern islands and reefs (Sweatman and Berkelmans 2012).

Examples of observations are provided below.

For the **whole of Torres Strait scale habitat assessment in Appendix 2B** (coral reefs, intertidal seagrass and subtidal seagrass) there were clear examples of patterns across habitats such as:

- While there was limited and infrequent exposure to Primary waters in all assessment periods for all habitats, some exposure was evident for coral reefs in all years where data was available. For coral reefs and intertidal seagrass, the area of exposure to Primary waters (while limited) was greatest during July–August 2008 and February–March 2009 and 2013.
- The area of exposure to Primary and Secondary waters was greatest in February–March for all years for all habitats. The exception was for subtidal seagrass during July–August 2008 when the results were comparable.
- For July–August and February–March, the greatest frequency and areas of exposure for all habitats typically occurred in 2008 and 2015. In February–March, reduced frequency exposure was evident in 2017 and 2018 compared to other years.

Further analysis of these preliminary results is required to understand the potential environmental drivers of the seasonal and inter-annual variability in exposure. In particular, Fly River discharge data would be highly valuable.

For the **habitat assessment (Appendix 2C)** there were clear examples of differences between sites including:

- **Boigu and Saibai Islands** – dominated by Primary and Secondary waters. Greatest frequency of exposure to Primary waters of all sites, with greatest exposure in February–March. Primary waters evident in all years (where data available) in February–March except for 2009. Some occurrence of Tertiary waters in July–August. Greatest variability between years in July–August.
- **Warrior Reefs (seagrass and reef sites)** – dominated by Secondary waters and, to a lesser extent, Tertiary waters, with some exposure to Primary waters in February–March in most years where data was available, and in July–August in 2008 and 2015. Patterns between seasons and between years are reasonably consistent.
- **Erub and Poruma Islands and Mer** – dominated by Secondary waters, particularly in February–March, and Tertiary waters in July–August. Some limited exposure to Primary waters in some years, particularly in February–March.
- **Masig and Ugar Islands** – dominated by Tertiary waters and, to a lesser extent, Secondary waters, with very limited exposure to Primary waters, predominantly in February–March. Greatest variability between years in July–August.

- **Bramble Cay** – dominated by Tertiary and marine waters and, to a lesser extent, Secondary waters, with limited exposure to Primary waters in some years, predominantly in February–March. Patterns between seasons and between years are reasonably consistent.

As above, further analysis of these preliminary results is required to understand the potential environmental drivers of the seasonal and inter-annual variability in exposure, which may include local wind and wave action and currents, and could be done to some extent with the hydrodynamic model. Cloud coverage interferes with image collection and can influence the results by providing a bias towards the period where there is lower cloud cover. Again, Fly River discharge data would be highly valuable.

A preliminary analysis of the correlation between frequency of exposure at the habitat sites and the annual NINO3.4 sea surface temperature anomaly (NINO3.4, index for ENSO events) is presented in Section 4.7. The results showed weak negative relationships between annual NINO3.4 and the annual frequency of exposure to Primary (CC1–4) and Secondary (CC5) water types, with the strongest correlation observed at Bramble Cay ($r^2 = 0.5$, $p < 0.1$) and Masig Island ($r^2 = 0.3$, $p < 0.1$). However, further analysis of this result, as well as of the exposure datasets (Appendices 2B, 2C and 2D) is required to understand potential environmental drivers of the seasonal and inter-annual variability in exposure to turbid waters. Furthermore, cloud coverage interferes with image collection and can influence the results by providing a bias towards periods where there is lower cloud cover, which is typically associated with less rainfall (Appendix 2A, Figure 2A to 2D and Table 4-4).

4.6.2 Recommendations

The results of the frequency of habitat exposure to river plume waters identifies some coral reef and seagrass habitats in the northern Torres Strait that are frequently or very frequently exposed to turbid waters. However, these results are preliminary and further assessment is required. These conditions could be driven by resuspension and/or the influence of fluvial discharges from PNG rivers potentially including the Fly River discharge under certain conditions. It is therefore important to identify the source of Primary (CC1–4) and Secondary (CC5) waters at specific sites by combining sediment and water sampling with habitat health surveys to provide more insight into whether the discharge from the Fly River is likely to be a major driver of the condition of coral reef and seagrass habitats in the northern areas of the Torres Strait.

4.7 Testing correlations between water quality monitoring datasets

Purpose: Analyse trends and correlations existing between the different environmental and water quality datasets collected through the NESP Project 2.2.1 (Section 4.3, 4.4, 4.5 and 4.6).

Highlights

- Comparison of salinity and rainfall data collected at Masig Island and Bramble Cay as well as the frequency of exposure to satellite CCs and ENSO events measured at several reef and seagrass sites revealed valuable correlations between datasets.
- Negative weak relationships were observed between annual NINO3.4 and the annual frequency of exposure to Secondary water type (satellite CC5) at different seagrass and coral reef sites.
- The strongest correlations were observed at Bramble Cay where the strong La Nina event of 2010–2011 and strong El Niño event of 2015–2016 resulted in the highest and lowest frequency of exposure to secondary waters at Bramble Cay, respectively.
- At Masig Island, less saline waters were measured during the SE trade wind season (July) and more saline waters during the monsoon season (December).
- Results from this study underline the importance of integration of data from a range of sources to provide comprehensive assessments of water quality trends, and the need to continue the acquisition of multiple datasets to adequately support these analyses.

4.7.1 Overview of methods and results

Salinity and rainfall at Masig Island

An assessment of the potential correlation between salinity levels, potential plume intrusions and local rainfall at Masig Island was undertaken as an example of the value of the combined datasets in identifying specific influences from the Fly River plume. This is the only location where all datasets were available for a long enough period to be valid for analysis.

Daily averages of the local rainfall data collected at Masig Island using the continuous loggers (Figure 4-31, top) were compared with the salinity measurements collected by local rangers at Masig Island (Figure 4-31, bottom). The blue shaded rectangles highlight the periods of peak rainfall. There was no correlation between local salinity and periods of higher local rainfall during the monitoring period.

However, the observed patterns of measured salinity showed lower salinity values around July 2017 (winter) and higher values around December 2018 (summer) (Figure 4-31). This is opposite to the expected monsoonal pattern where intense summer rains produce localised lower salinities and long dry winters produce higher salinities. This may indicate background processes that drive lower salinity waters southward from the fresh water input from PNG rivers to Masig Island in the winter via the trade winds, with this reversing in the summer as the trade winds ease. Given the volume of freshwater discharged from PNG rivers versus those in Cape York, if this pattern is sustained then it may indicate a signal of periodic water movement from PNG down to Masig Island. To measure this will require longer time series using oceanographic grade instruments.

Monthly MODIS CC composites maps were also processed for July and December 2016 and 2017 (Figure 4-27a–d) to illustrate changes in water compositions that can exist around Masig

Island between the periods of lower and higher water salinity (the periods of imagery analysis are shown in Figure 4-31 as orange shaded rectangles). In all of these periods, Masig Island was located outside of the main zone of exposure to coloured waters (Figure 4-27). However, a trend toward the presence of larger coloured areas in July (compared to December) was visible in 2016 (before the local salinity monitoring commenced) and even more so in 2017, which corresponds to lower salinity measurements. These results also complement the results of the hydrodynamic modelling, which highlights the greater influence of the Fly River plumes during the SE trade wind season (see Section 4.2).

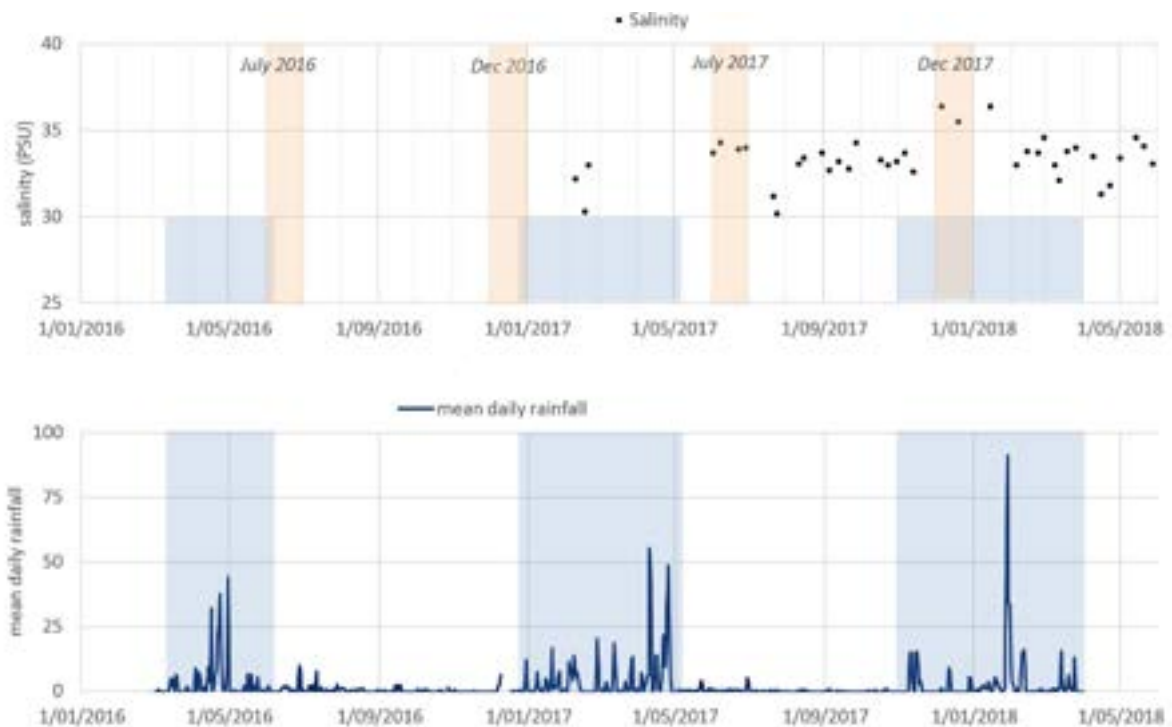


Figure 4-31: Mean adjusted salinity and rainfall measured at Masig Island. MODIS CC composite maps of July 2016, December 2016, July 2017 and December 2017 are presented in Figure 4-32.

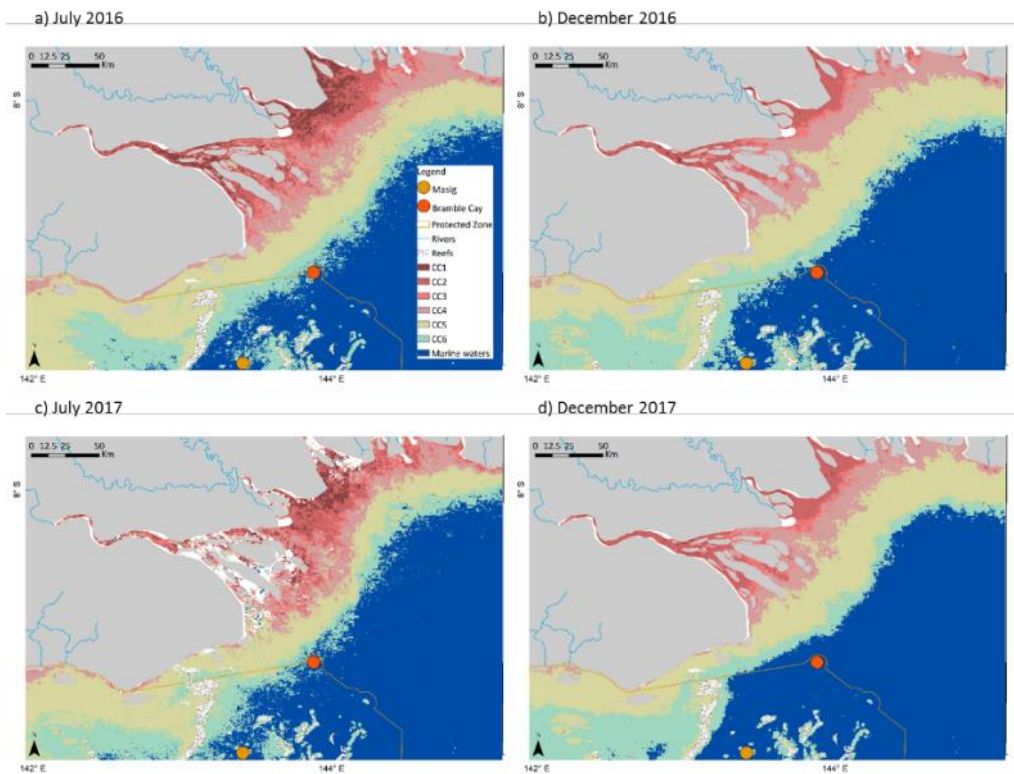


Figure 4-32: Monthly MODIS CC composite maps: median CC category per pixel over a) July 2016, b) December 2016, c) July 2017 and d) December 2017.

Examples of daily MODIS true colour and CC maps were also selected to illustrate water compositions around late June and early July 2017, when a peak in turbidity unrelated to local environmental conditions was reported at Bramble Cay. Figure 4-33 shows a panel of true colour (Figure 4-33a,b) processed CC maps (Figure 4-33c,d), and time series of turbidity and wind data recorded at Bramble Cay (Figure 4-33e from Section 4.4). Despite the intense cloud coverage observed on 29 June and 8 July 2017, daily MODIS true colour and CC maps showed large turbid areas off the Fly River mouth extending as far as Bramble Cay; Bramble Cay was exposed to CC6 (Tertiary waters) on 8 July 2017. These daily images are only examples and do not necessarily describe typical/systematic trends in water composition associated with higher coastal turbidity at Bramble Cay. The monthly MODIS CC composite maps also confirmed that Bramble Cay was exposed to CC6 waters during July 2017 (Figure 4-32c). During this period, intense (> 25 kph) SE winds were recorded (Figure 4-32e), indicating a probable correlation between the observed conditions at Bramble Cay and Fly River discharge.

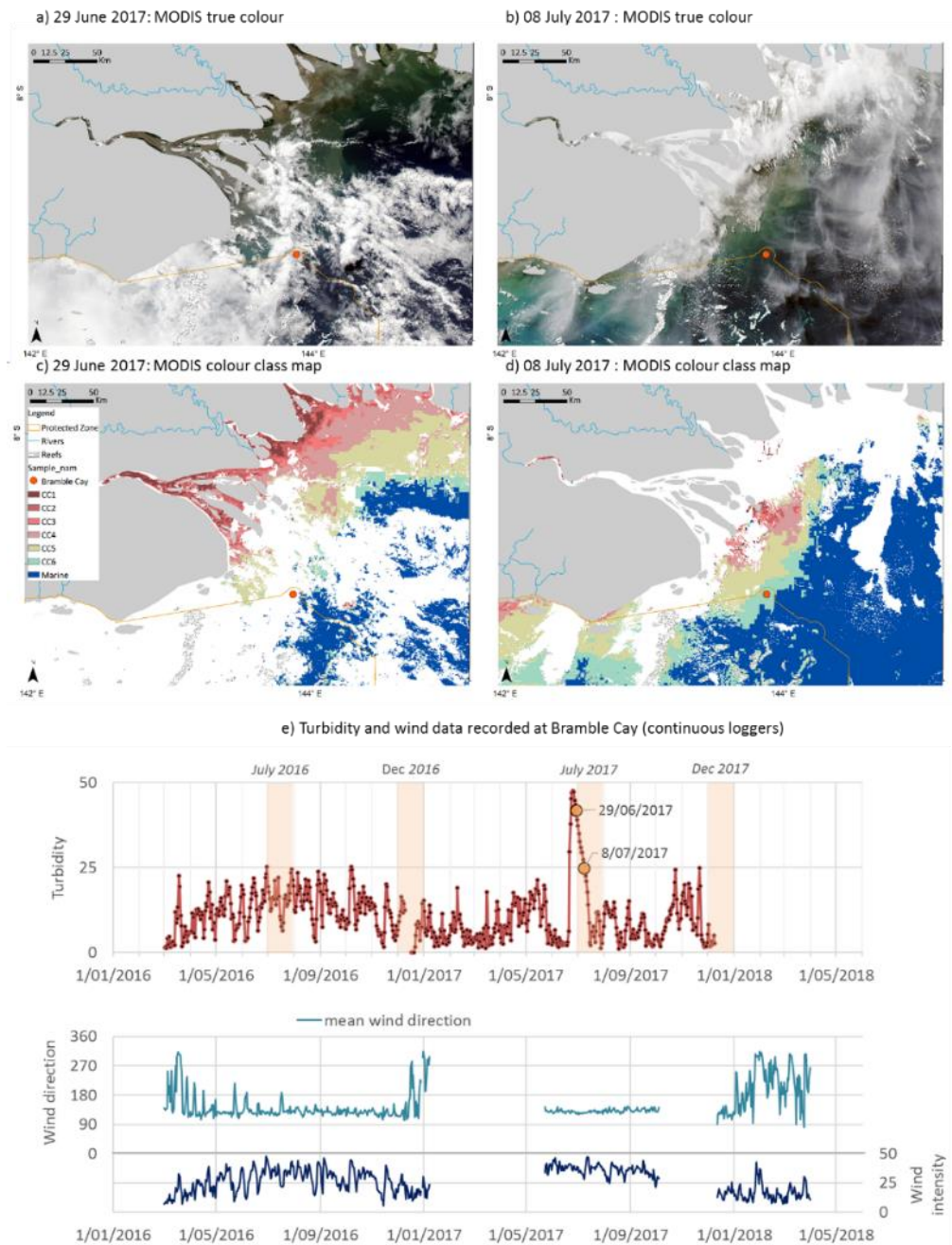


Figure 4-33: Examples of daily MODIS (a, b) true colour and (c, d) CC maps recorded around a turbidity spike measured at Bramble Cay and (e) turbidity and wind data recorded at Bramble Cay (e).

Frequency of exposure and ENSO events

The annual frequency of exposure of a selection of coral reef and seagrass sites to different satellite CCs has been extracted through the water quality exposure assessment (Section 4.6). Multi-annual mean frequencies of exposure were calculated and analyses were undertaken to evaluate the potential influence of the inter-annual ENSO cycles on the annual exposure to the CCs for each selected seagrass and reef site (Table 4-7).

The correlations between the annual NINO3.4 and the annual frequency of exposure to water types are shown in Table 4-7 and Figure 4-37.

Table 4-7: Correlations between the annual NINO3.4 and the annual frequency of exposure to Primary (CC1–4) and Secondary (CC5) and Tertiary (CC6) water types. The significant correlations are highlighted in bold text.

Site Location	stat.	NINO3.4		
		Primary	Secondary	Tertiary
Bramble Cay	r	-0.16	-0.69	0.53
	p	0.66	0.03	0.12
Warrior RF	r	-0.30	0.11	0.02
	p	0.40	0.76	0.96
Warrior SG	r	-0.30	-0.42	0.28
	p	0.41	0.22	0.44
Erub (E2R1)	r	-0.07	-0.26	0.38
	p	0.85	0.47	0.28
Poruma (P1R1)	r	-0.42	-0.16	0.15
	p	0.23	0.67	0.68
Ugar (U1R1)	r	-0.46	0.01	0.32
	p	0.18	0.97	0.36
Masig (Y1R1)	r	-0.50	-0.56	0.24
	p	0.14	0.09	0.51
Mer (M3R1)	r	-0.35	-0.34	0.40
	p	0.33	0.34	0.25
Boigu	r	-0.21	0.17	0.14
	p	0.56	0.64	0.70
Saibai	r	-0.60	0.45	0.37
	p	0.07	0.19	0.29

Weak negative and not significant relationships were generally observed between annual NINO3.4 and the annual frequency of exposure to Primary (CC1–4) and Secondary (CC5) water types (Table 4-7). Exception were at Bramble Cay, Masig Island and Saibai Island where correlations between annual NINO3.4 and the annual frequency of exposure to Primary (at Saibai Island) and Secondary (at Bramble Cay and Masig Island) water types were negatives, but significant ($p < 0.1$).

The strongest correlations between annual NINO3.4 and the annual frequency of exposure to Secondary waters was observed at Bramble Cay ($r^2 = 0.5$, $p < 0.1$, Figure 4-37b). The strong La Nina event of 2010–2011 (NINO3.4 = -1.64 in 2011) and strong El Niño events of 2015–16 (NINO3.4 = 2.6 in 2016) resulted in the highest (60% of the time) and lowest (12% of the time) frequency of occurrence of secondary waters at Bramble Cay (Figure 4-37a). The second strongest correlation was observed at Masig Island ($r^2 = 0.3$, $p < 0.1$, Figure 4-37d). The link between frequency of exposure to Secondary waters and El Niño events was however less consistent at Masig Island, with the highest frequency of occurrence of Secondary waters observed in 2016 and 2018 (29% of the time) and the lowest (9% of the time) in 2009 (Figure 4-37c). Correlations between annual NINO3.4 and the annual exposure to Primary waters at Bramble Cay and Masig islands were also negative but weaker ($R^2 = 0.02$ and $R^2 = 0.2$, respectively) than with the annual exposure to Secondary waters and not significant ($p > 0.1$).

At Saibai Island, the strong La Nina event of 2010–2011 (NINO3.4 = -1.64 in 2011) and strong El Niño events of 2015–16 resulted in the second highest (20% of the time, the highest was in 2008: 39%) and lowest (4.5% of the time) frequency of occurrence of Primary waters at Saibai Island (Figure 4-37e). Correlations between annual NINO3.4 and the annual exposure to Secondary waters at Saibai Island positive and not significant ($p > 0.1$).

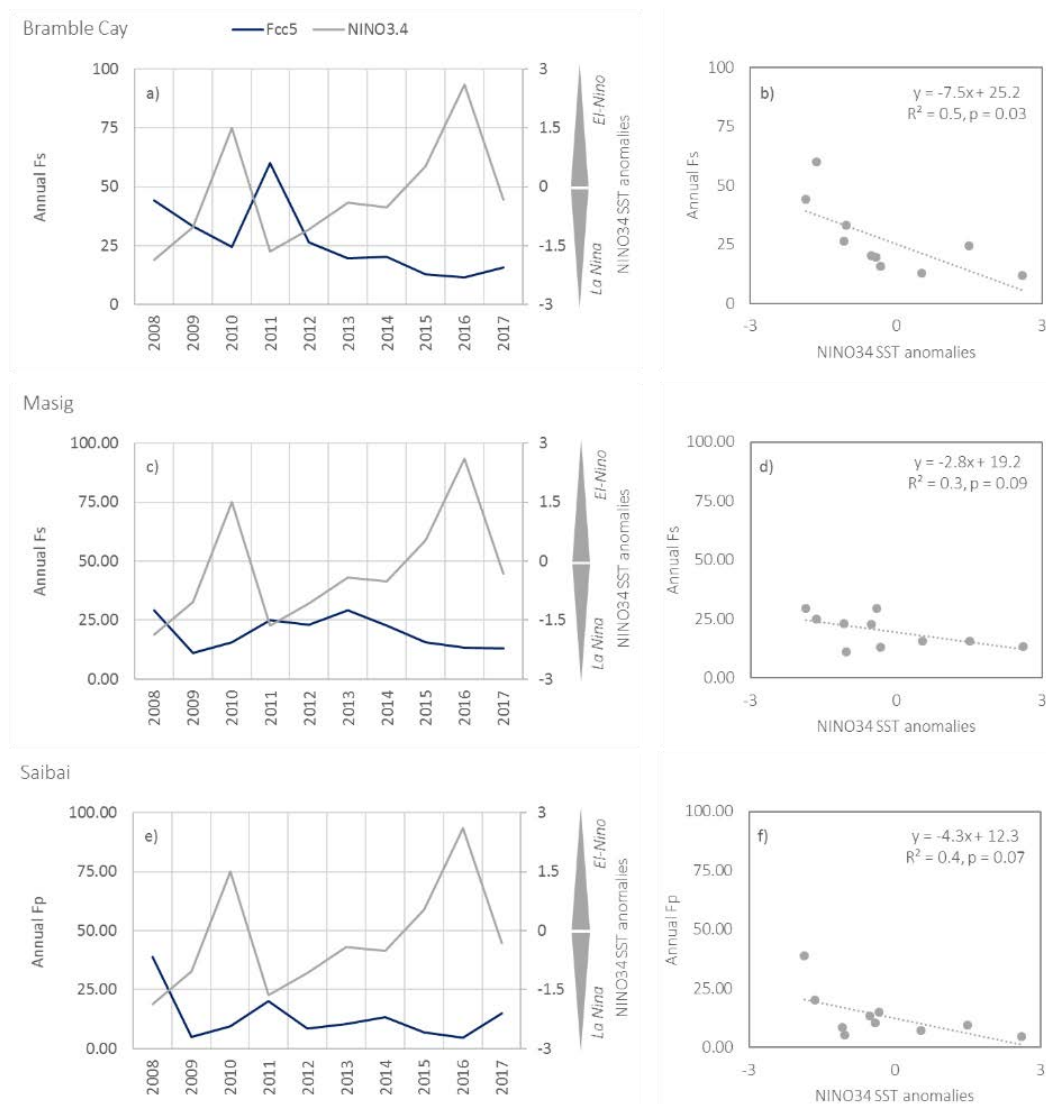


Figure 4-34: Time series (a, c) and correlation coefficients (b, d) between annual NINO3.4 and the annual frequency of exposure to Secondary waters (CC5) at (a, b) Bramble Cay and (c, d) Masig Island as between annual NINO3.4 and the annual frequency of exposure to Primary waters (CC5) at (e, f) Saibai Islands.

Recommendations

These datasets were used to evaluate the potential influence of the inter-annual ENSO cycles on the exposure of seagrass and coral reef sites to the Primary (CC1-4) and Secondary (satellite CC5) water type. Negative weak relationships were observed between annual NINO3.4 and the annual frequency of exposure to Primary (at Saibai Island) and secondary (at Bramble Cay and Masig Island) water types. The strongest correlation was observed at Bramble Cay where the strong La Nina event of 2010–2011 and strong El Niño event of 2015–

2016 resulted in the highest and lowest frequency of exposure to Secondary waters at Bramble Cay.

Results from this study underline the importance of integration of data from a range of sources to provide comprehensive assessment of water quality trends in the Torres Strait. We recommend that TSRA maintain the salinity monitoring at the northern sites, continuous loggers and remote sensing programmes undertaken through the NESP Project 2.2.1 and to facilitate the development of integrated water quality monitoring tools to maximise the value of these results.

4.8 Gene sequencing in sediment samples

Purpose: Targeted gene sequencing in sediment samples (in collaboration with CSIRO under NESP Project 2.2.2) to investigate whether microbial (bacterial and eukaryotic) and infaunal sedimentary (eukaryotic) biodiversity in the region reflected gradients of sediment properties or contaminants discharged from PNG rivers. Diversity metrics and community composition from each sediment core were compared to the metal/metalloid and organic/physico-chemical analyte profiles obtained from the same sample.

Highlights:

- Advances in genetic sequencing now allow information about entire communities, including macro- and micro-organisms to be captured. Micro-organisms have proven highly sensitive to metal contaminants and provide an additional line of evidence to understand the consequences of environmental change.
- This study collected subtidal sediments from 22 locations throughout the Torres Strait using a sediment corer deployed at between 2 and 20 m depth. Each core was sub-sampled to analyse: 1) the sediment community with targeted gene sequencing; 2) sediment particle sizes; and 3) carbon and nitrogen. Sediment metal concentrations were also measured in NESP Project 2.2.2 (Apte et al. 2018). Bacterial communities were sequenced using the 27f-519r primers for the V1-V3 region of the 16S rRNA gene. Eukaryotic communities were sequenced with the 1391f-EukBr primers for the V9 region of the 18S rRNA gene. All sequencing was undertaken at the Ramaciotti Centre.
- Analysis of the sediment community identified nine distinct biological assemblages for the bacteria and seven distinct biological assemblages for the eukaryotes. Community diversity varied spatially (between locations) with no clear patterns in the Torres Strait
- Changes in sediment community composition were related to several potential environmental drivers, including sediment fines content and metals. However, the amount of variation explained was low. Metal concentrations were generally low (see Apte et al. 2018) suggesting that the correlations are unlikely to be causative, the results might also suggest that the microbial assemblage is particularly sensitive. Alternatively, several of the metals found in this study (e.g. Cu, Zn, Fe) are essential micronutrients at low levels and are potentially associated with other nutrients such as sulphur, CO₃ and organic ligands, which could be selectively utilized by some microbes. Therefore, observed relationships could be a consequence of nutritional preferences rather than toxicological sensitivity.
- Spatial changes at most locations appeared to correlate best with a decreasing gradient of fines content from east to west. Metal concentrations best explained the

community composition at the three sampling locations closest to the PNG mainland. Here, community change may be related to terrestrial run-off. Sediments in the NW of the Torres Strait had higher fractions of sand, whereas samples towards the centre of the region showed higher percentages of fine sediment. Analyses of grain sizes within the fine fraction suggested differences from east to west, as well as potential relationship to island and Fly River influences. These particle size differences may have affected background metals as well as the quality of sediment interstices, the suitability of particles for ingestion by various faunal species and their desirability as substrate for various microbes.

- This study opportunistically collected seagrass samples from three locations in the Torres Strait. Samples were swabbed for gene analysis and analysed for tissue metal concentrations. The gene samples remain in storage, but the metal results of this component are included in Section 4.9 and Appendix 3.

4.8.1 Overview of methods and results

Macroinvertebrates have traditionally been used for the monitoring of environmental impacts in water and sediment (e.g. Smith et al. 2011; Sun et al. 2012). However, modern technologies now allow the use of genetic material for monitoring and development of biotic indices of entire communities, including microorganisms (Baird and Hajibabaei 2012; Chariton et al. 2015; Aylagas et al. 2017). Microorganisms are highly sensitive to environmental change (Sun et al. 2012; Birrer et al. 2018), and thus present an optimal basis for early detection of impacts or detection of subtle impacts. Sampling of sediment communities using targeted gene analysis was used here to investigate ecological change in relation to environmental differences in the Torres Strait and to explore potential relationships to metals and sediments in particular.

Profiles of relative metal concentrations within and across sample matrices can be used to identify whether concentrations reflect a gradient, which declines with distance from the Fly River mouth, or whether concentrations are more strongly related to background factors such as location, grain size and organic content. By integrating the targeted gene analysis directly with the metal profile data obtained from the other sample matrices, a biodiversity assessment can be developed that relates directly to environmental contaminant profiles. An order of magnitude more species are identified than using conventional sediment sorting. This is partly because the genetic sequencing captures data from microscopic organisms that are typically too difficult to consider, but also because many macrofaunal species will not have been formally described by taxonomists. The sequence separation allows macrofaunal OTUs (Operational Taxonomic Units) to be derived.

While macrofauna of the Torres Strait sediments have been surveyed previously (Long and Poiner 1994), the microbial biodiversity in the Torres Strait or adjacent regions has, to the best of our knowledge, not yet been investigated. Hence, this work provides the first description of the entire (bacterial and eukaryotic) sediment communities in the Torres Strait, and provides background information for future studies in this and adjacent regions.

The 22 sampling sites were arranged along a grid to maximise spatial coverage (Table 4-8 and Figure 4-35) and were adjusted on site to 2 to 20 m depth and sandy/fine sediment (rather than coral or shell) using the depth sounder on the boat. Sediment samples were collected from 2 to 16 October 2016. For DNA extraction, five replicate 50 mL sediment samples were

collected from each site from the top 2 cm in sterile containers, directly from the seafloor by divers or by corer from a boat (in cases where wildlife presented a risk to divers). Furthermore, 30 mL samples were taken to determine the level of carbon and nitrogen in the sediment. All samples were frozen within minutes of sampling and stored in the dark.

Sediment grain size, carbon/nitrogen analyses were measured on one replicate per sampling location. Grain size measurements were made by wet sieving through stainless steel sieves. Grain sizes were divided into three size classes: gravel (> 2 mm), sand (< 2 mm and > 63 µm) and fines (< 63 µm). Samples were then oven dried (24 h at 60 °C) and fine fractions quantified with a Malvern Particle Size Analyzer.

Samples were kept frozen and in the dark until C/N analyses. Sediments were oven dried at 60° C for 48 h, then homogenised using a mortar and pestle. Between 5 and 20 mg of dried and ground sediments were weighed into tin capsules. Samples were analysed using an isotope-ratio mass spectrometry (IRMS) with a Delta V Advantage IRMS located at the Mark Wainwright Analytical Centre, UNSW. Duplicates were run every 10 samples and final results compared to certified reference materials National Institute of Standards and Technology (NIST)-40 and NIST-41 in every run.

DNA was extracted from 1 gram of homogenised sediment using the DNeasy PowerSoil Kit (Qiagen) according to the manufacturer's instructions. DNA was stored at -20°C until subsequent processing.

Bacterial communities were sequenced using the 27f (5'-3': AGAGTTTGATCMTGGCTCAG; Lane 1991) and 519r (5'-3': GWATTACCGCGGCKGCTG; Turner et al. 1999) primers for the V1–V3 region of the 16S rRNA gene. This region of the 16S rRNA gene is frequently used to study bacterial communities in soils, sediments and biofilms (Bissett et al. 2016; Lawes et al. 2016; Birrer et al. 2018). Eukaryotic communities were sequenced using the 1391f (5'-3': GTACACACCGCCCGTC) and EukBr (5'-3': TGATCCTTCTGCAGGTTACCTAC) primer set (Amaral-Zettler et al. 2009) for the V9 region of the 18S rRNA gene. Sequencing was performed across one lane each at the Ramaciotti Centre for Genomics on the Illumina MiSeq platform. Paired-end sequencing of 2 × 300 base pair (bp) and 2 × 150 bp were used for 16S and 18S, respectively.

Table 4-8: Sampling location details including locations (Figure 4-32), coordinates, depth and sampling date.

Location	Latitude	Longitude	Depth (m)	Date sampled
1	09°46.821'S	142°58.033'E	7.3	3/10/2016
G	09°36.026'S	143°05.000'E	9.2	3/10/2016
2	09°30.748'S	143°17.117'E	16.7	4/10/2016
J	09°30.081'S	143°31.274'E	11.8	4/10/2016
3	09°08.362'S	143°52.427'E	13.1	5/10/2016
K	09°21.612'S	143°51.594'E	5.9	5/10/2016
I	09°26.074'S	143°46.709'E	7.3	5/10/2016
M	09°39.419'S	143°45.041'E	5.1	6/10/2016

N	09°34.670'S	143°46.275'E	11.4	6/10/2016
X	09°36.607'S	143°35.828'E	7.6	7/10/2016
O	09°44.735'S	143°24.281'E	21.8	8/10/2016
P	09°44.370'S	143°26.334'E	15.3	9/10/2016
E	09°28.940'S	143°05.835'E	4.3	10/10/2016
8	09°22.466'S	142°36.312'E	-	11/10/2016
A	09°20.909'S	142°44.401'E	-	11/10/2016
B	09°34.258'S	142°36.772'E	11.7	12/10/2016
9	09°45.389'S	142°37.543'E	0.5	13/10/2016
C	09°50.051'S	142°32.890'E	16.3	13/10/2016
10	10°09.438'S	142°30.028'E	11.9	13/10/2016
F	10°08.567'S	142°48.789'E	20	14/10/2016
11	10°31.180'S	143°05.465'E	13.5	15/10/2016
D	10°27.469'S	142°26.181'E	13	16/10/2016

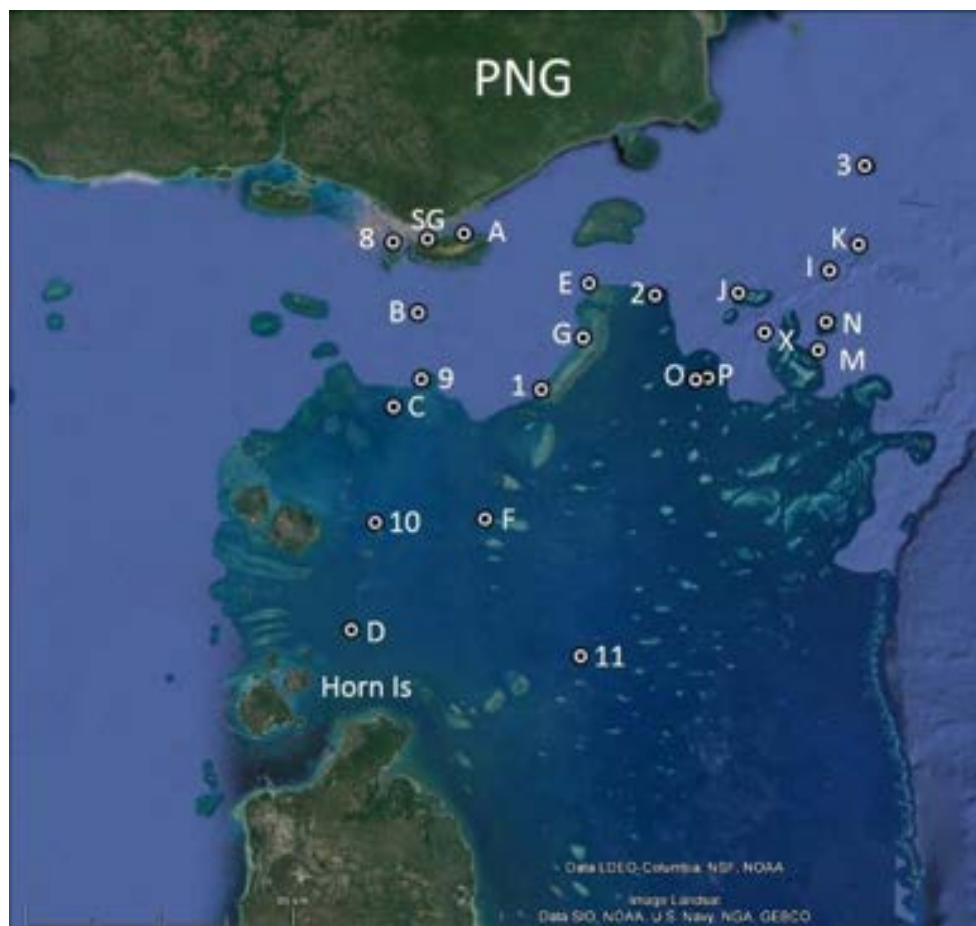


Figure 4-35: Map of sediment sampling locations in the Torres Strait (Table 4-8).

The amplicon sequence data was processed using genome-wide haplotyping (available at <https://doi.org/10.4225/08/59f98560eba25>), an in-house amplicon clustering and classification pipeline built around tools from USearch (Edgar 2013) combined with locally-written tools for demultiplexing, trimming and generating operational taxonomic unit (OTU) tables.

All data analyses were performed in R version 3.3.2 (R Core Team 2016) and the *DESeq* (Anders and Huber 2010), *vegan* (Oksanen et al. 2016), *phyloseq* (McMurdie and Holmes 2014), and *geosphere* (Robert et al. 2017) R packages. Package *ggplot2* (Wickham 2009) was used to generate all plots. Differences were considered as statistically significant if $p \leq 0.05$. Full details are provided in Appendix 3.

Sediment characteristics

Sediments from all locations mainly comprised sand and silt (Table 4-9). Only three locations (3, I and J) had a small amount of gravel in them because sites were selected to avoid coral and large shells (Santmire and Leff 2007). Generally, sediments selectively sampled in the northwest of the Torres Strait had higher fractions of sand, whereas samples towards the centre of the region showed higher percentages of fine silty sediment (Figure 4-36). Analyses of grain sizes within the fine fraction suggested differences from east to west, as well as potential relationship to island and Fly River influences. Grain size was not correlated with latitude or longitude ($p > 0.1$), and only marginally correlated with depth ($p = 0.07$, $\text{cor}(\text{fines and depth}) = 0.412$).

Carbon and nitrogen in selectively sampled Torres Strait sediment were highly variable (Table 4.9). $\delta^{15}\text{N}$ values had an average of 6.4, and ranged from 1.5 to 16.9, whereas $\delta^{13}\text{C}$ values had an average of 90.6 and ranged from -1,008 to 1,655. Percentages of ^{15}N and ^{13}C averaged at $< 0.1\%$ and 5.9% , respectively, and ranged between $0.02\text{--}0.08\%$ and $3\text{--}8.5\%$. None of the stable isotope values correlated with latitude, longitude or depth ($p > 0.1$). Therefore, a spatial signature could not be found of pollution from terrestrial sources. Nonetheless, these stable isotope values served as potential explanatory parameters for the community compositions.

Table 4-9: Proportion of sediment from each grain size class at each location. Gravel: > 2 mm, sand: < 2 mm and > 63 µm, fines: < 63 µm.

Zone	Location	Gravel (%)	Sand (%)	Fines (%)
1	2	0	11.5	88.5
1	E	0	32.6	67.4
2	3	0.2	96.9	2.8
2	K	0	100	0
3	8	0	8.4	91.6
3	A	0	61.2	38.7
3	B	0	95.9	4.1
4	1	0	97	2.9
4	9	0	44.1	55.9
4	C	0	24.7	75.3
4	G	0	3.9	96
4	I	0.5	88.4	11.2
4	J	0.5	95	4.5
4	M	0	75.1	24.9
4	N	0	25	75
4	O	0	13.7	86.3
4	P	0	33.3	66.7
4	X	0	71.1	28.9
5	10	0	23.1	76.9
5	11	0	38.8	61.2
5	D	0	84.5	15.5
5	F	0	18.4	81.6

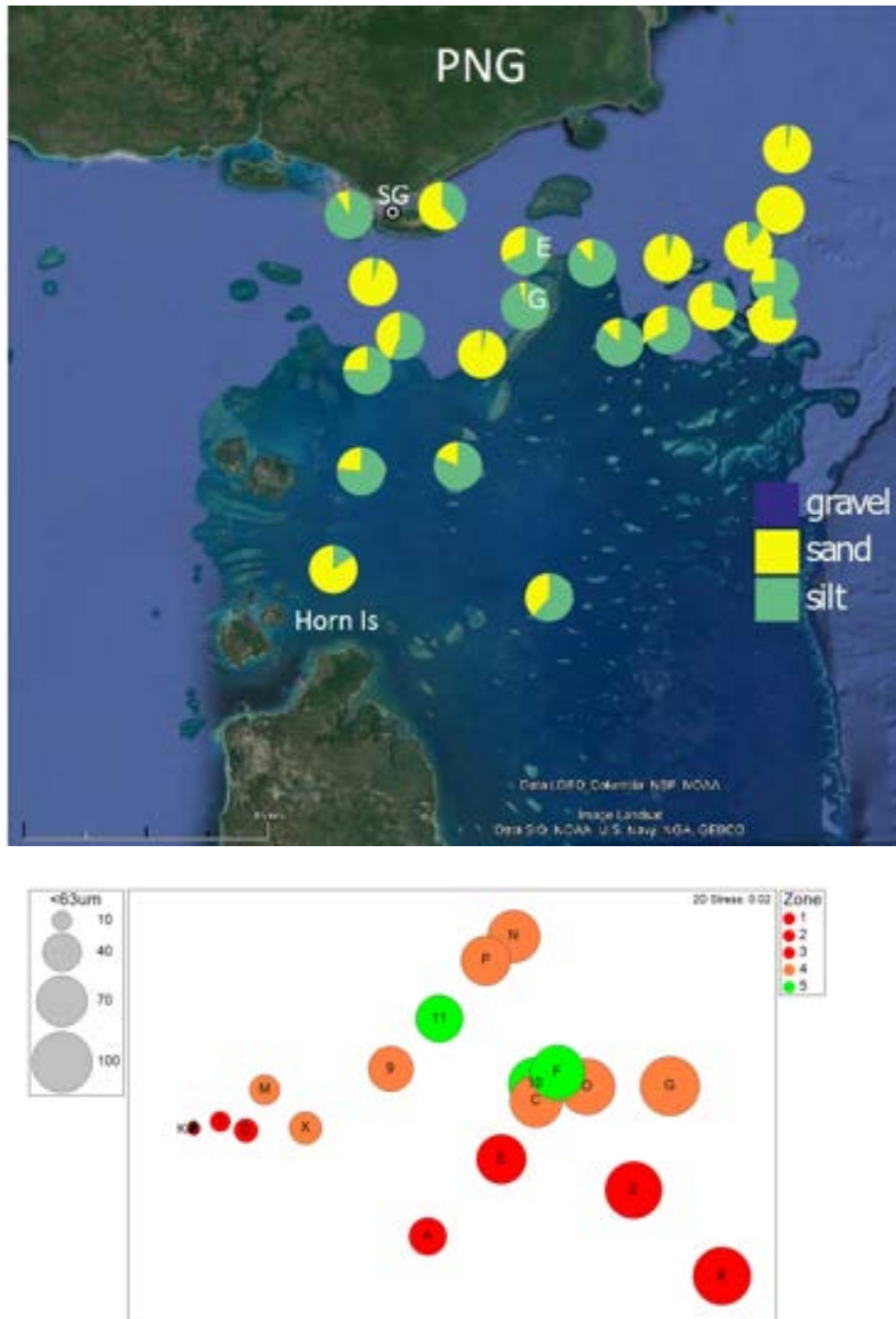


Figure 4-36: (Upper) Spatial distribution of the different sediment grain sizes. Only seagrass leaves were collected from location SG whereas sediment and seagrass leaves were collected from locations E and G. Gravel: > 2 mm, sand: < 2 mm and > 63 µm, fines: < 63 µm. (Lower) Bubble plot of grain size differences among locations in the fines fraction (<63µm). Letters/numbers relate to the locations. Colours relate to the risk of Fly River influence (H = Red, M = Orange, L = Green).

Table 4-10: Carbon and nitrogen results for each location. BDL indicates values below detection limits.

Zone	Location	$\delta^{15}\text{N}$	^{15}N (%)	$\delta^{13}\text{C}$	^{13}C (%)
1	2	10.2	0.02	-553	5.6
1	E	4.4	0.04	319	BDL
2	3	7.8	0.03	-1009	5.1
2	K	2.3	0.02	-240	5.7
3	8	5.8	0.08	645	3
3	A	1.5	0.03	12	3.9
3	B	6.2	0.02	-172	6
4	1	11.5	0.02	686	6.2
4	9	3.3	0.06	-145	5.3
4	C	16.9	0.03	63	BDL
4	G	2.8	0.08	3	6.5
4	I	7.2	0.03	-193	7.6
4	J	7.1	0.02	-169	6.9
4	M	5.7	0.02	-247	5.7
4	N	6.3	0.07	-162	8.5
4	O	BDL	BDL	BDL	BDL
4	P	7.3	0.04	14	BDL
4	X	8.1	0.02	1656	5.4
5	10	7.5	0.03	1378	4.7
5	11	5.5	0.03	38	6.2
5	D	5.2	0.03	43	5.6
5	F	2.8	0.06	-66	8.1

Sediment community

Environmental DNA was extracted and quantified using a Nanodrop (COMP). DNA concentrations ranged between 2.4 and 39.2 ng/ μL , with an average of 12.8 ng/ μL . DNA extractions were successful and sufficient DNA was obtained for amplicon sequencing (minimum requirement: 15–20 μL with a minimum DNA concentration of 5–10 ng/ μL). Successful amplification of the ribosomal genes in all samples was confirmed by visible bands in an agarose gel post polymerase chain reaction (PCR) amplification and gel electrophoresis using the 16S primer set (27f/519r), prior to sample submission for sequencing. Non-specific amplifications, other genes apart from 16S rRNA genes, or primer-dimers (where primers bind to each other), were not observed.

Biological assemblages at sampling locations

Based on Bray-Curtis dissimilarities of the bacterial community composition and an arbitrary cut-off of 0.2, we determined nine distinct biological assemblages (Figure 4-37 and Figure 4-38). Six of these groups only comprised one sampling location, suggesting that these locations are highly distinct from others. The 10 most abundant orders in each biological assemblage are shown in Table 4-11.

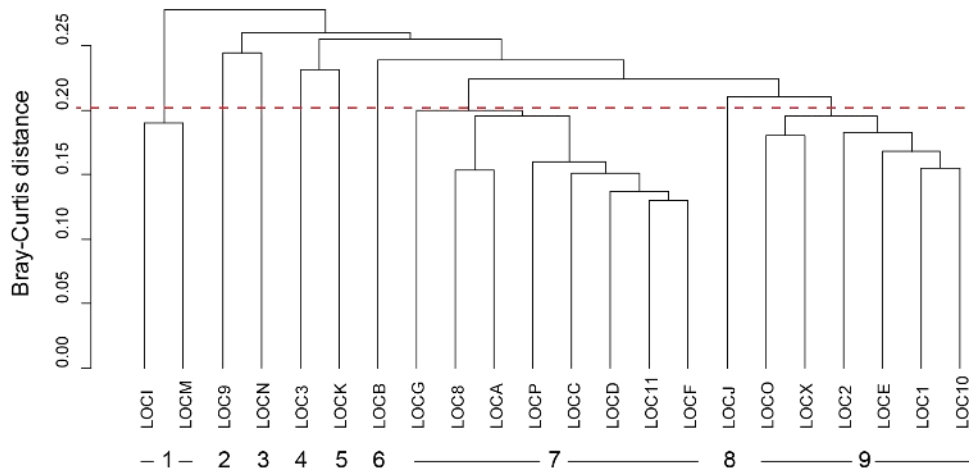


Figure 4-37: Dendrogram of all locations based on bacterial community composition (median of all replicates). The Bray-Curtis dissimilarity index was used in combination with the unweighted pair-group method with arithmetic mean (UPGMA). Based on a cut-off of 0.2 (red dotted line), the locations were grouped into nine biological assemblages.

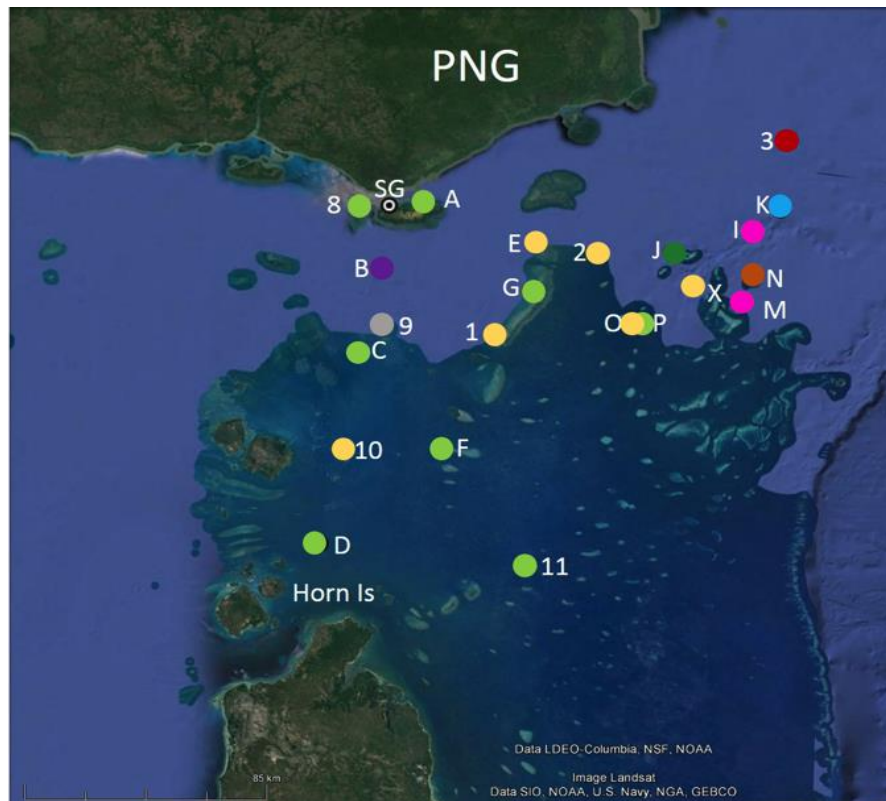


Figure 4-38: Map depicting the different biological assemblages based on bacterial community composition. Each colour represents a different group.

Table 4-11: Ten most abundant orders in each bacterial biological assemblage.

Biological assemblage 1	Biological assemblage 2	Biological assemblage 3
Cyanobacteria Family I	Cellulomonadaceae	Clostridiales_Incertae Sedis III
Simkaniaceae	Acidimicrobiaceae	Nitrosomonadaceae
Desulfohalobiaceae	Litoricolaceae	Thioalkalispiraceae
Hyphomicrobiaceae	Sporichthyaceae	Syntrophobacteraceae
Acidimicrobiaceae	Sinobacteraceae	Fibrobacteraceae
Cyanobacteria Family XII	Peptostreptococcaceae	Bradymonadaceae
Thioalkalispiraceae	Euzebyaceae	Acidimicrobiaceae
Peptostreptococcaceae	Thioalkalispiraceae	Sinobacteraceae
Alteromonadaceae	Bradymonadaceae	Peptostreptococcaceae
Acidimicrobinae_incertae_sedis	Clostridiaceae 1	Acidimicrobinae_incertae_sedis
Biological assemblage 4	Biological assemblage 5	Biological assemblage 6
Nitrosomonadaceae	Oceanospirillales_incertae_sedis	Syntrophorhabdaceae
Fibrobacteraceae	Cohaesibacteraceae	Acidimicrobinae_incertae_sedis
Bradymonadaceae	Acidimicrobinae_incertae_sedis	Solirubrobacteraceae
Peptostreptococcaceae	Thioalkalispiraceae	Desulfohalobiaceae
Alteromonadales_incertae_sedis	Nitrosomonadaceae	Gaiellaceae
Paenibacillaceae	Acidimicrobiaceae	Pseudomonadales_incertae_sedis
Acidimicrobinae_incertae_sedis	Psychromonadaceae	Nitrospiraceae
Thioalkalispiraceae	Clostridiales_Incertae Sedis III	Hyphomicrobiaceae
Cohaesibacteraceae	Fibrobacteraceae	Cyanobacteria Family III
Acidimicrobiaceae	Trueperaceae	SAR11
Biological assemblage 7	Biological assemblage 8	Biological assemblage 9
Acidimicrobinae_incertae_sedis	Simkaniaceae	Acidimicrobinae_incertae_sedis
Syntrophobacteraceae	Nitrosomonadaceae	Acidimicrobiaceae
Sinobacteraceae	Rhizobiales_incertae_sedis	Rhizobiales_incertae_sedis
Rhizobiales_incertae_sedis	Desulfohalobiaceae	Sinobacteraceae
Acidimicrobiaceae	Acidimicrobinae_incertae_sedis	Thioalkalispiraceae
Gaiellaceae	Rubritaleaceae	Hyphomicrobiaceae
Hyphomicrobiaceae	Hyphomicrobiaceae	Bradymonadaceae
Thioalkalispiraceae	Syntrophorhabdaceae	Peptostreptococcaceae
Bradymonadaceae	Gaiellaceae	Kordiimonadaceae
Desulfuromonadaceae	Acidimicrobiaceae	Gaiellaceae

Based on Bray-Curtis dissimilarities of the eukaryotic community composition and an arbitrary cut-off of 0.2, we determined seven distinct biological assemblages (Figure 4-39 and Figure 4-40). Four of these groups only comprised one sampling location, whereas one group comprised 14 locations. This suggests that the majority of samples are similar in eukaryotic compositions and, those which are not, are unique. The 10 most abundant orders in each biological assemblage are shown in Table 4.12.

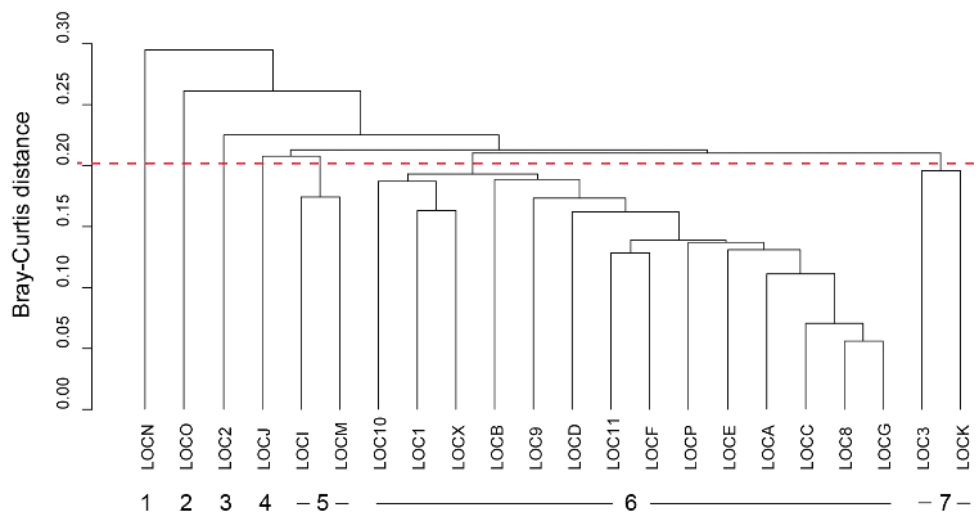


Figure 4-39 Dendrogram of all locations based on eukaryotic community composition (median of all replicates). The Bray-Curtis dissimilarity index was used in combination with the UPGMA. Based on a cut-off of 0.2 (red dotted line), the locations were grouped into seven biological assemblages.

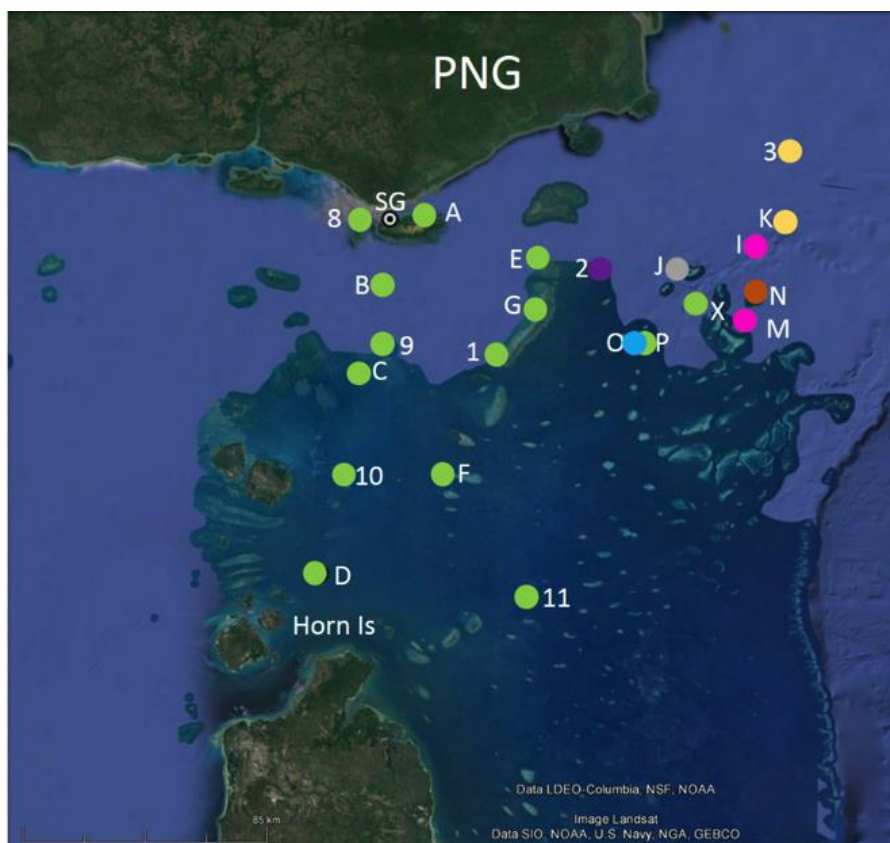


Figure 4-40: Map depicting the different biological assemblages based on eukaryotic community composition. Each colour represents a different group.

Table 4-12: Ten most abundant orders in each eukaryotic biological assemblage.

Biological assemblage 1	Biological assemblage 2	Biological assemblage 3	
Diplogasterida	Nuculanoida	Cucurbitales	
Cucurbitales	Cucurbitales	Salpida	
Trichocephalida	Arhynchobdellida	Ellobiopsida	
Asterocladales	Ellobiopsida	Strigeidida	
Endogonales	Neoelectales	Lingulida	
Acrochaetiales	Aspidosiphonidormes	Neocallimastigales	
Squamata	Valvatida	Parachela	
Hemiaulales	Hemiaulales	Scorpiones	
Solemyoida	Scleractinia	Hemiaulales	
Neocallimastigales	Neocallimastigales	Gonorynchiformes	
Biological assemblage 4	Biological assemblage 5	Biological assemblage 6	Biological assemblage 7
Cucurbitales	Cucurbitales	Cucurbitales	Cephalobaenida
Endeostigmata	Asterocladales	Trichocephalida	Cucurbitales
Pennatulacea	Acrochaetiales	Endeostigmata	Reticulosphaerales
Nanaloricida	Monoblepharidales	Ostreoida	Trichocephalida
Triganglionata	Gelidiales	Nuculanoida	Schizocladiales
Acrochaetiales	Fucales	Cymatosirales	Peltigerales
Euryalida	Ectocarpales	Neocallimastigales	Ostreoida
Acrosymphytales	Phaeophilales	Caryophyllales	Neocallimastigales
Gigartinales	Pedinomonadales	Trebouxiales	Collodaria
Lobulomycetales	Chlorodendrales	Paraliales	Climacospheniales

Sequence characteristics for the 16S dataset

A total of 7,853,187 high-quality non-chimeric bacterial sequences were obtained with an average of 73,390 sequences per sample (min: 33,020, max: 241,500). Sequences were assigned to 48,118 OTUs, of which 26,631 and 19,747 were classified down to phylum and species level, respectively. The calculated rarefaction curve (Figure 4-41) shows saturation.

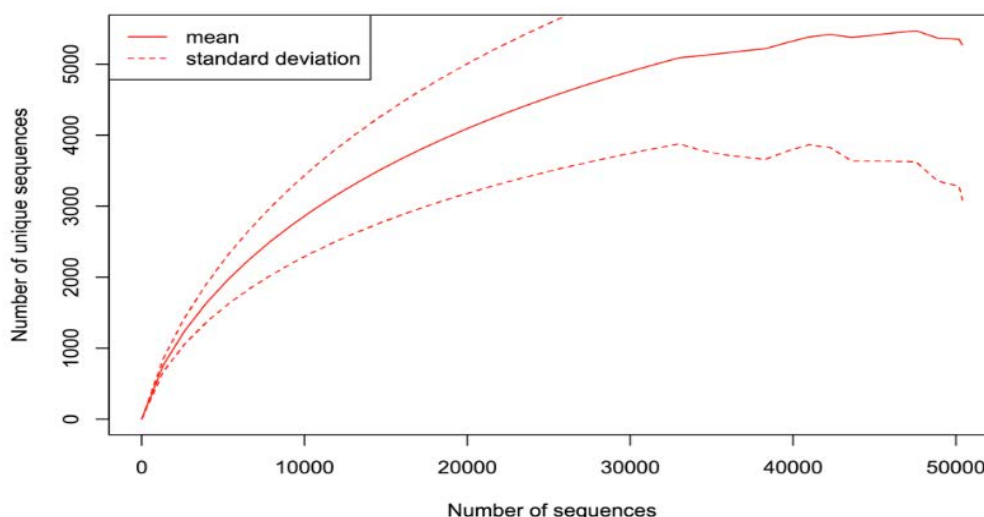


Figure 4-41: Rarefaction curve displaying the recoveries from the extraction and PCR process.

Bacterial diversity

Community diversity (Shannon index) and richness (number of observed species) were calculated (Table 4-12 and Figure 4-42). Diversity and richness for bacterial communities per sample ranged from 10.17 to 10.61 and from 510 to 10,350, respectively. An analysis of variance (ANOVA) of both diversity and richness showed that these parameters changed significantly between sampling locations ($p < 0.0001$).

Table 4-13: Bacterial (16S) and eukaryotic (18S) richness and diversity (mean $\pm$ sd, n=5) in the sediment samples at each location.

Zone	Location	16S		18S	
		Shannon Index	Diversity	Shannon Index	Diversity
1	2	10.28 $\pm$ 0.01	6838 $\pm$ 1063	9.92 $\pm$ 3481	0.04 $\pm$ 432
1	E	10.29 $\pm$ 0.01	7053 $\pm$ 536	10.02 $\pm$ 1399	0.12 $\pm$ 484
2	3	10.24 $\pm$ 0.02	8165 $\pm$ 798	9.95 $\pm$ 2360	0.09 $\pm$ 641
2	K	10.28 $\pm$ 0.01	6258 $\pm$ 332	10.00 $\pm$ 3065	0.03 $\pm$ 265
3	8	10.35 $\pm$ 0.05	5026 $\pm$ 2044	10.07 $\pm$ 734	0.20 $\pm$ 576
3	A	10.29 $\pm$ 0.02	6328 $\pm$ 837	9.94 $\pm$ 1318	0.05 $\pm$ 233
3	B	10.28 $\pm$ 0.01	5909 $\pm$ 328	9.92 $\pm$ 2125	0.09 $\pm$ 521
4	1	10.27 $\pm$ 0.02	7375 $\pm$ 644	9.97 $\pm$ 3476	0.05 $\pm$ 931
4	9	10.23 $\pm$ 0.02	4585 $\pm$ 566	9.99 $\pm$ 1414	0.06 $\pm$ 231
4	C	10.33 $\pm$ 0.03	4104 $\pm$ 1509	10.16 $\pm$ 735	0.12 $\pm$ 494
4	G	10.47 $\pm$ 0.13	2477 $\pm$ 2429	10.19 $\pm$ 441	0.16 $\pm$ 422
4	I	10.21 $\pm$ 0.01	6250 $\pm$ 1174	10.02 $\pm$ 2332	0.09 $\pm$ 848
4	J	10.25 $\pm$ 0.01	6414 $\pm$ 602	9.93 $\pm$ 3087	0.08 $\pm$ 475
4	M	10.20 $\pm$ 0.02	6407 $\pm$ 962	9.90 $\pm$ 2562	0.08 $\pm$ 573
4	N	10.25 $\pm$ 0.01	7584 $\pm$ 194	9.84 $\pm$ 4472	0.05 $\pm$ 214
4	O	10.29 $\pm$ 0.02	9615 $\pm$ 773	9.76 $\pm$ 3337	0.02 $\pm$ 478
4	P	10.31 $\pm$ 0.01	6107 $\pm$ 518	9.96 $\pm$ 1717	0.07 $\pm$ 212
4	X	10.28 $\pm$ 0.02	9067 $\pm$ 1187	9.96 $\pm$ 2952	0.10 $\pm$ 1273
5	10	10.29 $\pm$ 0.01	6443 $\pm$ 365	9.86 $\pm$ 2440	0.02 $\pm$ 573
5	11	10.29 $\pm$ 0.01	6266 $\pm$ 511	9.97 $\pm$ 1568	0.08 $\pm$ 685
5	D	10.29 $\pm$ 0.02	6084 $\pm$ 758	9.87 $\pm$ 1878	0.07 $\pm$ 601
5	F	10.29 $\pm$ 0.03	6179 $\pm$ 372	9.93 $\pm$ 15434	0.08 $\pm$ 584

Bacterial community composition

The following 11 dominant phyla (> 1% of all sequences across all samples) were present in all samples and accounted for over 75% of all bacterial sequences across all samples: *Proteobacteria* (38.15% of all sequences), *Bacteroidetes* (10.96%), *Planctomycetes* (8.98%), *Acidobacteria* (3.8%), *Actinobacteria* (2.63%), *Chloroflexi* (2.63%), *Firmicutes* (1.98%), *Cyanobacteria/Chloroplast* (1.93%), *Verrucomicrobia* (1.76%), *Spirochaetes* (1.42%) and *Parcubacteria* (1.34%). Within the most dominant phylum (*Proteobacteria*), sequences were assigned to seven classes, of which *Deltaproteobacteria* (15%), *Gammaproteobacteria* (10%) and *Alphaproteobacteria* (6.8%) were the most abundant.

Orders that contributed more than 1% to all sequences across all samples were *Planctomycetales* (4.94%), *Flavobacteriales* (3.64%), *Myxococcales* (3.57%), *Desulfobacterales* (3.14%), *Sphingobacteriales* (2.93%), *Chromatiales* (2.16%), *Gammaproteobacteria_incertaines_sedis* (2.06%), *Cytophagales* (1.85%), *Phycisphaerales* (1.51%), *Rhizobiales* (1.5%), *Spirochaetales* (1.42%), *Anaerolineales* (1.41%), *Bdellovibrionales* (1.33%), *Acidimicrobiales* (1.27%), *Rhodospirillales* (1.25%), *Rhodobacterales* (1.19%) and *Clostridiales* (1.12%) (Figure 4-43).

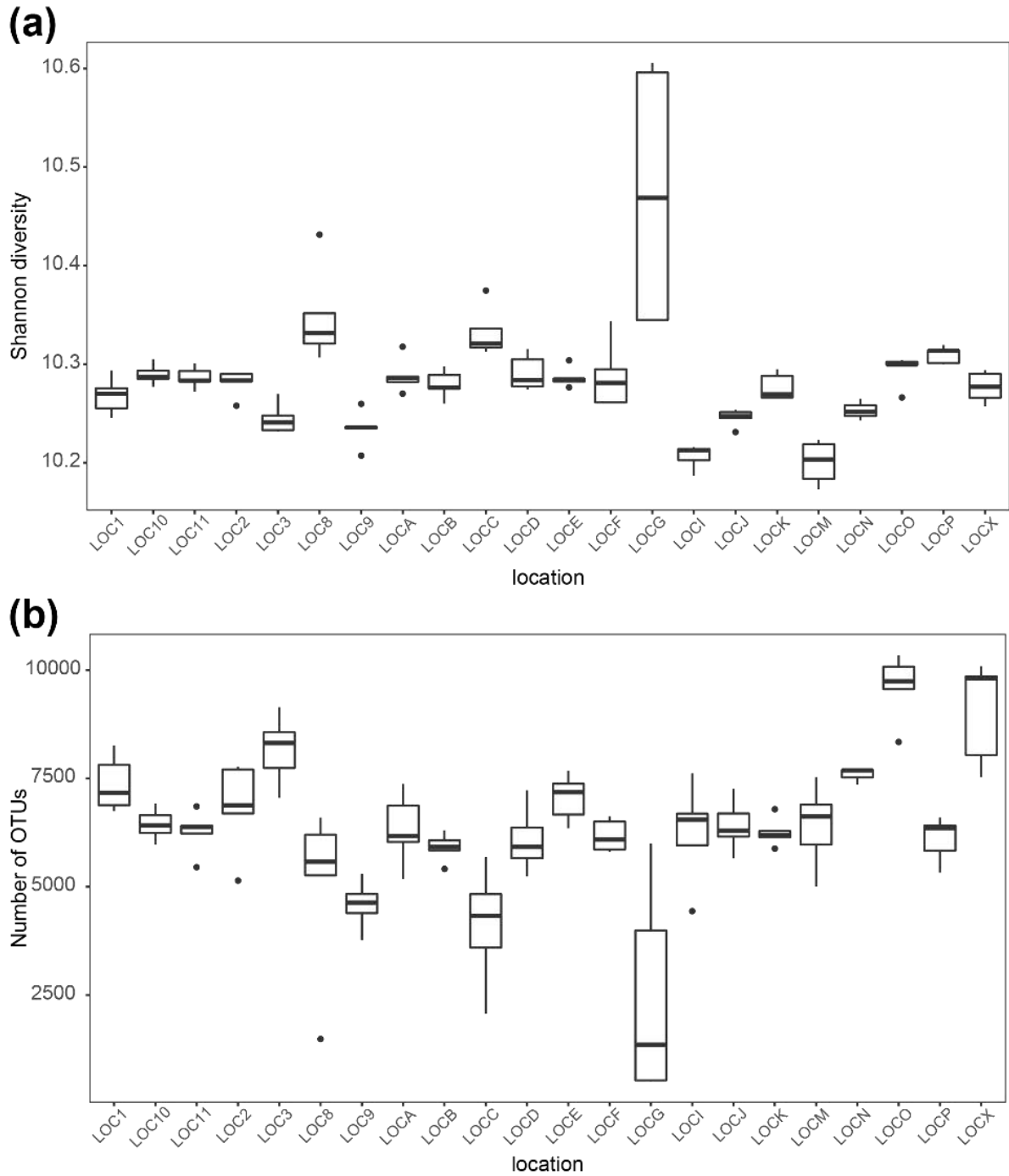


Figure 4-42: Bacterial (a) diversity and (b) richness in the sediment samples at each location.

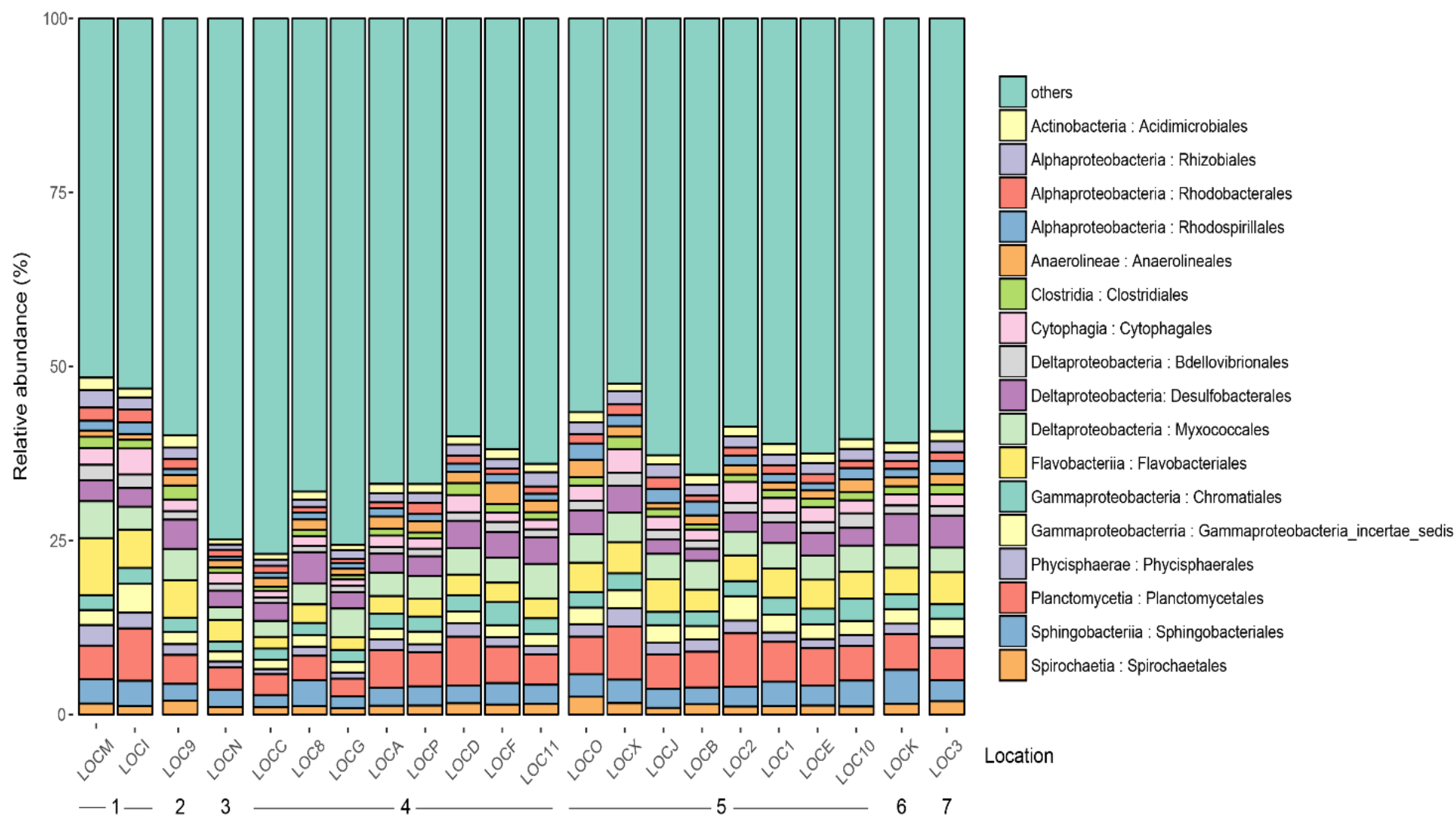


Figure 4-43: Bacterial community structure in all sampling locations. Only abundant bacterial orders with more than 1% abundance across all samples and their respective phyla are shown. Locations across the x-axis are grouped according to biological assemblages.

Generally, samples from locations in the biological assemblages 3 and 4 showed more sequences belonging to non-dominant orders, which suggests a lower evenness of the bacterial community within the sediment. Evenness commonly increases in contaminated sites, by increasing the dominance of certain species (Johnston and Roberts 2009) and shifting microbial resources to survival mode (Schimel et al. 2007). Lower evenness often correlates with higher diversity and is typically seen in more pristine ecosystems.

At the family level, *Phycisphaera* (1.51%), *Parcubacteria_genera_incertae_sedis* (1.34%), *Acidobacteria_Gp22* (1.3%) and *Candidatus Carsonella* (1.19%) contributed more than 1% to the total number of sequences. At the lowest taxonomical level, only one species was dominant: *Planctomycetes brasiliensis*.

Sequence characteristics for the 18S dataset

The composition and diversity of eukaryotic communities in the Torres Strait sediments were assessed by Illumina™ MiSeq sequencing targeting the V9 region of the 18S rRNA gene. This region of the 18S rRNA gene is frequently used to study eukaryotic communities in soils and sediments (Bissett et al. 2016; Birrer et al. 2018).

Paired-end sequencing data were merged, quality-filtered and denoised using USEARCH. Sequences. They were then classified using the SILVA 128 database and non-eukaryotic reads were removed.

A total of 6,514,385 high-quality non-chimeric eukaryotic sequences were obtained with an average of 60,880 sequences per sample (min: 5,536, max: 121,800). Sequences were assigned to 34,814 OTUs, of which 24,487 and 20,035 were classified down to phylum and species level, respectively. The calculated rarefaction curve (Figure 4-44 shows saturation).

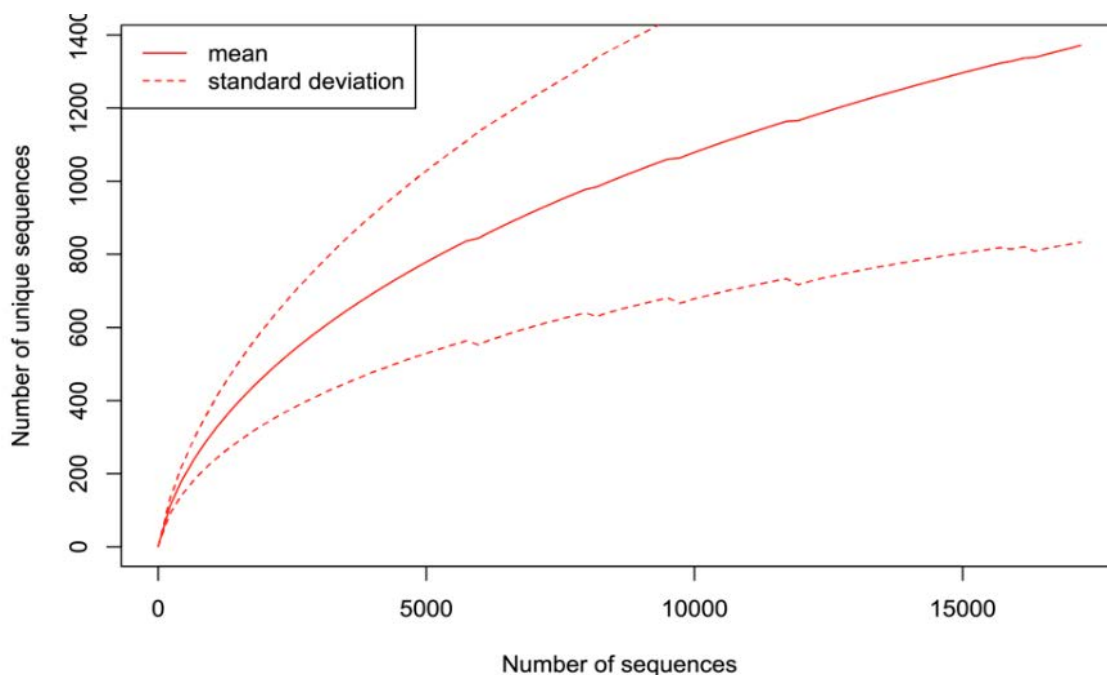


Figure 4-44: Rarefaction curve displaying the recoveries from the extraction and PCR process.

Eukaryotic diversity

Community diversity (Shannon index) and richness (number of observed species) were calculated (Table 4-13 and Figure 4-45). Diversity and richness for eukaryotic communities per sample ranged from 9.73 to 10.3 and from 131 to 4,703, respectively. An ANOVA of both diversity and richness showed that these parameters changed significantly between sampling locations ($p < 0.0001$).

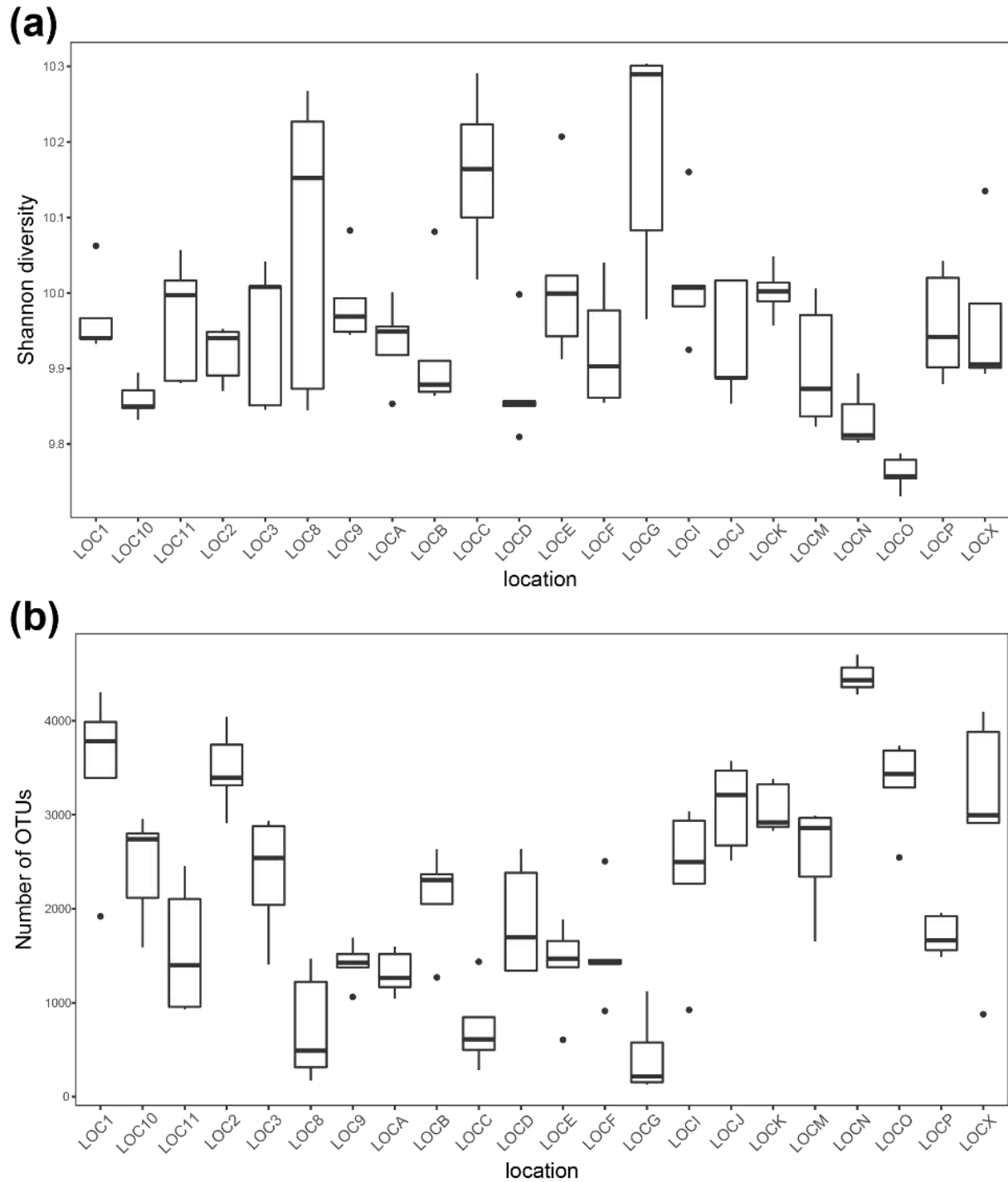


Figure 4-45: Eukaryotic (a) diversity and (b) richness in the sediment samples at each location.

Eukaryotic community composition

The following 17 dominant phyla (> 1% of all sequences across all samples) were present in all samples and accounted for over 58% of all eukaryotic sequences across all samples: *Bacillariophyta* (13.54% of all sequences), *Myzozoa* (11.16%), *Cercozoa* (6.71%), *Nematoda* (4.44%), *Foraminifera* (3.91%), *Arthropoda* (3.72%), *Euglenozoa* (3.64%), *Apicomplexa* (3.45%), *Bigyra* (2.48%), *Annelida* (1.79%), *Platyhelminthes* (1.54%), *Ciliophora* (1.49%), *Xenacoelomorpha* (1.43%), *Ochrophyta* (1.42%), *Chlorophyta* (1.4%), *Amoebozoa* (1.16%) and *Rhodophyta* (1.03%). Within the most dominant phylum (*Bacillariophyta*), sequences were assigned to four classes, of which *Bacillariophyceae* (10.79%) was the most abundant.

Orders that contributed more than 1% to all sequences across all samples were *Naviculales* (4.6%), *Syndiniales* (3.44%), *Rotaliida* (2.72%), *Aconchulinida* (2.58%), *Gymnodiniales* (2.44%), *Thalassiophysales* (2.08%), *Peridiniales* (2.03%), *Monhysterida* (1.82%), *Bacillariales* (1.81%), *Harpacticoida* (1.58%), *Thraustochytriida* (1.56%), *Diplonemida* (1.18%) and *Sphenomonadales* (1.08%) (Figure 4-46).

At the family level, *Vampyrellidae* (2.58%), *Catenulaceae* (2.08%), *Gymnodiniaceae* (2.03%), *Naviculaceae* (2.01%), *Bacillariaceae* (1.81%), *Thraustochytriaceae* (1.56%) and *Sphenomonadidae* (1.08%) contributed more than 1% to the total number of sequences. At the lowest taxonomical level, only the following uncultured species were dominant: uncultured eukaryote (15.01%), uncultured marine eukaryote (2.97%) and uncultured stramenopile (1.31%).

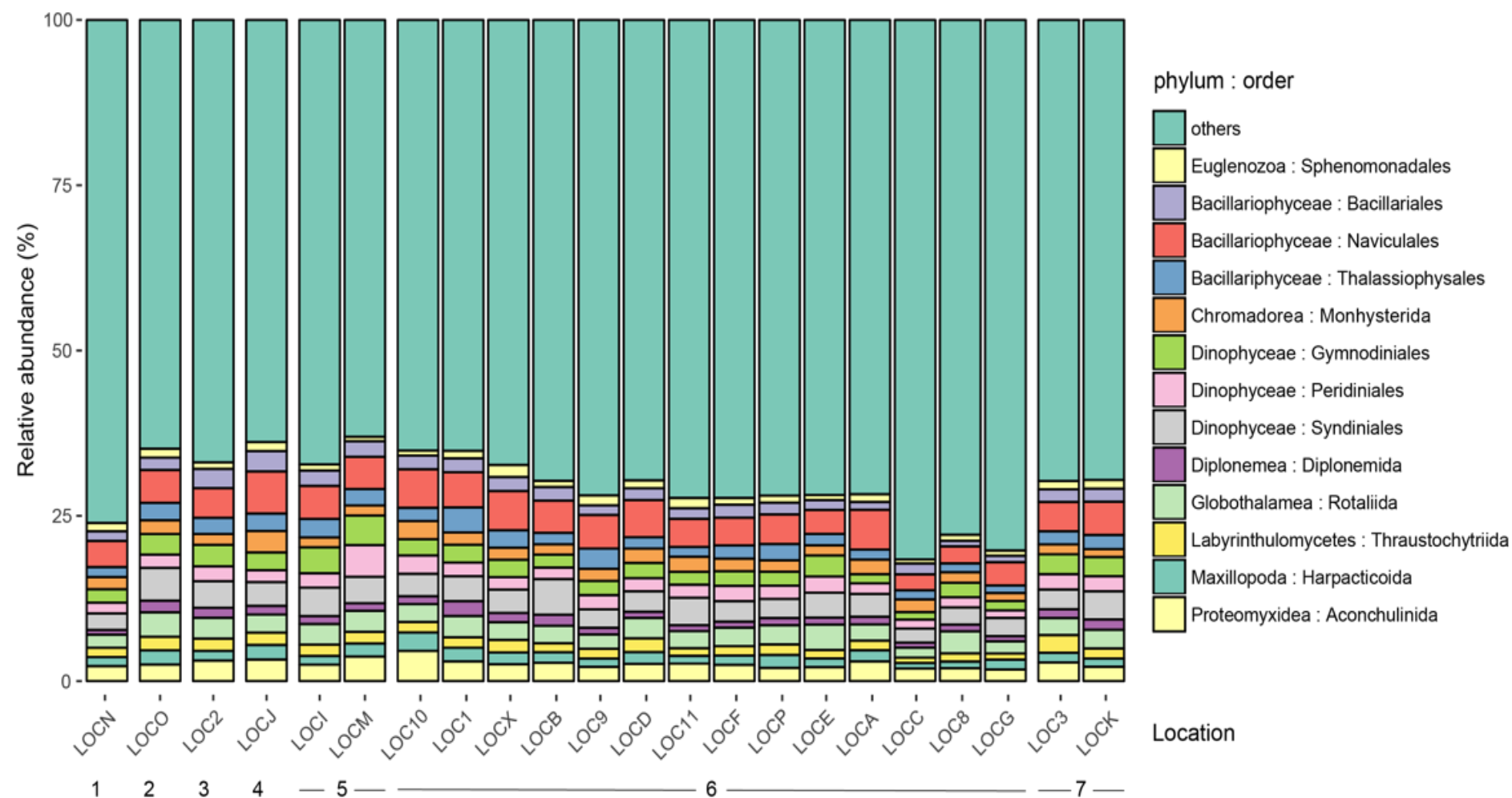


Figure 4-46: Eukaryotic community structure in all sampling locations. Only abundant eukaryotic orders with more than 1% abundance across all samples and their respective phyla are shown. Locations across the x-axis are grouped according to biological assemblages.

The following taxa represent major groups of benthic infauna: *Nematoda*, *Polychaeta*, *Crustacea*, *Mollusca*, *Platyhelminthes* and *Echinodermata*. The majority of these were also identified and measured in the nearby Gulf of Carpentaria (Long and Poiner 1994). Therefore, we decided to take a closer look at the abundances of these in our samples. Relative abundances of these taxa of interest per sample can be seen in Figure 4-47.

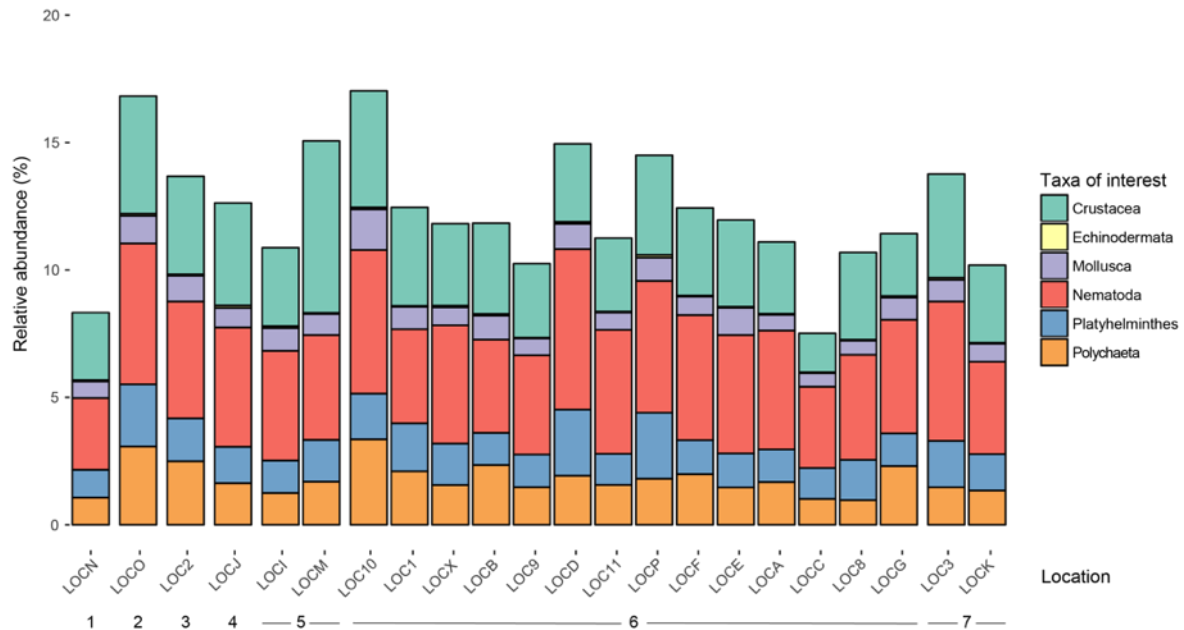


Figure 4-47: Relative abundance of eukaryotic taxa of interest in all sampling locations. Locations across the x-axis are grouped according to biological assemblages.

The order of abundance within the Torres Strait sediment of the taxa of interest was: *Nematoda* (4.44%) > *Crustacea* (3.43%) > *Polychaeta* (1.74%) > *Platyhelminthes* (1.54%) > *Mollusca* (0.82%) > *Echinodermata* (0.055%). This order of abundance has some similarities to what Long and Poiner (1994) found in the Gulf of Carpentaria, i.e. trends in the larger macrofauna. However, Long and Poiner's (1994) survey was conducted by manual sorting and identification of macrofauna, therefore some of the smaller taxa (e.g. *Nematoda* and *Platyhelminthes*) may not have been detected in their work. With modern molecular techniques, we are now able to detect even such small eukaryotes (and much smaller even), which is a major advantage of the approach used in our survey of the Torres Strait sediment.

Within the phylum *Nematoda*, the classes *Chromadorea* and *Enoplea* consisted of 3.81% and 0.63% of sequences across all samples, respectively. Crustaceans consisted of three classes: *Maxillopoda* (2.6%), *Ostracoda* (0.69%) and *Manacostraca* (0.13%). The three most abundant polychaete classes (total of 11 classes) were *Phyllodocida* (0.77%), *Spionida* (0.23%) and *Terebellida* (0.17%). *Platyhelminthes* consisted of four classes: *Turbellaria* (1.38%), *Trematoda* (0.06%), *Monogenea* (0.05%) and *Cestoda* (0.04%). Within the phylum *Mollusca*, sequences were assigned to the following four classes: *Gastropoda* (0.42%), *Bivalvia* (0.039%), *Polyplacophora* (0.02%) and *Scaphopoda* (0.003%). *Echinodermata*, which was the least abundant among the taxa of interest, was divided into five classes: *Ophiuroidea* (0.027%), *Holothuroidea* (0.012%), *Echinoidea* (0.008%), *Crinoidea* (0.005%) and *Asteroidea* (0.004%).

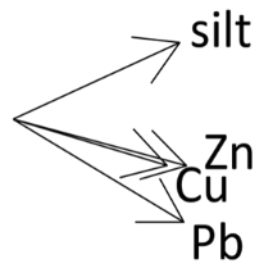
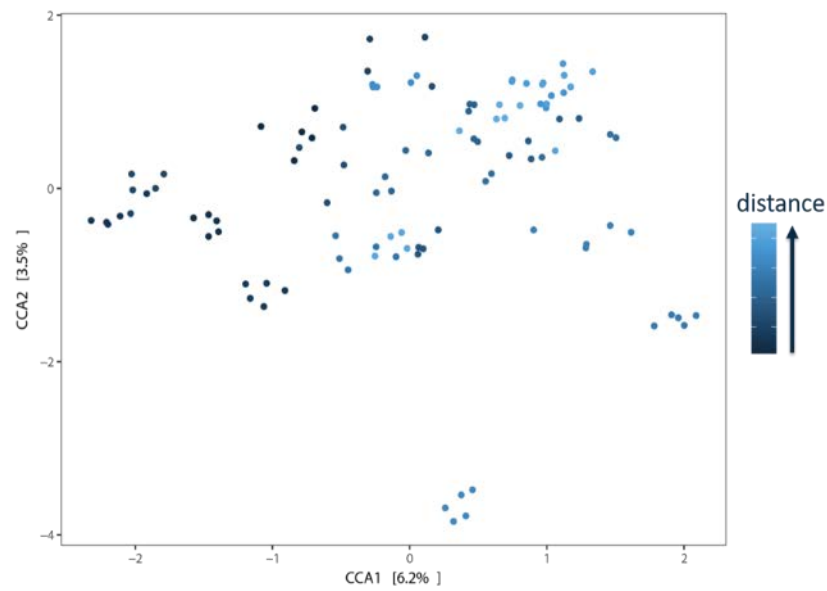
Exploring relationships between measured environmental variables and sediment community composition

The relationship between all environmental variables and the bacterial and eukaryotic community composition were tested using permutational multivariate analysis of variance (PERMANOVA) with distance matrices. All measured environmental variables were significantly related to the bacterial and eukaryotic community, but each explained a low amount of the variation: 2.2–7.7% and 2.7–6.6%, respectively (Table 4-14). There also remained some variation in the ecological datasets that could not be explained by the measured variables (16.4% in the bacteria and 31.2% in the eukaryotes) suggesting that additional environmental factors could influence their distribution beyond what was measured here.

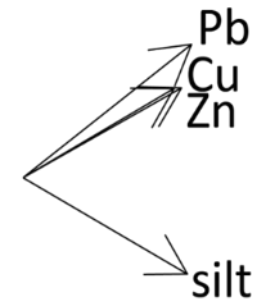
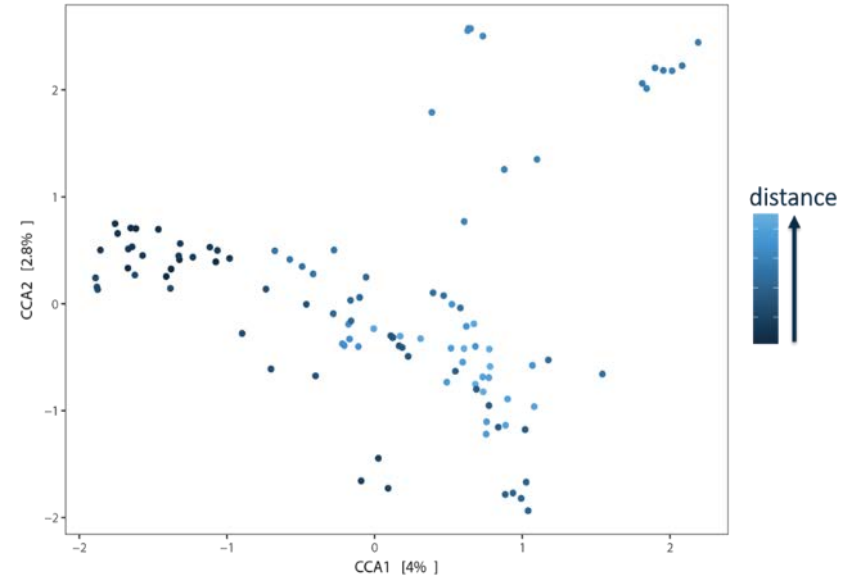
Table 4-14: Results from the PERMANOVAs, showing which environmental variables are related to the bacterial community composition (p -value ≤ 0.05) and to what extent (R^2). Significant p -values are highlighted in bold.

Environmental parameter	16S		18S		Comments
	R^2 (%)	p -Value	R^2 (%)	p -value	
Fines	7.2	0.001	4.8	0.001	
Latitude	5.3	0.001	4.9	0.001	
Longitude	7.4	0.001	6.4	0.001	
Distance from location 3	7.5	0.001	6.6	0.001	
Depth	6.6	0.001	5.9	0.001	No info for locations 8 and A
Co	6.8	0.001	5.2	0.001	
Cu	6	0.001	4.8	0.001	
Ni	7	0.001	5.3	0.001	
Pb	7.7	0.001	5.7	0.001	
Zn	6.5	0.001	5.3	0.001	
$\delta^{15}\text{N}$	2.2	0.003	2.7	0.001	No info for location O
^{15}N (%)	7.5	0.001	5.2	0.001	No info for location O
$\delta^{13}\text{C}$	2.9	0.002	2.8	0.001	No info for location O
^{13}C (%)	3	0.002	3.2	0.001	No info for location O

To test for a potential relationship between indicators of the Fly River plume and the sediment community, the relationship of the sediment fines fraction and metal with the eukaryotic community structure was visualised with canonical correspondence analysis (CCA) based on Bray-Curtis distances. Fines and metals were used here because these were expected to increase in locations influenced by the Fly River discharge. Samples in the visualisation (CCA) were colour-coded according to their distance to location 3, which was closest to the Fly River mouth (Figure 4-48a and b).



a)



b)

Figure 4-48. CCA plot of the (a) bacterial community composition and (b) eukaryotic community composition based on Bray-Curtis distances. Each dot corresponds to one sample and is colour-coded according to the distance from location 3, where darker dots represent samples closer to the Fly River mouth. Directions of change in which fines and metals are related to the community composition are shown as arrows below the CCA plot. The length of the arrows indicates the relative significance with which these variables explain patterns in community change.

Bacterial (Figure 4-48a) and eukaryotic (Figure 4-48b) community composition of samples changed with distance from the Fly River. The CCA suggests that many of the communities were changing along a gradient of fines content, which generally increases with increasing distance from the Fly River mouth. This spatial variation in sediment grain size has been reported previously and attributed to marine sand being transported westward deep into the Torres Strait from the Coral Sea (Saint-Cast 2008). The CCA also suggests that metal concentrations best explain changes in community composition in Locations A, B and 8 (see Figure 4-48). These are the three sampling locations closest to the PNG mainland and may be influenced by terrestrial run-off.

However, it should be acknowledged that these spatial differences could relate to changes in a number of environmental factors (Table 4-14). Several of the metals found in this study (e.g. Cu, Zn, Fe) are essential micronutrients at low levels and are potentially associated with other nutrients such as sulphur, CO₃ and organic ligands, which could be selectively utilized by some microbes. Therefore, observed relationships could be a consequence of nutritional preferences rather than toxicological sensitivity. Furthermore, analyses of grain sizes within the fine fraction suggested differences from east to west, as well as potential relationship to island and Fly River influences. These particle size differences may have affected background metals as well as the quality of sediment interstices, the suitability of particles for ingestion by various faunal species and their desirability as substrate for various microbes.

The small percentage of variability explained by the CCA axes (6.2% and 3.5%, and 4% and 2.8% for bacteria and eukaryotes, respectively), suggests that changes in community composition are likely to be driven by environmental variables, which we have not measured or tested. In addition, microbial communities are highly variable both temporally and spatially, hence, the changes we see in the Torres Strait communities could also simply be due to natural spatial and temporal variability. Therefore, we could not detect a clear signature of the Fly River discharge in the sediment microbial community at this level of temporal and spatial sampling.

4.8.2 Recommendations

The limited data collected does not show a clear impact of the Fly River plume on the sediment microbial communities.

As the seagrass sampling was done opportunistically, all seagrass meadows were not sampled comprehensively. Further sampling of seagrass meadows would be useful to assess the effects of the Fly River and island influences by investigating contaminant concentrations in seagrasses and seagrass health (by measuring the associated microbial communities in seagrass biofilms). Changes in metal concentrations and microbial communities associated with seagrasses could have a direct impact on seagrass meadow health, and thus on seagrass-feeding mammals, which have an important dietary role in the Torres Strait communities.

4.9 Metal analysis in seagrass leaves

Purpose: Pilot study to determine metal concentrations in seagrass leaves from the area suspected to be influenced by Fly River discharge in the northern Torres Strait. Metal concentrations in seagrass may be concentrated up the food chain into green turtles and dugong.

Highlights:

- Metal concentrations in seagrass leaves were opportunistically sampled as a pilot study with the NESP Project 2.2.2 sampling campaign in October 2016 (see Apte et al. 2018). Three locations were sampled: Saibai Island, northern Warrior Reef and Warrior Reef.
- Statistical analysis showed that seven metals (silver, arsenic, cobalt, manganese, nickel, antimony and uranium) differed significantly among the three locations sampled. Metal concentrations in seagrass leaves were generally correlated with metal values from sediments sampled at the same three locations.
- Heavy metal concentrations in the Torres Strait seagrass samples were highest at the inshore sampling location (Saibai Island) compared with offshore sampling sites at Warrior Reef. However, metal concentrations in all samples were low and indicative of an unpolluted environment.
- The Torres Strait results can be compared to results from the GBR where 1) seagrass was collected from Upstart Bay and Cleveland Bay, close to the coast and under the influence of the Burdekin River (and other coastal discharges), and the Howick Island group (well offshore and not under strong influence of river runoff), and 2) where seagrass was collected at a wide range of GBR coastal sites between Cape York and Moreton Bay. The Saibai Island site compares well with the coastal sites influenced by coastal discharges. Coastal discharge may also explain the patterns between Saibai Island and Warrior Reef.
- This small pilot study demonstrates the potential value of this technique in assessing regional differences and provides an important link between seagrass habitats and potential metal exposure for dependent species such as turtle and dugong. Further studies of metal analysis in seagrass leaves are recommended in the northern Torres Strait seagrass meadows to provide additional biologically relevant data on the presence of metals in the region.

4.9.1 Overview of methods and results

Seagrass sampling program description

Seagrass was sampled opportunistically during the survey of sediment and metals in the Torres Strait region at three locations: Warrior Reef (G), northern Warrior Reef (E) and near Saibai Island (SG) (Table 4-15 and Figure 4-49). Seven plants from each location were sampled manually and frozen immediately. The sampled seagrass was either *Halophila* or *Thalassia* species.

Table 4-15: Sampling location details for metal analysis in seagrass leaves including location, coordinates, depth and sampling date.

Location	Latitude	Longitude	Depth (m)	Date
Warrior Reef (G)	09°36.026'S	143°05.000'E	9.2	3/10/2016
northern Warrior Reef (E)	09°28.940'S	143°05.835'E	4.3	10/10/2016
Saibai Island (SG)	09°21.897'S	142°38.201'E	0.5	12/10/2016

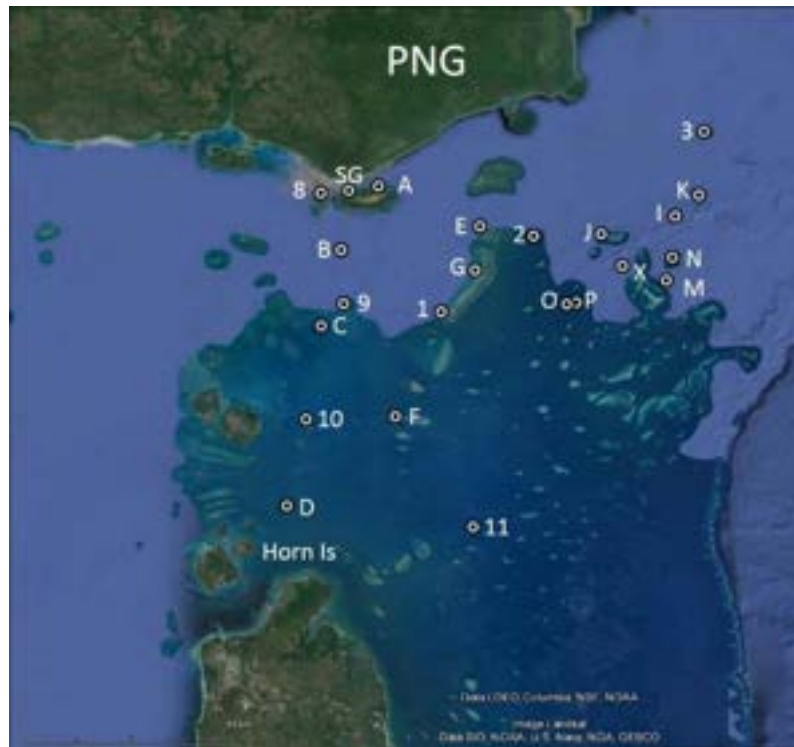


Figure 4-49: Map of the sampling locations for the collection of seagrass leaves (Site G, E and SG only). Note: Map also shows all of the sites sampled for the genomics (Section 4.8) and sediment metal analysis conducted by CSIRO in NESP Project 2.2.2.

Seagrass metal analysis

Sediment was rinsed from the seagrass leaves with Milli-Q® water before a minimum of 200 mg of seagrass was weighed for each sample. The seagrass was then freeze dried using a Christ Alpha 1-2 LD Freeze Dryer. The dried seagrass samples were digested by high pressure/temperature nitric acid microwave digestion using a CEM MARS6 Microwave (in-house Method C-225). Metals were quantified by Inductively coupled plasma mass spectrometry (ICP-MS) using an Agilent 8800 ICP-MS (in-house method C-209). The certified reference material DOLT-4 (fish liver) was digested and analysed with the samples as a check on accuracy. Recoveries and limits of detection are reported in Appendix 3.

Statistical analysis

All data analyses were completed in R version 3.3.2 (R Core Team 2016). Package *ggplot2* (Wickham 2009) was used to generate plots. Differences were considered statistically significant if $p \leq 0.05$. To determine significant differences of metal concentrations between the different locations, univariate ANOVAs were conducted. Tukey's Honest Significant Difference

test was used to determine the locations between which the metal concentrations differed significantly.

Metal concentrations measured in each seagrass sample are shown in Appendix 3, Table A3-2. These values largely reflected metal values found within the sediments of the same locations (sediment data only available for the northern Warrior Reef [E] and Warrior Reef [G] sites (Figure 4-50).

Heavy metal concentrations in the Torres Strait seagrass levels were higher at the inshore sampling location (Saibai Island) than that at the offshore sampling sites at Warrior Reef. However, metal concentration in all samples were low (Figure 4-51) and indicative of an unpolluted environment (Govers et al. 2014).

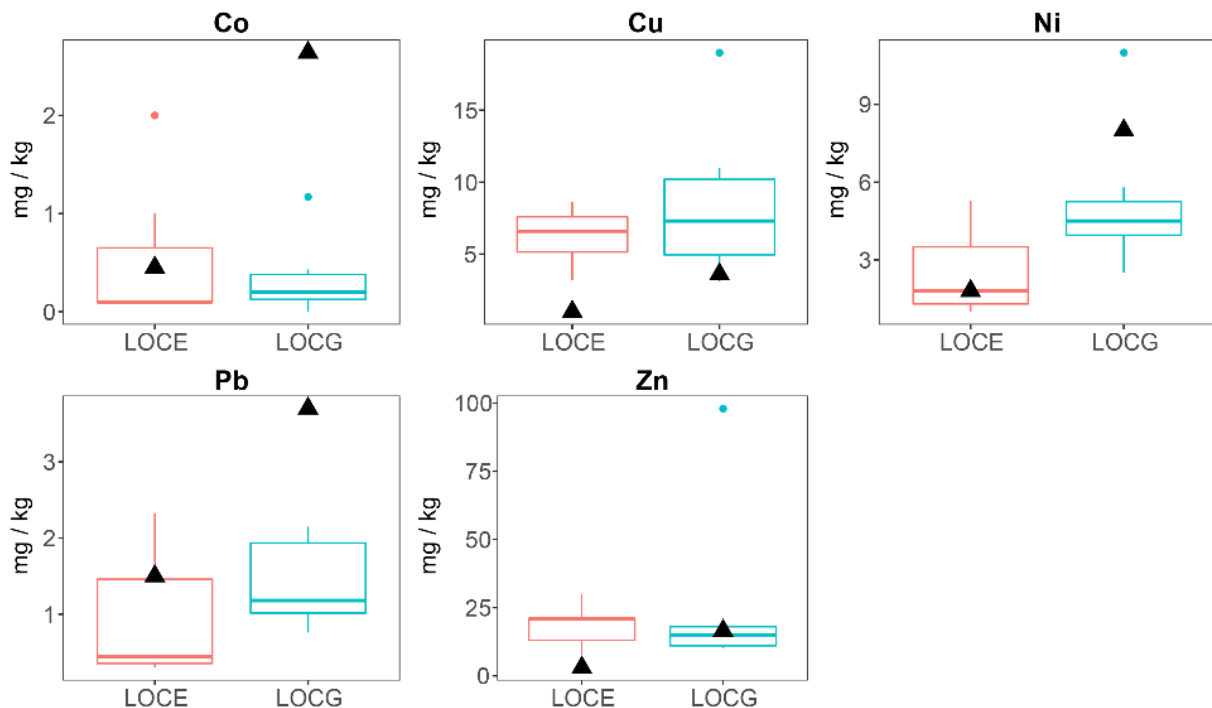


Figure 4-50: Comparison of metal concentrations within the seagrass leaves and sediment samples at northern Warrior Reef (E) and Warrior Reef (G). Boxplots show metal concentrations within the seagrass leaves and the black triangles show the corresponding mean of metal concentrations within the sediment. Colours are according to locations.

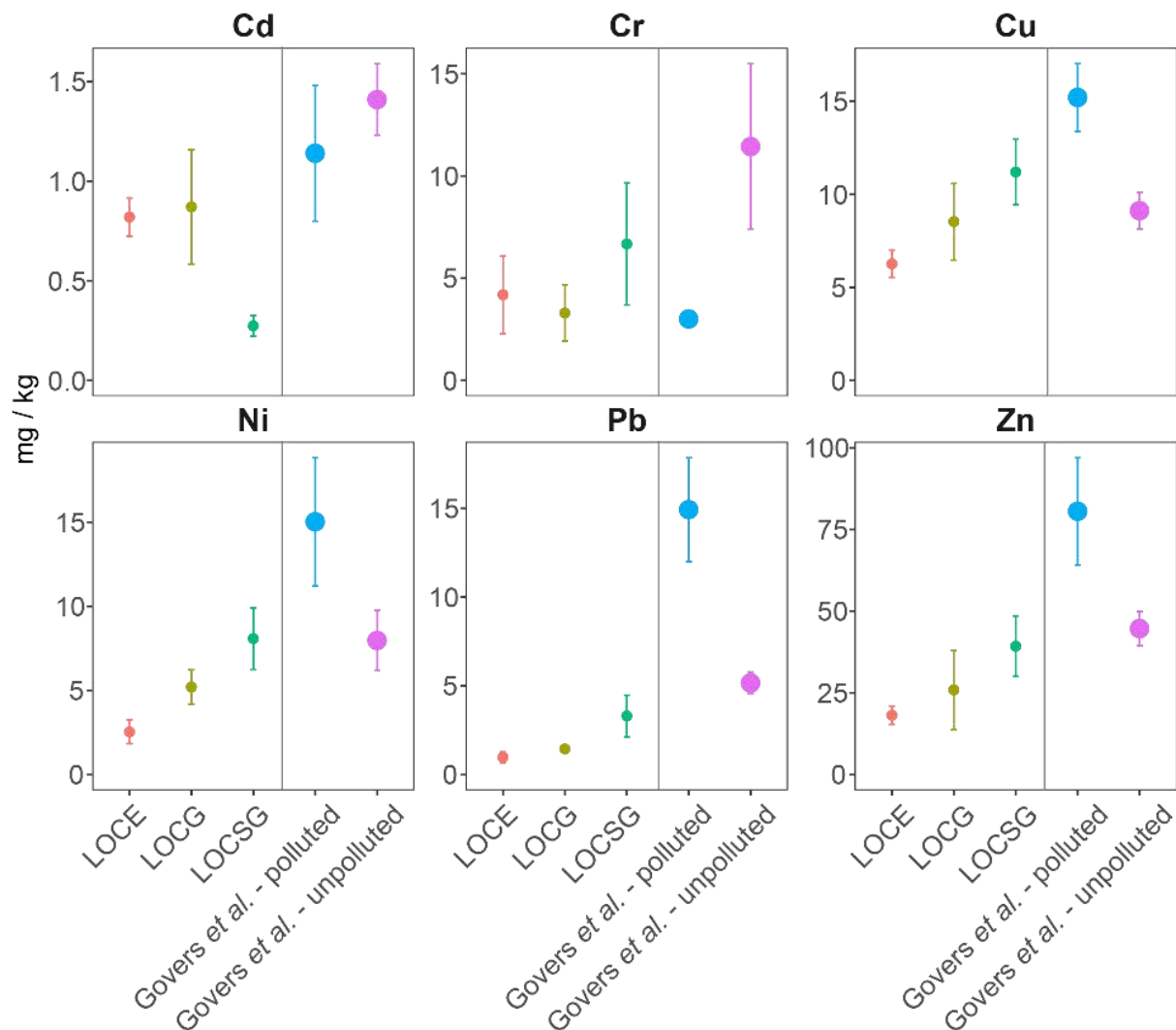


Figure 4-51: Comparison of a suite of metal concentrations (Cd, Cr, Cu, Ni, Pb and Zn) at the northern Warrior Reef (E), Warrior Reef (G) and Saibai Island (SG) sites, shown with values from polluted and unpolluted sites in a global meta-analysis (Govers et al. 2014). Values are shown as mean metal concentration ($\pm$ SE) in mg/kg and colours correspond to the different locations.

The statistical analysis showed that seven metals were found at significantly different concentrations at the three locations: silver, arsenic, cobalt, manganese, nickel, antimony and uranium; in addition, three metals were marginally insignificant: cadmium, lead, and samarium (Appendix 3, Table A3-4 and Figure 4-52).

The northern Warrior Reef site (E) had significantly higher silver and antimony than that of Warrior Reef (G) and Saibai Island (SG), but lower concentrations of nickel (Appendix 3, Table A3-4). With this exception, the Saibai Island site showed larger differences than that of the Warrior Reef sites, with higher metal concentrations for cobalt, manganese and lead, and marginally higher concentrations for nickel and zinc (Appendix 3, Table A3-4). However, the Saibai Island site had considerably lower concentrations of cadmium than at the Warrior Reef sites. It should be noted that the Torres Strait Baseline Study identified cadmium as a marine metal in the region, which is 'naturally' found at elevated concentrations (Gladstone 1996).

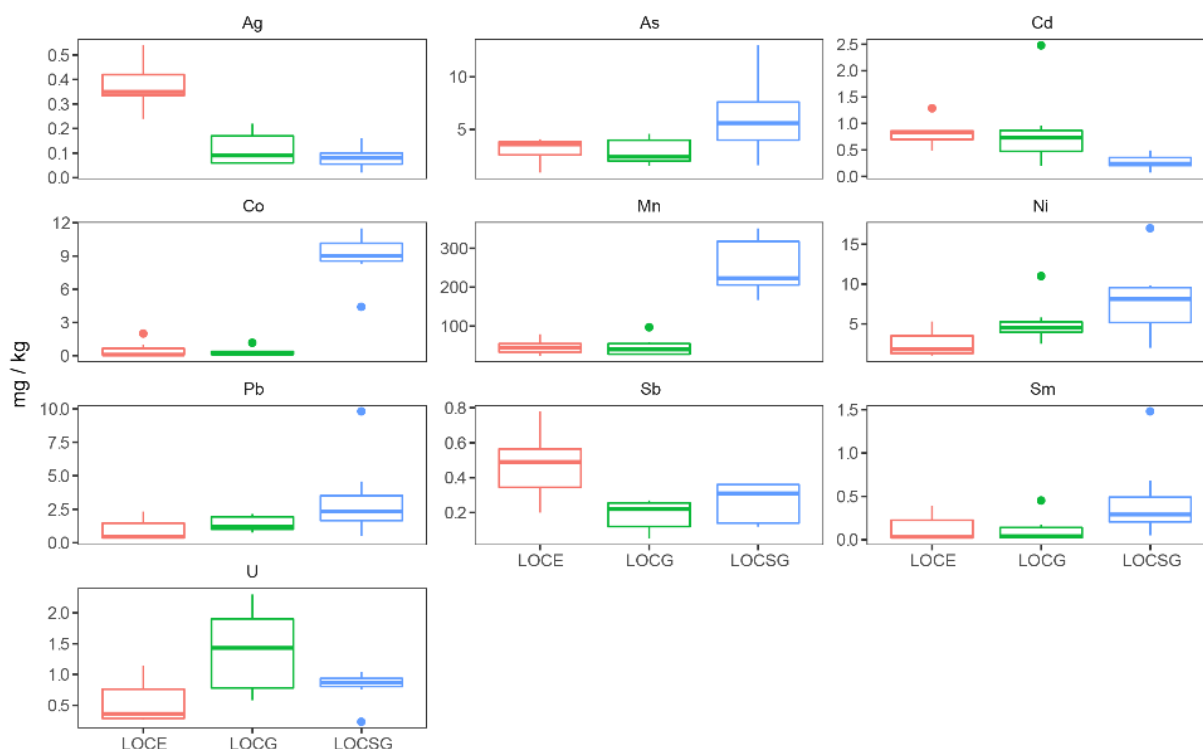


Figure 4-52: Selected metal concentrations in seagrass leaves sampled from Saibai Island (SG) and Warrior Reefs (E and G). Colours are according to location. Post-hoc test results showing which locations were significantly different can be found in Appendix 3, Table A3-4.

In general, seagrass leaf metal concentrations from the Warrior Reef sites reflected concentrations found in the adjacent sediment and largely corresponded to concentrations found in other unpolluted sites globally (Govers et al. 2014). This suggests that even though metal concentrations are present at these sites at concentrations that may indicate land-based metal inputs (see sediment metals results by Apte et al. 2018), the concentrations within the seagrasses do not show elevated pollution.

The results can be compared to GBR seagrass collected as part of the Rivers to Reef to Turtles (RRT) Project (Thomas et al. 2019) and by Haynes (2001). In the RRT study, seagrass was collected close to the coast and under the influence of the Burdekin River (and other coast discharges) from Upstart Bay and Cleveland Bay, and the Howick Island group (located well offshore in the northern GBR and not under strong influence of river runoff) (Thomas et al. 2018). In Haynes (2001), seagrass was collected from coastal sites between Cape York and Moreton Bay including from Lloyd Bay, Princess Charlotte Bay, Flinders Island, Bathurst Bay, Cairns, Cardwell, Pallarenda, Cleveland Bay, Upstart Bay, Newry Bay, Shoalwater Bay, Gladstone, Hervey Bay and Moreton Bay. In comparison to these studies, the Saibai Island (SG) site from the present study compares well with the coastal sites in the GBR having higher concentrations of metals in general, relatively higher concentrations of arsenic, cobalt and manganese, with higher (but not significantly) concentrations of nickel, lead, zinc, and copper and lower concentrations of cadmium (see Appendix 3, Tables A3-3 and A3-4 and Figure 4-50, Figure 4-51 and Figure 4-52). Similarly, out of the metals with well-defined toxicity, arsenic, cobalt, copper, manganese, nickel, lead and zinc were higher at the coastal sites in the GBR than that in the one offshore site (Howick Island group) in the comparative datasets (Thomas et al. 2018; Haynes 2001). This is attributed to the proximity to river discharge in the GBR with all coastal sites subject to considerable river influence whereas the Howick Islands have

minimal river influence. In the case of the Saibai Island site, the influence of river discharge is also likely to be important, with this site in the main zone of the Fly River plume transport during the SE trade wind season (see Sections 4.2 and 4.3) and also possibly influenced from PNG coastal rivers such as the Mai Kussa and Wassi Kussa. Of particular interest, cobalt and manganese were found at much higher concentrations at the Saibai Island site (SG) (but based on limited data at this time) than that at the Warrior Reef sites (more offshore) and in the GBR RRT studies these metals are suspected to be correlated with poor green turtle health (Villa et al. 2017).

4.9.2 Recommendations

Results from this limited study showed higher concentrations of some metals in the seagrass site close to the PNG coast exposed to land-based influences compared to sites further offshore; these mirror results found in the central GBR. These results have potential implications for green turtle health that feed on these important seagrass habitats. However it should be noted that the sediment metal levels were all below ANZECC SQG-Low values suggesting that existing concentrations are well below accepted toxicity thresholds.

Further sampling and analysis of seagrass from this NW region of the Torres Strait, both near the coastal areas of Boigu and Saibai Islands, further offshore and closer to the mouth of the Fly River including the northern Warrior Reefs and Bramble Reef, is required to assess the risk to turtle, dugong and human health in the region. With strategically selected sites, the possible influence of the Fly River discharge could be separated from the influence of other coastal rivers. However, the presence of seagrass is not fully known across this area and so an initial step for sampling would be to conduct surveys for the presence of seagrass including species variation. Seagrass on reef tops (intertidal) across this area will have a different species composition than for the sub-tidal seagrass already sampled, with likely more *Thalassodendron* sp. as sampled in the Torres Strait Baseline Study (Dight and Gladstone 1993; Gladstone 1996).

To more closely understand the potential impact of metal concentrations on seagrass feeding species (e.g. turtles and dugongs) within the Torres Strait, a larger seagrass survey should be conducted. Ideally, at least a number of seagrass meadows of lower impact and an equal number of high impact would be sampled extensively. Following valuable comments from the reviewers, it is suggested that Boigu and Saibai are selected as priority sites for detecting terrigenous impacts from the PNG mainland (potentially including contributions from Fly River) with an additional 3 or 4 sites in locations where fluvial inputs from the Fly River are more likely to be the dominant source of terrigenous contaminants (e.g. between Bramble Cay and Warrior Reef). A further 3 sites to the SW of these areas (e.g. near Erub, Ugar and SW of Bramble Cay) would establish the beginnings of effect-gradient transects which could be extended further SW in the future if any emerging impacts are detected at the pre-existing locations. Sampling would be planned post-monsoon and post-trade wind season where possible.

In addition to measuring the metals within the leaves, metal concentrations within the biofilm should be examined, as these can serve as an indicator of metal concentrations in the surrounding water (Marín-Guirao et al. 2005). Furthermore, an examination of the biofilm communities and the level of connectivity within these in a seagrass meadow can provide detailed information on the health of such ecosystems (Lawes et al. 2016).

4.10 Other analysis

4.10.1 Bioaccumulation of metals – oysters and DGTs

The deployment of oysters as an indicator of trace metal presence or accumulation was assessed as part of this project but was not progressed due to logistical difficulties and the need for ongoing support from the TSRA rangers, which was not considered feasible with existing workloads. However, the proposed deployment of oysters for biological monitoring was investigated as follows:

- A reliable source of oysters was secured, which could be readily obtained and were cultured to a similar size for deployment.
- In liaison with UNSW, a revised deployment method was developed to provide greater security and an enhanced chance of survival of oysters by using cages and a new anchoring system ('Wombats').
- Approvals for oyster deployment at several locations throughout the region was obtained from the Department of Agriculture and Fisheries. Permissions from the relevant Prescribed Body Corporates were also received to conduct the work in their areas.
- Coordination of the deployments (attempted in October, November and December 2016) was very challenging, and two field trips scheduled to Bramble Cay, Erub Island and Masig Island failed due to logistical challenges.
- Following several discussions with local rangers and TSRA staff, it was determined that the ongoing trial of this method (including regular checking of the deployment to check specimen survival and security) would require a greater time commitment by the rangers than what was currently available.

An additional and very important factor was that the oysters are only likely to be a useful technique if the method can be repeated for comparison periodically, which requires an ongoing funding commitment.

The deployment of DGTs is also still a possibly useful technique for further studies of metals in the Torres Strait region. However, more sophisticated multiple DGT devices with long-term deployability need to be explored to overcome the logistical issues associated with the DGTs that were trialled previously (refer to Section 3.2). For example, DGT devices such as the newly developed THOË system (Moreton et al. 2017) could be investigated. THOË is an autosampler that uses 'passive sampler' type DGT™ for the analysis of a range of analytes including metals and/or organic compounds dissolved in water. This device allows the acquisition of time series data over pre-programmed time periods, can be deployed for months at a time and may provide a very cost-effective solution for continuous data collection (sets of serial data).

5. DISCUSSION

The value of the multiple lines of evidence in this project has been demonstrated in the results, however, it is appreciated that it is challenging to piece together all of the highlights of the findings into a cohesive message. To support this need, a summary of the results is presented here.

5.1 Water quality characteristics and exposure

Field salinity measurements showed that the northern Saibai Island is exposed to lower salinity waters and greater salinity ranges than the islands in the mid and southern region of the Torres Strait (Section 4.5 and Figure 5-1). The lowest mean salinities were measured at the most northern site, Saibai Island (29.0 ± 2.6 PSU) and the highest mean salinities were measured at the more southern sites, Iama (35.3 ± 0.7 PSU), Warraber (35.0 ± 1.2 PSU) and Poruma Islands (34.8 ± 0.9 PSU). The mean salinity measurement at Masig Island was lower than expected for an open ocean site with 33.3 ± 1.3 PSU, and the range was relatively high (6.3 PSU) in the 39 samples collected between June 2017 and May 2018. A longer-term dataset is required to assess whether or not this greater range in salinity values may be an indicator of river influence. The greatest range between the highest and lowest salinity values was measured at Saibai Island (10.7 PSU) indicating a periodic influence from an external freshwater source. Insufficient data was collected at Boigu and Erub Islands at the time of reporting (< 5) to allow any meaningful analysis.

Analysis of the frequency of exposure of coral reef and seagrass habitats to waters with higher turbidity using remote sensing imagery has identified that sites around Saibai and Boigu Islands are most frequently exposed to turbid waters compared to islands and habitats in the central and southern areas of the Torres Strait (Section 4.7 and 4.6 and Figure 5-1). This was measured by a greater frequency of exposure to the most turbid satellite colour classes (CC1–4 and CC5) at Saibai and Boigu Islands, respectively, and then at Warrior Reef than at Ugar or Poruma Islands, for example. Saibai and Boigu Islands and Warrior Reefs were also always nearly exposed to coloured waters (on average > 95% of the time exposed to CC1–6 between 2008 and 2018) (Figure 4-30). Inversely, sites such as Bramble Cay, Ugar, Erub, or Masig Islands were less frequently exposed to coloured waters and were classified more than 10% of the time, on average, as ambient marine waters in the satellite imagery.

The source of both turbidity and brackish waters observed around Boigu and Saibai Islands cannot be directly attributed to the Fly River at this stage, however, multiple lines of evidence in the present study including the hydrodynamic modelling, remote sensing and exposure assessment indicate that this is highly likely the case at certain times of the year. The lower salinity levels and greater salinity ranges observed at Boigu and Saibai Islands might also be related to the influence of other coastal rivers. For example, Boigu Island had the lowest mean field salinity value (Section 4.5 and Figure 5-1) and it is likely that this site is affected by freshwaters from the Wassi Kussa and Mai Kussa Rivers (located north and NW of Boigu Island). Similarly, the higher turbidity levels and frequency of exposure to the most turbid satellite colour classes may be linked to other coastal rivers (e.g. at Boigu Island) or resuspension of fine sediments sourced locally or from the Fly River (e.g. at Saibai Island or Warrior Reefs).

Identifying water quality and ecosystem health threats to the Torres Strait from runoff of the Fly River

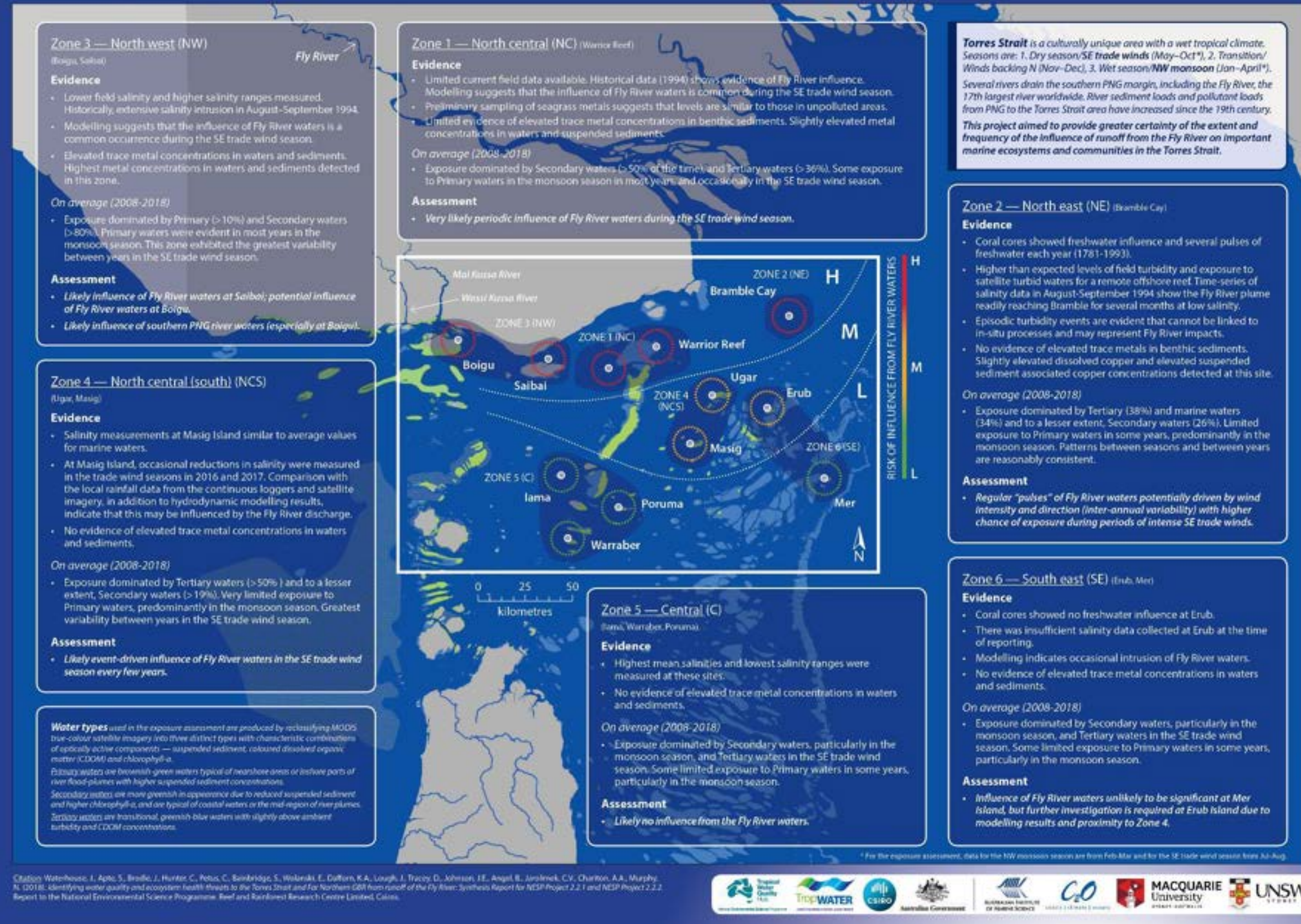


Figure 5-1: Summary of the highlights of the NESP Project 2.2.1 for each component, within the different zones of influence as defined in Table 4-3. Zone 1: North; Zone 2: North East; Zone 3: North West; Zone 4: South; Zone 5: Central and Zone 6: Eastern. Map prepared by D. Tracey, TropWATER, JCU.

Fly River plume waters have been detected across the northern Torres Strait, east of the Warrior Reefs, as far west as Saibai Island and south to Masig Island in previous studies (Wolanski et al. 2013; Martins and Wolanski 2015), which also validates the observations above. **It is therefore reasonable to assume that habitats located in the NE corner of the Torres Strait Protection Zone, north of Masig Island and east of Boigu Island, are located in a higher potential risk area of exposure to brackish and turbid waters from or derived from the Fly River, as well as from or derived from local PNG river discharges (see Figure 5-1).** While this movement of water from the Fly River is a historical pattern, the estimated 40% increase in sediment discharge associated with the operation of the Ok Tedi mine is likely to have changed the characteristics of the sediment and contaminant concentrations in this region.

The exploration of model and remote sensing scenarios allowed proposing preliminary hypotheses on which environmental and ocean conditions are likely drivers of turbid river waters reaching the Torres Strait. The modelling results (Section 4.2) showed that the Fly River plume intruded the Torres Strait the most during the SE trade wind season (i.e. when the ΔMSL across the Strait was $> 0^6$), and that the plume intruded the least in the Torres Strait during the monsoon season when $\Delta\text{MSL} < 0^7$.

These results are corroborated by the field salinity data recorded at Masig Island (Section 4.7). These data show less saline waters at Masig Island in July (SE trade wind season) and, inversely, the presence of more saline waters during December/January (monsoon season). As noted, this might be the result of winter trade winds that episodically push water from PNG down to Masig Island bringing coastal water with lower salinity. The results from the remote sensing component of this project (Section 4.7, Figure 4-11) also showed the occurrence of larger coloured (CC1–6) and areas of Primary water type (turbid, CC1–4) in the southwest Fly District during the trade wind season than during the monsoon season (Section 4.3 and Section 4.7). However, Masig Island was located outside of the main zone of influence of coloured waters (Section 4.3). Further assessment is required to confirm the long-term patterns of turbidity and salinity at Masig Island and what may be the main drivers of any exposure to ‘Fly River water’ around Masig Island, as well as around the northern Warrior Reefs and Saibai Island (Figure 5-1).

El Niño events decrease the strength of the Pacific Trade Winds, reduce the rainfall over northern Australia and New Guinea (as confirmed in Sections 4.1 and 4.7) and have been shown to create large negative perturbation to the relatively constant sediment discharge of the Fly River (Ogston et al. 2008). This may explain the negative relationships observed between annual NINO3.4 and the following:

- i. The areas in km^2 of coloured and turbid/Primary waters measured in the southwest Fly District (CC1–6 and CC1–4, respectively; Section 4.3), and;
- ii. The annual frequency of exposure to the Secondary water type at the seagrass and coral reef sites (Section 4.7).

⁶ $\Delta\text{MSL} > 0$: the MSL Gulf of Papua > MSL Gulf of Carpentaria

⁷ $\Delta\text{MSL} < 0$: the MSL Gulf of Papua < MSL Gulf of Carpentaria

However, the determination coefficients of (ii) were weak and the annual exposure to Secondary waters at these sites is likely the result of the complex oceanographic drivers of water circulation in the region rather than directly correlated to the ENSO cycles.

Luminescent lines have been observed in corals from Bramble Cay back to 1781, which along with previous observations (Wolanski et al. 2013) confirmed a freshwater influence at this site (Section 4.1). Several pulses of freshwater each year have been observed, likely associated with atmospheric or oceanic processes pushing the freshwater onto the reef. The strongest correlation between ENSO and annual exposure to Secondary waters was observed at Bramble Cay where the strong La Nina event of 2010–2011 and strong El Niño event of 2015–2016 respectively resulted in the highest/lowest frequency of exposure to secondary waters at Bramble Cay (Section 4.7). This is likely to highlight an influence of the ENSO cycles on the inter-annual exposure of Bramble Cay to Fly River waters (Figure 5-1). Intra-annual patterns of turbidity at Bramble Cay were generally related to local wind driven resuspension of in-situ material (Section 4.4). However, a peak in turbidity was also observed in June 2017 that seems to be unrelated to local wind-driven resuspension. Analysis of satellite imagery in this period showed evidence of Fly River discharge entering the region and corresponded to a prolonged period of intense (> 25 kph) SE trade winds (Section 4.7, Figure 4-33e). Further assessment is required to confirm these observations. The presence of fine terrestrial material at Bramble Cay, which seems to make up much of the resuspension, indicates that there is long-term transport of material from PNG as the nearest source. However, this may be via more complex long-term transport mechanisms rather than short-term episodic events, which in turn will have implications for the potential transport of contaminants into the area.

At the whole of Torres Strait scale, there is a limited area of coral reefs that are regularly exposed to turbid waters (the northern reefs) and 60% (or 1,778 km²) of the coral reef area is exposed to Primary and Secondary waters infrequently (0–10%). Intertidal and subtidal seagrass habitats have a greater exposure to turbid waters, with 46% (or 173 km²) and 58% (163 km²) of intertidal and subtidal seagrass areas being exposed 10–40% of the time to turbid waters. Northern Torres Strait habitats contain complex and important seagrass and reef communities potentially threatened by changes in water quality (Carter et al. 2014). **It is however important to note that exposure to turbid conditions does not necessarily equate to risk or consequences for the coral and seagrass reef habitats.**

In addition, it is emphasised that these turbid conditions do not necessarily provide a direct link to influence from the Fly River discharge only, as these conditions could also be driven by inputs from other rivers, wind driven resuspension, long-term transport of sediments through currents and noise in the data, especially in the primary signal. The drivers of turbidity are also likely to vary between locations.

While the exposure results provide a useful estimate of the spatial extent of exposure of habitats in the Torres Strait to Fly River discharge, no direct link between exposure to river plume waters and reef condition can be made with these data at this time. Analysis of the likely environmental impact of these influences is a subsequent step which was outside the scope of the current study.

Contaminants

The results from the partner project, NESP Project 2.2.2 (Apte et al. 2018), also support the evidence obtained through this project. Results on the concentrations of mine-derived contaminants in waters and sediments at 21 sites across the Torres Strait showed that the metal concentrations were generally low, and only dissolved copper and cobalt concentrations exceeded the 99% protection levels of the Australia & New Zealand Water Quality Guidelines (2000) in two water samples taken in the vicinity of Saibai Island. The concentrations of both these metals were below the 95% protection levels of the guidelines. All other metals sampled (e.g. mercury, cadmium, nickel and lead) were below guideline values but were detected in higher concentrations in the northern Torres Strait, including around Saibai Island and Bramble Cay. The sources of the higher concentrations of metals in the north remain to be fully identified. As indicated by the results in Project 2.2.1, correlations between suspended sediment and metals indicate that the most likely source of metals are inputs of sediments and waters from the PNG mainland including the Fly River.

The analysis of microbial community composition in sediment samples identified strong relationships to sediment grain size and water depth. Overall, sediment community diversity and composition were significantly related to metal and fines content; however, this did not correspond to a clear signature of the Fly River discharge in the sediment bacterial community. Seagrass samples were also collected from three locations in the Torres Strait, swabbed for gene analysis and analysed for tissue metal concentrations. The seagrass metals were highly correlated with the sediment metal analysis conducted by CSIRO. This is discussed further below.

5.2 Biological indicators

5.2.1 Sediment community surveys

Advances in genetic sequencing now allow information about entire communities, including macro- and micro-organisms to be captured. Micro-organisms have proven highly sensitive to contaminants and provide an additional line of evidence to understand the consequences of subtle environmental changes. Sediment communities in the sediment of the Torres Strait were sampled to create an ecological baseline and investigate relationships to potential environmental drivers that might be associated with the Fly River plume. Known environmental drivers of sediment communities were also sampled and included sediment grain size, and carbon and nitrogen content at 22 sites throughout the Torres Strait. Comparisons were made between the sediment communities and sediment metal concentrations (measured by the NESP Project 2.2.2; Apte et al. 2018).

Ecological baseline: The sediment communities of the Torres Strait did not show patterns that corresponded to the different zones defined in Figure 5-1, but rather followed a trend of changing community composition from east to west. Analysis of the sediment community identified nine distinct biological assemblages for the bacteria and seven distinct biological assemblages for the eukaryotes. Zones 1, 3, 4 and 5 were most similar, while there was significant variation in the microbial communities at locations within Zones 2 and 6 (Figure 5-1). Based on the bacterial community composition, the Torres Strait region could be divided into nine distinct biological assemblages, whereas the eukaryotic community composition resulted in seven biological assemblages. Within the bacterial community throughout the Torres Strait

region, 11 dominant phyla, 17 dominant orders and 4 dominant families were found. The eukaryotic community showed 17 dominant phyla, 13 dominant orders, and 7 dominant families. No previous bacterial community analyses are believed to have been undertaken in the region.

Potential environmental drivers: Changes in microbial community composition were correlated to several potential environmental drivers, including sediment fines content and metals. However, the amount of variation explained by each variable was low. Furthermore, metal concentrations were generally low (see NESP Project 2.2.2 results, Apte et al. 2018) suggesting that the correlations are unlikely to be causative, although an alternative explanation could be that the microbial assemblage is particularly sensitive.

Spatial changes at most locations appeared to correlate best with a decreasing gradient of silt content from east to west. Metal concentrations best explained the community composition at the three sampling locations closest to the PNG mainland. Here, community change may be related to terrestrial run-off. Sediments in the NW of the Torres Strait had higher fractions of sand, whereas samples towards the centre of the region showed higher percentages of fine silty sediment.

In conclusion, a clear signature of the Fly River discharge could not be detected in the sediment microbial communities.

5.2.2 Metals in seagrass leaves

Seagrass leaves from three distinct seagrass meadows near Saibai Island and the Warrior Reefs were sampled opportunistically. These were located within Zones 1, 3 and 4. Metal concentrations were measured within the seagrass leaves to assess the potential influence from the Fly River plume in these locations.

In comparison to global metal concentrations in seagrass leaves, all three locations corresponded more to unpolluted sites than to polluted sites. The site near Saibai Island had the highest concentrations while the sites near the northern Warrior Reefs were lower. Significantly different metal concentrations were found among the three measured sites. The site near Saibai Island (SG) (within Zone 3) had higher concentrations of chromium, copper, nickel, lead and zinc. In contrast, cadmium was slightly elevated at the northern Warrior Reef site (E) (in Zone 1) and the Warrior Reef site (G) (in Zone 4) compared to Saibai Island, but the difference was less than 0.5 mg/kg. The elevated metal concentrations at Saibai suggest higher concentrations of some metals at those areas are more likely to be influenced by land-based run-off compared to offshore sites, which agrees with similar work in the GBR, and has potential implications for green turtle and dugong health.

As the seagrass sampling was only done opportunistically and spatial coverage was limited further sampling of seagrass meadows would be useful to investigate concentrations of seagrass metals and seagrass health (by measuring e.g. shoot density, canopy cover, shoot height, above ground biomass, productivity, and leaf area index, Wood and Lavery 2001). Changes in metal concentrations and microbial communities associated with seagrasses could have a direct impact on seagrass meadow health, and thus on seagrass-feeding mammals, which have an important dietary role in the Torres Strait communities.

5.3 Integration of data sources

Results from this project underline the importance of integration of data from a range of sources to provide a comprehensive assessment of water quality and environmental conditions. For instance, examples of daily MODIS true colour images and colour class maps were selected to illustrate changes in water compositions that can exist in the study area during periods of lower and higher field salinity at Saibai and Masig Islands (Section 4.7). While these images are only examples and do not describe typical/systematic trends in water composition associated with low/high field salinity measurements, they provide a useful geospatial context to the field measurements.

Integration of all these different data sources (see Figure 5-2 for an illustration) will facilitate the development of integrated water quality monitoring tools for application in the region and, where relevant, elsewhere (Figure 5-2).

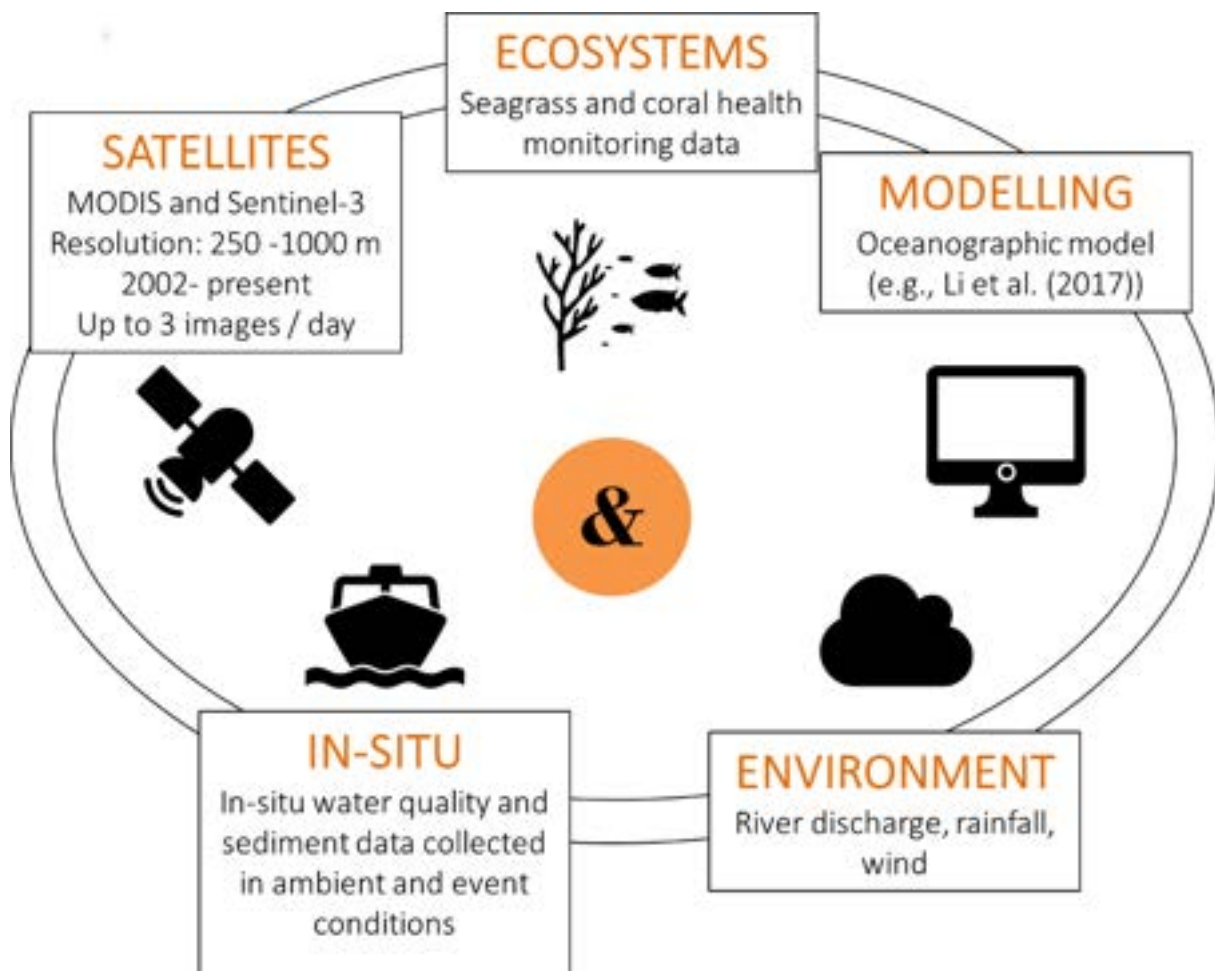


Figure 5-2: Conceptual scheme showing the importance of integrating different data sources to provide comprehensive water quality and environmental assessments.

6. FUTURE WORK

Each of the components of this study have identified recommendations for future work in the previous sections. Many of these are relatively short term. In addition to these, there is a need to establish a longer-term monitoring program that enables the assessment of Torres Strait ecosystems and drivers, to support natural resource management of this highly environmentally, socially and culturally valuable region.

In summary, it is recommended that focused additional work is undertaken in the north and northwestern Torres Strait—including areas around Boigu Island, Saibai Island, Warrior Reefs, Masig Island, Erub Island and Bramble Cay. The recommended activities to be conducted in the region during 2019–2020 are outlined below:

1. Field surveys of trace metals in waters and sediments to assess the sources and fate of elevated turbidity and metals in the northern Torres Strait region (NESP Project 2.2.2): Intended to provide a more comprehensive understanding of the trace metal concentrations found in the northern Torres Strait (Saibai, Boigu and Dauan Islands) and their potential sources (Fly River or other coastal discharges). This would include studies to 1) Develop methods for tracing mine-derived sediments in the Torres Strait; 2) Characterise trace metal distributions in the northern Torres Strait; and 3) Undertake Fly River plume tracking through the tracing method developed and intensive sampling campaigns for the collection and analysis of sediment and metals in the northern Torres Strait.
2. **Collection of anecdotal evidence of plume intrusions (NESP Project 2.2.2):** This would provide further insight to the views of the local community regarding the influence of the Fly River into the Torres Strait. To date, interviews have been conducted by CSIRO with community members from Boigu and Saibai Islands (see Apte et al. 2018). This effort should be extended to include other areas of interest and engage scientists and local experts with long-term experience in the region.
3. **Continued and expanded deployment of *in-situ* loggers for salinity and turbidity:** This would provide real-time meteorological, salinity and turbidity data stations at Bramble Cay and Masig Island (both continuing with the addition of salinity probes), and new stations at Saibai Island and the northern Warrior Reefs to assist in the assessment of the temporal variability in the frequency of exposure of brackish and turbid waters. This will provide continuation of real time meteorological and turbidity data to provide measures of Fly River intrusions and to provide ocean models with *in-situ* parameters for model runs. Continued turn-over of the real time ocean temperature loggers in the region would assist in understanding and measuring coral bleaching conditions.
- The initial data and model runs completed under the NESP project show that there are events where coastal PNG waters can influence the waters in the northern Torres Strait, such as around Bramble Cay and occasionally down to the central Torres Strait (e.g. Masig Island). The data are too sparse to reliably filter out local *in-situ* events (such as the re-suspension of local material through local wind conditions or low salinity events from localised rainfall), and thus the **collection of additional *in-situ* data would help identify what conditions promote the influx of PNG water into the northern Torres Strait.**

4. **Continued targeted local salinity monitoring:** It is recommended to continue weekly salinity monitoring by TSRA rangers at northern locations to link results to the logger data where applicable and provide local data from additional locations. Present salinity monitoring is conducted at Boigu, Saibai, Erub, Masig, Iama, Poruma and Warraber Islands. It would be desirable to continue this effort in these locations, with priority for Boigu, Saibai, Erub and Masig Islands. We will also explore the application of the simple 'Eye on the Water' phone application to measure Forel Colour Classification (linked to the remote sensing true colour analysis), which could be collected during routine ranger activities where possible/appropriate. This component can also provide an avenue for local engagement and capacity building for TSRA rangers.
5. **Continued remote sensing of river plumes:** Continuation of the 10-year dataset of satellite remote sensing data of plume movement, turbidity and frequency of the exposure of turbid waters in the northern Torres Strait is recommended. The additional field data would increase the confidence of the results. This data would assist in linking the *in-situ* data to potential sources, help distinguish local and Fly River influences and assess changes between 2008 and 2020.
6. **Development of sediment transport model:** The newly developed model would support assessment of the transport of Fly River-derived sediment and its particulate metals in the Torres Strait region. Field data would be available from the *in-situ* loggers, *in-situ* data (recent plus Wolanski's 1990s studies) and potentially remote-sensing. The model would assist in predicting changes in the distribution of particulate material from the Fly River over time.
7. **Further analysis of trace metals in seagrass leaves:** It is recommended that the pilot study be expanded to assess the ecological relevance of trace metal concentrations in seagrass and associated sediments, an important food source for the highly valued turtle and dugong populations in the region, by analysing trace metals in seagrass leaves. Boigu and Saibai would be selected as priority sites for detecting terrigenous impacts from the PNG mainland (potentially including contributions from Fly River) with an additional 3 or 4 sites in locations where fluvial inputs from the Fly River are more likely to be the dominant source of terrigenous contaminants (e.g. between Bramble Cay and Warrior Reef). A further 3 sites to the SW of these areas (e.g. near Erub, Ugar and SW of Bramble Cay) would establish the beginnings of effect-gradient transects which could be extended further SW in the future if any emerging impacts are detected at the pre-existing locations. Sampling would be planned post-monsoon and post-trade wind season where possible.
8. **Deployment of DGTs to assess site-based metal concentrations:** Investigation of the potential application of new DGT devices that may overcome logistical issues previously experienced in local studies in the past as a way of measuring accumulation of metals at high priority sites such as Saibai and Boigu Islands, near the Warrior Reefs and Bramble Cay.

Combined interpretation of the findings including a full risk assessment to ecosystems and communities would also be required to assess the overall implications of the results of the exposure of sediments and trace metals to ecosystems in the northern Torres Strait and the probable sources. This would include a qualitative assessment of the potential implications of the findings for the local community. To support this work, local coral and seagrass monitoring efforts need to continue (and be reported) to support analysis of the potential correlations between the results and ecosystem health.

In the longer term, it is recommended that investigations would extend to include a repeated and in-depth examination of biological indicators and seafood as conducted as part of the Torres Strait Baseline Study in the early 1990s. The potential implication of the results of these studies would need to consider potential human health issues, and it is thus recommended that the need to repeat the studies is fully substantiated by the above short-term studies before making such a commitment.

7. REFERENCES

- Alongi, D.M., Tirendi, F., Robertson, A.I. (1991). Vertical profiles of copper in sediments from the Fly Delta and Gulf of Papua (Papua New Guinea). *Marine Pollution Bulletin*, 22(5), 253-255.
- Álvarez-Romero, J., Devlin, M., da Silva, E., Petus, C., Ban, N., Pressey, R., Kool, J., Roberts, J., Cerdeira-Estrada, S., Wenger, A., Brodie, J. (2013). A novel approach to model exposure of coastal-marine ecosystems to riverine flood plumes based on remote sensing techniques. *Journal of Environmental Management*, 119, 194-207.
- Amaral-Zettler, L.A., McCliment, E.A., Ducklow, H.W., Huse, S.M. (2009). A method for studying protistan diversity using massively parallel sequencing of V9 hypervariable regions of small-subunit ribosomal RNA genes. *PLoS One*, 4, e6372.
- Anders, S., Huber, W. (2010). Differential expression analysis for sequence count data. *Genome Biology*, 11, R106.
- Angel, B.M., Apte, S.C., Simpson, S.L., Jarolimek, C.V., Jung, R.F. (2010a). Behaviour of copper in the Fly River Estuary, Papua New Guinea. CSIRO Water for a Healthy Country Technical Report. Prepared for Ok Tedi Mining Ltd, 45pp.
- Angel, B.M., Hales, L.T., Simpson, S.L., Apte, S.C., Chariton, A.A., Shearer, D.A., Jolley, D.F., (2010b). Spatial variability of cadmium, copper, manganese, nickel and zinc in the Port Curtis Estuary, Queensland, Australia. *Marine and Freshwater Research*, 61, 170-183.
- Angel, B.M., Apte, S.C., Simpson, S.L., Jarolimek, C.V., King, J.J. (2014). The behaviour of trace metals in the Fly Estuary, Papua New Guinea. CSIRO Water for a Healthy Country Technical report EP14897, 91pp.
- Anthony, K.R.N. (2000). Enhanced particle-feeding capacity of corals on turbid reefs (Great Barrier Reef, Australia). *Coral Reefs*, 19(1), 59-67.
- Anthony, K.R.N., Fabricius, K.E. (2000). Shifting roles of heterotrophy and autotrophy in coral energetics under varying turbidity. *Journal of Experimental Marine Biology and Ecology*, 252, 221-253.
- ANZECC and ARMCANZ (Australia and New Zealand and Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand), 2000. Australian and New Zealand guidelines for fresh and marine water quality. Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand, Canberra.
- Apte, S.C., Angel, B.M., Hunter, C., Jarolimek, C.V., Chariton, A.A., Murphy, N. (2018). Impacts of mine-derived pollution on Torres Strait environments and communities. Report to the National Environmental Science Programme. Reef and Rainforest Research Centre Limited, Cairns, 117pp.
- Australian Government (2014, 18 July 2014). The context of water planning in Queensland. 2015. <http://archive.nwc.gov.au/library/topic/planning/report-card/queensland>.
- Aylagas, E., Borja, Á., Tangherlini, M., Dell'Anno, A., Corinaldesi, C., Michell, C.T., Irigoien, X., Danovaro, R., Rodríguez-Ezpeleta, N. (2017). A bacterial community-based index to assess the ecological status of estuarine and coastal environments. *Marine Pollution Bulletin*, 114, 679-688.

- Ayukai, T., Wolanski, E. (1997). Importance of biologically mediated removal of fine sediments from the Fly River plume, Papua New Guinea. *Estuarine, Coastal and Shelf Science*, 44, 629-639.
- Bainbridge, S.J. (2016). Torres Strait temperature logger and real time ocean monitoring stations; update May 2016. Townsville, AIMS.
- Bainbridge, S.J. (2017). Temperature and light patterns at four reefs along the Great Barrier Reef during the 2015–2016 austral summer: understanding patterns of observed coral bleaching. *Journal of Operational Oceanography*, 1-14.
- Bainbridge, S., Berkelmans, R. (2014). The use of climatologies and Bayesian models to link observations to outcomes; an example from the Torres Strait. *Environmental Science: Processes & Impacts*, 16(5), 1041-1049.
- Bainbridge, S.J., Berkelmans, R., Sweatman, H., Weeks, S. (2015). Monitoring the health of Torres Strait Reefs – Final Report. Report to the National Environmental Research Program. Reef and Rainforest Research Centre Limited, Cairns, 93pp.
- Bainbridge, Z., Lewis, S., Bartley, R., Fabricius, K., Collier, C., Waterhouse, J., Garzon-Garcia, A., Robson, B., Burton, J., Wenger, A., Brodie, J. (2018). Fine sediment and particulate organic matter: a review and case study on ridge-to-reef transport, transformations, fates, and impacts on marine ecosystems. *Marine Pollution Bulletin*, 135, 1205-1220.
- Baird, D.J., Hajibabaei, M. (2012). Biomonitoring 2.0: A new paradigm in ecosystem assessment made possible by next-generation DNA sequencing. *Molecular Ecology*, 21, 2039-2044.
- Baker, E.K. (1991). Copper and Zinc Distribution in the Sediments of the Fly Delta and Torres Strait. Sustainable Development for Traditional Inhabitants of the Torres Strait Region: Proceedings of the Torres Strait Baseline Study Conference. D. Lawrence and T. Cansfield-Smith (eds) pp. 87-96.
- Baker, E.K. (1998). Sedimentation in a tidally dominated delta and the impact of upstream mining: Fly River, Papua New Guinea. Unpublished PhD thesis, University of Sydney.
- Baker, E.K., Harris, P.T., Beck, R.W. (1990). Cu and Cd associated with suspended particulate matter in Torres Strait. *Marine Pollution Bulletin*, 21(10), 484-486.
- Bielmyer, G.K., Grosell, M., Bhagooli, R., Baker, A.C., Langdon, C., Gillette, P. and Capo, T.R., (2010). Differential effects of copper on three species of scleractinian corals and their algal symbionts (*Symbiodinium* spp.). *Aquatic Toxicology*, 97(2), 125-133.
- Bégin, C., Schelten, C.K., Nugues, M.M., Hawkins, J., Roberts, C., Côté, I.M. (2016). Effects of protection and sediment stress on coral reefs in Saint Lucia. *PLoS ONE*, 11(2), p.e0146855.
- Bejarano, I., Appeldoorn, R.S. (2013). Seawater turbidity and fish communities on coral reefs of Puerto Rico. *Marine Ecology Progress Series*, 474, 217-226. <https://doi.org/10.3354/meps10051>
- Benthuisen, J.A., Oliver, E.C., Feng, M., Marshall, A.G. (2018). Extreme marine warming across tropical Australia during austral summer 2015–2016. *Journal of Geophysical Research: Oceans*, 123, 1301-1326.
- Bessell-Browne, P., Negri, A., Fisher, R., Clode, P.L., Jones, R. (2017). Impacts of light limitation on corals and crustose coralline algae. *Science Reports*, 7(1), 11553. doi: 10.1038/s41598-017-11783-z.

- Bielmyer, G., Grosell, M., Bhagooli, R., Baker, A., Langdon, C., Gillette, P., Capo, T. (2010). Differential effects of copper on three species of scleractinian corals and their algal symbionts (*Symbiodinium* spp.). *Aquatic toxicology* (Amsterdam, Netherlands). 97, 125-33. 10.1016/j.aquatox.2009.12.021.
- Birkeland, C. (1988). Geographic comparisons of coral-reef community processes. In Choat, J.H., D. Barnes, M.A. Borowitzka, J.C. Coll, P.J.Davies, P. Flood, B.G. Hatcher, D. Hopley, P.A. Hutchings, D. Kinsey, G.R. Orme, M. Pichon, P.F. Sale, P. Sammarco, C.C. Wallace, C. Wilkinson, E. Wolanski and O. Bellwood (eds.) *Proceedings of the 6th International Coral Reef Symposium: Vol. 1: Plenary Addressess and Status review*. Townsville, Australia. pp. 211-220.
- Birrer, S.C., Dafforn, K.A., Simpson, S.L., Kelaher, B.P., Potts, J., Scanes, P., Johnston, E.L. (2018). Interactive effects of multiple stressors revealed by sequencing total (DNA) and active (RNA) components of experimental sediment microbial communities. *Science of the Total Environmeny*, 637, 1383-1394.
- Biscéré, T., Lorrain, A., Rodolfo-Metalpa, R., Gilbert, A., Wright, A., Devissi, C., Peignion, C., Farman, R., Duvieilbourg, E., Payri, C.,Houlbrèque, F. (2017). Nickel and ocean warming affect scleractinian coral growth. *Marine Pollution Bulletin*, 120(1-2), 250-258.
- Bissett, A., Fitzgerald, A., Meintjes, T., Mele, P.M., Reith, F., Dennis, P.G., Breed, M.F., Brown, B., Brown, M.V., Brugger, J., Byrne, M., Caddy-Retalic, S., Carmody, B., Coates, D.J., Correa, C., Ferrari, B.C., Gupta, V.V.S.R., Hamonts, K., Haslem, A., Hugenholtz, P., Karan, M., Koval, J., Lowe, A.J., Macdonald, S., McGrath, L., Martin, D., Morgan, M., North, K.I., Paungfoo-Lonhienne, C., Pendall, E., Phillips, L., Pirzl, R., Powell, J.R., Ragan, M.A., Schmidt, S., Seymour, N., Snape, I., Stephen, J.R., Stevens, M., Tinning, M., Williams, K., Yeoh, Y.K., Zammit, C.M., Young, A. (2016). Introducing BASE: The Biomes of Australian Soil Environments soil microbial diversity database. *Gigascience* 5, 1-11.
- Bolton, B.R., Pile, J.L., Kundapen, H. (2009). Texture, geochemistry, and mineralogy of sediments of the Fly River system. In: *The Fly River Papua New Guinea. Environmental Studies in an Impacted Tropical River System*, ed. B.R. Bolton. Burlington, MA, USA: Elsevier, pp. 51-112.
- Bonanno, G., Orlando-Bonaca, M. (2017). Trace elements in Mediterranean seagrasses: accumulation, tolerance and biomonitoring. A review. *Marine Pollution Bulletin*, 125(1-2), 8-18.
- Bonanno, G., Orlando-Bonaca, M. (2018). Trace elements in Mediterranean seagrasses and macroalgae. A review. *Science of The Total Environment*, 618, 1152-1159.
- Borai, E.H., El-Sofany, E.A., Abdel-Halim, A.S., Soliman, A.A. (2002). Speciation of hexavalent chromium in atmospheric particulate samples by selective extraction and ion chromatographic determination. *TrAC Trends in Analytical Chemistry*, 21(11), 741-745.
- Brock, J., Bielmyer, G.K., (2013). Metal accumulation and sublethal responses in the sea anemone, *Aiptasia pallida* after waterborne exposure to metal mixtures. *Comparative Biochemistry & Physiology*, 158, 150–158.
- Brown, C.J., Jupiter, S.D., Lin, H.-Y., Albert, S., Klein, C., Maina, J.M., Tulloch, V.J., Wenger, A.S., Mumby, P.J. (2017). Habitat change mediates the response of coral reef fish populations to terrestrial run-off. *Marine Ecology Progress Series*, 576, 55-68.

- Burdon-Jones, C., Denton, G. (1984a). Metals in marine organism from the Great Barrier Reef Province, Part 1. Baseline survey. Final Report to the Australian Marine Science and Technologies Grants Committee. 155pp.
- Burdon-Jones, C., Denton, G. (1984b). Metals in marine organism from the Great Barrier Reef Province, Part 2. Regional and seasonal variations. Final Report to the Australian Marine Science and Technologies Grants Committee. 162pp.
- Burdon-Jones, C., Denton, G., Jones, G., McPhie, K.A. (1982). Regional and seasonal variations of trace metals in tropical Phaeophyceae from North Queensland. *Marine Environmental Research*, 7(1). doi: 10.1016/0141-1136(82)90048-4.
- Butler, J.R.A., Bohensky, E., Skewes, T., Maru, Y., Busilacchi, S., Rochester, W., Katzfey, J., Wise, R.M. (2015). Drivers of change in the Torres Strait region: status and trends. Report to National Environmental Research Program. Reef & Rainforest Research Centre Limited, Cairns, 60pp.
- Canestrelli, A., Fagherazzi, S., Defina, A., Lanzoni, S. (2010). Tidal hydrodynamics and erosional power in the Fly River delta, Papua New Guinea. *Journal of Geophysical Research*, 115, F04033. <http://dx.doi.org/10.1029/2009JF001355>
- Carter, A.B., Rasheed, M.A. (2016). Assessment of key dugong and turtle seagrass resources in north-west Torres Strait. Report to the National Environmental Science Programme and Torres Strait Regional Authority. Reef and Rainforest Research Centre Limited, Cairns, 40pp.
- Carter, A.B., Rasheed, M.A. (2018). Torres Strait Seagrass Long-term Monitoring: Dugong Sanctuary, Dungeness Reef and Orman Reefs., Centre for Tropical Water & Aquatic Ecosystems, Research Report No. 18/17, James Cook University, Cairns, 35 pp.
- Carter, A.B., Taylor, H.A., Rasheed, M.A. (2014). Torres Strait mapping: Seagrass consolidation, 2002–2014. JCU Publication, Report no. 14/55, Centre for Tropical Water & Aquatic Ecosystem Research, Cairns, 47pp.
- Carter, A.B., Mellors, J.M., Rasheed, M.A. (2018). Torres Strait Seagrass 2018 Report Card. Centre for Tropical Water & Aquatic Ecosystems, Research Publication 18/25, James Cook University, Cairns, 65 pp.
- Chappell, J. (1992). Effects of increased sediment flux in the Fly River delta: Sediment core results. Australian National University, School of Biogeography & Geomorphology. Report to Ok Tedi Mining Limited dated 1992, 64pp.
- Chariton, A.A., Stephenson, S., Morgan, M.J., Steven, A.D.L., Colloff, M.J., Court, L.N., Hardy, C.M. (2015). Metabarcoding of benthic eukaryote communities predicts the ecological condition of estuaries. *Environmental Pollution*, 203, 165-174.
- Coles, R.G. McKenzie, L.J., Campbell S.J. (2002). Chapter 11: The seagrasses of eastern Australia. In: *The World Atlas of Seagrasses: Present Status and Future Conservation*, eds. E.P. Green, F.T. Short, M.D. Spalding. University of California Press, pp. 119-133.
- Coles, R., Grech, A., Rasheed, M., McKenzie, L. (2012). Evaluating risk to seagrasses in the Tropical Indo-Pacific Region. Proceedings of the 12th International Coral Reef Symposium, Cairns, Australia, 9-13 July 2012. Mini-Symposium 15b–Seagrasses and seagrass ecosystems.

- Collier, C., Waycott, M. (2009). Drivers of change to seagrass distributions and communities on the Great Barrier Reef: Literature Review and Gaps Analysis. Report to the Marine and Tropical Sciences Research Facility. Reef and Rainforest Research Centre Limited, Cairns (55pp.)
- Collier, C.J., Lavery, P.S., Ralph, P.J., Masini, R.J. (2009). Shade-induced response and recovery of the seagrass *Posidonia sinuosa*. *Journal of Experimental Marine Biology and Ecology*, 370(1-2), pp.89-103.
- Collier, C.J., Waycott, M., McKenzie, L.J., (2012). Light thresholds derived from seagrass loss in the coastal zone of the northern Great Barrier Reef, Australia. *Ecological Indicators*, 23, 211–219.
- Collier, C.J., Adams, M., Langlois, L., Waycott, M., O'Brien, K., Maxwell, P., McKenzie, L., (2016). Thresholds for morphological response to light reduction for four tropical seagrass species. *Ecological Indicators*, 67, 358–366.
- De'ath, G., Fabricius, K. (2010). Water quality as a regional driver of coral biodiversity and macroalgae on the Great Barrier Reef. *Ecological Applications*, 20, 840-850.
- de Bakker, D., van Duyl, F.C., Bak, R.P.M., Nugues, M.M., Nieuwland, G., Meesters, E.H. (2017). 40 Years of benthic community change on the Caribbean reefs of Curaçao and Bonaire: the rise of slimy cyanobacterial mats. *Coral Reefs*, 36, 355-367. DOI link: <https://doi.org/10.1007/S00338-016-1534-9>.
- Denton, G.R.W., Burdon-Jones, C. (1986a). Trace metals in algae from the Great Barrier Reef. *Marine Pollution Bulletin*, 17(3), 98-107.
- Denton, G.R.W., Burdon-Jones, C. (1986b). Trace metals in corals from the Great Barrier Reef. *Marine Pollution Bulletin*, 17(5), 209-213.
- Denton, G.R.W., Burdon-Jones, C. (1986c). Trace metals in surface water from the Great Barrier Reef. *Marine Pollution Bulletin*, 17(3), 96-98.
- Devlin, M., Teixeira da Silva, E., Petus, C., Wenger, A., Zeh, D., Tracey, D., Álvarez-Romero, J., Brodie, J. (2013). Combining in-situ water quality and remotely sensed data across spatial and temporal scales to measure variability in wet season chlorophyll-a: Great Barrier Reef lagoon (Queensland, Australia). *Ecological Processes*, 2(1), 31.
- Devlin, M., Petus, C., Teixeira da Silva, E., Tracey, D., Wolff, N., Waterhouse, J., Brodie, J., (2015). Water quality and river plume monitoring in the Great Barrier Reef: An overview of methods based on ocean colour satellite data. *Remote Sensing*, 7, 12909-12941. doi: 10.3390/rs71012909
- Dietrich, W.E., Day, G.M., Parker, G. (1999). The Fly River, Papua New Guinea: Inferences about river dynamics, floodplain sedimentation and fate of sediment, In: *Varieties of Fluvial Form*, eds. A. J. Miller, A. Gupta. New York: John Wiley, pp. 345-376.
- Dight, I.J., Gladstone, W. (1993). Torres Strait baseline study: pilot study final report June 1993. Great Barrier Reef Marine Park Authority Research Publication No. 29. Townsville, Great Barrier Reef Marine Park Authority.
- Duckworth, A., Giofre, N., Jones, R., (2017). Coral morphology and sedimentation. *Marine Pollution Bulletin*, 125 (1–2), 289–300.

- Duke, N.C., Burrows, D., Mackenzie, J.R. (2015). Mangrove and freshwater wetland habitat status of the Torres Strait Islands. Biodiversity, biomass and changing condition of wetlands. Report to the National Environmental Research Program. Reef and Rainforest Research Centre Limited, Cairns, 100pp.
- Edgar, R.C. (2013). UPARSE: Highly accurate OTU sequences from microbial amplicon reads. *Natural Methods*, 10, 996-998.
- Environmental Systems Research Institute (ESRI). (2013). ArcGIS: Release 10.2. Environmental Systems Research Institute (ESRI). Redlands, CA.
- Erftemeijer, P.L., Riegl A., Hoeksema B.W., Todd, P.A. (2012). Environmental impacts of dredging and other sediment disturbances on corals: A review. *Marine Pollution Bulletin*, 64, 1737–1765.
- Esslemont, G. (2000). Heavy metals in seawater, marine sediments and corals from the Townsville section, Great Barrier Reef Marine Park, Queensland. *Marine Chemistry*, 71(3–4), 215-231.
- Evans-Illidge, E. (1997). Heavy metals in commercial prawn and crayfish species in Torres Strait. Great Barrier Reef Marine Park Authority, Townsville. 125pp.
- Fabricius, K., De'ath, G., McCook, L., Turak, E., Williams, D.M.c.B. (2005). Changes in algal, coral, and fish assemblages along water quality gradients on the inshore Great Barrier Reef. *Marine Pollution Bulletin*, 51, 384-398.
- Fabricius, K.E., De'ath, G., Humphrey, C., Zagorskis, I., Schaffelke, B. (2013). Intra-annual variation in turbidity in response to terrestrial runoff on near-shore coral reefs of the Great Barrier Reef. *Estuarine, Coastal and Shelf Science*, 116, 57-65.
- Fabricius, K.E., Logan, M., Weeks, S., Brodie, J. (2014). The effects of river run-off on water clarity across the central Great Barrier Reef. *Marine Pollution Bulletin*, 84, 191–200.
- Fabricius, K.E., Logan, M., Weeks, S.J., Lewis, S.E., Brodie, J. (2016). Changes in water clarity related to river discharges on the Great Barrier Reef continental shelf: 2002-2013. *Estuarine, Coastal and Shelf Science*, 173, A1–A5.
- Ferguson, A.J.P., Gruber, R., Potts, J., Wright, A., Welsh, D.T., Scanes, P., (2017). Oxygen and carbon metabolism of *Zostera muelleri* across a depth gradient – implications for resilience and blue carbon. *Estuarine Coastal Shelf Science*, 187, 216–230.
- Ferrier-Pages C, Schoelzke V, Jaubert J, Muscatine L, Hoegh-Guldberg O (2001) Response of a scleractinian coral *Stylophora pistillata*, to iron and nitrate enrichment. *Journal of Experimental Marine Biology and Ecology*, 259, 249–261
- Forsberg, J., Dahlgvist, R., Johan, G. (2006). Trace metal speciation in brackish water using diffusive gradients in thin films and ultrafiltration: comparison of techniques. *Environmental Science and Technology*, 40, 3901-3905.
- Fuentes, M.M.P.B., Bell, I., Hagihara, R., Hamann, M., Hazel, J., Huth, A., Seminoff, J.A., Sobtchik, S., Marsh, H. (2015). Improving in-water estimates of marine turtle abundance by adjusting aerial survey counts for perception and availability biases. *Journal of Experimental Marine Biology and Ecology*, 471, 77-83.

Galloway, W.E. (1975). Process framework for describing the morphologic and stratigraphic evolution of a deltaic depositional system, In: *Deltas: Models for exploration*, ed. M. L. Broussard. Houston: Houston Geological Society, pp. 87-98.

Gissi, F., Stauber, J., Reichelt-Brushett, A., Harrison, P.L. and Jolley, D.F., (2017). Inhibition in fertilisation of coral gametes following exposure to nickel and copper. *Ecotoxicology and Environmental Safety*, 145, 32-41.

Gehrke, P.C., Sheaves, M.J., Terry, J.P., Boseto, D., Ellison, J.C., Figa, B.S., Wani, J. (2011). Chapter 7: Vulnerability of freshwater and estuarine fish habitats in the tropical Pacific to climate change. In: *Vulnerability of tropical Pacific fisheries and aquaculture to climate change*, eds. J.D. Bell, J.E. Johnson, A.J. Hobday. Noumea, New Caledonia: Secretariat of the Pacific Community.

Gladstone, W. (1996). Trace metals in sediments, indicator organisms and traditional seafoods of the Torres Strait. Great Barrier Reef Marine Park Authority, Townsville. <http://hdl.handle.net/11017/262>

Govers, L.L., Lamers, L.P.M., Bouma, T.J., Eygensteyn, J., de Brouwer, J.H.F., Hendriks, A.J., Huijbers, C.M., van Katwijk, M.M. (2014). Seagrasses as indicators for coastal trace metal pollution: a global meta-analysis serving as a benchmark, and a Caribbean case study. *Environmental Pollution*, 195, 210-217. <https://doi.org/10.1016/j.envpol.2014.08.028>.

Hagihara, R., Cleguer, C., Preston, S., Soltzick, S., Hamann, M., Shimada, T., Marsh, H. (2016). Improving the estimates of abundance of dugongs and large immature and adult-sized green turtles in Western and Central Torres Strait. Report to the National Environmental Science Programme. Reef and Rainforest Research Centre Limited, Cairns, 53pp.

Hamann, M., Smith, J. Preston S., Fuentes, M.M.P.B. (2015a). Nesting green turtles of Torres Strait. Report to the National Environmental Research Program. Reef and Rainforest Research Centre Limited, Cairns, 13pp.

Hamann, M., Smith, J., Preston, S. (2015b). Flatback turtles of Torres Strait. Report to the National Environmental Research Program. Reef and Rainforest Research Centre Limited, Cairns, 8pp.

Harris, P.T. (2001). Environmental management of Torres Strait: a marine geologist's perspective. In: Gostin, V.A., (Ed), *Gondwana to Greenhouse*, pp. 317-328. Geological Society of Australia, Sydney.

Harris, P.T., Baker, E.K., Cole, A.R., Short, S.A. (1993). A preliminary study of sedimentation in the tidally dominated Fly River Delta, Gulf of Papua. *Continental Shelf Research*, 13, 441–472.

Harris, P.T., Heap, A.D., Passlow, V., Hughes, M., Porter-Smith, R., Beaman, R.J., Hemer, M., Daniell, J., Buchanan, C., Watson, T., Collins, D. (2002). Cross-shelf Sediment Transport in the Torres Strait-Gulf of Papua Region. *Geoscience Australia Survey*, 234, p.132.

Haswell-Elkins, M., Satarug, S., O'Rourke, P., Moore, M., Ng, J., McGrath, V., Walmsby, M. (2008). Striking association between urinary cadmium level and albuminuria among Torres Strait Islander people with diabetes. *Environmental Research*, 106(3), 379-383.

Haynes, D. (2001). Pesticide and heavy metal concentrations in Great Barrier reef sediment, seagrass and dugongs (*Dugong dugon*). Unpublished PhD thesis, University of Queensland. 187pp.

- Haynes, D., Johnson, J.E. (2000). Organochlorine, heavy metal and polyaromatic hydrocarbon pollutant concentrations in the Great Barrier Reef (Australia) environment: A review. *Marine Pollution Bulletin*, 41(7–12), 267-278.
- Haynes, D., Kwan, D. (2001). Trace metal concentrations in the Torres Strait environment and traditional seafood species, 1997–2000. Report prepared for the Torres Strait Regional Authority, Thursday Island, Queensland Australia.
- Haynes, D., Kwan, D. (2002). Trace metals in sediments from Torres Strait and the Gulf of Papua: concentrations, distribution and water circulation patterns. *Marine Pollution Bulletin*, 44(11), 1309-1313.
- Haywood, M.D.E., Browne, M., Skewes, T., Rochester, W., McLeod, I., Pitcher, R., Dennis, D., Dunn, J., Cheers, S., Wassenberg, T. (2007). Improved knowledge of Torres Strait seabed biota and reef habitats. Report to the Marine and Tropical Sciences Research Facility. Reef and Rainforest Research Centre Limited, Cairns, 115pp.
- Haywood, M.D.E., Pitcher, C.R., Ellis, N., Wassenberg, T. J., Smith, G., Forcey, K., McLeod, I., Carter, A., Strickland, C., Coles, R. (2008). Mapping and characterisation of the inter-reefal benthic assemblages of the Torres Strait. *Continental Shelf Research*, 28, 2304-2316.
- Hedley, J.D., McMahon, K., Fearn, P. (2014). Seagrass canopy photosynthetic response is a function of canopy density and light environment: a model for *Amphibolis griffithii*. *PloS ONE*, 9(10), p.e111454.
- Hemer, M., Harris, P.T., Coleman, R., Hunter, J. (2004). Sediment mobility due to currents and waves in the Torres Strait–Gulf of Papua region. *Continental Shelf Research*, 24(9), 2297-2316. doi: 10.1016/j.csr.2004.07.011.
- Hughes, T.P., Kerry, J.T., Álvarez-Noriega, M., Álvarez-Romero, J.G., Anderson, K.D., Baird, A.H., Babcock, R.C., Beger, M., Bellwood, D.R., Berkemans, R., Bridge, T.C. (2017). Global warming and recurrent mass bleaching of corals. *Nature*, 543(7645), 373.
- Hughes, T., Kerry, J., Simpson, T. (2018a). Large-scale bleaching of corals on the Great Barrier Reef. *Ecology*, 99, 501-501.
- Hughes, T.P., Kerry, J.T., Baird, A.H., Connolly, S.R., Dietzel, A., Eakin, C.M., Heron, S.F., Hoey, A.S., Hoogenboom, M.O., Liu, G., McWilliam, M.J. (2018b). Global warming transforms coral reef assemblages. *Nature*, 556(7702), 492.
- Humanes, A., Ricardo, G.F., Willis, B.L., Fabricius, K.E., Negri, A.P. (2017a). Cumulative effects of suspended sediments, organic nutrients and temperature stress on early life history stages of the coral *Acropora tenuis*. *Scientific Reports*, 7, 44101.
- Humanes, A., Fink, A., Willis, B.L., Fabricius, K.E., de Beer, D., Negri, A.P. (2017). Effects of suspended sediments and nutrient enrichment on juvenile corals. *Marine Pollution Bulletin*, 125(1-2), 166-175.
- International Union for Conservation of Nature. (1995). The Fly River Catchment, Papua New Guinea – A Regional Environmental Assessment. Published in collaboration with the Department of Environment and Conservation, Boroko, Papua New Guinea. IUCN, Gland, Switzerland and Cambridge, UK. 96pp.
- Johnson, J.E., Welch, D.J., Marshall, P.A., Day, J., Marshall, N., Steinberg, C.R., Benthuyssen, J.A., Sun, C., Brodie, J., Marsh, H., Hamann, M., Simpfendorfer, C. (2018). Characterising the values and connectivity of the northeast Australia seascape: Great Barrier Reef, Torres Strait,

Coral Sea and Great Sandy Strait. Report to the National Environmental Science Program. Reef and Rainforest Research Centre Limited, Cairns, 81pp.

Johnston, E.L., Roberts, D.A. (2009). Contaminants reduce the richness and evenness of marine communities: A review and meta-analysis. *Environmental Pollution*, 157, 1745-1752.

Jones, R., Ricardo, G.F., Negri, A.P., (2015). Effects of sediments on the reproductive cycle of corals. *Marine Pollution Bulletin*, 100 (1), 13–33.

Kilminster, K., McMahon, K., Waycott, M., Kendrick, G.A., Scanes, P., McKenzie, L., O'Brien, K.R., Lyons, M., Ferguson, A., Maxwell, P., Glasby, T., Udy, J. (2015). Unravelling complexity in seagrass systems for management: Australia as a microcosm. *Science of the Total Environment*, 534, 97–109.

Knutson, D.W., Buddemeier, R.W., Smith, S.V. (1972). Coral chronometers: seasonal growth bands in reef corals. *Science*, 177, 270-272.

Lane, D.J. (1991). 16S/23S rRNA Sequencing, In: *Nucleic Acid Techniques in Bacterial Systematics*, eds. E. Stackebrandt, M. Goodfellow. New York, NY: John Wiley & Sons, pp. 115-175.

Larcombe, P., Costen, A., Wolfe, K.J. (2001). The hydrodynamic and sedimentary setting of nearshore coral reefs, central Great Barrier Reef shelf, Australia: Paluma Shoals, a case study. *Sedimentology*, 48, 811-835.

Lawes, J.C., Neilan, B.A., Brown, M. V, Clark, G.F., Johnston, E.L. (2016). Elevated nutrients change bacterial community composition and connectivity: High throughput sequencing of young marine biofilms. *Biofouling*, 32, 57-69. <https://doi.org/10.1080/08927014.2015.1126581>

Lawrey, E. P., Stewart M. (2016). Mapping the Torres Strait reef and island features - Extending the GBR features (GBRMPA) dataset. Report to the National Environmental Science Programme. Reef and Rainforest Research Centre Limited, Cairns, 117pp.

Li, Y., Martins, F., Wolanski, E. (2017). Sensitivity analysis of the physical dynamics of the Fly River plume in Torres Strait. *Estuarine, Coastal and Shelf Science*, 194, 84-91.

Livingston, R.J., McGlynn, S.E., Niu, X. (1998). Factors controlling seagrass growth in a gulf coastal system: water and sediment quality and light. *Aquatic Botany*, 60, 135-159.

Long, B., Poiner, I. (1994). Infaunal benthic community structure and function in the Gulf of Carpentaria, northern Australia. *Marine and Freshwater Research*, 45, 293.

Longstaff, B.J., Dennison, W.C. (1999). Seagrass survival during pulsed turbidity events: the effects of light deprivation on the seagrasses *Halodule pinifolia* and *Halophila ovalis*. *Aquatic Botany*, 65, 105-121.

Lough, J.M. (2010). Climate records from corals. *Wiley Interdisciplinary Reviews – Climate Change*. doi: 10.1002/wcc.39.

Lough, J.M. (2016). Coral core records of the north-east Torres Strait. Report to the National Environmental Science Programme. Reef and Rainforest Research Centre Limited, Cairns, 24pp. <http://nesptropical.edu.au/wp-content/uploads/2017/01/NESP-TWQ-2.2.1-INTERIM-REPORT-1.pdf>

Lough, J.M., Cooper, T.F. (2011). New insights from coral growth band studies in an era of rapid environmental change. *Earth-Science Reviews*, 108, 170-184.

- Lough, J.M., Barnes, D.J., McAllister, F.A. (2002). Luminescent lines in corals from the Great Barrier Reef provide spatial and temporal records of reefs affected by land runoff. *Coral Reefs*, 21, 333-343.
- Lough, J.M., Lewis, S.E., Cantin, N.E. (2015). Freshwater impacts in the central Great Barrier Reef: 1648-2011. *Coral Reefs*, 34, 739-751.
- Macdonald, R.K., Ridd, P.V., Whinney, J.C., Larcombe, P., Neil, D.T. (2013). Towards environmental management of water turbidity within open coastal waters of the Great Barrier Reef. *Marine Pollution Bulletin*, 74(1), 82-94.
- Mallela, J., Roberts, C., Harrod, C., Goldspink, C.R. (2007). Distributional patterns and community structure of Caribbean coral reef fishes within a river-impacted bay. *Journal of Fish Biology*, 70, 523-537.
- Marín-Guirao, L., Atucha, A.M., Barba, J.L., López, E.M., García Fernández, A.J. (2005). Effects of mining wastes on a seagrass ecosystem: Metal accumulation and bioavailability, seagrass dynamics and associated community structure. *Marine Environmental Research*, 60, 317-337. <https://doi.org/10.1016/j.marenvres.2004.11.002>
- Marsh, H., Kwan, D. (2008). Temporal variability in the life history and reproductive biology of female dugongs in Torres Strait: the likely role of sea grass dieback. *Continental Shelf Research*, 28(16), 2152-2159.
- Martins, F., Wolanski, E. (2015). The pattern and intrusion of the Fly River flood plume to the Gulf of Papua and the Torres Strait: Preliminary numerical modelling results. TropWATER Report 38/15, James Cook University, Townsville, Australia, 25 pp.
- McKenzie, L., Yoshida, R., Grech, A., Coles, R. (2010). Queensland seagrasses. Status 2010 – Torres Strait and East Coast. Fisheries Queensland (DEEDI), Cairns, 6pp.
- McMahon, K., Collier, C., Lavery, P.S. (2013). Identifying robust bioindicators of light stress in seagrasses: a meta-analysis. *Ecological Indicators*, 30, 7-15.
- McMurdie, P.J., Holmes, S. (2014). Waste not, want not: Why rarefying microbiome data is inadmissible. *PLoS Computational Biology*, 10, e1003531.
- Menegário, A.A., Yabuki, L.N.M., Luko, K.S., Williams, P.N., Blackburn, D.M. (2017). Use of diffusive gradient in thin films for in situ measurements: a review on the progress in chemical fractionation, speciation and bioavailability of metals in waters. *Analytica Chimica Acta*, 983, 54-66.
- Millman J.D., Meade, R.H. (1983) World-wide delivery of river sediments to the oceans. *Journal of Geology*, 91, 1-21.
- Millman, J.D., Syvitski, J.P.M. (1992). Geomorphic/ tectonic control of sediment discharge to the ocean: the importance of small mountainous rivers. *Journal of Geology*, 100, 525-544.
- Moreton, B., Ratti, F., Fernandez, J-M, Laurent, A. (2017). Development and use of an automatic sequential passive sampler for the monitoring of dissolved metals in marine environments. Presented at: DGT2017 6 - 8 September 2017 Gold Coast, Australia. <http://ael-environnement.nc/wp-content/uploads/2018/08/Moreton-et-al-2017c-3.pdf>
- Munksgaard, N.C., Parry, D.L. (2001). Trace metals, arsenic and lead isotopes in dissolved and particulate phases of North Australian coastal and estuarine seawater. *Marine Chemistry*, 75(3), 165-184.

- NOAA (2013). Table of climate indices. Accessed on March 22, 2017; <http://www.cpc.ncep.noaa.gov/data/indices/ersst3b.nino.mth.81-10.ascii>.
- Núñez-Nogueira, G., Pérez-López, A. and Santos-Córdova, J.M. (2019). As, Cr, Hg, Pb, and Cd Concentrations and Bioaccumulation in the Dugong *Dugong dugon* and Manatee *Trichechus manatus*: A Review of Body Burdens and Distribution. *International Journal of Environmental Research and Public Health*, 16(3), p.404.
- O'Brien, D., Mellors, J., Petus, C., Martins, F., Wolanski, E., Brodie, J. (2015). Monitoring and assessment of water quality threats to Torres Strait marine waters and ecosystems. Progress Report August 2015 Report No. 15/43. Townsville, James Cook University.
- O'Brien, K.R., Adams, M.P., Ferguson, A.J.P., Villarreal-Samper, J., Maxwell, P.S., Baird, M.E., Collier, C.J. (2018). Chapter 10 Seagrass resistance to light deprivation. In (Eds) A.W.D. Larkum, G. Kendrick and P.J. Ralph. *Seagrasses of Australia: Structure, Ecology and Conservation*, pp287-311. Springer, Cham. ISBN 978-3-319-71354-0.
- O'Brien, K.R., Waycott, M., Maxwell, P., Kendrick, G.A., Udy, J.W., Ferguson, A.J.P., Kilminster, K., Scanes, P., McKenzie, L.J., McMahon, K., Adams, M.P., Samper-Villarreal, J., Collier, C., Lyons, M., Mumby, P.J., Radke, L., Christianen, M.J.A., Dennison, W.C. (2018). Seagrass ecosystem trajectory depends on the relative timescales of resistance, recovery and disturbance. *Marine Pollution Bulletin* 134, 166-176.
- Ogston, A. S., Sternberg, R. W., Nittrouer, C. A., Martin, D. P., Goní, M. A., Crockett, J. S. (2008). Sediment delivery from the Fly River tidally dominated delta to the nearshore marine environment and the impact of El Niño, *Journal of Geophysical Research*, 113, F01S11, doi:10.1029/2006JF000669.
- Oksanen, J., Blanchet, F.G., Kindt, R., Legendre, P., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Wagner, H. (2016). *vegan: community ecology package*. R package version 2.0–7. 2013. URL <http://CRAN.R-project.org/package=vegan>.
- Osborne, K., Miller, I., Johns, K., Jonker, M., Sweatman, H. (2013). Preliminary report on surveys of biodiversity of fishes and corals in Torres Strait. Australian Institute of Marine Science, Townsville, Australia.
- Pandolfi, J.M., Bradbury, R.H., Sala, E., Hughes, T.P., Bjorndal, K.A., Cooke, R.G., McArdle, D., McClenachan, L., Newman, M.J.H., Paredes, G., Warner, R.R., Jackson, J.B.C. (2003). Global trajectories of the long-term decline of coral reef ecosystems. *Science*, 301, 955.
- Patel, P., Bielmyer-Fraser, G.K., (2015). The influence of salinity and copper exposure on copper accumulation and physiological impairment in the sea anemone, *Exaiptasia pallida*. *Comparative Biochemistry & Physiology* 168, 39–47.
- Peters, E.C., Gassman, N.J., Firman, J.C., Richmond, R.H., Power, E.A. (1997). Ecotoxicology of tropical marine ecosystems 16(1), 12–40. doi:10.1002/etc.5620160103.
- Petus, C. (2013). Application of MODIS remote sensing imagery for monitoring turbid river plumes from PNG in the Torres Strait region: A test study. Report to the National Environmental Research Program. Reef and Rainforest Research Centre Limited, Cairns, 20pp.
- Petus, C., Collier, C., Devlin, M., Rasheed, M., McKenna, S. (2014). Using MODIS data for understanding changes in seagrass meadow health: a case study in the Great Barrier Reef (Australia). *Marine Environment Research*, 98, 68–85.
- Petus, C., Devlin, M., Thompson, A., McKenzie, L., Collier, C., Teixeira da Silva, E., Tracey, D. (2016). Estimating the exposure of coral reefs and seagrass meadows to land-sourced

contaminants in river flood plumes of the Great Barrier Reef: Validating a simple satellite risk framework with environmental data, 2016. Remote Sensing, 8(3), 210. doi:10.3390/rs8030210

Petus, C., Waterhouse, J., Brodie, J. (2018). Using MODIS satellite imagery to provide a large-scale baseline of the composition of coastal waters and the Fly River plume in the Gulf of Papua - Torres Strait (GPTS) regions, Technical Report to the National Environmental Research Program, Tropical System Hub, 32pp.

Pickup, G., Marshall, A.R. (2009). Geomorphology, hydrology, and climate of the Fly River system. In: The Fly River Papua New Guinea. Environmental Studies in an Impacted Tropical River System, ed. B. R. Bolton. Burlington, MA, USA: Elsevier, pp. 3-49.

Poiner, I.R., Peterkin, C. (1996). Seagrasses. In: The State of the Marine Environment Report for Australia. Technical Annex: 1, eds. L.P. Zann, P. Kailola, P. Townsville, Australia: Great Barrier Reef Marine Park Authority, pp. 40-45.

Pratchett, M.S., Hoogenboom, M.O. (2019). Disturbances and pressures to coral reefs. The Great Barrier Reef: Biology, Environment and Management, p.131.

PythonWin 2.7.3 (default, Apr 10 2012). Python IDE and GUI Framework for Windows. Copyright 1994-2008 Mark Hammond. Python is Copyright (c) 2001-2012 Python Software Foundation.

Quast, C., Pruesse, E., Yilmaz, P., Gerken, J., Schweer, T., Yarza, P., Peplies, J., Glöckner, F.O. (2013). The SILVA ribosomal RNA gene database project: Improved data processing and web-based tools. Nucleic Acids Research, 41, 590-596.

R Core Team (2016). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna. <https://www.R-project.org/>

Reichelt-Brushett, A.J., Harrison, P.L. (2000). The effect of copper on the settlement success of larvae from the scleractinian coral *Acropora tennisi*. Marine Pollution Bulletin 41(7–12), 385–391.

Reichelt-Brushett, A.J., Harrison, P.L. (2005). The effects of selected trace metals on fertilization success of several scleractinian coral species. Coral Reefs 24, 524–534.

Reichelt-Brushett, A., Hudspeth, M. (2016). The effects of metals of emerging concern on the fertilisation success of gametes of the tropical scleractinian coral *Platygyra daedalea*. Chemosphere, 150, 398-406.

Richmond, R.H., Tisthammer, K.H. and Spies, N.P., (2018). The effects of anthropogenic stressors on reproduction and recruitment of corals and reef organisms. Frontiers in Marine Science, 5, p.226.

Robert J. H., Williams, E., Vennes, C. (2017). geosphere: Spherical Trigonometry. r Packag. version 1.5-5 <https://cran.r-project.org/web/packages/geosphere>.

Robertson, A.I., Daniel, P.A., Dixon, P., Alongi, D.M. (1993). Pelagic biological processes along a salinity gradient in the Fly delta and adjacent river plume (Papua New Guinea). Continental Shelf Research, 13, 205-224.

Rodriguez, I.B., Lin, S., Ho, J., Ho, T.Y. (2016). Effects of trace metal concentrations on the growth of the coral endosymbiont *Symbiodinium kawagutii*. Frontiers in Microbiology, 7, p.82. DOI=10.3389/fmicb.2016.00082.

- Sabdon, A. (2009). Heavy metal levels and their potential toxic effect on coral *Galaxea fascicularis* from Java Sea, Indonesia. *Research Journal of Environmental Sciences* 3(1), 96–102.
- Saint-Cast, F., Condie, S.A., Geoscience Australia (2006). Circulation modelling in Torres Strait. Geoscience Australia, Canberra, 2006. Saint-Cast, F. (2008). Multiple time-scale modelling of the circulation in Torres Strait-Australia. *Continental Shelf Research*, 28, 2214–2240.
- Santmire, J.A., Leff, L.G. (2007). The influence of stream sediment particle size on bacterial abundance and community composition. *Aquatic Ecology*, 41, 153–160.
- Schaffelke, B., Thompson, A., Carleton, J., Davidson, J., Doyle, J., Furnas, M., Gunn, K., Skuza, M., Wright, M., Zagorskis, I. (2009). Reef Rescue Marine Monitoring Program: Final report of AIMS activities 2008/09. Great Barrier Reef Marine Park Authority, Townsville. Schimel, J.P., Balser, T.C., Wallenstein, M. (2007). Microbial stress-response physiology and its implications for ecosystem function. *Ecology*, 88, 1386–1394. <https://doi.org/10.1890/06-0219>
- Schliep, M., Pernice, M., Sinutok, S., Bryant, C.V., York, P.H., Rasheed, M.A., Ralph, P.J. (2015). Evaluation of reference genes for RT-qPCR studies in the seagrass *Zostera muelleri* exposed to light limitation. *Scientific Reports*, 5, Art. 17051. doi:10.1038/srep17051.
- Schwarz, J.A., Mitchelmore, C.L., Jones, R., O'Dea, A., Seymour, S. (2013). Exposure to copper induces oxidative and stress responses and DNA damage in the coral *Montastraea franksi*. *Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology*, 157(3), 272–279.
- Siddiqui, S., Bielmyer-Fraser, G.K., (2015). Responses of the sea anemone, *Exaiptasia pallida*, to ocean acidification conditions and copper exposure. *Aquat. Toxicol.* 167, 228–239.
- Smith, J.G., Brandt, C.C., Christensen, S.W. (2011). Long-term benthic macroinvertebrate community monitoring to assess pollution abatement effectiveness. *Environmental Management*, 47, 1077–1095.
- Smith, I., Moise, A., Inape, K., Murphy, B., Colman, R., Power, S., Chung, C. (2013). ENSO-related rainfall changes over the New Guinea region. *Journal of Geophysical Research: Atmospheres*, 118(19).
- Sobtzick, S., Hagihara, R., Penrose, H., Grech, A., Cleguer, C., Marsh, H. (2014). An assessment of the distribution and abundance of dugongs in the Northern Great Barrier Reef and Torres Strait. A Report for the Department of the Environment, National Environmental Research Program (NERP). August 2014.
- Statton, J., McMahon, K., Lavery, P., Kendrick, G.A. (2018). Determining light stress responses for a tropical multi-species seagrass assemblage. *Marine Pollution Bulletin* 128, 508–518.
- Stauber, J.L., Apte, S.C., Rogers, N. (2009). Speciation, bioavailability and toxicity of copper in the Fly River system. In: *The Fly River Papua New Guinea. Environmental Studies in an Impacted Tropical River System*, ed. B. R. Bolton. Burlington, MA, USA: Elsevier, pp. 375–408.
- Sun, M.Y., Dafforn, K.A., Brown, M. V, Johnston, E.L. (2012). Bacterial communities are sensitive indicators of contaminant stress. *Marine Pollution Bulletin*, 64, 1029–1038.

- Sweatman, H., Johns, K., Jonker, M., Miller, I., Osborne, K. (2015). Report on second monitoring field trip to Torres Strait, January 2014. Report to the National Environmental Research Program. Reef and Rainforest Research Centre Limited, Cairns, 10 + ivpp.
- Sweatman, H., Berkelmans, R. (2012). Summary of existing information on coral reef communities of the Torres Strait. NERP Project 2.3 Report, Australian Institute of Marine Science, Townsville, 7pp.
- Thomas, C.R., Bennett, W., Simmonds, A., Turner, R., Honchin, C., Madden Hof, C.A., Bell, I., Brodie, J.E. (In review). Metal screening in green turtle forage, Great Barrier Reef, Australia. Science of the Total Environment.
- Torres Strait Regional Authority (TSRA) (2016). Torres Strait Long Term Coral Reef Monitoring Project 2015-2016. Author: Simpson, T. Published by the Torres Strait Regional Authority Land and Sea Management Unit, Thursday Island, Queensland. (56pp).
- Thomas, C.R., Bennett, W., Simmonds, A., Turner, R., Honchin, C., Madden Hof, C.A., Bell, I., Brodie, J.E. (2018). Metal screening in turtle forage, sediment, seagrass and river water of the Great Barrier Reef, pp 37 – 78. Chapter 3 In: Rivers to Reef to Turtles Project Final Report, WWF, Australia. 192pp. https://www.wwf.org.au/ArticleDocuments/394/WWF_RRT.pdf.aspx
- Thompson, A., Costello, P., Davidson, J., Logan, M., Coleman, G., Gunn, K., Schaffelke, B. (2017). Marine monitoring program. Annual Report for inshore coral reef monitoring: 2015 to 2016. Report for the Great Barrier Reef Marine Park Authority, Great Barrier Reef Marine Park Authority, Townsville, 133pp.
- Turner, S., Pryer, K.M., Miao, V.P., Palmer, J.D. (1999). Investigating deep phylogenetic relationships among cyanobacteria and plastids by small subunit rRNA sequence analysis. Journal of Eukaryotic Microbiology, 46, 327-338.
- Villa, C.A., Flint, M., Bell, I., Hof, C., Limpus, C.J., Gaus, C. (2017). Trace element reference intervals in the blood of healthy green sea turtles to evaluate exposure of coastal populations. Environmental Pollution, 220, 1465-1476. <https://doi.org/10.1016/j.envpol.2016.10.085>.
- Vonk, J.A., Smulders, F.O., Christianen, M.J., Govers, L.L. (2018). Seagrass leaf element content: A global overview. Marine Pollution Bulletin, 134, 123-133.
- Walsh, J.P., Ridd, P.V. (2009). Processes, sediments and stratigraphy of the Fly River delta. In: The Fly River Papua New Guinea. Environmental Studies in an Impacted Tropical River System, ed. B. R. Bolton. Burlington, MA, USA: Elsevier, pp. 53-76.
- Walsh, J.P., Nittrouer, C.A., Palinkas, C.M., Ogston, A.S., Sternberg, R.W., Brunskill, G.J. (2004). Clinoform mechanics in the Gulf of Papua, New Guinea. Continental Shelf Research, 24, 2487-2510.
- Waterhouse, J., Brodie, J., Wolanski, E., Petus, C., Higham, W., Armstrong, T. (2013). Hazard assessment of water quality threats to Torres Strait marine waters and ecosystems. Report to the National Environmental Research Program. Reef and Rainforest Research Centre Limited, Cairns, 70pp.
- Waterhouse, J., Lønborg, C., Logan, M., Petus, C., Tracey, D., Lewis, S., Howley, C., Harper, E., Tonin, H., Skuza, M., Doyle, J., Costello, P., Davidson, J., Gunn, K., Wright, M., Zagorskis, I., Kroon, F., Gruber, R. (2018). Marine Monitoring Program: Annual Report for inshore water quality monitoring 2016-2017. Report for the Great Barrier Reef Marine Park Authority, Great Barrier Reef Marine Park Authority, Townsville, 153pp.

- Welch, D.J., Johnson, J.E. (2013). Assessing the vulnerability of Torres Strait fisheries and supporting habitats to climate change. Report to the Australian Fisheries Management Authority. C2O Fisheries, Australia, 114pp.
- Wells, J.N., Rasheed, M.A. (2017). Port of Townsville Annual Seagrass Monitoring and Baseline Survey: September - October 2016. James Cook University, Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER), Cairns, 54 pp.
- Wenger, A.S., Fabricius, K.E., Jones, G.P., Brodie, J.E. (2015). Effects of sedimentation, eutrophication, and chemical pollution on coral reef fishes. Chapter 15 in Ecology of fishes on coral reefs, p145. Cambridge University Press.
- Wenger, A.S., Harvey, E., Wilson, S., Rawson, C., Newman, S.J., Clarke, D., Saunders, B.J., Browne, N., Travers, M.J., McIlwain, J.L., Erftemeijer, P.L.A., Hobbs, J.-P.A., McLean, D., Depczynski, M., Evans, R.D. (2017). A critical analysis of the direct effects of dredging on fish. *Fish Fish.* <https://doi.org/10.1111/faf.12218>.
- Wickham, H. (2009). *ggplot: Using the Grammar of Graphics with R*. Springer, New York.
- Wolanski, E., Gibbs, R.J. (1995). Flocculation of suspended sediment in the Fly River estuary, Papua New Guinea. *Journal of Coastal Research*, 11, 754-762.
- Wolanski, E., Norro, A., King, B. (1995a). Water circulation in the Gulf of Papua. *Continental Shelf Research*, 15, 185-212.
- Wolanski, E., King, B., Galloway, D. (1995b). Dynamics of the turbidity maximum in the Fly River estuary, Papua New Guinea. *Estuarine Coastal Shelf Science*, 40, 321-337.
- Wolanski, E., King, B., Galloway, D. (1997). Salinity intrusion into the Fly River estuary, Papua New Guinea *Journal of Coastal Research*, 13, 185-212.
- Wolanski, E., Spagnol, S., King, B. (1999). Patchiness in the Fly River plume, Papua New Guinea. *Journal of Marine Systems*, 18, 369-381.
- Wolanski, E., Lambrechts, J., Thomas, C.J., Deleersnijder, E. (2013). A high-resolution model of the water circulation in Torres Strait. NERP project 4.4 Report. TropWATER, James Cook University, Townsville, Australia.
- Wolanski, E., Andutta, F., Deleersnijder, E., Li, Y., Thomas, C.J. (2017). The Gulf of Carpentaria heated Torres Strait and the Northern Great Barrier Reef during the 2016 mass coral bleaching event. *Estuarine, Coastal and Shelf Science*, 194, 172-181.
- Wood, N., Lavery, P. (2001). Monitoring seagrass ecosystem health – the role of perception in defining health and indicators. *Ecosystem Health*, 6(2), 134-148. <https://onlinelibrary.wiley.com/doi/abs/10.1046/j.1526-0992.2000.00015.x>
- Woods, R.M., Baird, A.H., Mizerek, T.L., Madin, J.S. (2016). Environmental factors limiting fertilisation and larval success in corals. *Coral Reefs*, 35(4), pp.1433-1440.
- Zhang, H., Davison, W. (1999). Diffusional characteristics of hydrogels used in DGT and DET techniques. *Analytica Chimica Acta*, 398, 329-340.

8. APPENDIX 1: SALINITY MONITORING TRAINING MANUAL

WEEKLY Torres Strait Salinity and Temperature Monitoring

Background

Thank you for taking part in this study of the intrusion of the Fly River plume into the Torres Strait.

Eric Wolanski and others at James Cook University have carried out field studies in the Torres Strait in past years and found that the Fly River plume reaches part of the Torres Strait region, particularly in the northern areas. However, the data is insufficient to determine whether this influence is frequent or rare. Mathematical models suggest that this influence is frequent during the trade wind season.

Your study is important to assess how strong and how common this influence is and to start assessing whether it presents a pollution risk to Torres Strait communities.

Dr Eric Wolanski and Dr Caroline Petus will compile the data from the various islands and email you a report of what the Fly River does in the Torres Strait this year.

This kit contains:

- An overview of the project
- Sampling instructions (this sheet)
- Sample recording sheet
- 2 x handheld salinity-temperature meters
- 2 x 1 L salinity standards
- 2 x jars for using the salinity standard
- A bucket and rope for collecting local seawater



SAMPLING INSTRUCTIONS

Salinity meters

You have been supplied with a salinity-temperature meter (AZ Instrument Corp. Water Quality Meter – Salinity meter, Model #8371). Each meter is labelled with a unique number—please record this number for your site.

The meter requires 4 × LR44 batteries, which are provided.

You have been given two meters, with one to be kept as a backup. The meters should be changed over after 6 months. Please send the used meter back to us for servicing or replacement; we will send you a reminder email.

Selecting your site

The monitoring site needs to be in at least 1 metre of water, and can be the end of a jetty or off the beach. The site should not be located close to any land based freshwater influence such as a stream or discharge area.

The position of the site should be carefully recorded using a GPS. The same site should be visited each time.

Sampling

Once a week, as much as possible on the same day of the week, collect seawater using a bucket at the selected site. The exact time of sampling is not important. Take the sample to a location where you can easily conduct the monitoring (e.g. a car park area or bench). It is best to avoid periods of heavy rainfall.

Write down the date, time, estimate of the tide and wind, and your name and contact details on the sample recording sheet (see example attached).

Remove the cap from the meter, turn on the meter using the top On/Off button.

Fill the small jar with the salinity standard from the 1 L bottle; dip the probe into the water to the top of the probe and measure the salinity once the reading is stable. Write that value on the sample



recording sheet. Throw away the water from the small container every second time you use it. If you are unsure, use a new standard.

Next, submerge the probe in the seawater sample in the bucket to the top of the probe. Read the temperature and the salinity result on the meter, and write down these values on the sample recording sheet. Then, turn off the meter, rinse it with freshwater, dry it with a tissue and put the cap back on for storage.



Submitting the data

Please submit your results through the TSRA Fulcrum system after each measurement.

If this is not accessible, take a photo of the sample record sheet at the end of each month that should contain four measurements (one measurement per week) and send it to Dr Eric Wolanski and Jane Waterhouse TropWATER, James Cook University.

E-mail: eric.wolanski@jcu.edu.au Mobile: 0408 897 107

jane.waterhouse@jcu.edu.au Mobile: 0409 053 367

If any salinity reading is below 33 ppt, please send us a text message or email with the value and location and we will review the satellite image for that time.

Questions or problems?

Do not hesitate to call Eric on 0408 897 107 or Jane on 0409 053 367 if you have any questions or issues with the equipment.

Thank you for your participation!

IMPORTANT TO KNOW!

Before you use the meter:

1. Put the probe in a cup or jar of tap water for 30 minutes before use. You don't need to turn the unit on to do this.

If you are using a new meter:

1. Put the batteries in from the box.
2. Put the probe in a cup or jar of tap water for 30 minutes before you use it. You don't need to turn the unit on to do this.

When you are taking the reading, make sure you stir the probe around in the water to remove any air bubbles.

9. APPENDIX 2: ADDITIONAL DETAILS OF THE EXPOSURE ANALYSIS

Appendix 2A: Cloud frequency

Cloud coverage interferes with image collection and can influence the results by providing a bias towards the period where there is lower cloud cover, i.e. less rainfall. Analysis of the cloud frequency over the complete time series (2008–2018, except 2011 when NASA had data issues) at the 10 selected habitat sites (see

Table 4-3 within the report) is shown in Figure A2-1, Figure A2-2 and Figure A2-3 and shows the highest cloud frequency at the Saibai and Boigu Islands and Warrior Reef sites, and greater cloud frequency in July–August (within the SE trade wind season) than in February–March (within the monsoon season) at most locations.

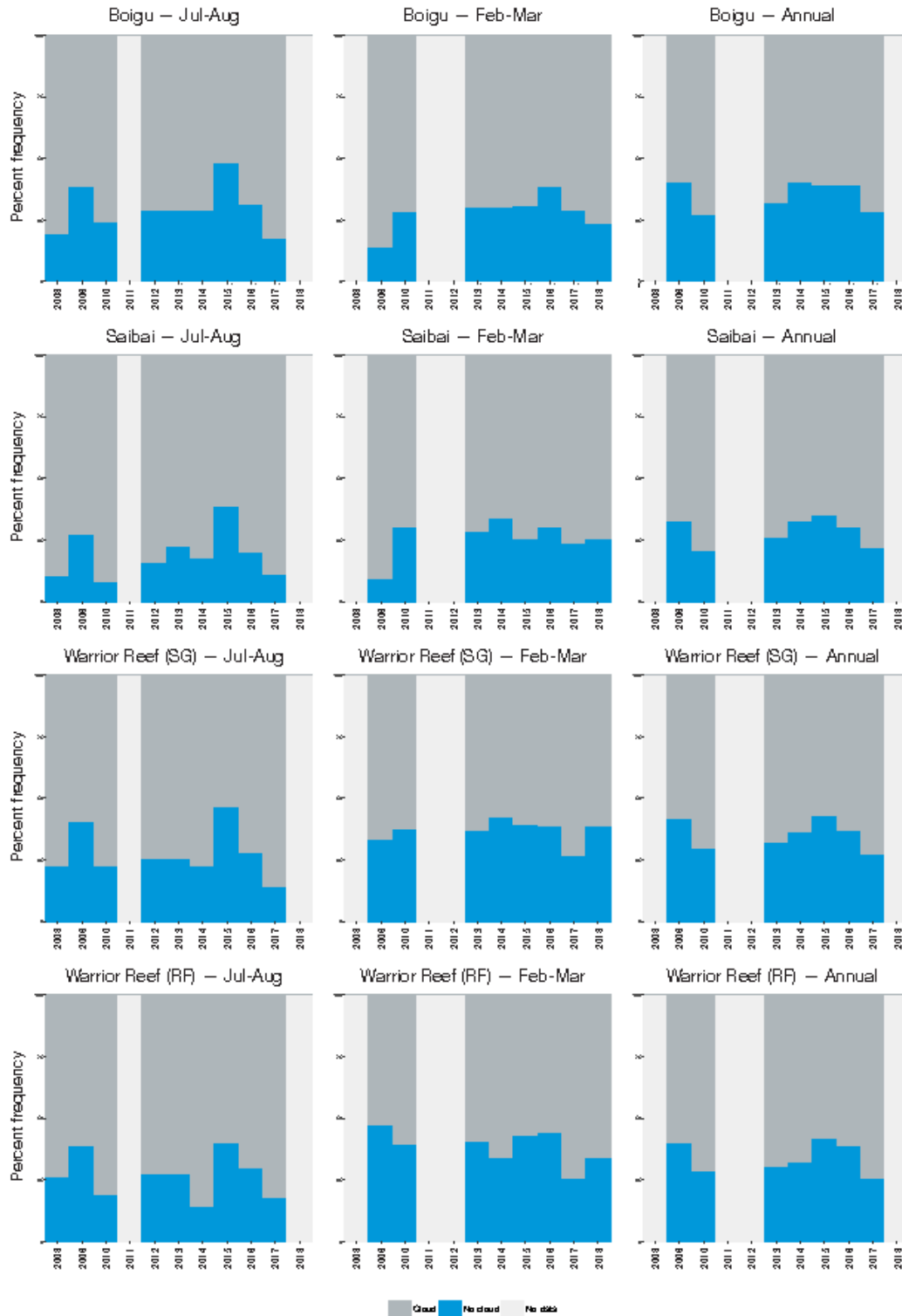


Figure A2-1: Cloud frequency (with clear skies shown in blue) at selected habitat sites in the NE Torres Strait from 2008 to 2018 during July–August (within the SE trade wind season), February–March (within the monsoon season) and annual. Site codes in brackets after each site name (TSRA 2016).

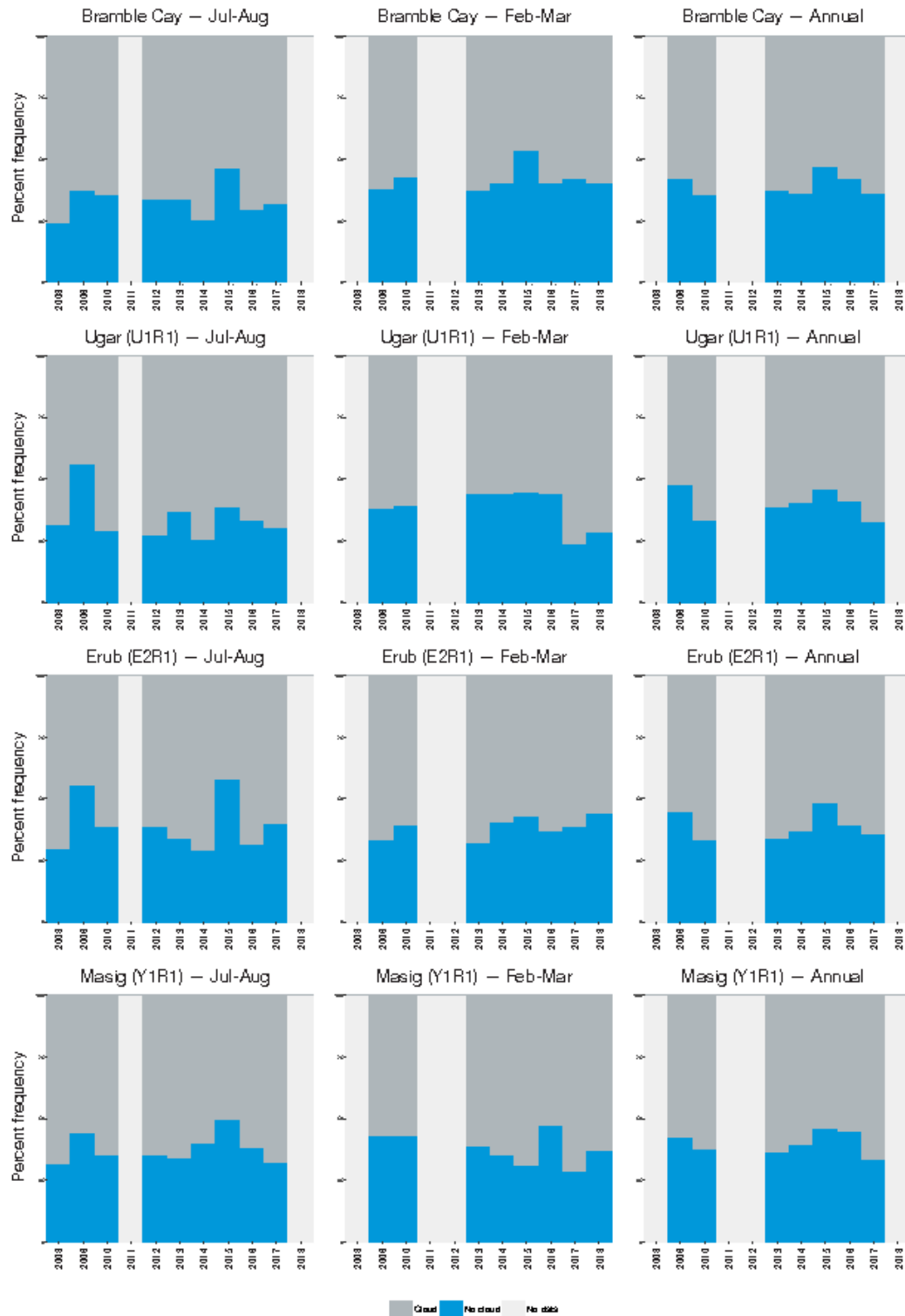


Figure A2-2: Cloud frequency (with clear skies shown in blue) at selected habitat sites in the NE Torres Strait from 2008 to 2018 during July–August (within the SE trade wind season), February–March (within the monsoon season) and annual. Site codes in brackets after each site name (TSRA 2016).



Figure A2-3: Cloud frequency (with clear skies shown in blue) at selected habitat sites in the NE Torres Strait from 2008 to 2018 during July–August (within the SE trade wind season), February–March (within the monsoon season) and annual. Site codes in brackets after each site name (TSRA 2016).

Appendix 2B: Coral reef and seagrass exposure (whole Torres Strait scale)

Analysis of the frequency of exposure to turbid waters (Primary and Secondary water types) across the Torres Strait region in different seasons and between years showed distinct variations, with higher frequency of exposure to turbid waters in some periods. Examples are provided below.

For **coral reefs** (Figure A2-4):

- While there was limited and infrequent exposure to Primary waters in all assessment periods, some exposure was evident in all years where data was available. This area of exposure was greatest in July–August 2008 and February–March 2009 and 2013.
- The area of exposure to Primary and Secondary waters was greatest in February–March for all years.
- While the annual exposure to Primary and Secondary waters was reasonably consistent between years, there was greater variability between years in February–March, with higher exposure evident in 2008, 2013 and 2015 and reduced exposure evident in 2016, 2017 and 2018 compared to other years. Similar patterns of higher exposure were observed during the July–August period but not the reduced exposure.

For **intertidal seagrass** (Figure A2-5):

- While there was limited and infrequent exposure to Primary waters in all assessment periods (more constrained than for coral reefs), some exposure was evident in all years where data was available. This area of exposure was greatest in July–August 2008 and February–March 2009 and 2013.
- As for coral reefs, the area of exposure to Primary and Secondary waters was greatest in February–March for all years.
- For July–August and February–March, the greatest frequency and areas of exposure occurred in 2008 and 2015. In February–March, reduced frequency exposure was evident in 2016, 2017 and 2018 compared to other years.

For **subtidal seagrass** (Figure A2-6):

- There was very limited exposure to Primary waters in all assessment periods.
- The area of exposure to Primary and Secondary waters was greatest in February–March for all years, except in July–August 2008 when the results were comparable.
- For July–August and February–March, the greatest frequency and areas of exposure occurred in 2008 and 2015. In February–March, reduced frequency of exposure was evident in 2017 and 2018 compared to other years.

Further analysis of these preliminary results is required to understand the potential environmental drivers of the seasonal and inter-annual variability in exposure. In particular, Fly River discharge data would be highly valuable.

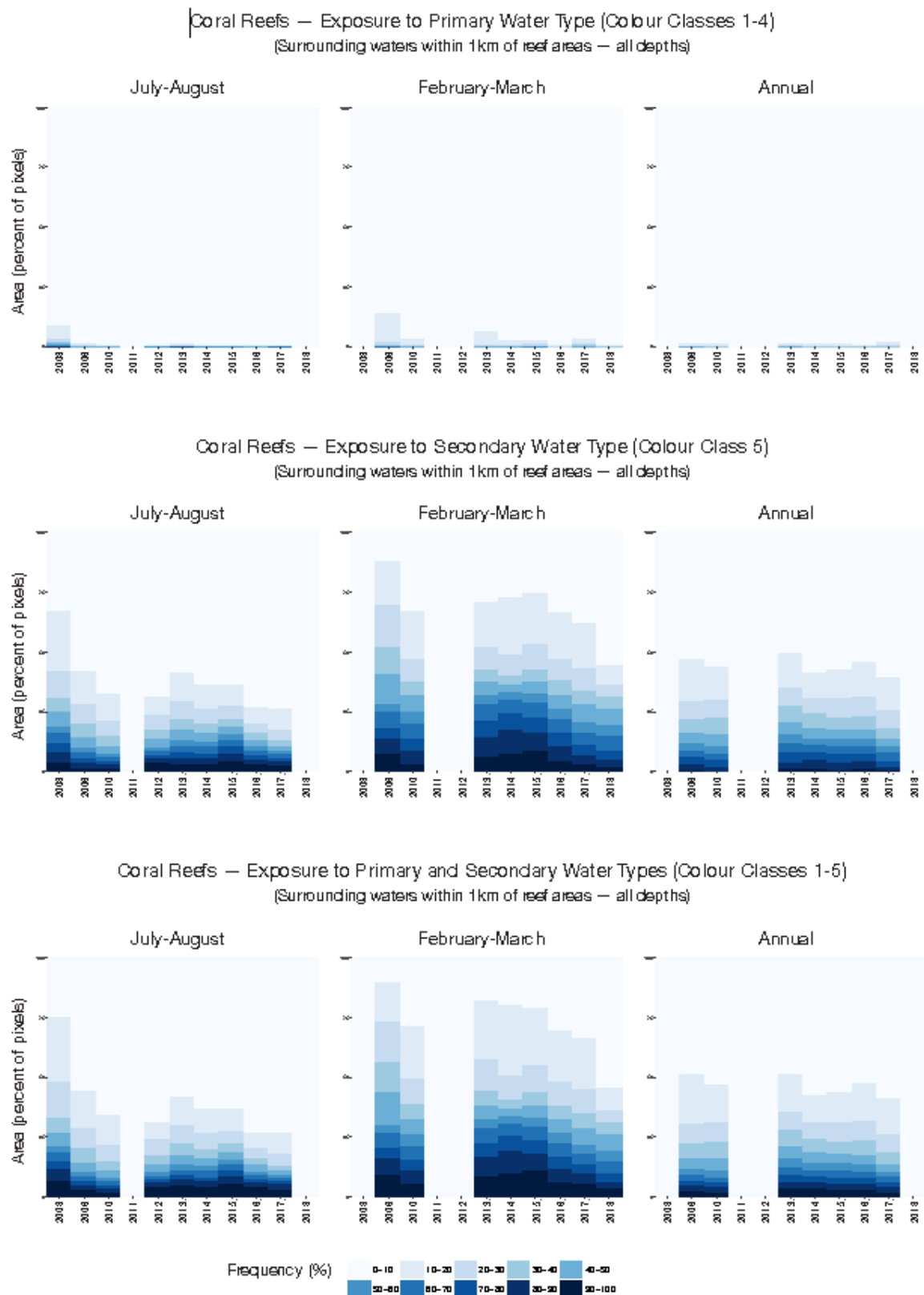


Figure A2-4: Areas (percent of pixels) of (first panel) coral reefs exposed to Primary (top), Secondary (centre) and Primary + Secondary (bottom) water types in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual.

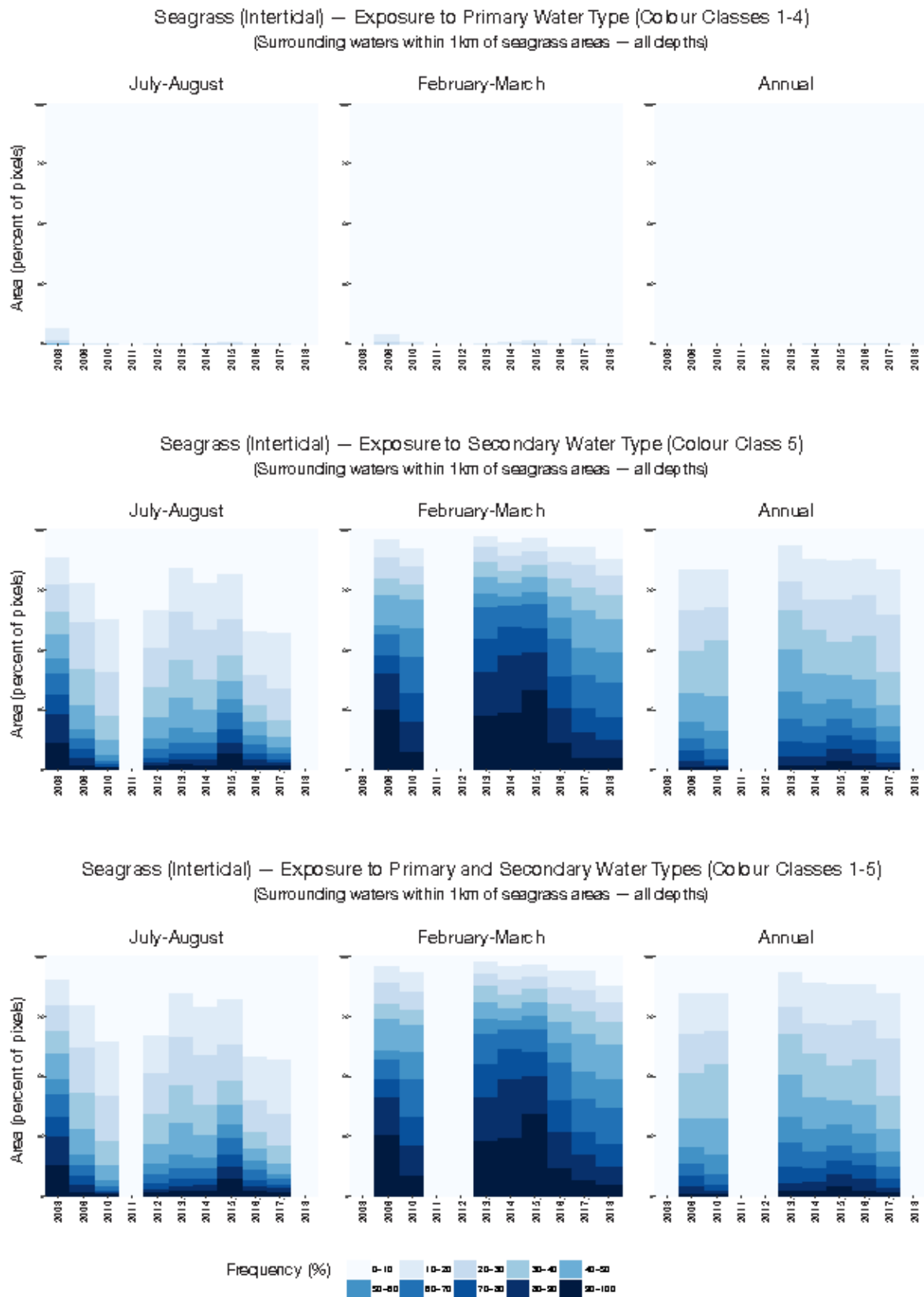


Figure A2-5: Areas (percent of pixels) of intertidal seagrass exposed to Primary (top), Secondary (centre) and Primary + Secondary (bottom) water types in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual.

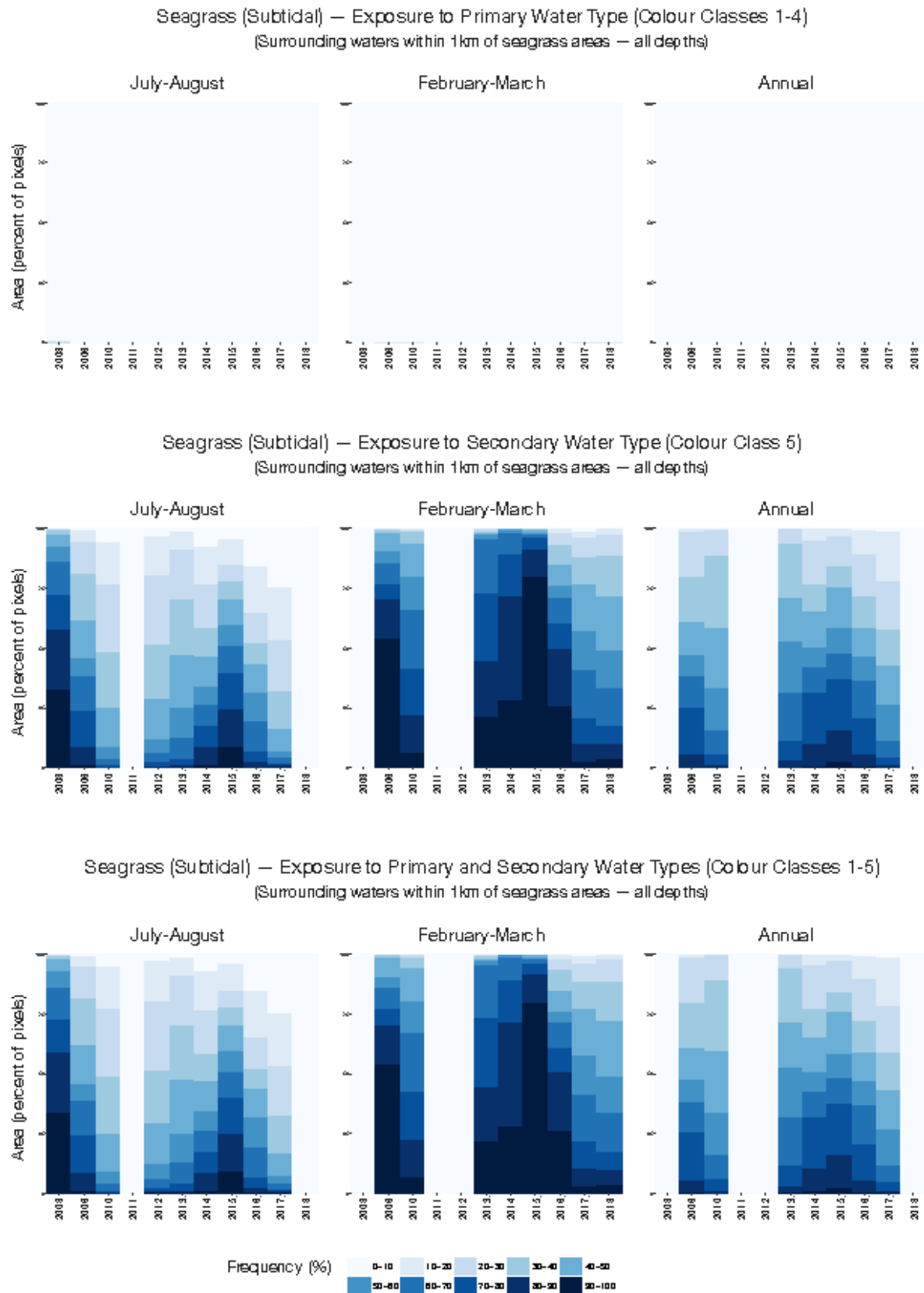


Figure A2-6: Areas (percent of pixels) of subtidal seagrass exposed to Primary (top), Secondary (centre) and Primary + Secondary (bottom) water types in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual.

Appendix 2C: Exposure at different habitat sites

Analysis of the frequency of exposure to the six colour classes at the 10 selected habitat sites (described in

Table 4-3 of the main report) in different seasons and between years (Figures A2-7, A2-8 and A2-9) showed distinct variations, with higher frequency of exposure to the more turbid colour classes (CC1 to CC5) at some sites and in some periods.

Examples are provided below.

- **Boigu and Saibai Islands** – similar patterns, both dominated by Primary and Secondary waters. Greatest frequency of exposure to Primary waters of all locations, with greatest exposure in February–March. Primary waters evident in all years (where data available) in February–March except for 2009. Some occurrence of Tertiary waters in July–August. Greatest variability between years in July–August.
- **Warrior Reefs (seagrass and reef sites)** – dominated by Secondary waters and, to a lesser extent, Tertiary waters, with some exposure to Primary waters in February–March in most years where data was available, and in July–August 2008 and 2015. Patterns between seasons and between years are reasonably consistent.
- **Bramble Cay** – dominated Tertiary and marine waters and, to a lesser extent, Secondary waters, with limited exposure to Primary waters in 2009 and 2013–2017 (when occurred, was greatest in February–March). Patterns between seasons and between years are reasonably consistent.
- **Ugar Island** – dominated by Tertiary waters and, to a lesser extent, Secondary waters, with very limited exposure to Primary waters in 2009 and 2013–2014 (occurred in February–March only). Patterns between seasons and between years are reasonably consistent.
- **Erub Island** - dominated by Secondary and Tertiary waters with limited exposure to Primary waters in February–March in 2009 and 2014–2017 (similar patterns to Bramble Cay) and some occurrence in July–August in 2008, 2012, 2013 and 2017. Dominated by Secondary waters in February–March. Greatest variability between years in July–August.
- **Masig Island** – dominated by Tertiary waters and to a lesser extent, Secondary waters, with very limited exposure to Primary waters in February–March in 2013–2014 and some occurrence in July–August 2008. Greatest variability between years in July–August.
- **Poruma Island** – dominated by Secondary waters, particularly in February–March, and Tertiary waters in July–August. Some limited exposure to Primary waters in February–March during 2014–2015 and 2017. Greatest variability between years in February–March.
- **Mer** – dominated by Secondary waters, particularly in February–March and, to a lesser extent, Tertiary waters, particularly in July–August. Some limited exposure to Primary waters in February–March in 2014, 2016 and 2018, and July–August in 2008, 2009 and 2012.

As above, further analysis of these preliminary results is required to understand the potential environmental drivers of the seasonal and inter-annual variability in exposure, which may include local wind and wave action and currents and could be done to some extent with the hydrodynamic model. Again, Fly River discharge data would be highly valuable.

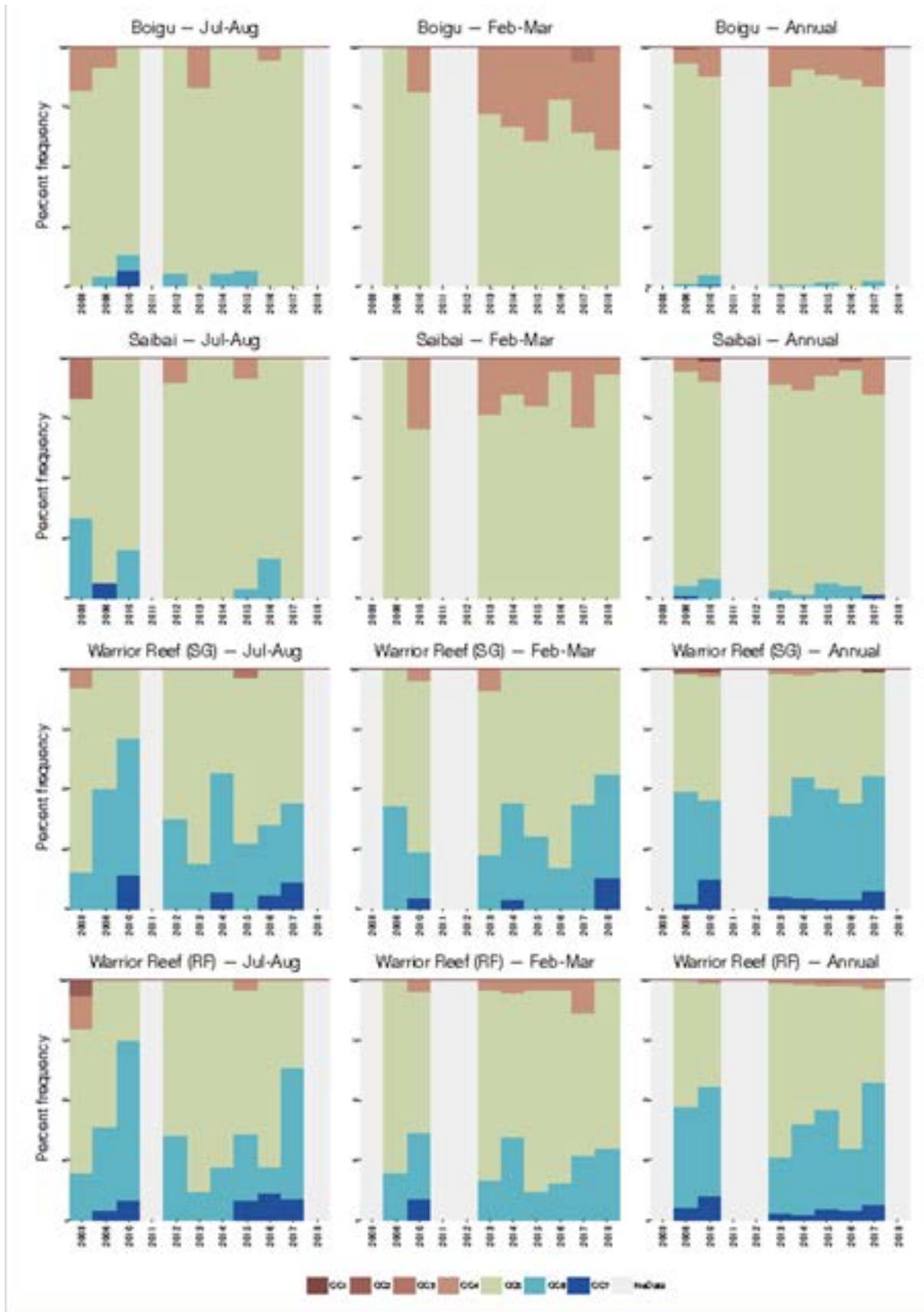


Figure A2-7: Frequency of exposure to different CCs at habitat sites at Saibai and Boigu Islands and Warrior Reefs in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).



Figure A2-8: Frequency of exposure to different CCs at habitat sites at Bramble Cay, and Ugar, Erub and Masig Islands in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).



Figure A2-9: Frequency of exposure to different CCs at habitat sites at Poruma Island and Mer in the northern Torres Strait from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).

Appendix 2D: Exposure in different potential zones of influence

Analysis of the frequency of exposure to the six colour classes in the six potential zones of influence (described in Section 4.6.1 of the main report) in different seasons and between years showed distinct variations, with higher frequency of exposure to the more turbid colour classes (CC1 to 5) at some zones and in some periods (Figures A2-10–A2-15).

These figures are best interpreted using the maps shown in Figure 4-30 of the main report.

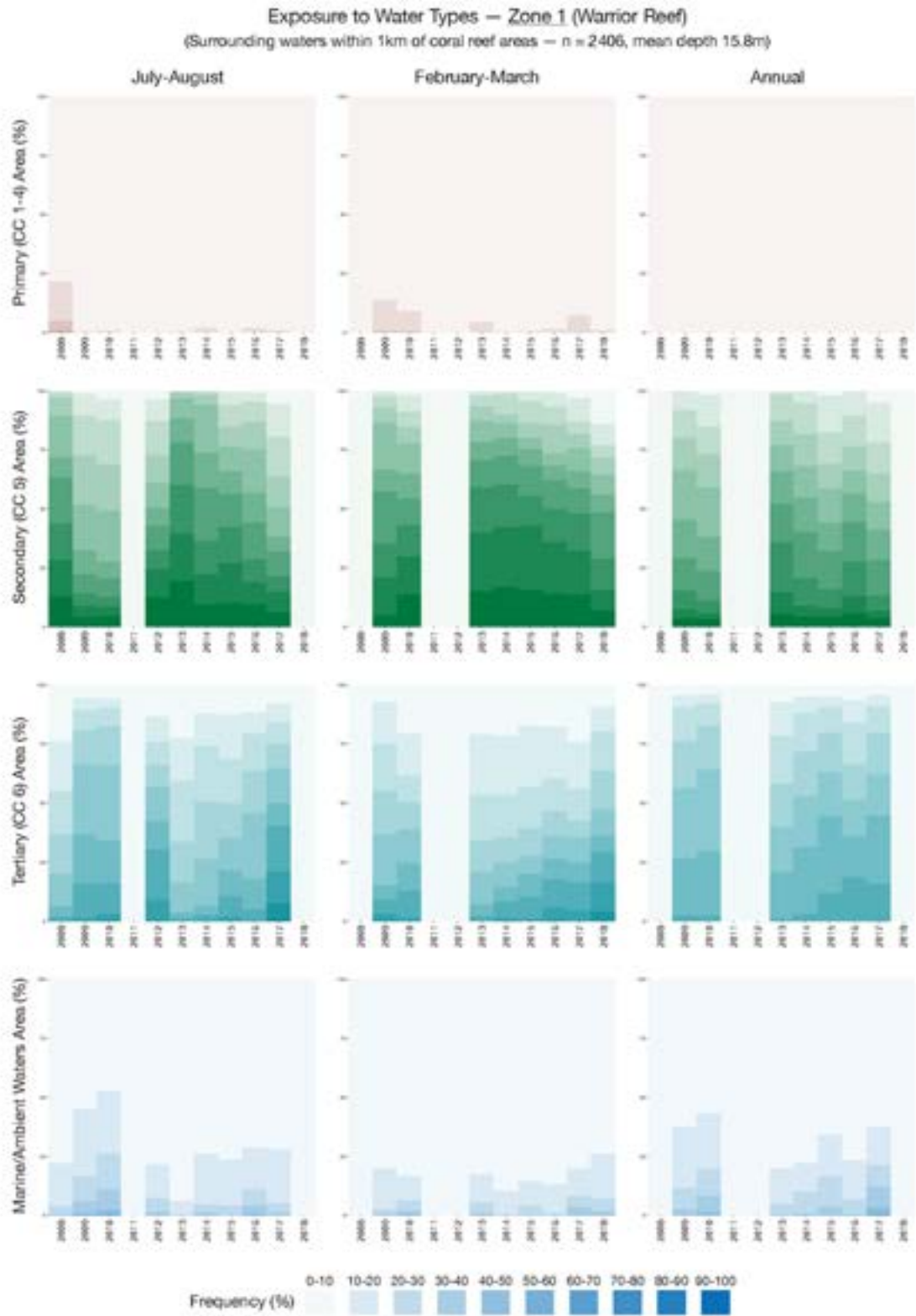


Figure A2-10: Frequency of exposure to different CCs in Zone 1 (North) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).

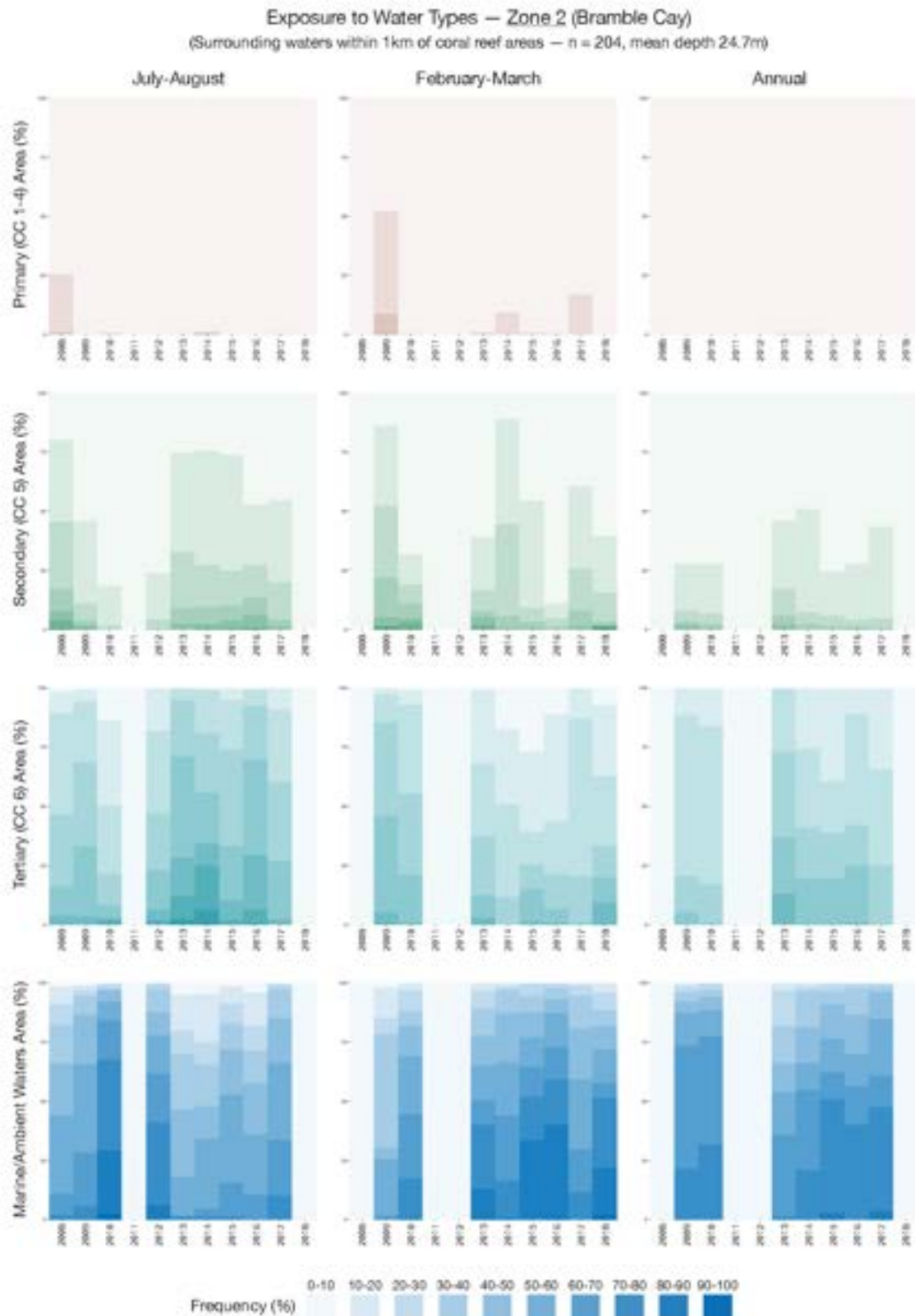


Figure A2-11: Frequency of exposure to different CCs in Zone 2 (North East) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).

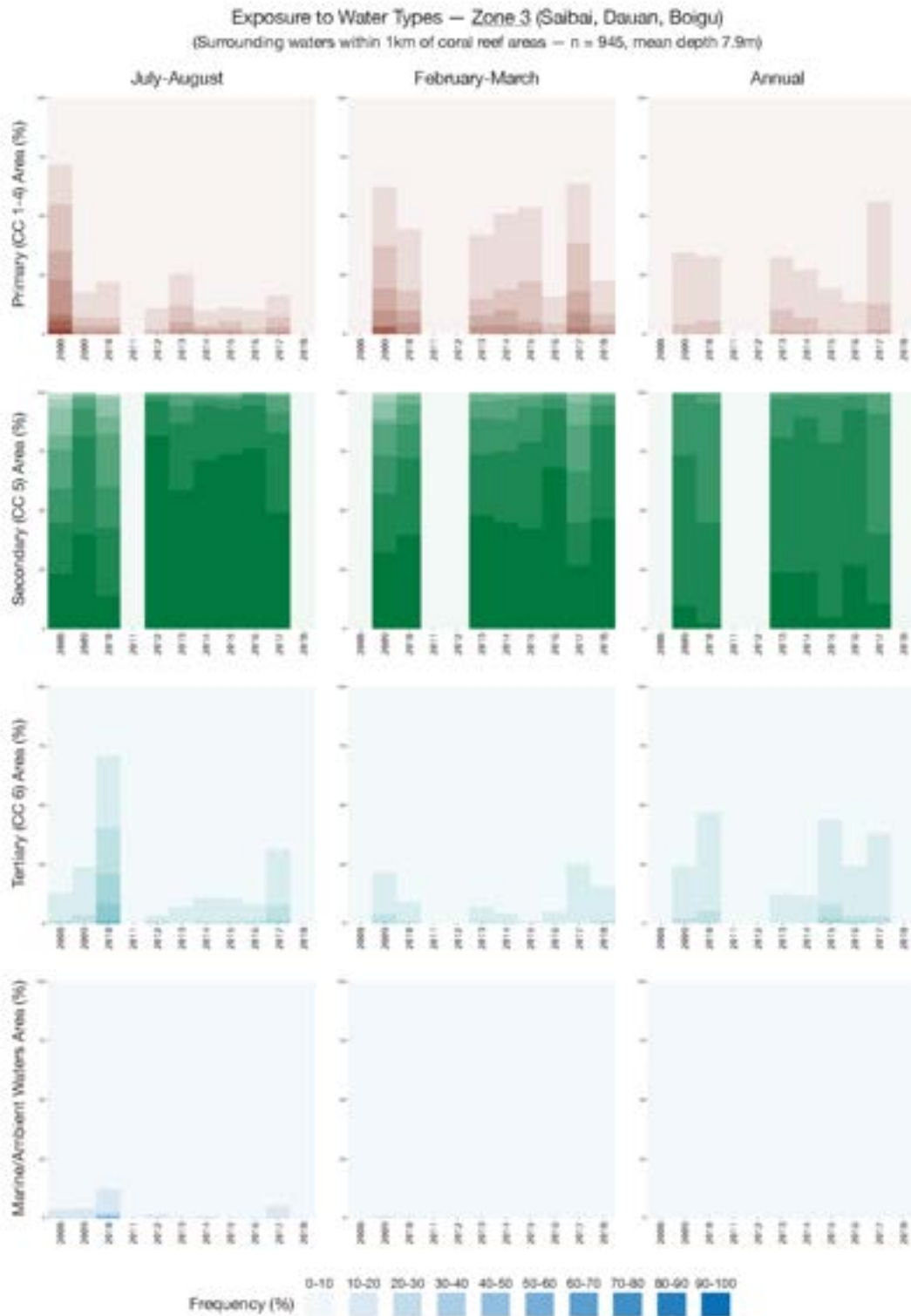


Figure A2-12: Frequency of exposure to different CCs in Zone 3 (North West) from 2008 to 2018 during July-August, February-March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).

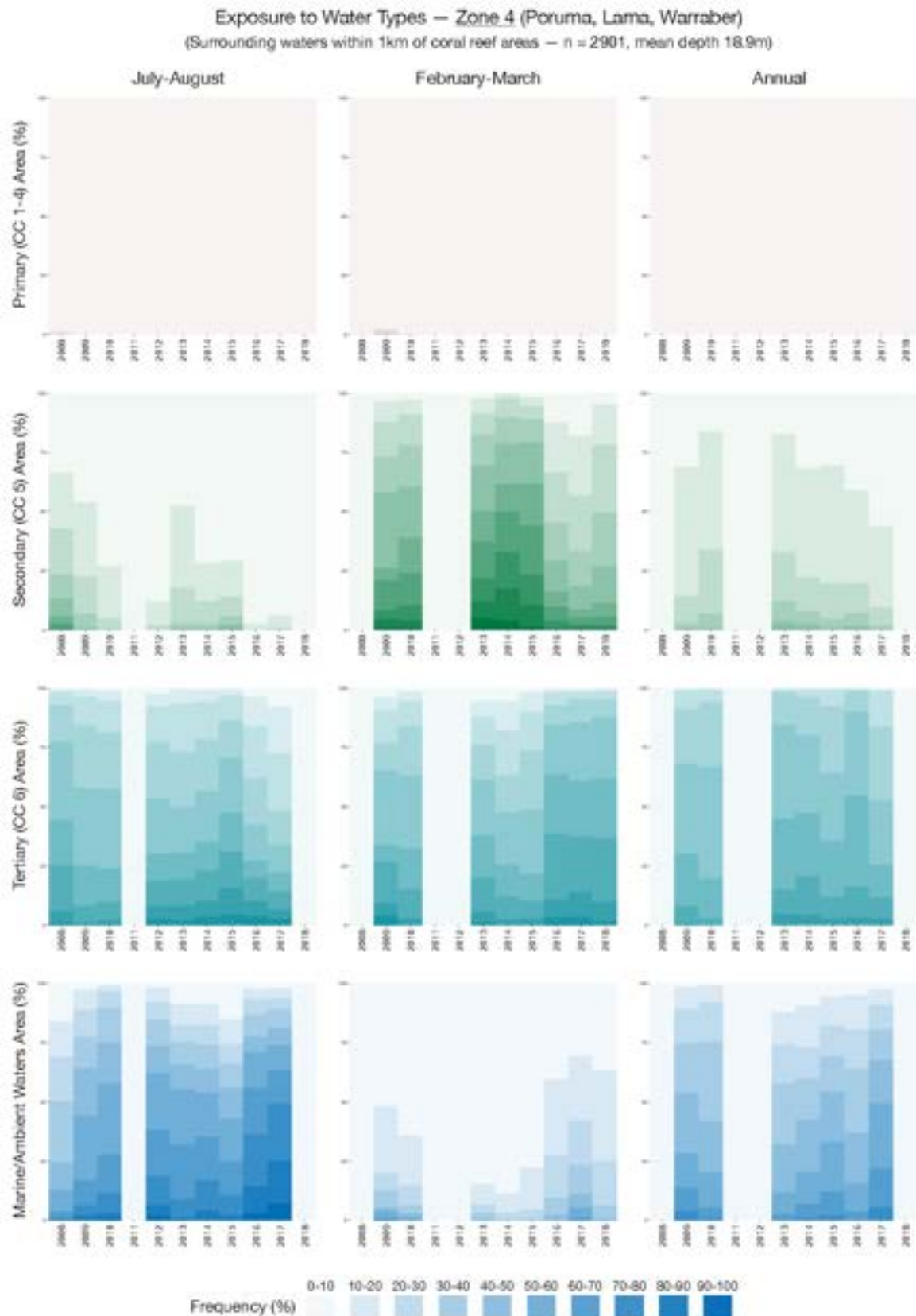


Figure A2-13: Frequency of exposure to different CCs in Zone 4 (South) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).

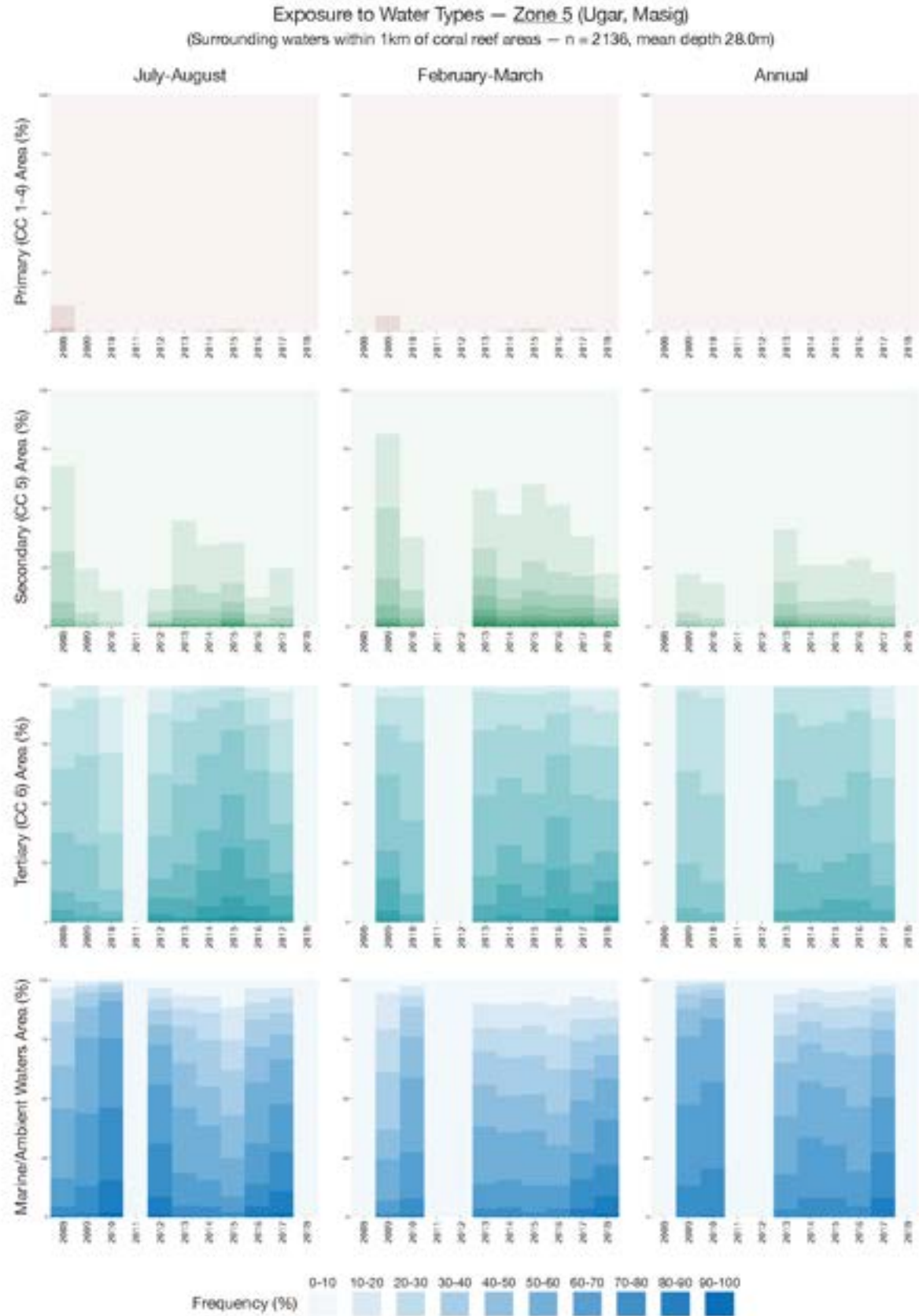


Figure A2-14: Frequency of exposure to different CCs in Zone 5 (Central) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).

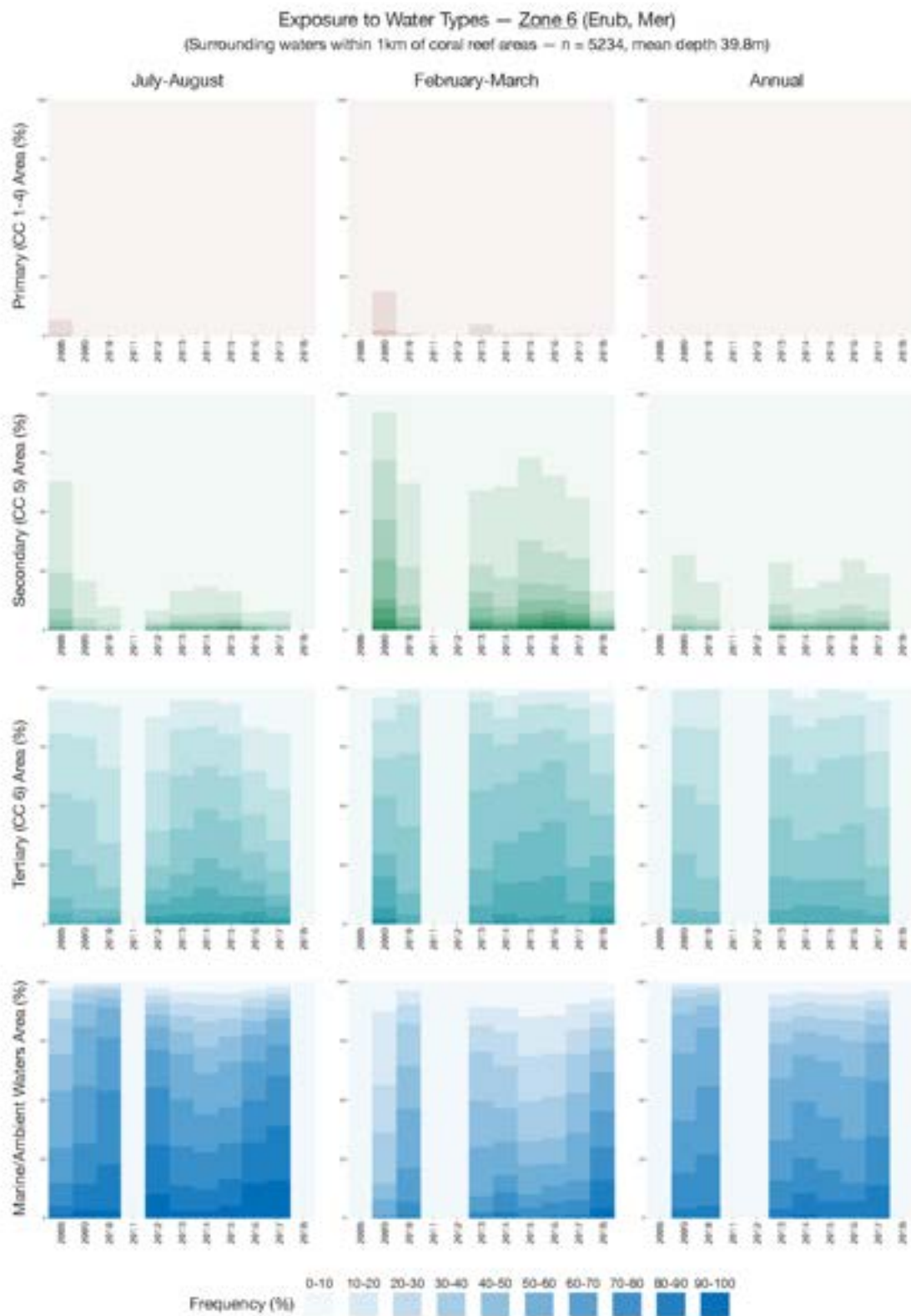


Figure A2-15: Frequency of exposure to different CCs in Zone 6 (East) from 2008 to 2018 during July–August, February–March and annual. Monitoring site codes in brackets after each site name (TSRA 2016).

10. APPENDIX 3: GENE SEQUENCING IN SEDIMENT SAMPLING AND SEAGRASS METAL SURVEYS

Statistical analyses of gene sequences

Prior to any analysis, bacterial and eukaryotic sequence datasets were variance stabilised using the *DESeq* package. This is necessary in sequencing data to be able to compare across samples despite differing library sizes (McMurdie and Holmes 2014), which are an artefact of modern sequencing techniques. Dissimilarities between sampling locations were calculated using the Bray-Curtis distance measure. Dissimilarities were subsequently sorted using the UPGMA and visualised using the *as.dendrogram* function in the *vegan* package (as seen in Long and Poiner 1994). Rarefaction curves of both 16S and 18S datasets were calculated using the *vegan* package. Alpha diversity (Shannon index) and diversity of each sample were calculated using the *diversity* and *specnumber* functions in R. The relationship between measured environmental variables and community compositions was tested using PERMANOVA using distance matrices with 10,000 random permutations using the *adonis* function (Oksanen et al. 2016). Canonical correspondence analysis (CCA) based on Bray-Curtis dissimilarities was used to visualise the differences in community composition between samples and the environmental parameters associated with these differences. Using the *adonis* function for PERMANOVA and CCA provides an alternative to both parametric MANOVA and to ordination methods such as distance-based redundancy analysis. The function partitions sums of squares of a multivariate dataset and because inputs are linear predictors, the function can describe how variation is attributed to different experimental treatments or uncontrolled covariates. Sediment metal concentrations measured by Apte et al. (2018) grain size fractions and C/N values were investigated for relationships to the microbial community. Distance between sampling locations was calculated based on the coordinates using the *distm* function in the *geosphere* package (Tables A3-1–A3-4).

Table A3-1: Actual and measured metal concentrations in certified reference materials (DOLT-4; fish liver) used to calculate recoveries and the limits of detection (LOD) for the analysis. Values shown as mg/kg dry weight or % (recoveries).

Metals	Ag	As	Cd	Cu	Fe	Hg	Ni	Pb	Se	Zn
Certified Reference Material	0.95	8.42	24.8	32.3	2073	2.64	0.787	0.158	8.8	116±6
DOLT-4	0.93±0.07	9.66±0.62	24.3±0.8	31.2±1.1	1833 ± 75	2.58±0.22	0.97±0.11	0.16±0.04	8.3±1.3	124
% Recovery	102	87	102	104	113	102	81	99.0	106	107
LOD	0.04	0.327	0.03	2	32	0.1	0.07	0.05	0.08	0.4

Table A3-2: Metal concentrations measured in each seagrass sample. All values are shown as mg/kg dry weight.

Metals	SG							E							G						
	rep1	rep2	rep3	rep4	rep5	rep6	rep7	rep1	rep2	rep3	rep4	rep5	rep6	rep7	rep1	rep2	rep3	rep4	rep5	rep6	rep7
Ag	0.16	0.1	0.06	0.1	0.05	0.08	0.02	0.43	0.41	0.34	0.24	0.35	0.33	0.54	0.22	0.06	0.06	0.14	0.06	0.2	0.09
As	1.6	5.6	8.3	13	5.3	6.9	2.7	4.08	1.59	0.89	3.56	4	3.66	3.58	4.46	1.84	4.58	1.56	2.4	3.5	2.17
Ba	1.9	29.6	19.5	100	22.8	8.1	6.8	3.85	3.01	1.95	1.81	1	16.3	11.5	17.4	2.1	9.89	7.15	2.11	4.19	5.39
Be	0.02	0.11	0.21	0.57	0.08	0.07	0.07	0.01	0.02	0.01	0.01	0.01	0.16	0.16	0.04	0.01	0.16	0.04	0.01	0.03	0.01
Bi	0.01	0.03	0.09	0.16	0.03	0.02	0.02	0.005	0.01	0.005	0.005	0.01	0.01	0.07	0.48	0.1	0.03	0.01	0.005	0.1	0.07
Cd	0.07	0.34	0.23	0.37	0.49	0.24	0.18	0.87	0.81	0.59	0.85	1.29	0.49	0.84	2.48	0.65	0.2	0.3	0.78	0.96	0.73
Ce	0.4	2.44	4.97	11.5	2.27	1.58	1.49	0.14	0.68	0.09	0.1	0.22	3.56	3	1.15	0.1	3.73	1.42	0.19	0.62	0.26
Co	4.4	8.8	8.3	9	11.5	9.9	10.4	0.1	0.3	0.1	0.1	0.1	2	1	0	0.1	1.17	0.43	0.15	0.33	0.2
Cr	1.1	4.9	7.4	24	3.6	2.8	2.9	1	2.1	1.7	0.5	1	12	11	3.2	0.5	11	4.2	0.7	2.4	1.1
Cs	0.07	0.51	0.69	2.2	0.28	0.23	0.23	0.01	0.08	0.01	0.01	0.01	0.57	0.55	0.01	0.13	0.47	0.17	0.01	0.09	0.01
Cu	3.6	18	11	15	12	11	7.8	7	4.7	3.2	6.6	8.2	5.6	8.6	19	9.4	3.1	3.4	11	6.5	7.3
Dy	0.04	0.22	0.47	0.96	0.22	0.15	0.12	0.018	0.038	0.015	0.019	0.029	0.304	0.289	0.14	0.01	0.39	0.14	0.02	0.07	0.03
Er	0.014	0.107	0.201	0.415	0.097	0.07	0.054	0.014	0.018	0.008	0.011	0.018	0.14	0.16	0.043	0.009	0.182	0.081	0.016	0.031	0.012
Eu	0.02	0.09	0.19	0.4	0.08	0.05	0.06	0.01	0.02	0.01	0.01	0.01	0.11	0.11	0.03	0.01	0.13	0.05	0.01	0.02	0.01
Ga	0.16	1.03	1.15	5.03	0.53	0.64	0.64	0.08	0.28	0.06	0.05	0.08	1.8	1.45	0.13	0.89	1.06	0.42	0.11	0.37	0.17
Gd	0.06	0.32	0.73	1.51	0.33	0.22	0.16	0.01	0.05	0.01	0.01	0.03	0.44	0.37	0.11	0.01	0.54	0.19	0.03	0.08	0.03
Hf	0.01	0.04	0.05	0.16	0.04	0.03	0.03	0.01	0.02	0.01	0.02	0.02	0.07	0.06	0.02	0.03	0.03	0.02	0.02	0.03	0.05
Hg	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05

	SG							E							G						
Metals	rep1	rep2	rep3	rep4	rep5	rep6	rep7	rep1	rep2	rep3	rep4	rep5	rep6	rep7	rep1	rep2	rep3	rep4	rep5	rep6	rep7
Ho	0.01	0.04	0.08	0.18	0.04	0.03	0.03	0.01	0.02	0.01	0.01	0.01	0.06	0.06	0.01	0.01	0.08	0.03	0.01	0.01	0.01
In	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.05	0.07	0.02	0.02	0.28	0.04	0.02	0.02	0.06	0.03	0.02
La	0.23	1.03	1.81	4.83	0.77	0.64	0.58	0.08	0.24	0.04	0.05	0.12	1.57	1.35	0.32	0.04	1.6	0.61	0.1	0.3	0.13
Lu	0.01	0.02	0.02	0.04	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.02	0.05	0.02	0.03	0.01	0.01	0.01	0.01
Mn	194	223	296	338	351	167	217	26	47	24	40	45	79	63	97	28	58	28	28	52	40
Mo	0.57	1.55	3.07	1.19	1.78	1.71	1.29	2.14	1.76	0.77	2.53	1.82	1	1.38	1.9	0.94	0.58	0.52	1.6	2.72	1.06
Nb	0.02	0.05	0.11	0.19	0.06	0.02	0.06	0.02	0.04	0.02	0.02	0.02	0.27	0.11	0.02	0.02	0.06	0.05	0.02	0.02	0.02
Nd	0.2	1.34	2.64	5.75	1.09	0.81	0.66	0.07	0.25	0.05	0.05	0.1	1.62	1.47	0.5	0.04	1.89	0.73	0.07	0.29	0.12
Ni	2	9.8	9.3	17	8.1	6.3	4	1	1.8	1.2	1.4	1.8	5.3	5.2	11	4.5	4.7	2.5	3.8	5.8	4.1
Pb	0.53	2.45	4.58	9.81	2.35	1.75	1.56	0.41	1.2	0.3	0.3	0.44	2.33	1.72	2	1.18	2.15	1.87	1.05	0.98	0.76
Pr	0.05	0.31	0.62	1.39	0.24	0.19	0.16	0.02	0.07	0.01	0.01	0.03	0.44	0.38	0.13	0.02	0.46	0.17	0.03	0.08	0.03
Rb	1.7	7.7	7.4	24.3	5.8	6.1	5.9	3.2	2.2	1.1	3	2.3	8	7.5	9	2.4	5.4	3.2	2.5	4	2.2
Re	0.005	0.008	0.005	0.005	0.005	0.005	0.01	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.005	0.01	0.005	0.005	0.005	0.005	0.01
Sb	0.13	0.36	0.15	0.12	0.36	0.31	0.36	0.78	0.45	0.24	0.49	0.64	0.2	0.49	0.22	0.27	0.05	0.09	0.15	0.27	0.24
Se	0.1	0.42	0.33	0.5	0.21	0.29	0.12	0.3	0.22	0.08	0.32	0.79	0.39	0.48	0.04	0.93	0.24	0.15	0.04	0.15	0.36
Sm	0.05	0.3	0.68	1.48	0.29	0.24	0.17	0.02	0.06	0.01	0.03	0.03	0.39	0.39	0.11	0.01	0.45	0.17	0.02	0.04	0.02
Sn	0.03	0.22	0.15	0.53	0.08	0.08	0.07	0.03	0.06	0.12	0.03	0.03	0.14	0.17	0.18	0.22	0.17	0.53	0.68	0.07	0.11
Tb	0.008	0.047	0.098	0.215	0.037	0.029	0.028	0.006	0.009	0.003	0.005	0.011	0.052	0.047	0.003	0.007	0.066	0.026	0.014	0.014	0.009
Te	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.15	0.37	0.45	0.15	0.15	0.8	0.66	0.15	0.15	0.15
Tm	0.01	0.01	0.02	0.05	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.02	0.01	0.01	0.02	0.01	0.01	0.01	0.01
Th	0.05	0.36	0.74	1.81	0.29	0.21	0.17	0.02	0.07	0.01	0.01	0.03	0.52	0.43	0.01	0.01	0.53	0.17	0.01	0.1	0.03
Tl	0.02	0.02	0.03	0.12	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.05	0.04	0.02	0.02	0.04	0.02	0.02	0.04	0.02
V	3	13	15	47	8	8	6	2	4	2	3	6	20	47	10	2	10	3	1	9	3
U	0.23	0.87	0.98	0.86	0.89	1.04	0.75	0.29	0.41	0.27	0.28	0.36	1.14	1.11	1.43	0.58	2.16	1.64	0.86	2.3	0.7
Y	0.18	0.95	2.08	3.98	0.98	0.72	0.53	0.1	0.29	0.06	0.08	0.16	1.92	1.74	1.14	0.08	2.28	1	0.15	0.37	0.17
Yb	0.02	0.07	0.15	0.3	0.06	0.03	0.04	0.009	0.018	0.006	0.006	0.015	0.12	0.112	0.078	0.015	0.151	0.074	0.032	0.025	0.016
Zn	8.7	85	36	55	36	27	27	21	13	7.7	13	30	21	21	98	20	10	11	16	11	15

	SG							E							G						
Metals	rep1	rep2	rep3	rep4	rep5	rep6	rep7	rep1	rep2	rep3	rep4	rep5	rep6	rep7	rep1	rep2	rep3	rep4	rep5	rep6	rep7
Zr	0.2	1.5	1.5	5.1	1.5	0.9	0.9	0.2	0.9	0.4	0.2	0.5	2	1.9	1.8	0.5	0.9	0.5	0.4	0.8	1

Table A3-3: ANOVA comparing differences in seagrass metal concentrations among locations. Significant p-values are highlighted in bold.

Metal	Df	SumSq	MeanSq	Fvalue	Pr(>F)
Ag	2	0.36	0.18	34.74	<0.000
As	2	48.12	24.06	4.14	0.033
Ba	2	2004.00	1002.20	2.51	0.109
Be	2	0.06	0.03	2.04	0.159
Bi	2	0.03	0.02	1.63	0.224
Cd	2	1.53	0.77	3.49	0.0523
Ce	2	27.60	13.80	2.27	0.132
Co	2	334.60	167.29	86.68	<0.000
Cr	2	42.80	21.38	0.64	0.541
Cs	2	0.95	0.48	2.25	0.135
Cu	2	85.20	42.61	2.32	0.127
Dy	2	0.19	0.10	2.15	0.146
Er	2	0.03	0.02	1.84	0.187
Eu	2	0.04	0.02	2.55	0.106
Ga	2	3.13	1.57	1.34	0.287
Gd	2	0.54	0.27	2.50	0.111
Hf	2	0.00	0.00	1.09	0.358
Hg	2	0.00	0.00	1.00	0.387
Ho	2	0.01	0.00	1.81	0.192
In	2	0.01	0.00	1.35	0.285
La	2	4.18	2.09	1.92	0.175
Lu	2	0.00	0.00	0.71	0.504
Mn	2	202596.00	101298.00	48.24	<0.000
Mo	2	0.37	0.19	0.35	0.712
Nb	2	0.01	0.00	0.99	0.393
Nd	2	7.49	3.74	2.46	0.114
Ni	2	107.60	53.79	4.67	0.023
Pb	2	21.31	10.66	2.97	0.0768
Pr	2	0.39	0.19	2.13	0.148
Rb	2	91.10	45.54	2.06	0.156
Re	2	0.00	0.00	1.19	0.327
Sb	2	0.31	0.15	7.33	0.005
Se	2	0.04	0.02	0.35	0.711
Sm	2	0.52	0.26	2.63	0.099
Sn	2	0.14	0.07	2.37	0.122
Tb	2	0.01	0.01	2.53	0.108
Te	2	0.10	0.05	1.47	0.256
Tl	2	0.00	0.00	0.37	0.694
Tm	2	0.00	0.00	0.71	0.505
U	2	2.54	1.27	5.35	0.015
V	2	296.00	148.00	0.86	0.442
Y	2	2.11	1.06	1.05	0.372

Metal	Df	SumSq	MeanSq	Fvalue	Pr(>F)
Yb	2	0.01	0.01	1.12	0.348
Zn	2	1602.00	800.80	1.43	0.265
Zr	2	2.99	1.50	1.33	0.288

Table A3-4: Tukey's Honest Significance Difference test results showing differences in seagrass metal concentrations among locations. Significant p-values are highlighted in bold.

Metal	Locations	diff	lwr	uppr	p_{adj}
Ag	G-E	-0.26	-0.36	-0.16	<0.000
	SG-E	-0.3	-0.39	-0.2	<0.000
	SG-G	-0.04	-0.14	0.06	0.610
As	G-E	-0.12	-3.41	3.17	0.995
	SG-E	3.15	-0.14	6.44	0.062
	SG-G	3.27	-0.02	6.56	0.052
Cd	G-E	0.05	-0.59	0.69	0.977
	SG-E	-0.55	-1.18	0.09	0.102
	SG-G	-0.6	-1.24	0.04	0.069
Co	G-E	-0.19	-2.08	1.71	0.965
	SG-E	8.37	6.48	10.27	0.000
	SG-G	8.56	6.66	10.46	0.000
Mn	G-E	1	-61.51	63.51	0.999
	SG-E	208.86	146.35	271.37	<0.000
	SG-G	207.86	145.35	270.37	<0.000
Ni	G-E	2.67	1.95754	7.30	0.327
	SG-E	5.54	0.91	10.17	0.018
	SG-G	2.87	-1.76	7.50	0.278
Pb	G-E	0.47	-2.11	3.05	0.889
	SG-E	2.33	-0.25	4.92	0.081
	SG-G	1.86	-0.72	4.45	0.185
Sb	G-E	-0.29	-0.48	-0.09	0.005
	SG-E	-0.21	-0.41	-0.02	0.033
	SG-G	0.07	-0.13	0.27	0.635
Sm	G-E	-0.02	-0.44	0.41	0.995
	SG-E	0.33	-0.10	0.75	0.157
	SG-G	0.34	-0.09	0.77	0.133
U	G-E	0.83	0.17	1.49	0.013
	SG-E	0.25	-0.41	0.92	0.607
	SG-G	-0.58	-1.24	0.09	0.094

