Nutrient Assimilative Capacity of Queensland Estuaries

Overview

The focus of this study is on assimilative capacity of estuaries to sustainably receive and process nutrients with the view to better regulate point source discharges, particularly within the Great Barrier Reef catchment in Queensland. Point source activities that discharge nutrients to an estuary have the potential to significantly influence receiving water quality, particularly during ambient, non-event periods. This document summarises project work to date prior to publication of a full technical report later in 2025.

Estuaries in Queensland are tropical/subtropical and are tidally dominated. The nature of an estuary can vary significantly from location to location. The hydrodynamics of tidal estuaries is particularly complex and highly variable due to tidal forces and other factors that affect mixing. As a result, assessing impacts to water quality can be complex, requiring significant monitoring data and information over time and space to properly characterise an estuary. Because of this complex and dynamic nature, there is currently no clear methodology, or associated predictive tools, to easily assess the assimilative capacity of nutrients in estuaries.

A review of studies of assimilative capacity of waterways around the world has been undertaken. In general, assimilative capacity is defined by an ‘acceptable level of change’ that a waterbody can sustain and typically relates to ‘bio-stimulants’ rather than toxicants (Masini et al., 1992). For aquatic ecosystems, this would ideally consider ecosystem functioning, preservation of diversity and species abundance, for example. However, there are significant challenges in monitoring and assessing many of these aspects of aquatic ecosystem health, including cost, sampling effort/access, and lack of guidelines/methodology, particularly for estuaries. Perhaps as a result, most studies on assimilative capacity use a single water quality indicator, such as total nitrogen or chlorophyll-a concentration, and a related water quality standard or objective for that indicator to define the maximum acceptable change. Information is also needed on the current/historical levels of the chosen water quality indicator. Where the current condition is below the water quality standard for that indicator, the difference in concentration is commonly referred to as the available assimilative capacity. Most studies then rely on hydrodynamic and biogeochemical models to estimate the additional input load (mass per time) that could occur without exceeding the concentration that marks available assimilative capacity. A well-known example of this is the total maximum loads (TDMLs) that have been adopted by the United States Environmental Protection Agency for managing anthropogenic loads of nitrogen and phosphorus to waterways (Hashemi Monfared et al., 2017).

Conceptual understanding of tidal estuaries in Queensland has been developed in terms of hydrodynamics, nitrogen and phosphorus. This includes an assessment of important environmental processes, largely based on work undertaken by Ryan et al (2003) as part of the Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management in the early 2000’s. For example, Figure 1 shows a conceptual model for nitrogen in tide dominated estuaries. Tidally dominated estuaries in central and north Queensland can go for long periods between event flows. During these periods, water mixing, flushing and residence times are dominated by tidal input, typically with a proportion of the total estuary volume moving out of the estuary each day (Ryan et al., 2003).

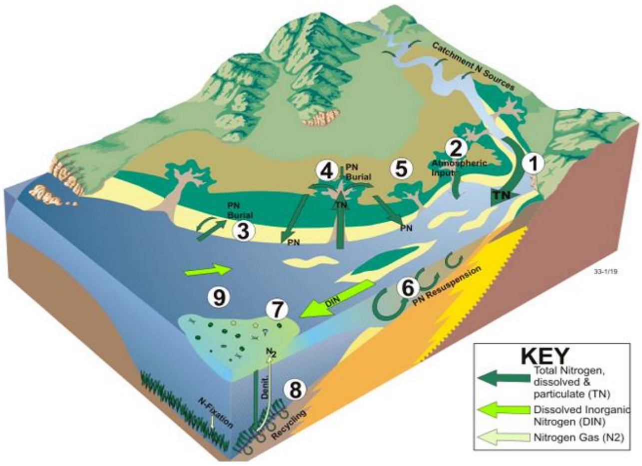

Figure 1: Conceptual model of nutrient dynamics in tide-dominated estuaries (Ryan et al., 2003). 1 Catchment runoff; 2 Atmospheric deposition; 3 Intertidal deposition; 4 Mangrove uptake/export; 5 Salt flat burial/export; 6 PN deposition/resuspension; 7 Phytoplankton update/conversion; 8 Seagrass uptake; and 9 Export to marine environment.

This large exchange of seawater results in significant quantities of constituents, such as nitrogen, being exported to the marine environment. The conceptual model includes multiple nitrogen (N) removal processes such as N burial, mangrove assimilation, etc. However, for the purposes of this report denitrification is assumed to be the main N removal process in estuaries.

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Estuary Water Quality Objectives and Indicators

Water Quality Objectives (WQOs) are generally based on water quality guidelines and are defined for many estuaries in Queensland (Environmental Protection Policy 2019 - Water and Wetland Biodiversity). WQOs can be used to define the upper concentration limit of a constituent, such as TN and TP, and be used to help define assimilative capacity.

For estuaries, WQOs are typically defined for total nitrogen (TN), total phosphorus (TP), ammonia, oxidised nitrogen (NO_x), filterable reactive phosphorus (FRP), suspended solids, and chlorophyll-a concentrations and dissolved oxygen saturation. WQOs are also defined for different water types, including enclosed coastal/lower estuary, mid estuary and in some cases, upper estuary. Therefore, up to three different water types and related WQOs may apply depending on the characteristics of the estuary. A further potential complication is that estuaries may be defined by different levels of protection, which are high ecological value (HEV), slightly disturbed (SD), moderately disturbed (MD), and highly disturbed. Most waters are MD and the WQOs are typically based on 80th percentiles of reference monitoring (QWQG, 2009). HEV areas have more stringent guidelines based on 20th, 50th and 80th percentiles of reference sites.

The choice of indicator for defining assimilative capacity of nutrients in estuaries is a critical decision. Typically, in Queensland, point source releases are regulated based on TN and TP concentrations and loads. Mass balance and conservative models can easily be developed for TN and TP, as compared to other nutrient indicators. As a result, TN and TP have been a particular focus of this study. Dissolved nutrients have also been investigated, and we have found that assessing oxidized nitrogen is also important, particularly for impacted systems. As mentioned, chlorophyll-a has been used for some assimilative capacity studies, and although an important ecological indicator of waterway health, reliable model predictions can be challenging and usually require biogeochemical models.

Condition Assessment

The first step to determine assimilative capacity involves assessing the current condition (or ecological status) of the receiving water. As part of this review, we assessed three estuaries in Queensland using historical water quality monitoring data collected by the Queensland Government. The choice of estuary was based on the availability of nutrient and physico-chemical data, duration of the dataset (> 10 years), the number of estuary monitoring locations (minimum of four), and whether the estuary was in the Reef catchment. The three estuaries chosen were the Moresby River Estuary, which is potentially impacted by point sources; the Baffle Creek Estuary, which is considered a reference system; and the Daintree River Estuary, which has high freshwater inflow. It should be noted that none of the systems have monitoring locations throughout the entire estuary or in the freshwater reaches which limited the review.

The condition assessment of these estuaries was done by analysing the available water quality data (typically the last 10 years) and comparing results to WQOs for each indicator for each monitoring location. Typically, the methodology recommended in the Queensland Water Quality Guidelines (2009) for assessing physico-chemical indicators in MD systems is to ensure that the annual median, typically from 12 monthly samples, lies within the guideline (generally based on the 80th percentile of the reference data). However, annual medians can vary from year to year due to rainfall and other factors, which means that the assimilative capacity would vary from year to year. For this reason, a condition assessment based on long-term data percentiles was also assessed and compared to the annual medians, as a potential method for determining assimilative capacity from long-term data.

Condition assessment of TN for the Moresby River, the Baffle Creek, and the Daintree River estuaries is presented in 2, 3 and 4, respectively. The boxes show the 25th, 50th and 75th percentiles of long-term TN data. The upper and lower range of annual medians calculated for each year of the 10-year period are shown by the dotted lines, while the solid line represents the middle value.

The assimilative capacity can be assessed along an estuary at locations where data has been collected. Figure 2 and 3 clearly shows that the highest annual median TN value of both the Moresby and Baffle (based on the upper dotted line) exceed WQOs at the upper estuary monitoring locations. This is despite the Baffle being a reference system. In comparison, both systems show lower concentrations in the middle to lower estuary locations, suggestive of higher assimilative capacity. Only the Daintree River (Figure 4) had TN levels below WQOs, potentially due to freshwater influence and low nutrient inputs.

The results show a relatively large variation in annual median concentration for each monitoring locations (as shown by the two dotted lines). Interestingly, the 25th and 75th percentiles of the data provide a reasonable estimate of the range of annual medians for the three systems. However, the 75th percentile is above the highest annual median in all cases. Therefore, the long-term 75th percentile could be used as a conservative approach to determining assimilative capacity, where sufficient data exists.

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Mixing Plots and Assimilation Assessment

Mixing plots are a tool to examine the transformation of water quality constituents in an estuary by comparing constituent and salinity concentrations along an estuary. Our case study estuaries had mixing plots developed for TN, NO_x, ammonia and TP as shown in Figures 5, 6, 7 and 8, respectively.

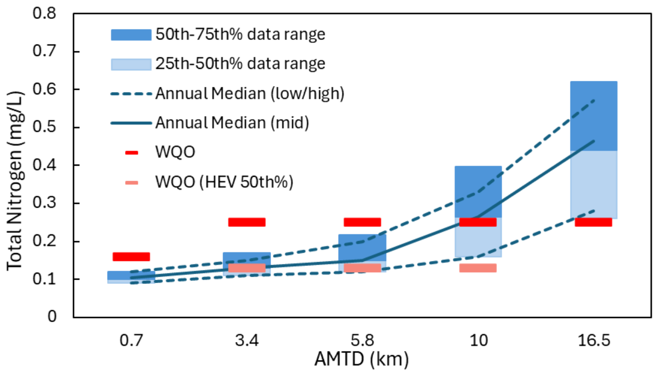

Figure 2: Box plots of total nitrogen at sampling points along the Moresby River Estuary. AMTD is adopted middle thread distance.

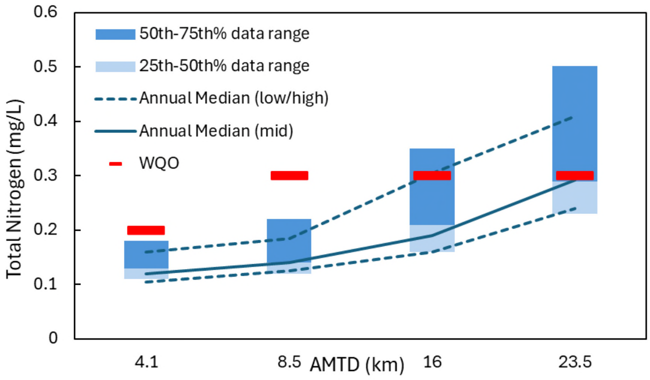

Figure 3: Box plots of total nitrogen at various sampling points along the Baffle Creek Estuary. AMTD is adopted middle thread distance.

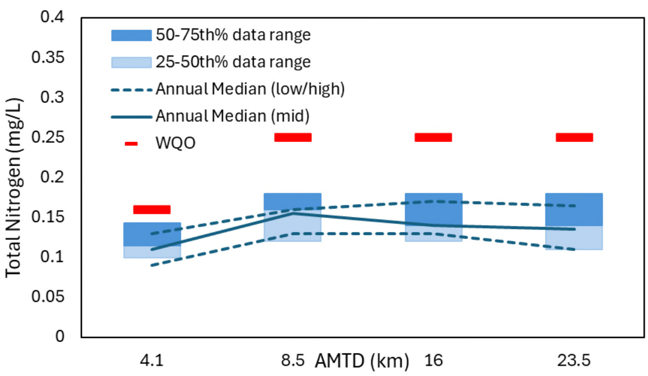

Figure 4: Box plots of total nitrogen at sampling points along the Daintree River Estuary. AMTD is adopted middle thread distance.

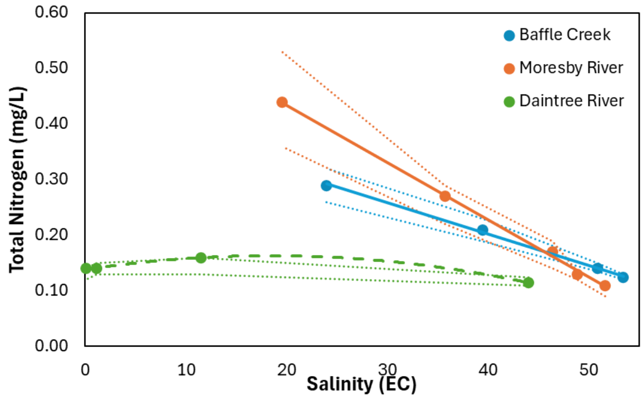

Figure 5: Mixing plot of total nitrogen versus salinity for the case study estuaries (2014 to 2024 data). Solid lines are linear regressions. Dashed line shows interpolated concentrations. Dotted lines are bootstrapped 95% confidence intervals.

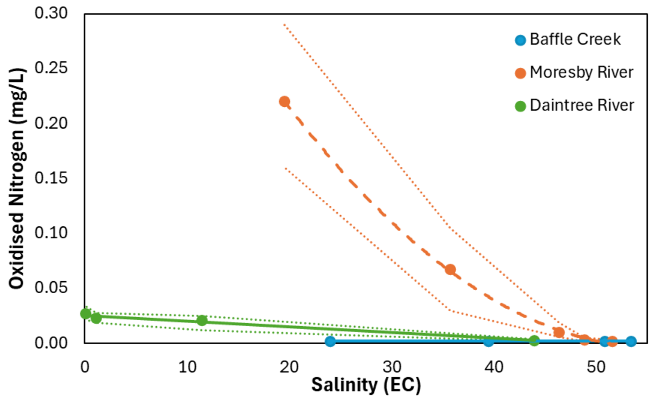

Figure 6: Mixing plot of oxidised nitrogen versus salinity for the case study estuaries (2014 to 2024 data). Solid lines are linear regressions. Dashed line shows interpolated concentrations. Dotted lines are bootstrapped 95% confidence intervals.

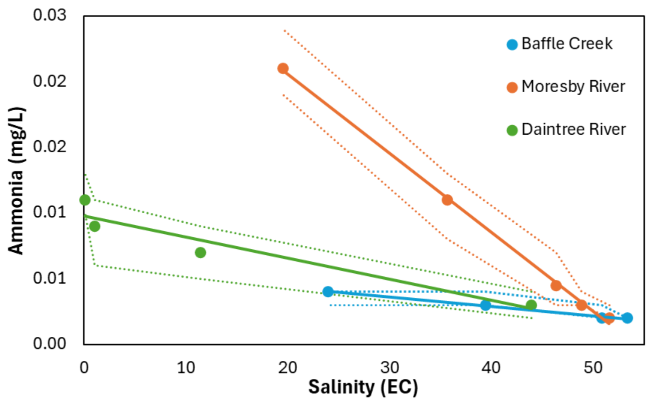

Figure 7: Mixing plot of ammonia versus salinity for the case study estuaries (2014 to 2024 data). Solid lines are linear regressions. Dotted lines are bootstrapped 95% confidence intervals.

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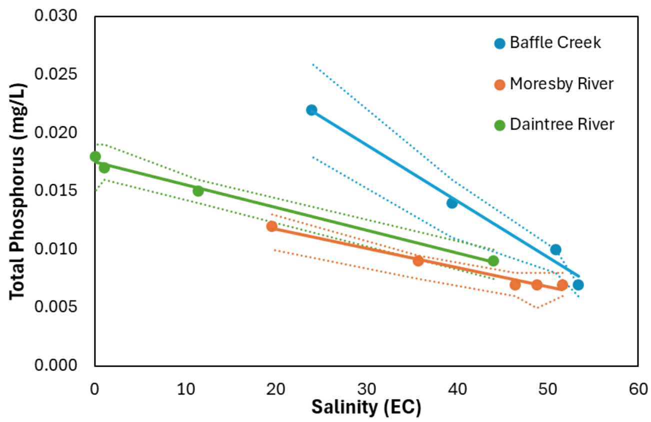

Figure 8: Mixing plot of total phosphorus versus salinity for the case study estuaries (2014 – 2024 data). Solid lines are linear regressions. Dotted lines are bootstrapped 95% confidence intervals.

These plots show the median value at each location based on the long-term, 10-year data set. Individual mixing plots were also examined for each sampling day but showed significant variation in both EC and constituent concentrations between sampling days. Nonetheless, the overall shape of the mixing plots (not shown here) was generally similar to the median plots. To account for this variation, the plots also show the 95% boot strapped values for each location using methods described by Carpenter and Bithell (2000) and Mangiafico (2016).

Linear regression lines are included in each plot for each system to illustrate the potential for mixing/flushing to describe the relationship between EC and the constituent concentrations. In the case where there is significant processing of a constituent throughout the estuary, such as from denitrification, the measured values would be lower than linear values in the estuary and show a sagging curve. In the case where there was input of point source nutrients at one or more points in the estuary, the results would rise above linear and show a rising curve.

For our case study estuaries, the three systems generally showed near linear correlations between TN and salinity ($R^2 > 0.996$), except for TN in the Daintree system (Figure 5). This suggests that the net change in total nitrogen across these estuaries is likely to be largely described by hydrodynamics and mixing. The non-linear relationship in the Daintree points to potential accumulation (convex shape). This may be due to nitrogen inputs from a major tributary in the lower estuary, the high freshwater flow in this area, and the similar levels of TN concentration between the ocean and the upper reaches of the estuary. Regardless, none of the three estuaries showed sagging curves for TN. This suggests that net loss of nitrogen from biogeochemical processes is not significant enough to influence overall changes in TN throughout the estuary. If denitrification is the main process involved, it would be consistent with low rates of total nitrogen removal via denitrification that have been found in other large estuaries, such as the Brisbane River, estimated by Newham et al. (2024) to be up to 2-3% of TN load.

Oxidised nitrogen for the Moresby had a significant sag in the curve rather than a linear relationship (Figure 6). This may be a result of algal or bacterial action that converted NO_x to other forms of nitrogen, such as organic nitrogen. This sag is not evident for the Daintree or the Baffle which have near linear relationships, noting the NO_x levels in the Baffle are extremely low. Ammonia concentrations were well described by linear relationships for all systems noting the low levels involved (Figure 7). Total phosphorus concentrations were also largely described by a linear relationship (Figure 8), except in the Moresby, where there is a significant sag in the curve indicating possible loss of phosphorus through processes such as flocculation and deposition (Coelho et al., 2004).

Simple Water Quality Models and Residence Times

Simple hydrological models were first used for assessing the residence times of our case study estuaries. Residence time can help understand how quickly an estuary is flushed and thereby how long point source nutrients would accumulate within the estuary. Our review of water quality data across the depth for the three systems, which is not included in this summary, showed that water column is well-mixed during ambient, non-event periods, and thus tidal prism models (and other simple models) are potentially applicable. Considering that Queensland estuaries are tidally dominated, we examined three simple models for calculating residence time: freshwater fraction; tidal prism; and estuary residence time with box models.

The tidal prism models assume complete mixing within the estuary as well as with respect to the flow of freshwater and tidal exchange with the ocean (Lucas and Deleersnijder, 2020). Freshwater fraction models examined are useful for estuaries with large freshwater inputs (Sheldon and Alber, 2006). We also examined simple box models which are a steady-state model that organizes an estuary into boxes and uses salinity and freshwater flow to calculate exchange rates between 'boxes' or sections of an estuary (Officer, 1980). Exchange coefficients are then used to simulate how much time it takes for a constituent that is evenly distributed within an estuary to reduce to 36.7% (also known as the e-fold time). This is referred to as estuary residence time (ERT).

A comparison of the residence time for the case study estuaries using the three different simple models is shown in Figure 9. There was a large variability in the prediction of residence time between models and systems. The freshwater fraction model was very dependent on the predicted freshwater flow for each system. Gaining accurate information for this was difficult given a lack of local stream gauging. Therefore, we believe the freshwater fraction model results are less reliable. The tidal prism model relied strongly on estimates of bathymetry, depth and volume. Although estimated with more confidence than freshwater flows, this may have been a contributing factor to inaccuracies. In both cases, the estuaries all had several tributaries, the contribution of which was difficult to quantify but will have affected the results. In comparison, the box model relied mainly on measurements of salinity and the water quality constituent (TN) to estimate residence time. For this reason, we believe the box model predictions are more accurate when considering the available data.

Based on the assessment using simple models, we believe that each of our case study systems have a relatively short residence time, likely between one and six days during ambient conditions. In this case, the water quality in these systems could change significantly over the period of weeks and will be largely flushed within one month. Therefore, we propose that assessing these systems in relation to monthly changes is more relevant than over longer periods, such as annually.

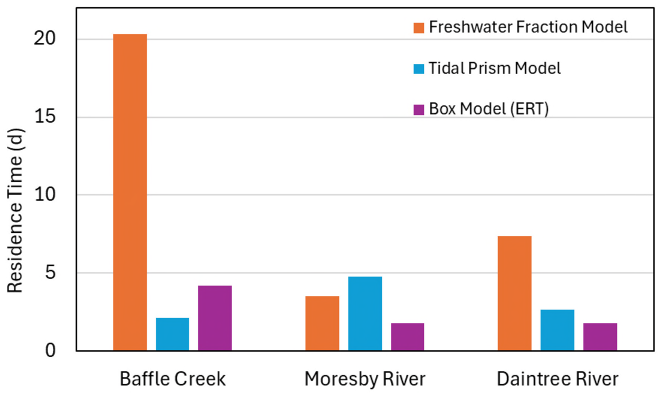

Figure 9: Residence times for the three case study estuaries using three types of simple hydrological models.

The exchange rates of box models can also be used to calculate the residence time of a simulated pulse of a point source release within a specific box within an estuary, referred to as the pulse residence time (Miller and McPherson, 1991). We used the box model to calculate pulse residence time (PRT) of each box within each estuary as shown in Figure 10. In all cases, PRT increases with distance up the estuary. The ERT was found to be representative of the PRT towards the lower end of the estuary. The residence time at the upper estuary boxes were significantly higher. For example, for Baffle Creek, the ERT was 5.6 days compared to the PRT of the upper box of 16 days. These results are consistent with the observation of poorer mixing and flushing in the upper parts of an estuary. We can also hypothesise that the amount of denitrification and other loss processes, e.g. N burial, uptake by mangroves and macrophytes, will likely vary accordingly with these residence times (see next section).

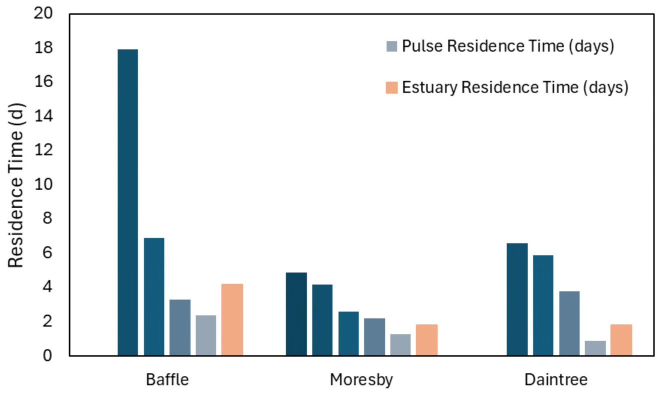

Figure 10: Box model estimates of residence time of the case study estuaries, including pulse residence time (lighter grey boxes are closer to the ocean) using simulated release from various points in the estuary.

Estimating Denitrification Rates

As TN and TP are released into an estuary from a point source, a change in concentration is expected to occur in the surrounding waters. The residence time estimates from the simple models give us an indication of how long nutrients will stay within the estuary. Other processes, such as denitrification are only likely to substantially contribute to nitrogen losses if the water residence times are long enough. As a result, longer residence times may allow for more nitrogen loss through denitrification. An empirical model developed by Seitzinger et al. (2006) relates the residence time of the estuary with an estimated denitrification rate ($R^2 = 0.62$). We used this model to estimate denitrification rates from box model residence times to conceptualise how denitrification rates might differ throughout an estuary (Table 1). This assumed that denitrification was the major loss term for nitrogen.

Table 1: Box model estimates of pulse residence time of our case study estuaries with corresponding denitrification rate estimates using the Seitzinger et al. (2006) empirical model.

Estuary	Residence Time (days)	Denitrification Range	Denitrification Overall
Moresby River	1.3 - 4.9	0.9 - 3.5%	1.3%
Daintree River	1.8 - 6.6	0.3 - 4.0%	1.3%
Baffle Creek	2.4 - 17.9	1.7 - 12.8%	3.0%

The predicted overall denitrification rates of the estuaries were generally at or below 3%. Lower residence times, and therefore less denitrification, is predicted in the middle/lower estuary locations, which coincides with the locations identified with available assimilative capacity. Therefore, for releases to these lower parts of the estuary, denitrification is likely to play a small role in the removal of TN. However, denitrification rates were much higher in the upper reaches of each estuary (e.g. 13% in the upper reaches of Baffle Creek), due to higher residence times.

Estimating Export Rates

Box models can also be used to estimate the export rates of TN and TP into the ocean from each estuary. Historical measurements of TN and TP concentrations throughout the estuaries and the ocean are required to get an accurate estimate of TN/TP export. We used regressions with currently available monthly monitoring data to estimate ocean TN and TP concentrations. The Moresby River, an impacted estuary, had the highest export rate of TN of the three estuaries examined but the lowest TP export rate (Table 2). The Moresby River has two point source inputs to the estuary; one to the upper estuary and one near the mouth of the river. The Daintree River has the next highest TN export, potentially due to higher freshwater and catchment input. The TN export was 5.8 kg/h less than the Moresby River. Baffle Creek is an example of a relatively unimpacted estuary, and it had the lowest TN export, 8.7 kg/h less than the Moresby River. At this stage, it is unclear why the TP export from the Moresby River is the lowest.

Table 2: Summary box model estimates of export rates (kg/h) of total nitrogen and total phosphorus.

Estuary	TN Export (kg/h)	TP Export (kg/h)
Moresby River	20.4	0.59
Daintree River	14.6	1.26
Baffle Creek	11.7	1.53

End-of-system export rates can be compared with typical point source input loads. The monthly export rates of TN were determined for eight aquaculture facilities based on the TN release data collected by Ramsay et al. (2020) in 2018 (Table 3). Maximum monthly TN releases ranged from 0.2 to 6.3 kg/h.

The end-of-system export rates for the Moresby River provides an example of a system which currently exceeds assimilative capacity for TN while the Daintree River has available assimilative capacity for TN. In addition, the residence times determined from the box models for the two systems are relatively similar. Assuming the systems are therefore comparable, this would suggest that the addition of TN to the Daintree in the order of 5.8 kg/h is potentially unsustainable and could exceed assimilative capacity in the middle/upper locations of the estuary. On the other hand, lower additional loads would have more chance of being sustainable.

Further work is required to convert point source load inputs at different locations in the estuary into changes in instream water quality, either using box models or more detailed hydrodynamic assessment, to provide more accurate predictions of sustainable loads for estuaries.

Table 3: Summary of the peak monthly TN release rates from eight aquaculture facilities using 2018 data from Ramsay et al. (2020)

Facility Label	Type	Peak Monthly Export (kg TN/h)
F5	Prawn, Barramundi	6.3
F8	Prawn, Seafood	3.2
F7	Prawn, Other, Seafood	3.2
F6	Prawn	2.6
F1	Prawn	2.1
F2	Prawn, Other, Seafood	0.9
F4	Barramundi	0.5
F3	Barramundi	0.2

Conclusions

The assimilative capacity of estuaries to receive nutrients can be defined using a primary water quality indicator, such as total nitrogen concentration, and a related water quality objective. This is consistent with approaches adopted around the world for point source regulation.

Water quality objectives are available for estuaries in Queensland for a range of relevant water quality indicators. Total nitrogen was used as the primary indicator for assessment of nutrient assimilative capacity in an estuary. Oxidized nitrogen was found to be important, particularly for impacted systems.

Other than dilution, denitrification has been documented as the main biogeochemical process expected to remove nitrogen in estuaries. In comparison, phosphorus is likely removed through precipitation with iron or aluminium. This study examined the potential significance of assimilatory processes relative to hydrodynamic processes for three case study estuaries using long term water quality measurements, mixing plots and application of box models.

To assess the assimilative capacity of an estuary, it is important to define its current condition. This was done using annual medians of monthly monitoring results and comparing these values to water quality objectives. However, results can vary significantly from year to year. As an alternative, the 75th (and 25th) percentiles of the long-term, 10-year data were found to conservatively estimate the upper (and lower) bounds of annual medians.

Limited or no assimilative capacity for total nitrogen was found for the middle/upper estuary monitoring locations for the Moresby and Baffle estuaries. This was not the case for the Daintree estuary which has high levels of freshwater inflow. It should be noted that the assessment was limited to locations where monitoring data was available, which did not include the very upper part of any estuary. In comparison, all three systems showed available assimilative capacity of total nitrogen in the lower/middle estuary locations. For total phosphorus, there was generally assimilative capacity along the estuary, even for the Moresby Estuary which is most impacted. As a result, total nitrogen was the primary focus of this study.

Mixing plots based on the median values of 10 years of data were found to generally show a linear relationship between constituents and salinity concentrations along each estuary. In other words, the long-term concentration of constituents, such as total nitrogen, oxidised nitrogen, ammonia and total phosphorus, can be largely described by hydrodynamics and the net biogeochemical assimilation is relatively small. Some significant assimilation of oxidised nitrogen (and total phosphorus) was observed in the Moresby estuary. However, this did not correspond to any significant net assimilation of TN throughout the estuary.

Simple models are potentially suitable for assessing the assimilative capacity of estuaries in Queensland. Our case study estuaries are tidally dominated and were found to be well-mixed across depth under ambient conditions using Queensland government salinity data. Tidal prism models appear more suitable than freshwater fraction models but require accurate information on volumes (bathymetry) while freshwater fraction models require accurate information on freshwater flows. Box models appear to provide the best potential to assess estuaries in terms of residence times, export rates and concentrations along the estuary, but strongly rely on accurate salinity and constituent monitoring data along the estuary and at its boundary conditions.

The residence times of the three case study estuaries were found to be relatively short under ambient conditions (at around 1 to 6 days). Pulse residence times were estimated by box models to simulate input from points in the estuary. The overall estuary residence time, as described by box models, was found to be comparable to a pulse residence time for the lower estuary locations (boxes). The pulse residence time for the upper estuary locations were larger, 2 to 6 times higher than the overall estuary residence time.

Considering the low residences times estimated for the three case study estuaries, complete turnover of water quality is likely to occur within one month. Therefore, assimilative capacity, and related sustainable nutrient loads, should be assessed monthly rather than over a longer-term, annual basis.

Overall estuary denitrification rates were estimated using an empirical model that relies on residence times. Overall rates were 3% or less. Relative denitrification was much higher for upper estuary locations, 4 to 13%. However, the upper estuary locations are not suitable to receive point source inputs due to the poor flushing in these locations, as indicated by the higher residence times.

End-of-system export rates were estimated for the three case study systems, and ranged from 12 to 22 kg/h and 0.6 to 1.5 kg/h for nitrogen and phosphorus, respectively. The difference in nitrogen export rates between two of the systems, one with and one without assimilative capacity, was approximately 6 kg/h. In comparison, the maximum monthly release rates from eight aquaculture facilities ranged from 0.2 to 6.3 kg N/h. It can be concluded that these larger point source release rates have the potential to exceed assimilative capacity in the middle/upper parts of the estuary while the lower rates are more likely to be sustainable.

Further work is being undertaken to investigate whether box models can be reliably used to predict instream concentrations along an estuary simulating point source load inputs at different locations in the estuary. However, simple models, particularly box models, appear suitable for assessing estuary condition and estimating potential nutrient export rate when good historical water quality data is available.

Findings from this study regarding simple models should be compared to more detailed hydrodynamic and biogeochemical modelling and assessment. In addition, this study did not look at the initial mixing of point source releases and potential acceptable mixing zones. This is a further area of work that could be assessed using hydrodynamic models.

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April 2025