Tropical Water Quality Hub

National Environmental Science Programme

Final Report

Evaluating wetland restoration success: feral pig fencing for conservation of Round Hill Reserve

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Australian Government

National Environmental Science Programme

Supported by the Australian Government's
National Environmental Science Program
Project 3.3.2 Evaluating wetland system repair projects in GBR catchments

National Library of Australia Cataloguing-in-Publication entry:
978-1-925514-70-4

This report should be cited as:
Waltham, N.J., Buelow, C.A., and Iles, J.A. (2020) *Evaluating wetland restoration success: feral pig exclusion fencing in the Round Hill Reserve.* Report to the National Environmental Science Program. Reef and Rainforest Research Centre Limited, Cairns (96pp.).

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

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

Cover photograph: (front and back) Roundhill Reserve feral pig fence. Images: S. Jackson..

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ACRONYMS

ANOVA .... Analysis of Variance
ASS.............. Acid Sulfate Soil
BMRG .... Burnett Mary Regional Group
BN................ Bayesian Network
C .... Carbon
DIN................ Dissolved Inorganic Nitrogen
DIWA .... Directory of Important Wetlands in Australia
DO .... Dissolved Oxygen
EC................. Electrical conductivity
ERT…………..Ecologically Relevant Targets
FHA............... Fish Habitat Area
FRP............... Filterable Reactive Phosphorus
GBR.............. Great Barrier Reef
H-BBN .... Habitability Bayesian Belief Network
HES…………..High Ecological Significance
ID .... Inner Diameter
JCU............... James Cook University
LOI................ Loss on Ignition
MC .... Moisture Content
NH3............... Ammonia
N .... Nitrogen
NESP .... National Environmental Science Program
NP................. National Park
NRM………… Natural Resource Management
OC .... Organic Carbon
OM .... Organic Matter
P .... Phosphorus
PN................. Particulate Nitrogen
PP .... Particulate Phosphorus
QLD .... Queensland
QPWS .... Queensland Parks and Wildlife Service
SVL .... Snout Ventral Length
TDN............... Total Dissolved Nitrogen
TDP............... Total Dissolved Phosphorus
TL .... Total length
TN .... Total Nitrogen
TP .... Total Phosphorus
TWQ.............. Tropical Water Quality
TSS............... Total Suspended Solids
UAV .... Uncrewed Aerial Vehicle
USA .... United States of America

ABBREVIATIONS

ºC.......................Celsius
dbRDA...............Distance based redundancy analysis
mg L^{-1} ....Milligrams per litre
mS cm^{-1} ....MilliSiemens per centimetre
nMDS.................Non-metric multidimensional scaling
NOx....................Nitrogen oxides
PERMANOVA....Permutational multivariate analysis of variance
PERMDISP ....Homogeneity of within-group multivariate dispersions
µg L^{-1} ....Micrograms per litre

ACKNOWLEDGEMENTS

This project would not have happened without the tireless support of the late Maree Prior. You were a joy to work with in the field, never afraid to dive into the work. You will be greatly missed by the Queensland coastal conservation community.

We thank the Australian Government's National Environmental Science Program (NESP) Tropical Water Quality (TWQ) Hub for funding this research project. We acknowledge the contribution of stakeholders who have provided advice and input into the project, with specific reference to Fiona Dinte (community volunteer), and Shane Jackson (Queensland Parks and Wildlife Service) for assistance in the field. Special thanks also to Dr Eric Nordberg, James Cook University, for providing expertise and assistance in surveying herpetofauna, and to Sophie Walker (Masters Minor project, James Cook University) for classifying aerial imagery. We also thank the numerous Queensland Parks and Wildlife Rangers who worked long days with us, and providing access to machinery and equipment to complete this project. We also thank Mike Ronan for tireless support, mentorship and advice. We also thank the two reviewers that provided comments which improved this report.

EXECUTIVE SUMMARY

The wetlands along the Great Barrier Reef coastline have been modified and altered over the past few decades (though the rate of change has slowed in more recent years). These landscape changes have been mostly to accommodate increasing urbanisation and associated infrastructure, but in response to expansion of agricultural areas particularly along low-lying coastal areas. A consequence of these disturbances has been a reduction in the services and values provided by the remaining wetlands, with many in a degraded state with poor water quality, or impacted by invasive plants and animals and altered hydrology. In the past few years, there has been recognition and a desire to protect coastal wetlands, and undertake restoration programs that maximise the services and values provided by wetlands. The success of these efforts needs to be supported by scientific data, which is generally in limited supply for managers challenged with implementing successful restoration programs that delivery broader conservation and protection outcomes.

Feral pigs cause major damage to coastal wetlands across northern Australia. Mitigation attempts have focused on the pig population, relying on aerial shooting, baiting or trapping, which generally have not been successful to date in reducing the population more long term. Using fences that boarder a wetland has received much attention, as a way to exclude feral pigs from accessing wetlands, and both limiting access to these resources while at the same time providing an opportunity for wetlands to recover. In November 2016 the Burnett Mary Regional Group (BMRG) constructed an exclusion fence surrounding the Round Hill Reserve, a coastal wetland in the upper reaches of Round Hill creek, located in the Baffle catchment. The primary objective of this restoration activity was to control feral pig (*Sus scrofa*) damage directly to the wetland. Round Hill creek was identified as a restoration priority because it is an important Fish Habitat Area (FHA), and is a Directory of Important Wetlands in Australia (DIWA) listed wetland. In collaboration with BMRG and Queensland Parks and Wildlife Services (QPWS), the desired value and services, with a focus on recovery of water birds, native vegetation, improved water quality and general aquatic fauna determined, were set for this wetland restoration site.

Between July 2017 and November 2019 (proximal to post wet season and late dry season), surveys of the native vegetation cover and extent was completed using drone aerial analysis, and quadrat surveys. Fish community were examined using fyke nets set to soak overnight during surveys, while water sampling was also completed during each survey. This sampling approach was established and agreed to by the stakeholders involved in the first inception meeting. In addition to the fenced wetland area (Eurimbula National Park, Round Hill Reserve), four additional wetland sites were included, consisting of two sites approximately 60km to the south, and another site located in the Eurimbula National Park approximately 20km to the north. The Eurimbula National Park Round Hill Reserve is the only wetland site here with a conservation fence in place, other sites were open to cattle and pig access.

A summary of the results are provided below:

- Of the five wetlands surveyed here, the fenced wetland in Eurimbula National Park supported a more diverse floralistic community than other sites visited. This was mostly due to the presence of freshwater species which were absent elsewhere. The fenced wetland site is positioned high in the estuary meaning it is least frequently inundated by tides, and more regularly inundated by freshwater flows after rainfall events. There were some limitations in the vegetation sampling design which made it difficult to compare UAV derived vegetation types with what was measured on the ground during quadrat surveys. Quadrats were generally placed close to or on boundaries between vegetation types, this was especially the case with 'high' sites which were placed at the saltmarsh-mangrove boundary. Mangroves overhead of quadrats were not recorded as the vegetation survey was focussed on what was within the quadrat on the ground. This may have been because the trunk itself was not within the quadrat, while branches and foliage were overhead. The conservation fence at the fenced wetland site was ineffective at protecting fragile marsh vegetation from trampling. Drone imagery and field observations identified higher vegetation cover in marsh areas outside of the fenced enclosure in areas where cattle were unable to access due to the natural barrier of a bend in the creek. The management practice of opening the fence to allow cattle to enter the wetland to graze (and presumably also feral pigs) needs to cease in order to ensure the fencing mitigation is effective.

- Saltmarsh is a common coastal tidal wetland vegetation species in the study region. Here saltmarsh soil carbon decomposition was investigated using constructed fiberglass mesh bags containing 200 g of the root and rhizome material that were position on the wetland soil surface and buried at 10cm depth, at two positions within each wetland (parallel with the low and high tide lines determined by the boundary with mangroves and saltmarsh/terrestrial vegetation. After 20 weeks *in situ* the bag weights were measured with difference from the starting weight determined as the decomposition rate (calculated as a daily rate). Overall the rate of organic matter decomposition was generally highest when buried compared to surface deposited bags. The organic matter contained in the surface litter bags are exposed to photochemical and physical degradation, hence would likely undergo rapid leaching. Burial (accretion) of saltmarsh litter reduces the rate of organic matter decomposition, hence both increasing carbon stock and reducing CO_2 release to the atmosphere. Once burial commences (either experimentally, or by accretion), the organic carbon contained in the litter bags is no longer exposed. Interestingly, the recovery rate of bags was lowest at the high tide position for those bags deployed at the surface; by this we assume that by the process of the cattle eating saltmarsh grass, they have also consumed the decomposition bags, which highlights the ineffectiveness of the current management of these wetlands and fencing to exclude feral pigs.

- Water quality measured at sites for physio-chemical conditions (temperature, conductivity, dissolved oxygen and pH) in addition to nutrients and total suspended sediments varied among wetlands and surveys. The data is limited because of the nature of this study, however, there was some evidence of seasonal cycling, and elevated nutrient concentrations in the post wet season surveys (but this cannot be confirmed). Dissolved oxygen was at times critically low, below asphyxiation thresholds known for some of the fish species captured, which might also explain the low number of fish species. Most of the nitrogen present was in the filterable fraction, while TN:TP and TDN:TDP ratios indicated that nitrogen was generally in excess. Total suspended sediments ranged between 7 and 140 mg/L, with the higher results measured after cattle had recently access the wetland site.

- A total of 12 fish species were detected across the wetland sites. The small bodied fish Gobiidae (true goby) and *Pseudomugil signifier* (Pacific blue-eye) were the most abundant species encountered. Two exotic species; *Oreochromis mossambicus* (mozambique tilapia) and *Gambusia affinis* (eastern gambusia) were present, but not all wetland sites. *Macrobrachium Rosenbergii* (giant river prawn) was present at only a single wetland site, so too were the *Metapenaeus bennettae* (greasy-back prawn) and *Metapenaeus macleayi* (school prawn) which are both commercially and recreationally targeted species in Queensland. Species richness was highest at the fenced wetland with 15 species present. Overall, the number and abundance of fish species captured in these wetlands was low when compared to the species known to occur in the region. This result is probably due to the limited connectivity these wetlands have with downstream estuaries, with most sites drying out almost completely during the late dry season, thereby these wetland areas probably provide limited fish habitat values.

- Twelve herpetofauna species were present across the five wetland sites (5 skink, 1 gecko, 5 frogs, and 1 snake species). This number is considered was low and probably reflects the short survey completed and during winter months, where warmer months would probably results in more species recorded. The highest diversity and abundance was from RH3, which paradoxically is one of the most disturbed sites (with cattle and pig access). The introduced cane toad (*Rhinella marina*) was detected at three sites.

- More waterbird species were detected during the post-wet season surveys (March 2018) in comparison to dry season surveys (July 2018). Water availability and depth are key factors determining use of wetlands by waterbirds, and a decrease in water availability during the dry season many explain the pattern here of fewer waterbirds detected in July 2018. Overall, the services and values provided by saltpan areas in Queensland are generally unknown, with only some data relating to fish habitat use. The data indicate that a range of species probably utilise these supratidal areas, which does give rise to their protection and conservation even if they only support a subset of species known more broadly in coastal estuaries, for example fish. Under sea level rise scenarios however, the frequency and duration of connection with downstream estuaries might increase giving rise to model that these areas might become important future habitat, highlighting the need to still conserve and protect them from feral pig and cattle impacts.

The key conclusions and recommendations:

- Restoration targeting the threat of feral pig (and cattle) damage caused to coastal wetlands is important and will continue to be necessary in the future. Evidence presented here portrays that fencing for wetland vegetation community values has indeed achieved much more so than when compared to wetlands that remain exposed to feral pig damage. There were some indication that the vegetation community, fish and bird diversity werereturning after the fence was erected. However, in the subsequent years of this study, it was apparent that protection of wetland values was limited by the continual access to the fenced wetland by cattle as part of land tenure arrangements with adjacent land holders. The effort of installing fences, building earth ramps for fauna movement and pig baited trapping needs to be also supported by changes to the land tenure arrangement, to ensure that the values and services aspired to here are reached for the investment in this sensitive wetland habitat.

- Maintenance of the fence will continue to be challenging and will require a long term commitment from state government and/or the Burnett Mary River Group to cover these costs. Routine inspection of fences and completing general maintenance as necessary will preserve the investment, and ensure the supporting infrastructure is in working order. More trials like the earth ramp are needed to build a knowledge base towards a design that is most effective for arthropod movement over fences.

- The value of these wetlands with low frequency of tidal connection presently offer little opportunity for fish species. The fact that only a small number of species were caught here supports this contention, where more fish might be initially able to access wetlands during summer rainfall or very high tides, however, dry over the coming months depending on further tides and rainfall. During these times water quality would deteriorate, and fish would be exposed to predation by water birds. While their value as fish habitat might be limited under current climatic conditions, regarding whether fencing beginning present or not, further sea level rise might render these upper tidal areas more important for fish habitat in the future.

- Finally, conservation fencing for wetland values protection is important and should be implemented in broader GBR catchments were the impact of feral pig damage is persistent and extensive. However, a clear understanding and determination of the values need to be set with stakeholders, to avoid situations where broader expectations are not met (for example, movement of terrestrial wildlife across wetland regions) and on-going expensive maintenance programs. Aerial shooting along with baited traps continue to be used in northern Australia, however, fences if designed and constructed to a high standard could result in a functioning and productive coastal wetland system for a much reduced cost. It is clear here that allowing cattle to enter fenced wetland should cease as this will only continue to impact on the broader value and services of the wetland – it is counterintuitive to continue permitting cattle access to wetlands fenced for feral pig control. Under a market mechanism scheme (e.g. blue carbon, or water quality markets) these coastal wetlands could become more valuable as part of climate change adaption, and could effectively earn more generated income compared to using these ecosystems for a late dry season cattle feed area.

1.0 INTRODUCTION

1.1 Coastal wetland values and services

Coastal wetlands (palustrine and lacustrine) located on floodplains away from riverine channels, or located high in the upper intertidal zone, support rich aquatic plant and fauna communities, and nutrients (Jiang et al. 2015). During high water levels, for example spring high tides or catchment freshwater flow, interconnecting riverine channels create a linking network of waterbodies that persist and eventually become disconnected and dry out (Datry et al. 2018). Aquatic organisms occupying these upper coastal wetlands face a shifting land-water margins, which results in wetlands supporting a non-random assortment of aquatic and semi-aquatic species (Arrington & Winemiller 2006; Pander et al. 2018). The duration, timing and frequency that off-channel wetlands sustain lateral connection to primary rivers is a determining factor in broader aquatic ecology and production (Galib et al. 2018; Hurd et al. 2016). In addition to connection, environmental conditions become important including water quality (Arthington et al. 2015, Godfrey et al. 2016), access to shelter to escape predation, and available food resources (Jardine et al. 2012). Efforts by managers to restore wetland ecosystem values is increasing (Waltham et al. 2020a), though access to data demonstrating success are limited, which becomes fundamental when attempting to assess biodiversity return for the funding invested by government or private sector markets (Weinstein and Litvin 2016, Waltham et al. 2019).

The value and services provided by coastal wetlands is well known and are among the most biodiversity ecosystems on the planet (Mitsch and Gosselink 1993, Gopal 2013, Weinstein and Litvin 2016, Schuerch et al. 2018). However because of their position along the coast they are also targeted for some level of modification, either for road transport, housing estates, heavy industry, agricultural production or recreation (Spalding 2007, Arkema et al. 2015, Agboola et al. 2016). This results in a complete loss of these ecosystems, while those remaining areas suffer some level of disturbance and a fragmented landscape (McKinney et al. 2009, Waltham and Sheaves 2015, Weber and Wolter 2017).

When assigning significance to a wetland or making management decisions about a wetland, it is important that decision makers take into account the full range of ecological, economic and social values a wetland provides (services and beneficiaries, see Figure 1). Historically, wetland management decisions have favoured either wetland conversion or management for a single ecosystem service such as water supply or fish nursery areas. As wetlands become scarcer and under more pressure, and as we develop a better understanding of the full range of values provided by them, the best options will increasingly involve managing wetlands for a broader array of services and in alignment with the wise use principles of the Ramsar Convention.

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 what values and services a wetland provides including its location, size, type and condition. Most wetlands may indeed 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, for example, bird habitat, fish habitat, water quality, hydrology, or cultural. Once the values are set and agreeed to by relevant stakeholders, avaiable management strategies can be implemented.

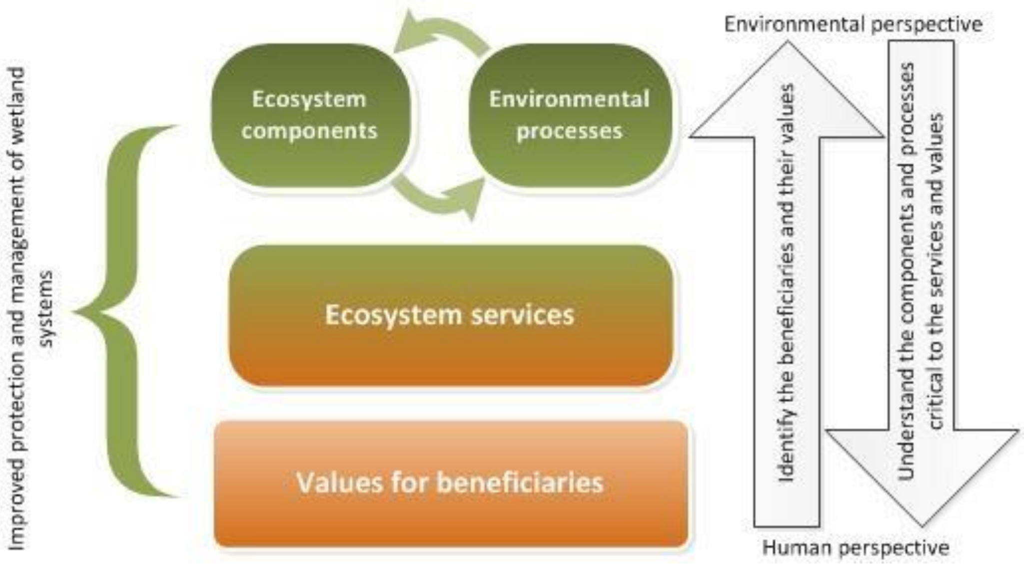


Figure 1. Conceptual framework for implementation projects following an ecosystem components and process approach to coastal wetland restoration (Source Wetland*Info*, Queensland Government).

1.2 Conservation fences

Conservation fences are a common tactic to ameliorate threatening processes from acting against individual species or for conservation of sensitive ecosystem habitats (Woodroffe et al. 2014). While conservation fences have been successful in achieving species or habitat protection (Woodroffe et al. 2014, Durant et al. 2015), they also have negative indirect effects on non-target species (Loarie et al. 2009, Rey et al. 2012), resulting in an ongoing conservation paradox for managers (Ferronato et al. 2014, Jakes et al. 2018). Emerging evidence suggests that fencing effects non-target species, for example, by disruption to dispersal processes, and increased mortality (via increased exposure to unfavourable conditions or predators; Spencer (2002)). By far these impacts are greatest on vagile animals which have evolved behavioural life history traits that allow them to inhabit landscapes characterised by spatial and temporal variability, and are therefore susceptible to limited access to resources or responding to local pressures (predation, climate conditions). However, with every conservation fence there exists the opportunity to evaluate the design efficacy, and implement supplementary modifications and improvements as part of a process of continual improvement (Loarie et al. 2009) towards maximising species conservation.

1.3 Feral pig disturbance in GBR catchments

Across northern Australia feral pigs (*Sus scrofa*) contribute wide-scale negative impact on wetland vegetation assemblages, water quality, biological communities and wider ecological processes (Fordham et al. 2008, Krull et al. 2013). Feral pigs have an omnivorous diet supported by foraging or digging plant roots, bulbs and other below-ground vegetation throughout terrestrial and wetland areas (Barrios-Garcia and Ballari 2012). This feeding strategy has a direct negative impact on wetland aquatic vegetation, which gives rise to soil erosion, benthic sediment resuspension, and reduced water clarity and eutrophication which becomes particularly critical late-dry season. Only a few studies have quantified the negative impacts feral pigs have on coastal wetlands (Mitchell and Mayer 1997, Doupe et al. 2010, Steward et al. 2018, Waltham and Schaffer 2018), limiting the ability of land managers to assess the consequences of feral pig destruction (Fordham et al. 2008), or indeed other large invasive species (Ens et al. 2016). Strategies focused on reducing or removing feral pigs from the landscape have been employed since their introduction to Australia (Fordham et al. 2006), including poison baiting, aerial shooting, and trapping using specially constructed mesh cages (Ross et al. 2017). Attempts to exclude feral pigs have also included building exclusion fencing for conservation outcomes by directly limiting access to essential resources (Nordberg et al. 2019). The installation of fences around wetlands has only been recently examined in Australia (Doupe et al. 2009b, Waltham and Schaffer 2018), suggesting fences may be less effective in providing freshwater refugia from pig damage, particularly in ephemeral wetlands that will dry out before the wet season rain reconnects them again with the primary rivers. The except to this however, is where wetlands are more permanent in the landscape, either supported by groundwater springs, that contain water permanently – fencing for the protection of the wetland services and values is therefore necessary (Figure 2). In addition, fences may prevent non-target terrestrial fauna from accessing wetlands which becomes particularly relevant late-dry season when wetlands function as regional water points for animals in the landscape (Waltham unpublished data). While small terrestrial species including birds, snakes and lizards are generally able to access fenced wetlands (Ross et al. 2017), access or escape from drying wetlands by other species, including macropods or freshwater turtles may be hindered (Waltham and Schaffer 2016). The inherent problem of wildlife fencing that focuses on a single target species (usually an invasive species) at local spatial scales needs further consideration (Jakes et al. 2018) as part of broader wildlife conservation and resource management strategies.

Figure 2. Conceptual representation in restoring and protecting wetland services and values. By fencing the wetland to remove the threat of feral pig damage to wetlands, it is hoped that flora and fauna, along with water quality will improve following this mitigation effort (Graphic prepared by J. Thomas, NESP Northern Australia Environmental Resources Hub)

1.3 Round Hill Reserve

In November 2016, the Burnett Mary Regional Group (BMRG) constructed an exclusion fence surrounding the Round Hill Reserve, a coastal wetland in the upper reaches of Round Hill creek, located in the Baffle catchment (Figure 4). The primary objective of this restoration activity was to control feral pig (*Sus scrofa*) damage directly to the wetland (Figure 3). Round Hill creek was identified as a restoration priority because it is an important Fish Habitat Area (FHA) and a Directory of Important Wetlands in Australia (DIWA) listed wetland (Queensland Government 2018b, a) (Figure 4).

Disturbance in Round Hill Reserve is largely attributed to feral pigs (*Sus scrofa*) and stock access, which can increase rates of erosion and sediment run-off to the GBR. Further negative consequences of feral pig foraging in wetland ecosystems include: damage to soil structure and habitat, destruction of wetland vegetation, spread of weeds and diseases, turtle and bird egg predation, and altered wetland ecosystem functioning (i.e. changes in aquatic macrophyte communities, increased wetland turbidity, prolonged anoxia, and pH imbalances; (Doupé et al. 2010).

Figure 3. Round Hill Reserve impacted by feral pigs before the installation of the fence (source S. Jackson).

1.4 Project objectives

Given the major threat that feral pigs pose to coastal wetlands and GBR catchments for example changing above and below ground vegetation, soil structure, wetland aquatic vegetation, water quality, and the processes including fish and waterbird reproduction and movement of fish, exclusion fences have been developed as a control strategy to reduce or eliminate the impact of this wetland treat. However, as with any restoration activity, evaluation of the mitigation efficacy is needed, and in the context of the components, processes and values changed and restored (see Figure 1).

The broader Round Hill Creek catchment is managed through multiple land titles (i.e. Queensland Government, Queensland Parks and Wildlife, and private landholder), and this project endeavours to provide the scientific information that these stakeholders need to continue effective management of this area. At the first stakeholder meeting with Queensland Parks and Wildlife, Burnett Mary NRM and private landholder, it was identified that the Round Hill Reserve held important biodiversity values with respect to flora and fauna. This wetland has important national and state conservation values, and therefore the feral pigs were considered to be compromising these services and values. Installing the fence around the most heavily feral pig impacted area was funded by Burnett Mary NRM and Queensland Government, as a strategy to protect the wetland services and values from further impact from feral pigs. The objective of this case study is to collect a series of field flora and fauna data to establish the services provided by Round Hill Reserve after the fencing, but to also sample nearby tidal wetland areas that remain impacted by feral pigs, but also cattle under historical land tenure arrangements between state and private landholders. These data join an increased understanding of the impacts that feral pigs, and cattle, have on wetlands (see Australian Government's Northern Australia Environmental Research Hub, https://www.nespnorthern.edu.au/projects/nesp/feral-animal-management/).

2.0 STUDY LOCATION

2.1 Study location and values evaluation approach

To evaluate wetland restoration efforts in the Round Hill Reserve, an array of wetland indicators have been assessed to track whether the services of this wetland are, or are likely, to return following the fencing investment. Broadly, the indicators evaluated change in the following wetland ecosystem components: water quality conditions, soil properties, vegetation, and biodiversity.

The study design consisted of measuring the same services at the Round Hill Reserve and multiple reference wetlands that are located nearby – the final sites were established at the first stakeholder meeting in mid-2017 with representatives from JCU, BMRG, and QPSW (Table 1, Figure 4). This provided additional context for evaluating the efficacy of pig exclusion fencing, and was necessary given the lack of baseline data available for making comparisons before and after fence construction more broadly in the GBR catchments. Sampling and surveying occurred at seasonal intervals (i.e. pre- and post-wet season) through-out March 2018 to July 2019.

Table 1. Restored and reference wetland location descriptions.

Code	Location	Management or Disturbance
RH1	Round Hill Reserve 24°13'55.35"S 151°49'55.45"E	Declared Fish Habitat Area (FHA) and listed as Nationally important wetland (DIWA). Adjacent to Eurimbula National Park. Feral pig exclusion fencing installed November 2016 (Queensland Parks and Wildlife, Burnett Mary Regional Group), however seasonal cattle grazing permitted.
RH2	Littabella Down reference 24°36'35.72"S 152°7'25.90"E	State-owned land with road causing tidal restriction, cattle grazing and feral pig disturbance.
RH3	Littabella Up reference 24°36'49.40"S 152°7'38.17"E	State-owned land with road causing tidal disconnection, cattle grazing and feral pig disturbance.
RH4	Round Hill reference 24°13'54.17"S 151°51'11.47"E	Privately-owned land with cattle grazing and feral pig disturbance.
RH5	Eurimbula NP reference 24°9'11.89"S 151°47'34.90"E	Located within Eurimbula National Park, with cattle grazing and feral pig disturbance.

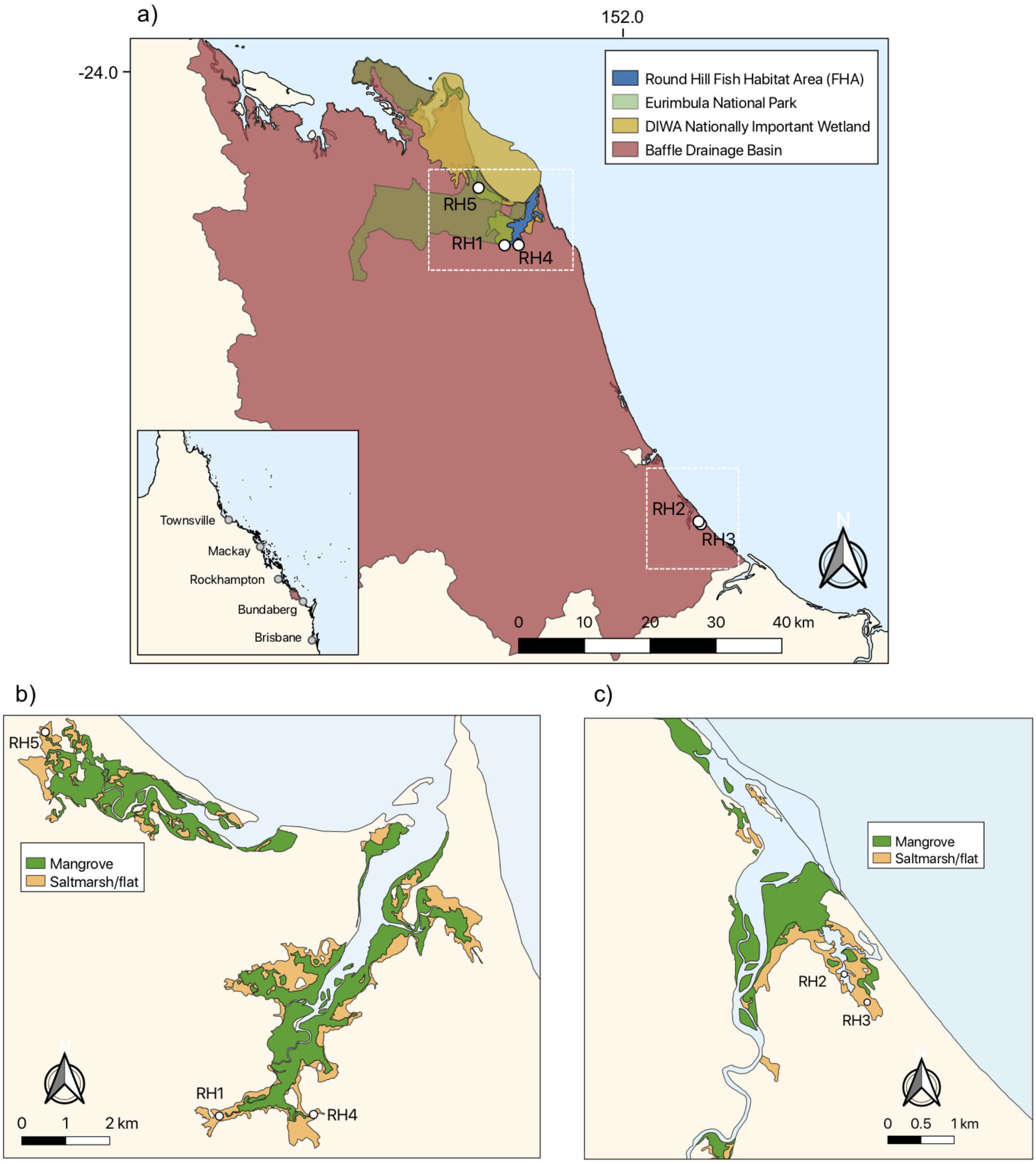


Figure 4. a) Baffle Drainage Basin (in red) and five survey locations: Round Hill Reserve (RH1), Littabella reference (RH2 & RH3), Round Hill reference (RH4), Eurimbula NP reference (RH5); b) Inset map of RH1, RH4, and RH5; c) Inset map of RH2 and RH3. Mangrove and saltmarsh spatial data layers sourced from: State of Queensland (Department of Environment and Science) (2019).

2.2 Restoration success indicators

2.2.1 Water quality

Water quality was measured at Round Hill Reserve and reference sites with a combination of high frequency loggers and grab samples at pre- and post- wet season survey intervals. This provided an understanding of water quality within the wetland. High frequency loggers were deployed in wetlands attachment to star pickets in Round Hill Reserve and both Littabella wetland sites. In addition, ambient water chemistry was measured with a calibrated multiprobe (Hydrolab Quanta) in addition to the light attenuation using a Secchi tube. Water samples were also collected by hand for measurement of nutrient concentrations and were analysed at the TropWATER Water Quality Laboratory, Townsville.

2.2.2 Vegetation

Vegetation change at Round Hill and reference sites were measured through a combination of vegetation mapping (via aerial satellite imagery and drone photographs) and on-the-ground vegetation surveys. On-the-ground vegetation surveys evaluated percent cover of vegetation and bare areas in quadrats deployed at low and high elevations.

In addition to vegetation change, the following saltmarsh soil characteristics were measured at low and high elevations: soil respiration rates, carbon storage, salinity, oxygen availability, and habitat for invertebrates. Soil respiration was measured with an oxygen probe-chamber apparatus, while salinity and oxygen availability were measured from soil samples taken with a soil corer. Carbon storage was measured by collecting the roots and rhizomes of saltmarsh (*Sporobolus virginicus*), placing them in mesh bags, and deploying mesh bags at the sediment surface and 10 cm below the soil surface for ~27 weeks (number of days deployed ranged between 189 and 194 at each site). Decomposition rates were measured over the 27-week period to provide an indication of saltmarsh carbon loss.

For all surveying/sampling procedures outlined above, five replicates were placed at ‘low’ and ‘high’ saltmarsh elevations along 3 transects at each site (total samples/site). The ‘low’ saltmarsh elevation was defined by the mangrove tree line and the ‘high’ saltmarsh elevation was defined by the terrestrial tree line at each site (Figure 5).

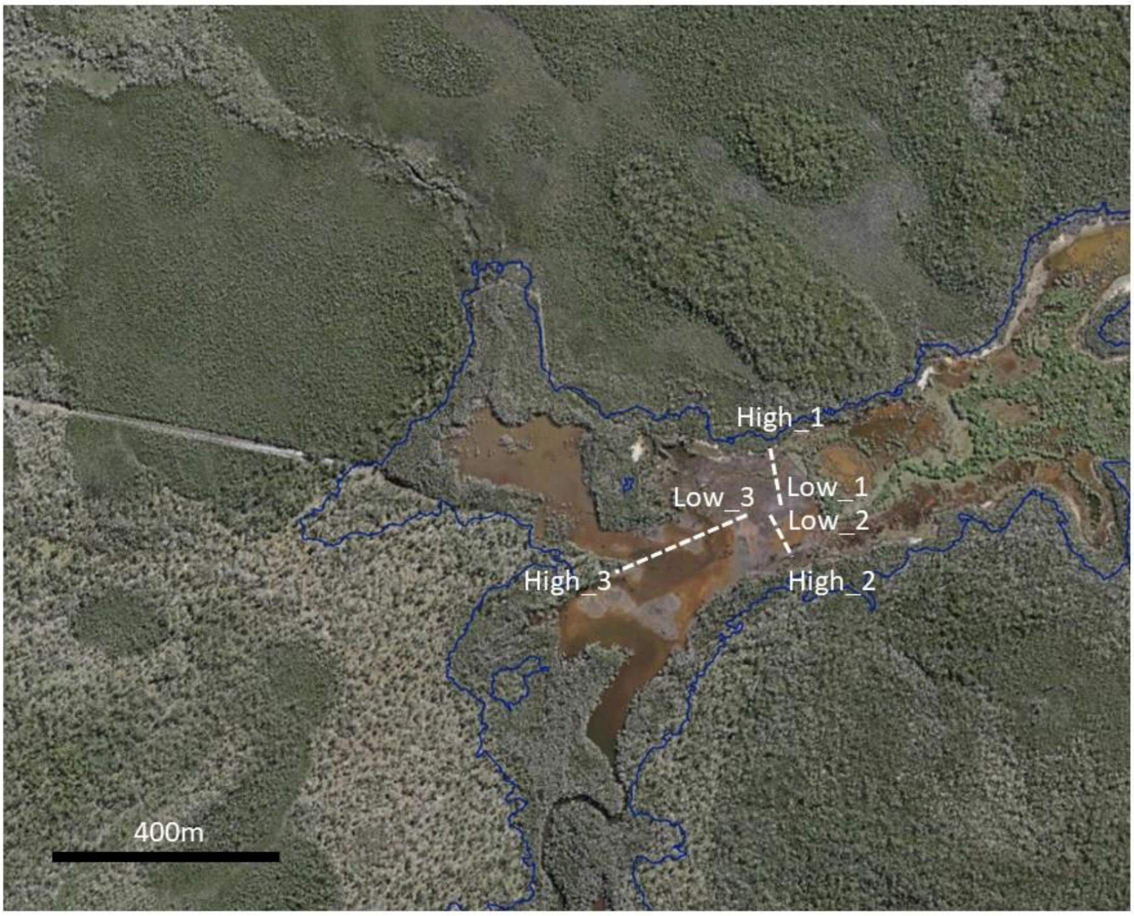


Figure 5. Example transect design for the Round Hill Reserve survey location. Three low elevation survey points were placed at the start of the mangrove tree line, and corresponding high elevation survey points were placed at the start of the terrestrial tree line. Five replicates at each survey point were spaced ~2-3 meters apart and in parallel with the tree line. Blue line is the Highest Astronomical Tide level (Queensland Government Globe spatial data).

2.2.3 Biodiversity

Feral pig control can maintain and improve wetland ecosystem functioning for species survival and diversity. Biodiversity at Round Hill and reference study sites was measured across several taxa including: herpetofauna, fish and bird assemblages. Sampling methods are detailed below.

3.0 VALUES AND SERVICES

3.1 Carbon dynamics: soil properties and saltmarsh decomposition

3.1.1 Introduction

Saltmarsh ecosystems provide a range services to coastal ecosystems including nursery habitat and feeding opportunity for a range of wildlife species (Kneib 1984a, Minello and Zimmerman 1992, Austen and Warwick 1995, Connolly et al. 1997). For example, fish visit flooded marsh areas to access food, but this feeding window is limited to the duration and extent of inundation (Minello 2000, Hollingsworth and Connolly 2006, Baker et al. 2020). When not inundated with tidal water, saltmarsh areas provide habitat for birds as an rich area for feeding (Shriver et al. 2004), as well as many invertebrate species that occupy both saltmarsh vegetated and unvegetated areas (Kneib 1984b, Evin and Talley 2002). Saltmarsh is also recognised globally for its ability to sequester carbon (Rogers et al. 2019). This accumulation effectively occurs in two complementary ways (components); 1) ability for saltmarshes to maintain high rates of primary production and low soil respiration (Mitsch and Gosselink 1993, Mitsch et al. 2013); and 2) these plants slow water velocity and facilitate suspended organic matter deposition on the surface (Mcleod et al. 2011). Once bound in saltmarsh plants or deposited and accumulating on the surface of these wetlands, the carbon accumulates when waterlogged and anoxic soil conditions limit microbe-driven carbon respiration belowground at a rate that is less than carbon loss (processes). Additionality of carbon in saltmarsh ecosystems is collectively coined '*blue carbon*' (Nellemann et al. 2009, Lovelock and Duarte 2019), and while important as part of future climate adaption (Alongi 2018), these ecosystems are at direct risk from numerous pressures from industrial and urban expansion (Macreadie et al. 2017a). Other pressures that may not be obvious on saltmarsh areas include physical disturbance by both domestic and feral animals (Meyer et al. 1995, Laffaille et al. 2000, Davidson et al. 2017, Persico et al. 2017). Physical disturbances from pig (*Sus scrofa*) and cattle (*Bos Taurus*) pugging can range from reducing vertical accretion of marsh surfaces because of trampling and compaction of soils, alter temperature dynamics, in addition to changing the physical properties of the soils (Smith and Odum 1981, Daehler and Strong 1995, Persico et al. 2017, Mason et al. 2019).

The aims of this study were: i) assess the amount of carbon stored in saltmarsh sediments, ii) measure the rate of carbon decomposition in saltmarsh sediments; and iii) determine whether tidal position, subsurface depth and disturbance gradient affect the amount of carbon stored and/or the rate of carbon decomposition in saltmarsh sediment. These data provide a basis for understanding what the restrictions on feral animal access means for carbon storage potential in coastal tidal wetlands in north Queensland.

3.1.2 Methods

Site description

Five saltmarsh locations were selected, including the restoration location RH1 (Round Hill Reserve), and nearby four reference locations: RH2 (Littabella Down), RH3 (Littabella Up), RH4 (Round Hill), and RH5 (Eurimbula NP) (Table 1). Within each of these wetlands, three transects were allocated, starting at the low marsh and extending to the high marsh (high marsh is defined as the ecotone transition between saltmarsh and terrestrial vegetation, that is only inundated during spring high tides; low marsh is the area more frequently inundated representing the transition between mangroves to saltmarsh). This resulted in three low and three high marsh sites within each of the five wetlands. At each low and high marsh positions, five replicate sites were chosen for this study, giving a total of 60 low marsh and 60 high marsh sites across the five wetlands.

Soil temperature

Soil temperature loggers (Hobo UA-001-64 Pendant Temp, Onset Computer Corporation, Massachusetts, USA) were deployed in saltmarsh sediments from July 2018 to January 2019. For each location, paired loggers were placed on the soil surface ('surface') and buried 10 cm in the sediment ('bottom'). Soil temperature was logged every 20 minutes for the duration of the bag deployment. Water temperature was concurrently measured in the estuary adjacent to the Littabella wetlands over the same deployment interval at RH2 and RH3.

Sediment moisture content and organic matter content

Three cores were extracted from within each wetland from each of three low and high marsh positions, giving a total of 30 cores across the five wetlands. A PVC corer (ID 50 mm) was used to extract core samples. Sediment cores was pushed manually into the surface until failure, and then gently removed to produce the sectioned sediment core. Each core was then split into 5 cm increment slices for the determination of moisture content and organic matter content back in the laboratory. Wet sediment slices were initially weighed on a 2 decimal place balance. Sediments were then oven dried at 60 °C for 48 h to a constant moisture content and reweighed. Moisture content was calculated as:

$$MC\ (\%) = \left(\frac{wet\ (g) - dry\ (g)}{wet\ (g)}\right) \times 100 \qquad \text{[Equation 1]}$$

Where: MC% = percent moisture content of the sediment
wet (g) = wet weight in grams
dry (g) = dry weight in grams

Oven dried sediment was ground in a mortar and pestle to break up peds then 2 g was weighed into porcelain crucibles and combusted in a muffle furnace at 525 °C for 3 h. The initial and final sediment weights were used to calculate percent organic matter content (%OM)

$$OM(\%) = \left(\frac{initial\ (g) - final\ (g)}{initial\ (g)}\right) \times 100 \qquad \text{[Equation 2]}$$

Where: OM(%) = percent organic matter in the sediment
initial (g) = initial weight in grams
final (g) = final weight in grams

Sediment electrical conductivity and pH

We prepared a 1:5 (w/v) sediment:water slurry to measure sediment electrical conductivity (mS cm^{-1}) and pH. Sediment:water slurries were sonicated for 10 min then shaken by hand at regular intervals for 1 hr. Electrical conductivity of the suspension was measured after allowing 30 minutes settling time with a handheld meter (WTW Cond 315 with TetraCon 325 sensor).

The suspension was then shaken by hand and the pH measured using a benchtop pH meter (Orion 3-Star pH benchtop with Thermo Ag/AgCl probe).

Carbon decomposition

Saltmarsh soil carbon storage potential was measured by collecting the roots and rhizomes of saltmarsh plants (*Sporobolus virginicus*). *S. virginicus* was collected from a coastal wetland near Townsville, by hand and upon returning to the laboratory, was washed to remove any soil or loose organic material/fragments, and dried for 48 hrs at 60 °C until constant weight. This species is the dominant in coastal wetlands in the region and was therefore considered the most approporiate choice to test the model of additionality (a more full study in the future could include more wetland species to determine overall carbon additionality). Carbon decomposition bags were constructed with fiberglass mesh (50 mm x 50 mm, 1 mm mesh) and contained approximately 200 g of the root and rhizome material (Figure 6). Each bag had a small piece of waterproof paper inserted with a unique bag code which assisted post decomposition weighing and data analysis. Bags were deployed at six sites (3 high marsh, 3 low marsh) within each of the five survey locations (RH1, RH2, RH3, RH4, RH5). A total of 150 decomposition bags were deployed across the saltmarsh locations. At each high marsh and low marsh site, five decomposition bags were deployed at the surface, and five were deployed directly below the surface bag at 10 cm depth. Surface bags were tethered using fine monofilament fishing line to metal wires that were inserted into the sediment to stop the bags from being dispersed. The bags were deployed parallel with the low or high tide line, and spaced ~2-3 m apart. Decomposition bags were left *in situ* for 20 weeks. At the end of the deployment period the bags were collected, and returned to the laboratory where they were gently washed to remove any soil or animals, dried for 48 hrs at 60 °C, and weighed. The rate of decomposition over the period was calculated to provide an indication of saltmarsh carbon loss with the following equation:

$$k = \left(\frac{W_{initial} - W_{final}}{W_{initial}}\right) \times \frac{1000}{t} \qquad \text{[Equation 3]}$$

Where:

k = decomposition rate (mg g^{-1} d^{-1})
W_{final} = Final weight (g)
$W_{initial}$ = Initial weight (g)
t = number of days

As we know organic decomposition follows an exponential decay:

$$W_t = W_0 e^{-kt} \qquad \text{[Equation 4]}$$

solved for k:

$$k = \frac{\log\left(\frac{W_{final}}{W_{initial}}\right)}{t} \qquad \text{[Equation 5]}$$

Where:

k = decomposition constant (d^{-1})

Figure 6. Example of surface deployed decomposition bag at a high marsh site. The bag was tethered to an aluminum spike in wetland.

Data analysis

Statistical analysis was performed in R (R Core Team 2018). Permutational analysis of variance (PERMANOVA) on rates of carbon decomposition was performed with the Vegan package in R (Anderson 2001, Oksanen et al. 2019). Analysis of multivariate homogeneity of group dispersions was conducted prior to running the PERMANOVA model. A constant was added to the non-diagonal dissimilarities such that all eigenvalues were non-negative in the underlying principal co-ordinates analysis (Method 1; Legendre and Anderson 1999). Pseudo-F values were produced from a model run of 9999 permutations constrained by site. Significance for each term was assessed sequentially from first to last.

3.1.3 Results and Discussion

Physical drivers

Air temperatures at the Seventeen Seventy meteorological station (Station 039314) were measured in the range of ~ 10 to 30 °C during the study period (Figure 7). There was a general warming trend over the 6 months from July (winter) to January (summer) due to expected seasonal changes. There were numerous rainfall events throughout the study period (Figure 7). The estuaries of the region are semi-diurnal macrotidal tide-dominated systems (Figure 8). Water temperature measured at Littabella estuary ranged from ~ 5 to 40 °C and followed a similar warming trend to air temperature due to season (Figure 9). Short-term declines in water temperatures occurred around rainfall events in October and December 2018.

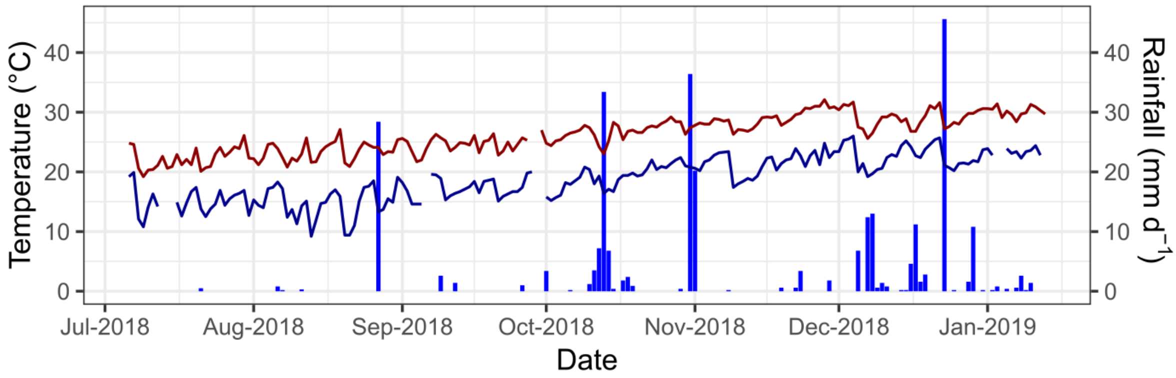


Figure 7. Observed daily temperature maximum (red), minimum (blue) and rainfall (bar) at Seventeen Seventy (Station 039314). Source: Bureau of Meteorology.

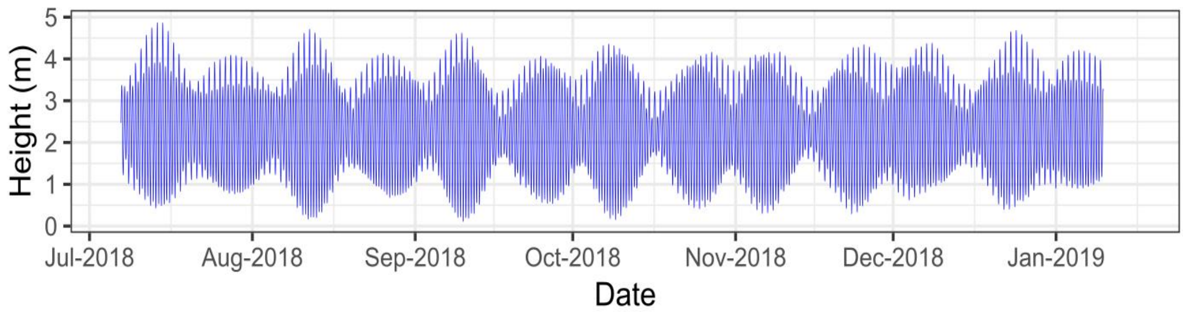


Figure 8. Tides recorded at the Gladstone gauge (Auckland Point 052027A), 25km northwest of the saltmarsh sites. Source: Transport and Main Roads, Queensland.

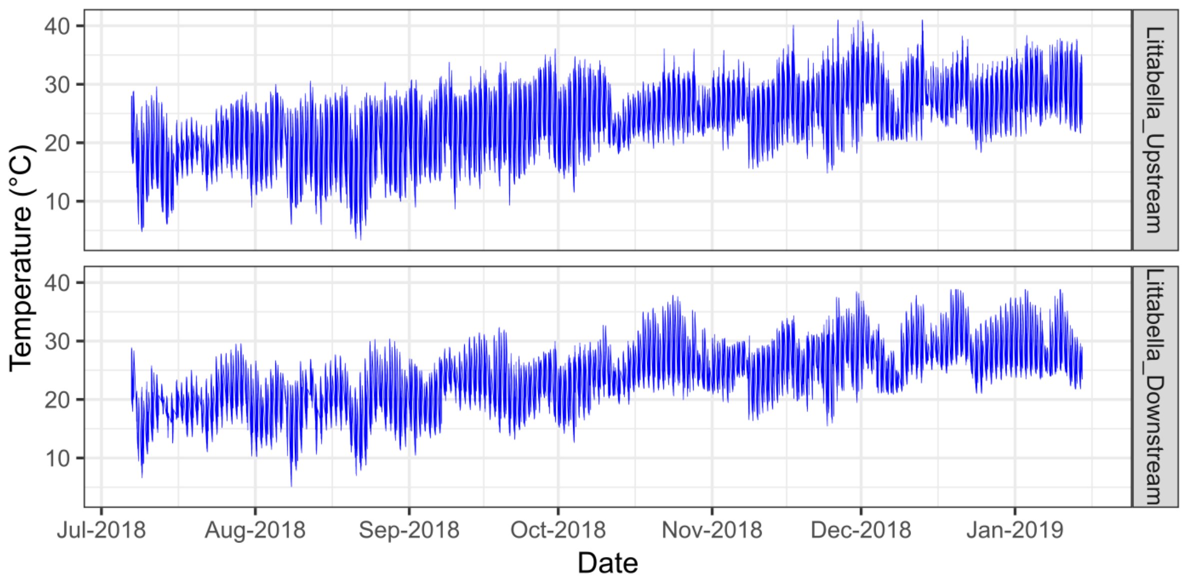


Figure 9. Water temperature measured in Littabella estuary at the upstream (RH3) and downstream (RH2) locations

Sediment temperature

Sediment temperature was recorded at 20 min intervals over the duration of the study on the surface and 10 cm below (Figure 10). Sediment temperature was logged from July 2018 to January 2019. Temperature loggers deployed at the surface at RH2 and RH4 stopped early (December 2018) for unknown reasons, all other loggers successfully recorded data for the entire period. Sediment temperature was higher at the surface than at depth, with diel variations also higher at the surface than at depth. Sediment temperature increased over the study period at rates of between 0.06 to 0.09 °C day^{-1} (approximated by linear model) in response to the seasonal transition from winter (July) to summer (Jan). Sediment temperatures were primarily coupled with air temperature, although temperature was also influenced by tidal inundation cycles. Delta-T values were above zero indicating that instantaneous sediment temperature was warmer at the surface than at depth (Figure 11). Short periods where delta-T was < 0 were attributed to tidal influence, and/or the early morning prior to sunrise.

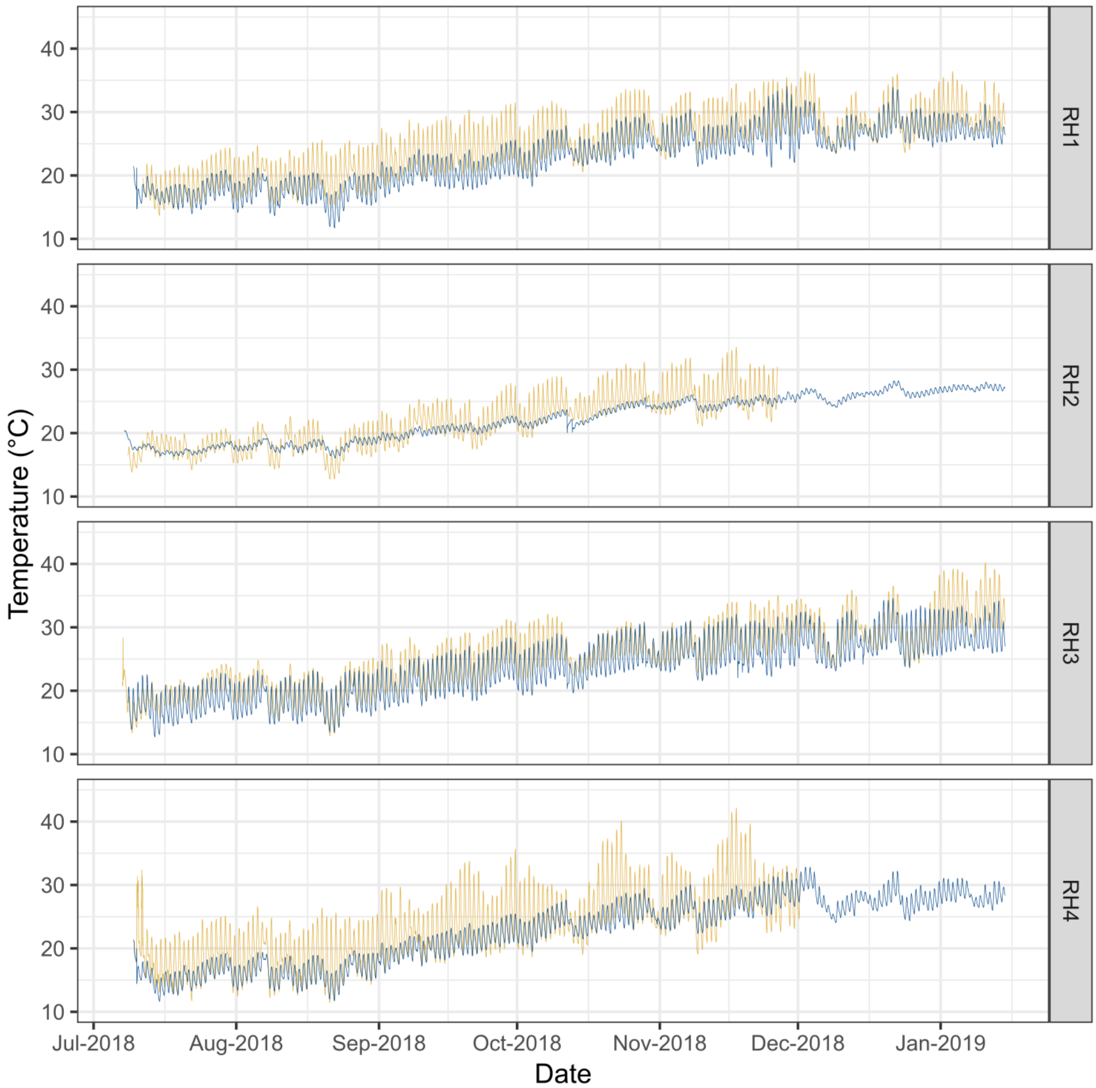


Figure 10. Saltmarsh sediment temperature measured at surface (yellow) and bottom (blue) at Round Hill Reserve (RH1, RH4) and Littabella Estuary (RH2, RH3). Temperature records for surface loggers at RH2 and RH4 discontinue beyond November 2018 due to logger malfunction.

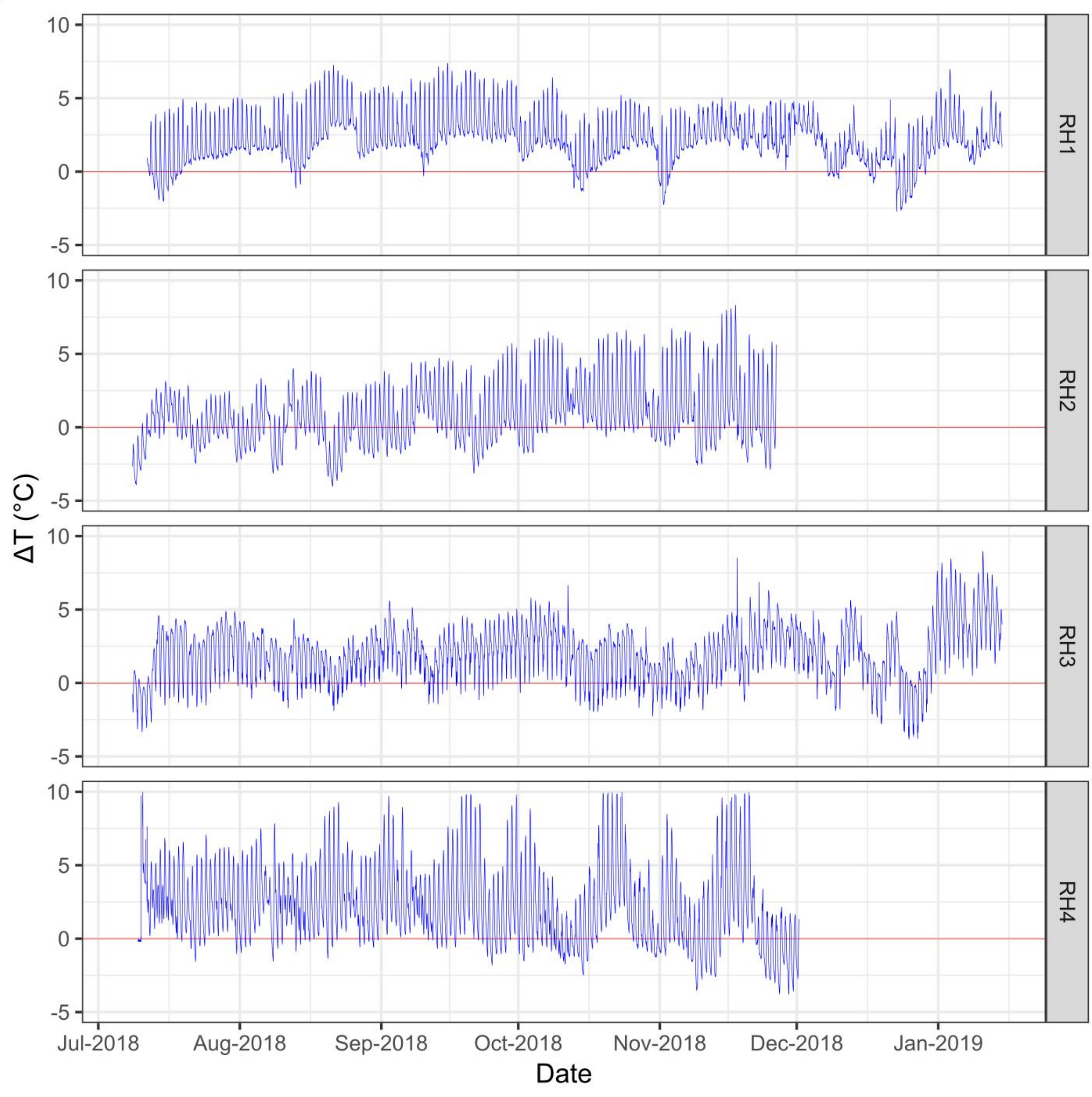


Figure 11. Difference in temperature (ΔT) between surface and bottom sediments at Round Hill Reserve (RH1, RH4) and Littabella Estuary (RH2, RH3).

Soil carbon profiles

Soil cores were recovered to a maximum depth of 115 cm, with generally deeper cores possible for ‘low’ compared to ‘high’ marsh sites at all wetlands. Soil moisture content ranged from 10 to 70 %. Moisture content was highest towards the sediment surface and generally decreased to a minimum at ~20 cm depth before either remaining stable (RH1, RH2, RH3) or increasing again with depth (RH4, RH5) (Figure 12). Loss on ignition indicated that there was relatively higher organic matter content in the soil surface compared to soil recovered much deeper in cores (Figure 13). We attribute this to shallow rooted saltmarsh vegetation and/or detrital organic matter, whereas the odd peak of organic matter content at depth was due to inclusion of woody roots. The depth profiles may also indicate carbon decomposition following deposition and subsequent diagenesis. There was a log-linear relationship between soil loss on ignition and moisture content (Figure 14), indicating that soils with high organic matter content retain more water. Soil electrical conductivity ranged from 1.1 to 48.9 mS cm^{-1} (Figure 15). At Round Hill (RH1) the soil electrical conductivity was highest at the surface, then diminished to a stable level for the remainder of the depth profile. Low marsh sites at RH1 had higher EC than high marsh sites. All other locations had higher soil EC at high marsh sites. Soil pH ranged from 2.3 to 7.6 with median value of 5.3 (Figure 16). Soils were generally more acidic at depth at RH4 and RH5 at low marsh sites.

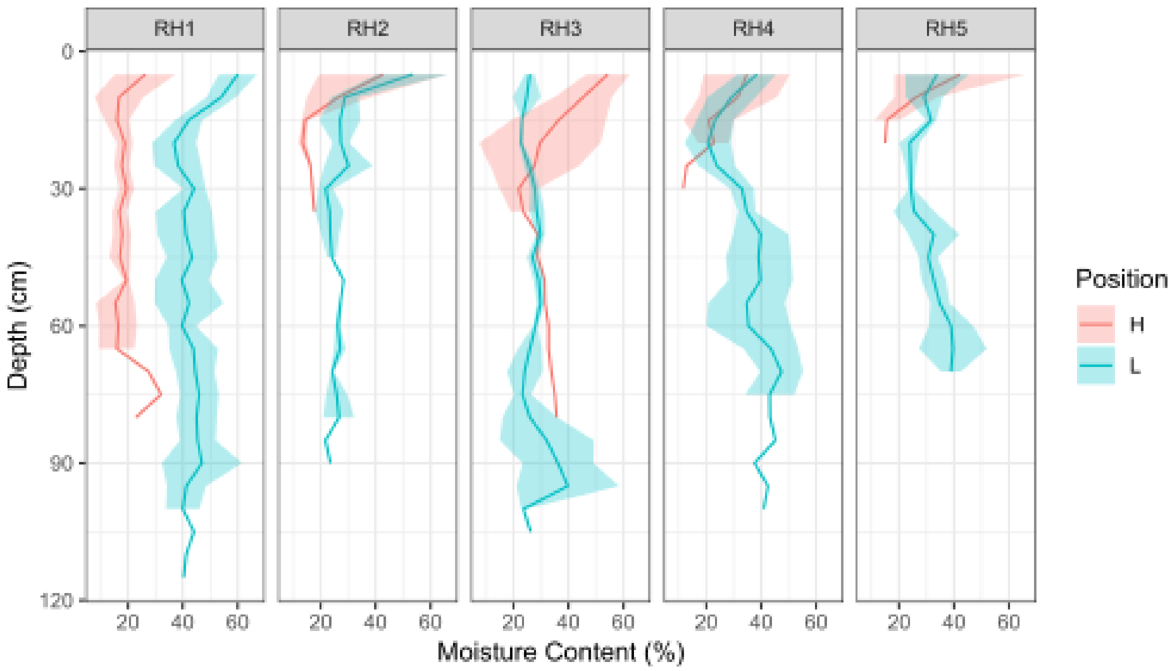


Figure 12. Mean moisture content (% w/w) of soils collected from high (H) and low (L) marsh sites at Round Hill Saltmarsh sites RH1 to RH5. Coloured shading denotes one standard deviation from mean (n = 3).

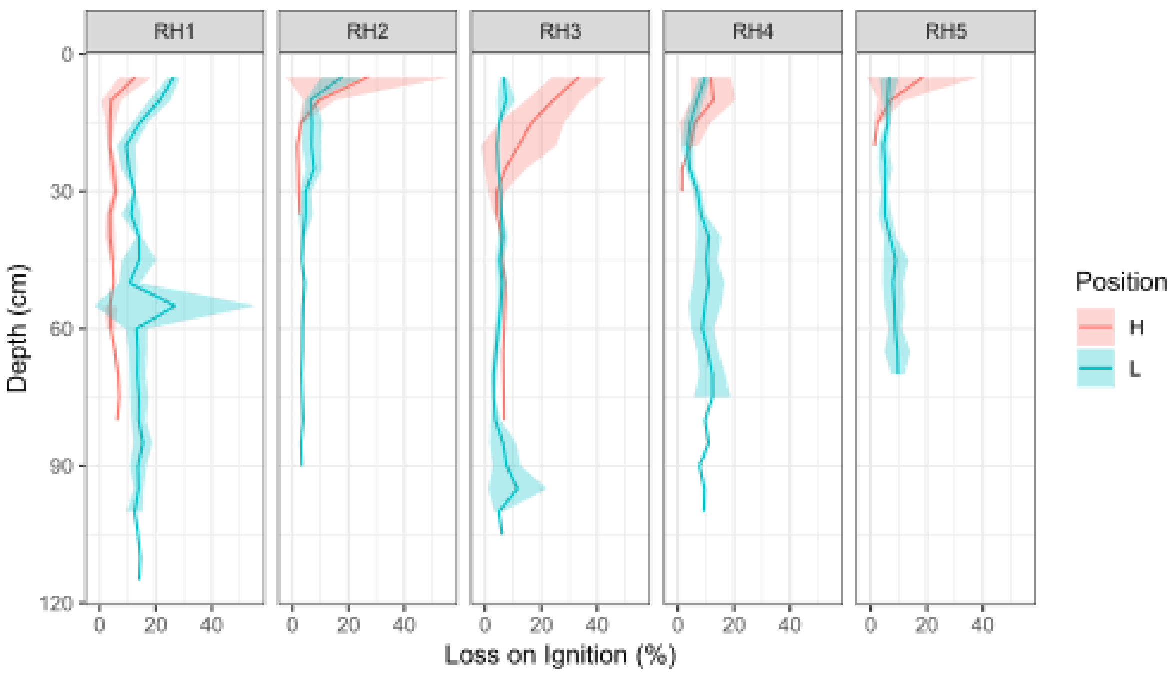


Figure 13. Mean loss on ignition (% w/w) of soils collected from high (H) and low (L) positions at Round Hill Saltmarsh sites RH1 to RH5. Coloured shading denotes one standard deviation from mean (n = 3).

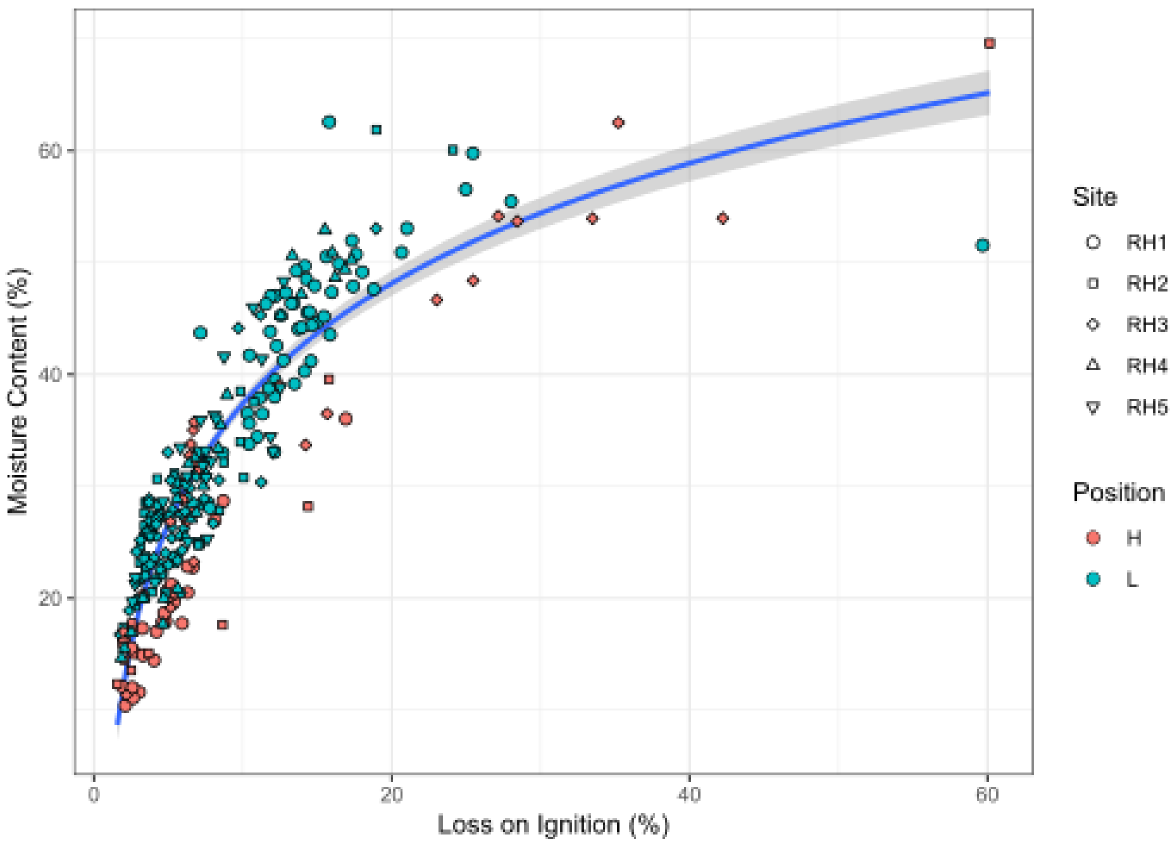


Figure 14. Round Hill saltmarsh soil moisture content increases with organic matter content (as determined by loss on ignition). Blue line indicates the linear model fitted to all data with formula y = log(x), adj-R^2 = 0.55, P < 0.001. Grey shading represents the 95% confidence interval.

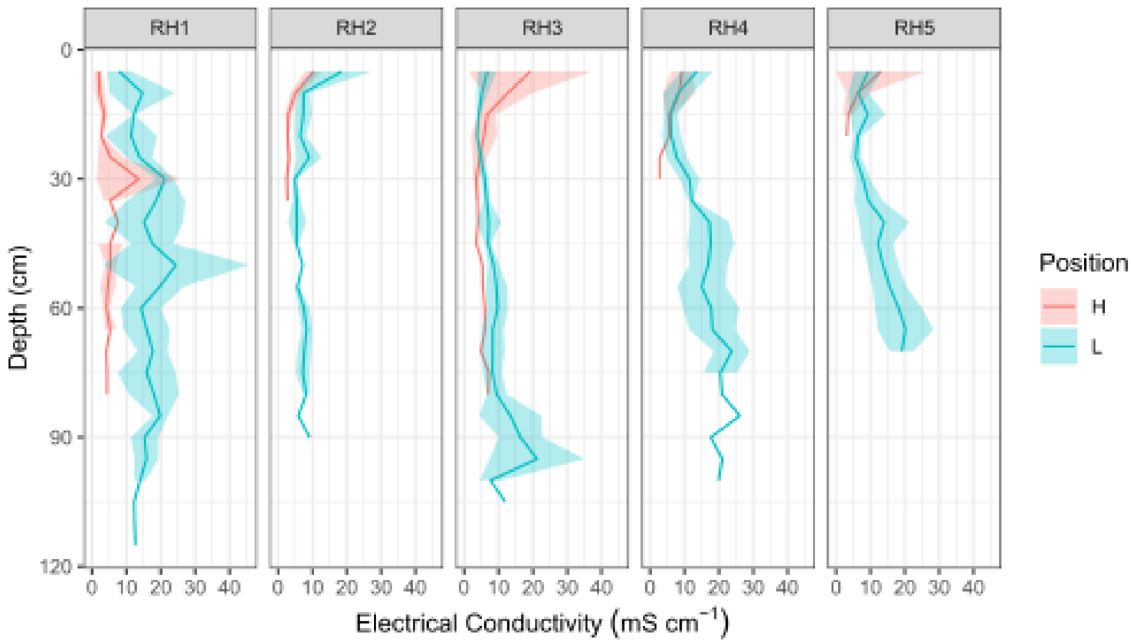


Figure 15. Mean electrical conductivity (mS cm^{-1}) of soils collected from high (H) and low (L) marsh sites at Round Hill Saltmarsh sites RH1 to RH5. Coloured shading denotes one standard deviation from mean (n = 3).

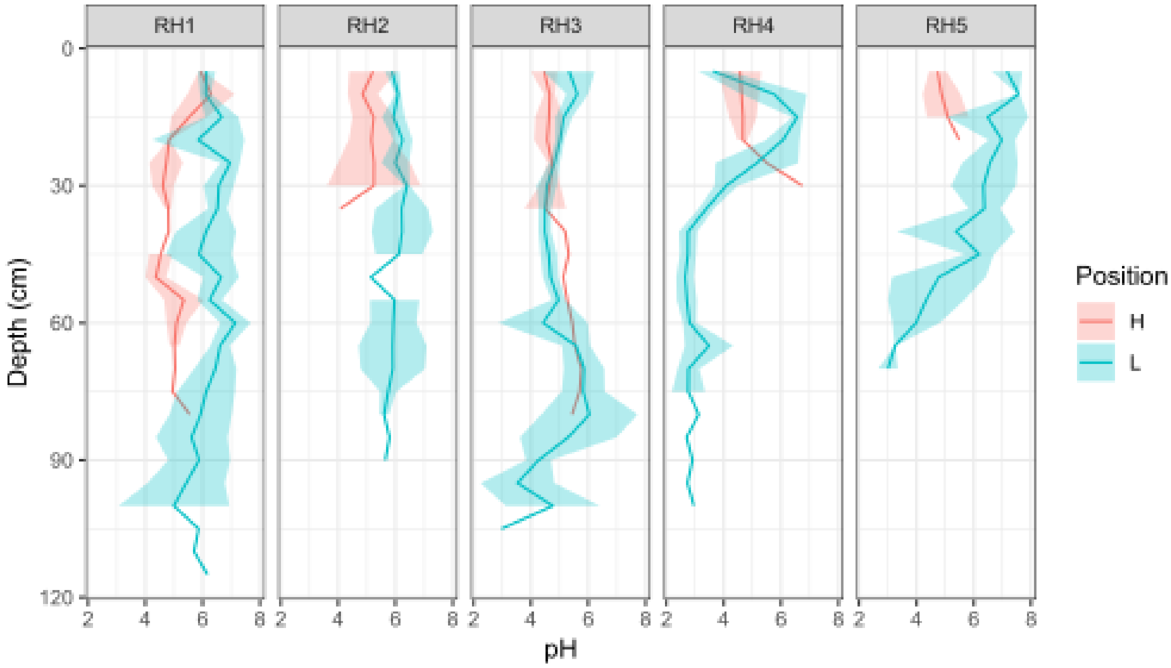


Figure 16. Mean pH of soils collected from high (H) and low (L) positions at Round Hill Saltmarsh sites RH1 to RH5. Coloured shading denotes one standard deviation from mean (n = 3).

Carbon stock estimates

Carbon stock was calculated from LOI values for the top 30 cm of saltmarsh sediments (Figure 17). Assuming OC = LOI × 0.58 (the so called van Bemmelen factor), carbon stock ranged 13.7 – 72.7 Mg OC ha^{-1} with a mean value of 38.7 Mg OC ha^{-1}. Two-way analysis of variance was used to compare carbon stock between marsh position and saltmarsh site. Carbon stocks were significantly larger in the low marsh than the high marsh position (ANOVA: $F_{1,20}$ = 12.8, P = 0.002), and there was a significant difference in carbon stock between the saltmarsh sites (ANOVA: $F_{4,20}$ = 8.6, P < 0.001). There was a significant interaction between saltmarsh position and site (ANOVA; $F_{4,20}$, P = 0.045).

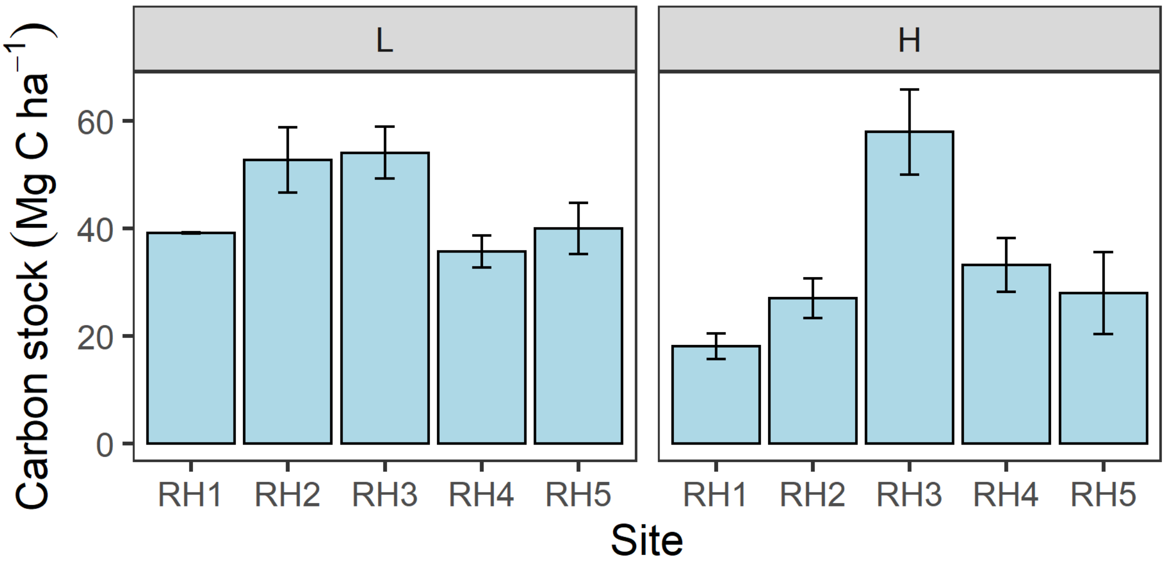


Figure 17.Carbon stock (mg C ha^{-1}) in the top 30 cm of sediment calculated at two positions (low marsh and high marsh) across five saltmarsh wetlands. Values are mean with error bars indicating one standard error from the mean (n = 3).

Carbon decomposition

The percent of deployed decomposition bag that were recovered after 20 weeks is shown in Figure 18. Significantly less decomposition bags were recovered from the high marsh areas than the low marsh (two-way ANOVA: $F_{1,56}$ = 15.5, P < 0.001), while significantly more decomposition bags were recovered from the 'deep' substrate position than those deployed on the surface (two-way ANOVA: $F_{1,56}$ = 6.1, P = 0.017). There was no interaction between marsh position and substrate position. The recovery rate of surface litter bags was between 6.7% and 93.3% for high marsh compared to 46.7 – 100 % for low marsh. Generally the percent recovery for high marsh was poor (median = 32%), with the exception of RH4 (93.3%). Cattle stock are generally not permitted access to the RH4 wetland site by the property owner, rather they are kept to the adjoining lots. The recovery of decomposition bags from the fenced wetland in Round Hill Reserve (RH1) were similar to the reference wetland sites where cattle freely access them. We attribute this to landholders intermittently allowing cattle into the fenced wetland to feed when vegetation conditions are favourable. Interestingly the recovery rate of surface litter bags from the low marsh areas, on the mangrove/saltmarsh boundary, were similar among wetlands, with the exception being RH2 which had around 50% surface return rate. For the decomposition bags that were buried in the sediments, most were recovered, however, at RH2 the return rate was lowest at both high and low marsh sites – a likely reason for missing bags could be because the buried bags where tethered to a wire rod, which was also tethered to surface bags which when disturbed by cattle removed the rod, hence making it difficult to find all the buried bags.

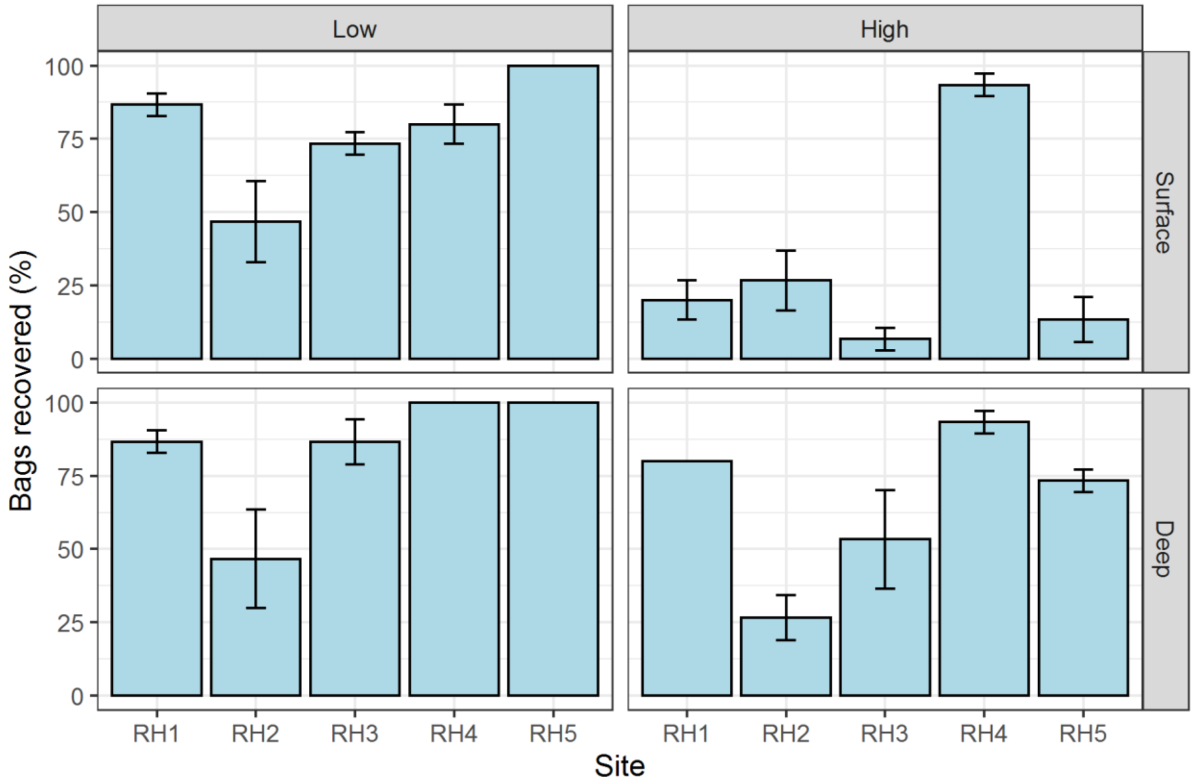


Figure 18.Decomposition bag recover percentage for high and low marsh, on the surface and buried at depth. Shown are the mean and one standard error from the mean (n = 5)

The mean daily carbon decomposition rate was 2.01 ± 0.77 mg g^{-1} d^{-1} (Range: 0.19 to 4.44 mg g^{-1} d^{-1}) across the study region (Figure 19). Carbon decomposition rates were higher at high marsh sites compared to the low marsh sites (PERMANOVA: $F_{(1,192)}$ = 28.27, P = 0.0001). Carbon decomposition rates were significantly higher at the 'Bottom' substrate position compared to the 'Surface' (PERMANOVA: $F_{(1,221)}$ = 41.59, P = 0.0001) (Table 2). There was a significant interaction between substrate position and tidal position (PERMANOVA: $F_{(1,221)}$ = 6.58, P = 0.0123). Substrate position (depth) gave the strongest effect.

Table 2. Permutational test for analysis of variance (PERMANOVA) of the rate of carbon decomposition in saltmarsh soils.

Source	***df***	***SS***	***R²***	***F***	***Pr (>F)***
Tidal Position	1	12.272	0.10651	28.272	**0.0001**
Substrate Position	1	18.052	0.15667	41.585	**0.0001**
Tidal Position * Substrate Position	1	2.858	0.0248	6.584	**0.0123**
Residual	189	82.042	0.71202		
Total	192	115.224	1		

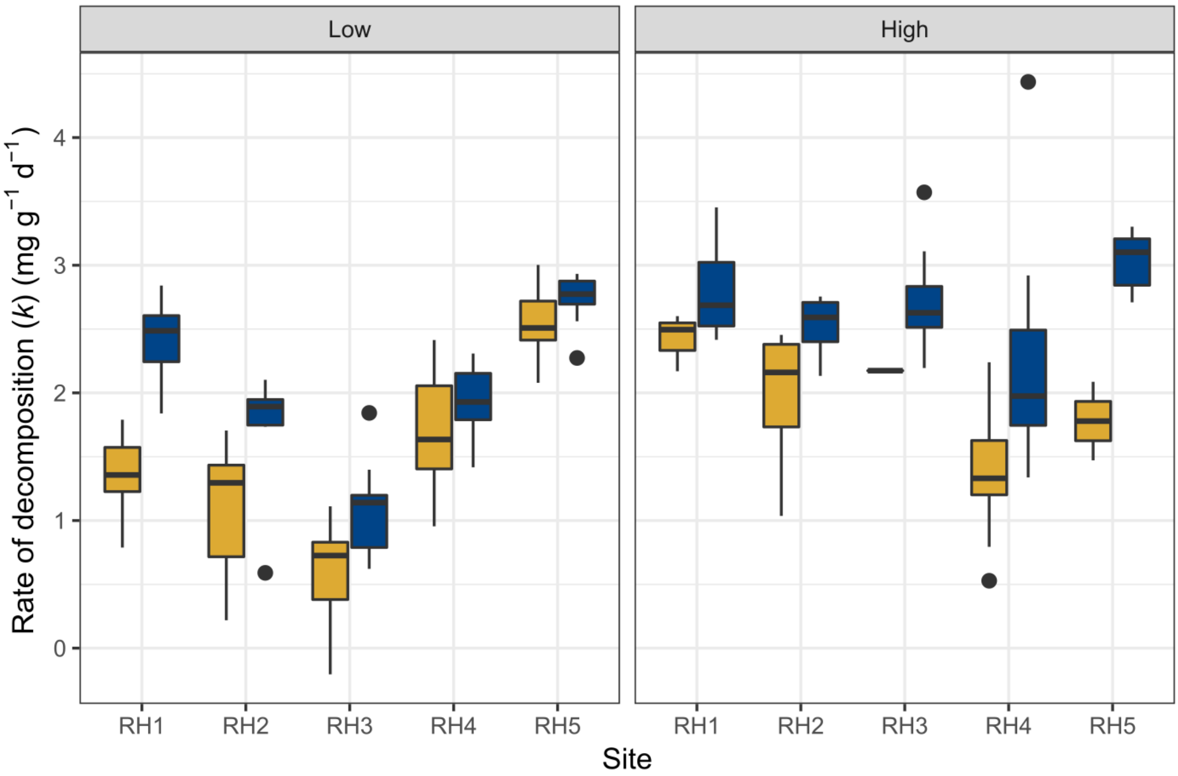


Figure 19. Carbon decomposition rates for Surface (orange) and Bottom (blue) substrate positions at low and high marsh positions.

i) Carbon stored in saltmarsh sediments

Saltmarsh wetlands are increasingly recognised for their carbon storage ecosystem service along with other 'blue carbon' coastal ecosystems such as mangroves and seagrass (Mcleod et al. 2011, Macreadie et al. 2017b, Lovelock and Duarte 2019). Yet saltmarsh ecosystems across recent decadal-century scale have been under pressure from both land-use practices and anthropogenic climate change driven sea level rise (Kelleway et al. 2017, Rogers et al. 2019). The value of this ecosystem service at a global scale is highly relevant (Beaumont et al. 2014). Hence, increasing our understanding of the carbon stock of saltmarsh wetlands such as those throughout the Round Hill area in southeast Queensland is important. Here in this study we have estimated organic carbon content from sediment loss on ignition in lieu of direct measurement of soil organic carbon by elemental analysis. We acknowledge that a simple LOI-OC conversion factor may not accurately represent OC content, and that a conversion factor of 0.58 is likely imprecise (Pribyl 2010) and leads to an underestimation of blue carbon stock (Ouyang and Lee 2020). Hence the results from this study should be interpreted in this light. We estimated organic carbon content of the top 30 cm to be ~38 Mg OC ha^{-1}, which is comparable to saltmarsh wetlands across Australia where carbon stock in top 30 cm ranges 43 – 117 Mg OC ha^{-1} (Kelleway et al. 2016, Macreadie et al. 2017a). Landscape position within the saltmarsh may affect carbon stock either due to vegetation association (Ford et al. 2019) or topography and tidal inundation (Brown et al. 2016). For example we showed that carbon decomposition is slower and carbon stock is higher at 'low' marsh sites (Figure 20), and attribute this to more frequent inundation maintaining anoxic conditions (Wang et al. 2019). This slowing of decomposition allows for the accretion and long-term storage of carbon deeper in the sediment column.

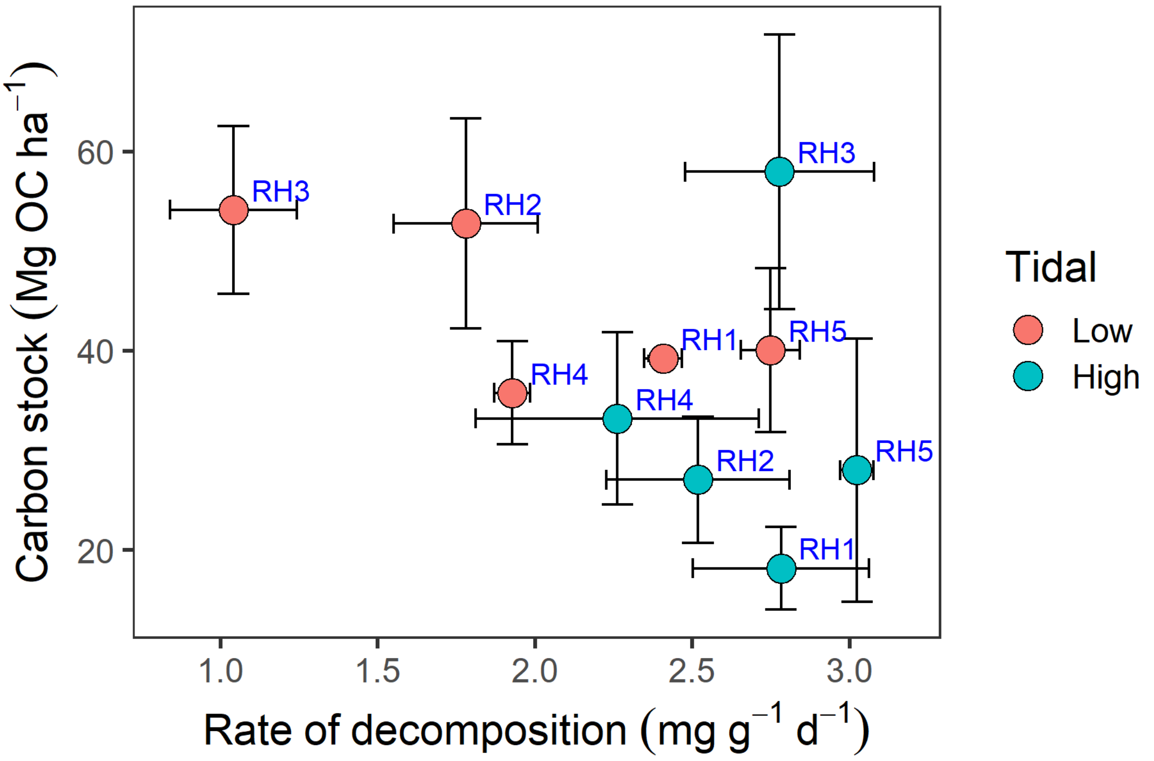


Figure 20. Relationship between rate of decomposition and carbon stocks in Roundhill saltmarsh sediments in low and high tidal positions. Shown are mean values and standard deviation (n = 3).

ii) Rate of organic matter decomposition in saltmarsh sediments

Organic material in coastal wetlands decomposes in response to leaching and microbial degradation (Valiela et al. 1985). In this study the organic matter decomposition rate was measured on and within sediment across a range of saltmarsh wetlands along a disturbance gradient. The decomposition rates measured in this study were almost double the rates measured in Florida saltmarsh (Persico et al. 2017). The higher rates in our system could not be explained by soil temperature as our measured soil temperatures were approximately 10 °C cooler than in the Florida study. A common problem with wetlands in the GBR catchment is the increased N loading due to agricultural runoff (Waterhouse et al. 2012, Brodie et al. 2013, Brodie et al. 2016). Increasing nitrate in saltmarsh increases the rate of organic matter decomposition and hence reduces the amount of C stored in sediment (Bulseco et al. 2019). This may explain why we recorded higher rates here but would need to be more formally tested. For example, this would need to be tested against other know factors which affect organic matter decomposition rates in intertidal systems such as inundation regime and salinity (Wang et al. 2019), bioturbation by crustaceans (Wang et al. 2010, Fanjul et al. 2015) and other biota (Kostka et al. 2002). Furthermore, we would expect to see a reduction in organic matter being retained by sediments where the removal of above-ground vegetation via grazing occurs.

Overall the rate of organic matter decomposition was generally highest when buried compared to surface deposited bags. The organic matter contained in the surface litter bags are exposed to photochemical and physical degradation, hence would likely undergo rapid leaching. Burial (accretion) of saltmarsh litter reduces the rate of organic matter decomposition, hence both increasing carbon stock and reducing CO_2 release to the atmosphere. Once burial commences (either experimentally, or by accretion), the organic carbon contained in the litter bags is no longer exposed.

iii) How tidal position, subsurface depth and disturbance gradient affect the amount of carbon stored and/or the rate of carbon decomposition in saltmarsh sediment.
The organic matter content (and hence carbon) stored in sediments was strongly related to subsurface depth, with shallow sediment containing more organic matter than at depth. There was generally steep decline in organic matter content in the first 30 cm with OM content then stabilising throughout remainder of the soil profile. This indicates important process of burial and sediment accretion, where much of the organic matter decomposition occurs in that first period of burial, with more labile forms of carbon first to leach and/or be metabolised and respired as CO_2 by sediment microbial activity. As organic matter ages with burial the content remaining is progressively more recalcitrant, as the labile forms are consumed and remineralisation becomes a more prominent process. With depth also comes anoxic conditions which are more favourable for the preservation and storage of organic matter.

Saltmarsh restoration has been shown to improve carbon stock via increased accumulation rates (Burden et al. 2019). Management interventions such as feral and domestic animal exclusion via fencing may lead to a recovery of vital ecosystem services such as carbon accretion. An important consideration in this study is the number of lost decomposition bags, with the highest of lost bags occurring at the surface, and in the wetlands that still allow cattle access. By this we assume that by the process of the cattle eating saltmarsh grass, they have also consumed the decomposition bags. This highlights the ineffectiveness of the current management practice where fencing is installed to exclude feral pigs and other wildlife while still allowing for cattle to access the wetland for seasonal grazing.

3.2 Vegetation extent and composition

3.2.1 Introduction

The use of remotely sensed imagery to map and monitor wetlands vegetation communities has numerous benefits as it allows for increased access to remote areas, and particularly resolves important safety concerns with operating in tropical northern Australia (Gray et al. 2018). It is considered to be more cost effective than utilising traditional field surveys (line intercept surveys or quadrats), as little to no labour is required. Images are rapidly captured, allowing managers to take snapshots of extensive environments. This has the added benefit of encouraging repeated sampling (Adam et al. 2009, Arroyo et al. 2010, Dronova 2015, Gray et al. 2018), and consistent information input into management by building libraries of wetland condition over time (Adam et al. 2009). Having historical imagery of wetlands, particularly pre-disturbance or before restoration effort, creates a starting baseline of reference for future comparison (Adam et al. 2009, Gray et al. 2018).

Despite advances in the spectral and spatial resolution of satellites remotely sensed imagery using satellites comes with wetland-specific challenges. Wetland vegetation remains more difficult to detect than its terrestrial counterparts (Adam et al. 2009) though this is dependent on degree of diversity in wetland vegetation (Zomer et al. 2009). Unmanned aerial vehicles (UAVs) can overcome some of these challenges in part through the higher spatial resolution afforded by this technology which increases the likelihood of features being identified (Arroyo et al. 2010, Zweig et al. 2015, Abeysinghe et al. 2019, Al-Najjar et al. 2019). Using UAVs to capture remotely sensed imagery has particular advantages, including the ability to plot the flight path to control for factors which may make image interpretation difficult for example sun glare or to account for the tides (Duffy et al. 2018). While image classification is now considered highly accurate, ground truthed information is also required to assess classification accuracy (Zomer et al. 2009) which can be collected concurrently with UAV flights reducing field time.

The user-controlled nature of UAVs permits revisit time for sites to be increased beyond that of satellites, allowing for increased temporal resolution of imagery. An important concept for wetlands in particular as wetlands have naturally variable extents dependent on the season (Blanchette et al. 2018). The flexibility of UAV flight times means that the temporal resolution of imagery can be tailored to what is appropriate for the required monitoring or the scientific question being posed (Berni et al. 2009), allowing for rapid collection of information for input into management actions (Mullerova et al. 2017). This combined with the low skill level required to operate and process UAV imagery means drones can be used for routine monitoring (Turner et al. 2016, Seymour et al. 2018). It is for these reasons that the use of UAVs to investigate wetland vegetation and land cover extent is becoming more popular line of wetland survey (Gray et al. 2018).

To examine the vegetation community in the fenced wetland in Round Hill Reserve, and nearby wetlands, this study employed the use of drone UAV technology to collect a series of repeated aerial images of each wetland, which was used in GIS software to map vegetation community extent and change during this project. In addition, quadrats were used to more explicitly measure vegetation species composition and coverage at both high and low marsh areas in each wetland. These data provide a baseline understanding of the vegetation community in wetlands in the Round Hill Reserve region.

3.2.2 Methods

Image acquisition and analysis

Aerial imagery of the saltmarsh vegetation was collected using an unmanned aerial vehicle (UAV) (DJI Phantom 4). UAV’s were flown 3 times at each of the 5 sites in July 2018, January 2019 and June 2019 in order to encompass changes seen in the wet (January) and dry season (June/July) (Table 4). 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 (Figure 21). The image resolution was set to 0.3 m. A total of 15 stitched orthomosaic images with 3 bands (Red, Green and Blue) covering 3 survey events at 5 sites was produced. A polygon was delineated on each image outlining the high tide line. Where the high tide line could not be distinguished, the tree line was used as a proxy. This polygon was buffered 5 m, to account for any potential error in delineating the polygon and each image was clipped to the buffered polygon. Three of the 15 orthomosaic images were discarded due to misalignment. One image was stitched incorrectly (RH1, July 2018) and the flight path was incorrect for 2 images (RH4 January 2019, RH4 June 2019). Hence, these three scenes were excluded from the vegetation classification analysis.

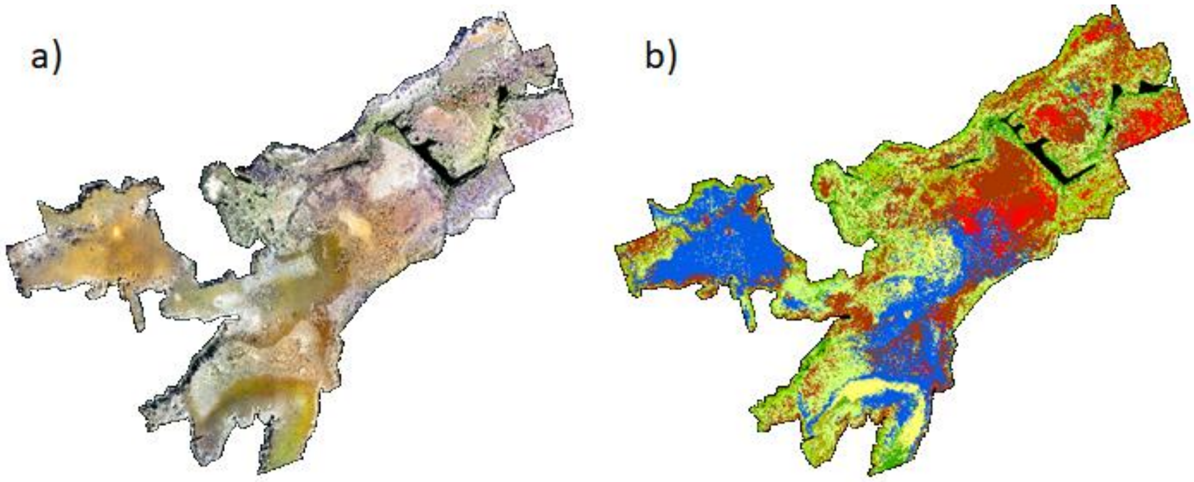
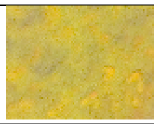
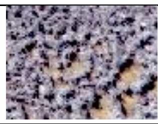
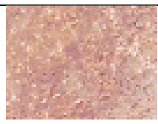
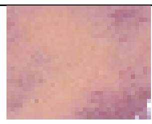
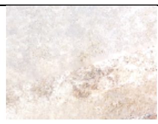


Figure 21. Example of a) orthomosaic image produced by georectifying and stitching together multiple UAV acquired images, and b) classification of the orthomosaic by vegetation types that were collected here.

Vegetation types were identified for classification into 8 classes (Table 3). A training set was created for each orthomosaic image by assigning a minimum of 30 polygons for each vegetation 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 is 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 (Esri 2016). In order to calculate the proportion of each land 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.

Table 3. Classes and associated classification cues used to generate training samples for pixel based image classification in ArcGIS

Class	Classification cues	Image chip examples
Trees (terrestrial)	Dark grey to green colour, shadows may be present	  
Trees (mangrove)	Brighter green colour than Terrestrial trees, found near/along bodies of water	 
Aquatic plants	Present in body of water, green to yellow colour	
Grass and sedges	Rough texture, grey to green colour	  
Shrubs and herbs	Rough texture, more height evident than Grass class	
Water	Sun glint evident, range of colours from blue to grey	 
Acid sulphate soil	Red coloured soil	 
Bare soil	Colour range from sandy white to dark brown, fine looking texture	  

The accuracy of classification was assessed by assigning 100 randomly distributed points per orthomosaic image 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 on the image to confirm it had been classified correctly, any errors were manually corrected – we also used a selection of the quadrat field data to assist with the model training (see below). 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. The confusion matrix computes user and producer accuracy with values ranging 0 – 1. User accuracy is a measure of Type 1 error, known as errors of commission, this measures the proportion of all accuracy assessment points which have been incorrectly classified as a known class (for example Bare soil being classified as ASS). Producer accuracy is a measure of Type 2 error, known as errors of omission, this measures the quality of the training samples by dividing the number of correctly classified accuracy points by the number of accuracy points assigned that class. A value closer to 1 meaning the classification is more accurate and the map produced is more representative of reality.

Vegetation surveys

On-the-ground quadrat surveys (1 × 1 m) of vegetation percent cover and height were conducted at each survey location (Figure 22). Quadrat surveys were placed adjacent to where carbon decomposition bags (Section 3.2) were deployed, following the same study design (i.e. five replicate surveys at each of three 'high' and low' elevation survey points within a survey location (RH1, RH2, RH3, RH4, RH5)). Within each quadrat the percent cover of live and dead vegetation was estimated, and if percent cover was >20% for a particular species, five individual plants were randomly chosen for height measurement (cm).

Figure 22. Example quadrat used in vegetation surveys in Round Hill wetlands to measure vegetation community.

Vegetation surveys were conducted at the five locations at high and low marsh positions, and within the area of the deployed saltmarsh litter bag decomposition experiment. Species percent cover and height was measured with 1 m^2 quadrats. Mean percent cover was calculated for each transect (n = 5). Surveys were repeated in March 2018, July 2018, and June 2019 (Table 4). Species richness and diversity indices (Shannon's 1-D and Simpsons H') were calculated at the five wetland locations for each tidal position. Vegetation species which made up < 0.1 % of quadrat cover were excluded from multivariate analysis. Percent cover was normalised and arc-sine square root transformed prior to calculating Bray-Curtis distances for non-metric multidimensional scaling. It is acknowledged that using quadrats may not necessarily provide a full representation of the vegetation community in a wetland, however, this approach could be well supplemented with the use of drone technology.

Table 4. Summary of when vegetation quadrat surveys and UAV surveys were conducted.

	Vegetation quadrat survey	UAV survey
March 2018	✓	-
July 2018	✓	✓
January 2019	-	✓
June 2019	✓	✓

3.2.3 Results and Discussion

Vegetation surveys

A total of 19 plant species were detected across all locations (Table 5). Vegetation species richness ranged from 0 to 5 plant species. RH1 (Round Hill) had the highest species diversity with diversity being higher in 'low' tidal positions compared to 'high' positions (Table 8). Shannon's H' index was higher than Simpson's 1-D or all locations and tidal positions.

Table 5. Plant species detected at wetland locations during quadrat surveys.

Species	Common name
Algae	Algae
Avicennia sp.	Mangrove
Bruguiera sp.	Mangrove
Cyperus eragrostis	Tall flatsedge
Eleocharis dulcis	Water chestnut
Eragrostis curvula	African lovegrass
Grass	Grass
Juncus articulates	Jointed rush
Juncus usitatus	Common rush
Leaf litter	NA
Nymphaea	Water lily
Portulaca bicolor	Pigweed
Sarcocornia quinqueflora	Beaded glasswort
Suaeda australis	Austral seablite
Sporobolus virginicus	Salt couch

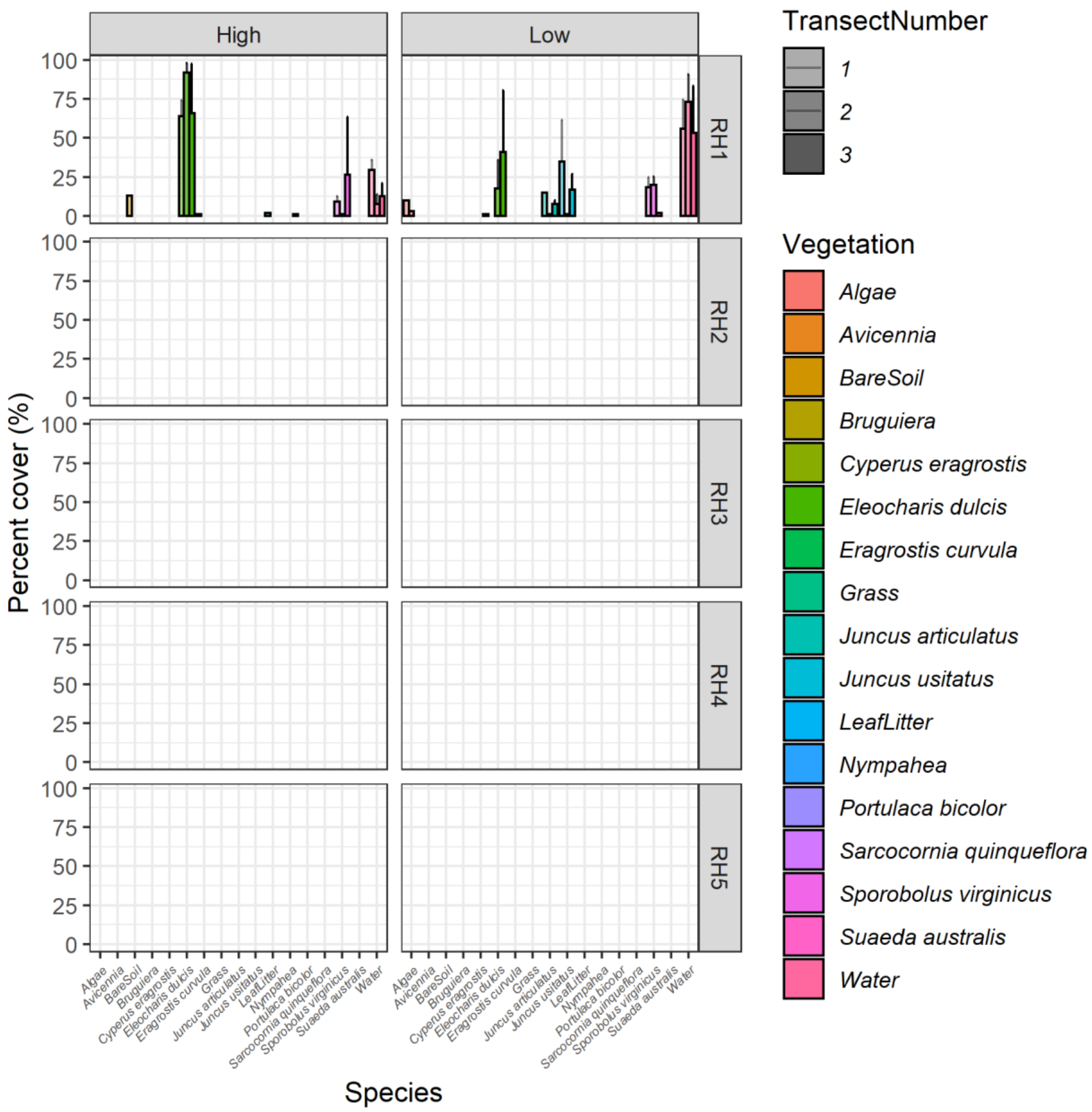


Figure 23. Vegetation percent cover in March 2018 quadrat surveys at high and low tidal positions at five wetland locations. Colours indicate plant species while opacity represents transect number. Mean values and standard deviations are reported (n = 3). Note: sites RH2 – RH4 were not surveyed in March 2018.

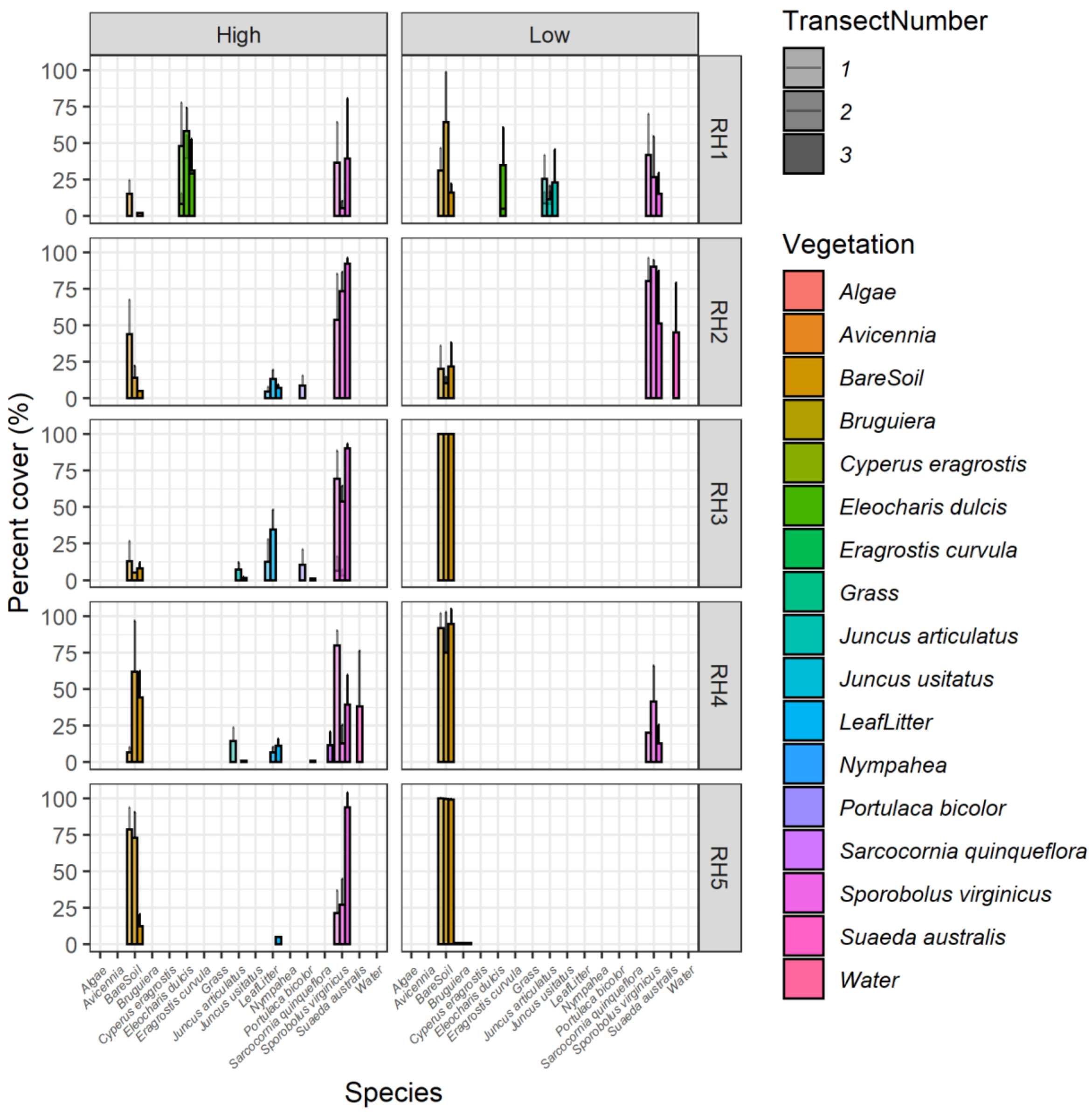


Figure 24. Vegetation percent cover in July 2018 quadrat surveys at high and low marsh positions at five wetland locations. Colours indicate plant species while opacity represents transect number. Mean values and standard deviations are reported (n = 3).

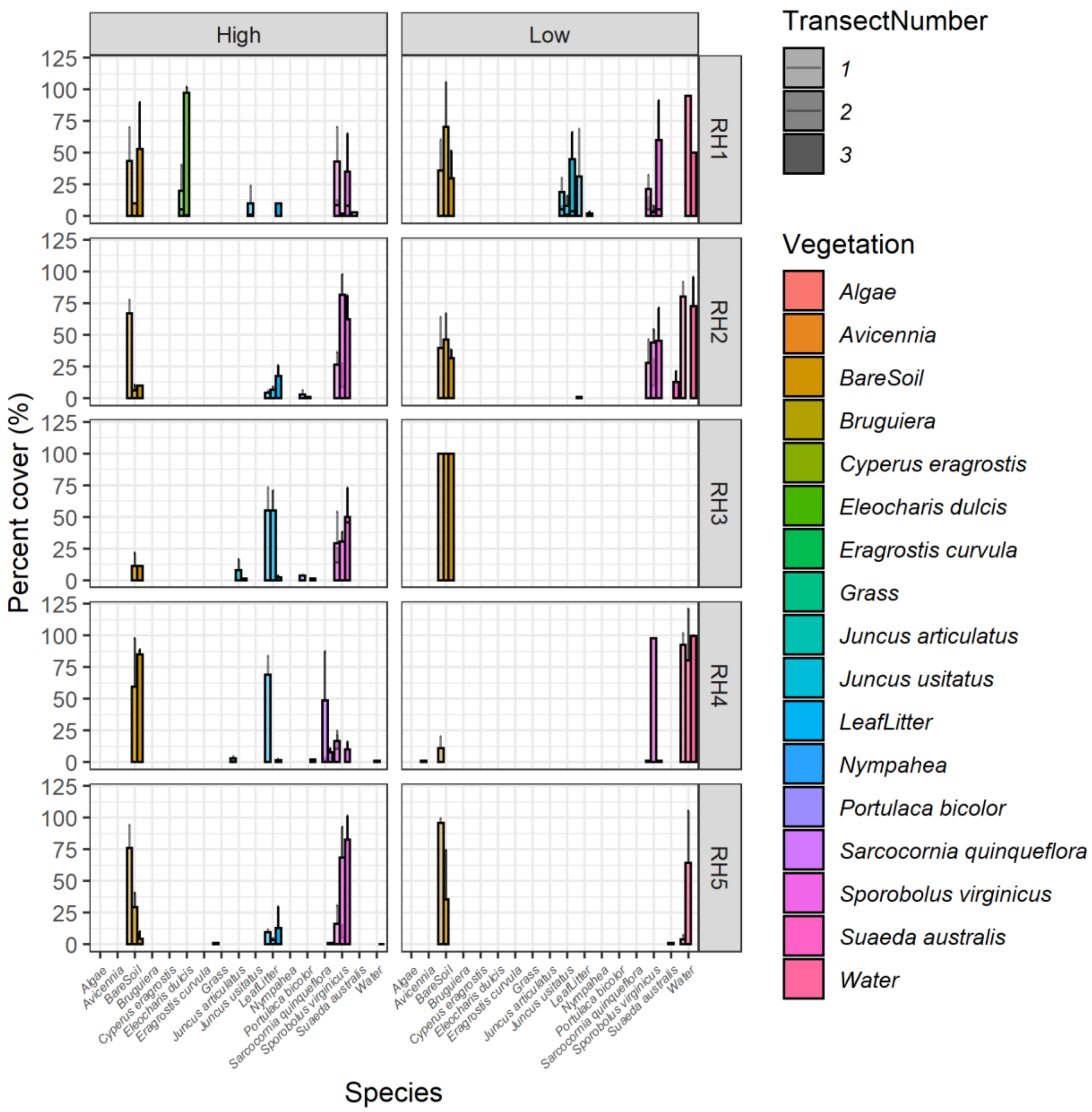


Figure 25. Vegetation percent cover in June 2019 quadrat surveys at high and low marsh positions at five wetland locations. Colours indicate plant species while opacity represents transect number. Mean values and standard deviations are reported (n = 3).

Saltmarsh vegetation percent cover was compared temporally (trip) and spatially (tidal position and location) in multivariate space. Multivariate dispersions were equal for trip and tidal position but not equal across locations (Table 6). Distance to centroids were significantly higher for RH1 and RH4 (Round Hill) than RH2, RH3, and RH5 indicating a much larger spread of samples in multivariate space for this location (PermDisp: $p < 0.001$; TukeyHSD: RH1 vs RH2/RH3/RH5 $p < 0.001$, RH4 vs RH2/ RH3 $p < 0.001$). Permutational analysis of variance indicated that there was a significant difference in vegetation cover between trips (PERMANOVA: *P*(perm) = 0.001), locations (PERMANOVA: *P*(perm) = 0.001), and tidal position (PERMANOVA: *P*(perm) = 0.006) (Table 7). We performed non-metric multidimensional scaling on the vegetation percent cover following normalization and arc-sine square-root transformation (Figure 26). The scaling converged with k = 5 dimensions.

Table 6. Permutation tests for homogeneity of multivariate dispersion (PERMDISP) of vegetation percent cover between Trip (Jul-2018, Jun-2019), Location (RH1, RH2, RH3, RH4, RH5) and Position (Low, High). Permutation: free. Number of permutations: 9999.

	df	*SS*	*MS*	*Pseudo-F*	*P (Perm)*
Trip	1	0.1518	0.151791	1.9069	0.1677
Residuals	207	16.4778	0.079603		
Location	4	3.3872	0.8468	17.508	**< 0.001**
Residuals	204	9.8671	0.04837		
Position	1	0.159	0.159049	2.2876	0.128
Residuals	207	14.392	0.069526		

Table 7. Permutational multivariate analysis of variance (PERMANOVA) of vegetation percent cover between Trip (Jul-2018, Jun-2019), Location (RH1, RH2, RH3, RH4, RH5) and Position (Low, High). Permutation: free. Number of permutations: 9999. Terms added sequentially (first to last).

	df	*SS*	*Pseudo-F*	*R^2*	*P (Perm)*
Trip	1	3.063	0.01667	3.6862	**0.001**
Location	4	11.374	0.06193	3.4225	**0.001**
Position	1	1.401	0.00763	1.6861	**0.006**
Residuals	202	167.829	0.91377		
Total	208	183.667	1		

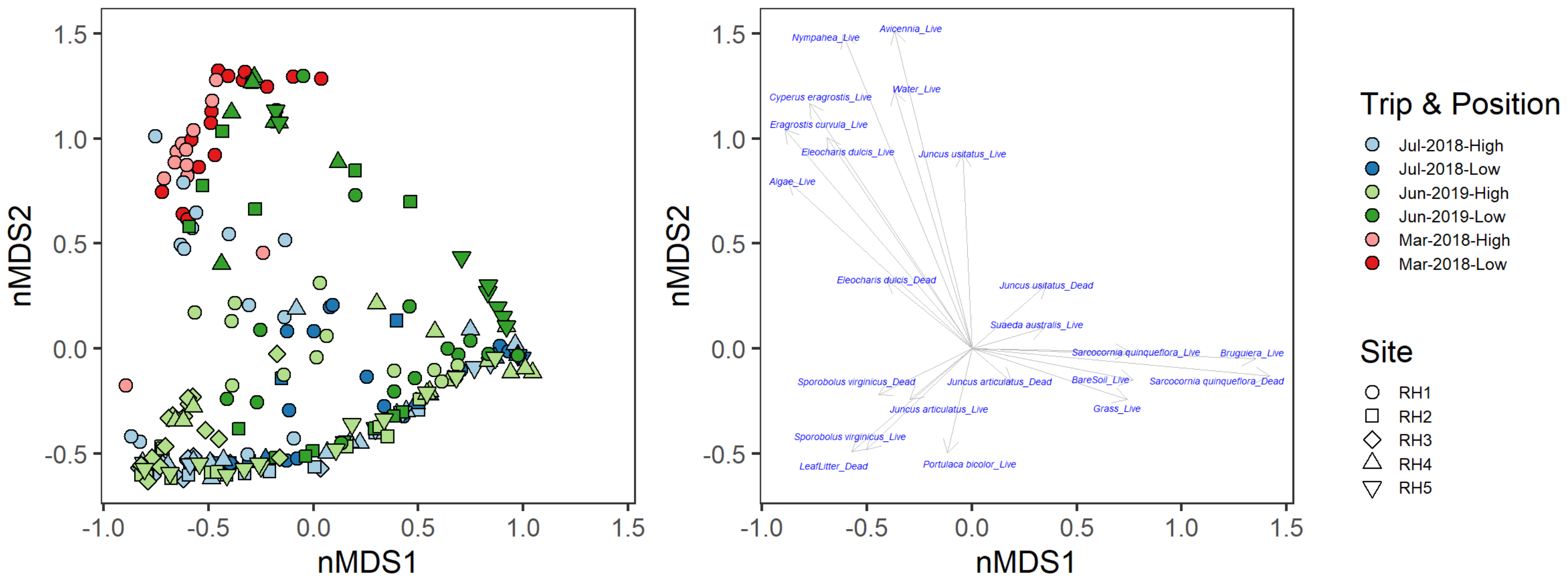


Figure 26. Bray-Curtis non-metric multidimensional scaling of normalized arc-sine square-root transformed percent cover values. 95 % ellipses. Stress = 0.046.

Table 8. Diversity indices for saltmarsh vegetation. Species richness is given as median, Shannon's 1-D and Simpson's H' indices are reported as mean and standard deviation (n = 5).

Trip	Code	Location	Tidal position	Species richness	Simpson's 1-D	Shannon's H'
Mar-18	RH1	Round Hill	High	2	0.11 ± 0.11	0.2 ± 0.18
			Low	2	0.27 ± 0.25	0.44 ± 0.41
Jul-18	RH1	Round Hill	High	2	0.13 ± 0.18	0.2 ± 0.26
			Low	2	0.31 ± 0.21	0.49 ± 0.33
	RH2	Littabella Down	High	1	0.07 ± 0.14	0.11 ± 0.21
			Low	1	0.03 ± 0.1	0.05 ± 0.15
	RH3	Littabella Up	High	2	0.12 ± 0.15	0.21 ± 0.21
			low	-	-	-
	RH4	Daniels	High	2	0.15 ± 0.17	0.25 ± 0.26
			Low	1	0	0
	RH5	Knuckies	High	1	0	0
			Low	1	0	0
Jun-19	RH1	Round Hill	High	1	0.1 ± 0.18	0.15 ± 0.26
			Low	2	0.2 ± 0.22	0.29 ± 0.31
	RH2	Littabella Down	High	2	0.05 ± 0.1	0.1 ± 0.15
			Low	1	0.01 ± 0.03	0.02 ± 0.07
	RH3	Littabella Up	High	2	0.17 ± 0.17	0.27 ± 0.25
			Low	1.5	0.12 ± 0.15	0.2 ± 0.23
	RH4	Daniels	High	1	0	0
			Low	1	0.01 ± 0.04	0.03 ± 0.08
	RH5	Knuckies	High	1	0	0
			Low	2	0.11 ± 0.11	0.2 ± 0.18

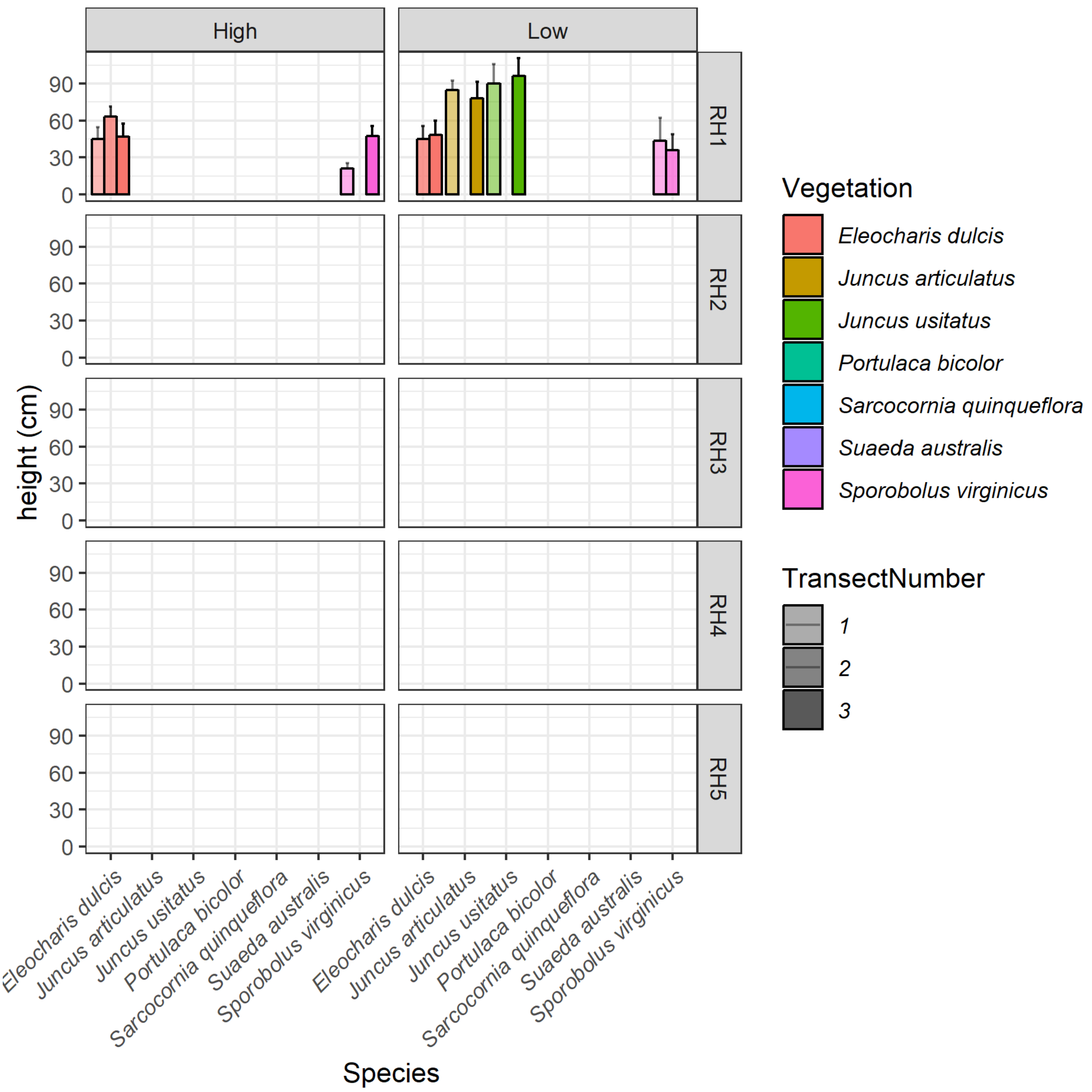


Figure 27. Vegetation height in March 2018 quadrat surveys at high and low tidal positions at five wetland locations. Colours indicate plant species while opacity represents transect number. Mean values and standard deviations are reported (n = 3). Note: sites RH2 – RH4 were not surveyed in March 2018.

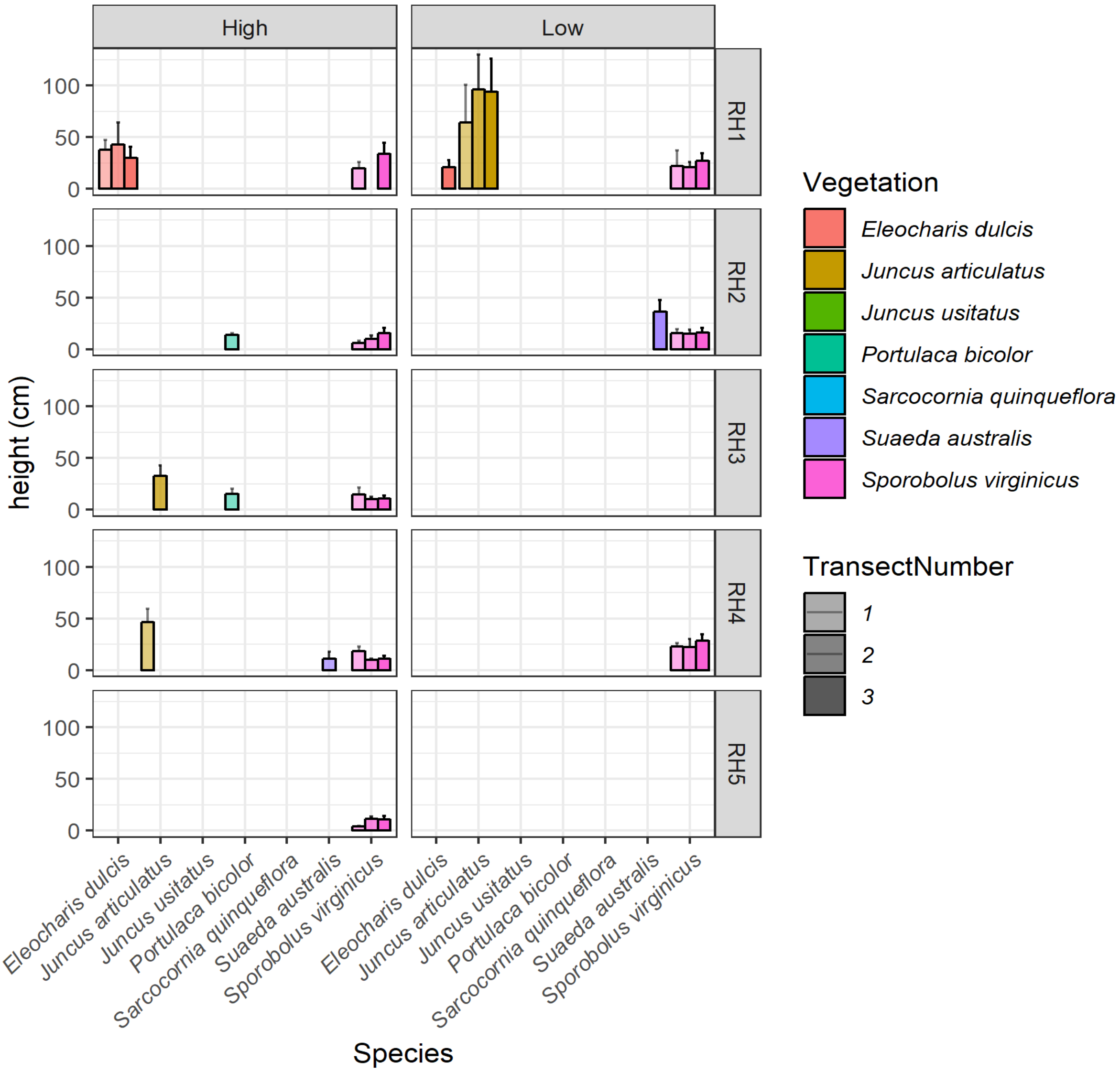


Figure 28. Vegetation height in July 2018 quadrat surveys at high and low tidal positions at five wetland locations. Colours indicate plant species while opacity represents transect number. Mean values and standard deviations are reported (n = 3).

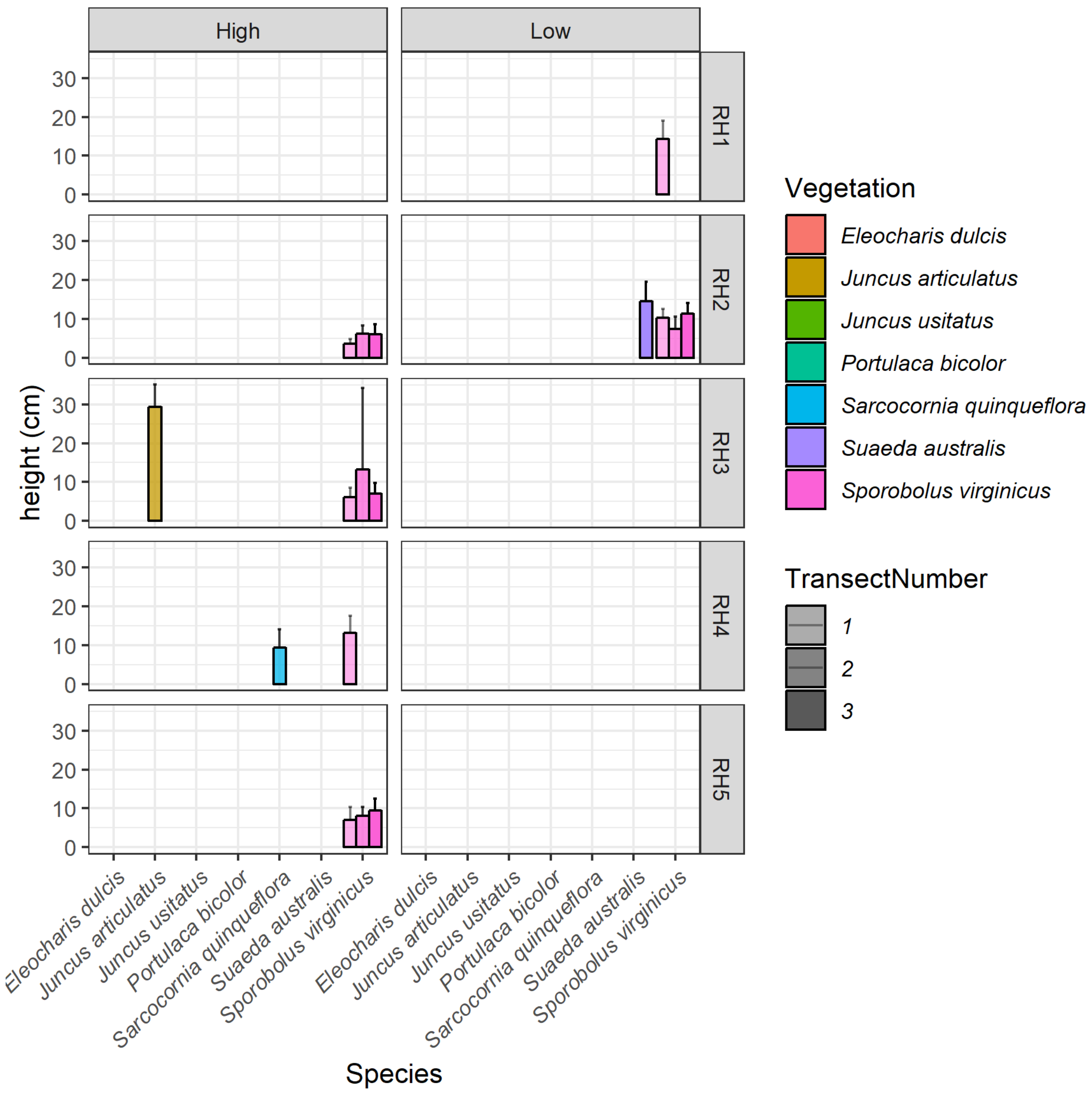


Figure 29. Vegetation height in June 2019 quadrat surveys at high and low tidal positions at five wetland locations. Colours indicate plant species while opacity represents transect number. Mean values and standard deviations are reported (n = 3)

Table 9. Permutation tests for homogeneity of multivariate dispersion (PERMDISP) of vegetation height between Trip (Jul-2018, Jun-2019), Location (RH1, RH2, RH3, RH4, and RH5) and Position (Low, High). Permutation: free. Number of permutations: 9999.

	df	*SS*	*MS*	*Pseudo-F*	*P (Perm)*
Trip	1	0.1518	0.151791	1.9069	0.145
Residuals	207	16.4778	0.079603		
Location	4	136239	34060	115.44	**0.001**
Residuals	760	224226	295		
Position	1	0.159	0.159049	2.2876	0.144
Residuals	207	14.392	0.069526		

Table 10. Permutational multivariate analysis of variance (PERMANOVA) of vegetation height between Trip (Jul-2018, Jun-2019), Location (RH1, RH2, RH3, RH4, RH5) and Position (Low, High). Permutation: free. Number of permutations: 9999. Terms added sequentially (first to last).

	df	*SS*	*Pseudo-F*	*R^2*	*P (Perm)*
Trip	1	22955	0.04071	41.052	**< 0.001**
Location	4	84754	0.15032	37.894	**< 0.001**
Position	1	32280	0.05725	57.73	**< 0.001**
Residuals	758	423842	0.75172		
Total	764	563831	1		

Image analysis

Twelve images were included in vegetation analysis. Three of the 15 orthomosaic images were discarded due to misalignment. One image stitched incorrectly (RH1, July 2018) and the flight path was incorrect for 2 images (RH4 January 2019, RH4 June 2019). The Aquatic plant land cover type was only identified at one site (RH1) so was excluded from further analysis.

RH1: As the stitching of imagery could not be completed for images taken at RH1 in July 2018, only change from January to June 2019 is presented (Figure 30). Two land cover categories had large increases from January 2019 to June 2019, these were Acid sulphate soil (ASS) and Water both of which doubled from one-time period to the next (Figure 31). Others, such as Bare soil, Terrestrial trees and Shrub land cover decreased by half over 6 months from January 2019 to June 2019. Both the Grass and Mangrove category experienced minimal changes, though Mangrove cover increased from January to June 2019 and Grass cover decreased over the 6 months.

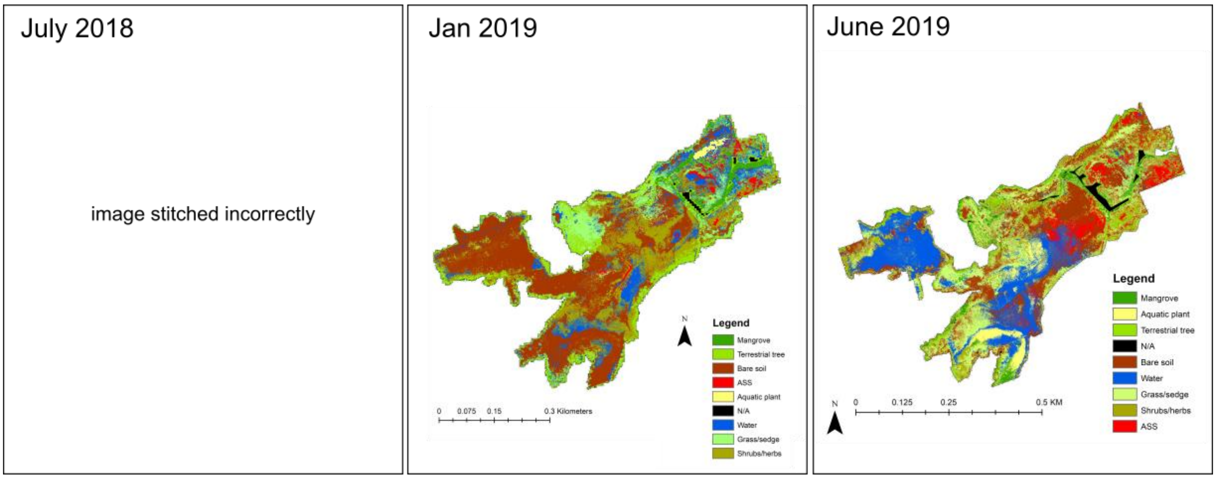


Figure 30. Vegetation classification from aerial imagery at Round Hill Reserve (RH1) in January 2019 and June 2019.

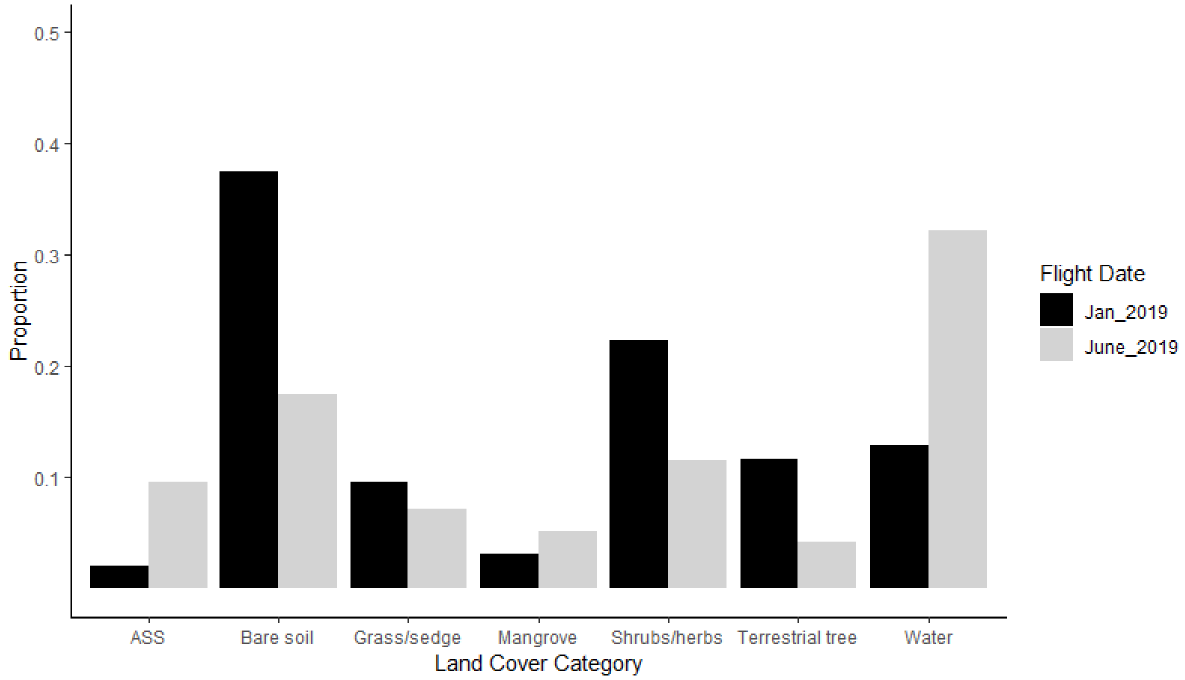


Figure 31. Proportion of land cover from classified imagery (Maximum likelihood classification performed in ArcGIS) at site RH1 taken from UAV imagery. Graph shows comparison between imagery taken in January 2019 and June 2019.

RH2: Vegetation classifications derived from aerial images acquired from Littabella Down reference site (RH2) in July 2018, January 2019, and June 2019 are shown in Figure 32. The proportion of land cover identified as Acid Sulphate Soils (ASS) shows a decreasing trend over time, although this decrease is not consistent, with a large change in proportions from July 2018 to January 2019 and only a slight decrease from January 2019 to June 2019 (Figure 33). The proportion of land classified as Bare soil shows an initial decrease from July 2018 to January 2019 and then an increase to 0.382 in June 2019. Both the Grass and Mangrove category have similar classified proportions across all 3 time series (although the proportion of Mangrove is less than that of Grass) with a slight drop from January 2019 to June 2019. The Terrestrial tree category shows the converse trend, with similar proportions in July 2018 and January 2019 and an increase in June 2019. The Shrubs and Water category show similar trends (although the proportion of Water is generally higher than that of the Shrub), with an increase in proportions of land cover from July 2018 to January 2019 followed by a decrease to June 2019 (although the June 2019 proportion is higher than that in July 2018.

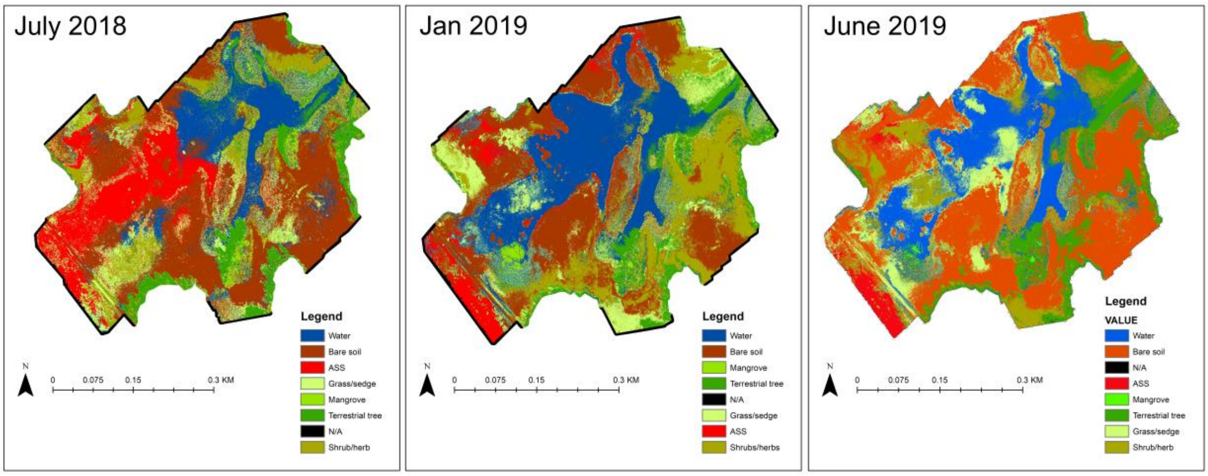


Figure 32. Vegetation classification from aerial imagery at Littabella Down (RH2) in July 2018, January 2019, and June 2019.

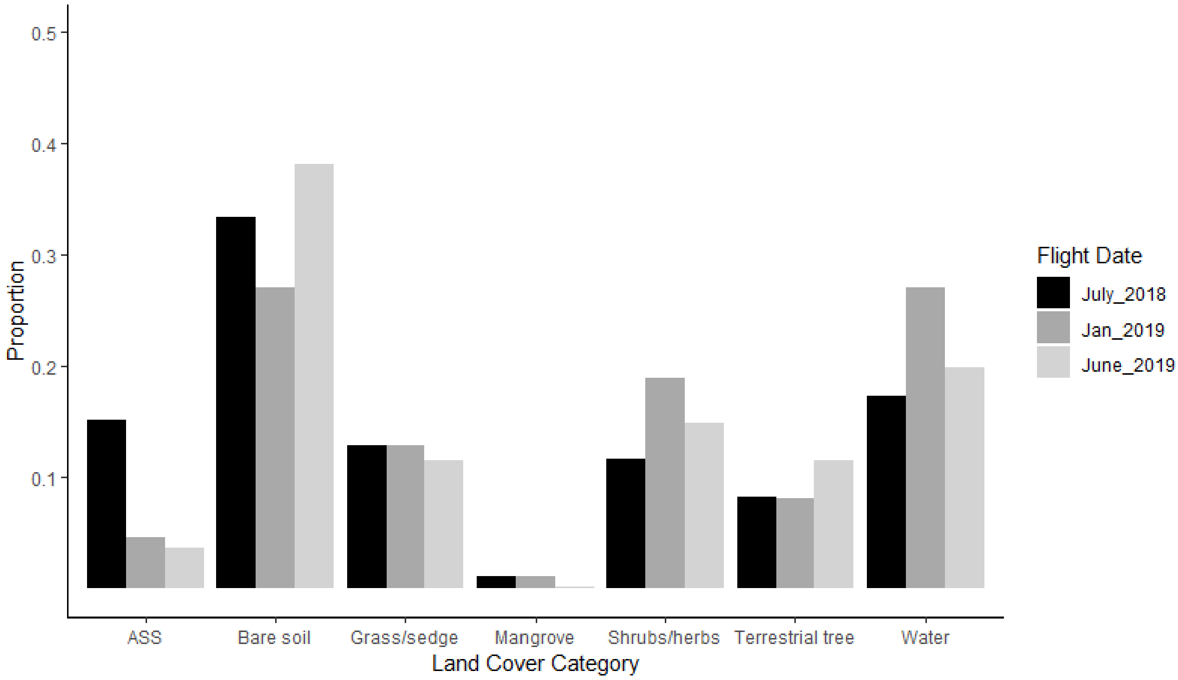


Figure 33. Proportion of land cover from classified imagery (Maximum likelihood classification performed in ArcGIS) at site RH2 taken from UAV imagery. Graph shows comparison between imagery taken in July 2018, January 2019 and June 2019.

RH3: Vegetation classifications derived from aerial images acquired from Littabella Up reference site (RH3) in July 2018, January 2019, and June 2019 are shown in Figure 34. ASS show a slight decrease between July 2018 and January 2019, but a large increase to June 2019 (Figure 35). This may have compensated for a loss of Bare soil observed between January 2019 and June 2019 or the decreased proportion of the Grass category observed at over the same time period. No Mangrove was identified as part of the classification and proportions of Terrestrial tree identified remained relatively ranging from 0.0586 in July 2018 to 0.0664 in June 2019. The proportion of Shrub classified remains similar between July 2018 and January 2019, with an increase to June 2019. The proportion of Water classified shows a marked decrease from July 2018 to none identified in January 2019, with an increase in June 2019.

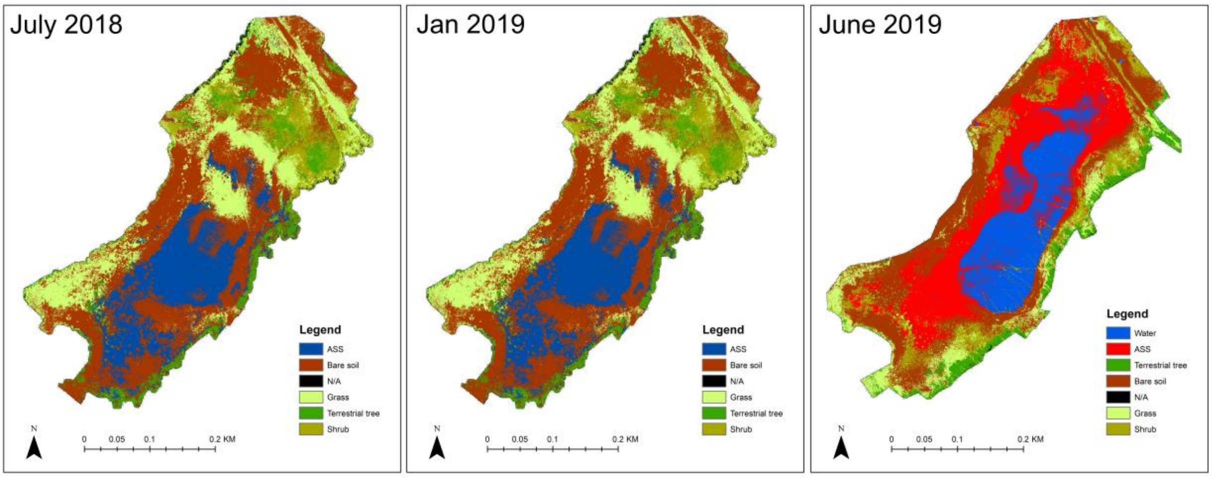


Figure 34. Vegetation classification from aerial imagery at Littabella Up (RH3) in July 2018, January 2019, and June 2019.

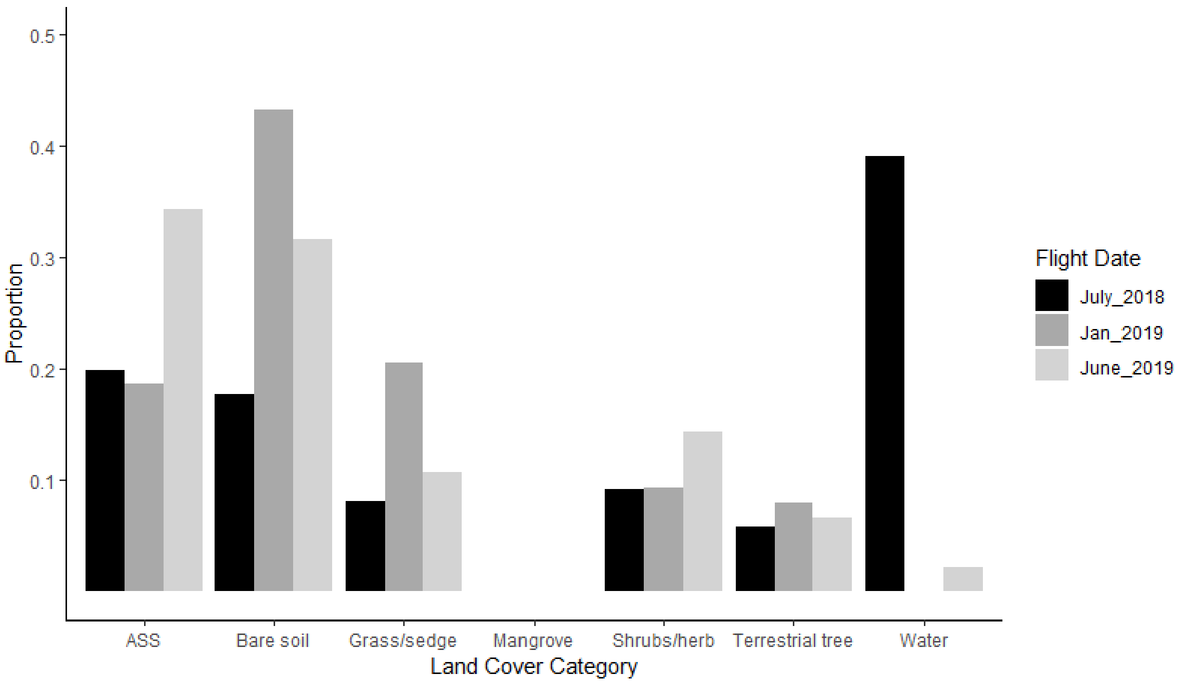


Figure 35. Proportion of land cover from classified imagery (Maximum likelihood classification performed in ArcGIS) at site RH3 taken from UAV imagery. Graph shows comparison between imagery taken in July 2018, January 2019 and June 2019.

RH4: Vegetation classifications derived from aerial images acquired from Round Hill reference (RH4) in July 2018, January 2019, and June 2019 are shown in Figure 36. As the flight path was incorrect for images taken at RH4 in January and June 2019, only results from July 2018 could be presented. Low levels of the ASS, Grass, Mangrove and Shrub category were found at RH4 (proportions <0.1) (Figure 37). The dominant land cover was Water followed by Bare soil and Terrestrial trees.

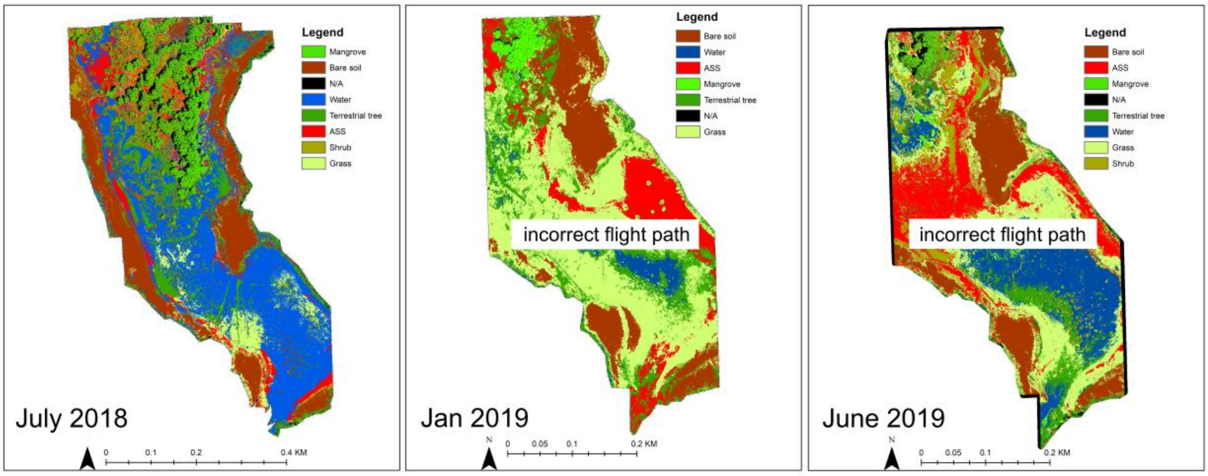


Figure 36. Vegetation classification from aerial imagery at Round Hill reference (RH4) in July 2018, January 2019, and June 2019.

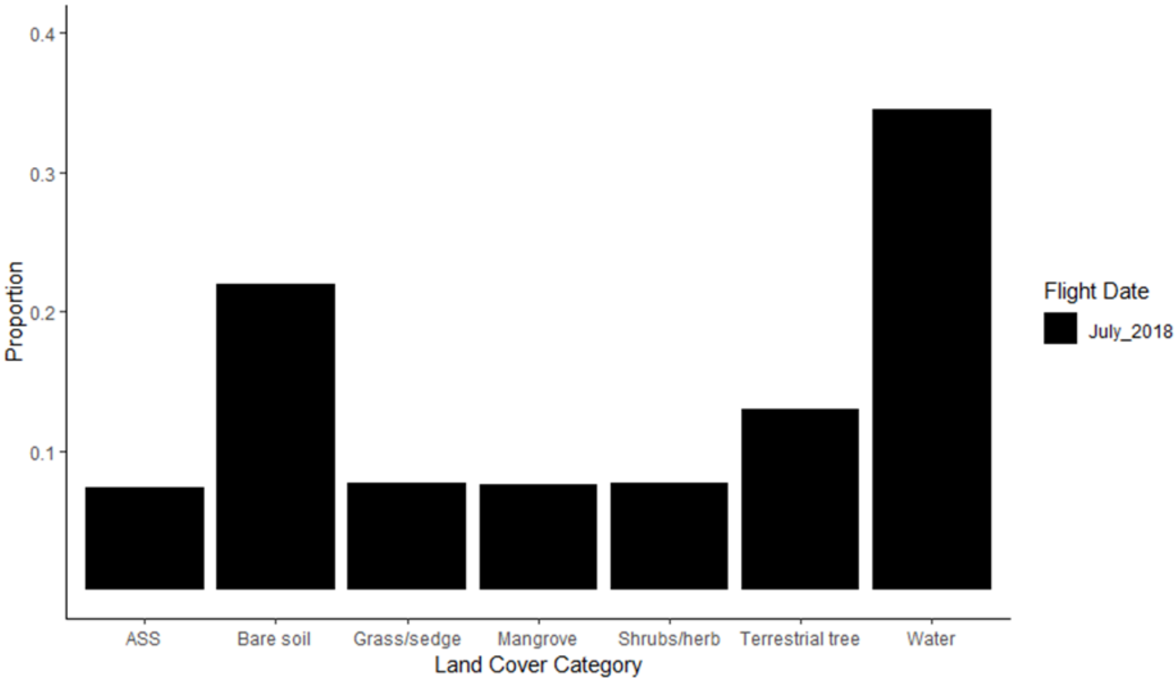


Figure 37. Proportion of land cover from classified imagery (Maximum likelihood classification performed in ArcGIS) at site RH4 taken from UAV imagery. Graph shows comparison between imagery taken in July 2018.

RH5: Vegetation classifications derived from aerial images acquired from Eurimbula NP reference (RH5) in July 2018, January 2019, and June 2019 are shown in Figure 38. No ASS was found in any of the time periods for RH5 (Figure 39). The Bare soil identified shows a slight increase from July 2018 to January 2019 with a decrease to below July 2019 levels in June 2019. The proportion of Grass land cover identified is similar between July 2018 and June 2019 but is much higher in January 2019. The Mangrove land cover shows small changes in proportion, decreasing from July 2018 to January 2019 and then increasing in June 2019. While the Shrub category has a similar pattern, the subsequent increase from January 2019 to June 219 is sharper. The Terrestrial tree and Water categories show similar trends over time with the highest proportion being identified in July 2018, followed by a decrease to January 2019 and the proportion remaining constant for June 2019.

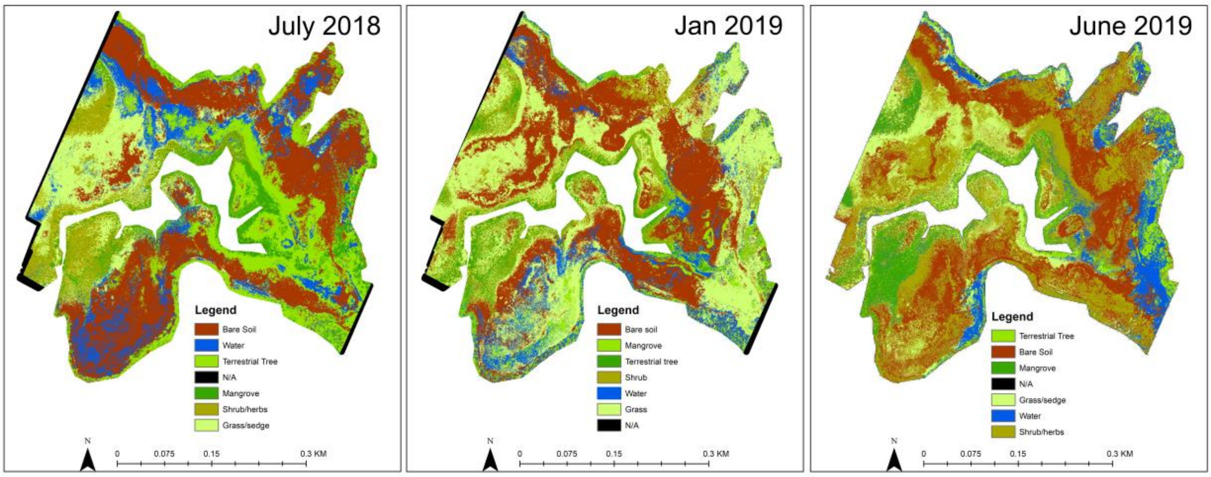


Figure 38. Vegetation classification from aerial imagery at Eurimbula NP (RH5) in July 2018, January 2019, and June 2019.

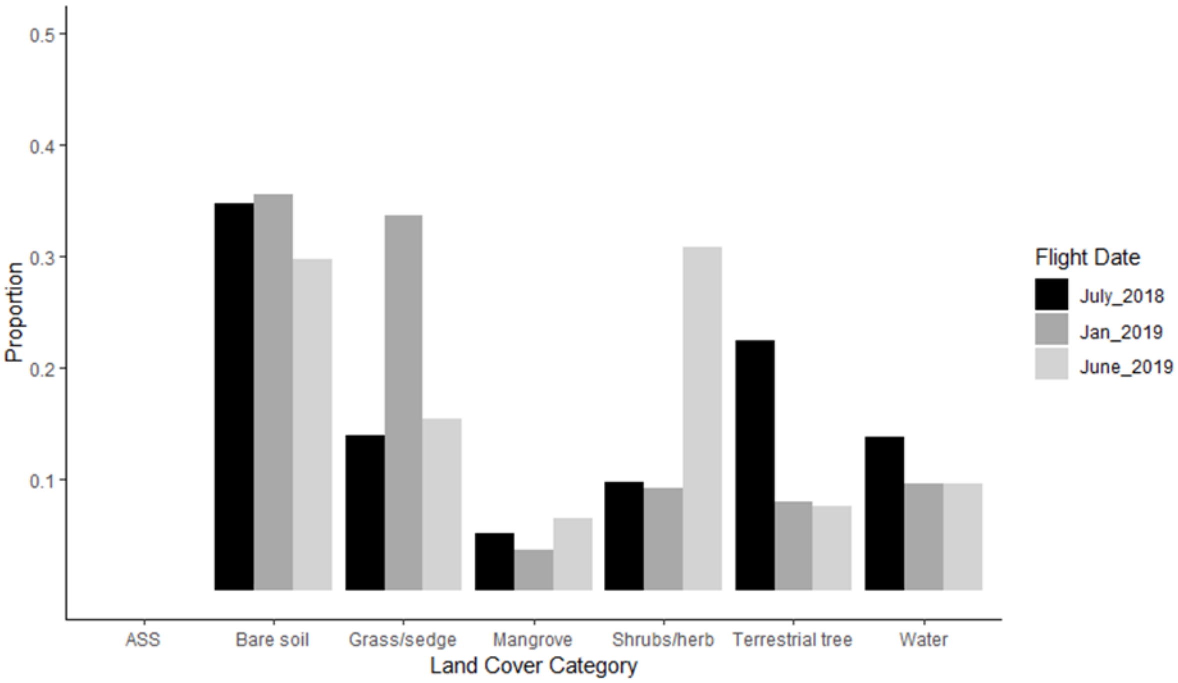


Figure 39. Proportion of land cover from classified imagery (Maximum likelihood classification performed in ArcGIS) at site RH5 taken from UAV imagery. Graph shows comparison between imagery taken in July 2018 January 2019 and June 2019.

Changes in land cover – class trends: While all land cover types were variable out of the twelve orthomosaic images the Water (4 orthomosaics) and Bare soil classes (6 orthomosaics) were consistently the most dominant classes present at each site (Figure 40). Only 2 other land classes appeared as the most dominant land cover at any other site or time, this was ASS at site RH3 in June 2019 and Shrub at site RH5 in June 2019. All classes, expecting Mangroves and ASS, were present at all sites.

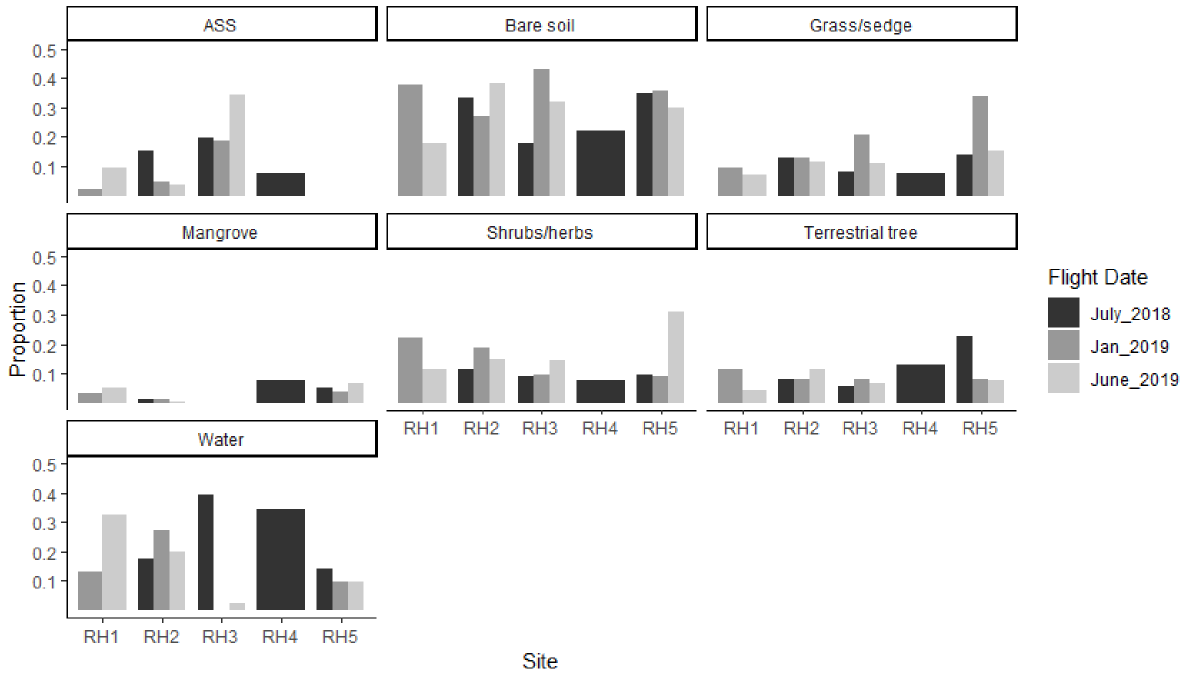


Figure 40. Proportion of land cover by class and site from classified imagery (Maximum likelihood classification performed in ArcGIS). Imagery compares images taken in July 2018, January 2019 and June 2019.

Accuracy assessment – by vegetation class: The average user accuracy for this classification by site is overall low (ranging from 0.390 to 0.637) (Figure 41). Producer accuracy was overall low though higher than the user accuracy ranging from 0.436 to 0.727. Site RH3 had the highest mean producer and user accuracy across all 5 sites (Figure 8). Within each site the mean producer and user accuracy was similar, being with 0.1 of each other.

Accuracy assessment – by land cover type: Both mean producer and mean user accuracy across classes was variable (Figure 9). Bare soil, Mangrove and Water were the classes with the highest user accuracy (0.827, 0.800 and 0.656 respectively), meaning these are the classes that are most accurately represented by the classification. While other categories (ASS, Grass, Shrub, Terrestrial tree) had lower accuracy. Though ASS and Shrub user accuracy was low, producer accuracy for these two classes were among the highest of all categories (0.844 and 0.587 respectively). The Mangrove, Bare soil, Terrestrial tree, Water and Shrub categories all possessed similar producer accuracy (ranging from 0.472 to 0.587) suggesting equally poor training samples in these categories. The Grass category had among the lowest mean user and the lowest producer accuracy (0.250 and 0.316 respectively). Though producer accuracy across all categories was on average lower than user accuracy.

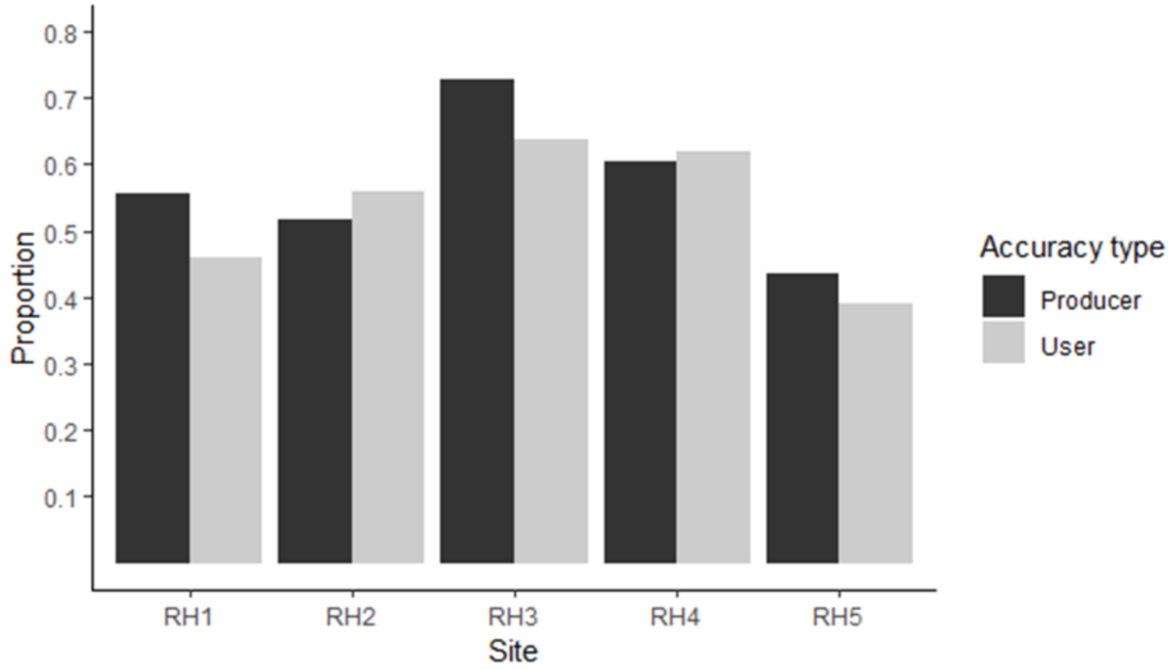


Figure 41. Comparison of mean producer and user accuracy. Metric includes the mean accuracy of all vegetation classes at all surveys available for each site. Accuracy values were calculated using the compute confusion matrix tool in ArcGIS.

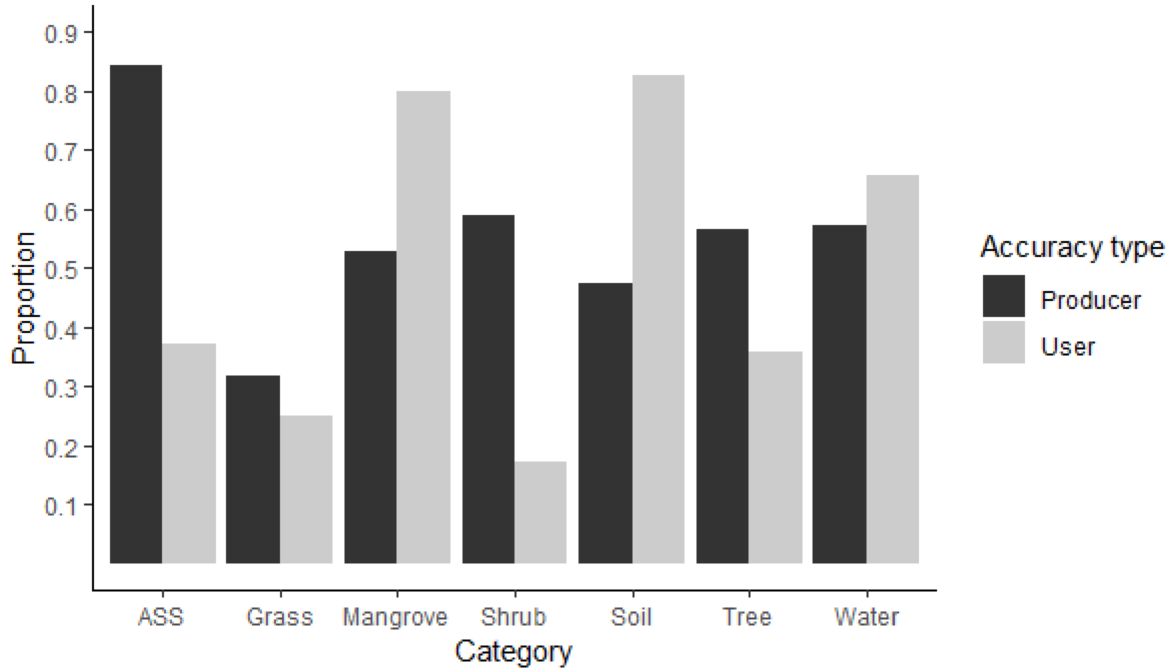


Figure 42. Comparison of mean producer and user accuracy for each class. Metric includes the mean accuracy of all classes at all flight dates available for that class. Accuracy produced using the compute confusion matrix tool in ArcGIS.

Future research in this area should focus on increasing the accuracy of classification so that the benefits of utilising UAVs can be fully maximised. Accuracy could be increased with the addition of hyperspectral imagery. For example, other studies have found that the inclusion of bands beyond the RGB increases the accuracy of classification (Abeysinghe et al. 2019, Al-

Najjar et al. 2019). In particular the ability to incorporate infrared and calculate the Normalised Different Vegetation Index may assist with increasing accuracy of vegetation categories which are spectrally similar in the RGB bands (such as the Grass and Shrub category) (Ruwaimana et al. 2018). This is reinforced by the finding presented here that ASS had the highest accuracy across all classes, which would have been the most spectrally different to other classes due to its high values in the red band. Object based classification is an alternate approach (Heumann 2011, Dronova et al. 2012, Körting et al. 2014, Fu et al. 2017, Luo and Mornya 2019), although this approach may not increase classification accuracy for wetland environments (Duro et al. 2012, Duffy et al. 2018). Combining multiple approaches (i.e. RGB classification, object-based classification, and inclusion of hyperspectral data) may offer the highest accuracy (Fu et al. 2017). Amalgamating classes to test if this would improve accuracy offered some increased accuracy and confidence in the method, a finding mirrored by others (Dronova et al. 2012, Ruwaimana et al. 2018), though the accuracies in these studies was higher than found here.

The identification of general trends (i.e. that land cover has changed) remains useful information and the failings identified as part of this study in the ability to quantify land cover change still represent a springboard for the advancement of wetland classification. This shows that while UAVs have the ability to become a staple tool for environmental management (and are on their way to becoming one) there is still work to be done in ensuring that the benefits of UAVs can be implemented in a useful and accurate manner.

This study investigated the species composition and coverage of saltmarsh vegetation across five coastal wetlands. The intention was to demonstrate the efficacy of installing exclusion fencing to limit vegetation damage due to feral pigs and cattle trampling. While the fencing was ineffective at limiting vegetation trampling due to management practice of allowing cattle into the enclosure, these data provide a baseline understanding of the vegetation community in the wetland in the Round Hill Creek Reserve and others in the region. Prevention of grazing elsewhere has been shown to increase vegetation cover and reduce bare ground (Jensen 1985), and the restoration of a detrital food web (Andresen et al. 1990). Grazing reduces sediment deposition and accretion, hence changing the sediment topology. As saltmarshes generally display vegetation zonation with elevation, the species which re-establish following destocking may be affected by this for some time until natural sedimentation and accretion processes are reinstated. Ungrazed sites show vegetation succession (due to natural processes), while grazed sites did not change (Jensen 1985).

The vegetation communities at the five wetland sites were primarily composed of stands of salt tolerant marsh species. The dominant vegetation type was *Sporobolus virginicus* (Salt couch), with large areas bare soil (saltpan). *S. virginicus* dominated most high and low elevation positions, with *Juncus sp.* present mostly in higher elevation positions in the marshes. High marsh areas were characterised by the presence of *Juncus usitatus* (Common rush) and less commonly *J. articulates* (Jointed rush). *Suaeda australis* (Austral seablite) was present at the Roundhill Creek sites RH1 and RH4, but not detected elsewhere. The succulent *Portulaca bicolor* was present at RH4, and the Litabella estuary sites RH2, and RH3. Freshwater species *Eleocharis dulcis* (Bulkuru / Spike-rush), *Nympahea sp.* (Water lily), and the introduced *Eragrostis curvula* (African lovegrass) were present at the RH1 Round Hill site, but not encountered elsewhere. Mangroves were generally not encountered within quadrats, although

Bruguier sp. (Orange mangrove) was detected at the wetland site in Eurimbula NP (RH5), and *Avicennia marina.* (Grey mangrove) was detected at RH4.

The Round Hill Creek reserve site (RH1) supported a more diverse floristic community than other sites visited in this study. This was mostly due to the presence of freshwater species which were absent elsewhere. The RH1 site is positioned high in the estuary meaning it is least frequently inundated by tides, and more regularly inundated by freshwater flows after rainfall events. There were some limitations in the vegetation sampling design which made it difficult to compare UAV derived vegetation types with what was measured on the ground during quadrat surveys. Quadrats were generally placed close to or on boundaries between vegetation types, this was especially the case with 'high' sites which were placed at the saltmarsh-mangrove boundary. Mangroves overhead of quadrats were not recorded as the vegetation survey was focussed on what was within the quadrat on the ground. This may have been because the trunk itself was not within the quadrat, while branches and foliage were overhead.

The conservation fence at Round Hill site RH1 was ineffective at protecting fragile marsh vegetation from trampling. Drone imagery and field observations identified higher vegetation cover in marsh areas adjacent to RH1 outside of the fenced enclosure in areas where cattle were unable to access due to the natural barrier of a bend in the creek. The management practice of opening the fence to allow cattle to enter the wetland to graze (and presumably also feral pigs). The data collected in this study will be useful in monitoring vegetation recovery when land-use management practices change.

3.4 Water Quality and Faunal Biodiversity

3.4.1 Introduction

Coastal ecosystems (e.g., mangroves, coral reefs, seagrass, and freshwater wetlands/swamps) have been modified or lost completely because of urban and agricultural expansion in many places (Mitsch and Gosselink 1993, Duarte et al. 2008, Elliott and Whitfield 2011, Murray et al. 2019). This has prompted coastal managers to launch programs to restore coastal floodplains with the objective of bringing back functionally equivalent, or as close as possible, to previous ecosystems (Creighton et al. 2016). Major large-scale coastal floodplain restoration and rehabilitation activities have been implemented in many places (Phelps et al. 2015, Hine et al. 2017, Elliott et al. 2019), although long-term commitment needed for maintenance and monitoring success (that is appropriately funded) is often lacking and leads to gaps in knowledge about effective practices (Sheaves et al. 2014, Zedler 2016). This information deficit could pose a risk of funders (government, private companies) having reduced confidence in the biological and conservation return on their investment (Bullock et al. 2011, Alexander et al. 2016, Bayraktarov et al. 2016). To be effective, measuring and evaluating the services and values of an ecosystem following some level of restoration must be able to show whether the management actions were effective, and whether they were cost-effective (Elliott and Whitfield 2011, Elliott et al. 2016). Access to restoration data demonstrating success and cost-effectiveness will only become more necessary as managers push for more government or private funding to up-scale restoration of coastal ecosystems in response to the United Nations General Assembly Decade on Ecosystem Restoration (2021-2030) (Waltham et al. 2020a).

Round Hill Reserve holds important conservation value given it is proximity to a Declared Fish Habitat Area (FHA), part of a Directory of Important Wetland, and is within the Eurimbula National Park. The impact of feral pig damage on the wetland biodiveristy and water quality services and values is investigated here, to serve the purpose of providing a baseline for future comparison, but to also examine how the fencing mitigation improves or protects these services and values. Completing these surveys assists in evaluating the compontents delivering water quaity and biodiversity services and values (or not) – for example changes in inundation patterns, changes in the sediment and vegetation, runoff from the catchment surrounding. These data are necessary for managers to understand the processes contributing to the changes in values and which of these processes are important for further investment and management effort (see Figure 1).

3.4.2 Methods

Water quality

Physicochemical water quality parameters were measured at each wetland and creek site coinciding with fish fyke net and boat electrofishing surveys. Depth profiles were measured with a multiparameter probe (Hydrolab Quanta) for temperature (°C), electrical conductivity (mS cm^{-1}), dissolved oxygen (mg L^{-1} and %sat), and pH. Three measurements were recorded at each depth with mean and standard deviation reported. A secchi tube was used to measure turbidity (NTU) at each site. Water samples were collected to measure nutrients (TN: total nitrogen, TDN: total dissolved nitrogen, NH_3: Ammonia, NO_x: Nitrate/Nitrite, PN: particulate nitrogen, TP: total phosphorus, TDP, total dissolved phosphorus, FRP: filterable reactive phosphorus, PP: particulate phosphorus, µg L^{-1}), Chlorophyll-*a*, Phaeophytin-*a* (µg L^{-1}), and total suspended solids (TSS: mg L^{-1}) concentrations at each survey location. Water samples were analysed at the TropWATER Water Quality Laboratory (Townsville, QLD). Fish habitat was quantified at each creek site based on proportional representation of stream structure, substrate type, and organic structure (living and detrital).

Fish surveys (fyke net)

Fish surveys were conducted in saltmarsh wetlands during the post-wet (March 2018) and dry seasons (July 2018, June 2019). Fyke nets (wing width: 5 m, mesh size: 1 mm) were set at three wetland locations (Figure 43). Nets were set within the fenced area at Round Hill Reserve (RH1) and at two reference sites; Littabella Down (RH2), and Round Hill (RH4) with three nets set at each Low and High tidal positions. Nets were set in the afternoon and retrieved the following morning. Fish were identified and measured (SL: standard length) before being returned to the water except declared fish species that were euthanized and disposed of under Queensland legislation. Water depth was insufficient to deploy fyke nets at the other reference sites Littabella Up (RH3) and Eurimbula NP (RH5).

We performed constrained ordinations on community (fish) and environmental data (water quality) distance based redundancy analysis (dbRDA) with Bray-Curtis dissimilarity indices. Stepwise model selection using permutation tests (ordistep) was used to reduce the model to the environmental variables which best explained the spread of community data. Multivariate analysis was performed in R (version 3.5.1) with the ‘vegan’ package.

Figure 43. Retrieving fish fyke net in Round Hill wetland with the field team.

Herpetofauna surveys

Herpetofauna were surveyed at the five saltmarsh locations (RH1, RH2, RH3, RH4, RH5) in July 2018. A variety of trapping methods and techniques were used to monitor herpetofauna around wetlands and adjacent woodland habitats. Drift fences and funnel traps, arboreal cover boards, Bioacoustic audio recorders, wildlife trail cameras, and active searches were used in unison to sample reptile and amphibian communities. Each trap array consisted of all the above trapping techniques. Twenty-five m long drift fences (approximately 40cm high) were dig into the ground approximately 5-10 cm into the soil to ensure no animals were able to pass underneath the fence (Figure 44). Fences were held in place by a series of 10-15 metal pegs (~45 cm long) to keep the fence upright. Each fence was dug into the ground with a mattock. Fences were installed diagonally at the wetland – woodland interface to allow both wetland associated and woodland associated animals to encounter the drift fence. Three paired funnels traps (a total of 6 per drift fence, 3 on each side) were installed at the ends and the middle section of each drift fence.

Five arboreal coverboards (12mm closed-cell foam) were temporarily attached to the main trunks of trees approximately 1.5 m from the ground using two elastic bungee cords (Figure 45). Each arboreal coverboard provides artificial bark habitat that can be removed to capture sheltering wildlife that typically hide under loose or peeling tree bark. The use of artificial bark prevents long term damage to tree bark through peeling in search of sheltering animals. Wildlife infrared game cameras and remote audio recorders were used to passively monitor wildlife. Wildlife game cameras were strapped to the main trunks of trees within 50 m of the drift fence array to target medium to large animals that would not be captured in funnel traps. Similarly, bioacoustic audio recorders were strapped to trees within the trapping array to document bird and frog calls. Active searches were also conducted at each site to systematically target animals using a timed-constrained search. Searches consisted of visual encounters of animals in the leaf litter, on or under logs and rocks, on the trunks of trees, or other opportunistic encounters. Each site was monitored for a total of two days, with traps being checked each morning. All captured individuals (from trapping and active searches) were identified, measured, and then released at their capture locations. Snout-vent length (SVL) and total length (TL) were measured with callipers, while body weight was measured on a small scale.

Figure 44. Drift line and traps set up in Littabella wetland.

Figure 45. Arboreal board used to survey herpetofauna in wetlands.

Waterbird surveys

Waterbird species presence was surveyed using camera traps (motion-detect setting) during two survey periods. Camera traps were initially attached to trees at Round Hill Reserve (RH1, 5 cameras) for between 24 and 48 hrs in March 2018. Camera traps were re-deployed at Round Hill Reserve (RH1, 1 camera) for between 24 and 48 hrs in July 2018 along with three reference sites: Littabella Down (RH2, 2 cameras), Littabella Up (RH3, 1 camera), and Round Hill (RH4, 1 camera).

3.4.3 Results and Discussion

Water quality

Water temperature at wetland locations ranged from 18 to 31 °C (Table 11). At the end of the wet season (March 2018) the Round Hill Reserve wetland (RH1) was fresh with electrical conductivity < 2 mS cm^{-1}. We attribute this to fresh inputs from rain- and ground- water. In the dry season (July 2018 and June 2019) all wetland locations were saline to hypersaline with electrical conductivity ranging from 43 to 84 mS cm^{-1}. Dissolved oxygen (DO) concentrations ranged from 27.5 to 117.3 % saturation. The lowest DO concentrations were measured at the end of the wet season at Round Hill (RH1) at depth although an overlying stratified layer of warmer and more oxygenated water was also present. Understanding the processes contributing to available oxygen in waters is complex. In coastal waters it is very common for daily DO minima to fall below asphyxiation thresholds for fish, and while the results here are probably not likely low enough to present potential risks to fish, the results are spot measures and give no impression of the longer term variability in conditions that would be expected to occur in these types of coastal wetlands. For example, low DO are commonly recorded in mangrove channels and back waters in saltmarshes (Dubuc et al. 2017, Mattone and Sheaves

2017) which does present biological challenges (Davis et al. 2014, Mattone and Sheaves 2019). A possible reason for few fish species recorded in the wetlands here might be linked to poor DO conditions that occur in these upper estuarine waters, which we were not able to capture. For the species recorded here might have well adapted strategies to deal with extended periods of low DO (Cheek et al. 2009, Tiedke et al. 2014). For example fish in the wild that are exposed to concentrations below thresholds can rise to the water surface to utilise aquatic surface respiration (the process of selectively inhaling water from the thin surface water layer that is in immediate contact with the air) or air gulping – some fish species undertake this response, most notably is the tilapia (*Oreochromis mossambicus*) which was recorded here and a species that is known to adapt to critically low levels of available DO (Butler and Burrows 2007). The capacity for fish to adapt to critical conditions safely depends on the timing of the oxygen sag, predation risks, and sufficient water depth to complete this response (Flint et al. 2018).

pH ranged from 5 to 8 (Table 11). pH is also potentially subject to the same kinds of biogenic fluctuations as DO due to the consumption of carbon dioxide by aquatic plants and algae during the day (photosynthesis) and net production of carbon dioxide at night. If respiratory oxygen consumption is predominant then in coastal waters dissolved oxygen concentrations will be lower become lower along with pH values. Here oxidised acid sulphide soils were evident in all wetlands, particularly during late dry season when wetlands were well drained and the soils were damage by feral pigs or cattle activities. In some areas there was evidence of iron bacteria present, which is general common in coastal waters in Queensland (Luke et al. 2017)

Water column nutrients: TN ranged from 699 to 6775 µg L^{-1} across locations while TDN ranged from 560 to 6306 µg L^{-1} indicating that most nitrogen present was in the dissolved fraction (Table 12). Exceptionally high nitrogen concentrations were measured at RH1 in July 2018 and Littabella Up (RH3) in June 2019 when compared with other sites and survey times. TP ranged from 8 to 114 µg L^{-1} while TDP ranged from 5 to 47 µg L^{-1}. TN:TP and TDN:TDP ratios indicated that nitrogen was generally in excess for all locations (i.e. >> 16:1). Chlorophyll-*a* concentrations ranged from 0.95 to 29.33 µg L-1. Chl-*a* concentration was highest in the post-wet (March 2018) and when TDN concentrations were in excess. TSS ranged from 6.9 to 140 mg L^{-1} (Table 12).

Table 11. Water quality measured during fyke net surveys.

Trip	Code	Location	Depth	Temp	Conductivity	Dissolved Oxygen		pH
			(m)	(°C)	(mS cm^{-1})	(%sat)	(mg L^{-1})	
Mar-18	RH1	Round Hill Reserve (fenced)	0.05	26.29	2	68.6	5.66	6.67
			0.1	30.98 ± 1.79	1 ± 1	84.7 ± 24.47	6.87 ± 2.34	6.73 ± 0.13
			0.15	24.36	1	27.3	2.49	7
			0.25	26.25 ± 2.27	1 ± 0	45.76 ± 30.48	3.89 ± 2.51	6.69 ± 0.26
			0.3	25.16	1	27.5	2.15	6.22
			0.5	27.8	1	37.1	3.27	6.79
Jul-18	RH1	Round Hill Reserve (fenced)	0.1	22.34 ± 4.08	59 ± 2	114.53 ± 6.09	8.57 ± 0.89	7.28 ± 0.17
	RH2	Littabella Down (reference)	0.1	28.8 ± 0.88	76 ± 3	105.97 ± 40.25	7.5 ± 0.47	7.5 ± 0.47
	RH3	Littabella Up (reference)	0.1	24.45 ± 1.01	63 ± 3	86.03 ± 5.42	5.41 ± 0.25	5.38 ± 0.2
	RH4	Round Hill (reference)	0.1	17.55 ± 0.71	77 ± 2	65.87 ± 1.46	5 ± 0.32	8.08 ± 0.1
	RH5	Eurimbula NP (reference)	0.1	23.03 ± 3.16	43 ± 7	90 ± 15.82	6.55 ± 1.1	8.11 ± 0.44
Jun-19	RH1	Round Hill Reserve (fenced)	0.1	25.1 ± 1.55	68 ± 5	95.45 ± 22.79	6.48 ± 1.9	6.19 ± 1.35
			0.15	27.05 ± 3.05	64 ± 3	93.75 ± 9.55	5.78 ± 0.2	5.04 ± 2.67
	RH2	Littabella Down (reference)	0.1	22.27	84	77.6	5.14	7.92
			0.15	21.64	82 ± 3	116.97 ± 29.57	7.75 ± 1.15	7.95 ± 0.25
	RH3	Littabella Up (reference)	0.1	27.46	56	94.1	6.44	6.77
			0.15	-	-	111.2	7.08	7.09
	RH4	Round Hill (reference)	0.1	26.32 ± 2.19	77 ± 1	117.23 ± 26.74	7.34 ± 1.29	7.7 ± 0.13
			0.3	22.87	78	83.2	5.88	7.63
	RH5	Eurimbula NP (reference)	0.59	22.34	69	58.9	5.19	7.68
			0.7	25.81	76	46.8	3.48	7.86

Table 12. Water sample results for nutrients, total suspended sediments and chorlophyll-*a* from wetlands in this study

Parameter	Location	RH1			RH2	RH3	RH4	RH5	
	Trip	Mar-18	Jul-18	Jun-19	Jun-19	Jun-19	Jun-19	Jul-18	Jun-19
Total Nitrogen	$\mu g\ L^{-1}$	1606.7 ± 645.5	4108.5 ± 3771	1059	699	1533	922	1797 ± 553.4	967
Total Phosphorus	$\mu g\ L^{-1}$	78.3 ± 33.5	74.5 ± 55.9	56	30	8	29	72.3 ± 5.8	111
Total Dissolved Nitrogen	$\mu g\ L^{-1}$	763 ± 52.3	3576.5 ± 3860.1	1015	673	1517	919	1386.3 ± 615.9	560
Total Dissolved Phosphorus	$\mu g\ L^{-1}$	16.3 ± 1.5	23.5 ± 16.3	5	15	5	10	28 ± 16.8	17
Particulate Nitrogen	$\mu g\ L^{-1}$	-	532 ± 89.1	44	26	16	3	410.7 ± 73.2	407
Particulate Phosphorus	$\mu g\ L^{-1}$	-	7 ± 8.5	51	15	3	19	17 ± 7.5	94
Ammonia	$\mu g\ L^{-1}$	-	3144 ± 4420.8	-	-	-	-	32.5 ± 36.1	-
Nitrate	$\mu g\ L^{-1}$	-	7 ± 7.1	-	-	-	-	6.7 ± 3.1	-
Nitrite	$\mu g\ L^{-1}$	-	1.3 ± 1.1	-	-	-	-	< 1	-
Filterable reactive phosphorus	$\mu g\ L^{-1}$	-	16.5 ± 7.8	-	-	-	-	11 ± 10.4	-
Chlorophyll-*a*	$\mu g\ L^{-1}$	19.1 ± 7	16 ± 18.8	12.06	0.95	1.37	6.03	4.4 ± 0.3	16.73
Phaeophytin-*a*	$\mu g\ L^{-1}$	11.2 ± 8.8	3.7 ± 0.5	8.89	< 0.2	0.78	1.36	1.7 ± 0.5	4.64
Total Suspended Solids	$mg\ L^{-1}$	53.7 ± 5.7	77 ± 32.5	25	6.9	16	7.9	28.3 ± 0.6	140

Fish surveys (fyke net)

A total of 12 fish species were detected across the wetland sites (Table 13). The small bodied fish Gobiidae (true goby) and *Pseudomugil signifier* (pacific blue-eye) were the most abundant species encountered. Two exotic species; *Oreochromis mossambicus* (mozambique tilapia) and *Gambusia affinis* (eastern gambusia) were present at Round Hill (RH1) and Daniels (RH4), but were not detected at Littabella Down (RH2). *Macrobrachium Rosenbergii* (giant river prawn) was present at Round hill (RH1) only, while *Metapenaeus bennettae* (greasy-back prawn) and *M. macleayi* (school prawn) were present at Daniels (RH4). Species richness was highest at the low position at Round Hill (RH1) with 15 species present. The following 7 species were exclusively present at this site: *Hypseleotris compressa* (empire gudgeon), *Melanotaenia splendida* (eastern rainbowfish), *Mugil cephalus* (sea mullet), Ambassidae (glassfish), *Anguilla reinhardtii* (longfin eel), and *Elops hawaiensis* (hawaiian giant Herring).

Ordination by non-metric dimensional scaling (nMDS) was conducted on a Bray-Curtis dissimilarity matrix of fish presence/absence data (Figure 47). Much of the spread in multivariate space could be explained by season, with March 2018 (post-wet season) plotting separately from July 2018 and June 2019 (dry season). This can be explained by the presence of numerous fish species at Round Hill (RH1) in March 2018 which were not detected at other locations or sampling times. Multivariate dispersions were equal for trip and tidal position but not equal across locations (Table 14). Distance to centroids were significantly higher for RH1 (Round Hill) indicating a much larger spread of samples in multivariate space for this location (PermDisp: $p = 0.0032$; TukeyHSD: $p = 0.0025$). Permutational analysis of variance indicated that there was a significant difference in fish populations between trips (PERMANOVA: P(perm) = 0.0001) and between tidal position (PERMANOVA: P(perm) = 0.0231) (Table 15).

Table 13. Number of fish and crustaceans caught during fyke net surveys at Daniels, Littabella Down, and Round Hill. Fyke nets were positioned at 'Low' and 'High' tidal positions. Asterisk (*) indicates exotic species.

Species	Common name	RH1 Round Hill		RH2 Littabella Down		RH4 Daniels	
		High	Low	High	Low	High	Low
Ambassidae	Glassfish	1	80	0	-	1	0
Anguilla reinhardtii	Longfin Eel	0	2	0	-	0	0
Elops hawaiensis	Hawaiian Giant Herring	0	1	0	-	0	0
*Gambusia affinis**	Eastern gambusia*	11	159	0	-	30	10
Gobiidae	True goby	4	168	501	-	131	212
Hypseleotris compressa	Empire gudgeon	4	17	0	-	0	0
Melanotaenia splendida	Eastern rainbowfish	1	31	0	-	0	0
Mugil cephalus	Sea mullet	0	7	0	-	0	0
Mugilidae	Mullet	1	16	0	-	1	0
*Oreochromis mossambicus**	Mozambique tilapia*	0	1	0	-	8	0
Pseudomugil signifer	Pacific blue-eye	0	88	76	-	385	49
Redigobius bikolanus	Speckled goby	0	5	0	-	0	0
Metapenaeus bennettae	Greasy-back Prawn	0	0	0	-	73	1
Macrobrachium Rosenbergii	Giant river prawn	25	23	0	-	0	0
Metapenaeus macleayi	School Prawn	0	0	0	-	29	0
Australoplax tridentata	Tuxedo Crab	0	0	1	-	0	0
*Bufo marinus**	Cane toad*	0	5	0	-	0	0

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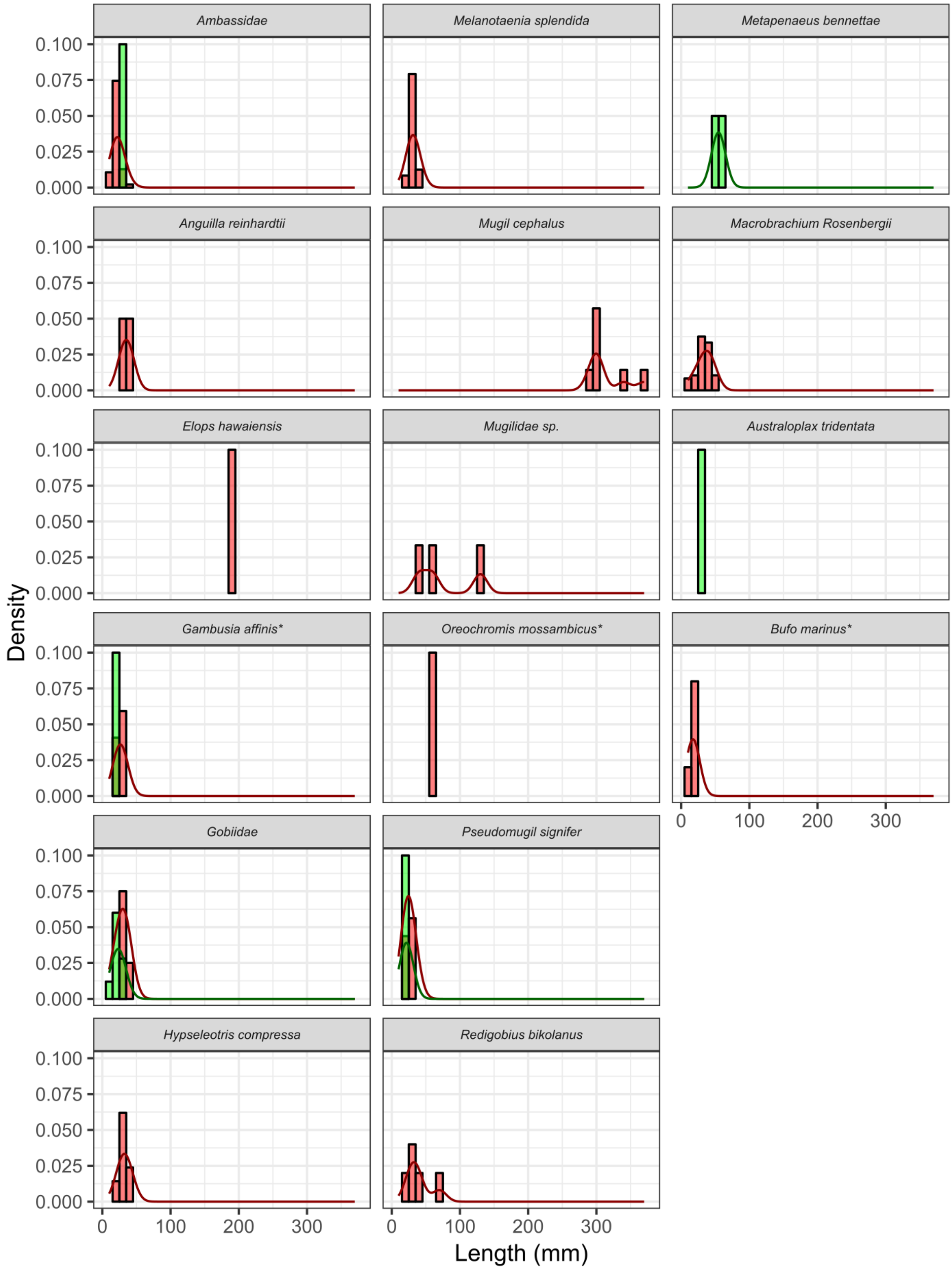

Figure 46. Fish size densities from fyke net surveys in March 2018 (red) and June 2019 (green). Bin width = 5 mm. Solid lines indicates kernel density estimates.

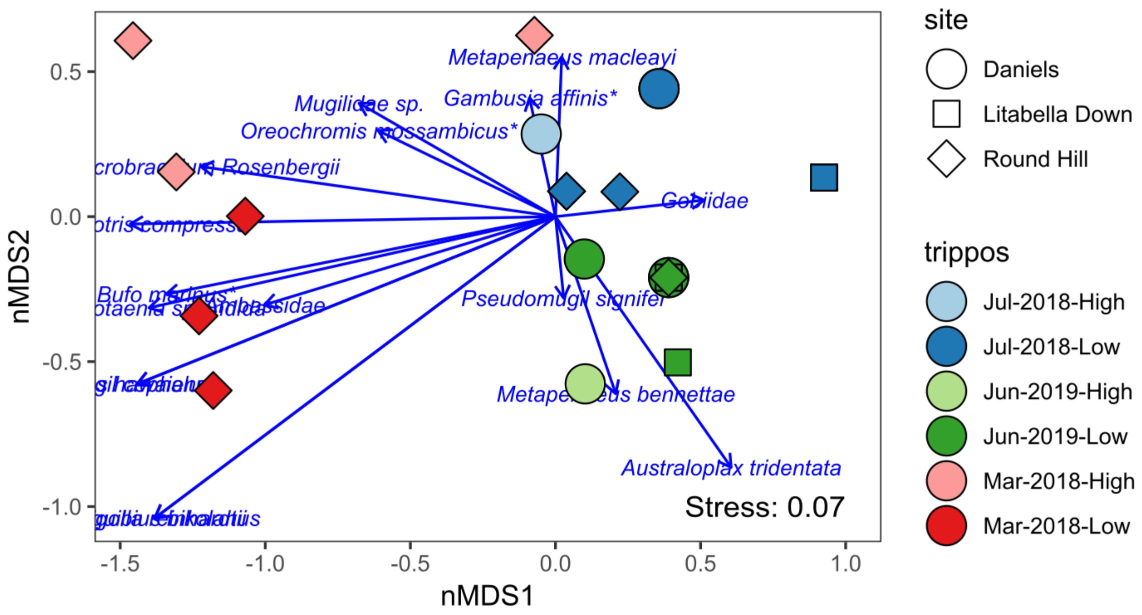

Figure 47. Results from fyke net fish sampling around wetlands. RH1 (Round Hill), RH2 (Littabella Down), and RH4 (Daniels). Non-metric multidimensional scaling (nMDS), presence/absence data. Bray-Curtis dissimilarity matrix. Stress = 0.07.

Table 14. Permutation tests for homogeneity of multivariate dispersion (PERMDISP) between Trip (Mar-2018, Jul-2018, Jun-2019) Location (Round Hill, Littabella, Daniels) and Position (Low, High). Permutation: free. Number of permutations: 9999.

	df	*SS*	*MS*	*Pseudo-F*	*P (Perm)*
Trip	2	0.14003	0.070016	3.0879	0.0666
Residuals	20	0.45349	0.022674		
Location	2	0.32848	0.16424	7.9923	**0.0036**
Residuals	20	0.41099	0.02055		
Position	1	0.04301	0.043012	0.979	0.3422
Residuals	21	0.92265	0.043936		

Table 15. Permutational multivariate analysis of variance (PERMANOVA) of fish presence-absence from fyke net surveys. Permutation: free. Number of permutations: 9999. Terms added sequentially (first to last).

	df	*SS*	*MS*	*Pseudo-F*	R^2	*P (Perm)*
Trip	2	2.3858	1.19291	6.8845	0.37295	**0.0001**
Location	2	0.61	0.30498	1.7601	0.09535	0.0702
Position	1	0.4558	0.45579	2.6304	0.07125	**0.0231**
Residuals	17	2.9457	0.17327	0.46046		
Total	22	6.3972	1			

Stepwise model selection for distance based redundancy analysis (dbRDA) indicated that water temperature, electrical conductivity, and dissolved oxygen best explained the differences in fish community data for fyke net surveys (dbRDA: $adj\text{-}R^2 = 0.3475$, $F_{3,16} = 4.3723$, $p = 0.001$) (Figure 48). Removing pH from the model did not significantly change the outcome ($F = 1.0522$ $p = 0.3977$). dbRDA axis 1 was dominated by electrical conductivity and dissolved oxygen, and significantly explained 63.5 % of fitted, and 28.6 % of total model variance ($F_{3,16} = 8.3255$ $p = 0.001$). dbRDA axis 2 was dominated by water temperature and significantly explained 32.4 % of fitted, and 14.6 % of total model variance ($F_{3,16} = 4.2558$, $p = 0.009$). Electrical conductivity significantly explained the variance on axis 1 ($F_{3,16} = 7.7713$, $p = 0.001$), Water temperature significantly explained the variance on axis 2 ($F_{3,16} = 3.3913$, $p = 0.008$). Dissolved oxygen was retained in the model but only partially explained variance and was not significant ($F_{3,16} = 1.9544$, $p = 0.100$). With more data the frequency and duration of inundation should be investigated to decipher fish community patterns in these types of coastal wetland systems (Sheaves et al. 2019).

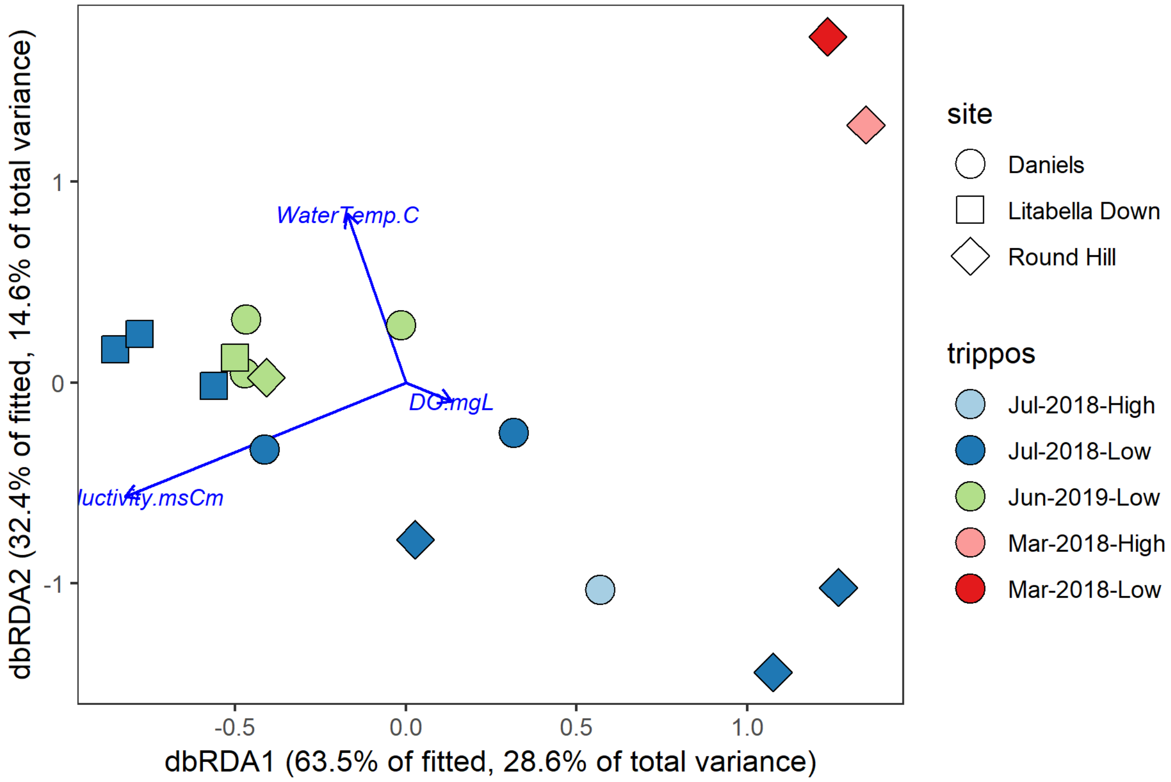

Figure 48. Distance based redundancy (dbRDA) plot illustrating reduced model based on fish community surveyed with fyke nets and the fitted environmental variables.

Herpetofauna survey

Fauna surveys detected a total of twelve species present across the five wetland sites (5 skink, 1 gecko, 5 frogs, and 1 snake species; Figure 49). The number of individuals observed at each location is presented in Figure 50. Overall, there was low captures all around (to be expected for a winter survey and for a short duration). The highest diversity and abundance was from RH3, which ironically is one of the most disturbed sites (with cattle and pig access). The size and weight of a subset of individuals collected in funnel traps were recorded (Table 16). The introduced cane toad (*Rhinella marina*) was detected at three sites.

A)

B)

Figure 49. A) Southern spotted velvet gecko (*Oedura tryoni*); and B) great brown broodfrog (*Pseudophryne major)* captured in wetlands (Photos source Dr E Nordberg, JCU).

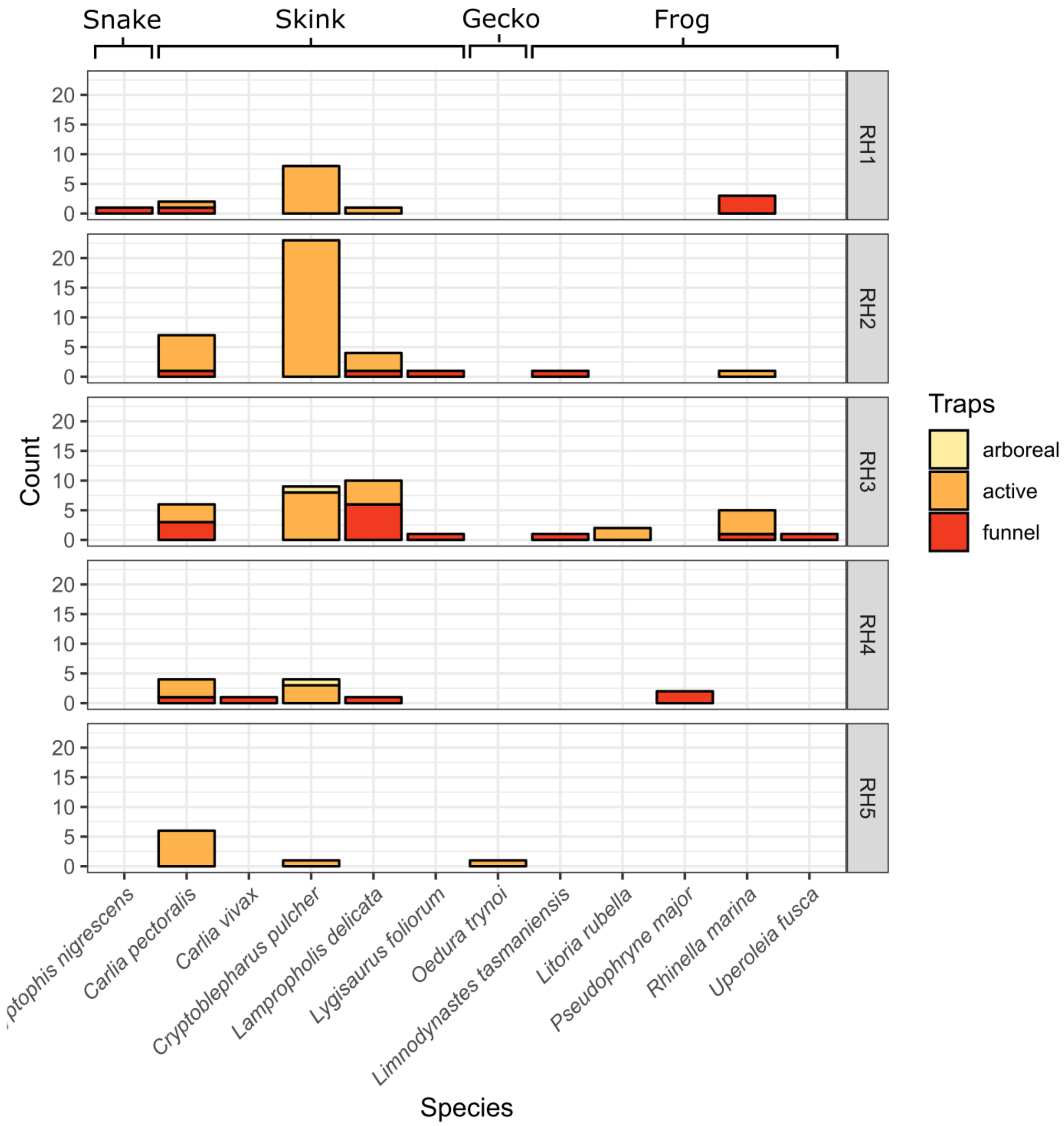

Figure 50. Number of individuals encountered during herpetological surveys.

Table 16. Size and body weight of lizards and frogs collected in funnel traps during the survey of saltmarsh wetlands. The snout-ventral length (SVL) of frogs and lizards and the total length (TL) of lizards was measured. Mean and standard deviation of the population are presented for sites where more than one individual was encountered.

Location	Species	Common name	TL (mm)	SVL (mm)	Mass (g)	n
RH1						
	Carlia pectoralis	Open-litter rainbow skink	113	42	5.4	1
RH2						
	Carlia pectoralis	Open-litter rainbow skink	115	42	5.6	1
	Lampropholis delicata	Delicate skink	49	19	-	1
	Lygisaurus foliorum	Tree-base litter-skink	63	27	0.4	1
	Limnodynastes tasmaniensis	Spotted Marsh Frog	-	27	2.4	1
RH3						
	Carlia pectoralis	Open-litter rainbow skink	107 ± 13.5	44 ± 1.7	3.3 ± 1.8	3
	Lampropholis delicata	Delicate skink	89 ± 15.1	36.5 ± 3.8	2.9 ± 2.2	6
	Lygisaurus foliorum	Tree-base litter-skink	60	30	0.6	1
	Limnodynastes tasmaniensis	Spotted Marsh Frog	-	23	1.2	1
	Uperoleia fusca	Dusky Toadlet	-	19	4.9	1
RH4						
	Carlia pectoralis	Open-litter rainbow skink	97	37	5.1	1
	Carlia vivax	Tussock rainbow-skink	122	46	5.1	1
	Lampropholis delicata	Delicate skink	35	27	3.6	1
	Pseudophryne major	Large toadlet	-	28 ± 1.4	4.3 ± 0.1	2
RH5						
	Oedura trynoi	Southern spotted velvet gecko	139	83	12.8	1

Waterbird survey

Waterbird assemblages were surveyed using camera traps at the Round Hill Reserve (RH1) in March 2018 (post-wet season) and in July 2018 (dry season) at all survey locations except for the Eurimbula NP reference (RH5), due to logistical constraints.

Table 17 provides a list of bird species present at each survey location and whether they are listed as a wetland indicator bird species of the Seventeen Seventy-Round Hill Fish Habitat Area (FHA; Queensland Government, 2019). Brolgas (*Grus rubicunda*) are listed as a wetland indicator species for the FHA (Queensland Government 2019).

Brolgas (*Grus rubicunda*) were observed during the March 2018 surveys (two adults and one juvenile) in the fenced Round Hill Reserve area (RH1; Figure 51), suggesting that the Round Hill Reserve may provide important breeding and foraging habitat for this species. However, continued surveys are required to determine the relative importance of the fenced Round Hill Reserve as Brolga foraging/breeding habitat in comparison to other wetlands throughout the region. Brolgas are omnivorous, foraging on a diversity of resources including fish, snails, bivalves, terrestrial vertebrates, and insects (Barker and Vestjens 1989). White-faced herons (*Egretta novaehollandiae*) were also observed foraging in the Round Hill Reserve (RH1) where fyke nets were deployed (Figure 52). White-faced herons forage primarily on fish, crustaceans, and insects (Barker and Vestjens 1989).

Table 17. Bird species list from camera trap photos.

Survey Period	Survey Location	Site Code	Number of taxa detected	Common name	Latin name	Round-Hill FHA Wetland Indicator Species?
Mar-18	Round Hill Reserve	RH1	6	White-faced heron	*Egretta novaehollandiae*	No
				White-necked heron	*Ardea pacifica*	Yes
				Brolga	*Grus rubicunda*	Yes
				Black-necked stork	*Ephippiorhynchus asiaticus*	Yes
				Wedge-tailed eagle	*Aquila audax*	No
				Egret	Unable to identify to species in photographs, but possible Eastern great egrets (*Ardea alba modesta*) or Intermediate egrets (*Ardea intermedia*)	Yes
Jul-18	Round Hill Reserve	RH1	0	-	-	-
	Littabella Reference	RH2	0	-	-	-
	Littabella Reference	RH3	1	Black-winged stilt	*Himantopus himantopus*	No
	Round Hill Reference	RH4	0	-	-	-

Figure 51. Two adult Brolgas (*Grus rubicunda*) and a juvenile in the fenced Round Hill Reserve (RH1; March 16th, 2018).

Figure 52. Fish and waterbird surveys at Round Hill Reserve (RH1; March 24th, 2018): a) fyke-net deployment, b) fish identification and measurement, c) and d) White-faced herons (*Egretta novaehollandiae*) foraging in the wetland where fyke-net was deployed (red-dashed circles identify individual birds).

High intertidal coastal saltpans provide important habitat for waterbirds (Velasquez 1992, Rocha et al. 2017, Lei et al. 2018). Overall, in this study more waterbird species were detected during the post-wet season surveys (March 2018) in comparison to dry season surveys (July 2018). Water availability and depth are key factors determining use of wetlands by waterbirds (Sebastián-González and Green 2014), and a decrease in water availability during the dry season many explain the pattern here of fewer waterbirds detected in July 2018.

Overall, the services and values provided by saltpan areas in Queensland are generally unknown, with only some data relating to fish habitat use (Russell and Garrett 1983, Hollingsworth and Connolly 2006). The data indicate that a range of species probably utilise these supratidal areas, which does give rise to their protection and conservation even if they only support a subset of species known more broadly in coastal estuaries, for example fish (Sheaves and Johnston 2009). For example, while water birds use these supratidal areas, there has been no examination of the benthic invertebrate community that might be supporting bird foraging. Under sea level rise scenarios however, the frequency and duration of connection with downstream estuaries might increase giving rise to model that these areas might become important future habitat, highlighting the need to still conserve and protect them from feral pig and cattle impacts.

In addition to biodiversity values itself, there are other beneficiaries (see Figure 1) of these wetlands, and indeed in a restored and enhanced state. A key motivation for the fencing program here was to keep feral pigs out of the national park and to allow for the plant and birdlife communities to return, and while there was some evidence of this in the initial years, clearly the need to maintain the fences and keep out cattle access is necessary. There are also possible first Nation people values that have not been included here. Identifying and engaging local indigenous groups in the management of these wetlands is an important next step, and can open a pathway for lasting positive environmental outcomes and shared learnings (Waltham et al. 2018).

4.0 SYNTHESIS

4.1 Efficacy of feral pig exclusion fencing

Conservation fences that are designed and built to protect or restore sensitive areas continue to be used, providing an effective means of controlling movement of species within/across the landscape. Fencing wetlands to control for feral animal access to wetlands is increasing in northern Australia, and can be an effective measure of the treat of damage caused to wetlands. However, depending on the identified value of the wetland, this mitigation measure has received mixed conclusions in coastal wetlands owing to the differences in conservation values identified. For example, fencing for the purposes of fish habitat in northern Queensland was considered less effective where wetlands are ephemeral and likely to dry out before the start of the wet season rain (Doupé et al. 2010). Conversely, fencing wetlands for protection of freshwater turtles in the same dry landscape has been suggested as important and necessary, preventing pigs from accessing wetlands to consume estivating turtles (Waltham et al. In Press).

For Round Hill Reserve, the installation of the conservation fence has been an important and necessary response to begin to protect and enhance the services and values of this wetland area. The evidence of historical damage caused by pigs to the wetland remains, as scares in the wetland soil and bare areas within the fence that haven't yet recovered with vegetation. During the first year of surveys, in the post wet season survey, this wetland had responded well with obvious regrowth of water sedge plants, fish and birds using the wetland (these observations were made as part of the initial site reconnaissance). Compared to the other wetlands in the region that were heavily impacted, with conceivably few fish and bird life.

An important determination in this study has been that while the fence has been effective in excluding pig access, the land tenure in place with the adjacent landholder that includes allowing cattle into the fenced wetland during the late dry season months seems counterproductive towards achieving the values and services set for this wetland restoration project. The current arrangement of allowing cattle into the wetland for the purposes of feeding directly on saltmarsh continues to cause damage to the wetland. This damage was most evident by the direct reduction and trampling of saltmarsh areas in the wetland, which during the last survey in July 2019 areas of the wetland appeared to be affected by acid soil oxidation with low pH water recorded and very fish and birds present. While the data here doesn't quite support statistically this direct impact on the wetland, vegetation surveys and UAV mapping provides at least some important evidence to revisit the land tenure arrangement, to more appropriately management this arrangement – or indeed, change the tenure arrangement to restrict total this access arrangement. This value of the wetland directly to local flora and fauna, blue carbon additionality services, and the broader value the wetland has in its location and connection with downstream fish habitat areas and internationally recognised wetland of importance, and tourism and local economy, raises the need to evaluate the economic trade here against cattle accessing the wetland by an adjacent landholder. Evaluating land tenure arrangements in state park is a wider challenge for managers, when considering fire break management and access to state park through private property, but in this case here, the investment of fencing the wetland preventing the threat of feral pig damage to then allow cattle in to damage the vegetation and soils requires review.

4.2 Maintenance of fences in coastal areas

Installation of conservation fencing is expensive and requires on-going maintenance as fences can be easily damaged following sever weather events, fire, stock or fallen timber branches (Negus et al. 2019). In addition, the location of the Round Hill fence at the upper tidal boundary of the national park resulted in an additional pressure on fences, corrosion of the metal pickets and fencing wire used here. In just three years following the installation of the fences, long sections more than several hundred meters has corroded and need replacing (Figure 53). Attempts to replace fencing was completed using stainless-steal pickets and fencing wire, but has been at a considerable cost.

Inspection of conservation fencing is necessary every few months to ensure they remain in good working order. Any repair works could then be attended to quickly and with minimal effort, avoiding more expensive and larger repair works that may eventuate. In addition, inspect of fences should be completed immediately following the wet season, after serve storms or fire which may compromise fences. For example, high rainfall in early 2017 resulted in the Round Hill Reserve wetland flooding, which washed large timber logs into the fence, which accumulated along with leaves and twigs (Figure 54). This accumulation of debris can result in a hot spot if a fire were to pass through the area. The collection of wood fuel would burn hot, and potentially compromising the aluminium fencing and timber posts used to tighten the tension. To reduce this risk requires manually removing the timber away from the fence, either by hand or machinery. These works could require many days of work and machinery, but would further protect the investment in the fence.

Figure 53. Round Hill fence after 3yrs exposed to tidal wetland soils. Corrosion compromises the integrity of the base of the fence, and potentially permitting feral pig access to the wetland. Fencing maintenance will be on-going in coastal areas subject to harsh marine conditions, but is presumably costly.

Figure 54. Rain and flooding causes accumulation of timber and organic material along fence lines. This becomes a major problem for fences during fire when the timber burns hot causing damage to fences requiring expensive maintenance work

4.3 Conservation fence improvement

Emerging evidence suggests that fencing effects non-target species, for example, by disruption to dispersal processes, and increased mortality (via increased exposure to unfavourable conditions or predators; Spencer 2002)). By far these impacts are greatest on vagile animals which have evolved behavioural life history traits that allow them to inhabit landscapes characterised by spatial and temporal variability, and are therefore susceptible to limited access to resources or responding to local pressures (predation, climate conditions). With every conservation fence there exists the opportunity to evaluate the design efficacy, and implement ongoing modifications as part of a process of continual improvement (Loarie et al. 2009).

Recent examples where conservation fences have impacted on non-target species is in northern Australia where freshwater turtles, that have a terrestrial movement requirement during certain times of the year, are restricted by fencing installed around wetlands. In this case, the services and values provided by wetlands to turtles becomes a barrier in place over protection of the wetland for other values or services. For example, in Round Hill Reserve, the wetland fencing for protection of vegetation communities would present a challenge for prospecting turtles trying to move across the landscape. In a manipulative study where fence designs were modified by removing a small piece of vertical wire and creating a gate for passage (Figure 55), turtles were found to rapidly locate gates after prospecting and trying to fit through meshing areas (Waltham et al., In review). This simple design change can be implemented retrospectively to fences, and would increase the value of wetland fencing projects to include turtle conservation.

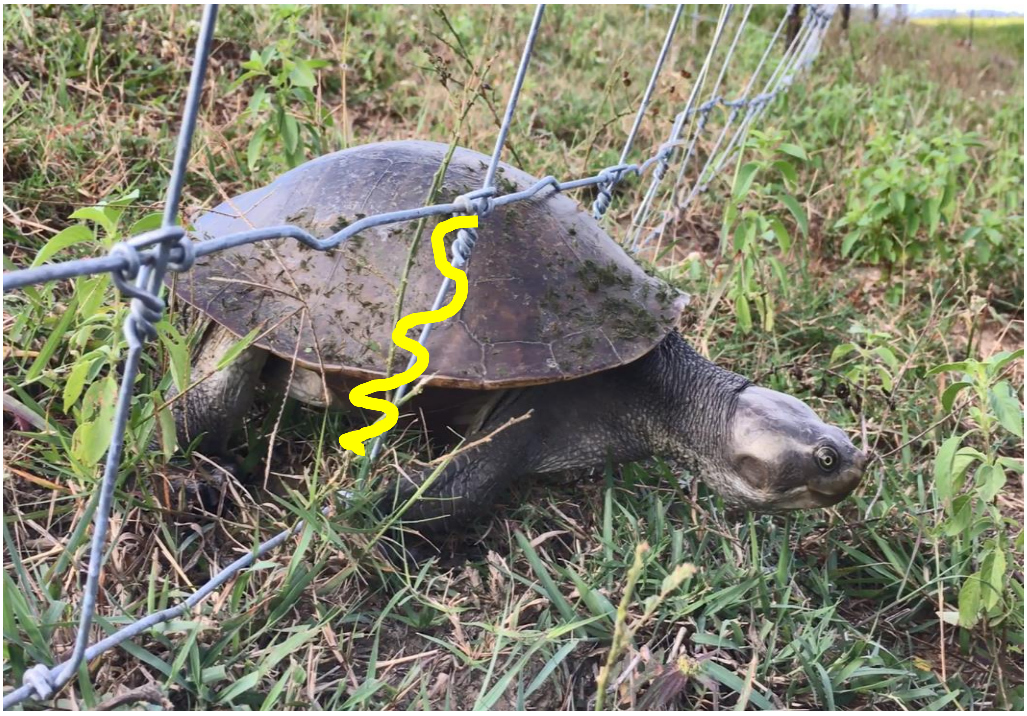

Figure 55. Fences around wetlands will prevent feral pig access, however, for freshwater turtles that move around the landscape, fences pose an obvious barrier. A simple conservation measure that allows turtles to move across floodplains is to remove a small piece of vertical wire, shown here, allowing prospecting turtles to pass through (Waltham et al., in review).

Conservation fences restrict movement and access for larger fauna species moving across landscapes or even from direct access to wetlands. An example are kangaroos that inhibit the Round Hill Reserve, the wetland fence in this case would be preventing access to feeding and shelter areas within the fenced area. Indeed, if they are able to access the wetland over the fence, the ability to move outside the fence is restricted or impossible. As part of a trial, Queensland Parks and Wildlife Service Rangers' constructed an earth ramp either side of the fence where kangaroos could pass over the fence (Figure 56). The ramp can be approached from either side of the fence, and effectively allowing kangaroos to clear the fence, while still preventing feral pigs and cattle from clearing the fence (Figure 57). A motion triggered camera was installed near to the ramp, and recorded at least one incident of a wallaby that is believed to have used the ramp, and successfully clearing the fence. This trial, similar to the clipping of fencing wire for turtles to pass through fences, provides evidence that continual adjustments and modifications applied to conservation fencing advances the ability to manage the threat while also maximizing the ecosystem value and services provided through wetland restoration. Maintenance of the ramp is minimal, a few hours of manual work to reconfigure the ramp and ensure the fence hasn't been compromised following rainfall/flooding or animal trampling.

Figure 56. Fences around wetlands also prevent animal movement, such as kangaroos, from accessing resources in the landscape. In Round Hill Reserve, Queensland Parks and Wildlife Services constructed an earth ramp to trial whether kangaroos would use it to move across fence boundaries by jumping over it, while at the same time preventing access by feral pigs and cattle (photo S Jackson, Queensland Government)

A)

B)

Figure 57. Motion detection camera in Round Hill Reserve capture: A) a wallaby using the earth ramp to pass over the fence; and B) cattle are not able to pass over fences when using the ramp. (Photo S Jackson, Queensland Government)

4.3 Limiting cattle stock access to wetlands

The impact of livestock on terrestrial ecosystems is well known (Daskin and Pringle 2016), however, livestock impacts on saltmarsh are less known (Hobbs 2001, Davidson et al. 2017, Muenzel and Martino 2018). In a comprehensive review of the literature, the key properties affected by livestock grazing, include increased plant richness, reduced invertebrate richness and herbivorous invertebrate abundance, reductions in plant material and altered soil condition and carbon storage (Davidson et al. 2017). The results of this review outlined that the use of salt marshes by livestock affects multiple ecosystem services, which creates a situation where trade-offs in services need to be discussed with stakeholders. In the case of Round Hill Reserve, allowing cattle access inside the fenced wetland area seems entirely counterproductive in terms of investing in funding to install the fence and to continue maintaining this asset in reducing the threat of feral pig damage to the wetland. By preventing feral pigs from accessing the wetland, to only then allow cattle access it under past land agreements requires an overview, not just for this location, but across Queensland more broadly (Figure 58). Evidence of cattle impact on saltmarsh areas is evident in every location where cattle are permitted access to these wetlands.

A)

B)

Figure 58. Clear evidence of the impact from cattle grazing in saltmarsh areas: A) Round Hill Reserve showing cattle grazing and quad bike impact on saltmarsh inside the fenced wetland, compared to downstream areas where cattle are not able to access; and B) example from Mungalla wetland where cattle access saltmarsh areas.

There is a case that cattle consuming saltmarsh is an effective means of transferring stored carbon in this plant into cattle production (usually for human food consumption). While this is plausible, and is the case elsewhere as sheep graze saltmarsh (Laffaille et al. 2000), there are important secondary consequences of cattle in wetlands. The most evident, and was observed in Round Hill Reserve, is the reduced canopy height of saltmarsh because of direct consumption of saltmarsh. Here particularly during the late dry season survey in 2018 the saltmarsh length was smallest where cattle had access to marsh areas, including the fenced wetland in Round Hill Reserve. The only wetland that had minimal direct damage to saltmarsh from cattle was RH4, which was on a private property where the wetland area was fenced from access and with cattle not permitted inside. Grazing in saltmarsh areas has been also linked to increased soil temperature, as a result of reduced shading or even just reduced vegetation (Persico et al. 2017), but also compacts soils and anaerobic respiration – a consequence of increased soil temperature is higher rates of evaporation and thereby higher soil salinity (which was shown here) (Davidson et al. 2017).

Grazing also alters saltmarsh patch heterogeneity through direct consumption or trampling on vegetation. This process results in fewer arthropods (Puzin and Pétillon 2019), and invertebrates (Kneib 1984b), all of which decreases both available plant and food resources for juvenile fish and crustaceans accessing saltmarsh areas (Minello and Zimmerman 1983, Hollingsworth and Connolly 2006, Mazumder et al. 2006), but also water birds (Gu et al. 2018).

4.4 Invasive fish management

The presence of tilapia (*Oreochromis mossambicus*) in the wetlands here were surprising and represent a new record for their location in Queensland waters. Tilapia are common on floodplains on the Burdekin and indeed constitute a large biomass (Davis et al. 2017, Waltham and Fixler 2017). This species is known to be a menace, it is both oxygen and temperature tolerant, can impact on native species (Doupe et al. 2009a) and will out compete native species for resources and habitat (Russell et al. 2012). Although predominately a freshwater species, it is known to deal with saltwater, in fact in some parts of its distribution it present in hypersaline conditions (Whitfield et al. 1981). Because of this tolerance to saltwater, it is considered that this species spreads its range during summer flooding, and is apparently moving south along the Queensland coast toward and into New South Wales.

Removing tilapia from Round Hill Reserve will be difficult given it seems to have a well-established population based on the numbers caught during sampling here. It is possible that removing this species from the Reserve is impossible, even dedicated efforts to fish or net this species out of the area will be near impossible given the size of the area and sampling around mangrove pool areas is difficult. Fortunately, tilapia where only present at one wetland site, so it is possible that presently the population is spatially restricted to a small area. The best cause of action at present will be an active education program in the Reserve and local community to destroy this fish if caught, which is required under Queensland legislation. The education would be an important means of ensure humans do not translocate this fish species, which would result in further spread.

4.5 Values based approach to wetland conservation

The characteristics of high value coastal wetlands provide the basis for predictive model construction. Bayesian Network (BN) models are becoming more commonly used in natural resources management (Nyberg et al. 2006, Gawne et al. 2011, McDonald et al. 2015). These tools let managers define cause-effect relationships directly through a simple graphical structure that are built around data or information from authoritative experts and/or collected data. The advantage of this approach is it presents likely probability outcomes for end use in the decision framework that is superior where it is not supported by on a basis of bias opinions or ideas, in arriving at the final management decision.

In the scenario here, a BN model was established to examine the values and services provided by wetlands that are fenced from pigs and cattle access in total, unfenced wetlands and fenced wetlands but where cattle are allowed to still access the protected wetland for a specific use. The data collected in this case study provide a superior opportunity to examine the scenario presented, to provide an easily measurable set of parameters that can be used as input into a simple Habitability Bayesian Belief Network (H-BBN). For the purposes of the BN here, the input values for a scenario of wetland fencing was examined for an estimate of fish habitability. The input variables are qualitative (Table 18), and can be collected at one time without need for long term data logging or extensive surveys (although tidal connectivity is best assessed via a pressure logger left in place for at least one lunar tidal cycle), providing the ability to assess a large range of sites rapidly. However, this is best seen as an initial screening tool as the a more detailed hydrodynamic model would require far more resources and data (Abbott et al. 2020). A set of nodes representing the major drivers on fish assemblages occupying wetlands were examined. These included habitat cover (here saltmarsh), food availability, (here invertebrate), water quality, tidal connectivity and water permanency. The cause-effect relationships between each node were manually entered, based on a probability between 0% (rare or never possible) to 100% (always or certain cause-effect relationship). The description details outline in Table 17 provide insight into the decisions made, where were based on the study team expertise, data from this study or supported by the literature.

Table 18. Simple qualitative input variables for the H-BBN

Category	Levels	Description
Management	Fenced	Fence permanently encompassing wetland, no damage and in full working order
	Unfenced	No fence around wetland
	Fenced Open	Fence permanently encompassing wetland, no damage and in full working order, except fence manually opened for cattle feeding
Saltmarsh Cover	High	>70% cover in 1mx1m quadrat
	Impacted	>70% cover in 1mx1m quadrat with evidence of disturbance by pigs or cattle
	Bare	0% cover in 1mx1m quadrat
Invertebrate community	High	>20 burrows 1mx1m quadrat
	Medium	Between 10 and 20 burrows 1mx1m quadrat
	Low	<10 burrows 1mx1m quadrat
Water quality	High	Water quality within acceptable guidelines for aquatic ecosystem protection
	Medium	Water quality generally accepted for aquatic ecosystem protection
	Degraded	Water quality below guidelines for aquatic ecosystem protection
Tidal connectivity	Always	Wetland is always connected at every high tide
	Often	Wetland is connected every spring high tide (4-10 times a year)
	Rare	Wetland connected <4 times a year
Water permanency	High	Water always in wetland
	Medium	Water present in wetland 60-80% of time
	Low	Water present in wetland <60% of time
	Absent	Water drains each low tide, exposing wetland soils

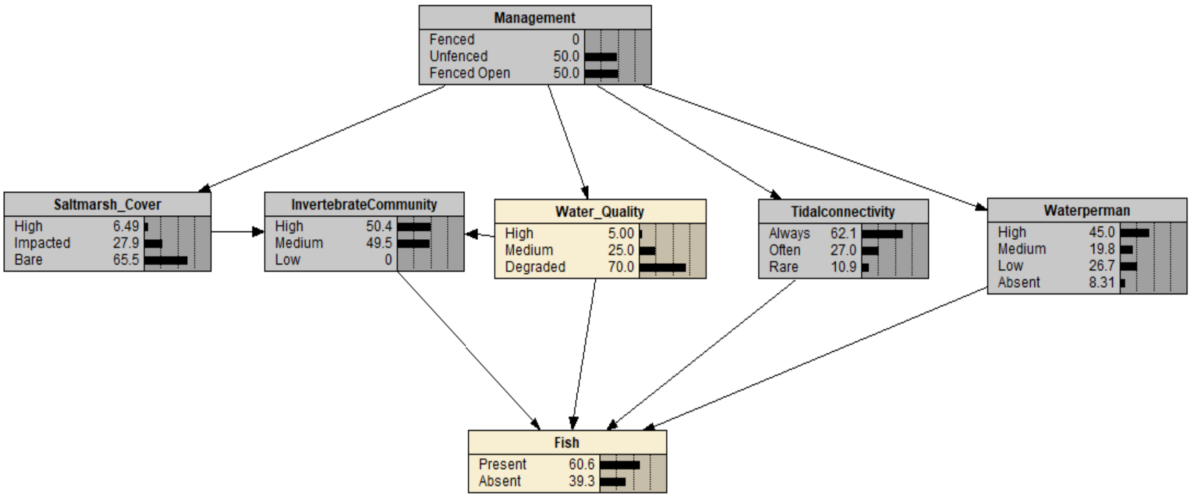

Figure 59. A Habitability Bayesian Belief Network (H-BBN) model examining the likelihood for fish based on qualitative wetland fencing attributes.

In this model scenario, the H-BBN model provides a tool for estimating the likely value of fencing, not fencing or fencing but allowing cattle to access the wetland. Although the model was developed for Round Hill Reserve it is a useful scenario to assess similar management decisions elsewhere in the GBR, because the decision node rules here would apply elsewhere. This model knowledge is a first step towards a superior management decision when considering fencing coastal wetlands. In the model here for fencing, tidal connectivity and water permanency were both critical in determining whether fish values are achieved. This model outcome aligns with many other studies on floodplains where connectivity (particularly frequency and duration) is important for fish movement (Freitas et al. 2010, Abrial et al. 2019). The probability of fish being present in fenced wetlands seems less influenced by saltmarsh cover and invertebrate community, which could be symptomatic of the available data that fish access to saltmarsh areas is a function of connection duration and depth (Hollingsworth and Connolly 2006, Hewitt et al. 2020).

The model also provides the ability to examine the likely probability of fish habitat use on the wetland where the management action is no fencing, or fencing that is open to cattle access for feeding. Here with fish habitat values being the end outcome, regardless of no fencing or fencing with cattle access, both result in an absence of fish when connectivity is “always” and water permanency is highest. This means that for wetlands near to primary estuaries or rivers with frequency exchange the impact of pigs, or fencing and cattle, has only a slight probably of higher than 50% presence of fish. Conversely, less permanency and/or connectivity will result in fish being absent from wetlands presumably because water quality conditions and saltmarsh habitat degradation increases. This is an important finding as it suggests that when pigs are damaging wetlands, installing a fence to prevent pig access is moot because cattle will only cause an impact on the wetland, to-this-end, fencing would not be an effective use of available funding.

The application of this modelling tool can be further scaled up to consider multiple values and services for the three management approaches to feral pig control here (Figure 60). When fish values are considered in the context of other wetland values and services, such as freshwater turtles, water birds, carbon additionality, climate resilience, terrestrial habitat etc.

the framework here in the modelling allows comparison of the likely multiple values and services achieved for the pig management approach. From these analysis, fencing would achieve many more values and services, but also when combined with the addition of turtle gates and earth ramps for terrestrial wildlife. For managers, the BN provides a more superior approach to management decisions, allowing for a mitigation decision to be assessed among a causal web of interacting factors and probabilities of multiple states of attributes and values/services. The framework here can be further refined and the probabilities improved with more data, reducing the need for expert input leading to more confidence in the model likely output results. This framework is also simple to operate and could be easily run by community and government groups now that the model platform has been established.

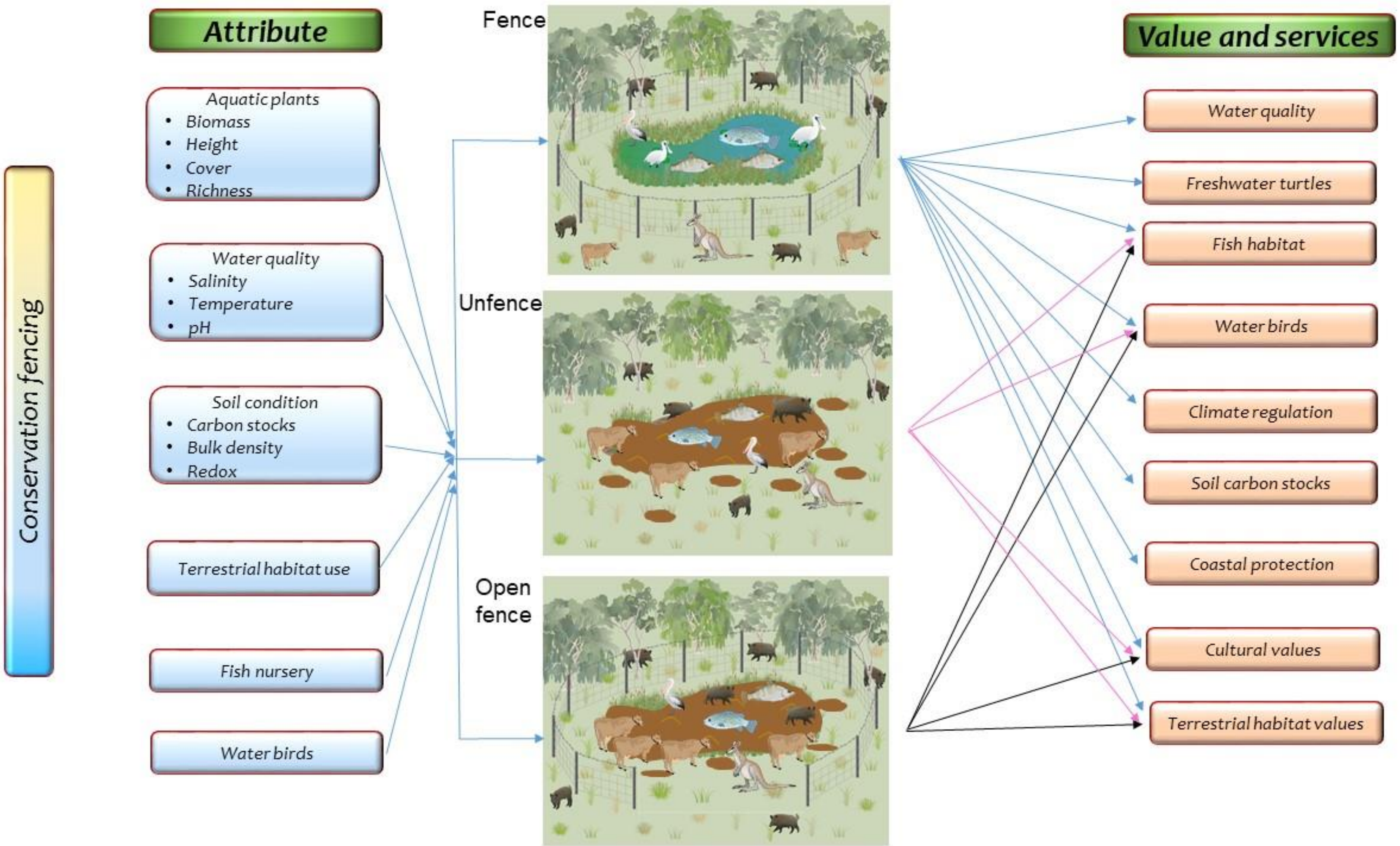

Figure 60. H-BBN for wetland fencing scenarios in Round Hill Reserve. The attribute node data are the mechanisms for estimating the likely values and services for the three fencing scenarios. The arrows illustrate the positive output values/services that are possible for the three scenarios. The unfenced and fencing with open access to cattle would provide similar values, while fencing is likely to offer the most number of values/services, particularly in combination with some further fencing modifications such as turtle gates for passage and earth ramps allowing arthropods to clear fences.

Understanding of the factors leading to nursery value of species that utilise estuaries as nurseries is still evolving (processes, Figure 1). Consequently, it is vital that the understanding established here is validated in other areas, and similar understanding developed for other species, both those with similar life-histories and habitat requirements, and those with different needs. Changing climates and sea levels are likely to lead to increasing shifts in the habitats and conditions available, and changes in the specific needs of various species (components). This will be complicated by concomitant changes in the ranges over which species occur. As a consequence, it is important that similar studies are undertaken with a view to understanding the effect of these cumulative changes on nursery ground values, the ability of habitats to continue provide nursery value, and the consequences for commercial, recreational and indigenous fisheries outcomes.

4.6 Final remark

This project provides a case study example of the kind of data that would be necessary to understand and contextualise the components and processes that exist in coastal wetlands impacted by feral animals. The impact of pigs was obvious in these wetlands, and probably contributed to changes in vegetation, soil properties, and water quality, which themselves probably impact on underlying biological processes such as fish and bird habitat. Erracting a fencing around the wetland here provided an opportunity for some of the desired services return. However, over the course of this study the land tenure with the adjacent landholder contributed to lost services (i.e., water quality deteriorated, vegetation community was disturbed or lost, and wadder bird numbers were lower – we acknowledge though that these observations are based on a few data points). From the data here, it seems important to not manage the treat of pigs, but rather manage for the processes such as improving vegetation communities and fish production. In addition, allow cattle to access the wetland needs further consideration and review of land tenure arrangements as this is clearly a processes that continues to disturb the services expected of these coastal wetlands.

5.0 CONCLUSIONS

Restoration targeting the threat of feral pig (and cattle) damage caused to coastal wetlands is important and will continue to be necessary in the future. Evidence presented here portrays that fencing for wetland vegetation community values has indeed achieved much more so than when compared to wetlands that remain exposed to feral pig damage. There were some initial indication that the vegetation community, fish and bird diversity was probably returning following the fencing. However, in the subsequent years of this study, it was apparent that protection of wetland values was limited by the continual access to the fenced wetland by cattle as part of land tenure arrangements with adjacent land holders. The effort of installing fences, building earth ramps for fauna movement and pig baited trapping needs to be also supported by changes to the land tenure arrangement, to ensure that the values and services aspired to here are reached for the investment in this sensitive wetland habitat.

Maintenance of the fence will continue to be challenging and will require a long term commitment from state government and/or the Burnett Mary River Group to cover these costs. Routine inspection of fences and completing general maintenance as necessary will preserve the investment, and ensure the supporting infrastructure is in working order – for example the earth ramp. More trials like the earth ramp are needed to build a knowledge base towards a design that is most effective for arthropod movement over fences. Trials similar to the freshwater turtle gate design that allows them to pass through when the wire mesh is too small for the shell width.

The value of these wetlands with low frequency of tidal connection seems to offer little opportunity for fish species. The fact that only a small number of species were caught here supports this point, where more fish might be initially able to access wetlands during summer rainfall or very high tides, however, dry over the coming months depending on further tides and rainfall. During these times water quality would deteriorate, whereby fish would be exposed to predation by water birds. While their value as fish habitat might be limited under current climatic conditions, regarding of fencing beginning present or not, further sea level rise might render these upper tidal areas to become more important for fish habitat.

Conservation fencing for wetland values protection is important and should be implemented in broader GBR catchments were the impact of feral pig damage is persistent and extensive. However, a clear understand and determination of the values need to be set with stakeholders, to avoid broader expectations and on-going expensive maintenance programs. Aerial shooting along with baited traps continue to be used in northern Australia (Ross et al. 2017), however, fences if designed and constructed to a high standard could result in a functioning and productive coastal wetland system for a much reduced cost. It is clear here that allowing cattle to enter fenced wetland should cease as this will only continue to impact on the broader value and services of the wetland – it is counterintuitive to continue permitting cattle access to wetlands fenced for feral pig control. Under a market mechanism scheme (e.g. blue carbon; Lovelock and Duarte (2019), or water quality markets; (Waltham et al. 2020b)) these coastal wetlands could become more valuable as part of climate change adaption, and could effectively earn more generated income compared to using these ecosystems for a late dry season cattle feed area (Muenzel and Martino 2018).

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