Final Report

Aquafin CRC

A whole-of-ecosystem assessment of environmental issues for salmonid aquaculture

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(FRDC Project No. 2004/074)

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ASAS
Atlantic Salmon
AQUACULTURE
SUBPROGRAM

CSIRO

Australian Government
Fisheries Research and
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University of Tasmania

A whole-of-ecosystem assessment of environmental issues for salmonid aquaculture

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Appendices

Abbreviations

ANOVA	Analysis of Variance
Aquafin CRC	Cooperative Research Centre for the Sustainable Aquaculture of Finfish
CMR	CSIRO Marine Research
CMAR	CSIRO Marine and Atmospheric Research (from July 1st, 2005)
CRQ	Community respiration quotients = TCO_2/O_2 flux ratio
CSIRO	Commonwealth Scientific and Industrial Research Organization
DBL	diffusive boundary layer
DIN	dissolved inorganic nitrogen
DO	dissolved oxygen
DPIW	Department of Primary Industries and Water (from April 5th, 2006)
DPIWE	Department of Primary Industries, Water and Environment
ESD	Ecologically Sustainable Development
FRDC	Fisheries Research and Development Corporation
HAB	Harmful algal bloom (phytoplankton)
HES	Huon Estuary Study (Environmental Research for Integrated Catchment Management and Aquaculture – FRDC project No. 96/284)
MECO	Model for Estuaries and Coastal Ocean
OM	Organic matter
PAR	photosynthetically active radiation
RPI	relative preference indices
SCFA	Standing Committee on Fisheries and Aquaculture
STP	Sewage treatment plant
TAFI	Tasmanian Aquaculture and Fisheries Institute
UTas	University of Tasmania

Non-Technical Summary

2004/074	Aquafin CRC - FRDC Salmon Subprogram: A whole-of-ecosystem assessment of environmental issues for salmonid aquaculture

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Objectives

This project had the following objectives:

1. Identification, characterisation and modelling of the key oceanographic and ecological features of the Huon Estuary and D'Entrecasteaux Channel and how these may affect or limit salmon cage farming, together with an assessment of possible industry responses.
2. Inventory of the sources of nutrients in this region, including those from salmon farms, their spatial and temporal variation, nutrient cycling, and impacts on pelagic and benthic production.
3. Definition of the factors driving the phytoplankton ecology of this region, especially interactions among phytoplankton and zooplankton (including jellyfish).
4. Determination of the role of carbon remineralisation in sediments with nutrient release into the water column in relation to the varying spatial and temporal environmental conditions.
5. Design of a new monitoring system and adaptive management strategy for use by industry and DPIWE together with definition of associated indicators and standards

Objective	Chapter	Technical Reports *
1	2, 3, 7	Herzfeld, M. (2008). Numerical hydrodynamic modelling of the D'Entrecasteaux Channel and Huon Estuary: Phase II.
2	3	Wild-Allen, K. (2008). Huon Estuary and D'Entrecasteaux Channel biogeochemical model scenario simulations for 2002 and 2009: farm impacts on seasonal pelagic biogeochemistry. Data are included in other reports.

3	**4, 5, 7**	Bonham, P., Rousseaux, C. and Thompson, P. (2008). Effects of grazing by microzooplankton on phytoplankton in the Huon Estuary and D'Entrecasteaux Channel. Clementson, L.A., Blackburn, S.I., Berry, K.M. and Bonham, P.I. (2008). Temporal and spatial variability in phytoplankton community composition in the mouth of the Huon River Estuary. Clementson, L.A., Blackburn, S.I., Thompson, P.A., Berry, K.M. and Bonham, P.I. (2008). Phytoplankton community composition during spring blooms in North West Bay and Port Esperance, Tasmania. Swadling, K.M., Macleod, C.K., Foster, S., and Slotwinski, A.S. (2008). Zooplankton in the Huon Estuary and D'Entrecasteaux Channel: community structure, trophic relationships and role in biogeochemical cycling.
4	**6**	Holdsworth, D.G., Revill, A.T., Volkman, J.K. and Swadling, K. (2008). Lipid biomarkers in sediment traps and sediments from North West Bay, Tasmania. Macleod, C., Revill, A., Volkman, J. and Holdsworth, D. (2008). Characterisation of the benthic environment of the D'Entrecasteaux Channel and Huon Estuary. Revill, A.T., Holdsworth, D.G., Volkman, J.K. and Swadling, K. (2008). Fluxes of organic matter and lipids to sediments in the Huon Estuary, Tasmania.
5	**8, 9**	Macleod, C. and Foster, S. (2008). Evaluation of selected alternative approaches for monitoring nutrient enrichment associated with caged marine salmonid aquaculture in Tasmania. Thompson, P., Wild-Allen, K. Macleod, C. Swadling, K. Blackburn, S. Skerratt, J. and Volkman, J. (2008). Monitoring the Huon Estuary and D'Entrecasteaux Channel for the effects of finfish aquaculture.

*** The technical reports are provided as pdfs on the accompanying CD. For a listing see Appendix 4 (page 197).**

This project has established a detailed set of data that provides a good picture of the environmental conditions in the Huon Estuary and D'Entrecasteaux Channel, where two-thirds of Tasmania's salmonid industry is located. These show generally good environmental conditions with occasional periods of high phytoplankton abundance and low dissolved oxygen. These data have been used to calibrate and validate sophisticated 3-dimensional hydrodynamic and biogeochemical models of the region that capture the main physical and biological processes.

Specific process studies in sub-regions of the domain have been carried out to shed more light on particular aspects of the nutrient-phytoplankton-zooplankton relationships and sediment biogeochemistry. Our measurements and modelling results indicate that the salmonid industry is a significant contributor of nutrients to this region and that these have led to measureable increases in phytoplankton abundance. As a result, the industry is reaching a size where significant further expansion in the same region could lead to deleterious environmental effects. A monitoring program has been devised that will allow any effects to be identified so that appropriate action can be taken to ensure a healthy and sustainable salmonid industry in southeast Tasmania.

A hydrodynamic modelling approach has been developed which uses a series of embedded models. A high resolution model was developed for the Huon Estuary and D'Entrecasteaux Channel which used boundary conditions from an intermediate model, which, in turn, was nested in a larger regional model that was driven with products derived from global models or altimetry and climatology on its open boundaries. The intermediate hydrodynamic model was calibrated to temperature and salinity data collected in the field at the northern and southern entrances to D'Entrecasteaux Channel and the mouth of the Derwent Estuary from December 2003 to March 2004. A coarse-scale local model suitable for coupling with biogeochemical and sediment transport modules was used to investigate the longer term trends of environmental indicators in response to aquaculture farming practices.

High resolution hydrodynamic models were also developed to investigate the mixing zone characteristics around a fish farm cage. These analyses revealed that a continuous release of tracer from the cage results in a plume emanating from the cage in the form of a long narrow ribbon. The position and concentration of the plume is dependent on the *in situ* flow conditions, and possesses large variability in space and time. It is possible for a plume having concentration of 10% (i.e. 10:1 dilution) of the source concentration to exist up to 0.5 km from the cage on occasion. Particle tracking was used to investigate the 'footprint', or spatial distribution of particles in the sediment, for various settling velocities of particles. For settling rates typical of feed pellets or faecal material, the footprint is restricted to the cage site. However, a settling rate of an order of magnitude less than this results in footprints in the form of an elongated plume with typical length ~0.5 km and width ~200 m emanating from the cage site. The direction of this plume is controlled by sub-surface currents, and may be in the opposite direction to surface currents.

A detailed picture of the phytoplankton ecology of this region and the factors that control phytoplankton abundance has been established. Major changes in the plankton ecology have been observed, including the establishment in 2002 of the heterotrophic dinoflagellate *Noctiluca scintillans* as a major herbivore in this ecosystem. *N. scintillans* is believed to have arrived in Tasmania in association with a strengthening of the East Australian Current. This species accounted for up to 20% of mesozooplankton abundance in autumn and it is capable of both suppressing zooplankton abundance and reducing the sedimentation of faecal pellets to the seafloor. The dominant phytoplankton species in the ecosystem changed dramatically from 1997 to 2003 with significant increases in dinoflagellates of the genus *Ceratium*, small flagellates, and species of the diatom genus *Skeletonema.* The abundance of the dominant toxic dinoflagellate, *Gymnodinium catenatum*, has fluctuated enormously. We have obtained evidence that *N. scintillans* will feed on *G. catenatum*, with the potential to influence the intensity and extent of blooms. Process studies in the Huon Estuary and North West Bay have demonstrated the rapid changes in phytoplankton dynamics that can occur due to changes in physical and chemical parameters over time scales of hours. The involvement of particular bacteria, *Alcanivorax* and *Marinobacter* spp. in the growth and bloom development of *G. catenatum* has been demonstrated for the first time. Studies of the mortality of the blooms show a slower decline of autumn blooms compared with summer blooms.

New information on mesozooplankton community composition and structure has been obtained for sites throughout the Channel, Huon and North West Bay from November 2004 to October 2005. The composition of this mesozooplankton community is typical of inshore, temperate marine habitats, with seasonally higher abundance in summer and autumn and lower numbers in winter and spring. Copepods were the largest contributors to total abundance across all seasons and stations, while cladocerans and appendicularians were proportionally abundant in spring and summer. The faecal pellets of these three main groups, along with those of krill and amphipods, also contributed significantly to material sedimenting to the bottom sediments as shown by sediment trap deployments.

The grazing impacts of microzooplankton and mesoplankton on the phytoplankton communities were examined during several process studies. Microzooplankton grazers included high proportions of ciliates, heterotrophic dinoflagellates and tintinnids, with smaller numbers of copepod nauplii, appendicularians and rotifers. Grazing rates of microzooplankton reached as high as 96% of daily primary production, while that of mesozooplankton herbivores was never greater than 20%.

The particle size distribution in sediments has been mapped throughout the Channel, Huon and North West Bay. Faunal populations were broadly similar throughout the main part of D'Entrecasteaux Channel and lower Huon Estuary, but there was a clear distinction between communities associated with lower salinity sites from those at fully marine sites. These community distinctions do not appear to be specifically correlated with differences in depth or sediment type either individually or in combination.

Mesocosm experiments were carried out to understand the role of sediments in organic matter degradation and release or uptake of nutrients. Sediment respiration rates in the Huon and lower D'Entrecasteaux Channel were relatively low and showed evidence for microphytobenthic activity at some sites, whilst sites in the Upper Channel appeared more productive, with greater oxygen draw-down over time. Benthic effluxes of nutrients were low in the Huon Estuary. On all occasions, the DIN flux was dominated by nitrate, which was always released from the sediment to the overlying water. Aerobic conditions were prevalent in the upper 3–9 mm for the sediment, providing an environment conducive for aerobic processes like nitrification.

Organic loading of these sediments with labile carbon has shown that the nitrification-denitrification pathway is susceptible to being overwhelmed, due to increased respiration and reduction in the pool of oxygen in the sediments. This results in large amounts of ammonium being released into the water column and therefore makes the estuary more susceptible to blooms of algae. Therefore, the maintenance of 'healthy' sediments is seen as important to the long-term sustainable use of the estuary.

A number of alternative measurement strategies for eutrophication were examined during the course of the project including measurements of fouling of ropes and plates suspended in the water column. There were marked differences in fouling communities between plates and ropes suggesting that ropes would not be a useful monitoring approach. We also found different communities between sites, strong temporal (seasonal) differences, increased numbers of species in summer, and an increased number of individuals and species at aquaculture sites relative to other sites during winter sampling. *Ulva* was a common algal fouling species at all times of year and so we investigated its growth as a measure of nutrient production.

A major success of the project was the development of a three-dimensional coupled hydrodynamical, sediment and biogeochemical model to evaluate the environmental impact of salmonid fish farms in the Huon Estuary and D'Entrecasteaux Channel. The model simulates the seasonal cycling of organic and inorganic carbon, nitrogen, phosphorus and oxygen through multiple phytoplankton, zooplankton, nutrient and detrital pools and was validated against field observations collected in 2002. Model analysis of scenario simulations, parameterised without fish farm inputs, with fish farm inputs for 2002 and with projected inputs for 2009, allowed us to identify the system-wide spatial and temporal environmental footprint of the industry which need to be assessed relative to environmental guidelines. Various statistical metrics were devised to highlight the spatial and temporal effects of this increased production. These results were communicated to the salmonid industry, the regulatory agency DPIW, the Marine Farming Planning Review Panel and to the Aquafin CRC board.

The projected 2009 scenario simulation shows larger, more prolonged and wider spatial impact of farm loads on pelagic nutrient concentrations, chlorophyll and bottom water oxygen saturation than in 2002 and a shift from oligotrophic to mesotrophic conditions over ca. 54% of the region. These results are consistent with the elevated nutrient discharge simulated in 2009 which was roughly 3 times 2002 values. Whilst farm expansion in the D'Entrecasteaux Channel was significantly larger than the expansion in the Huon Estuary, farm impacts were consistently high in the Huon Estuary, suggesting this region is more susceptible to farm waste. This could be because the Huon Estuary has a slower flushing time compared with the D'Entrecasteaux Channel and/or because it accumulates farm waste from elsewhere due to the residual and estuarine circulation. Other regions consistently impacted include the northern D'Entrecasteaux Channel, North West Bay and Barnes Bay. These regions are located in the north of the region and downstream of most of the farm leases with respect to the residual south to north circulation. Port Esperance and Great Taylors Bay in the south were also affected by farm loads, likely due to local discharge in excess of flushing.

A long-term monitoring strategy for the industry in the D'Entrecasteaux Channel and Huon Estuary has been developed to ensure the sustainability of the salmonid industry. Our major goal was to design a monitoring program with the capacity to detect the effects of those processes judged to be most threatening to the Huon and

D’Entrecasteaux ecosystem at the whole-of-ecosystem level. The monitoring program we have proposed is designed to provide knowledge of how well the ecosystem is functioning with an increased nutrient load and to allow any significant temporal trend(s) in ecological indicators to be detected. A major challenge was converting indicators of ecological condition into recommended quantitative performance measures that can be used in a regulatory manner to adaptively manage the ecosystem. These recommendations have been extensively discussed with industry and DPIW representatives and refined through an iterative process. Agreement has been reached in principle about the major elements of the plan and work is underway to implement the strategy.

Near the end of the project a decision was made by the marine farming planning review board to impose a limit on the salmonid industry in the Huon and D’Entrecasteaux based on the industry’s projections for salmon production in 2009. This decision was informed in part by the scenario of likely effects on phytoplankton populations produced by our project using the environmental modelling suite. If the industry is to expand further it will either have to demonstrate that it is not having serious environmental impacts (which can only be achieved by a comprehensive monitoring program of the type suggested by our project), or place new production outside of the Huon Estuary and D’Entrecasteaux Channel or find ways to reduce or ameliorate nutrient inputs (e.g. by polyculture although our work shows that this will not assimilate all the nutrients produced by the salmonid industry).

The option of expanding to the seaward side of Bruny Island is already under consideration by the industry which has ambitions for a doubling of production compared with 2007 figures. While the region off-shore Bruny Island is better flushed and larger in extent than the Huon and D’Entrecasteaux region the same general environmental issues will apply and an environmental risk assessment will be needed. There is the added complication of connection with the Derwent estuary and any assessment of nutrient effects must take into account efforts already underway in the estuary to reduce nutrient inputs from sewage treatment plants. There is thus a need to consider industry expansion in the context of multiple use management of this ecosystem. A new umbrella concept called INFORMD (Inshore Network for Observation and Regional Management: Derwent-Huon) which links research activities carried out by CSIRO and TAFI with the needs of stakeholders will assist this.

In summary, the project has generated considerable knowledge of the possible environmental effects of salmonid aquaculture at an ecosystem level and led to the development of the monitoring and modelling tools that can be applied to other fish species or to other regions. Our research has been materially assisted by the industry’s support for this project and their demonstrated commitment to ensuring the environmental sustainability of their operations. Also, the involvement of staff from the Tasmanian Department and Primary Industries and Environment (DPIWE, now DPIW) Marine Farming Branch was essential to ensure that our research was relevant to the effective regulation of salmonid farming in Tasmania.

Chapter 1

Introduction and Background to the Study

Finfish aquaculture in Australia is strongly supported by the State Governments of Tasmania and South Australia and it is generally well accepted by the general public in those States. A strong attraction is its wealth-creation and provision of employment in regional areas. The Atlantic salmon and southern bluefin tuna (SBT) finfish industries, Government regulators in Tasmania and South Australia and the Aquafin CRC agreed on the need to be proactive and work together to understand farm-environment interactions at system-wide scales to ensure the long-term sustainability of these industries and to minimise the risk of environmental controversies in Australia of the type that have become common overseas.

The salmonid aquaculture industry depends on a healthy and suitable marine environment to maintain production and profitability. Over recent years, the industry in Tasmania has faced a number of environmental challenges such as phytoplankton blooms including the recent occurrence of the red-tide alga *Noctiluca*, occasional periods of very warm waters which reduce growth rates, high salinities and episodes of low oxygen and reduced water quality. Salmon farms remain a significant point source of nutrients into the marine environment at a time when other sources such as sewage treatment plants (STPs) are reducing their loads. While stocking densities used in Tasmania are generally lower than those overseas, expansion of the industry is put at risk without an accurate idea of the amount of fish that can be stocked (i.e. the carrying capacity of the ecosystem) which permits both a healthy and productive industry and at the same time protects marine environmental values.

Evaluation of this combination of issues and risks constitutes a significant scientific challenge. System-wide environmental effects such as the frequency and composition of phytoplankton blooms and jellyfish swarms, and possible effects on benthic communities may affect industry production and profitability. Equally, regulators and the public need assurance that marine ecosystems will not undergo unacceptable environmental change as industry develops. Industry and managers require a capability to resolve and predict environmental effects due to changes in the adjacent offshore ocean, for example due to climate change, changes in catchment loads, and effects due to the industry itself. Advances in observation technologies, in scientific understanding, and in modelling capability are needed to underpin both long-term planning and short-term operational decisions. The goal of this CRC project was to work with industry and regulators to provide an environmental information and prediction system which allows each to manage environmental risk.

In Tasmania, salmonid aquaculture is regulated through the Department of Primary Industries and Water (DPIW) with specific proposals heard by the Tasmanian Aquaculture Review Panel. DPIW expressed a particular need to understand the nutrient budget in salmon-growing areas as part of their assessment of sustainable yields. Salmon farmers expressed a specific industry need for early warning of the advent of a phytoplankton bloom, and early warning of the likely level of threat of the bloom.

This project is part of the Environment Research Program of the CRC for Sustainable Aquaculture of Finfish ("Aquafin CRC"), and employed funds invested out of the CRC's Commonwealth grant and by FRDC and other Participants of the CRC to address environmental issues affecting the Atlantic salmon and southern bluefin tuna finfish industries. The initial 3.5 years of research were carried out as part of project 4.2 "System-wide issues for sustainable salmonid aquaculture" (Volkman et al., 2006). The present report describes research carried out as a continuation of this project as project 4.2(2) entitled "A whole-of-ecosystem assessment of environmental issues for salmonid aquaculture".

The first phase of the CRC salmon-related research (Project 4.1, FRDC 2000/164) provided considerable knowledge about the effects of stocking and fallowing on sediment health as measured by benthic fauna, microbiology and biogeochemical factors (Macleod et al., 2004). Extension of this work to industry and regulators was achieved in the form of a handbook and workshops in 2004. With the completion of this work, it was agreed to place more emphasis on environmental issues at the system-wide scale, particularly those associated with possible eutrophication. In particular, the overall assimilative capacity of the region in response to release of nutrients, organic matter and associated changes in ecosystem function was to be assessed.

The emphasis on eutrophication derives from the FRDC-funded Huon Estuary Study (Environmental Research for Integrated Catchment Management and Aquaculture - Project No. 96/284; abbreviated to HES hereafter; Butler et al., 2000) which concluded that the Huon Estuary could sustain a doubling of the 1997 salmon farming level with some risk of increased summer algal blooms while a quadrupling would put the system at substantial risk of prolonged algal blooms due to nutrient inputs. Salmon farming in the Huon was then voluntarily capped and production expanded to neighbouring waters in the D'Entrecasteaux Channel. In 2003, algal blooms have been recognised as a major concern by salmon farmers and thus we have put additional emphasis on this topic.

Approximately 30% of the nutrients added through fish feed are removed from the marine ecosystem as fish at salmon harvest, and the remainder are released to the environment as metabolic wastes and uneaten feed throughout the stocking cycle. About 80% of the total nutrient losses from fish farming are dissolved and in a form readily available to macroalgae, microalgae and plants such as seaweeds. The released nutrients may change ambient N/P ratios and can alter the ratio of key phytoplankton species (e.g. from diatoms to dinoflagellates). Eutrophication will become a problem if the assimilation capacity of a salmon-growing area for nutrients is exceeded. This could lead to phytoplankton blooms, which may include harmful species (HABs), or be manifested as increased production of benthic microalgae, macroalgae or epiphytic algae. Effects of increased nutrients can also be seen at higher trophic levels, by altering the abundance and species composition of zooplankton, which in turn can affect fish communities.

Possible effects of increased nutrients into the Huon and D'Entrecasteaux Channel were assessed using ecological and physical indicators, including measurements of nutrients, dissolved oxygen (DO) and chlorophyll, in the first phase of the project (Volkman et al., 2006). Many of the potential ecological indicators investigated showed high levels of natural variability. Phytoplankton species composition and abundance is clearly an important indicator of ecosystem health. In project 4.2(2), results collected from the HES study and the CRC Broadscale monitoring project (CRC 4.4) were reassessed for trends. In addition, a number of specific process studies were carried out in sub-regions of the domain to investigate in more detail specific ecological processes such as zooplankton grazing that had not been considered in phase 1. A major goal of this second phase was to further develop our biogeochemical model of the domain so that environmental effects at the system-wide level could be captured and the model used to run scenarios investigating effects of future loads.

Excess nutrients may also result in increased macroalgal growth. Seasonal monitoring of intertidal macroalgal composition using direct measurements and cliff top photography in Phase 1 had shown their potential as simple monitoring tools for eutrophication, but seasonal variation in abundance and community composition limit its applicability. Alternative monitoring techniques such as settlement plates were investigated as indicators of nutrient enrichment.

Zooplankton play an important role in biogeochemical cycling, but are not generally considered as primary indicators of eutrophication. Very little is known about zooplankton from the Huon Estuary and D'Entrecasteaux Channel region, apart from some baseline information in the MSc. study of Cazassus (2003) who examined zooplankton communities on the east coast of Tasmania. Zooplankton directly affect the material fluxes between particulate and dissolved organic matter by modifying sinking particulate fluxes and cycling inorganic and organic materials throughout the water column. Zooplankton feeding can also result in major changes to phytoplankton abundance and species composition. These processes need to be understood for improved calibration of the biogeochemical models. Specific manipulative experiments will be conducted to further understand the effects of zooplankton grazing on primary production.

Locally a number of large blooms of zooplankton have been reported in recent years; most notably the large jellyfish blooms in 2000 and 2002 and blooms of salps, ctenophores and other smaller zooplankton. These blooms can significantly affect the productivity of the system and therefore need to be incorporated into any attempt to understand or model ecosystem change.

DPIW and industry agreed to adopt an adaptive management approach to regulation and management of system-wide environmental effects of salmon farms in the Huon Estuary and D'Entrecasteaux Channel. Through an adaptive management subproject involving researchers and representatives from industry and DPIW, we set out to design a monitoring project that would identify any specific environmental issues and provide sufficient data to identify any trends over time. The data obtained could also be used to validate predictions from the biogeochemical model.

Modelling and monitoring both play essential and synergistic roles in adaptive management. System understanding, encapsulated in models, underpins the choice of environmental indicators. Monitoring of these indicators provides direct assessment of the environmental state against agreed environmental objectives and targets. Models are needed for interpretation of monitoring results, especially for attribution of cause where environmental change is detected. Model predictions provide a basis for the development of management decision rules, while monitoring provides feedback on the outcomes of past management decisions, and data for testing and refining system understanding and model prediction.

In the first phase of this project, we developed state-of-the-art physical and biogeochemical models for the Huon Estuary and D'Entrecasteaux Channel (Volkman et al., 2006). These models have been further refined and calibrated in this second phase. These models provide industry and managers with understanding and predictive capability for system-wide sustainability of finfish industries. Models can be used to underpin design and testing of more informative and more cost-effective data acquisition and monitoring programs.

Models require knowledge of key processes such as the sources and fates of nutrients in the system, roles of sediments in nutrient release and oxygen consumption, good data sets for water column properties and well characterized boundary conditions.

This project also benefitted the SBT aquaculture and South Australian regulators by providing information on how the system-wide effects of that industry can be assessed. In particular, our monitoring and modelling approach has been applied to waters off Port Lincoln in collaboration with SARDI scientists. It should be noted that SBT farming is carried out in much more open marine environments than salmon farming in Tasmania, with a very different benthic habitat and food web structure, but the general approach is transferrable.

There is a need within Australia to adopt a more consistent approach to the management of finfish industries. A national approach is needed that can build on existing experience in the different States. We believe that the environmental projects within the Aquafin CRC provide a demonstration how this can be achieved by substantially improving communication between researchers, different finfish industries and regulators, adoption of common approaches such as SCFA ESD, adaptive management and modelling and development of generic management and monitoring tools that can be readily adapted to different fish species and environments.

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Chapter 2

Numerical Hydrodynamic Modelling of the D'Entrecasteaux Channel and Huon Estuary

2.1. Introduction

The physical oceanography of the D'Entrecasteaux Channel and Huon Estuary was described by Herzfeld et al. (2005) as a component of Phase I of the current project (Volkman et al., 2006). This study described the nested models used in the numerical investigations, forcing applied to the model, model calibration and various analyses to assist in characterizing the system including momentum balance, residual flow, flushing rates and system connectivity using passive tracers and particle tracking. Our subsequent research has focused on processes not captured in the earlier analyses, including those that occur at larger and smaller scales than those captured by the original model. These processes include the large-scale interaction of water on the continental shelf and slope with the water residing in D'Entrecasteaux Channel, with an emphasis on the transport of oceanic nitrogen into the coastal regions. High resolution models were also constructed at scales suitable to resolve flow in and around the farm cages. These models provided insight into transport of contaminants delivered into the cages and distributions of material settling out of the cages. In addition, data collected during Phase I was used to calibrate the intermediate nested model in order to extend its usefulness to periods outside the original model calibration period of 2002. This was subsequently used to create a long term scenario encompassing the years 2002 to 2005, which was again coupled to the biogeochemical model to assess long term changes in the system.

The hydrodynamic contribution to Phase II can be summarized as:

- Calibration of the intermediate model,
- Development of a 4 year scenario,
- Investigations of large scale onshore–offshore coupling,
- Investigations of cage scale processes.

2.2. Intermediate model calibration

The model used in the Phase I study was the highest resolution model in a 3 nested suite of models (e.g. Fig. 3.3 of Herzfeld et al., 2005). In Herzfeld et al. (2005), the regional and intermediate models were used to propagate basin-scale phenomena into the local domain via the subscription of sea level, temperature and salinity open boundary conditions. Only the local model was calibrated to data collected in the field. Additionally, this model was also dependent on measured data for the T/S open boundary conditions, thus restricting its use to the period for which these data were available (2002). The calibration of the intermediate model facilitates the nesting of the local-scale model over any time period, or for scenario simulations, independent of field derived forcing data (particularly data required to quantify the open boundary conditions). This effectively removes the reliance on field-derived data and therefore extends the period the model may be simulated to any period where surface atmospheric data, river flows and open boundary data for the regional model are available. The latter may now be obtained from global products such as synTS or BRAN. SynTS is a product of spatial and temporal distributions of temperature, salinity and sea level where surface distributions derived from satellite altimetry are projected through the water column using climatological correlations (Ridgway et al., 2006). The resolution of synTS is 0.25 degrees. BRAN is a data assimilating global model with high resolution (10 km) around the Australasian region (Oke et al., 2005), based on the global model MOM4 (Griffies et al., 2005). These products can provide sea levels containing background mean state and low frequency oscillations, and T/S distributions containing mesoscale variability suitable for forcing the regional open boundary. A tidal signal is added to the sea level derived from the global products.

Moorings were deployed during December 2003 to March 2004 for the purpose of providing temperature and salinity data to calibrate the intermediate model. CTD (conductivity, temperature, depth) instruments were moored at the surface and bottom at each end of the channel and at the mouth of the Derwent Estuary (three surface moorings were deployed in the northern channel and Storm Bay and two in the southern channel, on the eastern and western sides of the channel respectively). The locations of these moorings are listed Table 2.1. The duration of deployment for all moorings was December 2003 to March 2004.

Table 2.1. Details of moored instruments

Mooring	Latitude	Longitude	Depth (m)	Instruments
Iron Pot: IPb	-43.043	147.379	17.3	CTD
Iron Pot: IPe	-43.051	147.417	1.6	CT
Iron Pot: IPw	-43.039	147.338	1.9	CT
North Channel: NC	-43.063	147.352	1.2	CT
North Channel: NCb	-43.059	147.346	16.8	CTD, ADCP
South Channel: SCe	-43.425	147.094	2.3	CT
South Channel: SCw	-43.410	147.023	2.5	CT
South Channel: SCb	-43.423	147.069	51.0	CTD

CT = temperature and salinity; Sea-Bird SBE37-SM
CTD = temperature, salinity and depth; Sea-Bird SBE37-SM
ADCP = RDI 600 kHz acoustic Doppler current profiler

The measured data revealed that surface waters are consistently cooler and fresher than bottom waters. There is large diurnal variability in the surface temperature, due to the large amount of heat input from solar radiation during the day, and subsequent cooling at night. Deeper water beyond the influence of the surface mixed layer is isolated from this effect (e.g. the southern channel). Temperatures on the western and eastern sides of the Channel or Derwent do not exhibit marked differences. The salinity time series indicates a flood event occurs in late January and early February, causing a rapid decrease in salinity. The salinity change is most pronounced in the northern channel, consistent with the favoured up-channel propagation of flood plumes from the Huon (Herzfeld et al., 2005). Bottom salinities do not experience large changes as a result of the buoyant nature of fresh water increasing stability of the water column and inhibiting vertical mixing. Salinities on the western and eastern sides of the Channel/estuary show significant differences as a result of the Derwent and Huon flood plume's favoured exit from the estuary and southern channel on the eastern shore respectively.

Initial T/S values for the intermediate model calibration were derived from the global product OFAM SPINUP4. This product was a non-data assimilating version of the global model MOM4, and at the time of calibration was the most reliable global product available. Surface atmospheric fluxes and river flows were derived in a manner analogous to that of Herzfeld et al. (2005). Open boundary conditions were derived from the regional model (i.e. nested open boundaries) where the regional model again used the global model OFAM SPINUP4 for its T/S and sea level (background mean + low frequency) open boundary conditions. The global tidal model of Eanes and Bettadpur (1995), using the methodology of Cartwright and Ray (1990)**,** was used to generate amplitudes and phases of 14 tidal constituents at every open-boundary node of the regional domain. These were then used to reconstruct the tide at every time-step on the regional open boundary. The resulting sea level distributions used as regional boundary forcing therefore contain the background mean state, low frequency oscillations and variability on sub-diurnal time-scales.

The nesting strategy allowed solutions of the intermediate model to be generated using only standard meteorological measurements, which were subsequently compared to the measured data (Figs. 2.1 and 2.2). Modelled and measured T/S magnitudes are comparable, as is generally the temporal trends and timing of events. Magnitudes of diurnal variations are also comparable, although the model tends to over-predict the surface diurnal variation in the temperature solution possibly due to the treatment of short wave radiation input at the surface. Overall the intermediate model calibration is suitable for generating open boundary solutions for nested local scale models

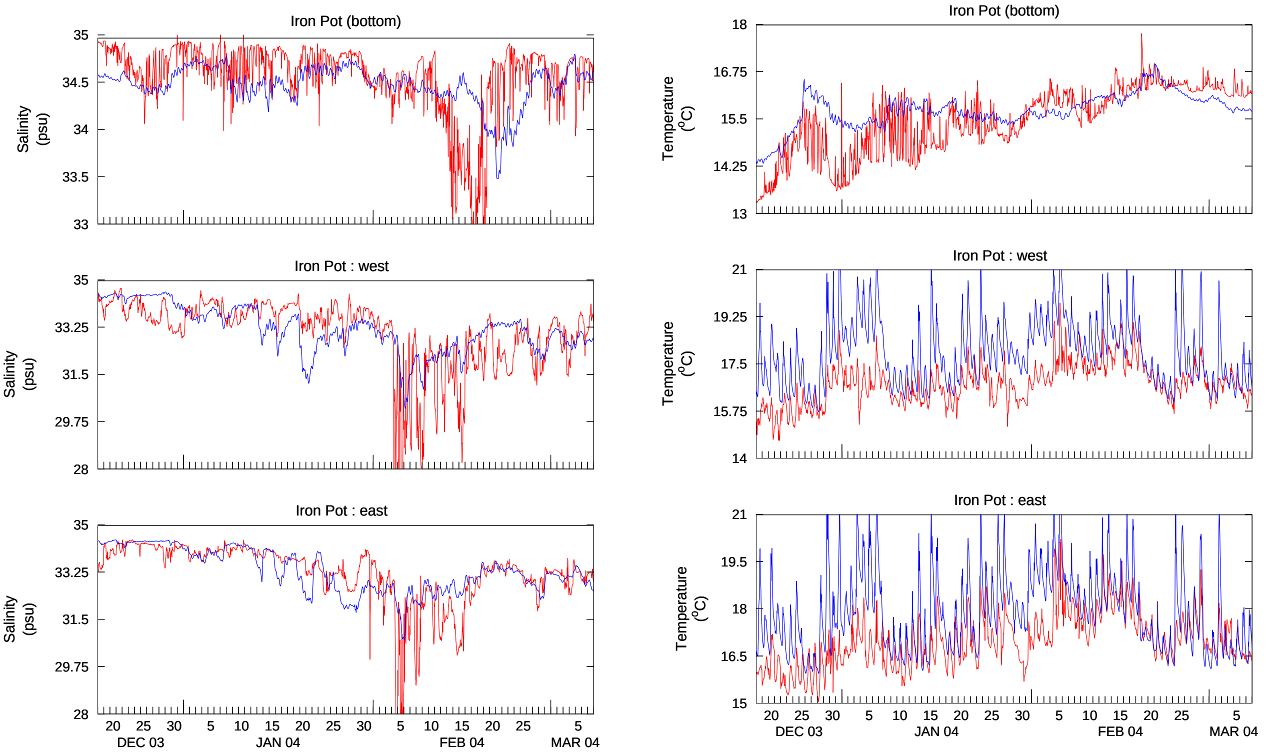


Figure 2.1. Intermediate model calibration in Storm Bay. Blue = modelled, red = measured.

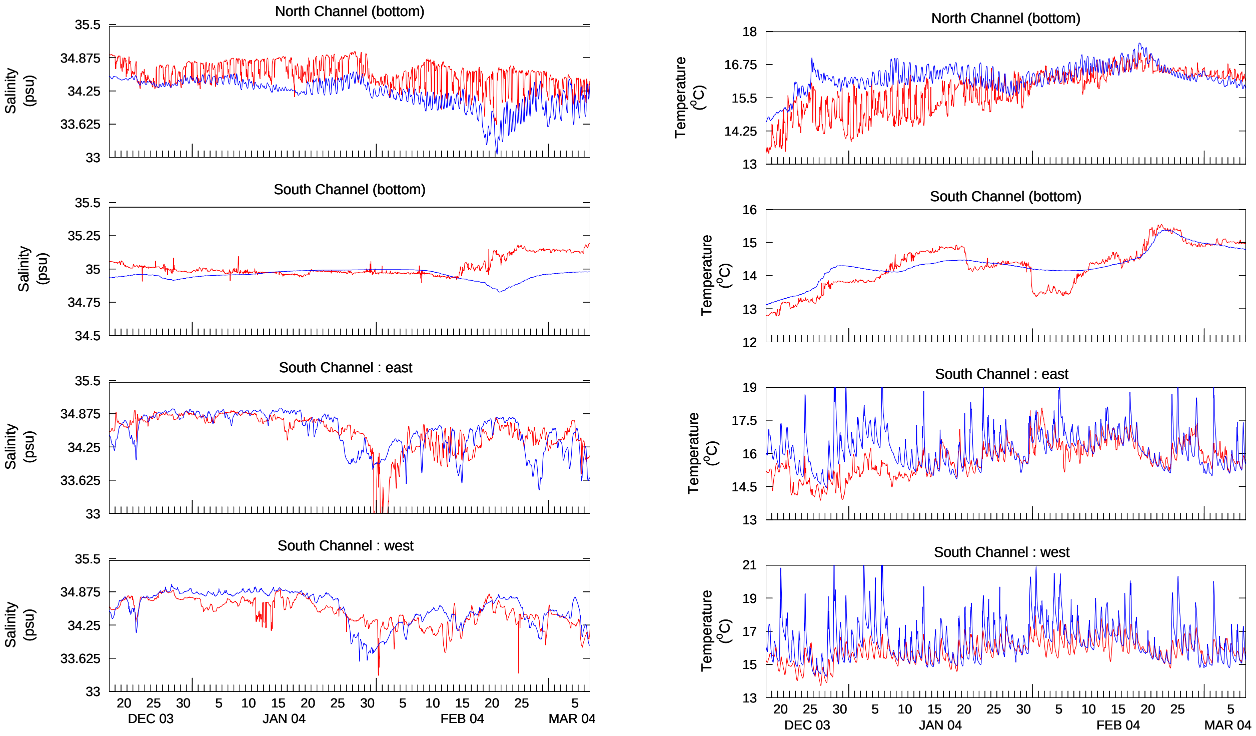


Figure 2.2. Intermediate model calibration in D'Entrecasteaux Channel. Blue = modelled, red = measured.

2.3. Four year scenario

The calibration of the intermediate model allowed the local-scale model to be simulated for any time period without the requirement of measured data being available to prescribe the northern and southern T/S open boundaries. These boundary conditions were prescribed via the nesting process. A four year scenario encompassing the period 2002 to 2005 was performed to produce coarse local-scale model output suitable for coupling with the biogeochemical model, so as to assess longer term trends of environmental indicators in response to aquaculture farming practices. This 4-year scenario required surface atmospheric fluxes and river flow data used by the model to be assembled, and this was performed in a manner analogous to that described by Herzfeld et al. (2005, Sections 5 & 6). The forcing for the nested suite is described in detail in Herzfeld (2008).

The year 2002 from the 4 year scenario was compared to measured data to validate the nested forcing strategy. Comparisons analogous to those provided in Section 7.8 of Herzfeld et al. (2005) (i.e. the data forced coarse model) are presented as Fig. 2.3. Note that the accuracy of the data forced coarse model deteriorated in comparison to the high resolution model due to the decrease in resolution, but was considered acceptable. The 4-year scenario produces comparable results to that of the data-forced coarse model. Surface temperature follows the annual cycle well, although tends to be slightly cooler than the measurements in winter, especially in surface waters in the northern Channel (also observed in the data forced coarse model) and bottom waters in the southern Channel. Bottom waters at the northern open boundary deviate from temperature measurements in early summer. Surface salinity is similar to the data-forced coarse model, where the model under-predicts salinity throughout the channel, especially mid-channel. The annual trend is consistent with measurements, although more variability is evident since measurements were only sampled monthly.

A consistent periodic cycle in the temperature solution is observed, reflecting the annual cycle of summer heating and winter cooling. Summer temperatures peak at just under 20 ^{o}C, and winter minimums are approximately 10 ^{o}C, resulting in about a 10 ^{o}C annual oscillation. The salinity solution exhibits much more variability, where trends are regulated by floods originating at the head of the Huon (characterized by rapid decreases in salinity), which is well specified in the model hence resulting in reasonable surface distributions. The largest of these floods occur in winter and spring where salinity decreases to almost 30 in 2002 and 2003, although it is observed that floods impact surface salinity in the southern Channel at all times of the year. The year 2005 appears to be a year where the least amount of freshwater input occurred.

Measured bottom salinity does not correlate well with surface salinity at the southern Channel locations (Fig. 2.3), indicating intrusions of shelf water through the southern boundary dominate bottom salinity rather than vertical mixing of fresh Huon flows. The reverse is true in the northern Channel, where bottom and surface salinity correlate indicating fresh Huon River flow influences bottom water in the northern Channel. This is consistent with the favoured flow direction of Huon flood water being up-Channel. However, since surface fresh water inhibits vertical mixing by stabilizing the water column, vigorous mixing must exist for the bottom waters to freshen during a flood. This mixing is hypothesised to occur at the narrowest point of the Channel where bottom depths decrease to ~10 m and current flows are maximum due to the geographic constriction to result in large vertical mixing within the bottom boundary layer.

Bottom salinity derived from the model at the northern boundary does not represent the flood response well (Fig. 2.3, ctd1). This may be due to the mid-Channel grid-spacing in the intermediate model being too coarse to adequately resolve the mid-Channel region and accurately represent the mechanism described above, resulting in northern open boundary forcing for salinity (derived from the intermediate model) that is not influenced by Huon floods. Away from the influence of the open boundary the coarse model appears to represent the bottom-freshening mechanism well. In the southern Channel differences are also observed between model and measurements on the southern boundary. The magnitude of these differences are consistent with those observed in the intermediate model calibration (late Feb, Fig. 2.2). Trends in bottom salinity in the southern Channel are largely controlled by intrusions of shelf water, which ultimately is dictated by the initial distribution of salt over the shelf and slope. This initial condition is prescribed using synTS, which does not include an altimetry component for salinity; i.e. salinity is estimated on the basis of climatological correlations alone. It is probable that error in the initial condition salinity distribution at depth is responsible for the deviation observed in bottom salinity at the southern boundary. This error is propagated through the nesting process to the coarse local model via the salinity open boundary condition derived from the intermediate model. Using more accurate salinity global products is likely to improve solutions of the coarse model, but the performance can be considered remarkably good overall.

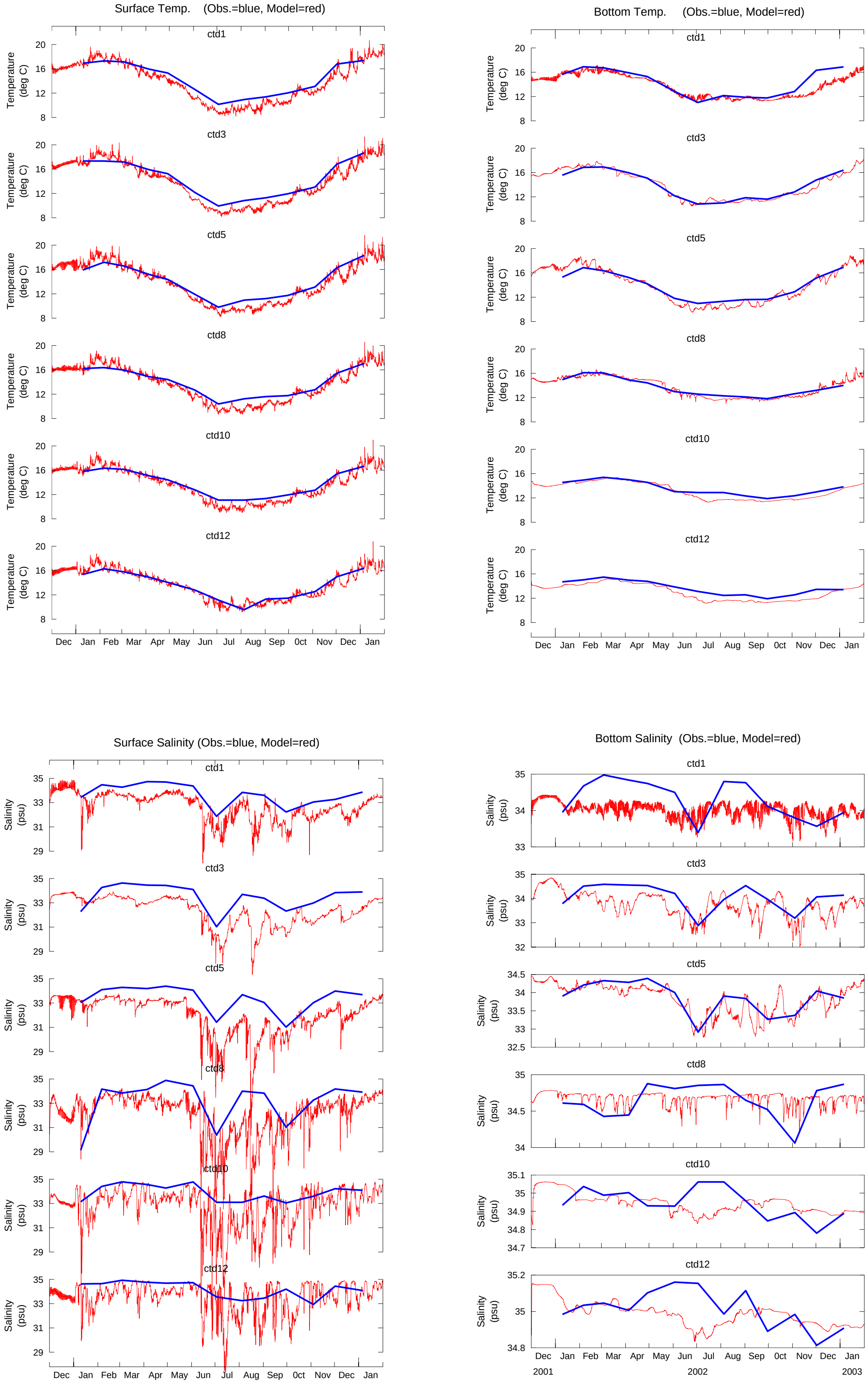


Figure 2.3. Comparison of coarse local modelled and measured T/S for 2002. red = modelled, blue = measured.

2.4. Large scale investigations

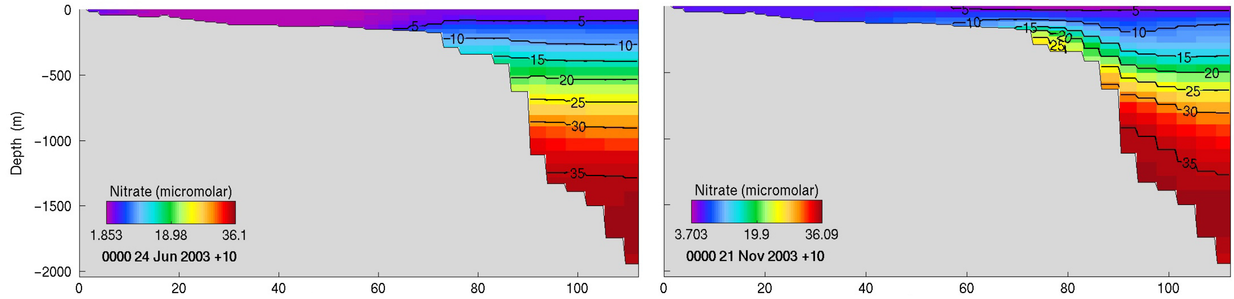
The Phase I modelling revealed that one important factor for accurate prediction of temperature and salinity (T/S) distributions in the model was the accurate prescription of T/S data on the open boundaries of the model. This suggests that the regional oceanography plays a role in determining the water properties in the D'Entrecasteaux and Huon system, since the T/S at the open boundaries (especially at the southern end of Bruny Island) are largely maintained by the offshore/shelf environment. Therefore, any comprehensive attempt to understand the distribution of tracers in the D'Entrecasteaux and Huon, including nutrients as well as T/S, should be undertaken within the context of a larger scale regional study which includes offshore – coastal interactions and impact of larger scale phenomena on the coastal zone. The latter include the impact of the East Australian Current on the east coast, the Zeehan Current on the west coast and eastward propagation of coastally trapped waves around the west coast.

These larger scale processes may be communicated into the regional model by nesting the regional model in global products that resolve these processes. SynTS was considered the optimum product for this nesting process, hence the regional model forced with synTS was used to investigate onshore-offshore interactions, and possible pathways of shelf water into the D'Entrecasteaux. This was accomplished using two passive tracers: one an idealized tracer designed to assess water entering D'Entrecasteaux from origins beneath 50 m depth, and another representing the distribution of nitrate initialised and boundary forced by CARS (CARS, Climatological Atlas of Regional Seas; Ridgway et al. 2002).

The passive tracer was included in the model having an initial condition of zero above 50 m depth and a value of 100 below 50 m. This tracer was boundary forced with a value of 100 below 50 m and had a no-gradient condition above 50 m. This boundary configuration corresponded to a constant injection of tracer through the boundaries at depth but allowed tracer to exit the boundaries in surface layers, analogous to a deep pool of nutrient (e.g. nitrogen) residing on the shelf break, and allowed an estimate to be made of how much tracer is transported into the surface layer from depth. If this tracer were assumed to be a normalized nitrate concentration, it provides an indication of how much nitrate makes its way into the surface layers of the D'Entrecasteaux from depth. The 4-year scenario was completed and the evolution of tracer (passive and nitrate) in the D'Entrecasteaux coastal zone analysed.

The nitrate climatology in southern D'Entrecasteaux Channel shows an increase to ~4 micromolar during spring. Minimum concentrations of ~1 micromolar occur in late summer and early autumn. The prescription of the nitrate climatology on the open boundaries of the models will therefore also contain this trend to result in increased nitrate during spring from boundary forcing alone. However, the surface nitrate concentrations in southern D'Entrecasteaux show peaks in excess of 4 micromolar, suggesting that additional nitrate is input into the system. For the regional model, nitrate attains a maximum of ~6 micromolar in late spring and early summer and a minimum of ~2–3 micromolar in late autumn and early winter. A secondary maximum of ~8 micromolar is evident in April for 2003. Concentration of surface passive tracer in southern D'Entrecasteaux also show distinct peaks in spring and minimums in winter, indicating that tracer, and possibly nitrate, during times of maximum concentration must originate from depths beneath 50 m.

Vertical sections originating in the southern D'Entrecasteaux Channel and extending to the boundary of the regional grid were obtained from the regional model for nitrate in winter and spring (e.g. Fig. 2.4 for 2003). These sections show that generally in winter there is no tendency for up-slope movement of high nitrate concentration water, characterized by a positive gradient of the nitrate contours towards the coast. In contrast, late spring exhibits significant uplift of high concentration nitrate water, especially at depth (e.g. 35 micromolar). This is particularly evident for 2003, and suppressed in 2005. The uplift of nitrate at depth extends over the slope and onto the shelf to result in elevated concentrations of nitrate, again particularly evident in 2003. This increased shelf nitrate appears to be absent in winter (e.g. concentration in winter 2003 was < 15 micromolar and over 25 micromolar in spring 2003).



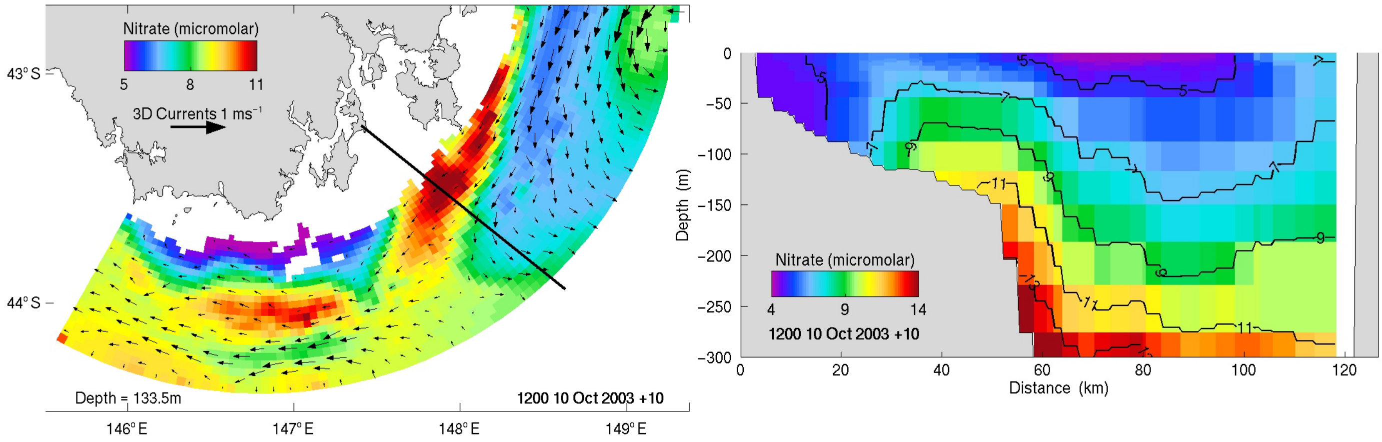
(a) Winter (b) Spring

Figure 2.4. Nitrate section south of D'Entrecasteaux, 2003.

The high nitrate water over the slope does not appear to be a continuous feature along the slope at depth, rather isolated cells directly south of Tasmania and south of the Tasman Peninsula are evident throughout the spring (Fig. 2.5). Sections of nitrate along a transect (marked in Fig. 2.5a) show a distinct uplift of high nitrate water onto the shelf (Fig. 2.5b). Further offshore from the high nitrate water is a band of low nitrate originating from the open boundary and associated with south-westward flow, which is also evident in the surface nitrate and flow. This flow is associated with an anti-cyclonic feature near the eastern offshore boundary, which is also evident in the sea surface solution where a pool of high elevation supports the anti-cyclonic flow. This circulation is not a boundary artefact, but is supported by a positive sea-surface height anomaly observable in synTS sea level anomaly distributions. It is hypothesized that during spring the existence of this eddy drives anti-cyclonic flow that impinges on the slope to generate up-slope flow in the bottom boundary layer, thus bringing high nitrate water from depth onto the slope. This anti-cyclonic eddy appears to be a consistent feature during spring (Fig. 2.6) and is often associated with a cyclonic eddy to the south-west. The presence of this eddy is consistent with the long term mean sea level distribution (Ridgway, 2007; their Fig. 7) which indicates an anticyclonic eddy persists off south-eastern Tasmania during spring, albeit further south than is observed in Fig. 2.6. The year 2005 appears anomalous in that the anti-cyclonic eddy resides further north than usual, with the associated cyclonic eddy assuming a position off Tasman Peninsula. This situation would tend to suppress any bottom-boundary layer driven upwelling.

The elevation gradient at a location adjacent to Tasman Peninsula and further offshore is indicative of the strength of anti-cyclonic (south-westwards) flow in the region. Analysis of this gradient from the model reveals considerable variability, however, the filtered gradient shows that generally south-westward flow is present during spring, and during winter flow is reversed. Generally a cyclonic eddy resides off Tasman Peninsula during winter (e.g. Fig. 2.7b) which would support clockwise circulation and north-eastward flow on the slope. An anti-cyclonic eddy often is observed further south of this cyclonic eddy. The year 2003 appears anomalous in that elevation gradients would support south-westward flow at times during winter; here the cyclonic eddy has shifted north further than usual (Fig. 2.7a), with the corresponding anti-cyclonic eddy stationed off Tasman Peninsula thus capable of supporting south-westward flow.

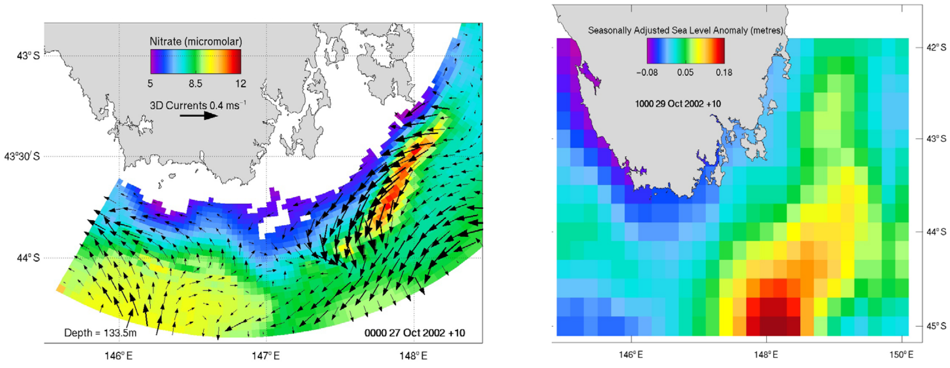
In summary, during spring high nitrate water is advected from depth onto the slope and shelf. This high nitrate water is available for transport into the coastal zone, e.g. D'Entrecasteaux Channel, via local processes (e.g. wind driven flow). The mechanism responsible for vertical advection of high nitrate water onto the shelf is proposed to be supported by large scale processes. Specifically, an eddy characterized by positive sea-surface height anomaly resides off Tasman Peninsula during spring. This eddy supports anti-cyclonic flow throughout the water column, resulting in south-westward flow along the continental slope south of Tasman Peninsula. This flow creates onshore (up-slope) transport in the bottom boundary layer where it engages with the slope; i.e. bottom boundary layer driven upwelling occurs. This mechanism is generally absent in winter, when a cyclonic eddy characterized by low sea surface height anomaly resides south-east of Tasman peninsula.



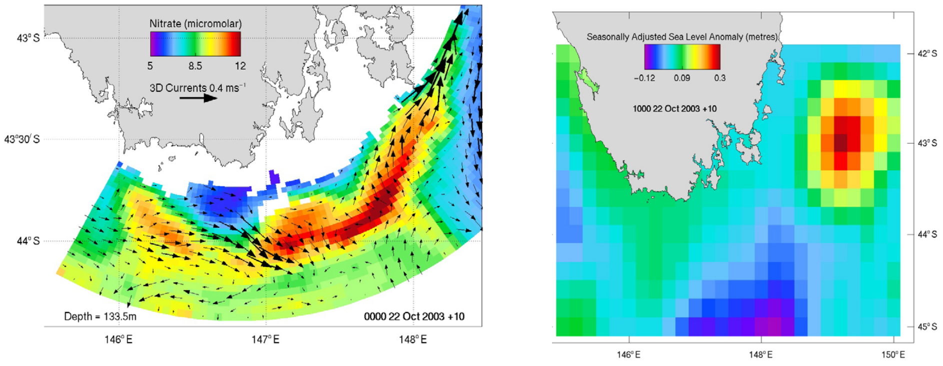
(a) Plan at 133 m depth

(b) Section

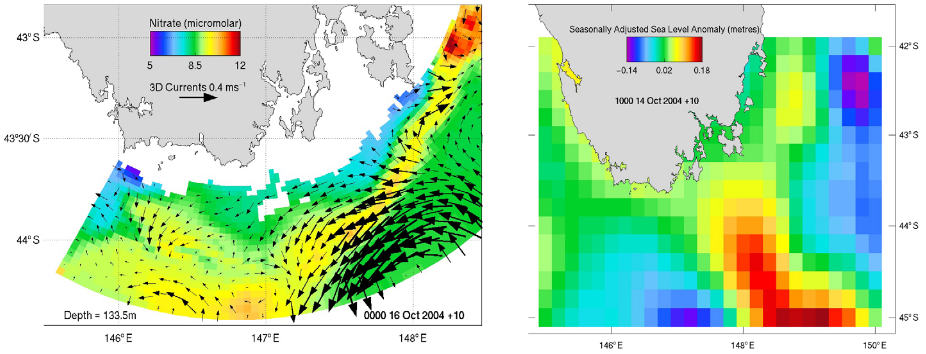
Figure 2.5. Nitrate concentrations and currents on October 10th, 2003.



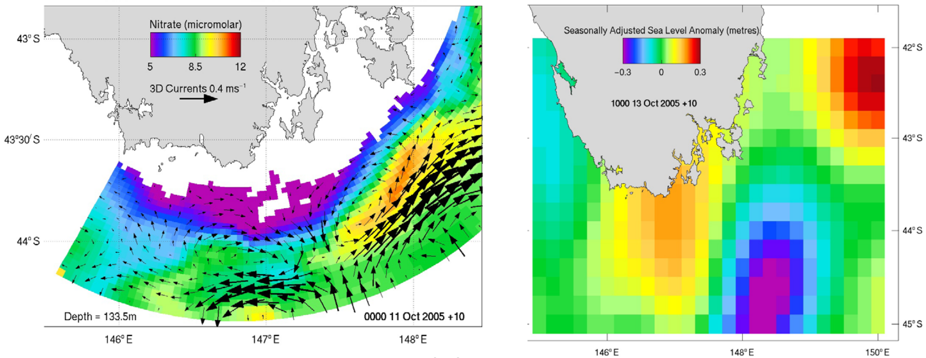
(a) 2002



(b) 2003

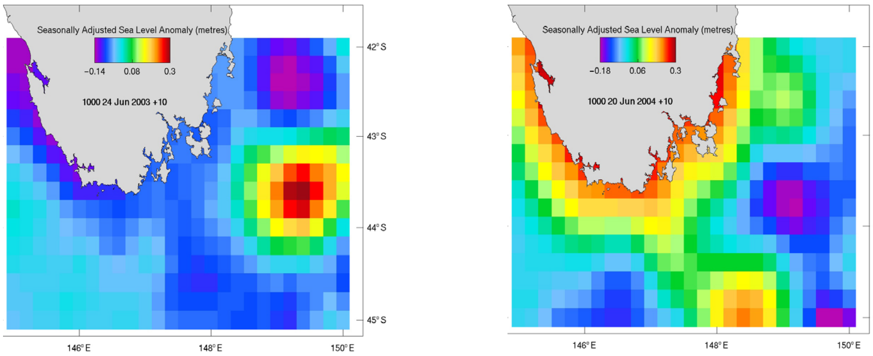


(c) 2004



(d) 2005

Figure 2.6. Spring nitrate and currents at 133 m and sea surface height anomaly.



(a) 2003 (b) 2004

Figure 2.7. Winter sea surface height anomaly.

Fluxes of nitrate and passive tracer were integrated over the southern entrance to D'Entrecasteaux Channel and averaged for 30 day periods. This was performed for both regional and intermediate models. Total fluxes per month (30 days) into D'Entrecasteaux from the open ocean, derived from the regional and intermediate models, are displayed in Fig. 2.8. A positive flux indicates a flux into the Channel.

As seen above, the regional grid resolves processes that may lead to uplift of high nitrate water from depth over the slope, however, the resolution of this model in the southern D'Entrecasteaux Channel is marginal to accurately predict flow into the Channel. Solutions from the intermediate grid are more reliable in this respect since the entrances to the Channel are adequately resolved, and if this model is forced with nitrate derived from the larger scale regional grid on its offshore boundary then it is possible for high nitrate water to be also delivered onto the shelf through the nesting process. However, nitrate is not a conservative tracer as is assumed in the model, and will undergo uptake and recycling in the surface layers. Forcing the intermediate model with nitrate climatology on the offshore open boundary implicitly represents the non-conservative behaviour of nitrate in deeper water, but cannot represent increases in shelf nitrate due to local uplift mechanisms. Consequently, nitrate in the coastal zone generally reflects the open boundary condition (i.e. climatology), resulting in lower concentrations. These arguments should be considered when analyzing model output, and in reality nitrate fluxes into D'Entrecasteaux probably lie between the under-resolved regional model and the intermediate model forced with climatology. Output from the regional model and intermediate model using regional model open boundary nitrate data (regional forced) and CARS open boundary data (CARS forced) are included in Fig. 2.8.

The fluxes derived from all models show the same basic annual trend (Fig. 2.8). The CARS forced intermediate model (red curves) exhibits a more distinct annual cycle and smaller fluxes due to the periodic nature of the climatological forcing on the open boundary. The regional forced intermediate model (green curves) fluxes are in closer agreement with the regional model fluxes (blue curves) and are larger than

CARS forced fluxes (almost twice as large) due to the local delivery of high nitrate water onto the shelf. This model also shows substantial export of nitrate from the Channel during winter.

It can be observed that the passive tracer and nitrate follow the same basic temporal trend, with largest fluxes in spring and minimum fluxes during late summer and early autumn. The average maximum flux of nitrate into the Channel during spring appears to be approximately 200 T per month. Considerable inter-annual variability is evident, where a secondary maximum is observed in April 2003, particularly in the regional model. The similarity in temporal trends between the passive tracer and nitrate suggests that the same mechanism is primarily responsible for delivering material into the Channel. Since no passive tracer existed in surface waters above 50m depth, and could only be brought from depth to the surface, this suggests that the majority of nitrate is also delivered into the D'Entrecasteaux from below 50 m in offshore waters.

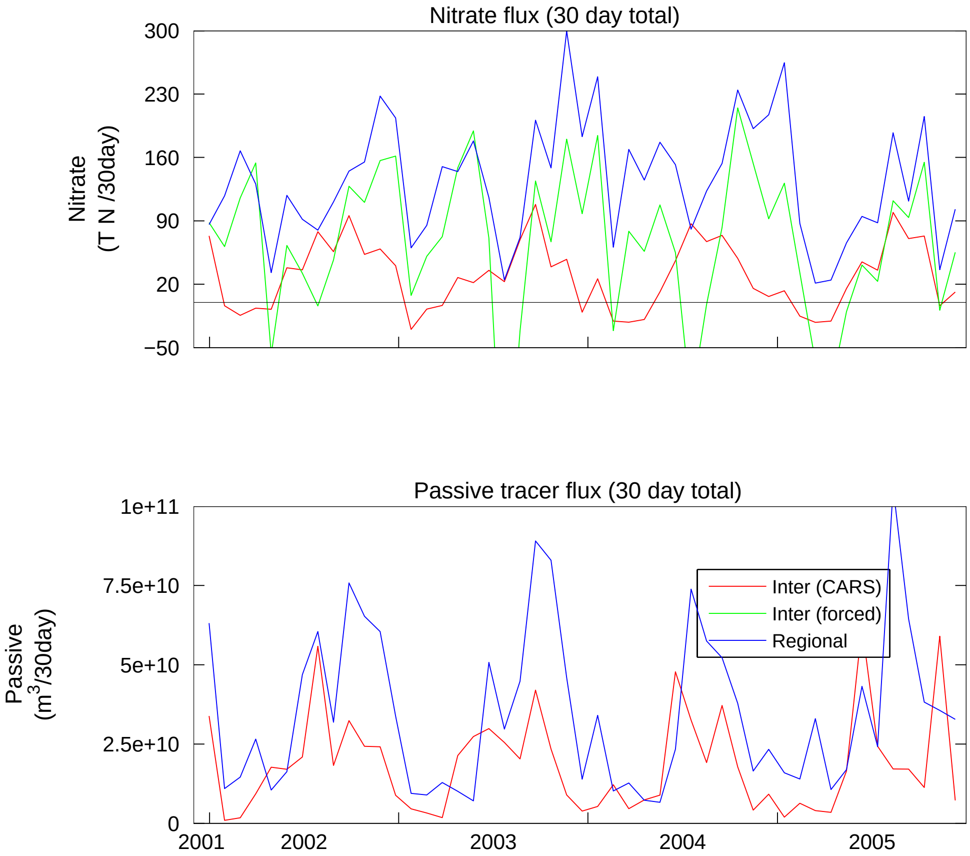
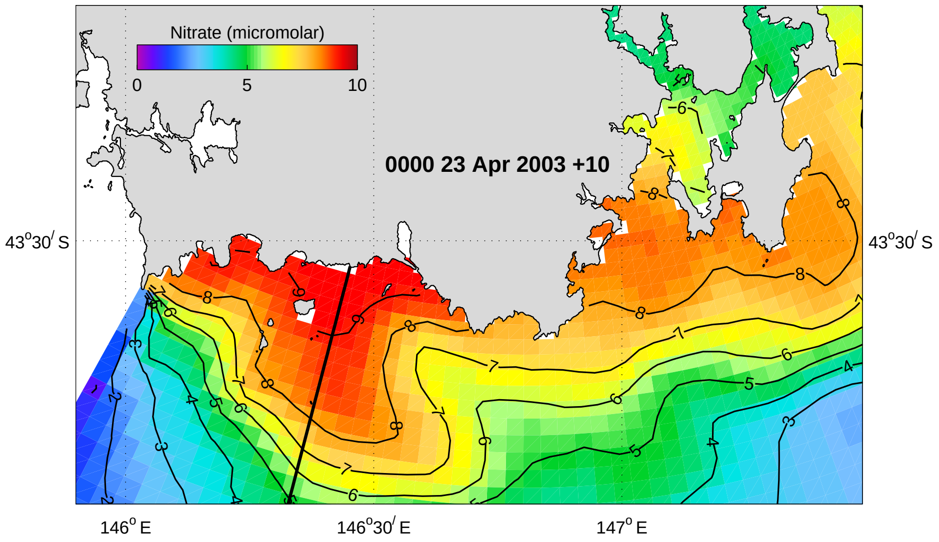


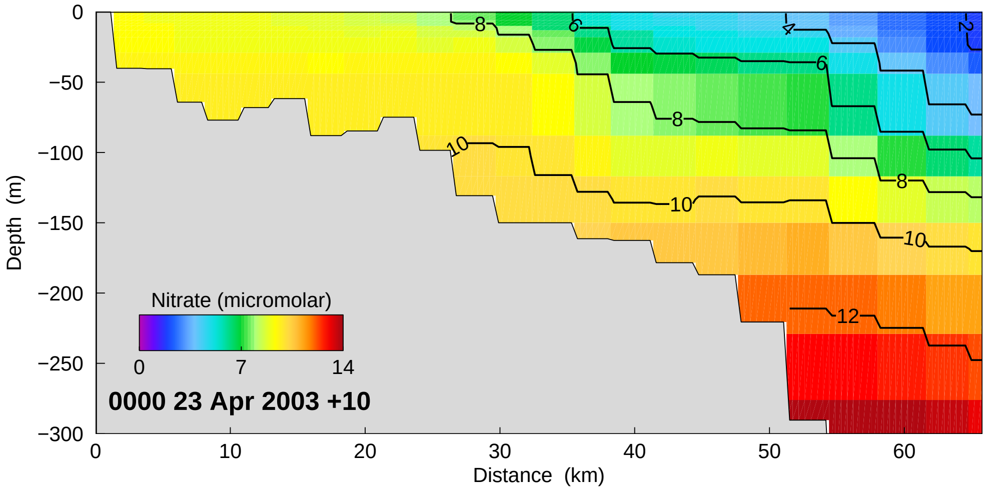
Figure 2.8. Comparison of nitrate flux through the southern D'Entrecasteaux Channel.

An anomaly in the nitrate trend for the regional model is the large fluxes (and concentrations) in April of 2003 and 2004. The surface nitrate concentration near Maatsuyker Island during Autumn 2003 more than doubles during this period, to peak at ~10 micromolar. This increase is attributed to local upwelling along the southern Tasmanian coast. The intermediate model does not capture this phenomenon well due to its smaller geographic extent. Surface distributions and cross-shore sections (Fig. 2.9) show elevated nitrate east of Maatsuyker Island and an uplift of nitrate over the shelf and slope to depth up to 300 m. The uplift over the shelf is also seen in the density structure, where the upward tilt of the isopycnals is present to depths of ~800 m. The evolution of the high surface nitrate is traced from late summer 2003 to late autumn. The high concentration first appears off the southern tip of Bruny Island in mid-March, then develops a secondary cell on the southern Tasmanian coast in late March. Numerous cells develop during April leading to an apparent band of high concentration in late April. It is proposed that the mechanism responsible for this shelf and slope uplift is the interaction of the East Australian Current (EAC) with the slope, where a westward EAC flow over the slope drives an onshore flow in the bottom boundary layer to result in bottom driven upwelling (e.g. Cresswell, 1994; McLean-Padman and Padman, 1991). The elevated nitrate on the shelf may then be brought into the coastal zone by local wind-driven processes.

Particle tracking was performed to ascertain the relative influence of the EAC and the Zeehan Current in the Channel region. Particles were seeded on the western (red particles) and northern (blue particles) open boundaries of the regional model. The distribution of particles is therefore representative of the trajectories of the EAC (blue) and Zeehan (red) currents. A snapshot is displayed in Fig. 2.10. Fig. 2.10a is typical of the general situation, where the Zeehan Current propagates eastward along the coast and pushes the EAC offshore as it flows in a clockwise sense around Tasmania. In the winter the Zeehan Current dominates the region. This situation is consistent with the studies of Cresswell (2000) and Ridgway (2007). In summer the EAC is present to ~147^{o}E (Fig. 2.10b) and attains a position nearer the coast. Generally only in February or March is there a possibility that the EAC approaches the coast and may enter the D'Entrecasteaux Channel. However, there is considerable inter-annual variability in the EAC position relative to the southern Tasmanian coast; e.g. in April 2003 the EAC intruded close to the coast (Fig. 2.11b), consistent with the bottom boundary layer induced upwelling described above, whereas in 2002 the EAC was pushed further offshore by the Zeehan Current (Fig. 2.11a).



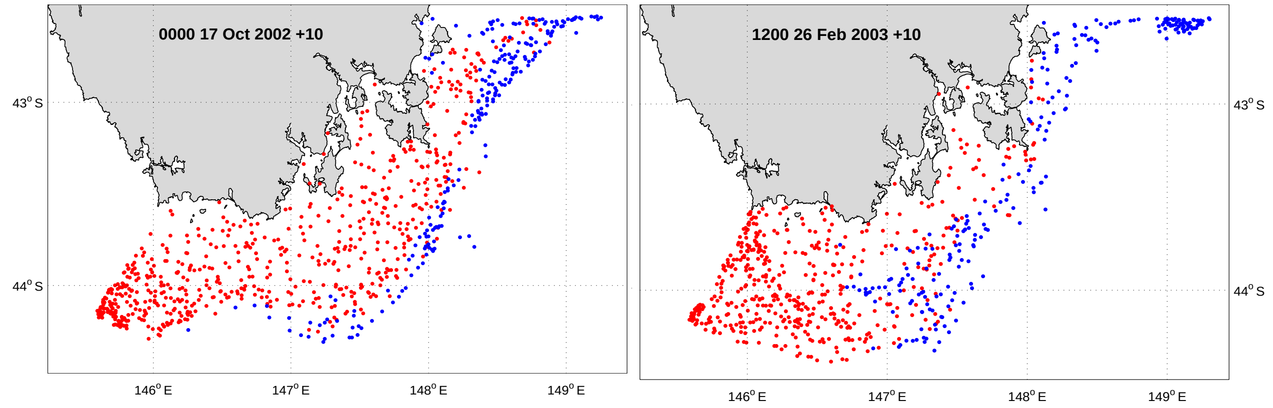
(a) Plan view with section indicated



(b) Section view

Figure 2.9. Nitrate concentrations around Maatsuyker Island.

In summary, the nitrate concentration in southern D'Entrecasteaux undergoes an annual cycle, driven by exchange with shelf water. Elevated nitrate concentrations occur as a result of larger fluxes through the southern Channel entrance, with largest fluxes in spring (~200 T per month) and minimum fluxes during late summer and early autumn.

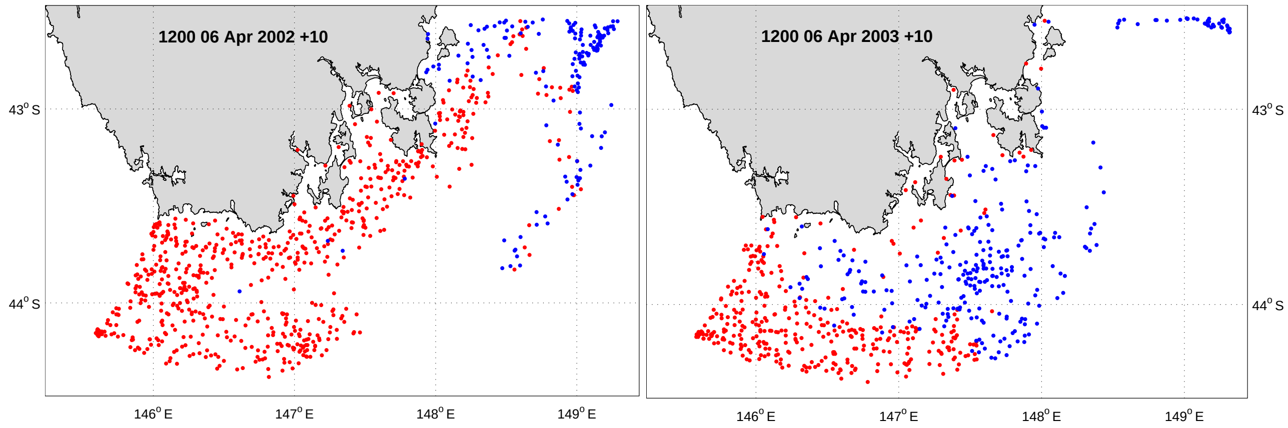


(a) Spring

(b) Summer

Figure 2.10. Snapshot of particle distributions.

The elevated nitrate on the shelf is hypothesized to be the result of large scale uplift of nitrate over the slope, due to bottom boundary layer driven upwelling as a consequence of an anti-cyclonic eddy stationed off Tasman Peninsula during spring. Local upwelling is possible during March and April, confined to the southern Tasmanian coast and southern tip of Bruny Island. This upwelling is hypothesized to be the result of onshore flow in the bottom boundary layer over the shelf and slope due to the passage of the EAC over the slope south of Tasmania. The EAC exhibits considerable variability in its passage down the west coast of Tasmania, and the upwelling events of this nature probably only coincide with strong EAC intrusions into southern Tasmanian waters.



(a) 2002

(b) 2003

Figure 2.11. Snapshot of particle distributions in April.

2.5. Small scale investigations

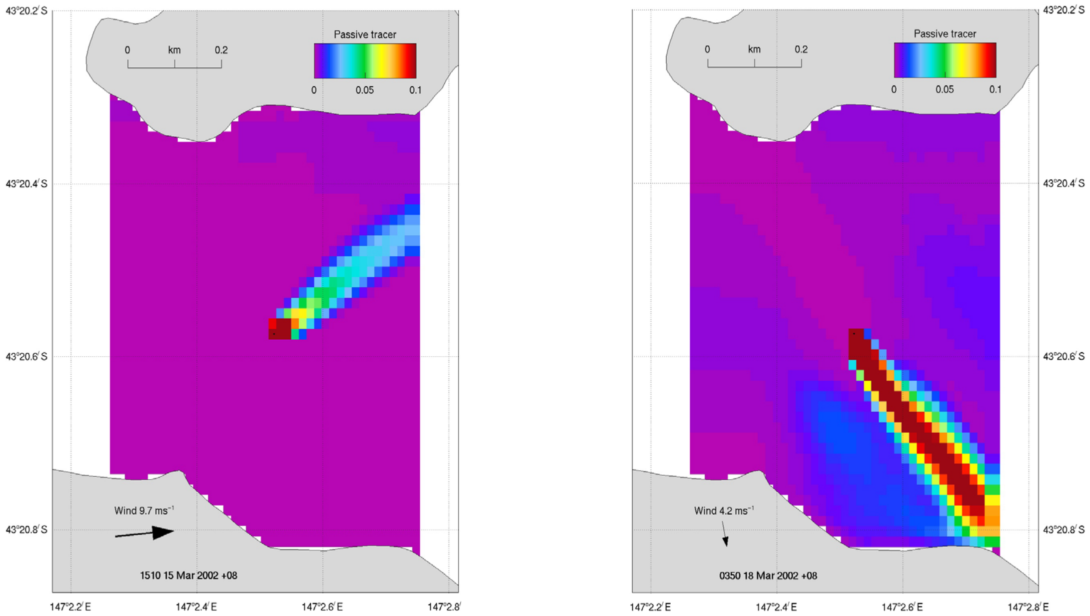
Mixing zones exist on small spatial scales around point sources of dissolved tracer release, where the concentration of the tracer is elevated above background values. The extent of this zone is dependent on the local transport characteristics and is typically of the order of several hundred metres. A fish farm site can be considered a point source of nutrients and consequently has an associated mixing zone, the extent of which is of interest from the perspective of impact analyses and sampling strategy. The model may be used to characterize mixing zones around several farm sites by performing analyses on the distributions of idealised tracers released at each site. This process will involve nesting a high resolution grid, encompassing the farm site and mixing zone, in the local model. Several of these high resolution models have been developed for the Stringers Cove farm site in Port Esperance. The Stringers Cove site was chosen since this was the focus of the Fallowing Project in the first phase of the Aquafin environmental research and familiarity with the site characteristics exists. This site has also been identified as an area particularly susceptible to eutrophication effects, and consequently was targeted as the location for a field-based process study in Phase II. The model was designed to cover scales of up to a kilometre and able to resolve the farm site itself, i.e. the resolution was several metres. The spatial scales employed in this model make execution speed very slow (< 1:1), and the model is only suitable of observing events on timescales of several tidal cycles.

The high resolution model aims to resolve the farm structure itself, and model code modification has been necessary to allow resolution of a solid structure of several metres depth floating on the surface to exist with free flow beneath this structure. This should allow phenomena inside the cage to be observed as well as outside, and may provide insight into the role of the cage structure as a barrier to flow. This high resolution model is referred to as the polar grid. An alternative grid (referred to as the rectilinear grid) which did not resolve the farm structure was also constructed to characterize transport properties around a farm site at high resolution. Details of these high resolution models are included in Herzfeld (2008).

Passive tracers and particles tracking were used in the rectilinear grid to assess the fate of tracers released from a mid-domain farm site. Continuous releases were performed where a unit concentration (or continuous stream of particles) was input to the source cell, and instantaneous releases where the source cell was initialized with a unit concentration (or 1000 particles). The simulation period was March 14th to 18th, 2002. Tides were in the neap phase during this period and relatively small. Winds were generally light westerlies of ~3–5 m s^{-1}, occasionally strengthening to 12 ms^{-1} south or north-westerlies. This created not very energetic forcing conditions around the farm site resulting in low surface velocities, with a mean of 0.11 m s^{-1} over the simulation period. The low energy environment resulted in relatively slow dissipation of tracer from the release point corresponding to a near worst-case scenario for tracer flushing.

The tracer was released into the surface cell for the continuous release, which resulted in the plume assuming the form of a narrow ribbon extending from the source with width of ~100 m. The length of the ribbon is several hundred metres;

concentrations in the plume are generally less than 0.1 and on occasion the length of the ribbon having concentrations of this magnitude reaches the eastern boundary, i.e. the length is > 300 m. The plume generally is transported eastward in the direction of the wind. Snapshots of maximum and minimum plume extents are displayed in Fig. 2.12. Note that the plume concentration and position is dependent on the flow conditions in effect at the time, and will vary under different forcing conditions, hence different lease sites are expected to have different plume dispersion characteristics. Based on this analysis, it appears possible that the plume emanating from the Stringers farm cage may contain tracer with concentration of 10% of the tracer concentration within the cage up to distances of ~0.5 km from the cage site.



(a) Small extent from source (b) Large extent from source

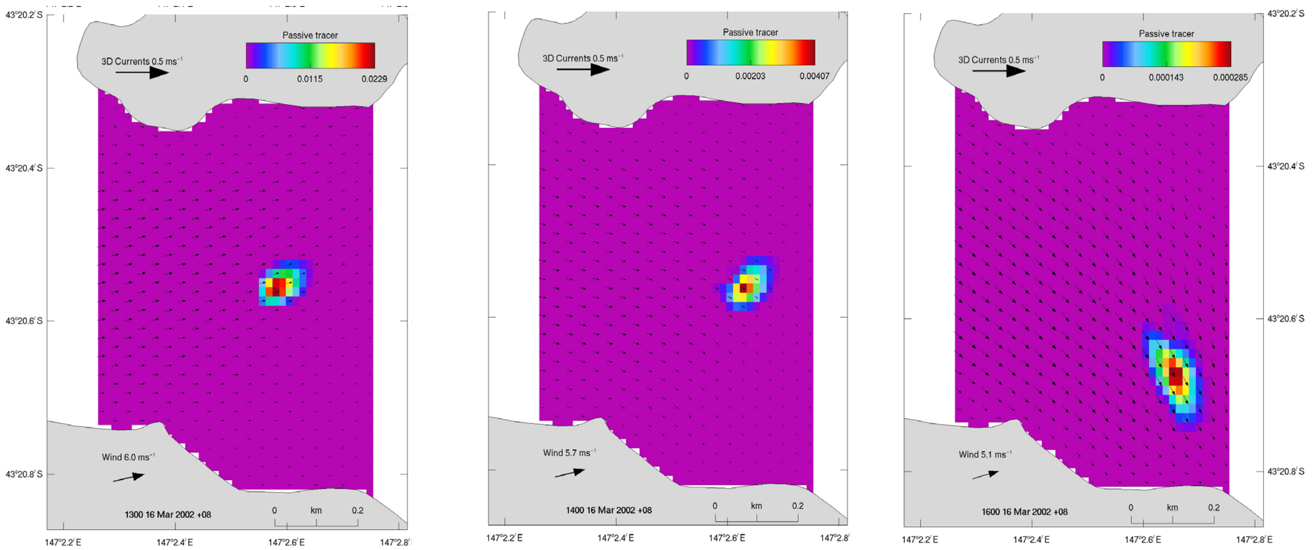
Figure 2.12. Snapshots of passive tracer plume extents.

Instantaneous releases were also investigated, where tracer with concentration of 1 was initialized within the top 5 m of the model cell corresponding to the cage site, and allowed to disperse over time. This depth distribution corresponded to ~1/8 the water column depth at the release site. The tracer was re-initialised at this cell at 12 hour intervals. The plume trajectory is traced for one such release in Fig. 2.13. It is observed that the plume does not simply diffuse around the release site but is advection dominated, and is transported away from the site as a coherent pool, slowly diffusing as it does so. The tracer appears to be influenced by the currents at depth rather than the surface flow, as the tracer pool's trajectory does not always correlate with surface currents. This is especially evident in the latter stages of the pool's life. On occasion the surface pool is advected in the opposite direction to the surface currents (e.g. March 16th 00:00). This behaviour suggests the surface signature is the result of vertical mixing, and transport is sub-surface advection dominated. Note that the colour bar scale changes with time in Fig. 2.13; surface concentrations of 0.002 (e.g. 0.2% or 500:1 dilution) the initial concentration are found within 0.5 km distance from the farm site after 8 hours.

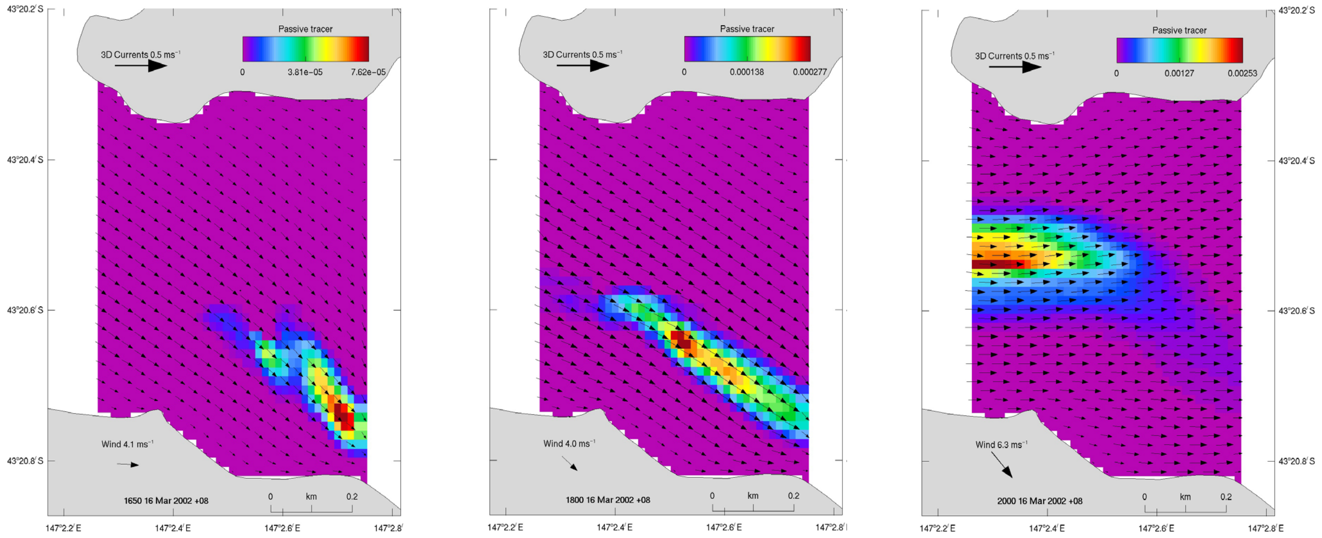
Particles released over the top 5 m of the water column exhibit similar behaviour to the instantaneous release. Selected trajectories over the 6 hour period from 15:00 to 21:00 March 15th reveal that particles are capable being transported up to 0.5 km in 6 hours. Some of these particle trajectories oppose the wind direction, indicating transport with sub-surface flow that has rotated anti-clockwise (left) relative to the surface flow. This flow rotation is consistent with an Ekman spiral in the Southern Hemisphere. The vertical profiles of current components at the release site at 12:00 March 18th reveal that currents reverse direction at ~5 m depth, attaining maximums at 10–12 m then linearly decreasing to zero at the bottom.

The high resolution models may also be used to diagnose the 'footprint' of particles that settle from the water column at any given velocity, i.e. the spatial distribution of particles that have settled into the sediment. Both un-eaten feed pellets and faeces sink to the sediments where they accumulate and break down, consuming oxygen as they do so to create anoxic environments. This is the principal reason that the practice of fallowing is adopted, so that bottom sediments are periodically given the chance to recover. It has been estimated that 25–30% of dry weight feed is excreted from Atlantic salmon in the aquaculture environment (NCC, 1990; Butz and Vens-Capell, 1982). Chen et al. (1999) estimated that the settling velocity of Atlantic salmon faecal pellets lies between 5.3 and 6.6 cm s^{-1}. These authors noted no significant dependency of the sinking velocity on pellet size, but did find that sinking rates were significantly dependent on salinity with higher rates for lower salinity. Also noted was the tendency for faecal pellets to absorb water or disintegrate as they fall through the water, thus reducing their settling rate.

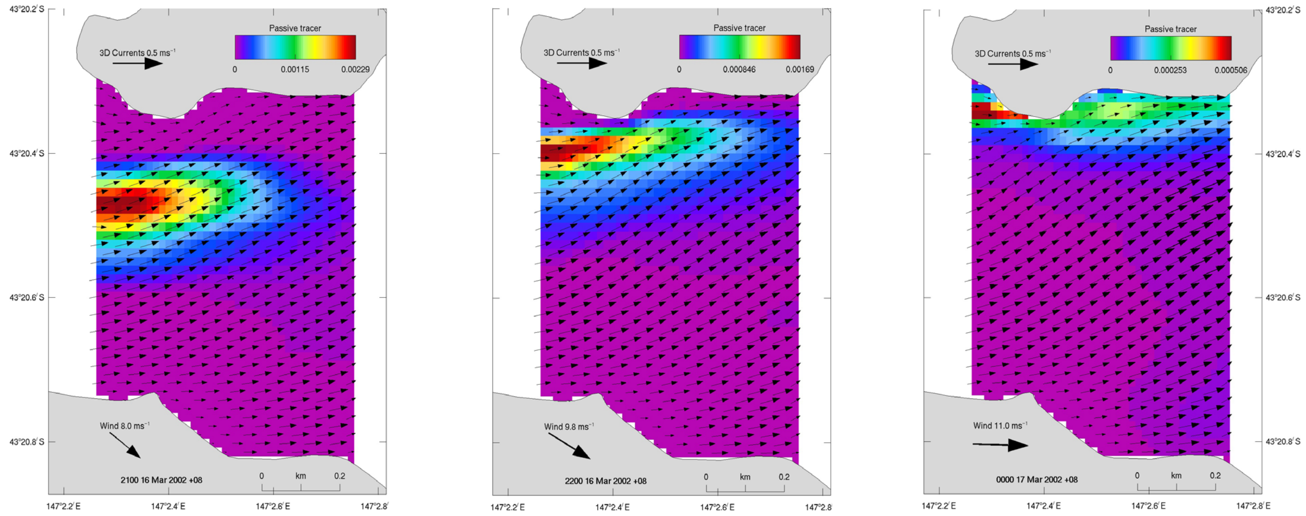
Wong and Piedrahita (2000) found that the settling rates of faecal material from rainbow trout were 0.7 cm s^{-1}, which is of the same order of magnitude as pellets from Atlantic salmon. Feed pellets are reported to settle faster than faeces, with rates of 6 to 14 cm s^{-1} (Chen et al., 1999), suggesting that the footprint of faecal pellets is likely to be greater than that of feed pellets. A settling rate of ~5.5 cm s^{-1} equates to about 4.7 km day^{-1}, hence in 40 m water depth a pellet would take ~12 minutes to reach the bottom. Assuming a mean current of 0.11 m s^{-1}, the pellet may be advected ~80 m laterally in this time. Given that sub-surface currents are less than surface currents and current sub-surface current reversals are possible, the footprint is likely to be smaller than this distance, so that settling farm waste is likely to be confined to the vicinity of the cage. It is instructive to diagnose the footprint from material settling with velocities less than those reported above; Fig. 2.13 shows the footprints for material settling at rates of (a) 30 m day^{-1} (0.035 cm s^{-1}) and (b) 600 m day^{-1} (0.7 cm s^{-1}).



(a) 1 hour after release **(b) 2 hours after release** **(c) 4 hours after release**

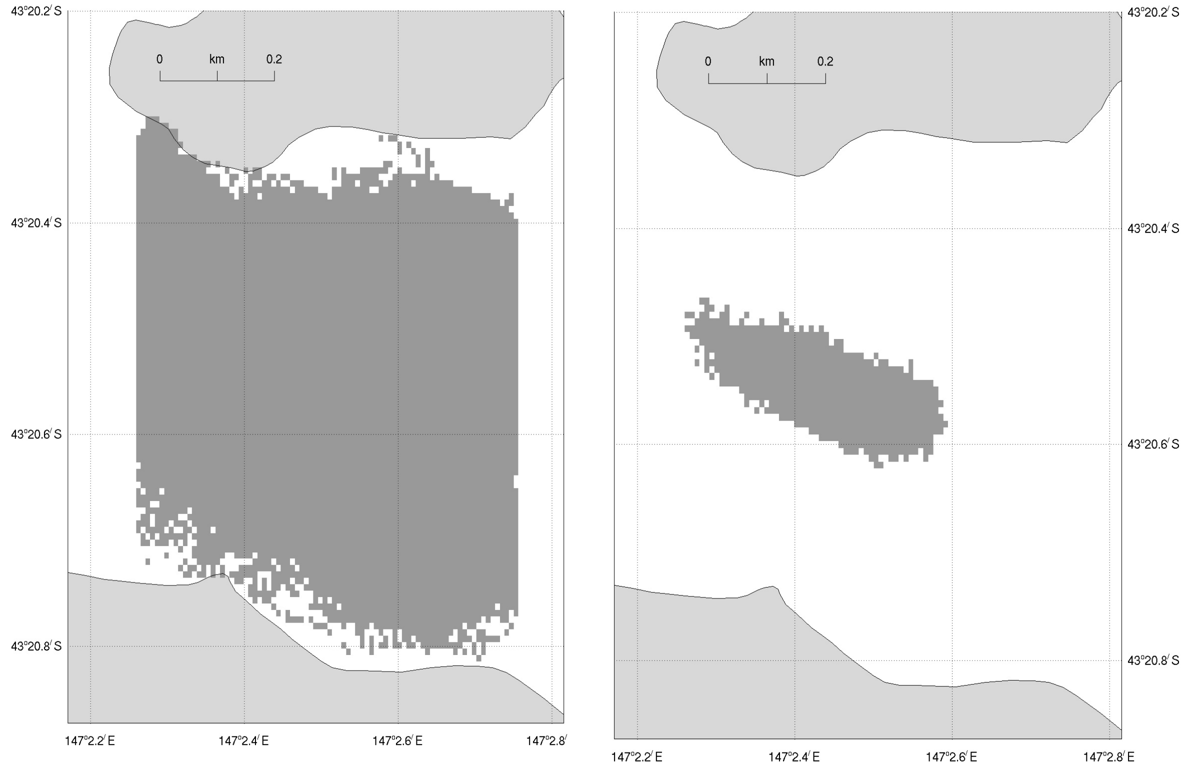


(d) 5 hours after release **(e) 6 hours after release** **(f) 8 hours after release**



(g) 9 hours after release **(h) 10 hours after release** **(i) 12 hours after release**

Figure 2.13. Example of instantaneous tracer release.



(a) 30 m day^{-1} settling rate **(b) 600 m day^{-1} settling rate**

Figure 2.14. Footprints from particulate material settling from the water column.

As anticipated, the slower settling rate deposits particulate material over a large area encompassing almost the whole domain. At a rate of 600 m day^{-1} the particles are confined to an plume of length ~500 m and width ~200 m originating from the cage. The plume is oriented north-west from the cage site, largely in the opposite direction to surface currents, again demonstrating the influence of sub-surface flow. The polar high resolution model was used to investigate the footprints associated with higher settling rates. Footprints corresponding to a settling rate of 10 cm s^{-1} (8640 m day^{-1}) reveal the particles are confined to the vicinity of the cage (< 20 m). This rate can be considered a mean of faecal material and feed pellets.

2.6. Summary

The intermediate model was calibrated to temperature and salinity data collected in the field at the northern and southern entrances to D'Entrecasteaux Channel and the mouth of the Derwent Estuary during December 2003 to March 2004. This model was nested in a larger regional model, which was driven with global products derived from global models or altimetry + climatology on its open boundaries. The modelled temperature and salinity compared favourably to measured data. This configuration was then used to simulate a longer period from 2002 to 2005 inclusive. The data collected in Project 4.2 were used to validate the coarse resolution local model.

Modelled data compared well with data from the measurement-forced coarse models. This indicates that the nesting process using the calibrated intermediate model in conjunction with global products is suitable for creating accurate model output for any simulation period, removing the requirement of collecting open boundary forcing data from the field. The coarse local model for the long term simulation is suitable for coupling with biogeochemical and sediment transport modules to investigate the longer term trends of environmental indicators in response to aquaculture farming practices.

The 4-year simulation using the larger scale models was used to assess the transport of nitrate into D'Entrecasteaux Channel. A seasonal cycle of nitrate flux into southern D'Entrecasteaux was observed, with maximum flux of ~200 T per month in spring, and minimum fluxes in late summer and early autumn. The increased fluxes during spring are due to the intrusion of high-nitrate water from depth onto the slope and shelf, which is then available for transport into the coastal zone by local processes. The up-slope intrusion is hypothesized to be due to the continental-slope interaction of large scale currents resulting from a persistent anticyclonic gyre stationed off south-east Tasmania in spring, producing upwelling in the bottom boundary layer. Local uplift of high-nitrate water on the slope and shelf is possible during March and April, confined to the southern Tasmanian coast and southern tip of Bruny Island. This also increases the flux of nitrate into the southern D'Entrecasteaux Channel. This uplift is hypothesized to be the result of onshore flow in the bottom boundary layer over the shelf and slope due to the passage of the EAC over the slope south of Tasmania. The EAC exhibits considerable variability in its passage down the west coast of Tasmania, and the upwelling events of this nature probably only coincide with strong EAC intrusions into southern Tasmanian waters during February to April. It is only during these times that there is the possibility of the EAC interacting with the shelf south of Tasmania; other times of the year the shelf is dominated by eastward transport of water in the Zeehan Current.

High resolution models were developed to investigate the mixing zone characteristics around a fish farm cage. These analyses revealed that a continuous release of tracer from the cage results in a plume emanating from the cage in the form of a long narrow ribbon. The position and concentration of the plume is dependent on the *in situ* flow conditions, and possesses large variability in space and time. It is possible for a plume having concentration of 10% (i.e. 10:1 dilution) of the source concentration to exist up to 0.5 km from the cage on occasion. Instantaneous releases of tracer from the cage site result in the tracer's transport with the flow away from the cage in the form of a coherent pool, rather than simple diffusion around the cage site. This pool slowly diffuses horizontally as it is advected. The rate of advection of this pool is dependent on the flow conditions at the time and hence subject to considerable variability. Sub-surface currents often determine the trajectory of this pool, with the surface signature of tracer being the result of vertical mixing of tracer from depth. In these cases the surface signature of the pool may move in the opposite direction to the surface currents.

Particle tracking was used to investigate the ‘footprint’, or spatial distribution of particles in the sediment, for various settling velocities of particles. For settling rates typical of feed pellets or faecal material, e.g. of the order of 10 cm s^{-1}, the footprint is restricted to the cage site. However, settling rate an order of magnitude less than this result in footprints in the form of an elongated plume with typical length ~0.5 km and width ~200 m emanating from the cage site. The direction of this plume is controlled by sub-surface currents, and may be in the opposite direction to surface currents.

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Chapter 3

Nutrient Cycles

3.1. Introduction

The Huon Estuary and D'Entrecasteaux Channel are water bodies in southern Tasmania (~ 43°S and 147°E) in a temperate zone with prevailing westerly winds. Both water bodies have considerable exchange with more offshore waters and these are influenced by the regional circulation (Ridgway, 2007). In summer the East Australian Current (EAC) brings water down the east coast and into this region (Fig. 3.1). In winter the region is influenced by the Zeehan Current, often postulated as an extension of the Leeuwin Current bringing water all the way from Western Australia. Based on climatology the regional waters and both currents are relatively low in nutrients, particularly in terms of silica (Fig. 3.2).

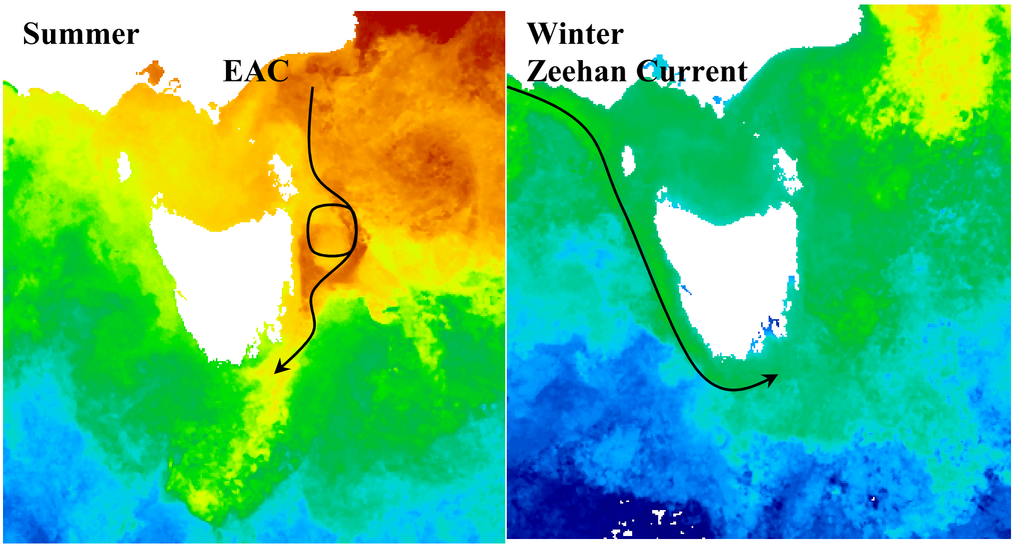


Figure 3.1. A simple schematic diagram of the two ocean currents with major influence on the sources of offshore marine waters into the Huon Estuary and D'Entrecasteaux Channel.

The dominant inputs of dissolved inorganic N (DIN) and dissolved inorganic P (DIP) into the Huon Estuary and D'Entrecasteaux Channel are by exchange with the regional marine waters. Correspondingly the dominant temporal patterns in surface nutrient concentrations are typical of temperate latitude cycles with greater concentrations during winter and relatively depleted concentrations during summer. Average annual concentrations are low relative to many temperate, coastal ecosystems with surface mean values in D'Entrecasteaux Channel of ~ 0.6 µM NO_3 and ~ 0.3 µM PO_4. The average concentrations of DIN, DIP and dissolved reactive silicate suggest that productivity of this ecosystem is predominantly limited by the availability of N.

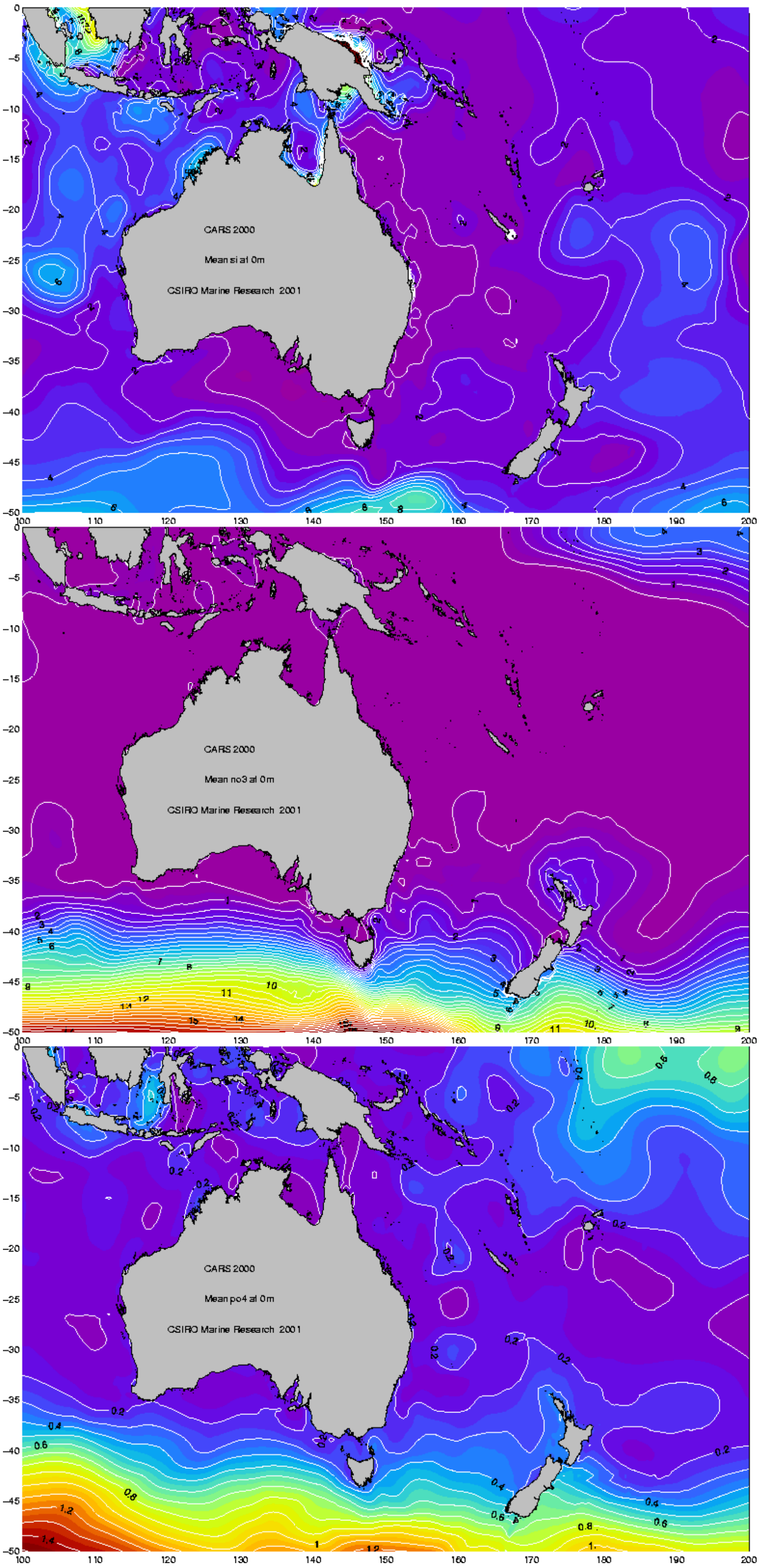


Figure 3.2. Regional nutrients: Analysis of all CSIRO surface data for silicate (top panel), nitrate (middle panel) and phosphate (bottom panel).

The relatively low nutrient inputs into this ecosystem result in a low productivity, or oligotrophic environment. The Huon Estuary and D'Entrecasteaux Channel are microtidal so that short term (daily) exchanges are not large and wind forcing can be a significant factor in determining transport and exchange. Although data are sparse, the land-derived and river-supplied inputs of DIN and DIP are relatively small. The Huon River is the major source of fresh water; it has a seasonal flow with a long-term average of ~ 40 $m^3 s^{-1}$ in summer (Dec-Feb) rising to ~ 185 $m^3 s^{-1}$ during winter (June-August) with high natural variability between days, months and years (Fig. 3.3). The Huon Estuary is a salt wedge with seasonally variable surface outflow and inflow at depth. The Huon River is a major source of refractory nitrogen and dissolved silica to the Huon Estuary and D'Entrecasteaux Channel. The catchment includes large areas of native vegetation, some rural and pastoral sections with small urban centres. There are 10 sewage treatment plants (STP) discharging into the Huon Estuary and D'Entrecasteaux Channel servicing about 36,000 people.

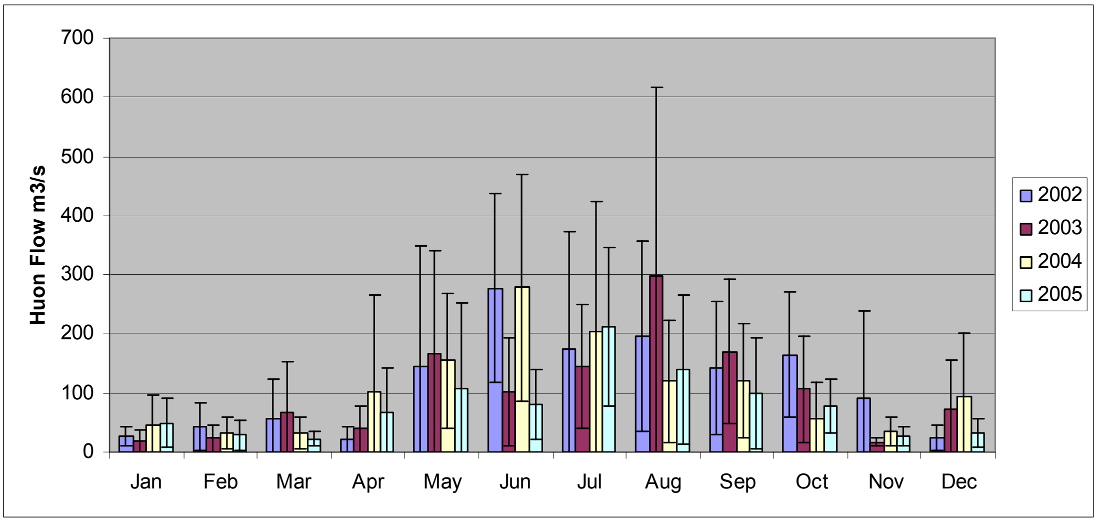


Figure 3.3. Monthly mean Huon River flow in $m^3 s^{-1}$ estimated from averaged daily flows in each month. Error bars are ± 1 standard deviation of averaged daily flow.

The Huon Estuary and D'Entrecasteaux Channel are important regional centres for the farming of salmonids. Over the past 20 years the farming of salmonids has grown rapidly. The salmon industry has done a lot to reduce its environmental impact often working proactively with state and federal agencies. Together they have implemented a successful program to monitor and eliminate potential problems associated with rapidly sinking waste particulates (largely fish faeces and feed pellets) that can cause sediment degradation. The industry's rapid growth, however, has made them a major source of nutrients (primarily from fish excretion and particulate remineralization) into this environment. These nutrients are dispersed throughout the pelagic environment, potentially resulting in more subtle, but more widespread, impacts on both the pelagic and benthic ecology of the region. While the absolute magnitude of the nutrient inputs from fish farming is still less than other natural inputs, their more labile form and the seasonal timing of the inputs results in a larger impact than would be suggested by a simple N budget. This chapter summarizes the data on nutrients in the Huon Estuary and D'Entrecasteaux Channel. Where data are available it compares the relative loads from different sources.

3.2. A retrospective analysis of the nutrient data from the Huon Estuary Study from 1996 to 1998

During the period from 1996 to 1998 CSIRO conducted a major FRDC-funded project in the Huon Estuary (Butler et al., 2000). The sampling intensity in terms of spatial and temporal coverage of the Huon Estuary was greater than in our more recent research. The data set resulting from the Huon Estuary Study (HES) provided a baseline of the Estuary's nutrient status which can be used to contrast with subsequent observations. This permitted a retrospective statistical analysis of the nutrient data to characterize the previous nutrient status of the Estuary. Five sample locations (Fig. 3.4) were sampled weekly. Approximately 20 sites were sampled quarterly for 2 years. Sampling consisted of discrete bottle samples at fixed surface and ~ 2 m above bottom with a mid-depth sample at various depths depending upon location and date.

Weekly samples across the five sites had overall mean depths of 0, 2.5 and 15 m and were pooled by season, plotted and then analyzed for vertical and temporal trends in nutrient concentrations (Fig. 3.5). Nitrate, phosphate and ammonium concentrations showed statistically significant effects of season and depth ($P < 0.001$, 2 way ANOVA, data were not homogenous or normally distributed). Concentrations of nitrate, phosphate and ammonium increased significantly with depth while silicate concentrations were greatest at the surface (annual mean = 16.5 µM) and significantly (Kruscal Wallis, $P < 0.001$) lower near the bottom (annual mean = 3.2 µM). Temporal trends were consistent across all depths and are presented as running medians of depth-averaged concentrations (Fig. 3.5), with the exception of nitrate where all data are shown. There were no significant seasonal patterns in silicate concentrations. Depth-averaged nitrate and phosphate concentrations peaked in winter (Fig. 3.5).

Mean summer surface concentrations of nitrate (0.15 µM) and phosphate (0.21 µM) were significantly less than other seasons (Kruscal Wallis, $P < 0.001$ for both). Depth averaged ammonium concentrations peaked strongly in autumn (Fig. 3.5). On average the DIN (dissolved inorganic N = NO_3+NO_2+NH_4) to dissolved inorganic P (DIP) ratio in surface waters was 1:1 and the DIN to dissolved Si ratio was 1:33.

Using surface nutrient data from the 20+ sites collected quarterly (Fig. 3.4) a simple correlation with salinity showed that riverine inputs were the major source of dissolved silicate to the ecosystem (Fig. 3.5). Correlations of phosphate and nitrate concentration with salinity are strongly influenced by a few (<8) observations at very low salinities. If these few observations of salinity below 0.1 are removed then there are 256 remaining surface observations of salinity and concentrations of both phosphate and nitrate show positive correlations ($P < 0.001$, 0.006, respectively) with salinity. The nitrate data further suggest that the positive relationship would be stronger if concentrations of zero nitrate were not included. These results suggest that the Huon River was not the major source of either nutrient.

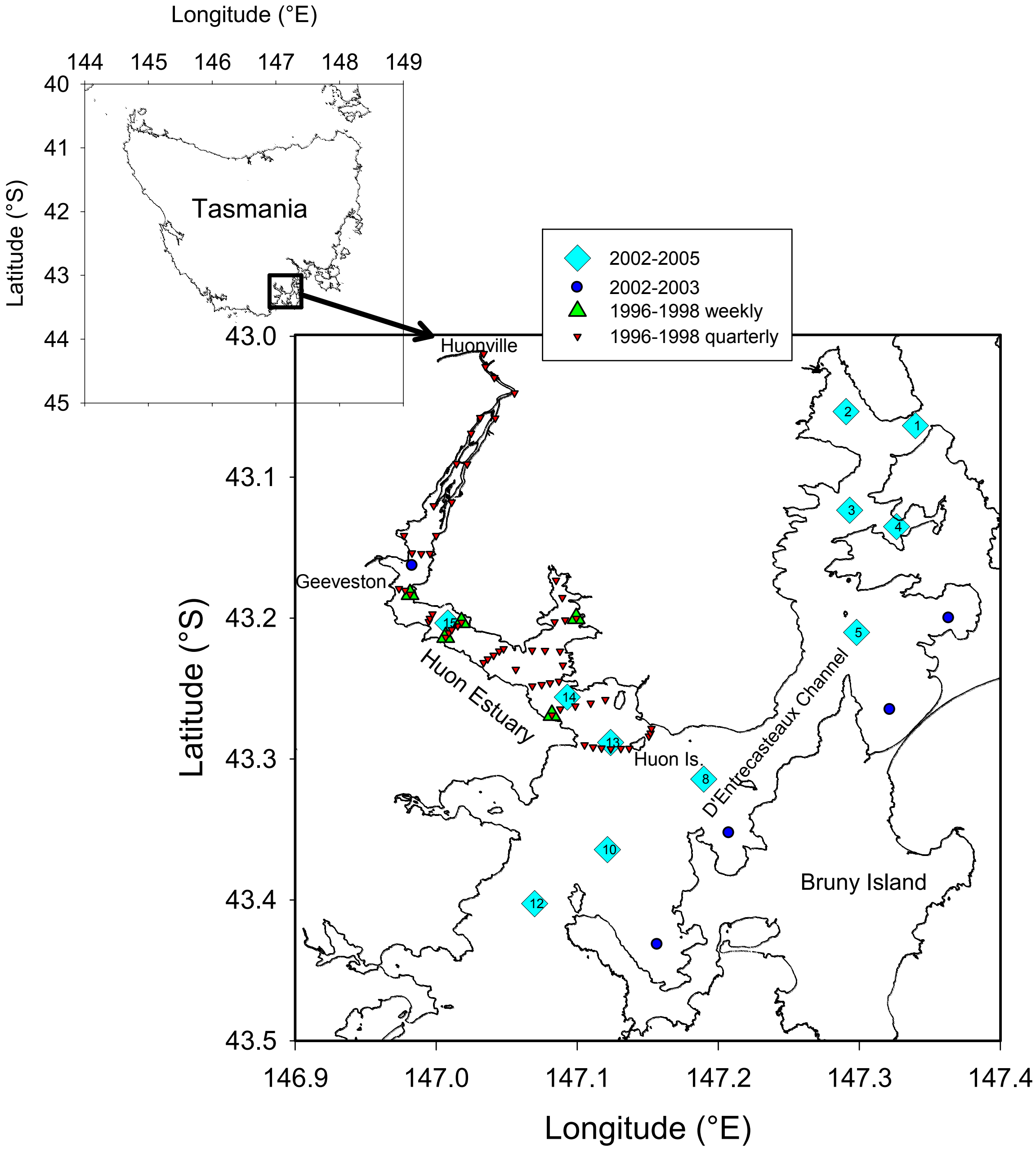


Figure 3.4. Map of study region showing location of sampling sites.

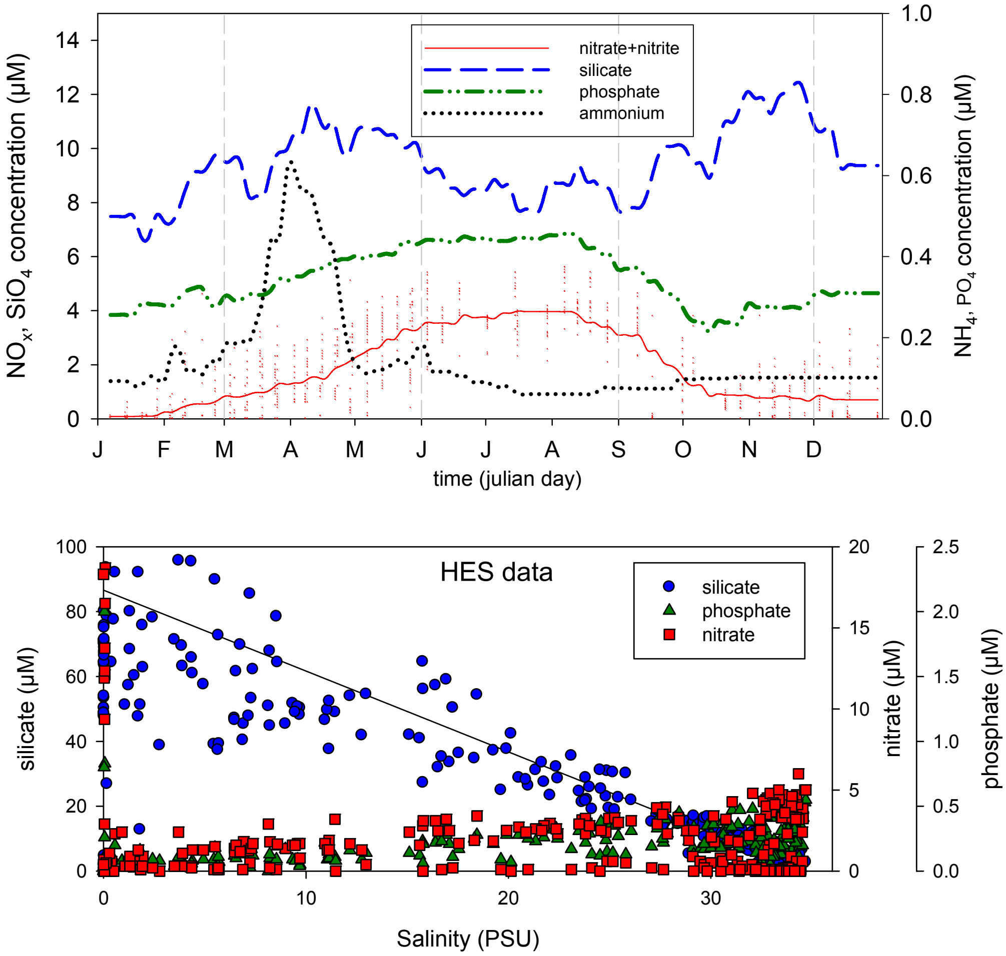


Figure 3.5. Analysis of Huon Estuary nutrient data collected from 1996–1998. Upper panel is from weekly sampling at 5 sites and 3 depths. For silicate, phosphate and ammonium only the running average is shown, for nitrate all data (+) and the running average are presented. Lower panel is from quarterly sampling at ~ 20 sites.

From the retrospective analysis of the Huon Estuary data it can be seen that the nutrient concentrations in this ecosystem are temporally variable. They are also spatially variable, both horizontally and vertically. Because of the heterogeneity there has been no attempt to make a direct comparison between the 1996–1998 data and those data collected more recently (2002–2005). The temporal variability of silicate is largely due to variation in rainfall in the Huon River catchment and runoff. Nitrate and phosphate concentrations peak during the winter intrusion of offshore waters originating from depth. Ammonium concentrations, especially in subsurface waters peak during autumn. The increase is hypothesized to have occurred due to weaker exchange and increased remineralization.

3.3. A review of nutrients in D'Entrecasteaux Channel (DEC) during 2002–2003

Nutrient data from 2002–2003 were summed across the 12 stations in D'Entrecasteaux Channel (Fig. 3.4) and presented to illustrate the temporal variation. Nitrate concentrations were considerably greater during winter than other seasons (Fig. 3.6). Surface and bottom nitrate concentrations were similar in June, but diverged during the development of the spring phytoplankton bloom. Surface nitrate concentrations were very low from October 2002 to April 2003.

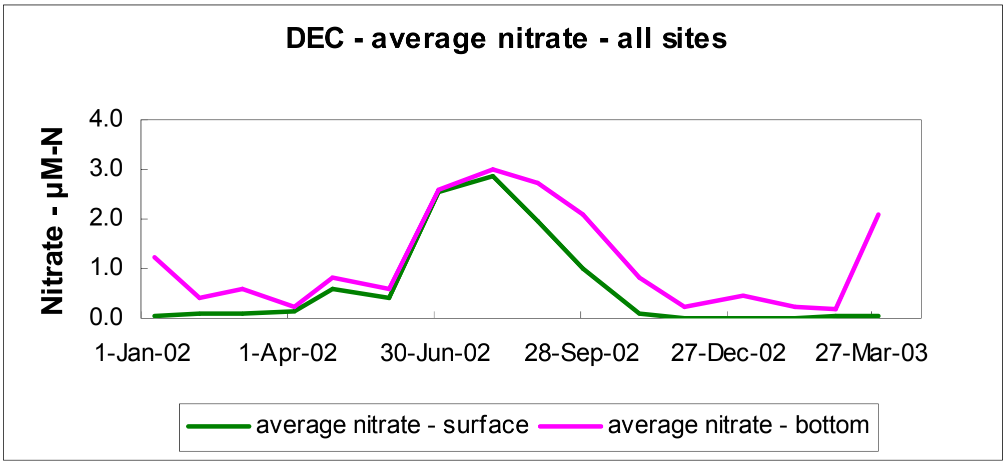


Figure 3.6. Surface and bottom nitrate concentrations, averaged across 12 sites in the D'Entrecasteaux Channel.

Surface and bottom nitrite concentrations were also elevated throughout the winter and appeared to show a double peak in concentration with the highest nitrite concentrations measured in June 2002 followed by a smaller peak in spring (Fig. 3.7).

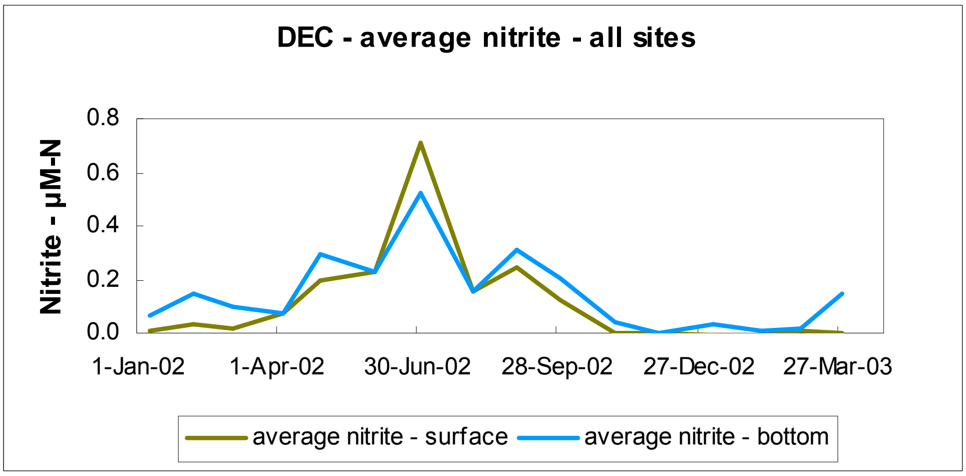


Figure 3.7. Surface and bottom nitrite concentrations, averaged across 12 sites in the D'Entrecasteaux Channel.

In contrast, there appeared to be no definite temporal pattern in bottom ammonium concentrations (Fig. 3.8). The average surface ammonium concentrations were elevated only in April-May 2002 and again in January 2003 (Fig. 3.8). The data were fitted to a general linear model that indicated ammonium concentrations were significantly greater at depth than at the surface (Table 3.1). There were also significant differences between sites (Table 3.1) with shallow sites in Isthmus Bay, Great Bay and Green Island near the middle of Bruny Island having the lower NH_4^+ concentrations than either the upper or lower end of the Channel especially in samples from depth (~ 2 above bottom, Fig. 3.8). To reduce the possible number of statistical comparisons the site west of Partridge Island was selected as a *control* and Bonferroni corrected multiple *t*-tests indicated that Isthmus Bay was significantly lower in ammonium concentration than the *control* site.

Table 3.1. Results of statistical analyses for $[NH_4^+]$

Source of Variation	DF	SS	MS	F	P
Site	11	1.91	0.174	2.411	0.008
Depth	1	1.64	1.638	22.721	<0.001
Date	15	2.90	0.193	2.679	0.001
Residual	165	11.89	0.0721		
Total	383	36.68	0.0958		

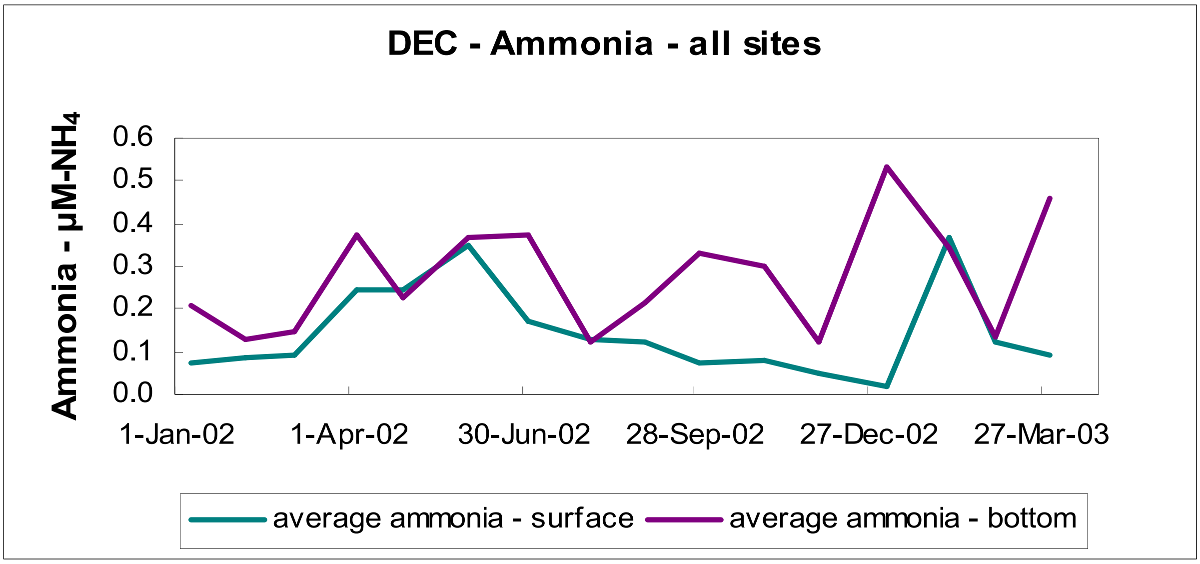


Figure 3.8. Surface and bottom ammonium concentrations averaged across 12 sites in the D'Entrecasteaux Channel.

Average phosphate concentrations at both surface and bottom were about 0.2 µM in the summer of 2002–2003 at the start of the survey, rising to a sustained level above 0.3 µM throughout autumn and winter 2002 (Fig. 3.9).

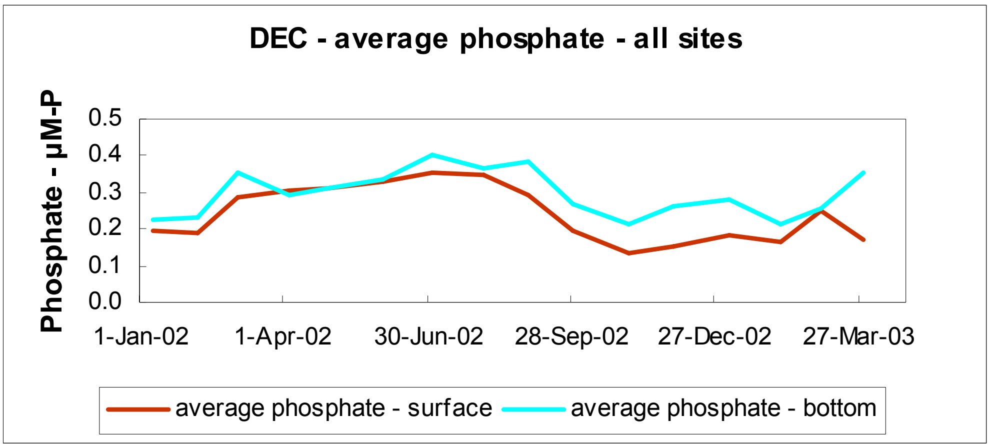


Figure. 3.9. Surface and bottom phosphate concentrations, averaged across 12 sites in the D'Entrecasteaux Channel.

Silicate concentrations near the bottom did not vary substantially throughout the sampling period, with a broad peak in winter 2002 (Fig. 3.10). In contrast, surface concentrations showed a sharp peak at almost 10 µM in June 2002 with indications another peak would probably occur in winter 2003. These winter peaks are presumed to be caused by inputs of silicate from river discharge as they are in the Huon Estuary. Quite low surface concentrations of silicate were observed in November 2002 and January 2003 reflecting local consumption, presumably by diatoms.

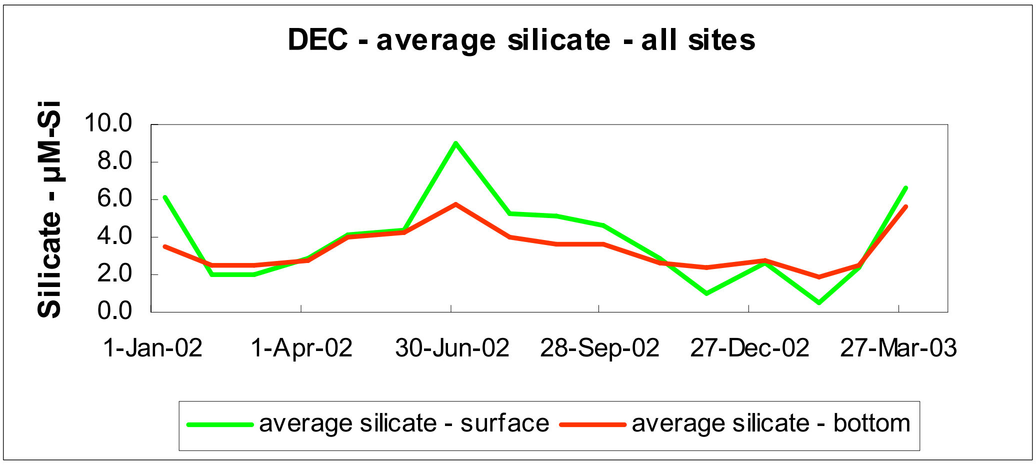


Figure 3.10. Surface and bottom silicate concentrations, averaged across 12 sites in the D'Entrecasteaux Channel.

The nutrient concentrations observed in D'Entrecasteaux Channel indicated similar seasonal patterns to those observed in the Huon Estuary during 1996–1998. Nitrate and phosphate concentrations peaked during the winter intrusion of offshore waters that originated from depth.

3.3.1. 2004–2005 nutrient measurements in the D'Entrecasteaux Channel and Huon Estuary

Nutrient sampling recommenced on September 20th, 2004 at 11 sites (1, 2, 3, 4, 5, 8, 10, 12, 13, 14, 15) in the combined Huon Estuary and D'Entrecasteaux Channel (Fig. 3.4) and continued until June 8th 2005. Approximately monthly sampling produced a total of 262 samples from the surface and near-bottom at each site. Nitrate, phosphate and silicate concentrations were all significantly different (Wilcoxin signed ranked test, P < 0.001) between surface and bottom (Fig. 3.11).

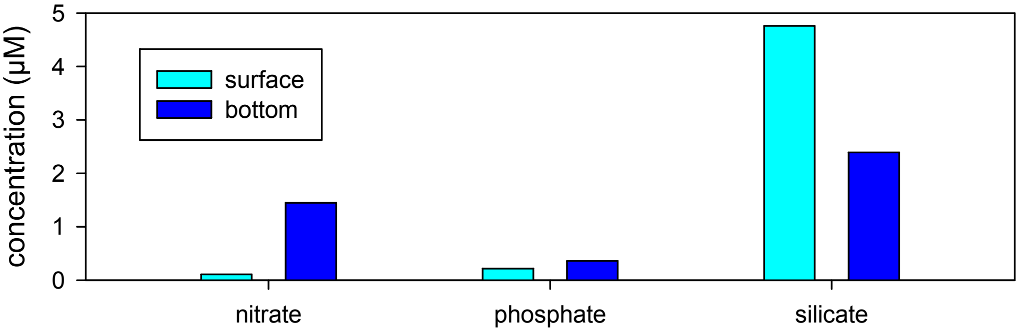


Figure 3.11. Median nutrient concentrations at the surface and near bottom from 11 sites (Fig. 3.4) in the combined Huon Estuary and D'Entrecasteaux Channel during 2004-2005.

The median nitrate concentration in the surface was 0.111 µM rising to 1.45 µM near the bottom. Median phosphate concentrations were 0.22 µM at the surface and 0.36 µM near the bottom. Silicate concentrations were greater at surface with median concentrations of 4.76 falling to 2.39 near the bottom.

Averaged surface nutrient concentrations from all 11 sites showed strong seasonal trends. The most pronounced temporal pattern was a significant (P < 0.001) decrease in nitrate at all sites to a summer mean of 0.023 ± 0.176 µM, or more than 2 orders of magnitude lower than the mean winter concentration (Fig. 3.12). Mean surface phosphate concentrations were also significantly (P < 0.001) reduce during summer but these fell by only 50% relative their winter mean. Surface silicate concentrations did not show a significant (P = 0.063) seasonal change although they followed a similar trend to phosphate and nitrate by being lowest during summer.

Considering the surface nutrient concentrations from the perspective of potentially limiting nutrient(s) the calculated molar N:P ratio was always much less than the 16:1 (Fig. 3.12) which would provide balanced growth for phytoplankton (Redfield et al., 1963). The N:P ratio reached its most extreme value of 0.096 during summer, suggesting nitrogen was ~ 170 times more limiting than phosphate. The ratio of silicate to nitrate ration required for a range of diatom species is ~ 1:1 (Redfield et al., 1963). The median surface silicate concentration was 44 times greater than that for nitrate suggesting that silicate was always present in excess.

The only nutrient observed to have a significant spatial pattern in surface concentrations was silicate (Fig. 3.13). Silicate was significantly (P < 0.001) greater at locations up the Huon Estuary relative to those in the D'Entrecasteaux Channel.

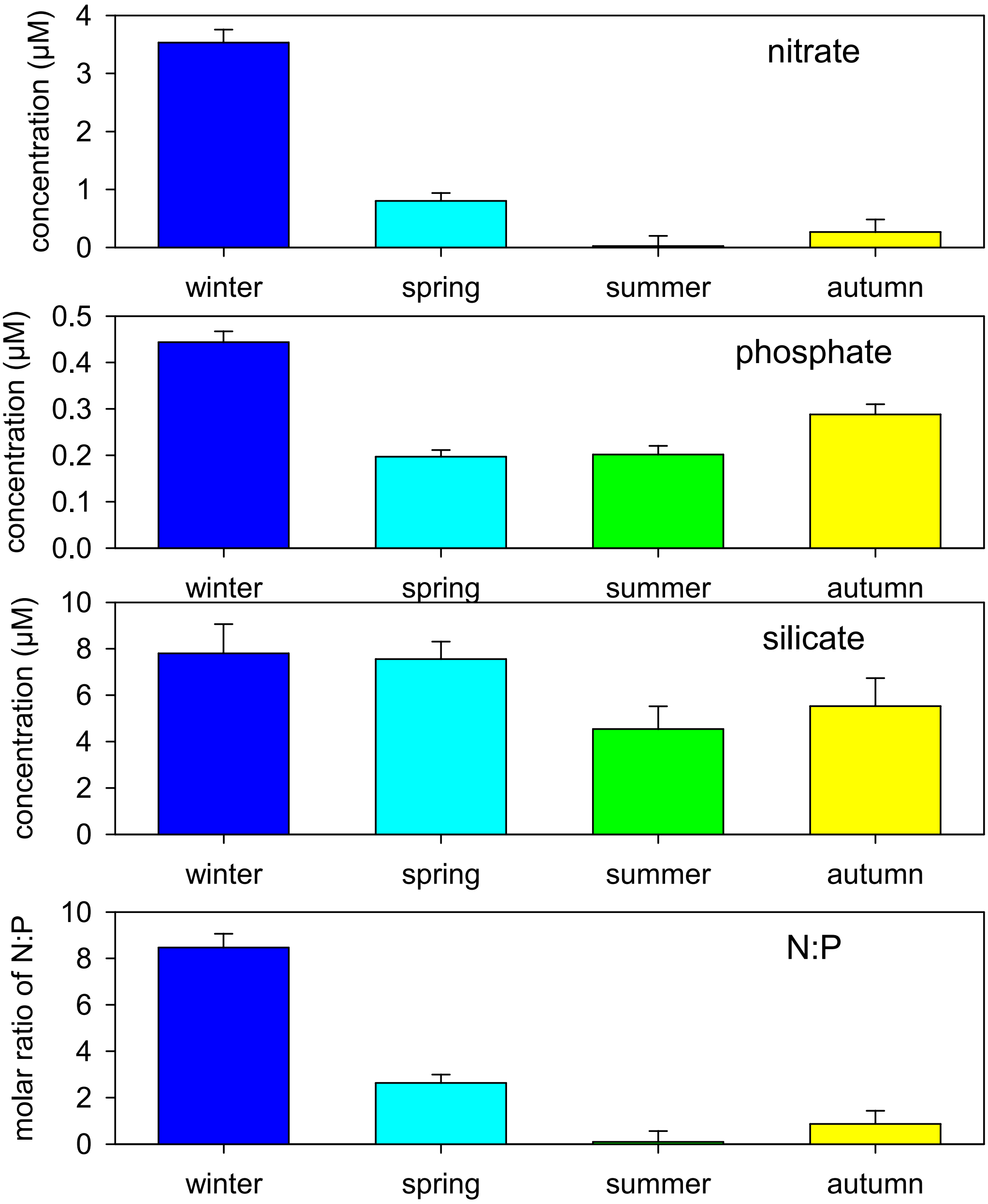


Figure 3.12. Mean surface nutrient concentrations at all 11 sites (Fig. 3.4) during the 4 seasons of the year (winter = June, July, August; spring = September, October, November; summer = December, January, February; autumn = March, April, May) in the combined Huon Estuary and D'Entrecasteaux Channel during 2004–2005.

From the 2004–2005 sampling we can conclude that nitrate and phosphate concentrations peaked during the winter intrusion of offshore waters originating from depth. Silicate concentrations were greater in the Huon Estuary than in D'Entrecasteaux Channel. Nitrogen was the nutrient most likely to limit phytoplankton growth, especially during summer.

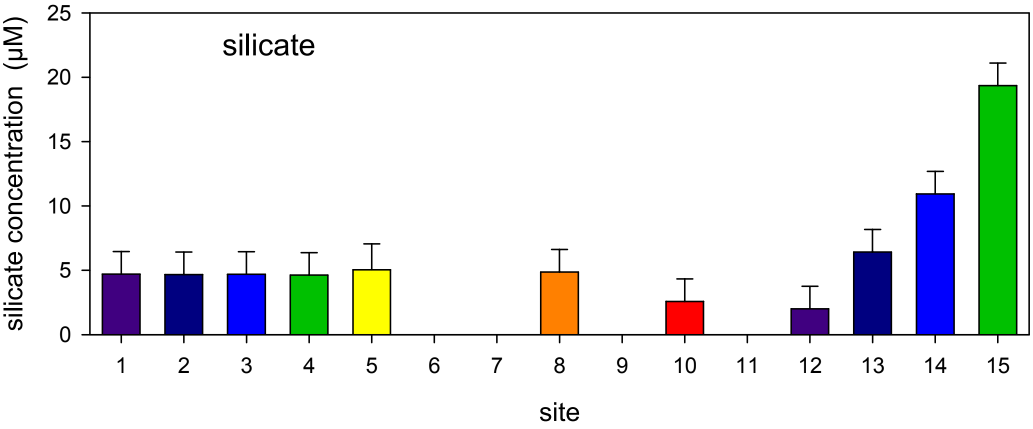


Figure 3.13. Mean +1 standard error surface silicate concentrations at each of the 11 sites (see Fig. 3.4 for locations) in the Huon Estuary and D'Entrecasteaux Channel during 2004-2005.

3.4. A brief review of sewage inputs

Points of sewerage discharge in the Huon Estuary are: Cygnet, Ranelagh, Geeveston, Kermandie and Dover. In all cases, the sewerage is secondary treated, but nutrient loads are not measured. The sewage treatment plants (STP) in the Huon Estuary provide sewerage services for about 13,000 people. Samples collected during the Huon Estuary Study (Butler et al., 2000) showed elevated nutrients in the Kermandie River downstream of the STP at Geeveston. Silicate and especially ammonium concentrations were highly elevated relative to other similar river inputs to the Huon Estuary. The lack of flow data for the Kermandie River makes it difficult to estimate the total loads into the Huon Estuary from this source. The input of these nutrients into the Huon Estuary at Hospital Bay is likely to be influenced by local rainfall.

In the D'Entrecasteaux Channel most STPs discharge secondary treated waste water. Local discharge points are: Electrona (~ 1100 person capacity (pc)), Margate (lagoon treatment ~1500 pc), (Howden (tertiary treatment ~300 pc) and Woodbridge (secondary treatment ~110 pc). Nearby communities discharging into waters just north of D'Entrecasteaux Channel include Blackmans Bay (17,500 pc but 18,000 live in the area) and Taroona (3000 pc). The presently connected population (Woodbridge to Taroona) is 23,200 persons (year 2005) and is projected to increase to 28,600 persons by 2010 (4.3% pa growth) and to 44,000 persons by 2030. The present total average daily flow for these wastewater plants is 5.3 Ml per day (year 2005) and is projected to increase to 6.6 Ml per day by year 2010 (i.e. 25% increase) and 10.3 Ml per day by year 2030, nearly double the present flow.

With the present number of people connected to the sewage system (Taroona to Dover) the total load of N into the Huon Estuary and D'Entrecasteaux Channel region from these STPs should be on the order of 120 tonnes (Baskin, 2003) per year.

STP nutrient loads into the Huon and D'Entrecasteaux are small in comparison to marine, river and farm loads into the region and are likely to exhibit less seasonal variation.

3.5. A brief review of fish farm N inputs

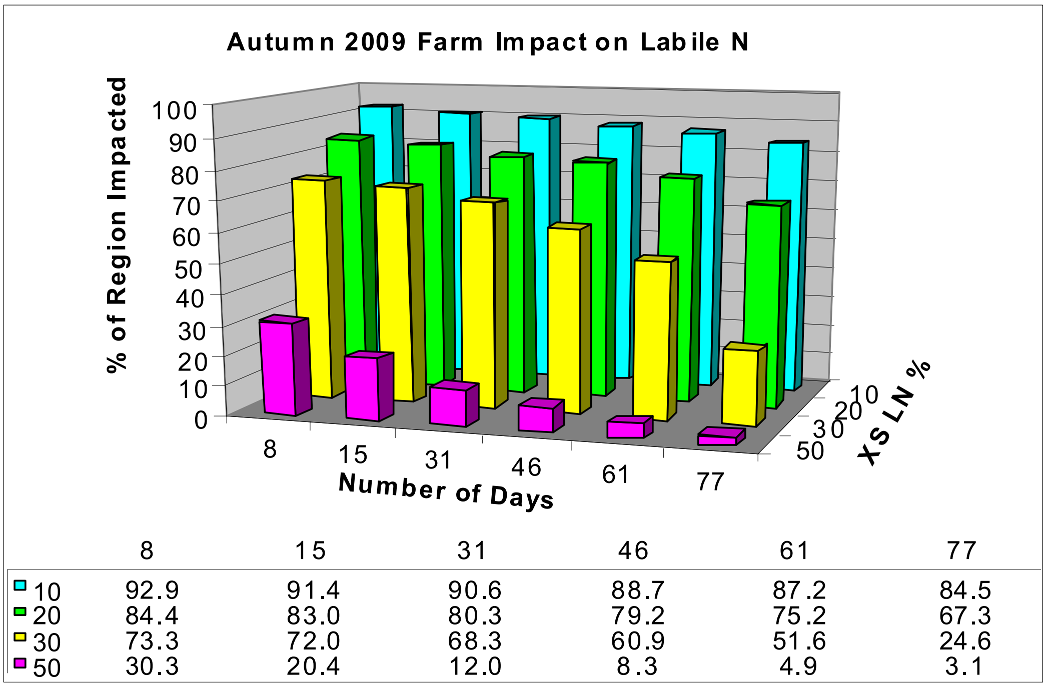
In Tasmania the Huon Estuary and more recently the D'Entrecasteaux Channel have been important regional centres for the growth of salmonid aquaculture since the late 1980s. Nitrogen is added as fish food and the load to the environment is calculated as N in fish feed input minus N in harvested salmon. Given improvements in feed technology and food conversion ratios the fish are presumed to consume the vast majority of the supplied food with very little directly reaching the bottom (Macleod et al., 2004; 2006, 2007; Crawford et al., 2001; Crawford, 2003) and approximately 86% of the N input into the environment is presumed to be excreted by the fish in a soluble form (Gowen and Bradbury, 1987). The majority of this is expected to be ammonium (once in seawater) and ammonium is a preferred N source for phytoplankton and some other aquatic plants.

The amount of nitrogen added to the marine environment as farm waste in 2002 was estimated to be 843 tonnes across both the Huon Estuary and D'Entrecasteaux Channel (Table 3.2; data from Tasmanian Department of Primary Industries and Water and adjusted for food conversion efficiency, N harvested as fish, solid versus dissolved waste). Of the 843 tonnes of added nitrogen approximately 120 tonnes was in particulate form and 723 tonnes was soluble waste. Farm waste is discharged throughout the year, but has the largest impact on the marine ecosystem in summer and autumn when it augments naturally depleted surface nutrient concentrations and enables additional phytoplankton growth. The salmon aquaculture industry would like to expand within the Huon and D'Entrecasteaux Channel. It has plans to grow more fish and in doing so would release more waste products. If the planned expansion eventuates the amount of finfish waste nitrogen being input into the ecosystem is expected to rise from 843 to 2590 tonnes, or ~ 210% from 2002 to 2009 (DPIW).

The 3D biogeochemical model combined with the hydrodynamic model can be used to estimate the impact of this proposed rise in feed input on the concentrations of labile N within the region. The results are complex but can be summarized spatially and temporally to show, for example, that 75% of the region will experience a 20% increase in depth integrated labile nitrogen for 2 months in autumn (Fig. 3.14), relative to the same environment with no farm inputs. More details of the biogeochemical modelling can be found in Chapter 7.

Table 3.2. Total nitrogen and phosphorus inputs as fish feed in 2002 and the projected rise to 2009. Data supplied by the Tasmanian Department of Primary Industries and Water and adjusted for food conversion efficiency, N harvested as fish, and solid versus dissolved waste.

	Total N	Total P
	tN y^{-1}	tP y^{-1}
2002		
Total Huon	313	54
Total Channel	530	92
Total Farms	843	146
Projected increase 2002 to 2009		
Total Huon	229	41
Total Channel	1518	267
Total Farms	1747	308



	8	15	31	46	61	77
🟦 10	92.9	91.4	90.6	88.7	87.2	84.5
🟩 20	84.4	83.0	80.3	79.2	75.2	67.3
🟨 30	73.3	72.0	68.3	60.9	51.6	24.6
🟪 50	30.3	20.4	12.0	8.3	4.9	3.1

-

Figure 3.14. Modelled effects of potential 2009 farm inputs on background nutrient concentrations.

3.6. A brief review of nutrient inputs from sediments

In shallow coastal ecosystems the recycling of nutrients from sediments can be an important source supporting autotrophic growth especially during periods when supply from other sources is low. A summary of results is presented here, but full details are available in the technical reports by Holdsworth et al. (2008), Revill et al. (2008) and Macleod et al. (2008) and Chapter 6. The vertical flux of nutrients to the sediments was measured during several specialized process studies conducted in the Huon Estuary, North West Bay and Port Esperance. During the Huon Estuary process studies the sediment trap data indicated significant differences in the amount of material being deposited during April 2005 than during September, 2005 (Revill et al., 2008). There was a much greater deposition rate of material that was more organic rich during April 2005.

In a sandy and more exposed coastal environment in South Australia the benthic fluxes of NH_4^+ from the sediment to the water column were ~ 2.2 mmol m^{-2} d^{-1} (Lauer et al., 2007). They ranged from negative (flux into sediments) to ~20 times greater at other locations in Australia and around the world (Table 3.3). In the Huon Estuary the average effluxes of ammonia, nitrate, phosphate and silicate were 1.3, 10, 1.6 and 33 µmol m^{-2} h^{-1} or several orders of magnitude lower (Chapter 6) than in South Australia. On all occasions the DIN flux in the Huon Estuary was dominated by nitrate.

Table 3.3. Benthic nutrient fluxes reported in other estuarine or marine studies.

Location	NH_4 (mmol m^{-2} h^{-1})	NO_3+NO_2 (mmol m^{-2} h^{-1})	FRP (mmol m^{-2} h^{-1})	Source
Humber Estuary	–0.28 to 1.1	–1.8 to 0.63	–0.046 to 0.092	Mortimer et al. (1998)
Tamar Estuary	–0.028 to 0.171	–0.283 to 0.37	0 to 0.033	Watson et al. (1995)
North Carolina estuaries	0 to 0.45	0 to 0.006	–0.008 to 0.046	Fisher et al. (1982)
Port Phillip Bay, Victoria	0.004 to 0.5	–0.05 to 1	0.002 to 0.079	Nicholson et al. (1996)
Lake Illawarra, NSW	0.458 to 2.63	0.04 to 0.31	–0.04 to 0	Qu et al. (2003)
Port River Day	–0.67 to 2.75	–1.46 to 0.133	–0.386 to 0.203	Jenkins (2005)
Port River Night	0 to 3.21	–1.46 to 0.089	–0.383 to 0.241	Jenkins (2005)

Extrapolation of these measurements (from Chapter 6) to the whole Huon Estuary suggests that the sediments were only a minor source of nutrients, providing approximately 96 tonnes of inorganic nitrogen, 32 tonnes of phosphate and 586 tonnes of silicate per annum. Similarly the time required for nutrients from the sediments to replace those in the water column (turnover times) were much longer in the Huon Estuary than other places in Australia and around the world. In South Australia estimated turnover times for ammonium were less than 1 day, similar to those observed in the Potomac River Estuary (Callender and Hammond, 1982). In the Huon Estuary turnover times averaged 430, 367, 846 and 246 days for ammonium, nitrate, phosphate and silicate, respectively.

3.7. Comparing nutrient concentrations across ecosystems

Data from several temperate Australian water bodies were used to provide some context for the nutrient status of the Huon Estuary and D'Entrecasteaux Channel (Fig. 3.15). Mean annual concentrations were lowest in D'Entrecasteaux Channel < Huon Estuary < Derwent Estuary (data supplied by the Derwent Estuary Program for 3 stations at the seaward end of the Estuary) < Swan River (WA), Canning River (WA). Concentrations of ammonium are useful indicators of nitrogen cycling. High concentrations are not common in Australian ecosystems but can occur where there are very high inputs (sewage discharge or very intense agriculture) or poor N cycling due to hypoxic or anoxic conditions in the water column or sediments. In the Huon Estuary dissolved oxygen concentrations tend to be lowest in late summer, simultaneous with high ammonium concentrations (Butler et al., 2000). The strong positive relationship between summer ammonium and summer chlorophyll-*a* would support hypothesis that the ammonium added to the water column by this process can stimulate an algal bloom in these N-limited waters.

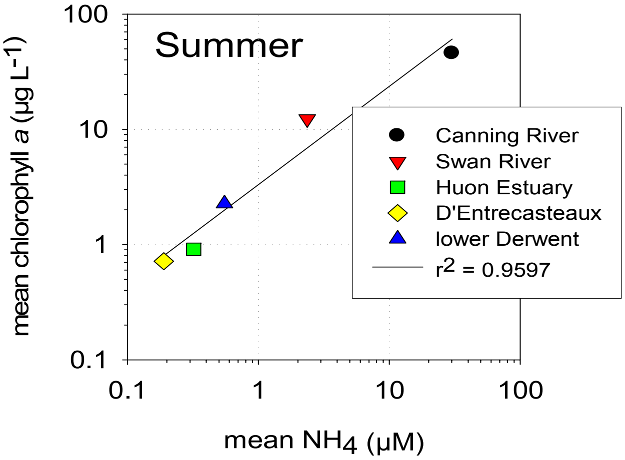


Figure 3.15. Mean summer (Dec, Jan, Feb) NH_4^+ concentration and mean summer chlorophyll-*a* for 5 different temperate water bodies in Australia.

3.8. Summary annual N budget

The nitrogen (N) budget of the ecosystem was estimated from the 3D biochemical model (Chapter 2) coupled to the 3D hydrodynamic model (Chapter 7). The resulting budget is an accurate assessment of the exchange of water between the Huon Estuary and D'Entrecasteaux Channel with the adjacent water bodies (Derwent Estuary and Storm Bay). Because the nutrient budget incorporates a biogeochemical model it includes many different biogeochemical transformations within the Huon Estuary and D'Entrecasteaux Channel and is, potentially, less well constrained than the physics. The nutrient budget was calculated using nutrient input data from

2002–2003 in conjunction with the calibrated and validated hydrodynamic and biogeochemical models run for that time period. It is estimated that about 2746 tonnes of nitrogen per year are exported in surface outflows from the Huon and Channel while exchange at depth imports ~ 1546 tonnes (Fig. 3.16). The major conclusion from this budgeting exercise is that the Huon and D'Entrecasteaux Channel are substantial net exporters of nutrients, sending ~ 1197 tonnes of N into the neighbouring marine environments each year.

3.9. Changes from 1996–1998 to 2002–2003

Relative to the N budget produced during the Huon Estuary Study (HES; Butler et al., 2000) the new budget includes both the Huon Estuary and D'Entrecasteaux Channel. This geographic expansion of the area included in the budget makes direct comparisons complicated. Furthermore the 2002–2003 year was different from the HES period (1996 to 1998) in terms of river flow. Based on the gauging station at Frying Pan Creek the major fresh water input from the Huon River in 2002 was ~ 50% greater than the average flow during 1996–1998. Measurements of nutrients in the Huon River in 2002–2003 were extremely sparse but suggest that concentrations might have risen from previous years; unfortunately, given the inherent variability in these sorts of measurements and the sparse data sets it is difficult to be confident in this result. Regardless, the product of increased flow and the available data on nutrient concentrations, plus the addition of freshwater flows into the D'Entrecasteaux Channel, mean that the estimated total N input from freshwater rose to ~ 1118 tonnes per annum from ~ 600 tonnes in 1996–1998. Other annual nutrient inputs have also risen due to increased population in the expanded region (up by ~ 100 tonnes N) and increased salmon production (up by ~ 82 tonnes N).

3.10. Acknowledgements

We thank our CSIRO and TAFI colleagues, including the Huon Estuary Study Team, for their contributions. In particular we acknowledge Simon Wilcox, Sam Foster, Kate Berry, Ros Watson and Val Latham for their care and diligence in sampling and analysis.

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Chapter 4

Phytoplankton distribution and autecology in the Huon River Estuary and D'Entrecasteaux Channel

4.1. Introduction

The phytoplankton ecology of the Huon River Estuary and D'Entrecasteaux Channel is a key component of the region's environmental status, influencing both marine ecosystem health and also its value for both recreation and aquaculture and other commercial activities. Increasing anthropogenic pressures were recognized as having the potential to increase eutrophication (Butler et al., 2000), and consequent environmental degradation. Phytoplankton are the key vector for the transformation of the nutrients in these water bodies from a dissolved to particulate form (Lohrenz, 2003). The type, abundance and distribution of phytoplankton that develops in response to increased anthropogenic nutrient loading are important in determining the severity of any negative environmental impacts (Smayda, 1997).

An important feature of the phytoplankton community in the Huon River Estuary since the mid 1980s is the formation of harmful algal blooms (HABs) of the introduced toxic dinoflagellate *Gymnodinium catenatum* during summer and autumn-winter (Hallegraeff and Sumner, 1986; Hallegraeff et al., 1995; McMinn et al., 1997; Butler et al., 2000). Large blooms of *G. catenatum* have also been reported in the D'Entrecasteaux Channel (Hallegraeff et al., 1995), and in the Derwent Estuary and nearby coastal waters (Coughanowr, 1997). In some years, *G. catenatum* HABs have caused shellfish farms to close for up to half of the year because of the high levels of paralytic shellfish toxins (Hallegraeff et al., 1995).

While much is known about many of the factors regulating these blooms (e.g. Butler et al., 2000), there remain other factors that are poorly understood, making modelling and prediction, and consequently management, difficult. As well, in recent years new HAB-forming microalgae have bloomed in the region. Notable is the large predatory dinoflagellate *Noctiluca scintillans* that has apparently extended its range from eastern Australia waters to become resident in Tasmanian waters. It is likely to have been introduced by the East Australian Current (Hallegraeff, 2002; Dela-Cruz, 2003). The impact of this species on the phytoplankton dynamics in the Huon Estuary and D'Entrecasteaux Channel is unknown.

This chapter presents information on the phytoplankton distribution and ecology in the Huon Estuary and D'Entrecasteaux Channel, examining the similarities and differences between these two areas, including long term trends in the region. Through this system-wide information as well as detailed field studies of phytoplankton and other processes over short periods up to several days in selected locations, we provide new knowledge and understanding of the temporal and spatial regulation of phytoplankton dynamics. For the ecologically significant HAB species, a combination of laboratory experimental approaches and field investigations has yielded important new understanding of the factors that influence the development of blooms. Overall these studies have facilitated improved parameterisation of the biogeochemical models in the CSIRO Environmental Modelling Suite (CSIRO–EMS) and the development of a proposed Monitoring Program for the Huon River Estuary and D'Entrecasteaux Channel.

4.2. Material and Methods

4.2.1. Temporal and spatial sampling for phytoplankton in the Huon Estuary and D'Entrecasteaux Channel

The temporal and spatial sampling for phytoplankton in the Huon Estuary and D'Entrecasteaux Channel has varied in terms of method, location and intensity. In the Huon Estuary Study (HES, Butler et al., 2000) the Huon Estuary only was sampled approximately weekly at 5 sites (1996–1998), or quarterly at about 20 sites (1996−1997) (Fig. 4.1). For this project, sampling occurred mainly at monthly intervals at a variable number of sites that also varied in their location (2000–2005). Sampling consisted of CTD casts and either discrete bottle samples at fixed and variable depths, or integrated samples using a 12 m long tube. Samples for cell counts were preserved in acid Lugol's iodine solution (Parsons et al., 1984). Samples for pigments were filtered onto Whatman GF/F filters (nominal pore size ~ 0.7 μm) and stored in liquid nitrogen prior to analysis.

4.2.2. Process studies

Process studies were undertaken in three locations – mouth of the Huon River Estuary, North West Bay and Port Esperance – to gain in-depth knowledge of biochemical fluxes on short time scales. Samples for the analysis of nutrients, phytoplankton speciation, pigment composition and concentration and the absorption properties of the dissolved and particulate fractions of the water column, micro and mesozooplankton and sediments were collected. Six sites in the mouth of the Huon River (Fig. 4.2) were studied during two process studies in autumn (12th and 13th April) and spring (20th and 22nd September) 2005. The sampling sites were close to, but not within, an area leased for salmon farming. For further details see Clementson et al. (2008a). Eight sites in North West Bay (Fig.4.3) were studied during 3rd and 4th October, 2006, and 8 sites in Port Esperance during 1st and 2nd November, 2007. In North West Bay a diel experiment was conducted at station 2 for a 23 hour period over 3rd and 4th October 2006. For further environmental and sampling details see Clementson et al. (2008b). The processes studied during these short but intensive sampling efforts included the uptake of different forms of nitrogen by phytoplankton from several depths sampled at midnight and midday; microzooplankton grazing and the impact of selective grazing on phytoplankton dynamics; mesozooplankton distribution, abundance and grazing. These results are given in Armstrong (PhD Thesis in preparation), Thompson et al. (2008) and Swadling (2008).

4.2.3. Pigment analyses

Up to 3 litres of water was filtered through a 47 mm diam. glass fibre filter (Whatman GF/F). The filter was then stored in liquid nitrogen until analysis. Samples were extracted over 15 to 18 hours in an acetone solution before analysis by HPLC using a C_8 column and binary gradient system with an elevated column temperature following a modified version of the Van Heukelem and Thomas (2001) method. Pigments were identified by retention time and absorption spectrum from a photo-diode array (PDA) detector and concentrations of pigments were determined from commercial and international standards (Sigma, USA; DHI, Denmark). A more detailed description of this method can be found in the associated technical report by Clementson et al. (2008a). Where data from this report are compared to data from the Huon Estuary Study and previous D'Entrecasteaux Channel studies, the methods used for pigment and microscopic analysis have been reported in the appropriate reports (Butler et al., 2000; Volkman et al., 2006).

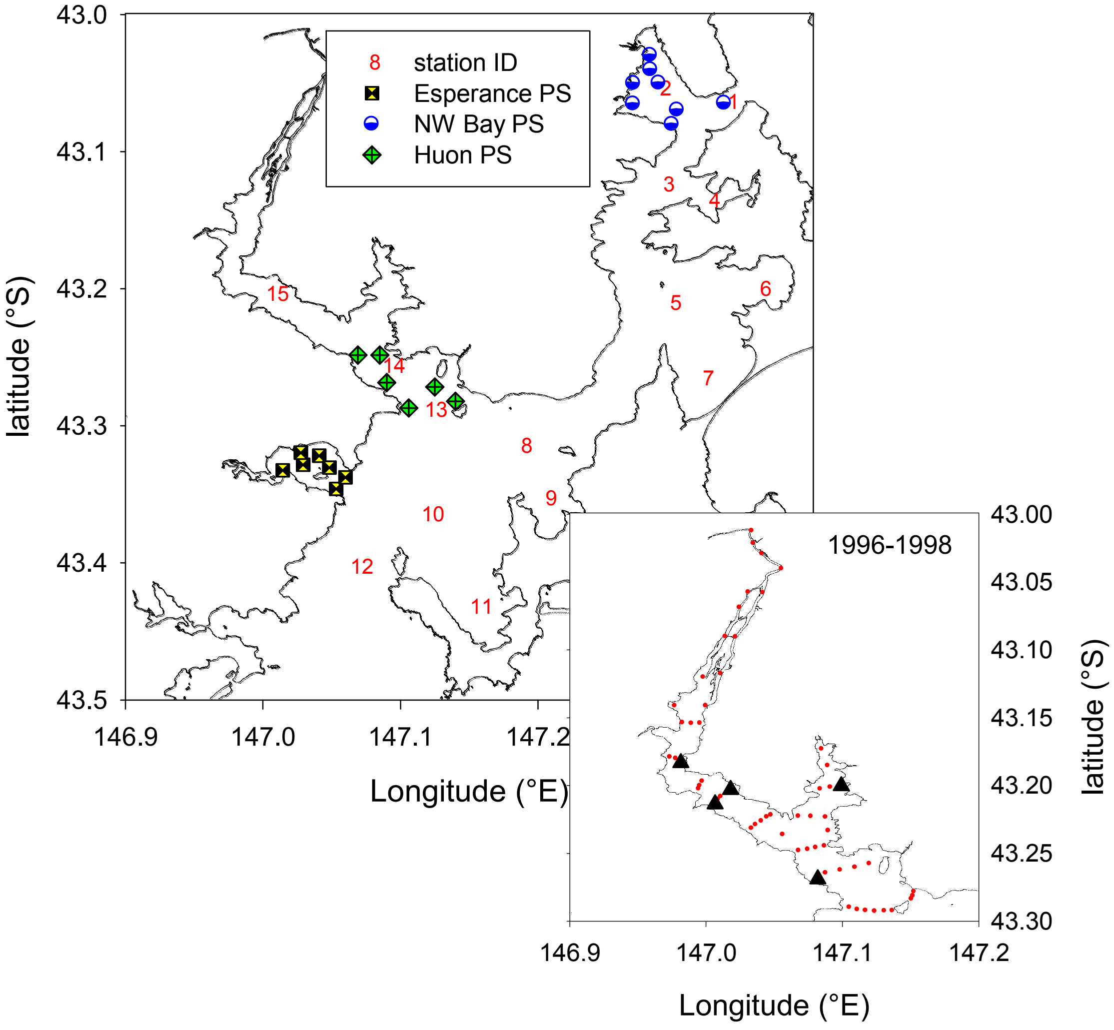


Figure 4.1. Sample sites in the Huon Estuary and D'Entrecasteaux Channel. Sites (1–15) were sampled from 2001–2005 (not continuously and not all sites sampled at all times) and are identified by number in other Figures and text. From HES study, (1996–1998) sites (▲) were sampled weekly while (+) were sampled quarterly.

4.2.4. Phytoplankton identification, cell counts and biovolumes

Phytoplankton samples of ~ 1 l were preserved in Lugol's solution, concentrated by repeated settling and 1 ml was then examined in a Sedgwick Rafter counting chamber using an inverted microscope. For most microplankton (cells generally larger than 20 µm diameter) at least 10% of a single slide was enumerated at 100× magnification (except when there were dense blooms of one or more microplankton species, when at least 20% was scanned). For nanoplankton, (2 to 20 µm in diameter) the chamber was examined at 400× magnification until at least 200 cells of the dominant nanoplankton "taxon" had been counted. Small flagellates (< 10 µm in diameter) in the nanoplankton were assigned to a single group (reported as "small flagellates"). Microzooplankton, including non-photosynthetic dinoflagellates, were identified by reference to literature information and corroborated by examining hundreds of live samples where pigmented versus non-pigmented cells were easily distinguished. The biovolumes of counted cells were estimated from standard geometries detailed in Hillebrand et al. (1999). Five species of *Skeletonema* have been reported from south-east Australia (Kooistra et al., 2008). Because high resolution microscopy or molecular screening is needed to correctly identify which species is present, for the purpose of this report *Skeletonema* is reported as *Skeletonema* spp.

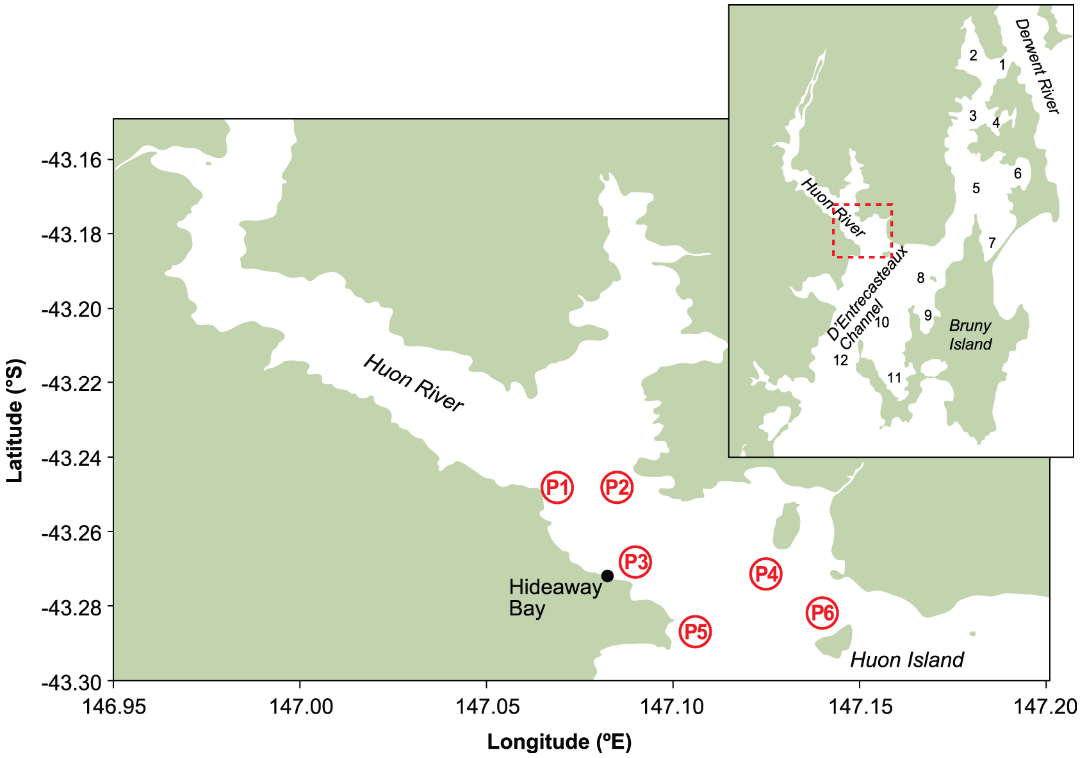


Figure 4.2. Map of the study area with station locations used during the process studies at the mouth of the Huon River Estuary in April 2005 and September 2005. The numbers in D'Entrecasteaux Channel on the inset map refer to the sites sampled during the monthly monitoring studies of 2002–03. Sites 6, 7, 9 and 11 were not sampled in 2004–05.

4.2.5. Particulate, detrital and CDOM absorption

Methods used to determine the absorption coefficients for the dissolved and particulate fractions of the water samples followed NASA protocols (Pegau et al., 2003) and are described in detail in the technical report by Clementson et al. (2008a).

4.2.6. Nutrient analysis

Prior to analysis, samples collected for nitrate/nitrite and phosphate analyses were allowed to thaw before equilibrating to room temperature along with the samples collected for silicate analysis. Each of the tubes was well mixed and placed in the auto-sampler of a Lachat QuickChem 8000 series Automated Ion Analyser. The concentration of each of the nutrients was determined using QuickChem methods as follows: nitrate/nitrite (31-107-04-1-A), phosphate (31-115-01-1-G) and silicate (31-114-27-1-D) which had been calibrated against standard solutions.

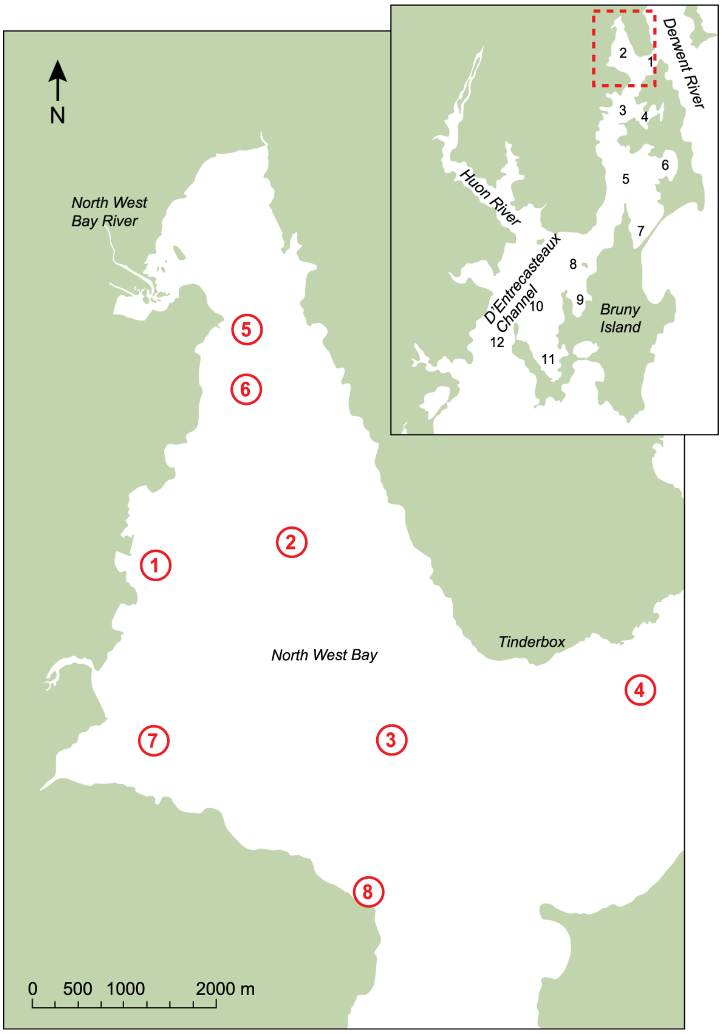


Figure 4.3. Map of North West Bay with site locations used during the process study in October 2006. The numbers in D'Entrecasteaux Channel on the inset map refer to the sites sampled during the monthly monitoring studies of 2002–03. Sites 6, 7, 9 and 11 were not sampled in 2004–05.

4.2.7. Statistical treatment of the data

A rigorous assessment of trends in phytoplankton populations over time requires that the spatial and temporal sampling is consistent, or that variations can be controlled for, post sampling, by ‘blocking’ or ‘binning’. For example having 10 samples in winter and 1 in summer during one particular year will not provide a meaningful comparison with another year where the seasonal sampling was reversed. These sorts of temporal problems can be addressed by binning samples by time so that the 10 observations become a single mean value for winter. Similarly having 10 samples in 1 location and 1 in another will not allow a meaningful comparison between years if the sampling intensity in the 2 locations was later reversed. Again this can be handled by binning the data to produce a single mean for each location. Therefore the data sets were blocked or binned to overcome the different spatial and temporal sampling regimes. Where noted, data were pooled into months or seasons (summer = December, January, February; autumn = March, April, May; winter = June, July, August; spring = September, October, November), and multiple depths were pooled to give a water column average. To facilitate inter-annual comparisons, data were also pooled into years that were defined as August to July (e.g. 1996–97 = August 1996 to July 1997). Some temporal trends were extracted from the data sets by using a running average, running median or locally weighted regression that smoothed over 10% of the available data to estimate any given point. If the data were normally distributed and homogeneous, or could be made normally distributed and homogeneous by transformation, they were analyzed by ANOVA or a general linear model followed by a standard post hoc test where alpha was adjusted for multiple comparisons (Bonferroni *t*-tests). Except where noted, non-parametric techniques were used if the data could not be transformed to be homogenous and normally distributed. Where possible, data from the Huon Estuary Study (Butler et al., 2000) were used to extend the time series analyses.

4.2.8. Laboratory culture studies of HAB species

Microalgal strains were obtained from the CSIRO Collection of Living Microalgae http://www.cmar.csiro.au/microalgae/, and, in the case of *Noctiluca scintillans*, isolated from natural blooms. Bacterial strains, as isolated and reported in Green et al. (2004) were kindly provided by Dave Green, Scottish Association for Marine Science. Controlled laboratory studies of *G. catenatum* used culturing, growth estimation and analysis methods established in previous studies (e.g. Blackburn et al., 2001; Doblin et al., 1999, 2000). Algal-bacterial interactions studies used methods described by Bolch et al. (2004). New culturing techniques for *N. scintillans* are described in detail in Holmes (2005).

4.3. Results and discussion

4.3.1. Temporal and spatial patterns of phytoplankton distribution the Huon River Estuary and D’Entrecasteaux Channel: 1996 to 2005

From the approximate monthly sampling that commenced in 2001 there were 393 measurements of chlorophyll-*a* using a mixture of surface and integrated samples at 15 stations (Fig. 4.1). The time series of chlorophyll-*a* concentration versus date showed very little apparent pattern or long term trends (Fig. 4.4). Analysis by site showed that D’Entrecasteaux Channel sites tended have a lower concentration of chlorophyll-*a* than sites in the Huon Estuary (Fig. 4.4) with the lowest average concentration at site 6 (Great Bay, Bruny Island). The average concentrations tended to rise sharply with distance up the Huon Estuary (sites 10, 13–15).

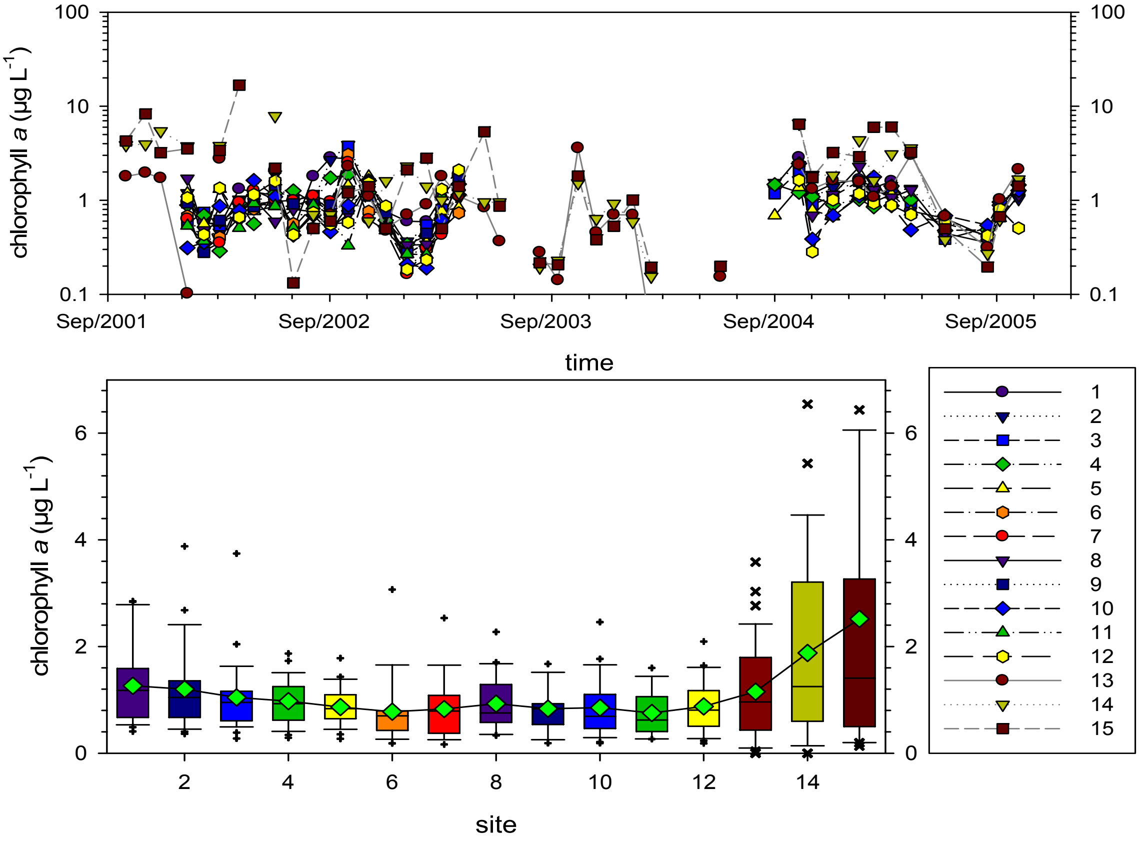


Figure 4.4. Chlorophyll-*a* concentrations in the Huon Estuary and D’Entrecasteaux Channel measured from 2001 to 2005. Sites are identified in Fig. 4.1. For box plots the edges of the boxes encompass from the 25th percentile to the 75th percentile while the capped bars enclose the 10th to 90th percentiles and the outliers are shown as (+ or x). The mean for each station is also shown (◆).

There were identifiable temporal patterns in phytoplankton abundance (assessed by concentration of chlorophyll-*a*) on a seasonal basis (Fig. 4.5). Spring blooms tended to occur in October when average chlorophyll-*a* concentrations rose sharply above 2 μg l^{-1}. Autumn blooms were evident in March and April, but tended to be greatest in May.

The major phytoplankton species were averaged across all 15 sites to examine regional temporal variation. There were few patterns evident in a simple time series of these major species (Fig. 4.6). Most species showed some decline during winter.

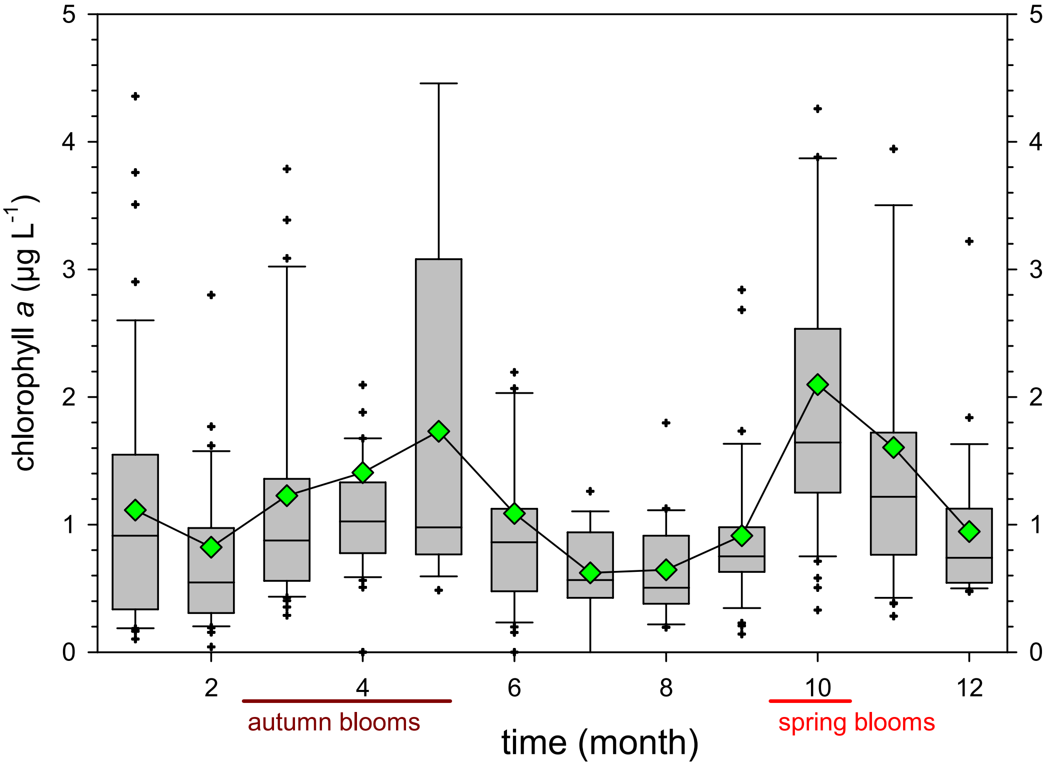


Figure 4.5. Chlorophyll-*a* concentrations in the Huon Estuary and D'Entrecasteaux Channel measured from 2001 to 2005 averaged and plotted by month. The edges of the boxes encompass from the 25th percentile to the 75th percentile while the capped bars enclose the 10th to 90th percentiles and the outliers are shown as (+). The mean for each month is also shown (◆).

The Tasmanian Department of Health and Human Services conducts the Tasmanian Shellfish Quality Assurance Program (TSQAP) to ensure the safety and regulatory compliance of Tasmanian shellfish. This involves phytoplankton identification and counts from shellfish farms State-wide by NATA-accredited phytoplankton taxonomists (Analytical Services Tasmania, AST). In the Huon Estuary and D'Entrecasteaux Channel region, TSQAP has samples sites in Recherche Bay, Hastings Bay, Cloudy Bay Lagoon, Little Taylors Bay, Port Esperance, Deep Bay, Fleurty's Point and Great Bay. In general TSQAP sampling intensity varies depending upon the known risk of toxic blooms which is dependent upon site history and recent cell counts. In the Huon Estuary and D'Entrecasteaux Channel region there is monthly sampling at times of low risk rising to weekly sampling at times of high *G. catenatum* abundance.

TSQAP kindly made available their last 8 years of data on *G. catenatum* from the Huon/Channel region representing more than 500 cell counts from a mixture of fixed depth (usually 0.3 m) and integrated (0–10 m) samples. The resulting data show considerable variation between sites and years with a notable reduction in *G. catenatum* during 2006 (Fig. 4.7). The mean value was 490 cells l^{-1} (although the detection limit is ~ 100 cell l^{-1} for the purpose of calculating a mean, no detected cells = 0). The further analysis of the cell density data for spatial or temporal characteristics (e.g. site means by season or year) is complicated by the temporally variable sampling regime triggered by cell density, but suggests that *G. catenatum* blooms were more prevalent during 2002 than 2006, 2007 and more persistent in Deep Bay and Port Esperance than at the other locations. Inter-annual variation appears to be a feature of *G. catenatum* blooms, having previously been identified from analyses of bloom-associated shellfish toxicity during 1986–1994 (Hallegraeff et al., 1995).

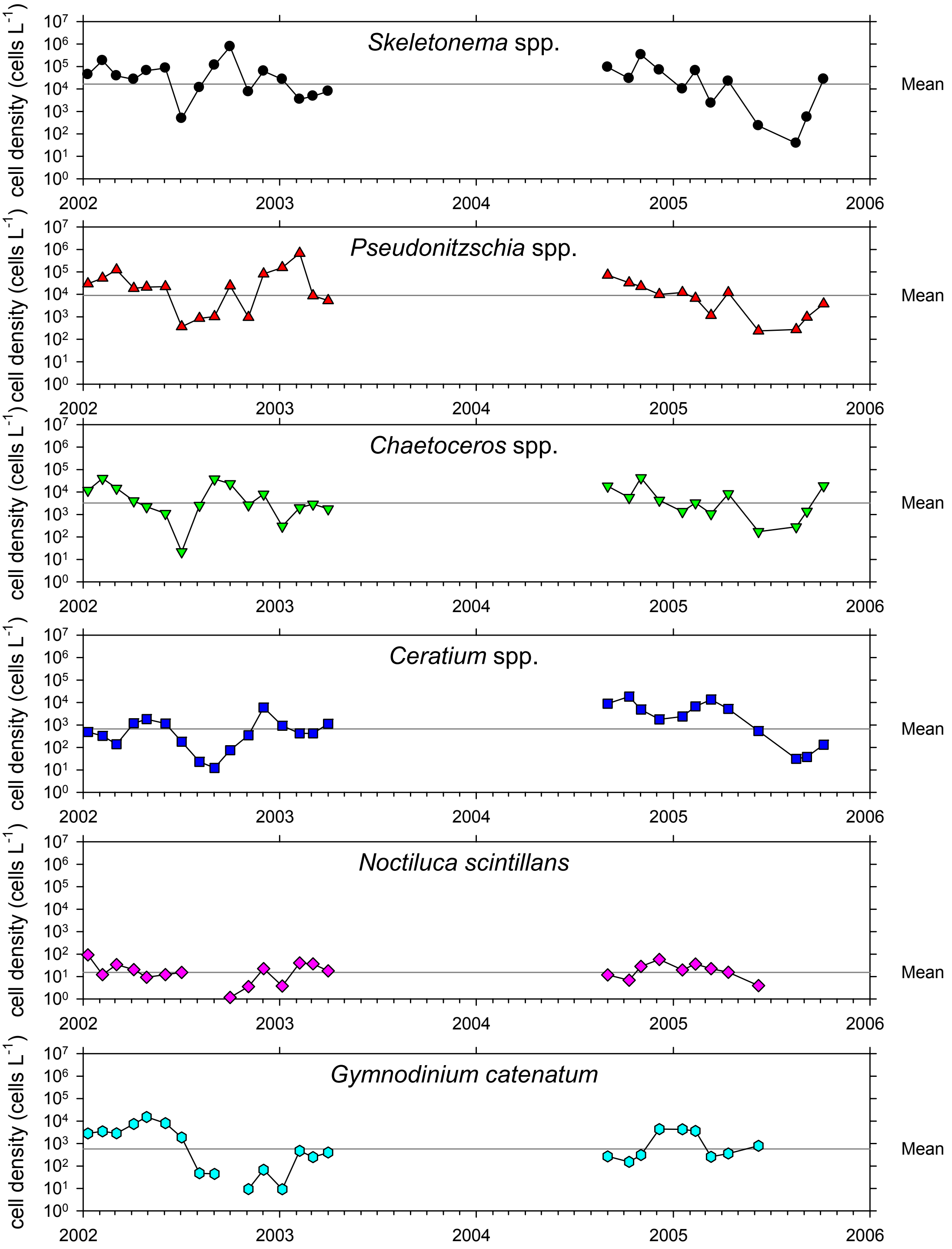


Figure 4.6. The mean cell densities of key phytoplankton taxa averaged across 15 sites in the Huon Estuary and D'Entrecasteaux Channel (see Fig. 4.1 for locations) over time.

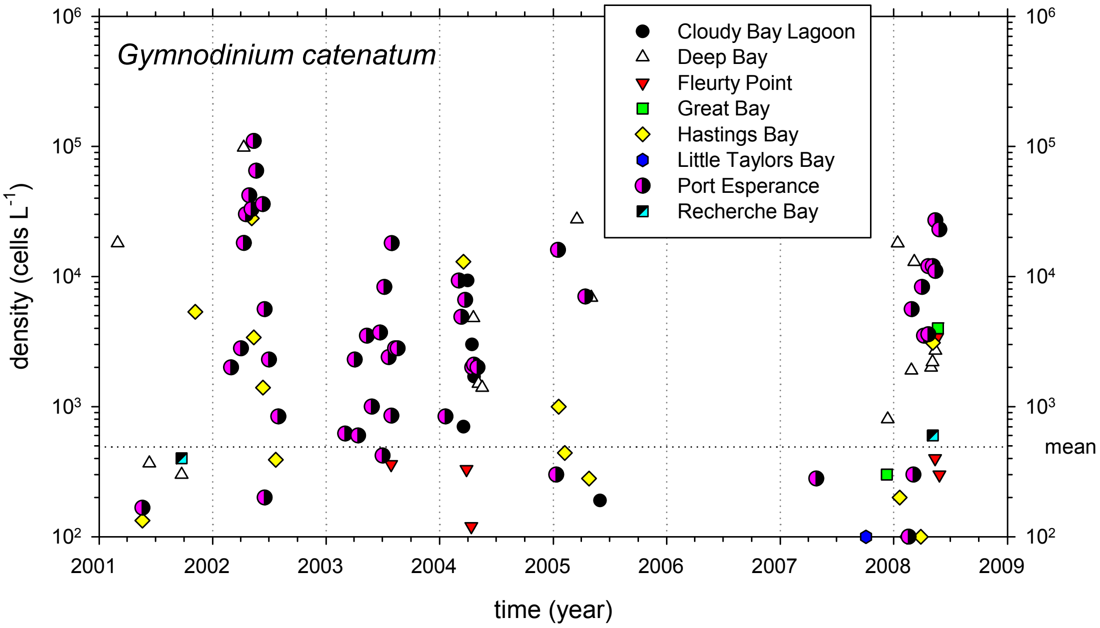


Figure 4.7. Tasmanian Shellfish Quality Assurance Program data on the abundance of *Gymnodinium catenatum* at 7 sites in the Huon Estuary and D'Entrecasteaux Channel from 2001 to 2008.

There was a strong tendency for the seasonal abundance of the two dominant classes (diatoms and dinoflagellates) to vary significantly. Given the differences in cell sizes, a simple comparison by cell numbers can be misleading. To overcome this potential bias the major marker pigments from each taxa (fucoxanthin = diatoms, peridinin = dinoflagellates) were used to investigate relative dominance. The pigment samples were from 0–12 m depth averaged samples at 15 sites in each month from September 2004 to October 2005. The results show that the spring bloom (October) was a mixture of diatoms and dinoflagellates while the autumn bloom (March) was predominately dinoflagellates (Fig. 4.8). Microscopic examinations revealed differences in the dinoflagellates between these seasons, with the spring dinoflagellates being largely *Ceratium* spp., while the autumn dinoflagellates are mostly *G. catenatum*.

The relative biomass of the main taxa found in the Huon Estuary and D'Entrecasteaux Channel was estimated from cell counts converted into biomass in samples at 15 stations from 2001 to 2005. The dinoflagellate *Ceratium* was the dominant genus (Fig. 4.9), followed by *Skeletonema* spp., *G. catenatum*, *Chaetoceros* spp., and *Pseudonitzschia* spp. These taxa showed a very strong spatial separation, with dinoflagellates being much more abundant in the Huon Estuary (sites 13, 14, 15) than in D'Entrecasteaux Channel and with diatoms more abundant at the northern end of D'Entrecasteaux Channel than elsewhere. However the heterotrophic dinoflagellate *Noctiluca scintillans* was most abundant in North West Bay (site 2; see Fig. 4.1 for location).

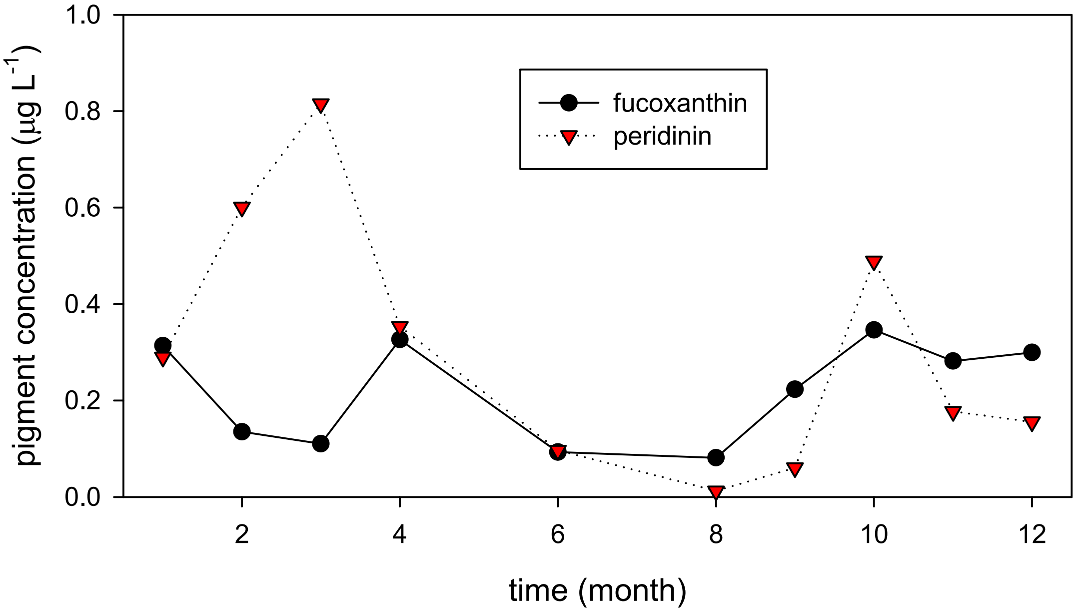


Figure 4.8. The concentrations of peridinin (=dinoflagellates) and fucoxanthin (=diatoms) from 0–12 m depth integrated samples and averaged across 15 sites (see Fig. 4.1 for locations) collected between September 2004 and October 2005 and plotted against month of the year.

There were significant inter-annual differences in abundances of some taxa. Data from two stations sampled in 1997 (n = 78) and 1998 (n = 51), that were sampled again in 2004 and 2005 (n = 21), were compared. The data were pooled into seasons and then analysed for differences by 3 way ANOVA (year, season and station) followed by Bonferroni *t*-tests for difference between years with 1996–97 as the control year. *Ceratium* spp., *Chaetoceros* spp., and small flagellates were all less abundant in 1996–97 than 1997–98. *Ceratium* spp., small flagellates and *Skeletonema* spp. were more abundant in 2004–05 than 1996–97 and *Pseudonitzschia* spp. were less abundant. Mean annual cell density of *G. catenatum* was greatest in 1997–98, but variability between stations was high and no statistically significant temporal pattern was observed (Table 4.1).

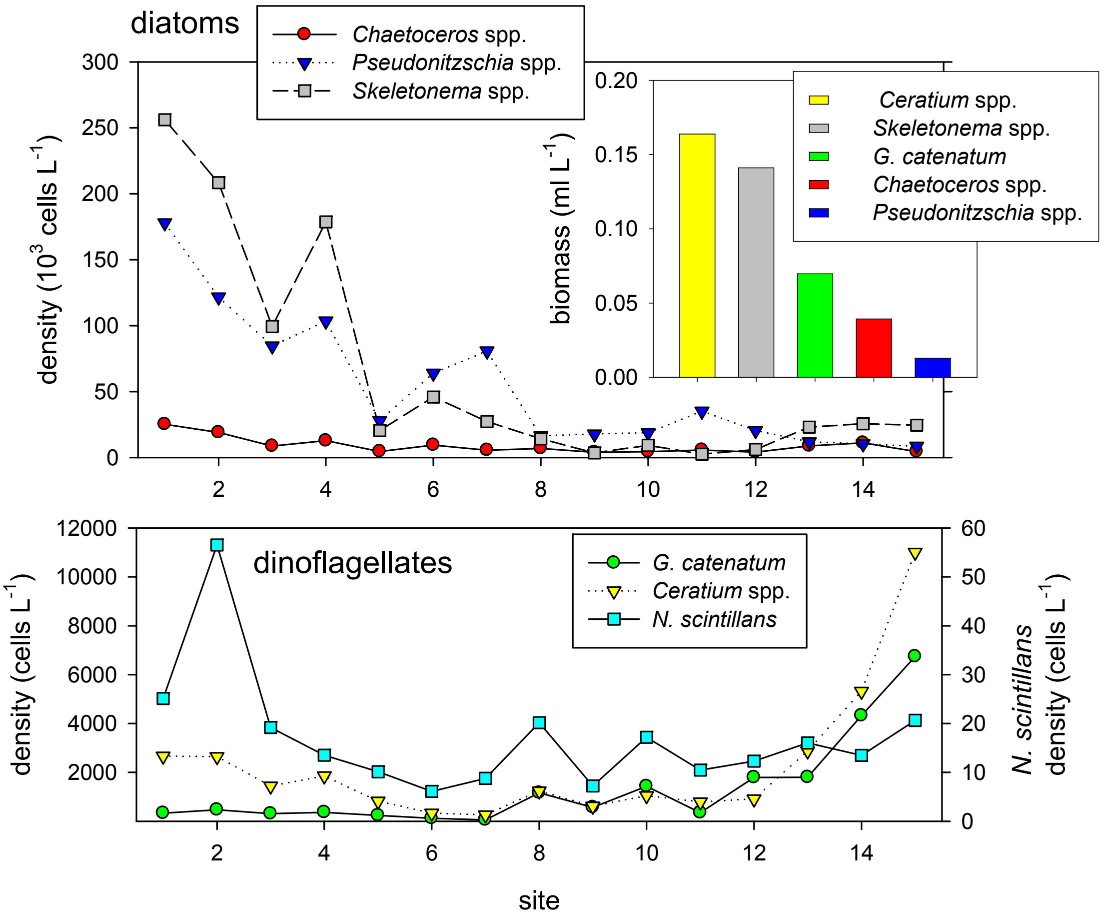


Figure 4.9. The temporally averaged (2001–2005) densities of the major taxa plotted against sites (for site locations see Fig. 4.1). The temporally averaged (2001–2005) density of *Noctiluca scintillans* is also shown (lower panel, cell density is on right hand Y axis). Bar graph shows the biomass (ml cell volume l^{-1} sample volume) for the major taxa averaged over all samples (2001–2005) and all stations (1–15).

Inter-annual comparisons were also made using HPLC pigment data from three stations in the Huon Estuary that were sampled in 1996–97 (n = 160), 1997–98 (n = 159) and again in 2004–05 (n = 41) with sampling by integrating tube and pigment analysis by HPLC. The data were binned by season (winter, spring, summer and autumn) and analysed by 3 way ANOVA followed by Bonferroni *t*-tests to detect significant differences between stations, seasons and years. (Table 4.2). Mean annual concentrations of chlorophyll-*a*, peridinin, 19'-butanoyloxyfucoxanthin, fucoxanthin, 19'-hexanoyloxyfucoxanthin, prasinoxanthin, diadinoxanthin and alloxanthin all increased significantly between 1996–97 and 2004–05 while lutein and zeaxanthin declined. The 20-fold increase in peridinin from 1996–97 to 1997–98 was not statistically significant due to high spatial variability in 1997–98, but the 60 fold increase between 1996–97 and 2004–05 was. The ratio of peridinin:chlorophyll-*a* increased ~12 times (1996–97 to 2004–05), while the fucoxanthin:chlorophyll-*a* ratio only fell 10% suggesting a marked shift in the balance between dinoflagellates and diatoms. The relative rise in peridinin:chlorophyll-*a* was greatest during spring between 1996–97 to 1997–98 and 1997–98 to 2004–05 (data not shown).

Table 4.1. A comparison of species densities at 2 stations in 1996–97 (n = 78), 1997–98 (n = 51) and the same 2 stations again in 2004–05 (n = 21) with sampling by integrating tube (adapted from Thompson et al., 2008). Years were defined as August to July (e.g. 1996–97 = August 1996 to July 1997), data were binned by season (winter, spring, summer and autumn) and data analysed by 3 way ANOVA followed by Bonferroni *t*-tests to detect significant differences between stations, seasons and years. Underlined values are significantly different from 1996–97 ($P < 0.05$), italics ($P < 0.01$), bold ($P < 0.001$).

Taxon	1996–97 mean (10^6 cells m^{-3})	1997–98 mean (10^6 cells m^{-3})	2004–05 mean (10^6 cells m^{-3})
Ceratium spp.	0.61	7.18	**15.3**
Pseudonitzschia spp. [a]	68.2	101.7	*10.1*
Chaetoceros spp.	25.0	65.6	12.2
Gymnodinium catenatum [a]	0.01	40.8	4.7
Small flagellates [a]	424	**674**	**934**
Skeletonema spp. [b]	0.12	0	**21.8**

[a] failed Kolmogorov – Smirnov test for normality and passed the Levene median test for equal variance.

[b] failed Kolmogorov – Smirnov test for normality and the Levene median test for equal variance.

4.3.2. Temporal and spatial variability in phytoplankton dynamics – process studies

The process studies were designed to improve our knowledge of the key biogeochemical fluxes on short time scales in the Huon Estuary and D'Entrecasteaux Channel. The three locations – the mouth of the Huon Estuary, North West Bay and Port Esperance – were selected to give insight into the similarities and differences of processes in different parts of the region, with both North West Bay and Port Esperance being coastal embayments off the channel, at the northern and southern ends, respectively. For the mouth of the Huon Estuary process studies were made in both autumn and spring and for North West Bay and Port Esperance the studies were in spring.

Table 4.2. A comparison of pigment concentrations and pigment ratios at 3 stations in 1996–97 (n = 160), 1997–98 (n = 159) and the same 3 stations (see Fig. 4.1) again in 2004–05 (n = 41) with sampling by integrating tube and pigment analysis by HPLC (adapted from Thompson et al., 2008). Years were defined as August to July (e.g. 1996–97 = August 1996 to July 1997), data were pooled by season (winter, spring, summer and autumn) and data analysed by 3 way ANOVA followed by Bonferroni *t*-tests to detect significant differences between stations, seasons and years. Underlined values are significantly different from 1996–97 (P < 0.05), italics (P < 0.01), bold (P < 0.001).

Pigment (µg l^{-1}) or pigment ratio (µg:µg)	1996–97 mean	1997–98 mean	2004–05 mean	% change 1996 to 2004
chlorophyll-*a*	0.66	1.36	**2.00**	203
peridinin	0.011	0.26	**0.67**	5991
19′-butanoyloxyfucoxanthin [†]	0.00006	0.000	**0.009**	15514
fucoxanthin [a]	0.111	0.123	**0.263**	137
prasinoxanthin [a]	0.0064	0.0039	**0.044**	588
19′-hexanoyloxyfucoxanthin	0.018	0.029	0.053	194
diadinoxanthin [a]	0.029	0.160	0.142	390
alloxanthin	0.032	0.024	*0.053*	66
zeaxanthin [a]	0.022	**0.011**	**0.003**	-86
lutein	0.00066	0.0007	0.000	-100
peridinin:chl-*a* [a]	0.023	**0.148**	**0.290**	1161
fucoxanthin:chl-*a*	0.169	0.108	*0.152*	-10

[a] failed Kolmogorov – Smirnov test for normality and passed the Levene median test for equal variance.

4.3.2.1 The mouth of the Huon Estuary – process studies

Phytoplankton community composition in the mouth of the Huon Estuary in autumn 2005 (12th and 13th April) as well as in spring (20th and 22nd September) was dominated numerically by small flagellates (Fig. 4.10). However phytoplankton biomass estimates (cell volume) showed a very different story between autumn and spring (Fig. 4.11): in autumn the biomass was dominated by diatoms and dinoflagellates, as is the case for the Huon Estuary and D'Entrecasteaux Channel area in general at that time of year (Fig. 4.6 this study; Butler et al., 2000), whereas in spring small flagellates dominated the phytoplankton. The marker pigments also showed the distinct differences in composition between autumn and spring, with a higher proportion of chlorophyll-*b* (chlorophytes) and alloxanthin (cryptophytes - components of the small flagellates) in spring and very low peridinin (dinoflagellates) compared with autumn marker pigment composition (Fig. 4.12). This, along with elevated concentrations of nutrients and low biomass spread throughout the water column, indicates that the mouth of the Huon Estuary was still experiencing 'winter conditions' in 'spring' i.e. late September.

Samples collected on the two consecutive days in April showed high variability in nutrient composition, phytoplankton composition, pigment concentration and composition, and absorption coefficients of the dissolved and particulate fractions (full data are shown in Clementson et al., 2008a). We consider that this is primarily due to changes in weather conditions over the 2 days, whereas there was uniformity in samples collected two days apart in September. Such variability over short time periods highlights the difficulties in tracking short time scale variations in biomass and composition in such dynamic environments (see technical report by Clementson et al. (2008a) for more details).

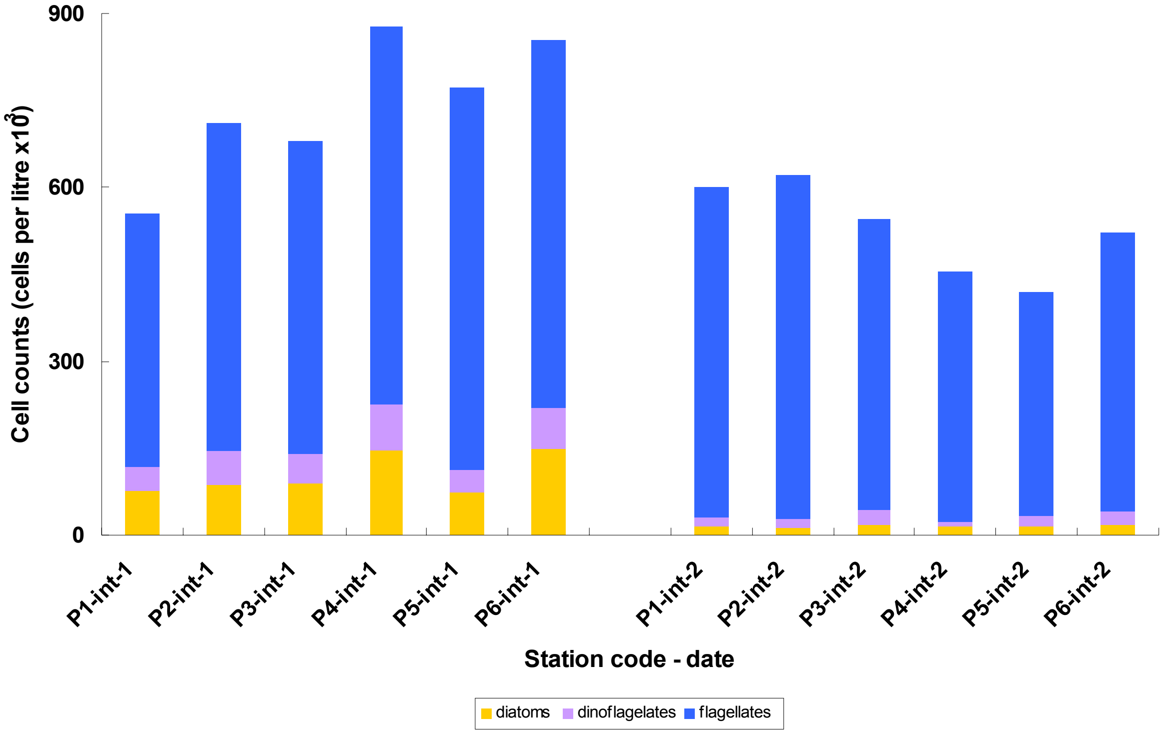


Figure 4.10. Phytoplankton composition shown as cell counts for the 12 m integrated samples averaged for 12th and 13th April 2005 (Int-1) and averaged for 20th and 22nd September 2005 (Int-2) at the mouth of the Huon Estuary.

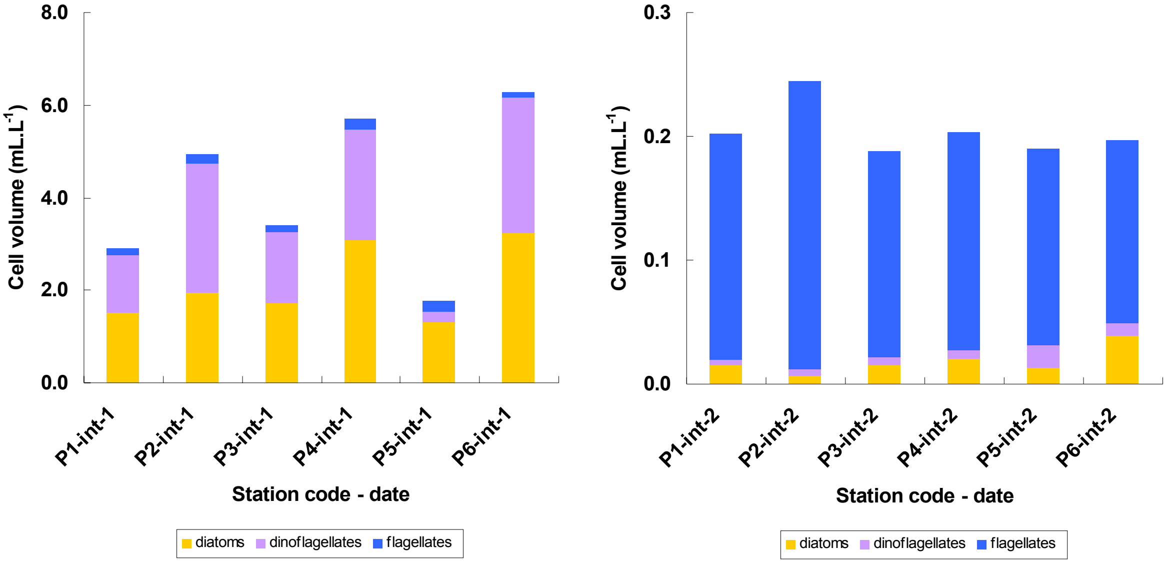


Figure 4.11. Phytoplankton composition shown as cell volume for the 12 m integrated samples averaged for April, 12th and 13th, 2005 (autumn) (Int-1) and averaged for September 20th and 22nd, 2005 (spring) (Int-2) at the mouth of the Huon Estuary.

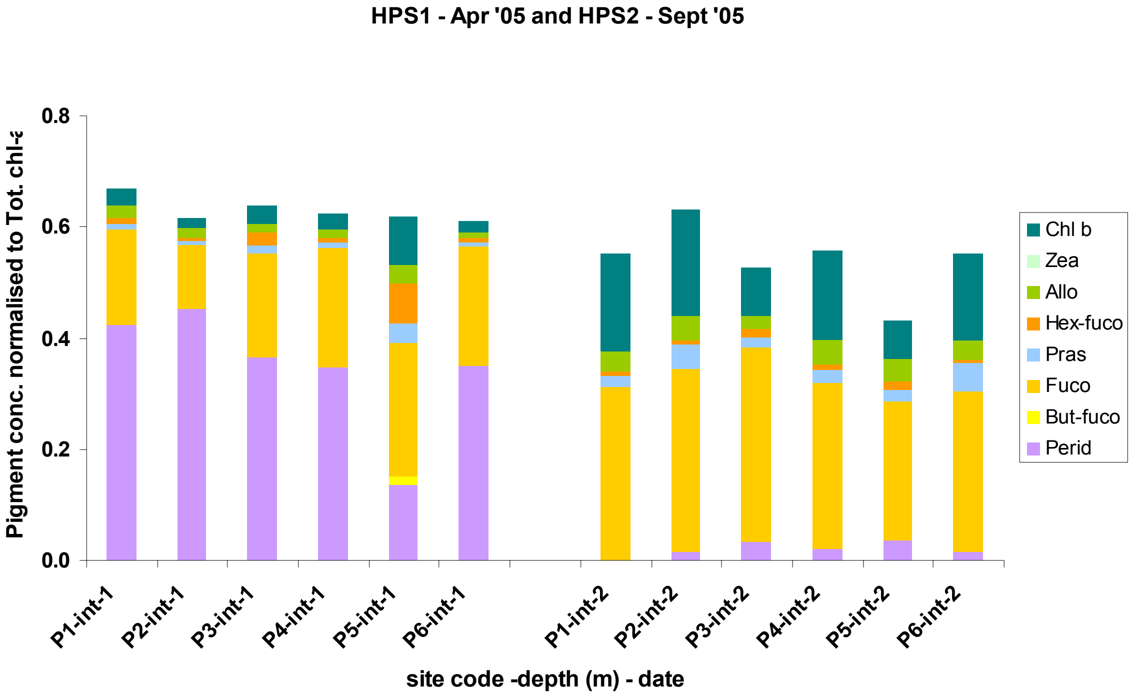


Figure 4.12. The composition of marker pigments for the 12 m integrated samples averaged for April 12th and 13th, 2005 (autumn) (Int-1) and averaged for September 20th and 22nd, 2005 (spring) (Int-2) at the mouth of the Huon Estuary. Pigments are Perid (peridinin); But-fuco (19′-butanoyloxyfucoxanthin); Fuco (fucoxanthin); Pras (prasinoxanthin); Hex-fuco (19’-hexanoyloxyfucoxanthin); Allo (alloxanthin); Zea (zeaxanthin); Lut (lutein); Chl b (chlorophyll-*b*).

4.3.2.2 North West Bay process study

In the North West Bay Process Study, measurements of microzooplankton and mesozooplankton grazing were conducted during a spring bloom in October 2006. In conjunction, the phytoplankton community along with nutrients and physical conditions were measured at 8 sites (Fig. 4.3) on consecutive days: October 3rd and 4th. During the process study North West Bay was characterised by variable concentrations of both phytoplankton biomass and nutrient concentrations considered to be due to the interleaving of marine with rather fresher water layers within the bay during early October 2006. Nutrient concentrations also showed no pattern between sites or with depth, and on average silicate >> phosphate > nitrate > nitrite (see Clementson et al., 2008b for more detail).

Averaged over North West Bay, the spring bloom reached 3.2 µg chlorophyll-*a* l^{-1} or about 4 times the annual mean concentration for the entire D'Entrecasteaux Channel. Phytoplankton cell numbers in North West Bay showed high numbers of small flagellates, but on a biomass basis (cell volume) the phytoplankton was dominated by dinoflagellates and diatoms, with the dominant dinoflagellate being *Gonyaulax digitale*, a relatively large species (35 µm diam.), while the dominant diatoms were the small species such as *Skeletonema* spp. and to a lesser extent *Thalassiosira* spp. and *Chaetoceros* spp. The marker pigments also demonstrated the dominance by dinoflagellates and diatoms. For more detailed information see Clementson et al. (2008b).

Continuous fluorescence data at 5.5 m depth obtained from an instrument package deployed on a mooring just prior to the process study (Fig. 4.13; Clementson et al., 2008b), suggested diurnal migration of phytoplankton. It is possible that the cyclic fluorescence signal could have been caused by a tidal cycle repeatedly advecting a patch of phytoplankton back and forth past the sensor, however this is regarded as unlikely from the physical characteristics of the water at that time (temperature and salinity profiles; M. Herzfeld, personal communication).

During the 23-hour diel study, the marker pigments at the surface and 20 m depth (peridinin for dinoflagellates and fucoxanthin for diatoms; see Fig. 4.14) showed an increase in peridinin at depth, corresponding with an overall increase in chlorophyll-*a*, during the night. This again suggests vertical migration by a motile phytoplankton species. Microscopic analyses showed *Gonyaulax digitale* was the dominant dinoflagellate species present (3.8×10^4 cells l^{-1}). Similar strong vertical migration of the HAB dinoflagellate *G. catenatum* was observed during the Huon Estuary Study (Butler et al., 2000) over a period of 20 days.

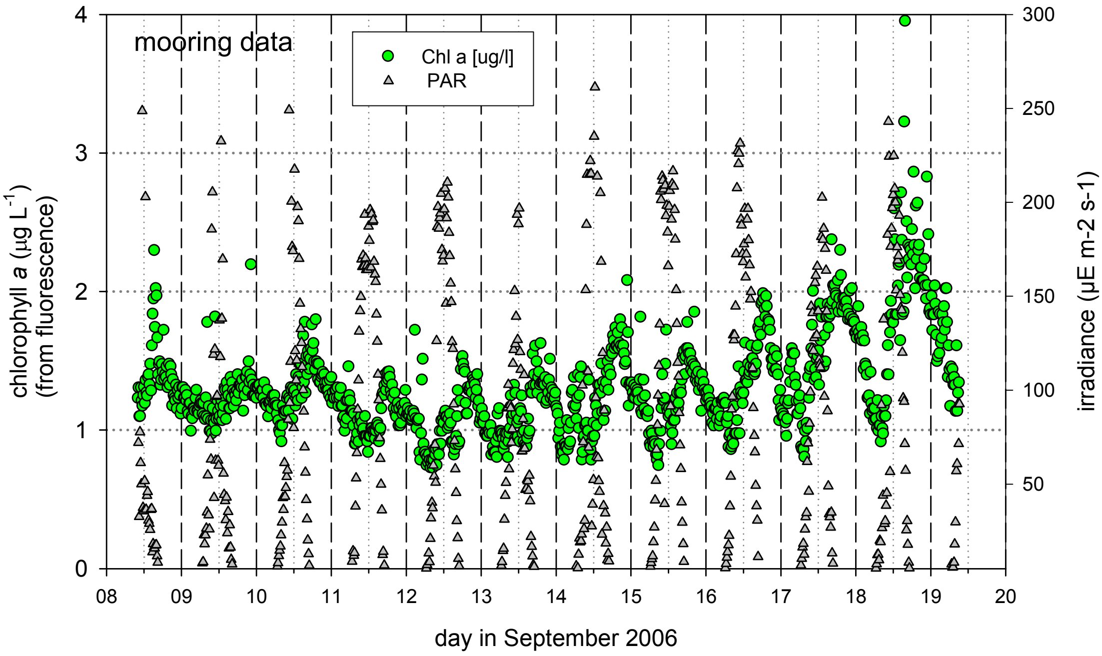


Figure 4.13. Fluorescence and irradiance data from a mooring at ~6 m depth in North West Bay during September 2006.

The concept of phytoplankton using vertical migration to access deep nitrogen reserves (Cullen, 1985; MacIntyre et al., 1997) is at odds with the nitrogen concentrations observed during this study with both nitrate plus nitrite and ammonia concentrations less than 0.1 µM in the surface waters and also in the 20 m layer for most of the process study (Clementson et al., 2008b). Instead, the data suggest the possibility of an internal circadian rhythm regulating the vertical migration. Evidence for endogenous vertical migration in dinoflagellates was reported by Eppley et al. (1968) for *Cachonina niei,* with vigorous migration over a depth of 10 m observed for 5 days after nitrogen depletion. The vertical migration of the HAB dinoflagellate *G. catenatum* observed in the Huon Estuary also took place even though surface nitrate plus nitrite concentrations were high (5.2 µM) and bottom ammonia concentrations were low (Butler et al., 2000).

The absorption coefficient at 440 nm for the detrital fraction (a_d) which is composed of suspended mineral material and heterotrophic microalgae, also showed an increase in the 20 m layer during the night hours (Fig. 4.15), suggesting that the heterotrophic dinoflagellate *Polykrikos schwartzii* was also vertically migrating along with *G. digitale* to deeper waters during the night hours. *P. schwartzii* is a large organism (140 µM × 40–70 µM) which feeds by the engulfment of entire prey and could easily feed on *G. digitale* (~35 µM). Further studies of *P. schwartzii* are needed to verify whether their migration is due to a deliberate response to the migration of the *G. digitale* as a food source and whether the migration is also due to an internal circadian rhythm which results in an opportunistic coincidence.

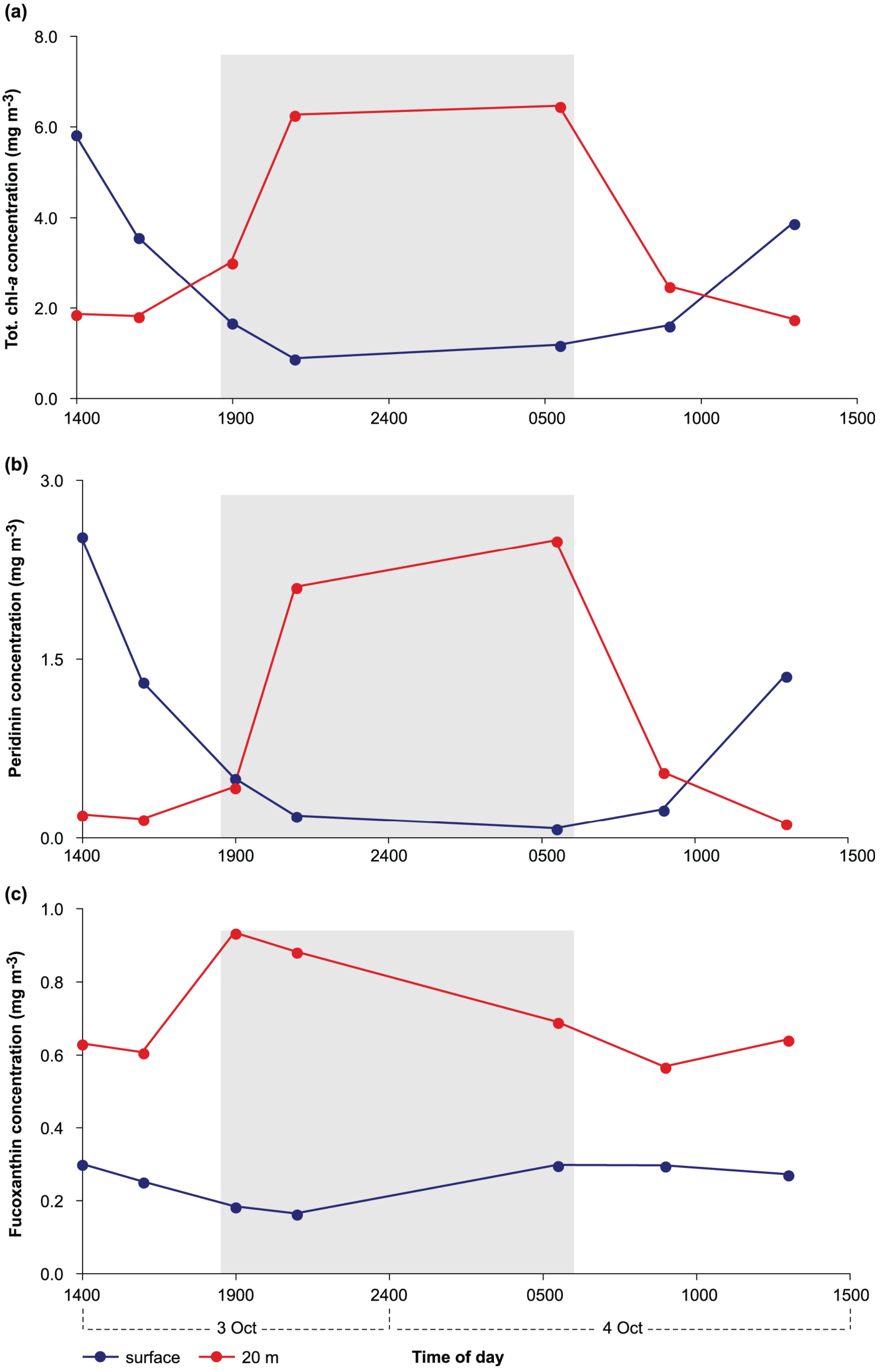


Figure 4.14. North West Bay diel study: the concentration of (a) chlorophyll-*a,* (b) peridinin and (c) fucoxanthin in surface and 20 m samples collected from site 2 on October 3rd to 4th, 2006. The grey area represents the time between sunset (October 3rd) and sunrise (October 4th).

There have been few studies on the role and behaviour of heterotrophic dinoflagellates, but it has been noted that these organisms often concentrate at depths associated with their prey (Hansen, 1991; Lessard, 1991). In a study off the Californian coast, *Polykrikos kofoidii* was observed to avoid the surface waters and form maximum cell concentrations in the subsurface layers (Blasco, 1978), and during microscopic examination of depth-integrated samples collected during the Huon Estuary Study, *Polykrikos* spp. was observed eating the chain forming dinoflagellate *G. catenatum* (N. Parker, personal communication).

4.3.2.3 Port Esperance process study

Port Esperance has similar physical characteristics to North West Bay. It is an embayment off D'Entrecasteaux Channel, although being 25 km further south it has a different hydrodynamic environment (Herzfeld et al., 2005). Sampling was made at 8 sites (Fig. 4.1) during 1st and 2nd November, 2007. In contrast to North West Bay, in Port Esperance all nutrient concentrations (i.e. phosphate, silicate, nitrate and ammonium) tended to be greater at depth. Port Esperance waters were dominated by diatoms, particularly those of the genera *Chaetoceros, Rhizosolenia, Guinardia* and *Leptocylindrus* (Clementson et al., 2008b). Large dinoflagellates such as *Ceratium* spp. were similar in abundance in Port Esperance compared with North West Bay and, the total biovolume of other dinoflagellates such as *Gonyaulax digitale* was much less than in North West Bay (Fig. 4.16). Rather surprisingly, given the differences in taxa, the marker pigments in Port Esperance were similar to North West Bay (Clementson et al., 2008b).

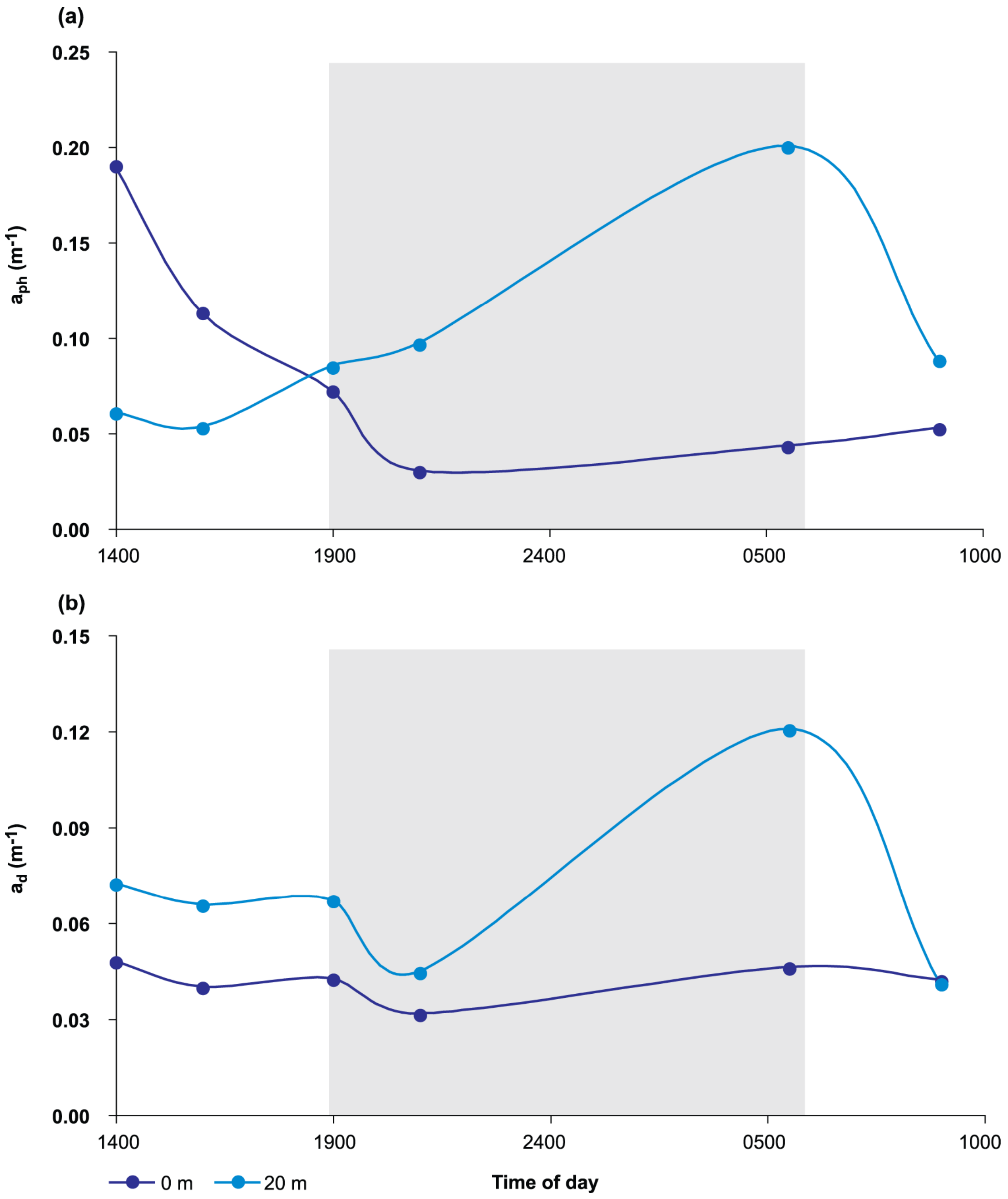


Figure 4.15. North West Bay diel study: Absorption coefficients for (a) phytoplankton and (b) detrital matter for surface and deep (20 m) water samples plotted against time of day. The grey area represents the time between sunset (October 3rd 2005) and sunrise (October 4th 2005).

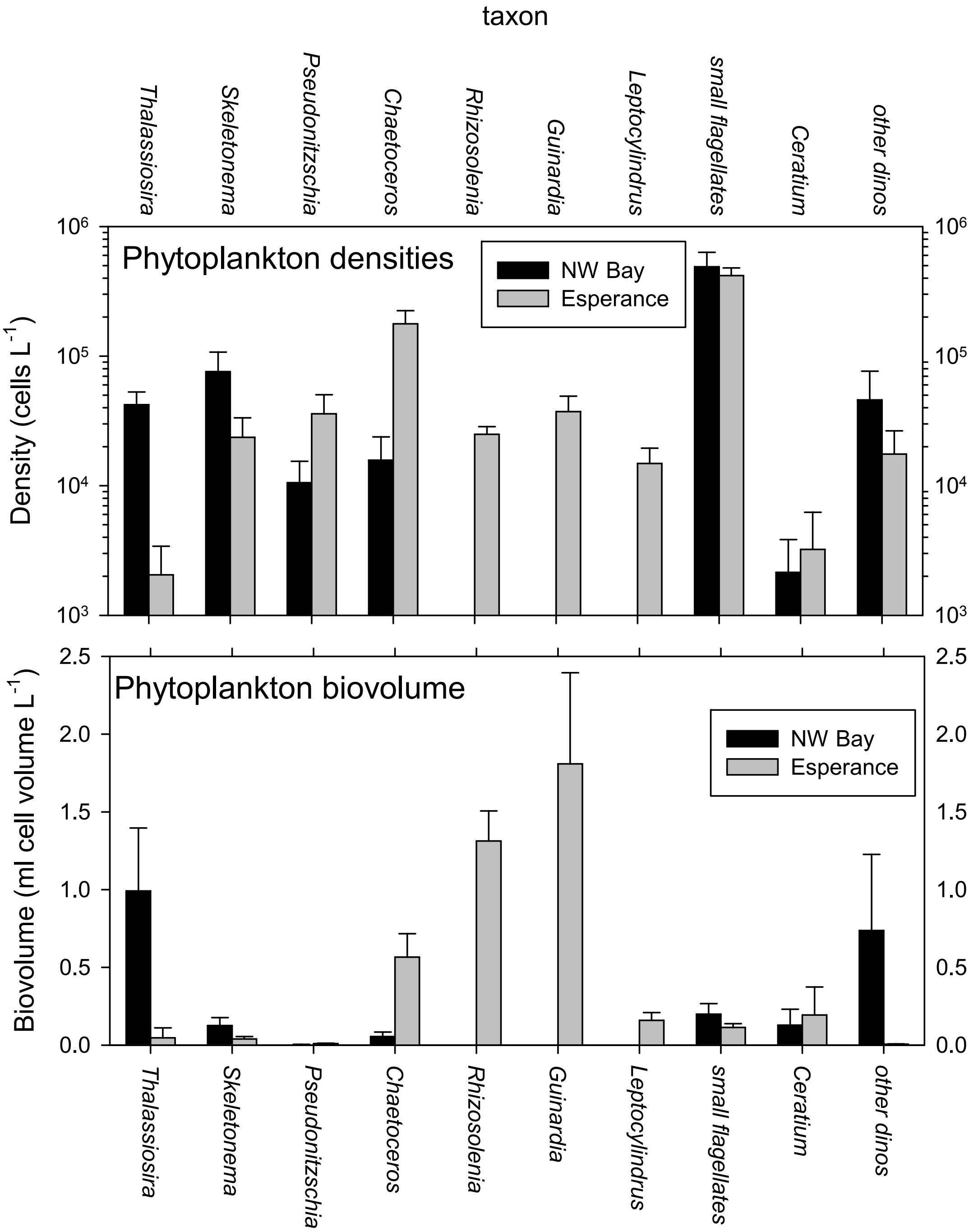


Figure 4.16. Phytoplankton composition by cell counts and biovolumes from depth integrated samples and averaged over all sites in Port Esperance and North West Bay during process studies in spring (November 2007 for Port Esperance; October 2006 for North West Bay).

4.4. Aspects of the autecology of the harmful algal bloom-forming toxic dinoflagellate *Gymnodinium catenatum*

Previous field studies have evaluated the factors influencing *G catenatum* bloom formation and development (Butler et al., 2000; Hallegraeff et al., 1995), while laboratory studies have identified the life history of *G. catenatum* (Blackburn et al., 1989) and the importance and process of vertical migration in the success of this species (Doblin et al., 2006). Several key aspects of the autecology were investigated to gain a deeper understanding of the bloom dynamics of this species: growth on different forms of nitrogen, mortality of *G. catenatum* and algal-bacterial interactions. The outcomes of these studies provide the basis for improved parameterisation of the biogeochemical modelling as part of the CSIRO Environmental Modelling Suite (CSIRO–EMS).

4.4.1. Nitrogen preferences for growth of *Gymnodinium catenatum*

To understand the link between nutrients and HAB blooms in the Huon Estuary it is important to understand the role of key nutrients such as nitrogen (N) and the uptake capacity of HAB species. Preference for, and growth of, *G. catenatum* on nitrate (NO_3^-), ammonium (NH_4^+) and urea were investigated in a series of laboratory experiments. Results demonstrated that *G. catenatum* grows equally well on nitrate (NO_3^-), ammonium (NH_4^+) and urea, but this alga takes them up preferentially in the order: $NH_4^+ > NO_3^- >$ urea (Fig. 4.17). In addition, *G. catenatum* is capable of initial specific NH_4^+ uptake rates ~20 times its specific growth rate i.e. ‘surge uptake’. ‘Surge uptake’ occurs following a period of N starvation, and is identified by an initial increase in uptake sufficient to overcome the deficit in N quota. The ability of *G. catenatum* to respond to pulses of N by ‘surge uptake’ when in N-starved conditions, combined with the physiological flexibility of *G. catenatum* to utilise NH_4^+, NO_3^- or urea, increases this species ability to compete for available N in the environment and would contribute to the ecological success of *G. catenatum* in the Huon Estuary and other ecosystems.

The effect of NH_4^+, NO_3^- and urea on growth rate of phytoplankton species was also investigated by Levasseur et al. (1993) who showed that *Thalassiosira pseudonana*, *Chaetoceros gracilis*, *Dunaliella tertiolecta* and *Gymnodinium sanguineum* were all capable of growth using NH_4^+, NO_3^- and urea as sole N sources. However, growth rates were lower for all these species when grown on urea compared with the other N sources. The ability to grow equally well on urea may be an important advantage for *G. catenatum* over other phytoplankton species that cannot grow using urea as a sole nitrogen source or have substantially reduced growth rates on nitrogen in this form. This nitrogen uptake and utilisation capacity would contribute to development and maintenance of blooms of *G. catenatum.*

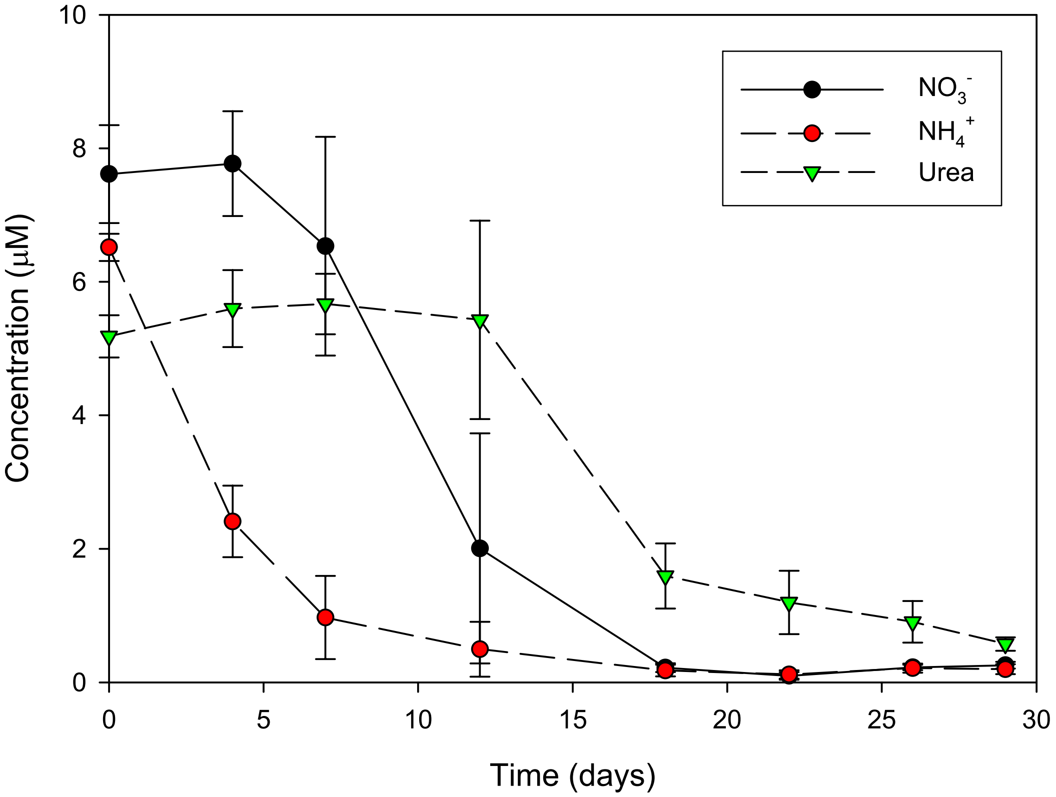


Figure 4.17. Preferential uptake of nitrogen species by *Gymnodinium catenatum*. Mean NO_3^-, NH_4^+ and urea concentrations for *G. catenatum* over the 29 day experiment period (data from cultures of HU02, VS04 and LVO1 strains). Error bars are +1 SEM (n=3).

4.4.2. Mortality rates of *Gymnodinium catenatum* harmful algal blooms

Phytoplankton mortality is rarely addressed specifically in ecological models and is generally handled by a general loss or grazing term. The mortality rates of naturally occurring blooms of *G. catenatum* were calculated from data on blooms between 1986 and 2005. Mortality rates of *G. catenatum* from the naturally occurring blooms differed with whether they were summer or autumn-winter blooms (Fig. 4.18): autumn bloom mortality average was 0.02 day^{-1} and summer bloom mortality average was 0.25 day^{-1}. Mortality rates increased when blooms were disturbed by severe physical events e.g. heavy rain in 1998 was associated with a mortality rate of 0.32 day^{-1}.

Controlled laboratory experimentation demonstrated mortality rates that concurred with field observations (between 0.018 and 0.24 day^{-1} in culture). Mortality rates tested at different temperatures in culture demonstrated lower mortality rates at lower temperatures (J. Skerratt, unpublished data). The lower autumn-winter bloom mortality rates explain the capacity for *G. catenatum* to sustain blooms over winter when water temperatures are suboptimal for growth (Blackburn et al., 1989). The cool winter temperatures decrease the natural mortality rates of this species. Therefore, in spite of the temperatures being less than optimum for growth, the species survives over winter and can sustain a substantial population. Short periods of light, such as those *G. catenatum* would experience during vertical migration through the water column, and high levels of humic substances have been found to be adequate for the vegetative cells of *G. catenatum* to survive (Doblin et al., 1999). The mortality results for *G. catenatum* will be incorporated into the CSIRO-EMS in order to improve the representation of phytoplankton bloom dynamics.

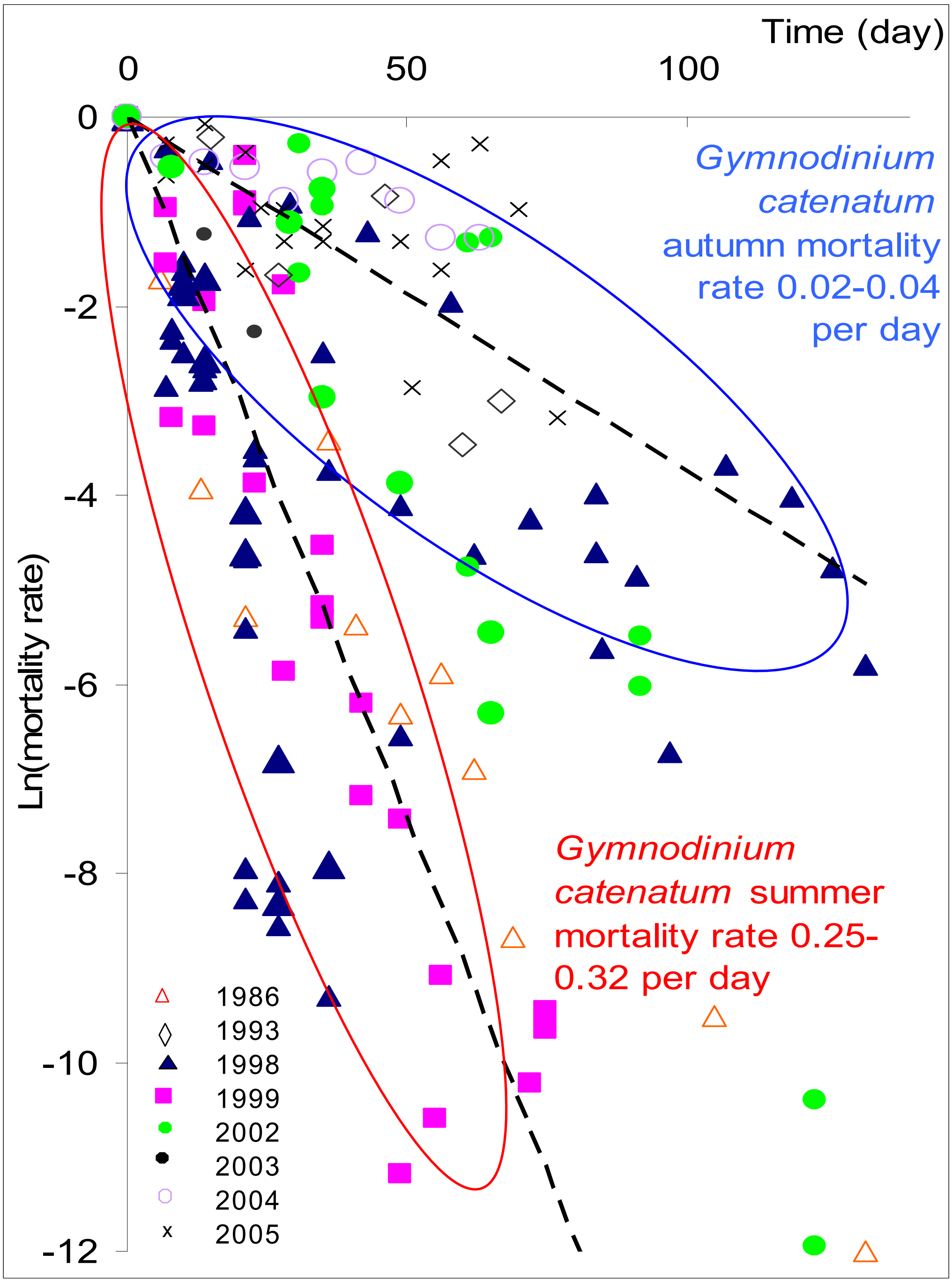


Figure 4.18. Field data for *G. catenatum* blooms and bloom decay rates between 1986 and 2005 at various sites in the Huon Estuary and D'Entrecasteaux Channel. The range of mortality rates for summer and autumn blooms are given.

4.4.3. Interactions of *Gymnodinium catenatum* and bacteria: molecular ecology of harmful algal blooms

Bacteria have been shown to play a role in aspects of algal life, for example positive stimulation of growth, promotion of sexuality, inhibition of cyst formation, antagonism mediated by the production of algicidal factors, and potentially influencing toxin production (Doucette et al., 1998; Uribe et al., 2003). *Gymnodinium catenatum* has been shown to be dependent on (at least) two bacterial strains, *Marinobacter* sp. (strain DG879) and *Alcanivorax* sp. (DG881), for its survival and growth (Bolch et al., 2004). Both strains were initially isolated from a culture of a Tasmanian strain of *G. catenatum*, GCDE08 (Green et al., 2004), are hydrocarbon degrading, and belong to the gamma-(γ)-proteobacteria. Though closely related, not all *Alcanivorax* strains isolated from *G. catenatum* are growth-stimulating. It appears that only a single nucleotide 16S rDNA mutation separates the growth-stimulating DG881 from another non-stimulatory *Alcanivorax* strain (DG813) isolated from a Korean strain of *G. catenatum*. It has therefore been concluded that only certain species within the *Alcanivorax* genera are able to stimulate growth of *G. catenatum* (Bolch et al., 2004). The 'model' algal–bacterial culture system developed by Bolch et al. (2004) was further developed for controlled laboratory experimentation with germination of cysts harvested and sterilized with H_2O_2, after which specific bacterial cultures were added back to the cysts to create unibacterial *G. catenatum* cultures. These studies demonstrated that *G. catenatum* cultures cannot sustain growth without the addition of bacteria (*Marinobacter* and *Alcanivorax* strains) (Fig. 4.19).

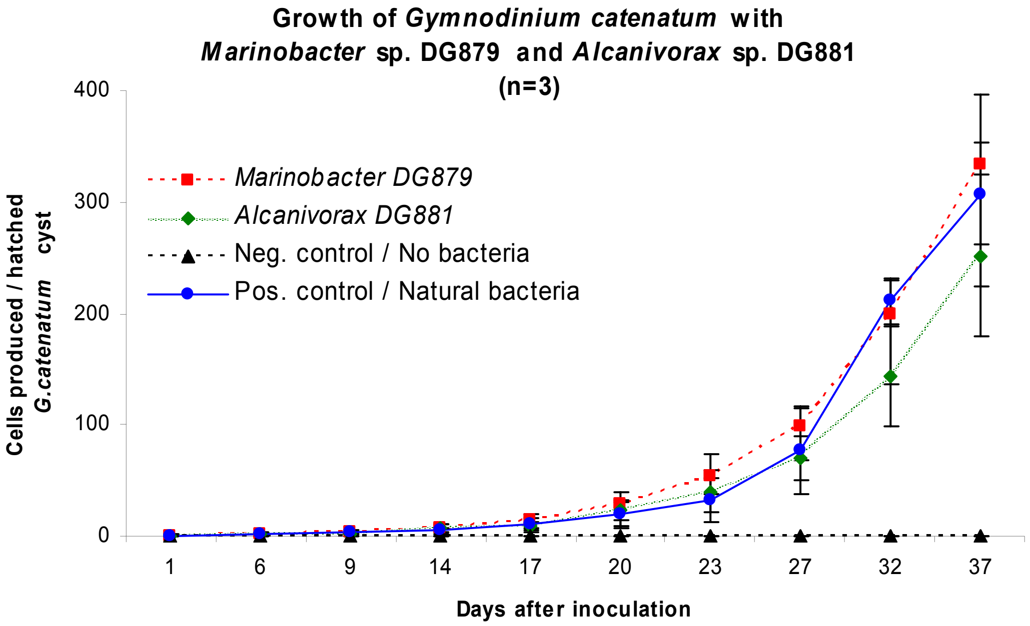


Figure 4.19. Growth of *Gymnodinium catenatum* with bacterial strains *Marinobacter* sp. DG879 and *Alcanivorax* sp. DG881. The positive controls are sterilized cysts to which a 10 µm filtrate of the original *G. catenatum* culture was added, allowing re-entry of the entire naturally occurring bacterial flora. Negative controls are *G. catenatum* cysts that had been sterilized to remove bacteria and then placed in sterile growth medium.

To investigate the effect of bacteria on natural populations of *G. catenatum*, quantitative molecular detection methods (qPCR probes) were developed. The probes allow both detection and quantification of the bacteria, thereby making it possible to determine what proportion of the total bacterial community they occupy at a specific point in time. The quantitative real-time PCR probes are FAM-labelled TaqMan MGB (Monor Groove Binding) probes, using Applied Biosystems-Primer Express 2.0 software. Both sequence detection primers and probes were developed from the 16-23S rDNA Intergenic Spacer Region (ISR) of the two bacterial species, and verified for species specificity using a range of non-target bacterial cultures. Probes and primers were validated using uni-bacterial culture model systems containing *G. catenatum* with either *Marinobacter* (DG 879) or *Alcanivorax* (DG 881) added.

Fig. 4.20 and 4.21 show the close growth interactions between *G. catenatum* and bacteria, using molecular detection of bacteria. The reason for the apparent decrease in bacterial cell numbers in the *Alcanivorax* (DG881) treatment between day 5 and 25 is unknown, but as the density of *G. catenatum* cells appears unaffected, it is possible the decrease is an artifact of DNA extraction. The *Marinobacter* strain and *G. catenatum* show the classic growth curve with a logarithmic growth phase followed by the stationary (plateau) growth phase. The extended stationary phase, particularly for the culture with the *Marinobacter* strain (>100 days), demonstrate the significant advantage provided by the presence of the bacteria on *G. catenatum* growth. Experimental studies by Blackburn et al. (1989), with no added bacteria, typically showed a short stationary phase (up to days 20 to 40) followed by a death phase. The presence of bacteria that can be detected readily by molecular means has the potential for early warning of development of *G. catenatum* blooms. This is currently under investigation by the CSIRO Wealth from Oceans National Research Flagship.

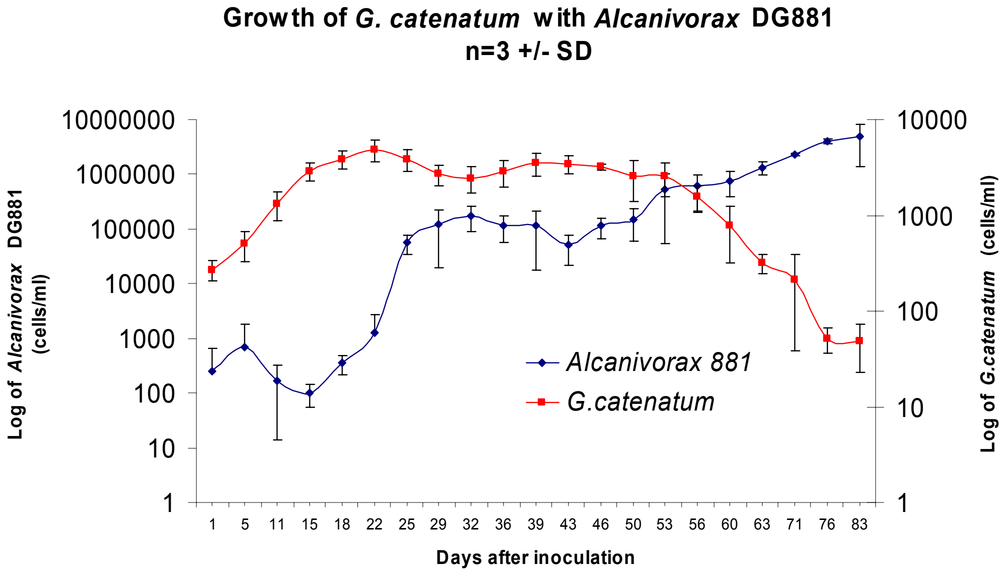


Figure 4.20. Growth of *G. catenatum* with *Alcanivorax* strain DG881 (measured as amount of bacterial DNA per µl converted to cells ml^{-1}).

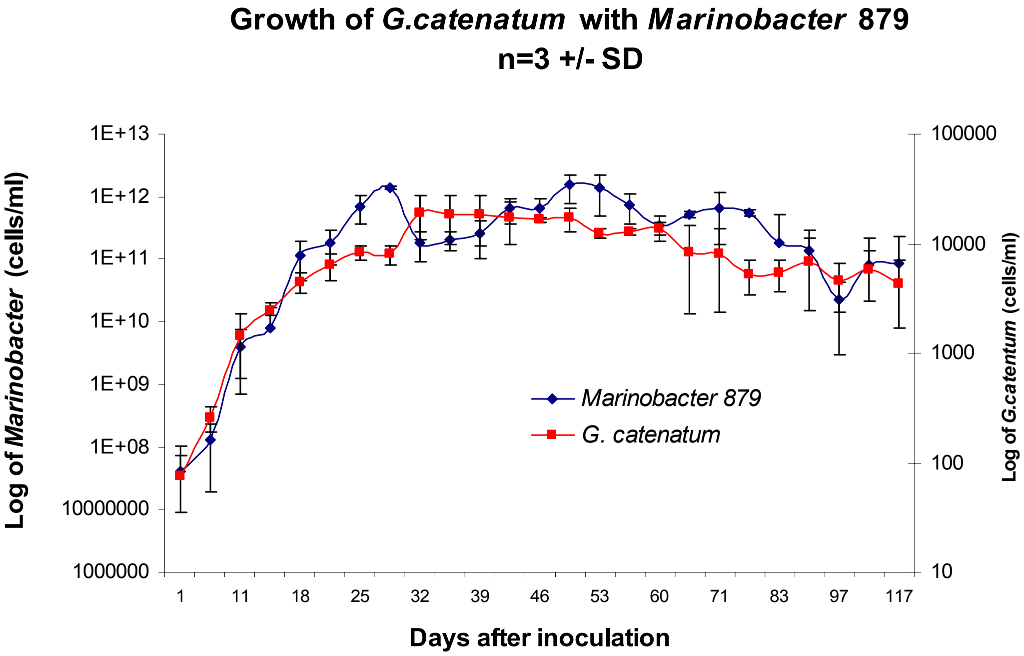


Figure 4.21. Growth of *G. catenatum* and *Marinobacter* strain 879 (measured as amount of bacterial DNA per µl converted to cells ml^{-1}).

4.5. Autecology of other harmful algal bloom species in south east Tasmania: *Noctiluca scintillans*

Noctiluca scintillans was first recorded in Australian waters in the 1970s from the New South Wales coast; the first evidence of range extension to Tasmanian waters was a bloom recorded in 1994. The East Australian Current is considered the most likely transport mechanism for *N. scintillans* cells to Tasmanian waters (Rintoul and Bullister, 1999; Hallegraeff, 2002; Dela-Cruz et al., 2003), and since 1994 there has been an apparent increase in frequency and distribution of blooms with an over-wintering population evident from 2004 (Grundle, 2004).

N. scintillans is a voracious heterotrophic dinoflagellate species that uses a mucus-producing tentacle to capture phytoplankton and zooplankton prey from surrounding waters. It can form extensive blooms which can result in the deaths of fish and other marine life due to depletion of oxygen in the water by dense declining blooms and/or the release of high concentrations of ammonia from large vacuoles that are a feature of *N. scintillans* cells (Okaichi and Nishio, 1976). Laboratory experiments have shown that ammonia concentrations of 225 µg l^{-1} are toxic to Atlantic salmon (Knoph and Thorud, 1996), but there are no published ammonia concentrations associated with fish kills. *N. scintillans* densities of about 10,000 cells per litre are defined as red tides and, based on an average cell ammonia concentration of 0.038 µg per cell, would produce ichthyotoxic ammonia concentrations in the surrounding seawater (Holmes, 2005). Lipid analysis of red tide populations exhibited increased free fatty acid fractions containing polyunsaturated fatty acids which have a history of known ichthyotoxicity. The occurrence of *N. scintillans* in the Huon Estuary and D'Entrecasteaux Channel region therefore has potentially serious implications for the Atlantic salmon aquaculture industry.

The ecological impact of this species on the Tasmanian marine ecosystem is uncertain. To improve our understanding of *N. scintillans* bloom dynamics and the impact of this new HAB species in south-east Tasmanian waters including the Huon Estuary and D'Entrecasteaux Channel, the distribution of *N. scintillans* blooms in Tasmanian waters from 2001 to 2005 was investigated (Albinsson, 2004; Thompson et al., 2005), together with the feeding preferences on ecologically relevant phytoplankton species, including the HAB species *G. catenatum* (Holmes, 2005).

4.5.1. *Noctiluca* bloom distribution

During a study of the Huon Estuary and D'Entrecasteaux Channel where samples were collected monthly from 15 sites between January 2002 and March 2003 (Thompson et al., 2005), *N. scintillans* was found to have the greatest abundance at the two sites at the northern end of the channel during the summer months (Fig 4.22a). In a following study of 11 of these sites between September 2004 and October 2005, the abundance of *N. scintillans* rarely exceeded 80 cells l^{-1} (Fig 4.22b). However, it is worth noting that in spring the *N. scintillans* blooms were only recorded in the northern region of the channel (Fig. 4.23), and with the progression to summer there was a trend of increasing number of blooms overall, with a population peak in December 2004, as well as an increase in the number of blooms in the southern part of Huon Estuary and D'Entrecasteaux Channel region (Fig. 4.23).

4.5.2. Culture studies and feeding preferences of *Noctiluca scintillans*

As part of the project we achieved the first ever successful long-term culture of *N. scintillans* (Holmes, 2005), enabling controlled experimentation of the interaction of *N. scintillans* in culture with individual 'prey' microalgal species that are known to be part of the phytoplankton community in south east Tasmanian waters. These experiments revealed that the HAB-forming dinoflagellate *Gymnodinium catenatum* was a suitable food source for *N. scintillans,* along with the diatom *Thalassiosira pseudonana*. Prey concentrations of 300 cells ml^{-1} proved to be suitable for maintaining a balanced predator-prey interaction between *N. scintillans* grazing rates and *G. catenatum* growth rates. Suitability of prey species was highest for motile species with a high prey encounter rate. Initial *N. scintillans* growth rate was highest with the smaller prey species *Tetraselmis suecica* and *Thalassiosira pseudonana,* but prey growth rates for these smaller species were too high and prey concentration could not be maintained at a level suitable for ongoing trials of these species (Holmes, 2005). *N. scintillans* growth rates when fed *Gymnodinium catenatum* were almost equivalent to maximum growth with smaller prey after an initial adaptation period had occurred.

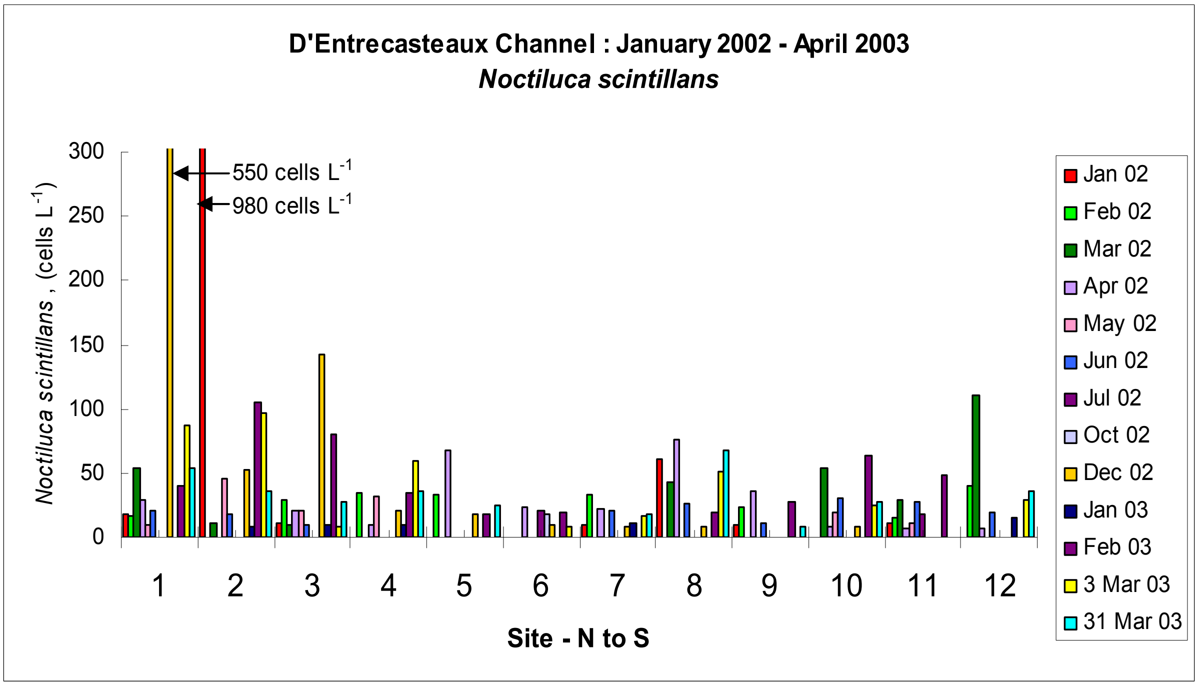
Elucidation of the growth response to environmental variables showed that *N. scintillans* growth increased with decreasing salinity and increasing temperature. Specific growth rate increased from 0.24 at 12 °C to 0.47 divisions per day at 23 °C and decreased from 0.6 divisions per day at a salinity of 20 to 0.33 divisions per day at a salinity of 30. Culture conditions of 17 °C for temperature and 28 for salinity were selected for optimal maintenance of co-existing predator-prey cultures. Bloom formations that occur during spring when *N. scintillans* growth rate is low can be explained by the abundance of phytoplankton prey at this time, i.e. a non-limiting food source would support consistent, albeit slow, growth. Active cell division at 10 °C and survival at 7 °C supports the field evidence that *N. scintillans* populations can survive low Tasmanian winter temperatures.

The increased growth rate of *N. scintillans* at low salinity in laboratory experiments suggests that growth would be possible in fresher surface waters which are frequently observed in the Huon and D'Entrecasteaux region. With *N. scintillans* now a permanent member of the phytoplankton community in south-east Tasmania, its ability to adapt to changes in environmental conditions increases the potential for bloom formation on both temporal and spatial scales. Its capacity to feed on the HAB-forming *G. catenatum* has a potentially positive effect by minimising blooms of this toxic species.

4.6. Conclusions

There were strong spatial and temporal patterns in the distribution and abundance of phytoplankton taxa within the Huon River Estuary and D'Entrecasteaux Channel. On seasonal time scales, mixed blooms of dinoflagellates and diatoms occurred in spring (October) and blooms of predominantly dinoflagellates occurred in autumn (March–May). Over a longer time period (~1996–2005) there has been a significant increase in *Ceratium* spp. and *Skeletonema* spp. and small flagellates. The toxic HAB species, *Gymnodinium catenatum,* was in low abundance in 1997 and 2006.

The pigment concentrations indicate a marked rise in biomass (measured as chlorophyll-*a*) from 1996 to 2005 and changes in the phytoplankton community composition with peridinin (dinoflagellates), fucoxanthin (diatoms), 19′-butanoyloxyfucoxanthin (small flagellates) all increasing. The two dominant taxonomic phyla, diatoms and dinoflagellates, showed a strong tendency to dominant in different regions. Diatoms, especially *Skeletonema* spp. and *Pseudonitzschia* spp., tended to dominate in the northern parts of D'Entrecasteaux Channel. Dinoflagellates such as *Ceratium* spp. and *G. catenatum* were more abundant towards the top end of the Huon Estuary. The spatial differences in diatom versus dinoflagellate abundance is hypothesized to be related to the increased stratification and circulation of a salt wedge estuary (Huon Estuary) both potentially favouring dinoflagellates over diatoms (Tyler and Seliger, 1978).



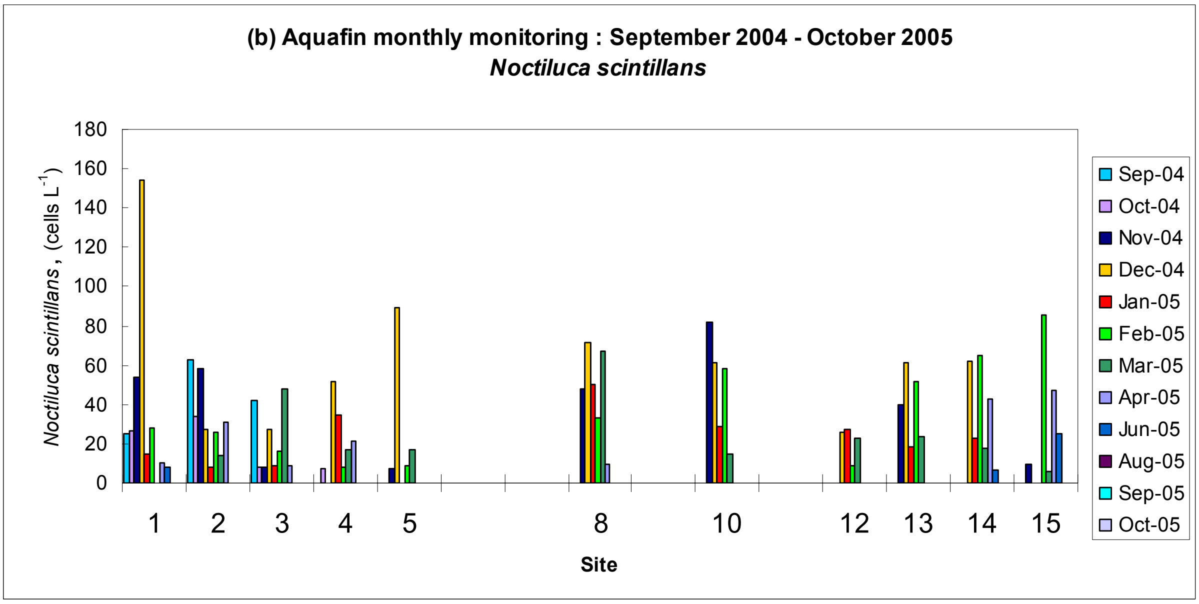


Figure 4.22. Abundance of *Noctiluca scintillans* at sites in the Huon and D'Entrecasteaux region for (a) 2002–2003 and (b) 2004–2005. Sites 6, 7, 9 and 11 were not sampled during 2004–05.

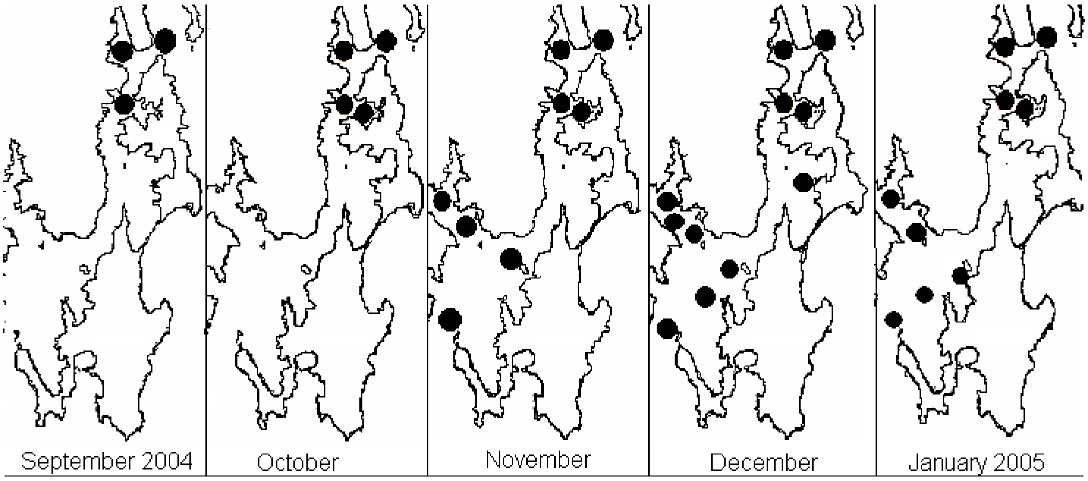


Figure 4.23. Recorded observations of *N. scintillans* in the Huon Estuary and D'Entrecasteaux Channel from September 2004 to January 2005.

The long term increase in phytoplankton biomass in the D'Entrecasteaux Channel and Huon Estuary is consistent with increased nutrient loading. From the limited number of comparable sites the increase between 1996–2005 was substantial (~200%). The biogeochemical (BGC) modelling as part of the CSIRO-EMS (Chapter 7) overcomes some of the limitations of the spatial sampling and deals with eutrophication in a more holistic manner. The long term change in species composition is both intriguing and challenging to explain. It has been shown that dinoflagellates are increasingly dominant over large parts of the world's oceans (Edwards and Richardson, 2004) and the advection of these populations into coastal embayments where they flourish is a well known phenomenon among some dinoflagellate taxa (Smayda and Reynolds, 2001). The strengthening East Australia Current (Rintoul and Bullister, 1999) is the presumed mechanism for the recent arrival of *Noctiluca scintillans* (Thompson et al., 2008) from further north (Dela-Cruz et al., 2003). This change in the regional oceanography may be bringing to Tasmania new taxa or a different mixture of taxa including more *Ceratium* species from the more northern waters (Wood, 1954, 1964).

Short time scale process studies at selected times of the year in the mouth of the Huon River Estuary (autumn and spring), North West Bay (spring) and Port Esperance (spring) have illustrated the change in phytoplankton dynamics due to changes in physical and chemical parameters over time scales of hours. Over a 24-hour period during April 2005 at the mouth of the Huon River Estuary, weather conditions determined the position of nutrient-replete surface waters, therefore changing the dominant algal group within the upper water column. In a system such as the Huon River Estuary and D'Entrecasteaux Channel where flows of fresh surface water converge with marine waters and motile species of phytoplankton thrive, the dynamics of the phytoplankton community can change rapidly. Interpretation of the results from discrete samples provides only a limited insight into what the ecosystem was like at the time of collection. Samples collected on a regular basis such as weekly in the previous Huon Estuary Study or monthly as in this project can really only be used to determine longer term trends or changes in such an ecosystem. To more fully understand dynamic systems such as the Huon River

Estuary and D'Entrecasteaux Channel, automated profiling instruments which operate at small time scales would be beneficial.

Gymnodinium catenatum remains a significant HAB species in the Huon River Estuary and D'Entrecasteaux Channel. Aspects of the dynamics of *G. catenatum* that will facilitate fine tuning of the parameterisation of the biogeochemical modelling of this species include demonstration of the effective utilisation of all nitrogen sources for growth with a preference for ammonia, the necessity of particular surface bacteria, *Alcanivorax* and *Marinobacter* spp., for growth and bloom development, as well mortality rates of blooms, with slower decline of autumn blooms compared with summer blooms. These differences in natural mortality seasonally would contribute to the maintenance of autumn-winter blooms. Molecular probes developed for detecting algal-bacterial interactions have the capacity to be developed as new monitoring tools for bloom detection. There is good evidence that the recently arrived HAB species *N. scintillans*, having extended its range from eastern Australia, now has a significant influence on the phytoplankton dynamics in the region, both as a bloom-forming species in its own right and also due to its capacity to effectively feed on *G. catenatum,* with the potential to influence the intensity and extent of phytoplankton blooms.

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Chapter 5

Zooplankton Ecology and Grazing in the Huon Estuary and D'Entrecasteaux Channel

5.1. Introduction

An important issue for the management of the Huon Estuary and greater D'Entrecasteaux Channel ecosystem is improved understanding of the factors that influence nutrient cycling. Increases in algal blooms and/or greater algal biomass are typical responses to nutrient loading and have been predicted for the Huon Estuary if inputs of nitrogen were to rise (Butler et al., 2000). The Huon Estuary Study documented seasonal cycles of algal biomass in the Huon and quantified the various processes of "loss", including consumption, sedimentation to the bottom and transportation out of the ecosystem. The relative importance of these processes is fundamental to our capacity to predict the response of this ecosystem to nutrient loading. For example, if most of the nitrogen input to the ecosystem is quickly buried deep into sediments or advected out to sea then local nitrogen inputs may have little impact on Huon Estuary ecology. Conversely, if the inputs of nitrogen are largely captured by phytoplankton, then grazed and recycled within the water column, the local ecosystem is much more susceptible to the deleterious effects of increased nutrient loading.

In this chapter we present an overview of the ecology of key grazers on phytoplankton in the Huon Estuary and D'Entrecasteaux Channel, namely the microzooplankton and mesozooplankton. More detailed information can be found in the accompanying technical reports by Bonham et al. (2008) and Swadling et al. (2008) which are included on the CD-ROM that accompanies the final report.

Studies of zooplankton were not included in our original studies, but began with the initiation of regular sampling for mesozooplankton in October 2004 and grazing studies in September 2005, as well as the continuation of targeted grazing studies for microzooplankton, which had begun in September 2003. In this chapter we define microzooplankton, which include ciliates, rotifers, heterotrophic dinoflagellates and some crustacean larvae, as falling within the size range of 20 to 200 μm, while mesozooplankton are organisms within the range of 200 μm – 2 mm (Sieburth, 1978). Mesozooplankton typically include small crustaceans and larval stages of benthic animals such as molluscs and polychaetes, along with specialist groups including arrow worms and jellyfish. The term zooplankton will be used to refer to both microzooplankton and mesozooplankton together. We will outline the dynamics of the mesozooplankton community over an annual cycle and highlight specific grazing studies designed to determine the relative grazing impacts of microzooplankton and mesozooplankton on the phytoplankton.

Zooplankton play key roles in pelagic food webs by controlling phytoplankton production. Furthermore, because of their critical role as prey for larval and juvenile fish, the dynamics of zooplankton populations have a substantial influence on recruitment to fish stocks. Also, zooplankton play important roles in carbon flux in the marine environment, both in its recycling in upper waters and sequestration into sediments. For example, smaller faecal pellets, such as those produced by microzooplankton, are often retained in surface waters and remineralised by bacteria and other grazers. Larger pellets, along with heavy cells such as diatom frustules, sink more rapidly and so contribute carbon to the sediments. It is generally accepted that communities dominated by microzooplankton grazing, in association with microbial food webs, will contribute to the regeneration of production using ammonia and organic nitrogen (i.e. recycled nutrients from within the system). In contrast mesozooplankton are typically associated with the classical food chain (e.g. diatoms – copepods – fish) and export (= new) production based on nitrate input from outside the system (e.g. upwelling, river runoff) (Buitenhuis et al., 2006). Generally, the contribution of mesozooplankton to vertical particle flux is much higher than that of microzooplankton, though the two are often linked together in most biogeochemical models (Buitenhaus et al., 2006).

An increase in nutrients into the Huon and D'Entrecasteaux ecosystem could result in different responses by the phytoplankton and, subsequently, the zooplankton and the remainder of the ecosystem. Firstly, zooplankton could ameliorate algal blooms by direct consumption, which could create nutrient supply conditions with altered N: P ratios, potentially favouring other algal species with different nutrient preferences. Secondly, zooplankton consumption could increase the amount of material contributing to the sediment flux and hence to the sequestration of carbon into the sediments. Thirdly, a rapid increase in phytoplankton production could result in short or long-term changes in the community dynamics of zooplankton. Zooplankton typically respond to increases in phytoplankton production by a corresponding increase in small, rapidly reproducing taxa, such as protists, cladocerans and copepods. This increase might occur in just 1 to 2 weeks. If the nutrient enrichment is episodic, these small-bodied colonisers gradually give way to larger-bodied taxa and the community will revert to its earlier composition. However, if the enrichment persists the community structure may switch, with different organisms becoming dominant. Thus, the responses of zooplankton to increases in nutrient supply can be complex and require sound understanding of both their community dynamics and their tropic interactions.

5.2. Zooplankton community dynamics

Mesozooplankton community composition and structure were examined throughout the Huon and D'Entrecasteaux ecosystem from November 2004 to October 2005. The composition of the mesozooplankton community over an annual cycle was typical of inshore, temperate marine habitats (e.g. Swadling and Bayly, 1997; Vieira et al., 2003), with seasonally higher abundance in summer and early autumn and lower numbers in winter and spring (Fig. 5.1). Copepods were the largest contributors (> 80%) to total abundance across all seasons and stations, while cladocerans and appendicularians were proportionally more abundant in spring and summer. Meroplanktonic larvae of benthic animals showed short-term peaks in abundance and were often absent from the water column for extended periods. North West Bay and the Channel had a higher representation of typically marine species, including the copepods *Calanus australis* and *Labidocera cervi*, while truly estuarine species, such as the copepod *Gladioferens pectinatus*, were more important in the Huon Estuary.

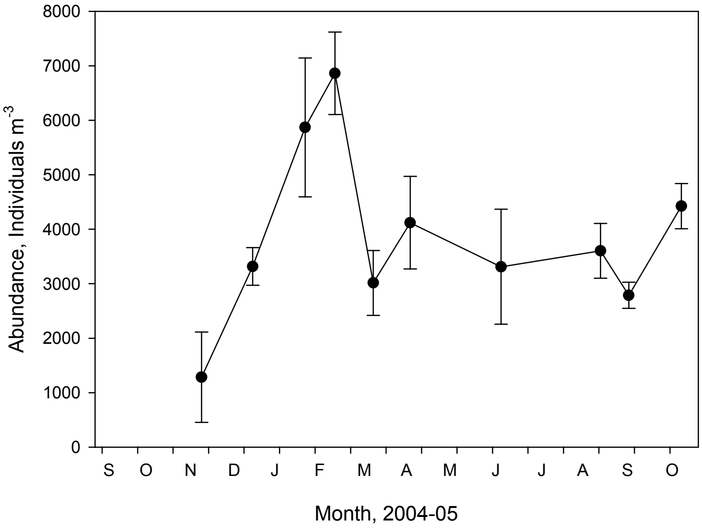

Figure 5.1. Mesozooplankton abundance over an annual cycle, mean density ± 1 se (n = 3) from November 2004 to October 2005 in the Huon Estuary. Sites in D'Entrecasteaux Channel and North West Bay showed similar seasonal cycles.

Spatially, there was considerable horizontal variation in the distribution of mesozooplankton (Fig. 5.2). In the Huon Estuary in autumn, the highest abundances were found at the sites furthest upriver. Overall, the dominant species were the copepods *Paracalanus indicus* (30%), *Oithona* spp. (13%), *Acartia tranteri* (8%), and *Centropages australiensis* (2%), the heterotrophic dinoflagellate *Noctiluca scintillans* (17%), cladocerans (7%) and appendicularians (larval tunicates) (6%). In the Huon Estuary, there were differences in the species that contributed to variation at each site. Sites 1, 2 and 6 were all dominated by *Paracalanus indicus*, the ghost shrimp *Lucifer hanseni* and *Centropages australiensis*. Site 5 revealed a large number of benthic larvae and was generally lacking in copepods and *Noctiluca scintillans*. Sites 3 and 4 were intermediate between these two groups.

In North West Bay in early October there was again patchiness in the distribution of mesozooplankton, with the western shore generally showing higher numbers than the eastern shore (Fig. 5.2). Throughout the bay the dominant organisms were *Paracalanus indicus* and eggs and larvae of *Centropages australiensis*, which together contributed an average of 78% to total abundance. Other contributors were *Acartia tranteri* (5%), *Oithona* spp. (4%) and small leptomedusae (3%). Zooplankton often time their reproduction to take advantage of bloom conditions. For example, they may release their fertilised eggs so that hatching larvae have a rich food supply or adults may feed voraciously while they can to convert the energy of the food into eggs. During this study of North West Bay it appeared that *Centropages australiensis* was exploiting this early increase in food supply, indicated by the presence of large numbers of their eggs and larval stages, while other species had not yet begun to increase their populations.

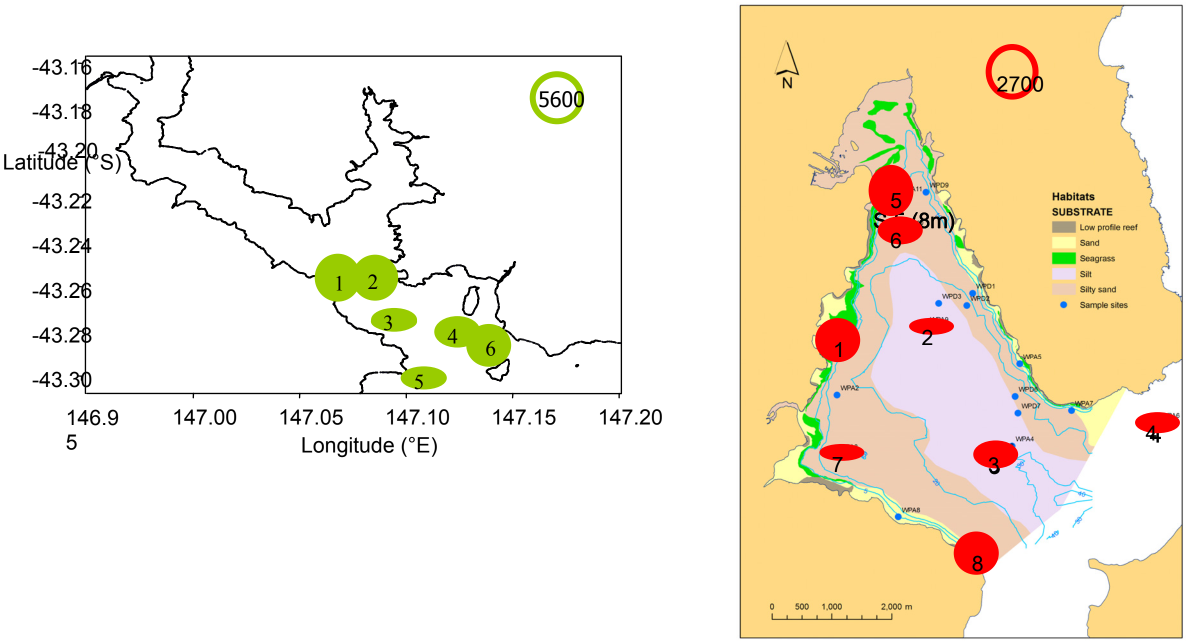

Figure 5.2. Spatial variation in mesozooplankton abundance (individuals m^{-3}). A) shows 6 sites sampled in the Huon Estuary in April 2005. B) shows 8 sites sampled in North West Bay in October 2006. Circles are proportional to the highest abundance at each location. Note that scales are different.

The microzooplankton (< 200 µm) community was assessed during experiments to measure grazing rates. These experiments took place in the Huon Estuary (n = 7), at Port Esperance (n = 1) and in North West Bay over the period from 2003 to 2007. Samples from various depths were collected and 5 to 13 samples from each site were counted and identified into the following groups: ciliates (small), heterotrophic dinoflagellates, tintinnids, radiolarians, copepod (nauplii), bivalve larvae, appendicularians, rotifers and *Polykrikos schwartzii*. In live samples, *P. schwartzii* was observed to be an important predator of autotrophic, chain-forming dinoflagellates. Its potential role as a major predator in this ecosystem provided the impetus to identify and enumerate *P. schwartzii* separately from other heterotrophic dinoflagellates.

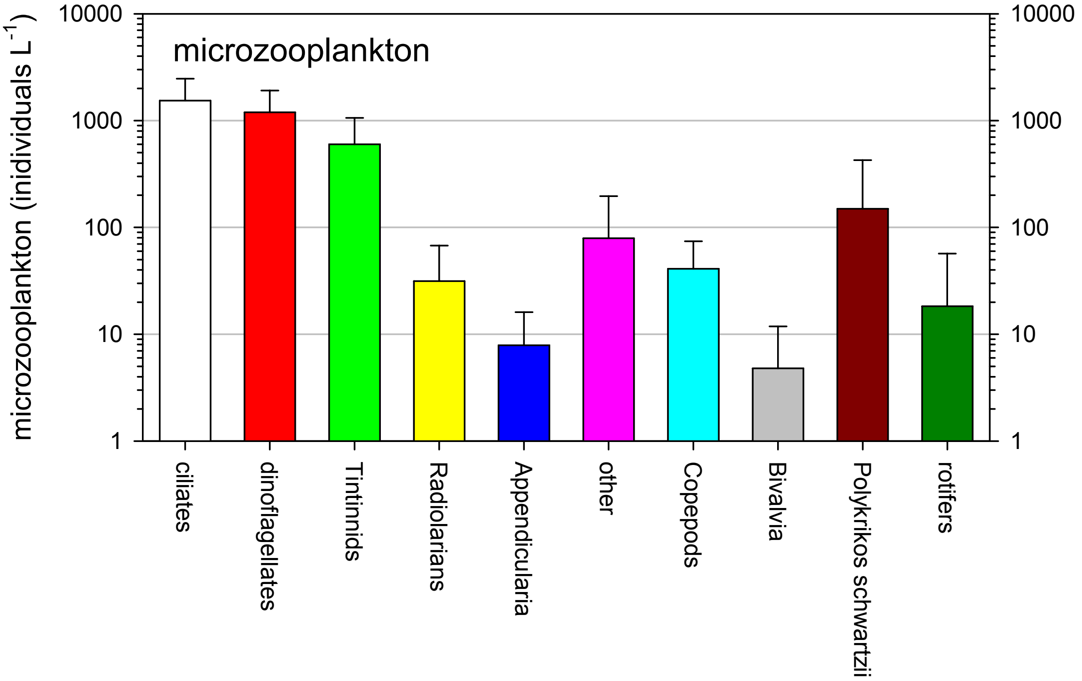

Figure 5.3. The mean and standard deviation of microzooplankton (< 200 µm) during the period from 2003 to 2007 in Huon Estuary (n = 7), at Port Esperance (n = 1) and in North West Bay. Samples from various depths were collected and 5 to 13 replicates from each site were counted and identified into the following groups: ciliates (small), heterotrophic dinoflagellates, tintinnids, radiolarians, copepod (nauplii), bivalve larvae, appendicularians (larval tunicates), rotifers and *Polykrikos schwartzii.*

Ranked by density, the dominant microzooplankton were ciliates, with 42% of the community (Fig. 5.3), followed by heterotrophic dinoflagellates (33%) and tintinnids (16%) with less than 10% in the other categories. The microheterotroph community observed in the Huon Estuary and D'Entrecasteaux Channel was similar in composition to those observed in coastal water bodies from Nova Scotia, Canada (Gifford, 1988) to South Africa (Froneman and McQuaid, 1997), with dominance by ciliates (small, aloricate) followed by tintinnids. Densities of both types of ciliates (aloricate + tintinnids) were ~ 2140 l^{-1}, very similar to the 2100 l^{-1} reported for waters around New Zealand (James et al., 1998; Hall et al., 2004). The major difference between microheterotrophs in the Huon Estuary and other coastal locations in the world seems to be the greater importance of heterotrophic dinoflagellates including *Polykrikos schwartzii.* Heterotrophic dinoflagellates are only rarely reported as the dominant grazers in coastal ecosystems but when present they appear to be quite selective grazers (e.g. Johnson et al., 2003). Relatively large numbers of heterotrophic dinoflagellates are characteristic of the oceanic Australasian region north of the subtropical convergence (Wood, 1954) and the seas around Tasmania (Wood, 1964).

5.3. Grazing

The grazing impacts of microzooplankton and mesozooplankton on the phytoplankton communities were examined during several process studies. The nature of these groups is such that different experimental techniques are employed to study their grazing rates. However, experiments were run in concert at Hideaway Bay in September 2005 and North West Bay in October 2006. In addition, during the North West Bay experiments, and later ones in Port Esperance in November 2007, mesozooplankton were directly added to microzooplankton dilution experiments to study the impacts of the larger grazers on the grazing dynamics of the smaller organisms.

Microzooplankton grazers included high proportions of ciliates, heterotrophic dinoflagellates and tintinnids, with smaller numbers of copepod nauplii, appendicularians and rotifers. Studies of mesozooplankton grazers were restricted to dominant omnivorous copepods (e.g. *Acartia tranteri*, *Paracalanus indicus*, and *Centropages australiensis*) cladocerans and appendicularians. Grazing rates of microzooplankton reached as high as 125% of daily primary production, while those of mesozooplankton herbivores were never greater than 20%. Trophic interactions between the species are complicated by the recent arrival of the heterotrophic dinoflagellate *Noctiluca scintillans* to the region. This species grazes heavily on phytoplankton, smaller zooplankton and faecal pellets. *Noctiluca scintillans* accounted for up to 20% of mesozooplankton abundance in autumn and it is capable of both suppressing zooplankton abundance and reducing the sedimentation of faecal pellets to the seafloor.

5.4. Microzooplankton grazing

Research on microzooplankton has suffered from historical neglect, largely because its size range overlaps with that of its prey and so experimentation can be difficult. However, in more recent times there has been an increase in strong experimental evidence that this group is one of the most important in biogeochemical cycles of important biological elements (Calbet and Landry, 2004; Sherr and Sherr, 2007; Thompson et al., 2008). In fact, recent research has shown that the majority of phytoplankton may be grazed by microzooplankton in some ecosystems and that their grazing impact usually exceeds that of mesozooplankton. They are also important contributors to mesozooplankton diet, especially under oligotrophic conditions (Calbet, 2008).

The principal location for microzooplankton grazing experiments was close to the mouth of the Huon Estuary, at the Hideaway Bay aquaculture facility of Huon Aquaculture Company Pty Ltd. Four experiments were carried out: one in September 2003, one in November 2003, one in February 2004, and a fourth in July 2004. Two further experiments were carried out at this site in April 2005, and one in September 2005 as part of process studies in the region. In October 2006, three experiments were performed at North West Bay, and in November 2007 the final three experiments were carried out at Port Esperance.

The technique developed to measure grazing rates on phytoplankton by microzooplankton uses a serial dilution of field samples (Landry and Hassett, 1982), and is referred to as the “grazing dilution technique”. The technique involves the collection of seawater containing prey items and microzooplankton with water bottles. The collected seawater is then diluted with filtered seawater collected from the same source. A series of dilution treatments is set up in replicate bottles, creating a gradient of grazer abundance. The treatments are then incubated under *in situ* light and temperature conditions for 24 hours. This technique has been widely used and enables the partitioning of grazing by different groups of organisms, as well as the capacity to compare with studies from other systems. The grazing dilution technique and calculations used in these experiments were adapted by Brian Griffiths (CSIRO Marine and Atmospheric Research) from the method of Landry and Hassett (1982). For experimental detail consult the accompanying technical report by Bonham et al. (2008).

In oceanic waters, nutrients approximately equal to those in the water mass, plus ammonia to a final concentration of 2 µM, would be added to each experimental bottle, to ensure that nutrients were equally available to phytoplankton at all dilutions. Nutrient addition at sites in the Huon Estuary and D’Entrecasteaux Channel was deemed unnecessary. Seasonal variation in N:P ratios suggest there may be nitrate limitation at some sites in the Huon in the summer months, particularly in times of dense microalgal blooms (Butler et al., 2000). However, the Huon Aquaculture site is near the mouth of the estuary, water at the site is generally well-mixed and a residual store of nitrate is available in bottom waters at the marine end of the estuary (Butler et al., 2000).

In the North West Bay process study (October 2006), three grazing experiments were performed to attempt to determine the effects of additional mesozooplankton. One set of samples was screened through a 200 µm mesh insert as usual, one set was not screened (presumed to have normal mesozooplankton densities), and additional mesozooplankton were added to the third set. At Port Esperance, there were also three experiments. All three sets of samples were screened through a 200 µm mesh insert as usual; the second set had 10 mesozooplankton added to each incubated bottle, and the third set had 30 additional mesozooplankton in each incubated bottle. These additional mesozooplankton were removed from the incubated bottles at the end of the experiment, before filtration. Initial samples were preserved at the start of each experiment to determine the concentrations of microzooplankton and their prey items. At the end of each experiment further samples were preserved for counting of cells and grazers.

Phytoplankton daily net growth rates are shown in Fig. 5.4 and indicate the seasonally variability evident in growth rates. The highest net growth rates occurred during the spring and late summer algal blooms. The final series of experiments (Port Esperance) suggested that the addition of mesozooplankton to the experimental bottles might have affected phytoplankton growth.

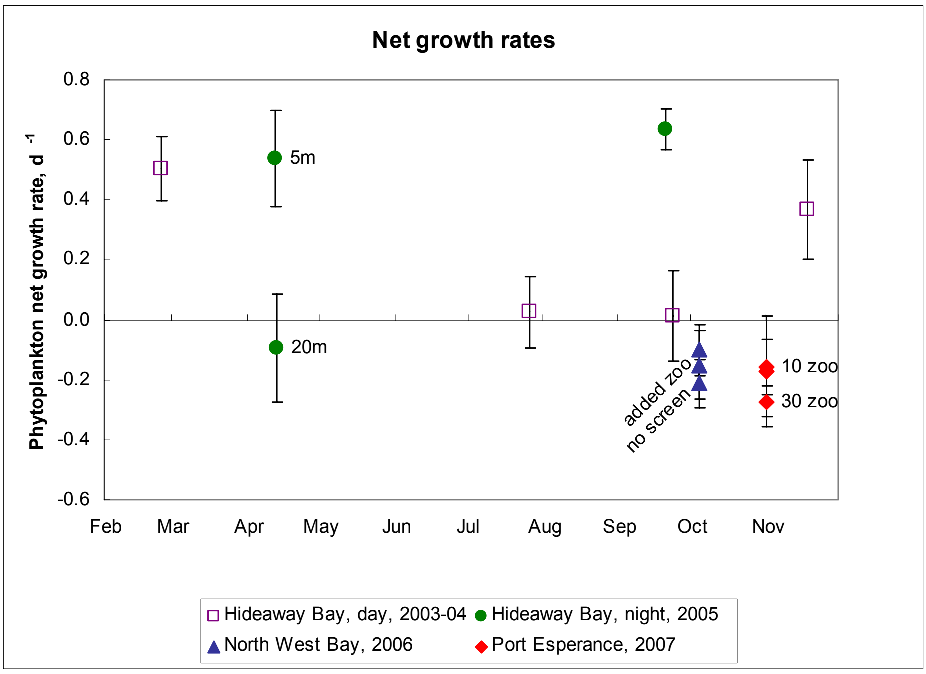

Figure 5.4. Net growth rates (μ_n) for sites in the Huon River and D'Entrecasteaux Channel, in 2003-2007. Error bars represent standard errors of the net rates. "zoo" = added mesozooplankton.

Growth rates in the North West Bay and Port Esperance are probably not directly comparable with the Huon diatom-dominated experiments, since these experiments were sampled during unusual events, one a vertically migrating spring dinoflagellate bloom, and the second during a late-spring bloom composed of senescing diatoms.

The microzooplankton grazing rates were calculated from each experiment (Fig. 5.5) and the overall grazing pressure on the phytoplankton standing stock was between 3 and 56%, while the proportion of new production grazed was between 7 and 125%.

Gross phytoplankton growth rates varied over the annual cycle in 2003-04 from about zero to 1.3 d^{-1}. The upper gross growth rate (1.3 d^{-1}) is approximately the maximum that can be expected for temperate phytoplankton species under the prevailing temperature conditions (Eppley, 1972). Net phytoplankton growth rates based on chlorophyll-*a* were sometimes small, but increased sharply during the blooms both during spring and in the late summer bloom reaching 0.37 to 0.5 d^{-1} (~40% of gross growth rates) during November 2003 and February 2004. That net growth rates may be small at certain times of the year is perfectly consistent with the observations of changes in widespread chlorophyll-*a* concentrations in the Huon Estuary and nearby D'Entrecasteaux Channel. For example, averaged over 12 sites in D'Entrecasteaux Channel the observed increase in chlorophyll-*a* concentration was a net growth rate of 0.005 d^{-1} during the annual transition from winter minimum to spring maximum.

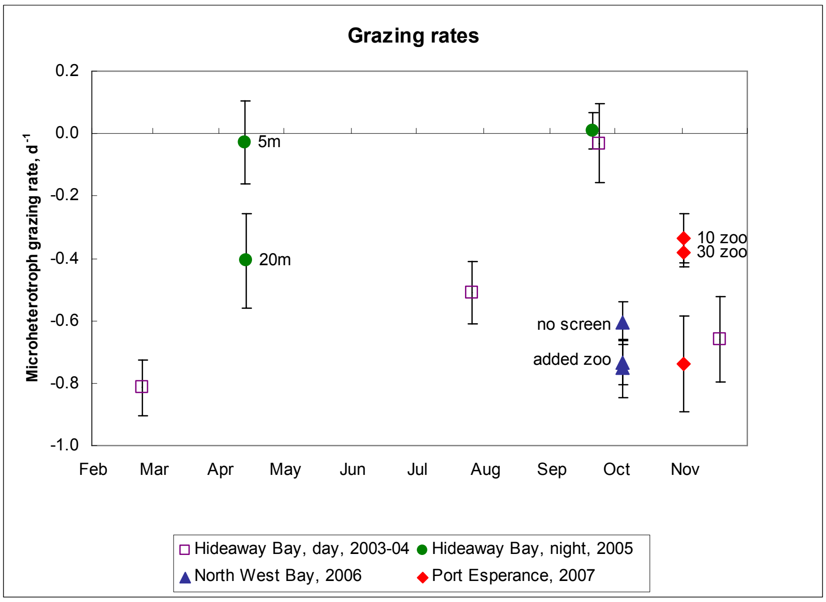

Figure 5.5. Grazing rates (*g*) for sites in the Huon River and D'Entrecasteaux Channel, in 2003–2007. Error bars represent standard errors of the rates. "zoo" = added meso-zooplankton.

Given that our experiments measured gross growth rates that are consistent with the demonstrated capability of the species present in this ecosystem and that net growth rates observed from local sampling and estimated by the grazing dilution technique were generally lower, sometimes considerably lower, we can conclude that microzooplankton grazing was always capable of consuming a substantial portion of the daily primary production. In other ecosystems around the world reported grazing rates were often lower, but range from ~ 40% to 100+% of primary production (see review by Calbet and Landry, 2004). As our observations indicate grazing rates in the Huon were always greater than 69% of gross production and reached 96% in winter they are at the upper end of those observed elsewhere. This comparison suggests that the Huon Estuary is characterized by a high degree of top-down control on phytoplankton.

The available data indicate that grazing pressure was always high in the Huon Estuary. Although this conclusion should be tempered by the lack of resolution in time and space it indicates that top-down control is an important component of the pelagic ecology in this ecosystem. For a phytoplankton bloom to occur a species or community must escape this strong grazing pressure for sufficient time to bloom. At the phytoplankton community level there are only a few mechanisms that would allow a general escape from microzooplankton predation. For example, there is often a lag between the increasing growth rate of phytoplankton relative to zooplankton in early spring, and the net result is a spring bloom. Spring blooms occur most years in the Huon Estuary and D'Entrecasteaux Channel, indicating this uncoupling does occur. During summer it is possible that grazing pressure is reduced as a result of a trophic cascade. Grazing by mesozooplankton on microzooplankton has been shown to reduce grazing on phytoplankton by 50% (Hansen et al., 1993).

We have observed episodic high densities of salps (tunicates) that appear to remove most of the particles from the water column. Subsequent to these salp "blooms" there may be an opportunity for phytoplankton growth to escape top-down control. It is also possible that *Noctiluca scintillans* may act in the same manner. In general the published accounts of trophic cascades involve non-visual feeders such as salps, ctenophores, medusae (as in Table 1 in Verity and Smetacek, 1996) or *N. scintillans.*

Selective grazing upon particular taxa was periodically observed in our experiments. The most compelling example came from a series of experiments conducted near the head of the Huon Estuary (latitude: 46.163°S, longitude: 146.975°E). During March 2005 an autumn phytoplankton bloom of > 80% dinoflagellates largely dominated by *Ceratium* (*C. tripos*, *C. furca* and *C. lineatum*), *Gymnodinium catenatum* and *Dinophysis* species and 2% diatoms was observed. The dominant microzooplankton were heterotrophic dinoflagellates (40%), followed by ciliates (36%) and tintinnids (16%), with an overall average density of 1,177 ± 1,795 (±1 standard deviation) individuals l^{-1}. The density of microzooplankton peaked on March 21 when the heterotrophic dinoflagellate *Polykrikos schwartzii* made up more than 50% of the total. Over the duration of this dinoflagellate bloom there was a strong preference by the microzooplankton to graze on diatoms in spite of their low abundance. The diatom marker pigment, fucoxanthin, had consistently greater pigment-specific grazing rates than those for the dinoflagellate marker pigment, peridinin (Fig. 5.6). The results clearly demonstrate that the microzooplankton preferentially graze upon diatoms.

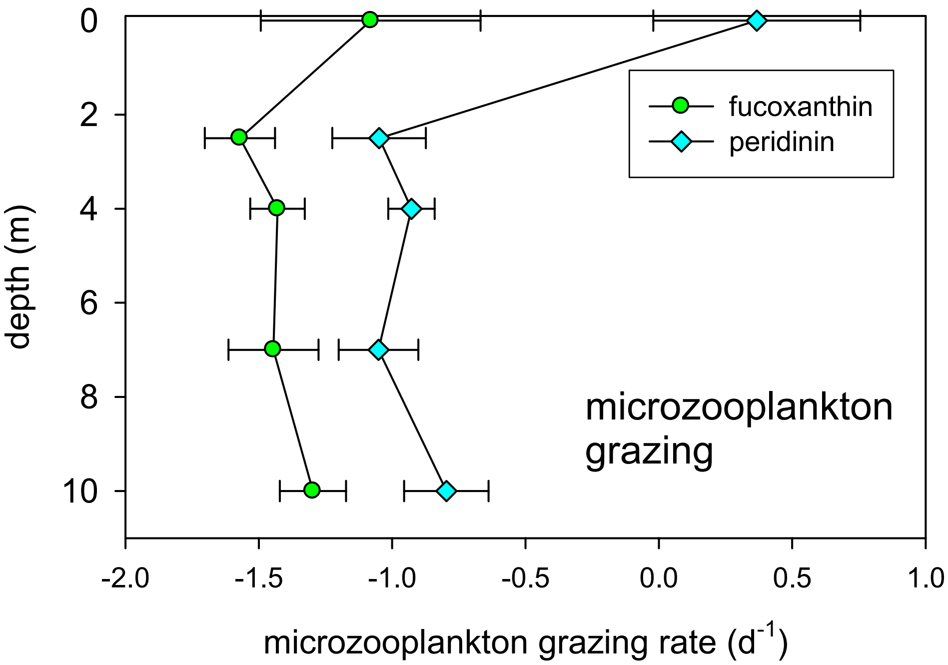

Figure 5.6. Pigment-specific rates of mortality for fucoxanthin and peridinin in a 10 meter water column in the Huon Estuary during March 2005.

5.5. Mesozooplankton grazing

Grazing experiments were conducted on two occasions: in Hideaway Bay, Huon Estuary on September 20–21, 2005, and North West Bay on October 1–2, 2006. Mesozooplankton for these experiments were obtained from slow tows with a Bongo net fitted with a closed cod-end, to reduce damage to the animals. Undamaged individuals were sorted from the cod-end, identified and a known number of one species was then placed in 1 litre bottles containing surface seawater. The bottles were incubated under *in situ* light and temperature for 24 hours. Natural seawater without additions was used, with the only modification being the removal of mesozooplankton grazers using a 200 µm mesh screen. The density of grazers was adjusted according to their size and literature clearance rates of congeneric species. Between 10 and 20 grazers were added to each experimental bottle. Microscope counts of initial and final subsamples of the incubation water were used to calculate clearance rates for each species. See the accompanying technical Report by Swadling et al. (2008).

Clearance rates (ml individual^{-1} day^{-1}) were converted to weight-specific ingestion rates [µgC (µg bodyC)$^{-1}$ d^{-1}] by multiplying the calculated clearance rate by phytoplankton carbon (µgC l^{-1}) and dividing the result by organism-carbon. Finally, grazing rate (µgC m^{-3} h^{-1}) for the mesozooplankton assemblage was determined by multiplying the above ingestion rate by biomass. The daily grazing rate of mesozooplankton was assessed by multiplying the hourly rate by 24. Many zooplankton are thought to exhibit diel periodicity in feeding, whereby they feed at higher rates at night, so these experiments integrate a 24 hour period and give no indication of day/night variation.

There was considerable variation in the clearance rates of the species examined. It is likely that the appendicularian *Oikopleura dioica* did not graze, as the animals were stressed by the incubation procedure and lost their feeding ‘houses’ during transfer to the experimental bottles. Clearance rates for the copepods and cladocerans are in accordance with literature values for congeneric species, and it is interesting that the clearance rates were so high given that the spring bloom had not yet developed in the region.

Small copepods, notably *Oithona* spp. and the cladoceran *Evadne* sp., always accounted for the highest proportion of grazing in each experiment. This observation agrees with that of Morales et al. (1991), who concluded that, while large species can dominate the biomass, numerically dominant small species may contribute the most to grazing pressure, and hence carbon turnover. It is now becoming clear that small copepods, including *Oithona* spp., are very common throughout the world's oceans and more work is needed to clarify their position in the marine trophic web.

Table 5.1. Mean (± S.D.) daily clearance and ingestion by copepods in the Huon Estuary, September 2005; n is the number of measurements.

Species (and Stage)	Clearance (ml ind^{-1} d^{-1})	Weight (μg C ind^{-1})	Rations (% body C d^{-1})	n
Acartia tranteri (adult females)	20.58 (1.14)	5.65	35	4
Paracalanus indicus (adult females)	12.89 (2.96)	4.87	18	3
Oithona spp. (CV and adults)	5.54 (1.68)	1.02	61	3
Evadne sp.	8.83 (8.99)	5.89	74	3
Calanus australis (adult females)	14.87 (11.02)	13.17	27	2
Centropages australiensis (CIV to adults)	21.13 (1.20)	8.75	59	3
Oikopleura dioica	0.38 (1.75)	3.5	2	2

5.6. Stable isotopes and food web dynamics

Stable isotope ratios, primarily carbon-13 to carbon-12 ($\delta^{13}C$) and nitrogen-15 to nitrogen-14 ($\delta^{15}N$), play an important role in food-web studies. This is because discrimination between each pair of isotopes occurs at every linkage in the food web. For example, animals metabolise ^{12}C more efficiently than ^{13}C due to the differences in atomic mass. Therefore, animals generally have higher concentrations of ^{13}C and ^{15}N in their body than is present in their food. Isotope ratios are expressed in terms of their per mil difference from a standard material, with values typically ranging from -30‰ to 5‰ for carbon (versus a calcite standard which is close to the mean composition of seawater), and -2‰ to 10‰ for nitrogen (versus atmospheric N_2).

The offsets between animals and their food are typically on the order of 1‰ for $\delta^{13}C$ and 3‰ to 4‰ for $\delta^{15}N$, but with considerable variation among different organisms. A review of both the principles of this approach and a number of aquatic ecosystem case studies is available in Fry (2006).

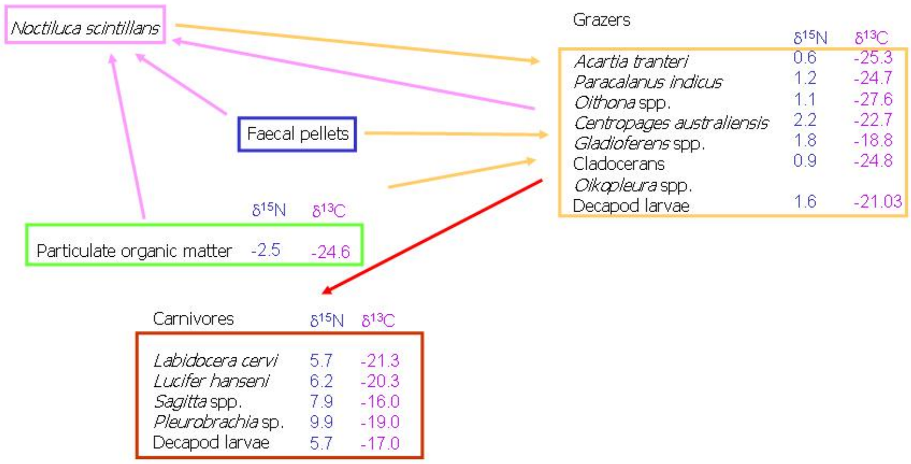

Figure 5.7. Food web in the Huon Estuary based on stable isotope analyses for δ^{13}C and δ^{15}N. Note that no measurements were made for *Noctiluca scintillans*, faecal pellets or *Oikopleura* sp.

Stable isotope analysis revealed consistent differences between species measured from the Huon Estuary and sites in the D'Entrecasteaux Channel. Fig. 5.7 shows a summary of δ^{15}N and δ^{13}C values that were measured for the dominant mesozooplankton species in the ecosystem. Analysis of δ^{15}N values showed that those animals listed in the grazer group were the first trophic level above the particulate organic matter (POM), with enrichment of approximately 3.1 to 4.7 ‰ above the POM. Average δ^{15}N values of the carnivore group were 8.2 ‰ to 12.4 ‰ above that of the POM, indicating two to three trophic transfers. This suggests that some carnivores (e.g. copepods and decapod larvae) were preying directly on the grazers, while, in turn, they were being consumed by higher order carnivores (e.g. the ctenophore *Pleurobrachia* spp.). δ^{13}C values suggest that several of the dominant grazers (e.g. *Acartia tranteri*, *Paracalanus indicus* and cladocerans) were largely ingesting POM available in the pelagic environment. Other taxa, such as *Gladioferens* spp. and decapod larvae, show different profiles, indicating that their diet might have originated from other sources including benthic material. *Gladioferens* spp. are known to hook onto macrophytes so they can maintain their position in the water column and so it is possible that they also graze particulate matter from the surface of these plants.

5.7. Summary

This study has shown that microzooplankton and mesozooplankton both make significant contributions to the ecology of the Huon, Channel and North West Bay ecosystems. While microzooplankton make an enormous contribution to grazing on primary production compared to mesozooplankton, the products of mesozooplankton make a strong contribution to carbon (and other elements) cycling.

In terms of the trophic characteristics of this system, it is interesting that the grazing impact of microzooplankton on phytoplankton is high, an average of 69–125% of primary production being consumed per day. This value contrasts with the smaller overall impact of mesozooplankton, between 10 to 30% per day. The smaller contribution by mesozooplankton to total community grazing in oligotrophic systems is variable, but certainly diverges from the traditional view of the classical herbivorous food chain (diatoms-copepods-fish; Calbet, 2008). It should be remembered, however, that copepods and large metazoans need longer developmental times (usually weeks to months) than microzooplankton (days), and it is likely that in many instances they do not cope with strong fluctuations in food supply.

An overall higher relevance of microzooplankton as a control factor of phytoplankton populations does not mean that large mesozooplankton are not important in the food webs of the Huon and D'Entrecasteaux. Even if their grazing impact is generally lower than that of microzooplankton they are still important agents for structuring food webs and they remain a crucial link between primary production and fish. As our study has shown, mesozooplankton can suppress phytoplankton growth both by direct consumption and by changing the grazing rates of microzooplankton. For example, ciliates and heterotrophic dinoflagellates are important food for copepods, often equaling phytoplankton in the amounts consumed by the crustaceans. However, it is also worth noting that mesozooplankton grazing is species-specific and while some species might play important roles in trophic cascades others are opportunistic feeders that have little impact on the path of nutrients through the system.

One important uncertainty in this ecosystem is the effects of the recent arrival of *Noctiluca scintillans*. This large (up to 2 mm) dinoflagellate has only been observed in southern Tasmanian waters since the 1990s, and it appears that increasing seawater temperatures are contributing to its expanding distribution; it has now established year-round populations in some sheltered waters on the east coast of Tasmania. *N. scintillans* is capable of ingesting a wide variety of prey, including diatoms, other protists and small mesoplankton, while, in turn, it is consumed by mesozooplankton. Thus it is both a grazer of, and a competitor with, local plankton, as well as a potentially significant prey item. The dinoflagellate occasionally forms massive blooms and under these conditions the abundance of dominant mesozooplankton species is reduced. Furthermore, the copepods and cladocerans that are associated with a *N. scintillans* bloom often appear in poor condition (K. Swadling personal observation). In future, the impacts of stressors such as introduced species into the Huon and D'Entrecasteaux ecosystem need to be understood to ensure the continued good health of the planktonic component. Any changes could upset the tight trophic coupling that is evident in this system, resulting in unpredictable changes to the planktonic food web.

5.8. Acknowledgements

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5.9. References

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Chapter 6

Sediment Biogeochemistry

6.1. Introduction

The sediment biogeochemistry of the Huon Estuary was studied as part of the PhD project of Dean Thomson at the University of Tasmania supervised by Dr Chris Burke and Dr John Volkman (Thomson, 2008). Sediments were collected in March, July and November in 2004 and in April 2005. These were returned to the laboratory and incubated to determine fluxes of the major nutrients. To represent the terrestrial and marine end members of the estuary, locations were situated in the upper and lower reaches respectively (Fig 6.1) Additional studies of sediment biogeochemistry and organic matter fluxes to sediments were carried out in Port Esperance, North West Bay and D'Entrecasteaux Channel by C. Macleod, A. Revill, D. Holdsworth and J. Volkman. Details of these studies can be found in the accompanying technical reports (Holdsworth et al., 2008; Macleod et al., 2008; Revill et al., 2008).

This chapter is largely based on Chapter 4 in Thomson's PhD thesis. The sources of organic matter and rates of decomposition were investigated as well as fluxes of nutrients liberated during organic matter decomposition. The study aimed to develop a conceptual understanding of benthic respiration and nutrient cycling in the Huon Estuary and the influence of organic carbon loadings on these processes. This study also sought to evaluate the ecological significance of nutrient inputs from sediments compared with other nutrient sources to the estuary.

6.2. Organic matter remineralisation

Estuaries act as an important link between the land and the sea and therefore play a crucial role in coastal ecosystems. High sedimentation rates in estuaries mean that they play a crucial role in recycling carbon and nutrients to the broader ecosystem. Labile organic matter (such as that produced by phytoplankton) depositing onto the seafloor is rapidly remineralised leading to release of CO_2 back into the water column. However, organic matter that is more refractory (such as that derived from terrestrial plants) is degraded only slowly and so much of it will be sequestered within the sediments (Kristensen, 2000). If sedimentation rates are high, a substantial proportion of the labile organic matter may also be buried and degraded at depth in the sediment by both aerobic and anaerobic processes.

Particulate organic nitrogen (PON) is remineralised during organic matter decomposition. The remineralised nitrogen can be released back into the overlying water column, as dissolved organic nitrogen (DON), ammonium or nitrate, which can be readily available for uptake by phytoplankton, or in the case of ammonium, reassimilation by heterotrophic bacteria during organic matter decomposition within the sediment (Lomstein et al., 1998; Kristensen, 2000). Alternatively the nitrogen can be simultaneously converted to di-nitrogen gas (N_2) by denitrification or Anaerobic Ammonium Oxidation (the so-called ANAMMOX reaction) and therefore no longer available for phytoplankton uptake.

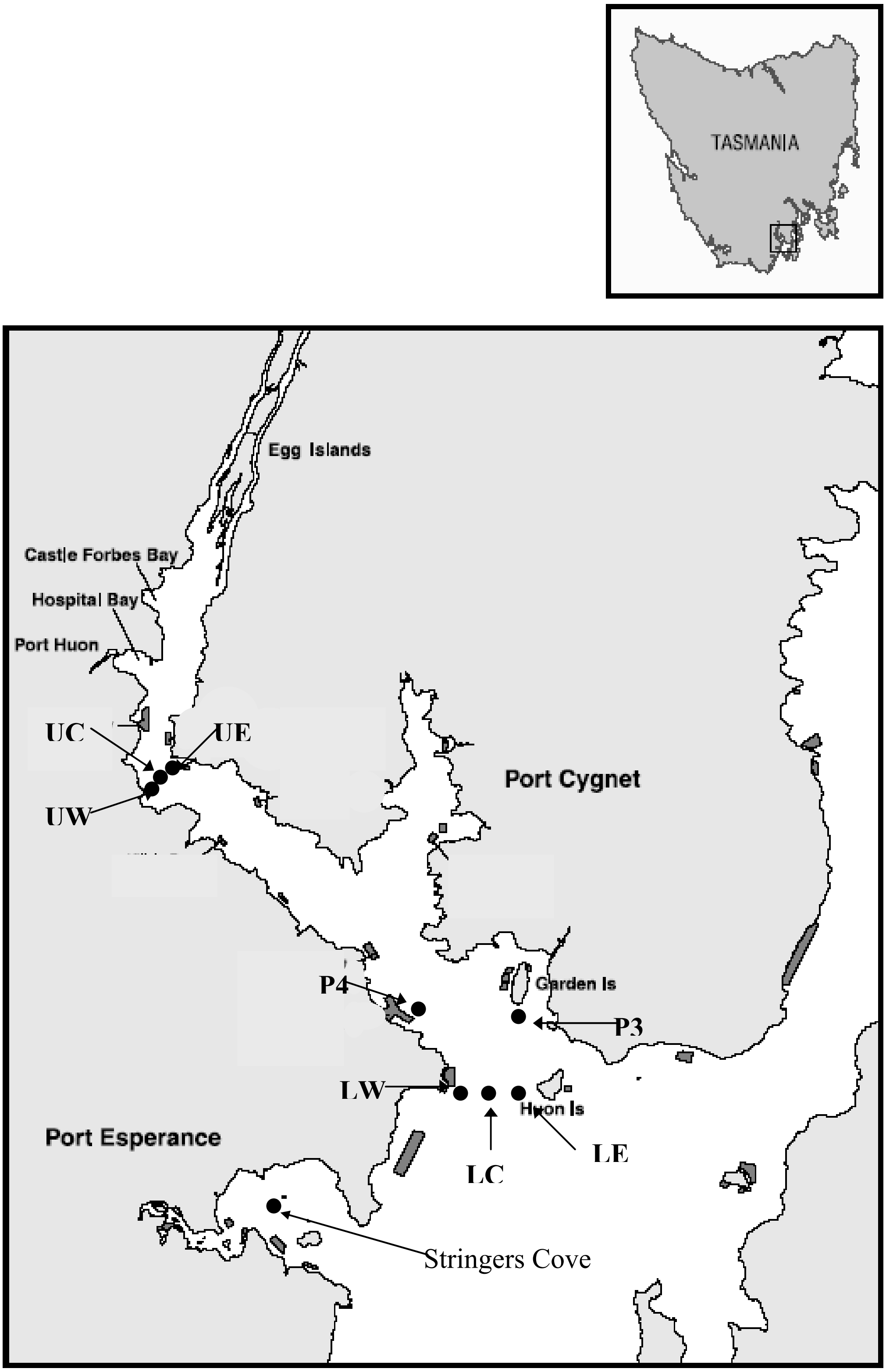


Figure 6.1. Sampling sites for the sediment biogeochemistry. The sampling locations include the upper estuary stations UE, UC and UW, the lower estuary stations LE, LC and LW, and additional sampling stations at Hideaway Bay (P4) and Garden Island (P3). Also included was a sampling station at Stingers Cove for comparative purposes.

6.3. Organic matter sources in the Huon Estuary

Geochemical parameters such as C/N ratio and stable carbon isotope values indicated that all sites had a mixture of organic matter derived from phytoplankton, bacteria and terrestrial plants. Sediments from the upper estuary contained a greater proportion of terrestrial organic matter as shown by slightly more negative $\delta^{13}C$ values and higher C/N ratios (Table 6.1). The latter also contained much higher contents of organic carbon (8.1 vs. 4.1 % dry wt; Table 6.1). Only minor differences were found between sampling stations within each location. Lipid biomarkers showed a strong terrestrial signature which was greater in winter, most likely due to an increase in input from catchment runoff (Thomson, 2008).

Table 6.1. Organic carbon and nitrogen as % dry wt. (%C_{org} and %N_{org}), stable isotope values $\delta^{13}C$ and $\delta^{15}N$, carbon:nitrogen ratios in organic matter (C/N), silt/clay fraction and porosity in surface sediments of the study locations

	%C_{org}	%N_{org}	C/N	$\delta^{13}C$ ‰	$\delta^{15}N$ ‰	% sediments < 63 µm	Porosity
LW							
March	4.5	0.5	9.0	-21.7	7.7	-	
July	3.8	0.4	9.5	-19.7	7.6	51.5	
November	3.6	0.4	9.0	-20.3	8.3	-	0.73
LC							
March	4.7	0.5	9.4	-21.3	7.4	-	
July	4.6	0.5	9.2	-20.3	7.4	43.7	
November	4.5	0.5	9.0	-19.9	8.1	-	0.81
LE							
March	3.4	0.3	11.3	-20.0	7.3	-	
July	3.9	0.4	9.8	-21.1	7.4	53.3	
November	3.8	0.4	9.5	-21.0	8.0	-	0.80
UW							
March	7.7	0.6	12.8	-22.3	5.2	-	
July	7.8	0.6	13.0	-22.3	5.1	60.1	
November	7.7	0.5	15.4	-22.2	5.9	-	0.87
UC							
March	8.1	0.6	13.5	-21.7	4.6	-	
July	8.2	0.6	13.7	-22.3	4.7	64.3	
November	8.0	0.5	16.0	-22.3	5.3	-	0.83
UE							
March	8.2	0.5	16.4	-22.4	4.2	-	
July	8.8	0.6	14.7	-21.9	4.1	57.0	
November	8.7	0.5	17.4	-22.1	4.8	-	0.77
Average Lower Stations	4.1	0.4	9.5	-20.6	7.7	49.5	0.78
Average Upper Stations	8.1	0.6	14.8	-22.2	4.9	60.5	0.82

6.4. Benthic metabolism

Sedimentary organic matter can be remineralised either through aerobic or anaerobic pathways. Organic matter can generally be characterised as having the following chemical composition:

$$\mathbf{(CH_2O)}_{\boldsymbol{a}}\ \mathbf{(NH_3)}_{\boldsymbol{b}}\ \mathbf{(H_3PO_4)}_{\boldsymbol{c}} \qquad \mathbf{(1)}$$

The numerical value of *a*, *b* and *c* largely depends on the source of the organic matter and stage of decomposition.

When organic matter is initially deposited onto the sea floor, organisms such as benthic fauna, bacteria and fungi (Kristensen, 2000) aerobically decompose the organic matter according to the following stoichiometry:

$$\mathbf{(CH_2O)}_{\boldsymbol{a}}\ \mathbf{(NH_3)}_{\boldsymbol{b}}\ \mathbf{(H_3PO_4)}_{\boldsymbol{c}} + \boldsymbol{d}\mathbf{O_2} \rightarrow \boldsymbol{a}\mathbf{CO_2} + \boldsymbol{b}\mathbf{NH_3} + \boldsymbol{c}\mathbf{H_3PO_4} + \boldsymbol{d}\mathbf{H_2O} \qquad \mathbf{(2)}$$

Because aerobic respiration occurs in a relatively thin oxidised upper layer in coastal sediments, a large fraction of the organic matter can be buried into anoxic layers. Here organic matter is decomposed anaerobically according to the following stoichiometry:

$$\mathbf{8(CH_2O)}_{\boldsymbol{x}}\ \mathbf{(NH_3)}_{\boldsymbol{y}}\ \mathbf{(H_3PO_4)}_{\boldsymbol{z}} \rightarrow$$
$$\boldsymbol{x}\mathbf{CH_3CH_2COOH} + \boldsymbol{x}\mathbf{CH_3COOH} + \mathbf{3}\boldsymbol{x}\mathbf{CO2} + \mathbf{3}\boldsymbol{x}\mathbf{H_2} + \mathbf{yNH_3} + \boldsymbol{z}\mathbf{H_3PO_4} \qquad \mathbf{(3)}$$

Anaerobic metabolism proceeds in a sequence of metabolic processes that generally change with depth. The following order in which anaerobic metabolism occurs will depend on the availability of electron acceptors (Kristensen, 2000):

- Mn^{4+} Manganese oxide reduction
- NO_3^- Denitrification
- Fe^{3+} Iron oxide reduction
- SO_4^{2-} Sulphate reduction
- CO_2 Carbon dioxide reduction

In general, aerobic and anaerobic respiration contributes approximately 50% each to total respiration with sulphate reduction often the largest contributor within anaerobic respiration in marine sediments (Kristensen, 2000), although recent studies have shown dissimilatory iron and manganese reduction to be important in some environments (Arnosti and Holmer, 2003). Benthic metabolism can be influenced by a number of factors including organic enrichment, bioturbation/irrigation and temperature (Aller, 1994; Thamdrup et al., 1998).

Benthic respiration is directly influenced by the source and supply of labile organic matter. In sediments that undergo organic enrichment either following phytoplankton blooms or by anthropogenic inputs, sediments may become anaerobic due to the depletion of oxygen within the sediments. The oxygen penetration depth is directly influenced by the supply of labile organic carbon, with shallow oxic zones related to high respiration rates due to increased organic loading (Kristensen, 2000).

Temperature affects the rate of both anaerobic and aerobic respiration. Thamdrup et al. (1998) showed that the rate of aerobic respiration in marine sediments increased by a factor of 2–3 for a temperature increase of 10 ^{o}C. Likewise, studies have shown that sulphate reduction increases by a factor of 3 for a temperature increase of 10 ^{o}C (Westrich and Berner, 1988 and references therein).

Benthic respiration can also be influenced by a variety of benthic faunal processes including bioturbation and irrigation. Bioturbation can enhance the movement of particles within sediments, which leads to greater substrate exposure, and hence an increase in organic matter decomposition. Sediment particles may also be transported between anaerobic and aerobic environments resulting in increased re-oxidation and possibly increased remineralization (Aller, 1994). Irrigation and ventilation within and around worm burrows can also enhance benthic respiration by supplying oxygen and other oxidized compounds (electron acceptors) at depth in the sediments (Kristensen, 2000).

6.5. Nutrient recycling

During the remineralisation of organic matter, inorganic nutrients are released as shown by the simplified stoichiometric equations for aerobic (equation 2) and anaerobic respiration (equation 3) above. Of particular interest in Australian coastal environments is the nitrogen cycle, since nitrogen is generally the limiting nutrient for phytoplankton growth. The ammonium produced may then be assimilated by bacteria, released from the sediment, become adsorbed onto sediment particles or oxidized to NO_3^- (Blackburn and Henriksen, 1983; Klump and Martens, 1983).

6.5.1. Nitrification

Nitrification is the process whereby ammonium is oxidised to nitrite (by nitrosofying bacteria such as *Nitrosomonas* sp.) and thence to nitrate (by nitrifying bacteria such as *Nitrospina* sp.) (Ward, 2000). This can be represented by the following stoichiometry:

$$\mathbf{NH_4^+ + 1.5O_2 \rightarrow NO_2^- + H_2O} \qquad \mathbf{(4)}$$

$$\mathbf{NO_2^- + 0.5O_2 \rightarrow NO^-_3} \qquad \mathbf{(5)}$$

Nitrification is typically an aerobic process, and is generally confined to the oxic zone within sediments. In some cases, nitrification may also be coupled to manganese IV reduction within the anaerobic zone (Hulth et al., 1999). Nitrification is regulated by a number of factors including the availability of O_2 and NH_4^+, as well as salinity and pH. These, in turn, are often controlled by the metabolism of bacteria, benthic algae and macrofauna.

Nitrification can be significantly inhibited by the activity of benthic diatoms (Henriksen and Kemp, 1988) and heterotrophic bacteria (Strauss and Lamberti, 2000), possibly through competition for NH_4^+. Rates of nitrification are generally highest around macrofauna burrows, due to the occurrence of relatively high concentrations of NH_4^+ within an oxic environment (Henriksen and Kemp, 1988). Nitrification is important in the nitrogen cycling pathway as it is the precursor to denitrification.

6.5.2. Denitrification

Denitrification is the reduction of NO_3^- to N_2 gas mediated by bacteria (Hulth et al., 2005). The process of denitrification requires the availability of NO_3^- (or NO_2^-), organic carbon and anoxic conditions. Denitrification is represented by the following stoichiometry:

$$\mathbf{(CH_2O)_{106}(NH_3)_{16}(H_3PO_4) + 94.4HNO_3 \rightarrow 106CO_2 + 55.2N_2 + H_3PO_4 + 177.2H_2O\ (6)}$$

The bacteria involved in denitrification can obtain nitrate from either the water column or from nitrification in sediments. When nitrate for denitrification is obtained from nitrification, the process is called coupled nitrification–denitrification (Capone, 2000). Factors that may influence denitrification include oxygen, NO_3^-, and organic matter concentrations (Seitzinger, 1990). High rates of denitrification are often found in systems with high NO_3^- concentrations in the water column (Trimmer et al., 2000), while denitrification rates in systems with low concentrations of NO_3^- depend entirely on nitrification occurring within the sediment.

6.5.3. ANAMMOX

An alternative pathway to denitrification is anaerobic ammonium oxidation (ANAMMOX) whereby ANAMMOX bacteria convert nitrite and ammonium to N_2 (Hulth et al., 2005). However the ANAMMOX reaction can only take place in the presence of both ammonium and nitrite in an anoxic environment. The ANAMMOX reaction is strongly inhibited by the presence of dissolved oxygen (Hulth et al., 2005). Nedwell et al. (1999) suggested that the ANAMMOX reaction is most likely to occur in hyper-nutrified environments. Since that study however, Kuypers et al. (2005) discovered the occurrence of ANAMMOX in the oxygen minimum zone of the Benguela upwelling off the coast of Namibia implying that it could be more widespread.

6.5.4. Dissimilatory nitrate reduction to ammonium (DNRA)

This process is mediated by bacteria that couple the oxidation of organic carbon or reduced Fe and S to the reduction of nitrate by nitrite to ammonium (Hulth et al., 2005). It appears to be most important in organic-rich sediments with low NO_3^- concentrations (Nedwell et al., 1999). In a recent study, sediments enriched by fish farm effluent were shown to have DNRA rates 7 times higher than denitrification, while sediments unaffected by the fish farm effluent had insignificant rates of DRNA (Christensen et al., 2000). This is important for nutrient cycling as it can lead to large supplies of ammonium to the water column, where otherwise it may have been removed from the environment by denitrification or ANAMMOX.

6.6. Sediment nutrient fluxes in the Huon Estuary

The Huon Estuary is strongly stratified, with two-layer circulation driven by relatively high annual freshwater discharge from the Huon River (Butler et al., 2000). It also has high annual NO_x and phosphate concentrations in deep water at the mouth of the estuary in the D'Entrecasteaux Channel. This, combined with the 2-layer circulation, results in a large influx of NO_x and phosphate into the estuary bottom waters (Butler et al., 2000).

There is, however, some doubt about the source of NO_x in bottom waters at the mouth of the estuary during summer due to high nitrite: nitrate ratios. These observations suggest that a substantial fraction of the nitrate may be due to nitrogen recycling and nitrification in the water column and sediments, in both the lower Huon Estuary and the D'Entrecasteaux Channel. If this is the case, it may have significant implications for the impact of finfish-farm loads on the estuary (Butler et al., 2000).

The aim of the research reported here was to measure rates of nutrient fluxes at the sediment-water interface concentrating on: ammonia, nitrite, nitrate, phosphate and silicate across both temporal and spatial scales (for details see Thomson, 2008). Flux stoichiometry was then used to define the processes that are of importance in the recycling and burial of nutrients in Huon Estuary sediments. Additionally, ammonia, nitrate and nitrite in pore waters were measured during the last sampling campaign in spring to compare measured nutrient fluxes with diffusive fluxes calculated from sediment pore water profiles.

6.7. Methods

6.7.1. Sediment collection

The upper and lower estuary locations (Fig. 6.1) were visited three times in 2004, in March, July and November, using a small research vessel. The two locations were sampled on separate occasions, generally 3 days to a week apart, due to resource and labour limitations. Results from Hideaway Bay and Garden Island (Fig. 6.1) sampled in April 2005 serve as a comparative study of the benthic processes with the upper and lower estuary locations. At each site a box corer was used to take intact sediment samples. From each box-core sample, an undisturbed sub-core was taken using 25 cm long polyethylene tubes with an inner diameter of 9.7 cm to measure benthic respiration and nutrient exchange. Three replicates were obtained at each site.

The remaining surface sediment (top 1 cm) that was not collected in the sub-cores was collected from the box core and placed into glass jars and frozen until analysed for stable isotope ($\delta^{13}C$ and $\delta^{15}N$) measurements, carbon and nitrogen content, lipid biomarkers and sediment grain size. Cores (4.8 cm inner diameter, i.d) for nutrient porewater analysis were taken separately, in duplicate, in November 2004 at the Upper and Lower locations and again in April 2005 at Hideaway Bay and Garden Island.

The cores were then held in an ice-cooler at *in situ* temperatures. Near-bottom water (~160 l) was collected using an 8 l Niskin bottle and stored in 20 l plastic carboys. Temperature and salinity were determined in surface and bottom water on each occasion using a digi-thermo thermometer and refractometer respectively. Secchi depth was also measured in July and November. The cores and water were then transported back to the laboratory within six hours of sampling. The bottom water was placed into an incubation trough and glass aquarium (60 l) and brought to *in-situ* temperatures using a thermostatically controlled recirculating water bath and temperature thermostat probe respectively.

Prior to submerging the cores for benthic respiration and nutrient exchange into the incubation trough, the sediment volume was standardised so that there was a sediment depth between 8 and 10 cm and from 11 to 13 cm of overlying water. The uncapped cores were then left to re-equilibrate in darkness overnight prior to measurements. To ensure a good exchange between the water phase of the core and the exterior seawater during the re-equilibration period, a Teflon-coated stirrer bar (5 cm length) was suspended ~5 cm above the sediment surface in each core. This was driven by an external magnet rotating at 50–60 rpm.

6.7.2. Sediment-water nutrient fluxes

Rates of nutrient exchange (ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), phosphate (PO_4^{3+}) and silica (Si) in the dark, as well as O_2 and TCO_2, were measured on the day after sampling following re-equilibration. The incubations were started by capping the cores and waiting 1 hour prior to making the first measurement. Oxygen and pH were measured by inserting electrodes into the core water sampling port. Water samples for nutrient and alkalinity analysis determinations were taken from the core water by withdrawing 40 ml of sample into a plastic syringe through a Luer lock valve fitted to the lid. The water withdrawn from the core was simultaneously replaced with water from a gravity-fed reservoir. The water samples were then filtered through Acrodisc 0.45 µm cellulose acetate filters into 10 ml screw-cap polypropylene containers; alkalinity samples were stored at 4 °C in the dark and analysed within 2 weeks of sampling while nutrient samples were stored at -20 °C and were analysed within 3 months of sampling unless otherwise stated in the text. Measurements were taken every 2–4 hours depending on the rate of decrease in oxygen saturation. Core incubations were terminated prior to dissolved oxygen (DO) decreasing to 80% saturation of the starting concentration and total incubation times ranged between 8 to 12 hours.

6.7.3. Nutrient porewater analysis

For porewater extractions, cores were rapidly sectioned under atmospheric conditions at 0.5 cm intervals from 0 to 2 cm, at 1 cm intervals from 2 to 4 cm, and at 2 cm intervals from 4 to 8 cm. Slices were placed in a centrifuge tube and then centrifuged at 2000 rpm for 10 minutes. The water was then extracted and filtered through Acrodisc 0.4 micron filters into 10 ml polystyrene tubes and frozen at -20 °C for analysis of NH_4^+, NO_2^- and NO_3^-. Analytical methods for nutrient analysis are given in Thomson (2008).

6.7.4. Sediment–water flux calculations

The flux across the sediment-water interface was calculated as:

$$flux = \alpha - \alpha_w \times \frac{V}{A}$$

Where:

α = linear regression of analyte concentration (corrected for addition of replacement water) versus time in sediment core (µmol l^{-1} h^{-1})

α_w = linear regression of analyte concentration (corrected for addition of replacement water) versus time in “blank” core

V = water column volume (l) and,

A = sediment surface area (m^2).

The linear regression used 3 or 4 data points. The flux was only taken as being significant if the standard error of the slope of the line was less than the magnitude of the flux (Cook et al., 2004).

6.7.5. Modelled sediment NH_4^+ production rates

Estimates of the upward flux of NH_4^+ within the sediments were calculated according to the method described in Cook et al. (2004). Briefly NH_4^+ fluxes were calculated using Fick's first law:

$$F = -\varphi(D_s + D_e)\frac{\Delta C}{\Delta x}$$

Where φ is porosity, $\Delta C/\Delta x$ is the concentration gradient obtained using linear regression of the linear portion of the concentration profiles (4 to 5 data points), D_e is the enhanced sediment diffusion of solutes and D_s is the sediment diffusivity. D_s was calculated according to Iversen and Jorgensen (1993). An estimate of enhanced diffusion (D_e) was obtained by comparing the measured fluxes of oxygen to those calculated from O_2 micro-profiles (See Chapter 3 in Thomson, 2008 for more details). D_e was calculated according to Berg et al. (2001):

$$D_e = D_s\left(\frac{F_{meas}}{F_{calc}} - 1\right)$$

Where F_{meas} is the measured flux in intact cores and F_{calc} is the calculated diffusive flux profile assuming $D_e = 0$. Diffusion coefficients for NH_4^+ were obtained from Li and Gregory (1974) and corrected for temperature. Diffusion coefficients for O_2 were obtained from Rasmussen and Jorgensen (1992). NH_4^+

6.8. Results

6.8.1. Physico-chemical water column conditions

Physico-chemical water column conditions are reported in Table 6.2. Salinity was stable at the Lower location, but dropped slightly in winter at the Upper location, reflecting its position in the mixing zone of the estuary and exposure to greater freshwater inflows during winter months. Water temperature followed a temporal pattern, with highest recorded in March (late summer/early autumn) and lowest in July (winter). Nitrate concentrations were low, reflecting the low nutrient runoff from the catchment, and peaked in winter, most likely derived from the marine end of the estuary (Butler et al., 2000). Ammonium and phosphate concentrations were generally low and variable, reflecting the low inputs to the estuary. Silicate (strictly this is reactive Si as measured by molybdenum blue complexation and mostly consists of silicic acid $Si(OH)_4$) varied over the sampling period and was highest in July at both study locations, when it was most likely derived from the terrestrial end of the estuary (Butler et al., 2000).

Table 6.2. Physico-chemical conditions of bottom water at the Upper and Lower locations during 2004. (nm – not measured).

		Lower	Upper
Water Depth (m)		25-37	10–19
Salinity (‰)	- Mar	35	35
	- Jul	35	28–34
	- Nov	35	34
Temp (^{o}C)	- Mar	15.5	15.7
	- July	10.3	8.0–8.2
	- Nov	13.0	12.8
Secchi Depth (m)	- Mar	nm	nm
	- Jul	1.5	<1
	- Nov	2.0	<1
Ammonium (µM)	- Mar	0.8	1.3
	- Jul	0.1	0.3
	- Nov	0.3	3.1
Nitrate (µM)	- Mar	0.7	0.6
	- Jul	5.1	5.0
	- Nov	1.6	1.5
Phosphate (µM)	- Mar	0.5	0.3
	- Jul	0.6	2.0
	- Nov	0.4	0.5
Silicate (µM)	- Mar	3.8	1.5
	- Jul	9.0	18.1
	- Nov	5.3	5.8

6.8.2. Sediment nutrient fluxes

Fluxes of dissolved inorganic nitrogen (DIN) $[NH_4^+ + NO_2^- + NO_3^-]$ were always directed out of the sediment, with the highest mean flux rate recorded in March at UW (18.1 ± 9.0 µmol m^{-2} h^{-1}) and the lowest mean flux rate recorded in July at UM (0.9 ± 4.7 µmol m^{-2} h^{-1}). DIN fluxes were found to be significantly different between the 3 sampling periods (Kruskal-Wallis, $P < 0.05$). Mann-Whitney pair-wise comparisons found that March had significantly higher flux rates compared to July ($P = 0.00001$) and to November ($P = 0.010$). July had significantly lower flux rates compared to November ($P = 0.00001$).

Kruskal-Wallis analysis was also used to determine differences across spatial scales. No significant difference was found between stations at either the lower ($P = 0.2202$) or the upper ($P = 0.4158$) locations. All the data from the three stations were then pooled for each of the locations and the two locations were compared on an annual basis. The annual mean flux at the upper and lower locations was not significantly different ($P = 0.5913$).

Mean fluxes of ammonium were generally directed out of the sediment during March with the exception of site LC (Figs. 6.2 and 6.3). During July all sites except LE had mean ammonium fluxes directed into the sediment. In November, all sites at the lower estuary had small fluxes directed out of the sediment, but all sites at the upper estuary location ammonium were directed from the water column to the sediments. During the course of the study, none of the ammonium fluxes differed significantly between stations at the upper location (Kruskal-Wallis, $p = 0.8394$) while at the lower location, LC and LE were significantly different (Mann-Whitney, $p = 0.0170$). The highest ammonium efflux rate across all sites during the sampling period occurred at site LE during March with a mean flux of 6.4 ± 3.7 µmol m^{-2} h^{-1}out of the sediment.

Nitrate was always directed out of the sediment (Figs. 6.2 and 6.3) and was the dominant form of DIN flux (mean range 3.7 to 17 µmol m^{-2} h^{-1}). Nitrite was a small component of DIN with a flux of ≤ 1 µmol m^{-2} h^{-1} and always directed into the sediment. The highest nitrate efflux rate across all sites during the sampling period occurred at site UW during November with a mean flux of 17.0 ± 1.2 µmol m^{-2} h^{-1}out of the sediment. Two-way analysis of variance (ANOVA) was used to compare flux rates with time and location as fixed factors. Significant interactions between location x station ($F = 5.982$, df = 2,30; $P = 0.007$) and between location x time ($F = 10.576$, df = 2,30, $P = 0.000$) were found.

Fluxes of silicate were directed out of the sediment at most sites with the exception of sites UM and UE during winter (Figs. 6.2 and 6.3). Two-way ANOVA was used to compare flux rates with time and location as fixed factors. A significant interaction was found between location and time ($F = 4.976$, df = 2,28, $P = 0.014$).

Fluxes of phosphate were directed out of the sediment with mean rates always < 3 µmol m^{-2} h^{-1}. No phosphate fluxes at the lower estuary sites during summer were reported due to erratic changes in concentration over the incubation period leading to non-significant flux results. DON concentrations were small with changes over time generally non-linear, leading to insignificant flux results, so they were not reported.

6.8.3. Porewater profiles

Ammonium concentrations at all sites increased down the profiles to a maximum between 20 and 40 µM at a depth of 4 cm before decreasing again (Fig. 6.4). Calculated rates of ammonium upward fluxes were generally low, with the highest rate recorded at UW (5.7 ± 1.2 µmol m^{-2} h^{-1}) and the lowest rate recorded at Garden Island in April 2005 (1.9 ± 1.2 µmol m^{-2} h^{-1}).

Nitrate concentrations (Fig. 6.5) generally peaked in the top cm of the sediments, within the oxygenated zone for all sites. Nitrate concentrations were highest in the Lower estuary sites with LW recording 12.1 (± 1.5) µM in the top cm. The highest concentration in the Upper estuary sites was 3.5 (± 0.6) µM at UM. Nitrate concentrations in the surface sediments were greater than nitrate concentrations in the water column (see Table 6.2). No nitrite was detected in the porewaters.

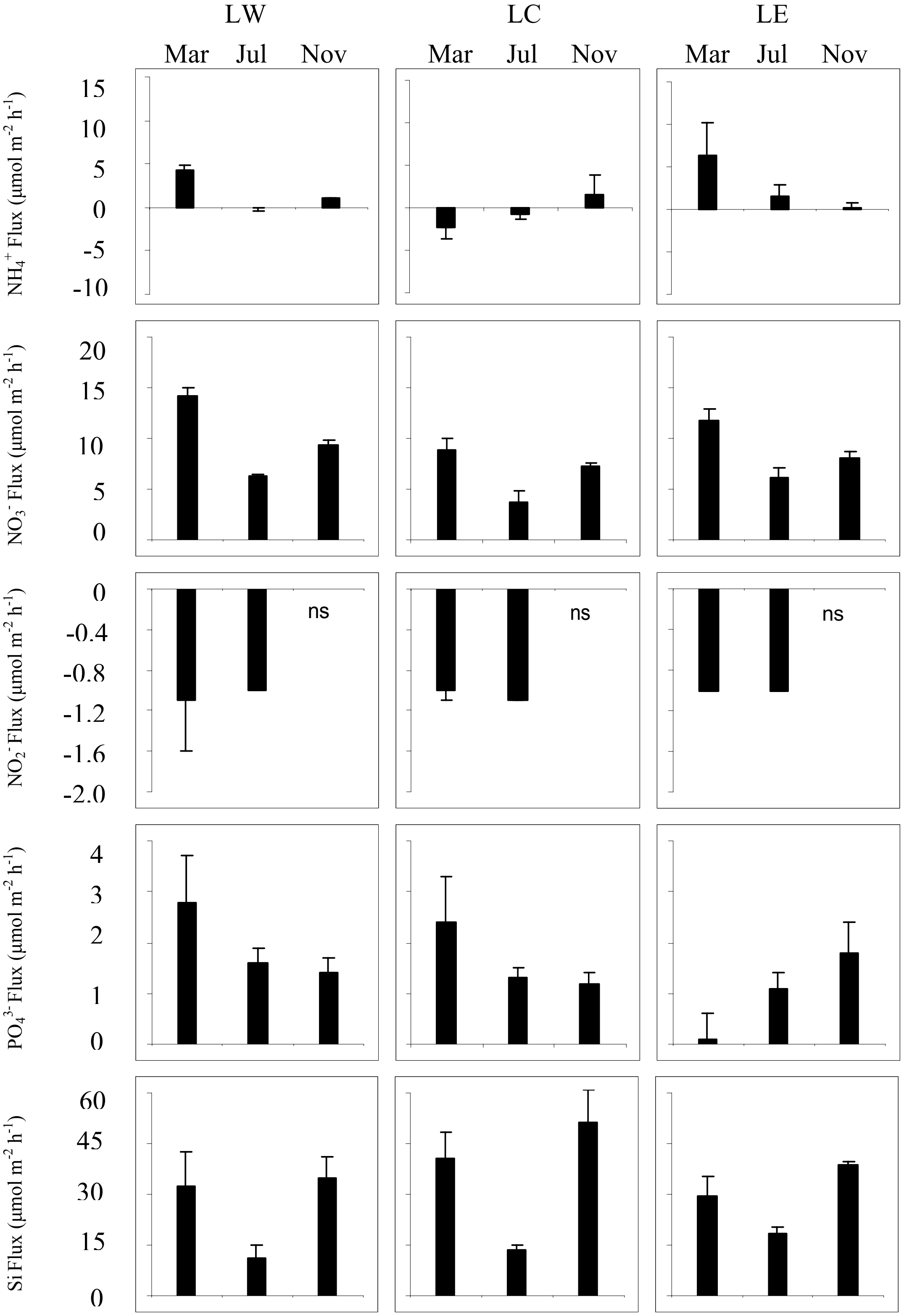


Figure 6.2. Fluxes of NH_4^+, NO_2^-, NO_3^-, $PO_4^{3-}$and Si at the sediment–water interface at the three sample stations during March, July and November at the Lower estuary sites of the Huon Estuary. Error bars represent the standard error of the mean (n = 2 or 3). N.B. where no error bars are present, the datum only represents one flux result. ns = fluxes not significant because of erratic changes over the incubation period.

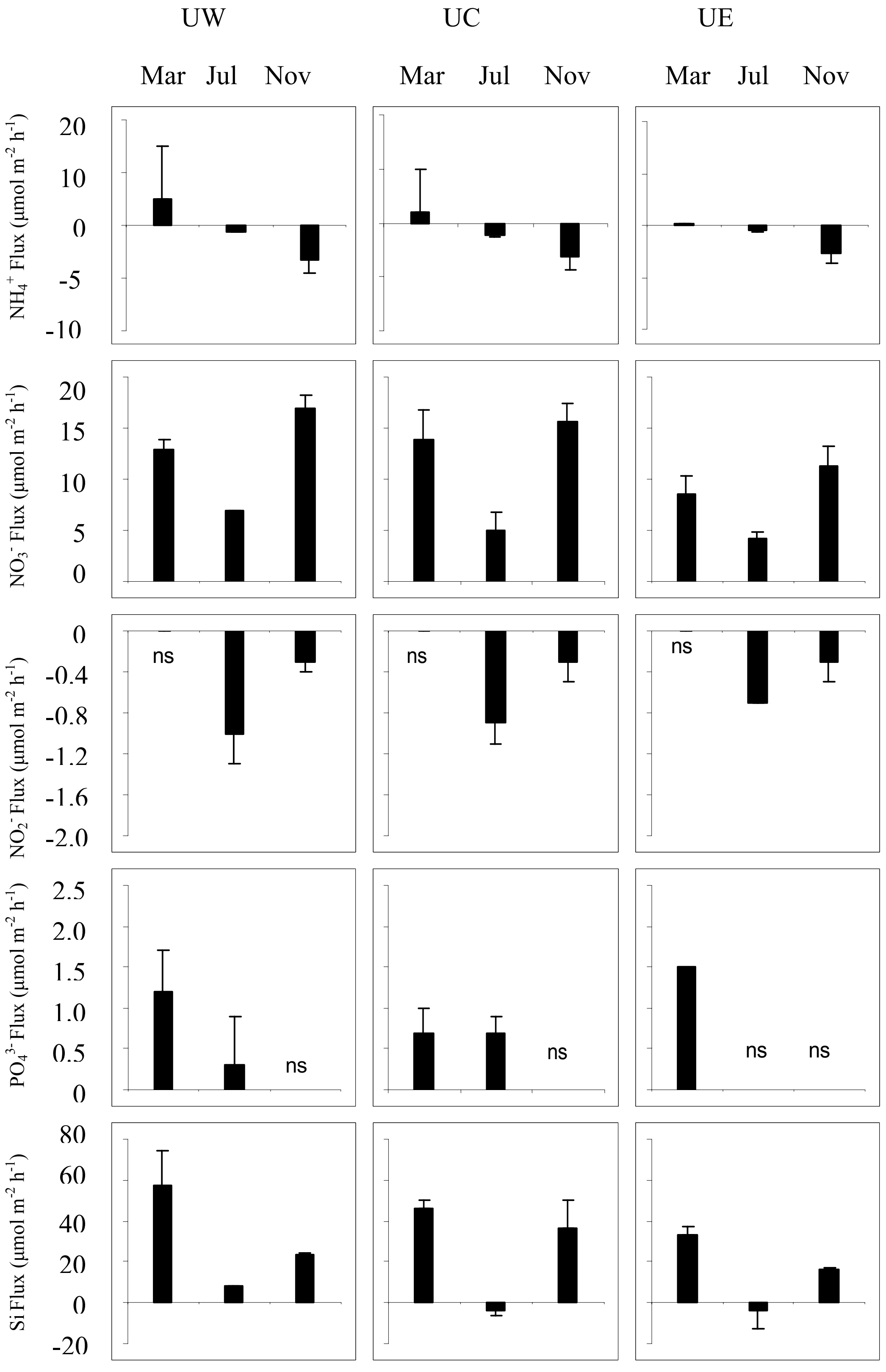
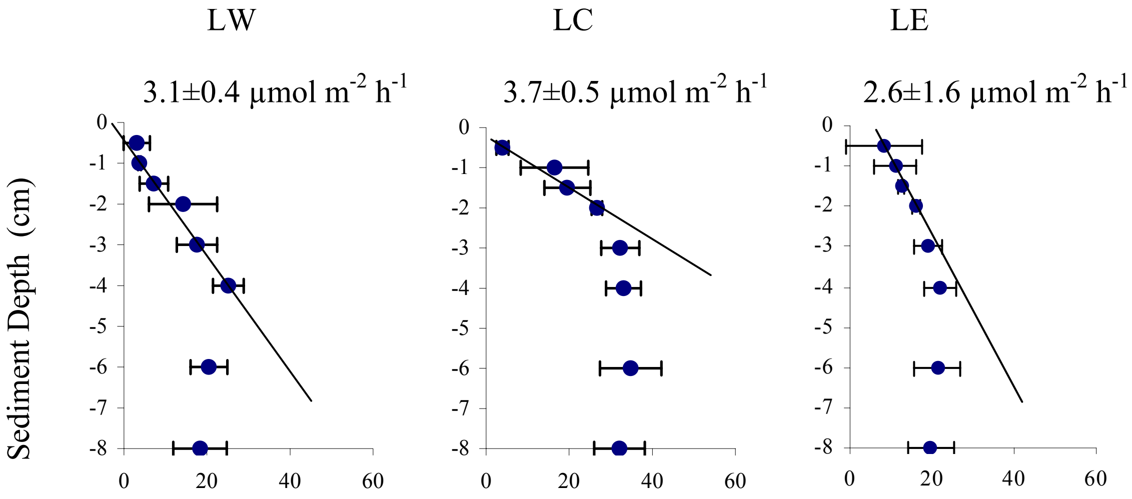
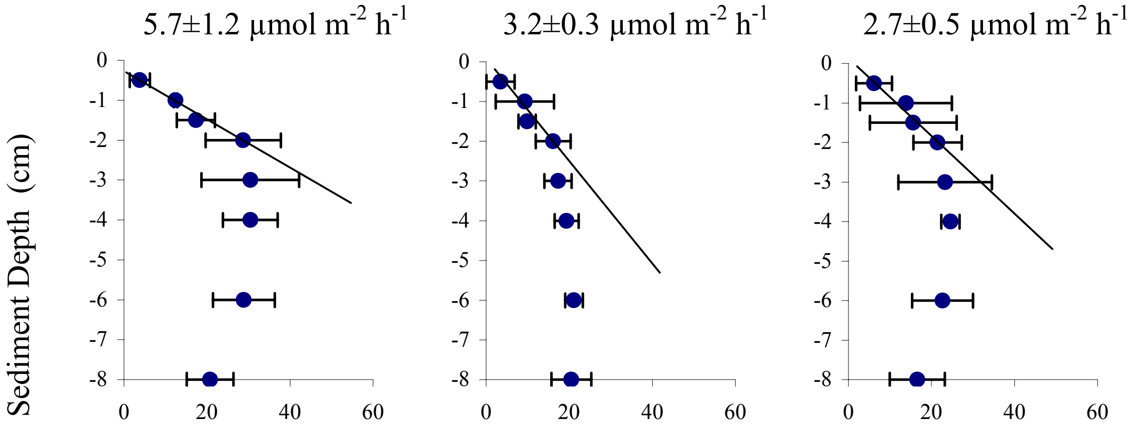


Figure 6.3. Fluxes of NH_4^+, NO_2^-, NO_3^-, $PO_4^{3-}$and Si at the sediment–water interface at the three sample stations during March, July and November at the Upper estuary sites of the Huon Estuary. Error bars represent the standard error of the mean (n = 2 or 3). N.B. where no error bars are present, the datum only represents one flux result. N.S = fluxes not significant because of erratic changes over the incubation period.

(a) Lower estuary sites



(b) Upper estuary sites



(c) April 2005 Sampling

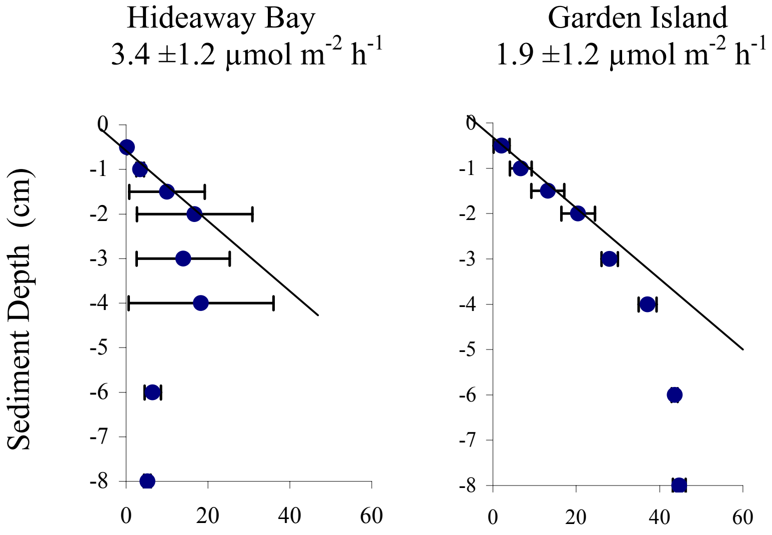
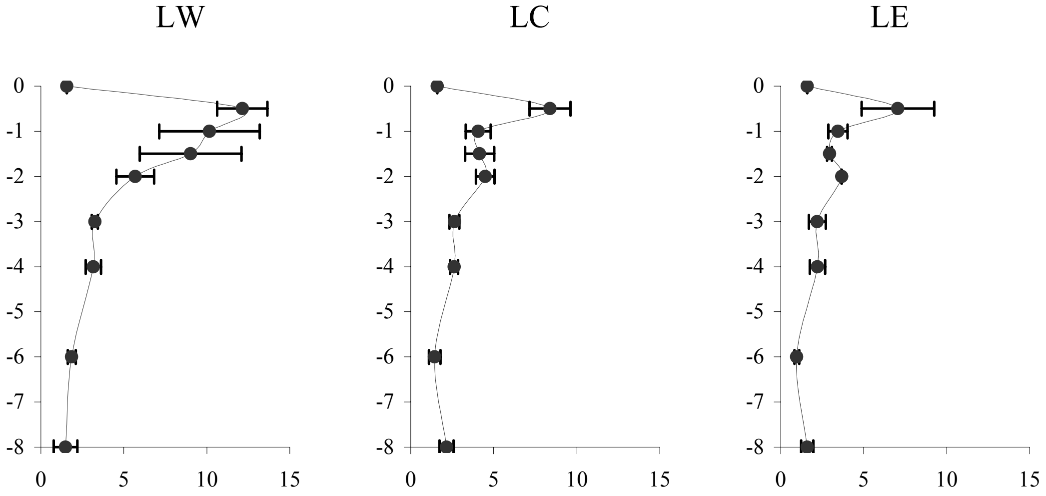
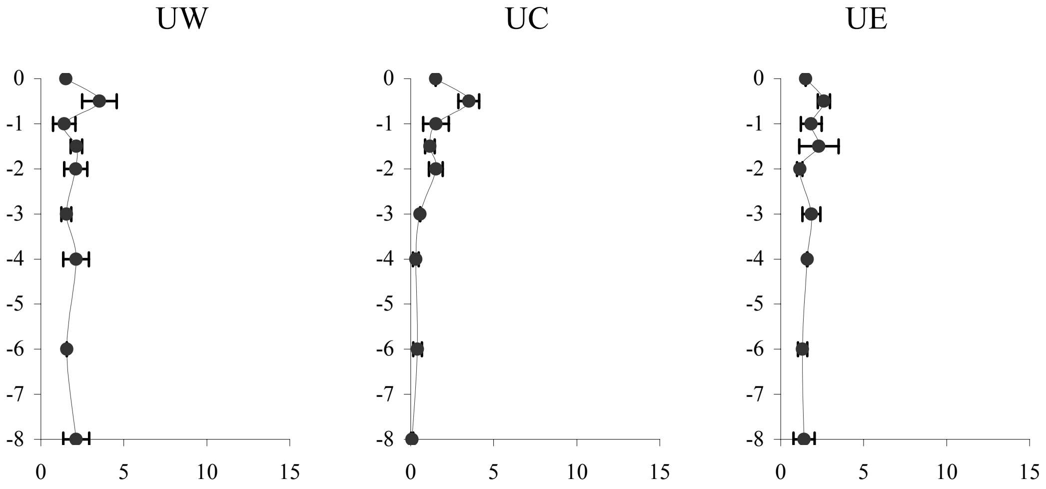


Figure 6.4. Porewater profiles of NH_4^+ and regression lines used to calculate net upward fluxes of NH_4^+ via diffusion and enhanced diffusion at (a) Lower estuary sites and (b) Upper estuary sites in November 2004 and (c) Garden Island and Hideaway Bay during sampling in April 2005 (Error bars = standard deviation of mean (n = 2). Regression was chosen as the best fitting line according to r^2.

a) Lower estuary stations



b) Upper estuary stations



(c) April 2005 Sampling

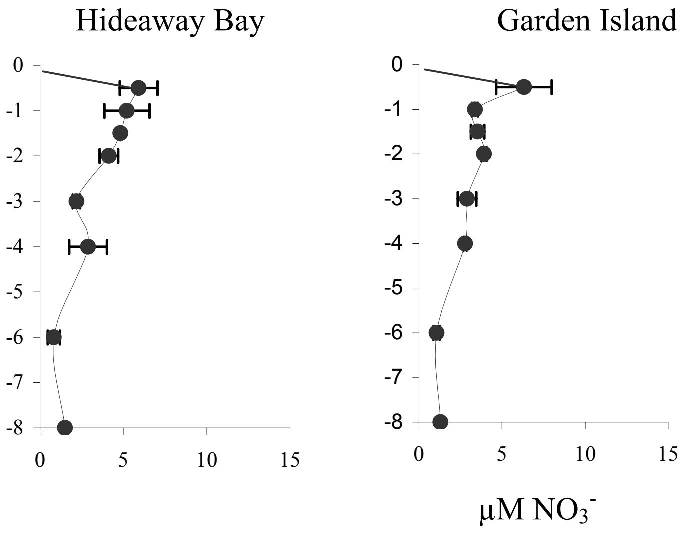


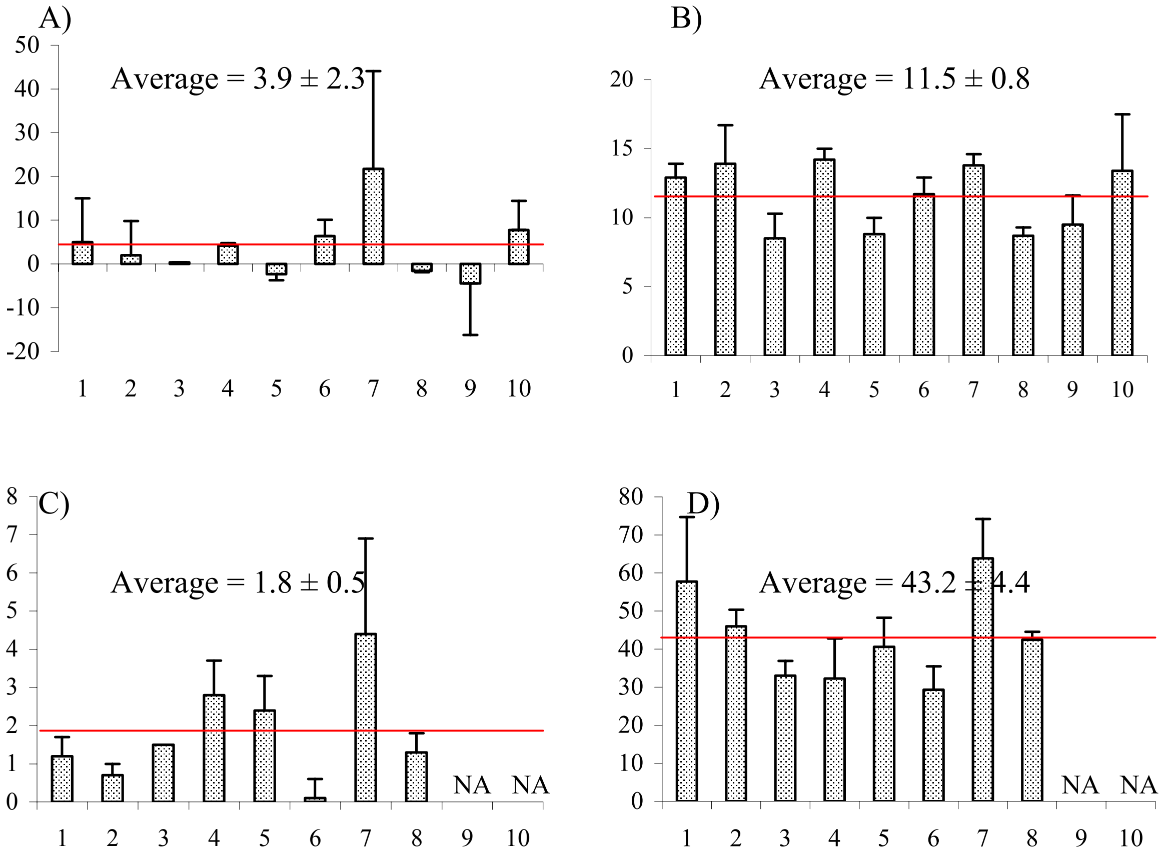
Figure 6.5. Porewater profiles of NO_3^- at (a) Lower estuary sites and (b) upper estuary sites in November and (c) Garden Island and Hideaway Bay during sampling in April 2005 (Error bars = standard deviation of mean (n = 2)

6.8.4. Spatial variability of nutrient fluxes

Benthic nutrient fluxes often have large spatial variability, which may occur due to the presence or absence of benthic fauna which irrigate or bioturbate the sediments (Caffrey, 1995; Kristensen, 2000; Glud et al., 2003), the quantity and source of organic matter in the sediments (Forja et al., 1994; Hall et al., 1996), the sediment texture (Caffrey, 1995) and the presence of benthic algae (Sundbäck et al., 1991; Caffrey, 1995; Magalhaes et al., 2002; Cook et al., 2004). All these factors can have a significant effect on nutrient fluxes across the sediment-water interface. In this study, spatial variability was examined across a number of scales including sediment cores <10 m apart (cores within a station), cores 0.5–2 km apart (between stations in the same location) and cores >20 km apart (between the two locations).

Small-scale nutrient flux variability was examined by calculating the coefficient of variation (SD/mean) as a measure of precision between replicate cores at the same sampling station. Ammonium fluxes exhibited considerable spatial heterogeneity in values, but all other fluxes were, with one or two exceptions, less variable in time and space. Ammonium fluxes were highly variable both between cores within a station and between cores within a location: some cores released ammonium to the overlying water column while other cores at the same station or location consumed ammonium from the water column. Possible causes include the presence or absence of fauna in the cores during incubation, the variable nature of ammonium retention in the sediments due to adsorption onto sediment particles, ammonium uptake by bacteria, nitrification and the presence of 'hot spots' of organic carbon on the sediment surface. Also the relatively small core diameter means that any very small cm-scale spatial variability is not averaged out. Site 7 seems to be an outlier with respect to both ammonium and silicate fluxes (Fig. 6.6)

Broader spatial variability within the Huon Estuary and surrounding bays was also examined by including data from a separate study conducted in 2005 from Hideaway bay and Garden island and from Stringers Cove during a 2001–2 study. Nitrate and silicate fluxes had relatively high precision, with coefficients of variation of 21% and 29% respectively, in comparison, ammonium and phosphate fluxes were highly variable with coefficients of variation equal to 189% and 76% respectively. The presence of benthic fauna does not explain this variability as no significant correlation was found between macrofauna abundance and ammonium fluxes. Also, nitrate and silicate fluxes should also show a greater variability than we observed if there was significant bioturbation. Benthic algae can also be ruled out as a source of variability as no photosynthetic activity occurred at the surface of the sediments. Also the Huon Estuary has extremely limited light penetration due to highly coloured waters and this severely inhibits benthic algal activity.



Number	Location	Station	Sampling date	Temperature
1	Upper Huon Estuary	UE	06/03/2003	15.6
2	Upper Huon Estuary	UC	06/03/2003	15.6
3	Upper Huon Estuary	UW	06/03/2003	15.6
4	Lower Huon Estuary	LE	01/03/2003	15.7
5	Lower Huon Estuary	LC	01/03/2003	15.7
6	Lower Huon Estuary	LW	01/03/2003	15.7
7	Hideaway Bay	P3	12/04/2005	15.7
8	Garden Island	P4	12/04/2005	15.7
9	Stringers Cove (Macleod et al., 2004)	R2	22/02/2001	~16.0
10	Stringers Cove (Macleod et al., 2004)	R2	18/02/2002	~16.0

Figure 6.6. Spatial patterns of benthic nutrient fluxes: A) Ammonium, B) Nitrate, C) Phosphate, D) Silicate during late summer/early autumn. Temperatures ranged between 15–16 °C for all sampling periods (n.d.= no data). Redline indicates average flux. Fluxes = μmol m^{-2} h^{-1}. Error bars reflect the standard error of the mean.

The high variability in phosphate and ammonium fluxes could be explained by the removal of phosphate (Hall et al., 1996) and ammonium (Mackin and Aller, 1984; Blackburn, 1986) by adsorption in oxidised sediments, in particular by clay (Hall et al., 1996). Ammonium may also be removed by nitrification (Blackburn and Henriksen, 1983; Nedwell et al., 1993) or incorporated into bacterial biomass through assimilation during the decomposition of organic matter (Blackburn, 1986; van Duyl et al., 1993; Lomstein et al., 1998). Oxygen concentration can also play a major regulatory role in benthic phosphate release (Sundby et al., 1986) and in nitrification. However, the Huon Estuary bottom water is generally well oxygenated throughout the year (Butler et al., 2000) as are the surficial sediments. During this study, oxygen penetration depths of 3–9 mm were observed, comparable to depths obtained by Macleod et al. (2004) who reported a range of 4.7–7.1 mm for the oxic zone in unimpacted sites near the Huon Estuary. All this would suggest that dissolved oxygen was not sufficiently variable to account for the observed changes in ammonium or phosphate fluxes.

6.8.5. Temporal variability of nutrient fluxes

A change in water temperature can alter flux rates by causing a change in the rate of microbially mediated organic matter remineralisation (Klump and Martens, 1989). Several studies have found that temperature has a significant effect on nutrient fluxes including studies undertaken in Port Phillip Bay (Nicholson and Longmore, 1999), in the Bay of Cadiz (Forja et al., 1994) and in Alfacs Bay, Ebro delta (Vidal et al., 1997). In addition to temperature effects, temporal variability of nutrient fluxes may also be attributed to the annual supply and source of organic matter (Caffrey, 1995). Previous findings by a number of researchers have shown that spring phytoplankton blooms can often be associated with increased nutrient flux rates (Hammond et al., 1985; Caffrey, 1995; Berelson et al., 2003).

At the lower estuary stations, ammonium fluxes were generally directed out of the sediment in both March and November, with highest effluxes in March, whilst ammonium effluxes were generally directed into the sediments in July. Silicate fluxes however were highest in November and lowest in July. In contrast, fluxes of ammonium at the upper stations were directed out of the sediments only in March, and were directed into the sediments during both July and November, with the highest uptake rates observed in November. In November at the upper location stations, the relatively high uptake rates of ammonium coincided with the largest nitrate fluxes observed during the study. The relatively large nitrate effluxes and uptake of ammonium by the sediments suggests that nitrification may have been an important process at this time (November). Potentially nitrifiers may have being limited by ammonium production in the sediments and therefore obtained the rest of their ammonium requirements from the water column. Interestingly, the high ammonium uptake rate relative to the rest of the ammonium fluxes observed during this study, occurred when ammonium concentrations in the water column were highest (3 μM). This may suggest that sediments have the potential to act as net sinks of ammonium when concentrations in the water column are elevated and therefore may provide an important regulatory mechanism for ammonium levels in the estuary.

The temporal trends in the data were further explored by running correlations between temperature and nutrient fluxes (Table 6.3) on both the lower and upper stations. Significant correlations were found between temperature and nitrate and DIN at the lower stations and for nitrate, DIN and silicate at the upper stations and therefore may explain most of the difference between fluxes across the different study times. Temperature however could not explain differences in ammonium and phosphate fluxes with no significant correlation between these two variables and temperature found (Table 6.2).

In addition to temperature, correlations were also run between organic carbon contents in the surface sediments and nutrient fluxes. No significant correlations were found between sediment organic carbon contents and nutrient fluxes and therefore temporal variability of the nutrient fluxes observed during this study could not be attributed to the organic content of the sediments.

Table 6.3. Correlations between temperature and nutrient fluxes at both the Lower and Upper locations

Location	**Variable**	r^2	**P value**
Lower	Nitrate	0.85	Significant at 0.01 level (2-tailed)
	Ammonium	0.42	Not Significant
	DIN	0.75	Significant at 0.05 level (2-tailed)
	Silicate	0.65	Not Significant
	Phosphate	0.29	Not Significant
Upper	Nitrate	0.68	Significant at 0.05 level (2-tailed)
	Ammonium	0.36	Not Significant
	DIN	0.93	Significant at 0.01 level
	Silicate	0.91	Significant at 0.01 level (2-tailed)
	Phosphate	0.48	Not Significant

6.9. Nutrient Cycling

6.9.1. Nitrogen

The ratio of measured fluxes TCO_2:DIN can be used as an indicator of DIN reactions within sediments assuming that organic material undergoing degradation in the sediments conforms to Redfield proportions (Berelson et al., 2003). If it does, then it is expected that resultant fluxes of carbon and nitrogen would have a ratio approximating 6.6C:1N (Berelson et al., 2003; Giblin et al., 1997). During this study, the ratio of TCO_2 (used as a proxy for oxidised organic carbon) to DIN flux strongly deviated from the expected. The measured ratios were always above 6.6 and had a range between 14.4 and 724.7 (average 45.4) at all stations, including the samples taken at Hideaway Bay and Garden Island (Fig. 6.7a).

The higher than expected ratios suggest that a large portion of DIN released during organic matter decomposition is retained in the sediments. Assuming TCO_2 flux reflects remineralisation of Redfield material, it was calculated that 54–99% (average 85%) of the nitrogen remineralised in sediments at all stations was not released to the overlying water as DIN.

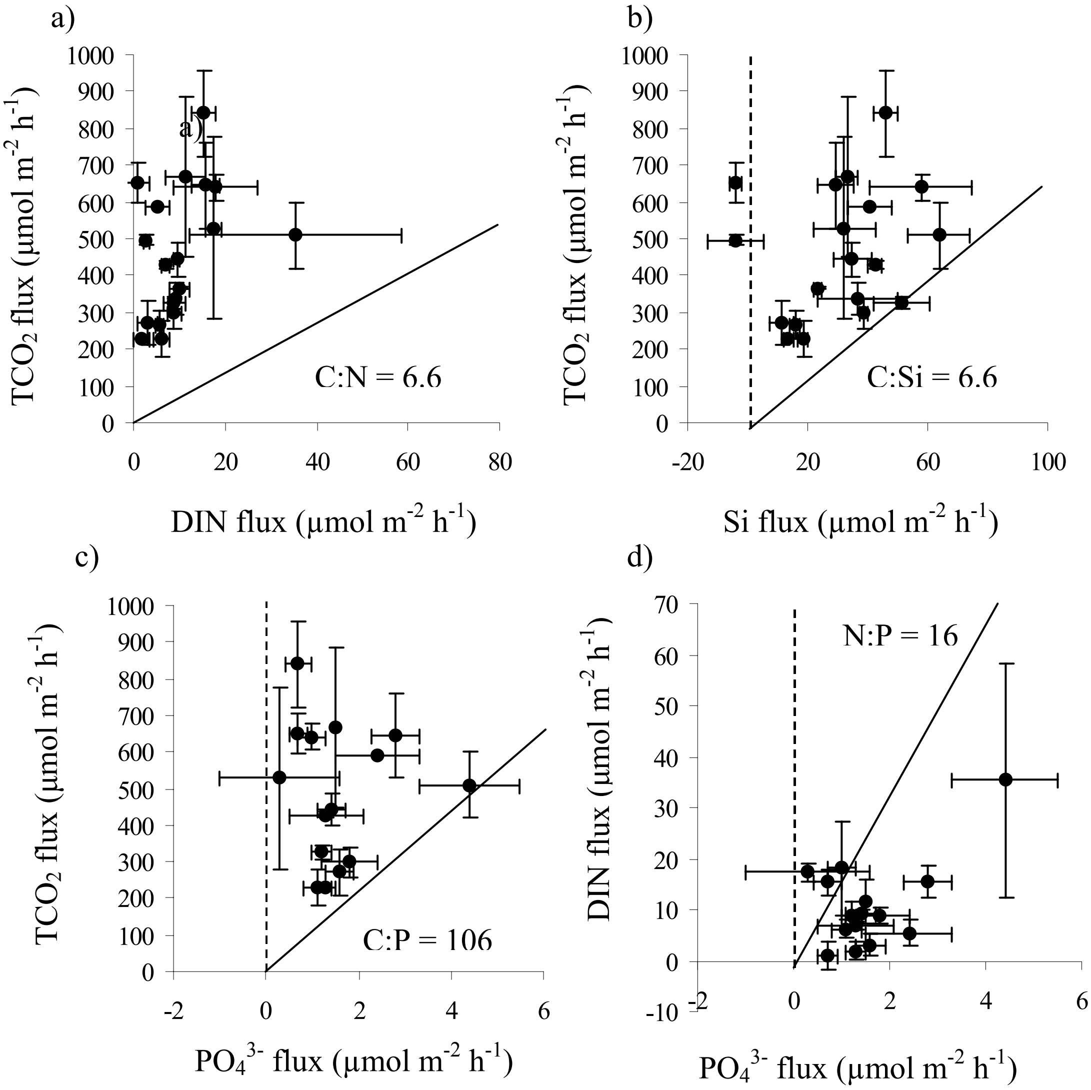


Figure 6.7. Nutrient flux stoichiometry. Line represents the Redfield ratio. Error bars = SE of mean (n = 2 – 3).

One possible mechanism that may account for this ‘missing’ nitrogen is the conversion of ammonium to nitrate by nitrification and then nitrate conversion to di-nitrogen gas (N_2) by denitrification (Hulth et al., 2005). On all occasions during this study, the DIN flux was dominated by nitrate, which was always released from the sediment to the overlying water. Nitrite fluxes were always taken up by the sediment from the overlying water, while ammonium fluxes were erratic, but almost always smaller than nitrate effluxes. Aerobic conditions were prevalent in the upper 3–9 mm of the sediment during this study providing an environment conducive for aerobic processes like nitrification. Peaks of nitrate within the aerobic zone observed from nitrate pore water profiles (Fig. 6.5) also provided evidence that nitrate is produced in the oxic zone. All these observations suggest that the effluxes of nitrate and influxes of nitrite and ammonium in Huon Estuary sediments are associated with intensive nitrification (Hall et al., 1996), stimulated by the presence of relatively deep oxygenated zones.

Numerous studies have found that a large portion of nitrate produced in sediments is denitrified resulting in the release of N_2 into the overlying water column. Seitzinger (1987) found that 80% of the nitrate produced in Ochlockonee Bay was denitrified. Jenkins and Kemp (1984) observed tight coupling between nitrification and denitrification in the Patuxent River where over 99% of nitrate produced in sediments was reduced to N_2 while Henriksen et al. (1981) found approximately 50% of the nitrate produced was reduced to N_2. Macleod et al. (2004) observed that denitrification accounted for 39 to 60% of total nitrogen fluxes in Stringers Cove, which is adjacent to the Huon Estuary and has similar sediment geochemistry.

Direct denitrification measurements were not made during this study, but estimates of denitrification rates (DR) can be made using the following equation (Geoscience Australia, 2007):

$$DR = TDIN_p - DIN_m$$

Where $TDIN_m$ is the measured dissolved inorganic nitrogen flux and $TDIN_p$ is the total dissolved inorganic nitrogen flux predicted on the basis of stoichiometric relationship with benthic carbon dioxide flux. This is calculated by the following equation (Geoscience Australia, 2007):

$$TDIN_p = C_{ox} \text{ x } (N:C)$$

Where C_{ox} is benthic carbon dioxide flux and N:C is the stoichiometric relationship of the decomposing organic matter (Redfield N:C ratio was used in these calculations).

The denitrification efficiency (DE) of the sediments can also be calculated as (Geoscience Australia, 2007):

$$DE\% = (TDIN_p - DIN_m) \text{ x } 100/\ TDIN_p$$

Fig. 6.8 shows calculated denitrification rates and efficiencies during the current study. Denitrification rates were estimated to be highest in March at the upper location and highest in November at the lower location. Denitrification efficiencies were also estimated to be high, between 80 and 98%, and were most efficient during July when nitrate concentrations were elevated in the water column. These estimates of denitrification imply that most of the DIN is ultimately converted to N_2.

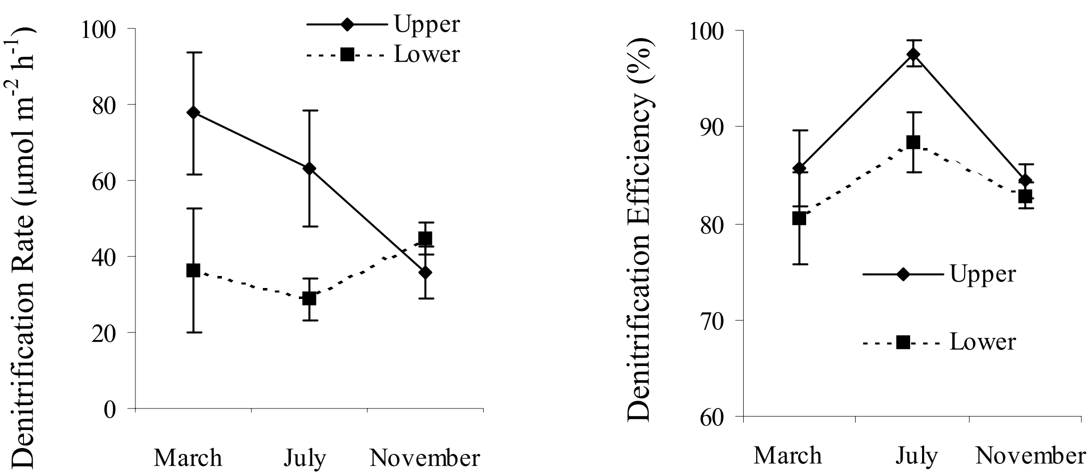


Figure 6.8. Calculated denitrification rates and denitrification efficiencies during the current study at the upper and lower locations.

Support for the case for nitrification-denitrification occurring in Huon Estuary sediments is given by the porewater profiles of oxygen and NO_3^- (Fig. 6.5). Nitrate was produced in the oxic zone of the sediments indicating the presence of nitrification. Furthermore, nitrate penetrates into the suboxic zone of the sediments where it is consumed, presumably by the denitrifiers. This supports the classical view of coupled nitrification-denitrification where nitrification takes place in the upper oxic zone of the sediments and denitrification in the suboxic zone just below the oxic/suboxic interface (Hulth et al., 2005).

However an alternative pathway to this classical view is that of anaerobic ammonium oxidation (ANAMMOX) which bypasses the aerobic nitrification phase. ANAMMOX is the oxidation of ammonium coupled to the reduction of NO_2^- which results in the production of N_2 (Trimmer et al., 2003a,b). Anaerobic ammonium oxidation has recently been reported to account for as much as 24 and 60% of N_2 formation in continental shelf sediments, but values as low as 2% have been reported for eutrophic shallow coastal bay sediments (Thamdrup and Dalsgaard, 2002).

The stoichiometric approach used in the study however has some limitations as it assumes that the missing nitrogen from the DIN has been denitrified, when it may be assimilated by bacteria (Lomstein et al., 1998) or benthic algae in the sediment. However, the latter can probably be ruled out as no benthic algal activity was detected in the sediments. Another assumption made using the stoichiometric approach is that the organic matter being remineralised conforms to the Redfield ratio. However, if the C:N ratio is significantly higher, then this assumption will overestimate the predicted DIN release and therefore the denitrification rate.

The calculated denitrification rates in this study are high compared with results from a study by Macleod et al. (2004) in which observed mean denitrification rates ranged between 8–19 µmol m^{-2} h^{-1} in unimpacted sediments at Stringers Cove adjacent to the Huon Estuary. The surface sediments in Stringers Cove have similar organic matter content, oxygen penetration depth and DIN, nitrate and ammonium fluxes to those measured in the Huon Estuary here. Therefore the calculated denitrification rates in this study most likely overestimate actual rates of denitrification and therefore future work should incorporate direct measurements of denitrification.

DON fluxes may also account for some of the missing nitrogen, however they are unlikely to be significant. DON fluxes are only significant when there is a high deposition of fresh phytoplankton (Cook et al., 2004 and references therein; Sloth et al., 1995). During this study, no algal blooms were apparent during sampling periods and results from lipid biomarkers suggested that phytoplankton accounted for only a small proportion of organic matter in the sediments. DON fluxes in this study were considered to be insignificant because changes in DON concentrations over time were small and non-linear. This leads to the conclusion that DON fluxes are not likely to be a significant source of the missing nitrogen flux in this case.

6.9.2. Phosphate

TCO_2 to PO_4^{3-} flux ratios (Fig. 6.7d) indicate strong phosphate retention is occurring in sediments at all sites throughout this study. The flux ratios ranged between 116–1762, which is higher than predicted by the 106:1 Redfield ratio. A possible reason for this is that the efflux of remineralised phosphate is controlled in the surface oxic layer by sorption to iron oxyhydroxides (Hopkinson et al., 2001) and is only released when iron (III) is reduced to iron (II) (Butler et al., 2000). The oxic nature of the sediments in this study suggests that most of the iron is in the form of Fe^{3+} and therefore binding phosphate to sediment particles.

PO_4^{3-} was also compared to DIN with the flux ratio (16:1) expected from material with Redfield composition undergoing degradation. The majority of DIN:PO_4^{3-} flux ratios (Fig. 6.7d) fell below 16:1, suggesting the majority of nitrogen was either retained in the sediments or exported as N_2 to the overlying water.

6.9.3. Silicate

The TCO_2:Si flux ratio (Fig. 6.7b) ranged between 6.3 and 24.3 for all sites over the sampling period, with an average ratio of 14.3. Some cores had a ratio close to the expected ratio of 6.6 for Redfield material (Berelson et al., 2003) indicating that the organic matter undergoing decomposition was largely phytoplanktonic, in particular cores LC and LE measured in November. However, only a weak positive correlation ($r^2 = 0.39$) existed between TCO_2 production and silicate fluxes, suggesting that the sources of organic carbon are variable in their silicate content.

6.10. Comparison with other estuaries

It is often difficult to compare flux rates between different studies as different techniques were used. However, one way to compare the importance of nutrient fluxes between environments is to compare the time it would take for the sediments to replace the standing stock of nutrients in the water column. Using a similar method to that used by Hammond et al. (1985) and Friedl et al. (1998) the standing nutrient stock in the water column was divided by the average temporal flux for different solutes at both locations (Table 6.4). In other words, the nutrient (e.g. silicate) concentration was multiplied by the depth of the water column and then divided by the daily nutrient flux. The turnover rates or replacement time in this study generally suggest that the water column nutrient concentrations are likely to be dominated by processes other then from benthic regeneration.

The turnover rates of nutrients showed a distinct temporal trend with replacement time shortest in March and November when relatively high benthic recycling is occurring and water column nutrient concentrations are low. Comparatively, July had the longest replacement times, when benthic recycling is slow due to lower water temperatures and water column nutrient concentrations are relatively high.

Table 6.4. Replacement time (number of days) of water column nutrients by benthic fluxes. Water column nutrients and depth are taken from Table 6.2. Benthic fluxes are taken from Figs. 6.2 and 6.3.

	NH_4^+	NO_3^-	PO_4^{3-}	Si
Lower Estuary				
Average depth = 30 m				
March	432	76	347	140
July	625	1156	561	777
November	400	244	354	159
Upper Estuary				
Average depth = 15 m				
March	254	31	191	20
July	NA	633	2778	NA
November	NA	64	NA	143

The turnover rates for nutrients in the Huon Estuary are considerably longer compared with other coastal systems. Hammond et al. (1985) found the replacement time for a deepwater station in San Francisco Bay was between 20 to 80 days for ammonia, an order of magnitude lower than this study. In the Potomac River Estuary, benthic ammonium fluxes were sufficient to replace ammonium water concentrations three times a day (Callender and Hammond, 1982). The replacement times for silicate and phosphorus were 3 and 26 days respectively (Callender and Hammond, 1982), while replacement times in San Francisco Bay for silicate and phosphorus were 150 to 300 days and 250 to 1000 days respectively (Hammond et al., 1985). Contrastingly, in this study we observed replacement times between 191 and 2778 days for phosphate and 20 to 777 days for silicate, all of which indicates that the sediments are not currently a major source of nutrients in the Huon Estuary.

The relationship between bacterial respiration and ammonia release can be illustrated by the relationship between sediment oxygen consumption (used as proxy for bacterial respiration) and ammonium release. As sediment oxygen consumption increases so does ammonium release. Presumably this occurs due to the depletion of the sediment oxic layer, and inhibition of nitrification. Furthermore, dissimilatory nitrate reduction to ammonium (DNRA) may also contribute significantly to the ammonium flux (at the expense of nitrate) in sediments that become organically enriched. Christensen et al. (2000) found that DNRA increased 7-fold in sediments underneath fish cages in comparison to unimpacted sediments. Fig. 6.9 shows the relationship between ammonium and oxygen fluxes across a number of near-shore environments. This figure demonstrates that ammonium regeneration occurs at much higher rates in eutrophic systems than in systems like the Huon Estuary.

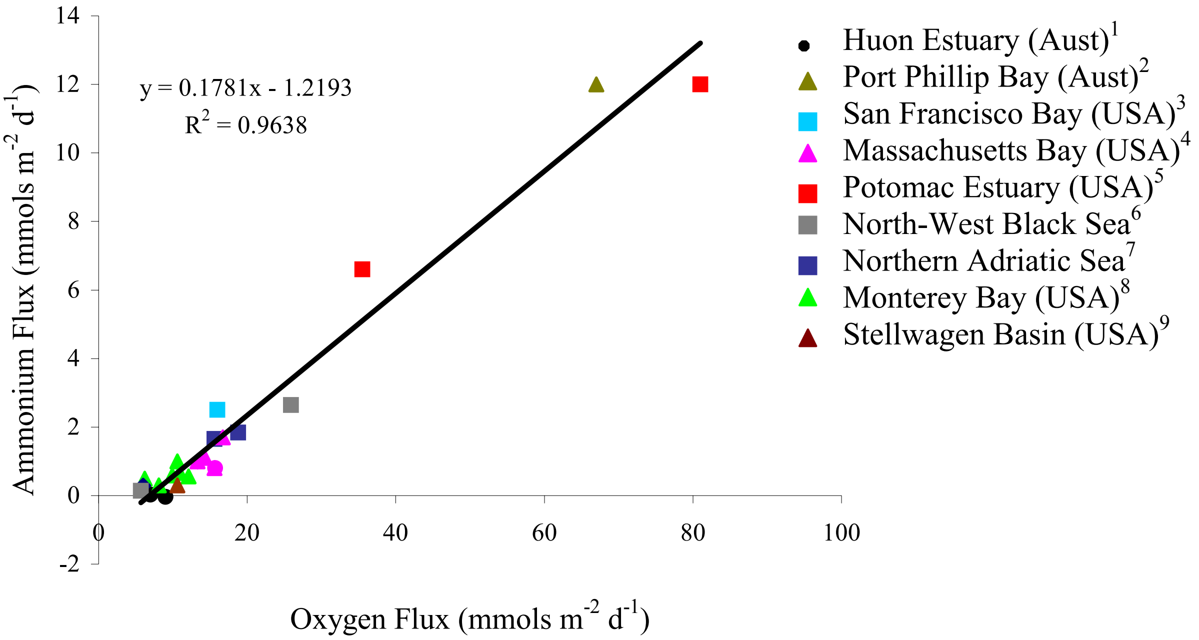


Figure 6.9. Relationship between ammonium fluxes and oxygen consumption in near-shore environments. Square boxes represent eutrophic environments. Numbers in superscript refer to the following references: 1: this study; 2: Berelson et al. (1998); 3: Hammond et al. (1985); 4: Hopkinson et al. (2001); 5: Callender and Hammond (1982); 6: Friedl et al. (1998); 7: Hammond et al. (1999); 8: Berelson et al. (2003); 9: Hopkinson et al. (2001).

To determine the significance of the sediments as a source of nutrients, we have compared annual nutrient inputs from sediments with inputs from other sources calculated in the HES. All the data for each nutrient from the 3 sampling trips in 2004 were pooled and an average annual flux calculated. The average annual flux was then multiplied by the surface area of the estuary floor (77.4 km^2), excluding the 5% associated with intertidal areas, to get an estimate of estuary-wide nutrient inputs from sub-tidal sediments. It is noted that the extrapolated Huon Estuary inputs from sediments are only approximate as they are based on only a few laboratory measurements. However, sediment biogeochemistry results in the HES suggest that apart from small regions of sand, mud (i.e. sediment grain size < 63 μm) was the dominant component over almost the entire estuary floor in its middle and lower reaches. Therefore, the extrapolation of the results from this study to the whole estuary is not unreasonable for a first estimation of the importance of nutrients from benthic regeneration.

This calculation shows that the sediments are only minor sources of both nitrogen and phosphate compared to other sources, which supports argument that sediments have low respiration rates and potentially significant denitrification and/or ANAMMOX rates. The major source of nitrogen to the Huon Estuary comes from the marine system (Butler et al., 2000), which is approximately 8 times higher than the Huon River, the next largest source (Table 6.5). Marine waters are also the major source of phosphate to the estuary, about 9 times higher then the Huon River, the next most prominent source. The sediments provide about 32 tonnes of inorganic phosphate the third most important source, but approximately 2 orders of magnitude below marine sources. Overall the sediments provide nutrients with an N:P ratio of 3, well below “Redfield ratio” of 16:1, the ratio that phytoplankton assimilate N and P. This means that phytoplankton must obtain nitrogen from sources other than from the sediments to meet their nutritional requirements.

Table 6.5. Annual loadings of nutrients to the Huon Estuary (tonnes)

Source	TDN	NO_3	NH_4	TDP	Reference
Huon River	966	51	23	142	Wild-Allen et al. (2004)
Kermandie River	32	9	8	1	Wild-Allen et al. (2004)
STP outfalls	28	-	-	9	Butler et al. (2000)
Fish farms	268	-	-	25	Wild-Allen et al. (2004)
Atmospheric	18	-	-	-	Butler et al. (2000)
Marine	7654			1272	Butler et al. (2000)
Sediments	96	91	12	32	Current study

6.11. Carbon loading experiments

A laboratory experiment was conducted to observe the response of Huon Estuary sediments to increasing loads of labile organic carbon (see Thomson, 2008 for details). The addition of the cyanobacterium *Spirulina* as a source of labile organic matter produced a dramatic increase in the fluxes of all analytes. These fluxes correlated well with increasing carbon loading, but the rate change occurred in two distinct stages for oxygen and ammonium. There was a rapid change in oxygen and ammonium flux rates and oxygen penetration with increasing carbon loading. However when the carbon load became >20.2 g C m^{-2} the change in flux rates and decrease in oxygen penetration slowed significantly for these analytes.

The point at which the rate change of fluxes slowed possibly indicates that the biogeochemical system switched from one that was dominated by aerobic respiration to one dominated by anaerobic respiration. A number of trends in the data indicate this including increasing CO_2:O_2 ratio, increasing alkalinity fluxes out of the sediments, most likely due to sulphate reduction (an anaerobic metabolic process), and declining oxygen penetration depths. Ammonia rather than nitrate became the dominant DIN flux as the sediments became more anaerobic. Nitrate effluxes switched to nitrate fluxes into the sediment, suggesting that denitrifiers could no longer obtain sufficient nitrate *in situ* due to the lessening of nitrification. As the sediments became anaerobic, direct denitrification became dominant, whereby denitrifiers obtain their nitrate requirements from the water column, as evidenced by uptake of nitrate by the sediments. Dissimilatory nitrate reduction to ammonium (DNRA) may also have contributed to the large efflux of ammonium as sediments became more anaerobic.

6.12. Conclusions

Benthic fluxes of nutrients were low in the Huon Estuary. Average effluxes of ammonia, nitrate, phosphate and silicate were 1.3, 10.1, 1.6 and 32.5 µmol m^{-2} h^{-1} respectively. An extrapolation of these measurements to the whole estuary revealed that the sediments were only a minor source of nutrients, providing approximately 96 tonnes of inorganic nitrogen, 32 tonnes of phosphate and 586 tonnes of silicate per annum.

On all occasions, the DIN flux was dominated by nitrate, which was always released from the sediment to the overlying water. Aerobic conditions were prevalent in the upper 3–9 mm for the sediment, providing an environment conducive for aerobic processes like nitrification. Peaks of nitrate within the aerobic zone of the sediment observed from nitrate pore water profiles also provided additional evidence that nitrate was been produced in the oxic zone, most likely due to nitrification. All these observations suggest that the effluxes of nitrate and influxes of nitrite and ammonium in Huon Estuary sediments are associated with intensive nitrification stimulated by the presence of relatively deep oxygenated zones. The net efflux of nitrate from the sediments suggests that they act as net regenerators of nitrogen rather than as nitrogen assimilators.

The benthic effluxes of nitrogen however were smaller than expected from carbon oxidation rates. The low N:P ratio of benthic fluxes (approx. 3:1) indicates that processes such as denitrification may be an important nitrogen elimination process. However another potential pathway for nitrogen released during organic matter remineralisation is for the decomposing bacteria to reassimilate some of the ammonium, the rest being nitrified, due to the low nitrogen content of the organic matter been decomposed.

In systems like the Huon Estuary, DIN fluxes are dominated by nitrate produced by nitrification, which in turn is partially converted to N_2 by denitrification. Phosphate is also adsorbed within the sediments due to the oxic conditions. This contrasts with highly eutrophic systems, where nitrification becomes inhibited due to a reduction in oxygen availability and DIN fluxes become dominated by ammonia. Denitrification efficiency is also reduced. In such systems, phosphate is released at a greater rate due to the reduction in the adsorption capacity of the sediments due to decreased oxic conditions

While this study cannot rule out the potential for sediments to provide ammonium to the water column during summer, as no field campaign was conducted during a summer algal bloom period, our results indicate that deposited nitrogen is successfully recycled by the nitrification-denitrification pathway and that on many occasions the sediments acted as net sinks of ammonium; or that ammonium is reassimilated into microbial biomass or directly lost as N_2 *via* ANAMMOX. In either case, sources of ammonium and nutrients in general are retained in the sediments or converted to N_2 and therefore are no longer biologically available in the water column.

Overall, the sediments are relatively minor contributors of nutrients to the broader Huon Estuary. The sediments are net contributors of ammonium overall, but showed on many occasions that they can act as sinks for ammonium as well. Furthermore, ammonium inputs from the sediments are small compared with inputs from the Huon River and fish farming. The sediments provide 7 to 8 times more nitrate to the water column than ammonium presumably due to intensive nitrification, however it is still dwarfed by the marine inputs of nitrate. The sediments also provide phosphate to the water column, however once again this is only minor in comparison to sources from the ocean and the Huon River.

Oxygen and CO_2 fluxes, in particular low CO_2:O_2 and alkalinity:O_2 flux ratios, indicated that aerobic respiration was the main pathway for carbon degradation in Huon

Estuary sediments. Modelled oxygen consumption profiles showed that the majority of oxygen was consumed near the surface, most likely due to aerobic heterotrophic bacteria. A small oxygen consumption peak was also observed at the sub-surface anoxic/oxic interface most likely due to sulphide oxidation.

Spatial and temporal differences in respiration were noted and these were related to carbon contents and temperature. Differences also occurred between total versus diffusive oxygen uptake rates, and was likely due to the presence of benthic fauna. Respiration in Huon Estuary sediments compared well with other deep coastal sediments whereby rates of TCO_2 and O_2 fluxes were very similar to Monterey Bay in California and the southern Kattergat in the Baltic Sea

A number of important knowledge gaps have been identified from this study that should be included in any future research undertaken in the Huon Estuary. The next step would be to carry out a set of carefully designed experiments that simultaneously measure bacterial nitrogen assimilation, ANAMMOX, denitrification and nitrification. These experiments should be carried out under both unimpacted and organic-enriched sediments as it is still unknown exactly how benthic metabolism would be affected if farming does greatly increase. Furthermore, how these individual processes (i.e. nitrification, ANAMMOX, denitrification and DNRA) collectively influence nutrient fluxes from the sediment under unimpacted and impacted conditions requires further study.

6.13. References

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Chapter 7

Biogeochemical Modelling

7.1. Biogeochemical model description and calibration

The biogeochemical model simulates the cycling of carbon, nitrogen, and phosphorus through dissolved and particulate organic and inorganic forms in the water column and surface sediments of coastal marine systems (Fig. 7.1). It is dynamically coupled to a high resolution 3D hydrodynamic model 'SHOC' (Herzfeld et al., 2005) and a multilayer sediment model (MECOSED; Margvelashvili, 2003), and incorporated into the CSIRO Environmental Modelling Suite. Dissolved nutrients are advected and diffused throughout the model domain in an identical fashion to temperature and salinity whilst particulate substances sink and are resuspended in the same way as sediment particles.

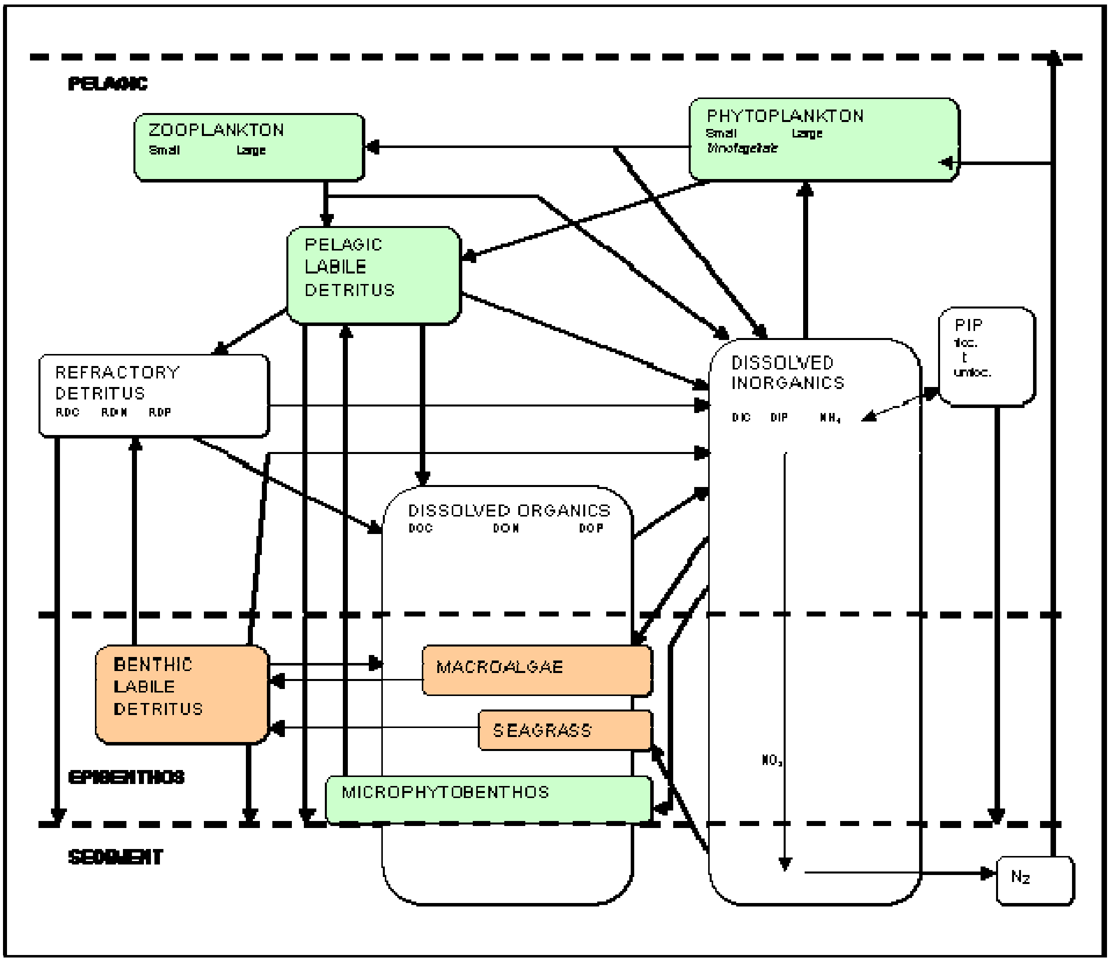


Figure 7.1. Schematic diagram of the biogeochemical model compartments, links and vertical layers. Green compartments have fixed nutrient content at Redfield ratio (106C:16N:1P); brown compartments are fixed at Atkinson ratio (550C:30N:1P).

The biogeochemical model includes four groups of microalgae (small and large phytoplankton, dinoflagellates and microphytobenthos) and two macrophytes (seagrass and macroalgae) which grow at a 24 hour mean rate determined by access to dissolved nutrients (nitrogen and phosphate) and photosynthetically active radiation (PAR) (Baird

1999). Autotrophs take up ammonium and nitrate with equal preference and phosphate and dissolved inorganic carbon are taken up by phytoplankton at Redfield ratio (106C:16N:1P) and by macrophytes at Atkinson ratio (550C:30N:1P). Ambient PAR is calculated from incident surface 24 hour mean PAR attenuated by sea water, coloured dissolved organic substances, organic and inorganic particles. Chlorophyll-*a* concentration is calculated by a fixed nitrogen to chlorophyll ratio (7 mgN/mgChl). Micro- and meso-zooplankton graze on small and large phytoplankton respectively, at rates determined by swimming speed and particle encounter rate. A fraction of grazed material is released as dissolved and particulate carbon, nitrogen and phosphate and further detrital material accumulates through mortality. Detritus and dissolved organic substances are remineralised into inorganic carbon, nitrogen and phosphate with labile detritus transformed most rapidly, refractory detritus slower and dissolved organic material transformed over the longest timescales. The evolution (by photosynthesis) and utilization (by respiration and remineralisation) of dissolved oxygen is also included in the model and depending on prevailing concentrations, facilitates the oxidation of ammonia to nitrate its subsequent denitrification to nitrogen gas which is then lost from the system.

The model was initialized in December 2001 with tracer concentrations and boundary fluxes derived from observations made throughout the region or historical data (Wild-Allen et al., 2005). Flow and nutrient loads were estimated for the Huon, Esperance, Kermandie and North West Bay Rivulet based on observations (Fig.7.2). Sewerage and wastewater discharge is generally small and diffuse along the coast and was not included in the model. Finfish farm inputs for 2002 were derived from monthly feed data from 20 salmonid farms in the region assuming that all feed pellets were consumed by the fish and that no overfeeding occurred (Fig.7.2). Of the total feed 5% and 0.8% was discharged by the fish as waste nitrogen and phosphorus respectively in dissolved and particulate forms. Waste was discharged at the farm sites as a point source discharge evenly distributed between 0.5 m and 12 m and diffused spatially throughout the immediate grid cell.

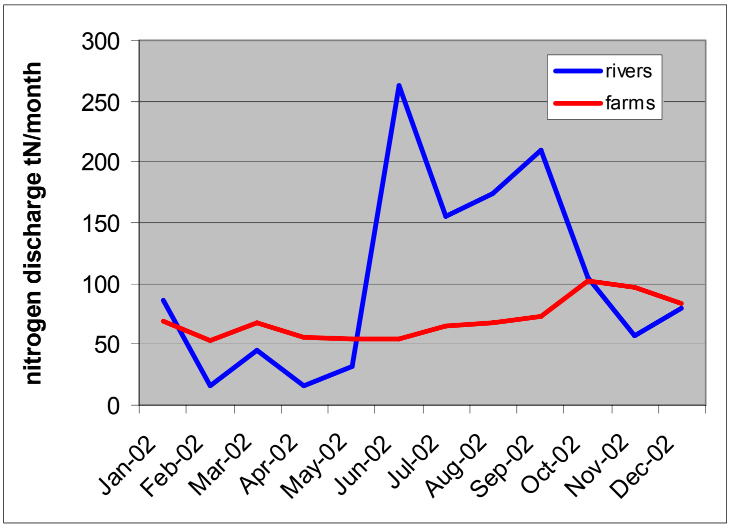


Figure 7.2. Monthly nitrogen input into the model domain for the sum of all farm sites and the sum of the Huon, Kermandie, Esperance and North West Bay rivers in 2002.

The model was calibrated against nutrient, phytoplankton and dissolved oxygen data collected throughout the region during the Broad Scale Monitoring program (Thompson et al., 2004). The model reproduced the observed spatial and temporal dynamics of dissolved nutrients, chlorophyll, phytoplankton biomass and oxygen in the Huon Estuary and D'Entrecasteaux Channel (Fig.7.3). Observed nitrogen, chlorophyll and large phytoplankton biomass concentrations were particularly well reproduced in the D'Entrecasteaux Channel and side bays. The phosphorus cycle, which has a greater number of dissolved and particulate phases, was adequately simulated. Modelled small phytoplankton biomass throughout the region lacked the observed seasonal winter maxima, and dinoflagellate biomass was poorly represented, particularly in the Huon where observed autumn blooms were absent. Modelled oxygen concentrations suggest possible over-estimation of horizontal exchange of bottom waters between D'Entrecasteaux Channel and the mouth of the Huon Estuary.

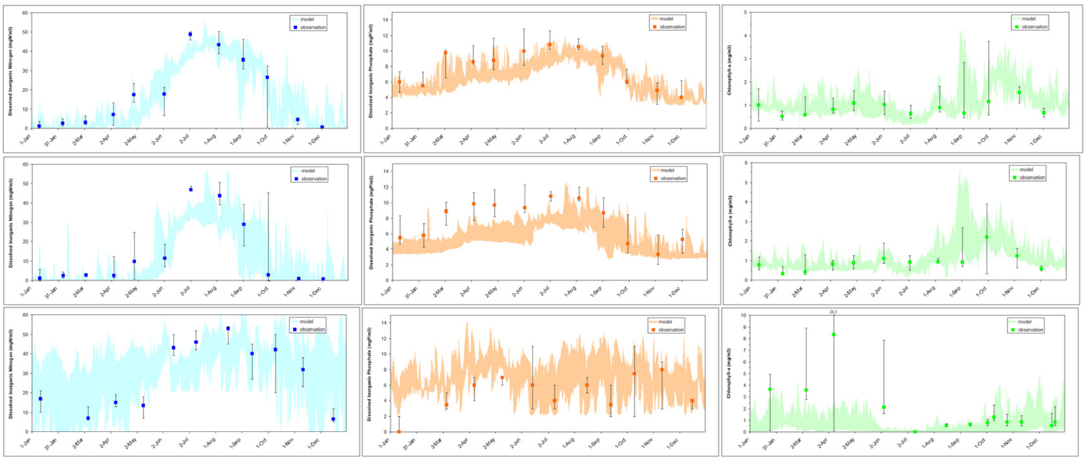


Figure 7.3. Summary calibration plots comparing modelled and observed DIN (left), DIP (centre) and chlorophyll (right) in the D'Entrecasteaux Channel (upper), the D'Entrecasteaux side bays (middle) and the Huon Estuary (lower).

The model in its current formulation is considered to provide an adequate simulation and explanation of the seasonal dynamics of nutrient cycling and phytoplankton abundance in the D'Entrecasteaux Channel and side Bays. However model results in the Huon show poorer agreement with observations, and this is considered to be due to still-unresolved aspects of dinoflagellate dynamics in the estuary.

7.2. Model insights into regional biogeochemistry

The biogeochemical model has provided a description of the biogeochemical dynamics of the Huon Estuary and D'Entrecasteaux Channel with unprecedented spatial and temporal resolution. Model results, supported by observations, show that in 2002 the spring bloom commenced in the northern part of the D'Entrecasteaux Channel and progressed south (Fig. 7.4).

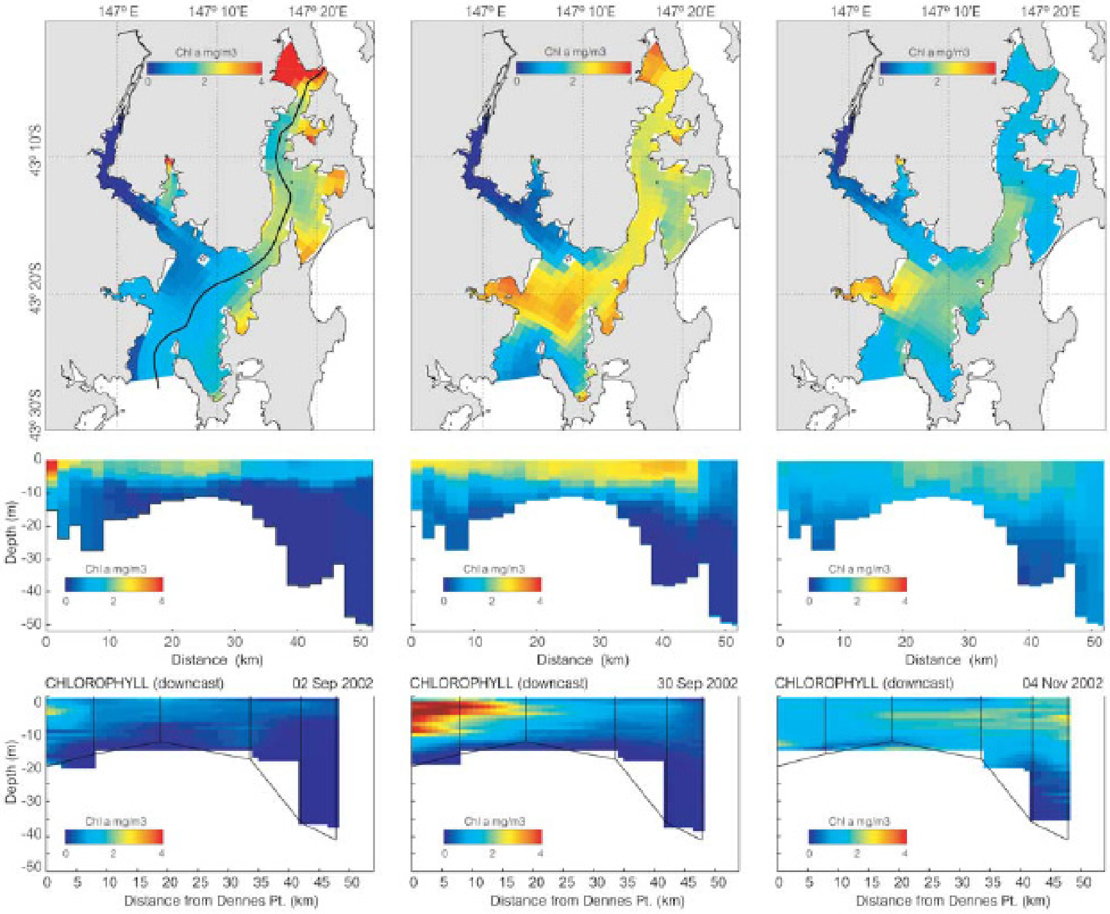


Figure 7.4. Modelled chlorophyll-*a* concentration in surface waters (upper panel), along a cross section through the D'Entrecasteaux Channel (middle) and derived from observations of fluorescence (lower panel) show the progression of the spring bloom from north in early spring (left) to south in late spring (right).

In the Huon Estuary the model was used to investigate the occurrence of dinoflagellate blooms. By including a parameterisation for dinoflagellate vertical migration, the model was able to demonstrate a plausible mechanism for the retention and accumulation of dinoflagellates in the estuary by the interaction of diel vertical migration with the estuarine circulation and river plume dynamics (Fig. 7.5) (Wild-Allen et al., 2005).

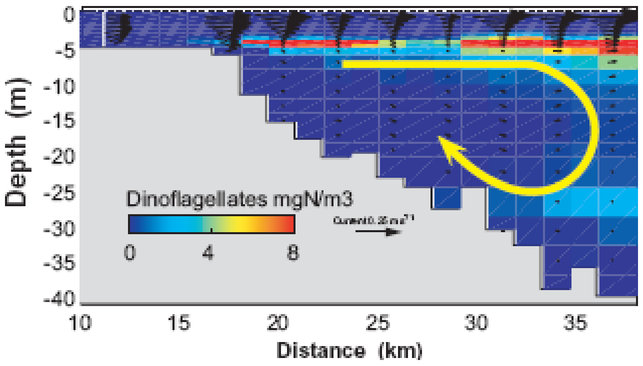


Figure 7.5. Modelled concentration of dinoflagellates and estuarine circulation along a cross section through the Huon Estuary.

Analysis from the hydrodynamic model shows that the region is well ventilated and flushed approximately every 2.5 weeks, with residual circulation from south to north (Herzfeld et al., 2004). Huon river water has high attenuation which limits phytoplankton growth in the upper estuary and in areas influenced by the river plume. Localized stratification due to river influence and favourable local winds results in episodic patchiness in phytoplankton growth during summer and autumn throughout the region. Resuspension of bottom material in shallow bays and constricted channels can elevate nutrient concentrations but also reduce available light, resulting in complex phytoplankton dynamics.

The model has shown that the biogeochemical dynamics of the Huon Estuary and D'Entrecasteaux Channel has high spatial and temporal variability resulting from the complex interaction of coastal morphology, hydrodynamics, local weather, opaque river water and a range of nutrient sources.

7.3. Evaluation of modelled nutrient sources and sinks

The biogeochemical model has enabled us to calculate total nutrient budgets for the Huon Estuary and D'Entrecasteaux Channel by integrating the load crossing the boundaries (assuming physics is approximately right). As shown in Table 7.1, most of the nitrogen in 2002 was contributed from the surrounding marine waters (60%: mostly delivered in winter), with similar contributions from the Huon River (23%) and salmonid farms (17%). However, the form of the nitrogen source is quite different with almost all of the farm-derived nitrogen being labile and most of the river-derived nitrogen being refractory (Fig. 7.6).

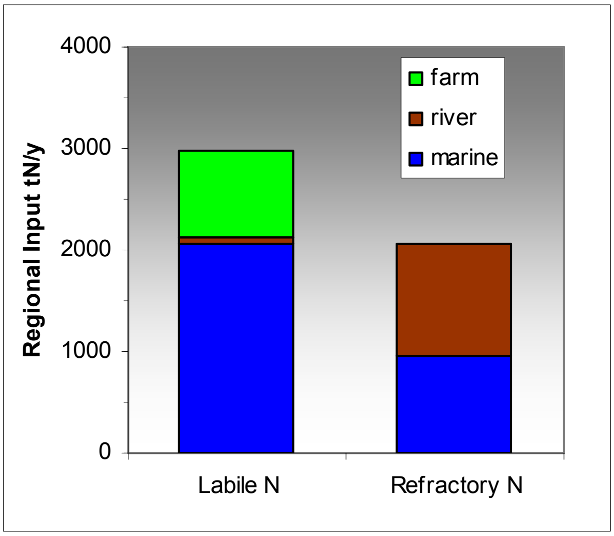


Figure 7.6. Contributions from salmon farms, Huon River and marine waters to labile and refractory nitrogen.

Total regional nitrogen sinks have been calculated (Fig. 7.7). For this purpose the D'Entrecasteaux Channel was divided into upper and lower basins since these 2 sub-regions show slightly different characteristics. As shown, most of the nutrients are flushed from the system. Denitrification is thought to remove only 8–15% of the nitrogen, although we have yet to confirm this directly.

Table 7.1. Sources of nitrogen

Source	**Total nitrogen**
marine	60%
river	23%
salmon farm	17%

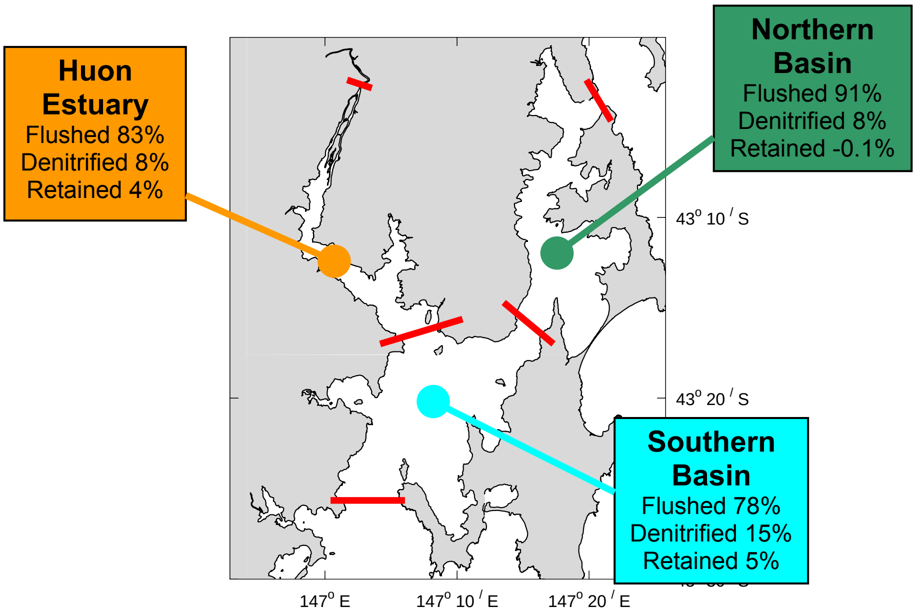


Figure 7.7. Estimates of amount of nitrogen flushed, denitrified and retained in 3 sub-regions of the Huon Estuary and D'Entrecasteaux Channel.

A regional nitrogen budget was estimated from the model simulations and shows an annual net in-flux of nutrients from rivers and farms and export of nutrients at the marine boundaries (Fig. 7.8). Recalling that 60% of the regional nitrogen is supplied across the marine boundary this export represents the excess of nitrogen that is not processed (buried or denitrified) within the region over the course of the year.

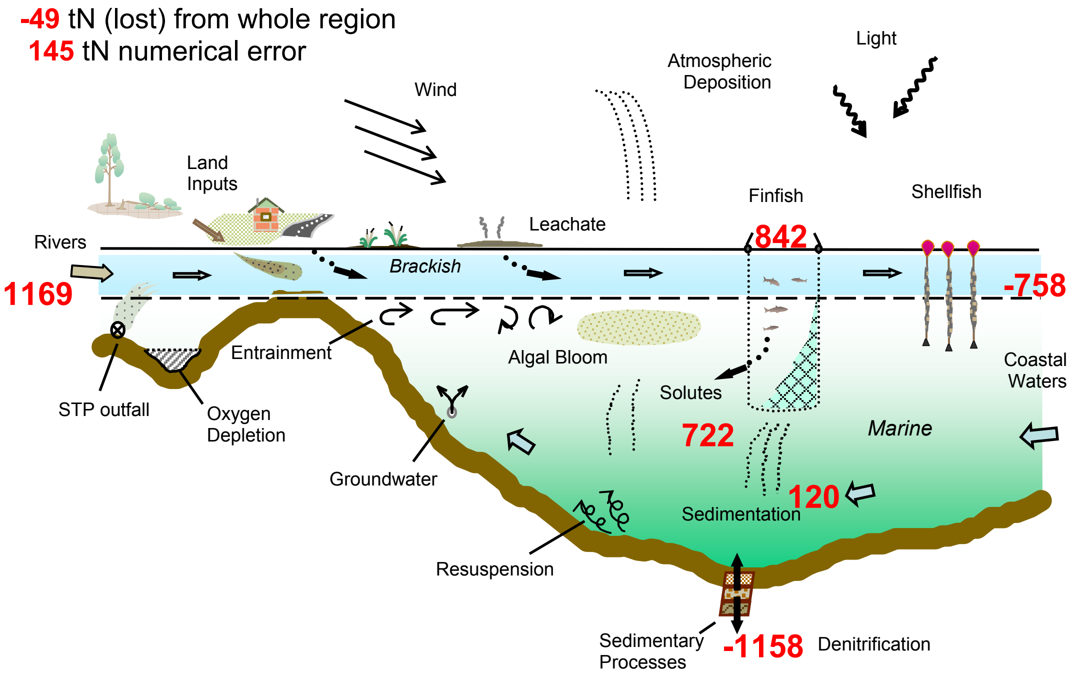


Figure 7.8. Modelled nutrient budget for the Huon Estuary and D'Entrecasteaux Channel.

7.4. Farm impacts scenario simulations

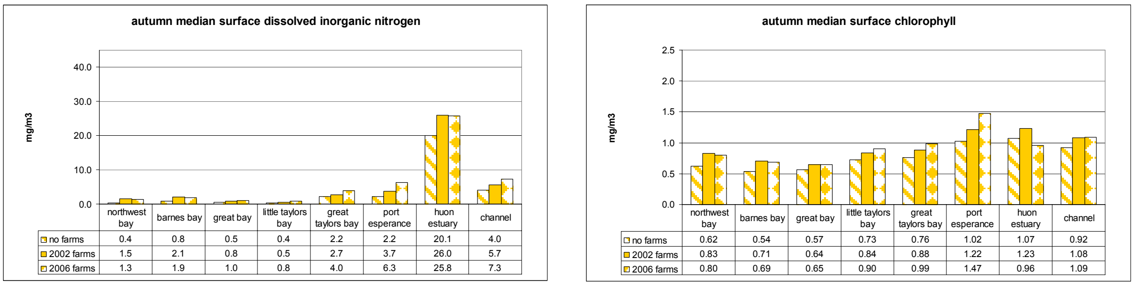
The impacts of farm discharges on the regional biogeochemistry were investigated by comparing scenario simulations with and without farm loads. In general, farm discharges had greatest impact on the nutrient and phytoplankton fields in summer and autumn by alleviating seasonal near-surface nutrient limitation and thus promoting phytoplankton growth. Riverine and marine fluxes of nutrient into surface waters are comparatively small during this period and the farm nutrient loads enhance the ambient surface nutrient and chlorophyll concentration. Farm discharges had a small impact on simulated dissolved oxygen concentrations likely due to efficient ventilation of the water column and general short flushing time of the region.

Table 7.2. Comparative farm nutrient load discharge scenarios into the Huon Estuary and D'Entrecasteaux Channel.

Input	No Farms	2002	2006	2009
Nitrogen	0	843	1216	2590
Phosphorus	0	146	213	454

Scenario simulations of conditions without farm waste, with 2002 loads, 2006 loads and projected 2009 loads show progressive nutrient enrichment, increase in phytoplankton chlorophyll and reduction in bottom water dissolved oxygen. Scenarios included variations in spatial and temporal farm loads and demonstrated a corresponding variability in impacts.

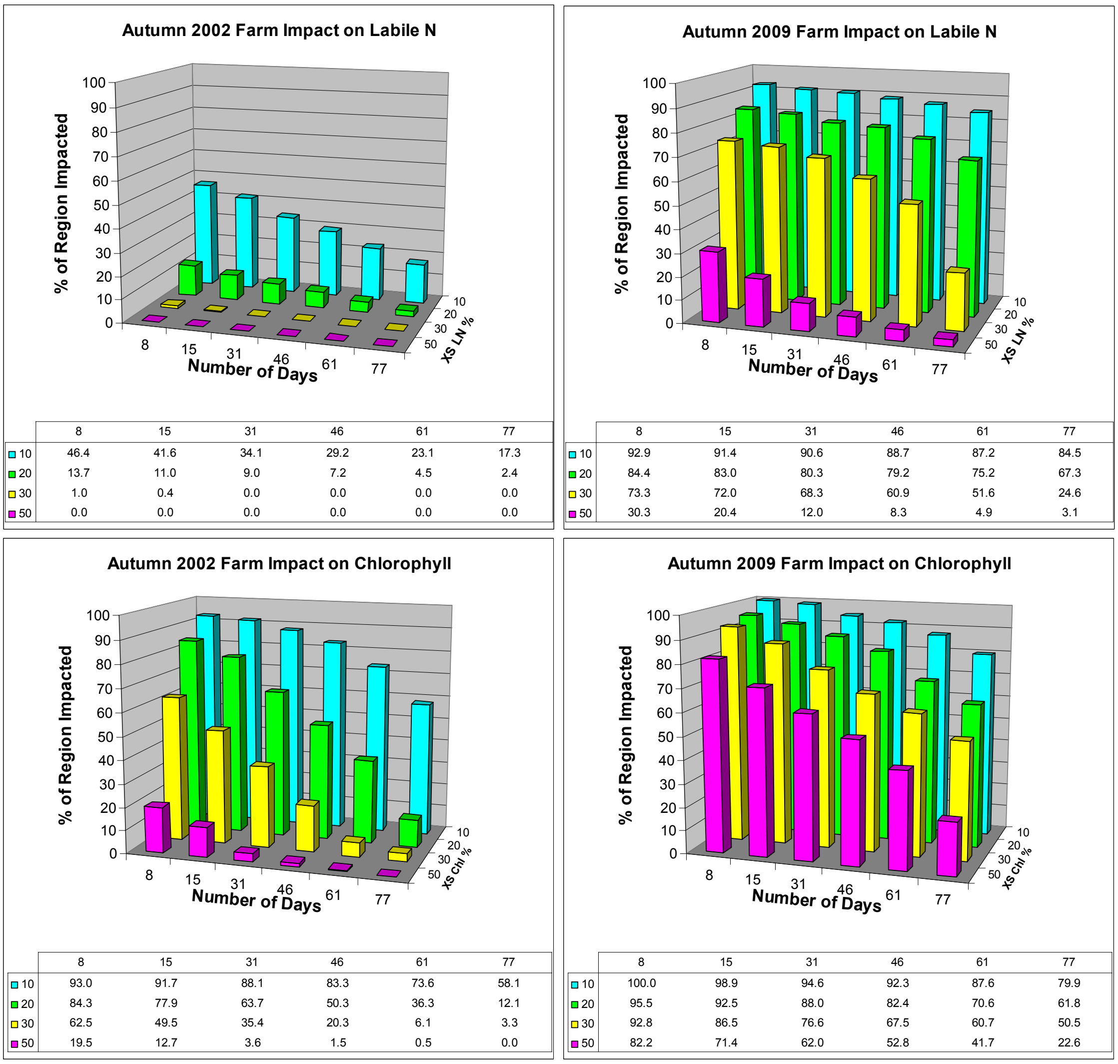
Useful metrics to communicate regional spatial and temporal farm impacts integrated effects over seasons and sub-regions of the model (Fig.7.9) or reported % area and duration of a threshold level of impact (Fig.7.10).



	northwest bay	barnes bay	great bay	little taylors bay	great taylors bay	port esperance	huon estuary	channel
▨ no farms	0.4	0.8	0.5	0.4	2.2	2.2	20.1	4.0
■ 2002 farms	1.5	2.1	0.8	0.5	2.7	3.7	26.0	5.7
▨ 2006 farms	1.3	1.9	1.0	0.8	4.0	6.3	25.8	7.3

	northwest bay	barnes bay	great bay	little taylors bay	great taylors bay	port esperance	huon estuary	channel
▨ no farms	0.62	0.54	0.57	0.73	0.76	1.02	1.07	0.92
■ 2002 farms	0.83	0.71	0.64	0.84	0.88	1.22	1.23	1.08
▨ 2006 farms	0.80	0.69	0.65	0.90	0.99	1.47	0.96	1.09

Figure 7.9. Seasonal DIN (upper) and chlorophyll (lower) surface concentrations in subregions of the model in autumn for scenarios with no farms, 2002 loads and 2006 loads.



	8	15	31	46	61	77
🟦 10	46.4	41.6	34.1	29.2	23.1	17.3
🟩 20	13.7	11.0	9.0	7.2	4.5	2.4
🟨 30	1.0	0.4	0.0	0.0	0.0	0.0
🟪 50	0.0	0.0	0.0	0.0	0.0	0.0

	8	15	31	46	61	77
🟦 10	92.9	91.4	90.6	88.7	87.2	84.5
🟩 20	84.4	83.0	80.3	79.2	75.2	67.3
🟨 30	73.3	72.0	68.3	60.9	51.6	24.6
🟪 50	30.3	20.4	12.0	8.3	4.9	3.1

	8	15	31	46	61	77
🟦 10	93.0	91.7	88.1	83.3	73.6	58.1
🟩 20	84.3	77.9	63.7	50.3	36.3	12.1
🟨 30	62.5	49.5	35.4	20.3	6.1	3.3
🟪 50	19.5	12.7	3.6	1.5	0.5	0.0

	8	15	31	46	61	77
🟦 10	100.0	98.9	94.6	92.3	87.6	79.9
🟩 20	95.5	92.5	88.0	82.4	70.6	61.8
🟨 30	92.8	86.5	76.6	67.5	60.7	50.5
🟪 50	82.2	71.4	62.0	52.8	41.7	22.6

Figure 7.10. % of region (area) where day mean depth integrated labile nitrogen (upper) and chlorophyll (lower) exceeds threshold over background (no farm) level.

In 2002, 21.9% of the region could be classified as mesotrophic (annual mean chlorophyll in top 13 m = 1–3 mg m^{-3} after Smith, 1998). In the 2009 scenario, the area of mesotrophic classification had increased to 54% of the total region (Table 7.3; Fig.7.11). Impacts were not evenly distributed throughout the region. Elevated chlorophyll concentrations relative to background levels were most significant in the northern part of the D'Entrecasteaux Channel, Little Taylors Bay, North West Bay, Port Esperance and Cygnet. Absolute changes in chlorophyll were most significant in the southern part of the D'Entrecasteaux Channel and Great Taylors Bay which have higher concentrations naturally. There was no evidence of eutrophication (annual mean chlorophyll in top 13 m > 3 mg m^{-3} after Smith, 1998) in any model simulation. Note, however, that the annual mean near-surface chlorophyll concentration in shallow water off Cygnet reached 2.7 mg/m^3 in the 2009 simulation, which was the closest to eutrophic conditions simulated in the region.

Table 7.3. Percent age of region (by area) in each classification.

	oligotrophic	mesotrophic	eutrophic
No Farms	90.2	9.8	0.0
2002	78.1	21.9	0.0
2009	46.0	54.0	0.0

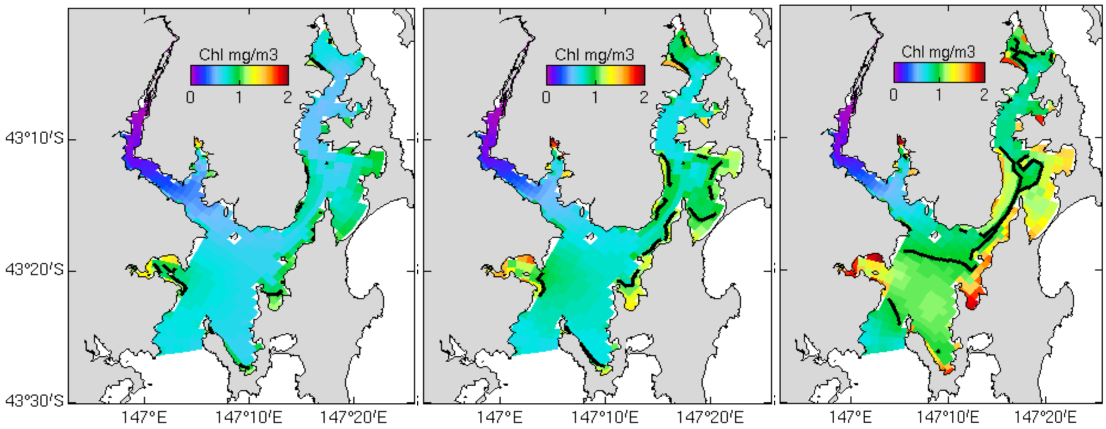


Figure 7.11. Annual mean chlorophyll concentration in the top 13 m of the water column from the model scenario without farm inputs (left); with 2002 farm inputs (middle); and with 2009 scenario farm inputs (right). Contour line denotes transition from oligotrophic (<1 mg m^{-3}) to mesotrophic (1–3 mg m^{-3}) classification after Smith (1998).

7.5. Monitoring system design and evaluation

Early warning of nutrient enrichment is vital for adaptive and sustainable ecosystem management, but sub-lethal impacts are often diffuse across variables, difficult to distinguish from natural variability and spatially and temporally affected by ambient flow and sediment regimes. Sophisticated models can now simulate the biogeochemistry of coastal ecosystems and explore a range of alternate futures. The biogeochemical model was used to identify optimal locations for environmental monitoring in south-east Tasmania, where the expanding salmon farming industry impacts coastal waters (Fig. 7.12).

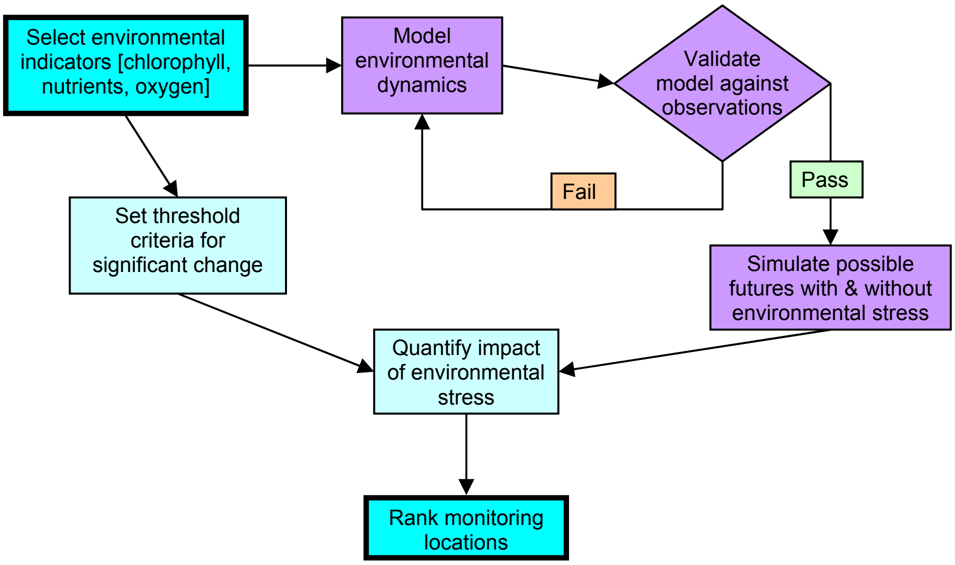


Figure 7.12. Monitoring program design process.

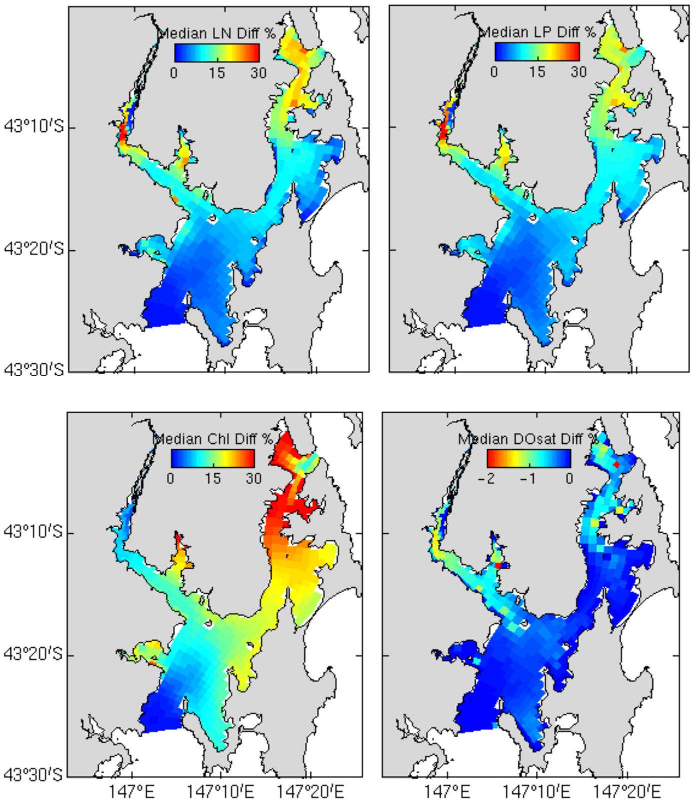


Figure 7.13. Annual mean % difference in day mean depth integrated labile nitrogen, phosphorus, chlorophyll and bottom water oxygen between model runs with and without farm inputs.

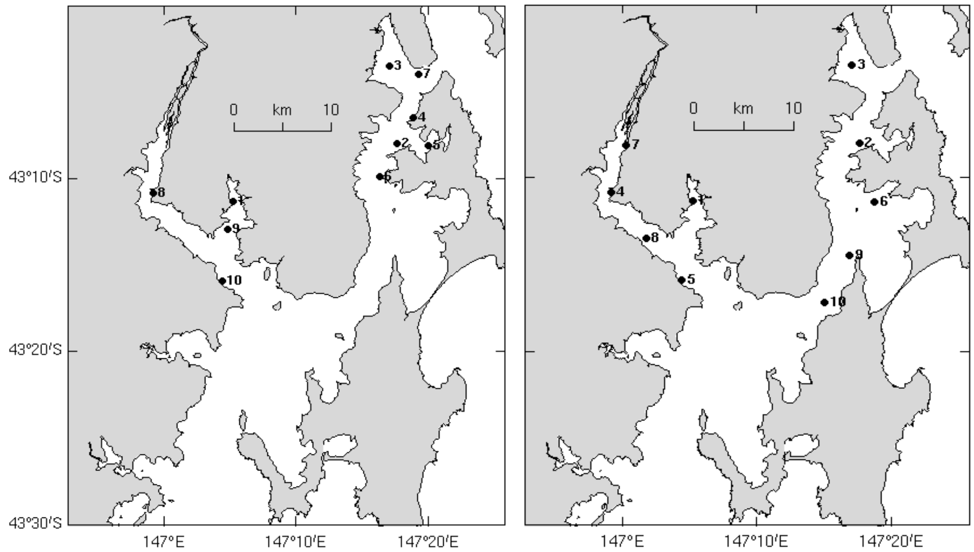


Figure 7.14. The 10 most likely places to detect impacts of fish farm loads on pelagic labile nitrogen, phosphorus, chlorophyll and bottom water oxygen concentration with stations > 2.5 km apart (left) and > 5 km apart (right).

Annual means of daily differences between model simulations with and without fish farm inputs (Fig. 7.13) were used to locate the places with smallest and largest fish farm impact (on average over the whole year). Maps of ranked location were calculated for depth-integrated labile nitrogen, phosphorus, chlorophyll and bottom water oxygen concentration, and summed to find the sites most frequently impacted for all substances. These are the best places to detect the biogeochemical impacts of fish farms over the course of a year (Fig. 7.14).

The location of stations most likely to be suitable for detecting fish farm impacts is consistent with our understanding of the residual regional circulation which generally flows from south to north via the estuarine circulation of the Huon (Herzfeld et al., 2004) and would facilitate accumulation of farm impacts in the Huon and north D'Entrecasteaux Channel. In this analysis, the location of stations (Fig. 7.15) is optimal for the year 2002. However, the analysis has been repeated for other years including the 2009 scenario simulation for input to the monitoring program design proposed in Chapter 8. Further details can be found in Thompson et al. (2008).

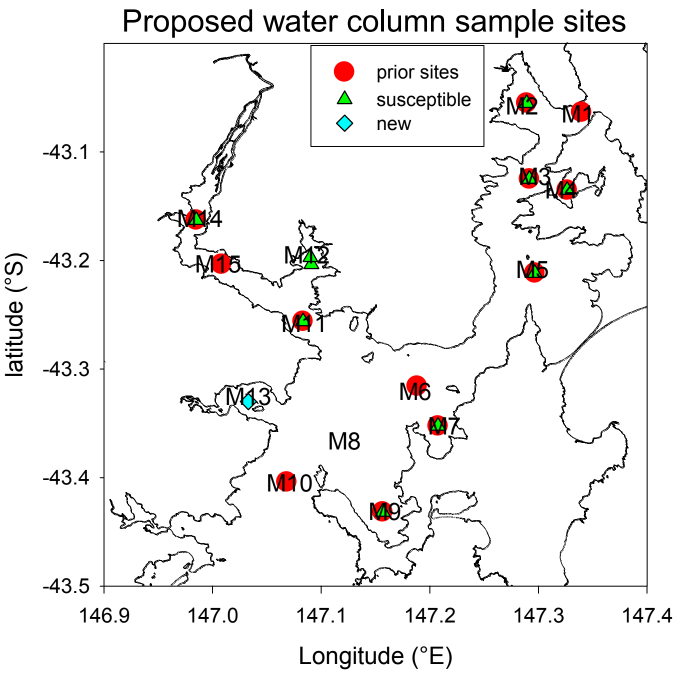


Figure 7.15. Proposed water column sampling sites for an environmental monitoring program.

The 2009 model simulation was sub-sampled at the proposed monitoring locations to evaluate the adequacy of the proposed temporal sampling frequency. For near-surface chlorophyll (mean concentration over surface 0–12 m), model results fell below the level 1 threshold at full temporal resolution and monthly sampling intervals, but sampling only once or twice a summer could trigger a level 1 response (Fig. 7.16).

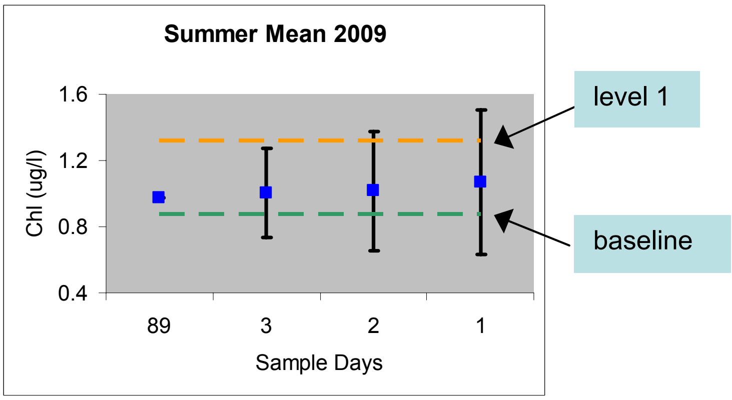


Figure 7.16. Modelled 2009 summer mean near-surface (0–12 m) chlorophyll concentration sampled at proposed monitoring stations at contrasting frequencies, and possible maximum/minimum concentration.

The biogeochemical model has been used to investigate far-field effects of nutrient inputs from a range of sources. Fish farm nutrient inputs are better known than some of the other sources of nutrients, and improvements in model skill could be anticipated if other nutrient sources (primarily rivers and STPs) were better characterised through improved monitoring.

7.6. Interannual variability

The 2002 model simulation was extended to the end of 2005 to investigate interannual variation in the region. The hydrodynamics of the region are strongly influenced by influxes across the marine boundary, simulated by multiple nesting of the model into a global domain, and by river flow, derived from observations (Fig. 7.17) (Herzfeld et al., 2004). There was considerable variability in monthly flow between years although integrated annual flow showed smaller differences, with 2002 slightly wetter and 2005 drier than other years (Table 7.3).

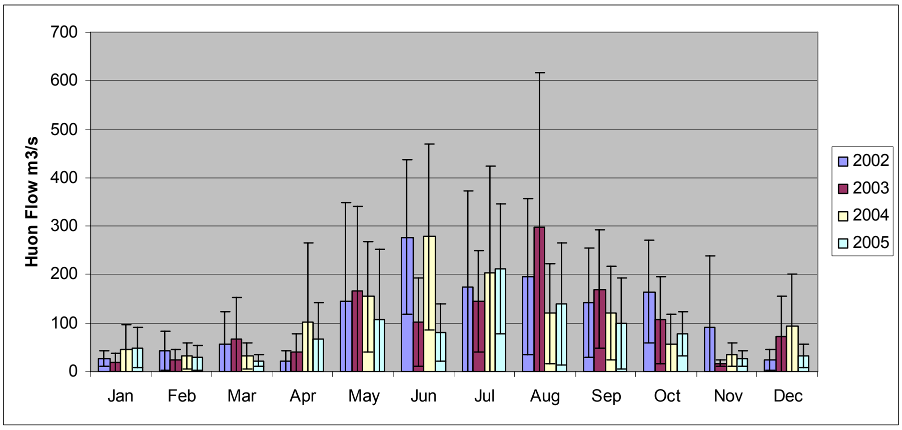


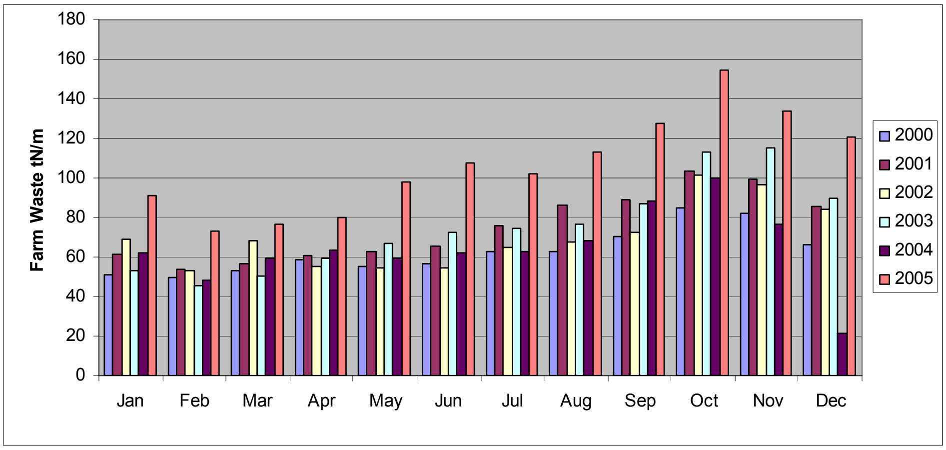
Figure 7.17. Monthly mean and standard deviation in Huon River flow derived from daily observations.

Table 7.4. Total annual Huon River Flow in $m^3 s^{-1}$.

	2002	2003	2004	2005
Annual flow	3.69E+09	3.25E+09	3.37E+09	2.49E+09

Biogeochemical model boundary conditions were derived for nitrogen from a relationship between observed nitrogen and temperature and for phytoplankton biomass from remote sensing observations of sea surface colour in Storm Bay. River nutrient concentrations were assumed constant at 2002 estimates (Wild-Allen et al., 2005) with loads varying according to flow. Biogeochemical influxes to the model were less well constrained than for the 2002 simulation, but a review of the model calibration showed that they provided an adequate first order estimate in the absence of *in-situ* observations for subsequent years.

Farm waste inputs were estimated from DPIW feed data assuming all pellets were consumed by the fish who discharged 5% and 0.8% of the total feed as waste nitrogen and phosphorus respectively in dissolved and particulate forms (in Wild-Allen et al., 2005). Seasonal waste input is lowest in February and highest in October (Fig. 7.18). The annual load dropped slightly in 2004, but increased significantly in 2005.



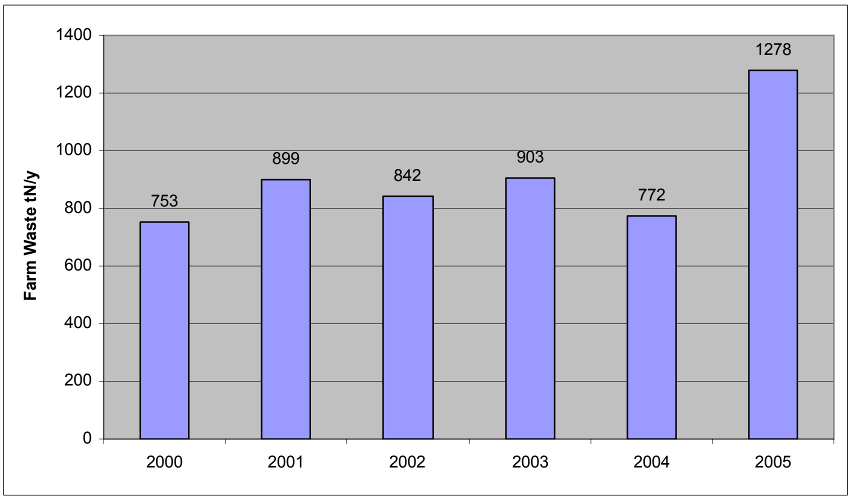


Figure 7.18. Annual farm waste nitrogen input by season (upper) and year (lower).

Natural inter-annual variability was evaluated from the 4 year model simulation with no farm waste input. Results show considerable variability in monthly nutrient, chlorophyll and bottom water oxygen concentrations. Maximum monthly surface DIN, near-surface chlorophyll (mean concentration 0–12 m), and minimum bottom water dissolved oxygen % saturation were evaluated from full spatial fields at 5 day intervals.

Maximum monthly surface DIN concentrations are high in the Huon Estuary for much of the year excluding winter (Fig. 7.19). This likely results from a combination of local river DIN from the Kermandie, stratification limiting vertical mixing and high attenuation limiting phytoplankton uptake. Other places with naturally high surface DIN include Great Taylor Bay.

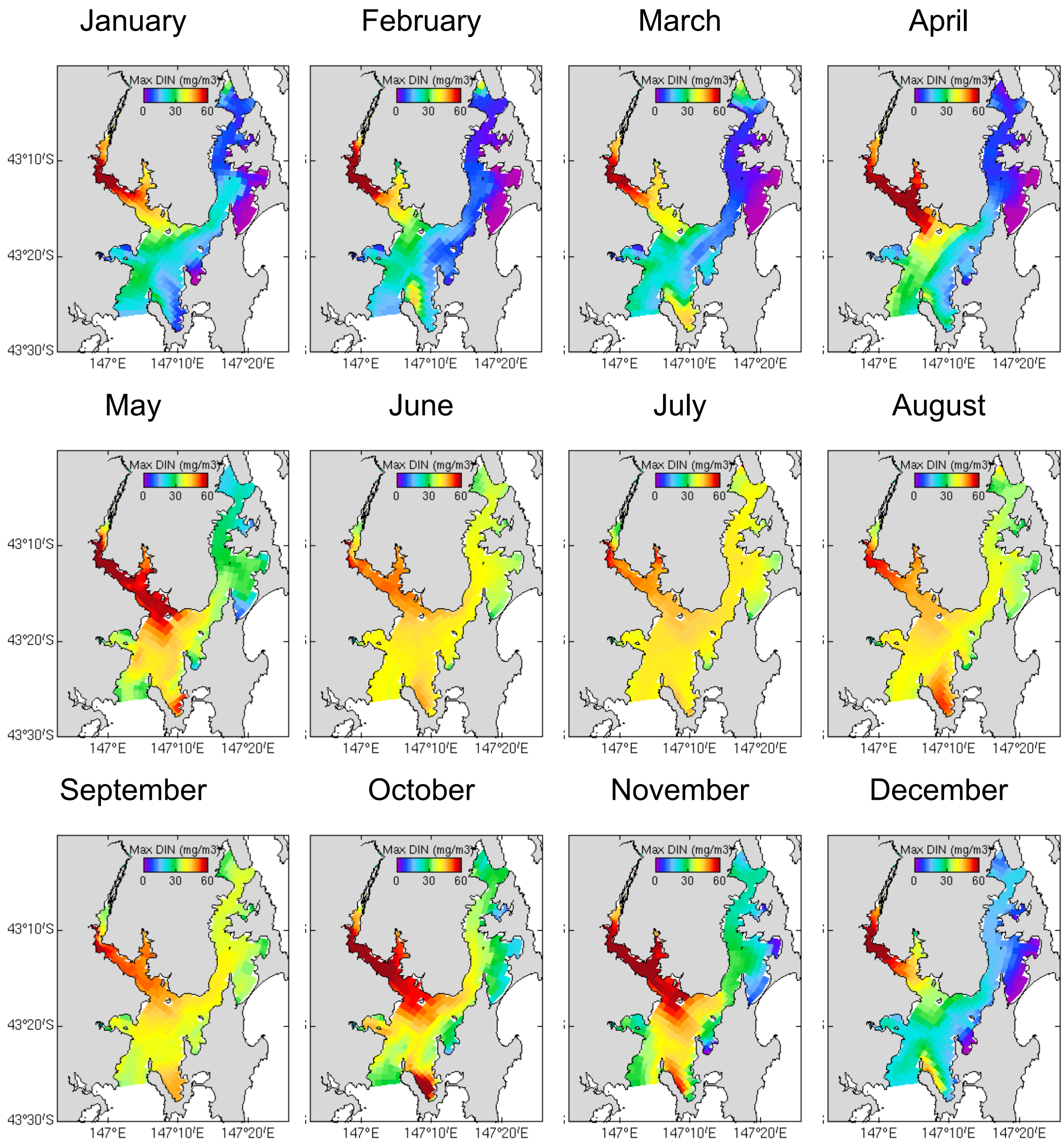
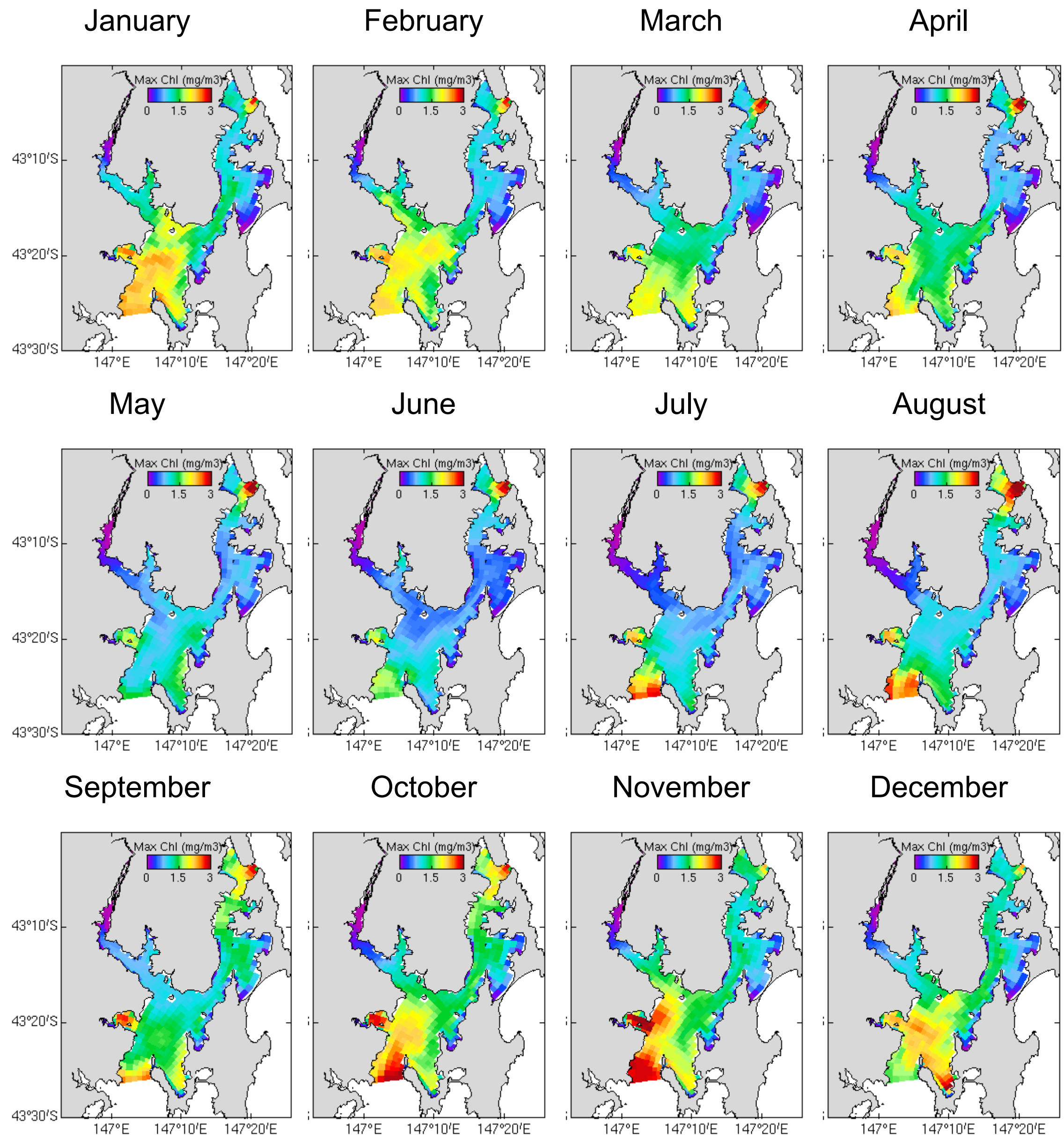


Figure 7.19. Monthly maximum surface DIN concentrations in the absence of fish farms calculated from modelled data at 5 day intervals over 4 years.

Chlorophyll has highest natural concentration in the southern D'Entrecasteaux Channel, Port Esperance and Great Taylors Bay (Fig. 7.20). Note that elevated concentrations at the boundary of the model should be treated with caution as they could reflect influx of phytoplankton-enriched waters due to farming activity as well as natural inputs since the boundary conditions were derived from remote sensing observations over the 4 year period rather than from direct measurements made outside of the model domain.


January February March April

May June July August

September October November December

Figure 7.20. Monthly maximum near-surface chlorophyll (0–12 m) concentrations in the absence of fish farms calculated from modelled data at 5 day intervals over 4 years.

Bottom water oxygen saturation is naturally depleted in the Huon Estuary, the southern D’Entrecasteaux and the mouth of North West Bay (Fig. 7.21). Lowest concentrations occur in summer and autumn when stratification restricts vertical exchange.

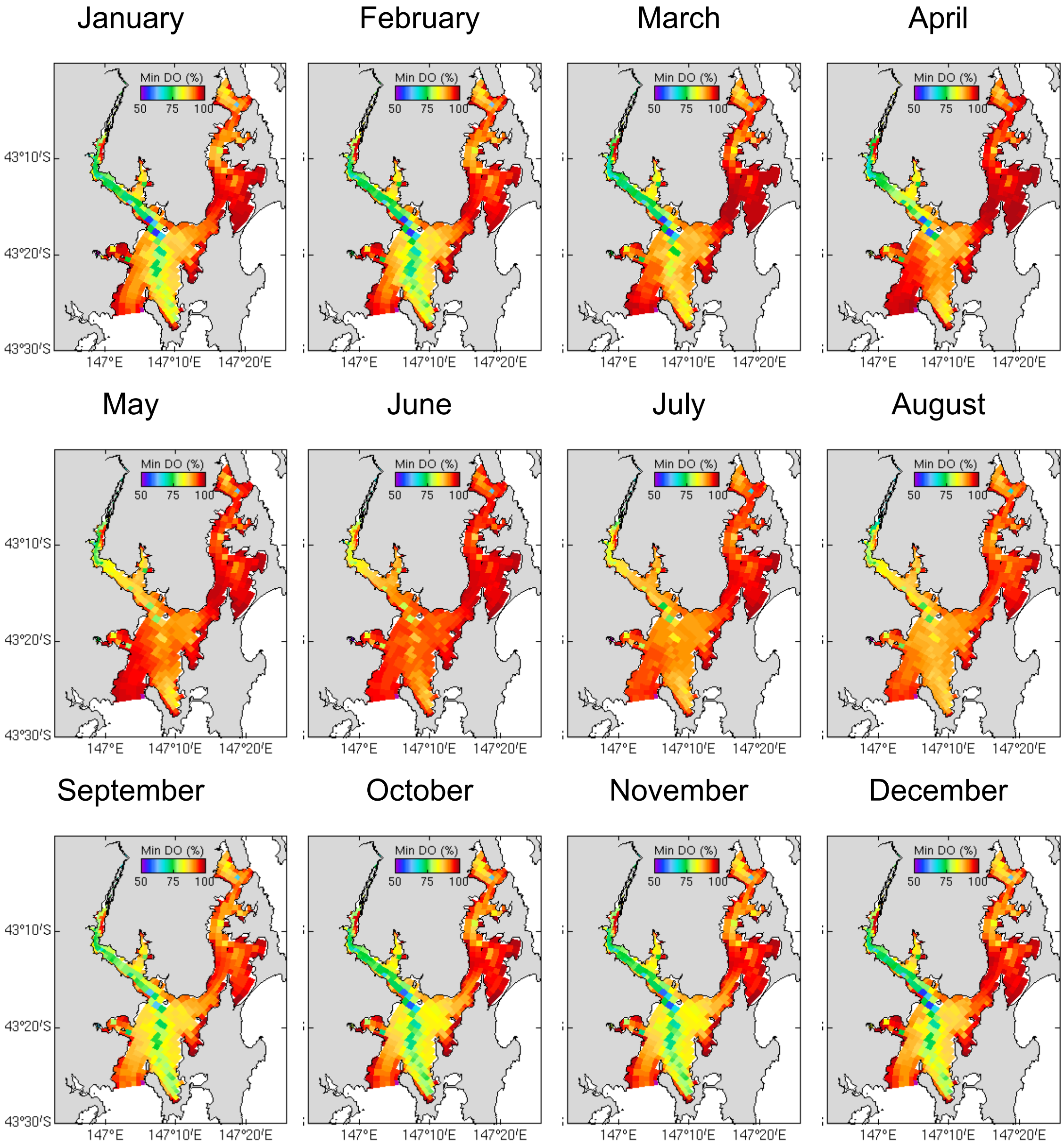


Figure 7.21. Monthly minimum bottom water dissolved oxygen saturation in the absence of fish farms calculated from modelled data at 5 day intervals over 4 years.

To evaluate whether farm impacts could be detected in excess of natural interannual variability, model data from 4 years of simulation with farm inputs were compared with the monthly maximum and minimum concentration simulated by the model without farm inputs. Spatial maps show the % of sample days when farm impacts exceed natural monthly maxima or minima for each year (Fig. 7.22).

In general this analysis shows the likelihood is small of detecting farm impacts in excess of natural system variability throughout much of the region. Even in the most likely places only 60% of surface DIN, 30% of near-surface chlorophyll or 30% of bottom water oxygen samples will exceed natural variability. The probability of detecting farm impacts exceeding natural variability does not appear to be closely correlated with the interannual variation in farm loads suggesting that spatial and temporal variability in hydrodynamics and marine and river influxes dominate the regional biogeochemistry.

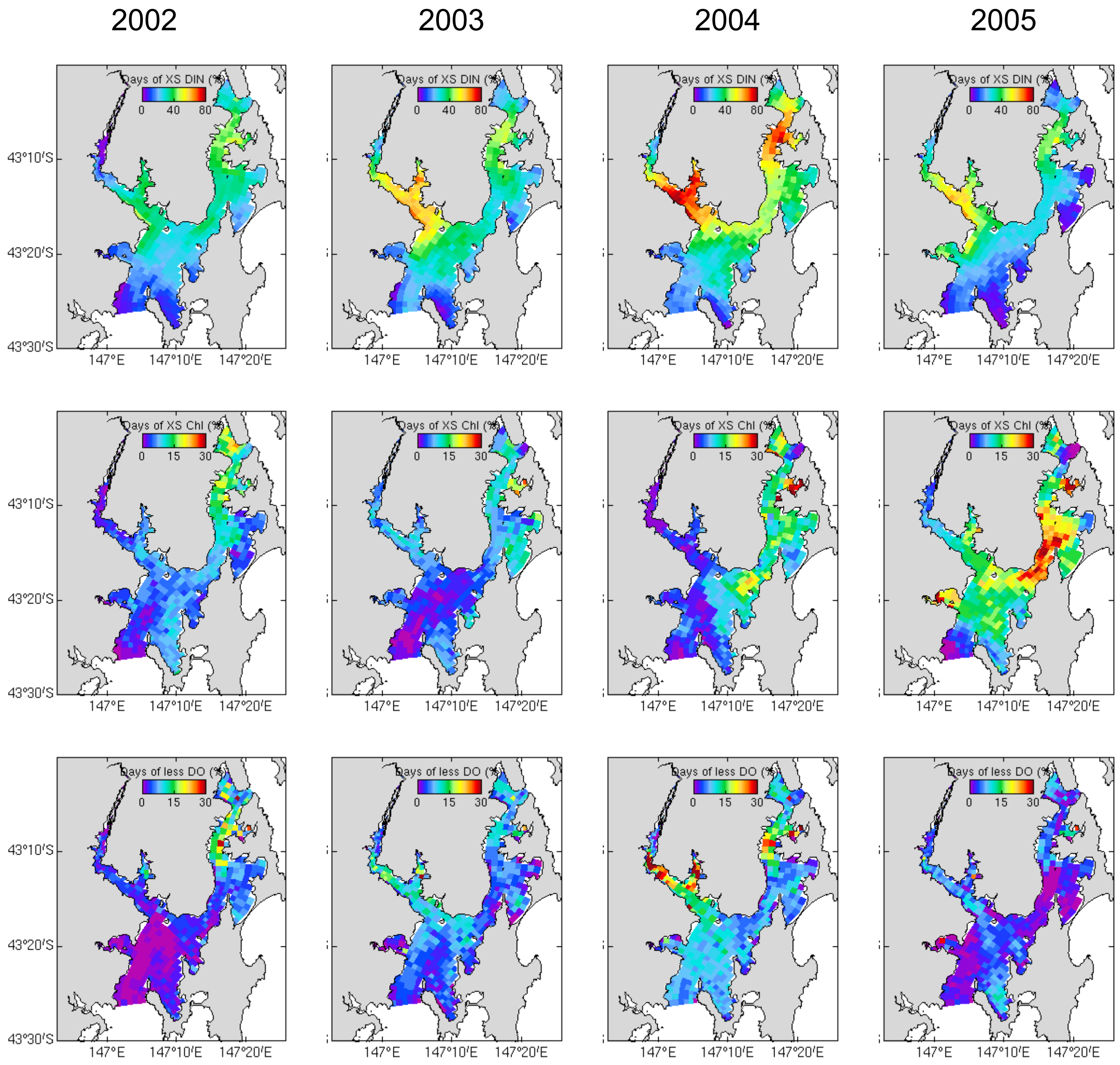


Figure 7.22. The % of sample days when concentrations exceed natural monthly maximum surface DIN (upper) or near-surface chlorophyll (middle) and were less than natural monthly minimum bottom water oxygen saturation (lower) for each modelled year including farm loads.

These model results suggest that it is necessary to collect data with high temporal resolution to characterise the impact of fish farm nutrients in a region with high natural variability.

7.7. Summary

A sophisticated biogeochemical model has been implemented within a coupled 3D hydrodynamic and sediment model to simulate the cycling of carbon, nitrogen and phosphorus through dissolved and particulate organic and inorganic forms in the pelagic and surface sediments of the Huon Estuary and D'Entrecasteaux Channel. The model has been validated against observations and captures the essential seasonal dynamics of nutrients and plankton in the D'Entrecasteaux Channel and side bays, and slightly less well in the Huon Estuary. Model insights into regional biogeochemistry include the evolution of the spring phytoplankton bloom from north to south and simulation of a plausible mechanism for the retention and accumulation of dinoflagellates in the Huon Estuary. The model has shown that the biogeochemical dynamics of the region has high spatial and temporal variability resulting from the complex interaction of coastal morphology, hydrodynamics, local weather, opaque river water and a range of nutrient sources.

Analysis of model results shows that in 2002 about 60% of the nitrogen supplied to the region was sourced from marine waters, 23% from rivers and 17% from fish farms. Nutrient sources vary from mostly refractory (rivers) to mostly labile (farm waste) with corresponding impact on phytoplankton assimilation. Further analysis shows that significant amounts of nitrogen are flushed from the system and in 2002 there was a net export of 758 tonnes of N into adjacent waters. Scenario simulations were used to evaluate the impact of contrasting farm waste loads on the regional biogeochemistry. The differences in modelled nutrients, chlorophyll and dissolved oxygen between model runs with and without farm waste loads, were summed over sub-regions to communicate the progressive increase in farm impact from 2002 through 2006 and for a hypothetical 2009 scenario. Summary statistics integrated the time and area over which farm impacts exceeded given thresholds and communicated the shift from a naturally oligotrophic system to one with 54% by area classified as mesotrophic.

The scenario simulations were used to identify sites most frequently impacted by farm waste nutrients to inform the design of a coastal monitoring program. The effectiveness of the monitoring program was then evaluated for pelagic chlorophyll by sub-sampling the 2009 simulation at the proposed locations and at varying temporal resolution. Given 2002 meteorology and hydrodynamics, monthly sampling of the model at the proposed sites did not trigger a level 1 response.

A four year model simulation [triple nested in a global model and forced with real meteorology, river flows, biogeochemical boundary conditions derived from a limited set of observations and real farm loads], was used to investigate interannual variability. During this period 2002 was a slightly wetter year and 2005 was slightly drier with elevated farm loads. Natural interannual variability was assessed from a repeat simulation without farm loads and demonstrated a considerable range in results. Results from all years were summarised to show regions where naturally high surface DIN, near-surface chlorophyll and depleted bottom water oxygen had occurred. Further analysis showed the likelihood is small of detecting farm impacts in excess of natural system variability throughout much of the region. The probability of detecting farm impacts exceeding natural variability did not appear to be closely correlated with interannual variation in farm loads suggesting that spatial and temporal variability in hydrodynamics, and marine and river influxes dominate the regional biogeochemistry. This analysis suggests that it is necessary to collect data with high temporal resolution to characterise the impact of fish farm nutrients in a region with high natural interannual variability.

7.8. Acknowledgments

Peter Thompson and Peter Oke are thanked for many helpful discussions.

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Chapter 8

Monitoring the Huon Estuary and D'Entrecasteaux Channel for the Environmental Effects of Finfish Aquaculture

8.1. Introduction

The Huon Estuary and D'Entrecasteaux Channel form part of the Bruny Bioregion in the southwest of Tasmania. The region is largely oligotrophic with low concentrations of nutrients and high water quality. Over the last 200 years considerable changes have occurred both in these water bodies and their surrounding catchments. Changes such as increased sedimentation, nutrient inputs, sewage, habitat loss and regulation of freshwater flows are recognized as potential threats (Jordan et al., 2002, RPDC 2007). The major natural sources of nutrients into the region include the intrusion of oceanic waters during winter and various inputs from land, streams and rivers (Butler et al., 2000). Over the past 30 years the amount of aquaculture production (mostly salmonids) in the region has grown dramatically to the point where nutrients from finfish farming are the largest known anthropogenic source of nitrogen to the ecosystem.

In this chapter the potential of various parameters and indicators was assessed for their value in monitoring the environmental condition of the Huon Estuary and D'Entrecasteaux Channel. The overall goal was to design a monitoring system that will ensure the environment remains healthy and capable of sustaining a wide range of uses even when under stress from increasing nutrient loads. The chapter includes a brief review of the principles of monitoring, details on the process of selecting suitable indicators and a rigorous review of existing environmental data. Modeling was extensively used to improve the performance of the monitoring system. Converting the environmental indicators into performance measures that can be used in a regulatory manner has resulted in a 3-tiered approach designed around detecting and responding to symptoms of increased environmental risk. A fuller account is provided in the technical report by Thompson et al. (2008) on the CD that accompanies this report.

The net result is a proposed monitoring program that is relatively inexpensive, sensitive and robust.

8.2 Background on designs of monitoring programs

The basic design principles for modern impact assessment were published by Roger Green (1979) and have been subsequently adopted by many regulators such as the US EPA and Commonwealth and State Governments in Australia. The 5 proposed designs for assessment and inference (Fig. 8.1) have been shown to apply to many situations, but there is no single design that will assess impacts that have already commenced, are ongoing, widespread or may manifest themselves in areas remote from the disturbance site. For example, adding CO_2 to the atmosphere or adding nutrients to the ocean both require the combination of assessment methods 2, 3, 4 and 5 (Green, 1979); with the inclusion of type 1 if possible.

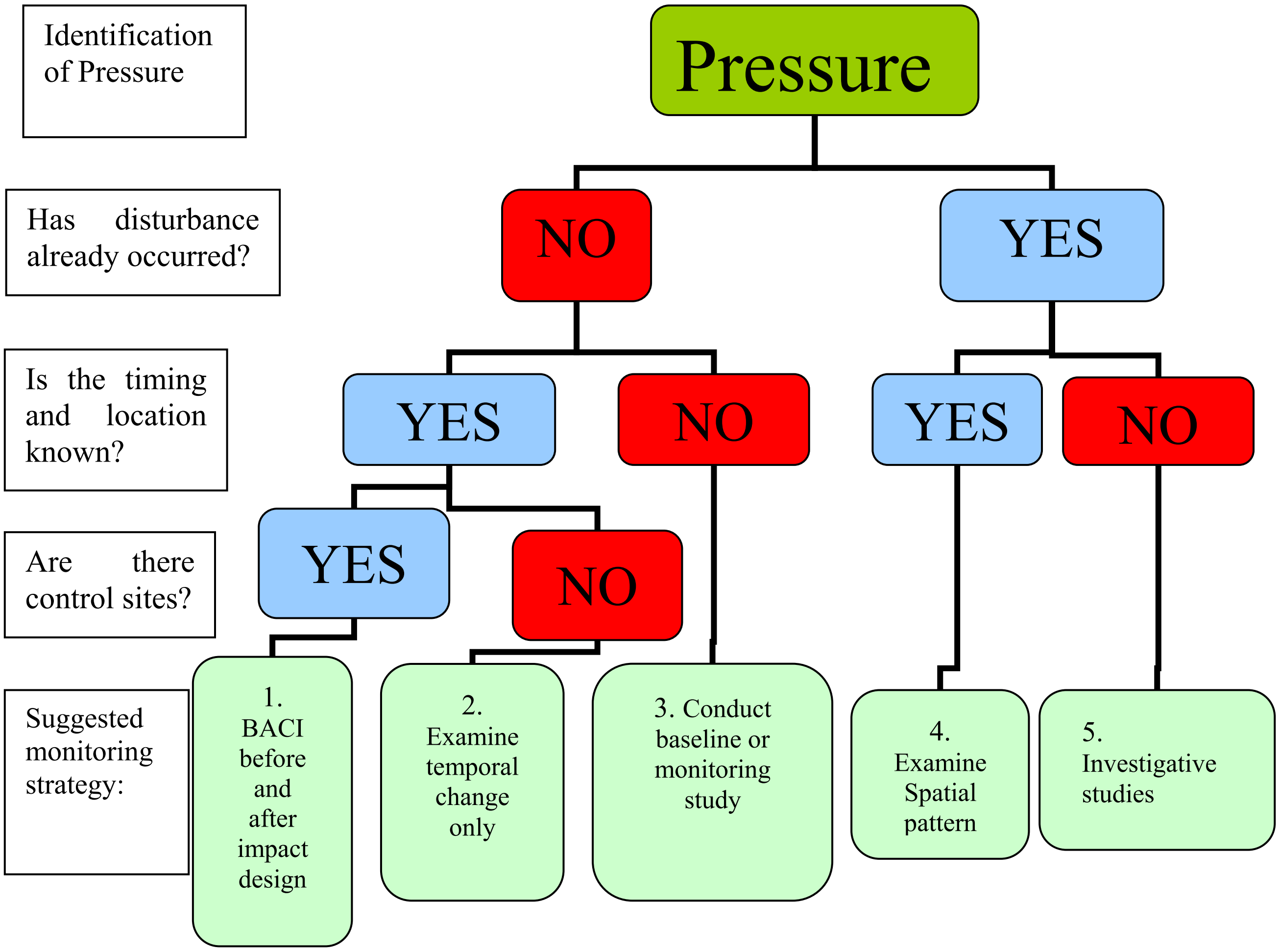

Figure 8.1. Identifying 5 monitoring designs for making an inference about possible environmental impacts to be used in impact assessment. From Green (1979), also ANZECC/ARMCANZ (2000).

While we consider the Huon Estuary and D'Entrecasteaux Channel to be under pressure from a number of stressors, the estimated 800 tonnes of extra nitrogen being added as a result of finfish aquaculture was identified as a 'pressure' of primary concern in the development of this monitoring program. In this report, we recommend an impact assessment procedure based upon both a coupled hydrodynamic and biogeochemical model and conventional monitoring where the assessment of temporal change relative to a defined baseline is used as the primary method of inference of an impact. Both approaches are proposed for use in regulatory mode within an adaptive management framework.

In its simplest form, the process of designing the monitoring program has the following steps:

1. observe the environment
2. select indicators
3. determine number, location and frequency of sampling
4. select performance measures
5. set trigger values
6. determine responses/actions if trigger values are exceeded
7. monitor environment

To detect change over time it is essential that there be some consistency in indicators and locations. The greater the consistency the more capability the monitoring program will have to detect change over time. It is always possible to add indicators, sample sites and increase the frequency of sampling as such changes will not detract from the capability of the monitoring program. Other aspects of the monitoring program that are reasonably flexible and can, with suitable stakeholder consultation, be changed at any time include: performance measures, trigger values and actions. The latter are suitable for continuous improvement using an adaptive management approach.

8.3. What to measure?

The D'Entrecasteaux Channel and Huon Estuary are oligotrophic to mesotrophic water bodies. The D'Entrecasteaux Channel is characterized by good trophic coupling such that undesirable algal blooms are rare. For both the Huon Estuary and D'Entrecasteaux Channel nutrient cycling is generally efficient and dissolved nutrients do not accumulate in the water column during summer. A diverse range of habitats support a high diversity of biota including seagrasses, macroalgae, phytoplankton, zooplankton, fish and other taxa. While the ecosystem is judged to be in good health it is under stress from a number of increased human uses of both the aquatic ecosystem and adjacent terrestrial ecosystem. A list of these pressures + stressors and their potential impacts was compiled (Appendix 1 in Thompson et al., 2008). This list is similar to those produced by others when considering similar issues (e.g. Ward et al., 1998). A range of potential indicators or parameters that could be measured to quantify the degree of pressure or stress was constructed from the list, but the range was considered too broad to be implemented in a practical monitoring program. The range of parameters was reduced to a subset in a two step process with an iterative loop described below.

In the first step of this process of selecting indicators it was necessary to agree on the list of 'good' environmental indicators or parameters that could be measured. We scored the list of potential parameters/indicators against the following criteria that characterize what a good parameter should be: anticipatory, sensitive, diagnostic, broadly applicable, correlated to actual environmental effects, timely and cost effective, regionally and socially relevant, easy to measure, constant in space and time (i.e. high signal: noise ratio) and non-destructive (Appendix 2 in Thompson et al., 2008; after Maher et al., 1994). In this process each parameter/indicator was rated 0–10 against each criterion and those with the highest scores were selected for risk assessment. The highest scoring parameters were: phytoplankton biomass, frequency of algal blooms, the concentration of dissolved oxygen, changes in nutrient cycling as manifest by the increase in concentrations of NH_4^+, organic enrichment of the sediments, increases in bacteria involved in the remineralization of the organic matter, and pathogenic bacteria. A great number of possible parameters or indicators scored badly in this process and were eliminated from further consideration. Pathogens that might be transferred between wild and farmed organisms were not considered 'good' for monitoring purposes due to the expected low signal:noise ratio and were not considered further. Other possible indicators were also eliminated from further consideration in this process. *The remaining parameters can all be summarized as indicators of uncoupling between trophic levels.*

In the second step we considered the value of these parameters to a monitoring program in terms of their contribution to risk (= *likelihood* x *consequence*, after Fletcher et al., 2004). For example, is a particular parameter likely to change, and if so how much will this change harm the environment? Where possible, existing data sets were examined and the frequency of deleterious conditions used to establish likelihood. Deleterious conditions were defined from both national and international guidelines. We note that guidelines most commonly deal with concentrations (e.g. 2 ppm DO). We identified two other components of risk: the spatial and temporal extent. In the current (ESD) type risk assessments the temporal frequency is dealt with in the consequences part of the assessment, but the spatial extent is not explicitly considered. For some indicators, like DO, it is the duration and spatial extent rather than frequency that are better indicators of risk. At this time there is no agreed framework for including duration or spatial extent into ESD type risk assessments. For the purpose of this report we have considered widespread and persistent deleterious conditions as having greater consequences. Observations and modelling of DO, nutrients and chlorophyll-*a* were used for information on spatial and temporal extent of the pressures. We applied the risk assessment framework (Table 8.1 and Appendix 3 in the technical report by Thompson et al., 2008); after Fletcher et al., 2004) to identify the parameters selected in step 1 that are associated with a high (> 12) risk to the environment.

In the iterative loop portion of this process we confirmed that the conclusions derived from the 2 steps above were consistent with our understanding of the major pressures on the ecosystem and their likely impacts.

Table 8.1. Risk Matrix – numbers in cells are the product of the consequence and the likelihood ratings = the risk (adapted from Fletcher et al., 2004).

		Consequence rating					
Likelihood rating		**Negligible**	**Minor**	**Moderate**	**Severe**	**Major**	**Catastrophic**
		0	**1**	**2**	**3**	**4**	**5**
Remote	**1**	0	1	2	3	4	5
Rare	**2**	0	2	4	6	8	10
Unlikely	**3**	0	3	6	9	12	15
Possible	**4**	0	4	8	12	16	20
Occasional	**5**	0	5	10	15	20	25
Likely	**6**	0	6	12	18	24	30

Increasing risk

Consequence ratings are multiplied by likelihood ratings to give risk categories with values of 0 – 6 considered low; 7 – 12 (░) moderate, > 12 (▓) high (after Appendix 3 in Thompson et al., 2008; adapted from Fletcher et al., 2004).

We identified some risks that we believe are high because they are likely to occur and would have severe to major consequences on ecosystem functioning. Such risks are defined as having the potential to change the population dynamics resulting in different species/groups becoming major components of the region (after Fletcher et al., 2004; see Appendix 3 in Thompson et al. (2008) for details). Recovery from such an impact is possible but it might require a change in the degree of stress and could take months to years to occur. For example, there is evidence of both an increase in phytoplankton biomass and a change in phytoplankton community composition in the Huon Estuary between 1997 and 2004 (risk factor ~ 18). Similarly there is evidence of occasional low dissolved oxygen concentrations which could have 'major' consequences for ecosystem functioning (risk factor ~ 20). In the iterative loop we confirmed that the conclusions derived from this 2-step process were consistent with our understanding of the major stressors and their likely impacts. We consider the indicators and parameters identified to be the best candidates for a monitoring system to be used in the Huon Estuary and D'Entrecasteaux Channel.

8.4. What has been, or is being, done in terms of regional environmental assessment

In Tasmania there is a legislative requirement that all finfish aquaculture operations (farms) undertake environmental monitoring both within farms and at reference positions outside the farm boundaries. Although there may be some monitoring requirements that are lease-specific, depending on the farmed animal and particular environmental conditions of the given area, it is a requirement that all subtidal farms undertake an initial assessment of the local current conditions, bathymetric profile and seabed characteristics (including habitat mapping and collection of video footage). All finfish leases are also required to undertake a baseline assessment of the sediment conditions including evaluation of sediment redox potential, sulfide levels, particle size and an evaluation of the infaunal community structure. Between February 1997 and

January 2003, a total of 240 environmental surveys for finfish aquaculture have been conducted, all of which included video assessments and 88 of which included benthic infaunal surveys. To date the results of this monitoring program have indicated very few instances (~ 4%) where an effect on the benthos was observed beyond the lease boundary and in all of these instances it was determined to be either incorrect pen positioning or remnant farming activity rather than the dispersal of particulate organic material from farming operations within the lease area (Woods et al., 2004). Consequently it would seem unlikely that the industry is having a major impact on the benthos outside the lease areas. Ongoing monitoring requires that video assessments be undertaken annually at all finfish farm leases. This includes evaluation of video footage at approximately 45 to 50 reference locations throughout the Huon and D'Entrecasteaux Channel.

In addition many finfish aquaculture operations monitor a range of environmental parameters (i.e. DO, temperature, salinity and phytoplankton) at a wide range of sites as part of their general farm management routines. These data may be extremely useful in an ongoing monitoring program. However, some data will be more reliable than others; some information is obtained routinely and frequently, while other data may only be available sporadically. Sampling and analytical methods also vary and consequently the results may not be comparable. For example, some operators use taxonomists accredited by the National Association of Testing Authorities (NATA) and some do not, some sample phytoplankton with nets while others use bottle samplers. The Tasmanian Shellfish Quality Assurance Program (TSQAP) samples phytoplankton at all shellfish leases on a regular basis (weekly to monthly depending upon assessed risk of a toxic phytoplankton bloom). This provides information on up to 29 sites, approximately 9 of which are in the Bruny region. Sample methods are standardized and analyses are undertaken in the Algal Unit, Analytical Services Tasmania (AST) by skilled, NATA-accredited phytoplankton taxonomists.

Other major environmental or water quality sampling programs in the region include the Derwent Estuary Program (http://www.derwentestuary.org.au/) and some local sampling in association with sewage treatment plants (STP). Monitoring programs for marine and aquatic environments are currently undergoing active design at both national and state level with reports under development or available in draft form (e.g. Scheltinga et al., 2006; Temby and Crawford, 2006). We have every expectation that the material presented herein will be compatible with those programs.

8.5. Where and when to measure

In keeping with well-established principles, sampling to characterize the marine environment may be either random in time and space or stratified. The choice between random and stratified sampling is dependent upon whether the environment has recognizable patterns in the parameter of interest (Green, 1979). Where there are recognizable patterns then stratified sampling is a better option (Green, 1979; Jassby et al., 1997) to maximize the benefit/$cost of monitoring. Cost effectiveness was a major consideration in the design of this monitoring system to detect ecosystem-wide effects of eutrophication. To keep costs down sample frequency was kept low whenever dedicated sampling from a vessel was required. To keep analytical costs low the number of samples was constrained but their location was chosen in an effort to make them 'representative' of a large area. Sites that were susceptible to impacts were identified from existing data and by the coupled hydrodynamic and BCG modelling and added to complete the list of proposed sites. Additional sampling from TSQAP, local industry and local government can add valuable information to the backbone of sampling proposed here.

A three dimensional coupled hydrodynamic, sediment and biogeochemical model was applied to inshore waters of southern Tasmania to evaluate the environmental impact of the existing salmonid fish farms. The model simulates the seasonal cycling of organic and inorganic carbon, nitrogen, phosphorus and oxygen through multiple phytoplankton, zooplankton, nutrient and detrital pools and is validated against observations collected in 2002. Model analysis of scenario simulations, with and without fish farm inputs (based on 2002 data for 21 farms), identified the system-wide spatial and temporal environmental footprint of the industry. This information was used to guide the selection of locations and periods of the year when the ecosystem is most susceptible to fish farm waste. Relatively un-impacted sites and times can also be identified and selected to provide control sites.

8.5.1. Evaluation of change

Changes in modelled substances between the simulations with and without farm inputs were summed for each model grid square. The number of days in the year when inputs from fish farms resulted in model-simulated nitrogen, phosphorus, chlorophyll and DO different from background (i.e. without fish farm inputs) were estimated (chlorophyll-*a* example in Fig. 8.2). Portions of the Huon Estuary and D'Entrecasteaux Channel were consistently impacted for all four with highest impacts in the northern section of the Channel (Fig. 8.2 and Appendix 4 in Thompson et al., 2008).

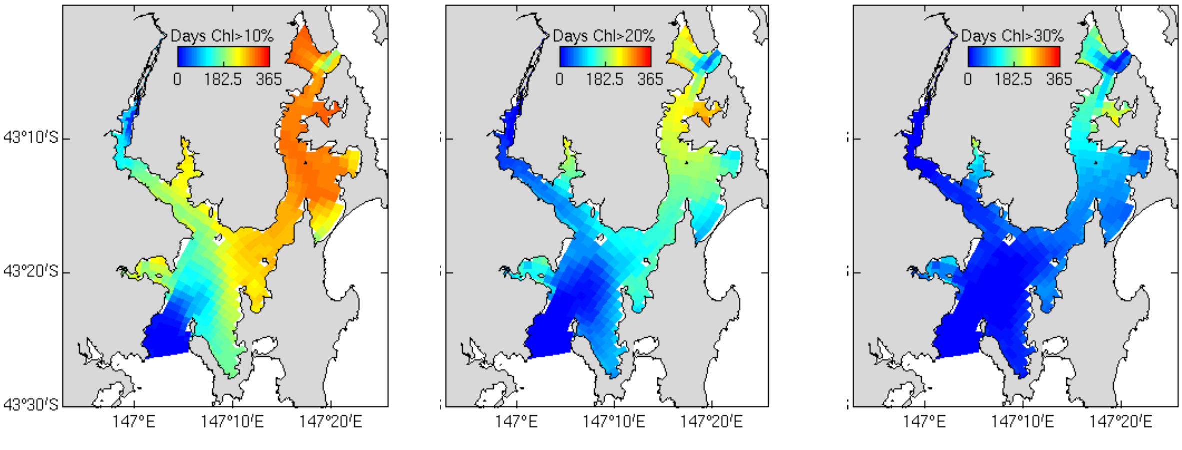

Figure 8.2. The number of days in the year when model simulated chlorophyll-*a* with fish farm inputs exceeded threshold levels of 10, 20 and 30% excess over background levels (estimated from the model without fish farm inputs, 2002 data).

8.5.2. Quantifying the impact of stress

Significant impacts from fish farming were summarised into % of regional area and duration (>1, >3 and >6 months) where mean depth-integrated labile nitrogen, phosphorus, chlorophyll exceeded, and bottom water oxygen saturation was depleted, relative to background levels. This showed that 27% of the region experienced a >10% increase in labile nitrogen for more than 6 months of the year due to fish farming activity, and 68% of the region had a >10% increase in chlorophyll-*a* for more than 6 months. Details for other parameters are in Appendix 4 in Thompson et al., 2008.

Table 8.2. Proportion of region (as % of area) where daily mean depth-integrated labile nitrogen, phosphorus, chlorophyll-*a* exceed thresholds of 10, 20 and 30% over background levels and bottom water oxygen saturation is less than thresholds of 1, 5 and 10% compared with background level (2002 data). Other parameters are in Appendix 4 in Thompson et al. (2008).

Depth-integrated chlorophyll-*a*	Chl > 10%	Chl > 20%	Chl > 30%
> 1 month	92.5	82.1	53.3
> 3 months	88.4	56.0	19.1
> 6 months	68.0	20.1	3.6

8.5.3. Susceptible sites

Annual means of daily differences, between model simulations with and without fish farm inputs were used to locate the places with smallest and largest fish farm impact (on average over the whole year). Maps of ranked location were calculated for depth-integrated labile nitrogen, phosphorus, chlorophyll and bottom water oxygen saturation, and summed to find the sites most frequently impacted for all substances. These are the best places to detect the biogeochemical impacts of fish farms over the course of a year (Fig. 8.3).

The location of stations most likely to detect fish farm impacts is consistent with our understanding of the residual regional circulation which generally flows from south to north by the estuarine circulation of the Huon (see Fig. 4.8 in Appendix 4 of Thompson et al., 2008) and would facilitate accumulation of farm impacts in the Huon and north D'Entrecasteaux Channel. The exact location of stations is optimal for the year 2002, but the results may be generalised to other years with similar farm loads and locations experiencing similar meteorological and river flow conditions.

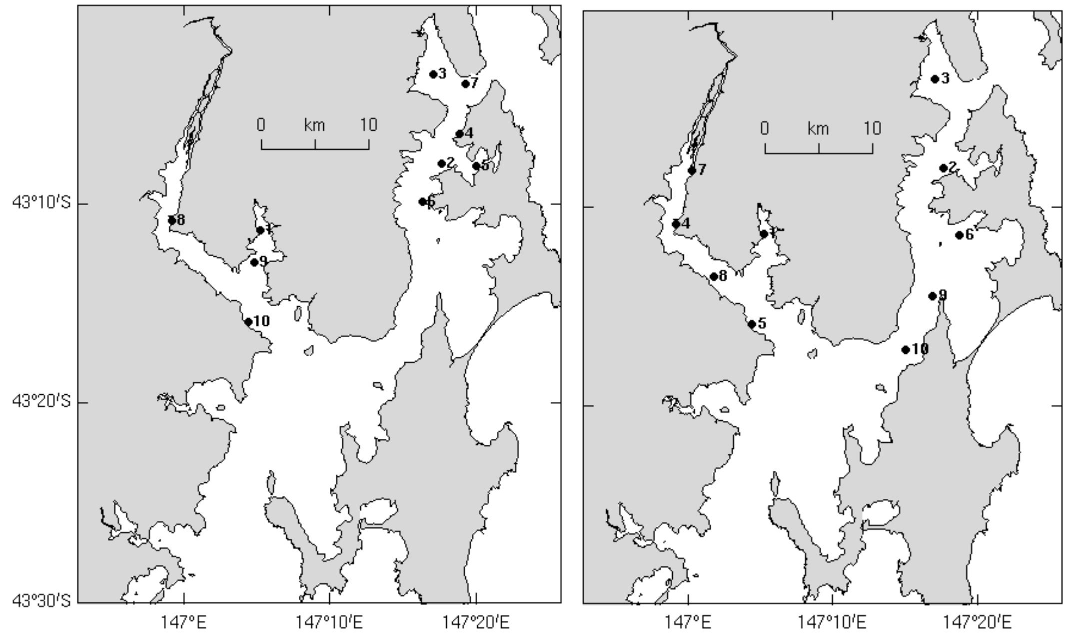

Figure 8.3. The 10 most likely places to detect impacts of fish farm loads on pelagic labile nitrogen, phosphorus, chlorophyll-*a* and bottom water oxygen saturation with stations > 2.5 km apart (left) and > 5 km apart (right) based on 2002 data.

8.6. Water column (chlorophyll-*a*, phytoplankton, DO, nutrients)

Based upon our assessment (Appendix 5 in Thompson et al., 2008) we propose that water column parameters (phytoplankton biomass, nutrients and DO) should be measured ~ monthly and sites should be located ≥4 km apart if well connected. Measurements more frequently or closer together are not fully independent (Appendix 5 in Thompson et al., 2008). We propose a transect along the centre of the channel in the Huon Estuary (3 sites) and similarly 5 sites in D'Entrecasteaux Channel (Fig. 8.4) rather than select sites at random (Jassby et al., 1997). If the monitoring program is to rely upon modelling to assist with attribution of cause and effect there is a need to have sites at the boundaries where most of the non-anthropogenic dissolved inorganic nutrients enter the ecosystem. We further recommend that sites defined as susceptible (Appendix 4 in Thompson et al., 2008) be monitored. Combining these approaches and deleting the overlapping sites gives a total of 14 sites (8 mid-channel plus 6 bays). We propose that chlorophyll-*a* and phytoplankton should be sampled as integrated samples over a reasonable fraction of the euphotic depth. For comparison with previous data a 12 m long tube could be used. If the water column is shallower than 12 m then a sample from the surface to 1 m above the bottom is recommended. Phytoplankton species counts are proposed to be sampled at the same spatial and temporal frequency as phytoplankton biomass (Table 8.3).

DO and nutrients should be sampled at the surface and 1 m above the bottom. Additional sampling from TSQAP and any farm site that can provide routine measurements from a defined location at a suitable depth should be included. We note that surface samples for DO are the least valuable and a standard depth of 3–5 m could be established (many farms have a 5 m cable on their DO meter). While this seems like a small number of sites, we note that Strobel et al. (2000) concluded that 5 sites per estuary were adequate for comparisons across multiple estuaries.

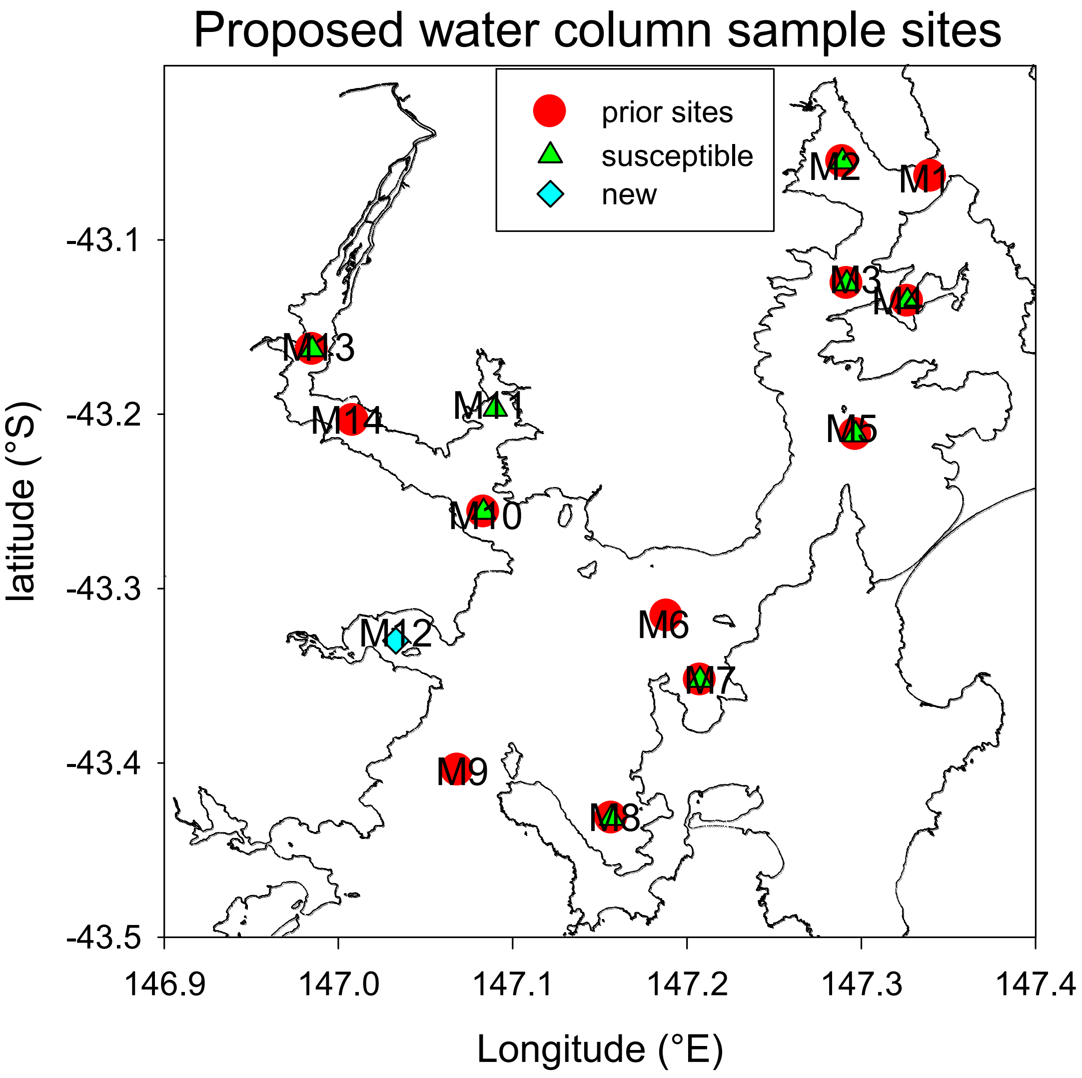

Figure 8.4. Proposed water column (pelagic) sample sites for DO, chlorophyll-*a*, nutrients and phytoplankton. Sites are classified as prior (some baseline data are available) or susceptible based on the modelling (Appendix 4 in Thompson et al., 2008) or needed to define the boundary conditions for future modelling.

Table 8.3. Proposed monitoring program parameters, spatial and temporal frequency (REC = recommended, OPT = optional, but will be useful).

Parameter	Temporal	Spatial	Measurement
Sediment chemistry	annual	14 sites (see Appendix 6 in Thompson et al., 2008)	Redox potential organic content stable isotopes particle size sulfide concentration metal concentrations antibiotics
Benthic infauna biota	annual	14 sites (see Appendix 6 in Thompson et al., 2008)	Multivariate assessment of infaunal species abundance
Other benthic biota	Annual	1 site	Seagrass shoot density % chlorophytes
Water column nutrients	REC: Monthly in water column. OPT: Continuous at station 12 by correlation with temperature.	At 8 mid channel ~ 10 km apart + 6 bays, surface and + 1 m above bottom. Single mooring with CTD at model boundary (OPT).	NH_4, NO_3, PO_4, Si, TN and TP
Phytoplankton community	REC: Monthly OPT: more frequently during major blooms or at farm sites.	As for nutrients but integrated over depth, 8 sites mid channel + 6 bays.	HPLC pigments and cell counts
Phytoplankton biomass	REC: Monthly, integrated sample	As for nutrients but integrated over depths, 8 mid channel + 6 bays.	Chlorophyll-*a*
Water column DO	REC: Monthly in summer OPT: monthly year round, weekly at farms sites throughout summer or year round.	As for nutrients, 8 sites down centre of Huon Estuary and D'Entrecasteaux Channel + 6 bays, bottom only. At farms, one consistent point and depth ≥ 5 m.	Dissolved oxygen in mg l^{-1}, need temperature and salinity to calculate percent saturation (pressure and altitude effects to be ignored).

More spatial sampling will improve the precision of any measurement and may reduce overall costs due to a lower frequency of exceeding of trigger values.

8.7. Benthic sampling and sediments

Localized monitoring of sediments already exists and is adequate for its purpose (Appendix 6 in Thompson et al., 2008). The more challenging problem of how to detect a system-wide change in sediment characteristics that could be attributed to any particular influence has not been resolved. A review of the local, national and international literature plus a detailed assessment of the extensive local data has yielded no parameter that could be confidently used as an indicator across the Huon Estuary and D'Entrecasteaux Channel and therefore turned into a suitable performance measure (Appendix 6 in Thompson et al., 2008). Many indicators were considered and their potential use as a performance measure was assessed but they were rejected. The reasons for rejection varied but a large number of potential indicators suffered from high spatial variability leading to low statistical power and therefore required a very large number of samples and was considered not suitable for a system-wide monitoring program.

Very different biological communities and sediment characteristics were observed at a range of spatial scales (Appendix 6 in Thompson et al., 2008), sometimes associated with spatial heterogeneity in bottom type but making the definition of a system-wide monitoring strategy problematic. However, the potential consequences associated with a significant impact on the benthic ecosystem are of sufficient concern that some monitoring of the environment is warranted.

It is recommended that there should be sampling of benthic infauna, sediment chemistry and benthic macrophytes in areas deemed susceptible to fish farming impacts. In the proposed plan (Appendix 6 in Thompson et al., 2008), the majority of these sites are consistent with existing farm monitoring programs. In addition it is concluded that there is a need for further research to determine a suitable suite of benthic indicator(s) that can be translated into a range of performance measures and used to support sustainable management of the resource at a regional scale.

Based on the BGC and hydrodynamic modelling, which shows some regions are more likely to be impacted than others, benthic monitoring for sediment chemistry and benthic infauna is proposed for the same 14 sites in the Huon Estuary and D'Entrecasteaux Channel and one control site in Recherche Bay. These are to be sampled once per year. In addition a program of monitoring seagrass and benthic macrophytes at one *sentinel* location and at a control site is proposed.

8.8. Preferred techniques

With regard to sampling and analytical methodology we have limited our recommendations. As long as appropriate quality control and quality assurance (QA&QC) measures are adopted a range of techniques are likely to be suitable. There is a need for data quality to be high if the ecosystem is to be managed properly. For this reason some form of QA&QC should be undertaken as part of any monitoring program. We advocate for QA&QC procedures to be implemented at the outset of any monitoring. The quality control processes should be adequate to determine whether the data obtained for each parameter have a suitable level of precision and accuracy. Guidelines for QA&QC are provided (Appendix 10 in Thompson et al., 2008). We further note that sampling methodology can impact on data quality, so that to ensure high quality and uniform sampling we recommend a single, well-trained person or agency conduct as much of the sampling as possible. A range of published methodologies is available in the literature for the analysis of samples for all proposed parameters. We do not propose to be prescriptive regarding analytical methods but have provided some guidance with regard to those methods known to be satisfactory for samples from the local environment (Appendix 10 in Thompson et al., 2008).

We acknowledge the lack of independent control sites with which to compare the Huon Estuary and D'Entrecasteaux Channel, but the furthest southward site off Brown's Point (Fig. 8.4, site M9), the Derwent Estuary and Recherche Bay may provide useful comparisons in terms of assessing temporal trends. An ongoing comparison of several water bodies will improve any effort to attribute cause should one of the proposed indicators show deterioration over time. The observation of simultaneous and similar changes in several ecosystems relative to their own baseline may indicate a larger scale regional pattern of change not associated with local stress or use.

We note that, should other existing monitoring programs continue, there is scope for reducing costs and (potentially) improving data quality by combining this monitoring with existing programs.

8.9. Converting indicators into performance measures

Having reviewed the local, state, national and international literature on ecological indicators and assessed these for their usefulness in the Huon Estuary and D'Entrecasteaux Channel we considered how the selected indicators might be converted into *performance measures* that regulators, industry and others may use to assess any changes to the state of the environment. To be useful as a performance measure any indicator must be measured in a defined manner and converted into some sort of statistic. There are many statistics that could be used as a performance measure ranging from a single measurement at a single site to the central tendency (e.g. mean, median or mode) from many measurements at different times at many sites. The general issues that pertain to sampling and that are relevant to performance measures plus trigger levels are discussed in the next paragraph.

Indicators that make good performance measures are ecosystem characteristics that are relatively stable under normal conditions, that is, in the absence of perturbation or pressure. When a performance measure exceeds a defined threshold (a trigger level) then some action is required. The trigger level must be defined as some degree of change relative to the normal, or mean, condition. The degree of change cannot be so large that the ecosystem is profoundly different before action is taken. Because water quality (WQ) parameters (e.g. chlorophyll-*a,* dissolved nutrients and DO) are highly variable in time and space single measurements are not good indicators of ecosystem status. Typically it is necessary to average over multiple measurements and it is appropriate that monitoring sites be distributed throughout the region of interest. More sampling increases the confidence of detecting whether a real change has occurred. In the Huon Estuary and D'Entrecasteaux Channel we know that there are significant variations in WQ parameters between seasons and between locations (Appendix 5 in Thompson et al., 2008). We have predicted that changes in WQ will not be uniformly distributed over this region with some bays predicted to be at greater risk (Appendix 4 in Thompson et al., 2008). Given the need for sites to be ~ 10 km apart to be statistically independent we recommend that 8 mid-channel sites and 6 bays be sampled to characterize any system-wide change over time. Using 14 sites significantly improves the precision of the estimated mean WQ indicator. It should also be possible to investigate change in bays (n = 6) or channels separately.

Designing a temporal sampling strategy is complex. There is a natural and large seasonal dynamic, as well as some interannual variation, in WQ indicators. Observations and modelling both suggest that summer and autumn are periods when potential impacts due to anthropogenic pressures are likely to be manifest. A monitoring program designed solely to detect change due to anthropogenic pressure could reasonably sample only throughout summer and autumn. The major difficulty with such a program is the inability to interpret the ecosystem's behaviour should a problem develop. Consider, for example, that a particular WQ indicator exceeds its trigger value during summer monitoring. Any effort to understand why this occurred, whether undertaken by an expert or using models, will fail unless there are sufficient data available to quantify the large seasonal and interannual variability. Accordingly, performance measures and trigger values are based upon observations obtained during summer while WQ monitoring is recommended to continue year round.

8.10. Setting trigger levels

There are many options for setting trigger levels. The ANZECC (2000) guidelines refer to triggers as: "*A trigger for further investigation will be deemed to have occurred when the median concentration of n independent samples taken at a test site exceeds the eightieth percentile of the same indicator at a suitably chosen reference site. Where suitable reference site data do not exist, the comparison should be with the relevant guideline value published in this document*".

While completely valid, this approach fails to consider the high seasonal variability that occurs in Tasmania for most of the WC parameters proposed for use here. It also defines the need for some action based upon a single high value. Furthermore it requires comparison with a reference site which is not deemed feasible in this instance. For the water column parameters the proposed trigger values are based upon a comparison of seasonal or annual means across years. Given the proposed sampling regime(s) this is likely to represent a statistically robust measurement and a conservative trigger level. The latter portion of the recommendation quoted above (using ANZECC guidelines) cannot be used here as the guidelines do not have values for most WQ parameters for Tasmania, although values are given for SE Australia. Trigger values can also be set using relative change or absolute change. For example, changes in the dissolved oxygen concentrations might have a trigger level of a change in annual mean concentration by 10% (i.e. a trigger level relative to baseline) or no observations less than 5 ppm persisting for more than 1 month. Both types of trigger values have merit, the former trigger level acknowledges some change is acceptable and stipulates how much can occur before some action is required. The latter trigger level (5 ppm) is an absolute concentration where values below are known to harm some organisms that normally inhabit the ecosystem. Where appropriate both absolute and relative trigger values are specified.

Table 8.4. Proposed trigger levels for various parameters with 3 levels of trigger values

Parameter	Standard or Baseline value[1]		Level 1	Level 2	Level 3
	Huon Estuary	D'Entrecasteaux Channel	(low risk)	(moderate risk)	(high risk)
Sediment biota (infauna)	To be determined TBD	To be determined TBD	Significant change over time since start of assessments at one or more sites + other indicators TBD	Significant change in multivariate community structure at 1 site since last assessment + other indicators TBD	Significant change in multivariate community structure at ≥ 2 or more locations since last assessment + other indicators TBD
Seagrass and other macrophytes	TBD	TBD	Significant change over time or relative to control site.	As level 1 + TBD	As level 1 +TBD
Sediment chemistry	ANZECC guidelines for metals and TBD	ANZECC guidelines for metals and TBD	Significant change over time at one site. TBD	Significant change at 2 sites in ≥ 2 indicators. Exceeds ANZECC guidelines for low metal concentrations. TBD	Significant change at ≥ 3 sites in ≥ 2 indicators. Exceeds ANZECC guidelines for high metal concentrations. TBD
Nutrients	Summer NH_4^+ surface = 0.32 µM Bottom = 0.42 µM	Summer NH_4^+ surface = 0.12 µM Bottom = 0.27 µM	summer mean up 25%, or 3 successive annual means > baseline, or mean for any one site +50%	summer mean up 50%, or 8/10 annual means > baseline for any site, or mean for any single site up 200%	Summer mean +100%, Or summer mean > 1 µM (~ ANZECC)
Chlorophyll-*a*	sites 10 to 14 annual = 1.4 $\mu g\ l^{-1}$ summer = 1.7 $\mu g\ l^{-1}$	Sites 1 to 9 summer mean = 0.66 $\mu g\ l^{-1}$ annual mean = 0.80 $\mu g\ l^{-1}$	Any site[5]: annual mean +100%; or Average summer mean +50%	Any site[5]: annual mean +200%; or Average summer mean +100%; or Average annual mean +50%	Any site[5]: annual mean +400%; or Average summer mean +200%; or Average annual mean +100%
Phytoplankton blooms	7% obs. > 3x median chla	3.6% obs. > 3x median chla	% obs. > 3x median rise 50%	% obs. > 3x median rise 100%	% obs. > 3x median rise 200%
Harmful Phytoplankton *G. catenatum*[4]	TSQAP data from 7 areas 1997–2007	TSQAP data from 7 areas 1997–2007	# of days 7 areas are closed to shellfish harvest due to HAB >226 days	# of days 7 areas are closed to shellfish harvest due to HAB >336 days	Not defined
Absolute DO[3]	Channel mean > 6 ppm Bay mean > 5 ppm	Channel mean > 6 ppm Bay mean > 5 ppm	Any 2 channel observations ≤ 6 ppm Any 2 bay observations ≤ 5 ppm	50% of channel observations ≤ 6 ppm 50% of bay observations ≤ 5 ppm Any 2 observations < 2 ppm	Channel mean ≤ 6 ppm Bay mean ≤ 5 ppm Any 2 measurements < 1 ppm
Relative DO (percent saturation)	Set at 20th percentile from 1st year of observations	Set at 20th percentile from 1st year of observations	[2]Number of observations below baseline increases 50%	[2]Number of observations below baseline increases 100%	Mean falls 10% from baseline (~ ANZECC)

[1] If not specified then the first year will define '*baseline*' condition.
[2] Conditional, may need further refinement.
[3] sample locations avoid known and localized areas of low DO (Appendix 5 in Thompson et al., 2008), trigger values may need to be adjusted for temperature if ecosystem warms significantly.
[4] see Appendix 9 in Thompson et al. (2008) for full details.
[5] site means are in Table 5.5 in Appendix 5 of the technical report by Thompson et al. (2008).
TBD = to be determined

8.11. Response levels

Following from Fletcher (2000) we recommend a three-level, staged-approach (Table 8.5) to dealing with deleterious changes in ecosystem functioning as indicated by parameters exceeding their trigger values (Table 8.4). Very simply, the three levels represent our opinion regarding the relative risk to ecological functioning represented by the increasing temporal or spatial extent of ecological conditions that are increasingly above the baseline. A similar approach but slightly different responses are proposed depending upon whether the parameter exceeding its trigger level is from the water column or from the sediments (pelagic or benthic). In this report we have included some possible responses to exceeding the trigger levels, but we acknowledge the details require refining in consultation with stakeholders. In particular deciding the responses for each level of risk is the responsibility of the management authority.

While acknowledging the limits of our current understanding of marine ecosystems, and this ecosystem in particular, the level 3 trigger levels have been selected to represent a major negative change in ecosystem function. As proposed, Level 3 responses require an increase in investigative effort in terms of either data analysis and interpretation of the already collected data, or more data collection to determine the spatial extend of the deterioration. It is our opinion that, if the ecosystem should really exceed trigger level 3, a significant increase in the investment of resources into understanding the underlying causes would be warranted. There would also be a need for increased activity by government regulators and the engagement of other stakeholders. We anticipate that if the ecosystem is to recover from level 3 the stress must be reduced or some other remedial action implemented.

On the assumption that the ecosystem continues to be heavily used a full review of monitoring data (a level 2 response) is recommended at least every 5 years regardless of whether the data collected by the monitoring exceed any trigger values.

Table 8.5. Proposed responses when parameters exceed trigger levels as identified in Table 8.4.

Response level	Water column	Benthic sediment chemistry and infauna algal macrophytes and seagrass
1 (low risk)	Add more spatial and/or temporal sampling.	Resample affected site/s ASAP. Increase spatial sampling to evaluate extent of change.
2 (moderate risk)	Commission expert review of the monitoring data, the monitoring strategy and a model run[1] as part of a report to government, industry and other stakeholders. Review to make management recommendations.	Resample affected site ASAP. Extend sampling to find limit of change for infauna and sediment chemistry. Expert review of dataset to establish appropriate follow up sampling program and to provide management recommendations. TBD[2] for seagrass and algal macrophytes.
3 (high risk)	Commission a study of the cause, to include more field observations and new modelling if deemed necessary. Formal expert report to government, industry and stakeholders (management intervention and remedial action may be necessary).	Resample affected sites ASAP. Extend sampling to find limit of change for infauna and sediment chemistry. Expert review of dataset to establish appropriate follow up sampling program and to provide management recommendations TBD[2] for seagrass and algal macrophytes. Formal, expert report to government industry and stakeholders (management intervention and remedial action may be needed)

[1] The existing biogeochemical (BGC) model predicts the effects of added nutrient loads to the Huon Estuary and D'Entrecasteaux Channel and therefore can be useful in partitioning attribution. We note the existing model is coupled to a hydrodynamic model derived from a limited temporal period (2002) and therefore the exact location of a particular ecosystem response may not be accurate in other years. Given the potential for significant interannual variation in hydrodynamics and two open boundaries in the model domain there will be some uncertainty about the BGC model output without updating the hydrodynamics. Regardless we consider the BGC model is likely to give reasonable estimates of overall ecosystem responses in terms of chlorophyll-*a* as well as some indication of the possible causes. Therefore a BGC model run is recommended as part of the level 2 responses. Level 2 responses should include stakeholder involvement and may require remedial action. We note the possibility that the collective stakeholders may consider the change(s) in ecosystem status acceptable given the possible social and economic costs of any possible remedial action(s).

[2] TBD = to be determined

8.12. Summary

The D’Entrecasteaux Channel is naturally oligotrophic with strong trophic coupling and nutrient recycling processes that result in very few algal blooms. In contrast, the Huon Estuary is currently mesotrophic with increased primary production and some uncoupling between trophic levels resulting in increased frequency and density of algal blooms. The most significant threats to the ecological functioning of the D’Entrecasteaux Channel and the Huon Estuary ecosystem are further increases of organic matter and nutrient inputs (eutrophication). The most common symptoms of eutrophication include increased nutrient concentrations, trophic decoupling, increased algal blooms, hypoxia, changes in nutrient cycling, reductions in water clarity, eventual loss of benthic macroflora, changes in food web structure and organic enrichment of the bottom waters and sediments, and/or changes in benthic fauna and microbial processes (Grey, 1992; EPA, 2003).

The parameters that might be used to monitor for these symptoms or effects of eutrophication were assessed for their suitability using criteria that define a ‘good’ indicator (Maher et al., 1994). Initially a very comprehensive list of parameters was considered but many were discarded as unsuitable. The final list of parameters is compatible with similar monitoring programs in Australia (e.g. ANZECC, 2000) and worldwide. The reduced list of parameters was subjected to risk assessment (after Gavine and McKinnon, 2002; Fletcher et al., 2004). The risk assessment considers both the likelihood (almost certain to none) and consequence (insignificant to catastrophic) of a change in ecosystem functioning, with the potential for harm used to estimate risk. Of all the parameters considered, low dissolved oxygen (DO) and changes in trophic structure were identified as ‘high risk’ and most in need of monitoring. Based on recent measurements in the Huon Estuary low DO concentrations are ‘likely’ (‘the event will probably occur ~ once per year’) and the consequences of low DO was considered ‘major’ (recovery to take many years). Therefore low dissolved oxygen is a high risk with three identifiable components: concentration, spatial extent and temporal extent. Changes in trophic coupling, manifested as algal blooms and changes in phytoplankton community composition, were also rated as a high risk and consequently they are an important component of the proposed ecological monitoring.

It is likely that existing or proposed ecosystem uses will have both positive and negative impacts on the local populations of various plants and animals. Many of these impacts would be difficult to quantify, given the natural variability of these populations over time and in space (low signal:noise ratio). Our major goal was to design a monitoring program with the capacity to detect the effects of those processes judged to be most threatening to Huon and D’Entrecasteaux ecosystem at the whole of ecosystem level. The monitoring program proposed here is designed to provide knowledge of how well the ecosystem is functioning with an increased nutrient load and to allow any significant temporal trend(s) to be detected. We believe that a major improvement in our proposed program, relative to more conventional programs, is the application of modelling to focus monitoring on the areas that are most susceptible to eutrophication. Within any monitoring program there is the potential for some localized impacts on ecosystem functioning or populations to be undetected. Very localized impacts are classified as ‘minor’ (Fletcher et al., 2004) and are not likely to cause a significant change to ecosystem functioning. One notable exception is the initial colonization by new invasive species. New arrivals such as *Gymnodinium catenatum*, *Noctiluca scintillans*, Northern Pacific sea star (*Asterias amurensis*), wakame (*Undaria pinnatifida*), and New Zealand screw shell (*Maoriculpus roseus*), have all had significant impacts on this ecosystem. Therefore any use of the ecosystem that facilitates colonization by pest species, particularly introduced pests, should be minimized. Very localized reductions in populations of many marine organisms are not considered a major risk to the ecology because of their broadcast spawning, high fecundity and lack of strongly differentiated genotypes at small spatial scales (Elliott et al., 2002). Along with localized changes there are two other types of environmental impacts not well monitored by the proposed plan. One of these is the possible ecological impacts of aquaculture on rare species. Users of any natural resource have an obligation to ensure their activities do not adversely impact on populations of plants or animals known to be endangered, threatened or protected by Local, State, Commonwealth or international law, treaty or convention. It is recommended that a separate review of possible impacts on endangered, threatened or protected species be undertaken. Secondly, the proposed plan does not monitor for impacts possibly caused by some other users of the marine environment including contaminant disposal (for example: CO_2, sewage, pesticides, herbicides, solid wastes, litter) and the relevant authorities may wish to consider expanding this or other monitoring programs to include these indicators.

A major challenge was the development of baselines and trigger values for the selected parameters; in order to convert the science into a usable management tool. A thorough review of existing data and high resolution biogeochemical modelling was used to estimate baselines and trigger values. The modelling was also used to optimise site selection reducing the cost of monitoring and increasing its sensitivity. The trigger values are performance measures that the ecosystem must meet. There is a series of trigger values designed around the proposed monitoring that incorporate the 3 dimensions of risk (intensity, duration and spatial extent). If the monitoring indicates the ecosystem does not meet a particular trigger level then a series of management actions is recommended. Extensive consultation with stakeholders in the regulatory and industrial sectors indicates uptake is very likely with implementation possible in 2008. We believe the major improvements in our proposed monitoring program include lower cost and more robust monitoring with an explicit incorporation of risk and direct links to recommended management actions.

8.13. References

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Chapter 9

Conclusions and Recommendations

9.1. Benefits and adoption

The project generated considerable knowledge of key processes in the ecosystem, and the possible environmental effects of salmonid aquaculture at an ecosystem level, of value to both industry and regulators, and leading to the development of the monitoring and modelling tools, which can also be applied to other fish species or to other regions.

The three-dimensional coupled hydrodynamical, sediment and biogeochemical model was used to evaluate the environmental impact of salmonid fish farms in the Huon Estuary and D'Entrecasteaux Channel.

Scenario simulations, parameterised without fish farm inputs, with fish farm inputs for 2002 and with projected inputs for 2009, allowed us to identify the system-wide spatial and temporal environmental footprint of the industry. Various statistical metrics were devised to highlight the spatial and temporal effects of this increased production. These results were communicated to the salmonid industry, the regulatory agency DPIW and the Marine Farming Planning Review Panel.

A long-term monitoring strategy for the industry in the D'Entrecasteaux Channel and Huon Estuary was developed to ensure the sustainability of the salmonid industry, having the capacity to detect the effects of those processes judged to be most threatening to Huon and D'Entrecasteaux ecosystem at the whole-of-ecosystem level. The monitoring program we have proposed is designed to provide knowledge of how well the ecosystem is functioning with an increased nutrient load and to allow any significant temporal trend(s) in ecological indicators to be detected.

Indicators of ecological condition were converted into recommended quantitative performance measures that can be used in a regulatory manner to adaptively manage the ecosystem. These recommendations have been extensively discussed with industry and Tasmanian DPIW representatives and refined through an iterative process. Agreement has been reached in principle about the major elements of the plan and work is underway to implement the strategy.

Near the end of the project a decision was made by the marine farming planning review board to impose a limit on the salmonid industry in the Huon and D'Entrecasteaux based on the industry's projections for salmon production in 2009. This decision was informed in part by the scenario of likely effects on phytoplankton populations produced by our project using the environmental modelling suite.

9.2. Further development

Process studies in the Huon River estuary and North West Bay demonstrate the rapid changes in phytoplankton dynamics that can occur due to changes in physical and chemical parameters over time scales of hours. Samples collected on a regular basis such as weekly in the previous Huon Estuary Study or monthly as in this project can really only be used to determine longer term trends or changes in such an ecosystem. To more fully understand dynamic systems such as the Huon Estuary and D'Entrecasteaux Channel, automated profiling instruments which operate at small time scales need to be utilized.

If the industry is to expand further it will either have to demonstrate that it is not having serious environmental impacts, or place new production outside of the Huon Estuary and D'Entrecasteaux Channel or find ways to reduce or ameliorate nutrient inputs. The possibility of "co-culture" or "polyculture" of organisms that take up nutrients (such as macroalgae or molluscs) is actively under consideration overseas and could be implemented in Tasmania. Since it is the whole-of-ecosystem effects that are important, the nutrient uptake need not occur immediately adjacent to the fish farm and so a range of options for sites are possible. The opportunity for developing new macroalgal or molluscs industries in the D'Entrecasteaux Channel needs to be assessed.

The option of expanding to the seaward side of Bruny Island is already under consideration by the industry. While the region off-shore Bruny Island is better flushed and larger in extent than the Huon and D'Entrecasteaux region, the same general environmental issues will apply and some form of environmental risk assessment will be needed. There is the added complication of connection with the Derwent estuary and any assessment of nutrient effects must take into account efforts already underway in the estuary to reduce inputs from sewage treatment plants. There is thus a need to consider industry expansion in the context of multiple use management of this ecosystem. A new program called INFORMD (Inshore Network for Observation and Regional Management: Derwent-Huon) may assist this.

The INFORMD program will augment existing research, planning and management in several ways: (1) delivering a 'state of the art' observation network of fixed and mobile sensors including genomic approaches to monitor key elements of the ecosystem in close to real-time, (2) providing an information management system to allow information from the new observation network and from existing monitoring systems to be integrated and mutually accessible, and (3) developing new models for data interpretation, prediction and comparison management options. A major objective is to provide support tools that make the information and predictions accessible and relevant to industry, planners, policy developers and managers.

9.3. Planned outcomes

The project has delivered fully the planned outcomes:

9.3.1. Project outcomes

The major outcome of this project will be a scientifically rigorous assessment of environmental issues associated with salmon farming in southern Tasmania. This information will be made available to all stakeholders and will provide the basis for ensuring the long-term sustainability of the industry and an underpinning resource for decisions relating to industry expansion.

This project will provide salmon farmers greater knowledge and understanding of the marine environment in which they have farms, with an emphasis on key aspects of the environment including phytoplankton and jellyfish blooms, and water quality attributes such as dissolved oxygen and nutrient status.

Both farmers and regulators will benefit from the development and implementation of an adaptive management strategy and associated monitoring regime addressing the assimilative capacity of salmonid aquaculture in the Huon and D'Entrecasteaux Channel. This will be both less expensive and more informative than the present monitoring regime.

Benefits will also flow to the tuna industry in South Australia and other finfish industries elsewhere through extension activities and joint research with other research providers (this was achieved through CRC project 4.6, FRDC 2005-059).

In particular, the project has:

- established a detailed set of environmental data that provides a good picture of the environmental conditions in the Huon Estuary and D'Entrecasteaux Channel, where two-thirds of Tasmania's salmonid industry is located. These show generally good environmental conditions with occasional periods of high phytoplankton abundance and low dissolved oxygen.

- used these environmental data to calibrate and validate sophisticated 3-dimensional hydrodynamic and biogeochemical models of the region that capture the main physical and biological processes.

- carried out specific process studies in sub-regions of the domain to shed more light on particular aspects of the nutrient-phytoplankton-zooplankton relationships and sediment biogeochemistry.

- demonstrated from measurements and modelling results indicate that the salmonid industry is a significant contributor of nutrients to this region and that these have led to measureable increases in phytoplankton abundance.

- demonstrated that there is a limit to the extent that industry can expand without having deleterious environmental effects. Information from this project informed a decision by the Marine Farming Planning Review Panel to limit industry expansion in the Huon Estuary and D'Entrecasteaux Channel to a feed input projected by industry to be reached by 2009-2010.

9.3.2. Commonwealth Agreement outcomes:

- An ability to predict the environmental impact of cage aquaculture at the system-wide (e.g. estuary) scale
- Improved monitoring of the environmental performance of cage aquaculture operations
- Better community understanding of aquaculture and the environment

9.4. Summary of results from the major components of the study

9.4.1. Hydrodynamics

The Huon River Estuary is a microtidal salt-wedge estuary with a spring tidal range of about 1 m. Flushing times for the entire estuary have been estimated at 5.6 days during low flow and 2.5 days during high flow. The surface layer within the entire estuary is flushed more rapidly at every 1.3 days during low flow and every 0.6 days during high flow.

The hydrodynamic modelling approach used a series of embedded models where the high resolution model for the Huon Estuary and D'Entrecasteaux Channel took its boundary conditions from an intermediate model, which, in turn, was nested in a larger regional model that was driven with global products derived from global models or altimetry and climatology on its open boundaries. The intermediate hydrodynamic model was calibrated to temperature and salinity data collected in the field at the northern and southern entrances to D'Entrecasteaux Channel and the mouth of the Derwent Estuary during December 2003 to March 2004. The modelled temperature and salinity compared favourably to measured data. This configuration was then used to simulate a longer period from 2002 to 2005 inclusive. The data collected in the first phase of the project was used to validate the coarse resolution local model. Model data comparisons were comparable to the comparisons using measurement-forced coarse models. This indicates the nesting process using the calibrated intermediate model in conjunction with global products is suitable to create accurate model output for any simulation periods, without the requirement of collecting open boundary forcing data from the field. The coarse local model for the long term simulation is suitable for coupling with biogeochemical and sediment transport modules to investigate the longer term trends of environmental indicators in response to aquaculture farming practices.

The 4-year simulation using the larger scale models was used to assess the transport of nitrate into D'Entrecasteaux Channel. A seasonal cycle of nitrate flux into southern D'Entrecasteaux was observed, with maximum flux of ~200 T per month in spring, and minimum fluxes in late summer and early autumn. The increased fluxes during spring are due to the intrusion of high-nitrate water from depth onto the slope and shelf, which is then available for transport into the coastal zone *via* local processes. The up-slope intrusion is hypothesized to be due to the continental slope interaction of large scale currents resulting from a persistent anticyclonic gyre stationed off south-east Tasmania in spring, producing upwelling in the bottom boundary layer. Local uplift of high nitrate water on the slope and shelf is possible during March and April, confined to the southern Tasmanian coast and southern tip of Bruny Island. This also increases the flux of nitrate into southern D'Entrecasteaux. This uplift is hypothesized to be the result of onshore flow in the bottom boundary layer over the shelf and slope due to the passage of the East Australian Current (EAC) over the slope south of Tasmania. The EAC exhibits considerable variability in its passage down the east coast of Tasmania, and the upwelling events of this nature probably only coincide with strong EAC intrusions into southern Tasmanian waters during February to April. It is only during these times that there is the possibility of the EAC interacting with the shelf south of Tasmania; at other times of the year the shelf is dominated by eastward transport of water in the Zeehan Current.

High resolution hydrodynamic models were developed to investigate the mixing zone characteristics around a fish farm cage. These analyses revealed that a continuous release of tracer from the cage results in a plume emanating from the cage in the form of a long narrow ribbon. The position and concentration of this plume is dependent on the *in situ* flow conditions and exhibits a high degree of variability in space and time. The modelling suggests that on occasion plumes with concentrations up to 10% (i.e. 10:1 dilution) that of the source could exist up to 0.5 km from the cage. Models of instantaneous releases of tracer from the cage site show that the tracer is transported with the flow away from the cage in the form of a coherent pool, rather than simple diffusion around the cage site. This pool slowly diffuses horizontally at a rate that is dependent on the flow conditions at the time, hence subject to considerable variability. Sub-surface currents often determine the trajectory of this pool and the surface signature of the pool may move in the opposite direction to the surface currents.

Particle tracking was used to investigate the ‘footprint’, or spatial distribution of particles in the sediment, for various settling velocities of particles. For settling rates typical of feed pellets or faecal material, e.g. of the order of 10 $cm.s^{-1}$, the footprint is restricted to the cage site. However, a settling rate of an order of magnitude less than this results in footprints in the form of an elongated plume with typical length ~0.5 km and width ~200 m emanating from the cage site.

9.4.2. Phytoplankton and zooplankton ecology

During the course of this project major changes in the plankton ecology have been observed, including the establishment in 2002 of the heterotrophic dinoflagellate *Noctiluca scintillans* as a major herbivore in this ecosystem. *N. scintillans* is believed to have arrived in Tasmania in association with a strengthening of the East Australian Current. The dominant phytoplankton species in the ecosystem changed dramatically from ~1997 to ~2003 with significant increases in dinoflagellates of the genus *Ceratium*, small flagellates and species of the diatom genus *Skeletonema.* The abundance of the dominant toxic dinoflagellate, *Gymnodinium catenatum*, has fluctuated enormously. In conjunction with shellfish farm closure data from the Tasmanian Shellfish Quality Assurance Program our measurements of *G. catenatum* indicate population lows in 1997 and 2006 with greater abundances during 2003–2005.

There is good evidence that *N. scintillans* effectively feeds on *G. catenatum*, with the potential to influence the intensity and extent of blooms. Studies of the dynamics of *G. catenatum* carried out to fine tune the parameterisation of the biogeochemical modelling demonstrate for the first time the involvement of particular bacteria, *Alcanivorax* and *Marinobacter* spp., for growth and bloom development. Studies of the mortality of the blooms show a slower decline of autumn blooms compared with summer blooms. Molecular probes developed for detecting the algal-bacterial interactions have the potential for development as new monitoring tools for bloom detection.

Process studies in the Huon Estuary and North West Bay demonstrate the rapid changes in phytoplankton dynamics that can occur due to changes in physical and chemical parameters over time scales of hours. Over a 24-hour period during April 2005, changes in weather conditions determined the position of nutrient-replete surface waters thereby changing the dominant algal group within the upper water column. Samples collected on a regular basis such as weekly in the previous Huon Estuary Study or monthly as in this project can really only be used to determine longer term trends or changes in such an ecosystem. To more fully understand dynamic systems such as the Huon Estuary and D'Entrecasteaux Channel, automated profiling instruments which operate at small time scales need to be utilized.

Mesozooplankton community composition and structure were examined throughout the Channel, Huon and North West Bay from November 2004 to October 2005. The composition of the mesozooplankton community was typical of inshore, temperate marine habitats, with seasonally higher abundance in summer and autumn and lower numbers in winter and spring. Copepods were the largest contributors to total abundance across all seasons and stations, while cladocerans and appendicularians were proportionally abundant in spring and summer. The faecal pellets of these three main groups, along with those of krill and amphipods, contributed significantly to material recovered from sediment traps. Meroplanktonic larvae of benthic animals showed short-term peaks in abundance and were often absent from the water column for long periods. Spatially, North West Bay and the Channel had a higher representation of typically marine species, including *Calanus australis* and *Labidocera cervi*, while truly estuarine species, such as the copepod *Gladioferens pectinatus*, were more important in the Huon Estuary.

The grazing impacts of microzooplankton and mesozooplankton on the phytoplankton communities were examined during several process studies. Microzooplankton grazers included high proportions of ciliates, heterotrophic dinoflagellates and tintinnids, with smaller numbers of copepod nauplii, appendicularians and rotifers. Studies of mesozooplankton grazers were restricted to dominant omnivorous copepods (e.g. *Acartia tranteri, Paracalanus indicus*, *Centropages australiensis*) cladocerans and appendicularians. Grazing rates of microzooplankton reached as high as 96% of daily primary production, while that of mesozooplankton herbivores was never greater than 20%. Overall, microzooplankton are a major control on phytoplankton populations while the large mesozooplankton play a major role in structuring the food webs of the Huon and D'Entrecasteaux. Trophic interactions between the species are complicated by the recent arrival of the heterotrophic dinoflagellate *Noctiluca scintillans* to the region. This species grazes heavily on phytoplankton, smaller zooplankton and faecal pellets. *N. scintillans* accounted for up to 20% of mesozooplankton abundance in autumn and it is capable of both suppressing zooplankton abundance and reducing the sedimentation of faecal pellets to the seafloor.

9.4.3. Sediment ecology and geochemistry

Habitat and sediment types and benthic infaunal community structure have been mapped throughout the Channel, Huon and North West Bay. Faunal populations were broadly similar throughout the main part of D'Entrecasteaux Channel and lower Huon Estuary. Although communities associated with lower salinity sites could be broadly distinguished from those at fully marine sites, community distinctions were not specifically correlated with differences in depth or sediment type, either individually or in combination. Mesocosm experiments showed that sediment respiration rates differed markedly between regions within the Huon and D'Entrecasteaux systems. Sites in the lower Channel showed similar responses, with relatively low respiration rates and no evidence of microphytobenthic activity, whilst sites in the Upper Channel appeared more biologically active, with greater oxygen draw-down over time suggesting that sites in the Upper Channel may have a greater assimilative capacity than sites in the lower Channel.

Benthic fluxes of nutrients were low in the Huon Estuary. Average effluxes of ammonia, nitrate, phosphate and silicate were 1.3, 10.1, 1.6 and 32.5 μmol m^{-2} h^{-1} respectively. An extrapolation of these measurements to the whole estuary revealed that the sediments were only a minor source of nutrients, providing approximately 96 tonnes of inorganic nitrogen, 32 tonnes of phosphate and 586 tonnes of silicate.

On all occasions, the DIN efflux was dominated by nitrate, which was always released from the sediment to the overlying water. Aerobic conditions were prevalent in the upper 3–9 mm for the sediment, providing an environment conducive for aerobic processes like nitrification. Peaks of nitrate within the aerobic zone of the sediment observed from nitrate pore water profiles also provided additional evidence that nitrate was being produced in the oxic zone, most likely due to nitrification. All these observations suggest that the effluxes of nitrate and influxes of nitrite and ammonium in Huon Estuary sediments are associated with intensive nitrification stimulated by the presence of relatively deep oxygenated zones. The net efflux of nitrate from the sediments suggests that they act as net regenerators of nitrogen as opposed to nitrogen assimilators.

Organic loading of these sediments with labile carbon has shown that the nitrification-denitrification pathway is susceptible to being overwhelmed, due to increased respiration and reduction in the pool of oxygen in the sediments. This results in large amounts of ammonium being released into the water column and therefore makes the estuary more susceptible to blooms of algae, of which toxic dinoflagellates such as *G. catenatum* are especially important. Therefore maintenance of 'healthy' sediments is seen as important to the long-term sustainable use of the estuary.

9.4.4. Alternative measures of eutrophication

A number of alternative measurement strategies for eutrophication were examined during the course of the project including the rate of fouling of ropes and plates suspended in the water column. There were marked differences in fouling communities between plates and ropes, leading to the conclusion that ropes would not be a useful monitoring approach. We also found different communities between sites, strong temporal (seasonal) differences, increased numbers of species in summer, and an increased number of individuals and species at aquaculture sites relative to other sites at winter sampling. *Ulva* was a common algal fouling species at all times of year and consequently this led to an investigation of its growth as a measure of nutrient production. However there were difficulties associated with artificially propagating settlement and changes in gross growth did not necessarily reflect nutrient uptake.

We also carried out experiments to identify the capacity of *Ulva* to remove nutrients under local environmental conditions. *Ulva* has a relatively high nutrient removal potential relative to many other algal species. There did not appear to be any threshold nutrient level for assimilation, although nutrient assimilation capacity was reduced at high light levels. These findings suggest that under certain conditions this alga may be useful to compensate for periods of high nutrient inputs, but that it is unlikely to be a major mitigation strategy, unless there was a market for production, as the amount of algae needed to offset farm inputs would be considerable.

9.4.5. Biogeochemical model

A three-dimensional coupled hydrodynamic, sediment and biogeochemical model was applied to inshore waters of southern Tasmania to evaluate the environmental impact of salmonid fish farms in the Huon Estuary and D'Entrecasteaux Channel. The model simulates the seasonal cycling of organic and inorganic carbon, nitrogen, phosphorus and oxygen through multiple phytoplankton, zooplankton, nutrient and detrital pools and was validated against observations collected in 2002. Model analysis of scenario simulations, without fish farm inputs, with fish farm inputs for 2002 and with projected inputs for 2009, allowed us to identify the system-wide spatial and temporal environmental footprint of the industry. These were then assessed relative to published environmental guidelines. A number of statistical metrics have been devised showing the spatial and temporal effects of this increased production. These results have been communicated to the salmonid industry, the regulatory agency DPIW, the Marine Farming Planning Review Panel and to the Aquafin CRC board.

The 2009 scenario simulation shows larger, more prolonged and wider spatial impact of farm loads on pelagic nutrient concentrations, chlorophyll and bottom water oxygen saturation than in 2002 and a shift from oligotrophic to mesotrophic conditions over 54% of the region. These results are consistent with the elevated waste discharge simulated in 2009 which was roughly 3 times 2002 values. Whilst farm expansion in the D'Entrecasteaux Channel was significantly larger than the expansion in the Huon Estuary, farm impacts were consistently high in the Huon Estuary, suggesting this region is more susceptible to farm waste. This could be because the Huon Estuary has a slower flushing time compared with the D'Entrecasteaux Channel and/or because it accumulates farm waste from elsewhere due to the residual and estuarine circulation. Other regions consistently impacted include the northern D'Entrecasteaux Channel, North West Bay and Barnes Bay. These regions are located in the north of the region and downstream of most of the farm leases with respect to the residual south to north circulation. Port Esperance and Great Taylors Bay in the south were also affected by farm loads, most likely as a result of localised nutrient discharge in excess of the flushing capacity of these embayments.

9.4.6. A proposed monitoring strategy

A long-term monitoring strategy for the industry in the D'Entrecasteaux Channel and Huon Estuary has been developed to ensure the sustainability of the salmonid industry. Our major goal was to design a monitoring program with the capacity to detect the effects of those processes judged to be most threatening to Huon and D'Entrecasteaux ecosystem at the whole-of-ecosystem level. The monitoring program we have proposed is designed to provide knowledge of how well the ecosystem is functioning with an increased nutrient load and to allow any significant temporal trend(s) in ecological indicators to be detected. A major challenge was converting indicators of ecological condition into recommended quantitative performance measures that can be used in a regulatory manner to adaptively manage the ecosystem. These recommendations have been extensively discussed with industry and DPIW representatives and refined through an iterative process. Agreement has been reached in principle about the major elements of the plan and work is underway to implement the strategy.

9.4.7. Future considerations

Near the end of the project a decision was made by the Marine Farming Planning Review Board to impose a limit on the salmonid industry in the Huon and D'Entrecasteaux based on the industry's projections for maximum salmon production (nominally to occur in 2009). This decision was informed in part by the scenario of likely effects on phytoplankton populations produced by our project using the environmental modelling suite. If the industry is to expand further it will either have to demonstrate that it is not having serious environmental impacts (which can only be achieved by a comprehensive monitoring program of the type suggested by our project), or place new production outside of the Huon Estuary and D'Entrecasteaux Channel or find ways to reduce or ameliorate nutrient inputs. It is difficult to see that major reductions in nutrient inputs can be achieved through better feeds, husbandry practices or improved salmon breeds since these are already highly developed. Nonetheless selective breeding may provide some benefit here. The possibility of "co-culture" or "polyculture" of organisms that take up nutrients (such as macroalgae or molluscs) has been suggested over a number of years, but has yet to achieve acceptance by the industry. This is actively under consideration overseas and could be implemented in Tasmania. Since it is the whole-of-ecosystem effects that are important, the nutrient uptake need not occur immediately adjacent to the fish farm and so a range of options for sites are possible. The opportunity for developing new macroalgal or molluscs industries in the D'Entrecasteaux Channel needs to be assessed.

The option of expanding to the seaward side of Bruny Island is already under consideration by the industry which has ambitions for a doubling of production compared with 2007 figures. While the region off-shore Bruny Island is better flushed and larger in extent than the Huon and D'Entrecasteaux region the same general environmental issues will apply and some form of environmental risk assessment will be needed. There is the added complication of connection with the Derwent estuary and any assessment of nutrient effects must take into account efforts already underway in the estuary to reduce inputs from sewage treatment plants. There is thus a need to consider industry expansion in the context of multiple use management of this ecosystem. A new program called INFORMD (Inshore Network for Observation and Regional Management: Derwent-Huon) may assist this.

INFORMD is being established as a collaborative umbrella arrangement between CSIRO, University of Tasmania, the Tasmanian government and stakeholder organisations to support sustainable development of the Derwent-Huon Estuary and adjacent coastal ecosystems. INFORMD will build on the complementary skills and background experience of CSIRO and the University of Tasmania through its various Departments and the Tasmanian Aquaculture and Fisheries Institute. The INFORMD program will augment existing research, planning and management in several ways: (1) delivering a 'state of the art' observation network of fixed and mobile sensors including genomic approaches to monitor key elements of the ecosystem in close to real-time, (2) providing an information management system to allow information from the new observation network and from existing monitoring systems to be integrated and mutually accessible, and (3) developing new models for data interpretation, prediction and comparison management options. A major objective is to provide support tools that make the information and predictions accessible and relevant to industry, planners, policy developers and managers. Consequently the INFORMD program may provide a more comprehensive means to monitor and model the environmental effects of salmonid aquaculture in this region thus increasing management and public confidence that the industry can expand without undue environmental impact.

Finally, the project has developed considerable knowledge of the possible environmental effects of finfish aquaculture that can be applied to other fish species or to other regions. As a first demonstration, the environmental modelling suite has been applied over the past 3 years to the southern bluefin tuna farming region off Port Lincoln through the Aquafin CRC "Risk and Response" project 4.6 (FRDC 2005/059). Our results and approaches have been widely disseminated through conference proceedings, newsletters, popular articles, scientific papers and books as detailed in the Appendices. This has helped to raise awareness of the need for rigorous scientific assessment of the environmental effects of finfish aquaculture and also demonstrated that the industry can operate successfully and sustainably in southern Tasmania.

Appendix 1

Intellectual property

Following the cessation of the Aquafin CRC on June 30th, 2008, IP generated by the project has been assigned to the two main contributing research organisations CSIRO through its Division of Marine and Atmospheric Research and the University of Tasmania through the Tasmanian Aquaculture and Fisheries Institute.

Metadata records have been written for all data collected during the project.

Those data which have been primarily analysed by CMAR staff are described in the CSIRO Marine and Atmospheric Research Laboratories Information Network (MarLIN). MarLIN can be found at:

http://www.cmar.csiro.au/marlin/

search->free text-> aquafin

Metadata stored in MarLIN is published to the Australian Spatial Data Dictionary (ASDD). The ASDD can be accessed at:

http://asdd.ga.gov.au/tech/zap/basic.html

Metadata describing data primarily analysed by Tasmanian Aquaculture and Fisheries Institute (TAFI) have been entered into the BlueNet Metadata Entry and Search Tool (MEST). The MEST can be accessed at:

http://bluenet.its.utas.edu.au/geonetwork/srv/en/main.home

All information generated by the project as reported in this final report and associated technical reports is public domain.

Appendix 2

Staff contributing to the project

Staff Member	Role	Organisation
Dr John Volkman	Project Leader	CSIRO
Dr John Parslow*	Modeller	CSIRO
Dr Peter Thompson	Phytoplankton Ecologist	CSIRO
Dr Andrew Revill	Biogeochemist	CSIRO
Dr Susan Blackburn	Phytoplankton Ecologist	CSIRO
Dr Michael Herzfeld	Hydrodynamic Modeller	CSIRO
Dr Karen Wild-Allen	Biogeochemical Modeller	CSIRO
Ms Rebecca Esmay	Isotope analysis	CSIRO
Ms Pru Bonham	Phytoplankton/zooplankton	CSIRO
Dr John Andrewartha	Modelling support	CSIRO
Dr Pavel Sakov*	Modelling support	CSIRO
Ms Ros Watson*	Nutrient analysis	CSIRO
Mr Daniel Holdsworth	Organic analysis	CSIRO
Ms Val Latham*	Nutrient analysis	CSIRO
Ms Kate Berry	Nutrient analysis	CSIRO
Ms Lesley Clementson	Pigment analysis	CSIRO
Dr Jennifer Skerratt	Modelling support	CSIRO
Ms Leanne Wilkes	Database management	CSIRO
Dr Kerrie Swadling	Zooplankton ecology	University of Tasmania
Dr Christine Crawford	Marine biologist	University of Tasmania
Dr Catriona Macleod	Benthic environment	University of Tasmania
Mr Sam Foster	Field support	University of Tasmania
Ms Anita Alexander	Marine biology	University of Tasmania
Mr Tim Alexander	Marine biology	University of Tasmania
Ms Susan Forbes	Field support	University of Tasmania
Dr Chris Burke*	PhD supervisor	University of Tasmania
Dr Chris Bolch*	PhD supervisor	University of Tasmania
Dr John Purser*	PhD supervisor	University of Tasmania
Mr Dean Thomson	PhD student	University of Tasmania
Mr Paul Armstrong	PhD student	University of Tasmania
Ms Maria Albinsson	PhD student	University of Tasmania
Ms Angela Holmes	PhD student	University of Tasmania
Ms Cecile Rousseaux	BSc(Hons) student	University of Tasmania
Mr Colin Shepherd	Tasmanian State Government	DPIW
Dr Dom O’Brien	Salmon Industry	TSGA

*: staff members who contributed to phase 1 of the project (2002–2006). All CSIRO staff from Marine and Atmospheric Research in Hobart, Tasmania.

Appendix 3

Communications

Presentations at conferences, and meetings with stakeholders and CRC participants, from Aquafin CRC Environment program project staff related to salmonid aquaculture.

2002:

Armstrong, P., Thompson, P. and Bolch, C. (2002). *Nutrient dynamics and phytoplankton blooms in the Huon Estuary*. First Aquafin CRC Conference, Hobart, September 24.

Crawford, C., Foster, S., Jordan, A., Thompson, P., Bonham, P., Parslow, J., Herzfeld, M. and Volkman, J. (2002). *Progress towards the development of a monitoring program for broad-scale effects of salmon aquaculture*. First Aquafin CRC Conference, Hobart, September 24

Forbes, S.E., Macleod, C.K. and Mitchell, I.M. (2002). *What can sedimentation rate tell us about the environmental impact of marine finfish culture in Tasmania*. Australian Marine Science Association Conference, Fremantle, July.

Forbes, S.E., Macleod, C.K. and Mitchell, I.M. (2002). *Design, development and field trial of a sediment trap for use in evaluating organic loading associated with cage finfish culture*. Australian Marine Science Association Conference, Fremantle, July.

Herzfeld, M., Parslow, J., Andrewartha, J. and Sakov, P. (2002). *Physical modelling of the Huon and D'Entrecasteaux Channel*. First Aquafin CRC Conference, Hobart, September 24.

Macleod, C., Forbes, S., Volkman, J., Revill, A., Holdsworth, D., Burke, C., and Bissett, A. (2002). *One thing leads to another – the relationship between biology and chemistry in sediment recovery*. First Aquafin CRC Conference, Hobart, Tasmania, September 24.

Macleod, C., Forbes, S., Volkman, J., Revill, A., Holdsworth, D., Burke, C. and Bissett, A. (2002). *Development of novel methods for the assessment of sediment condition and determination of management protocols for sustainable finfish cage aquaculture operations – Progress Report*. ASAS Conference, Hobart, Tasmania, July.

Macleod, C., Forbes, S., Volkman, J., Revill, A., Holdsworth, D., Burke, C. and Bissett, A. (2002). *A multidisciplinary approach for evaluation and management of recovery processes in sediments associated with marine finfish culture in Tasmania, Australia*. American Society of Limnology and Oceanography Conference, Victoria, Canada, June.

Volkman, J.K. (2002). *Environmental research for Aquaculture ESD*. Sustainable Australian Aquaculture: Practical Solutions to Achieving ESD. Melbourne, July 14-18.

Volkman, J.K. (2002). *Environmental research for the salmonid industry within the Aquafin CRC Environment Program*. The Second Scientific Conference of the Atlantic Salmon Aquaculture Subprogram, Hobart, July 8.

Volkman, J.K. (2002). *Finfish aquaculture and the Australian environment*. Aquafest Conference, Hobart, Tasmania, September 19-22.

Volkman, J.K., Parslow, J.P., Butler, E.C.V., Blackburn, S. and Revill, A.T. (2002). *Environmental issues for salmonid farming in southern Tasmania*. American Society of Limnology and Oceanography Summer Meeting "Interdisciplinary Linkages in Aquatic Sciences and Beyond", Victoria, British Columbia, Canada. June 10-14.

Wildish, D.J., Hargrave, B.T., Macleod, C.K. and Crawford, C.M. (2002). *Detection of organic enrichment near finfish net-pens by sediment profile imaging at SCUBA-accessible depths*. Imaging the Benthos Conference, Aberdeen, Scotland, April.

2003:

Armstrong, P., Bolch, C., Thompson, P. and Blackburn, S. (2003). *Understanding phytoplankton bloom in the Huon Estuary*. Second Annual Conference of the Aquafin CRC.

Forbes, S. and Macleod, C. (2003). *Development of quantitative visual assessment techniques for the assessment of sediment condition at marine finfish farms.* Biodiversity and Conservation Conference, Hobart, Tasmania, October.

Forbes, S. and Macleod, C. (2003). *Development of quantitative visual assessment techniques for the assessment of sediment condition at marine finfish farms.* Poster at Aquafin CRC Conference, Hobart, Tasmania, October.

Herzfeld, M. (2003). *Numerical modelling explained*. Second Annual Conference of the Aquafin CRC.

Macleod, C., Forbes, S., Bissett, A., Burke, C., Gibson, J., Holdsworth, D., Nichols, P., Revill, A. and Volkman, J. (2003). *Links between physical/chemical and ecological techniques for the assessment of sediment condition.* Poster, Aquafin CRC Conference, Hobart, Tasmania, October.

Macleod, C., Forbes, S., Volkman, J., Revill, A., Holdsworth, D., Burke, C., Bissett, A., Gibson, J. and Nichols, P. (2003). *Development of novel methods for the assessment of sediment condition and determination of management protocols for sustainable finfish cage aquaculture operations – an update on research progress*. ASAS Conference, Hobart, Tasmania, May.

Parslow, J. (2003). *Adaptive management of coastal environments*. Second Annual conference of the Aquafin CRC.

Parslow, J.P., Herzfeld, M., Andrewartha, J. and Sakov, P. (2003). *Environmental modelling to support salmon aquaculture in Huon-D'Entrecasteaux*. In Battaglene, S.C. and Cobcroft, J.M. (Eds). The Third Scientific Conference of the Atlantic Salmon Aquaculture Subprogram, May 21st, 2003, CSIRO Marine and Atmospheric Research Marine and Atmospheric Laboratories, Hobart.

Thompson, P. and Bonham, P. (2003). *Contrasting phytoplankton ecology in adjacent coastal water bodies: Huon Estuary and D'Entrecasteaux Channel.* In Battaglene, S.C. and Cobcroft, J.M. (Eds). The Third Scientific Conference of the Atlantic Salmon Aquaculture Subprogram, May 21, 2003, CSIRO Marine and Atmospheric Research Marine and Atmospheric Laboratories, Hobart.

Thomson, D., Volkman, J., Burke, C. and Purser, J. (2003). *Sediment biogeochemistry – benthic-pelagic coupling of nutrients in the Huon Estuary*. Second Annual Conference of the Aquafin CRC.

Volkman, J.K. (2003). *The Aquafin CRC Environment program: highlights of salmon-related research*. In Battaglene, S.C. and Cobcroft, J.M. (Eds). The Third

Scientific Conference of the Atlantic Salmon Aquaculture Subprogram, May 21st, 2003, CSIRO Marine and Atmospheric Research Marine and Atmospheric Laboratories, Hobart.

2004:

Armstrong, P., Bolch, C., Thompson, P. and Blackburn, S. (2004). *Nutrient dynamics and phytoplankton blooms in the Huon Estuary, southeast Tasmania, Australia*. Poster, XI International Conference on Harmful Algal Blooms, November 14-19, Cape Town, South Africa

Blackburn, S., Jameson, I., Johnston, C., Frampton, D., Mansour, M.P., Nichols, P., Robert, S., Volkman, J., Bolch, C. and Tredici, M. (2004). *An Australian perspective on biodiversity and applications of microalgae*. Tenth International Congress for Culture Collections. This won the best poster award.

Blackburn, S., Bolch, C., Haskard, K.A. and Hallegraeff, G.M. (2004). *Reproductive compatibility among four global populations of the toxic dinoflagellate Gymnodinium catenatum (Dinophyceae)*. Poster, Tenth International Congress for Culture Collections.

Forbes, S. and Macleod, C. (2004). *Development of quantitative visual assessment techniques for the evaluation of sediment condition*. “Estuaries and Change” ECSA37-ERF 2004, Ballina, NSW, June.

Forbes, S., Macleod, C., Crawford, C., Volkman, J., Holdsworth, D., Revill, A., Burke, C. and Bissett, A. (2004). *A simple and effective method for farm-based environmental monitoring of caged marine finfish*. Talk, Australasian Aquaculture Conference, Sydney, September.

Macleod, C., Forbes, S., Crawford, C. and Moltschaniwskyj, N. (2004). *Organic enrichment and infaunal community response – Do some systems cope better?* “Estuaries and Change” ECSA37-ERF 2004, Ballina, NSW, June.

Revill, A.T., Cook, P. and Volkman, J.K. (2004). *Carbon flow between bacteria and micro-phytobenthos on a temperate mudflat – insights from bulk and compound-specific isotope analysis.* IsoEcol: Isotopes in Ecological Studies conference, Wellington, New Zealand and Estuaries and Change Conference, Ballina, NSW.

Thomson, D., Volkman, J., Burke, C. and Purser, J. (2004). *Sedimentary biogeochemistry of the Huon Estuary*. Estuaries and Change Conference, Ballina, NSW.

Thomson, D.C., Burke, C.B., Purser and J., Volkman, J.K. (2004). *Benthic nutrient cycling in the Huon Estuary*. Poster, Australasian Aquaculture Conference, Sydney, September.

Volkman, J.K., Holdsworth, D.G., Revill, A.T., Cook, P., Macleod, C. and Forbes, S. (2004). *Organic matter degradation in sediments under salmon sea cages*. Estuaries and Change Conference, Ballina, NSW.

Volkman, J.K., Holdsworth, D.G., Revill, A.T., Macleod, C. and Forbes, S (2004). *The organic geochemistry of sediments under salmon sea cages*. Australian Organic Geochemistry Conference, Leura, NSW.

2005:

Thomson, D., Burke, S., Purser, J. and Volkman, J. (2005). *Benthic nutrient cycling of the Huon Estuary*. Aquafin CRC Conference, Hobart, July 5-7.

Volkman, J.K. (2005). *Environment program highlights*. Aquafin CRC Conference, Hobart, July 5-7.

Volkman, J.K. (2005). *Environmental risks, adaptive management and regional monitoring*. Aquafin CRC Conference, Hobart, July 5-7.

Volkman, J.K. (2005). *Indicators for finfish aquaculture*. Ecological Indicators Workshop, CSIRO Marine Laboratories, July 11-12, Hobart.

Volkman, J.K. and Lyne, V. (2005). *Climate and seasonal changes to key environmental parameters*. Aquafin CRC Conference, Hobart, July 5-7.

2006:

Albinsson, E., Blackburn, S. and Legrand, C. (2006). *The effect of Noctiluca scintillans on harmful algal species of south eastern Australia.* 12th International Conference on Harmful Algae, Copenhagen, Denmark. International Society for the Study of Harmful Algae.

Armstrong, P.A., Thompson, P., Bolch, C.J.S. and Blackburn, S.I. (2006). *Gymnodinium catenatum preference for and growth on nitrate, ammonium and urea.* 12th International Conference on Harmful Algae, Copenhagen, Denmark. International Society for the Study of Harmful Algae.

Blackburn, S.I., Wild-Allen, K., Doblin, M.A., Armstrong, P.B., Bolch, C.J., Thompson, P.A. and Hallegraeff, G.M. (2006). *Vertical migration: a key strategy for the ecological success of the toxic dinoflagellate Gymnodinium catenatum in south east Tasmania, Australia.* 12th International Conference on Harmful Algae, Copenhagen, Denmark. International Society for the Study of Harmful Algae. p. 38. http://www.bi.ku.dk/hab/docs/P&A_Book.

Holmes, A.K., Hallegraeff, G.M. and Blackburn, S.I. (2006). *Culture and ichthyotoxicity of the red tide dinoflagellate Noctiluca scintillans.* 12th International Conference on Harmful Algae, Copenhagen, Denmark. International Society for the Study of Harmful Algae.

Macleod, C.K., Moltschaniwskyj, N., Crawford, C. and Forbes, S. (2006). *Site selection – further considerations for environmental sustainability of cage aquaculture*. Australasian Aquaculture Conference, Adelaide, August.

Macleod, C.K., Shepherd, C.J. and O'Brien, D.P. (2006). *Environmental sustainability of aquaculture operations in Australia - a farmers, a managers and a researchers perspective*. Australasian Aquaculture Conference, Adelaide, August.

Skerratt, J., Ratkowsky, D., Holmes, A. and Blackburn, S. (2006). *Field and laboratory mortality and bloom decay rates of Gymnodinium catenatum: Improving parameters in coastal models.* 12th International Conference on Harmful Algae, Copenhagen, Denmark. International Society for the Study of Harmful Algae.

Swadling, K.M., Macleod, C.K. and Foster, S. (2006). *Zooplankton dynamics in a southeastern Australian estuary*. Liege, Belgium, May 8–12.

Thompson, P. A., Bonham, P. and Swadling, K. (2006). *Selective grazing as a factor in phytoplankton blooms*. Global challenges facing oceanography and limnology: ASLO Summer meeting, Victoria, British Columbia, Canada.

Volkman, J. (2006). *Environmental management – status of current project and refinement of focus*. Tasmanian Salmonid Growers Association, Industry R&D Workshop. December 6.

Volkman, J. K. (2006*). Environmental research for finfish aquaculture in Australia.* Open Ocean aquaculture – moving forward. International workshop at East-West Center, Honolulu. October 21–29.

Wild-Allen, K., Parslow, J., Herzfeld, M. and Thompson, P. (2006). *Modelling the effects of salmon farms on plankton ecology in Tasmanian coastal waters.* Challenger Conference for Marine Science, Oban, UK.

2007:

Albinsson, E, Blackburn, S. and Legrand, C. (2007). *Bloom distribution and feeding characteristics of the dinoflagellate Noctiluca scintillans in south eastern Tasmania, Australia*. 4th Aquafin CRC Conference, May 15-17, Barossa Valley. Poster.

Albinsson, E., Blackburn, S. and Bolch, C. (2007). *Harmful algal bloom dynamics: microalgal bacterial interactions*. 4th Aquafin CRC Conference, May 15-17, Barossa Valley. Oral Presentation.

Armstrong, P., Thompson, P.A., Bolch, C.J.S. and Blackburn, S. (2007). *Gymnodinium catenatum – preference for and growth on nitrate, ammonium and urea*. 4th Aquafin CRC Conference, May 15-17, Barossa Valley. Poster.

Herzfeld, M., Andrewartha, J. and Middleton, J. (2007). *Hydrodynamic modelling used in Aquafin CRC numerical studies*. 4th Aquafin CRC Conference, May 15-17, Barossa Valley. Poster.

Macleod, C., Swadling, K. and Foster, S. (2007). *Mud, bugs and weeds: an overview of salmonid environmental research*. 4th Aquafin CRC Conference, May 15-17, Barossa Valley. Oral Presentation.

Rousseaux, C., Descy, J.P. and Thompson, P.A. (2007). *A study on the effects of microzooplankton grazing on the diel vertical migration (DVM) of dinoflagellates in the Huon Estuary (Tasmania).* Australasian Society of Phycology and Aquatic Botany (ASPAB) annual conference program and abstracts, Hobart, Tasmania. p. 36.

Rousseaux, C., Waite, A., and Thompson, P.A. (2007). *Phytoplankton growth and microzooplankton grazing: comparing two ecosystems off Australia*. Marine science in a changing world: the Australian Marine Sciences Association 45th Annual Conference. AMSA 2007, Melbourne, Australia. Australian Marine Sciences Association.

Skerratt, J., Ratkowsky, D., Holmes, A. and Blackburn, S. (2007). *Mortality rates of Gymnodinium catenatum: improving microbiological parameters in coastal biogeochemical models*. 4th Aquafin CRC Conference May 15-17, Barossa Valley. Poster.

Swadling, K., Alexander, A., Blackburn, S. and Holmes, A. (2007). *Effects of a toxic dinoflagellate Gymnodinium catenatum on grazing rates, egg production and survivorship of marine copepods*. 4th Aquafin CRC Conference, May 15-17, Barossa Valley and 4th International Zooplankton Production Symposium, Hiroshima, Japan, June.

Thompson, P. (2007). *Proposals for monitoring*. Atlantic Salmon Aquaculture Subprogram Industry Workshop, Hobart, November 30.

Thompson, P.A. and Bonham, P.I. (2007). *Phytoplankton blooms, top down or bottom up control?* Australasian Society of Phycology and Aquatic Botany (ASPAB) annual conference program and abstracts, Hobart, Tasmania. p. 43.

Volkman, J.K. (2007). *Aquafin CRC Environmental Program - Review of progress*. Aquafin CRC Board Meeting, Melbourne, November 29.

Volkman, J.K. (2007). *Australian research to ensure sustainability of aquaculture*. BioEco 2007 International Conference for Bioeconomy, Tianjin, China, June 26-28. (Abstracts, pp. 221-223).

Wild-Allen, K. (2007). *Biogeochemical modelling in coastal waters*. 4th Aquafin CRC Conference, May 15-17, Barossa Valley. Oral Presentation.

Wild-Allen, K. (2007). *Use of a coastal biogeochemical model to design an environmental monitoring program*. 4th Aquafin CRC Conference, May 15-17, Barossa Valley. Poster.

Wild-Allen, K. (2007). *Use of a coastal biogeochemical model to design an environmental monitoring program*. International Symposium on Integrated Coastal Zone Management, ICES/CIEM, Norway.

Wild-Allen, K.A. and Volkman, J.K. (2007). *Biogeochemical model scenario for 2009*. Atlantic Salmon Aquaculture Subprogram Industry Workshop, Hobart, November 30.

2008:

Blackburn, S.I., Jameson, I., Johnston, C., Nichols, C.M., Frampton, D.M.F., Mansour, M.P., Nichols, P.D., Robert, S. and Volkman, J.K. (2008). *Microalgae Bioapplications Research at CSIRO, Australia.* Algal Culture Collections 2008, June 8-11, hosted by the Culture Collection of Algae and Protozoa SAMS Research Services Ltd., Dunstaffnage Marine Laboratory, Oban, Scotland, UK

Blackburn, S.I., Jameson, I., Johnston, C., Nichols, C.M., Frampton, D.M.F., Mansour, M.P., Nichols, P.D., Robert, S. and Volkman, J.K. (2008). *Microalgae bioapplications research, CSIRO, Australia.* 11th International Conference on Applied Phycology, Galway, Ireland, June 21-27,

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Interim Report for Phase 1 of the Project

The interim report for the first phase of project 4.2 was submitted to the CRC on March 9th, 2006. A pdf is provided on the CD accompanying the present report.

Volkman, J.K., Parslow, J., Thompson, P., Herzfeld, M., Wild-Allen, K., Blackburn, S., Crawford, C., Bonham, P., Holdsworth, D., Sakov, P., Andrewartha, J.R. and Revill, A. (2006). *System-wide environmental issues for sustainable salmonid aquaculture*. Interim report for Aquafin CRC project 4.2.

Interim Technical Reports

The interim report also included a number of technical reports on a CD plus animations from the hydrodynamic and biogeochemical models. These are provided as pdfs on the CD accompanying the present report.

Herzfeld, M., Parslow, J., Sakov, P. and Andrewartha, J.R. (2005). *Numerical hydrodynamic modelling of the D'Entrecasteaux Channel and Huon Estuary*. Aquafin CRC Technical report.

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Appendix 4

Technical Reports Associated with this Final Report

These technical reports provide additional detail on various components of our research. Copies as pdfs are available on the CD that accompanies this final report.

Bonham, P., Rousseaux, C. and Thompson, P. (2008). Effects of grazing by microzooplankton on phytoplankton in the Huon Estuary and D'Entrecasteaux Channel.

Clementson, L.A., Blackburn, S.I., Berry, K.M. and Bonham, P.I. (2008). Temporal and spatial variability in phytoplankton community composition in the mouth of the Huon River Estuary.

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Revill, A.T., Holdsworth, D.G., Volkman, J.K. and Swadling, K. (2008). Fluxes of organic matter and lipids to sediments in the Huon Estuary, Tasmania.

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