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Final Report

Exploring the potential of watercourse repair on an agricultural floodplain

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Cover photographs: (front) Turbid irrigation channel on the Burdekin floodplain. Image: Nathan Waltham. (back) Palustrine wetland. Image: Nathan Waltham.

This report is available for download from the NESP Tropical Water Quality Hub website:
http://www.nesptropical.edu.au

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ACRONYMS

BR^P .... Phosphorus Biodeposition Rates
DO .... Dissolved Oxygen
GBR .... Great Barrier Reef
GBRWHA...... Great Barrier Reef World Heritage Area
nMDS............ Nonmetric Multidimensional Scaling
NESP .... National Environmental Science Program
NQDT............ North Queensland Dry Tropics
NRM.............. Natural Resource Management
TropWATER..Centre for Tropical Water and Aquatic Ecosystem Research
TDN............... Total Dissolved Nitrogen
TDP............... Total Dissolved Phosphorus
TN .... Total Nitrogen
TWQ.............. Tropical Water Quality
UAV .... Unmanned Aerial Vehicle

ABBREVIATIONS

cm ....centimetre
ha ....hectare
m ....metre
mm ....millimetre
mS/cm ....milliseimens per centimetre

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EXECUTIVE SUMMARY

The Burdekin floodplain includes Australia's largest sugarcane production district. Concomitant with sugarcane development, the floodplain and waterways are in poor ecological condition, with altered hydrology, poor water quality, and extensive invasive weeds and noxoius fish. Considerable effort continues to be made on reversing the decline in water quality reaching the Great Barrier Reef (GBR), particularly the Bowling Green Bay Ramsar wetland located along the coastal strip. The floodplain now comprises a highly modified distribution network used to move Burdekin dam water, which is turbid, across the floodplain to provide shallow groundwater aquifer recharge and surface water resources for two distinct sugarcane irrigation schemes. The floodplain has been the subject of research and restoration for many years, however, it remains in poor condition. This case study took the opportunity to partner with major restoration activities under the direction of local restoration practitioners charged with managing overall threats on the floodplain, in an effort to restore the services and values possible. This case study supports and builds on limnological knowledge on the floodplain (see Davis et al., 2014 review). The key findings in this case study were:

- Overall, for as long as sugar production occurs on the floodplain and delivery of excessive available nutrients reaches local creeks, aquatic invasive plants will require management intervention. The issue is more critical for the clear water lagoons, located at the end of the distribution network or off-channel section, where submerged aquatic vegetation biomass is high because of a deeper euphotic zone. High plant biomass (particularly species such as water hyacinth, which forms densely packed mats, and submerged aquatic vegetation such as Ceratophyllum sp.) places a high net demand on available oxygen in the water column. This high oxygen demand results in extreme diel cycling compared to turbid lagoons which generally have lower daily amplitudes and higher minimum values. Maintaining most creeks on the floodplain in a moderately turbid state is probably important in reducing the threat of eutrophic-driven hypoxia and fish kills (without floodplain-scale water treatment of sugar farm runoff), in addition reduces directly the need to apply chemicals to treat aquatic plants (both emergent and floating). This management approach is counterintuitive to the broader GBR initiatives set under the Reef 2050 plan which have centered on reducing land derived suspended sediment. In managing the threat of aquatic plant overgrowth and poor water quality conditions, the strategy thereby is to keep the floodplain creek network in a turbid state. In doing so, this will give the best chance of better dissolved oxygen (DO) cycling, thereby reducing conditions potentially leading to fish kills.

- Drying down palustrine areas on the floodplain through better irrigation water use efficiency has been a critical management need for many years. Installation of recycle pits (similar to the one on Saltwater Creek) situated in the lower ends of the farms to capture irrigation water runoff before reaching shallow coastal wetlands, coupled with improved irrigation water delivery efficiency seems a plausible endeavour at reinstating the natural wetting and drying cycles of coastal wetlands. In doing so the objective is to self-regulate invasive floristic species and improve fish connectivity. While the data here were limited and not able to test this model owing to the short timeframes of this project, it requires further investigation and tracking. Constructing more recycle pits would also increase the number of habitat opportunities for freshwater bivalve, which were also shown here to provide bioengineering services.

- Weed removal using mechanical harvesting approaches while resulting in immediate removal of large amounts of organic material, also will almost immediately improve available oxygen and connectivity for fish to access the floodplain. Although this was considered a successful restoration case, without ongoing maintenance, the longer term return for the investment will be lost. Treating the threat of weed infestation and not focusing on the ecosystem services and converting these services to values by engagement of local beneficiaries (i.e., an ecosystem with appropriate levels of nutrients, vegetation, DO etc, to support healthy flora and fauna levels) was a major learning in this demonstration case study. Indeed, this has been the case in Crooked Waterhole where local land managers have not kept up with the weed maintenance and the creek has slowly reverted back into an overgrown weed reach again.

- Reinstating fish passage on the floodplain is challenging owing to the number of physical barriers (engineering structures), biological barriers (weeds chokes), and chemical barriers (dissolved oxygen). Prioritised efforts to remove and reduce these barriers to fish movement, has been important, with some progress. However, considerable funds to redesign engineered structures are necessary to reinstate fish passage across the floodplain, in addition to ongoing management of the biological and chemical barriers that occur across the floodplain.

- An environmental levy (approximately $20,000/yr AUD, funded by farmers, Burdekin Shire Council, a local Natural Resource Management group, and Lower Burdekin Water Board) in Sheep Station Creek was enacted in 2003 to fund an ongoing aquatic weed removal program. It involves the use of an aquatic weed harvester that removes floating vegetation for composting in conjunction with boat weed spraying). This funding and maintenance program is critical to the delivery of irrigation water across the network, but is also promoting productive fish habitat. A resurvey of Sheep Station Creek nearly 20 years after commencing the levy and maintnenace revealled the creek continues providing habitat for many of the same species recorded previously, with the exception of several invasive fish species. Dissolved oxygen concentrations in the creek were geneally better than other places on the floodplain, a consequence of fewer aquatic weeds. This funding model and maintenance program serves as a model for other floodplain systems throughout the GBR.

Managing the Burdekin floodplain for multiple services will continue to be challenging and will require a whole-of-system approach (see Queensland WetlandInfo for details). The support, involvement and commitment from all stakeholders is key to the success of any management program. There are a number of key learnings from this project that combined with knowledge and converting services into values by engagement of key beneficiaries, need to be considered and used by stakeholders. The management required on the floodplain to protect and maximising the ecosystem services outcomes, offers an important lesson for agricultural development on other floodplains in northern Australia. There are no ‘off the shelf’ guides. Local data and knowledge of each floodplain is necessary. The detail and scale of data necessary to effectively manage agricultural development on floodplains should not be underestimated, and the Burdekin is a case in point. With expansion of new agricultural developments and changes in floodplains to more hybrid, novel, ecosystems will reduce ecosystem services. This case study advocates the need to manage wetland restoration activities for the values and services, and to not focus on the components and processes.

1.0 INTRODUCTION

1.1 Whole-of-system approach to floodplain wetland restoration

Floodplains contain transitional watercourses that connect, often intermittently, estuaries with freshwater riverine and palustrine ecosystems (Mitsch and Gosselink 2016). By providing connectivity between habitats, and diverse transitional habitats in themselves, floodplain water features are a key feature in supporting fisheries and biodiversity (Nagelkerken et al. 2015). A major challenge facing coastal managers is protecting coastal floodplains for conservation values and services, while recognising further land use change (Arantes et al. 2019). Many fish species occupying this biologically rich ecosystem have migration lifecycle requirements that are precisely cued to the rising waters and floodplain connection, thereby successfully accessing seasonal habitats for feeding and breeding (Godfrey et al. 2016; Hurd et al. 2016; Abrial et al. 2019; Cruz et al. 2020). Duration and frequency of hydrological connectivity is a major context-specific driver of floodplain functionality (Whitfield et al. 2012; Elliott et al. 2016; Waltham et al. 2019). Barramundi (*Lates calcarifer*), for example, migrate downstream to spawn around river mouths and bays, high spring tides wash eggs and larvae into coastal swamps (Russell and Garrett 1988). Once juveniles are approximately one year old (Power et al. 2019), a proportion migrate upstream during the wet season to occupy deeper wetlands, such as large rivers, lakes and lagoons (Sawynok 1998; Robins et al. 2006; Crook et al. 2020). Any human interference or disturbance that modifies or prevents this function, without on-going (expensive) management and maintenance action, could have lasting consequences on lost opportunities to support fish movement to upstream habitats that may be returned to the fishery and estuarine population on subsequent flood cycles (Anjos et al. 2008).

Despite their value as productive habitat, the contribution that floodplains provide to fishery production remains an often overlooked habitat feature in the seascape mosaic, meaning that the contribution to broader production estimates is probably under-estimated and incomplete (Galib et al. 2018; Peel et al. 2019). Given their location in the coastal network, floodplains are often extensively modified as part of urban sprawl, industry expansion, agricultural development, or even transport corridors, resulting in habitat fragmentation or loss of these coastal ecosystems. Estimates suggest we are losing in the order of 95 km^2 of global floodplain wetlands each year (Mitsch and Gosselink 2016), and this is not likely to slow with continuing urban and industrial expansion in many coastal locations (Davidson 2014). In response, managers are moving towards implementing large scale restoration programs (Barbier 2013; Waltham et al. 2020c), though data presenting cases where these efforts return strong environmental dividends for the investment are rare (Sheaves et al. 2020). The need for more examples of the values and services arising from restored floodplain ecosystems is apparent (Zedler 2016; Oosterlee et al. 2018; Weinstein et al. 2019).

Focusing on fish assemblage structure and diversity provides an important step towards recognising the need for conservation of wetlands on floodplains for these important services and values (i.e., fish production), however, examining the habitat condition that fish occupy is equally important. For example, dissolved oxygen (DO%) and temperature are most studied determinants given both are directly linked to acute and chronic effects on fish growth, reproduction and overall fitness (Heath 1995). This is particularly pertinent for floodplains, which are increasingly hindered by excessive aquatic plant growth as a consequence of excessive nutrients and increased hydrology residents time (Villamagna and Murphy 2010).

On floodplains, low DO events are linked periods to the development of thick beds of submerged aquatic vegetation cover (Kaenel et al. 2000; Miranda and Hodges 2000). The densely packed canopy of floating leaves lower light by shading, and reduces water temperature (Greenfield et al. 2007), but will also prohibit gas exchange, suppressing DO further and increasing asphyxiation risks to local wetland fauna (Villamagna and Murphy 2010).

Wetlands provide many services which are valued by humans, but not all wetlands provide the same values or services, e.g., one wetland may be primarily valued for its natural features while other wetlands might be considered more important for their productivity or tourism values. There are a range of factors that influence the values and services a wetland provides including its location, size, type and condition. Most wetlands have multiple values and managing wetlands effectively involves balancing these values to achieve the best outcomes economically, socially and environmentally (WetlandsInfo, accessed September 2020).

In order to consider wetland values and services a common framework and terminology is required. The values and services provided by a wetland are dependent on the wetland’s structural assembly and processes. It is important to understand the components and processes at the scale (context) at which they function ecologically (Bradley et al. 2020), this may be broader than the wetland itself and incorporate the larger landscape.

1.2 Connectivity on a developed floodplain

Australia faces a legacy of degraded coastal wetland habitats from more than 200 years of urban/industrial development and agricultural intensification (Creighton et al. 2016). Of particular interest is the Great Barrier Reef (GBR) lagoon, a World Heritage Area (GBRWHA) and National Marine Park, protected under an assortment of international agreements, and national and state legislation and policies (Box 1). On-going poor water quality resulting from catchment agricultural runoff and intensification (Brodie and Waterhouse 2012; Brodie et al. 2016; Waterhouse et al. 2016), and the loss of coastal habitats concomitant with coastal development (Waltham and Sheaves 2015), has placed major pressure on reef resilience (DEHP 2016). Conservation and repair of the Great Barrier Reef coastal wetland ecosystems’ have come into focus following media converging on the point that the reef health and resilience has been compromised, particularly around major agricultural centres (DEHP 2016; Waltham et al. 2019), with ecosystem protection and restoration a key performance measure in the development of long term strategic planning policies (GBRMPA 2015).

Box 1. GBR wetlands management framework.

Legislation governing Australia's Great Barrier Reef (GBR), a World Heritage Area and National Marine Park, reflect a desire for coastal wetland restoration. In response to the loss of coastal wetlands that are reducing the GBR's resilience to future disturbance, the overarching framework for managing the GBR, the *Reef 2050 Long-term Sustainability Plan* (Reef 2050 Plan hereafter), and its subordinate *Wetlands in the Great Barrier Reef Catchments Management Strategy 2016-2021* (Reef wetlands strategy) and *The Reef 2050 Water Quality Improvement Plan 2017-2022* (Reef WQ Plan), all recognise the need to maintain and improve the net extent and condition of wetlands, within a whole-of-system catchment management approach. In addition to the Reef 2050 Plan framework, there are numerous laws regulating the development or disturbance of coastal habitats, including the Australian Government's *Environment Protection and Biodiversity Conservation Act 1999* and the Queensland Government's *Environmental Protection Act 1994*, *Fisheries Act 1994*, *Marine Parks Act 2004*, *Planning Act 2016*, *Vegetation Management Act 1999* and *Water Act 2000*. While the *Queensland Environmental Offsets Act 2014* provides a framework to facilitate environmental offsets that may lead to restoration of degraded habitats or creation of wetlands with demonstrable benefits.

Whole-of-system catchment management is an integrated approach to catchment management and the protection, maintenance and restoration of wetland systems (catchment strategy) which is outlined in the Reef wetlands strategy (DEHP 2016). The approach integrates information on natural and human values of a wetland with catchment functioning. Most wetlands have multiple values and managing wetlands effectively involves balancing these values to achieve the best outcomes economically, socially and environmentally. Restoring wetland services first requires defining the values desired, for example, bird habitat, fish habitat, water quality, carbon abatement, hydrology, or cultural. These values will differ between stakeholders and locations. Ecosystem repair strategies seem to be most effective when values of all stakeholders are incorporated, a process best facilitated through discussions to set objectives early in the project lifecycle. Often there will be trade-offs between values, for example, wetlands designed for high nitrogen attenuation are often poor for carbon abatement. The nature of the catchment can heavily influence the ability of a wetland to support given values, integrating desired values with catchment functioning is, therefore, critical to successful restoration. Once the values are set and agreed to by relevant stakeholders, wetland restoration projects can then be prioritised and implemented in a way that supports the identified values and catchment constraints, maximising the likelihood of a successful restoration that is generally accepted (WetlandsInfo, accessed September 2020).

The Burdekin catchment is Australia's largest sugar cane production region (Davis et al. 2014). Concomitant with on-going expansion of sugar cane since the 1980's, the Queensland Government regional ecosystem mapping estimates that 23% of pre-European freshwater wetland extent remain, with most in a fragmented state (GBRMPA 2015). Those remaining have altered hydrology, poor water quality (O'Brien et al. 2016), extensive invasive aquatic vegetation (macrophytes, (Waltham and Fixler 2017), and engineered barriers impeding fish migration (Perna and Burrows 2005; Burrows et al. 2012). The Governments of Queensland and the Commonwealth of Australia recognised the need to reverse the decline in water quality reaching the GBR (GBRMPA 2014). As part of this response, funding for the "*Delivering biodiversity dividends to the Barratta Creek, Burdekin Catchment*" project was granted in 2013, with the aim of restoring ecological function into the creek/floodplain and the offshore Bowling

Green Bay Ramsar wetland. While the broader project covered riparian tree planting, fire management, feral animal management and extension training in land management for farmers, this study examined water quality and fish community occupying Lochinvah wetland. This wetland was treated with herbicides to control invasive aquatic plants (particularly species listed under Australian legislation as Weeds of National Significance; WONS) – most notably the water hyacinth (*Eichhornia crassipes*) which is one of the world’s most prevalent invasive aquatic plants (Toft et al. 2003; Villamagna and Murphy 2010; Montiel-Martínez et al. 2015). Water hyacinth occurs in wetlands on floodplains across northern Queensland, particularly the Burdekin floodplain because of permanent water and high nutrients (Waltham et al. 2020b). Clearing this declared WONS has become an obligation for local government agencies and natural resource management groups (Burrows et al. 2012), particularly in the lower Burdekin floodplain (Figure 1). Hyacinth growth in the Burdekin floodplain is fueled by an extensive water distribution network (over 1,500km of linear channels) that deliver irrigation water to sugar cane farms, with the tail water (rich in nutrients, sediments and herbicides) then flowing via coastal wetlands to Bowling Green Bay (Davis et al. 2014).

The Burdekin floodplain is dominated by highly modified distributary channels of the lower Burdekin River, which are used as a distribution network for turbid Burdekin River water for use in sugar cane irrigation and shallow groundwater recharge. Historically, the coastal floodplain was largely used for cattle grazing country. Given limited grass growth during the dry season, supplemental forage crops were created during the 1950s by using earth bunds to exclude seawater from otherwise tidally inundated drainage depressions along the coast. By preventing seawater intrusion, these areas became inundated with fresh water and permitted the growth of ponded pastures for late dry season cattle forage (Wildin 1991). Most of these bunded pastures remain today (Challen and Long 2004), and are now more perennial due to the near continual discharge of regulated flow tailwater to the lower catchment.

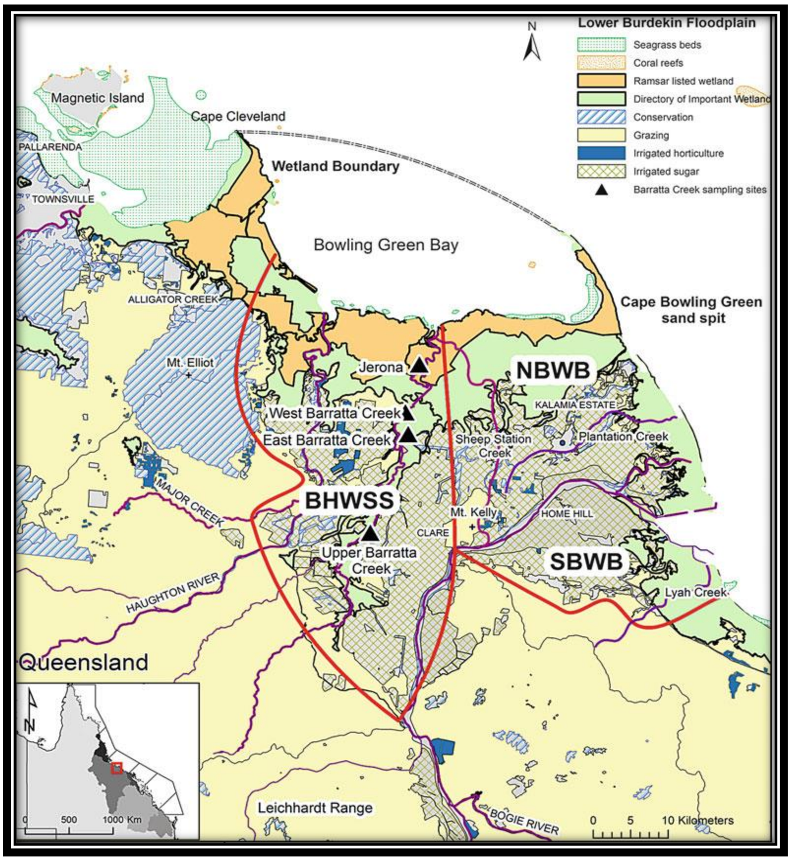


Figure 1: Map of lower Burdekin waterways depicting major land uses and catchment water quality sampling locations. Note the area of internationally (RAMSAR) that are recognised in the legislation,Pg and nationally recognised wetlands. Land use categories are based on 2004 Queensland Land Use Mapping (QLUMP) shape files obtained from the Queensland Department of Natural Resources and Water (DNRW) for the Burdekin catchment. Boundaries of the major floodplain irrigation scheme areas are indicated by red boundaries; BHWSS Burdekin Haughton Water Supply Scheme; NBWB North Burdekin Water Board; SBWB South Burdekin Water Board (Davis et al. 2014).

1.3 Project outline

This research study was designed to report on current restoration projects that are underway on the Burdekin floodplain by North Queensland Dry Tropics NRM (NQDT) and Greening Australia. In doing so we are building on the existing reports and knowledge that is available on the wetland (see Davis et al., 2014 for detailed review of historical data). Here we present a series of case study investigations into novel coastal wetland repair options that are proposed or being trialed on the Burdekin floodplain. These include:

1. Characterise saltwater ingress into palustrine wetland areas (to control freshwater invasive weeds) and track water hydrology in irrigation channels;
2. Mechanical removal of aquatic weeds to improve water quality and increase fish habitat;
3. Examine the potential bioengineering role of freshwater bivalves in processing available nutrients in floodplain wetlands, particularly given the potential role that building more recycle pits has on this floodplain as part of larger irrigation water management; and

4. Examine whether an on-going aquatic weed management partnership continues meeting the objective of water quality and fish habitat management.

The findings here will also assist other practioneers in the region, and will provide important data for the Queensland Government's Wetlands Program (Queensland Government).

For the purposes of this research, palustrine wetlands are defined as vegetated, non-riverine or non-channel systems and include billabongs, swamps, bogs, springs and soaks, and have more than 30% emergent vegetation (https://wetlandinfo.des.qld.gov.au/wetlands/ecology/aquatic-ecosystems-natural/palustrine/).

2.0 CASE STUDY ONE: USING SALTWATER INGRESS TO CONTROL FRESHWATER INVASIVE WEEDS

2.1 Overview

The delivery of irrigation water across the Burdekin floodplain continues to challenge managers where excessive (fresh) water, that is nutrient-laden, reaches sensitive palustrine wetland ecosystems. This results in expansion of freshwater aquatic weeds into mangrove and saltmarsh areas on the floodplain, weed chokes and poor water quality (hypoxic black water events during summer months) (Burrows et al. 2012). Managing the apparent oversupply of irrigation water into the palustrine areas has been a key focus of research and restoration efforts for many years. However, recent attempts to increase saltwater ingress through earth manipulation has had some success (Abbott et al. 2020). Here we have partnered with NQDT to characterise how saltwater and aquatic weeds respond to different irrigation water management approaches. The results are of interest to restoration practioneers, land holders and local water boards responsible for delivery of irrigation water across the floodplain.

2.2 Methods

2.2.1 High frequency loggers

Water depth, temperature and electrical conductivity were monitored by loggers (CTD-Diver, Eijkelkamp Soil & Water, Netherlands) at three study locations including: A) Jerona; B) Kalamia lagoon; and C) Saltwater Creek (Figure 2). The loggers were positioned at the bottom of the water column (~ 10 cm above the soil surface), logged ambient conditions every 20 minutes, and were downloaded during routine maintenance visits. Each logger was installed inside a PVC pipe (3m height, 90mm diameter) and fastened to a star picket (Abbott et al. 2020).

Traditional methods of controlling invasive aquatic weeds on the floodplain have included the use of chemical spraying (aerial, boat or from bank), and mechanical removal (Perna et al. 2012; Waltham and Fixler 2017). More recently, there has been an interest in the use of saltwater as a more ‘natural’ approach (Abbott et al. 2020). Reid et al (2018) examined the tolerance of four north Queensland weeds and one native freshwater sedge to different salinity treatments (% of seawater) and exposure durations. Using these data, we are able to calculate the frequency that conductivity (as a surrogate for salinity) occurred between 0-16, 16-27.5, 27-.5-38.5, 38.5-55, >55 mS/cm (upper limits equivalent to salinity at 30%, 50%, 70% and 100% seawater respectively) for durations of <7 days, 7-14 days, 14-21 days and >21 days. Using these same thresholds, we are able to directly compare exposure periods with weed tolerance.

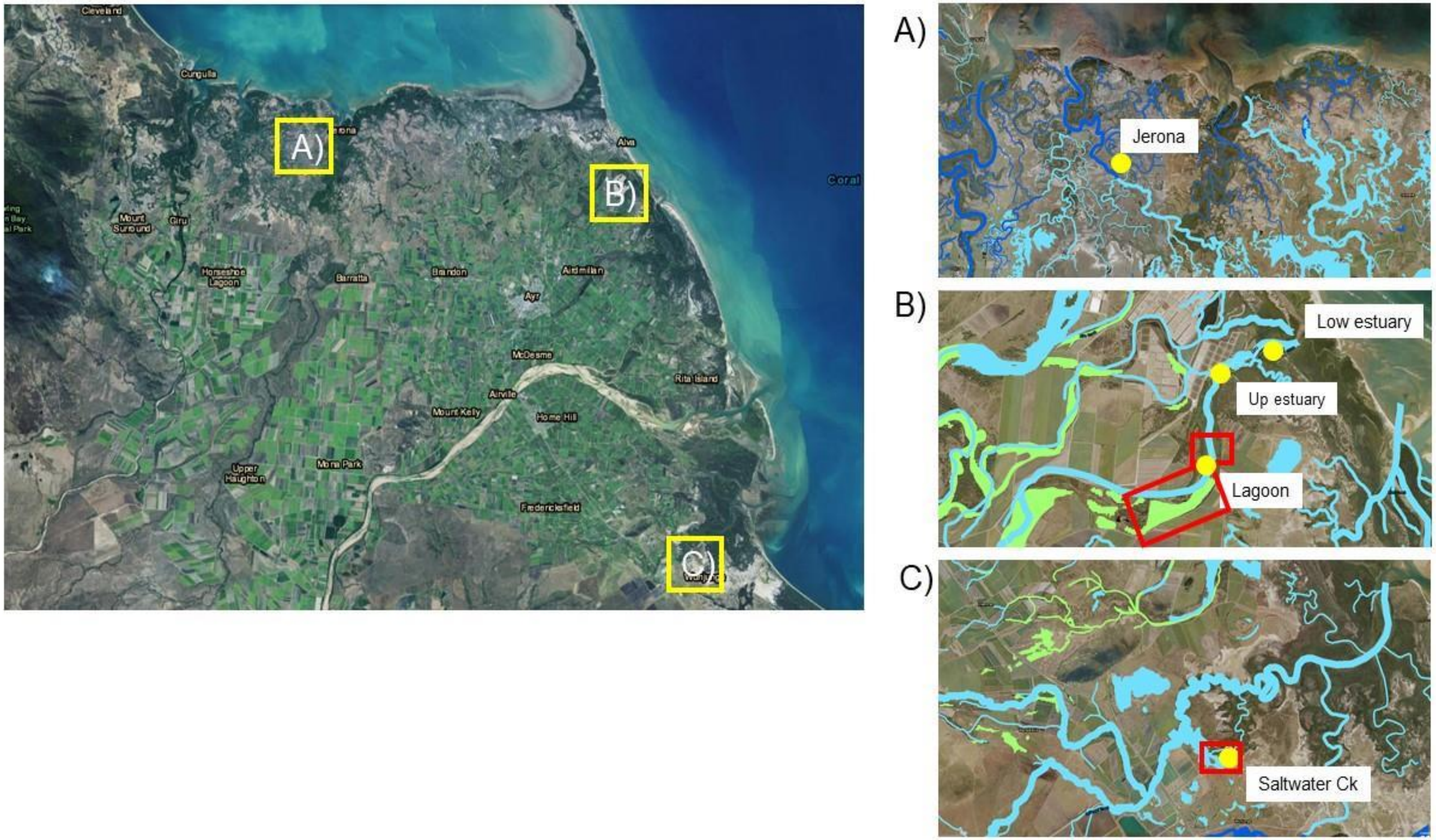


Figure 2: Location of CTD diver loggers measuring saltwater ingress on the floodplain (yellow circles). Inserts – overlay of Aqua Conservation Assessment data (WetlandInfo, Queensland Government) – Blue very high conservation value, light blue high conservation value and green medium conservation value. Red polygons are drone imagery analysis.

2.2.2 Field vegetation community

Aerial imagery of the study wetland area was collected using an unmanned aerial vehicle (UAV) (DJI Phantom 3). UAVs were flown at the Kalamia and Saltwater Creek sites between July 2018 and June 2020 to capture images during the wet (January) and dry seasons (June/July). Individual images were georectified and stitched together using the online platform 'DroneDeploy' (www.dronedeploy.com) to create an orthomosaic image for each survey at each site. The image resolution was set to 0.3 m. Drone images were stitched orthomosaic images with 3 bands (Red, Green and Blue). The high tide line or wetland extent was manually digitised using a polygon on each image. Where the boundary line could not be distinguished, the tree line was used as a proxy. This polygon was buffered by 5 m, to account for any potential error in delineating the high tide line and each image was clipped to the buffered polygon.

Image classification and accuracy assessment

Ground cover types were identified for classification into dominant classes. A training set was created for each orthomosaic image by assigning a minimum of 30 polygons for each ground cover type. In some cases, not all classes were present within an image; for example, some images contained no visible aquatic plants or water. In other cases, very small extents of the class were evident in images, such as small patches of acid sulfate soils which resulted in less than 30 training samples for that class. Image classification was performed with the 'Maximum Likelihood Classification' tool in ArcGIS (Version 10.7.1) using equal *a priori* probability weighting. In order to calculate the proportion of each ground cover class, the number of pixels in each class was extracted from ArcGIS. The proportion was calculated by dividing the number of pixels in each class by the total number of pixels (obtained by adding the number of pixels of each class together). In some cases, images contained a black border, where this was applicable it was included in classification as a N/A category. Any areas classified as N/A were removed from the total number of pixels so the proportion encompassed only the wetland extent.

Classification accuracy was assessed against 100 randomly distributed points that were manually classified and independent of the training dataset. Points were created using the 'Create Accuracy Assessment Points' tool in ArcGIS. The distribution of points across each orthomosaic was different between sites, while the position of points remained static within each site between survey events. Classification data was added to these points from the final classified image using the 'Update Accuracy Assessment Points' tool. Each point was then truthed by visually inspecting the image to confirm it had been classified correctly, any errors were manually corrected (these data points were also compared to field data records of vegetation community spatial data collected during field work). A confusion matrix for each site and survey was developed using the 'Compute Confusion Matrix' tool in ArcGIS to assess the accuracy of each classification. Classification accuracy was assessed using Cohen's Kappa (McHugh 2012) against 100 random points that had been manually classified independent of the training data. A value closer to 1 meaning the classification is more accurate and the map produced is more representative of reality.

2.3 Results and discussion

Conductivity in Kalamia estuary was highest in the lower estuary ranging between 18 – 55 mS/cm, while the mid-region ranged between 1 – 55 mS/cm, and the upper estuary logger located downstream of the bund wall was more fresh, particularly following February 2018 where levels remained below approximately 5mS/cm (Figure 3). It is interesting to note here in this time series that conductivity within the area of the bund wall reached levels indicative of salinity concentrations conducive for destroying freshwater weeds on two occasions, and really only for a few weeks (Table 1, Figure 3). These data present a case that the tidally driven inflow of marine water through the lower estuary rarely reaches the upper estuary, and for periods long enough to be impactful on controlling invasive freshwater weeds that exist downstream of the bund wall. The possible reason for this relates to either the leaking of the fish ladder on the earth wall, water level in the lagoon is maintained above the earth wall height and spilling into the upper estuary (particularly after rainfall events), or extraction of incoming tidal waters from the estuary by the local aquaculture facility. From this assessment, it seems unlikely that licenced extraction of incoming tidal waters in the estuary are the cause of freshwaters in the upper estuary, given several high tide events were allowed to “pass” by the facility during the project, and was not detected on the upstream logger. It is more probable that the management of freshwater in the lagoon, either through the fish ladder or overtopping the bank is the cause of the lower salinity conditions in the upper estuary.

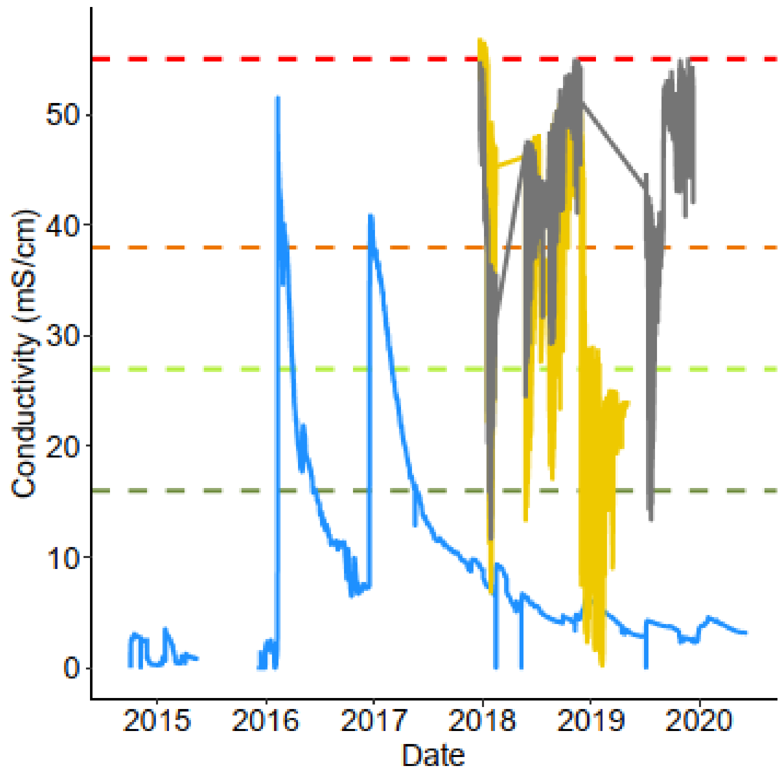


Figure 3: Continuous conductivity (mS/cm) at the Kalamia Lagoon (Lagoon) (blue), upper estuary site (gold) and the lower estuary site (grey) (see Figure 2). Dashed lines represent salinity equivalence at 30% seawater (dark olive), 50% seawater (light olive), 70% seawater (orange) and 100% seawater (red).

Table 1: The frequency, over the entire measured record, that the Kalamia Lagoon logger (Blue line in Figure 9) occurred at given conductivity levels (mS/cm) for given durations (days). NA indicates that the given conductivity was not held for the given continuous period at all.

Duration	Cond 16 - 27	Cond 27 - 38	Cond 38 - 55	Cond >55
0 - 6	1	NA	NA	NA
7 - 13	NA	NA	2	NA
14 - 21	NA	NA	1	NA
30 - 59	NA	1	NA	NA
60 - 89	NA	1	NA	NA
90 - 119	1	NA	NA	NA
120 +	1	NA	NA	NA

Conductivity measured at Jerona, following the bund wall removal, ranged between 1 and 90 mS/cm (Figure 5). This fluctuation is probably in response to regular (approximately every month) high tides which extend from the nearby estuary and reach the restoration site. In fact, conductivity reached levels much higher than seawater (approximately 55mS/cm) presumably in response to evapoconcentration. Rapid reduction in conductivity, to below 10mS/cm is likely a response to rainfall events in the region. The most noteworthy is the February 2019 decline which coincides with the tropical monsoon that delivered record breaking rainfall amounts in Townsville (032040), and widespread flooding. However, conductivity again increased by March 2019 towards 55mS/cm (i.e. marine values). The fluctuating conductivity at this restoration site is above tolerance thresholds for freshwater aquatic weeds. This is supported by reports that following the earth wall breach and saltwater ingress events, massive dieback of freshwater weed choke occurred.

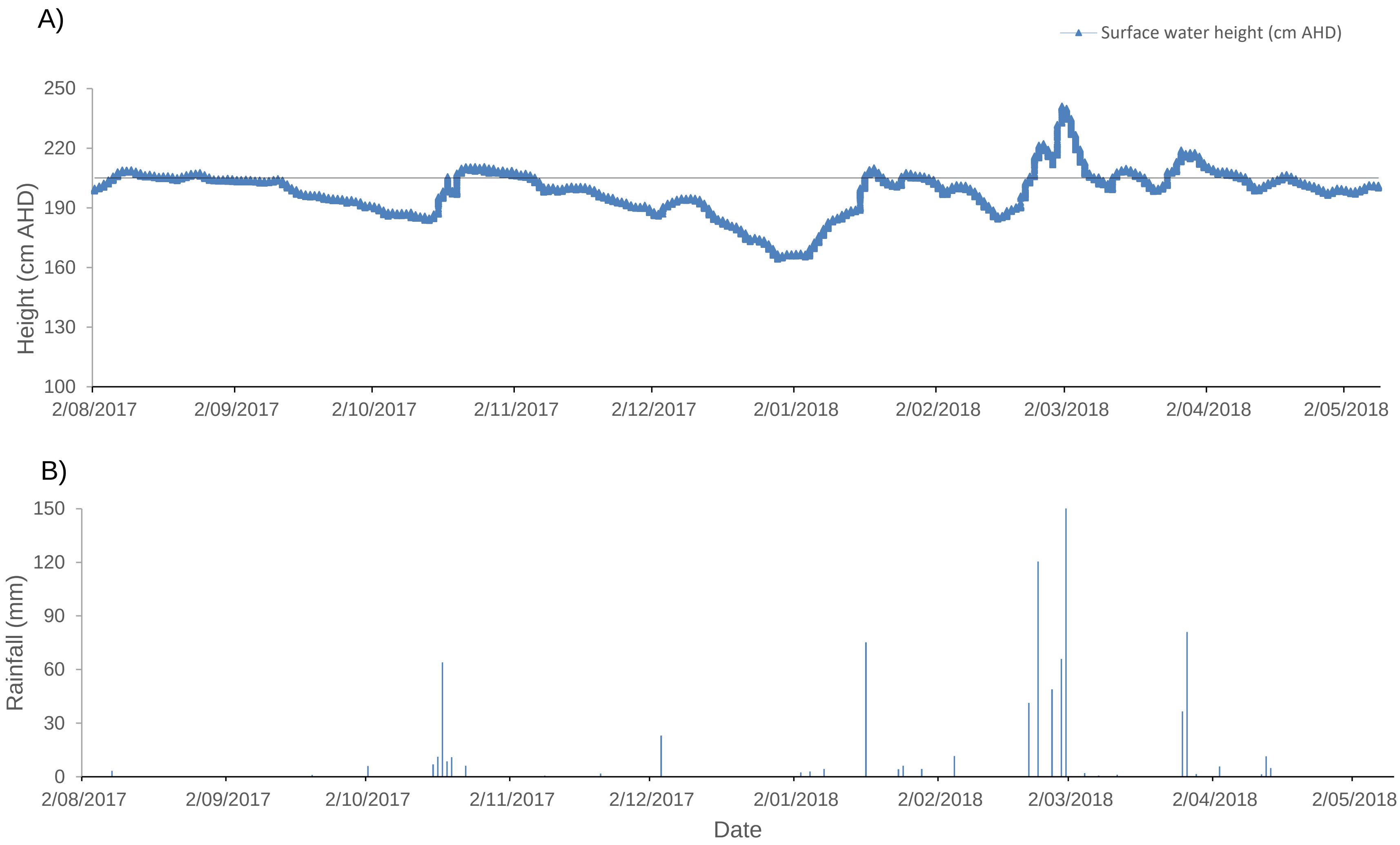


Figure 4: (A) Pyott's Lagoon surface water (blue) and spillway (grey) heights. (B) daily rainfall (mm) as measured at Burdekin Shire Council (Station 033001).

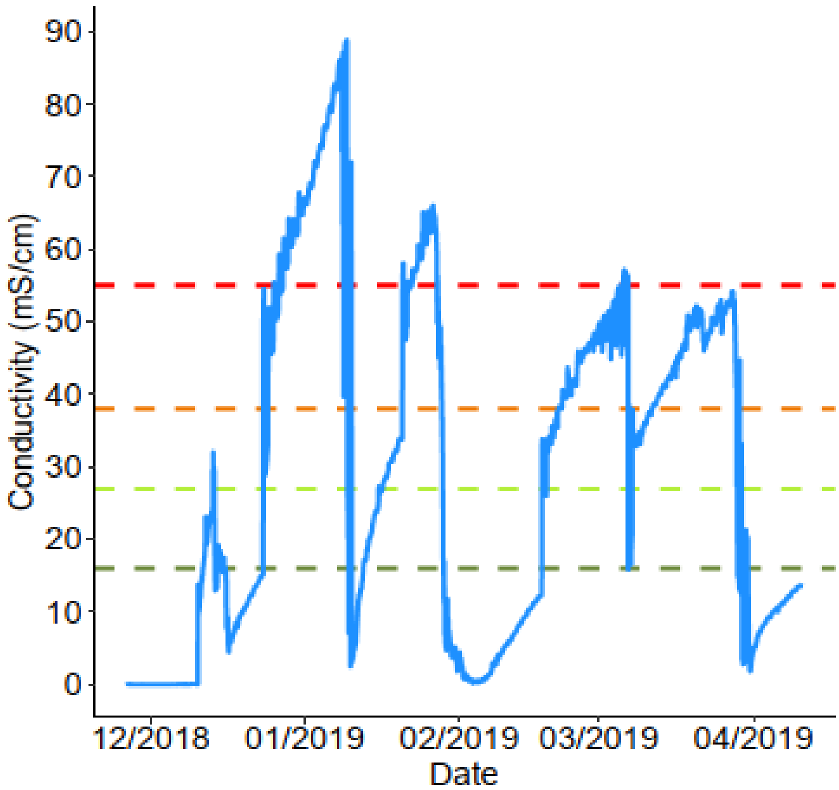


Figure 5: Continuous conductivity (mS/cm) at the Jerona site. Dashed lines represent salinity equivalence at 30% seawater (dark olive), 50% seawater (light olive), 70% seawater (orange) and 100% seawater (red).

Table 2: The frequency, over the entire measured record, that the Jerona site occurred at given conductivity levels (mS/cm) for given durations (days). NA indicates that the given conductivity was not held for the given continuous period at all.

Duration	Cond 16 - 27	Cond 27 - 38	Cond 38 - 55	Cond >55
0	1	1	NA	1
7	NA	1	2	1
14	2	1	2	1
30	1	1	NA	NA
NA	5	5	5	4

Conductivity measured at Saltwater Creek was persistently below the levels necessary to destroy invasive weeds, remaining below 3mS/cm during the logging period (Figure 5). This provides evidence there is minimal chance that saltwater reached the logger and therefore implies no effective destruction of freshwater aquatic weeds in the wetland area upstream of the logger.

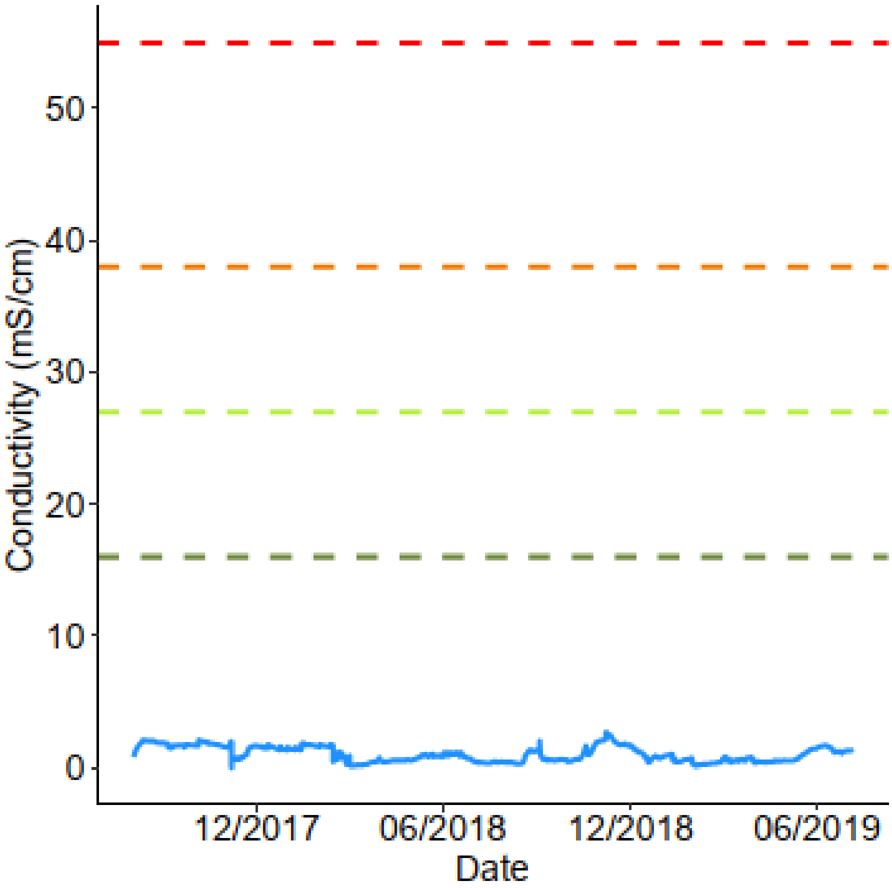


Figure 6: Continuous conductivity (mS/cm) at the Saltwater Creek site. Dashed lines represent salinity equivalence at 30% seawater (dark olive), 50% seawater (light olive), 70% seawater (orange) and 100% seawater (red).

2.3.3 Drone surveys

Wetland vegetation classification derived from drone images for Saltwater Creek are shown in Figure 7. It was not possible to decipher a measurable change in the habitat features at this restoration site, at least over the 12mth project (Table 3). The changes in habitat features could reflect seasonality. Importantly for this project site, it seems too soon after the installation of the recycle pit to measure any reduction in aquatic weeds – this would require future repeated imagery analysis, which is possible given that now the drone methodology and processing methodology has been developed. Similarly, little data is also available for Kalamia lagoon. However, until further management intervention in the area occurs, it will be/maybe difficult to detect any measurable reduction in freshwater weeds. Despite the very high conservation value of these restoration sites, or adjacent areas, there were no new marine plants identified during the field work and processing of imagery. However, this could change in the future under a scenario of increased saltwater ingress at these sites.

Table 3: Proportion of land cover from classified imagery (Maximum likelihood classification performed in ArcGIS) at Saltwater Creek taken from UAV imagery. Graph shows comparison between imagery taken in July 2019, October 2019 and June 2020.

Habitat (%)	July 2019	October 2019	June 2020
Open water	3.1	9.4	22.5
Soil	18.6	19.7	14.8
Green grass/plants	48.4	39.3	0.0
Brown grass/plants	24.5	25.7	44.3
Aquatic vegetation	5.4	5.9	18.4
Total (%)	100	100	100

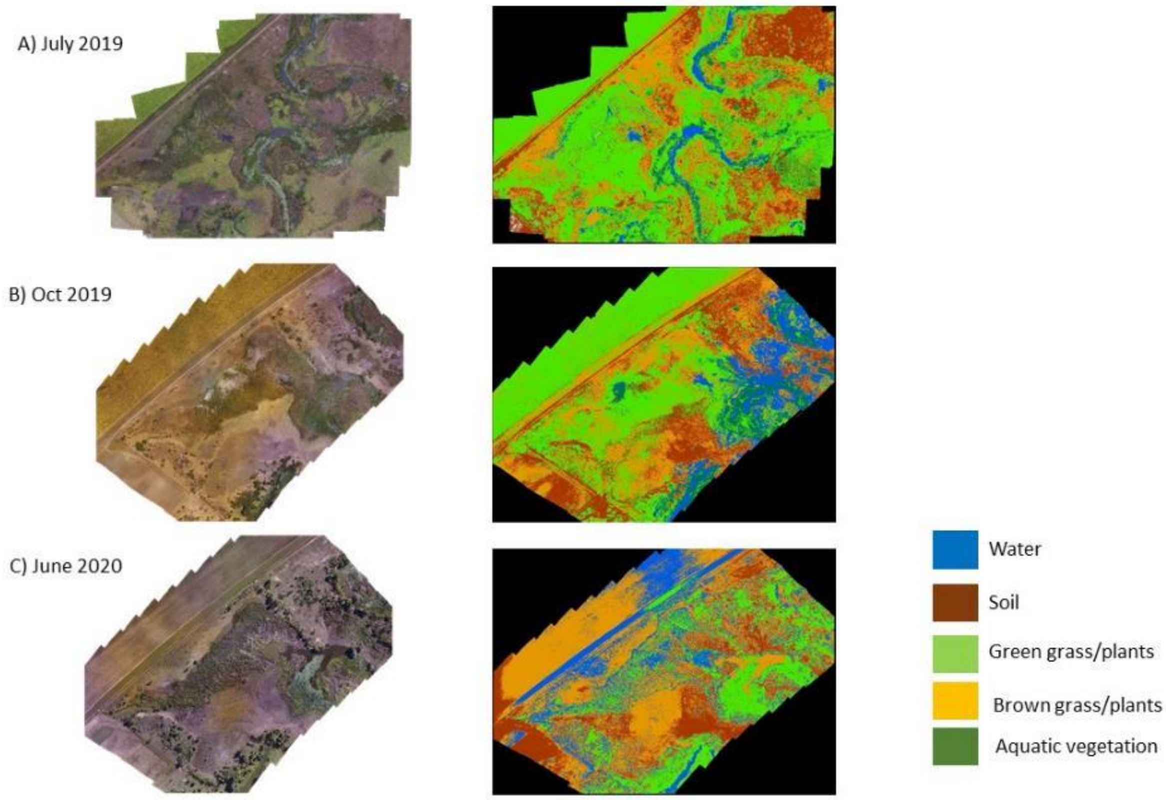


Figure 7: Vegetation classification from aerial imagery at Saltwater Creek: a) July 2019; b) Oct 2019; and c) June 2020.

3.0 CASE STUDY TWO: FRESHWATER BIVALVES FOR WATER QUALITY IMPROVEMENTS

3.1 Overview

The extent to which freshwater bivalves provide ecosystem services, such as biofiltration and nutrient storage, is highly dependent on their environmental context, e.g. hydrology, available food and nutrients, dissolved oxygen levels, and water temperature (Strayer 2017). Water temperature will affect bivalve physiology (Hochachka and Somero 2002), most notably their filtration capacity (Baker and Hornbach 2001; Ganser et al. 2015). Also, when thermally stressed, some bivalve species will switch from anabolic to catabolic processes, which leads to increased dissolved oxygen consumption, and higher nutrient excretion rates (Spooner and Vaughn 2008). These thermoregulatory responses pose important implications when considering climate change scenarios (Lopes-Lima et al. 2018). In addition to environmental context, the size of individual bivalves can influence their biofiltration and biodeposition rates and should be considered when quantifying capacity to provide ecosystem services.

Farmers on the floodplain have noticed the presence of freshwater bivalves in their water drainage and delivery network, and were interested in their role in processing nutrients. We investigated the capacity of freshwater bivalves (*Corbicula australis*) to filter and improve water quality, particularly in the context that if more recycle pits are constructed as part of a broader irrigation water management and water quality program, then the role of these bivalves is important in the processing of nutrients. This native freshwater bivalve is present throughout the floodplain system. We selected six freshwater wetlands (Figure 8) for the collection of *C. australis,* and measured their filtration and biodeposition rates. Three of the selected wetlands were natural and three were artificially constructed, allowing measurements to occur under a range of ambient environmental conditions between October and December 2017. Laboratory experiments were also conducted with individuals collected from these wetlands to determine the influence water temperature has on filtration and biodeposition rates (see experimental procedures outlined below). Filtration experiments measured the rate of chlorophyll *a* removal via filter-feeding by individual bivalves (i.e., clearance rate); while biodeposition experiments measured the rate at which individuals deposited faeces and psuedofaeces by proxy of total nitrogen and total phosphorus content of their biodeposits (i.e., biodeposition rate).

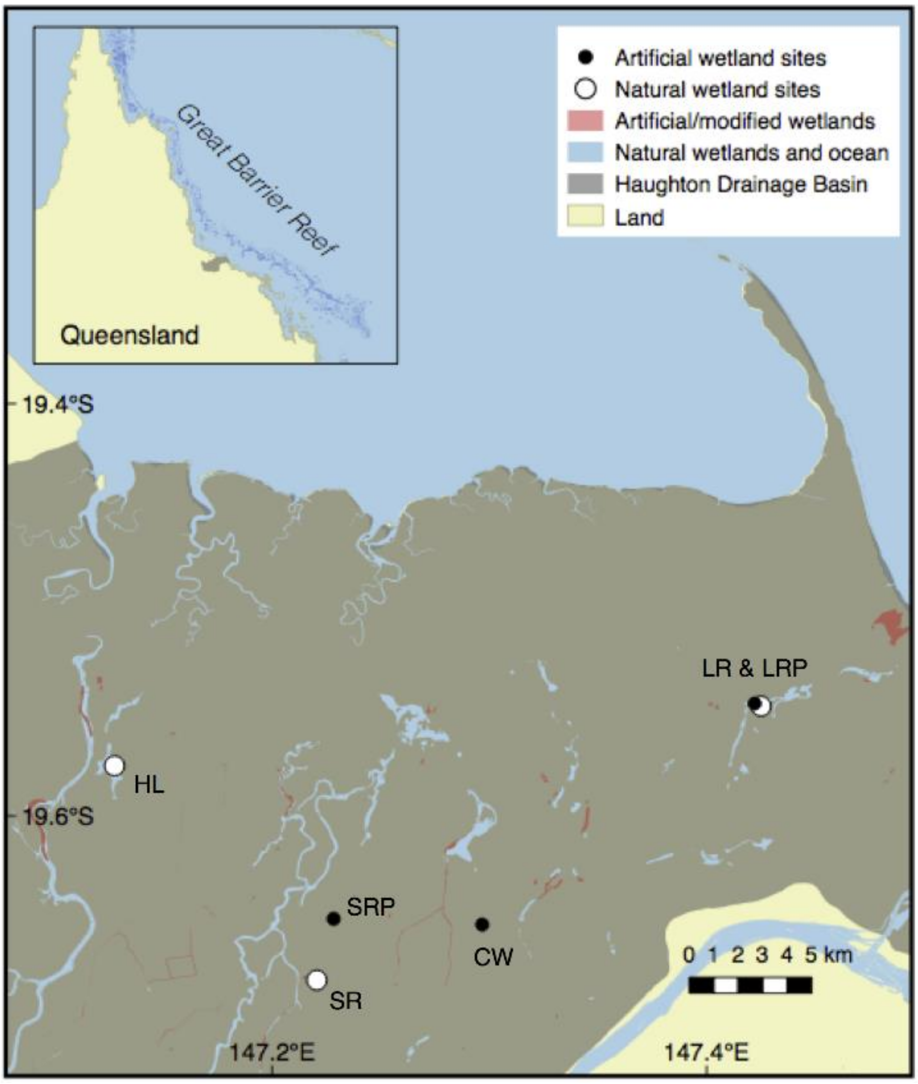


Figure 8: Map of natural and artificial coastal wetland study sites in the Haughton catchment, Queensland, Australia: Horseshoe lagoon (HL), Sheep Station recycle pit (SRP), Sheep Station river (SR), Constructed wetland (CW), Lilliesemere river (LR), and Lilliesmere recycle pit (LRP).

3.2 Methods

3.2.1 Bivalve population density sampling, size range measurement and collection

Population density

While six wetland sites were used for collection of bivalves and measurement of filtration and biodeposition rates in the field, only four were randomly sampled to estimate bivalve population densities. Sediment was collected from the benthic surface of wetlands with a Ponar grab operated from a boat (grab samples were ~ 2.4 L in volume). One litre of sediment was sieved (1 mm mesh size) from each grab sample to determine bivalve density, and all measures of bivalve density were reported as number of individuals/m^2 (1 litre of sediment = 0.01 m^2).

Size range measurement and collection

During October to December 2017, 30-40 bivalves were collected from each wetland either along the shoreline with a sieve (1 mm mesh size), or by boat with a Ponar grab. Calipers were used to measure the anterior-to-posterior shell length and dorsal-to-ventral shell width of each individual, to a precision of 0.1 mm. Sixteen individuals from each wetland were randomly selected and kept for testing filtration and biodeposition rates in the field.

In December 2017, an additional 60 bivalves were collected (pooled from the six wetland sites) for temperature experiments and transferred to an outdoor laboratory at James Cook University, QLD, Australia. The bivalves were held in aerated plastic containers (100cm X 45cm X 20cm) with sediment and water collected from the wetlands for one week to acclimate to the new conditions. During the holding period, bivalves were fed phytoplankton cultured from the wetland water (Spooner and Vaughn 2008).

3.2.2 Field experimental procedures

Environmental condition and water temperature profiling

A calibrated Quanta multi-probe was used to measure water temperature (°C), conductivity (mS/cm), dissolved oxygen saturation (%), and pH at the bottom sediment boundary layer of each wetland. Secchi depth (cm) was measured to estimate light attenuation. Chlorophyll a samples (n=3) were collected in black bottles, placed on ice, and filtered through a glass fiber filter (GF-75, 0.45µm pore size) within 24 hours following collection. To determine nutrient concentrations (TN = total nitrogen and TP = total phosphorus; n = 1), water samples were collected in 60 ml falcon tubes and immediately placed on ice for transport to the laboratory. Subsequently, chlorophyll a, and nutrient samples were analysed at the TropWATER Water Quality Laboratory, James Cook University, QLD, Australia.

To profile wetland water temperatures experienced by bivalves throughout the year, Hobo high frequency temperature loggers (Onset Corporation Bourne, Massachusetts) were deployed at two depths: 1) surface (0.2 m below water surface); and 2) bottom (0.1 m above the profile bottom) at the Lilliesmere River (LR) site Figure 8. The surface logger was attached to the underside of a 0.15 m diameter buoy to shield it from the sun at all times as direct exposure could produce erroneous results (Wallace et al. 2015; Wallace et al. 2017). Loggers were set to record data every 20 min from July 2017 to April 2018.

Filtration and biodeposition experiments

Prior to initiation of field experiments, sediment and algae were removed from bivalves by gentle scrubbing with a soft toothbrush. For the filtration experiments, individual bivalves were placed in 500 ml plastic cylindrical containers filled with water collected from the wetland (n = 8). For biodeposition experiments, individual bivalves were placed in 60 ml falcon tubes containing pure Milli-Q water (n = 8). Both filtration and biodeposition experiments had two additional water-filled containers without bivalves to serve as controls. Experimental and control containers (16 in total) were placed in a water bath filled with water collected from the wetland and allowed 20 minutes to equilibrate to the ambient water temperature before experiments began (Figure 9).

Following water temperature equilibration, the filtration and biodeposition experiments ran for a total incubation period of two hours (Cyr et al. 2017), with temperature recorded every ten minutes. Following the two-hour incubation period, bivalves were removed from their experimental containers, transferred to a plastic collection bag, and stored frozen on ice. Once in the laboratory, the shell length and width (mm) of each experimental bivalve was measured, body tissue was removed and oven-dried at 60°C for 48 hours, and tissue dry weight (g) was recorded.

Following removal of bivalves, the water remaining in each experimental container was placed on ice and transported to the TropWATER Water Quality Laboratory at James Cook University, QLD, Australia for analysis. For filtration experiments, the final chlorophyll a concentration following the two-hour incubation period was measured and clearance rate (CR; (L/bivalve/hr) was calculated with the following equation (Coughlan 1969):

$$\mathrm{CR} = \frac{V}{T} \left\{ \ln\frac{C_0}{C_t} - \ln\frac{C_{0'}}{C_{t'}} \right\} \qquad (1)$$

where V is the volume (L) of water, T is the incubation time (h), C0 is the initial chlorophyll a concentration, Ct is the final chlorophyll *a* concentration. C0' and Ct' are initial and final chlorophyll a concentrations, respectively, in the controls and account for any change in chlorophyll a concentration during the incubation period.

For biodeposition experiments, the total nitrogen (TN) and total phosphorus (TP) deposited by each bivalve was measured. Additionally, total dissolved nitrogen (TDN) and total dissolved phosphorus (TDP) excreted by individual bivalves was measured in a subset of the samples (TDN: n = 19 and TDP: n = 17). Biodeposition rates (BR; μg/bivalve/hour) were calculated separately for total nitrogen and total phosphorus with the following equation:

$$\mathrm{BR} = \frac{V}{T}\{C_t - C_{t'}\} \qquad (2)$$

where V is the volume (L) of water, T is the incubation time (h), Ct is the total deposited nutrient concentration (TN or TP), and Ct' is the total nutrient concentration in the controls.

a)

b) c)

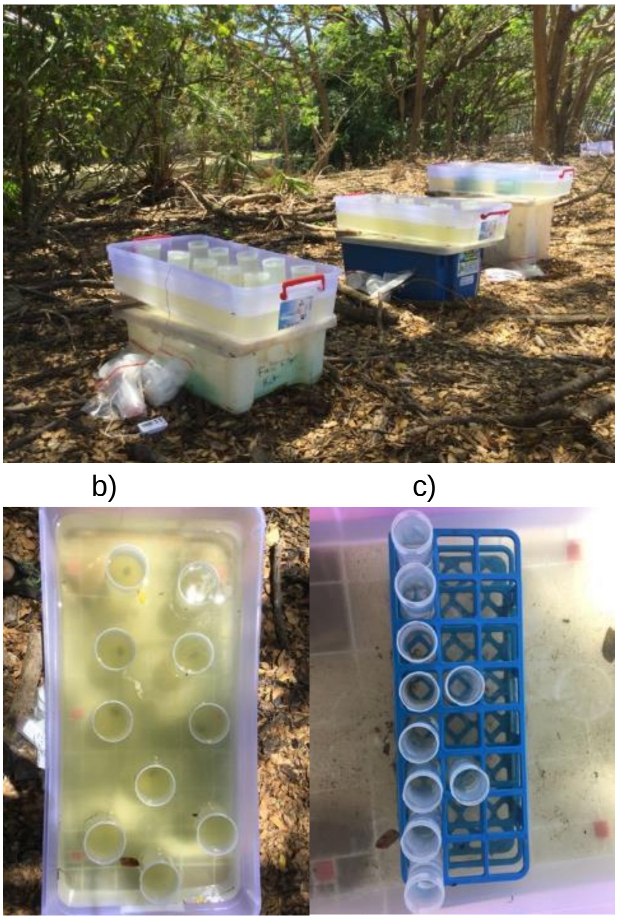


Figure 9: Experimental set-up in the field: a) water baths with experimental containers holding individual bivalves; b) overhead view of filtration experiments (500 ml plastic containers); and c) overhead view of biodeposition experiments (60 ml falcon tubes).

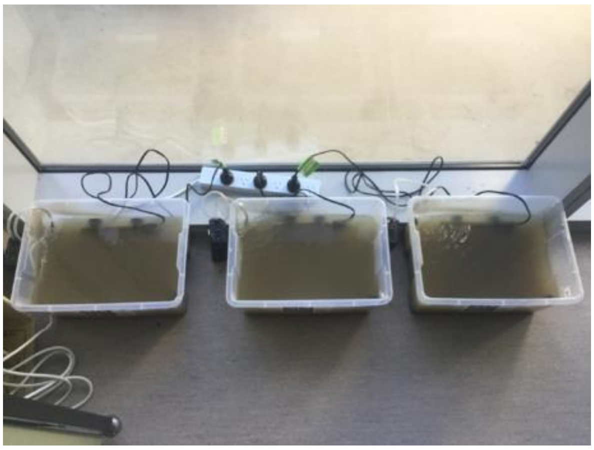
3.2.3 Laboratory experimental procedures

Temperature acclimation

Laboratory experiments were conducted to determine the effect of temperature on bivalve filtration and biodeposition rates. Three temperature acclimation tanks (35cm X 25cm X 25cm) were set up in an indoor laboratory with sediment and water collected from the study wetlands and were placed near large windows to provide a natural diel light cycle (see Supplementary Materials for experimental set-up in the laboratory). Aquarium thermostats (100 watt) were used to control temperature in the acclimation tanks to a constant 23°C, 27°C, or 32°C. Sixteen bivalves were held in each temperature treatment and allowed to acclimate for one week. The bivalves were fed phytoplankton (here measured as chlorophyll-a) cultured from wetland water on a daily basis to ensure that levels remained sufficient (6-8 μg/L) throughout the acclimation period (Figure 10).

After temperature acclimation, filtration and biodeposition experiments followed the same protocol as field experiments (n = 8 for either filtration or biodeposition experiments in each of three temperature treatments). Aquarium thermostats (250 watt) were used to maintain water baths and experimental containers at treatment temperatures during the two hour incubation period.

a)



b)

Figure 10: Experimental set-up in the laboratory: a) temperature acclimation tanks and b) water baths with experimental containers during the two hour incubation period.

3.2.4 Statistical analysis

Kolmogorov-Smirnov (K-S) tests determined if there were differences in the shapes of bivalve size and density distributions between natural and artificial wetlands. A difference in the mean size of bivalves in natural and artificial wetlands was also evaluated using a two-sample t-test. Assumptions of normality and equal variance were assessed by visual inspection of a quantile-quantile plot and Levene's test for homogeneity of variance. Analysis of covariance (ANCOVA) was used to define the slope of the relationship between shell length (mm) and tissue dry weight (mass (g)) for bivalves measured in natural and artificial wetlands with the following model:

$$\log_{10}DW = a + b \times \log_{10}L + c \times W + d \times [W \times \log_{10}L] \quad (3)$$

where DW is the tissue dry weight (g), L is shell length (anterior-to-posterior in mm), W is wetland type (i.e., artificial or natural), and the interaction term is in square brackets.

ANCOVA was also used to define the allometric relationships between tissue dry weight (g) and clearance or biodeposition rates with the following model:

$$\log_{10}Y = a + b \times \log_{10}DW + c \times W + d \times [W \times \log_{10}DW] \quad (4)$$

where DW is the tissue dry weight (g), Y is clearance rate (CR, L/bivalve/hr) or biodeposition rate (BR, μg/bivalve/hour), W is wetland type (i.e., artificial or natural) or temperature treatment (i.e., 23°C, 27°C, or 32°C), and the interaction term is in square brackets. Only clearance and biodeposition rates above zero were included in the analysis.

Assumptions of linearity, normality, and homogeneity of variance were assessed by visual inspection of a quantile-quantile plot and a plot of residual vs. fitted values. ANCOVA also assumes homogeneity of slopes among covariate groups, and this was evaluated through significance of the interaction term included in the model. If an interaction term was significant, this indicated that the slope of the relationship differed between covariate levels (e.g., natural vs. artificial wetlands, or temperature treatments). Where slopes differed, relationships were defined separately.

If the allometric relationship for clearance or biodeposition was significant ($p < 0.05$), the equation defining the allometric relationship was used to calculate the average rates of bivalve clearance and biodeposition. This allowed the size distribution of bivalves in natural and artificial wetlands to be accounted for in clearance and biodeposition estimates and involved the following steps: 1) Equation 3 was used to predict individual body mass (dry weight) from shell lengths measured in artificial and natural wetlands; and 2) a boot-strapping procedure (999 iterations) was used to calculate average clearance and biodeposition rates from predicted dry weights with Equation 4. If the allometric relationship for clearance or biodeposition was not significant, average clearance and biodeposition rates were calculated directly without accounting for bivalve size distributions.

Following calculation of average clearance and biodeposition rates, either a two-sample t-test or a one-way ANOVA was used to determine whether the mean rates differed between artificial and natural wetlands or among temperature treatments. Assumptions of normality and equal variance were assessed by visual inspection of a quantile-quantile plot and Levene's test for homogeneity of variance across groups. Mean bivalve population densities (individuals/m^2) measured in artificial and natural wetlands were multiplied by the estimates of hourly clearance and biodeposition rates. This quantified the hourly rate of water clearance and nutrient biodeposition provided by freshwater bivalves in artificial and natural wetlands (L/hour/m^2).

Finally, the correlation between total nutrient biodeposition (total nitrogen (TN) or total phosphorus (TP)) and total dissolved nutrient excretion (total dissolved nitrogen (TDN) or total dissolved phosphorus (TDP)) by individual bivalves was evaluated using a Pearson correlation test. Assumptions of normality, linearity, and homogeneity of variance were evaluated as described above. All statistical analyses were performed in R (R Core Team 2017). Linear models were fit using the R function 'lm', and bootstrapping was performed using the R function 'bootCase' in the car package.

3.3 Results

3.3.1 Bivalve density and size distribution

Bivalve populations showed clumped density distributions in both natural and artificial wetlands ranging from 0 to 10,500 individuals/m^2 (Table 4; Figure 11; Figure 13). The constructed wetland had the highest bivalve density (maximum = 10,500 individuals/m^2) relative to natural wetlands (maximum = 1,700 – 7,400 individuals/m^2) (Table 4). The shape of bivalve density distributions did not differ between artificial and natural wetlands (K-S test statistic (D) = 0.346, p = 0.151; Table 4), however, the mean density was higher in artificial wetlands (2,417.6 ± 789.6 individuals/m^2) compared to natural wetlands (710.3 ± 280.9 individuals/m^2) (Table 4).

Shell lengths ranged from 6.1 to 24.3 mm, and mean size was higher in artificial wetlands (13.6 ± 0.48 mm) compared to natural wetlands (12.4 ± 0.27 mm; t = 2.60, df = 167.74, p = 0.01; Figure 11; Table 4). The frequency distribution of shell lengths (mm) also differed between natural and artificial wetlands (K-S test statistic (D) = 0.298, p = 0.001; Table 4). Visual comparison of the frequency distributions shows that individuals with shell lengths > 15 mm were more frequent in artificial wetlands compared to natural wetlands (Table 4).

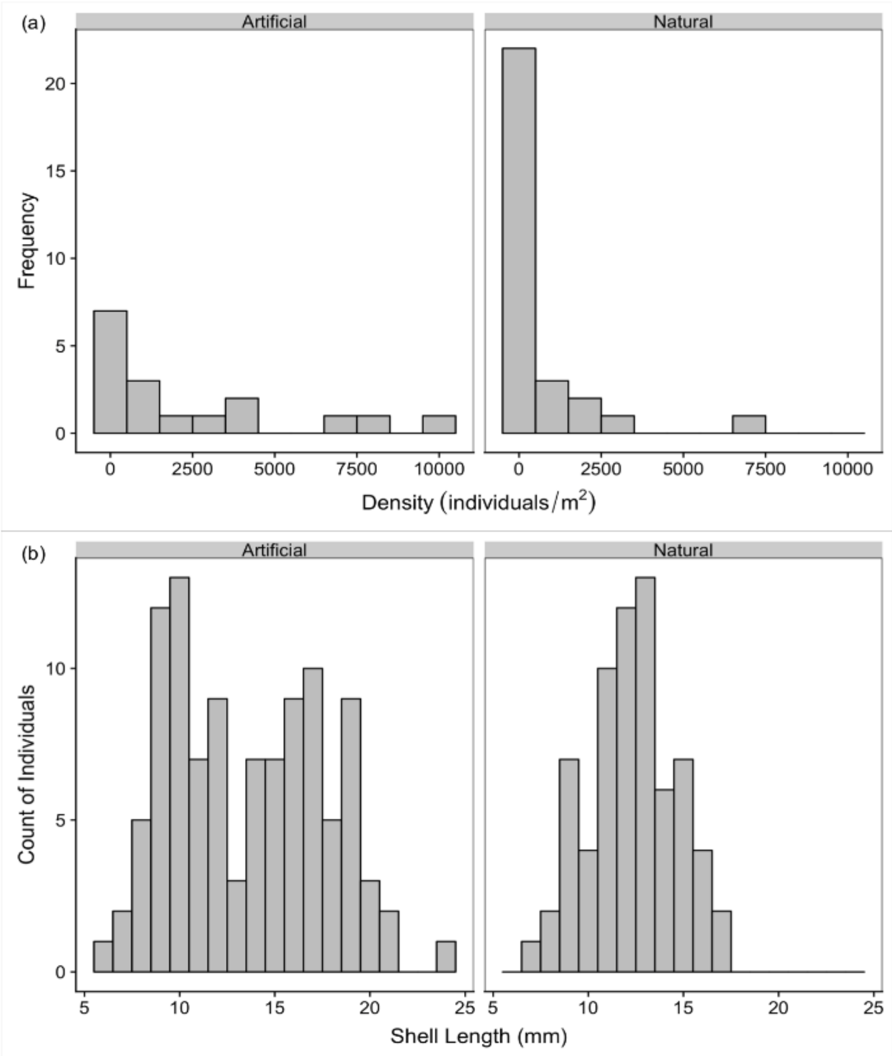


Figure 11: Frequency distributions of bivalve density (a) and size (b) in artificial and natural wetlands.

Table 4: Estimates of individual bivalve weight, clearance and biodeposition rates in the field and in temperature-controlled laboratory experiments.

		Field				Laboratory					
	Measured or predicted	**Natural wetlands**	**n**	**Artificial wetlands**	**n**	**Temperature 23°C**	**n**	**Temperature 27°C**	**n**	**Temperature 32°C**	**n**
Shell length (mm)	Measured	12.4 ± 0.27	68	13.6 ± 0.48	105	-		-		-	
Density (individuals/L)	Measured	7.1 ± 2.81	29	24.2 ± 7.90	17	-		-		-	
Tissue dry weight (g)	Predicted with Equation 5	0.016 ± 0.001*	-	0.024 ± 0.002**	-	-		-		-	
Clearance rate	Measured (L/bivalve/hr)	0.19 ± 0.04	20	0.34 ± 0.07	20	0.37 ± 0.07**	6	0.12 ± 0.03*	5	0.12 ± 0.03*	4
	Extrapolated (L/hour)	1.35	-	8.23	-						
Phosphorus biodeposition rate	Predicted (μg P/bivalve/hour) with Equation 6 (field) or Equation 7 (temperature treatments 23°C and 27°C), or measured (temperature treatment 32°C)	0.77 ± 0.02*	-	0.87 ± 0.03**	-	0.83 ± 0.02*			-	1.16 ± 0.1**	6
	Extrapolated (μg P/hour)	5.47	-	21.05	-						
Nitrogen biodeposition rate	Predicted (μg N/bivalve/hour) with Equation 8 (field) or measured (laboratory)	2.62 ± 0.11*	-	3.35 ± 0.19**	-	2.9 ± 0.5	8	4.6 ± 1.3	7	5.5 ± 1	6
	Extrapolated (μg N/hour)	18.60	-	81.07	-						

Note: Clearance and biodeposition rates that were significantly different between natural and artificial wetlands or temperature treatments are marked by asterisks (* and **).

3.3.2 Wetland seasonal water temperature profiles

The cumulative frequency of surface and bottom water temperature recorded in winter (July 2017 to October 2017) and summer (November 2017 to April 2018) at Lilliesmere River shows that summer water temperatures are higher at the surface and bottom of the wetland when compared to winter months (Table 5; Figure 12). To put the controlled temperature experiments (23°C, 27°C, and 32°C) into context, the percentage of time these water temperatures occurred during summer or winter months was calculated. During the winter, bottom and surface water temperatures of 23°C were recorded 16 - 26% of the time, while water temperature was always above 23°C during summer months. Water temperatures of 27°C were recorded 8 - 10% of the time during the winter, and 12 - 33% of the time during summer months (surface and bottom temperatures, respectively). Water temperatures never reached 32°C at the bottom of the wetland during either summer or winter months, and rarely at the surface of the wetland (1% of the time during winter months, and 6% of the time during summer months).

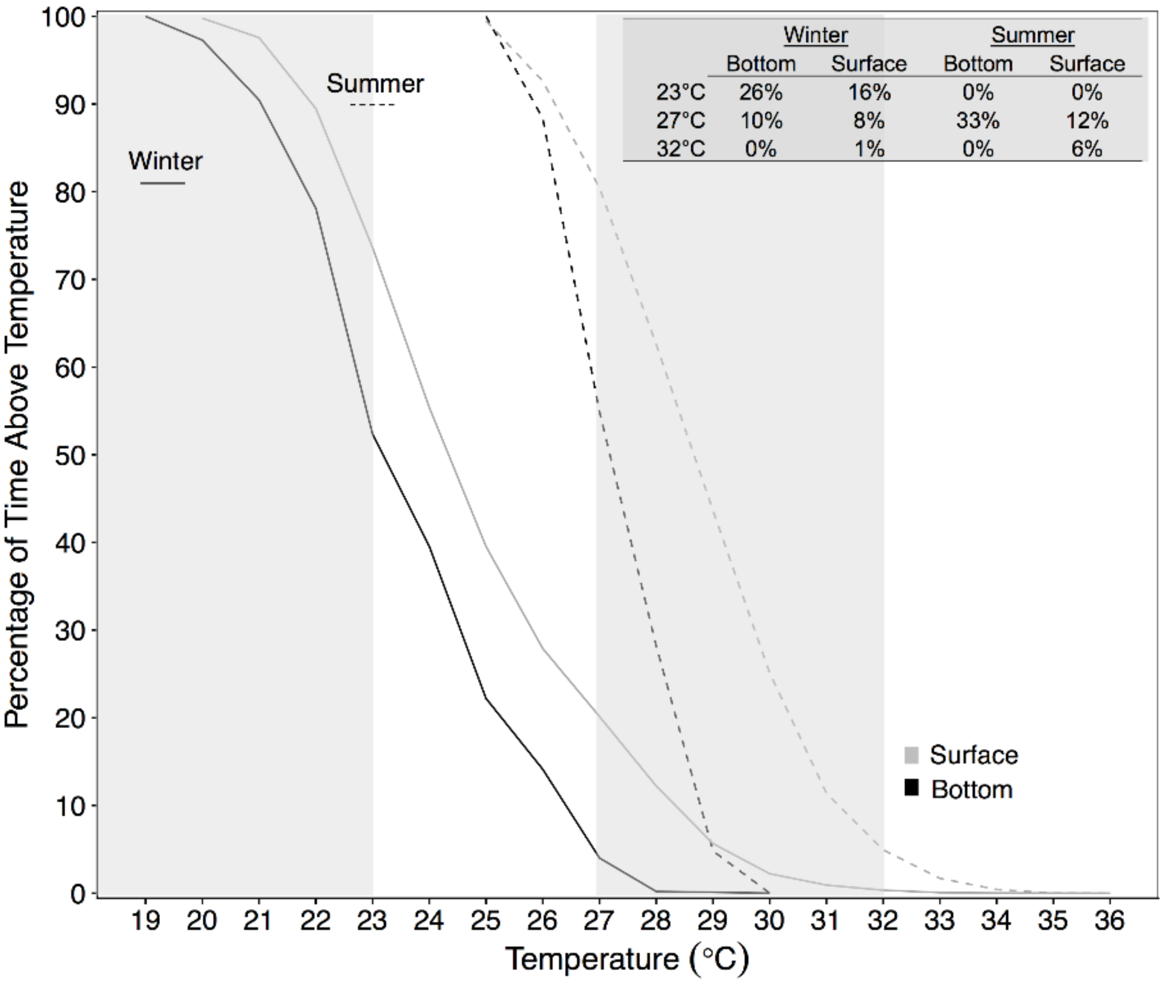


Figure 12: Cumulative frequency plot showing the percentage of time above temperatures in Lilliesmere River. Solid lines represent winter months (May to October) while dashed lines represent summer months (November to April) for temperatures logged at the bottom (black) and surface (grey) of the wetland. The inset table shows the percentage of time at each temperature: 23°C, 27°C, and 32°C.

Table 5: Environmental conditions measured in wetlands during field and laboratory experiments. The range of bivalve population density measurements in four of the study wetlands are also provided.

Site or Treatment	Date	Temperature (°C)	Conductivity ms/cm	Dissolved oxygen (%)	pH	Chlorophyll *a* (µg/L) ± se	TN (µg N/L)	TP (µg P/L)	Density (Ind/L)	Filtration experiments			Biodeposition experiments		
										n	Shell length (mm)	Incubation temperature (°C) ± se	n	Shell length (mm)	Incubation temperature (°C) ± se
Field															
Natural wetlands															
Horseshoe Lagoon	29/11/17	30.9	348	89.9	7.6	11 ± 0.9	1018	170	0 - 74	8	14.1 - 19.7	32.8 ± 1	8	12.0 - 14.0	32.8 ± 0.8
Lilliesmere River	27/10/17	27.1	266	46	8	11.3 ± 0.51	1018	170	0 - 17	8	11.6 - 15.8	29.6 ± 0.6	8	10.8 - 13.9	30.7 ± 0.5
Sheepstation River	19/12/17	32	207	80.3	8	5.4 ± 0.28	432	27	-	8	14.9 - 18.4	31 ± 0.4	8	13.7 - 18.3	31 ± 0.4
Artificial wetlands															
Lilliesmere Recycle Pit	27/10/17	28	277	87.5	9.5	46.1 ± 2.31	1332	827	0 - 38	8	17.7 - 21.1	30.3 ± 0.6	8	14.9 - 20.4	30.7 ± 0.5
Sheepstation Recycle Pit	18/12/17	31.5	268	66.3	7.1	6.5 ± 0.02	1250	279	-	6	16.6 - 19.8	32 ± 1.4	5	16.7 - 18.3	32 ± 1.4
Constructed Wetland	29/11/17	27.1	345	35	6.8	7.9 ± 0.7	771	80	0 - 105	7	11.3 - 15.6	32.8 ± 0.8	8	8.2 - 13.2	32.8 ± 0.8
Laboratory															
Temperature 23°C	13/12/17	23.2	-	99.1	-	7.9 ± 0.26			-	8	11.9 - 20.3	23.4 ± 0.1	8	11.5 - 19.5	23.4 ± 0.1
Temperature 27°C	13/12/17	27.3	-	98.2	-	9.6 ± 0.24	1304*	1239*	-	8	14.8 - 19.8	27.2 ± 0.2	7	11.5 - 18.7	27.2 ± 0.2
Temperature 32°C	13/12/17	31.7	-	98.3	-	6.8 ± 1.2			-	8	11.6 - 19.5	32.5 ± 0.1	8	14.5 - 20.3	32.5 ± 0.1

*Note: water used in temperature experiments was sourced from the same wetland, and so only one measurement each of TN and TP was taken to represent water used in all three temperature experiments.

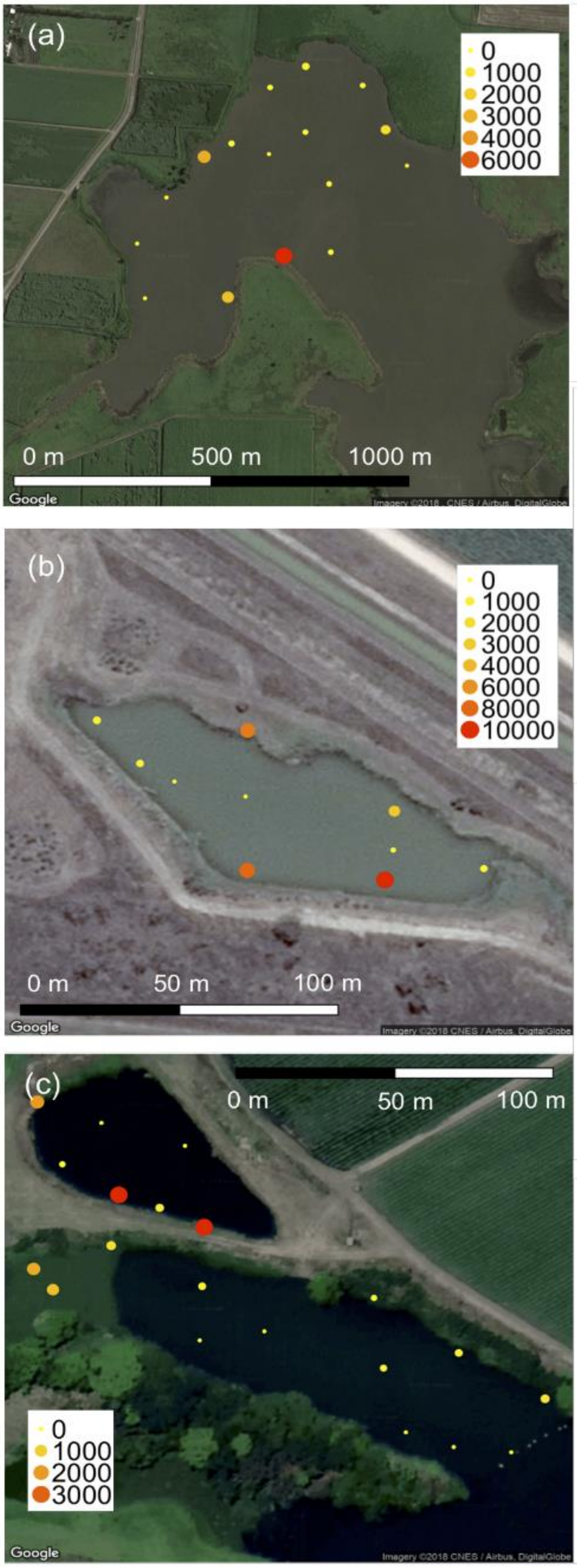


Figure 13: Density distribution of bivalves (individuals/m^2) at: a) Horseshoe lagoon, b) Constructed wetland, and c) Lilliesmere recycle pit (top) and Lilliesmere River (bottom)

3.3.3 Length-mass relationship

The slope of the relationships between shell length (mm) and tissue dry weight (g) did not differ between artificial and natural wetlands (t = 0.74, p = 0.45), and so the bivalve length-mass relationship for all wetlands was best described without an interaction term as follows:

$$\log_{10}DW = 2.9(\pm 0.17) \times \log_{10}L - 4.9(\pm 0.2) \qquad (5)$$

where DW is the tissue dry weight (g), L is shell length (anterior-to-posterior in mm), and coefficients are presented with standard errors ($R^2 = 0.77$, $p < 0.001$, n = 86). The average tissue dry weight predicted from Equation 5 using measured shell lengths differed between artificial wetlands (0.024 mm ± 0.002 g) and natural wetlands (0.016 mm ± 0.001 g) (t = 3.84, df = 151.26, p = <0.001).

3.3.4 Clearance rates

Individual bivalve clearance rates ranged from 0.002 to 1.17 L/bivalve/hr and did not show a significant relationship with tissue dry weight (g) in field experiments ($R^2 = 0.06$, p = 0.57, n = 33; Figure 14) or in laboratory experiments ($R^2 = 0.77$, p = 0.06, n = 11). Given there was no allometric relationship between bivalve mass and clearance rate, average clearance rates were calculated directly without accounting for body size for both field and laboratory experiments. Average clearance rate was higher in artificial wetlands (0.34 ± 0.07 L/bivalve/hour) than in natural wetlands (0.19 ± 0.04 L/bivalve/hour; Figure 14), but this difference was not statistically significant (*t* = 1.72, df = 29.61, p = 0.09). The average clearance rate was higher in the 23°C temperature treatment (0.37 ± 0.07 L/bivalve/hour) compared to the 27°C and 32°C temperature treatments (0.13 ± 0.03 and 0.12 ± 0.03 L/bivalve/hour, respectively; *F* = 7.64; df = 2, 12; p = 0.007).

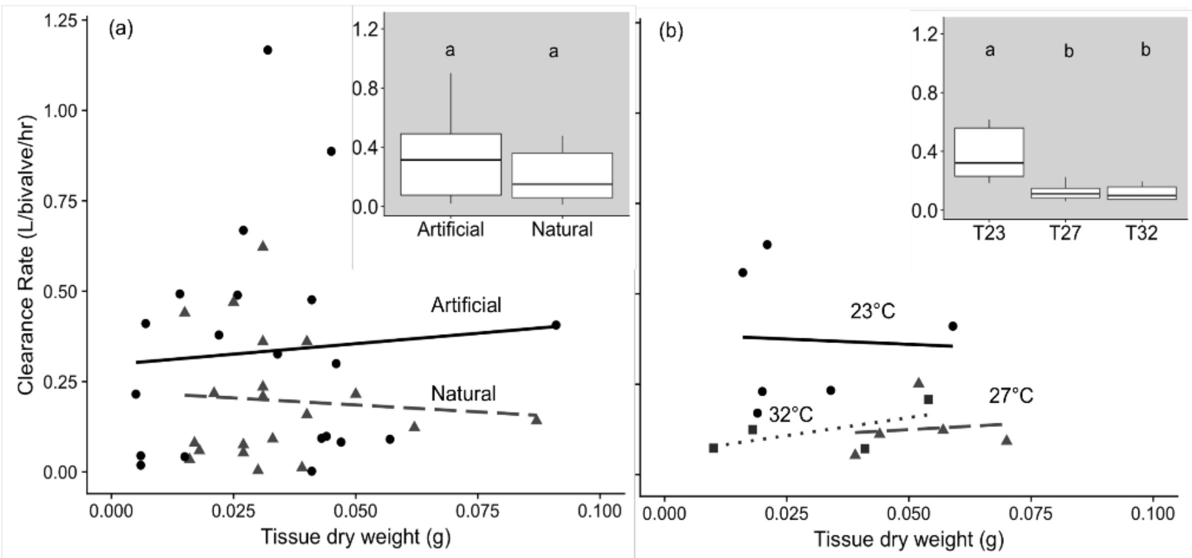


Figure 14: Individual bivalve clearance rates (L/bivalve/hr) measured in the field (a) and under controlled temperatures in the laboratory (b). Letters (a or b) above box plots indicate whether average rates differ significantly (p < 0.05) among groups.

3.3.5 Biodeposition rates

Phosphorus biodeposition rates (BR^P) ranged from 0.3 to 4.26 μg P/bivalve/hour and increased with tissue dry weight (g) in field experiments (Table 5). The slopes of BR^P for artificial and natural wetlands did not differ (t = -1.9, p = 0.065), and therefore the overall BR^P for natural and artificial wetlands was best described without an interaction term by the following model:

$$\log_{10} BR^P = 0.967(\pm 0.23) + 0.42(\pm 0.13) \times \log_{10} DM \quad (6)$$

(R^2 = 0.19, p = 0.003, n = 40; Figure 15). Using the allometric relationship for BR^P (Equation 6) and the size distribution of bivalves in artificial and natural wetlands (Figure 11), we calculated that BR^P were on average higher (0.87 ± 0.03 μg P/bivalve/hour) in artificial wetlands compared to natural wetlands (0.77 ± 0.02 μg P/bivalve/hour, Figure 15) (t = 2.86, df = 166.93, p = 0.004).

Phosphorus biodeposition rates (BR^P) increased with tissue dry weight (g) in 23°C and 27°C temperature treatments but remained relatively constant in the 32°C treatment (Table 5). There was a difference in slope among the temperature treatments, confirmed by a significant interaction term in the linear model (t = -3.7, p = 0.003). To define the allometric relationship for the 23°C and 27°C temperature treatments, which had similar slopes (t = -0.4, p = 0.679), the model was refit without the 32°C treatment data. The overall BR^P at 23°C and 27°C was best described without an interaction term by the following model:

$$\log_{10} BR^P = 1.8(\pm 0.49) + 1.2(\pm 0.31) \times \log_{10} DM \quad (7)$$

(R^2 = 0.56, p = 0.002, n = 13; Figure 15). Using the allometric relationship for BR^P at the 23°C and 27°C temperature treatments (Equation 7) and the size distribution of bivalves across all study wetlands, we calculated that bivalves would deposit on average 0.83 ± 0.02 μg P/bivalve/hour within a temperature range of 23-27°C (Figure 15). The average BR^P in the 32°C treatment was calculated without accounting for bivalve size as 1.16 ± 0.1 μg P/bivalve/hour (Table 2), and was significantly higher relative to the combined average in 23°C and 27°C temperature treatments (t = 3.44, df = 5.48, p = 0.02). Total phosphorus (TP) deposited by individual bivalves was strongly correlated with dissolved phosphorus excretion by the same individuals (r = 0.99, t = 36.95, df = 17, p = <0.001).

Nitrogen biodeposition rates (BR^N) ranged from 0.03 to 18.7 μg N/bivalve/hour and increased with tissue dry weight (g) in field experiments (Figure 15). There was no significant interaction between the slopes of BR^N for artificial and natural wetlands (t = -0.1, p = 0.91). Therefore, the overall BR^N for natural and artificial wetlands was best described without an interaction term by the following model:

$$\log_{10} BR^N = 1.67(\pm 0.24) + 0.68(\pm 0.14) \times \log_{10} DM \quad (8)$$

(R^2 = 0.37, p < 0.001, n = 39). Using the allometric relationship for BR^N (Equation 8) and the size distribution of bivalves in artificial and natural wetlands (Figure 11), we calculated that the average BR^N was higher (3.35 ± 0.19 μg N/bivalve/hour) in artificial wetlands compared to natural wetlands (2.62 ± 0.11 μg N/bivalve/hour) (t = 3.37, df = 160.42, p = <0.001).

Temperature controlled BR^N (μg N/bivalve/hour) did not show a significant relationship with tissue dry weight (g) in laboratory experiments (R^2 = 0.41, p = 0.18, n = 18; Figure 15). Therefore, average BR^N were calculated without accounting for body size as: 2.9 ± 0.5 μg N/bivalve/hour in the 23°C treatment, 4.6 ± 1.3 μg N/bivalve/hour in the 27°C treatment, and 5.5 ± 1 μg N/bivalve/hour in the 32°C treatment. Average BR^N were not significantly different among temperature treatments (F = 1.662; df = 2, 17; p = 0.219). Total nitrogen (TN) deposited by individual bivalves was strongly correlated with dissolved nitrogen excretion by the same individual (r = 0.91, t = 9.85, df = 19, p = <0.001).

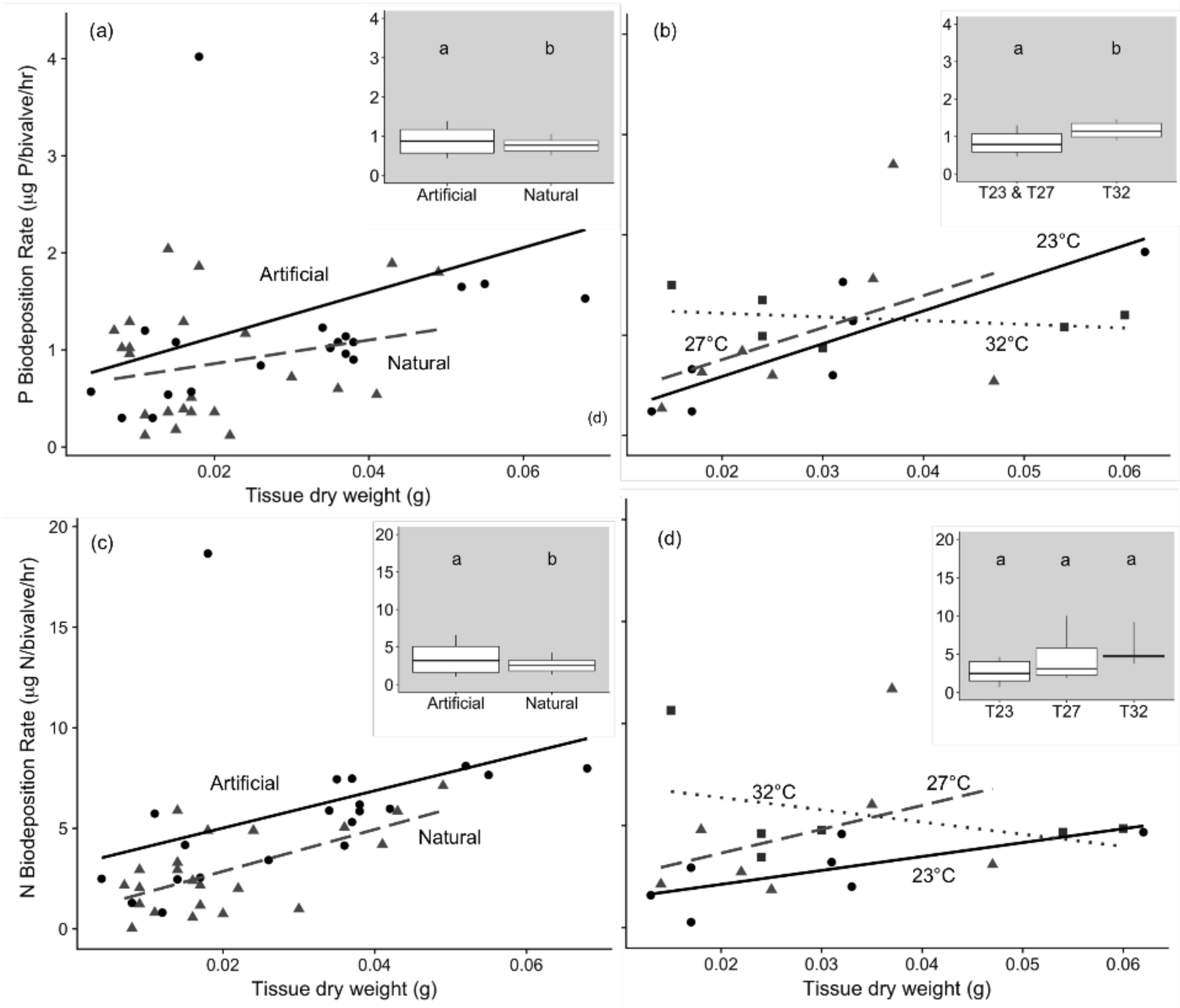


Figure 15: Bivalve biodeposition rates (μg/bivalve/hr) measured in the field and under controlled temperatures in the laboratory for phosphorus (P; a, b) and nitrogen (N; c, d). Letters (a or b) above boxes indicate whether average rates differ significantly (p < 0.05) among groups.

3.4 Discussion and conclusions

Native freshwater bivalves (*Corbiucla australis*) provide ecological processes that could help restore coastal wetland water quality in GBR catchments. Biodeposition, a component of bivalve nutrient cycling, was highly dependent on wetland context (i.e., artificial vs. natural). Their capacity to deposit nutrients in sediment was elevated in artificial wetlands due to increased biodeposition rates, larger individuals and denser populations. Our results also show that the capacity of *C. australis* to deliver the ecosystem service of water filtration is likely to change seasonally, as their filtration rate was higher in temperatures that only occur during cooler winter months. Although filtration rates did not differ between artificial and natural wetlands or increase with bivalve size, greater population densities meant that extrapolated filtration rates were highest in artificial wetlands. However, to fully understand the role of *C. australis* in providing a net water quality benefit to the GBR, more research is needed to quantify wetland nutrient budgets, determine the influence of water flow, and understand the fate of biodeposits (e.g., translocated via water flow or concentrated in sediment with potential to contribute to nitrification-denitrification processes).

Bivalve biofiltration rates did not increase with bivalve size or differ between natural and artificial wetlands. However, filtration rates were higher in cool water temperatures (23°C) that only occurred during winter months. These results are counter-intuitive; it has been well established that filtration rates generally increase with bivalve size and with higher temperatures (until the acute effects thermal stress threshold is reached; Spooner and Vaughn 2008). The temperature data from our study wetland shows that bivalves frequently experience temperatures between 21 and 29°C, depending on season, and regardless of context (natural or artificial). Therefore, it is possible that 23°C is within the optimal range for filtering by *C. australis*, and that 32°C is likely to be approaching their thermal effects limit. However, testing a wider range of temperatures at more frequent intervals is needed to more clearly delineate the thermal effects thresholds of *C. australis* and, therefore, maximise water quality benefits, which is a priority in this agricultural landscape located adjacent to the GBRWHA.

Although we did not find a difference in bivalve filtration rate between wetland types (i.e., natural vs. artificial), greater bivalve population densities in artificial wetlands conferred a higher extrapolated filtration rate (821.9 L/hr/m^2) compared to natural wetlands (134.9 L/hr/m^2). This result may be magnified by increased water residency in artificial wetlands that could allow bivalves to filter greater amounts of water, when compared to natural (flowing) sites examined here. By virtue of their construction, artificial wetlands receive less water flow than most natural wetlands in this region (Davis et al. 2014), and previous research has shown that high water discharge can reduce filtration rates to a fraction of their typical values (Vaughn 2010; Pigneur et al. 2014). Likewise, less rain and catchment water flow during winter months in natural wetlands may compound the higher filtration rates found in cool winter water temperatures, contributing further to seasonal differences in bivalve filtration rates.

Biodeposition

Corbicula australis deposited higher amounts of phosphorus and nitrogen in artificial wetlands relative to natural wetlands. This was partly attributed to the greater frequency of larger individuals in artificial wetlands, as size was positively associated with phosphorus and nitrogen biodeposition rates, but also because population density was higher in artificial wetlands. It is possible that elevated nutrient levels in artificial wetlands increase food availability, causing individual bivalves to reach larger sizes and populations to achieve greater densities. However, previous research suggests that high nutrient levels can have both positive and negative effects on bivalve functioning. Up to a certain threshold, high nutrient levels are likely to improve food, host, and habitat availability for bivalves, leading to increases in their growth, fecundity, and population size; while extreme eutrophication is likely to decrease food quality and reduce dissolved oxygen levels to fatal limits (Strayer 2014) – this is also a known as a stoichiometric knife-edge (Elser et al. 2000; Hessen et al. 2013).

The nitrogen biodeposition rates of *C. australis* did not change under different water temperature treatments, however phosphorus biodeposition rate was highest at 32°C. *C. australis* is rarely exposed to this temperature – only during summer months near the surface/edge of the wetland (1-6% of the time). Therefore, our data suggest that it is unlikely for temperature changes associated with seasonality to influence biodeposition by *C. australis*, although it does imply that climate change could be an important consideration in the future. It is also interesting to note that both nitrogen and phosphorus biodeposition rates were de-coupled from their allometric relationships when exposed to 32°C water temperatures. This may indicate that *C. australis* experiences physiological stress at 32°C, leading to increased dissolved oxygen consumption and higher nutrient excretion rates (Spooner and Vaughn 2008). However, as previously recommended for biofiltration, biodeposition rates should be further tested under a wider temperature range at more frequent intervals (and possibly under different dissolved oxygen rates in laboratory experiments).

Although it was not a focal part of this study, we have limited data to show that biodeposition provides a good proxy for the excretion of dissolved nutrients by *C. australis*. Therefore, it is likely that these bivalves are playing a role in making nutrients more easily bioavailable to support higher productivity in coastal wetland food webs. This can provide a secondary ecosystem service benefit of higher ecosystem productivity and biodiversity to the broader region, potentially linking into the terrestrial ecosystems surrounding freshwater wetlands (Vaughn 2018). Bottom-up food web processes driven by freshwater bivalve nutrient excretion have been well documented, with up to 74% of nutrients channelled through aquatic food webs passing through bivalves (Atkinson et al. 2014), increasing production across all trophic levels (Atkinson and Vaughn 2015), and reducing nitrogen limitation (Atkinson et al. 2014). We would expect *C. australis* excretion rates to follow patterns similar to biodeposition rates, with higher excretion of dissolved nutrients in artificial wetlands relative to natural wetlands. However, increasing the availability of nutrients in wetlands may not be an important or valuable service in catchments of the GBR. For example, nitrogen excretion by freshwater bivalves has been found to increase in streams near agricultural land use, but because nitrogen concentrations were already elevated due to agricultural run-off, any additional nitrogen provisioning by bivalves was not important for ecosystem functioning (Spooner et al. 2013). Further research is necessary to understand the nitrogen cycle and processing across this floodplain.

The importance of environmental context

Overall, our findings underscore the importance of wetland context for ecosystem service provisioning by freshwater bivalves. Artificial and natural wetlands differed in the size range and density of their bivalve populations, and this either contributed to or compounded differences in filtration and biodeposition capacity. Seasonal context (i.e., temperature exposure) was also found to influence the rate of filtration performed by *C. australis.* However, to establish a more comprehensive understanding, further research is needed to directly investigate other factors that might underpin wetland and seasonal context dependence such as: food particle size and availability, dissolved oxygen levels, water flow, and seasonal effects of life history. For example, corbiculids are known to demonstrate higher filtration rates during spawning (Viergutz et al. 2012), release nutrients during mass mortality events (Cherry et al. 2005), and population sizes can be highly variable depending on season and location (Sousa et al. 2012; Zając et al. 2016). Furthermore, sediment composition/chemistry and wetland hydrology will determine whether biodeposits accumulate in sediment and contribute to nitrification and/or denitrification processes or, alternatively, are translocated to connected coastal wetland ecosystems (Vaughn 2018). Although the exact mechanisms are unclear, corbiculid biodeposition has been found to increase sediment denitrification, especially in high nutrient waters (Turek and Hoellein 2015; Welsh et al. 2015). This could be a particularly important process in the context of agricultural nutrient run-off to the GBR.

Net water quality benefit

This study provides a first step towards understanding the potential water quality benefits provided by native freshwater bivalves (*C. australis*) as part of a larger campaign towards improving water quality and resilience of the GBR (Waterhouse et al. 2017). However, further research is needed to affirm their efficacy in providing a net water quality benefit. This will require quantifying the nutrient budget of artificial and natural wetlands and acquiring a better understanding of the life history of *C. australis*. For example, without an estimate of their average life span, it is unknown whether any realised nutrient storage benefits negate nutrient release following the death and decomposition of *C. australis*. The lifespan of *Corbicula fluminea* is known to be shorter (1 - 5 years; Sousa et al. (2008) in comparison to other freshwater bivalve species (generally 10 – 25 years; Haag and Rypel (2011). If the lifespan of *C. australis* is similarly short, their ability to act as a nutrient sink in coastal wetlands may also be limited. However, there is evidence that a large proportion of the nitrogen and phosphorus content of freshwater bivalve biomass is stored in shell tissue rather than soft tissue (Atkinson and Vaughn 2015). Furthermore, corbiculids typically produce shell tissue at a higher rate than soft tissue and, depending on water chemistry and flow rate, their shells can accumulate in aquatic ecosystems and serve as long-term nutrient storage (Strayer and Malcom 2007).

4.0 CASE STUDY THREE: MECHANICAL HARVESTER REMOVAL OF INVASIVE AQUATIC WEEDS TO IMPROVE WATER QUALITY AND FISH HABITAT

4.1 Overview

We examine here a coastal restoration project conducted along Crooked Waterhole (-19.5172º, 147.086º) on the Burdekin floodplain, a heavily weed-choked creek and degraded system on the Burdekin floodplain (Figure 16). This system is representative of creek systems on the floodplain draining to the Bowling Green Bay RAMSAR wetland and into the GBR lagoon. The project trialled the removal of aquatic weed mats, predominantly comprised of Water Hyacinth, from a 7km section of creek to identify potential for this restoration activity, and providing case study data to assist managers in meeting ecosystem restoration and protection targets set in the Reef 2050 Plan.

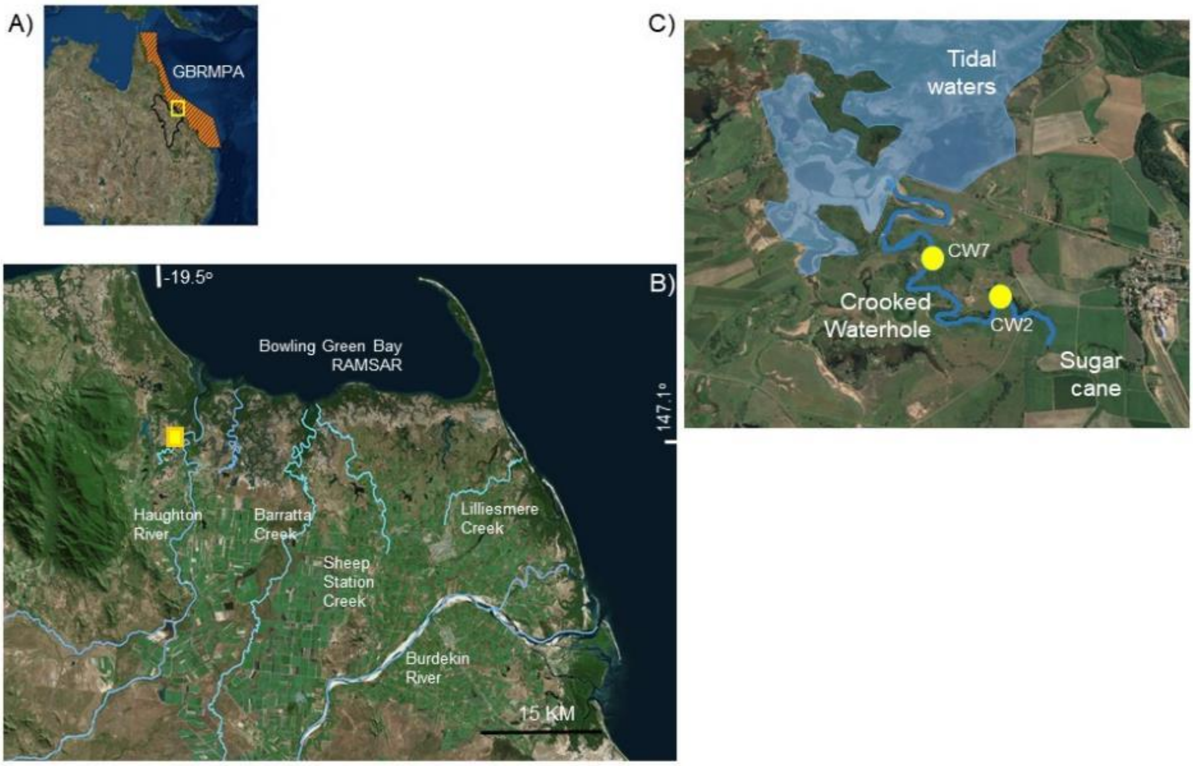


Figure 16: (a) Location map of Burdekin catchment, adjacent to the Great Barrier Reef Marine Park Area (GBRMPA) Australia; (b) location of Crooked Waterhole on the Burdekin floodplain (yellow square); and (c) Crooked Waterhole and sample sites (CW2 and CW7) connected to tidal water areas of the Bowling Green Bay.

Around the 1950s, large areas of tidally influenced coastal drainage depressions were bunded by landholders (i.e., surrounded by earth banks) to enhance grass growth for cattle production (Challen and Long 2004). This allowed the formation of seasonally ponded pastures (Abbott et al. 2020), and protected coastal freshwater resources, including shallow groundwater aquifers, from tidal intrusion. Most of these bunded pastures are still in place today and are perennial due to the near-continual discharge of regulated tailwater (excess water that flows directly from cane paddocks untreated and unfiltered into adjacent watercourses). Water in these bunded pastures is typically high in nutrients and herbicides (Davis et al. 2017) and flows directly from cane farms located in the lower catchment to the floodplain and RAMSAR wetland. Aquatic ecosystem impacts driven by altered hydrology and poor water quality associated with intensive use of the creek's land and water resources include elevated sediment and nutrient loads, loss of dry season zero-flow periods, lack of seasonal water draw down, loss of riparian vegetation, eutrophication and proliferation of submerged, floating and emergent aquatic plants comprising native and exotic weed species. The altered hydrology contributes to weed-chokes and water quality problems: for example, irrigation water flow under mats of Water hyacinth results in dissolved oxygen (DO) depletion (Waltham and Fixler 2017).

Removal of Water Hyacinth chokes elsewhere (see Queensland WetlandInfo) has resulted in improved open water conditions at relatively small scales (Butler et al. 2009). For example, Perna et al. (2012) detailed how poor water quality at Sheep Station Creek in the Burdekin floodplain resulted in few fish species. However, after the removal of Water hyacinth there was an almost immediate improvement in DO, and subsequent recruitment of native fish numbers. A similar treatment was therefore planned for the Crooked Waterhole to further trial the potential benefits that could arise from applying such treatments at a larger scale in the catchment.

In November 2016, after a planning period by project partners comprising of local government, Greening Australia, James Cook University and landholder to set the values and services for this restoration project, a weed choke (consisting primarily of Water hyacinth) was removed along a 7km section of Crooked Waterhole. This was achieved using a floating excavator (amphibious especially designed for shallow aquatic weed removal, Figure 17). As the weed choke covered an area of approx. 5 ha that extended approx. 0.2m underwater, approximately 155 tonne dry weight of biomass was removed from the creek and deposited on the banks. This was left to decompose for later use as mulch for onsite tree planting areas. The removal of aquatic weeds is calculated to have removed approximately 800kg of nitrogen (Reddy et al. 1989) from the waterway, a preferable option to poisoning the weed and leaving it to sink and rot in situ, further adding more nutrient to the aquatic ecosystem. Following weed excavation, spot spraying approximately every 6 months (using Round-up Bi-active® at a rate of 1mL per 1L) was used to maintain this waterway open and weed-free.

This study examined the water quality and fish assemblages directly before and subsequently after weed removal at Crooked Waterhole.

Figure 17: Civil Plus Construction's amphibious excavator with Greening Australia staff (left), property owner (centre), and excavator operator (right). This floating excavator completed the works in just seven days, progressively moving along the creek, picking up the floating invasive weeds and relocating to the banks for drying and later used for mulching on nearby tree planting (Photo source: Greening Australia).

4.2 Methods

The water quality and fish community in this creek system was monitored before weed initial maintenance works (November 2016) and every 2-3 months thereafter to November 2018. A calibrated multi-probe data logger (Hydrolab, OTT Hydromet, USA) was deployed in the near-surface water layer (0.2 m) to monitor diel cycling of temperature, pH, electrical conductivity and DO (%) measured at 20 min intervals at two sites within the upstream and downstream ends of the managed area (CW 2 and CW 7). The Hydrolab was deployed over a few days during fish surveys, both before the restoration works, but then also to measure water conditions following works and during follow up spray campaigns.

Fish sampling was completed using a Smith-Root 2.5 GPP generator boat-mounted electrofishing unit (voltage changed depending on water salinity and depth), where sampling involved use of the single pass technique (five minute shots following similar studies elsewhere on the floodplain, Waltham and Fixler, 2017), and cast nets (8 ft drop with 10mm the occurring at CW7). All field work was completed in the same two study locations as the water sampling using the same field team and gear. The maximum water depth was 2.2m, although the creek become progressively shallower as the dry season progressed each year. All fish were identified and released alive, except declared non-native fish species that were euthanased following Queensland Fisheries Permit 18712 and animal ethics permit A2178. Cast nets were only used before weed removal when the waterway was choked and boat use was not possible. Following weed removal, the electrofishing boat was used on a regular basis to assist in characterising the fish community over time. The exception was December 2016 and October 2018 at the downstream site when the water was too salty to effectively electrofish and the cast net was used again. Use of cast nets to sample fish on floodplains is well established in the region (Sheaves and Johnston 2009). No statistical test of the data is possible given two different sampling methods were used (electrofishing and cast nets).

4.3 Results and discussion

4.3.1 Water quality

Prior to restoration works DO levels were critically low in Crooked Waterhole, as would be expected with such high levels of aquatic weed infestation (Table 6). This contrasts with a previous survey of Crooked Waterhole before weed infestation where water quality was more suitable to support fish (Burrows and Perna 2005). Following the weed removal in 2016 and subsequent maintenance spraying, DO improved from being persistently zero % to cycling between 40 and 100% (Table 6). This was likely to be due to a combination of wind driven re-aeration, in addition to an overall reduced organic oxygen consuming load on the creek. Previous studies of aquatic weed removal corroborate the findings here (Perna and Burrows 2005; Sharma et al. 2016; Yan et al. 2016), reinforcing the effectiveness of aquatic weed removal for improving coastal wetland water quality conditions for fish.

4.3.2 Fish community

Prior to weed removal, only three fish species were detected (one native species, Snakehead *Giurus margaritacea*, and two invasive species, Three Spot Gourami, *Trichopodus trichopterus*, and Mozambique tilapia, *Oreochromis mossambicus*) by electrofishing, and these were species deemed most tolerant to low oxygen. In the first year after weed removal, a total of 15 fish species was recorded (Table 7), with the most abundant native species in the upstream reaches being Empire Gudgeon (Hypseleotris *compressa*), Tarpon (*Megalops cyprinoides*), and Glassfish (*Ambassis agrammus*). However, the most abundant species was the invasive Mozambique Tilapia, a species widespread across the Burdekin floodplain (Davis & Moore, 2016). Seven diadromous species (fish that need to migrate between freshwater and marine water as part of their lifecycle ecology) were recorded. The most abundant was Barramundi which is known to access saltwater during wet season flood connection and will then undertake a return migration back to freshwater areas during subsequent flows (Staunton-Smith et al. 2004; Crook et al. 2020).

Interestingly, the fish assemblage recorded after works were similar to that recorded before the excessive build-up of aquatic weeds in this creek, although still only a subset of all the species known to occur on the Burdekin floodplain. The fish recorded represent broad feeding guilds, including insectivores (Eastern rainbow fish, *Melanotaenia splendida inornata*), herbivores (Bony bream, *Nematolosa erebi*), and predators (Barramundi).

A)

B)

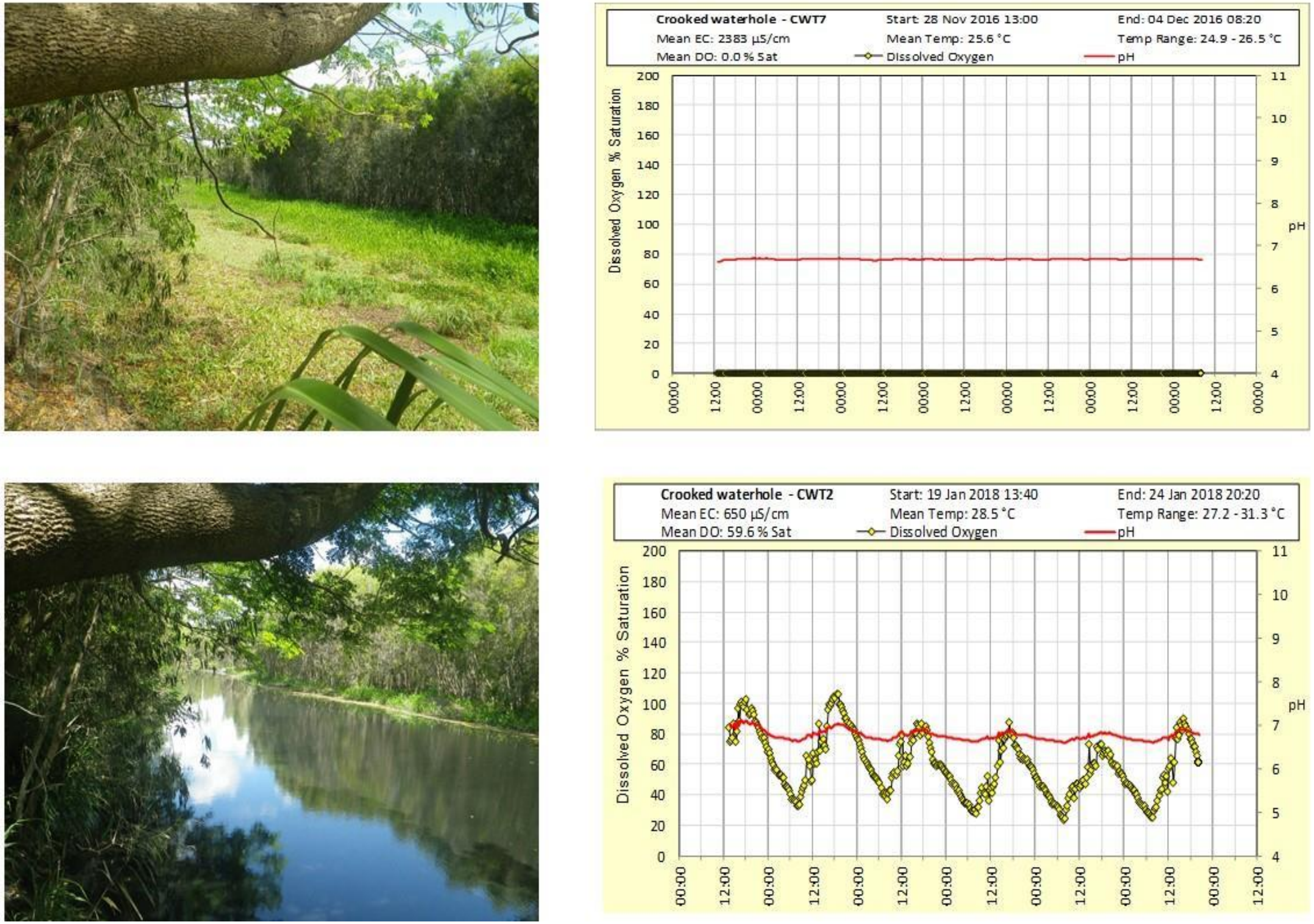


Figure 18: A) Photograph of creek before major aquatic plant removal and corresponding Hydrolab time series plot; and B) Photograph of creek in January 2018 during wet season and with ongoing follow up maintenance works and corresponding Hydrolab time series plot. Plots show summary statistics during each logging period (about 1 week in duration) for dissolved oxygen, and pH.

Table 6: Summary water quality conditions at the two sites in the Crooked Waterhole restoration area. Data are taken from logged (20min) time series before restoration works (Nov 2016) until Nov 2018.

Site	Start Time	Finish Time	Min Temp (°C)	Max Temp (°C)	Mean Temp (°C)	Min EC (µS/cm)	Max EC (µS/cm)	Mean EC (µS/cm)	Min pH	Max pH	Mean pH	Min DO (%)	Max DO(%)	Mean DO (%)
CWT2	11/11/2016	14/11/2016	24.86	26.47	25.65	953.00	978.00	967.81	6.42	6.56	6.53	0.00	0.00	0.00
CWT2	14/12/2016	16/12/2016	26.24	30.32	27.63	1011.00	1036.00	1021.42	6.63	6.88	6.76	2.40	85.50	31.43
CWT2	24/01/2017	7/02/2017	28.18	36.98	31.28	736.00	817.00	781.77	6.71	7.13	6.82	0.00	83.00	3.17
CWT2	3/03/2017	20/03/2017	26.15	29.36	27.48	369.00	674.00	649.15	6.65	6.88	6.76	0.00	3.98	1.44
CWT2	27/10/2017	5/11/2017	22.34	32.00	28.03	40.00	675.00	591.87	6.37	7.20	6.91	2.50	66.10	30.48
CWT2	19/01/2018	24/01/2018	27.24	31.30	28.51	311.00	1061.00	650.30	6.61	7.11	6.77	23.00	105.60	59.60
CWT2	23/02/2018	25/02/2018	22.48	27.82	24.75	255.00	2915.00	454.31	6.56	8.12	7.15	0.00	101.60	62.07
CWT2	6/03/2018	16/03/2018	23.69	32.09	28.56	159.00	279.00	223.65	6.48	7.52	6.70	0.00	159.30	31.67
CWT2	13/08/2018	26/08/2018	18.14	22.85	20.10	876.00	958.00	914.94	7.77	8.12	7.92	36.50	84.80	59.16
CWT2	24/10/2018	2/11/2018	27.69	33.02	30.04	385.00	1217.00	1060.38	7.24	8.22	7.75	21.90	140.50	85.30
CWT7	11/11/2016	14/11/2016	26.44	30.53	28.29	2407.00	2438.00	2417.75	6.91	7.14	7.00	0.00	0.00	0.00
CWT7	28/11/2016	4/12/2016	24.85	26.54	25.61	2337.00	2438.00	2383.44	6.62	6.71	6.68	0.00	0.10	0.00
CWT7	14/12/2016	16/12/2016	27.40	32.39	29.46	2215.00	2327.00	2280.72	7.15	7.64	7.25	2.00	142.20	50.03
CWT7	24/01/2017	7/02/2017	27.56	33.71	29.51	761.00	796.00	779.08	6.37	6.71	6.57	0.00	14.20	0.97
CWT7	3/03/2017	20/03/2017	26.84	29.30	28.03	802.00	920.00	859.60	6.26	6.49	6.37	0.00	45.50	1.16
CWT7	27/10/2017	5/11/2017	26.43	31.58	29.14	2497.00	2904.00	2816.73	7.08	7.55	7.35	2.80	98.40	59.73
CWT7	19/01/2018	24/01/2018	28.16	33.75	29.76	2438.00	3616.00	3521.44	8.19	8.81	8.51	26.90	191.90	100.32
CWT7	6/03/2018	8/03/2018	28.52	33.05	30.47	164.00	194.00	179.10	6.51	6.61	6.56	0.00	27.80	3.76
CWT7	14/08/2018	20/08/2018	19.47	24.61	21.61	3037.00	4253.00	4190.61	6.96	7.31	7.13	76.60	115.60	97.14
CWT7	24/10/2018	2/11/2018	27.40	31.45	29.15	2645.00	5689.00	5504.32	7.98	8.56	8.24	0.00	235.90	73.37

Table 7: Summary of fish data collected at Crooked Waterhole. CW2 is the freshwater site located upstream of the rock road crossing, CW7 is below the rock road crossing in the upper tidal region of the creek. The 2007 data are from Burrows & Perna (2007) which were recorded as presence/absent only. Fish catch during this study are total catch over the duration of sampling, shown as power on (mins). Fish catch for cast nets were from 10 throws, while visual observation was over 20mins.

	Wetland	CW 2	CW 2	CW 2	CW 2	CW 2	CW 2	CW 2	CW 7	CW 7	CW 7	CW 7	CW 7
	Date	**24/01/2007**	**4/04/2007**	**14/12/2016**	**17/05/2017**	**17/08/2017**	**27/04/2018**	**24/10/2018**	**14/12/2016**	**17/05/2017**	**17/08/2017**	**27/04/2018**	**24/10/2018**
	Method	**Efish**	**Efish**	**Cast**	**Efish**	**Efish**	**Efish**	**Efish**	**Cast**	**Efish**	**Efish**	**Efish**	**Visual**
	Power on (mins)				**35**	**35**	**50**	**35**		**40**	**35**	**35**	
Species	**Common name**												
Anguilla reinhardtii	Marbled eel	1	1		1	2	3	10		1		3	
Nematolosa erebi	Bony bream	1	1		3					2			
Craterocephalus stercusmuscarum	Fly-specked hardyhead	1	1										
Melanotaenia splendida	Eastern rainbow fish	1	1			20	12						
Lates calcarifer	Barramundi		1			2		9			13	2	3
Glossamia aprion	Mouth almighty	1	1										
Giurus margaritacea	Snakehead gudgeon	1	1	4			3	25	1		2		
Hypseleotris compressa	Empire gudgeon		1	3		33	5	15			40	5	
Hypseleotris sp. 1	Midgley's carp gudgeon	1	1					3					
Porochilus rendahli	Rendahl's catfish					1							
Ambassis agrammus	Glass fish	1	1			5	42	1			3		
Toxotes chatareus	Archer fish	1	1										
Oxyeleotris lineolatus	Sleepy cod	1	1					2					
Megalops cyprinoides	Tarpon		1		5	11	6	16		5	35	3	20
Amniataba percoides	Bared grunter	1	1										
Leiopotherapon unicolor	Spangled perch	1	1										
Neoaris graeffei	Lesser salmon catfish		1										
Chanos chanos	Milkfish							1					
Selenotoca multifasciata	Banded scat										10		
*Gambusia holbrooki**	Gambusia	1	1				5	3					
*Trichogaster trichopterus**	Three spot gourami			10	20	2	10	11	3				
*Oreochromis mossambicus**	Tilapia			15	25	10	25	30	20	1	55	20	10
	Species Richness	13	17	4	5	9	9	12	3	4	7	5	3

5.0 CASE STUDY FOUR: LONG TERM REMOVAL OF AQUATIC INVASIVE WEEDS

5.1 Overview

Sheep Station Creek is a system flowing through the Burdekin floodplain (Figure 19). Fish and water quality in this creek have not been studied since a pilot project in 2001/2002, which found that irrigation water flowing under weed mats composed mostly of water hyacinth suffered major depletion of dissolved oxygen (Perna et al. 2012). Poor water quality affected fish populations, and the removal of weed biomass resulted in an almost immediate improvement in levels of dissolved oxygen in the water and a gradual recruitment and expansion of native fish populations from adjacent remnant habitat (Perna 2003). Studies on weed removal at other wetland areas on this floodplain (Waltham and Fixler 2017) and other areas of coastal north Queensland (Veitch et al. 2007) have produced similar results. Wherever floating weeds such as water hyacinth have been removed on the floodplain, water quality improves and fish assemblages begin to recover (Perna and Burrows 2005; Veitch et al. 2008).

Motivated by this outcome, Burdekin Shire Council (local government authority), the Lower Burdekin Water Board, riparian landholders (farmers), and NQ Dry Tropics Natural Resource Management (NRM) formed an alliance in 2001 to share the costs (approximately $20,000 AUD each year) to implement an aquatic weed removal program along Sheep Station Creek (mechanical and aerial/vessel-based spraying) in an effort to maintain water delivery for irrigation and to protect fish habitat values. Despite observed improvements, including in water quality and fish populations, during the initial period of the restoration program, no follow-up evaluation has been completed. We repeat the field surveys of Perna et al., (2012) to examine whether the ongoing maintenance still achieve the goals set in 2001, which focused primarily on maintaining high quality water and fish habitat despite being located on a highly disturbed floodplain. The results of this survey support a more holistic plan to develop cooperative partnerships on other coastal floodplains in an attempt to expand restoration efforts and success as we advance towards the Reef 2050 wetland conservation and enhancement targets.

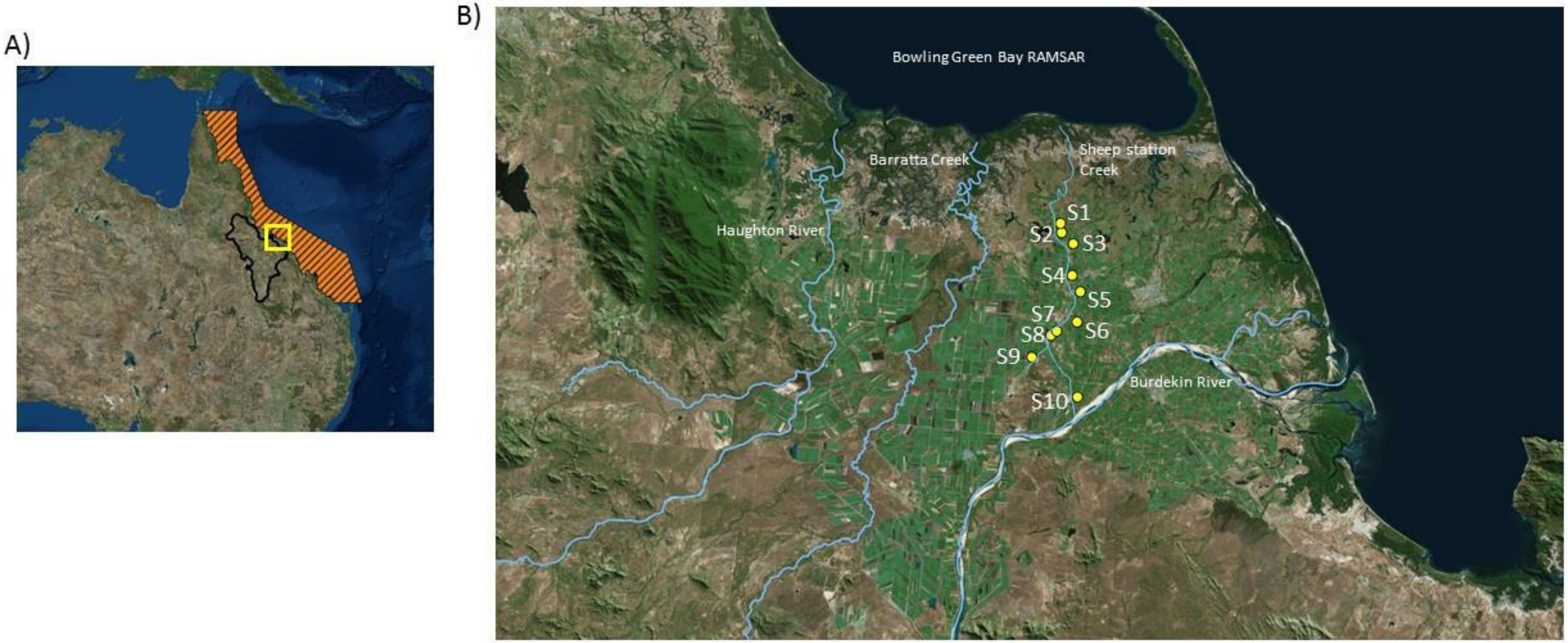


Figure 19: A) Location of the Burdekin drainage basin next to the Great Barrier Reef in northern Queensland B) Site map of Sheep Station Creek on the Burdekin floodplain. For site codes see (Waltham et al. 2020b).

5.2 Methods

5.2.1 Fish sampling

The fish assemblages in lagoons along Sheep Station Creek that were previously examined in 2001/2002 – though data were published later as (Perna et al. 2012); was resurveyed in the same lagoons, except in the current study that included several additional lagoons including Landos (S3), Churches (S4) and Aherns (S10), at approximately the same time of year, September 2017 (late dry season) and May 2018 (post-wet season) (Figure 19). Unlike the 2001/02 study, which combined boat electrofishing and gill nets at selected sites, fish sampling during this study utilized a Smith-Root 2.5 GPP generator boat-mounted electrofishing unit (the same vessel as the earlier study). Sampling involved the use of a single-pass technique following a standardized protocol (five minute shots, depending on the size of the water body – between 15 to 30 shots), with effort standardized to the number of individuals caught per minute with effort covering open water and vegetated littoral areas to provide an overall inventory of fish in lagoons. All fish were identified and released alive, except non-native species that were euthanized following Australian law, under JCU permit number 187102.

5.2.2 Water quality

To characterise physiochemical conditions in the lagoons, a calibrated Hydrolab multiprobe data logger (compared to the 2001 (Perna et al., 2012) study which used spot measures thereby no comparison with the data is possible) was deployed in the near-surface water layer (0.2 m below the surface) for between 5 and 7 days during the late dry season (September 2017) to quantify the diel periodicity of water temperature, dissolved oxygen, conductivity, and pH recorded at 20 min intervals. (Hydrolabs were not deployed during the post-wet season survey because that is the period when conditions are more stable). Water clarity was measured with a Secchi disc, with the Secchi depth/total depth ratio (Zu %) calculated. Lagoons in which the ratio was greater than 20% were designated clear lagoons, and lagoons in which the ratio was less than 20% were designated turbid lagoons (Butler and Burrows 2007).

5.2.3 Data analysis

Nonmetric multidimensional scaling (nMDS) was used to evaluate differences in the species composition of fish catches during the more recent sampling campaign (statistical comparison with the 2001 data is not possible given that results are based on two sampling methods, as opposed to the single method used in the recent surveys). The nMDS ordination ($k = 2$) was performed on presence-absence data for comparison using a Jaccard dissimilarity matrix. A permutational multivariate analysis of variance (PERMANOVA) and subsequent pairwise comparisons were performed to evaluate whether differences in species composition were statistically significant (between late dry and post wet season as fixed factors, while sites were random factors). Similarity percentage analysis (SIMPER) identified which fish species contributed to at least 70% of the differences between groups.

5.3 Results and discussion

5.3.1 Water quality

A summary of the mutliparameter probe data is shown in Table 8. In turbid and clear lagoons (Table 8), the water temperature had a similar rhythmic diel cycle, ranging between 24°C and 25°C (early hours of the morning) and 27°C (mid- to late afternoon before sunset) each day. The highest overall water temperature was recorded in turbid (29.7°C) rather than clear (27.8°C) lagoons, while the daily lowest temperatures were recorded in clear (24.2°C) rather than turbid (25.3°C) lagoons. Clear lagoons also had a higher hourly amplitude (0.64°C/hr) than turbid lagoons (0.52°C/hr).

Dissolved oxygen (DO% saturation) has distinct diel periodicity (Figure 20) with concentrations as low as 7% saturation in the morning reaching concentrations of 210% coincident with the mid- to late afternoon. The highest and lowest DO% values were recorded in clear lagoons (Figure 18). Clear lagoons had a higher hourly amplitude (82.8% saturation) than turbid lagoons (24.8% saturation).

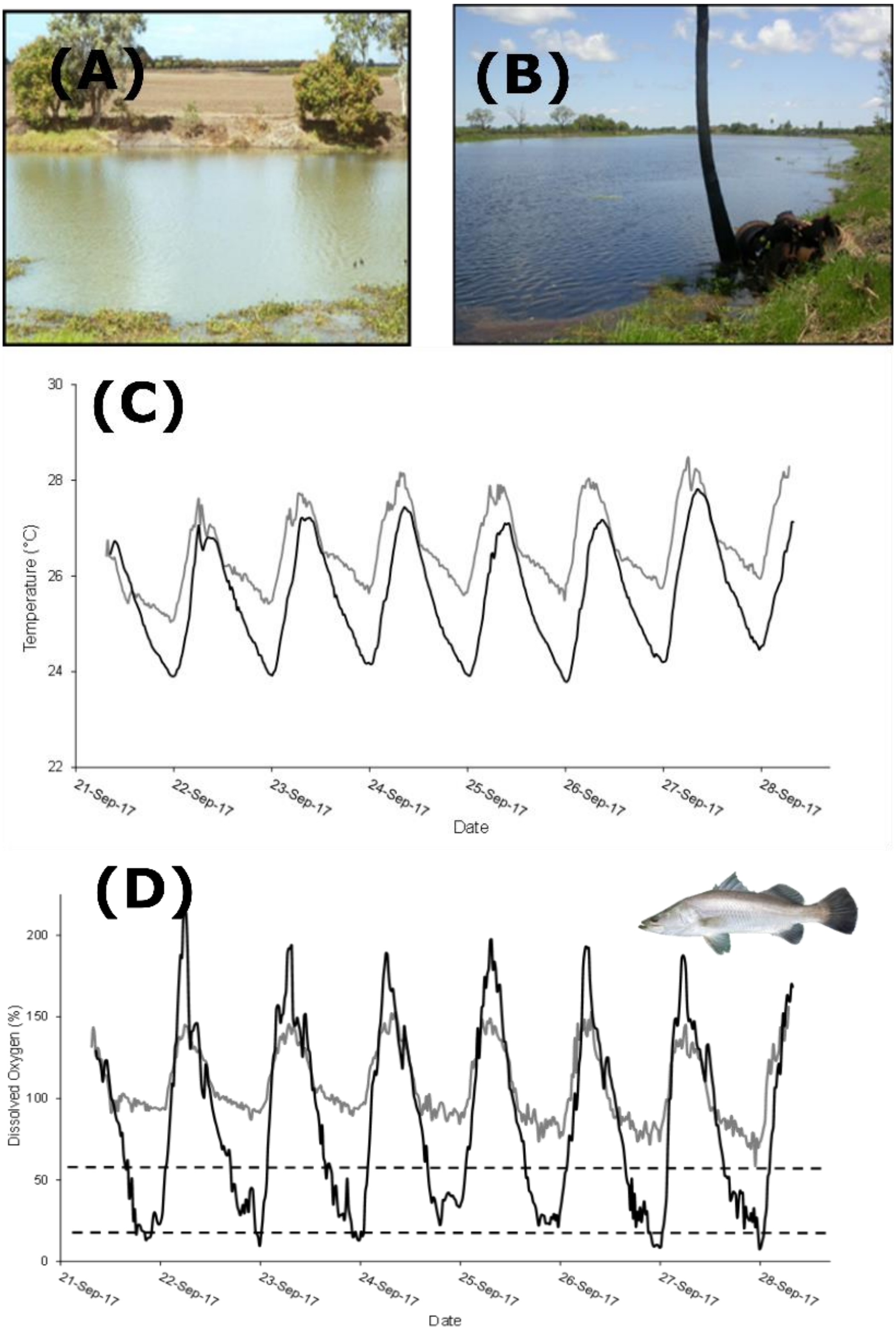


Figure 20: Example (May 2018) of (A) a turbid lagoon (Churches lagoon); and (B) a clear water lagoon (Castenelli lagoon). Time series plot of (C) water temperature (°C), and (D) dissolved oxygen (% saturation) recorded using the Hydrolab (logging 20 min intervals) at sites. Light grey lines for turbid site and black lines for clear site. Dashed lines of dissolved oxygen (% saturation) limits for barramundi (*Lates calcarifer*) are the chronic trigger value (62.5%) and acute trigger value (16%) from Butler and Burrows (2007).

Table 8: Summary of water quality parameters measured in ten lagoons in Sheep Station Creek during September 2017, including water clarity based on euphotic depth ratio (clear > 20%, turbid < 20%)

				Water temp (oC)			Sp Cond (µS/cm)			Diss Oxy (%)			pH			Water clarity
Lagoon	**Code**	**Lat**	**Long**	**Median**	**Max**	**Min**	**Median**	**Max**	**Min**	**Median**	**Max**	**Min**	**Median**	**Max**	**Min**	
Pegararo	S1	-19.508685°	147.326625°	27.8	30.8	25.8	241.6	248.0	237.0	92.2	166.7	51.1	7.9	8.8	7.2	Clear
Goriza	S2	-19.517263°	147.327734°	26.6	28.5	25.0	224.3	231.0	220.0	107.8	156.2	58.5	8.1	8.8	7.2	Clear
Landos	S3	-19.527734°	147.338555°	26.1	30.9	23.7	222.0	289.0	198.0	99.0	178.7	16.9	7.4	8.7	6.5	Turbid
Churches	S4	-19.556833°	147.337770°	25.6	27.8	23.8	214.0	222.0	209.0	85.4	215.2	7.6	7.8	9.3	7.0	Turbid
Dick's	S5	-19.572030°	147.345258°	26.9	29.1	24.8	214.5	216.0	212.0	78.5	118.8	34.6	7.6	8.5	7.1	Turbid
Castenelli	S6	-19.599702°	147.341990°	27.1	29.5	25.1	227.7	229.0	217.0	122.4	156.0	98.4	8.4	8.8	8.0	Turbid
Payards	S7	-19.612050°	147.317839°	24.2	30.9	23.2	211.2	286.0	205.0	90.0	169.9	54.3	7.7	9.1	7.2	Turbid
Kelly's	S8	-19.608271°	147.323218°	24.7	30.5	23.6	210.2	335.0	200.0	87.2	133.8	57.0	7.7	8.8	6.9	Turbid
Red Lilly	S9	-19.632546°	147.300098°	24.0	27.7	22.4	276.6	282.0	255.0	57.8	126.2	16.9	6.8	8.1	6.5	Clear
Aherns	S10	-19.669135°	147.342376°	24.3	30.0	23.1	209.4	360.0	205.0	80.4	102.8	68.1	7.4	7.6	7.3	Turbid

5.3.2 Fish community

In total, 2,267 individuals from 24 species were identified during the two seasonal sampling periods, which is slightly more than in the 2001 survey (Table 9). The most abundant species (44% of total catch) was the fly-specked hardyhead (*Craterocephalus stercusmuscarum*) which was found at every site except Payard and Ahern's lagoons. The eastern rainbowfish (*Melanotaenia splendida*) (15%) was the next most abundant (found at all sites), followed by bony bream (*Nematalosa erebi*) (14%) found at all sites except Ahern's lagoon. Interestingly, the invasive tilapia (*Oreochromis mossambicus*) was captured during this study, survey, along with several native fish species including eastern gudgeon (*Hypseleotris compressa*), sleepy cod (*Oxyeleotris lineolatus*), pacific short-finned eel (*Anguilla obscura*) and barred grunter (*Amniataba percoides*) but all were not captured in the 2001.

More fish species were captured during the late dry season survey (24 species), along with a higher overall total number (1,446) compared to the post-wet season survey (19 species, 821 fish) (PERMANOVA, Pseudo-F=2.92, P<0.02). Figure 21 shows those fish species contributing to the seasonal split in the ordination, with *Anguilla reinhardtii* (15.1% SIMPER), *Megalops cyprinoides* (14.7%) and *Hypseleotris compressa* (14.3%) contributing most to the ordination pattern, with more individuals caught during the late dry season. Non-native fish contributed to 15.4% of the total catch pooled for the two surveys, while native fish comprised the remaining 84.6% (Table 9).

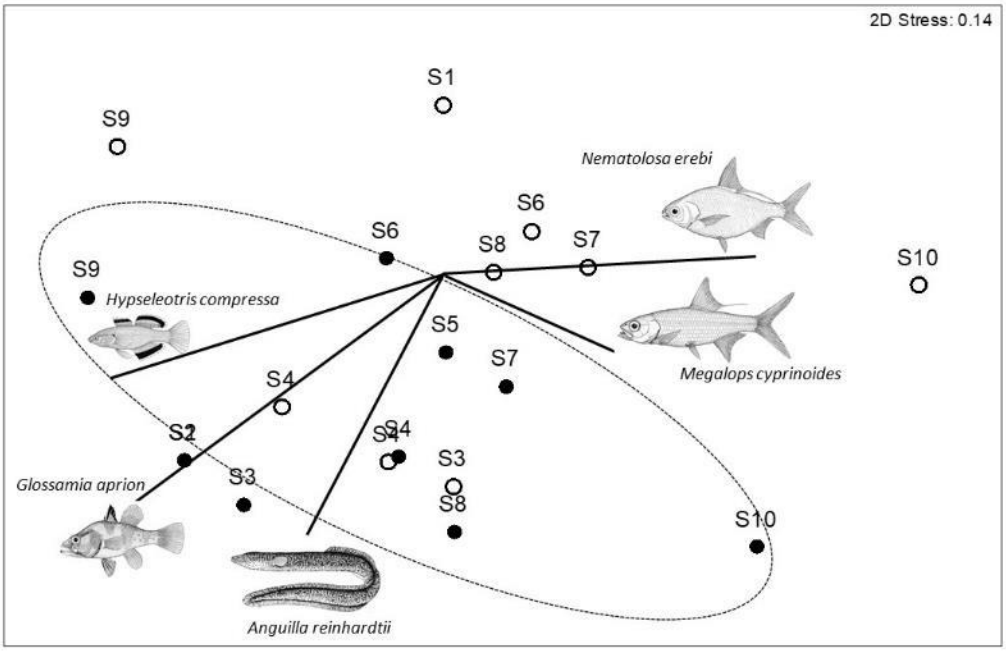


Figure 21: nMDS ordination (stress = 0.14) for September 2017 (closed circles) and May 2018 (open circles), vector line directions are towards fish species contributing to the position of lagoons in the ordination. The sites within the dotted line are the September 2017 survey (late dry season). Fish images taken from Pusey et al. (2004).

Table 9: Comparison of native and non-native fish species occurrence in Sheep Station Creek in 2001 and in the present study, and total floodplain diversity (γ – diversity) (Davis and Moore, 2016). * signifies species recorded in surveys. § signifies diadromous species or species with suspected estuarine affiliation. Non-native species are listed separately below

Family	Species	Common Name	Floodplain γ -diversity	Sheep Station Creek 2001	This study	Percentage of total catch
Native species						
Ambassidae	*Ambassis agrammus*	Sailfin glass perch	*	*	*	0.1
Anguillidae	*Anguilla reinhardtii* §	Speckled longfin eel	*	*	*	3.7
	Anguilla obscura §	Pacific short-finned eel	*		*	0.2
Apogonidae	*Glossamia aprion*	Mouth Almighty	*	*	*	2.6
Ariidae	*Neoarius graeffei*	Lesser salmon catfish	*			0.1
Atherinidae	*Craterocephalus stercusmuscarum*	Fly speckled hardyhead	*	*	*	43.7
Belonidae	*Strongylura krefftii*	Freshwater longtom	*		*	0.1
Centropomidae	*Lates calcarifer §*	Barramundi	*	*	*	2.0
Chanidae	*Chanos chanos §*	Milkfish	*			0.0
Clupeidae	*Nematalosa erebi*	Bony bream	*	*	*	13.9
Eleotridae	*Giuris margaritacea §*	Snakehead gudgeon	*	*	*	0.1
	Hypseleotris compressa §	Empire gudgeon	*		*	0.4
	Hypseleotris klunzingeri	Western carp gudgeon	*			0
	Hypseleotris galii	Firetail gudgeon	*			0
	Hypseleotris spp. 1	Midgeley's carp gudgeon	*	*	*	0.4
	Mogurnda adspersa	Purple-spotted gudgeon	*	*	*	0.1
	Oxyeleotris lineolatus	Sleepy cod	*		*	1.5
Elopidae	*Elops hawaiensis §*	Giant Herring	*			
Engraulidae	*Thryssa scratchleyi §*	Freshwater anchovy	*			
Gobiidae	*Redigobius bikolanus §*	Speckled goby	*			
Kuhliidae	*Kuhlia rupestris §*	Jungle perch	*			
Lutjanidae	*Lutjanus argentimaculatus §*	Mangrove jack	*			
Megalopidae	*Megalops cyprinoides §*	Tarpon	*	*	*	1.4
Melanotaeniidae	*Melanotaenia splendida*	Eastern rainbow fish	*	*	*	14.6
Scatophagidae	*Scatophagus argus §*	Spotted scat	*			
	Selenotoca multifasciata §	Banded scat	*			
Synbranchidae	*Ophisternon gutturale*	Swamp eel	*	*		
Terapontidae	*Amniataba percoides*	Barred grunter	*		*	0.1
	Hephaestus fuliginosus	Sooty grunter	*			
	Leiopotherapon unicolor	Spangled perch	*	*	*	0.1
Toxotidae	*Toxotes chatareus*	Seven-spot Archerfish	*	*	*	0.1
	Neosilurus ater	Black catfish	*	*	*	0.2
	Neosilurus hyrtlii	Hyrtl's tandan	*	*		
	Arrhampus sclerolepis	Snub-nosed garfish			*	0.1
	Porochilus rendahli	Rendahl's catfish	*	*	*	0.1
	Scortum parviceps	Small-headed grunter	*			
Non-native fish species						
Cichlidae	*Oreochromis mossambicus*	Tilapia	*		*	12.5
Osphronemidae	*Trichopodus trichopterus*	Three-spot gourami	*	*	*	0.1
Poeciliidae	*Gambusia holbrooki*	Eastern mosquitofish	*	*	*	1.8
		Total species	**38**	**19**	**24**	**100**

6.0 DISCUSSION AND RECOMMENDATIONS

6.1 Aquatic plant management

Land use changes have altered the hydrology, limnology and ground/surface water interface; meaning, the Burdekin floodplain will only ever remain in a modified, hybrid, state providing a subset of the ecosystem services that it once did. For as long as sugar production occurs on this floodplain and delivery of excessive available nutrients reache local creeks (Brodie et al. 2016; Waterhouse et al. 2017), aquatic invasive plants will continue to require some form of management intervention and mostly in the form of aquatic treatment. The issue continues to be more critical for the clear water lagoons, located at the end of the distribution network or off-channel sections (Perna et al. 2012), where submerged aquatic vegetation biomass is high because of a deeper euphotic zone. High plant biomass (particularly species such as water hyacinth, which forms densely packed mats, and submerged aquatic vegetation, such as Ceratophyllum sp.) places a high net demand on available oxygen in the water column, resulting in extreme diel cycling compared to turbid lagoons which generally have lower daily amplitudes and higher minimum values. Maintaining the Sheep Staton Creek in a moderately turbid state along its entire linear length is probably important in reducing the threat of eutrophic-driven hypoxia and fish kills (in the absence of floodplain-scale water treatment of sugar farm runoff), mostly owing to the suppression of submerged aquatic vegetation (Burrows and Butler 2012) – this has been the focus entirely on the floodplain wetland network values as a functioning ecosystem in its own right for the past few decades.

This management approach is counterintuitive when considering broader GBR initiatives set under the Reef 2050 plan which have centered on reducing land derived suspended sediment that are known to cause effects on offshore (downstream) coral reefs (Bartley et al. 2017) and seagrass meadows (Coles et al. 2015), which form part of the Bowling Green Bay RAMSAR wetland complex (DEHP 2016). In managing the threat of aquatic plant overgrowth and poor water quality conditions, the strategy thereby is to keep the floodplain creek network in a turbid state, as a means towards at least giving the best chance of better DO cycling, thereby reducing changes of fish kills.

The impact of water hyacinth on water and habitat quality is a challenge for managers requiring considerable investment control (Masifwa et al. 2001; Villamagna and Murphy 2010). Although costly, the return in terms of water delivery for irrigation, flood risk immunity, water quality conditions, lowered risk of fish kills, and habitat value should not continue to be undersold even in a directive of budget cuts by funding partners. While spraying herbicide chemicals is the most cost-effective way to control floating plants, it can have detrimental secondary effects when the organic material decomposes, creating a massive oxygen demand on the water column (Villamagna and Murphy, 2010; Waltham and Fixler, 2017). Alternative approaches are available, such as mechanical excavation, a technique considered to be the most effective ecosystem-responsive technique (Masifwa et al., 2001; Greenfield et al., 2007; Güereña et al., 2015), or even removing barriers and allowing tidal ingress which is possible in some places along the coast (Abbott et al., 2020). Opportunities to decompose vegetation material on nearby sugarcane farms are possible and have the added benefit of supplementing fertilizer application to cane fields.

The capacity of submergent plants to produce sufficient oxygen to sustain the system hinges on two key factors: 1) light availability and 2) background respiration rates (the principal source of which are microbes).

1) During prolonged dry spells, the water is generally sufficiently clear for submergent plants throughout most of the waterbody to obtain light. However, when plant biomass reaches high levels, production of oxygen by the parts of the plants (and the epiphytic algal communities they support) that lie beneath the underwater canopy may become light limited due to self-shading – hence low DO levels may develop in the deeper water layers. Shading from floating and emergent macrophytes, and light attenuation due to turbidity caused by algal blooms, may also gradually reduce oxygen production rates in the underlying water column. These effects are generally manifested gradually over periods of months and cause a general decline in DO levels, especially in deeper water layers. These effects can be exacerbated (over periods of days to weeks) by heavily overcast conditions which reduce the intensity of sunlight and substantially increase the proportion of the waterbody that is unable to support photosynthesis. Acute/rapid oxygen sags can occur (over periods of hours to days) if the turbidity and/or colour of the water suddenly increases to the point where a large proportion of the submergent plants are unable to obtain sufficient light to photosynthesise. Sudden increases of that sort are most commonly a consequence of inflows of turbid stormwater, especially in connection with first-flush events (i.e., the first rain storms that occur after a prolonged dry period). These effects can be exacerbated if the inflows cause a sudden rise in water level (further decreasing the light available to plants growing on the bottom.

2) Though most aquatic organisms (including plants) consume oxygen, fluctuations in background consumption rates are largely a function of microbial respiration (resulting from the decomposition of organic matter). During dry spells instream/riparian production of plant detritus is the major source of organic matter, although there are times and places where excrement from waterbirds may be a contributing factor. Accordingly as biomass increases over time, so do respiratory oxygen consumption rates. Most of this organic matter settles to the bottom either directly (e.g., as leaf litter) or indirectly in the form of animal faeces; hence respiration rates near the bottom increase disproportionately often resulting in low DO levels. Hypoxic bottom waters of this sort are a natural feature of most standing waters and do not generally present a risk to aquatic organisms provided that the hypoxic water remains near the bottom. However, if the hypoxic bottom waters come to the surface they can cause acute problems. Such episodes, termed turnover events, can occur if the surface water layer cools suddenly causing the water column to mix. The sudden drop in air temperatures during storm events and the low temperature of rain water runoff are a potential cause of turnovers. Since rain runoff also typically contains significant quantities of oxygen-demanding organic matter the overall effects of storm water inflows can be dramatic potentially resulting in severe oxygen sags. Attendant increases in water depth can also result in drowning of terrestrial weeds along the water's edge and entraining organic matter in the riparian zone – further increasing oxygen demand.

In our experience in northern Queensland, rain events will deliver turbid stormwater that dramatically reduces the available light. Without access to light, submerged aquatic plants begin to decompose where oxygen consuming bacteria break down the organic material, or indeed will not become established due to low available light (Figure 22). This process creates a huge oxygen demand, which leads to extended periods of critically low oxygen. Warm conditions (commonly experienced during summer months) further reduces the available oxygen in the water. In addition, stormwater flow may also disturb hypoxic waters along the edge areas particularly where large areas of floating vegetation exist, moving it out into the open water areas of the river. This process has the potential to spread poor water quality conditions more broadly throughout the river. Extended cloudy days, again which generally occur following rainfall periods, further limits recovery.

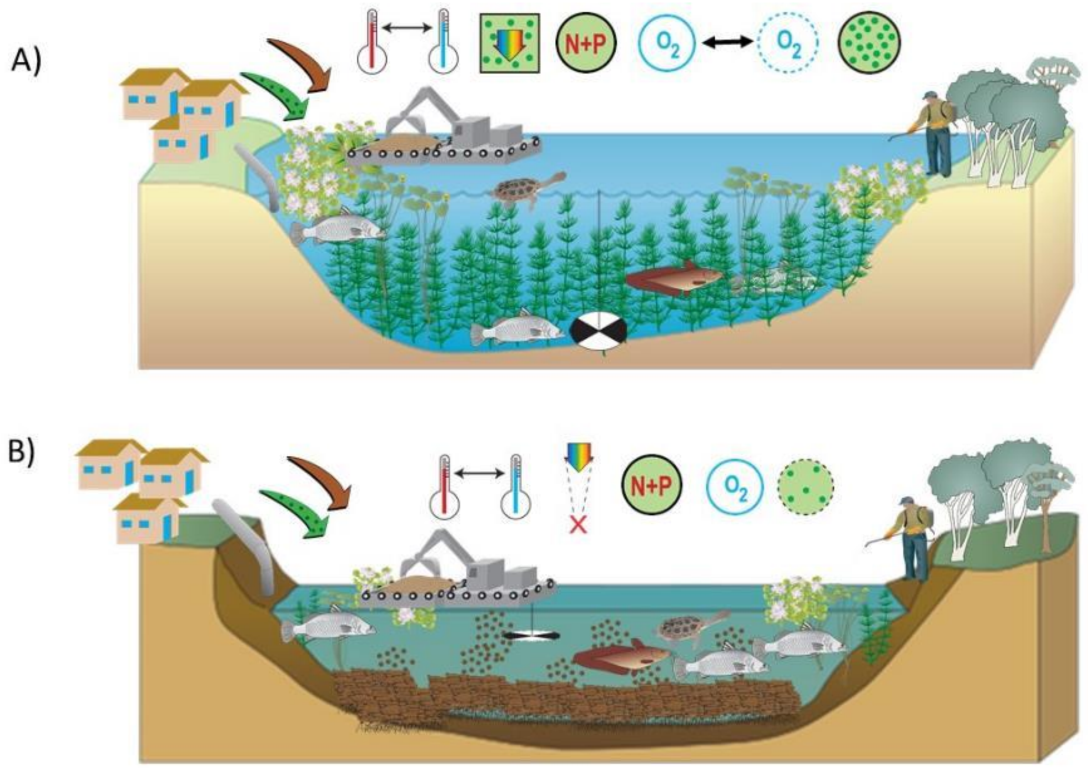


Figure 22: Conceptual diagram of conditions in tropical freshwaters. A) excessive nutrients from urban runoff provides conditions to promote excessive submerged and floating aquatic plant growth. DO cycling is high, while water transparency is high, in addition to water temperature conditions that also vary spatially between open waters and edge vegetated areas. Maintenance efforts including some edge spraying to reduce hypoxic conditions in the epilimnion, while mechanical removal of vegetation is necessary to reduce the potential DO sag from occurring after rainfall; and B) following rainfall, water flow and turbidity cause DO sag, which if conditions do not permit recovery, a fish kill will occur (Butler and Burrows 2007).

6.2 Long term maintenance

The increase in fish species richness following works in Crooked Waterhole is an important result and highlights that, for coastal low-lying areas immediately upstream of tidal areas, restoration leading to reconnection of the seascape extends usable areas for fish habitat. This conclusion was best supported by the presence of juvenile Barramundi following the first wet season connection with downstream areas after the works, permitting juveniles to migrate upstream areas. As the season progressed, Barramundi captured in the creek were larger in size, and were even caught in the upstream areas of Crooked Waterhole. The possible reasons for this may be either because larger individuals had migrated upstream during the later stages of the wet season flow, or they were juveniles that had grown as the dry season progressed.

Either way, the results are positive, particularly when coupled with the improved water quality conditions (i.e., higher DO), when previously this was not the case. However, a concern was the number of Mozambique Tilapia across all size ranges, as this is an introduced species that negatively impacts native fish species (Doupe et al. 2009). Mozambique Tilapia are widespread in the region and reducing numbers will be impossible given it is highly tolerant to poor water quality including low DO and high temperatures (Butler and Burrows 2007).

The investment in aquatic weed removal and on-going maintenance not only improved water quality but also survival of fish reaching the restored waterway compared to prior condition in the creek. To retain the advantages gained and avoid the return of poor conditions, there is an on-going need to continue weed maintenance works in this creek. (For example, a site inspection in Crooked Waterhole in April 2020 recorded large areas of floating aquatic weeds that require maintenance). Hence, satisfactory outcome of this project relies on on-going spot-spraying maintenance of aquatic weeds. Without this – and similar works in other nearby waterways - excessive weed will continue to choke parts of the Burdekin floodplain and prevent upstream fish recruitment during wet season flows. The investment in aquatic weed removal and on-going maintenance is critical, not only for improved water quality but also survival of fish reaching the restored waterway. A funding model similar to the Sheep Station Creek maintenance program, which has continued for almost 20 years, is probably the best example of long term maintenance necessary on the floodplain. Establishing a similar partnership arrangement is necessary in all areas of the Burdekin floodplain.

6.3 Reducing tailwater delivery

Installation of recycle pits on the Burdekin floodplain might serve as a positive way forward in addressing many challenges with water management, such as efficiencies in water delivery and management of tailwater. Recycle pits are a structure designed to capture irrigation run-off (tailwater) for re-use in the production area (Queensland WetlandInfo, accessed September 2020). If designed, sited and operated correctly, recycle pits have water quantity and quality benefits by ensuring tailwater, and any associated sediments, nutrients and pesticides, are re-used on the farm and do not enter natural waterways and wetlands. Unlike many other treatment systems used in agricultural production systems (Wallace et al. 2020), recycle pits are engineered to not treat the water, rather they rely on the water being captured and re-used on the farm (Figure 23). If situated in the lower ends of the catchment, they can capture irrigation water running off into the shallow coastal wetlands (including Bowling Green Bay Ramsar wetland). The capture of this water coupled with improved irrigation water delivery efficiency can reinstate the natural wetting and drying cycles which self regulates invasive floristic species and improves fish connectivity after the initial capital investment. Further research should examine the fertiliser avoidance/efficiencies that recycling nutrient-laden water in recycle pits offers to farmers.

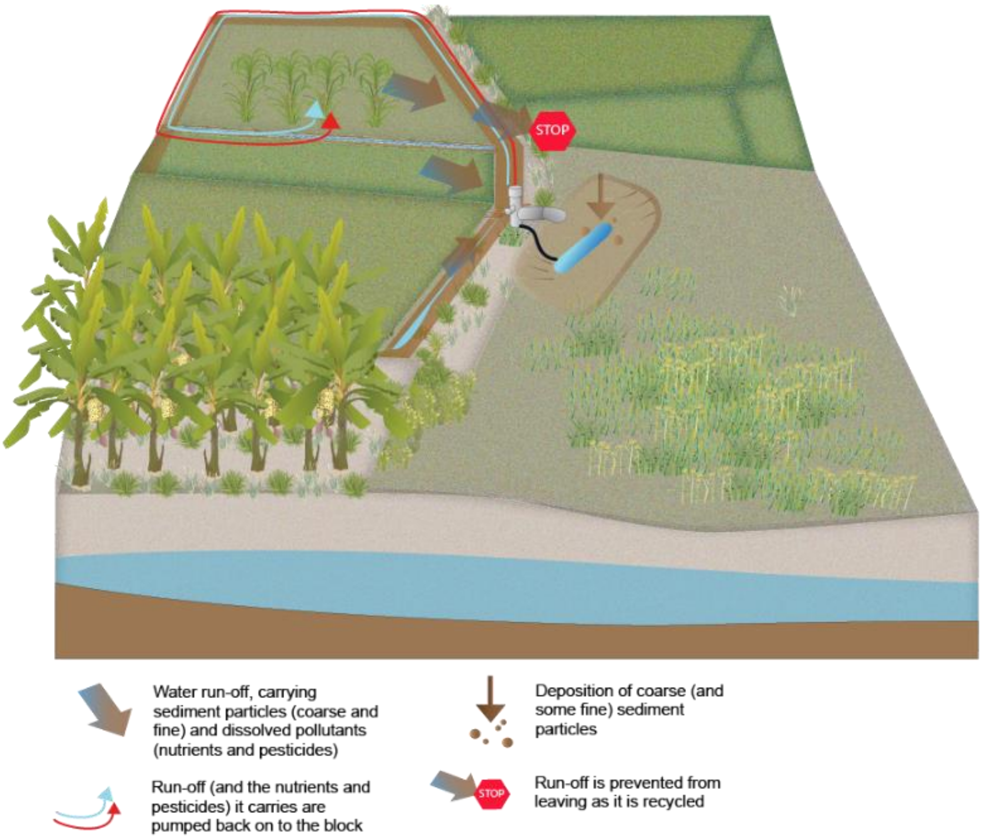


Figure 23: Conceptual example of a recycle pit that captures runoff water from agricultural property (and the nutrients and pesticides) and pumps the water back to the property to reuse the water and available nutrients (source Queensland WetlandInfo).

Recycle pits have the potential to reduce the losses of sediments, nutrients and pesticides to downstream waterways, however, need to be operated correctly in order to improve water quality. That is, to ensure captured water is rapidly re-used on the farm and sufficient capacity is maintained in the recycle pit to capture all irrigation tailwater, without overflowing to downstream waterways (more details can be found on WetlandInfo, Queensland Government). The use of recycle pits in achieving water quality improvements on the Burdekin floodplain and costing analysis is being completed in a companion NESP project (see Waltham et al. (2020a)). The colonisation of freshwater bivalves would also deliver effective bioengineering services in recycle pits.

6.4 Connectivity on the floodplain

Modification and works lower on the floodplain in upper tidal and palustrine areas, such as removing sediment build up, installation of fish passage structures, mitigating weed chokes, and controlling/reducing the over delivery of irrigation water continues to be a major challenge for the Burdekin floodplain. The most critical challenge is the oversupply of freshwater that reaches the floodplain, the persistence of invasive aquatic weeds and low dissolved oxygen availability for fish (Butler et al., 2009). Efforts to work with water delivery managers is critical and requires more attention and investment in order to strike a balance between water supply to irrigators with which that is necessary for successful fish passage. In Kalamia, tidal water rarely reaches the fish ladder, which is probably the result of freshwater leaking through or over the top of the earth bund wall, or even through the fish ladder itself. In an attempt to repair the system, a water balance management plan is necessary which will require a system of better management of water delivery through the Lilliesmere network. Wet season rainfall is the exception, where large volumes of freshwater reach Kalamia estuary. However, following the wet season, the delivery of freshwater and transfer past the final lagoon system into the upper estuary will continue to result in being overgrown by freshwater weeds. This seems to be a challenge in many places across the Burdekin floodplain (e.g., Saltwater Creek restoration site). Based on the laboratory trials (Reid et al. 2018), only a low level of saltwater is necessary to achieve the desired outcome of weed removal. Saltwater ingress controlling freshwater plant chokes was successful in Jerona, a site located at the upper intertidal areas that is more frequently connected, probably each month (NQDT per. comms), but that does not receive tailwater from the irrigation channel network.

Barramundi are a good surrogate indicator of floodplain connectivity. A lack of connectivity to coastal wetlands ultimately means less available habitat for fish. A simple agent-based model was used to simulate the passage of an artificial fishery migrating from the coast up the watercourse network to wetlands, and assess the extent of accessible wetlands with and without manmade barriers in place (see Appendix A). Across the Burdekin coastal waterway network, barriers were predicted to prevent access to ~1,006 km of waterway. With mapped barriers, approximately 1,979 km of waterways were considered accessible, in comparison to approximately 2,985 km estimated to be accessible in the absence of barriers (Figure 24). This equates to approximately 60,500 ha of wetland currently accessible, with an additional 24,500 ha potentially available in barrier-absent conditions (total = 88,000 ha). While this analysis is theoretical and uses arbitrary preferences, further analysis could examine the preference selection probabilities of highly valued migratory species, such as barramundi and mangrove jack.

Between 2010 and 2019, the average annual Barramundi catch on the Burdekin catchment or coastline was 46,653 kg (Department of Agriculture and Fisheries, 2020), which retail for ~$30/kg (total value ~$1.4 million). If the population productivity was limited by accessible wetland habitat and increased proportionally to an increase in habitat availability, then removing all barriers to the Burdekin's coastal wetlands could increase the local fishery value ~45% to over $2 million annually (Table 10). This is a very coarse approximation; however, further analysis exploring the impact of available coastal wetland habitat on commercial fisheries is required, in addition to the potential confounding in the analysis caused by stocking barramundi on the floodplain which does occur from time-to-time.

Table 10: The type and extent (ha) of wetland habitats potentially accessible to an artificial diadromous fishery simulated across the coastal Burdekin catchment, Queensland.

Habitats	Extent accessible with no barriers	Extent accessible with barriers	Habitat available if barriers removed
Artificial/ highly modified wetlands (dams, ring tanks, irrigation channel	723.3	544.9	178.5
Coastal/ Sub-coastal floodplain grass, sedge and herb swamps	13055.4	1673.0	11382.4
Coastal/ Sub-coastal floodplain lakes	29.1	0.0	29.1
Coastal/ Sub-coastal floodplain tree swamps (Melaleuca and Eucalypt)	2118.8	1998.3	120.5
Coastal/ Sub-Coastal floodplain wet heath swamps	0.5	0.5	0.0
Coastal/ Sub-coastal non-floodplain grass, sedge and herb swamps	4.3	0.0	4.3
Coastal/ Sub-coastal non-floodplain soil lakes	437.3	0.3	437.0
Coastal/ Sub-Coastal non-floodplain tree swamps (Melaleuca and Eucalypt)	5953.0	2687.7	3265.2
Coastal/ Sub-Coastal saline swamps	1454.6	121.0	1333.6
Estuarine - Mangroves and related tree communities	21779.6	18947.3	2832.3
Estuarine - salt flats and saltmarshes	25318.6	21926.5	3392.1
Estuarine - water	4009.9	3721.0	288.9
Riverine	13113.7	8916.0	4197.7
Total	**87998.2**	**60536.6**	**27461.6**

The barramundi (*Lates calcarifer*) is widespread across northern Australia (Pusey et al. 2004; James et al. 2017), though it is important to note that individuals caught were probably part of a broader fish stocking restoration program given that not a single specimen captured was small enough to indicate successful recruitment from lower coastal tidal reaches. The fact that no juveniles were caught supports the notion that migration during the wet season is hindered by the extent and number of flow control structures/constrictions located on the floodplain, which are necessary for regulating the supply of water for irrigation (Alluvium 2007). Recruitment of juvenile barramundi have been seen elsewhere on the Burdekin floodplain, where migration barriers are minimal and restoration work via removal of aquatic weeds low on the floodplain permit successful connection. Returning adult barramundi and new recruits would also benefit from the installation of fish passage structures, automated gates or even low-level rock ramped bunds at the end of the irrigation delivery area. Improved irrigation management would also minimise water losses from the delivery area to wetlands between the irrigation area and the coast (part of the Bowling Green Bay Ramsar wetland), which are the direct cause of substantial weed chokes (para grass; *Brachiaria mutica*) that reduce connectivity between fresh and saltwater sections of this system (Burrows et al. 2012).

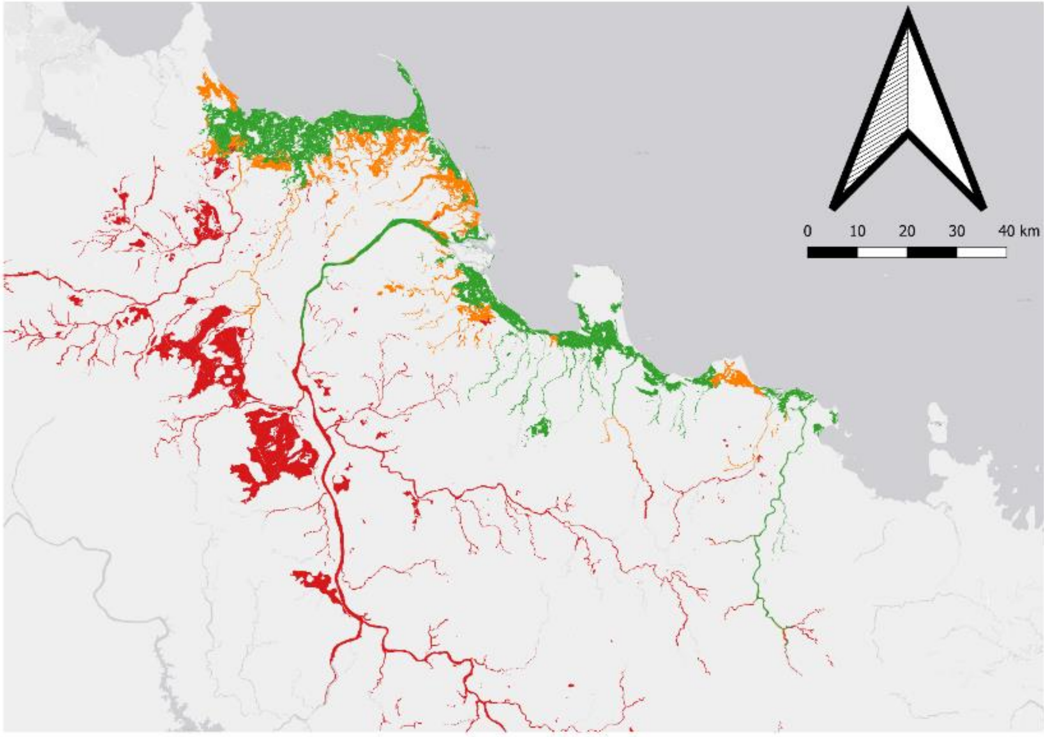


Figure 24: Burdekin coastal wetlands predicted to be accessible by an artificial diadromous fishery with existing barriers present (green), with existing barriers removed (orange), or inaccessible with or without barriers (red).

The findings here are important given a strategic focus and recognition that freshwater floodplain wetlands form an important biological component, though are heavily modified and are in a poor state of condition in the GBR seascape (Arthington et al. 2015; Waltham et al. 2019). The Reef 2050 Plan has a target that '*there is no net loss of the extent, and a net improvement in the condition, of natural wetlands and riparian vegetation that contribute to Reef resilience and ecosystem health'*. An action set out in the *Wetlands in the Great Barrier Reef Catchments Management Strategy 2016-21* is that '*on-ground actions, education and scientific research are necessary to improve management and repair of wetlands in the GBR catchments*'.

The research completed in this case study demonstrate that for the objective of providing high-quality fish habitat, the management of aquatic weeds is necessary and requires long-term funding to ensure the intervention efforts are not lost in the future. A much shorter-term maintenance commitment would probably not achieve the results shown here, and though successful in terms of still providing fish habitat, more maintenance work is necessary to further capitalise on the biological return on the funding investment. The funding model in Sheep Station Creek seems to continue meeting the services and values identified by the community (i.e., keeping the creek open and weed free for fish habitats), though this same funding model needs to be rolled out to other sections of the Burdekin floodplain, and will unfortunately be a necessary expense for as long as excessive nutrients reach the floodplain. In reviewing the literature, it seems that few studies exist that have had such a long-term maintenance investment (Duarte et al. 2009). In fact, many do not present a strong case demonstrating that the project goals were met which could mean the funding model designed to support the restoration investment and effort has likely failed (Li and Pennings 2018).

6.5 Recommendations

Managing the Burdekin floodplain for multiple services will continue to be challenging because of its heavily modified condition. Management of the services provided by the Burdekin floodplain will require a whole-of-system approach, with involvement and commitment from all stakeholders. The success here will only be possible with this support and focus by all interest and landholder groups. There are a number of key learnings from this project: that combined with knowledge held more broadly on the floodplain held by our research group, need to be considered and used by stakeholders.

The management needed on the floodplain in terms of protecting and enhancing the services, provides an important lesson for other agricultural development enterprises on floodplains in northern Australia. There are no ‘off the shelf’ guides - local scientific limnological data and knowledge of each floodplain where development is proposed is necessary. The detail and scale of data necessary to effectively manage agricultural development on floodplains should not be underestimated, and the Burdekin is a case in point where some 30 yrs after the Burdekin dam was built we are still learning more about this complex wetland floodplain.

Funding for continued maintenance after restoration works is necessary. The Sheep Station creek funding model examined here is a case in point, but this model needs expanding across the entire Burdekin floodplain. Management of water delivery across the floodplain also requires more attention, to ensure sensitive palustrine wetlands are protected and enhanced. For as long as this floodplain remains in use for agricultural production, there will need to be on-going funding support for maintenance, and monitoring and evaluation to ensure the ecosystem services and values are protected.

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APPENDIX 1: METHDOLOGY FOR MODELLING DIADROMOUS FISH CONNECTIVITY ON THE BURDEKIN FLOODPLAIN

A simple agent-based model was used to simulate the passage of an artificial fishery species migrating from the coast up the watercourse network to wetlands. The watercourse network used was that derived by Geosciences Australia (from a 9 second DEM) for the National Surface Hydrology Database. This network was spatially limited to the coastal bioregion as described in "Module 8. Native Vegetation Clearing" in the Queensland State Development Assessment Provisions (SDAP). In the Burdekin region, there are 5208 river reaches (network segments representing part of a watercourse), that average 2.7 km in length. This network was spatially joined to the Queensland wetland mapping geodatabase (Environmental Protection Agency and Agency, 2005; Department of Environment and Science, 2019). The decision for an individual to move to a new reach, or remain present, is based on the random selection of the adjacent and present reaches, weighted by preference probabilities assigned to various attributes of each river reach. The overall probability of reach selection was the product of preference probabilities applied to each attribute.

River reach selection preferences (and rationale) were assigned for the following attributes:

Temperature. Along the Queensland east coast, Barramundi migrate upstream largely during the Wet Season (Nov – Mar), and have an optimal temperature range of 25-30°C, with optimal larval survival between 27-28°C. Feeding activity is greatly reduced below 24°C and movement almost ceases below 22.5°C. Barramundi have also been recorded in waters as high as 35°C (Pusey et al. 2004; Heupel et al. 2011). Given this, preference probabilities for mean catchment air temperature during the wettest quarter (as a surrogate for water temperature during the peak wet season migration period) were assigned as follows: 0% selection for reaches below 22°C, 20% at ≤24°C or ≥30°C, and 80% for 25-30°C.

Flow. Many watercourses across Queensland are intermittent, this limits their ability to support fisheries long term and provide for fish passage. Given this, perennial streams were assigned 80% preference, while intermittent were assigned 20% preference.

Riparian vegetation. Riparian vegetation can help in the provision of preferable habitat for Barramundi by filtering runoff (e.g., nutrients and sediment), regulating stream temperature, providing instream cover that limits algal and macrophyte growth, and provides terrestrial invertebrate inputs for sustenance. Riparian vegetation can also contribute to instream wood, which provides habitat for prey species and refuge from predators. For these reasons, river reaches with at least 20% shrub cover were assigned 80% preference probability, while those with less coverage were assigned a 20% preference probability.

Waterfalls. Where a reach has a waterfall, then these were considered as natural and permanent barriers, and assigned a 0% preference.

Artificial barriers. Barriers to fish passage were a collation of mapped barriers from the NESA database, Queensland wetland mapping bund locations (Environmental Protection Agency 2005; Department of Environment and Science 2019), NQ Dry Tropics fish barrier GIS database and the authors local knowledge. Reaches with a mapped barrier were assigned a 0% preference.

For this exercise, two simulations were ran to estimate the type and extent of wetland habitats accessible to fish migrating from the coast. In the first scenario, all artificial barriers were deemed unpassable (i.e., a 0% preference); while the second scenario examined movement in the absence of any barriers. Both simulations used a population of 10 000 individuals and 50 day.week.yr? timestep. A wetland was considered accessible if any of the individuals visited at least once. All analysis was carried out using QGIS and R (R Core Team, 2016; QGIS.org, 2020).

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