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1 Introduction

1.1 Background

Tweed Shire Council (TSC) were co-opted to assist the NSW Department of Environment and Climate Change (changed to DECCW and now NSW OEH) in carrying out a routine water quality monitoring program along the Tweed River and Rous River estuaries during 2007 and 2008. This was intended to assist DECC in the development of models underpinning the CERAT and DEFIRE programs (DECCW 2009; DECCW and ABER 2009). TSC suspended its existing routine water quality monitoring program in order to adopt the program designed by DECC and ABER. TSC opted to continue the monitoring program beyond the initial period proposed by DECC, and has continued to monitor all sites on a monthly basis. This report provides an analysis of the data collected in this program, and provides advice to TSC about the design of any ongoing water quality monitoring programs in the Tweed estuary. This report is intended to supersede the draft report provided to TSC in September 2010.

1.2 Monitoring program aims

The primary aim of the monitoring program designed by DECC and ABER was to provide supplementary data to assist in the development of estuarine response models being undertaken by DECC as part of the CERAT and DEFIRE projects. It was seen as necessary to redesign the existing TSC sampling program (which focussed all effort on the immediate vicinity of the sewage treatment plant outfalls) in order to better cover the entire estuarine gradient. It was anticipated that this approach would provide a more comprehensive indication of 1) ecosystem function, 2) the relative impacts of the STP outfalls, and 3) the impacts (if any) of the diversion of effluent from the Murwillumbah STP in December 2007 for reuse in the Condong sugar mill.

1.3 Report aims

This report aims to:

1. Analyse spatial and temporal trends in the water quality data collected in the Tweed estuary between 2007 and 2011
2. Synthesise this analysis into a clear conceptual understanding of interactions between anthropogenic and natural pressures, water quality and ecological outcomes in the estuary
3. Provide a more quantitative analysis of the relative impacts of catchment and sewage effluent inputs to the estuary using a simple salt balance approach.
4. Provide an analysis of the potential impacts of changes to effluent management at the Murwillumbah sewage treatment plant.
5. Provide feedback and recommendations to TSC regarding ongoing water quality monitoring in the Tweed estuary

2 Interpretative background

2.1 Systems overview

This section provides a map for understanding the linkages between catchment pressures, physico-chemical water quality attributes ('stressors'), and ecological outcomes in the estuary (Figure 1). While not all of the complexities are represented in Figure 1, this map is intended to foster a 'systems view' of the various physical and biological aspects that go together to make up the Tweed estuary. This will provide an integrated interpretive framework to better understand water quality data.

Pressures are defined here as human activities occurring in the catchment that have direct impacts on water quality (stressors) in the estuary, and indirect impacts on the ecology of the estuary. There are other natural pressures that impact on water quality stressors such as climatic controls (e.g. rainfall and seasonal temperature cycles). These natural pressures account for variation in stressors that is partly independent of, but also interacts with, human pressures. In trying to interpret water quality data and infer ecological impacts it is important understand the temporal and spatial nature of pressures impacting the system. Further, these linkages provide a strong conceptual basis for more quantitative attempts to understand system behaviour (e.g. modelling).

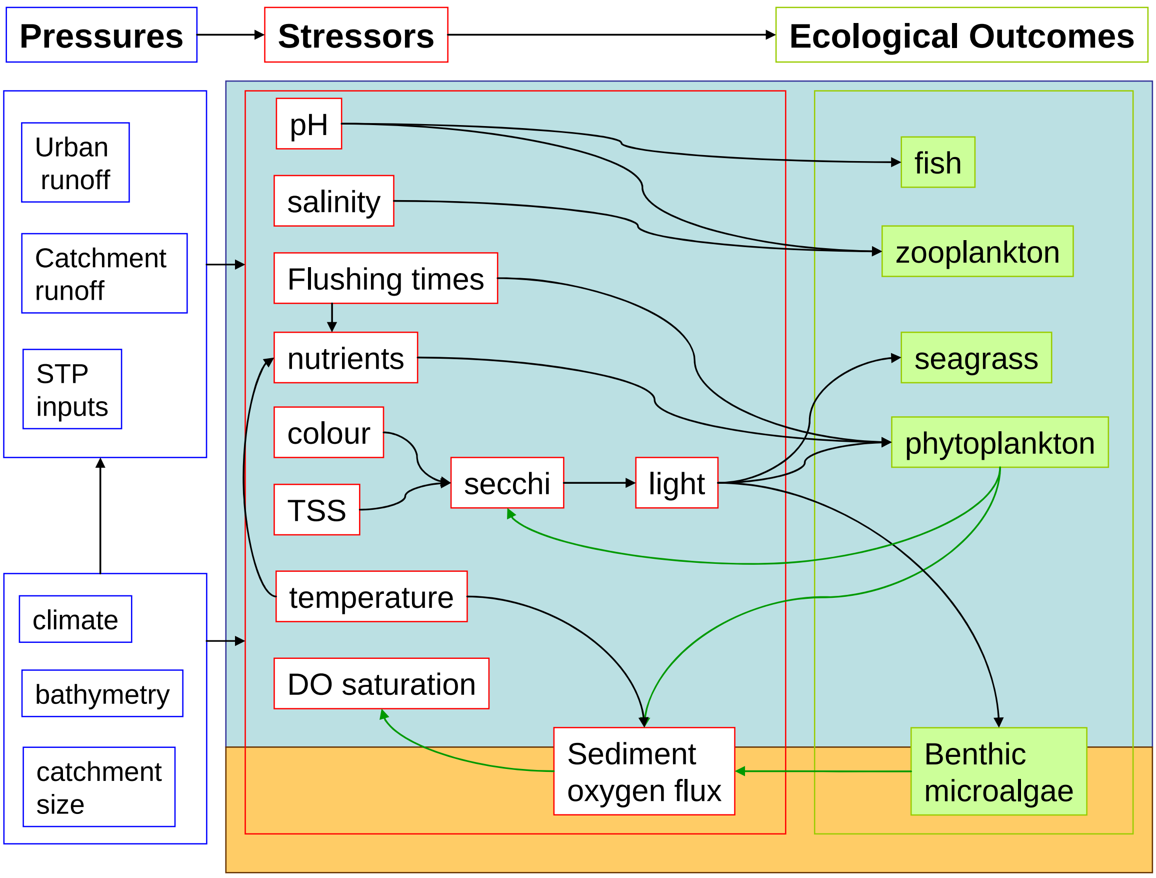


Figure 1 Conceptual model showing the linkages and feedbacks between pressures, stressors and ecological outcomes in estuarine systems.

Figure 1 also shows that there are important linkages and feedbacks between various stressors and ecological outcomes. These will be discussed more fully throughout the report in order to better explain observed trends in the data. These linkages and feedbacks are central to understanding the flow on effects of different pressures, and for informing the architecture of ecosystem response models.

2.2 Guide to interpretation of water quality results

Water quality data analysis is presented in a logical order of parameters that follow the hierarchy of interactions occurring in estuarine ecosystems:

1. Basic physico-chemical parameters (temperature, salinity and pH) are presented first as these stressors are primary determinants of water chemistry and ecology.
2. Nutrients are presented next as they constitute the link between the basic physic-chemical stressors and the primary ecological response (primary productivity by phytoplankton and benthic algae and macrophytes).
3. Chorophyll-a, used as a proxy for phytoplankton biomass, follows nutrients in recognition of this primary link between stressors and ecological response.
4. Total suspended solids and their control over water clarity form the primary forcing factors of light climate, a major stressor controlling ecological responses. This section also elucidates feedbacks from phytoplankton on water clarity.
5. Dissolved oxygen saturation is dealt with last as the important stressor is impacted by many interactions between other stressors and feedbacks from ecology.

2.3 Freshwater flows

Water quality stressors and ecological outcomes in estuaries are highly influenced by freshwater flows, due to their control over inputs of catchment material (i.e. TSS and nutrients) and water residence times along the estuary. Estuarine function within different reaches varies dramatically as a function of the progression between high and low flow conditions. Any interpretation of water quality must therefore place the data into context with the flow conditions prevailing at the time of collection. The data in this report has been grouped according low (<10th percentile), median (50th percentile) and high (>90th percentile) flows. Flow statistics for this grouping were derived from total daily catchment discharges estimated using the OEH catchment export model (see section 6).

2.4 Spatial trends

Data has been analysed for spatial trends occurring during low, median and high flow conditions (see Figure 2A for example output). The sample sites chosen for this study constitute a smooth progression along the estuarine gradient (i.e. they display minimal overlap in terms of salinity), and have been grouped according to proposed functional zones (lower estuary, transition, middle estuary, upper estuary, and Rous estuary) to aid in discussion of results. The spatial analysis summarises data from each site using boxplots generated by the statistical package SPSS v13.0. An explanation of the boxplot output is presented in Figure 2B.

2.5 Temporal trends

Data has been also analysed for trends occurring over monthly, seasonal or annual timescales. This analysis uses visual presentation of timeseries plots of data from all sites, and multi-factorial ANOVAs (factors: month, year) using the statistical package SPSS v13.0. The results of this analysis are summarised in plots showing monthly variation in average estuary concentrations for each year (Figure 3). Significant differences between years were detected using LSD post hoc tests and are reported at the $p<0.5$ level.

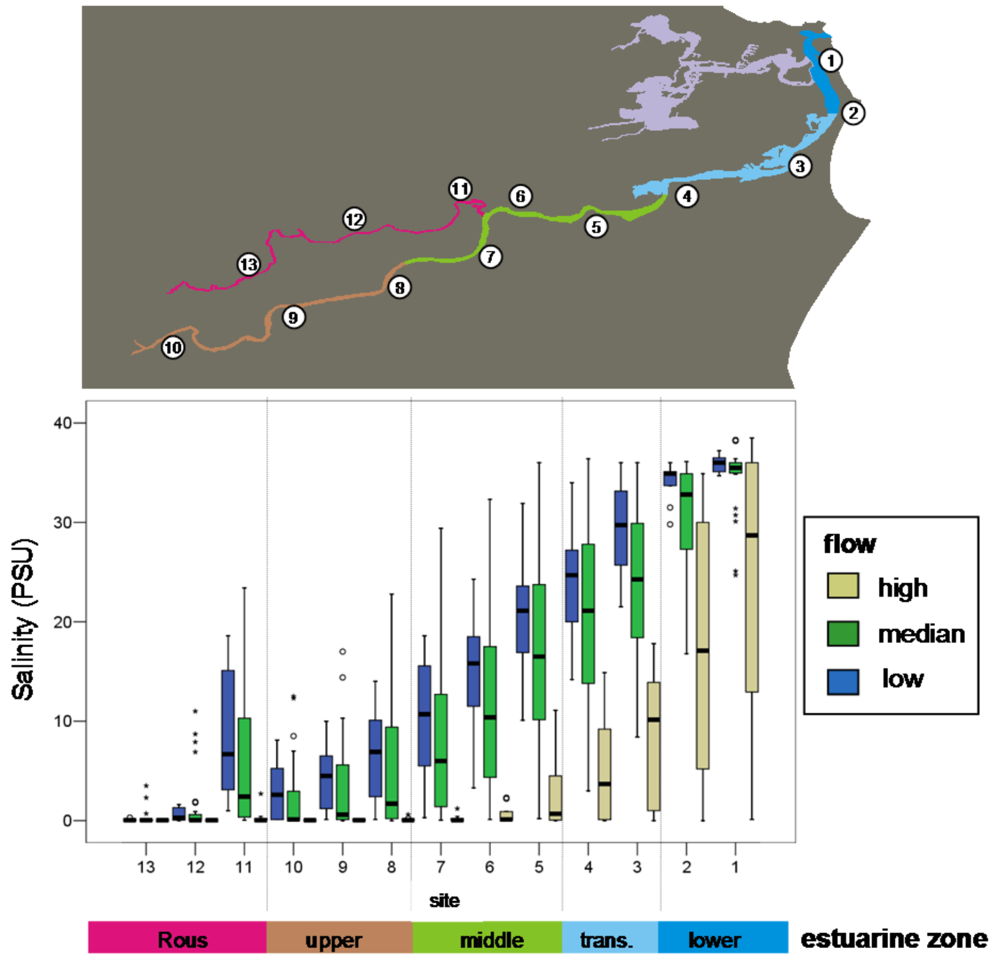


Figure 2A Spatial presentation of salinity statistics during low, median, and high flow conditions. The grouping of sites according to estuarine zone is based on various attributes outlined in section 5. The definition of boxplot statistics is presented below.

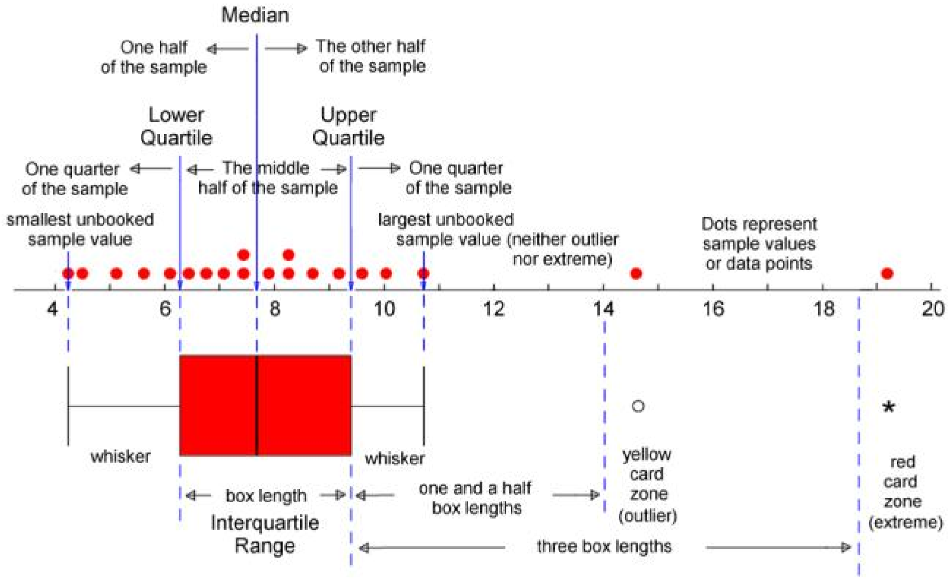
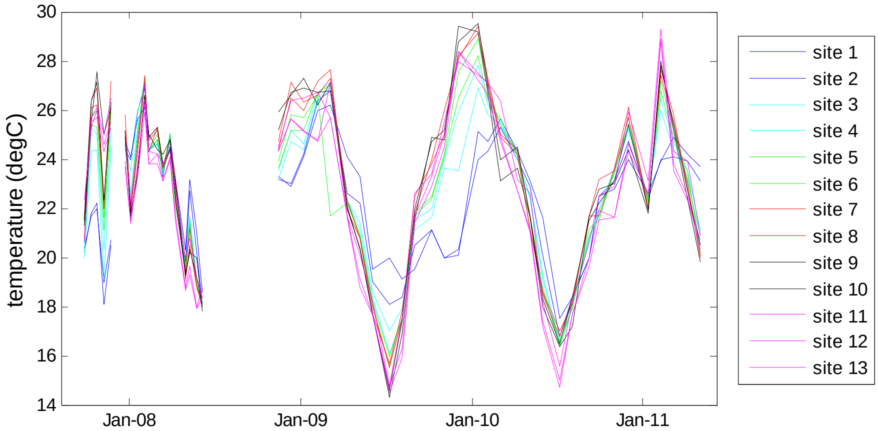


Figure 2B Explanation of boxplots generated by SPSS v13.0.



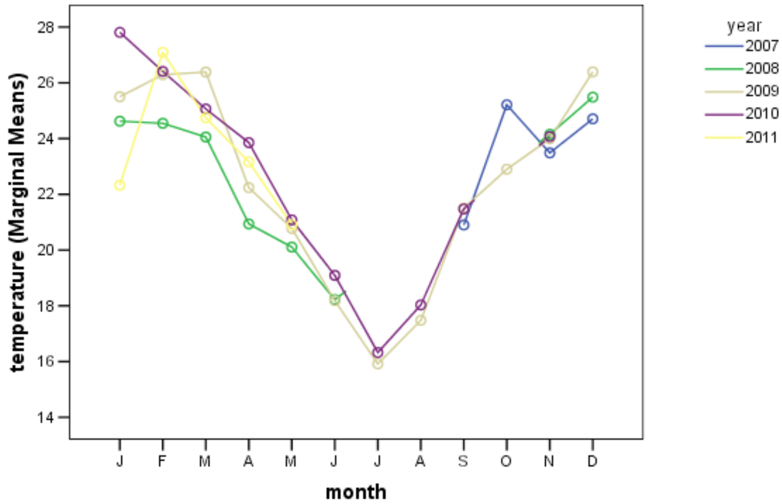


Figure 3 Temporal variation plots for temperature, displaying strong seasonal variation and minimal inter-annual variation. Both sources of variation were significant ($p<0.05$).

3 Monitoring program methods

3.1 Sampling period

Sampling commenced on the 25/09/07 and was carried out fortnightly at 13 sites along the estuaries until 4/06/08 (Figure 4). After this date there was a brief pause in sampling u ntil it recommenced on a monthly basis on 3/09/08.

3.2 Sample collection

Samples were collected midstream commencing from the estuary mouth at Tweed Heads as near to high tide as possible[1]. At each site physico-chemical properties (salinity, temperature, pH, dissolved oxygen) were measured in surface water. Depth profiles of physico-chemical properties were also measured at 1 site in the lower middle and upper estuary. Samples for nutrients, total suspended solids, colour, and chlorophyll-a were collected from a depth of 20cm at each site. Dissolved organic and inorganic nutrient samples were filtered immediately

[1] While high tide sampling was identified as important at the start of the project, it was not possible to maintain this practice throughout the project due to staff limitations.

through a 0.45µm cellulose acetate filter, and all samples were stored on ice until return to the laboratory for analysis.

3.3 Laboratory Analysis

Nutrient, chlorophyll and TSS analyses were undertaken by the TSC laboratory within 3 days of sample collection. Duplicate samples were sent to the DECC laboratory in Lidcombe, NSW for nutrient and colour analysis during the first 6 months of the monitoring period.

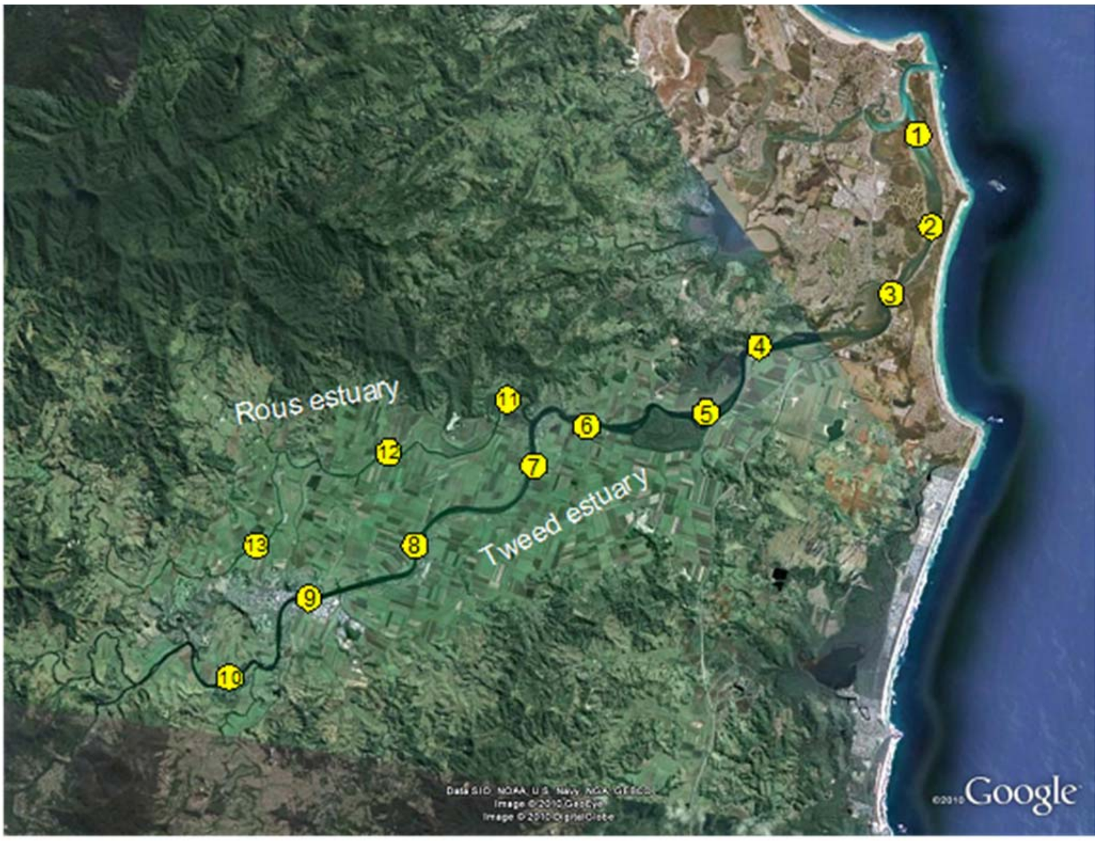


Figure 4 Location of sample sites along the Tweed and Rous estuaries.

4 Climate

4.1 Rainfall

The study period was characterised by a range of below and above average rainfall years, as shown in Figure 5. The study period commenced in 2007 which experienced generally well below average rainfall for the bulk of the year until December which experienced some large storm events resulting in an above average monthly total. This ushered in an above average rainfall summer in 2008, followed by a below average autumn, a relatively wet winter and a dry spring. 2009 experienced well above average rainfall for the bulk of the year, while 2010 and the first part of 2011 experienced generally average rainfall. The rainfall patterns over this succession of years highlights the inter-annual variability of rainfall in the northern rivers which is a primary control over water quality in the Tweed estuary and Cobaki-Terranora broadwaters.

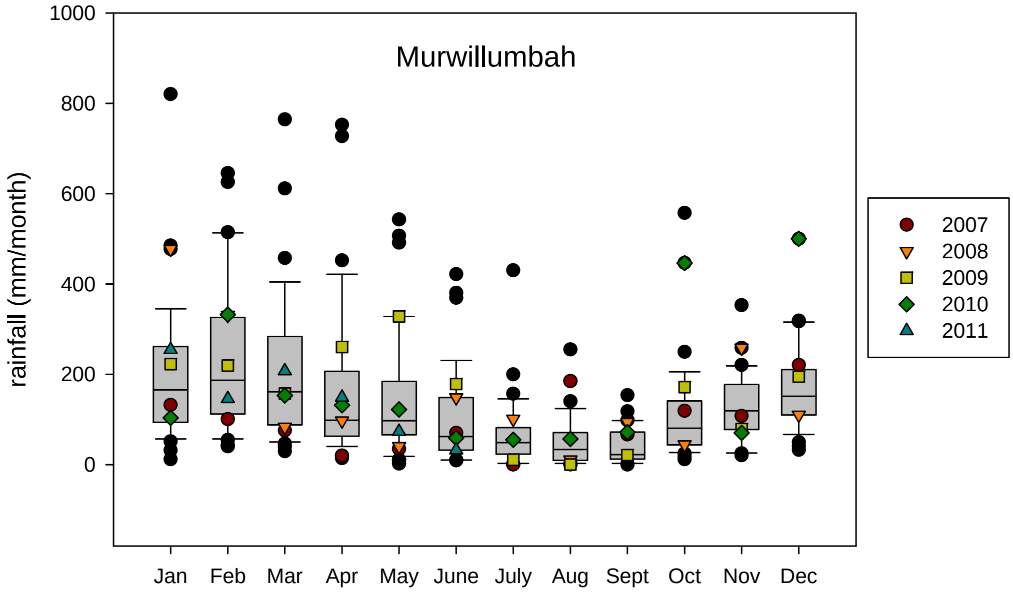


Figure 5 Rainfall statistics (boxplots) of rainfall at Murwillumbah from 1972 to present, with monthly totals for each year of the study period included as dots.

4.2 Freshwater discharges

4.2.1 Event-based and seasonal patterns

Modelled freshwater discharges to the Tweed estuary display distinct short period spikes (days to weeks) relating to runoff events, and longer period oscillations (annual) relating to wet and dry seasons (Figure 6). The salinity structure can be impacted by large flood events which can flush the estuary fresh to the mouth, but is generally more closely related to seasonal oscillations in freshwater discharge. As catchment flows drop below the low flow threshold, brackish conditions penetrate to upstream of Murwillumbah and the upper Rous. In contrast, during prolonged high flow conditions in the wet season, salinities become depressed into the lower estuary and freshwater conditions prevail throughout the upper estuary and into the middle estuary (downstream of Murwillumbah to the confluence with the Rous estuary).

4.2.2 Timing of sample collection

While the results of sample efforts can be influenced by recent runoff events (e.g. floods), it is clear that the cumulative discharge over the preceding months is a more important influence. A more important consideration is the timing of sampling relative to the semi diurnal tidal cycle, which can dramatically affect results of a geographically fixed site sampling strategy. Figure 7 shows that salinity at a fixed site within the Tweed estuary can vary by more than 10 PSU over a single tidal cycle. As such, the results of the current study are subject to considerable error since the sampling time did not consider the state of tide. As time of sample collection was not recorded, it is not possible to correct for this error.

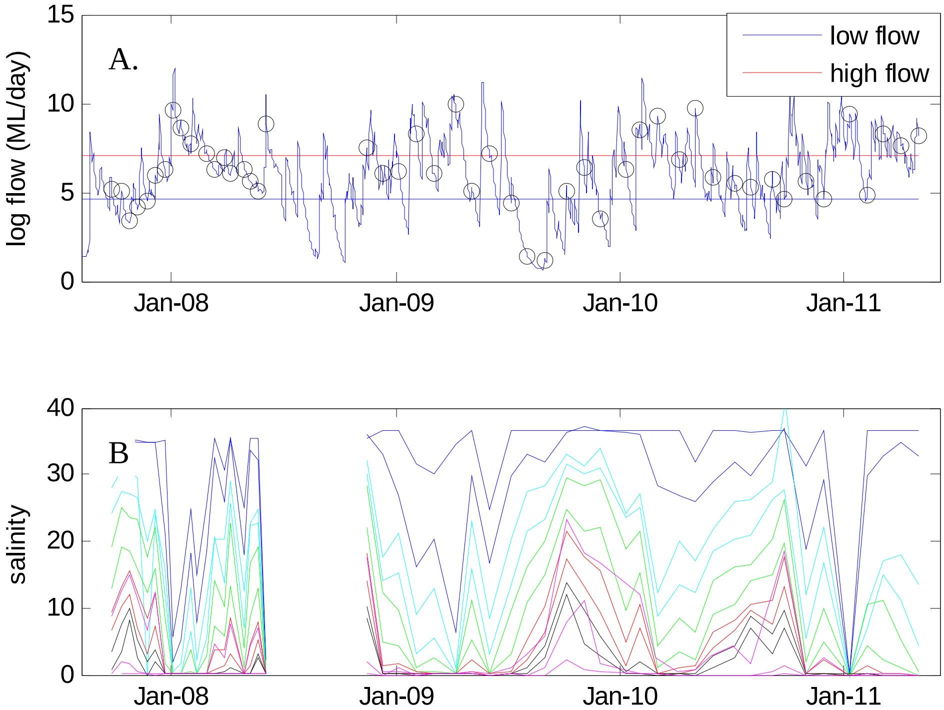


Figure 6 A. Total freshwater discharge (note log scale) to the Tweed estuary during the study period showing timing of sample runs. The thresholds of low and high flow conditions (10th and 90th percentile flows) are indicated. **B.** Salinity of the Tweed estuary over the study period.

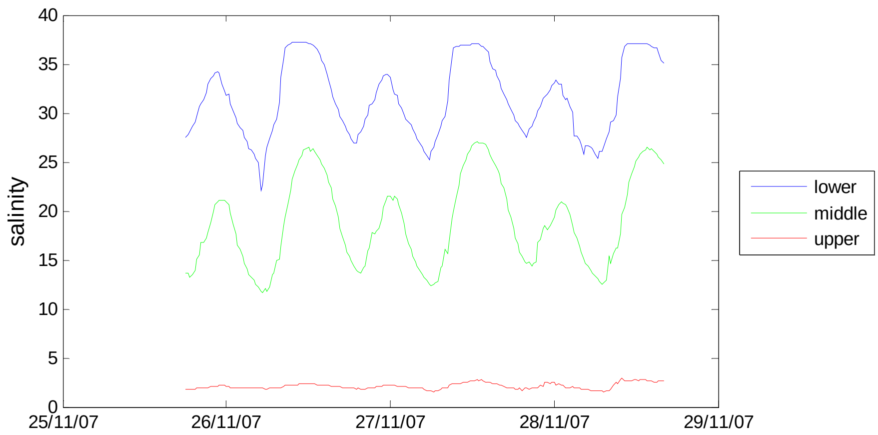


Figure 7 Salinity variation over three days at three locations in the Tweed estuary (lower = downstream of Pacific Highway bridge; middle = downstream of Stotts Island; upper = upstream of Murwillumbah). Unpublished data from ABER/DECCW.

5 Water quality results

5.1 Temperature

5.1.1 Background

Temperature provides an important ancillary measure used in the calculation of salinity and dissolved oxygen saturation. Temperature can be an important factor in causing vertical stratification of water bodies, especially poorly flushed coastal lakes. Temperature can impact on biogeochemical processes (e.g. the rate of organic matter breakdown in sediments), and also limit ecological processes and diversity.

5.1.2 Spatial trends

Spatial trends in temperature were highly influenced by flow conditions at the time of sampling (Figure 8). During low flow, temperature generally increased slightly moving up the estuary, most likely reflecting a combination of solar heating and increasing residence times progressing up the estuary. This spatial gradient decreased progressively during median and high flow conditions, reflecting the homogenisation of water throughout the estuary and greatly reduced residence times as freshwater flow increases.

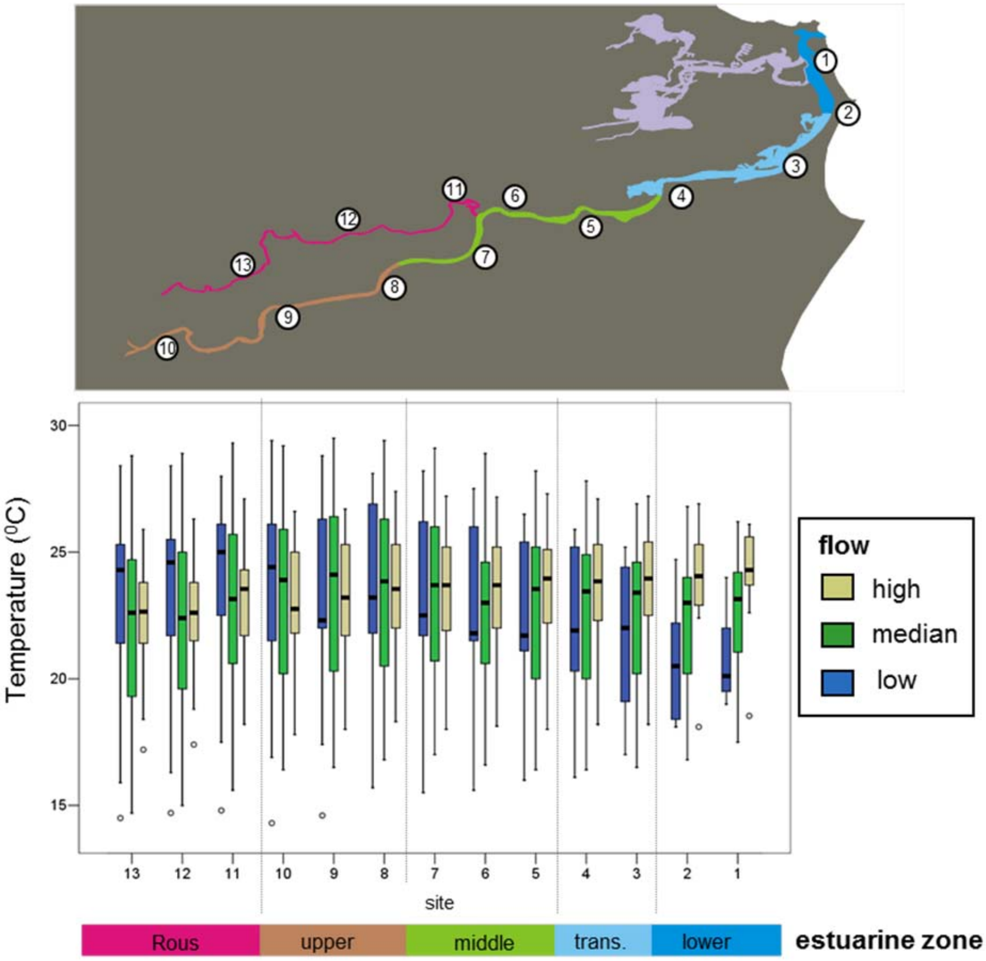


Figure 8 Spatial variation in temperature throughout the Tweed estuary during low, median and high flow conditions.

5.1.3 Temporal trends

There was a strong seasonal pattern to temperature with mid summer maxima (up to 29^{0}C) and mid winter minima (down to 14^{0}C; Figure 9). The mid summer maxima in the upper to middle estuary were highly influenced by flow conditions, with higher temperatures experienced during low flow conditions due to solar heating and long residence times. Similarly, winter minima were also influenced by flow conditions with lower temperatures experienced during low flow conditions. This effect is due to the moderating impact of freshwater inflows on temperature during this season. The seasonal temperature oscillation at the lower estuary sites was considerably less than the upper estuary sites, and the summer maxima lagged by up to 2 months. This is due to the overriding influence of ocean water.

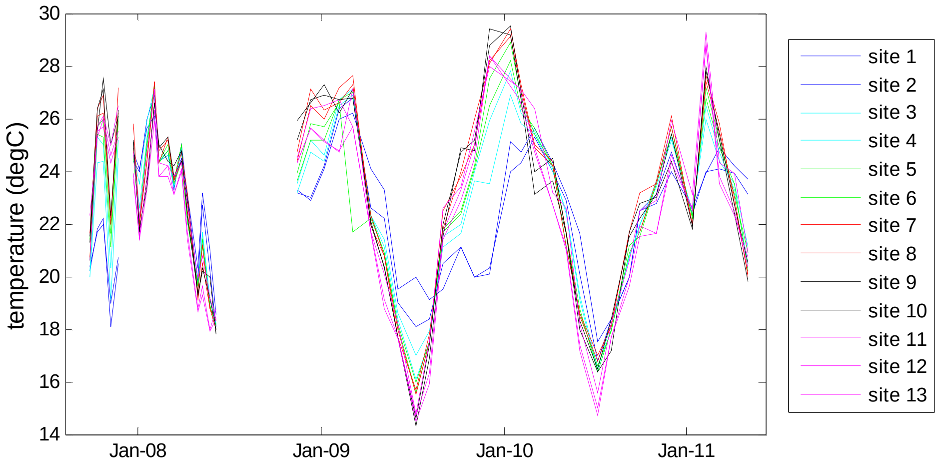


Figure 9 Temporal variation in temperature in the Tweed estuary.

5.1.4 Inter-annual variation

There was significant inter-annual variation in temperature during the summer-autumn months (Figure 10), most likely reflecting the influence of variable flow conditions between the years. Inter-annual variation became insignificant during the winter-spring months due to lesser influence from freshwater inflow. Inter-annual temperature variation also occurs in oceanic water due to the variable strength of upwelling in the coastal zone during the spring dry season which brings cooler oceanic water into the lower estuary (Davies 2005).

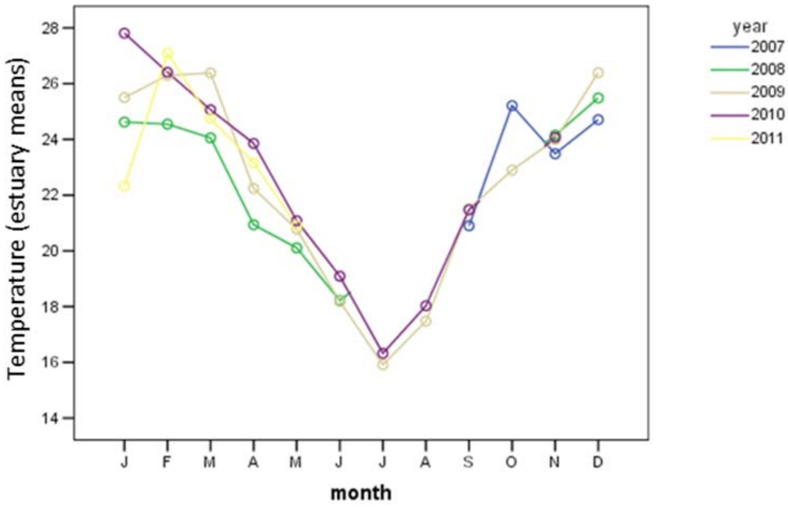


Figure 10 Inter-annual variation in mean estuary temperature over the study period.

5.2 Salinity

5.2.1 Background

Salinity is a measure of dissolved salts in water, which can be used as a conservative tracer (i.e. a tracer whose concentration remains unchanged by any processes within the estuary). Salinity thereby provides a measure of the relative proportions of oceanic water and freshwater at any point along the estuary. The vertical salinity profile (stratification) also determines a major aspect of the mixing structure of the estuary, ranging from well mixed (no vertical stratification) to highly stratified (surface freshwater underlain by marine water). Salinity is a major driver of estuarine ecology.

5.2.2 Spatial trends

Salinity decreased predictably moving upstream from the estuary mouth to the upper estuary during all flow conditions (Figure 11). The highest salinities were consistently recorded during low flow conditions when tidal influence was greatest. There was considerable overlap in salinities under low and median flow conditions, while high flow salinities were significantly lower. The overlap between low and median flow conditions most likely arises from 1) the arbitrary threshold imposed to delineate these categories (i.e. some sample runs may have fallen very close to either side of the cutoff), and 2) the error introduced by variable state of tide at the time of sampling.

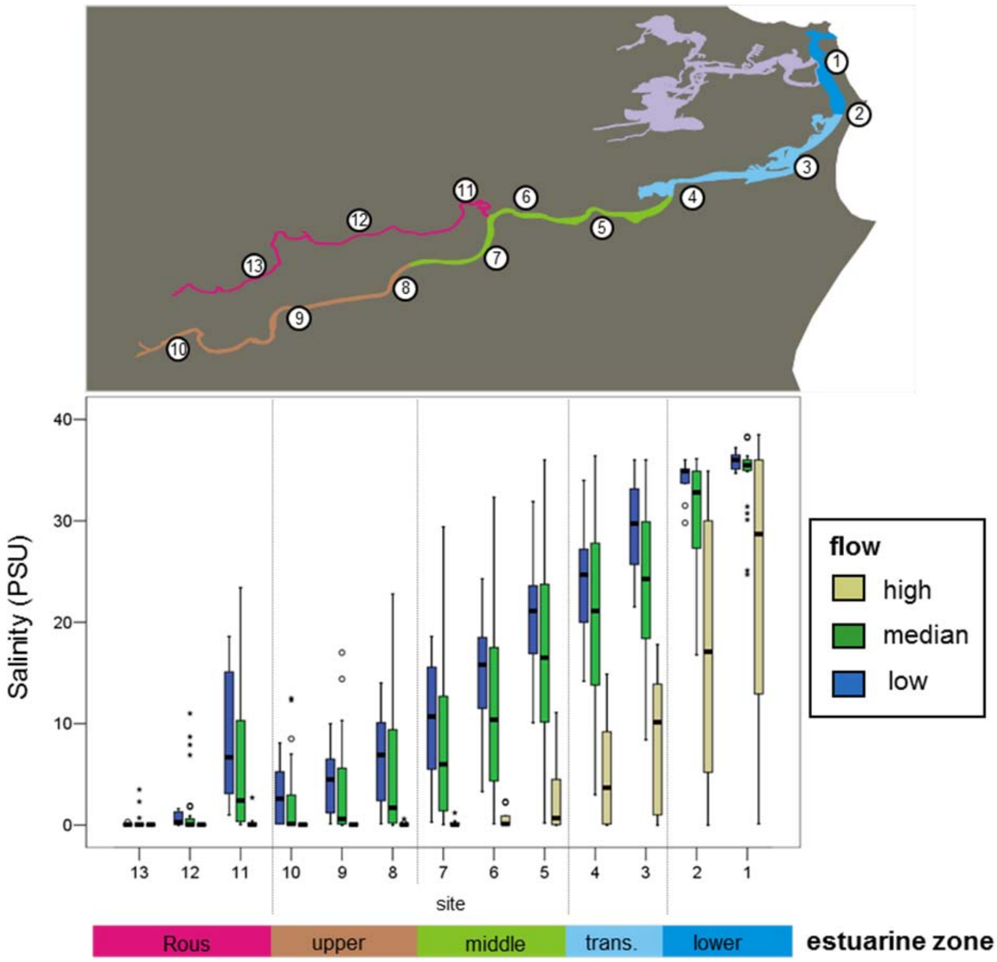


Figure 11 Spatial variation in salinity throughout the Tweed estuary.

5.2.3 Temporal trends

Seasonal trends in salinity are less clear than those apparent for temperature, due to the variable nature in the timing, duration and magnitude of freshwater runoff events (Figure 12). Freshwater influence increases greatly throughout the estuary during the summer-autumn wet season. This can occur rapidly in response to a flood event (e.g. Jan 2008), or progressively with the onset of more frequent but smaller runoff events (e.g. the 2009 wet season). Recovery of brackish estuarine conditions occurs as the frequency and severity of runoff events diminishes into the winter-spring dry season. The highest salinities were generally recorded during the spring-early summer months.

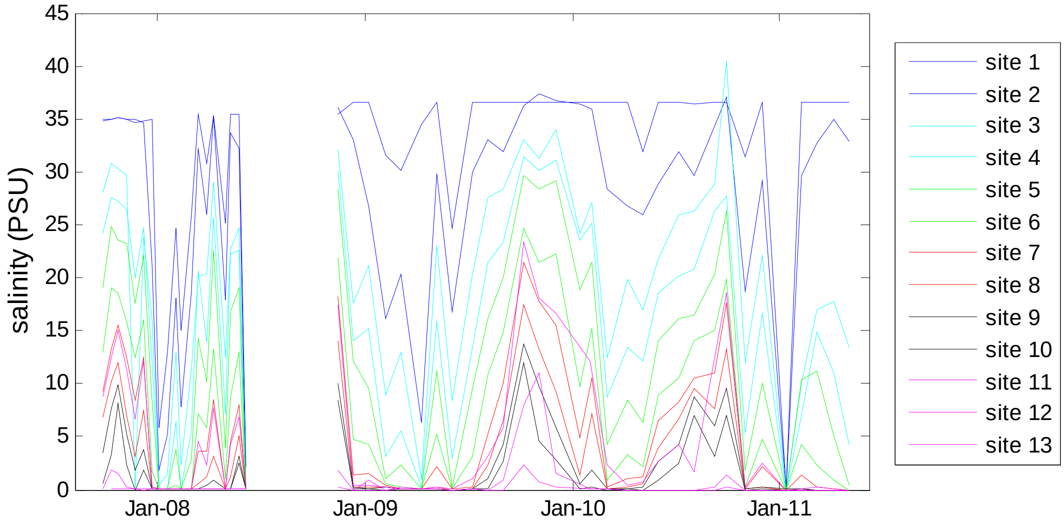


Figure 12 Temporal variation in salinity in the Tweed estuary during the study period.

5.2.4 Inter-annual variation

All years during the study period displayed the seasonal progression between lower salinities during the summer-autumn months and higher salinities during the late spring-early summer months (Figure 13). The greatest inter-annual variability occurs during the summer, reflecting variation in the onset of the wet season. High variability also occurs in the early winter months reflecting variability in the duration of the wet season.

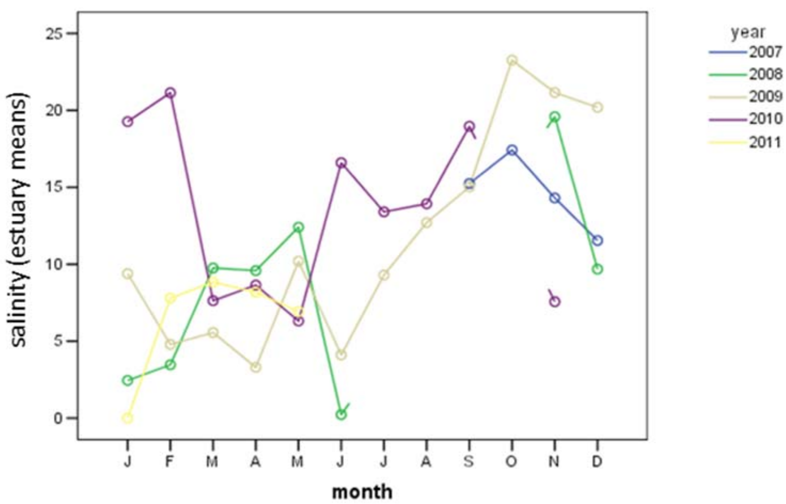


Figure 13 Inter-annual variation in mean estuary salinity over the study period.

5.2.5 Ecological implications

Salinity is a major forcing factor for estuarine ecology, controlling species distributions of phytoplankton, zooplankton benthic invertebrates and fish. As such, salinity determines the seasonally dynamic boundaries of ecological zones along the estuary. Salinity also controls geochemical processes such as flocculation and settling of suspended solids, which in turn controls light attenuation.

5.2.6 Management implications

The management and release of freshwater from the weir above Murwillumbah is the major anthropogenic pressure on salinity in the Tweed estuary. The weir clips low to median freshwater discharges to the upper estuary thereby significantly increasing salinity of this zone. This has implications for phytoplankton blooms in the upper estuary, as well as migratory pathways for fish species dependent on low salinity to freshwater conditions (e.g. bass). Releases of freshwater effluent from STPs have very minor impacts on salinity in the Tweed estuary.

5.3 pH

5.3.1 Background

pH is a measure of how acid or alkaline a water body is. It approximates p[H], the negative logarithm (base 10) of the molar concentration of dissolved hydronium ions (H3O+); a low pH indicates a high concentration of hydronium ions, while a high pH indicates a low concentration. In most natural waters pH will vary between 6.5 in freshwater and 8.2 in oceanic water. Sources of acid water in coastal systems include humic-rich groundwater (pH~ 4.5) and acid sulfate soil runoff (pH ~ 2.5 – 3). pH can also vary in response to biological processes, driven lower by organic matter breakdown and higher by photosynthesis. Low pH (e.g. in association with acid sulfate runoff) can cause physiological stress to estuarine biota in itself, however most stresses are associated with increased toxicity of many elements (e.g. aluminium) at low pH.

5.3.2 Spatial trends

pH generally decreased moving upstream from the estuary mouth reflecting the mixing of oceanic water (high pH ~8.18) and freshwater (lower pH ~ 6.6) along the estuarine gradient (Figure 14). During low flow conditions when freshwater inputs to the estuary are minimal and residence times are long, it is likely that internal biological processes (i.e. the production and consumption the of organic matter) form the primary control over spatial trends in pH[2]. A common pattern recorded during low flow was reduced pH in the middle estuary coinciding with sags in dissolved oxygen. A slight increase in pH occurred at upper estuary sites during low flow which is most likely due to Slightly lower pH along the estuary during median flow conditions relative to low flow most likely reflects the increased influence of freshwater runoff. In contrast, the greatly reduced pH during high flow conditions most likely reflects the influx of acid sulfate soil runoff (pH ~ 3.5 – 4)[3]. Consistently lower pH measurements in the Rous estuary indicate a relatively greater influence of acid sulfate soil runoff in this tributary.

[2] pH is driven down in aquatic systems where the consumption of organic matter by bacteria exceeds the production by photosynthesis.
[3] Note that spurious pH measurements collected during the beginning of 2008 have not been omitted from the dataset for this analysis.

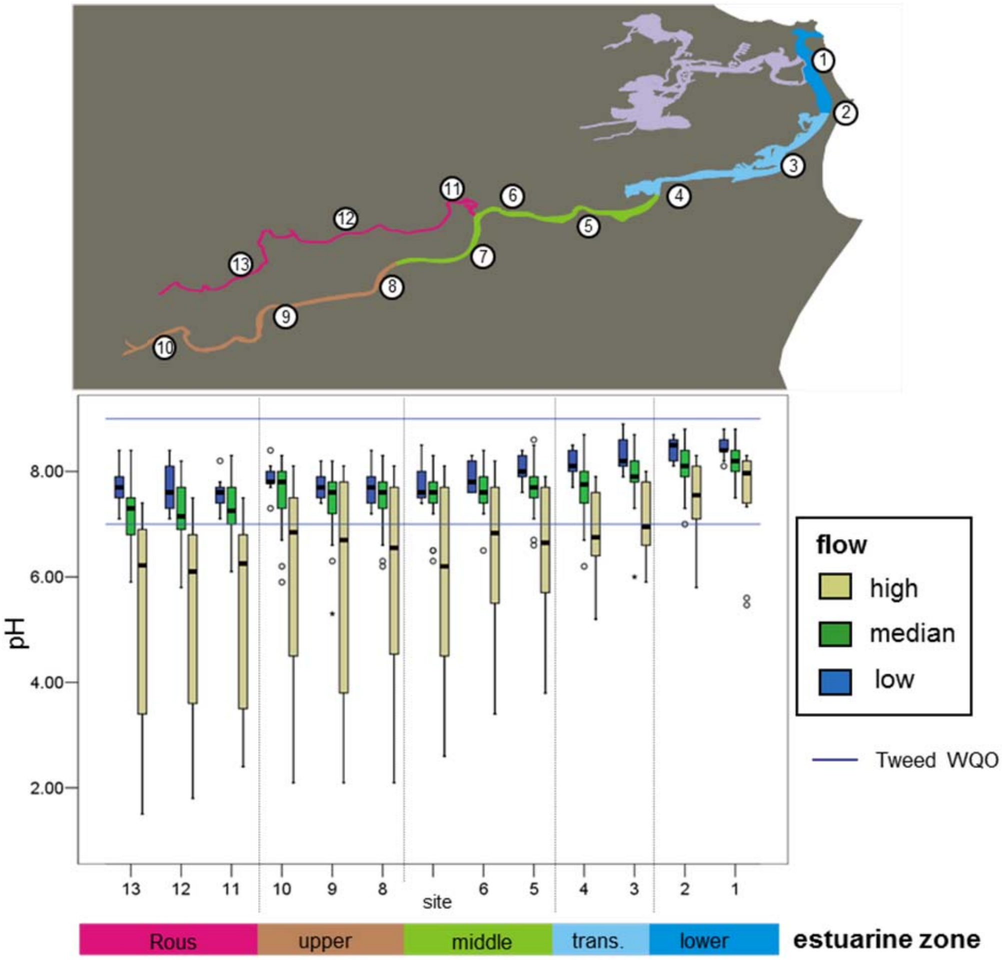


Figure 14 Spatial variation in pH throughout the Tweed.

5.3.3 Temporal trends

Temporal trends in pH were primarily driven by the relative importance of freshwater inflow and acid sulfate soil runoff, with a secondary influence due to biological processes during low flow conditions. Excluding the data from early 2008, there evidence of one event where acid sulfate soil runoff may have had a primary influence on the estuary as a whole (April 2009).

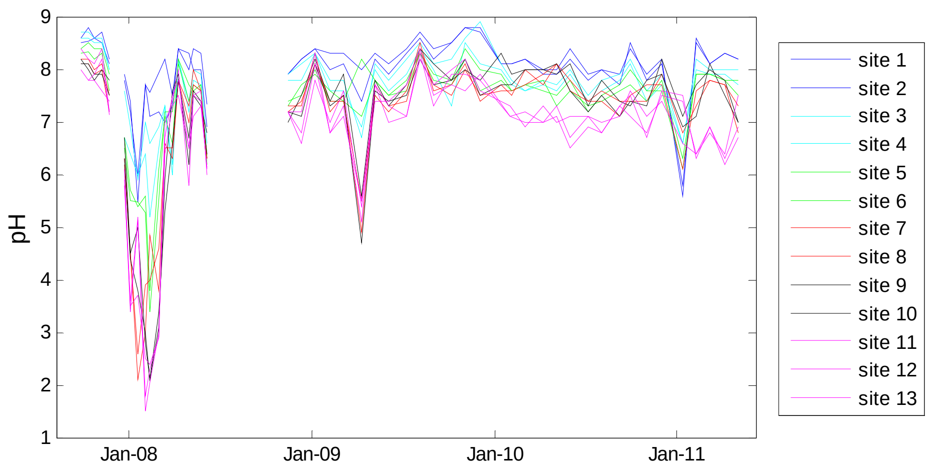


Figure 15 Temporal variation in pH in the Tweed estuary during the study period.

5.3.4 Inter-annual variation

Due to the uncertain value of pH data collected during late 2007 and early 2008 it is difficult to analyse inter-annual variation. Excluding the spurious data, it would appear that inter-annual variation in pH is insignificant, with the exception of years where major acid sulphate soil runoff events occur.

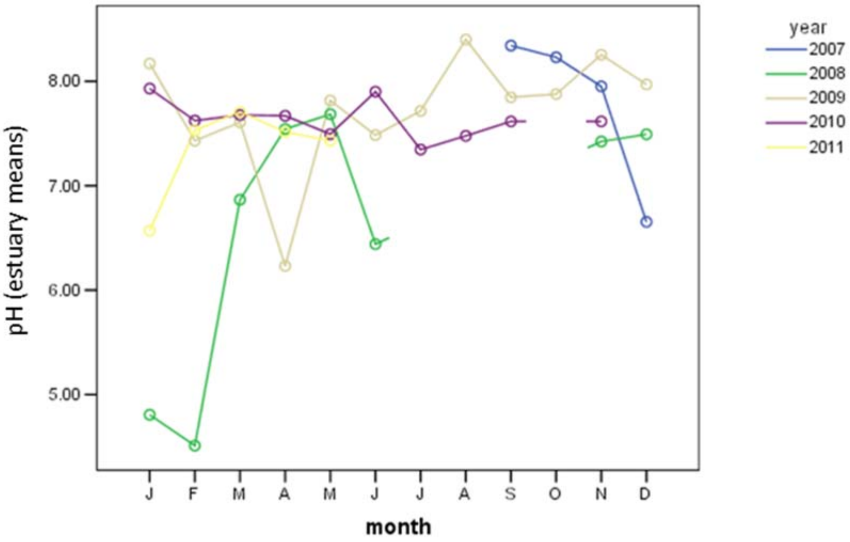


Figure 16 Inter-annual variation in mean estuary pH over the study period.

5.4 Total Nitrogen

5.4.1 Background

Total nitrogen and phosphorus represent the sum of dissolved inorganic, dissolved organic and particulate nutrients. Due to the rapid assimilation of dissolved inorganic forms (see section 5.6 below), total nutrients are generally dominated by particulate forms (e.g. phytoplankton cells) and dissolved organic forms. Nitrogen is commonly regarded as the limiting nutrient for primary production in estuarine ecosystems (Doering, Oviatt et al. 1995). Total nutrient concentrations provide a useful measure of the trophic status of an aquatic ecosystem, however concentrations should be augmented with information on other nitrogen forms.

5.4.2 Spatial trends

Total nitrogen (TN) concentrations ranged between <50μg.L-1 to 1240 μg L^{-1} during the study period. There was a consistent trend of low TN concentrations in the lower estuary increasing to peak in the middle to upper estuary and diminishing towards the top of the estuary (Figure 17). TN concentrations were significantly higher during high flow conditions ($p<0.05$), followed by low flow and median flow conditions.

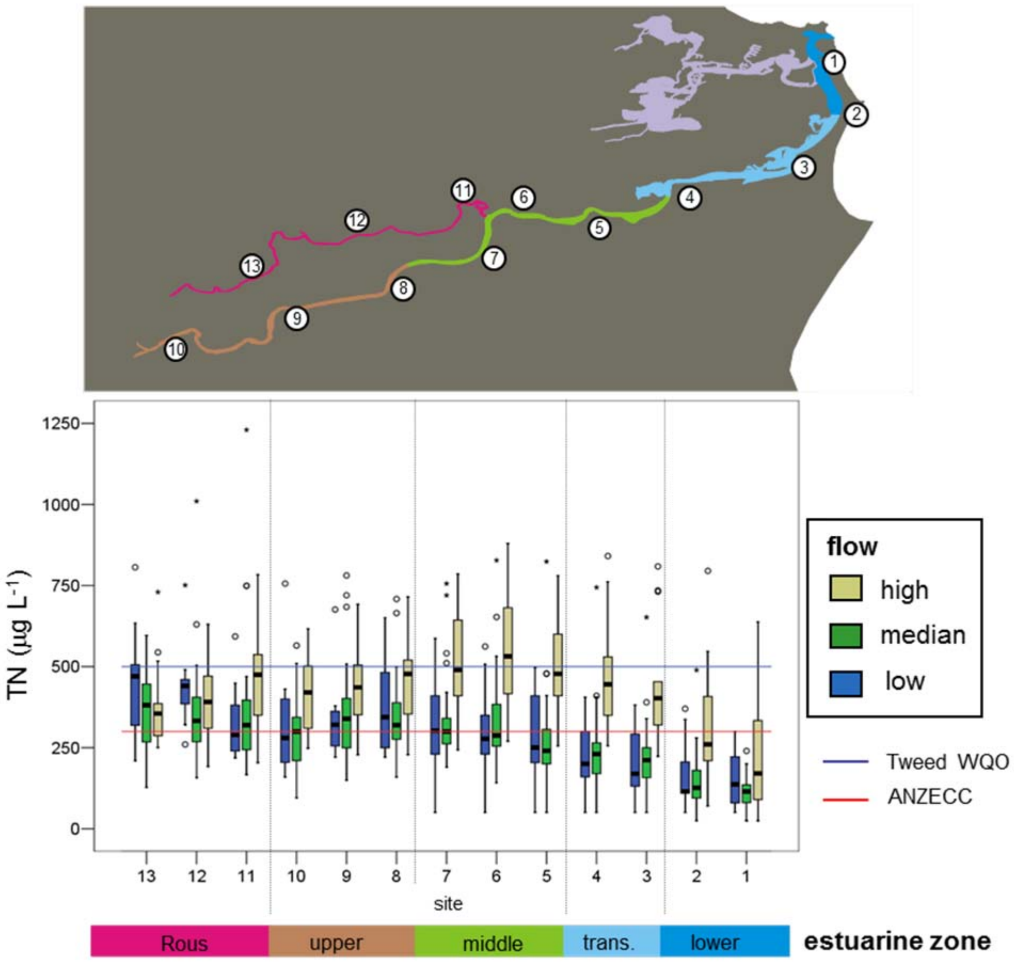


Figure 17 Spatial variation in total nitrogen concentrations throughout the Tweed estuary.

5.4.3 Temporal variation

There were no clear temporal trend in TN concentrations during study period (Figure 18). Higher concentrations during high flow times resulted in a weak seasonal trend of elevated concentrations during the summer – autumn wet season. Using the upper most sites in each estuary arm as proxies for freshwater input concentrations (site 10 in the Tweed and site 13 in the Rous), it can be seen that there is a weak positive relationship between temperature and TN concentration (Figure 19). This indicates that variation in TN concentrations most likely arises from the timing of high flow events relative to the seasonal temperature cycle. It is also likely that temporal trends are obscured by continual inputs from STPs along the estuary throughout the year.

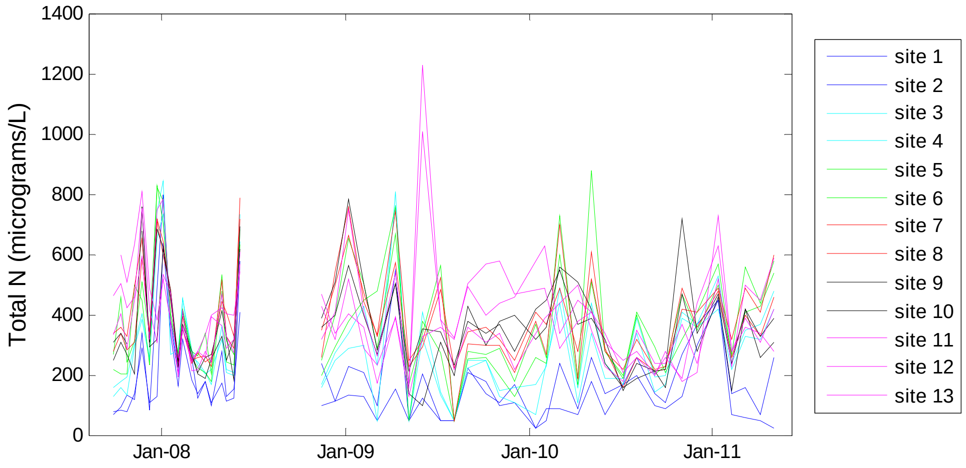


Figure 18 Temporal variation in TN concentrations in the Tweed estuary.

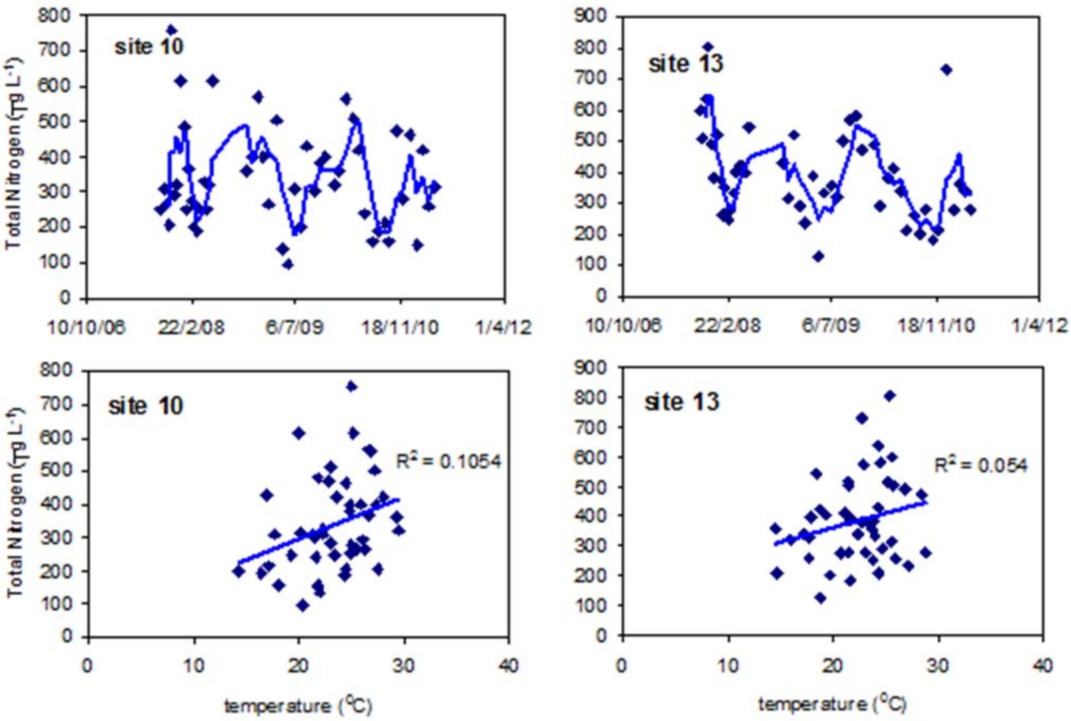


Figure 19 Temporal variation in TN concentrations at sites 10 and 13. The bottom plots show the temperature dependence of TN in freshwater runoff at these sites.

5.4.4 Inter-annual variability

There was inter-annual variability in TN concentrations during the study period caused by highly variable concentrations through the summer – autumn wet season (Figure 20). This variation is primarily caused by interaction between the timing and magnitude of freshwater runoff events and the seasonal temperature cycle.

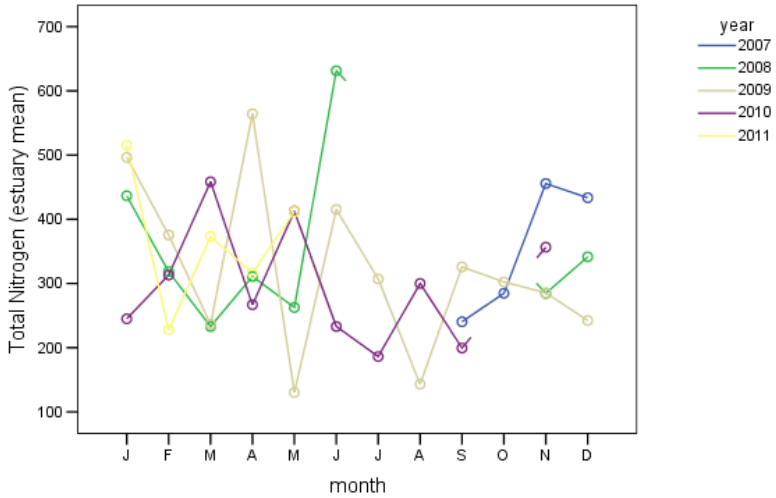


Figure 20 Inter-annual variability in mean estuary total nitrogen concentrations in the Tweed estuary during the study period.

5.4.5 Ecological implications

Total nitrogen concentrations were closely related to oxygen saturation in the Tweed estuary (see section 5.15), however the nature of this link is not entirely clear. It is likely that TN concentrations are in part controlled by STP inputs during low to median flow conditions (see section 6), therefore a significant portion of the TN pool represents bio-available and biologically bound N (i.e. phytoplankton biomass). The linkages between this material and oxygen saturation are discussed more fully in section 5.15.

5.4.6 Comparison with ANZECC guidelines

Input concentrations

Total nitrogen concentrations at the upper most estuary site (10) were above ANZECC (2000) guidelines for more than 75% of the time during high flow conditions indicating that the bulk of rural runoff does not comply with conditions for the maintenance of aquatic ecosystems in Southeast Australian estuaries (Figure 17). Site 10 complied with ANZECC guidelines during low to median flow conditions for 50% of the time.

Estuary concentrations

Total nitrogen concentrations exceeded the guideline thresholds for greater than 75% of the time during high flow conditions throughout the transition, middle and upper estuary sites. Compliance was better during low to median flow conditions, with only the upper estuary and Rous estuary sites exceeding thresholds for greater than 50% of the time.

5.5 Total Phosphorus

5.5.1 Background

Total phosphorus represents the sum of dissolved inorganic, dissolved organic and particulate nutrients. Phosphorus is not regarded as limiting primary production in estuaries (Thompson and Hosja 1996), however the role of TP limitation is some coastal systems is currently being debated (Eyre and McKee 2002; Glibert, Heil et al. 2006). In particular, phosphorus availability can control the occurrence of nitrogen-fixing organisms such as cyanobacteria which are commonly associated with toxic blooms (Karl, Michaels et al. 2002).

5.5.2 Spatial trends

Total phosphorus (TP) ranged between below detection (<5 μg L^{-1}) to 320 μg L^{-1} during the study period. There was a general trend of low concentrations in the lower estuary rising to a peak in the middle estuary and diminishing towards the top of the estuary (Figure 21). TP concentrations in the Rous estuary were consistently high throughout the study. There was a trend of increasing TP concentrations with flow, with high flow conditions having significantly higher concentrations than both median and low flow conditions. During low flow, TP concentrations were generally below detection limits in the lower and middle estuary.

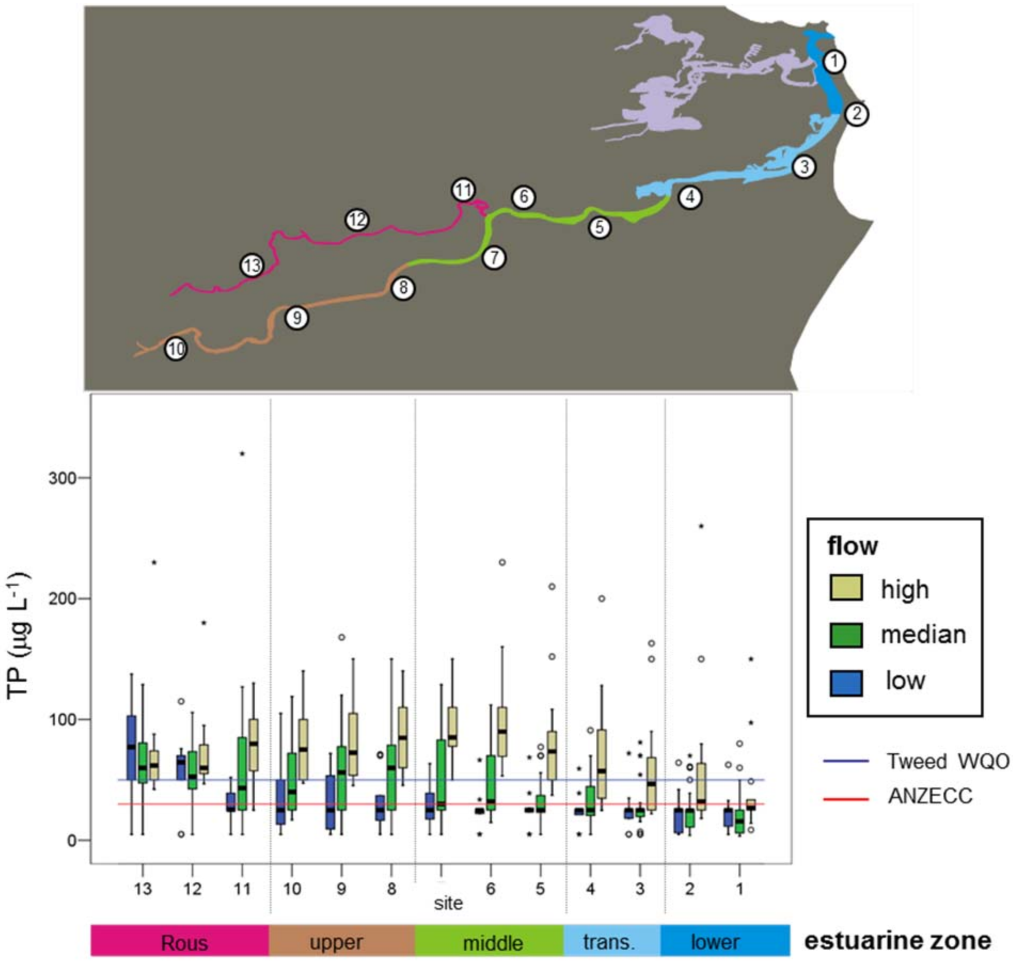


Figure 21 Spatial variation in total phosphorus concentrations throughout the Tweed estuary.

5.5.3 Temporal variability

There was a weak seasonal trend of elevated TP concentrations during the summer – autumn wet season, followed by trace concentrations during late winter – spring (Figure 22). Using the upper most sites in each estuary arm as proxies for freshwater input concentrations (site 10 in the Tweed and site 13 in the Rous), it can be seen that there is a weak positive relationship between temperature and TP concentration (Figure 23). This indicates that variation in TP concentrations most likely arises from the timing of high flow events relative to the seasonal temperature cycle.

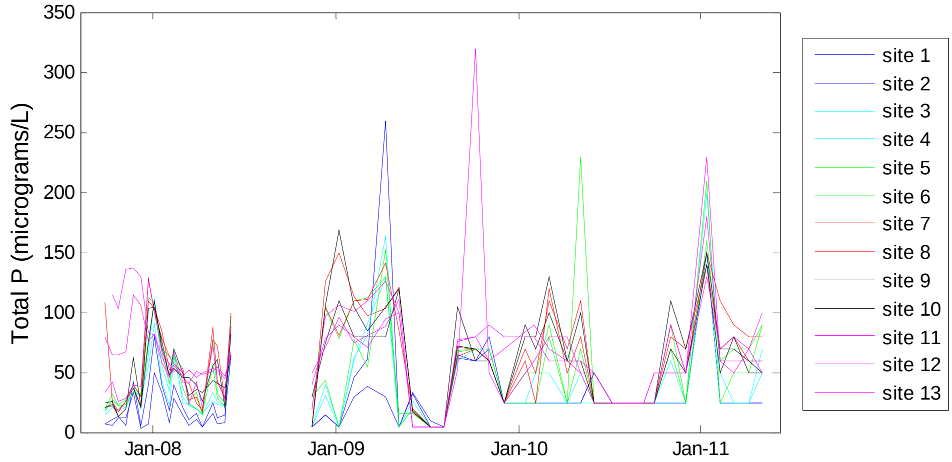


Figure 22 Temporal variation in TP concentrations in the Tweed estuary.

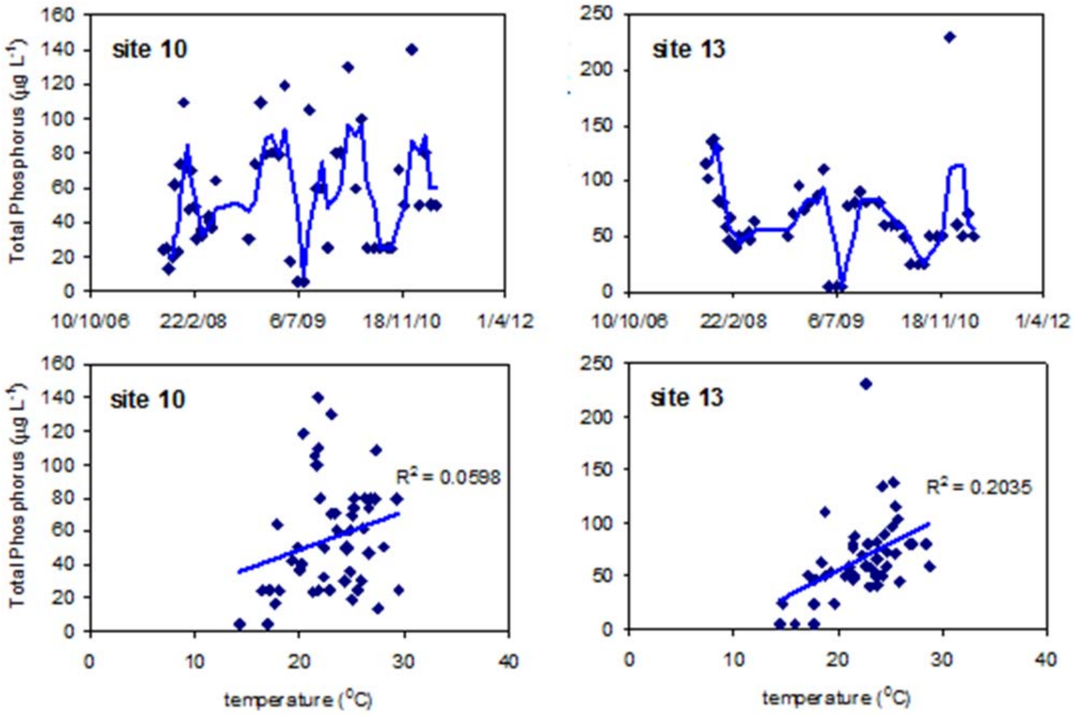


Figure 23 Variation in TP concentrations at sites 10 and 13. The bottom plots show the temperature dependence of TP concentrations in freshwater runoff.

5.5.4 Inter-annual variability

There was significant inter-annual variability in total phosphorus concentrations ($p<0.05$), with different years experiencing highly variable concentrations during the summer – autumn wet season and spring dry season (Figure 24).

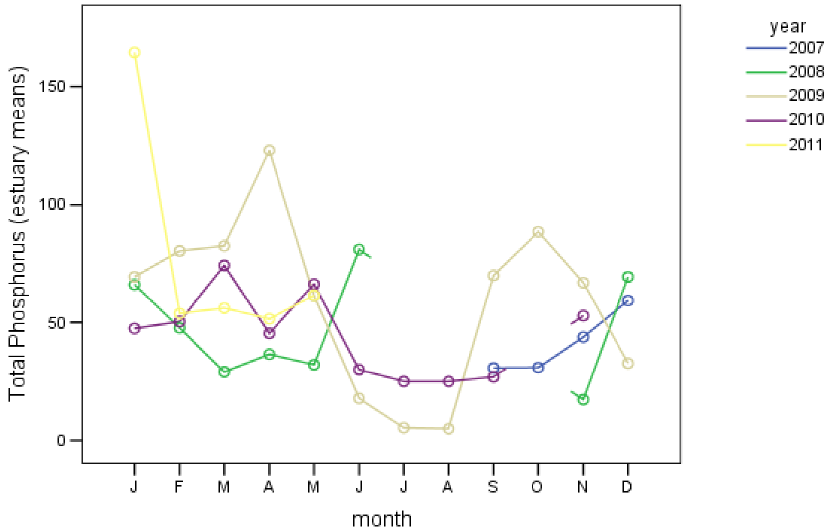


Figure 24 Inter-annual variability in mean estuary total phosphorus concentrations in the Tweed estuary during the study period.

5.5.5 Ecological implications

The true ecological implications of TP concentrations in the Tweed estuary are difficult to ascertain due to variable data quality throughout the monitoring period. This has greatly reduced the resolution of temporal and spatial trends, and in particular obscured valuable comparisons with TN (see section 9.5 for further analysis on this).

5.5.6 Comparison with ANZECC guidelines

Input concentrations

Total phosphorus concentrations at site 10 were more variable and exceeded guidelines more than 50% of the time. Total phosphorus concentrations at the upper most Rous estuary site (13) were consistently above guideline thresholds due to the influence of effluent from the Murwillumbah STP.

Estuary concentrations

Total phosphorus concentrations exceeded the guideline thresholds for greater than 75% of the time during high flow conditions throughout the entire estuary. The middle and upper estuary sites exceeded thresholds for greater than 50% of the time during median flow conditions. Compliance was better during low flow conditions, with only the upper estuary exceeding thresholds for greater than 50% of the time. Rous estuary sites generally exceeded thresholds for the bulk of the time during all flow conditions.

5.6 Ammonium

5.6.1 Background

Dissolved inorganic nitrogen (DIN = ammonium + nitrate + nitrite) and phosphorus (DIP) constitute the bio-available fraction of the total nutrient pool. DIN and DIP concentrations are generally low in pristine Australian estuaries reflecting 1) low concentrations in catchment runoff, and 2) high rates of biological uptake within the estuary. Due to a high bio-availability and rapid assimilation by aquatic plants, ambient concentrations of DIN and DIP rarely reflect input loadings.

5.6.2 Spatial trends

Ammonium (NH_4^+) concentrations varied between below detection (<5 µg L^{-1}) and 179 µg L^{-1} during the study period (Figure 25). There was a trend of low concentrations in the lower and upper estuary with a peak in concentrations in the middle estuary. The location of the mid estuary peak varied with flow: during high flow conditions the peak was located at site 3; during median flow the peak shifted upstream to site 4; during low flow the peak shifted further upstream to sites 5 and 6. There was a trend for increasing concentrations with flow.

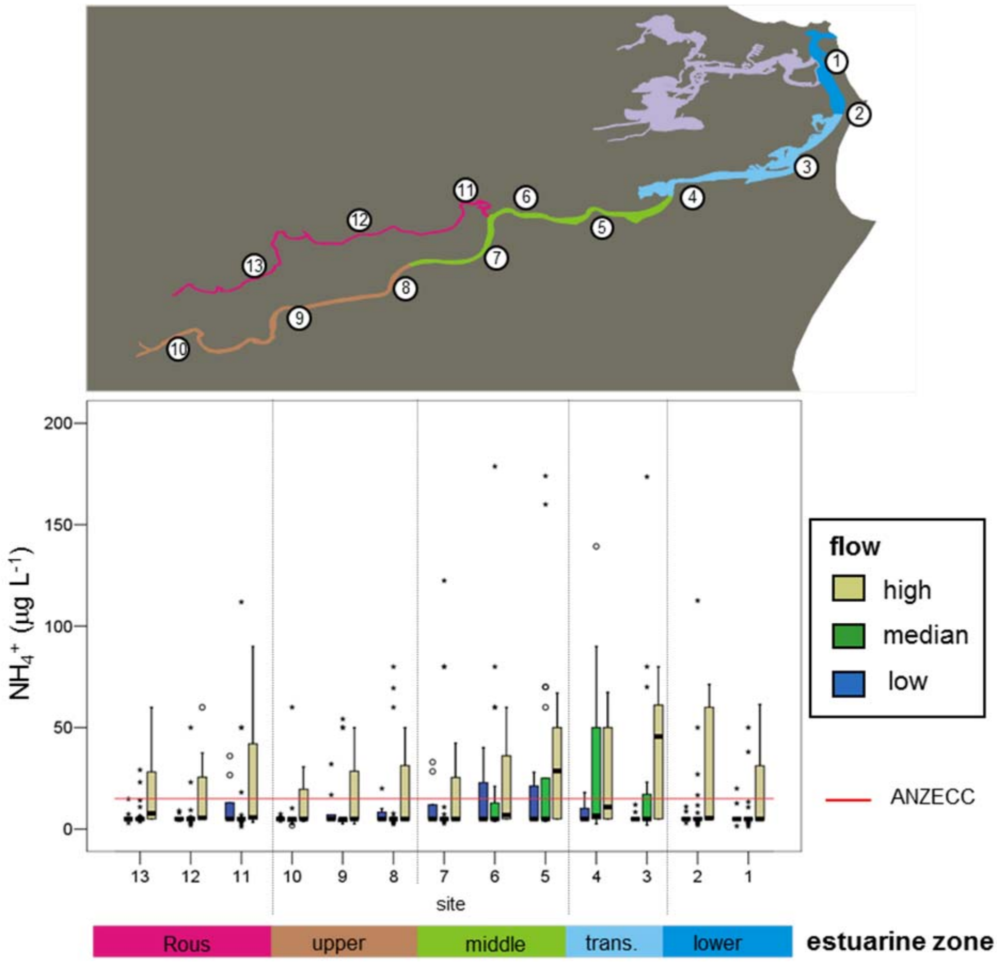


Figure 25 Spatial variation in ammonium concentrations throughout the Tweed estuary.

5.6.3 Temporal variability

Data quality for the bulk of the study period was poor, therefore analysis of temporal trends is not possible for the period 2009 to 2011. Based on the 2007 to 2008 data[4], it appears that ammonium concentrations are highest during the summer wet season and diminish as flows decrease into winter and spring.

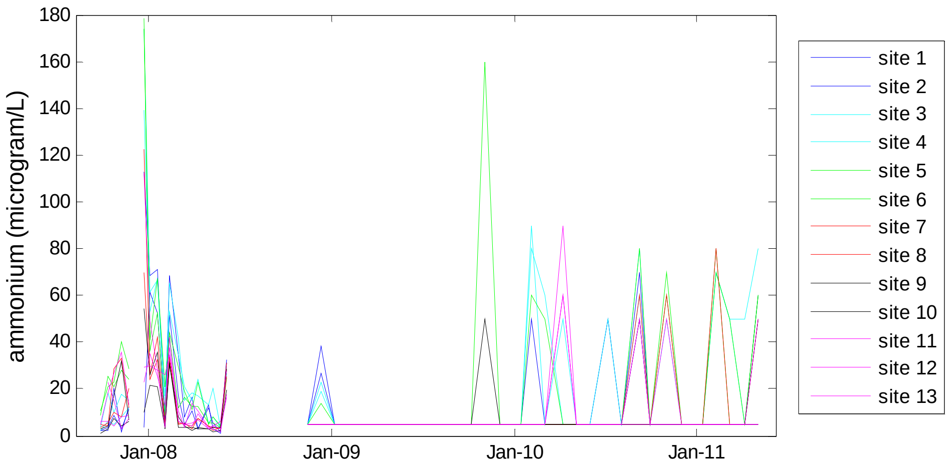


Figure 26 Temporal variation in ammonium concentrations in the Tweed estuary during the study period.

5.6.4 Inter-annual variability

Poor data quality makes an assessment of inter-annual variability not possible for ammonium.

5.6.5 Comparison with ANZECC guidelines

Input concentrations

Ammonium concentrations at site 10 were generally below ANZECC guideline thresholds, except during high flow when they exceeded the threshold for greater than 25% of the time.

Estuary concentrations

Ammonium concentrations exceeded the guideline thresholds for greater than 25% of the time during high flow conditions at the middle to lower estuary sites. Compliance was better during low to median flow conditions, with only sites 3 – 5 exceeding thresholds for greater than 25% of the time. Rous estuary sites generally exceeded thresholds for greater than 25% of the time during high flow conditions.

[4] Data from this initial period is from analysis carried out by NSW DECCW.

5.7 Oxidised Nitrogen

5.7.1 Spatial trends

Oxidised nitrogen (nitrate + nitrite) ranged between detection limits and 358 μg L^{-1} during the study period. As observed for ammonium, there was a middle estuary peak in oxidised nitrogen which migrated slightly upstream with reduced flow (Figure 27). The highest concentrations were consistently recorded during high flow conditions, with greatly reduced concentrations during median flow and trace concentrations limits during low flow conditions. Oxidised nitrogen was generally highest in the Rous estuary during all flow conditions.

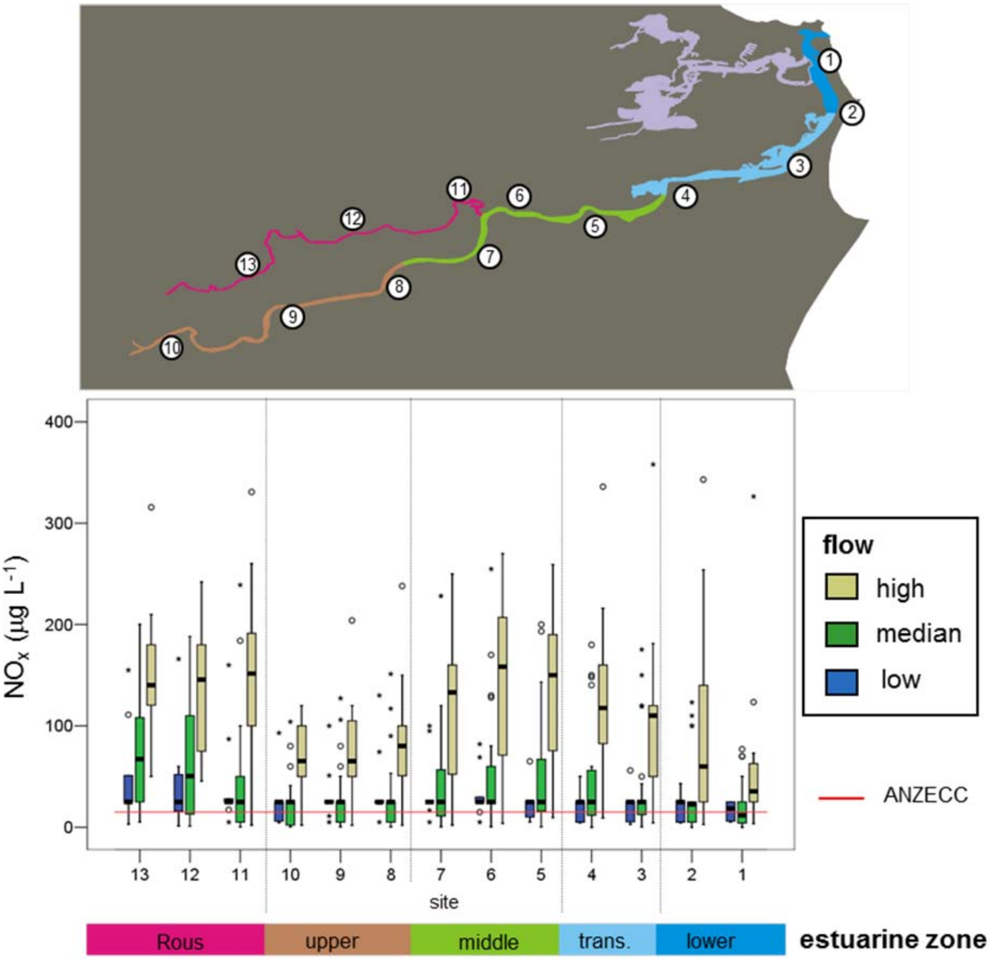


Figure 27 Spatial variation in oxidised nitrogen concentrations throughout the Tweed estuary.

5.7.2 Temporal variability

There was a weak seasonal trend of elevated oxidised nitrogen concentrations during the summer – autumn wet season most likely reflecting the higher concentrations prevalent during high flow (Figures 28 and 29). As with total nutrients there was a temperature dependency for the NO_x concentrations in freshwater runoff, with higher concentrations occurring during cooler months. Expressed as a percentage of total nitrogen, this temperature dependency is clearly evident at sites 10 and 13 (Figure 28). High variability on monthly timescales may also arise from uptake and release associated with the boom-bust cycle of phytoplankton blooms. It is also likely that high variability at this scale arises from poor analytical protocols.

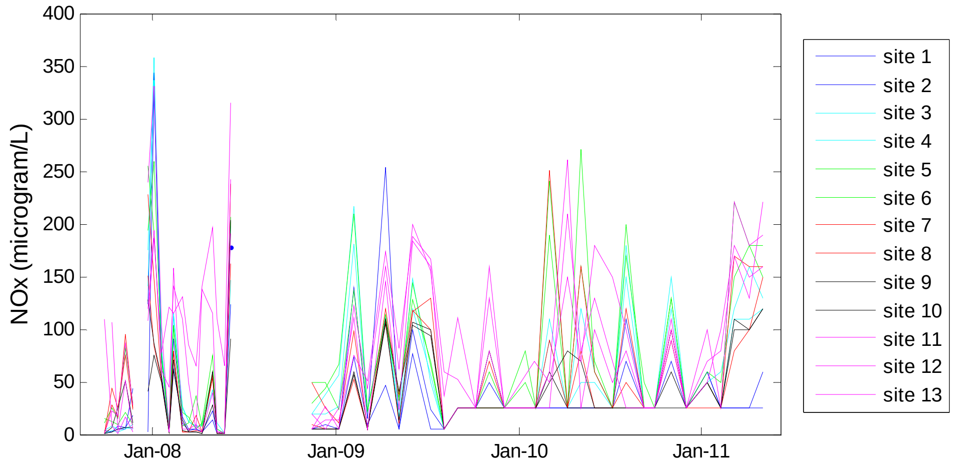


Figure 28 Temporal variation in oxidised nitrogen concentrations in the Tweed estuary during the study period.

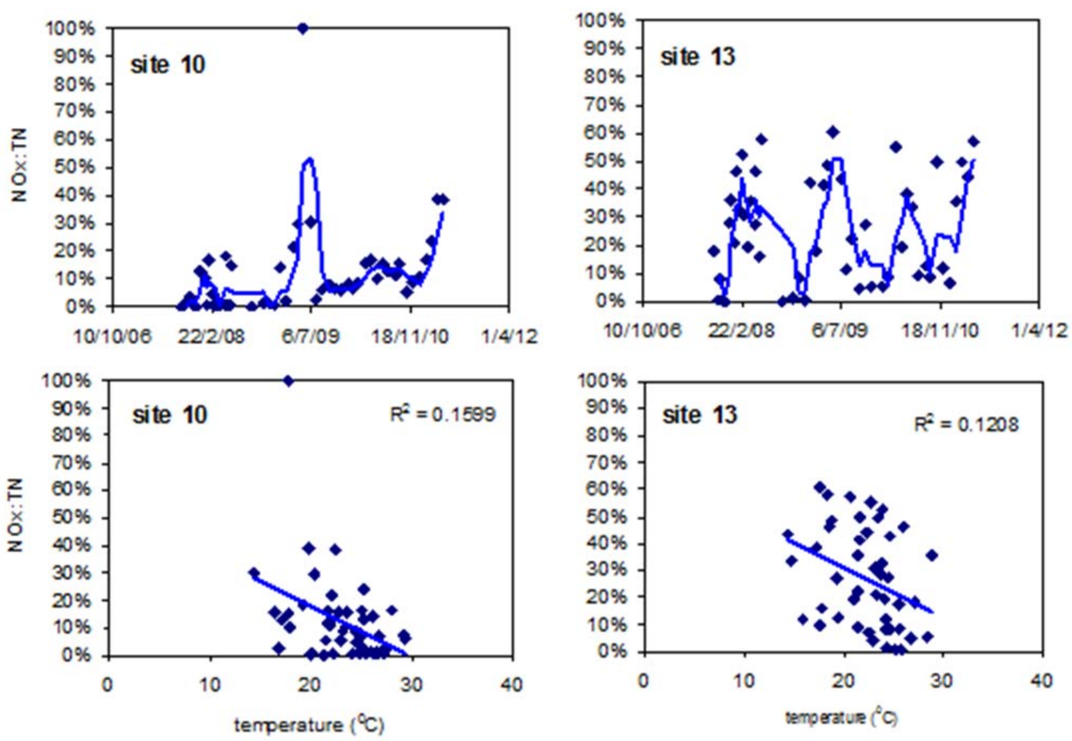


Figure 29 Variation in the relative importance of NOx to TN (expressed as a percentage) at sites 10 and 13. The bottom plots show the temperature dependence of the NOx:TN ratio.

5.7.3 Inter-annual variability

Poor data quality makes an assessment of inter-annual variability not possible for oxidised nitrogen.

5.7.4 Comparison with ANZECC guidelines

Input concentrations
Oxidised nitrogen (NOx) concentrations at site 10 were generally below ANZECC guideline thresholds, except during high flow when they exceeded the threshold for greater than 75% of the time.

Estuary concentrations
NO_x concentrations exceeded the guideline thresholds for greater than 75% of the time during high flow conditions and greater than 50% of the time during median flow conditions throughout the entire estuary. Compliance was better during flow conditions, with only sites 1 – 5 exceeding thresholds for greater than 25% of the time. Rous estuary sites generally exceeded thresholds for greater than 25% of the time during high flow conditions[5].

[5] Note that detection limits for NO_x were above the ANZECC guideline threshold for NO_x of 15µg L^{-1} therefore a true assessment of compliance was not possible for the current dataset.

5.8 Dissolved organic nitrogen

5.8.1 Background

Dissolved organic nitrogen (DON) commonly comprises the largest fraction of total nitrogen in Australian estuaries and rivers (Eyre 2000). DON itself is made up of a large array of different compounds which are analytically grouped and reported as a single pool. Many of these compounds are refractory (i.e. biologically unavailable), however there is increasing evidence to suggest that some DON compounds are very bio-available and may be important to the ecology of certain systems (Bronk, See et al. 2007). DON was only measured during the initial phase of this project when samples were analysed by NSW OEH.

5.8.2 Spatial trends

DON concentrations ranged from detection limits to 475 μg L^{-1} during the study period. Spatial trends were less distinct for DON than DIN, but were generally characterised by concentrations increasing towards the middle estuary. There was a significant increase in DON during high flow conditions, with a number of sample times characterised by highest DON concentrations adjacent to the confluence with the Rous estuary. This is consistent with DON inputs from low lying floodplain catchment along the middle estuary (see also section 6.4.5).

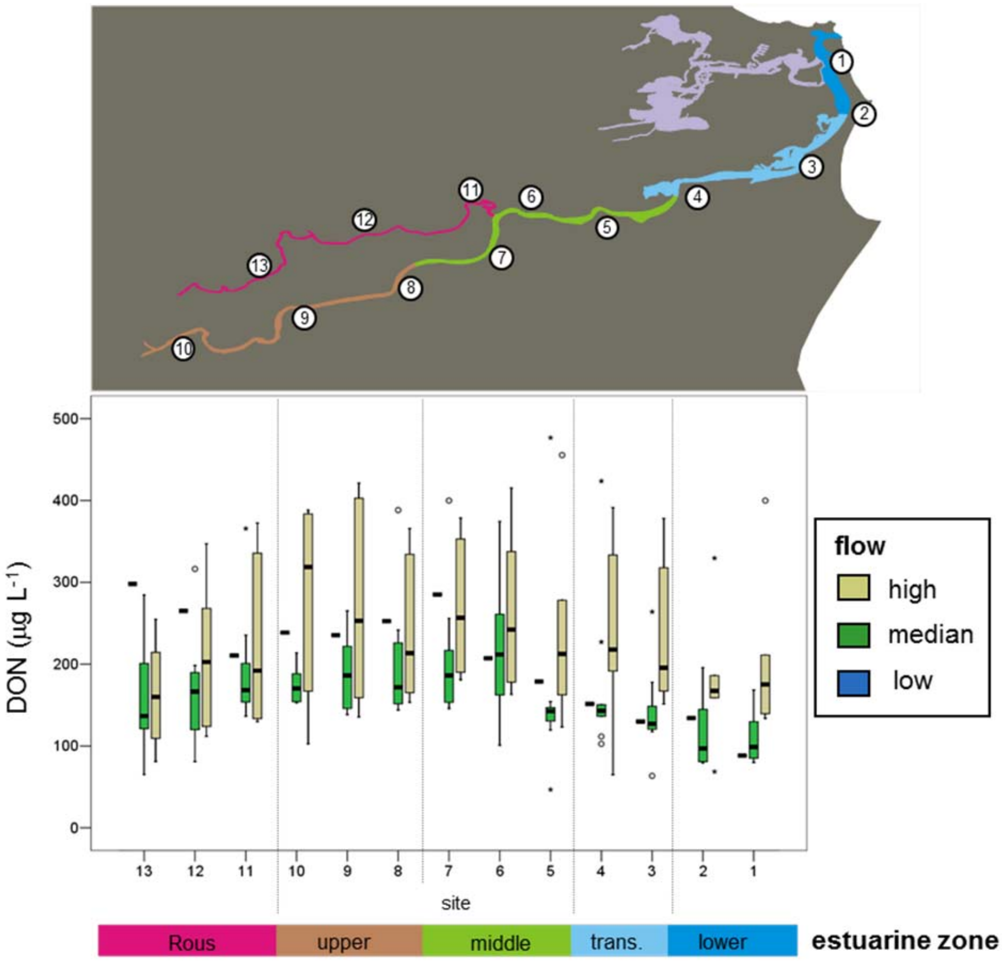


Figure 30 Spatial variation in dissolved organic nitrogen concentrations throughout the Tweed estuary

5.8.3 Temporal trends

The highest DON concentrations were observed during the first half of the study period and reduced significantly after February 2008 (Figure 31). Variability along the estuarine gradient was also reduced after this date. There was an increase in DON concentrations during the final high flow sample time in early June 2008. DON consistently accounted for between 40% and 80% of the total nitrogen pool.

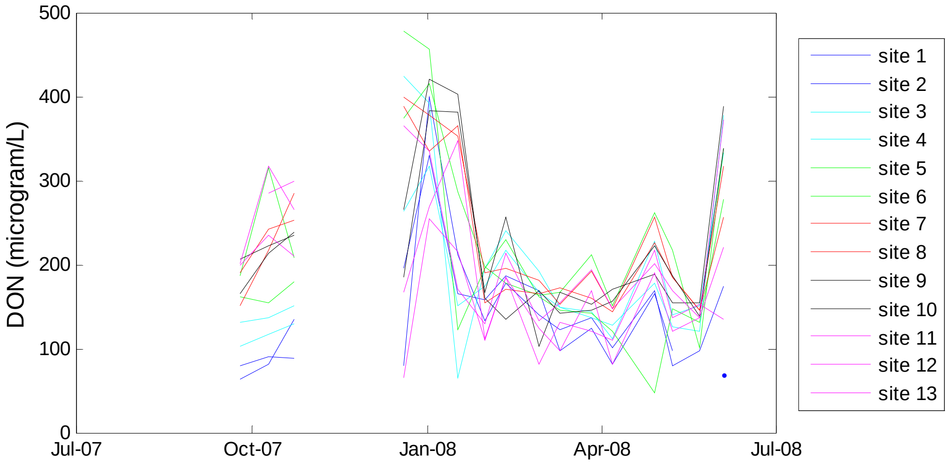


Figure 31 Temporal variation in dissolved organic nitrogen concentrations in the Tweed estuary during the study period.

5.8.4 Ecological implications

The coincidence of peak DON concentrations with high flow conditions along the middle estuary suggests inputs from low lying sub-catchments adjacent to this reach. The inundation of cane land and Melaleuca swamps can commonly result in high concentrations of DON in overland runoff and groundwater once these areas drain to the estuary. Other major sources of DON are likely to be associated with the release of dissolved organic substances from phytoplankton during low to median flow times. As such, the makeup of the DON pool is likely to vary spatially and temporally. The ecological implications of DON are currently unknown for the Tweed estuary, however it is likely that inputs from certain sources during certain times of the year are implicated in stimulating phytoplankton growth in the estuary. This topic is the subject of ongoing research within the NSW OEH.

5.8.5 Management implications

Due to the large contribution of DON to the TN pool, it imperative to understand the ecological significance of this material more fully. This will allow more targeted strategies to be developed for reducing bio-available nitrogen to the Tweed estuary. It is suggested that DON is measured in ongoing water quality monitoring efforts, and that managers continue to source updates of scientific research into the ecological significance of DON.

5.9 Dissolved inorganic phosphorus

5.9.1 Background

Dissolved inorganic phosphorus (DIP = phosphate) is the bio-available form of phosphorus in aquatic systems and is one of the primary nutrients controlling algae and macrophyte growth. The current paradigm is that phosphorus is not generally limiting to photosynthetic production in estuaries, however this is currently being questioned for a range of systems.

5.9.2 Spatial trends

DIP ranged between 0.7 and 65 μg L^{-1} during the study period. As with dissolved inorganic nitrogen, there were distinct spatial gradients in DIP concentrations (Figure 32), although these appeared to vary over time. In general, the highest DIP concentrations were recorded in the middle estuary, with concentrations during high flow conditions being significantly higher than other times. During the first three sample times, DIP concentrations tended to be highest between sites 3 and 6. After January 2008, there were commonly two concentration peaks along the estuarine gradient: one at or near site 3 and another peak in the upper estuary. Concentration tended to be consistently higher at site 10 than 13.

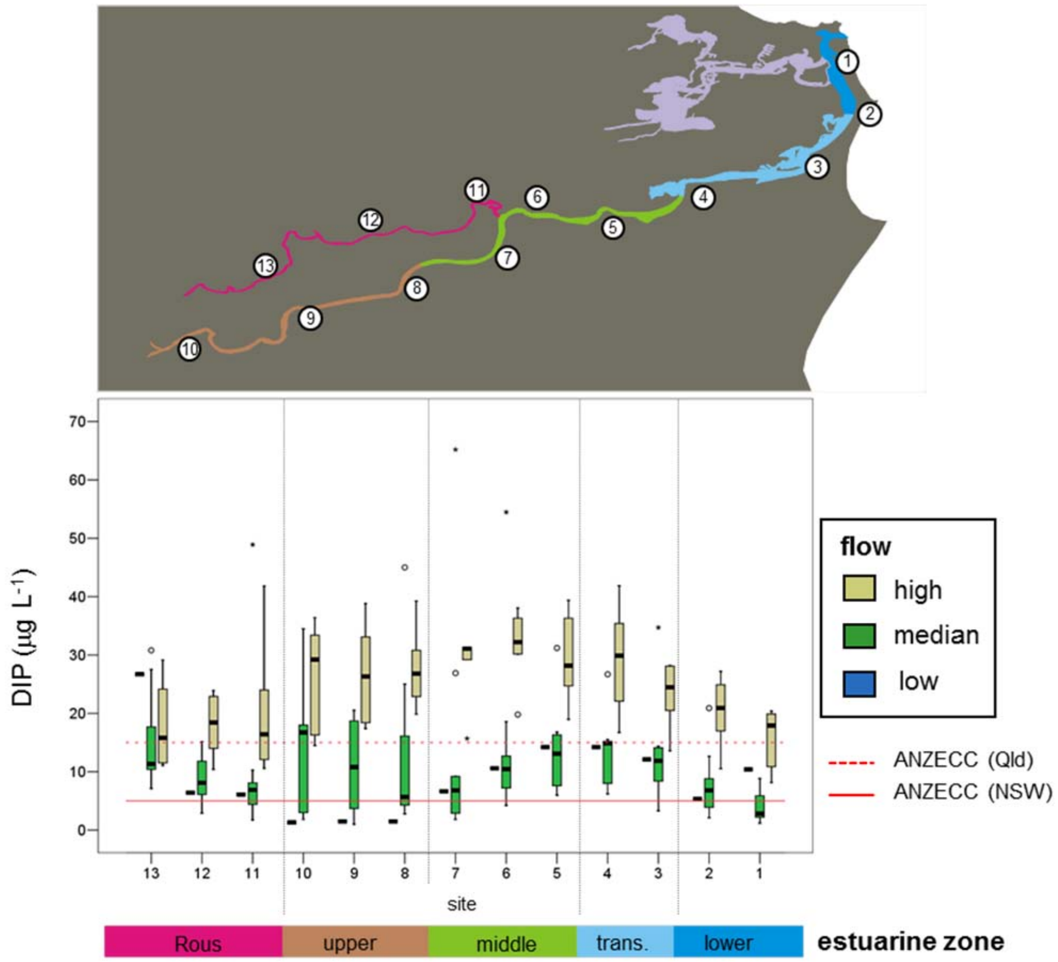


Figure 32 Spatial variation in dissolved inorganic phosphorus concentrations throughout the Tweed estuary.

5.9.3 Temporal trends

Overall, DIP concentrations peaked in Dec 2007 and then steadily declined over the next 6 months (Figure 33). There was a large increase in DIP concentrations during the last high flow sample time.

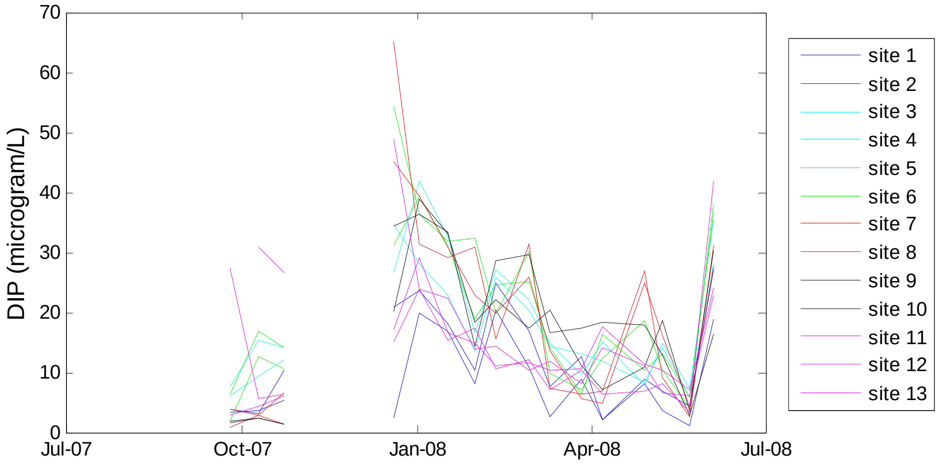


Figure 33 Temporal variation in dissolved inorganic phosphorus concentrations in the Tweed estuary during the study period.

5.9.4 Ecological implications

The ecological implications of DIP are considered relative to dissolved inorganic nitrogen and are therefore addressed in the following section.

5.10 DIN:DIP ratios

5.10.1 Background

The ratio between inorganic nitrogen and phosphorus (DIN:DIP) is commonly used to infer which nutrient is potentially limiting production within the system (Fisher, Melack et al. 1995). Most microalgae (e.g. phytoplankton) require a relatively constant DIN:DIP of 16 for growth (Redfield 1934). Where DIN:DIP falls below 16 it is generally held that the system is nitrogen limited, however in certain cases where DIP is available in high concentrations, phytoplankton such as cyanobacteria (blue green algae) can meet their nitrogen needs via the fixation of atmospheric nitrogen (Karl, Michaels et al. 2002).

5.10.2 Current study

Due to poor data quality and high detection limits for inorganic nutrients during the period 2009 – 2011 only the first part of the dataset will be analysed for nutrient ratio trends. The DIN:DIP ratio was predominantly well below 16 throughout the Tweed estuary during low and median flow conditions, and increased during high flow primarily in the lower estuary (Figure 34). DIN:DIP ratios were consistently higher in the Rous estuary, most likely reflecting high DIN concentrations in STP effluent.

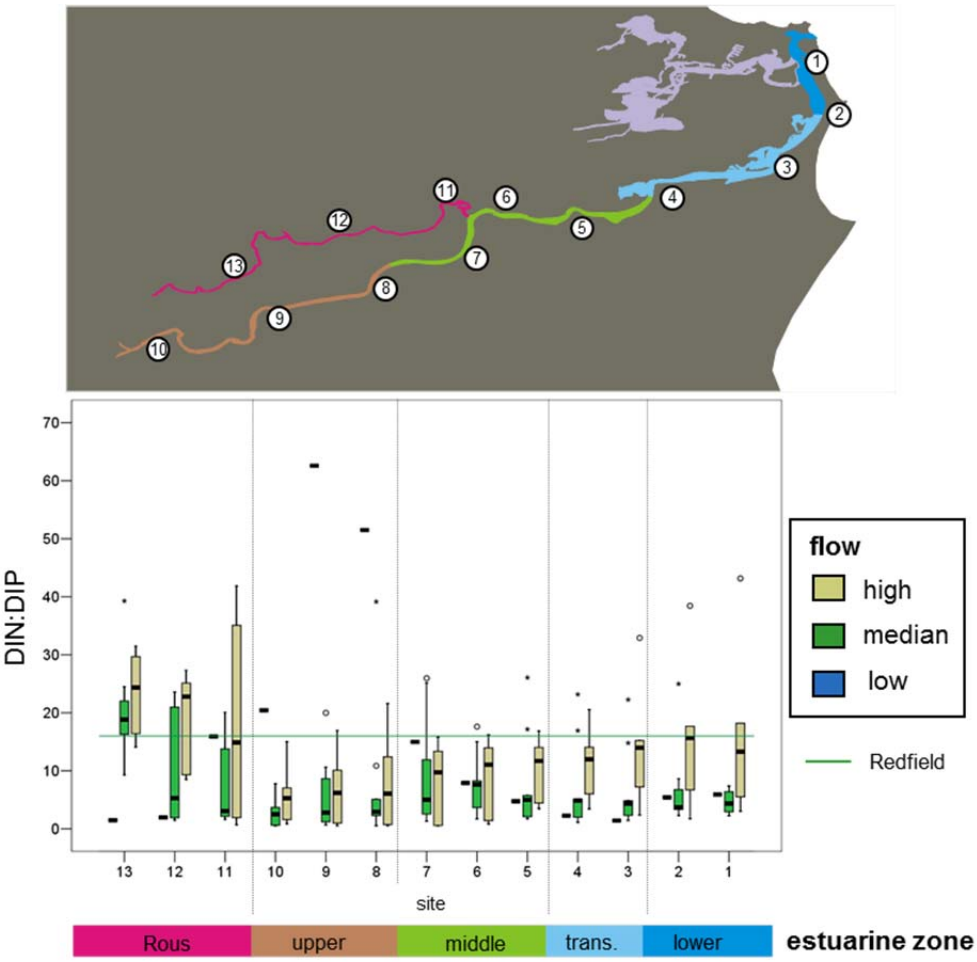


Figure 34 Spatial variation in DIN:DIP ratios throughout the Tweed estuary.

5.10.3 Ecological implications

The predominance of low DIN:DIP ratios throughout the Tweed estuary, especially with diminishing flow, suggest that the system currently tends towards N limitation. In contrast, N limitation is less severe in the Rous estuary which most likely partially accounts for the higher phytoplankton biomass in this reach.

Aside from the influence of freshwater inputs, the DIN:DIP ratio can also be significantly influenced by internal processes (e.g. sediment – water fluxes) especially as water residence times increase. During this study, the occurrence of high DIN:DIP ratios tended to be associated with low dissolved oxygen (DO) saturation (Figure 35). Low DO saturation tended to be spatially aligned with high sediment oxygen demand (see section 5.15 for full details). Remineralised DIP tends to become bound to iron compounds in the sediment, hence it is likely that high DIN:DIP ratios in sediment – water fluxes may also be a primary influence on the water column. This was confirmed by measurements of sediment-water nutrient fluxes made in 2007 – 2008 (DECCW and ABER 2009) and implicates internal recycling as an important factor contributing to ecosystem stress in the Tweed estuary.

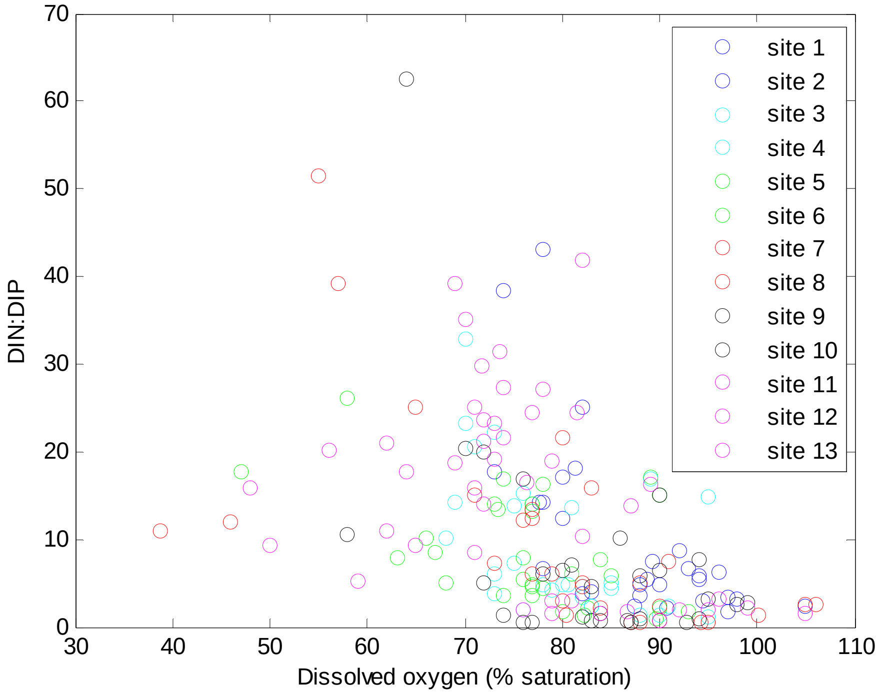


Figure 35 The negative relationship between dissolved oxygen saturation and DIN:DIP ratios during the study period. The association of high DIN:DIP ratios and low DO saturation are two indicators of eutrophication arising from organic enrichment of the sediments.

5.10.4 Management implications

Management strategies should focus on reducing DIN inputs to the system, especially during low to median flow conditions. However, it is important to stress that any nutrient reduction strategies undertaken as part of STP upgrades should consider the reduction of both DIN and DIP in order to maintain the DIN:DIP ratio in effluent at approximately 16. Current research from northern USA suggests that imbalances in effluent DIN:DIP ratios can have far reaching ecological implications (Anderson, Glibert et al. 2002), and that best results in maintaining ecosystem processes are achieved by reducing both DIN and DIP together.

5.11 TN:TP ratios

5.11.1 Background

The TN:TP ratio can also be used to infer which nutrient is most likely limiting productivity in a system. However in many Australian systems where autotrophic production exceeds inorganic nutrient supply (i.e. inorganic nutrients are maintained at very low concentrations), the TN:TP commonly reflects the composition of phytoplankton (i.e. TN:TP = 16). Variation can occur when concentrations of DON are high (e.g. where groundwater inputs dominate), or when other abiotic processes affect one of the nutrients. The latter case is most likely to apply to phosphorus which can be readily bound to particulate iron in estuarine environments under oxidising or saline conditions.

5.11.2 Current study

In contrast to DIN:DIP ratios, TN:TP ratios were consistently above 16 throughout most of the initial monitoring period. During the first part of monitoring period, TN:TP increased progressively towards the upper estuary (Sept to Oct 2007; Figure 36). This spatial trend was reversed, with ratios increasing towards the estuary entrance during subsequent sample times. These trends in TN:TP were primarily controlled by variation in total phosphorus concentrations (Figure 25) and were independent of any nitrogen species, suggesting abiotic controls over phosphorus.

A feasible explanation for the observed trends in TN:TP is the binding of phosphorus with iron hydroxides liberated by acid sulfate soil runoff, and the subsequent removal of particulate P from the water column via flocculation. Two physico-chemical properties are implicated: 1) pH, which determines the redox state of iron in the water (low pH = high dissolved reduced iron; high pH = conversion to iron hydroxides), and 2) salinity, which influences iron state and exerts a powerful control over flocculation (Figure 26).

During the first part of monitoring period which occurred under low flow conditions, acid water would be largely contained within the soil profile and any runoff neutralised within the field drains. As such, the influence of iron hydroxide would have been restricted to the upper estuary. Hence the TN:TP increased towards the upper estuary during this time as high phosphorus concentrations emanating from STP effluent were advected upstream by tidal mixing and reacted with iron hydroxides.

After the onset of the wet season when low pH acid runoff impinged more directly on the main estuary, the formation of iron hydroxides was most likely shifted down the estuarine gradient (i.e. as acid runoff was mixed and neutralised within the estuary). Phosphorus is likely to be scavenged by suspended iron hydroxides at intermediate pH (~ 6 – 7), and then subsequently flocculated as this water was mixed with oceanic water in the lower estuary. Hence, the increase in TN:TP can be seen to result from the physico-chemical transformations of P along pH and salinity gradients.

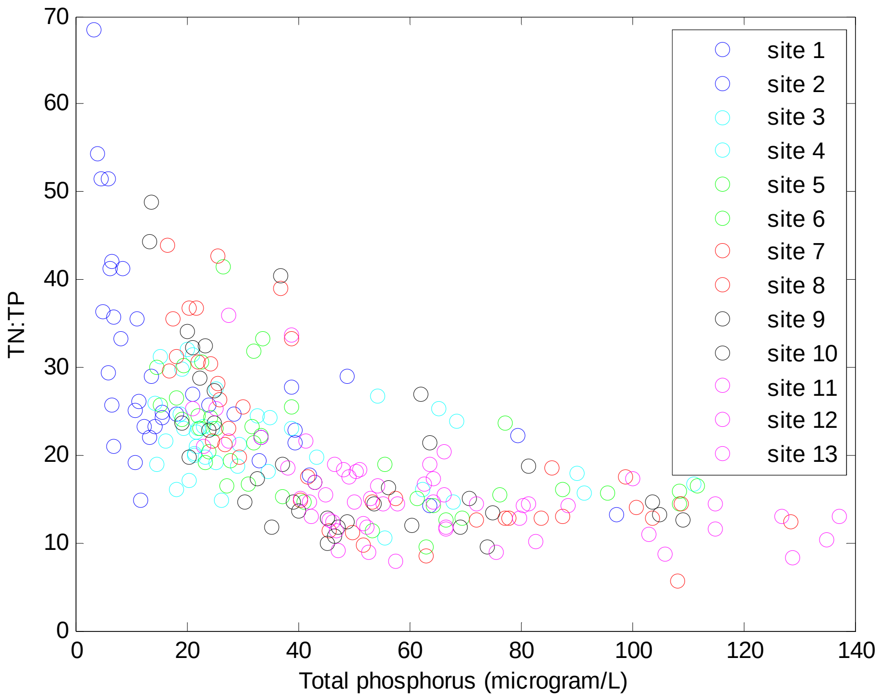


Figure 36 The marked increase in the TN:TP ratio towards the lower Tweed estuary results from the removal of phosphorus from the water column.

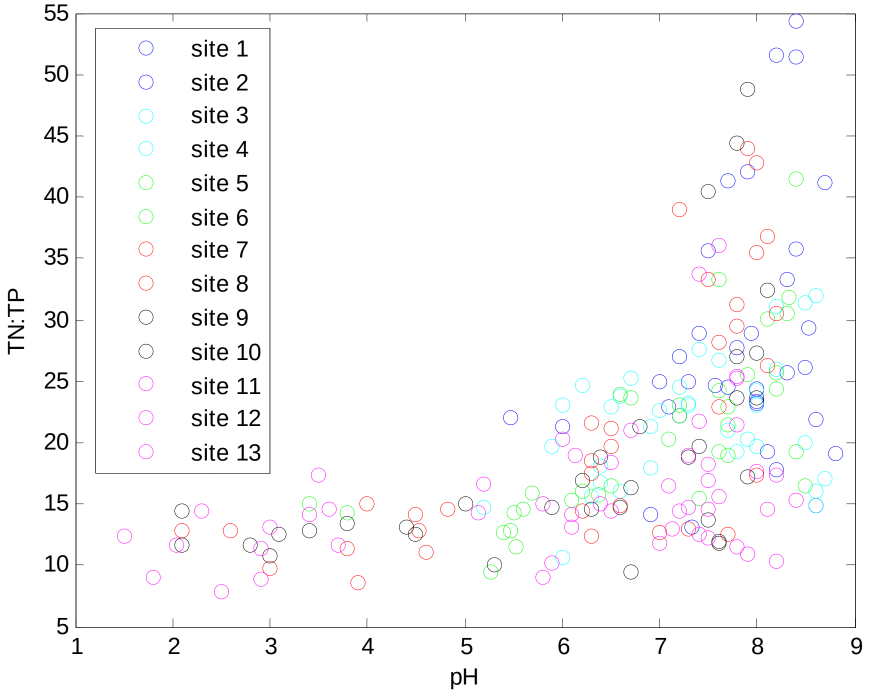


Figure 37 The removal of phosphorus and hence the increase in TN:TP towards the lower estuary is related to pH indicating the likely flocculation of iron bound phosphorus.

5.12 Chlorophyll-a

5.12.1 Background

Chlorophyll-a is a measure of the primary photosynthetic pigment present in all microscopic, single cell algae in the water column (phytoplankton). Chlorophyll-a concentrations are commonly used as a proxy measure of the total biomass of phytoplankton in aquatic systems. Phytoplankton comprise one of the major primary producers in estuarine systems and are therefore an important base for many foodchains. Stimulation of phytoplankton growth due to nutrient pollution is a major aspect of eutrophication, which in extreme cases causes organic enrichment and deoxygenation of sediments and greatly decreases water clarity.

5.12.2 Spatial trends

Chlorophyll-a concentrations ranged from below detection (0.1 µg L^{-1}) to 57 µg L^{-1} during the study period (Figure 38). There was a consistent trend of low chlorophyll-a concentrations in the lower estuary increasing to a peak in either the middle or upper estuary. Chlorophyll-a was consistently highest in the Rous estuary.

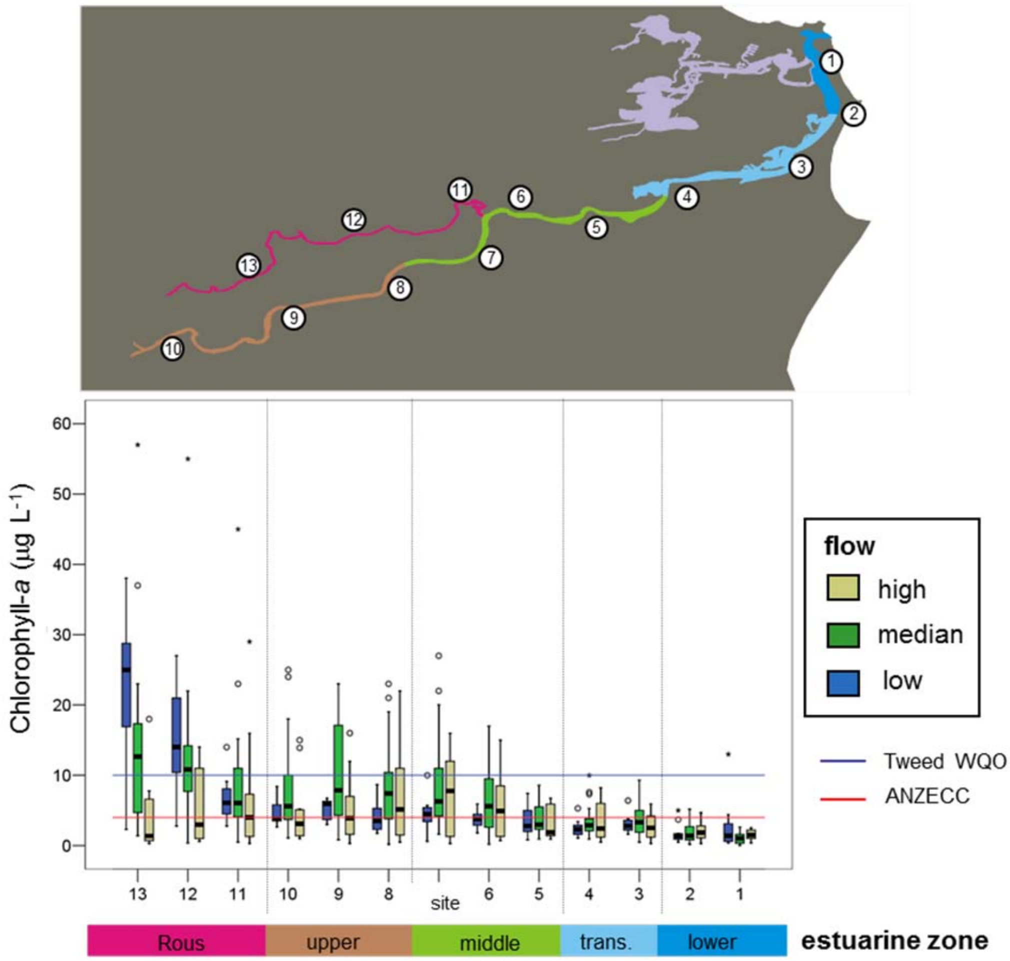


Figure 38 Spatial variation in chlorophyll-a concentrations throughout the Tweed estuary.

There was no significant trend with flow conditions in the Tweed estuary, however concentrations tended to be lowest during low flow conditions. In contrast, there was a clear significant trend ($p<0.001$) of increasing chlorophyll-a concentrations with diminishing flow in the middle to upper Rous estuary (Figure 38).

5.12.3 Temporal variation

There was a high level of variation in chlorophyll-a concentrations on monthly to bimonthly timescales, most likely reflecting the boom-bust nature of phytoplankton blooms in estuarine systems. Underlying this variation was a seasonal trend of higher concentrations during the summer – autumn period when both solar radiation (light) and nutrient supply (freshwater inflow) are greatest (Figure 39). Overall, February and March had the highest concentrations ($p<0.05$) during the study period. Concentrations in the Rous estuary were largely uncoupled from the Tweed estuary, as illustrated by the large persistent bloom in the Rous during the second part of 2009.

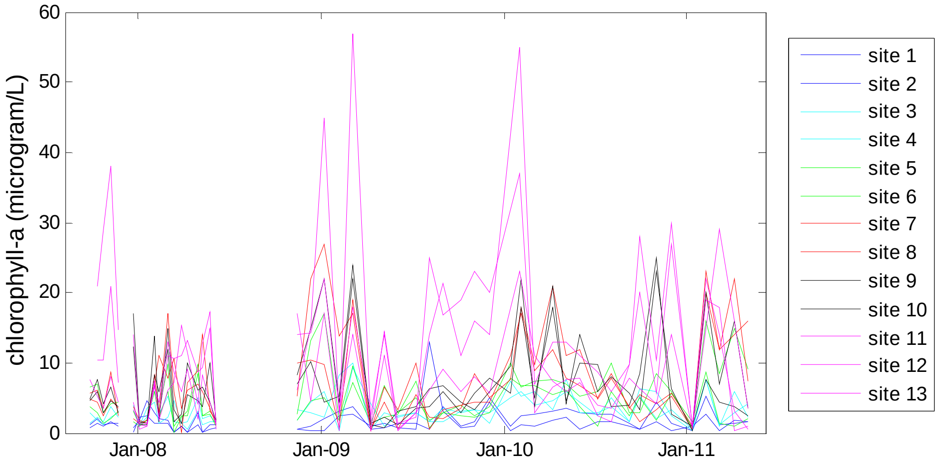


Figure 39 Temporal variation in chlorophyll-a concentrations in the Tweed estuary during the study period.

5.12.4 Inter-annual variability

There was significant inter-annual variability in chlorophyll-a concentrations ($p<0.05$) during the study period (Figure 40). The source of this variability was during the summer – autumn period, where the timing and severity of phytoplankton blooms varied greatly according to primary forcing factors such as flow and residence times. In contrast, inter-annual variability was minimal during the late winter – spring period (Figure 40), when variation in forcing factors was much lower.

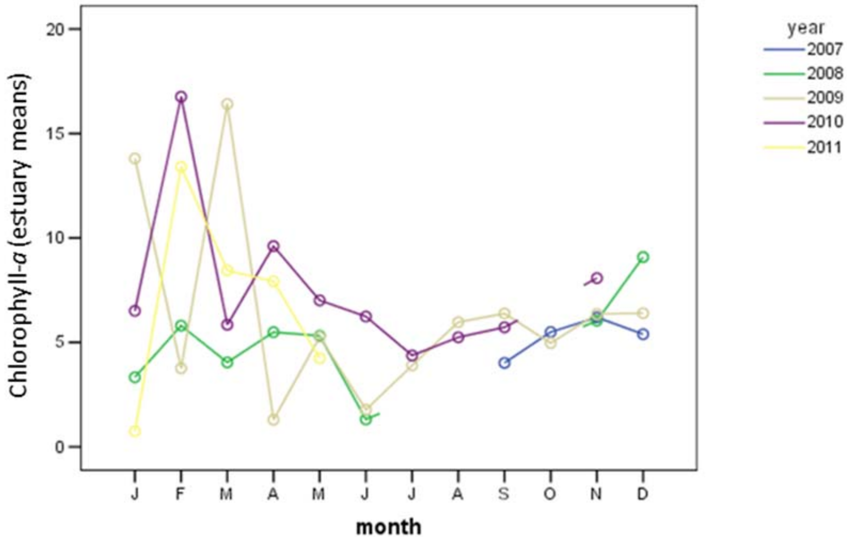


Figure 40 Inter-annual variability in mean estuary chlorophyll-a concentrations in the Tweed estuary during the study period.

5.12.5 Phytoplankton model

Background

A simple empirical model was developed to demonstrate the impacts of dissolved inorganic nitrogen ($NH_4^+ + NO_x$) inputs and freshwater flushing times on phytoplankton biomass along the estuary. This model is adapted from a similar approach used successfully in the nearby Brunswick estuary (Ferguson, Eyre et al. 2004). The model derives a dimensionless estimation of phytoplankton biomass from the multiplication of modelled DIN[6] ($DIN_{modelled}$) and freshwater flushing times (FWRT):

$$PHY = DIN_{modelled}^{DINfact} * FWRT^{FWRTfact}$$

Where DINfact and FWRTfact are constants that allow for non-linear response of phytoplankton biomass to the two factors. The model was tested against mean chlorophyll concentrations for low, median and high flow conditions.

Results

A combination of modelled DIN and freshwater residence times can explain a significant part of the observed variation in phytoplankton biomass along the Tweed estuary (Figure 41). The relationship between DIN loading and increased biomass is self explanatory: increased DIN availability stimulates growth. The interaction with freshwater flushing times arises from the increase in time available for utilisation of available DIN by phytoplankton. Biomass increase also proceeds much faster once water residence times exceed the cell doubling time of phytoplankton (Gay 2002). It should be noted that the current estimation of freshwater flushing time does not indicate the true water residence times along the estuary which integrate the flushing effects due to freshwater inflows and tidal flows.

The results also show that the slope for the dimensionless output versus measured chlorophyll-a is lower for low flow conditions (Figure 41). This suggests that other factors limit biomass.

6 Modelled DIN is derived from the salt balance model presented in section 6.

For example, as flow decreases and water residence times increase, light penetration improves (see section 5.14) and the relative importance of DIN uptake by benthic microalgae and denitrification increases significantly (Ferguson, Eyre et al. 2007). The resultant competition for DIN means that phytoplankton biomass per unit DIN loading is significantly less. In addition, grazing of phytoplankton biomass would be expected to increase as flow diminishes, thereby causing another limiting factor on biomass accumulation. This competition is significantly reduced in the Rous estuary where DIN inputs from the STP are relatively much larger. This gives rise to the observed increase in chlorophyll-a with decreasing flow (Figure 38), and explains the decoupling of phytoplankton growth in the Rous and Tweed estuaries.

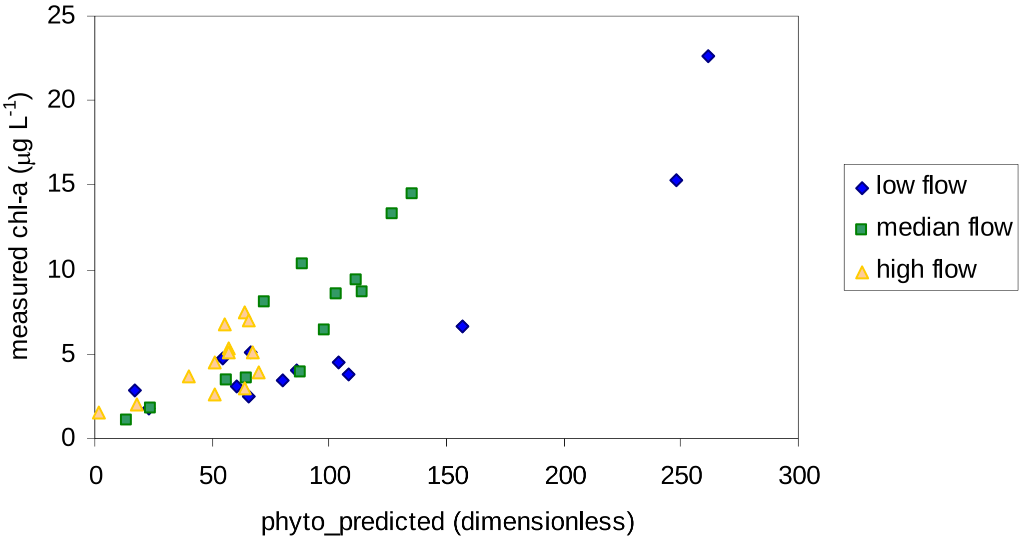


Figure 41 Comparison predicted phytoplankton (phyto_predicted) and measured mean chlorophyll-a concentrations during low, median and high flow conditions.

5.12.6 Ecological implications

Chlorophyll-a concentrations in the Tweed and Rous estuaries are consistently higher than ANZECC (2000) guidelines for the maintenance of aquatic ecosystems (Figure 38). In addition, there is considerable evidence to link phytoplankton blooms in the estuary with low dissolved oxygen saturation (see section 5.15 for full details). As such, chlorophyll-a should be regarded as an indication of ecosystem stress in these systems.

5.12.7 Management implications

The phytoplankton model (see above) confirms DIN loading as a primary factor influencing phytoplankton blooms in the estuary. In addition, improved light climate during low flow conditions allows for higher competition by benthic processes for DIN resources thereby limiting phytoplankton growth. As such, management efforts should focus on reducing DIN inputs and improving water clarity during median and high flow conditions. This can best be achieved by STP management during median flow and catchment management during high flow (see section 6: Salt balance modelling).

5.13 Total suspended solids

5.13.1 Background

Total suspended solids (TSS) is a measure of the combined concentration of particulate matter (comprising inorganic sediments, organic matter and phytoplankton) in the water column. The relative contribution of these constituents varies widely according to position along the estuary, state of tide and state of flow. TSS is a major driver of water clarity, impacting on the light climate of the water column and sediments.

5.13.2 Spatial trends

Total suspended solids (TSS) concentrations ranged between 0.5 and 185 mg L^{-1} during the study period (Figure 42). TSS concentrations were general lowest at the estuary mouth, increased towards the middle estuary, and diminished towards the upper estuary. TSS in the Rous estuary were generally high relative to the main Tweed estuary. There was a clear and consistent increase in TSS with flow category, with the highest TSS concentrations recorded during high flow conditions.

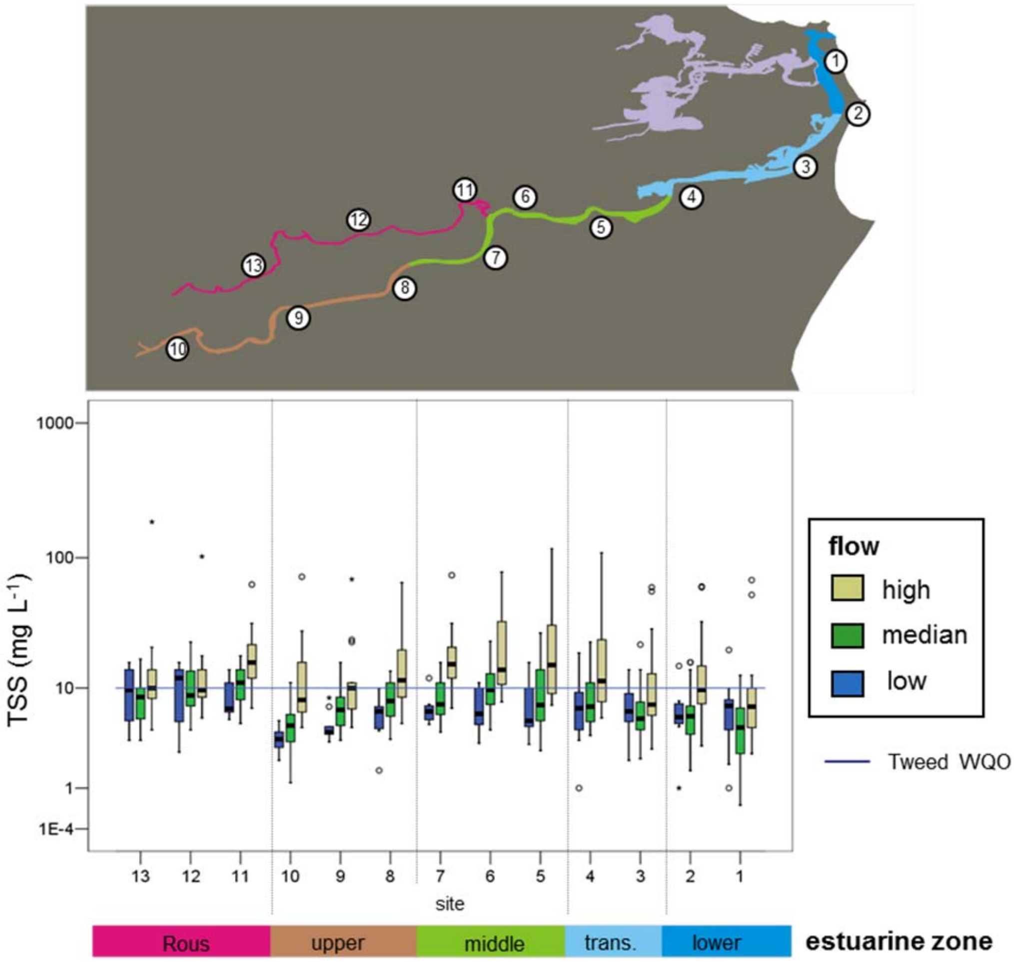


Figure 42 Spatial variation in total suspended solids throughout the Tweed estuary.

High frequency data (every 20 minutes) collected by in situ dataloggers (ABER/NSW DECCW unpublished data) suggest that resuspension of bottom sediments may also play a significant role in elevating TSS in the middle estuary (Figure 43). There are regular spikes in turbidity apparent at the middle estuary site which coincide with the highest tidal currents occurring on mid incoming and outgoing tides (Figure 44). In contrast, the lower estuary site experiences peaks in turbidity coinciding with low tide when more turbid water from upstream is advected into the lower estuary. During high tide, turbidity drops as the site becomes influenced by clear oceanic water.

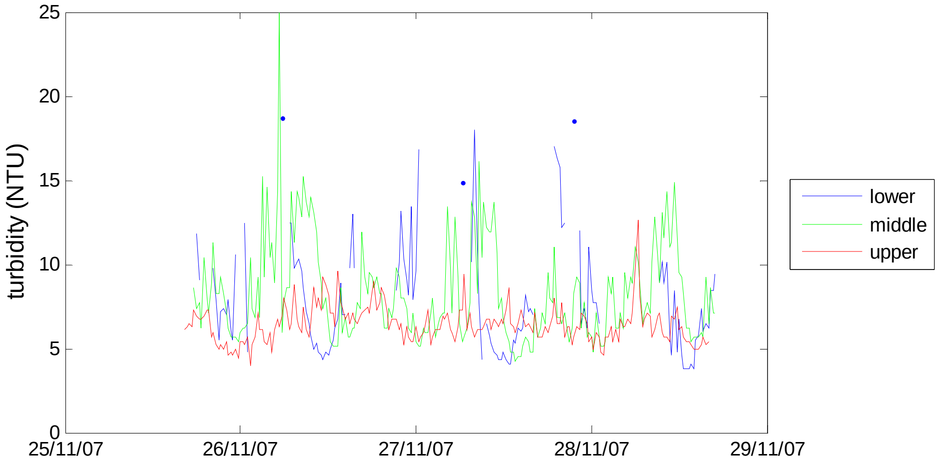


Figure 43 Turbidity variation over three days at three locations in the Tweed estuary (lower = downstream of Pacific Highway bridge; middle = downstream of Stotts Island; upper = upstream of Murwillumbah). Unpublished data from ABER/NSW DECCW.

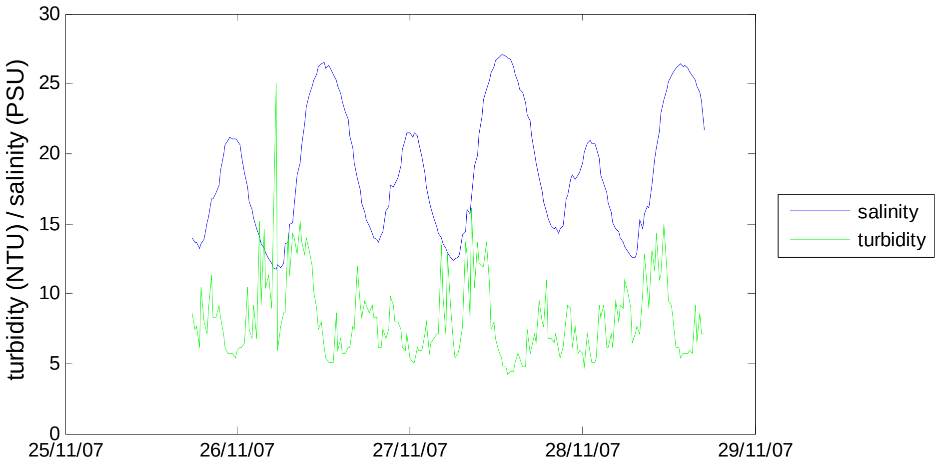


Figure 44 Salinity and turbidity variation over three days at the middle datalogger site (Stotts Island). Using salinity as a proxy for tide state, note that the main spikes in turbidity occur at mid incoming tide and lesser spikes occur during the outgoing tide.

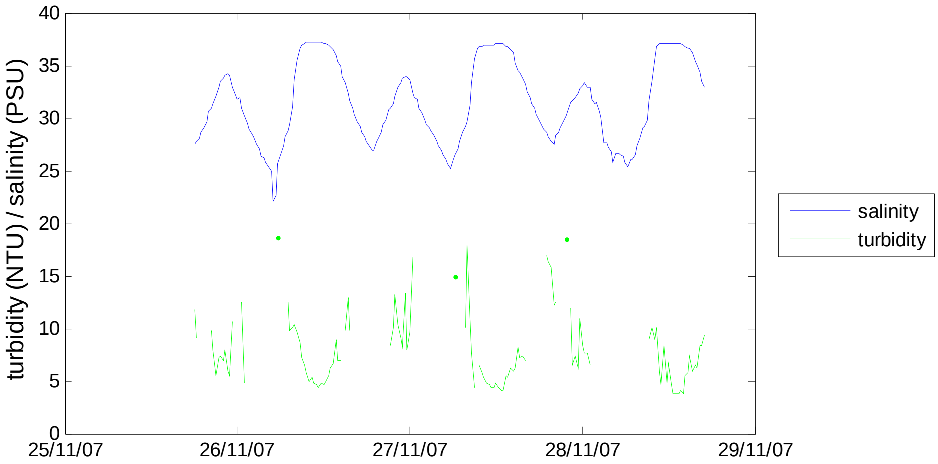


Figure 45 Salinity and turbidity variation over three days at the lower (Pacific Highway) datalogger site. In contrast to the middle estuary, increases in turbidity coincide with low tide when the lower estuary is influenced by water from upstream.

5.13.3 Temporal trends

There was high variability in TSS concentrations over monthly to bimonthly timescales, reflecting variability in flow conditions at the time of each sampling effort. It is also clear that state of tide at the time of sample collection will also significantly affect the results due to resuspension (see Figure 44). Overall, TSS tended to be highest during the summer-autumn wet season months, with large spikes occurring during flood events (Figure 46). This is supported by the strong trend of increasing TSS concentrations with flow (Figure 42).

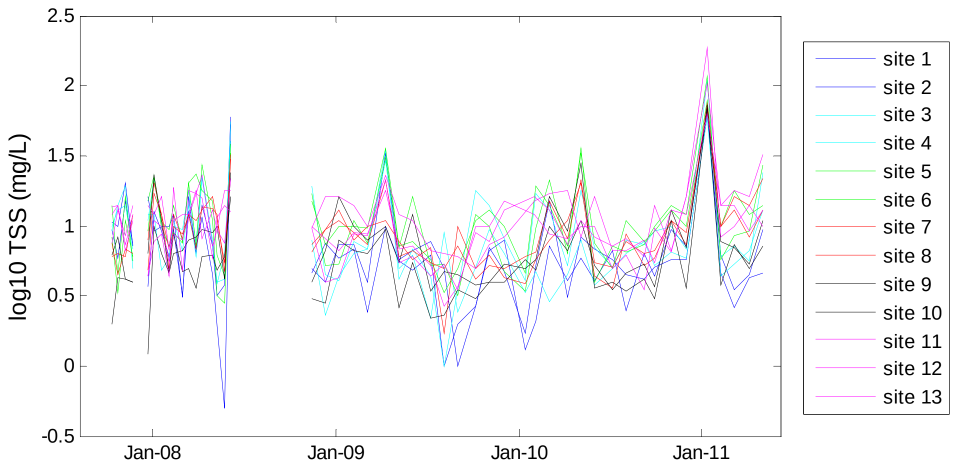


Figure 46 Temporal variation in total suspended solids in the Tweed estuary during the study period.

5.13.4 Inter-annual variability

Inter-annual variability in TSS concentrations was minimal, excluding the extreme high flow events (floods) which may occur at any time during the wet season window (summer – early winter). Due to the relatively rapid decay in high TSS concentrations post-flood, it is likely that the routine sampling strategy would have missed some TSS peaks during the study period.

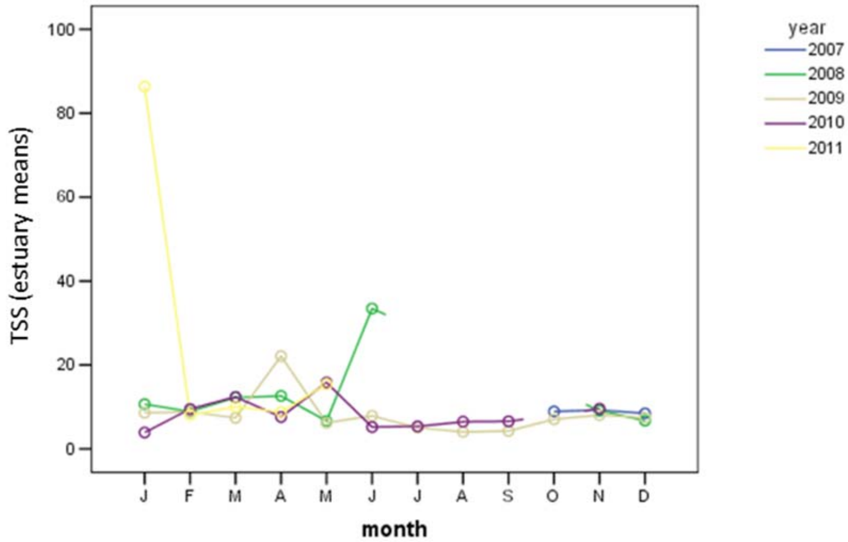


Figure 47 Inter-annual variability in mean estuary TSS concentrations in the Tweed estuary during the study period.

5.13.5 Ecological implications

TSS is a critical ecological indicator due to the importance of good water clarity in maintaining ecosystem processes in the Tweed and Rous estuaries. The relative importance of TSS to water clarity is discussed more fully in section 5.14. It is clear that different processes control TSS concentrations during different flow conditions: inorganic sediments from diffuse catchments during high to median flows, giving way to organic internal sources (e.g. phytoplankton) and tidally-induced resuspension during median to low flows. It is likely that a portion of TSS loads delivered to the estuary during high flows are deposited and subject to tidal resuspension as flows diminish.

5.13.6 Management implications

Management strategies should focus on reducing TSS in catchment runoff during high and median flow times in order to reduce bed loads of fine grained sediments that are susceptible to resuspension. Large flood events tend to override catchment management strategies and in general deliver most of their load to the continental shelf, bypassing the estuary. Strategies to reduce phytoplankton blooms during median flow will significantly reduce TSS concentrations in the middle and upper reaches of the estuary.

5.14 Water clarity (secchi depth)

5.14.1 Background

Water clarity (as measured by secchi depth) is a critical attribute that relates to the attenuation of light and its impact on primary production (pelagic and benthic) within aquatic ecosystems. The total attenuation of photosynthetically active radiation (PAR) with depth through the water column (denoted by Kd_{PAR}) is due to the individual contributions of particulate and dissolved constituents, as well as the water itself[7]. Broadly, the constituents of concern are inorganic suspended solids, organic suspended solids, phytoplankton cells, and coloured dissolved organic matter (CDOM). In the current study, TSS includes both inorganic solids, organic solids and phytoplankton (measured as chlorophyll-a), therefore individual contributions of these fractions are difficult to derive. However, chlorophyll-a will be treated as a separate variable in order to give an indication of its importance. Hence, Kd_{PAR} can be estimated as:

$$Kd_{PAR} = Kd_{water} + Kd_{chlorophyll}*Chl + Kd_{TSS}*TSS + Kd_{CDOM}*CDOM(colour)$$

Kd_{PAR} can be directly estimated from secchi depth using the relationship:

$$Kd_{PAR} = 1.44 / \text{secchi depth}$$

This relationship does not vary between systems (Figure 48), and assuming secchi depth has been collected carefully and to a high level of precision, the estimation of Kd_{PAR} is a valuable indicator of a core ecosystem process.

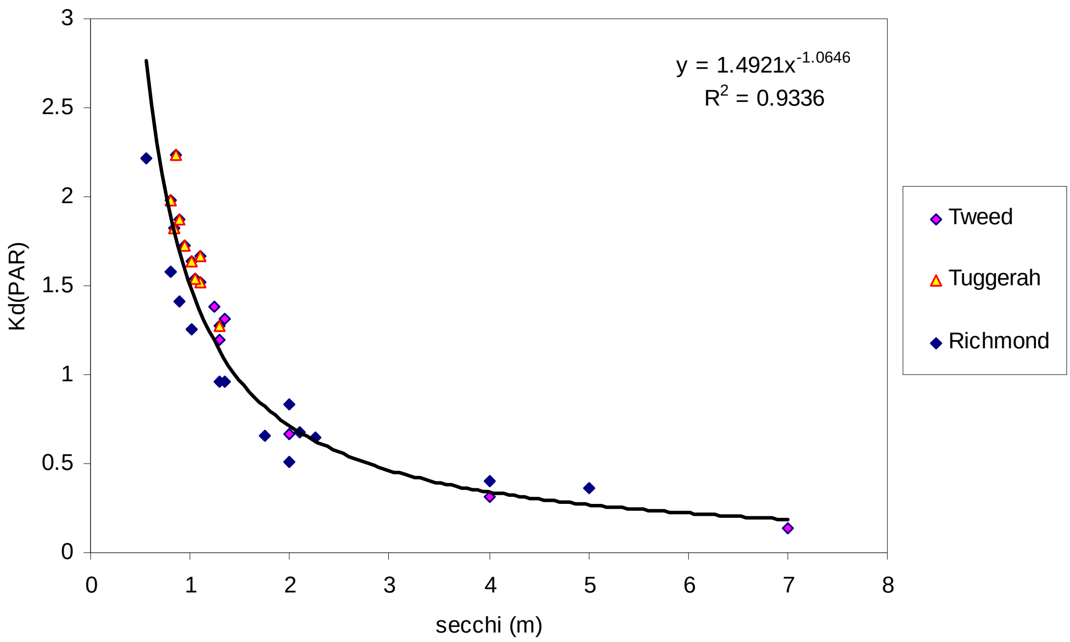


Figure 48 The relationship between K_{PAR} and secchi in Tuggerah Lakes and the Richmond and Tweed estuaries. The line of best fit (y = 1.492x-1.0646) is similar to published values (e.g. K_{PAR} = 1.44 / secchi; (Kirk 1994).

[7] Attenuation of PAR is due to absorption and scattering of light due to optically active constituents within the water. For the purposes of this analysis, KPAR will be defined as the sum of absorption and scattering.

5.14.2 Spatial trends

Secchi depths ranged from less than detection (0.1m) to greater than 6m (Figure 49). There was a consistent spatial trend of greatest secchi depths in the lower estuary becoming progressively more shallow towards the upper estuary. The Rous estuary consistently had shallower secchi depths than the main Tweed estuary. There was a significant reduction in secchi depths with increased flow at all sites in the Tweed estuary ($p<0.05$). In contrast, this trend was not significant in the middle to upper Rous estuary.

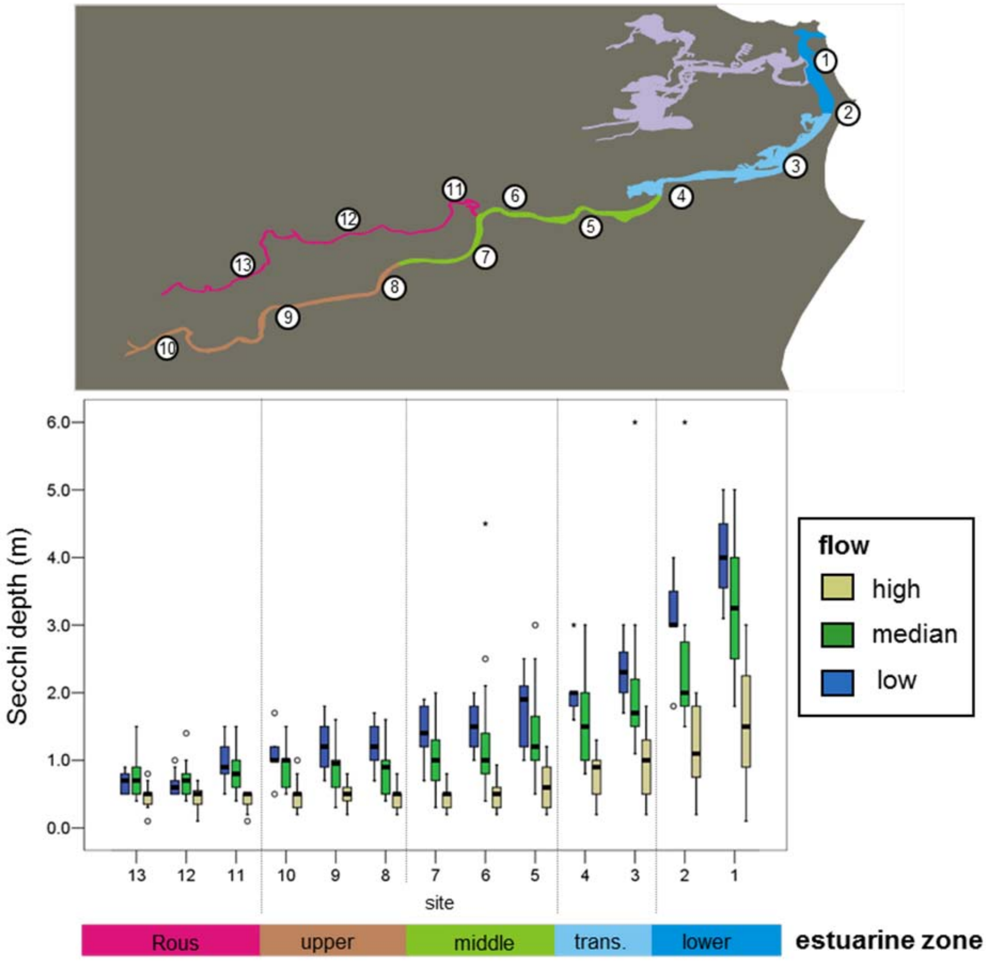


Figure 49 Spatial variation in secchi depths throughout the Tweed estuary.

5.14.3 Temporal variation

There was a clear seasonal trend of poor water clarity (shallow secchi depth) during the summer – autumn wet season, followed by a gradual improvement through winter into spring (Figure 50). This trend did not hold for the Rous estuary which generally experienced poor water clarity throughout the winter – spring period.

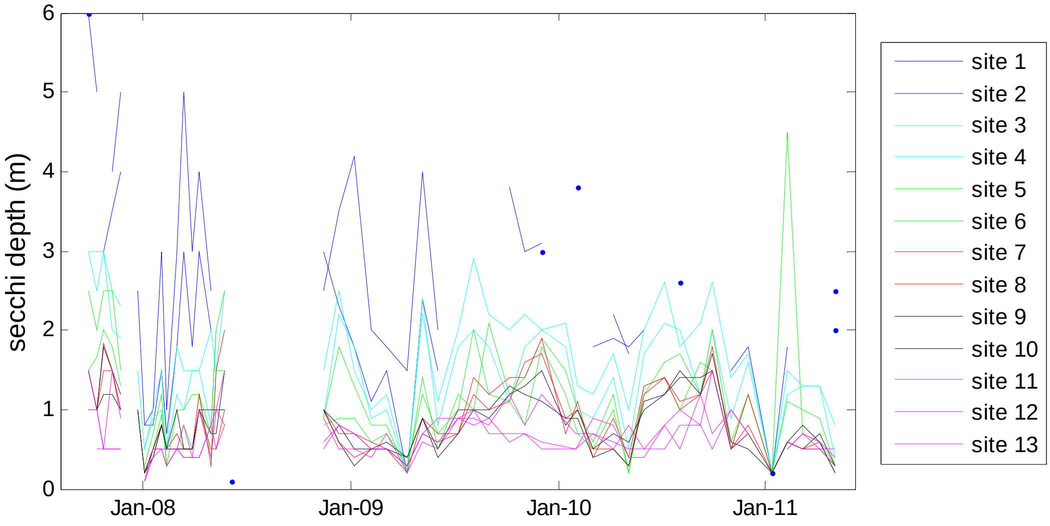


Figure 50 Temporal variation in secchi depth in the Tweed estuary during the study period.

5.14.4 Inter-annual variability

There was significant inter-annual variability in secchi depths (Figure 51), due to the high temporal variability in constituent attenuating properties (TSS, chlorophyll-*a* and dissolved colour). These are in turn primarily controlled by variability in freshwater inflow and residence times.

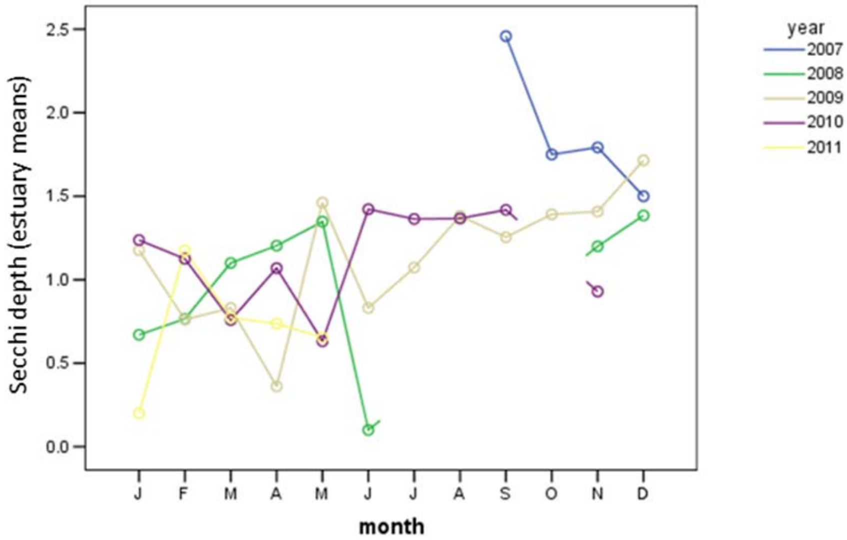


Figure 51 Inter-annual variability in mean estuary secchi depth in the Tweed estuary during the study period.

Contributions to Kd_{PAR}

Light attenuation (Kd_{PAR}) has been assessed in terms of its constituent attenuating properties in Figure 52. A multi-linear regression model was used to assess the relative importance of these constituents over Kd_{PAR} (Figure 53). The results of this analysis show that colour has the greatest bearing over Kd_{PAR}, followed by TSS and then chlorophyll-*a*. The relative contributions varied with flow however, and were complicated by the interdependence of the constituents on each other. For example, chlorophyll-*a* becomes 3 times more important during low flow conditions when TSS concentrations due to freshwater inputs are minimal. During this time colour becomes closely related to chlorophyll-a, reflecting the release of dissolved organic exudates by phytoplankton. In contrast, the relative importance of TSS increases with flow and colour becomes associated with dissolved organics in freshwater runoff.

The comparisons in Figure 52 show that secchi depth measurements were subject to considerable error in collection. In addition, the default precision of 0.1m resulted in coarse results. For example, during higher turbidity times (i.e. when Kd_{PAR} exceeded 2) there was a great increase in identical readings across wide ranges of TSS and other constituents. This most likely arises from the coarse lumping of readings up or down into 0.1m categories.

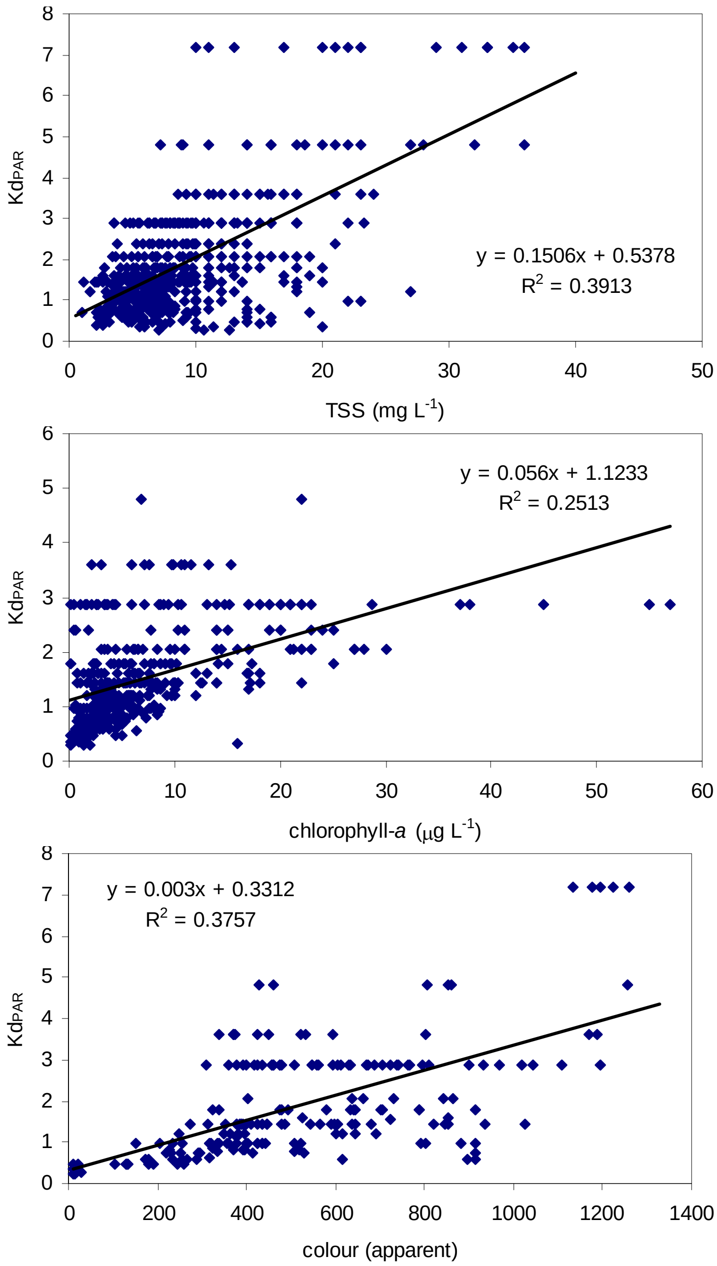


Figure 52 The relationship between individual water quality attributes and Kd_{PAR}

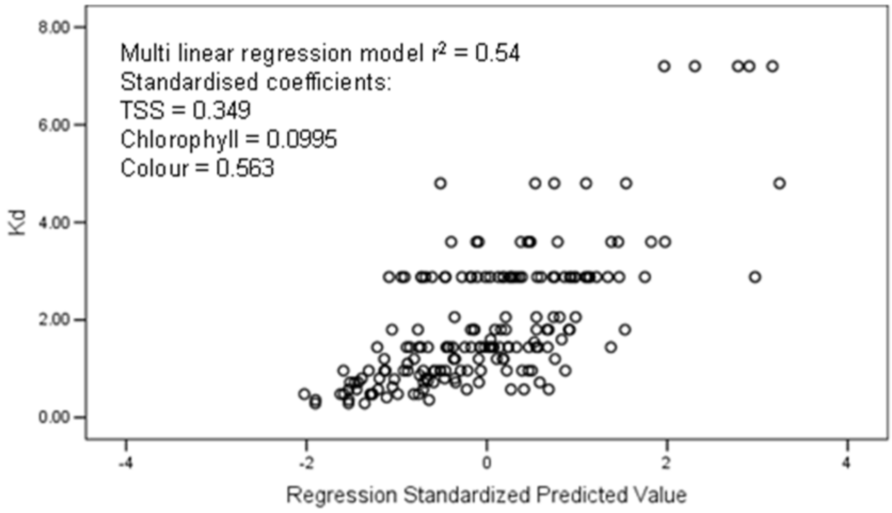


Figure 53 Multiple linear regression model of Kd_{PAR} (dependent variable) and TSS, chlorophyll-a, and colour (independent variables). The standardised coefficients give an indication of the relative influence of each variable over Kd_{PAR}.

5.14.5 Ecological implications

The impact of water clarity over light climate has important implications for dissolved oxygen saturation in the Tweed estuary. There is evidence that breakdown of organic matter in the sediments is responsible for a significant part of the observed oxygen sag along the estuary. This oxygen consumption is offset by benthic microalgae photosynthesis (an oxygen producing process), which is closely related to the light reaching the sediments. The net sediment oxygen flux is therefore the difference between oxygen production (photosynthesis) and consumption (organic matter breakdown). As such, it follows that better water clarity will decrease biological oxygen demand within the system and result in increased oxygen saturation.

Data collected during the DEFIRE project show that sediments in the Tweed estuary become a net source of oxygen when light reaching the sediment surface is greater than 200 µEinsteins (or approximately 10% of average midday sun during summer; Figure 54).

Good water clarity also increases the relative importance of photosynthesis by benthic microalgae (along the entire estuary) and seagrass (in the lower estuary). Benthic microalgae in particular constitutes a major source of organic carbon in these shallow ecosystems, and oxygen production greatly increases the diversity of sediment habitats. Increased benthic productivity by seagrass and benthic microalgae has important implications for the nature and diversity of estuarine foodwebs, which are likely to depend on these primary producers under pristine conditions.

Production by benthic microalgae and seagrass also constitute a major temporary sinks of nutrients, thereby helping to limit the occurrence of phytoplankton blooms (e.g. during low flow conditions in this study). Further, benthic microalgae can indirectly stimulate denitrification (a major permanent sink of nitrogen), and promote the retention of DIP in sediments.

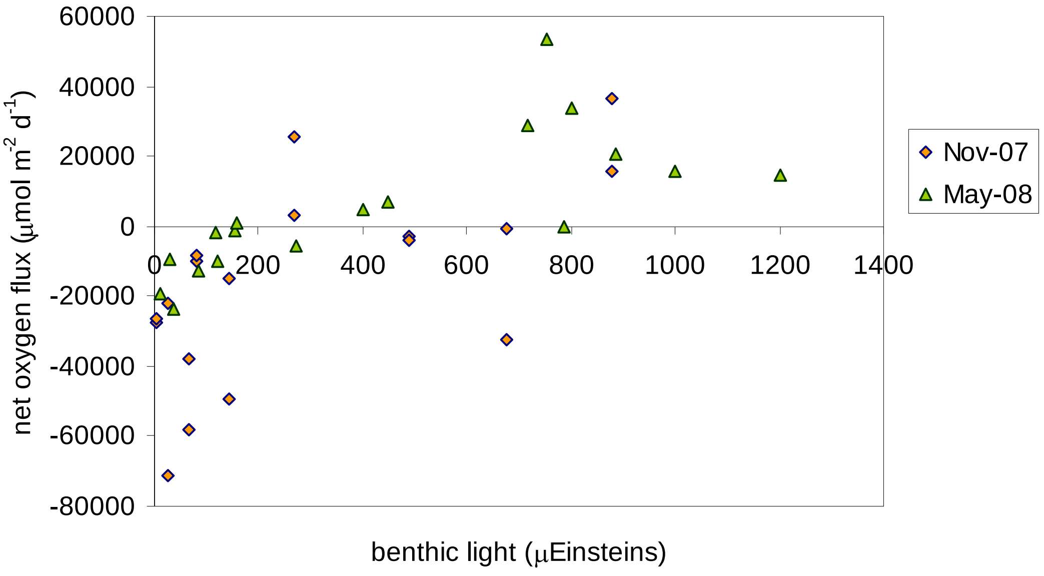


Figure 54 Net oxygen flux from sediments (positive = production of oxygen to the water; negative equals consumption by sediments). Sediments become a net source of oxygen as light improves due to the impact of photosynthesis by benthic microalgae.

5.14.6 Management implications

Management of good water clarity in the Tweed should be regarded as a high priority in order to 1) improve oxygen saturation and 2) increase productivity by benthic microalgae and seagrass. Strategies should be directed towards reducing chlorophyll-a concentrations during low to median flow times, and inorganic TSS during high flow times.

5.15 Dissolved oxygen saturation

5.15.1 Background

Dissolved oxygen (DO) is a measure of the concentration of free oxygen in solution, with the maximum potential concentration controlled by atmospheric pressure, temperature and salinity. Oxygen solubility decreases with increasing temperature and salinity. As such, the concentration of DO in estuaries varies naturally according to location along the estuarine gradient and season. It is therefore common to correct for these factors by expressing DO in terms of percent saturation (i.e. DO % sat = measured concentration / maximum potential concentration at the temperature and salinity of the sample). Any deviations from 100% saturation are largely due to biological or chemical processes in the water body which consume or produce oxygen. Oxygen consuming processes include oxidation of acid sulfate runoff products and the biological breakdown of organic matter. Oxygen producing processes include photosynthesis by phytoplankton, seagrass and benthic algae.

DO is a key indicator of environmental stress, with low DO (hypoxia = DO < 3) causing stress to a range of estuarine biota and impacting on biogeochemical cycling. Hypoxia can occur in response to acute events (e.g. “blackwater” runoff following flood events), or as a chronic impact of eutrophication[8]. The common expression of eutrophication in riverine estuaries is an increase in phytoplankton blooms in the upper and middle reaches caused by excessive nutrient loadings[9]. While DO saturation is an important and most commonly reported indicator of stress, there are other significant implications such as reduced water clarity (with flow effects on benthic productivity) and shifts in species assemblages.

5.15.2 Spatial trends

There was a consistent trend of highest dissolved oxygen (DO) saturation in the lower estuary (sites 1 and 2) progressing to a DO sag at the middle estuarine sites before improving slightly at the upper estuarine sites (Figure 55). It is likely that these trends are primarily driven by biological processes, in particular the production and breakdown of organic matter. Hypoxia tended to be greatest in the middle to upper Tweed estuary sites during low flow conditions, most likely due to the greatly increased residence times. In contrast, DO saturation improved towards the upper estuary during median flow conditions. The Rous estuary displayed opposite trends, with DO saturation increasing upstream during low flow conditions, and decreasing upstream during median flow conditions. DO saturation was lowest in the middle estuary during high flow times which may reflect inputs of low DO water from low lying catchments adjacent to this reach. This is confirmed by high frequency logger data collected during the high flow event in June 2008 (Figure 56).

[8] Eutrophication is defined as an increase in organic matter supply to an ecosystem. This can occur due to direct inputs of organic matter (e.g. organic rich effluent with a high BOD), or as the result of nutrient enrichment which causes algal blooms within the system.

[9] An estuary’s susceptibility to eutrophication is largely determined by interactions between hydrology (freshwater inputs and tidal flows) and morphology (depth). Upper and middle estuarine reaches tend to be more susceptible due to longer water residence times which allow phytoplankton blooms to develop. Greater water depths in the middle to upper reaches result in a higher proportion of the water column below the compensation depth (i.e. light limited) increasing the relative importance of heterotrophic (oxygen consuming) processes. Light limited conditions at the sediment-water interface reduce the relative importance of benthic productivity as an oxygen source and nutrient sink, resulting in sediments mediating internal nutrient recycling and oxygen consumption.

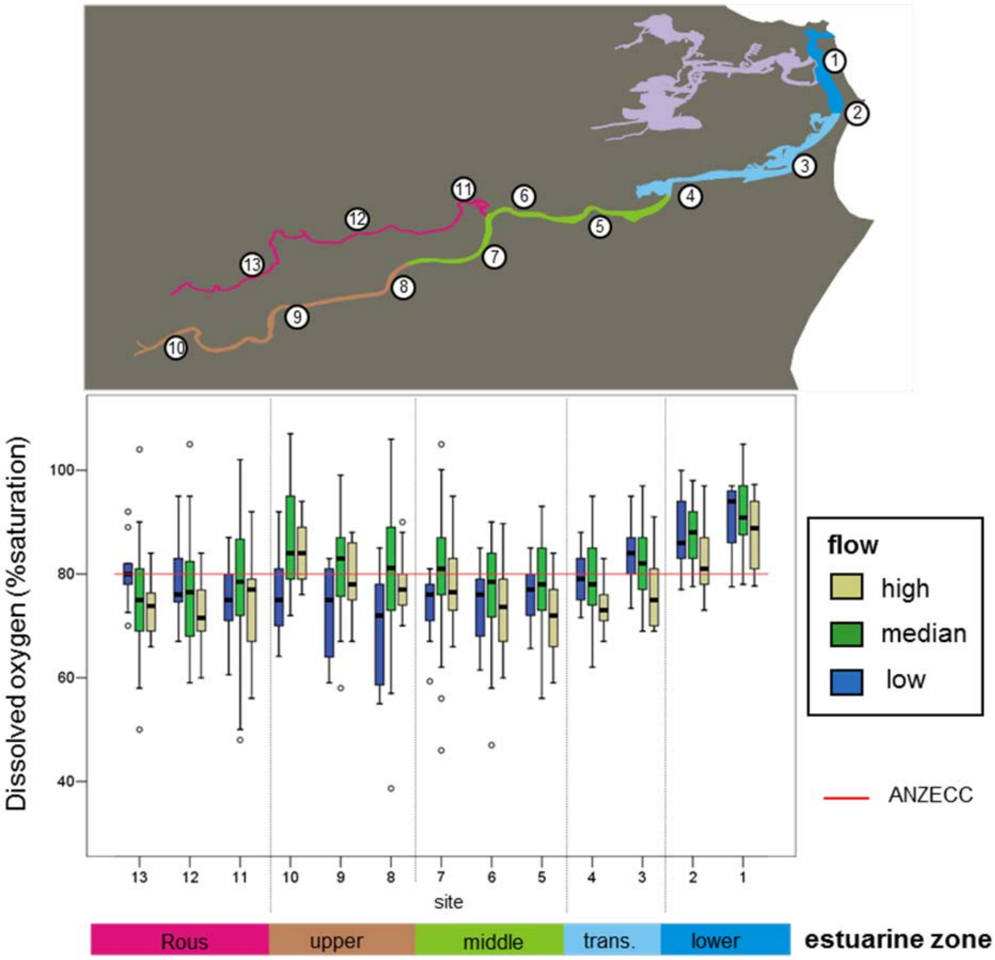


Figure 55 Spatial variation in dissolved oxygen saturation throughout the Tweed estuary.

5.15.3 Temporal trends

There were no clear seasonal trends in DO saturation during the study period, with variation tending to occur over monthly or bimonthly timescales rather than clear seasonal or annual timescales (Figure 57). Spatial patterns in DO saturation showed consistent sags (or hypoxia) in the middle to upper estuary during most sample runs, with only the severity of hypoxia varying between runs. Given the primary dependence of DO saturation on biological processes, This suggests that spatial controls (e.g. channel morphology) the production and breakdown of organic matter along the estuarine gradient are more important than seasonally varying factors such as light, temperature and freshwater inflows.

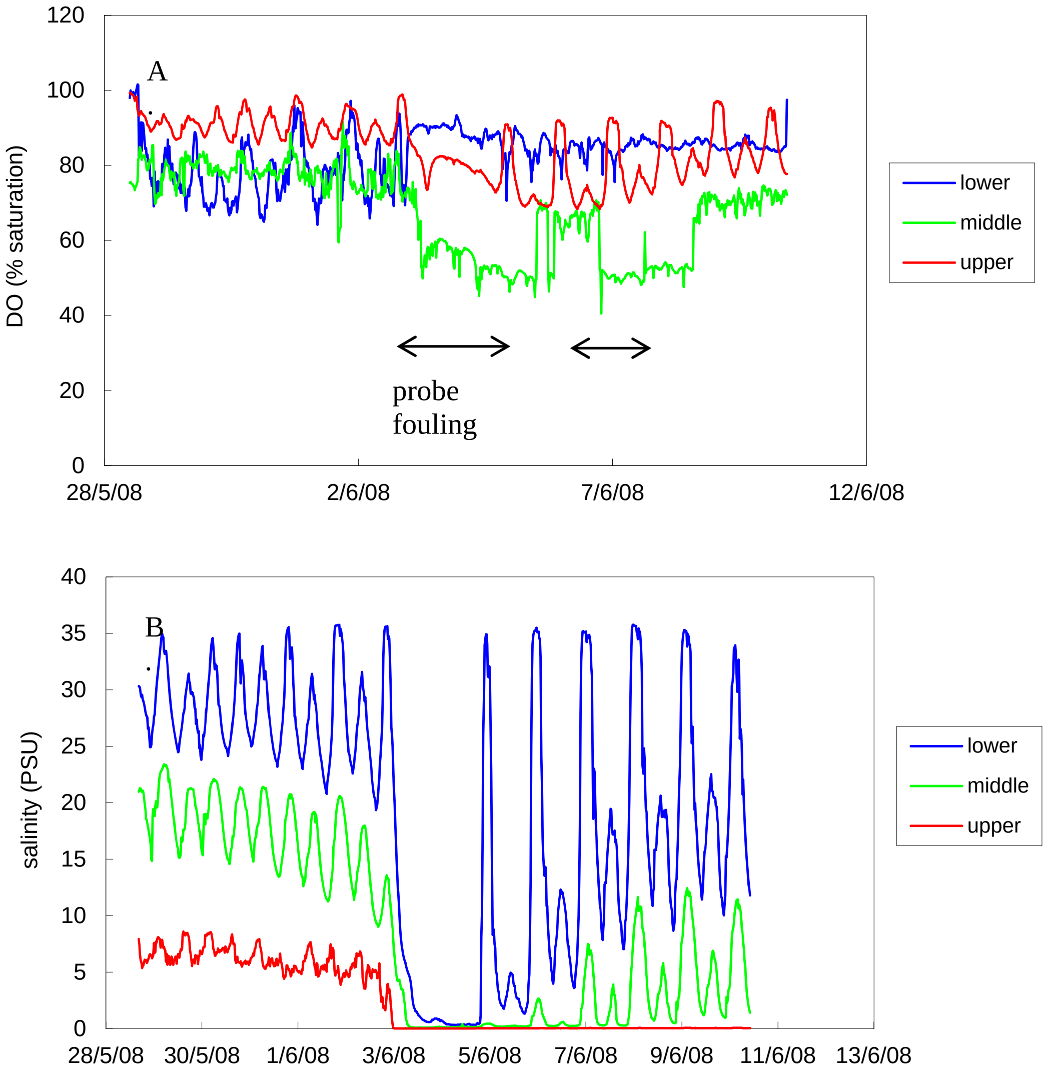


Figure 56 **A.** Dissolved oxygen saturation and **B.** salinity variation over one week at three locations in the Tweed estuary (lower = downstream of Pacific Highway bridge; middle = downstream of Stotts Island; upper = upstream of Murwillumbah). A flood event occurred on the 3/6/08 causing the estuary to be flushed fresh to the mouth. Note: the oxygen probe on the middle datalogger most likely became periodically fouled during the high flow causing an approximate 20% decrease in readings. Regardless of this artefact, DO saturation was lowest in water entering the middle estuary. Unpublished data from ABER/NSW DECCW.

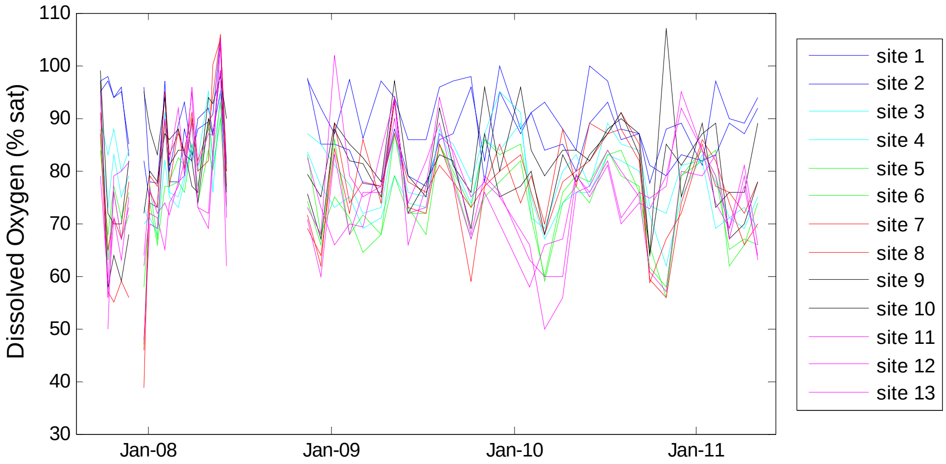


Figure 57 Temporal variation in dissolved oxygen saturation in the Tweed estuary during the study period.

5.15.4 Inter-annual variation

There was significant inter-annual variation in DO saturation during this study ($p<0.05$; Figure 58), arising from highly variable seasonal progression in saturation between different years. This is most likely due to the interaction of freshwater flows and temperature (i.e. the timing of wet season flows varied).

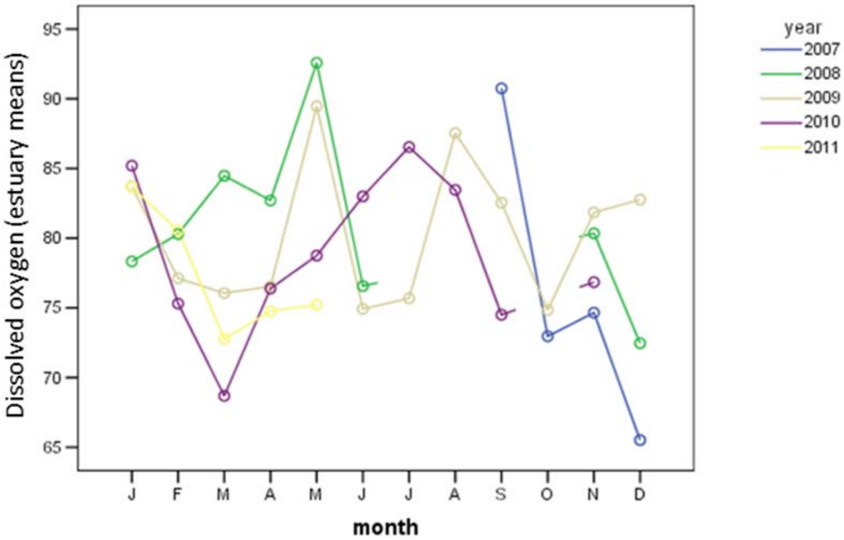


Figure 58 Inter-annual variation in mean estuary dissolved oxygen saturation over the study period.

5.15.5 Stratification

Physico-chemical profiles taken by TSC at sites 3, 6 and 13 are summarised in Figure 59. Detailed profiles taken by ABER on 27/11/07 are presented in Figure 60. There was significant vertical stratification apparent in dissolved oxygen saturation, with saturation decreasing with depth. The degree of vertical saturation tended to be greatest in the middle to upper estuary during low to median flow conditions (Figure 60). Conversely, while under saturation was commonly recorded during high flow sample times, vertical stratification tended to be minimal due to better mixing and shorter residence times.

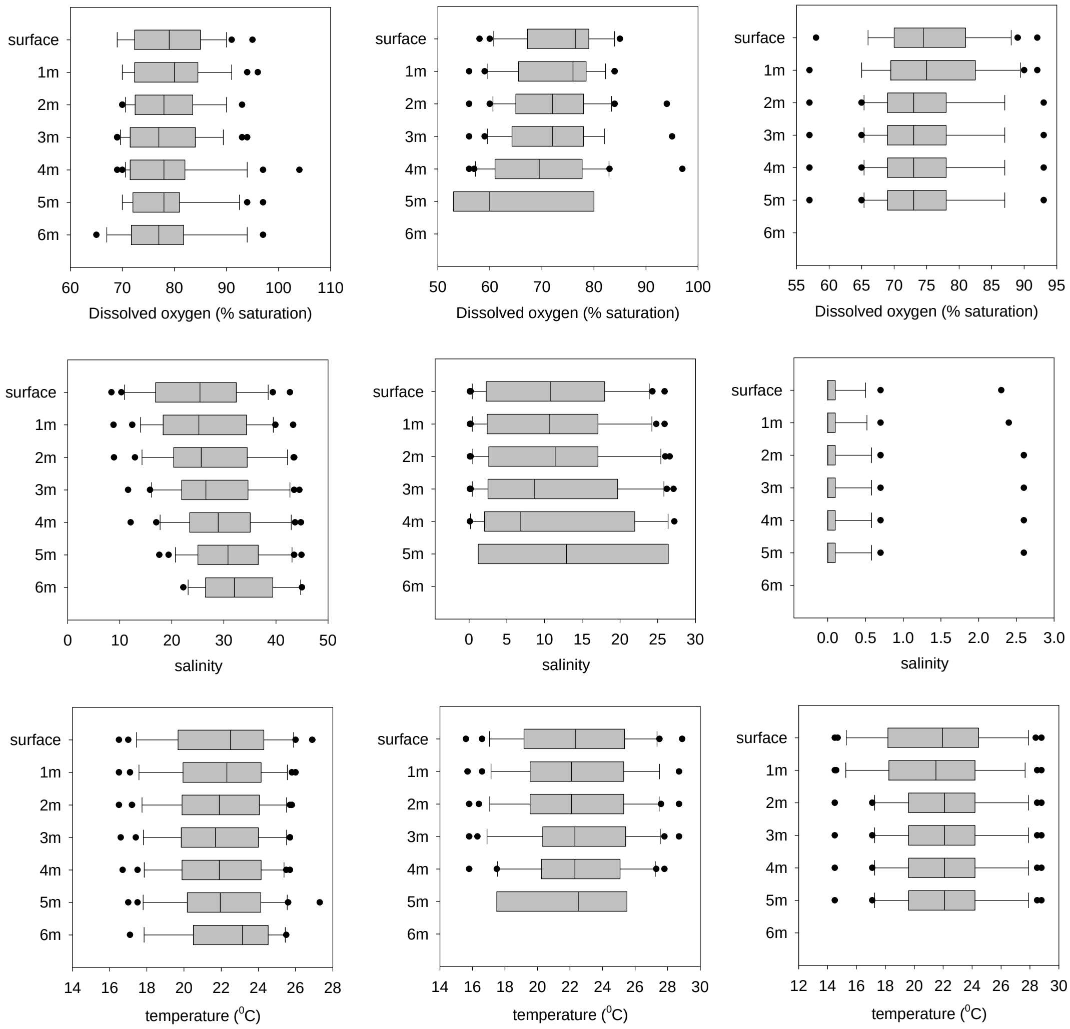


Figure 59 Boxplot summaries of physico-chemical profiles taken by TSC staff at sites 3, 6 and 13.

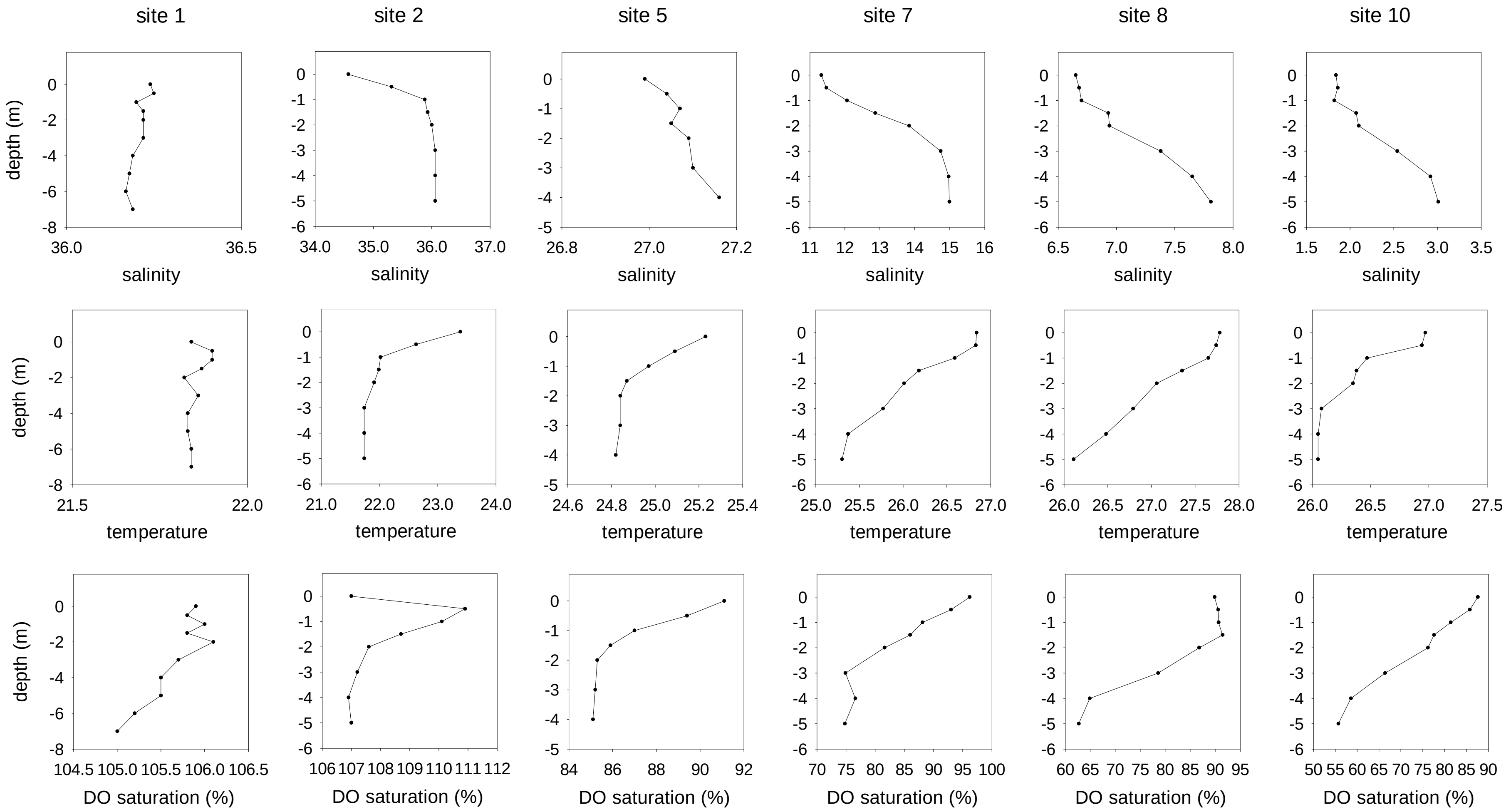


Figure 60 Depth profiles of salinity, temperature and dissolved oxygen saturation at selected sites under low to median flow conditions on the 27/11/07.

5.15.6 Dissolved oxygen sags

Mechanisms – low to median flow

The results of this survey indicate that the Tweed estuary is prone to moderately severe sags in dissolved oxygen saturation along the middle to upper estuary reaches during low to median flow conditions. Theses reaches are most susceptible to hypoxia due to the coincidence of residence times long enough to allow phytoplankton bloom development, more channelized morphology and mesohaline salinity conditions which enhance the deposition of phyto-detritus to the sediments. In contrast, more rapid flushing and shallower morphology in the lower estuary serve to make it more resilient to eutrophication in response to nutrient loadings from the Kingscliff STP.

The location and severity of the sag varied between sample times, most likely due to differences in freshwater inputs, stage of the spring / neap tidal cycle, and state of tide at the time of sampling. These factors determine the structure of the estuarine salinity gradient and the location of phytoplankton blooms which are likely to be the main source of organic matter within the system. The most severe DO sags occur upstream of the Rous estuary confluence during low flow which most likely reflects the significant increase in water residence times in this reach and greater oceanic influence downstream of the confluence. The DO sag is generally less severe during median flow conditions due to a reduction in water residence times.

The role of sediments

DO profiles show a marked decrease in oxygen saturation with depth (Figure 60) indicating that the primary cause of deoxygenation during low to median flow conditions is likely to be the breakdown of organic matter in the sediments. This is supported by measured rates of sediment oxygen consumption, which tend to be greatest at the middle estuary sites (Figure 61). Preliminary biogeochemical modelling of the Tweed estuary (ref DEFIRE) indicates that phytoplankton biomass is one of the primary organic matter inputs to the sediments in this system (Figure 62) and is therefore closely linked to sags in DO saturation[10].

[10] The deposition rate of phyto-detritus is controlled by 1) the biomass of phytoplankton in the water column (as indicated by chlorophyll-*a* concentrations), 2) the settling velocity of the phytoplankton cells, and 3) the depth of the water column. Research in the Tweed and Richmond estuaries suggests that the settling velocity is also positively related to salinity due to coagulation of cells. As such, an indirect measure of phyto-detritus deposition has been developed using chlorophyll-*a* and salinity, and this is closely related to measured sediment oxygen demand (Figure 22).

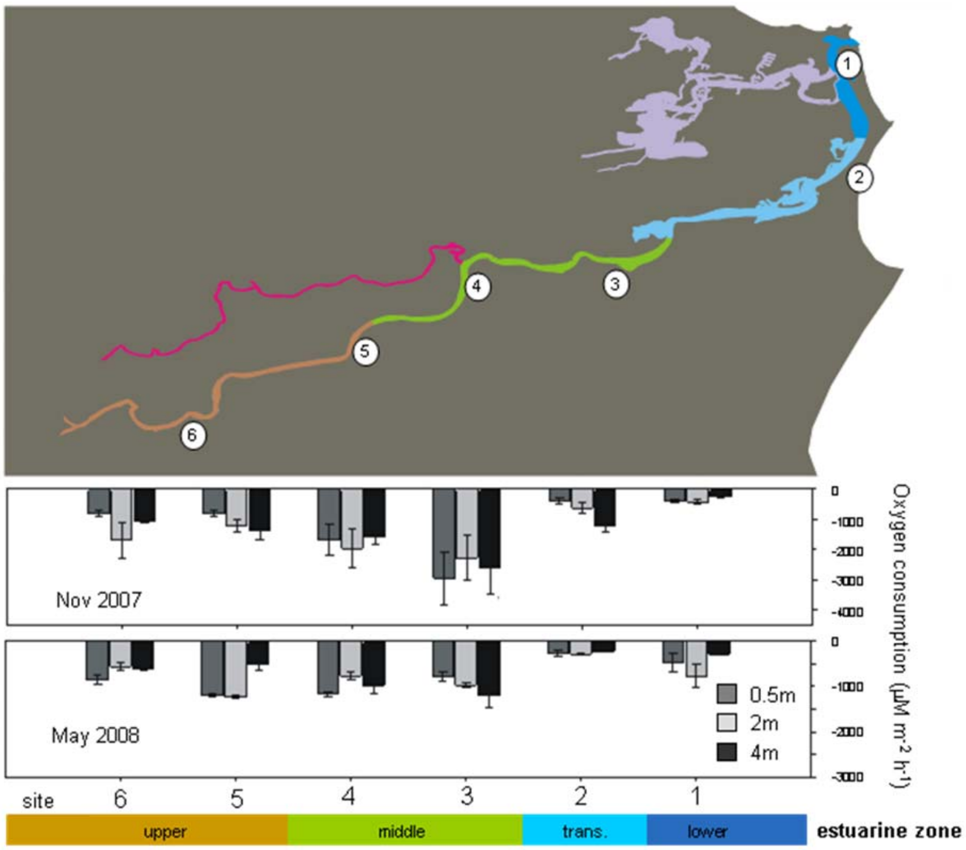


Figure 61 Oxygen consumption by sediments (mean ±SD) at 6 sites along the Tweed estuary (data from ABER / DECCW). The middle estuary consistently had the highest rates of consumption.

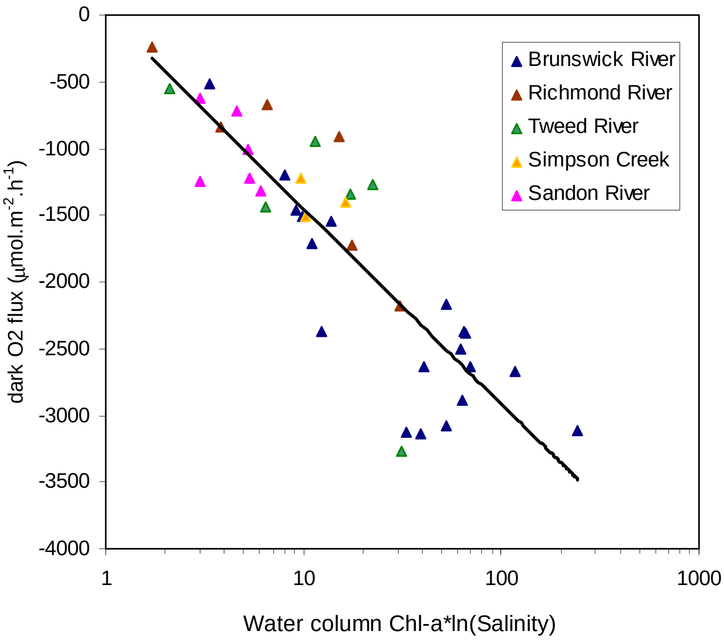


Figure 62 Relationship between deposition of phyto-detritus to sediments (estimated as chlorophyll * ln[salinity]) and sediment oxygen consumption.[11]

[11] Points for Richmond, Tweed, Simpsons and Sandon estuaries are annual means (n = 9 - 12) for sites along the estuarine gradient. Brunswick points are sample means (n = 5) for an upper estuary site.

Mechanisms – high flow
The input of hypoxic water from low lying catchments adjacent to the middle estuary most likely accounts for reduced DO saturation along this reach during high flow (see Figure 56). This phenomenon is common in northern NSW estuaries and is due to the inundation of swamps and low lying pastures during floods. Residual organic matter and dying pastures contribute a high BOD which deoxygenates runoff.

5.15.7 Ecological Implications

Patterns of DO saturation observed during this study indicate that the middle estuary is currently stressed with regards to dissolved oxygen saturation, with stress also occurring in the upper estuary during low flow periods. Stress tends to increase with depth and will therefore be significant for demersal fish / crustacean species which feed on or inhabit sediments. Although not dealt with in this report, high rates of sediment oxygen consumption and pelagic hypoxia imply high rates of nutrient regeneration from the sediments and a potential decrease in denitrification efficiency (Eyre and Ferguson 2009), which causes a positive feedback in the stimulation of phytoplankton blooms.

5.15.8 Management implications

The current STP loading rates are implicated in stimulating phytoplankton blooms in the Rous estuary, and middle to upper reaches of the Tweed estuary resulting in moderate to severe hypoxia during low to median flow times. There was a clear relationship between total nutrient concentrations and dissolved oxygen saturation (Figure 63), indicating that TN loadings are directly linked to ecosystem stress. The data show that TN concentrations in excess of 400 μg L^{-1} lead to moderate to severe hypoxia throughout the estuary. As such, the current ANZECC (2000) threshold guideline for total nitrogen (300 μg L^{-1}) represent an appropriate threshold guideline for the maintenance of aquatic systems in the Tweed estuary.

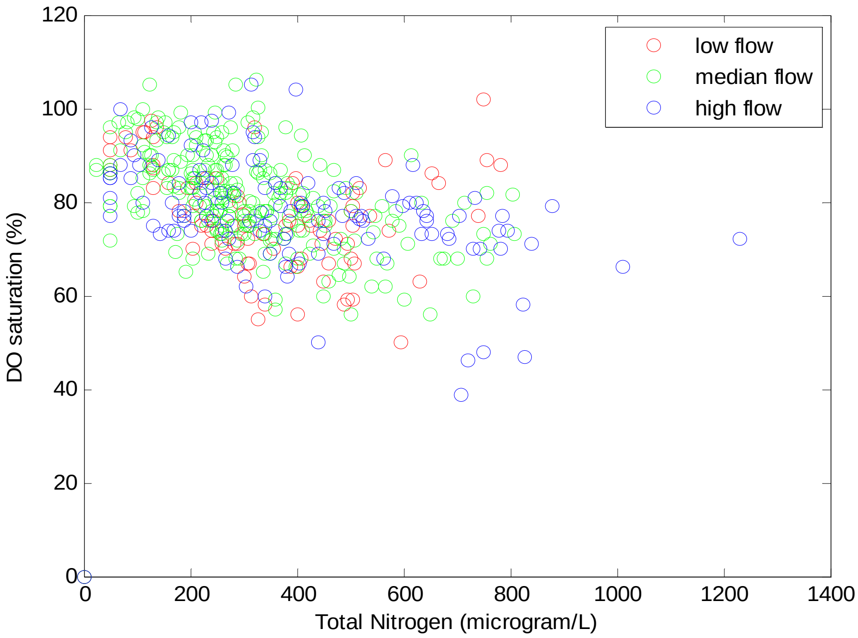


Figure 63 The relationship between total nitrogen and dissolved oxygen saturation in the Tweed and Rous estuaries. The ANZECC (2000) guideline threshold for the maintenance of aquatic systems in estuaries (300 μg L^{-1}) corresponds to a median dissolved oxygen saturation of approximately 80%.

6 Cobaki - Terranora Broadwaters

6.1 Salinity

6.1.1 Spatial trends

Salinity was generally highest at the Terranora Creek / Tweed River sites (10 to 16), slightly lower at the nexus sites (4, 5, and 9) and broadwater sites (3 and 8), and significantly lower at the estuary sites (1, 2, 6, and 7; Figure 64). This spatial trend likely reflects a combination of proximity to major freshwater inputs and the efficiency of tidal flushing along the system. Salinity diminished with flow at all sites, however the difference between median and high flow salinities was most marked, especially in the Terranora Creek sites and in particular the Boyds Bay and Jack Evans Boat Harbour sites (12 and 15 respectively). These sites 12 and 15 are located in more poorly flushed (with respect to tide) backwaters and receive direct inputs of urban stormwater, hence their buffering capacity is lower than the channel reaches of Terranora Creek. The estuary sites tended to be completely fresh during high flow, but rapidly became brackish during median to low flow.

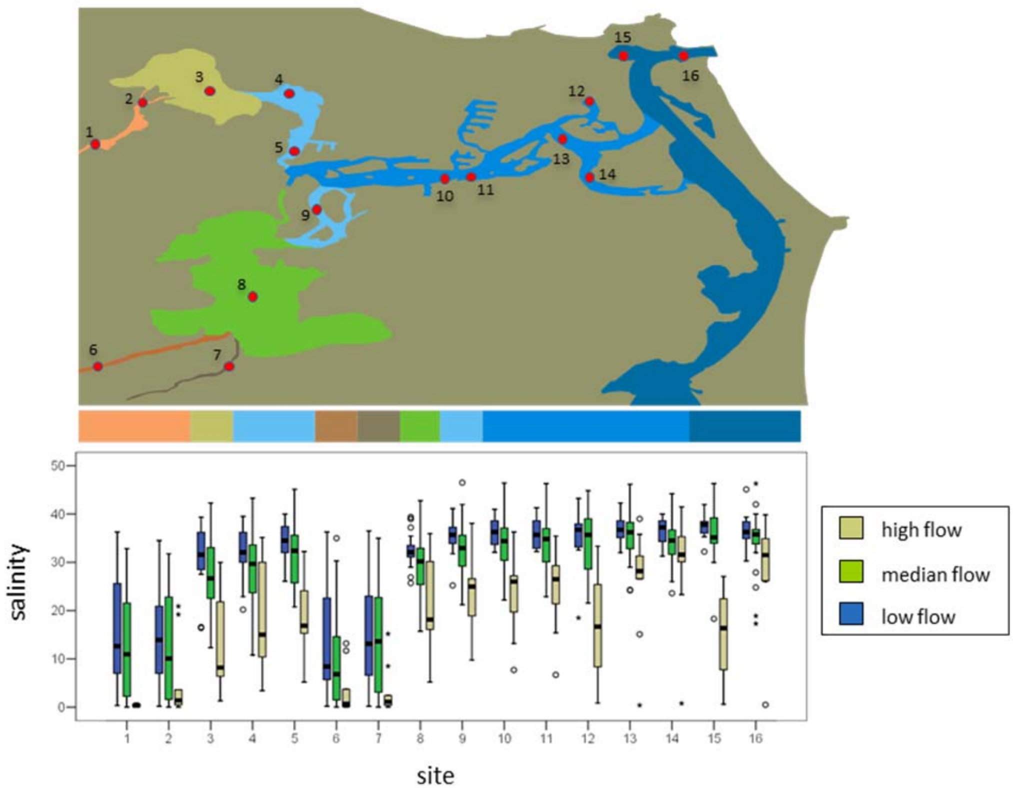


Figure 64 Spatial variation in salinity throughout the Cobaki-Terranora system.

6.1.2 Temporal variability

Temporal variation in salinity was marked by relatively brief spikes of lower salinity throughout the Terranora Creek, nexus and broadwater sites, and more periods of low salinity at the estuary sites (Figure 65). This pattern reflects the rapid recovery of estuarine conditions following high flow events throughout much of the system. The period of extremely high salinity (>36PSU) during 2010 most likely reflects poor probe calibration. While it is possible that broadwater sites may experience salinity in excess of seawater (36 PSU) due to evaporation during extreme low flow periods, this is unlikely to occur during a relatively wet period as in 2010.

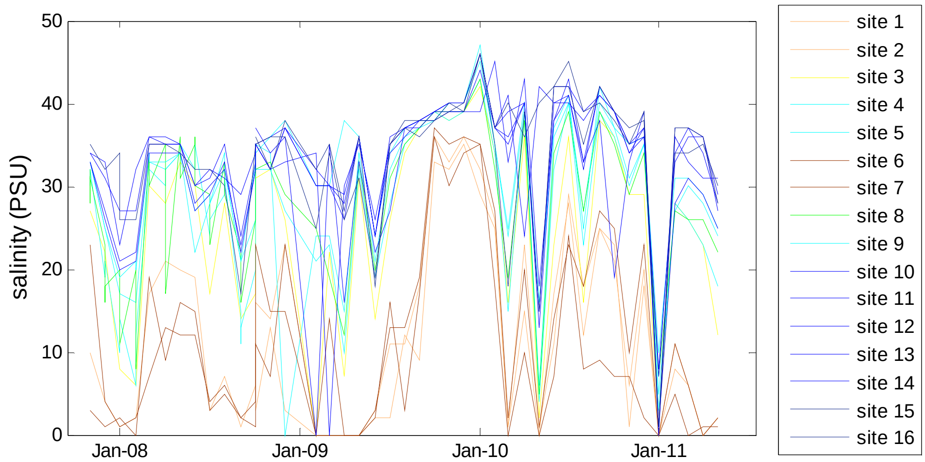


Figure 65 Temporal variation in salinity in the Cobaki-Terranora system during the study period.

6.1.3 Interannual variability

There was significant interannual variation in salinity in the Cobaki- Terranora system during the study period (Figure 66). Variability tended to be greatest during the summer / autumn, reflecting variability in the magnitude and timing of major rainfall events during the wet season.

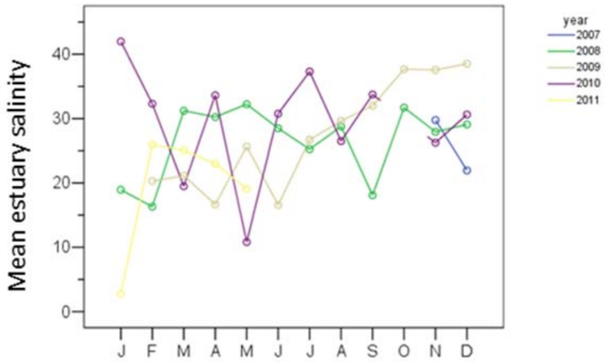


Figure 66 Interannual variation in salinity in the Cobaki-Terranora system during the study period.

6.2 Temperature

6.2.1 Spatial trends

There were no significant spatial trends in temperature in the Cobaki-Terranora system during the study period (Figure 67). Temperature tended to be lowest during low flow and highest during high flow. It is likely that this trend reflects the predominance of higher flows during the summer-autumn period, when ambient temperatures are greatest.

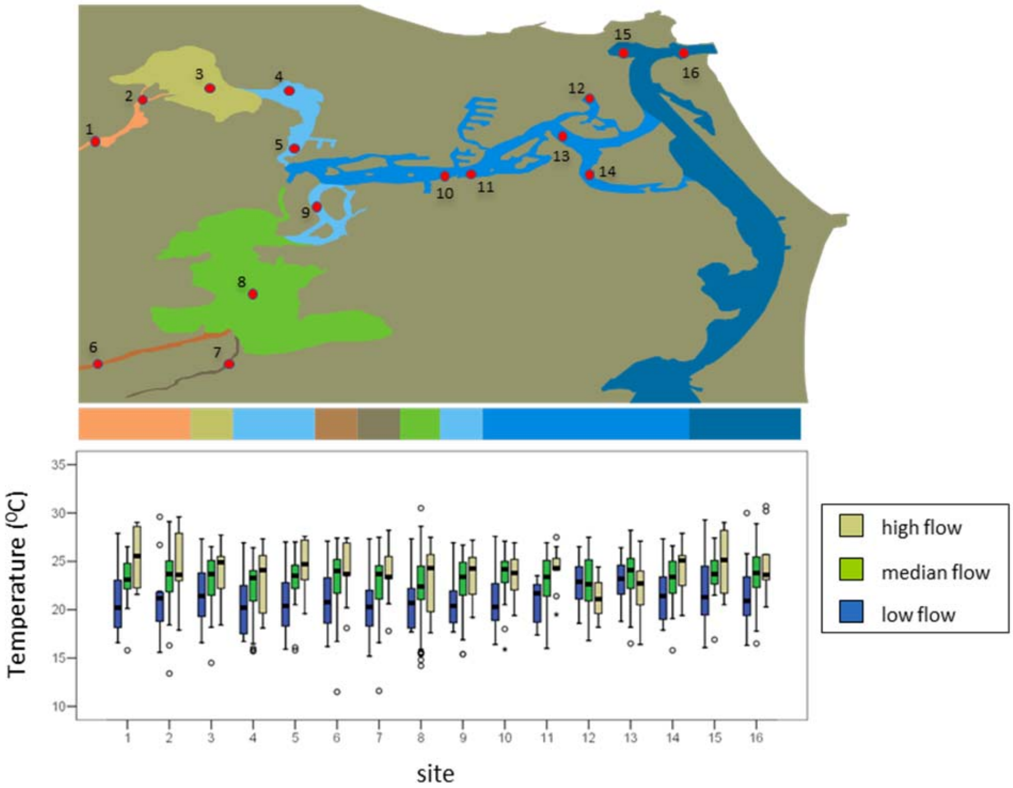


Figure 67 Spatial variation in temperature throughout the Cobaki-Terranora system.

6.2.2 Temporal variability

There was a clear seasonal cycle in temperature throughout the Cobaki-Terranora, with lowest temperatures recorded in winter to early spring, and highest temperatures recorded during summer to early autumn (Figure 68). There was considerable noise in the summer temperature maxima most likely introduced by the impacts of high flow events. The Boyds Bay site (12) was consistently out of phase with the remainder of sites, recording winter maxima and late spring minima.

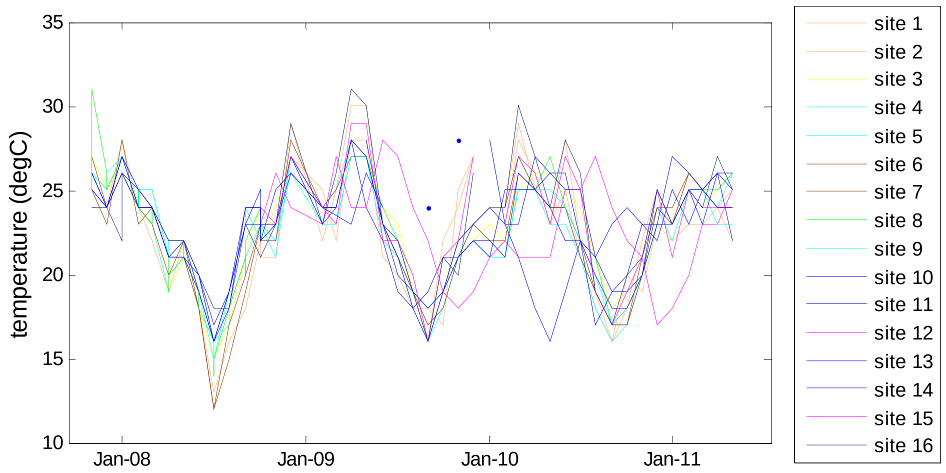


Figure 68 Temporal variation in temperature in the Cobaki-Terranora system during the study period.

6.2.3 Interannual variability

There was significant interannual variability in temperature in the Cobaki-Terranora system during the study period (Figure 69). The source of this variability was the markedly different seasonal cycle in temperature during 2008 compared to 2009 and 2010.

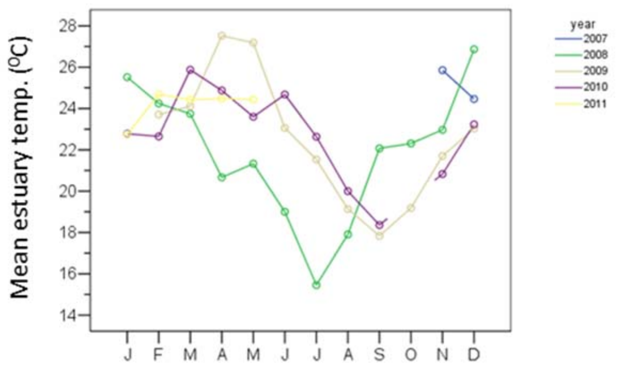


Figure 69 Interannual variation in temperature in the Cobaki-Terranora system during the study period.

6.3 pH

6.3.1 Spatial trends

pH tended to be lowest at the estuary sites, increasing towards the Tweed estuary sites (Figure 70). There was a significant reduction in pH with increasing flow at all sites. These spatial and flow related trends primarily reflect the mixing of freshwater (pH~ 6.5) with oceanic water (pH ~ 8.2) along the estuarine gradient of the system.

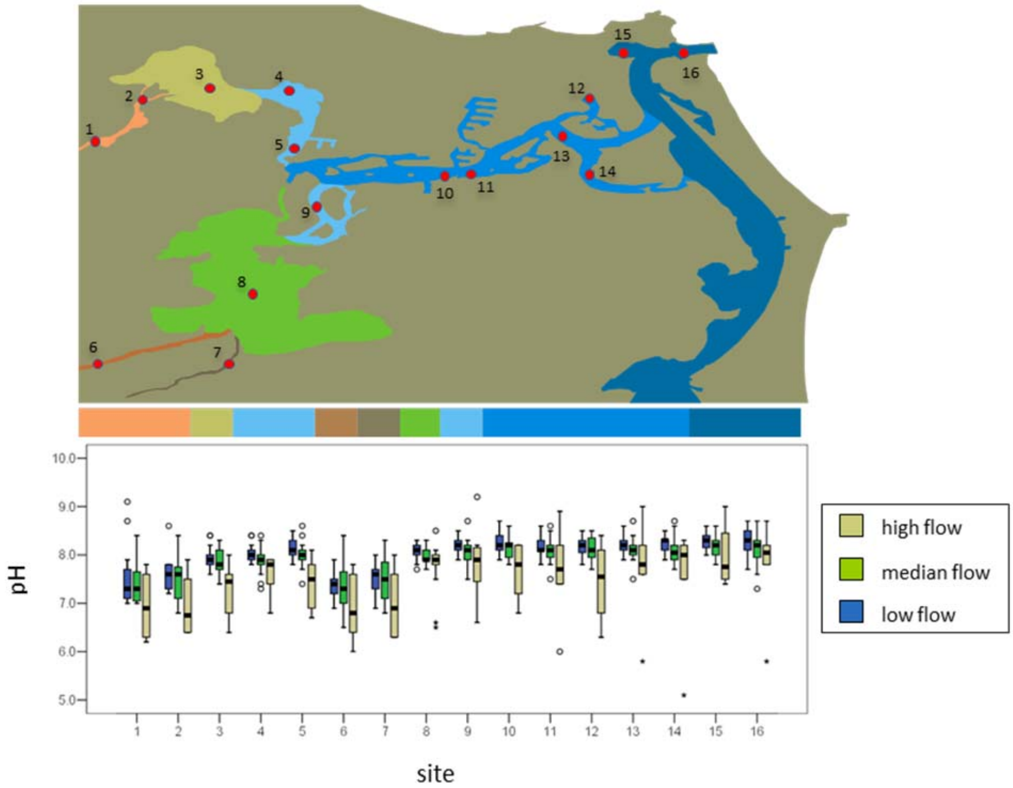


Figure 70 Spatial variation in pH throughout the Cobaki-Terranora system.

6.3.2 Temporal variability

Temporal variability in pH was related to variability in freshwater inputs, and tended to be greatest at the estuary sites reflecting the greater influence of freshwater inputs at these sites (Figure 71). Lowest pH values were recorded at the lower Terranora Creek and Tweed estuary sites during a high flow event in Jan 2011. The cause of these low values is unclear but may have been due to acid sulfate soil runoff impacting the lower Tweed estuary at the time. It is notable that pH was low at all sites in the Cobaki-Terranora at this time, while middle to upper Tweed estuary sites recorded higher pH values. This suggests that the source of acid water was located in the Cobaki-Terranora system and discharges impacted the lower Tweed estuary.

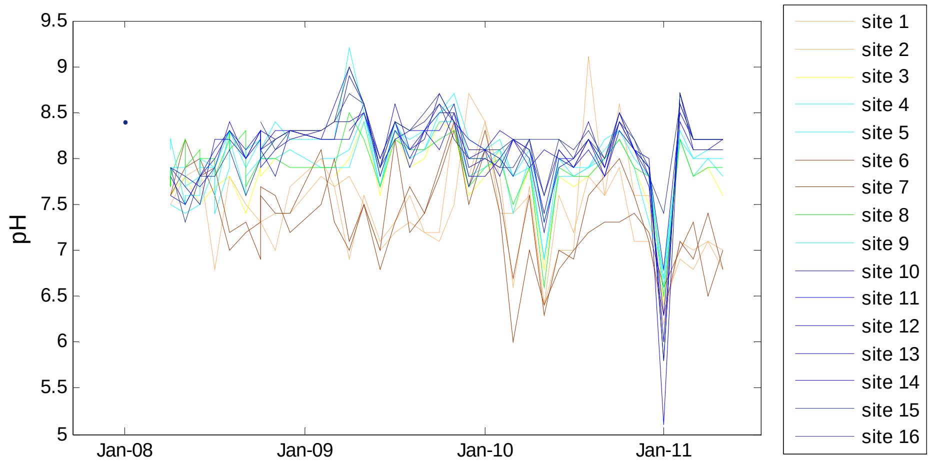


Figure 71 Temporal variation in pH in the Cobaki-Terranora system during the study period.

6.3.3 Interannual variability

There was significant interannual variability in pH in the Cobaki-Terranora system during the study period (Figure 72). This variability was primarily associated with variation in freshwater inflows but also due to the influence of the acid runoff event during Jan 2011.

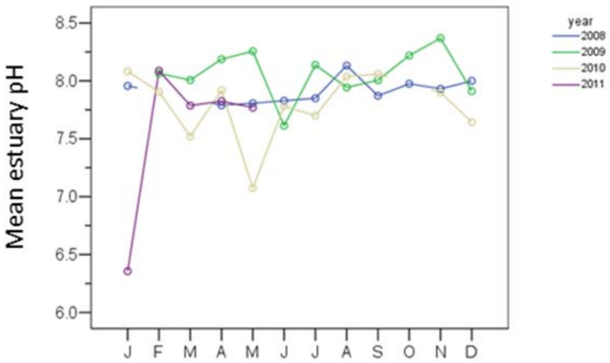


Figure 72 Interannual variation in pH in the Cobaki-Terranora system during the study period.

6.4 Total nitrogen

6.4.1 Spatial trends

Total nitrogen concentrations were greatest at the estuary sites and diminished through the broadwater and nexus sites, with lowest concentrations recorded throughout the Terranora Creek (Figure 73). There was a significant increase in TN concentrations during high flow, with this increase greatest at sites receiving direct freshwater inputs (e.g. the estuary sites, Boyds Bay and Jack Evans Boat Harbour).

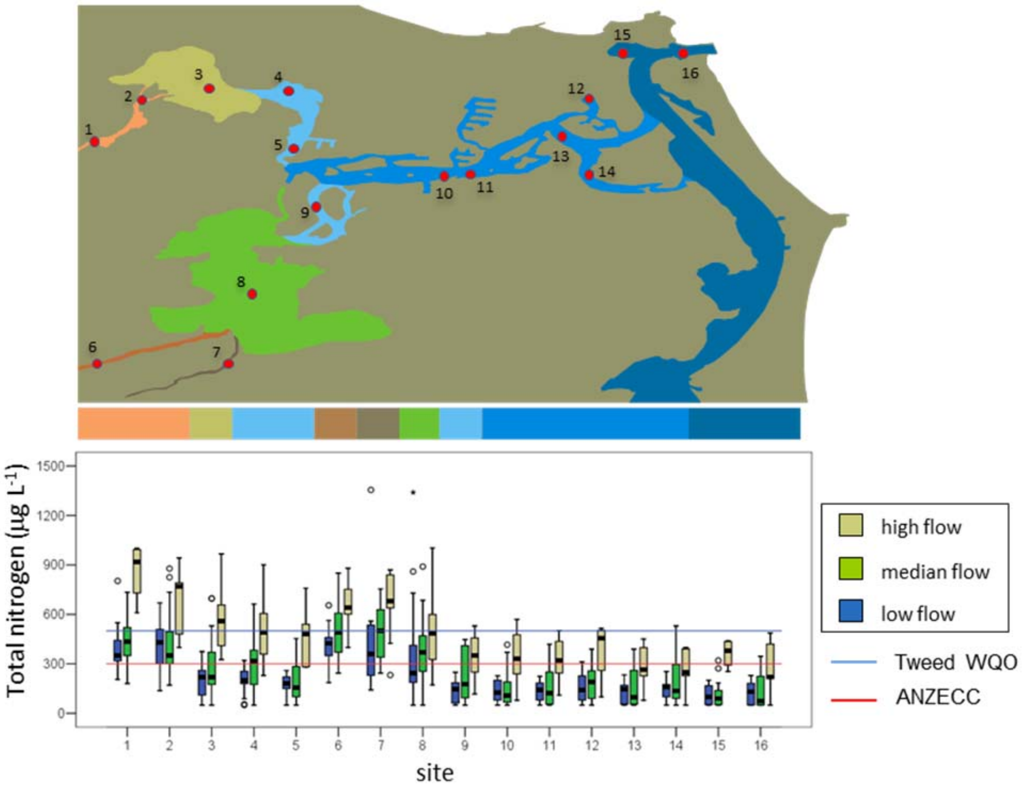


Figure 73 Spatial variation in TN concentrations throughout the Cobaki-Terranora system.

6.4.2 Temporal variability

Temporal variability in TN concentrations was primarily associated with variability in freshwater inflow (Figures 74 and 75). There was an apparent decrease in TN concentrations after Jan 2009, however this shift coincides with an obvious change in analytical protocol and data quality (evidenced by the discontinuation of Total Kjeldahl Nitrogen analysis and changes in detection limits).

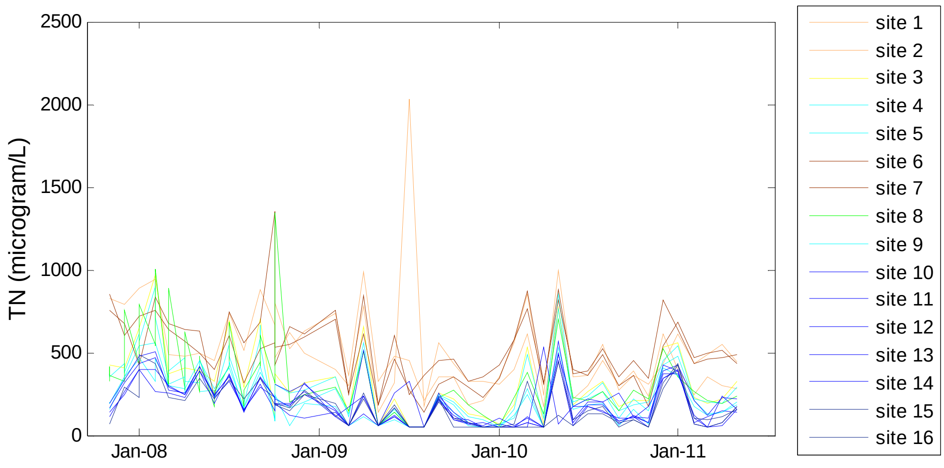


Figure 74 Temporal variation in TN concentrations in the Cobaki-Terranora system during the study period.

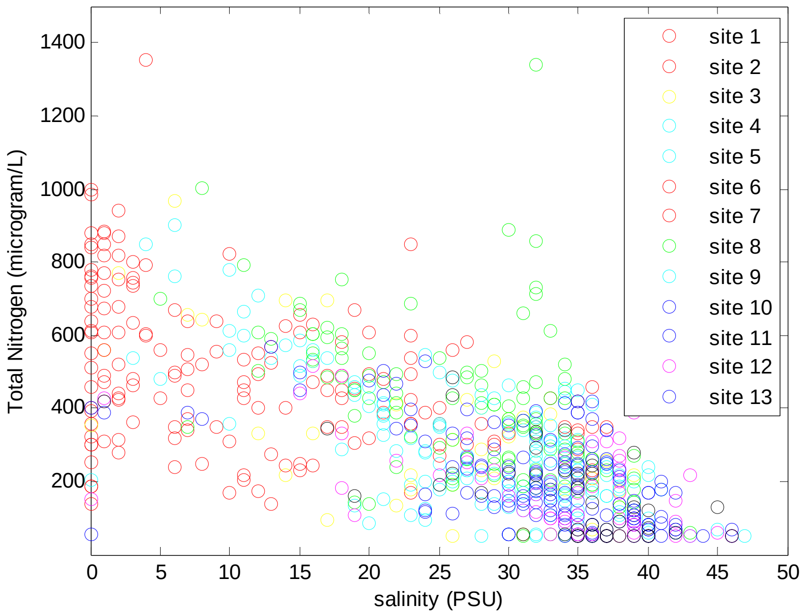


Figure 75 The relationship between salinity and TN concentrations in the Cobaki-Terranora system.

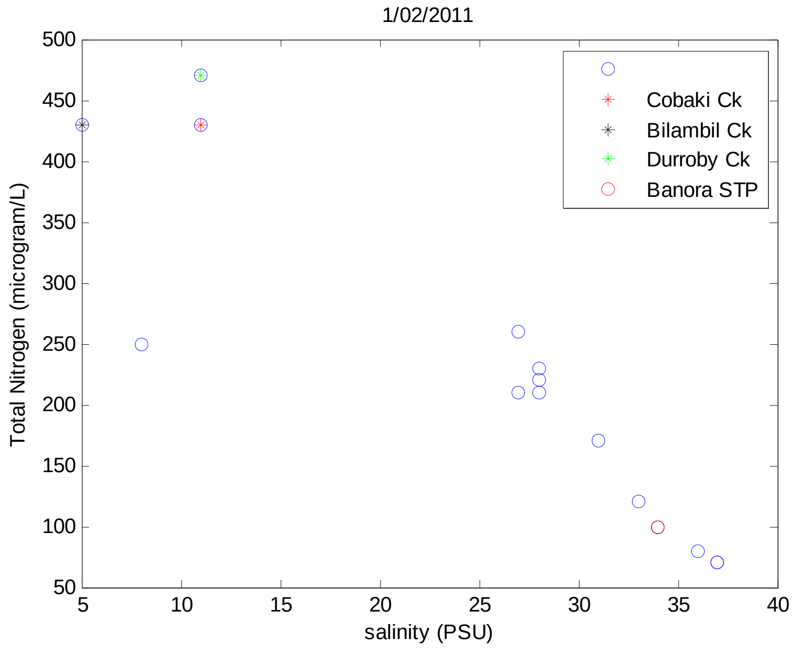


Figure 76 The relationship between salinity and TN concentrations in the Cobaki-Terranora system on 1/02/2011. The plot shows conservative mixing of all major freshwater inputs with oceanic water. The site downstream of the Banora Point STP plots in line with the rest of the data indicating minimal influence of this input over TN concentrations in Terranora Creek.

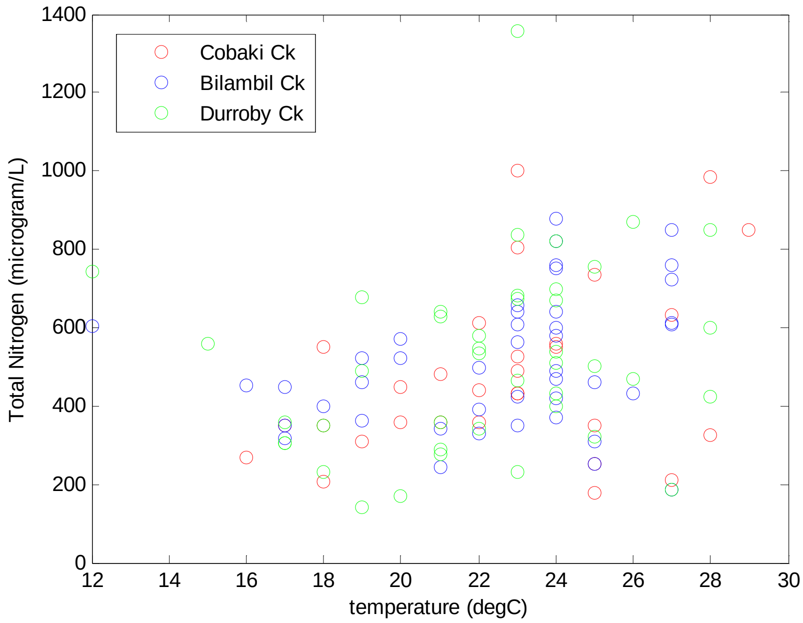


Figure 77 The relationship between temperature and TN concentrations at the estuarine sites.

6.4.3 Interannual variability

There was significant interannual variability during the study period, reflecting interannual variability in freshwater inputs (Figure 78). However this result is in part due to artefacts associated with analysis described above.

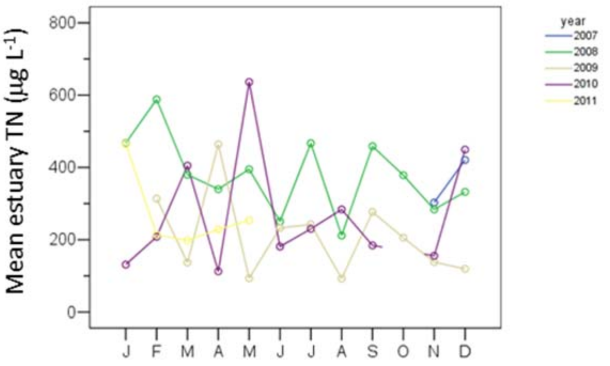


Figure 78 Interannual variation in TN concentrations in the Cobaki-Terranora system during the study period.

6.4.4 Comparison with water quality objectives

Input concentrations

Total nitrogen concentrations at the estuary sites (1, 2, 6 and 7) were above ANZECC (2000) guidelines[12] for more than 75% of the time during low and median flows and 100% of the time during high flow conditions indicating that the bulk of rural runoff does not comply with conditions for the maintenance of aquatic ecosystems in Southeast Australian estuaries (Figure 73). Figure 77 shows that TN concentrations in all of the estuary sites increased with temperature indicating that exceedence of the guidelines was more likely during summer.

Estuary concentrations

TN concentrations in Cobaki Broadwater were below the ANZECC (2000) guidelines for greater than 75% of the time during low flows, and greater than 50% during median flows. TN exceeded the guidelines for greater than 90% of the time in the Cobaki Broadwater during high flow. TN was lower than guidelines for greater than 50% of the time during low flow, but exceeded the guidelines for more than 50% and 75% of the time during median and high flows respectively. TN in Terranora Creek and the nexus were below the ANZECC guidelines for 90% and 75% of the time during low and median flows respectively, and exceeded the guideline during high flow for approximately 50% of the time.

According to the Tweed Water Quality Objectives, TN was below guidelines throughout most of the system during low and median flows, and exceeded the guidelines at the estuary sites for greater than 75% of the time and the broadwater sites for greater than 50% of the time during high flow.

6.4.5 Ecological and management implications

Total nitrogen concentrations were positively correlated with chlorophyll-*a* concentrations, indicating likely nitrogen limitation of phytoplankton growth in the Cobaki – Terranora Broadwater system (Figure 79)[13]. The TN:chl-*a* relationship in Figure XX

[12] Note that the line on Figure 73 indicates the ANZECC guideline for maintenance of aquatic health in estuaries. The TN threshold for lowland rivers is 350 µg L^{-1}, therefore compliance according to this guideline is slightly better.

[13] There was a much weaker correlation between TP concentrations and chlorophyll-*a* concentrations, however this result may be influenced by the association with biologically unavailable TP and TSS

shows that the ANZECC (2000) guideline for TN (300 μg L^{-1}) correlates to median and 80%ile chlorophyll-*a* concentrations of approximately 2.5 and 4 μg L^{-1} respectively, hence the adoption of the ANZECC (2000) guideline for TN is appropriate for complying with guidelines for chlorophyll-*a*.

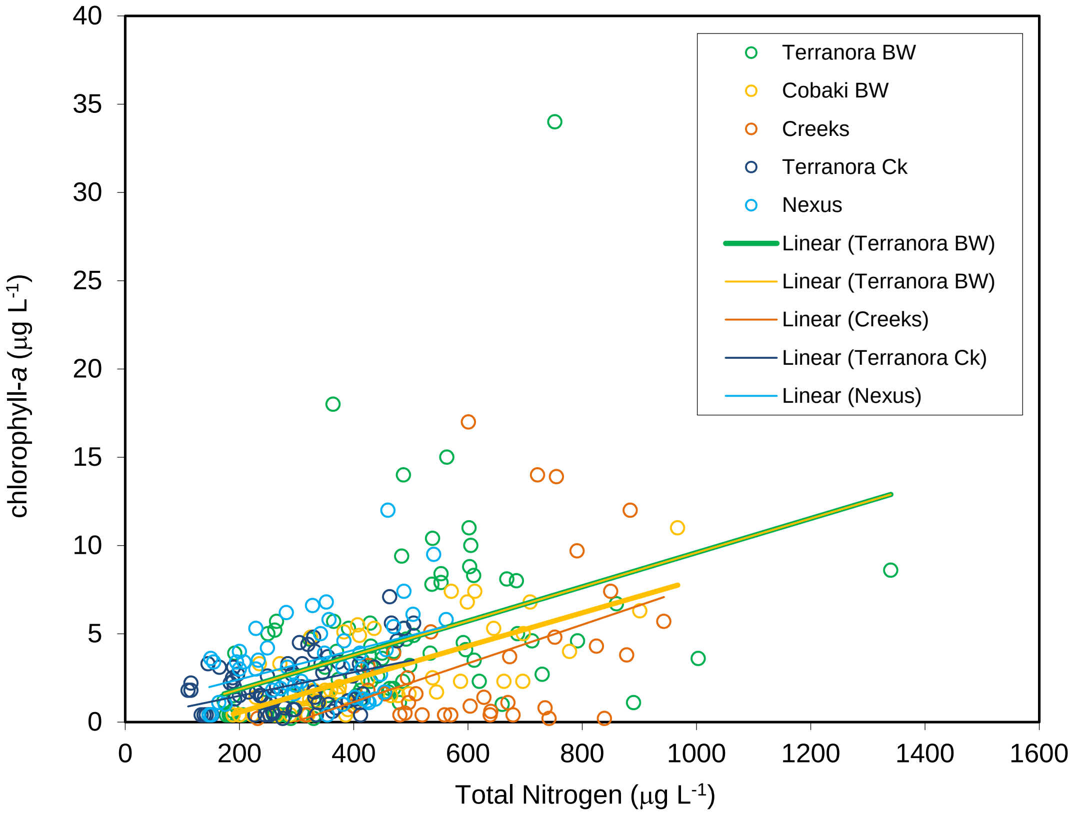


Figure 79 The relationship between TN and chlorophyll-*a* concentrations (EHMP data only).

concentrations. As such, assessment of N limitation should be regarded with caution and efforts to reduce nutrient inputs to the system should consider reducing both N and P.

6.5 Total phosphorus

6.5.1 Spatial trends

Total phosphorus concentrations were greatest at the estuary sites (1, 2, 6 and 7), were lower in the broadwaters and diminished to generally below detection limits throughout Terranora Creek (Figure 80). There was an increase in TP concentrations during high flow in the estuary sites and at the Boyds Bay and Jack Evans Boat Harbour sites (12 and 15). There was no discernible difference between TP concentrations during low and median flow. TP concentrations in Terranora broadwater (site 8) were highly correlated to TSS concentrations, and tended to increase with diminishing flow. These trends implicate wind / tidal driven resuspension of sediments as a significant source of TP in this part of the system.

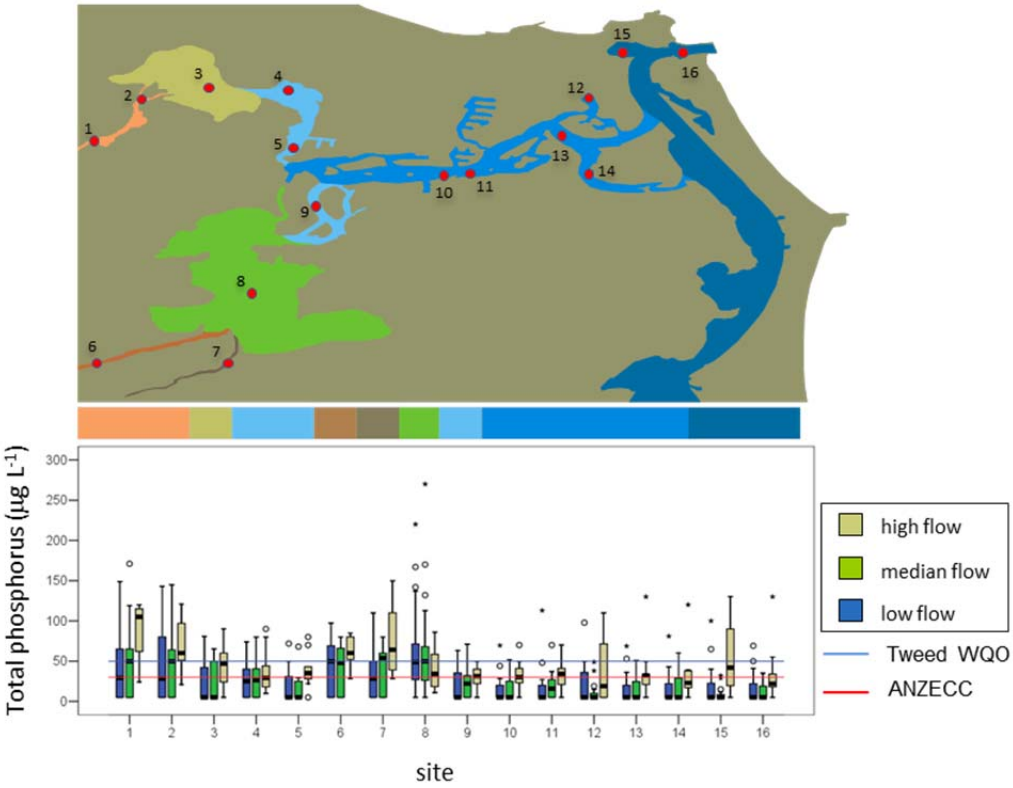


Figure 80 Spatial variation in TP concentrations throughout the Cobaki-Terranora system.

6.5.2 Temporal variability

There were no apparent seasonal trends in TP concentrations during the study period, with the bulk of variability explained by slightly higher concentrations during high flow (Figure 81). The dramatic decline in data quality post Jan 2009 greatly reduced the ability to assess temporal trends in the data.

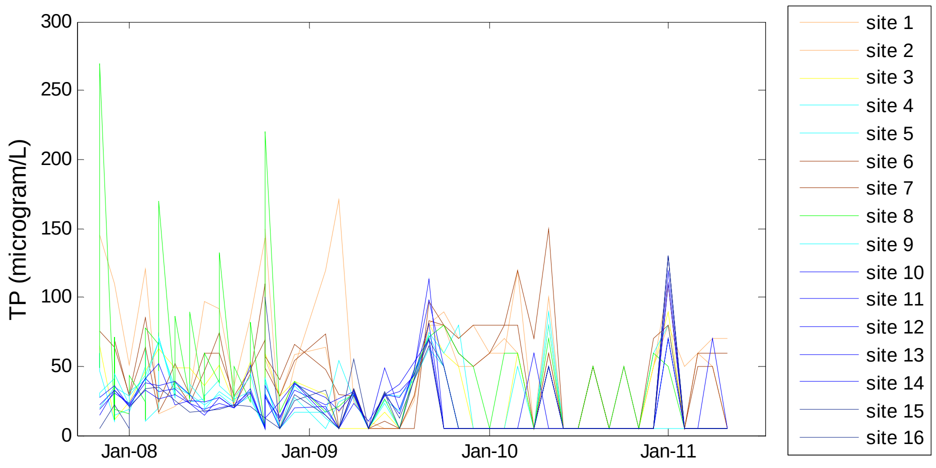


Figure 81 Temporal variation in TP concentrations in the Cobaki-Terranora system during the study period.

6.5.3 Interannual variability

There was significant interannual variability in TP concentrations, reflecting variability in rainfall across the study period (Figure 82). This result is unreliable however due to the variation in data quality.

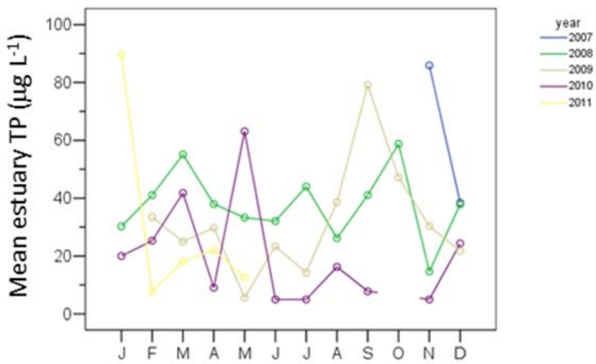


Figure 82 Interannual variation in TP concentrations in the Cobaki-Terranora system during the study period.

6.5.4 Comparison with water quality objectives

Input concentrations

Total phosphorus concentrations at the estuary sites (1, 2, 6 and 7) were above ANZECC (2000) guidelines for 50% of the time during low flow, more than 50% of the time during median flow and more than 75% of the time during high flow (Figure 80).

Estuary concentrations

TP concentrations throughout the Cobaki Broadwater and Terranora Creek were below the ANZECC (2000) guidelines for greater than 75% of the time during low flows and median flows, and exceeded guidelines for approximately 50% of the time during high flows. TP concentrations in the Terranora Broadwater exceeded the guidelines for 75% of the time during low and median flows due to the effects of wind / tide driven resuspension.

6.5.5 Ecological and management implications

There was only a weak positive correlation between TP and chlorophyll-*a* concentrations during the study. The strong positive correlation between TP and TSS, especially in the Terranora Broadwater indicates biologically unavailable phosphorus bound to inorganic sediment. As such, this material may mask any relationship between biologically available phosphorus and chlorophyll-*a*, therefore any assessment of the role of phosphorus in limiting productivity in this system cannot be readily made with the available data. It is recommended that any management actions aimed at the reduction of nutrients to the system consider both N and P.

6.6 Ammonium

6.6.1 Spatial trends

Ammonium concentrations tended to be highest at the estuary, broadwater and nexus sites and diminish to below detection limits in Terranora Creek (Figure 83). Relationships between ammonium concentrations and flow were spatially variable: there was a significant increase in concentrations throughout Terranora Creek and a slight increase in Cobaki Broadwater during high flow, while concentrations tended to be highest during low flow in Terranora Broadwater.

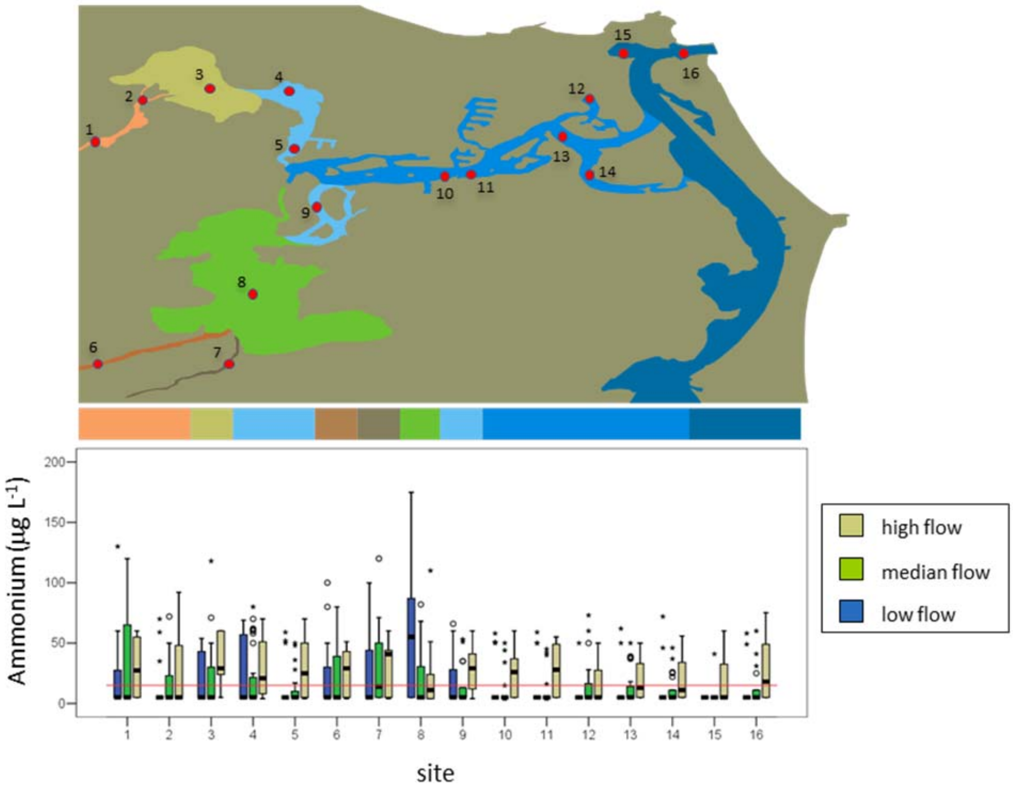


Figure 83 Spatial variation in ammonium concentrations throughout the Cobaki-Terranora system.

6.6.2 Temporal variability

Poor data quality post Jan 2009 means that only the first part of the dataset will be assessed for temporal trends. There was an increase in ammonium concentrations with the onset of drier conditions during the winter – spring of 2008 (Figure 84). This is trend was most apparent in the estuary sites, where ammonium tended to increase with salinity (Figure 85). This suggests internal recycling of nitrogen as ammonium from the sediments as the most likely cause. In contrast, elevated ammonium concentrations in the Terranora Broadwater were associated with high TSS concentrations implicating wind-driven resuspension of sediments as a likely cause.

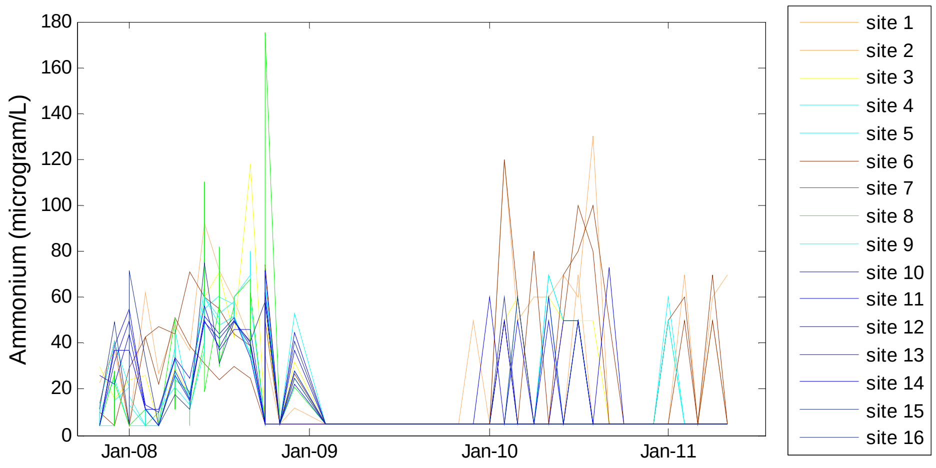


Figure 84 Temporal variation in ammonium concentrations in the Cobaki-Terranora system during the study period.

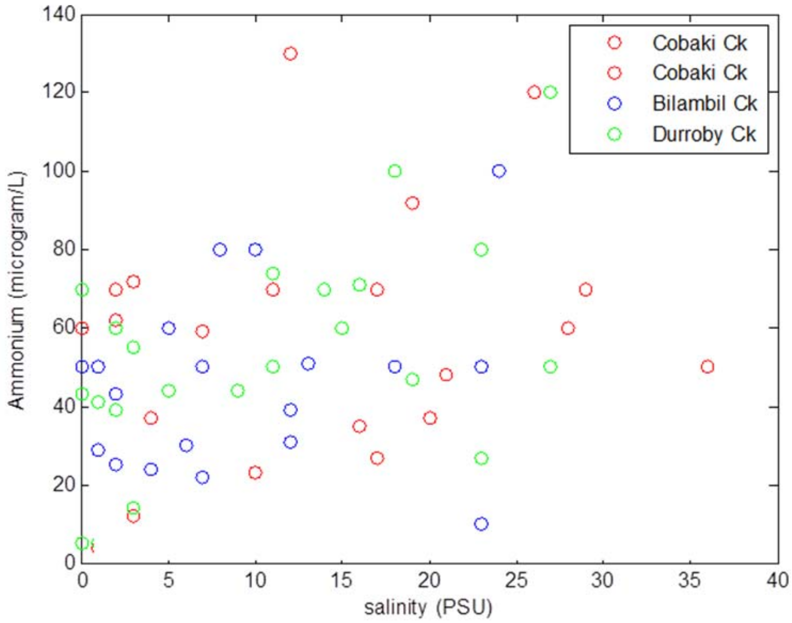


Figure 85 relationship between salinity and ammonium concentrations in the Cobaki-Terranora system during the study period.

6.7 Oxidised nitrogen

6.7.1 Spatial trends

There were few spatial or flow related trends in oxidised nitrogen concentrations during the study period. The notable exception was at the Durroby Creek estuary site which recorded elevated concentrations during all flow conditions, with the highest concentrations occurring during low flow (Figure 86).

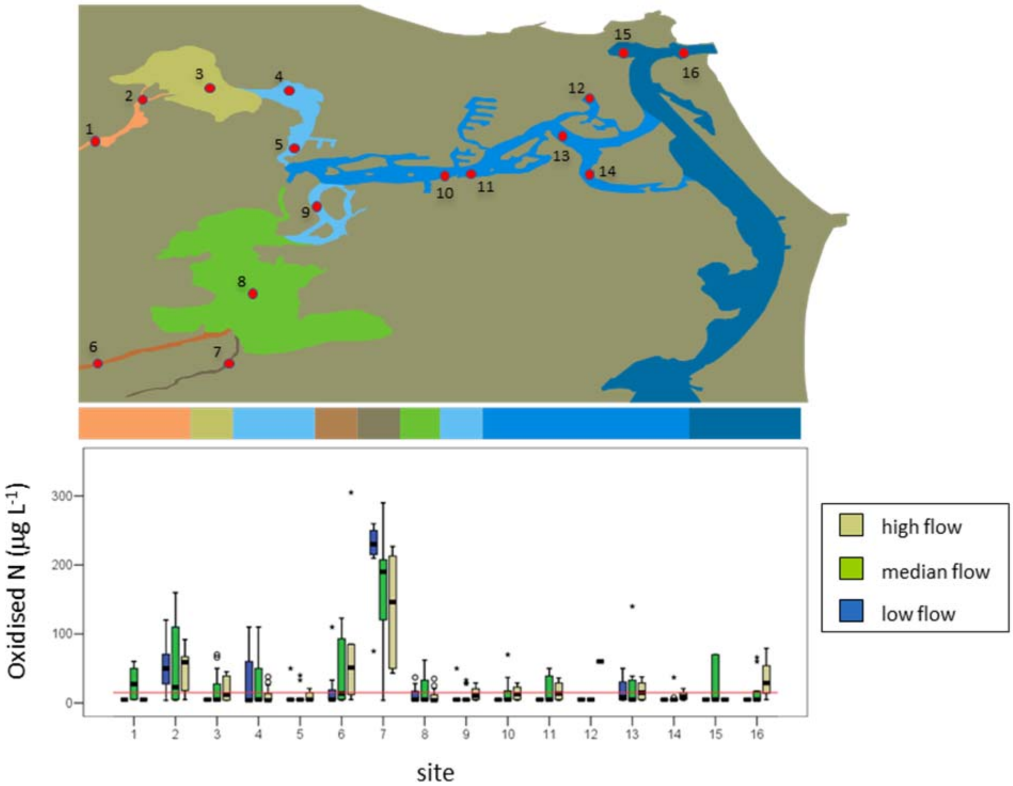


Figure 86 Spatial variation in oxidised N concentrations throughout the Cobaki-Terranora system.

6.8 Chlorophyll-a

6.8.1 Spatial trends

Chlorophyll-*a* concentrations tended to be highest at the estuary sites (1, 2, 6 and 7) and diminish to low concentrations throughout the broadwaters, nexus and Terranora Creek (Figure 87). There was a general increase in chlorophyll-*a* concentrations with flow throughout the broadwater and Terranora Creek sites, most likely explained by the stimulation of phytoplankton growth by freshwater nutrient inputs in the more marine dominated parts of the system where nutrient concentrations are commonly low enough to limit productivity. In contrast, chlorophyll-*a* concentrations increased with diminishing flow at the estuary sites. Nutrients are less likely to be limiting at the estuary sites, hence greater phytoplankton biomass occurs when water residence times increase (i.e. during low flow). Within individual sample runs, there was commonly a linear decrease in chlorophyll-*a* concentrations along the salinity gradient (Figure 89). This trend most likely reflects the dual controls of nutrient supply and residence times, both of which become more limiting towards the oceanic end of the system.

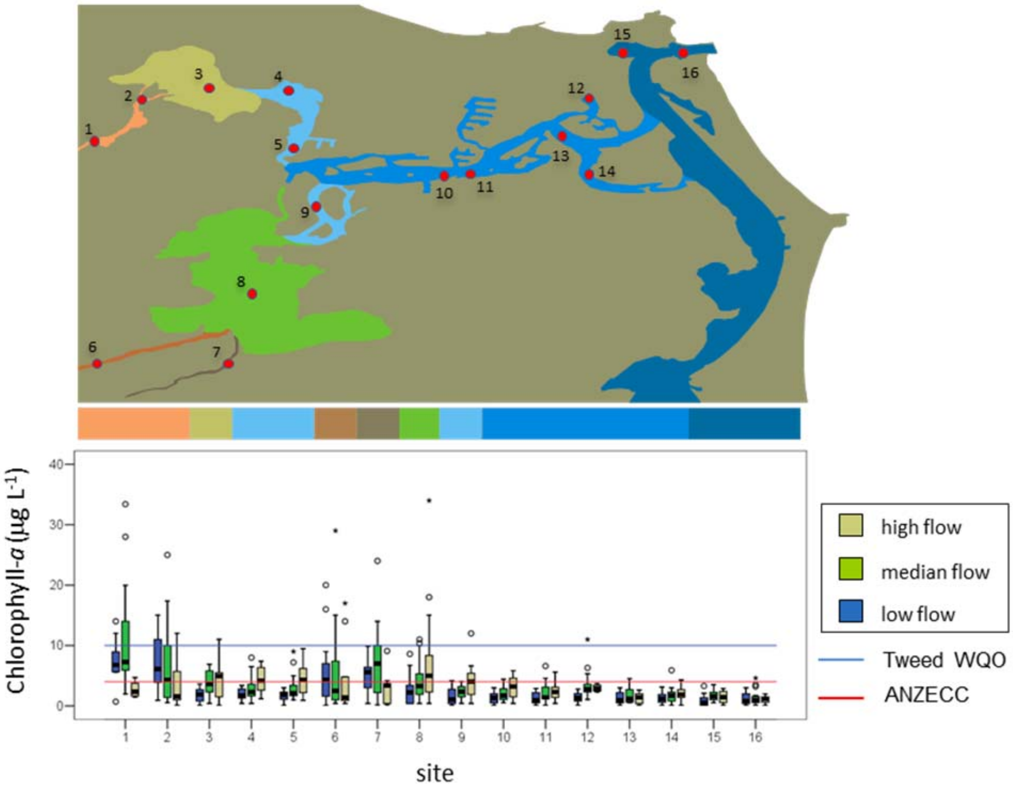


Figure 87 Spatial variation in chlorophyll-*a* concentrations throughout the Cobaki-Terranora system.

6.8.2 Temporal variability

There was a clear increase in chlorophyll-*a* concentrations during summer reflecting seasonal patterns of the primary drivers of phytoplankton growth (temperature, freshwater nutrient inputs, and light; Figure 88).

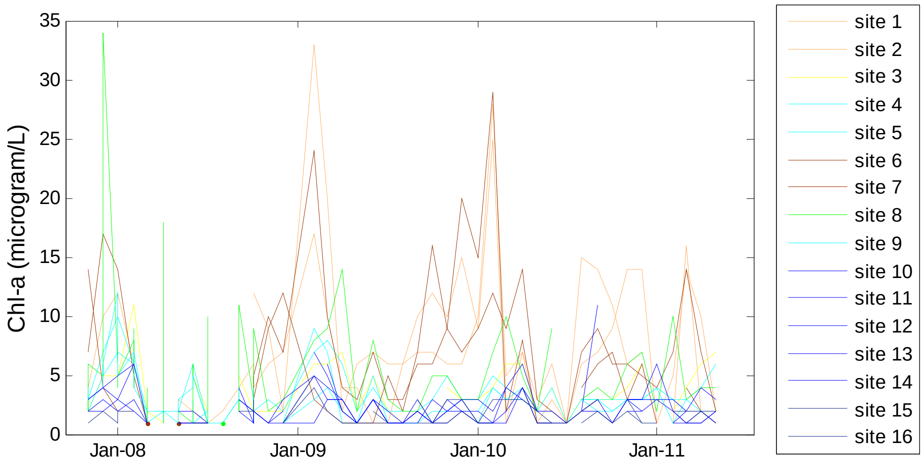


Figure 88 Temporal variation in chlorophyll-*a* concentrations in the Cobaki-Terranora system during the study period.

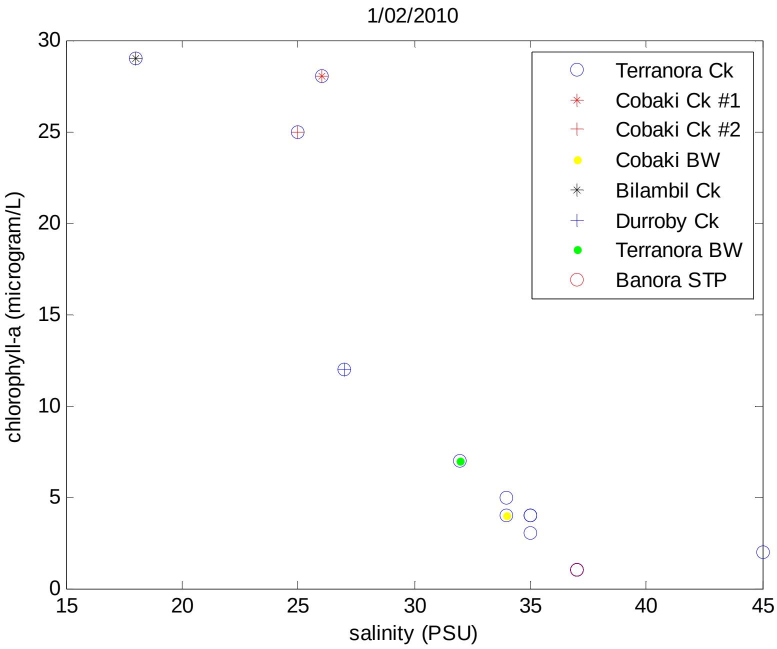


Figure 89 The relationship between salinity and chlorophyll-*a* concentrations on the 1/02/2010.

6.8.3 Interannual variability

There was significant interannual variability in chlorophyll-*a* concentrations during the study period (Figure 90). This result arises due to variability in the timing and magnitude of freshwater inflows during the summer – autumn wet season.

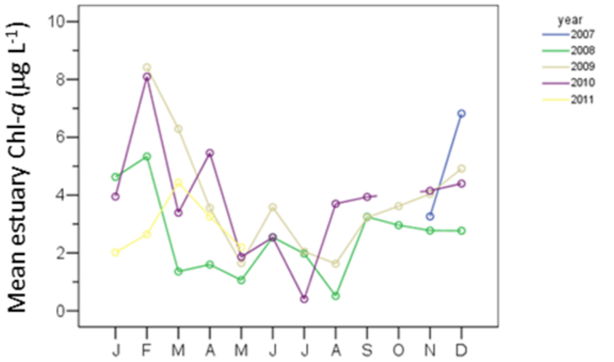


Figure 90 Interannual variation in chlorophyll-*a* concentrations in the Cobaki-Terranora system during the study period.

6.8.4 Ecological and management implications

Data show that the estuarine reaches of the Cobaki Terranora Broadwater system (sites 1, 2, 6 and 7) experience relatively severe phytoplankton blooms during dry and median flow periods suggesting moderate eutrophication. However, there was no clear relationship between chlorophyll-*a* concentrations and dissolved oxygen saturation in these reaches as might be expected if phyto-detritus was implicated in organic enrichment of sediments. This may be due to the confounding effect of oxygen production by phytoplankton which (depending on the time of day dissolved oxygen is measured) will mask the true oxygen status of the system. A more indicative measure of the impacts on oxygen status is the diel swing in oxygen saturation (i.e. the difference between dissolved oxygen measured at dawn when saturation will be at its lowest for the 24 hour cycle, and at around 2pm when it will be at its greatest).

The Cobaki and Terranora Broadwaters both experience periodic phytoplankton blooms which increase the organic matter content of sediments and thereby impact on oxygen status of the system. This is supported by the negative relationship between chlorophyll-*a* concentrations and DO saturation in the broadwaters (Figure 91). These data indicate that the ANZECC (2000) guideline for chlorophyll-*a* (4 $\mu g\ L^{-1}$) corresponds to a median dissolved oxygen saturation of ~80%, hence the ANZECC (2000) guideline for chlorophyll-*a* seems appropriate for the broadwaters. Data from the nexus sites (4, 5, and 9) indicate that the ANZECC (2000) guideline for chlorophyll-*a* corresponds to a median dissolved oxygen saturation of <80%, hence the ANZECC (2000) guideline for chlorophyll-*a* may be slightly high for these sites (Figure 92). Data from the Terranora Creek and Tweed entrance sites (10 – 16) indicate that the ANZECC (2000) guideline for chlorophyll-*a* corresponds to a median dissolved oxygen saturation of ~80%, hence the ANZECC (2000) guideline for chlorophyll-*a* seems appropriate for these sites (Figure 93).

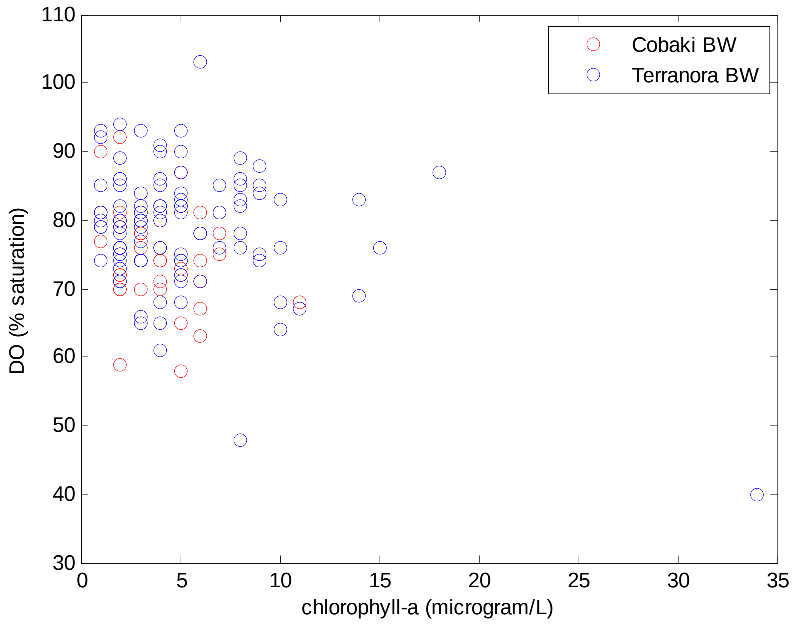


Figure 91 The relationship between chlorophyll-*a* concentrations and DO saturation in the Cobaki and Terranora Broadwaters.

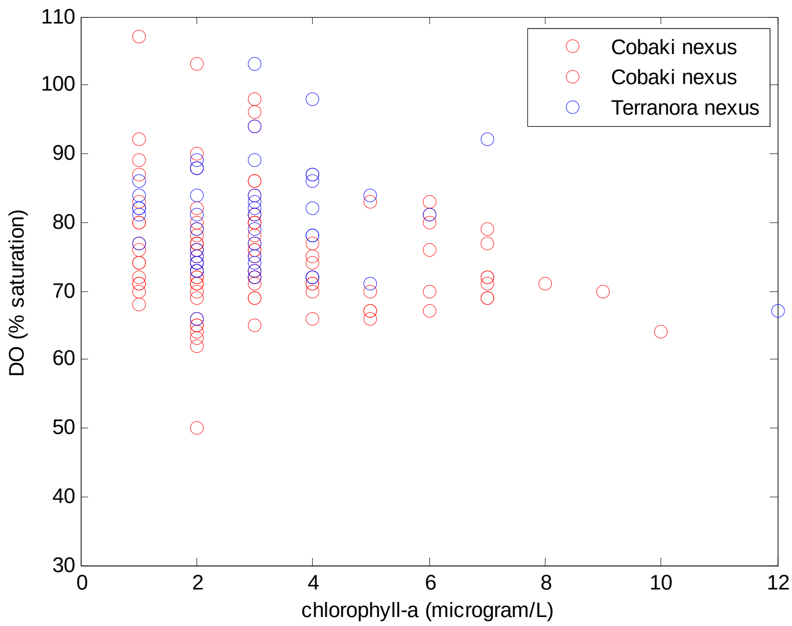


Figure 92 The relationship between chlorophyll-*a* concentrations and DO saturation in the Cobaki and Terranora nexus sites.

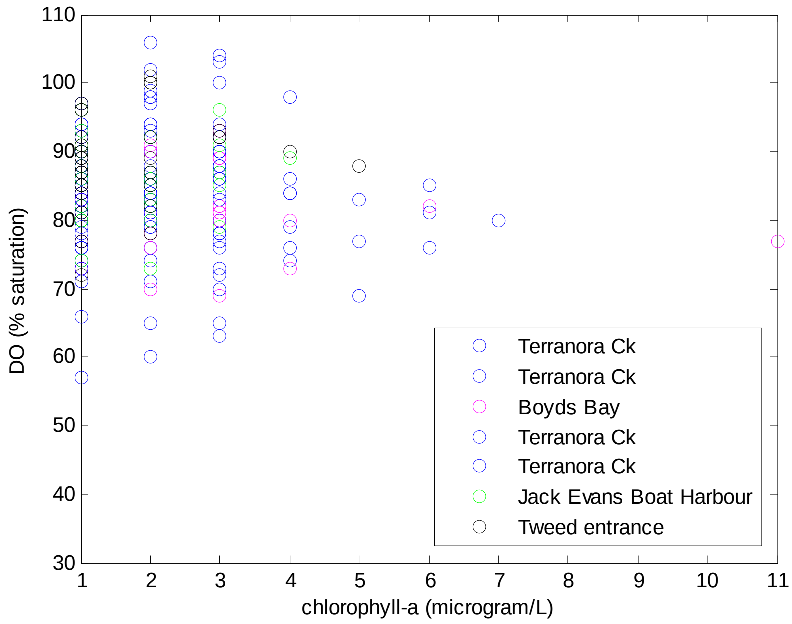


Figure 93 The relationship between chlorophyll-*a* concentrations and DO saturation in the Terranora Creek and Tweed River sites.

6.9 Total suspended solids

6.9.1 Spatial trends

Total suspended solids (TSS) during the study period were consistently highest at the estuarine and broadwater sites and decreased in the Terranora Creek sites (Figure 94). Terranora Broadwater recorded some extremely high TSS concentrations due to the occurrence of wind / tide driven resuspension of sediments in Trutes Bay. In general, there was an increase in TSS with flow at the estuarine sites and at the Boyds Bay and Jack Evans Boat Harbour sites (12 and 15) indicating the influence of more turbid freshwater runoff. There was no indication of elevated TSS concentrations at site 2 relative to site 1, indicating that the current monitoring strategy did not detect significant impacts due to runoff from the Cobaki Lakes construction site (see further discussion of this issue below).

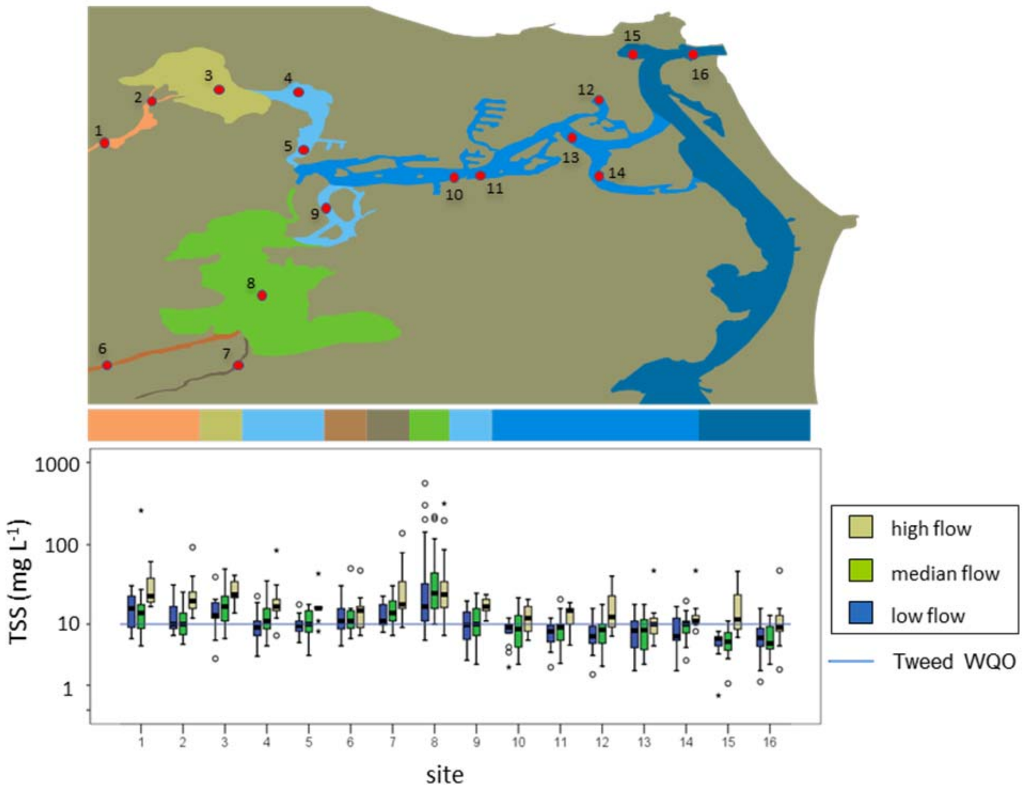


Figure 94 Spatial variation in TSS concentrations throughout the Cobaki-Terranora system.

6.9.2 Temporal variability

There were no clear temporal trends in TSS throughout the system (Figure XX). The first part of the dataset (up till Oct 2008) comprises data collected as part of the Cobaki Terranora Broadwaters EHMP (IWC 2009), and as such includes multiple sites in the Terranora and Cobaki Broadwaters. Figures 95 and 96 show that these sites routinely captured the high turbidity events associated with resuspension in the broadwaters, while subsequent sampling largely missed this phenomenon. Likewise, the current routine sampling strategy most likely misses the extreme spikes in TSS associated with high flow events. An example of this is during the runoff event which occurred between the 4th and the 6th May 2010 in response to over 150mm of rain on the 4th and a further 11mm of rain on the 5th. The aerial photo taken on the 6th May (Figure 97) shows a highly turbid plume of runoff from the Cobaki Lakes development site impinging on the estuary and Cobaki Broadwater. However TSS data collected on the 5th May shows no extremely high concentrations as might be expected from the aerial photo (Figure 98). It is possible that water quality at the time of sampling was dominated by upper catchment runoff and runoff from the low lying Cobaki Lakes site was impounded until water levels in the estuary receded sufficiently to allow it to drain. Alternatively, there may be serious issues with the analysis of TSS during this study. A common problem associated with the analysis of high flow TSS samples is the settling out of suspended material during sample storage and the incomplete recovery of this material upon analysis. Regardless of the reasons for this anomaly between observations and the data, it is clear that the current strategy does not adequately assess TSS in the Cobaki Terranora system.

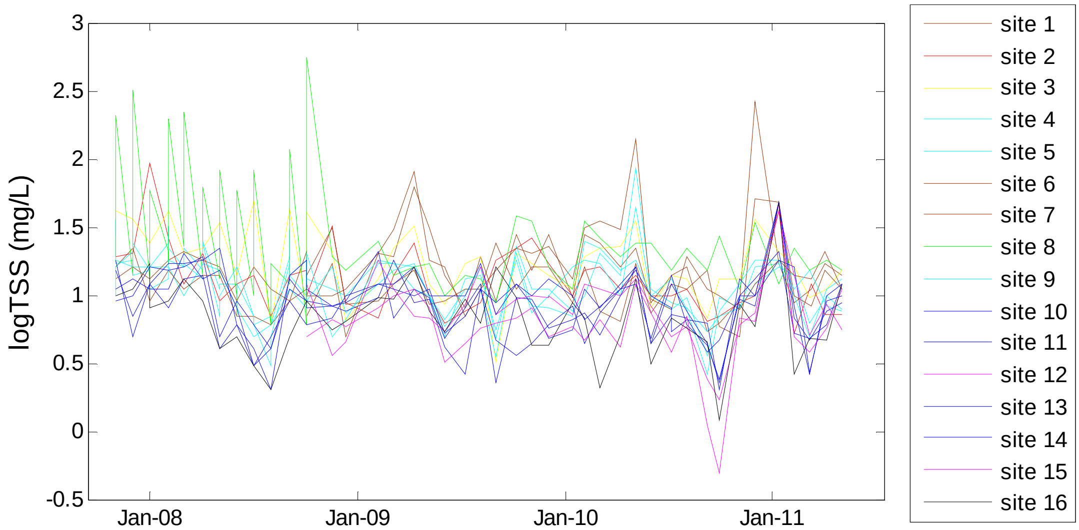


Figure 95 Temporal variation in TSS concentrations in the Cobaki-Terranora system during the study period.

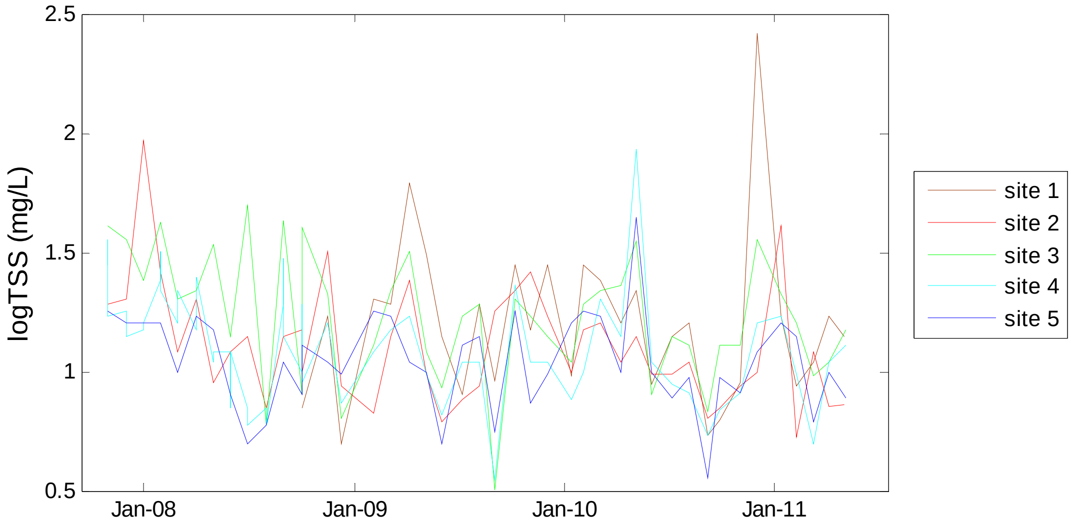


Figure 96 Temporal variation in TSS concentrations in the Cobaki Broadwater system during the study period.

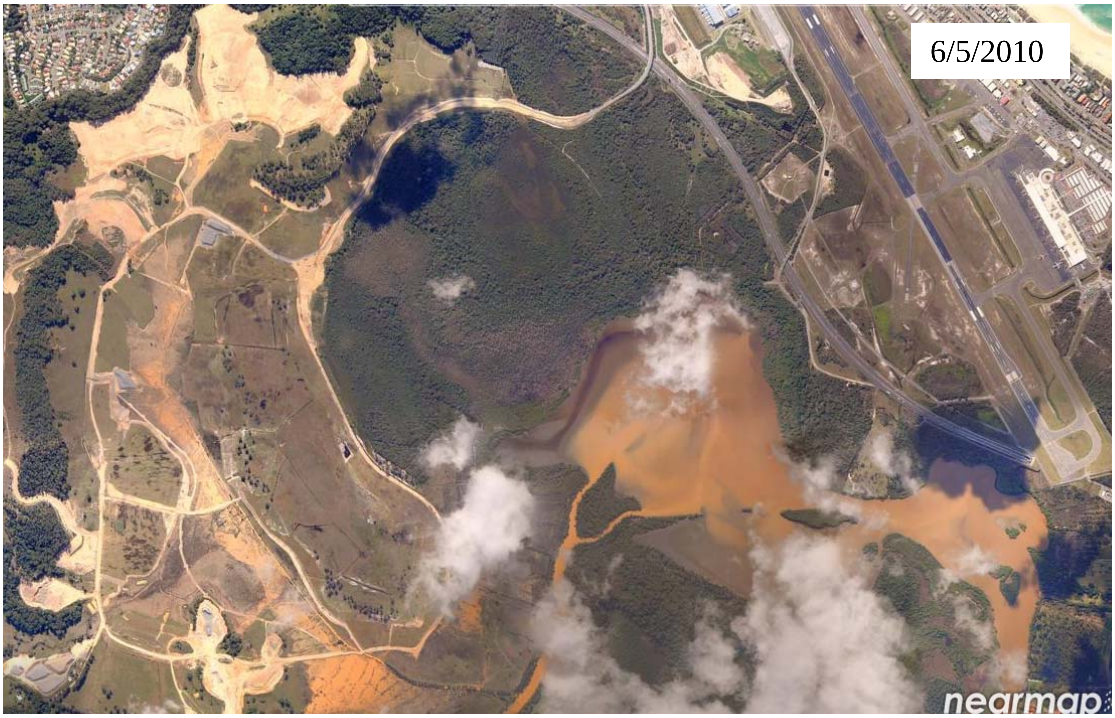


Figure 97 Aerial photo of the Cobaki Broadwater on the 6/05/2010 following over 150mm of rain over the preceding two days. The image clearly shows highly turbid runoff from the Cobaki Lakes construction site influencing the broadwater and nexus reaches.

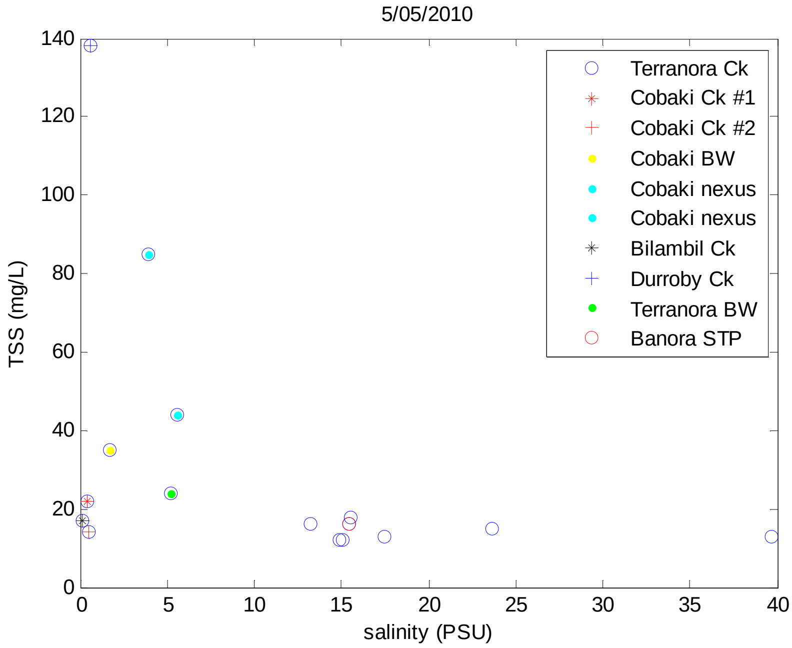


Figure 98 The relationship between salinity and TSS on the 5/05/2010. There is no evidence of the highly turbid runoff clearly shown in the aerial photo taken a day later. Potential reasons for this anomaly are given in the text.

6.9.3 Interannual variability

There was significant interannual variation in TSS during the study period (Figure 99), reflecting variability in the timing and magnitude of freshwater inputs and artefacts associated with changed sampling regime described above. Variability tends to be least during late winter when freshwater inflow and winds are at a seasonal low.

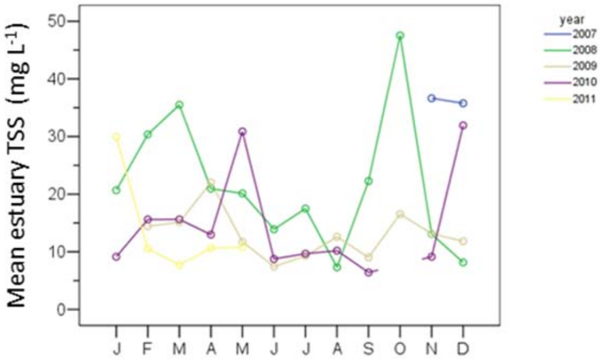


Figure 99 Interannual variation in TSS concentrations in the Cobaki-Terranora system during the study period.

6.9.4 Ecological and management implications

High turbidity (TSS) in the Cobaki – Terranora Broadwater system occurs in response to wind / tide driven resuspension events (mainly in the Terranora Broadwater), and during high flow runoff events. As flows diminish after floods, the settling of fine grained TSS in the broadwater and nexus reaches of the system most likely contribute to the bed load of material available for resuspension by winds and tides. Light limitation due to TSS is likely to be a significant issue for the ecology of the estuarine, broadwaters and nexus reaches. In particular, seagrasses are sensitive to extended periods of high turbidity which can cause major dieback and loss of beds. As such, management strategies should be directed at reducing the generation of fine grained TSS during runoff events. In general, the Terranora Creek reach of the system is somewhat buffered against the impacts of TSS due to the predominance of marine conditions which tend to flocculate out material and maintain good water clarity.

It appears that the current monitoring strategy does not adequately sample high flow events and may therefore give a false indication of the severity of their impacts. The highly turbid runoff emanating from the Cobaki Lakes development site is of particular concern as a large proportion of this material will be deposited in the broadwater and nexus reaches. This may cause smothering of benthic habitats, and the fine bed load is subsequently available for resuspension by tidal currents and wind waves. It is recommended that any compliance monitoring of Cobaki Lakes development should include routinely maintained turbidity loggers situated upstream and downstream of the confluence between the main drainage from the site and Cobaki Creek. Council should also consider opportunistic sampling during high flow events to better understand the nature of TSS inputs to the system. Finally, it is recommended that sampling and analysis protocols be reviewed to identify any issues that may compromise data quality.

6.10 Secchi

Secchi data for the study period are presented in Figure 100. Due to various artefacts associated with collection these data are not suitable for analysis in this report. In particular data were not collected at some sites, while at others secchi depths reaching the bottom were recorded as greater than an arbitrary depth relating to the specific location of sampling. There is therefore no way of ascertaining an accurate secchi depth for the bulk of samples at the clearer Terranora Creek sites. These values were removed from the dataset, hence Figure XX is skewed towards poorer water quality times and does not provide any indication of true spatial trends.

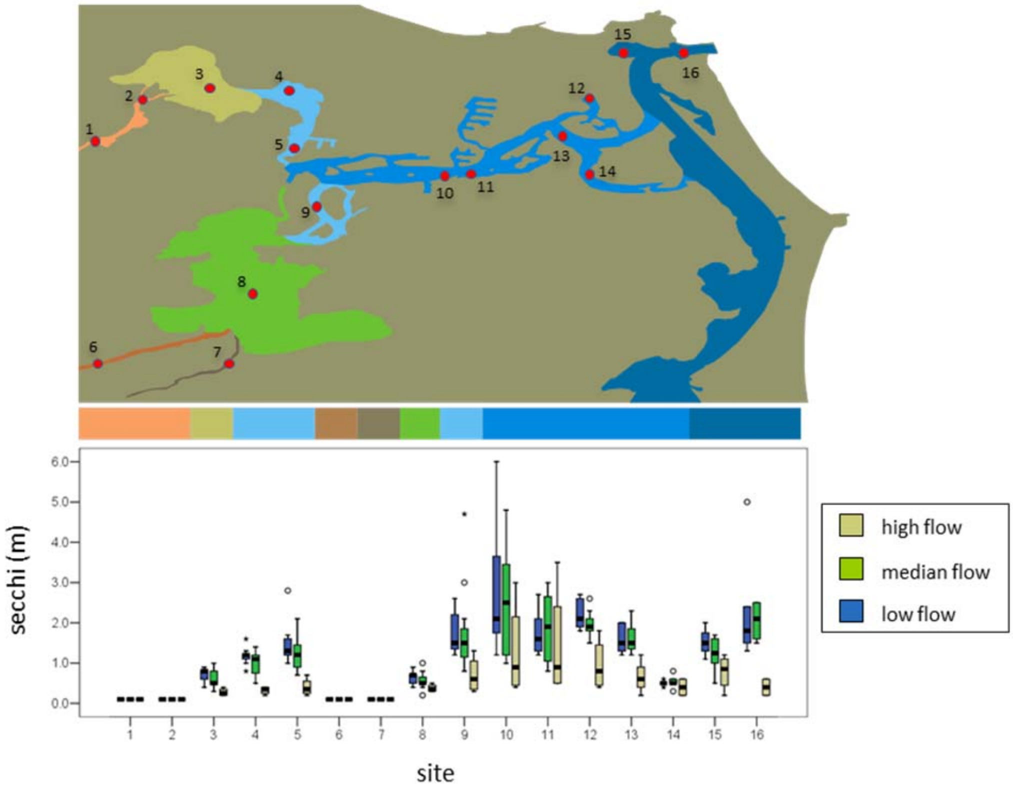


Figure 100 Spatial variation in secchi depths throughout the Cobaki-Terranora system. Note that data were not collected at the estuarine sites.

6.11 Dissolved oxygen

6.11.1 Spatial trends

Dissolved oxygen saturation was generally below 100% along the entire system, and was consistently poorest at the estuary sites and best at the lower Terranora Creek and Tweed entrance sites (Figure 101). There was a general decrease in DO saturation with increasing flow throughout the broadwater and Terranora Creek sites, reflecting the influence of freshwater inflows. In contrast, flow dependence was not as clear at the estuary sites most likely due to the relatively greater influence of internal processes which influence DO saturation (e.g. phytoplankton O_2 production and sediment O_2 consumption). Internal processes are more important in reaches where residence times are longer.

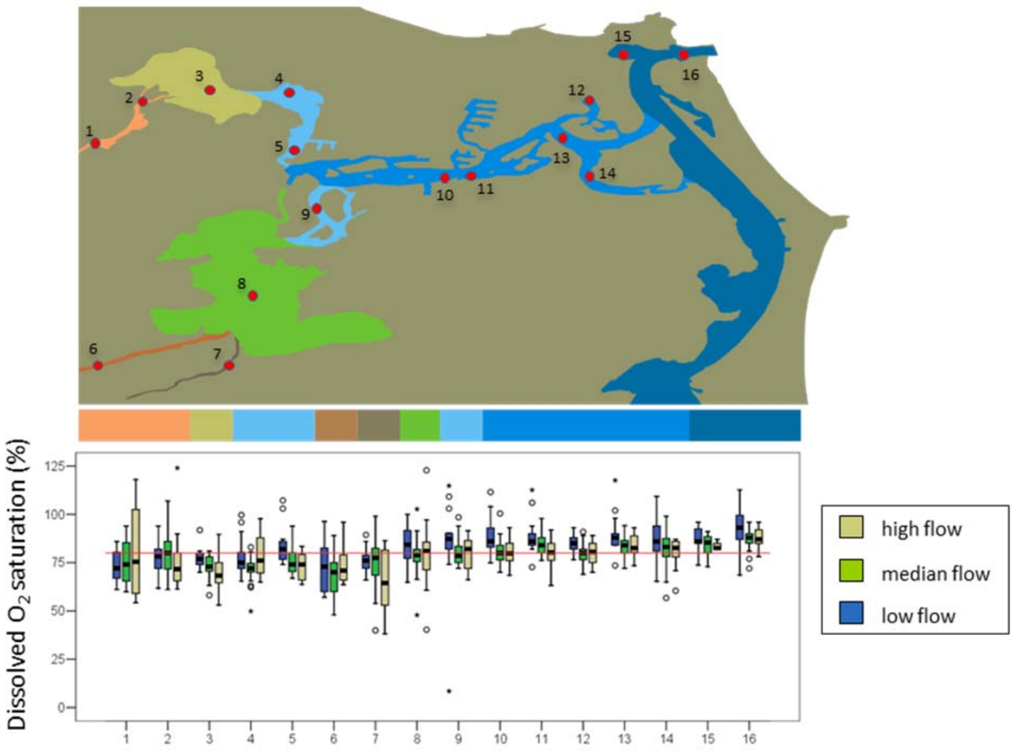


Figure 101 Spatial variation in dissolved oxygen saturation throughout the Cobaki-Terranora system.

6.11.2 Temporal variability

There was a weak seasonal trend of good DO saturation during spring and poor saturation during summer (Figure 102). This primarily reflects the influence of freshwater inflows over processes that impact on DO saturation. Figure XX shows that there was negative relationship between TN concentrations and DO saturation apparent during all flow conditions.

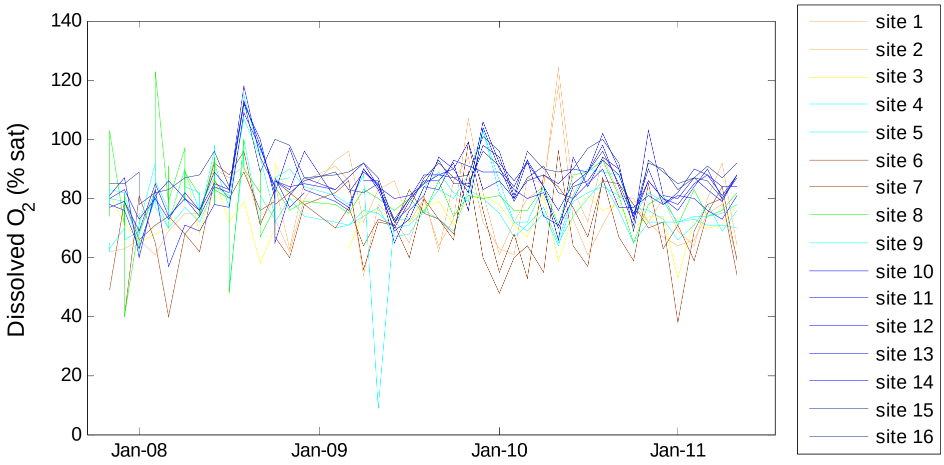


Figure 102 Temporal variation in dissolved oxygen saturation in the Cobaki-Terranora system during the study period.

6.11.3 Interannual variability

There was significant interannual variability in dissolved oxygen saturation throughout the Cobaki Terranora Broadwater system during the study period (Figure 103).

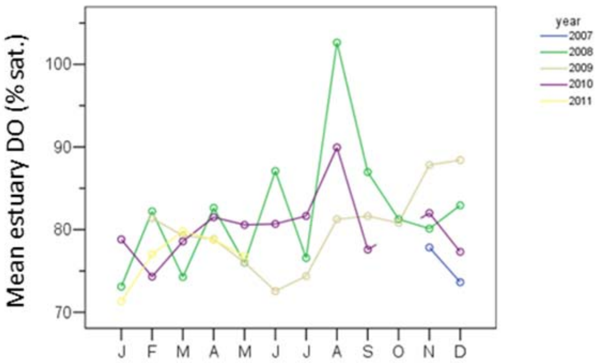


Figure 103 Interannual variation in dissolved oxygen saturation in the Cobaki-Terranora system during the study period.

6.11.4 Ecological implications

This study shows that the estuary reaches in the Cobaki Terranora Broadwater system are subject to periods of moderate hypoxia which is directly linked to nitrogen loadings (Figure 104). Hypoxia in the estuarine reaches is likely to impact on their value as fish habitat, especially species that exclusively inhabit the freshwater / brackish interface. The broadwaters and Terranora Creek are less prone to hypoxia due to better tidal flushing and shallower morphology which increases the relative importance of oxygen production by seagrasses and benthic microalgae.

6.11.5 Management implications

The clear relationship between total nitrogen concentrations and dissolved oxygen saturation (Figure 104) indicates that managing nitrogen loadings to the system will have a direct positive impact on the oxygen status of the Cobaki Terranora Broadwater system. The data show that TN concentrations in excess of 400 µg L^{-1} lead to a predominance of moderate hypoxia throughout the system. As such, the current ANZECC (2000) threshold guideline for total nitrogen (300 µg L^{-1}) represent an appropriate threshold guideline for the maintenance of aquatic systems in the Cobaki Terranora Broadwater system.

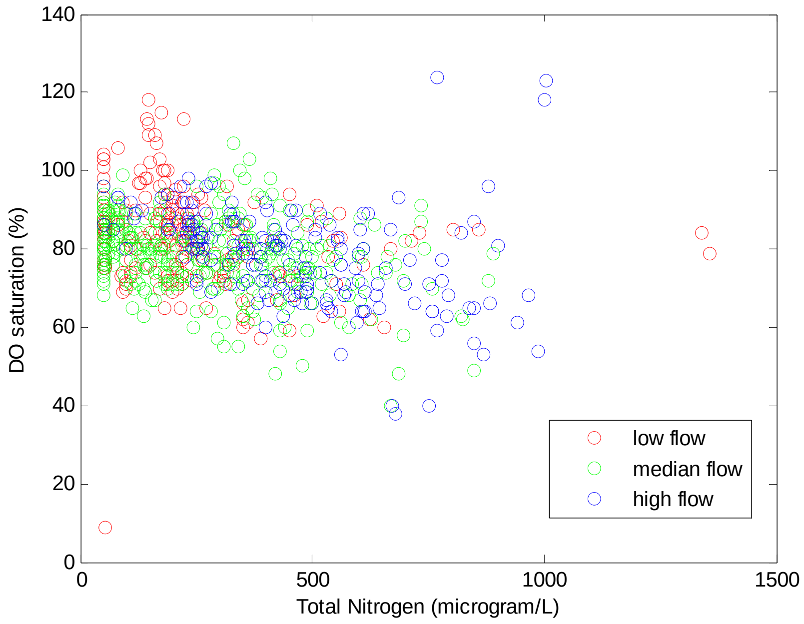


Figure 104 The relationship between total nitrogen and dissolved oxygen saturation in the Cobaki Terranora Broadwater system. The ANZECC (2000) guideline threshold for the maintenance of aquatic systems in estuaries (300 µg L^{-1}) corresponds to a median dissolved oxygen saturation of approximately 80%.

7 Salt balance modelling

7.1 Approach

A simple one dimensional (1D) salt balance model was used to assess the relative impacts of different effluent release scenarios on nutrient concentrations throughout the Tweed estuary. The 1D salt balance model uses salinity as a tracer of mixing between seawater and freshwater flows, with the salinity at any point in the estuary determined by the relative importance of these two flows (Fischer, List et al. 1979). Effluent entering the estuary at any point will be diluted upstream and downstream of the source as a function of mixing. The 1D salt balance model predicts the concentration of dissolved pollutants emanating from the effluent discharge as they are diluted along the estuary. For full details on the theory underpinning the salt balance model please refer to the Appendix.

7.2 Estuarine model setup

The model represents the estuary as a series of 1D boxes with the water quality survey sites situated at the centre of each box (Figure 105). The volume of each box is estimated from bathymetric surveys (ABER/NSWDECCW unpublished data) and box length. Average salinity distributions along the estuary were estimated from sample surveys for 10th, 25th, 50th, 75th, and 90th percentile freshwater flows (Table 1).

Figure 105 Boundaries of the 1D boxes used in the salt balance model. The centre of each box corresponds to the location of sample sites used in the current study.

7.2.1 Catchment model

Freshwater inflows to the estuarine model were based on a catchment model developed by Jocelyn Delacruz of NSW OEH. The model provides daily flows from 41 sub-catchments in the Tweed catchment (Figure 106). Freshwater flows were modelled using the 2Csalt package and calibrated against measured flow from the PINEENA database.

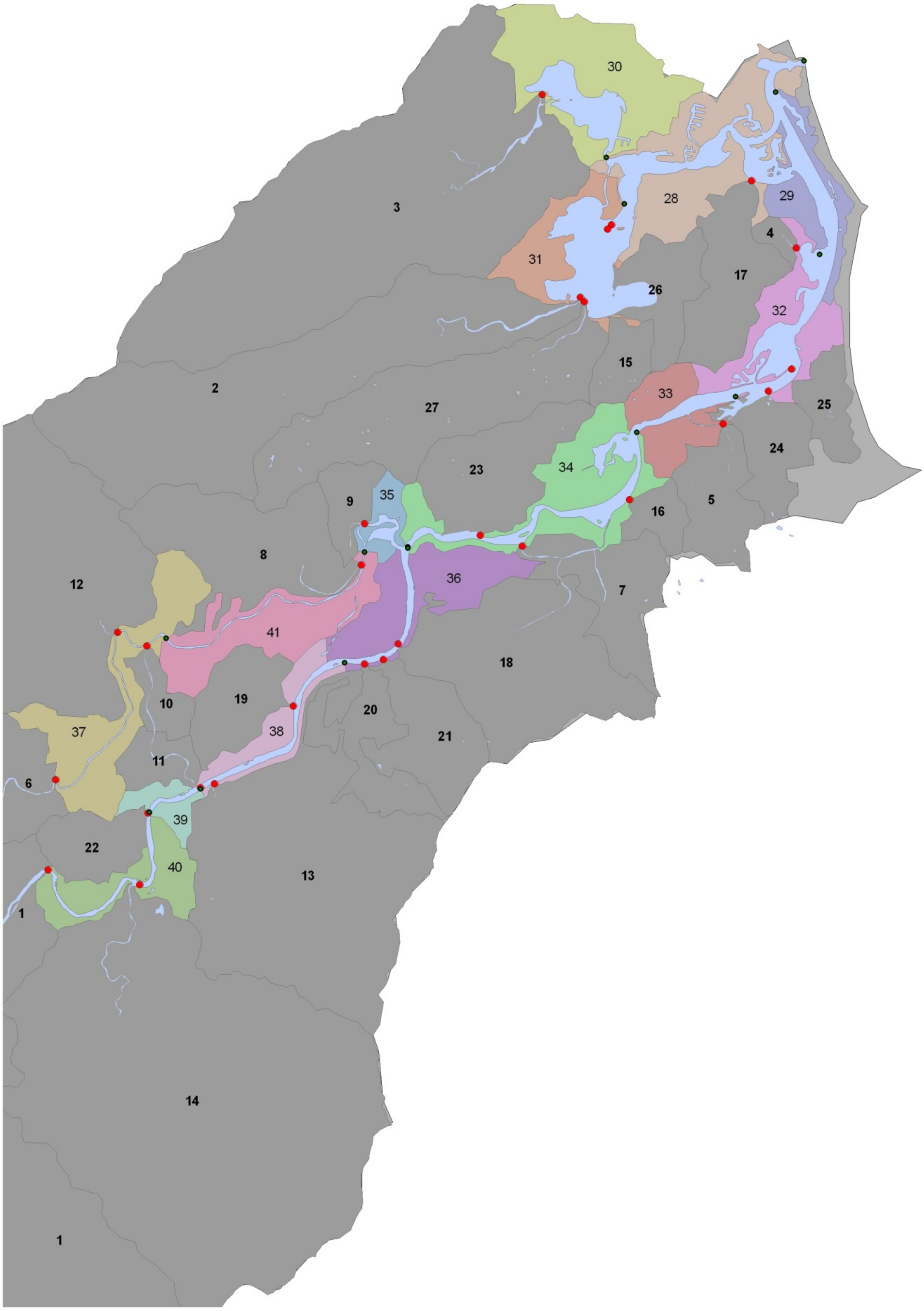


Figure 106 Sub-catchment map used in the catchment discharge model. Red dots indicated the input points to the estuary for various sub-catchments.

Table 1 Relative contribution of diffuse freshwater (FW) and STP inflows (ML/day) to each of the estuarine boxes represented in the salt balance model. The contribution of STP inflows is also given as a percentage of the total input flow (STP+FW) to that box (note that this does not include the cumulative flows entering the box from upstream sub-catchments, only sub-catchment directly entering the box). This simulation did not include the diversion of effluent from the Murwillumbah STP to Condong sugar mill. Freshwater flushing time (Flush Time) is the replacement time (in days) of the freshwater in the box by inflows to the box.

	10%ile				25%ile				50%ile				75%ile				90%ile			
Box	FW flow	STP flow	% STP	Flush Time	FW flow	STP flow	% STP	Flush Time	FW flow	STP flow	% STP	Flush Time	FW flow	STP flow	% STP	Flush Time	FW flow	STP flow	% STP	Flush Time
1	18	14	45%	1	64	14	18%	0	258	15	5%	1	1472	15	1.0%	0	6021	17	0.29%	0.1
2	8	12	59%	2	30	12	28%	6	119	12	9%	3	746	12	1.6%	1	3147	12	0.38%	0.7
3	8	4	34%	47	30	4	12%	34	117	4	3%	11	742	5	0.6%	4	3133	7	0.23%	1.1
4	8	3	25%	29	28	3	8%	19	112	3	2%	5	727	3	0.5%	2	3089	6	0.18%	0.5
5	8	3	26%	75	27	3	9%	52	110	3	2%	15	719	3	0.5%	4	3061	6	0.18%	1.1
6	7	3	27%	84	26	3	9%	42	105	3	3%	12	705	3	0.5%	3	3017	6	0.18%	0.7
7	3	0	0%	360	10	0	0%	138	44	0	0%	34	451	0	0.0%	4	2184	0	0.00%	0.9
8	2	0	0%	545	9	0	0%	192	39	0	0%	46	425	0	0.0%	5	2106	0	0.00%	1.0
9	2		0%	624	8		0%	207	36		0%	48	411		0.0%	5	2056		0.00%	0.9
10	1		0%	940	5		0%	304	22		0%	71	323		0.0%	5	1547		0.00%	1.1
11	4	3	41%	38	13	3	16%	23	52	3	5%	6	229	3	1.4%	2	749	6	0.73%	0.5
12	2	3	52%	69	9	3	22%	30	36	3	7%	9	186	3	1.7%	2	607	6	0.90%	0.6
13	1	3	72%	83	4	3	39%	44	17	3	14%	15	105	3	3.0%	3	348	6	1.56%	0.8

Table 2 Effluent flows and quality at the various STPs discharging to the Tweed estuary.

STP	statistic	Flow (ML/day)	Nitrogen Oxidised (mg/L)	NH_4^+ (mg/L)	BOD (mg/L)	Chl-a (µg/L)	DIP (mg/L)	TN (mg/L)	TP (mg/L)
Kingscliff	10th %ile	2.45	0.25	0.08	1.40	1.50	0.05	1.23	0.14
	25th %ile	2.57	0.56	0.13	2.40	7.20	0.06	2.09	0.99
	50th %ile	2.71	1.12	0.30	4.20	28.90	0.08	3.16	2.57
	75th %ile	2.92	2.33	0.65	6.60	70.00	0.18	4.38	6.40
	90th %ile	3.25	3.50	1.77	10.32	119.50	0.43	7.45	7.41
Murwillumbah 2ndary	10th %ile	1.81	1.67	0.13	1.10	1.00	0.05	3.35	0.10
	25th %ile	2.25	2.69	0.22	1.40	4.50	0.07	4.21	0.13
	50th %ile	2.76	4.05	0.38	2.00	13.00	0.11	5.30	0.21
	75th %ile	3.40	5.78	0.56	3.35	27.25	0.24	7.05	0.40
	90th %ile	4.62	7.55	0.85	6.40	58.90	0.51	8.89	0.75
Murwillumbah tertiary	10th %ile	0.00	3.05	0.09	1.02	0.40	0.07	4.08	0.06
	25th %ile	0.19	4.27	0.15	1.40	0.50	0.09	4.90	0.10
	50th %ile	1.18	5.30	0.24	3.10	0.90	0.16	5.83	0.16
	75th %ile	1.74	6.41	0.39	5.60	1.20	0.36	6.81	0.34
	90th %ile	2.08	7.62	0.53	8.64	2.28	0.60	7.85	0.61
Banora Point	10th %ile	6.00	2.10	0.05	1.20	2.70	1.03	3.70	1.94
	25th %ile	8.47	2.94	0.08	1.60	6.00	1.68	4.26	2.80
	50th %ile	10.32	3.63	0.14	2.40	16.00	2.11	4.90	4.01
	75th %ile	11.71	4.32	0.36	4.50	45.85	2.79	5.50	5.03
	90th %ile	13.40	4.95	2.13	9.00	91.00	3.64	6.42	5.95
Tumbulgum	10th %ile	0.03	0.25	0.56	1.00	0.60	0.06	2.40	0.18
	25th %ile	0.04	0.53	0.91	1.00	1.30	0.10	3.00	0.29
	50th %ile	0.05	1.16	1.49	1.50	3.60	0.19	4.10	0.49
	75th %ile	0.06	2.52	2.54	2.00	13.00	0.45	6.49	0.84
	90th %ile	0.08	4.20	5.91	3.20	29.00	0.90	10.40	1.55

7.3 Results

7.3.1 Freshwater inflows

The salt balance model provides insights into the delivery of freshwater to the estuary and the resultant freshwater flushing times[14] (Table 1). Figure 107 shows that more than half of the total freshwater discharge to the estuary enters via the Rous estuary during low and median flow conditions. This is partly due to the clipping of freshwater flows by the weir above Murwillumbah. The relative importance of flows from the Tweed sub-catchments above the Rous confluence greatly increases during high flow conditions. Freshwater flushing times reflect this pattern, with a large increase in flushing times upstream of the Rous confluence (Table 1). Actual water residence times (taking into account tidal flushing) would be less than shown in Table 1, however the general spatial patterns would be similar with significantly higher residence times in the upper Tweed estuary.

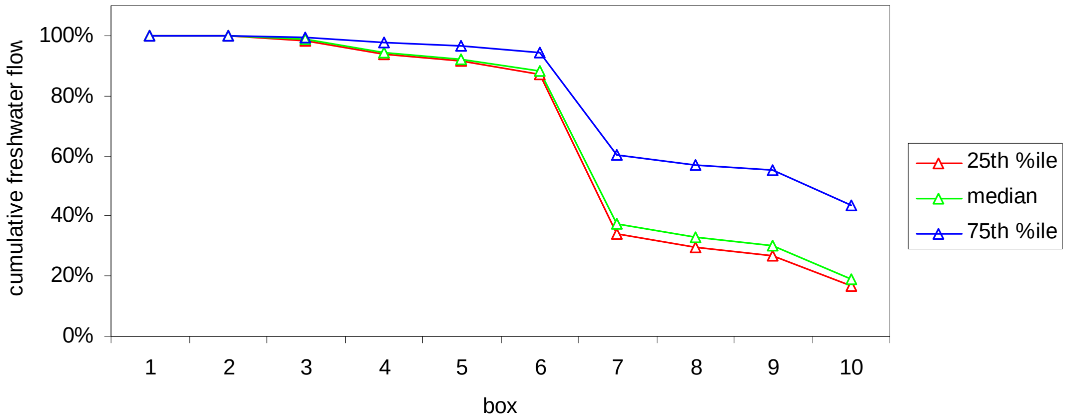


Figure 107 The relative contribution of freshwater inputs along the Tweed estuary to total catchment discharge during different flow conditions. The large increase in cumulative flow at box 6 is due to the contribution from the Rous estuary sub-catchments.

7.4 Existing condition scenarios

The estuarine model was tested against water quality data collected as part of the current study for the range of flow conditions and salinity structures described in Table 1. For the existing condition scenarios, the TN concentrations of catchment inputs were held constant across all urban and rural catchments, with the exception of the sub-catchments entering the Rous estuary which were held at 60% of the Tweed sub-catchments (based on analysis of routine data). The contribution of STP effluent was estimated from the average concentration and flows for each of the STPs (Table 2).

7.4.1 Existing conditions – low flow

During low flow, inputs from the diffuse catchments are small, and in the case of the main Tweed River are zero due to the impact of the weir above Murwillumbah.

[14] Freshwater flushing times calculated by the salt balance model are the time in days equal to the replacement of all freshwater within the box by freshwater entering the box from upstream. This is not equal to the water residence times which would include flushing due to tidal flows.

During this period effluent discharges from sewage treatment plants (STPs) become relatively more important (Table 1).

The results show that the model is able to closely predict the distribution of TN concentrations along the estuary (Figure 108). In order to replicate the observed range in total nitrogen concentrations during low flow it was necessary to vary the STP effluent loadings by the measured variation in effluent quality and flow. The higher upper range for measured TN in the lower estuary most likely arises from 1) variable TN concentrations in oceanic water; and 2) occasional sampling of the lower estuary sites during low or ebb tide. Increasing the TN concentration of oceanic water to upper end of observed range (100 μg L^{-1}) provides a closer fit for the lower estuary sites. The lower limits of observations form the estuary mouth (site 1) to the middle estuary (site 7) most likely arise from samples taken during high tide. The salt balance model only represents 'average' conditions along the estuary so it reasonable that it does not reproduce the variation observed in the lower estuary (where salinity varies widely).

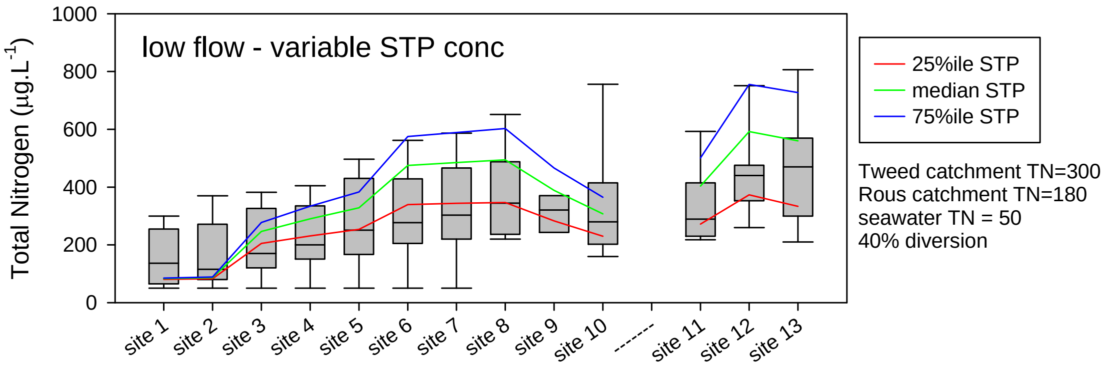


Figure 108 Comparison of modelled and observed TN concentrations during low flow (10th percentile) conditions. Boundary conditions for catchment TN concentrations were estimated from measured data from the upper most sites in the Tweed and Rous estuaries. The diversion of effluent to Condong was held at 40% (average). STP inflows were held at the 25th percentile of recorded flows (TSC data).

7.4.2 Existing conditions – median flow

The median flow category used to group data for analysis in section 5 encompasses all flows falling within the 25th and 75th percentile and therefore includes a wide range of salinity conditions in the estuary. As such, the existing conditions scenario for median flow was iterated for 25th, median, and 75th percentile flows and salinity conditions (Figure 109). An additional simulation was included to account for higher TN concentrations in catchment runoff and the maximum diversion rate of effluent from Murwillumbah STP to the Condong sugar mill. Contributions from STP effluent become relatively smaller during median flow conditions, but still account for up to 28% of total inflows at the lower end of the median flow category.

The results show that the model is able to closely predict the distribution of TN concentrations along the estuary (Figure 109). Results for the 25th percentile simulation are very similar to those for low flow (10th percentile), indicating that some sample runs included in the median flow category were perhaps more closely aligned with low flow conditions. This is an unavoidable artefact of the arbitrary categorisation of data according to flow statistics. Nonetheless, the categorisation according to flow has provided a useful tool to aid in the understanding of observed patterns in the data.

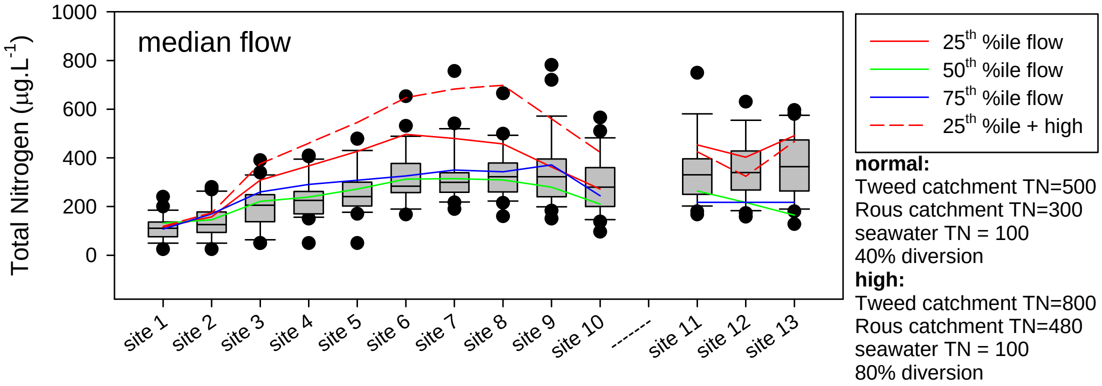


Figure 109 Comparison of modelled and observed TN concentrations during median flow (25^{th} to 75^{th} percentile) conditions. Boundary conditions for catchment TN concentrations were estimated from measured data from the upper most sites in the Tweed and Rous estuaries. The diversion of effluent to Condong was held at 40% (average) except for the 'high' scenario where diversion was held at 80% (maximum observed).

7.4.3 Existing conditions – high flow

During high flow conditions STP contributions are reduced to <0.3% of total flow (Table 1) and model results are not sensitive to variations in effluent flow or quality. Given the dominance of freshwater inflows to the estuary, the TN concentrations along the estuary are controlled by the quality (TN concentration) of TN in freshwater inflows which can be highly variable (McKee, Eyre et al. 2000).

The model was capable of predicting the observed range of TN concentrations along the estuary by varying the concentration of freshwater inputs (Figure 110). The range chosen (300 – 1200 µg L^{-1} is based on measured data from nearby systems (McKee, Eyre et al. 2000). The 1200 µg L^{-1} TN simulation reproduced peak concentrations in the estuary, but over estimated concentrations in the upper estuary (Figure 69). This suggests that high flow events may be characterised by inputs of TN-rich water (e.g. humic-rich swamp water or runoff from sugar cane lands) from the sub-catchments discharging to the middle estuary. To account for this, an additional scenario ('high_mid') was run where the concentration of TN in the upper Tweed sub-catchments was reduced to 900 µg L^{-1} (to closer agree with the upper limit of observations at site 10) and TN concentrations in runoff from sub-catchments draining to boxes 6 – 8 were elevated to 1900 µg L^{-1}. Runoff from the Rous was elevated to 1200 µg L^{-1} for this scenario. The 'high_mid' scenario provides a much better fit with the upper limits of observations and lend credence to the hypothesis that large runoff events may be comprised of nitrogen-rich discharges from the middle estuary sub-catchments. In reality, runoff from each sub-catchment will vary in quality for the duration of any event and between events. Further understanding of this issue can only be gained by more in depth, flow-weighted monitoring of water quality.

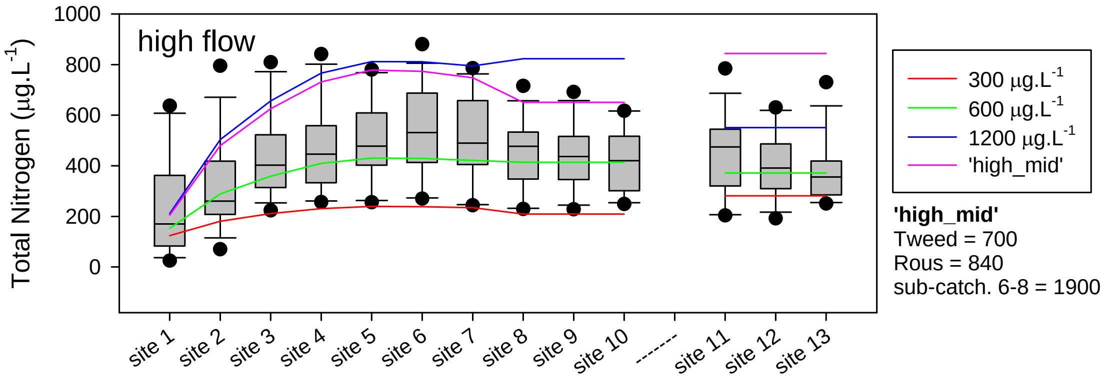


Figure 110 Comparison of modelled and observed TN concentrations during high flow (90th percentile) conditions. Boundary conditions for catchment TN concentrations were estimated from measured data from nearby systems (Brunswick and Richmond Rivers). The diversion of effluent to Condong was held at 40% (average). Concentrations in catchment runoff for the 'high_mid' scenario were amended for middle estuary sub-catchments as indicated in the text.

7.5 STP scenarios

The salt balance model was used to test the impacts of variable STP effluent inputs to the estuary under different flow conditions. Freshwater inflows from diffuse catchments were held constant for all scenarios to highlight the impact of STPs. A no STP scenario ('catchment') was included to indicate the likely TN concentrations in the absence of STP inputs.[15] The quantity and quality of STP effluent inputs for the 'mean STP' and max STP' were estimated from statistical analysis of data provided by Tweed Shire Council. The 'mean STP' scenario holds effluent flow and TN concentrations at the average of available data, while the 'max STP' scenario holds both effluent flow and TN concentrations at the maximum observed values from available data. While these conditions may not commonly co-occur, this was chosen to indicate a likely 'worse case' scenario.

7.5.1 Variable STP performance scenarios

The results of the variable STP performance scenarios are shown in Figure 111. The most striking feature of these results is the effect of flow on the relative impact of STP inputs to the Tweed system. During low flow conditions, inputs from the catchment alone result in vary low TN concentrations along the estuary, while the inclusion of average STP inputs results in TN concentrations at the upper limit of observed water quality, and maximum STP inputs result in TN concentrations three fold higher than observed water quality. The results for median flow indicate a much reduced impact due to STP inputs, while during high flow the impacts due to STP inputs are imperceptible above the 'catchment' scenario. This indicates that effluent

[15] Note that while the 'catchment' scenario excludes STP inputs it cannot be regarded as a true approximation of pre-settlement 'pristine' conditions as it holds all catchment inputs at a common flat value. In reality, catchment inputs are highly variable in space and time, even under 'pristine' conditions.

management aiming to minimise impacts on the estuary is most critical during low and medina flow conditions.

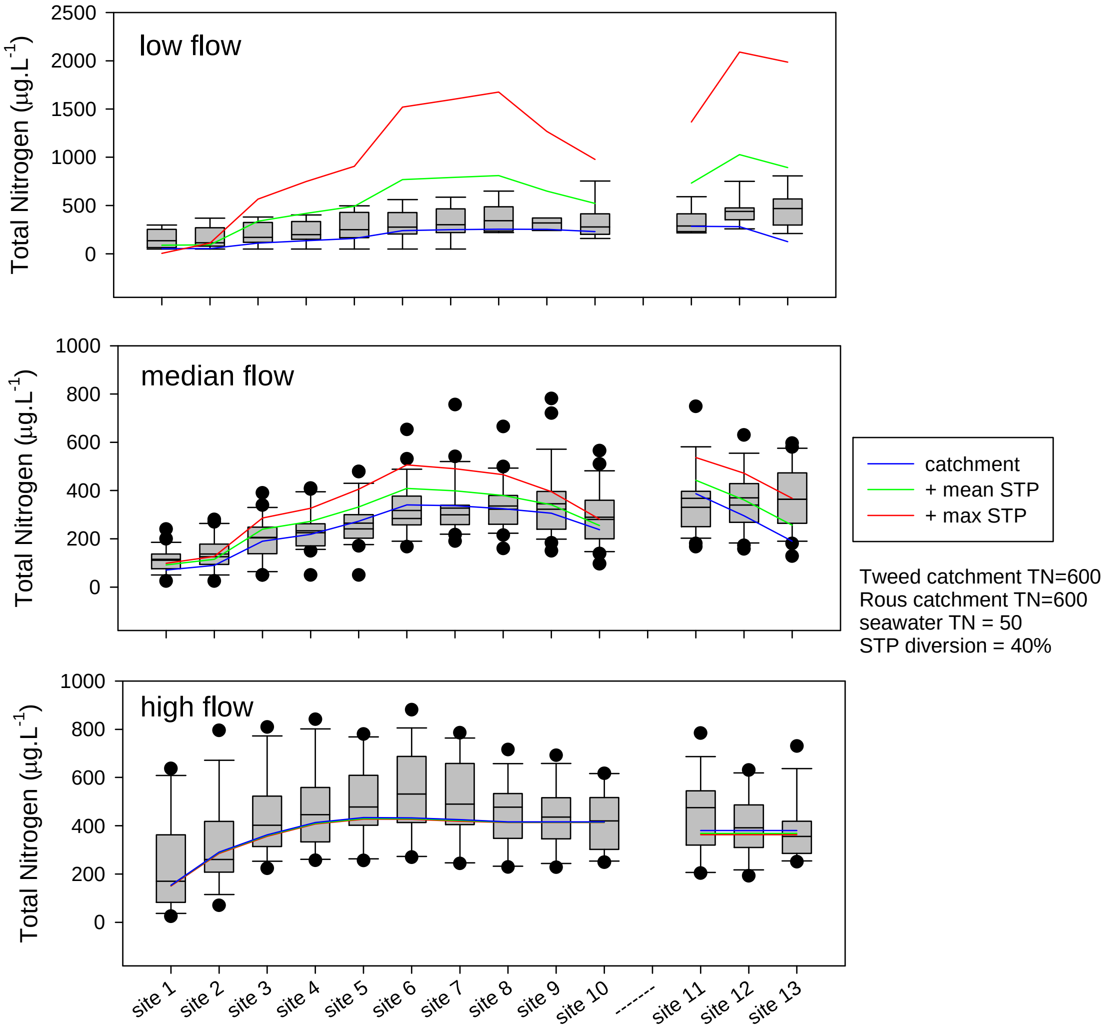


Figure 111 Effects of varying STP contributions on the distribution of TN throughout the estuary.

7.6 Oxidised nitrogen

Oxidised nitrogen (NO_x) constitutes a major fraction of treated sewage effluent (Table 2) and is recognized as a primary limiting nutrient for the development of phytoplankton blooms in estuaries (O'Donohue, Glibert et al. 2000). NO_x inputs along estuarine gradients are subject to various sources (e.g. benthic fluxes) and sinks (e.g. uptake by phytoplankton and benthic microalgae, and/or denitrification), therefore measured concentrations during low to median flow conditions are commonly far less than modelled concentrations using the simple salt balance approach (which assumes conservative mixing; (Ferguson, Eyre et al. 2004).

Model results show that STP effluent discharges can potentially greatly increase NO_x availability in the Tweed and Rous estuaries (Figure 112). As with TN, the greatest impacts are apparent during low flow conditions when catchment discharges are relatively small and NO_x concentrations in freshwater runoff are low (Figure 112).

Measured NO_x concentrations are far below modelled concentrations due to the combination of biological uptake and denitrification. In particular, modelled inputs to the Tweed estuary from the Rous are likely to be too high as no account is made of uptake in the Rous estuary. This results in artificially high concentrations in the Tweed estuary both up and downstream of the confluence with the Rous. Much of the effluent-derived NO_x in the Rous would be converted to particulate N (i.e. phytoplankton) before being delivered to the Tweed estuary. At least part of this load would be delivered to sediments in the middle reaches of the Tweed and subject to remineralisation by bacteria. As such, benthic fluxes of DIN (NH_4^+ and NO_x) will result in elevated NO_x concentrations during low to median flow conditions, thereby representing an indirect impact of sewage effluent inputs.

Model results for median flow conditions show that the relative increase in NO_x concentrations above catchment inputs due to STP effluent inputs is much smaller than for low flow conditions (Figure 113). In addition, model results are closer to measured concentrations as freshwater flushing times are shorter, thereby limiting the potential for biological processing in the estuary.

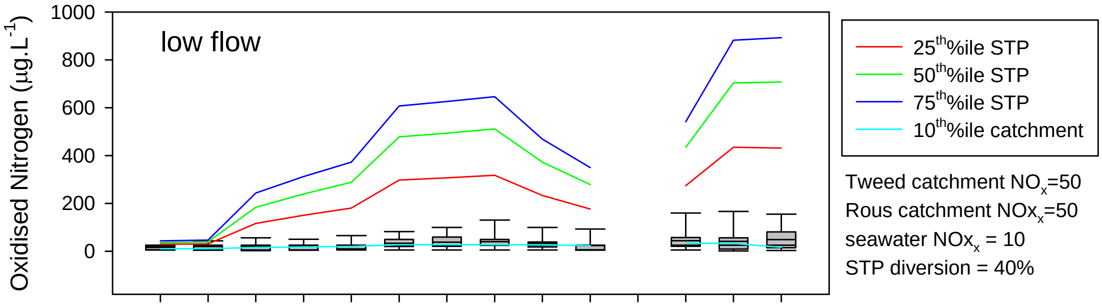


Figure 112 Modelled concentrations of NO_x along the Tweed and Rous estuaries during low flow conditions. The significantly lower measured concentrations are most likely due to rapid biological processing of effluent in the estuary see text for details).

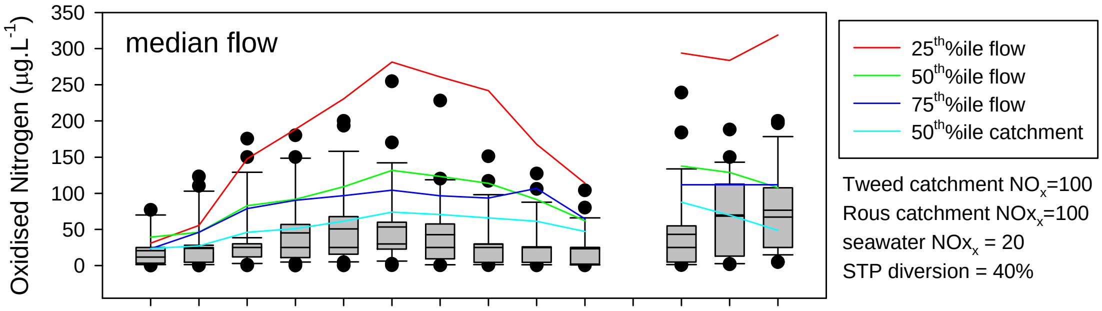


Figure 113 Modelled concentrations of NO_x along the Tweed and Rous estuaries during median flow conditions.

8 Murwillumbah STP upgrade and effluent diversion

8.1 Background

The monitoring period for this study spanned to distinct management regimes for the Murwillumbah STP. Until late Dec 2007 (exact timing unavailable), all effluent was released to the Rous estuary just upstream of site 13 (scenario 1). After this date, the treatment train was upgraded and a portion of effluent (~40%) was routed to the Condong sugar mill for use as cooling water, after which it was discharged to the Tweed estuary just upstream of site 8. The remainder of the effluent (~60%) was discharged to the Rous estuary (scenario 2).

Data from each scenario has been grouped according to low and median flow conditions and presented in Figures 114 and 115. To aid in interpretation of results, the salt balance model has been used to simulate each average conditions (i.e. salinity and TN concentrations) for each scenario (Figure 116) and to assess the impacts of variable rates of diversion (Figure 117). For the average condition simulations catchment TN concentrations were varied according to the average temperature for each scenario (see section 5.4 for discussion about temperature dependency of catchment TN concentrations). All STP inputs were held at median conditions for flow and TN concentrations. For scenario 2 simulations 40% of the total effluent flow from the Murwillumbah STP was diverted to Condong. No change in effluent quality was included for scenario 2 based on the statistics for the tertiary treated effluent (Table 2).

8.2 Seasonal context

Monitoring during Scenario 1 (pre upgrade and effluent diversion) occurred during the spring to summer dry season and encompassed mostly low and median flow conditions. Monitoring during Scenario 2 (post upgrade with variable effluent diversion) occurred across a full range of flow conditions and for a much longer period. In order to make valid comparisons, data for each scenario has been grouped according to flow conditions so that salinity structure and diffuse catchment inputs are similar for each scenario. It should be noted that the Scenario 1 period was greatly under-sampled to provide a true indication of prevailing conditions.

Figure 114 A. shows that median salinities were similar between scenarios for each flow category, however the range of salinities represented in scenario 2 under low flow conditions was greater. Temperatures in Scenario 1 were skewed to the upper end of the temperature range reflecting the limitation of the monitoring period to spring – summer (Figure 114 B). This has implications for the valid comparison of the scenarios due to the temperature dependence of nutrient concentrations in freshwater catchment runoff (see section 5.4).

8.3 Measured nitrogen concentrations

Figure 115 shows an apparent reduction in both NO_x and TN in scenario 2 for both low and median flow groupings. However, data quality and insufficient sampling of scenario is likely to have influenced this result, and in the case of NO_x preclude the application of rigorous statistical analysis.

In the case of NO_x, the increase in detection limit during the second phase of the monitoring period when TSC undertook analyses meant that low concentration samples were defaulted to the detection limit value and only sporadic samples registered values above detection limit. Inter –laboratory comparisons of replicate samples analysed at both DECC and TSC show that the TSC lab routinely returned below detection limit values for samples that the DECC lab measured relatively high NO_x concentrations. Hence, it is likely that data for the scenario 2 period were influenced by erroneously low NO_x concentrations.

Total nitrogen concentrations were significantly higher during scenario 1 under low flow conditions ($p<0.05$). However, it is important to note that temperature during scenario 1 were at the high end of the seasonal range, therefore catchment TN concentrations were at a seasonal high (see Figure 19). Accounting for the temperature dependent range in catchment TN concentrations in the salt balance model was sufficient to account for the difference between scenarios 1 and 2 (Figure 116), even with the change in effluent management. The difference between scenarios 1 and 2 during median flow conditions was less marked and not significant (Figure 116). Again, temperature dependent differences in catchment TN concentrations were sufficient to explain this difference (Figure 116).

The model results for testing variable rates of diversion to Condong predicted a shift in the TN concentration peak from the middle to the upper Tweed estuary reach (Figure 117). This was pattern was confirmed by distributions of measured TN, however there was no increase in concentrations in the upper estuary as predicted. Again, this was most likely due to the temperature dependent variation in catchment concentrations that was not accounted for in the variable diversion rate simulations.

8.4 Chlorophyll-a

There was no difference in chlorophyll-a between scenarios for either flow conditions (Figure 114). This suggests that any actual changes in effluent management were insufficient to reduce nitrogen loading enough to see a significant reduction in phytoplankton biomass. The reasons for this may include: 1) the primary control of water residence times over phytoplankton biomass accumulation applied equally to both scenarios; 2) that effluent quality was unchanged hence total DIN loading was the same, only slightly redistributed; and 3) that internal recycling of DIN may contribute a large portion of the DIN demand by phytoplankton. This internal recycling may continue for some years after changes in catchment pressures due to build ups of organic matter in the sediments.

8.5 Dissolved oxygen

There were significant increases in DO saturation during low and median flows, due primarily to increases in the middle estuary (Figure 115). As with TN, this may in part be explained by higher rates of temperature dependent oxygen consumption during scenario 1. It is noteworthy that BOD loading would have slightly increased under scenario 2 (Table 2), hence DO saturation appears to be uncoupled from STP organic matter loading.

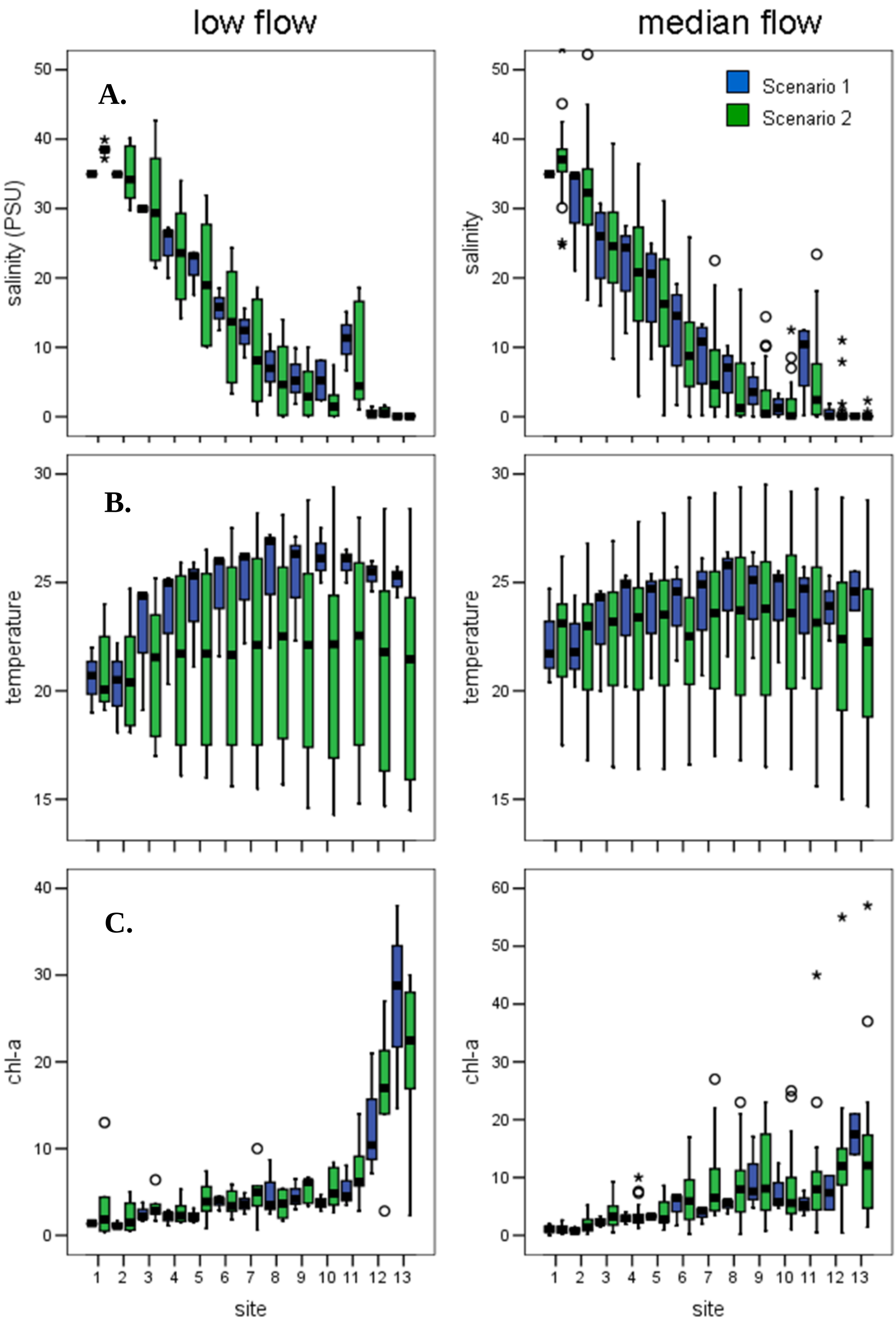


Figure 114 A. Salinity, **B.** temperature and **C.** chlorophyll-a grouped according to Scenario 1 and 2 for low and median conditions.

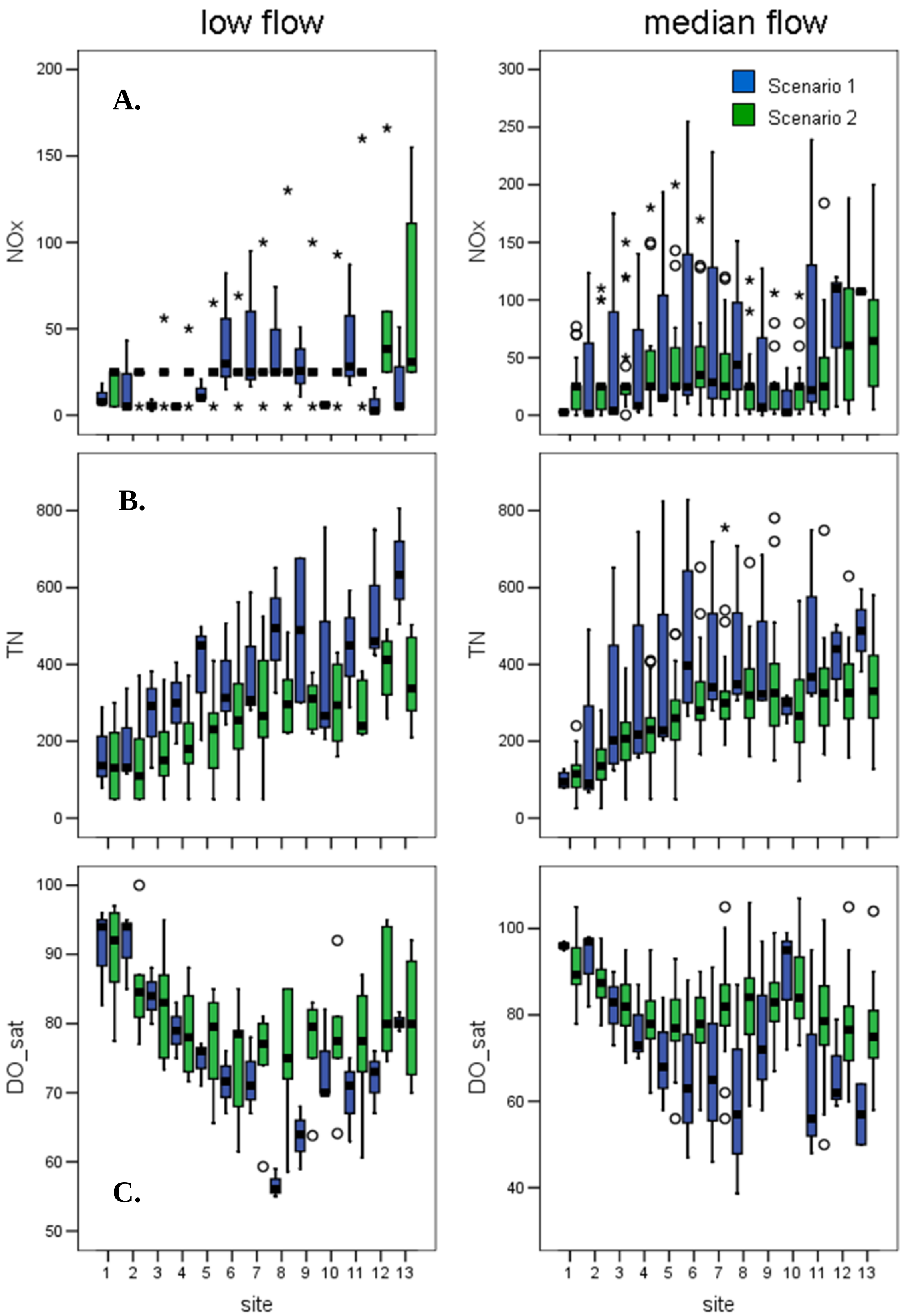


Figure 115 A. NO_x, **B.** TN and **C.** DO saturation grouped according to Scenario 1 and 2 for low and median conditions.

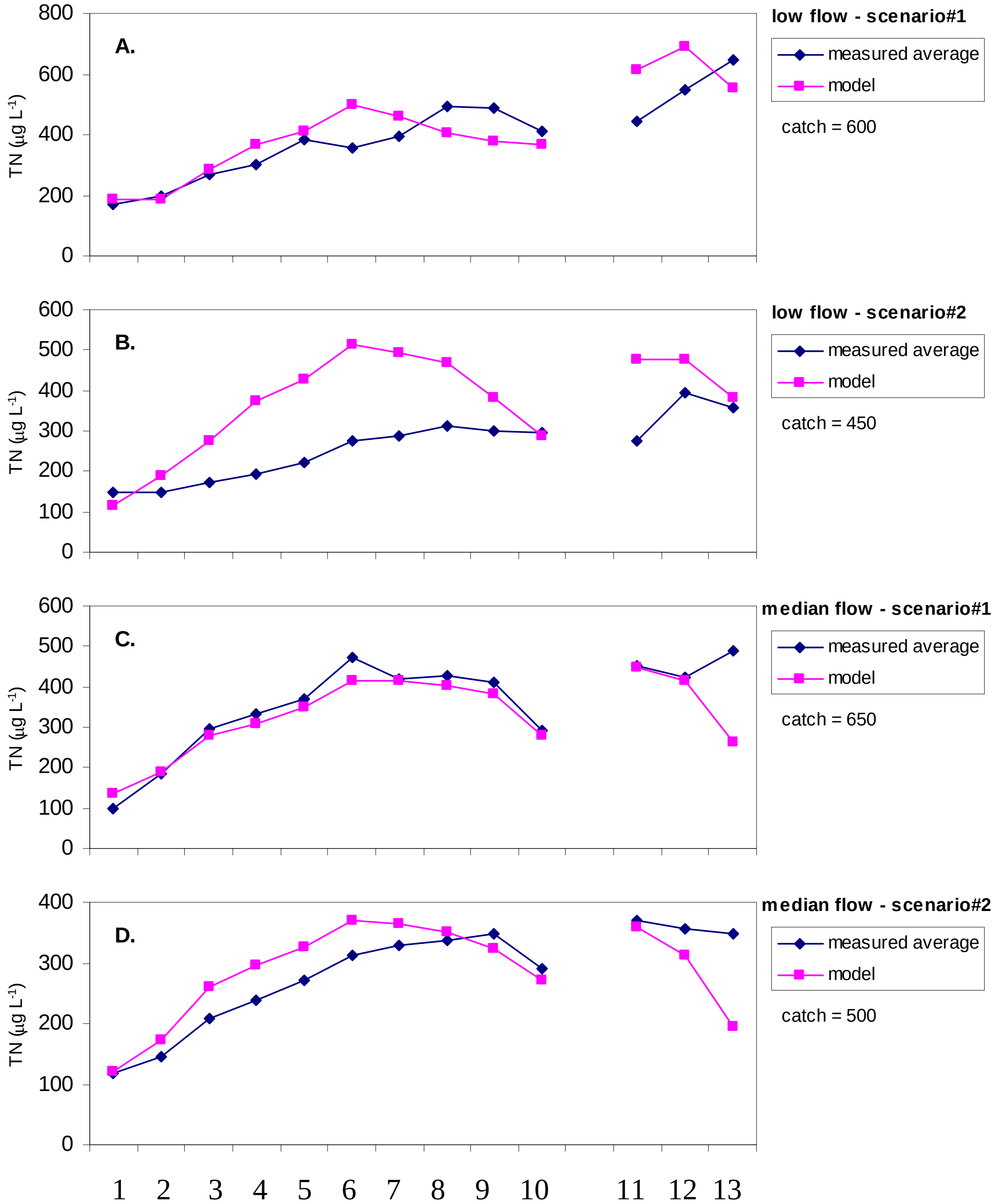


Figure 116 Salt balance modelling of the STP management scenarios compared with averages of measured concentrations during each scenario.

8.6 Modelling effluent diversion to Condong sugar mill

The impacts of variable rates of effluent diversion to Condong sugar mill for use as cooling water were tested using the salt balance model. Rates of diversion were estimated assuming that all effluent from the tertiary treatment pond at Murwillumbah STP was transferred to Condong, and that the difference between flows from the secondary treatment pond and the tertiary pond was discharged into the Rous estuary. Analysis of available data from Tweed Shire Council indicated that the proportion transferred to Condong varied between 0 and 80% of the secondary treatment pond flows. It was assumed that effluent was concentrated 7 times during the cooling process before discharge to the Tweed estuary adjacent to the Condong sugar mill (i.e. box 8 in the salt balance model).

The results show that the effects of diversion are most dramatic during low flow times and have the potential to increase TN concentrations in the upper Tweed estuary significantly. This has serious implications for the development of phytoplankton blooms in the middle to upper reaches of the Tweed estuary as freshwater flushing times in these reaches are vary long (see Table 1).

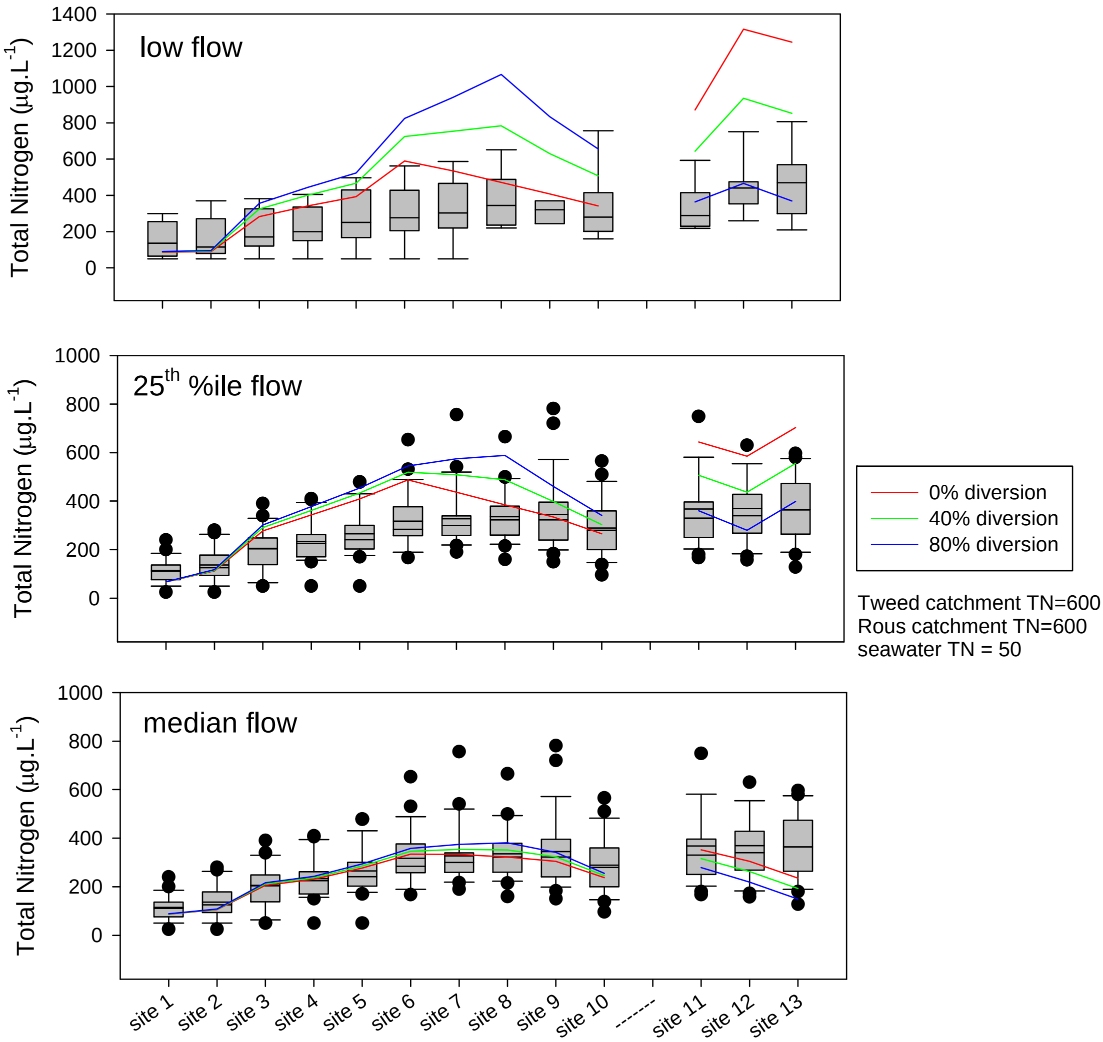


Figure 117 Impacts of effluent diversion to the Condong sugar mill under low and median flow conditions. (note: high flow conditions were excluded from this analysis due to the imperceptible impacts during high flow).

8.7 Summary

The results of the comparison between effluent management scenarios are inconclusive, due mostly to the lack of data from a full annual cycle in scenario 1. The skewing of monitoring to the warmer part of the dry season is likely to have introduced a number of confounding artefacts (due primarily to temperature) which preclude any rigorous comparison of results between the scenarios. In addition, data from the Murwillumbah STP suggests that 1) there has been little change in effluent quality, and 2) the diversion rates were highly variable, therefore it is likely that any beneficial impacts on the estuarine ecology would have been minimal.

9 Conceptual model of the Tweed estuary

This section presents a draft conceptual model of estuarine function specifically for the Tweed estuary based on evidence provided by this study, preliminary analysis of morphometrics, biogeochemical process measurements carried out as part of the DEFIRE project, and preliminary modelling of the estuary. The results indicate that the Tweed estuary broadly fits conceptual models of biogeochemical and ecological function for riverine estuaries in northern NSW (Eyre 2000).

Morphology[16] and hydrodynamics interact to create three broad functional zones along the main arm of the Tweed estuary: upper, middle and lower (Figure 118). Each has it's own set of primary attributes including morphology, sediment types, salinity regime, and water residence times. The ecology and resilience to disturbance (e.g. eutrophication) of each zone varies due to interactions between these attributes. It is important to note that there is considerable plasticity in zone boundaries due to seasonal variation in freshwater inflows which can significantly alter the spatial characteristics of salinity regimes and water residence times. The zone of greatest variability in this respect exists between the middle and lower estuary. As such, we propose a transition zone which shares many similar morphometric attributes with the middle zone but has a broader annual salinity range (Table 3).

Temporal and spatial variation in water quality can be broadly explained by considering the processes represented in Figure 119. In general, the degree of internal processes and transformation of nutrients depends on the water residence times. Hence during high flow times, water quality along the estuary reflects catchment inputs, while during the dry season biological uptake of inorganic nutrients dominates. Improvement of water clarity during the dry season increases the relative importance of benthic productivity as a nutrient sink. Inputs of nutrients from STPs and recycling from the sediments dominate during median to low flow conditions.

[16] Morphology refers to the channel dimensions, and can be characterised by metrics such as depth, channel width, channel / shoal ratio. The distribution of areas within discrete depth ranges is termed "hypsometry".

Table 3 Functional habitat zone attributes along the estuarine gradient of the Tweed estuary.

	Functional / habitat zone upper	Middle	transition	Lower	broadwater
Hypsometry*					
Channel width – mean	100	170	180	480	1280
Channel width-stdev	25	75	250	350	375
Depth – mean	4	6	5	3	0.4
Width / depth	25	30	35	160	3200
Channel / shoal	4.7	0.6	0.55	0.4	0.06
Salinity	0 – 5	5 – 10	8 – 30	15 – 35	15 – 35
Residence times**	> 20	10 – 20	2 – 10	< 3	< 10
Pelagic / benthic GPP	1 – 10	0.8 – 15	0.1 – 5	0.05 – 0.1	0.05 – 0.1
NEM***	H >95%	H > 75%	H = 100%	H = 0	H < 5%
	B < 5%	B < 25%	B = 0	B = 0	B < 10%
	A = 0	A < 5%	A = 0	A = 100%	A > 90%
Foodchains****	P, D	P, D	D, P, B	B, D	B, D

* Hypsometry units = metres
** Residence time units = days
*** NEM categories: H = heterotrophic; B = balanced; A = autotrophic
**** Foodchain categories: P = pelagic; D = detrital; B = benthic

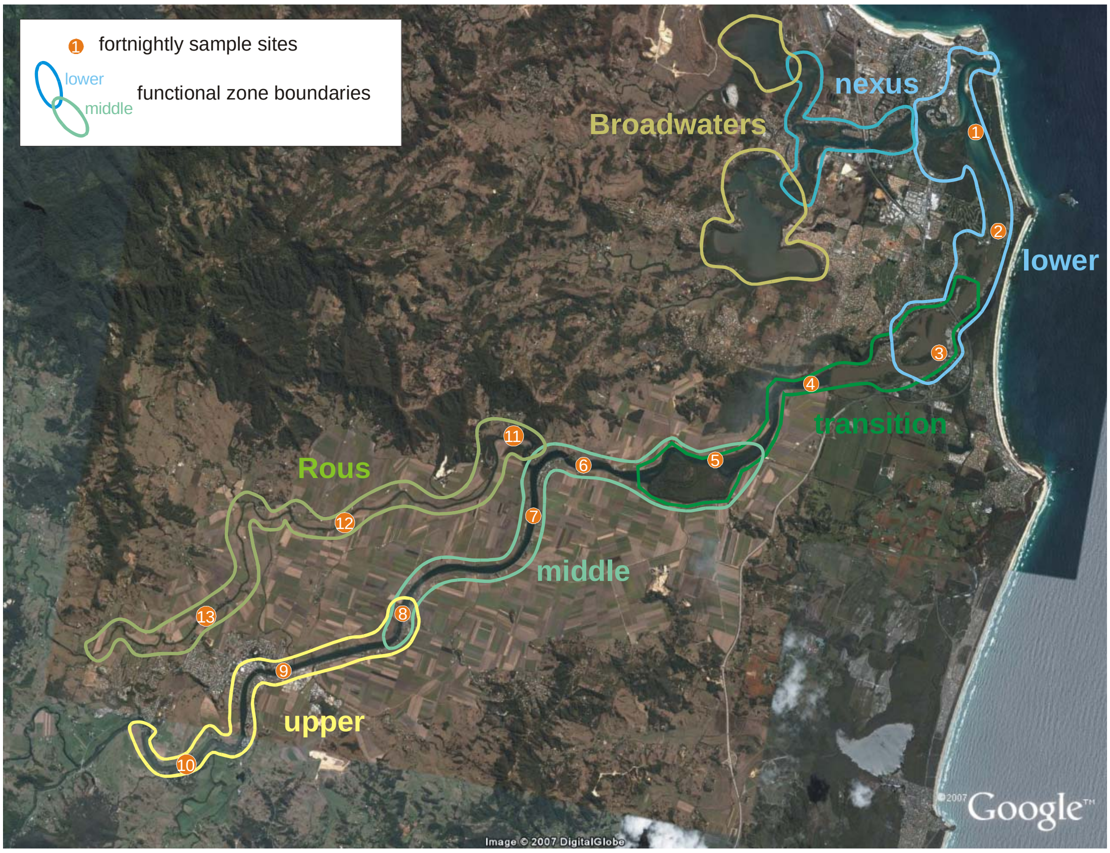


Figure 118 Proposed functional zones within the Tweed estuary.

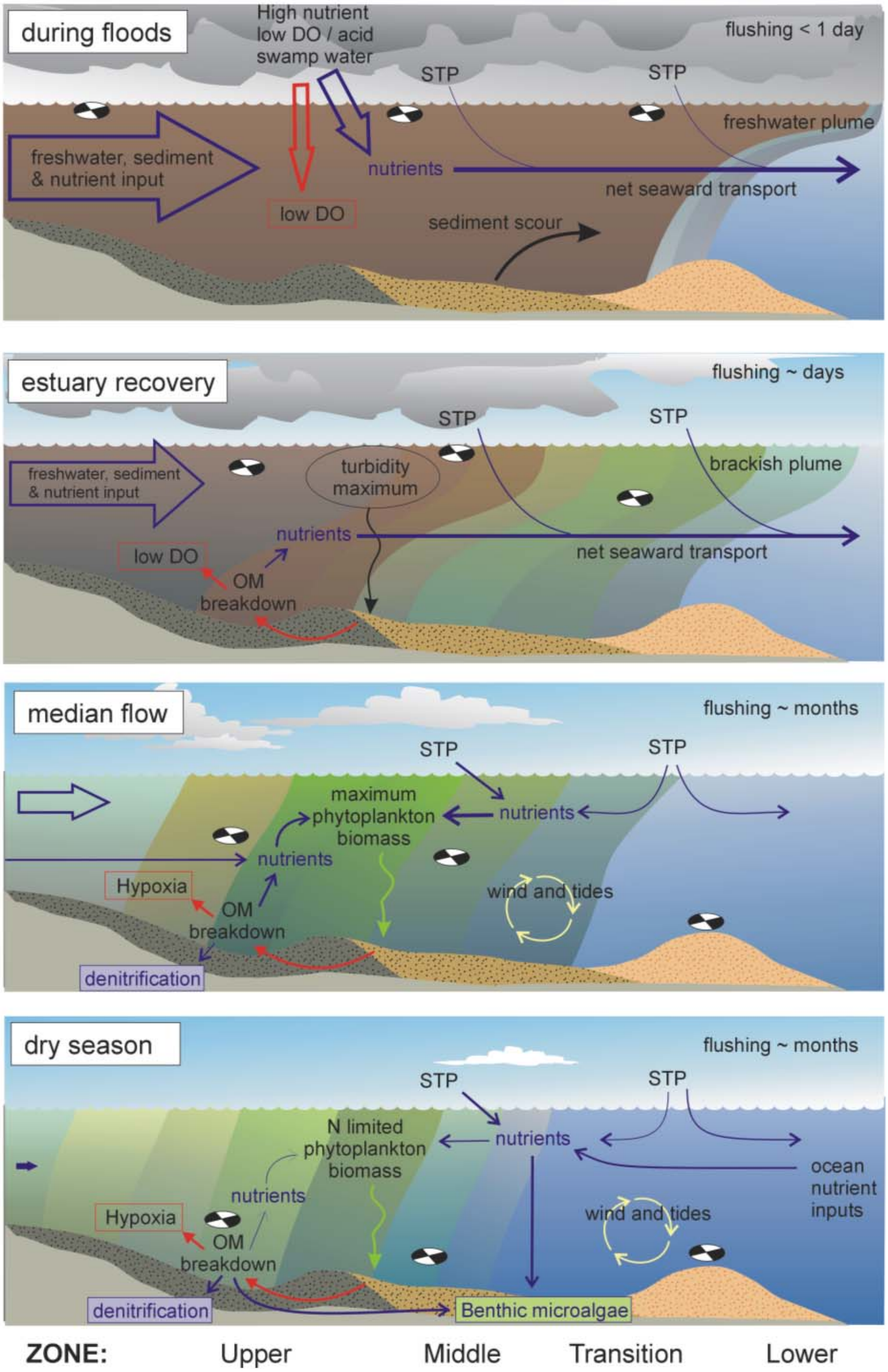


Figure 119 Seasonal progression of estuarine function in the Tweed estuary. The relative magnitude of freshwater inflows is indicated by the block arrows on the left of each stage. The relative importance of nutrient cycling pathways is indicated by the size of blue arrows. Light penetration depth is indicated by the secchi symbols.

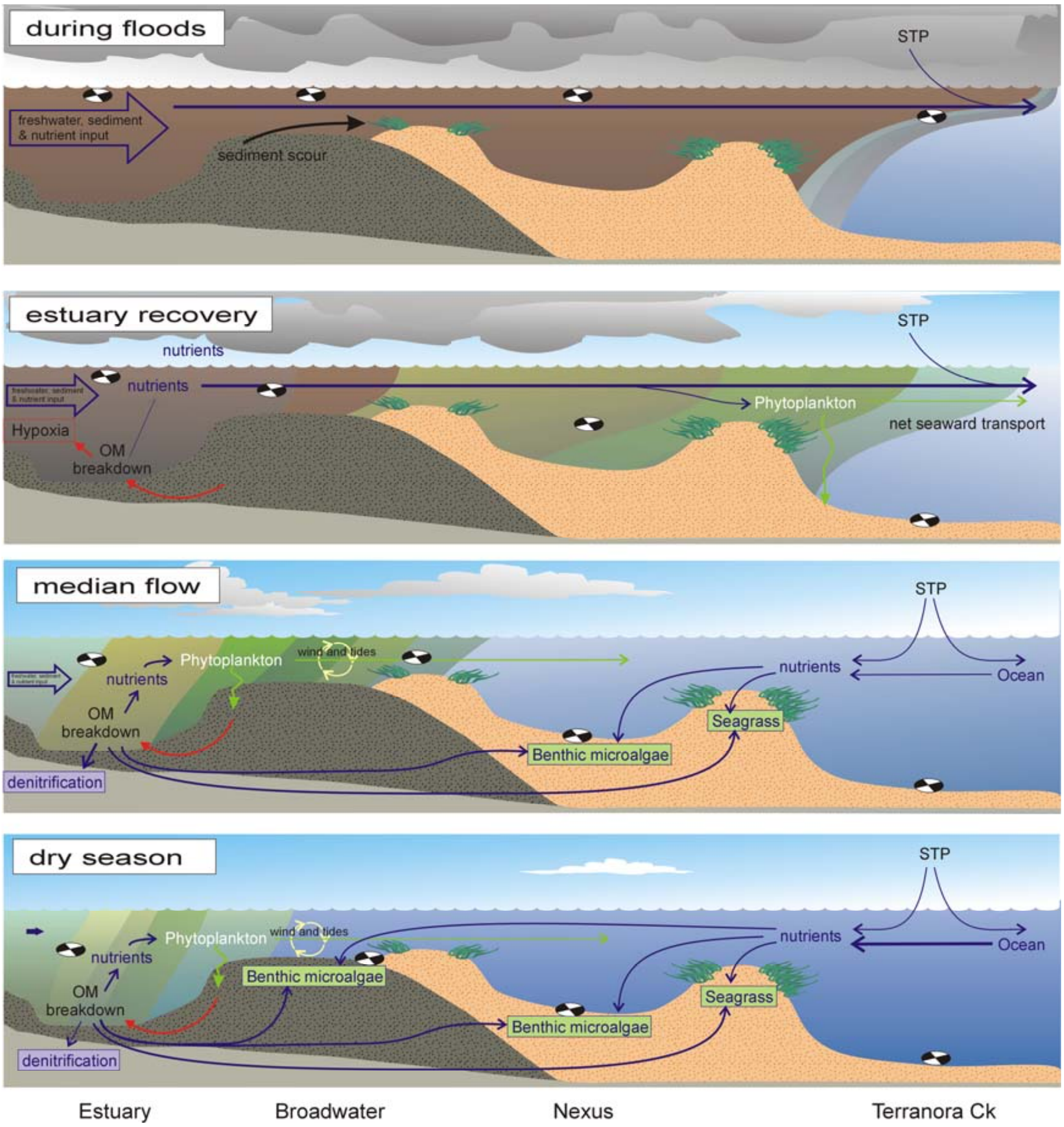


Figure 120 Seasonal progression of estuarine function in the Cobaki – Terranora system. The relative magnitude of freshwater inflows is indicated by the block arrows on the left of each stage. The relative importance of nutrient cycling pathways is indicated by the size of blue arrows. Light penetration depth is indicated by the secchi symbols.

10 Recommendations for ongoing monitoring

10.1 Summary of current study strategy

The sampling strategy employed during this study yielded a dataset of variable quality that nonetheless provided good spatial and temporal coverage of water quality along the Tweed and Rous estuaries. The dataset allowed sufficient resolution to identify the impacts of point source inputs against the background of diffuse catchment inputs and tidal mixing. The study provided good insights into estuary function in response to diffuse and point source inputs. The spatial and temporal coverage provided an appropriate dataset for development of a simple 1D salt balance model.

10.2 Ongoing monitoring considerations

The adoption of any ongoing water quality monitoring strategy in the Tweed estuary should take into account the needs of all stakeholders to better rationalise strategy aims and resources. In developing a future strategy, Tweed Shire Council should carefully consider 1) the reasons for undertaking monitoring, 2) clear aims and objectives, 3) the best rationalisation of available resources, 4) the need for low level nutrient analysis, 5) the archiving, end use and interpretation of data, and 6) appropriate system specific guidelines to compare data against.

10.3 Sampling locations

The use of boat based sampling in the Tweed estuary provided a far superior result than bank based sampling by allowing the collection of mid stream samples and efficient travel between sample locations. The adoption of fixed sampling locations rather than sampling according to salinity (e.g. a sample taken every 2 PSU) loses some resolution in describing processes associated with the mixing of freshwater runoff with oceanic water along the estuarine gradient. However, there are significant advantages in the fixed location approach: 1) quicker and easier to find sample locations, 2) the entire estuary is sampled every time, therefore floodplain inputs are always accounted for, 3) better accounts for morphological impacts on water quality (e.g. depth), 4) data better suited for modelling. Despite the fact that the salinity gradient was periodically under sampled, the current sample locations still provided a reasonable description of salinity dependent processes.

10.4 Sampling timing

The monthly sampling frequency provided the minimum resolution required to describe seasonal changes in water quality, however some resolution was lost to properly describe dynamic changes to the estuary structure through the wet season. Ideally, routine sampling in the estuary should be supplemented with event based sampling of major freshwater inputs to the system. This provides important information about catchment concentrations which are the primary driver of water quality in the estuary during high flow. These data are invaluable contributions to the development of a robust ecosystem response model of the estuary.

10.5 Sample analysis

A primary limitation of the current study was poor analytical protocols for low level nutrients in estuarine waters. Inter-laboratory comparisons undertaken between TSC and NSW OEH laboratories in Lidcombe NSW of duplicate samples taken during the first 6 months of the monitoring period. These comparisons revealed that TSC laboratory protocols resulted in the following artefacts:

1. Total nitrogen concentrations were routinely overestimated by 30 – 40%.
2. Total phosphorus concentrations were routinely underestimated, with many high concentration samples analysed as 'below detection' by TSC.
3. Many inorganic nutrient concentrations (NH_4^+, NO_x and DIP) were routinely underestimated, with many high concentration samples analysed as 'below detection'.

These artefacts seriously compromise data quality and add noise to environmental trends. This reduces the data's utility in 1) interpreting critical estuarine functions such as N:P ratios; and 2) comparisons with ANZECC guidelines (many samples that were 'below detection' were in fact likely to have exceeded guidelines).

These artefacts most likely arise from variable salt effects on the colorimetric methods used for nutrient analysis. Unless analytical protocols are carefully adjusted to account for these effects, samples taken from along an estuarine gradient (i.e. with variable salinity) and analysed in a single analytical run will yield highly unreliable data. It is highly recommended that TSC liaise with NSW OEH to improve their analytical protocols.

In addition, physico-chemical field data was unreliable due to the occurrence of many salinity values in excess of 40 for lower estuarine samples[17]. This suggests poor probe calibration protocols and / or in appropriate probe for the accurate measurement of salinity across the natural estuarine range.

10.6 Interpretation framework

Due to the highly variable nature of estuarine water quality, all monitoring data need to be placed into context with the environmental conditions at the time of sampling. For example, the increase in nutrient concentrations in the lower estuary during the wet season is a natural response to higher freshwater flows and diminished oceanic water influence. It would be misleading to interpret this result as necessarily a decline in estuary health rather than a natural part of the annual cycle in water quality.

It is therefore recommended that an interpretation framework be developed that provides reach specific (e.g. lower, middle and upper estuary) indicator thresholds accounting for season and freshwater flow. This should ideally be done using an appropriately formulated response model of the estuary which accurately describes the physical and biological responses to environmental drivers. This approach is consistent with recommendations contained within the (ANZECC 2000) guidelines.

[17] Seawater along the NSW northern rivers coastline varies between 35 and 36.5 PSU.

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13 Glossary

Algal bloom – the rapid growth of phytoplankton resulting in a high biomass in the water column.
Ammonium – inorganic nitrogen compound (NH_4^+) readily available to marine plants for growth.
Anoxic – an oxygen-free environment.
Anthropogenic – any phenomenon caused by human activities.
Autotrophic – refers to organisms that are able to fix carbon dioxide to form organic matter (e.g. plants and algae). May also refer benthic community metabolism (see also Benthic community metabolism).
Benthic – belonging to the bottom, or sediments, of the estuary.
Benthic community metabolism – refers to the sum of metabolism by all benthic plants and animals over 24 hours, e.g. benthic community metabolism may be net autotrophic (production of organic matter by algae exceeds breakdown by bacteria), or alternatively net heterotrophic (breakdown of organic matter by bacteria exceeds production by algae).
Benthic fluxes – exchange of nutrients and gases between the sediments and water column.
Benthic microalgae – microscopic algae living in the surface sediments.
Biochemical oxygen demand (BOD) – a measure of the amount of oxygen that will be consumed by biological and chemical processes over a given time period (usually 5 days).
Biomass – The living weight of plant or animal material (organic matter).
Chlorophyll-*a* – The green pigment in plants used to capture and use energy from sunlight to form organic matter (see photosynthesis). Concentrations of chlorophyll-a are used as an indicator for phytoplankton and benthic algae biomass.
Coupled nitrification-denitrification – where the processes of nitrification and denitrification occur in close proximity within the sediment.
Denitrification – the reduction of bio-available nitrate (NO3-) to unavailable nitrogen gas (N_2) by bacteria.
Dissolved inorganic nitrogen (DIN) – the total of all inorganic nitrogen compounds; ammonium (NH_4^+) + nitrite (NO_2^-) + nitrate (NO_3^-). DIN is recognised as the most bioavailable form of nitrogen used by plants for growth.
Dissolved organic nitrogen (DON) - the total of all organic nitrogen compounds; includes urea, proteins, amino acids etc. DON can be used by both plants and bacteria for growth.
Dissolved inorganic phosphorus (DIP) – the most bio-available form of phosphorus. Also known as phosphate (PO_4^{3-}), reactive phosphorus, or dissolved reactive phosphorus (DRP).
Diurnal – refers to biological rhythms occurring in phase with the light / dark cycle.
Ecosystem – refers to all the biological and physical parts of a biological unit (e.g. an estuary, forest, or planet) and their interconnections.
Eutrophication – the process of nutrient enrichment of a water body resulting in the increase in plant biomass (algal blooms) and bacterial decay (heterotrophic activity). Often results in a reduction in species diversity, visual amenity, and the prevalence of toxic algal species.
Food web – the predator / prey interactions of an ecosystem.

Freshwater residence time – the time (in days) that freshwater stays within an estuary before being transported to the sea by advection and tidal mixing.
Grazing – the eating of plants (e.g. phytoplankton) by animals (e.g. zooplankton).
Heterotrophic - refers to organisms that source their energy from the breakdown of organic carbon (e.g. bacteria, humans). May also refer benthic community metabolism (see also Benthic community metabolism).
Light attenuation – the absorbance of sunlight by dissolved and particulate matter in a water body.
Mole – a chemical mass unit defined as containing a set number of atoms or molecules (Avagardro's number = 6.022 * 1023)
Nitrate – an inorganic nitrogen compound (NO_3^-).
Nitrification – the oxidation of ammonium to nitrate by bacteria.
Nutrient budget – a simple model quantifying nutrient loadings (by weight) to a waterway from different sources over a given time period (e.g. one year).
Nutrient limitation – the restriction of phytoplankton growth by the low concentration (availability) of a nutrient.
Oxygen saturation – the amount (concentration) of oxygen dissolved in water expressed as a percentage of the maximum concentration of oxygen possible at a given temperature and salinity.
Pelagic – belonging to the water column.
Phosphate – see dissolved inorganic phosphorus.
Phytoplankton – microscopic single-cell plants growing in the water column.
Primary production – the formation of organic matter by autotrophs (e.g. phytoplankton).
Pristine – undisturbed by human activities such as urban and agricultural development, pollution, erosion, weed infestations etc.
Redfield ratio – the relative proportions of nitrogen and phosphorus required by phytoplankton for growth (16:1).
Stratification – where there is a distinct difference in salinity (or dissolved oxygen) between the surface and bottom water in the water column of an estuary.
Sulphate reduction – the bacterial breakdown of organic matter in anoxic sediments using sulphate instead of oxygen. Produces hydrogen sulphide, the 'rotten egg gas' smell common in muddy sediments.
Total nitrogen – the total of all inorganic and organic forms of particulate and dissolved nitrogen.
Total phosphorus - the total of all inorganic and organic forms of particulate and dissolved phosphorus.
Turbidity – a measure of the amount of light-attenuating particles in a water body.
Well mixed - where there is a little difference in salinity (or dissolved oxygen) between the surface and bottom water in the water column of an estuary.

14 Appendix

14.1 Salt balance model theory

The 1D salt balance model treats discharge of freshwater from each sub-catchment and STP effluent discharges as a point source inputs to boxes along the estuary. Dilution of these inputs along the estuary is based on the following assumptions and calculations:

Let Q_s = the flow of seawater
Q_f = the flow of freshwater
Q_e = input flow

The salt balance in the estuary requires that:

$$QsSs = (Qs + Qe + Qf)*S$$

where Ss = seawater salinity
S = the average salinity in the vicinity of the input point

Solving for Qs gives

$$Qs = (Qe + Qf)*S / (Ss – S)$$

The total flow (Qt) available for diluting the input flow equals

$$Qt = Qs + Qe + Qf = (Qe + Qf)*Ss / (Ss – S)$$

The average nutrient concentration (C) in the estuary at the point of input due to loading from the sub-catchment or STP equals

$$C = M / Qt$$

where:

M = the nutrient loading from the sub-catchment or STP in units of mass per unit time

If mixing (or dilution) of the input material is conservative, the concentrations downstream of the input point can be calculated

$$Cx = Cd * (Ss – Sx) / (Ss – Sd)$$

where:

subscript x = any point in the estuary
subscript d = the input point

The concentrations upstream of the input can also be calculated

$$Cx = Cd * (Sx / Sd)$$