Marine and Coastal

National Environmental Science Program

RESEARCH REPORT
Project 3.4

BETTER MANAGEMENT OF CATCHMENT RUNOFF TO MARINE RECEIVING ENVIRONMENTS IN NORTHERN AUSTRALIA

Research Report

M.A. Burford, N. Waltham, D. Irvine, P. Cartwright, A. Brooks, S.J. Faggotter

DECEMBER 2024

Preferred citation

Burford, M.A., Waltham, N., Irvine, D., Cartwright, P., Brooks, A., Faggotter, S.J. (2024) Better Management of Catchment Runoff to Marine Receiving Environments in Northern Australia. Report to the Reef and Rainforest Research Centre, Cairns, Queensland.

ISBN 978-1-922640-25-3

Acknowledgement

This project is supported with funding from the Australian Government under the National Environmental Science Program. The Marine and Coastal Hub is funded by the Australian Government under the National Environmental Science Program.

Cover images

Front: Norman River by Michele Burford.

Back: Gulf flood plume, satellite image by Paula Cartwright.

This report is available for download on the NESP Marine and Coastal Hub website:

www.nespmarinecoastal.edu.au

Contents

List of figures

Acknowledgement

The Marine and Coastal Hub acknowledges Aboriginal and Torres Strait Islander people as the first peoples and Traditional Owners and custodians of the land and waterways on which we live and work. We honour and pay our respects to Elders past, present and emerging.

Aboriginal and Torres Strait Islander peoples represent the world's oldest living culture. We celebrate and respect this continuing culture and strive to empower Aboriginal and Torres Strait Islander peoples.

Executive summary

There are many catchments in northern Australia where increased catchment development is proposed. This is largely in the form of irrigation development but also increased cattle stocking rates. Given the relatively low levels of such development in many catchments to date, there is a strong desire to maintain the integrity of coastal and marine receiving environments after the implementation of future developments. The baseline understanding of water quality in receiving marine environments and in the contributing catchments is very limited across much of northern Australia, making management and other development decisions very challenging. However, there are examples of intensive grazing and irrigation developments in northern Australia, e.g. in the Lower Burdekin Delta, adjacent to the Great Barrier Reef coastline, where lessons can be learnt to fast-track understandings and management and set testable hypotheses about the potential impacts of development in other northern catchments. Additionally, studies have been done on the potential impacts of water development on estuaries, coastal floodplains and the coast in other areas of northern Australia, e.g. Gulf of Carpentaria. This project aims to take advantage of these existing examples to improve the quality of decision-making around the impact of terrestrial runoff on the marine environment, providing a template for decision-makers.

Specifically, this project aimed to:

1. In a literature review, summarise what is known about the impacts of terrestrial runoff on the productivity and health of marine environments in northern Australia and examine the relevance for four catchments with proposed development, i.e., Gilbert and Flinders (Qld), Daly (NT) and Keep (WA) rivers.
2. Using current and historical satellite imagery over the study catchments with flow hydrographs, define the distribution of freshwater river plumes for sediment and nutrients, and their relationship to river flow to examine future plume extent under future development and climate scenarios.
3. Examine changes in mangrove distribution using change analysis modelling to determine greenness of mangrove forests in study locations and relationships with catchment hydrology.
4. Test hypotheses developed for the Flinders and Gilbert systems on other river systems earmarked for further water development to determine the critical nature of nutrient inputs from catchments in fuelling estuarine and coastal productivity, and potentially assess groundwater contributions to estuarine flow using isotopic measures.
5. Share information with key stakeholders to disseminate knowledge from these studies and propose methods for future modelling, monitoring and research to fill in knowledge gaps.

Aim 1 – Literature review

The literature review examined the following key development pressures likely in northern Australia:

a. Non-irrigated agriculture (cattle grazing)
b. Irrigated agriculture
c. Mining
d. Urban development

Additionally, the review incorporates the impact of other stressors, such as climate change and environmental degradation due to feral animals and weeds.

The review focused on four catchments in the wet-dry tropics where there has been relatively little development to date, but pressures are increasing: the Flinders and Gilbert Rivers in the Gulf of Carpentaria region, the Daly River in the NT, and the Keep River on the NT/WA border.

Many estuaries, coastal floodplains and coasts across northern Australia are already impacted by erosion due to broadscale low-intensity cattle grazing, with some catchments, particularly on the east coast, also being impacted by irrigated agriculture. Studies in the GBR have shown that widescale livestock grazing can significantly increase erosion processes in catchments, resulting in an increase in sediment and particulate nutrient loads to waterways. The impacts on habitats and species are wide-ranging, but a key indicator of an ecosystem impact is the smothering of seagrass beds by sediment, resulting in seagrass loss, and thus, loss of sites for feeding and refuge for fish, crustaceans and other species. This impacts biodiversity and is a major risk to endangered species using these habitats.

The proposed development activity likely to have the greatest additional impact on estuaries, coastal floodplains and the coast across northern Australia is expansion of water development for irrigated agriculture. Other activities, such as mining and urban centres, may also have significant effects, but it is unlikely to be at the same scale as irrigated agriculture. Climate change will undoubtedly also have major compounding impacts. This includes increased temperature and evaporation, as well as sea level rise. Weather patterns are also likely to change with a potential increase in extreme events, such as megadroughts, intense rainfall and more frequent cyclones. Northern Australia already has some of the most hydrological extreme river systems globally, and therefore the estuarine, coastal floodplains and coastal ecosystems may be highly vulnerable to an increase in the frequency and severity of extreme events. There is limited information on where and when the greatest risks from climate change are likely to be, which makes it difficult to factor these changes into decisions about development.

The greatest challenge to the Flinders, Gilbert, Daly and Keep River estuaries, coastal floodplains and nearshore is likely to be the combined effects of erosion effects, water development for irrigation, and climate change impacts. Although it is difficult to predict the final outcome of this combination of stressors, there are likely to be synergistic effects, meaning that a precautionary and strategic approach to catchment development is warranted. This includes the need to remediate the impacts of current animal grazing on erosion processes. Significant knowledge gaps remain in terms of thresholds of ecosystem change in response to catchment stressors, synergistic effects of stressors, and viable remediation actions for existing ecosystem impacts.

Aim 2 – Flood plume mapping

The flood plume mapping study focussed on the Flinders, Gilbert and Daly River catchments, and investigated the relationship between wet season river flows, flood plume extent, and primary productivity (measured as chlorophyll-*a)* in adjacent coastal seas. Hydrological data from 2003 to 2023 was analysed for the Flinders, Gilbert, and Daly Rivers to determine peak flow events and their corresponding flood plume sizes using MODIS satellite imagery. The study found that flood plumes were highly variable across the 20-year period, with significant events recorded in 2019 and 2023 and strong relationships between 7-day river flows and plume extents for all rivers.

Primary productivity, measured as chlorophyll-*a* concentration, was significantly associated with plume sizes in the southern Gulf of Carpentaria and Anson Bay, specifically for tertiary plumes from the Flinders, Gilbert, and Daly Rivers. Future climate projections for northern Australia are highly uncertain but there are indications of potential reductions in rainfall by 2070-2099. This could lead to decreases in flood plume extent and associated primary productivity, with implications for higher trophic levels.

This research highlights the critical connection between river flows, coastal flood plumes, and marine productivity in northern Australia. The findings underscore the importance of maintaining environmental water flows to sustain coastal ecosystems and fisheries, particularly in the context of increasing water allocation pressures and the potential impacts of climate change on regional rainfall patterns.

Aim 3 – Mangrove mapping

To identify and investigate potential environmental drivers, such as river flow and rainfall, on the growth and canopy composition of mangrove forests along the Eastern coast of the Gulf of Carpentaria (GoC), we used a multidecadal mangrove dynamics dataset developed by Digital Earth Australia (DEA). Here, we compared mangrove canopy density to river flow to assess whether the two were correlated and to assess whether decreases in base flow that may occur under an increased irrigation extraction scenario are likely to impact mangroves in the GoC. We also arranged the data by 'wetness of year' for each of the study catchments to identify whether there was a clear pattern of lower mangrove canopy cover during low-flow years.

Overall, we did not find a strong relationship between mangrove canopy density and river flow (for the areas examined in this study), with mangrove canopy cover changes more likely to be dominated by regional sea level fluctuation and tropical cyclones that cross through the region. These analyses provide data useful in the assessment of water resource development and water plan reviews proposed in the eastern Gulf region, which is planned over the coming years by the Queensland Government.

Aim 4 – Nutrients and mudflats

This aim focussed on the impact of nutrient additions on primary productivity on mudflats in estuaries, an indication of the importance of catchment nutrient inputs in the wet season for stimulating productivity. In our experiments, Norman, Daly, Flinders and Adelaide River mudflats all had similar rates of oxygen flux and had a statistical increase in oxygen fluxes (as a measure of primary productivity) with the addition of nutrients. Although the addition of nutrients caused a very rapid increase in primary production, i.e. in a couple of days, at times it took a few more days for a statistically significant increase in primary production to occur. This reflects the heterogeneous nature of mudflats. The implications of this study are that all estuaries in this study were nutrient depauperate, and therefore a reduction in nutrient loads from increased freshwater extraction is ultimately likely to decrease primary production on mudflats.

Aim 5 – stakeholder engagement

Stakeholder engagement involved a series of presentations to state and Commonwealth government, NRMS, TO group and a range of other stakeholders. There was a total of eight presentations. There were challenges in connecting with some TO groups, despite repeated attempts.

1. Literature review

Executive summary

Northern Australia contains some of the most pristine tropical estuaries, coastal floodplains and coasts globally. In large part this is because of the low human population pressure across this region. However, catchments are under pressure from significant areas of cattle grazing, causing erosion and other environmental impacts (e.g. spreading weeds) and mining activity is increasing. In some catchments of the Great Barrier Reef (GBR) there is substantial intensive agriculture, e.g., sugarcane, and numerous urban centres, mostly on the coast. Much of our understanding of the impacts of development on estuaries, coastal floodplains and coasts in northern Australia is based on studies in the GBR catchments.

Most of northern Australia has a wet-dry tropical climate, meaning that annually there is a relatively short wet season followed by an extended dry season with little or no rain. This stretches from Cape York to the Kimberley, and also includes some catchments in the GBR region. Climatically, the wet-dry tropics have high interannual variability in rainfall and hence flow. There is growing pressure to further develop this area, including water development for irrigated agriculture, increased intensification of cattle grazing, expansion of urban centres and mining activities. All these activities have the potential to impact the ecological health of estuaries, coastal floodplains and coasts, via freshwater flow alteration and increased pollutant loads. Therefore, this review examined existing studies of the functioning of estuaries, coastal floodplains and coasts in northern Australia, and the current and possible impacts of catchment development. The implications of the research were focused on four estuaries (Flinders, Gilbert, Daly and Keep River estuaries) where there is increasing pressure from catchment development.

The Flinders and Gilbert Rivers are intermittently flowing major river systems in the Gulf of Carpentaria region, Queensland with current and proposed water entitlements for irrigated agriculture. The Daly River, in the Northern Territory, is a groundwater-fed, perennially flowing river system that already has a greater area of irrigated agriculture, with plans for expansion. The Keep River is also an intermittently flowing river on the Northern Territory/Western Australian border, receiving runoff from established irrigated agriculture companies, with further development planned.

1.1. Introduction

Estuaries, coastal floodplains and coastal environments in northern Australia remain relatively undeveloped compared with southern Australia, and many other areas of the world (Halpern et al., 2008). The GBR region is an exception, where there has been significant agricultural and urban development for many decades (e.g., Davis et al., 2017; Shishaye et al., 2020). However, there is growing interest in further development of northern Australia with a focus on areas suitable for irrigated agriculture and mining and, in some regions, further expansion of urban centres (Australian Government, 2015). Key examples of areas earmarked for irrigated agriculture are the southern Gulf of Carpentaria, Queensland, i.e., Flinders and Gilbert River catchments (Dale et al. 2024), the Daly River catchment in the

Northern Territory, and the Ord River Irrigation Area in Western Australia. Development of these catchments has the potential to impact downstream environments such as the estuaries and the coast.

Much of northern Australia is in the wet-dry tropics (Beck et al., 2018), a climatic regime having a monsoon-driven wet season followed by an extended dry season. The scale of the rainfall events is highly variable from year to year. The other climatic regime is the wet tropics which is primarily in northeastern Australia, adjacent to the GBR. It has higher and more predictable rainfall. Many of the river systems in the wet-dry tropics cease to flow in the dry season, meaning that estuaries may become hypersaline (e.g., Andutta et al., 2011). A small number of rivers are groundwater-fed, with a constant baseflow, e.g., the Gregory River (Queensland) and Daly River (NT). The tidal regimes of estuaries in northern Australia also vary widely, with some regions, e.g., the Kimberley, being macrotidal, while others, e.g., the Gulf of Carpentaria have a much smaller tidal range. This is likely to impact the degree to which freshwater flow and its associated materials will impact the estuary, not to mention the frequency and duration of connection the estuaries have with coastal floodplains and marine ecosystems, such as mangroves (Sheaves, 2005).

Development can have a range of pressures on natural systems, including waterways. A number of reviews have been undertaken in northern Australia focussed on actual and potential impacts of water development, but they have been focussed primarily on freshwater systems (e.g., Warfe et al., 2011; Brooks and Spencer, 2016; Douglas et al., 2005, 2019; King et al., 2021). However, the impact of development, including water development on estuaries and nearshore environment, has received less attention. The exception is the GBR region where impacts of anthropogenic impacts on coastal habitats have been studied for many years (e.g. Brodie et al., 2012; Murphy et al., 2013; Davis et al., 2017; Waterhouse et al., 2017; Pearson et al., 2021; Tait et al., 2023). There is scope to use our learnings from studies in the GBR, and other studies throughout northern Australia, to inform our understanding of the potential impacts of development on estuaries and the coast. This review focuses on these learnings.

The impacts of development on estuaries, coastal floodplains and coasts can be grouped by the types of development (catchment change), the stressors associated with these types of development, and the known and anticipated impacts on habitats, processes and species (Appendix Table 1). The catchment changes most likely include increasing irrigated agriculture, agricultural activity (livestock), mining and urban development. Climate change is also likely to transform estuaries and coasts. Development activities can have a wide range of direct and indirect impacts on habitats, processes and species. This includes habitats such as mangroves, coastal wetlands, mudflats, and seagrass beds. Within these habitats, studies have identified key risks to species, including biodiversity and overall species abundance.

1.2. Focal estuaries / coastal floodplains / coast

The four estuaries under pressure from development, as previously mentioned, are the Flinders/Gilbert, Daly and Keep Rivers. They all have different rainfall, flow and land use characteristics.

1.2.1. Flinders / Gilbert River estuaries

The Flinders River starts in the Great Dividing Range, extends westward into Gulf Savannah country towards Julia Creek then heads north to drain through a delta into the Gulf of Carpentaria. The Gilbert River rises below Conical Hill in the Einasleigh Uplands, draining the eastern slopes of the Gregory Range and the western slopes of the Newcastle Range, north of Hughenden. A third of the catchment is a vast estuarine delta largely consisting of coastal tidal flats and mangrove swamps. Both catchments are dominated by cattle grazing and, more recently, an increasing number of agricultural irrigation schemes. Modelled annual end-of-system flows are available for the two rivers, as shown in Figure 1.1. Both rivers have highly variable flows from year to year. However, the Flinders River has higher interannual variability than the Gilbert River. The Flinders River estuary may have multiple years of little or no flow.

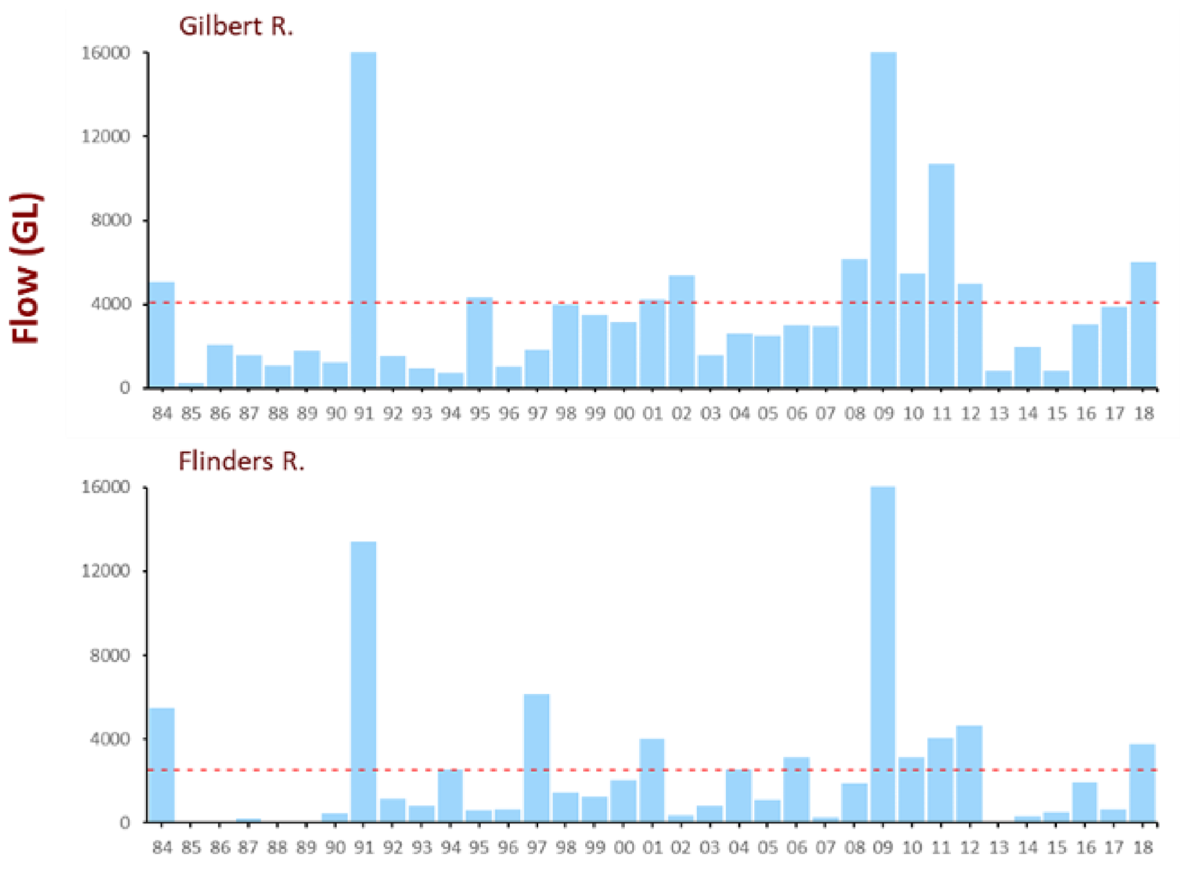


Figure 1-1. Long-term annual flow (GL) based on modelled end-of-system flow (Lerat et al., 2013; Petheram and Yang, 2013; Philip et al., 2018) for Gilbert and Flinders Rivers adapted from Burford and Faggotter (2021). The dashed line shows the long-term average annual flow.

The Gilbert and Flinders River estuaries are characterised as having simple meandering shallow river channels with some small tidal creeks, fringed with a relatively narrow line of mangroves, behind which are extensive salt flats that are only inundated during the wet season or at the highest astronomical tides (Figures 1.2, 1.3). The tidal regime is mesotidal, i.e., 2–4 m.

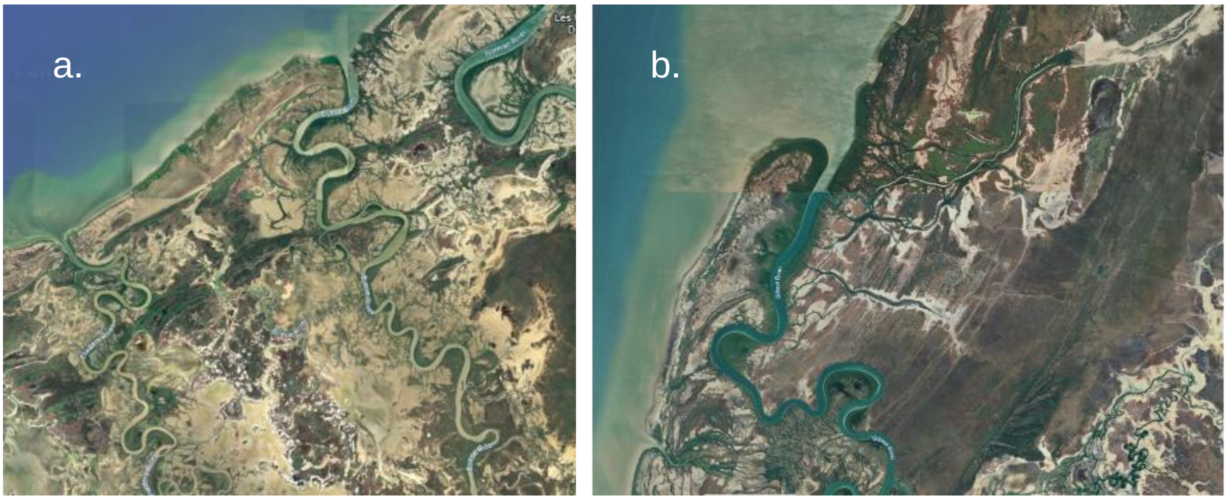


Figure 1-2: a. Satellite Imagery of the Flinders (and Bynoe which is a tributary of the Flinders) River estuaries, and b. Gilbert River estuary in the Gulf of Carpentaria (source: Google Earth).

The main development pressure for the Flinders and Gilbert River catchments is from irrigated agriculture (Dale et al., 2024). There are a number of proposals in the pipeline, e.g. Hipco (https://hipco.com.au/project-details/), Three Rivers Irrigation project (https://www.statedevelopment.qld.gov.au/coordinator-general/assessments-and-approvals/coordinated-projects/projects-discontinued-or-on-hold/three-rivers-irrigation-project) and Gilbert River Irrigation project (https://www.etheridge.qld.gov.au/downloads/file/599/gilbert-river-irrigation-project-brief). The Gulf Water Plan for these rivers is currently being reviewed by the Queensland Government to revise water entitlements relating to these rivers. Additionally, there is pressure for development of more mines in the Flinders River catchment, e.g., a vanadium mine near Julia Creek (https://www.statedevelopment.qld.gov.au/coordinator-general/assessments-and-approvals/coordinated-projects/current-projects/richmond-julia-creek-vanadium-project). Mines have a requirement for water as well as the potential for environmental impacts on rivers and downstream.

There is no regular monitoring of the habitats or water quality of the Flinders or Gilbert River estuaries due to logistical and financial issues.

1.2.2. Daly River estuary

The Daly River catchment (53,000 km^2) is situated south of Darwin, Northern Territory. Mean annual rainfall grades from about 600 mm in the south to over 1300 mm in northern parts of the catchment, with most of the rain falling during the summer monsoon season from November to March. Dry season flows are maintained by discharge from limestone aquifers in the central part of the basin (Wasson et al., 2010). The mean annual flow is 8,317 GL (at Mt Nancar).

Figure 1-1*: a. Flinders River estuary in flood, b. Flinders River estuary during the dry season, c. Gilbert River estuary during the dry season, and d. Gilbert River estuary during the wet season.*

The native vegetation in the Daly catchment consists of Eucalyptus woodlands and open forests, with riparian communities of Melaleuca forests, closed monsoon forests and open Eucalyptus forests on levees (Lamontagne et al., 2005). About 5% of the catchment has been cleared for cropping and plantation forestry and much of the remainder is used for low-density cattle grazing (Townsend and Padovan, 2005). The main pressures on the Daly River catchment are from further development of irrigated agriculture extracting surface and groundwater, and to a lesser degree, mining, e.g. gas fracking.

The Daly River estuary has a simple estuary mouth flanked by intertidal mudflats, beyond which is a fringing mangrove forest (Figure 1.4). This estuary is shallow, with numerous sand and mud banks emerging at low tide, particularly in the lower 30 km. The estuary is 100 km long and is macro-tidal with a peak spring tidal range of about 6 m at the mouth (Wolanski et al., 2004). Tidally-driven sediment resuspension processes dominate the estuary. There have been dramatic changes to the Daly River estuary channel over the last few decades, i.e., 8 km^2 increase in total estuary area from 1972 to 2006 as a result of erosion of the floodplain at a rate of 1.75 Mt per year (Wasson et al., 2014). However, there has been little published research on the extent to which changes to the estuary may be contributing to sedimentation within the upper estuary. This process is important because of the potential for the macrotides in the Daly River to transport sediments upstream towards the head of the estuary.

There is no regular monitoring of the habitats or water quality of the Daly River estuary due to logistical and financial issues.

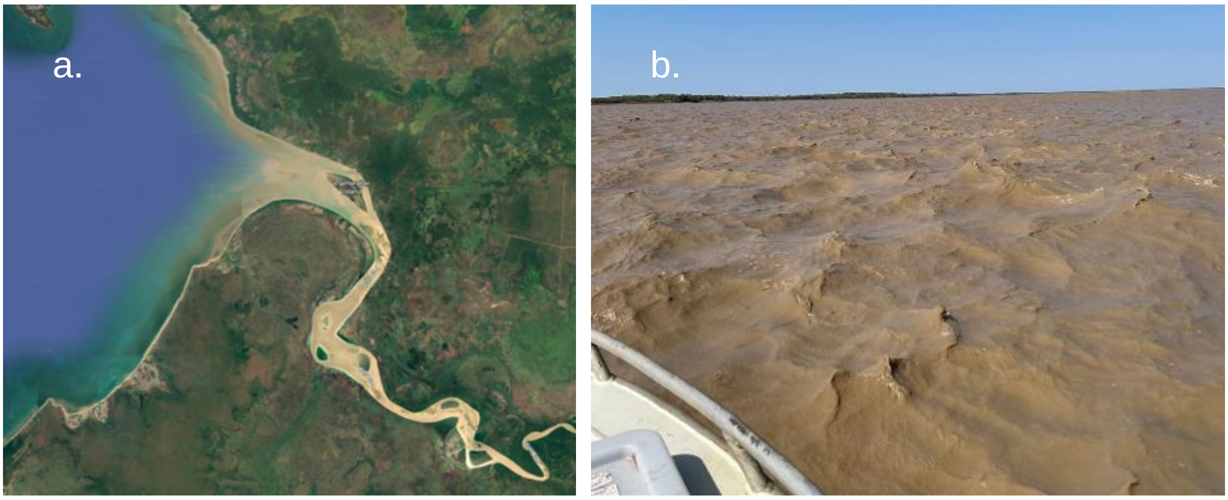


Figure 1-2: a. Daly River estuary (source: Google Earth) and b. View from a boat during the dry season showing the highly turbid waters.

1.2.3. Keep River estuary

The Keep River region lies approximately 100 km south of Joseph Bonaparte Gulf, between the Ord and Victoria Rivers in northwestern Australia (Figure 1.5). The catchment area is 6,000 km^2. The river has a mean annual outflow of 500 GL. Much of the catchment is used for low-intensity cattle grazing.

In 2008, the Ord Irrigation Expansion Project was approved by the Western Australian Government to develop irrigated agriculture on the Weaber Plain. The development makes water from the Ord Rivers system and drainage water from the development flows into the Keep River. Such development had the potential to affect water quality of the downstream aquatic environment of the lower Keep River, particularly in relation to threatened species. Possible increases in salinity, nutrients, suspended sediment, heavy metals and farm chemicals delivered in run-off were of particular interest.

In 2011, the Commonwealth Minister approved the Goomig Farmlands development, subject to it meeting 22 conditions relating to protecting the downstream surface water environment. One of these conditions required a baseline water quality monitoring program, including one site in the Keep River estuary, which was undertaken from 2010-2013. The lower Keep system was designed as slightly to moderately disturbed due to both natural, i.e. macrotidal, climate variability, groundwater discharge, etc. and anthropogenic factors (rangeland cattle grazing). There is no ongoing monitoring of the Keep River estuary (Bennett and George, 2014).

Figure 1-1. Keep River estuary (red marker), Goomig irrigation area (near Kununurra, labelled), and Ord River (source: Google Earth).

1.3. Flow alteration and the impacts

Flow alteration is likely to be a key factor impacting estuaries in northern Australia, due to the development of water resources. The high level of unpredictability of rainfall/runoff events in the wet-dry tropics, coupled with high temperatures leading to high evaporation rates, means that the availability of water for agricultural development can also be variable. To accommodate this variability, water development often involves building infrastructure, such as on- and off-channel dams and weirs, that can store significant volumes of water. Increasing the number and the scale of water storages in northern Australia has been proposed in Federal Government white papers (e.g., Australian Government, 2015), with some examples currently under construction, including the Adelaide River Off-stream Water Storage in the NT (Northern Territory Government, 2023). These potential new water supplies may have a wide range of uses, such as for irrigated agriculture, animal watering, mining operations, electricity generation, and urban drinking water supplies.

Water extraction may reduce the scale, duration and timing of flow. Northern Australian ecosystems, including estuarine and coastal environments, are highly adapted to variable flow but require a minimum flow volume to maintain productivity and biodiversity (e.g., Burford and Faggotter, 2021; Leahy and Robins, 2021; Lowe et al., 2022). Studies in the Gulf of Carpentaria region have predicted that water extraction for irrigated agriculture can substantially reduce the nutrient and sediment loads transported downstream to estuaries, coastal floodplains and the coast (Burford and Faggotter, 2021; Plagányi et al., 2023). This can impact primary productivity in the water column and sediment, as northern Australian estuaries have been shown to be highly nutrient-limited (Burford et al., 2008, 2011, 2012; Burford and Faggotter, 2021). In the Gulf region, there are extensive supratidal mudflats, or salt flats, which are only inundated at the highest astronomical tides, and when freshwater flooding occurs. A study showed that these environments, far from being benign, are important sources of nutrients upon wetting, as well as being areas of significant primary production by benthic algal species, with flow-on effects to food availability for higher trophic levels (Burford et al., 2016). A modelling study of the Ord River estuary by Parslow et al., (2003) estimated reductions in nutrient load by 30% due to water extraction, for example, will lead to approximately a 25% reduction in predicted water column chlorophyll *a* at the same flow. This suggests that phytoplankton biomass in the upper estuary responds almost proportionally to nutrient load, and points to the critical nature of nutrient inputs for productivity. Coastal and estuarine productivity, therefore, relies on freshwater flows and the associated nutrients and sediment to fuel productivity. A reduction in flow due to water development for irrigated agriculture will, therefore, impact on productivity in estuaries and the coast.

Wet season flow to estuaries can have negative short-term impacts on meiofauna and macrobenthos, as well as primary producers, that inhabit intertidal mudflats. For example, a study in an estuary in the Gulf of Carpentaria showed that it causes osmotic stress for animals and plants growing in intertidal mudflats due to the shift from marine to freshwater, as well as smothering of mudflats with catchment-derived sediment loads (Duggan et al., 2014). Large events appear to act more as a disturbance event, than a subsidy for estuarine benthic biota. Lowe et al. (2022) showed that species that can burrow, such as polychaetes (dominant species), are more resilient to changes in salinity than species with less burrowing ability, such as bivalves. Despite the short-term negative effects of freshwater flows, in the months after the wet season, recovery is rapid with these species re-establishing. The increased primary productivity, as a result of the nutrient inputs from freshwater flows, increased food availability (Burford and Faggotter, 2021).

Mangrove forests rely on periodic freshwater flows and/or rainfall, and associated nutrients and sediment to sustain them. One study has shown an increase in mangrove area in the Gulf of Carpentaria region up until 2014 (Asbridge et al., 2015). However, this was prior to the mass mangrove dieback across the Gulf due to a sea level anomaly (Duke et al., 2021). A modelling study in the Gulf Carpentaria predicted declines in mangrove abundance with increased water extraction in the catchment (Plagányi et al., 2023). However, the importance of freshwater for mangroves may be offset by high mortality rates and slow recovery rates during periods of drought and after tropical storms (Lovelock et al., 2009, Feller et al., 2015a). This is because nutrients, especially nitrogen, stimulate the growth of shoots relative to roots, which causes physical instability of the trees (Lovelock et al., 2009). The decrease in root biomass can also enhance the subsidence of soils (McKee et al., 2007). Studies have shown that in the GBR, high nutrient loads delivered in floodwaters could be a contributor to the localised dieback of mangrove forests when followed by periods of low rainfall and, thus, high salinity or after tropical storms and cyclones (Asbridge et al., 2015; Feller et al., 2015).

During a period of very low rainfall, mangroves in Port Douglas and Hinchinbrook Channel had significantly greater canopy loss in trees fertilised with nitrogen at the most saline site (Lovelock et al., 2009).

Freshwater flowing into estuaries and nearshore drives fluctuations in growth, reproduction and survival of many species. One key impact on the physiology of many species is the change in salinity regime. Much of the research on the effect of flow alteration on species globally has focussed on fisheries species (Broadley et al., 2022). Regulation of freshwater flows coupled with human-induced changes around coastal areas is believed to have contributed to the decline of many fish and invertebrate stocks globally. A study by O'Mara et al. (2023) on effects of flow alteration on fisheries species in northern Australia showed that much of the research has focused on direct flow-biota relationships, although there have also been some modelling studies. Barramundi (*Lates calcarifer*) was the species most studied (Figure 1.6). There have been very few studies directly measuring the impact of flow alteration.

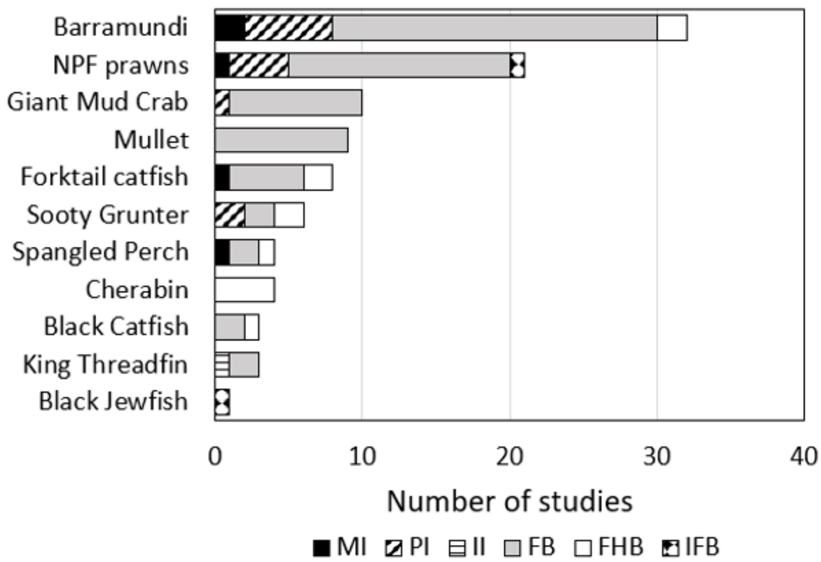


Figure 1-6. Categories of flow/impact focus for fisheries species in northern Australia. PI = predicted impact, II = inferred impact, FB = direct flow-biota relationships studied, FHB = flow-habitat-biota (indirect flow) relationships, IFB = inferred flow-biota relationship from observation (From O'Mara et al., 2023).

Barramundi catch rates have been linked with the scale of freshwater flows (Robins et al., 2005). River discharge has been shown to have a strong positive effect on juvenile growth rates for young Barramundi. Water resource development could potentially reduce juvenile Barramundi growth rates, and modelling studies have estimated the scale of reduction in growth rate as a result of water extraction scenarios, e.g. Nullinga dam on the Mitchell River (Leahy and Robins, 2021). The commercially caught mud crab (*Scylla serata*) also relies on freshwater flow for transport and stimulation of food supply (Robins et al., 2005). Modelling studies have shown that this species will also be impacted by alteration in flow (Playangi et al., 2023). Novak et al. (2017) also showed that wet season water discharge was critical for transporting the crustacean *Macrobrachium* larvae from the river to the estuary in order for them to grow and develop.

Banana prawns are an important commercial fishery in northern Australia and use estuaries as refuges and for feeding during the post larval and juvenile phases. Freshwater flows act as a cue to move from estuaries into deeper waters each year, due to osmotic stress in the low salinity and a reduction in food supply (Vance and Rothlisberg, 2020; Duggan et al., 2014; Lowe et al., 2022). Adults are then caught by commercial fishing in deeper waters. Modelling studies for the Flinders, Gilbert and Mitchell Rivers, Gulf of Carpentaria, have shown it is important to maintain flow in low flow years to ensure catch is maintained (Plaganyi et al., 2023). Therefore, water development and particularly water extraction in low flow years can have a negative impact on catch. If proposed dams are built, e.g. dams on the Mitchell River, and higher water extraction in adjacent rivers, the reduction in catch in a low flow year has been estimated to be 53% (Broadley et al., 2020). Duggan et al. (2019) used a Bayesian modelling approach to determine thresholds of freshwater flow that affect the likelihood of prawn catches and suggested modifying flow reduces the probability of high catches. Additionally, Turschwell et al. (2022) quantified the effect of flow alteration on fisheries catch. The economic impacts of a reduction in flow on banana prawn catch have also been estimated. Profit could be reduced between 7-12% if currently granted entitlements and planned allocations are extracted from Gulf rivers, or by 22% if additional dams are constructed in the Mitchell River (Smart et al., 2021).

Water development may also involve building dams that will alter the flow regime, such as changing flow from intermittent to constant baseflow, e.g., as is the case in the Ord River scheme where the Ord River dam controls flow downstream to supply irrigated agriculture and generate hydroelectric power. The construction of dams can also alter the loads of nutrients and sediments, as well as the characteristics of those loads. Several studies reported that reservoirs, such as the Burnett, Pioneer, Burdekin, Tully and Barron, trap a significant proportion of the total suspended solid loads in rivers and, by inference, particulate nitrogen and phosphorus loads (Kroon et al., 2012; Lewis et al., 2013). Reservoirs also transform particulate nutrients into dissolved inorganic nutrients as a result of the long water residence time and anoxic bottom conditions. These dissolved inorganic nutrients may then be released downstream, stimulating primary productivity (Brodie et al., 2015). However, reservoirs can also reduce primary productivity downstream. For example, a study of the Ord River, a region where irrigated agriculture occurs, suggested that the presence of Lake Argyle, built by damming the Ord, impacted nutrients and primary productivity via a reduction in wet season flows downstream, with a resultant decrease in nutrients (Burford et al., 2011).

On-channel dams impact the connectivity of rivers. Many marine and freshwater fish species in northern Australia rely on this connectivity to move up and down river systems (Robins et al., 2005). In Queensland, the significance of connectivity of estuarine wetlands has been linked to inshore fisheries production (Meynecke et al., 2008), the composition of estuarine fish communities (Sheaves and Johnston, 2008) and the capacity of wetlands to keep pace with sea level rise (Lovelock et al., 2011). Many fish species use coastal wetlands as nurseries or breeding grounds, and habitat removal or alteration disrupts their life cycles (Meynecke et al., 2008; Sheaves et al., 2015; Harris et al., 2016). Building of infrastructure for irrigated agriculture, such as dams and weirs, can impede movement of these species that may move up and down river systems, and into estuaries and the coastal zone (Ferrier et al., 2021). This includes endangered species such as sawfish, whip rays and river sharks, which depend on estuaries, rivers and tidal channels for part of their lifecycle. The freshwater sawfish species (*Pristis pristis*), which is critically endangered, has been shown to rely in the first few years almost entirely on large wet season foods, and the brief periods of highest water levels within these years, to replenish juvenile populations in the Fitzroy River (WA) nursery (Lear et al., 2019). Plaganyi et al. (2023) modelled the effect of water extraction and dam building on sawfish in the Gulf of Carpentaria and found all scenarios were predicted to result in major declines in stocks.

Therefore, the challenge is to ensure that flow extraction volumes, including timing and duration, do not have adverse impacts on downstream ecosystems.

1.4. Pollutants and their impacts

1.4.1. Irrigated agriculture

In the wet tropics of the GBR region, large areas of irrigated agriculture can increase the loading of pollutants to rivers and downstream estuaries. Brodie et al. (2012) and more recent studies in the 2022 Scientific Consensus Statement for the GBR (https://reefwqconsensus.com.au/themes/) highlighted the importance of the export of suspended sediment, nutrients and photosystem II-inhibiting (PSII) pesticides, with most of these pollutants being delivered during river floods. Sugar cane cultivation is a major form of irrigated agriculture and has been shown to be a major contributor to dissolved inorganic nitrogen and pesticide loads to the GBR (Waterhouse et al., 2012; Fraser et al., 2017; Masters et al., 2017; Lewis et al., 2021; McCloskey et al., 2021), via fertilizer and pesticide application (Bainbridge et al., 2009; Mitchell et al., 2009; Thorburn and Wilkinson, 2013; Connolly et al., 2015). Fertilizers are typically inorganic forms, or simple organic forms, such as urea, that are rapidly transformed into ammonium. Studies have shown that the dominant form of nutrient runoff from catchments with irrigated agriculture has changed from dissolved organic nitrogen and phosphorus to dissolved inorganic nutrients, such as nitrate and phosphate (Prosser and Wilkinson, 2022). This change has occurred along with a range of other anthropogenic impacts such as increased weed infestations, hydrological modifications, and climate change.

Nutrient runoff leads to consistently elevated nutrient concentrations in inshore waters in the GBR compared with concentrations in offshore waters, especially in the Wet Tropics, Burdekin and Fitzroy regions (Brodie et al., 2012; Fabricius et al., 2016; Webster et al., 2006). A number of studies, such as Brodie et al. (2012), Fabricius et al. (2010), and Furnas et al. (2005), determined that much of the sediment and nutrients discharged from rivers settles out quickly and is deposited within 25 km of the coast, but may occasionally extend further out. One example was a plume extending for 70 km^2 and covering an area of ~200 km^2, following the 2010-2011 Fitzroy River flood (Jones and Berkelmans, 2014). Floodplumes can reach coral reefs and seagrass meadows in the GBR but there is high inter-annual variation in exposure (Devlin et al., 2012; Álvarez-Romero et al., 2013; Devlin et al., 2015a).

Elevated nutrient loads from rivers in the GBR are linked with higher chlorophyll *a* concentrations in the GBR lagoon and are typically highest during the wet season (e.g., Devlin et al., 2013a; Oke et al., 2015). The highest concentrations have typically been observed at the outer edge of a flood plume (Oubelkheir et al., 2023) and may be up to 50 times higher than background concentrations (Brodie et al., 2010). Baird et al. (2021) have predicted that chlorophyll *a* concentrations could be reduced by between 0.01-0.10 mg m^{-3} if there was a reduction in anthropogenic nutrient loads, as proposed in the Reef 2050 Water Quality. The variability in timing and intensity of floods also affected the spatial distribution and temporal dynamics of water clarity (Fabricius et al., 2014; Fabricius et al., 2016) and marine microbial communities (Angly et al., 2016).

Herbicides and pesticides transport from land to water is a significant concern in the GBR. Brodie et al. (2012) showed frequent exceedances of pesticides above Australian water quality guidelines in rivers, streams and estuaries draining to the GBR, including the pesticides atrazine, diuron and metolachlor. Metolachlor is a herbicide used for grain growing, and Murphy et al. (2013) found that the loads (dominated by dissolved, rather than particulate forms) entering waterways are highly dependent on the timing and characteristics of the first flow rainfall events after application. Where 40% or greater crop cover is maintained, zero till has been found to be effective at reducing herbicide loss to waterways. Gallen et al. (2014) found that a wide range of PSII herbicides and other pesticides (e.g. terbutryn, imidacloprid and imazapic) were also detectable in the GBR. Flood plume monitoring showed that PSII herbicides are generally 'conservatively mixed', that is, concentrations become increasingly diluted as the freshwater discharge progressively mixes with seawater (Devlin et al., 2015b). The half-lives of PSII herbicides, ametryn, atrazine, diuron, hexazinone and tebuthiuron have been shown to be greater than a year (Mercurio et al., 2015), indicating high persistence and explaining their year-round presence in the GBR (Gallen et al., 2014). Given that herbicide persistence is higher in low-light conditions, it is likely that limited degradation would occur during transport in wet season flood plumes (Mercurio et al., 2014; Mercurio et al., 2015).

Understanding the impacts of pollutants from irrigated agriculture, and their interacting effects, in estuaries and the coast is complicated. Extreme conditions co-occur with elevated discharge including cloudiness and low incoming solar radiation (leading to low light), large waves and currents during storms can have direct physical effects on organisms as well as indirect effects through resuspension, and other water quality variables that co-vary with discharge (e.g. dissolved nutrients, herbicides, low salinity) leading to cumulative pressures. There are also lag times in biological responses, with some accumulating over multiple years (Lambert et al., 2021).

Estuarine wetlands are an important habitat in many areas of northern Australia and may be either adjacent to estuaries and coasts or in low-lying areas above the mangrove forests. They may be regularly inundated or have infrequent inundation. Wetlands may have a relatively high capacity for nutrient retention in low and medium flood events and can play a role in protecting the marine environment from land-derived nutrient pollution (Valiela and Cole, 2002; Alongi and McKinnon, 2005; Adame et al., 2010a). However, in the long term, nutrient enrichment may have negative consequences affecting vegetation structure and composition and reducing their capacity for nutrient retention (Verhoeven et al., 2006; Reef et al., 2010). Coral reefs can also be impacted by pollutants associated with freshwater flows. A review by Brodie et al. (2012) showed that chronic exposure to herbicides from intensive agriculture can lead to decreased photosynthesis rates in coral, bleaching, partial colony mortality, reduced tissue lipid content and reduced fecundity in corals.

Much can be learnt about the interacting effects of grazing and irrigated agriculture in catchments such as the Burdekin River. This river has similar grazing pressure to the northern Queensland catchments, but irrigated agriculture is also common. In these catchments, irrigated agriculture has led to significant increases in nutrient loads flowing to waterways associated with fertilizer application as well as increased pesticide and herbicide runoff (Brodie and Mitchell, 2005; McCloskey et al., 2021). On the other side of the ledger, the Burdekin Falls Dam has acted as a significant sediment trap since its construction in the 1980s (Lewis et al., 2013) which has counteracted some of the sediment and particulate nutrient supply increased by grazing-induced gullying (Shellberg et al., 2016; McCloskey et al., 2021). The potential effects of the development of irrigated agriculture on soil erosion have been examined in the Gilbert/Etheridge River system (Brooks and Spencer, 2016). This catchment already has significant impact of cattle grazing, particularly on erosion risks. The study flags that the geomorphic and soil characteristics will exacerbate erosion risk from further development, with 106 active gullies within just one proposed development. The study proposes mechanisms to reduce erosion including substantial buffers around all drainage lines and existing gullies.

1.4.1.1. Non-irrigated agriculture development (livestock)

Much of northern Australia has not been subjected to irrigated agriculture, but cattle grazing dominates 54% of the area (ACLUMP, 2016). Cattle often use natural waterbodies, such as creeks, waterholes and wetlands, for drinking water supplies. The grazing and watering activities of cattle can result in significant erosion, typically gully and stream channel erosion, resulting in the loss of sediment to waterways. This can silt up waterways and reduce light availability for primary productivity. Across northern Australia, the initiation of gully erosion in association with cattle grazing has had profound impacts on catchment scale sediment budgets (Brooks et al., 2009; Caitcheon et al., 2012; Shellberg et al., 2016) The impacts of irrigated agriculture on catchment material fluxes, therefore, need to be understood within the context of these changes to catchment sediment budgets.

Studies in the Daly, Mitchell and Flinders River catchments in northern Australia have shown that both sediment channel and gully erosion processes are important contributors to sediment loads from catchments into estuaries and coastal environments (Caitcheon et al., 2012). An estimated 90% of the sediment load was shown to be from subsoils. This erosion was initiated by clearing of forests, introduction of grazing stock, drainage of valley bottoms, and clearing of riparian vegetation, commencing 180 years ago. they propose that the elevated sediment loads increase the risk of flooding, due to a reduced capacity of river channels and increased likelihood of channel blockages. Another study in the Mitchell catchment showed that the high density of cattle grazing in riparian zones during the dry season decreased perennial vegetation cover along hollows and steep river banks, increasing the potential for gully erosion (Shellberg et al., 2010, 2016). In the Daly River, NT, the dominant process in cattle grazing areas is also gully erosion (Wasson et al., 2014). The alluvial gullies produce about 24% of the sediment input to the main channel. However, another change that has been reported in the study is that river discharge has increased along with rainfall. This has led to a widening of the channel by river flow and mass failure of the banks due to elevated groundwater levels in the floodplain, and large floods. This has implications for sediment loads in the estuary, but the fate of this sediment is unclear.

Some GBR catchments, e.g. Normanby basin, have significant erosion because of overgrazing cattle, cattle pads along river frontage, poorly designed and maintained roads and fence lines, as well as intensive agricultural activity (Brooks et al., 2013). Sediment source tracking has shown considerable sediment production from small alluvial tributaries and alluvial gullies. The study found that the contribution was far greater than expected, with impacts being cumulative across the catchment. They are now important drivers of water quality at the catchment scale. These high erosion rates also cause increased sediment and nutrient loads entering the adjacent marine waters, i.e., Princess Charlotte Bay (Howley et al., 2018). Clays <4 µm diameter have been shown to be preferentially transported to the estuary, with an estimated 46% sediment delivery ratio. In the estuary, these suspended sediments are then affected by tidal resuspension processes, causing significant DIN release (Howley et al., 2021). Seagrass and coral ecosystems are then exposed to flood plumes containing dissolved and particulate nutrients, and suspended sediment elevated above ambient levels, which is likely to have significant effects on the health of these ecosystems.

A study in the Victoria River catchment, NT, also highlighted the role of agriculture, combined with increased rainfall, in exacerbating gully erosion, and identified the need to protect riparian zones (McCloskey et al., 2016).

The Burdekin River is a well-studied major river in the GBR region, with extensive agriculture and significant erosion issues. Studies have shown that most fine sediment (<63 µm diameter) delivered from the Burdekin River is retained in coastal waters within 50 km of the river mouth (Delandmeter et al., 2015; Lewis et al., 2014), but wind- and tide-driven resuspension can result in remobilisation of this fraction (Bartley et al., 2014a). Studies have also recognised that the mineral fraction <20 µm travels the furthest in riverine flood plumes, forms the nucleus of organic-rich sediment flocs, is more easily resuspended from the seabed and hence likely disproportionately contributes to reductions in water clarity (Fabricius et al., 2016; Bainbridge et al., 2018; Bainbridge et al., 2021). This fraction, which also contains particulate nitrogen and phosphorus can form organic-rich flocs and result in increased turbidity in coastal areas following major discharge events from the Burdekin River (Bainbridge et al., 2012; Fabricius et al., 2014; Lewis et al., 2014; Lewis et al., 2015). There were few studies on the impacts of particulate nutrients, making it difficult to account for the risk they pose to GBR ecosystems (Bainbridge et al., 2018). The floodplain has an extensive distribution channel network that delivers irrigation water for use by sugar cane farmers. These channels may have poor water quality, reducing oxygen levels and causing issues for fish survival (Waltham et al., 2020a; Waltham et al., 2020b).

Particulate organic nutrients from erosive processes may deposit in sediments in estuaries and at the mouths of rivers but later remineralise, releasing dissolved nitrogen and phosphorus to the overlying water column. Additionally, particulate nutrients may be released as ammonium, as suspended sediments reach saline waters (Garzon-Garcia et al., 2021). Therefore, nutrient concentrations in inshore areas remain elevated even after the flood plumes disperse (e.g. Howley et al., 2018; Lonborg et al., 2018; Marion et al., 2021).

Seagrass meadows are a critical habitat in northern Australia’s inshore environments, supporting megafauna such as dugong and turtles and providing ecosystem services that make them a high conservation priority (e.g., Cullen-Unsworth and Unsworth, 2013; Coles et al., 2015). Seagrass beds can be impacted by freshwater inputs, with the major water quality pollutants that undermine seagrass resilience being fine sediment, elevated nutrients and herbicides (Brodie et al., 2017). Therefore, both non-irrigated agricultural development, and irrigated agricultural development impact on seagrass beds. Studies have shown widespread impacts on seagrass along the GBR coast from multiple years of above-average rainfall and extreme weather events (in 2009-2011; McKenzie et al., 2012; Rasheed et al., 2014; Petus et al., 2014; Petus et al., 2016). Since that time, inshore seagrass meadows in most areas have only recovered slowly. The capacity of seagrass meadows to recover following disturbance depends on the interaction between light availability, nutrient loads, habitat properties and the availability of propagules to establish new populations (Grech et al., 2016; McKenzie et al., 2016). There has been some evidence of recovery in Cape York, Mackay Whitsunday and Burnett Mary regions, while abundance in the Burdekin region has shown substantial recovery (Davies et al., 2016; McKenzie et al., 2016). In the Wet Tropics, some seagrass beds have failed to recover their abundances in 2015 (McKenzie et al., 2016), while others, including Cairns Harbour, have shown signs of recovery (York et al., 2016). In the Fitzroy region, ongoing disturbances, including tropical cyclone Marcia, have delayed recovery. By contrast, the meadows to the immediate north of the GBR in the Torres Strait have remained relatively stable over similar time frames (Carter et al., 2014; Carter et al., 2015; Sozou et al., 2016).

Studies of seagrass meadows on the coastal environment near the Burdekin River in the GBR catchment have shown that seagrass biomass declined when suspended sediment loads increased during high periods of freshwater flow, reducing light availability (Lambert et al., 2021). However, there was a time delay in response. Additionally, different seagrass species, depending on their growth requirements, varied in their response and recovery times. Baird et al. (2021) characterised the impact of sediments on light availability for seagrasses, using a coupled hydrodynamic-biogeochemical model, to provide information about the scale of the loads that are likely to have an effect.

The loss of seagrass from reduced water quality and physical disturbance as a result of floods and tropical cyclones in the GBR is known to have significant flow-on effects on dugong and green turtle populations that feed in seagrass meadows (Preen and Marsh, 1995; Marsh et al., 2011; Meager and Limpus, 2012). Malnutrition makes these herbivores prone to disease, and lack of food may force them to move long distances to find alternative sources. As a consequence of the widespread loss of seagrass along the developed coast of Queensland in early 2011, stranding rates of sea turtles and dugongs increased dramatically and produced high mortality rates. As seagrass abundance improved in many regions in 2015, dugong mortalities decreased. In contrast, turtle mortalities have declined since 2011.

High concentrations of suspended sediment can interfere with filter feeding, alter the quantity and quality of light available for photosynthesis by coral symbionts, and smother corals with a fine layer of sediment that requires mucus production and energy to clear (Jones et al., 2015). Suspended sediments also impact the reproductive cycle and early life histories of corals (Jones et al., 2015b). Developing embryos and larvae can tolerate exposure to suspended sediments by having mechanisms to remove particles (Ricardo et al., 2016), however, colonisation of reef surfaces by coral spat is threatened by the deposition of fine, terrigenous sediments (Perez et al., 2014; Jones et al., 2015).

1.4.1.2. Mining

Mining occurs throughout northern Australia, and there are many mines in close proximity to waterways. The effects can be either chronic, due to ongoing mining activity, or acute, due to an accidental spillage. There is very limited scientific information freely available on the effect of mining on the hydrology and pollutant loads in estuaries and coasts. However, some studies on the effect on freshwater systems have been conducted which provides some insights into potential estuarine effects.

One study of the long-term effects of the Ranger Uranium mine in the downstream freshwater Magela Creek, NT, found elevated concentrations of magnesium and other mining-related solutes, low dissolved oxygen and risks for fish from the solutes, especially under low and recessional flow (Crook et al., 2021). However, no adverse behavioural responses were found in fish in concentrations of magnesium four times the chronic exposure limit in mine water discharge. Van Dam et al. (2002) have proposed a program for assessing the effects of uranium mining on aquatic ecosystems based on a four-tiered best-practice approach, including 1) the derivation of local water quality guideline trigger values, 2) direct toxicity assessment of mine waters prior to their release, 3) creekside or in situ monitoring for early warning of adverse effects during mine water release, and 4) longer-term monitoring of macroinvertebrate and fish communities. Bioaccumulation in aquatic biota is also important to assess, both in the context of ecosystem health, as well as the health of local Aboriginal people who consume aquatic animals.

Downstream of the Mt Isa mine, Queensland, water and sediment in Lake Moondarra and the Leichhardt River have been shown to frequently exceed Australian government sediment guidelines for copper, lead and zinc (Taylor et al., 2009). Lake Moondarra is a potable water supply for the township of Mt Isa. Dry season analysis of water-soluble copper, lead and zinc concentrations within pools showed that Australian government low trigger guidelines were exceeded in 100%, 46% and 100% of cases, respectively. The impacts on biota were also examined by assessing the metal content of the tissue of seven fish from Lake Moondarra. Tissue metal values were generally low with only a few samples having copper and zinc values in excess of the recommended Australian retail guideline values for fish suitable for human consumption.

Another study of an accidental release of metal-contaminated waters from the Lady Annie Copper Mine in northwest Queensland found that nearby creeks and floodplains were contaminated, with copper contamination as the principal element of concern (Taylor and Little, 2013). Approximately 43% of channel surface and 31% of floodplain surface samples exceeded the Australian guideline value for sediments. However, only the first 5 km from the release site had elevated levels, and the authors concluded that the legacy risk posed to grazing cattle was considered low.

The Century zinc mine at Lawn Hill in the southern Gulf is another example of the release of metal-contaminated waters which occurred in 2009 and 2022. The 2022 spill occurred via a break in the pipeline 30 km from Century Mine, along the 304km slurry pipeline used to transport zinc concentrate to a Karumba port, with 575 tonnes of zinc concentrate released into the environment. In this example, the spill was rapidly cleaned up and is unlikely to have major impacts on adjacent rivers, but it highlights the potential for impact if the break occurred near a river system.

1.4.1.3. Urban development

Much of the urban area in northern Australia is along the east coast of Queensland, adjacent to the GBR, with more than 10% linear of the coastal line in the GBR being transformed into hard engineering structures (e.g. seawalls, boat ramps, ports; Waltham and Sheaves, 2015). An investigation into the presence of traffic-derived metals within road, stream and estuarine sediments in the Cairns catchment (in the GBR region) found distinctly elevated zinc values in road sediments due to abundant tyre rubber shreds (as verified by SEM-EDS and correlation analysis; Pratt and Lottermoser, 2007). By comparison to the road sediments, background stream sediments taken upstream from roads had relatively low median Pb, Pd, Pt and Zn concentrations (7.3 mg/kg Pb, 0.01 mg/kg Pd, 0.012 mg/kg Pt, 62 mg/kg Zn). Mobilisation of dust and sediments from road surfaces also resulted in relatively elevated Pb, Pd, and Pt concentrations and non-radiogenic Pb isotope ratios in the local downstream coastal stream and estuarine sediments. However, the levels in biota were not studied, so the impact of these elements on estuarine health is unclear.

A study in Darwin Harbour found statistically significant contributions of urban metal sources to harbour sediment close to Darwin City (Munksgaard et al., 2019). Copper, zinc and lead concentrations were elevated in tidal flat sediment near Darwin City, i.e., 28%, 29% and 20%, respectively, compared to the levels found in the remaining areas of the harbour. However, metal and metalloid levels in intertidal flat and mangrove creek sediment in Darwin Harbour are generally well below Australian sediment quality guidelines. Urban areas can also have higher nutrient loads, due to both stormwater and treated sewage discharge. In a study of Darwin Harbour, Fortune et al. (2020) found that urbanisation is increasing the nitrogen loads. Studies have shown measurable effects on urban tidal creeks, including changes in nutrient processing rates, i.e. denitrification, benthic nutrient fluxes, between creeks impacted and unimpacted by sewage inputs (Burford et al., 2012; Smith et al., 2012; Fortune et al., 2022).

Dredging is another activity that results in coastal and estuarine impacts. Studies have shown that the turbidity implications of dredging activities in GBR reef environments are highly complex over time frames from hours to weeks (Jones, et al., 2015a). The direct and indirect effects of dredging are severe within the dredging footprint and could be significant at local and regional scales (McCook et al., 2015).

In terms of wastewater treatment plants, pharmaceuticals and personal care products have been detected in effluent discharging into rivers that flow into the GBR lagoon (O'Brien et al., 2014; Scott et al., 2014). These studies found that in the majority of cases, pharmaceuticals measured in treated sewage were at generally low concentrations, i.e., between 10 and 500 ng L^{-1}, with some higher concentrations, e.g., up to 2.3 μg L^{-1} for iopromide (used in X-ray imaging; O'Brien et al., 2014). In river water, the painkiller, paracetamol was reported in the Fitzroy region at a concentration of 4.1 μg L^{-1} (Scott et al., 2014). Another study in Darwin Harbour found that the chemicals present in the highest concentrations in treated sewage were acesulfame, paracetamol, cholesterol, caffeine, DEET and iopromide (20 μg/L, 17 μg/L, 11 μg/L, 11 μg/L, 10 μg/L and 7.6 μg/L, respectively). Wastewater effluent had higher concentrations of DEET than reports in other studies (French et al., 2015).

1.5. Climate change effects

Climate change has a range of effects in estuaries and the nearshore, including increases in temperature and evaporation, sea level rise, alterations in rainfall patterns and extreme meteorological events. These changes will impact a range of factors linked to the ecosystem health and biodiversity of estuaries and the nearshore. Close et al. (2012) summarised a range of potential impacts of climate change and development on aquatic ecological assets in northern Australia. The most accurate parameter projections are for increased temperatures and sea level rise. Studies have found potential for saltwater intrusion in the coastal plains of the Mary River in the Northern Territory (Knighton et al., 1992; Mulrennan and Woodroffe 1998). Bayliss et al. (2011) did a study on the risk of sea level rise in 53 basins in northern Australia and found that the following rivers with low typography, such as the Adelaide River basin, followed by the Mary River catchment in the NT, and in Queensland, the southern Gulf of Carpentaria were most at risk from sea level rise (with the highest risk being the Mornington Inlet).

All species have a range of temperature tolerances, and it is unclear how the projected increases in temperature will affect many individual species in estuaries and the nearshore. A study of temperature tolerances of estuarine fish in a tidal lake in northern Australia showed that a 2.3 °C climate warming (based on 2100 local climate prediction) raised the water temperature by 1.3 °C (Waltham and Sheaves, 2017). This led to a doubling of time that water temperatures were in excess of acute effect temperatures (AET) for fish at the surface, but also the bottom waters that presently provide thermal refugia for fish. A recent study showed the projected range of groundwater temperature increase (Benz et al., 2024). It is likely that this increase in groundwater temperature will exacerbate any increase in surface water temperature due to atmospheric warming.

Climate change will also increase evaporation rates, and whilst there is limited information on how much of an impact this will have, many northern Australian estuaries are already hypersaline late in the dry season (e.g., Burford and Faggotter, 2021). The southern Gulf of Carpentaria region is projected to have increased aridity with reduced stream flow, exacerbating hypersalinity (https://longpaddock.qld.gov.au/qld-future-climate/adapting/water/). Further increases in salinity above seawater salinity are likely to impact on the productivity of a range of species. The most poorly understood effects of climate change are changes in weather patterns that impact rainfall, extreme events such as the magnitude and frequency of cyclones and alterations in ocean currents. This will potentially impact flood magnitude and frequency and hence catchment erosion processes. Storm surges can also interact with cyclones and storm events, further exacerbating coastal erosion and the erosion of the coastal plain.

King et al. (2015) suggested that climate change is also likely to exacerbate the impacts of water development and highlighted the challenges around determining the effect of concurrent stressors, such as changing land use patterns, increased sediment input, and toxicant input.

Other potential climate change effects are alterations to the magnitude and frequency of cyclones, with potential flow-on effects to flooding. Storm surges may also change – this is known to be a driver of erosion of the coastal plain.

1.6. Other challenges

There are a range of other stressors that impact on estuarine habitats and processes. This includes invasive species such as feral pigs and buffalo. Feral ungulate populations have considerable impacts on soil, water quality, waterhole hydrology, vegetation, fire regimes and the spread of exotic plants, and resulting changes to habitat have flow-on effects for native wildlife (Mihailou and Massaro, 2021). These impacts are similar to those of domesticated ungulates, primarily cattle, that dominate northern Australia's landscape. Whilst most of the research on impacts of feral ungulates has focussed on freshwater habitats, such as wetlands and floodplains (e.g. Marshall et al., 2019; Waltham and Schaffer, 2021), it is likely that habitats adjacent to estuaries and the coasts, e.g. mangroves and salt flats, are also likely to be impacted. There is a need to develop a better understanding of the impacts of feral animals on these habitats and their ecosystem functioning. Weeds are also an issue in the riparian zone of estuaries. An example of this is rubber vine infestation in many floodplain areas of northern Australia, including the Gulf of Carpentaria catchments (e.g. Fensham 1996).

1.7. Summary – implications for estuaries in the study

The review of the literature has outlined a range of threats to estuaries from agricultural development, both irrigated and non-irrigated, urban expansion and mining. These activities have the potential to impact habitats, species and processes, depending on both the scale and type of activities within each catchment and the nature of the receiving environment.

Below is outlined key risks for the four focal estuaries in our study: Flinders, Gilbert, Daly and Keep Rivers.

1.7.1. Flinders and Gilbert Rivers, Queensland

The Flinders and Gilbert Rivers in the southern Gulf are considered together as they share many characteristics in terms of development, and estuary characteristics, as well as both being in the southeast Gulf region. These rivers have had extensive cattle grazing for many years, which resulted in significant erosion across the catchment due to lthe oss of vegetation cover and the development of gully and riverbank erosion (Table 1.1, Figure 1.7). The eroded sediment and associated nutrients are transported into estuaries and the nearshore during the wet season and may increase silting up of the river mouth but can also facilitate the creation of new habitat for mudflats and mangroves. The nutrients transported to the estuary are key to stimulating primary production with flow-on effects to higher trophic levels.

In the short term, irrigated agricultural development will increase water extraction, reducing sediment and nutrient transport, with the likely effect of reducing primary and secondary production, and decreasing the catchment of commercial and recreational fish species (Table 1.1, Figure 1.7). Whilst on-river storages appear unlikely, they would further reduce sediment and nutrient transport downstream to estuaries.

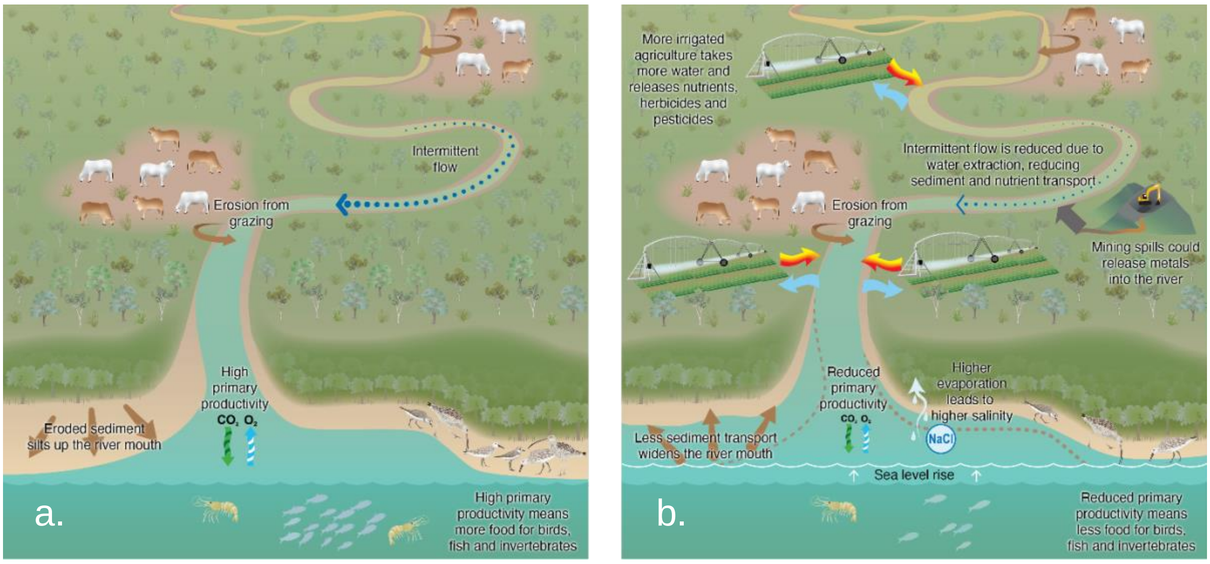


Figure 1-7 a. Current and b. future potential pressures and impacts of development on estuaries, coastal floodplains and the coast of the Flinders and Gilbert River systems.

In the longer term, if the scale of irrigated agriculture becomes sufficiently large, nutrient, pesticide and herbicide additions will begin to impact the rivers, and potentially downstream estuaries, as has been seen in the Great Barrier Reef catchments (Table 1.1). This being the case, there is the potential for both stimulation of primary productivity, from nutrients, but also suppression of primary production, particularly from herbicides. Irrigated agriculture expansion is likely to expand urban centres, which are currently in the mid-to-upper catchments, but the footprint is likely to remain small, with the main impacts adjacent to the urban centres near the towns rather than downstream in the estuaries. Mining is also being proposed and developed, particularly in the Flinders catchment. There are ecological risks from metals transported downstream if containment is insufficient, or during extreme events, e.g. major flooding. Spills have already occurred related to another mine in the southern Gulf region (e.g. https://www.abc.net.au/news/2022-10-26/gulf-of-carpentaria-new-century-pipe-spill/101579282).

In terms of mitigation of impacts of development, there are a number of mechanisms that can be used. Firstly, maintenance or development of high-quality buffers in the riparian zones for existing and future agricultural development. This is key to reducing inputs such as sediment and nutrients. In some areas, engineering works may be needed to reduce gully erosion, but as the areas involved are vast, there would need to be prioritisation of these areas, as has been demonstrated in GBR catchments (Doriean et al., 2021; Brooks et al., in press). Additionally, as many of these impacts are legacy effects, developing strategies for gaining funding are critical.

Enforcement of existing government regulations regarding the environmental impacts of current and future developments is critical.

1.7.2. Daly River, Northern Territory

The Daly River, NT, is one of the NT's iconic rivers and is under considerable pressure due to the exploitation of the surface and groundwater for irrigated agriculture. Historically, it has had similar pressures to the Flinders and Gilbert Rivers from extensive cattle grazing. The Daly River estuary, coastal floodplain and coast are less well studied than the Flinders and Gilbert Rivers and differ from these two systems in 1) having perennial river flow rather than intermittent flow, and 2) being a macrotidal system rather than mesotidal.

Grazing has resulted in erosion across the catchment due to loss of vegetation cover, and development of gully and riverbank erosion (Appendix Table 1, Figure 1.8). The eroded sediment and associated nutrients are transported into estuaries, coastal floodplains and the coast. However, relative to the natural erosive processes in the estuary which are widening the estuary, these inputs are relatively minor. The nutrients transported to the estuary along with the sediment are key to stimulating primary production with flow-on effects to higher trophic levels.

Significant irrigated agricultural development is already occurring, which is likely to increase water extraction and reduce sediment and nutrient transport, with the likely effect of reducing primary and secondary production and decreasing the catchment of commercial and recreational fish species (Appendix Table 1, Figure 1.8).

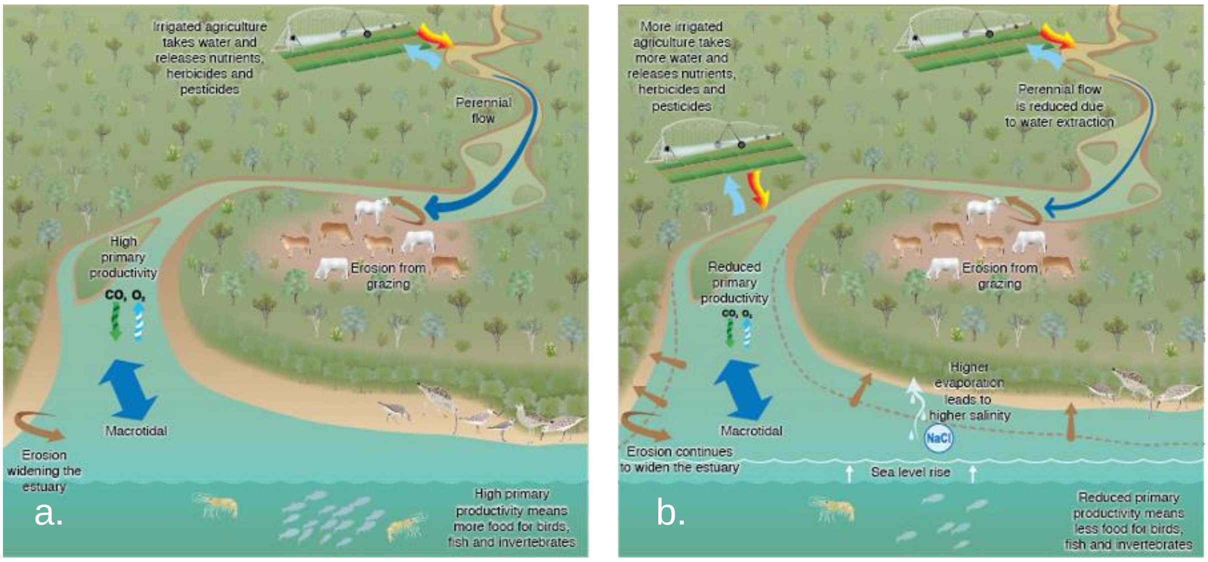


Figure 1-8 a. Current and b. future potential pressures and impacts of development on estuaries, coastal floodplains and the coast of the Daly River system.

In the longer team, if the scale of irrigated agriculture becomes sufficiently large, nutrient, pesticide and herbicide additions will begin to impact the rivers, and potentially downstream in estuaries, as has been seen in the Great Barrier Reef catchments (Appendix Table 1). This being the case, there is the potential for both stimulation of primary productivity, from nutrients, but also suppression of primary production, particularly from herbicides.

In terms of mitigation of impacts of development, there are a number of mechanisms that can be used. Firstly, maintenance or development of high-quality buffers in the riparian zones for existing and future agricultural development. This is key to reducing inputs such as sediment and nutrients. In some areas, engineering works may be needed to reduce gully erosion but as the areas involved are vast, there would need to be prioritisation of these areas. Additionally, as many of these impacts are legacy effects, developing strategies for gaining funding are critical.

Enforcement of existing government regulations regarding the environmental impacts of current and future developments is critical.

1.7.3. Keep River, Western Australia

The Keep River, WA, shares some similarities with the Flinders, Gilbert and Daly River systems in that it has extensive cattle grazing and likely has the same challenges in terms of erosion and sediment loads to the estuary. Like the Daly River estuary, it has a macrotidal system. The same mitigation strategies outlined for the Flinders, Gilbert and Daly rivers are also likely to be effective in the Keep River.

In terms of irrigated agriculture, the Keep River already receives discharge water from the Goomig Irrigation area in the Kimberley (Appendix Table 1, Figure 1.9). It differs from the NT and QLD rivers in that water is not extracted from the Keep River for irrigation, as water is taken from the Ord River. Findings from water quality monitoring have raised concerns about the potential impacts of pollutants from irrigation. To date, it appears that this is primarily in the river rather than the estuary, but there is potential for pollutants, e.g. pesticides and herbicides, to impact the estuary.

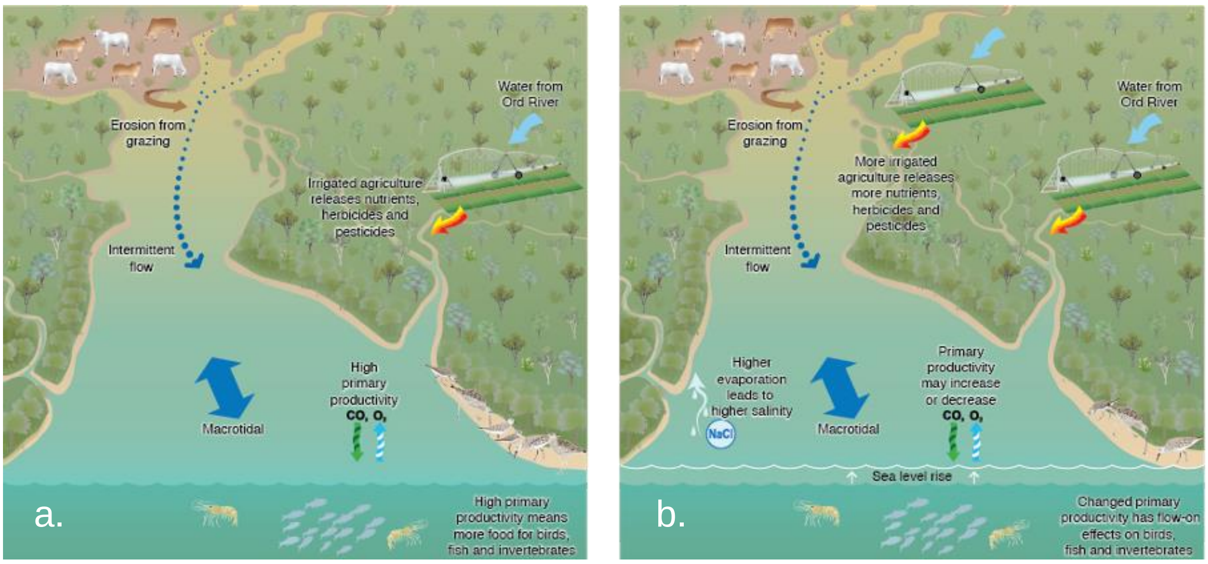


Figure 1-9 a. Current and b. future potential pressures and impacts of development on estuaries, coastal floodplains and the coast of the Keep River system.

Table 1.1. Summary of development types and their impact on the four estuaries in our study.

Development type	Pathway to effect	Impact	Flinders / Gilbert	Daly	Keep
Cattle grazing	Erosion	Changes to estuary/floodplain geomorphology, increase in particulate nutrients Sediment smothering seagrass beds	Significant erosion already occurring	Significant erosion already occurring	Significant erosion already occurring
Irrigated agriculture	Increased nutrient & pesticide/ herbicide loads	At low levels, nutrients increase estuarine productivity At higher levels, nutrients and pesticides/herbicides negatively impact estuarine ecosystem health	Little development to date Significant development proposed	Area of irrigated agriculture increasing rapidly	Already has a significant area of irrigated agriculture
	Reduced freshwater flow to estuary	Reduced nutrient inputs decrease estuarine and coastal productivity Reduced inundation of coastal floodplain			
Mining	Potential for pollutants to enter waterways	Negative impact on ecosystem health	Mining activity increasing	Some mining activity	Little mining activity
Urban	Increased pollutant loads	Pollutants may negatively impact estuarine ecosystem health	Currently little expansion of urban centres	Few towns and little expansion proposed	Almost no towns and little expansion proposed

1.8. Future research

There have been a number of studies on catchment impacts on estuaries and the coast, with far fewer studies on coastal floodplains. Despite this, there has been limited understanding and very little integration of geomorphic, hydrological and biochemical drivers within and between catchments. This means that it is difficult to assess the scale of development that is likely to have significant ecosystem impacts, e.g. thresholds or tipping points. Additionally, synergistic effects are poorly understood, such as catchment development combined with changes in climate-related parameters. Predictive models do exist for some regions but are often not validated with field data and, as such, may have high levels of uncertainty. Therefore, in order to aid decision-making on the scale and type of catchment development that is likely to be sustainable, a more integrated approach to research on catchments is needed.

1.9. References

ACLUMP., 2016. Australian Collaborative Land Use and Management Program. https://www.agriculture.gov.au/sites/default/files/abares/aclump/documents/Land%20use%20in%20Australia%20at%20a%20glance%202016.pdf

Adame, M.F., Neil, D., Wright, S., Lovelock, C.E., 2010a. Sedimentation within and among mangrove forests along a gradient of geomorphological settings. Estuarine, Coastal and Shelf Science, 86, 21–30.

Alongi, D.M., McKinnon, A.D., 2005. The cycling and fate of terrestrially-derived sediments and nutrients in the coastal zone of the Great Barrier Reef shelf. Marine Pollution Bulletin, 51, 239–252.

Alvarez-Romero, J.G., Devlin, M., Teixeira da Silva, E., Petus, C., Ban, N.C., Pressey, R.L., Kool, J., Roberts, J.J., Cerdeira-Estrada, S., Wenger, A.S., & Brodie, J., 2013. A novel approach to model exposure of coastal-marine ecosystems to riverine flood plumes based on remote sensing techniques. Journal of Environmental Management, 119, 194–207. https://doi.org/10.1016/j.jenvman.2013.01.036

Andutta, F.P., Ridd, P. V, Wolanski, E., 2011. Dynamics of hypersaline coastal waters in the Great Barrier Reef. https://doi.org/10.1016/j.ecss.2011.06.009

Angly, F.E., Heath, C., Morgan, T.C., Tonin, H., Rich, V., Schaffelke, B., Bourne, D.G., Tyson, G.W., 2016. Marine microbial communities of the Great Barrier Reef lagoon are influenced by riverine floodwaters and seasonal weather events. PeerJ, 2016(1). https://doi.org/10.7717/peerj.1511

Asbridge, E., Lucas, R., Accad, A., Dowling, R., 2015. Mangrove Response to Environmental Changes Predicted Under Varying Climates: Case Studies from Australia. Current Forestry Reports 1, 178–194. https://doi.org/10.1007/s40725-015-0018-4

Australian Government, 2015. Our North, Our Future: White Paper on Developing Northern Australia. https://infrastructure.gov.au/sites/default/files/documents/nawp-fullreport.pdf

Bainbridge, Z.T., Brodie, J.E., Faithful, J.W., Sydes, D.A., Lewis, S.E., 2009. Identifying the land-based sources of suspended sediments, nutrients and pesticides discharged to the Great Barrier Reef from the Tully–Murray Basin, Queensland, Australia. Marine and Freshwater Research, 60(11), 1081–1090. https://doi.org/10.1071/MF08333

Bainbridge, Z.T., Wolanski, E., Álvarez-Romero, J.G., Lewis, S.E., Brodie, J.E., 2012. Fine sediment and nutrient dynamics related to particle size and floc formation in a Burdekin River flood plume, Australia. *Marine Pollution Bulletin*, *65*(4–9), 236–248. https://doi.org/10.1016/J.MARPOLBUL.2012.01.043

Bainbridge, Z., Lewis, S., Bartley, R., Fabricius, K., Collier, C., Waterhouse, J., Garzon-Garcia, A., Robson, B., Burton, J., Wenger, A., Brodie, J., 2018. Fine sediment and particulate organic matter: A review and case study on ridge-to-reef transport, transformations, fates, and impacts on marine ecosystems. Marine Pollution Bulletin, 135, 1205–1220. https://doi.org/10.1016/j.marpolbul.2018.08.002

Bainbridge, Z., Lewis, S., Stevens, T., Petus, C., Lazarus, E., Gorman, J., Smithers, S., 2021. Measuring sediment grain size across the catchment to reef continuum:

Improved methods and environmental insights. Marine Pollution Bulletin, 168. https://doi.org/10.1016/j.marpolbul.2021.112339

Baird, M.E., Mongin, M., Skerratt, J., Margvelashvili, N., Tickell, S., Steven, A.D.L., Robillot, C., Ellis, R., Waters, D., Kaniewska, P., Brodie, J., 2021. Impact of catchment-derived nutrients and sediments on marine water quality on the Great Barrier Reef: An application of the eReefs marine modelling system. Marine Pollution Bulletin, 167. https://doi.org/10.1016/j.marpolbul.2021.112297

Bartley, R., Bainbridge, Z.T., Lewis, S.E., Kroon, F.J., Wilkinson, S.N., Brodie, J.E., Silburn, D.M., 2014a. Relating sediment impacts on coral reefs to watershed sources, processes and management: A review. In Science of the Total Environment (Vols. 468–469, pp. 1138–1153). Elsevier. https://doi.org/10.1016/j.scitotenv.2013.09.030

Bayliss, P., Kennard, M., Bartolo, R., Close, P., 2011. An assessment of risks to aquatic ecosystems in northern Australia from development and sea level rise threats Final report for NAWFA-Chapter 4.

Beck, H.E., Zimmermann, N.E., Mcvicar, T.R., Vergopolan, N., Berg, A., Wood, E.F., 2018. Data Descriptor: Present and future Köppen-Geiger climate classification maps at 1-km resolution. https://doi.org/10.1038/sdata.2018.214

Bennett, D., George, R., 2014. ‘Goomig Farmlands development: baseline water quality in the lower Keep River’, Resource management technical report 393, Department of Agriculture and Food, Western Australia, Perth.

Benz, S.A., Irvine, D.J., Rau, G.C., Bayer, P., Menberg, K., Blum, P., Jamieson, R.C., Griebler, C., Kurylyk, B.L., 2024. Global groundwater warming due to climate change. Nature Geoscience 2024 17:6, 17(6), 545–551. https://doi.org/10.1038/s41561-024-01453-x

Broadley, A., Stewart-Koster, B., Kenyon, R.A., Burford, M.A., & Brown, C.J., 2020. Impact of water development on river flows and the catch of a commercial marine fishery. Ecosphere, 11(7). https://doi.org/10.1002/ECS2.3194

Broadley, A., Stewart-Koster, B., Burford, M.A., Brown, C.J., 2022. A global review of the critical link between river flows and productivity in marine fisheries. Rev Fish Biol Fish 32, 805–825. https://doi.org/10.1007/s11160-022-09711-0

Brodie, J.E., Mitchell, A.W., 2005. Nutrients in Australian tropical rivers: changes with agricultural development and implications for receiving environments. Marine and Freshwater Research, 56, 279–302.

Brodie, J., Schroeder, T., Rohde, K., Faithful, J., Masters, B., Dekker, A., Brando, V., Maughan, M., 2010. Dispersal of suspended sediments and nutrients in the Great Barrier Reef lagoon during river-discharge events: Conclusions from satellite remote sensing and concurrent flood-plume sampling. Marine and Freshwater Research, 61(6), 651–664. https://doi.org/10.1071/MF08030

Brodie, J.E., Kroon, F.J., Schaffelke, B., Wolanski, E.C., Lewis, S.E., Devlin, M.J., Bohnet, I.C., Bainbridge, Z.T., Waterhouse, J., Davis, A.M., 2012. Terrestrial pollutant runoff to the Great Barrier Reef: An update of issues, priorities and management responses. https://doi.org/10.1016/j.marpolbul.2011.12.012

Brodie, J., Burford, M., Davis, A., da Silva, E., Devlin, M., Furnas, M., Kroon, F., Lewis, S., Lønborg, C., O'Brien, D., Schaffelke, B., Bainbridge, Z., 2015. The relative risks to water quality from particulate nitrogen discharged from rivers to the Great Barrier Reef in comparison to other forms of nitrogen. Centre for Tropical Water & Aquatic Ecosystem Research, James Cook University, Townsville. 100 pp.

Brodie, J.E., Lewis, S.E., Collier, C.J., Wooldridge, S., Bainbridge, Z.T., Waterhouse, J., Rasheed, M.A., Honchin, C., Holmes, G., Fabricius, K., 2017. Setting ecologically relevant targets for river pollutant loads to meet marine water quality requirements for the Great Barrier Reef, Australia: A preliminary methodology and analysis. Ocean & Coastal Management, 143, 136-147.

Brooks, A.P., Spencer, J., Doriean, N.J., Thwaites, R., Daley, J., Pietsch, T., Stout, J. in press. The effectiveness of alluvial gully remediation in Great barrier reef catchments. *International Soil and Water Conservation Research*. July 15, 2024 https://www.sciencedirect.com/science/article/pii/S2095633924000510

Brooks, A.P., Shellberg, J.G., Knight, J., Spencer, J., 2009. Alluvial gully erosion: An example from the Mitchell fluvial megafan, Queensland, Australia. *Earth Surface Processes and Landforms*, *34*(14), 1951–1969. https://doi.org/10.1002/ESP.1883

Brooks, A., Spencer, J., Olley, J., Pietsch, T., Borombovits, D., Curwen, G., Shellberg, J., Howley, C., Gleeson, A., Simon, A., Bankhead, N., Klimetz, D., Eslami-Endargoli, L., Bourgeault, A., 2013. Sediment Sources, Sinks & Drivers on the Cape York Savannah An Empirically-based Sediment Budget for the Normanby Basin. http://www.reefrescueresearch.com.au

Brooks, A., Spencer, J., 2016. Developing guidelines for agriculture development in northern Australia to avoid and reduce accelerated erosion. The case of the Gilbert/Etheridge River Integrated Agriculture Project.

Burford, M.A., Alongi, D.M., McKinnon, A.D., Trott, L.A., 2008. Primary production and nutrients in a tropical macrotidal estuary, Darwin Harbour, Australia. https://doi.org/10.1016/j.ecss.2008.04.018

Burford, M.A., Revill, A.T., Palmer, D.W., Clementson, L., Robson, B.J., Webster, I.T., 2011. River regulation alters drivers of primary productivity along a tropical river-estuary system. Marine and Freshwater Research, 62(2) https://doi.org/10.1071/MF10224

Burford, M.A., Revill, A.T., Smith, J., Clementson, L., 2012. Effect of sewage nutrients on algal production, biomass and pigments in tropical tidal creeks. Marine Pollution Bulletin, 64(12), 2671–2680. https://doi.org/10.1016/j.marpolbul.2012.10.008

Burford, M.A., Valdez, D., Curwen, G., Faggotter, S.J., Ward, D.P., O'Brien, K.R., 2016. Inundation of saline supratidal mudflats provides an important source of carbon and nutrients in an aquatic system. Mar Ecol Prog Ser 545. https://doi.org/10.3354/meps11621

Burford, M.A., Faggotter, S.J., 2021. Comparing the importance of freshwater flows driving primary production in three tropical estuaries. Mar Pollut Bull 169, 112565. https://doi.org/https://doi.org/10.1016/j.marpolbul.2021.112565

Caitcheon, G.G., Olley, J.M., Pantus, F., Hancock, G., Leslie, C., 2012. The dominant erosion processes supplying fine sediment to three major rivers in tropical Australia, the Daly (NT), Mitchell (Qld) and Flinders (Qld) Rivers. *Geomorphology*, *151–152*, 188–195. https://doi.org/10.1016/j.geomorph.2012.02.001

Carter, A., Taylor, H., Rasheed, M., 2014. Torres Strait Mapping: Seagrass Consolidation, 2002-2014, TropWATER Report no. 14/55. Centre for Tropical Water & Aquatic Ecosystem Research, James Cook University, Cairns. 48 pp.

Carter, A., Taylor, H., Rasheed, M., 2015. Torres Strait Dugong Sanctuary – Deepwater Seagrass Monitoring 2010-2015. JCU Publication, Report no. 15/24. Centre for Tropical Water & Aquatic Ecosystem Research, Cairns. 24 pp.

Close, P.G., Dobbs, R., Davies, P., 2012. Summary Report – Assessment of the likely impacts of development and climate change on aquatic ecological assets in Northern Australia. A report for the National Water Commission, Australia. Tropical Rivers and Coastal Knowledge (TRaCK) Commonwealth Environmental Research Facility, Charles Darwin University, Darwin.

Coles, R.G., Rasheed, M.A., McKenzie, L.J., Grech, A., York, P.H., Sheaves, M., McKenna, S., Bryant, C., 2015. The Great Barrier Reef World Heritage Area seagrasses: Managing this iconic Australian ecosystem resource for the future. Estuarine, Coastal and Shelf Science, 153, A1–A12. https://doi.org/10.1016/j.ecss.2014.07.020

Connolly, N.M., Pearson, R.G., Loong,D., Maughan, M., Brodie, J., 2015. Water quality variation along streams with similar agricultural development but contrasting riparian vegetation. Agriculture, Ecosystems & Environment, 213, 11–20. https://doi.org/10.1016/J.AGEE.2015.07.007

Crook, D., Wedd, D., Adair, B., King, A., Mooney, T., Harford, A., Humphrey, C., 2021. Fish migration in Magela Creek and potential impacts of mining-related solutes. Charles Darwin University, Darwin.

Cullen-Unsworth, L., Unsworth, R., 2013. Seagrass meadows, ecosystem services, and sustainability. Environment, 55(3), 14–28.

Dale, A.P., Turnour, J., Burns, S., Burford, M., Stewart-Koster, B., Waltham, N., Burrows, D., Douglas, M., Bock, E. and Baresi, U. 2024. Regional planning and assessment for sustainable development in the Gilbert River Catchment. Governance Systems Analysis. A report to the National Environmental Science Program. Reef and Rainforest Research Centre (RRRC), Cairns.

Davies, J.N., Rasheed, M.A., 2016. Port of Townsville Annual Seagrass Monitoring: September 2015. James Cook University Publication, Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER), Cairns. 50 pp.

Davis, A.M., Pearson, R.G., Brodie, J.E., Butler, B., 2017. Review and conceptual models of agricultural impacts and water quality in waterways of the Great Barrier Reef catchment area. Mar Freshw Res. https://doi.org/10.1071/MF15301

Delandmeter, P., Lewis, S.E., Lambrechts, J., Deleersnijder, E., Legat, V., Wolanski, E., 2015. The transport and fate of riverine fine sediment exported to a semi-open system. Estuarine Coastal and Shelf Science, 167, 336–346.

Devlin, M.J., McKinna, L.W., Álvarez-Romero, J.G., Petus, C., Abott, B., Harkness, P., Brodie, J., 2012. Mapping the pollutants in surface riverine flood plume waters in the

Great Barrier Reef, Australia. Marine Pollution Bulletin, 65(4–9), 224–235. https://doi.org/10.1016/J.MARPOLBUL.2012.03.001

Devlin, M.J., Teixeira Da Silva, E., Petus, C., Wenger, A., Zeh, D., Tracey, D., Álvarez-Romero, J.G., Brodie, J., 2013. Combining in-situ water quality and remotely sensed data across spatial and temporal scales to measure variability in wet season chlorophyll-a: Great Barrier Reef lagoon (Queensland, Australia). http://www.ecologicalprocesses.com/content/2/1/31

Devlin, M.J., Petus, C., da Silva, E., Tracey, D., Wolff, N.H., Waterhouse, J., Brodie, J., 2015a. Water quality and river plume monitoring in the Great Barrier Reef: An overview of methods based on ocean colour satellite data. Remote Sensing, 7(10), 12909–12941. https://doi.org/10.3390/rs71012909

Devlin, M., Lewis, S., Davis, A., Smith, R., Negri, A., Thompson, M., Poggio, M., 2015b. Advancing Our Understanding of the Source, Management, Transport and Impacts of Pesticides on the Great Barrier Reef 2011-2015. A report for the Queensland Department of Environment and Heritage Protection. Tropical Water & Aquatic Ecosystem Research, James Cook University, Townsville. 150 pp.

Doriean, N.J., Bennett, W.W., Spencer, J.R., Garzon-Garcia, A., Burton, J.M., Teasdale, P.R., Brooks, A.P. 2021. Intensive landscape-scale remediation improves water quality of an alluvial gully located in a Great Barrier Reef catchment. Hydrology and Earth System Sciences, 25(2), 867-883.

Douglas, M.M., Bunn, S.E., Davies, P.M., 2005. River and wetland food webs in Australia's wet-dry tropics: general principles and implications for management. Mar Freshw Res 56, 329–342. https://doi.org/10.1071/MF04084

Douglas, M.M., Jackson, S., Canham, C.A., Pusey, B.J., Loomes, R., Setterfield, S.A., 2019. Conceptualizing Hydro-socio-ecological Relationships to Enable More Integrated and Inclusive Water Allocation Planning. One Earth 1, 361–373. https://doi.org/10.1016/j.oneear.2019.10.021

Duggan, M., Connolly, R.M., Whittle, M., Curwen, G., Burford, M.A., 2014. Effects of freshwater flow extremes on intertidal biota of a wet-dry tropical estuary. Journal Title Marine Ecology Progress Series. MARINE ECOLOGY PROGRESS SERIES Mar Ecol Prog Ser 502, 11–23. https://doi.org/10.3354/meps10719

Duggan, M., Bayliss, P., Burford, M.A., 2019. Predicting the impacts of freshwater-flow alterations on prawn (Penaeus merguiensis) catches. Fish Res 215, 27–37. https://doi.org/10.1016/J.FISHRES.2019.02.013

Duke, N.C., Hutley, L.B., Mackenzie, J.R., Burrows, D. 2021. Processes and factors driving change in mangrove forests: An evaluation based on the mass dieback event in Australia's Gulf of Carpentaria. In *Ecosystem collapse and climate change*, pp. 221-264. Cham: Springer International Publishing.

Fabricius, K.E., Logan, M., Weeks, S., Brodie, J., 2014. The effects of river run-off on water clarity across the central Great Barrier Reef. Marine Pollution Bulletin, 84(1–2), 191–200. https://doi.org/10.1016/j.marpolbul.2014.05.012

Fabricius, K.E., Logan, M., Weeks, S.J., Lewis, S.E., Brodie, J., 2016. Changes in water clarity in response to river discharges on the Great Barrier Reef continental shelf:

2002–2013. Estuarine, Coastal and Shelf Science, 173, A1–A15. https://doi.org/10.1016/J.ECSS.2016.03.001

Feller, I.C., Dangremond, E.M., Devlin, D.J., Lovelock, C.E., Proffitt, C.E., Rodriguez, W., 2015a. Nutrient enrichment intensifies hurricane impact in scrub mangrove ecosystems in the Indian River Lagoon, Florida, USA. Ecology 96, 2960–2972. https://doi.org/10.1890/14-1853.1

Feller, I.C., Dangremond, E.M., Devlin, D.J., Lovelock, C.E., Proffitt, C.E., Rodriguez, W., 2015b. Nutrient enrichment intensifies hurricane impact in scrub mangrove ecosystems in the Indian River Lagoon, Florida, USA. Ecology 96, 2960–2972. https://doi.org/10.1890/14-1853.1

Fensham, R.J., 1996. Land clearance and conservation of inland dry rainforest in north Queensland, Australia. Biological Conservation, 75(3), 289–298. https://doi.org/10.1016/0006-3207(95)00057-7

Ferrier, R.C., Jenkins, A., Watson, I., Ash, A., Petheram, C., Barber, M., Stokes, C., 2021. Development in the Northern Rivers of Australia. Handbook of Catchment Management 2e 465–497. https://doi.org/10.1002/9781119531241.CH19

Fortune, J., Butler, E.C.V., Gibb, K., 2020. A decade of nitrogen inputs to a tropical macrotidal estuary of Northern Australia, Darwin Harbour. Regional Studies in Marine Science, 36, 101275. https://doi.org/10.1016/J.RSMA.2020.101275

Fortune, J., Kaestli, M., Butler, E.C.V., Gibb, K., 2022. Denitrification in intertidal sediments of a tropical estuary subject to increasing development pressures. 84, 53. https://doi.org/10.1007/s00027-022-00885-0

Fraser, G., Rohde, K., Silburn, M., 2017. Fertiliser management effects on dissolved inorganic nitrogen in runoff from Australian sugarcane farms. Environ Monit Assess, 189, 409. https://doi.org/10.1007/s10661-017-6115-z

French, V.A., Codi King, S., Kumar, A., Northcott, G., McGuinness, K., Parry, D., 2015. Characterisation of microcontaminants in Darwin Harbour, a tropical estuary of northern Australia undergoing rapid development. *The Science of the Total Environment*, *536*, 639–647. https://doi.org/10.1016/J.SCITOTENV.2015.07.114

Furnas, M., Mitchell, A., Skuza, M., Brodie, J., 2005. In the other 90%: Phytoplankton responses to enhanced nutrient availability in the Great Barrier Reef Lagoon. Marine Pollution Bulletin, 51(1–4), 253–265. https://doi.org/10.1016/j.marpolbul.2004.11.010

Gallen, C., Devlin, M., Thompson, K., Paxman, C., Mueller, J., 2014. Pesticide monitoring in inshore waters of the Great Barrier Reef using both time-integrated and event monitoring techniques (2013 - 2014). The University of Queensland, The National Research Centre for Environmental Toxicology (Entox).

Garzon-Garcia, A., Burton, J.M., Lewis, S., Bainbridge, Z., de Hayr, R., Moody, P., Brodie, J., 2021. The bioavailability of nitrogen associated with sediment in riverine plumes of the Great Barrier Reef. *Marine Pollution Bulletin*, *173*(Pt A). https://doi.org/10.1016/J.MARPOLBUL.2021.112910

Grech, A., Wolter, J., Coles, R., McKenzie, L., Rasheed, M., Thomas, C., Waycott, M., Hanert, E. 2016. Spatial patterns of seagrass dispersal and settlement. Diversity and Distributions, 22(11), 1150–1162. https://doi.org/10.1111/DDI.12479

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., Watson, R., 2008. A global map of human impact on marine ecosystems. Science (1979) 319, 948–952. https://doi.org/10.1126/SCIENCE.1149345

Harris, J.H., Kingsford, R.T., Peirson, W., Baumgartner, L.J., 2016. Mitigating the effects of barriers to freshwater fish migrations: the Australian experience. Marine and Freshwater Research, 68(4), 614–628. https://doi.org/10.1071/MF15284

Howley, C., Devlin, M., Burford, M., 2018. Assessment of water quality from the Normanby River catchment to coastal flood plumes on the northern Great Barrier Reef, Australia. Marine and Freshwater Research, 69(6), 859–873. https://doi.org/10.1071/MF17009

Howley, C., Shellberg, J., Olley, J., Brooks, A., Spencer, J., Burford, M., 2021. Sediment and nutrient sources and sinks in a wet-dry tropical catchment draining to the Great Barrier Reef. Marine Pollution Bulletin, 165, 112080. https://doi.org/10.1016/J.MARPOLBUL.2021.112080

Jones, A.M., Berkelmans, R., 2014. Flood Impacts in Keppel Bay, Southern Great Barrier Reef in the Aftermath of Cyclonic Rainfall. PLoS One 10;9(1):e84739. doi: 10.1371/journal.pone.0084739. PMID: 24427294; PMCID: PMC3888408

Jones, R., Fisher, R., Stark, C., Ridd, P., 2015a. Temporal Patterns in Seawater Quality from Dredging in Tropical Environments. PLOS ONE, 10(10), e0137112. https://doi.org/10.1371/JOURNAL.PONE.0137112

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

King, A.J., Townsend, S.A., Douglas, M.M., Kennard, M.J., 2015. Implications of water extraction on the low-flow hydrology and ecology of tropical savannah rivers: An appraisal for northern Australia. Freshwater Science, 34(2), 741–758. https://doi.org/10.1086/681302

King, A., Dostine, P., Crook, D., Keller, K., Schult, J., Waugh, P., Townsend, S., Tyler, K., Wedd, D., Roberts, B., Tjuwaliyn-Wagiman Aboriginal Corporation., 2021. Environmental water needs of the Daly River, Northern Territory Final report.

Knighton, A.D., Woodroffe, C.D., Mills, K., 1992. The evolution of tidal creek networks, mary river, northern Australia. Earth Surface Processes and Landforms, 17(2), 167–190. https://doi.org/10.1002/ESP.3290170205

Kroon, F.J., Kuhnert, P.M., Henderson, B.L., Wilkinson, S.N., Kinsey-Henderson, A., Abbott, B., Brodie, J.E., Turner, R.D.R., 2012. River loads of suspended solids, nitrogen, phosphorus and herbicides delivered to the Great Barrier Reef lagoon. https://doi.org/10.1016/j.marpolbul.2011.10.018

Lambert, V., Bainbridge, Z.T., Collier, C., Lewis, S.E., Adams, M.P., Carter, A., Saunders, M.I., Brodie, J., Turner, R.D.R., Rasheed, M.A., O'brien, K.R., 2021. Connecting targets for catchment sediment loads to ecological outcomes for seagrass using multiple lines of evidence. https://doi.org/10.1016/j.marpolbul.2021.112494

Lamontagne, S., Cook, P.G., O'Grady, A., Eamus, D., 2005. Groundwater use by vegetation in a tropical savanna riparian zone (Daly River, Australia). Journal of Hydrology, 310(1–4), 280–293. https://doi.org/10.1016/J.JHYDROL.2005.01.009

Leahy, S., Robins, J., 2021. River flows affect the growth of a tropical finfish in the wet-dry rivers of northern Australia, with implications for water resource development. Hydrobiologia 848. https://doi.org/10.1007/s10750-021-04641-7

Lear, K.O., Gleiss, A.C., Whitty, J.M., Fazeldean, T., Albert, J.R., Green, N., Ebner, B.C., Thorburn, D.C., Beatty, S.J., Morgan, D.L., 2019. Recruitment of a critically endangered sawfish into a riverine nursery depends on natural flow regimes. Sci Rep 9. https://doi.org/10.1038/s41598-019-53511-9

Lerat, J., Egan, C., Kim, S., Gooda, M., Loy, A., Shao, Q., Petheram, C., 2013. Calibration of river models for the Flinders and Gilbert catchments A technical report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, part of the North Queensland Irrigated Agriculture Strategy.

Lewis, S.E., Bainbridge, Z.T., Kuhnert, P.M., Sherman, B.S., Henderson, B., Dougall, C., Cooper, M., Brodie, J.E., 2013. Calculating sediment trapping efficiencies for reservoirs in tropical settings: A case study from the Burdekin Falls Dam, NE Australia. Water Resources Research, 49(2), 1017–1029. https://doi.org/10.1002/WRCR.20117

Lewis, S.E., Olley, J., Furuichi, T., Sharma, A., Burton, J., 2014. Complex sediment deposition history on a wide continental shelf: Implications for the calculation of accumulation rates on the Great Barrier Reef. *Earth and Planetary Science Letters*, *393*, 146–158. https://doi.org/10.1016/j.epsl.2014.02.038

Lewis S., Bartley R., Bainbridge Z., Wilkinson S., Burton J., Bui E., 2015. Burdekin sediment story. Report No. 1 for NO Dry Tropics NRM, Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) Publication, James Cook University, Townsville.

Lewis, S.E., Bartley, R., Wilkinson, S.N., Bainbridge, Z.T., Henderson, A.E., James, C.S., Irvine, S.A., Brodie, J.E., 2021. Land use change in the river basins of the Great Barrier Reef, 1860 to 2019: A foundation for understanding environmental history across the catchment to reef continuum. Marine Pollution Bulletin, 166, 112193. https://doi.org/10.1016/J.MARPOLBUL.2021.112193

Lønborg, C., Álvarez–Salgado, X.A., Duggan, S., Carreira, C., 2018. Organic matter bioavailability in tropical coastal waters: The Great Barrier Reef. Limnology and Oceanography, 63(2), 1015–1035. https://doi.org/10.1002/LNO.10717

Lovelock, C.E., Ball, M.C., Martin, K.C., Feller, I.C., 2009. Nutrient Enrichment Increases Mortality of Mangroves. PLoS One 4, 5600. https://doi.org/10.1371/journal.pone.0005600

Lovelock, C.E., Bennion, V., Grinham, A., Cahoon, D.R., 2011. The Role of Surface and Subsurface Processes in Keeping Pace with Sea Level Rise in Intertidal Wetlands of Moreton Bay, Queensland, Australia. Ecosystems 14, 745–757. https://doi.org/10.1007/s10021-011-9443-9

Lowe, V., Frid, C.L.J., Venarsky, M., Burford, M.A., 2022. Responses of a macrobenthic community to seasonal freshwater flow in a wet-dry tropical estuary. Estuar Coast Shelf Sci 265. https://doi.org/10.1016/j.ecss.2021.107736

Marion, G.S., Jupiter, S.D., Radice, V.Z., Albert, S., Hoegh-Guldberg, O., 2021. Linking isotopic signatures of nitrogen in nearshore coral skeletons with sources in catchment runoff. Marine Pollution Bulletin, 173(Pt B). https://doi.org/10.1016/J.MARPOLBUL.2021.113054

Marsh, H., O'Shea, T.J., Reynolds III, J. 2012. Ecology and conservation of the Sirenia : dugongs and manatees. Cambridge University Press.

Marshall, J.C., Blessing, J.J., Clifford, S.E., Negus, P.M., Steward, A.L., 2020. Epigeic invertebrates of pig-damaged, exposed wetland sediments are rooted: An ecological response to feral pigs (Sus scrofa). Aquatic Conservation: Marine and Freshwater Ecosystems, 30(12), 2207–2220. https://doi.org/10.1002/AQC.3468

Masters, B., Tahir, N., Fries, J., Mortimore, C., Enderlin, N., Armour, J., 2017. Paddock-scale water quality monitoring of nitrogen fertiliser management practices in sugarcane cropping: 2014–2016 technical report, Wet tropics Region. Department of Natural Resources and Mines, Brisbane, Qld.

McCloskey, G.L., Wasson, R. ., Boggs, G. ., Douglas, M., 2016. Timing and causes of gully erosion in the riparian zone of the semi-arid tropical Victoria River, Australia: Management implications. *Geomorphology*, *266*, 96–104. https://doi.org/10.1016/J.GEOMORPH.2016.05.009

McCloskey, G.L., Baheerathan, R., Dougall, C., Ellis, R., Bennett, F.R., Waters, D., Darr, S., Fentie, B., Hateley, L.R., & Askildsen, M. 2021. Modelled estimates of dissolved inorganic nitrogen exported to the Great Barrier Reef lagoon. Marine Pollution Bulletin, 171, 112655. https://doi.org/10.1016/j.marpolbul.2021.112655

McCook, L.J., Schaffelke, B., Apte, S.C., Brinkman, R., Brodie, J., Erftemeijer, P., Eyre, B., Hoogerwerf, F., Irvine, I., Jones, R., King, B., Marsh, H., Masini, R., Morton, R., Pitcher, R., Rasheed, M., Sheaves, M., Symonds, A., Warne, M.St.J., 2015. Synthesis of current knowledge of the biophysical impacts of dredging and disposal on the Great Barrier Reef: Report of an Independent Panel of Experts, Great Barrier Reef Marine Park Authority, Townsville.

McKee, K.L., Cahoon, D.R., Feller, I.C., 2007. Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. Global Ecology and Biogeography 16, 545–556. https://doi.org/10.1111/j.1466-8238.2007.00317

Mckenzie, L., Collier, C., Waycott, M., Unsworth, R., Yoshida, R., Smith, N., 2012. Monitoring inshore seagrasses of the GBR and responses to water quality (Issue 13). www.seagrasswatch.org

McKenzie, L.J., Collier, C.J., Langlois, L.A., Yoshida, R.L., Smith, N., Waycott, M., 2016. Marine Monitoring Program—Inshore Seagrass, Annual Report for the sampling period 1st June 2014 – 31st May 2015. Centre for Tropical Water & Aquatic Ecosystem Research, James Cook University, Cairns.

Meager, J.J., Limpus, C.J., 2012. Marine wildlife stranding and mortality database annual report 2011. III. Marine Turtle. Conservation Technical and Data Report, 2012(3), 1–46.

Mercurio, P., Flores, F., Mueller, J.F., Carter, S., Negri, A.P., 2014. Glyphosate persistence in seawater. Marine Pollution Bulletin, 85(2), 385–390. https://doi.org/10.1016/j.marpolbul.2014.01.021

Mercurio, P., Mueller, J.F., Eaglesham, G., Flores, F., Negri, A.P., 2015. Herbicide persistence in seawater simulation experiments. PLoS ONE, 10(8). https://doi.org/10.1371/journal.pone.0136391

Meynecke, J.-O., Lee, S.Y., Duke, N.C., 2008. Linking spatial metrics and fish catch reveals the importance of coastal wetland connectivity to inshore fisheries in Queensland, Australia. Biol Conserv 141, 981–996. https://doi.org/10.1016/j.biocon.2008.01.018

Mihailou, H., Massaro, M., 2021. An overview of the impacts of feral cattle, water buffalo and pigs on the savannas, wetlands and biota of northern Australia. Austral Ecology, 46(5), 699–712. https://doi.org/10.1111/AEC.13046

Mitchell, A., Reghenzani, J., Faithful, J., Furnas, M., Brodie, J., Mitchell, A., Reghenzani, J., Faithful, J., Furnas, M., Brodie, J., 2009. Relationships between land use and nutrient concentrations in streams draining a 'wet-tropics' catchment in northern Australia. Marine and Freshwater Research, 60(11), 1097–1108. https://doi.org/10.1071/MF08330

Munksgaard, N.C., Hutley, L.B., Metcalfe, K.N., Padovan, A.C., Palmer, C., Gibb, K.S. 2019. Environmental challenges in a near-pristine mangrove estuary facing rapid urban and industrial development: Darwin Harbour, Northern Australia. In Regional Studies in Marine Science (Vol. 25). Elsevier B.V. https://doi.org/10.1016/j.rsma.2018.11.001

Mulrennan, M. E., & Woodroffe, C. D. (1998). Saltwater intrusion into the coastal plains of the Lower Mary River, Northern Territory, Australia. *Journal of Environmental Management*, *54*(3), 169–188. https://doi.org/10.1006/JEMA.1998.0229

Murphy, T., Dougall, C., Burger, P., Carroll, C., 2013. Runoff water quality from dryland cropping on Vertisols in Central Queensland, Australia. Ecosystems and Environment 180, 21–28. https://doi.org/10.1016/j.agee.2011.07.023

Northern Territory Government, 2023. Adelaide River Off-stream Water Storage project [WWW Document]. URL https://watersecurity.nt.gov.au/darwin-region-water-supply/adelaide-river-off-stream-water-storage-project.

Novak, P.A., Garcia, E.A., Pusey, B.J., Douglas, M.M. 2017. Importance of the natural flow regime to an amphidromous shrimp: A case study. Marine and Freshwater Research, 68(5), 909–921. https://doi.org/10.1071/MF16034

O'Brien, D., Lewis, S., Gallen, C., O'Brien, J., Thompson, K., Eaglesham, G., Mueller, J., 2014. Barron River pesticide monitoring and Cairns WWTP WQ assessment. 14/40, Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) Publication, James Cook University, Cairns.

Oke, P.R., Larnicol, G., Jones, E.M., Kourafalou, V., Sperrevik, A.K., Carse, F., Tanajura, C. A.S., Mourre, B., Tonani, M., Brassington, G.B., Le Henaff, M., Halliwell, G.R., Atlas, R., Moore, A.M., Edwards, C.A., Martin, M.J., Sellar, A.A., Alvarez, A., Demey, P., Iskandarani, M., 2015. Assessing the impact of observations on ocean forecasts and reanalyses: Part 2, regional applications. Journal of Operational Oceanography, 8, s63–s79. https://doi.org/10.1080/1755876X.2015.1022080/ASSET/83507201-FEF1-4423-99C2-03A9588FCB12/ASSETS/IMAGES/TJOO_A_1022080_F0010_C.JPG

Oubelkheir, K., Ford, P.W., Cherukuru, N., Clementson, L. A., Petus, C., Devlin, M., Schroeder, T., Steven, A.D.L., 2023. Impact of a Tropical Cyclone on Terrestrial Inputs and Bio-Optical Properties in Princess Charlotte Bay (Great Barrier Reef Lagoon). Remote Sensing, 15(3). https://doi.org/10.3390/rs15030652

O'mara, K., Beesley, L.S., Kopf, R.K., Burford, M.A., Douglas, M.M., Stewart-Koster, B., Kennard, M.J., 2023. Water abstraction impacts on flow dependent fisheries species of the Northern Territory, Australia - a synthesis of current knowledge and future research needs. FRDC Project No. 2021-114.

Parslow, J., Margvelashvili, N., Palmer, D., Revill, A.T., Robson, B., Sakov, P., Volkman, J.K., Watson, R., Webster, I., 2003. The Response of the Lower Ord River and Estuary to Management of Catchment Flows and Sediment and Nutrient Loads.

Pearson, R.G., Connolly, N.M., Davis, A.M., Brodie, J.E., 2021. Fresh waters and estuaries of the Great Barrier Reef catchment: Effects and management of anthropogenic disturbance on biodiversity, ecology and connectivity. Mar Pollut Bull 166, 112194. https://doi.org/10.1016/J.MARPOLBUL.2021.112194

Perez, K., Rodgers, K.S., Jokiel, P.L., Lager, C.V., Lager, D.J., 2014. Effects of terrigenous sediment on settlement and survival of the reef coral Pocillopora damicornis. PeerJ, 2014(1), e387. https://doi.org/10.7717/PEERJ.387/FIG-5

Petheram, C., Yang, A., 2013. Climate data and their characterisation for hydrological and agricultural scenario modelling across the Flinders and Gilbert catchments. A technical report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, part of the North Queensland Irrigated Agriculture Strategy. CSIRO Water for a Healthy Country and Sustainable Agriculture flagships, Australia.

Petus, C., Collier, C., Devlin, M., Rasheed, M., McKenna, S. 2014. Using MODIS data for understanding changes in seagrass meadow health: A case study in the Great Barrier Reef (Australia). Marine Environmental Research, 98, 68–85. https://doi.org/10.1016/J.MARENVRES.2014.03.006

Petus, C., Devlin, M., Thompson, A., McKenzie, L., da Silva, E.T., Collier, C., Tracey, D., Martin, K. 2016. Estimating the Exposure of Coral Reefs and Seagrass Meadows to Land-Sourced Contaminants in River Flood Plumes of the Great Barrier Reef: Validating a Simple Satellite Risk Framework with Environmental Data. Remote Sensing 2016, Vol. 8, Page 210, 8(3), 210. https://doi.org/10.3390/RS8030210

Philip, S., Davies, P., Hughes, J., Karim, F., Marvanek, C., Petheram, C., Taylor, A., Ticehurst, C., Vanderzalm, J., Wang, B., Watson, I., 2018. Chapter 2: physical environment of the Mitchell catchment C. Petheram, I. Watson, C. Bruce, C. Chilcott (Eds.), Water Resource Assessment for the Mitchell Catchment. A Report to the Australian Government From the CSIRO Northern Australia Water Resource Assessment, Part of the National Water Infrastructure Development Fund: Water Resource Assessments, CSIRO, Australia (2018).

Plagányi, É., Kenyon, R., Blamey, L., Robins, J., Burford, M., Pillans, R., Hutton, T., Hughes, J., Kim, S., Deng, R.A., Cannard, T., Jarrett, A., Laird, A., Lawrence, E., Miller, M., Moeseneder, C., 2023. Integrated assessment of river development on downstream marine fisheries and ecosystems. Nat Sustain 7, 31–44. https://doi.org/10.1038/s41893-023-01238-x

Pratt, C., Lottermoser, B.G., 2007. Mobilisation of traffic-derived trace metals from road corridors into coastal stream and estuarine sediments, Cairns, northern Australia. Environmental Geology, 52(3), 437–448. https://doi.org/10.1007/S00254-006-0471-2

Preen, A., Marsh, H. 1995. Response of dugongs to large-scale loss of seagrass from Hervey Bay, Queensland Australia. Wildlife Research, 22(4), 507–519. https://doi.org/10.1071/WR9950507

Prosser, I.P., Wilkinson, S.N., 2024. Question 4.4 How much anthropogenic dissolved nutrient (nitrogen and phosphorus species) is exported from Great Barrier Reef catchments? In Waterhouse J, Pineda M-C, Sambrook K (Eds) 2022 Scientific Consensus Statement on land-based impacts on Great Barrier Reef water quality and ecosystem condition. Commonwealth of Australia and Queensland Government.

Rasheed, M.A., McKenna, S.A., Carter, A.B., Coles, R.G. 2014. Contrasting recovery of shallow and deep water seagrass communities following climate associated losses in tropical north Queensland, Australia. Marine Pollution Bulletin, 83(2), 491–499. https://doi.org/10.1016/J.MARPOLBUL.2014.02.013

Reef, R., Feller, I.C., Lovelock, C.E., 2010. Nutrition of mangroves. Tree Physiology, 30, 1148–1160. doi:10.1093/treephys/tpq048

Ricardo, G.F., Jones, R.J., Negri, A.P., Stocker, R., 2016. That sinking feeling: Suspended sediments can prevent the ascent of coral egg bundles. Scientific Reports 2016 6:1, 6(1), 1–7. https://doi.org/10.1038/srep21567

Robins, J.B., Halliday, I.A., Staunton-Smith, J., Mayer, D.G., Sellin, M.J., 2005. Freshwater-flow requirements of estuarine fisheries in tropical Australia: A review of the state of knowledge and application of a suggested approach, in: Marine and Freshwater Research. pp. 343–360. https://doi.org/10.1071/MF04087

Scott, P.D., Bartkow, M., Blockwell, S.J., Coleman, H.M., Khan, S.J., Lim, R., McDonald, J.A., Nice, H., Nugegoda, D., Pettigrove, V., Tremblay, L.A., Warne, M.S.J., Leusch, F.D.L., 2014. A national survey of trace organic contaminants in Australian rivers. Journal of Environmental Quality, 43(5), 1702–1712.

Sheaves, M., 2005. Nature and consequences of biological connectivity in mangroves systems. Mar Ecol Prog Ser 302, 293–305

Sheaves, M., Baker, R., Nagelkerken, I., Connolly, R.M., 2015. True Value of Estuarine and Coastal Nurseries for Fish: Incorporating Complexity and Dynamics. Estuaries and Coasts 38, 401–414. https://doi.org/10.1007/s12237-014-9846-x

Sheaves, M., Johnston, R., 2008. Influence of marine and freshwater connectivity on the dynamics of subtropical estuarine wetland fish metapopulations. Mar Ecol Prog Ser 357, 225–243. https://doi.org/10.3354/meps07292

Shellberg, J.G., Spencer, J., Brooks, A.P., & Pietsch, T.J., 2016. Degradation of the Mitchell River fluvial megafan by alluvial gully erosion increased by post-European land use change, Queensland, Australia. Geomorphology, 266, 105–120. https://doi.org/10.1016/J.GEOMORPH.2016.04.021

Shishaye, H.A., Tait, D.R., Befus, K.M., Maher, D.T., Reading, M.J., 2020. New insights into the hydrogeology and groundwater flow in the Great Barrier Reef catchment, Australia, revealed through 3D modelling. https://doi.org/10.1016/j.ejrh.2020.100708

Smart, J., Mcmahon, J., Curwen, G., Burford, M., 2021. Estimating the economic impact of potential extractions from Gulf rivers on the banana prawn fishery Final report (component 4).

Smith, J., Burford, M.A., Revill, A.T., Haese, R.R., Fortune, J., 2012. Effect of nutrient loading on biogeochemical processes in tropical tidal creeks. Biogeochemistry, 108(1–3), 359–380. https://doi.org/10.1007/s10533-011-9605-z

Sozou, A., Scott, E., Rasheed, M., 2016. Long-term Seagrass Monitoring in the Port of Thursday Island: March 2016. TropWATER Report no. 16/29. James Cook University, Centre for Tropical Water & Aquatic Ecosystem Research, Cairns, 45 pp.

Tait, D.R., Santos, I.R., Lamontagne, S., Sippo, J.Z., McMahon, A., Jeffrey, L.C., Maher, D.T., 2023. Submarine Groundwater Discharge Exceeds River Inputs as a Source of Nutrients to the Great Barrier Reef. Environ Sci Technol 57, 15627–15634. https://doi.org/10.1021/ACS.EST.3C03725/SUPPL_FILE/ES3C03725_SI_001.PDF

Taylor, M.P., Mackay, A., Kuypers, T., Hudson-Edwards, K., 2009. Mining and urban impacts on semi-arid freshwater aquatic systems: the example of Mount Isa, Queensland. Journal of Environmental Monitoring, 11(5), pp.977-986.

Taylor, M.P., Little, J. A., 2013. Environmental impact of a major copper mine spill on a river and floodplain system. Anthropocene, 3, 36–50. https://doi.org/10.1016/J.ANCENE.2014.02.004

Thorburn, P.J., Wilkinson, S.N., 2013. Conceptual frameworks for estimating the water quality benefits of improved agricultural management practices in large catchments. Agriculture, Ecosystems & Environment, 180, 192–209. https://doi.org/10.1016/J.AGEE.2011.12.021

Townsend, S.A., Padovan, A.V., 2005. The seasonal accrual and loss of benthic algae (Spirogyra) in the Daly River, an oligotrophic river in tropical Australia. Marine and Freshwater Research, 56, 317-327. http://dx.doi.org/10.1071/MF04079

Turschwell, M.P., Stewart-Koster, B., Kenyon, R., Deng, R.A., Stratford, D., Hughes, J.D., Pollino, C.A., 2022. Spatially structured relationships between white banana prawn (Penaeus merguiensis) catch and riverine flow in the Northern Prawn Fishery, Australia. J Environ Manage 319. https://doi.org/10.1016/j.jenvman.2022.115761

Valiela, I., Cole, M.L., 2002. Comparative evidence that salt marshes and mangroves may protect seagrass meadow from land-derived nitrogen loads. Ecosystems 5, 92–1023.

van Dam, R.A., Humphrey, C.L., & Martin, P., 2002. Mining in the Alligator Rivers Region, northern Australia: Assessing potential and actual effects on ecosystem and human health. Toxicology, 181–182, 505–515. https://doi.org/10.1016/S0300-483X(02)00470-5

Vance, J., Rothlisberg, P.C., 2020. The biology and ecology of the banana prawns: Penaeus merguiensis de Man and P. indicus H. Milne Edwards. Advanced Marine Biology 1–139. https://doi.org/10.1016/bs.amb.2020.04.001

Verhoeven, J.T.A., Arheimer, B., Yin, C., Hefting, M.M., 2006. Regional and global concerns over wetlands and water quality. Trends in Ecology & Evolution, 21, 96–103. doi:10.1016/j.tree.2005.11.015

Waltham, N.J., Sheaves, M., 2015. Expanding coastal urban and industrial seascape in the Great Barrier Reef World Heritage Area: Critical need for coordinated planning and policy. Marine Policy, 57, 78–84. https://doi.org/10.1016/J.MARPOL.2015.03.030W

Waltham, N.J., Sheaves, M., 2017. Acute thermal tolerance of tropical estuarine fish occupying a man-made tidal lake, and increased exposure risk with climate change. *Estuarine, Coastal and Shelf Science*, *196*, 173–181. https://doi.org/10.1016/J.ECSS.2017.06.032

Waltham, N.J., Schaffer,J., 2021. Will fencing floodplain and riverine wetlands from feral pig damage conserve fish community values? Ecology and Evolution, 11(20), 13780–13792. https://doi.org/10.1002/ECE3.8054

Waltham, N.J., Pyott, M., Buelow, C., Wearne, L., 2020a. Mechanical harvester removes invasive aquatic weeds to restore water quality and fish habitat values on the Burdekin floodplain. Ecological Management & Restoration, 21(3), 187–197. https://doi.org/10.1111/EMR.12427

Waltham, N.J., Coleman, L., Buelow, C., Fry, S., Burrows, D., 2020. Restoring fish habitat values on a tropical agricultural floodplain: Learning from two decades of aquatic invasive plant maintenance efforts. Ocean and Coastal Management, 198. https://doi.org/10.1016/J.OCECOAMAN.2020.105355

Warfe, D.M., Pettit, N.E., Davies, P.M., Pusey, B.J., Hamilton, S.K., Kennard, M.J., Townsend, S.A., Bayliss, P., Ward, D.P., Douglas, M.M., Burford, M.A., Finn, M., Bunn, S.E., Halliday, I.A., 2011. The “wet-dry” in the wet-dry tropics drives river ecosystem structure and processes in northern Australia. Freshw Biol 56, 2169–2195. https://doi.org/10.1111/J.1365-2427.2011.02660.X

Wasson, R., Furlonger L., Parry, D., Pietsch, T., Valentine, E., Williams, D., 2010. Sediment sources and channel dynamics, Daly River, Northern Australia. Geomorphology, 114(3), 161-174.

Wasson, R.J., Brooks, A.P., Olley, J.M., Knight, J., 2014. Erosion, sediment transport and deposition in the Daly River catchment: Implications for catchment management Sand dynamics in Darwin Harbour: a tool for coastal management View project. https://www.researchgate.net/publication/265392441

Waterhouse, J., Brodie, J., Lewis, S., Mitchell, A., 2012. Quantifying the sources of pollutants in the Great Barrier Reef catchments and the relative risk to reef ecosystems. https://doi.org/10.1016/j.marpolbul.2011.09.031

Waterhouse, J., Brodie, J., Tracey, D., Smith, R., Vandergragt, M., Collier, C., Petus, C., Baird, M., Kroon, F., Mann, R., Sutcliffe, T., Waters, D., Adame, F., 2017. Scientific Consensus Statement 2017: A synthesis of the science of land-based water quality impacts on the Great Barrier Reef, Chapter 3: The risk from anthropogenic pollutants to Great Barrier Reef coastal and marine ecosystems. State of Queensland, 2017.

Webster, I., Atkinson, I., Bostock, H., Brooke, B., Douglas, G., Ford, P., Hancock, G., Herzfeld, M., Leeming, R., Lemckert, C., Margvelashvili, N., Noble, B., Oubelkheir, K., Radke, L., Revill, A., Robson, B., Ryan, D., Schacht, C., Smith, C., Smith, J., Vicente-Beckett, Wild-Allen, K., 2006. The Fitzroy Contaminants Project-A study of the nutrient and fine-sediment dynamics of the Fitzroy Estuary and Keppel Bay. https://ozcoasts.org.au/wp-content/uploads/pdf/CRC/42_fitzroy_contaminants_screen.pdf

Wolanski, E., Williams, D., Spagnol, S., Chanson, H., 2004. Undular tidal bore dynamics in the Daly Estuary, Northern Australia. https://doi.org/10.1016/j.ecss.2004.03.001

York, P.H., Reason, C.L., Scott, E., Sankey, T., Rasheed, M., 2016. Seagrass Habitat of Cairns Harbour and Trinity Inlet: Annual Monitoring Report 2015. Centre for Tropical Water & Aquatic Ecosystem Research, James Cook University, Cairns.

Section 1: Appendix

Table A1. Linking changes in catchment activities through to impacts.

		Impacts		
Catchment change	**Stressor relevant to estuary and marine environment**	**Habitat**	**Processes**	**Species**
Increased irrigated agriculture	Increased flow abstraction	Reduced freshwater flow for mangrove growth & survival	Reduced nutrient loads = reducing primary productivity throughout estuaries at low levels of development Increased nutrient loads at high levels of development	Reduced food supply affecting species numbers
			Reduced connectivity to floodplains and river	Affects migrating species, reproduction, feeding, growth
		Reduced river flows, increasing the extent of saltwater inland	Impacts on riparian vegetation	Stress and loss of some species due to inability to sustain in elevated salinity
	Fertiliser application		Increased algal bloom and low DO risk	Reduced biodiversity e.g. loss of large fish species due to low DO
	Pesticide/ herbicide application	Wetlands, mangroves	Non-functioning food webs	Reduced biodiversity
	Flow regulation	In some cases, trapping sediment upstream reduces sediment deposition for mangroves, mudflats, beach sand	Reduced nutrient loads = reducing primary productivity throughout estuaries	Reduced food supply affecting species numbers
Increased agricultural	Increased erosion	Increased sediment flux in	Increase water temp and decrease in DO	Reduced biodiversity, fish kills

activity (livestock)		coastal floodplains		
			Increase in nutrients and algal bloom, low DO	Reduced biodiversity, fish kills
Mining	Increased erosion	Increased sediment flux in coastal floodplains	Increase water temp and decrease in DO	Reduced biodiversity, fish kills
			Increase in algal blooms, low DO	Reduced biodiversity, fish kills
	Alteration of groundwater levels	Damage to groundwater-dependent ecosystems	Loss of freshwater refugia	Reduced biodiversity, species loss, disruption to bird migration
	Increased pollutants in surface and groundwater	Potential water contamination	Addition of toxic metals reduce DO, increase turbidity	Toxic harm to species (including edible)
Urban development	Stormwater inputs	Increase in sediments during development phase, then increase in nutrients and other contaminants from urban living	Increase in algal blooms, low DO	Reduced biodiversity, fish kills
			Addition of toxic metals reduce DO, increase turbidity	Toxic harm to species (including edible)
	Sewage inputs	Increase in nutrients and possible DBP's (depending on treatment)	Increase in algal blooms, low DO	Reduced biodiversity, fish kills
			DBP's endocrine effects	???
	Industry inputs	Potential water contamination	Addition of toxic metals reduce DO, increase turbidity	Toxic harm to species (including edible)
Climate change	Increase in water temps and evaporation rates	Mangrove, mudflat habitats	Physiological stress for PP	Physiological stress for range of species

	Increasing severity of storms	Destruction of habitats e.g. mangroves, wetlands, beaches	Disruption to life-cycles	Many species use mangroves for nesting, spawning, growth etc. Beaches for nesting species
	Sea level rise	Destruction of habitats e.g. mangroves, wetlands, beaches	Disruption to life-cycles	Many species use mangroves for nesting, spawning, growth etc. Beaches for nesting species
		Enhanced seawater intrusion in rivers and coastal aquifers	Salinisation of surface and groundwater, loss of riparian vegetation	Stress and loss of some species due to inability to sustain in elevated salinity
	Changes in rainfall and ocean current patterns	Destruction of habitats e.g. mangroves, wetlands, beaches	Disruption to life-cycles	Many species use mangroves for nesting, spawning, growth etc. Beaches for nesting species
		Very large floods and mega-droughts impacting whole landscapes	Increased algal bloom and low DO risk	Reduced biodiversity
		Enhanced seawater intrusion in coastal aquifers	Salinisation of surface and groundwater, loss of riparian vegetation	Stress and loss of some species due to inability to sustain in elevated salinity
		Increased or decreased groundwater recharge, altering groundwater levels	Various, depending on the direction of alteration of the water table	
			Addition of toxic metals reduce DO, increase turbidity	Toxic harm to species (including edible)

2. Flood plume mapping

2.1. Introduction

Northern Australia is notable for its expansive river catchments, which represent vital components of the region's hydrological and ecological systems. The wet season, extending approximately from November to April, constitutes a pivotal period during which copious rainfall inundates the landscape, profoundly impacting floodplains, and inland ecosystems, before emptying into the ocean and extending outward, sometimes for 100's of kilometres, as large plumes of sediment and nutrient-laden water. Many coastal processes, plants, and animal species rely on this freshwater influx that reduces the hyper-salinity of the late dry season, and brings nutrients that drive the coastal and offshore productivity vital for fisheries species and migrating birds (Burford & Faggotter, 2021; Lowe et al., 2022). Wet season rainfall, however, is highly variable across northern Australia, with approximately one in five years considered a low-flow year. During these 'dry' summers, the reduction in freshwater entering the ocean can impact commercial fisheries and higher trophic species through reduced food availability, hyper salinity, and other processes, e.g. changes to triggers such as flooding that are key emigration cues (Broadley et al., 2020).

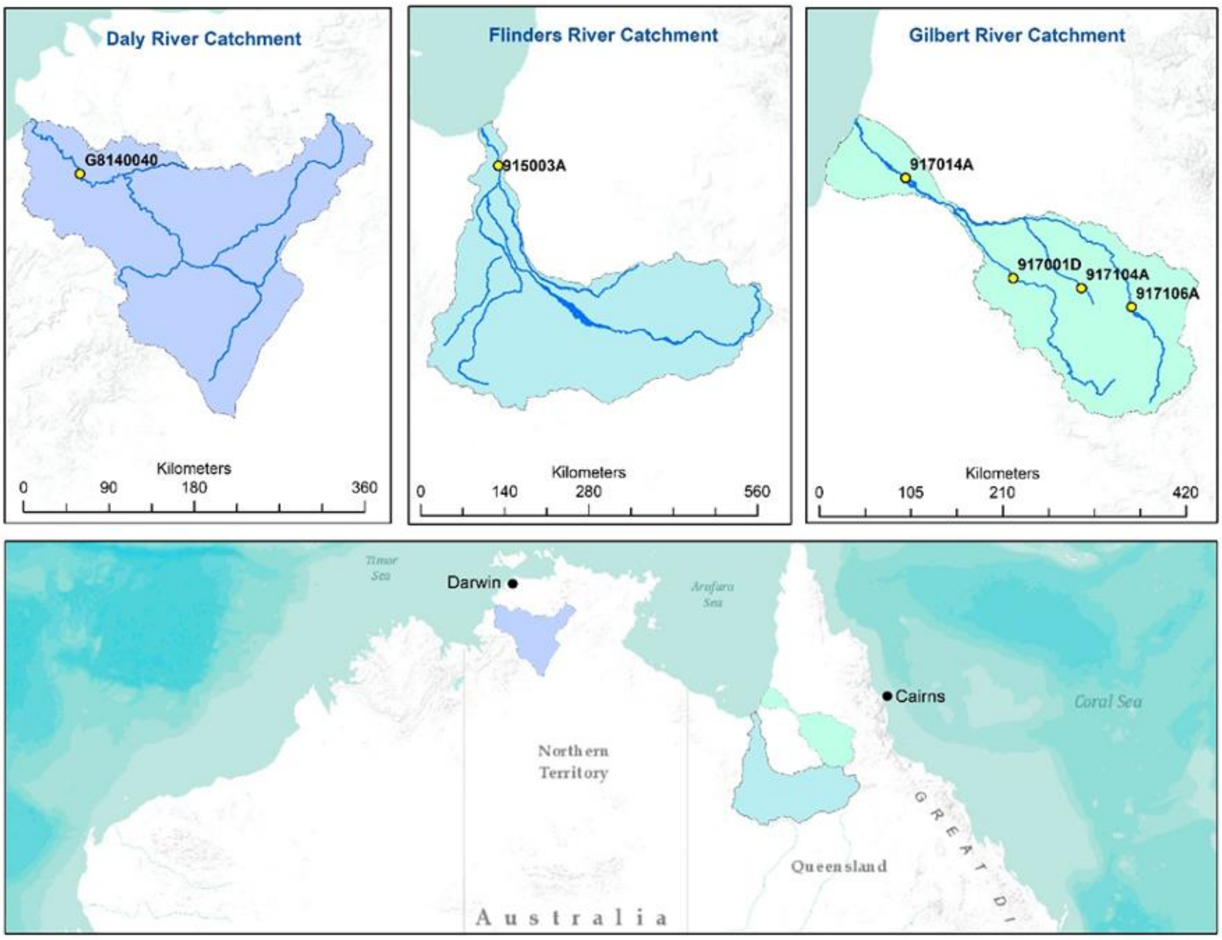
In recent years, there has been increased interest in northern Australian river catchments as a resource for further water development to support activities such as irrigated agriculture and critical minerals mining (CSIRO, 2013; DNRME, 2018). This has highlighted the need for an increased understanding of how water extraction might impact not only freshwater/floodplain ecosystems but also offshore productivity and the ecology of coastal ecosystems. Previous studies have shown that higher wet season flows are associated with increased catches of fish and crustaceans, including prawns (Broadley et al., 2020; Robins et al., 2005). The Northern Australia Prawn Fishery, for example, accounted for 21% of Australia's fishery income in 2021, making it the country's largest fishery. Potential reductions in prawn catch due to future water extraction during low flow years have been estimated to be over 50% (498-646 tonnes less catch; Broadley et al., 2020). Given that climate change adds further uncertainty regarding future rainfall levels in the region (Ridder et al., 2021), it is crucial to understand the relationship between wet season river flows and the extent of coastal flood plumes to assess the ecological resilience of northern Australia under increased water allocation within catchments.

This report aims to explore how wet season rainfall events in the Flinders, Gilbert and Daly River catchments drive flood plume extent and primary productivity in the adjacent coastal waters and assess how a reduction in environmental water through climate change and/or water re-allocation could reduce plume extent and coastal primary productivity across northern Australia.

2.2. Methods

2.2.1. Study region

Three notable river systems in northern Australia - the Flinders, Gilbert, and Daly – were selected for this study, each exhibiting catchments of substantial dimensions and ecological importance (Figure 2.1). The Flinders and Gilbert Rivers are located in northern Queensland, and both discharge into the Gulf of Carpentaria within 200 km of each other. Despite this proximity, the catchments exhibit different environments, water quality, and anthropogenic pressures.



Station name	Station Number	Commence date	Latitude	Longitude	Distance from sea	Catchment area
Daly River-Mount Nancar	G8140040	19/11/1952	-13.7662	130.7113	80 km	53,000 km^2
Flinders River at Walkers Bend	915003A	12/12/1969	-18.1617	140.8582	103 km	106,300 km^2
Gilbert River at Bourke Development Rd	917014A	11/02/2015	-17.1683	141.7675	102 km	39,100 km^2
Etheridge River at Roseglen	917104A	14/01/1967	-18.3064	143.579	350 km	867 km^2
Gilbert River at Rockfields	917001D	14/01/1967	-18.2025	142.876	267 km	10,990 km^2
Einasleigh River at Einasleigh	917106A	15/02/1968	-18.5002	144.0959	420 km	8,244 km^2

Figure 2-1. Map of the study area showing the Daly, Flinders and Gilbert catchments, their major tributaries, and the hydrological stations used in the flow analysis of this study. Note that the only downstream hydro station for the Gilbert River was offline. The Daly River catchment is located in the Northern Territory and discharges in Anson Bay in the Timor Sea. The volume of water discharged into the seas is the second highest of any river in Australia (CSIRO, 2009). Major streams within the catchment include the Katherine and Douglas Rivers, with perennial flow supported by significant groundwater input, a finite resource increasingly utilised for new agricultural and mining developments (Currell et al., 2024; Lamontagne et al., 2021). The streams and rivers in the Flinders, Gilbert and Daly River catchments are classified as Class 10 rivers, i.e. predictable summer highly intermittent flows (Kennard et al., 2010), and during the dry season, the rivers are often reduced to a series of drying waterholes.

2.2.2. Hydrological data

Hydrological flow data from the Flinders and Gilbert River catchments between 2003-2023 was retrieved from the Queensland Government Water Monitoring Information Portal (https://water-monitoring.information.qld.gov.au/). For the Daly River, hydrological data was extracted from the Bureau of Meteorology Water Data Online portal (http://www.bom.gov.au/waterdata/).

Downstream hydrological stations were selected for flow analysis (Figure 2.1). For the Flinders River, this was station 915003A (Flinders River at Walkers Bend), ~70 km upstream from the river mouth. This station has been continually gauging data since 1969. For the Gilbert River, data from station 917014A (Gilbert River at Bourke Development Road), approximately 100 km from the Gulf was used. Unfortunately, this station only re-started gauging data in February 2015 after a 25-year hiatus and this limited the Gilbert analysis to a shorter period of analysis. For the Daly River, the station was G8140003 (Daly River- Mt Nancar), ~70 km upstream from where the river enters Anson Bay in the Timor Sea.

Flow hydrographs were created for each river, and peak flow events for each year from 2003- 2023 were identified. For the Gilbert River, flow levels for the years 2003-2015 (when downstream station 9170014A was not operational) were obtained from three upstream tributaries (Einasleigh, Etheridge and Gilbert Rivers, station numbers 917106A, 917104A and 917001D respectively), and the flow levels were summed. Calendar years (as opposed to fiscal years that are often used to encapsulate wet season rainfall) were used, as very little flow occurred before January each wet season, and the analysis was related to peak flow events of 7-day durations, not annual rainfall averages.

2.2.3. Flood plume analysis

A search was conducted for satellite imagery from the Moderate Resolution Imaging Spectroradiometer (MODIS) true colour, corrected radiance products, covering the period immediately following peak flow events. The MODIS satellite images from the south-eastern Gulf of Carpentaria region that encompassed the Flinders and Gilbert River outflows (Coordinate limits: -18.580, 137.250, -14.361, 142.031) and from Anson Bay region that encompassed the Daly River outflow (Coordinate limits: -12.673, 129.446, -14.009, 130.534) were downloaded in GeoTIFF format from the NASA Worldview Snapshots Portal (https://wvs.earthdata.nasa.gov/). The products were loaded onto ArcMap (ArcGIS Desktop 10.8) for spatial plume size analysis.

Flood plume categories were defined as in Devlin and Schaffelke (2009), that were based on the plume concentration of water quality parameters that can be distinguished through ocean colour remote sensing, as follows:

(i) Primary water types were defined as having a high total suspended mineral (TSM) load, minimal chlorophyll (Chl-*a*) and high coloured dissolved and organic matter (CDOM).

(ii) Secondary water types were defined as regions where CDOM is still high. However, the TSM has been reduced and increased light and nutrient availability has prompted phytoplankton growth. Thus, the secondary plume exhibits high Chl-*a*, high CDOM and low TSM.

(iii) Tertiary water types are the regions of the plume that exhibit no elevated TSM and reduced amounts of Chl-*a* and CDM when compared with that of the secondary plume. Tertiary plumes can be described as being the transition between a secondary plume and ambient conditions.

Primary, secondary, and tertiary plumes were drawn using the ArcMap polygon tool, and the spatial extent in square kilometres for each plume type was calculated. This hand-digitising method has been shown to provide better accuracy when determining plume boundaries over shallow benthic features compared to semi-automated processes that can struggle to distinguish seafloor from turbid water (Evans et al., 2012).

To determine the relationship between flow and plume size, linear regression analyses were conducted between plume extent (sum of primary, secondary, and tertiary plumes) and 7-day hydrological flows. The analysis utilised complete hydrological data from 2003 to 2023 for the Flinders River and Daly River. For the Gilbert River, hydrological data from 2016 to 2023 only, was utilised.

2.2.4. Primary productivity

MODIS-aqua Level 2 ocean colour satellite products from the same satellite pass as the GeoTIFF products used to identify plumes, were downloaded from NASA (https://oceancolor.gsfc.nasa.gov) and the chlorophyll-a (chlor_a) bands opened in software package SeaDAS version 8.3.0 (NASA). The 'chlor_a' band, as described in Hu et al. (2019), returns the near-surface concentration of chlor_a in mg m^{-3} (µg L^{-1}). Each product was subsetted to the same spatial extent which was a) the south-east section of the Gulf of Carpentaria defined by the oceanic region south of latitude 15.5°S and bounded by the mainland, and b) the region of Anson Bay in the Timor Sea defined by the latitudes 13.0°S to -13.55°S, and longitudes 129.7°E to 130.3°E. The statistical analysis tool was run to quantify mean chlor_a following each flow event. Because primary productivity commonly increases as secondary/ tertiary plumes expand seaward, multiple products encompassing lags of 3-5 days (where cloud-free satellite imagery was available) were used to assess quantifiable changes in the chlor_a spatial extent in the week following peak flow events. Regression analyses were conducted between 7-day flow during peak flood events and total plume size, and 7-day flow and tertiary plume size.

2.2.5. Climate change and rainfall

Future climate predictions were retrieved from the Climate Change Web Portal (Earth Systems Research Laboratory, 2014), developed by the National Oceanic and Atmospheric Administration's (NOAA) Earth System Research Laboratory to collate and regionally downscale (to approximately 1° spatial resolution) the climate model outputs from the Coupled Model Intercomparison Project phase 6 (CMIP6; Eyring et al., 2016). The portal calculates the anomaly as the difference in the mean precipitation between the future climate (we used 2070–2099) and the model baseline reference period of 1985–2014, under the different Shared Socio-economic Pathways (SSPs). Six climate models were chosen for use in the analysis as they have shown the best performance against the historical climate of northern Australia (Ridder et al., 2021). The models used were CESM2-WACCM (NSF-DOE-NCAR, USA), FGOALS-G3 (Chinese Academy of Sciences), INM-CM5-0 (INM, Russia), ACCESS-CM2 (CSIRO-BOM, Australia), MRI-ESM2-0 (MPI-M, Germany) and NorESM2-MM (NCC, Norway). The scenario SSP5-8.5 was used as it projects the most global warming of all Shared Socio-economic Pathways, representing the continuation of a fossil fuel intensive world. In research applications, SSP5-8.5 is often used as the climate signal is strongest under this emissions scenario, making the signal most easily identifiable from the background noise of natural climate variability. Projected future rainfall anomalies, both seasonal and annual, were extracted for each climate model and the percentage increase or decrease in rainfall against baseline values (1995-2014) was calculated for each region (Gilbert, Flinders, and Daly catchments).

The projected percentage reduction in rainfall was applied to the 7-day flow – plume size regression models, with the assumption that the percentage reduction in rainfall would be the same as the percentage reduction in flow, and the projected change in flood plume extent in square km calculated.

2.3. Results

2.3.1. Annual flows

The Flinders River hydrological flow was highly variable (Figure 2.2) across the study period 2003-2023, with the highest sustained flows in 2009 (total annual flow of 19,503,455 ML), followed by 2019 (7,563,696 ML) and 2011 (5,029,243 ML). The most significant flood event took place in 2019 where daily flow peaked at 652,117 ML. In contrast, very low flows were experienced in 2013 (total annual flow of 58,949 ML), with low flows also in 2014 (327,320 ML) and 2007 (397,489 ML).

In contrast to the Flinders River, the Gilbert River (Figure 2.2) had more sustained flows but smaller peak flood events. The highest annual flows were in 2023 (total annual flow of 12,320,221 ML), 2009 (9,345,647 ML *upstream data only) and 2021 (7,152,598 ML), and the lowest flow was in 2022 (2,656,258 ML). Peak downstream flood events were similar across most years between 2016-2023 where they did not exceed 353,000 ML/day.

The Daly River (Figure 2.2) exhibited the largest flow volumes of the three catchments, with 2011 having an annual flow of 23,820,198 ML, followed by 2004 with 19,762,590 ML and 2008 15,520,934 ML. The year 2019 saw record flooding in the catchment, with hydrological stations becoming disabled for most of the wet season and flows were unable to be recorded accurately. Low flow years include 2022 (3,826,776 ML) and 2020 (2,564,811 ML).

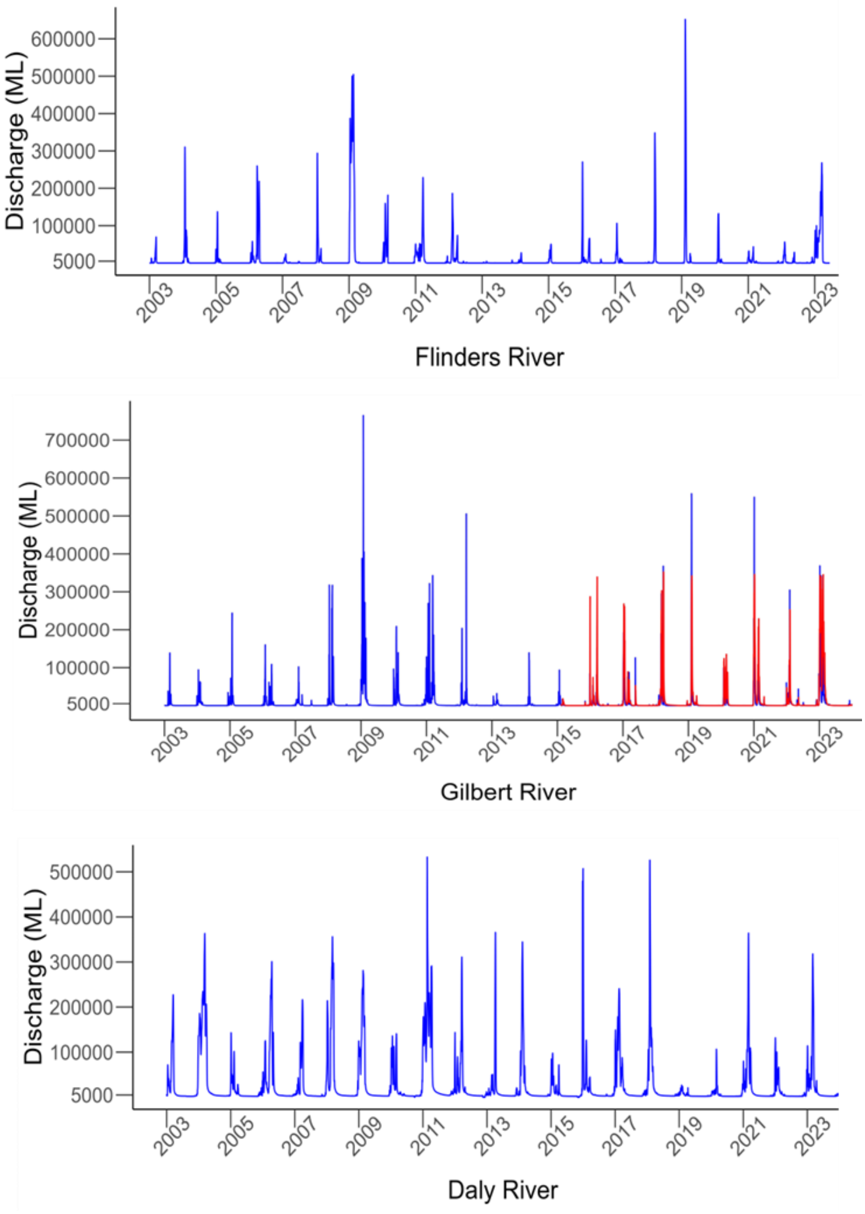


Figure 2-1. Daily flow (megalitres) at the Flinders River, Gilbert River, and Daly River from 2003 - 2023, from hydrological stations 915003A, 917014A, and G8140040 respectively. Note that for the Gilbert River, three upstream stations, that are located on the major tributaries of the Gilbert Catchment (917104A, 917106A, and 917001D; see Figure 2.1 for the location of these stations) are shown in blue (sum of the three stations), while downstream data that is only available from 2015, is shown in red. Note also that Daly River data was missing during the 2019 wet season floods.

2.3.2. Flood plume mapping

Over the 20-year study period, flood plumes in the Gilbert, Flinders and Daly Rivers were highly variable (Figure 2.3). The largest plume event from the Flinders River occurred in 2019 (plume size 6603 km^2), coinciding with the largest peak flood event ever recorded in the river, while the next largest plumes were found in 2008 and 2009 (3703 km^2 and 3589 km^2 respectively). The Gilbert River had its largest plumes of the study period in 2009 (6159 km^2), followed by 2012 and 2019 (4927 km^2 and 4233 km^2 respectively), with all years from 2006 – 2010 also having large plumes (>3,000 km^2). The Daly River had its largest plume events of this study in 2014 (2265 km^2), 2016 (2268 km^2), and 2023 (2180 km^2), where an extended flow event over several weeks led to a larger-than-normal tertiary plume. Note that 2019 was potentially a larger event for the Daly River. However, flooding caused the loss of hydrological data for most of the wet season and there were few opportunities for cloud-free remotely sensed imagery.

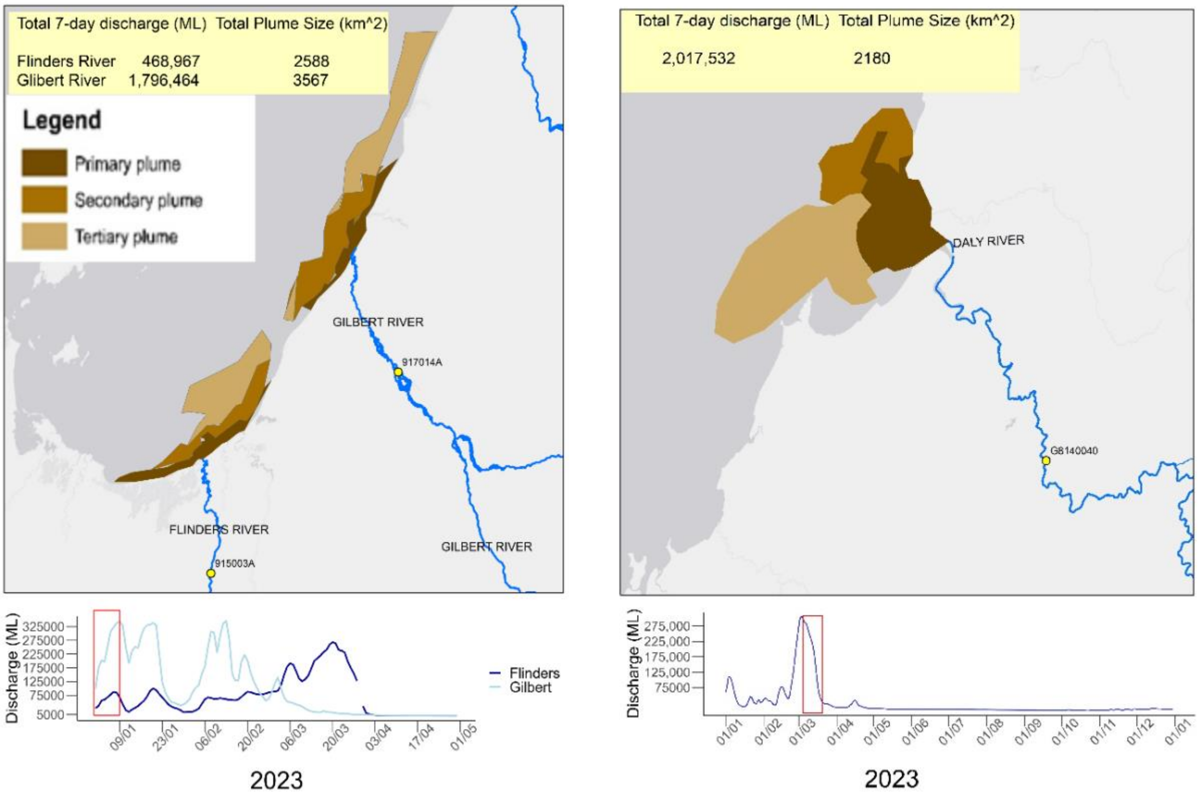


Figure 2-1. Flood plumes from the Flinders, Gilbert and Daly rivers following 7-day peak flow events, during each wet season from 2003 – 2023. Hydrographs at bottom of each map show the 7-day period (red box) immediately prior to each plume event. Yellow displays the 7-day total discharge (ML) from each river, and the total plume size (primary, secondary and tertiary plume combined km^2).

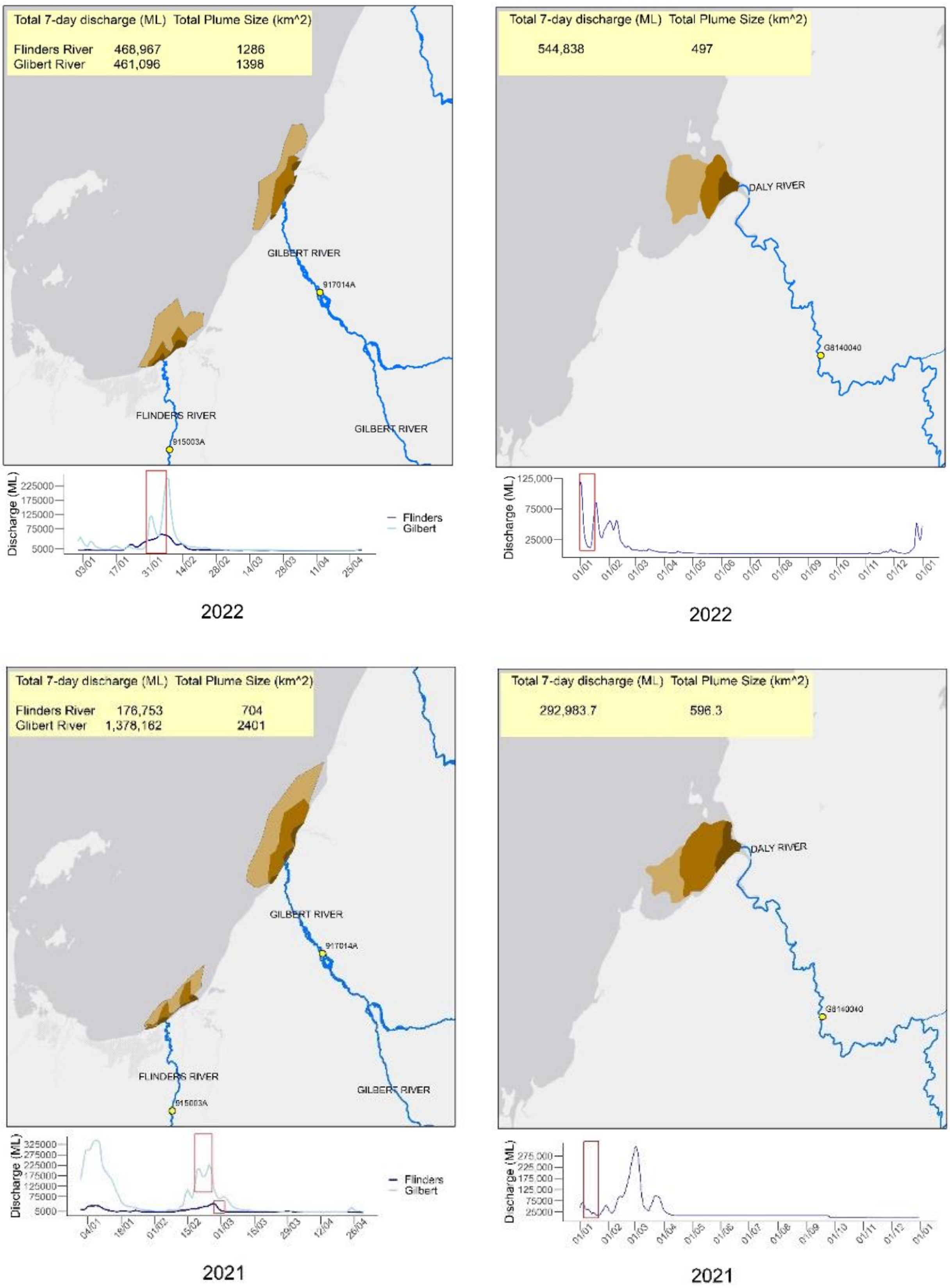


Figure 2.3 cont.

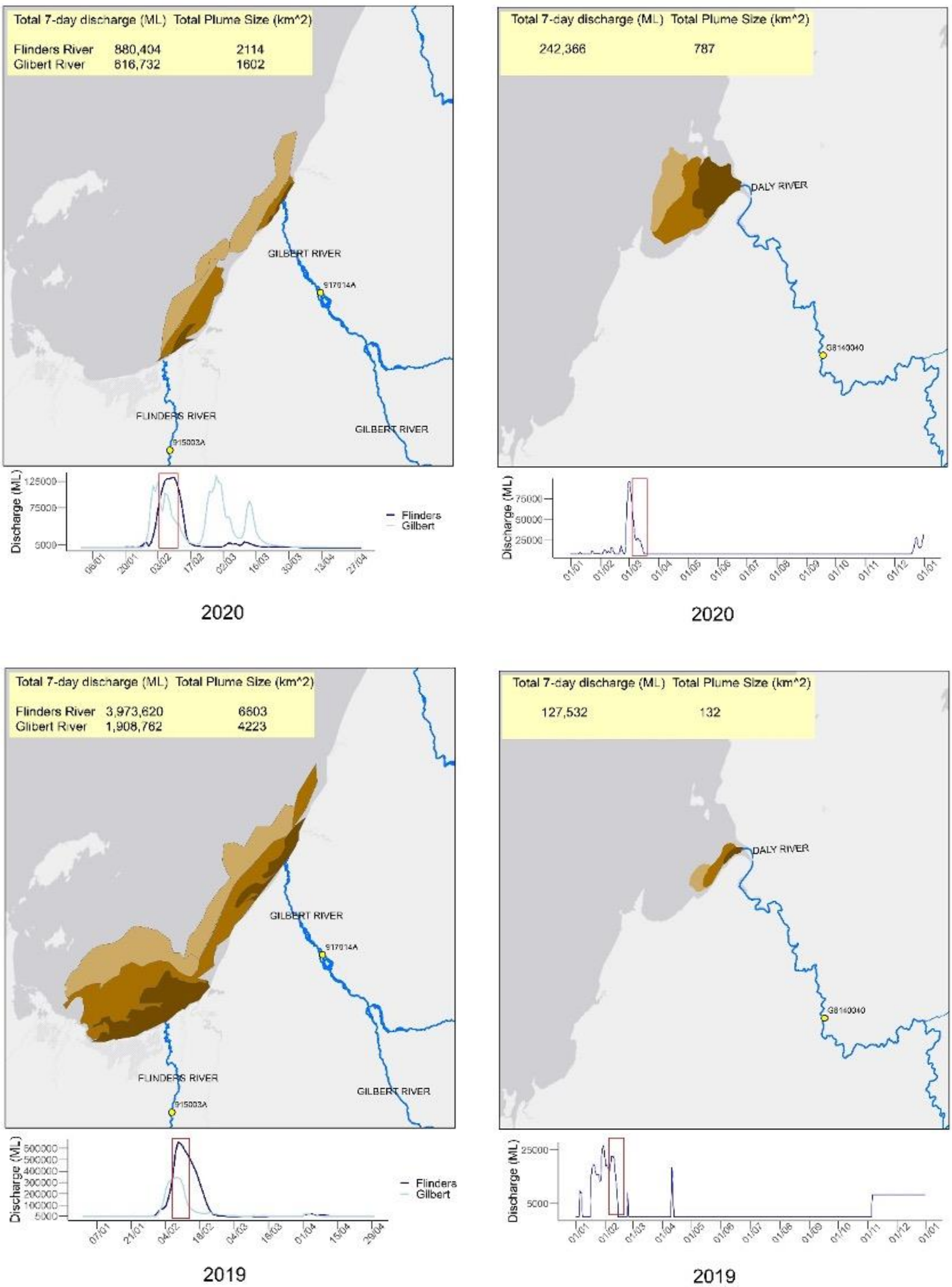


Figure 2.3 cont.

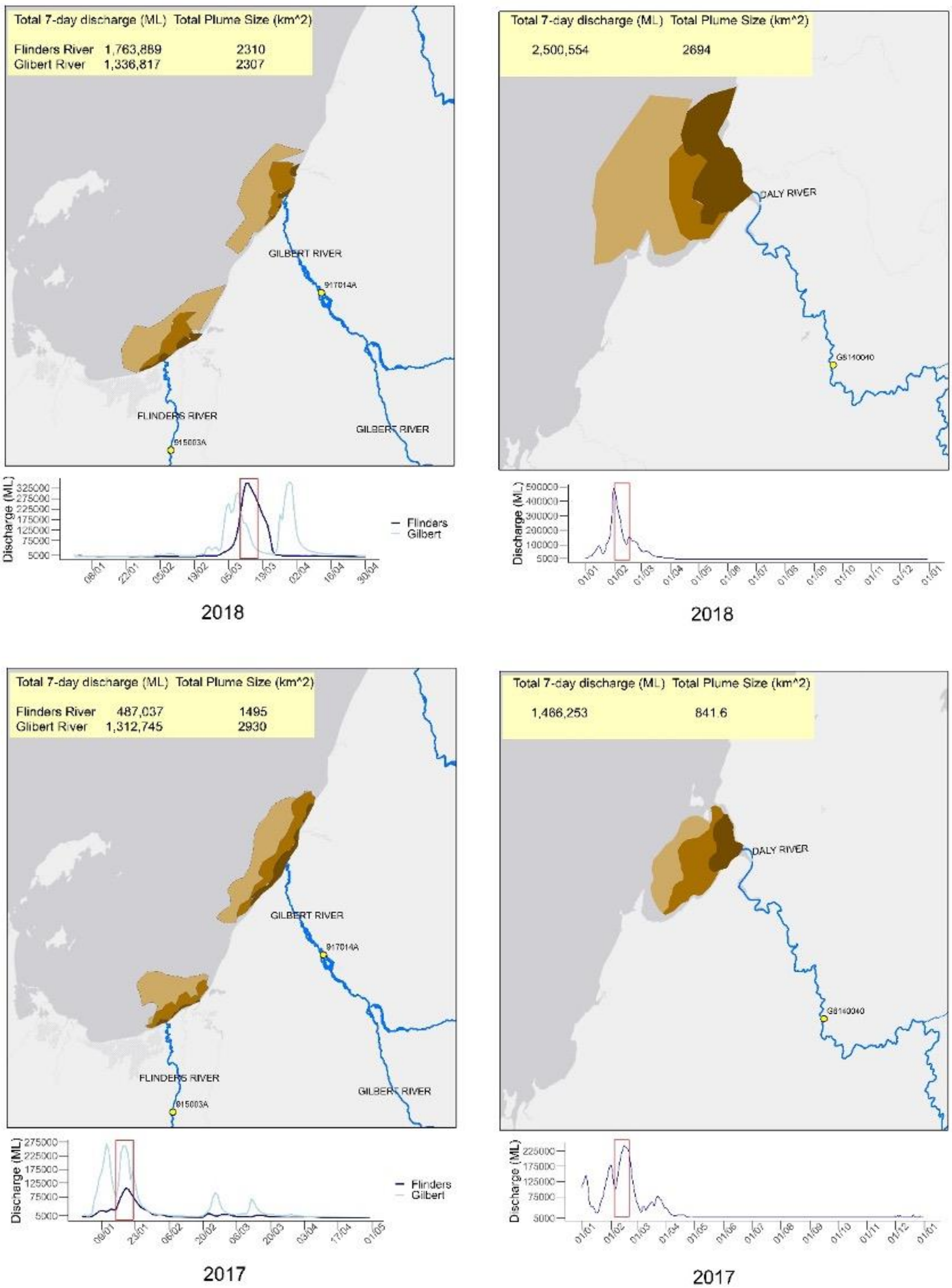


Figure 2.3 cont.

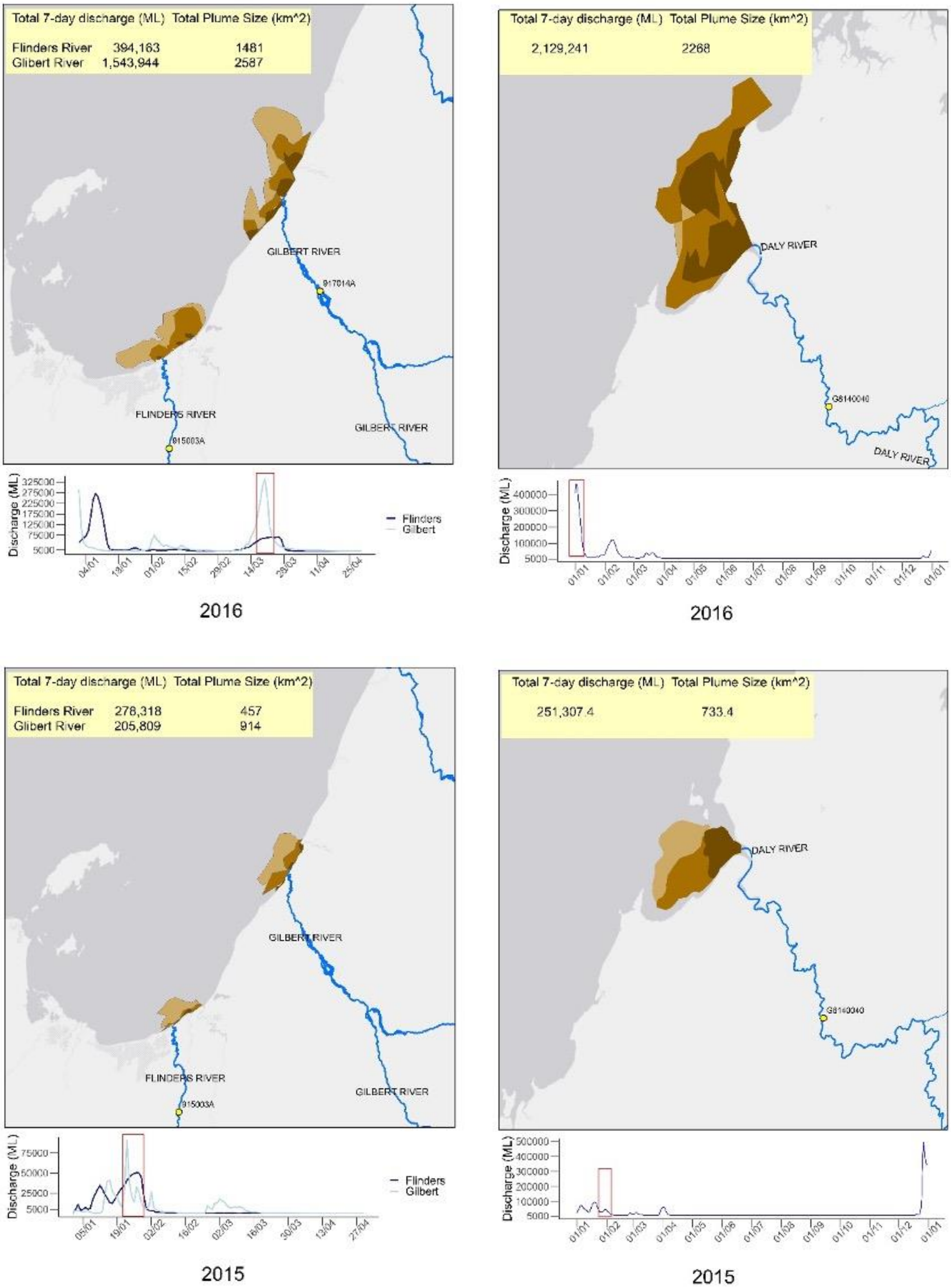


Figure 2.3 cont.

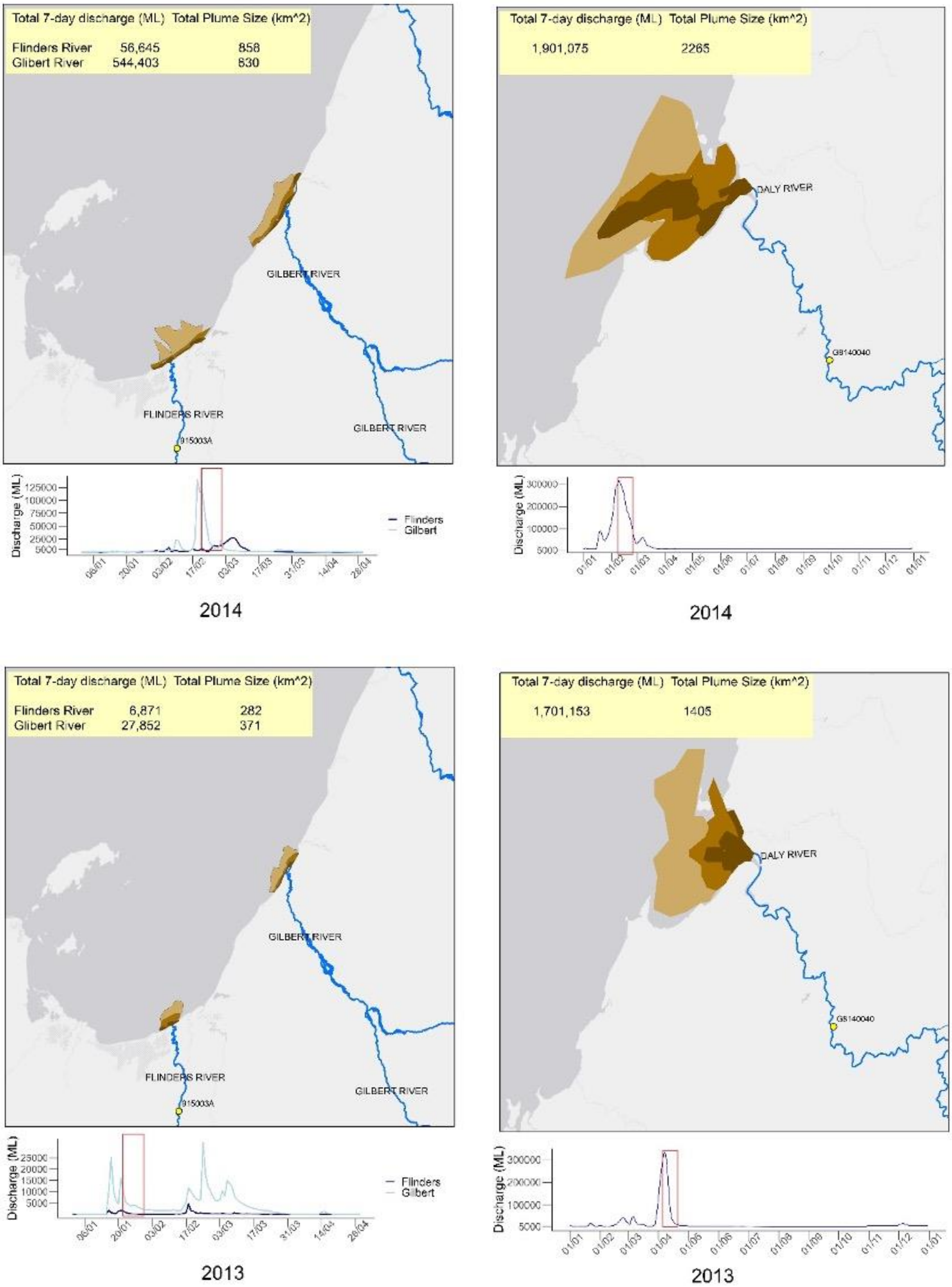


Figure 2.3 cont.

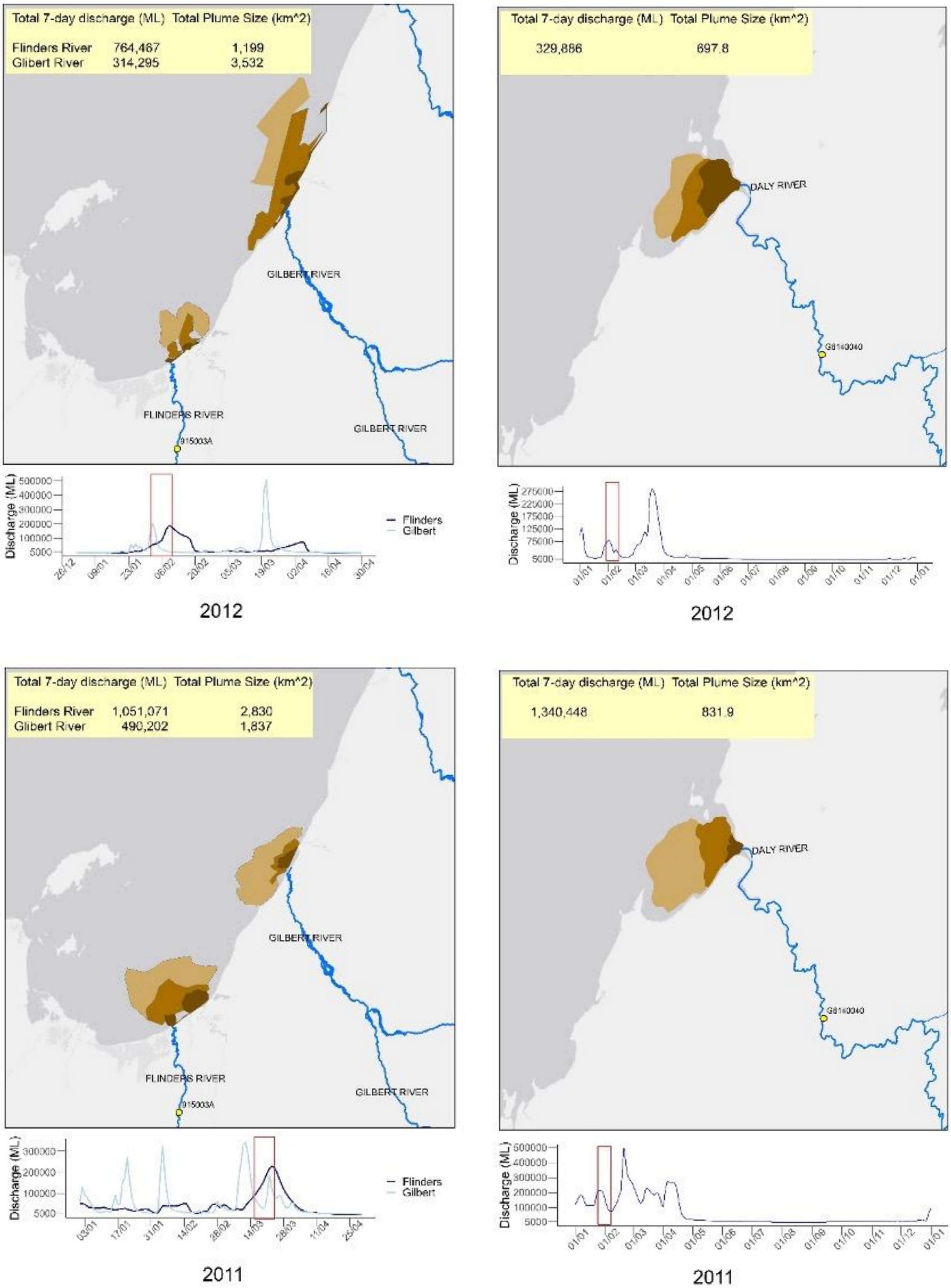


Figure 2.3 cont.

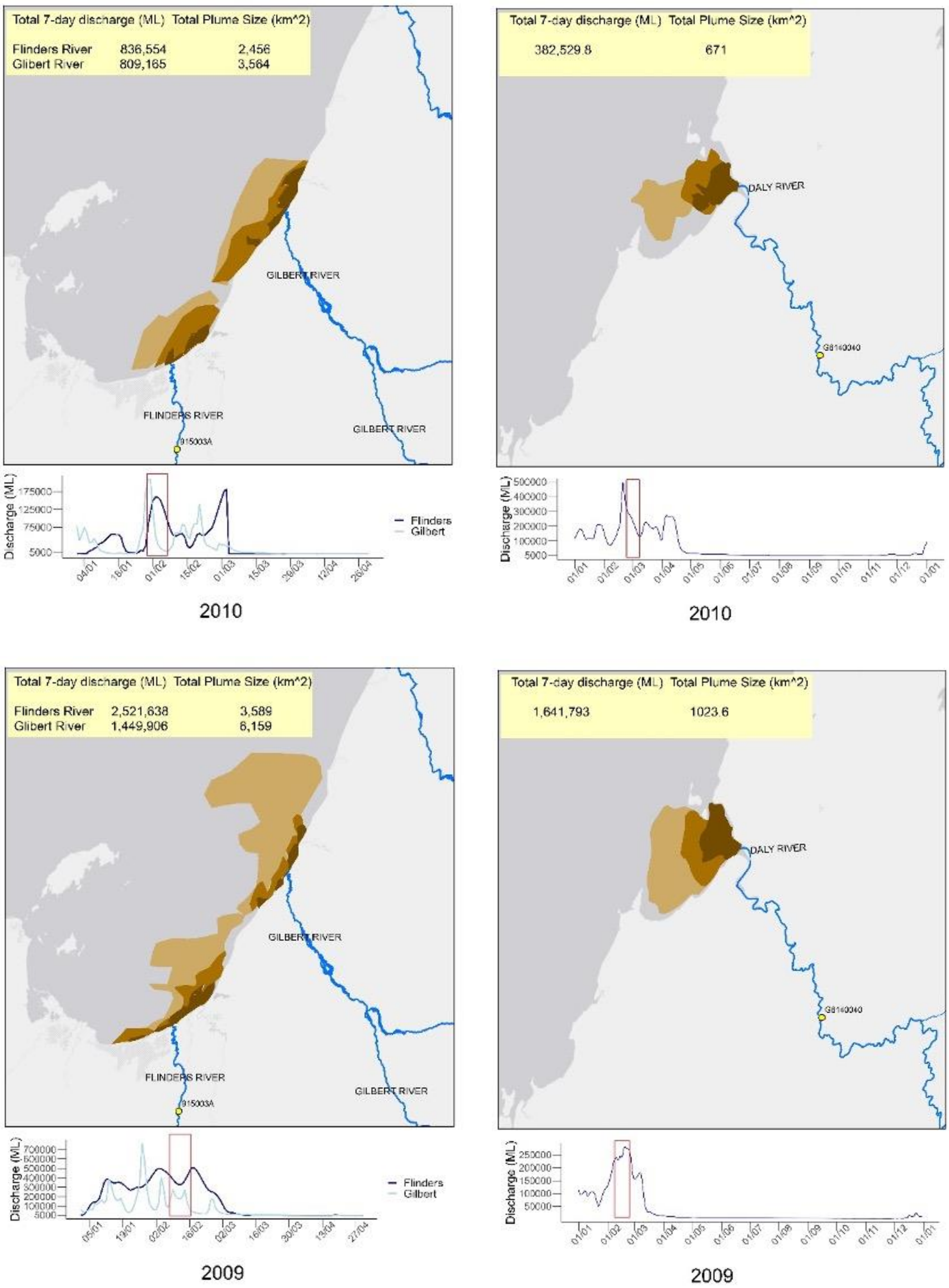


Figure 2.3 cont.

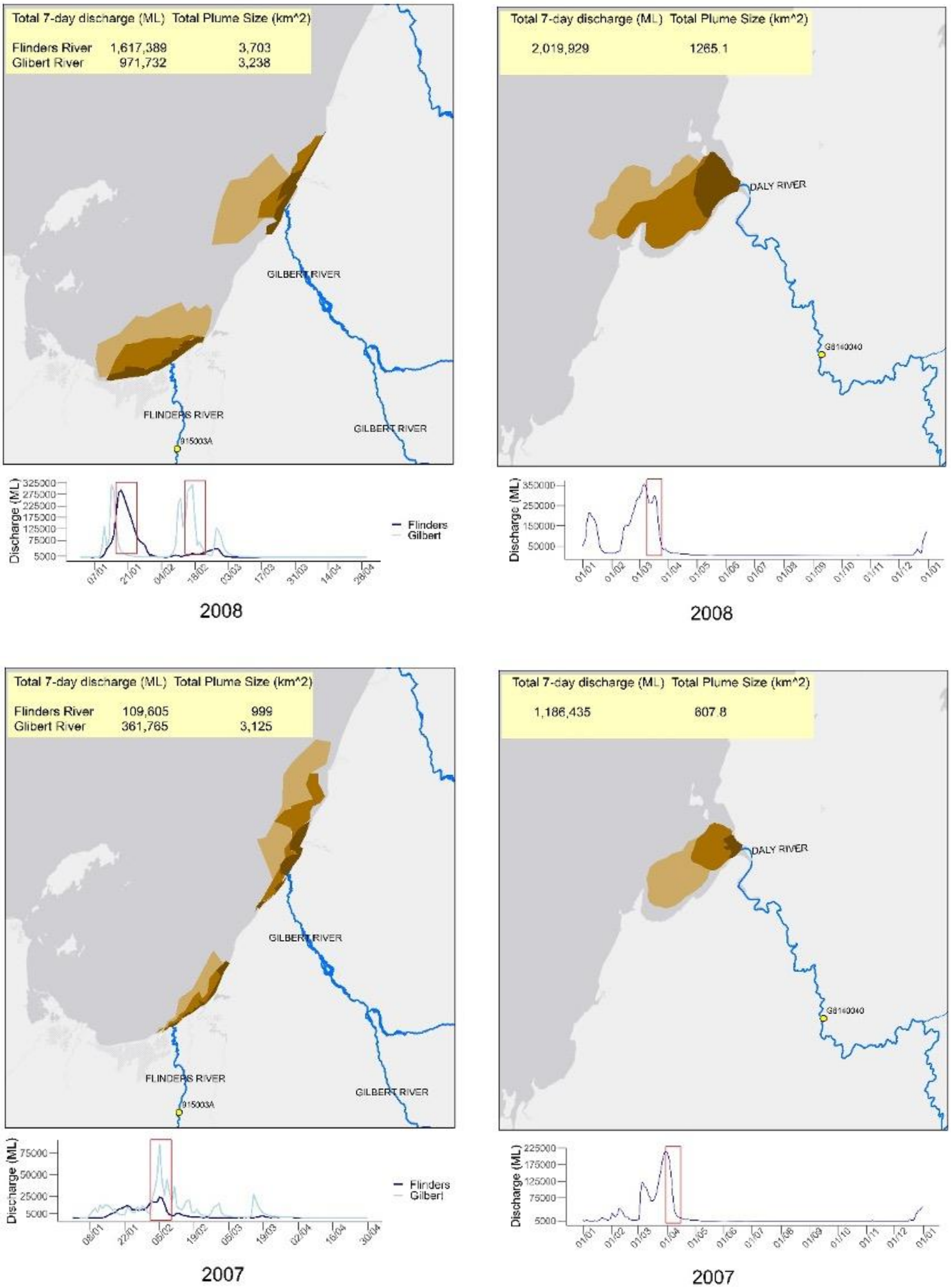


Figure 2.3 cont.

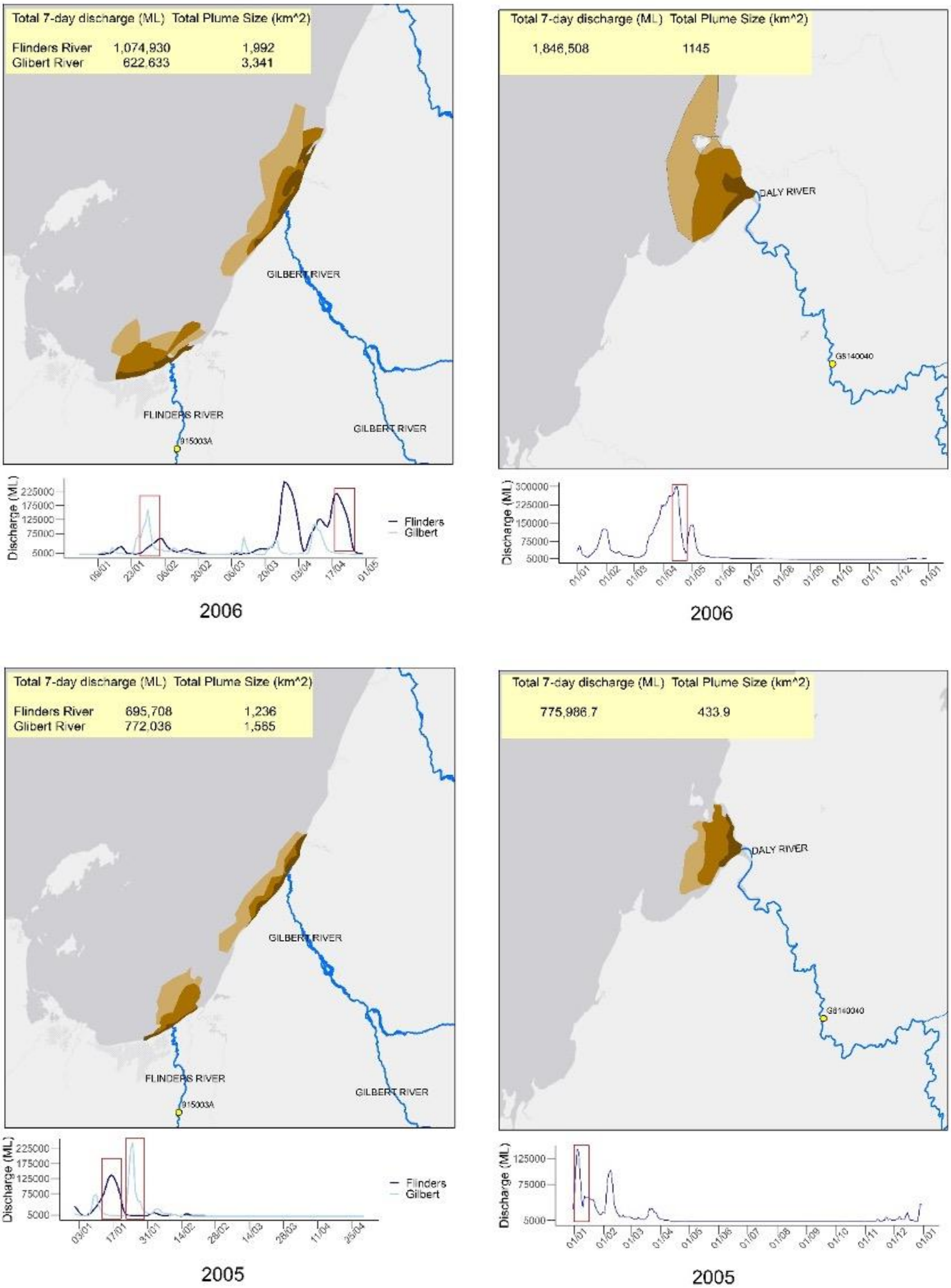


Figure 2.3 cont.

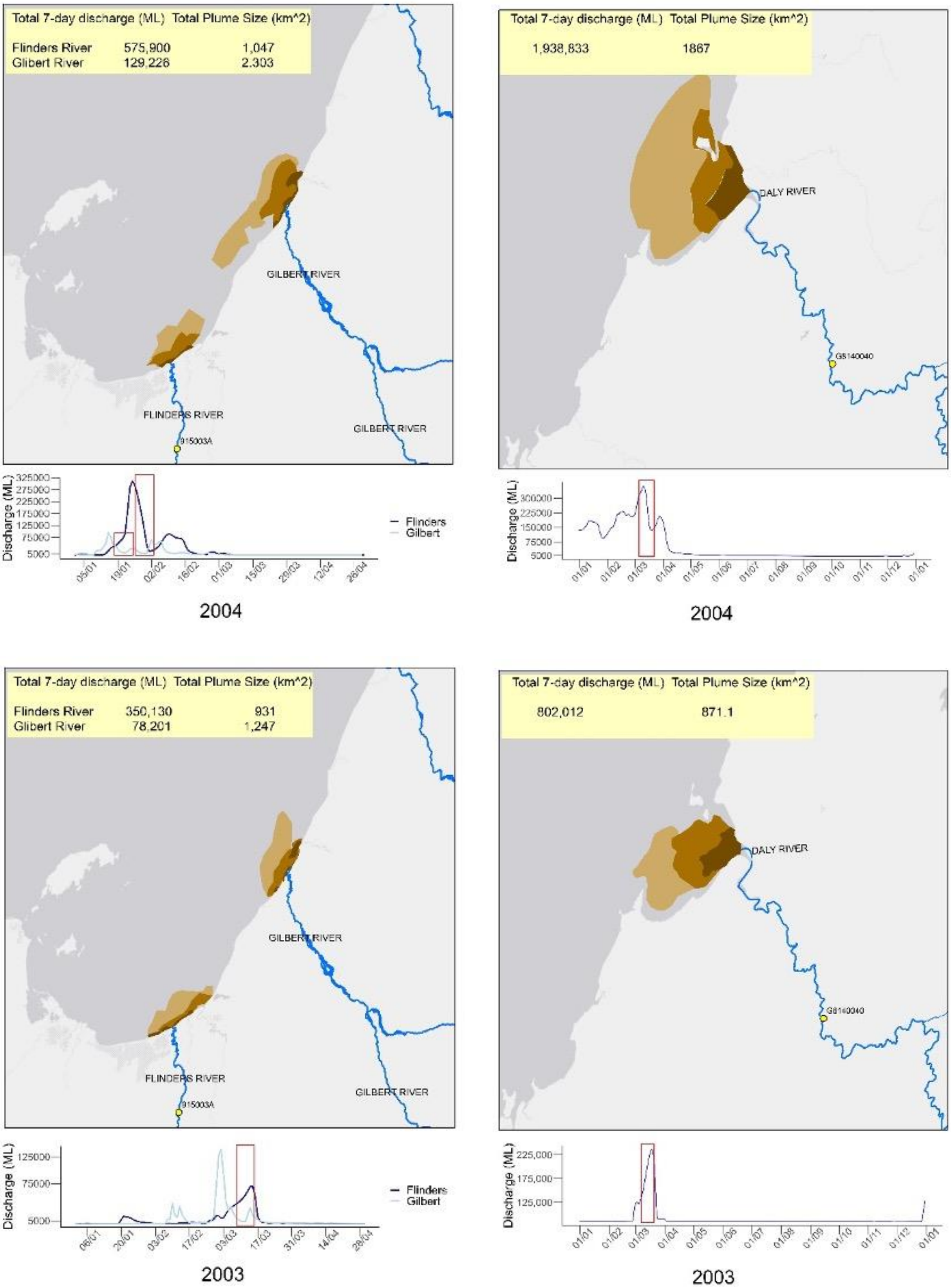


Figure 2.3 cont.

2.3.3. Linear regression analysis

Scatterplots illustrating the relationships between river flow (7-day total) and plume size (sum of primary, secondary, tertiary) are presented in Figure 2.4. The regression coefficients, standard errors, and significance levels are summarised in Appendix 1. Linear regression analyses examining the relationship between river flow and plume size for the Flinders, Gilbert and Daly Rivers were as follows.

The regression model for Flinders River explained a substantial proportion of the variance in plume size, with an R-squared value of 0.8613. The F-statistic was significant ($F(1, 19) = 117.9$, $p < 0.001$), indicating that the regression model as a whole was statistically significant. The model was specified as Flinders plume size (sq km) = 678.0 + 0.0014 * 7-day flow (ML).

The regression model for Gilbert River explained a substantial proportion of the variance in plume size, with an R-squared value of 0.848. The F-statistic was significant ($F(1, 6) = 33.48$, $p < 0.01$), indicating that the regression model as a whole was statistically significant. The model was specified as Gilbert plume size (sq km)= 436.4 + 0.0017 * 7-day flow (ML).

The regression model for Daly River explained a smaller proportion of the variance in plume size compared to the Gilbert and Flinders models with an R-squared value of 0.706. The F-statistic was however still highly significant ($F(1,19) = 45.5$, $p < 0.001$). The model was specified as Daly plume size (sq km) = 193.8 + 0.000776 * 7-day flow (ML).

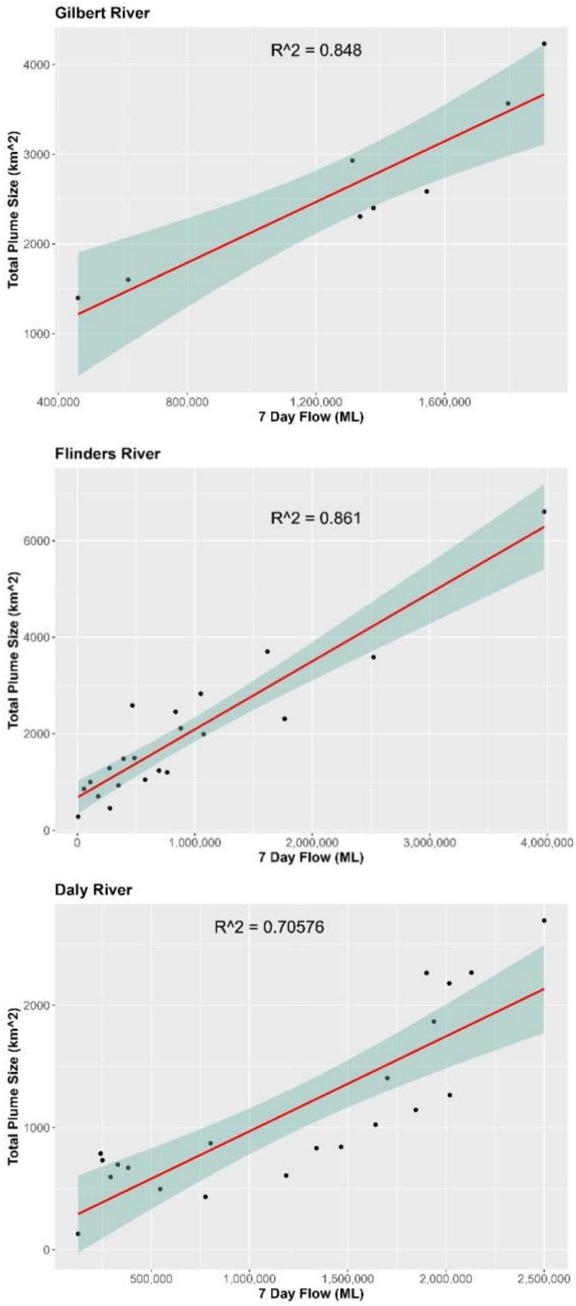
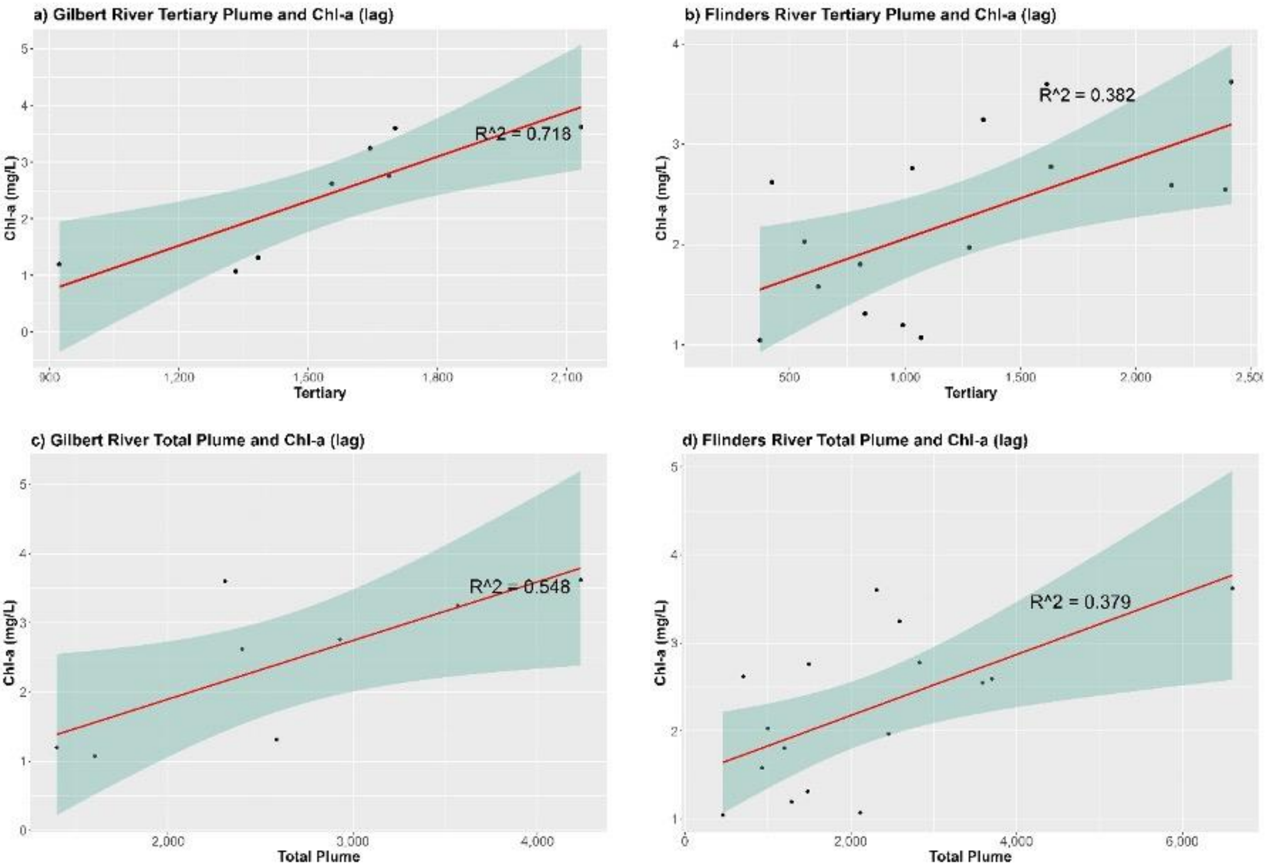


Figure 2-1. Relationship between total plume size and 7-day flow from the Flinders, Gilbert and Daly catchments. The plots show a linear regression model with a 95% confidence interval (green shaded area) around the regression line (red). The model explains a proportion of the variance in plume size as indicated by the R^2 value on each plot.

2.3.4. Primary productivity

Chlorophyll-*a* presence in the southern Gulf of Carpentaria was significantly associated with flood plume size from the Flinders and Gilbert Rivers (Figure 2.5; model summary results in supplementary material). The strongest relationships were found between tertiary plume size and chlorophyll-*a* (lag), with r-squared values of 0.383 and 0.718, respectively (Figures 2.5 a & b). The relationship between total plume size (combined primary, secondary and tertiary plume) was less strong, but still significant, for the Flinders and Gilbert Rivers, with r-squared values of 0.548 and 0.379, respectively (Figures 2.5 c & d). In contrast, the Daly River only had a significant relationship between tertiary plume size and lagged chlorophyll-*a* (Figure 2.5e). There was no significant relationship between non-lagged chlorophyll-a and the total plume size at the direct 7-day flood event for any river, indicating that it took days for the chlorophyll-*a* to develop after each flood event.



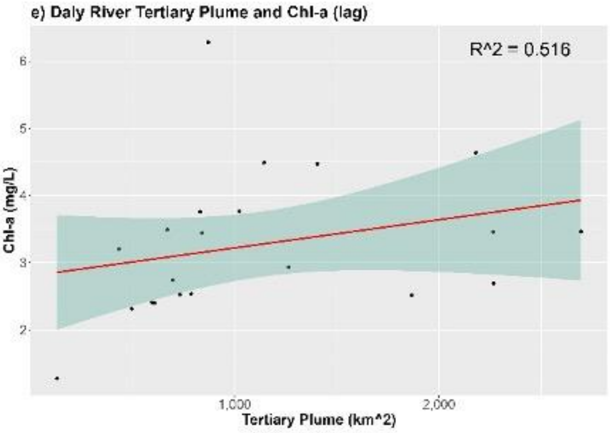


Figure 2-1. Relationship between total plume size/ tertiary plume size, and chlorophyll-a concentration (Chl-a) at three river catchments across northern Australia. a) Gilbert River tertiary plume and chlorophyll-a production; b) Flinders River tertiary plume and chlorophyll-a production; c) Gilbert River total flood plume (primary, secondary and tertiary combined) and chlorophyll-a production; d) Flinders River total flood plume (primary, secondary and tertiary combined) and chlorophyll-a production; e) Daly River tertiary plume and chlorophyll-a production. The plots show a linear regression model with a 95% confidence interval (green shaded area) around the regression line (red). The model explains a proportion of the variance in Chl-a as indicated by the R^2 value on each plot. The confidence interval reflects the uncertainty in the estimated relationship, with a narrower interval suggesting higher confidence in the model's prediction.

2.3.5. Climate change and rainfall

Projected rainfall anomalies from an ensemble of six CMIP6 global climate models, under climate forcing scenario SSP5-8.5 are presented in Table 2.1.

Table 2.1. Future annual (Ann) and seasonal (JFM- January, February, March; AMJ – April, May, June; JAS – July, August, September; OND – October, November, December) precipitation anomalies by 2070-2099 under climate forcing scenario SSP5-8.5 for the Flinders, Gilbert and Daly catchments.

Flinders catchment precipitation anomalies (mm) – Mean & standard deviation (SD)											**% change**
SP5-8.5	**Ann**	**SD**	**JFM**	**SD**	**AMJ**	**SD**	**JAS**	**SD**	**OND**	**SD**	
CESM2-WACCM	-254	38	-178	29	-24	13	-20	8	-43	7	-26.5
FGOALS-G3	-9	21	-8	12	12	9	-6	4	-22	10	-1.5
INM-CM5-0	81	55	61	37	-31	6	-10	6	46	13	7.7
ACCESS-CM2	-95	19	-45	10	-26	9	-16	6	-5	4	-16.2
MRI-ESM2-0	35	34	71	20	-13	10	-17	3	-2	6	8.2
NorESM2-MM	-237	21	-67	28	-17	8	-39	11	-97	15	-23.6
Ensemble	**-80**	**31**	**-28**	**23**	**-16**	**9**	**-18**	**6**	**-20**	**15**	**-10.4**

Gilbert catchment precipitation anomalies (mm) – Mean & standard deviation (SD)											**% change**
SP5-8.5	**Ann**	**SD**	**JFM**	**SD**	**AMJ**	**SD**	**JAS**	**SD**	**OND**	**SD**	
CESM2-WACCM	-277	21	-210	15	-19	5	3	10	-59	3	-22.5
FGOALS-G3	-117	61	-75	36	-13	13	1	2	-37	6	-14
INM-CM5-0	62	68	93	8	-19	3	-14	2	30	22	6.9
ACCESS-CM2	-119	13	-56	13	-39	7	-20	5	-6	3	-30
MRI-ESM2-0	15	22	61	33	-32	10	-11	1	-21	8	4.7
NorESM2-MM	-239	25	-43	11	-9	4	-23	7	-145	18	-22.6
Ensemble	**-112**	**35**	**-38**	**19**	**-22**	**7**	**-11**	**5**	**-40**	**10**	**-13.8**

Daly catchment precipitation anomalies (mm) – Mean & standard deviation (SD)											**% change**
SP5-8.5	**Ann**	**SD**	**JFM**	**SD**	**AMJ**	**SD**	**JAS**	**SD**	**OND**	**SD**	
CESM2-WACCM	-277	21	-210	15	-19	5	3	10	-59	3	-22.5
FGOALS-G3	-117	61	-75	36	-13	13	1	2	-37	6	-14
INM-CM5-0	62	68	93	8	-19	3	-14	2	30	22	6.9
ACCESS-CM2	-119	13	-56	13	-39	7	-20	5	-6	3	-30
MRI-ESM2-0	15	22	61	33	-32	10	-11	1	-21	8	4.7
NorESM2-MM	-239	25	-43	11	-9	4	-23	7	-145	18	-22.6
Ensemble	**-112**	**35**	**-38**	**19**	**-22**	**7**	**-11**	**5**	**-40**	**10**	**-13.8**

Predicted future precipitation anomalies from the ensemble (average of all six models) showed reduced rainfall for all river catchments, with annual reductions by 2070-2099 (from a baseline of 1985-2014 average rainfall) of 13.8% for the Gilbert River, 10.4% for the Flinders and 5.1% for the Daly River. There was large variation between the six individual climate models, with predictions ranging from a 26% decrease (Flinders catchment under CESM2-WACCM modelling) to a 13.6% increase (Daly catchment under MRI-ESM2-MM) in annual rainfall. Seasonal anomalies also varied between models, with the ensemble predicting highest reductions during the wet season months of January- March and October-December. The Gilbert and Flinders catchment also show large reductions in April to June and July to September rainfall, with the Daly showing smaller reductions over the dry season.

When the projected percentage decreases in rainfall were applied to the linear models describing flow contribution to flood plume size, reductions in the extent of flood plumes were in the range of 26 km^2 – 579 km^2 for the Flinders River, 108 km^2 – 448 km^2 for the Gilbert River and 5 km^2 to 100 km^2 for the Daly River, when based on the historic flows used in this study.

2.4. Discussion

Flood plumes from all three catchments (Flinders, Gilbert, and Daly) were highly variable over the 20 years of the analysis, corresponding to annual catchment rainfall variability. Flood plumes from the Flinders and Gilbert Rivers often spread out alongshore before dispersing as tertiary plumes offshore with whatever prevailing wind direction/ currents were prevalent at the time following the flood. The Daly River, in contrast, has a smaller, partly enclosed bay and while the flood plume generally remained confined to the bay, the effect of wind/ currents on the direction of the tertiary plume extent was also obvious.

The relationships between flood plume extent and hydrological flows, and flood plume extent and chlorophyll-*a* production were significant for all three rivers in this study, with the Flinders and Gilbert Rivers exhibiting stronger relationships than the Daly River. The regions that were analysed in regard to the chlorophyll-a impact zone were largely different in size (southern Gulf of Carpentaria ~ 50,000km^2, Anson Bay ~ 2,500km^2) and this likely affected the residence time for nutrient-laden river plumes to stay in the coastal regions. It may be that an analysis of chlorophyll-a across a more nuanced time period than the 7-day total flow used in this study could capture a stronger relationship. Another factor is the shape and exposure of the regions; the southern Gulf is bounded within the southern regions of a large, enclosed body, while Anson Bay is smaller and opens directly into the Timor Sea and is, therefore, more exposed to currents that can move bodies of water away faster.

This study has quantified the spatial extent of flood plumes in three catchments across northern Australia. However, it does not quantify Total Suspended Sediments (TSS), i.e., the total discharge (in weight) of sediment. The most accurate method for quantifying TSS over large spatial scales is taking water samples and in-situ radiometry for ground-truthing satellite reflectance data. However, sampling in the Gulf of Carpentaria and Anson Bay during wet season and across peak flow events is rarely feasible due to the remoteness and flooded access of these regions. Whilst applying third-party algorithms for quantifying TSS from satellite data is also feasible (e.g., Cartwright et al., 2021), the very high turbidity within the primary flood plume during the peak flow events of this study saturated the satellite sensor, and despite processing filters such as cloud and stray light masks being removed, regions containing very high sediment loads were masked. An example of this can be seen in Figure 2.6, where the chlorophyll reflectance band displays no data (white space) in the concentrated primary plume region of the flood event.

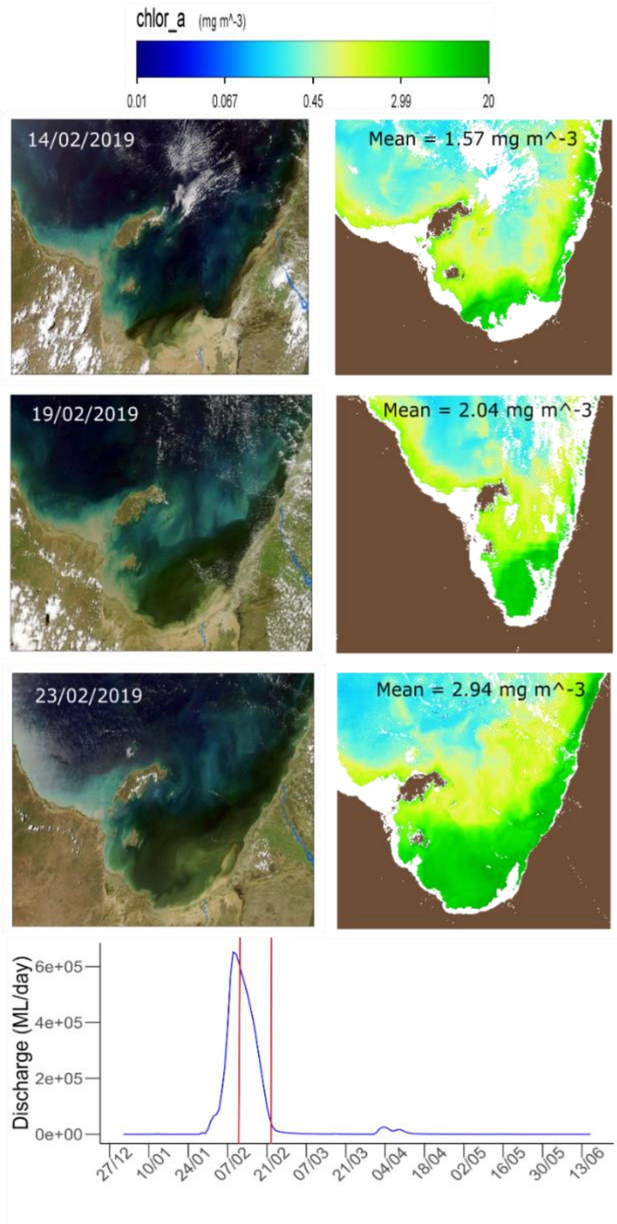


Figure 2-1. True colour images (left) and chlorophyll-a reflectance band data (right) immediately following a large flood event (top), 5 days post event (middle) and 9 days post event (bottom). The bottom graph shows the flow hydrograph over the period of flood plume development depicted in the images.

While this limitation affects TSS quantification, it does not limit chlorophyll-a (Chl-a) quantification, as Chl-a is less associated with the primary plume but arises as the primary plume evolves into secondary and tertiary plumes. Chl-a increases as the tertiary plume expands in the week/s following major flood plume events, as evident in Figure 2.6, where Chl-a increases dramatically as the primary plume dissipates and the tertiary plume spreads. Note that the mean values of Chl-a quantified in Figure 2.6 are an average of a large region where Chl-a values range from <1mg/m^3 - >20 mg /m^3 as referenced on the chlor_a scale.

Australia's Northern Prawn Fishery (NPF) is divided into 15 statistical regions where catch is recorded separately (Figure 2.7).

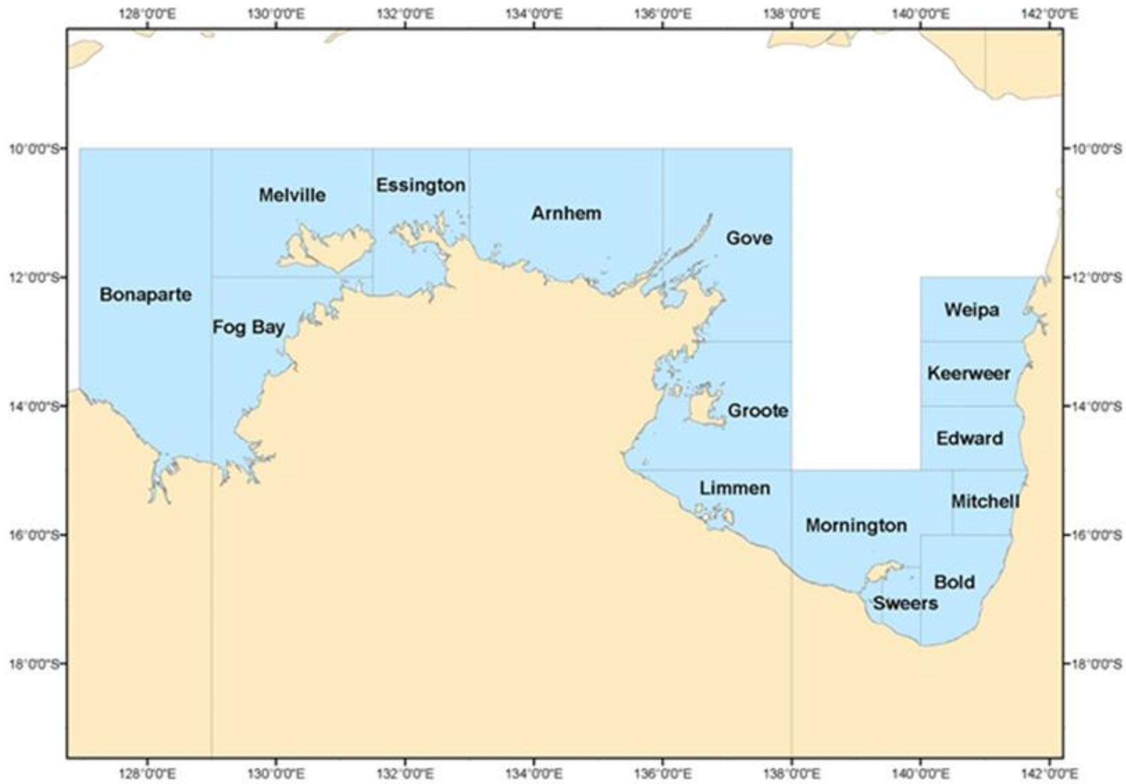


Figure 2-2. Statistical areas of the Northern Prawn Fishery. The Gilbert and Flinders Rivers discharge into the Bold region, while the Daly River discharges into the Fog Bay region.

The Bold fishery, where the Flinders and Gilbert Rivers enter the Gulf of Carpentaria, is one of the highest-producing regions in the Northern Prawn Fishery, producing 65% to 200% more banana prawns than any other region across the fishery (Figure 2.8). It's possible that nutrient-laden plumes remaining 'trapped' is driving the large primary productivity, and increasing prawn catch in this region. The phenomenon of ocean currents becoming somewhat trapped in the shallow head regions of the Gulf of Carpentaria is well-documented in oceanographic studies (Wolanski, 1993). Shallow coastal embayments are often associated with large prawn fisheries; for example, Exmouth Gulf and Shark Bay, both in Western Australia, supply over 2,000 tonnes of prawns annually (Kangas et al., 2015; Kangas et al., 2015a). This emphasises the importance of the river catchment-derived nutrient flows to this part of the Gulf of Carpentaria, not only for the productivity of the Northern Prawn Fishery, but for driving the unique coastal ecology that is vital for migrating shorebird populations and other estuarine, coastal, and marine species. Further studies into the relationship between oceanic currents and primary productivity in the head region of the Gulf would benefit our understanding of the contribution of river-sourced nutrients to the ecology of this region.

The Fog Bay fishery (Figure 2.7) where the Daly River empties into the ocean, is a larger region than the Bold fishery, with a more complex coastline and multiple rivers contributing nutrient-laden freshwater flows to the ocean (e.g., Ord, Keep, and Victoria rivers), making it difficult to separate individual sources of nutrient supply driving primary productivity. Despite being a larger region, the Fog Bay fishery is less productive than the Bold fishery (Figure 2.8), further emphasising the unique value of the Bold Fishery to Australia's largest fishing resource and the economic value of protecting the Flinders River from excessive water extraction.

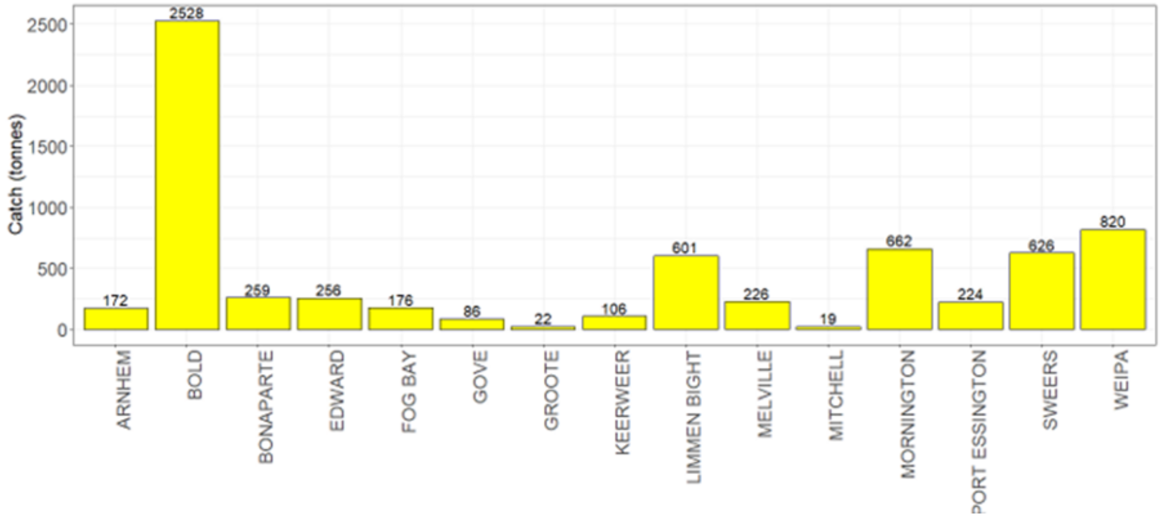


Figure 2-3. Total catch of Banana Prawns for each statistical area of the NPF in 2023 (Taken from: NPF Annual Data Summary 2023).

Climate change models for predicting future rainfall events across northern Australia do not offer consistent predictions for future rainfall, with model projections ranging from decreased rainfall to no change in rainfall, to increased rainfall, by 2079-2099. When combined as an ensemble, the six models predict an overall decrease in rainfall, with the largest reductions predicted to occur in the Gilbert River catchment (13.8%) followed by the Flinders (10.4%). Seasonally, the Gilbert River catchment would see most of its rainfall decrease in the wet season months from October to March. Because this is the time when flood events occur, it is possible that the effects of reduced rainfall will be skewed towards reductions in flood plume extent compared to the historical extents – which are presented here. The Flinders catchment would see its largest decrease in rainfall from January to March, with the remainder of the year seeing a fairly equal shortfall in rain. This is also likely to affect flood plume spatial extent and, thereby, primary productivity / prawn catch in the highest contributing region of the Northern Prawn Fishery. Further, the dry season reduction in rain could indicate more severe droughts during low flow years, giving way to increased erosion and turbidity that could affect nearshore processes when floods do return.

The Daly River catchment shows the smallest reduction in rainfall of the three catchments (5.1% less rainfall than historical levels), however, almost all the reduction in Daly catchment rainfall is predicted to occur during the wet season (Table 1). The catchment relies on the wet season deluge to recharge aquifers and support surface flows through the dry season (Barton & Pantus, 2010; Smerdon et al., 2012). The water allocation practices of the Northern Territory Government have been criticised by scientists and Traditional Owners for the lack of a formal water allocation plan and the practice of giving free water licences to developers (Currell et al., 2024; O'Donnell et al., 2022). As this practice is based on the assumption of guaranteed high annual wet season rainfall, the potential reduction in rainfall, particularly in the arid zones of the catchment, should be considered when water allocations are granted.

2.5. Conclusions

Wet season river flow is highly variable across northern Australia, with coastal flood plumes and primary productivity directly related to flow events. In the southern Gulf of Carpentaria, river flood plumes contribute to the largest prawn catches in Australia. Climate change projections show that large decreases in wet season rainfall will affect the Flinders, Gilbert, and Daly River catchments by 2079 – 2099, potentially resulting in reduced plumes of up to 580 km^2. A whole-of-catchment approach that includes the receiving marine environment should be considered when making water allocation policies.

2.6. References

Broadley, A., Stewart-Koster, B., Kenyon, R. A., Burford, M. A., & Brown, C. J. (2020). Impact of water development on river flows and the catch of a commercial marine fishery. *Ecosphere*, *11*(7), e03194.

Burford, M. A., & Faggotter, S. J. (2021). Comparing the importance of freshwater flows driving primary production in three tropical estuaries. *Marine Pollution Bulletin*, *169*, 112565.

Barton, C.L. & Pantus, F.J., Daly River Catchment Water Model Catchment Report: Progress Report, TRaCK, Australian Rivers Institute, Griffith University, Nathan, Qld, August 2010.

Cartwright, P. J., Fearns, P. R., Branson, P., Cuttler, M. V., O'leary, M., Browne, N. K., & Lowe, R. J. (2021). Identifying metocean drivers of turbidity using 18 years of MODIS satellite data: implications for marine ecosystems under climate change. *Remote Sensing*, *13*(18), 3616.

CSIRO. (2009). Water in the Timor Sea Drainage Division. A report to the Australian Governemnt from the CSIRO Northern Australia Sustainable Yields Project. CSIRO Water for a Healthy Country Flagship, Australia , 508pp

CSIRO. (2013) Agricultural resource assessment for the Flinders catchment. An overview report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, part of the North Queensland Irrigated Agriculture Strategy. Canberra, A.C.T.: CSIRO; 2013. csiro:EP1313098. https://doi.org/10.4225/08/584ee5d194764

Currell, M., Jackson, S., & Ndehedehe, C. (2024). Risks in the current groundwater regulation approach in the Beetaloo region, Northern Territory, Australia. *Australasian Journal of Water Resources*, *28*(1), 47–63. https://doi.org/10.1080/13241583.2023.2297949

Devlin, M., & Schaffelke, B. (2009). Spatial extent of riverine flood plumes and exposure of marine ecosystems in the Tully coastal region, Great Barrier Reef. *Marine and Freshwater Research*, *60*(11), 1109-1122. https://doi.org/https://doi.org/10.1071/MF08343

DNRME. (2018). *Minister's Performance Assessment Report, Water Plan (Gulf) 2007*. Brisbane, Australia

Evans, R. D., Murray, K. L., Field, S. N., Moore, J. A. Y., Shedrawi, G., Huntley, B. G., Fearns, P., Broomhall, M., McKinna, L. I. W., & Marrable, D. (2012). Digitise This! A Quick and Easy Remote Sensing Method to Monitor the Daily Extent of Dredge Plumes. *PLoS One*, *7*(12), e51668. https://doi.org/10.1371/journal.pone.0051668

Eyring, V., Bony, S., Meehl, G.A., Senior, C.A., Stevens, B., Stouffer, R.J. and Taylor, K.E., 2016. Overview of the Coupled Model Intercomparison Project Phase 6 (CMIP6) experimental design and organization. *Geoscientific Model Development*, *9*(5), pp.1937-1958.

Hu, C., Feng, L., Lee, Z., Franz, B. A., Bailey, S. W., Werdell, P. J., & Proctor, C. W. (2019). Improving Satellite Global Chlorophyll a Data Products Through Algorithm Refinement and Data Recovery. *Journal of Geophysical Research: Oceans*, *124*(3), 1524-1543.https://doi.org/https://doi.org/10.1029/2019JC014941

Kangas, M I, Sporer, E C, Hesp, S A, Travaille, K L, Moore, N, Cavalli, P, and Fisher, E A. (2015), *Exmouth Gulf Prawn Managed Fishery*. Department of Fisheries, Perth, Western Australia. Report 1. https://library.dpird.wa.gov.au/fr_msc/1

Kangas, M I, Sporer, E C, Hesp, S A, Travaille, K L, Brand-Gardner, S J, Cavalli, P, and Harry, A V. (2015a), *Shark Bay Prawn Managed Fishery*. Department of Fisheries, Perth, Western Australia. Report 2. https://library.dpird.wa.gov.au/fr_msc/2

Kennard, M. J., Pusey, B. J., Olden, J. D., Mackay, S. J., Stein, J. L., & Marsh, N. (2010). Classification of natural flow regimes in Australia to support environmental flow management. *Freshwater Biology*, *55*(1), 171-193.

Lamontagne, S., Suckow, A., Gerber, C., Deslandes, A., Wilske, C. and Tickell, S., 2021. Groundwater sources for the Mataranka springs (northern territory, Australia). *Scientific Reports*, *11*(1), p.24288.

Lowe, V., Frid, C. L. J., Venarsky, M., & Burford, M. A. (2022). Responses of a macrobenthic community to seasonal freshwater flow in a wet-dry tropical estuary. *Estuarine, Coastal and Shelf Science*, *265*, 107736. https://doi.org/https://doi.org/10.1016/j.ecss.2021.107736

NASA Ocean Biology Processing Group. (2018). Sea-viewing Wide Field-of-view Sensor (SeaWiFS) Level-2 Ocean Color Data, version R2018.8, NASA Ocean Biology Distributed Active Archive Center. doi: 10.5067/ORBVIEW-2/SEAWIFS/L2/OC/2018. Accessed on 2019/02/29.

O'Donnell, E., Jackson, S., Langton, M., & Godden, L. (2022). Racialized water governance: the 'hydrological frontier'in the Northern Territory, Australia. *Australasian Journal of Water Resources*, *26*(1), 59-71.

Ridder, N. N., Pitman, A. J., & Ukkola, A. M. (2021). Do CMIP6 Climate Models Simulate Global or Regional Compound Events Skillfully? *Geophysical Research Letters*, *48*(2), e2020GL091152. https://doi.org/https://doi.org/10.1029/2020GL091152

Robins, J. B., Halliday, I. A., Staunton-Smith, J., Mayer, D. G., & Sellin, M. J. (2005). Freshwater-flow requirements of estuarine fisheries in tropical Australia: A review of the state of knowledge and application of a suggested approach. Marine and Freshwater Research, Scott, J. D., Alexander, M. A., Murray, D. R., Swales, D., & Eischeid, J. (2016). The climate change web portal: A system to access and display climate and Earth system model output from the CMIP5 archive. *Bulletin of the American Meteorological Society*, *97*(4), 523-530.

Smerdon, B. D., Gardner, W. P., Harrington, G. A., & Tickell, S. J. (2012). Identifying the contribution of regional groundwater to the baseflow of a tropical river (Daly River, Australia). Journal of Hydrology, 464, 107-115.Tebaldi, C., Debeire, K., Eyring, V., Fischer, E., Fyfe, J., Friedlingstein, P., Knutti, R., Lowe, J., O'Neill, B., Sanderson, B. and Van Vuuren, D., 2021. Climate model projections from the scenario model intercomparison project (ScenarioMIP) of CMIP6. *Earth System Dynamics*, *12*(1), pp.253-293.

Wolanski, E. (1993). Water circulation in the Gulf of Carpentaria. *Journal of Marine Systems*, 4(2-3), 401-417.

Section 2: Appendix

Table A 2.1. Model summary of relationship between Gilbert River tertiary flood plume size and mean Chlorophyll-a in the southern Gulf of Carpentaria.

```
lm(formula = chla_lag ~ Tertiary, data = Gilb_ch)

Residuals:
     Min       1Q   Median       3Q      Max 
-0.79752 -0.43370  0.06151  0.43767  0.76082 

Coefficients:
              Estimate Std. Error t value Pr(>|t|)   
(Intercept) -1.6176664  1.0584819  -1.528  0.17730   
Tertiary     0.0026185  0.0006701   3.907  0.00792 **
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 0.6211 on 6 degrees of freedom
Multiple R-squared:  0.7179,	Adjusted R-squared:  0.6709 
F-statistic: 15.27 on 1 and 6 DF,  p-value: 0.007916
```

Table A 2.2. Model summary of relationship between Gilbert River total flood plume size (sum of primary, secondary and tertiary plumes) and mean Chlorophyll-a in the southern Gulf of Carpentaria.

```
lm(formula = chla_lag ~ Plume_sum, data = Gilb_ch)

Residuals:
     Min       1Q   Median       3Q      Max 
-1.08097 -0.26360 -0.07319  0.15255  1.44355 

Coefficients:
             Estimate Std. Error t value Pr(>|t|)  
(Intercept) 0.2026639  0.8704429   0.233   0.8236  
Plume_sum   0.0008469  0.0003139   2.698   0.0357 *
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 0.786 on 6 degrees of freedom
Multiple R-squared:  0.5482,	Adjusted R-squared:  0.4729 
F-statistic:  7.28 on 1 and 6 DF,  p-value: 0.03567
```

Table A 2.3. Model summary of relationship between Flinders River tertiary flood plume size and mean Chlorophyll-a in the southern Gulf of Carpentaria.

```
lm(formula = chla_lag ~ Tertiary, data = flin)

Residuals:
    Min      1Q  Median      3Q     Max
-1.0412 -0.5306 -0.1358  0.4871  1.0466

Coefficients:
             Estimate Std. Error t value Pr(>|t|)
(Intercept) 1.2513880  0.3774881   3.315  0.00511 **
Tertiary    0.0008061  0.0002737   2.945  0.01066 *
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.7015 on 14 degrees of freedom
  (5 observations deleted due to missingness)
Multiple R-squared:  0.3825,    Adjusted R-squared:  0.3384
F-statistic: 8.672 on 1 and 14 DF,  p-value: 0.01066
```

Table A 2.4. Model summary of relationship between Flinders River total flood plume size (sum of primary, secondary, and tertiary plumes) and mean Chlorophyll-a in the southern Gulf of Carpentaria.

```
lm(formula = chla_lag ~ plume_sum, data = flin)

Residuals:
    Min      1Q  Median      3Q     Max
-1.1432 -0.4228 -0.1606  0.4237  1.3163

Coefficients:
             Estimate Std. Error t value Pr(>|t|)
(Intercept) 1.4850688  0.3113269   4.770 0.000299 ***
plume_sum   0.0003457  0.0001183   2.922 0.011140 *
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.7036 on 14 degrees of freedom
  (5 observations deleted due to missingness)
Multiple R-squared:  0.3789,    Adjusted R-squared:  0.3345
F-statistic:  8.54 on 1 and 14 DF,  p-value: 0.01114
```

Table A 2.5. Model summary of relationship between the Daly River total flood plume size (sum of primary, secondary, and tertiary plumes) and mean Chlorophyll-a in Anson Bay.

```
lm(formula = Chl ~ plume_sum, data = Daly)

Residuals:
    Min      1Q  Median      3Q     Max
-1.0754 -0.4453 -0.3031  0.3684  3.1075

Coefficients:
             Estimate Std. Error t value Pr(>|t|)
(Intercept) 1.8776906  0.5068408   3.705   0.0015 **
plume_sum   0.0014845  0.0005541   2.679   0.0148 *
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.9133 on 19 degrees of freedom
Multiple R-squared:  0.2742,    Adjusted R-squared:  0.236
F-statistic: 7.178 on 1 and 19 DF,  p-value: 0.01484
```

Table A 2.6. Model summary of relationship between the Daly River tertiary flood plume size and mean Chlorophyll-a (lagged) in Anson Bay.

```
Call:
lm(formula = chl_lag ~ Tertiary, data = Daly)

Residuals:
     Min       1Q   Median       3Q      Max 
-0.73542 -0.32437 -0.02718  0.38211  0.74860 

Coefficients:
             Estimate Std. Error t value Pr(>|t|)    
(Intercept) 1.9755032  0.2692811   7.336 2.49e-05 ***
Tertiary    0.0014838  0.0004548   3.263  0.00854 ** 
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.4932 on 10 degrees of freedom
  (9 observations deleted due to missingness)
Multiple R-squared:  0.5156,	Adjusted R-squared:  0.4672 
F-statistic: 10.64 on 1 and 10 DF,  p-value: 0.008538
```

Table A 2.7. Model summary of relationship between river flood plume extent (sum of primary, secondary, and tertiary plumes) and 7-day flow events in the Flinders River.

```
Call:
lm(formula = plume_sum ~ Flow7, data = F)

Residuals:
   Min     1Q Median     3Q    Max 
-858.7 -424.4  100.0  246.4 1247.8 

Coefficients:
             Estimate Std. Error t value Pr(>|t|)    
(Intercept) 6.780e+02  1.653e+02   4.101 0.000608 ***
Flow7       1.412e-03  1.300e-04  10.860 1.37e-09 ***
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 550.1 on 19 degrees of freedom
Multiple R-squared:  0.8613,	Adjusted R-squared:  0.854 
F-statistic: 117.9 on 1 and 19 DF,  p-value: 1.371e-09
```

Table A 2.8. Model summary of relationship between river flood plume extent (sum of primary, secondary, and tertiary plumes) and 7-day flow events in the Gilbert River.

```
Call:
lm(formula = Plume_sum ~ flow7, data = G)

Residuals:
   Min     1Q Median     3Q    Max 
-463.8 -375.1  105.0  203.3  564.4 

Coefficients:
             Estimate Std. Error t value Pr(>|t|)   
(Intercept) 4.364e+02  4.041e+02   1.080  0.32175   
flow7       1.693e-03  2.926e-04   5.786  0.00117 **
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 398.5 on 6 degrees of freedom
Multiple R-squared:  0.848,	Adjusted R-squared:  0.8227 
F-statistic: 33.48 on 1 and 6 DF,  p-value: 0.001166
```

Table A 2.9. Model summary of relationship between river flood plume extent (sum of primary, secondary, and tertiary plumes) and 7-day flow events in the Daly River.

```
Call:
lm(formula = plume_sum ~ ML, data = D)

Residuals:
    Min      1Q  Median      3Q     Max
-506.81 -402.29   54.83  344.52  595.74

Coefficients:
             Estimate Std. Error t value Pr(>|t|)
(Intercept) 1.938e+02  1.639e+02   1.183    0.252
ML          7.761e-04  1.150e-04   6.751 1.89e-06 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 396.1 on 19 degrees of freedom
Multiple R-squared:  0.7058,    Adjusted R-squared:  0.6903
F-statistic: 45.57 on 1 and 19 DF,  p-value: 1.892e-06
```

3. Mangrove extent changes and links to changes in environmental conditions in Gulf of Carpentaria

3.1. Introduction

Large-scale loss of coastal wetland ecosystems has been regularly reported in the media and literature as a direct response to land use changes and clearing for urban and industrial expansion, agricultural and aquaculture enterprise growth and through shoreline modifications for the purposes of navigation or shipping (Murray et al. 2022; Yin et al. 2021). There are also indirect drivers of coastal wetland loss because of land use change altering flow and hydrology causing coastal erosion, or the most obvious, changing climate conditions which have caused massive loss of coastal vegetation areas (Bugnot et al. 2021). These losses are important to humans because of the direct ecosystem services that coastal wetlands provide not only in terms of cultural heritage and supporting biodiversity and species assemblages, but also for the economic opportunities such as eco-tourism or commercial fisheries. The loss of these coastal ecosystems continues to be a major challenge for managers who are responsible for both the approval of further direct loss due to development expansion and the conservation and protection of these ecosystems at the same time (Bell-James et al. 2024; Piccolo et al. 2024; Rummell et al. 2023).

Tackling the loss of coastal wetlands has been met with global initiatives that call to halt further loss of these ecosystems and to begin to restore them, which has been outlined in several United Nations declarations. While these initiatives have been important, they are also perhaps ambitious, with a range of limitations and barriers currently in place, including legislation approvals and the funding short fall between available funding and that which is necessary. Nonetheless, efforts are underway in many places to restore coastal ecosystems, such as mangroves, seagrass and tidal marshes, with varying levels of success (Canning et al. 2021; Piccolo et al. 2024; Saunders et al. 2024). While these restoration efforts are necessary, with the learnings hopefully useful to guide future projects to maximise success, the challenges will still remain, particularly when considering the unknown role that future climate change will have on restoration and its success.

Mangroves are a major ecosystem in coastal settings within subtropical regions for the role they play in stabilising shoreline areas, carbon capture and long-term storage in above and below-ground reserves, processing sediments and nutrients in waterways, and the habitat provided for a range of aquatic and terrestrial species. They are, however, dynamic plants, given their location within the intertidal zone along coastal areas, floodplains and estuaries in transitional areas (Robertson and Duke 1990).

There has been a recent focus on major water resource development and expansion of agriculture as part of an agenda to increase development prospects in northern Australia. This could potentially cause a reduction of freshwater availability and flow into adjacent or downstream environments. Given the important role of mangrove ecosystems and their specialised needs, it is crucial to understand how they will be affected by these changes in water management.

Using a multidecadal mangrove dynamics dataset developed by Digital Earth Australia (DEA), the aim of our study was to identify and investigate potential environmental drivers, such as river flow and rainfall, on the growth and canopy composition of mangrove forests along the Eastern coast of the Gulf of Carpentaria (GoC). We compared mangrove canopy density to river flow to assess whether the two were correlated and to assess whether decreases in base flow that may occur under an increased irrigation extraction scenario are likely to impact on mangroves in the GoC. We also arranged the data by ‘wetness of year’ for each of the study catchments to identify whether there was a clear pattern of lower mangrove canopy cover during low flow years. Overall, we did not find a strong relationship between mangrove canopy density and river flow (for the areas examined in this study), with mangrove canopy cover changes more likely to be dominated by regional sea level fluctuation and tropical cyclones that cross through the region. These analyses provide data useful in the assessment of water resource development and water plan reviews proposed in the eastern Gulf region, which is planned over the coming years by the Queensland Government.

3.2. Methods

3.2.1. Study area

The focus of this investigation was on the eastern catchments of the Gulf of Carpentaria, Queensland, Australia, including the Flinders River catchment (extending to the Morning River catchment), Norman River catchment, Gilbert Rivers catchment, Staaten River catchment, and the Mitchell River catchment. Catchments with small sections of coastline were merged with larger catchments (e.g., Staaten and Gilbert; Norman, Morning, and Flinders) to result in three main catchments – Mitchell, Gilbert, and Flinders (Figure 3.1). Spanning across approximately 330km of coastline, these catchments support approximately 213km^2 of mangrove habitat which has been experiencing a long-term expansion both inland and along the coast (Saintilan et al. 2022) until a large-scale dieback in 2015, with losses of approximately 7400 ha or 6% (Duke et al. 2017).

Most of the study area falls within the Gulf Plains bioregion and two climate zones – tropical and grassland. The area typically experiences a tropical monsoonal climate with a distinct hot and humid wet season and a cooler dry season (CAFNEC) but is considered semi-arid or a drier tropical area of Australia (Duke et al. 2017). Annual temperatures range from approximately 16 to 37°C and mean annual rainfall ranges from 265mm around Karumba up to 363mm in Kowanyama (BoM).

The recent plan for changes in water management and increased agriculture development within the region is concerning as the Gilbert River area has already been under the stress of irrigated agriculture for the last fifty years (Karim et al. 2015; Petheram et al. 2018; Petheram and Yang 2013). It is also of significant concern for both recreational and commercial fisheries that depend on the health of the local watershed. The area supports a major commercial fishery where a number of fisheries-targeted species have closely coupled lifecycle ecology processes to flood flows and access to coastal wetland ecosystems, such as prawns, barramundi and mud crabs (Leahy et al. 2022; Robins et al. 2005), with a gross value of production of $91.7 million for the Northern Prawn Fishery for the 21/22 fiscal year and $23.6m for the Gulf of Carpentaria Inshore Fishery for the 19/20 fiscal year (Webley and Probst 2020).

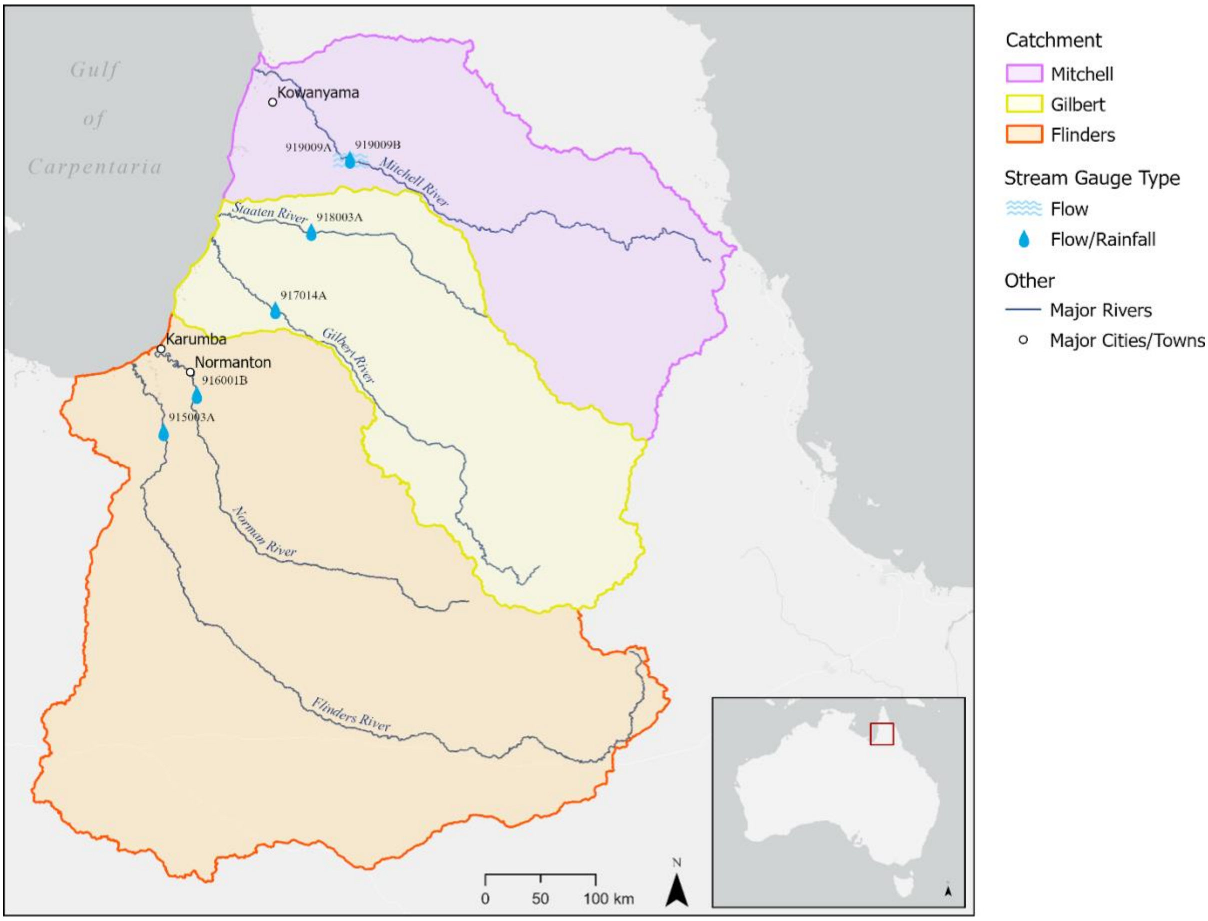


Figure 3-1. Mangrove extent changes and links to changes in environmental conditions in the Gulf of Carpentaria investigated in this study. Shown are the locations of the flow/rainfall stations used in the mangrove cover analysis for each catchment. Data provided by the State of Queensland Department of Natural Resources, Mines and Energy 2019 (watercourses) and Department of Environment, Science, and Innovation, Queensland 2016 (drainage basins).

3.2.2. Mangrove extent change analysis

3.2.2.1. DEA Mangroves dataset

Observable changes in mangroves over time were assessed using the DEA mangrove dataset outlined in Lymburner et al. (2020) which characterises the maximum mangrove areal extent and canopy coverage across Australia from 1987 to 2016. By the time of this study, the dataset extended to 2022. For quantifying the extent and density of canopy cover, the initial boundaries of mangrove extent were defined using a subset of data from the Global Mangrove Watch (Bunting et al., 2018) to focus the analysis. Satellite data from available observations of the Landsat 5 TM, Landsat 7 ETM, and Landsat 8 OLI were provided by the DEA as analysis-ready datasets (i.e., satellite data has been corrected, standardised, and orthorectified (Krause et al. 2021)). Using the Landsat fractional cover, the 10th percentile green photosynthetic fraction (GV_{10}) was calculated for each year, which identified vegetated areas that were consistently green throughout the year; a distinctive characteristic of mangroves (Lymburner et al. 2020). Canopy cover, or Planimetric Canopy Cover (PCC%), is considered as “the proportion of the forest floor covered by the vertical projection of the tree crowns” (Korhonen and Pihlanto 2006) and is summed over a unit area (Lymburner et al. 2020). PCC% was determined using LiDAR ≤ 1m resolution canopy height models (CHMs) at various sites across

Australia. The 50% of the CHMs were used for validation and the remaining 50% were used to link GV_{10} with PCC%, which involved quantifying PCC% within Landsat pixels on which the GV_{10} was projected, resulting in annual Landsat PCC% estimates (Lymburner et al. 2020). Using similar classification methods of Australia's State of the Forest Report (2013), PCC% was classified by forest type into three classes - Woodland (20-50% cover), Open Forest (30-80% cover), and Closed Forest (>80% cover).

3.2.2.2. Assessing changes in canopy coverage

The DEA provides numerous Python-based workflow notebooks to work within their programming environment, DEA Sandbox (https://www.ga.gov.au/scientific-topics/dea/for-developers/introduction-to-the-dea-sandbox), which utilises Jupyter Notebooks for analysing both the spatial data that they provide as well as external datasets. For this study, we adapted the workflow of the Introduction to DEA Mangroves notebook. Using the DEA Mangroves dataset, all mangrove extent within the outlined area was queried for the years falling between 1988 and 2022. Focus areas were a regional area encompassing mangroves directly around the mouth of each major river and the larger river catchments. Areas around each river mouth ranged from approximately 22 to 68 km^2. The area extent was intended to capture mangroves that we hypothesised would be directly impacted by changes in river discharge. To limit memory usage in the DEA Sandbox, a reduced catchment outline that still captured the inland extent of the mangrove dataset was used when querying data for the overall catchment analyses (Supplementary 11). With the resulting mangrove extents, the total area (km^2) of each canopy cover classification was calculated for every catchment and river mouth area annually from 1988 to 2022 (Section 3.2).

Substantial variations in mangrove dynamics across both the overall catchment and river mouth areas were investigated. This approach enabled the observation of trends in coastal mangroves, particularly those directly affected by river discharge, while also capturing the broader impacts on upstream mangroves and their influence on the general catchment. Moreover, differences in trends between the catchments were observed.

3.2.3. Environmental data

Annual mangrove extent and variations in forest type were assessed alongside average flow rates and total rainfall data collected at stream gauge stations on each of the major rivers (Section 3.1). Stations were selected based on closest proximity to the river mouth to capture the most representative flow rate the coastal mangroves were receiving. The following stations were selected: Flinders River at Walkers Bend (915003A; -18.161675, 140.8582), Norman River at Glenore Weir (916001B; -17.860025, 141.128726), Gilbert River at Burke Development Road (917014A; -17.168245, 141.767484), Staaten River at Dorunda (918003A; -16.531488, 142.059679), Mitchell River at Dunbar (919009B; -15.942376, 142.374261), and Mitchell River at Koolatah (919009A; -15.9509, 142.3772). Two stations were used for the Mitchell River as they were within 1km of each other and thus considered representative of the same area. Data from station 919009B ranged from 1987 to 2010, while data from station 919009A ranged from 2009 to 2023. Due to the large gap of missing data in the 919009B dataset, data from the 919009A station was used to cover the missing period. Any overlapping data was averaged.

Stream flow data was obtained from the Water Monitoring Information Portal (WMIP; https://water-monitoring.information.qld.gov.au/) managed by the Queensland Government for all but one station, Mitchell at Koolatah, which was obtained from the Bureau of Meteorology (BoM) Water Data Online portal (http://www.bom.gov.au/waterdata/). All rainfall data was obtained from the Queensland Government Water Monitoring Information Portal due to higher data quality based on the QA assessment in comparison to the BoM Water Data Online datasets.

Stream flow discharge and rainfall data were assessed for additional quality control using the QA codes provided with the datasets. Data that was missing, deemed less than "Fair" quality, or rated lower than a quality code "C" were removed prior to analysis, unfortunately resulting in large data gaps (seen in Section 3.1). To compensate for these gaps and to remove seasonality, the discharge datasets were averaged per calendar year during analysis. Additionally, discharge data was averaged both per calendar year and water year to account for any differences that may have arisen (Supplementary 1-5).

To identify relationships between canopy extent and river discharge, mangrove extent datasets were ranked annually based on the average discharge recorded for each river, from the year with the lowest average discharge to the year with the highest (see Section 3.3). To further investigate these relationships, linear models were applied to analyse canopy classification and total extent against the average discharge per year (Supplementary 6-10).

3.3. Results

3.3.1. Flow and rainfall

3.3.1.1. Flinders River Catchment

Rainfall and flow discharge recorded in the Flinders River catchment are presented in Figure 3.2. These results outline a strongly seasonal pattern with most of the rainfall and flow occurring during the wet season months, approximately December to March each year. There is also a pattern of strong interannual variability in rainfall and, thereby, flow. Major rainfall years and flow are interspersed among multiple years of lower rainfall and much lower average discharge volumes recorded across this time series. The 2009 wet season was the highest in this time series, while since 2009, rainfall has been similar from year to year but flow discharge has generally been much lower.

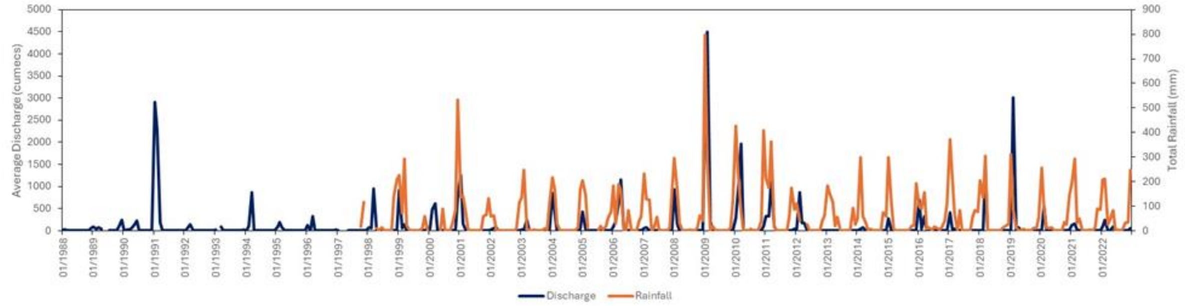


Figure 3-1. Monthly discharge and rainfall recorded in Flinders River catchment.

3.3.1.2. Norman River Catchment

Rainfall and flow discharge recorded in the Norman River catchment are presented in Figure 3.3. These results outline a strongly seasonal pattern with most of the rainfall and flow occurring during the wet season months, approximately December to March each year. There is also a pattern of strong interannual variability in rainfall and, thereby, flow. Major rainfall years and flow are interspersed among multiple years of lower rainfall and much lower average discharge volumes recorded across this time series. The 2009 wet season was the highest in this time series, while since 2009, rainfall has been similar from year to year but flow discharge has generally been much lower. There were also several years in a row with very low discharge flow, particularly between 1992 and 1998, and 2001 and 2006 (unfortunately, no rainfall records are available at the station during this time).

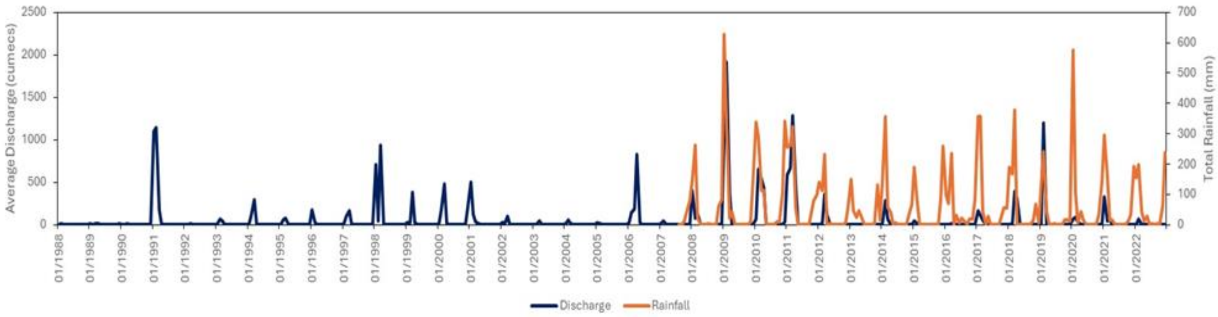


Figure 3-1. Monthly discharge and rainfall recorded in Norman River catchment.

3.3.1.3. Gilbert River Catchment

Rainfall and flow discharge recorded in the Gilbert River catchment are presented in Figure 3.4. These results outline a strongly seasonal pattern with most of the rainfall and flow occurring during the wet season months, approximately December to March each year. There is also a pattern of strong interannual variability in rainfall and, thereby, flow. For example, the 2014 and 2015 wet seasons were much lower when compared to the 2017 and 2020 wet seasons.

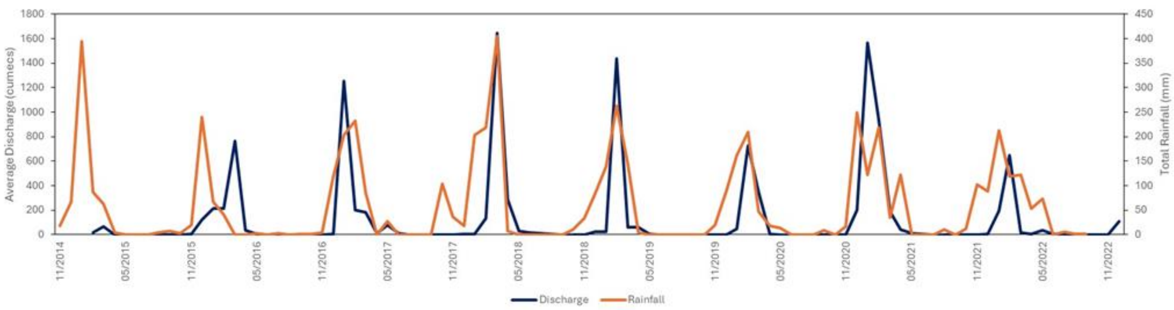


Figure 3-1. Monthly discharge and rainfall recorded in Gilbert River catchment.

3.3.1.4. Staaten River Catchment

Rainfall and flow discharge recorded in the Staaten River catchment are presented in Figure 3.5. These results outline, once again, a strongly seasonal pattern with most of the rainfall and flow occurring during the wet season months, approximately December to March each year. There is also a pattern of strong interannual variability in rainfall and, thereby, flow. Major rainfall years and flow are interspersed among multiple years of lower rainfall and much lower average discharge volumes recorded across this time series. There were several years in a row with very low discharge flow, particularly between 1994 and 1996, and 2014 and 2016. Unfortunately, there are no rainfall records available at the station prior to 2003.

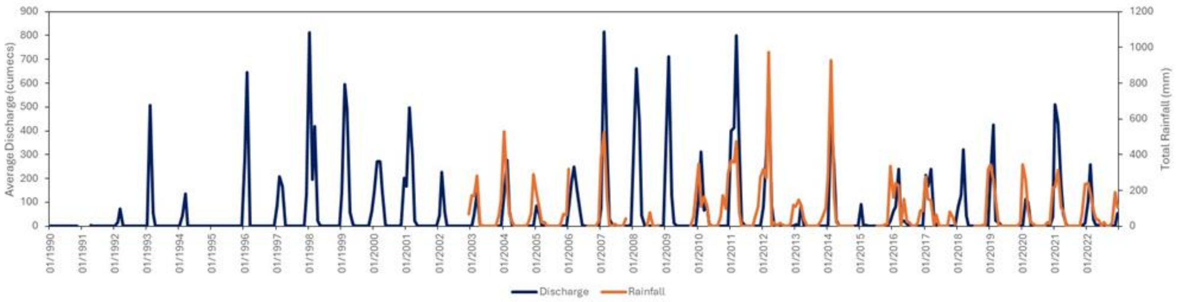


Figure 3-1. Monthly discharge and rainfall recorded in Staaten River catchment.

3.3.1.5. Mitchell River Catchment

Rainfall and flow discharge recorded in the Mitchell River catchment are presented in Figure 3.6. These results also show a strongly seasonal pattern with most of the rainfall and flow occurring during the wet season months, approximately December to March each year. There is also a pattern of strong interannual variability in rainfall and, thereby, flow. Major rainfall years and flow are interspersed among multiple years of lower rainfall and much lower average discharge volumes recorded across this time series. The 2008 wet season was the highest in this time series, while since 2010, rainfall has been generally similar from year to year but flow discharge has been much lower. There were also several years in a row with relatively low discharge flow, particularly between 2002 and 2007, and 2012 and 2019. Unfortunately, rainfall data at the station was not available prior to 2010.

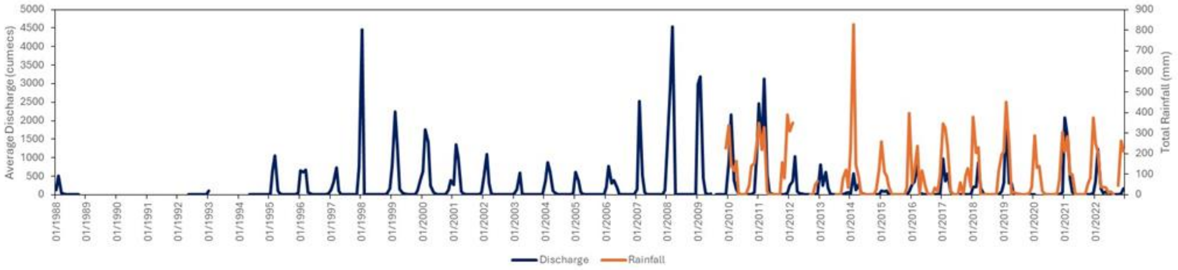


Figure 3-1. Monthly discharge and rainfall recorded in Mitchell River catchment.

3.3.2. Mangrove canopy cover 1988 to 2022

3.3.2.1. Flinders River Catchment

Annual changes in canopy cover and area of mangroves in the Flinders River estuary are shown in Figure 3.7. Within this study area, the area of woodland ranged between 0.16 km^2 and 1.09 km^2, open forest ranged between 0.17 km^2 and 1.5 $km^{2,}$ while closed forest was generally lower than open forest each year, ranging between 0 and 1.06 km^2. Open forest and woodland cover areas were generally similar from year to year, while closed forest was more variable from year to year, with major increases in extent between 1998 to 2003, 2004 to 2007, 2008 to 2014, and after 2016. For the years between these increases in extent, there were obvious major reductions in the extent of closed forest, with a corresponding increase in woodland extent. Over the past decade, however, the extent of woodland and open woodland has remained consistent.

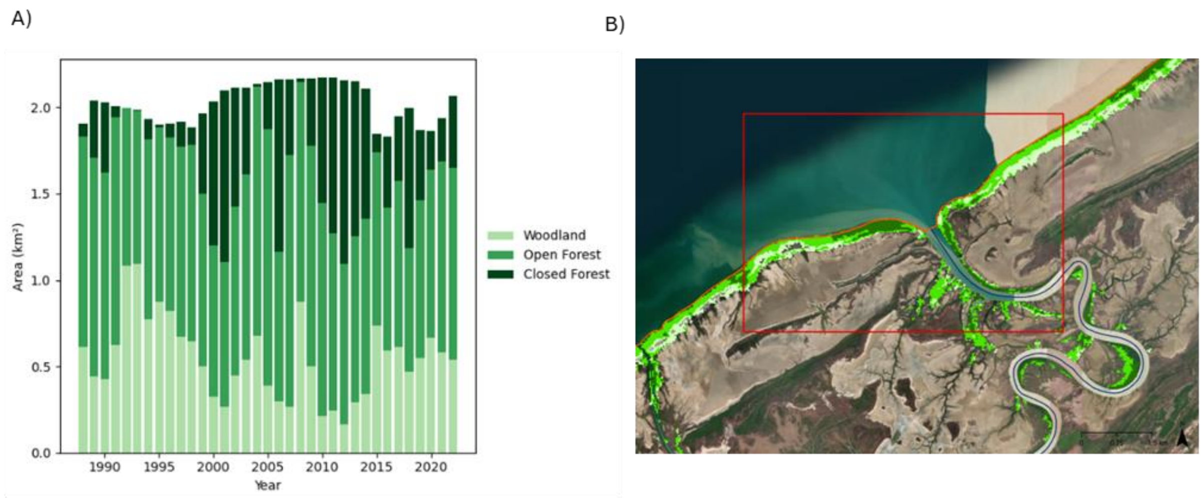


Figure 3-1. a) Annual mangrove canopy area cover classes (km^2) between 1988 and 2022; and b) mangrove canopy extent in Flinders River estuary.

3.3.2.2. Norman River Catchment

Annual changes in canopy cover and area of mangroves in the Norman River estuary are shown in Figure 3.8. Within this study area, the area of woodland ranged between 0.18 km^2 and 1.03 km^2, open forest ranged between 0.83 km^2 and 1.7 km^2, while closed forest was generally lower than open forest ranging between 0.1 and 1.7 km^2. Open forest and woodland cover areas were generally similar from year to year, while closed forest was more variable from year to year.

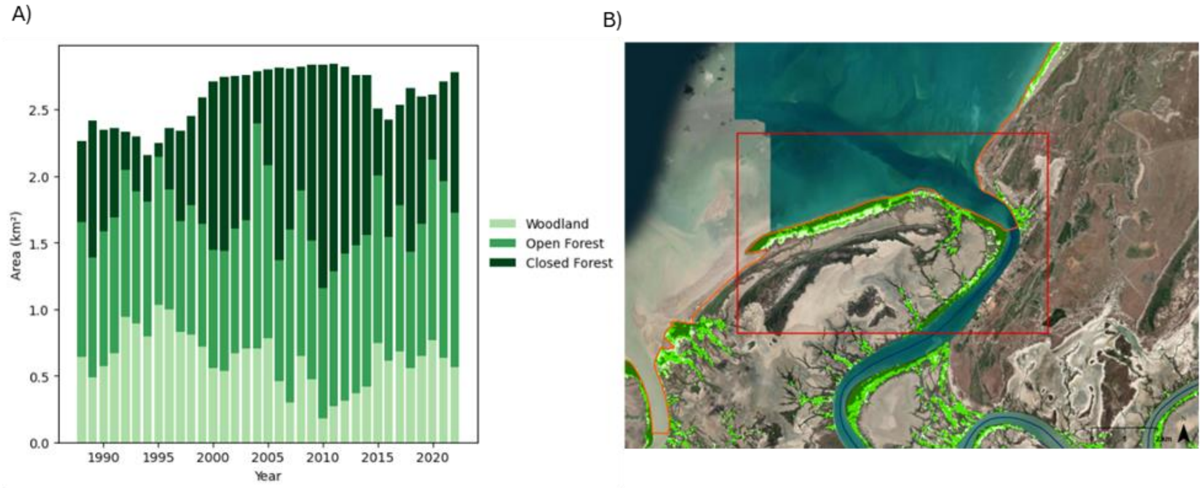


Figure 3-1. a) Annual mangrove canopy area cover classes (km^2) between 1988 and 2022; and b) mangrove canopy extent in Norman River estuary.

3.3.2.3. Flinders, Morning and Norman Catchment

Annual changes in canopy cover and area of mangroves in the catchment area consisting of the Flinders, Morning, and Norman catchments are shown in Figure 3.9. Within this study area, the area of woodland ranged between 15.1 and 32.8 km^2, open forest ranged between 16.9 and 36.4 km^2, while closed forest was generally lower each year, ranging between 0.75 and 21.9 km^2. Open forest and woodland cover areas were generally similar from year to year, while closed forest was more variable from year to year. There were obvious impacts of a cyclone in 1995 that resulted in a reduction of closed mangrove forests which occurred over the subsequent three years, and loss of closed forests in 2015 associated with the major dieback in the region, though the closed forest cover increased rapidly to be more similar to area extent prior to the dieback. There also seems to be a slow increasing tread in closed forest extent over the past 10 years, which is potentially replacing woodland area.

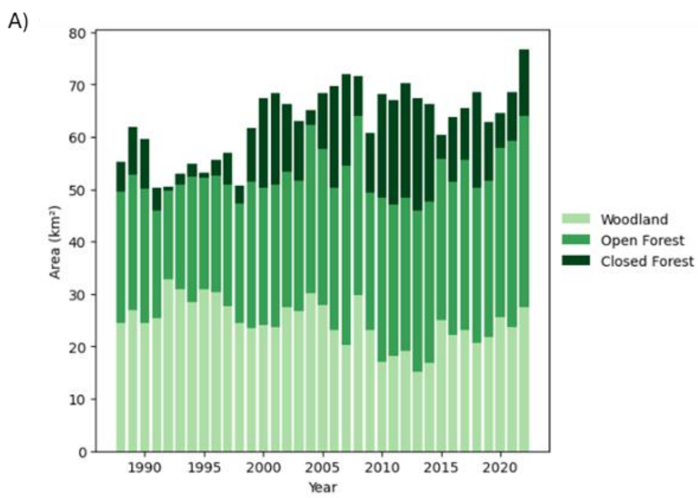
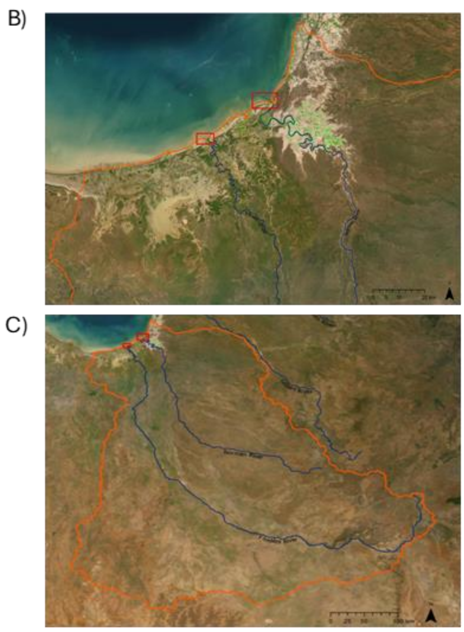


Figure 3-1. a) Annual mangrove canopy area cover classes (km^2) between 1988 and 2022; and b) mangrove canopy extent in Norman River catchment.



3.3.2.4. Gilbert River Catchment

Annual changes in canopy cover and area of mangroves in the Gilbert River estuary are shown in Figure 3.10. Within this study area, open forest area ranged between 0 and 1.24 km^2 and the area of closed forest ranged between 0 and 1.23 km^2, while woodland was generally lower each year, ranging between 0.03 and 1.0 km^2. There were obvious impacts of a cyclone in 1995 that resulted in the massive loss of open and closed mangrove forests, an overall extent of approximately 69%, and the loss of closed forests in 2016 associated with the major dieback event.

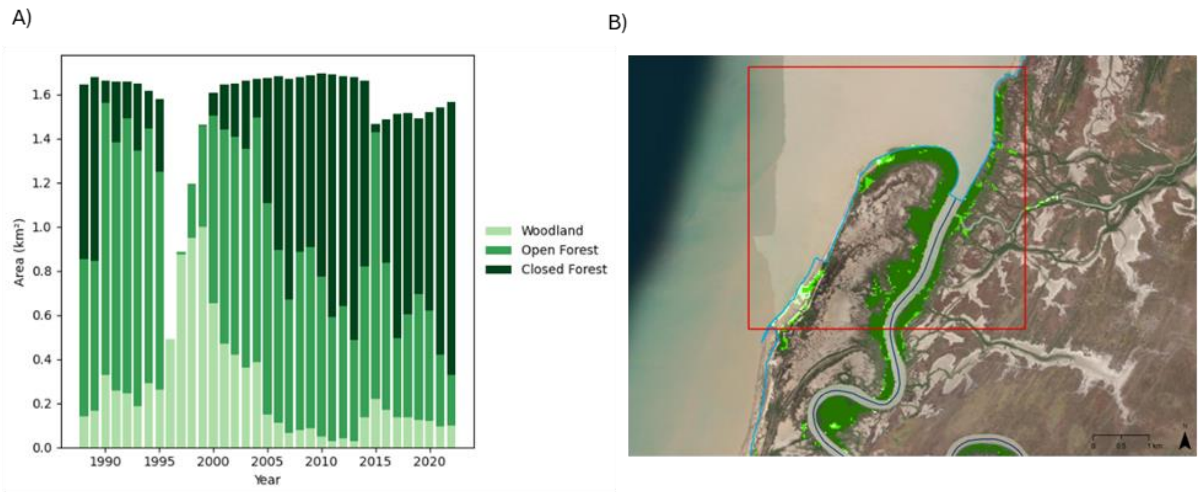


Figure 3-1. a) Annual mangrove canopy area cover classes (km^2) between 1988 and 2022; and b) mangrove canopy extent in Gilbert River estuary.

3.3.2.5. Staaten River

Annual changes in canopy cover and area of mangroves in the Staaten River estuary are shown in Figure 3.11. Within this study area, the area of closed forest ranged between 0 and 0.55 km^2, open forest ranged between 0.01 and 0.53 km^2, while woodland was generally lower each year, ranging between 0.02 and 0.56 km^2. There was a major loss of closed forest in 1995, with a reduction of approximately 60%, until 1999, when it had started to recover. During this same period, the open forest area was significantly reduced and slowly recovered. Following the year 1999, there has been an obvious reduction in mangrove woodland extent, which has been gradually replaced with an increase in areas of open and closed mangrove forest. There was also a slight reduction in closed forest in 2015, likely due to the dieback event, but this quickly increased again.

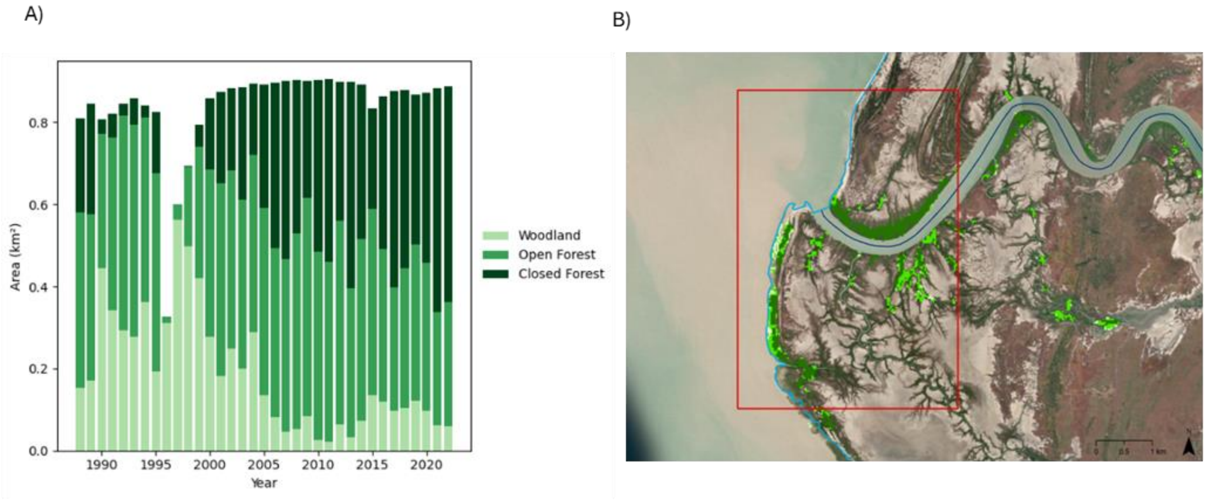
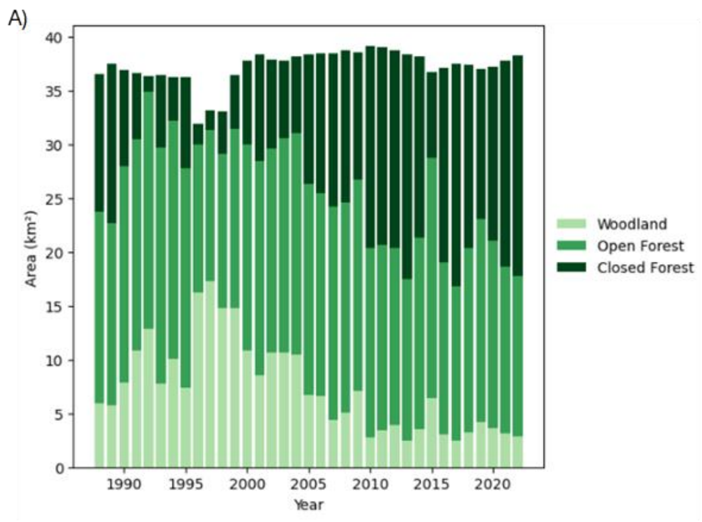


Figure 3-1. a) Annual mangrove canopy area cover classes (km^2) between 1988 and 2022; and b) mangrove canopy extent in Staaten River estuary.

3.3.2.6. Gilbert and Staaten Catchment

Annual changes in canopy cover and area of mangroves in the catchment area consisting of the Gilbert and Staaten catchments are shown in Figure 3.12. Within this study area, the area of open forest ranged between 13.73 and 22.37 km^2, closed forest ranged between 1.50 and 20.80 km^2, and woodland was generally lower, ranging between 2.45 and 17.28 km^2. There were obvious impacts of a cyclone in 1995 that resulted in a substantial reduction of closed and open mangrove forests which occurred over the subsequent three years. Additional loss of closed forests occurred in 2015 associated with the major dieback in the region, though the closed forest cover increased rapidly to a similar extent prior to the dieback. There also seems to be a slowly decreasing trend in woodland extent, which is becoming replaced with closed forest, with open woodland remaining relatively similar since 1995.



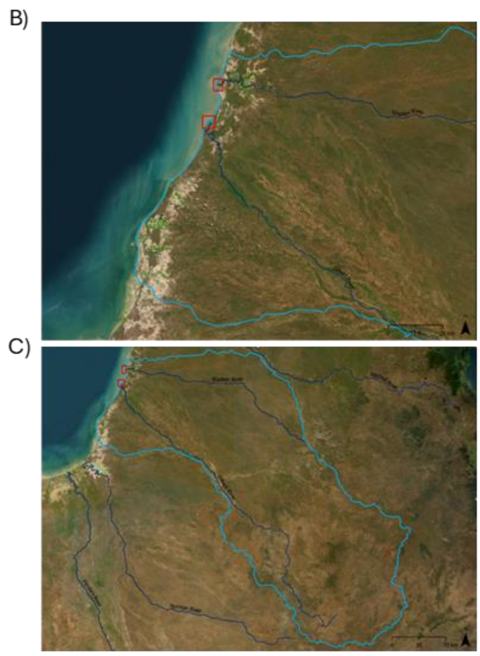


Figure 3-1. a) Annual mangrove canopy area cover classes (km^2) between 1988 and 2022; and b) mangrove canopy extent in Gilbert and Staaten catchment area.

3.3.2.7. Mitchell River Catchment

Annual changes in canopy cover and area of mangroves in the Mitchell River estuary are shown in Figure 3.13. Within this study area, the area of closed forest ranged between 0.37 and 2.79 km^2, open forest ranged between 0.95 and 2.41 km^2, while woodland was generally lower, ranging between 0.09 and 0.85 km^2. There was a significant loss in closed forest area from 1993 until 1995, after which it increased. During this same period, the woodland area increased, though it has slowly been reducing in extent since 1997. In the years after 1997, there has been a gradual increase in closed mangrove forest extent, with open forest generally having a steady, consistent trend. There was no apparent change in mangrove coverage for all categories in 2015.

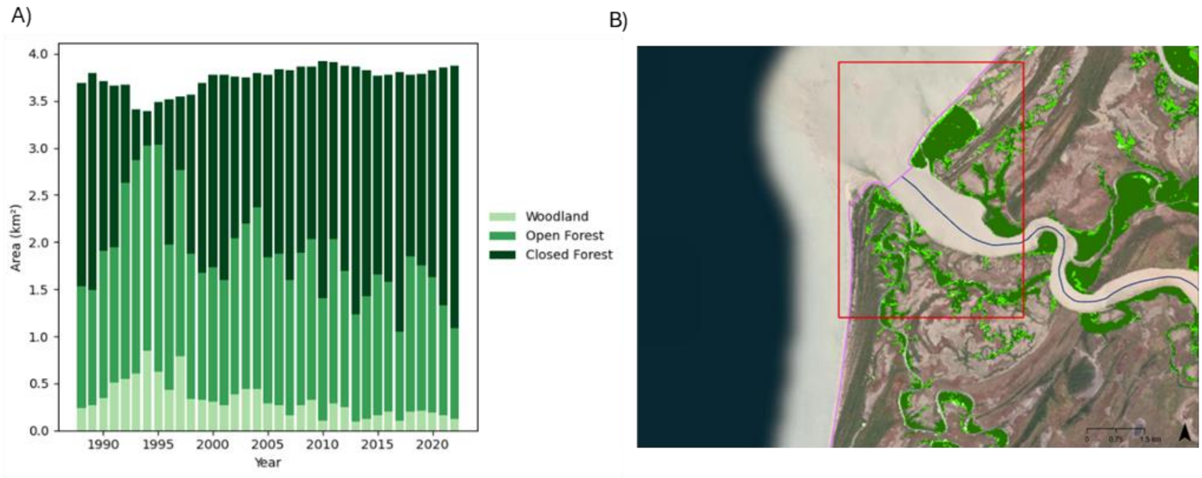


Figure 3-1. a) Annual mangrove canopy area cover classes (km^2) between 1988 and 2022; and b) mangrove canopy extent in Mitchell River estuary.

Annual changes in canopy cover and area of mangroves in the Mitchell catchment are shown in Figure 3.14. Within this study area, the area of closed forest ranged between 8.48 and 36.41 km^2, open forest ranged between 17.54 and 31.64 km^2, while woodland was generally lower each year, ranging between 2.10 and 13.65 km^2. Open and closed forest cover areas were generally similar from year to year, while woodland was consistently lower from year to year. There was a significant loss in closed forest from 1992 to 1994, but it steadily increased thereafter. While there was an increasing trend of woodland cover during that period, after 1994, woodland seemed to have a slow decline. Open forest cover remained relatively steady throughout the total period.

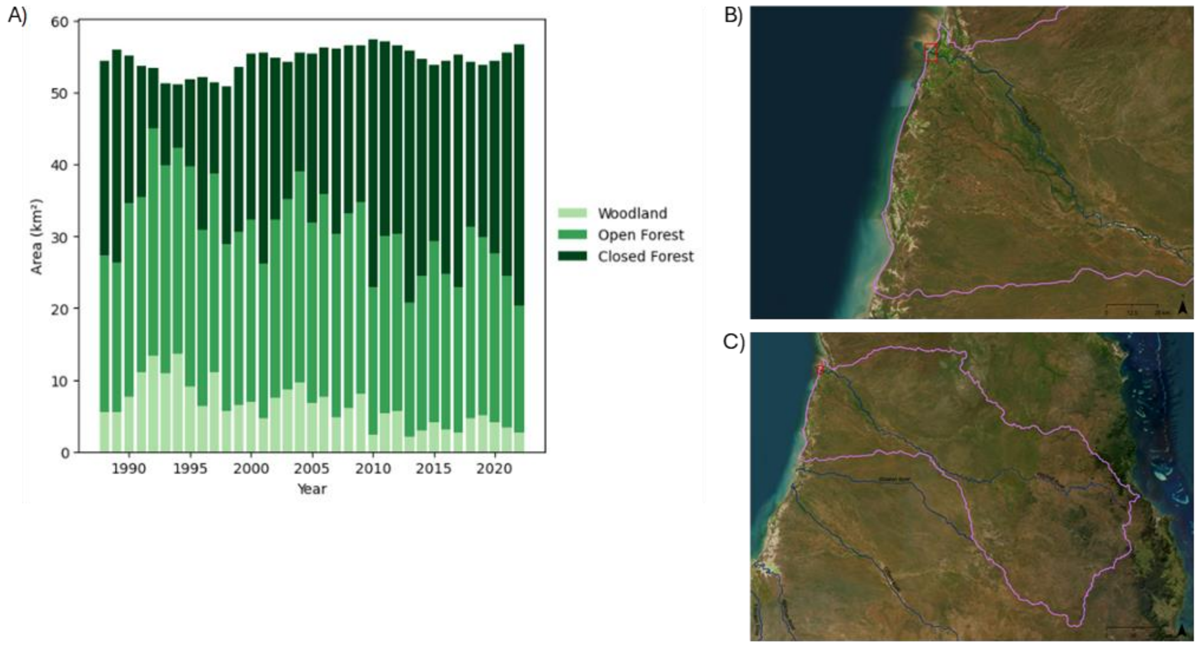


Figure 3-2. a) Annual mangrove canopy area cover classes (km^2) between 1988 and 2022; and b) mangrove canopy extent in Mitchell catchment area.

3.3.3. Relationship between canopy cover and discharge

3.3.3.1. Flinders River Catchment

The pattern of mangrove cover from year to year compared to annual discharge recorded at the gauging station in the Flinders catchment is presented in Figure 3.15. Here, the area (km^2) of mangrove woodland and open and closed forest for each year has been ordered from the year with the lowest recorded discharge to the highest. Comparing these environmental variables, there appears to be no apparent relationship between any mangrove area cover and discharge, despite interannual variability in mangrove area.

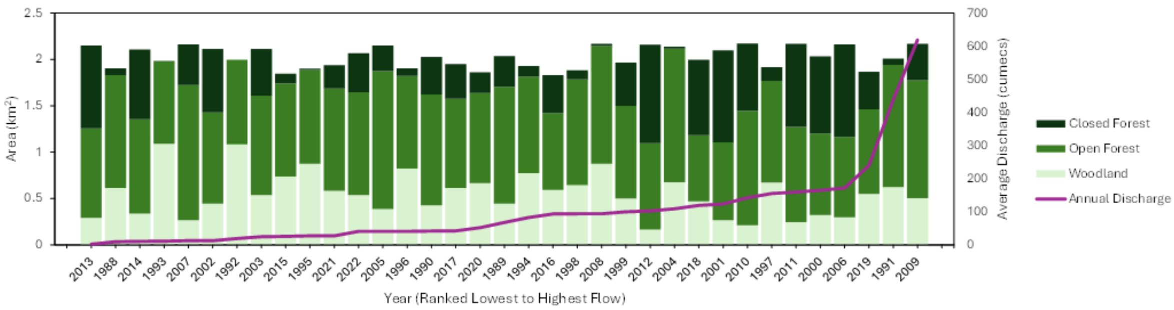


Figure 3-1. Mangrove forest areas (km^2) ranked lowest to highest according to average annual discharge in the Flinders River.

3.3.3.2. Norman River Catchment

The pattern of mangrove cover from year to year compared to annual discharge recorded at the gauging station in the Norman catchment is presented in Figure 3.16. Here, the area (km^2) of mangrove woodland and open and closed forest for each year has been ordered from the year with the lowest recorded discharge to the highest. Comparing these environmental variables, there appears to be no apparent relationship between any mangrove area cover and discharge, despite interannual variability in mangrove area.

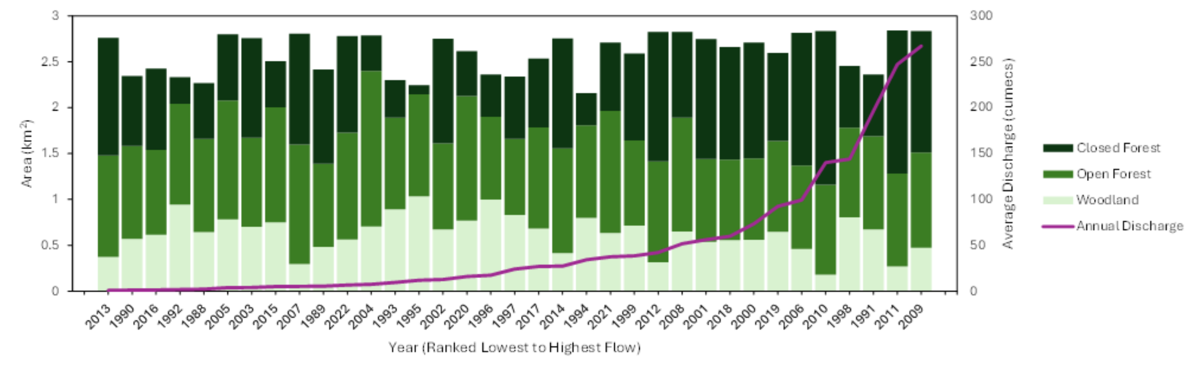


Figure 3-1. Mangrove forest areas (km^2) ranked lowest to highest according to average annual discharge in the Norman River.

3.3.3.3. Gilbert River Catchment

The pattern of mangrove cover from year to year compared to annual discharge recorded at the gauging station in the Gilbert catchment is presented in Figure 3.17. When the area (km^2) of mangrove woodland, open and closed forest for each year is ordered from the year with the lowest recorded discharge to the highest, there appears to be no apparent relationship between any mangrove cover and discharge, though the discharge data has only commenced in 2015. A longer dataset might reveal a different pattern in this catchment.

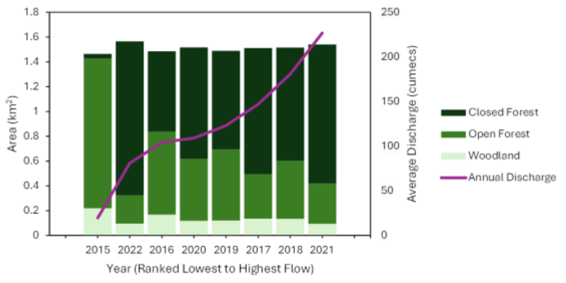


Figure 3-1. Mangrove forest areas (km^2) ranked lowest to highest according to average annual discharge in the Gilbert River.

3.3.3.4. Staaten River Catchment

The pattern of mangrove cover from year to year compared to annual discharge recorded at the gauging station in the Staaten catchment is presented in Figure 3.18. The ranking according to annual discharge, again, appears to show no apparent relationship with any mangrove area cover despite interannual variability in mangrove area. This is particularly the case in 1996, 1997 and 1998 where the extent of closed forest was zero, despite each year having a different annual discharge volume.

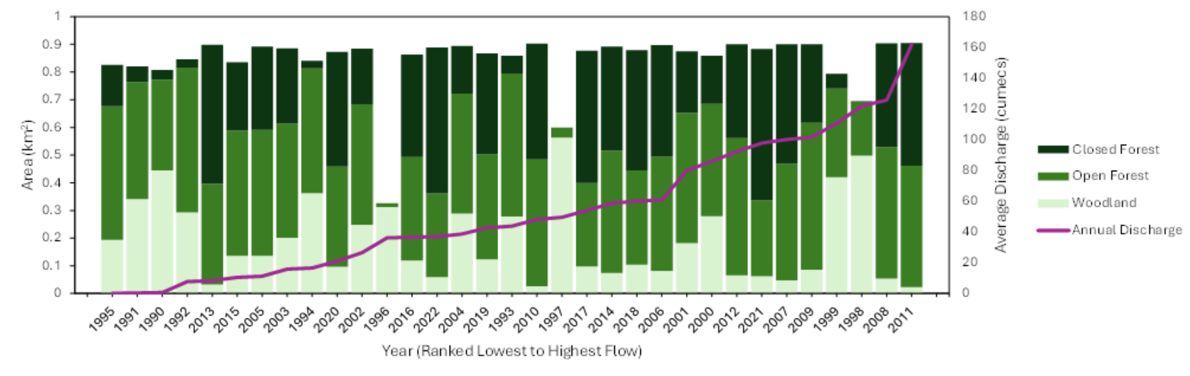


Figure 3-1. Mangrove forest areas (km^2) ranked lowest to highest according to average annual discharge in the Staaten River.

3.3.3.5. Mitchell River Catchment

The pattern of mangrove cover from year to year compared to annual discharge recorded at the gauging station in the Mitchell catchment is presented in Figure 3.19. The ranking of annual discharge, again, appears to show no apparent relationship with any mangrove area cover despite interannual variability in mangrove area. While it seems like some of the highest closed forest area records occur when annual discharge is greater than 200 cumecs, this is not actually the case, as similar closed forest areas were recorded during years when discharge flow was approximately 100 cumecs or lower.

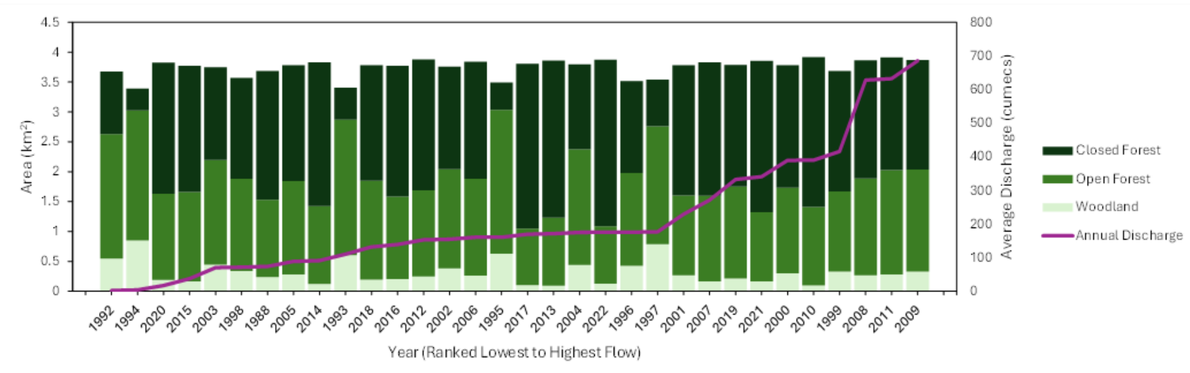


Figure 3-1. Mangrove forest areas (km^2) ranked lowest to highest according to average annual discharge in the Mitchell River.

3.4. Discussion

According to Australia's Department of the Environment and Energy (2016), “mangroves and saltmarshes have historically been undervalued and considered by many to be wastelands”, making them some of the least understood ecosystems along Australia's coasts (Cresswell et al. 2018). Major stress events, such as the 2015/2016 large-scale mangrove dieback in the GoC underscore the urgent need for continued research into these coastal systems. Closing significant data gaps and improving our understanding of the stressors affecting these ecosystems, along with the biology and resilience of mangroves, is essential for developing best management practices for future protection and preservation. The need becomes increasingly critical as ongoing rapid population growth and the impending impacts of climate change intensify these pressures (Rogers et al. 2016). Given the focus on major water resource development and expansion of agriculture in the eastern Gulf of Carpentaria region, understanding the influence of upstream discharge, rainfall, and other environmental factors on mangrove growth is especially crucial as mangroves along arid coasts have been predicted to feel these pressures most severely through increasing salinities, rising temperatures, and decreasing availability of freshwater (Alongi et al. 2016).

Our study concentrated on the importance of freshwater influx into the ecosystem via the region’s major watercourses, with river discharge and rainfall as our priority factors of interest. River discharge, as one of the most direct measures of freshwater input to the area, plays a vital role in sustaining healthy mangrove ecosystems. Most importantly for growth and resilience, it influences sediment deposition and salinity levels. Sediment that becomes suspended in river discharge facilitates substrate accretion downstream, potentially increasing the ability to contend with sea level rise (Pernetta 1993). It also allows for increased propagation of mangrove seedlings (Ellison 2019), which supports the ecosystem’s resiliency and recovery from disturbance. Furthermore, some mangroves, depending on their salt tolerance, are vulnerable to sudden increases in salinity, with hypersalinity potentially leading to mortality (Harris et al. 2017). Therefore, a reduction of flow downstream and, thus, influx of freshwater into the ecosystem could be detrimental.

The significance of discharge becomes even more apparent when considering its association with rainfall. While rainfall not only has a direct importance on mangrove health through precipitation, but also indirectly through runoff from river catchments (Ewel et al. 1998). It has been identified as a driver for species diversity (Robertson and Duke 1990), with areas receiving higher rainfall and high fluvial inputs of freshwater supporting greater numbers of species (Burford et al. 2010; Kenyon et al. 1999; Vance et al. 2002). This is of particular concern within the study area, as several studies found that the Gulf of Carpentaria hosted fewer than twenty species (Blaber et al. 1994; Brewer et al. 1994), despite surrounding areas, including New Guinea, containing higher levels of species richness. Further findings suggested that species richness was greatest in regions of moderate salinity, highlighting the essential role of freshwater input, particularly in terms of the “amount, duration, frequency and regularity of runoff“ (Ball and Luk 1998). Species richness may also contribute to overall ecosystem resilience. As Bernhardt and Leslie (2013) suggested, biological diversity in coastal marine ecosystems “increases the range of biological responses and the odds that species can compensate for one another if some are lost.” Additionally, greater diversity within communities has been shown to enhance recovery from disturbances (Bernhardt and Leslie 2013) through efficient resource use, leading to higher productivity (Duffy 2009).

The results of our study suggest no apparent relationship between river discharge nor rainfall with mangrove extent and forest composition – at least for the data that were available here for this analysis. While we did observe high extent values and significant proportions of closed forest during high discharge periods, it was not exclusive to the higher periods of discharge; it was equally likely to observe the same extent and closed forest coverage during times of low river discharges in the catchments examined. There was the potential for a lag effect, with growth or loss occurring after a period of significant discharge, however, this was not observed with the data available. Additionally, when assessing the linear relationship between areal extent and average discharge per year (Supplementary 6 – 10), only three models resulted in significant relationships – the woodland and closed forest for the Norman River estuary region ($p = 0.031$ and $p = 0.0095$, respectively), and the total area for the Mitchell River estuary region ($p = 0.013$). However, R^2 values were low (R2 = 0.13, 0.19, and 0.19, respectively). Despite observing distinct periods of loss and growth in both mangrove extent and composition, when compared with average discharge, the analyses suggested no evidence of these changes being directly linked to varying levels of discharge.

Overall, rainfall exhibited strong seasonal patterns with most occurring within the wet season months (approximately December to March), which corresponded with peaks of discharge. As such, we would expect a similar relationship between rainfall and mangrove dynamics as seen with discharge. However, due to the limitations of data availability with both rainfall and discharge, it would be premature to make any significant conclusions on the impacts of mangroves within the catchments.

Our study focused on obtaining data only from gauge stations closest to the river mouths as the discharge would be most representative of the flow the mangroves were receiving, but it resulted in several areas having significantly limited datasets; most notably, the Gilbert River, which only had eight years of recorded discharge data. This limitation reduced the timeframes of comparable data that could be used for analysis for each area. Other datasets were missing substantial gaps of data throughout the time period. While we tried to mitigate this by averaging annual discharge, it increased the probability of error in calculating the averages. the use of additional data from other stations could not only help to fill in the existing gaps and create a more robust dataset, but it could also present opportunities for expanding the study as well.

Significant changes in mangrove cover extent and/or composition were identified on the floodplains examined in this study. These negative changes appear closely linked to cyclones, and other climatic factors or fluctuations in environmental variables, such as sea level (Duke et al. 2017). The most prominent periods of decline were observed between years 1995 and 1998, as well as 2015 and 2016. The loss of overall mangrove extent, and more specifically, closed forest, between 1995 and 1998 suggested significant effects from the Severe Tropical Cyclone Barry; a Category 3 storm that made landfall between the Staaten and Gilbert rivers in early January 1996 (BoM). The latter decline is a well-known widespread mangrove dieback where canopy cover was reduced by approximately 6% in the Gulf of Carpentaria in 2015 due to prolonged periods of drought, elevated temperatures, and a significant drop in sea level (Duke et al. 2017). Changes were largely seen within the closed forest classification. These events suggest that major changes in mangrove canopy can be less attributed to a single stressor but likely in response to an additive interaction of stressors (i.e., higher temperatures increasing moisture loss, decreased rainfall, and more dramatic fluctuations in sea level), which could be exacerbated by a decline in the downstream flow of freshwater.

Nevertheless, our study highlighted the resilient and dynamic nature of mangrove ecosystems. While significant loss was observed primarily in the more coastal sections of mangroves, less loss was evident when considering the catchment as a whole (Figures 3.7 - 3.14). This suggests that mangroves upstream were less affected or perhaps that coastal mangroves are more vulnerable, emphasising the important role of sea level while also suggesting a potential relationship with freshwater influx in the ecosystem. Moreover, despite notable declines in canopy cover, recovery rates were relatively quick, with closed forest and extent achieving pre-disturbance coverage in less than five years. However, these recovery times followed isolated yet significant disturbances. More frequent disturbances could potentially extend these recovery periods.

DEA Mangroves proved to be an invaluable resource. Initial intentions were to utilise a Normalised Difference Vegetation Index (NDVI) time series for assessing canopy growth and composition. However, it was found that the 10th percentile green photosynthetic fraction (GV_{10}) provided more dependable mapping and addressed potential issues proposed with the NDVI. Concern lay with the variability presented while exclusively using NDVI caused by “the differences in canopy openness and the complex seasonality associated with overstorey and understorey vegetation” (Lymburner et al. 2019). The DEA Mangroves was limited, though, in that it is calculated per calendar year as opposed to water year, which would have been more relevant for our study; an advantage the NDVI time series would have given. However, little difference was seen when comparing calendar year and water year of the discharge data (see Appendix).

3.4.1. Future research

With present data gaps and limited data availability during this study, further research is necessary to have a firm understanding of the influence of environmental variables, specifically discharge and rainfall, on mangrove dynamics in the eastern Gulf of Carpentaria. For example, the DEA provides other resources, such as the DEA Waterbodies dataset, that could be used to assess percentages of wet areas along rivers over the same time period, which could be interpreted as equivalent to discharge volumes. Another potential direction would be to incorporate additional station gauges to interpolate downstream flow during the missing timeframes. This could also be used to develop a catchment-wide hydrological network, opening up opportunities for additional hypotheses.

Previous studies, such as Duke et al. (2017), have highlighted the significant role of sea level influencing mangrove density, often in conjunction with other environmental factors like rainfall. Therefore, investigating the relationship between sea level and river discharge and examining how inundation from both impacts changes in mangrove forests, would be beneficial.

By utilising another DEA dataset, the DEA Fractional Cover Percentiles Calendar Year, assessments could be made to evaluate the influence of discharge and rainfall on inland wetlands and upstream riparian vegetation. This approach offers a catchment-wide perspective of potential impacts of introduced agricultural development into the area.

3.5. References

Alongi, D., D. Murdiyarso, J. Fourqurean, J. Kauffman, A. Hutahaean, S. Crooks, C. Lovelock, J. Howard, D. Herr, and M. Fortes. 2016. Indonesia's blue carbon: a globally significant and vulnerable sink for seagrass and mangrove carbon. *Wetlands Ecology and Management* 24: 3-13.

Ball, J.E., and K.C. Luk. 1998. Modeling spatial variability of rainfall over a catchment. *Journal of Hydrologic Engineering* 3: 122-130.

Bell-James, J., R. Foster, N. Shumway, C.E. Lovelock, J. Villarreal-Rosas, C.J. Brown, D.A. Andradi-Brown, M.I. Saunders, N.J. Waltham, and J.A. Fitzsimons. 2024. The Global Biodiversity Framework's ecosystem restoration target requires more clarity and careful legal interpretation. *Nature Ecology & Evolution* 8: 840-841.

Bernhardt, J.R., and H.M. Leslie. 2013. Resilience to climate change in coastal marine ecosystems. *Annual review of marine science* 5: 371-392.

Blaber, S.J.M., D.T. Brewer, and J.P. Salini. 1994. Comparisons of fish communities of tropical estuarine and inshore habitats in the Gulf of Carpentaria, northern Australia. In Changes in Fluxes in Estuaries, ed. O. Olsen, 363-372.

Brewer, D.T., S.M. Blaber, D.A. Milton, and J.P. Salini. 1994. Aspects of the biology of Caranx bucculentus (Teleostei : Carangidae) from the Gulf of Carpenteria, Australia. *Australian Journal of Marine and Freshwater Research* 45: 413 - 427.

Bugnot, A., M. Mayer-Pinto, L. Airoldi, E. Heery, E. Johnston, L. Critchley, E. Strain, R. Morris, L. Loke, and M. Bishop. 2021. Current and projected global extent of marine built structures. *Nature Sustainability* 4: 33-41.

Burford, M.A., R.A. Kenyon, M. Whittle, and G. Curwen. 2010. River flow impacts on estuarine prawns in the Gulf of Carpentaria, 165: Fisheries Research and Development Corporation.

Canning, A.D., D. Jarvis, R. Costanza, S. Hasan, J.C. Smart, J. Finisdore, C.E. Lovelock, S. Greenhalgh, H.M. Marr, and M.W. Beck. 2021. Financial incentives for large-scale wetland restoration: Beyond markets to common asset trusts. *One Earth* 4: 937-950.

Duffy, J.E. 2009. Why biodiversity is important to the functioning of real-world ecosystems. *Frontiers in Ecology and the Environment* 7: 437-444.

Duke, N.C., J.M. Kovacs, A.D. Griffiths, L. Preece, D.J. Hill, P. Van Oosterzee, J. Mackenzie, H.S. Morning, and D. Burrows. 2017. Large-scale dieback of mangroves in Australia's Gulf of Carpentaria: a severe ecosystem response, coincidental with an unusually extreme weather event. *Marine and Freshwater Research* 68: 1816-1829.

Ellison, J.C. 2019. Biogeomorphology of mangroves. In *Coastal wetlands*, 687-715: Elsevier.

Ewel, K., R. Twilley, and J. Ong. 1998. Different kinds of mangrove forests provide different goods and services. *Global Ecology & Biogeography Letters* 7: 83-94.

Harris, T., P. Hope, E. Oliver, R. Smalley, J. Arblaster, N. Holbrook, N. Duke, K. Pearce, K. Braganza, and N. Bindoff. 2017. Climate drivers of the 2015 Gulf of Carpentaria mangrove dieback.

Karim, F., D. Dutta, S. Marvanek, C. Petheram, C. Ticehurst, J. Lerat, S. Kim, and A. Yang. 2015. Assessing the impacts of climate change and dams on floodplain inundation and wetland connectivity in the wet–dry tropics of northern Australia. *Journal of Hydrology* 522: 80-94.

Kenyon, R.A., M.D.E. Haywood, D.S. Heales, N.R. Loneragan, R.C. Pendrey, and D.J. Vance. 1999. Abundance of fish and crustacean postlarvae on portable artificial seagrass units: daily sampling provides quantitative estimates of the settlement of new recruits. *Journal of Experimental Marine Biology and Ecology* 232: 197-216.

Korhonen, H., and A. Pihlanto. 2006. Bioactive peptides: Production and functionality. *International dairy journal* 16: 945-960.

Krause, J.R., A. Hinojosa-Corona, A.B. Gray, and E. Burke Watson. 2021. Emerging sensor platforms allow for seagrass extent mapping in a turbid estuary and from the meadow to ecosystem scale. *Remote Sensing* 13: 3681.

Leahy, S.M., D.R. Jerry, B.B. Wedding, J.B. Robins, C.L. Wright, A. Sadekov, S. Boyle, D.B. Jones, S.M. Williams, and M.T. McCulloch. 2022. Barramundi origins: Determining the contribution of stocking to the barramundi catch on Queensland's east coast.

Lymburner, L., P. Bunting, R. Lucas, P. Scarth, I. Alam, C. Phillips, C. Ticehurst, and A. Held. 2020. Mapping the multi-decadal mangrove dynamics of the Australian coastline. *Remote Sensing of Environment* 238: 111185.

Murray, N.J., T.A. Worthington, P. Bunting, S. Duce, V. Hagger, C.E. Lovelock, R. Lucas, M.I. Saunders, M. Sheaves, and M. Spalding. 2022. High-resolution mapping of losses and gains of Earth's tidal wetlands. *Science* 376: 744-749.

Pernetta, J. 1993. *Mangrove forests, climate change and sea level rise: hydrological influences on community structure and survival, with examples from the Indo-West Pacific*: IUCN.

Petheram, C., J. Gallant, P. Stone, P. Wilson, and A. Read. 2018. Rapid assessment of potential for development of large dams and irrigation across continental areas: application to northern Australia. *The Rangeland Journal* 40: 431-449.

Petheram, C., and A. Yang. 2013. Climatic data and their characterisation for hydrological and agricultural scenario modelling across the Flinders and Gilbert catchmentsA recent examination of the non-riverine and riverine areas of the catchment concluded that the aquatic values of the Flinders catchment were moderate to high, with respect to the extent of naturalness, special features, diversity and richness of flora and fauna, and threatened species (Rollason and Howell, 2010). The most highly scored areas included the Porcupine Gorge and White Mountain National Park region, upper Stawell, Cloncurry and Corrella River region, and parts of the Saxby River (Rollason and Howell, 2010). : CSIRO Water for a Healthy Country and Sustainable Agriculture Flagship, Australia.

Piccolo, R.L., C.J. Brown, D.A. Friess, C.E. Lovelock, B.M. Brown, C. Buelow, and M.I. Saunders. 2024. Feasibility of habitat restoration under multidisciplinary constraints. *Ocean & Coastal Management* 249: 106977.

Robertson, A.I., and N.C. Duke. 1990. Mangrove fish-communities in tropical Queensland, Australia: spatial and temporal patterns in densities, biomass and community structure. *Marine Biology* 104: 369-379.

Robins, J.B., I.A. Halliday, J. Staunton-Smith, D.G. Mayer, and M.J. Sellen. 2005. Freshwater-flow requirements of estuarine in tropical Australia: a review of the state of knowledge and applicaiton of a suggested approach. *Marine & Freshwater Research* 56: 343-360.

Rummell, A.J., H.P. Borland, J.J. Hazell, J.D. Mosman, J.X. Leon, C.J. Henderson, B.L. Gilby, and A.D. Olds. 2023. Connectivity shapes delivery of multiple ecological benefits from restoration. *Biological Conservation* 288: 110358.

Saintilan, N., K.E. Kovalenko, G. Guntenspergen, K. Rogers, J.C. Lynch, D.R. Cahoon, C.E. Lovelock, D.A. Friess, E. Ashe, and K.W. Krauss. 2022. Constraints on the adjustment of tidal marshes to accelerating sea level rise. *Science* 377: 523-527.

Saunders, M.I., T. Cannard, M. Fischer, M. Sheppard, A. Twomey, R. Morris, M.J. Bishop, M. Mayer-Pinto, F. Malcolm, and M. Vozzo. 2024. A roadmap to coastal and marine ecological restoration in Australia. *Environmental Science & Policy* 159: 103808.

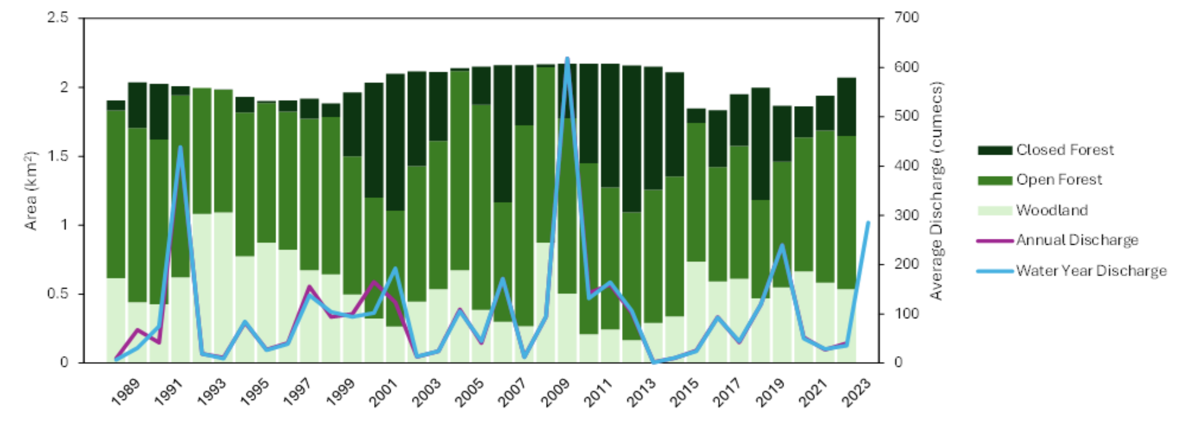
Vance, D.J., M.D. Haywood, D.S. Heales, R.A. Kenyon, N.R. Loneragan, and R.C. Pendreyne. 2002. Distribution of juvenile penaeid prawns in mangrove forests in a tropical Australian estuary, with particular reference to *Penaeus merguiensis*. *Marine Ecology Progress Series* 228: 165-177.

Webley, J., and T. Probst. 2020. Allocation of fisheries management costs to Queensland's commercial fisheries for the 2017-18 and 2018-19 financial years: The process of allocation.

Yin, H., Y. Hu, M. Liu, C. Li, and J. Lv. 2021. Ecological and environmental effects of estuarine wetland loss using keyhole and landsat data in Liao River Delta, China. *Remote Sensing* 13: 311.

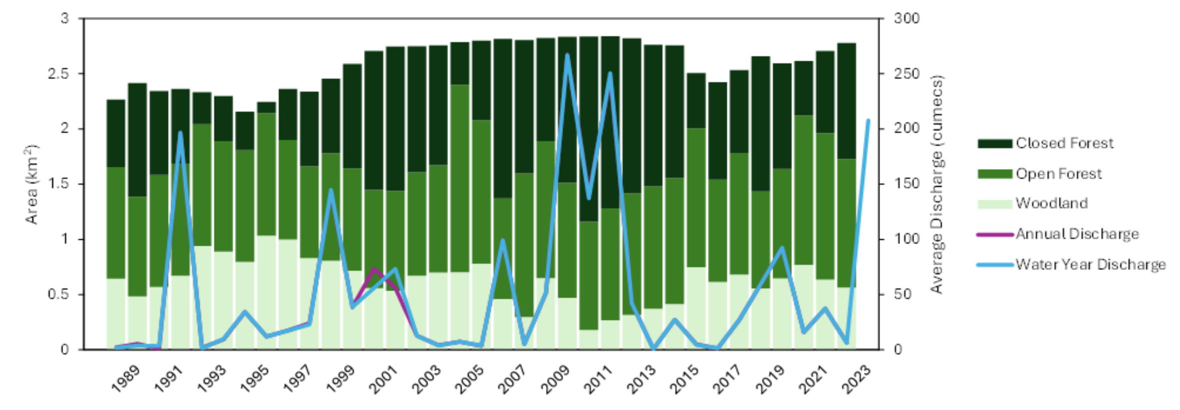
Section 3: Appendix

Flinders River



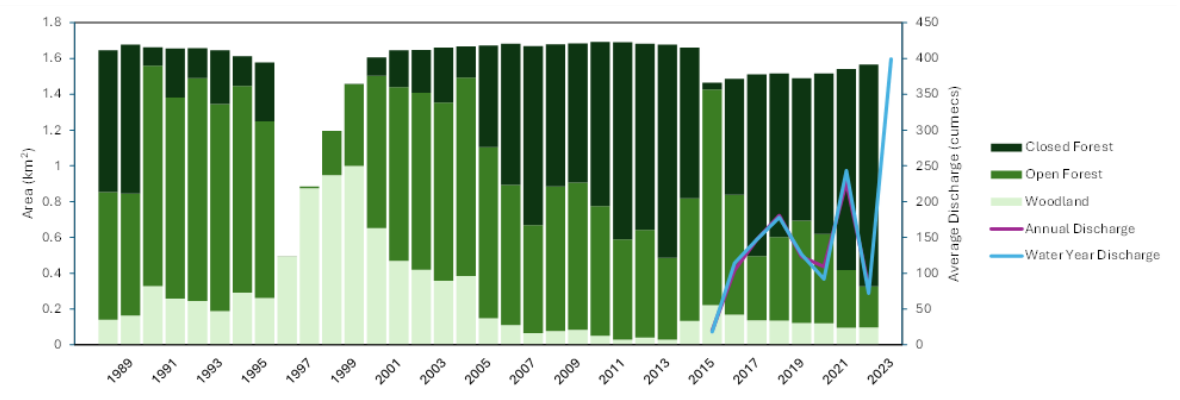
Supplementary 1. Area of mangrove cover between 1988 and 2022 in Flinders River plotted with annual and water year discharge.

Norman River



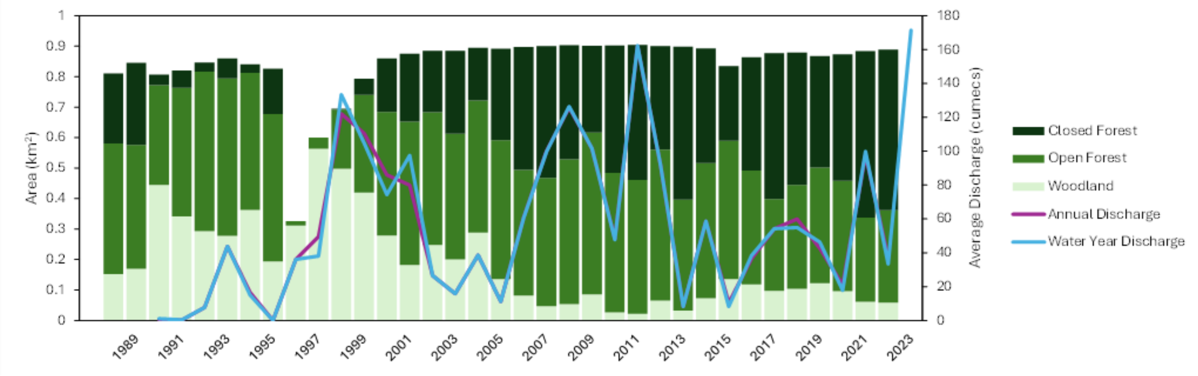
Supplementary 2. Area of mangrove cover between 1988 and 2022 in Norman River plotted with annual and water year discharge.

Gilbert River



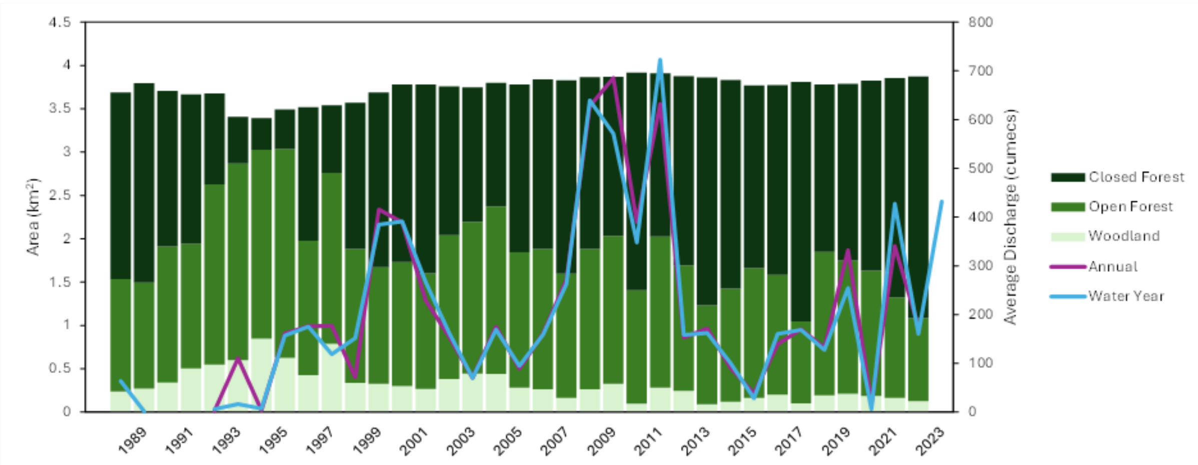
Supplementary 3. Area of mangrove cover between 1988 and 2022 in Gilbert River plotted with annual and water year discharge.

Staaten River



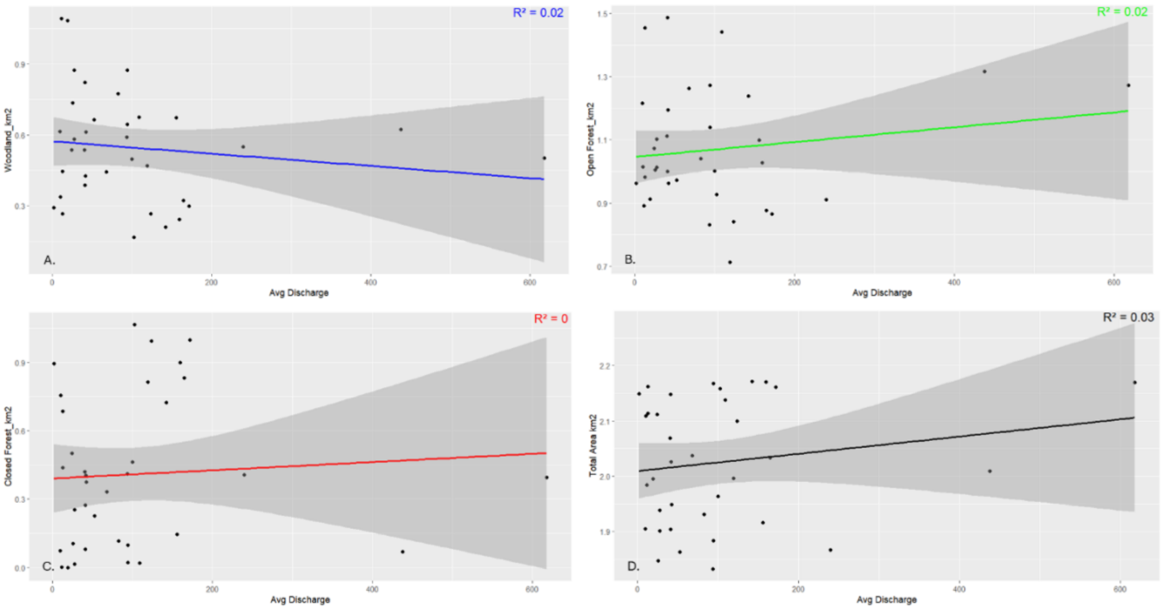
Supplementary 4. Area of mangrove cover between 1988 and 2022 in Staaten River plotted with annual and water year discharge.

Mitchell River



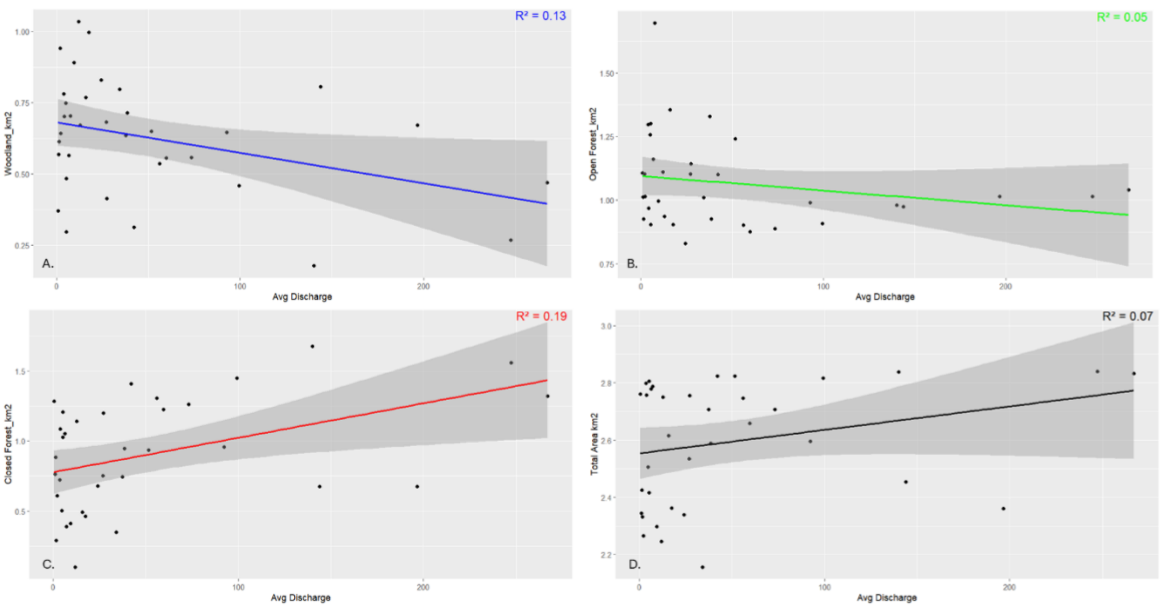
Supplementary 5. Area of mangrove cover between 1988 and 2022 in Mitchell River plotted with annual and water year discharge.

Flinders River



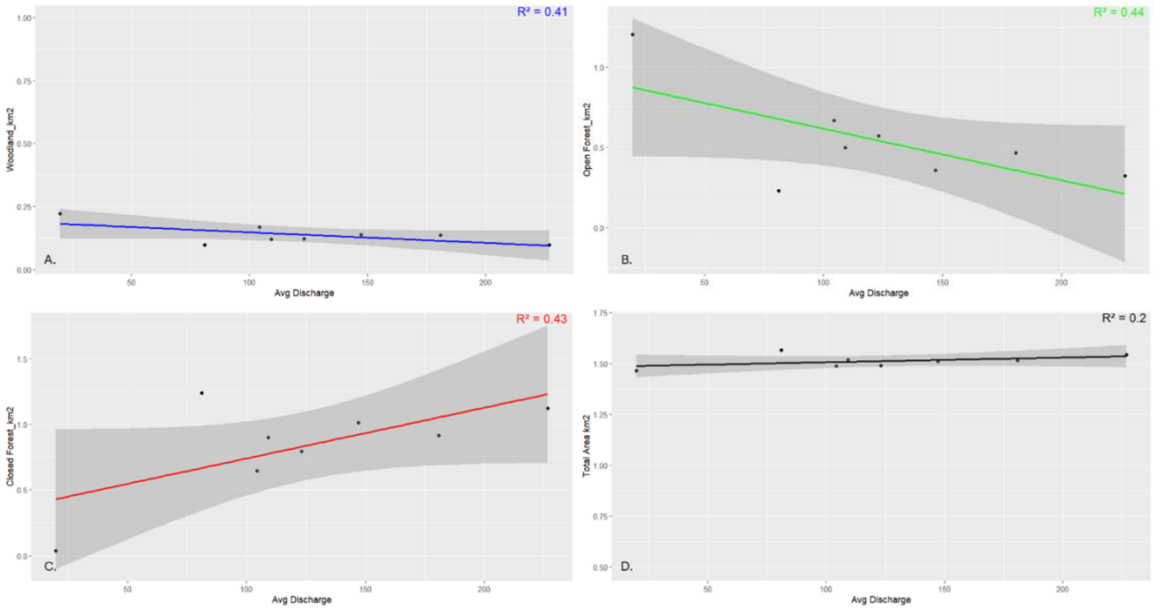
Supplementary 6. Linear model plots for the Finders River. Each canopy coverage and average discharge per year (A-C); Total area extent and average discharge per year (D) with corresponding R-squared values. Darker grey area represents the 95% confidence interval. All p-values > 0.05.

Norman River



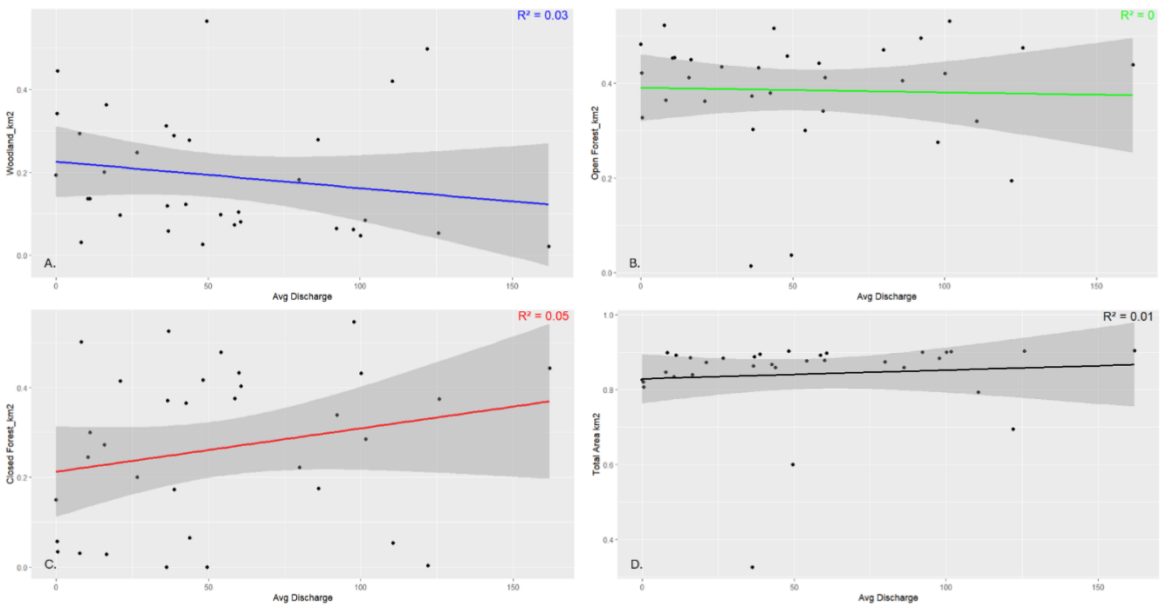
Supplementary 7. Linear model plots for the Norman River. Each canopy coverage and average discharge per year (A-C); Total area extent and average discharge per year (D) with corresponding R-squared values. Darker grey area represents the 95% confidence interval. P-values for B & D > 0.05. P-value for A < 0.05 (p = 0.03126). P-value for C < 0.01 (p = 0.009485).

Gilbert River



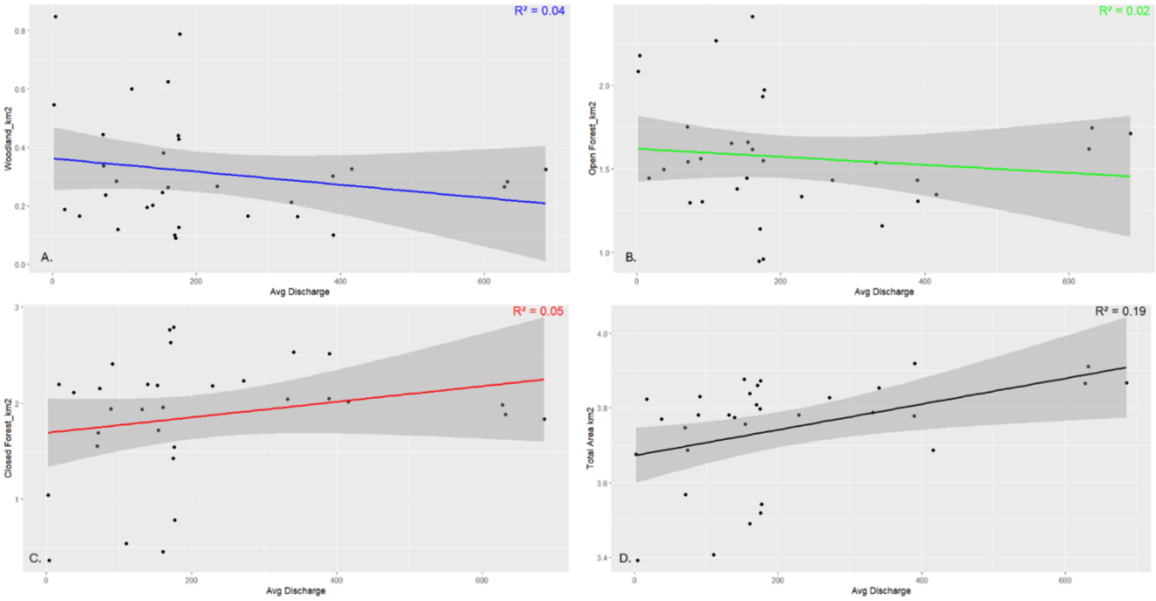
Supplementary 8. Linear model plots for the Gilbert River. Each canopy coverage and average discharge per year (A-C); Total area extent and average discharge per year (D) with corresponding R-squared values. Darker grey area represents the 95% confidence interval. All p-values > 0.05.

Staaten River



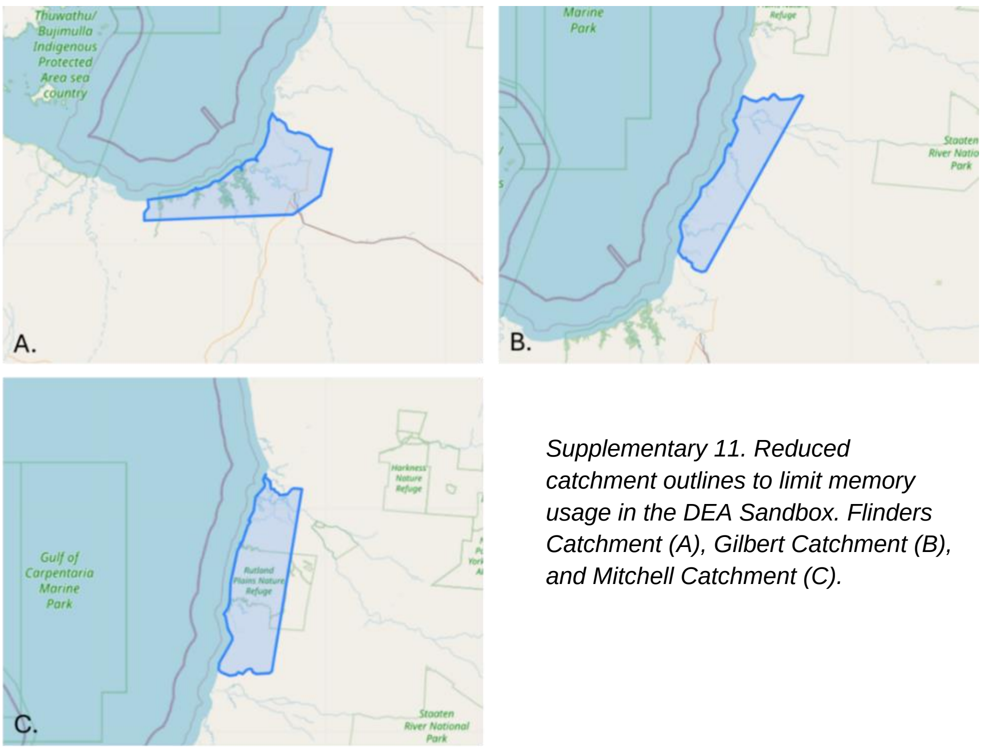
Supplementary 9. Linear model plots for the Staaten River. Each canopy coverage and average discharge per year (A-C); Total area extent and average discharge per year (D) with corresponding R-squared values. Darker grey area represents the 95% confidence interval. All p-values > 0.05.

Mitchell River



Supplementary 10. Linear model plots for the Mitchell River. Each canopy coverage and average discharge per year from (A-C); Total area extent and average discharge per year (D) with corresponding R-squared values. Darker grey area represents the 95% confidence interval. P-values for A-C > 0.05. P-value for D < 0.05 (p = 0.01315).

Reduced Catchments



Supplementary 11. Reduced catchment outlines to limit memory usage in the DEA Sandbox. Flinders Catchment (A), Gilbert Catchment (B), and Mitchell Catchment (C).

4. Testing nutrient status of estuarine mudflats

4.1. Introduction

Intertidal mudflats are an important habitat in estuaries and are often the site with much of the primary productivity, especially in highly turbid estuaries where light limits productivity in the water column. Many estuaries in northern Australia are characterised by highly turbid waters (Burford et al. 2008, 2011, 2012). These turbid waters may be caused by tidal mixing, e.g. macrotidal areas, wind mixing, catchment erosion, and the inherent properties of the suspended sediment. Mudflats in northern Australia provide habitat for many species, including meiofauna and macrofauna, fish and crustacean species, and migratory shorebirds and seabirds (e.g. Duggan et al., 2014; Lowe et al., 2022; Venarsky et al. 2022; Burford et al. 2020, 2021).

Previous studies in the Mitchell, Flinders, Norman and Gilbert River estuaries in the Gulf of Carpentaria, in both intertidal mudflats and in the water column, have demonstrated that the aquatic ecosystems are chronically nutrient-limited (Burford et al. 2012, Burford and Faggotter 2021). Studies have shown that the deeper waters of the Gulf are also highly nutrient-limited, and as such, freshwater flows are the principal mechanism for nutrient inputs (except for via nitrogen fixation, Burford 2009).

This means that freshwater flows, and their associated nutrients, are fundamental to ensuring estuaries remain productive. This is significant because it means that any reduction in freshwater flows, and associated nutrients, during the wet season has the potential to impact primary production. Therefore, water development may have significant impacts on estuarine primary productivity, with flow-on effects on higher trophic levels, depending on the volumes of water removed.

Therefore, this study had the following aims:

- Test the effect of reducing water column nutrients on primary production of mudflat algal (= microphytobenthos) to test how nutrient reductions, with reduced freshwater flow may ultimately impact estuaries. This was tested on the Norman, Flinders, Daly and Keep River systems.
- Test the effect of increasing nutrient inputs on the primary productivity of mudflats in the Daly and Adelaide Rivers, NT, and validate previous findings for the Norman and Flinders Rivers. This work is designed to demonstrate the critical nature of freshwater nutrient inputs.

4.2. Methods and Results

The methods used for the primary productivity experiments followed that of Burford and Faggotter (2021). In essence, mudflat samples were collected in the intertidal zone (to a set depth of approximately 3 cm) in the Norman, Flinders, Daly and Adelaide Rivers. These samples were transported back to Brisbane in plastic containers and cores were inserted into the containers to get subsamples of the sediment. The core bottoms were then bunged, and filled with seawater. Between 5 and 6 replicate cores were collected from between 2 to 4 sites within each estuary (Table 4.1). Samples were then left to equilibrate for 24 h before experiments were conducted and experiments were done at 30°C (with a temperature controller) in full sunlight.

There were three treatments: 1) A control with intertidal mud and overlying water from the sampling sites, 2) treatment with nutrients (ammonium, phosphate) added to the overlying water (final nitrogen and phosphorus concentrations 0.924 mg N L-1, and 0.127 µg P L-1 respectively), and 3) treatment with low nutrient seawater (ocean water) as the overlying water (Table 4.1). For treatment 2, more nutrients were added every 2-3 days, whilst for treatment 3, low nutrient seawater replaced the overlying water every 2-3 days.

In order to measure oxygen flux rates (as a measure of primary productivity), cores were periodically filled with seawater and capped to be fully sealed, with no air bubbles. Oxygen levels were measured while cores were incubated in the sun. A Presens® fiber-optic oxygen sensor (FIBOX) was used to measure oxygen-sensitive optode patches glued to the inside wall of each core (Presens; Duggan et al. 2014). This allowed measurements of oxygen concentrations in each core over time without needing to open up the cores. Multiple readings were done during the morning when oxygen production was at its maximum. Dark cores were also incubated but respiration rates were typically at or below detection limits.

After each day of reading, caps were removed and the samples left in ambient conditions until the next day of reading.

Table 4-1. Experimental design used for oxygen flux experiments using mudflat cores from multiple estuaries.

Estuary	# Sites	# replicate cores	Treatments
Norman (QLD)	2 = high nutrients 3 = low nutrients	5	Control Nutrient addition (high) Low nutrient seawater periodic exchange
Daly (NT)	3	5	Control Nutrient addition (high) Low nutrient seawater periodic exchange
Flinders (QLD)	4	6	Control Nutrient addition (high & low) Low nutrient seawater periodic exchange

Adelaide (NT)	4	6	Control Nutrient addition (high & low) Low nutrient seawater periodic exchange

The first estuary tested was the Norman River estuary. This involved collection at two intertidal mudflat sites in the estuary. The incubations were conducted over four days. There were statistical differences in the oxygen flux rates (as a measure of primary productivity) at both sites between the control (adjacent water on top of cores) and the treatment with nutrients added periodically during the study (Figure 4.1). However, there were no statistical differences in the oxygen flux rates between the control and the treatment with regular addition of low-nutrient seawater.

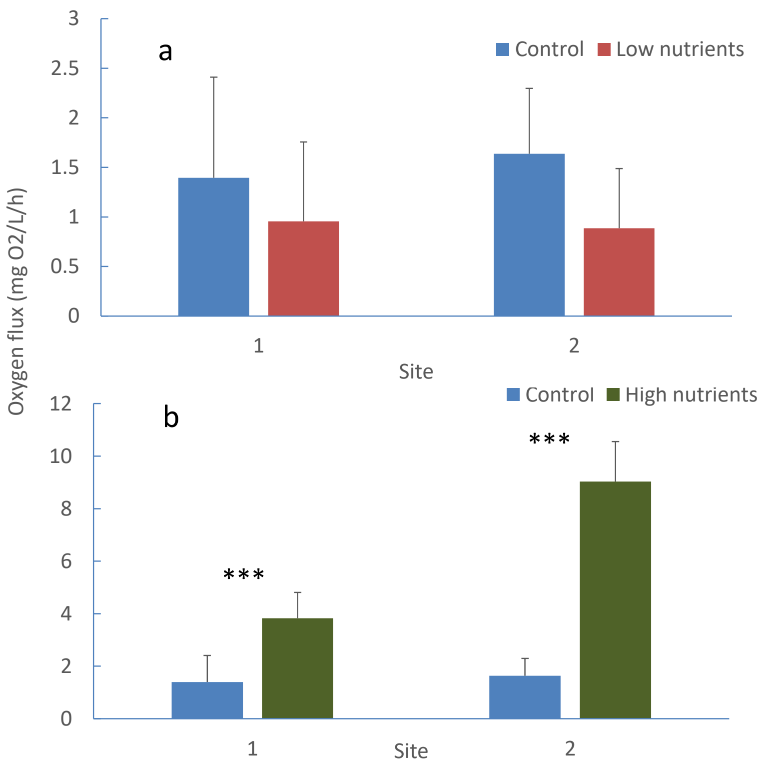


*Figure 4-1: Mean (SD) oxygen flux (mg O2/L/h) in core chamber experiments for mudflat sites in the Norman River estuary in June 2023. a) comparison of control cores with cores that were regularly flushed with low nutrient seawater on day 4. b) Comparison of control cores with cores where nutrients were added every 3 days as measured on day 4. *P<0.05, **P<0.01, **P<0.005.*

The second estuary tested was the Daly River estuary. For this estuary, three sites were tested with nutrient additions, and two sites were tested to determine whether low-nutrient seawater would reduce the oxygen flux rates. For the nutrient additions, flux rates were higher at two of the three sites compared with the control (Figure 4.2). Removal of nutrients with low-nutrient water reduced the flux rates at one of the two sites.

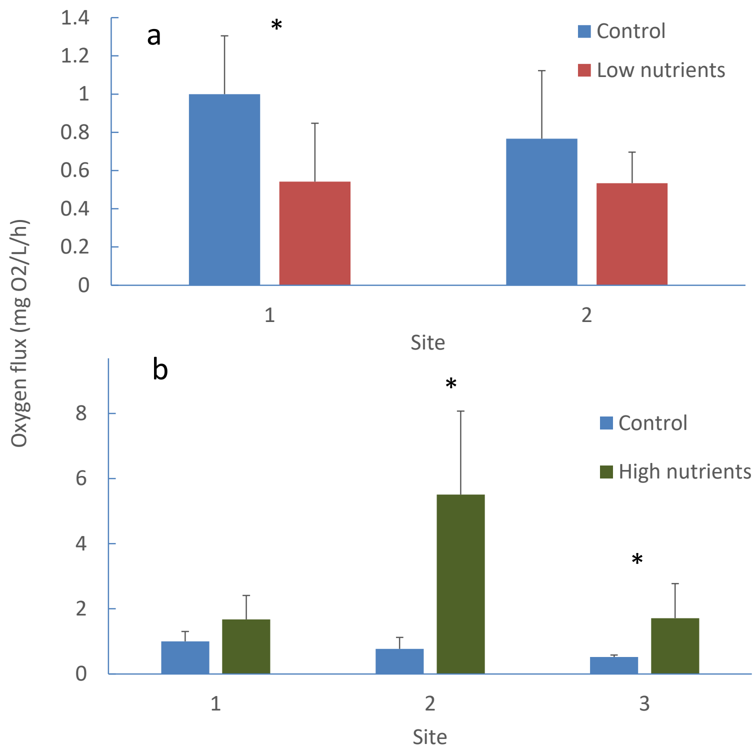


*Figure 4-2: Mean (SD) oxygen flux (mg O2/L/h) in core chamber experiments for mudflat sites in the Daly River estuary in Sept 2023. a) comparison of control cores with cores that were regularly flushed with low nutrient seawater on day 7, b) comparison of control cores with cores where nutrients were added every 3 days as measured on day 7. *P<0.05, **P<0.01, **P<0.005.*

While there were indications of a reduction of oxygen flux in low-nutrient water, and an increase with nutrient addition in both the Norman and Daly Rivers, this was not always statistically significant. Therefore, for the two remaining estuaries, i.e. Flinders and Adelaide River estuaries, six replicates were used, rather than five, the number of sites was increased to four, and for the nutrient addition, experiments were run for at least one week (Table 4.1).

For the Flinders River estuary mudflats, three of the four sites had a decrease in oxygen flux on day three when low-nutrient water periodically replaced the site water (Figure 4.3). In addition, all four sites had an increase in flux rates on day eight when nutrients were periodically added.

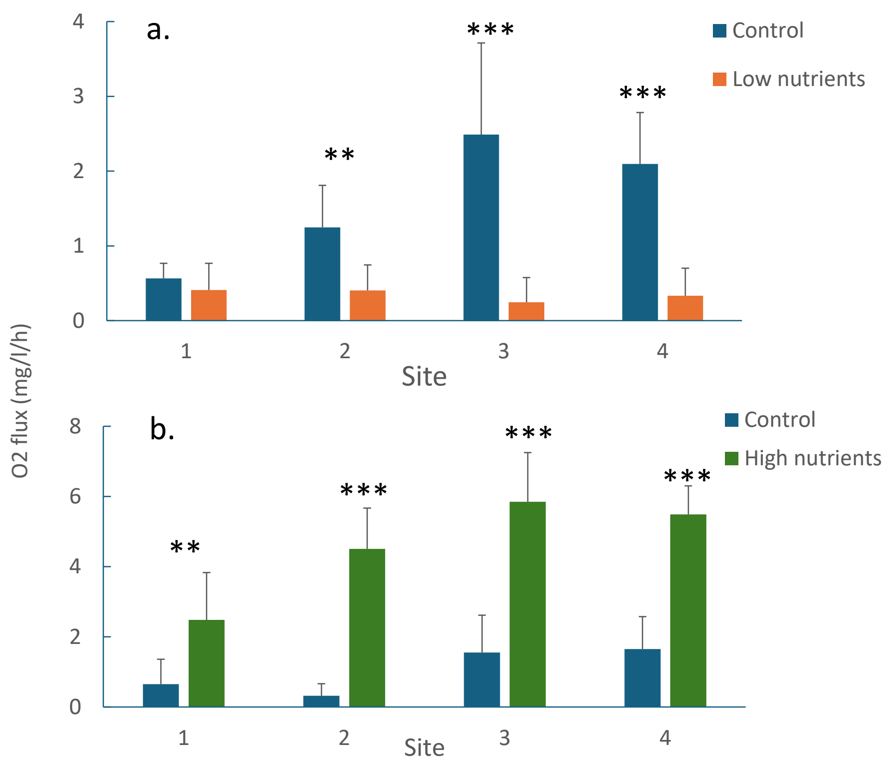
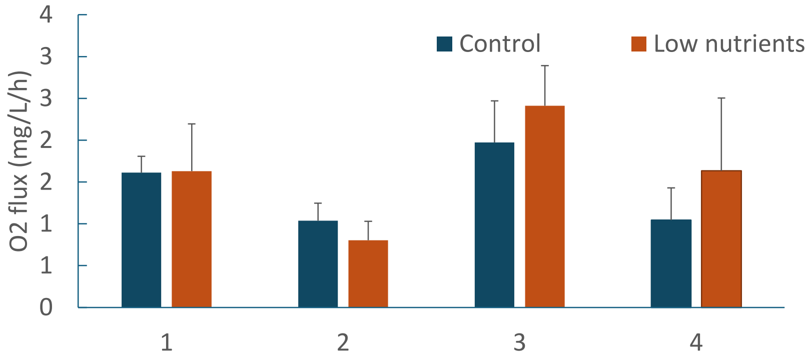


*Figure 4-3: Mean (SD) oxygen flux (mg O2/L/h) in core chamber experiments for mudflat sites in the Flinders River estuary in August 2024. a) comparison of control cores with cores that were regularly flushed with low nutrient seawater on day 3, b) comparison of control cores with cores where nutrients were added every 3 days as measured on day 8. *P<0.05, **P<0.01, **P<0.005.*

For the Adelaide River estuary mudflats, only one of the four sites had a decrease in oxygen flux by day seven when low nutrient water periodically replaced the site water (Figure 4.4). All four sites had an increase in flux rates by day seven when nutrients were added periodically throughout the experiment.



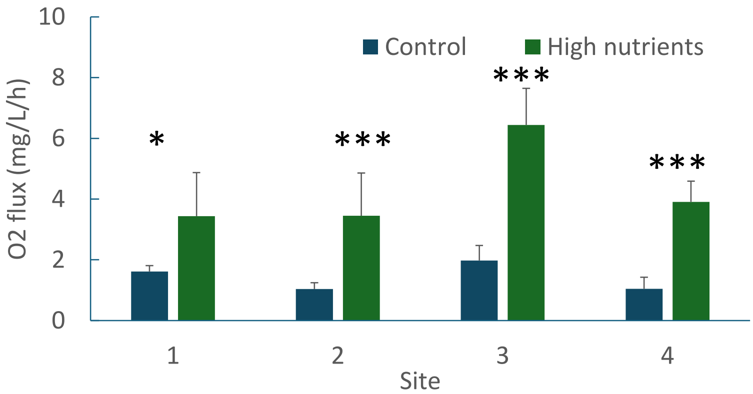


*Figure 4-4: Mean (SD) oxygen flux (mg O2/L/h) in core chamber experiments for mudflat sites in the Adelaide River estuary in August 2024. a) comparison of control cores with cores that were regularly flushed with low nutrient seawater on day 7, b) comparison of control cores with cores where nutrients were added every 2 to 3 days as measured on day 7. *P<0.05, **P<0.005.*

4.3. Discussion and Conclusions

In our experiments, Norman, Daly, Flinders and Adelaide River mudflats all had similar rates of oxygen flux, and a statistical increase in rates with the addition of nutrients. This broadens and substantiates a previous study on the Mitchell, Gilbert and Flinders mudflats that showed that nutrient addition stimulated primary production (Burford and Faggotter 2021). Although the addition of nutrients caused a very rapid increase in primary production, i.e. in a couple of days, at times it took a few more days for a statistically significant increase in primary production to occur. This reflects the heterogeneous nature of mudflats.

The use of low-nutrient seawater to reduce oxygen flux rates resulted in a statistical decrease in flux rates only at some sites in each estuary over the timeframe of each incubation (up to 9 days). This method is designed to demonstrate that addition of water with low nutrients will rapidly decrease primary productivity rates. Although this method shows some promise, it is likely that incubations will need to be run for longer in order to see statistical differences across all sites.

The implications of this study are that all estuaries in this study were nutrient depauperate, and therefore a reduction in nutrient loads from increased freshwater extraction will ultimately decrease primary production on mudflats.

4.4. Acknowledgements

We wish to thank the skipper and crew of the Kerry D in Karumba, Queensland, Peter Kyne, Erica Garcia and Julia Constance at Charles Darwin University for help and support in field work to the Daly and Adelaide River estuaries.

4.5. References

Burford, M.A., Faggotter, S.J. 2021. Comparing the importance of freshwater flows driving primary production in three tropical estuaries. *Marine Pollution Bulletin,* 169, 112565

Burford, M.A., Alongi, D.M., McKinnon, A.D., Trott, L.J. 2008. Primary production and nutrients in a tropical macrotidal estuary, Darwin Harbour, Australia. *Estuarine, Coastal and Shelf Science*, 79, 440-448.

Burford, M.A., Rothlisberg, P.C., Revill, A. T. 2009. Sources of nutrients driving production in the Gulf of Carpentaria, Australia – a shallow tropical shelf system. *Marine and Freshwater Research* 60, 1044-1053.

Burford, M. A., Revill, A.T. Palmer, D.W. Clementson, L., Robson, B.J., Webster, I.T. 2011. River regulation alters drivers of primary productivity along a tropical river-estuary system. *Marine and Freshwater Research,* 62, 141-151.

Burford, M.A., Webster, I.T., Revill, A.T., Kenyon, R.A., Whittle, M., Curwen, G. 2012. Controls on phytoplankton productivity in a wet-dry tropical estuary. *Estuarine, Coastal and Shelf Science,* 113, 141-151.

Burford, M.A., Faggotter, S.J., Kenyon R. (2020). *Project 1.4: Contribution of rivers to the productivity of floodplains and coastal areas of the southern Gulf of Carpentaria – Component 1*. NESP Northern Australia Environmental Resources Hub final report.

Burford, M.A., Faggotter, S.J., Lowe, V., Venarsky, M., Frid, C., Ronan, M., Bush, R., Edwards, 2021. D. *Project 3.6: The vulnerability of food supplies for migratory shorebirds to altered flow in the southern Gulf of Carpentaria.* NESP Northern Australia Environmental Resources Hub final report. Griffith University, Brisbane.

Duggan, M., Connolly, R.M., Whittle, M., Curwen, G., Burford, M.A. 2014. Effects of freshwater flow extremes on intertidal biota of a wet-dry tropical estuary. *Marine Ecology Progress Series,* 502, 11-23.

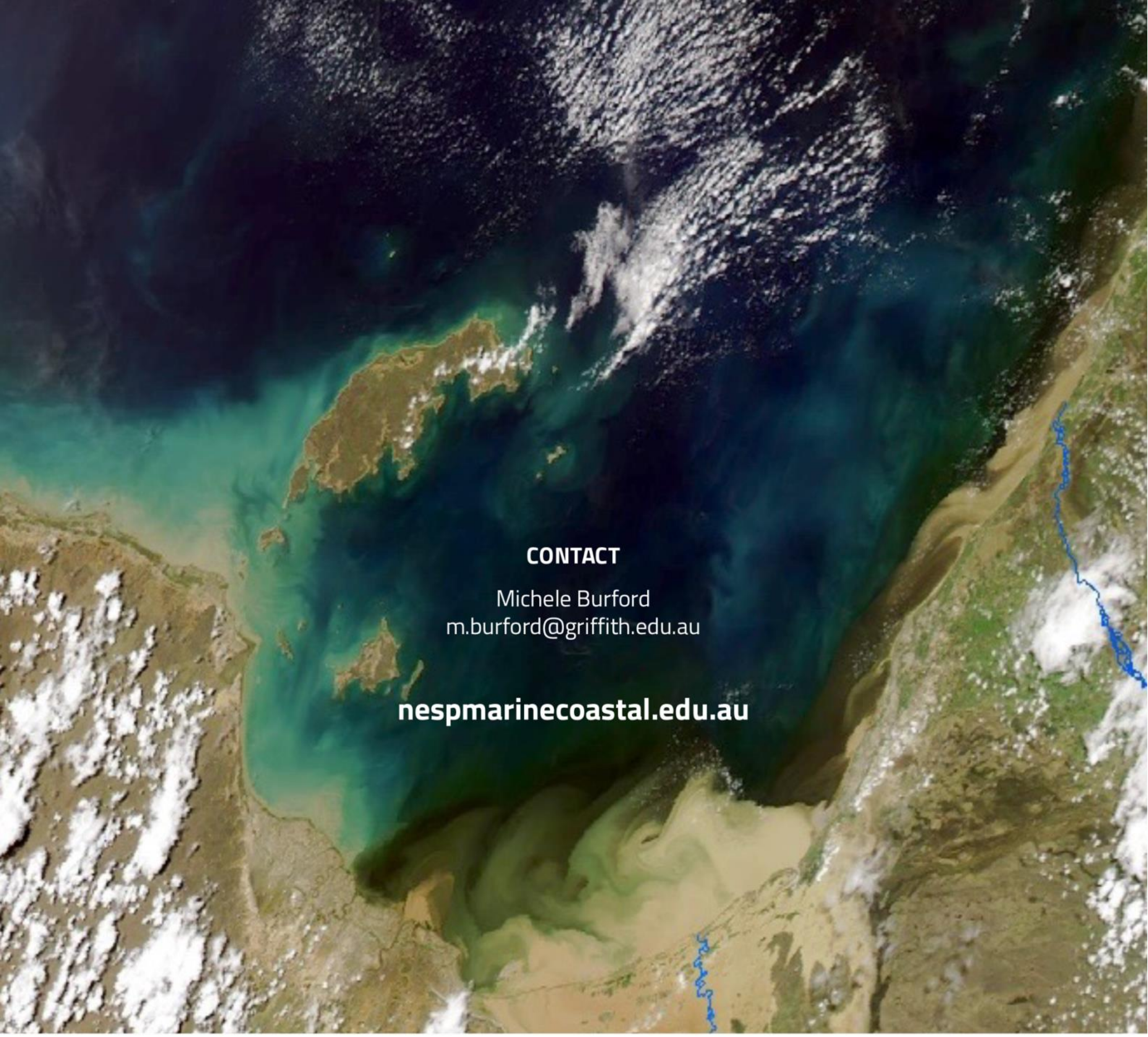
Lowe, V., Frid, C.L.J., Venarsky, M., Burford, M.A. 2022. Responses of a macrobenthic community to seasonal freshwater flow in a wet-dry tropical estuary. *Estuarine, Coastal and Shelf Science,* 265, 107736

Venarsky, M.P., Lowe, V., Frid, C.L.J., Burford, M.A. 2022. Flow regime influences benthic biota biodiversity, but not abundance or biomass, in intertidal mudflats and sandflats in wet-dry tropical estuaries. *Estuarine, Coastal and Shelf Science,* 271, 107858.

5. Stakeholder engagement

There have been a range of activities focussed on stakeholder engagement over the lifetime of the project:

- **May 2023**
  Presentation of project aims with the steering committee (Teams meeting) (Invitees: A. Dale (JCU), P. Waugh (NT DEPWS, G. Penton (SGNRM), R. Zuks (WA DPIRD), J. Coysh (Qld DRDMW), A. Curro (CRCNA), R. Dann (DCCEEW), A. Jarrett (NPF), J. Marshall (Qld DES), Z. Williams (Savannah NRM), RDA)
- **May 2023**
  Presentation at Carpentaria Land Council Aboriginal Corporation (CLCAC) marine protected areas workshop (Cairns)
- **March 2024**
  Presentation at CLCAC to provide updated information on the project
- **March 2024**
  Attended Keep River catchment management scoping study meeting (Teams)
- **August 2024**
  Presentation at CLCAC on project findings (Burketown)
- **September 2024**
  Presentation to CRCNA Advisory group for the Gilbert Catchment (teams)
- **September 2024**
  Final report sent to steering committee
- **Throughout the project**
  Attempts to engage with Northern Land Council and Kimberley Land Council on multiple occasions were unsuccessful.



CONTACT

Michele Burford
m.burford@griffith.edu.au

nespmarinecoastal.edu.au

Marine and Coastal
National Environmental Science Program

GRIFFITH UNIVERSITY
QUEENSLAND AUSTRALIA

National Environmental Science Program

This project is supported with funding from the Australian Government under the National Environmental Science Program.