Efficacy of different water treatment wetland systems to inform system repair projects in Great Barrier Reef catchments

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A Report for Reef Catchments

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EXECUTIVE SUMMARY

The loss and modification of catchment ecosystems is contributing to increased nutrients and sediments load reaching the Great Barrier Reef (GBR) lagoon. Attempts to improve water quality by targeting land use practice change has been shown to not be sufficient to reach water quality targets. There is a need for new, innovative approaches to reduce nitrogen loads that are cost effective, and provide longer term viability (and maybe profitability) for land owners. A possible strategy to achieve water quality improvement is through the use of water treatment engineered structures designed to incept and treat water before reaching receiving waters. This report provides an appraisal of water quality data recorded for treatment wetlands constructed near Mackay, Queensland.

Sites were sampled at irregular intervals between 17-10-17 and 28-03-20. Sampling frequencies and intensities varied between sites and was carried out manually at most sites over the duration of the study but autosamplers were employed at the inlet and outlet of Bakers Creek wetland in 2019 and 2020. This yielded a total of 74 samples for Bakers Creek inlet and 70 samples for the outlet, which is substantially greater than the number of samples collected at other study site (see below).

Samples were analysed for suspended solids, total and dissolved inorganic nutrients, and a range of up to 91 pesticides comprising several, but not all, of the pesticides that have previously been detected in cane farm drainage systems throughout the GBR catchment area.

Water level, temperature and EC data obtained from JCU loggers were available for the Bakers Creek inlet and outlet over the 2019 and 2020 wet seasons, and DO logging data were also available for those two sites during the 2020 wet season. Due to the availability of this additional background data and the more intensive sampling regime adopted at the Bakers Creek wetland, this report focuses heavily on that study site.

Bakers Creek Wetland Treatment System

- The available data provide some qualitative indications that the Baker's Ck wetland may be capable of effecting a reduction in the quantities of pesticides being released into the receiving environment. There are currently insufficient data to be able to confirm that finding or to quantify the magnitude of the effects. However, subject to the proviso outlined in the next dot point, the initial indications are sufficiently favourable to justify further investment to refine monitoring techniques and attend to a few existing technical sampling design issues that have been raised in this report, in order to be able to carry out a quantitative assessment of contaminant import and export rates and loads, which is the only basis upon which the wetland's performance can be accurately gauged.
- Logging records indicate that, except for very brief periods on the peak of flow events, EC levels at the wetland inlet are 30 to 60 % higher than at the outlet. This suggests that a significant proportion of the water emerging from the wetland originated from, an as yet unidentified and unmonitored, source with lower EC concentrations than the inlet drain. Before proceeding with any further monitoring it would be highly advisable to carry out a one-off investigation to confirm that the EC records are correct, and if so, ascertain the source of the additional water and determine the feasibility of including that site in future monitoring programs. If it proves infeasible to monitor the source (for example if it is groundwater inflow or diffuse overland flow), any further attempts at performance monitoring would be largely futile. If the identified source proves to be one or more surface input points along the length of the wetland, the feasibility of being able to account for their effects would need to be evaluated carefully before any commitment to intensive routine monitoring could be justified.
- The suspended sediment (SS) and nutrient data that have been collected to date do not provide an adequate basis for reaching any conclusions regarding the performance of the wetland. Current indications are that, due to the flashy nature of the hydrographs in this system and the highly transient nature of associated contaminant pulses entering the wetland, the sampling frequencies employed for event monitoring would need to be increased substantially in order to obtain more representative samples. It is also evident that high frequency sampling would need to be continued

for the duration of the falling limb of events in order to account for the wetland's detention capacity and ensure that export load is properly quantified.

- It is clear that contaminant concentration data alone, even when supported by water level data (which provide qualitative indications of hydrographic variations), do not provide an adequate basis for quantitatively assessing wetland water treatment efficiency, because the potential significance of each concentration value can differ by more than three orders of magnitude depending on the discharge rate at the time. Quantitative assessments require accurate flow gauging data in order to be able to calculate discharge loads and flux rates. Moreover, in order to be able to ensure that monitoring activities are correctly timed it is necessary to have some knowledge of the wetland's water residence time under different flow conditions because if it is detaining water long enough to perform the desired water treatment function there will never be any direct real-time correspondence between the contaminant concentrations at the inlet and outlet.
- The monitoring methods employed to date have evolved over the course of the project and entailed pilot grab sampling during 2017, manual (low frequency) event sampling in 2018, and automated (medium intensity) event sampling supported by depth and EC logging in 2020. If future performance monitoring is to be attempted, the following additional improvements and refinements would need to be implemented in order to be effective:
    - Installation of accurate flow gauges at all inlets and at the outlet is essential. This could entail construction of flow control devices and depth loggers, and the development of stage-discharge curves so that flow rates can be calculated from water level measurements, or the use of A-V (area-velocity) sensors which yield direct flow measurements (provided that the cross-sectional area of the drain has been accurately determined).
    - Employ the highest sampling rates that are logistically feasible, noting that most samplers can be programmed to take multiple subsamples thus yielding composite samples representative of the period over which each sample bottle is filled.
    - Continue autosampling through the entire event hydrograph, including the tail.
    - Consider installing turbidity sensors in order to obtain a continuous record of particulate contaminant fluctuations. By comparing turbidity data to the trends exhibited by lab SS results it is usually possible to ascertain whether the lab samples are accurately representative (and adjust the autosampling program accordingly if necessary). Turbidity logging records can provide a basis for selecting samples for lab analysis if the number of samples collected exceeds the allocated analytical budget.
    - Include pH and EC in the laboratory analysis suite for many, if not all, samples and routinely conduct field pH measurements whenever conducting manual sampling. (pH values are necessary to confirm the suitability of the water for release to the receiving environment and as an aid for assessing the potential toxicity of parameters such as ammonia. Lab EC values are useful for validating the EC logging data).
    - Determine if there is an alternative analytical method or service provider that can be used to avoid the requirement to dilute pesticide samples in cases where one or two pesticides are present at high concentrations.
    - Periodic evaluations of the health and limnology of the wetland is also recommended for consideration in all wetland projects. This is not required for assessing performance per se, but it is required in order to determine the potential source of performance failures and to gain an understanding of what types of biological communities, conditions and processes yield the best outcomes.
- Conditions in the wetland were severely hypoxic for the first two months of the 2020 wet season. This is not necessarily an unfavourable outcome from a water treatment perspective as it allows denitrification to occur. However, it does mean that the waters contained insufficient oxygen to support most local fish species other than a few low DO specialists such as tarpon and eels, and that the conditions would likely have provided competitive advantage to hypoxia-tolerant noxious exotics such as Gambusia and Tilapia. This linked with high water temperatures, occasional occurrences of potentially toxic ammonia concentrations, frequent detections of 18 different herbicides, some which occurred at levels well above ecosystem protection guidelines, suggests that

this wetland is not desirable fish habitat, at least during the first few months of the wet season. There are therefore grounds to suggest that it might be beneficial to install fish exclusion devices.

- After the third wet season flush in 2020, pesticide levels appeared to have generally declined and DO levels began exhibiting the kinds of daily cycling that are typical of natural wetlands suggesting that habitat conditions towards the end of wet were more conducive to aquatic fauna; however, it is unlikely that favourable conditions would have been sustained (for example in previous years the wetland has experienced algal blooms at various stages during the year.

Other sites Monitored in this Study

- Bakers Creek wetland is the only study site where *in situ* dataloggers were deployed during the course of this project. Accordingly there are no water level or EC records available to contextualise the water sampling results obtained from the Sandy Creek TT, Sediment Basin or cane drain study sites. This makes it impossible to confidently assess the potential significance of the data obtained from these sites. Moreover, water sampling at these sites was conducted manually and was far less intensive than the autosampling campaigns which were carried out at the Bakers Creek wetland. It is therefore considered highly unlikely that the available water quality data accurately represent the hydrographic variations that occurred at these other sites over the course of the project. Strictly speaking that invalidates the use of comparisons between inlet and outlet concentrations as a basis for assessing contaminant removal and the only scientifically defensible conclusion that can be drawn is that the data are inadequate to allow meaningful assessment.
- The above comments notwithstanding, if the water quality data collected at these sites during periods of flow were to be taken at face value the following tentative conclusions could be drawn:
  - The Sandy Creek TT data comprise just nine samples collected from each of 3 sampling points – inlet (SC1), middle (SC5) and outlet (SC7). The results indicate that concentrations of SS and total N, and to a lesser extent oxidised N (NOx), at the outlet were slightly lower than the inlet, while total and reactive P concentrations at the outlet were significantly lower than the inlet. This implies that there may have been some minor removal of particulate N and NOx, and significant removal of phosphorus. However, (even if the above-mentioned comments questioning the representativeness are ignored) these conclusions carry very little statistical weight due to the small number of available data points. Notably, the highest SS concentrations were actually reported at SC5 (within the wetland system), suggesting the possibility that there may have been plugs of turbid water passing through the system that were not detected at the inlet or outlet. There were no potential indications of any pesticide removal.
  - The sediment basin data comprised only seven samples collected from the inlet (SB1) and outlet (SB7). This very limited, and almost certainly unrepresentative dataset provided no indication of any potentially significant contaminant removal.
  - The Cane Drain dataset comprised 23 samples collected from the inlet (CD1) and outlet (CD7). The differences between these two sampling points were only minor and not statistically significant. Nonetheless concentrations of SS, total N, total P, NOx and ammonia were actually slightly higher at the outlet than they were at the inlet; thus supporting the conclusion that there was no evidence of contaminant removal.

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1 INTRODUCTION

The loss and modification of catchment ecosystems is contributing to increased nutrients and sediments load reaching the Great Barrier Reef (GBR) lagoon (Brodie et al. 2010, Brodie and Pearson 2016). Broad scale vegetation clearing and catchment urban, agricultural and industrial development (Waltham and Sheaves 2015) has led to the widespread loss and degradation of freshwater wetlands, forested floodplains, woodlands, rainforests and other terrestrial and aquatic ecosystems in the catchment (GBRMPA 2009, Brodie et al. 2013, Adame et al. 2019b, Waltham et al. 2019b). These ecosystems are essential for a healthy, resilient GBR because they can offer some assistance in trapping catchment sediments and nutrients (Adame et al. 2019a), slow surface water flow, improve coastal hydrological connectivity and provide habitat for a range of freshwater and marine species (Sheaves et al. 2012, Great Barrier Reef Marine Park Authority 2014).

Declining reef health will potentially have a significant impact on the environmental values and economic return generated by the GBR as an asset. Stoeckl *et al.* (2014) conclude that changes in the environment in the GBR would have a major impact on national and regional economies. Those authors found, through a major survey of visitors and residents, that degradation of environmental values would have real impacts in the tourism industry, including reductions in tourist satisfaction, reduced numbers of tourists visiting the region, reductions in the length of visits, and fewer repeat visits. Mustika *et al.* (2016) examined the potential implications of environmental deterioration for business and non-business visitor expenditures in the GBR. The authors concluded that nature-based tourism is an important source of income for the region. 90% of visitors came to the region for at least one nature-related reason, and that substantial environmental degradation could reduce visitor expenditure, and thus local tourism income, by at least 17%.

The impact of agricultural runoff on coral reefs is not limited to the GBR (Roebeling et al. 2011), with around a quarter of the total global reef area (Burke 2011), and a range of other aquatic ecosystems impacted by agricultural pollution (Verhoeven et al. 2006, Flanagan and Richardson 2010). Countries around the world have implemented programs in an attempt to address the impact of agricultural pollution on aquatic ecosystems. These have included the regulation of nitrogen fertiliser use on crops (Kronvang et al. 2008), soil and water conservation (Chu et al. 2009), reduced livestock stocking density (Kronvang et al. 2008), and conversion of agricultural land to alternative production systems or natural ecosystems (Frisvold 2004). The management of diffuse pollutants from agricultural land uses is therefore a key issue throughout the world and management approaches, including the construction of strategically positioned treatment wetlands, implemented in other countries may also be applicable in the GBR catchments (Waltham et al. 2020b).

There is a need for new, innovative approaches to reduce nitrogen loads that are cost effective, and provide longer term viability (and maybe profitability) for land owners (Wallace et al. 2020b, Waltham et al. 2020b). The need for more examples of floodplain scientific investigations to understand the services and values (Zedler 2016), and the threats limiting these potential is only going to become more necessary as governments, industry and community groups respond to the United Nations recent declaration of a decade on ecosystem restoration (Waltham et al. 2020a).

Connectivity of wetlands and drainage channels crossing floodplains provide essential habitat for a range of flora and fauna that have vital cultural, social and economic values. Because of their low-lying positions coastal wetland and rivers receive runoff from urban, agricultural and industrial areas. There is an urgent need for managers to implement strategies and plans to halt coastal wetland ecosystem value loss and degradation, and to commence large-scale programs to repair and restore connectivity, water quality and habitat conditions. While these restoration efforts are vital, access to relevant and appropriate data demonstrating success of project sites, and therefore a positive return on the investment, are lacking (Waltham et al. 2021).

In planning restoration projects, it is important to recognise that stakeholders (beneficiaries) have different and sometimes conflicting views or priorities when determining coastal wetland ecosystem services. For example, placing high value on services such as the freshwater extraction for agriculture from floodplains can directly undermine cultural ecosystem service values related to aquatic biodiversity (Boulton et al., 2016), not to mention reduce duration and frequency of water connection across floodplains which has biological consequences (Baran et al., 2001; Rayner et al., 2009). Ecosystem repair strategies seem to be most effective when values of all stakeholders are incorporated, a process best facilitated through discussions to set objectives early in the project lifecycle (Sheaves et al. 2014; Zedler 2016; Guerrero et al. 2017). Scale is another important aspect, e.g. local-scale improvement of fish habitat *vs.* catchment-scale amelioration of agricultural fertilizer loads exported to coastal waters. Focusing at an appropriate scale is important not only for informing technical aspects of the restoration management activities, but also ensures appropriate management bodies are involved (Butler et al., 2013).

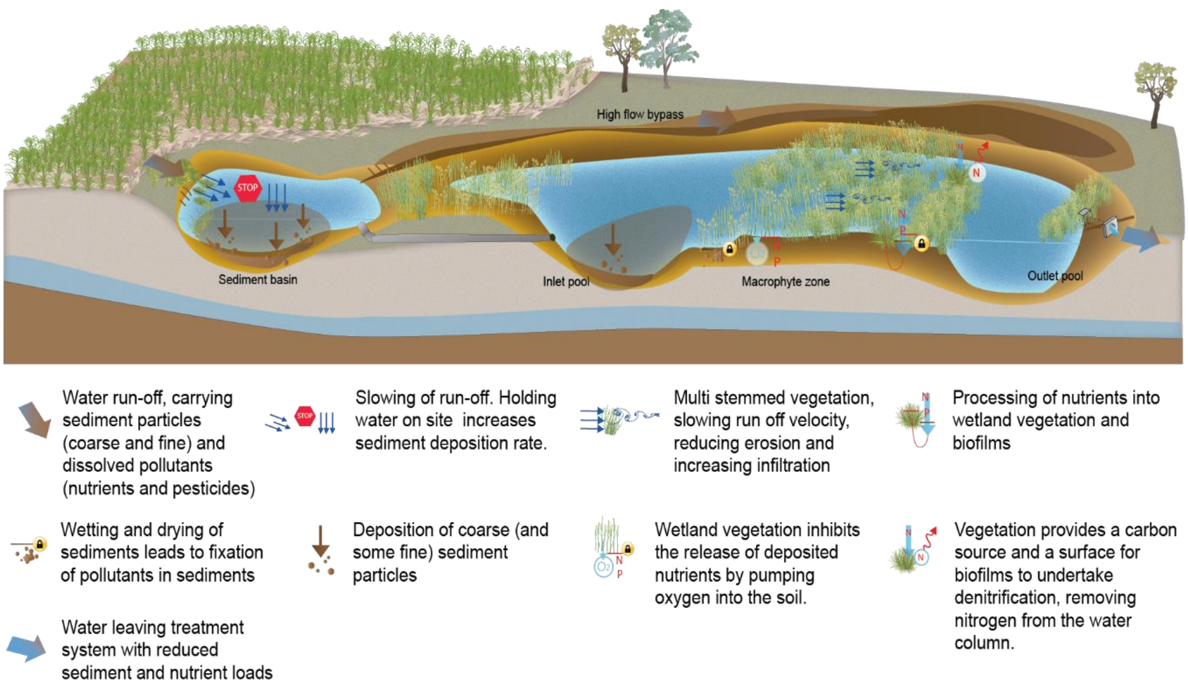


Figure 1 Example of treatment train wetland conceptual diagram (WetlandInfo, Queensland Government, accessed November 2019).

Reef Catchments – water treatment wetland project

Between 2013-2016 Reef Catchments; funded by the Australian Government, constructed two large treatment train systems, within the Sandy and Bakers Creek sub catchments. The focus of the investment was in line with an understanding that within these sub-catchments, little opportunity existed to improve water quality beyond the farm gate. Furthermore, despite a large proportion of farm practices being at accepted industry standard, significant levels of water quality pollution (nutrients, sediment and pesticides) were still being recorded at end of catchment. Unfortunately, the previous investment, did not support the ongoing ability to monitor and evaluate the treatment train system performance or develop science based reports or general communication products to explain the benefits of, and promote treatment trains within a rural landuse context. Additionally, the inability to continue performance assessment of the constructed treatment train is likely a key limitation given the full performance of any treatment train is not realised until aquatic macrophyte zones and riparian vegetation are fully established – which they were since this study commenced in 2017 on treatment train sites. More recently Reef Catchments has been awarded funding by the Queensland Department of Environment and Science (Queensland Reef Water Quality Program) to continue monitoring the efficacy of the works completed and compare these structures to traditional sediment basins and a cane drain.

1.1 Baker Creek and Sandy Creek - Mackay

The Sandy Creek treatment train system, is a three chambered system covering 1.3ha, treating discharge from 50ha of sugarcane production land and bushland area. Establishment costs for the treatment train totalled $100,000 of which construction costs and landholder's cash and in kind totalled $56,500.

The Bakers Creek treatment train system, is a four chamber system covering 2.1ha, effectively treating discharge from approximately 500ha of Bakers Creek sub catchment of which approximately 50% of available land is used for sugarcane production. Establishment costs for the treatment train totalled $155,000 of which construction costs and landholder's cash and in-kind totalled $26,889.

The Sandy Creek sediment basin covers an area of approximately 0.82 ha and drains approximately 45ha. Reef Catchments has also been sampling a cane drain approximately 650m long at the inlet and the outlet to assess any changes in concentration along the drain.

Available land for the construction of large wetlands is almost non-existing on sugarcane properties especially those within Plane and Sandy Creek sub-catchments. Existing waterways treatment structures established on farms are structures built 'in line' within drainage channels rather than 'off line' due to land being used for the production of sugarcane. The most common structure is a traditional sediment basin within a drain which can be pumped for irrigation. Despite the work undertaken by Reef Catchments and others little information on the efficacy of these treatment train structures in the GBR is available and its contribution towards the Reef 2050 Plan with focus on improving habitat, water quality and enhancing connectivity. The basic lack of treatment train performance information within the GBR is a key issue in being able to promote broader uptake of these systems within the GBR context.

<u>Project description</u>

Reef Catchments has completed a three year monitoring campaign to assess the established constructed wetland treatment systems in the Sandy and Bakers Creek sub catchments plus the sediment basin and the cane drain. For the Treatment systems samples were collected at the inlet and outlet for all analyses (sediments, nutrients and pesticides). Additional samples were also collected after the macrophytes zone and analysed for sediments and nutrients. Reef Catchments staff have collected three ambient samples and three event samples per year with staff attempting to collect a sample during events on the rise, peak and fall of the event. Data from this project will be will be used to assess the relative impacts toward improved water quality resulting from farm practices improvements versus treatment systems, as well as, showcasing water quality improvement achieved as a whole, from the various actions at sub-catchment scale. The total improvement gives a direct ability to assess if water quality targets set within Mackay Whitsunday Isaac Water Quality Improvement Plan and the Reef 2050 Plan more broadly.

Water quality monitoring in the final year was restricted to the Bakers Creek Treatment train and the cane drain. Two avalance ISCO auto samples were established at the inlet and the outlet (Figure 2), together with depth and pressure gauge. As the Sandy Creek treatment wetland and the sediment basin have been dropped off the monitoring for the 2019/20 wet season more samples were collected along the hydrograph. This report presents the water samples collected by Reef Catchments from these treatment systems, and provides some interpretation and recommendations to further examine the efficacy of treatment wetlands.

2 METHODOLOGY

2.1 Site locations

The location of the constructed treatment systems is location in Figure 2.1, including the Cane drain, Bakers Creek wetland, Sandy Creek treatment train, and sediment basin. Full location details and sample numbers is presented in Table 3.1.

Figure 2.1 Location map of sites along Sandy Creek Catchment. Site codes in inserts match Table 3.1.

Figure 2.2 Baker Creek wetland outlet has a rock ramp fish ladder, a condition of approval for this constructed wetland. Here fish sampling is underway during wet season flow to estimate fish movement upstream using the ladder (photos source Catchment Solutions)

2.2 Rainfall

Rainfall has been recorded daily at the Plane Creek Sugar Mill station since 1910. Analysis of these data (Figure 2.3) reveals that the highest accumulative wet season (November to March) rainfall occurred 1990/91 (3379 mm) (Table 2.2). Annual summer rainfall total recorded prior to the study were below long term average, in fact, the 2017/18 wet season total was below the 20th percentile of historical records, while 2016/2017 was within the 5th percentile.

Figure 2.3 BOM wet-season (Nov - March) rainfall data recorded at Plane Creek Sugar Mill (station number 33059) ranked in order of decreasing total rainfall (mm). Blue bars show total rainfall over the past few years, red bars cover wet season prior to this survey.

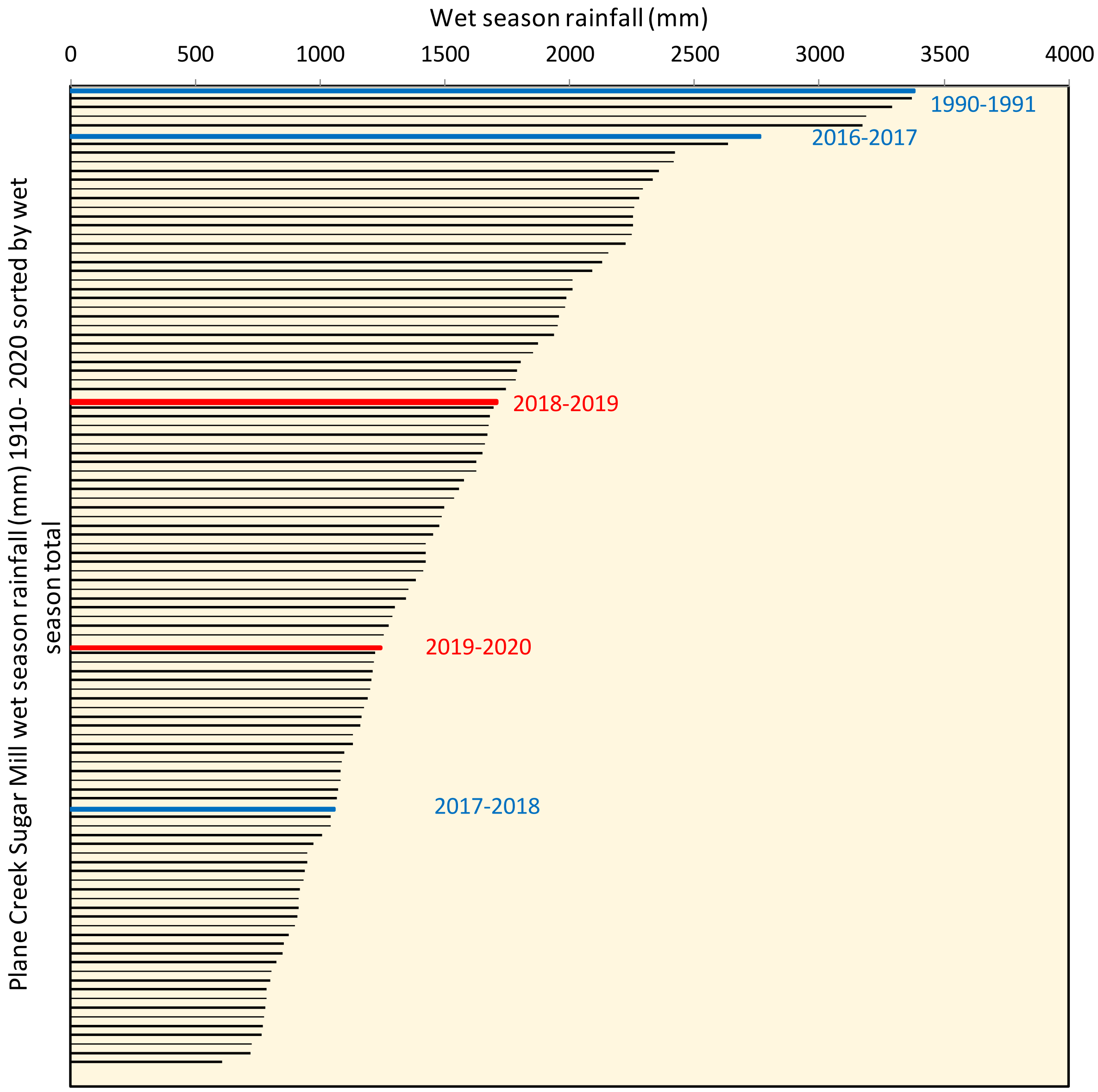


Table 2.2 Summary wet season (Nov – March) statistics of rainfall recorded at Plane Creek Sugar Mill station

Statistic	Wet season rainfall (mm)
Minimum	606 (1930/1931)
Maximum	3379 (1990/1991)
95th percentile	2988.6
50th percentile	1399.0
5th percentile	773.9
2017/18 wet season total	1059.2
2018/19 wet season total	1707.9
2019/20 wet season total	1240.2

2.3 Water sampling

2.3.1 Auto-samplers

Before implementing the project, Reef Catchments identified several locations and appropriate indicators to assist in initiating monitoring. Rainfall gauges closest to the project sites were identified to be used in predicting possible monitoring events including the new rain gauge station at Ooralea (Schmidtke Rd) AL., which is less than 3km from the site, facilitating the monitoring of localised events.

Reef Catchments set a threshold for initiating a monitoring event, the threshold is 30mm of rain in a 24-hour period. The 30mm value was developed through the use of runoff models on the Australian Landscape water balance website (http://www.bom.gov.au/water/landscape).

The grab samples aimed to be collected during rising, the peak and the fall of an event. Monitoring locations have been marked on QLDGlobe to ensure project staff are collecting the samples from the same location regardless of who is collecting the sample. Manual grab sample collection was undertaken by trained personnel, including the use of an extendable sampling pole, enabling collection of the samples from a well-mixed, representative section of the stream /wetland.

All water samples analysed for the project were obtained and preserved in accordance, following methods and standards outlined in the Environmental Protection (Water) Policy Monitoring and Sampling Manual (DERM 2009).

Water quality sampling targeted ambient conditions and rain events, especially the initial rainfall runoff events of the wet season (between November and February). This period is when the majority of pesticides and fertilizers are mobilized from cropping areas and enter local waterways (Rohde and Bush 2009, Agnew et al. 2011). Each set of samples collected consisted of a pesticide sample bottle, a TSS bottle (no preservative) and nutrient sample bottle (citric acid). Samples were retrieved from each location and kept cold (under 4ºC) until delivered to the Mackay ALS environmental office for processing and dispatch

Additionally, the use of the Isco Avalanche Autosampler was trialled with a level actuator trigger at both Inlet and the Outlet at Bakers Creek Treatment train to start sampling as water levels rise above 15cm from ambient levels (Figure 2.4). However, it proved to be unreliable and to avoid missing the first flushes entering and leaving the system a programmed start of the sampler was implemented. Therefore, using weather forecast and the latest river heights and rainfall for the Ooralea (Schmidtke Rd) Alert station an optimal start for the sampler was determined for each sampled event. The Isco Avalanche was programmed to capture a sample every 3 hours or until water levels dropped below the sampler intake.

The interpretation of the rain event prior to sending samples to the laboratory contributed to determine if all samples should be analysed with the aim to capture 3 samples on the rise, one around at the peak and four samples at the fall of each significant rain event, which is kept refrigerated under 4ºC until collection. Due to limited budget, prolonged events that had more than one peak and lasted for more than 3 days were sampled every 6 hours aiming to capture all the changes in the hydrography.

To assess the range of pollutants and the concentrations before entering the wider wetland the samples were analysed for:

- Total Soluble Solids (TSS low level)
- Nutrients (8 different analytes)
- Pesticide (90 different analytes)

Analysis of all samples for pesticides was undertaken by ALS Environmental Sydney and Nutrients and TSS undertaken by ALS Environmental, Brisbane. The laboratory is accredited by the National Association of Testing Authorities (NATA, Australia).

Figure 2.4 Autosampler set up at the outlet of Bakers Creek (BC7). Also the high frequency logger set up using PVC protective pipe

2.3.2 Grab water sampling

Water samples collected by hand (collected by Reef Catchments) were taken well away from the bank and 15 to 30 cm below the water surface, with the mouth of the sampling vessel facing into the current. If flow is absent the sample container was swept gently through the water column to minimise intake of water that has been in contact with the outside of the container and/or the grasping hand. Care was taken to ensure that the bottom sediment was not disturbed and that surface films were not collected. Except where otherwise stated, standard sampling and preservation methods were employed (DERM 2009). Water samples for filterable nutrients were syringe-filtered on site with an unused disposable plastic 60 mL syringe, 0.45µm Sartorius minisart filters. All water samples were kept on ice, in an esky, until processing at the analytical laboratory.

2.3.3 Depth loggers

Water depth, temperature and electrical conductivity were monitored by loggers (CTD-Diver, Eijkelkamp Soil & Water, Netherlands) located in two permanent positions in Bakers Creek (Figure 2.5). The loggers captured data from the bottom of the water column (~ 10 cm above the soil surface) every 20 minutes, and were downloaded only at the end of the two deployment periods. Each logger was installed inside a PVC pipe (1.5m height, 90mm diameter) that was attached to a steel star picket, next to the auto-sampler intake area. Loggers were attached to a stainless steel wire cord that was attached to the top of the PVC pipe for easy retrieval (downloading the data and maintenance). CTD-Diver logger was deployed between January and May 2019, and December 2019 and April 2020, while the HOBO dissolved oxygen logger was deployed only between December 2019 and April 2020 at both the inlet and outlet.

Figure 2.5 Deploying high frequency logger into the inlet to Baker Creek wetland (BC1)

3 RESULTS

3.1 Overview of Data Availability

Sites in Table 3.11 were sampled at irregular intervals between 17-10-17 and 28-03-20. Sampling frequencies and intensities varied between sites (see Figure 3.1.1) and was carried out manually at most sites over the duration of the study but autosamplers (see methods) were employed at BC1 and BC7 in 2019 and 2020. Water level, temperature and EC data obtained from JCU loggers are available for BC1 and BC7 over the 2019 and 2020 wet seasons, and DO logging data are also available for those two sites during the 2020 wet season. Since depth data are only available for two years at two sites, rainfall data has been employed in Figure 3.1.2 to provide some basis for contextualising the timing of sampling events over the entire dataset.

Table 3.1.1 Treatment devices, site codes, total number of samples collected for this report and the latitude and longitude for sites

Treatment Train (TT)	Location	Site Code	Samples	Latitude	Longitude
Bakers Creek Wetland	Inlet	BC1	74	-21.20559	149.13249
	Macrophyte Bed	BC3	7	-21.20770	149.13406
	Middle	BC5	27	-21.20870	149.13415
	Outlet	BC7	70	-21.21138	149.13325
Sandy Creek TT	Inlet	SC1	13	-21.25662	148.94012
	Middle	SC5	12	-21.25503	148.94181
	Outlet	SC7	13	-21.25368	148.94335
Sediment basin	Inlet	SB1	11	-21.29351	149.04671
	Outlet	SB7	9	-21.29284	149.04835
Cane Drain	Inlet	CD1	23	-21.19815	149.13332
	Outlet	CD7	22	-21.20399	149.13243

Accurate assessments of wetland water treatment performance require evaluation of contaminant loads and flux rates and an understanding of key performance drivers such as water residence times, and mixing, and dispersion characteristics to ensure that samples have been collected at the correct time intervals to be able to validly compare inlet and outlet. In order for the wetland to perform efficiently as a water treatment device it must retard the water flow and detain it long enough for contaminants to be dispersed and mixed within the waterbody and allow time for biophysical processing to occur – hence if the wetland is functioning well any brief contaminant pulse that enters the wetland should be diluted and emerge gradually from the outlet over a prolonged period.

The approved methodology for this project did not include any hydrographic monitoring. The water level data obtained from the JCU loggers provide an indication of hydrographic variations but in the absence of current velocity data and/or a reliable flow control structure such as a properly designed V-notch weir it has not been possible to calculate flow rates contaminant fluxes or loads (or water residence times, dispersion and mixing characteristics within the wetland) with the data provided. It would possibly be feasible to calculate a very coarse first-order estimate of the water volume entering the wetland based on catchment area and rainfall data; but since there is no way of reconstructing the dynamics of the discharges over the course of each day, the estimates would be of no value for wetland performance assessment (i.e. there is no means of determining when any particular plug of water that enters the wetland will emerge at the outlet). This issue is discussed further in later sections of this report.

A bathymetric survey of the BC wetland was conducted and the data provided on 21 May 2020. This yielded data which may eventually provide some basis for calculating first-order volume estimates, but the information arrived too close to the reporting deadline to be included in this current report. Moreover, it may still be challenging to obtain reliable flow rate estimates given that the outlet structure at BC7 is a rock wall which functions as a leaky weir (making it challenging to predict stage discharge relationships or confidently determine precisely when discharge ceased), and there is also an unmonitored inlet to the wetland upstream of the outlet the effects of which are undetermined, downstream of BC5, in addition to potential groundwater influences on the wetland which are unknown at this stage.

Figure 3.1.1 Number of samples collected per day over the course of this study (17-10-17 to 28-02-20). Total N=281.

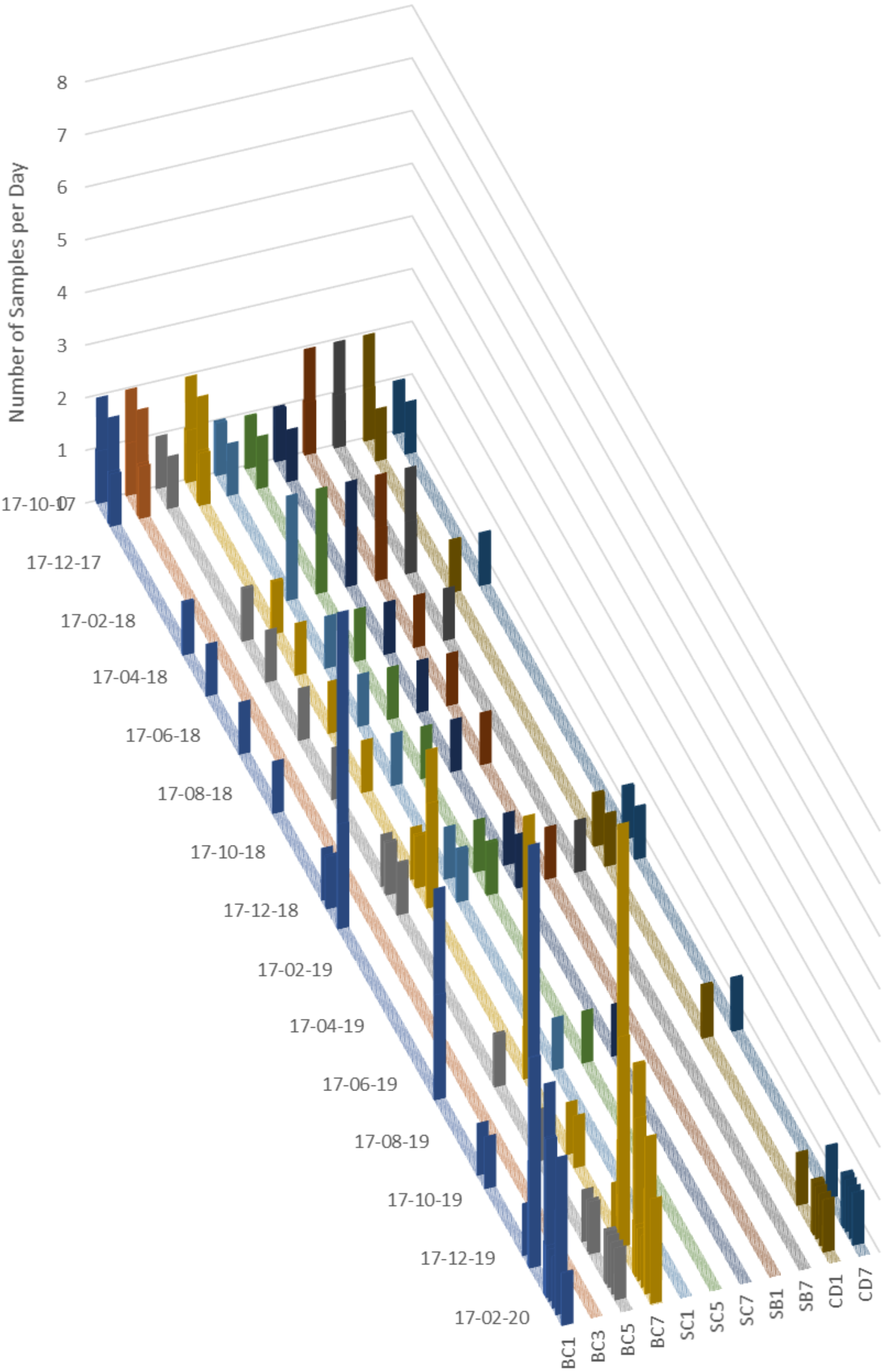
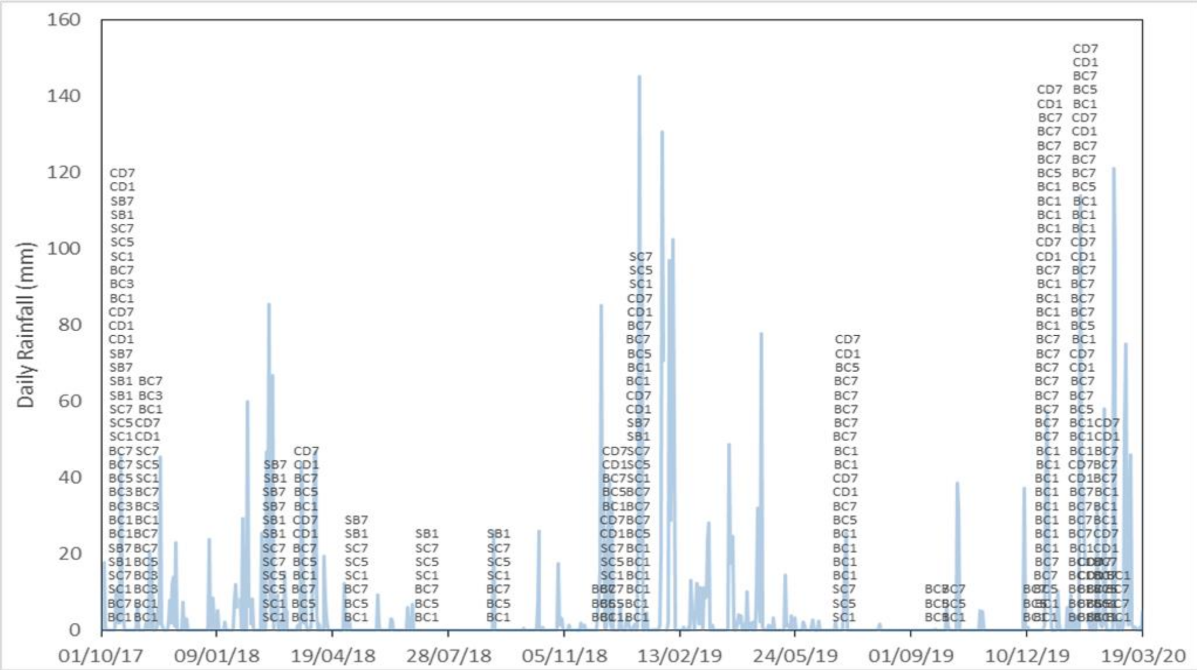


Figure 3.1.2 Timing of sampling events relative to rainfall over the course of this study



Due to these information deficiencies the analyses presented in this report are constrained to dealing with contaminant concentration data and it must be stressed that this does not yet provide a reliable basis for gauging the performance of a wetland treatment system. In fact it is possible that the sampling regime adopted here might not be adequate to obtain a reliable performance assessment even if load calculations were to prove possible. That is because the event-based auto-sampling campaigns that have been implemented to date (see Figure 3.1.3) have not encompassed enough of the falling limb of the hydrograph to account for the wetlands detention capacity. Water residence times and therefore treatment efficiency can be expected to decrease with increasing inflow rates and there will almost certainly be a point at which the wetland ceases to detain water long enough to have any effect on water quality. The hydrographs in Figure 3.1.3 suggest that this point may have been reached during the 2019 wet season and again in March 2020, as there is little indication of any time lag at the outlet. However, there are indications of a noteworthy time lag during January and February 2020 as can be seen clearly in Figure 3.1.4.

During that period the plug of water delivered to the wetland during the peak of a brief inflow event would be expected to become more dilute and disperse as it passes through; however, because outflows are more prolonged than inflows the total load of contaminant released could potentially be the same as the inlet even though concentrations at the outlet are noticeably lower. Conversely if water residence times are very high evapoconcentration can lead to concentration increases even though the export load remains the same (NOx concentrations for example could remain unchanged even if there is significant removal occurring). In order to account for complex interactive effects of this kind it is necessary to continue relatively intensive sampling through to the tail of the hydrograph.

The sampling effort has been concentrated mainly around event peaks, which has an important bearing on the interpretation of the data presented in this report as it increases the likelihood of summary statistics showing that contaminants concentrations at the inlet are generally higher than the outlet. Conversely, the sampling intervals employed to date have ranged from 2.75 to 6.0 hours (for example peak rainfall occurs at night and sampling the next day could miss the initial rise of the hydrograph) and due to the flashy nature of the hydrographs at these study sites and the rapidity of contaminant concentration fluctuations, it is unclear whether this is sufficient to ensure detection of brief but extreme concentration pulses at both the inlet and the outlet. Accordingly, there may be cases where a concentration peak is detected at the outlet but not the inlet. (The probability of such occurrences could be decreased in the future without increasing analysis costs by employing compositing sampling techniques – e.g., by programming the samplers to collect ten 100 mL subsamples at regular intervals over the course of 3 hours to collect a one litre composite, rather than taking a single discrete one litre sample every 3 hours).

Figure 3.1.3 Timing of sampling events at BC1 and BC7 relative to rainfall and water level (where available)

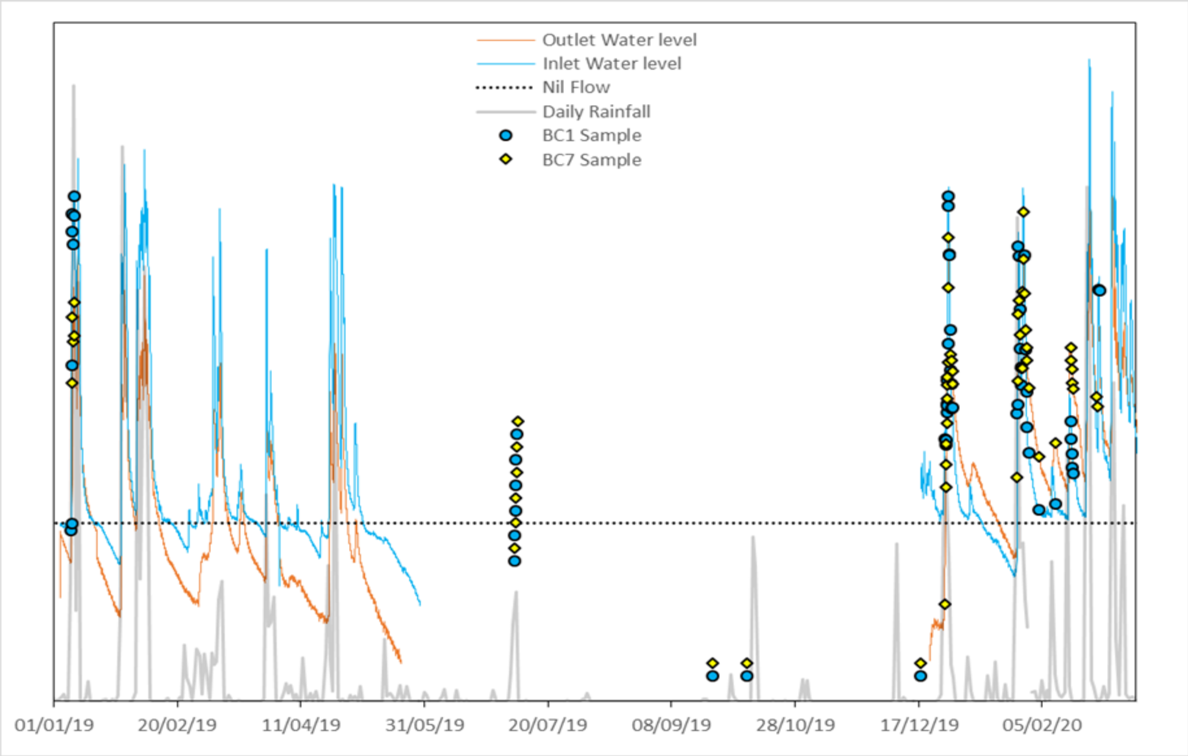
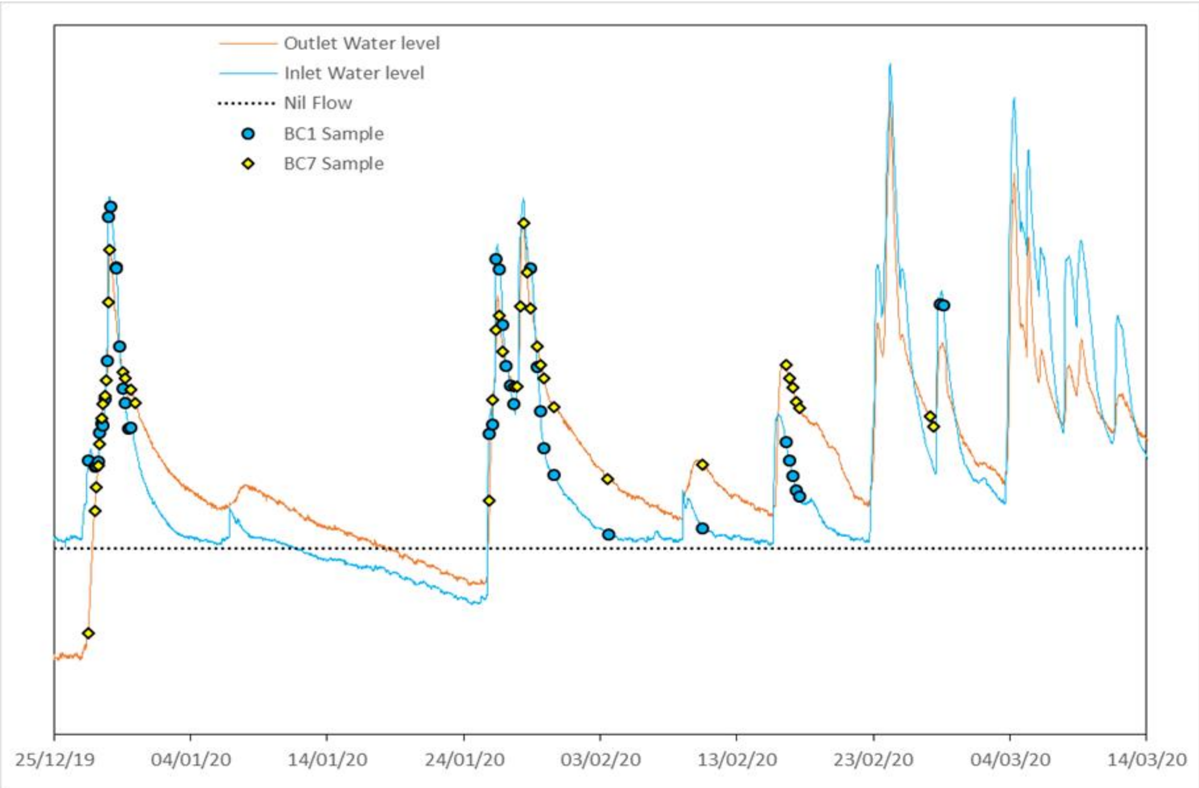


Figure 3.1.4 Timing of sampling events at BC1 and BC7 in 2020 relative to water level



Water samples were analysed for the parameters listed in Table 3.1.2. In some cases, particularly with pesticides, the laboratory found it necessary to dilute samples by 10 to 100-fold in order to resolve matrix interferences and/or contamination due to the presence of elevated concentrations of one or more analytes. This yielded unacceptably high reporting limits (RLs) for all of the other analytes in those samples (e.g., results of <20 µg/L were reported for a number of pesticides that typically occur at concentrations in the order of <0.02 to 5 µg/L). These results were excised from the statistical analysis as there was no way to compute a meaningful substitute value. The potential ramifications of this data censoring are discussed where relevant in subsequent sections. In cases where results were below the normal RL for that parameter, a surrogate value equal half of the RL was employed for the purposes of graphical display and non-metric statistical analyses. Due to the non-normal data distributions and the high frequency of surrogate values, metric statics were not employed.

Table 3.1.2 Laboratory analysis parameters

Parameter	Abbrev	Unit	Reporting Limit	No. of Valid Results
Suspended Solids	SS	mg/L	1	279
Total Kjeldahl Nitrogen as N	TKN	mg/L	0.01	280
Total Nitrogen as N	TN	mg/L	0.01	280
Total Phosphorus as P	TP	mg/L	0.01	279
Nitrate as N	NO3	mg/L	0.01	275
Nitrite as N	NO2	mg/L	0.1	275
Nitrite + Nitrate as N	NOx	mg/L	0.1	275
Ammonia as N	NH3	mg/L	0.01	280
Reactive Phosphorus as P	FRP	mg/L	0.01	279
91 Pesticides (see Tables 3.4.1 and 3.4.2)	-	µg/L	See tables 3.4.1 and 3.4.2	<=233, See tables 3.4.1 and 3.4.2

Between 220 and 233 usable results were obtained for 88 different pesticide residues. In December 2019 the analytical laboratory were able to add three additional analytes (DKN, Imazapic and Isoxaflutole) to the analysis suite and this yielded 64 to 66 results for those three pesticides. The analysis suite comprised numerous pesticides that are commonly detected in cane farming areas, but did not include 2,4-D, MCPA, Fluroxypyr and Imazethapyr, each of which have previously been detected in the Sandy Creek catchment area (Wallace et al. 2017b), or Chlorothalonil, Fipronil, Haloxyfop, Metsulfuron-methyl and Triclopyr, which are commonly used in cane farming (King et al. 2017a, b).

The methodology approved by the Reef Catchment steering committee for this project did not specify monitoring of EC and pH. Electrical conductivity (EC) data (indicative of salt concentrations) are available from the two JCU datalogging stations at BC1 and BC7 but no equivalent data are available for other sites (or for BC1 and BC7 prior to 2019). EC data are a valuable and inexpensive interpretative aid, so in the future it would be advisable to determine EC levels (either in the field or laboratory) whenever a sample is collected.

There are no pH data available for any site. Wetlands commonly experience large fluctuations in pH levels over a variety of temporal and spatial scales. The nature and timing of these variations are an important indicator of wetland health and function, and in some cases may impact on the suitability of the water for release to receiving environments.

At a minimum it would be advisable in the future to conduct laboratory pH analyses on all samples to ensure that there are no values that present a potential risk to receiving waters (for example through acid sulphate generation, which is not an uncommon occurrence in some wetlands). Ideally the samples would be equilibrated with air prior to submission for analysis in order to be able to infer what the pH will likely be if released into a well-aerated receiving environment. This is because carbon dioxide (CO_2) dissolves to form carbonic acid, and accordingly pH levels are potentially subject to substantial but often brief fluctuations due to transient natural variations in carbon dioxide saturation levels – which can be caused for example by the rapid temperature and pressure changes that occur when rain water reaches the ground, photosynthetic CO_2 consumption by plants and/or by respiratory production of CO_2 by most aquatic organisms.

Spot field measurements of pH, when compared to air-equilibrated lab measurements, can provide some indications of CO_2 saturation levels at the time of sampling and therefore some insights into the potential sources and consequences of pH variations. However, continuous datalogging is recommended as the only reliable means of effectively monitoring and interpreting pH fluctuations.

3.2 *In Situ* Logging Data

The interpretation of the Bakers Ck wetland water sampling results presented in later sections of this report are underpinned by the JCU logging data which provide some basic understanding of the ways in which flow rates and ionic composition vary between inlet and outlet.

3.2.1 Hydrographic variations in water level and electrical conductivity (EC)

The water level and EC results obtained from the dataloggers at the inlet (BC1) and outlet (BC7) of the Bakers Creek wetland during the 2019 and 2020 wet seasons are plotted in Figure 3.2.1a. The 2020 data are more closely examined in Figure 3.2.1b. Note that the morphology and therefore depth-flow relationship differ between monitoring locations, so the water level data have been normalised in order to more easily compare the shapes of the two hydrographs.

Figure 3.2.1a Water level and EC records for the 2019 and 2020 wet season

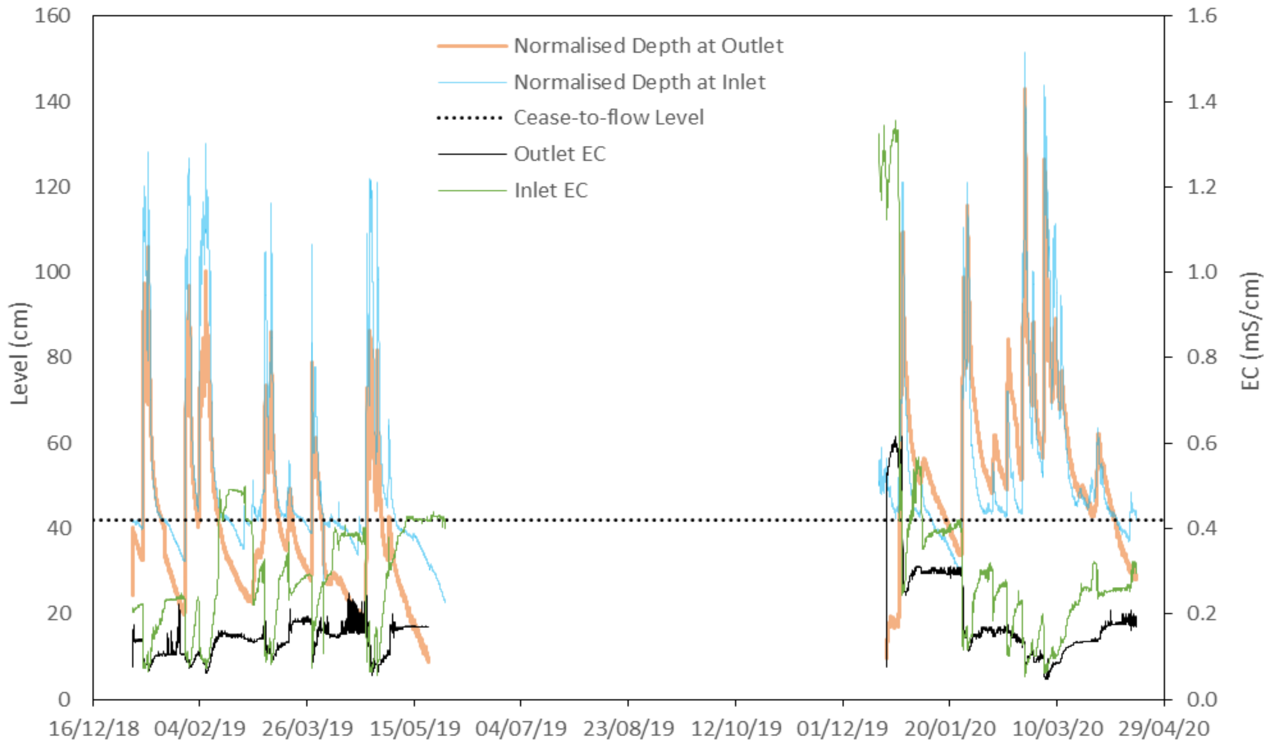
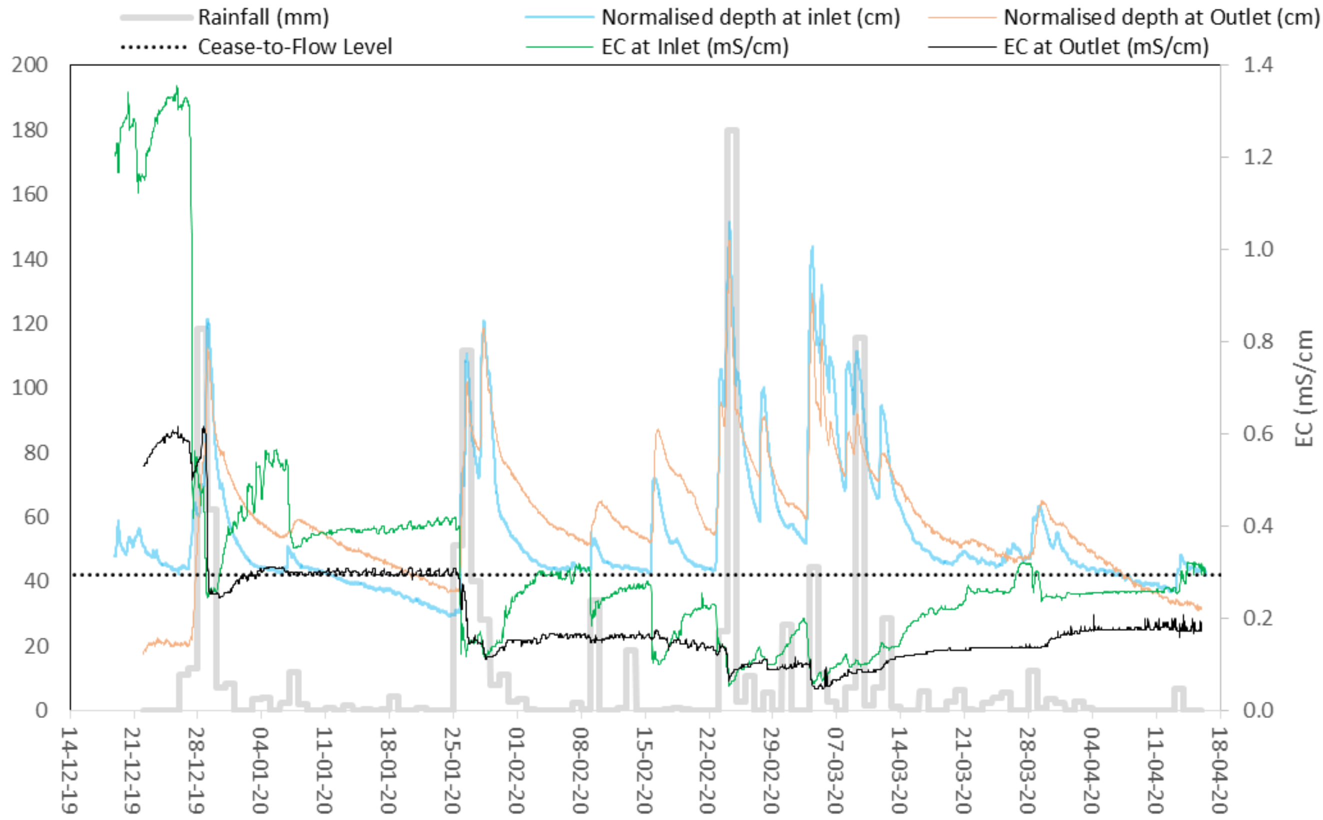


Figure 3.2.1b Water level and EC variations over the 2020 wet season



It is evident from the plots that EC values at the inlet (BC1) were consistently higher than the outlet (BC7) except for brief periods during the peak of flow events at which time the values at both sites fell to similar (lower) levels. Under stagnant conditions prior to the commencement of wet season flows the EC levels at both sites were noticeably higher than normal, especially at BC1 where values ranged up to 1.35 mS/cm which was more than twice the levels recorded at BC7 and high enough to potentially be stressful to some stenohaline wetland organisms. The moderately elevated levels at BC7 were possibly due to the effects of evapoconcentration (which is in turn a natural consequence of prolonged water residence time, stagnation, high water temperatures and shallow water depth). These factors may also have had a bearing on the results for BC1; however, the much higher levels at that site suggest the potential involvement of groundwater input to the drain.

EC levels at both sites fell rapidly down to 0.245 mS/cm on the rising limb of the first-flush event on 29/12/19 and fell to progressively lower levels on the rising limb of the subsequent large scale events on 28/1/20, 24/2/20 and 4/3/20, reaching a minimum of 0.05 mS/cm (which is very low by most standards). Notably the EC levels at BC1 increased on the falling limb of each event, reaching a maximum of 0.55 mS/cm after the first flush and levels in the order of 0.3 mS/cm on the tail of subsequent events. In contrast EC values at BC7 stabilised at 0.41 mS/cm on the tail of the first-flush, fell to 0.12 mS/cm on the next event and remained below 0.2 mS/cm for the remainder of the wet.

The rises in EC levels at BC1 occur each time rainfall (and hence overland surface runoff) ceases, and is almost certainly due to inputs of moderately saline soil water and/or groundwater which sustain flow in the drainage system for extended periods between rain events. Similar effects are often observed in natural streams throughout this region (the rate and extent of EC increases being dependent on the hydrochemistry of the groundwaters proximal to each stream). This effect was not evident in the outlet end of the wetland. There are insufficient data to confidently determine why that is the case; however, it appears that the wetland may be receiving additional inputs of low salinity water (from the unmonitored drain upstream of the outlet or groundwater inflows) and/or that the amounts of water delivered to the wetland on the tail of the hydrograph are generally too low compared to the standing water volume to significantly affect the quality of the water contained in the opposite end of the wetland.

The fact that inflowing EC levels are between 30% and 60% higher than the EC values at the outlet most of the time (effectively whenever it is not still raining) is a significant finding and indicates that the water in the wetland usually has a fundamentally different composition to that in the inlet drain. Ostensibly it seems that a significant proportion of the water discharged from the wetland in the aftermath of flow event peaks either originates from an as yet unidentified and unmonitored source or it comprises mainly waters delivered by the inlet drain during the flow peak that has been detained for several days and which is still discharging from the wetland on the tail of the hydrograph. Regardless of which is the case these findings cast further doubt on the validity of attempting to assess the wetland's contaminant removal capacity by comparing concentrations at the inlet and outlet, especially given that the available data include very few samples representative of the prolonged stages of the hydrograph during which the wetland discharges are occurring. Since the quantity and quality of the unidentified input(s) is unknown, it is not possible to infer whether these effects would result in negative or positive biases in the results, and it is conceivable that the effects would vary between parameters and over time.

3.2.1 Variations in dissolved oxygen - an indicator of denitrification potential

It is apparent from the datalogging results plotted in Figure 3.2.2a that dissolved oxygen (DO) saturation levels at both BC1 and BC7 are subject to substantial fluctuations over a variety of time scales. Note that similar levels of spatial variation can also occur throughout the water column. These DO sensors were deployed 10 cm off the bottom and based on experience with other wetlands in the region it is considered likely that temporal variations nearer to surface would have been equally substantial but that saturation levels would have been somewhat higher on average. Saturation levels at both sites generally declined during the peak of the hydrograph. This was likely due to the combined effects of inputs of organic matter (and respiratory uptake of DO through microbial decomposition), decreased light penetration (due to increased water depth and turbidity) thus preventing photosynthetic DO production by plants and also phytoplankton (most of which are washed downstream during events). That is to say that in the absence of biological DO production the rate of DO uptake by biota exceeds the rate of oxygen uptake from the overlying air. There were brief periods on the tail of the first-flush event when biogenic DO production resumed (firstly at BC1 and then at BC7) but hypoxic conditions returned during the second flow event and instream DO production did not re-establish until mid to late March 2020. The fact that DO levels are primarily driven by biological processes is most clearly evident in Figure 3.2.2b which shows that DO levels increased during the day (due to photosynthetic production by plants and phytoplankton) and declined overnight (due to respiratory uptake by plants and other organism in the absence of photosynthetic production).

Figure 3.2.2a Water level and DO (dissolved oxygen) variations over the 2020 wet season

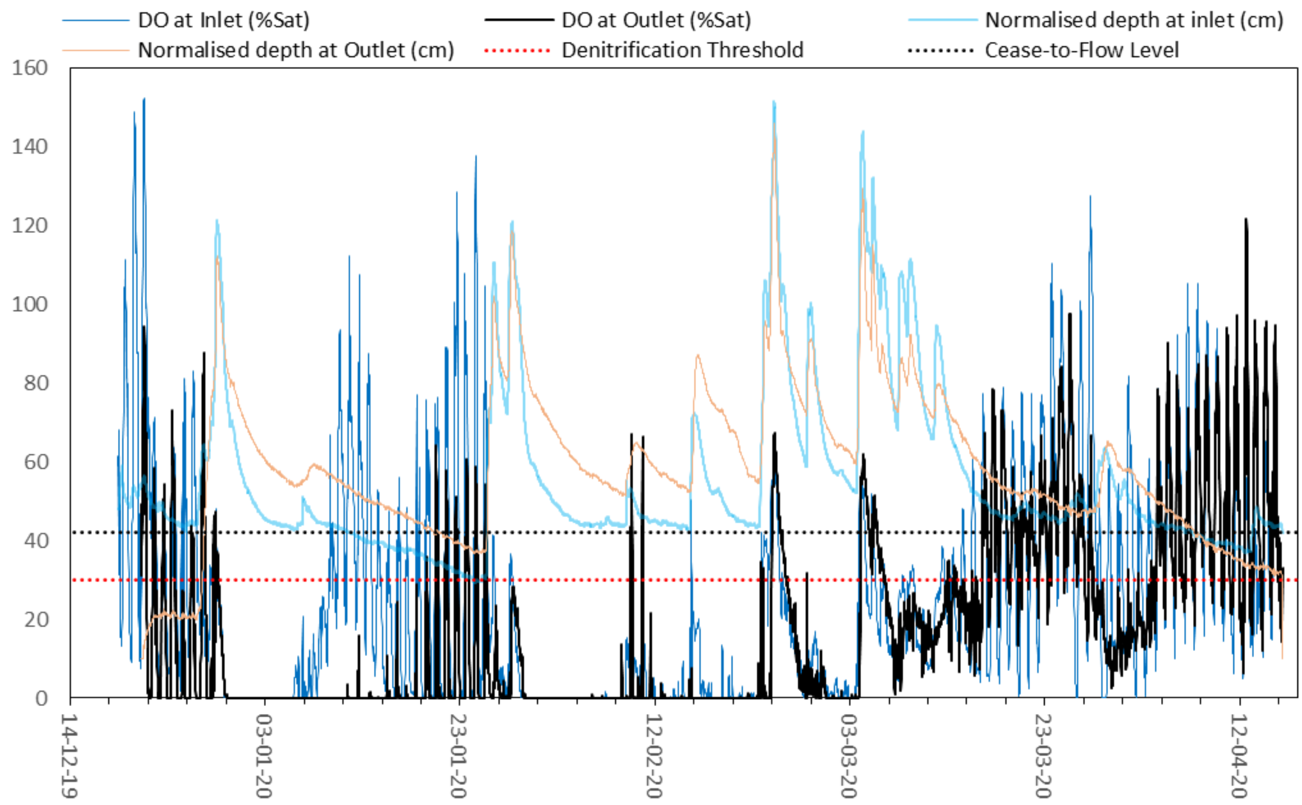
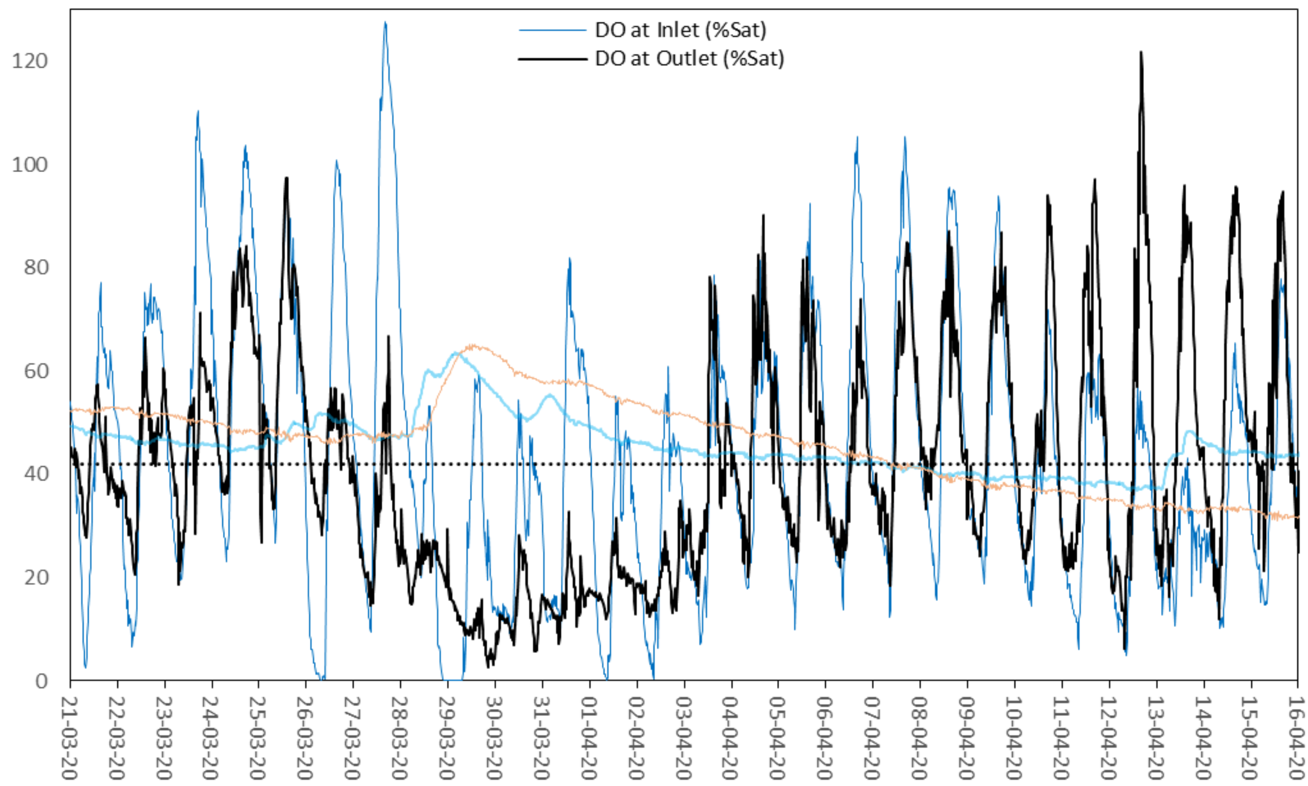


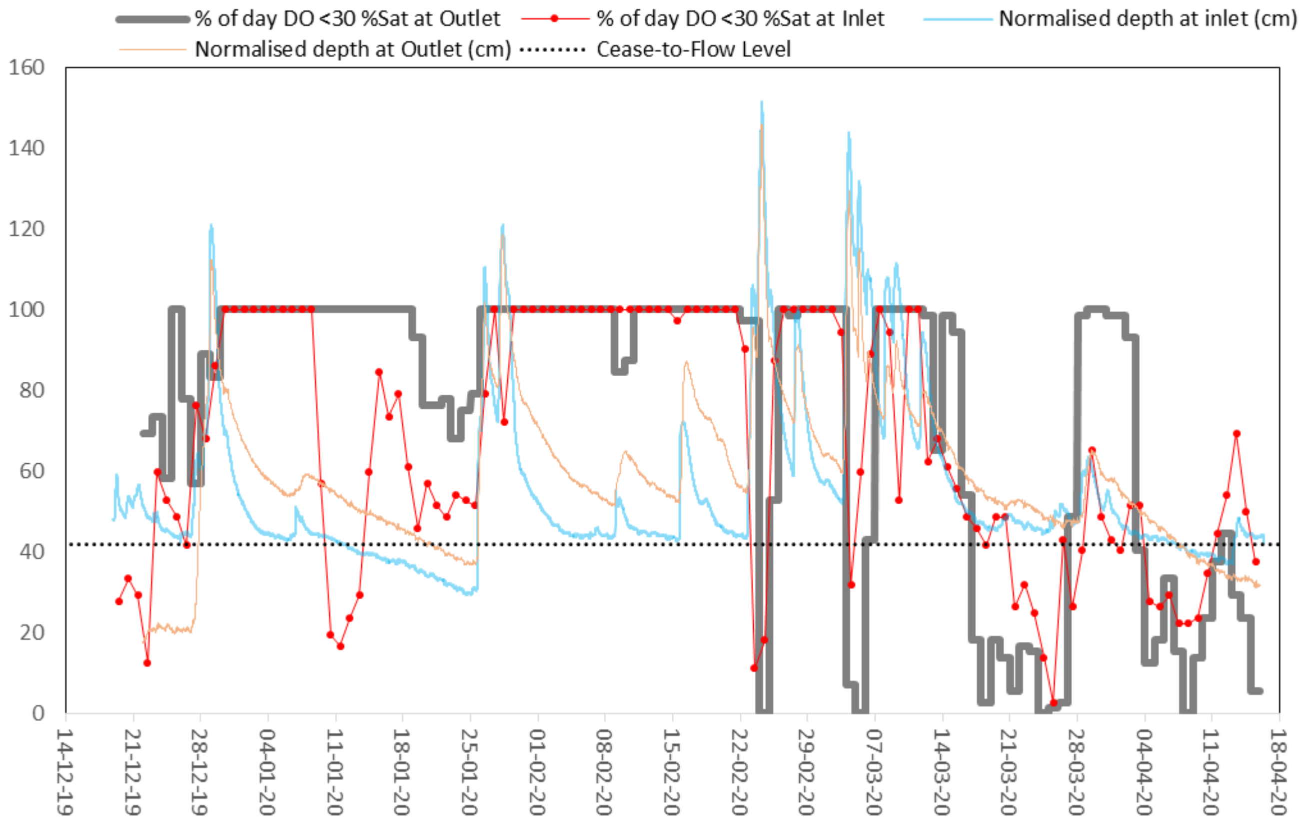
Figure 3.2.2b Daily DO variations over the 2020 wet season



Low DO concentrations are a natural and often inevitable characteristic of many natural wetlands in north Queensland (Perna et al. 2012, Waltham and Fixler 2017) but are nonetheless generally considered undesirable by most environmental practitioners. However, in cases such as this where microbial denitrification may be a desired wetland function / ecosystem service, hypoxic conditions are necessary

(Adame et al. 2019a). The percentage of time each day that DO levels were below the 30 % saturation threshold for denitrification at each site is plotted in Figure 3.2.2. It can be seen that DO concentrations were 100% conducive to denitrification at BC1 (and indeed for much of the time at BC7) for the majority of January and February and the first half of March. These are favourable conditions for treatment of water containing high concentrations of nitrogen (N) in the form of nitrate (provided that water residence times are sufficient and there is sufficient organic matter present to sustain the microbes).

Figure 3.2.3 Temporal variations in the percentage of time each day that DO levels were below 30 % Saturation (below which microbial denitrification occurs and many native fish species begin to experience stress)



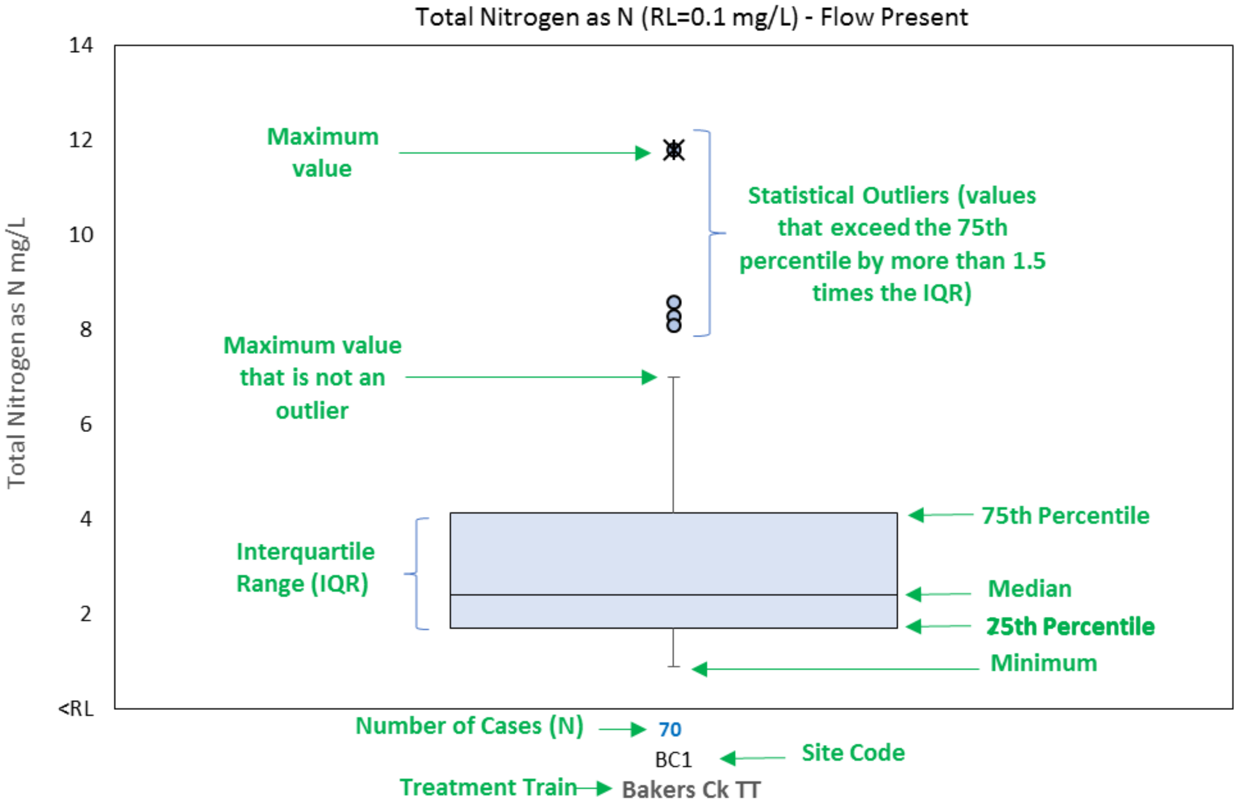
However, it is not necessarily the most desirable scenario for treating water containing high levels of ammonia, urea or organic nitrogen. Under persistently hypoxic conditions microbes convert organic N into ammonia which, unless taken up by plants, will persist until DO levels increase sufficiently for it to be oxidised to form nitrate. In such cases it is necessary to expose the water to normoxic conditions long enough for conversion to nitrate (i.e., nitrification) to occur in order for denitrification (under subsequent hypoxic conditions to be effective).

In treatment systems this can be accomplished either by dividing the system into sequential sections each of which maintain persistently oxic or hypoxic/anoxic conditions as required, or by sustaining controlled biogenic daily DO cycling similar to that evident in Figure 3.2.2b in order to achieve sufficient DO production to convert ammonia and some organic forms of N to nitrate during the day and maintain sufficiently low DO levels to support denitrification and ammonification (decomposition of organic N to form ammonia) during the night. The Bakers Ck and Sandy Ck wetlands are multi-pond systems and could theoretically/potentially be functioning in either of these ways; however, detailed biophysical limnological surveys and DO profiling of individual ponds under a range of flow conditions (neither of which have yet been conducted) would be required in order to determine if that potential is being realised and if not, what alterations might be needed to accomplish that performance objective.

3.3 Suspended solids and Nutrients

Throughout the remainder of this report summary statistics for each parameter and site are displayed using a modified version of the traditional Tukey box and whisker plots as shown in Figure 3.3.1. Note that there are instances in this dataset where the number of samples at some sites is quite low and also cases where a large proportion of samples yielded the same value (generally <=RL). In these cases the boxes must be interpreted carefully because for example the minimum, 25th percentile, median and 75th percentile may all be the same (thus yielding no box) and/or the interquartile range may be so low that all detections appear as outliers. Note that when N values are low the calculated percentiles may be very low confidence estimates. For example binomial probability calculations indicate that a minimum number of samples of six is required in order to be 95% confident that the true median value lies between the minimum and maximum values.

Figure 3.3.1 Annotated example of the Box and Whisker Plots employed throughout this report. In cases where numerous results are close to or below the reporting limit (RL) the RL is shown as a dotted reference line (•••••••).

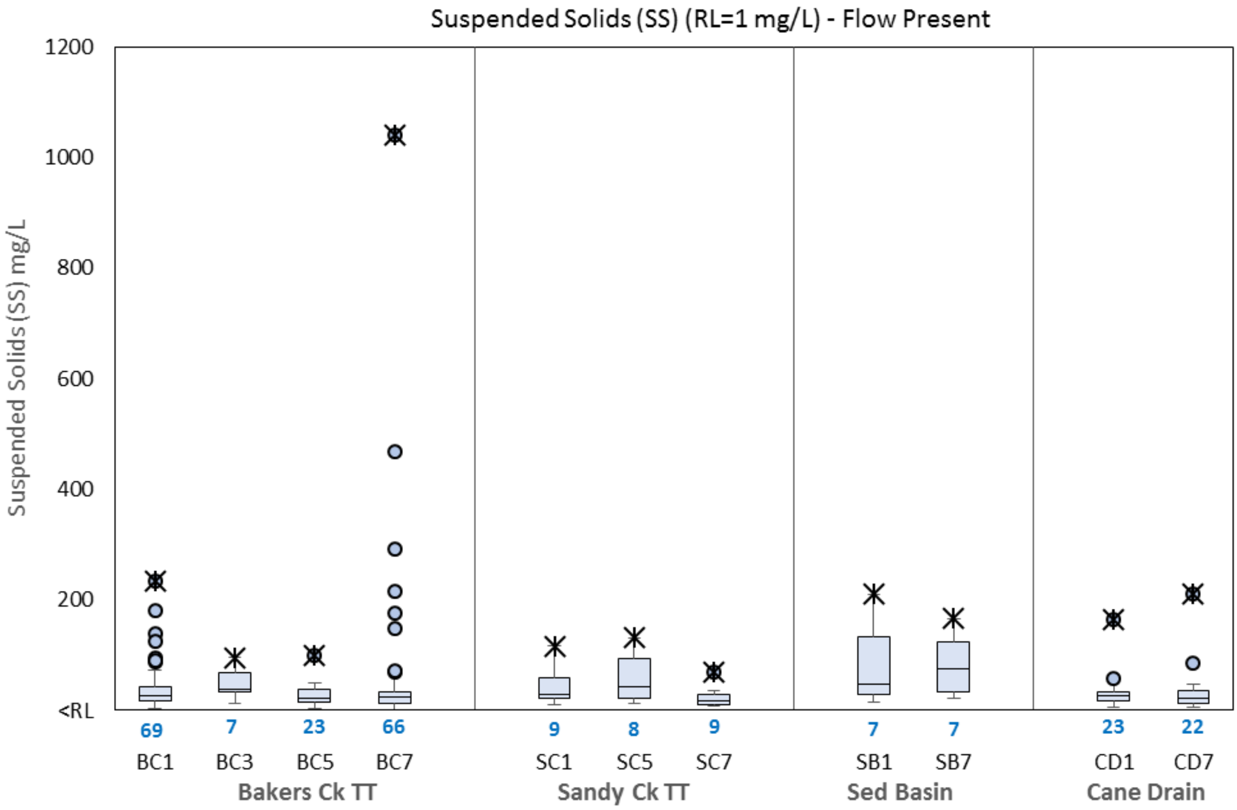


3.3.1 *Flow Present*

Samples indicative of ambient water quality during periods of no-flow are useful for assessing the condition and health of the wetland and its capacity to support various ecological functions but they do not provide a basis for evaluating its stormwater treatment efficiency. Accordingly, ambient water quality data have been analysed separately in Section 3.3.3.

The data presented in this section have been obtained by pooling all results from samples collected at each site any time that flow was present between 19/10/17 and 28/2/20. The data for sites BC1 (inlet) and BC7 (outlet) comprise a mixture of autosampling and manual grab sampling results. The autosampling data are very useful for examining temporal/hydrographic variations and in that context have been examined separately later in this report. Interpretation of the pooled data is underpinned by the fact that the autosampling runs did not encompass the whole hydrograph and consequently it is very likely that a significant proportion of the contaminants introduced to the wetland during those events was not sampled at the outlet (i.e. the outlet data are not representative and cannot validly be compared to the inlet). Accordingly the validity of displaying these data as boxplots is somewhat questionable and invites potential misinterpretation. However, we have resorted to that approach in the absence of any better method of summarising the data. The manual sampling data cover a wider variety of hydrographic conditions and there inclusion in the pooled dataset therefore partly ameliorate the potential bias introduced by the autosampling results. For example the extreme outliers shown in suspended solids (SS) plot in Figures 3.3.2 indicate that there have been some occasions where the SS concentrations discharging from the wetland have been substantially higher than any of the concentrations ever recorded at the other monitoring points in this program. That may possibly be a valid finding but it may also simply be a consequence of the adopted sampling regime. That possibility is discussed in the next section.

Figure 3.3.2 Spatio-temporal variations in suspended solids concentrations during periods of flow over the course of the study



Similar extreme outliers are evident in the total N (TN) and total P (TP) results (see Figures 3.3.3 to 3.3.5). However, even if these outliers are ignored, the data provide no indication of any potentially significant removal of particulate contaminants (SS, TN and TP) or reactive phosphorus, which again, may simply be a consequence of the adopted sampling regime (Figure 3.3.5).

Figure 3.3.6 suggests that the Nitrite + Nitrate (NO_x) concentrations at BC1 were slightly lower than BC7, however, the differences are very subtle. Moreover, as discussed in Section 3.2, the fact that the EC levels at BC7 were 30 to 60% lower than BC1 at the times when many of these samples were taken suggests that a significant proportion of the water emerging from BC7 did not originate from BC1. This suggests that direct comparisons between BC1 and BC7 may not provide a valid basis for assessing the performance of the Bakers Creek wetland.

Figure 3.3.7 provides little evidence that the wetland has had any significant effect on ammoniacal N concentrations although the median value at BC7 was slightly higher than BC1, and there were two outliers at BC7 that exceeded the 0.9 mg N/L default ANZECC 2000 Australian Water Quality Guideline trigger value

TV for protection of aquatic ecosystems from toxic effects (cf one exceedance at BC1). The toxicity of ammonia (and the TV) increases substantially with pH, the default TV being applicable to waters with pH<=8. The potential significance of the reported ammonia values cannot be interpreted in the absence of pH data, but it is pertinent to note that the TV for pH 9 is 0.18 mg N/L for freshwaters and 0.14 mg N/L for marine receiving waters, which can be compared to a median of 0.12 mg N/L and a 75th percentile of 0.17 mg N/L at BC7. In contrast the TV for pH 6 is 2.57 mg N/L which is substantially higher than any of the results reported here. This highlights the value of monitoring pH values in further sampling campaigns in treatment wetlands.

Figure 3.3.3a Spatio-temporal variations in Total Nitrogen concentrations during periods of flow over the course of the study

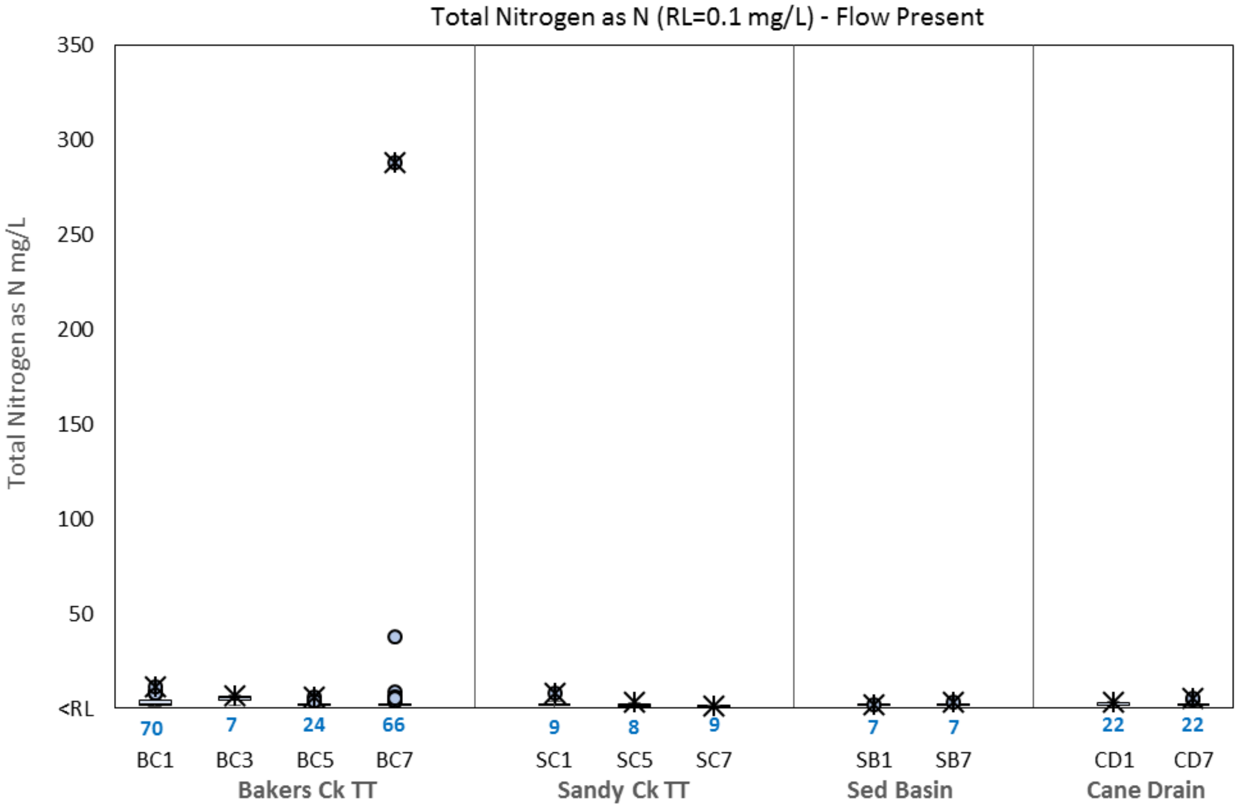


Figure 3.3.3b Spatio-temporal variations in Total Nitrogen concentrations during periods of flow. Extreme outliers of 288 mg/L and 38.1 mg/L at BC7 not shown

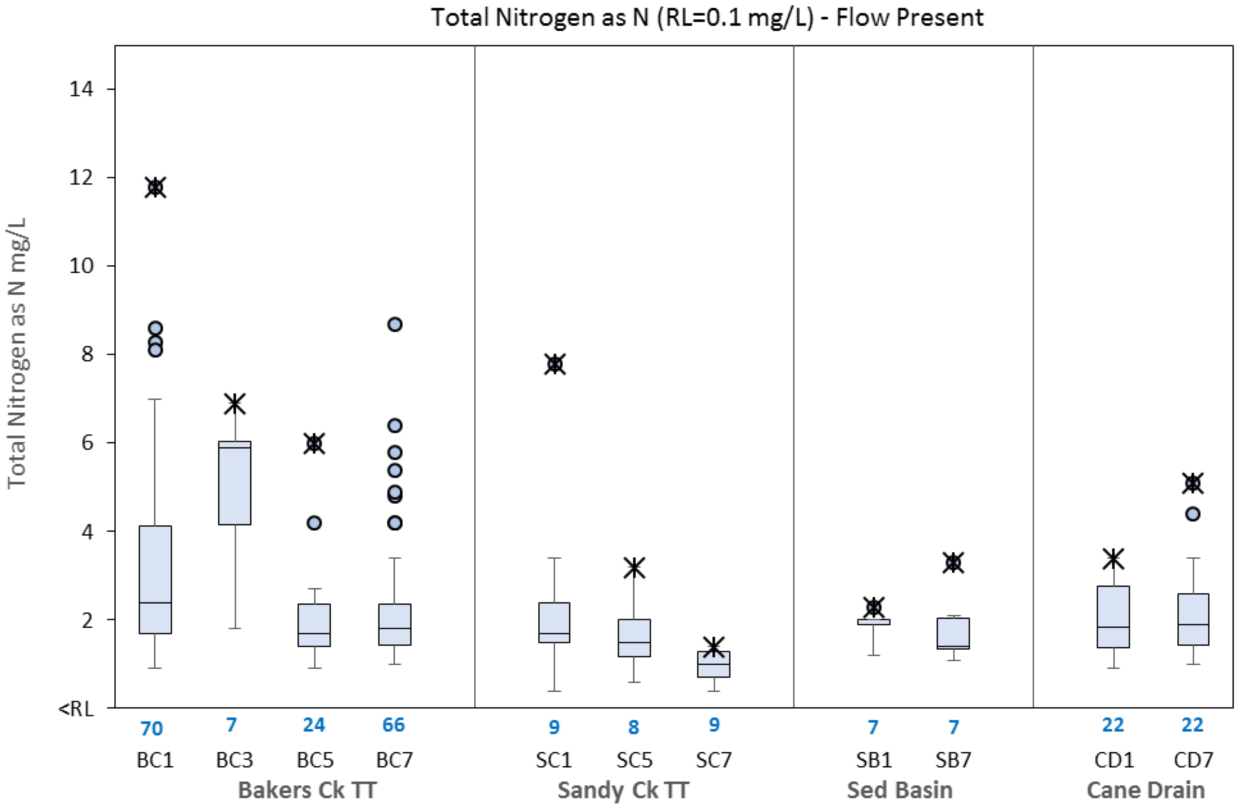


Figure 3.3.4a Spatio-temporal variations in total phosphorus concentrations during periods of flow

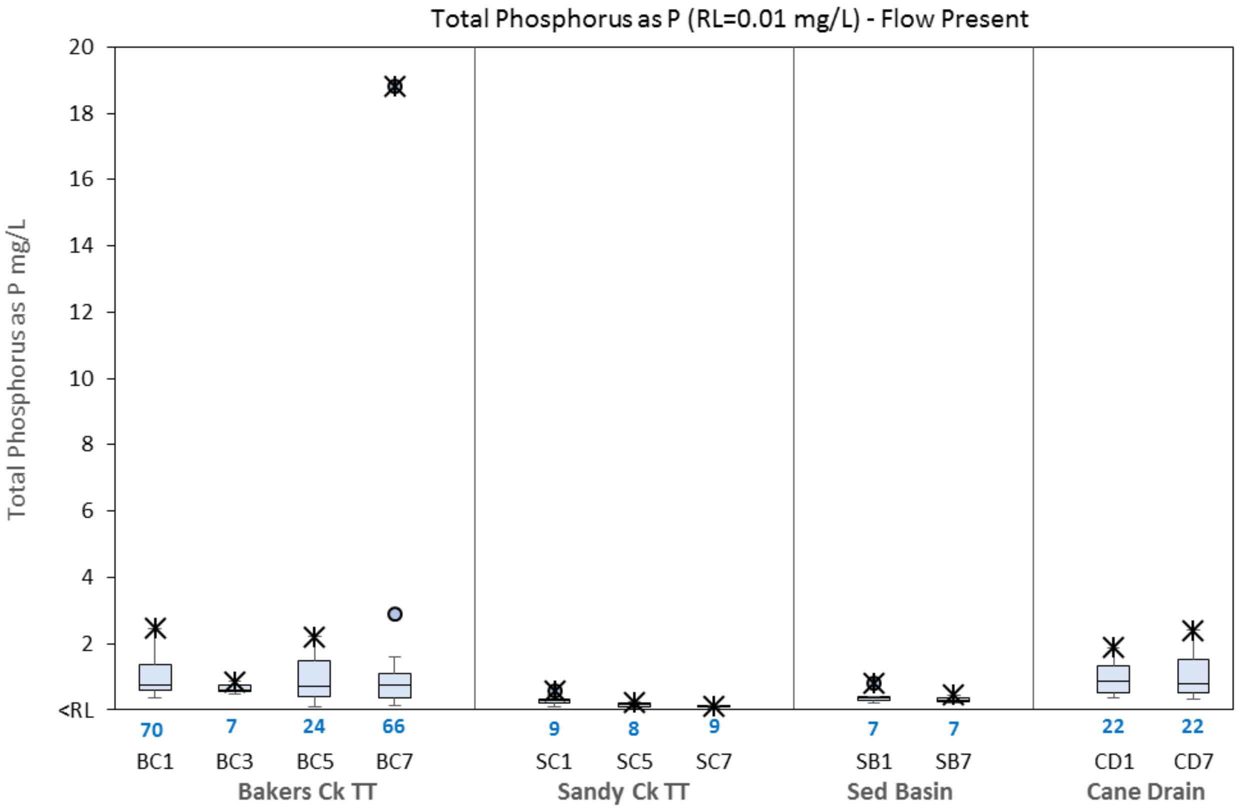


Figure 3.3.4b Rescaled version of Figure 3.3.4 with extreme outlier at BC7 not shown

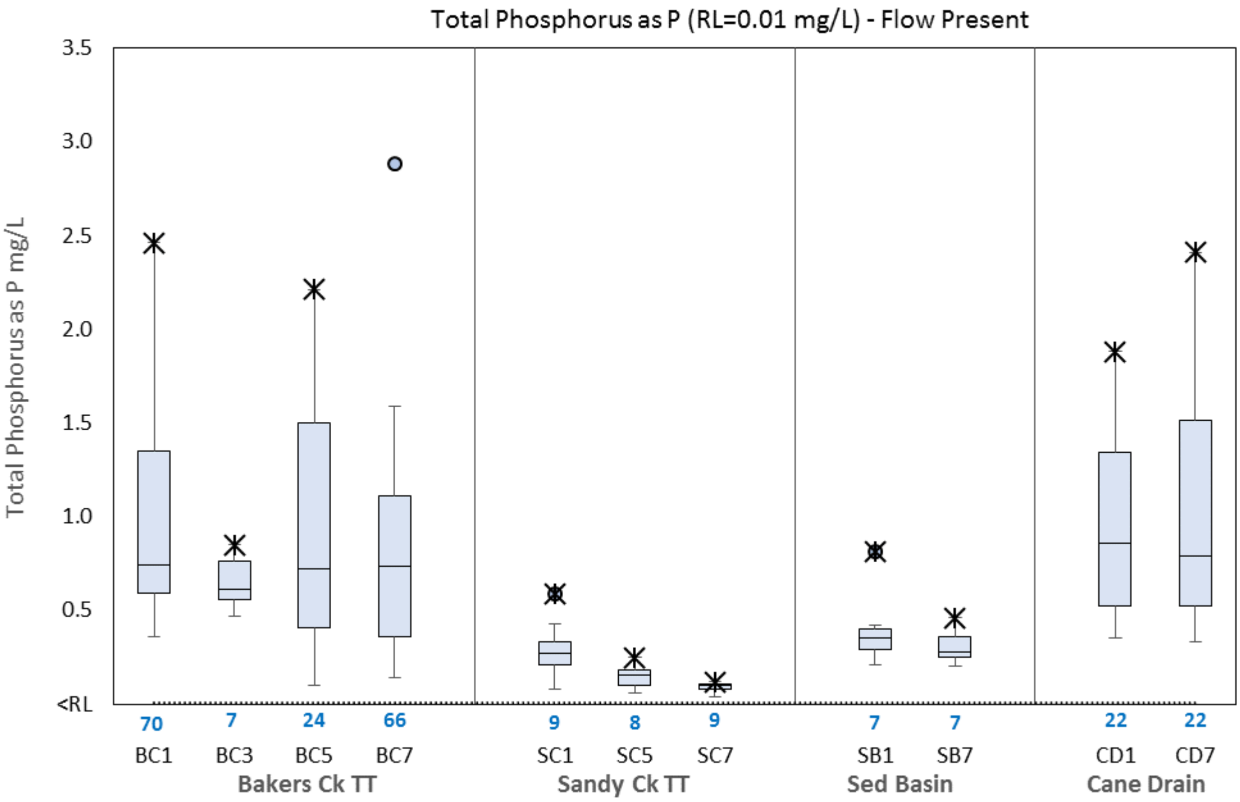


Figure 3.3.5 Spatio-temporal variations in reactive phosphorus concentrations during periods of flow

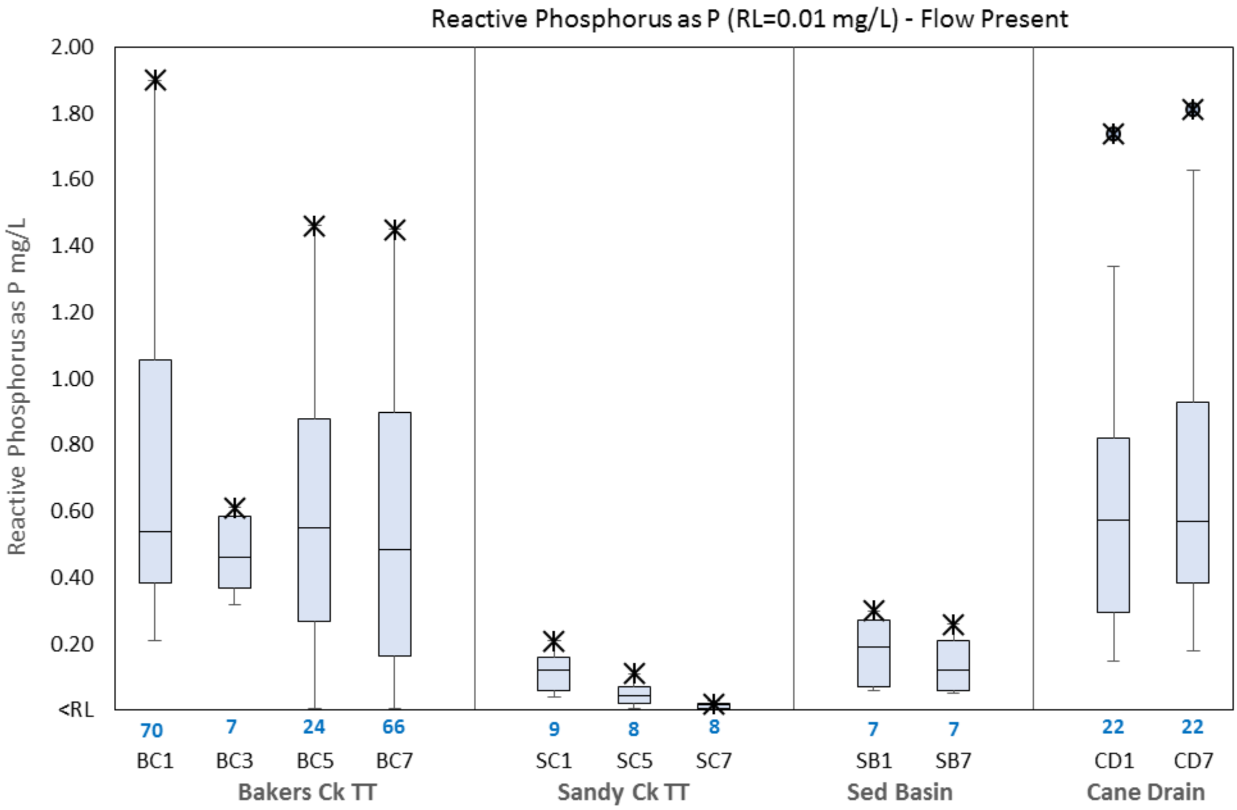


Figure 3.3.6 Spatio-temporal variations in Nitrite + Nitrate concentrations during periods of flow

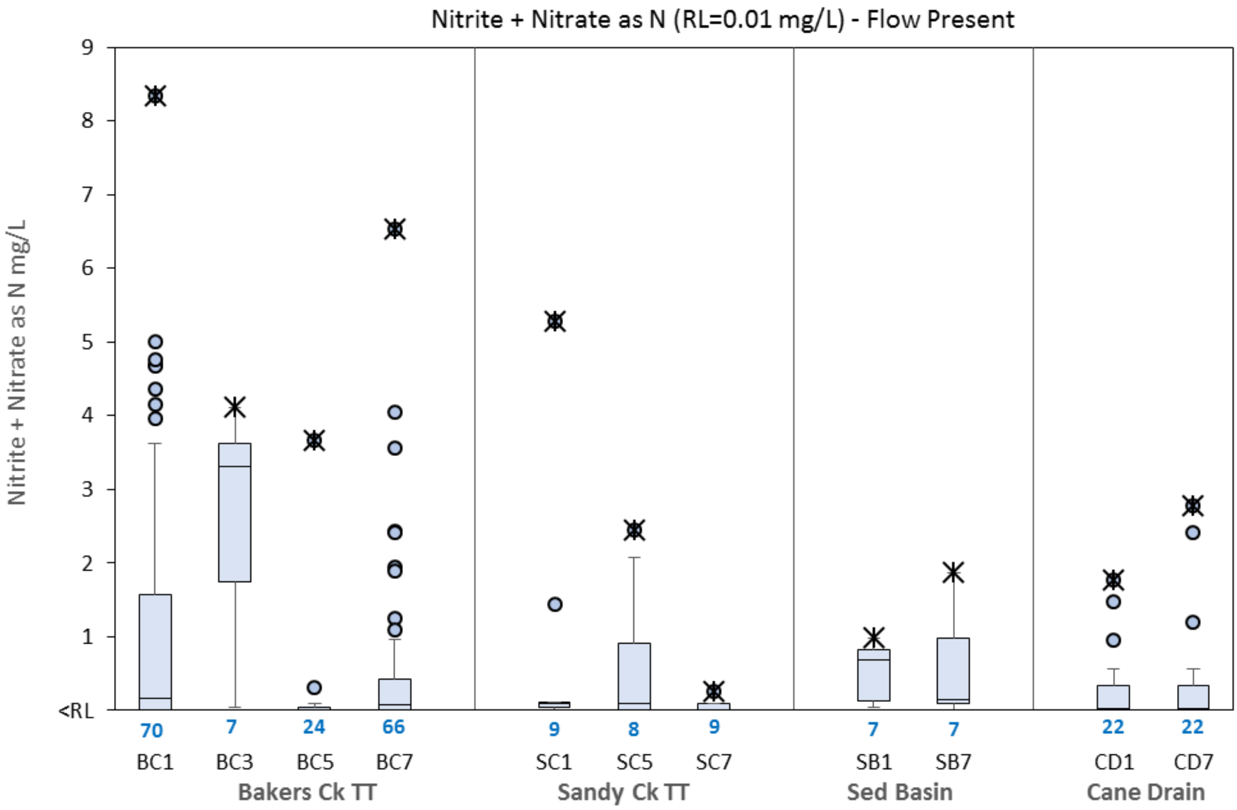
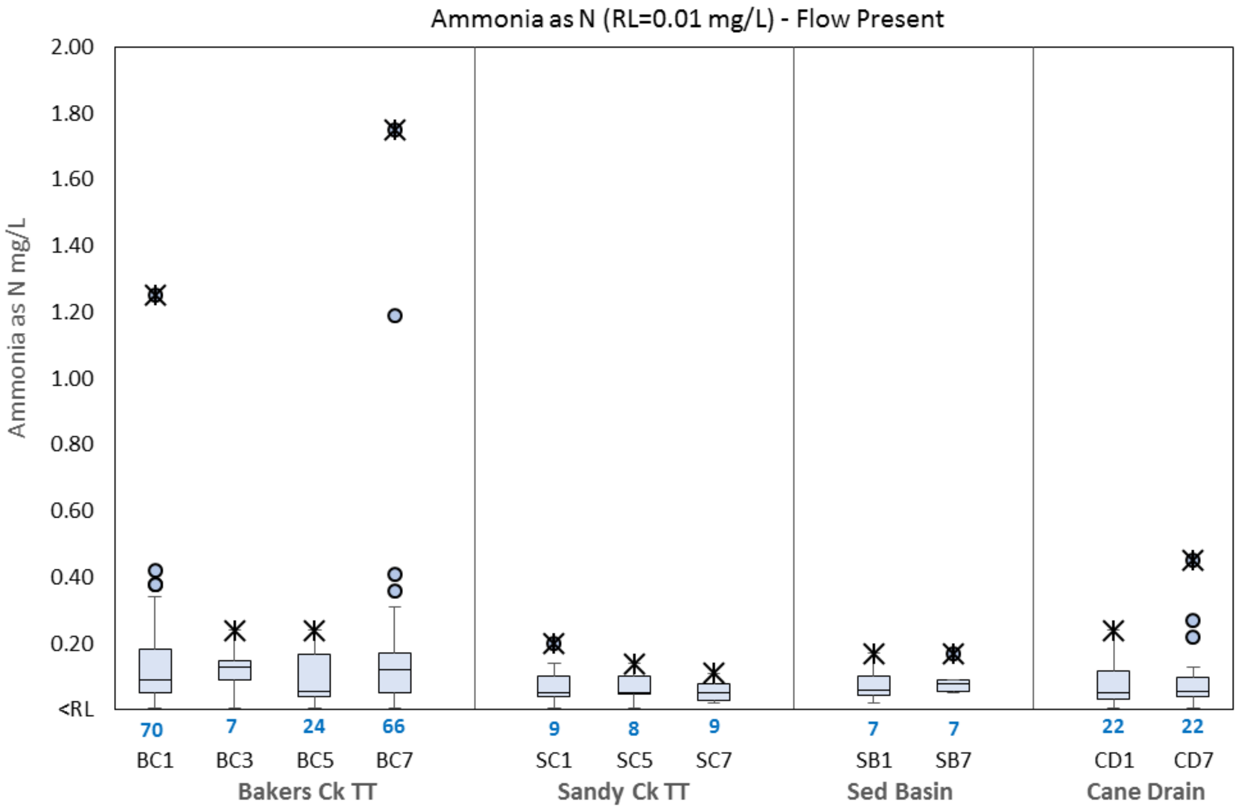


Figure 3.3.7 Spatio-temporal variations in total ammonia concentrations during periods of flow

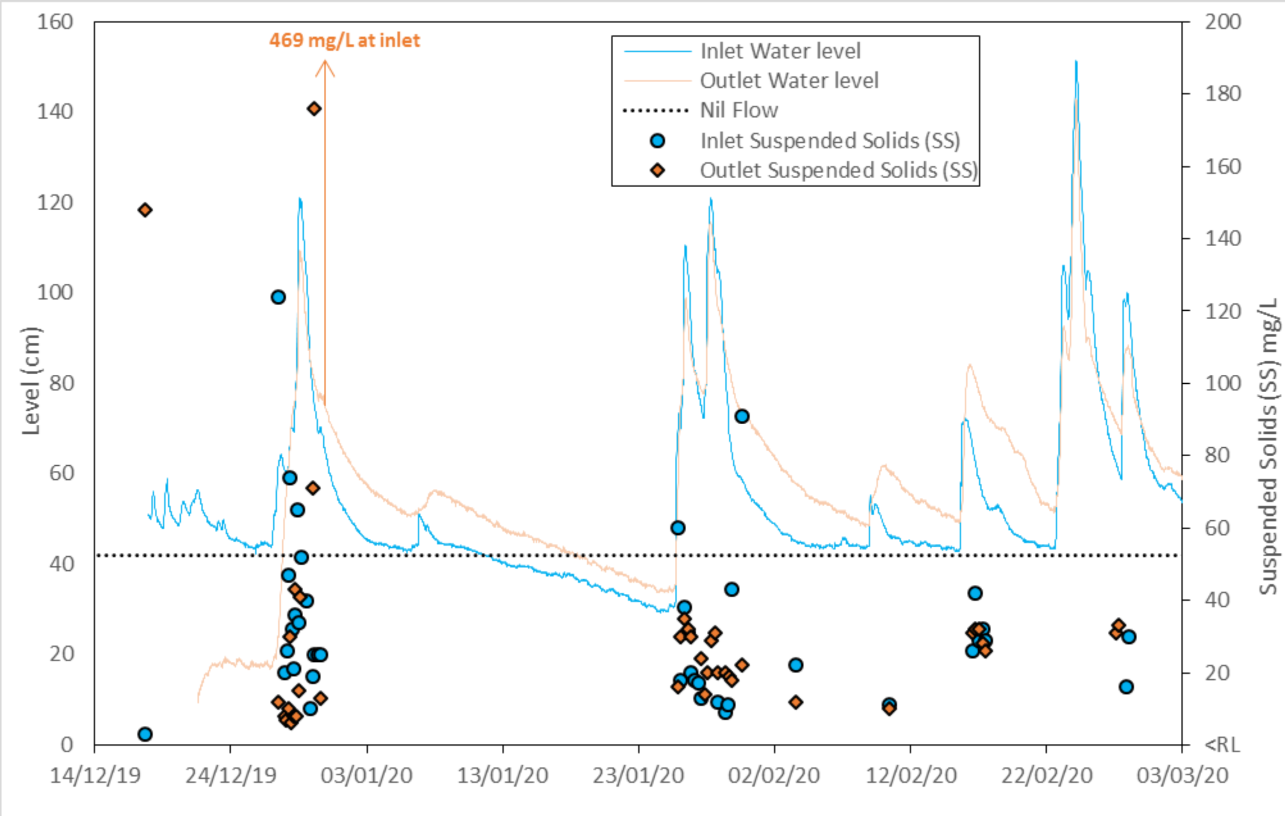


3.3.2 Event dynamics in Bakers Creek Wetland

All of the extreme BC7 SS concentrations shown previously in Figure 3.3.2 were reported in 2019. The highest result of 1040 mg/L was obtained on 24/9/19 and the third highest value of 292 mg/L was obtained on 18/10/19. Both were one-off samples taken during the dry season at a time when the depth loggers were not in place. Rainfall records suggest that discharge rates would have been moderate at the time, although it is understood that there could potentially have also been some irrigation runoff present at the time. Regardless it seems unlikely that the total load of particulates discharged would have been large compared to the quantities released during large scale wet season events, although in the absence of flow volume data there is no way of confirming if that was the case.

Notably, in both cases the concentrations at BC1 (17 mg/L and 3 mg/L, respectively) were orders of magnitude lower than BC7. A similar outcome was reported on 17/12/19 (see Figure 3.3.7a) prior to the first flush event at the end of December 2019. These results all raise the possibility that a brief pulse of turbid water had already passed though BC1 by the time each sample was taken but, due to the time taken for pulses to pass through the wetland, they were detected at the outlet. This highlights the inadvisability of employing one-off sampling tactics.

Figure 3.3.7a Temporal variations in suspended sediment concentrations during flow events



The dynamics of contaminant discharges from wetlands during flow events are difficult to predict, especially in cases such as this where critical drivers such as flow rates, detention times and standing water volumes have not yet been determined. However, it would seem reasonable to assume that BC1 would behave like a normal low order stream in which particulate contaminant concentrations typically increase rapidly on the rising limb (usually by orders of magnitude on first-flush and large-scale events), peak just before maximum discharge is reached and then fall back to moderate levels on the falling limb. In small flashy watercourses the concentration increases that occur on the rising hydrograph may not be sustained and may instead manifest as multiple very brief peaks rather than a single peak; nevertheless, there is still usually a clear correlation to the hydrograph. However, as can be seen in Figures 3.3.7b and c (SS), Figure 3.3.8 (TN) and Figure 3.3.9 (TP) the BC1 inlet site exhibited none of these expected characteristics.

Figure 3.3.7b Temporal variations in suspended sediment concentrations during the Dec 2019 flow event

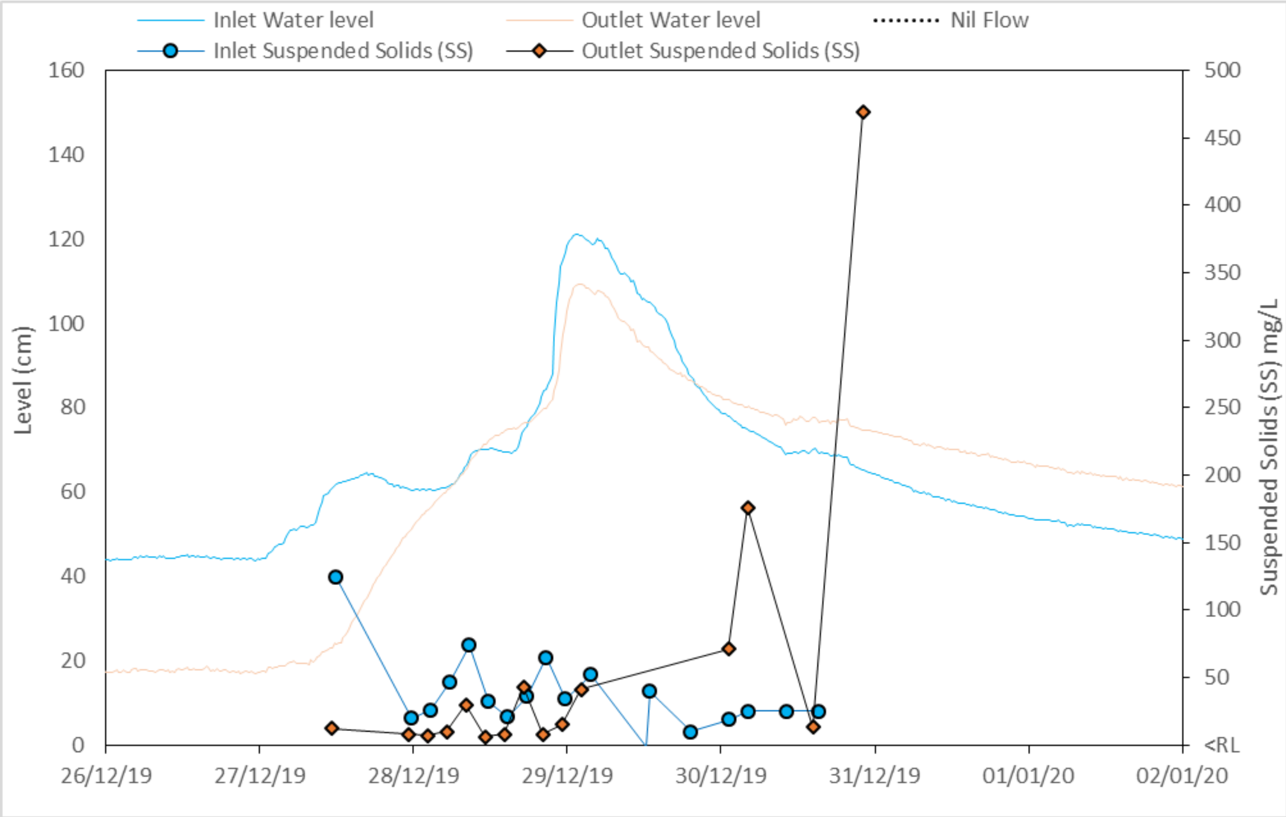


Figure 3.3.7c Temporal variations in suspended sediment concentrations during the Jan 2020 flow event

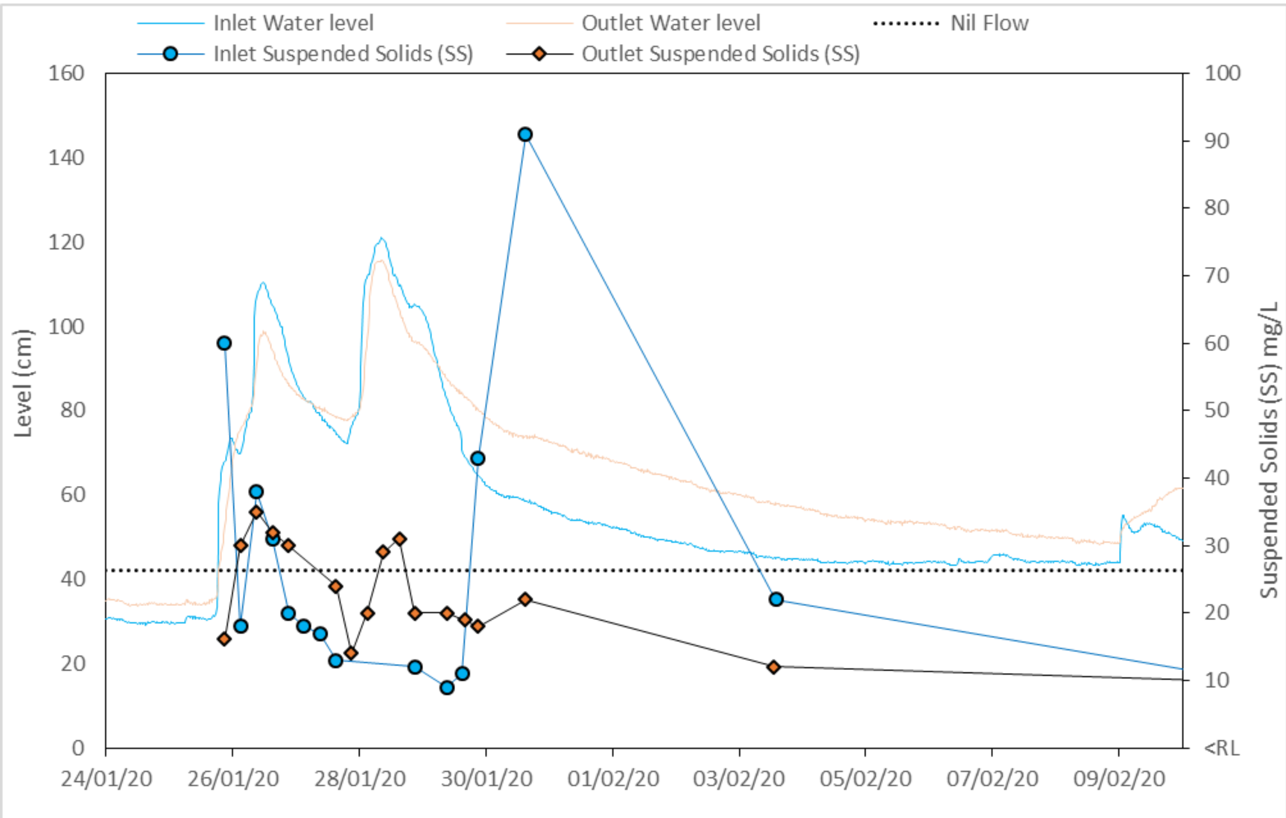


Figure 3.3.8 Temporal variations in total nitrogen concentrations during flow events

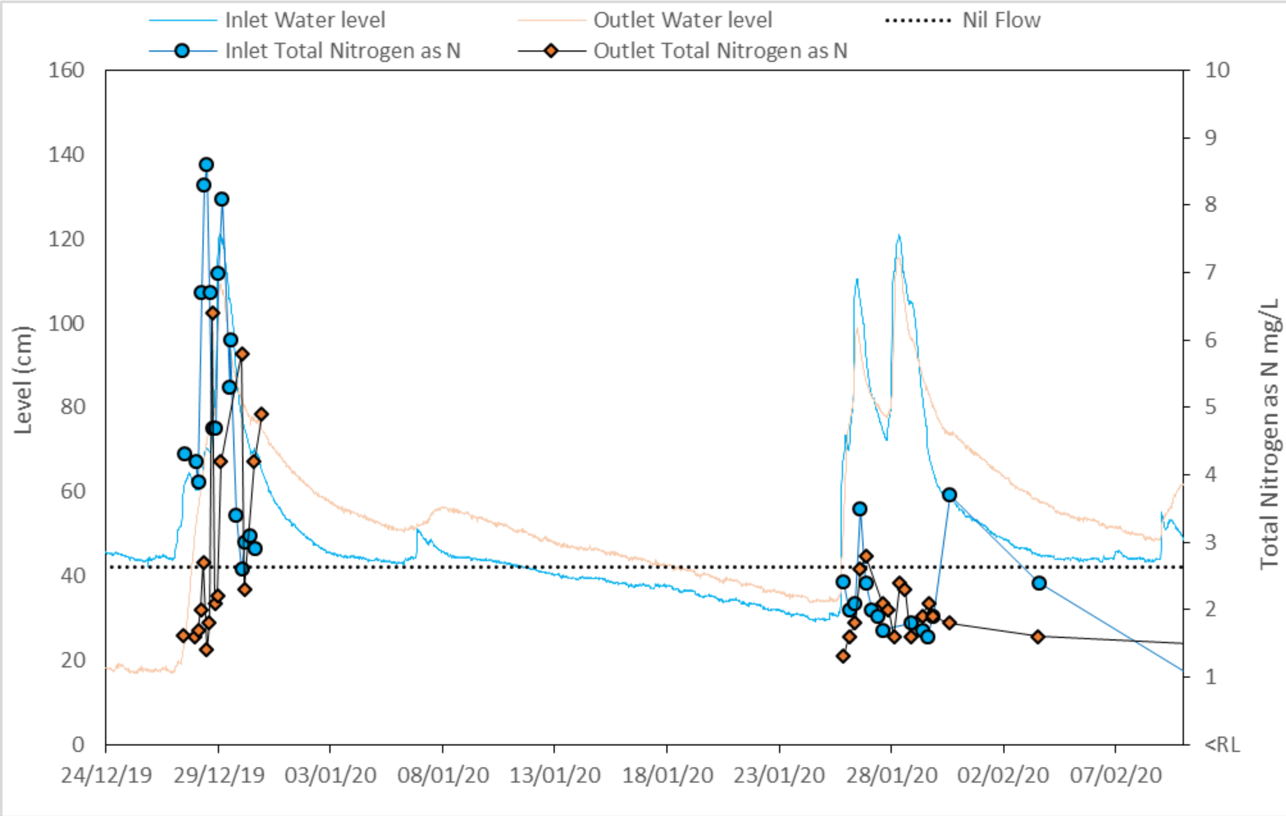
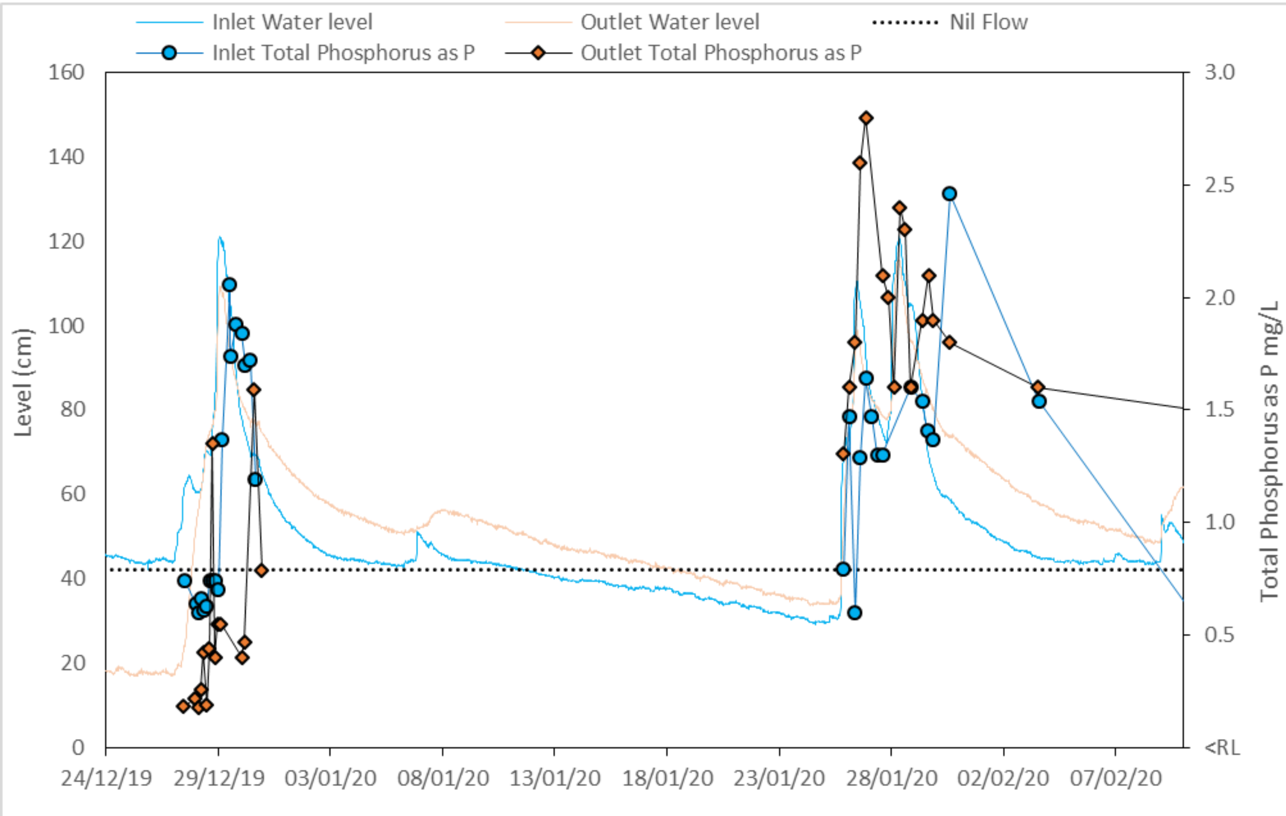


Figure 3.3.9 Temporal variations in total phosphorus concentrations during flow events



The SS concentration dynamics evident in Figures 3.3.7b and c are in fact as atypical as any these authors have ever seen. Further monitoring is clearly needed to confirm that the observed patterns are reproducible and not an artefact of the adopted sampling regime, and if necessary, to ascertain the source of the anomalous behaviour. There are a number of cases where SS varied by an order of magnitude between samples collected three hours apart, and it is noteworthy that the first sample collected at BC7 at the beginning of each of the two events shown in the figures was already elevated. This suggests the possibility that brief but intense SS peaks could have been missed between samples, and accordingly it would be advisable in future investigations to employ shorter sampling intervals (as described in Section 3.1).

It is also salient to note that the highest SS concentrations at BC1 on the first event and BC7 on the second event were each reported from the last sample collected on their respective autosampling runs. It is therefore difficult to infer the potential duration (and hence significance) of the high values. In order to avoid problems of this sort it would be highly advisable to ensure that high frequency autosampling is maintained for the full duration of events – ideally until outflows cease. Such tactics generate large numbers of samples and can become cost-prohibitive but there are a number of techniques that can be employed in order to address that potential problem. For example carefully contrived compositing techniques can be used to reduce the number of analytical samples and/or initial analyses can be performed on just a few parameters in order to ascertain discharge patterns and the information used to select a subset of samples for detailed analysis. As an adjunct or alternative to the latter approach it would be feasible to install turbidity loggers in order to obtain continuous data indicative of real-time fluctuations in particulate contaminant concentrations.

The elevated SS concentration reported on the tail of the event shown in Figure 3.3.7c is a potentially significant anomaly. The result correlates to TN and TP results and is therefore not likely an analytical error but since it is represented by only a single sample the possibility that it was simply a consequence of a brief localised disturbance to bottom sediments in the vicinity of the sampler, and therefore not indicative of the water flowing through the drain. However, if results are replicated and confirmed in the future they would deserve further investigation. In our experience SS pulses usually only occur on the falling limb of the hydrograph in situations where a stormwater detention structure such as a sediment basin has filled up and begun to overflow.

The data in Figures 3.3.8 (TN), 3.3.10 (NOx) and 3.3.11 (NH_3) suggest that on this occasion the first-flush in January 2020, was large enough to significantly deplete catchment supplies of nitrogen, as evidenced by the lower concentrations that were recorded at both the inlet and outlet during the subsequent flow event. As noted elsewhere in this report, due to existing data deficiencies, especially the lack of water quality data representative of the receding hydrograph (i.e. the stages circled in red on Figure 3.3.10), and uncertainties regarding the source of the atypical SS dynamics and EC discrepancies between the inlet and outlet, it is not feasible to assess how the wetland performed during the 2020 first-flush event. However, given the likelihood that water residence times in the wetland were quite low during the peak of this moderately large-scale event, there are theoretical grounds to suspect that contaminant removal capacity would have been low. That may not always be the case though because in this region it is not uncommon for drainage systems to experience a number of smaller-scale pre-flush events such as the one which occurred at the end of December 2019 (circled in green on Figure 3.3.10). These may generate sufficient runoff to refill the wetland without producing much outflow or flushing, and in some years it may take several smaller scale events of that kind to achieve a first-flush. The single pre-flush event shown in Figure 3.3.10 occurred almost immediately prior to the first-flush and would probably not have had much effect on wetland performance, but if there had been a number of these kinds of events spread over several weeks (which can happen some years) water residence times and therefore contaminant removal capacity would almost certainly have been substantially enhanced.

NOx concentrations discharging from the wetland at the beginning of the second event were low and increased during the event while NH3 levels exhibited the reverse trend. This is consistent with expectations given that DO levels in the wetland prior to the event were low enough to support denitrification and prevent oxidation of ammonia. In contrast to nitrogen there were no indications of potential TP (Figure 3.3.9) or reactive phosphorus (Figure 3.3.12) depletion in the aftermath of the first-flush.

It is pertinent to note that in agricultural landscapes of this kind variations between events must be interpreted very cautiously and it would be unwise to draw any conclusions regarding the performance of a stormwater drainage management system based on just a few events. This is because the already large natural between-event variations are amplified by numerous anthropogenic factors such as the timing of agrochemical applications relative to the event, the type of chemical used, catchment conditions prior to the event (e.g., ploughed field vs mature crop) and the stage of the farming cycle (e.g., plant vs ratoon).

Figure 3.3.10 Temporal variations in NO_x concentrations during flow events

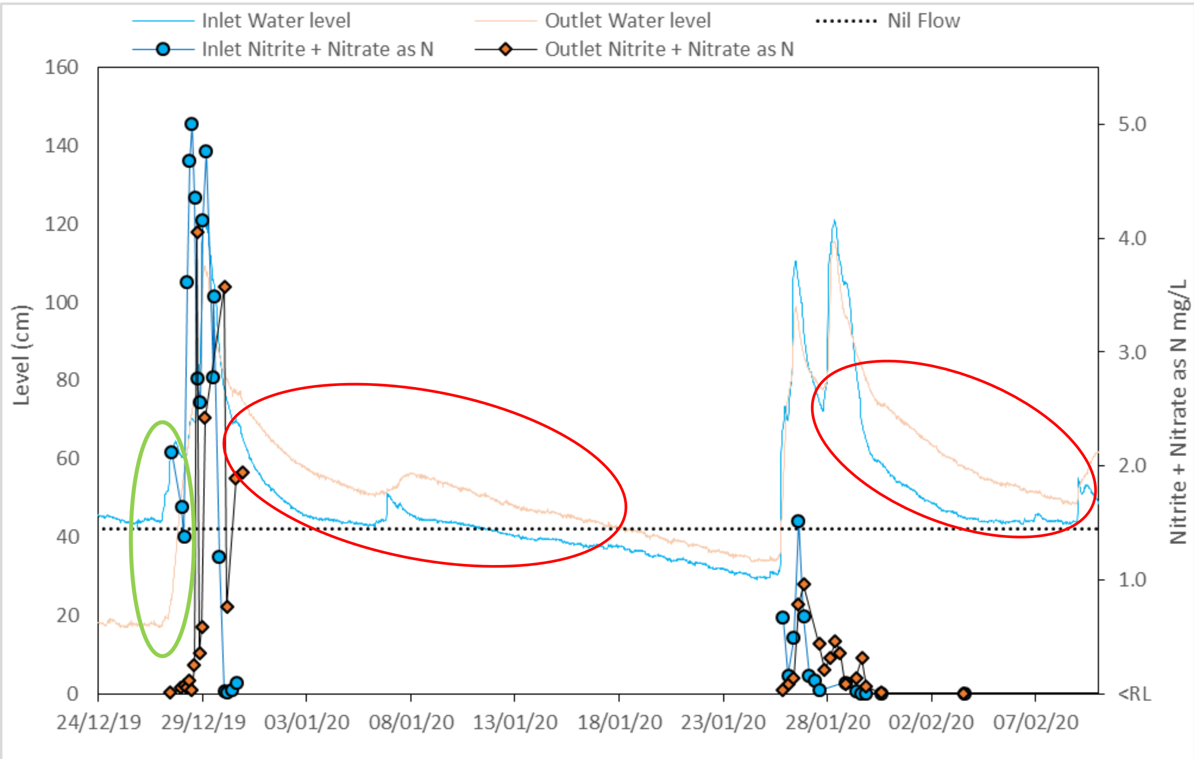


Figure 3.3.11 Temporal variations in ammonia concentrations during flow events

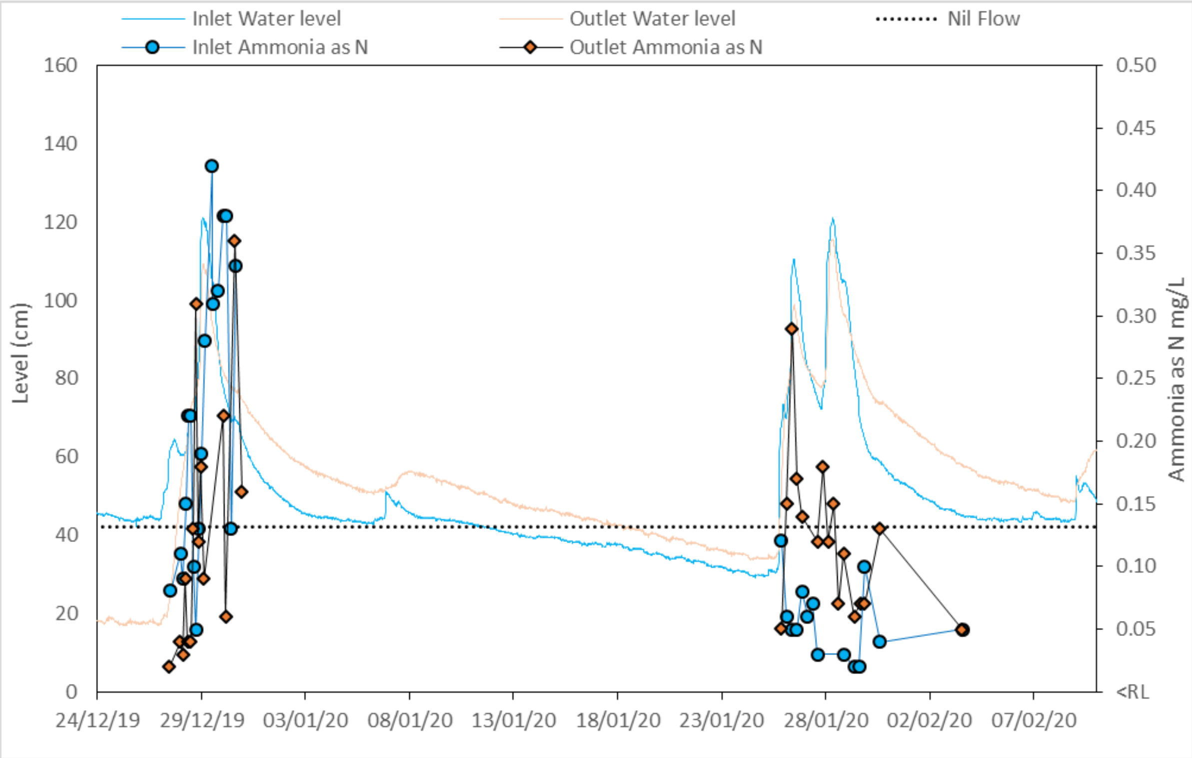
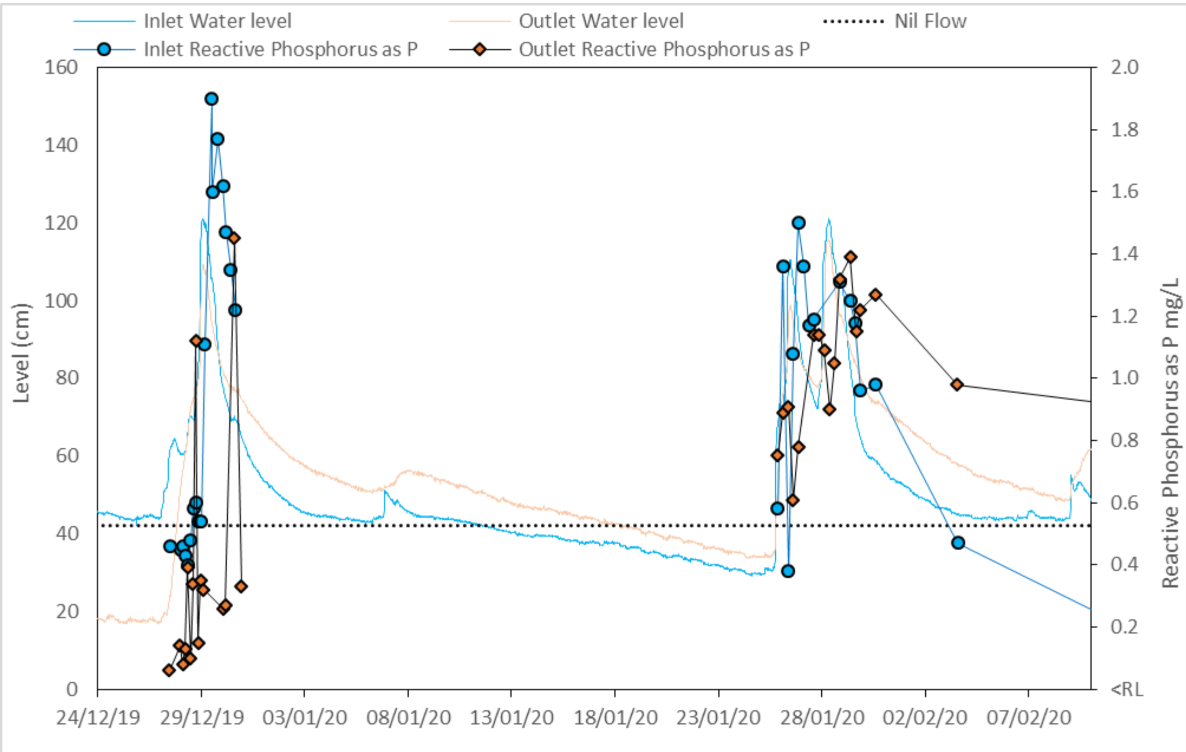


Figure 3.3.12 Temporal variations in reactive phosphorus concentrations during flow events



3.3.3 Flow Absent

As mentioned previously, water quality data collected under still water conditions (i.e. data obtained from manual samples collected at times when there was no visible surface outflow and autosampler samples taken when the water level was below the cease-to-flow point) do not provide a basis for assessing wetland water treatment performance but the data are useful for ascertaining if there is any evidence of any ambient water quality problems that could potentially undermine the health and functionality of the wetland. The available results indicate that ammonia (Figure 3.3.13) is the only one of the parameters being dealt with in this section that could conceivably present a potential risk of direct toxic effects on fauna inhabiting the wetland. All ammonia results were well below the default ANZECC TV of 0.9 mg/L (which is applicable to waters with pH 8). However, if pH levels were 9.5, for example, the TV would be 0.1 mg/L which is close to the 75th percentile of the overall dataset for the BC wetland.

In the absence of pH data or any knowledge of the waters pH buffering capacity it is difficult to infer whether extreme pH values of that sort are likely to occur in this wetland. However, TropWATER commonly encounter pH values in the 8.8 to 9.5 range in wetlands that support high levels of submergent macrophyte and/or phytoplankton productivity (Waltham and Fixler 2017, Waltham and Schaffer 2018, Waltham et al. 2019a), and which exhibit DO cycling levels similar to those observed in the Bakers Creek wetland. Therefore, until proven otherwise (by monitoring pH levels) it would be prudent to assume that extreme pH values and consequent ammonia toxicity would potentially occur. This in addition to the low DO levels discussed previously would make this wetland undesirable as fish habitat.

Chlorophyll concentrations (indicative of phytoplankton biomass) have not been monitored in the wetland; however; there have been anecdotal reports of algal blooms. The daily DO cycling records presented earlier tend to confirm that phytoplankton is high at times, especially at the outlet end of the wetland. This is also evidenced by the total nutrient data (phytoplankton being a significant contributor to particulate N and P concentrations) as can be seen in Figure 3.3.14. These results in combination with the daily DO cycling patterns suggest that SS levels are sufficiently low to allow enough light penetration to support plant growth. That contention is supported by the moderate ambient SS concentrations (Figure 3.3.15).

Figure 3.3.13 Spatio-temporal variations in ammonia concentrations during periods of no flow

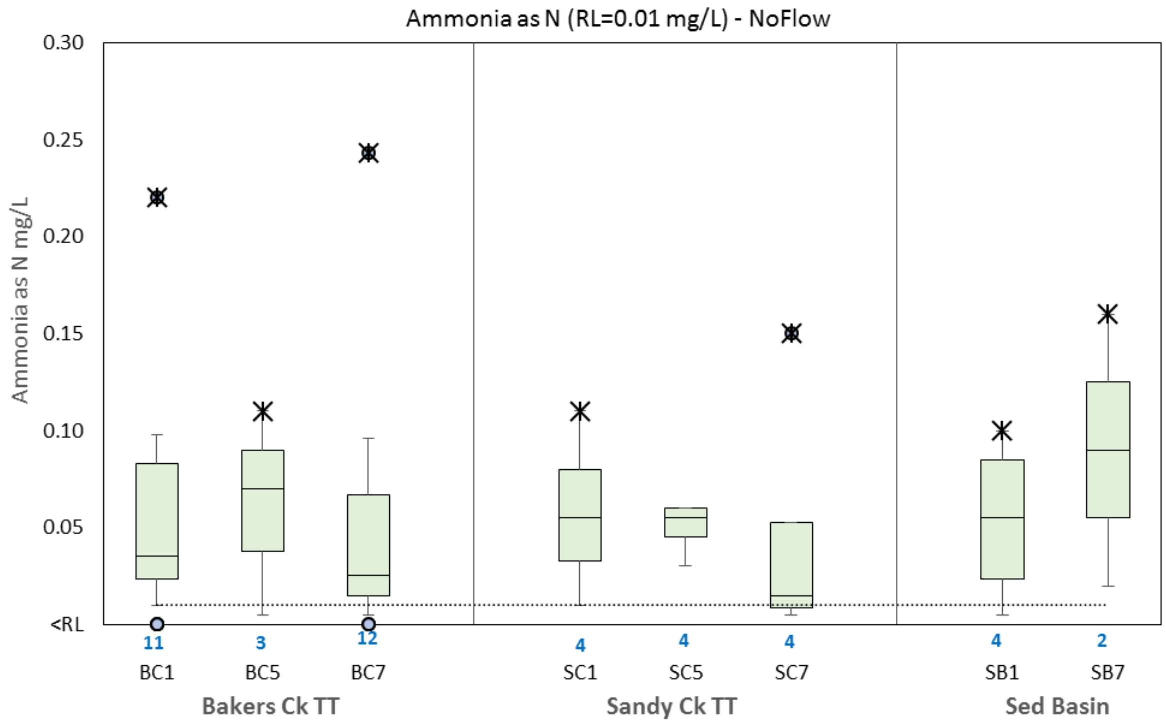


Figure 3.3.14 Spatio-temporal variations in TP concentrations during periods of no flow

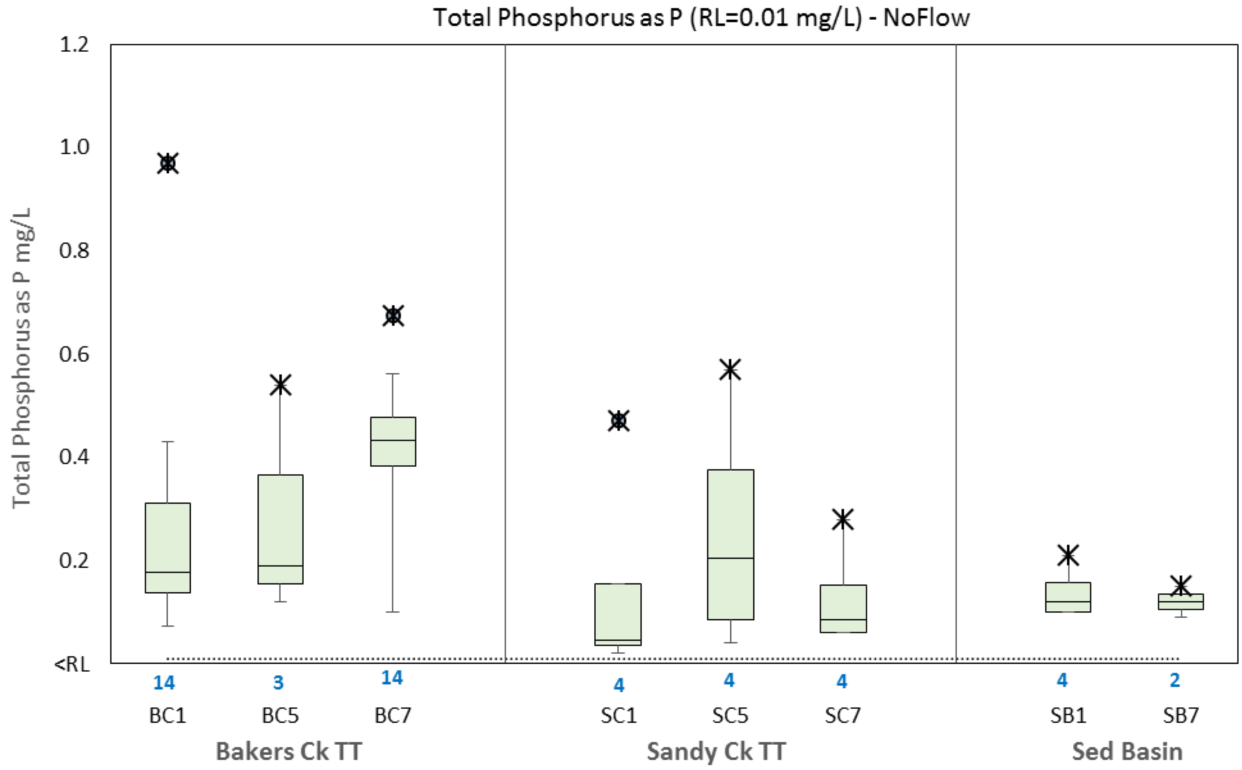


Figure 3.3.15 Spatio-temporal variations in SS concentrations during periods of no flow

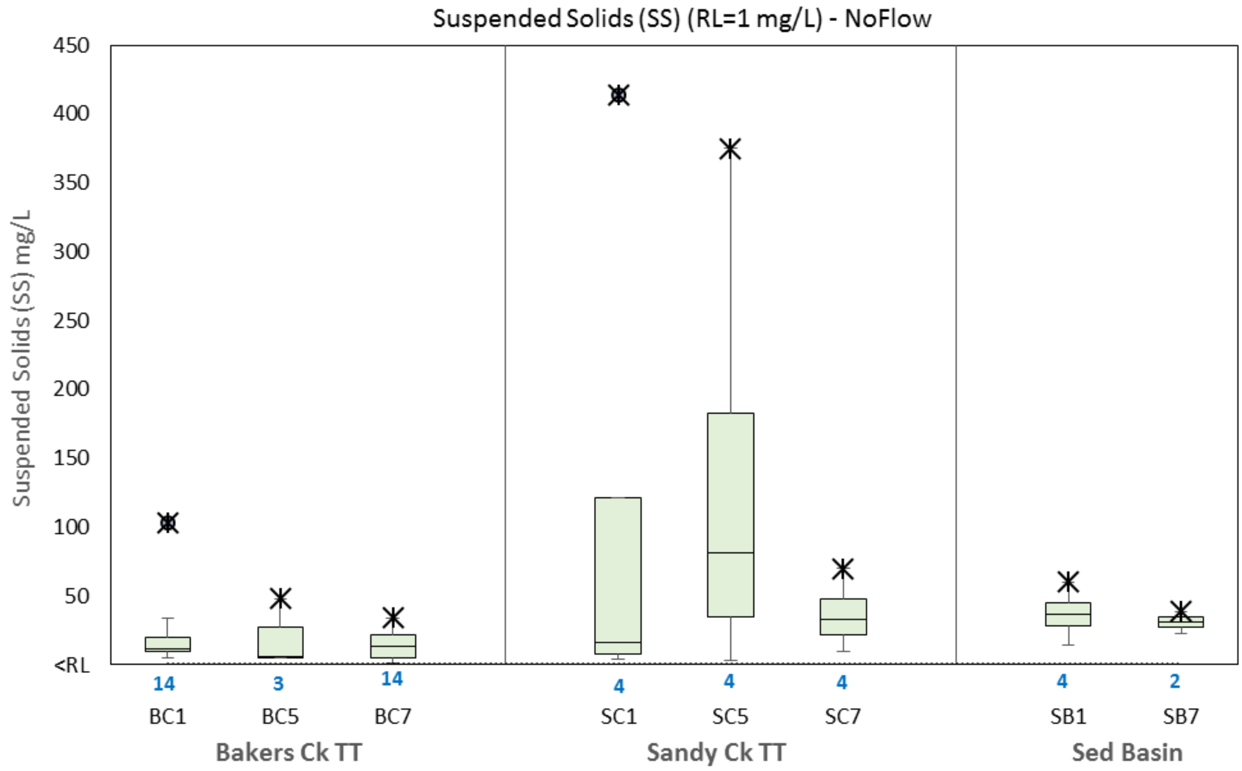
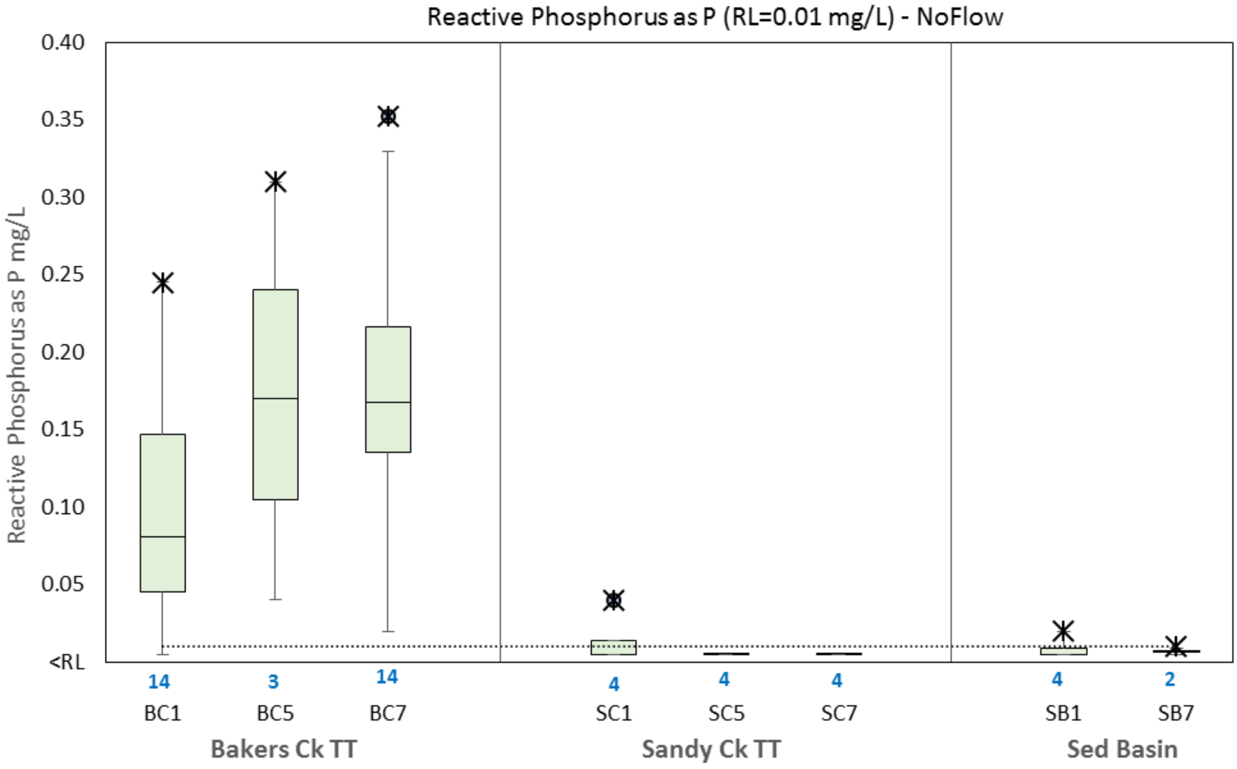


Figure 3.3.16 Spatio-temporal variations in reactive P concentrations during periods of no flow



3.4 Pesticides

The 18 pesticides that reported at least one detection throughout the entire dataset are listed in Table 3.4.1. The remaining 73 pesticides which did not report any detections are listed along with their reporting limits in Table 3.4.2. DKN (98.5%) and Imazapic (97%), which along with Isoxaflutole have only been monitored since Dec 2019, reported the highest detection frequencies. DKN is a derivative of isoxaflutole but the parent chemical was not detected in this study. Four other herbicides; Atrazine (91.4%), Imidacloprid, Diuron, Hexazinone, and Metolachlor (78.3%), listed in decreasing order, were detected in more than 75% of samples. While Metribuzin (55.1%), Ametryn, Propazine, Simazine, Fluometuron (15.9%) were reported in more than 15% of samples.

Table 3.4.1 Pesticides that reported at least one detection. PGV = Proposed Guideline Value.

Pesticide	Reporting Limit (µg/L)	Samples	Detections	Percent Detections	Freshwater		Marine	
					PGV for 95% protection (µg/L)	Percent Exceedances	PGV for 95% protection (µg/L)	PGV for 99% protection (µg/L)
Diketonitrile (DKN)	0.1	66	65	98.5	-	-	-	-
Imazapic	0.1	66	64	97.0	0.41	50.0	0.44	0.049
Atrazine	0.01	233	213	91.4	13	7.3	1.4	0.6
Imidacloprid	0.01	142	125	88.0	0.11	21.8	0.13	0.057
Diuron	0.02	226	194	85.8	0.23	69.5	0.67	0.43
Hexazinone	0.02	230	193	83.9	1.1	47.4	2.5	1.8
Metolachlor	0.01	226	177	78.3	0.71	9.7	0.084	0.0002
Metribuzin	0.02	227	125	55.1	2.6	18.1	2.7	2.0
Ametryn	0.01	220	89	40.5	0.33	0.0	0.61	0.1
Propazine	0.01	220	89	40.5	3.1	0.0	4.6	2.2
Simazine	0.02	220	45	20.5	10	0.0	63	28
Fluometuron	0.01	220	35	15.9	-	-	-	-
Bromacil	0.02	220	16	7.3	3.6	0.0	1.1	0.23
Propiconazole	0.05	220	7	3.2	-	-	-	-
Tebuconazole	0.01	220	5	2.3	-	-	-	-
Benomyl	0.01	220	4	1.8	-	-	-	-
Chlorpyrifos	0.02	220	1	0.5	0.01	0.5	0.009	0.00004
Thiamethoxam	0.02	220	1	0.5	-	-	-	-

PGV Reliability Classification
Very High
High
Moderate
Low
Very Low

In order to contextualise the data, ecosystem protection guidelines (where available) have been included in Table 3.4.1. The water quality guidelines for pesticides have been in a state of flux for a number of years and are currently under Federal review. For all pesticides other than Atrazine, the guidelines provided here are proposed values (PGVs) which were obtained from King *et al* (2017a and b) but which have not yet been formally sanctioned. The guideline for Atrazine has been obtained from the ANZECC 2000 Australian Water Quality Guidelines (AWQGs). It can be seen that apart from DKN, for which no GVs are currently available, all of the pesticides with a detection frequency greater than 50% reported some exceedances, the highest exceedance rates being reported for diuron (69.5%), Imazapic (50%), Hexazinone (47.4%), Imidacloprid (21.8%) and Metribuzin (18.1%).

All of these pesticides are herbicides and would be expected to potentially inhibit photosynthetic activity in the wetland, at least periodically. This could potentially be a factor contributing to the low photosynthetic DO production rates which were observed between the first and second flush events during the 2020 wet season; however, the daily DO cycling that was recorded in the wetland during the later stages of the wet season indicate that photosynthetic activity had largely been re-established.

Table 3.4.2 Pesticides that reported no detections. Number of samples was 220 in all cases expect for Isoxaflutole which was analysed on 66 samples.

Pesticide	Reporting Limit (µg/L)	Pesticide	Reporting Limit (µg/L)	Pesticide	Reporting Limit (µg/L)
Azinphos-methyl	0.02	Omethoate	0.01	Flusilazole	0.02
Azinphos-ethyl	0.02	Parathion	0.2	Hexaconazole	0.02
Bromophos-ethyl	0.01	Parathion-methyl	2	Paclobutrazole	0.05
Carbofenothion	0.02	Phorate	0.1	Penconazole	0.01
Chlorfenvinphos	0.02	Pirimiphos-ethyl	0.01	Cyprodinil	0.01
Chlorpyrifos-methyl	0.02	Pirimiphos-methyl	0.01	Pyrimethanil	0.02
Coumaphos	0.01	Profenofos	0.01	Tebuthiuron	0.02
Cyproconazole	0.02	Prothiofos	0.1	Chlorsulfuron	0.2
Demeton-O & Demeton-S	0.02	Sulfotep	0.005	Cyanazine	0.02
Demeton-S-methyl	0.02	Sulprofos	0.05	Cyromazine	0.05
Diazinon	0.01	Temephos	0.02	Prometryn	0.01
Dichlorvos	0.2	Terbufos	0.01	Terbuthylazine	0.01
Dimethoate	0.02	Tetrachlorvinphos	0.01	Terbutryn	0.01
Disulfoton	0.05	Triazophos	0.005	Diclofop-	0.05
EPN	0.05	Trichlorfon	0.02	Fenarimol	0.02
Ethion	0.02	Trichloronate	0.5	Irgarol	0.002
Ethoprophos	0.01	Aldicarb	0.05	Oxyfluorfen	1
Fenamiphos	0.01	Bendiocarb	0.1	Isoxaflutole*	0.01
Fenchlorphos (Ronnel)	10	Carbaryl	0.01	Oxamyl	0.01
Fenitrothion	2	Carbofuran	0.01	Thiobencarb	0.01
Fensulfothion	0.01	3-Hydroxy Carbofuran	0.02	Thiodicarb	0.01
Fenthion	0.05	Methiocarb	0.01	Pendimethalin	0.05
Malathion	0.02	Methomyl	0.01	Trifluralin	10
Mevinphos	0.02	Molinate	0.1		
Monocrotophos	0.02	Difenoconazole	0.02		

3.4.1 *Flow Present*

Statistical summaries of the data obtained from samples collected while flow was present are displayed in Figures 3.4.1 to 3.4.12 which encompass the 12 most frequently detected pesticides. It is important to remember that, as detailed in Section 3.1, it has been necessary to censor these data by removing <RL values obtained from diluted samples which yielded unacceptably high RL values. This creates an imbalance that must be taken into consideration when interpreting the box and whisker plots.

Notably samples were diluted (usually by 100-fold), if just one pesticide (most commonly Atrazine) was present in excessive concentrations. In most case this meant that the sample in question yielded no usable results for any other pesticide, even though it is likely that they would have contained elevated concentrations of several of them. Accordingly, it is possible that the number of extreme values reported for pesticides other than Atrazine may have been underestimated here. In some cases this has possibly affected the top half of the box and whiskers plot (i.e., the 75th percentile and above), but the median and lower percentiles are unlikely to have been affected.

As previously discussed, it is uncertain whether the event sampling conducted to date have been of sufficient frequency and duration to detect pesticide pulses at both ends of the wetland. The validity of employing concentration data (rather than load data) to assess wetland performance has also been questioned and there are indications (from the EC data) that there may be unmonitored inputs to the wetland, which if present, would invalidate any performance assessment that could be conducted, until the source has been identified and monitored.

Figure 3.4.1 Spatio-temporal variations in atrazine concentrations during periods of flow. Note that some data has been censored (refer to text).

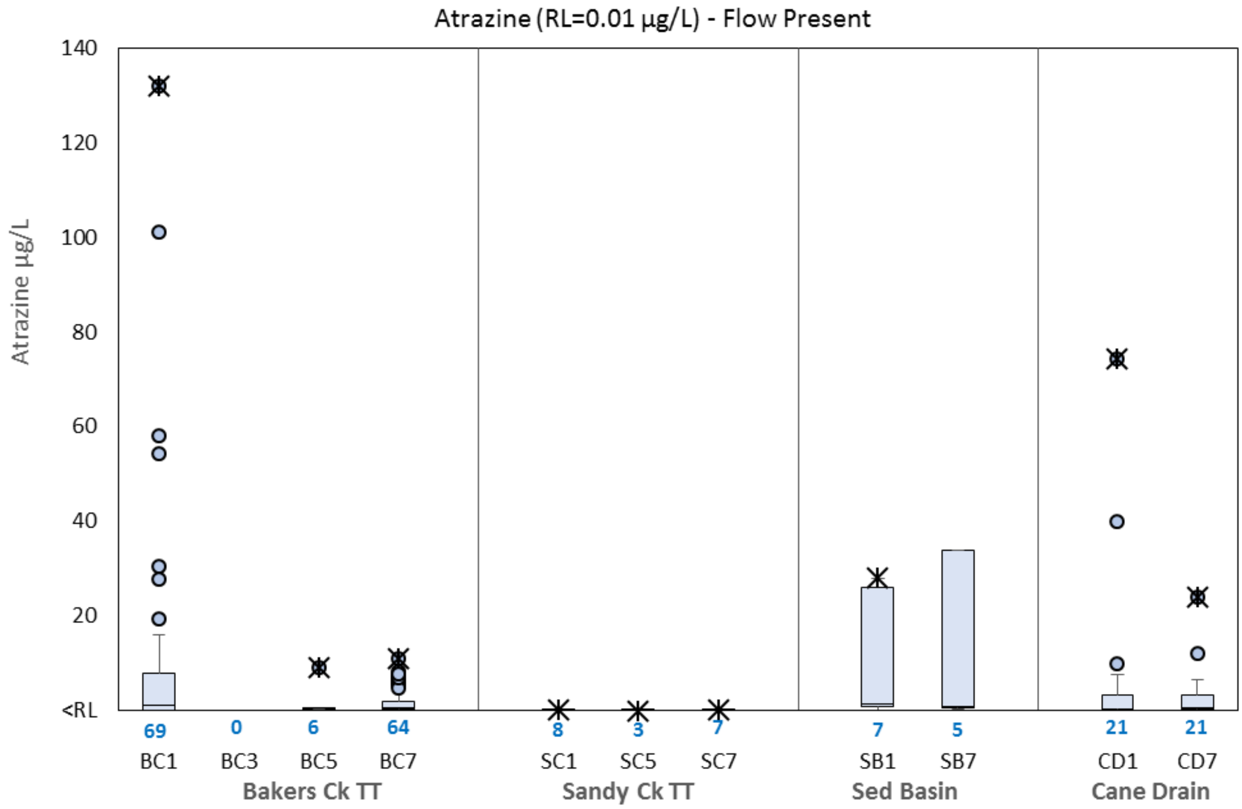


Figure 3.4.2 Spatio-temporal variations in imidacloprid concentrations during periods of flow. One extreme outlier of 8.8 µg/L at SB7 not shown. Note that some data has been censored (refer to text).

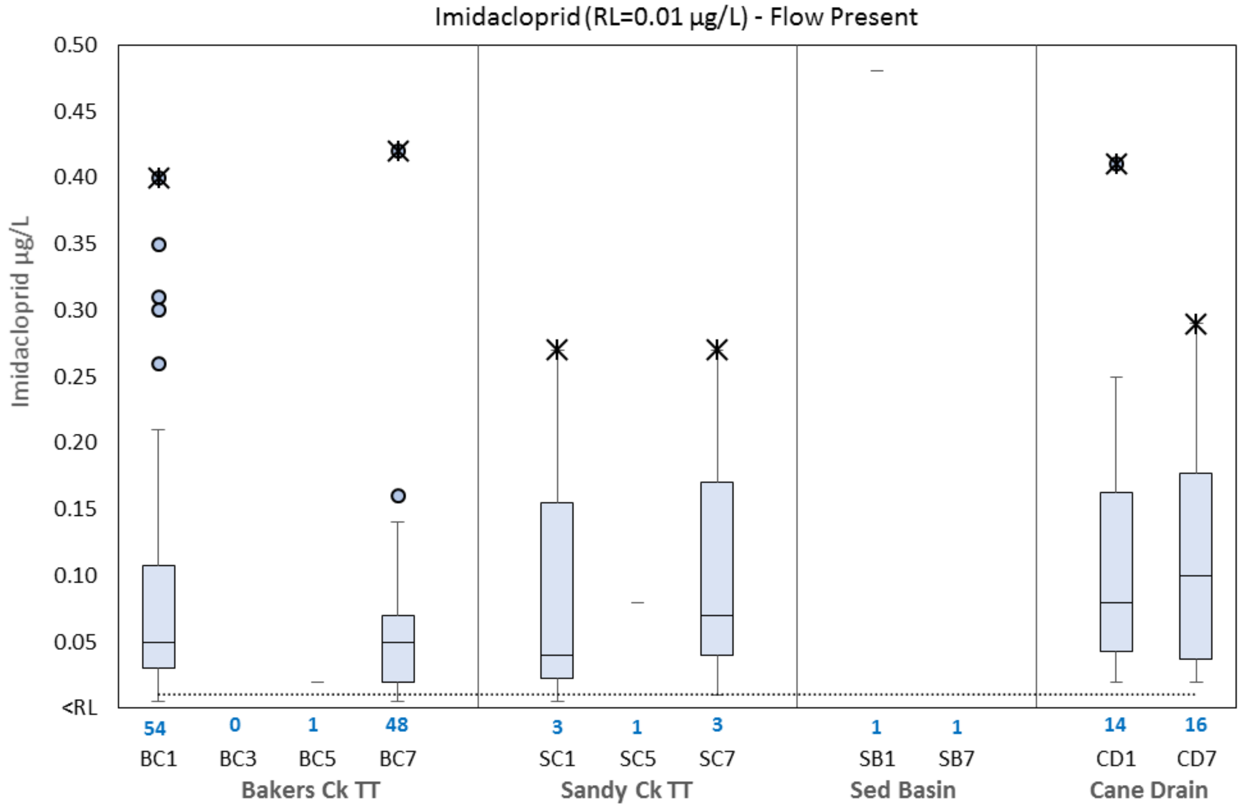


Figure 3.4.3 Spatio-temporal variations in diuron concentrations during periods of flow. Note that some data has been censored (refer to text). Outliers of 38.8, 85 and 91 µg/L at BC1 are not shown.

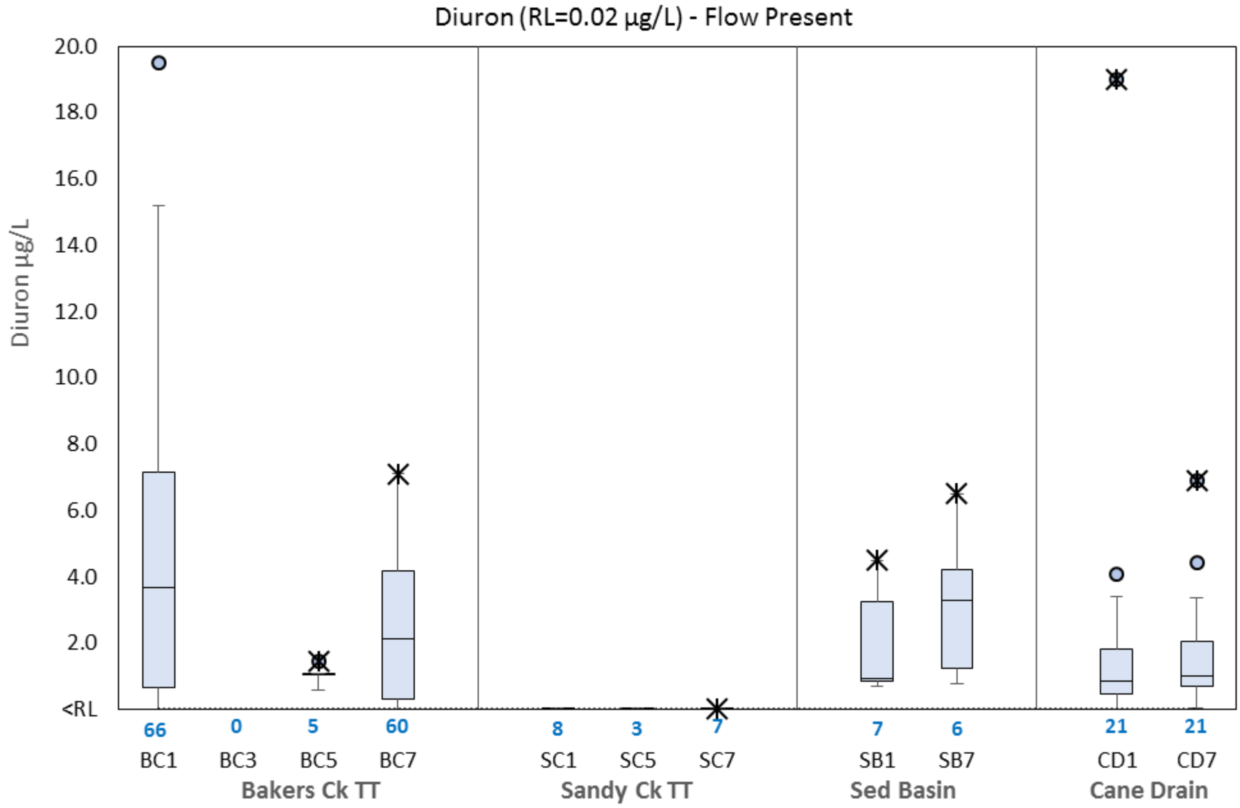


Figure 3.4.4 Spatio-temporal variations in DKN concentrations during periods of flow. Note that some data has been censored (refer to text).

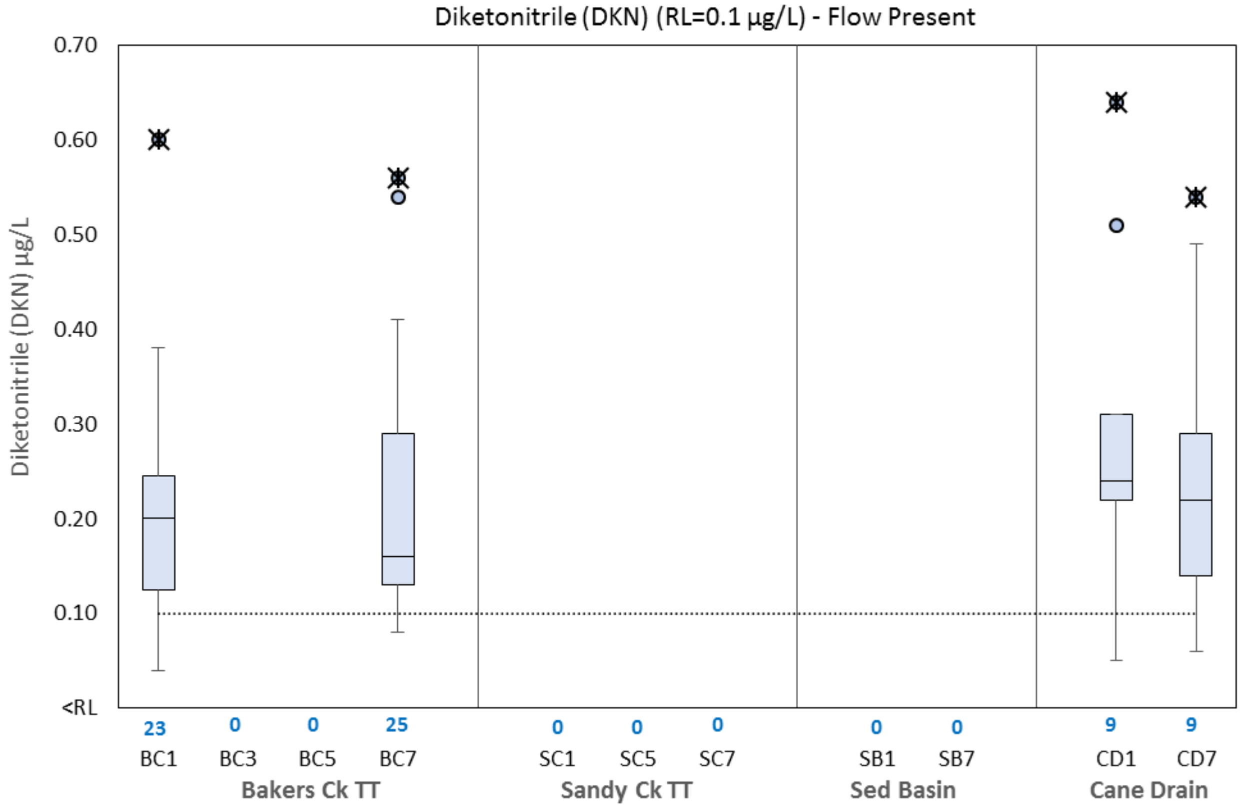


Figure 3.4.5 Spatio-temporal variations in Imazapic concentrations during periods of flow. Note that some data has been censored (refer to text).

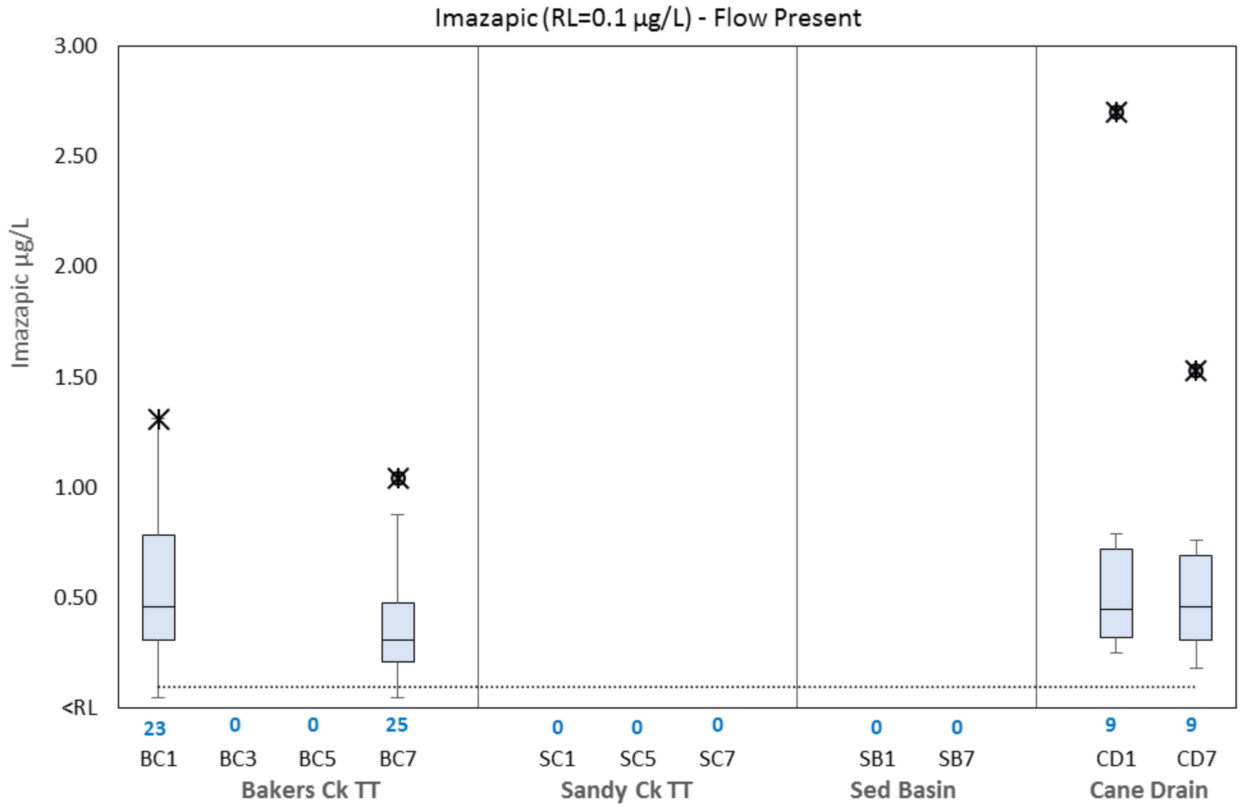


Figure 3.4.6 Spatio-temporal variations in hexazinone concentrations during periods of flow. Note that some data has been censored (refer to text).

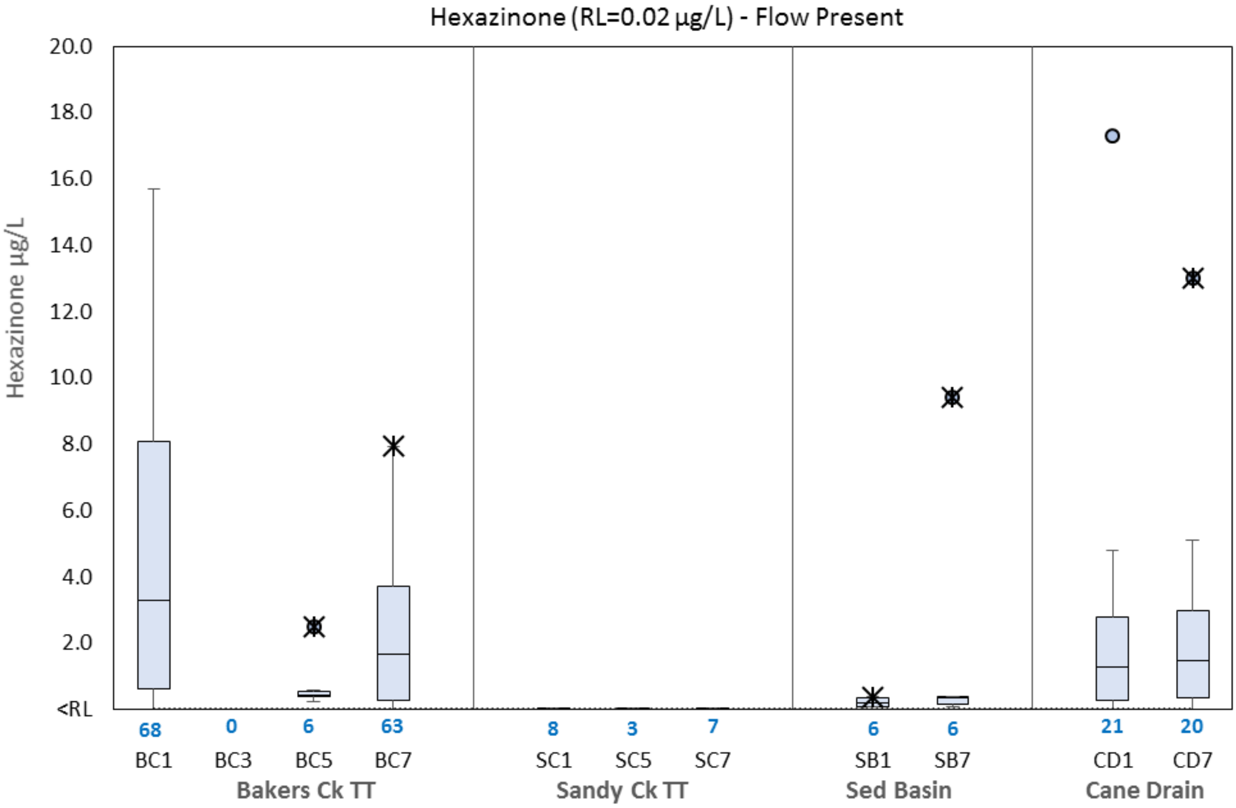


Figure 3.4.7 Spatio-temporal variations in metalochlor concentrations during periods of flow. Note that some data has been censored (refer to text). Extreme outliers of 5.2 µg/L at CD1 and 3.7 µg/L at BC1 not shown

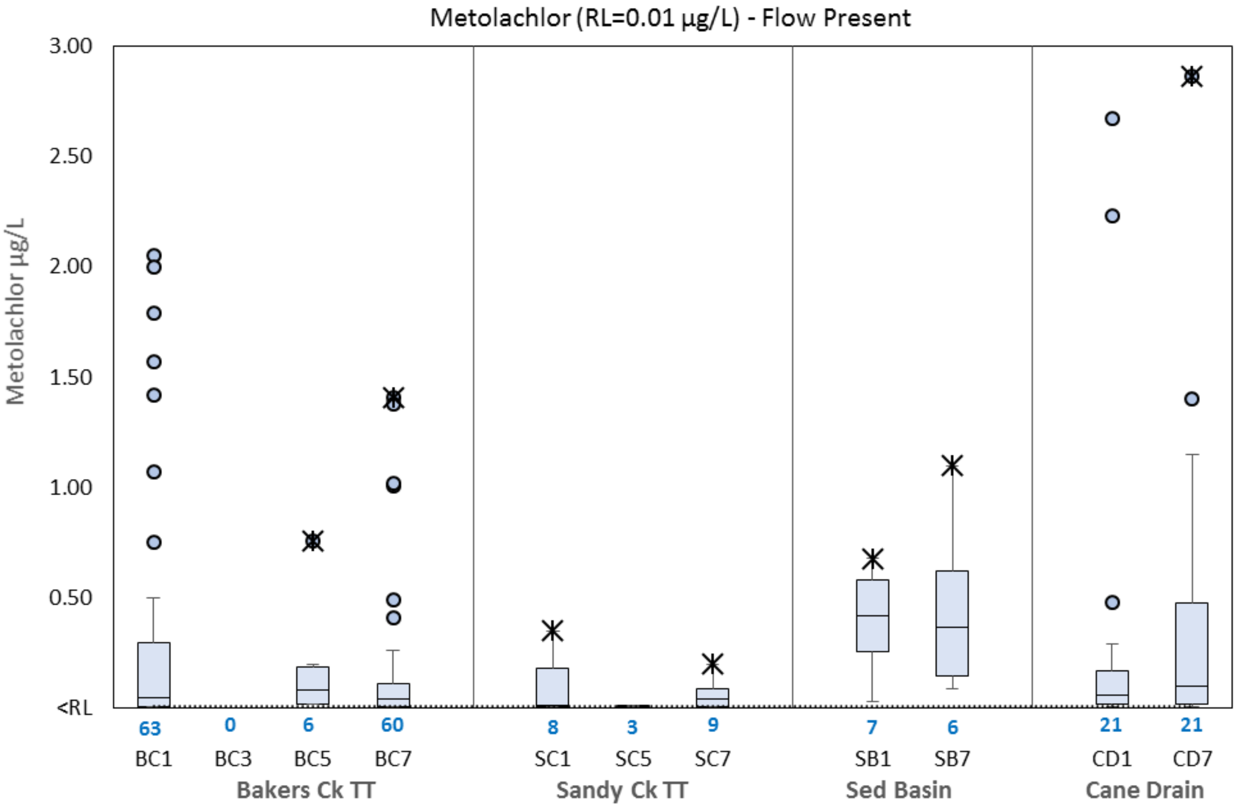


Figure 3.4.8 Spatio-temporal variations in metribuzin concentrations during periods of flow. Note that some data has been censored (refer to text). Extreme outliers of 43.8, 52 and 52.9 µg/L at CD1 not shown

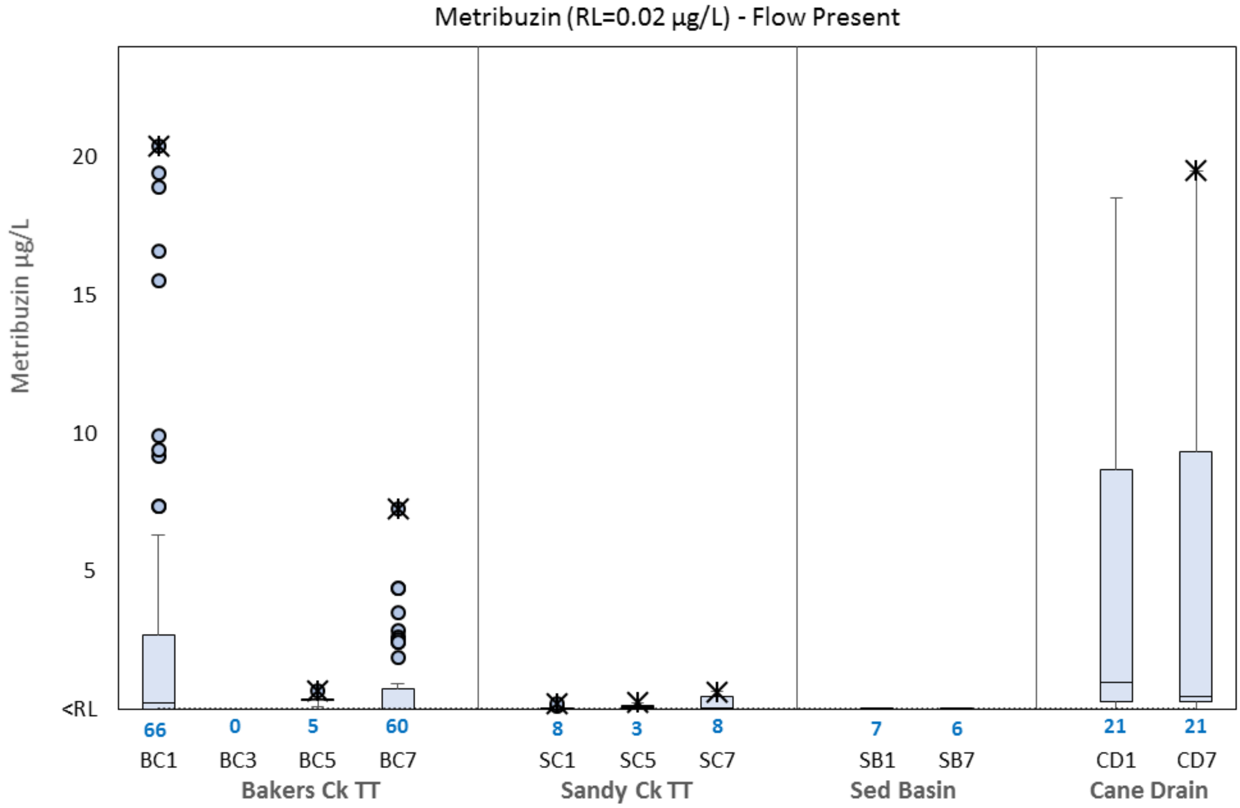


Figure 3.4.9 Spatio-temporal variations in ametryn concentrations during periods of flow. Note that some data has been censored (refer to text).

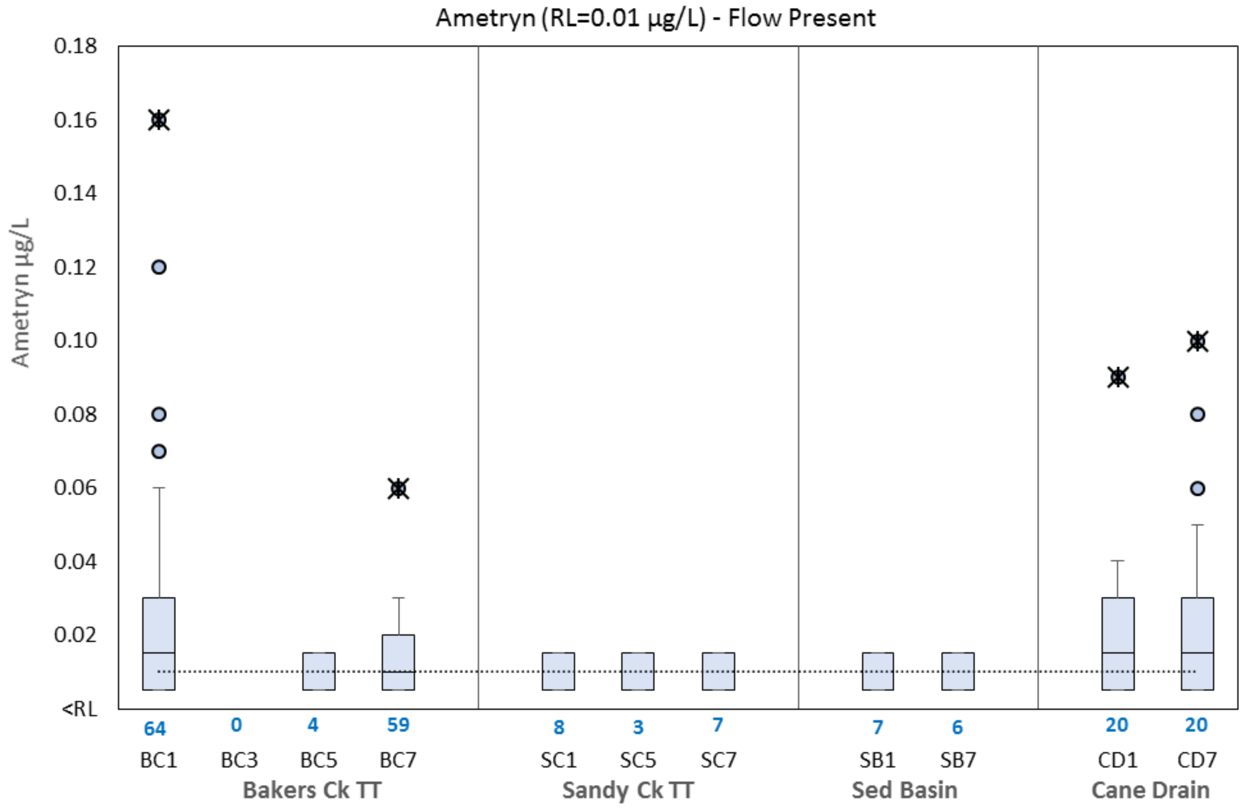


Figure 3.4.10 Spatio-temporal variations in propazine concentrations during periods of flow. Note that some data has been censored (refer to text).

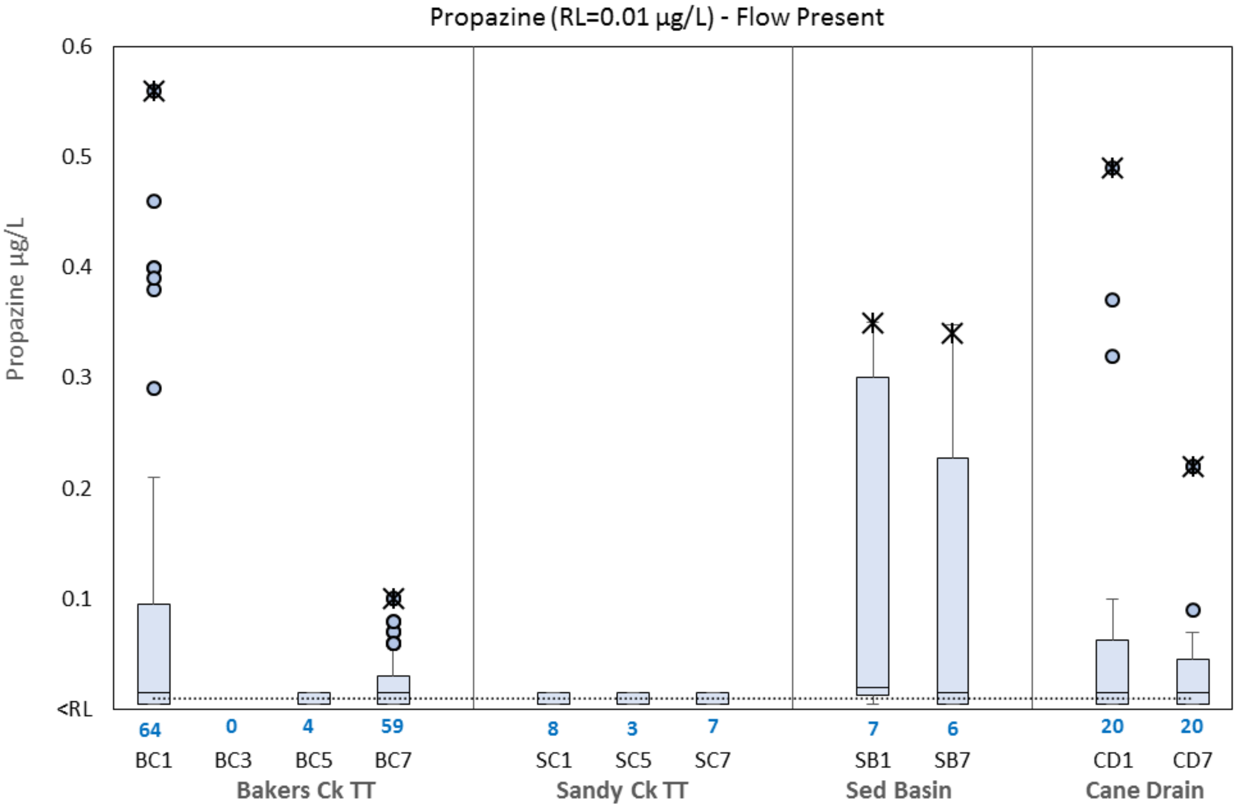


Figure 3.4.11 Spatio-temporal variations in simazine concentrations during periods of flow. Note that some data has been censored (refer to text).

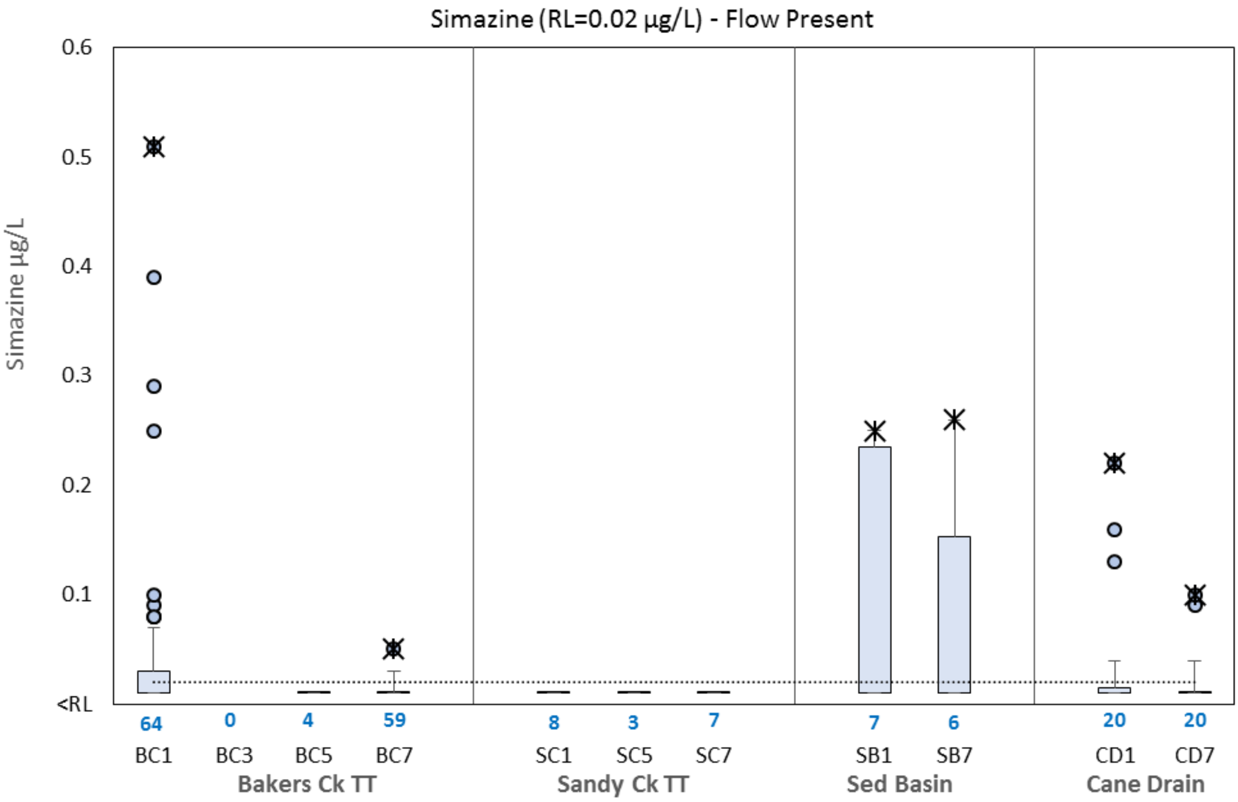
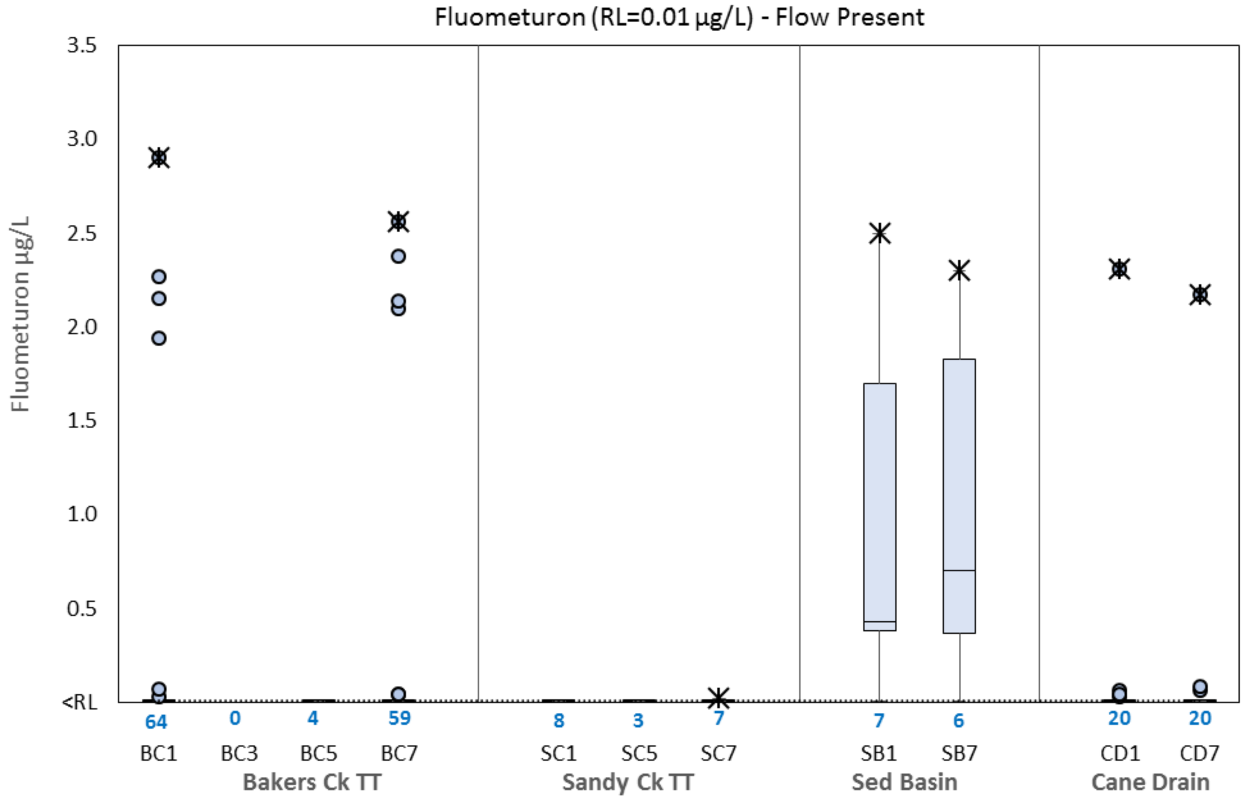


Figure 3.4.12 Spatio-temporal variations in Fluometuron concentrations during periods of flow. Note that some data has been censored (refer to text).



These existing uncertainties notwithstanding, if the plots in Figure 3.4.1 to 3.4.12 were to be taken at face value, there are indications that the wetland may have had some subtle effects on most of the pesticides presented here, other than DKN and Fluometuron (for which there was no difference between inlet and outlet concentrations) and Imidacloprid (which is a marginal case). The effects implied is a general reduction in the number and magnitude of extreme concentrations and a slight reduction in median concentrations. The net benefit of reducing peak concentrations cannot be quantified in the absence of flow and load data (i.e. removal of an extreme value that occurred during a period of low flow may be of little consequence but removal of a high peak could potentially have a substantial effect on event-mean concentrations and export loads). However, the next section which displays temporal variations in pesticide concentrations in the context of depth variations (indicative of hydrographic fluctuations) provides some qualitative indications of the potential significance of the observed differences.

3.4.2 Event dynamics in Bakers Creek Wetland

Figure 3.4.13(Atrazine), 3.4.15(Diuron), 3.4.16(Hexazinone), 3.4.17(Metalochlor), 3.4.19(Propazine), and 3.4.20 (Simazine) indicate that all of the extreme concentrations recorded at the inlet during these events occurred on the rising limb and peak of the hydrograph (when discharge volumes and therefore contaminant loads would have been high) and no equivalently elevated levels were recorded at the outlet. If one accepts the premise that there were no further releases of elevated concentrations from the outlet later in the hydrograph and that there was no substantial dilution from an unmonitored input source, then these results imply that the wetland may have effected a reduction in export load of those particular pesticides.

Imidacloprid (Figure 3.4.14), is once again a marginal case, as a few high values (including the highest reported value) were recorded at the outlet during times of high flow. Similar comments are applicable to Metalochlor (Figure 3.4.17) and Imazapic (Figure 3.4.22). DKN is the only pesticide for which the available data for these events provide no indication of any potential for a reduction in export load.

Figure 3.4.13 Temporal variations in atrazine concentrations during flow events

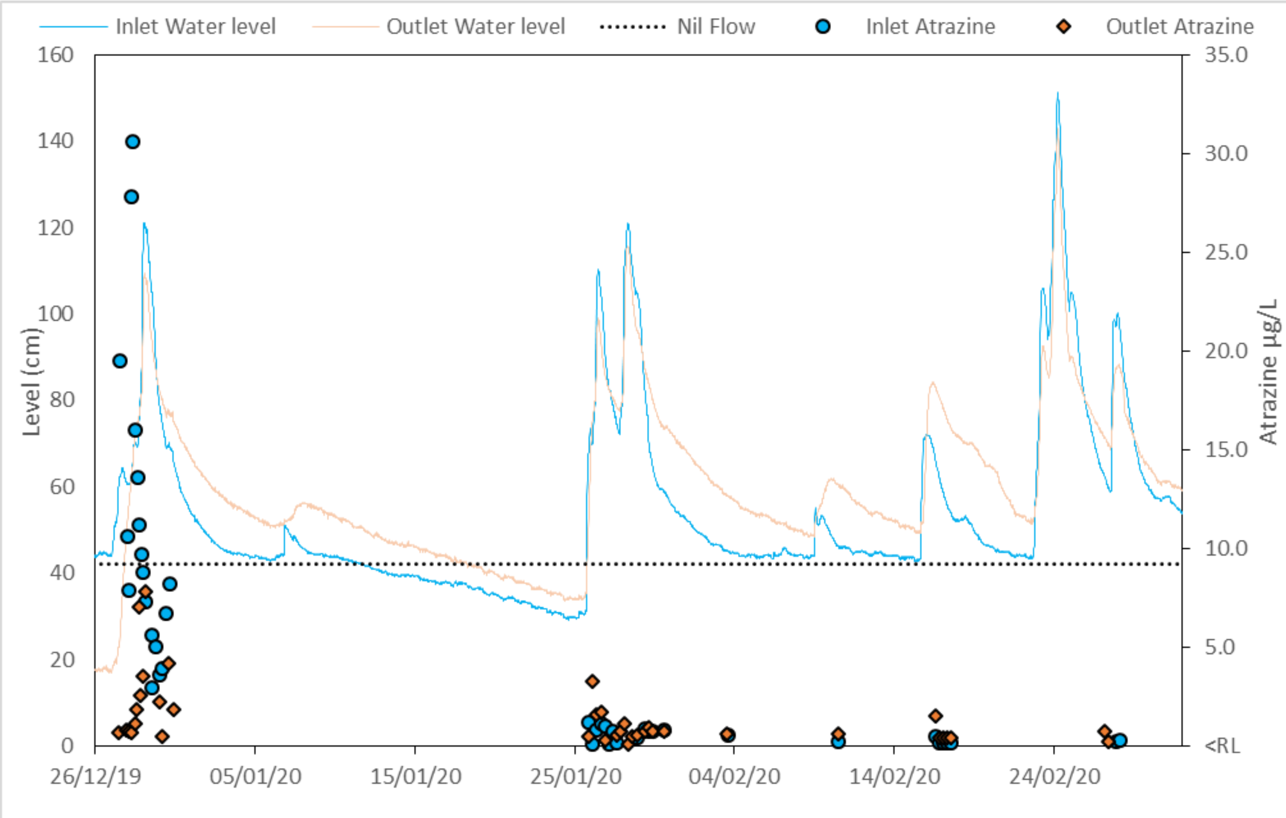


Figure 3.4.14 Temporal variations in imidacloprid concentrations during flow events

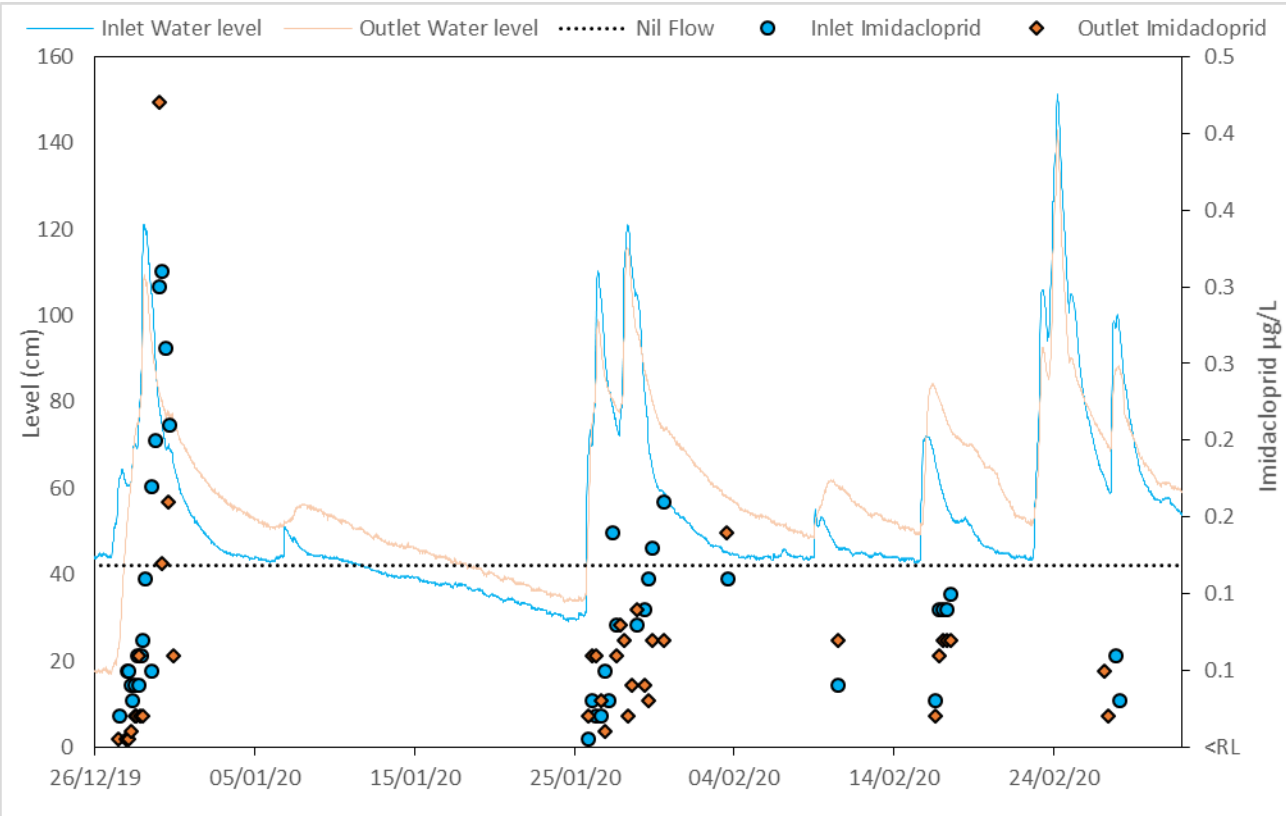


Figure 3.4.15 Temporal variations in diuron concentrations during flow events

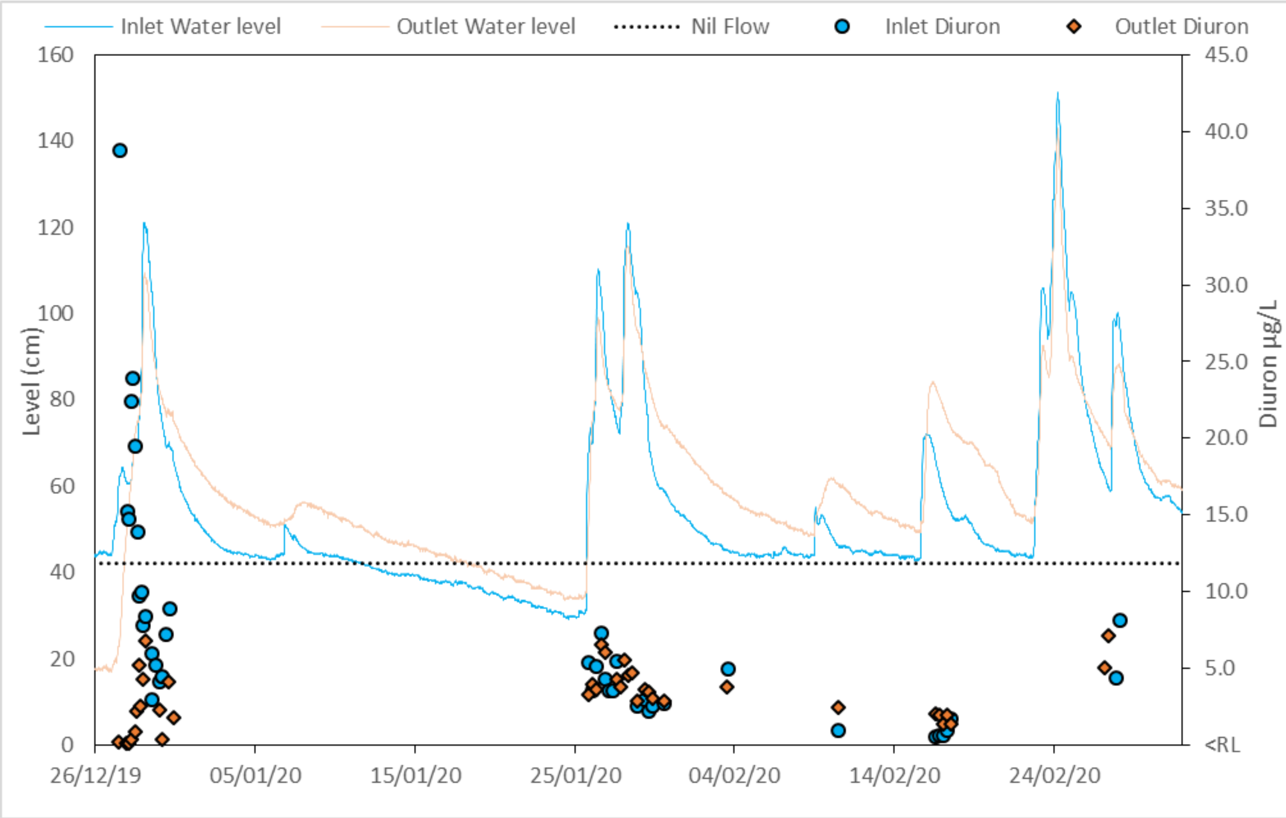


Figure 3.4.16 Temporal variations in hexazinone concentrations during flow events

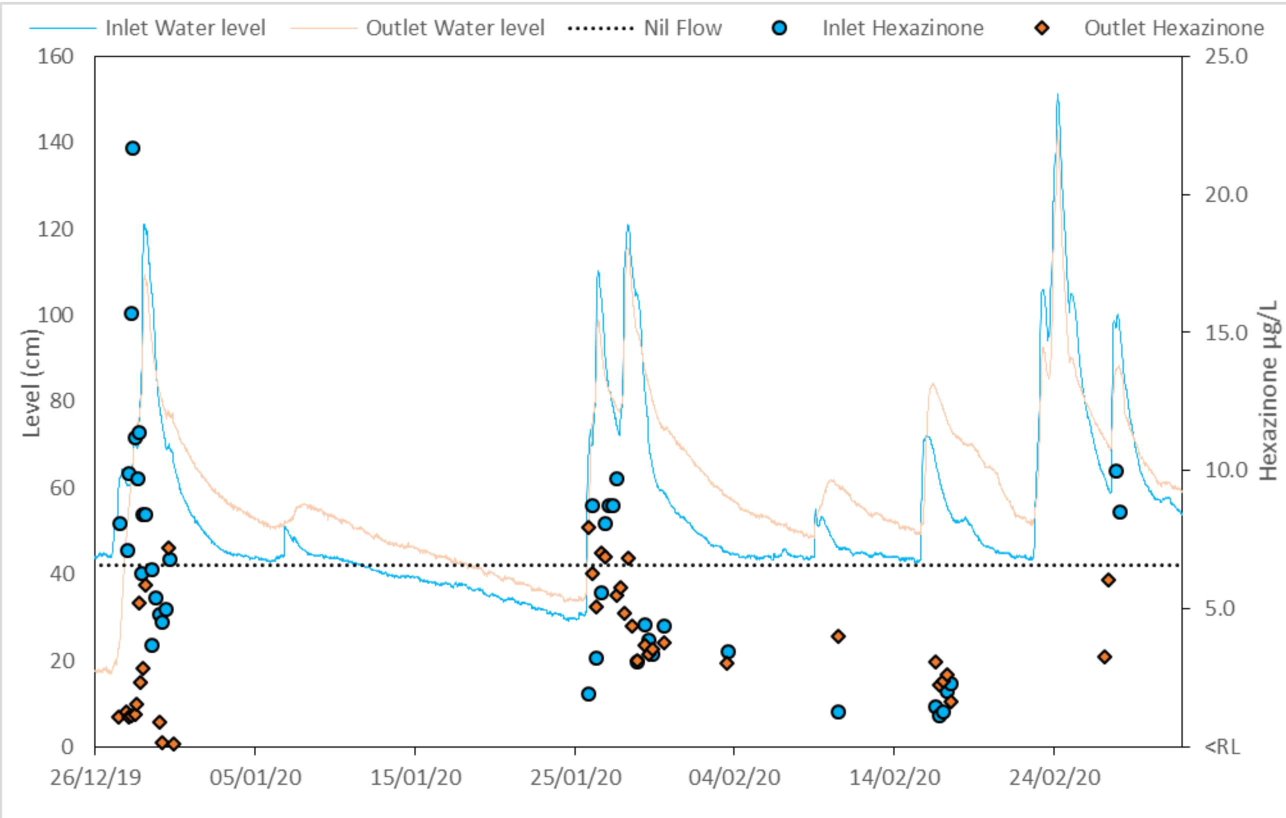


Figure 3.4.17 Temporal variations in metalochlor concentrations during flow events

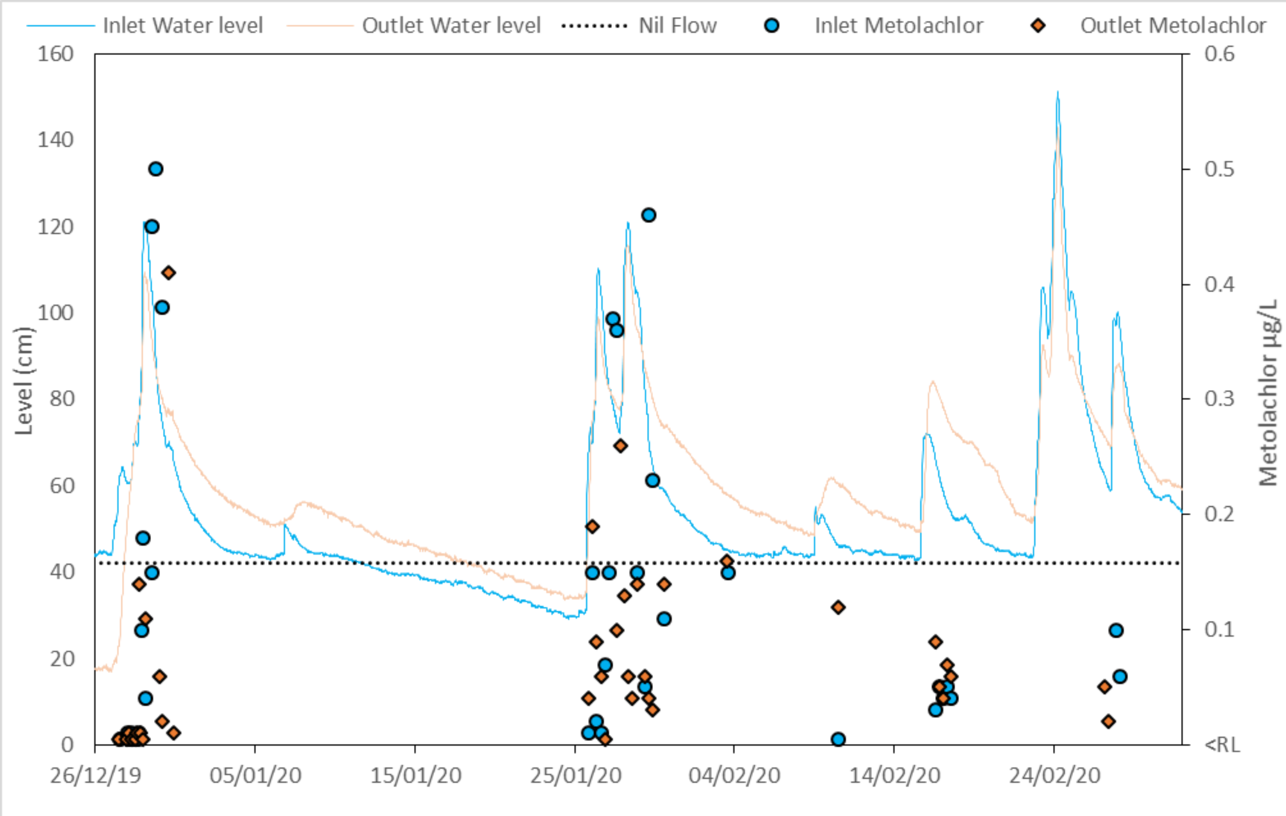


Figure 3.4.18 Temporal variations in metribuzin concentrations during flow events

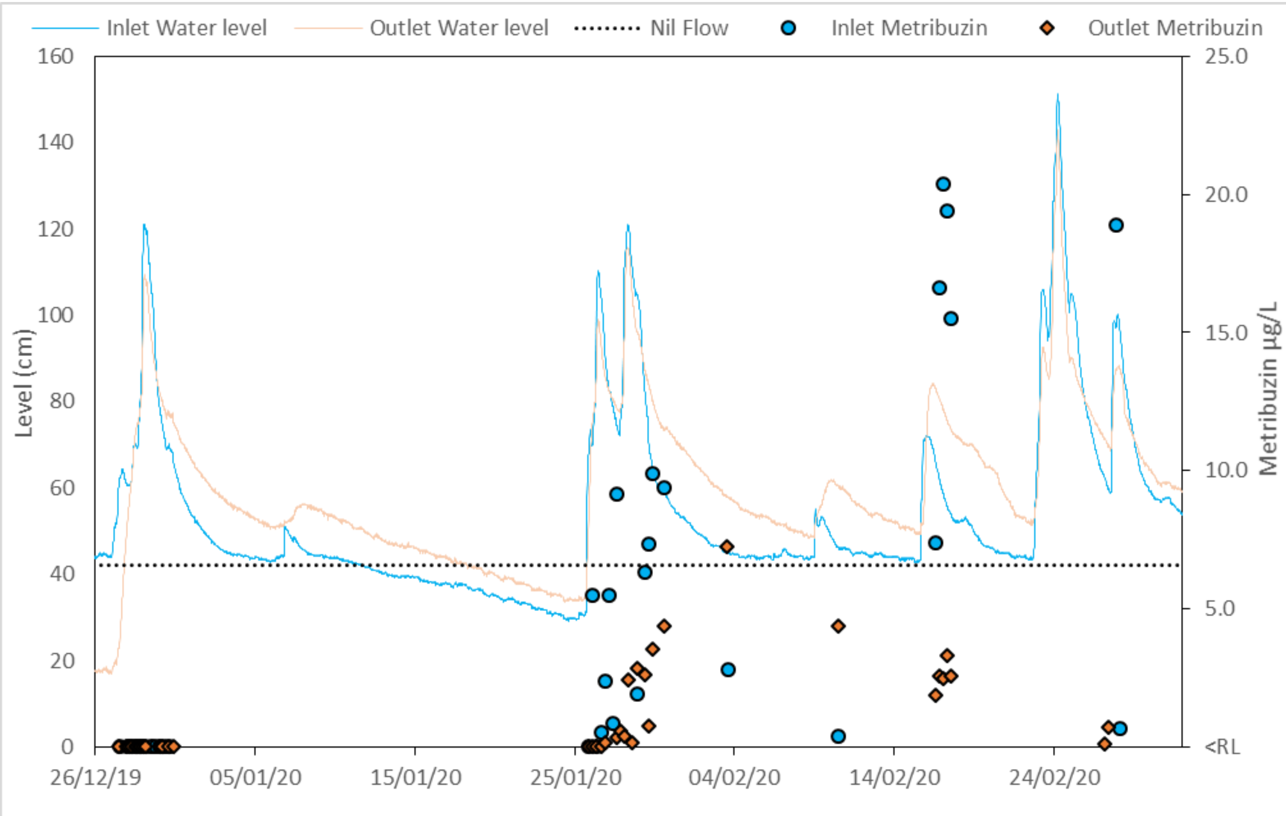


Figure 3.4.19 Temporal variations in propazine concentrations during flow events

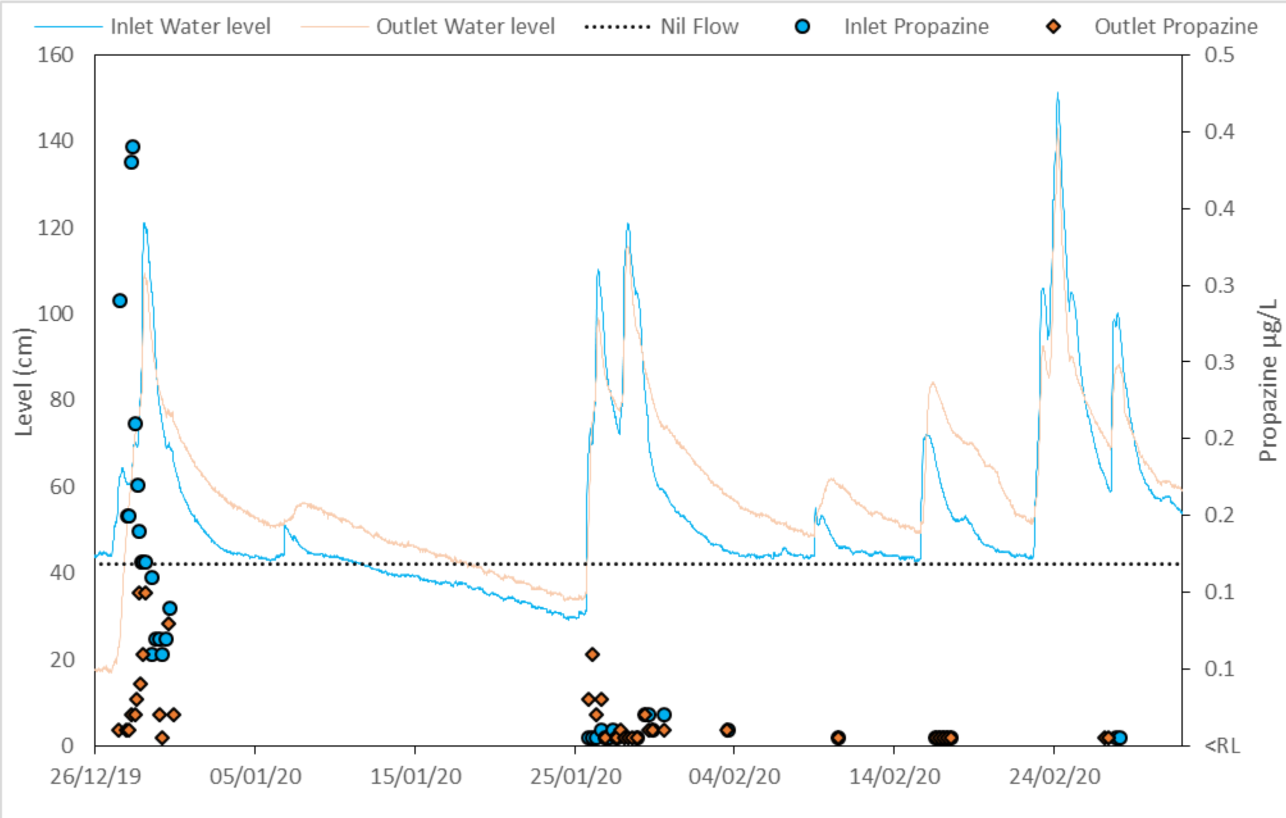


Figure 3.4.20 Temporal variations in simazine concentrations during flow events

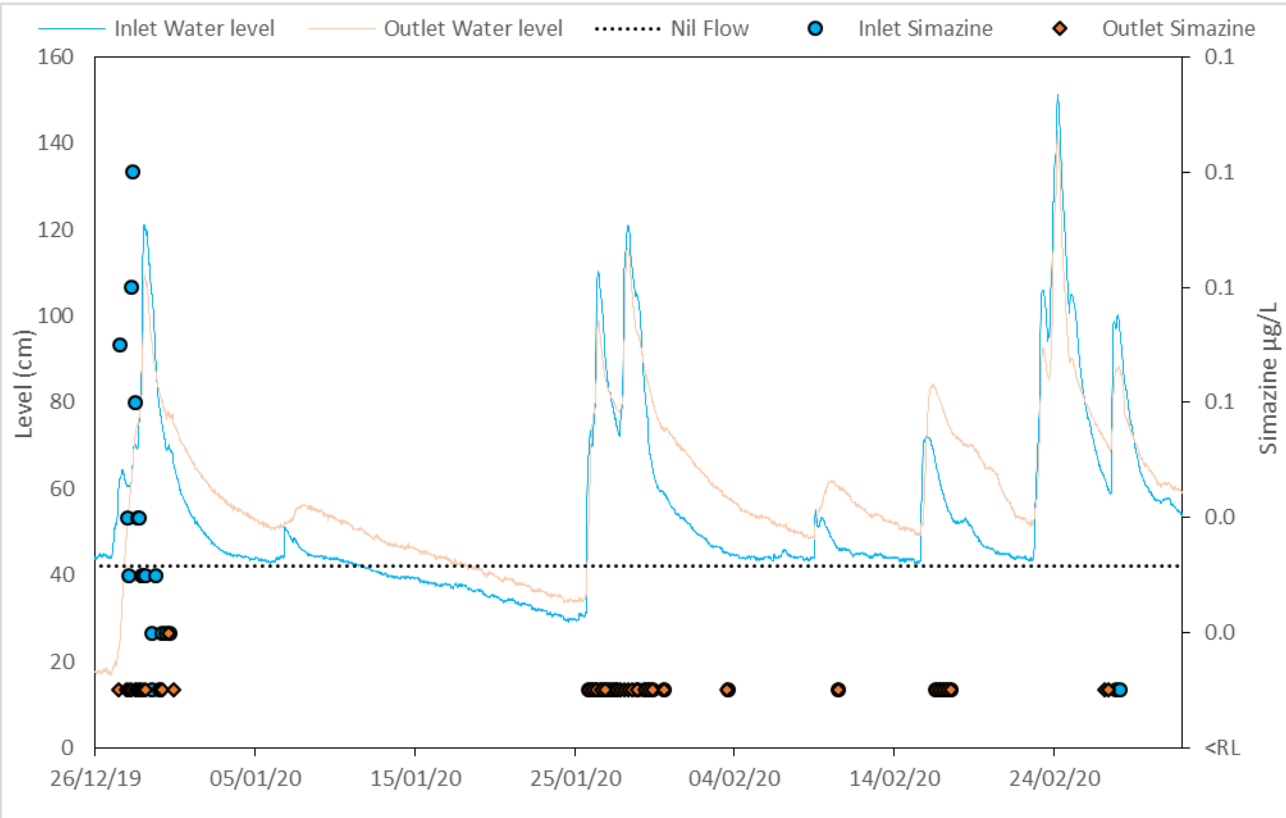


Figure 3.4.21 Temporal variations in DKN concentrations during flow events (not analysed in first event)

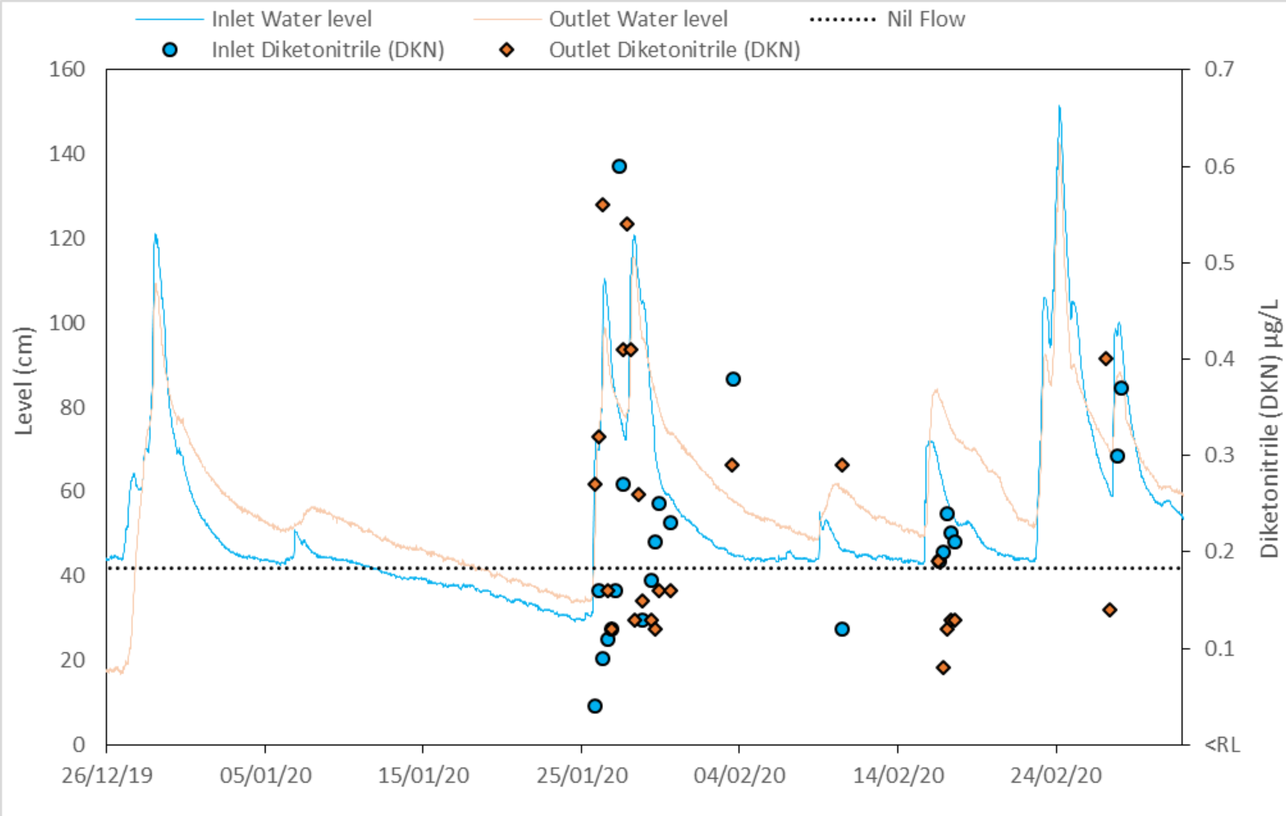
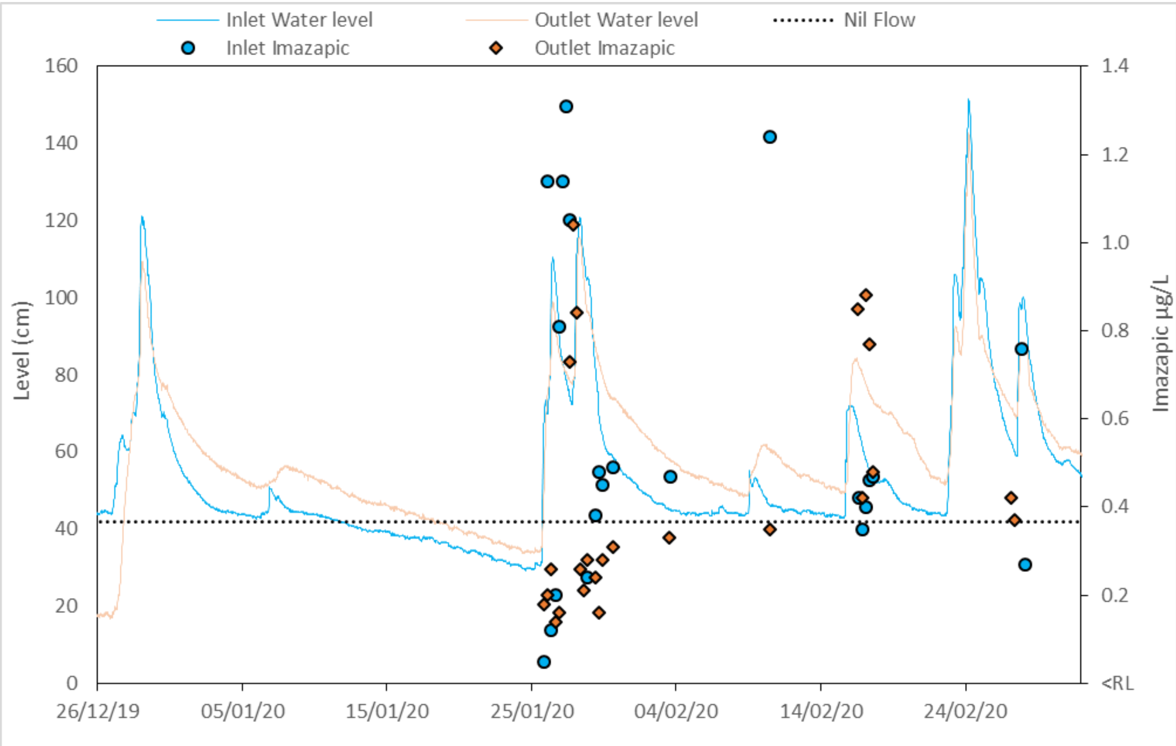


Figure 3.4.22 Temporal variations in imazapic concentrations during flow events (not analysed in first event)



3.4.2 Flow Absent

Table 3.4.3 indicates that a range of pesticide residues were detected in the wetland under still conditions, however, there was only one PGV exceedance (for Diuron).

Table 3.4.3 Pesticides that reported detections in the absence of flow.

Sites	Pesticide	Reporting Limit (µg/L)	Samples	Detections	Percent Detections	Freshwater: PGV for 95% protection (µg/L)	Freshwater: Percent Exceedances	Marine: PGV for 95% protection (µg/L)	Marine: PGV for 99% protection (µg/L)
Bakers Creek Sites (BC1 - BC7)	Atrazine	0.01	8	7	87.5	13	0.0	1.4	0.6
	Diuron	0.02	8	8	100.0	0.23	12.5	0.67	0.43
	Hexazinone	0.02	8	8	100.0	1.1	0.0	2.5	1.8
	Metolachlor	0.01	8	3	37.5	0.71	0.0	0.084	0.0002
	Metribuzin	0.02	8	5	62.5	2.6	0.0	2.7	2.0
	Ametryn	0.01	8	6	75.0	0.33	0.0	0.61	0.1
	Fluometuron	0.01	8	2	25.0	-	-	-	-
Sandy Ck and Sed Basin Sites (SC1 to SC7 and SB1 to SB7)	Atrazine	0.01	14	10	71.4	13	0.0	1.4	0.6
	Diuron	0.02	14	7	50.0	0.23	35.7	0.67	0.43
	Hexazinone	0.02	14	6	42.9	1.1	14.3	2.5	1.8
	Metolachlor	0.01	14	9	64.3	0.71	7.1	0.084	0.0002
	Metribuzin	0.02	14	2	14.3	2.6	0.0	2.7	2.0
	Ametryn	0.01	14	1	7.1	0.33	0.0	0.61	0.1
	Propazine	0.01	14	1	7.1	3.1	0.0	4.6	2.2
	Fluometuron	0.01	14	2	14.3	-	-	-	-
	Bromacil	0.02	14	3	21.4	3.6	0.0	1.1	0.23

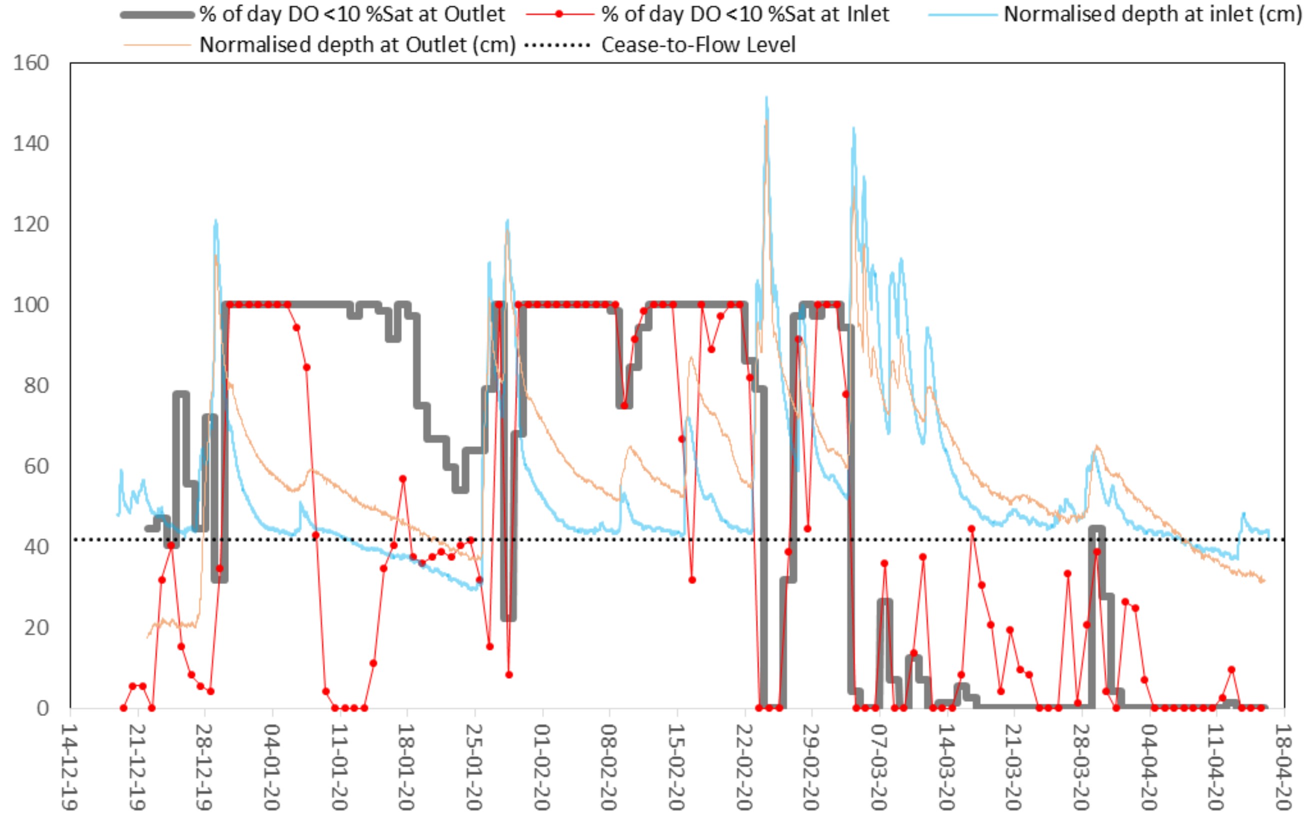
PGV Reliability Classification
Very High
High
Moderate
Low
Very Low

3.5 Dissolved Oxygen (DO) Availability for Aquatic Fauna

The very low oxygen levels required to support denitrification in the wetland system pose questions regarding the suitability of water treatment wetlands as habitat for aquatic biota. The 30% saturation threshold for denitrification is also the DO concentration below which many DO-dependent native fish species begin to experience acute stress (Butler and Burrows 2007). The data presented previously in Figure 3.2.3a which showed that DO levels were below the threshold value for prolonged periods during February and March therefore suggest that conditions in the wetland would have been unsuitable for such species.

Available data (Butler and Burrows 2007) indicate that at DO levels below 10% saturation, sensitive fish species risk acute asphyxiation, and most native fish (other than a few specially adapted species) and some macroinvertebrates experience acute stress. Accordingly, the data presented in Figure 3.5.1 suggest that conditions in the wetland during January and February were entirely unsuitable as habitat for most fish species and some sensitive invertebrate species.

Figure 3.5.1 Temporal variations in the percentage of time each day that DO levels were below 10 % Saturation (below which most native fish species begin to experience stress)



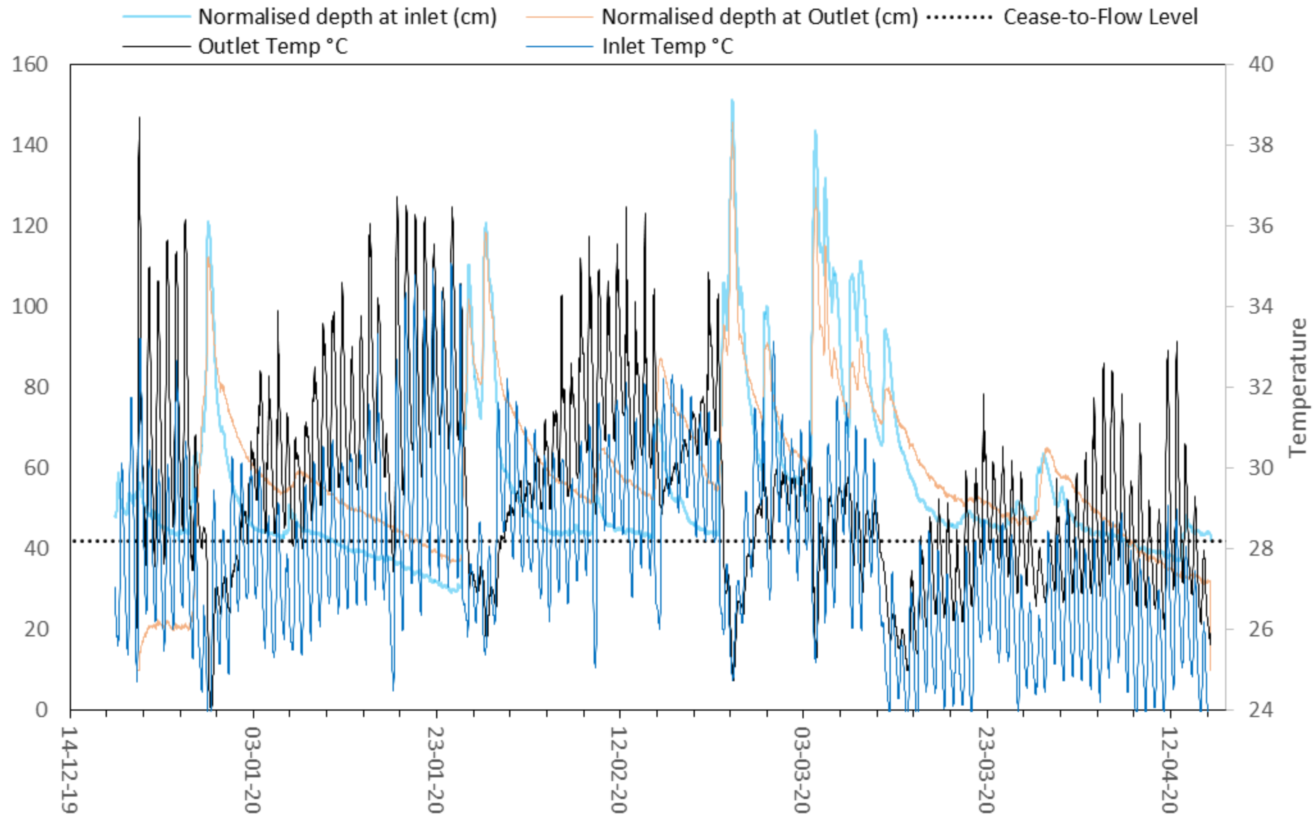
Most local fish can contend with brief exposure to low DO levels by rising to the surface to perform ASR (aquatic surface respiration, which involves breathing in the very thin surface water layer in direct contact with the air). This can generally be done relatively safely if the low DO levels are confined to night-time hours (as is the case when natural daily cycling is present); however, in order to perform ASR in daylight hours they must expose themselves to substantial hazards (e.g., high temperatures, sunburn, exposure, predation risk and inability to feed). Accordingly it is not uncommon to find that fish inhabiting severely hypoxic waters ultimately die from exposure or hyperthermia rather than asphyxiation. As can be seen in Figure 3.2.3 fish inhabiting the Bakers Creek wetland would have been forced to the surface most of the time during the 2 month period following the first flush thus exposing them to mortal risk.

The potential risks presented by hypoxic water conditions are exacerbated in this region by high temperatures and strong sunlight. The mean water temperature 10 cm above the bottom at BC7 was 28 ^{0}C and as can be seen in Figure 3.5.2 values ranged up to 38ºC. Depth profiling data are unavailable for these sites but based on experiences with other regional wetlands (Waltham and Fixler 2017, Waltham and Schaffer 2018, Wallace et al. 2020a) daily maximum surface water temperatures would almost certainly have been significantly higher (by at least 2 to 5 ^{0}C). Accordingly, it is likely that most fish species would have suffered some thermal stress (Wallace et al. 2015, Wallace et al. 2017a) if forced to the surface. Most fish would also have sustained unusually high metabolic rates thus exacerbating the situation by increasing their oxygen requirements.

Hypoxia-sensitive fish species on the Herbert floodplain have been observed (unpublished personal observation) actively feeding by swimming in and out of anoxic plumes discharging from cane drains (presumably the piscatorial equivalent of humans diving under water to collect food), so the possibility of fish entering the wetland at times when DO levels were low cannot be completely disregarded; however, it seems likely that persistently low DO levels at the outlet would generally serve as a chemical fish passage barrier for most unspecialised fish species. If so, that would mean that any such fish that enter the wetland during the brief periods when DO levels are adequate (generally during events) would be prevented from escaping once

DO concentrations have fallen back down to threatening levels. It would therefore be advisable to consider installing physical passage barriers to avoid that contingency.

Figure 3.5.2 Water temperature variations over the 2020 wet season



4 CONCLUSIONS AND RECOMMENDATIONS

Bakers Creek Wetland Treatment System

- The available data provide some qualitative indications that the Baker's Ck wetland may be capable of effecting a reduction in the quantities of pesticides being released into the receiving environment. There are currently insufficient data to be able to confirm that finding or to quantify the magnitude of the effects. However, subject to the proviso outlined in the next dot point, the initial indications are sufficiently favourable to justify further investment to refine monitoring techniques and attend to a few existing technical sampling design issues that have been raised in this report, in order to be able to carry out a quantitative assessment of contaminant import and export rates and loads, which is the only basis upon which the wetland's performance can be accurately gauged.
- Logging records indicate that, except for very brief periods on the peak of flow events, EC levels at the wetland inlet are 30 to 60 % higher than at the outlet. This suggests that a significant proportion of the water emerging from the wetland originated from, an as yet unidentified and unmonitored, source with lower EC concentrations than the inlet drain. Before proceeding with any further monitoring it would be highly advisable to carry out a one-off investigation to confirm that the EC records are correct, and if so, ascertain the source of the additional water and determine the feasibility of including that site in future monitoring programs. If it proves infeasible to monitor the source (for example if it is groundwater inflow or diffuse overland flow), any further attempts at performance monitoring would be largely futile. If the identified source proves to be one or more surface input points along the length of the wetland, the feasibility of being able to account for their effects would need to be evaluated carefully before any commitment to intensive routine monitoring could be justified.
- The suspended sediment and nutrient data that have been collected to date do not provide an adequate basis for reaching any conclusions regarding the performance of the wetland. Current indications are that, due to the flashy nature of the hydrographs in this system and the highly transient nature of associated contaminant pulses entering the wetland, the sampling frequencies employed for event monitoring would need to be increased substantially in order to obtain more representative samples. It is also evident that high frequency sampling would need to be continued for the duration of the falling limb of events in order to account for the wetland's detention capacity and ensure that export load is properly quantified.
- It is clear that contaminant concentration data alone, even when supported by water level data (which provide qualitative indications of hydrographic variations), do not provide an adequate basis for quantitatively assessing wetland water treatment efficiency, because the potential significance of each concentration value can differ by more than three orders of magnitude depending on the discharge rate at the time. Quantitative assessments require accurate flow gauging data in order to be able to calculate discharge loads and flux rates. Moreover, in order to be able to ensure that monitoring activities are correctly timed it is necessary to have some knowledge of the wetland's water residence time under different flow conditions because if it is detaining water long enough to perform the desired water treatment function there will never be any direct real-time correspondence between the contaminant concentrations at the inlet and outlet.
- The monitoring methods employed to date have evolved over the course of the project and entailed pilot grab sampling during 2017, manual (low frequency) event sampling in 2018, and automated (medium intensity) event sampling supported by depth and EC logging in 2020. If future performance monitoring is to be attempted, the following additional improvements and refinements would need to be implemented in order to be effective:
    - Installation of accurate flow gauges at all inlets and at the outlet is essential. This could entail construction of flow control devices and depth loggers, and the development of stage-discharge curves so that flow rates can be calculated from water level measurements, or the use of A-V (area-velocity) sensors which yield direct flow measurements (provided that the cross-sectional area of the drain has been accurately determined).

   - Employ the highest sampling rates that are logistically feasible, noting that most samplers can be programmed to take multiple subsamples thus yielding composite samples representative of the period over which each sample bottle is filled.
   - Continue autosampling through the entire event hydrograph, including the tail.
   - Consider installing turbidity sensors in order to obtain a continuous record of particulate contaminant fluctuations. By comparing turbidity data to the trends exhibited by lab SS results it is usually possible to ascertain whether the lab samples are accurately representative (and adjust the autosampling program accordingly if necessary). Turbidity logging records can provide a basis for selecting samples for lab analysis if the number of samples collected exceeds the allocated analytical budget.
   - Include pH and EC in the laboratory analysis suite for many, if not all, samples and routinely conduct field pH measurements whenever conducting manual sampling. (pH values are necessary to confirm the suitability of the water for release to the receiving environment and as an aid for assessing the potential toxicity of parameters such as ammonia. Lab EC values are useful for validating the EC logging data).
   - Determine if there is an alternative analytical method or service provider that can be used to avoid the requirement to dilute pesticide samples in cases where one or two pesticides are present at high concentrations.
   - Periodic evaluations of the health and limnology of the wetland is also recommended for consideration in all wetland projects. This is not required for assessing performance per se, but it is required in order to determine the potential source of performance failures and to gain an understanding of what types of biological communities, conditions and processes yield the best outcomes.
- Conditions in the wetland were severely hypoxic for the first two months of the 2020 wet season. This is not necessarily an unfavourable outcome from a water treatment perspective as it allows denitrification to occur. However, it does mean that the waters contained insufficient oxygen to support most local fish species other than a few low DO specialists such as tarpon and eels, and that the conditions would likely have provided competitive advantage to hypoxia-tolerant noxious exotics such as Gambusia and Tilapia. This linked with high water temperatures, occasional occurrences of potentially toxic ammonia concentrations, frequent detections of 18 different herbicides, some which occurred at levels well above ecosystem protection guidelines, suggests that this wetland is not desirable fish habitat, at least during the first few months of the wet season. There are therefore grounds to suggest that it might be beneficial to install fish exclusion devices.
- After the third wet season flush in 2020, pesticide levels appeared to have generally declined and DO levels began exhibiting the kinds of daily cycling that are typical of natural wetlands suggesting that habitat conditions towards the end of wet were more conducive to aquatic fauna; however, it is unlikely that favourable conditions would have been sustained (for example in previous years the wetland has experienced algal blooms at various stages during the year.

Other sites Monitored in this Study

- Bakers Creek wetland is the only study site where *in situ* dataloggers were deployed during the course of this project. Accordingly there are no water level or EC records available to contextualise the water sampling results obtained from the Sandy Creek TT, Sediment Basin or cane drain study sites. This makes it impossible to confidently assess the potential significance of the data obtained from these sites. Moreover, water sampling at these sites was conducted manually and was far less intensive than the autosampling campaigns which were carried out at the Bakers Creek wetland. It is therefore considered highly unlikely that the available water quality data accurately represent the hydrographic variations that occurred at these other sites over the course of the project. Strictly speaking that invalidates the use of comparisons between inlet and outlet concentrations as a basis for assessing contaminant removal and the only scientifically defensible conclusion that can be drawn is that the data are inadequate to allow meaningful assessment.

- The above comments notwithstanding, if the water quality data collected at these sites during periods of flow were to be taken at face value the following tentative conclusions could be drawn:
    - The Sandy Creek TT data comprise just nine samples collected from each of 3 sampling points – inlet (SC1), Middle (SC5) and outlet (SC7). The results indicate that concentrations of SS and total N, and to a lesser extent oxidised N (NOx), at the outlet were slightly lower than the inlet, while total and reactive P concentrations at the outlet were significantly lower than the inlet. This implies that there may have been some minor removal of particulate N and NOx, and significant removal of phosphorus. However, (even if the above-mentioned comments questioning the representativeness are ignored) these conclusions carry very little statistical weight due to the small number of available data points. Notably, the highest SS concentrations were actually reported at SC5, suggesting the possibility that there may have been plugs of turbid water passing through the system that were not detected at the inlet or outlet. There were no potential indications of any pesticide removal.
    - The sediment basin data comprised only seven samples collected from the inlet (SB1) and outlet (SB7). This very limited, and almost certainly unrepresentative dataset provided no indication of any potentially significant contaminant removal.
    - The Cane Drain dataset comprised 23 samples collected from the inlet (CD1) and outlet (CD7). The differences between these two sampling points were only minor and not statistically significant. Nonetheless concentrations of SS, total N, total P, NOx and ammonia were actually slightly higher at the outlet than they were at the inlet; thus supporting the conclusion that there was no evidence of contaminant removal.

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Supplementary

Hydrology and denitrification in the Bakers Creek water treatment wetland system

Jim Wallace

For inclusion in Report No. 20/01

February 2021

EXECUTIVE SUMMARY

This study of a multi-component 2.5 ha constructed wetland system near Mackay, north Queensland was established to try and determine the efficacy of these types of wetlands for improving water quality entering the Great Barrier Reef lagoon. To do this a water balance model was constructed using water depth measurements made in the wetland inlet basin, between 25 December 2018 and 30 September 2020. The water balance model allowed daily values of run-in and drainage to be calculated, which were combined with nitrogen concentration estimates to derive a value for the filtering of dissolved (*DN*) and particulate nitrogen (*PN*) of 107 kg ha^{-1} year^{-1}, or 48% of the nitrogen input to the wetland.

The largest losses of nitrogen where via drainage which accounted for 52% of the total input to the wetland system. Most of the drainage from the wetland occurred as outflow ,77%, with losses to groundwater accounting for the remaining 23%. This means that 83 kg ha^{-1} year^{-1} of *DN* flowed out of the wetland and 24 kg ha^{-1} year^{-1} entered the groundwater. Outflow also carries sediment and *PN* out of the wetland, with 35% leaving the wetland, equivalent to a *PN* loss of 10 kg ha^{-1} year^{-1}.

The next largest loss of nitrogen was by denitrification of *DN* at 88 kg ha^{-1} year^{-1}. This value is dependent on the rate of denitrification used (971 µg L^{-1} d^{-1}), but substitution of higher and lower rates from other studies of wetlands in north Queensland showed that although denitrification loss may change by 8 to 15%, the overall filtering percentage of *DN* and *PN* only changes by between 3 and 6 %. This is because denitrification is less than half of the nitrogen loss from the wetland.

The filtering of nitrogen in Bakers Creek can be compared to two other studies of wetlands in north Queensland. The first is another constructed wetland near Babinda where the filtering of *DN* and *PN* was higher at 56% (Wallace et al., 2020). This wetland is in a very high rainfall area (annual rainfall 4287 mm) and it was frequently filled by deep floods. Once these receded, water was held in the wetland for long periods, as it was operated as a closed system (no outflow), allowing time for denitrification. In contrast, the Bakers Creek wetland is an open system where water can flow through. However, annual rainfall is much lower (1585 mm), leading to more moderate high flows and long periods when water is below the cease to flow level. This, combined with the permanent presence of water in its biodiversity pond, again means that there is sufficient time for significant denitrification to occur.

Another example of an open flow through wetland is Kyambul lagoon in the Tully catchment (McJannet et al 2012 a,b). This natural riverine wetland filtered little or no nitrogen or sediment over the long term. This was a consequence of the very large fluxes of water and nutrients that entered and left the wetland very quickly during large floods in the wet season. With the very short residence time of water in this wetland there was

little/no time for denitrification or sedimentation. Clearly, the filtering capacity of a wetland is highly dependent on its hydrology, with open wetlands only able to filter significant amounts of sediment and nutrients where rainfall is comparatively low and water residence times are high.

The filtering in Bakers Creek is greatly enhanced by its ability to retain water for long periods. This mainly occurs in the final component of this wetland, the biodiversity pond. This is the largest and deepest part of this wetland system which always contained water, even during the dry season. Of the total *DN* removed by denitrification in Bakers Creek, 89% occurred in the biodiversity pond. This illustrates the value of large, permanent deep pools in a wetland system, whether they are natural or constructed.

We estimate that the Bakers Creek constructed wetland system can remove just under half of the nitrogen that enters it and ~ 65% of its sediment input. These types of wetlands should therefore be able to improve the quality of water entering the Great Barrier Reef lagoon when constructed downstream of areas generating sediments and nutrients. However, to do this they need to retain water for significant amounts of time, which they may do in comparatively low rainfall areas. In higher rainfall areas open, flow through wetlands may not be able to do this, so constructed wetlands need to be able to retain water in these regions in order to have significant filtering capacity.

1 Wetland bathymetry, volume and area

Prior to the construction of the Bakers Creek wetland system, the site topography was surveyed (AWC 2015). Four of the 11 transects made across the main components of the wetland are shown in Figure 1.

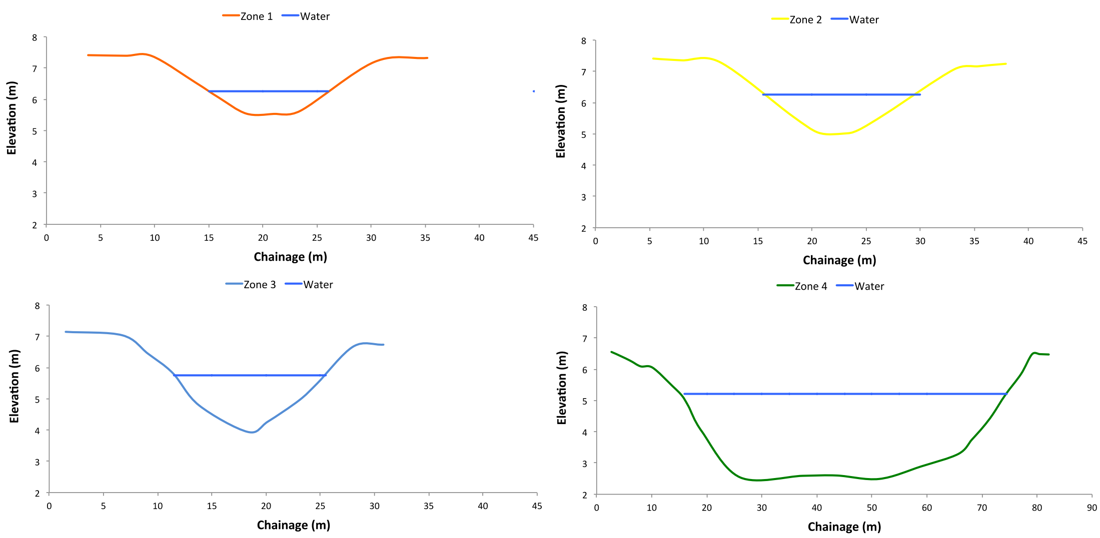

Figure 1. Ground level surveys across the Bakers Creek wetland system (a) Zone 1; sediment basin, (b) Zone 2; macrophyte zone, (c) Zone 3; irrigation pond and (d) Zone 4; biodiversity pond. The depth of water to the overflows in each zone is also shown.

The sediment basin is about 25 m wide with a flat bottomed V shape. It has an overflow at 6.25 m and when full to this point has maximum depths ranging from 0.7 to 1.0 m. The next, macrophyte zone, has a similar shape, but is slightly deeper at 1.1 to 1.3 m when full to the same overflow height. The irrigation pond (zone 3) is significantly deeper at 1.8 to 2.5 m when full to its lower overflow (at 5.75 m). The biggest and deepest zone 4, the biodiversity pond, is ~ 70 m wide with depths from 2.1 to 2.7 m when full to it lower overflow of 5.2 m.

These transect data can be used to calculate how the area of the wetland, and the volume of water within it change with wetland depth (Figure 2).

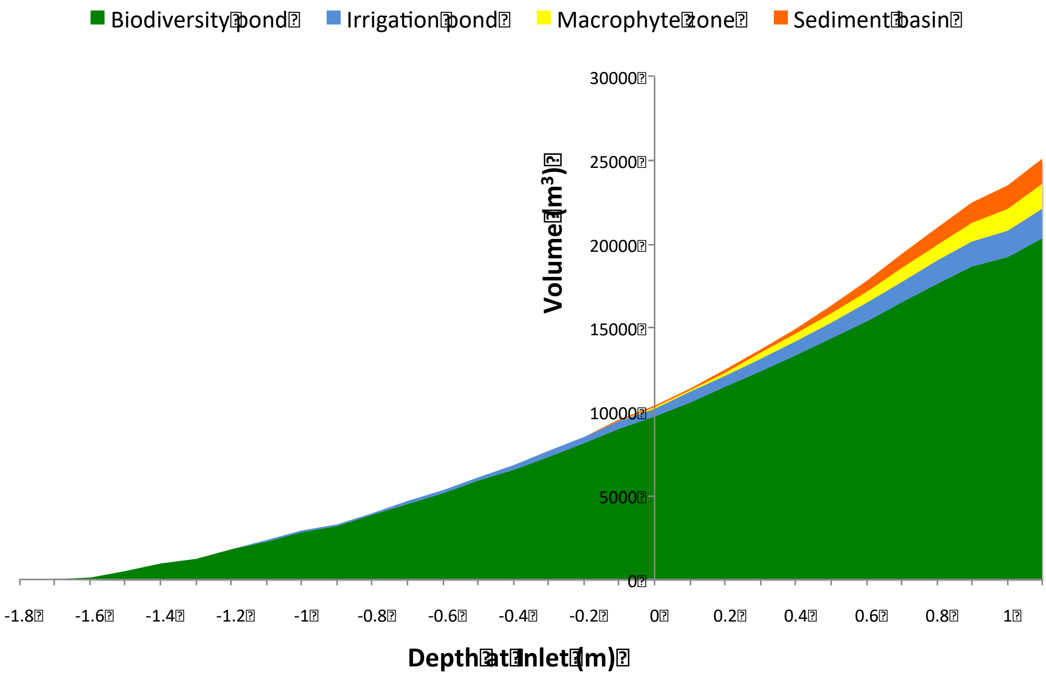

Figure 2. Change in water volume with depth in the inlet sediment basin in Zone 1 (green), Zone 2 (blue), Zone 3 (yellow) and zone 4 (orange).

The maximum volume of water in the entire wetland system is 24254 m^3, which occurs when it is full to the overflows in each of the wetland components. At this stage ~ 81% of the water is in the biodiversity pond, the remainder being held in the other 3 components in approximately equal amounts. As the depth in the wetland decreases, so does the water volume and when *d* = 0 at the inlet in the sediment basin, there is no water left in this component and only small amounts in the macrophyte zone (1%) and irrigation pond (5%). As depth decreases further in Zone 2 to 4 the vast majority of the water is held in the biodiversity pond, which doesn't empty until the depth is 1.8 m below the reference ground level at the inlet.

The changes in the total wetland water volume and area with depth are shown in Figure 3. The maximum area of water is 19399 m^2, which again occurs when it is full to the overflows in each of the wetland components. At this stage 71% of the wetland water area is in the biodiversity pond, with about 10% in each of the other 3 components. As the depth decreases, the percentage of wetland area in the biodiversity pond increases exceeding 83% when the inlet depth drops below zero.

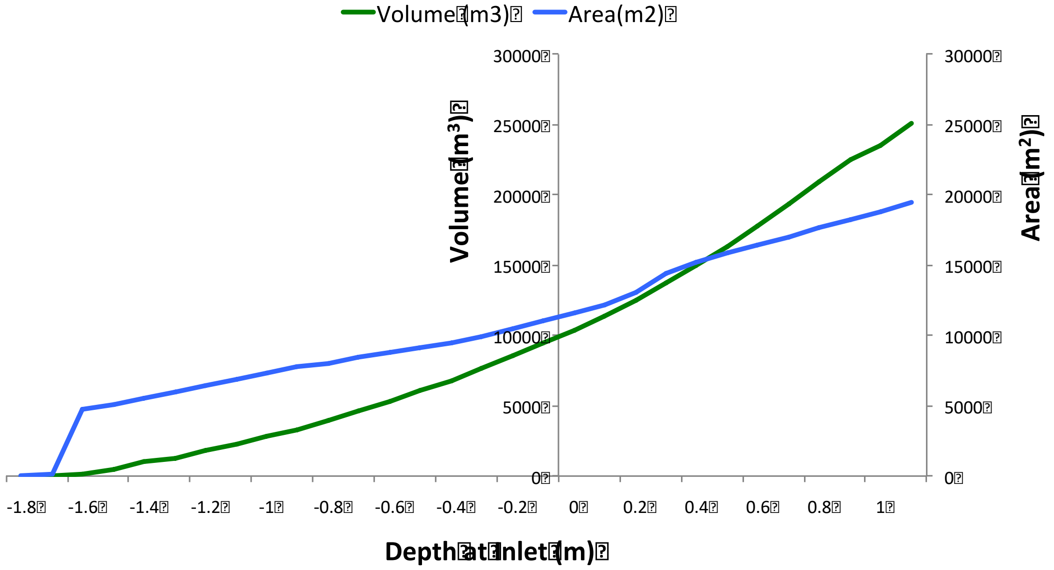

Figure 3. Change in total water volume (green) and area (blue) with depth in the inlet sediment basin.

The above relationships between wetland depth, area and volume are used in the calculation of wetland nitrogen concentration and the different nitrogen loss rates which are expressed per unit volume (dissolved nitrogen) and area (nitrogen in drainage); see Section 2.2.

2 METHODOLOGY

2.1 Water balance

A schematic diagram of the key components of the Bakers Creek wetland system is shown in Figure 4.

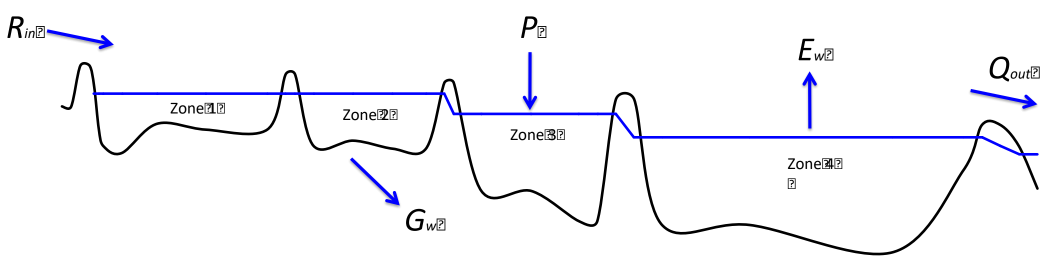

Figure 4. Schematic diagram of the Bakers Creek wetland system showing its four main components; Zone 1, the sediment basin, Zone 2, the macrophyte zone, Zone 3, the irrigation pond and Zone 4, the biodiversity pond. The water balance symbols are explained in the text.

The change in depth (δd) of the wetland system on any day is given by the difference between water entering and leaving it. This can be expressed as:

$$\delta d = (P + R_{in}) - (DR_w + E_w), \quad (1)$$

where P is the rainfall directly entering the wetland, R_{in} is the water which flows into the wetland from its surrounding catchment, and DR_w is the total drainage from the wetland. E_w is rate of evaporation from the open water in the wetland using the method described below.

2.1.1 Evaporation

The rate of evaporation from the wetland (E_w) was estimated using the energy balance model described by McJannet et al. 2008 and Wallace et al. 2015. The main input of energy to the model is solar radiation and the main losses are via heat conduction to the atmosphere and evaporation. The model requires daily weather data, which were obtained for the location from the Scientific Information for Land Owners (SILO) database (http://www.nrw.qld.gov.au/silo/). The SILO database consists of interpolated meteorological variables on a 0.05° (5 km) grid for the whole of Australia (Jeffrey et al. 2001). The SILO variables used in the evaporation model are air temperature, vapour pressure, solar radiation and rainfall and the way these variables are used to calculate daily evaporation are described by McJannet et al. (2008) and Wallace et al. (2015).

2.1.2 Wetland drainage

Drainage (DR_w) from the wetland was calculated from the daily decrease in wetland depth recorded in the rain free periods following rain events. As water also evaporates from the wetland during this time, daily drainage estimates are given by the change in water level (δd) minus the evaporation (E_w) on each day (assuming there is also no run-in R_{in} to the wetland; see equation (1)). Drainage (DR_w) occurs as both outflow (Q_{out}) from the wetland system outlet and groundwater seepage (G_w) along the entire wetland. We will see later in the results section how the relative contributions of these two components can be deduced.

Drainage from the wetland was separated into two phases; 1) 'rapid' drainage which occurred when the water level was above a threshold value of 26 cms and 2) 'slow' drainage when the water level was below 26 cms (see section 3.1.2).

2.1.3 Water run-in from the surrounding catchment

Estimates of the amount of water that flowed into the wetland from its surrounding catchment (R_{in}) during rainfall were made using a simple runoff coefficient model that assumes that R_{in} is a fixed fraction of rainfall, C (e.g. see Pilgrim and Cordery 1993).

$$R_{in} = C * P \quad (2)$$

As small amounts of rainfall do not generally produce runoff due to losses from interception and depression storage in the land surface (Critchley and Siegert, 1991), only events > 5 mm were used to calculate R_{in}. Above this threshold the value of R_{in} is also affected by the wetness of the surrounding catchment, with less runoff occurring when it is dry. To account for this in a simple way we calculated daily values of the soil moisture deficit (SMD_c) in the surrounding catchment as the difference between rainfall (P) and catchment evaporation (E_c). Unless the catchment is saturated, E_c will be less than the wetland evaporation, E_w, and mainly controlled by the soil moisture deficit and following Shuttleworth (1993) is given by:

$$E_c = K_c * E_w \quad (3)$$

where K_c is a 'crop coefficient' with a value between 0 and 1 depending on the soil moisture deficit (SMD_c). K_c = 1 when the soil is reasonably wet (SMD_c < 100 mm) after which K_c decreases linearly until it reaches zero at the maximum soil moisture deficit SMD_{max} of 200 mm; typical of a 2 m deep sandy loam soil (Burk and Dalgleish, 2013). Run-in to the wetland is then calculated as,

$$R_{in} = C * P \left(1 - \frac{SMD_c}{SMD_{max}}\right) \quad (4)$$

The value of the catchment runoff coefficient, C was then estimated by minimising the root mean square difference between the measured and modelled depths in each wet season.

2.2 Wetland sediment and nitrogen balance

2.2.1. Nitrogen balance model

The Bakers Creek wetland system was constructed to improve water quality due to its ability to filter sediment and nutrients that enter it. Here we estimate this filtering capacity by calculating the difference between the amounts of sediment and nutrient entering and leaving the wetland. To do this we use the water balance model described above (Figure 1) in combination estimates of particulate settlement and water denitrification within the wetland. This gives a nitrogen balance model for the wetland where the inputs of total nitrogen (TN_{in}) to the wetland can be compared with the total losses of nitrogen (TN_{out}) over time. Daily inputs of total nitrogen (kg) from run-in (R_{in}) and rainfall (P) are given by:

$$TN_{in} = PN_{in} + DN_{in} \quad (5)$$

where PN_{in} and DN_{in} are inputs of particulate nitrogen and dissolved nitrogen respectively. Each of these is given below as:

$$PN_{in} = PN_c^{in} * R_{in} * A_w \tag{6}$$

where PN_c^{in} is the concentration of particulate nitrogen in the water flowing into the wetland. The water balance model gives R_{in} as a depth and this is converted to a volume of water using the area of the wetland on each day, A_w. Dissolved nitrogen inputs are given by:

$$DN_{in} = DN_c^{in} * R_{in} * A_w + DN_c^r * P * A_w \tag{7}$$

where DN_c^{in} is the concentration of dissolved nitrogen in the water flowing into the wetland. The addition of dissolved nitrogen to the wetland from the direct input of rainfall (*P*) was made by assuming an average rainfall concentration, TN_c^r, of 500 µg L^{-1}. This figure was taken from the review of the nitrogen composition of precipitation by Eriksson (1952) who reported values of 410 and 630 µg L^{-1} for rainfall samples in Queensland, Australia. As the nitrogen input in rainfall is very small, ~ 2.5%, it is not necessary to have a more accurate value of TN_c^r.

Nitrogen losses from the wetland water can occur via three main processes, sedimentation of particulate nitrogen (*PN*), denitrification of dissolved nitrogen (*DN*) in the water column and losses of both *PN* and *DN* in water draining from the wetland. Some of the *PN* may be in the form of organic detritus particles which are not necessarily attached to inorganic particles. Furthermore, significant amounts of the *DN* may be in the form of dissolved organic nitrogen (*DON*), which will only denitrify when it is converted to nitrate. These latter two points are discussed in more detail in Section 4. In the current nutrient model total daily losses of nitrogen (kg) from the wetland are given by:

$$TN_{out} = (PN_c - PN^{rate}) * V_w + (DN_c - DN^{rate}) * V_w + (PN_c + DN_c) * DR_w * A_w \tag{8}$$

where PN_c and DN_c are the concentrations of particulate nitrogen and dissolved nitrogen respectively and their daily loss rates are PN^{rate} and DN^{rate}. Both loss rates are effective daily rates calculated from measured changes in *TN* and estimates of *PN* made using total suspended sediment (*TSS*) measurements (see Appendix 1 a&b). Nitrogen concentrations are converted to a mass of nitrogen (kg) using the volume of the wetland on each day, V_w. The water balance model gives the daily drainage DR_w as a depth and this is again converted to a volume of water using the area of the wetland on each day, A_w .

The derivation of the values of the nitrogen concentrations and their daily loss rates in Equations (5) and (6) are described in more detail below.

2.2.2. Particulate nitrogen

When water enters the wetland the suspended sediment and particulate nitrogen (*PN*) adhered to it will settle to the bottom of the water column over time. The rate at which this occurs depends mainly on the particle size distribution of the sediment. For the estimation of sediment and *PN* losses in the Bakers Creek wetland we have used values of total suspended sediment (*TSS*) measured in the inlet basin after a series (four) of high flow events that entered it in 2017, 2019 and 2020. Details of how the rate of sedimentation was estimated from these data are given in Appendix 1a.

To calculate the loss of nitrogen due to sedimentation it is also necessary to know the PN_c concentration of the water entering the wetland. Measurements of total nitrogen concentration (*DN* and *PN*) were also made for the above four flow events where the average peak concentration was 5060 µg L^{-1} (Appendix 1b). *PN* was not measured separately in this study, so we have used measurements of *TSS* to estimate the mean percentage of *PN* as 13% (Appendix 1b). Using this figure here gives a peak *PN* concentration of 662 µg L^{-1} and we have used this (as PN_c^{in}) to calculate the amount of *PN* that entered the wetland during all inflow events (see Equation 6).

Loss of nitrogen due to sedimentation was then calculated from the relative *TSS* settlement rate given in Appendix (1a) and the peak *PN* concentration (662 µg L^{-1}). This gave a daily average *PN* loss rate of 154 µg L^{-1} d^{-1} (PN^{rate} in Equation 8), which was applied when the *PN* concentration in the wetland was above zero.

2.2.3. Dissolved nitrogen

Another potential pathway for nitrogen loss from the wetland is through gaseous nitrogen (N_2) loss from the water through the process of denitrification. Although many factors can influence the rate of denitrification, the main ones are nitrate supply, readily available carbon and dissolved oxygen (*DO*) concentrations (Groffman 1994).

We have made our estimates of denitrification from the water column using a mean daily denitrification rate derived from total nitrogen measurements made in the inlet basin of the Bakers Creek wetland system (see Appendix 1b). This gave an average rate of denitrification (DN^{rate} in Equation 8) following high flow events of 791 µg L^{-1} d^{-1}. The average peak concentration of dissolved nitrogen in these events, DN_c^{in}, was 4398 µg L^{-1} and this value was used to calculate the input of *DN* to the wetland for all flow events (see Equation 7). This figure is similar to the nitrogen concentration in high flow events in another wetland near Tully, north Queensland (McJannet et al., 20212b). The above denitrification rate is lower than the total potential denitrification rate found in Babinda (972 µg L^{-1} d^{-1}; Wallace et al., 2020), but higher to that found at five other wetland locations in the Tully catchment (Adame et al., 2019), i.e. 623 µg L^{-1} d^{-1}.

A critical parameter in estimating denitrification in water is its dissolved oxygen (*DO*) concentration, as it has been reported that denitrification can only occur when *DO* is low (e.g. Rivett et al. (2008). This study summarised a number of reports showing that the maximum *DO* level at which denitrification occurred varied between 0.2 and 4 mg L^{-1}, with typical values around 30% saturation (equivalent to ~ 2 mg L^{-1}). We have used the measurements of *DO* made in the Bakers Creek wetland recorded every 20 minutes, 10 cm above the bottom of the water column, to derive the amount of time each day that *DO* was below 30% saturation (Waltham et al., 2020). *DO* values in this location varied between 0 and 150% for the period when measurements were made (19 December 2019 to 15 April 2020) and these data were used to calculate the percentage of time DO was below the above threshold value on each day. Daily denitrification was then adjusted to allow for this. On days where DO data were not measured the mean value for the whole measurement period was used, i.e. 67%.

3 RESULTS

3.1 Wetland water balance

3.1.1. Wetland water depth

Figure 5 shows that the water depth in the inlet basin was close to zero at the end of December 2018 but rose to ~ 90 cms following heavy rain between 8 and 11 January 2019 (293 mm). Once the rainfall stopped, there was an initial rapid decline in water depth, followed by a slower decline as the depth fell below ~ 26 cms. Each subsequent major rainfall event increased the water depth again to over 90 cms. As the wet season declined from May 2019 onwards, water levels decreased steadily reaching zero by the beginning of June 2019. By the time logging resumed in mid-December 2019 the water depth in the inlet basin was already 30 cms and this increased and decreased following rainfall events as in the previous wet season. It is clear that water was flowing through the inlet basin during both the wet seasons studied, although there were significant periods of time when depth was quite shallow (34% time < 20 cms and 17% time < 10 cms). This does not include the times when shallow depths would have occurred during the (unmonitored) dry season.

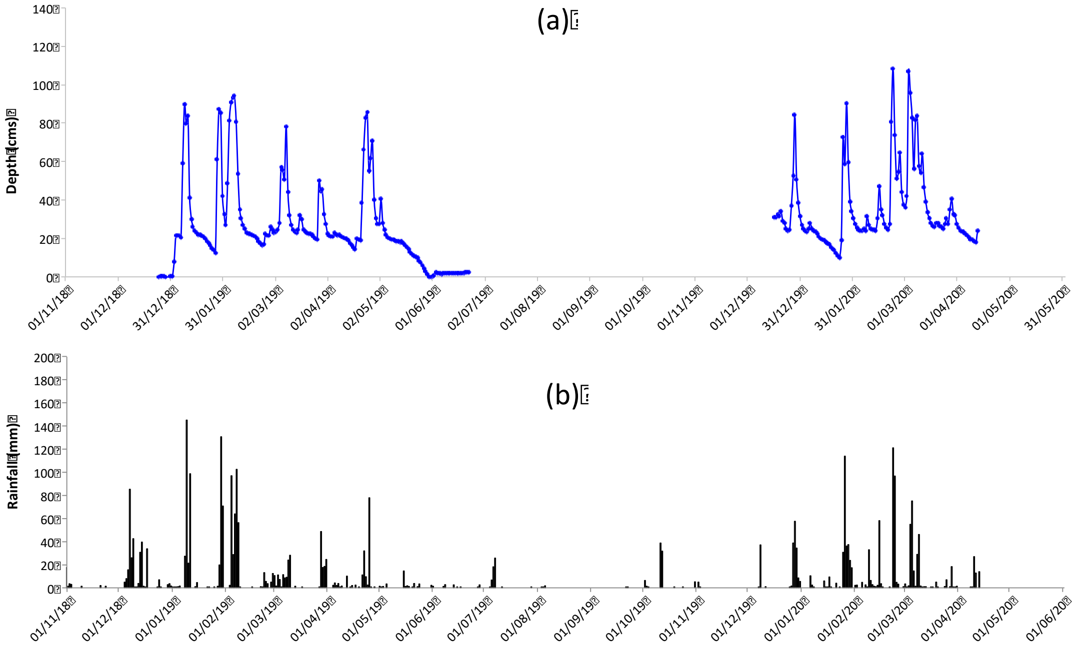

Figure 5 (a) Changes in daily average depth at the wetland system inlet between 25 December 2018 and 25 April 2020 and (b) Daily rainfall during the same period

3.1.2. Wetland drainage

When rainfall stops the water depth in the wetland decreases due to drainage. Estimates of drainage were made from the rate at which the water level in the wetland dropped after the rainfall events in 2018/19 and 2019/20 (Figure 5). The drop in water level on each day after rainfall stopped was adjusted for losses due to evaporation (calculated from weather data – see section 2.1.1). Figure 6 shows the relationship between drainage and wetland depth. When the water depth was below ~ 26 cms the drainage rate was constant relatively slow and constant, averaging 5.2 mm day^{-1} (similar to the daily losses by evaporation). As water depths rose above 26 cms, the drainage rate increased rapidly, reaching over 300 mm day^{-1}. A linear regression fitted to the data when depth was > 26 cms has the form;

$$DRw = 10.0\ d - 254 \qquad (5)$$

and despite the scatter in the data the correlation coefficient is quite high (r^2 = 0.92). Equation 5 can therefore be used to estimate drainage from the wetland when d is above 26 cms (i.e. the value of d when DR_w = 5.2 mm day^{-1}). Below this level DR_w = 5.2 mm day^{-1}.

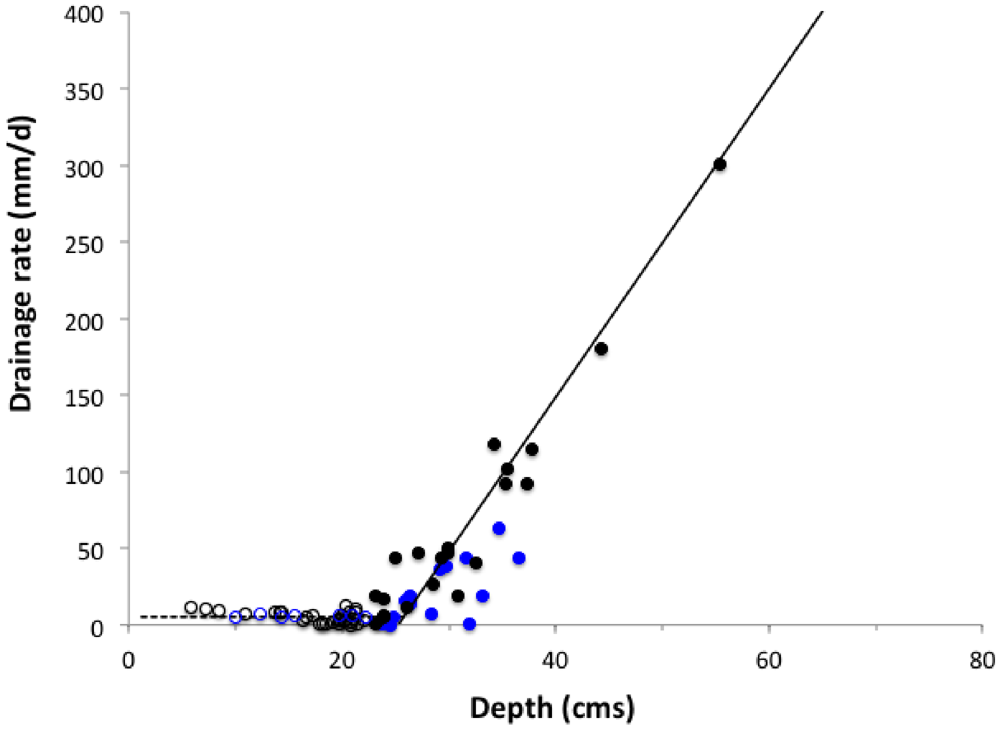

Figure 6. The relationship between drainage from the wetland and depth in 2018/19 (black dots) and 2019/20 (blue dots). When the water is above 26 cms (solid dots) the linear regression has the form, $DR_w = 10.0\ d - 254$ ($r^2 = 0.92$). The dashed line shows the constant value of drainage when the depth is below 26 cms.

With only depth data available it is not possible to separate total drainage (DR_w) into its outflow (Q_{out}) and groundwater seepage (G_w) components. However, given the relatively high values of DR_w obtained when depths were > 26 cms, it is likely that drainage at this time is dominated by outflow over a solid structure at the outlet from the wetland system. It is likely that flow over the structure stops once the wetland depth drops below 26 cms and further drainage losses are mainly due to groundwater seepage.

3.1.3. Run-in to the wetland

The amount of water running into the wetland system was estimated in each wet season by adjusting *C* (see equation 4) to obtain the minimum root mean square difference (RMSE) between the measured and modelled daily depths. The resultant fit between the modelled and measured depths is shown in Figure 7. In the 2018/19 wet season the optimised value of *C* was 5.8, meaning that nearly six times as much water entered the wetland system from the surrounding catchment as entered it directly as rainfall. This run-in includes any flow into the wetland system from adjacent drains and creeks as well as overland flow along the length of the wetland system. The optimised value of *C* in the increases in flow from adjacent drains and creeks, possibly due to clearance of weeds 2019/20 wet season was 7.0, higher than in the previous season. This may be a result of and/or other obstructions.

3.1.4. Wetland water balance model

The water balance model described in Section 2.1 was used to simulate the wetland depth in the inlet basin. Water entering the wetland on each day (R_{in}) was estimated using rainfall data (see Section 2.1.3) and direct input of rainfall (P) was added to this. Losses of water due to drainage (DR_w) were calculated from depth as described above in Section 3.1.2. Evaporation losses were estimated using daily weather data, see Section 2.1.1. Changes in wetland depth were then calculated using Equation 1. The resultant modelled water depths over the entire study period from 25/12/2018 to 30/09/20 are shown in Figure 7. Modelled depths are fairly close to the measured depths and the peak depths are reasonably well captured. Drainage analysis (see Section 3.1.2) suggests that flow through the inlet basin ceased when depths fell below ~ 26 cms and modelled depths were usually less that the measured depths below this threshold. For all depths the root mean square difference between measured and modelled values (RMSE) was 12.5 cms in the 2018/19 wet season and 15.7 cms in the 2019/20 wet season. Overall the model therefore reproduces wetland depths fairly well and has the advantage that it can predict depths and water balance components when there are no measured values.

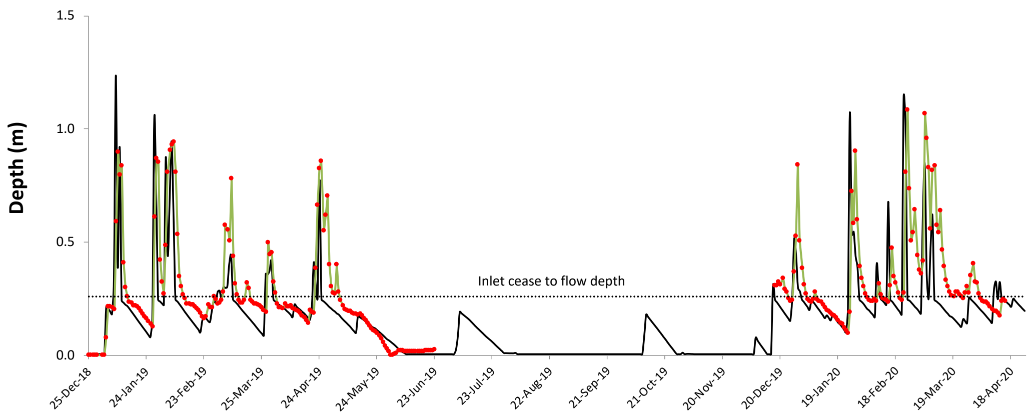

Figure 7. The change in wetland depth (red dots & green line) during the 2018/19 1nd 2019/20 wet seasons. Also shown is the depth modelled (black line) using rainfall data and the water balance model described in section 2.1. The depth at which flow through the inlet basin ceases is also shown (dashed line).

3.1.5. Seasonal water balance

A summary of the seasonal water balance of the wetland system over the study period is shown in Figure 8. The largest water input to the wetland is via run-in from the surrounding catchment, which was over 5 times the rainfall input in the wet seasons. In the dry seasons both rainfall and run-in were much smaller, making up only 9% and 3% respectively of their totals over the entire study period. Loss of water from the wetland system was dominated by drainage, which accounted for ~ 90% of the total loss, the rest being evaporation. However, evaporation made up 45% of the wetland loss during the dry seasons, but the absolute amounts were still much less than in the wet seasons. Most (92%) of the evaporation occurred from the open water in the wetland, but there was also a small component of evaporative loss from the wetland soil, which occurred when there was no water in the wetland.

We have separated drainage into its outflow (Q_{out}) and groundwater seepage (G_w) components by assuming that Q_{out} only occurred when the wetland depth was greater than 26 cms (see Section 3.1.2) and that G_w was constant throughout at 5.2 mm day^{-1}. This revealed that outflow was ~ 90% of the wetland drainage in the wet seasons. As depths were always < 26 cms in the dry season, there was no outflow and drainage at these times was 100% groundwater seepage.

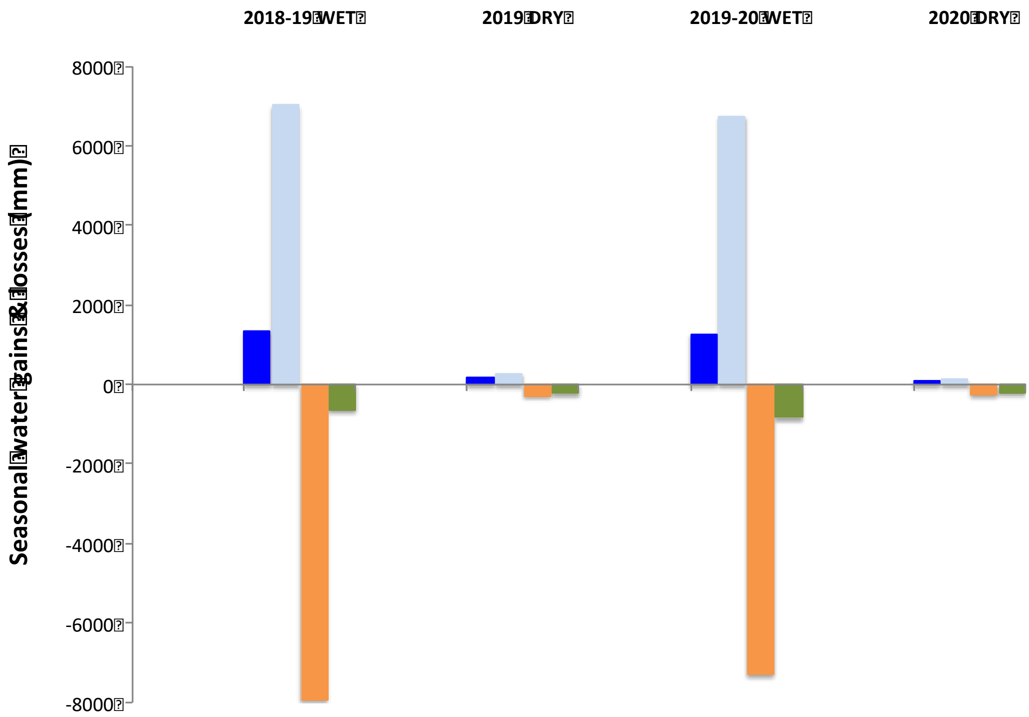

Figure 8. A summary of the seasonal gains and losses of water from the wetland system for the two wet and two dry seasons modelled. Inputs as rainfall (blue), Run-in (light blue) and losses as drainage (orange) and evaporation (green) are shown for each season.

3.1.6. Wetland denitrification

Estimates of wetland nitrogen inputs and losses have been made over the period 25 December 2018 to 30 September 2020 (645 days) using the wetland nitrogen balance model described in Section 2.3. The depth of water in the inlet sediment basin (Zone 1) calculated using the water balance model was used for the entire period (see Figure 7). As there is still water in Zones 2, 3 and 4 of the wetland when the depth in Zone 1 is zero, total wetland volume and area were calculated using the maximum depth in the deepest part of the wetland (Zone 4) – this is given by adding 1.8 m to the inlet depth (see Figure 3).

Figure 7 has shown that over the period simulated there were numerous high flow events that sharply raised the water level and volume of water in the wetland. These events brought large amounts of nitrogen into the wetland, up to 74 kg of *DN* and 11 kg *PN* in a single day Figure 9a. Much smaller amounts of *DN* entered in the rainfall, a total of 24 kg for the entire period, compared with 952 kg of *DN* brought in by inflow. This was over six times the amount of *PN* that entered as inflow, 143 kg for the entire period. The total nitrogen input (*DN* + *PN*) that entered the wetland during the study was therefore 1120 kg, equivalent to a loading of 196 kg ha^{-1} $year^{-1}$.

As soon as water entered the wetland, nitrogen was lost from the water column as *PN* settlement, denitrification and in drainage water, Figure 9b. The largest loss was via drainage which carried 585 kg of nitrogen out of the wetland, 91% as *D*N and 9% as *PN*. Denitrification removed 442 kg of *DN* from the wetland over the study period with *PN* settlement within the wetland amounting to 93 kg. All loss mechanisms were greatest during high flow events, with comparatively little nitrogen loss when wetland depths and flows were relatively low.

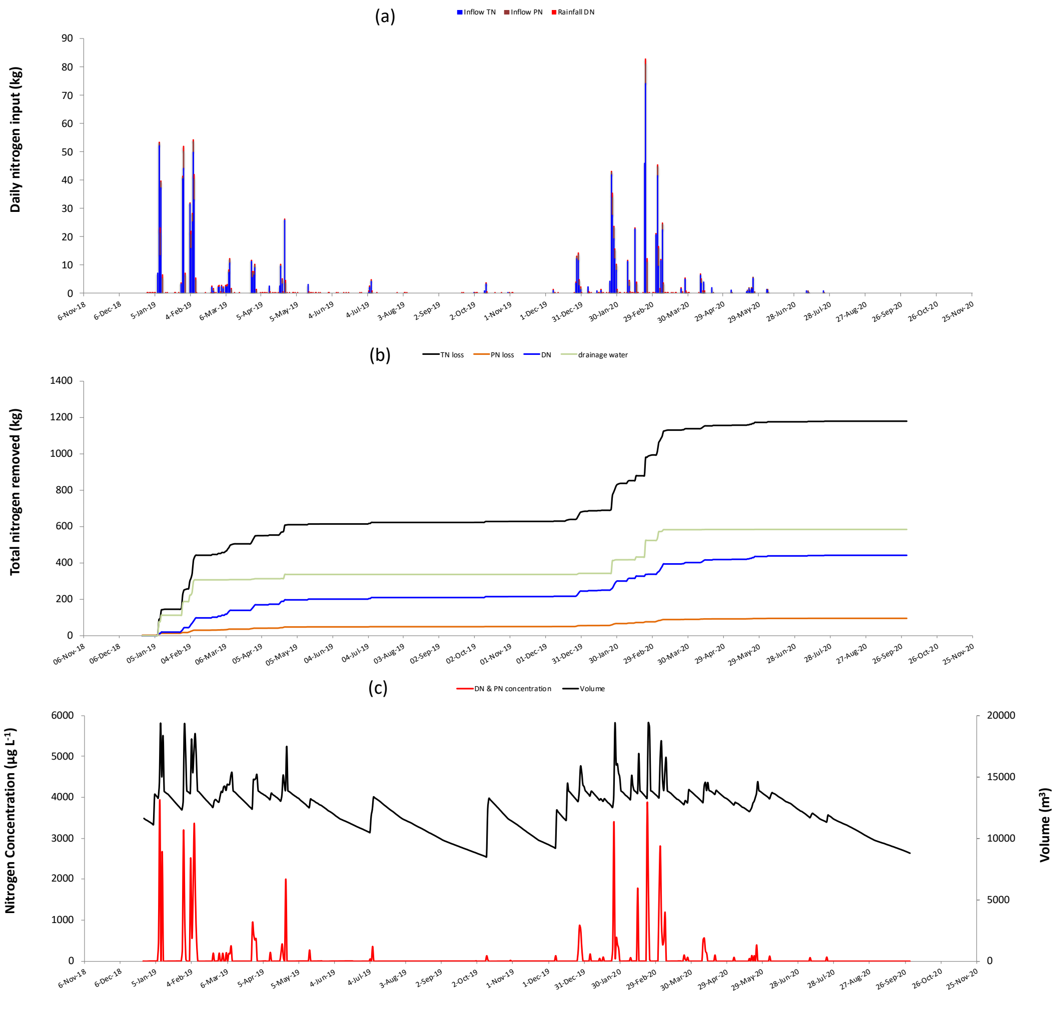

Figure 9. Annual time series of (a) daily nitrogen inputs from DN inflow (blue), PN inflow (brown) and rainfall DN (red), (b) cumulative nitrogen losses by drainage (green), water column denitrification (blue), PN loss (brown) and total nitrogen loss (black) and (c) wetland water nitrogen concentration (red) and volume (black).

The model simulation of the variation in total nitrogen concentration in the wetland is shown in Figure 9c. Peak concentrations between 3000 - 4000 µg L^{-1} coincided with the high inflow events, but these decreased quickly in the days following peak. This is a consequence of the rapid losses of nitrogen in the drainage, along with the high denitrification rate used and relatively fast settlement of particulate nitrogen. This rapid loss of nitrogen leads to the concentration of nitrogen in the wetland being close to zero for most (87%) of the time.

Table 1 summarises the nitrogen balance of the wetland over the entire (~ 2 year) study period. The inflow flow events brought a total of 1120 kg of nitrogen into the wetland, equivalent to 224 kg N ha^{-1} year^{-1}. In this analysis we have estimated that 83% of this (196 kg N ha^{-1} year^{-1}) was *DN* and 13% (29 kg N ha^{-1} year^{-1}) was *PN*. The largest loss of *DN* was via drainage (55%), with denitrification accounting for the remaining 45%. Drainage losses also accounted for 35% of the *PN* that entered the wetland, however, the majority (65%) settled on bottom of the wetland. The total amount of nitrogen (*DN* + *PN*) filtered by the wetland was therefore 107 kg N ha^{-1} year^{-1}, or 48% of the input. The amount of nitrogen filtered varied with individual high flow events, ranging from 11% for short duration events (4 - 6 days), to 93% for longer events (20 - 30 days).

Table 1. Summary of wetland annual nitrogen balance

	kg ha^{-1} year^{-1}	*% of input*
Nitrogen input of *DN*	196	
Nitrogen input of *PN*	29	
Total Nitrogen input	224	
Drainage loss of *DN*	107	*-55*
Drainage loss of *PN*	10	*-35*
Wetland denitrification *DN* loss	88	*-45*
Particulate settlement *PN* loss	19	*-65*
Total filtering of nitrogen by wetland	107	*-48*

4 DISCUSSION

This study was established to try and determine the efficacy of a multi-component constructed wetland for improving water quality entering the Great Barrier Reef lagoon. Using a combination of water and nitrogen balance models we have been able to estimate that the Bakers Creek wetland system would filter ~ 48% of the nitrogen entering it. This result is dependent on the key parameters used in the models and their values are discussed below.

The largest losses of nitrogen were via drainage, accounting for 52% of the total input to the wetland system. We tested how sensitive these drainage nitrogen losses were to uncertainties in the water balance model estimates of drainage. A 20% change in drainage produced a ~10% change in the total nitrogen lost in drainage. However, reducing drainage losses increases the concentration of nitrogen in the wetland (and vice versa) and this increases the loss of nitrogen by denitrification and *PN* loss. This effect, combined with *DN* and *PN* losses being around half of nitrogen input, means that net result is to only change the wetland filtering by ~ 5%.

Most of the drainage from the wetland occurred as outflow (Q_{out}), 77%, with losses to groundwater (G_w) accounting for the remaining 23%. As This wetland drainage losses of *DN* occur in the same proportions, this means that 83 kg ha^{-1} year^{-1} was lost as Q_{out} and 24 kg ha^{-1} year^{-1} as G_w. Outflow also carries sediment and *PN* out of the wetland, with 35% leaving the wetland, equivalent to a *PN* loss of 10 kg ha^{-1} year^{-1}.

The next largest loss of nitrogen is by denitrification of *DN*. We have estimated that this accounts for 88 kg ha^{-1} year^{-1} (Table 1) and this is clearly dependent on the rate of denitrification used, i.e. 791 µg L^{-1} d^{-1}. Higher rates of potential denitrification, 972 µg L^{-1} d^{-1}, have been found in a constructed wetland near Babinda, north Queensland (Wallace et al., 2020). Using this value in our model increases the loss of *DN* by denitrification by 8%. As a consequence, there is a reduction in nitrogen concentration in the wetland, which leads to reduced losses of *DN* in the drainage water. However, since loss of *DN* by denitrification is just under half of the nitrogen input to the wetland, the net effect is to only increase the wetland filtering to 51%. Wallace et al., (2020) used a mean denitrification rate of 427 µg L^{-1} d^{-1}, derived from a range of tropical wetland studies in north Queensland, to estimate *DN* losses in their wetland. If we adopt this figure here, denitrification drops by 15%, but again as denitrification is less than half of the total *N* loss, the net effect is that total filtering only falls to 42%.

Another important parameter affecting denitrification and hence wetland filtering is the dissolved oxygen (DO) level. It has been reported that denitrification can only occur when *DO* is low (e.g. Rivett et al. (2008), below a threshold of around 30% saturation (equivalent to ~ 2 mg L^{-1}). We used measurements of *DO* in the Bakers Creek wetland made 10 cm above the bottom of the water column. This gave the average percentage of time DO was below the 30% saturation threshold of 67%. However, DO is likely to increase towards the surface reaching 100% saturation at the surface. Hence, the percentage of time DO was below 30% saturation at the surface would be zero. Across the entire water column therefore the average percentage of time DO was below the above threshold can be approximated as 67/2 = 33.5 % of the time. Using this value in our nitrogen balance model reduces denitrification by 16% and leads to a net nitrogen filtering of 41%. This assumes that denitrification can take place across the entire water column. This calculation illustrates the importance of having measurements of DO in any system under study both near the surface and bottom of the wetland.

Denitrification can only occur if the nitrogen is in the form of nitrate. Our nutrient analysis data show that, on average, nitrate was only 34% of the total dissolved nitrogen, the majority being *DON*, 63%, with a small additional amount of nitrogen as ammonia (NH_3) – ~3%. In order to denitrify the *DON* component of the nitrogen pool it must first be nitrified and this requires an adequate DO supply. As a rule of thumb, DO needs to be > 2 mg L^{-1} (30% saturation) for *DON* to be nitrified. From our DO concentration data, we calculated that this occurred ~ 33% of the time near the wetland bottom and much more frequently nearer the surface. This should provide adequate time for *DON* to be nitrified. Further evidence for this comes from the consistently low nitrate fraction of *DN* which indicates that nitrification and denitrification were proceeding at similar rates, thus preventing accumulation of nitrate. It is possible that in addition to denitrification there was uptake of nitrate by plants and algae.

Other mechanisms could also have removed nitrogen from the wetland water. For example, when DO was very low it is also possible that microbes could have been converting *DON* into NH_3, which could be taken up by plants. If this was happening, the consistently low NH_3 concentrations observed indicate that plants took up NH_3 as fast as it was being produced. There is also the possibility that there was direct uptake of *DON* (e.g. as urea and amino acids) by plants, algae, fungi and bacteria. However, as our estimates of 'denitrification' were derived from the decline in total dissolved nitrogen values measured within the wetland after high flow events (Appendix 1b), they represent the net effect of gaseous denitrification and any of the above additional nitrogen removal processes that may have been taking place.

To understand the factors that affect the filtering capacity of wetlands it is useful to compare different types in contrasting rainfall climates. For example, the overall filtering of *DN* and *PN* in the Babinda wetland was 56%, with denitrification accounting for 36% and *PN* losses 20% (Wallace et al., 2020). In Bakers Creek we found lower overall filtering of 48% with denitrification accounting for 40% and *PN* losses 8%. The two wetlands have very different hydrological regimes. Babinda is in a very high rainfall area (annual rainfall 4279 mm) and in the wet season the wetland is frequently filled by deep floods (Wallace et al., 2020). Once these floods have receded, water is held in the wetland for long periods (up to 60 days) as there was no outflow from this wetland (the outlet gates were kept closed). In contrast, the Bakers Creek wetland is an open system where water can flow through as long as the depth is over 26 cms. However, as the annual rainfall is only 1585 mm, this only occurred for ~ 15% of the time in the wet season and never in the dry season. As a result, water was retained in this wetland throughout the year (see Figure 9c). In addition, different denitrification rates were used in the two studies, 427 µg L^{-1} d^{-1} in Babinda and 971 µg L^{-1} d^{-1} in Bakers Creek, and the overall effect of this and the different hydrological regimes led to broadly similar nitrogen filtering of 107 and 120 kg ha^{-1} $year^{-1}$ respectively. Note that had the two wetlands had the same denitrification rate (427 µg L^{-1} d^{-1}), then there would have been a bigger difference in nitrogen filtering, 92 and 120 kg ha^{-1} $year^{-1}$ at Bakers Creek and Babinda respectively.

Another example of an open flow through wetland is Kyambul lagoon in the Tully catchment (McJannet et al 2012 a,b). This 3 year study of a natural riverine wetland found that little or no nitrogen was filtered, and no sediment was removed in the long term. This result was a consequence of the very large fluxes of water and nutrients that entered and left the wetland during large floods in the wet season. This led to very short residence times of water in this wetland, with ~95% of the annual flow staying in the wetland for less than 10 hrs. Small amounts of denitrification occurred during low flow conditions in the dry season, but these were much less than 1% of the long-term nitrogen input. This study is in stark contrast with the flow through wetland system at Bakers Creek, where 45% of the *DN* that entered it was denitrified and it also trapped 65% of the sediment input. Clearly, the filtering capacity of a wetland is highly dependent on its hydrology, with open wetlands only able to filter significant amounts of sediment and nutrients where rainfall is comparatively low and water is retained for significant periods of time.

The effectiveness of the constructed wetland at Babinda in a very high rainfall area is largely because it can hold and retain water for significant periods, thereby allowing denitrification and sedimentation to take place. The filtering in Bakers Creek is also enhanced by its ability to retain water for long periods. This mainly occurs in the final component of this wetland, the biodiversity pond. This is the largest and deepest part of this wetland system which always contained water, even during the dry season (see Figure 9c). Of the total *DN* removed by denitrification in Bakers Creek, 89% occurs in the biodiversity pond and this only varied between 85 and 90 % for individual high flow events. This illustrates the value of large, deep permanent pools in a wetland system, whether they are natural or constructed.

We have estimated that the Bakers Creek constructed wetland system can remove around half of the nitrogen that enters it and ~ 65% of its sediment input. These types of wetlands should therefore be able to improve the quality of water entering the Great Barrier Reef lagoon if they are placed downstream of areas that generate sediments and nutrients. However, to do this they need to retain water for significant amounts of time, which they may do in comparatively low rainfall areas, particularly if they have outlet structures that stop outflow while there is still significant water in the wetland. In higher rainfall areas open, flow through wetlands may not be able to do this, so constructed wetlands need to be closed in these areas in order to have significant filtering capacity.

Appendix 1 (a): Sediment settlement rate.

Measurements of the concentration of suspended sediments (*TSS*) in the water that entered the Bakers Creek wetland were made during four high flow events in October and November 2017, December 2019 and January 2020. These data are used here to estimate the initial *TSS* and subsequent rate at which this falls for flow events generated by high rainfall in the catchment surrounding the wetland.

Figure A1 shows the time trend in *TSS* for the above four high flow events. Measurements were made in the water in the inlet basin of the wetland during each event.

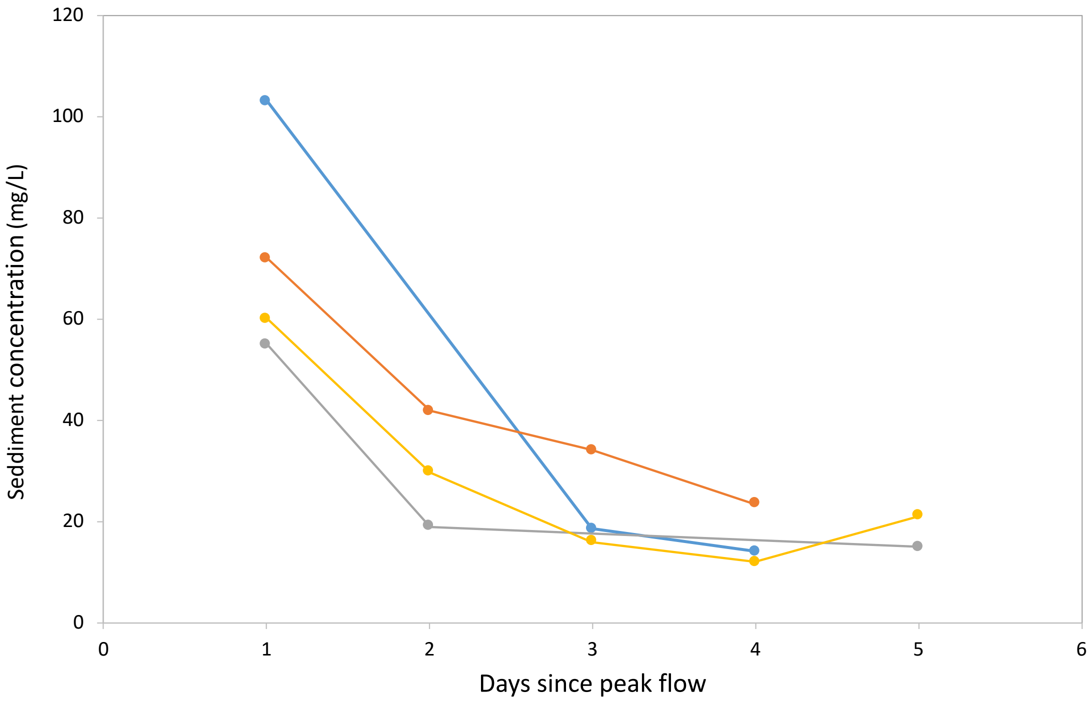

Figure A1. The change in total suspended sediment (TSS) concentration with time after the start of high flow for four events; 17-20/10/2017 (blue), 9-13/11/2017 (grey), 27-30/12/2019 (brown) and 25-29/01/2020 (yellow).

The highest *TSS* concentrations were recorded on the first day and ranged from 55 to 103 mg/L. Following this *TSS* declined and reached ~ 20 mg/L 4 to 5 days after peak flow. Similar *TSS* values were recorded in overbank flood waters in the Tully-Murray catchments (Wallace *et al*., 2020 – Appendix 1).

The data in Figure A1 can be combined to give an estimate of how rapidly *TSS* concentration declines after reaching its peak value in a high flow event. This can be done by plotting the relative TSS_{rel} (i.e. the ratio of *TSS* to the peak value TSS_p) against the number of days since the peak *TSS* occurred, as shown in Figure A2.

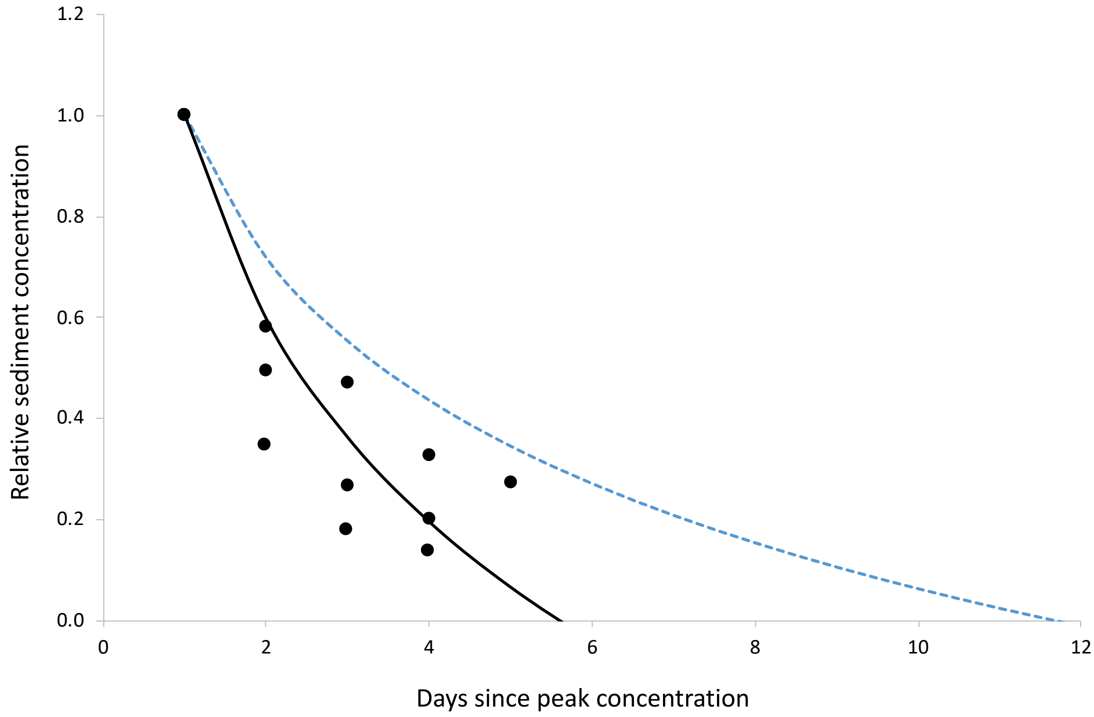

Figure A2. The decline in relative TSS concentration with days (t) after the TSS peak concentration. The fitted line has the form TSS_{rel} = -0.581*ln (t) +1 ; r^2 = 0.96. Also shown in blue is the equivalent curve derived from TSS data in Tully-Murray flood waters.

In all 4 high flow events, relative *TSS* declined at broadly similar rates, which is quite rapid initially as the larger sediment particles settle out of the water. Smaller particles settle more slowly, and so *TSS* concentration falls less quickly and it takes ~ 6 days for all of the sediment to precipitate from the water.

The logarithmic line fitted through the data points for all four events has the form;

$$TSS_{rel} = -0.581 * TSS_p * \ln(t) + 1, \quad \text{(A1)}$$

and this equation can be used to estimate the actual value of *TSS* on any day after the start of a high flow event, if the peak *TSS* concentration is known.

The equivalent equation derived using data from the Tully-Murray floodplain (Wallace *et al*., 2020 – Appendix 1 a) is also shown in Figure A2 for comparison. There *TSS* concentration declined more slowly, taking ~ 12 days for all the sediment to settle. This could be because the sediment particle size distribution in the Bakers Creek contains more larger particles which precipitate more rapidly.

To simplify the calculation of *TSS* and hence particulate nitrogen concentration, *PN*, we derived a fixed daily rate which gave the best match to the logarithmic equation A1. Using the peak *PN* concentration of 662 µg

L^{-1} (see Appendix 1(b) below) this gave a daily average *PN* loss rate of 154 µg L^{-1} d^{-1}, which was applied when the *PN* concentration in the wetland was above zero.

Appendix 1 (b): Nitrogen decay rate.

Measurements of the total nitrogen (*DIN, DON*, *PN* and NH_3) concentration in the water that entered the Bakers Creek wetland were also made during the above four high flow events. These data are used here to estimate the peak *TN* concentration and subsequent rate at which this falls for flow events generated by high rainfall in the catchment surrounding the wetland.

Figure A3 shows the time trend in *TN* in the water in the inlet basin of the wetland for the four high flow events. The highest *TN* concentrations were recorded on the first or second day following each high flow event and ranged from 2667 to 6350 µg L^{-1}. *TN* declined following this to between 1800 to 3000 µg L^{-1} 2 to 4 days after peak flow. The average peak *TN* concentration for all four flow events was 5060 µg L^{-1}. Total dissolved nitrogen (*TN*) contained 63% *DON* and 34% *DIN* (mostly nitrate – 93%), with a small additional amount of nitrogen as ammonia (NH_3) – ~3%.

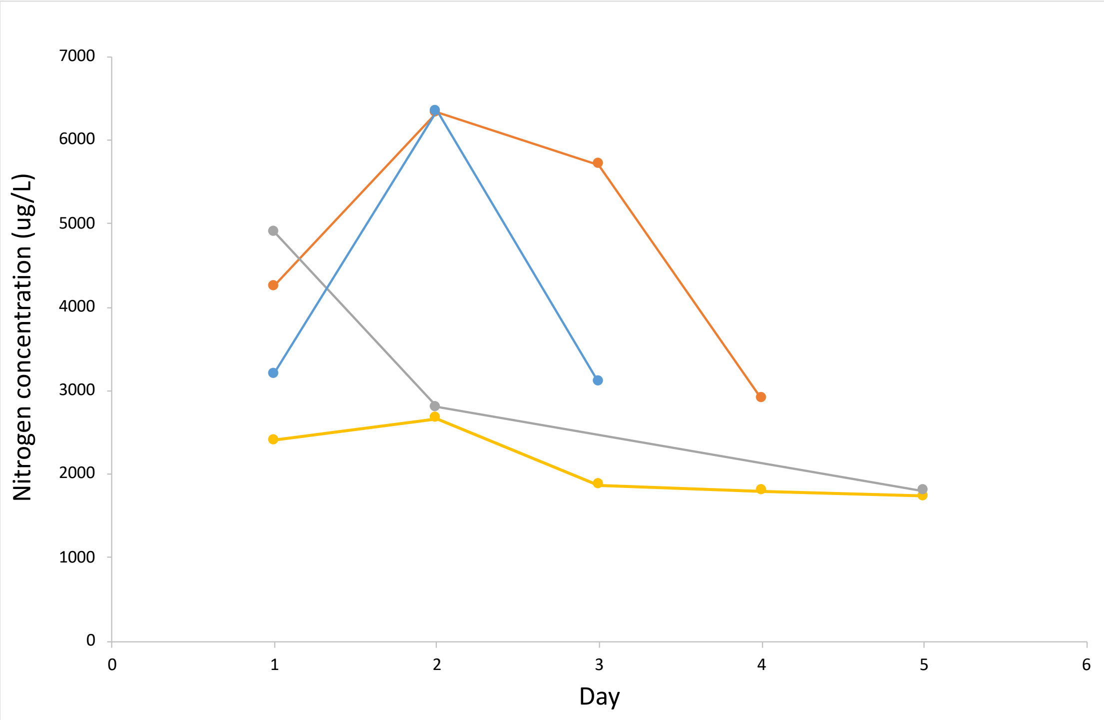

Figure A3. The change in total dissolved nitrogen concentration (TN) with time after the start of high flow for four events; 17-20/10/2017 (blue), 9-13/11/2017 (grey), 27-30/12/2019 (brown) and 25-29/01/2020 (yellow).

The total nitrogen values in Figure A3 include particulate nitrogen, *PN*, and this needs to be subtracted from *TN* to give the concentration of dissolved nitrogen *DN* (*DIN, DON* and NH_4). *PN* was not measured separately in this study, so we have derived approximate values using the relationship between *TN* and *TSS* shown in Figure A4. The correlation is very weak, since *TN* is much more dependent on *DIN* and *DON* concentrations than *TSS*. However, there is a positive slope to the regression, indicating that increasing *TSS* also increases

TN. The slope of the regression can therefore be used to estimate *PN* concentration, i.e. as 14.3 * *TSS* / 1000 (µg/L). The values of PN estimated in this way are plotted against measured values of *TN* in Figure A5. Although there is considerable scatter in the data, the average percentage of *PN* can be estimated from the slope of the line through the origin. This gives a value of 13%, somewhat lower than the mean *PN* fraction of 30% reported by McJannet et al., (2012b) for another wetland nitrogen balance study in north Queensland. However, their study was in a very high rainfall area where there were frequent over bank floods which would have mobilised a greater amount of sediment than in the lower rainfall Bakers Creek catchment.

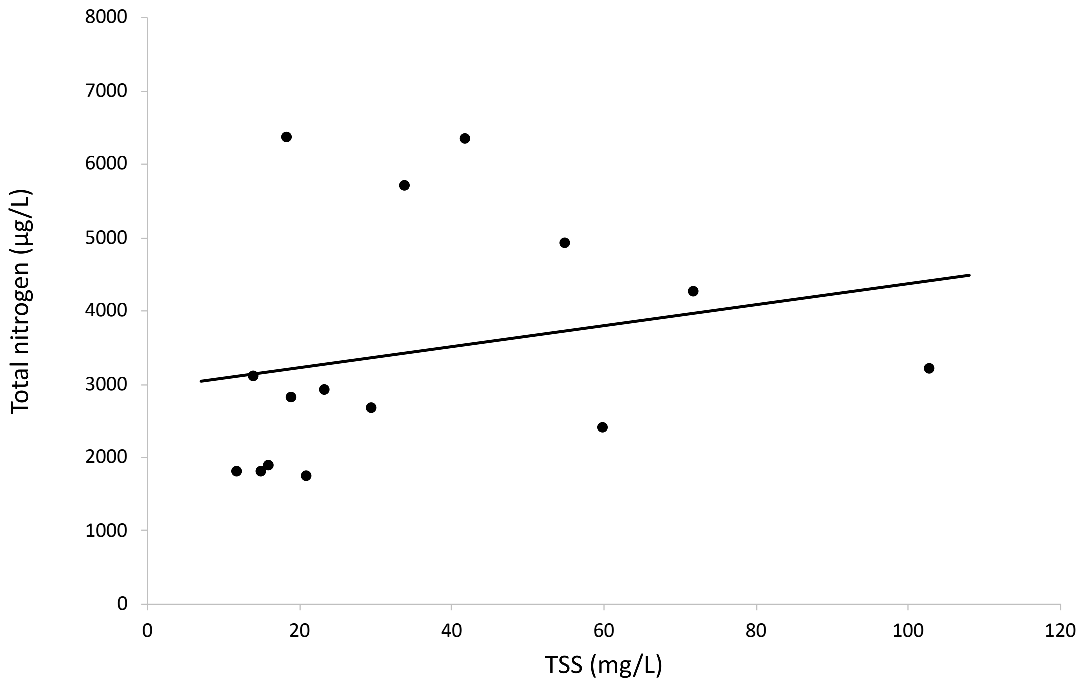

Figure A4. The relationship between total nitrogen concentration (*TN*) and total suspended solids concentration (*TSS*). The fitted line has the form $TN = 14.3 * TSS + 2943$; $r^2 = 0.05$.

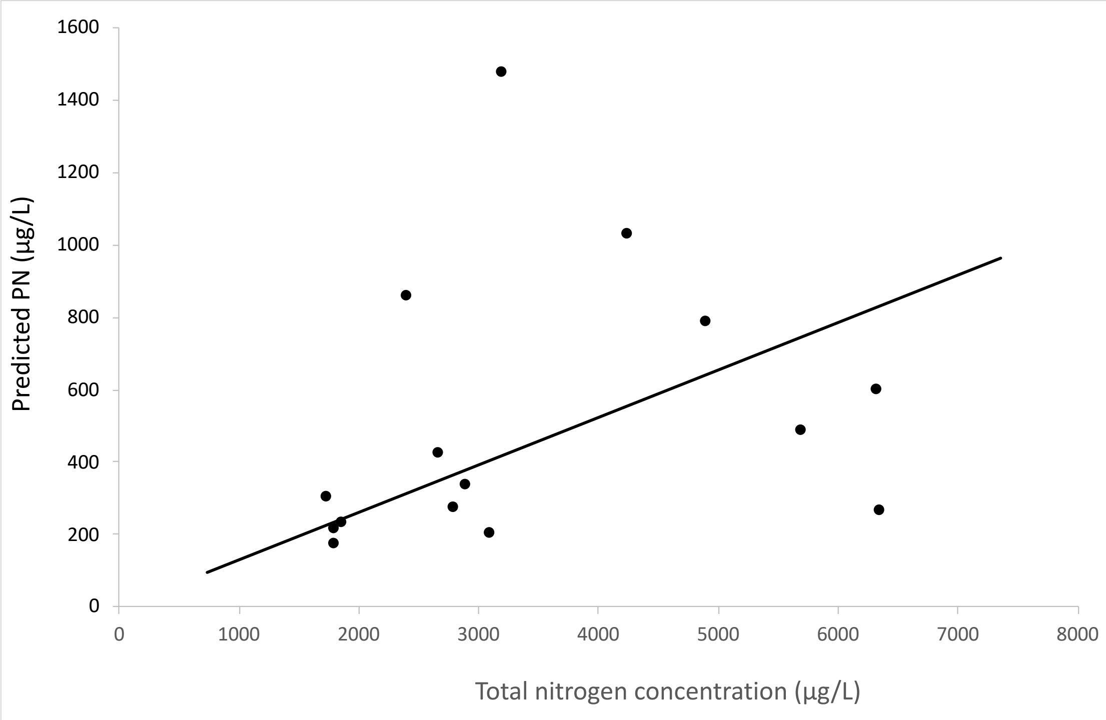

Figure A5. The relationship between predicted particulate nitrogen concentration (*PN*) and measured total nitrogen concentration (*TN*). The line fitted through the origin has the form *PN* = 0.13 * *TN* ; r^2 = 0.63.

Dissolved nitrogen (*DN*) values were estimated from the *TN* data in Figure A3 by subtracting 13% and the rate at which the remaining *DN* declined after reaching its peak value in each flow event calculated. This was done by plotting the relative DN_{rel} (i.e. the ratio of *DN* to the peak value DN_p) against the number of days since DN_p occurred, as shown in Figure A6. This shows that *DN* decreased to about half of its peak value by around 4 days after the peak concentration.

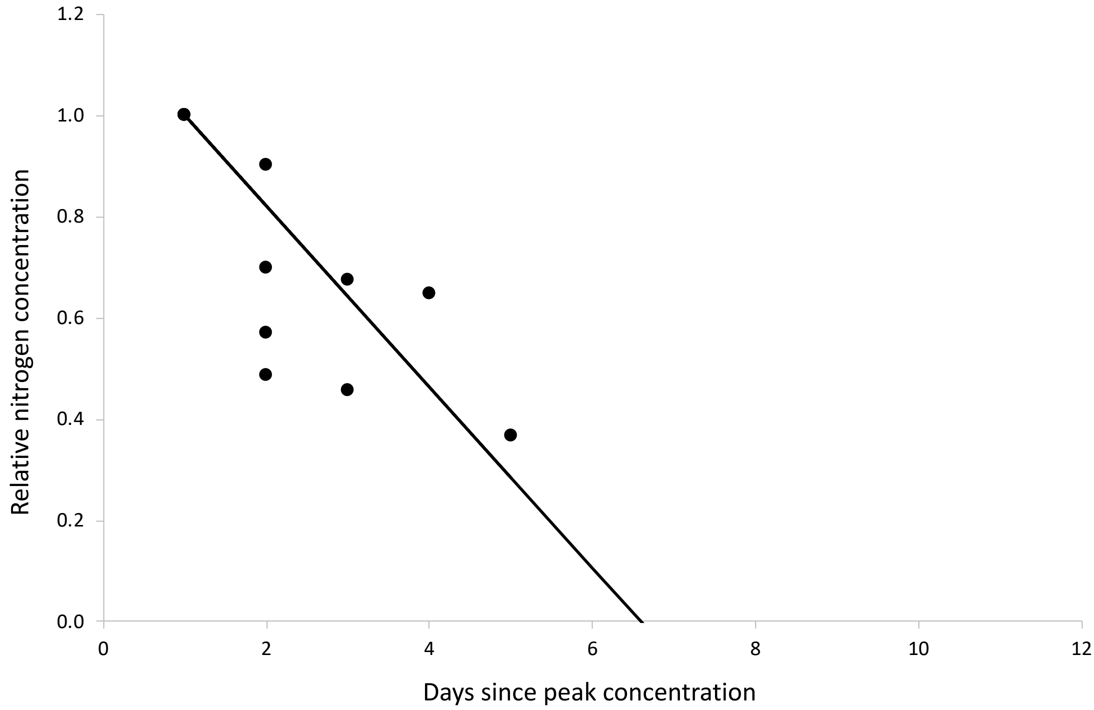

Figure A6. The decline in relative dissolved nitrogen (*DN*) concentration with days (*t*) after the *DN* peak concentration. The fitted line has the form DN_{rel} = -0.180 * *t* +1.18 ; r^2 = 0.82.

Although there is considerable difference in the rate at which DN_{rel} declines in each high flow event, the data from all 4 events can be combined to give an estimate of the average rate of *DN* decay. This is given by fitting a straight line fitted through the data which gives a relationship of the form;

$$DN_{rel} = -0.180 * DN_p * \mathrm{t} + 1.18, \qquad \text{(A2)}$$

and this equation can be used to estimate the actual value of *DN* on any day after the start of a high flow event, if the peak *DN* concentration is known. Equation A2 also predicts that it takes just over 6 days for all of the nitrogen to be lost from the water.

The average rate of denitrification (of dissolved nitrogen) following high flow events can now be estimated using Equation A2. This requires a value for the peak *DN* concentration during the four high flow events shown in Figure A3, and their average peak concentration is 4399 µg L^{-1}. Using this value, the predicted values of *DN* on the subsequent 6 days given by Equation A2 correspond to a daily denitrification rate of 791 µg L^{-1}. This value assumes that all of the decline in *DN* is due to denitrification and this would be an overestimate if water entered the wetland after peak *DN* with a lower nitrogen concentration than the water already in it.

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