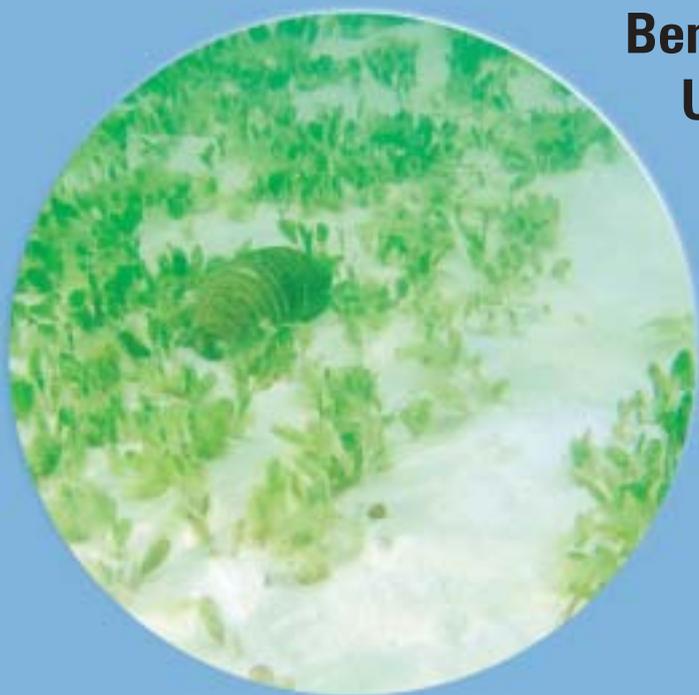**Cooperative Research Centre for Coastal Zone, Estuary & Waterway Management**

Benthic habitat function: Understanding benthic community metabolism using dynamic system models

R. Johnstone

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THE UNIVERSITY
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Estuary & Waterway Management

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Executive summary

Results from previous studies in Moreton Bay have highlighted the increased load of nitrogen (N) and phosphorous (P) that is reaching the bay compared to historical levels (e.g. Dennison & Abal, 1999). Concomitantly there has been a growing concern over the frequency of events such as toxic algal blooms, which have been attributed to alteration of the *natural* nutrient dynamics in the bay and the development of eutrophic conditions. Further, research undertaken by Johnstone (2001) has shown that, depending on the habitat type and location, the level of nutrient remineralisation and community respiration can vary significantly so that different areas of the bay demonstrate a varying capacity to deal with nutrient inputs.

Despite the studies mentioned above, our understanding of how the key benthic habitats function within Moreton Bay, and how they might contribute to the overall performance and capacity of the bay ecosystem is limited. Whilst there has been extensive sampling and monitoring of water column nutrients and some measurement of dynamics, there have been only sporadic and patchy assessments made of the benthos function in the bay. Accordingly, our understanding of the significance that sediment or benthic habitat processes might have for nutrient dynamics overall is limited.

This project utilised a case study area on the Eastern Banks of Moreton Bay to compare the nutrient dynamics and community metabolism of different habitat types with their classification based on their plant community composition. The site selection and classification utilised existing maps and information from the Queensland Environmental Protection Agency (EPA), Department of Primary Industries and Fisheries (DPIF), and the remote sensing validation work conducted by Phinn and group (Phinn and Dekker, 2004).

Benthic metabolism and nutrient flux measurements were undertaken using benthic chambers, as have been used elsewhere with good success (Johnstone *et al.*, 1990; Heutel and Rusch, 2000; Wild *et al.,* 2004).

The results of the benthic metabolism and flux measurement work were used to produce dynamic systems models. The models aimed to capture and characterise key elements of benthic habitat types with regard to biomass, nutrient, and oxygen fluxes in an idealised and controlled experimental environment. The intent of the modelling is to allow managers to formulate hypotheses of nutrient dynamics for each of the eco-types being studied and to then test these against the models and the data that underpins them. Accordingly, on discussion with potential end-users about their critical areas of concern, or aspects to be tested, the models can evolve to allow alteration of the

associated parameters and influences according to the field data available from this study and elsewhere.

The benthic flux measurements highlighted both the temporal and spatial variability in the net exchange of nutrients that occurs between the benthos and the overlying water. Much of the spatial variability is likely due to variations in the geochemistry of the underlying sediments as well as the infauna that exist within them (e.g. Krantzberg G, 1985; Kristensen E., 2000). Importantly, however, there was a tendency toward the decreased release of dissolve inorganic nitrogen (DIN) from sediments in winter where the sediment had a significant primary producer component. In summer the efflux of DIN was more widespread among sites and a consistent release of DIN was observed across most of the study areas. This seasonal variability may result from differences in the geochemical and Redox conditions within sediments underlying seagrass and algal communities. Under warmer conditions in summer, oxygen solubility is likely to be lower and thus sediments are likely to be more reducing due to the heightened community metabolism and relatively lower availability of free dissolved oxygen (Boudreau and Jorgensen, 2001). This in turn would likely to lead to larger pools of dissolved nitrogen that may exceed uptake rates by sediment bound organisms and plants. This relationship between Redox potential, temperature and nutrient pools has been reviewed elsewhere and noted in different marine sediments (Thamdrup *et al.*, 1998). By comparison to the DIN fluxes, no exchange of soluble reactive phosphorous was observed in summer, one site showed a net uptake in winter.

Also of note in the summer benthic flux work is the general agreement between results for DIN in the three habitat types of seagrass-algae, sparse seagrass, and sparse seagrass-algae. This implies that the seagrass component may be very significant in influencing nutrient exchanges but also suggests that in summer, these three habitat definitions may exist based on composition but not in terms of nitrogen exchange rates. In winter this situation did not hold and the three habitat types behaved with great variability so that any statistical similarity could not be defined. Further fieldwork involving larger replicate numbers may help resolve this aspect further. Notably, the open sediment areas showed some seasonal variation but behaved similarly in summer and winter with stronger fluxes observed in summer.

As noted above, the use of habitat classification systems based on their plant community composition does not always accurately represent their performance in terms of nutrient dynamics and this may vary between seasons. In this regard, it is critical that any models or management approaches that use a community composition-based classification system to describe their ecosystem must

undertake process studies before ascribing a remineralisation or nutrient dynamic role. From the results in this study, it is clear that not all seagrass or seagrass-algal assemblages behave the same and so pooling of these may have negative impacts on estimates of their role within the wide ecosystem.

With regard to the modelling aspect of the project, the modelling process proved very useful in identifying both the limits of current knowledge on Moreton Bay benthic nutrient dynamics as well as in helping to capture the key elements that warrant further research in order to adequately understand the influences that anthropogenic nutrient loads may have on the benthos. For example, by simulating the different habitats and their function with the model it is clear that, depending on where a nutrient load may be deposited (open sediment or seagrass area), the load may enter an environment where there is already a net release of materials (e.g. open sediment in summer) or an environment where there is likely to be a net uptake of dissolved nutrients. In the later case the potential for nutrient removal from the water column is higher and so outcomes such as phytoplankton blooms may be lower within the immediate timeframe of load deposition. Conversely, the load may reinforce the release of nutrients in the immediate term and augment phytoplankton growth.

Similarly, in the composite model, the user is able to test notions about how different relative ratios of the benthic plant community might influence nutrient dynamics. This provides insight into why it is necessary to not focus only on particular species or assumed assemblages, but to also consider that there are a diverse range of community compositions that may serve a particular role in nutrient removal and control. On this later aspect, the models have been used to elicit responses and to augment a discussion between scientists, engineers and managers as part of a wider discussion on algal blooms in Moreton Bay (Healthy Waterways Partnership, March 2006). This set of discussions validated the modelling approach and indicated that such integrated framework and model approaches are very useful in both the communication of ideas as well as in reaching consensus on what the key issues may be from a technical and managerial perspective.

Background and aims of the project

Results from previous studies in Moreton Bay have highlighted the increased load of nitrogen (N) and phosphorous (P) that is reaching the bay compared to historical levels (e.g. Dennison and Abal, 1999). Concomitantly, there has been a growing concern over the frequency of events such as toxic algal blooms, which have been attributed to alteration of the 'natural' nutrient dynamics in the bay and the development of eutrophic conditions. Further, research undertaken by Johnstone (2001) has shown that, depending on the habitat type and location, the level of nutrient remineralisation and community respiration can vary significantly so that different areas of the bay demonstrate a varying capacity to deal with nutrient inputs.

Despite the studies mentioned above, our understanding of how the key benthic habitats function within Moreton Bay, and how they might contribute to the overall performance and capacity of the bay ecosystem is limited. Whilst there has been extensive sampling and monitoring of water column nutrients and some measurement of dynamics, there have been only sporadic and patchy assessments made of the benthos function in the bay. Accordingly, our understanding of the significance that sediment or benthic habitat processes might have for nutrient dynamics overall is limited.

Against this background, a recent review by Wulff and Johnstone (in prep.) highlighted the hydrodynamic modelling that has been undertaken by the Queensland EPA, and the significant benthic mapping that has been achieved by Phinn and Roelfsema from The University of Queensland (Phinn and Dekker, 2004). The existence of these data sets and models provides a unique opportunity to integrate them with the benthic data to obtain models against which we can test our current understanding of ecosystem function at a level of detail and accuracy previously unavailable to us. This would strengthen the confidence and integrity of the support given to management and decision–makers responsible for the Moreton Bay ecosystem.

The central aim of this study was to augment the information available on benthic habitat metabolism, its interaction with the overlying water column, and the variation this might have seasonally. Together with spatial information being produced by concomitant studies, this project would then seek to provide an integrated model that could be used to assess the significance of these processes at different scales.

It is anticipated that the results from the project will provide the basis of new conceptual models and interactive models that can be used to underpin ecosystem management and appraisal for Moreton Bay. The level of detail that

the results will provide will greatly enhance the accuracy and integrity of any attempts to understand and model the role of benthic processes and habitats within the larger Moreton Bay ecosystem.

Approach and methods

General approach

The project research was limited to a case study area where the questions being posed could be tested and yet the results obtained could be applied to other locations in the bay to consider the wider hypotheses being tested. In this light, the approach of the project has been to initially undertake a broad survey to identify a suitable case study area and then to study established sites within this area. The initial survey examined previous benthic studies undertaken in the bay to identify the major habitat types and their major components, as well as to assess how 'representative' any particular study area may be of the bay or of reasonably common habitat types in such estuarine ecosystems. The survey utilised existing maps and information from the Queensland Environmental Protection Agency (EPA), Department of Primary Industries and Fisheries (DPIF), and the remote sensing validation work conducted by Phinn and group (Phinn and Dekker, 2004).

Selection of habitat types and locations

All of the fieldwork and benthic data in this project were collected from habitats on the Eastern Banks in Moreton Bay. The location of the study area is given in Figure 1. Although this area has been described in great detail in numerous studies (Phinn and Dekker, 2004; Dennison and Abal, 1999), it is important to note the following features:

- The banks are subject to significant oceanic influence due to their proximity to one of the main exchange channels for the bay that flows between the banks and Amity Point on North Stradbroke Island (see Figure 1).

- Because of their eastern location and exposure to oceanic influences, the banks are not necessarily representative of the Seagrass and algal communities in the western bay where waters are typically more turbid, the sediments different in their geochemistry, and the influence of terrestrial runoff is intermittent but high.

- Much of the Eastern Banks is either fully exposed or lies in very shallow water (≤15 cm) at low tide. This is especially so on the spring low tide and this restricts direct flux measurements to high tide conditions. This is discussed further below.

As noted earlier, this project had a direct interaction with the CRC remote sensing project of Phinn and Roelfsema and so this collaboration was used to identify habitats that had the following attributes:

- could be identified by the methods used in the remote sensing project and, thus, mapped accurately under the remote sensing methodologies;

- represented a significant spatial component (>15%) as a definable habitat or zone on the remote sensing maps for the wider ecosystem of the case study area – Eastern Banks; and

- could be validated using direct field survey techniques.

Within this context, an initial survey was conducted to assess the spatial extent and likely composition of the dominant communities in the study area. It should also be noted that these initial assessments were guided in part by the maps already produced by the Coastal CRC remote sensing team (Phinn and Dekker, 2005). An example of one such map is included below in Figure 2.
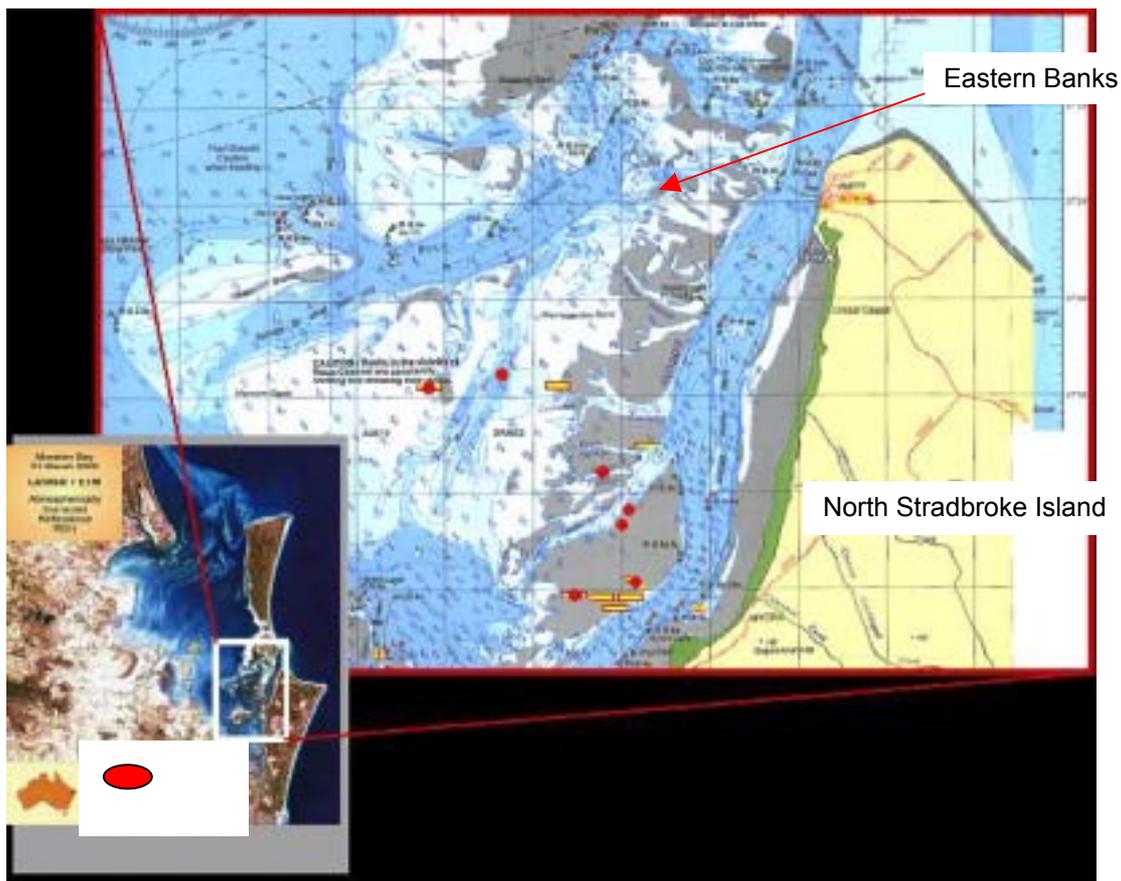
Figure 1: Moreton Bay. The location of the Eastern Banks in Moreton Bay. The red and yellow markers represent the initial pilot study sites used to test methodologies.
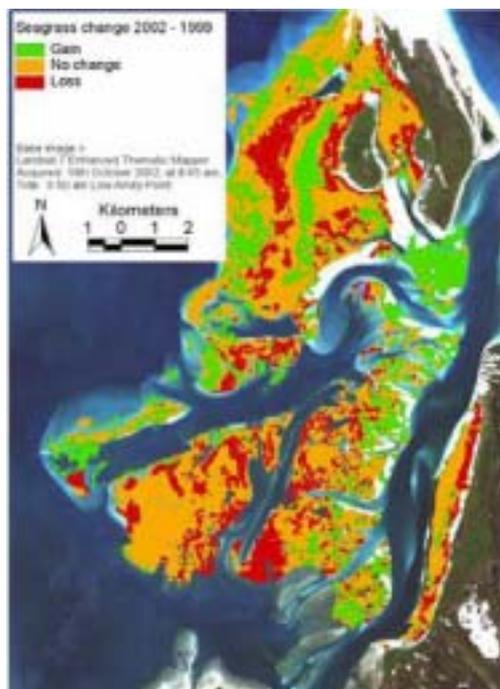

Figure 2: Map of seagrass habitat change over time derived from remote sensing data for Eastern Banks (from Phinn and Dekker, 2004).

Community metabolism and nutrient fluxes

The project undertook community metabolism and nutrient flux measurements in each of the benthic habitats defined in the case study area within Moreton Bay.

Benthic metabolism and nutrient flux measurements were undertaken using benthic chambers, as have been used elsewhere with good success (Johnstone *et al.*, 1990; Heutel and Rusch, 2000; Wild *et al.*, 2004). Chamber incubations were conducted over approximately 8 hours with close scrutiny of temperature and dissolved oxygen values to ensure that neither became high nor low enough to negatively impact flux rates. A picture illustrating the benthic chambers is presented in Figure 3.

Benthic metabolism and flux measurements were undertaken during summer (November 2004) and winter (August 2005) to assess seasonal differences in the rates observed.

Eight replicate chambers were deployed at each location. Four of the eight were randomly assigned black covers to provide a dark rate determination. The remaining four were left open to ambient light conditions.

Figure 3: Benthic chamber as used in benthic metabolism and benthic flux measurements. The dispersion attachment for mixing is not shown in this picture.

Temperature and light intensity were measured in chamber and external waters using a HOBO combined temperature and light logger. Dissolved oxygen was measured using a YSI dissolved oxygen sensor attached to a TPS logger. Recordings were made every 30 seconds for temperature and light, and approximately every two hours for dissolved oxygen.

Water samples were taken from the chambers every two hours using a 60ml syringe and needle. The needle was pushed through the propyl rubber membrane on the respective sampling port on each occasion to extract the sample. All water samples were immediately filtered through a 0.45 µm filter and

stored at -25oC until analysed. Analysis took place within two weeks of sample collection and used the methods of Yoshida et al., (1984).

Community species composition

Species composition was determined by using photo-quadrate and direct observation to classify species using the definitions of McMahon (2005). The level of classification was limited to verifying that the areas under consideration complied with the classes of habitat defined by Phinn and Dekker (2004) in their remote sensing derived map of Eastern Banks. This was further confirmed through discussion with remote sensing team members.

Based on a preliminary examination of the remote sensing maps and data from pilot field surveys, the habitat validation work was able to quickly focus on the main habitat types including seagrass areas, areas of open sediment, mixed algal/seagrass areas, and algal dominated areas. The areas selected for the study also sought to account for the range of input loads and environmental factors influencing the different habitats. However, limited resources hindered this significantly so a complete coverage across the potential range of values and a fully extensive spatial data set were not possible. This limited the strength of some of the statistical testing and comparisons. Ultimately this field element was given a lower priority to allow for greater input to the metabolic studies and model construction. In order to ultimately define representative sites for community metabolism studies a combination of the initial surveys, validation surveys and the species maps developed by Phinn and Dekker (2004) were used. This is discussed in the results section.

Model development

The models built in this project used the STELLA[1] Research Package V8.1. This is a dynamic systems modelling package that is freely available as a runtime version on the web and is thus very accessible to potential end-users without any direct cost. Also, the STELLA package has been used with great success in similar research elsewhere (e.g. Costanza *et al.*, 1998; Arquitt and Johnstone, 2004)

The modelling component of the project was undertaken in parallel with benthic metabolism studies conducted at Eastern Banks, Moreton Bay. The model assumes an area of biomass of one square metre and a volume of overlying

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[1] STELLA is the registered trademark of ISEE Systems Inc. www.iseesystems.com

water of one cubic metre. We assume that there is no exchange of biomass, nutrients, oxygen or water with a surrounding environment beyond the immediate overlying water, i.e. no external sink[2]. Daylight period is the only external influence and in the form reported here is the primary forcing function for the modelled system. The model attempts to capture and characterise key elements of benthic habitat types with regard to biomass, nutrient, and oxygen fluxes in an idealised and controlled experimental environment. As noted previously, the intent of the modelling is to allow managers to formulate hypotheses of nutrient dynamics for each of the eco-types being studied and to then test these against the models and the data that underpins them. Accordingly, on discussion with potential end-users about their critical areas of concern, or aspects to be tested (see footnote 2), the model can be modified to allow alteration of the associated parameters and influences according to the field data available from this study and elsewhere.

From the habitat maps and field validation work five benthic biomass/habitat types were ultimately defined and depicted within this modelling framework. They were:

1. Seagrass/algal;

2. Sparse seagrass;

3. Sparse algal/seagrass;

4. Algal dominated; and

5. Open sediment.

As discussed later, these complied with the classification proposed by Phinn and Dekker (2004) from their remote sensing work and provided a representative coverage of metabolic range for each habitat type defined for the Eastern Banks by the remote sensing maps. A detailed definition of each habitat class is given in Phinn and Dekker (2005).

The model was constructed so that hypotheses are tested through simulation exercises allowing the user to assess the robustness of the hypotheses and to compare simulated data with data obtained through field experiments. Field experiments also allowed us to update the model with more accurate parametric and structural assumptions. As a consequence, beyond this project the model can in turn help provide focus to field investigations by identifying influential parameters through automated sensitivity analysis. A description of the initial

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[2] This is the initial state of the model. The intention is that beyond the life of this project users may then link this model to a wider ecosystem model where a sink term will be necessary.

general model structure is provided in Figure 4. The later model form is presented in the section on models.
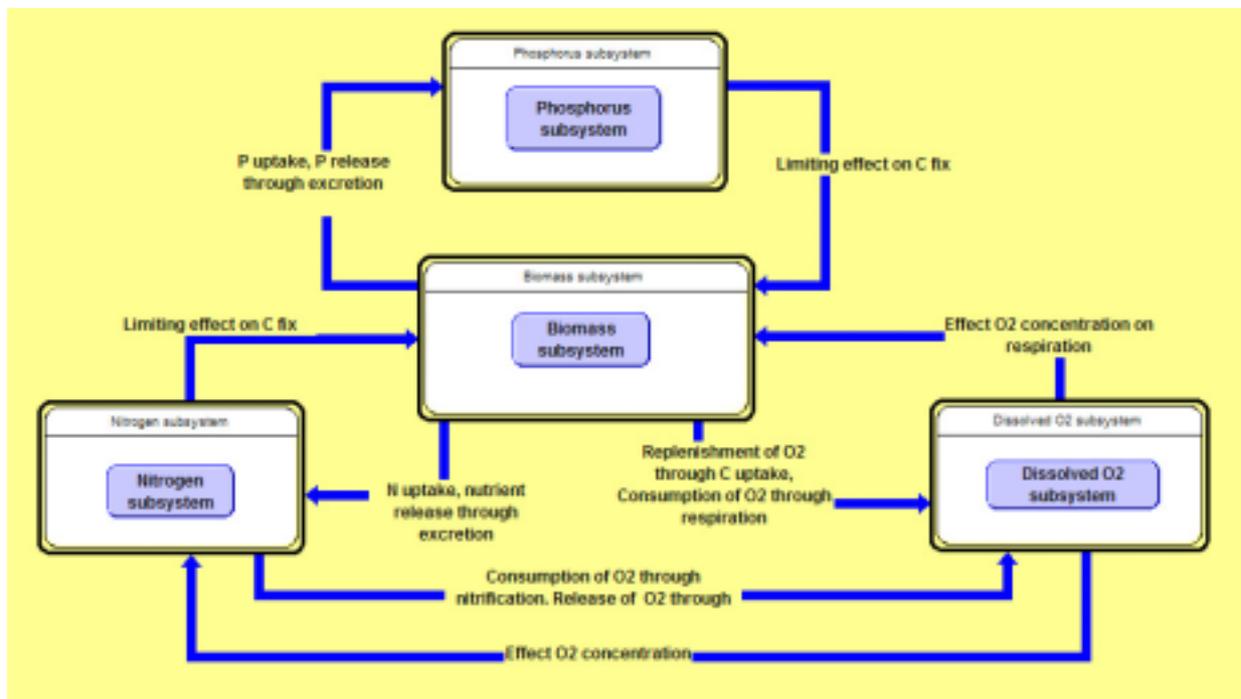

Figure 4: Summary diagram of the main components to the initial benthic habitat model. This structure is applied to each habitat using the field and literature data available to derive the interaction terms.

Results

Habitat descriptions

As noted in the approach taken, an initial, pilot, assessment of the habitats was made of the benthos on the Eastern Banks as well as an assessment of the relative seagrass and algal composition of these different areas. These initial surveys lead to the definition of four habitat types illustrated in Figure 5.

Although these habitats could be identified readily in the field, it became difficult to ascribe borders to each type without a major survey that far exceeded the resources of the current project. As a consequence, their use in providing representative metabolic and nutrient flux estimates was considered to be limited. In this light, the existing remote sensing maps for the banks were reconsidered and after deliberation with the remote sensing team (Roelfsema, pers. comm.) sites were chosen that were central to the community types discernable by the remote sensing methods, and which represented the bulk of the benthos at the banks. Species maps used for site location are provided in Appendix 1.

The habitat or community types were then defined as 1) seagrass/algal; 2) sparse seagrass; 3) sparse algal/seagrass; 4) algal dominated; and 5) open sediment.

These categories covered the relative compositions reflected in the remotely sensed communities and were easily identifiable within the map locations taken from the remote sensing maps.

With the exception of the open sediment habitat type, all other areas were within a depth range of up to approximately 2.5 m depending on the state of tide. None of the study areas were exposed during low tide and typically had a minimum of 50 cm of overlying water. The open sediment habitat ranged in depth from exposed areas to >3 m depth at low tide. For the current study, only areas within the same depth range as the other habitats were considered.

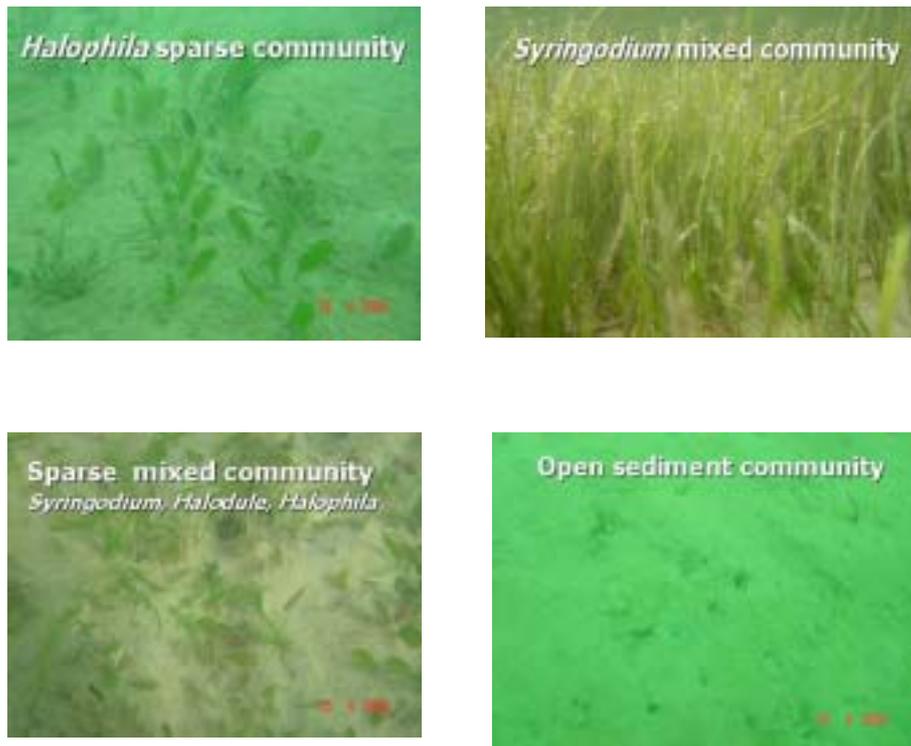The location for these sites is presented on Figure 6.

Figure 5. Images of the four initial habitat types defined from field surveys in the pilot study.

Figure 6: Location of study sites based on field observation and remote sensing maps.

Nutrient dynamics

In order to characterise the nutrient behaviour of the benthos on Eastern Banks a number of surveys and benthic metabolism studies were undertaken.

The initial study was aimed to provide a gross assessment of the porewater nutrient concentrations at the initial sites chosen for the Banks. The results showed that there were significant concentrations of dissolved inorganic nitrogen (DIN) and soluble reactive phosphorous (SRP) at different locations across the banks area. This suggested that there was strong potential for nutrient exchange between the water column and benthos. The results are summarised below and in Figure 7. The figures illustrate the concentration of each dissolved nutrient species with depth within the sediments at each location. From this concentration gradient and the porosity estimates made on the sediments (Johnstone, unpublished) it is possible to estimate the potential flux of a nutrient species between the sediment porewater and the overlying waters.

As noted, this aspect of the study was intended to identify the potential for sediment fluxes so that the ensuing chamber studies could be established with confidence and approximate ranges of expected fluxes could be used as a coarse validation of the flux rates observed.
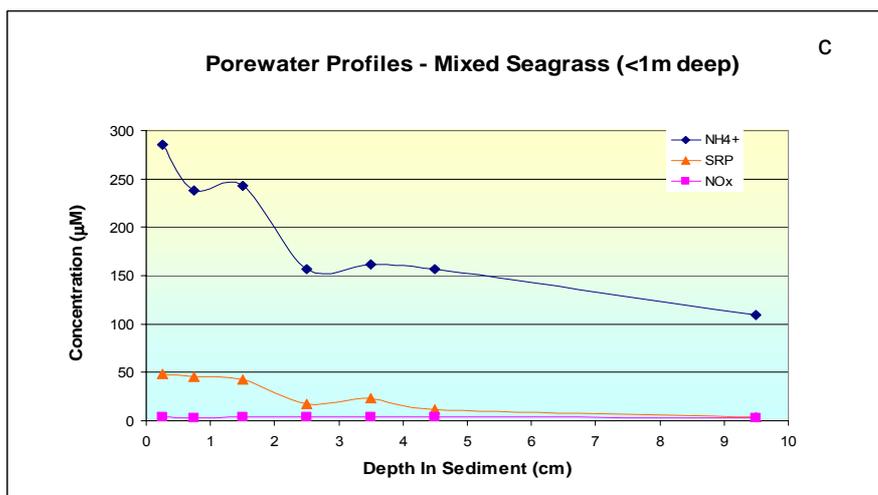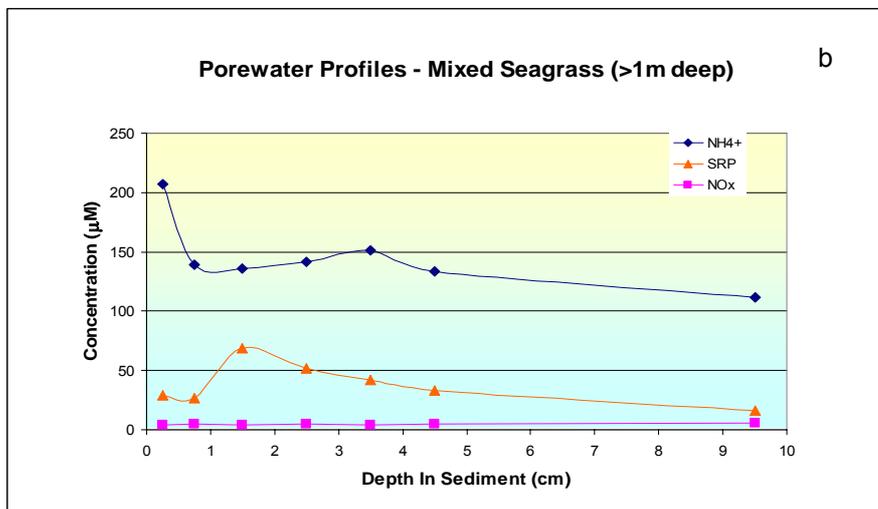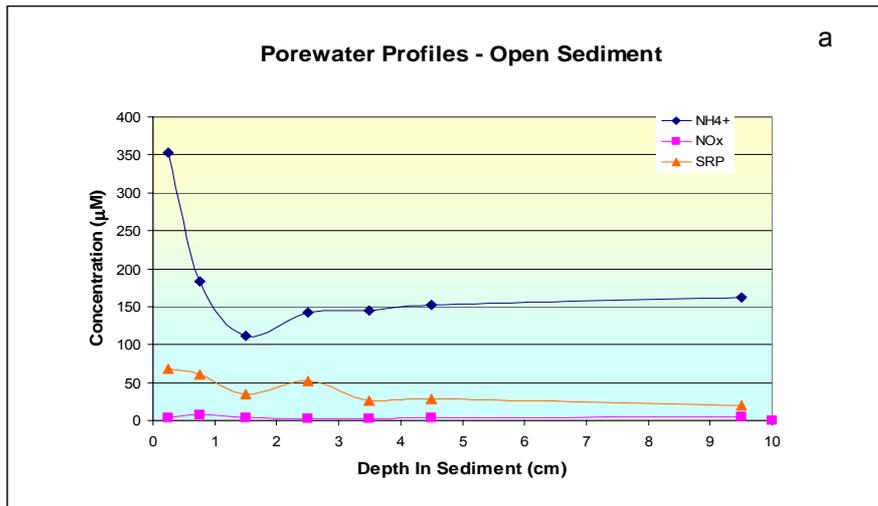
Figure 7: Porewater nutrient profiles for broad habitat types at Eastern Banks. (a) open sediment; (b) mixed seagrass deeper than 1 m; (b) mixed seagrass in less than 1 m water depth.

Summer nutrient fluxes

Nutrient flux rates between the benthos and overlying water column were within the range reported elsewhere (e.g. Zeigler and Benner, 1999; Stockenberg and Johnstone, 1997; Niencheski and Jahnke, 2002) and were in the mid range of rates reported for a similar sub-tropical seagrass-dominated ecosystem in the south eastern USA (Zeigler and Benner, 1999). Notably, Zeigler and Benner (1999) reported ammonium flux rates ranging between -228 and 336 $\mu M.m^{-2}.d^{-1}$, so that the benthos either consumed or released DIN depending on location and time of measurement. This compares with the range observed in the current study of between -205 and 45 $\mu M.m^{-2}.d^{-1}$. The observed rates were also within the range observed for other locations within Moreton Bay (Johnstone, in prep.).

As reported in numerous benthic nutrient studies (e.g. Zeigler and Benner, 1999; Stockenberg and Johnstone, 1997), it is not uncommon that the flux rates observed from benthic communities may vary significantly; both within study sites and between different study locations. In the case of the current study, it was necessary to consider the level of variation in order to assess whether there is the potential to obtain a mean rate (with confidence limits) that could be used as a representative rate for the given habitat. As shown in Figures 8, 9, and 10 each of the study habitats demonstrate the sort of variability that is reported in the literature for DIN fluxes, however, there is significant congruence between the mean rates observed from some habitats despite their different classification using floral species composition. The chamber that fell outside the range of the other replicates was observed to have a large animal burrow under it. This is likely to have produced the elevated total DIN concentrations due to animal irrigation of the underlying sediments.

The best example of this is seen in Figure 8 which presents the mean flux values for DIN in the three habitats which all have a seagrass component. Despite one chamber having a significantly different set of values, the remaining values all fall within a range that is not significantly different between habitat types. Notably, this group of habitats are collectively different from the remaining two defined habitat types, 'algae/sediment', and 'open sediment'.

In the case of 'algae/sediment' sites, one site showed a significant difference with a net uptake of DIN compared with the release seen at the other locations within the habitat. This variation may be due to a number of factors such as the presence or absence of large benthic fauna or the particular level of biomass of one of the algal components. This later aspect is still being evaluated.

In comparison to the other habitat types, the open sediment measurements all showed a release of DIN into the water column across a range of rates. Notably, however, none of the chambers showed a significant deviation from the mean

pattern and all rates fell within those observed previously in other locations from Moreton Bay (Johnstone, in prep*.).*
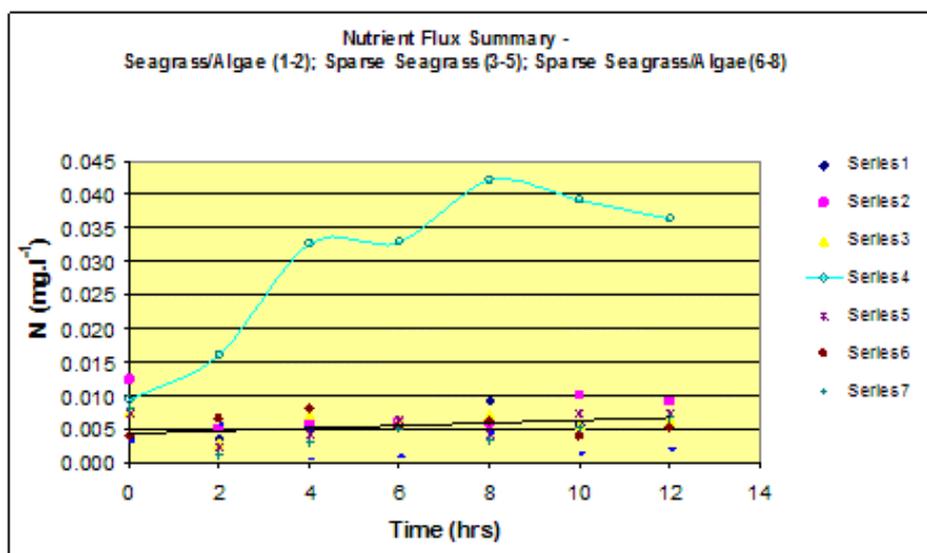

Figure 8: Mean DIN flux values for chamber experiments conducted in three seagrass influenced habitats at Eastern (Amity) Banks. Each point represents an individual replicate value.
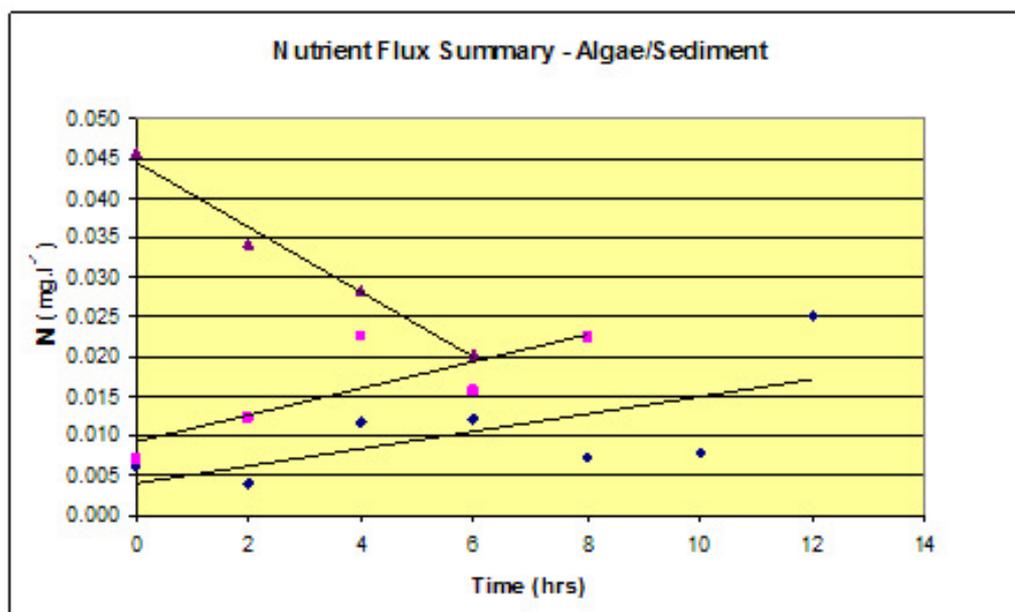

Figure 9: Mean DIN flux values for chamber experiments conducted in algal/sediment habitats at Eastern (Amity) Banks. Each point represents an individual replicate value.
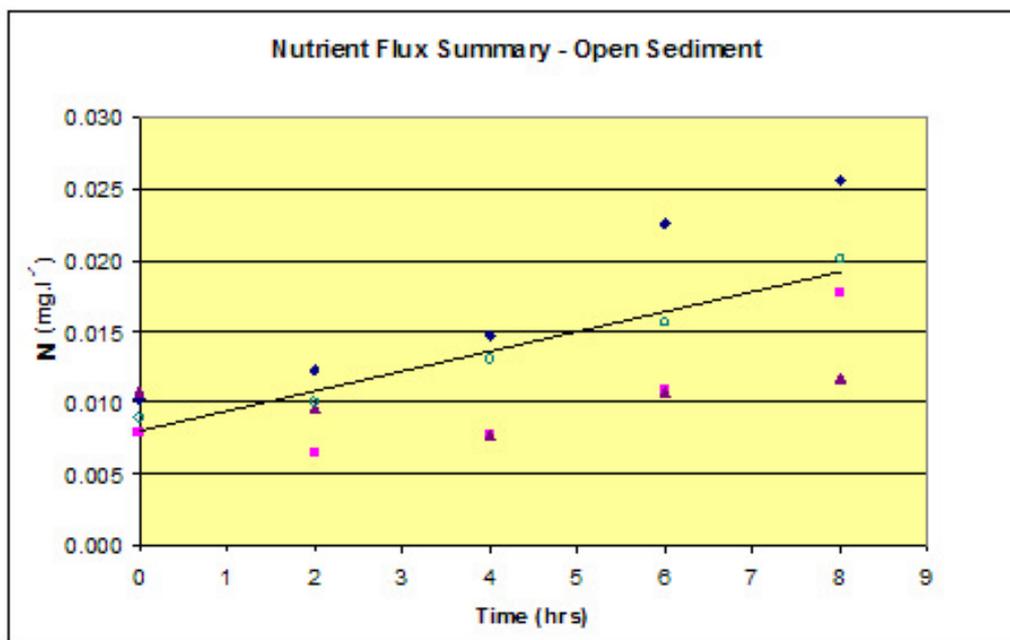
Figure 10: Mean DIN flux values for chamber experiments conducted in open sediment habitats at Eastern (Amity) Banks. Each point represents an individual replicate value.

Winter nutrient fluxes

As with the observed summer flux rates, the rates observed in winter were within the range reported elsewhere and within the range reported for a similar seagrass-dominated ecosystem in south eastern USA (Zeigler and Benner, 1999).

Winter nutrient fluxes showed high variability within most habitats and thus significant fluxes were only observed in some specific cases. The most notable of these is in the case of SRP, which only showed a significant flux in one habitat, seagrass/algal (Figure 11). The reasons for this are not clear but may be related to the animal activity that was observed in this area whilst sampling. Further work is needed to resolve this fully.

In the case of nitrate/nitrite (NOx) fluxes, the only significant flux was observed in the open sediment (Figure 12). In view of the mixture of processes supported in the more complex habitat types comprised of varying amounts of algae and seagrass, the occurrence of a clear trend might be less likely due to competition between uptake and release by the different processes. Conversely, the open sediment showed a net uptake of NOx. This might be expected in view of the potential for denitrification in these sediments, which would utilize the available NOx as substrate.

The other major component to the DIN pool, ammonium, showed no significant flux in three of the five habitat types with the only significant fluxes recorded in the open sediment and sparse seagrass habitats (Figure 13). Again, this might be expected given the lack or very low standing stock of benthic primary producers (consumers of DIN) in these two habitat types. This is consistent with observations made elsewhere (e.g. Stockenberg and Johnstone, 1997).
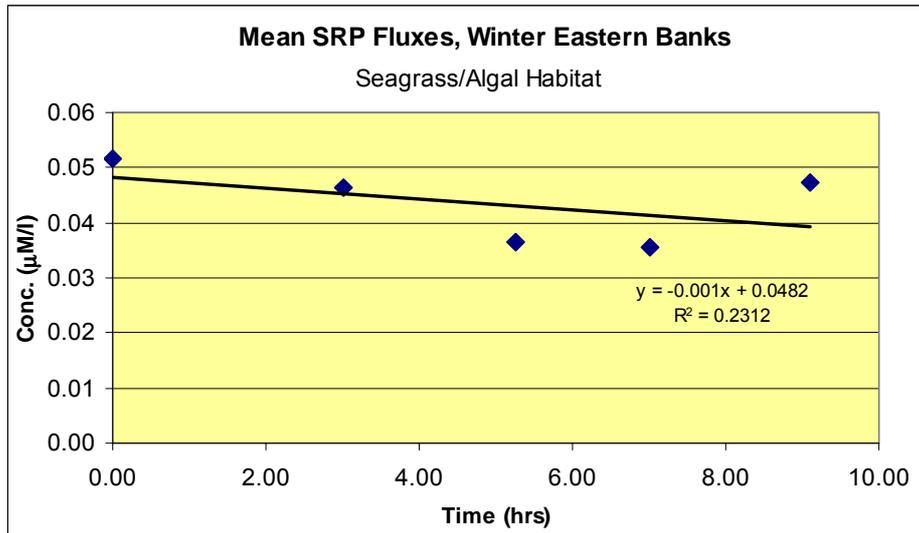

Mean SRP Fluxes, Winter Eastern Banks
Seagrass/Algal Habitat

$y = -0.001x + 0.0482$
$R^2 = 0.2312$

Figure 11: Mean winter SRP flux for seagrass/algal habitat at Eastern Banks. Linear regression line and r^2 values are given.

Mean NO_x Fluxes, Winter Eastern Banks
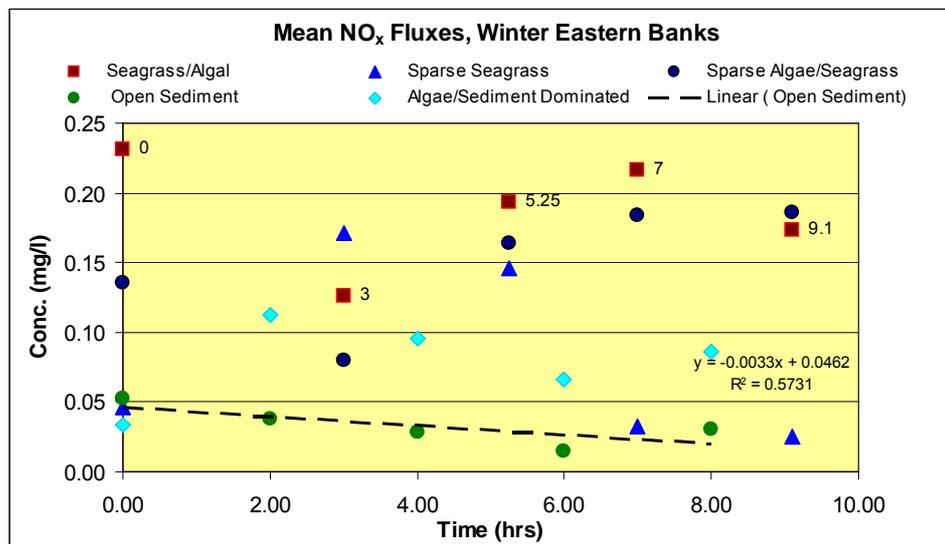
$y = -0.0033x + 0.0462$
$R^2 = 0.5731$

Figure 12: Mean winter NOx fluxes Eastern Banks. Linear regression is for open sediment habitat type. Linear regression line and r^2 values are given.
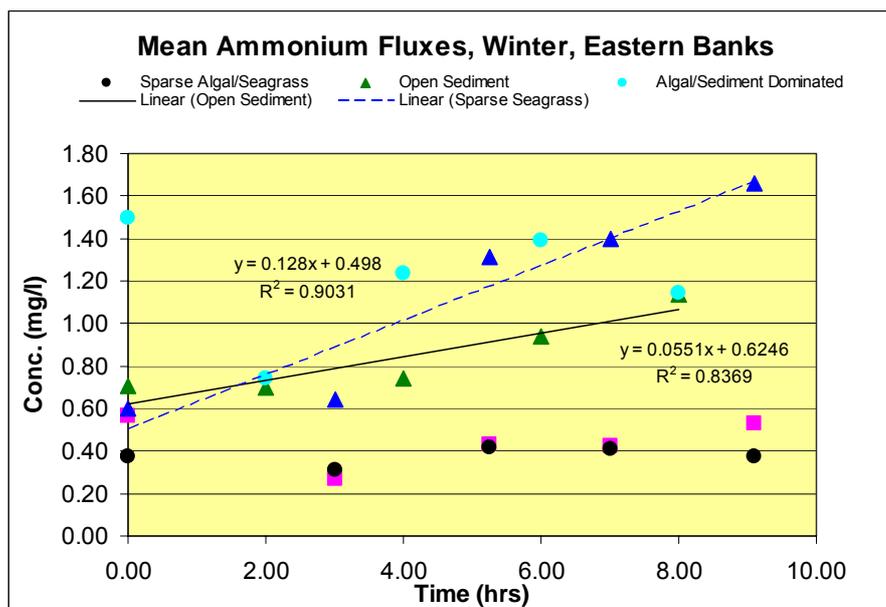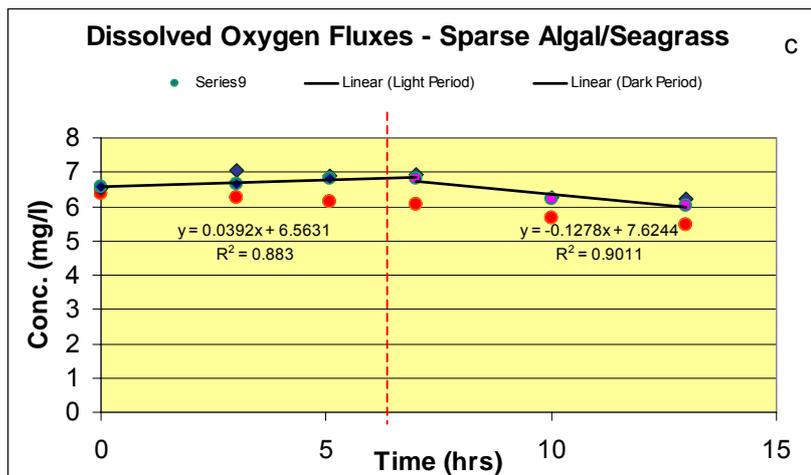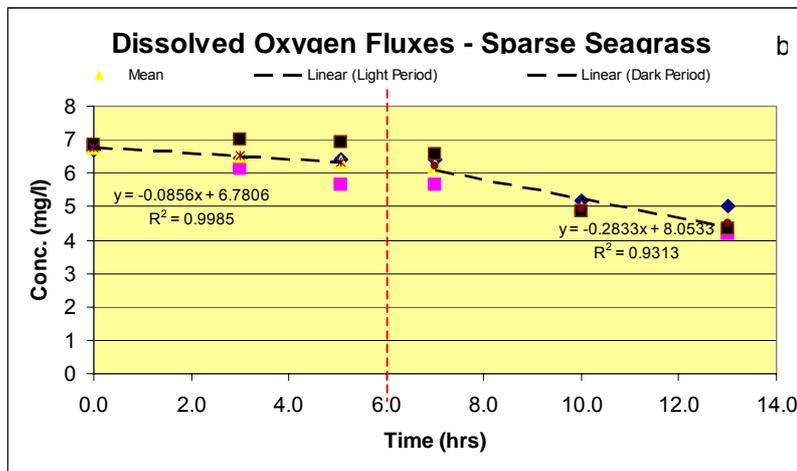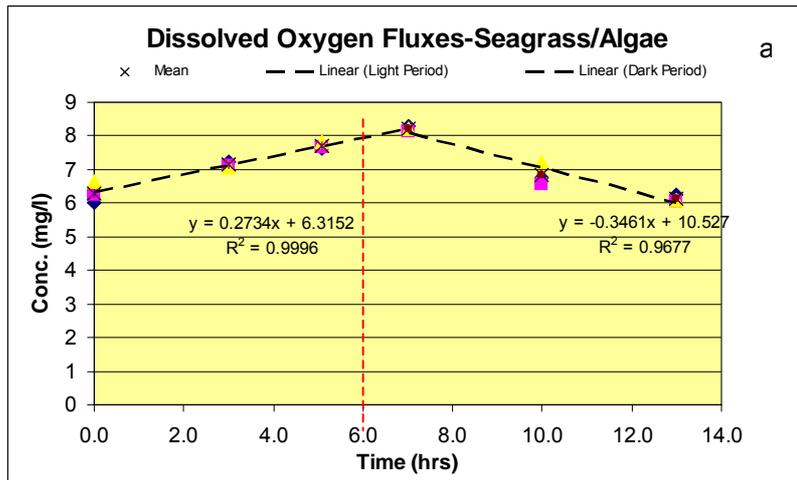
Figure 13: Mean winter ammonium fluxes, Eastern Banks. Linear regression show the only significant fluxes observed. Linear regression line and r^2 values are given. Each point represents an individual replicate value.

Community metabolism

In both the summer and winter sampling periods, habitat oxygen fluxes varied according to the relative representation of primary producing organisms (algae or seagrass) and sediments. As shown in Figure 14a, the most pronounced fluxes of dissolved oxygen were observed in the seagrass/algal habitat and a clear diurnal signal was apparent in all habitats with an algal or seagrass component. By comparison, the open sediment showed a clear net uptake of oxygen as might be expected in the absence of primary producers. The mean dissolved oxygen fluxes for all habitats are presented in Figures 14 a, b, c, d, and e.

Figure a: Dissolved Oxygen Fluxes-Seagrass/Algae

Figure b: Dissolved Oxygen Fluxes - Sparse Seagrass

Figure c: Dissolved Oxygen Fluxes - Sparse Algal/Seagrass

Dissolved Oxygen - Alga/Sediment Dominated   d

$y = -0.1933x + 7.1211$
$R^2 = 0.97$

$y = 0.0675x + 6.2303$
$R^2 = 0.9926$

Dissolved Oxygen Fluxes - Open Sediment   e

$y = -0.1382x + 6.505$
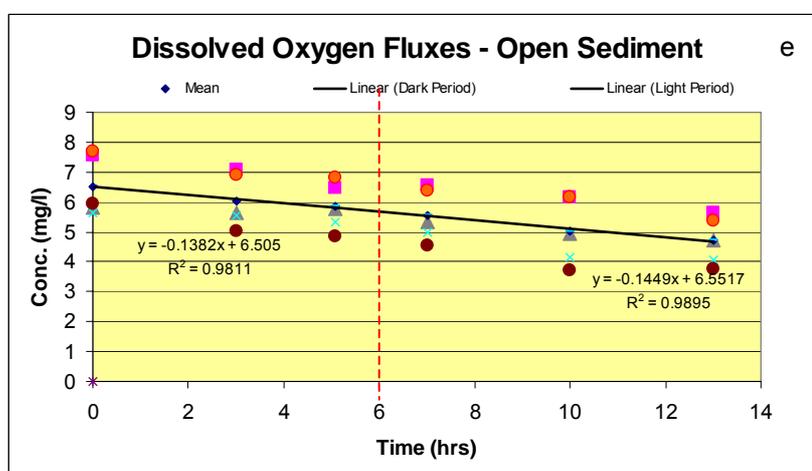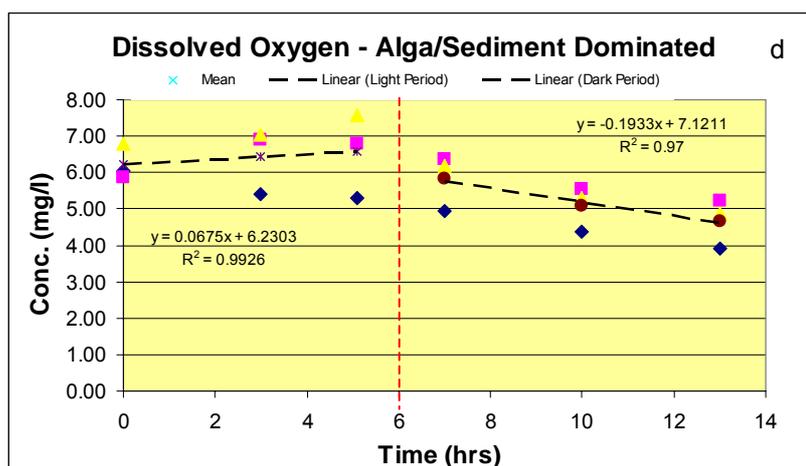$R^2 = 0.9811$

$y = -0.1449x + 6.5517$
$R^2 = 0.9895$

Figure 14 a, b, c, d and e: Mean Dissolved Oxygen flux values for chamber experiments conducted in each of the habitat types defined at Eastern Banks. Symbols represent individual replicate chamber measurements. The vertical dotted line represents the shift from the light period to the dark period. Each point represents an individual replicate value.

In terms of functional attributes for the respective habitats, the community metabolism results highlighted the gradient from highly autotrophic communities in the seagrass and algal dominated habitats, to the strongly heterotrophic communities in the open sediment. This is important when considering both the potential release of nutrients to the overlying water column as well as the sites where different primary and secondary production, and remineralisation processes are likely to dominate. These attributes are central to the management perspective that can be applied or tested with the models constructed from the field data.

The models

As noted previously, dynamic systems models were constructed using the STELLA modelling package. It should also be noted, however, that another more flexible, but less user–friendly package, VenSim was also used to compare the performance of some model components under the two packages and with different time-steps.

At the beginning of field data collection, an initial model framework was contrived to help guide the sampling and to provide insight into the information and processes that would need to be documented for a robust model to be produced. The initial model framework is presented in Figure 4. Once the field data was sufficient to describe the interaction terms, a second generation model framework was constructed and this has been the basis from which the initial and subsequent model forms were built. The framework is presented in Figure 15.

Model sensitivity analysis

In order to establish the internal logic and rigor of the dynamic systems model built from the benthic habitat flux data, the model was subject to a sensitivity analysis. In essence, this initial sensitivity analysis was aimed at establishing the basis for later examination of potential load and habitat variation scenarios.
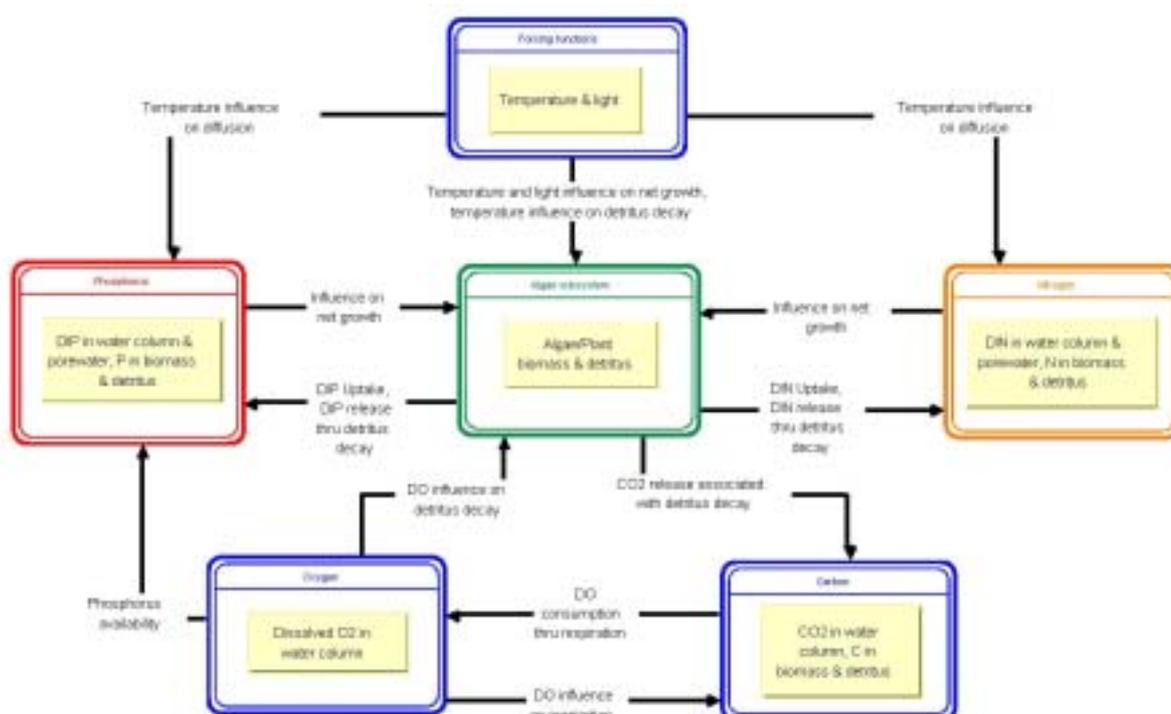

Figure 15: Component structure of second generation benthic model used for simulation of habitat nutrient responses relative to composition.

Sensitivity of the model behaviour was analysed for each of the five habitats defined from the fieldwork. The simulated variables are the water column concentrations of dissolved oxygen, NH_4, NOx, and PO_4. The distinguishing parametric values for each habitat are initial carbon in biomass and fractional respiration rate.

Table 1 gives carbon in biomass and fractional respiration rates for each habitat type.

Table 1: Model habitats with assumed initial biomass values and fractional respiration rates.

Habitat type	Habitat type description	Carbon in biomass (g.m^{-2})	Fractional respiration rate (l.hr^{-1})
1	Seagrass/algal	60	0.0005
2	Sparse seagrass	25	0.0005
3	Sparse algal/seagrass	20	0.001
4	Algal dominated	10	0.001
5	Open sediment	1	0.001

Figure 16 shows simulated time paths of dissolved oxygen for each habitat type. The time paths are numbered to correspond to habitat type and are overlaid for easy visual comparison.
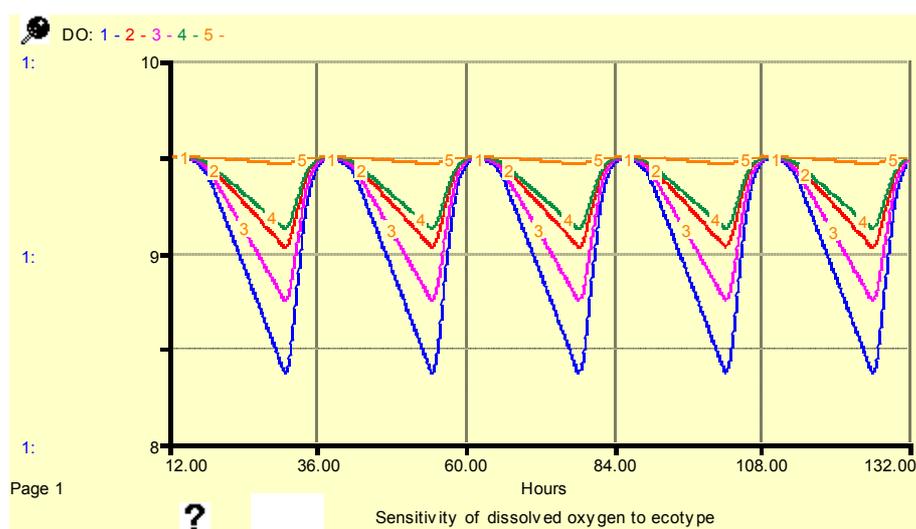

Figure 16: Sensitivity of dissolved oxygen concentration to habitat type. The numbers on each curve correspond to the habitat type being simulated within the model. (See Table 1 for key to habitat numbers).

The degree of diurnal dissolved oxygen fluctuation is a function of the respiration rate. The respiration rate is the product of carbon in biomass and the fractional respiration rate multiplied by an oxygen limitation factor. As expected, seagrass/algal habitat causes the most extreme fluctuation of dissolved oxygen due to high comparative biomass. Sparse algal/seagrass causes more extreme oscillation than sparse seagrass habitat, even though its biomass is lower this is due to a higher fractional respiration rate associated with an algal component. Algal dominated oxygen oscillations are less extreme than for the sparse seagrass habitat because of lower biomass. Open sediment with very low biomass causes very limited oxygen oscillation under normal water mixing regimes where oxygen supply from the overlying water is constant.

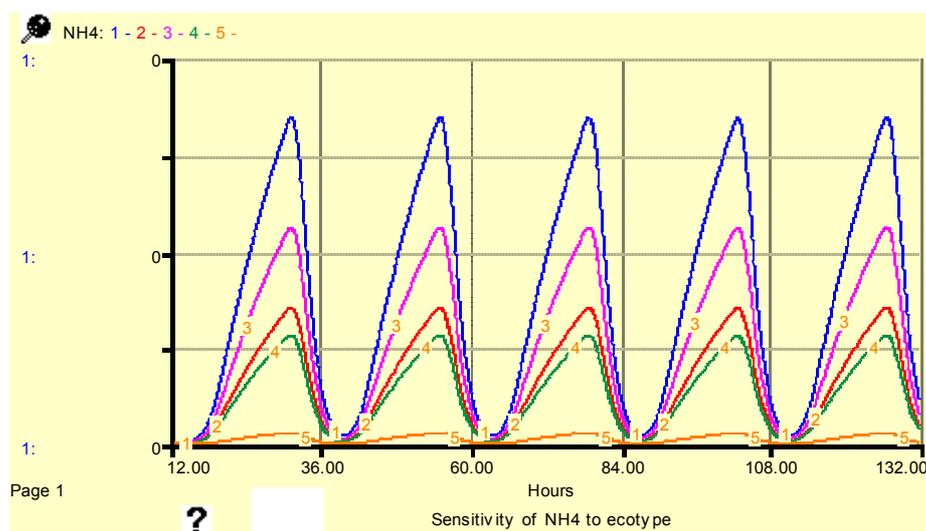Figures 17, 18, and 19 show simulation time paths of water column NH_4, NOx, and PO_4 for each habitat type.

Figure 17: Sensitivity of NH_4 concentration to habitat type. The numbers on each curve correspond to the habitat type being simulated within the model (See Table 1 for key to habitat numbers).
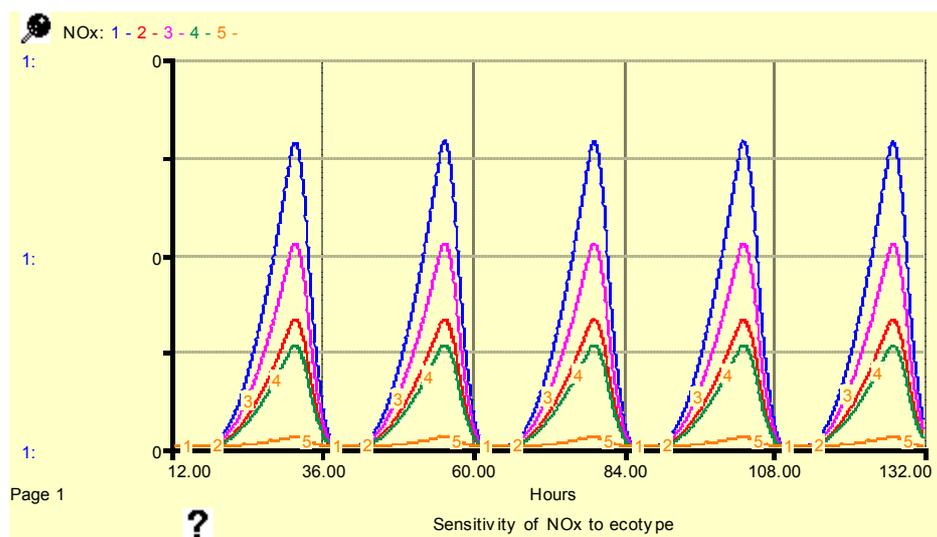
Figure 18: Sensitivity of NOx to habitat type in the simulation model. The numbers on each curve correspond to the habitat type being simulated within the model. (See Table 1 for key to habitat numbers).

Figure 19: Sensitivity of PO$_4$ to habitat type in the simulation model. The numbers on each curve correspond to the habitat type being simulated within the model. (See Table 1 for key to habitat numbers).
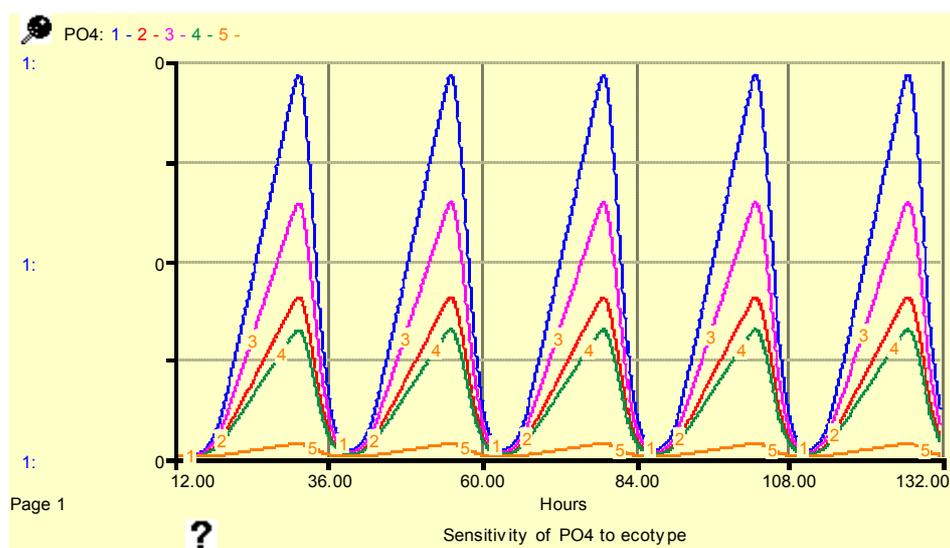
In this form of the model, each nutrient is released into the water column through excretion and there is no component to account for the currently unknown contribution of grazing and cell rupture due to physical damage. The excretion rate is assumed to equal the respiration rate, which is a common practice in benthic biomass modelling (EPA, 1985). The nutrients are taken up during carbon uptake based on fixed ratio of elemental nutrient to biomass carbon. As

expected the nutrient oscillations correspond to the oscillations of oxygen, i.e. seagrass/algal shows the greatest degree of oscillation, sparse algal/seagrass the second greatest, and so on. This is explained by the different levels of autotrophic contribution to the uptake of nutrients and to the oxygen budget in each habitat type. Oxygen concentrations in the sediment porewaters is influenced by water column concentrations and, in turn, can influence the form and relative availability of nutrient species such as NOx and NH_4^+ (e.g. Wild *et al.*, 2004).

Model refinement and development

At the time of writing this report the model interface is designed to allow for the examination of nutrient and metabolic responses due to fixed conditions defined by the field data. This is an important and necessary basis from which other forms of the interface and potential refinements of the model can be made. In this initial form, the model is easily interacted with to examine the basic performance of the habitats within the range of current knowledge. From this point further discussion with potential end-users will provide the information necessary to refine the interface and the elements that end-users desire to manipulate to test their key notions and management intentions. This dialogue is ongoing and has included representatives from the Queensland EPA as well as from the SEQ Healthy Waterways Partnership, a consortium of local coastal governments in south eastern Queensland. Many of these bodies are directly responsible for managing water quality and its impacts on benthic communities such as those encompassed by this project and the models.

It should also be noted that the model is also being used as part of the discussion associated with the nitrogen and phosphorous budgets in Moreton Bay. This discussion is seeking to improve our understanding of the two nutrient species and to identify the key knowledge gaps that need to be addressed if we are to make this understanding sufficiently robust and suitable for management decision support. This process has been used to improve some of the models sub-components and the interface so that it better reflects the needs of users from different backgrounds and levels of systems understanding.

At completion of this project, a new project being conducted by Johnstone has provided the means to collect further benthic flux data and to expand the spatial coverage of the flux estimation work across a wider variety of habitat types throughout Moreton Bay. In this light, it is intended that the model and the process components will be continually refined into the future so that their predictive and decision support capacity is improved.

Summary and conclusions

As noted in the introduction, current efforts to manage anthropogenic nutrient loads to coastal waters continue to be hampered by a number of key aspects. Firstly, there continues to be a paucity of field based data on ecological processes and their behaviour or response under different circumstances of climate, season, and human influence. Secondly, there is still the tendency to use few or single process measurements (field– or laboratory–based) to estimate the function of whole ecosystems at various scales of time and space, and this often clumps habitats (or gradients of habitat type) into one functional unit. Exacerbated by the first two aspects, the third element that often hinders current decision–making in coastal resources management is a suitably integrated framework or knowledge base that allows both scientists and managers to interact over testing key notions or perspectives that they may have about ecosystem function or the behaviour of different ecosystem components.

The project reported here has aimed to address these considerations with a focus on the questions of variation within and between specific 'habitat' types, the implications this variability may have for the function of these habitats from a nutrient remineralisation perspective, and how this may alter the manner in which we consider these habitats in terms of nutrient load management.

The benthic flux measurements highlighted both the temporal and spatial variability in the net exchange of nutrients that occurs between the benthos and the overlying water. Much of the spatial variability is likely due to variations in the geochemistry of the underlying sediments as well as the infauna that exist within them (e.g. Krantzberg G, 1985; Kristensen E, 2000). Importantly, however, there was a tendency toward the decreased release of DIN from sediments in winter where the sediment had a significant primary producer component. In summer, the efflux of DIN was more widespread among sites and a consistent release of DIN was observed across most of the study areas. This seasonal variability may result from differences in the geochemical and Redox conditions within sediments underlying seagrass and algal communities. Under warmer conditions in summer, oxygen solubility is likely to be lower and thus sediments are likely to be more reducing due to the heightened community metabolism and relatively lower availability of free dissolved oxygen (Boudreau and Jorgensen, 2001). This in turn would likely lead to larger pools of dissolved nitrogen that may exceed uptake rates by sediment bound organisms and plants. This relationship between Redox potential, temperature and nutrient pools has been reviewed elsewhere and noted in different marine sediments (Thamdrup *et al.*, 1998). By comparison to the DIN fluxes, no exchange of soluble reactive phosphorous was observed in summer. One site showed a net uptake in winter.

Also, of note in the summer benthic flux work is the general agreement between results for DIN in the three habitat types of seagrass-algae, sparse seagrass, and sparse seagrass-algae. This implies that the seagrass component may be very significant in influencing nutrient exchanges but also suggests that in summer, these three habitat definitions may exist based on composition but not in terms of nitrogen exchange rates. In winter this situation did not hold and the three habitat types behaved with great variability so that any statistical similarity could not be defined. Further field work involving larger replicate numbers may help resolve this aspect further. Notably, the open sediment areas showed some seasonal variation, but behaved similarly in summer and winter with stronger fluxes observed in summer.

As indicated above, the use of habitat classification systems based on their plant community composition does not always accurately represent their performance in terms of nutrient dynamics and this may vary between seasons. In this regard, it s critical that any models or management approaches that use a community composition-based classification system to describe their ecosystem must undertake process studies before ascribing a remineralisation or nutrient dynamic role. From the results in this study, it is clear that not all seagrass or seagrass-algal assemblages behave the same and so pooling of these may have negative impacts on estimates of their role within the wide ecosystem.

With regard to the modelling aspect of the project, the modelling process proved very useful in identifying both the limits of current knowledge on Moreton Bay benthic nutrient dynamics as well as in helping to capture the key elements that warrant further research in order to adequately understand the influences that anthropogenic nutrient loads may have on the benthos. For example, by simulating the different habitats and their function with the model it is clear that, depending on where a nutrient load may be deposited (open sediment or seagrass area), the load may enter an environment where there is already a net release of materials (e.g. open sediment in summer) or an environment where there is likely to be a net uptake of dissolved nutrients. In the later case the potential for nutrient removal from the water column s higher and so outcomes such as phytoplankton blooms may be lower within the immediate timeframe of load deposition. Conversely, the load may reinforce the release of nutrients in the immediate term and augment phytoplankton growth.

Similarly, in the composite model, the user is able to test notions about how different relative ratios of the benthic plant community might influence nutrient dynamics. This provides insight into why it is necessary to not focus only on particular species or assumed assemblages but to also consider that there are a diverse range of community compositions that may play a varying role in nutrient

removal and control. On this later aspect the models have been used to elicit responses and to augment a discussion between scientists, engineers and managers as part of a wider discussion on algal blooms in Moreton Bay (Healthy Waterways Partnership. March 2006). This set of discussions validated the modelling approach and indicated that such integrated framework and model approaches are very useful in both the communication of ideas as well as in the reaching of consensus on what the key issues may be from a technical and managerial perspective.

At completion of this project the models provide a solid foundation for future development of more sophisticated and elaborate frameworks that can fully test the contribution that benthic habitats may have at a whole of ecosystem scale. As noted, however, issues such as the variability within and between habitat types, the apparent role of seasonality and environmental conditions, and the lack of continuous agreement between community based classification and habitat remineralisation function all require further work. This is critical in order to more fully resolve these aspects and consolidate our understanding of how these issues may alter the capacity of the benthos to process a given load.
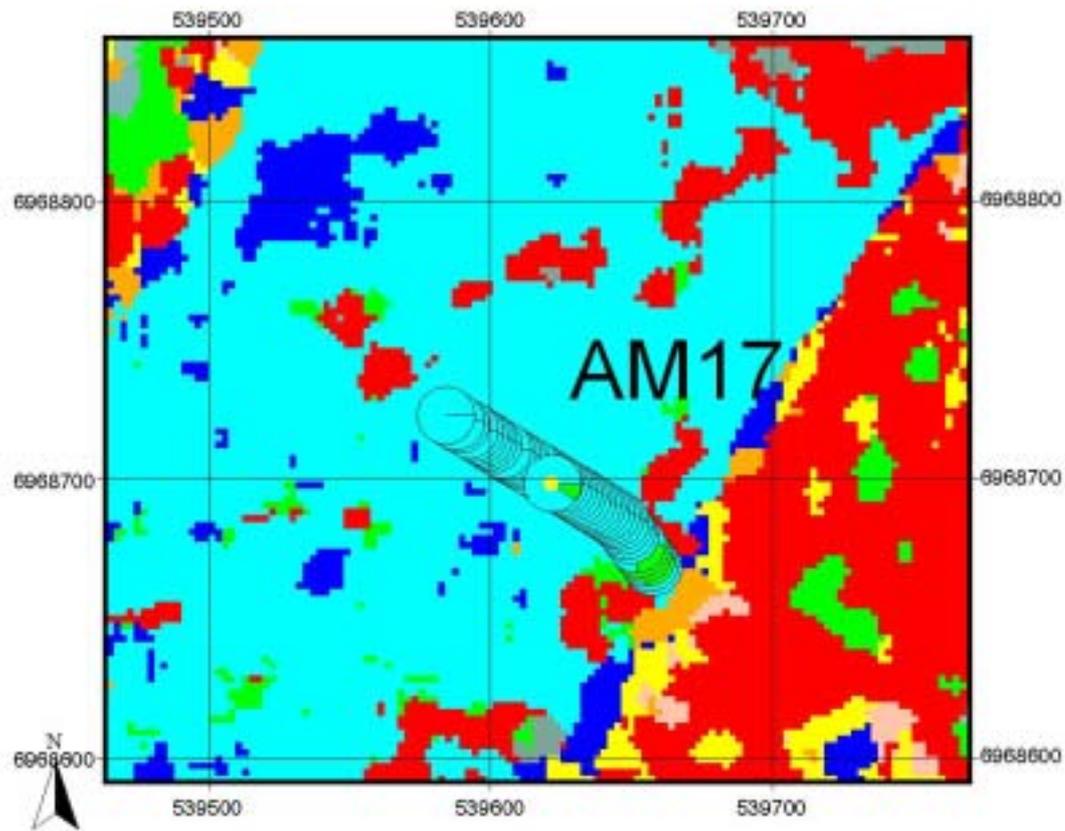
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APPENDIX 1
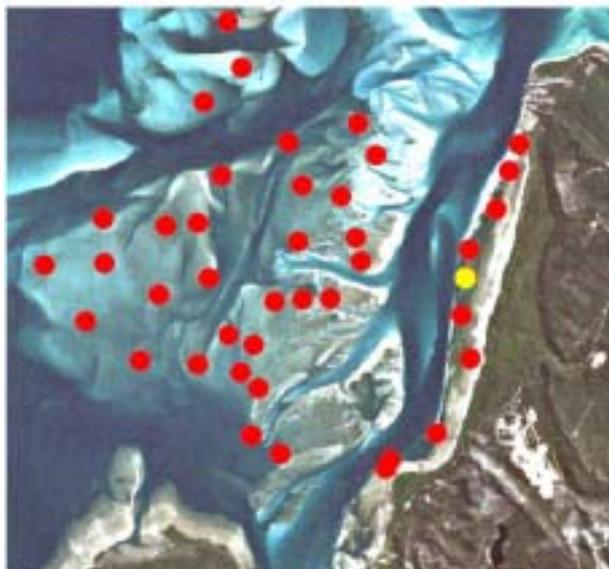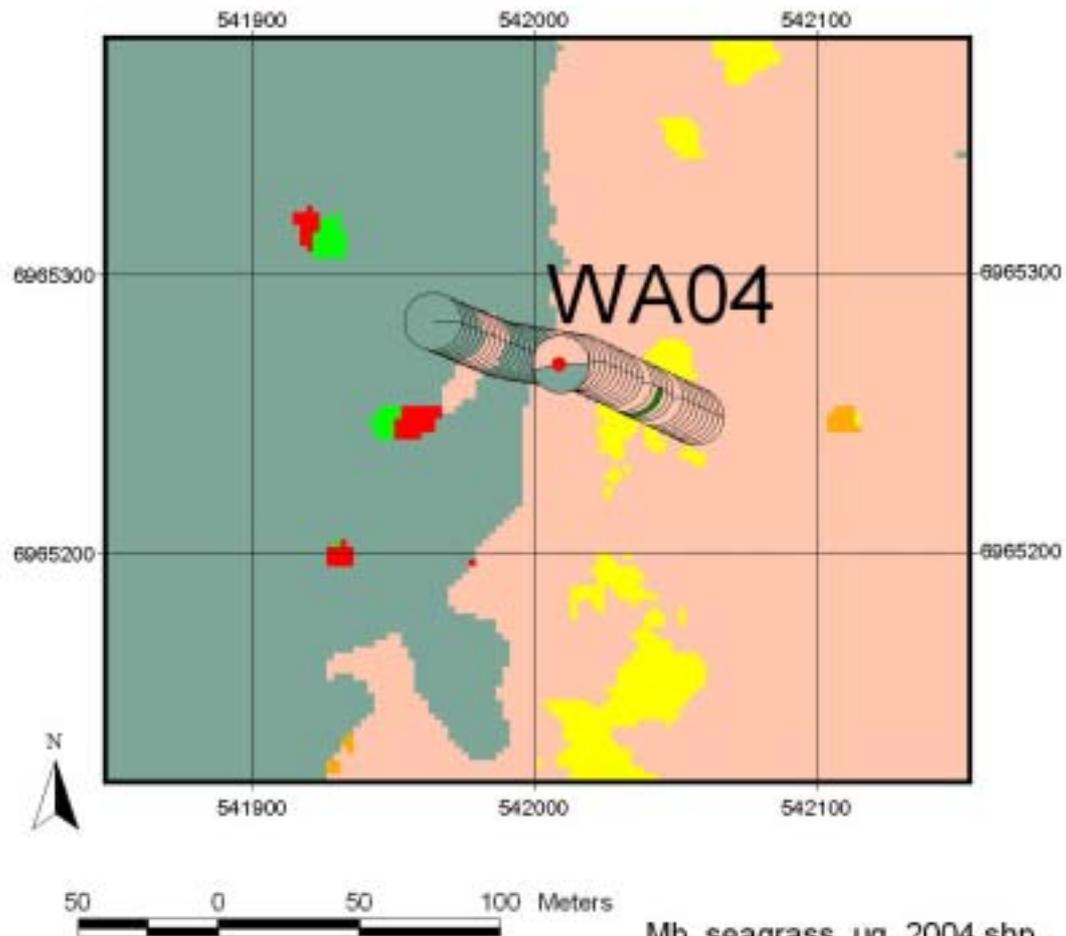
Remote sensing derived species maps for Eastern Banks
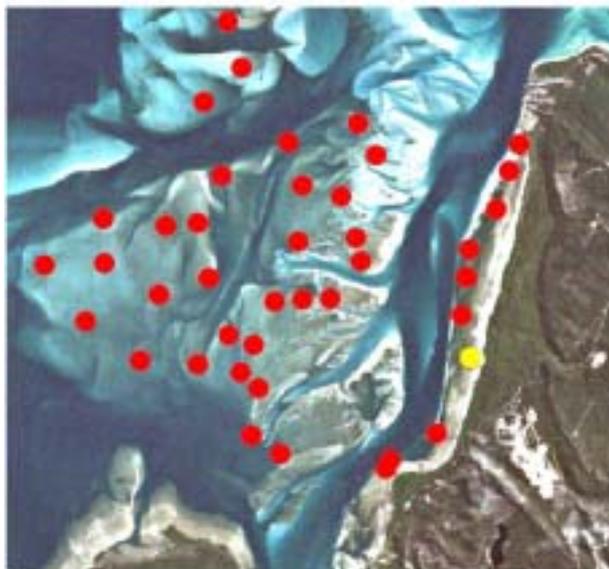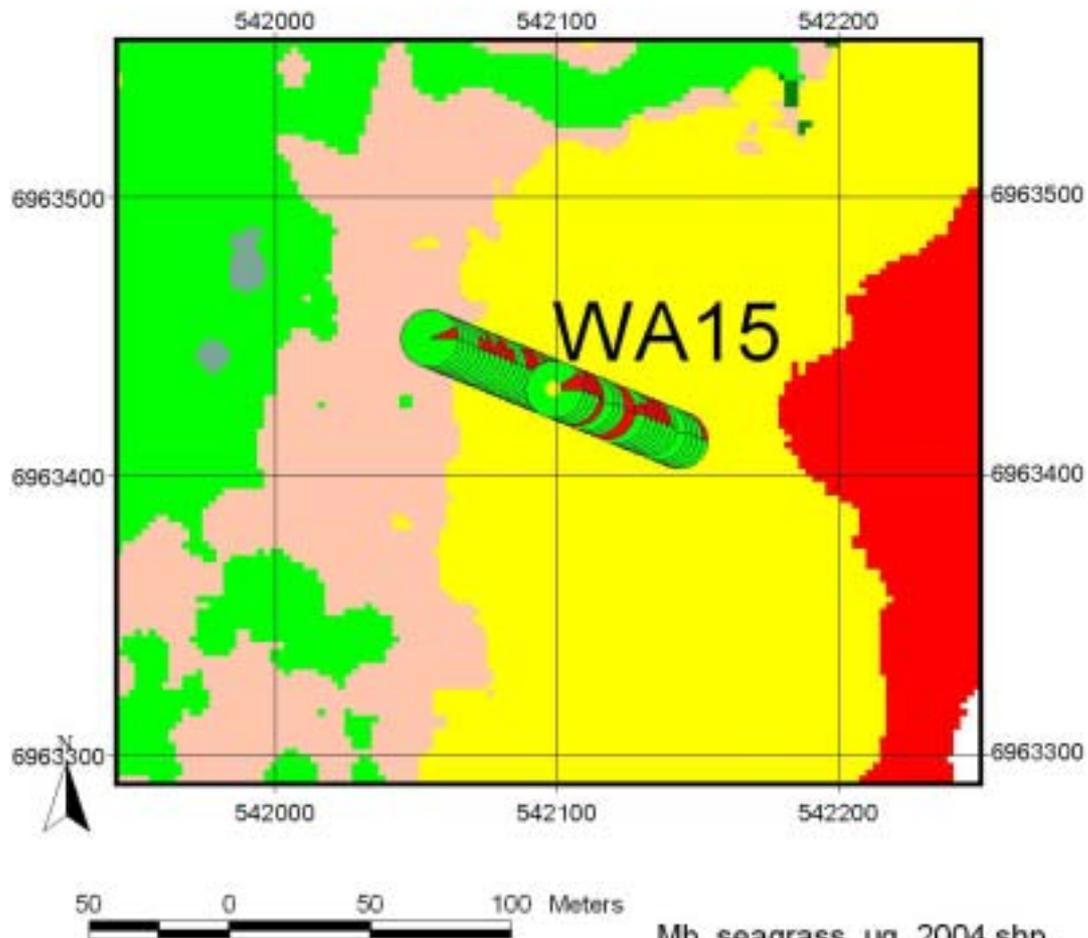
(Phinn and Dekker, 2005)