Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads

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Abstract

Coastal embayments are important ecosystems with high biodiversity and social values often under threat from human activities. Biota living in the intertidal zone of coastal embayments are adapted to a dynamic environment dominated by the tidal flooding and ebbing of water with periods of tidal exposure. Seasonal changes in key biota, their influence on intertidal zone primary production, and the consequences for intertidal ecology, appear to be driven by a combined effect of anthropogenic nutrient delivery, climate factors and intertidal zone hydrodynamics.

The objective of this project was to apply a research approach in field surveys with the involvement of the community and a range of modelling techniques to increase understanding of how sub-tropical catchment hydrology, nutrient delivery together with physical forcings that control intertidal zone hydrodynamics and environment variables affects the growth behaviour of marine cyanobacteria, *Lyngbya Majuscula* (hereafter Lyngbya) in an hypertidal coastal embayment. Lyngbya is a cyanobacterium that is found worldwide in tropical and subtropical coastal habitats, typically in low levels. During some parts of its life cycle it is toxic and has the potential to affect nearly the entire food web. The main research question was why Lyngbya occurs in an hypertidal coastal environment which generally discourage algal bloom with strong vertical mixing and strong flushing capacities. Finding answers for this question is challenging because the causes of toxic Lyngbya blooms are complex and multifaceted. This is more challenging when logistical issues confronted in remote and ungauged environments on top of the scientific problems. This study provides insight into the catchment, climate, ecological and physical processes controlling light and nutrients affecting Lyngbya blooms.

A remote, ungauged catchment and intertidal zone, of Roebuck Bay, near Broome in sub-tropical Western Australia was the study site of this research. This Ramsar listed hypertidal coastal embayment is under pressure from Broome’s urbanisation with the discharge of nutrients from waste water effluent and fertilisers occurring through surface and groundwater pathways. This
increasing wetland nutrient loads, is threatening the sensitive ecosystem of Roebuck Bay. In recent years, blooms of Lyngbya have risen in frequency and severity in the intertidal zone. The overall aim of this research was to investigate how changes in catchment hydrology, nutrient delivery, hydrodynamics and environmental conditions combine to affect nutrient dynamics, and influence the risk of Lyngbya growth. In order to achieve this, a two-year stormwater quality and quantity sampling campaign was undertaken to address knowledge and information gaps in the historical catchment and embayment data. These data were then analysed using statistical analyses and numerical modelling tools.

The research was arranged into four parts. In the first part, Broome catchment was instrumented to quantify nutrient export to Roebuck Bay with a community participated sampling campaign. This led to a quantitative estimate of surface hydrology and nutrient dynamics from this region that displays distinct episodic wet season conditions. Nutrient delivery to the Roebuck Bay embayment was identified to be greatest from the original Broome town site sub-catchments compared to sub-catchments that have been progressively urbanised since 2000. Most sub-catchments showed a distinct seasonal first flush with an initial 30% of runoff volume containing 40-70% of the nutrient load, resulting in periodic shock loading of nutrients to the bay.

The second research component analysed spatiotemporal changes in impervious surface areas over time to understand how a progressive increase in urbanisation has changed nutrient wash-off. With the increase in impervious surfaces, annual stormwater runoff volume and nutrient concentrations were predicted as having increased by 57% between 1981 and 2012. Nutrient loads increased from 378 g/ha/year and 64 g/ha/year in 1981 to 588 g/ha/year and 100 g/ha/year in 2012 for total nitrogen and total phosphorous respectively.

The third research component focuses on understanding the significance of physical and environmental variables on the seasonal occurrence of Lyngbya blooms in the intertidal zone of Roebuck Bay. A three-dimensional coupled hydrodynamic and water quality model, TUFLOWFV-AED, was used to investigate the hydrodynamics and nutrient dynamics in intertidal zone. Findings
suggested that nutrient availability is a necessary precondition for blooms but does not guarantee blooms will occur. The timing of nutrient loading in the context of other temporally varying parameters is critical. These parameters include physical and environmental properties such as temperature, turbulence intensity, tidal asymmetry, tidal exposure, photosynthetic active radiation, and water retention time.

The final study offers a mechanistic description of the next generation of ecohdrological model required to predict Lyngbya biomass in an intertidal zone of a coastal ecosystem. The model structure is based on three key state variables of Lyngbya biomass; 1) benthic submerged, 2) floating and 3) benthic exposed. It links these three states with key environmental factors for growth: bioavailability of nutrients (including iron, phosphorus, nitrogen and dissolved organics); light, salinity and temperature regimes; biological and physical processes such as growth, mortality, respiration, sloughing, floating and beach wracking. Data requirements for improved setup and validation of the model for supporting ongoing management decisions are outlined in this study.

This thesis highlights the importance and challenges of the integrated approach applied in this study to a remote subtropical coastal environment to achieve an improved understanding of how interacting driving factors affect nutrient pathways and lead to Lyngbya blooms. It is envisioned that this approach can be used in other similar systems under threat from increased nutrient loads and subsequent algal blooms. A key finding on Lyngbya is that the tidally exposed niche that favours it in competition for light is consistent with the nutrient rich thin submarine groundwater layer in tidally dominated aquatic environments.
Statement of originality

This thesis is my own composition, all sources have been acknowledged and my contribution is clearly identified in the thesis. For any work in the thesis that has been co–published with other authors, I have the permission of all co–authors to include this work in my thesis. The thesis has been substantially completed during the course of enrolment in this degree at UWA and has not previously been accepted for a degree at this or another institution.

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Dedication

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Publications arising from this thesis

- Gunaratne GL (2014) What’s lurking in the Roebuck bay coastal catchment?, “Science on the Broome Coast” seminar series organised by the Roebuck Bay Working Group and Yawuru Land and Sea Unit, Broome, Australia. 03 September, 2014 (Invited talk) (Part of chapter 2 and part of chapter 3)
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<td>Table A1</td>
<td>The Manning's n values for Broome drains in each sub-catchment. These values were compiled from literature (French 1986, Streeter et al. 1987, FHWA 2011) about Manning's n. Values were carefully selected following the guide explained by of Arcement and Schneider (1989).</td>
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## List of abbreviations

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<th>Abbreviation</th>
<th>Description</th>
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<tbody>
<tr>
<td>AED</td>
<td>Aquatic Ecodynamics</td>
</tr>
<tr>
<td>ANZECC</td>
<td>Australian and New Zealand Environment and Conservation Council</td>
</tr>
<tr>
<td>BOM</td>
<td>Bureau of Metrology</td>
</tr>
<tr>
<td>CLL</td>
<td>Cleared Lands</td>
</tr>
<tr>
<td>DEM</td>
<td>Digital Elevation Model</td>
</tr>
<tr>
<td>DO</td>
<td>Dissolved Oxygen</td>
</tr>
<tr>
<td>DOC</td>
<td>Dissolved Organic Carbon</td>
</tr>
<tr>
<td>DPaW</td>
<td>Department of Parks and Wildlife</td>
</tr>
<tr>
<td>EMC</td>
<td>Event Mean Concentration</td>
</tr>
<tr>
<td>FABM</td>
<td>Framework for Aquatic Biogeochemical Modelling</td>
</tr>
<tr>
<td>FRP</td>
<td>Filterable Reactive Phosphorous</td>
</tr>
<tr>
<td>GOTM</td>
<td>General Ocean Turbulence Model</td>
</tr>
<tr>
<td>HDR</td>
<td>High Density Residential</td>
</tr>
<tr>
<td>IND</td>
<td>Industrial</td>
</tr>
<tr>
<td>IPCC</td>
<td>The International Panel on Climate Change</td>
</tr>
<tr>
<td>LBE</td>
<td>Benthic Lyngbya biomass on mudflats at the absence of tidal waters</td>
</tr>
<tr>
<td>LBS</td>
<td>Benthic Lyngbya biomass submerged in waters</td>
</tr>
<tr>
<td>LDR</td>
<td>Low Density Residential</td>
</tr>
<tr>
<td>LF</td>
<td>Floating Lyngbya biomass in waters</td>
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<tr>
<td>Lyngbya</td>
<td>Lyngbya Majuscula</td>
</tr>
<tr>
<td>MDR</td>
<td>Medium Density Residential</td>
</tr>
<tr>
<td>N-\text{NH}_4^+</td>
<td>Ammonia-Nitrogen</td>
</tr>
<tr>
<td>N_E</td>
<td>Damköhler Number</td>
</tr>
<tr>
<td>N-\text{NO}_x</td>
<td>Nitrate/Nitrite - Nitrogen</td>
</tr>
<tr>
<td>PAR</td>
<td>Photosynthetic Active Radiation</td>
</tr>
<tr>
<td>PCA</td>
<td>Principal Component Analysis</td>
</tr>
<tr>
<td>POC</td>
<td>Particulate Organic Carbon</td>
</tr>
<tr>
<td>P-\text{PO}_4^{3-}</td>
<td>Orthophosphate</td>
</tr>
<tr>
<td>RBWG</td>
<td>Roebuck Bay Working Group</td>
</tr>
<tr>
<td>SMFF</td>
<td>Seasonal Mass First Flush</td>
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<tr>
<td>SWNL</td>
<td>Seasonal Wash-Off Nutrient Loading</td>
</tr>
<tr>
<td>TIA</td>
<td>Total Impervious Area</td>
</tr>
<tr>
<td>TKE</td>
<td>Turbulent Kinetic Energy</td>
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<tr>
<td>Acronym</td>
<td>Description</td>
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<tr>
<td>TN</td>
<td>Total Nitrogen</td>
</tr>
<tr>
<td>TP</td>
<td>Total Phosphorous</td>
</tr>
<tr>
<td>TUFLOW FV</td>
<td>Two dimensional unsteady flow finite volume</td>
</tr>
<tr>
<td>UDA</td>
<td>Undeveloped Vegetated Area</td>
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<tr>
<td>UWA</td>
<td>The University of Western Australia</td>
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<tr>
<td>WSUD</td>
<td>Water Sensitive Urban Design</td>
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<tr>
<td>WWTP</td>
<td>Waste Water Treatment Plant</td>
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Chapter 1  Introduction

1.1  Background and Motivation

1.1.1  Coastal wetlands

Wetlands are biologically diverse transitional areas between terrestrial and aquatic environments, characterised by shallow water overlying waterlogged soil and interspersed with submerged or emergent vegetation. In coastal areas, wetland habitats may include salt marshes, mangroves, intertidal mudflats, seagrass beds and shallow sub-tidal habitats effectively as a transition between terrestrial and marine ecosystems. Coastal wetlands provide foraging and nursery habitats for many fish, crustaceans and molluscs, including species with commercial and recreational value (Mitsch and Gosselink 2000, Delgado and Stedman 2011). They are also important feeding, breeding and roosting areas for water birds, including migratory species, and refuge areas for inland species in times of drought (GeoScience Australia 2011).

Wetlands also act as a buffer between land and sea as they prevent erosion, reduce currents, retain nutrients, attenuate waves and encourage sediment deposition and accretion (Nichols 1983, Bird 1984). They are dynamic and rich ecosystems globally recognized not only as areas of high ecological value but also of economic and social values (Curran and MacKenzie 2007). In Australia, ninety-five percent of the population lives around coastal wetlands, whose natural beauty and rich biological diversity provide the tourism and fisheries industries with billions of dollars each year.

1.1.2  Human-induced pressures

The degradation of wetlands causes a loss of critical habitat, biodiversity and ecosystem services. Wetland degradation also often increases runoff and associated sediment and nutrient loads to coastal waterways and embayments (Ward et al. 1998). Wetland habitats have been significantly altered through natural and anthropogenic forces and this alteration is becoming a major
environmental management issue in most parts of the world (Moser et al. 1996, Berzas et al. 2000, Gunaratne et al. 2011). Hydrological alterations are well documented to be one of the major anthropogenic activities that leads to degradation in the ecological character and functioning of wetland systems degrading their ability to deliver ecosystem services (USEPA 2001).

1.1.3 Hydrological regimes

The hydrology of a wetland defines its extent and determines species composition (Mitsch and Gosselink 1993). Hydrological conditions also influence the soils and biogeochemical cycling of nutrients, which in turn influence the species richness and abundance of biota (Kadlec and Knight 1996). It is well established that knowledge of wetland hydrology is essential to understanding, quantifying, and evaluating wetland functions and processes (Good et al. 1978, Carter 1986, Hipsey et al. 2010). Despite the importance of having a quantitative understanding of the hydrology of natural wetlands, the hydrological processes often remain poorly understood. Gaining this data is complicated by difficulties in measuring water inflows and outflows, changes in storage and the relatively large variability and error in field measurements (Winter 1981, Hunt et al. 1996).

Changes in water abstraction, land-use and climate variability all have the capacity to affect the regional hydrological cycle (DeWalle et al. 2000), and dependent ecosystems specially wetlands. Catchments respond to clearing of vegetation by producing more frequent and larger amounts of runoff, with peak flow rates depending on the soil moisture condition (Amatya et al. 1997). Both the soil moisture and the characteristics of storm events are influenced by seasonal climate variability, which in turn affects the runoff generation pattern and suspended sediment loads (Singh 1997). Stormwater runoff is sensitive to urbanisation density (particularly hardpan area), stormwater system design, vegetation water use, rainfall event size, frequency, and water table position prior to storm events (Harder et al. 2007).
1.1.4 Nutrient loading and water quality

Water quantity problems in wetlands are also often accompanied by water quality problems. One of the most common water quality problems is nutrient enrichment and associated eutrophication. This has been emerging as a significant problem in estuaries and coastal wetland systems and is considered a major focus for environmental management agencies (Turner and Rabalais 1994). Physical processes in coastal ecosystems serve as the driving forces for water and nutrient exchanges with the adjacent coastal sea (Dyer 1973). Hydrological processes (such as tidal driven nutrient flushing) may be dramatically affected by coastal re-engineering. Agricultural and urban development in the adjoining watershed, typically delivers significant nutrient loading to coastal waters (Correll et al. 1992, Valiela et al. 1992).

Loading of the critical nutrients (generally nitrogen and phosphorus but in some cases iron) to many coastal waters has greatly increased over the last few years, and will further increase as populations grow (Lee et al. 2006). Nitrogen is considered to be the limiting nutrient in most coastal systems due to denitrification processes, and export of N from the land to the coast is strongly influenced by population density and anthropogenic nitrogen inputs (Harris 2001). Coastal eutrophication leads to a number of detrimental environmental impacts including harmful algal blooms, loss of coral reef and seagrass habitats, and hypoxia (National Research Council 2000, Rabalais 2002, Rabalais et al. 2002). Ecosystem responses to anthropogenic stresses have been researched during the past three decades (Odum 1985, Hopkinson and Vallino 1995, Rapport et al. 1998, Boesch 2006); however, well documented assessments of disturbance effects on coastal wetlands at the ecosystem level has been scarce despite the impacts being widespread (Gunaratne et al. 2010, Paerl and Otten 2013).
1.1.5 Wetland hydrodynamics and ecological modelling

The water and mass exchanges between the main body of embayments and their fringing wetlands exert significant influence on wetland ecology (Hsu et al. 1998). Therefore, the hydrodynamics and water quality in a coastal wetland system are the major factors to be considered in conservation and restoration of coastal wetlands. Coastal wetland hydrodynamics control a variety of coastal processes including tidal flushing, nutrient dispersion, current patterns, sedimentation, erosion, and water levels (Le Hir et al. 2000, Andutta et al. 2014).

Embayment morphology influences the time that nutrients are retained before being flushed out to adjacent water bodies. A thorough understanding of embayment circulation is required to accurately determine nutrient concentrations within the system. Their shallow depths both decrease their ability to dilute inputs and increase the secondary impacts of nutrients often recycled from the sediments (Howes et al. 2006). Therefore, water quality modelling of tidally influenced coastal wetlands must include a thorough evaluation of the hydrodynamics of the ecosystem.

Water quality models have been important management tools for several decades, providing answers to complex questions concerning ecological systems that cannot be addressed with field data alone (Yang et al. 2007). The complex hydrological environment in coastal wetlands can be represented by numerical models (Langevin et al. 2005). Numerical models provide a cost effective method for evaluating intertidal zone hydrodynamics because they require limited data collection and may be employed to assess numerically a range of management alternatives (Pattiaratchi et al. 2011). Computations related to coastal processes become relatively straightforward extensions of the hydrodynamic modeling once the hydrodynamics of a coastal ecosystem are clearly understood. For example, the spread of pollutants may be analysed from tidal current information developed by the numerical models or the effect of climate change (including sea level rise) could be evaluated (Hipsey et al. 2013, Hipsey 2014).
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The International Panel on Climate Change (IPCC) estimates that the global average sea level will rise between 0.18 m to 0.59 m in the next century (IPCC 2007). Rising sea level can impact coastal wetlands by changing the hydrology, in particular the hydrodynamics, including sediment dynamics of the coastal zone (Working Group on Sea Level Rise and Wetland Systems 1997). Surface elevation in the wetland changes measurably in response to processes occurring within the sediment, such as accretion, changes in the sediment/water balance, mechanical loading by storm tides, and the annual cycle of root growth and decay (Cahoon 1995). Effective prediction and management of these measures will require understanding the dynamics of coastal wetlands, including dependent ecosystems, well enough to be able to mitigate their response to human-induced pressures (Xu 2004, Brown et al. 2013).

The uncertainty over hydrological processes in different types of WA coastal catchments, in particular the complex nature of the relationship between wetlands and their catchments has not been well-studied (Marimuthu 2005). In addition, the effects of changes in terrestrial hydrology and nutrient fluxes have been poorly investigated at a site or catchment specific level. To better guide the sustainable management of coastal ecosystems, improved understanding of surface water hydrology, nutrient loading and hydrodynamics of the ecosystem is necessary. This understanding must include empirical evidence and the ability to numerically simulate them in the coastal landscape (Hipsey et al. 2010). This research will unravel the degradation pathways of a coastal wetland within the context of and human-induced activities.

1.2 Research Direction

The coastal watershed and wetland under investigation in this study is Roebuck Bay, near Broome in the Kimberley region of Western Australia (Fig. 1.1). Literature on the eco-hydrology and nutrient fluxes of coastal wetlands in the Broome area is lacking except for the baseline studies by Carew and Hickey
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(2000) on tidal inundation model, Vogwill (2003) on hydrogeology and Estrella (2013) effect of nutrient enrichment on invertebrates and shore birds. However, some studies on social/cultural values (Peggs 1903, Scherrer et al. 2008) and biodiversity assets (Tulp and Goeij 1994, Piersma et al. 2006, Rogers et al. 2006) have occurred over the last two decades.

**Figure 1.1** Roebuck Bay Ramsar site, Broome showing site boundary and key locations

The discharge of waste water effluent, fertilisers, pesticides and other compounds all threaten the sensitive ecosystems of Roebuck Bay (Vogwill 2003, Pearson et al. 2008). In recent years, Lyngbya blooms have risen in frequency and severity (Roebuck Bay Working Group 2011), leading to concerns for the effects on social, economic and conservation values. *L. majuscula* is a cyanobacterium that is found worldwide in coastal habitats (Osborne et al. 2001), typically in low levels. Elevated nutrient inflows have
been blamed for increased \textit{L. majuscula} density and blooms in other sites (refs).

Conservation of Roebuck Bay coastal habitats and subsequent ecosystem services is a management priority in Roebuck Bay (Roebuck Bay Working Group 2011, Maynard and Wilcox 1996). Understanding the dynamics of such nutrient flows requires knowledge of the effects of land use on nutrient discharge and the effects on the receiving wetland environment including internal hydrodynamics (Correll \textit{et al.} 1992).

Presence and growth of the town of Broome on the shore of Roebuck Bay has altered the hydrology of this coastal wetland. One of the major emerging issues is the town acting as a major source of anthropogenic nutrients into the ecosystem. This is occurring due to the disposal of partially treated (nutrient rich) wastewater effluent which is flowing into the Bay. In order to understand the threat posed by nutrient discharge, a detailed understanding of the physical characteristics and hydrological processes controlling water flow and transport of constituents through the mangrove-fringed coastal creeks of the tidal embayment is required. The effects of changing hydrological conditions and fluxes of nutrients into Roebuck Bay are unknown however these changes are thought to be threatening the social and biodiversity values (Pearson \textit{et al.} 2008, Roebuck Bay Working Group 2011).

\subsection{Study site description}

Broome occurs on the western extent of the Kimberley coastline, which meanders for over 3000 km to the Northern Territory border, and the rugged terrain makes much of it mostly inaccessible by land. In recent years, the area has gained increasing publicity and popularity largely because of its spectacular scenery, Aboriginal rock art and wildlife. Tourism, which operates largely in state waters, has been largely unregulated, unplanned and unmanaged. At the same time, interest has grown from the minerals and petroleum industry in accessing potentially highly lucrative resources of the Kimberley on and off
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shore. Between 1995 and 2005, the Shire of Broome's population increased by 55%. The Kimberley population exhibited one of the fastest growth rates of any region in Western Australia between 1996 and 2006 and is expected to more than double by the year 2031 (Kimberley Development Commission 2009).

The climate of the Broome region is semi-arid, monsoonal with a distinct wet season with high temperatures (November to March) and a dry season with moderate temperatures (April to October). From October to April (the wet season) maximum temperatures average over 33 °C while overnight minima are typically warm 26°C. By contrast, the dry seasons (May to September) are mild, with July average maximum and minimum temperatures being 27 °C and 12 °C. Over 75% of the average annual rainfall falls from January to March which is associated with thunderstorms and tropical lows or cyclones.

Tropical lows or cyclones can produce heavy rain in short periods, and often a significant proportion of the yearly total can fall in just one or two days. The average annual rainfall is 596.3 mm with evaporation being almost 5 times that at approximately 2700 mm. Mean humidity (at 9am) varies from 45% to 71% in the months of August and January (Bureau of Meteorology (BOM) 2013). The wet season climate, including extreme climatic events, can have a profound and unpredictable effect on field equipment and the environment (Vogwill 2003, Bureau of Meteorology (BOM) 2013).

The Roebuck Bay coastal wetland is located in the shire of Broome (17° 58' S to 18° 16' S and 122° 08' E to 122° 27' E) in the coastal tropics of the Kimberley region of Australia about 2200 km north of the Western Australian capital Perth. Roebuck Bay is a large 55,000 ha embayment of the Indian Ocean extending from Fisherman's Bend, immediately east of the town of Broome, to the south of Sandy Point (Fig. 1.1) (Ramsar 1998, Pearson et al. 2008). The coastal wetland was declared as a Ramsar site in 1990 and is dominated by intertidal mudflats (Ramsar 1998).
The Roebuck Bay site was listed for several reasons including, most notably, outstanding shorebird values. The soft bottom intertidal mudflats of the northern and eastern shores of Roebuck Bay are critical areas as it is where most of the benthos (the primary wader food) occurs. The high tide roosts at Bush and Sandy Points are also significant (Ramsar 1998).

The northern shore of Roebuck Bay contains the mouths of two major creeks – Dampier Creek and Crab Creek (Fig. 1.1) (Pearson et al. 2008). A long red cliff, 2-6 m in height, of pindan soil, overlying yellowish-red Broome Sandstone of Cretaceous age dominates the northern shore of the Bay. The embayment has a macro-tidal system, varies from only about 1 m on neaps to 8 to10 m on spring tides (Pepping et al. 1997), which exposes around 160 km2 of mudflat with a high level of benthic invertebrate species richness and abundance (Pearson et al. 2008).

The Roebuck Bay coastal wetland is significant with many migratory water bird species using the site as a stopover on the Asia-Pacific flyway due to the abundant food source, i.e. benthic invertebrates. The area also supports threatened and nationally endangered marine mammals. It was designated a “Wetland of International Importance” under the Ramsar Convention in June 1990 (Ramsar 1998).

Given these high conservation values there is an urgent need to evaluate the impact of changing hydrological conditions on this internationally recognised coastal wetland. Fluxes of nutrients and water quality into the coastal wetland due to natural and man-made activities need to be quantified in the context of the current and future sustainability of the town of Broome. This work also has scientifically wider implications for: hydrological and hydrodynamic investigations; characterizing water cycle variability across multiple scales; for understanding its interactions in the land-ocean system; and for monitoring changes in catchment hydrology in the context of anthropogenic pressures.
1.3 Research Objectives and approach

The Roebuck bay coastal ecosystem provides an excellent natural laboratory to look at and compute the spatial and temporal changes in embayment nutrient dynamics with harmful cyanobacteria blooms in response to the simultaneous interactions of multiple stressors such as groundwater, stormwater and nutrient delivery and how these manifest in a whole-of-system response. This research seeks to do that through a gathering of catchment and hydrology field data through community based participatory research approach, and available embayment data for the region.

Unlike more intensely studied coastal embayments in the northern hemisphere, catchment inputs to southern subtropical and tropical embayments are highly episodic resulting in ‘shock loading’ or immediate loading of water bodies with nutrients bring dramatic changes in water quality over a short period of time (Tsihrintzis and Hamid 1997). As a result they are considered to be particularly sensitive to large-scale land-use changes, reductions in rainfall, increased extraction of water from groundwater and extreme discharge events since these changes impact on the delivery, residence and subsequent transformations of nutrients (Ahern et al. 2006, Pointon et al. 2008). Understanding the complex interactions between embayment morphometry, surface runoff flows, tidal regimes, nutrient load, and phytoplankton growth is essential for predicting embayment biogeochemical response to continuously evolving anthropogenic pressures.

Specifically the focus of the study is on providing a better understanding of the stormwater and nutrient pathways how different factors interact together and affect the proliferation of harmful Lyngbya blooms in the intertidal zone along the catchment-embayment-continuum. Ultimately it is envisioned that this is required to improve current management practices towards long-term sustainable development of the increasingly populated coastal areas. It encompasses three main research focus areas, outlined next.
The first approach consists of three main research objectives. They are, to 1) quantify the stormwater discharge and nutrient export from the urbanising Broome catchment to Roebuck Bay; 2) understand the relationship between nutrient sources, nutrient loads, runoff volume and the seasonal first flush phenomenon; 3) assess the results in terms of the effectiveness of varying degrees of Water Sensitive Urban Design (WSUD) that have been implemented in different sub-catchments.

The second research focus aimed to identify changes in the stormwater runoff and nutrient export rates of the Roebuck bay catchment, over a three-decade period, in response to changes in external factors such as landuse change incorporated urbansiation. Specifically, this study attempted to answer the following three questions. 1) What is the extent of spatio-temporal changes in total impervious area?; 2) What is the effect of land use change on urban seasonal runoff volumes and nutrient loading; 3) What type of recommendations can be made for ongoing sustainable management of the coastal wetland.

The aim of the third research area was to construct a process based biomass model for Lyngbya in the intertidal zone of a subtropical embayment. Roebuck bay intertidal zone was used as a model application in predicting Lyngbya biomass in the wet season.

1.4 Structure of the thesis

The next three chapters of this thesis (Chapters 2, 3, 4 and 5) are written as self-contained manuscripts intended for publication as journal articles. The final chapter (fifth chapter) condenses the key findings and recommendations for future work. The three main chapters of the thesis align with the research objectives and methodological approaches outline above.

Chapter 2 focuses on the nutrient export characteristics of Broome catchment. It is an urbanising, ungauged and poorly studied coastal catchment of north-western Australia. It suggests that managing the early seasonal runoff with its
higher nutrient concentrations reduces nutrient export by as much as 70%. This work highlights the emerging use of WSUDs over conventional urban stormwater management is a key in stormwater management. The general framework adopted in this study to monitor the nutrient and surface hydrology in a cost effective and a community-based participatory research is relevant for other catchments that may similarly be unable to develop a long-term monitoring program despite challenges associated with rapid urbanisation.

Chapter 3 explores the effects of changing land use on seasonal nutrient wash-off processes in an urbanising coastal catchment in a sub-tropical climate. This work present an analysis to synthesize spatial-temporal changes in impervious surfaces, and their relative impacts, on changing surface hydrology and nutrient wash-off load, suited to a sub-tropical environment with a paucity of information. This study further discusses the importance of lower impact practises for drainage design in future developments.

Chapter 4 describes the effect of intertidal zone hydrodynamics and abiotic factors on seasonal Lyngbya occurrence in hypertidal coastal embayments. The coupled hydrodynamic and water quality model used here provided enough evidence to suggest that intertidal hydrodynamics and abiotic factors play a key role in Lyngbya bloom likelihood, and nutrient availability is a necessary, but not sufficient, pre-condition.

Chapter 5 describes a coupled hydrodynamic-biogeochemical model for assessment of Lyngbya growth and biomass accumulation in a hypertidal coastal embayment. This article offers a mechanistic description of an ecological model to predict Lyngbya biomass suited to support assessment of management decisions of intertidal coastal ecosystems. A framework for setting Lyngbya management priorities in Roebuck Bay and as a basis for future research into coastal ecosystem impacts from Lyngbya blooms is described here.
Chapter 6 summarises and discusses all findings of Chapter 2-5. The chapter also provides concluding remarks on the management implications arising from the findings of this study, study limitations and recommendations for future work.
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Chapter 2. Effect of seasonal flushing on nutrient export characteristics of an urbanising, remote, ungauged coastal catchment.

2.1 Abstract

The threat of increased nutrient loads on fragile coastal ecosystems has been widely acknowledged. Urbanisation is a primary driver of nutrient export in coastal catchments, however estimating the rate of export from urbanising, remote and ungauged communities with episodic rainfall characteristics has rarely been reported in tropical and sub-tropical areas of the world. In this study, the urbanising, remote and ungauged coastal catchment of Broome, north-western Australia, was instrumented to elucidate the effect of seasonal flushing on nutrient export to Roebuck Bay. A novel cost-effective framework involving a temporary gauging strategy and community-based participatory research was employed. This led to a quantitative estimate of surface hydrology and nutrient dynamics from this region that displays distinct episodic wet season conditions. Results suggest that of all sub-catchments investigated, the contribution of nutrient export to the Roebuck Bay embayment was larger from the older Broome town site sub-catchments compared to sub-catchments that have been progressively developed since 2000. The length of the first antecedent dry period of the wet season had little influence on the runoff coefficient but had a substantial influence on seasonal event mean concentrations. Most sub-catchments showed a distinctive seasonal first flush phenomenon with an initial 30% of runoff volume containing 40-70% of the nutrient load, which indicates there are opportunities for stormwater management to minimise impacts through adopting ‘water sensitive urban design’ (WSUD) principles.
2.2 Introduction

Stormwater runoff from urbanising catchments is the principal transport mechanism carrying natural and anthropogenically-derived pollutants from non-point sources into receiving waters such as wetlands (Hall and Ellis 1985). Typically nutrient export from urbanising catchments increases, leading to deterioration of water quality, blooms of nuisance algae and eutrophication of coastal wetland systems (Bowen and Valiela 2001, Bricker et al. 2008, Estrella 2013). Blooms of the marine cyanobacterium Lyngbya majuscula are increasingly being reported in warmer waters in response to anthropogenically derived nutrients delivered to coastal wetlands (Ahern et al. 2007). These blooms have been reported within Roebuck Bay, Broome, in north-western Australia (Pearson et al. 2008), which is the receiving water body of the catchment under investigation in this study.

A significant concern is the effect that nutrient enrichment and L. majuscula blooms may be having in the Roebuck Bay ecosystem, particularly given a large portion of the wetland was designated as a ‘wetland of international importance’ under the Ramsar convention in 1990. Roebuck Bay supports large numbers of migratory shorebirds and comprises the primary staging and over-wintering areas for shorebirds on their annual southwards migrations (Pearson et al. 2008). Apart from its ecological value, the site has great cultural importance for indigenous people and provides a range of essential ecosystem services for them (Curran and MacKenzie 2007). For sound management of the Bay, a detailed understanding of the hydrological processes controlling water flow and nutrient export through the drains is required.

To date, a large body of research has been carried out to characterize surface runoff and nutrient export from catchments of differing urban characteristics in Australia (Deletic 1997, Goonetilleke et al. 2005), Asia (Ebise 1991, Lee and Bang 2000, Huang et al. 2007, Luo et al. 2009), Europe (Bertrand-krajewski et al. 1998, Gnecco et al. 2005) and North America (Lee et al. 2004, Schiff and
However, there is a paucity of information for tropical regions of the world, and no published accounts for north-western Australia, which has a unique geomorphological and hydro-climatological context. The Broome areas soils are typically silty, clayey sand red-soil (locally known as Pindan), with typically gently sloping topography and occasional rocky cliffs or dune ridges at the coast (Emery et al. 2009). The area also experiences a distinct monsoonal climate, more than 90% of the region’s rainfall occurs during the wet season, heavily influenced by tropical cyclone activity (BOM) 2013). These factors suggest that hydrological and nutrient export characteristics are also likely to be somewhat unique.

North-western Australia is generally remote, however, Broome is a hotspot of urbanisation supporting mining, tourism and oil and gas industries within the region. Despite the rapid rate of urbanisation, water quality issues have not been a priority for management organisations and consequently the catchment remains ungauged. An ungauged catchment is one with poor records (in terms of both data quantity and quality) of hydrological observations to enable computation of hydrological variables of interest at the appropriate spatial and temporal scales, to the accuracy acceptable for practical applications (Sivapalan et al. 2003).

The problem of undertaking hydrological assessments in ungauged catchments is not new. Accurate and reliable hydrological predictions are important to local and regional communities who are increasingly being asked to make independent judgments about actions required to manage natural disasters, the natural environment and water resources in a sustainable manner (Lee and Bang 2000, Sivapalan et al. 2003). Whilst there is a growing body of literature on approaches for estimating hydrological dynamics in poorly and ungauged catchments (e.g. Sivapalan et al. 2003, McIntyre et al. 2005, Blöschl 2006, Blöschl et al. 2013), the problem of prediction of nutrient export from ungauged catchments remains as an area of limited work to date. Several open research questions remain, for example, understanding how rates of nutrient export vary
seasonally and depend upon antecedent conditions, identifying how export is linked to the history of land-use development and the degree to which ‘water sensitive urban design’ (WSUD) principles have been considered. Donofrio et al. (2009) defined WSUD as an integrated water management system that encompasses low-impact design, water conservation and recycling, water quality management, and urban ecology. The seasonal first flush occurs, when a long dry period allows contaminants to build up and the initial rainfall events of the season may create runoff events with higher pollutant concentrations (and loads) relative to events later in the season (Lee et al. 2004). In the Broome catchment, the export is highly episodic and is expected to experience a reduction in the net rate of export over the season, however without historical monitoring data it remains unclear how various sub-catchments with varying amounts of water sensitive development are behaving.

The objectives of this study were therefore to: 1) quantify the stormwater discharge and nutrient export from the Broome sub catchments to Roebuck Bay; 2) understand the relationship between nutrient sources, nutrient loads, runoff volume and the seasonal first flush phenomenon; 3) assess the results in terms of the effectiveness of varying degrees of WSUD that have been implemented in different sub-catchments.

In order to quantify the surface hydrological and nutrient export characteristics, we approached the study by adopting a temporary gauging strategy. The relatively remote nature of the study site and the episodic rainfall-runoff events associated with Monsoonal conditions presented logistical challenges in terms of gauging the catchment. The objectives of the study were therefore achieved through engagement of local indigenous community rangers and volunteers from community groups in a field campaign that included intensive stormwater sampling and water level gauging. The analysis framework provided herein demonstrates how priority areas for targeting nutrient management efforts can be identified in ungauged catchments, with a low cost temporary monitoring program.
2.3 Material and methods

2.3.1 Catchment and climate data

A 1 second (~30 m) Digital Elevation Model (DEM) data from GeoScience, Australia and ArcHydro hydrology modelling tools (Maidment 2002) were used to delineate the 11 topographic sub-catchments of Broome on the north shore Roebuck Bay (Fig. 2.1). The sub-catchments were divided into two main categories: Old Broome town sub-catchments and Broome North. Demco (DMC), Town beach (TB), Seaview shopping centre (SVS), Hamersly & Guy st (HGS), Matsos (MTS), Magabala books (MBB) and Airport (APT) are the Old Broome town sub-catchments, whilst Gubinge road (GB), Roebuck estate (RBE), V-notch weir (VN) and New town exit (NTE) are the Broome North sub-catchments (Fig 2.1).

All Broome sub-catchments have a mixture of residential, public open space and commercial land uses. Part of DMC, RBE and GB are relatively new residential areas progressively being built since 2000. VN and NTE sub-catchments are still under development for commercial and residential land uses. There are a number of wastewater disposal sites across Old Broome town sub-catchments, where public open space is irrigated with secondary treated wastewater. The four sub-catchments in Broome North have adapted WSUD principles within their stormwater drain network (SKM 2009). Landuse features for the catchment were mapped using existing aerial photos and ArcGIS 10 (ESRI) 2011), refined with ground survey. Impervious percentage of the catchment was measured directly from aerial photos by digitizing the boundaries of impervious surface areas (Brabec 2002).

Rainfall data with 1-min temporal resolution were obtained from the Bureau of Meteorology weather station 003003 located at Broome Airport (BOM 2013). These were used to calculate event rainfall depths, the antecedent dry period and rainfall intensity. Antecedent dry period was estimated as the number of dry days following a measurable rainfall event.
2.3.2 Water quantity measurement

The stormwater depths in drains for each sub-catchment were recorded at 15-min intervals for all rain events from December 2011 to March 2012. We used 11 Capacitance probe data loggers (Scott Parsons Electronics, Australia) along with three conductivity, temperature and depth (CTD) pressure transducers or
Divers (Schlumberger Water Services, Germany) to gauge runoff water levels at the 11 temporary gauging stations. These 11 sites were the main drain channels which discharge Broome catchment runoff directly to the embayment. All drain channels were surveyed for their cross sections and slope geometry using an AX-2S Auto level, made by Nikon, USA. With the availability of runoff depths and channel geometry information, average flow rates of drains at each gauging site were estimated using Manning's equation for open channel flows (refer to Appendix A for specific details on runoff calculation). Estimated flow rates from Manning’s equation were validated against measured flow rates (‘stick and clock’ travel time estimates).

By integrating the runoff hydrograph, runoff volumes for each event were calculated. Sequences of rapid rainfall events, resulting in continuous flow events with multiple peaks, were considered as a single rainfall event for the purposes of this investigation. The runoff coefficients, $\varphi$, for individual rainfall events were calculated using:

$$\varphi = \frac{Q_s \times 10^2}{R_s \times A}$$

(2.1)

where, $Q_s$ is the surface runoff volume caused by single rain event which is measured by mega litres (ML), $R_s$ (mm) rainfall depth, $A$ (ha) is the area of sub-catchment. The seasonal runoff coefficient was calculated similarly using the total runoff($Q_t$) divided by the total rainfall ($R_t$) for the entire wet season and are replaced respectively in equation 2.1. Further, event median runoff coefficients were calculated for each rainfall event using event runoff coefficients in individual sub-catchments.

### 2.3.3 Water quality measurements

Water samples were taken at the 11 temporary gauging stations over 15 rainfall events during the study period. During each event, sampling teams of...
volunteers were coordinated to undertake manual grab sampling at 15-min intervals during the first 60-min, and thereafter at 30-min intervals for receding flow stage. Samples were split into filtered (0.45 µm) and unfiltered samples, then frozen. Frozen samples were sent to the University of Western Australia analysis laboratory, where total nitrogen (TN), total phosphorous (TP), nitrate/nitrite - nitrogen (N-NO\textsubscript{x}), ammonia-nitrogen (N- NH\textsubscript{4}+) and orthophosphate (P-PO\textsubscript{4}^{3-}) contents were measured on a Lachat flow-injection analyser (Lachat Instruments, Milwaukee, USA). The analytical methods were as follows: unfiltered samples were digested by using the persulphate method for TN and TP; P-PO\textsubscript{4}^{3-} and TP were determined by the molybdate-ascorbic acid method; N-NO\textsubscript{x} and TN were determined using the cadmium reduction method and N- NH\textsubscript{4}+ were determined using the phenolate method (APHA 1998).

### 2.3.4 Data analysis approach

#### 2.3.1.1 Event Mean Concentration (EMC)

Nutrient concentrations fluctuate considerably during a rain event and a single index known as event mean concentration (EMC) is often used to characterize runoff water quality. EMC represents a flow weighted composite sample, effectively the total pollutant load divided by the total runoff volume:

\[
EMC = \frac{\int_{0}^{t} C(t) \times Q(t) \, dt}{\int_{0}^{t} Q(t) \, dt}
\]  

(2.2)

where \(C(t)\) is the time variable concentration (mg L\textsuperscript{-1}) and \(Q(t)\) is the runoff flow rate (m\textsuperscript{3} s\textsuperscript{-1}) discharged at time \(t\) (Huber 1993, Lee and Bang 2000). On average we had three to five samples per event per sub-catchment and were used to numerically integrate Eq. 2.2.
Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads

2.3.1.2 **Seasonal Mass First Flush (SMFF)**

Seasonal Mass First Flush (SMFF) defines the amount of mass of nutrients and/or contaminants in early season runoff events. It was adapted from the previously defined Mass First Flush (MFF) ratio which describes the fractional mass of pollutants emitted as a function of storm duration (Lee *et al.* 2003, Han *et al.* 2006). The dimensionless SMFF ratio can be expressed as:

\[
SMFF_n = \frac{\int_0^T EMC \times Q(t) dt / M}{\int_0^T Q(t) dt / V}
\]  

(2.3)

where \( SMFF_n \) is the seasonal mass first flush ratio, \( n \) is the index of rainfall-runoff event within the wet season, which corresponds to the percentage of the cumulative runoff volume, ranging from 0 to 100%, \( M \) and \( V \) are total nutrient loading and total runoff volume for the wet season, respectively.

2.3.1.3 **Statistical analysis of catchment behaviour**

The relationships among hydrological (runoff coefficient), catchment (area, impervious percentage), and climate (antecedent dry days, rainfall intensity) catchment variables were modelled using linear regression. Variables were checked for normality using ‘Q-Q plot’ and ‘Shapiro-Wilk’ test in R software with the ‘Rcmdr’ package (R Core Team 2014) and non-normal data were log transformed to approximate a normal distribution, before the regression analysis.

We then undertook Principal Component Analysis (PCA) as a multivariate data analysis technique to reduce the large total number of variables into a smaller, conceptually coherent set, known as the principal components (Dunteman 1989). PCA has been previously used for linking hydrology and water quality within urban environments (Goonetilleke *et al.* 2005, Sodré *et al.* 2005, Shrestha and Kazama 2007). In order to recognise the patterns in nutrient,
catchment and hydrological data and their correlations with sub-catchments, XLSTAT software (XLSTAT 2013, Addinsoft, New York) was used to perform the PCA. To apply this method, data were arranged into a data matrix with the selected nutrient, catchment and hydrological variables (Table 2.1) defining the columns, and the rows referring to the sub-catchments. This 12×11 raw matrix, was then subjected to pre-treatment to remove ‘noise’ which may interfere in the analysis (Kokot et al. 1998).

**Table 2.1 Nutrient, catchment and hydrological variables used in the PCA**

<table>
<thead>
<tr>
<th>Variables</th>
<th>Acronym</th>
<th>Unit</th>
</tr>
</thead>
<tbody>
<tr>
<td>SMFF₃₀ for TN</td>
<td>TNS</td>
<td>dimensionless</td>
</tr>
<tr>
<td>SMFF₃₀ for TP</td>
<td>TPS</td>
<td>dimensionless</td>
</tr>
<tr>
<td>SMFF₃₀ for N-NH₄⁺</td>
<td>NH4S</td>
<td>dimensionless</td>
</tr>
<tr>
<td>SMFF₃₀ for N-NO₃⁻</td>
<td>NOxS</td>
<td>dimensionless</td>
</tr>
<tr>
<td>SMFF₃₀ for P-PO₄³⁻</td>
<td>PO4S</td>
<td>dimensionless</td>
</tr>
<tr>
<td>Slope</td>
<td>SLP</td>
<td>dimensionless</td>
</tr>
<tr>
<td>Area(ha)</td>
<td>A</td>
<td>ha</td>
</tr>
<tr>
<td>Impervious percentage</td>
<td>IMP</td>
<td>%</td>
</tr>
<tr>
<td>Seasonal runoff coefficient</td>
<td>ROC</td>
<td>dimensionless</td>
</tr>
<tr>
<td>Number of treated waste water disposal sites</td>
<td>TWWD</td>
<td>dimensionless</td>
</tr>
<tr>
<td>Export coefficient for TN</td>
<td>TNEC</td>
<td>kg/ha/year</td>
</tr>
<tr>
<td>Export coefficient for TP</td>
<td>TPEC</td>
<td>kg/ha/year</td>
</tr>
</tbody>
</table>

The data matrix was column standardized, thus, each variable had equal weighting with a standard deviation of 1, as the pre-treatment. Using the selected principal components (PC1 and PC2) that describe the largest amount of variance, a bi-plot was produced to display relationships between variables and sub-catchments.
2.4 Results

2.4.1 Characteristics of rainfall and runoff

The number of events that occurred was largely a function of seasonal rainfall distribution. The majority of rainfall occurs in December, January, February and March as is typical of sub-tropical monsoonal climates in the southern hemisphere (Fig. 2.2).

![Figure 2.2](image-url)

Figure 2.2 Mean monthly rainfall (column bars) and mean number of days of rain $\geq 10$ mm (line + scatter) in Broome for 1939-2012 period and 2011/12 wet season (BOM 2013).

Of the 15 rainfall events that were captured during the 2011/12 wet season, a range of antecedent dry periods, rainfall intensities and runoff from the 11 sub-catchments were recorded (Fig. 2.3). The first rainfall was on 2 January 2012, a delayed onset of the wet season relative to typical years (mid December). There was a seasonal (first outflow) lag time of more than 8 days in the GB and VN in Broome North sub-catchments. The reasons for the seasonal lag time are likely the greater storage in these sub-catchments which have more heavily
engineered storm water systems (e.g. weirs and culverts) and larger contributing areas. Rainfall events with similar depths have different runoff volumes largely due to different rainfall intensities and antecedent dry periods. For example, for two rainfall events of 9.4 mm and 10.6 mm in the APT sub-catchment, the runoff volumes were 15.5 ML and 5.8 ML respectively. The antecedent dry period for the first rain event of the season is 267.2 days but for the rest of wet season it varied from 0.4 to 19.3 days. Longer antecedent dry periods (> 19 days) reduced the amount runoff generation.

Figure 2.3 Rainfall, rainfall intensity and antecedent dry period (a) and calculated runoff volume from sub-catchments (b) for the study period 2011/12 wet season

Rainfall events on 08/01/2012, 27/01/2012 and 21/02/2012, with magnitudes of 32.2 mm, 60.8 mm and 22.8 mm respectively, have 1 to 2-year, 1-h Average Recurrence Interval (ARI). The rainfall event on 29/02/2012 showed a 2.5-year, 1-h ARI and the rest of the events in the wet season were in the category of 1-year or <1-year, 1-h ARI. The wettest month of the 2011/12 wet season was
March, largely due to the cyclonic derived rainfall events, which occurred from 15 March-17 March 2012.

2.4.2 Relationships between catchment and hydrological parameters

Variation in land use occurs within most sub-catchments, categorised as: low density residential (LDR), medium density residential (MDR), high density residential (HDR), industrial (IND), undeveloped vegetated area (UDA) and cleared lands (CLL). Each sub-catchment presented different hydrological and catchment characteristics (Table 2.2).

### Table 2.2 Characteristics of urbanising sub-catchments in the Broome town

<table>
<thead>
<tr>
<th>Sub-Catchment ID</th>
<th>Catchment area (ha)</th>
<th>Impervious %</th>
<th>Seasonal runoff coefficient</th>
<th>Slope</th>
<th>Land use code</th>
</tr>
</thead>
<tbody>
<tr>
<td>APT</td>
<td>184.26</td>
<td>40.13</td>
<td>0.63</td>
<td>0.008</td>
<td>MDR+IND</td>
</tr>
<tr>
<td>DMC</td>
<td>12.82</td>
<td>36.01</td>
<td>0.46</td>
<td>0.02</td>
<td>MDR</td>
</tr>
<tr>
<td>HGS</td>
<td>10.33</td>
<td>25.98</td>
<td>0.44</td>
<td>0.059</td>
<td>MDR</td>
</tr>
<tr>
<td>MBB</td>
<td>30.71</td>
<td>31.06</td>
<td>0.22</td>
<td>0.002</td>
<td>MDR</td>
</tr>
<tr>
<td>MTS</td>
<td>20.57</td>
<td>43.47</td>
<td>0.53</td>
<td>0.059</td>
<td>MDR</td>
</tr>
<tr>
<td>SVS</td>
<td>8.51</td>
<td>28.96</td>
<td>0.23</td>
<td>0.012</td>
<td>MDR</td>
</tr>
<tr>
<td>TB</td>
<td>53.87</td>
<td>32.82</td>
<td>0.55</td>
<td>0.01</td>
<td>MDR</td>
</tr>
<tr>
<td>GB</td>
<td>172.57</td>
<td>27.58</td>
<td>0.51</td>
<td>0.016</td>
<td>HDR+UDA</td>
</tr>
<tr>
<td>RBE</td>
<td>156.5</td>
<td>22.69</td>
<td>0.32</td>
<td>0.014</td>
<td>HDR+UDA</td>
</tr>
<tr>
<td>VN</td>
<td>67.89</td>
<td>20.68</td>
<td>0.11</td>
<td>0.017</td>
<td>LDR+CLL</td>
</tr>
<tr>
<td>NTE</td>
<td>29.39</td>
<td>13.23</td>
<td>0.42</td>
<td>0.007</td>
<td>CLL+UDA</td>
</tr>
</tbody>
</table>

Impervious percentage and total catchment area of each sub-catchment had a weak linear correlation with seasonal runoff coefficient ($R^2 \leq 0.23$, $p =$ not significant) (Fig. 2.4(a) & (b)). Event median runoff coefficients had a large variation (range = 0.1 to 0.8) for relatively smaller antecedent dry periods (< 10 days), with substantially lower values for greater antecedent dry periods (> 19
days) (Fig. 2.4(c)). Event median runoff coefficient and antecedent dry period had a weak linear relationship ($R^2 = 0.09$, $p = \text{not significant}$). For most events, relatively higher rainfall intensities were able to generate higher event median runoff coefficients and these two variables showed no strong linear relationship but was statistically significant ($R^2 = 0.34$, $p < 0.05$) (Fig. 2.4(d)).

![Figure 2.4](image)

**Figure 2.4** Relationships between catchment, hydrologic and climate characteristics.

### 2.4.3 Runoff water quality

The EMC varies by several orders of magnitude throughout the wet season and showed higher values during early runoff events over all sub-catchments. The temporal variation of EMC over the 2011/12 wet season shows the highest values occur in the first or second event of the season and then drop dramatically for the remainder of the season (Fig. 2.5(a)-(e)). The first two
rainfall events were small (0.8 mm and 11 mm, only 4% of the total wet season rainfall) but by the end of the second event there was sufficient runoff generation for sampling from all sub-catchments, except GB and VN. GB and VN had an 8-day lag before outflow commenced. During these first two outflow events the highest EMC of the wet season was recorded in all sub-catchments where flow occurred.

Figure 2.5 The temporal variation of EMCs of TN (a), N-NH$_4^+$ (b), N-NO$_x^-$ (c), TP (d), P-PO$_4^{3-}$ (e) for the 2011/12 wet season from the 11 sub-catchments.

EMCs of Old Broome town sub-catchments outflows are all larger than those of Broome North sub-catchments. DMC has the largest for TN while MTS has the largest for N-NO$_x^-$, TP and P-PO$_4^{3-}$, all measured during the very first outflow event on the 2 January, 2012. TB and other large (> 50 ha) sub-catchments in Broome North such as GB and VN, showed a lag time until peak occurred for most nutrients. In these sub-catchments the peak occurred after the fourth rainfall event (15% of the total wet season rain) as multiple flow events were
required to transport solutes to outflow given the small, low intensity rainfall events which occurred. A small peak in EMCs in most catchments can be observed in relation to the rainfall event on the 18 February 2012 emphasising the influence of antecedent dry period, 19.3 days in this case.

### 2.4.4 Seasonal first flush effect

Nutrient export showed a strong seasonal first flush effect in the first 30% of the cumulative runoff volume for most sub-catchments (Fig. 2.6(a)-(e)).

**Figure 2.6** Cumulative nutrient loading percentage curves for TN (a), N-$\text{NH}_4^+$ (b), N-$\text{NO}_3^-$ (c), TP (d) and P-$\text{PO}_4^{3-}$ (e) against cumulative runoff volume percentages in the 11 sub-catchments. A seasonal first flush occurs if the data ascended above the 45° dashed line, which represents uniform nutrient loading throughout the wet season.
DMC showed a delayed seasonal first flush TN and N-NO\textsubscript{x}. TP, P-PO\textsubscript{4}\textsuperscript{3-} and NH\textsubscript{4}\textsuperscript{+} exports were uniform in APT. A slight dilution effect in the first 30% of the cumulative runoff volume was visible in GB for TP and P-PO\textsubscript{4}\textsuperscript{3-}.

Analysis of nutrient discharge from 11 sub-catchments over 15 rainfall events of the 2011/12 wet season showed higher SMFF values for all nutrients during the first 30% of the seasonal runoff volume (Fig. 2.7(a)-(e)). The median SMFF for TN at 0-10% of runoff volume was approximately 2.7, which means treatment of the initial 0-10% of runoff volume could reduce the TN load 2.7 times as much as treating an equal volume later in the season. The ratio between cumulative rainfall categories (i.e. 0-10%) and annual average SMFF were the highest in the 0-10% of runoff volume for all parameters. There was a slight increase in the ratios at 50-60% of runoff volume for TN, N- NH\textsubscript{4}\textsuperscript{+} and N-NO\textsubscript{x}, potentially due to the nutrient build up and wash off after the 19.3 antecedent dry days. Regardless of the antecedent dry period TP and P-PO\textsubscript{4}\textsuperscript{3-} showed decreasing patterns from the beginning to the end of the wet season.
Figure 2.7 SMFF ratios for TN (a), N-NH4+ (b), N-NO3- (c), TP (d) and P-PO43- (e) against cumulative runoff volume for the study period for all sub-catchments. The dashed line at 1.0 represents the SMFF where the nutrient loadings are uniform throughout the wet season (i.e., no seasonal first flush). Values greater than 1 indicate the seasonal first flush during the early part of the wet season.
2.4.5 Unravelling sub-catchment behaviour

The PCA establishes if relationships occur between the various sub-catchment properties and the hydrological responses, including nutrient export characteristics. The bi-plot shows both the ‘loadings’ (variables) and the ‘scores’ (sub-catchments) for two principal components (PC1 and PC2) which are able to explain more than 72% of the data variance (Fig. 2.8). The degree of correlation between individual sub-catchments and variables is inversely related to the angle between lines. Variables closer to sub-catchments indicate a high correlation while orthogonal variables represent sub-catchments with no correlation.

The MTS sub-catchment produced the largest total nutrient export rate among all sub-catchments while being a small catchment area with a relatively high slope. HGS has strong linkages with SMFF$^{30}$ for TP and is another sub-catchment with a relatively small area and high slopes. The PCA results suggest strong relationships between SMFF$^{30}$ for TN, N-$\text{NH}_4^+$, N-$\text{NO}_x^-$, P-$\text{PO}_4^{3-}$ and NTE and SVS sub-catchments. These four sub-catchments have relatively small impervious area percentages and seasonal runoff coefficients. APT, DMC, TB and GB have larger catchment areas and higher numbers of wastewater disposal sites, with resulting runoff having some of the largest EMC’s. Seasonal runoff coefficients and impervious areas have strong correlations with APT, TB and DMC. Larger area catchments such as GB, APT, MBB, VN and RBE displayed weak relationship with seasonal first flush effects as evident from the weak SMFF$^{30}$. Broome North sub-catchments showed ‘no to weak’ correlation with TN and TP export coefficients and seasonal mass first flush for TP. These correlations demonstrated the relative effectiveness WSUD over traditional stormwater runoff designs.
Figure 2.8 Bi-plot for the first two principal components with nutrient, catchment and hydrological variables and their correlations with sub-catchments. See Table 2.1 and 2.2 for acronyms.

2.5 Discussion

2.5.1 Stormwater discharge and nutrient export

The quantification of stormwater discharge and nutrient export, suggests a significant nutrient loading is occurring to Roebuck Bay. The quantification of stormwater discharge was challenging, mainly due to logistic issues, in this relatively remote area with very rapid rainfall-runoff responses and no permanent gauging sites. Manning’s roughness values used were taken from the literature for open channels with different bed conditions. Uncertainty in measured flow estimates occurs due to the method used to measure water
velocity (‘stick and clock’ travel time estimates). There was also uncertainty in water quality sampling, i.e. the use of grab sampling over automated sampling.

Weather forecasting played a major role in determining whether the grab sampling captured the variability over an event. It was vital to get the most reliable forecast in order to avoid unproductive mobilisations for rainfall events that did not occur. The number of collected samples and sampling times across each sub-catchment were different in each rainfall event. Some sampling teams missed early parts of some rain events due to travel distances. This can affect EMC calculation because samples collected early in the runoff event typically have higher concentrations than those collected later in the storm event. Receiving water bodies respond relatively slowly to storm inflows compared to the rate at which constituent concentrations change during a storm event. Thus, EMC is an important analytical parameter for evaluating the effects of stormwater runoff on receiving waters (Lee et al. 2002).

Due to their relatively larger areas (> 50 ha) sub-catchments such as TB and GB showed delayed responses for EMCs. The TN wash-off process can be described as “source limiting” as the amount of TN in wash-off is limited by the load available (Miguntanna et al. 2013), and we observed both nitrogen and phosphorus compounds as “seasonally source limiting” since the wash-off EMCs values decreased towards the end of the wet season.

Sampling had to be called off one week prior to the scheduled finish of the program due to safety concerns associated with field work during cyclonic rainfall events. Consequentially the contribution of cyclonic events to nutrient loading was not measured in this study but the loading due to cyclonic rainfall events is likely negligible for two reasons. Firstly, the cyclonic events occurred towards the end of the wet season, after two and half months of periods of frequent rainfall so there was little possibility for nutrient built up to occur. Secondly, these events are likely to have very low nutrient concentrations due to the large magnitude of stormwater flow they generate. On average, Broome
experiences extreme rainfall events from a cyclone about one in every four years and they typically occur between late November to April (BOM 2012).

Despite the limitations and uncertainty, the dataset provided by this community-based sampling campaign and analysis has provided useful insights that can inform management. In particular, quantifying the nutrient export coefficients shows the sub-catchments with the largest nutrient inflows to the bay (Fig. 2.9). Export of TN ranges from 0.2-3.8 kg ha\(^{-1}\) year\(^{-1}\) for Old Broome town and 0.3-0.5 kg ha\(^{-1}\) year\(^{-1}\) for Broome North. The export coefficient values for TP range from 0.04-0.5 kg ha\(^{-1}\) year\(^{-1}\) and 0.04-0.1 kg ha\(^{-1}\) year\(^{-1}\) for Old Broome town and Broome North respectively. Greater than 80% of the total nutrient exports for both TN and TP were from the Old Broome town sub-catchments. MTS contributed approximately one third of the total nutrient export for both TN and TP.

![Figure 2.9 Nutrient export coefficients from Broome coastal sub-catchments.](image)
Gunaratne, G.L.

Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads

The increased nutrient flows from the Old Broome town is related to the presence of abundant potential sources of non-point source contamination and poor water sensitive urban drainage design structures. Residential lawns, parks and golf courses are fertilised heavily. There are also areas irrigated with treated wastewater, both of which have been identified as significant sources of nitrates and phosphorus in urban runoff (Pescod 1992).

Comparing the nutrient export coefficient values from Broome with those reported from urban areas in other parts of Australia and USA (Table 2.3) suggests that Broome’s annual nutrient export is relatively low. However, nutrient export from the Broome sub-catchments only occurs for three to four months during the wet season with 82% of the total seasonal TN exports and approximately 78% of the total seasonal TP exports occurring during the first month of the wet season as a shock loading. Therefore it is not appropriate to compare the nutrient export coefficients on annual basis as many of these other areas discharge nutrients for longer periods.

<table>
<thead>
<tr>
<th>Urban catchment</th>
<th>TP export coefficient (kg ha(^{-1}) year(^{-1}))</th>
<th>TN export coefficient (kg ha(^{-1}) year(^{-1}))</th>
<th>Catchment size (ha)</th>
<th>Climate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Queensland, Australia</td>
<td>1</td>
<td>6.6</td>
<td>10(^{4})–10(^{5})</td>
<td>Sub-tropical</td>
</tr>
<tr>
<td>South-West Australia</td>
<td>0.4</td>
<td>2.5</td>
<td>10(^{1})–10(^{2})</td>
<td>Mediterranean</td>
</tr>
<tr>
<td>Victoria, Australia</td>
<td>1.4</td>
<td>4.5</td>
<td>10(^{6})–10(^{7})</td>
<td>Temperate</td>
</tr>
<tr>
<td>North America</td>
<td>1.2</td>
<td>5.5</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Broome, North-western Australia</td>
<td>0.12</td>
<td>0.55</td>
<td>10(^{5})–10(^{6})</td>
<td>Sub-tropical</td>
</tr>
</tbody>
</table>

The impact of the shock-loading or immediate loading of water bodies with nutrients brings dramatic changes in water quality over a short period of time.
Findings of Tsihrintziz and Hamid (1997) suggested the changes in water quality may become more permanent if frequent shock loadings take place and nutrients accumulate in the system. As these nutrients circulate in the bay, they may be sequestered in sediments (or biomass) at various distances from the point of origin. Consequentially it is not only the short term impact of shock loading that is causing algal blooms but the longer term accumulation of nutrients in the wider ecosystem. The timing of the intense nutrient export/shock loading may be critical in generation of *L. Majuscula* blooms. If conditions for rapid *L. Majuscula* growth occurred during the flush of nutrients, there would be a period of approximately one month where marine water nutrient concentrations would be greatly elevated. The effect of this needs to assessed using marine circulation models as future research work.

### 2.5.2 Seasonal first flush phenomenon

Our findings expand on those of Lee *et al.* (2004) who documented the seasonal first flush in a Mediterranean climate in California, USA, where the seasonal first flush was also observed for organics, minerals and heavy metals. Seasonal first flush is a significant nutrient transfer mechanism in this subtropical urbanising coastal catchment. Yet prediction of nitrogen and phosphorous based nutrient export associated with seasonal first flush from ungauged catchments has never been attempted previously. Dilution effects varied significantly across the sub-catchments, due to the higher runoff volumes caused by larger catchments such as GB and APT. GB is 12 ha smaller than APT but it had the strongest dilution effect, likely due to the engineered nature of flow controls and drain channels, which generate relatively high runoff volumes. APT has mostly non engineered drain channels (basically earth based and walled gullies). The reason for delayed seasonal first flush in DMC is likely due to the readily removable nature of nitrogen compounds that make relatively low concentrations in the wash-off at high amount of runoff generation.
2.5.3 Implications for nutrient surface runoff management

The improved understanding of nutrient export from different sub-catchments suggests that treating the early runoff in the season is likely to be several times more effective than treating the later runoff. The many drains in Old Broome town have not been designed to retain anything; their focus is to discharge to Roebuck Bay even for 1-year, 1-h ARI rainfall events.

Management through stormwater retention, nutrient attenuation and disposal or artificial groundwater recharge of flows resulting from early rain events could drastically reduce nutrient loading (and shock loading) to Roebuck Bay. The reduced nutrient loading from Broome North sub-catchments (when compared to Old Broome sub-catchments) is likely due to the adaptation of WSUD principles in the more recently built stormwater system. However, Broome North subdivisions were not complete during the study period so urban based nutrients (garden fertilisers etc.) would not be in widespread use yet. There are also currently not any wastewater disposal sites in these areas. These results should therefore be confirmed once Broome North is complete and established.

Small sub-catchments respond more rapidly to rainfall events, with a shorter time of concentration. With the exception of APT and TB all the other sub-catchments are relatively small in Old Broome. Dietz and Clausen (2008) showed that the impact of increased stormwater runoff on local streams due to changes in a smaller watershed can be dramatic. Therefore, integrating WSUD principles to a small sub-catchment will potentially produce greater benefits than if similar changes were made in a large catchment. Our findings confirm those of Lee et al. (2004) who identify that stormwater management to treat early runoff is more beneficial. Broome's stormwater managers can optimise...
outcomes by implementing management, particularly in the Old Broome town sub-catchments, for the early runoff events. Further, prioritising management practices according nutrient export hotspots, related to specific sub-catchments, as shown by the PCA, can lead to a more targeted stormwater management approach. Thus, using basic catchment, climate, water quality and water quantity data, together with community-based participatory research, the present study demonstrates a novel cost-effective framework to tackle nutrient stormwater issues in remote, ungauged catchments (Fig. 2.10).

![Diagram showing the framework for managing stormwater nutrients from an ungauged, remote, subtropical catchment.](image-url)

**Figure 2.10** Framework for managing stormwater nutrients from an ungauged, remote, subtropical catchment.
The framework would be particularly beneficial in ungauged catchments in developing areas not resourced sufficiently to monitor stormwater using more traditional approaches. Grab sampling, temporary gauging, surveying and levelling of drains was achieved through engagement of local indigenous rangers and local community volunteers, which promotes stakeholders participation.

2.6 Conclusion

This study reports on the nutrient export characteristics of an urbanising catchment in the ungauged and poorly studied coastal catchment of north-western Australia. Catchment, climate, water quality and quantity data collected during the 2011/12 wet season were examined to identify nutrient export values and quantify the seasonal first flush effect. Results indicate that sub-catchments with relatively high slopes, high impervious percentages and small catchment areas have strong (high nutrient loading in outflows per unit area) nutrient export characteristics. Sub-catchments with relatively low slopes, low impervious percentages with large catchment areas have weak correlation with nutrient export characteristics. This is also related to the more modern WSUD urbanisation practices within the larger catchments.

The existence of seasonal first flush creates opportunities for stormwater managers and engineers to minimise impacts by integrating WSUD principles to the existing stormwater management system, particularly in the Old Broome town sub-catchments. Results suggest that managing the early season runoff (first 30%) with its higher nutrient concentrations, will reduce nutrient export by as much as 70%. The emerging use of WSUDs in the new Broome North sub-catchments appears effective when compared to conventional urban stormwater management. The lack of wastewater disposal sites would also contribute to the lower nutrient export characteristics in Broome North. This
however needs to be reassessed after the developments are complete and fully occupied.

Whilst we acknowledge that the data-set cannot provide answers to all the questions required to support the management of this catchment, the outcomes provide direction to managers in terms of setting priority areas for nutrient intervention and provide an estimate of the spatiotemporal variability of loads to the bay. The general framework adopted here to monitor the nutrient and surface hydrology in a cost effective manner is relevant for other catchments that may similarly be unable to develop a long-term monitoring program despite challenges associated with rapid urbanisation.

2.7 Acknowledgments

The authors would like to thank Rangelands NRM, The University of Western Australia Department of Parks and Wildlife (DPaW) for funding the project. We greatly appreciate the volunteer work from DPaW, Yawuru and Roebuck Bay Working Group (RBWG). We are indebted to Darren Stevens at DPaW for his immense support in organising and coordinating the whole monitoring program. The authors wish to thank the staff of UWA Analytical Laboratory for performing the nutrients analyses of the collected samples.
Chapter 3. Effects of changing land use on seasonal nutrient wash-off processes in an urbanising coastal catchment in a sub-tropical climate

3.1 Abstract

Roebuck Bay near Broome, Western Australia, is an area with high biodiversity, social and economic values that has been subject to rapid growth of urban areas, and the associated impact to runoff quantity and quality. We present an analysis to synthesize spatial-temporal changes in impervious surfaces, and their relative impacts, on changing surface hydrology and nutrient wash-off load, suited to this sub-tropical environment with a paucity of information. Multi-year (1981-2012) planimetric data were used to quantify urbanisation rate and the increase in impervious area. Parsimonious deterministic models were then used to calculate the temporal range in seasonal stormwater runoff and nutrient wash-off, identified as a key processes impacting Roebuck Bay. Results on landscape change suggested over the past three decades impervious areas increased from 37.8 ha (5%) to 255.8 ha (34%) in the Broome catchment. Stormwater model results revealed with the increase of impervious surfaces, the annual stormwater runoff volume increased by 57% between 1981 and 2012. Wash-off model estimated 378 g/ha/year and 64 g/ha/year in 1981 and 588 g/ha/year and 100 g/ha/year in 2012 for TN and TP, amounting to an increase of 56% and 57% in TN and TP, respectively. Recommendations for storm water management to reduce the threat posed by urbanisation to Roebuck Bay include introducing lower impact practices (reduced fertilizer, wastewater irrigation etc), riparian buffers around drains and best management practices for drainage design in future developments.
3.2 Introduction

Urban expansion is a key driver of degradation of natural wetland environments and aquatic ecosystems (United States Environment Protection Agency (USEPA) 2001). Changing a pre-existing land use to urban, can alter the natural hydrologic characteristics through increased impervious area and contaminant wash-off (Tang et al. 2005). Increased density of urbanisation typically further increases the amount of impervious area, with further increased volumes of runoff generation (Lee and Heaney 2003). Increased runoff occurs primarily due to a decreased capacity to infiltrate precipitation, with an increase in the volume of surface water runoff, shorter rainfall to storm water discharge lag times and decreased recharge of groundwater (Coletti 2013). Some urban areas try to dispose of storm water in local groundwater compensating basins where it can infiltrate into aquifers. This locally increases groundwater recharge and reduces storm water discharge, but can lead to prolonged flooding in areas exposed to episodic rainfall events typical of the tropics and sub-tropics. Water quantity problems in urban catchments are also often accompanied by water quality degradation. Impervious areas accumulate nutrients and other contaminants which are washed off and delivered to aquatic systems during runoff events (Wood 2010). Studies have documented enrichment of coastal wetlands by land-derived nutrients as one of the most pervasive pressures to aquatic ecosystems over past few decades (Bowen and Valiela, 2001; Estrella, 2013). In sub-tropical climates, episodic rainfall events from tropical storms dominate the hydrological cycle during the wet season. The few studies conducted on urban hydrology in sub-tropical climates have emphasized the unique characteristics of urban runoff with first flush effects (Lee et al. 2004, Gunaratne et al. 2015). Hydrological data from these urban studies typically span only a few years and have not allowed the impact of imperviousness area change on the volume of water, nutrients and the seasonal first flush effect through time to be assessed.
Roebuck Bay is a Ramsar listed coastal wetland in sub-tropical North Western Australia. It supports large numbers of migratory shorebirds and comprises the primary staging and over-wintering areas for shorebirds on their annual southwards migrations (Pearson et al., 2008). Apart from its ecological value, the site has great cultural importance for indigenous people and provides a range of ecosystem services (Curran and MacKenzie 2007). Recent research efforts have discovered nutrients entering through surface runoff and groundwater seepage (Estrella 2013, Wright 2013, Gunaratne et al. 2015). As a result of elevated nutrient levels in the bay, blooms of *Lyngbya Majuscula* (a blue-green algae) has been occurring in the intertidal zone of Roebuck bay since 2005. It is important to not only identify but quantify the relative contribution of nutrient sources to Roebuck Bay, and how this has changed in relation to urbanization, to allow targeted management to occur.

In this study, we have characterised the variations in impervious area in the urbanising catchment of Roebuck Bay using high resolution satellite imagery for 1981-2012. We have then used parsimonious deterministic models as a way to estimate the seasonal stormwater runoff volume and nutrient wash-off load associated with the land use change. The objectives of the study are to: 1) document the extent of spatio-temporal changes in total impervious area; 2) create a simple model framework using empirical relationships to quantify the effect of land use change on urban seasonal runoff volumes and nutrient loading; and 3) make recommendations for ongoing sustainable management of the coastal wetland.

### 3.3 Material and methods

#### 3.3.1 Study site description

Roebuck Bay is a large (360km$^2$) embayment of the Indian Ocean, located immediately south of Broome, in the coastal sub-tropics of the Kimberley region,
Western Australia (Fig. 3.1). The rapid urban expansion of Broome town has been accompanied by an increased population. Broome has also become increasingly popular over past two decades as a tourist destination due to its attractive coastal location, cultural values, natural environment, and quality of lifestyle.

![Figure 3.1 Broome coastal sub-catchments and Roebuck bay; Source: (Gunaratne et al., 2014)](image)

Broome’s resident population of 22,350 people increases during the peak tourism season (April to November) to over 45,000 people as of 2011 (ABS, 2013). This rapid population growth has also lead to increased volume of wastewater generation which is disposed of in treatment plants and public open space irrigation sites around town. It is the altered land use, water/fertiliser use and wastewater disposal practices that have put the Roebuck Bay ecosystem under pressure due to increased nutrient and other contaminant wash-off loadings. The sub-tropical climate of Broome, with long dry periods and intense
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rainfall events creates a strong seasonal contaminant first flush effect and subsequent shock loading to the ecosystem. Broome’s climate receives an average of 615mm of rainfall per year and approximately 90% of it gets during the wet season (December to March) while having January as the wettest month with an average of 181mm monthly rainfall (BOM, 2013).

### 3.3.2 Catchment Characteristics

There are 11 sub-catchments (Fig. 3.1) with a range of unique characteristics (Table 3.1).

<table>
<thead>
<tr>
<th>Sub-catchment category</th>
<th>Sub-catchment name</th>
<th>Sub-catchment ID</th>
<th>Sub-catchment area (ha)</th>
<th>Seasonal runoff coefficient</th>
<th>Land use code</th>
</tr>
</thead>
<tbody>
<tr>
<td>Old Broome town sub-catchments</td>
<td>Airport</td>
<td>APT</td>
<td>184.26</td>
<td>0.63</td>
<td>MDR+IND</td>
</tr>
<tr>
<td></td>
<td>Demco</td>
<td>DMC</td>
<td>12.82</td>
<td>0.46</td>
<td>MDR</td>
</tr>
<tr>
<td></td>
<td>Hamersly/Guy St</td>
<td>HGS</td>
<td>12.01</td>
<td>0.37</td>
<td>MDR</td>
</tr>
<tr>
<td></td>
<td>Magabala Books</td>
<td>MBB</td>
<td>30.71</td>
<td>0.22</td>
<td>MDR</td>
</tr>
<tr>
<td></td>
<td>Matsos</td>
<td>MTS</td>
<td>27.03</td>
<td>0.41</td>
<td>MDR</td>
</tr>
<tr>
<td></td>
<td>Seaview Shopping</td>
<td>SVS</td>
<td>8.51</td>
<td>0.23</td>
<td>MDR</td>
</tr>
<tr>
<td></td>
<td>Town Beach</td>
<td>TB</td>
<td>53.87</td>
<td>0.55</td>
<td>MDR</td>
</tr>
<tr>
<td>Broome North sub-catchments</td>
<td>Gubinge Road</td>
<td>GB</td>
<td>172.57</td>
<td>0.51</td>
<td>HDR+UDA</td>
</tr>
<tr>
<td></td>
<td>Roebuck Estate</td>
<td>RBE</td>
<td>156.50</td>
<td>0.32</td>
<td>HDR+UDA</td>
</tr>
<tr>
<td></td>
<td>V_Notch</td>
<td>VN</td>
<td>67.89</td>
<td>0.11</td>
<td>LDR+CLL</td>
</tr>
<tr>
<td></td>
<td>New Town Exit</td>
<td>NTE</td>
<td>29.39</td>
<td>0.42</td>
<td>CLL+UDA</td>
</tr>
</tbody>
</table>

Broome coastal catchment made up of Old Broome sub-catchments, which have no storm water retention or nutrient stripping structures in place, and Broome North sub-catchments which do. Old Broome sub-catchments all have a mixture of residential, public open space and commercial land uses. RBE and GB, are relatively new residential areas progressively being built since 2000. VN and NTE sub-catchments are still under development for commercial and
residential land uses. The land uses in sub-catchments were made of medium density residential (MDR), high density residential (HDR), low density residential (LDR), industrial (IND), undeveloped area (UDA) and cleared lands (CLL).

### 3.3.3 Impervious data

Land use features for the catchment were mapped using existing aerial photos and ArcGIS 10 (ESRI 2011). These were then refined with local on ground survey. The impervious percentage of the catchment was accurately estimated from aerial photos by digitizing the boundaries of impervious surface areas (Brabec 2002) (Fig. 3.2).

**Figure 3.2** An example of digitizing of impervious area change using geo-referenced aerial imagery for part of GB sub-catchment in 1981 (left) and 2012 (right)
Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads

Multi-year planimetric data includes 1981, 1993, 2005, and 2012 mosaics which were based on digital aerial photos provided by Western Australian Land Information Authority (Landgate). Total impervious area (TIA) includes roads, buildings and airports. Sidewalks, driveways were not included because they were too narrow to delineate at the spatial resolution of our images. The resulting three classes however are considered to dominate the coverage of impervious areas.

### 3.3.4 Impervious area change

TIA for each sub-catchment and year were calculated by dividing IA by the corresponding sub-catchment area. The change in percent TIA was quantified by estimating annual percent TIA change (Wu and Thompson 2013) (Eq. 3.1):

\[
I_{\text{area}} = \frac{TIA_{\text{end}} - TIA_{\text{begin}}}{n}
\]  

(3.1)

where \(I_{\text{area}}\) is the annual percent TIA change (%), \(TIA_{\text{begin}}\) is the per-cent TIA (%) at the beginning of a time period, \(TIA_{\text{end}}\) is the percent TIA (%) at the end of a time period, and \(n\) is the length of a time period (years).

We also calculated annual TIS change rate (Eq. 3.2)

\[
I\% = \left( \frac{\sqrt{A_{\text{end}}} - 1}{A_{\text{begin}}} \right) \times 100
\]  

(3.2)

where \(I\%\) is the annual TIA change rate (%), \(A_{\text{begin}}\) is the TIA (ha) at the beginning of a time period, \(A_{\text{end}}\) is the TIA (ha) at the end of a time period, and \(n\) is the length of a time period (years).
3.3.5 Stormwater runoff

We estimated runoff for 1981, 1993, 2005 and 2012 using the 2011/2012 seasonal rainfall data as an idealized reference condition. The same rainfall is used to remove the effect of the highly variable rainfall, to isolate the effect of land use. Runoff was calculated as a product of seasonal rainfall, runoff coefficient and drainage area which was adapted from Chandler (1996) and Schueler (1987):

\[ R = P \times P_j \times R_v \times A / 100 \]  

(3.3)

where \( R \) is seasonal runoff volume (ML), \( P \) is seasonal rainfall (mm), \( P_j \) is fraction of annual rainfall events that produce runoff (usually 0.9), \( A \) is sub-catchment area and \( R_v \) is runoff coefficient. The runoff coefficient was calculated based on impervious cover percentage in the sub-catchment. This linear relationship can be expressed as \( R_v = 0.1502 + (0.0075 \times TIA) \) and it was built using 2011/2012 wet season data from all Broome sub-catchments (Gunaratne et al., 2014). The model was then calibrated using 2011/2012 observed rainfall and runoff data obtained from Gunaratne et al. (2014). After calculating \( R_v \) using respective multi-year TIA values, we estimated seasonal runoff volumes for 1981, 1993, 2005 and 2012.

3.3.6 Seasonal wash-off nutrient loading (SWNL)

Seasonal wash-off nutrient loading (SWNL) defines how the quantity of nutrients available for wash-off decreases exponentially with seasonal runoff volumes during the season. It was adapted from the previously defined wash-off nutrient loading which is a function of the amount present on the watershed and runoff volume of a rainfall event (Alley, 1981). The total load from wash-off for the storm events of the season is expressed as:

\[ L = P_1(1 - e^{-kV_t}) \]  

(3.4)
where $L$ is the seasonal nutrient wash off load, $P_1$ is initial seasonal load (kg); $k$ is seasonal wash-off coefficient (mm$^{-1}$), and $V_t$ is total seasonal runoff to time $t$. The exponential model was calibrated using water quality (total nitrogen (TN), total phosphorous (TP),) and quantity data collected over 2011/2012 wet season with the approach used by Charbeneau and Barrett (1998). Estimated runoff volumes, seasonal wash-off coefficients and initial seasonal nutrient load values for each sub-catchment were used in calculating SWNL for 1981, 1993, 2005 and 2012.

### 3.4 Results and discussion

#### 3.4.1 Impervious area

Impervious are has increased from 30 ha to 120 ha in Old Broome Town while it has increased from 7 ha to 130 ha in Broome North (Fig 3.3).

![Figure 3.3 Total Impervious area change in Old Broome town and Broome North for year 1981, 1993, 2005 and 2012](image-url)
For Old Broome Town it has a gradual rise of impervious area over the last three decades while Broome North has been largely developed after 2005. In 1981 the Broome North sub-catchments were essentially undeveloped (TIA values less than 10%) but by 2012, they varied from < 20 % to as much as 42%. Average annual percent TIA change for all sub-catchments was 0.70% from 1981 to 2012 (Fig 3.4).

In 1981, for all except SVS, the Old Broome sub-catchments had TIA percentage in between 10 and 20% by 2012 the TIA had increased to 30-40%. For old Broome town sub-catchments it ranges from 0.29% to 1.06% while in the Broome North it ranges from 0.61% to 1.24%. Almost all sub-catchments in Old Broome town showed a rapid increase in average annual TIA percent

Figure 3.4 (a) Total impervious area (TIA) (ha); (b) TIA percentage; (c) Annual average TIA percent change (%); (d) Annual TIA change rate percentage (%) for different time periods from 1981 to 2012 for Broome sub-catchments
change for the period of 1981-1993 while other sub-catchments in Broome North showed rapid increment from 2005 to 2012 (Fig 3.4). Impervious area has increased from 38 ha to 256 ha from 1981 to 2012. It was an increment of TIA from 4 % to 27.5% over the past three decades (Fig 3.5).

![Figure 3.5](image)

**Figure 3.5** Total impervious area (TIA) shaded in red for Broome subcatchments in (a) 1981 and (b) 2012.

### 3.4.2 Stormwater runoff

Observed and modelled seasonal runoff volumes showed very strong correlation ($R^2=0.85$) for 2011/2012 data (Fig. 3.6). For relatively smaller catchments, the simple method estimated more accurate values than relatively larger catchments. Seasonal runoff volume changed over past three decades across all sub-catchments.
Fig. 3.6 Observed Vs modelled seasonal stormwater runoff volumes for 2011/2012 wet season

Fig. 3.7 shows the change of seasonal runoff volume over past three decades. Old Broome town catchments showed 1.6 times increase while Broome North sub-catchments showed 2.1 times increase from 1981 to 2012 in seasonal runoff volumes.

The technique we used to calculate stormwater runoff was most suitable for assessing and comparing the relative stormwater runoff changes of different land use and stormwater management scenarios from relatively smaller catchments (<1 sq. mi.) (Chandler 1996). Even though this technique was advantageous in making reasonable estimates considering time and funds still there are situations this should not be used due to its limitations as explained by Chandler (1996).
3.4.3 Seasonal wash-off nutrient loading (SWNL)

Observed and predicted nutrient loading data for 2011/2012 wet season showed strong correlation with $R^2$ values 0.98 and 0.91 for TN and TP respectively (Fig. 3.8). Modelled seasonal TP and TN loading increased over time, for year 1981, 1993, 2005 and 2012 (Fig. 3.9). Results showed the SWNL from relatively smaller catchments such as DMC, HGS, MBB, MTS, SVS and NTE for TN, had an increment of average 1.2 times in 2012 compared to 1981. The larger sub-catchments increased their wash-off TN loading by approximately 1.7 times over the three decades. For TP, relatively smaller catchments showed 1.2 times average increment and larger catchments showed 1.8 times over the past three decades.
Knowledge of nutrient wash-off and associated physical processes is critical for the development of appropriate strategies for managing the effect of stormwater runoff on receiving waters (Charbeneau and Barrett, 1998). The simple exponential wash-off model we used to calculate wash-off nutrient load from seasonal runoff, based on event mean concentrations (EMCs) and is appropriate for evaluating effects of stormwater runoff on receiving waters.

3.5 Conclusions

Using multi-year high-resolution spatial planimetric data, we were able to accurately calculate TIA changes in Broome sub-catchments over approximately three decades. TIA increased consistently (although the rate of change varied) over the study period. Buildings and roads were the key contributors to expansion of TIA.
Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads

The simple runoff modelling used herein made reasonable estimations for stormwater runoff volumes and that led to more accurate results in wash-off load modelling. Runoff volumes and nutrient wash-off loads increased from 1981 to 2012 due to the change in TIA. Old Broome town sub-catchments and Broome North sub-catchments reacted differently to urban growth and subsequent nutrient wash-off loads over the study period. The outcome of this study can be used as a framework to characterise the long term catchment imperviousness impact on degrading water quality in receiving waters in areas with distinct episodic event driven wet season conditions and limited historical monitoring data. Further these results could be used to enhance local land use policy and planning by focusing efforts on nutrient export hot-spot areas.

Figure 3.9 Modelled seasonal wash-off nutrient loading (TN and TP) in Broome sub-catchments for year 1981, 1993, 2005 and 2012
3.6 Acknowledgement

The authors would like to thank Rangelands NRM, The University of Western Australia Department of Parks and Wildlife (DPaW) for funding the project and DPaW, Yawuru and Roebuck Bay Working Group (RBWG) for all in-kind support.
Chapter 4. The effect of intertidal zone hydrodynamics and abiotic factors on seasonal Lyngbya occurrence in hypertidal coastal embayments

4.1 Abstract

The incidence of harmful algal blooms in sub-tropical and tropical coastal waters has increased in recent years mainly due to human activities. Coastal environments with high tidal energy normally discourage algal blooms due to strong vertical mixing and high flushing capacity. Over the past decade, outbreaks of the cyanobacterium *Lyngbya majuscula* (Hereafter Lyngbya) have been recorded in different severity in every wet season in the intertidal zone of Roebuck Bay, Australia which is a hypertidal (tidal range >9m) coastal embayment listed as a Ramsar wetland. The puzzling fact is that the intertidal zone of the embayment has strong tidal forcing but has become increasingly favourable for Lyngbya blooms. Although considerable effort has been devoted to study various aspects of Lyngbya bloom formation, the physical processes controlling hydrodynamics of the bay and intertidal zone are still not completely understood. This led us to investigate the intertidal hydrodynamics and abiotic factors such as temperature, salinity and photosynthetic active radiation (PAR) to identify likely drivers that that may explain Lyngbya bloom occurrence. A three-dimensional coupled hydrodynamic and water quality model, TUFLOW FV-AED, was applied to fill this gap and to investigate how tidal elevation, tidal velocity, turbulence intensity, photosynthetic active radiation (PAR), salinity, temperature, retention time and stormwater and groundwater inputs behave in the intertidal coastal embayment. Turbulence intensity along the water depth changed according to flood-ebb tide cycles and spring-neap tide cycles indicating periodical vertical mix in the water column. However, when intermittent weak stratification occurs during favourable growth periods, Lyngbya can quickly proliferate. Salinity and temperature showed a weak stratification in the system, while PAR is a key factor specifically during tidal...
exposure where Lyngbya in some parts grows on film of nutrient rich water layer generated as a submarine groundwater discharge. Estimates of the retention time and Damköhler number (growth time scale vs residence time scale) suggested the opportunity for bloom formation to occur despite the large tidal range. The model provided enough evidence to suggest that intertidal hydrodynamics and abiotic factors play a key role in Lyngbya bloom likelihood, and nutrient availability is a necessary, but not sufficient, pre-condition. This work is an initial step towards developing a Lyngbya biomass model in the intertidal zone of tropical embayments.

4.2 Introduction

Over the last several decades coastal regions across the globe have experienced an increase in the incidence of toxic or otherwise harmful algal blooms (Anderson et al. 2002, Paerl et al. 2014). Blue green algae (cyanobacteria) are the earth’s oldest oxygenic photoautotrophs and have exploited recent anthropogenic alterations of aquatic environments, most notably nutrient enrichment, to increase in density and distribution (Paerl and Otten 2013). A good example of cyanobacterial ecological opportunism is the rapid growth and persistence of dense blooms in response to nutrient enrichment in a vast range of water environments including coastal wetlands (Paerl 1996). Extensive harmful cyanobacterial blooms pose a serious threat to water quality and coastal biodiversity. As the environment and economic effects of such blooms have increased over the past few decades, including losses to fisheries, biodiversity, tourism, and public health risk, the need for effective mitigation strategies has also become critical (Paerl and Fulton III 2006, O'Neil et al. 2012).

The specific focus of this study is the benthic cyanobacteria, Lyngbya, which can be a major contributor to the total primary productivity of sub-tropical and tropical coastal wetlands (Paerl and Otten 2013). Lyngbya blooms are most
common in coastal ecosystems where marine waters are warm, calm and stratified, and after rainfall events that deliver elevated nutrient loads (Albert et al. 2005, Bell and Elmetri 2007). Dyer (1997) classified coastal environments according to tidal range such that if the tidal range is less than 2 m, the environment is termed microtidal, if it is between 2 m and 4 m then it is mesotidal, when it is between 4 m and 6 m the environment is classified as macrotidal, and when it is greater than 6 m, then it is hypertidal. Lyngbya blooms are often found in enclosed, microtidal systems such as Moreton Bay, Australia (Dennison et al. 1999, Albal et al. 2005), but have also been reported to occur in mesotidal systems such as Florida, USA and Okinawa, Japan (Furnas 1989, Wang et al. 1999). The number of reports for these types of coastal ecosystems have been increasingly reported for Lyngbya blooms, with blooms not typically expected in macro and hypertidal ecosystems (Osborne et al. 2001).

The occurrence of Lyngbya in Roebuck Bay, a hypertidal coastal embayment (tidal range >9 m) near Broome in Kimberly coast, Australia (Pearson et al. 2008), therefore leads to an interesting paradox. Roebuck Bay is a Ramsar listed coastal wetland which supports over 170,000 wader birds of more than 120 species feeding on benthic organisms in an extensive intertidal mudflat (Ramsar 1998, Carew and Hickey 2000), placing it among the richest mudflats in the world (Piersma et al. 2006). Since the early 2000s nuisance blooms have been occurring challenging us to identify the causal mechanisms bearing in mind the large tidal excursion.

The tidal range drives the hydrodynamic processes of mixing and advection in coastal waterways and affects the potential for phytoplankton blooms (Chan and Hamilton 2001). It is generally assumed that embayments that are mesotidal, macrotidal or hypertidal are unlikely to experience blooms since they mix the water column and decrease the residence time of algae in the photic zone, and encourage off-shore export (Monbet 1992).

Alpine and Cloern (1992) suggested that, bloom formation requires that algal doubling time exceeds the hydraulic residence time, the ratio of these two time
scales is termed the Damkohler number for advection dominated systems (see Oldham et al., 2013). It is very important from a management perspective to understand that hydrodynamics drives transportation of dissolved or suspended pollutants and other substances (Dimberg and Bryhn 2014). Thus, water retention time (a function of hydrodynamics) may be decisive for which of a series of alternative management actions is most effective (Lee and Park 2013, Phelps et al. 2013). Tidal mixing can also cause fine sediment to be resuspended, and the resulting turbidity reduces light available for photosynthesis (Cloern 1991, Monbet 1992). Vertical mixing in the water column of a hypertidal environment changes according to high-low tide cycles and spring-neap tide cycles. Lizon et al. (1995) observed, despite vertical homogeneity in salinity, chlorophyll-a and nutrient profiles, a vertical gradient in photosynthetic parameters in vertical mixing areas in hypertidal eastern English Channel in UK. They suggested strong light penetration in the water column and the photo adaptation processes occurring at neap tide could be major factors inducing a high primary production.

In contrast, McKinnon et al. (2015) observed that rainfall and runoff from adjoining catchments lowered salinity in coastal waters in Kimberly-Browse region north-west Australia. They showed that the water column was stratified both in terms of salinity and chlorophyll fluorescence, though temperature stratification was weak where water depth was greater than 50 m. In the Kimberly coast, freshwater input therefore potentially constitutes a significant enrichment of nutrients and sediments into coastal waters that are characterised by high insolation, high temperatures, and extensive mixing that maintains high turbidity and a shallow euphotic zone (<10 m) (McKinnon et al. 2015). These conditions may not be favourable for primary production due to energetic vertical mixing caused by large tidal currents and relatively shallow bathymetry. In contrast, stratification that develops under microtidal conditions can encourage blooms because it deepens the photic zone and reduces turbulence (Koseff et al. 1993, Cloern 1996). Therefore algal blooms seldom occur in well-flushed coastal systems because freshwater inflows dilute
nutrients and cell densities, and because tidal mixing related turbidity reduces the length of time that algae can be in the photic zone. Overall, the possibility of bloom formation is typically low in hypertidal coastal environments.

With this in mind, the drivers of Roebuck Bay’s Lyngbya blooms are not yet fully understood. Although Roebuck Bay is of great economic, cultural and environmental importance (Vogwill 2003, Curran and MacKenzie 2007, McQueen 2013) the limited number of investigations conducted to date (Carew and Hickey 2000, Estrella 2013) are not able to explain why blooms are occurring when the potential for growth is thought to be low. One of the main remaining challenges is to investigate the interactions between Lyngbya blooms and bio-physical processes but are thought to be related to elevated nutrient inflow, temperature and tidal exposure (duration of irradiance) in the intertidal zone during the wet season. Broome is also experiencing rapid urbanisation and has been identified as the key source of nutrient discharges with elevated nutrient inflow during the wet season, particularly during first flush stormwater events (Hearn 2014, Gunaratne et al. 2015). This study attempts to resolve the hypertidal embayment Lyngbya bloom paradox through interpretation of hydrodynamic modelling in the context of past reported accounts of bloom occurrence.

During the past decade a several modelling studies have been carried out to explore Lyngbya growth in coastal wetlands (Arquitt and Johnstone 2004, Hamilton et al. 2007, Gibbes et al. 2008, Kehoe et al. 2012). Kehoe et al (2012) argued that at eutrophied sites in Queensland coastal waters of Australia, variation in parameters such as temperature and light (PAR) rather than nutrients were more likely to cause blooms. They showed that at least one major bloom of Lyngbya in Deception Bay coincided with high light conditions in the water column that were optimal for Lyngbya photosynthesis. Further they discussed that causes of Lyngbya blooms are complex and multifaceted and nutrient availability is a necessary but not sufficient precondition for bloom. These studies, however, did not directly incorporate comprehensive analysis of
the physical processes driving the system. Considering the hypertidal hydrodynamic characteristics of Roebuck Bay we can hypothesise that Lyngbya blooms are largely dependent on PAR, temperature, and governed by hydrodynamic flushing, but that nutrient availability must also be a critical driver as blooms have not been recorded historically.

This study investigates the hydrodynamic characteristics, including tidal mixing, tidal exposure, tidal asymmetry, retention time, and environmental variables including salinity, temperature, PAR, and nutrients within Roebuck Bay. The numerical hydrodynamic model TUFLOW FV was applied to simulate the three-dimensional non-linear shallow water equations, which has previously been applied over a wide range of hydrodynamic systems including coastal embayments (BMTWBM 2013). TUFLOW FV was coupled with the Fortran-based Aquatic Ecodynamic (AED) model (Hipsey et al. 2013) for analysing and predicting the physical forcings that control intertidal zone hydrodynamics and water quality in the water column of the embayment. Within this chapter, incorporation of primary production and biomass accumulation of Lyngbya was not undertaken as it was thought to be premature at this stage as no water column biomass data exists for calibration and validation. Also with Lyngbya predominantly being benthic cyanobacteria, we need a more complex model to simulate its behaviour under exposed and submerged conditions as well as the water column. Therefore we use the model to only to explore the effect of physical and environmental variables on Lyngbya blooms of Roebuck Bay’s intertidal zone, and assess these variables within the context of the historical Lyngbya data.

The objectives of the present study were therefore to: 1) characterise the mixing due to physical forcing that control hydrodynamics of the intertidal zone of the Roebuck Bay hypertidal embayment; 2) characterise the abiotic factors that may contribute to Lyngbya blooms; 3) quantify the flushing capacity of the embayment; 4) asses the significance of spatial and temporal distribution of nutrients in the intertidal zone for Lyngbya blooms;
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4.3 Material and methods

4.3.1 Study site description

Roebuck Bay (18.1167° S, 122.2667° E) is located in the coastal tropics of the Kimberley region of Australia about 2200 km north of the Western Australian capital Perth (Fig. 4.1). Sub-tropical monsoonal is the predominant climate regime of the Broome region, with a wet season of high temperatures (December to March) and a dry season of moderate temperatures (April to November) (Table 4.1).

Figure 4.1 Map of Lyngbya bloom areas, sampling sites, and other key locations in the Roebuck Bay study area (Adapted from Estrella 2013 and Hearn 2014)

Broome has an average annual rainfall of 596 mm/year with typically 90% of the rainfall occurring during the summer (wet season) period from December to
March, often associated with tropical monsoons and cyclones (Bureau of Meteorology (BOM) 2013).

Wind conditions for Roebuck Bay are fairly consistent throughout the year with a monthly average of 13.5 km/h (Table 4.1). During the wet season, the most typical wind speed is 20-25 km/h while during the dry season wind speeds are typically 30-35 km/h. Tropical cyclones generate strong winds, predominantly from the east. Westerly winds are dominant at other times including the 2011/12 wet dry seasons (Bureau of Meteorology (BOM) 2013).

Table 4.1 Mean monthly meteorological data obtained at Broome Airport station (data from 1941 to 2012). Highlighted columns show the wet season data.

<table>
<thead>
<tr>
<th>Climate Variable</th>
<th>Jan</th>
<th>Feb</th>
<th>Mar</th>
<th>Apr</th>
<th>May</th>
<th>Jun</th>
<th>Jul</th>
<th>Aug</th>
<th>Sep</th>
<th>Oct</th>
<th>Nov</th>
<th>Dec</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean maximum temperature (°C)</td>
<td>33.3</td>
<td>32.9</td>
<td>33.9</td>
<td>34.3</td>
<td>31.5</td>
<td>29.1</td>
<td>28.8</td>
<td>30.3</td>
<td>31.8</td>
<td>32.9</td>
<td>33.6</td>
<td>33.9</td>
</tr>
<tr>
<td>Mean minimum temperature (°C)</td>
<td>26.3</td>
<td>26</td>
<td>25.4</td>
<td>22.6</td>
<td>18.2</td>
<td>15.1</td>
<td>13.7</td>
<td>14.9</td>
<td>18.5</td>
<td>22.4</td>
<td>25.1</td>
<td>26.5</td>
</tr>
<tr>
<td>Mean rainfall (mm)</td>
<td>178.7</td>
<td>179.8</td>
<td>101.7</td>
<td>26.2</td>
<td>27.3</td>
<td>19.7</td>
<td>7.1</td>
<td>1.7</td>
<td>1.4</td>
<td>1.4</td>
<td>8.9</td>
<td>55.6</td>
</tr>
<tr>
<td>Mean number of days of rain &gt;= 10 mm</td>
<td>4.3</td>
<td>4.7</td>
<td>2.6</td>
<td>0.7</td>
<td>0.7</td>
<td>0.5</td>
<td>0.2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.3</td>
</tr>
<tr>
<td>Mean daily solar exposure (MJ/m²)</td>
<td>23.1</td>
<td>22.1</td>
<td>22.6</td>
<td>21.6</td>
<td>18.5</td>
<td>17.2</td>
<td>18.6</td>
<td>21.7</td>
<td>24.3</td>
<td>26.8</td>
<td>27.8</td>
<td>25.7</td>
</tr>
<tr>
<td>Mean 9am relative humidity (%)</td>
<td>70</td>
<td>74</td>
<td>69</td>
<td>56</td>
<td>48</td>
<td>47</td>
<td>46</td>
<td>45</td>
<td>49</td>
<td>54</td>
<td>58</td>
<td>64</td>
</tr>
<tr>
<td>Mean 9am cloud cover (okas)</td>
<td>5.4</td>
<td>5.7</td>
<td>4.2</td>
<td>2.8</td>
<td>2.6</td>
<td>2.2</td>
<td>1.9</td>
<td>1.5</td>
<td>1.7</td>
<td>2</td>
<td>2.6</td>
<td>4.3</td>
</tr>
<tr>
<td>Mean 9am wind speed (km/h)</td>
<td>13.8</td>
<td>12.9</td>
<td>11.4</td>
<td>11.7</td>
<td>13.9</td>
<td>14.3</td>
<td>14.3</td>
<td>13.9</td>
<td>13.9</td>
<td>13.9</td>
<td>14.2</td>
<td>14.5</td>
</tr>
</tbody>
</table>

Tides vary from about 0.9 m on neap tides to 8 to 10 m on spring tides, which exposes around 160 km² of intertidal mudflat (Carew and Hickey 2000). The intertidal area on the west side of Dampier Creek (From Port Jetty to Dampier Creek) is affected by groundwater discharge from a wastewater treatment plant, golf course and urban storm water runoff (Hearn 2014, Gunaratne et al. 2015). This area also presents a high density of Lyngbya during the wet season.
Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads (Estrella 2013). This area is characterized by sandy sediments (Piersma et al. 2006), the existence of seagrass meadows and a narrow fringe of mangroves (Fig. 4.2). It is a zone used by the community as a recreational area including fishing and shell fish collecting (Estrella 2013).

During the wet season, 11 sites (Fig. 4.1) discharge stormwater runoff to the intertidal zone from the Broome urban catchment (747 ha). The more recently developed residential areas in Broome North contribute higher volumes of storm water (Fig 4.2 (b)) but a relatively low amount of nutrients when compared to the older part of Broome (Gunaratne et al. 2015).

![Figure 4.2](image)

**Figure 4.2** (a) An aerial image of the intertidal zone of Roebuck Bay from Broome Port Jetty to Dampier creek mouth area. Sea grass meadows are visible when the water is at the low tide mark; (b) An aerial image of Broome peninsula with inflow stormwater runoff from recently developed Broome north residential sub-catchments to Dampier creek mangrove area (Images courtesy of Richard Costin and Fiona West).

The intertidal zone is also affected by nutrient rich submarine groundwater discharge throughout the year (Hearn 2014). We use the areas of groundwater discharge from Hearn (2014) and these are colour coded on Fig.4.1. The east coastal strip of Roebuck Bay (From Dampier creek to One Tree) is situated in the proximity of an extensive mangrove area and away from point source...
nutrient discharges from the Broome town. This area is characterised by muddy sediments and a wider fringe of mangal (Piersma et al. 2006). The use of this side of the Bay for recreational activities is more restricted due to the inconvenience in accessibility.

4.3.2 Lyngbya biomass distribution

Lyngbya sampling was carried out for three consecutive wet seasons from 2009 to 2011 along eleven transects in the intertidal zone of Roebuck Bay (Estrella 2013). Only the exposed benthic Lyngbya lying on the tidal flat in receding tide was sampled due to logistical challenges in collecting water column samples (Estrella 2013). Two main sampling areas were selected in the intertidal area to characterize the biomass distribution. One was the area directly affected by nutrient inflows from Broome catchment (TR8, TR7, TR6) and the other area was adjacent to an extensive area of mangroves, to the east of the Dampier creek (TR5.3, TR5.2, TR5.1, TR5, TR4, TR3, TR2) (Fig 5.1). TR1 was selected as a control site as no Lyngbya has previously been observed there. Each transect ran perpendicular to the shoreline for 1 km across the intertidal zone and was located by handheld GPS (Fig 4.1). Every 100 meters, three replicate samples of Lyngbya were collected using a quadrat (25cm x 25cm).

The development of Lyngbya in the 2010/11 and 2011/12 wet seasons (Fig. 4.3) showed an exponential initial growth phase from approximately December to February. This was followed by a peak phase in February, followed by the decay phase, characterised by a decline in Lyngbya biomass, from February to April (Estrella 2013).
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Figure 4.3 Temporal variations of Lyngbya dried biomass (g/m²) for two successive wet seasons (modified from Estrella 2013)

The exact timing of each phase varied among sites with varying extension of bloom areas and densities (Estrella 2013). Considering this distribution we have selected three sites (TR8-400m, TR5.1-400m, and TR1-400m) as our sites of greatest interest for simulation purposes (Fig. 4.3).

Lyngbya dried biomass measured at each site over 11 transects (between port of Broome and One Tree) in Roebuck Bay was mapped onto a digital base map using ArcGis 10.1. These data were interpolated using ‘kriging’ technique (Childs 2004) to create continuous spatial predictions of exposed benthic biomass in the study area. Temporal variation at individual sampling locations was also used to validate the model against seasonal trends in the data set.
4.3.3 Model description

4.3.3.1 Hydrodynamic model

To simulate hydrodynamics, TUFLOW Finite Volume (TUFLOW FV) was adopted which is a hydrodynamic model that solves the three-dimensional Non-Linear Shallow Water Equations (NLSWE) on a flexible grid (BMTWBM 2012, Barry et al. 2013) was applied. TUFLOW FV is a modern and extensively validated 3D hydrodynamic modelling package that has been successfully applied and proven across water environments including coastal and ocean waters (Guard and Teakle 2011, Hipsey et al. 2013). TUFLOW FV has the ability to simulate temperature, salinity and density stratification in order to fully resolve baroclinic (density) driven processes. Intimately linked with this ability is TUFLOW FV’s capability to accept and respond to spatially variable high temporal resolution atmospheric forcing data from global atmospheric circulation models (including air temperature, relative humidity, long and short wave radiation and wind speed and direction) to fully simulate atmospheric heat-exchange processes as required (Barry et al. 2013). In the current application, turbulent mixing of momentum and scalars has been calculated using the Smagorinsky scheme in a horizontal plane and through coupling with the General Ocean Turbulence Model (GOTM) (Umlauf and Burchard 2003) for vertical mixing. The hydrodynamic model was used to determine shear stress, tidal exposure and tidal asymmetry, including flushing of stormwater and submarine groundwater discharge.

4.3.3.2 Model mesh and bathymetry

This study utilised the modelling package SMS10.1 (Aquaveo 2010) and ArcGIS 10 (Environmental Systems Research Institute (ESRI) 2011) to construct the model mesh which covers the whole Roebuck Bay and adjacent marine waters of the Indian Ocean (Fig. 4.4).
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Figure 4.4 Roebuck Bay computational mesh with zoomed in regions showing the part of intertidal zones affected with *Lyngbya* blooms with high resolution (30m-40m) grid cells; (A) near golf course and waste water treatment plant, (B) near the east side of Dampier creek to One Tree

The mesh consists of triangular and quadrilateral elements of different size. The horizontal mesh resolution varied between 30 m and 40m (in the intertidal zone near Broome) while the lower horizontal resolution was applied to the other areas of the bay and the adjacent ocean. Five vertical sigma layers were used. The bathymetric data for the mesh was sourced from the best available data, which was a combination of depth sounder data collected during a 2011 December field survey in the intertidal zone (high resolution) and a recent bathymetric map (low resolution) from Broome Ports Authority.
4.3.3.3 Boundary forcing for hydrodynamic model

Semi-diurnal tide data were used to force the model at the open ocean boundary. Hourly sea surface elevation data at Broome port were obtained from Bureau of Meteorology, (BOM) WA. The urban catchment area of Roebuck Bay covers 747 ha, which drains stormwater runoff at 11 sites to the bay during the wet season (Fig. 4.1). Submarine ground water discharges to the bay approximately in the middle of intertidal zone from four main discharge boundaries as per the findings of Hearn (2014) (different colours were used to differentiate discharge boundaries in Fig. 4.1. Apart from tidal and freshwater inflows, the domain was forced by meteorological data, which include solar radiation, wind, air temperature, humidity, rain and cloud cover gauged at BOM station at Broome Airport (Fig. 4.1). Macrotidal and hypertidal waterways usually fall into tide-dominated geomorphic classifications. Therefore it is safe to assume that wave conditions for Roebuck Bay have a small effect when compared to its hypertidal currents.

4.3.4 Hydrodynamic model calibration

4.3.4.1 Manning’s ‘n’ Distribution

A simple Manning’s ‘n’ distribution (Fig. 4.5) was assumed based on information about embayment substrate from previous studies (Pearson et al. 2008, Estrella 2013). This consisted of a typical “ocean bed” bottom roughness value for the majority of the bay, with slightly lower values in the creek areas and slightly higher values for sandy and mangrove areas where Lyngbya blooms occurred.
4.3.4.2  Tidal data

Tidal data obtained from The University of Hawaii Sea Level Center (UHSLC) were used to assess model performance. Tides from UHSLC location (-18.001, 122.219) inside the embayment were plotted against modelled tide elevations for the same location. The comparison is favourable, with excellent phase and amplitude correlations (Fig. 4.6).
Vertical mixing, due to wind induced tidal currents, is increasingly implicated as a key factor regulating ecological dynamics at population, community and ecosystem levels of organization (Petersen et al. 1998). Paerl and Otten (2013) suggested that relatively low turbulence creates favourable environments for cyanobacterial blooms in coastal areas while relatively high turbulence is most likely not a major factor inducing bloom formation as it increases the vertical mixing. We have estimated turbulence intensity, $\varepsilon$, measured as the rate of dissipation of turbulent kinetic energy (TKE) in the intertidal zone to examine the relationship between Lyngbya blooms and turbulence. $\varepsilon$ is expressed as the kinetic energy per unit mass per second, with units of velocity squared ($m^2 s^{-2}$). The methods selected for the vertical mixing model included a second order turbulence mixing model with a dynamic k epsilon method for total kinetic
energy, a dynamic dissipation rate equation for length scale method and a constant stability function (Bruce et al. 2014).

4.3.6 Tidal asymmetry

Tidal asymmetry is produced by the distortion of the tidal wave propagating on the coastal shelf and entering bays and estuaries (Dronkers 1986). The difference between flood and ebb magnitudes is more important for bedload transport, whereas the difference in slack durations is fundamental for suspended matter transport. Among tidal asymmetries, residual eddies due to headlands (i.e Broome Peninsula) have also to be mentioned, as they are likely to induce specific sediment transport or deposition (Whitehouse and Mitchener 1998). In this study we investigated horizontal velocities and vertical velocities in selected locations of the intertidal zone understand the flood/ebb domination (duration and velocity) as it directly correlates to the flushing capacity.

4.3.7 Tidal exposure

Unlike other coastal environments, tidal exposure in hypertidal embayments is a defining property of the ecology of the intertidal zone. Tidal exposure is the duration of air exposure episodes in between flood and ebb tide in the intertidal zone of a coastal embayment. The areas exposed to air during prolonged low-tide periods, get the direct sunlight while the areas submerged, light intensity is decreased with the water depth.
4.3.8 Water quality model

TUFLOW FV coupled with the AED water quality modules within the Framework for Aquatic Biogeochemical Modelling (FABM) was chosen for this study (Fig 4.7).

FABM is a general programming framework for biogeochemical models which is relatively new open-source code base that links aquatic models simulation of aquatic biogeochemical and ecological dynamics (Bruggeman and Bolding 2014).

This model combination was chosen due to the high level of flexibility in grid generation, and flexible and customisable open-source water quality modules. The model is also recently applied in a similar capacity to simulate biogeochemistry within the Yarra River Estuary (Bruce et al. 2014), Moreton Bay (McAlister et al. 2014) and Caboolture Estuary (Adiyanti 2014).

In this study, the Aquatic Ecodynamic (AED) modules (Hipsey et al. 2013) that were linked to FABM are used and they allow simulation of the C, N, P, O, and Si cycles including inorganic nutrient, organic matter, phytoplankton and zooplankton components. Since the focus of this study was on physical and
environmental parameters on Lyngbya growth the AED configuration was simplified to include nutrient parameters, PAR and retention time.

### 4.3.9 Boundary forcing for water quality model

Water quality data, including totals and dissolved inorganic nutrients (total nitrogen-TN, total phosphorous-TP, ammonium-NH$_4^+$, nitrate-NO$_3^-$, phosphate-PO$_4^{3-}$), temperature and salinity were measured in situ for stormwater runoff over the 11 drains during 2011/12 wet season (Gunaratne et al. 2015). Submarine groundwater nutrient data were obtained from the work done by Hearn (2014) for the same time period. Roebuck Bay water quality data were measured before (07/12/2011), during (03/02/2012) and after (15/05/2012) for nutrient and water quality parameters at 10 locations in the intertidal zone using a Hydrolab Multiprobe. Nutrient samples were analysed at the UWA analytical laboratory (Gunaratne et al. 2015).

Forcing data inputs for the model simulations (measured data used to set the boundaries of the model) included water temperature, salinity, concentrations of phosphate, ammonium, and nitrate. It has been suggested in other studies (Ahern et al. 2006) that iron is a key micronutrient for Lyngbya growth but given the large amount of iron rich Pindan sediments, washed into the bay during runoff events, iron was not taken into account in water quality modelling.

### 4.3.10 Water quality calibration

The model was calibrated using the temperature and salinity data taken before and during the 2011/12 wet season across Y1-Y2 (20 km) longitudinal section in the intertidal zone (Fig 4.8). Field data were obtained from the water quality monitoring at 10 stations along the Y1-Y2 transect.
Figure 4.8 Sites of interest, transect locations used for calibration and simulations of water quality and hydrodynamic parameters.

The results of the model calibration for the water temperature and salinity at the starting point (X1) surface layer are shown in Fig. 4.9.

Figure 4.9 Observed and modelled salinity (a) and temperature (b)
Salinities are well predicted throughout the embayment with the model showing complete recovery of salt following inflow events (Fig. 4.8 (a)). Salinity values in dry and wet season were checked for statistical significance and t-test suggested that the null hypothesis can be rejected, means they are statistically significant. Water temperatures at all sites show a slightly underpredicted trend throughout the study period (Fig. 4.8 (b)). This may be related to overestimation of freshwater inflows as there is an uncertainty in groundwater discharge rate, temperature and location in the intertidal zone.

4.3.11 Light penetration in bloom sites

As well as nutrients, light is an important influence on the dynamics of phototrophic species. Lyngbya derive energy from light, specifically photosynthetically active radiation (PAR), via the photosynthetic process (Watkinson et al., 2005). Blooms of Lyngbya in Deception Bay coincided with optimal PAR levels for Lyngbya photosynthesis (Kehoe et al. 2012). Wind and induced resuspension of sediment particles can have a significant effect on the PAR at shallow coastal sites (e.g.(Walker 1981)). The amount of PAR reaching the benthic environment is a function of the amount of PAR entering the water’s surface, the depth of the water, and the absorption properties of the water (Kirk, 1994). The absorption of PAR in water is influenced by water itself plus, the particles suspended, and substances dissolved in it (Kirk 1994). Particles include sediment but also phytoplankton. Small particles scatter light more than larger particles causing higher rates of attenuation in PAR. Sediment particles tend to be denser than water and so tend to sink and settle; they require energy to be resuspended into the water column. Here we used the AED water quality model to estimate PAR and euphotic depth in the intertidal zone. The euphotic depth is the area in which photosynthesis takes place and the ratio of euphotic to aphotic depth is usually small in turbid waters (Grobbelaar 1990).
4.3.12 Retention time and transport of tracers

Retention time is the water constituents residence time inside an aquatic system and is broadly recognised as important descriptors of coastal hydrodynamics. It is a convenient parameter to represent the time scale of physical transport processes, hence it is often compared with the time scales of biogeochemical processes (Wan et al. 2013). We used the coupled model to predict the “tracer” retention time and distribution at key Lyngbya bloom sites in the intertidal zone. To model the fate and transport of stormwater and groundwater (as a conservative tracer in space and time) we evaluated tracer concentrations away from the discharge locations. For simulations, tracer concentrations at inflow boundaries used as ‘1’ and initial tracer concentration of the open ocean boundary was 0.

4.4 Results

4.4.1 Observed Lyngbya distribution

The interpolated, exposed, benthic Lyngbya (dried biomass/m²) in Roebuck Bay that was present during the study was spatially and temporally mapped between Port of Broome and One Tree, (Fig. 4.10). No Lyngbya has been reported in other parts of the bay thus far, although large areas on the east and south of Roebuck Bay are effectively unsurveyed. The measured biomass was temporally and spatially variable. However two hot spots (TR8 and TR5.1) of Lyngbya biomass were found in the bay for the study period, with TR8 the most severe. TR8 was located between Port of Broome and Town Beach immediately down gradient from the Golf course and waste water treatment plant. TR5.1 was located between Dampier Creek and One Tree.
4.4.2 Tidal mixing

Tidal elevation, turbulence intensity, horizontal and vertical velocities were estimated across two main Lyngbya bloom sites (TR8, TR5.1) and the control site TR1 in the intertidal zone. Simulated tidal elevations are found to be similar in all locations in the intertidal zone but showed slightly different values in Spring tides (Fig 4.11 (a)). Note that tidal elevation becomes constant when the location was exposed to the air. The simulated tidal currents (Fig 4.11 (b,c,d)) in the three locations indicate the existence of tidal velocity asymmetry. Fig. 4.11 shows a tidal velocity asymmetry for a typical neap and spring tide periods. For the whole study period, all sites showed ebb dominance at a daily scale.
means ebb period is longer than flood period. But when moving from TR8 to TR1, the ebb dominance is weaker and current velocities are stronger (Table 4.2).

Figure 4.11 Simulated tidal elevations (a) and simulated mean tidal velocities and turbulence intensities (b,c,d) for the three selected sites (TR8,TR5.1,TR1) in a typical spring and neap tides during the 2011/12 wet season. Positive values represent flood velocities while negative values represent ebb velocities.

TR1 has the highest values for both maximum and mean values of horizontal tidal velocities whiles TR8 shows the lowest. Vertical velocities are negligible (<0.0001 ms$^{-1}$) compared to horizontal velocity magnitudes in all locations. According to the model results, maximum flood velocity occurs, approximately in between low tide and mid tide during a neap tide in the intertidal zone (Fig 4.10 and Table 4.2). Spring tide flood velocities are relatively smaller than neap tide flood velocities. In contrast, Spring tide ebb velocities are higher than neap
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tide ebb velocities. Maximum ebb velocities occur approximately in between high tide and mid tide during spring tides. Larger horizontal velocities and fluctuations cause higher turbulence intensities. TR8, alternated between larger turbulence intensities ($>10^{-2} \text{ m}^2 \text{s}^{-2}$) and smaller ($<10^{-9} \text{ m}^2 \text{s}^{-2}$) in regular intervals (Fig 4.11 (b,c,d)). TR8 showed the greatest variability in velocity and turbulence with very calm and very turbulent conditions occur throughout tide cycles. During turbulent conditions sediment and water mixing may provide additional nutrients, while the calm conditions will favour Lyngbya actual growth. Turbulent conditions occur during flood and ebb mid tides while calm conditions occur at high tide mark. Also velocities are relatively lower at the beginning of ebb tide due to tidal asymmetry in the TR8. This asymmetry may be largely due to the headland effect (Hir et al. 2000) of Broome Peninsula and created tidally driven residual currents particularly in Broome Town area including TR8 helped anti-clock wise water circulation (Fig 4.12).

Table 4.2 Comparison of maximum and mean velocities in ebb and flood tides on selected sites

<table>
<thead>
<tr>
<th>Parameter_Description</th>
<th>TR8</th>
<th>TR5.1</th>
<th>TR1</th>
</tr>
</thead>
<tbody>
<tr>
<td>Max_Flood_Velocity (m$^{-1}$)</td>
<td>0.4</td>
<td>0.53</td>
<td>0.96</td>
</tr>
<tr>
<td>Max_Ebb_Velocity (m$^{-1}$)</td>
<td>-0.39</td>
<td>-0.91</td>
<td>-0.78</td>
</tr>
<tr>
<td>Mean_Ebb_Velocity (m$^{-1}$)</td>
<td>-0.09</td>
<td>-0.17</td>
<td>-0.14</td>
</tr>
<tr>
<td>Mean_Flood_Velocity (m$^{-1}$)</td>
<td>0.04</td>
<td>0.1</td>
<td>0.13</td>
</tr>
<tr>
<td>Max_Turbulence_Intensity (m$^{2}$s$^{-2}$)</td>
<td>0.197</td>
<td>0.013</td>
<td>0.021</td>
</tr>
<tr>
<td>Mean_Turbulence_Intensity (m$^{2}$s$^{-2}$)</td>
<td>2.92E-03</td>
<td>4.10E-04</td>
<td>5.98E-04</td>
</tr>
</tbody>
</table>

Predicted spatial variations in tidal velocities (Fig 4.12 and 4.13), probably result from variation of tidal range across the embayment, bathymetry, water depth, and the morphology of the surrounding intertidal area (e.g., mangrove vegetation retards flows during spring tide flood events; bed roughness due to different substrates, and the shape of the embayment). In the main Lyngbya bloom area (TR 8), along shore currents were reduced by the presence of the natural obstruction of Broome peninsula. Site TR 5.1 also has reduced velocities and is down gradient from the main bloom area. The velocities at
sites TR8 and TR5.1 are mainly cross shore but surrounding areas are mostly alongshore

Fig. 4.12 (a,b) Depth averaged velocity vectors showing tidal asymmetry during flood tide for consecutive two hours before high water in a typical tide cycle of Roebuck Bay. Areas with decreased velocities were (< 0.2m/s) encircled in red.

Fig. 4.14 presents the model predicted maximum velocity (over a mean spring tide) at several locations (the key Lyngbya sampling sites and control site) versus the reduced maximum water depth. The latter is defined as the maximum water height over the tide divided by the tidal range and represents a normalized distance to the shore when the bottom slope is uniform. The normalised depth was used so that the velocity variable can be compared equitably with respect to the water depth variable as was done in Hir et al. (200). This issue is more important for algorithms and models that are based on some sort of distance (Hir et al. 2000) such as water depth. In this case, water depths were normalised to the same ‘dynamic range’ which is tidal range, with no units.

TR8 and TR7 had the highest velocity difference at the two ends (high and low tide marks) of the intertidal zone. This means in near shore areas (high tide
mark) TR8 and TR7 had the lowest velocities (<0.2m/s). TR6, TR5.3, and TR5.1 sites had relatively higher velocities than TR8 and TR7. All TR8, TR7, TR6, TR5.3, and TR5.1 sites, velocities increase with the distance from the shore. In contrast, the results at site TR1 didn’t show a considerable change in velocity with increased distance from the shore and the model predicts relatively higher velocities with maximum of 0.96 m/s (Table 4.2).

**Figure 4.13** (a,b) Depth averaged velocity vectors showing tidal asymmetry during ebb tide for consecutive two hours before low tide in a typical tide cycle of Roebuck Bay. Areas with decreased velocities were (< 0.2m/s) encircled in red.
Figure 4.14 Distribution of computed maximum horizontal bottom velocities at different locations of the intertidal zone of Lyngbya sampling sites at mean spring tide. The reduced maximum water depth is the local maximum water height over the tide, divided by the tidal range.

Turbulence depth profile distribution along Y1Y2 transect corroborates the spatial velocity distribution, predicting that Lyngbya bloom sites (TR8 and TR5.1) experienced greater turbulence intensities with frequent high magnitude fluctuations (Fig. 4.15). Turbulence intensity is greatest at the mid tide mark during flood tide. At TR1, reduced fluctuations in simulated turbulence intensities were predicted along the Y1Y2 transect over the course of a tidal cycle. Turbulence intensities are always high in creek areas including Dampier Creek and Crab Creek. Greater variability in intensity was found in TR8 and TR5.1 sites.
Figure 4.15 Spatial distribution of turbulence intensity – TKE along Y1Y2 transect covering half a tide cycle (10 am to 6 pm). Colour bar shows the turbulence intensity – TKE (m$^2$s$^{-2}$). Figure (a-b) shows a rising tide, while (c-e) shows a falling with (c) the high tide.

4.4.3 PAR

Despite the increased intermittent water turbulence experienced by Lyngbya bloom areas in the intertidal zone, combined with high water turbidity, light penetration through the relatively shallow water column may be sufficient to sustain high rates of photosynthesis during nearly all periods of submergence. Model results showed, surface PAR was changed from 0-450 Wm$^{-2}$ while bottom PAR ranged from 0-350 Wm$^{-2}$ for the 2011/12 wet season in the intertidal zone (Fig. 4.16).
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Figure 4.16 Simulated tidal elevations (a) and simulated surface PAR (b) and simulated bottom PAR (c) for the three selected sites (TR8, TR5.1, TR1) during typical spring and neap tides during the 2011/12 wet season.

The depth of the euphotic zone is crucial for Lyngbya growth in the water column. Simulated results suggested that euphotic depth at the high tide mark is about 0-1.5 m and never exceeded 2 m (Fig. 4.17 (a)). Light completely penetrated the water column at the blooms sites (TR 8 and 5.1) when the tidal elevation is between mid and low tide. In terms of the effect on inflowing groundwater on light penetration, inflow increased light penetration in bloom forming areas (Fig. 4.17 (b)).

Storm water inflow also caused increased the euphotic depth (Fig. 4.17 (c)). During rain events, even though the solar irradiance is lower due to cloud cover, sufficient light for Lyngbya growth was able to reach the bottom of the intertidal zone at high tide mark. During these time periods, PAR variation (difference
between surface and bottom PAR) reduced in lower tidal elevations and created a favourable environment for Lyngbya (Fig. 4.17 (d)).

Figure 4.17 Simulated daytime PAR (Y1-Y2 transect) for the three selected sites (TR8, TR5.1, TR1) during typical spring tide (a) dry season (pre-stormwater inflows) at mid day; (b) wet season, rainy day at midday; (c) wet season, dry day at mid day; (d) wet season, dry day at sunset flood tide in a typical spring tide. Color bar represents the surface PAR in Wm$^{-2}$

Previous studies showed optimal PAR is in the range of 600 to 700 mmol quanta m$^{-2}$ s$^{-1}$ (approx. 333 – 388 Wm$^{-2}$) for primary production (Watkinson et al. 2005, Kehoe et al. 2012). According to our simulation results during the dry season from 11am to 2pm PAR is in the optimum range (above 350 Wm$^{-2}$) for Lyngbya occurrence (Fig 4.17(a)). During the rainy days PAR is below the optimum range largely due to reduced solar irradiation (Fig. 4.17 (b)). During antecedent dry periods, again the intertidal zone provides sufficient PAR at mid day (Fig. 4.17 (c). In general, before 11am and after 2pm PAR levels are not sufficient under any condition (Fig. 4.17 (d) for primary production.

At high tide, light never completely penetrated the water column largely due to the turbid environment created by a high tidal range. However during periods of
reduced tidal level, thin or absent layers of water in the intertidal zone encourage bloom formation by creating optimum PAR levels. The model simulations worked only under inundated conditions, and simulated maximum PAR was 450 Wm\(^{-2}\) during mid day with the minimum of 20 cm water layer (wet/dry limit). But under the exposed conditions, the intertidal zone could be an ideal environment for Lyngbya occurrence, when PAR exceeds 350 Wm\(^{-2}\).

### 4.4.4 Suspended solids

The combined effects of suspended particles and phytoplankton can greatly decrease PAR by reducing light availability in the intertidal zone. Dissolved matter that colours the water and suspended solids from land runoff and the resuspension of bottom materials can be more prevalent in coastal environments than in the open ocean (McPherson and Miller 1987). The presence of the Broome coastal catchment, which produces highly turbid stormwater flow, largely contributed to increased suspended solids concentrations in the intertidal zone (Fig.4.18). During the flood tide the suspended solids were pushed towards the beach and tend to stay in the intertidal zone (Fig. 4.18 (a,b,c)). During the ebb tide, the concentrations slowly spread beyond the intertidal zone (Fig. 4.18 (d,e)).
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**Figure 4.18** Temporal distributions of suspended solids during a typical tidal cycle during a typical tide cycle at TR8 on a rainy day. (a) shows the start of the flood tide and (f) again shows the start of the flood tide in the next tide cycle.

TR8 showed the highest suspended solid concentration among all sites according to simulated results, important in the context of the potential for P in particular to be adsorbed onto fine grained particles. Results at TR8 suggest that PAR levels in the euphotic zone are slightly lower than the rest of the coastal waters probably due to the higher concentration of suspended solids coming from the catchment (Fig. 4.17 (a)). Before the start of the wet season, PAR levels in the intertidal zone’s euphotic zone provides optimum light condition for Lyngbya occurrence (Fig. 4.19 (a)).
Material coming through groundwater discharge slightly reduced the light penetration in the intertidal zone (Fig. 4.19 (b)) according to simulation results, but these are likely to be low as groundwater discharge is typically not high in suspended sediments or coloured dissolved organic matter. During the stormwater runoff periods, due to the presence of greater amounts of suspended solids as is typical for runoff, PAR level reduced in the euphotic zone (Fig. 4.19 (c)).

Results showed the concentrations of suspended solids started to increase with the beginning of flood tide and started to decrease with the ebb tide in regular time intervals. Results also suggested lowest suspended solid concentrations in the intertidal waters occur just after the high tide water mark with the slack water. This is further decreased during spring tides.
4.4.5 Intertidal exposure

Thin layers of water in the intertidal zone encourage bloom formation by creating optimum PAR levels. Tidal exposure (Fig. 4.20) follows the tidal range trends as more exposure time with longer tidal ranges. TR1 shows longer periods of exposure than the others two. During neap tides exposure at all sites was minimal while at spring tide the exposure time ranged from 2 to 5 hours.

![Tidal exposure model predictions](image)

**Figure 4.20** Histograms of the tidal exposure occurrence during daytime for 2011/12 wet season for TR8, TR5.1 and TR1 sites in the Roebuck Bay

The tidal exposure model predictions were used to identify tidal exposure period, probability of a given exposure duration and, the total number of hours of exposure during a daytime in a wet season. For example, over the study period the model predicts a 13% chance of 0 hours of exposure during day time for TR8, 15% occurrence of 0 hours during daytime for TR5.1 and 10% occurrence of 4 hours during day time for TR1. Results suggested both TR8 and TR 5.1 have similar exposure while TR 1 is more commonly exposed than TR8 and TR5.1.
We further analysed the tidal exposure time over the wet season and simulated spatial distribution of monthly cumulative submerged time in days over the study period (Fig 4.21). Submerged time is the opposite of exposure time which is defined as the number of days a certain area is submerged under water. Monthly cumulative submerged time (Fig. 4.21), suggests that the majority of the intertidal zone in the Lyngbya bloom sites were exposed more than 50% (approximately 60 days) of the total time. During a typical month in the wet season exposure time of the Lyngbya bloom areas is 15 days.

**Figure 4.21** Monthly cumulative submerged time at the end of the first month (a); second month (b); third moth (c) and final month (d) of the 2011/12 wet season. Colour bar represents the submerged time period in days.

### 4.4.6 Water quality

Salinity and temperature were predicted to assess if stratification was occurring along the depth profiles. A slight temperature drop was predicted at groundwater discharge sites before the start of storm water runoff (Fig. 4.22 (a)). Site TR5.1 is also predicted to have a slight temperature drop, even though
it’s not close to the submarine groundwater sites, due to intertidal zone hydrodynamics. When stormwater discharge to the intertidal zone occurs a slight stratification (warmer over cooler) is predicted at the sites (TR8 and TR5.1) close to stormwater drains (Fig. 4.22 (b)).

![Simulated salinity and temperature before and during the wet season across the Y1 Y2 transect.](image)

Before the onset of wet season rainfall when only groundwater inflow is occurring, there was slight drop in salinity at groundwater discharge locations (Fig. 4.22 (c)). Under the influence of rainfall, lower salinities occur at the top of the intertidal zone, closer to stormwater drain sites (TR8, TR7, and TR6). Overall, the intertidal zone showed slightly reduced salinity concentrations (1-2ppt) during the wet season relative to the dry season (Fig. 4.22 (d)). TR5.1 surprisingly showed slightly higher salinity level compared to other sites suggesting poor horizontal mixing in the intertidal zone due to potentially
because of increased water retention time due to the low velocities and circulation hydrodynamics in the area (see below).

### 4.4.7 Retention time and tracer transport

Retention time and tracer simulation involved introducing pulses of tracers along with stormwater or groundwater inflows. The period of retention of the tracers (assumed to be conservative) was estimated using the model over the 2011/12 wet season. Intra wet seasonal variations in the retention time at selected sites (TR8, TR5.1 and TR1) are shown (Fig. 4.23).

![Figure 4.23](image)

**Figure 4.23** Retention time for TR8, TR5.1 and TR1 sites before and during the 2011/12 wet season and the estimation based on simulated freshwater tracer concentration.

TR8 showed slightly higher flushing capacity with lower retention times compared to other sites. Wet season started on 02/01/2012 and suddenly retention time dropped in all sites with the first stormwater flows. Major drops in
retention time coincided with stormwater flows. Comparatively larger flows created larger drops in retention time. For the month of January in 2012, average retention times for TR8, TR5.1 and TR1 are respectively 17.2 days, 17.9 days and 18.9 days. The sites with higher freshwater input (TR8 and TR5.1) have more flushing as evident from lower retention time, however this is not a large decrease. Overall all the areas in the intertidal zone have similar capacity of flushing with typical range of 16 to 22 days.

Stormwater derived nutrient circulation patterns in the embayment were also investigated using tracers. In the 2011/12 wet season stormwater runoff started on 02/01/2012 and typically mimics rainfall. Fig. 4.24 (a,b) shows the normalised tracer concentrations. Increased tracer concentrations occur at the sites prone (TR8, TR5.1) to blooms as compared to the site which isn’t (TR1), for Lyngbya blooms.

Figure 4.24 Hourly precipitation from BOM (a), stormwater runoff from 11 Broome sub-catchments (b) as estimated in Chapter 2 and (c) simulated distribution of normalised stormwater tracer concentration 2011/12.
TR1 was only influenced when the stormwater runoff volumes were of high magnitude and occurred over a prolonged period (Fig 4.24(c)) but the source of tracers at the two sites prone to blooms were different.

At TR8, stormwater nutrients originate at the drains in the Broome Town area while at TR5.1 the major nutrient suppliers are the stormwater drains in Broome North area (Fig. 4.25).

Figure 4.25 Spatial variation of simulated stormwater concentrations in the intertidal zone of Roebuck bay immediately after a stormwater runoff event. The main patch of tracers moved different directions with different dilutions with the tidal circulations. Colour bar shows the normalised stormwater tracer concentration.

Circulation within of the embayment was the key driver for stormwater transport. Dilution varied considerably (up to 3 orders of magnitude) spatially and
temporally depending on the timing of stormwater discharge relative to the tidal cycle.

The spatial distribution of simulated groundwater tracers suggested intertidal zone hydrodynamics had the ability to spread the groundwater nutrients across all Lyngbya bloom sites, with increased concentrations at the discharge proximal sites. Before the start of the 2011/12 wet season (during December) groundwater tracers circulated between Port Jetty and Dampier Creek during a typical tide (Fig 4.26 (a-c) impacting site TR 8.

![Image](https://example.com/image.png)

Figure 4.26 Partial variations of simulated groundwater tracers in the intertidal zone of Roebuck bay 24 December 2011 (a-c) and 19 February 2012 (d-f). Colour bar shows the normalised stormwater tracer concentration.

During February however the tracers also occurred to the east of Dampier Creek, impacting site TR5.1 during a typical tide (Fig 4.26 (d-f). It believed this
has to do with changes in wind direction inducing different intertidal zone hydrodynamics.

The temporal distribution of groundwater tracers at the 3 transect sites is shown in Fig. 4.27. The highest concentrations almost always occur near the site of groundwater discharge (TR 8). The lowest concentrations almost always occur at site TR 1 which is furthest away. The change in wind direction in February causes higher values to occur at site TR5.1 at the same time as site TR8 shows a marked dilution (approximately 50%). This corroborated the spatial distribution findings presented above.

Figure 4.27 Simulated temporal distribution of normalised groundwater tracer concentration at TR8, TR5.1 and TR1 sites at before and during the wet season of 2011/12.
4.4.8 Nutrients

Tracers are conservative (non-reactive), however, in reality complex biogeochemical reactions occur within the embayment which would cause nutrients to be bound to sediments, consumed or even produced (if for example nutrients were bound to sediments in previous years) (Miguntanna 2009). To better understand the likely concentrations of nutrients present from all sources in a non-conservative context, the temporal distribution nitrate-nitrogen (N-NO$_3^-$) and Filterable Reactive Phosphorus (FRP) was simulated before and during the wet season. FRP showed a higher concentration in TR8 and adjacent Broome town coastal strip from Jetty to Town Beach (Fig. 4.28).

Figure 4.28 Simulated FRP distribution of Roebuck Bay during 2011/12 wet season. Colour bar represents ‘mmol P/m$^3$.’
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The changes of FRP with sub diurnal tides showed, under exposed condition, the highest concentrations of FRP was recorded in the Roebuck Bay for both before and during the wet season. Flood tides tend to reduce nutrient concretions as opposed to ebb tides which create a nutrient rich environment particularly in the vicinity of TR8 site.

FRP and N-NO₃⁻ trigger values for chemical stressors and toxicants for slightly disturbed marine in shores of tropical Australia are 0.005 and 0.008 according to Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines (ANZECC 2000) (Fig. 4.29).

Figure 4.29 Simulated N-NO₃⁻ distribution of Roebuck Bay during 2011/12 wet season. Colour bar represents ‘mmol N /m³’.
Temporal variation of nutrients showed slightly higher concentrations during dry season than wet season (Fig. 4.30). Our simulated results showed FRP values are slightly higher than trigger values during the dry season and slightly lower during wet season. Conversely N-NO$_3^-$ results indicated 1000 times larger values than the trigger value throughout the year (Fig. 4.30).

At site TR8 high nitrate concentrations fluctuate from 5-20 mg/L, occurred frequently, driven by the proximal groundwater discharge. At site TR 5.1 the concentration of nitrate seemed to be more influenced by the timing of freshwater runoff but rarely exceeded 5 mg/l. TR1 concentrations only always slightly lower than TR5.1 concentrations.

**Figure 4.30** Simulated temporal distribution of nitrate-nitrogen (N-NO$_3^-$) and Filterable Reactive Phosphorus (FRP-P) concentration at TR8, TR5.1 and TR1 sites at before and during the wet season of 2011/12.
4.4.9 Gross Primary Production (GPP)

For simplicity we will define our use of “gross primary production (GPP)” as the potential for phytoplankton contribution to system production; “production” is either the mass of carbon fixed over a period of time (Hipsey et al. 2013). Due to the limited biomass data, this is used as a proxy for Lyngbya biomass production and the model simulations do not yet explicitly predict Lyngyba biomass production, however the GPP computation accounts for light, nutrients and temperature on the potential for photosynthesis. Our results suggested GPP in the intertidal zone of the Roebuck Bay is predicted to be considerably larger than the rest of the bay area (Fig. 4.31), particularly in the vicinity of the bloom sites due to less nutrient limitation. Gross primary production has increased from December to February and then started to decrease (Fig. 4.31) with the reduction of nutrient flows from the catchment.

**Figure 4.31** Spatial and temporal distribution of GPP in the roebuck Bay during the 2011/12 wet season. Colour bar represents the GPP in mmol C /hour.
4.4.10 Management scenario assessments

Model simulations were performed to test two ‘idealised’ management scenarios:

- Scenario 1 - halving the groundwater nutrient loading
- Scenario 2 - halving the stormwater nutrient loading

GPPP was used as a proxy to investigate the changes in Lyngbya biomass production with varying nutrient loadings. Simulation results showed during the dry season, halving of the groundwater nutrient load reduced the GPP in the intertidal zone significantly (Fig. 4.32 (a&b)).

During the wet periods, reduction of groundwater nutrient loads (scenario 1) largely reduced the GPP (Fig. 4.32 (d&e)). Halving of stormwater nutrient load didn’t show a significant effect compared to the original state of nutrient loading (Fig. 4.32 (f)).

Along with GPP, nutrient concentrations were investigated for each scenario with respect to the original condition. When groundwater discharge nutrient load was halved (scenario 1), the effect was significant in reducing nutrient levels in the intertidal zone of Roebuck Bay (Fig. 4.33). At TR8, nutrient levels are 30 times larger than TR1 for nitrate-nitrogen (Fig. 4.33 (a&b)). For FRP, it showed a similar trend where values at TR8 4 times larger than TR1 (Fig. 4.33(c&d)). During the wet season at TR8 showed slightly lower values than the dry season, conversely at TR1 wet season values are slightly higher than the dry season values. Which implies TR8 is largely affected by groundwater but TR8 is not.
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Simulation results suggested, halving of stormwater nutrient loads (scenario 1) only made slight changes of nutrient levels in the intertidal zone (Fig. 4.34). The reduction of nitrate-nitrogen levels are less than 1% at both TR8 and TR1 (Fig. 4.34 (a&b)). The reduction of FRP at TR8 is less than 2% but at TR1 it ranged from 1-5% (Fig. 4.34 (c&d)).
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Figure 4.33 Nutrient concentrations at TR8 and TR1 for original and halved groundwater nutrient load scenarios. (a) nitrate-nitrogen (N-NO$_3$) at TR8; (b) nitrate-nitrogen (N-NO$_3$) at TR1; (c) Filterable Reactive Phosphorus (FRP-P) at TR8; (d) Filterable Reactive Phosphorus (FRP-P) at TR1
Figure 4.34 Nutrient concentrations at TR8 and TR1 for original and halved stormwater nutrient load scenarios. (a) nitrate-nitrogen ($\text{N-NO}_3^-$) at TR8; (b) nitrate-nitrogen ($\text{N-NO}_3^-$) at TR1; (c) Filterable Reactive Phosphorus (FRP-P) at TR8; (d) Filterable Reactive Phosphorus (FRP-P) at TR1

4.5 Discussion

4.5.1 Model suitability and limitations

The hydrodynamics of Roebuck Bay are primarily a function of tidal forcing, climate (particularly wind), bathymetry and substrate roughness. We have used best available data for all these parameters, which are explicitly included in our model. Our results show that the model satisfactorily simulated Roebuck Bay hydrodynamics (tidal mixing, tidal exposure) and environmental variables (PAR, nutrients, salinity, temperature, and flushing capacity). There was an under
estimation of temperature in simulations probably due to the uncertainties of submarine ground water inflow volumes and temperatures or bias in the atmospheric forcing variables. Groundwater inflow along the north shore of Roebuck Bay east of Dampier Creek was not incorporated due to a lack of data and the reduced focus on this area compared to the Port to Town Beach area where the worst blooms occur and groundwater nutrient inflows have been reported. The uncertainty in temperature might affect dependant environmental variables slightly.

Only a single data set was available for calibration of tide, temperature and salinity but a comprehensive data set was not available for the validation of the model. This occurs due to a lack of pre existing monitoring and insufficient resources to collect a comprehensive, model calibration dataset during the course of this study. No data exists to calibrate tidal velocity and turbulence simulations. However, the hydrodynamics and water quality results showed good agreement with the limited available observations and available literature. For example Carew and Hickey (2000) used remote sensing to predict the Roebuck Bay intertidal area at 190 km$^2$, approximately 45% of the total bay area, with water velocities of up to 20 cm/sec mid tidal cycle. In this study, our results showed the maximum area exposed was 272 km$^2$ (51% of the bay area). For horizontal tidal velocities, our results showed tides travelling at mean speed of 13 cm/s in ebb and 9 cm/s in flood in mid tide cycle with the maximum speed of 69 cm/s and 63 cm/s in ebb and flood tides respectively. Carew and Hickey (2000)'s estimates were obtained with much less sophisticated methods than are used herein (our model incorporates hydrodynamics and bathymetry etc) however our results are in a good agreement as their estimate lies between our mean and maximum values.
Intertidal zone hydrodynamics

Roebuck Bay's intertidal zone showed nearly horizontal water movement over typical tide cycles. This occurs because the wave length of the tide is large relative to the intertidal zone width and there is a significant slope and bottom friction at upper intertidal zone (above mean sea level) (Le Hir et al. 2000). Our analysis agrees with Le Hir et al (2000), as site TR1 showed slightly higher tidal elevations than TR8 and TR1 has reduced substrate roughness (mud and silt) compared to TR8 (sandy bottom with seagrass beds).

Cross-shore currents which exceed the long-shore current occur when the intertidal zone is wide and/or when the alongshore currents are reduced by the presence of a natural or man-made obstruction, such as a headland, spit or breakwater (Le Hir et al. 2000). Flow in the coastal strip from Port Jetty to Dampier Creek appeared to be controlled by cross shore propagation (Fig. 4.12 (a)) due to the wider intertidal zone (> 5km) and/or headland effect from Broome Peninsula. The flow is mainly longshore in the intertidal zone bloom areas east of Dampier Creek (Fig. 4.12 & 4.13), controlled by the large scale tidal propagation.

Water flow in the creek areas (Dampier Creek, and Crab Creek) is mainly across-shore in the intertidal zone (Fig. 4.12 & 4.13). The intrinsic tidal asymmetry was observed in the upper part of the intertidal zone showing near zero velocities and turbulence intensities at high tide mark (Fig. 4.11). The intrinsic asymmetry is merely the fact that the upper part of an intertidal zone experiences only one slack at high water (Postma 1961, Le Hir et al. 2000). Sites TR8, TR5.1 and TR1 all have different velocities, highest at TR1 and lowest at TR8 which exhibited differential tidal asymmetries in favour of ebb flow. In general, the areas with decreased velocities were overlapped with key Lyngbya bloom sites (TR8 and TR5.1) and created calm environment which is one of the key factors for Lyngbya blooms. In contrast, tidal creek areas (Dampier creek and Crab creek) with non blooming sites generally showed higher velocities (>1m/s).
As a consequence of the intrinsic asymmetry, particles coming onshore during the flood begin to settle onto the upper intertidal zone when the bottom stress is below the critical value for deposition. Depositional processes are favoured by the decreasing velocities near the shore. This allows time for the partial consolidation of newly deposited cohesive material which may then be retained. The subsequent resuspension of the deposited material requires the ebb flow speed to exceed some minimum, and often this minimum is hardly reached in the upper intertidal zone except if the ebb current is locally dominant (Postma 1961). This process may retain particulate nutrients, introduced to the bay through surface and groundwater inflows, in the upper part of the intertidal zone which contribute to elevated nutrient levels in the intertidal zone and subsequent Lyngbya blooms.

Turbulence over a range of scales (cellular to ecosystem) plays an important regulatory role in cyanobacterial bloom dynamics (Reynolds 1987, Kehoe 2010). At TR8, turbulence intensity, showed a periodic variation between a minimum value of $0 \ (\approx 10^{-9})$ and $0.01 \text{ m}^2 \text{ s}^{-2}$ in a ebb and flood flows of a spring tide. Sites TR5.3 and TR1 exhibited moderate but continuous turbulence intensities (i.e. $10^{-3} - 10^{-4} \text{ m}^2 \text{ s}^{-2}$). Turbulence intensity stratification occurred, for 3-4 hours, during rising and falling waters but none was predicted at high water mark. Lyngbya prefers calm, vertically stratified conditions, given adequate nutrient supplies (O'Neil et al. 2012). According to the simulation results turbulence intensity at TR8 periodically (every tide cycle) experienced lower turbulence intensities than at TR1 throughout the study period. When intermittent weak stratification occurs during favourable growth periods, Lyngbya can propagate quickly (Albert et al. 2005). Shallow, periodically mixed intertidal zones in tropical and subtropical coastal ecosystems exhibit these tendencies, including Roebuck Bay. Increased or excessive turbulence however inhibit photosynthesis, nitrogen fixation and growth, causing disaggregation, filament damage and rapid death among diverse colonial genera in cultures and in natural settings (Paerl 1990). Optimal bloom growth often relies on mutually beneficial microbial consortial interactions with host cyanobacteria. Turbulence
can disrupt consortia and negatively affect growth (Paerl et al. 2000). The hydrodynamics of Roebuck Bay appear conducive to bloom formation.

4.5.3 The role of abiotic factors

The role of abiotic factors (i.e. salinity, water temperature and PAR) and ambient water quality parameters (i.e. dissolved inorganic nutrients and trace metal concentrations) have been shown to be critical for the formation of Lyngbya blooms in both coastal embayments (Albert et al. 2005, Watkinson et al. 2005). According to simulated results Roebuck Bay intertidal zone salinity in TR8, TR5.1 and TR1 declined from a mean value of 36.8 ppt, 37.2 ppt, 37.4 ppt before the wet season to a mean value of 36.2 ppt, 36.2 ppt, 36.4 ppt during the wet season respectively.

Even though Lyngbya has well adapted to a broad salinity tolerance with substantial photosynthetic activity from 9 to 36 ppt (Watkinson et al. 2005). With Roebuck Bay being mildly hypersaline (>37ppt) during the dry season, Sites TR5.1 and TR1 are less favourable for Lyngbya photosynthesis prior to onset of the wet season. During the wet season, stormwater runoff and increased groundwater discharge (Hearn, 2014) salinity drops and conditions are favourable for Lyngbya growth, particularly in areas of water body stratification. Part of the reason why the highest Lyngbya biomass is found at TR8 is likely related to its favourable environmental conditions all year round.

Simulated results of temperature at TR8, TR5.1 and TR1 showed mean values of 28.6 °C, 28.7 °C, 28.0 °C at the end of the dry season (December) and 28.2 °C, 28.2 °C, 27.5 °C during the wet season (February). 25 °C has been found to be suitable for Lyngbya growth in Moreton Bay but it can grow successfully at higher temperature (Watkinson et al. 2005). Elevated temperatures will favour cyanobacteria blooms since cyanobacteria normally exhibit optimal growth rates at high temperatures (to a maximum of 30 °C) (Coles and Jones 2000). In fact,
Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads

Recent studies have revealed that the predicted increase in global temperatures due to climate change, will increase the number of days favouring cyanobacteria blooms and anoxic events may become more frequent and last longer (Neumann et al. 2012). Temperature conditions are favourable for Lyngbya growth throughout the study time period.

Previous research work has already identified the role of nutrients in the development of Lyngbya. Measurements of physiological responses to nutrient addition have identified iron, nitrogen and phosphorus as being capable of stimulating Lyngbya growth on already established Lyngbya mats (Watkinson et al. 2005, Ahern et al. 2008). Roebuck Bay simulated nitrate and FRP results don’t seem to exceed water quality trigger values for slightly disturbed estuaries of tropical Australia (ANZECC 2000). But Estrella (2013) observed the sediment nutrient and pore water nutrient values were well above the trigger values. This indicates that Roebuck Bay nutrient has a problem with flushing. Even though nutrients are in elevated levels, it is a necessary but not sufficient precondition for Lyngbya blooms (Kehoe et al. 2012).

Anthropogenic nutrients are being discharged to Roebuck Bay. Stormwater and groundwater inflows to Roebuck Bay have been documented to contain considerable nutrient loads (Hearn 2014, Gunaratne et al. 2015). Estrella (2013) observed the elevated sediment nutrient and pore water nutrient concentrations during investigation of intertidal benthos across the study site. This, along with the presence of Lyngbya blooms, suggests that nutrients are not being flushed out of Roebuck Bay, in fact they may be accumulating according to our hydrodynamic predictions. Even though nutrients are in elevated levels, it is a necessary but not sufficient precondition for Lyngbya blooms (Kehoe et al. 2012). The environmental variables need to also be within Lyngbya’s tolerances, which they are in the case of Roebuck Bay.
4.5.4 Bloom timing

Lyngbya studies in Moreton Bay suggested that it is the abiotic parameters such as temperature and PAR, which are more temporally variable, that trigger bloom formation but additional nutrients (relative to background) must also be present (Kehoe et al. 2012). Lyngbya blooms in Roebuck Bay start to occur before the first stormwater flows. A number of factors may be contributing to this. The intertidal zone is typically covered with a film of water during tidal exposure. This occurs as much of the intertidal zone is receiving submarine groundwater discharge all year round, rich in nutrients in places (Hearn, 2014) (Fig. 4.35).

![Figure 4.35](image)

**Figure 4.35** Lyngbya blooms TR8, Roebuck bay Broome, Australia, on 19 December 2011. (a) Distribution of Lyngbya in mangroves; (b) close up of Lyngbya smothering mangrove roots

The effect of substrate roughness and low bathymetric gradients retarding tidal water outflow also contribute. This decreases the effects of drying creating an appropriate habitat for continued benthic Lyngbya growth. It is also likely that exposed parts of the intertidal zone may be able to support enhanced biomass growth as submarine groundwater discharge and stormwater runoff would deliver elevated nutrient concentrations and ensure desiccation is minimal during exposure periods.
Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads

The submarine groundwater discharge in the vicinity of TR8 is almost certainly rich in nutrients. It is located immediately down gradient (a few hundred meters) from a wastewater treatment plant (WWTP) and golf course irrigated with treated wastewater (Hearn 2014) (Fig 4.1). WWTP input of nutrients has severely affected coastal wetlands elsewhere with eutrophication in Tampa Bay, Florida (Greening and Janicki 2006), and in tributaries of Chesapeake Bay, USA (Kemp et al. 2005) some important examples.

As well as nutrients, light is an important influence on the dynamics of phototrophic species like Lyngbya. According our PAR simulations, maximum PAR in the film of water would be greater than 400 W/m\(^2\) during tidal exposure. Even at low tide levels, with the increase of euphotic zone maximum PAR should be around 400 W/m\(^2\) as in Roebuck Bay. The longer the duration of exposure, the higher the derived energy from light will be, encouraging Lyngbya biomass growth.

Apart from light and nutrients, vertical mixing affects phytoplankton growth (Cloern 1991). Our study (Fig. 4.11) shows relatively weak mixing in the bottom layer and relatively strong mixing in upper layer of Roebuck Bay’s water body. Koseff et al (1993) showed that regions of high biomass correspond to regions of low turbulent mixing. High amounts of turbulent mixing rapidly mixe the biomass leading to a nearly uniform distribution, which is then depleted by benthic grazing. We suspect that benthic grazing in Roebuck Bay is not a dominant force for control of Lyngbya spp. because it deters grazing by generalist herbivores such as fish, crabs, urchins, and other macroherbivores due to its toxic nature (Nagle and Paul 1999, Thacker and Paul 2004). The low turbulence in Roebuck Bay particularly in the lower part of the water column favour the growth and persistence of benthic Lyngbya. Further (Kehoe et al. 2012)
Nutrient flushing and retention time

Nutrient retention time in Roebuck Bay is determined by timing and magnitude of stormwater and groundwater inflows, embayment morphometry (i.e. embayment shape, bathymetry) and intertidal hydrodynamics. Roebuck Bay is continuously receiving anthropogenic nutrients throughout the year via groundwater inflow. Stormwater nutrients inflow typically only occurs during the wet season, with the greatest loads during the first flush, at the onset of the wet season. There has been a perception held by some stakeholders that Roebuck Bay’s high magnitude tidal regime would effectively flush all nutrients out of the system into the open ocean, diluting them to the point where impacts would be minimal. This is clearly not the case as Lyngbya blooms are persisting year after year. Our investigation into hydrodynamics suggests that the introduced nutrients will typically have long retention times, there will be persistent nutrients in the water column which may become adsorbed onto fine sediments particularly in the case of phosphorous.

There is variability in retention time depending on the timing of nutrient inflow, particularly in the case of stormwater as groundwater inflow is perennial. For example, if nutrients were discharged via storm water drains into the embayment at falling tide, they would be rapidly flushed out by ebb currents, which results in a very low residence time. Conversely, if they are released during flood currents, nutrients would be transported into the embayment (in particular the north shore of Roebuck Bay) with a long (typically almost 20 day) residence time. Consequentially the timing of the first flush of the storm water system could be critical in terms of bloom severity, if shock loading of nutrients occurs when conditions are right for biomass growth and long retention times then blooms will be greater in magnitude. Retention time also varies seasonally and spatially.

Lower retention times were observed before the wet season during December (10-12 days) while during the wet season (January-March) retention time ranged from 17-20 days. This may be due to the effect of intertidal
Gunaratne, G.L.
Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads

Hydrodynamics, tidal range was relatively low (6-7m) during December while tidal range was slightly higher (7-10m) in spring tides January-March. The change in prevailing wind direction (westerly during the dry season and easterly during the wet season) will also be contributing to this.

Retention time also depends on location. During January the average retention time is 17.2, 18 and 19 days at TR8, TR5.1 and TR1 respectively. The slightly lower retention time at TR8 suggests its frequent exposure to the freshwater inflows may assist in flushing at this location as well as its proximity to the headland of the Broome Peninsula. TR1 is a few kilometres away from stormwater and groundwater discharge locations, but has higher retention time. Apart from freshwater flows, variations in the hydrodynamics of each site also affect the retention time.

4.5.6 Damköhler Number: The balance of growth vs flushing

To assess the interplay between retention of nutrients and subsequent reaction from Lyngbya blooms in the embayment we are using the generalised form of the Damköhler Number ($N_E$). The Damköhler Number was recently adapted to characterise the fate of material in hydrologically and hydrodynamically connected (advection dominated) systems (Ocampo et al. 2006, Oldham et al. 2013). To our knowledge, this has not been previously applied to intertidal zone studies. For our particular case, the dimensionless $N_E$ is defined as the ratio between transport time scale ($\tau_T = \text{retention time}$) and reaction time scale ($\tau_R = \frac{1}{K_{\text{Lyngbya}}}$) where $K_{\text{Lyngbya}}$ is the specific growth rate. The growth rate of Lyngbya in Moreton Bay was found to be 0.33 day$^{-1}$ (doubling time ~ 3 days) (Watkinson et al. 2005). Thus information on average monthly retention time and $N_E$ in TR8, TR5.1 and TR1 were analysed to characterise Lyngbya's opportunity to proliferate during the study period (Table 4.3). These were
compared to published data on other Lyngbya studies to compare our hypertidal embayment to micro and mesotidal environments.

\( N_E \) gives an indication of the possibility for proliferation of Lyngbya. For study sites, with \( N_E << 1 \), doubling of biomass time is greater than retention time hence Lyngbya cannot bloom. When \( N_E >> 1 \), Lyngbya will have substantial opportunity to utilise available nutrients and potentially bloom. Alpine and Cloern (1992) argued that bloom formation requires net algal growth rate is less than hydraulic residence time (\( N_E >> 1 \)). Our results predict a Damköhler Number in Roebuck Bay greater that the \( N_E >> 32 \) (\( \tau_E = 10.71 \) and \( \tau_P = 0.33 \)) have potential for Lyngbya bloom formation.

### Table 4.3 Parameter values and data sources used to calculate the Damköhler Number

<table>
<thead>
<tr>
<th>Time period</th>
<th>Study Site</th>
<th>Typical retention time (days)</th>
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<td>10.71</td>
<td>32.45</td>
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</tr>
<tr>
<td>Jan</td>
<td></td>
<td>17.19</td>
<td>52.09</td>
<td>This study</td>
</tr>
<tr>
<td>Feb</td>
<td></td>
<td>17.17</td>
<td>52.04</td>
<td>This study</td>
</tr>
<tr>
<td>Mar</td>
<td></td>
<td>18.19</td>
<td>55.12</td>
<td>This study</td>
</tr>
<tr>
<td>Dec</td>
<td></td>
<td>10.89</td>
<td>33.01</td>
<td>This study</td>
</tr>
<tr>
<td>Jan</td>
<td>TR5.1_Roebuck Bay</td>
<td>17.99</td>
<td>54.52</td>
<td>This study</td>
</tr>
<tr>
<td>Feb</td>
<td></td>
<td>17.59</td>
<td>53.30</td>
<td>This study</td>
</tr>
<tr>
<td>Mar</td>
<td></td>
<td>19.02</td>
<td>57.64</td>
<td>This study</td>
</tr>
<tr>
<td>Dec</td>
<td>TR1_Roebuck Bay</td>
<td>11.24</td>
<td>34.06</td>
<td>This study</td>
</tr>
<tr>
<td>Jan</td>
<td></td>
<td>18.96</td>
<td>57.46</td>
<td>This study</td>
</tr>
<tr>
<td>Feb</td>
<td></td>
<td>18.50</td>
<td>56.05</td>
<td>This study</td>
</tr>
<tr>
<td>Mar</td>
<td>Annual Deception Bay,</td>
<td>20.00</td>
<td>60.61</td>
<td>This study</td>
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<tr>
<td></td>
<td>Queensland, Australia</td>
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<td></td>
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</tr>
<tr>
<td>Annual</td>
<td>Annual Tampa Bay, Florida,</td>
<td>55.00</td>
<td>166.67</td>
<td>(Gibbes et al. 2014)</td>
</tr>
<tr>
<td></td>
<td>USA</td>
<td></td>
<td></td>
<td>(Meyers and Luther 2008)</td>
</tr>
</tbody>
</table>

#### 4.5.7 Nutrient dynamics in the intertidal zone

The spatial variability in the distribution of stormwater and ground water tracers is high in Roebuck Bay, but the tracers from both sources are clearly affecting
the main Lyngbya bloom sites along the north shore of Roebuck Bay. The main bloom site at TR8 was predominantly influenced by proximal nutrient inflow (mostly groundwater but also storm water). Nutrients are however primarily transported to TR5.1 by the embayment’s particular hydrodynamics. At current levels of nutrient inflow it seems unlikely that nutrients will be (or have been) transported to TR1 advectively. Thus we suggest that in the vicinity of sites TR8 to TR6 there is a sufficient nutrient load in the water column, sediments and pore water to facilitate growth to differing degrees. Sites TR5.3 to TR2 only experience elevated nutrient levels during the wet season due to advective transport of nutrient inflows. Even at the relatively oligotrophic sites (i.e. TR5.3, TR5.1, TR2) Lyngbya can grow during the wet season if the conditions are right.

4.6 Conclusion

This study highlights that, contrary to some stakeholder opinions, Roebuck Bay’s intertidal zone is a suitable ecological niche for Lyngbya blooms and that existing levels of anthropogenic nutrients can create blooms. In fact we can conclude that storm water, groundwater and marine water hydrology of Roebuck Bay (including its urbanising catchment) facilitates Lyngbya blooms. In a hypertidal coastal embayment such as Roebuck Bay, water and solute exchange at the land-ocean boundary relies on intertidal hydrodynamics. The long retention time leads to persistence and possibly accumulation of nutrients in the intertidal zone, enhancing the ecological niche for Lyngbya which subsequently blooms. The tendency to bloom, however, is negated by effects of vertical mixing and settling, light climate and anthropogenic nutrients. We have put forward a model to understand the physical processes (which are controlled by intertidal hydrodynamics and environmental variables) critical to the ability of Lyngbya blooms to form in Roebuck Bay.
Our model does not have enough fidelity nor was it sufficiently calibrated and validated to comprehensively predict optimal management. Additional data collection, model development and model application is required prior to undertaking broad scale management. The model requires validation which requires field measurements of tides, currents, waves, salinity, temperature, bed roughness, meteorological parameters (wind, solar radiation, cloud cover and rainfall), and nutrient concentrations in storm, marine and groundwater. It is advised that the responsible agencies carry out or commission long term monitoring to facilitate long term modelling studies.

Any measures to reduce nutrient inflow will have benefit, however it is difficult based on our analysis to be conclusive about precise benefits of specific management actions. The model encapsulates the essential physical and environmental variables such as tidal velocity asymmetry, turbulence intensity, PAR, nutrient availability, salinity and temperature in a three dimensional numerical model. By applying the model to before and during the wet season, it is also possible to explain the hydrodynamic and environmental conditions that favour Lyngbya blooms in the intertidal zone during the mid to late wet season and why growth begins prior to the wet season. This is critical to understanding the complete temporal and spatial distribution of Lyngbya.

The model proved invaluable in explaining the complex myriad of spatial and temporal patterns of observed Lyngbya bloom occurrences in Roebuck Bay. This work is a first step towards developing a Lyngbya biomass model in the intertidal zone of tropical embayments, important for this and many other sites around the world.

4.7 Acknowledgements

The many stimulating discussions with Brendan Busch and Aditya Singh and computer modelling support from Casper Boon at UWA Aquatic Ecodynamics Research Group are well appreciated.
Chapter 5. A coupled hydrodynamic-biogeochemical model for assessment of Lyngbya growth and biomass accumulation in a hypertidal coastal embayment

5.1 Abstract

Extensive blooms of Lyngbya Majuscula (hereafter Lyngbya) have been recorded in tropical coastal waters across the globe in recent years, including Roebuck Bay north-western Australia. Such blooms are driven by environmental changes, with anthropogenic input of nutrients the major contributing factor, threatening coastal ecosystems as well as being harmful in instances of human contact. While there is a general consensus on factors regarding Lyngbya bloom initiation and growth, a clear process-based formulation of the bloom process is lacking. This article offers a mechanistic description of an ecological model to predict Lyngbya biomass suited to support assessment of management decisions of intertidal coastal ecosystems. The model structure is based on three key state variables of Lyngbya biomass; 1) benthic submerged, 2) exposed and 3) floating. It links these three states with key environmental factors for growth: bioavailability of nutrients (including iron, phosphorus, nitrogen and dissolved organics); light, salinity and temperature; biological and physical processes such as growth, mortality, respiration, sloughing, floating and beach wracking. Whilst there remains limited validation data for this site, the mechanistic nature of the model means it can be used to assessing the relative risk of Lyngbya accumulation for different scenarios. A framework for setting Lyngbya management priorities in Roebuck Bay and as a basis for future research into coastal ecosystem impacts from Lyngbya blooms is described.
5.2 Introduction

Much attention has been paid in recent years to the nutrient enriched coastal waters in tidal embayments, especially those experiencing a rapid increase in the frequency of harmful cyanobacterial blooms (Albert et al. 2005, Paul et al. 2005, Al-Thukair et al. 2007, Estrella 2013). *Lyngbya majuscula* (*Lyngbya* here onwards) is a toxic, filamentous marine cyanobacterium of concern for harmful algal blooms in recent years. *Lyngbya* is found in tropical to subtropical coastal inter-tidal habitats, typically with an increased frequency and severity of bloom. Increased frequency and severity of blooms of *Lyngbya* is often symptomatic of excessive inputs of anthropogenic nutrients as a result of landuse changes in the surrounding catchment (Dennison et al. 1999). Human activities such as dredging, inputs of treated municipal waste, and the discharge of nutrient laden freshwater through coastal canals have all been cited as contributing (Dennison et al. 1999, Paerl et al. 2008, Estrella 2013).

*Lyngbya* can end up as wrack on beaches, creating potential health risks, environmental risks, and are un-aesthetic in appearance, typically requiring removal by the local authorities at considerable cost (O'Neil and Dennison 2005). These blooms also create significant economic risks for coastal communities, impacting commercial and recreational fishing, tourism, human health, cultural values and restrict development activities in the coastal zone (Cardellina et al. 1979, Pearson et al. 2008). There is increasing evidence of an association between the increased distribution and proliferation of *Lyngbya* blooms and land-derived nutrients in stormwater runoff and groundwater (Ahern et al. 2006). Nutrient circulation and dissolved oxygen (DO) in embayments have also been identified, as shallow intertidal zones act as a store for nutrients (Suzuki and Matsukawa 1987). Understanding the dynamics of these bloom forming species is therefore crucial from an ecological and biogeochemical point of view. Scoping and consensus models have been previously used for this with respect to *Lyngbya* blooms in coastal waters (Arquitt and Johnstone 2004). Bayesian network models and random forest algorithms have also been
used to quantify the major factors and their pathways contributing to the
initiation of Lyngbya, for both scientific and management perspectives (Hamilton
*et al.* 2007, Johnson *et al.* 2010, Kehoe *et al.* 2012). Despite the above
advances and our increased understanding of the pathways by which nutrients
are delivered to ecosystems (Wright 2013, Gunaratne *et al.* 2015), the
pathways by which they are assimilated by Lyngbya remain poorly understood,
and the role of physical redistribution of biomass within coastal embayment is
unclear.

It is therefore the aim of this chapter to develop a conceptual framework for
modelling that can support management and identify the high priority research
areas required to quantify the poorly understood mechanistic features related to
Lyngbya biomass, nutrients and its relationship to hydrodynamic properties. We
present a comprehensive mechanistic model based on process data available
in the literature for the key ecological processes involved in the dynamics of
Lyngbya biomass in coastal waters. Besides the physiological
parameterisations, the model introduces expressions for the detachment,
sloughing and wrack dynamics, which are thought to be key but not yet
available within most model platforms. This model has been developed within
the open-source “Aquatic Ecodynamics” (AED) model library that is able to link
to a range of hydrodynamic drivers via the Framework for Aquatic
Biogeochemical Models (FABM; Bruggeman et al., 2014). Whilst data for
validation of the model is limited, we describe application of the proposed model
system for ongoing investigations into Lyngbya dynamics, and for assessing
management scenarios.

5.3 **Drivers of growth in intertidal coastal waters**

Lyngbya blooms in sub-tropical and tropical coastal waters have been linked to
a number of physical and environmental variables such as high benthic light,
warm water temperatures and ample bioavailable nutrients. These act
synergistically to cause a bloom of Lyngbya in coastal waters (Albert et al. 2005, Watkinson et al. 2005). Bioavailable nutrients generally come from the surrounding catchments via surface and groundwater pathways to the bloom sites (Dennison et al. 1999). A notable exception to this theory of nutrient transport from nearby terrestrial catchments is the occurrence of Lyngbya blooms within a system of offshore intertidal areas considered pristine with excellent water quality (for example, some oligotrophic sites in Moreton bay Australia) (Dennison and Abal 1999). In the absence of a strong, readily identifiable mechanism for transport of key Lyngbya nutrients to offshore intertidal areas with no strong surface water transport process to deliver nutrients from the adjacent terrestrial catchments is still a puzzle. Past researches have shown that the causes of toxic blooms of Lyngbya are complex and multifaceted. Tidally driven exchange mechanism (Gibbes et al. 2008) and submarine groundwater discharge (O’Neil and Dennison 2005) have been suggested as potential additional nutrient transport mechanisms (see chapter 4). The findings from Chapter 4 agreed with Kehoe et al. (2012), who suggested that nutrient availability is a necessary but not sufficient precondition for blooms, it is other parameters such as temperature and light that may trigger blooms. Nutrient retention and low dissolved oxygen content also are a potential mechanism and have been previously suggested as a key contributor for nutrient enrichment in shallow intertidal zones. Therefore it is suggested, when certain physical and chemical conditions are met, specific anatomical or metabolic adaptations give Lyngbya a competitive advantage, and allow them to proliferate and create a bloom.

### 5.4 Conceptual basis for Lyngbya model

The ecological processes essential for simulating Lyngbya biomass are summarised in Fig. 5.1. This forms the basis for the numerical model that is created, which includes interactions between nutrients, organic matter and
distinct Lyngbya pools. By embedding within a hydrodynamic platform, the coupled model allows a comprehensive simulation of processes that govern the transport and fate of water quality attributes included suspended sediment, dissolved inorganic nutrients, organic matter (dissolved and particulate), Lyngbya, and relevant fluxes at the air-water and sediment-water interface.

Figure 5.1 Graphical representation of conceptual model of L. Majuscula biomass and nutrient flux pathways

A schematic of the mechanistic model implemented is shown in Fig. 5.2. Lyngbya is found in three growth states in shallow intertidal embayments with semi-diurnal tides. They are 1) benthic submerged (LBS) represents benthic Lyngbya biomass submerged in waters; 2) floating (LF) represents floating Lyngbya biomass in waters and 3) benthic exposed (LBE) represents benthic Lyngbya biomass on mudflats at the absence of tidal waters.
Figure 5.2 Schematic representation of processes involved with Lyngbya biomass simulations.

LBE has not been taken into account when modelling Lyngbya in all previous research work and new algorithms are required to capture the physical processes that govern accumulation and redistribution within the intertidal zone. When LBS has acquired sufficient biomass, masses of filaments become buoyant due to entrapped oxygen bubbles, and float upwards (Beer et al. 1986), becoming LF, that can then be mobile through the embayment.

Lyngbya grows in shallow tropical areas with frequent exposure to diverse environmental stress factors such as desiccation during low tide or exposure to high fluxes of UV light (Jones et al. 2011). Prolonged desiccation leads to damage cells of cyanobacteria and such damage restricts many organisms to aqueous environments, but many cyanobacteria can tolerate the air-dried state for prolonged periods. Lyngbya showed a marked capacity to withstand the removal of their cellular water (Starkenburg et al. 2011). Therefore we assume LBE can grow under the exposed benthic conditions where water has drained away, or more typically, sits in shallow pooling and saturated sediments.
Growth, respiration, mortality, sloughing and beach wrack, upward floating and Lyngbya detritus decay were identified as key ecological processes of the model. Uptake of land based nutrients from water and sediments coming through surface runoff and ground water discharge is also supported, the importance of which was discussed in Chapter 4. Lyngbya detritus consumes dissolved oxygen (DO) when decomposing the associated dissolved and particulate organic material. Sloughing and shear forces disperse floating mats (LF) throughout the bay water. At different growth states, through different ecological processes such as growth, sloughing & beach wrack, respiration, mortality, decay, Lyngbya interact with nutrients (NH4, NO3, Fe, FRP) and organic matter such as dissolved organic carbon (DOC) and particulate organic carbon (POC).

These interactions are driven by, light (I), salinity (S), temperature (T) and hydrodynamic drivers, horizontal velocities (U), vertical velocities (V), and shear stresses (τ) in coastal waters. Actively growing algae release substantial amounts of DOC via photosynthetic extracellular release (Baines and Pace, 1991). This exudation of organic material occurs when algal carbon fixation exceeds synthesis of new cell material during periods of sufficient irradiance (Panzenböck, 2007) and under nutrient-depleted conditions (Berman-Frank and Dubinsky, 1999).

5.5 Parameterisation approach

Three Lyngbya growth state variables (LBS, LF, LBE) are required, with 8 state variables required to model the Lyngbya biomass with dynamic internal nutrient stores of N and P. In addition, four dissolved inorganic nutrients (FRP, NO3, NH4, Fe); three dissolved (DOC, DON, DOP) and three particulate (POC, PON, POP) detrital organic matter groups and dissolved oxygen (DO) make a total of 19 simulateable variables (Table 5.1). All variables are transported and subject to boundary forcing by the hydrodynamic model.
Table 5.1 State variables for Lyngbya biomass model. The variable code and its units are also given.

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<tr>
<th>State Variable</th>
<th>Symbol</th>
<th>Units</th>
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<td></td>
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<tr>
<td>Lyngbya benthic exposed</td>
<td>LBE</td>
<td>mg C m(^{-3})</td>
</tr>
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<td>Lyngbya benthic submerged</td>
<td>LBS</td>
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<td>mmol Fe m(^{-3})</td>
</tr>
<tr>
<td><strong>Environmental dependencies</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Light</td>
<td>I</td>
<td>W m(^{-2})</td>
</tr>
<tr>
<td>Temperature</td>
<td>T</td>
<td>C</td>
</tr>
<tr>
<td>Salinity</td>
<td>S</td>
<td>ppt</td>
</tr>
</tbody>
</table>

5.6 Mass balance

The resulting cyanobacteria process-based biomass model with 14 state variables and environmental dependencies is given below (Eq. (5.1) – (5.6)). All processes and parameters are discussed in details under ‘processes’.
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For benthic exposed Lyngbya biomass:

\[
\frac{\partial (L_{BE})}{\partial t} = R_{growth} \left( \text{Light attenuation + nutrient uptake (P, Fe) + N fixation} \right) - \\
R_{mortality+respiration} + R_{beachwrack} \pm R_{repartitioning} \quad (5.1)
\]

\[
\frac{\partial (L_{BE})}{\partial t} = \\
\mu_{\text{max Lyngbya}} \min[f(1), f(P), f(Fe)] f_{A_{\text{Lyngbya}}}^{T_{1}} (T) \left[ f_{N_{\text{Lyngbya}}} + f(N)_{\text{Lyngbya}} (1 - f_{N_{\text{Lyngbya}}}) \right] - \\
k_{r_{\text{Lyngbya}}}^{T-20} + k_{r_{\text{Lyngbya}}^{*}} \mu_{\text{Lyngbya}} \frac{R_{growth}}{R_{mortality+respiration}} + R_{beachwracking} \pm R_{repartitioning} \quad (5.2)
\]

For benthic submerged Lyngbya biomass:

\[
\frac{\partial (L_{BS})}{\partial t} = R_{growth} \left( \text{Light attenuation + nutrient uptake (P, Fe) + N fixation} \right) - \\
R_{sloughing} - R_{mortality+respiration} - R_{floating} \pm R_{repartitioning} \quad (5.3)
\]

\[
\frac{\partial (L_{BS})}{\partial t} = \\
\mu_{\text{max Lyngbya}} \min[f(1), f(P), f(Fe)] f_{A_{\text{Lyngbya}}}^{T_{1}} (T) \left[ f_{N_{\text{Lyngbya}}} + f(N)_{\text{Lyngbya}} (1 - f_{N_{\text{Lyngbya}}}) \right] - \\
k_{r_{\text{Lyngbya}}}^{T-20} + k_{r_{\text{Lyngbya}}^{*}} \mu_{\text{Lyngbya}} \frac{R_{growth}}{R_{mortality+respiration}}
\]
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\[- SL_{\text{max}} \left( \frac{\omega}{\omega_{\text{max}}} \right) \left( \frac{\mu_{\text{Lyngbya}}}{\mu_{\text{max LF}}} \right) - \frac{C_1 \left[ 1 - e \left( - \frac{1}{k_e} \right) \right] - C_3}{-C_3} \text{ for day time} \pm R_{\text{repartitioning}} \]

\[ R_{\text{floating}} \]

\[ (5.4) \]

For Lyngbya floating biomass:

\[ \frac{\partial (L_F)}{\partial t} = R_{\text{growth}} (\text{Light attenuation + nutrient uptake (P, Fe) + N fixation}) - R_{\text{mortality + respiration}} + R_{\text{sloughing}} + R_{\text{floating}} - R_{\text{beachwrack}} \quad (5.5) \]

\[ \frac{\partial (L_F)}{\partial t} = \frac{\mu_{\text{max Lyngbya}} \min[f(I), f(P), f(Fe)] f^{T_1}_{\text{Lyngbya}} (T) \left[ f_{\text{NF Lyngbya}} + f(N)_{\text{Lyngbya}} (1 - f_{\text{NF Lyngbya}}) \right]}{R_{\text{growth}}} - \frac{k_{r_{\text{Lyngbya}}} T^{-20} + k_{p_{\text{Lyngbya}}} \mu_{\text{Lyngbya}}}{R_{\text{mortality + respiration}}} + \frac{SL_{\text{max}} \left( \frac{\omega}{\omega_{\text{max}}} \right) \left( \frac{\mu_{\text{Lyngbya}}}{\mu_{\text{max LF}}} \right)}{R_{\text{sloughing}}} + C_1 \left[ 1 - e \left( - \frac{1}{k_e} \right) \right] - C_3 \text{ for day time} - R_{\text{beachwrack}} \quad (5.6) \]

\[ R_{\text{floating}} \]

5.7 Processes

5.7.1 Floating

In contrast to eukaryotic phytoplankton species, some bloom-forming cyanobacteria can form gas vesicles, which allow for buoyancy regulation (Walsby et al. 1997). Walsby (1981) states that for Lyngbya, gas vacuoles were
not discussed until 1981, when he suggested it was likely they occur and should be investigated. O’Neil et al (2012) reported that Lyngbya can form dense surface blooms when they episodically detach from their benthic substrates, especially under calm stratified conditions. In turbulent waters with intense wind mixing, floating does not provide much of an advantage as vertical mixing will distribute the cyanobacteria over the entire water column (Paerl and Huisman 2009). However, when temperatures increase and there is little wind mixing, the water column becomes stagnant and buoyant cyanobacteria will float upwards. This leads to the accumulation of dense surface blooms at the water surface as have been observed for freshwater Lyngbya species (Beer et al. 1986).

For floating filamentous Lyngbya in water, the vertical velocity is derived from the Stokes equation for a sphere (Eq. 5.7).

\[ v = 2gr^2(\rho - \rho_w)/9\eta \phi \]  

(Eq. 5.7)

where \( v \) is the floating velocity of Lyngbya, \( g \) is the gravitational acceleration, the size of the Lyngbya filament clump estimated as the radius \( r \) of a sphere of equal volume, the density of the Lyngbya \( \rho \), and the density \( \rho_w \) and viscosity \( \eta \) of the coastal water. A ‘form factor’ \( \phi \) is included to adjust for the non-spherical shape of filamentous Lyngbya as explained by Walsby et al. (2006).

Kromkamp and Walsby (1990) conceived a model to explain buoyancy regulation in a culture of filamentous cyanobacteria (Oscillatoria) which had previously been incubated at an irradiance flux \( I \) of 10 \( \mu \text{mol quanta m}^{-2} \text{s}^{-1} \). We adapted this model for Lyngbya floating process (Eq. 5.8).

\[ \frac{dp_{\text{Lyngbya}}}{dt} = \begin{cases} 
C_1 \left[ 1 - e \left( -\frac{1}{I_k} \right) \right] - C_3 & \text{for day time} \\
-C_3 & \text{for night time} 
\end{cases} \]  

(Eq. 5.8)

where \( \frac{dp_{\text{Lyngbya}}}{dt} \) is the rate of density change (kg m\(^{-3}\) min\(^{-1}\)), \( C_1 \) is the rate coefficient for rate of increase in density with time, \( C_3 \) is the minimum rate of decrease in density with time and \( K_\text{E} \) is the half saturation constant for the density response to irradiance.
5.7.2 Sloughing and beach wrack

Sloughing loss of attached phytoplankton depends on wind-induced orbital wave velocities and biomass density (Canale and Auer 1982). During peak bloom periods winds can transport detached and floating clumps of *Lyngbya* to the shore line where it accumulates as beach wrack (Albert et al. 2005). In Geographe Bay of Western Australia, beach wrack appeared to be removed from beaches and become available for long-shore transport and re-deposition on daily to weekly timescales. This resulted in patchy and temporally variable beach wracks (Oldham et al. 2010). In tropical and subtropical intertidal mangrove forests, where *Lyngbya* is present, it will grow on leaf and root litter, live roots, dense mats on the surrounding sediments and accumulations (5 - 20 cm deep) on a mangrove shoreline (Pittman and Pittman 2005) (Fig. 5.3).

Flindt et al (2007) showed that loss by sloughing requires a much higher current speed than the resuspension threshold for unattached macroalgae, making the attached macroalgae less sensitive to transport losses. They further discovered that substrate (or bed) roughness may enhance algal sloughing and change the transition between algal bed load and suspended load. For some phytoplankton biomass loss is substantial through sloughing. For example, *Cladophora* sp., (filamentous green algae) approximately 80% of the net production of the biomass is lost through sloughing; a significant portion of the biomass sloughed is deposited on the beaches as beach wracks (Higgins et al. 2005).
Here we use a model suggested by Canale and Auer (1982) to quantify sloughed biomass of floating Lyngbya. This model based on surface wind conditions, shear stress, empirically determined maximum sloughing rates and biomass (Eq. 5.9).

\[
R_{\text{Slough}} = SL_{\text{max}} \left( \frac{\omega}{\omega_{\text{max}}} \left( \frac{LF}{L_{\text{max}} \cdot LF} \right) \right)
\]  

(5.9)

where, \( R_{\text{Slough}} \) is specific loss or sloughing rate \( \text{(day}^{-1}) \), \( S \) is shear stress correction factor (dimensionless), \( L_{\text{max}} \) is maximum sloughing rate \( \text{(day}^{-1}) \), \( \omega \) is
daily wind speed (km h$^{-1}$), $\omega_{\text{max}}$ is maximum daily wind speed (km h$^{-1}$), and 
$\mu_{\text{max}}$ is maximum Lyngbya biomass (g DM m$^{-2}$).

5.7.3 Growth

The bloom-forming Lyngbya is a part of the diverse group of phototrophs and capture most of their energy from sunlight, using this energy source to drive nutrient uptake and cellular metabolism that results in cell maintenance and growth. Previous work has focussed on the role of nutrients (nitrogen, phosphorous, iron and environmental parameters (light, salinity, temperature) in the development of Lyngbya blooms (Ahern et al. 2008, Kehoe et al. 2012).

As the elemental composition of phytoplankton does not necessarily conform to the Redfield ratio (Geider and La Roche 2002 ), it is no longer acceptable to assume that knowledge of one element enables the concentration of other elements to be calculated (Baklouti et al. 2006). As a result it has become necessary to develop multi-nutrient models and to explore the possibilities of multi-nutrient limitation and co-limitation (see also Flynn 2005). The nitrogen fixing ability of Lyngbya suggests that elevated levels of nutrients other than nitrogen (N), such as phosphorus (P) and bioavailable iron (Fe), are contributing to the occurrence of the blooms (Bell and Elmetri 2007, Ahern et al. 2008). P and Fe are required by all algae for the synthesis of various cell constituents but diazotrophs, such as Lyngbya, require additional amounts of these nutrients for synthesis and activation of nitrogenise (Brand et al. 1983, Bell and Elmetri 2007).

5.7.4 Light attenuation

Light has an important influence on Lyngbya and it has exhibited a tolerance for elevated irradiances (up to approximately 1000 µmol quanta m$^{-2}$ s$^{-1}$) with an
optimal PAR level of 700 µmol quanta m\(^{-2}\) s\(^{-1}\) for primary production quantified in an ecophysiological analysis (Watkinson et al. 2005). The optimal value corresponded approximately to daily surface irradiance (0 to 700 µmol quanta m\(^{-2}\) s\(^{-1}\)) as per Watkinson et al (2005) for a subtropical coastal wetland during Lyngbya blooms.

Numerous different equations have been used in modelling of photosynthesis of phytoplankton (e.g., Baklouti et al (2006). In modelling photosynthesis in Lyngbya, we use the equations for phytoplankton from Webb et al (1974) (Eq. 5.10).

\[
f(I) = 1 - e^{-\frac{I}{I_k}}
\]  \hspace{1cm} (5.10)

where the term \(f(I)\) is defined as the ratio of \(\mu/\mu_{\text{max}}\) where \(\mu\) is the rate of carbon fixation and \(\mu_{\text{max}}\) is the maximum potential rate of carbon fixation, \(I\) is the incoming irradiance and \(I_k\) is the light intensity at which the photosynthetic rate is numerically equivalent to \(\mu_{\text{max}}\) if light saturation behaviour was absent (Talling 1957). \(I_k\) is 300 µmol quanta m\(^{-2}\) s\(^{-1}\) and \(\mu_{\text{max}}\) is 0.33 day\(^{-1}\) at a reference temperature of 25°C (Watkinson et al 2005).

In the model, light is attenuated through the water column, the floating Lyngbya (LF), benthic submerged Lyngbya (LBS), the seagrass and the microphytobenthos sequentially in the intertidal zone. In the absence of tidal waters in the inter-tidal zone, light is incident directly to benthic exposed Lyngbya (LBE). Wind and induced resuspension of sediment particles can have a significant effect on the benthic light attenuation in shallow coastal areas (Walker 1981). Both wind and tidal movement cause shear stress at the water sediment interface causing particles to be resuspended in the water column, particularly in intertidal areas, playing a key role shaping the PAR experienced by LBS (Kehoe et al. 2012).

For primary production, the shortwave (280-2800 nm) intensity at the surface is usually converted to the PAR based on the assumption that ~45% of the
incident spectrum lies between 400-700 nm (Kirk 1994). PAR penetrates into the water column according to the Beer-Lambert Law. PAR at the bottom of a layer of water, \( I_{\text{bot}} \) (mol quanta m\(^{-2}\) s\(^{-1}\)), is approximated by (Eq. 5.11):

\[
I_{\text{bot}} = I_{\text{top}} e^{-K_d dz}
\]

(5.11)

where \( I_{\text{top}} \) is the PAR at the top of the layer (mol quanta m\(^{-2}\) s\(^{-1}\)), \( dz \) the thickness of the layer (m) and \( K_d \) is the light extinction coefficient of the water (m\(^{-1}\)). In the intertidal zone, at the absence of water (\( dz = 0 \)), where \( I_{\text{bot}} = I_{\text{top}} \), surface irradiance directly affects both the LF and LBE state variables of Lyngbya.

The light extinction coefficient can be dynamically adjusted to account for variability in the concentrations of algal, inorganic and detrital particulates, and dissolved organic carbon levels based on user defined specific attenuation coefficients. \( K_d \) is given by the sum of the each attenuating component in the water (Eq. 5.12):

\[
K_d = K_w + K_{eSS} + K_{eDOC} + K_{ePOC} + K_{eLBS}
\]

(5.12)

where \( K_w \) is the background attenuation coefficient of water (m\(^{-1}\)), \( K_{eSS}, K_{eDOC}, K_{ePOC} \) and \( K_{eLBS} \) are the attenuation coefficients other components in the water column, suspended solids, dissolved organic carbon, particulate organic carbon and benthic submerged Lyngbya.

### 5.7.5 Nutrients

Nutrient limitation is frequently considered the cause of reduced growth of phytoplankton in natural environments and an important driver of competition that determines community composition (Oliver et al. 2012). The macronutrients nitrogen and phosphorus, and the micronutrient iron, all play a role in regulating photosynthesis and ultimately the growth of many cyanobacteria including
Lyngbya. Orthophosphate and inorganic forms of nitrogen such as nitrate and ammonia are the preferred sources of macronutrients for uptake by Lyngbya (Ahern et al. 2007). Lyngbya is capable of fixing nitrogen, phosphorus can often become a limiting nutrient controlling its growth, since nitrogen deficiencies can be partially overcome through use of atmospheric nitrogen (Jones 1990). Phosphorus has been shown to stimulate nitrogen fixation rates and filament growth of Lyngbya maintained in culture (Elmetri and Bell 2004). Similarly, phosphate additions have enhanced productivity for natural populations of Lyngbya collected from the field and grown under laboratory conditions (Watkinson et al. 2005). Micronutrients, such as iron can also be potentially limiting factors for N2 fixation (Paerl et al. 1994) and consequently, Lyngbya growth. Iron has been identified as an important limiting nutrient in Lyngbya productivity (Ahern et al. 2006). Given that the concentration of dissolved iron in seawater is generally extremely low iron must be transported from catchments to coastal waters with suspended sediment, and such loads are enhanced by land clearing and during periods of heavy rainfall (Gunnars et al. 2002).

At certain pH values (circa 8.2) of sea water Iron becomes insoluble (Rose 2005). These conditions also reducing phosphorus availability to Lyngbya (Ahern et al. 2006). However, in the presence of dissolved organic carbon (DOC), chelating agents can complex the iron and allow it to persist for much longer in a soluble form in seawater (Rose 2005). Organically chelated iron (FeEDTA) has been shown to be far more effective in enhancing Lyngbya growth and productivity than inorganic iron (Ahern et al. 2006).

5.7.5.1 Phosphorous limitation

It has been suggested that oceanic productivity depends, to a significant extent, on N2 fixing phytoplankton and therefore, the productivity in the oceanic regions as a whole could well be P-limited (Tyrrell 1999). The specific growth rate and N2 fixation rate of Lyngbya increased with P-PO$_4^{3-}$ concentration up to a
maximum value and thereafter remained constant. The growth rate of Lyngbya is extremely sensitive to small increases in concentration of $P-\text{PO}_4^{3-}$ above a threshold value (e.g., threshold value is 0.07 µM for Moreton Bay, Australia) (Elmetri and Bell 2004). This implies a shock load of nutrients with elevated $P$ levels can bring dramatic changes in water quality with harmful Lyngbya blooms (Gunaratne et al. 2015).

Phosphorous limitation of Lyngbya is expressed by using a simple Michaelis-Menten equation adapted from Hipsey et al. (2014) (Eq. 5.13):

$$f(P) = \frac{FRP}{FRP + K_{PLyngbya}} \tag{5.13}$$

where $K_{PLyngbya}$ is the half-saturation constant for the effect of phosphorous on the growth rate.

### 5.7.5.2 Iron limitation

Iron is required by cyanobacteria like Lyngbya in relatively large quantities (Kustka et al. 2002). This is problematic for marine cyanobacteria, where iron is highly insoluble in their habitats (Liu and Millero 2002) and the dissolved fraction is predominantly strongly bound to organic compounds (Witter and Luther III 1998). Under these conditions, marine microalgae have been forced to develop specialized mechanisms for iron acquisition; 1) formation of organic complexes and 2) reduction of ferric iron to the more soluble ferrous state (Rose et al. 2005). The former by release of cellular products that form strong complexes with iron specifically to aid iron uptake (siderophores) and in the latter by reduction of iron from Fe(III) to Fe(II) using either surface bound iron reductases (Maldonado and Price 2001) or by release of iron-reducing compounds (so-called electron shuttles) into the surrounding milieu (Hernandez and Newman 2001). Therefore, concentration of dissolved organic carbon (DOC)
is a crucial factor affecting iron solubility in the waters rather than the actual source of the organics (Albert et al. 2005).

Few ecosystem models deal with potential for micronutrient limitation or explicitly include micronutrients as state variables, but limitation by iron or other trace elements such as molybdenum may potentially be alleviated by periods when these micronutrients are present in high concentrations (Donnelly et al. 1997). Research by Gross and Martin (1996) suggests that the relationship between bioavailable iron concentration and Lyngbya growth response follows a Michaelis–Menten pattern. Iron limitation of Lyngbya takes the form (Eq. 5.14);

\[
f(\text{Fe}) = \frac{\text{Fe}(\text{II})}{\text{Fe}(\text{II}) + K_{\text{Fe\_Lyngbya}}} \tag{5.14}
\]

where \( K_{\text{Fe\_Lyngbya}} \) is the is the half-saturation constant for the effect of bioavailable iron on the growth rate.

In some modelling efforts, iron limitation was not taken into account even though availability has demonstrated to limit cyanobacteria growth (Robson et al. 2013). This may be an acceptable approximation, as there is a suggestion that cyanobacteria can take up iron and other minerals quickly, when they are available for short periods and store them for later use (Donnelly et al. 1997, Rose et al. 2005).

### 5.7.5.3 Nitrogen limitation and nitrogen fixation

Nitrogen limitation to primary production and other ecosystem processes is well documented. Cyanobacteria including Lyngbya can uptake a range of nitrogen sources including ammonium, nitrate, nitrite and dissolved organic nitrogen (DON) such as urea and some amino acids (Flores and Herrero 2005). The proportion of DON (present in these directly available forms) in tropical and sub
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tropical coastal waters have been poorly quantified, but is probably small (Robson et al. 2013). In this model, we assumed that DON is not available to Lyngbya in the intertidal environment.

Using constant (e.g. Redfield ratio) or user defined nitrogen concentration is one way of modelling N limitation in a relatively simple manner. Redfield ratio (106C:16N:1P) has been used widely in aquatic studies to determine the nutrient status of systems, link biogeochemical models of these elements and to estimate cellular production (Oliver et al. 2012). In a field experiment, concentrations of biomass carbon, nitrogen, phosphorus and iron of Lyngbya were calculated to be 179C:14N:1P:1.8Fe (Watkinson et al. 2005). We have used this C:N:P:Fe ratio as a constant cellular elemental stoichiometry and model the nitrogen limitation of Lyngbya using a simple Michaelis-Menten equation adapted from Hipsey et al (Hipsey et al. 2014) (Eq. 5.15):

\[
 f(N) = \frac{NH_4^++NO_3^-}{NH_4^++NO_3^-+K_{N_{Lyngbya}}} 
\]  

(5.15)

where \( K_{N_{Lyngbya}} \) is the half-saturation constant for the effect of nitrogen on the growth rate.

To understand N limitation, we must understand the controls of biological N fixation. Lyngbya has been reported to fix nitrogen only during the day (Jones 1990, Dennison et al. 1999), the night (Lundgren et al. 2003, Elmetri and Bell 2004) and both day and night (Jones 1992). This is somewhat contradictory but fixing nitrogen only during the day is an uncommon feature among non-heterocystous cyanobacteria like Lyngbya (Berman-Frank et al. 2001, Lundgren et al. 2001). We agree with the findings of Lundgren et al (2003) and Elmetri and Bell (2004) who showed that Lyngbya fix nitrogen and synthesize nitrogenase enzyme in all cells during the dark phase of a diel cycle which suggests that oxygenic photosynthesis and atmospheric \( N_2 \) fixation are incompatible processes for this species. The order of preference amongst the commonly available inorganic sources is \( NH_4^+ > NO_3^- > N_2 \) (Flores and Herrero
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2005). The nitrogen fixation function is flowed by limitation function as the former is only used as an indicator of the amount of dissolved inorganic nitrogen available.

No appropriate numerical approximations of N fixation by freshwater and estuarine cyanobacteria have been developed (LEVINE and LEWIS 1987, Howarth et al. 1999, Oliver and Ganf 2002). We use a more general model which has been widely used in coastal phytoplankton modelling (Zouiten et al. 2013, Machado and Imberger 2014). This has been adapted for coastal ecosystems, by Hipsey et al. (2014) who considered, nitrogen fixation is optimal when the value of the nitrogen limitation function, f(N), is zero, and it is minimum when f(N) is 1. The amount of nitrogen fixed by Lyngbya can be expressed as (Eq. 5.16):

\[
U_{N_2}(A_{Lyngbya}) = k_{NF_{max}}(1 - f(N))(A_{Lyngbya}) \tag{5.16}
\]

where \(A_{Lyngbya}\) is the biomass of Lyngbya in chlorophyll-a units, \(U_{N_2}(A_{Lyngbya})\) (mg N fixed (mg Chl-a) \(^{-1}\) day\(^{-1}\)) is the amount of fixed nitrogen for Lyngbya biomass (mg C day\(^{-1}\)) Lyngbya biomass, \(k_{NF_{max}}\) (mg N fixed (mg Chl-a) \(^{-1}\) day\(^{-1}\)) is the value when no NO\(_3^\) or NH\(_4^+\) is present and when the internal nitrogen stores are depleted. Under this relationship fixation is optimum when no dissolved inorganic nitrogen is present.

The metabolic capacity of the cells is compromised as the rate of nitrogen fixation increases, and the growth rate must therefore be reduced accordingly. The growth equation for Lyngbya is therefore written as (Eq. 5.17)

\[
\mu_{Lyngbya} =
\mu_{max, Lyngbya}\min[f(I), f(P), f(Fe)]T^{1}_{A Lyngbya}(T)\left[f_{NF_{Lyngbya}} + f(N)_{Lyngbya}\left(1 - f_{NF_{Lyngbya}}\right)\right] \tag{5.17}
\]
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where $f_{NF_{Lyngbya}}$ is the fraction of the maximum growth rate that is experienced under maximum nitrogen fixation. For the model, nitrogen uptake amount is dependent only on the $\mu_{Lyngbya}$. $f_{M_{Lyngbya}}(T)$ is a the Arrhenius function of temperature and $\mu_{max_{Lyngbya}}$ is the maximum growth rate (approximately 0.33 day$^{-1}$ at a reference temperature of 25°C) according to the results of Elmetri and Bell (2004)).

5.7.6 Respiration

Lyngbya respires in both day time and night time conditions and respiration rates ranging between 0.03 to 0.98 g C m$^{-2}$ day$^{-1}$ for Lyngbya in Morteon bay, Australia (Quigg et al. 2010). $\mu_{max_{Lyngbya}}$ term in the above nutrient limitation equation includes photorespiration effects inherently and therefore our model doesn’t consider a separate term. For Lyngbya mean specific net photosynthesis rate and mean specific dark respiration rate were estimated as 22.25 mg O$_2$ g(dry)$^{-1}$ h$^{-1}$ and 0.63 mg O$_2$ g(dry)$^{-1}$ h$^{-1}$ respectively (Doty 1971). The ratio between net photosynthesis rate and dark respiration rate was calculated as 35.3 which was higher than for other phytoplankton. It could be expected that a species whose net photosynthesis rate is many times larger than its respiration rate can grow at low light intensities (Buesa 1977). Respiration was modelled as a loss term together with mortality and excretion which are discussed below.

5.7.7 Lyngbya Excretion (Toxin & Metabolite Production)

Cyanobacteria excrete organic compounds, including organic and amino acids, peptides, alkaloids, carbohydrates and lipopolysaccharides (Paerl and Fulton III 2006). Due to the excretion of allelopathic substances by certain harmful algal bloom species, co-occurring phytoplankton species may experience haemolysis.
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(Arzul et al. 1999) resulting in damage to the cytoplasmic membrane and subsequent cell lysis. Some Lyngbya strains produce toxic secondary metabolites (Paul et al. 2001), including aplysiatoxins, debromoaplysiatoxin, and lyngbyatoxin A. These metabolites of Lyngbya, affect human health and ecological health and in particular marine reptiles (Osborne et al. 2001). A limited number of grazers have been observed feeding upon Lyngbya and it was reported that Lyngbya toxins may bioaccumulate through marine food chains (Capper et al. 2005). As per our knowledge, however there is a paucity of information on process based mechanism for Lyngbya metabolic loss through excretion.

5.7.8 Grazing

Crude extracts and isolated secondary metabolites of several benthic marine cyanobacteria such as Lyngbya spp. have been shown to deter grazing by generalist herbivores such as fish, crabs, urchins, and other macroherbivores (Nagle and Paul 1999, Thacker and Paul 2004).

However, specialized mesoherbivores, such as the sea hare Stylocheilus striatus, may preferentially consume Lyngbya (Pennings et al. 1996, Nagle et al. 1998). Secondary metabolites often play an important role in plant defence against herbivory (Stamp 2003) and one of the richest cyanobacterial source of secondary metabolites is the benthic filamentous Lyngbya majuscula (Moore 1981, Sitachitta and Gerwick 1998). Hence we do not explicitly model a grazing population in our biomass model due to unpalatable characteristics of Lyngbya upon key grazers, hence grazing will be volumetrically insignificant.
5.7.9 Mortality

In marine waters, a phytoplankton mortality term has been included in models for some time (Steele 1974, Walsh 1983, Fasham et al. 1990), but not much attention was given to understand the process of natural death as a loss term (Franklin et al. 2006). Phytoplankton cell mortality is a key process, at the ecosystem level, where it affects the balance of energy available to higher trophic levels via detrital food webs and the biogeochemical cycling of carbon. It is also key at the population and community levels, where it is critical to understanding bloom dynamics and species successions (Franklin et al. 2006).

For Lyngbya, mortality is poorly understood. Apart from natural death due to metabolic loss of nutrients, largely grazing and viral infection can cause the loss of Lyngbya biomass. For example, the rapid decline of the Deception Bay Lyngbya majuscula bloom suggests that cyanophages present within Lyngbya filaments and may play a key role in the ecophysiology and bloom dynamics of Lyngbya (Hewson et al. 2001). Metabolic loss of nutrients from mortality and excretion is proportional to the internal nitrogen to chl-a ratio multiplied by the loss rate and the fraction of excretion and mortality that returns to the detrital pool. Loss terms represented in the model include a lumped term for metabolic loss. The metabolic loss term, \( L \), is a lumped parameterization of respiration, natural mortality and excretion (Hipsey et al. 2014), and can be expressed as (Eq. 5.18):

\[
L = k_{rLyngbya} 20^{\text{g}} + k_{rPLyngbya} \mu_{gLyngbya}
\]

where \( k_{rLyngbya} \) is the basal loss rate coefficient, and \( k_{rPLyngbya} \) is the fraction of production which is lost during photosynthesis. Values of \( kr \), as a fraction of production, have been established for many marine and estuarine phytoplankton by Langdon (1993) and many others.
5.7.10 Environmental and physical drivers

5.7.10.1 Salinity

Salinity, does not necessarily act as barrier to cyanobacterial growth and proliferation, in estuarine and coastal ecosystems as evident by the presence of diverse epiphytic and epibenthic N2-fixing cyanobacteria (e.g., Calothrix, Lyngbya, Nostoc, Scytonema) (Potts 1994). Lyngbya has well adapted to a broad salinity tolerance with substantial photosynthetic activity from 9 to 36 ppt (Watkinson et al. 2005). Therefore effect of salinity on growth inhibition of Lyngbya was not considered in our model.

5.7.10.2 Importance of tidal circulation and flushing

Advective and shearing redistribution and vertical mixing of phytoplankton biomass is affected by the circulation and of coastal water bodies as discussed in Chapter 4. Tidal currents in mesotidal and macrotidal waterways (mean tidal range >2m) generally discourage blooms because they mix the water column and reduce the residence time of algae in the photic zone (i.e. surface zone of light penetration) (Monbet 1992). However, tidal range and associated processes such as tidal mixing, current velocity, light penetration, and sediment resuspension influence phytoplankton biomass in some coastal waters. One example is seasonal Lyngbya blooms in Roebuck Bay hypertidal (tidal range > 9m) coastal wetland in Broome, Australia (Pearson et al. 2008, Estrella 2013). In order to control Lyngbya blooms more effectively, hydrodynamics should be coupled with the nutrient dynamics and Lyngbya growth.
5.7.11 Implementation

TUFLOW FV model has been used to simulate the hydrodynamics of coastal embayments and is used here to simulate Roebuck Bay, Broome (see Chapter 4). TUFLOW FV was coupled with the Fortran-based Aquatic Ecodynamic (AED) model (Hipsey et al. 2013) for analysing and predicting the physical forcings that control intertidal zone hydrodynamics and water quality in the water column of the embayment. The ‘aed_lyngbya’ module developed here is a modification of the ecological model used in FABM-AED (Hipsey et al. 2012). This module has been designed to simulate the interactions between nutrients, organic matter and Lyngbya. Suggested parameters have been selected from field survey and published literature on Lyngbya and other cyanobacteria (Table 5.2) in modelling Lyngbya biomass.

A number of compartments of the FABM-AED model are unchanged such as dissolved inorganic nutrients (FRP, NO3, NH4), light attenuation and dissolved and particulate organic carbon (DOC, POC). Although the numerical code has been developed it has not been validated due to data limitations, but testing was undertaken to explore the operation of the above described functions.
Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads

Table 5.2 Summary of selected parameters used in ecological and water quality simulations

<table>
<thead>
<tr>
<th>Code</th>
<th>Description</th>
<th>Value</th>
<th>source</th>
</tr>
</thead>
<tbody>
<tr>
<td>p_name</td>
<td>Name of phytoplankton group</td>
<td></td>
<td>Lyngbya Majuscula</td>
</tr>
<tr>
<td>p_initial</td>
<td>Initial concentration of phytoplankton</td>
<td>2*</td>
<td>Field Survey</td>
</tr>
<tr>
<td>p0</td>
<td>Minimum concentration of phytoplankton (mmol C/m3)</td>
<td>1*</td>
<td>Field Survey</td>
</tr>
<tr>
<td>Ycc</td>
<td>carbon to chlorophyll ratio (mg C/mg chla)</td>
<td>70*</td>
<td>Albert 2005</td>
</tr>
</tbody>
</table>

**Growth parameters**

<table>
<thead>
<tr>
<th>Code</th>
<th>Description</th>
<th>Value</th>
<th>source</th>
</tr>
</thead>
<tbody>
<tr>
<td>R_growth</td>
<td>Phyto max growth rate @20C (/day)</td>
<td>0.33*</td>
<td>Elmetri and Bell 2004</td>
</tr>
<tr>
<td>theta_growth</td>
<td>Arhenius temperature scaling for growth function (-)</td>
<td>1.06</td>
<td>Hipsey et al 2013</td>
</tr>
<tr>
<td>T_std</td>
<td>Standard temperature (deg C)</td>
<td>25</td>
<td>Chung et al 2014</td>
</tr>
<tr>
<td>T_opt</td>
<td>Optimum temperature (deg C)</td>
<td>30</td>
<td>Chung et al 2014</td>
</tr>
<tr>
<td>T_max</td>
<td>Maximum temperature (deg C)</td>
<td>35</td>
<td>Chung et al 2014</td>
</tr>
</tbody>
</table>

**Light parameters**

<table>
<thead>
<tr>
<th>Code</th>
<th>Description</th>
<th>Value</th>
<th>source</th>
</tr>
</thead>
<tbody>
<tr>
<td>I_S</td>
<td>saturating light intensity (microE/m^2/s)</td>
<td>700*</td>
<td>Watkinson et al 2004</td>
</tr>
<tr>
<td>KePHY</td>
<td>Specific attenuation coefficient ((mmol C m^-3^-1)1 m^-1)</td>
<td>0.0051</td>
<td>Hipsey et al 2013</td>
</tr>
</tbody>
</table>

**Respiration parameters**

<table>
<thead>
<tr>
<th>Code</th>
<th>Description</th>
<th>Value</th>
<th>source</th>
</tr>
</thead>
<tbody>
<tr>
<td>f_pr</td>
<td>Fraction of primary production lost to exudation (-)</td>
<td>0.025</td>
<td>Hipsey et al 2013</td>
</tr>
<tr>
<td>R_resp</td>
<td>Phytoplankton respiration/metabolic loss rate @ 20 (degC)</td>
<td>0.02</td>
<td>Hipsey et al 2013</td>
</tr>
<tr>
<td>theta_resp</td>
<td>Arhenius temperature scaling factor for respiration (-)</td>
<td>1.04</td>
<td>Hipsey et al 2013</td>
</tr>
<tr>
<td>k_fres</td>
<td>Fraction of metabolic loss that is true respiration (-)</td>
<td>0.7</td>
<td>Hipsey et al 2013</td>
</tr>
<tr>
<td>k_fdom</td>
<td>Fraction of metabolic loss that is DOM (-)</td>
<td>0.3</td>
<td>Hipsey et al 2013</td>
</tr>
</tbody>
</table>

**Salinity parameters**

<table>
<thead>
<tr>
<th>Code</th>
<th>Description</th>
<th>Value</th>
<th>source</th>
</tr>
</thead>
<tbody>
<tr>
<td>S_bep</td>
<td>Salinity limitation value at maximum salinity S_maxsp (-)</td>
<td>5*</td>
<td>Field Survey</td>
</tr>
<tr>
<td>S_maxsp</td>
<td>Maximum salinity (g/kg)</td>
<td>38*</td>
<td>Field Survey</td>
</tr>
<tr>
<td>S_opt</td>
<td>Optimal salinity (g/kg)</td>
<td>36*</td>
<td>Field Survey</td>
</tr>
</tbody>
</table>

**Nitrogen parameters**

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<table>
<thead>
<tr>
<th>Parameter</th>
<th>Description</th>
<th>Value</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>N_0</td>
<td>Nitrogen concentration below which uptake is 0 (mmol P/m³)</td>
<td>0.17</td>
<td>Hipsey et al. 2013</td>
</tr>
<tr>
<td>K_N</td>
<td>Half-saturation concentration of nitrogen (mmol N/m³)</td>
<td>0.045</td>
<td>Chung et al. 2014</td>
</tr>
<tr>
<td>R_nfix</td>
<td>Nitrogen fixation rate (mmol N/mmol C/day)</td>
<td>0.151</td>
<td>Hipsey et al. 2013</td>
</tr>
</tbody>
</table>

Phosphorus parameters

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Description</th>
<th>Value</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>P_0</td>
<td>Phosphorus concentration below which uptake is 0 (mmol P/m³)</td>
<td>0.1935</td>
<td>Hipsey et al. 2013</td>
</tr>
<tr>
<td>K_P</td>
<td>Half-saturation concentration of phosphorus (mmol P/m³)</td>
<td>0.005</td>
<td>Chung et al. 2014</td>
</tr>
</tbody>
</table>

5.8 Discussion

When compared to previously used probabilistic approaches, the mechanistic approach considers more completely the factors describing Lyngbya accumulation in the intertidal environment. Mechanistic biogeochemical models are usually complex and, when designed to produce quantitative results, are often accessible only to expert modellers (Robson et al. 2008) and relies on the modeller capturing the important processes. The main disadvantage of the mechanistic approach is the primary assumption that all important processes have been captured. For example previously for cyanobacteria sloughing and beackwracking were not explicitly simulated in AED model and that means when simulating, it assumed the effect of sloughing on biomass is negligible. While mechanistic parameters can be constrained by theoretical interpretations, this exercise has not been extensively used may be due to the high level of complexity and computational costs (Robson et al. 2008).

Our model was developed to asses Lyngbya in the tropical and sub-tropical coastal wetlands with different intertidal environments (i.e. microtidal to hypertidal) across the world. Therefore this framework has created a modelling approach that can be integrated with catchment data to use as a decision
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support tool in managing Lyngbya blooms in the intertidal zone of coastal wetlands (Fig. 5.4).

Figure 5.4 Schematic diagram of using the model as a decision support system in Lyngbya management of intertidal coastal environments

Coastal wetland managers and stake holders can use this tool to sustainably manage the coastal wetland by identifying the areas of growth potential and timing of blooms, benefit of nutrient reduction. Previous researches, in managing the Brisbane river and Moreton Bay, Australia (Abal et al. 2001), Chesapeake Bay and Louisiana Coastal ecosystem, USA (Boesch 2006), highlighted the importance of orienting more scientific activity to providing the solutions needed for ecosystem restoration and building bridges crossing scientific and management barriers to more effectively integrate science and management. Through our study we have built this bridge connecting coastal wetland scientists and managers on how to tackle this Lyngbya blooms in intertidal zones. Lyngbya has been observed in tropical to subtropical climates.
Characterising the response of inter-tidal zone ecohydrology, to coastal hydrodynamics and anthropogenic nutrient loads in more than 20 different countries (Osborne et al. 2001). It is expected that the mechanistic model developed herein will be useful for complex ecological models of coastal wetlands with comprehensive data sets.
Chapter 6. General discussion

This study demonstrates how low cost field surveys (with community involvement) and a range of modelling techniques can increase our understanding of sub-tropical catchment hydrology. In this case the focus has been anthropogenic nutrient delivery together with physical forcings that control intertidal zone hydrodynamics and environment variables controlling the growth behaviour *Lyngbya majuscula* in a hypertidal coastal embayment.

In a global to local context, coastal wetlands are under immense pressure due to human activities (Faulkner 2004, Lee et al. 2006, Gunaratne et al. 2010), this study provide insights into ungauged catchment hydrology, nutrient delivery, intertidal hydrodynamics and ecological niche of Lyngbya in a sub-tropical climate. Intertidal systems are a key interface of the ocean, atmosphere, and terrestrial environments, and as such, are characterized by frequent fluctuations in temperature, ion concentration, desiccation, UV-irradiation, and wave action dominated by the tidal flooding and ebbing of water with periods of aerial exposure (Decho 2000). The relative frequency of these fluctuations causes both physical and biochemical challenges to organisms which inhabit this environment. The characteristics and intensities of such fluctuations may vary substantially (Giller et al. 1994). Coastal zones experience a variety of pressures (i.e. anthropogenic nutrients and climate change) and the consequences of these pressures often lead changes in the ecology of intertidal zone (i.e. algal blooms, sea level rise etc) (Raffaelli 1996).

In intertidal systems, experimental approaches have shown that simple models are often unable to predict patterns in species distributions and abundance due to the complex nature of physical and biological interactions across latitudinal gradients (e.g. (Helmuth et al. 2002, Hawkins et al. 2008).

Prediction of the three-dimensional distribution of algae growth in the intertidal zone is challenging due to the complex controls (Béchet et al. 2013). Mechanistic biogeochemical models are typically complex and, when designed
to produce quantitative results, are only accessible by expert modellers (Robson et al. 2008). Only a few modelling studies, the Fitzroy Estuary, Keppel Bay (Herzfeld et al. 2001, Robson et al. 2006) and Swan-Canning estuary (Hamilton and Turner 2001), are focused on biogeochemical dynamics of coastal environment including the intertidal zone. Even in these studies, the intertidal zone was not specifically and comprehensively discussed. This “forgotten margin” is one of the most complex areas to understand as a link between marine and terrestrial hydrology and is typically highly biodiverse. Very little data is collected here and very few studies are focused on it. In this regard our research work has significantly contributed to understanding of biogeochemical interactions and physical processes in the intertidal zone. Specifically, a coupled process based model (Chapter 4) was used, which captures the complex dynamics of the environment as well as biogeochemical interactions with Lyngbya in the intertidal zone.

6.1 Urbanisation pressure

Rapid urbanisation is occurring worldwide (Brabec 2002) and (Wu and Thompson 2013) including relatively remote areas like the focus area for this study. Broome supports substantial, developing mining, tourism and hydrocarbon industries in northern Australia. A particularly poignant example is the building of a gas hub 50 km north of Broome, at James Price Point, as part of the development of liquefied gas resources from Broome-Browse basin. This will require local accommodation for 100’s to 1000’s of additional workers, putting pressure on Broome to expand. The rate of population growth and urbanisation in Broome will increase wastewater disposal requirements and stormwater runoff, increasing the risk of Lyngbya blooms, accelerating the already substantial impacts on Roebuck Bay and its sensitive ecosystem. This
justifies the urgent need to assess Lyngbya management as discussed in previous chapters.

6.2 Participatory research approach in a remote community

A critical component of the novel, cost effective temporary gauging strategy used in Roebuck Bay catchment was the participatory research approach, involving the Broome community. This was achieved through a collaboration with local indigenous community rangers and volunteers from community groups in a field campaign that included intensive stormwater sampling, surveying and levelling of drains, mapping, frequent public communication and workshops (i.e. Science on Broome Coast Program, 2014). This has been invaluable to not only collect relatively low cost, high quality data but to engage with stakeholders encouraging them to get involved in natural resource management. The engagement and participation of the Broome community precipitated a learning opportunity for both the researchers and the local community. It has been a critical component of our improved understanding of hydrological processes inherent in the Roebuck Bay catchment. This also resulted in a unique experience for both researchers and the Broome community. This approach is a key in instilling a sense of ownership and management of the Roebuck Bay ecosystem to the local community.

To the best of our knowledge this is the first community based, participatory research project undertaken in Australian intertidal catchment hydrology. Success stories in participation of local communities in water resources management have been reported in the African region (Nare et al. 2006, Kongo et al. 2010). Our work is also in line with the recommendations of International Association of Hydrological Sciences (IAHS), in the formulation and implementation of appropriate science programmes, to achieve major advances in predictions in ungauged basins (Sivapalan et al. 2003).
6.3 Significance of stormwater and shock loading

Seasonal first flush is an important phenomenon in tropical and sub-tropical regions with distinct wet and dry seasons including Broome and many parts of North-West, North and North-East Australia (Deletic 1997, Goonetilleke et al. 2005). It is also an important driver of nutrient export and algal blooms worldwide (Ebise 1991, Lee and Bang 2000, Huang et al. 2007, Luo et al. 2009, Bertrand-krajewski et al. 1998, Gnecco et al. 2005, Lee et al. 2004, and Schiff and Tiefenthaler 2011). Nutrient export associated with seasonal first flush could bring a dramatic change to the receiving water bodies chemistry with a ‘shock loading’ over a short period of time as in Roebuck Bay (Gunaratne et al., 2015). As a consequence of shock loading every wet season, elevated nutrient levels are occurring in Roebuck Bay. These nutrients are derived from 11 sub-catchments in Roebuck Bay primarily from the older parts of Broome with storm water systems are not designed to restrict nutrient discharge. Recently developed parts of Broome have better designed storm water systems which utilise water sensitive urban design principles to restrict nutrient discharge (Gunaratne et al. 2015).

Historical loads of nutrients were also estimated using Parsimonious deterministic models. As anthropogenic nutrients from Broome circulate in the bay, they may exist in the water column for prolonged periods or may be sequestered in sediments (or biomass) at various distances from the point of origin. Consequentially it is not only the short term impact of shock loading that is causing algal blooms but the longer term accumulation of nutrients in the wider ecosystem controlled by intertidal hydrodynamics.

The timing of the shock loading may be critical in generation of Lyngbya blooms. If conditions for rapid Lyngbya growth occur during the first flush of nutrients there would be a period of approximately one month where marine and sediment pore water concentrations would be elevated due stormwater inflows alone. In 2011-12 in Roebuck Bay this (along with groundwater borne nutrients, see below) resulted in Lyngbya blooms at site TR8 in January and
February with blooms occurring at site TR5.1 in February (Gunaratne et al., 2015).

Coastal ecosystems are typically highly sensitive to changes in the environment, but are also typically resilient (i.e. able to recover after stress and shock) (Turner 2000). However if changes persist year after year, as is the case in Roebuck Bay, this has been shown to reduce the ‘resilience’ to respond to the stress and shock of environmental impacts such as algal blooms. This reduction in coastal resilience, is associated with a loss of nursery areas for fisheries, biodiversity, coastal defence functions, carbon reservoirs, and buffers to regulate nutrient fluxes worldwide (Folke et al. 1996). The presence of persistent algal blooms in Roebuck Bay should be a warning for local natural resource managers.

6.4 Significance of submarine groundwater discharge

Contaminants from coastal cities are discharged into aquifers where they may impact estuaries, rivers, beaches and coastal waters. As a result, hotspots of water pollution have been observed related to most coastal cities, releasing industrial wastes, rubbish through wastewater systems, sediments, heavy metals and nutrients (Zann 1996). Intertidal and marine nutrient input via submarine groundwater discharge (SGD) may play a significant role in nutrient cycling and primary productivity in the coastal ocean. SGD varies from neap to spring tide with semi-monthly period and also changes semi-diurnal periods due to changes in hydraulic gradient between the groundwater and seawater in the coastal zone (Taniguchi 2002, Xin et al. 2010).

In Broome the main contaminants identifies in previous studies (Vogwill 2003 and (Hearn 2014) are nutrients, the source identified as the Waste Water Treatment Plant (WWTP) and disposal sites. The disposal sites include the golf course and public open spaces. In Roebuck Bay we can expect increased SGD during tidal exposure which would reduce drying of the sediment due to the presence of a thin layer of water which persists over large areas during low tide
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(Hearn 2014). This groundwater discharge would also elevate nutrient levels in pore water in sites in the intertidal zone as was observed in (Estrella 2013).
Site TR8 has historically been the site of the worst Lyngbya Blooms, as was the case in the 2011-12 wet season. TR8 is immediately down gradient from the WWTP and its most active disposal site (the Broome golf course) (Hearn 2014). Thus blooms at TR8, which occurred in December 2011-12 prior to the first rainfall, suggest that the site had sufficient nutrients available, from previous years and/or groundwater discharge, for Lyngbya blooms to occur. Our hydrodynamic modelling supports this and also suggests nutrients from groundwater discharge, are contributing to blooms along most of the north shore of Roebuck Bay as well (TR5.1).

6.5 The ecological niche of Lyngbya and bloom formation
Lyngbya has been found worldwide in the intertidal zone of estuarine and coastal waters in both tropical and subtropical regions (Izumi and Moore 1987, Albert et al. 2005). Lyngbya blooms in sub-tropical and tropical coastal waters have been linked to a number of physical and environmental variables which act synergistically to cause a bloom of Lyngbya in coastal waters (Albert et al. 2005, Watkinson et al. 2005). Our findings agree with this.
In Roebuck Bay's intertidal zone, Lyngbya was first reported 2000 and every wet season since 2005. At TR8, the main bloom site, nutrient availability is not a sufficient precondition for a bloom to form. Abiotic factors are the key in initiating an actual Lyngbya bloom. The perennial availability of groundwater discharge related nutrients means that blooms at TR8 are inevitable once abiotic conditions are within Lyngbya’s preferred ecophysiological range, which would typically occur on an annual basis. It is only the magnitude of the bloom that is in question and this will relate to: magnitude of groundwater borne nutrient discharge, timing and magnitude of the stormwater first flush, climate and embayment hydrodynamics.
In contrast at site TR5.1 the arrival of transported nutrients, from storm and groundwater inflow, triggers Lyngbya blooms, hence why they occur later than at TR8. Nutrients are not likely perennially available in sufficient quantities, hence must be transported by embayment hydrodynamics. Once a bloom is established the nutrient supply may indeed become limiting, affecting the carrying capacity and duration of the bloom. This is in line with other studies which have identified blooms within a system of offshore intertidal areas considered relatively pristine (Moreton bay Australia, (Dennison and Abal 1999). The simulation of stormwater and groundwater tracers has demonstrated nutrient transport via intertidal zone hydrodynamics. Lyngbya blooming sites in the intertidal zone can occur many kilometres away from the nutrient discharge locations if abiotic conditions and hydrodynamics are suitable.

Benthic Lyngbya growth in exposed tidal areas has never been investigated before based on what is available in the public domain. Submerged growth has been more widely discussed (Kehoe et al. 2012). According to field observations in Roebuck Bay and other study sites such as Moreton Bay, sloughing and beach wracking may also result in substantial biomass redistribution into benthic exposed habitat. Therefore the mechanistic model framework presented herein includes growth in exposed conditions, beach wracking and sloughing.

Beach wracks during bloom periods have become problematic for Lyngbya impacted local communities through lost tourism and the cost of beach clean up (Albert et al. 2005). Nutrient stress of Lyngbya has been linked to the process of detachment and sloughing at the end of blooms (Kehoe et al. 2012). Rapid mortality due to viruses has also been identified as contributing to Lyngbya detachment and mortality (Hewson et al. 2001).

In summary the ecological niche of Lyngbya requires sufficient nutrient availability, suitable intertidal zone hydrodynamics and abiotic factors such as PAR, temperature, salinity to allow Lyngbya blooms to occur. The effect of Intertidal hydrodynamics on Lyngbya blooms has never been comprehensively studied previously. Therefore in predicting Lyngbya biomass it is important to
fully understand its ecological niche and use models which incorporate this understanding in the context of hydrodynamic drivers.

6.6 Numerical Modelling

In previous coupled hydrodynamic-biogeochemical models, the growth state in the tidal exposed area has never been comprehensively studied (Pregnall and Rudy 1985, Beukema 1991, Everett 1991) so in this alone this study represents a major step forward. We use a process based mechanistic model to predict biomass of Lyngbya with three state variables namely 1) benthic submerged, represents benthic Lyngbya biomass submerged in waters; 2) floating, represents floating Lyngbya biomass in the water column; 3) benthic exposed. Other than the main biological processes including growth, mortality and respiration our model includes sloughing and beach wracking to improve simulations of biomass in all three growth states. When compared to previously used probabilistic approaches, the mechanistic approach we used here considers more completely the factors describing Lyngbya accumulation in the intertidal environment. While mechanistic parameters can be constrained by theoretical interpretations, this exercise has not been extensively used may be due to the high level of complexity and computational costs (Robson et al. 2008).

Being benthic cyanobacteria, Lyngbya prefers to stay attached to seagrass or sediment (Estrella 2013) but typically detach when stressed (Bell and Elmetri, 2007). Once detached, sloughing and beach wracking are key process in the distribution of Lyngbya and its intra and inter-annual growth behaviour in the intertidal zone. Once extensive Lyngbya distribution and biomass data are available for calibration and validation our model can be used to predict intertidal biomass of Lyngbya.

This model was developed to asses Lyngbya in the tropical and sub-tropical coastal wetlands with different intertidal environments (i.e. microtidal to hypertidal) across the world. Therefore this framework has created a modelling
approach that can be integrated with catchment data to use as a decision support tool in managing Lyngbya blooms in the intertidal zone of coastal wetlands. It is expected that the mechanistic model developed herein will be useful for complex ecological models of coastal wetlands with comprehensive data sets.

### 6.7 Future Directions and Challenges - Roebuck Bay Lyngbya Assessment and Management

There may be other possible influences on Lyngbya bloom formation in Roebuck Bay that were not included in this study but these are likely subordinate to those described herein. For example the presence of large numbers of migratory wader birds during the wet season could introduce extra nutrients via faeces. The correlation between the location of groundwater and storm water inflow, the subsequent areas with long nutrient residence times and sites of the worst algal blooms suggests anthropogenic nutrients are the key driver of Lyngbya Blooms. If migratory wader faeces were an important contribution then the location of the worst blooms would be very different. The Town to Port Beach portion of the intertidal zone is not frequented by large numbers of wader birds, in fact wader numbers there have dropped in response to the presence of Lyngbya and the subsequent impacts to benthos (Estrella 2013).

Given the significant of the site ecologically, socially and economically more research is required to understand threats and their mitigation. Future monitoring strategies need to combine measurement of biological parameters (i.e. chlorophyll-a, Lyngbya benthic biomass, water column biomass, benthos and waders) with the abiotic parameters previously discussed. More stormwater, groundwater and embayment water quantity and quality data are also urgently required.

As the predominantly benthic Lyngbya was not explicitly modelled in our numerical model, only bay hydrodynamics and geochemistry were, it is
important that future work models Lyngbya biomass explicitly. No modelling framework was previously available to undertake this. We have clearly shown that the abiotic environment in Roebuck Bay is favourable for Lyngbya growth and bloom formation, but an explicit model is required to assess management in detail. Models need to include: Lyngbya in water column, benthic Lyngbya under both submerged and exposed conditions; Lyngbya sloughing and beach wracking; and hydrodynamics including environmental variables. Only through this holistic approach will we be able to get a sufficient understanding of Lyngbya to facilitate management. We must go beyond traditional algal biomass modelling which is limited to water column phytoplankton. If our model was extended to model Lyngbya holistically throughout its life cycle, this will be useful not only in Roebuck Bay but in other benthic phytoplankton- intertidal scenarios around the world.

### 6.8 Future Directions and Challenges - Global Lyngbya Assessment and Management

Managing receiving-water quality, ecosystem health and ecosystem service delivery is challenging in regions where extreme rainfall and runoff events occur episodically (i.e. tropical and sub-tropical coastal ecosystems in Australia), often intensifying land-degradation impacts (Leigh et al. 2013). As Lyngbya blooms have been reported in many tropical and subtropical coastal ecosystems over recent years (Paerl and Otten 2013) the main area of concern globally in Lyngbya management is tropical and sub-tropical coastal environments. In areas with this climatic regime, water flows (i.e. including extreme events such as tropical cyclones), mobilise large quantities of sediment, organic matter, nutrients and pollutants from upstream sources to adjacent wetlands and coastal waters within very short time periods (see Chapter 2). These events may occur infrequently or unpredictably, which can lead to uncertainty about the extent and frequency of monitoring required to encapsulate the full range of event characteristics and associated ecological responses. Further, their effects
on receiving waters are often superimposed on seasonal or inter-annual changes in the receiving-water bodies water quality and ecology. These impacts require management consideration that must be underpinned by detailed scientific understanding (Watson et al. 1996) preferably based on robust numerical tools. Our study has advanced these issues considerably.

Nutrient loading dynamics have changed substantially over the past several decades and practices such as bans on detergents containing phosphate and improved waste water treatment have been effective at reducing freshwater phosphorous loads, but less so for nitrogen (Howarth 2008). The emphasis on phosphorous reductions to control Lyngbya is based on knowledge that it can satisfy its metabolic requirements via nitrogen fixation (Elmetri and Bell 2004). However, at the ecosystem-level, for harmful cyanobacteria, only a fraction, usually far less than 50%, of primary production demands are met by nitrogen fixation, even when phosphorus supplies are sufficient (Howarth et al. 1988) hence nitrogen inflow from external source is required for blooms to form. Nitrogen and phosphorous co-limitation appears to be an important limitation on Lyngbya bloom formation in coastal environments (i.e. Roebuck Bay, Moreton Bay).

In addition to nutrient over-enrichment, climate change, in particular global warming is likely to favour CyanoHABs (Cyanobacterial harmful algal blooms). Lyngbya exhibits optimal growth rates at relatively high temperatures, usually in excess of 28 °C. Intense light absorption by dense surface CyanoHABs can locally increase water temperatures thereby creating a positive feedback that perpetuates further Lyngbya dominance (Hense and Burchard 2010). In addition to parallel reductions in N and P inputs, water quality managers are in dire need of effective tools, such as ours, to understand how to break the synergy between nutrient loading and hydrologic regimes made more favourable for CyanoHABs by climate change (Paerl and Paul 2012). While there is a need for remedial action in the catchments and riparian zones to improve receiving-water quality of degraded wetlands, there is an equally important need to increase the recognition of the value of receiving-waters as functioning ecosystems that support biodiverse communities, providing
substantial ecosystem services. Robust and defensible science and monitoring programs, effective communication of key findings and their implications are all essential elements for success. Also environmental models, to use as decision support, are critical both environmentally and economically. However, these alone cannot guarantee that preferred management actions will be achieved. This also requires the involvement of government agencies, as well as in the scientific and broader community, to provide leadership and a vision for change.
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APPENDIX A

Flow rate calculation through Manning’s Equation

Velocity of flow in open drain channels computed through manning’s equation and it is an empirical equation that applies to uniform flow in open channels and is a function of the channel velocity, flow area and channel slope (Arcement and Schneider 1989). Uniform flow was assumed (Streeter et al. 1987) for all Broome drain channels. Manning’s equation can be expresses as follows;

\[ V = \frac{k R^{2/3} S^{1/2}}{n} \]  

Eq. (A1)

where, \( V \) (m\(^{-1}\) s) is the mean velocity of flow, \( k \) (m\(^{1/3}\) s\(^{-1}\)) is a conversion factor, it is 1 for SI units. \( n \) (dimensionless) is the Manning coefficient which is a function of the culvert material. Values used for drain channels in each Broome sub-catchment for \( n \) compiled from previous studies (Table A1).

\( R \) (m) is the hydraulic radius which is defined as the ratio of flow cross sectional area and wetted perimeter. \( S \) (dimensionless) is slope of energy grade line; for uniform flows this equals the channel bottom or water surface which can be computed through the vertical distance divided by horizontal distance.

If velocity is known, the discharge \( Q \) (m\(^3\) s\(^{-1}\)) can then be computed as

\[ Q = VA \]  

Eq. (A2)

The results of Manning’s formula, an indirect computation of stream flow, have applications in flood-plain management, in flood insurance (Arcement and Schneider 1989). More error can be expected in estimating the average velocity by assuming a Manning’s \( n \), than by direct sampling (i.e. with a current flow meter), or measuring it across weirs. Therefore we used measured flow rates using “stick and clock” method to calibrate the Manning’s formula, even though the former had its own uncertainty.

VN sub-catchment had a V-Notch weir at the gauging station. Flow rate at V_Notch weir was calculated using V_Notch weir equation (Standards Australia 1991).
The notch angle of the weir was 53.02° and the coefficient of discharge was used as 0.578 ($C = 0.58$ typically used for a 90° V-notch weir).

**Table A1** The Manning’s $n$ values for Broome drains in each sub-catchment. These values were compiled from literature (French 1986, Streeter *et al.* 1987, FHWA 2011) about Manning’s $n$. Values were carefully selected following the guide explained by Arcement and Schneider (1989).

<table>
<thead>
<tr>
<th>Sub-catchment</th>
<th>Manning’s n</th>
<th>Nature of the channel</th>
</tr>
</thead>
<tbody>
<tr>
<td>APT</td>
<td>0.037</td>
<td>Clean, winding, some pools and shoals</td>
</tr>
<tr>
<td>DMC</td>
<td>0.013</td>
<td>Unfinished Concrete</td>
</tr>
<tr>
<td>HGS</td>
<td>0.013</td>
<td>Finished Concrete</td>
</tr>
<tr>
<td>MBB</td>
<td>0.030</td>
<td>Clean, straight, full stage, no rifts or deep pools, with more stones and weeds</td>
</tr>
<tr>
<td>MTS</td>
<td>0.028</td>
<td>Clean, straight, full stage, no rifts or deep pools</td>
</tr>
<tr>
<td>GB</td>
<td>0.048</td>
<td>Clean, winding, some pools and shoals with some weeds and more stones</td>
</tr>
<tr>
<td>NTE</td>
<td>0.025</td>
<td>Clean, straight, full stage, no rifts or deep pools</td>
</tr>
<tr>
<td>RBE</td>
<td>0.040</td>
<td>Clean, winding, some pools and shoals with some weeds and stones</td>
</tr>
<tr>
<td>SVS</td>
<td>0.025</td>
<td>Clean, straight, full stage, no rifts or deep pools</td>
</tr>
<tr>
<td>TB</td>
<td>0.048</td>
<td>Clean, winding, some pools and shoals with some weeds and more stones</td>
</tr>
</tbody>
</table>