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# Characterising benthic pelagic interactions in Macquarie Harbour - organic matter processing in sediments and the importance for nutrient dynamics

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# Executive Summary

**What the report is about**

The strategic growth of the Tasmanian Salmonid Industry over the next decade is contingent upon ecologically sustainable development in Macquarie Harbour. In coastal bays and estuaries, it is well known that sediment - water column interactions are a major driver of ecosystem condition and health. A key knowledge gap in Macquarie Harbour was a lack of ecological data on the capacity of sediments to process organic matter and nutrients, and the influence of these on bottom waters, particularly given the expectation of increased localised organic loads associated with expanded farming. This report describes the work conducted to address this knowledge gap. The first part of the study assessed the impacts of organic enrichment from salmon farming on nitrogen cycling processes in the sediments and included an assessment of the sediment's capacity to recover during fallowing. The information was achieved by conducting a series of incubations, at farm and harbour scales and repeated across seasons and farm production cycles, in which sediment - water column fluxes were measured. This component of the study was carried out by the Institute of Marine and Antarctic Studies (University of Tasmania) with technical support from Aquenal Pty Ltd. The results from the field incubations were used to re-calibrate and validate the sediment - water column interaction terms in the environmental model used in the Macquarie Harbour EIS. The modelling and subsequent interpretation of model outputs was carried out by DHI Water and Environment (Danish Hydrological Institute, DHI.

This report describes the approach and results of the field study as well as the subsequent modelling, and discusses the outcomes in the context of ecological sustainable development of the salmonid industry in Macquarie Harbour.

**Background**

This project has arisen from discussions with industry and state government regarding the scientific understanding necessary to support ecologically sustainable development of fish farming operations in Macquarie Harbour. The Adaptive Management process proposed for Macquarie Harbour seeks to ensure sustainable development and management of fish farming in the region. Critical to this is the implementation of a monitoring program focused on key environmental interactions within the harbour and a whole of harbour environmental predictive model. The modelling and monitoring carried out by DHI as part of the Macquarie Harbour EIS identified that current levels of understanding of the seabed interactions with bottom waters in this system could be improved. As a result this study was commissioned to provide that critical process understanding and to assist DHI with the re-calibration and validation of their environmental model of Macquarie Harbour.

**Aims/objectives**

The aims of this project were to:

1. Quantify sediment - water column nutrient fluxes at both the farm (local) and harbour (regional) scales
2. Generate sediment nutrient and dissolved oxygen respiration maps of Macquarie Harbour, including the release of nutrients from deposited farm waste
3. Calibration of sediment - water column interactions in the Macquarie Harbour environmental model using process information from 1 and 2 above
4. Identify ecologically relevant and practical indicators of key ecosystem processes.

**Methodology**

The project was carried out in two separate but complementary components. The first component, carried out by IMAS and Aquenal Pty Ltd, measured sediment - water column nutrient fluxes at sites on and off-farm within the Harbour. This was achieved by collecting sediment cores and measuring changes in nutrient concentrations over a fixed incubation period. The sediments were also processed for a range of key sediment variables (i.e. particle size, C/N ratio and isotopic composition ($\delta^{13}C$ and $\delta^{15}N$). Cores were collected across 2 spatial scales; farm (underneath, 50 and 1000m from cages) and harbour scales (lower,

middle and upper region), this enabled an understanding of the key environmental drivers of sediment nutrient cycling on and off-farm. The full survey was conducted in November 2012 with a sub set of sites revisited in January, May and September 2013.

The second component of the study was undertaken by DHI. The data on nutrient fluxes was used to generate sediment nutrient and DO respiration maps and to calibrate the environmental model of Macquarie Harbour.

**Results/key findings**

*Environmental conditions*

As expected, water column profiles demonstrated the highly stratified nature of Macquarie Harbour; showing the system to be characterised by low salinities and high dissolved oxygen conditions in the surface waters compared with higher salinities and low dissolved oxygen conditions in the bottom waters[1]. Surface salinity declined through the winter (May 2013) and spring (September 2013) surveys, consistent with the large step change increase in Gordon river flow that occurred in early 2013 due to increased power generation. Most notably, despite reasonably stable bottom water temperatures and salinities across the four surveys, dissolved oxygen of bottom waters declined significantly in winter and spring 2013, reaching 0.2-0.9 mg/L in the spring survey.

Although the sediments were generally similar across sites in terms of their grain size composition, the elemental (C/N ratio) and isotopic ($\delta^{13}C$ and $\delta^{15}N$) composition proved to be a good indicator of the source of sedimentary organic matter in Macquarie Harbour. The organic matter signature of sediments within the Harbour generally reflected a terrestrial source (high C:N and depleted $\delta^{13}C$), which allows for the signature of fish farm derived organic matter (lower C:N and more enriched $\delta^{13}C$ and $\delta^{15}N$ signatures)to be clearly differentiated.

*Benthic Nutrient Fluxes*

As might be expected, based on the relative levels of organic matter input, rates of organic matter mineralisation were significantly lower at harbour and farm control sites compared to cage sites[2]. Comparing the ratio of oxygen consumed with dissolved inorganic carbon produced during respiration highlighted the increasing role of anaerobic respiration at sites enriched by farm inputs. However, the fate of the reduced compounds produced during anaerobic respiration is difficult to determine based on benthic flux incubations and warrants further research (e.g. are they reoxidised in bottom waters creating a further oxygen deficit?)

Patterns of ammonium production largely reflected the patterns of respiration with higher rates at farmed sites compared with harbour and farm control sites. Nitrate fluxes were predominantly directed into the sediments at all sites, which is consistent with conditions in low oxygen environments where nitrification (the process by which ammonia is converted to nitrate in oxic conditions) is limited and denitrification (the process by which nitrate is converted to nitrogen gas in anoxic conditions) must rely on sourcing nitrate from the water column rather than from nitrate produced in the sediments via nitrification. This response is also consistent with expectations given oxygen consumption across sites; with the largest uptake of nitrate being at those farmed sites where oxygen consumption is highest, and thus, oxygen availability the lowest. Rates of denitrification (the process that permanently removes nitrogen from the

[1] bottom water refers to measurements taken 1-2m above the sediment

[2] when putting farm measurements in the context of harbour wide measurements, it is important to recognise that the total cage area (the area encompassing the cages plus a 100m buffer outside the cages) across the harbour represents ~0.25% of the total benthic area in Macquarie Harbour or ~ 2.5 % of the total benthic area below 15 m depth

system) also reflected patterns of organic enrichment, with higher rates at farmed sites compared with harbour and farm control sites. However, in the context of the total nitrogen budget, although rates of denitrification are higher at farm sites, the percentage of nitrogen removed via denitrification is in fact lower in sediments at farm sites compared to the broader harbour or farm control sites. This is due to the very high efflux of ammonium back into the water column at the farm sites.

Comparing the response from farm sites through time and in relation to different stages of the farming cycle showed that some, but not all sites, performed in a manner consistent with what might be expected during fallowing and stocking. The fact that not all sites performed in a manner consistent with what might be expected during fallowing and stocking suggests that other factors (e.g. changes in diet, conversion ratios, feeding regimes, bottom water conditions) can play a significant role in determining sediment condition.

*Modelling*

The ecological modelling recalibration was conducted by DHI, and as part of that process empirical data collected via this study was incorporated into the ecological template in the form of respiration and nutrient flux maps created directly from the collected benthic flux data. Together with other improvements to the model, the recalibrated model generally provided a good fit with the 12 months of water column data collected from Macquarie Harbour between October 2011 and September 2012.

It is important to acknowledge that the modelling is based on a set of assumptions regarding processes that define the biogeochemical interactions in the harbour and that these assumptions provided a reasonable fit to the October 2011-Sepember 2012 observations. However, it is possible that the processes dominating the ecology and hydrodynamics of the harbour may change based on external factors outside the realm of current knowledge. With this in mind it is clear that continued monitoring comprises an essential component of a sound adaptive management strategy, and that any shift in conditions or change in system understanding observed through such monitoring should require the model to be revised for future predictions. The very low bottom water dissolved oxygen conditions recorded in the final two surveys (May and September 2013) may indicate such a shift in conditions[3].

**Implications for relevant stakeholders**

*Industry Managers* – this study provides an improved understanding of both sediment function and the benthic response to farming in Macquarie Harbour. Prior to this study, there had been no measurement of benthic nutrient cycling for this system. The inclusion of local benthic flux data rather than literature values will improve the reliability of the environmental model and information used to assist in decision making in the harbour.

The study has also reinforced the importance of farm based management as a means to ensure that sediments recover after/ between farming cycles and can function most efficiently when subject to significant enrichment. However, the results suggest that the ecology and recovery dynamics in Macquarie Harbour may be different to other farming regions and further work is required to fully understand the drivers and management indicators of sediment function and recovery for farm affected sediments in the harbour.

*Government Regulators/ Managers*– this study has generated an improved understanding of sediment function in Macquarie Harbour and has enabled sediment function to be more appropriately represented

[3] Note, the model recalibration was based on the flux data collected during the first 2 surveys (Nov 2012 and Jan 2013) and water column observations from Oct 2011 to Sep 2012; the additional two surveys were a modification of the original project objectives to look specifically at sediment performance at different stages of farming activity.

in the environmental model currently used to assist decision making. It is clear that the Macquarie Harbour ecosystem and the associated biogeochemical processes are different from that previously described for other systems in Tasmania. Consequently, it is acknowledged that changes in environmental conditions and our understanding of system dynamics would require the model to be revised for future use. From a sediment monitoring perspective, bulk identifiers of organic matter source (C:N ratio and $\delta^{13}C$ and $\delta^{15}N$ signature) together with measured rates of respiration appear to be good environmental indicators of the footprint of farm derived organic matter and sediment function respectively.

**Recommendations/ Further Research**

Whist the work has improved our understanding of sediment function in Macquarie Harbour and led to improvements to the environmental model used to assist decision making in Macquarie Harbour, it has also identified a number of knowledge gaps that warrant further investigation as follows:

- The study highlighted the importance of anaerobic processes and the production of reduced compounds in benthic biogeochemistry of the harbour. If these reduced compounds are reoxidised in bottom waters the concomitant oxygen demand is not likely to be fully accounted for in benthic core incubations. The very low bottom water oxygen conditions in the final two surveys highlight the importance of understanding the major drivers of oxygen dynamics in bottom waters; the potential role of reduced compounds warrants investigation.
- Measurement of sediment function at some but not all sites showed patterns consistent with expectations during fallowing and stocking. This suggests that drivers other than stocking (e.g. changes in diet, conversion ratios, feeding regimes, bottom water conditions) are playing a significant role in determining sediment condition. A greater understanding of the drivers of sediment function in response to different stages of farming activity is likely to improve the effectiveness of farm based management of stocking and fallowing regimes.
- In the second half of the study a significant decline in bottom water oxygen conditions was evident. The causes of this decline and the implications for broader ecosystem dynamics warrants further investigation, and as such, may require the model to be revised for future use.

#### 1.1.1.1 Keywords

Salmon Aquaculture, *Salmo salar*, Macquarie Harbour, nutrients, sediment function, benthic processes, modelling, nitrogen, carbon, environmental management

# 1 Introduction

## 1.1 Background

Aquaculture is developing rapidly to meet the increasing global demand for animal protein, with an average global growth rate of 6.3% per year (FAO 2012). In Tasmania, the Salmonid industry plans to double production by 2030. To do so the industry must consider alternative production approaches and expansion into new and existing areas. Maintaining high environmental performance and ecological sustainability (a priority for both industry and its regulators) requires an understanding of how farming activities interact with the environment. Further development of salmon farming in Macquarie Harbour on Tasmania's west coast is central to the industries plans for growth over the next decade. A detailed and targeted environmental monitoring program and a whole of harbour environmental predictive model are at the core of the Adaptive Management process proposed to ensure sustainable development and management of fish farming in Macquarie Harbour. The modelling and monitoring carried out by DHI Water and Environment (DHI) as part of the Macquarie Harbour EIS identified that current levels of understanding of seabed interactions with bottom waters in this system could be improved.

In coastal bays and estuaries, sediments act as in important site for organic matter deposition and its subsequent mineralisation (Burdige 2006). Sediments can be a both a major source of nutrients and carbon to the overlying water column or/and a significant sink. Typically, a significant fraction of the organic material that settles to the seabed undergoes biologically mediated degradation and oxidation through a complex set of biogeochemical reactions (see below for more detail), with only a small fraction permanently buried. Thus, determining the role that Macquarie Harbour sediments play in regulating the transformation and fate of organic matter and nutrients and the response to additional organic matter inputs due to increased salmon farming is central to ensuring ecological sustainable development.

This study will provide the critical process understanding of sediment-water column interactions and the response to increased loads associated with expanded farming. As part of the work, DHI will use this data to re-calibrate the sediment water column interactions in the existing environmental model of Macquarie Harbour. This project is clearly aimed at addressing two of the priority areas within the FRDC Research, Development and Extension Plan 2010 - 2015 Environment Program, "Ecologically sustainable development" theme; i)"quantify the environmental carrying capacity of aquaculture operations" and ii)"develop and implement standardised environmental impact assessments and statements for the aquaculture sector". Importantly, the understanding gained from this study on sediment - water quality dynamics, the system response to additional organic matter inputs and the assimilation of this knowledge into ecosystem models will have direct application to system understanding and management of other temperate coastal ecosystems.

### 1.1.1 A brief background on organic matter processing and nitrogen cycling in coastal sediments

In sediments, most processes start when organic matter (OM) is mineralised. The primary source of organic matter in bays and estuaries is decaying plant materials (e.g. phytoplankton, seagrass) and animal faeces. In the case of sediments adjacent to salmon cages, the major source of OM is likely to be fish faeces and unconsumed feed . The mineralisation process of OM occurs through bacterial respiration and this process consumes oxygen ($O_2$) and produces carbon dioxide ($CO_2$). Initially ammonia is also produced, some of which is released back into the water column and this can fuel more phytoplankton growth (Figure 1-1). Where surface sediments are oxygenated (oxic) the ammonia is also converted to nitrate via the process of nitrification (Figure 1-1). The nitrate can then also be released back into the water column to fuel more phytoplankton growth, or under more anoxic

conditions, it can be converted to nitrogen gas via the process of denitrification (Figure 1-1). Nitrogen gas is then lost from the system, unavailable to fuel algal growth. There is also another completing process for nitrate, dissimilatory nitrate reduction to ammonia (DNRA). DNRA reduces nitrate to ammonia under anoxic conditions, thereby acting to recycle bioavailable nitrogen within the system rather than permanent removal via denitrification (Figure 1-1). The relative contribution of each of these processes in nitrogen cycling is critical because they determine how much nitrogen remains in the system as bioavailable nitrogen versus how much is lost from the system. In the absence of significant oceanic exchange denitrification is the critical natural process that can permanently remove nitrogen from the water column.

The oxygen concentration of bottom waters strongly influences the nitrogen transformation pathways described here. If concentrations of oxygen in bottom waters decline under excessive organic loading or limited water exchange, the sediments can become anoxic and the process that converts ammonia to nitrate (nitrification) is effectively shut down. Under these circumstances most of the ammonia is released back into the water column. Under these circumstances denitrification in the sediments must now rely on sourcing nitrate from the water column, because there is no longer nitrate produced in the sediments via nitrification. In anaerobic sediments, mineralisation may also take place through the processes of sulphate reduction and methanogenesis due to the lack of oxygen, which produce hydrogen sulphide and methane gases respectively.

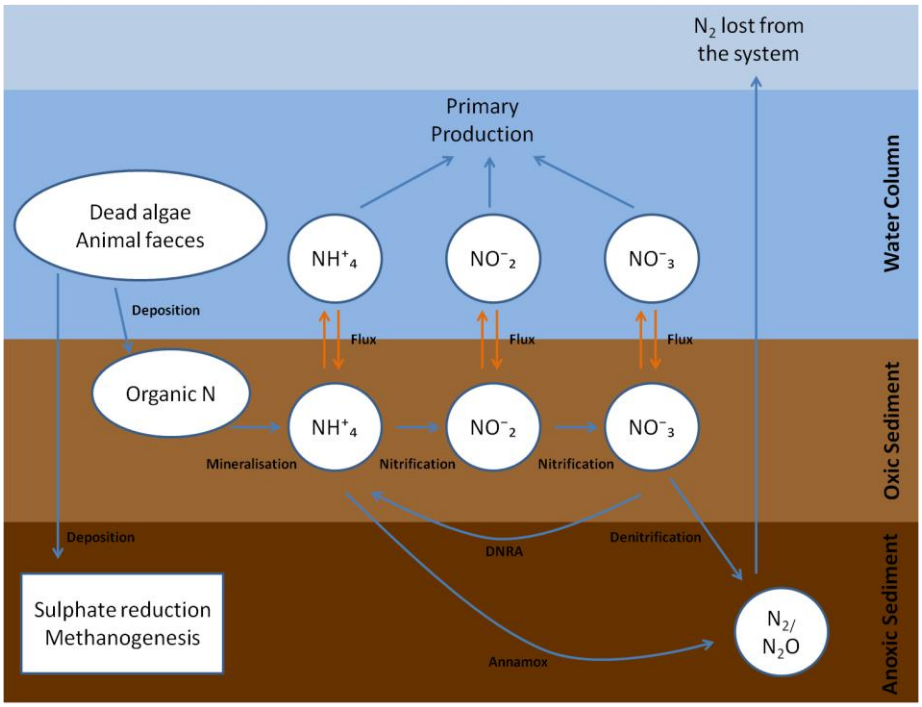


Figure 1-1 Simplified schematic and description of organic matter and nitrogen cycling in coastal sediments (see Burdige 2006 for a comprehensive text of the geochemistry of marine sediments)

## 1.2 Need

Strategic growth for the Tasmanian Salmonid Industry over the next decade is contingent upon ecologically sustainable development in Macquarie Harbour. A key knowledge gap in Macquarie Harbour is a lack of ecological data on the capacity of sediments to process organic matter and nutrients and the influence on bottom waters, particularly in response to increased loads associated with expanded farming. This was acknowledged in the EIS for industry expansion in Macquarie Harbour prepared by the Proponent (the three companies growing salmonids in Macquarie Harbour; Tassal, Huon Aquaculture and Petuna Aquaculture). The work proposed in this study feeds directly into the adaptive monitoring and modelling approach adopted to support decision making for marine

farming expansion in Macquarie Harbour, reducing the uncertainty in the environmental model, particularly with respect to bottom water predictions. This will be achieved via the collection of empirical process data and re-calibration of sediment water column interactions in the environmental model.

A limited understanding of sediment-water column processes is often lamented in other regions where ecosystem/biogeochemical model outputs are used to help guide environmental management decisions, and as such, this research is likely to have much broader R & D applications.

# 2 Objectives

Four objectives were defined for this project:

1. Quantify sediment - water column nutrient fluxes at both the farm (local) and harbour (regional) scales
2. Generate sediment nutrient and dissolved oxygen respiration maps for Macquarie Harbour, and identify the extent to which nutrients are released from sediment enriched with farm waste
3. Calibrate the sediment - water column interaction terms in the Macquarie Harbour environmental model using process information from 1 and 2 above
4. Identify ecologically relevant and practical indicators of key ecosystem processes.

Note, the field survey to meet objective 1 was to be repeated in November 2012 and January 2013 to correspond with different loads to the sediments through the production cycle. Following completion of the November 2012 survey and discussion with the project stakeholders, the number of sites in January 2013 survey was reduced and the project extended to re-survey a subset of sites in May and September 2013. This extension was designed to improve our understanding of sediment performance at different stages of farming activity, information that will assist farm management.

# 3 Methodology

## 3.1 Survey Design

To understand organic matter processing in Macquarie Harbour sediments and how it responds to increased organic loads associated with farming, sampling was conducted at both farm (local) and harbour (regional) scales. See Figure 3-1 for a map showing survey sites and Table 3-1 for site details. An important consideration in site selection was also to maximise overlap with previous and ongoing sediment and water column sampling programs in the harbour.

- At the farm scale this included 2 cage sites, a 50m from cage site and a control site 500 - 1000m from the farm in a similar depth. This was repeated at the Tassal Gordon (Zone 8; referred to as TAS 8) and Central Harbour (Zone 7; referred to as TAS 7) leases and Petuna Liberty Point lease (Zone 3; referred to as PET 3).
- At the harbour scale six sites were chosen including one site at each of the major boundaries (Hells gates - CHNa, King River-KR1 and Gordon River –WH1), 2 central harbour sites (CH, ECH), and another World Heritage Area site (WH2). Note, the 3 control sites for the farm scale assessment (TSC, CH2 and PET 3) also double as harbour wide sites.

To establish both harbour and farm scale sediment water column nutrient fluxes during periods that correspond with minimum and maximum loads to the sediments through the production cycle, the full suite of sites was surveyed in November 2012 and at a subset of sites in January 2013.This data was used for model calibration (objective 3). To improve our understanding of sediment performance at different stages of farming activity, the project was extended to re-survey a subset of sites in May and September 2013. See Table 3-2 for a complete list of sites and sample times and how the surveys meet the respective objectives.

## 3.2 Sample Collection

Sediment samples were collected using a box corer with a Perspex liner (surface area 0.0441 $m^2$; Figure 3-2), with 4 box cores collected at each site. For benthic flux analysis, 1 large (300 mm x 150 mm diameter) and 1 small core (300 mm x 44 mm) were taken from each box core to a sediment depth of ~ 100 mm (Figure 3-3). The large cores are used for the measurement of sediment fluxes of DO, dissolved inorganic carbon (DIC), $NH_4$, $NO_2$, $NO_3$ and $PO_4$ and the small cores are used for measuring rates of denitrification and DNRA using the isotope pairing technique (see descriptions of incubation techniques below). Cores were gently filled with bottom water collected from the site, capped and transferred to a bin filled with bottom water for transport back to the on - shore laboratory. To ensure that incubations conditions remained similar to bottom water conditions in the field, a bilge pump was used to collect 100 litres of bottom water (~ 1 m above the sediments). Water column profiles of salinity, temperature and dissolved oxygen were also taken at each site using a YSI 6600 V2 Multi Parameter Water Quality Sonde with YSI 650 MDS logger.

## 3.3 Sediment-water incubations

The cores were transferred to temperature control baths and allowed to equilibrate overnight in site water at *in situ* temperature and oxygen concentrations. All cores were stirred continuously throughout the incubation via a battery operated stirrer and suspended magnet (Figure 3-4); with the stirring rate set to ensure mixing of the water column but without agitating the sediment surface. Note, all incubations were undertaken in the dark given that little or no light reaches the sediment at the survey sites due to the depth and tannin stained water that is characteristic of Macquarie Harbour.

*Nutrient fluxes* - To start the incubation, the large cores are sealed and isolated. Samples are collected for dissolved nutrients ($NH_4$, $NO_2$, $NO_3$ and $PO_4$), DIC and alkalinity at 4 time intervals over the

course of the incubation. Samples for analysis of $NH_4$, $NO_2$, $NO_3$, $PO_4$ and alkalinity were filtered (0.45 μm; 30 mm polypropylene housing; Bonnet) and stored in 50-mL high density polyethylene sample bottles. Nutrient samples were frozen until analysis. Alkalinity samples were immediately preserved with 20 μl $HgCl_2$ and refrigerated until analysis. Samples for DIC were filtered (0.45 μm; 30 mm polypropylene housing; Bonnet) and preserved in a 12-mL Exetainer with 20 μl $HgCl_2$ and refrigerated until analysis. Dissolved oxygen and pH are measured at the same time intervals using a Hach HQ40d with DO and pH probes. The length of incubation is determined by the rate of oxygen depletion allowing for a total drop of no more than 10-20 % in oxygen saturation. This equated to incubations running for between 3 and 24 hrs. Sediment-water nutrient fluxes are calculated on the basis of the concentration change of analyte over time. The water volume and the surface area of the sediment are taken into consideration along with a correction for the addition or dilution of constituents by the replacement of site water when taking samples.

*Denitrification and DNRA* - Denitrification is determined using the isotope pairing technique with smaller cores (Nielsen 1992). Put most simply the nitrate that is converted to $N_2$ gas via the process of denitrification is measured by labelling the nitrate in the core using a different isotope than naturally occurs (i.e. $^{15}NO_3$ rather than $^{14}NO_3$). The isotope composition of the $N_2$ is then measured through the incubation providing an estimate of the rate of denitrification.

At the start of the incubation a 15 mL water sample is collected for $NO_3$ analysis and then 0.2 mL of labelled nitrate (0.05 mol $L^{-1}$ $^{15}N$-$NO_3$; 98%+, Cambridge Isotope Laboratories) is added to each core. The overlying water column is allowed to mix for ~1 min after the addition of the tracer, and a 15 mL sample is then taken to determine the initial concentration of $^{15}N$-$NO_3$ within the overlying water of the core. The sample water removed from the core is replaced with 15 mL of bottom water and the core sealed and isolated. The $^{15}NO_3$ then diffuses towards the denitrification zone and after a certain time the flux of $^{15}NO_3$ into the denitrification zone and the evolution rate of $^{15}N_2$ will be constant. The produced $^{15}N_2$ was then extracted as a time series. The times of sampling were similar to those of the nutrient flux incubations. Because collecting a labelled $N_2$ sample involves mixing the whole core only one of the four cores is sacrificed and sampled at each time interval. At the time of sampling, the DO is measured, a nutrient sample taken and then 1 mL of $ZnCl_2$ (50% w/v) is added to the core and the sediment and water is stirred using a 5 - 10 mm thick Perspex rod. The core was then allowed to settle before an $N_2$ sample was collected in a 12 mL Exetainer and preserved with 250 μL of $ZnCl_2$ 50% w:v until analysis.

The denitrification rate was calculated after the addition of $^{15}N$-$NO_3$ ($D_{15}$). The rate was determined from the linear relationship observed over time with respect to the excess $^{15}N$-labelled $N_2$ gas production (Nielsen 1992; Dalsgaard et al. 2000). The *in situ* denitrification rate produced from $^{14}N$-$NO_3$ ($D_{14}$) was calculated on the basis of the production of $^{14}N^{15}N$ and $^{15}N^{15}N$; from the $D_{14}$ rate the contribution of nitrification-driven denitrification ($D_n$) and water-column-driven denitrification ($D_w$) were calculated (Nielsen 1992; Dalsgaard et al. 2000).

The isotope pairing technique is based on the assumption that the amount of labelled nitrate added does not influence the denitrification rate ($D_{14}$; Risgaard – Peterson et al. 2004). This was tested using a concentration series experiment that was repeated on high and low organic enrichment sites. Sediment cores were collected and the denitrification incubation repeated at three different $^{15}N$-$NO_3$ concentrations.

To determine the amount of DNRA occurring within the core, a sample of $^{15}N$-$NH_4$ was collected at the end of the denitrification incubation. DNRA rates for $^{15}N$ were calculated from the production of $^{15}N$-$NH_4$ over the incubation period. See Roberts et al. (2012) for further detailed discussion on the assumptions and calculation of DNRA.

*Analytical methods* - All nutrient samples ($NH_4$, $NO_2$, $NO_3$ and $PO_4$) were analysed by the Water Studies Centre, Monash University using flow injection analysis (FIA) (Lachat Quichchem 8000 Flow injection Analyser, spectrophotometric detector). The analysis of nutrient samples via FIA followed the procedures in Standard Methods for Water and Wastewater (APHA 2005) including standard and quality-assurance checks. Alkalinity was analysed via gran titration using 100 μL aliquots of with 0.1

mol $L^{-1}$ HCL via a Metrohm Auto Titration System. Total carbon dioxide ($TCO_2$) was subsequently calculated from pH and alkalinity, the carbonate equations corrected for salinity and temperature (Millero 2006). DIC was analysed colourimetrically using a LI-7000 $CO_2/H_2O$ infrared gas analyser (LI-COR Biosciences, Lincoln, NE, USA) after acidifying the samples with 10% phosphoric acid.

## 3.4 Sediment properties

At the end of the nutrient flux incubations, subsamples for the measurement of sediment porosity, grain size, organic carbon and nitrogen content and their isotopic composition($\delta^{15}N$, $\delta^{13}C$) were collected from 2 of the cores using a 60 mL cut-off syringe down to ~20 mm depth. Porosity was determined from weight loss after drying the samples at 105$^{o}$C to a constant weight. For sediment grain size samples were dried in an oven at 60 °C and organics were removed from the sample using 10 % hydrogen peroxide. Sediment particle size was determined using a Saturn Digitiser 5200 laser diffractometer and is represented as the average percentage volume frequency for size fractions <0.063, >0.063 - <0.125, >0.125 - <0.25, >0.25 - <0.5, >0.5 - 1 and >1mm. Samples for carbon and nitrogen content and isotopic composition were ground and the sample for C analysis was acidified with a dilute HCl solution to dissolve solid carbonates. The samples were analysed at the Water Studies Centre (Monash University) on an ANCA GSL2 elemental analyser interfaced to a Hydra 20-22 continuous-flow isotope ratio mass-spectrometer (Sercon Ltd., UK). The precision of the elemental analysis was 0.5 µg for both C and N (n = 5). The precision of the stable isotope analysis was ±0.1‰ for $^{13}C$ and ±0.2‰ for $^{15}N$ (SD for n=5). Stable isotope data are expressed in the delta notation ($\delta^{13}C$ and $\delta^{15}N$), relative to the stable isotopic ratio of Vienna Pee Dee Belemnite standard ($R_{VPDB}$= 0.0111797) for C and atmospheric $N_2$ ($R_{Air}$ = 0.0036765) for nitrogen.

Table 3-1 Site coordinates and description.

| Site | Scale | POINT_X | POINT_Y | Depth (m) | Survey 1 (Nov) | Survey 2 (Jan) | Survey 3 (May) | Survey 4 (Sep) |
|---|---|---|---|---|---|---|---|---|
| WH2 | harbour | 370223.98 | 5309960.28 | 32 | harbour | | | |
| WH1 | harbour | 375844.82 | 5303913.08 | 7 | harbour | | | |
| KR1 | harbour | 361454.79 | 5325884.75 | 34 | harbour | | | |
| CHNa | harbour | 361224 | 5321325 | 14 | harbour | | | |
| ECH | harbour | 366930 | 5318194 | 18 | harbour | | | |
| CH1 | harbour | 366536.39 | 5314142 | 43 | harbour | | | |
| | | | | | | | | |
| TAS 8.1 (CAGE) | farm | 364254 | 5316675 | 37 | stocked | stocked | stocked | stocked |
| TAS 8.2 (CAGE) | farm | 364092 | 5316569 | 34 | fallowed | fallowed | fallowed | stocked |
| TAS 8.3 (50m from CAGE) | farm | 364064 | 5316610 | 34 | gradient | gradient | gradient | gradient |
| TSC | farm/harbour | 364678 | 5315867 | 35 | control | control | control | control |
| | | | | | | | | |
| TAS 7.1 (CAGE) | farm | 363113 | 5318398 | 37 | stocked | stocked | fallowed | fallowed |
| TAS 7.2 (CAGE) | farm | 363011 | 5318317 | 33 | stocked | stocked | fallowed | fallowed |
| TAS 7.3 (50m from CAGE) | farm | 362982 | 5318358 | 35 | gradient | gradient | gradient | gradient |
| CH2 | farm/harbour | 361896.32 | 5320351.61 | 36 | control | control | control | control |
| | | | | | | | | |
| PET 3.1 (CAGE) | farm | 361897 | 5315585 | 18 | stocked | fallowed | | |
| PET 3.2 (CAGE) | farm | 362007 | 5315209 | 19 | fallowed | stocked | | |
| PET 3.3 (50m from CAGE) | farm | 361896 | 5315635 | 19 | gradient | gradient | | |
| PET3 | farm/harbour | 362733.75 | 5313568.72 | 18 | control | | | |

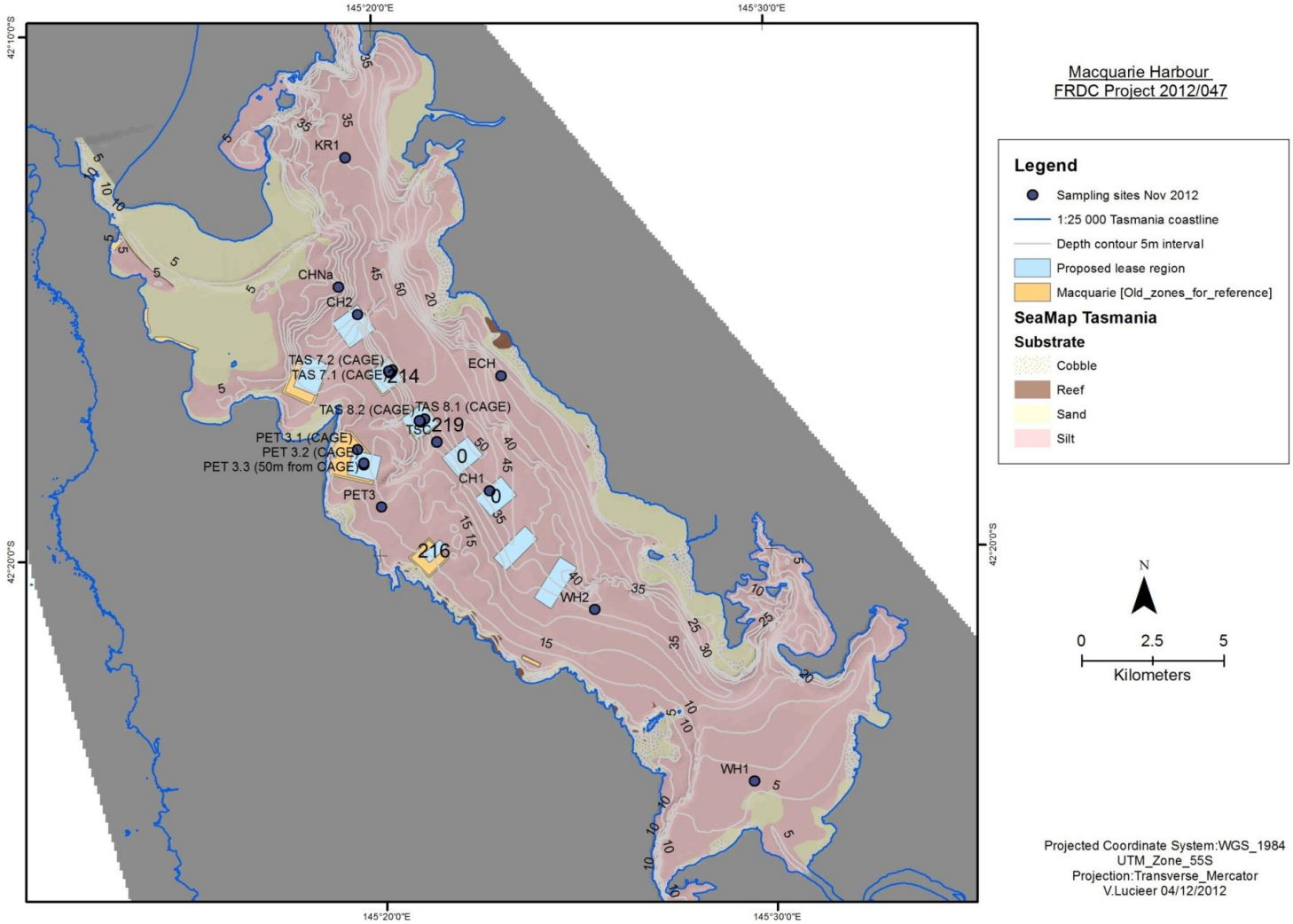


Figure 3-1 Map of Macquarie Harbour showing survey sites. Note the labelling at farm sites refers to the zone number (e.g. TAS 7, TAS 8 and PET 3), cage number (e.g., TAS 8.1,8.2; TAS 7.1,7.1;PET 3.1,3.2 refer to cages 1 and 2 at each farm) and 50m from cage site (e.g. TAS 8.3, TAS 7.3 and PET 3.3).

Table 3-2 Survey locations and times

| Site | Nov-12 | Jan-13 | May-13 | Aug-13 |
|---|---|---|---|---|
| WH2 | √ | | | |
| WH1 | √ | | | |
| KR1 | √ | | | |
| CHNa | √ | | | |
| ECH | √ | | | |
| CH1 | √ | | | |
| | | | | |
| TAS 8.1 (CAGE) | √ | √ | √ | √ |
| TAS 8.2 (CAGE) | √ | √ | √ | √ |
| TAS 8.3 (50m from CAGE) | √ | √ | √ | √ |
| TSC | √ | √ | √ | √ |
| | | | | |
| TAS 7.1 (CAGE) | √ | √ | √ | √ |
| TAS 7.2 (CAGE) | √ | √ | √ | √ |
| TAS 7.3 (50m from CAGE) | √ | √ | √ | √ |
| CH2 | √ | √ | √ | √ |
| | | | | |
| PET 3.1 (CAGE) | √ | √ | | |
| PET 3.2 (CAGE) | √ | √ | | |
| PET 3.3 (50m from CAGE) | √ | √ | | |
| PET3 | √ | √ | | |

Spatial/temporal snapshot / data for model calibration

Temporal response to farm management

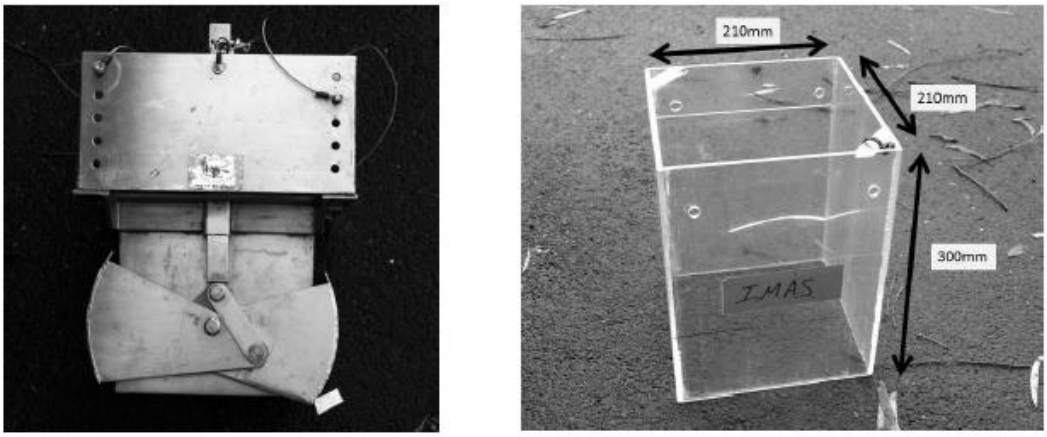


Figure 3-2 Box core (left) and Perspex liner used for sediment sampling.

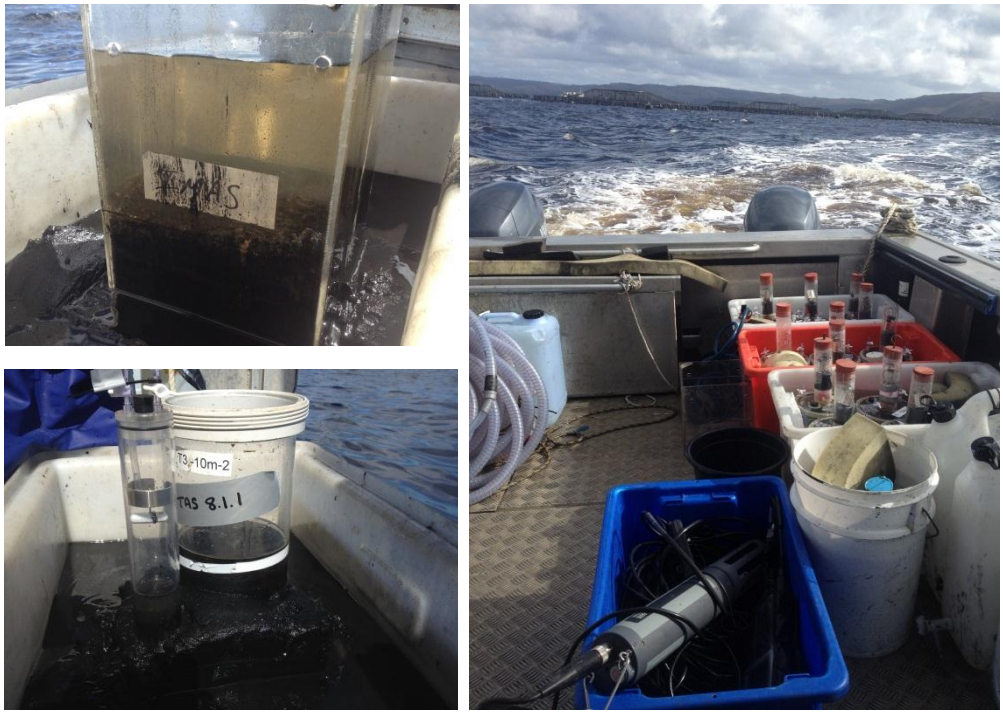


Figure 3-3 Pictures of Perspex liner after the box corer has been removed (top left), mesocosm core and mini-core (bottom left) and a full set of samples on return to the lab (right)

Figure 3-4 Pictures of lab set up (top left) - the large black tub acts a temperature control bath with circulating freshwater connected to the heater/chiller units, the clear Perspex aquaria hold the mesocosm and are filled with site bottom water (top right). The picture on the bottom left shows a close up of the sediment surface with faunal tubes and the picture on the bottom right shows the mesocosms with stirrer lids attached.

# 4 Results and Discussion

## 4.1 Variation in space

### 4.1.1 Environmental data

*Water Column* - Stratification of the water column based on temperature, salinity and dissolved oxygen was clearly evident at all of the sampling sites in the November survey (Figure 4-1, Table 4-1). Surface water[4] temperature ranged between 14.5°C and 16°C and the thermocline, or area of greatest temperature change, occurred between 3 and 10 m. Temperature then gradually increased by 1-2°C down to 20 m, below which it was relatively stable at 13.5-14.5 °C in bottom waters. Salinity in surface waters ranged between 5 and 10 ppt increasing to 30-31 ppt by 15 m. The halocline, or area of greatest change in salinity, occurred between 3 and 15 m. Dissolved oxygen (DO) concentrations in surface waters were > 8 mg/L before gradually decreasing < 5.5 mg/L in bottom waters. Not surprisingly, bottom water DO concentrations were highest at the two shallowest sites WH1 (7m) and CHN (14m). At the other sites bottom water DO was <2.5 mg/L, regularly <2 mg/L and at the Petuna sites less than <1 mg/L. Whilst stratification of the water column and low bottom water DO concentrations is a natural feature of Macquarie Harbour, the concentration recorded in this survey appear lower than those typically recorded in previous surveys (e.g. Koehnken 1996; Creswell et al., 1989; Carpenter et al., 1991). For example, during surveys as part of the Macquarie Harbour King River Study in the early 1990s, Koehnken (1996) reports an oxygen minimum of ~ 2.4 mg/L, very rare observations below this and no observations <1.6 mg/L.

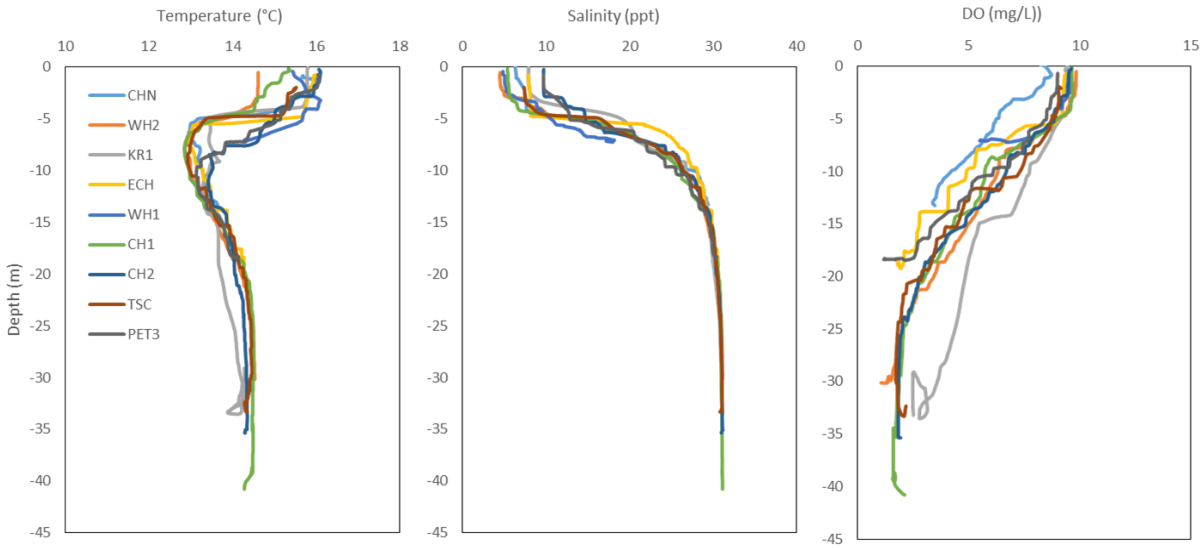


Figure 4-1 Water column profiles at all sampling sites in Macquarie Harbour taken in November 2012

[4] Surface water refers to measurements taken 0.5-1m below the surface and bottom water refers to measurements taken 1-2m above the sediment

*Sediments* – Sediments were predominantly silt (<62 µm) with a median grain ranging from 6.85 to 18.63 µm, with the exception of WH1 (44.4 µm) (Table 4-2). The slightly larger sediment grain size at WH1 is most likely due to its shallowness and proximity to the Gordon River. Habitat mapping of the harbour determined that 77% of the harbour is dominatedby silt habitat (Lucieer et al., 2009). The average organic carbon and nitrogen content per dry weight of sediment were greater than 5.5% C and 0.25% N, with the exception of CHN (1.29% C and 0.09% N). This most likely reflects the proximity of CHN to Hells gates and the greater ocean influence of marine sources of organic matter.

Table 4-1 Water column parameters measured at sampling sites in Macquarie Harbour in November 2012. No surface water (<1 m depth) data were available for TAS7.1 and TAS7.3.

| Parameter | Depth (m) | Temperature (°C) | | Salinity (ppt) | | DO (mg/L) | |
|---|---|---|---|---|---|---|---|
| | | Surface water | Bottom water | Surface water | Bottom water | Surface water | Bottom water |
| CHN | 13 | 15.7 | 13.6 | 6.3 | 29.4 | 8.2 | 3.5 |
| KR1 | 33 | 15.8 | 13.9 | 7.9 | 31.1 | 9.4 | 2.5 |
| WH1 | 7 | 15.4 | 14.3 | 4.9 | 18.2 | 9.6 | 5.5 |
| WH2 | 30 | 14.6 | 14.5 | 5.0 | 31.1 | 9.9 | 1.1 |
| ECS | 18 | 16.0 | 14.3 | 7.8 | 30.6 | 9.4 | 1.8 |
| CH1 | 40 | 15.3 | 14.3 | 5.4 | 31.1 | 9.6 | 2.1 |
| CH2 | 35 | 16.1 | 14.3 | 9.7 | 31.0 | 9.6 | 1.9 |
| TSC | 32 | 15.5 | 14.3 | 7.4 | 31.0 | 9.2 | 2.2 |
| PET3 | 18 | 16.0 | 14.1 | 9.7 | 30.4 | 9.0 | 1.2 |
| TAS7.1 | 36 | n.d | 14.3 | n.d | 31.2 | n.d | 1.7 |
| TAS7.2 | 26 | 15.4 | 14.4 | 11.3 | 31.0 | 9.2 | 1.1 |
| TAS7.3 | 26 | n.d | 14.3 | n.d | 31.1 | n.d | 1.3 |
| TAS8.1 | 36 | 15.1 | 14.3 | 8.0 | 31.0 | 9.9 | 1.6 |
| TAS8.2 | 36 | 15.5 | 14.4 | 8.3 | 31.1 | 9.5 | 1.3 |
| TAS8.3 | 35 | 16.2 | 14.4 | 8.6 | 31.0 | 9.4 | 1.3 |
| PET3.1 | 18 | 15.3 | 14.4 | 9.6 | 30.8 | 8.8 | 0.9 |
| PET3.2 | 20 | 15.9 | 14.3 | 8.9 | 30.6 | 9.0 | 0.8 |
| PET3.3 | 18 | 15.3 | 14.4 | 9.9 | 30.7 | 8.7 | 0.8 |

Similarly, the C:N molar ratio was lower at CHN (15.92) compared to the other harbour wide sites (> 20). Because terrestrial OM has a high C:N ratio (>20) compared to marine phytoplankton ( ~6.6), the results suggests that the sediment OM throughout the harbour is predominately terrestrial in nature; the lower ratio at CHN is consistent with a greater contribution of marine OM closer to Hells Gates. The depleted $\delta^{13}C$ of sediments at the harbour wise sites is also consistent with a predominately terrestrial source of OM.

At fish farm sites, the C:N ratio and isotopic signature of sediment OM clearly identified the influence of fish farm waste (feed and faeces; Table 4-2, Figure 4-2, Figure 4-3) . The nitrogen content of sediments adjacent to cages was significantly higher than elsewhere and the C:N molar ratio <13 compared to > 16 at the harbour wide sites. Nickell et al. (2003) in a study of salmon cage effects on benthic sediments in Loch Creran, Scotland also reported a lower C/N molar ratio in sediments adjacent to cages compared to reference stations (8 vs. 10-12‰) Although the C:N molar ratio of fish and faeces will vary depending on feed quality and a range of other factors, typical C:N ratios measured for feed and faeces range from 8-12 (Crawford et al., 2003; Chen et al., 2003; Wang

et al., 2013). C:N ratios have also been used as a proxy for carbon quality with lower ratios indicating increased organic matter lability; this is consistent with the elevated nutrient fluxes at the cage sites described below. Figure 4-3 also highlights the distinct isotopic signature of the cage sites, characterised by enriched $\delta^{13}C$ and $\delta^{15}N$ which we might expect from fish feed and faeces. The enriched $\delta^{13}C$ reflects carbon of a more marine source; terrestrial organic matter (25 to -33‰) and freshwater phytoplankton (-25 to -30‰) typically have a more depleted $\delta^{13}C$ signature compared with marine particulate organic matter (-22 to -18‰) (see Middelburg and Nieuwenhuize, 1998). Similarly, marine organic matter usually has $\delta^{15}N$ of 5 - 7‰ as derived from phytoplankton whereas terrestrial organic matter has $\delta^{15}N$ values $< 4$‰ (see Middelburg and Nieuwenhuize, 1998). The enriched $\delta^{15}N$ of organic matter associated with the salmon cages may also reflect the composition of the feed. $\delta^{15}N$ becomes more enriched at higher trophic levels, and as such, fish meal that contains small pelagic fish will have a more enriched signature.

The C:N ratio and isotopic signature of sediment OM at the 50 m from cage sites was similar to the control site at the Petuna lease, intermediate between the control and cage site at the Tassal Gordon lease and similar to the cage sites at the Tassal Central lease. This most likely reflects the relative influence of farm waste across leases at the 50 m site in November 2012. This is also consistent with the comparison of benthic fluxes (described below); the 50 m site at the Tassal Central lease had the highest flux relative to the control site, followed by Tassal Gordon lease. At the Petuna lease the flux at the 50 m and control site was similar.

Table 4-2 Sediment characteristics measurements from the top 0 to 3 cm of sediment from all sites

| Parameter | Porosity | Median Grain Size (μm) | C (%) | N (%) | C:N molar |
|---|---|---|---|---|---|
| CHN | 0.37 ± 0.05 | 18.63 ± 10.28 | 1.29 ± 0.22 | 0.09 ± 0.02 | 13.65 ± 0.23 |
| KR1 | 0.84 ± 0.02 | 8.68 ± 1.05 | 7.99 ± 0.16 | 0.39 ± 0.01 | 20.64 ± 0.86 |
| WH1 | 0.58 ± 0 | 44.40 ± 18.16 | 6.27 ± 4.10 | 0.29 ± 0.11 | 20.31 ± 7.53 |
| WH2 | 0.76 ± 0.08 | 10.48 ± 0.34 | 10.48 ± 0.15 | 0.52 ± 0 | 20.24 ± 0.40 |
| ECS | 0.79 ± 0 | 6.85 ± 0.05 | 5.95 ± 0.80 | 0.32 ± 0.02 | 18.55 ± 1.54 |
| CH1 | 0.81 ± 0.03 | 7.69 ± 0.98 | 8.36 ± 0.04 | 0.45 ± 0 | 18.77 ± 0.25 |
| TAS7.1 | 0.74 ± 0.03 | 5.94 ± 6.91 | 8.47 ± 1.35 | 1.06 ± 0.24 | 9.76 ± 3.69 |
| TAS7.2 | 0.78 ± 0.07 | 15.12 ± 1.34 | 10.19 ± 0.42 | 1.39 ± 0.06 | 8.55 ± 0 |
| TAS7.3 | 0.81 ± 0.03 | 10.79 ± 1.78 | 9.43 ± 0.22 | 1.30 ± 0.64 | 9.53 ± 4.46 |
| CH2 | 0.71 ± 0.05 | 6.74 ± 0.44 | 5.99 ± 0.41 | 0.33 ± 0.02 | 17.90 ± 0.58 |
| TAS8.1 | 0.64 ± 0.06 | 8.78 ± 0.33 | 9.70 ± 0.53 | 1.57 ± 0.40 | 7.40 ± 1.50 |
| TAS8.2 | 0.66 ± 0.05 | 10.49 ± 1.6 | 8.42 ± 1.39 | 0.93 ± 0.41 | 11.24 ± 3.22 |
| TAS8.3 | 0.74 ± 0.01 | 11.01 ± 0.42 | 8.14 ± 0.25 | 0.73 ± 0 | 12.93 ± 0.37 |
| TSC | 0.82 ± 0.03 | 9.47 ± 0.95 | 7.59 ± 0.92 | 0.41 ± 0.04 | 18.55 ± 0.58 |
| PET3.1 | 0.83 ± 0.03 | 12.56 ± 1.48 | 10.60 ± 2.00 | 0.91 ± 0.15 | 13.51 ± 0.36 |
| PET3.2 | 0.83 ± 0.02 | 12.19 ± 0.92 | 10.80 ± 3.85 | 1.14 ± 0.49 | 11.27 ± 0.95 |
| PET3.3 | 0.81 ± 0.02 | 11.03 ± 4.83 | 9.12 ± 0.20 | 0.54 ± 0.02 | 19.75 ± 0.26 |
| PET3 | 0.81 ± 0.02 | 11.54 ± 0.12 | 10.19 ± 0.21 | 0.50 ± 0.03 | 20.46 ± 0.81 |

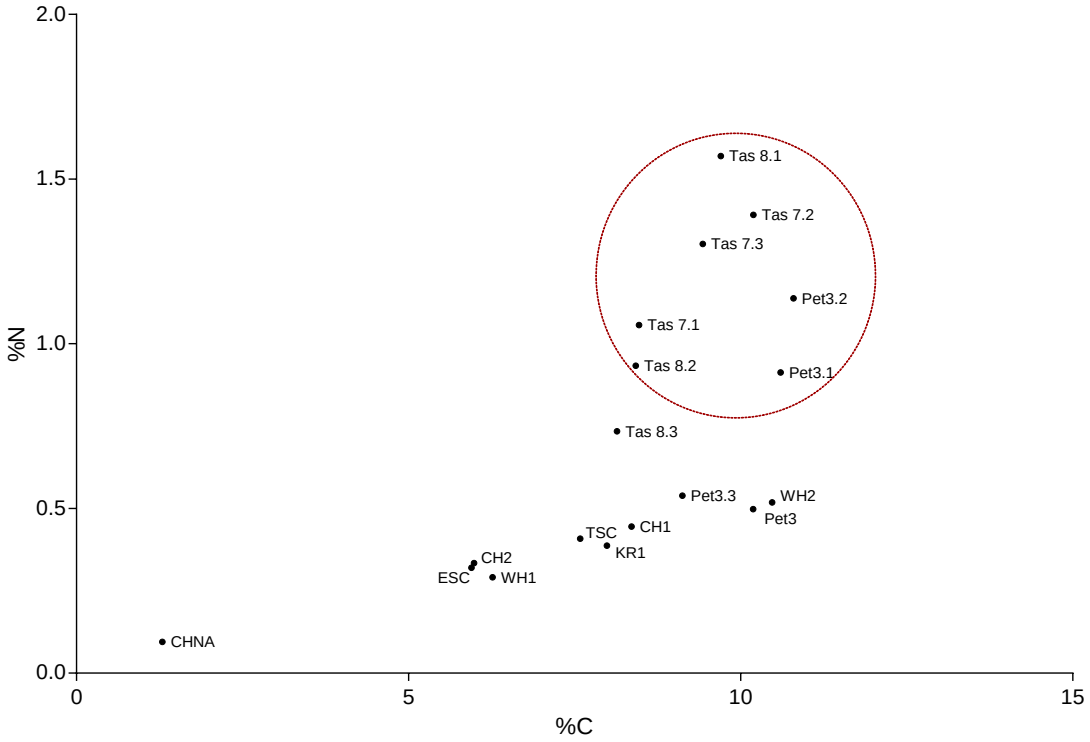


Figure 4-2 Percent carbon (C) and nitrogen (N) content of Macquarie Harbour sediments in November 2012. The red circle encompasses all of the sites located directly adjacent to cagess.

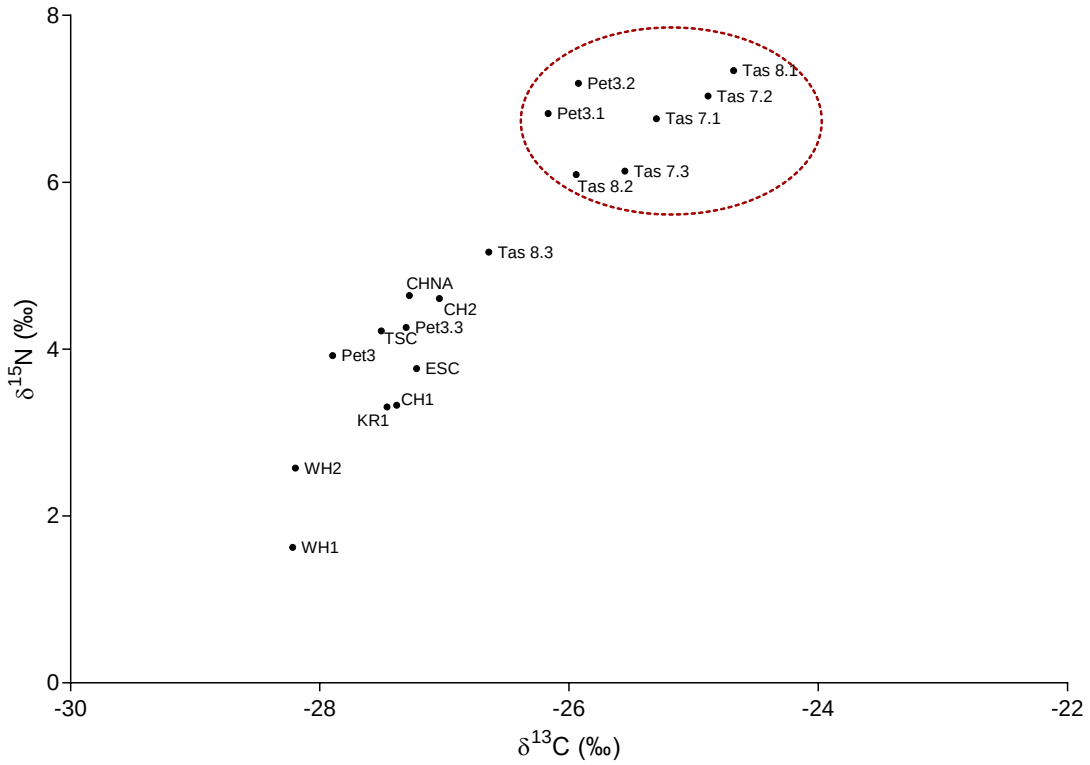


Figure 4-3 $\delta^{13}C$ and $\delta^{15}N$ of Macquarie Harbour sediments in November 2012. The red circle encompasses all of the sites located directly adjacent to cages.

### 4.1.2 Nutrient fluxes

Nutrient fluxes across both harbour and farm scales[5] were consistent with what might be expected based on organic matter loadings (Figure 4-4, Figure 4-5 & Figure 4-6).

*Respiration* - Rates of respiration measured via oxygen consumption were lower at harbour and control sites (i.e. typically < 500 $O_2$ µmol $m^{-2}$ $h^{-1}$) compared to the high rates observed at the cage sites (i.e. typically > 1000 $O_2$ µmol $m^{-2}$ $h^{-1}$) (Figure 4-4). The oxygen consumption rates at the cage sites are comparable to those recorded elsewhere under salmon cage aquaculture (e.g. Hargrave et al. 1993; Pereira et al. 2004). The exception was cage site 8.2 (645 $O_2$ µmol $m^{-2}$ $h^{-1}$) which was fallowed at the time of sampling. Production of dissolved inorganic carbon (DIC; Figure 4-4) (i.e. principally carbon dioxide produced during respiration) was similarly higher at the cage sites compared to the control and harbour wide sites. Both oxygen consumption and DIC production were higher at the 50 m from cage site compared to the control for Tassal Central and Gordon, while the rates were similar at the Petuna 50 m and control site.

In coastal sediments, particularly in low oxygen environments, anaerobic respiration of carbon is often dominant, in which case alternative oxidants (i.e. nitrate, manganese and iron hydroxides, sulphate) are used when the demand of oxidants exceeds the supply of oxygen[6] (Middleburg et al. 2004). In Macquarie Harbour sediments, the rate of dissolved inorganic carbon production (DIC) often exceeds the rate of oxygen consumption, particularly in enriched farm sediments, indicating anaerobic respiration is common (Figure 4-4). The reduced compounds produced during anaerobic respiration (e.g. hydrogen sulphide produced during sulphate reduction of organic matter) can then be reoxidised, potentially consuming oxygen in the process. Carpenter et al. (1991) reported the presence of sulphate-reducing bacteria in Macquarie Harbour. It is likely that there would be some reoxidation (utilizing oxygen) of reduced compounds in the water column that may not be captured in the incubations (e.g. reoxidation of ammonium and sulphide). However, reduced compounds are also likely to be removed from solution and bound in sediments (e.g. metal sulphides) and thus, may not lead to an increase in oxygen consumption (see Middleburg and Levin 2009). The degree to which these reduced compounds ultimately influence bottom water oxygen is difficult to estimate based solely on benthic flux incubations.

[5] when putting farm measurements in the context of harbour wide measurements, it is important to recognise that the total cage area (the area encompassing the cages plus a 100m buffer outside the cages) across the harbour represents ~0.25% of the total benthic area in Macquarie Harbour or ~ 2.5 % of the total benthic area below 15 m depth

[6] Note, during aerobic respiration oxygen is consumed and carbon dioxide produced in a 1:1 ratio, but during anaerobic respiration carbon dioxide is still produced but oxidants, such as sulphate, are used instead of oxygen. Put simply, when carbon dioxide production exceeds oxygen consumption this indicates the presence of anaerobic respiration pathways

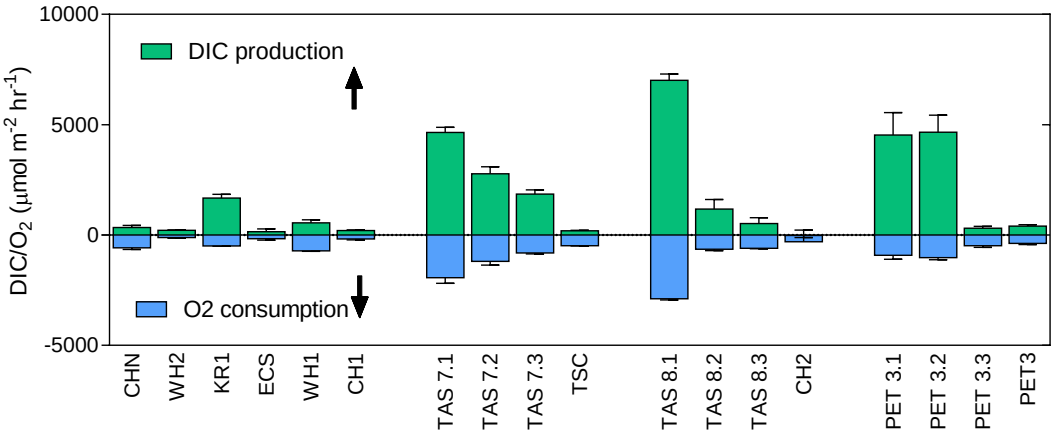


Figure 4-4 Sediment respiration: comparison of O2 and DIC fluxes (µmol $m^{-2}$ $h^{-1}$) (± SE) in November 2012. Note, sites 8.2 and 3.2 were fallowed at the time of sampling

*Nitrogen* - Ammonium fluxes from the sediment largely reflected the patterns of respiration with high rates at farmed sites compared with harbour and control sites (Figure 4-5). Ammonium fluxes were also elevated at the 50 m from cage site for the central harbour farms. Nitrate fluxes were predominately directed into the sediment at all sites (Figure 4-5). This is consistent with a low oxygen environment where the process of nitrification (conversion of ammonia to nitrate in oxic conditions) in sediments is limited and the denitrification process (conversion of nitrate to nitrogen gas in anoxic conditions) must rely on sourcing nitrate from the water column rather than from nitrification in the sediments (see Cornwell et al. 1999). This pattern again follows what we might expect based on oxygen consumption across the sites, with the largest uptake at farmed sites where oxygen consumption is highest, and thus, availability the lowest. The higher fluxes of phosphate out of the sediments at organically enriched sites is consistent with reduced sediment oxygen; phosphate is typically bound in oxidised sediments, but released from reduced anoxic sediments (Figure 4-6).

Rates of denitrification, the process that permanently removes nitrogen from the system, also reflected patterns of organic enrichment with higher rates at farmed sites compared with harbour and control sites (Figure 4-5). Bissett et al. (2009) also reported elevated rates of denitrification under salmon cages in southern Tasmania. The denitrification measurements also further highlight that the denitrifying bacteria are relying primarily on nitrate sourced from the water column rather than nitrate produced via sediment nitrification. This is most evident at the farmed sites where the percentage of denitrification based on water column nitrate is far higher than at harbour and control sites. This is again consistent with Bissett et al. (2009) who reported uptake of nitrate from the water column under salmon cages at the end of the stocking period. In contrast, there was no clear link between DNRA (i.e. conversion of nitrate to ammonium) and rates of respiration and organic enrichment in the November survey (but see section below on seasonal patterns) although the highest of the harbour wide sites was measured at the King River site where rates of respiration were relatively high. When comparing the rates of ammonium release from the sediments to the rates of DNRA, DNRA is clearly only responsible for a small percentage of the ammonium produced from the sediments.

Although the rates of denitrification, and thus, the permanent loss of nitrogen are higher at the cage sites it is important to put this in the context of the total nitrogen budget across the sediment-water column interface. Fluxes of nitrogen leaving the sediments as ammonium from cage sediments range from ~150-1000 µmol $m^{-2}$ $h^{-1}$ compared to ~15-50 µmol $m^{-2}$ $h^{-1}$ leaving via denitrification as nitrogen gas. Thus, denitrification is only removing a small percentage of the nitrogen from the system, the majority re-entering the water column as bioavailable nitrogen. At the harbour and control sites rates of denitrification are significantly lower (2-7 µmol $m^{-2}$ $h^{-1}$) but so too is the rate of ammonium regeneration (0-80 µmol $m^{-2}$ $h^{-1,}$ only the King River site is >20 µmol $m^{-2}$ $h^{-1}$) from the sediments. In summary, even though rates of denitrification are higher at the cage sites, the percentage of nitrogen

removed via denitrification is lower at the cage compared to the harbour and control sites due to the very high efflux of ammonium at the cage sites. Bissett et al. (2009) reported a similar pattern under salmon cages in southern Tasmania; at the end of the stocking period denitrification increased dramatically, yet it accounted for <15% of the total nitrogen efflux (compared to 40-60% at reference sites) because of the very high efflux of ammonia. Similarly, Christensen et al. (2000) reported denitrification activity 2.5 higher below fish cages than at reference stations, but that this only accounted for 0.1% of the additional nitrogen input to the system in connection to fish farming.

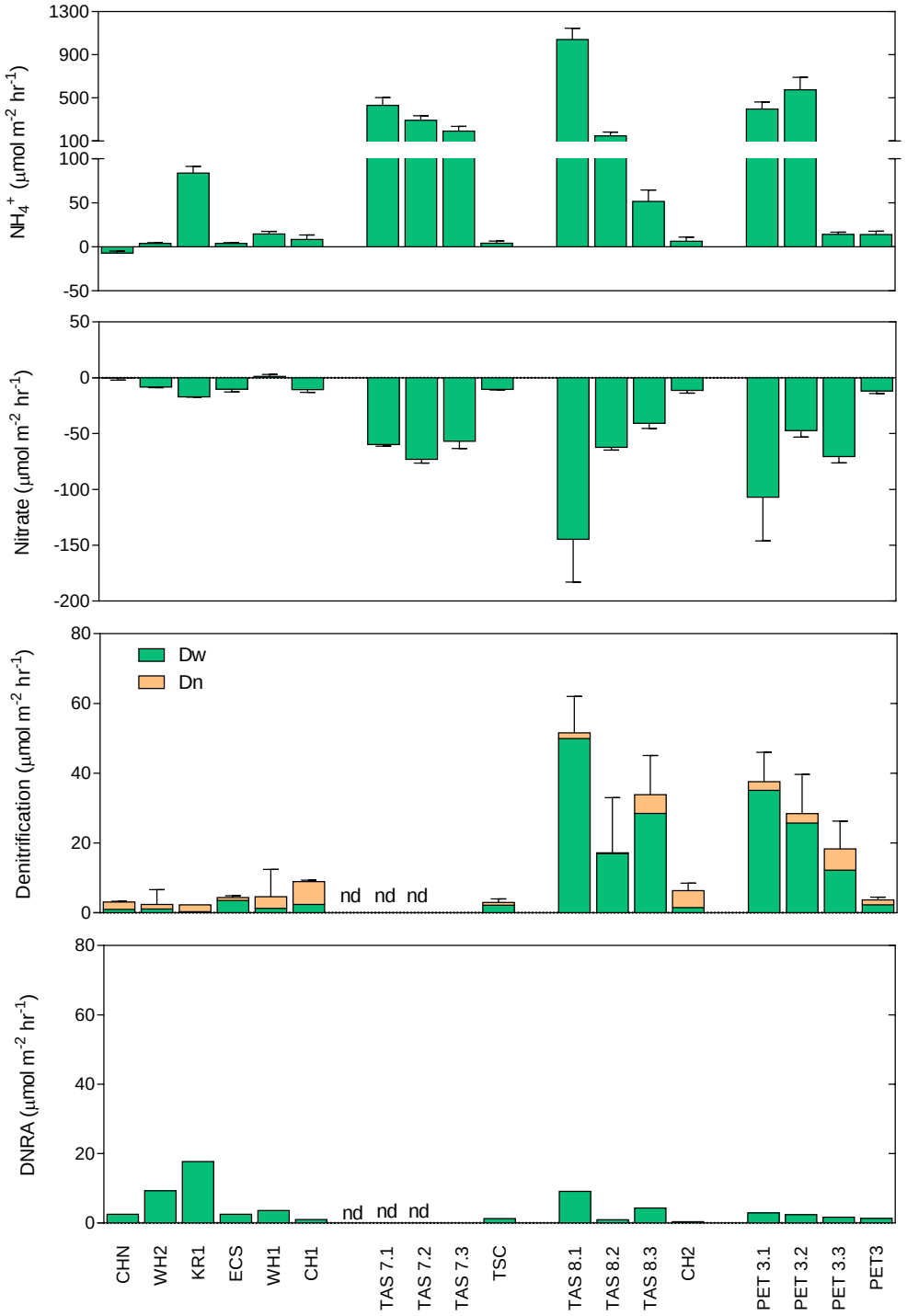


Figure 4-5 Sediment-water nitrogen fluxes from November 2012. Ammonium ($NH_4$), nitrate ($NO_3$), denitrification ($N_2$) and DNRA flux (µmol m$^{-2}$ h$^{-1}$) (± SE). For denitrification subscript w represents water-column-driven nitrate reduction and subscript n nitrate sourced from sediment nitrification. nd represents no

data. Note, sites 8.2 and 3.2 were fallowed at the time of sampling

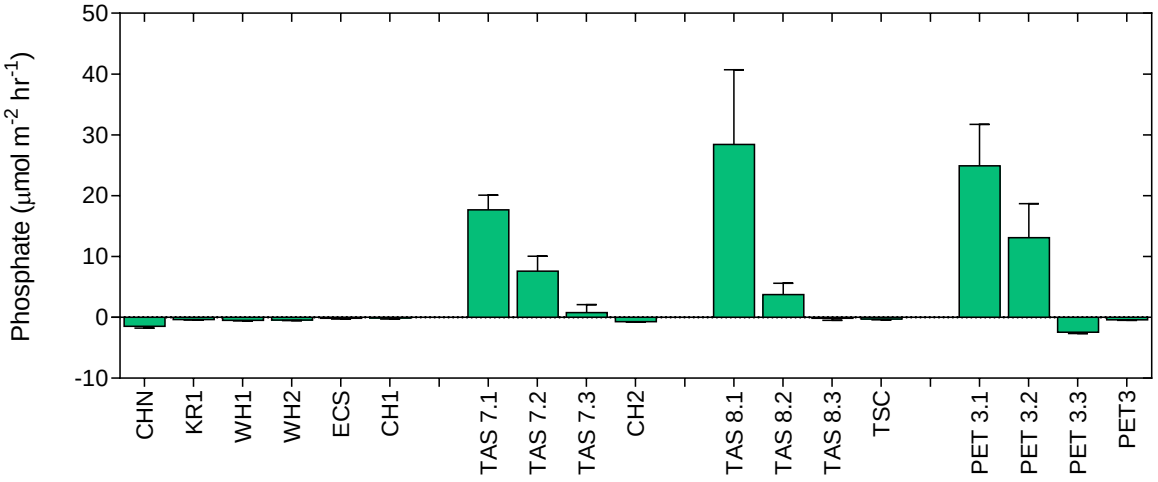


Figure 4-6 Sediment water phosphate flux (µmol $m^{-2}$ $h^{-1}$) (± SE) in November 2012. Note, sites 8.2 and 3.2 were fallowed at the time of sampling.

# 4.2 Variation in time

The original project scope was extended to include an assessment of the functional response of sediments to farm management. The two deeper water leases (Gordon and Central) surveyed in November and January were re-visited in May and September. This included flux measurements at the two cages, 50 m and 1000 m sites at each lease. Importantly, the two 1000 m control sites also provides useful information about background temporal variation in sediment nutrient fluxes in Macquarie Harbour sediments.

## 4.2.1 Environmental data

*Water Column* – Bottom water temperature and salinity remained very stable across the November 2012 and January, May and September 2013 surveys (Table 4-3); varying by only 1°C (14.3 – 15.3°C) and 1ppt (30.4-31.4ppt). Bottom water dissolved oxygen was variable across sites in November 2012 (1.3-2.7 mg/L) and January 2013(1.2-3.3 mg/L), however by May (0.4-1.6 mg/L) and September (0.2-0.9 mg/L) oxygen concentrations had clearly dropped. In the surface waters, salinity ranged between 8 and 11ppt in November and January 2012/13 before dropping in May (4-10ppt) and September 2013 (1-3ppt). This most likely reflects the large step change increase in Gordon River flows in early 2013 due to increased power generation at the Gordon Power Station in response to the fixed carbon price period (Hydro Tasmania 2013; Figure 4-7). The temperature of surface waters increased between spring (November 15-16°C) and summer (January 17-19°C) before decreasing in the autumn (May 10-12°C) and winter (September 10-12°C) surveys. Consistent with the drop in temperature and salinity, the dissolved oxygen content of surface waters increased between spring/summer (8-10 mg/L) and autumn/winter (10-12 mg/L).

*Sediments* – The carbon and nitrogen content of farm sediments remained distinct from the control sites for each of the surveys (Figure 4-8). Farm sediments had a higher nitrogen content and typically a higher carbon content. This is further highlighted in Figure 4-9 which shows the C:N molar ratio of cage, 50 m from cage and control sediments through time at the Central and Gordon leases. At both leases the C:N is lower at the cage sites, and intermediate at the 50 m sites compared to the control sites indicating the level of farm derived organic matter enrichment. Similarly, the $\delta^{13}C$ and $\delta^{15}N$ isotopic signature of the control sites remained depleted compared to the farm sediments across all surveys, again the 50 m sites was typically intermediate between the control and cage sediments (Figure 4-10). For both the C:N content and isotopic signature of sediments there was no clear link

with farming activity, the distinction between farm and control sediments remained regardless of fallowing.

Table 4-3 Water column parameters measured at sampling sites in Macquarie Harbour in November 2012, and January, May and September 2013. No surface water (<1 m depth) data were available for TAS7.1 and TAS7.3 in November 2012.

| | | Temperature (°C) | | Salinity (ppt) | | DO (mg/L) | |
|---|---|---|---|---|---|---|---|
| Parameter | Depth (m) | Surface water | Bottom water | Surface water | Bottom water | Surface water | Bottom water |
| Nov-12 | | | | | | | |
| CH2 | 35 | 16.1 | 14.3 | 9.7 | 31 | 9.6 | 1.9 |
| TSC | 32 | 15.5 | 14.3 | 7.4 | 31 | 9.2 | 2.2 |
| TAS7.1 | 36 | n.d | 14.3 | n.d | 31.2 | n.d | 1.7 |
| TAS7.2 | 26 | 15.4 | 14.4 | 11.3 | 31 | 9.2 | 1.1 |
| TAS7.3 | 26 | n.d | 14.3 | n.d | 31.1 | n.d | 1.3 |
| TAS8.1 | 36 | 15.1 | 14.3 | 8 | 31 | 9.9 | 1.6 |
| TAS8.2 | 36 | 15.5 | 14.4 | 8.3 | 31.1 | 9.5 | 1.3 |
| TAS8.3 | 35 | 16.2 | 14.4 | 8.6 | 31 | 9.4 | 1.3 |
| Jan-13 | | | | | | | |
| CH2 | 35 | 18.6 | 14.8 | 13.7 | 31.4 | 8.6 | 3.3 |
| TSC | 32 | 17.8 | 14.7 | 9.3 | 31.3 | 9 | 2.7 |
| TAS7.1 | 36 | 18.5 | 14.7 | 13 | 31.4 | 8.8 | 2.6 |
| TAS7.2 | 26 | 19 | 14.5 | 13.3 | 30.9 | 8.8 | 1.2 |
| TAS7.3 | 26 | 18 | 14.6 | 13.4 | 31.1 | 8.7 | 1.9 |
| TAS8.1 | 36 | 17.4 | 14.7 | 8.4 | 31.3 | 9.2 | 2.3 |
| TAS8.2 | 36 | 17.1 | 14.7 | 8.1 | 31.4 | 9.4 | 2.4 |
| TAS8.3 | 35 | 16.8 | 14.6 | 7.8 | 31.2 | 9.4 | 2.1 |
| May-13 | | | | | | | |
| CH2 | 37 | 11.9 | 15.2 | 9.6 | 31.1 | 10.2 | 1.61 |
| TSC | 33 | 10.9 | 15.2 | 4 | 30.4 | 11.1 | 0.6 |
| TAS7.1 | 35 | 11.4 | 15.2 | 7.9 | 31 | 10.5 | 0.4 |
| TAS7.2 | 25 | 11.5 | 15.2 | 8.7 | 31 | 10.4 | 1.02 |
| TAS7.3 | 25 | 11.4 | 15.2 | 8.6 | 31 | 10.5 | 1.09 |
| TAS8.1 | 31 | 10.6 | 15.3 | 4.2 | 31.2 | 11 | 0.98 |
| TAS8.2 | 33 | 10.7 | 15.2 | 4.1 | 31.2 | 11.3 | 0.93 |
| TAS8.3 | 35 | 10.9 | 15.2 | 4.1 | 31.2 | 11.1 | 0.83 |
| Sep-13 | | | | | | | |
| CH2 | 37 | 11.9 | 14.8 | 2.5 | 30.8 | 10.4 | 0.7 |
| TSC | 33 | 11.3 | 14.9 | 1.8 | 30.9 | 11 | 0.72 |
| TAS7.1 | 32 | 11.8 | 14.9 | 2.7 | 30.8 | 10.4 | 0.46 |
| TAS7.2 | 25 | 11.5 | 14.8 | 2.6 | 30.6 | 10.6 | 0.59 |
| TAS7.3 | 25 | 11.2 | 14.8 | 2.6 | 30.6 | 10.7 | 0.9 |
| TAS8.1 | 31 | 10.5 | 14.9 | 2.2 | 30.9 | 10 | 0.55 |
| TAS8.2 | 33 | 11.4 | 14.9 | 1.9 | 30.9 | 10.3 | 0.56 |
| TAS8.3 | 35 | 10.5 | 14.9 | 2 | 30.9 | 10.2 | 0.21 |

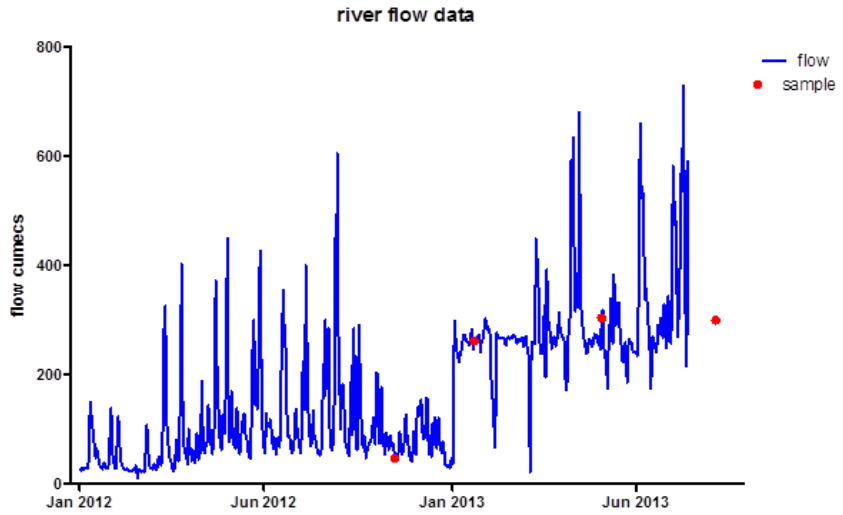


Figure 4-7 Gordon River flow before and during the study period. Red dots identify November, January, May and September flux surveys.

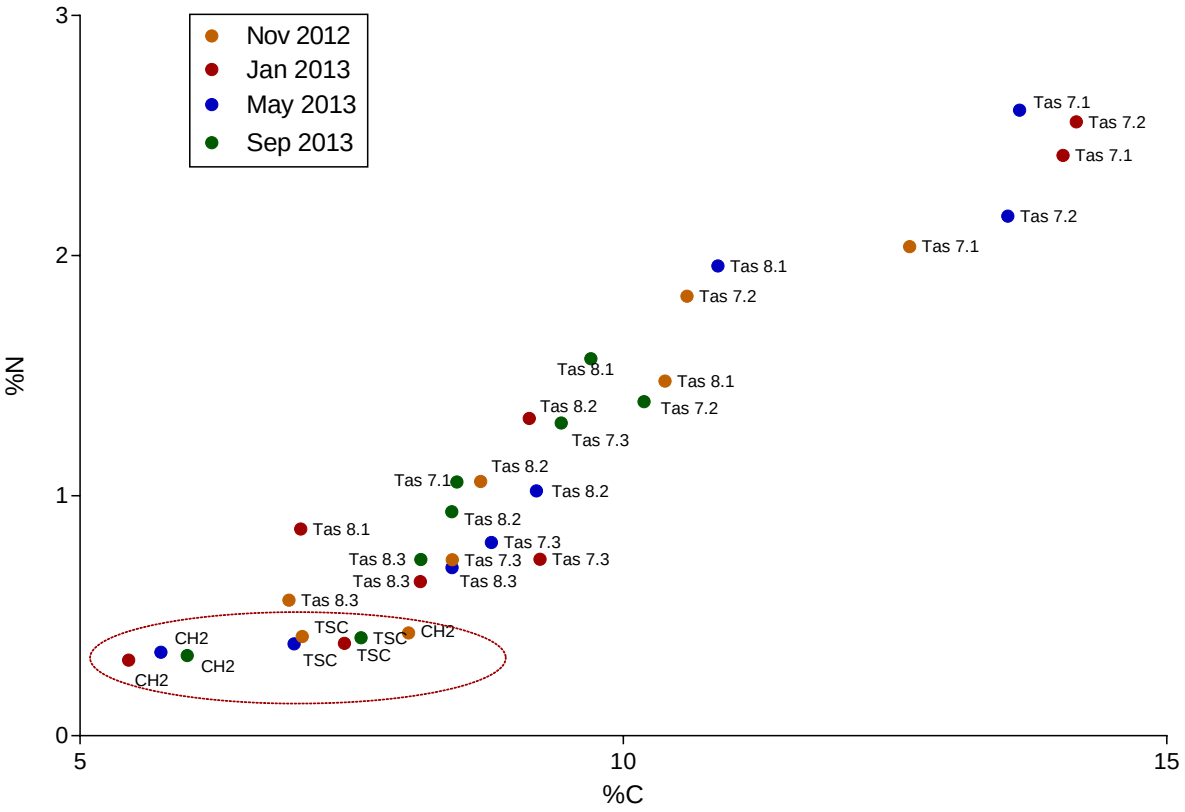


Figure 4-8 Percent carbon (C) and nitrogen (N) content of Macquarie Harbour through time. The red circle encompasses the control sites.

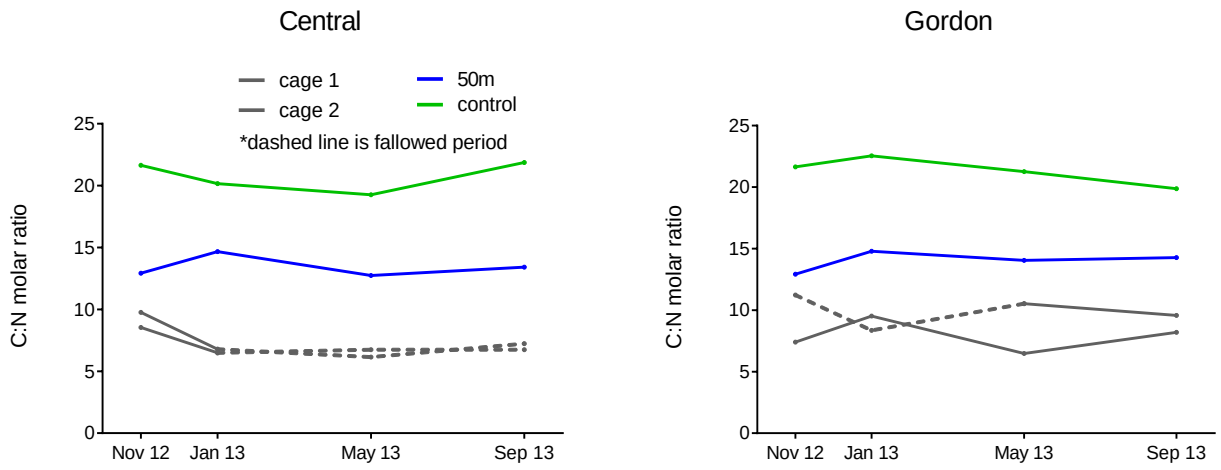


Figure 4-9 C:N molar ratio of Macquarie Harbour sediments through time at Central and Gordon leases.

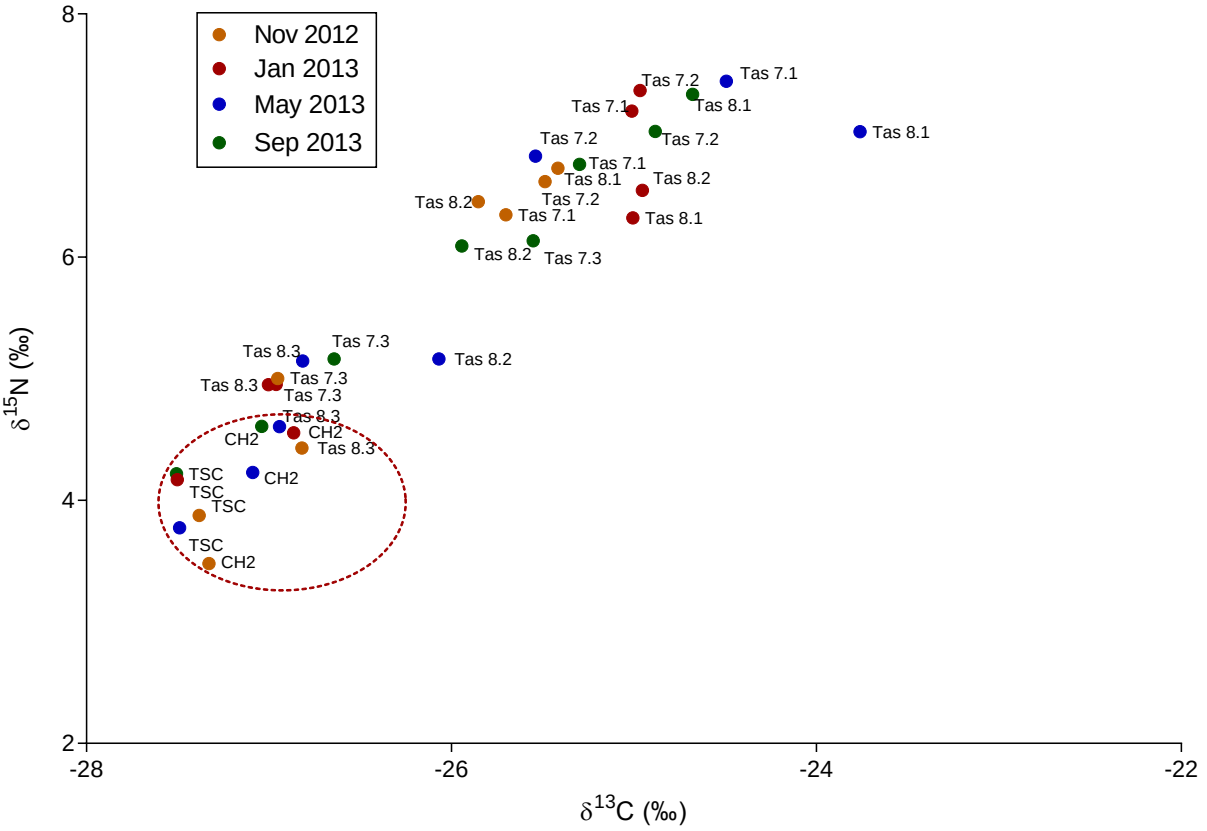


Figure 4-10 $\delta^{13}C$ and $\delta^{15}N$ of Macquarie Harbour sediments in the November 2012 and January, May and September 2013 surveys. The red circle encompasses the control sites.

### 4.2.2 Nutrient fluxes

Table 3-1 shows that the 2 cages at each of the leases were subject to different stages of farm management. At the Central Harbour lease cages 7.1 and 7.2 were stocked for the first 2 surveys and fallowed for the final two surveys. At the Gordon lease, cage 8.1 was stocked for the duration of the study while cage 8.2 was fallowed for the first 3 surveys before been stocked for the September survey.

As expected, rates of respiration (measured by oxygen) and ammonia production both declined at the Central Harbour lease cage sites during fallowing; indicative of sediment recovery (Figure 4-11). Similarly, phosphate production decreased during fallowing (Figure 4-11). There is some suggestion that nitrate uptake increased during fallowing and this may reflect increased rates of denitrification and or an increased reliance on nitrate from the water column for denitrification due to the decline in bottom water oxygen concentrations in the latter two surveys (see Table 4-3).

At the Gordon lease, rates of respiration were significantly higher at the stocked (8.1) compared to the fallowed cage (8.2) in the November survey as expected. However, in the January and May surveys rates of respiration and ammonia production were also elevated at the fallowed cage (8.2). Between May and September 2013 both sites were stocked, yet the change in sediment function between these surveys is consistent with sediment recovery (decrease in respiration and ammonia production). This suggests that drivers other than stocking are playing a significant role in determining sediment condition (also see separate discussion on the influence of bottom water DO on sediment biogeochemistry below). This could include improved food conversion ratios, changes in diet (and thus waste composition), season etc. It is also possible that the site may have shifted slightly due to slight cage movement in the prevailing conditions on the day of sampling.

Data for DNRA and denitrification are not yet available for the May and September 2013 surveys. In the November survey there was no clear link between rates of DNRA and OM enrichment, however, in January rates of DNRA had increased significantly and appear elevated at sites associated with OM enrichment from fish farms (Figure 4-12). Rates of denitrification are also elevated at the OM enriched sites, but in January, the rates of DNRA are now far more comparable to the rates of denitrification. Competition for nitrate between DNRA that converts the nitrate back into bioavailable ammonium versus denitrification which converts nitrate to nitrogen gas which is lost from the system is reported in numerous studies (see Roberts et al. 2012 and references therein). Christensen et al (2000) noted the importance of high organic carbon loading in the promotion of DNRA over denitrification beneath fish farms in Horsens Fjord, Denmark. High rates of DNRA over denitrification have also been attributed to anoxic conditions in bottom waters due to limited replenishment of nitrate because nitrate ammonifiers have a higher affinity for nitrate (Childs et al 2002). Other studies have attributed higher rates of DNRA to sulphate reduction in the sediments and subsequent sulphide inhibition of nitrification and denitrification (An and Gardner 2002). In this study it is difficult to isolate the control on the relative rates of DNRA and denitrification because high carbon loading also leads to highly reducing conditions.

*Effects of bottom water DO* - Over the course of the study, the very low bottom water oxygen conditions measured in the May (0.4-1.6 mg/L) and September 2013 (0.2-0.7 mg/L) indicate that the bottom waters at the study sites have shifted to hypoxic conditions (<2.0 mg/L) in May and anoxic conditions (<0.5 mg/L) in September. Depletion of oxygen in bottom waters is known to cause a decrease in the oxidation-reduction potential (redox condition) and a fundamental shift in sediment biogeochemistry and nutrient cycling processes (e.g. Strumm and Morgon 1970). Oxygen conditions this low are known to significantly alter nutrient cycling pathways conducive to the build-up of toxic compounds such as hydrogen sulfide and ammonia gas. The fraction of remineralised ammonium that is recycled from the sediments back into the water column is also controlled by redox conditions, typically increasing as bottom water oxygen concentrations decrease. This can occur due to the limited supply of oxygen for the nitrification process and/or hydrogen sulphide inhibition of nitrification and denitrification. Although the rates of denitrification were higher at the cage sites throughout this study, the percentage of nitrogen removed via denitrification is far lower at the cage

compared to the harbour and control sites. This is consistent with high OM enrichment and elevated oxygen demand at cage compared to the harbour and control sites. At the Gordon lease, the sediment response to stocking wasn't as expected; rates of respiration (measured as oxygen consumption and DIC production) and ammonium production decreased rather than increased. Changes in diet, feeding or site location (e.g. due to slight cage movement in the prevailing conditions on the day of sampling may explain this as discussed above, however, a major shift in the nutrient transformation pathways due to the anoxic conditions may also be responsible.

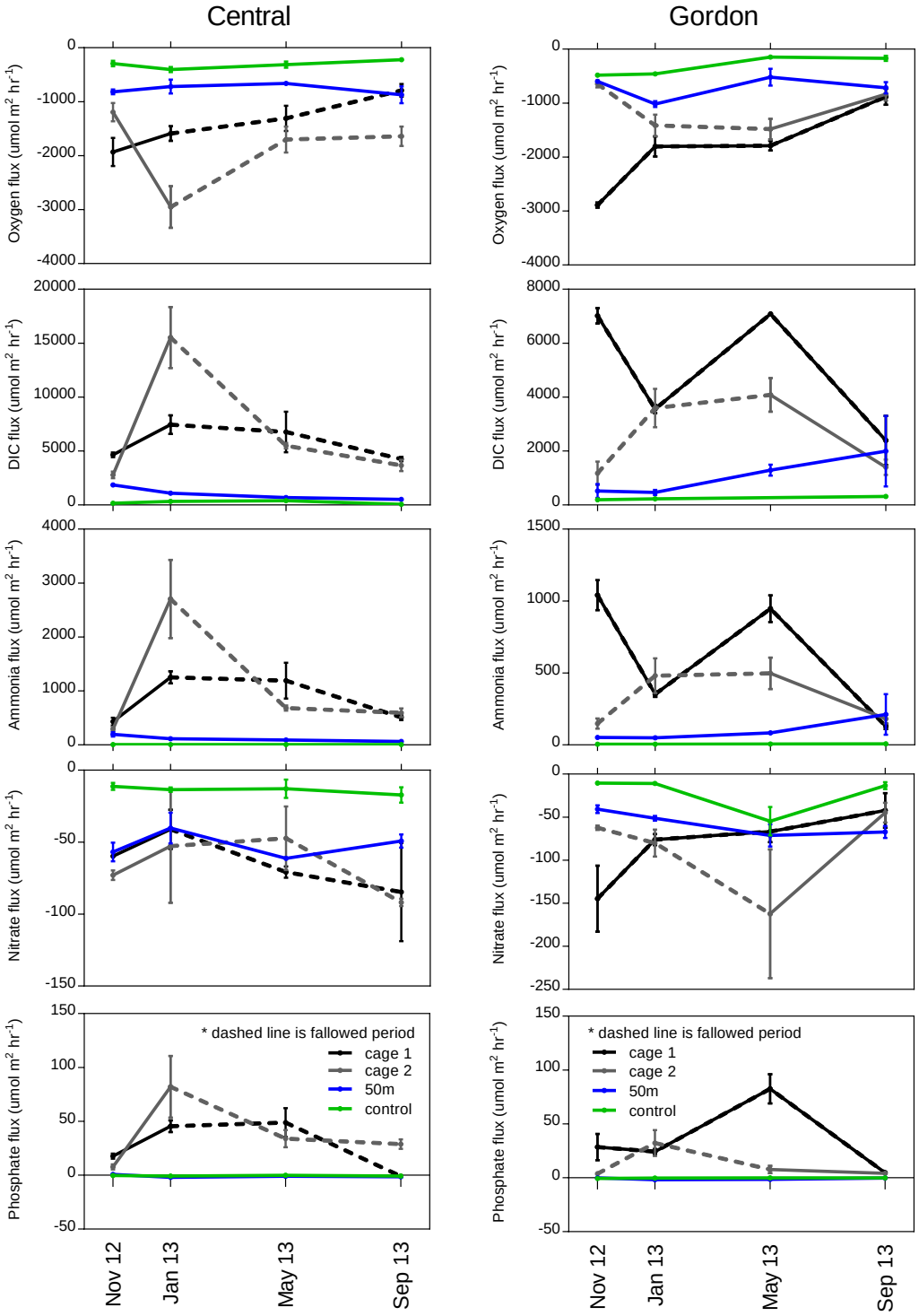


Figure 4-11 Sediment-water fluxes of $O_2$, DIC, $NH_4$, $NO_3$ and $PO_4$ (µmol $m^{-2}$ $h^{-1}$) (± SE) at Tassal Central (left column) and Gordon (right column) leases in November 2012 and January, May and September 2013.

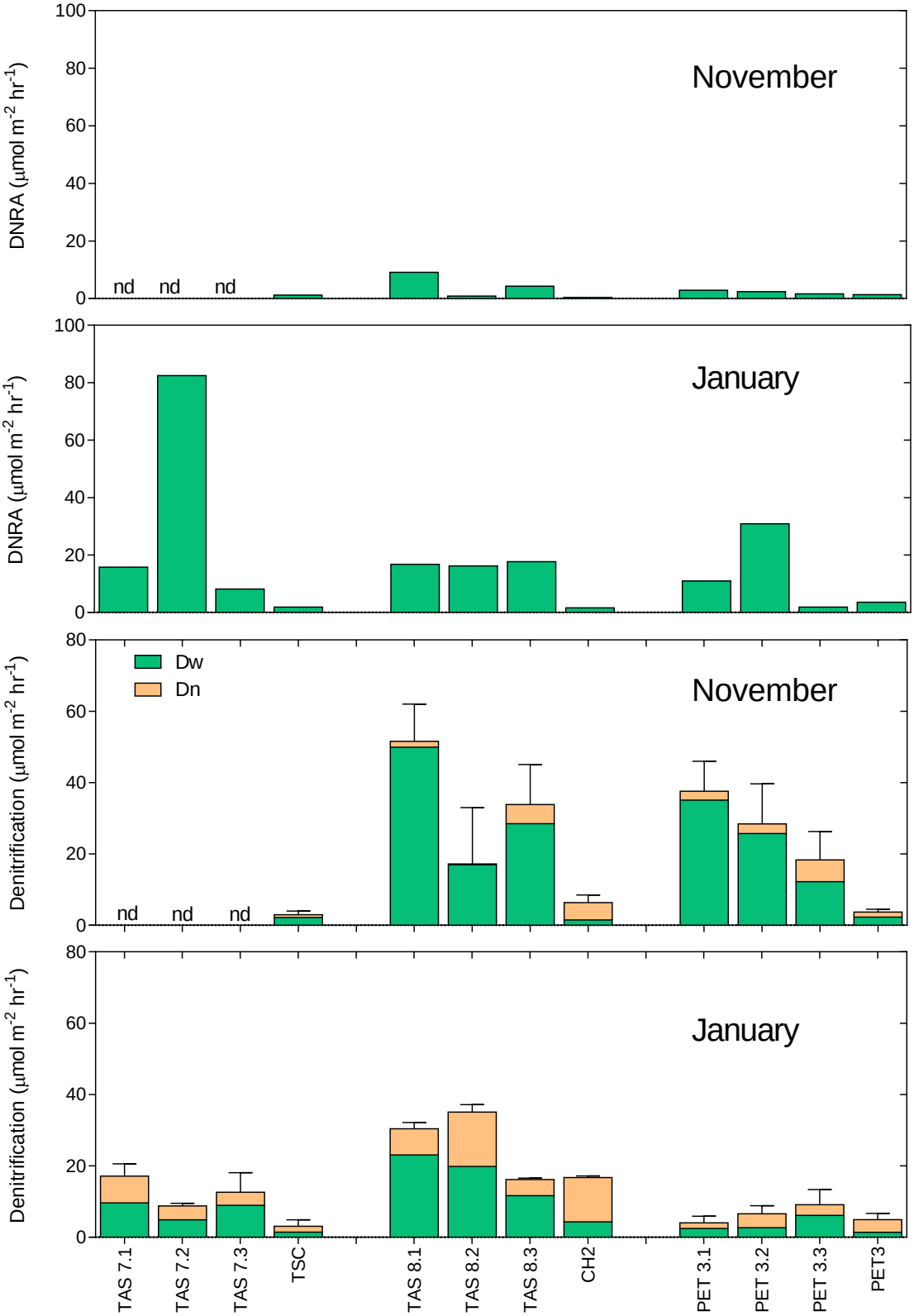


Figure 4-12 Sediment-water DNRA and denitrification ($N_2$) fluxes (µmol $m^{-2}$ $h^{-1}$)(± SE) in November 2012 and January 2013. For denitrification subscript w represents water-column-driven nitrate reduction and subscript n nitrate sourced from sediment nitrification. nd represents no data.

### 4.2.3 Indicators of sediment condition

Measuring the full range of variables that characterise the structure and function of benthic soft-sediments is an expensive exercise. Although numerous studies have identified environmental indicators to help characterise the response to organic enrichment, it is well documented that the relationship between indicator variables and how they change in response to enrichment is likely to vary across different sediment environments and in different systems (e.g. Keeley et al. 2013 and references therein). The extensive range of sediment properties and processes measured in this study provided an opportunity to identify indicators of sediment condition in response to organic enrichment in Macquarie Harbour.

In Macquarie Harbour sediments, the C:N ratio and isotopic signature of sediment organic matter appear to provide a good environmental indicator of the footprint of farm derived enrichment. This is largely because Macquarie Harbour sediments have an organic matter signature reflecting a terrestrial origin which provides a strong contrast to the organic matter signature associated with the farm sites. Despite a shift away from a dependence on marine sources of protein in fish meal in recent years, the signature of farm waste still appears distinct from that of natural sources of organic matter in Macquarie Harbour sediments.

Measured rates of organic matter mineralisation are also likely to provide a useful proxy for organic matter loading (see Eyre and Ferguson 2009). In this study, mineralisation rates, measured as either oxygen consumption or dissolved inorganic carbon (DIC) production, increased markedly at farm enriched sites, consistent with a relationship between loading and rates of mineralisation. However, it was also clear that organic enrichment of sediments and the subsequent demand for oxygen is often exceeding supply, with anaerobic respiration of carbon becoming dominant at farm enriched sites. This is best seen when DIC production is plotted as a function of oxygen consumption (Figure 4-13); DIC production clearly exceeding the theoretical 1:1 molar relationship between oxygen and dissolved inorganic carbon (DIC) expected for marine organic matter[7]. From a monitoring perspective, DIC provides the best measure of organic matter mineralisation because it is the end product of all respiration pathways (aerobic and anaerobic), however the ratio between $DIC/O_2$ provides useful insight into the pathways of organic matter mineralisation.

As a function of sediment carbon and nitrogen loads, there was a strong relationship between total carbon mineralisation and both the percentage of organic carbon (Figure 4-14) and nitrogen (Figure 4-15) in the sediments. Not surprisingly, there was also a clear relationship between C:N ratio of the sediment organic matter and rates of total carbon mineralisation (Figure 4-16). This suggests that the C:N ratio is useful proxy for carbon quality in Macquarie Harbour sediments, lower ratios indicating increased organic matter lability based on the relationship with rates of organic matter mineralisation.

Given the importance of nitrogen cycling in coastal systems like Macquarie Harbour, identifying cost effective indicators of critical processes, such as denitrification which permanently removes fixed nitrogen, is crucial for understanding ecosystem dynamics and effective management. In an

[7] Note, although the source of organic material will directly influence the expected relationship between oxygen and dissolved inorganic carbon (see Middleburg et al. 2004), the significant departure in DIC production from oxygen consumption remains consistent with anaerobic respiration

evaluation of denitrification measurements for 22 shallow coast ecosystems in Australia, Eyre and Ferguson (2009) found a strong relationship between carbon mineralisation and denitrification efficiency, with a marked decrease in denitrification efficiency (i.e. the percentage of total inorganic nitrogen released from the sediments as nitrogen gas and removed from the system) above carbon mineralisation rates of 1000-1500 µmol $m^{-2}$ $hr^{-1}$. The results of this study also demonstrate a clear relationship between denitrification efficiency and rates of carbon mineralisation, with denitrification efficiency declining markedly at carbon mineralisation rates > 1000µmol $m^{-2}$ $hr^{-1}$ (Figure 4-17). This is largely because of the increased recycling of nitrogen back to the water column in the form of ammonia at enriched sites (Figure 4-18).

Overall, the results from this study highlight that bulk identifiers of organic matter source (C:N ratio and $\delta^{13}C$ and $\delta^{15}N$ signature) together with measured rates of respiration appear to be good environmental indicators of the footprint of farm derived organic matter and sediment function respectively in Macquarie Harbour.

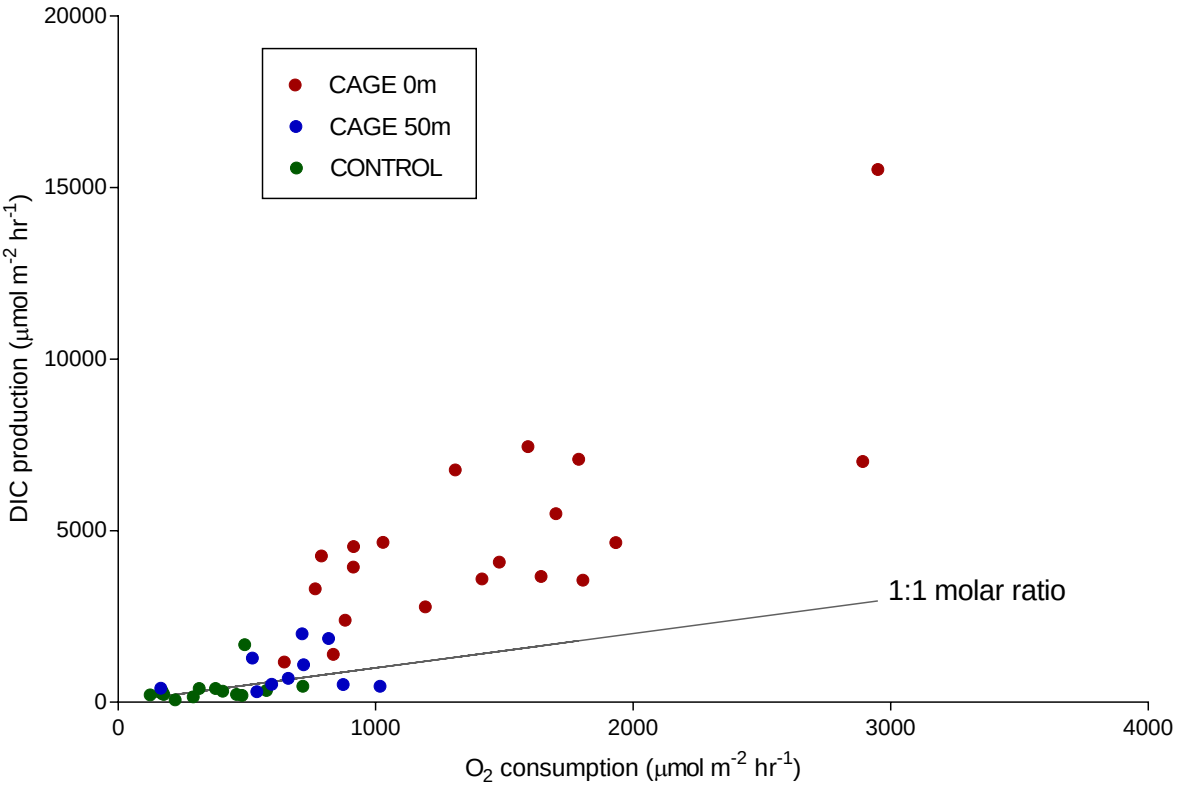


Figure 4-13 Relationship between dissolved inorganic carbon (DIC) production and oxygen consumption across all measurements in the study. The 1:1 molar ratio for DIC/$O_2$ expected for marine organic matter if aerobic processes dominate organic matter mineralisation is plotted to demonstrate the importance of anaerobic mineralisation processes at farm enriched sites.

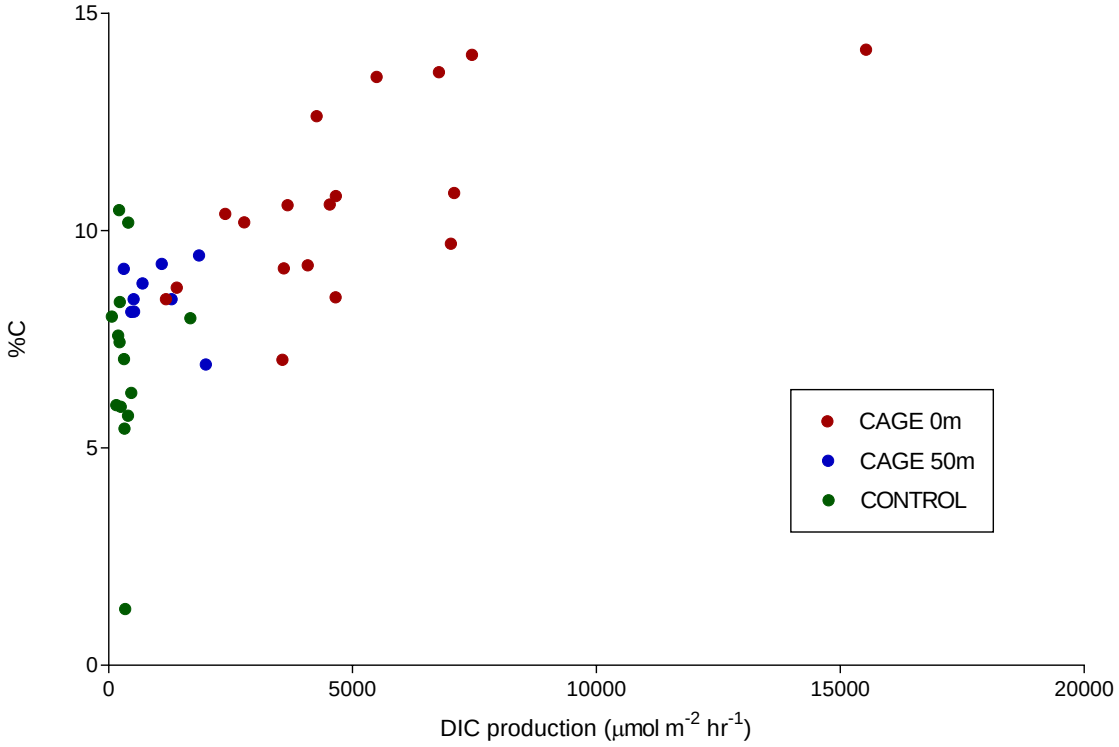


Figure 4-14 Relationship between sediment organic carbon content and carbon mineralisation (DIC production)

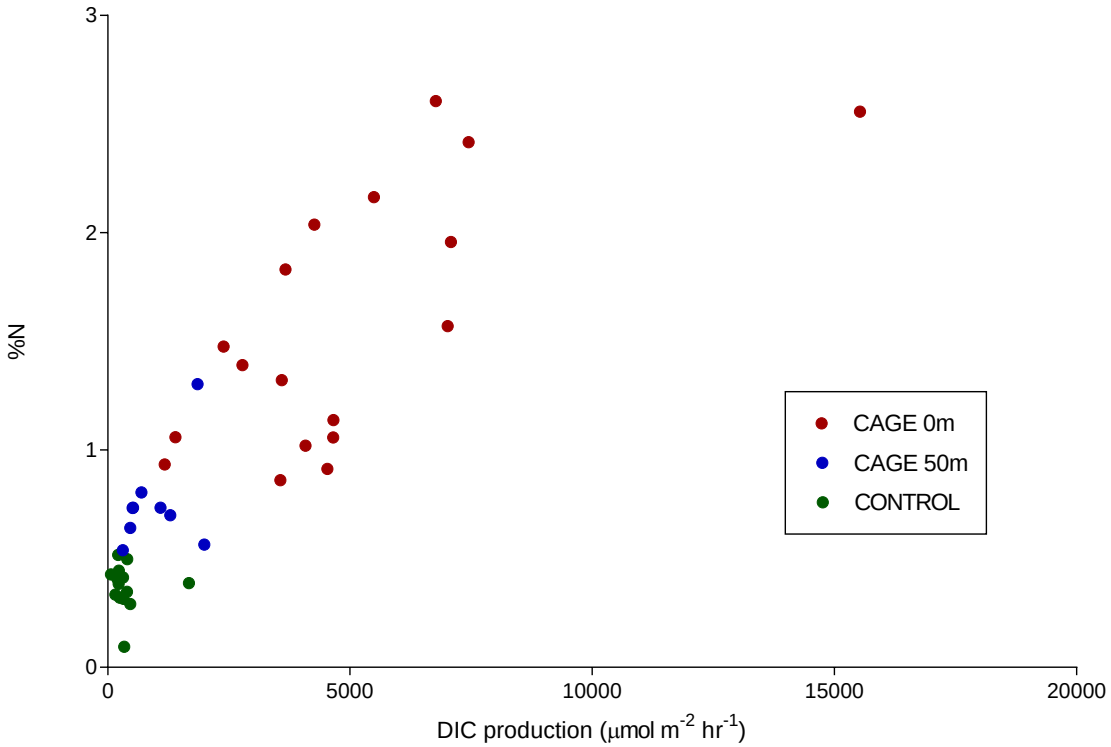


Figure 4-15 Relationship between sediment organic nitrogen and carbon mineralisation (DIC production)

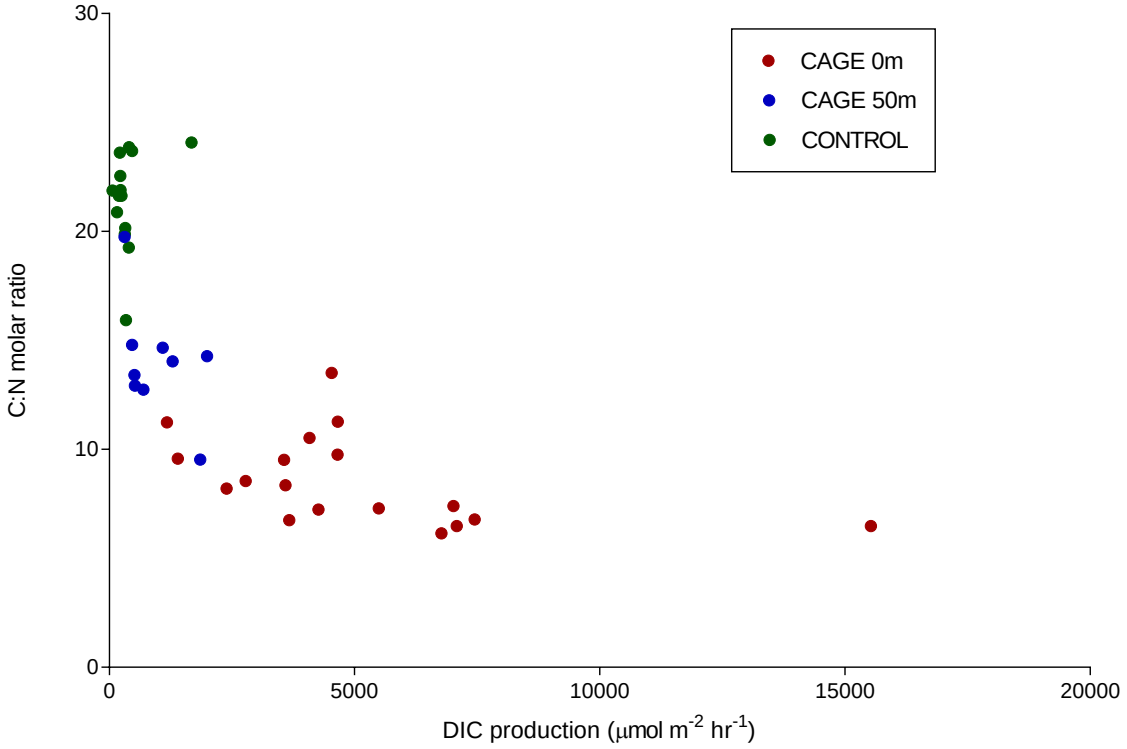


Figure 4-16 Relationship between sediment C:N ratio and carbon mineralisation (DIC production)

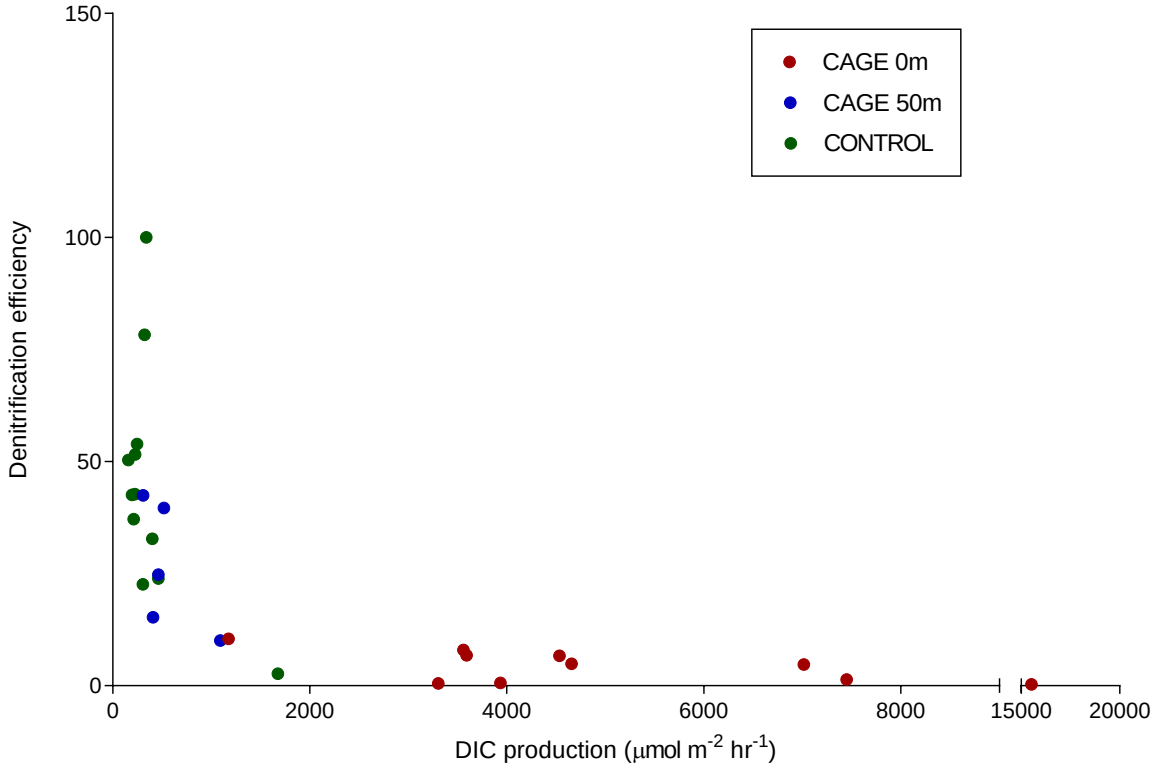


Figure 4-17 Denitrification efficiency (%) as a function of carbon mineralisation (DIC production)

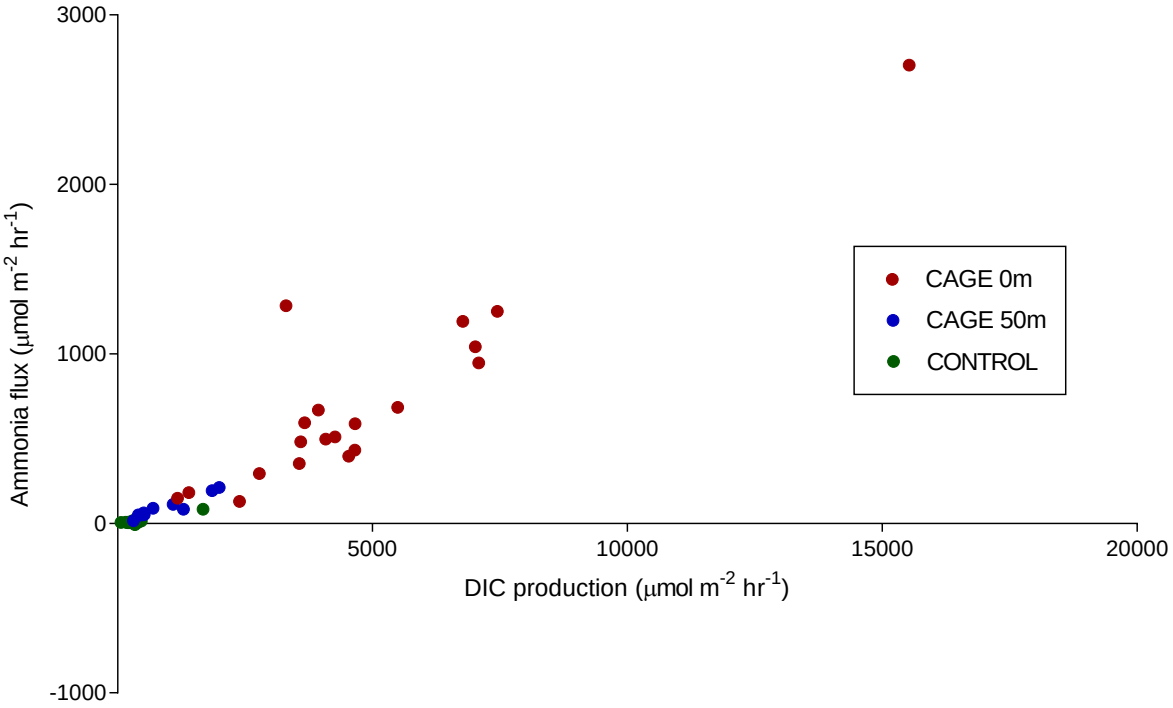


Figure 4-18 Ammonia production as a function of carbon mineralisation (DIC production)

## 4.3 Model calibration

The model used as part of the 2010/2011 Macquarie Harbour EIS has undergone a number of improvements as part of an update to understand the hydrodynamic, water quality and seabed processes within the harbour. Firstly, the forcing's behind the hydrodynamic model have been updated to include recent tidal, wind and river flow from October 2011 to September 2012. The hydrodynamics have also been improved by recalibrating the flows against ADCP collected in 2012 from 2 locations within the centre of the harbour. The re-calibrated hydrodynamic model has subsequently been run for a period of 1 year, divided into 1 month blocks. These results files have been saved and are used to provide the hydrodynamic forcing's for the water quality/ecological model calibration.

As part of the ecological modelling recalibration process, 12 months (October 2011-September 2012) of water column data (Dissolved oxygen, Nitrate, Ammonium, Chlorophyll A, TKN) have been collected at 13 sites and 4 different depths within Macquarie Harbour. This data has been reformatted so it can be used and compared against as part of the ecological modelling calibration process. The ecological modelling was undertaken using Ecolab which is essentially an equation solver whose processes can be coupled directly to the hydrodynamic/advection dispersion results. The model template handles nitrogen related exchanges similar to the processes described in Figure 1-1 where organic matter is deposited to the seabed and mineralised consuming oxygen. Ammonium is produced from the sediment back into the water column with some of it being taken up by the phytoplankton and some converted to nitrate via nitrification. Nitrate can be also taken up by the phytoplankton or converted to gas via denitrification (via a set of equations representing these processes). The model also takes into account riverine and ocean sources of nutrients (generally as source inputs into the model domain over time). The previous ecological modelling relied on sediment water interaction data from New Zealand salmon farms; specifically from farms in Big Glory Bay, Stewart Island.

To improve the seabed sediment-water column interface interactions, data collected in this study was inserted directly into the ecological template using maps created directly from the collected *in situ* benthic flux data. Each measurement was attributed to an area of influence surrounding it and

distinction was made between farming and non-farming areas according to the sampling program. The sediment flux part of the model template utilized sediment-water exchange rates of ammonium, nitrate and dissolved oxygen (Figures 4-4, 4-5 & 4-6), and implicitly integrates the denitrification removal of nitrates from the sediment. These rates are expressed as empirical values and aren't derived from equations, unlike the rest of the ecological template.

Sediment deposition footprints were also created as part of the modelling assessment, showing areas of deposition exceeding the organic enrichment threshold identified in the literature (Figure 4-22). Literature values were based on studies undertaken by Cromey et al. 2002 (for Scottish water bodies) and it highly recommended that future studies are undertaken to identify organic enrichment levels specifically within the harbour.

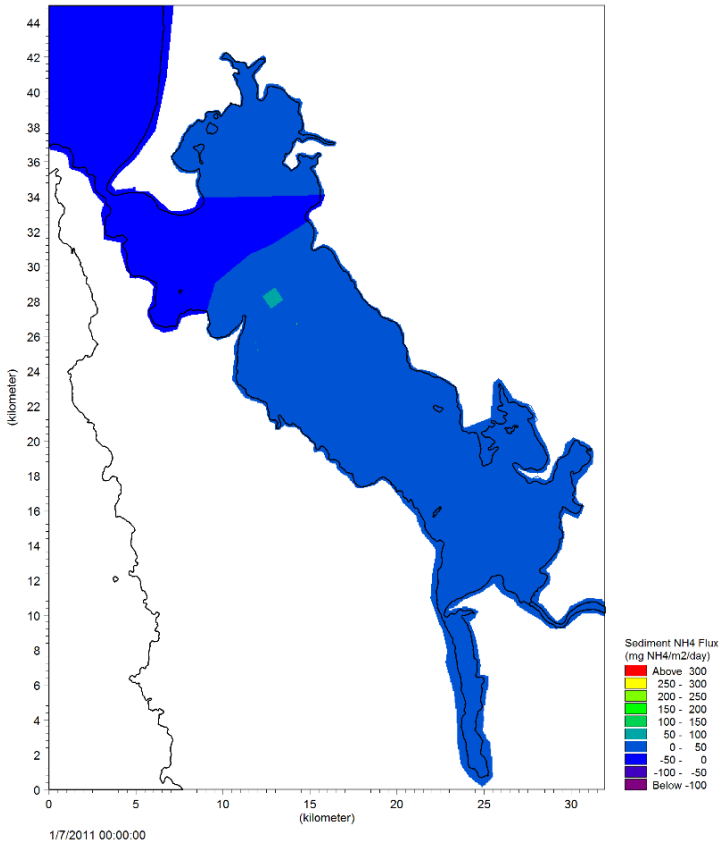


Figure 4-19 Map of benthic fluxes of Ammonia generated using incubation data (Calibration)

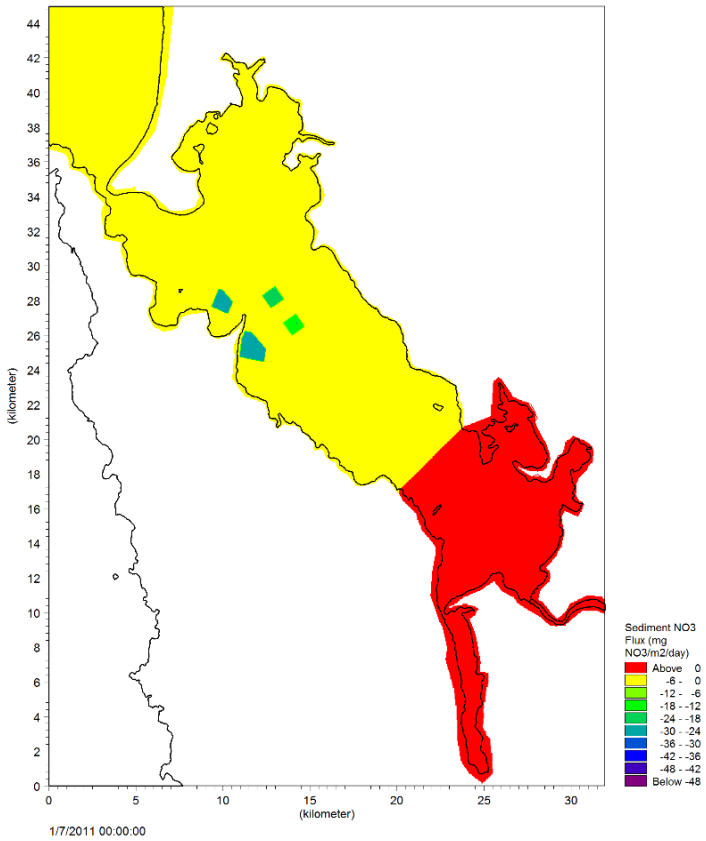


Figure 4-20 Map of benthic fluxes of Nitrate generated using incubation data (Calibration)

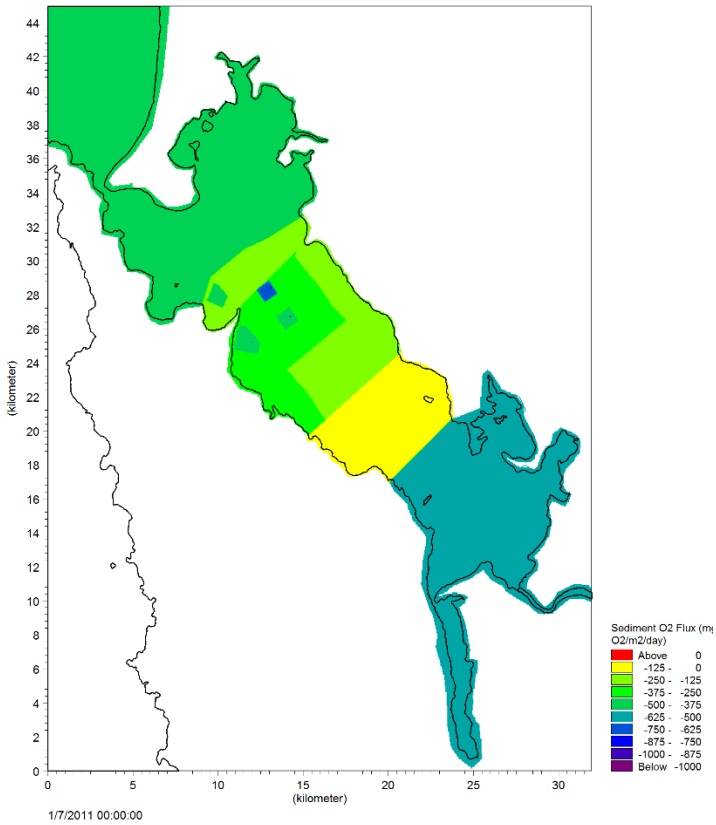


Figure 4-21 Map of benthic fluxes of oxygen generated using incubation data (Calibration)

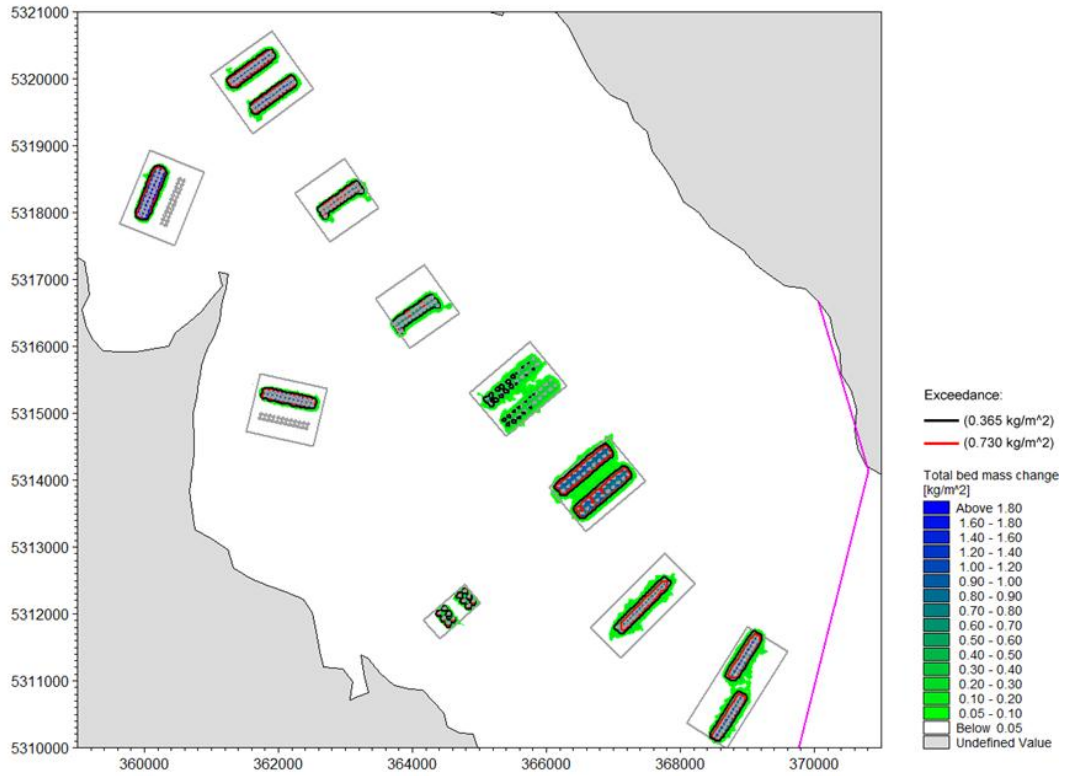


Figure 4-22 Example of the faeces deposition results (mass of Carbon per area, gC/m2/day) for 1 year of the Recalibrated Model.

Figure 4-24 and Figure 4-24 show examples of the model calibration at king river (KR1) and cosy corner (CC) respectively. Model results are shown against measured data over a period of one year. Generally the model calibration for nitrate and ammonium was very good. At times the model did tend to overestimate these nutrients but was able to readjust with time.

The new version of the sediment-water fluxes in the recalibrated model was a significant step forward in the biogeochemical model: it provided actual rates measured *in situ* with a degree of spatial variation in the place of previously used literature values from similar environments. This, along with the other improvements made by the refining of processes governing biogeochemical cycling in the Harbour provided a successful recalibration with the new water quality data collected from October 2011-September 2012.

However, no matter how accurate, model predictions alone cannot be considered sufficient in order to carefully manage the sustainability of a complex environment such as Macquarie Harbour. Currently, the modelling is based on a set of assumptions regarding the processes governing the biogeochemical interactions in the Harbour, and this set of parameters fit the collected data for the period modelled. However, in such a unique environment, it is possible that the processes dominating the ecology and hydrodynamics in the Harbour may change based on a set of external factors outside the realm of current knowledge. Farming is also not the only factor susceptible to change within the Harbour, and as such the management of this asset will be based on a sound adaptive management strategy. Continued monitoring will provide knowledge of the Harbour as farming evolves along with the other external factors. Any shift in the conditions of the Harbour can then be highlighted and the situation re-assessed if specific changes occur that need special attention. The model could potentially be revised in the future to address these changes and provide predictions along with desktop analysis of shifts in governing processes.

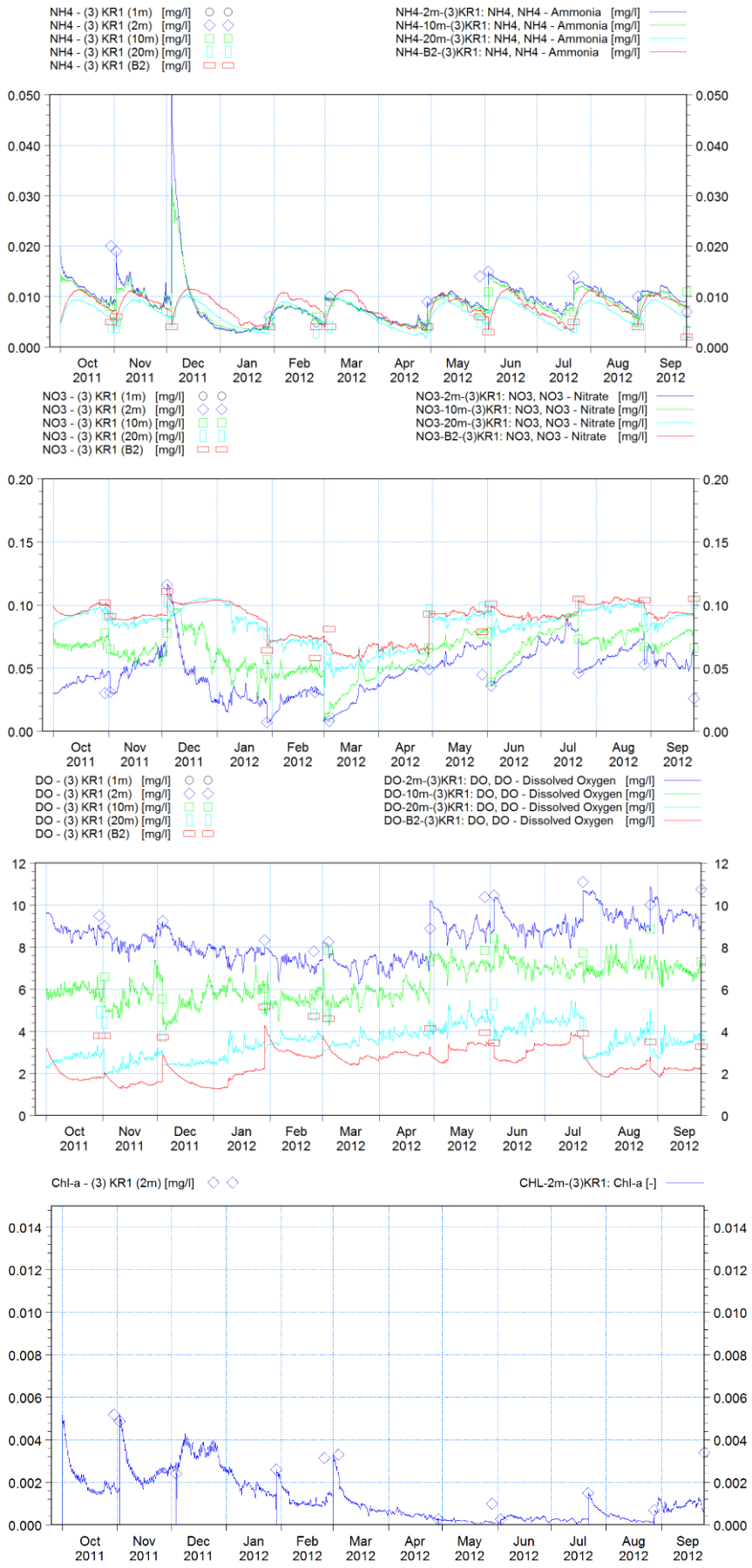


Figure 4-23 Calibration plots for Ammonia (NH4), Nitrates (NO3), Dissolved Oxygen (DO) and Chlorophyll-a (Chl-a) at the KR1 station. Monthly monitoring measurements (discrete markers) are shown against model results (solid lines).

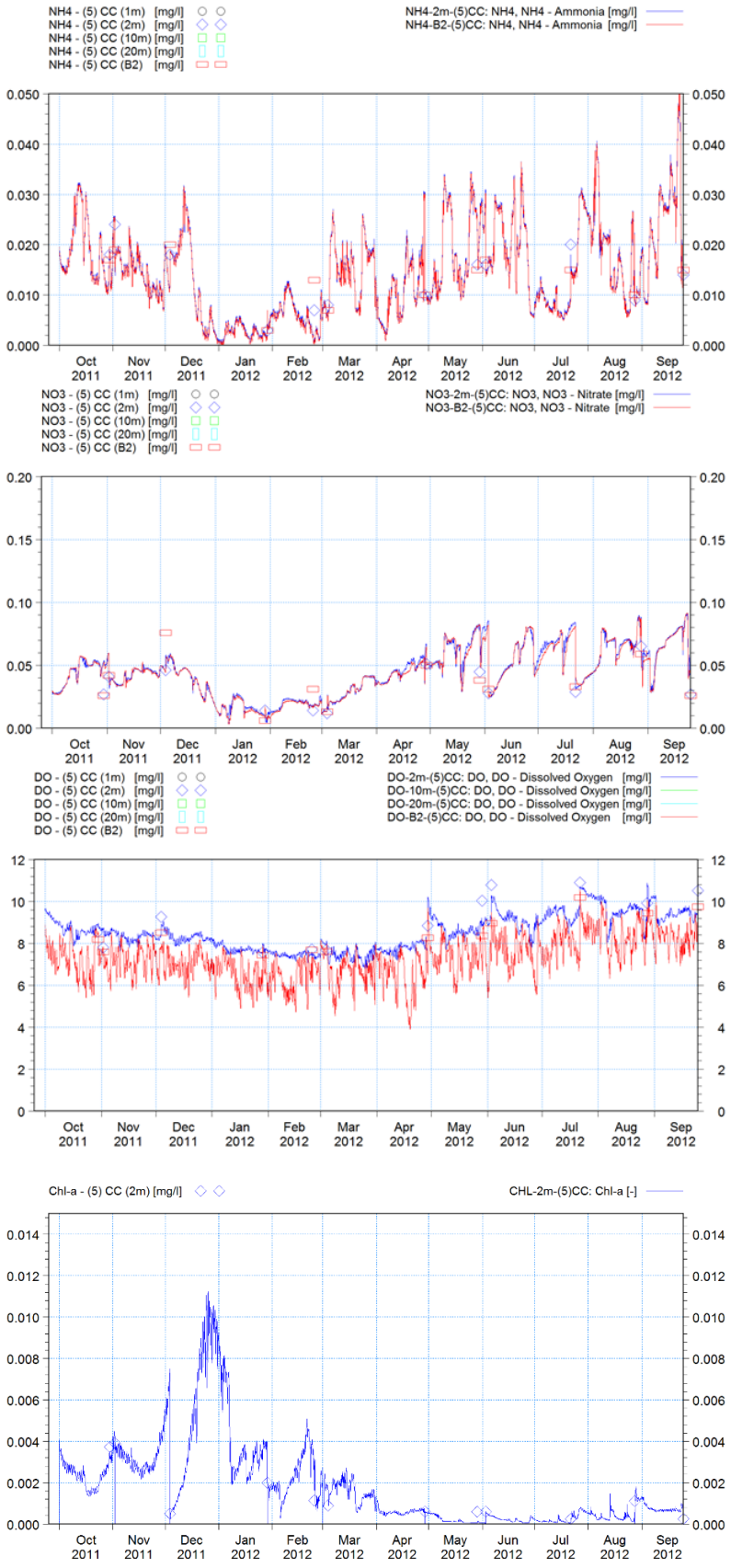


Figure 4-24 Calibration plots for Ammonia (NH4), Nitrates (NO3), Dissolved Oxygen (DO) and Chlorophyll-a (Chl-a) at the CC station. Monthly monitoring measurements (discrete markers) are shown against model results (solid lines).

# 5 Conclusion

The results of this study demonstrate the importance of organic enrichment and dissolved oxygen levels in regulating the cycling of nutrients between the seabed and bottom waters in Macquarie Harbour. The Macquarie harbour water column is naturally highly stratified and low dissolved oxygen concentrations in bottom waters are common. With little oxygen to penetrate the sediments, the results indicate that nitrification (the process by which ammonia is converted to nitrate in oxic conditions) in the sediment is largely inhibited, and as such, denitrification (the process by which nitrate is converted to nitrogen gas in anoxic conditions and permanently removed from the system) must rely on sourcing nitrate from the water column rather than from nitrate produced in the sediments via nitrification. At farmed sites subject to elevated levels of organic enrichment there is a concomitant increase in the rates of nitrate uptake, ammonia release and nitrogen gas production via denitrification. In terms of denitrification efficiency, the proportion of dissolved inorganic nitrogen converted to nitrogen gas and effectively removed from the system, the enriched sites are far less efficient with a greater proportion of the nitrogen returning to the water column as ammonia.

At different stages of farming activity, sediment function responded in a manner consistent with what might be expected during fallowing and stocking at some, but not all sites. This suggests that factors other than stocking and bulk feed inputs (e.g. changes in diet, conversion ratios, feeding regimes, bottom water conditions) are also playing a significant role in regulating sediment loads and the processing of organic matter.

The new version of the ecological model was a significant step forward because it provided actual benthic process rates measured *in situ* with a degree of spatial variation in the place of previously used literature values from similar environments. This, along with the other improvements made by the refining of processes governing biogeochemical cycling in the Harbour provided a successful recalibration with the new water quality data collected from Oct 2011-Sep 2012.

It is important to acknowledge that the modelling is based on a set of assumptions regarding processes that define the biogeochemical interactions in the harbour and that these assumptions provided a reasonable fit to the Oct 2011-Sep 2012 observations. However, it is possible that the processes dominating the ecology and hydrodynamics of the harbour may change based on external factors outside the realm of current knowledge. With this in mind it is clear that continued monitoring comprises an essential component of a sound adaptive management strategy, and that any shift in conditions or change in system understanding observed through such monitoring should require the model to be revised for future predictions. The very low bottom water dissolved oxygen conditions recorded in the final two surveys (May and September 2013) may indicate such a shift in conditions.

The study also highlighted the importance of anaerobic pathways of organic matter mineralisation in Harbour sediments, particularly at enriched farm sites. However, the fate of the reduced compounds produced during anaerobic respiration is difficult to determine based on benthic flux incubations and warrants further research. This is particularly important given the very low bottom water oxygen conditions recently observed, and the potential for the reduced compounds to be reoxidised in bottom waters creating a further oxygen demand.

# 6 Implications

This study has provided an improved understanding of sediment function and the benthic response to farming in Macquarie Harbour. Prior to this study, there had been no measures of benthic nutrient cycling for this system. The subsequent inclusion of local data on sediment-water column interactions rather than literature values has improved the reliability of the environmental model used by industry and government to assist decision making in the harbour. Nonetheless, it is also acknowledged that modelling is inevitably based on a number of assumptions and current knowledge of the processes governing the biogeochemical interactions in the Harbour. The current set of parameters and assumptions fits reasonably well to the collected water quality data (Oct 2011-Sep 2012), however, in such a unique environment it is possible the processes dominating the ecology and hydrodynamics of the harbour may change based on external factors outside the realm of current knowledge. With this in mind it is clear that continued monitoring comprises an essential component of a sound adaptive management strategy, and that any shift in conditions or change in system understanding observed through such monitoring would require the model to be revised for future predictions. The very low bottom water dissolved oxygen conditions recorded in the final two surveys (May and September 2013) may indicate such a shift in conditions.

The study has also reinforced the importance of farm based management as a means to ensure that sediments recover after/ between farming cycles and can function most efficiently when subject to significant enrichment. However, the results suggest that the ecology and recovery dynamics in Macquarie Harbour are unique and further work is required to fully understand the drivers and management indicators of sediment function and recovery for farm affected sediments in the harbour.

From a monitoring and management perspective, bulk identifiers of organic matter source (C:N ratio and $\delta^{13}C$ and $\delta^{15}N$ signature) together with measured rates of respiration appear to be good environmental indicators of the footprint of farm derived organic matter and sediment function respectively.

# 7 Recommendations and further development

Whist the work has improved our understanding of sediment function in Macquarie Harbour and led to improvements to the environmental model used to assist decision making in Macquarie Harbour, it has also identified a number of knowledge gaps that warrant further investigation as follows:

- The study highlighted the importance of anaerobic processes and the production of reduced compounds in benthic biogeochemistry of the harbour. If these reduced compounds are reoxidised in bottom waters the concomitant oxygen demand is not likely to be fully accounted for in benthic core incubations. The very low bottom water oxygen conditions in the final two surveys highlight the importance of understanding the major drivers of oxygen dynamics in bottom waters; the potential role of reduced compounds warrants investigation.
- Measures of sediment function at some but not all sites showed patterns consistent with expectations during fallowing and stocking. This suggests that drivers other than stocking (e.g. changes in diet, conversion ratios, feeding regimes, bottom water conditions) are playing a significant role in determining sediment condition. A greater understanding of the drivers of sediment function in response to different stages of farming activity is likely to improve the effectiveness of farm based management of stocking and fallowing regimes in Macquarie Harbour.
- In the second half of the study a significant decline in bottom water oxygen conditions was evident. The causes of this decline and the implications for broader ecosystem dynamics warrants further investigation, and as such, may require the model to be revised for future use.

# 8 Extension and Adoption

Both industry and government have been involved in this project from the outset, and as such have been provided with the findings to use in the adaptive management process for fish farming in Macquarie Harbour as the project has evolved. This includes access to the outputs from the re-calibrated environmental model. The study findings have also being used to inform individual company's environmental certification processes. The project findings will also be provided to government and industry as a final report for their information.

The extension of the project to look at sediment responses to farming practice has led to ongoing interactions with managers of the farms used in the study. As a consequence they have responded to information on sediment function and the observed response to farming practices as they became known. Notably, not all sediment responses to fallowing and stocking were consistent with expectations gained from farming in other regions. This suggests that drivers other than stocking (e.g. changes in diet, conversion ratios, feeding regimes, bottom water conditions) are playing a significant role in determining sediment condition in Macquarie Harbour. To understand these drivers, further work is included in an FRDC proposal currently under review.

## 8.1 Project coverage

There has been no media coverage of the results/findings to date. A summary of the results and findings have been presented at 2 international conferences; The Aquaculture Elsevier Conference, Canary Islands 2013, World Aquaculture Society, Adelaide 2014

*Abstract*

Resilience of Key Benthic Processes under Cage Aquaculture

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Cage aquaculture in marine systems can result in organic enrichment of the benthos, with excess feed and waste products altering the ecology and processes that determine the ultimate fate of nutrients. The capacity of sediments to efficiently process organic matter depends on the spatial and temporal dynamics of fish farm inputs and the nature of the receiving environment.

In coastal and estuarine ecosystems, understanding the processes that determine the fate of nitrogen is critical because excessive nitrogen can lead to coastal eutrophication. Although removal of nitrogen can occur via transport of nutrient-rich water offshore, in many water bodies the key ecosystem processes that determine the fate of nitrogen, including effective removal to the atmosphere via the process of denitrification, occur in the sediments. Understanding the response of nitrogen transformation pathways to farming operations, in particular the sediments' ability to recover such that the capacity for efficient removal of nitrogen is not irreversibly affected, allows for farm-based management of inputs which can optimise farming productivity and sustainability.

Macquarie Harbour in southwest Tasmania is a highly stratified, tannin rich system with a low residence time and naturally low bottom water dissolved oxygen conditions. This study assessed the impacts of organic enrichment from salmon farming on nitrogen cycling processes in the sediments including the capacity to recover during fallowing. This was achieved via sediment water column incubations conducted at farm and harbour scales repeated across seasons and farm production cycles. In this talk I will present results that demonstrate the effects of enrichment on key nitrogen cycling process including nitrification, denitrification, dissimilatory nitrate reduction and anammox and the resilience of the these processes during fallowing. The implications of these results for management at both farm and whole system scales will then be discussed.

# 9 Appendices

## 9.1 List of researchers and project staff

Dr Jeff Ross – Chief-Investigator, IMAS

Dr Catriona Macleod – Co-Investigator, IMAS

Neil Hartstein– Co-Investigator, DHI

Perran Cook - Co-Investigator, Monash University

Joe Valentine - Co-Investigator, Aquenal Pty Ltd.

Dr Vanessa Lucieer – Co-Investigator, IMAS

Bronagh Kelly - Research Assistant, IMAS

Adam Davey - Research Assistant, IMAS

Andrew Pender – Technical Assistant, IMAS

Malinda Auluck – Masters student, IMAS

Lance Searle, Petuna

Matt Barrenger, Tassal

Dom O'Brien, Huon Aquaculture Company

Graham Woods, DPIPWE. Marine Farming Branch

Eric Brain, DPIPWE. Marine Farming Branch

Adam Main, TSGA Executive Officer

## 9.2 Intellectual Property

There is no specific IP associated with this project.

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