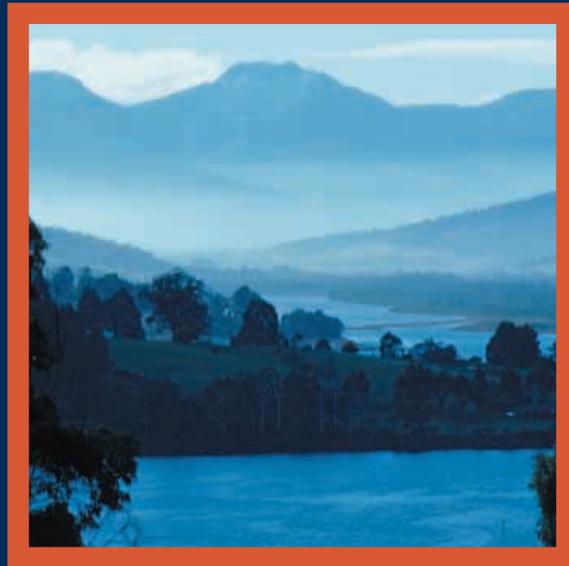


HUON ESTUARY STUDY

Environmental Research for Integrated Catchment Management and Aquaculture



CSIRO Huon Estuary Study Team

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Summary

The Huon River and its estuary in southeastern Tasmania have supported the mature enterprises of forestry and horticulture since the early days of the region's settlement. The community of the Huon valley is now developing a more broad-based economy with the new industries of tourism and aquaculture. Comprehensive and effective management of the catchment and waterways are fundamental to a prosperous future that is in harmony with the natural ecosystem. The Huon Estuary Study has been a three-year research program to improve knowledge and provide a scientific framework for such management of the estuarine zone of the Huon River.

Our project was founded on the need to evaluate the environmental quality and understand the working of the estuary as a *system*. Although the farming of finfish in these waters was a key reason for the research, we sought to look at this industry in the context of the entire estuary, and even beyond to the catchment where appropriate. Increasingly, it is recognised that estuaries and other coastal water bodies are managed well when the catchment (the natural zonation) is the domain for governance. For our investigations to contend with this complexity, we needed to study the key questions with an interdisciplinary (physics, chemistry and biology) approach that considered the important time and space scales.

The Huon Estuary is a strongly stratified waterway; the Huon River flows consistently throughout the year and drives a two-layer estuarine circulation, with a shallow, fresher layer flowing seaward over a deeper, marine bottom layer moving sluggishly upstream. Other features include: fast flushing times (the surface layer swept out in hours to days, and the whole water column in days up to one week), and low turbidity (but light penetration is diminished by high levels of coloured dissolved organic matter). Autonomous profiling systems deployed at two sites in the lower estuary showed rapid fluctuations in physical properties caused by run-off and wind mixing. Currents are generally weak ($< 0.2 \text{ m s}^{-1}$), and tidal currents even weaker in this microtidal system. The Huon and its neighbouring estuary the Derwent share many physical features in common and presumably function very similarly; knowledge from one can be applied to the other.

We find that the Huon Estuary is a 'substantially natural' waterway — meaning that it is not changed dramatically from its historic baseline. In general, its environmental quality is high and that of its two main source waters — the Huon River and coastal waters of southeast Tasmania — is very high. We flag three issues for further attention by managers. The first is the degraded quality of some lower catchment streams, apparently as a result of land-based activities in their subcatchment. Up to now, the effects of their deterioration remains localised. The second is dissolved oxygen. Oxygen depletion in deeper holes in the upper estuary may have natural causes, but we also observed a few instances of very low concentrations in bottom waters in the lower estuary in summer. The final issue of concern is the appearance of raised levels of ammonia and nitrite (readily available forms of nitrogen for microalgae) in bottom waters in the lower estuary after dense microalgal blooms. In combination with oxygen depletion, these increased nutrient concentrations indicate that sediments are recycling and releasing nutrients efficiently back to the water column, making the estuary more vulnerable to increases in nutrient loads.

The sediments of the estuary are mostly muds, small regions of sand occur near the mouth and in the upper reaches above Crowthers Bay. The muds have high organic matter content; much of this is historical — from land plants in the upper estuary, and from phytoplankton production in the middle and lower estuaries. This high proportion of organic matter contributes to the vulnerability of the bottom waters to oxygen depletion. Hospital Bay (Port Huon) sediments are different, apparently because of past sawmill and pulp mill operations on the shores of this water body.

A preliminary survey of contaminants (trace metals and pesticides) — in sediments and water of the estuary — revealed no levels above national guidelines, apart from higher concentrations of the organochlorine pesticides DDT and DDD in a sediment sample from Hospital Bay. However, a follow-up survey with more sensitive measurements of pesticides currently used in the catchment would be advisable.

In the Huon Estuary, we observed a close link between physical conditions (e.g. light, temperature, salinity layering) and populations of microalgae. Blooms of diatoms and dinoflagellates alternate from spring to autumn in most years. During both years of our field program (1996/97 and 1997/98), diatom blooms were observed in spring and late summer, but their biomass was quite low (1–2 mg chlorophyll *a* m⁻³). Dinoflagellate blooms were not observed in the first year, but dense blooms (typically 20 mg chlorophyll *a* m⁻³) of the toxin-producing *Gymnodinium catenatum* developed early in the summer and autumn of 1997/98 along with less intense blooms of other dinoflagellates. Nitrogen seems to be the limiting nutrient for microalgal production in estuary waters. Surface-water depletion of nitrogen is not such a constraint for dinoflagellates, because these microalgae can migrate vertically to draw on nutrients in bottom waters and return back up to the light for photosynthesis. By moving between outflowing surface waters and inflowing bottom waters, dinoflagellates also appear to avoid being flushed out of the estuary, and are able to accumulate high biomass levels.

We have identified the environmental conditions that support blooms of *G. catenatum* (largely corroborating earlier research), but still elusive are the factors that prevent blooms forming in apparently suitable years (e.g. 1996/97). Resting cysts of the dinoflagellate in sediments, and their mechanism of germination, could hold the key.

Key environmental issues in the Huon Estuary are associated with effects and fate of nutrient and organic matter loads from the catchment, from coastal waters, and from activities in the estuary, especially salmon farming. We combined field observations, budgets, and simple process models to understand nutrient cycling in the estuary. Some general conclusions for the critical nutrient nitrogen were that about half the available nitrogen (for biological production) entering the estuary, under existing conditions, comes from bottom waters at the marine boundary. The other half is split almost evenly between agricultural run-off and from operation of salmon farms. (Very large fluxes of marine nitrate circulate through the estuary in winter, but this is largely unutilised, owing to low light.) The Huon River discharges appreciable quantities of total nitrogen from the upper catchment, but it is mostly refractory organic nitrogen; the upper catchment is a negligible source of available nitrogen. Overall, the estuary is a substantial sink for total nitrogen, which must be either buried in sediments or lost to the atmosphere (as gaseous nitrogen through denitrification) — the balance between the two remains to be elucidated.

We looked at a number of aspects of finfish farming. At the scale of the lease site and its operation, our focus was on the fallowing of the sediments beneath sea-cages. A pilot study of fallowing of single cage sites in an operational farm revealed that most of the waste deposited directly beneath the cage, and in declining amounts moving away from it. After an 11-month interval without a cage overhead the anoxic conditions in surface sediments had returned to oxic. Nevertheless, we found that a significant part of the organic waste still remained. Further work is needed to find out whether this incomplete breakdown of organic matter has implications for re-occupation of the site.

Salmon farms may affect water quality nearby their sites. Our field observations yielded evidence of higher ammonia concentrations in surface and mid-depth waters close to the marine farm zones. However, phytoplankton levels were not significantly elevated at sites near fish farms: we conclude that nutrients disperse before phytoplankton can respond. Sporadic observations were also made of oxygen-depleted bottom waters near marine farms.

The effects of nutrient loads from salmon farms must be considered at estuary-wide scales. We used a simple algal bloom model to look at effects of increasing salmon farm loads to twice, four times and ten times present-day values. According to the model, doubling fish farms from present levels in the waterway will increase biologically available nitrogen by 50%. At four times the current loads, available nitrogen is doubled again, and the predicted phytoplankton biomass (measured as chlorophyll *a*) is about twice current levels. In this circumstance, nitrogen is scarcely limiting and a substantial risk of prolonged algal blooms would arise. At ten times the existing loads, there will be sufficient available nitrogen to saturate phytoplankton growth; the ecosystem will become eutrophic.

From experience gained in our observational program, we have proposed an automated, catchment-scale monitoring network, which should meet the broad requirements of the region. Marine farmers will be able to gain operational data from autonomous instrument systems strategically located in the estuary, and backed up by simpler devices about their farms. Regional environmental managers will be able to draw on the integrated network of estuarine instruments and sensors in the catchment and on tributary streams to monitor the effects of change, and also to derive product for environmental forecasting and scenario testing.

As a result of our research, we have identified a number of issues that merit further attention by environmental custodians of the Huon Estuary. It is strongly recommended that they be read in their context in Chapter 11 of the report. We advise:

- *that the current, near-pristine state of the upper Huon catchment should be carefully maintained because it is the main source of freshwater to the estuary, and that thorough measures are instituted to prevent localised degradation from human activities*
- *that management plans designed to prevent further deterioration of the lower subcatchments should be implemented, and steps taken to improve the quality of run-off from the degraded subcatchments*
- *that information pertaining to nutrient budget estimates in the Huon Estuary is kept up to date, and is made as comprehensive as possible (this could form part of a broader catchment audit process)*

- *that any development planned for Hospital Bay (or nearby in the main estuary) that could disturb the underlying sediments should require a survey of the sediment quality and characteristics*
- *that a formal risk assessment of the system's carrying capacity should be carried out to underpin any further expansion of finfish farming in the Huon Estuary*
- *that the salmonid-farming industry work with government and research agencies to establish a more detailed understanding of the environmental effects of different fallowing practices with the objective of developing generic guidelines*
- *that near-field effects of marine farming on water quality should be reconsidered, to ensure that regulations and compliance measures are appropriate*
- *that all users and other stakeholders of the Huon Estuary work in concert to develop a monitoring strategy, and foster the design of monitoring systems, to assist with informed decision-making about the use of the estuary and developments along its shores*
- *that any monitoring program seeking to identify causative factors for harmful algal blooms include measurements of parameters, at weekly intervals (or even daily at critical periods), that are indicators of environmental conditions known to support bloom events, along with integrated phytoplankton analysis, and*
- *given the importance of the Huon Estuary as a site for salmonid culture, and its potential role as a model system both nationally and internationally, we suggest that the development of an integrated decision-support system for the Huon Estuary should be considered.*

A set of recommendations on research gaps is also presented in the final chapter of the report.

1 INTRODUCTION

1.1 Background to Study

Aquaculture continues to expand rapidly in most States of Australia, as part of a global trend. Governments see in the industry an opportunity for sustainable development that benefits both regional economies and employment. At the same time, farmed production compensates partly for the declining catches from wild fisheries. One quarter of Tasmania's seafood production (~\$68 million) in 1994 was derived from mariculture – most of it from two species, Atlantic salmon and Pacific oysters (DPIF 1994a). By 1998, the figures had risen to over a third or ~\$75 million (DSD 1999). These figures do not include any value adding during processing.

The centre for salmonid production in Tasmania is the Huon River estuary. In 1994, about 80% of the State's salmonid harvest of an estimated 4000 tonnes per annum originated from fish farms in the estuary and the small neighbouring bay of Port Esperance (Tassal Ltd — *unpublished data*). This resulted in an estimated value of annual farm production at almost \$30 million¹, with about 170 people employed directly in the industry and \$16 million injected into the local economy. In that year, the marine farming zone of Huon River and Port Esperance included 21 marine farm leases (finfish, shellfish and seaweed) that occupied about 130 ha of the waterway, which is about 0.6% of its total surface area (DPIF 1994a). Expansion under the Marine Farming Development Plan (MFDP) for Huon River and Port Esperance, finalised in October 1996 (DPIF 1996), has seen the current lease area for marine farms expand to 354 ha. Atlantic salmon production from the Huon and Port Esperance Plan area exceeded 5000 tonnes in the year 1998/99 (DPIWE — *unpublished data*). A State Government review of the Huon River and Port Esperance Marine Farming Development Plan is in progress.

The Huon Estuary has not been as important an area for growing shellfish. This is despite most of the marine farms being licensed for shellfish (often jointly with finfish), and about half of them commercially growing either mussels or Pacific oysters. It is worth noting that in the Port Cygnet arm of the estuary only shellfish are now farmed. An introduction to the estuary, its setting, condition and environmental stresses, is given in the next chapter.

The procedure for granting the original salmon-farm leases in Tasmania during the 1980s was based on limited understanding of environmental effects and requirements. Inadequate legislation, a haphazard process for lease allocation and objections from sections of the general community were also difficulties. This situation culminated in a general State Government moratorium on further marine farm expansion in 1991, pending the development of marine farm planning schemes consistent with an integrated approach to coastal management. During this time of uncertainty, fish farmers improved farm management practices (reducing disease, optimising feed and feeding procedures, identifying stocking constraints, etc.) to raise production. Consideration of the environment of fish farms was generally deferred unless there was an obvious link to farm yield. However, future development of fish farming (and all aquaculture) is clearly dependent on the maintenance of a healthy aquatic environment.

¹ The estimate of farm gate value of production was made on the basis of an Atlantic salmon fetching \$9 kg⁻¹ HOGG (head-off, gilled and gutted).

Concerns are, (i) potential adverse effects of local environmental degradation from waste on finfish farms, leading to increased disease and poor growth, and (ii) perceived and real risks of the broader environmental effects of aquaculture on the regional coastal environment, and the ramifications of other human activities (especially in catchments) on environmental quality and fish farming.

The Tasmanian Government proclaimed the Marine Farming Planning Act (1995) in May 1996 to redress the statutory and management deficiencies of past legislation (e.g. Fisheries Act 1959) that did not cover adequately the development of marine farming. However, it could not itself remedy directly the lack of scientific knowledge required for effective environmental decision-making. This is succinctly stated in a discussion document on marine farm development plans (DPIF 1994a): “there is a general lack of detailed scientific information providing base-line environmental data on the marine environment”, and “there is a limited framework for consideration of the environmental consequences of a marine farm proposal”. The new legislation does prescribe that a draft marine farming plan, or an amendment to a plan, must include an environmental impact statement as part of the draft plan. Marine Resources (Tasmanian Department of Primary Industries, Water and the Environment — DPIWE) — as a condition of the 1995 Act — also requires baseline surveys (see following subsection) for all new salmon farms, and farm expansions of greater than 10%. So limited environmental information in marine farming zones has been collected from 1996. The legislation also allows, in principle, for the aquaculture industry to be compensated for environmental degradation caused by third parties, but establishing links depends upon adequate monitoring and scientific understanding of the local and regional environment.

The inadequacy of knowledge of the physics, chemistry and biology of the Huon Estuary system – the wellspring for the burgeoning aquaculture industry in the region – was the impetus for our project.

1.1.1 Previous environmental research on marine fish farming

Environmental issues for finfish farming in estuaries and coastal waters have been investigated overseas extensively [see Gowen and Bradbury (1987), Silvert (1992), Gowen and Rosenthal (1993), Wu (1995) and Buschmann et al. (1996) for reviews, and special issues of the *Journal of Applied Ichthyology* (Vol. 10, No. 4, 1994) and *Estuaries* (Vol. 18, No 1A, 1995), as well as Wu et al (1994) and Black (1996) for up-to-date reports]. Woodward (1989) reviewed the interaction between finfish farming and the environment for Tasmanian interests.

The environmental effects of finfish aquaculture upon its supporting aquatic system vary depending on the species being farmed and the location of the farming operation. They are known to include increased nutrient loading (nitrogen and phosphorus) and turbidity; addition of chemicals used in disease and parasite control; increased deposition of organic matter to sediments, decreased sediment redox potential; increased flux of hydrogen sulfide (H_2S , HS^-), methane and total ammoniacal nitrogen from sediments; altered benthic communities (e.g. microbes, macrofauna); escapes of farmed species; introduced parasites, diseases and pest species (including toxic microalgal blooms); and growth of existing parasite populations.

Deleterious external influences on finfish aquaculture involve seasonal or extreme conditions outside the acceptable range for farmed species (e.g. water temperature); degraded water

quality (deficient in oxygen, high suspended solids concentration, increased contaminant loading – heavy metals, pesticides, hydrocarbons...); increased incidence of problem microalgal blooms; occurrence of other problem biota (e.g. jellyfish); exposure to human and animal pathogens from effluents and run-off; and attacks by predator species, such as seals.

Comparatively little research into environmental interactions with aquaculture has been done in Australia. Recent studies of sediments and associated macrofauna around salmonid marine farms in southeast Tasmania (Ritz et al. 1989, Ye et al. 1991) suggest that farming in these waters affect only sediments near the cages (<30 m), and that these effects are reversible over time.

Woodward et al. (1992) made a two-year study of the waste produced from a salmon cage in the Huon Estuary within the general area to be studied by our project. Their data suggested that salmon farming was not having a detectable impact on natural algal blooms and nutrient cycling, although a comprehensive survey of the estuary was beyond the scope of their study. The possibility of effects must be recognised because of the release of high concentrations of nutrients (ammonia, nitrate and phosphate) from breakdown of the organic matter in the sediments and by direct excretion from the fish.

The success of finfish farming is critically dependent on water quality. A comprehensive picture of water quality is not yet available for the Huon River estuary, although some spot measurements have been made (in addition to the nutrient data from the Woodward et al. (1992) survey of 1988/89). Nutrient inputs from agriculture and sewage are identified as point sources, but remain a point of conjecture when their origin is diffuse. Wotherspoon and co-workers (1994) have made some estimates of a variety of nutrient sources to the Huon system. Faecal bacteria are reported intermittently in the estuary. Other suspected inputs include pesticides and herbicides from agriculture and forestry (Gallagher 1996).

1.1.2 Associated studies active in the Huon region

Finfish farmers routinely measure select water quality parameters – water temperature, salinity and dissolved oxygen – as a part of farm operation. They also monitor plankton regularly by microscopic examination of the contents of plankton net tows. As well, their fish-farming license is conditional upon them gathering additional environmental data — they are (DPIF 1996). For example, assessment (usually six-monthly) of the condition of the sediments beneath the farm is required. The monitoring, as specified by Marine Resources (DPIWE), is an underwater video transect of the seabed from within the farm to beyond the lease boundary. If a new lease, or an expansion of an existing lease by more than 10%, is foreshadowed, then an environmental baseline survey is required before marine farming operations begin. For salmonid fish farms, this survey includes analysis of sediments for grain size, organic carbon content (by loss on ignition), redox potential, stable isotopes (^{13}C and ^{15}N) and composition of the benthic community; a standard underwater video assessment must also be made. In addition, currents in the overlying water need to be recorded.

The 'Huon Catchment – Healthy Rivers Project' (HC-HRP) was set up in October 1993 by the Huon Valley Council to promote a sustainable development strategy for the Huon River and its catchment. One of the objectives of the project was "to provide a sound scientific knowledge base for long-term planning and management of the river's catchment, waters and sediments".

Since our study would furnish fundamental information for this objective, it was identified at the outset as integral to HC-HRP. The HC-HRP also nurtures community activities, such as water quality monitoring of streams by the region's Landcare, Coastcare and Waterwatch groups (more than 10 groups are currently monitoring tributaries to the Huon Estuary throughout much of the catchment), and the development of subcatchment management strategies.

Various State Agencies also mount programs in the Huon region. The Resources Management and Conservation Division of DPIWE gauges stream flow and makes intermittent measurements of water quality of the main tributaries of the Huon catchment. Run-off has been one of the factors implicated in promoting seasonal blooms of the toxic dinoflagellate *Gymnodinium catenatum* (DPIF 1994b). This microalga, thought to have been introduced into Tasmania in the late 1970s via ballast water, first bloomed extensively in 1986 (McMinn et al. 1997). It became the focus of a major study after the 1993 bloom, which closed some shellfish farms for 5 months (DPIF 1994b). The Tasmanian Shellfish Quality Assurance Program (under the Department of Health and Human Services) monitors water quality of all shellfish-growing areas around the State's coast. In addition to routine microbiological testing of water samples, plankton counts from net tows are made the levels of the dinoflagellate toxin in commercial shellfish are determined. It is also the authority responsible for closing shellfish farms when a threshold toxin concentration is exceeded in shellfish.

The Department of Primary Industry and Fisheries recognised that an effective environmental monitoring program for salmon farming was necessary to meet the requirements of the Marine Farming Planning Act (1995). In late 1996, a research project was begun at the Department's Marine Research Laboratories, Taroona — now part of the Tasmanian Aquaculture and Fisheries Institute (TAFI) — to investigate the best techniques to monitor the environment in and around salmon farms. Two sites with different environmental conditions — an estuarine site at Hideaway Bay in the Huon River, and a less protected coastal embayment at Nubeena, Tasman Peninsula — were studied over eighteen months. Physico-chemical parameters, benthic community structure and video recordings were obtained at a number of sites on both farms. In 1999, a draft report of findings and recommendations was presented to State Government and industry (Crawford et al. 1999).

As an adjunct to the above study of monitoring techniques, a pilot research project on developing fallowing protocols for finfish farms was conducted by the Marine Research Laboratories at the same two salmon farms. This project investigated the rate of recovery of the sediment after the removal of sea cages. Results from a collaborative study by CSIRO Marine Research, TAFI and Huon Aquaculture Pty. Ltd. on the sediment-chemical aspects of fallowing form the basis of Chapter 8 of this report (also McGhie et al, 2000). TAFI are currently preparing a report on the corresponding effects on the benthic infaunal community.

Another study on sediment recovery is proceeding on a farm site in North West Bay (~30 km to the northeast of the Huon Estuary). The site had been used intensively for salmon cage culture, but was vacated in mid-1999. This provided a unique opportunity for the Marine Environment section at TAFI to monitor the rate of recovery of a degraded farm site, rather than just the sediments beneath a single cage within an operational farm. Regular monitoring will continue until 2001.

1.2 Need

The potential for finfish farming to damage the aquatic environment can be diminished by sound management practices. Owing to a lack of scientific information on environmental conditions in the Huon Estuary, fish farms are mostly managed conservatively with respect to siting, stocking density and sediment fallowing. Nonetheless, sections of the public are concerned that finfish farming may be causing long-term ecological damage in some areas of the estuary, but the environmental quality and ecosystem records that might support or refute such a view do not generally exist.

From extensive discussions with managers from State Government, industry representatives and other stakeholders in the Huon Estuary, we identified the following issues as requiring scientific investigation.

- Stocking and fallowing practices, and local management of waste on finfish farms
- Nutrient sources, sinks and cycling throughout the estuary, especially in the neglected area of sediments
- The significance of individual nutrient inputs (including finfish farms) to the nutrient status of the estuary
- The links between nutrients, physical forcing and algal blooms, and the possibility of predictive management of toxic blooms in the future
- How and where contaminants are dispersed in the water column and sediments
- The extent to which the degradation of estuarine sediments beneath farm cages is reversible by fallowing
- Methods and technologies for monitoring of local environmental quality

In addition to the above, the Marine Farming Branch of DPIWE was looking to this study to provide an aquatic system context for setting environmental parameters to monitor performance of marine farms in the Huon Estuary, and elsewhere. A current license condition for finfish farming operations in Tasmania, mentioned above, requires them to make regular environmental surveys. The basis for this condition — set out in the MFDP — is to guard against “unacceptable environmental impact 35 m outside the boundary of the marine farming lease area” (DPIF 1996). The MFDP for the Huon River and Port Esperance is soon to be one of the first plans to be publicly reviewed.

1.3 Scope of This Study

From its inception, our study was envisaged as a fully integrated and multidisciplinary investigation of the Huon River estuary. Although aquaculture is the main enterprise associated with this waterway, our investigations would look not only at the effects of this activity, but also at other pressures over the whole estuary, and if necessary to the catchment beyond.

It was always seen as important to link our work with other research and observational activities in the Huon Estuary. The relationship with the Huon Catchment – Healthy Rivers Project has been mentioned above. We also established links with the fish-farming industry,

relevant government agencies (Marine Farming Branch and the Resource Management and Conservation Division of DPIWE), the Huon Valley Council and the University of Tasmania. Moreover, our project provided the logistics with which we could support several postgraduate research projects; details of these “associated” projects are included in the appendices. The benefits of many of these collaborations are evident in this report, and we trust that our research program provided benefits in return to our partners.

The Huon Estuary Study in its first stage was a field program to gather sufficient information to describe important aspects of the physics, chemistry and biology of the aquatic system. We looked to acquire data at different time and space scales, so as to effectively describe physical and biogeochemical processes active over, say, minutes within a few metres out to years or decades over the entire estuary. The focus of the physics was the estuarine structure and transport. That of the chemistry was mostly nutrients in the water column, organic biomarker compounds (including stable isotope measurements – ^{13}C and ^{15}N) in sediments, and pigment analysis of suspended particulate matter to provide a link to microalgal studies. The main theme for the biology was the investigation of phytoplankton ecology, looking at species succession, the occurrence of blooms, and factors influencing outbreaks of the toxic dinoflagellate *G. catenatum*.

The second stage of the project (often running in parallel with the first) was in the laboratory with the analysis of samples, acquisition of supporting information, construction of a project database and entering data therein. Interpretation of the gathered information was by both traditional data analysis and numerical models of physical and biogeochemical processes in the estuary. The models developed specifically for this work also provided some predictive capacity so that conditions outside those seen in the fieldwork could be investigated. Importantly, this phase of our work integrated knowledge across scientific disciplines to gain insight into total-system function — the interplay of physics, chemistry and biology in the estuary.

The final stage of the project was to communicate the outcomes of our work. Implications for environmental managers in State and local government and within the aquaculture industry were highlighted. However, we were also mindful of the interests of groups in the general community (typically working under the umbrella of the Healthy Rivers Project) who are striving to improve the condition of, and the public attitude to, the Huon waterways. Another issue we have tackled, mainly as a result of our experiences with field instruments and their output, is the matter of specific recommendations for effective long-term monitoring of the Huon Estuary. We have looked beyond the operational measurements needed in the aquaculture industry to the monitoring of the environmental health of the estuary, and in other instances to the entire catchment.

1.4 Project Objectives

The objectives of the study remained essentially as presented in our project application. They were to:

1. Determine the sources, distribution and cycling of nutrients (including those from finfish farming) in the Huon River estuary, and relate nutrients and physical parameters to algal dynamics
2. Evaluate the processes (and their rates) that contribute organic matter to sediments from finfish farming and natural sources; and the significance of this organic matter in the cycling of nutrients through the sediments
3. Determine the sedimentary distribution of organic matter around the fish cages that ensues from salmonid farm operation, and the time needed for degraded sediments to return to ambient conditions when cages are removed (the latter is a pilot experiment only)
4. Test the usefulness of different methods for monitoring the environmental quality of sediments and the water column to, (i) provide a scientific basis for the design of a monitoring framework for both industry and environmental managers, and (ii) give technical advice on optimising such a framework to address both localised impacts and general estuarine conditions

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2 STUDY CONTEXT AND DESIGN

2.1 Introduction

Study design is often neglected in environmental research — for estuaries as elsewhere. It tends to be considered perfunctorily at the outset of a study, and is then adjusted *ad hoc* throughout the operational phase when early results indicate shortcomings. Without doubt, a strategy for environmental sampling — deciding what to measure, and where, when, and how to measure it in a *system* context — should be the basis upon which any estuarine study is founded.

Before estuarine study design can begin in earnest, attention must be paid to three precursors: the study objectives, the estuary as a dynamical system — its key processes and responses, and the natural time and space scales of variation. Generally, this requires a preliminary gathering of all information relevant to the study estuary.

Published discussions of survey design in estuarine studies usually have a narrow perspective. Their attention has often been weighted toward a particular scientific discipline, or influenced by budgetary or regulatory considerations. Only a handful of reports on study design have had either a broad interdisciplinary basis (e.g. Harris et al. 1996, Nittrouer and DeMaster 1996 and references therein), grappled with the estuary as dynamic system (Morris et al. 1982, Morris 1985, Kjerfve and Wolaver 1988), or dealt with the estuary as the axis for myriad interactions influenced by its catchment, sediments, neighbouring coastal waters and the atmosphere (Morris 1985, Schoer and Duwe 1986).

From the outset, we decided to develop a number of observational strategies to cover the important spatial and temporal scales. Not all time and space scales are necessarily of interest. If it is decided that certain scales of variation are not relevant, they can be treated as noise. However, sampling strategies must take account of this noise, and ensure that the desired signal does not end up masked, aliased or biased. Concurrently, we looked to have an interdisciplinary study — integrating physics, chemistry and biology — to improve understanding of the ecological functioning of the estuary. The benefits of such an approach have been advocated previously (Panel on Estuarine Research Perspectives et al. 1983).

The basis for our design of the Huon Estuary Study is presented in this chapter. It begins with a description of the study area, the Huon Estuary and its catchment, and reviews past investigations. The basic elements of our study are then presented, and their part in an integrated, interdisciplinary scientific investigation of the waterway is discussed.

2.2 Study Area

2.2.1 The natural environment

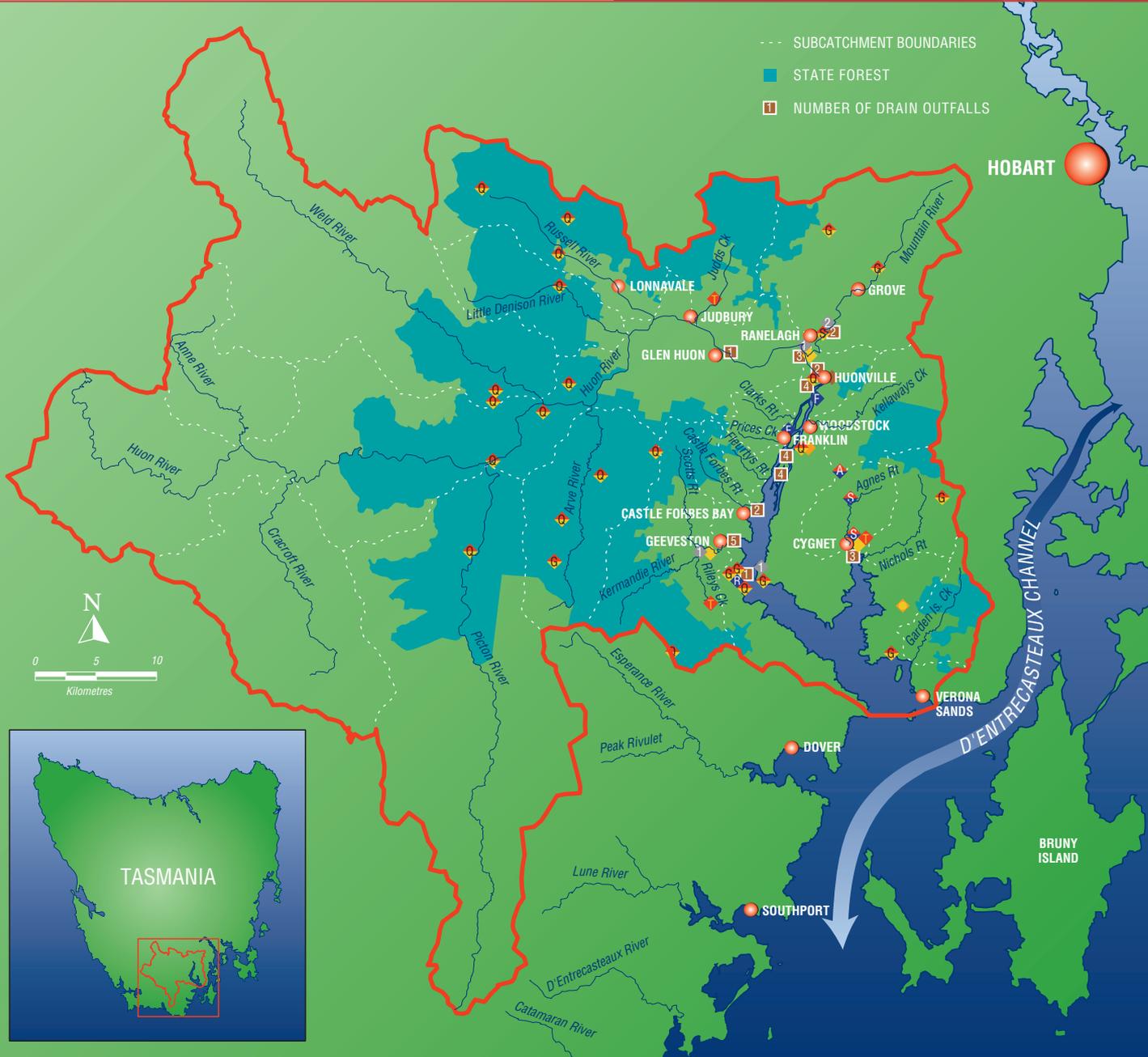
The Huon River estuary and its catchment are in southern Tasmania (Fig. 2.1) between latitude 42° 45' S and 43° 45' S, in a maritime climate that is dominated by zonal westerlies that produce changeable, cool temperate conditions. Average maximum air temperatures along the estuary shore range from 11°C in July to 22°C in January and February. The average annual rainfall ranges from over 2000 mm in the west of the catchment to slightly under 800 mm in the east. Rainfall is relatively uniform throughout the year, peaking July to October. The catchment area (Fig. 2.1) is about 3130 km². In the west highland plateaux and slopes are interspersed with stony crests and ridges, and rocky mountain tops in excess of 1200 m, falling away to rolling hills, gullies and valleys, and then undulating plains down to the river flats. In the east, the topography trends from rolling hills through lower slopes and gullies to drainage flats. Finally, along the estuarine shores and on the coast are marshes, beaches and rocky shores.

Geology, soils and vegetation all influence run-off — both quantity and quality — and, therefore, conditions in streams, estuaries, and even coastal waters. These aspects of the Huon catchment are summarised here. It is distilled from a number of key references: Mather (1955), Kirkpatrick and Dickinson (1984), Davies (1988), Pemberton (1989) and Gallagher (1996).

A simplified description of the geology of the catchment notes ancient Precambrian and some Cambrian formations in the west, typical of the southwest of Tasmania. These are mostly unmetamorphosed (sandstone, siltstone, conglomerate, etc.), but with metamorphic quartzite along the Arthur Range (southwestern boundary). They have been extensively eroded by running water and glaciers. In the central and eastern zones of the catchment, younger Permo-Triassic sediments (Parmeener Supergroup — mudstone, sandstone and shales) are intruded by Jurassic dolerite, and in a few local areas, by Tertiary basalt. The dolerite forms many of the high-altitude features. Quaternary deposits (alluvial and estuarine deposits of sand, silt, clay and gravel) are found along the valley floors of the main watercourses in the east and northeast. The hills between Cygnet and the Huon Estuary are a local anomaly, with Upper Carboniferous glacio-marine sediments intruded by Cretaceous syenite.

Soils have not been studied in detail in the Huon catchment, apart from a small area of agricultural interest, and sampling points for surveys in the *Land Systems of Tasmania* series (Davies 1988) and the *Forest Soils of Tasmania* handbook (Grant et al. 1995). The range of climatic, geological and topographic conditions favours diversity. Soils of the region vary from acidic to slightly alkaline. Their nutrient concentrations range from low to medium for total nitrogen, and from low to high for total phosphorus (Grant et al. 1995). The soils formed on sedimentary rocks of the Parmeener Supergroup are impoverished in both nitrogen and phosphorus. Peats are found in waterlogged soils with high rainfall, high humidity and low temperatures. This soil type is extensive in the west of the catchment, but also occurs on drainage flats near rivers, and in salt marshes of the lower-rainfall areas in the east. Deep peat layers might be expected in parts of the salt marshes around Franklin and Egg Islands (by analogy with the Boyer Marshes on the adjacent Derwent Estuary). Run-off from peat is

Figure 2.1.
The Huon Estuary and its catchment



LICENSED ACTIVITIES

(Data from April 1994)

- ◆ QUARRY
- ◆ GRAVEL
- ◆ SAND
- ◆ SAWMILL
- ◆ TIMBER PROCESSING & SAWMILL
- ◆ RENDERING PLANT
- ◆ FRUIT PROCESSING
- ◆ ABATTOIR
- ◆ SLAUGHTERHOUSE
- ◆ TIP
- ◆ SEWAGE TREATMENT PLANT
- ◆ SCREENING PLANT
- ◆ WASTE TRANSFER
- ◆ SEPTIC WASTE

strongly coloured with dissolved humic material, which can limit light penetration in river and estuary waters. Another issue for water quality — when vegetation is cleared — is the vulnerability of soils overlying Parmeener Supergroup geology to sheet, rill and gully erosion, which results in increased turbidity in streams.

The variability of the study area as described above produces a complex mosaic of vegetation types. Kirkpatrick and Dickinson (1984) identify at least 15 vegetation types across the Huon catchment. In broad categories, the headwaters of the Huon River in the far west are in buttongrass moorland interspersed with wet scrub. Crests and ridges have a lower band of *Eucalyptus coccifera* (snow gum), and are capped with central alpine vegetation complex. The central catchment is mostly *E. obliqua* (messmate stringybark) forests with pockets of rainforest, and small islands of moorland and wet scrub. Stands of *E. delegatensis* (gum-topped stringybark) are along the higher slopes of the north and northeast. Forestry Tasmania manages much of this corridor of State Forest (~125,400 hectares) for forestry operations. A patchwork of *Eucalypt* dry forest, *E. obliqua* wet forest, grassy and sclerophyll forests are found in the east. Land cleared for agriculture is mostly along the river flats and the rolling hills on either side of the estuary.

The Huon River's mean annual flow (measured just upstream of Judbury — Fig. 2.1) is $87 \text{ m}^3 \text{ s}^{-1}$, with monthly average flows of between $30\text{--}40 \text{ m}^3 \text{ s}^{-1}$ (Jan-Mar) and $125\text{--}130 \text{ m}^3 \text{ s}^{-1}$ (Jul-Aug). One-in-seven-year floods have flows in excess of $1300 \text{ m}^3 \text{ s}^{-1}$. Diversion of Huon River headwaters above Scotts Peak Dam (for hydro-electric power generation) for the past two decades has reduced the annual discharge of the Huon River system from 3000 to 2600 million cubic metres (Gallagher 1996, and references therein). Livingston (1995) estimated that after the dam was built, the median flow decreased by 15% and low flows by about 8%. He also examined the frequency and size of pre- and post-dam floods in the Huon River, and found a reduction in flooding in the Huon River consistent with the reduction in catchment area above Scotts Peak Dam.

Many streams enter the Huon Estuary below Judbury, but their relative discharges are all very small compared to the main river. Two rivers — the Mountain and the Kermandie — are the largest of these tributaries, but their individual annual discharges are less than 3% of that of the Huon River at Judbury (Wotherspoon et al. 1994, Bobbi 1998). Nevertheless, volume of water is not necessarily a good indicator of the load of solutes or suspended particulate matter delivered by different tributaries. This matter is addressed later in the report.

The Huon Estuary, like others in southern Tasmania, has been formed from the drowning of a river valley. The water flows from the estuary into the D'Entrecasteaux Channel between mainland Tasmania and Bruny Island. The entrance to the estuary is 20 km from the Southern Ocean in the south and 40 km to Storm Bay in the North (Fig. 2.1).

The Huon Estuary is characterised as a microtidal salt-wedge estuary; the salt water can penetrate (under low river flow) as far as Ranelagh — some 40 km upstream from the mouth of the estuary at Huon Island. The estuary can be partitioned into three geographic regions: (i) a deeper marine zone in the lower half of the estuary, (ii) the Port Cygnet arm, and (iii) a shallow brackish zone in the upper half of the estuary. The lower half of the estuary, from the mouth at Huon Island to Port Huon on Figure 2.2, has a length of about 21 km and a depth ranging from

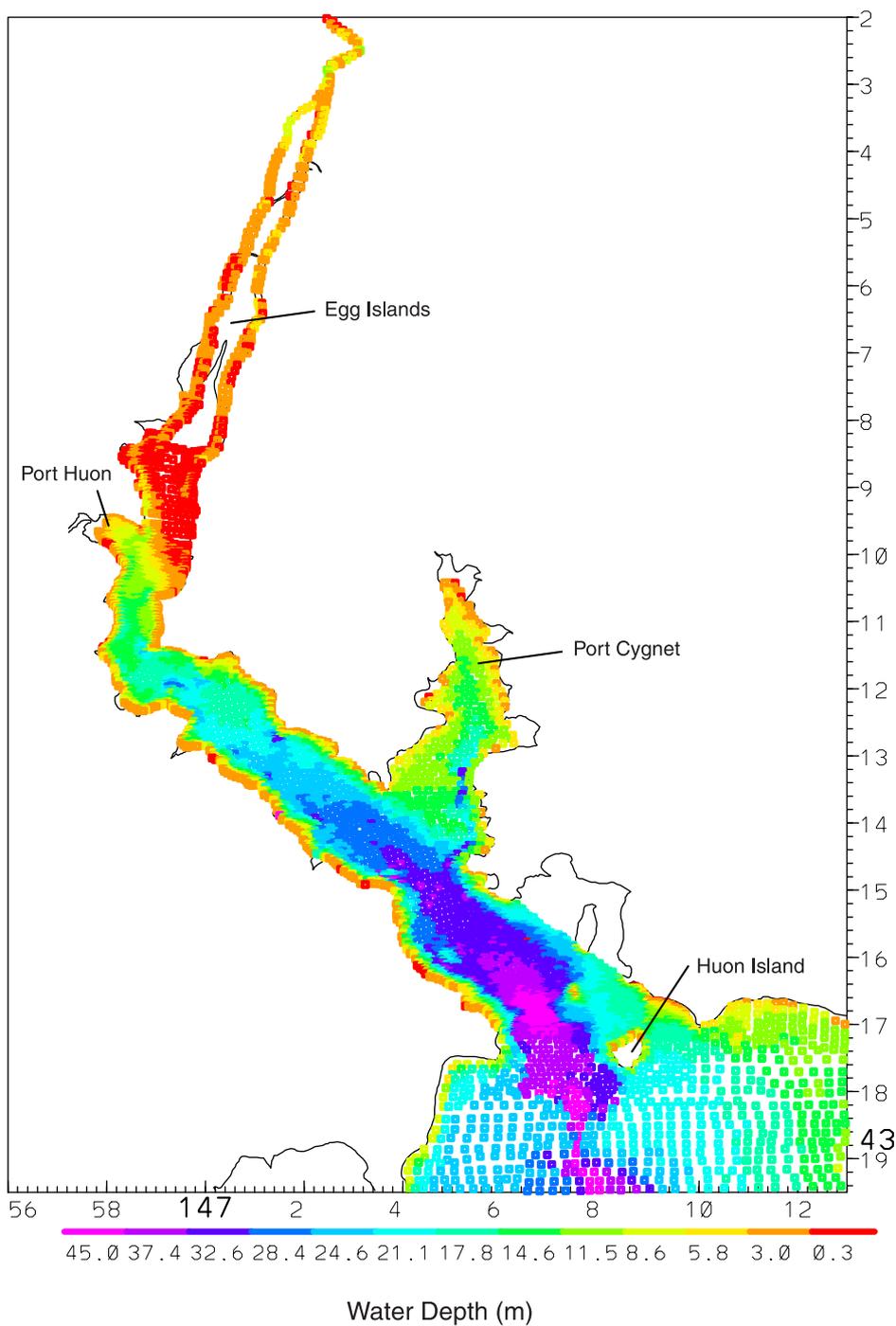


Fig. 2.2 Bathymetric map of the Huon Estuary and part of D'Entrecasteaux Channel. Isobaths (m) indicated on colour scale beneath map, and the scale on the axes is latitude and longitude.

about 40 m at the mouth to 10 m at Port Huon. The Port Cygnet arm is mid-way up the lower estuary; it is shallow, but marine-dominated, with very little freshwater input from the Agnes and Nicholls rivulets. The main estuary above Port Huon shoals rapidly from 10 m at Port Huon to a mean depth of around 4 m on the western side, and 2 m on the eastern side, of the Egg Islands. The deep salt-water layers penetrate another 19 km upstream to near Ranelagh (Fig. 2.1), whereas the surface waters range from brackish to freshwater under the influence of the Huon River.

2.2.2 Human activities in the catchment

The region has always been sparsely settled. The present population is approximately 13,000; of the main towns — Huonville, Cygnet, Geeveston and Franklin — only Huonville exceeds 1000 inhabitants. Since settlement over 160 years ago, commercial activity in the region has centred on forestry and agriculture (mainly horticulture, grazing and dairying) (Hammond 1995). These activities spawned associated industries, such as sawmilling, shipbuilding and fruit processing, along the banks of the estuary from late last century. Transport for the region was mostly provided by the waterway, so shipping itself was also an important activity. The estuary at one time had two important ports serving national and international markets (Port Huon and Port Cygnet), as well as a multitude of smaller wharves and jetties.

The “largest sawmill in Australasia” was at Whale Point on Hospital Bay (Port Huon) early this century (Row 1980). At its peak in 1922, a tramway network over 60 km long fed the mill from a timber concession in the hills around Geeveston. The sawmill employed 170 people and was cutting 22 cubic metres per day. An aquaduct from the Kermadie River supplied water to the mill to wash sawdust into Hospital Bay (D. G. Geeves — *pers. comm.*, November 1999). Later in the 1920s, the more easily accessible forests were cut out and the industry (which was never really profitable) declined and then closed in 1929 (Row 1980). On the same Whale Point site, a new secondary industry grew from eucalypt pulping trials before the Second World War by the Council for Scientific and Industrial Research (the predecessor to CSIRO). A neutral sulfite semi-chemical pulping mill (Australian Paper Manufacturers) operated there from 1962 to July 1991 (interim closure 1982–1986). It ceased operation when world prices for unbleached pulp fell by 30%. The waste from this type of mill was mostly suspended solids and organic matter (causing both biological oxygen demand and discolouration).

The legacy of wood processing on the shores of Hospital Bay for much of this century has left some environmental degradation (Chesterman 1995). Deposits of sawdust have changed the western foreshore markedly, filling in what was originally watercourse. In a 1980 survey, wood fibres and other wood-derived particles were also evident in sediments throughout Hospital Bay and out into the main arm of the estuary (Jones and Lawson 1980). While the pulp mill was operational, its discharge caused the surface sediments of Hospital Bay to be a soft, sticky mud that reeked of hydrogen sulfide. This “anaerobic blanket” has declined from a recorded 72 ha of bay sediments in 1982 to about 1 ha about the pulp-mill wharf in 1996 (Hyde 1996).

For much of the last decade, the Huon Valley has not had any secondary industry. Instead, the new enterprises of aquaculture and tourism have emerged to complement the established, but still evolving, horticulture and forestry. Along with this shift in commercial activity has come the realisation that a healthy environment is important for the future of the region.

Several current activities and facilities throughout the Huon catchment (Fig. 2.1), including marine farms (Fig. 2.3), have the potential to degrade the region's waterways. Historical water-quality data, although sporadic and not comprehensive, "indicate no serious water quality problems in the Huon region [including the estuary]" (Gallagher 1996). A recent problem for environmental management of the Huon Estuary was the introduction of *Gymnodinium catenatum* to the waterway in the 1970s (McMinn et al. 1997). This dinoflagellate is one of the causative organisms of paralytic shellfish poisoning, and blooms of this species have resulted in closures of shellfish farms in the estuary for periods of weeks and longer almost every summer and autumn since 1986.

The estuary has recently been evaluated for its conservation significance using ecological and physical attributes and assessment of human pressures in a classification of Tasmanian estuaries (Edgar et al. 1999). It was classified as Class C (moderate conservation significance); though having a higher than usual human population density in its catchment for that grading, this was outweighed by high population densities of aquatic fauna and "high fish and invertebrate diversity".

2.2.3 Other factors influencing the estuary

At the beginning of any estuarine study, the environmental setting or context of the water body must be examined. For example, are there often strong winds during a part of the year that cause sediment resuspension? Conversely, are there some months when calm periods favour microalgal blooms? What is the annual mean river flow, and when typically are peak flows? With this basic information covered, another question follows. What information is needed to discern the influence of external factors on an estuary, especially if we are looking to model it? Obvious influences are winds, river flows and tides, but storm surges and influx of groundwater are among the less obvious.

Winds

With respect to wind data over the Huon Estuary, we are limited by the absence of any regular meteorological station along its shoreline. Data from temporary installations (e.g. Huon Aquaculture Company's fish farm at Hideway Bay) appear very scant.

The nearest meteorological stations in the Huon catchment are at Geeveston (Forestry Tasmania, Cemetery Road, 43° 10' S, 146° 55' E, elevation 60 m), and an automatic weather station about 13 km WSW of Geeveston, in the Hartz Mountains (Keoghs Pimple, 43° 12' S, 146° 46' E, elevation 830 m). The meteorological station on South Bruny Island at the other end of D'Entrecasteaux Channel (Cape Bruny/Lighthouse, 43°29' S, 147° 09' E, elevation 55 m), 22 km south of Huon Island, is useful for zonal wind patterns. There is also a nearby station at Dover (Forestry Tasmania, Station Road, 43° 19' S, 147° 01' E, elevation 16 m) 8 km WSW of Huon Point. The Grove Research Station (Department of Primary Industries, Water Environment, 43° 19' S, 147° 05' E, elevation 60 m), 4 km NNE of Huonville on the Mountain River, also independently takes meteorological measurements.

Local winds are strongly influenced by the hilly topography of estuarine shores. A small network of automated stations in some estuaries could describe wind fields sufficiently, but it seemed unlikely in the Huon Estuary that we could deploy sufficient monitoring systems to

resolve the complexity. If small-scale winds were required (e.g. for 3-D hydrodynamic models), a limited network of field stations with interpolation from local wind models could suffice.

Run-off

A gauging station at Frying Pan Creek in the upper Huon River (maintained by Tasmanian Department of Primary Industries, Water and Environment) provides good records of flow for the main tributary. During our study, data from Rileys Creek, a tributary of the Kermadie River were also available for periods. An estimated scaling factor for the Kermadie River (at our site R6) is 7.1 times the flow at Rileys Creek station (Bobbi 1998). The simplest way to estimate flows on ungauged streams, and ultimately for the entire catchment, is by analogy with adjoining gauged subcatchments corrected for differences in rainfall and subcatchment area. However, some subcatchments can be quite different (in our case, the Port Cygnet catchment that includes Agnes and Nicholls Rivulets); intermittent flows in small ephemeral streams might be measured by snapshots with portable devices.

Groundwater

Water delivered from the land to the estuary can also come directly from groundwater. Little has been published on groundwater in the Huon region — either its quality or its quantity. Leaman (1967) reported on the groundwater of the Cygnet District. Apart from one bore contaminated with seawater, he found all groundwater in the district to have been derived from rain and snow. It had a relatively low total salt content, and there had been no contamination from other sources (naturally high levels of iron made it unsuitable for domestic purposes). Leaman (1967) estimated that if 20% of the precipitation in the catchment were added to the groundwater reserve, it would amount to 5000 million gallons ($\equiv 19$ million m^3) annually. The same volume would enter the estuary each year from this subcatchment of the Huon Estuary if an assumption that the groundwater reservoir is at a steady state were valid. Much of the groundwater would contribute to the base flow of streams in the Cygnet district. Its influence on water quality would be accounted for in the monitoring of the streams. If, however, an aquifer discharged directly to the estuary, its influence on estuarine water quality would need to be estimated separately.

Tides

In the absence of an operational tide gauge in the Huon Estuary, it is pragmatic to estimate tides in the lower estuary from either the tidal record or predictions for the Port of Hobart, which has the nearest tide gauge. Comparison of tidal records from a previous gauge installation at Port Huon with that of Hobart indicates that the tide at Port Huon is just one minute in advance of Hobart, and its amplitude is typically 83% of the reference (P. Davill, National Tidal Facility — *pers. comm.*, October 1999). Short-term gauging higher up in the Huon Estuary recorded a mean tidal amplitude of 0.9 m for the Huon River at the Huonville Bridge (reference in Gallagher 1996).

We have found no information on the frequency or magnitude of storm surges in the Huon Estuary.

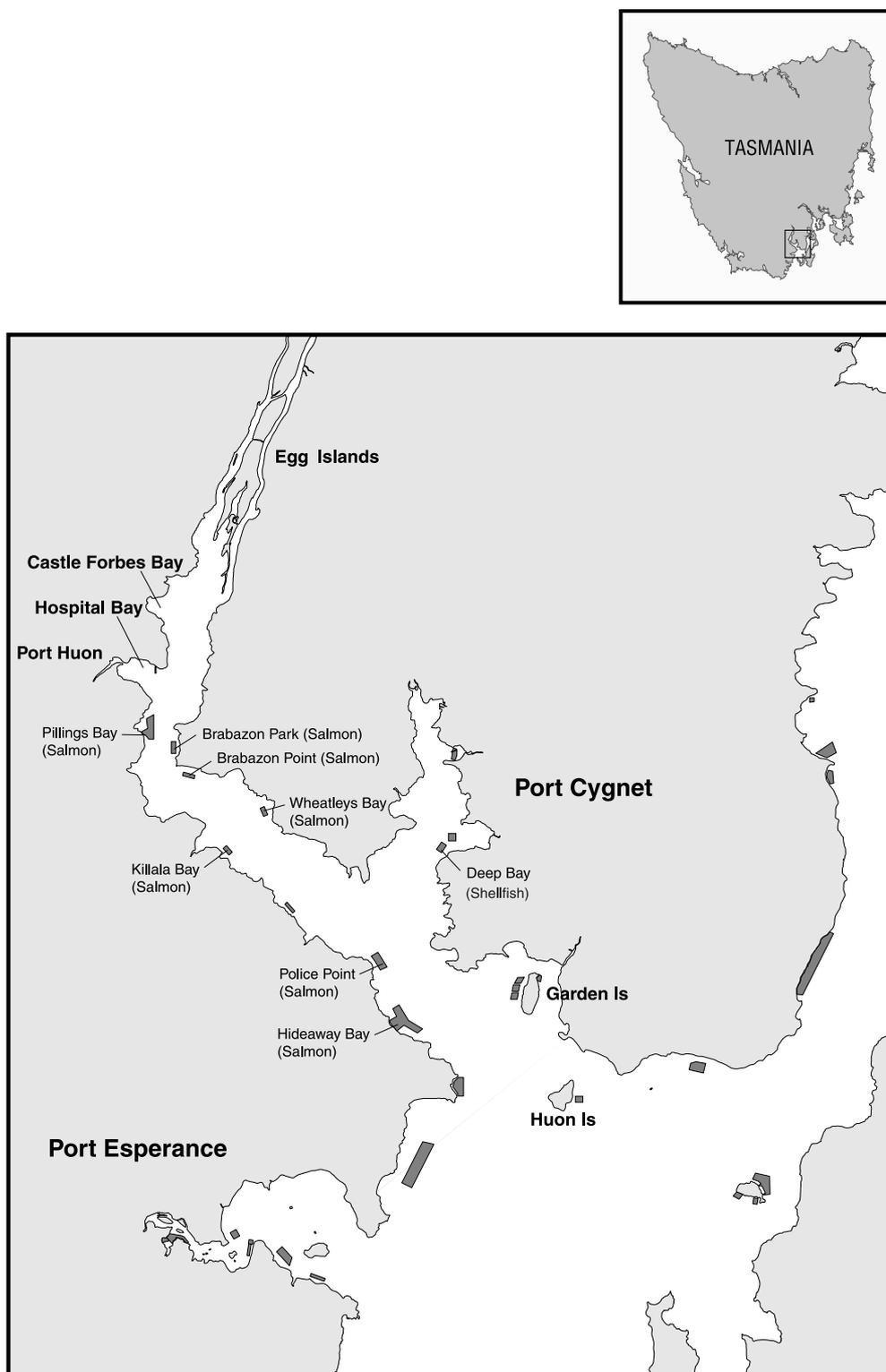


Fig. 2.3 Marine Farming Zones (incorporating existing lease sites) in the Huon River and Port Esperance Marine Farming Development Plan depicted as shaded areas. Zones in D'Entrecasteaux Channel, to the east of Huon Island, are part of a different Marine Farming Development Plan. Drawn from information supplied by DPIWE as of April 1999.

Bathymetry

Both part of the estuary and external to it, bathymetry is a key characteristic in determining estuarine hydrodynamics. We obtained soundings data from charts (1931–1965) produced by the Hobart Marine Board and the Royal Australian Navy. The bathymetry of the Huon Estuary and D'Entrecasteaux Channel (Fig. 2.2 and Lands Department 1984) shows a sill depth of around 15 m to the north in the channel, and from there northward to its opening to Storm Bay: its maximum depth is between 10 and 15 m. In contrast, to the south, a canyon always deeper than 30 m feeds directly to the narrow continental shelf off southeastern Tasmania. This feature suggests that the deep marine waters in the Huon Estuary could be influenced directly by waters funnelling in from the Southern Ocean.

2.3 Previous Investigations of the Huon Estuary

Previous hydrographic surveys in the Huon Estuary (Woodward et al. 1992, Burgess et al. 1993) had established some of the broad physical and chemical characteristics of the estuary. These included the limit of penetration of saltwater (up to the township of Ranelagh), and the complex hydrodynamical features of the middle estuary around the elbow below Port Huon. Some insight into the seasonality of nutrients and chlorophyll concentrations was also provided. A recent compendium of historical and current environmental information on the Huon River and Catchment (Gallagher 1996) also proved invaluable in early planning.

An earlier pilot study at Killala Bay in the Huon Estuary by CSIRO Division of Fisheries (April–July 1995 — Clementson et al. 1997) provided a useful starting point for monitoring short time-scale processes. This work is discussed further in the next section.

2.4 Development of a Survey Strategy for the Huon Estuary Study

This section describes the scientific approach to the Huon Estuary Study and gives an overview of the study's design. Information on field equipment, sampling procedures, laboratory instrumentation and methods is all detailed in appendices.

Our study began with a pilot, three-day survey of the entire Huon Estuary in March 1996. It comprised 86 stations at 59 sites, each of which were profiled using an oceanographic conductivity-temperature-depth (CTD) sensor (analogous to a physical station in Subsection 2.4.2). The main objective of this fieldwork was to characterise the estuary at about the resolution that we wanted for our regular spatial surveys of the full estuary. The pilot survey was also an opportunity to test a variety of instrumentation and sampling equipment, as well as chemical and biological field techniques. The disposition of stations is shown in Figure 2.4. A number of sites were re-occupied at different stages of the tidal cycle and on different days.

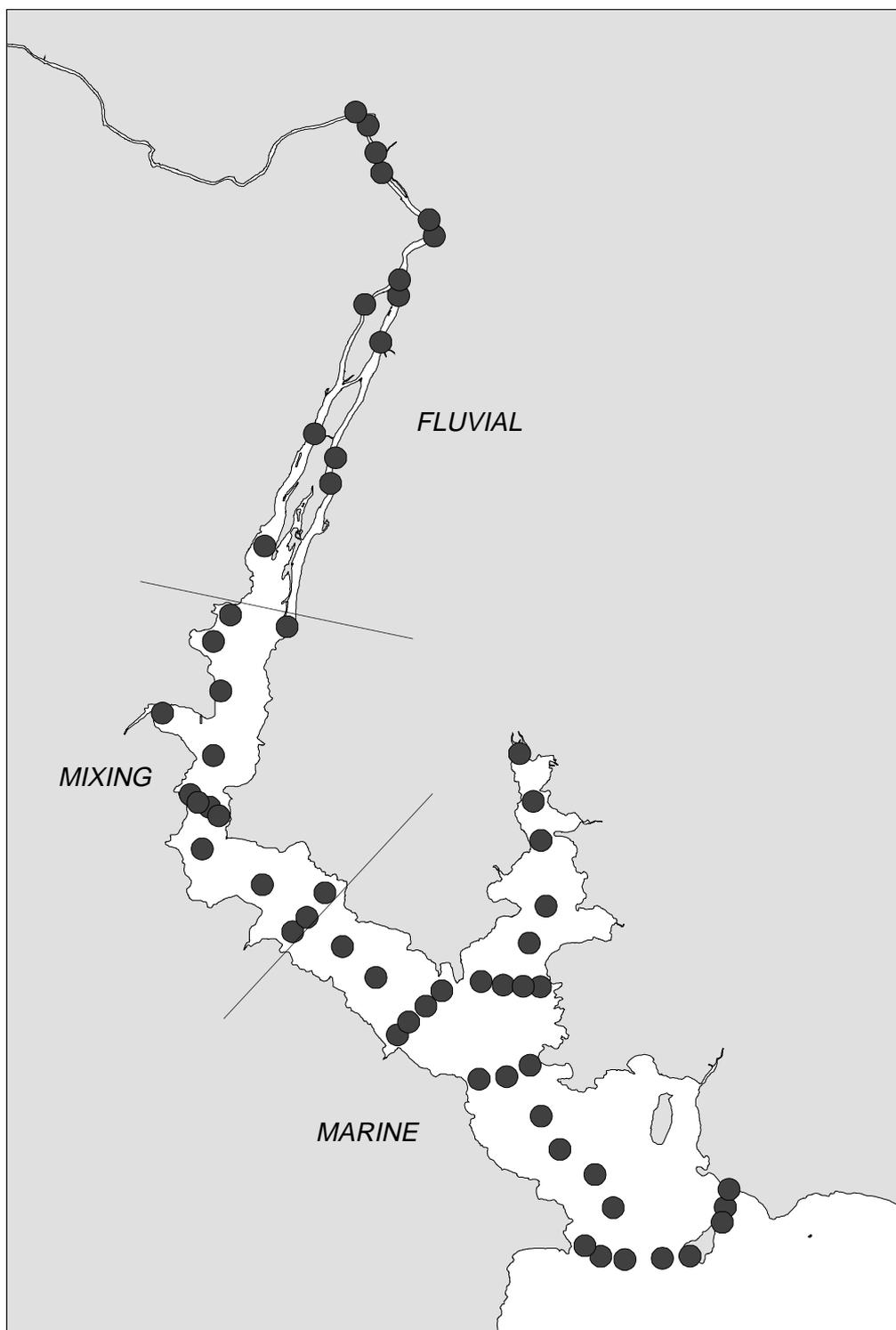


Fig. 2.4 Sampling sites are shown for the pilot survey (HES 1) of the Huon Estuary in March 1996. The approximate boundaries for the three zones of the estuary, fluvial (upper), mixing (middle) and marine (lower) are also shown. These will vary with stream discharge.

Results from the pilot survey (e.g. Fig. 2.5 – Morgan et al. 1996) and data from historical surveys of the estuary were our founding information. They were used in discussions within the HES project team, and also with research collaborators, to begin detailed planning for the spatial surveys and other monitoring activities.

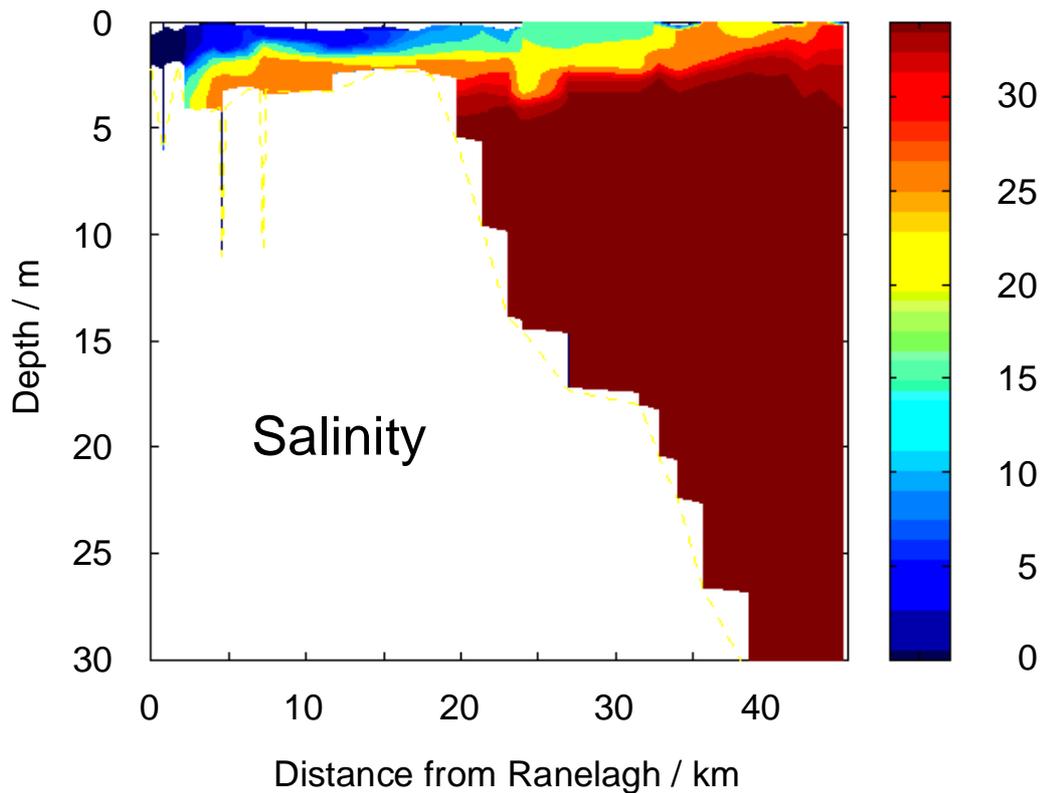


Fig. 2.5 Example of a survey result from HES 1: salinity section along the principal axis of the estuary. The salinity scale is shown as a colour gradation on the right.

Decisions as to what measurements to make in HES were determined mainly by the study objectives, and the need for fundamental measurements for mixing indices, water quality, sedimentary types, and conditions for microalgal growth. The core suite of measurements is presented in Table 2.1.

Table 2.1 Core set of measurements made in the water column during HES surveys, using either profiling instruments or discrete measurements in the laboratory.

HES — CORE WATER COLUMN MEASUREMENTS

temperature	dissolved reactive phosphorus
conductivity / salinity	total dissolved phosphorus
turbidity (OBS)	total phosphorus
dissolved oxygen	dissolved reactive silicon
<i>in situ</i> chlorophyll fluorescence	dissolved organic carbon
suspended particulate matter (+0.45 µm)	chl a
nitrate	spectral absorbance (350–700 nm)
nitrite	light quality (PAR or spectral irradiance)
total ammoniacal nitrogen	HPLC pigments (chlorophylls and carotenoids)
total dissolved nitrogen	net plankton (+20 µm, 10-m vertical haul)
total nitrogen	plankton samples (Lugol's preserved)

2.4.1 Geographic coverage — spatial surveys

The traditional approach to study design is the “whole-of-estuary survey”. As summarised by Morris (1985), “geographical sites are fixed according to the salinity gradient, and spanning the entire salinity range, including freshwater sampling”. More closely spaced sampling is made in selected regions to resolve known major inputs or other important influences. One-, two- or three-dimensional sampling is chosen according to estuarine type and investigational aims.

A survey of the entire estuary as described gives valuable information on (i) mixing processes, and behaviour of dissolved and particulate matter over the estuarine gradient, (ii) broad geographic distribution of properties, and (iii) if three-dimensional sampling is used in stratified estuaries, differences in properties (e.g. transport processes, chemical reactivity and microalgal distribution) between surface and deeper layers can also be discerned, and possibly inferred for sediment-water exchange of solutes.

Estuarine bounds

The marine boundary of the Huon Estuary was fixed across the opening to D'Entrecasteaux Channel (Huon Point - Huon Island - Ninepin Point) (Fig. 2.6). This is a functional limit. Other researchers have sought to use a mixing tracer (e.g. alkalinity–salinity relations — Muller et al. 1995) to define the extent of an estuarine plume in coastal waters. The freshwater end-member was selected at a gauging station (automated turbidity measurement) at Judbury, just upstream

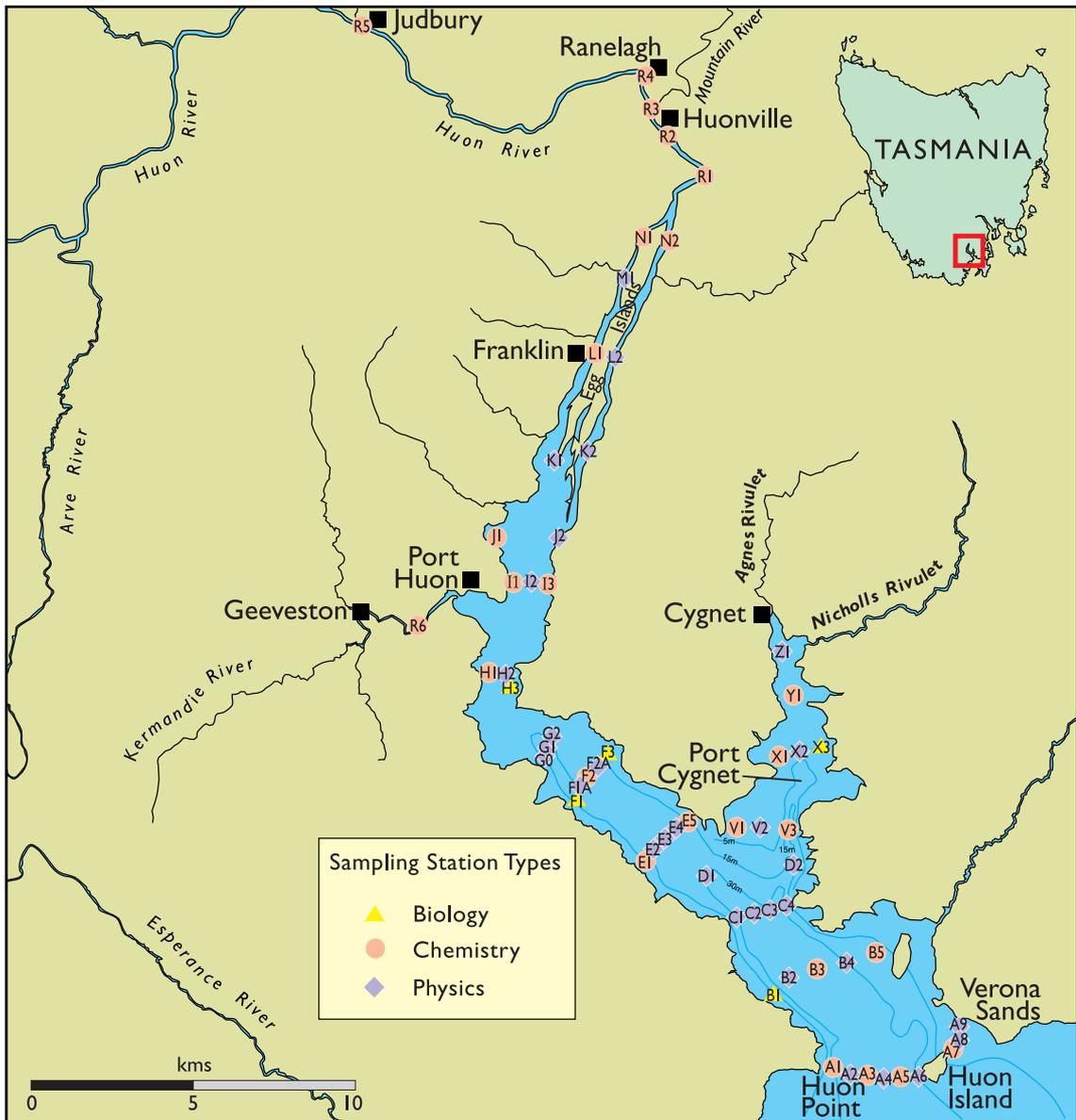


Fig. 2.6 The sampling sites for the Huon Estuary Study. The key to station types is indicated; biology stations were also those used in weekly / fortnightly monitoring.

of the first shallow rapids. This overcame two difficulties: the maximum penetration of saltwater (i.e. toe of salt wedge) was not known with certainty in the vicinity of Ranelagh, and locating a zero salinity by boat would be time consuming. The Judbury site (R5) was sampled by hand from the river bank, as was a freshwater end-member on another tributary, the Kermantie River (R6).

Disposition of stations

Survey stations were spread evenly throughout the estuary paying heed to inputs of significance, outfalls from sewage treatment plants, finfish aquaculture leases, major tributaries and population centres. We also identified at the outset two important horizontal gradients: the principal axial (or longitudinal) gradient from freshwater end-member to the marine end-member, and cross-estuary gradients. We sought to gain good cross-sectional information throughout the middle and lower estuary through a series of transverse sections. Tie-points for these sections were quite often narrow points of the estuary or aquaculture leases. Additional centre-line stations were included to resolve the axial gradient better. In the upper estuary, individual sites were chosen to delineate features (e.g. flow around Egg Islands, urban inputs, deep holes and tributaries).

Types of stations

The complexity of estuarine distributions of salinity and temperature was evident in historical observations (Woodward et al. 1992) and our own pilot survey. We needed a dense array of physical (hydrographic) measurements to both resolve these features and provide a basis for box inverse models of the estuary. A total of 63 stations was the balance struck between sufficient resolution and operational capacity in both the field and laboratory. Of these, 30 were also chemical stations (chemical analysis of water samples for nutrients and other water-quality parameters); and five of the chemical stations were upgraded to biological stations (enhanced characterisation of microalgal ecology) (Fig. 2.6).

Station locations were all fixed. The many sites of interest (i.e. point sources) throughout the estuary were influential in this decision, as was the utility of a grid of stations for developing box inverse models. A disadvantage was that samples for chemical and biological analysis would not cover the salinity range evenly to describe the estuarine gradient fully. Morris (1985) recommends covering the estuarine gradient in steps of one or two salinity units. We used a mix of fixed geographic location and salinity-determined stations in recent work in the Derwent Estuary (Plaschke et al. 1997) as a compromise.

Vertical resolution

At each station in a salt-wedge estuary, it is common to take just single samples from the two layers — surface and bottom. However, since *in situ* fluorescence measurements during the pilot survey suggested that phytoplankton biomass could be concentrated at or near the pycnocline, we opted to also sample at this feature, or at an obvious midwater fluorescence maximum. This would provide a sample with the maximal biological imprint, from an intermediate depth (henceforth referred to as the *mid-depth* sample).

Zonation of estuary

It is impossible to survey the estuary as described above in a single day, and even quite difficult in two days. Furthermore, the estuary cannot be practically divided into two. By carrying out the survey over three days, the appropriate partition of lower estuary (including Port Cygnet) — the marine zone, middle estuary — the mixing zone, and upper estuary — fluvial zone is realised (see Fig. 2.4). Such a zonation is relevant to most estuaries. Other advantages of this arrangement are (i) more flexibility to avoid bad weather in the lower estuary, (ii) more scope to carry out small additional tasks as the opportunity presents, and (iii) a manageable number of samples to be filtered and processed, during and at the end of each day.

Surveying the estuary over three days has one obvious drawback: there may be mismatches in data records from one day to the next, induced by tides, changes in river flow, winds and other external factors. We dealt with this problem to some degree by re-occupying a number of physical stations on different occasions during the survey. Furthermore, we also made more rapid surveys of the full estuary by other means. These are described below in Subsection 2.4.5.

Survey frequency

We chose to carry out the intensive estuary-wide surveys, as described above, at quarterly intervals to track seasonal changes in the Huon Estuary. These surveys would reveal changes in estuarine conditions associated with annual cycles, such as those of temperature, light, river discharge and biological activity. They would also show the relation of water quality and biological parameters to estuarine mixing (with salinity as the mixing index). They should also resolve geographic variations and anomalies in these same properties, influenced perhaps by discrete or diffuse inputs.

Nevertheless, in order to understand the full functioning of an estuary in an interdisciplinary study, it is essential to observe it over different temporal and spatial scales. Intermittent whole-of-estuary surveys fail this by missing out on spatial resolution (in our case, 63 sampling points over a surface area of 77 km²), short time-scale events (hourly, daily, weekly, ...), the importance of interfaces (air–sea, halocline, and sediment–water) and influence of adjoining environmental compartments (catchment — terrestrial and fluvial, coastal seawaters, atmosphere and sediments). Recognising the limitations of the quarterly seasonal surveys, we chose to supplement them, or integrate within them, other sampling procedures to cover different temporal and spatial scales. These procedures are described below.

2.4.2 Short time-scales — monitoring at key stations

Week by week is a useful interval to track succession in microalgal blooms (although not the history of an individual bloom), and it approximates the period of short-term cycles in weather. Moreover, weekly monitoring provides information on trends within a season — whether, for instance, there is a sharp step in a property at the change of seasons, or if there is a gradient over the duration of the season. However, manual sampling as for a chemical station (see above) and subsequent laboratory measurements were only manageable on a weekly or fortnightly rotation if they were done at a subset of stations in the whole-of-estuary network.

The five biological stations of the spatial survey were selected as the “key” stations. Although these five are in the middle and lower estuary, they bracket the region in which aquaculture is found. Furthermore, the Huon Estuary can be classified as a “biochemical” estuary (Church 1986): its small tides and low concentration of suspended particulate matter cause the geochemical (particulate) processes affecting solutes in the upper estuary to be overwhelmed by the biological activity in the lower estuary.

The five key stations could all be surveyed from a small boat in about three hours, giving a weekly or fortnightly snapshot of the estuary under close to synoptic conditions. Sampling at each station was also at three depths, but unlike the spatial survey the three depths were always fixed for each station (i.e. the mid-depth was not selected by reference to the CTD profile, as for spatial surveys). It was not practicable to download and view CTD profile data on the small open dinghy furnished by the fish-farm company (Tassal Ltd) for this work.

The fifth biological station was located at first in Crowthers Bay (Station J1, Fig. 2.6). This site was often within a high salinity anomaly. After HES 3 (October 1996), this station was moved to Wheatleys Bay (Station F3) to give improved cross-estuary definition, when compared to the station at Killala Bay (F1). The key station in Port Cygnet for weekly / fortnightly monitoring was not at, but close to, site X3 at a location in Deep Bay. It was given the site code X3B (from HES 6 onward the biological station for the seasonal estuarine surveys also moved to X3B, and X3 was downgraded to a physical station).

2.4.3 Shorter time-scales — autonomous continuous profilers

Conditions that influence the genesis of the bloom of an individual phytoplankter, the onset of oxygen-depleted conditions beneath fish cages during summer, and variation in solar irradiance and light penetration are all examples of processes that can change in the estuary from hour to hour. On time-scales of hours to days, changes in tide, wind and wave climate affect the pattern of circulation in estuaries, the location of fronts, and the extent of vertical mixing. In shallower waters, the perturbations extend to sediment resuspension and transport. This type of information cannot be collected manually, particularly if it is used routinely as input for numerical models.

Automated monitoring systems are the most practical way to gather short time-scale data. Such instruments can also capture rare or sudden events, for which it is impossible to plan surveys using conventional methods. An example is riverine discharge. Many Australian estuaries receive most of their freshwater and nutrient loads in rare, large, run-off events (Eyre 1998). Sampling these extreme events can be essential to gaining a reasonable estimate of a riverine budget for nutrients and suspended particulate matter.

Instrumentation to gather high frequency data over the full water column grew out of an earlier demonstration project at a finfish farm in the Huon Estuary from 1 May to 8 July 1995 (Clementson et al. 1997). The subsurface package consisted of a conductivity–temperature–depth (CTD) profiler, light sensor, an *in situ* fluorometer, and a prototype underwater spectroradiometer. The package was deployed at Killala Bay at a depth of 3 m, and measurements were made and recorded at 30-minute intervals (60 minutes for the spectroradiometer). It could be made to record a depth profile on manual command.

A key lesson from this experimental work was the importance of resolving vertical variation in the estuary. Not only was this important to follow changes in physical characteristics of the water column, but also to track the vertical migration of dinoflagellates, which can only be measured over the full water column.

Two automated continuous profiling systems were developed in-house for HES (Fig. 2.7). Each system raises and lowers a sensor package at intervals of about 1 hour obtaining data with a vertical resolution of ~ 0.5 m. The controller-logger unit and winch were mounted on a moored pontoon. These profiling systems were deployed with CTD, fluorescence, turbidity, light and dissolved oxygen sensors (Sherlock et al. 1999). We had originally intended to install the continuous profilers at each of the five biological stations to resolve the broad axial and transverse gradients in the estuary, but project funding proved insufficient. Operational units were installed at Killala Bay (Biological Station F1) in the middle estuary, and Hideaway Bay (Biological Station B1) in the lower estuary (see Figs 2.3 and 2.6). For ease of service and security, the instrument pontoons were moored at a corner of finfish-farm lease sites.

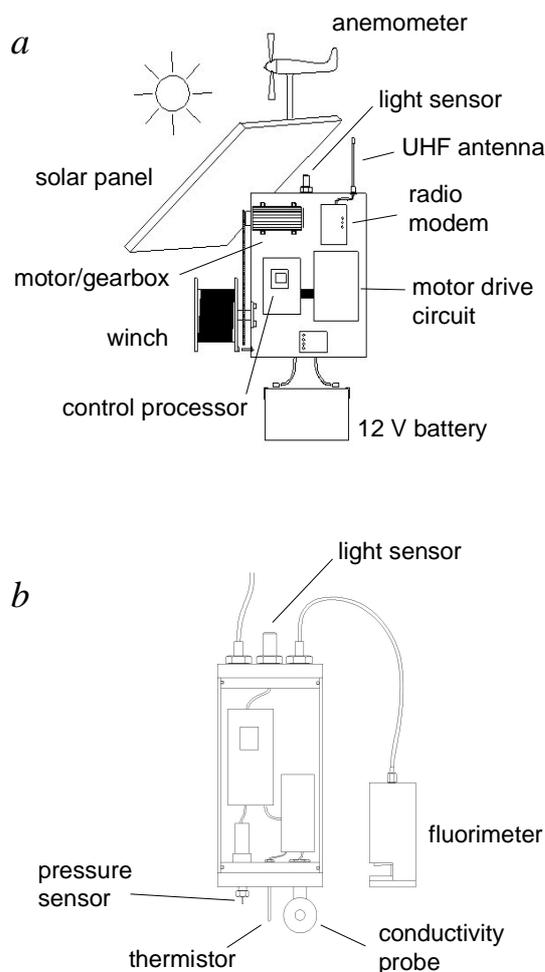


Fig. 2.7 Cutaway drawing of automated continuous profiler showing (a) surface unit that is mounted on a pontoon, and (b) sensor unit deployed by winch (from Sherlock et al. 1999).

We used the weekly or fortnightly manual sampling to calibrate and maintain the automated sensors. The calibration was made against the CTD units used for estuarine survey work. Regular maintenance to remove biofouling on the subsurface sensor package was necessary during the warmer months of the year.

2.4.4 Short time-scales and mapping of surface-water properties

The standard spatial surveys mentioned above took three days — one for each of the lower, middle and upper regions of the estuary. Variations in conditions at time-scales of one day or less are hidden in the survey data. On occasions, we took rapid physical (CTD only) profiles of the water column at a subset of the 61 estuarine stations in one day. This “physical snapshot” of the estuary would typically be done on the day preceding a standard spatial survey. For example, the entire estuary was covered in ~6 h with CTD profiles at 30 stations preceding HES 4 in February 1997. However, the tidal influence would still be present in these data.

To survey the whole estuary rapidly and with improved geographic coverage, we developed an intake system for use while our survey vessel was underway. The system operated while the vessel was moving forward at constant speed (typically 18 kts), with intake pipes mounted on the side of the hull at depths of 0.5 and 1.0 m. The intake manifold distributed water to flow-through cells of deck-mounted CTD units. Data compatibility with other surveys was assured by using the same CTD instruments that were used for profiling the water column. The full estuary from the mouth to just below Huonville could be mapped in 2–3 h, depending on the survey track. Using time as a reference, the sensor data were concatenated with simultaneously collected GPS information to provide maps of salinity, temperature and *in situ* fluorescence. A compact flow-analysis instrument is being developed to add nutrients to the array of mapped variables.

Although this mapping is limited mostly to the surface layer (depending on the depth of the pycnocline), it is these waters, as a buoyant brackish plume in the Huon Estuary, that are most changeable. They are influenced by transient forcing by winds, waves and short-term fluctuations in local run-off.

2.4.5 Long time-scales — sediment sampling and analysis

Estuaries act as a natural filter, retaining in their sediments much of the organic matter and minerals supplied naturally by rivers. Concentrations are often several orders of magnitude higher in the sediment than in the overlying water column, thus simplifying the analysis. A single estuary-wide survey will often suffice for gathering the required information.

Surface sediments provide an integrated picture of inputs over relatively short time-frames of a few years. Hence, they can indicate localised inputs in the context of an estuary-wide baseline, and also give a better view of long-term average inputs than what we can see in individual or even several snapshot surveys of estuarine waters.

Sediment cores provide a record, down their length, of estuarine conditions over years to decades and longer — depending on the sedimentation rate. The gradient with depth, especially for porewaters, yields estimates of the fluxes of solutes in or out of the sediment.

The selection of sites for sediment sampling used those of the spatial (water column) surveys as a basis. We were interested in relating conditions observed in the water column to those seen in the sediments. The 30 chemical stations provided a useful framework. Other stations were included on the basis of preliminary evidence of sediment type, or to provide better discrimination of local inputs (Fig. 2.8).

Interpretation of sediment data relies critically on a good characterisation of the sediment types and their distributions. Organic matter loadings and concentrations of contaminants are often strongly correlated with surface area and hence grain size (Mayer 1994), with low concentrations associated with sands and high concentrations associated with fine muds. Accordingly, we opted to determine a suite of primary characteristics (grain size, colour, loss on ignition, etc.), before more refined and specialist analyses were carried out.

A key issue in the Huon Estuary is organic matter in the sediments, the high content, its origins and susceptibility to degradation (Woodward et al. 1992). Much of it is derived from freshwater inputs of terrestrial origin, such as that from sub-alpine moorland and other peaty soils, with additional inputs from autochthonous phytoplankton and from salmon fish farms, sewage treatment plants and stormwater drains. Organic matter from these diverse sources has, not unexpectedly, very different compositions. The terrestrial input of high-molecular-weight humic substances (from such as the peat soils) seems to be degraded very slowly, and thus its remineralisation probably does not contribute greatly to nutrient loads. In contrast, organic matter of marine origin is more likely to result from phytoplankton production, and therefore to be more readily decomposed to recycle nutrients.

Our strategy here was to type the organic matter through a range of discriminating techniques — for example, organic biomarkers (fatty acids, sterols, etc. — O’Leary et al. 1999, Volkman et al. 1997), and stable isotope ratios of carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$).

Fallowing study

In our original project proposal, we suggested that we would do a pilot study of the breakdown of waste material deposited from a sea cage onto the sediments of a finfish-farm lease. The experiment would begin once the overlying cage had been removed. This fallowing study was not envisaged as providing another perspective on temporal or spatial scales; rather it would contribute to defining the time sediments would take to return to a ‘stable’ condition (unlikely to be its original condition). This stable condition would serve as a fallowing reference, and guard against the sediments continuing to degrade over multiple cycles of restocking and fallowing.

The estuarine sediment survey was a yardstick for the fallowing study, because it would provide information on reference conditions for similar sediments distant from finfish-farm operations. It would also be likely to provide some insight into the background rate at which organic matter breaks down.

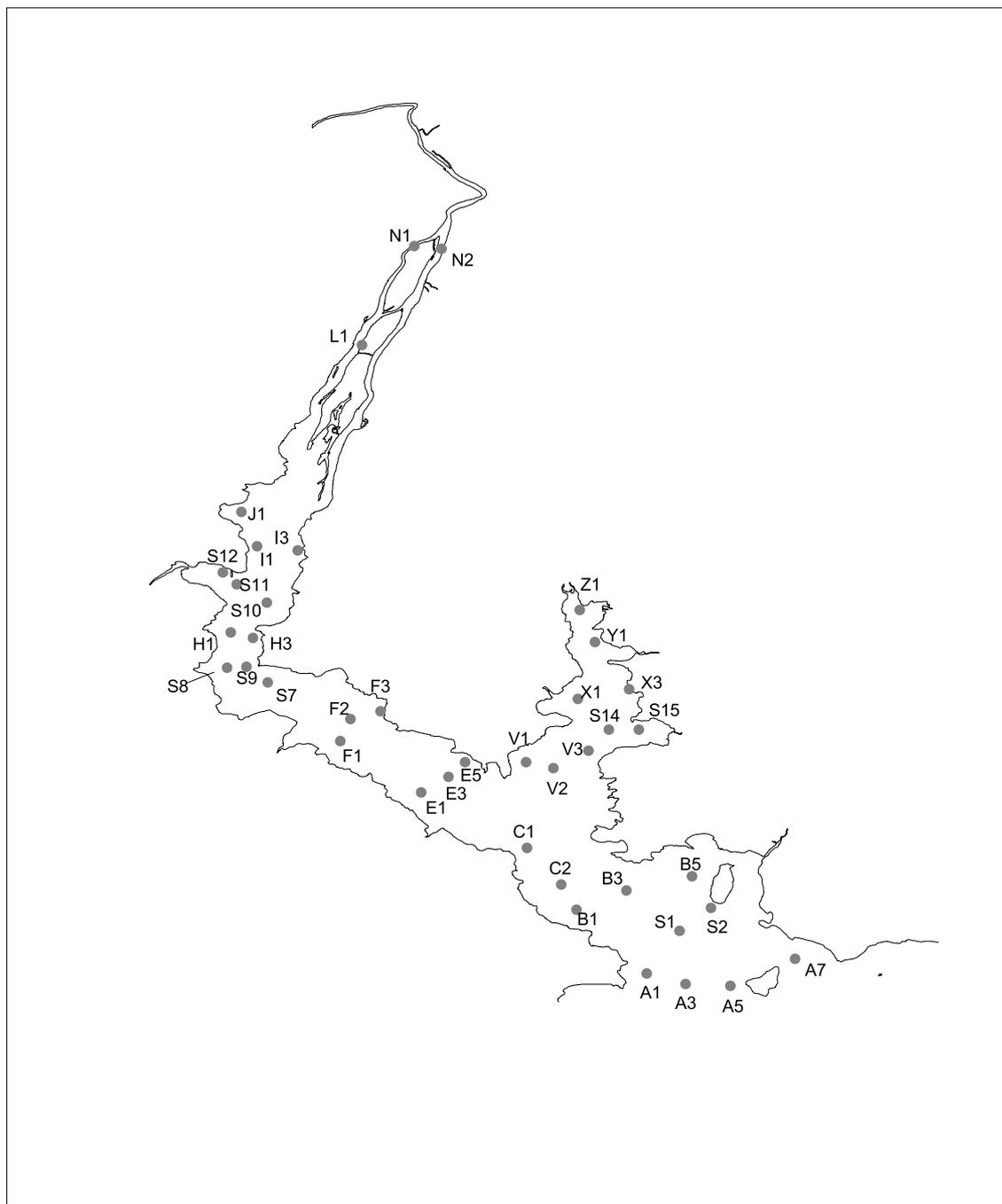


Fig. 2.8 Sampling sites occupied during the sediment survey of the Huon Estuary (May 1997).

2.4.6 Summary — covering relevant time and space scales

It was not practicable to examine all temporal and spatial scales relevant to this study in one survey procedure; it was necessary to integrate several different observational approaches. Some fieldwork was also separated for operational reasons. It is usual for surveys of water column and sediments to be done separately, and whereas the continuous profiling was done by an autonomous instrument, the weekly monitoring remained a manual task.

We devised the preceding program of survey and monitoring methods for HES to acquire the relevant observational data to meet the study objectives. A diagrammatic representation of the coverage of time and space scales is seen in Figure 2.9.

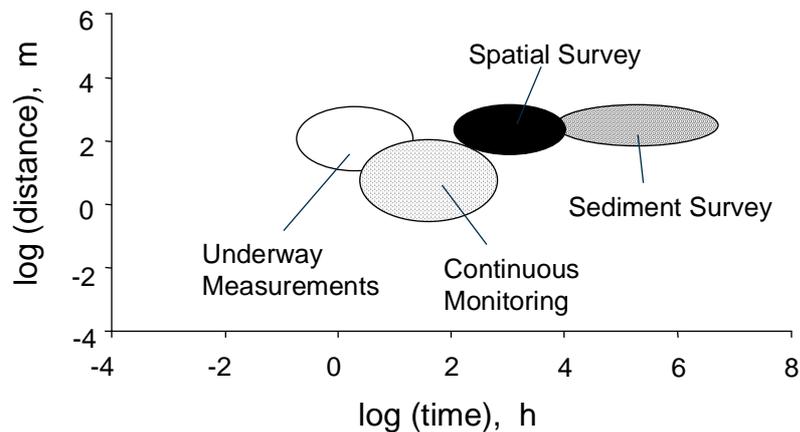


Fig. 2.9 A representation of the different time and space scales covered by the different observational procedures used in the Huon Estuary Study. See text for explanation of each.

The calendar of estuarine surveys is recorded in Table 2.2. Since we were not interested in looking into substrate assimilation at the molecular level of cell membranes, nor investigating the changes in estuarine characteristics since European colonisation, the spectrum of temporal and spatial observations has gaps at the low and high ends.

Table 2.2 Chronicle of estuarine surveys completed during the Huon Estuary Study, using survey vessel *Explorer*.

HES — TIMETABLE OF ESTUARINE SURVEYS	
Pilot Survey (HES 1)	26–29 March 1996
HES 2	2–5 July 1996
HES 3	8–11 October 1996
HES 4	24–28 February 1997
Sediments Survey	13–14 May 1997
HES 5	16–20 June 1997
HES 6	6–10 October 1997
HES 7	2–4 December 1997
HES 8	17–19 February 1998
HES 9	18–21 May 1998
HES 10	24–27 August 1998
Contaminants Survey (HES 10A)	31 August–3 September 1998

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3 PHYSICS OF THE ESTUARY

3.1 General Review

3.1.1 Introduction

The physics of an estuary determine its essential character — the framework and time-scale in which the system (natural *and* human-modified processes) operates. The layering or stratification of the water column provide niches for different life forms to prosper. Whereas, the flushing time of the estuary is a measure of the time that both biotic and abiotic activity are given to leave an imprint on the waters before they discharge out to sea. The underlying circulation pattern and transient currents together are the vehicle for dispersing these influences and their products, and also for replenishing the waters to begin the cycle afresh.

This chapter provides an overview of the physics of the Huon Estuary and describes the development of an inverse model, which may be used to provide advective and diffusive fluxes to transport and ecological models. Much of the physical data for the Huon has been processed within the framework of this inverse model, which involves the following discretisation:

- 1 The smallest spatial entity of the model is termed a *cell*.
- 2 Cells are arranged horizontally in *columns* and vertically in *layers*.
- 3 Each column and each layer is defined by a corresponding *index*.
- 4 A specific column is defined by a *column index*, the lowest index value being zero. The indices increase *down-estuary*.
- 5 Interfaces between columns are indexed according to the index of the *adjacent down-estuary cell*.

The arrangement of model columns used in the Huon model is shown in Figure 3.1. Since the arm adjacent to Port Cygnet (northeast of Column 11) receives little freshwater inflow and is not part of the main estuarine channel, it was omitted from the model.

3.1.2 Classification of the Huon Estuary

Hansen and Rattray's (1966) scheme for estuary classification (discussed, for example, by Dyer 1997, pp. 25-32) is probably the most widely used. Their scheme relies on two dimensionless parameters, which are assumed to be based on *tidally averaged* observations (or an approximation to them):

- the ratio of the velocities of the surface and depth-averaged currents, and
- the ratio of the *salinity contrast* to the depth-averaged salinity, where the salinity contrast is defined as the difference between the surface and bottom salinity.

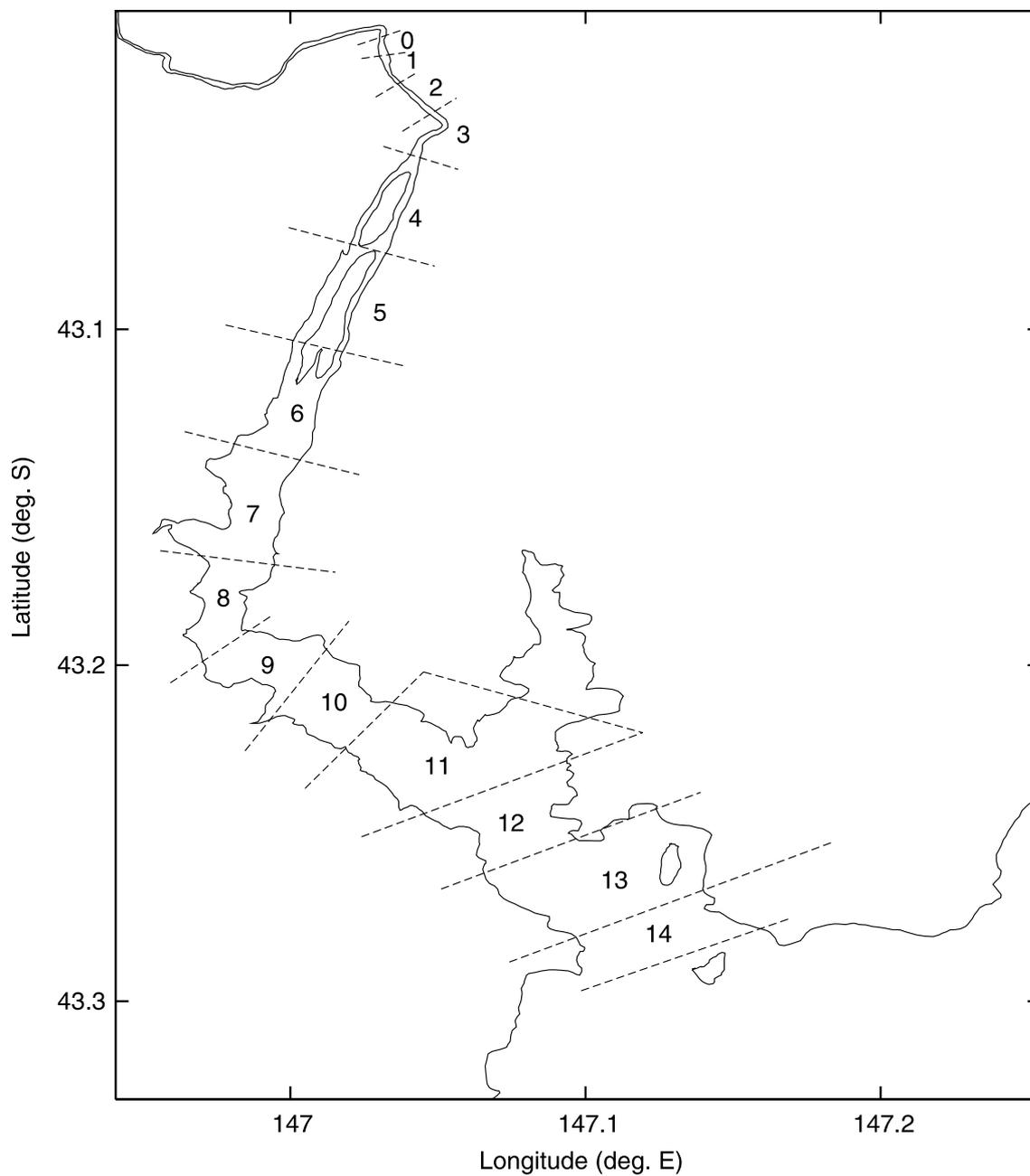


Fig. 3.1 Subdivision of Huon Estuary to form model columns.

An important output from the Hansen and Rattray classification is the value of the diffusive fraction, which describes the relative importance of ‘external’ mixing processes (e.g. tidal and wind mixing) and those due to the buoyancy-driven circulation of the estuary. The diffusive fraction is formally defined as the ratio between the *upstream* flux of salt attributable to external mixing processes and the *downstream* flux of salt attributable to the river flow. A strongly stratified estuary driven predominantly by buoyancy forcing has a diffusive fraction close to zero, while an embayment driven mainly by tides and winds has a diffusive fraction close to unity.

The value of the diffusive fraction, estimated by the Hansen and Rattray classification provides a useful indication of the type of physical model suitable for a given estuary. For low values of diffusive fraction, stratification is important and a layered model is required. As the diffusive fraction approaches unity, a vertically well-mixed model (i.e. one with only one layer) or a single box model becomes more appropriate.

The inverse model described in Section 3.4 was used to provide the classification parameters described above, for each model column. The results are shown in Figure 3.2, which is a copy of the classification diagram provided by Hansen and Rattray (1966) with values for the Huon Estuary overlaid. The ‘river’ end of the estuary is towards the top left-hand part of the figure, while the ‘ocean’ end is towards the bottom right-hand part. The small squares indicate values at the vertical interfaces between columns, omitting the interface at the extreme ‘river’ end where the salinity is close to zero most of the time. Each small square represents an average of surveys HES 2 (Jul ’96) to HES 10 (Aug ’98). Also shown on the figure are the defined estuary types (1 to 4, with ‘a’ and ‘b’ subdivisions) and contours of the diffusive fraction. This classification indicates that:

- 1 The ‘river’ end is close to Type 4, which is described as a ‘salt wedge’.
- 2 The remainder of the estuary falls close to the line separating Types 2a/2b and 3a/3b. Estuaries of Type 3a/3b are generally defined as ‘fjord-like’.
- 3 Most of the estuary falls to the right of the line separating Types 1a/1b and 2a/2b, which indicates that the current velocity reverses at depth.
- 4 The diffusive fraction over most of the lower estuary is close to zero indicating the dominance of the buoyancy-driven flow in transporting material horizontally.

3.1.3 Dimensions of the Huon Estuary

The approximate dimensions of the Huon Estuary, below Ranelagh (near cell 0 of Figure 3.1) are shown in the following table:

PROPERTY	VALUE
Length	39.1 km
Width at mouth	4.5 km
Surface area	77.4 km ²
Volume	1.38 km ³

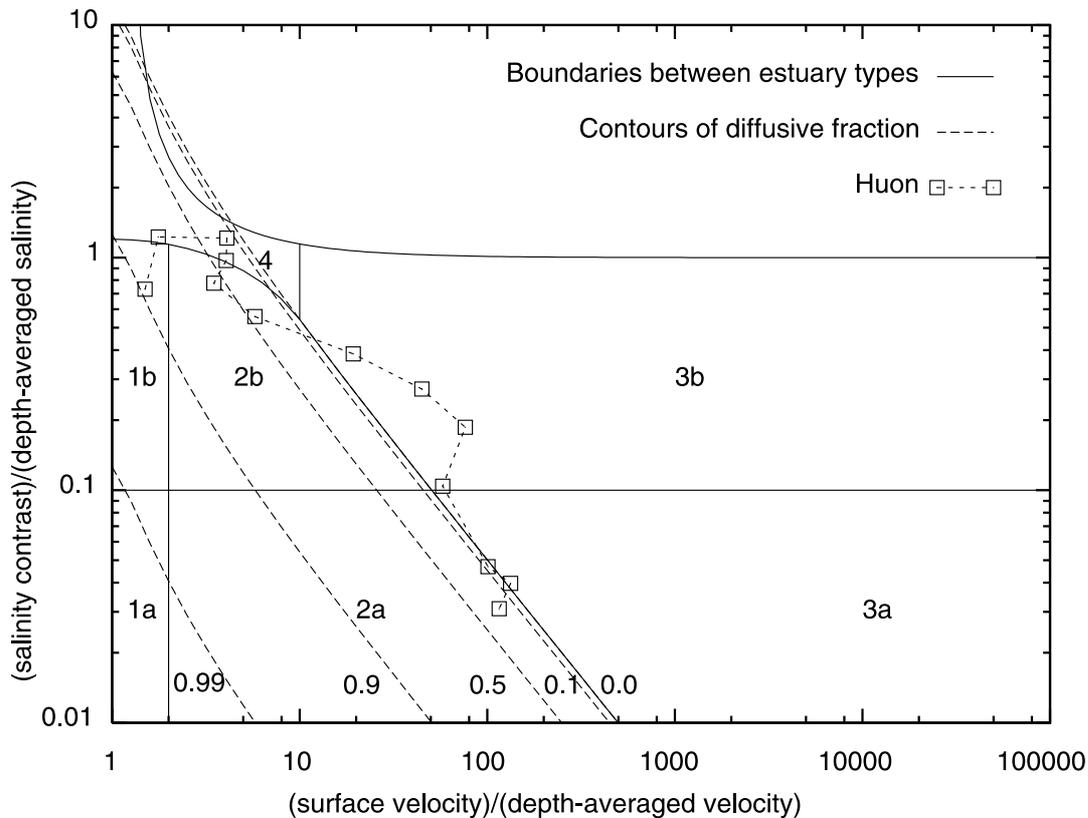


Fig. 3.2 Hansen-Rattray Diagram showing classification of the Huon Estuary.

Since the neighbouring Derwent Estuary has been much researched, it is useful to compare the physical properties of the Derwent and the Huon. Edgar et al. (1999), in their classification of Tasmanian estuaries, noted very close similarities for the Huon and Derwent estuaries in cluster analysis of their “physical variables” (catchment area, total annual run-off, estuarine perimeter length, etc.). Figure 3.3 shows the way in which the cross-sectional areas of the two estuaries vary along their length. The two curves have been aligned longitudinally to provide the best visual fit, and the positions of significant sites along each estuary have been indicated along the top of the Figure. Therefore, on a geometric basis, Huon Bridge is comparable with Bridgewater, and Port Huon is comparable with Bowen Bridge. The agreement between the curves is good and would be improved if Ralphs Bay were omitted from the Derwent data (Ralphs Bay is arguably not a part of the Derwent Estuary proper). However, there is an important difference between the estuaries in the upper reaches (above Port Huon/Bowen Bridge), where the cross-sectional area of the Derwent is significantly larger than that of the Huon. This difference has important implications regarding the penetration of the salt wedge into the estuary (Subsection 3.3.3). In addition, the Huon Estuary is about 10 km shorter than the Derwent Estuary.

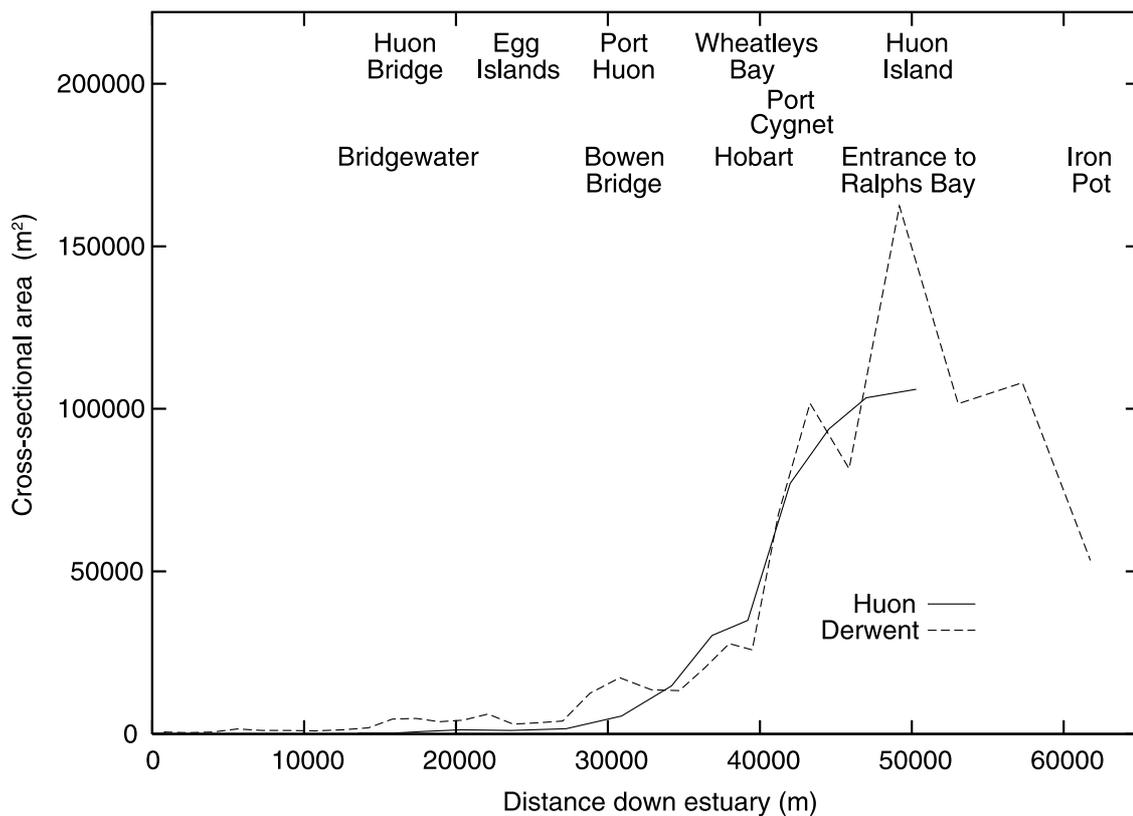


Fig. 3.3 Comparison of cross-sectional areas of the Huon and Derwent Estuaries.

3.1.4 Tidal motions

The mean tidal range at Huonville Bridge is 0.9 m (Gallagher 1996); that of Hobart is 0.8 m (Department of Defence 1998). The horizontal tidal excursion at any site in the estuary may be estimated by multiplying the *vertical* tidal excursion by the ratio of the total upstream surface area to the cross-sectional area at that point. This calculation is based on the (reasonable) assumption that the water level in the estuary at any one time is approximately horizontal. Figure 3.4 shows the variation of tidal excursion along the estuary, derived from this relationship and the tidal amplitude of the main semi-diurnal (M_2) tidal constituent (0.251 m; Department of Defence 1998). The figure shows that, at the model interface with index 6 (the southern end of the Egg Islands), a water particle moves between a point 570 m upstream and a point 570 m downstream of a given site during a tidal cycle. In all other regions in the estuary the tidal excursion is significantly less than this, and is typically ± 150 m. Maximum tidal velocities (in m s^{-1}) may be obtained by multiplying the excursion values by a factor of 0.00014 (the angular frequency of the M_2 tidal constituent), yielding typical values of 0.02 m s^{-1} . It should be noted that this estimate relates to the M_2 tidal constituent only. Inclusion of the other tidal constituents would increase the excursion values by a factor of ~ 1.6 .

These are still very small tidal excursions. They reflect the low tidal amplitude, and the lack of any large basin in the upper estuary. It is not surprising that flushing of the estuary is dominated by the two-layer overturning circulation.

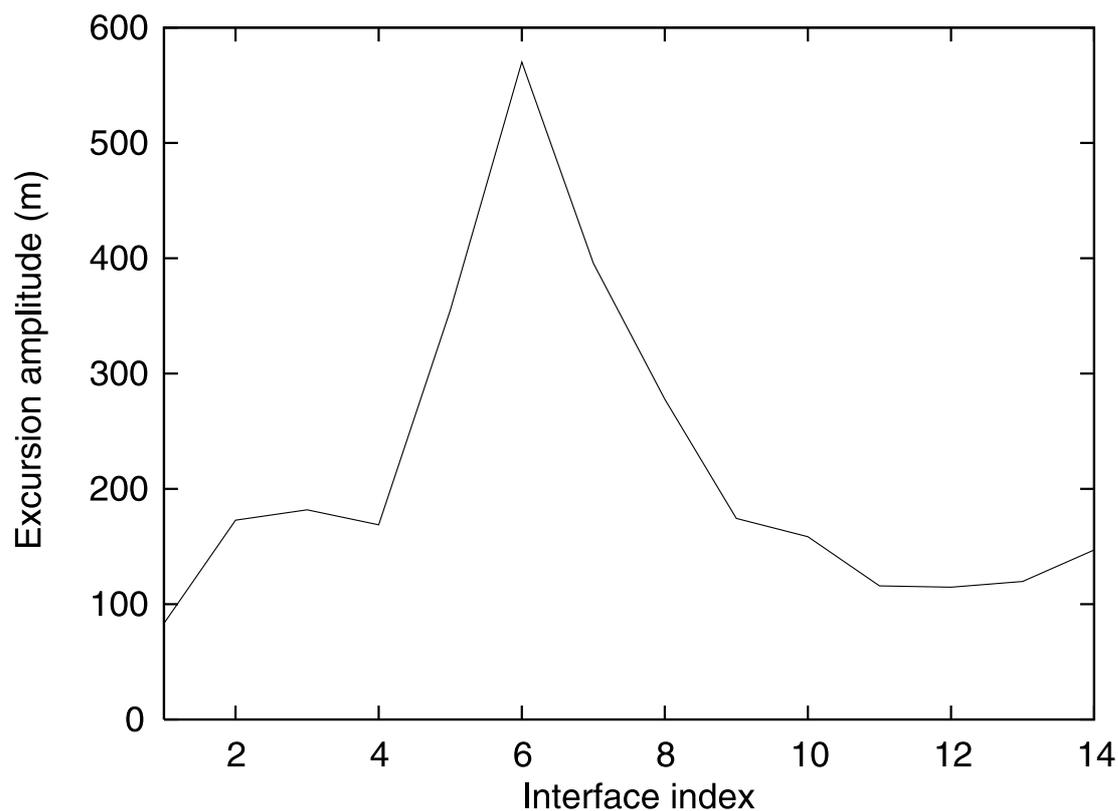


Fig. 3.4 Approximate amplitude of horizontal tidal excursion in Huon Estuary.

3.1.5 Winds

Winds were not measured as part of this study. However, indicative wind velocities for the estuary were obtained from observations made at Bruny Island and provided by the Bureau of Meteorology, Hobart. Composite data from neighbouring stations 94010 and 94198 were used. Figure 3.5 shows a scatter diagram of wind velocities (to) covering the 10-year period 1989 to 1998. The figure shows a spread of winds from all directions, but with a predominance of winds from the WSW. The 50-percentile wind speed is 5.6 m s^{-1} .

3.1.6 River flow

Daily river flow data from Frying Pan Creek were obtained for the years 1996 to 1998, inclusive. After data gaps were filled by interpolation, the flow values were multiplied by a factor of 1.5 to account for the extra catchment area below Frying Pan Creek and above Port Huon. The resultant data therefore represents the river flow approximately half way along the Huon Estuary. Figure 3.6 shows flow “exceedance” plots for the Huon

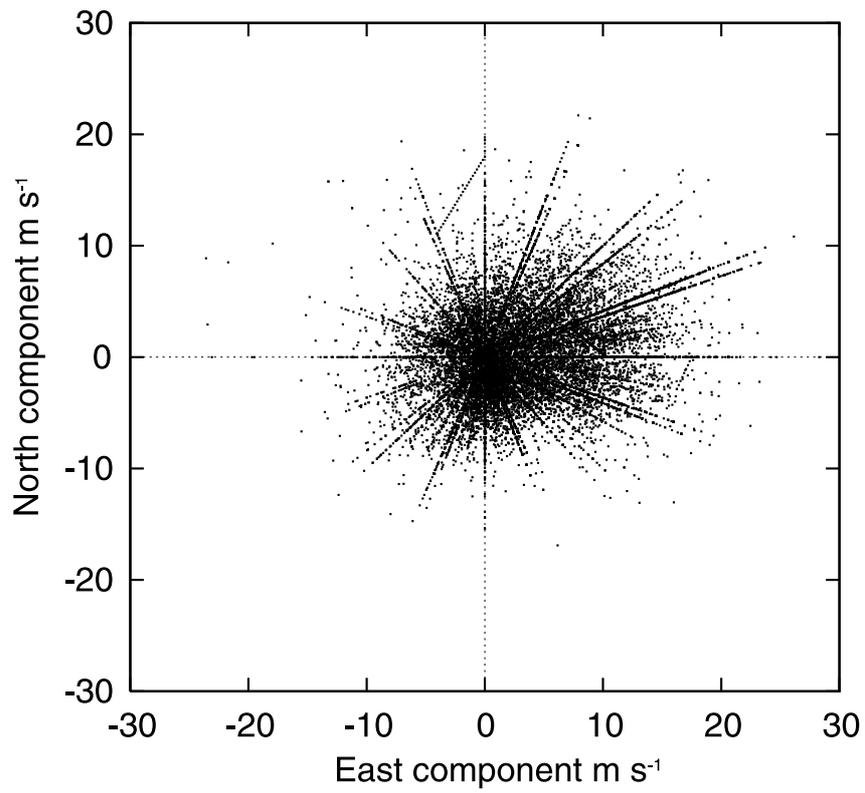


Fig. 3.5 Scatter diagrams of winds (to) for Bruny Island, 1989-1998.

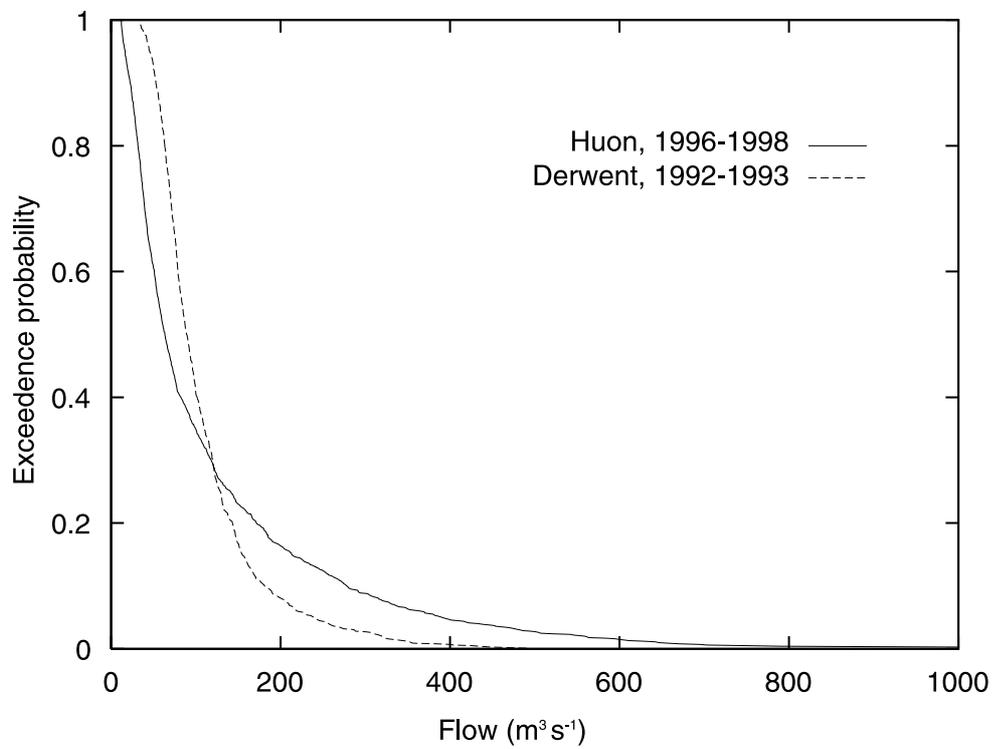


Fig. 3.6 Flow exceedance plots for the Huon and Derwent Estuaries.

and Derwent Estuaries (the latter for 1992 and 1993). While the average flows for the two estuaries are similar ($117 \text{ m}^3 \text{ s}^{-1}$ for the Huon and $108 \text{ m}^3 \text{ s}^{-1}$ for the Derwent), it is clear that the flow of the Derwent is more closely regulated than that of the Huon. During the periods of the Huon surveys HES 2 to HES 10 (Jul '96–Aug '98), the mean river flow was $90.3 \text{ m}^3 \text{ s}^{-1}$, with a standard deviation of $50.2 \text{ m}^3 \text{ s}^{-1}$.

Woodward et al. (1992) described four spatial surveys of salinity in the Huon Estuary made during 1989, when river flows ranged from 8.5 to $264 \text{ m}^3 \text{ s}^{-1}$. All surveys showed a pronounced tendency for the fresher water to flow down the eastern coast of the upper estuary and the northeastern coast of the lower estuary. This effect could be caused by a combination of:

- the effect of the rotation of the Earth (e.g. Dyer 1997, pp. 15, 19), and
- the predominance of the wind from the WSW (see above).

3.1.7 Currents

Current-meter data was provided by Aquatas Pty Ltd, Huon Aquaculture Company Pty Ltd, Tassal Ltd and the Tasmanian Department of Primary Industries, Water and Environment. The observed currents were weak (generally less than 0.2 m s^{-1}) and often near the limit of sensitivity of the instruments. In particular, the current meters were unable to satisfactorily resolve the tidal currents, which were shown in Subsection 3.1.4 to be typically 0.02 m s^{-1} . In addition, the deployment depth of the instruments was not always known. Therefore, only broad conclusions may be drawn from these observations.

Figure 3.7 shows a scatter diagram of current velocities at eleven sites. It is evident that the currents are highly variable, although overall they circulate clockwise (towards the southeast along the northeastern shore, and towards the northwest along the southwestern shore). This circulation is consistent with the tendency for fresher water to flow down the northeastern coast (Subsection 3.1.8; Subsection 3.3.1, Fig. 3.10; Subsection 3.3.2, Fig. 3.11).

3.1.8 Other studies of the Huon Estuary

Woodward et al. (1992) also attempted to relate the depth of the fresher surface layer to the surface salinity and to the river flow. They also reported some current-meter observations, but did not provide detailed results.

Burgess et al. (1993) described the results of a survey of the estuary conducted in March 1993. The survey included salinity measurements at transects on either side of the Egg Islands (at Franklin and Cradoc) and at a site near Police Point. The observations from Franklin and Cradoc indicated lower salinities on the eastern end of each transect, and (paradoxically) *lower* overall salinities in the Franklin (western) transect. The latter effect is probably caused by a difference in hydraulic resistance between the channels on either side of the Egg Islands.

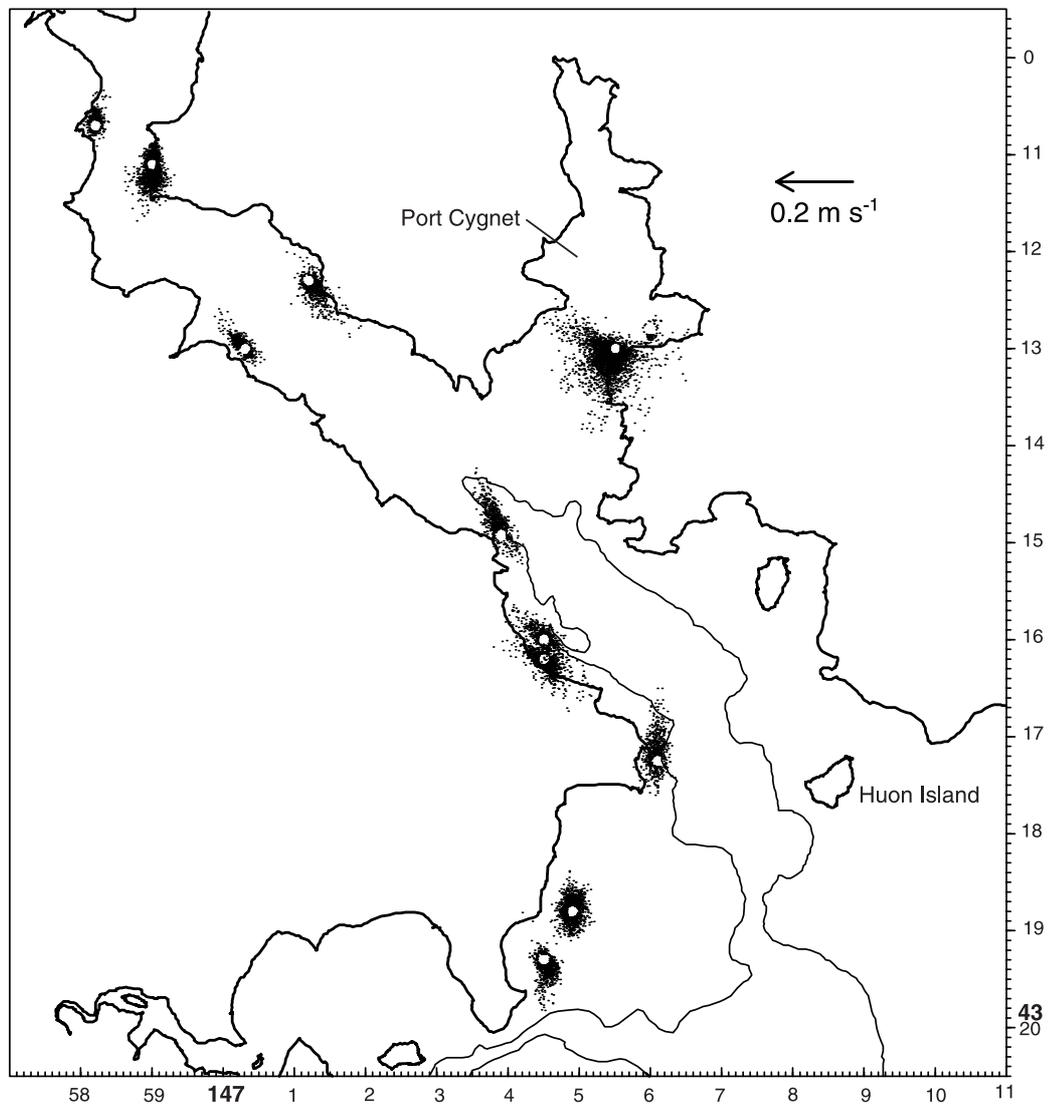


Fig. 3.7 Scatter diagram of current velocities measured by current meters. The thin line is the 30-m isobath; the arrow (top right) indicates the scale of each scatter plot. The axes are latitude and longitude in degrees and minutes.

Gallagher (1996) presented a general review of the Huon Estuary and catchment, but provided no new data of relevance to the physics of the estuary. Bobbi (1998) described the water quality of the Huon Catchment, but gave no information on the physics of the estuary.

Wallace (1998) used data from the present study to investigate statistical relations between the salinity variation along the estuary and their dependence on river flow (averaged over three days). Unfortunately, the flow rate and any resultant relation with salinity depend strongly on the averaging time (as Wallace himself noted, and as we note in Subsection 3.2.1); derivation of this relation must therefore probably await a better understanding of the appropriate averaging to be applied to the river flow.

3.2 Background to Physical Surveys

3.2.1 River flows

Daily river flows for the study period were derived as described in Subsection 3.1.6, and filtered using a one-sided exponential filter with a prescribed time constant. The filtered flow values for the central day of each survey are shown in Table 3.1 for a range of time constants. The time constant of zero represents unfiltered daily data. The table shows that the filtered river flow depends quite strongly on the choice of time constant.

Table 3.1 Daily flow rates of the Huon River at approximately Port Huon for the central day of each estuarine spatial survey (see Subsection 3.1.6). Different filters have been applied; a filter with a time constant of zero equates to the unfiltered daily flow.

SURVEY	TIME CONSTANT OF FILTER (DAYS)				
	0	1	3.5	6.7	14
HES 2 (Jul '96)	107.6	124.7	95.7	80.5	94.8
HES 3 (Oct '96)	249.9	195.5	170.3	188.5	212.7
HES 4 (Feb '97)	30.2	26.5	25.9	26.8	33.8
HES 5 (Jun '97)	64.0	75.9	89.2	89.6	85.8
HES 6 (Oct '97)	28.6	35.3	51.8	54.5	66.6
HES 7 (Dec '97)	122.1	175.7	164.6	140.4	129.3
HES 8 (Feb '98)	55.6	58.4	52.0	46.3	39.3
HES 9 (May '98)	81.6	62.9	58.7	74.8	89.7
HES 10 (Aug '98)	59.1	81.8	112.3	111.2	111.6

At low flows, the exponential flushing time of the whole estuary was estimated to be about 1 week (see Subsection 3.4.7). It would seem reasonable to use a time constant of this magnitude for the exponential filter to be applied to the river flow. However, subsequent trials with the inverse model indicated that the model exchange fluxes correlated best with *unfiltered* daily flow data (i.e. a time constant of zero). This may arise from the much shorter flushing time of the upper layer (Subsection 3.4.7). Runs of the transport model (Subsection 3.4.6) and the ecological model (Chapter 10) were therefore based on *unfiltered* daily river flow data.

3.2.2 Winds

Approximately three-hourly wind records were obtained from stations 94010 and 94198 on Bruny Island. Station 94010 covered the full study period, but was based on manual logging. Station 94198 is an automatic recorder installed near 94010 on 16 May 1997. A composite record was therefore produced using data from 94010 prior to 16 May 1997 (including, therefore, surveys HES 2, HES 3 and HES 4) and data from 94198 thereafter. Linear interpolation was used to fill gaps and to generate wind values every three hours relative to 00:00 Eastern Standard Time. Figure 3.8 shows ‘stick diagrams’ of the wind for the periods of surveys HES 1 to HES 10 (Mar ’96–Aug ’98). The vectors on each plot start one day before the first day of each survey.

3.3 CTD Observations

3.3.1 Vertical sections

Observations of salinity, temperature and dissolved oxygen, made with a vertical CTD (conductivity-temperature-depth) profiler during surveys HES 2 to HES 10 (Jul ’96–Aug ’98), were used to construct vertical sections along and across the estuary, as shown in the following table.

Table 3.2 Abbreviated names of different CTD sections of the Huon Estuary.

SECTION NAME	DESCRIPTION
huon_5m	Longitudinal section along whole of estuary, truncated below 5 m depth
huon_30m	Longitudinal section along whole of estuary, truncated below 30 m depth
east	Longitudinal section along channel to east of the Egg Islands
west	Longitudinal section along channel to west of the Egg Islands
cyg	Longitudinal section into Port Cygnet
a	Transverse section through stations A1 to A9
b	Transverse section through stations B1 to B5
c	Transverse section through stations C1 to C4
e	Transverse section through stations E1 to E5
f	Transverse section through stations F1 to F3
h	Transverse section through stations H1 to H3
v	Transverse section through stations V1 to V3
x	Transverse section through stations X1 to X3

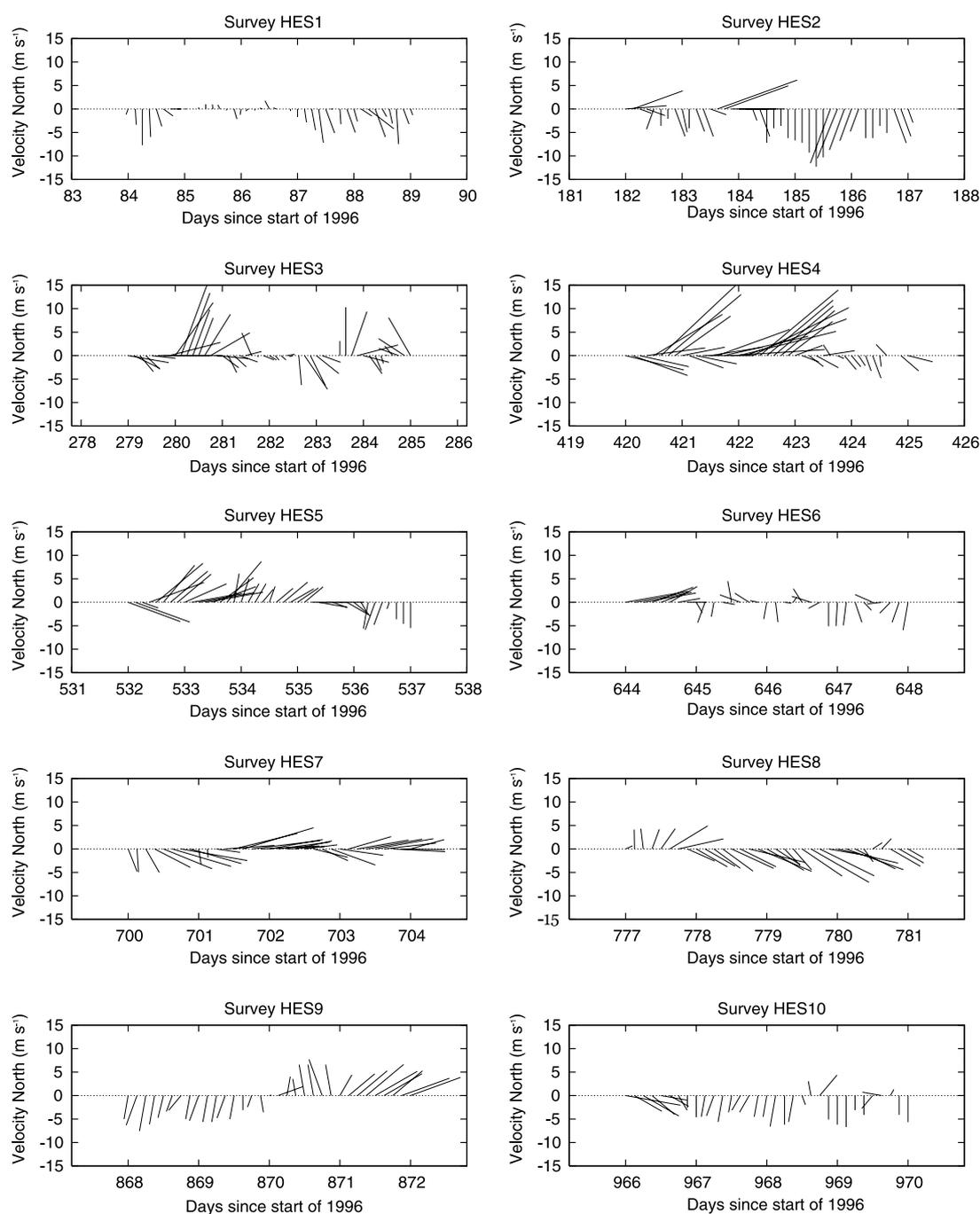


Fig. 3.8 Stick diagrams showing winds at Bruny Island during survey periods. The vectors start one day prior to the first day of each survey. The wind direction is from the point where the vector touches the horizontal axis to the far end of the vector.

The longitudinal variation of salinity for each survey is shown in Figure 3.9 (section 'huon_5m' in table on previous page) and the variation of salinity across the estuary mouth is shown in Figure 3.10 (section 'a' in table on previous page).

Figure 3.10 clearly shows the tendency of the fresher water to flow down the northeastern coast of the estuary.

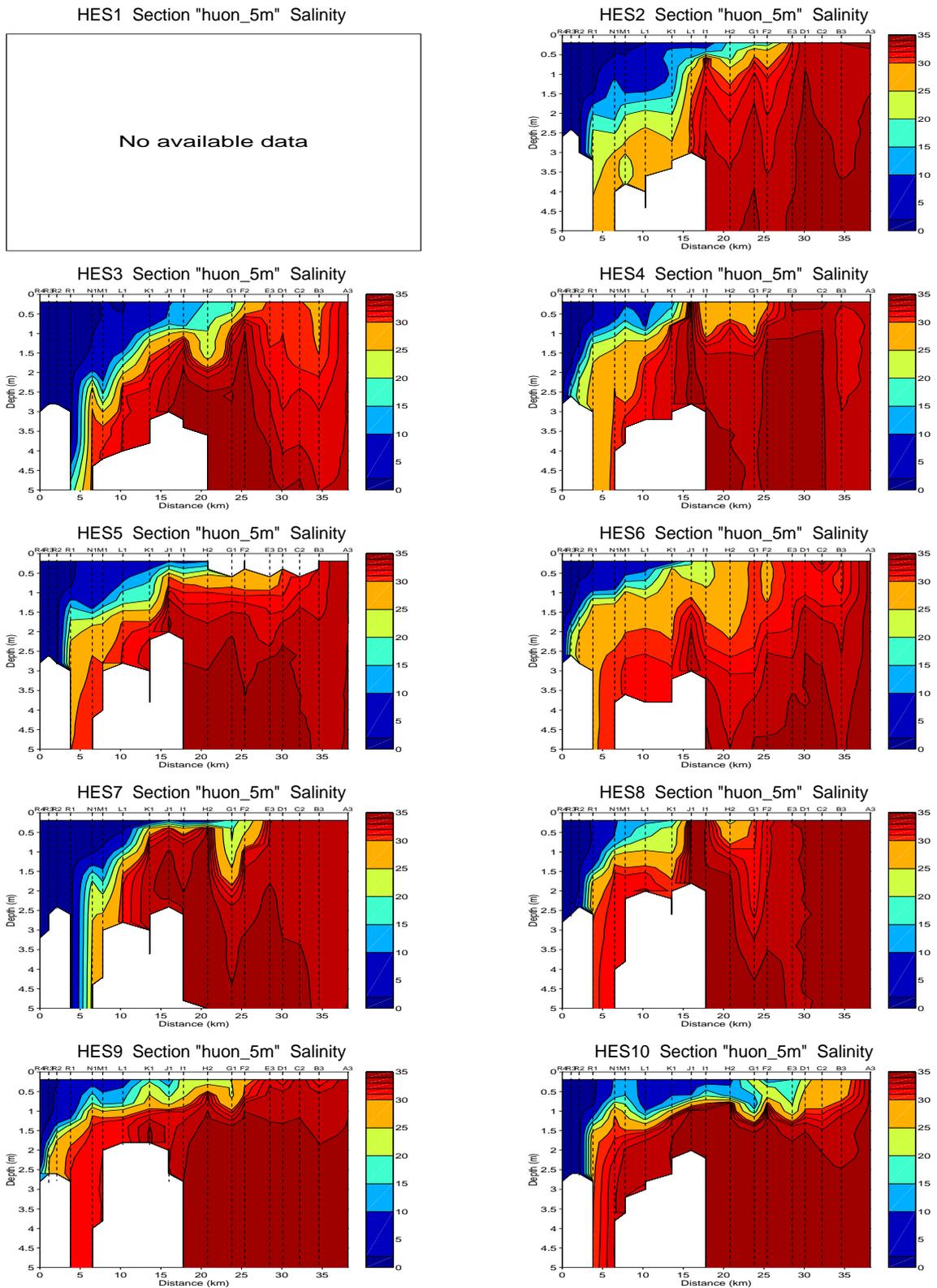


Fig. 3.9 Vertical section along the estuary showing salinity for each spatial survey. Distance shown is distance downriver from station R4.

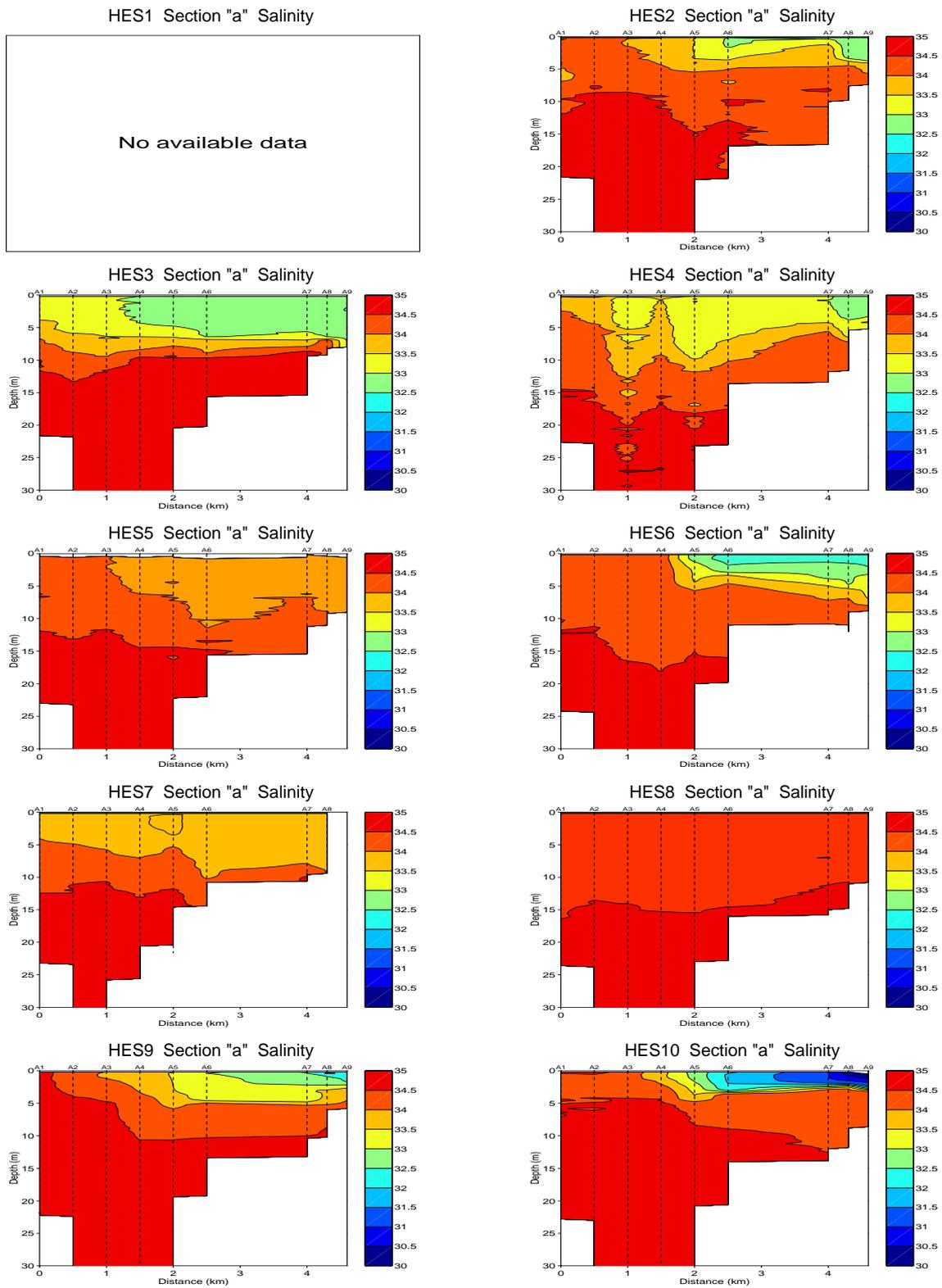


Fig. 3.10 Vertical section across the estuary at the mouth (Transect A), showing salinity for each spatial survey.

3.3.2 Surface salinity measured underway

A number of surveys of near-surface salinity, temperature and fluorescence were made using an underway sampling system from February 1997. As an example, Figure 3.11 shows the observed salinity at 0.5 m and 1.0 m depth on 18 May 1998 (Survey PS 4, just prior to HES 9) and 24 August 1998 (Survey PS 5, just prior to HES 10). The following features are evident:

- 1 Fresher water tends to flow down the eastern and northeastern coasts of the estuary; the effect is more pronounced on 18 May 1998.
- 2 There is considerable spatial patchiness, presumably caused by the action of the wind on the shallow low-salinity layer.
- 3 On 24 August 1998, there was a thin lens of fresh water (less than 1 m thick) in the Port Cygnet arm. The salinity contrast over the top one metre of the water column is around 25. Since there is little fresh water flow into Port Cygnet, this feature must be transient and is probably related to wind action.
- 4 There is little evidence that the fresher water flows preferentially to either the east or west of the Egg Islands.

3.3.3 Comparison with the Derwent Estuary

Figures 12 and 13 show the mean longitudinal variation of bottom and surface salinity along the Huon and Derwent estuaries, using the alignment previously derived from matching their cross-sectional areas (Fig. 3.3). The salinity data from the Derwent was derived from ten surveys by the CSIRO Coastal Zone Program between 1992 and 1994 (CSIRO — *unpublished data*). The error bars indicate the variation (plus and minus one standard deviation) over all surveys around the mean value. The salinity values were derived from data used to drive two-layer inverse models of the type described in Section 3.4, and so represent averages over upper and lower layers of the water column. A striking feature of Figures 3.12 and 3.13 is the greater penetration of salt (especially in the lower layer) into the Derwent Estuary than into the Huon. This is presumably caused by the large cross-sectional area of the Derwent above Bridgewater, compared with that of the Huon above Huon Bridge (Subsection 3.1.3 and Fig. 3.3). The relatively small penetration of salt above Huon Bridge is probably the main reason why the salinity structure in the Huon shows only a weak relationship with river flow (Subsection 3.4.5).

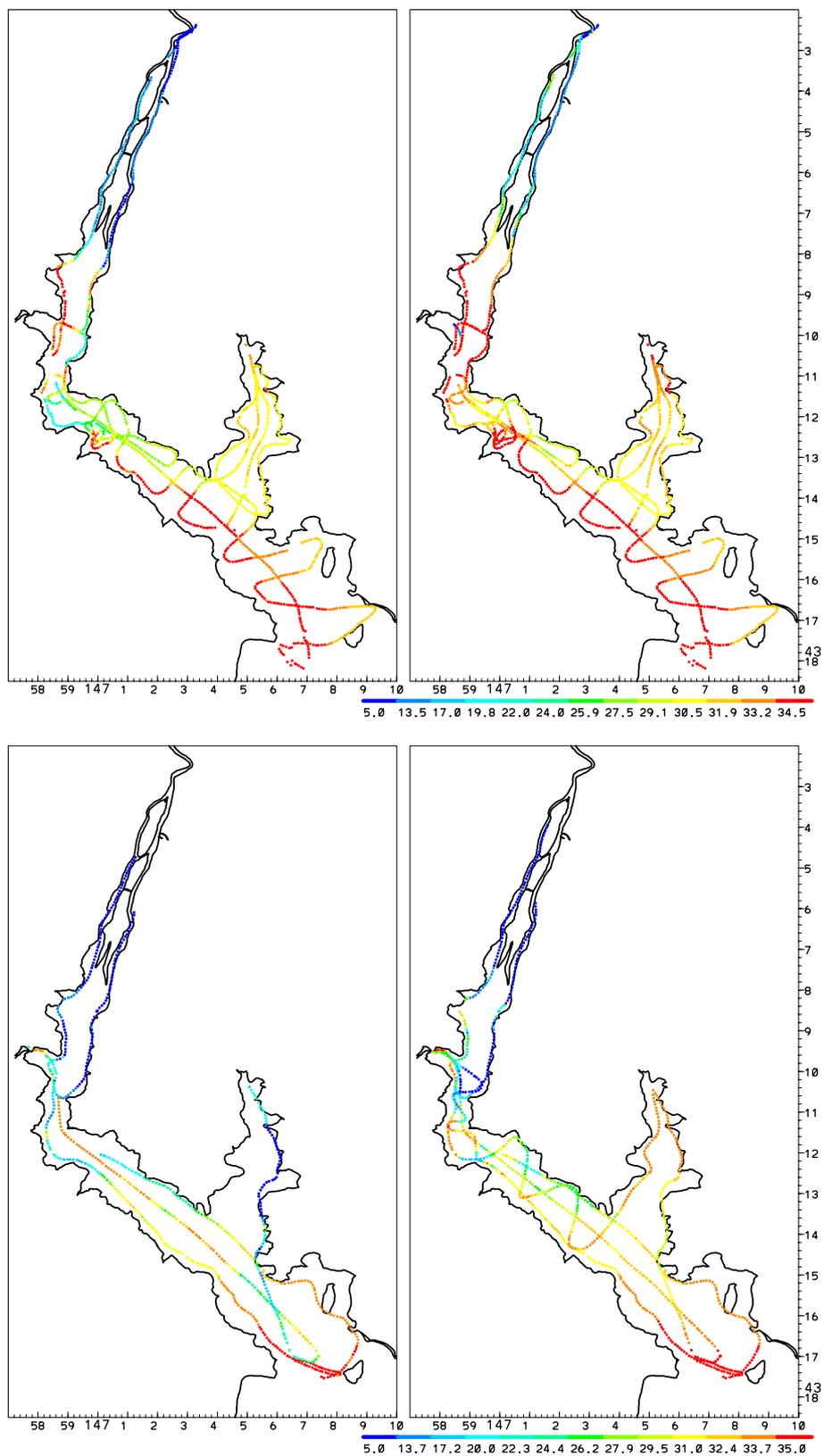


Fig. 3.11 Surface salinity observed underway. Top panels are for 18 May 1998; bottom panels are for 24 August 1998. Left hand panels are at 0.5 m depth; right hand panels are at 1.0 m depth. The axes are latitude and longitude in degrees and minutes, and the colour bar and scale represents salinity.

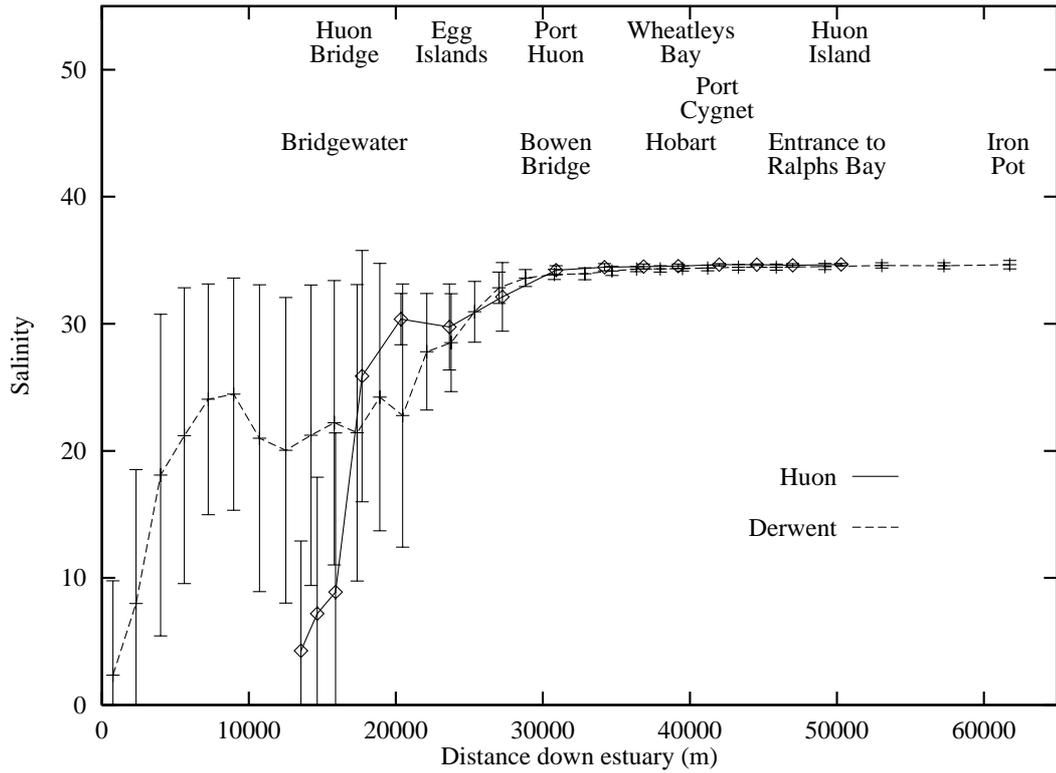


Fig. 3.12 Comparison of Huon and Derwent Estuaries: bottom salinity. See text for meaning of error bars.

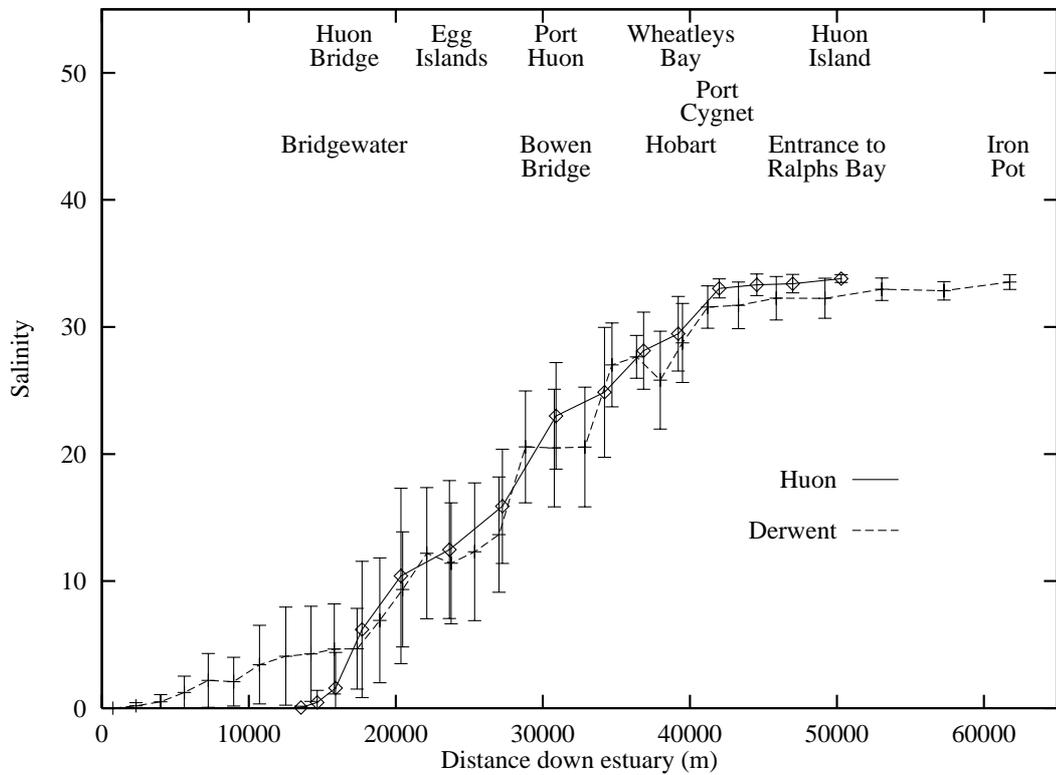


Fig. 3.13 Comparison of Huon and Derwent Estuaries: surface salinity. See text for meaning of error bars.

3.4 Inverse Modelling

3.4.1 Schematisation

The inverse model was based on a subdivision of the estuary into ‘columns’, ‘layers’ and ‘cells’ as described in Subsection 3.1.1, and used a combination of river flow and salinity data derived from surveys HES 2 to HES 10. The output is a set of exchange fluxes between model cells for each of a number of prescribed river flows. These exchange fluxes may then be used to drive generic estuarine transport or eutrophication models such as the *SEEM* model developed by CSIRO (Parslow et al. 1999b). The general schematisation of two-layer inverse and transport models is shown in Figure 3.14, where property concentrations are denoted by C , fluxes by q , column indices by i and layer identifiers by l or u . The model has been described more fully by Parslow et al. (1999a).

The estuary was subdivided horizontally into model columns approximately centred on survey transects. Since Port Cygnet is not part of the main estuary flow channel, it was excluded from the model. The subdivision is shown in Figure 3.1. Since the survey boundaries (i.e. the cross-sections north of column 0 and south of column 14) coincide approximately with survey stations, the plan areas of columns 0 and 14 (as shown in Figure 3.1) were doubled before being used to define the geometry of the model.

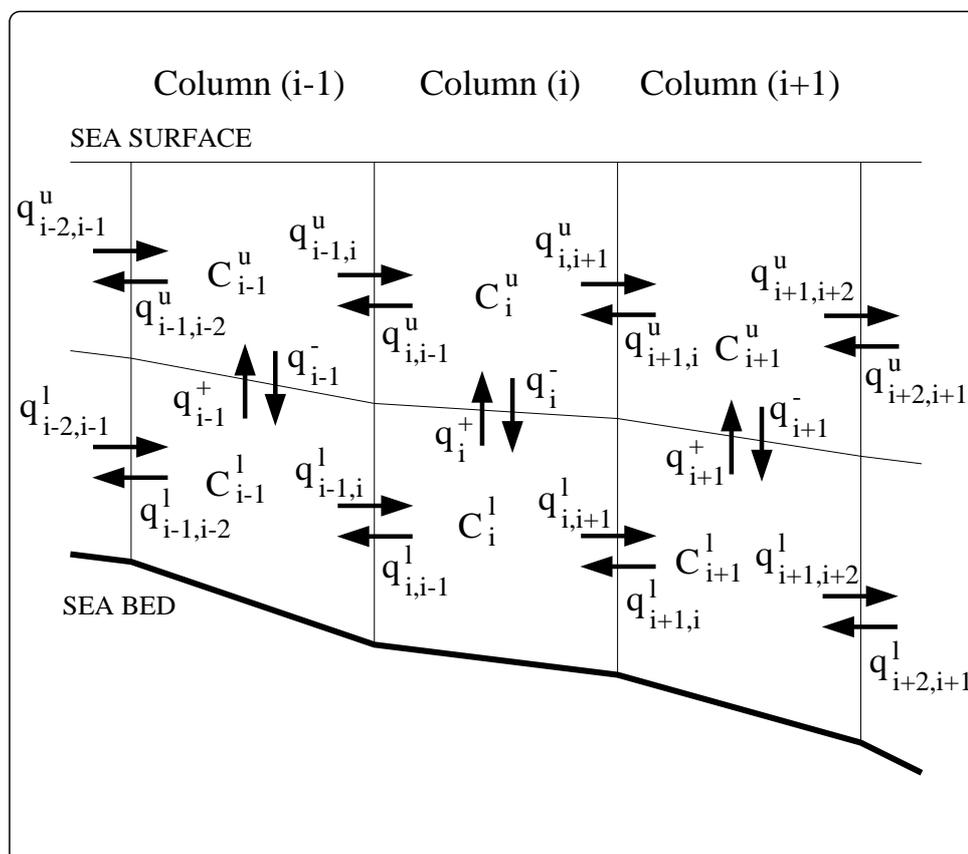


Fig. 3.14 Schematisation of two-layer inverse and transport models. See text for meaning of symbols.

3.4.2 Processing of salinity data

The following procedure was used to process the salinity data:

- 1 The pressure was corrected to actual depth, using the density derived from salinity and temperature.
- 2 Since the top 0.5 m of the water column was not satisfactorily sampled by the CTD (owing to problems related to priming of the pump), observations from the surface salinity bottle were used in this region, where possible.
- 3 The salinity data for each profile were binned into 0.1 m depth bins.
- 4 All data representative of a single model column were averaged for each survey.
- 5 A two-layer profile was fitted to the resultant salinity distribution for each column and for each survey.
- 6 A single layer depth (generally representative of the range of 'observed' layer depths) was selected for each column. The prescribed interface depths for each column are shown in the following table.

Table 3.3 Prescribed depths of the interface between surface and bottom layers for water columns of the cells depicted in Figure 3.1.

COLUMN INDEX	INTERFACE DEPTH (m)
00	1.4
01	1.4
02	1.4
03	1.4
04	1.4
05	1.4
06	1.4
07	1.4
08	1.4
09	1.4
10	1.4
11	5.5
12	5.5
13	5.5
14	8.6

- 7 Mean salinity values were derived for the upper and lower layers of each column for each survey, using the layer depths selected in (6).
- 8 Regions in which salinity increased in the upstream direction were removed by scanning the salinity from the *seaward* end and setting any 'saltier' values equal to the current minimum salinity.

3.4.3 Processing of bathymetric data

Processing of the bathymetry data involved the following procedure:

- 1 About 5900 bathymetric data points were digitised from Hobart Marine Board and Royal Australian Navy charts dating from 1931 to 1965. Some extra bathymetry values for Garden Island Bay (where the data are very sparse) were inserted by extrapolation from further offshore.
- 2 Positions were corrected to WGS84 by fitting the coastline on the above charts to modern coastline data.
- 3 The bathymetric data were corrected to mean sea level, using tidal data for Hobart.
- 4 A horizontal polygon was generated for each model column by intersecting the estuary cross-sections joining the two adjacent columns with the modern coastline.
- 5 The surface area of each column was derived from the polygons generated in (4).
- 6 The area of the horizontal interface within each column was derived by multiplying the surface area by the proportion of bathymetry points deeper than the prescribed interface depth within that column.
- 7 The depth of the lower layer for each column was derived by subtracting the prescribed interface depth from the average depth of all bathymetry points deeper than the interface depth.
- 8 The shape factor for the bottom layer (the ratio between the cross-sectional area and the product of interface width and layer depth) was chosen as unity for all columns, for consistency with (7).

3.4.4 Processing of river flow data

The river flow data were processed as described in Subsections 3.1.6 and 3.2.1. Selection of the appropriate filter length is described in Subsection 3.4.5.

3.4.5 Runs of the inverse model

The inverse model is a linear model and therefore the resultant exchange fluxes are proportional to the prescribed river flow. The model was therefore run with the geometry and salinity distributions described above, and unit river flows. Final exchange fluxes were obtained by multiplying the exchange fluxes output from the model by the actual river flows.

The required filter length for the river flow was selected as follows:

- 1 Model exchange fluxes were generated for each survey using the unfiltered daily river flow, and the filtered daily river flow with filter e-folding times of 1, 3.5, 6.7 and 14 days.

- 2 The correlation coefficient between each exchange flux and each of the sets of river flow data (point 1 above) was calculated.
- 3 The square of the correlation coefficient, when averaged over all fluxes, showed a maximum for *unfiltered* daily river flow (rather than for any of the filtered river flows).

Unfiltered daily river-flow data were therefore selected for forcing the inverse and transport models.

The salinity distributions in each survey showed considerable variability that could not be simply attributed to changes in river flow. The exchange fluxes derived from the inverse model showed similar unexplained variability. Therefore, for the purposes of generating exchange fluxes for use in transport models, the following procedure was adopted:

- 1 The exchange fluxes were linearly regressed against river flow.
- 2 'Synthetic' exchange fluxes were generated from the regression relationship obtained in (1) and two 'extreme' river flows: 25 and 250 m³ s⁻¹. (Since the resulting fluxes are linearly related to the exchange fluxes derived from the inverse model, they also obey conservation of volume.)
- 3 The two resulting sets of 'synthetic' exchange fluxes were adjusted to be consistent with an upstream differencing option for future use in transport models (Parslow et al. 1998a, Subsection 5.4.3 therein).

3.4.6 Simulations of salinity using the transport model

A simplified version of the *SEEM* model (Parslow et al. 1998b) was run to simulate the salinity observed during the surveys HES 2 to HES 10, using daily river flows from 1/1/96 to 26/9/98 (1000 days). The modelled and observed salinities at the times of the salinity surveys are shown in Figure 3.15. The agreement between model and observations is fair and provides an indication of the validity of the fluxes produced by the inverse model.

3.4.7 Flushing times

It is common practice to compute flushing times for estuaries and coastal embayments. These can provide a useful indicator of the relative importance of physical and biogeochemical processes. However, flushing times must be used cautiously with estuaries such as the Huon, which has strong downstream and vertical gradients.

The flushing time for any given volume is usually computed by dividing the volume by the volume flux through the boundaries. When this is applied to tracers, it is implicitly assumed that the volume interior can be treated as well-mixed with respect to the tracers. The Huon Estuary is certainly not well-mixed with respect to salinity or other conservative tracers.

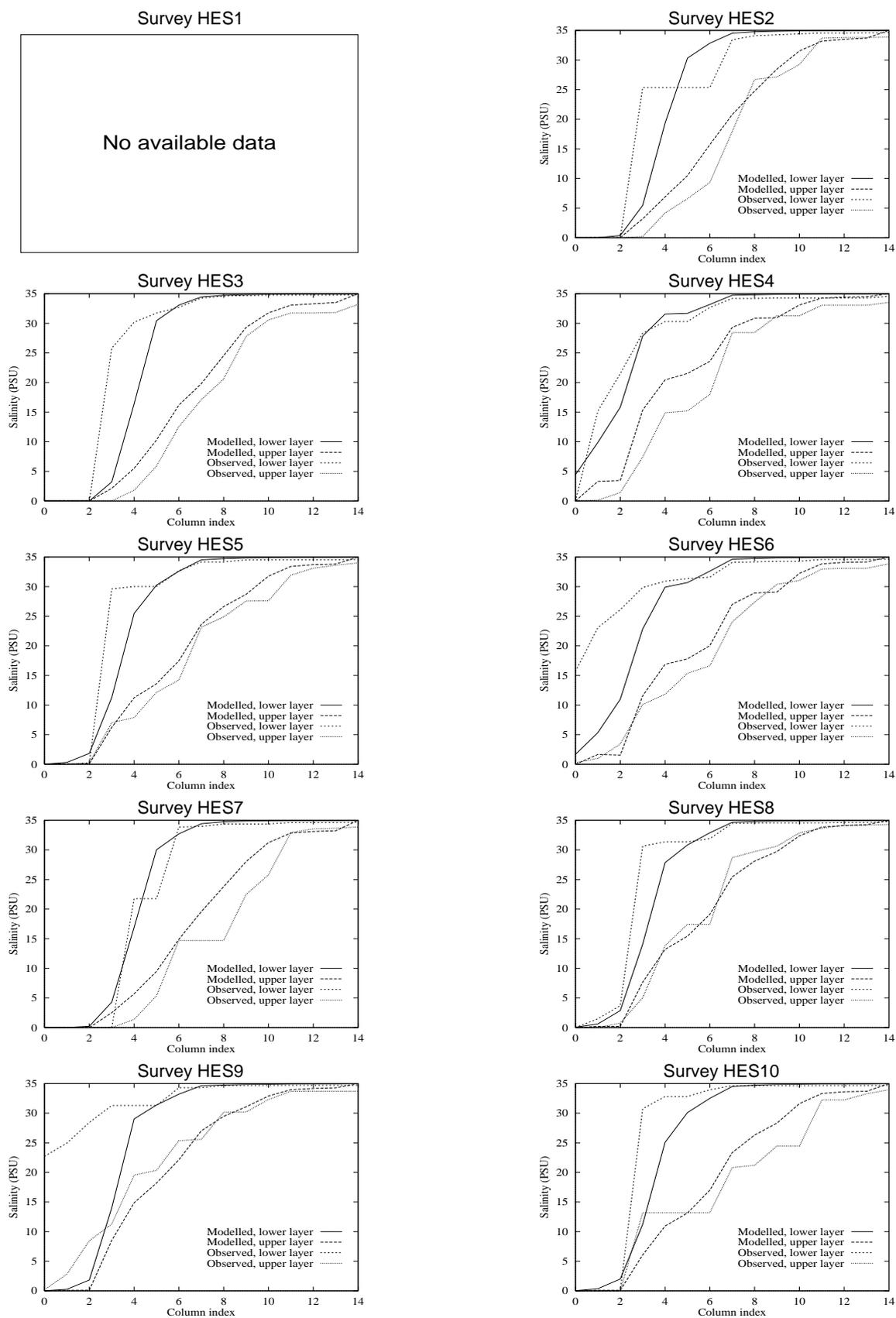


Fig. 3.15 Observed salinities and salinities simulated by the transport model.

To give some idea of the effects of these inhomogeneities, we have computed flushing times for the upper estuary (columns 0 to 6), upper plus middle estuary (columns 7 to 10), and whole estuary (columns 11 to 14). In each case, we compute flushing time for the upper layer alone, and for upper and lower layers combined. For each combination, we have divided the specified volume by the total downstream flux through the downstream face of the volume, as computed by the inverse model. We have computed flushing times for two extreme river flows: 25 and 250 m³ s⁻¹.

The results are shown in the table below. Flushing times for the surface layer alone are very short, especially at high flow rates. Flushing times for the entire water column are longer: about 6 days for low flows, but for high flows, about 1 day for the upper and middle estuary, and 2.5 days for the whole estuary. These short flushing times have significant implications for nutrient cycling and phytoplankton blooms in the estuary.

Table 3.4 Flushing times for the Huon Estuary and its different zones for low and high river flows.

	UPPER		UPPER + MIDDLE		UPPER + MIDDLE + LOWER	
River Flow (m ³ s ⁻¹)	25	250	25	250	25	250
Surface layer	1.1 d	0.2 d	0.8 d	0.2 d	1.3 d	0.6 d
Surface + Bottom	2.4 d	0.4 d	5.8 d	1.1 d	5.6 d	2.5 d

One must be cautious in interpreting these flushing times. The concentration achieved by an individual tracer from a point source will depend on its circulation throughout the estuary. For example, negatively buoyant particles may be retained more efficiently, as they sink into the bottom layer where the flow is predominantly upstream. Some of these effects are explored in Chapter 10.

3.5 Summary

The Huon Estuary is clearly a complex system, not unlike the neighbouring Derwent Estuary, with which it shares a similar inflow of fresh water (Subsection 3.1.6) and a similar geometry (Subsection 3.1.3). Longitudinal exchange in the estuary is dominated by the effect of the buoyancy-driven circulation caused by freshwater inflow (Subsection 3.1.2), although some features of the salinity distribution (e.g. its spatial patchiness) may be attributed to the effects of wind action (Subsections 3.1.8, 3.3.1 and 3.3.2). Currents are generally weak (< 0.2 m s⁻¹ – Subsection 3.1.7) with those due to tides being typically 0.02 m s⁻¹ (Subsection 3.1.4). There is clear evidence that the fresher water tends to flow down the eastern and northeastern coasts of the estuary, presumably under the influence of wind and of the rotation of the Earth.

An inverse model of the estuary was developed to provide exchange fluxes for transport and ecological models (Section 3.4). This model indicates that the exponential flushing time of the entire estuary is about *one week* at low river flows, declining to 2.5 days at high flows (Subsection 3.4.7). The surface layer is flushed even more rapidly.

3.6 References

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4 NUTRIENTS AND WATER QUALITY

4.1 Introduction

Water quality is the foundation of a healthy aquatic ecosystem. Deterioration in its quality usually results in loss of habitat, of biodiversity and of public amenity.

Estuaries are a transition zone between land and sea. In their confines, solutes and suspended particulate matter passing from catchments to coastal seawaters are modified or removed. Marine material conveyed from the coastal waters can also be processed there. Moreover, estuaries can be important zones of incipient production. The mix of conditions — physical and chemical — favours biological production. In turn, estuarine microalgae, macroalgae, seagrasses and other plant forms serve as food and shelter for a variety of fauna, including juvenile stages of many fishes.

The main sources of chemical substances in catchment streams are weathering of rock to release dissolved minerals; leaching of soils and breakdown of plant material to augment both dissolved organic and inorganic substances; and flushing of land surfaces to sweep in suspended solids. These natural pathways contribute to the chemical load of freshwaters discharging to an estuary. Water quality in estuaries is strongly influenced by the materials load delivered in surface run-off, but it can also be affected by direct inputs from groundwaters. Direct atmospheric deposition to the estuary surface, by way of rainfall or fallout of dusts, is another natural input; in Australia, it is usually a minor contribution.

Human practices in the catchment influence the natural processes — mostly by increasing the transport of material to waterways, or by adding new inputs to the system. Activities in the waters of an estuary and along its banks also impinge on environmental quality. The extent of anthropogenic effects depends on the size of the population, its distribution in the catchment, and the scale and type of enterprises in which it is engaged. In rural regions like the Huon, urban inputs are minimal. Point sources are linked to sewage-treatment-plant (STP) outfalls, leachates from tips or waste dumps (see Fig. 2.1), or direct discharge from agricultural operations (e.g. dairies and piggeries). Diffuse sources arise from wash-off from land recently cleared for forestry or farming. They also result from rainfall running over, and moving through, ground that has been treated (with fertilisers, pesticides or other substances) or modified by cropping or other farm activities. In addition, hydrologic modification — by impoundments, channel engineering, road building, irrigation and other activities affecting the natural course of run-off and streamflow — can cause contamination through point-source or diffuse inputs.

A range of natural in situ processes (Fig. 4.1) influences water quality and the cycling of chemical substances in estuaries. These include mixing between fresh and marine waters (from molecular diffusion up to larger-scale entrainment and stirring, and tidal exchange),

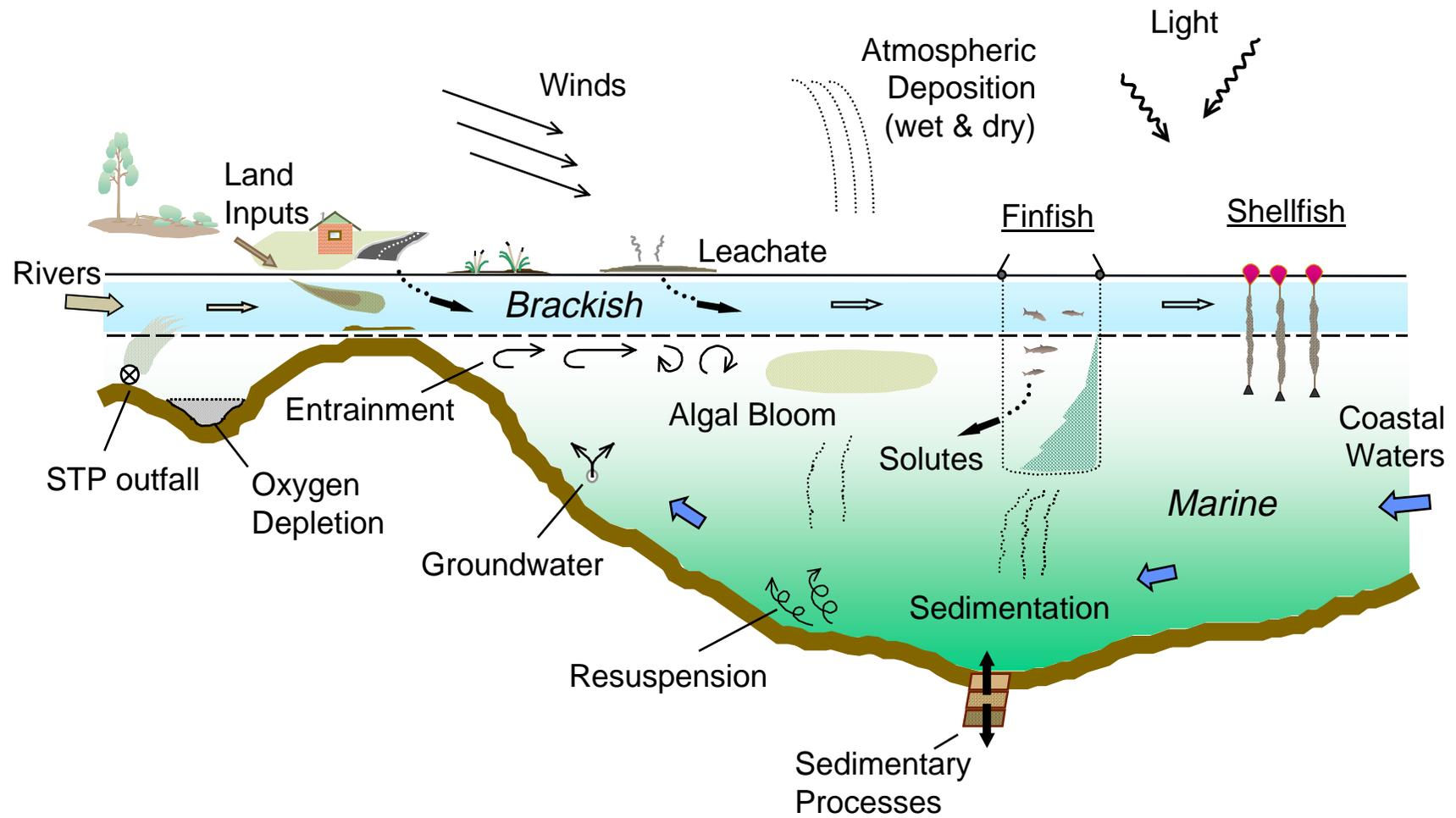


Fig. 4.1 Conceptual diagram of processes affecting nutrient cycling and transport in the Huon Estuary.

biological assimilation, flocculation, resuspension, decomposition and remineralisation of detritus and other suspended solids, sedimentation, redox processes, and exchange or concentration at interfaces (e.g. sediment-water, air-water and pycnocline). Adjacent natural features, such as fringing wetlands, mudflats, seagrass beds, and rocky shores, also play a role — usually beneficial.

The ultimate fate of most chemical substances in estuaries is either deposition to estuarine sediments, or transport to coastal waters. There are some exceptions to these two sinks; for example dissolved gases can be lost by evasion to the atmosphere. Denitrification in the water column or in sediments forms molecular nitrogen and nitrous oxide — either of which provides a route for nitrogen from estuary to atmosphere.

This chapter looks at the water quality of the Huon Estuary. Although it focuses on the nutrient elements nitrogen, phosphorus and silicon, it also covers suspended solids and dissolved oxygen. Trace inorganic and organic contaminants, in both waters and sediments, are discussed in Chapter 9. Important concepts in estuarine chemistry are presented in an introductory section immediately below. A brief summary of Australian water quality guidelines and a review of published information on water quality of the Huon Estuary follow. Then the distributions of nutrients, dissolved oxygen (DO) and suspended solids from our observations are discussed in both spatial and temporal contexts. We look to identify key influences — both within the estuarine water body and external to it. Cycling of nutrients over seasonal scales are considered, but the interplay between nutrients and microalgae is considered more fully in Chapter 5. Estimates of nutrient loads to the estuary — in fact, the presentation of nutrient budgets — are tackled in Chapter 10.

4.1.1 Estuarine chemistry

The behaviour of dissolved nutrients and other solutes in an estuary is usually related to salinity as an index of mixing. If the concentration of a chemical substance throughout an estuary is simply the dilution of one end-member (riverine or marine) with the other, the relation with salinity is a straight line joining its concentration value in river water with that in seawater. In this circumstance, the chemical substance is said to behave *conservatively*. If a chemical substance has been removed or added to during its transit of the estuary, this is usually manifest by a non-linear relation with salinity over the estuarine gradient. The behaviour is then said to be *non-conservative*.

This simple analysis can be complicated by the limitations of the field data in resolving linear and non-linear relations, or by artefacts arising from the non-synoptic [asynchronous] nature of observations over the full estuary. Short-term variations in river input or the influence of secondary tributaries can lead to misleading assessments of non-conservative behaviour (Morris 1985).

Whether an individual chemical species (e.g. NO_3^- , Mn^{2+} , cysteine, atrazine) or class of chemical compounds (e.g. DOC, organic mercury, PON, tannins) behaves conservatively or not during estuarine mixing depends on their reactivity and the nature of the estuary. Factors that influence the outcome have been mentioned above in Section 4.1 and earlier in Chapter 3 (e.g. type of estuary). The main one is time. An estuary with a rapid flushing time, in the order of days, does not give geochemical or biological processes an opportunity to change the concentration of

chemical constituents introduced in run-off or by other means. However, once the flushing time extends to weeks or months, the rates of reaction (geochemical and biochemical) are sufficient to discernibly add or remove chemical substances. Non-conservative behaviour becomes more prominent and prevalent.

This criterion is not adequate to fully delineate estuarine conditions that favour conservative and non-conservative mixing. Morris (1990) records that small estuaries (< 100 km in length) with dynamic and functional characteristics identical to large estuaries (\geq 200 km in length) are more likely to be non-conservative. He attributes this difference to the enhanced internal mobility of estuarine “resuspendable” sediment in the smaller estuaries, which typically have proportionately greater particle resuspension and transport over the estuarine (salinity) gradient.

As to where in the estuary substances are added or removed, it very much depends upon the type of estuary (Church 1986). For example, a fully mixed macrotidal estuary with a high suspended-solids load is more likely to function as a “geochemical reactor”. It is the interaction of solutes with particles that sees their removal or addition throughout the water column. Typically, the most reactive zone is the turbidity maximum at the low-salinity end of the estuary. In contrast, a microtidal, stratified estuary with a low concentration of suspended solids, and hence greater light penetration, will favour biological activity. The “biochemical reactor” is likely to be more pronounced toward the marine end, where light quality is better. Some estuaries can support both geochemical and biological reactors.

4.1.2 Water-quality guidelines for estuaries

Australian guidelines for the quality of natural waters evolved from local and international experience, beginning with the compilation of Hart (1974). The current guidelines for natural waters were produced by the Australia and New Zealand Environment and Conservation Council (ANZECC 1992) for fresh and marine waters. The most relevant category for natural waters is under ‘Protection of Aquatic Ecosystems’, which provides information on physico-chemical indicators and toxicants. Among the indicators are: salinity, temperature, suspended particulate matter / turbidity, DO, nutrients — ammonia ($\text{NH}_4\text{-N}$), nitrate ($\text{NO}_3\text{-N}$) and dissolved reactive phosphorus ($\text{PO}_4\text{-P}$) — and chlorophyll *a*. Apart from the nutrients (where concentration ranges are stipulated for estuaries) and DO, the other indicator guidelines suggest a threshold percentage by which the indicator should not depart from ambient or mean concentrations or values. The relevant guidelines for nutrient and water-quality parameters under the Protection of Aquatic Ecosystems criterion are presented in Table 4.1.

The ANZECC (1992) guidelines are currently being revised. Draft documents (ANZECC / ARMCANZ 1998) now available present these proposed new set of guidelines and the philosophy behind them. The premise from which they were derived is that “Australia and New Zealand contain a vast range of aquatic environments and ecosystem types, and in varying degrees of health, which requires a flexible approach to setting water quality objectives”. No standard set of guidelines will do. The new approach looks to tailor guideline “trigger values” for site-specific conditions, using hierarchical decision frameworks that are founded on a risk-analysis approach. A reference system or a set of baseline concentrations for the waterway of interest is the preferred course. However, when this information is not available, a set of ‘interim trigger levels’ is provided. These are included as part of Table 4.2.

Table 4.1 Guidelines for concentrations of nutrients and other water-quality parameters under the Protection of Aquatic Ecosystems criterion from the Australian Water Quality Guidelines for Fresh and Marine Waters (ANZECC 1992).

INDICATOR (Concentrations: $\mu\text{g L}^{-1}$) ^a	Riverine	Estuarine	Marine
Nitrate		10–100	10–60
Total Ammonia	20–30 ^b	< 5	< 5
Total Nitrogen	100–750		
Phosphate		5–15	1–10
Total Phosphorus	10–100		
Dissolved Oxygen (mg L^{-1})	> 6 (> 80–90% saturation)		
Suspended Particulate Matter	< 10% change in seasonal mean concentration		

a To convert $\mu\text{g L}^{-1}$ to μM , concentrations of N species need to be divided by 14, and P species by 31

b Concentration of undissociated ammonia (NH_3)

Table 4.2 Consolidated data on the concentrations of nutrients and other water-quality parameters during the Huon Estuary Study (1996–1998) from estuarine surveys and weekly / fortnightly monitoring.

WATER-QUALITY MEASUREMENTS (Concentrations, μM) ^a	Riverine (R5)	Estuarine (all sites)	Marine (A stns – deep)	ANZECC ^b river/estuary/sea
Nitrate + Nitrite	< 0.05–0.7	< 0.05–7.2	< 0.05–5.3	– / 0.36 / 4.3
Nitrite	< 0.03–0.06	< 0.03–2.25	< 0.03–1.49	—
Total Ammonia	0.05–0.14	< 0.02–2.60	0.06–1.00	– / 1.4 / 2.9
Total Dissolved N	7.5–15.4	< 0.5–26	2.8–11.5	—
Total Nitrogen	8.5–16.9	< 0.5–45	4.4–11.5	114 / 5.7 / 25
Dissolved Reactive P	< 0.03–0.09	< 0.03–0.71	0.18–0.64	– / 0.13 / 0.19
Total Dissolved P	< 0.08–0.54	< 0.08–1.86	0.25–0.69	—
Total Phosphorus	< 0.08–0.64	< 0.08–3.16	0.31–0.76	1.19 / 1.45 / 1.77
Dissolved Reactive Si	50.3–101	< 0.5 - 106	0.8–3.8	—
Dissolved Oxygen	330.2–349.6 ^d	9.1–370.0	205.5–275.6	—
DO (% saturation)	91.4–92.9 ^d	3.6–163	75.8–101.1	90 / 90 / 90
SPM ^c (mg kg^{-1})	0.8–3.0	0.1–12.6	0.1–1.8	6 / – / – ^e

a Concentrations in μM , except where indicated otherwise

b 'Interim trigger levels' for substantially natural ecosystems — Draft ANZECC / ARMCANZ guidelines (1998)

c Suspended particulate matter (\equiv nonfilterable residue) retained on 0.45- μm filter

d Two values only (HES 2 and 3, Jul. and Oct. 1996)

e Concentrations in mg L^{-1}

4.2 Background on Nutrients and Water Quality in the Huon Catchment and Estuary

A comprehensive, yet succinct, review of water quality information up to 1996 for the Huon catchment, its streams and waterways is provided by Gallagher (1996). The report's bibliography (and summary notes) includes a substantial number of early reports on Hospital Bay and the effects of the pulp mill, operating there from 1962 to July 1991 (interim closure 1982-1986). These are more relevant to Chapter 9, which deals with contaminants.

4.2.1 Nutrients and water quality in the catchment

Early measurements of water quality in the Huon River and tributaries were made by the State agencies and utilities (Department of Environment, Forestry Tasmania, Hydro-Electric Commission, and Rivers and Water Supply Commission). They were sporadic spot measurements often made as part of state-wide water-quality surveys, or monitoring associated with logging operations. The data are stored in the Tasmanian water-quality database HYDROL (contact Department of Primary Industries, Water and Environment for information). Gallagher (1996 – Appendix 4 therein) used HYDROL to compile median values for surface water from the headwaters of the Huon River and its tributaries down to the mouth of the estuary. Without sampling information (date, procedure, sample storage, etc.) and by their very sparseness, these data can be regarded only as indicative of conditions throughout the catchment.

More recently, Wotherspoon et al. (1994) sampled and determined some water-quality parameters in the Huon River and key tributaries over two days (7 & 9 September 1994). Bobbi (1998) has just reported on the most comprehensive survey to date of water quality in streams of the Huon catchment. His year-long study involved monthly monitoring at six sites (Huon River at Judbury, Mountain River at Ranelagh, Kermandie River at Kermandie, Rileys Creek upstream of Rileys Dam, Agnes Rivulet at Cygnet, and Nicholls Rivulet upstream of the tidal limit in Port Cygnet). In addition, he made two comprehensive snapshot surveys of streams throughout the catchment in summer 1996/97 and winter 1997.

Bobbi (1998) concluded that the streams in the upper Huon catchment (above Judbury) generally had high water quality. However, in the lower catchment, the Kermandie River and the Agnes Rivulet, and to a lesser extent Nicholls Rivulet and Prices Creek, were degraded. This was manifested by high concentrations of suspended solids (or turbidity level), nutrients and faecal coliforms. The water quality of the lower Kermandie River and Agnes Rivulet was affected by land-use practices in the subcatchments, and by effluents discharged from STPs and drainage from domestic septic tanks. Conditions in both Prices Creek and Nicholls Rivulet are also influenced negatively by land-use practices in their subcatchments; and the former suffers from its proximity to the township of Franklin. Some of the smaller streams above Huonville (Dickensons, Watsons and Fourteen Turn Creeks), flowing through populated areas, do present sporadic examples of diminished water quality from increased turbidity and faecal contamination.

4.2.2 Nutrients and water quality in the estuary

Apart from a scattering of spot measurements in the Huon Estuary (see Gallagher 1996 – Appendix 4 therein), only two surveys have been made of the water body.

Woodward and co-workers (1992) carried out a broad environmental survey of the middle and lower Huon Estuary during 1988 and 1989. This included water column sampling and analysis between June 1988 and December 1989. They used a ‘Lund tube’ to collect an integrated surface-layer sample (down to pycnocline) and also a sample of the integrated water column (down to the bottom or 20 m — whichever was the shallowest). The samples were analysed for the nutrients, dissolved reactive phosphorus, nitrate, nitrite and ammonia. Results are plotted in their report for 10–12 surveys (depending upon the nutrient) between July 1988 and August 1989.

All nutrients followed a seasonal cycle, as expected for an aquatic system influenced by biological activity. Concentrations of dissolved reactive phosphorus [range: 0.05 ~ 1.0 μM] were lower in the surface layer than in the full water column — especially in spring and summer. This trend was not so obvious at sites in the lower estuary; Woodward et al. 1992 suggest it was because the water column there became increasingly “mixed”. Nitrate concentrations [range: < 0.1 ~ 3.0 μM] “showed a marked seasonal change, more pronounced than that of phosphate”. Its prolonged near-zero concentrations in summer suggested that nitrate was a limiting nutrient for phytoplankton. The integrated surface-water concentrations lagged those of the full water column by about two months during the autumn “recovery”. Nitrite [range: < 0.03 ~ 0.8 μM] showed very similar behaviour to nitrate in its seasonal cycle. However, the period of depletion was much shorter than for nitrate. The highest nitrite concentrations over winter were observed in the middle estuary. They declined both seaward and landward. Ammonia concentrations [range: 0.1–12.7 μM] were very substantial at times in the estuary. The highest values were recorded in surface waters in winter (1989) at Brabazon Point, opposite Brooks Bay (downstream of Wheatleys Bay) and in Port Cygnet.

Burgess et al. (1993) measured physicochemical (temperature, salinity, turbidity, visibility [\equiv Secchi depth], pH, redox potential, dissolved oxygen and DOM [\equiv gilvin]) and biological (chlorophyll *a*, plankton, invertebrates and ‘vegetation’) properties at five sites in the Huon River and Estuary. The survey was done on a single two-day period in early autumn (29–30 March 1993). They found that water quality was generally good. DO was above 75% saturation at all sites. Turbidity (via light transmission) decreased from 31 Formazin Turbidity Units at Tahune on the upper Huon River to ~20 FTU at the Franklin–Cradoc transect, and dropped sharply to 1 FTU at Granny Gibbons Bay site in the lower Huon Estuary. Gilvins (g440: 12.78 to 0.58) showed a very similar trend to turbidity moving from fresh to marine water. Secchi depths were the reverse, increasing from about 2 m to ≥ 15 m moving from Tahune to Granny Gibbons Bay, as expected for increased water clarity at the marine end.

Other pertinent observations made by Gallagher (1996) include intermittent evidence of the elevated nutrient concentrations as a result of human settlements or activities (stormwater, effluents from STPs, drainage from septic tanks, leachate from refuse tips, and forestry operations in the upper catchment). He concludes his review with the statement that “available water quality data indicate no serious water quality problems in the Huon region”, but they are “drawn from a highly fragmented data set with severe limitations in terms of data validity and comparability”.

4.3 Nutrients and Water Quality in the Huon Estuary (1996–1998)

4.3.1 Sampling and analytical methods

Water samples for assessing water quality were collected by the means and procedures described in Chapter 2 and in the supplements to this report. Chapter 2 describes the basis of our survey design. The report supplements provide details on operations in the field, processing and storage of samples, and the methods used in physical and chemical analysis. They also provide information on the performance characteristics (Figures-of-Merit) of analytical methods, as well as the means of validation and quality assurance.

Precision salinity was determined by conductivity (Cowley 1999). Dissolved oxygen was determined by Winkler titration on water samples brought back to the laboratory with the minimum of delay. Standard flow-analysis procedures (Plaschke 1999) were modified slightly for the determination of the nutrients, nitrate, nitrite, dissolved reactive phosphorus and dissolved reactive silicon. Ammonia was determined by a newly developed flow injection method (Watson et al. — *in prep.*)

Some definitions are needed for the following subsections. *Dissolved* generally refers to analysis of the filtrate after passing water samples through a 0.45- μm filter membrane; a synonym also used now in aquatic chemistry is *filterable*. The exception to this statement is dissolved oxygen, where we are referring to the hydrated gas in the water sample (the sample is not filtered!). *Ammonia* is used in place of total ammoniacal nitrogen (i.e. ammonia + ammonium). The fraction of free ammonia is very relevant to finfish farmers, because free ammonia is very toxic to fish (Handy and Poxton 1993); it can be calculated from equilibrium thermodynamics given the pH of the water sample (ANZECC 1992). Nitrate + nitrite (or NO_x) are often reported together. Sometimes nitrate is used as an abbreviation for nitrate + nitrite, but here when we speak of nitrate or nitrite, we will be referring to the specific anionic form in each case.

By *phosphate* and *silicate*, we mean dissolved reactive phosphorus and dissolved reactive silicon. The variants of the ‘phospho-molybdenum blue’ method are not selective for orthophosphate, or orthosilicic acid and its ionic forms, but will also measure other reactive condensed phosphates and silicates (dimer only). Possibly some organic fraction of phosphorus and some colloidal silicic acid will also be included in the analytical measurements (Koroleff 1983, Robards et al. 1994).

The following are arbitrary classifications of groups of nitrogen and phosphorus species that arise from our choice of analytical procedure. *Total nitrogen* and *total phosphorus* are determined as nitrate and phosphate, respectively, after chemical oxidation of unfiltered water samples. *Total dissolved nitrogen* and *total dissolved phosphorus* are the same determinations made on filtered water samples. *Particulate nitrogen* and *particulate phosphorus* are the difference between total nitrogen and total dissolved nitrogen, and total phosphorus and total dissolved phosphorus, respectively. *Dissolved organic nitrogen* is determined by subtracting the summed concentration of dissolved inorganic nitrogen species (nitrate + nitrite + ammonia) from the concentration of total dissolved nitrogen. *Dissolved organic phosphorus* is determined by subtracting the concentration of directly measured phosphate from that of total dissolved

phosphorus. The dissolved organic phosphorus is likely to include a fraction of labile polyphosphates.

4.3.2 Overview of nutrients and water quality

In this subsection, the full collection of results from all spatial surveys and weekly / fortnightly monitoring is consolidated. The concentration range for each nutrient species, DO and suspended particulate matter is presented in Table 4.2, along with the ‘interim trigger levels’ for a substantially natural ecosystem from the new draft Australian Water Quality Guidelines for Fresh and Marine Waters (ANZECC / ARMCANZ 1998) as a reference.

The interim trigger levels are specifically not put forward as a standard national set of guidelines. Indeed, they have mostly been derived from observations of estuaries in lower latitudes than Tasmania. Nonetheless, it is useful to begin our discussion with the interim trigger levels, because there is no reference system or set of baseline values for the Huon Estuary. We can then move on to discuss instances where the trigger levels are exceeded as a result of natural processes or human activities.

The Huon River discharges the bulk of the freshwater (~80% — from catchment area–rainfall calculation) to the Huon Estuary. Its quality is very high. Our measurements were from the Huon River at Judbury (Site R5). Not only are the total nutrient concentrations below the interim trigger levels for a lowland river shown in Table 4.2, but they are also below those of an upland river (total N 24 μM and total P 1.1 μM). The guidelines do not categorise the form of nitrogen or phosphorus in rivers. The highest concentrations of suspended particulate matter in the Huon River were still only half of the interim trigger level. Only two dissolved oxygen determinations were made at R5 (both giving an oxygen saturation greater than 90% — the minimum guideline threshold). One year’s data (Oct ’96–Oct ’97), collected monthly at the same station by Tasmanian Department of Primary Industries, Water and Environment, gave oxygen saturation values in the range 91–105% (median: 95.8%). At the next site downriver, R4, oxygen saturation over all estuarine surveys ranged from 85–94% (median 88.8%). Our impression is that the river is well oxygenated above the head of the estuary.

The other tributary that we sampled was the Kermantie River (Site R6). Its water quality was poorer, as recognised in previous studies (Subsection 4.2.1). We consider its condition, along with that of its tributaries in a special case study below (Subsection 4.3.9).

In contrast to the Huon River, the highest concentrations observed in its estuary exceed the interim trigger levels for all of the listed estuarine water quality guidelines (Table 4.2). This is apart from dissolved oxygen saturation, for which a low value is of consequence, and suspended particulate matter, for which no estuarine or marine trigger levels are suggested. Rather than a reflection of poor water quality, the exceeding of the interim trigger levels is often a result of the regional differences anticipated in the framing of the revised ANZECC / ARMCANZ guidelines.

Dissolved oxygen concentrations in marine waters (A transect) were generally within the guidelines. Concentrations less than the trigger level of 90% saturation arise from consumption of dissolved oxygen in bottom waters in summer, probably as part of a natural seasonal cycle of decomposition of organic matter. Estuarine concentrations of dissolved oxygen in the Huon

Estuary vary over a wide range, at times grossly under the interim trigger level; but these occasions seem to be few, short-lived, and limited in the volume of water body affected. These circumstances are elaborated further in the next subsection.

Concentrations of nitrate + nitrite and dissolved reactive phosphorus in the Huon Estuary were routinely higher than their interim trigger levels, as were the 'marine' waters (subsurface waters of A transect). These high concentrations result mainly from the incursion of sub-Antarctic waters into D'Entrecasteaux Channel, as in neighbouring Storm Bay (Clementson et al. 1989).

The reason for total nitrogen exceeding its trigger level is twofold: the contribution of sub-Antarctic waters from the marine end is combined with the high concentrations of total nitrogen (mostly dissolved organic nitrogen) derived from the humic substances in many streams of the Huon region. Total phosphorus also has two influences. The highest concentrations of total phosphorus were observed in estuarine waters, which have inputs (mostly as dissolved reactive phosphorus) from the marine end, but sporadic sources were also evident in the estuary. Their proximity on several occasions to STPs suggests that they could be one such source.

The peak levels of total ammonia in the estuary were twice the interim trigger level. Its concentrations in marine waters were lower (and below the higher interim trigger level), but not down to the background concentrations observed in nearby Southern Ocean waters (< 80 nM in surface waters — Watson and Berry 1996). The inflow of the Huon River (R5) had low concentrations (< 200 nM) throughout our study. However, the peak concentrations of total ammonia in the estuary were typically patchy and transient. As with total phosphorus, total ammonia is influenced by natural processes and human activity. Its estuarine distributions will be elaborated in the following sections. Apparent sources for ammonia include STP effluents, occasionally salmon farms, and seasonal remineralisation. The last was particularly pronounced in bottom waters (and presumably in the surface sediments) of the summer of 1997/98, in the aftermath of dense microalgal blooms (see Subsection 4.3.5 below). Sediments can also be a diffusive source of ammonia, when their redox status at or just below their surface is reducing.

4.3.3 Estuarine distribution and behaviour of dissolved oxygen

Waters of the Huon Estuary in our surveys were generally close to oxygen saturation (80–100%), with no clear trend with salinity (Fig. 4.2), and small differences between surface and subsurface samples. The exceptions to these prevailing conditions were either localised or transitory.

Bottom samples in the vicinity of Egg Islands at sites R1, N1, N2 and L1 were systematically undersaturated. The strongly reducing (anoxic) sediments in this deltaic or depositional zone of the estuary around Egg Islands seemed to be the cause. Their consumption of dissolved oxygen (DO) appeared to be sufficient to draw down the concentration in the isolated subsurface (subhalocline) waters in the upper estuary. This depletion of DO peaked in summer (Fig. 4.3), when microbial respiration would be at its most pronounced. Nevertheless, we did not observe DO fall below 40% saturation in the upper estuary.

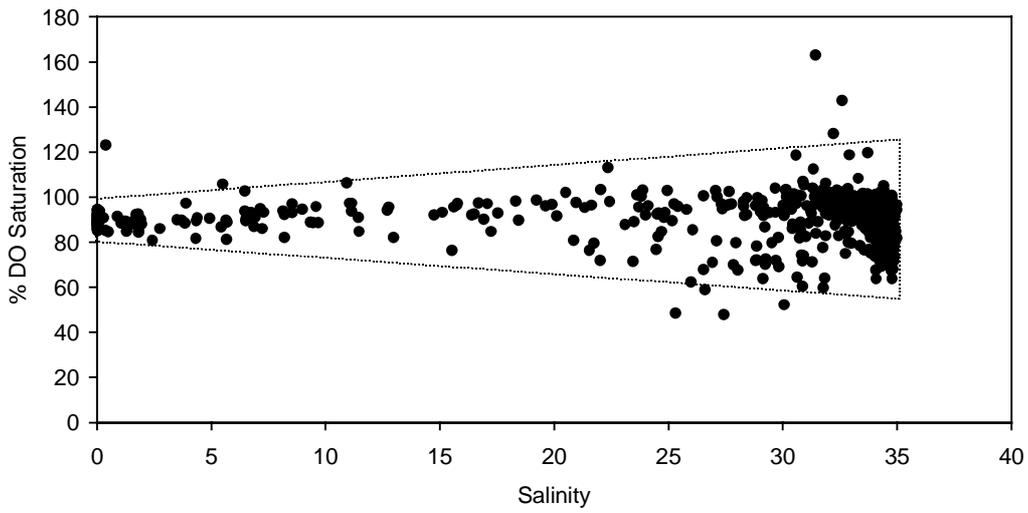


Fig. 4.2 The relation of dissolved oxygen saturation to salinity for HES surveys 2–10A.

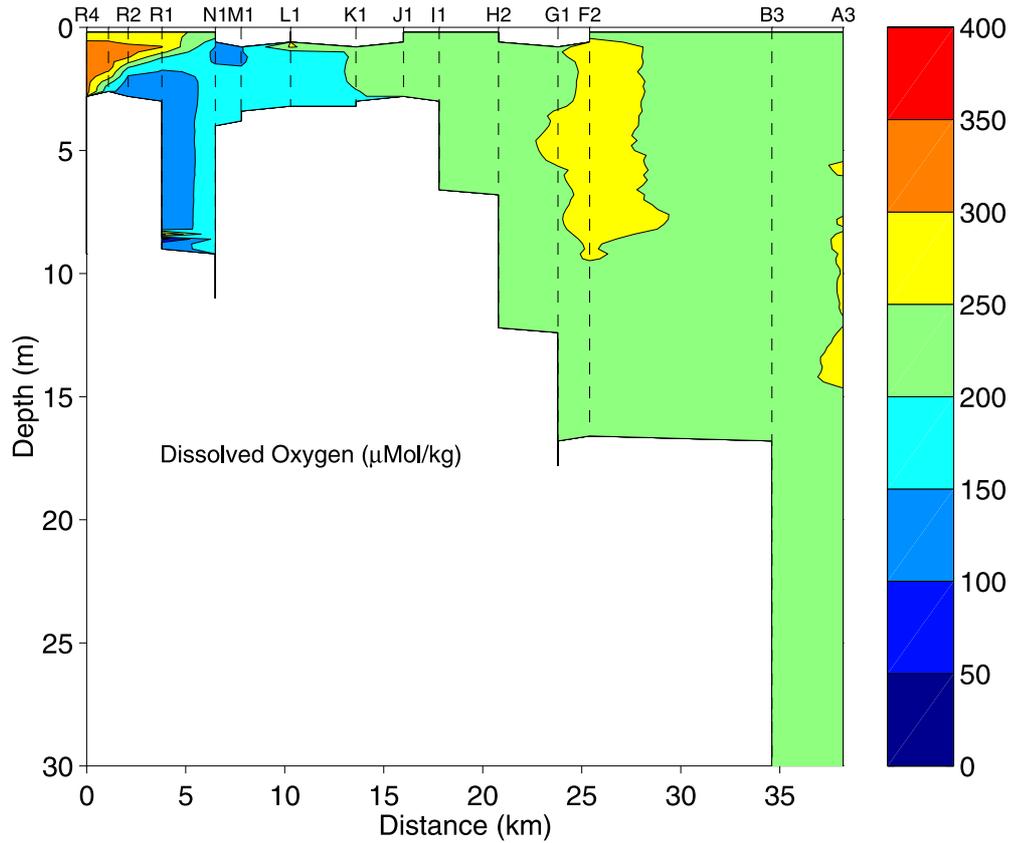


Fig. 4.3 Dissolved oxygen section along the main axis of the Huon Estuary (HES 4, February 1997), illustrating the depletion of oxygen in bottom waters of the upper estuary. Dissolved oxygen isopleths are defined by the colour gradient to the right.

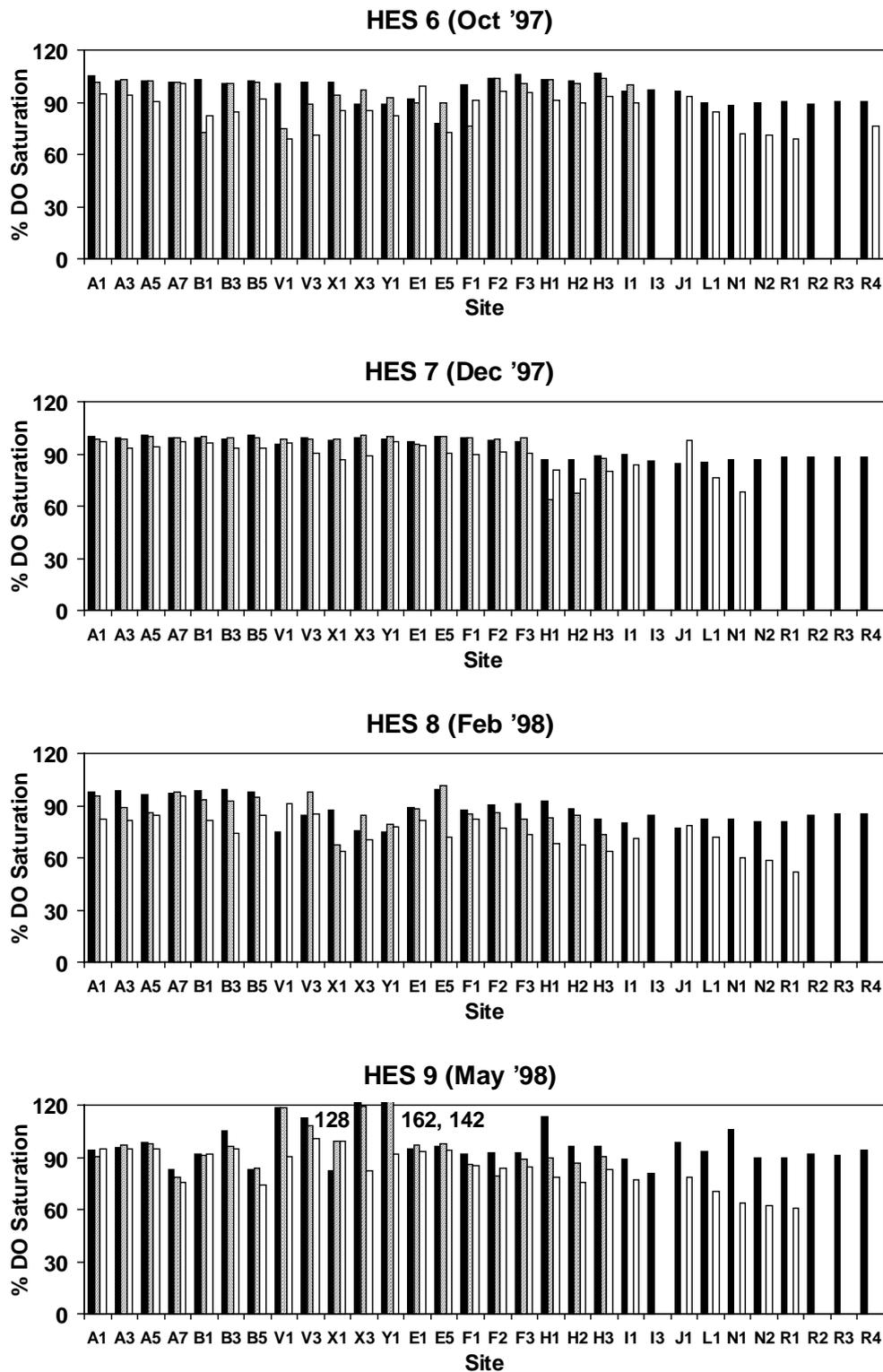


Fig. 4.4 Dissolved oxygen saturations at sites throughout the estuary for HES surveys 6–9 during late spring and summer of 1997/98 — surface (solid bar), mid-depth (hatched bar), bottom (open bar). The site denoted as X3 is X3B from HES 6 onward.

Occasionally during estuary-wide surveys (Fig. 4.4), other subsurface samples were found to be mildly undersaturated (60-80%). During the extensive phytoplankton blooms of 1997/98 (see Chapter 5), sporadic DO depletions were noted at several lower estuary sites (e.g. Port Cygnet stations during HES 6 (Oct '97), Brabazon Park vicinity [H2, H3] during HES 7 (Dec '97), middle of main estuary [transects E-H] as well as Port Cygnet in HES 8 (Feb '98) and a few scattered stations [A7, B5 and bottom waters of H sites] in HES 9 (May '98)). An anomaly during HES 9 (May '98) was the supersaturation of surface and mid-depth samples throughout much of Port Cygnet. An extreme value of 160% was recorded at site Y1 at the head of Port Cygnet. These conditions were during a dense microalgal bloom (see Subsection 5.4.1).

Monitoring at weekly and fortnightly intervals at the biological stations — all of which were adjacent to aquaculture leases (4 finfish, 1 shellfish) — revealed that the spatial surveys had not picked up much greater depletions of dissolved oxygen. The very lowest DO concentrations (down to a DO concentration of 9.1 μM , 3.6% saturation) were recorded near the salmon farm at Hideaway Bay (site B1) on four occasions during December 1997 and January 1998 (Fig. 4.5). Other instances of low DO concentrations were noted at Deep Bay (Port Cygnet, site X3B — shellfish farm), and Killala Bay (F1 — salmon farm).

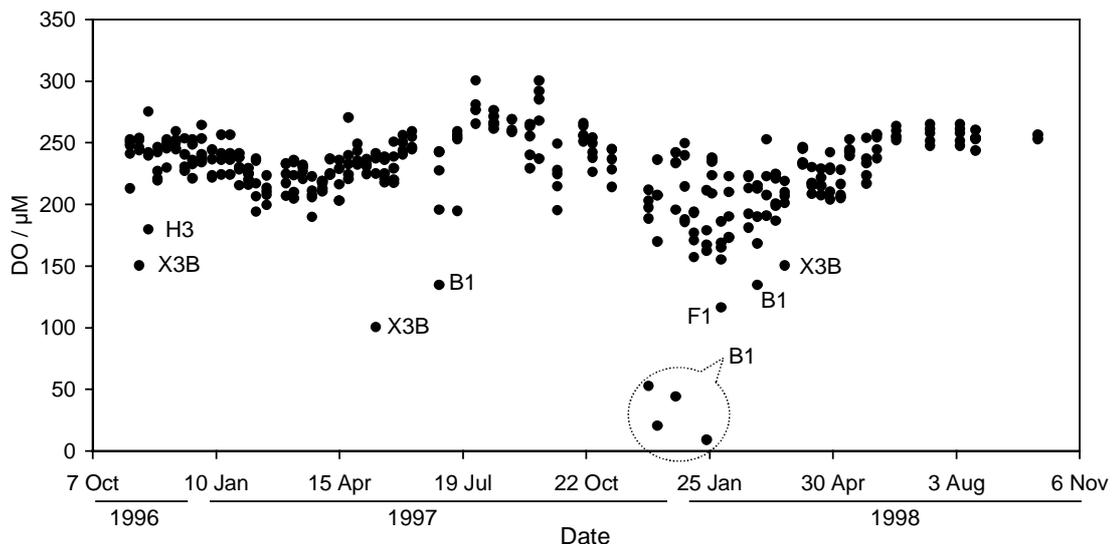


Fig. 4.5 Dissolved oxygen concentrations at the five biological sites during weekly / fortnightly monitoring (1996–1998). Sites are identified for oxygen-depleted conditions.

4.3.4 Estuarine distribution and behaviour of suspended particulate matter

Suspended particulate matter was measured in the spatial surveys only. However, the results from these surveys give a consistent picture, so it is unlikely that weekly or fortnightly monitoring at the five biological stations would have provided much more. Moreover, it was the upper estuary where the between-survey variability in concentrations of suspended particulate matter was observed.

As mentioned above (Subsection 4.3.2), we found the Huon Estuary low in concentration of suspended particulate matter (SPM). Both the freshwater¹ and marine end-members had very low concentrations: the former $\leq 3 \text{ mg kg}^{-1}$, and the latter $< 2 \text{ mg kg}^{-1}$. Apart from a very few instances, concentrations in the estuary were less than 6 mg kg^{-1} . SPM concentrations in surface waters of the upper estuary correlate with increased river flows (flow gauge at Frying Pan Creek) almost without delay (≤ 1 day). However, SPM concentrations in subhalocline, marine waters of the upper estuary lag flows in the Huon River by several days; the best correlation for the filters used in Chapter 3 (Table 3.1) was 6.7 days ($R^2 = 0.9282$; one outlier removed). Hence, ~ 7 days must elapse before the suspended particles have either settled or been mixed into these waters.

The upward curved relation of the concentration of SPM with salinity, peaking between 5 and 10 units, suggests that an ‘input’ of particulate matter occurred at these brackish salinities. An example is provided from HES 7 (Dec '97 – Fig. 4.6). We suspect that this feature was associated with flocculation of river-borne material. Detritus measured as the detrital absorption coefficient also had a corresponding peak in this salinity range (see Subsection 6.3.2).

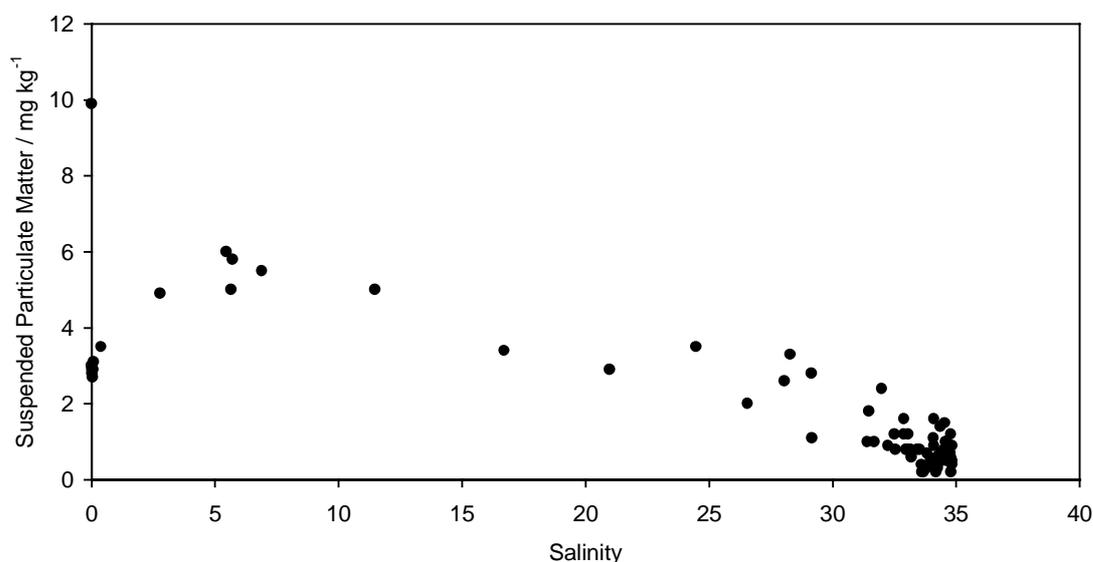


Fig. 4.6 The relation of suspended particulate matter with salinity for HES 7 (December 1997).

The Kermadie River was a consistent source of suspended solids. At site R6, the concentration was regularly between 6 and 12 mg kg^{-1} , but in HES 9 (May '98) it was markedly greater at 39 mg kg^{-1} . Additional site-linked inputs, smaller in scale, appear to be in the vicinity of Egg Islands (*viz.* L1, N1, N2 and R1), but this observation is complicated by salinities often being between 5 and 10 in this region.

¹ *freshwater* — for the purposes of this and the following subsections is the Huon River at R5 and whichever samples at other R sites have a conductivity reading that makes them indistinguishable from zero salinity

4.3.5 Estuarine distribution and behaviour of nitrogen species

In this subsection, we consider the individual species of inorganic nitrogen (nitrate, nitrite and ammonia) before discussing the broad categories of dissolved organic nitrogen, particulate nitrogen and total nitrogen.

Nitrate

Nitrate showed a pronounced seasonal (biological) cycle in its estuarine concentrations. Figure 4.5 depicts this behaviour in a plot of nitrate measurements from the weekly / fortnightly monitoring at the five biological stations (data for all three depths). In most estuarine waters, nitrate was strongly depleted, often to below the analytical detection limit ($< 0.05 \mu\text{M}$) in spring, summer and early autumn. In late autumn, it was restored to the incipient concentration in marine waters of $3\text{--}5 \mu\text{M}$ and remained steady through winter until rapid depletion in spring began another cycle.

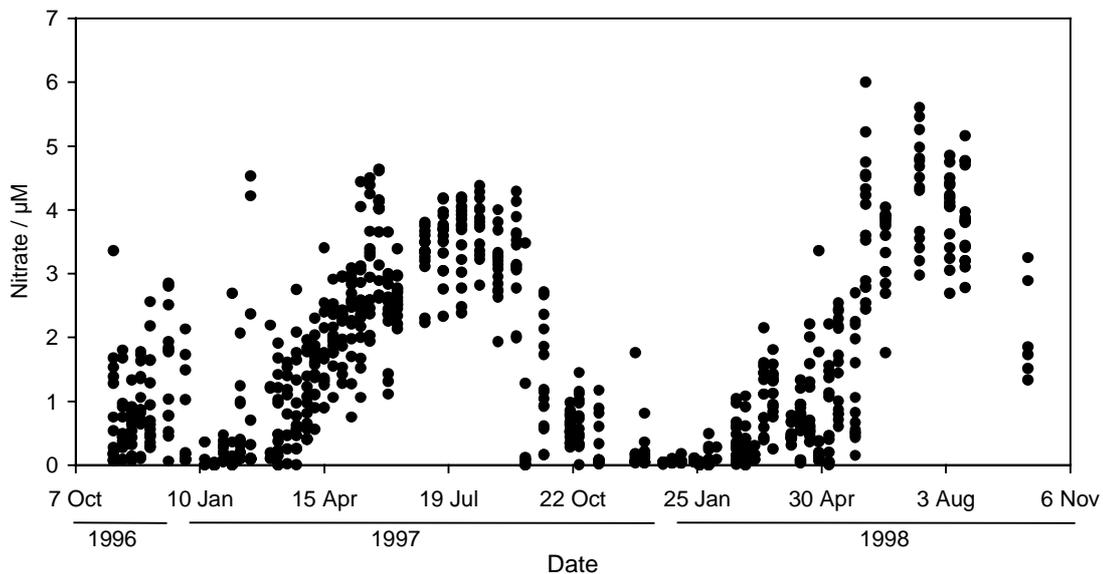


Fig. 4.7 Variation of nitrate concentrations at the five biological sites (3 depths each) with time, from weekly / fortnightly monitoring.

The estuarine distribution of nitrate was marked by low concentrations at the freshwater end ($< 1 \mu\text{M}$). In winter, nitrate behaved quasi-conservatively; the general trend against salinity was linear to the marine end-member concentration of $3\text{--}5 \mu\text{M}$. Surface and subsurface concentrations were usually similar at this time of the year. Interannual differences (or variation from one winter to the next in how much nitrate was replenished by remineralisation) caused a change in the slope of the nitrate-salinity relation. The main factor was change in the marine concentration of nitrate, although small fluctuations in the freshwater concentration were evident. In spring, summer and early autumn — under the influence of biological activity — nitrate was strongly removed from estuarine waters. However, a store of nitrate was present throughout the year in subsurface marine waters. This is seen graphically where high nitrate concentrations persist for some marine salinities in nitrate vs. salinity plots (Fig. 4.8).

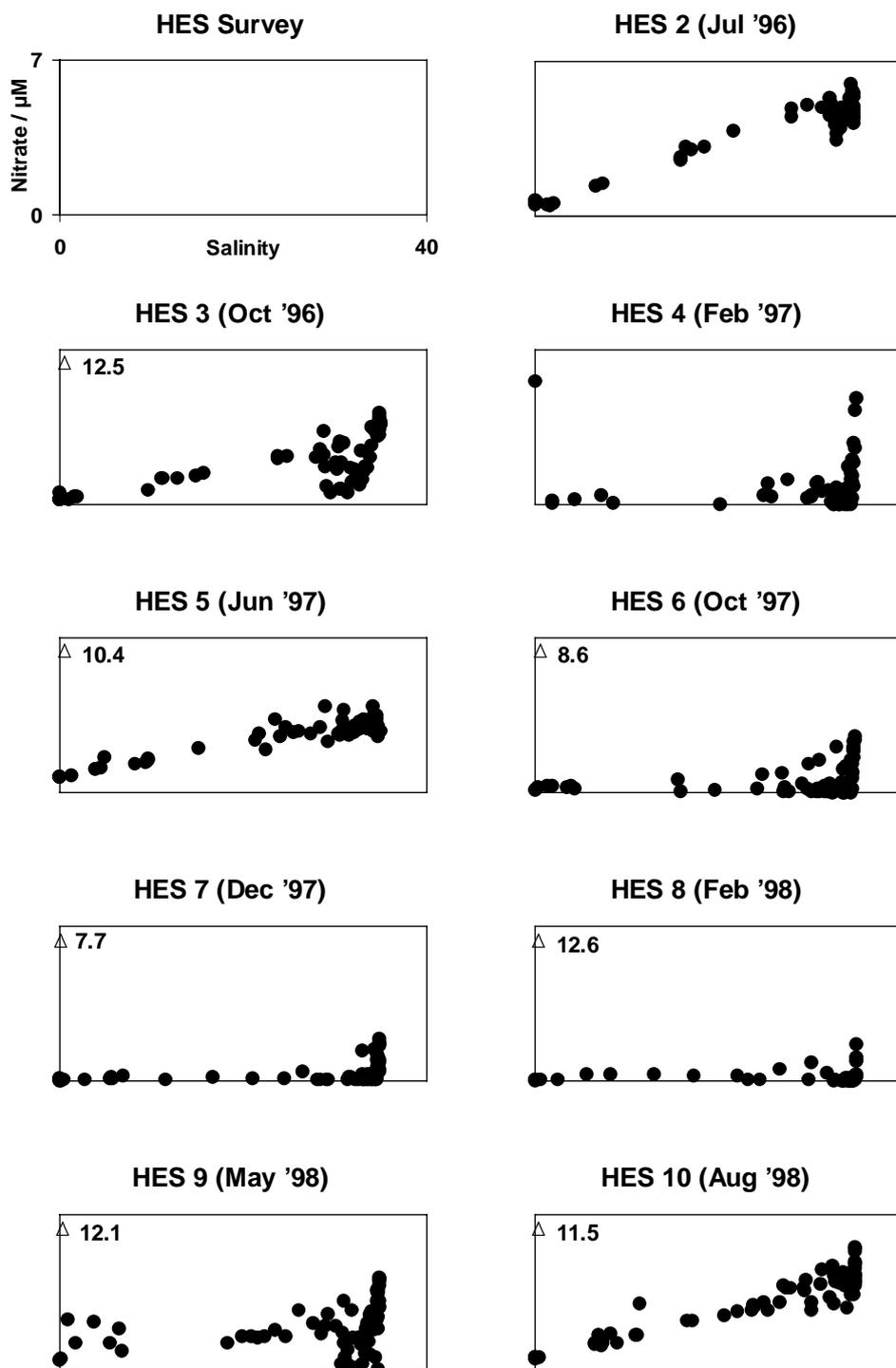


Fig. 4.8 The relation of nitrate to salinity for HES surveys 2–10. Indicated off-scale values are at site R6.

The contrast of the biologically active seasons (depicted here as ‘summer’) with the biologically dormant one for nitrate is presented schematically in Figure 4.9.

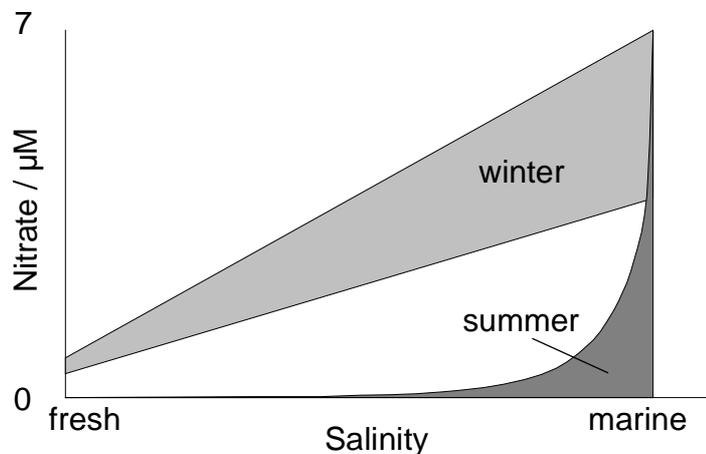


Fig. 4.9 Schematic diagram of the behaviour of nitrate against salinity in winter, contrasted with that for summer; spring and autumn have distributions in between the two.

Nitrate removal in the spring of 1996 (HES 3) was greatest at Port Cygnet surface stations; and in that summer (HES 4), the same region was distinguished by nitrate being absent from all depths. Furthermore, it is presumed that isolated biological activity in Port Cygnet during late autumn and winter, particularly that of 1998 (HES 9, May '98), caused the anomalous depletion of nitrate in this side-arm of the estuary.

Nitrite

Nitrite showed broadly similar seasonal and mixing trends to nitrate, but usually at an order of magnitude lower concentrations. For example, the freshwater end of the mixing line again had the lowest concentrations, but for nitrite they were $< 0.15 \mu\text{M}$. Site R6 on the lower Kermadie River had atypically high and variable concentrations of nitrite (range: $0.17\text{--}1.56 \mu\text{M}$; median: $0.64 \mu\text{M}$).

In winter, nitrite seemed to depart from conservative behaviour with scattered high concentrations in the bottom waters of the upper estuary (Fig. 4.10). However, a truly anomalous late autumn and winter was that of 1998 (HES 9, May '98 — see lower left panel): the nitrite concentrations were at least twice those observed in 1996 (HES 2, Jul '96) and 1997 (HES 5, Jun '97). At up to $2 \mu\text{M}$, nitrite was more than a third of the concentration of nitrate, and it behaved more like nitrate. It had a near-straight-line relation with salinity apart from surface samples from Port Cygnet and in the lower estuary, where it was strongly depleted. It is presumed that in these locations microalgae were satisfying nitrogen requirement by assimilating nitrite and nitrate in similar proportions.

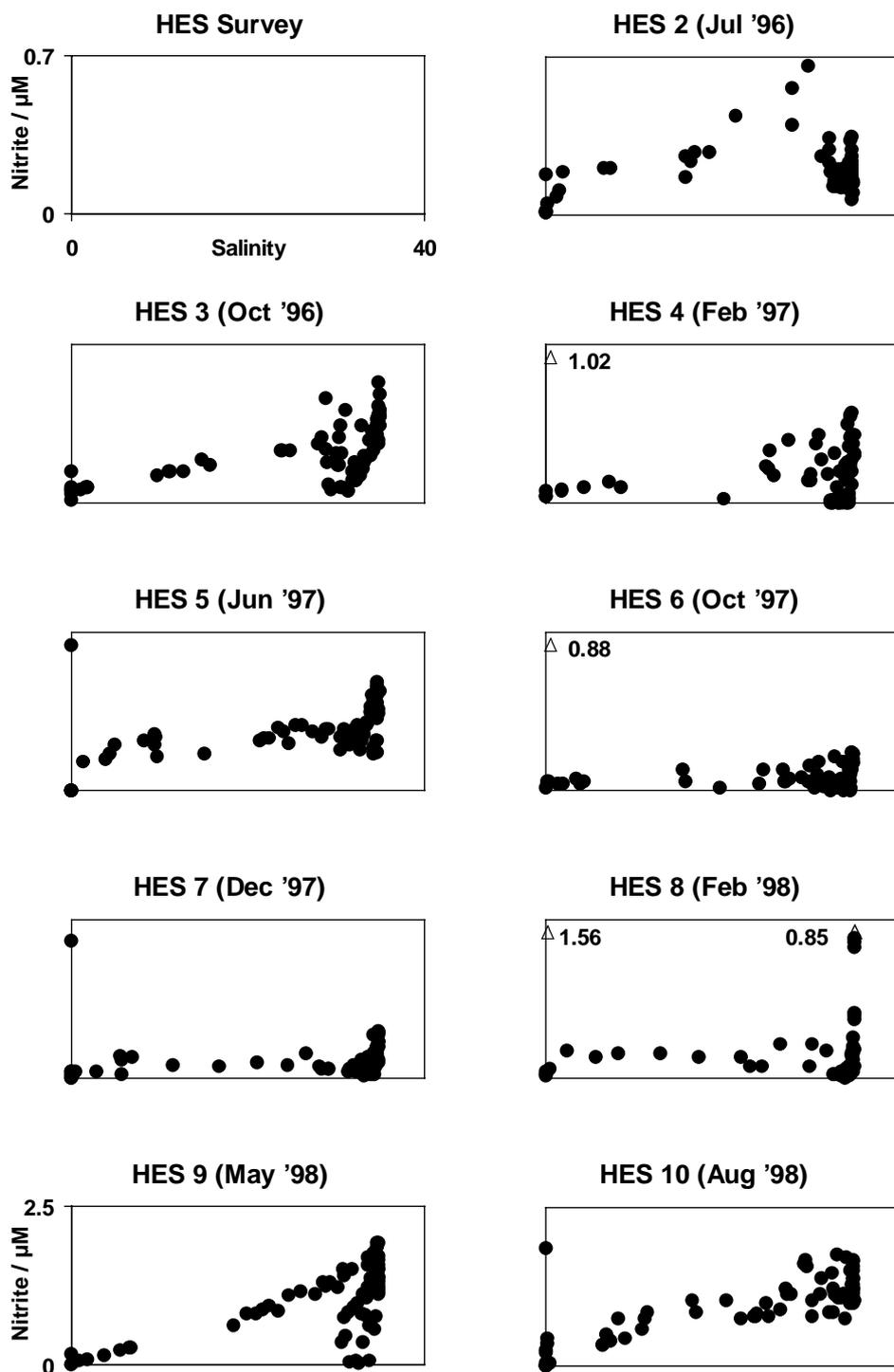


Fig. 4.10 The relation of nitrite with salinity for spatial surveys of the Huon Estuary, HES 2–10 (1996–1998). Note different scale for left-hand bottom panel only. Indicated off-scale values on y-axis are at site R6.

A probable explanation for the elevated concentrations of nitrite in mid-1998 begins with the dense microalgal blooms of the preceding spring, summer and autumn. Dead microalgal cells and detritus were deposited on the sediments of the lower estuary and Port Cygnet. The copious quantities of organic nitrogen were remineralised to ammonia (see below), presumably by heterotrophic activity during the warmer months. In the autumn, the ammonia was then nitrified to nitrite (i.e. $\text{NH}_4^+ \rightarrow \text{NO}_2^-$), some of which was presumably oxidised subsequently to nitrate.

Ammonia

Concentrations of ammonia in estuarine waters were generally low ($< 1 \mu\text{M}$). To measure ammonia reliably, a new sensitive method of determining ammonia was required, coupled with careful sample handling and processing (Watson — *in prep.*). These procedures were implemented from HES 4 (Feb '97) onward.

In surface waters, the relation with salinity was slightly inverse, with indication of ammonia input in the upper estuary. On several occasions, site R3, immediately downstream of the effluent discharge from the Ranelagh STP, had slightly higher ammonia concentrations, but still less than $1 \mu\text{M}$. The consistent source of ammonia in this part of the estuary was the reducing sediments around Egg Islands in the upper estuary (Fig. 4.11). Higher concentrations of ammonia (peak values in the range $0.5\text{--}3.0 \mu\text{M}$) were routinely detected in the subsurface waters of sites L1, N1, N2 and R1, and on occasion also pervaded their surface waters. All evidence pointed to an efflux of ammonia from the sediments. Ammonia concentrations were inversely, but weakly, related to those of dissolved oxygen in bottom waters for all HES surveys ($R^2: -0.4881$). Factors other than dissolved oxygen (as an indirect measure of redox potential) can influence ammonia distribution.

Another constant source of ammonia was the Kermadie River downstream of the Geeveston STP (site R6). Concentrations ($2\text{--}11 \mu\text{M}$) were an order of magnitude higher than in the estuary. Although the river empties directly to the middle estuary, its discharge is small, and so is its ammonia load (see Subsections 4.3.9 and 10.2.3).

In the lower estuary, concentrations at mid-depth were not too different from those at the surface. However, bottom waters were different. They had frequent examples of increased concentrations of ammonia, most evident in summer. Interannual variability was also manifest, with widespread ammonia concentrations of $\sim 1 \mu\text{M}$ in bottom waters in spring (HES 6, Oct '97) and summer (HES 7, Dec '97) of 1997. We postulate that this ammonia was generated from the remineralisation of organic nitrogen in the remains of dead microalgae deposited on the estuary floor after the crash of dense blooms. A residual trace of this process appeared to linger in bottom waters in the mouth of the estuary in autumn (HES 8, Feb '98, transect A).

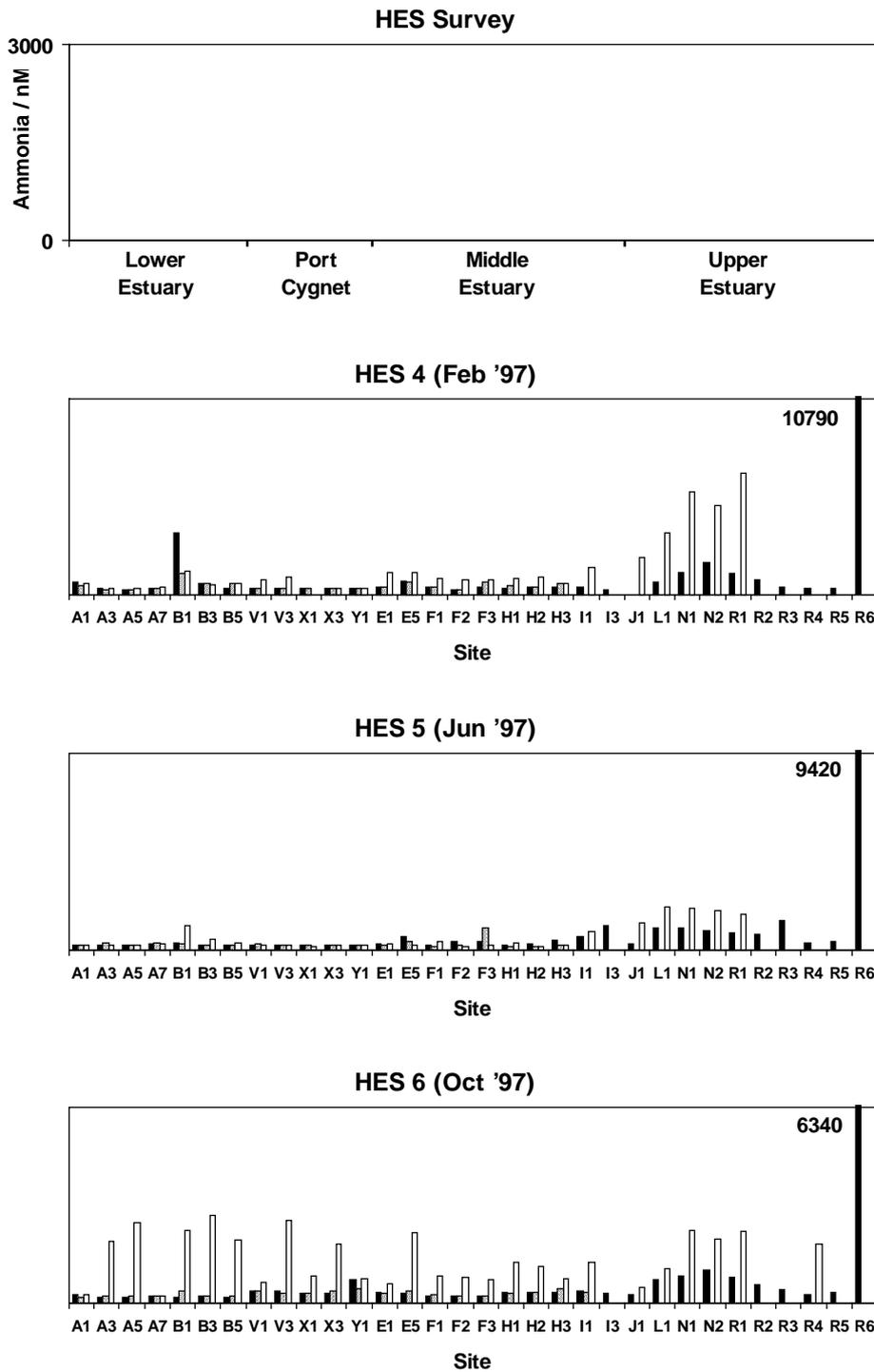


Fig. 4.11 Variation of ammonia concentration with site and depth during estuarine surveys (HES 4–6). Ammonia concentrations were routinely higher at sites around Egg Islands, such as L1 and N1, because of its efflux from anaerobic sediments. Key is surface (solid bar), mid-depth (hatched bar) and bottom (open bar). The site denoted as X3 is X3B from HES 6 onward.

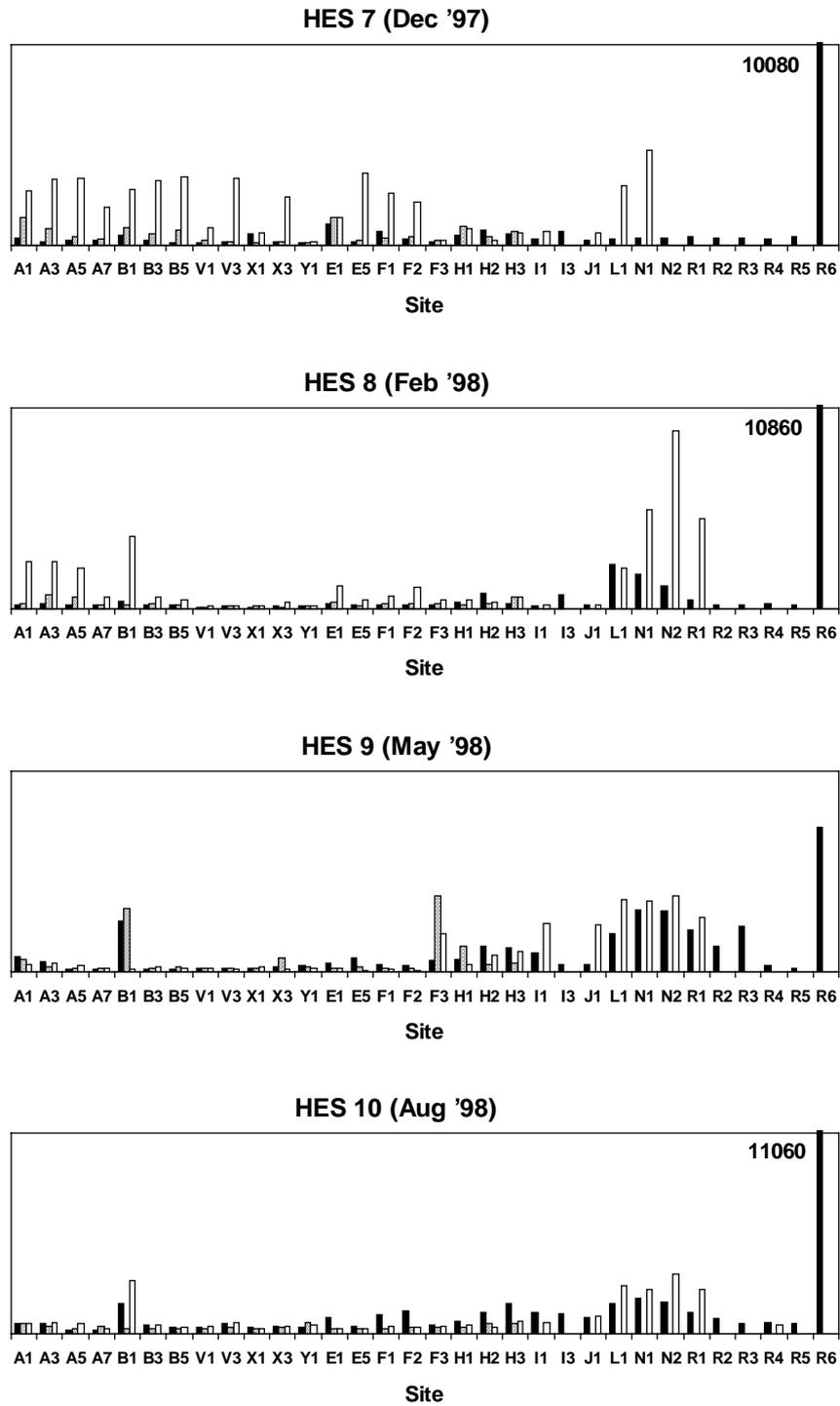


Fig. 4.11 (cont.) Variation of ammonia concentration with site and depth during estuarine surveys (HES 7–10). The ammonia concentration at R6 (Kermandie River below Geeveston STP) is universally off-scale with the adjacent number indicating its concentration. Key is surface (solid bar), mid-depth (hatched bar) and bottom (open bar).

Although fish — salmon included — excrete appreciable quantities of ammonia (Handy and Poxton 1993, Kibria et al. 1997), it is often difficult to detect evidence of this ammonia in the receiving environment near farmed fish because it is rapidly dispersed and is readily assimilated by micro-organisms.

We performed an ANCOVA (analysis of covariance) of ammonia concentrations *vs.* site from all surveys for which we had data (HES 4–10). In the one-way ANCOVA, we set the ammonia concentration as the dependent variable, the site as the independent variable, and each survey as the covariate (the influence of seasonality is removed with the survey treated as the covariate). We tested for significant differences in the mean concentrations of ammonia between sites A to I in the main channel, and V to Y in Port Cygnet. The region covers the marine part of the estuary, and where marine farms are located. Ammonia concentrations in the surface and mid-depth waters there were generally low and uniform (Fig. 4.11). The variations observed in bottom waters have already been discussed above.

The sequence of mean concentrations for surface and mid-depth waters is depicted in Figure 4.12.

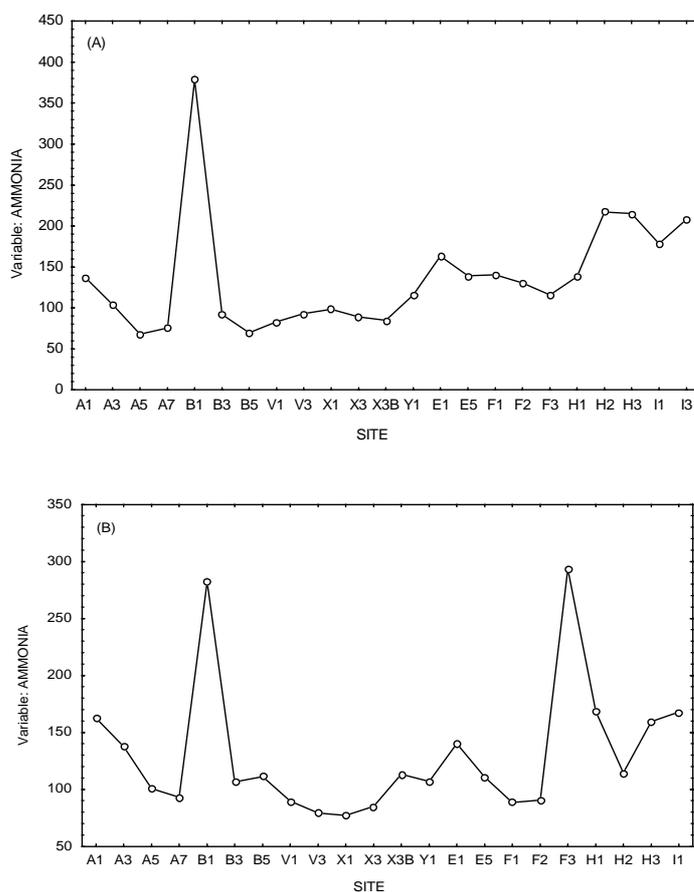


Fig. 4.12 Series of mean concentrations of ammonia at sites in the lower and middle reaches of the Huon Estuary and Port Cygnet (a — surface waters, b — mid-depth waters) considered in the ANCOVA test. The precision (coefficient of variation) of ammonia determinations was 5.7% at 800 nM.

The results of the ANCOVA test reveal that surface waters at site B1 were significantly different ($P < 0.005$) from all other sites in the middle and lower estuary. Mid-depth waters at this same site were significantly different ($P < 0.05$) from all others apart from A1, F3, H1, H3, X3 and I1. Similarly, at site F3, mid-depth waters were significantly different ($P < 0.05$) than all others in the middle and lower estuary, apart from A1, B1, H1, H3 and I1. An LSD (least squares difference) test for homogeneous groups ($\alpha = 0.05$) places B1 in a 'group' of its own for surface waters, and A1, B1, F3, H1, H3 and I1 in a distinct group for mid-depth waters (four of these six sites are beside salmon farms).

The detection of ammonia in waters near a fish farm is erratic; it depends on various factors, such as current direction, proximity of sampling sites to fish cages, farm biomass and stocking density, and fish condition and activity. Furthermore, the stocking of some leases is intermittent — Wheatleys Bay (site F3) is an example. The higher concentrations of ammonia noted at F3 did coincide with times that fish were present (T. Dix — *pers. comm.*, October 1999).

Stratification of the water column can constrain the excreted ammonia to certain depths. We believe that this explains why in the middle estuary that the influence of salmon farms on ammonia is only observed subsurface. The 'prescribed depths' of the interface — or halocline — between surface and bottom waters (Table 3.1 — generally representative of the range of 'observed' layer depths) are 1.4 m for the F sites (cell 10), and 5.5 m for the B sites (cell 13). If salmon are typically swimming at about 5-m depth, they are below the interface (in marine bottom waters) at site F3, and just above the interface (in surface waters) at site B1. Toward the mouth of the estuary, the halocline is more diffuse and the salinity contrast between surface and bottom layers is smaller. Therefore, mixing is more prevalent between the layers, and is probably the reason that higher ammonia concentrations are found at both depths at B1.

Dissolved organic nitrogen

Freshwater concentrations of dissolved organic nitrogen (DON) fell mostly between 9 and 16 μM ; marine concentrations were mostly 1–8 μM . The inverse relation with salinity was close to linear and strongly correlated for individual surveys (R^2 : -0.6403 to -0.9690). This is quite surprising given that this arbitrary grouping of organic nitrogen will include compounds as diverse as urea, which is labile and biologically mediated, and humic substances, which are expected to be refractory and effectively biologically inert. Carlsson et al. (1999) suggest that even humic substances are available to some communities of microbes and microalgae. Perhaps our results indicate the dominance of humic substances and other N-containing macromolecular compounds that are not processed by biota in the time-scale of estuarine transit. The estuarine distribution of this terrigenous organic nitrogen is simply the result of river water being diluted with seawater; marine DON must be similarly refractory (see also Section 10.3).

Variation in the riverine concentrations of DON in the Huon River (6.8–15.2 μM at R5, median: 10.1 μM) did not appear to be seasonal, nor was there any clear relation with riverflow. DON concentrations in the lower Kermadie River at site R6 (range: 3.8–19.0 μM , median: 12.9 μM) were slightly above the concentrations in freshwaters of the Huon River, but not to the same degree as other forms of nutrients. Higher concentrations in summer and lower concentrations in winter suggested a seasonal trend for this tributary.

Particulate nitrogen

Uniformly low concentrations of particulate nitrogen ($< 1.5\text{--}5\ \mu\text{M}$) were observed over the estuarine gradient, independent of season. Scattered high values in HES 4 (Feb '97) and HES 9 (May '98) coincided with high microalgal cell counts. The latter, a late autumnal bloom, provided extreme examples. Elevated DON concentrations observed at site Y1 (surface: $31\ \mu\text{M}$, mid-depth: $9\ \mu\text{M}$) in Port Cygnet, and site B5 ($11\ \mu\text{M}$) just outside, were linked with high cell counts of the dinoflagellate *Gymnodinium catenatum* (see Subsection 5.4.1).

Although suspended solids in the Kermadie River were always high (see Subsection 4.3.9), it did not follow that particulate nitrogen concentrations (at R6) were also high. In fact, they were similar to estuarine concentrations on all but HES surveys 4 (Feb '97 – $20\ \mu\text{M}$) and 9 (May '98 – $17\ \mu\text{M}$).

Total nitrogen

The distribution of total nitrogen (TN) in the Huon Estuary is influenced by DON as the dominant form of the element. Therefore, total nitrogen has an inverse relation with salinity. It also mirrors DON in that the relation for each survey appears linear.

Freshwater concentrations of TN are typically $10\text{--}20\ \mu\text{M}$; in marine waters it is $3\text{--}10\ \mu\text{M}$. Figure 4.13 depicts the domain for TN concentrations in the estuary from all of our estuarine surveys and the weekly / fortnightly monitoring at the biological stations. In winter, the TN concentrations are toward the upper bound; they move downward during times of biological uptake of inorganic nitrogen (possibly, too, with some minor removal of labile DON — urea, amino acids, amines, etc.). Data lying outside this domain are almost without exception unusually high concentrations of TN associated with rare instances of high particulate N. In turn, these high concentrations of particulate N (and other nutrient species, as well) are linked with high microalgal biomass during dense blooms, as measured by chlorophyll *a* data or microalgal cell counts.

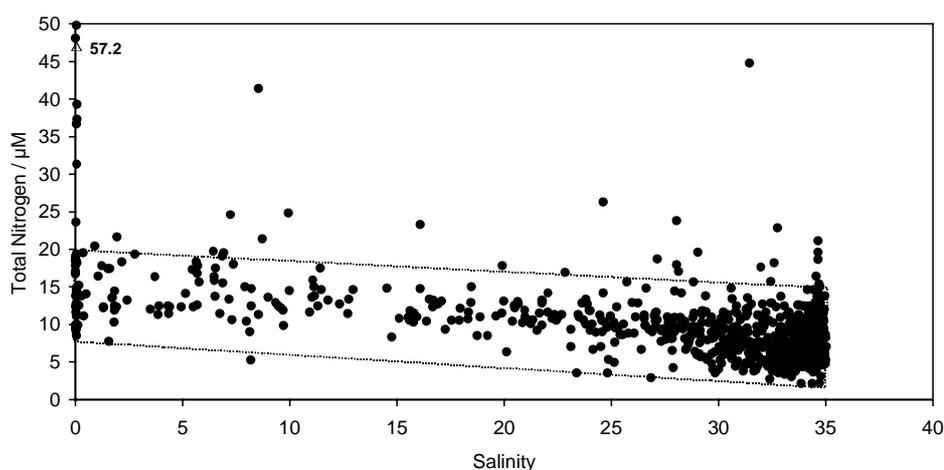


Fig. 4.13 The relation of total nitrogen to salinity for all HES data (spatial surveys and weekly / fortnightly monitoring). The high concentrations in freshwaters ($S = 0$) are from streams in the subcatchment of the Kermadie River.

The variability of TN in the freshwaters of the Huon River was possibly a result of different discharge rates coupled with seasonal factors. Webb and Walling (1985) observed that increased run-off in winter diluted nitrate in streamwaters, while the opposite was true in summer: increased run-off raised the nitrate concentration of the same streamwaters. Although it has not been demonstrated, it was conceivable that a similar process operated in the Huon catchment with DON (and other nitrogen species) in run-off.

4.3.6 Estuarine distribution and behaviour of phosphorus species

In this subsection, we consider the bioavailable form phosphate (as pointed out in Subsection 4.3.1, this measurement is more precisely described as filterable reactive phosphorus, a composite of several labile species), before discussing the broad categories of dissolved organic phosphorus, particulate phosphorus and total phosphorus.

Phosphate

Like nitrate, phosphate distribution in the Huon Estuary is markedly affected by biological activity. It manifests the same seasonal cycle with conservative estuarine mixing in winter, and removal in the other seasons. However, two key differences are that phosphate is depleted more gradually, and it is not depleted as much. An example is provided by a phosphate vs. salinity plot from HES 7 (Dec '97 – Fig. 4.14): the phosphate is depleted, but at most stations it is still above 50% of the winter concentration. At the same stage, nitrate was exhausted to concentrations at the detection limit or below, equating to >85% depletion of a typical middle-estuary concentration of nitrate in winter. This is possibly not unexpected behaviour for phosphate. The ratio of nitrate:phosphate was typically 10:1 in marine waters at the mouth of the estuary (see Subsection 4.3.8). If the microalgae assimilate N and P at Redfield ratios (16:1), P is in excess relative to N.

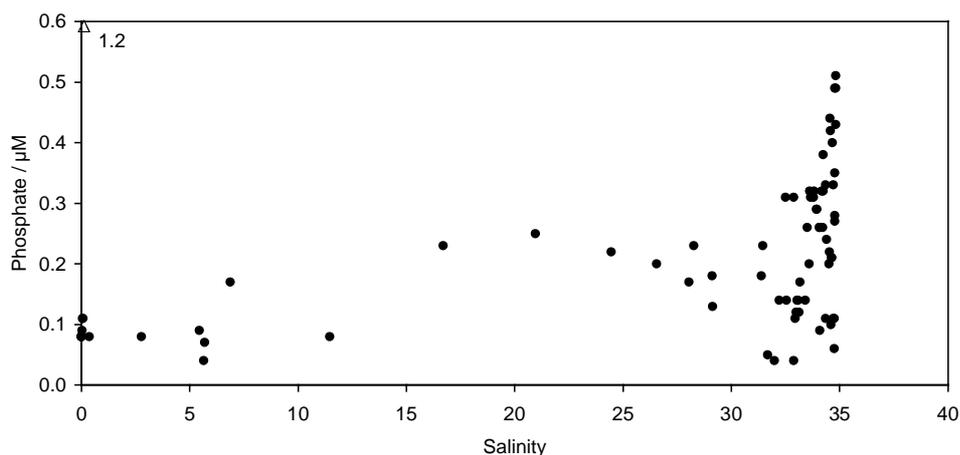


Fig. 4.14 The relation of phosphate (dissolved reactive phosphorus) to salinity for HES 7 (December 1997). The Kermadie River (site R6) value is off-scale as indicated.

On most surveys, the greatest depletion of phosphate was at sites in Port Cygnet; but during summer and autumn of 1997/98, Port Cygnet was matched or supplanted by sites along transects E, F and H in the main arm of the estuary (Fig. 4.15).

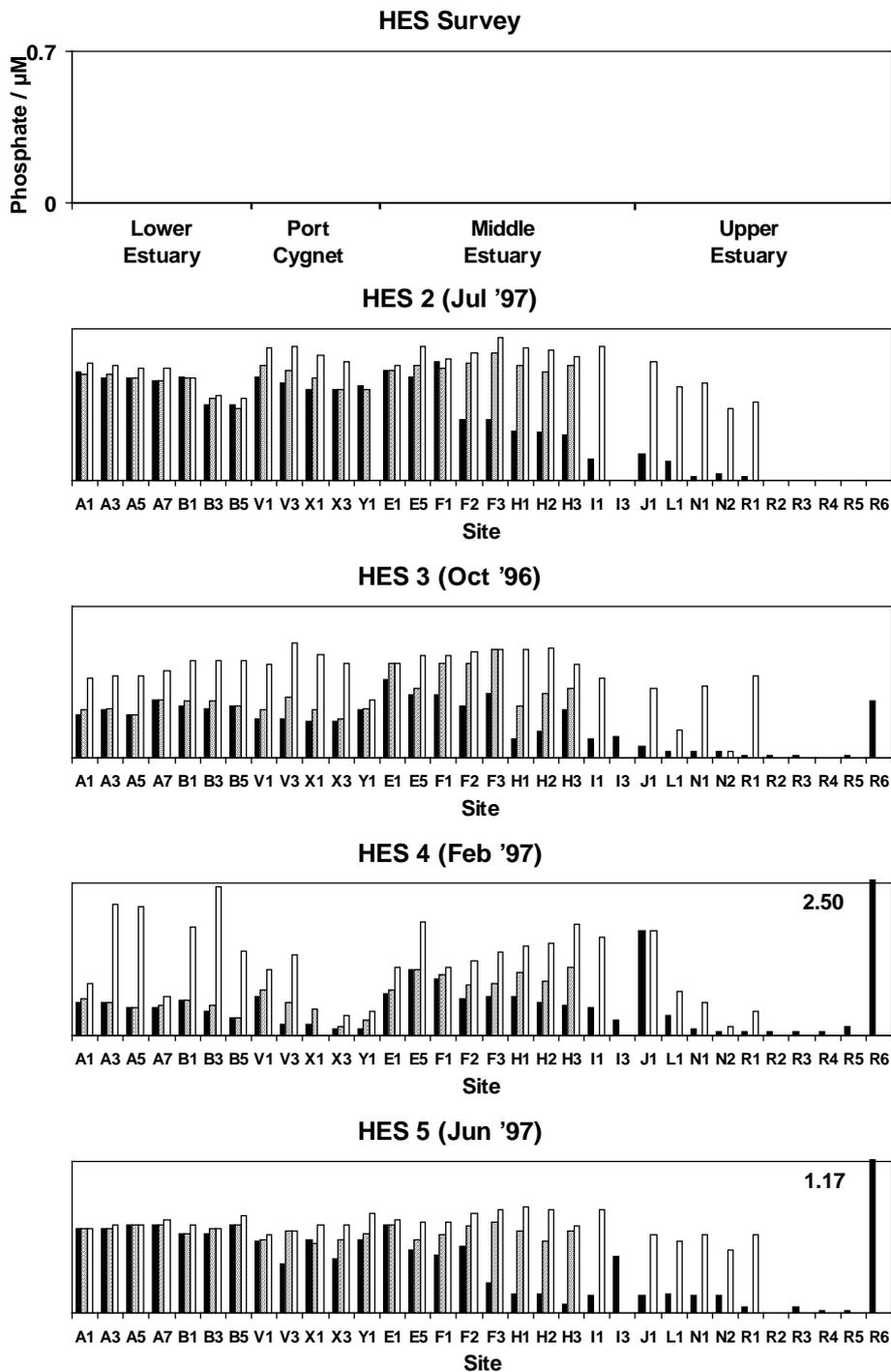


Fig. 4.15 Variation of phosphate concentration with site and depth during estuarine surveys (HES 2–5). The phosphate concentration at R6 (Kermadie River below Geeveston STP) is often off-scale, whereby the adjacent number indicates its concentration. Key is surface (solid bar), mid-depth (hatched bar) and bottom (open bar).

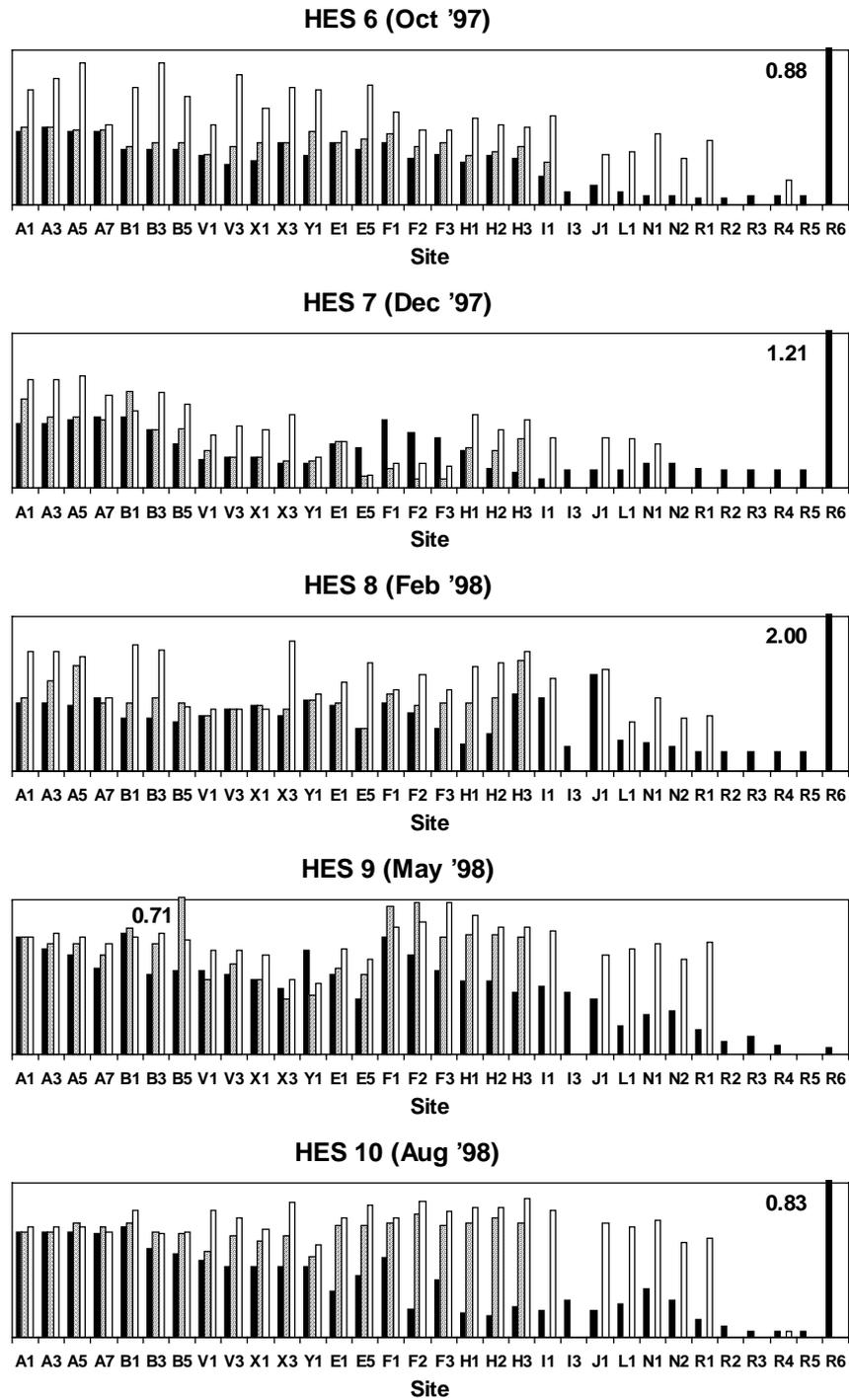


Fig. 4.15 (cont.) Variation of phosphate concentration with site and depth during estuarine surveys (HES 6–10). Key is surface (solid bar), mid-depth (hatched bar) and bottom (open bar). The site denoted as X3 is X3B from HES 6 onward.

The phosphate concentrations in freshwaters are low ($< 0.2 \mu\text{M}$), and upstream of most influences of agriculture and human habitation on the Huon River, even lower ($\leq 0.09 \mu\text{M}$ at R5). The exception is the lower Kermadie River at R6. Owing to land practices, a township and effluent from an STP, it was considerably higher, with a median concentration of $0.88 \mu\text{M}$.

In wintertime *surface* waters, the estuarine relation of phosphate with salinity seemed conservative. Marine concentrations varied from just under $0.3 \mu\text{M}$ to slightly above $0.5 \mu\text{M}$ for the three winters (1996–1998) that our study covered. This suggested some measure of interannual variability.

Almost without exception, phosphate concentrations in at least some bottom waters of the middle or lower estuary exceeded those in seawaters along the marine boundary (A transect). Similar observations in other estuaries have been put down to efflux of phosphate from reducing sediments (Maher and DeVries 1994).

On occasions, phosphate appears to be able to escape a thin capping layer of oxidised sediments. These effluxes are not always associated with other indicators of sedimentary input (e.g. ammonia, depleted DO in bottom waters). Phosphate bound to iron oxyhydroxides is released when iron (III) is reduced to iron (II). It is thought that the iron is trapped as sulfides in sulfur-rich marine sediments, and the phosphate may diffuse into bottom waters (Caraco et al. 1993, Butler et al. — *in prep.*). The sites at which increased concentrations of phosphate were observed did not appear to relate to their proximity to aquaculture leases.

Dissolved organic phosphorus (and condensed phosphates)

The estimation of dissolved organic phosphorus (DOP, including an unknown fraction of condensed phosphates) was more uncertain than most nutrient determinations discussed here. Phosphate made up the bulk of total dissolved phosphorus in the Huon Estuary, and so DOP was often a small, imprecise difference. This should be borne in mind when considering our results.

The concentration of DOP was uniform and mostly near zero (typical range: $< 0.1\text{--}0.2 \mu\text{M}$) throughout the estuary during our study. A seasonal trend could not be resolved unequivocally. The exception was the summer of 1997/98 (HES 7 & 8), when DOP concentration was slightly higher (typical range: $< 0.1\text{--}0.4 \mu\text{M}$) over the estuarine range, but again with no particular trend with salinity.

Freshwater concentrations of DOP appeared higher on occasions than those in estuarine waters (range: $< 0.1\text{--}0.4 \mu\text{M}$). The concentrations in the lower Kermadie River (R6) were higher still, with a range of $0.1\text{--}0.9 \mu\text{M}$ (median: $0.5 \mu\text{M}$).

Particulate phosphorus

As with DOP, particulate phosphorus concentrations are variable as a result of both analytical imprecision and environmental variability. Particulate P concentrations, too, are low (typical range: $< 0.1\text{--}0.3 \mu\text{M}$) and uniform over the estuarine gradient. Scattered high concentrations were observed during the summer of 1997/98 and into the late autumn (Fig. 4.16). These outliers were presumably associated with the dense microalgal blooms at this time. The surface water at site Y1 during HES 9 (May '98) had a particulate P concentration of $2.0 \mu\text{M}$.

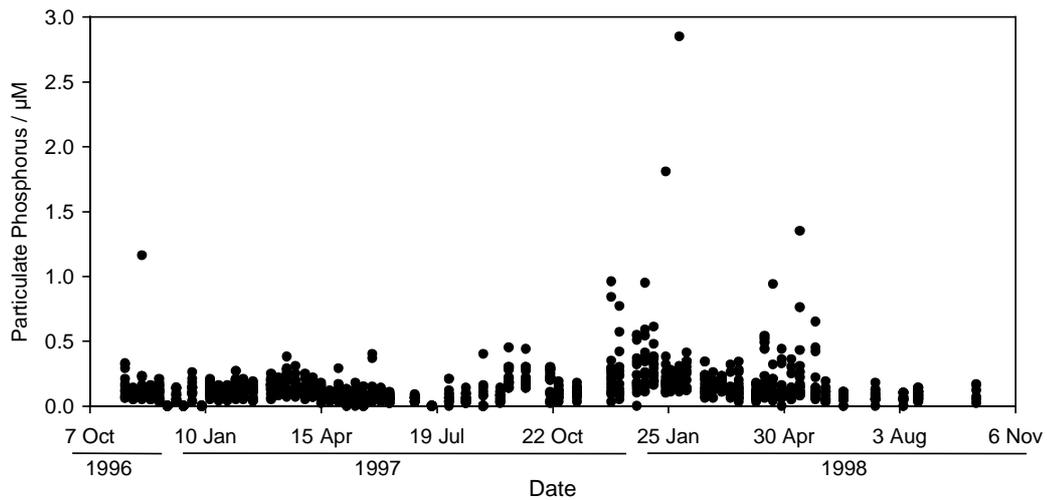


Fig. 4.16 Variation of particulate phosphorus concentrations at the five biological sites (3 depths each) with time, from weekly / fortnightly monitoring.

Particulate P concentrations in the freshwaters of the Huon River were very similar to those of the estuary. However, the lower Kermandie River (R6) was discernibly higher (range: 0.4–1.5 µM; median: 0.7 µM).

Total phosphorus

Total phosphorus (TP) displayed much the same behaviour as phosphate: concentration increasing with salinity down the estuary (Fig. 4.17). The result is not surprising, because phosphate was a major portion of the total at all salinities. However, TP also differed from phosphate in a number of important ways. It did not show obvious signs of removal during mixing, at least for mid- to high salinities. The corollary is that removal of phosphate resulted in production of particulate P and DOP (probably more of the former). This observation was especially true for the microalgal blooms over the summer and autumn of 1997/98.

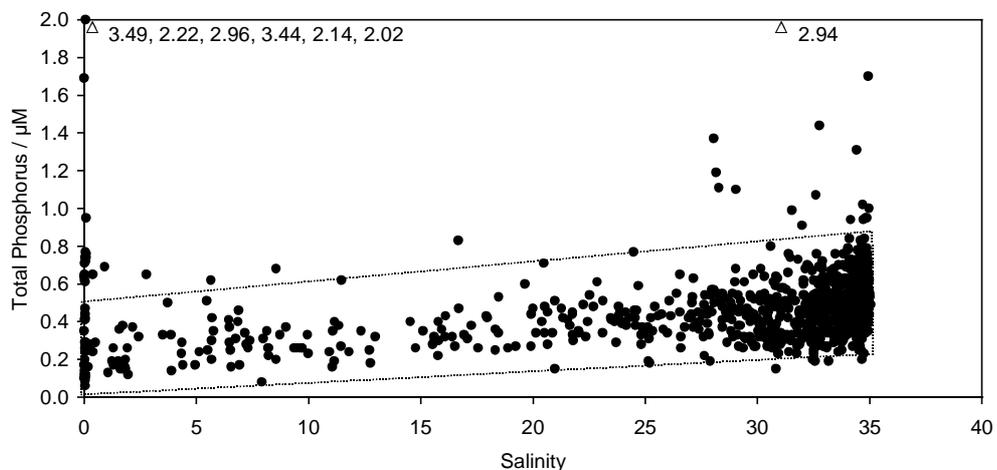


Fig. 4.17 The relation of total phosphorus to salinity for all HES data (spatial surveys and weekly / fortnightly monitoring). Several abnormally high concentrations in freshwaters (site R6) and in the estuary are outside the bounds of this graph (see Tables 4.2 and 4.3).

In the freshwaters, TP was quite variable from survey to survey, but we could find no correlation with streamflow in the Huon River. Its concentration frequently declined at low salinities (0-10). Removal by precipitation and flocculation with other solutes is conceivable. Iron or manganese oxyhydroxides in combination with humic substances are reported to remove phosphorus (e.g. Smith and Longmore 1980, Church 1984). However, it would appear to be DOP, rather than phosphate, which was removed in the low-salinity waters. This 'DOP' fraction in river waters could be colloidal forms of non-molybdate-reactive P (see 4.3.1 above). These forms of phosphorus — either genuinely organic or associated with Fe oxyhydroxides — would subsequently aggregate at low salinity, under the influence of increasing ionic strength.

Sporadic high concentrations of TP in the Huon Estuary are thought to be a combination of STP input and phosphorus within microalgal cells. Some examples were the surface sample at L1 near the township of Franklin (HES 4; Feb '97 – 0.68 μM); and the surface samples at Y1 (head of Port Cygnet) and mid-depth at B5 (off Gardners Bay) (HES 9, May '98 – 2.9 and 1.4 μM , respectively). Site R6 on the Kermantie River below the Geeveston STP is affected solely by effluent. The range of TP concentrations here during the estuarine surveys was 0.7– 3.5 μM (median: 2.1 μM)

4.3.7 Estuarine distribution and behaviour of silicate

Silicate had the expected inverse relation with salinity. Concentrations in the bottom waters of the A transect (approximating the marine end-member) were low, ranging between 0.8 and 3.8 μM . At the freshwater end, two sources were important — the Huon River and the Kermantie River. The Kermantie River had higher concentrations of silicate (range: 100–308 μM ; median: 239 μM) than the Huon River (range: 50–101 μM ; median: 72 μM). Their influence upon the relation of silicate with salinity is seen in Figure 4.18. With the Huon River dominant, the mixing curve was close to conservative. The slight addition of silicate close to the river end (S, 0–10) could result from dissolution of biogenic silicate (Eyre and Balls 1999), or from inputs of the Mountain River. However, the distinct change represented by HES surveys 2, 3 and 7 (Jul & Oct '96, Dec '97) in Figure 4.18 was caused by the Kermantie River. They are "broken-stick" plots — composed of two line segments of different gradient. The salinity at their point of intersection coincided with Port Huon, and the location of the Kermantie River plume. The effect was observed when rainfall exceeded 10 mm at Geeveston on the two days before we sampled the middle estuary.

The levels of silicate in surface and mid-depth waters of the lower estuary in summer are generally < 10 μM , and in places < 3 μM . It might not be the absolute concentrations of silicate alone that constrain the competitiveness of diatoms, but the N:Si and P:Si ratios that might favour dinoflagellates and other microalgae later in the growing season (see Chapter 5).

4.3.8 Nitrate-phosphorus relation and nutrient limitation

The nitrate:phosphate ratio in marine waters has been used as an indication of nutrient limitation when it departs sharply from the Redfield ratio of 16:1 (Redfield et al. 1963). Redfield made his observations on open-ocean systems. In coastal waters, a range of effects

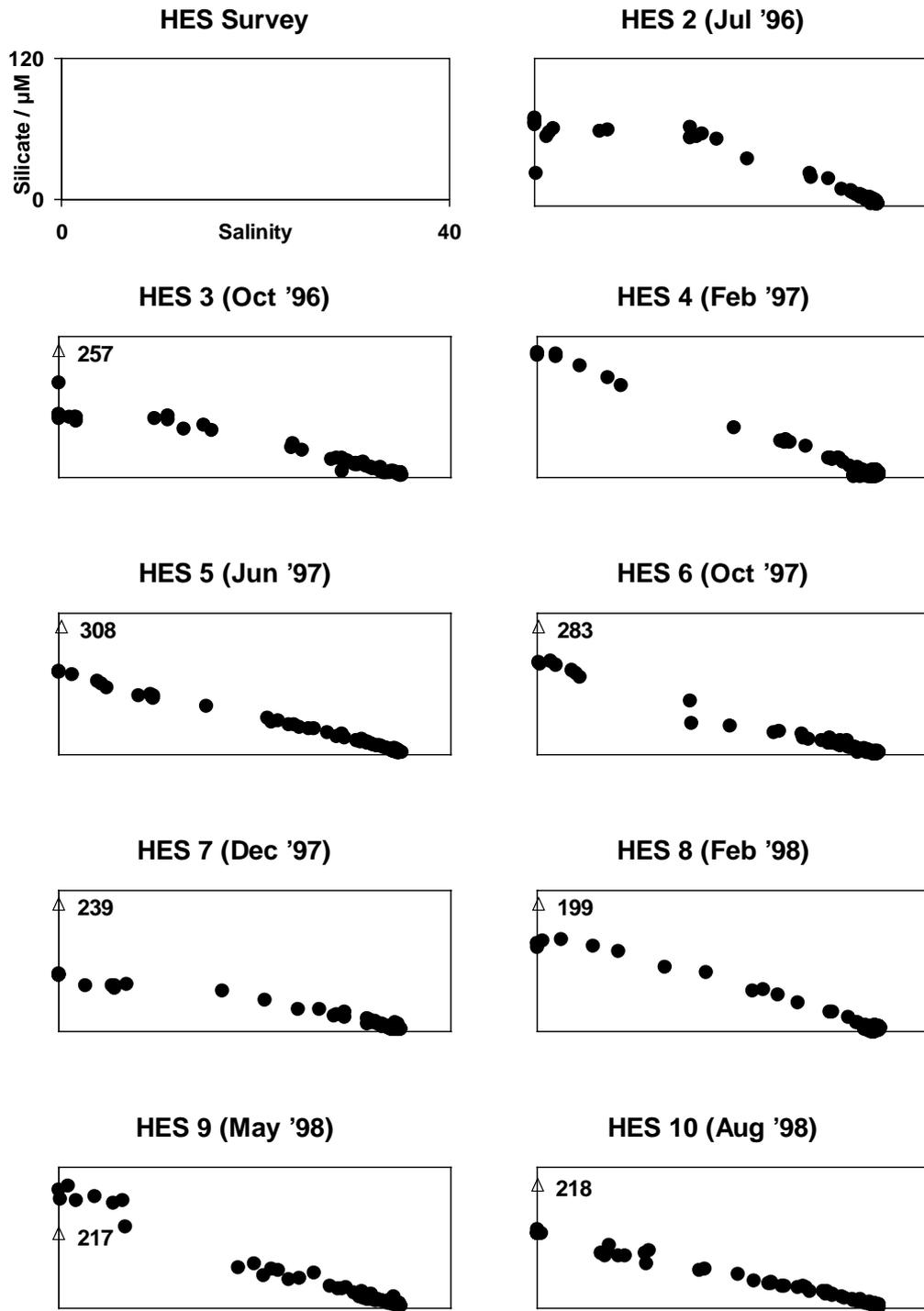


Fig. 4.18 The relation of silicate (dissolved reactive silicon) to salinity for individual surveys, HES 2 to 10. The scale for all graphs is shown in the top left panel. Indicated off-scale values on y-axis are at site R6.

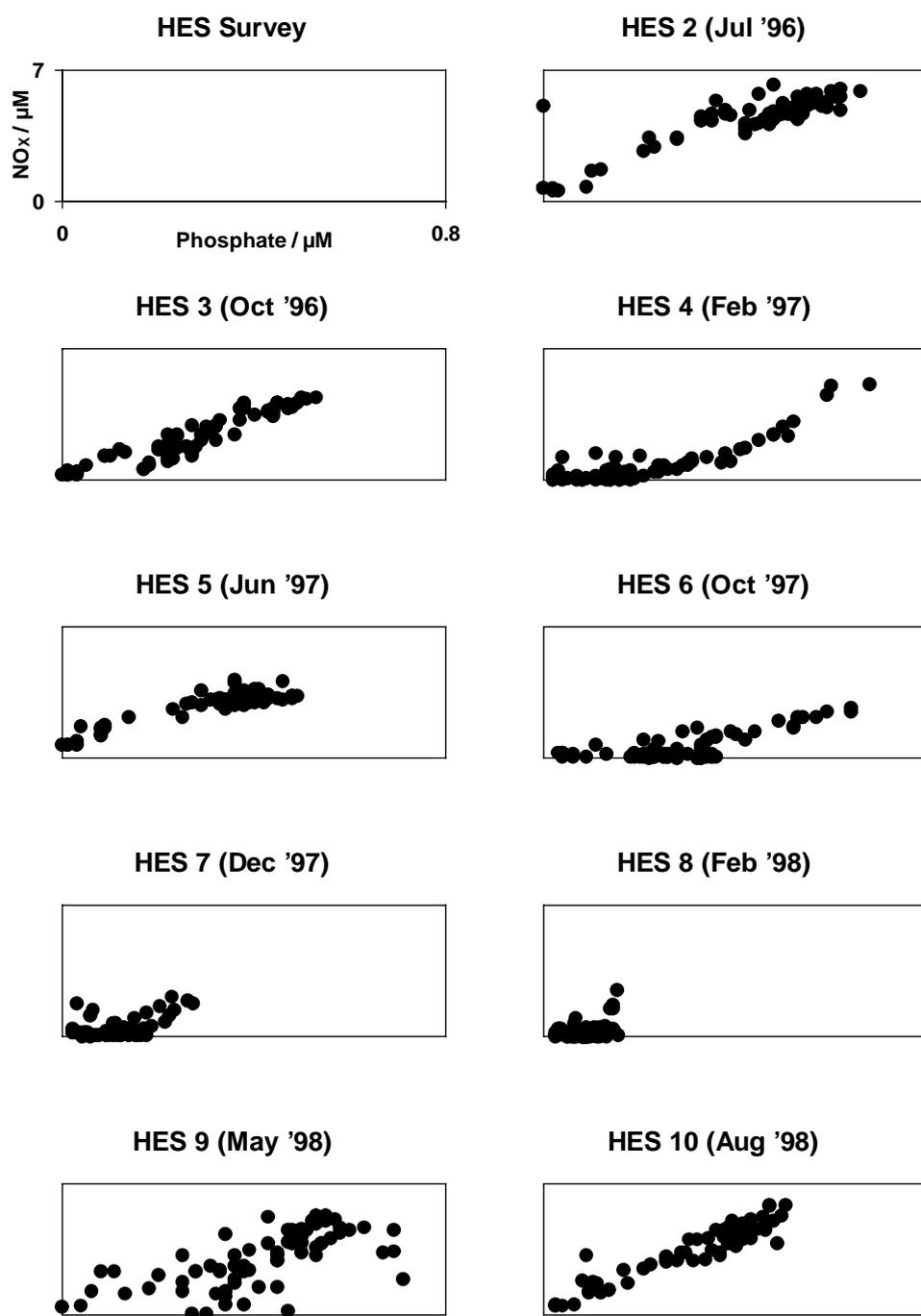


Fig. 4.19 Variation of nitrate + nitrite (NO_x) against phosphate for surveys, HES 2 to 10. The scale for all graphs is shown in the top left panel.

cause departures from the classic assimilation and remineralisation pathway for nutrients seen in pelagic waters. These include run-off from agricultural land, effluent from urban centres (treated sewage, stormwater, etc.) and discharges from industrial activities (via effluents or aerosols). Biological pathways for nutrients in coastal waters are also different and more complicated. Exchange with sediments, macrophytes, wetlands and other coastal features all leave their imprint. Nevertheless, gross changes in the nitrate:phosphate ratio can still be useful in assessing pressures on microalgal production in coastal waters.

Plots of nitrate + nitrite vs. phosphate for the spatial surveys (Fig. 4.19) showed seasonal changes, and differences between years. In the winters, a reasonably tight relation was observed, with a slope (N:P ratio) of 10 or slightly under. The slope declined in spring as biological activity increased — more sharply in the spring of 1997 (N:P \leq 5) than in that of the preceding year. In the summer of 1996/97 (HES 4), the slope was significantly depressed: nitrogen was depleted relative to phosphorus. In fact, the relation curved downward, the nitrogen depletion coinciding with microalgal blooms in the middle estuary (e.g. site L1) during that survey. The following summer (1997/98 — HES 7 and 8) was marked by strong removal of nitrogen, and to a lesser degree phosphorus, with N:P falling below 3 at most estuarine sites. This suggested nitrogen limitation of microalgal production. The nitrate + nitrite vs. phosphate plot in late autumn 1998 (HES 9) was very scattered, reflecting an incomplete and disconcerted replenishment of nitrogen via remineralisation. A tighter relation and N:P approaching 10 at the change of season (winter to spring, 1998 – HES 10) suggests that the remineralisation was complete. The same features were evident as a more continual record from the weekly / fortnightly monitoring at the five key estuarine stations (Fig. 4.20). Also clear from this figure is the duration and extent of nitrate depletion in the 1997/98 cycle relative to that of 1996/97.

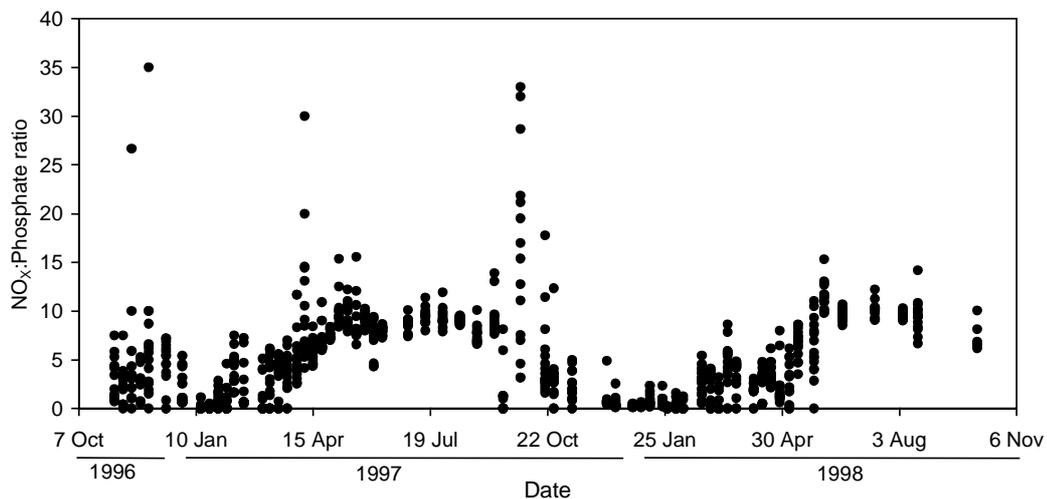


Fig. 4.20 Variation of NO_x / phosphate ratios at the five biological sites (3 depths each) with time, from weekly / fortnightly monitoring.

Another interesting ratio is that of nitrite:nitrate in bottom waters of the middle and lower estuary (Fig. 4.21). In the period of maximum depletion of dissolved inorganic nitrogen (summer 1997/98), this ratio began to increase in the middle estuary and Port Cygnet in

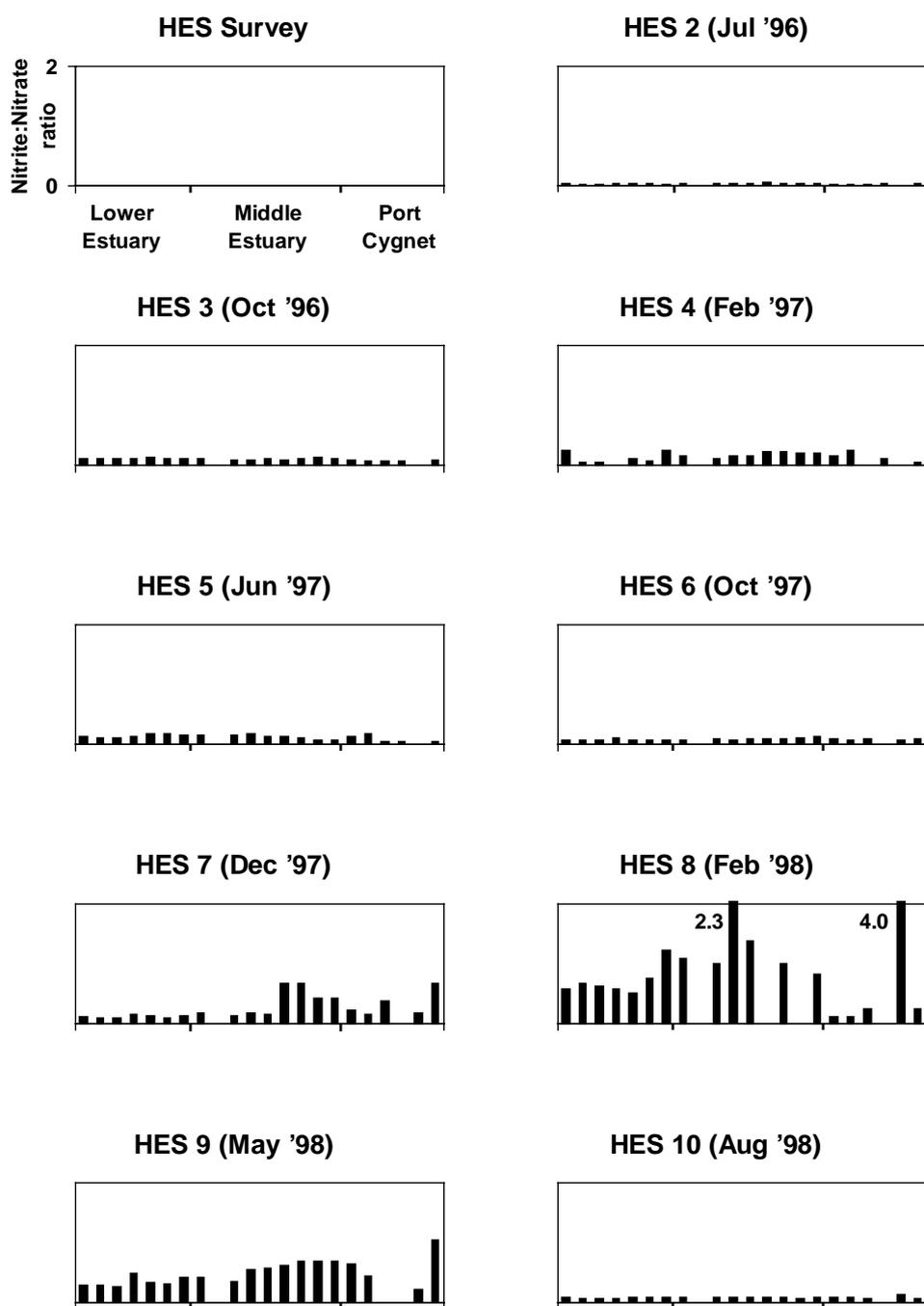


Fig. 4.21 Variation of nitrite / nitrate ratio of bottom waters with site for surveys, HES 2 to 10. The scale for all graphs is shown in the top left panel.

early summer (HES 7). It was strongly elevated in late summer (HES 8), and was still high in autumn (HES 9). For each of these observations, the nitrite:nitrate ratio increased away from the marine end of the estuary. It seemed that the ratio was increasing with the upstream flow (as part of estuarine circulation) of these bottom waters. We surmise that the increased nitrite:nitrate ratio was the result of the recycling of nitrogen in the surface sediments or adjacent bottom waters, at times of maximum depletion of dissolved inorganic nitrogen. This process was discussed above under the headings for nitrite and ammonia. Nitrate was observed in the bottom waters at all times of the year (see Fig. 4.8). An input from D'Entrecasteaux Channel (probably via the canyon to the south — see Subsection 2.2.3, Bathymetry) is an expected source, but in summers like that of 1997/98, it is conceivable that some nitrate is also supplied from within the estuary by recycling (discussed further in Chapter 10).

4.3.9 Case study: nutrients and water quality in the Kermandie River and its tributaries

A repeated comment in the preceding subsections was that water quality (and nutrients as a part) at site R6 on the lower Kermandie River was quite different to freshwaters of the Huon River; it was invariably poorer.

As part of HES 10 (27 August 1998), we did a limited survey of the subcatchment streams to seek further information on the water quality of the Kermandie River and its tributaries. The map of sampling sites (Fig. 4.22) also depicts subcatchments, possible point sources and mean annual rainfall.

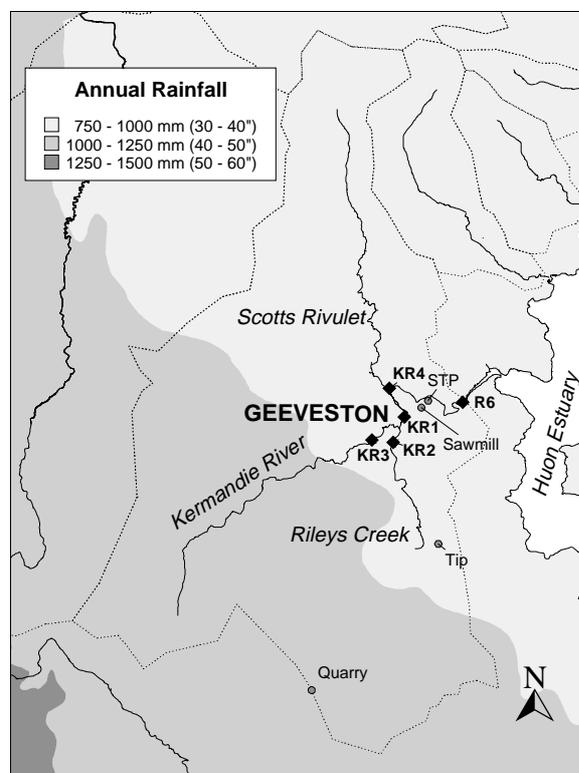


Fig. 4.22 Map of the Kermandie River Basin showing sampling sites on the tributaries and the main river (27 August 1998). Also depicted are licensed activities in the region and rainfall isohyets (modified from Gallagher 1996).

Results are presented in Table 4.3. The concentrations at R6 were not unusual when compared with data from all other surveys at this site. Analyte concentrations were close to the medians, although the ammonia concentration was the highest recorded. Our stream data is also consistent with Bobbi's (1998) catchment surveys of January and June 1997.

Table 4.3 Concentrations of nutrients and other water-quality parameters in the Kermantie River and its tributaries (27 August 1998) compared with data from lower Kermantie River site R6 from all surveys.

WATER-QUALITY MEASUREMENTS (Concentrations, μM)	KR1 ^a	KR2 ^a	KR3 ^a	KR4 ^a	R6 ^a	R6 ^b	R6 ^b
	HES10	HES10	HES10	HES10	HES10	range	median
Nitrate + Nitrite	14.2	16.9	9.6	18.4	12.0	6.6–14.2	11.8
Nitrite	0.07	0.10	0.06	0.12	0.52	0.14–1.56	0.64
Ammonia	0.26	0.74	0.26	0.95	11.06	2.16–11.06	9.75
Total Dissolved N	16.0	40.8	9.5	—	28.8	22.9–43.9	30.8
Total Nitrogen	18.3	39.3	11.1	37.3	31.3	23.6–57.2	36.7
Phosphate	0.08	0.05	0.03	0.12	0.83	0.03–2.50	0.88
Total Dissolved P	0.42	0.45	0.34	0.42	1.32	0.35–2.72	1.32
Total Phosphorus	0.61	0.76	0.42	0.95	2.00	0.72–3.49	2.14
Silicate	197.9	201.2	201.2	301.6	218.4	99.5–308.3	239.4
Dissolved Oxygen ^c	—	—	—	—	—	330.52	330.52
SPM (mg/kg)	5.4	10.9	2.6	8.0	10.8	6.8–39.3	9.7

a KR1: Kermantie River at Geeveston, KR2: Rileys Creek above Geeveston, KR3: Upper Kermantie River above Geeveston, KR4: Scotts Rivulet at Geeveston, R6: Lower Kermantie River below Geeveston and STP

b Range and median determined for results at R6 from HES surveys HES 3–10A (Oct. 1996–Sep. 1998)

c DO determined for HES 3 (Oct. 1996) sampling only

It is clear that at R6 water-quality parameters (ammonia, nitrite, phosphate and probably all other forms of phosphorus) were significantly different to all other KR sites. The STP was almost certainly the source, given the nature of these elevated nutrient species, and because R6 is just downstream of its discharge point.

Identifying other sources of nutrients in the Kermantie catchment, or causes for degradation of water quality, is not so straightforward. Nutrients (N, P and Si) and SPM concentrations were high in all streams; it was a matter of degree. Rileys Creek and Scotts Rivulet were of poorer quality than the Kermantie River at Geeveston and above. They were visually more turbid at the time of sampling (reflected in the higher SPM concentrations). The particulate matter very probably contributes to the higher concentrations of nutrients in these two streams. The degraded water quality in the catchment would therefore arise from diffuse sources, attributable to land practices (e.g. agriculture and forestry) and human settlement (road building, septic tanks, sillage and stormwater), and more subtly, to differences in geology and soils. The slightly better quality of the Upper Kermantie River probably reflects that much of its catchment lies in State Forest.

Silicates appear to derive from soil particles washed into streams. Some support for this statement arises from the catchment water-quality report by Bobbi (1998). He observes: “run-off from the gravel road which closely follows Scotts Rivulet down the valley appears to cause high turbidity in this stream and the silt load has visibly affected the stream bed”. Scotts Rivulet had 50% more silicate than the other Kermandie tributaries, which were themselves high (~200 μM).

4.3.10 Nutrients in groundwater and rainwater — a brief examination

As shown in Figure 4.1, groundwater and rainwater can be sources of nutrients to the Huon Estuary. We are not referring to the contribution via streams, which has already been covered above, but rather direct input to the estuary: subterranean discharge of groundwater into the estuary proper, and rain falling onto the surface of the water body. In both these cases, the quality of the source water becomes an issue, although not necessarily a significant ingredient, in estimating a nutrient budget for the system (see Chapter 10).

An exploratory survey was made of two groundwater samples from bores in the Huon region — one at the head of the estuary at Huonville, and another on the western shore of the lower estuary at Waterloo. We were unable to gain a third sample from the Port Cygnet subcatchment. The groundwater analyses are reported in Table 4.4.

Table 4.4 Concentrations of nutrients and other water quality parameters in groundwater of the Huon Catchment taken from two bores (2 April 1999).

LOCATION	Huonville ^a		Waterloo ^b	
	unfiltered	filtered	unfiltered	filtered
Salinity	0.467		0.527	
Conductivity, μmho	430		475	
Nitrate+Nitrite, μM	0.79	0.80	0.45	0.42
Nitrite, μM	0.02	0.02	0.05	< 0.02
Total Nitrogen, μM	7.5	7.8	11.1	9.8
Phosphate, μM	0.38	0.14	0.48	0.23
Total Phosphorus, μM	1.46	0.17	1.87	0.32
Silicate, μM		1200		505

a Bore near packing shed on Hansen Bros. orchard, 100 m to east of Huonville

b Bore behind house on Calvert Bros. orchard

Compared with freshwater in the Huon River, they had slightly higher conductivity and very similar concentrations of nitrogen species. However, their concentrations of phosphorus species were from two to ten times greater, and silicate at least an order of magnitude greater, than in the Huon River. The phosphorus concentrations were comparable to those observed in the lower Kermandie River (site R6 — see Table 4.3), although the bore-water samples contained mostly particulate phosphorus. Silicate concentrations exceeded even those recorded for the streams of the Kermandie subcatchment by a factor of two to four.

We did not analyse rainwater of the Huon region. Any cursory investigation suggests that Tasmania, as an island in the Southern Ocean with no landmass upstream for the dominant wind directions or for most of the wind rose, will have wet (and dry) atmospheric deposition with an almost exclusively marine signature. Measurements at the Baseline Atmospheric Program Station at Cape Grim, on the northwest tip of Tasmania, indicate typical concentrations of ammonium and nitrate ions in rainwater are about 4 μM and 5 μM , respectively (Ivey et al. 1996). These values may be high — arising from contamination during sample collection (G.P. Ayers — *pers. comm.*, November 1999). No comparable data are available on phosphorus or silicon in rainwater.

4.4 Summary and Conclusions

The Huon Estuary is fortunate that its two main inputs — the Huon River and the coastal seawaters from D'Entrecasteaux Channel and beyond — are of a high environmental quality. This natural advantage is enhanced by the small regional population, a lack of heavy industry on its shores, and a prevalence of human activities in the catchment that do not seriously degrade tributary streams.

The estuary might be broadly classified by Church's (1986) scheme as a "biochemical reactor" (see Subsection 4.1.1). In fact, it is close to a pure example of such an estuary; geochemical processes in the lower salinity zone are exceedingly limited by both the rapid flushing of the upper estuary and the very low concentrations of suspended particulate matter. In contrast, primary production in the lower estuary affects nutrient concentrations appreciably, and this influence would doubtless extend to other trace solutes.

Since there was no reference system or comprehensive set of water quality measurements for the Huon Estuary — the first step in the decision sequence of the draft ANZECC / ARMCANZ (1998) guidelines — we opted to use their interim trigger levels instead. Several chemical forms of the nutrients nitrogen and phosphorus exceeded these trigger levels for entirely natural reasons. The concentration of nitrate and phosphate in marine end-member waters is one example. Sub-Antarctic waters encroaching on coastal waters around southern Tasmania are much richer in nutrients than subtropical waters from which the marine trigger levels were derived. Regional differences were expected in drawing up the new guidelines (ANZECC / ARMCANZ 1998). They should be updated as soon as practicable, with at least a reference set of marine concentrations for sub-Antarctic waters.

From our assessment of the water quality in the Huon Estuary, it appeared to be generally a well-oxygenated system (>80% saturation). We found a pool of low-oxygen water (down to 40% saturation) in the bottom marine layer of the upper estuary, influenced by anaerobic sediments at the top end of the Egg Islands. During the warmer months, transient depletion of oxygen in bottom waters of the middle and lower estuary was sometimes very substantial (down to 3.6% saturation) near marine farms. Such occurrences are obvious hazards for finfish-farm operation, but we are not certain how widespread oxygen depletion is in the marine zone of the estuary. The broader ramifications for the estuary also need to be resolved.

Concentrations of suspended particulate matter were low throughout the estuary — usually < 6 mg kg⁻¹. This situation is favoured by its being a microtidal system and a drowned river valley with large tracts of its native catchment intact. Slightly higher SPM concentrations were

associated with perhaps natural processes around Egg Islands; discernibly higher levels result from human activities (e.g. Kermandie subcatchment).

Dissolved organic nitrogen was the main form of nitrogen. It verged on conservative behaviour, with higher concentrations at the river end deriving from terrestrial run-off. Particulate nitrogen was a small and fairly uniform fraction of nitrogen across the estuarine gradient. Therefore, total nitrogen had the characteristics of DON (inverse with salinity), but the mixing line was displaced up or down depending upon the seasonal influence of dissolved inorganic nitrogen. Sporadic high concentrations of particulate nitrogen coincided with high-density microalgal blooms.

Dissolved inorganic nitrogen was usually dominated by nitrate, especially in winter when it was close to conservative mixing (increasing concentration with salinity). However, during spring, summer and early autumn — particularly in 1997/98 — it was strongly removed from estuarine waters by biological activity. A residual store of nitrate was present throughout the year in bottom waters at the marine end of the Huon Estuary. Deeper marine waters entering via southern D'Entrecasteaux Channel were the likely source, but small concentrations might arise from internal recycling of nitrogen during periods of high biological production. Nitrite behaved similarly to nitrate, but was at 10% or less of nitrate's concentration, apart from late autumn 1998, when it was over a third. Remineralisation of detritus from the dense microalgal blooms of earlier that year is a credible explanation for this observation. Ammonia was also at low, but patchy, concentrations ($< 1.0 \mu\text{M}$) in the estuary. Higher levels were observed at times just above the sediments, near to finfish farms and downstream of sewage-treatment plants.

Phosphorus, mostly in the form of phosphate, increased from very low levels in freshwaters to higher concentrations in marine waters. It was also influenced by seasonal uptake by microalgae; however, it was not removed as rapidly or as extensively as nitrate. Scattered elevation of phosphate in bottom waters may have arisen from sedimentary input. Higher surface concentrations of phosphate near towns were possibly the only lingering signs of sewage input, because nitrogen species were rapidly stripped out of estuarine waters by microalgal uptake.

Dissolved organic phosphorus and particulate phosphorus were both generally at very small concentrations and were invariant with salinity. Particulate phosphorus, like particulate nitrogen, was occasionally measured at higher concentrations as a result of intense microalgal blooms. While total phosphorus (mostly as DOP) was on occasions seen at slightly higher concentrations in freshwaters, the excess was removed at low salinities (0–10), possibly by adsorption and flocculation processes.

Seasonal variation in N:P ratios suggested nitrogen limitation ($< 5:1$) in the summer months. It was even more pronounced ($< 3:1$) during the dense microalgal blooms of summer 1997/98. Definitive proof of nitrogen limitation can only come from culturing experiments in the laboratory, or from bioassay techniques (such as nutrient-induced fluorescence transients in microalgae — Wood and Oliver 1995).

Silicate in the estuary was dominated by terrigenous sources. Much higher concentrations were found in the Kermandie River than in the Huon River. When the former's discharge was sufficient, it added silicate to the middle estuary, and disrupted the more usual conservative relation with salinity. During summer in the lower Huon Estuary, silicate concentrations

of $\leq 3 \mu\text{M}$, or their relative proportion to nitrogen and phosphorus, were possibly limiting to diatoms.

Water quality in the Kermandie River and its tributaries was decidedly inferior to that of the Huon River. It was made worse by effluent from an STP in its lower reaches. As recorded by Bobbi (1998), the condition of the Kermandie River is shared by a small number of estuarine tributaries. For example, the main freshwater discharges to Port Cygnet — Agnes and Nicholls Rivulets — are similarly degraded. Fortunately, the streams with poor water quality have insignificant flows compared with the Huon River, although they may have localised negative effects in neighbouring embayments and inlets.

Two examples of groundwater from the Huon region had low conductivities and nitrogen species concentrations similar to the Huon River. Concentrations of total phosphorus were on par with those in the lower Kermandie River, but silicate was two to four times greater than the levels seen in any surface waters.

Aside from the aforementioned question of oxygen-deficient conditions in bottom waters of the estuary, other issues concerning nutrients and water quality remain to be resolved. These include the nature and reactivity of dissolved organic nitrogen. It would be valuable to determine the fraction of DON that is readily assimilable by microalgae — compounds such as urea, amines and amino acids. In addition, if a test were developed to measure the labile DON as against the refractory DON, it would make a useful measure of a nitrogen fraction that might be available to microalgae. Of special interest, too, are techniques that would enable tracing sources of nitrogen and phosphorus (e.g. discerning urea excreted by fish from that introduced by run-off). Stable isotope signatures are a possibility here.

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5 PHYTOPLANKTON DYNAMICS

5.1 Introduction

Phytoplankton are major primary producers in aquatic ecosystems. These microscopic plants, or microalgae, grow in response to both the sun's energy and the interplay of physical and chemical parameters to provide the basis of biological productivity in these systems.

Estuaries have unique characteristics compared with other aquatic systems because they are an interface for catchment/riverine influences and coastal/marine influences. Mallin et al. (1991) listed factors affecting biological productivity in estuaries: latitude, season, irradiance, temperature, flow, nutrient loading and recycling, grazing, and watershed geomorphology and development. Many of these factors interact to create a dynamic mosaic of physical, chemical and biological characteristics, with the phytoplankton dynamics being a highly responsive indicator of the net effect of these characteristics of the estuary. It is not uncommon for one factor to be the limiting factor for phytoplankton growth; for example, nitrogen is often considered the main regulator of estuarine phytoplankton production (Boynton et al. 1982). However, other nutrients and light are also regulators (Pennock and Sharp 1994 and references therein).

While the phytoplankton biomass and production are important in themselves, the composition of the phytoplankton also defines the character of the estuarine dynamics. The presence and diversity of major microalgal classes such as diatoms (Bacillariophyceae), dinoflagellates (Dinophyceae) and other flagellates represent the interplay of the estuary's characteristics, as does the presence of certain species, especially those that form mono-specific algal blooms. Particular microalgae will respond to specific conditions; for example diatoms grow well in silicate-rich waters, whereas dinoflagellates grow more slowly, but in a stratified or otherwise stable water column, dinoflagellates may dominate because they can vertically migrate and may be stimulated by factors in river run-off (Paerl et al. 1990, Doblin et al. 1999a). Phytoplankton composition can change rapidly depending on single or multiple factors. As some phytoplankton can double their numbers in less than a day while others take several days, the ecological picture of an estuary or parts of an estuary can change dramatically on relatively short time-scales. The flushing rate of the estuary also plays a role. Seasonal trends give a "big picture" that contains, but often masks, these rapid and potentially important changes on short time-scales.

The phytoplankton dynamics are essential for the natural biological activity of the estuary and also critical for our use of estuaries for aquaculture. Phytoplankton are the essential food supply for the shellfish and other filter-feeding aquaculture animals. For finfish, the relationship is more indirect, although microalgae (packaged in zooplankton) provide the nutrients for fish larvae. Farmed adult fish are fed nutrient-rich pelleted feeds. These feeds can also increase the nutrients that encourage natural phytoplankton growth in the estuary (Folke et al. 1994) and modify the phytoplankton species composition (Riegman 1995, Arzul et al. 1996). There are also some direct deleterious effects of phytoplankton on adult fish, such as anoxia and gill irritation (MacKenzie 1991, Albright et al. 1993, Koizumi et al. 1996).

An algal bloom is a proliferation of phytoplankton. A seasonal cycle of phytoplankton dynamics including algal blooms (usually in spring, but sometimes at other times of the year) is a natural characteristic of aquatic systems. However, under some circumstances, such as changes in physical conditions and increased eutrophication, dominant mono-specific blooms can develop (Paerl 1988, Mallin et al. 1993). These can be problematic because of their high biomass, which can deplete oxygen in the water column as the bloom decays. The problem of algal blooms is exacerbated when the bloom-forming species produces harmful substances such as toxins. In particular, some of the dinoflagellates produce paralytic shellfish toxins (PSTs), neurotoxins which are accumulated by shellfish and can cause illness and death in humans (Yentsch 1984).

5.2 Background to the Phytoplankton Dynamics in the Huon Estuary

Previous research and monitoring have demonstrated that the Huon Estuary is somewhat unusual with respect to phytoplankton dynamics. Blooms of various dinoflagellates are interspersed with diatom blooms and they often co-exist (Jameson and Hallegraeff 1994). Since 1986, irregular blooms of the PST-producing dinoflagellate *Gymnodinium catenatum* have been recorded in the Huon Estuary (Hallegraeff and Sumner 1986, Hallegraeff et al. 1995). As discussed in Chapter 2, the Huon Estuary catchment has a high proportion of wilderness rainforest, and tannin-rich, river run-off which possibly stimulates algal blooms, and particularly the toxic blooms (Hallegraeff et al. 1995, Doblin et al. 1999a). Blooms of *G. catenatum* have caused closures of shellfish farms in the Huon Estuary for periods of up to nine months. Less problematic blooms are also common (Jameson and Hallegraeff 1994, Tassal Ltd., and CSIRO unpublished data). However, there is little detailed information on the nature of the phytoplankton dynamics, the causes of the toxic blooms, and the impact on, or of, the aquaculture industry on these blooms (Woodward et al. 1992).

5.3 Huon Estuary Phytoplankton Dynamics 1996-98: Approach to the Study

The Huon Estuary Study made a comprehensive investigation of the abundance and composition of the phytoplankton in the estuary. Two complementary approaches to measuring phytoplankton abundance and composition were used. The first involved detailed microscopic examination of phytoplankton samples, with identification to both class and species level. Particular attention was paid to bloom-forming species. The composition was quantified as cell density (cells L⁻¹) and also, for the bloom species, as biomass based on estimates of cell volume. The second approach measured phytoplankton pigments (the light harvesting compounds of microalgae) with High Performance Liquid Chromatography (HPLC). Some pigments are very specific to a microalgal class so can be used as a biochemical “signature” of the phytoplankton composition. This technique also discriminates between the classes of small flagellates that cannot be identified by conventional light microscopy. As discussed above, strong spatial gradients and temporal fluctuations in phytoplankton abundance and composition are to be expected in estuaries, so the phytoplankton sampling strategy used in the study was designed

accordingly (see Chapter 2). Methods of both microscopic and HPLC examination of samples are described in detail in the supplements.

Spatial surveys provided near-synoptic pictures of the phytoplankton spatial distribution in the estuary at ~3-month intervals over the study period. On spatial surveys, 20 µm mesh phytoplankton net samples were collected at all chemical stations (see Chapter 2, Fig. 2.6), and the species identified. Samples were collected from three depths (surface, mid-depth and bottom) at 30 sites distributed throughout the estuary (chemical stations) for HPLC pigment analysis. Depth-integrated water column samples were collected at five of these sites in the middle to lower estuary (biological stations – see Chapter 2, Fig. 2.6) for microscopic analysis of all phytoplankton, including picoplankton (<2 µm) and nanoplankton (2-5 µm), and for HPLC pigment analysis.

To resolve short-term changes and algal bloom behaviour, intensive monitoring involving weekly or fortnightly sampling was carried out at the five biological stations throughout the study period. The biological stations were:

Hideaway Bay	(B1)
Port Cygnet	(X3b)
Wheatleys	(F3)
Killala Bay	(F1)
Brabazon Park	(H3)

At Killala Bay and Hideaway Bay, automated *in situ* instrumentation recorded profiles of salinity, temperature and fluorescence at 60 minute intervals, providing information on rapid changes in water column structure and phytoplankton density (see supplement for details on automated profiling system).

This intensive sampling program has resulted in a wealth of data. It is not feasible to show it all here; instead, we have extracted the principal spatial and temporal patterns, and illustrated these with examples.

5.4 Spatial Distribution of Phytoplankton Abundance and Composition

5.4.1 Biomass — chlorophyll *a*

We have used chlorophyll *a* (measured by HPLC) as a routine measure of phytoplankton biomass. For statistical analysis, we carried out a \log_{10} transformation, as $\log(\text{chl } a)$ was approximately normally distributed (see section 5.5.1). Further details of statistical procedures are provided in the supplement.

Differences among regions within the estuary

For purposes of statistical analysis, we divided the estuary into four regions: the upper estuary (sites L, N, R1, R2), the middle estuary (transects E to H), the lower estuary (transects A, B) and Port Cygnet (transects V, X and Y). Sites I and J lie between the upper and middle estuary and, because their properties tend to switch between these two regions, we have omitted them from this analysis. We carried out an analysis of variation (ANOVA) among regions for surface and mid-depth chlorophyll values separately.

Table 5.1 shows mean surface chl *a* values (geometric means) by region and survey. One-way ANOVAs showed significant differences in mean log(chl *a*) among regions within all surveys. Table 5.1 also indicates results of pairwise comparisons between specified regions. The mean surface log(chl *a*) in the upper estuary was significantly less than the mean in the middle or lower estuary ($P < 0.05$) in all surveys except HES 4 (Feb '97) and HES 5 (Oct '97). Mean surface log(Chl *a*) in the lower estuary was significantly less ($P < 0.05$) than in the middle estuary in Feb '97 (HES 4), Oct '97 (HES 6), Dec '97 (HES 7) and Feb '98 (HES 8), and significantly greater in Jun '97 (HES 5) and Aug '98 (HES 10). Mean surface log(chl *a*) in Port Cygnet was significantly greater than in the lower estuary in Oct '96 (HES 3), Jun '97 (HES 5) and May '98 (HES 9), but not significantly different on other surveys.

Table 5.1 Mean surface chlorophyll values (mg Chl *a* m⁻³) by region and survey. Values shown are geometric means. Outcomes of tests for equality of means between specified regions are shown.

	Upper	Middle	Lower	Port Cygnet	Upper vs Middle	Lower vs Middle	Lower vs P. Cygnet
HES 2 (Jul '96)	0.09	0.31	0.43	0.54	P<0.05		
HES 3 (Oct '96)	0.08	0.55	0.74	1.35	P<0.05		P<0.05
HES 4 (Feb '97)	0.69	1.11	0.91	0.86		P<0.05	
HES 5 (Jun '97)	0.13	0.13	0.29	0.49		P<0.05	P<0.05
HES 6 (Oct '97)	0.06	1.31	0.67	0.57	P<0.05	P<0.05	
HES 7 (Dec '97)	0.12	0.88	0.49	0.44	P<0.05	P<0.05	
HES 8 (Feb '98)	0.16	1.76	0.57	0.70	P<0.05	P<0.05	
HES 9 (May '98)	0.29	0.56	0.57	12.00	P<0.05		P<0.05
HES 10 (Aug '98)	0.11	0.22	0.39	0.48	P<0.05	P<0.05	

A comparable analysis was carried out for mid-depth chlorophyll values. Mean values by region and survey are shown in Table 5.2. Again, mean mid-depth log(chl *a*) in the upper estuary was significantly less than in the middle estuary on all surveys except during Feb '97 (HES 4). Mean mid-depth log(chl *a*) in the lower estuary was significantly less than in the middle estuary

in Feb '97 (HES 4), Oct '97 (HES 6), Dec '97 (HES 7) and Feb '98 (HES 8), and significantly greater in Oct '96 (HES 3). Mean mid-depth log(chl *a*) in Port Cygnet was significantly greater than in the lower estuary in Aug '98 (HES 10), and less in Feb '98 (HES 8).

Chlorophyll values in the upper estuary are consistently lower than in the middle estuary across most surveys for both surface and mid-depth samples. As we discuss below, phytoplankton communities in the fresh surface layer in the upper estuary are also quite distinct from those in the middle and lower estuary. Chlorophyll in the lower estuary is consistently less than in the middle estuary at both surface and mid-depth in the summer surveys, Feb '97 (HES 4), Oct '97 (HES 6), Dec '97 (HES 7) and Feb '98 (HES 8). However, differences between Port Cygnet and the lower estuary, although statistically significant in a few individual surveys, are not consistent between depths.

Table 5.2 Mean surface chlorophyll values (mg Chl *a* m⁻³) by region and survey. Values shown are geometric means. Outcomes of tests for equality of means between specified regions are shown.

	Upper	Middle	Lower	Port Cygnet	Upper vs Middle	Lower vs Middle	Lower vs P. Cygnet
HES 2 (Jul '96)	0.34	0.57	0.42	0.59	P<0.05		
HES 3 (Oct '96)	0.22	0.46	0.97	1.23	P<0.05	P<0.05	
HES 4 (Feb '97)	1.05	1.30	0.98	0.97		P<0.05	
HES 5 (Jun '97)	0.16	0.26	0.30	0.33	P<0.05		
HES 6 (Oct '97)	0.41	2.81	1.39	1.15	P<0.05	P<0.05	
HES 7 (Dec '97)	0.46	3.37	1.03	0.65	P<0.05	P<0.05	
HES 8 (Feb '98)	0.65	2.88	1.41	0.82	P<0.05	P<0.05	P<0.05
HES 9 (May '98)	0.22	3.91	3.46	8.87	P<0.05		
HES 10 (Aug '98)	0.16	0.73	0.59	0.96	P<0.05		P<0.05

Vertical gradients

For the combined lower and middle estuary there was significantly more chlorophyll *a* at mid-depths than at the surface (P<0.05) throughout the study, except for Oct '96 (HES 3). These differences are small in winter surveys - Jul '96 (HES 2) and Jun '97 (HES 5), but large and highly significant on all other surveys (Fig. 5.1a). Vertical gradients in chlorophyll *a* are not as large in Port Cygnet as in the main estuary (Fig. 5.1b). In Port Cygnet, differences between surface and mid-depth samples are significant (P < 0.05) for Jul '96 (HES 2), Oct '97 (HES 6), Dec '97 (HES 7), Feb '98 (HES 8) and Aug '98 (HES 10), and not significant at other times.

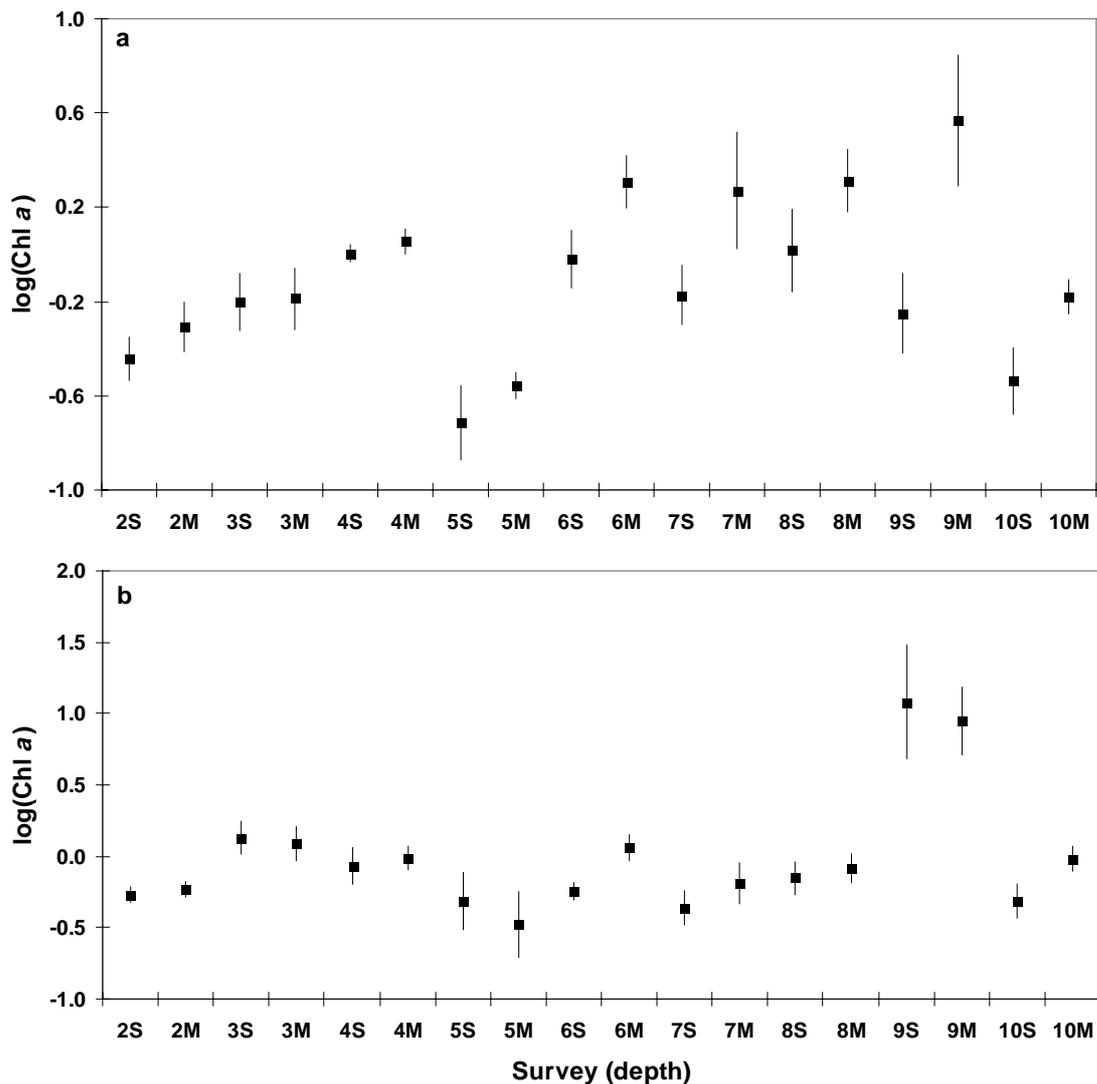


Fig. 5.1 Means and 95% confidence limits for log(Chl *a*) for surface (S) and mid-depth (M) samples for surveys 2 to 10, for (a) combined lower and middle estuary and (b) Port Cygnet.

Changes in chlorophyll a over time among surveys

Tables 5.1 and 5.2 show large changes in mean chlorophyll *a* over time i.e. among surveys. These changes are much larger in amplitude than differences among regions within surveys (at least if we exclude the upper estuary). Pooling all data within surveys in the middle and lower estuary and Port Cygnet, a one-way ANOVA shows highly significant variation among surveys in both surface and mid-depth chlorophyll (surface: $P = 10^{-13}$, mid-depth: $P = 10^{-30}$). Figure 5.1a shows mean (log chl *a*) with 95% confidence intervals (based on t-statistics) for surface and mid-depth samples, for the combined middle and lower estuary. Figure 5.1b shows a comparable plot for Port Cygnet stations. Figure 5.1a shows a consistent pattern of higher chlorophyll in mid-depth samples, which is exaggerated in the high biomass period in the spring, summer and autumn of 1997/98 (HES 6 to 9). By comparison, differences between mid-depth and surface samples in Port Cygnet are small (Fig. 5.1b), and the differences between

early surveys (1996/97) and the later surveys (1997/98) are much less pronounced, with only HES 9 (May '98) standing out as having particularly high biomass.

Horizontal patchiness

As an overall measure of spatial variability, we have computed the variance among samples within surveys, for the lower and middle estuaries combined, and for Port Cygnet. For the lower and middle estuaries combined, variances in surface log(chl *a*) are relatively uniform across surveys, and correspond to a coefficient of variation of chlorophyll *a* of about 55%. However, variances in mid-depth log(chl *a*) vary widely across surveys, and are particularly high in Dec '97 (HES 7) and May '98 (HES 9). This likely reflects the difficulty of sampling chlorophyll *a* concentrated in thin layers during dinoflagellate blooms, possibly exacerbated by diel vertical migration (see later sections). The mean variance in mid-depth log(chl *a*) corresponds to a CV of 63%.

Sample variances in Port Cygnet show significant variation across surveys in both surface and mid-depth samples, with particularly high values in May '98 (HES 9). The mean variance over surveys corresponds to a CV of 43% for surface samples and 35% for mid-depth samples.

Effect of proximity to fish farms

Four of the sites in the spatial survey (B1, F1, F3 and H3) are located close to finfish farms. These sites were also used for weekly monitoring, and two (B1 and F1) as sites for installation of automated profiling instruments. We have used the survey data to test whether these sites showed any significant and consistent differences in chlorophyll concentrations from other sites in the lower and middle estuary. We posed two specific questions:

- are sites B1, F1, F3 and H3 significantly different from all other sites in the lower and middle estuary combined?
- are sites F1, F3 and H3 (all in the middle estuary) significantly different on average from all other sites in the middle estuary.

For surface chlorophyll *a*, we obtained only one significant difference out of 18 tests (comparison of four farm sites with combined middle and lower sites in Feb '98 (HES 8)). This almost certainly occurred because there were strong differences between the lower and middle estuary regions, and three out of the four farm sites are located in the middle estuary. For mid-depth chlorophyll *a*, we also obtained only one significant result, Dec '97 (HES 7), where values at farm sites F1, F3 and H3 were high compared with other sites in the middle estuary ($P = 0.035$). Given only one significant result at 95% level in 18 tests, we conclude that there is no evidence of a consistent effect of farm proximity on chlorophyll *a*. Analysis of covariance, using all survey data, also showed chlorophyll *a* levels not to be significantly higher in the proximity of finfish farms, at either surface or mid-depth. This is not at all surprising: given the short flushing times for the estuary, comparable with time scales for phytoplankton growth,

we would expect nutrients from farms to be dispersed broadly within the estuary before there is any biomass response from phytoplankton.

5.4.2 Community composition – marker pigments

HPLC pigments that relate specifically to an algal class are termed ‘marker pigments’ (Jeffrey and Vesik 1997). Some of these marker pigments are found exclusively in one algal class (e.g. alloxanthin which is only found in Cryptophytes) while others are the principal pigments of one class but are also found in other classes (e.g. fucoxanthin in diatoms and others). The presence of these marker pigments can be used as a guide to phytoplankton composition, as well as to biomass. We have used the relative contribution of seven marker pigments (carotenoids) to indicate changes in phytoplankton composition throughout the estuary during the period of this study. These pigments, and the corresponding algal classes, are shown in table 5.3.

Table 5.3 Marker pigments (with abbreviations) and the algal classes they represent.

Pigment	Algal Class
Fucoxanthin (Fuco) and Diadinoxanthin (DD)	Diatoms
Peridinin (Perid)	Dinoflagellates
19'-hexanoyloxyfucoxanthin (19HF)	Haptophytes
Alloxanthin (Allo)	Cryptophytes
Prasinoxanthin (Pras)	Prasinophytes
Lutein (Lut)	Chlorophytes
Zeaxanthin (Zea)	Cyanobacteria

Plots of total carotenoid composition in the surface and mid-depth samples in Feb '97 (HES 4), Jun '97 (HES 5) and Dec '97 (HES 7) are shown in Figures 5.2a,b. In these surveys and in other surveys, the carotenoid composition of samples from fresh surface waters in the upper estuary differs markedly from that of surface and mid-depth samples in the middle and lower estuary and Port Cygnet. Over all surveys, 84% of surface samples in the upper estuary have no fucoxanthin (indicative of diatoms), compared with 19.4% in the middle estuary, and 4.8% and 2.8% in the lower estuary and Port Cygnet respectively. Fucoxanthin is successively replaced by alloxanthin, lutein and zeaxanthin in fresher surface waters. However, the mid-depth samples in the upper estuary, which are taken from the salt wedge, are more similar to middle estuary samples, although 19HF disappears, and lutein increases slightly.

In Feb '97 (HES 4), the carotenoid composition is rather homogeneous, horizontally and vertically, throughout the lower and middle estuary, but is dominated by fucoxanthin (diatoms) and 19HF (haptophytes), with small amounts of alloxanthin (cryptophytes) and zeaxanthin (cyanobacteria). A small amount of prasinoxanthin appears in the middle estuary, and traces of peridinin in transect A at the mouth of the estuary.

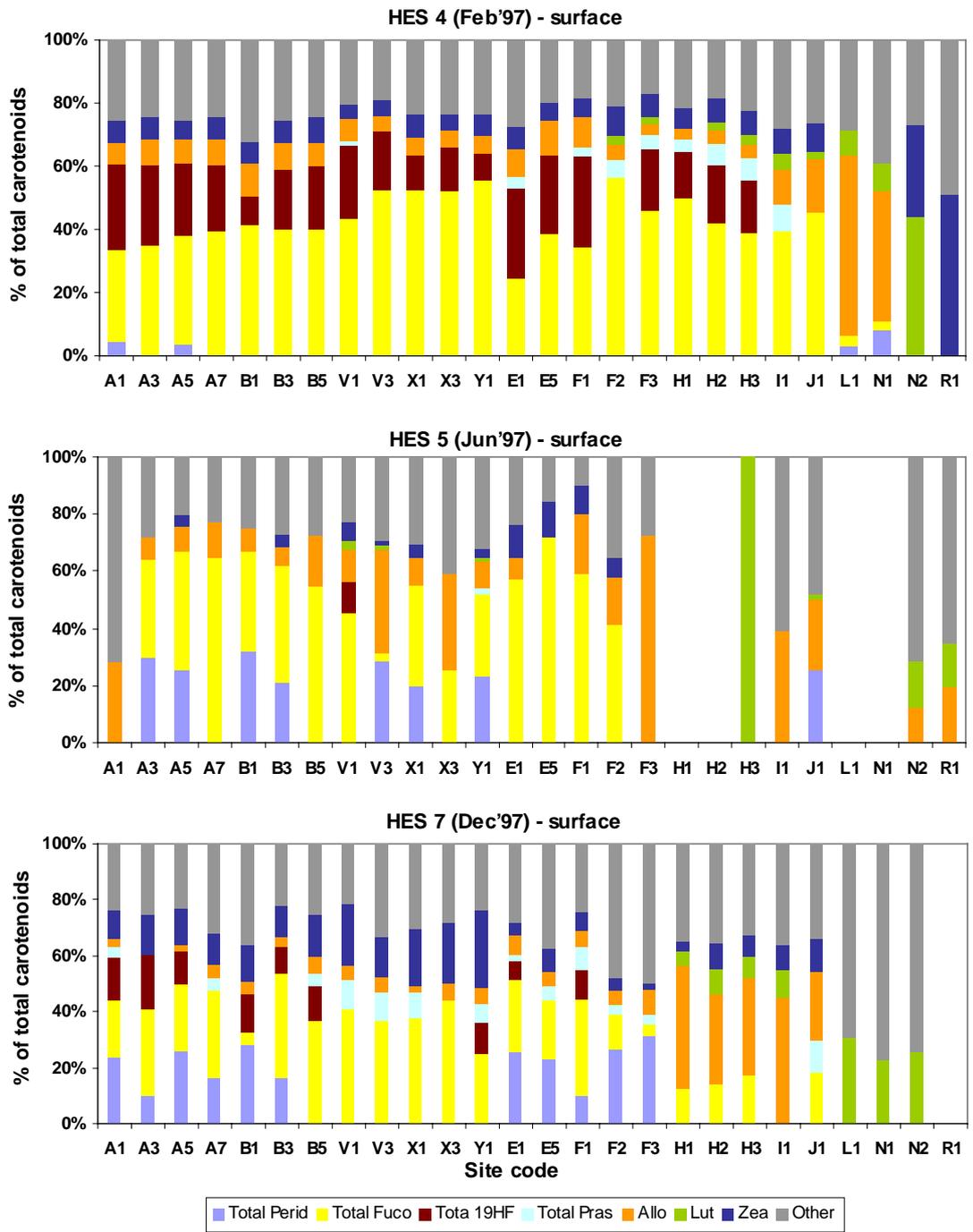


Fig. 5.2a Marker pigment composition in surface samples versus station number for HES 4 (Feb '97), HES 5 (Jun '97) and HES 7 (Dec '97).

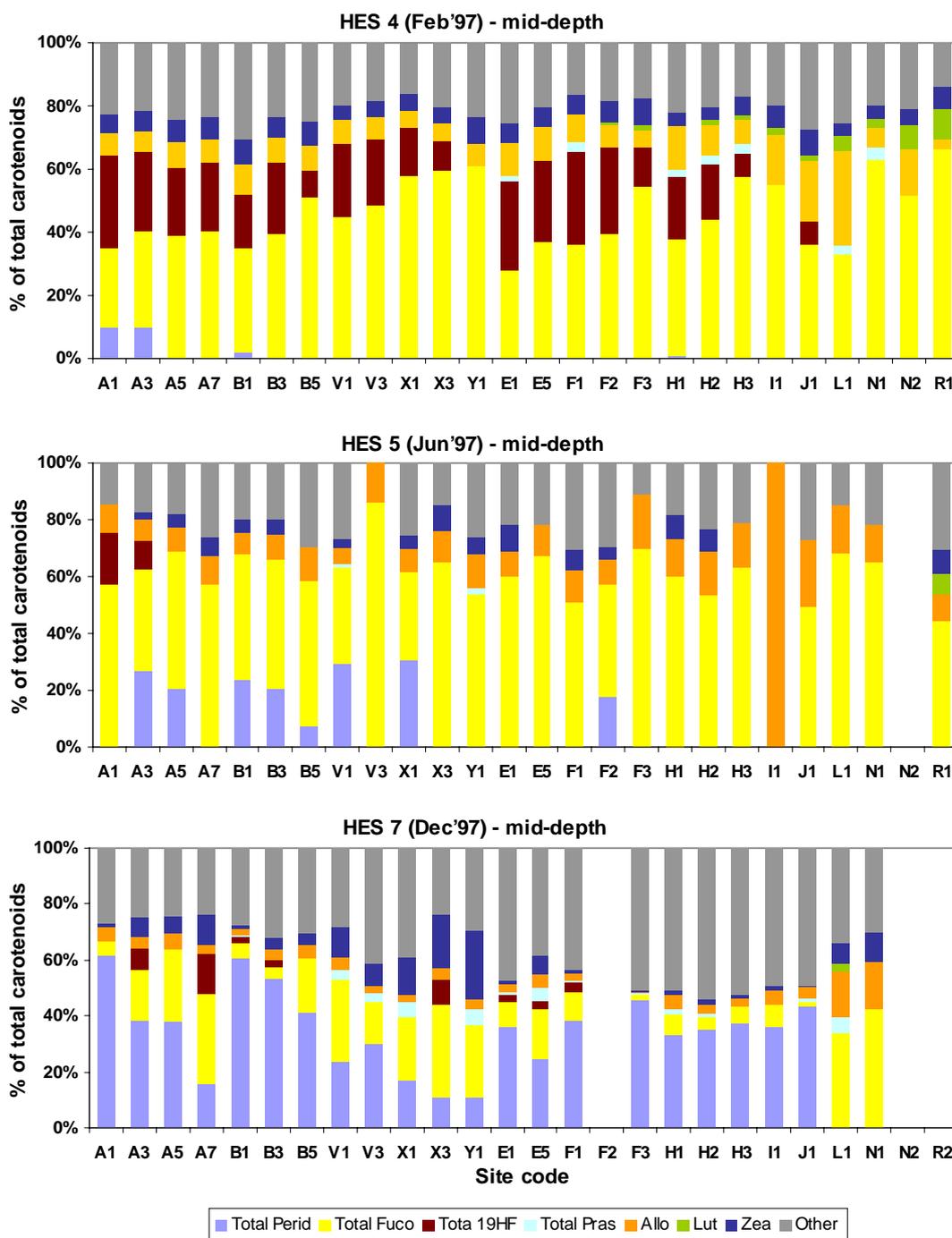


Fig. 5.2b Marker pigment composition in mid-depth samples versus station number for HES 4 (Feb '97), HES 5 (Jun '97) and HES 7 (Dec '97).

In Jun '97 (HES 5), despite the rather low winter chlorophyll *a* levels, there is significant spatial variation in carotenoid composition. Large amounts of peridinin (dinoflagellates) occur in some, but not all, stations in the lower estuary. In comparison with Feb '97 (HES4), alloxanthin is high at some sites (indicating the presence of cryptophytes), while 19HF (indicating haptophytes) is almost absent. Peridinin is present at J1, where surface chlorophyll *a* was also high. In Dec '97 (HES 7), peridinin is present in surface samples throughout most of the middle and lower estuary, but absent from the stations in or just outside Port Cygnet. However, peridinin is present in mid-depth samples from Port Cygnet, suggesting a strong subsurface vertical concentration of dinoflagellates. The haptophyte marker 19HF is also absent in Port Cygnet, but prasinoxanthin is higher there. Peridinin is present in mid-depth samples as far up-estuary as J1, but absent from L1 and N1.

The spatial surveys consistently show strong differences in pigment composition between low-salinity surface water in the upper estuary, and the rest of the estuary. These are not unexpected: the community in fresh surface waters may be a remnant riverine community. While there is also evidence of some vertical and horizontal spatial variation in the middle and lower estuary and Port Cygnet, the phytoplankton community composition there shows substantial spatial coherence, so that seasonal or interannual changes in pigment composition (e.g. from Feb '97 (HES 4) to Dec '97 (HES 7)) are evident throughout this region. Again this is not surprising, as the physical analysis indicates relatively rapid flushing of this region (Chapter 3, 10). This provides support for treating weekly data from the five biological stations as representative of the middle and lower estuary and Port Cygnet in our analysis of temporal variation in phytoplankton biomass and composition.

5.5 Temporal Variation in Phytoplankton Biomass and Composition

5.5.1 Chlorophyll *a*

*Frequency distribution for log(chl *a*)*

Figure 5.3 shows the frequency distribution of log(chl *a*) over all 5 sites and all dates (323 data points). This distribution shows a small upper tail, but is not highly skewed. The mean is -0.06 (geometric mean of $0.88 \text{ mg chl } a \text{ m}^{-3}$), with a standard deviation of 0.43 ($CV = 1.51$). The median (-0.10) is very close to the mean. The 90 percentile value of log(chl *a*) is 0.6 : that is, chl *a* exceeds $4 \text{ mg chl } a \text{ m}^{-3}$ about 10% of the time. This distribution of chl *a* levels is characteristic of a mesotrophic environment, and generally consistent with the moderate nutrient levels described in Chapter 4.

*Spatial and temporal variation in chlorophyll *a* and cell counts*

The variation over time in chl *a* at each of the five biological sites is presented in Figure 5.4. We subjected these data (log-scaled) to a 2-way ANOVA without replication, with date and sites as the factors. Most of the variation in log(chl *a*) is accounted for by variation across time ($P = 10^{-47}$). Site effects are not significant: i.e. there is no consistent difference in time-averaged log(chl *a*) among the sites. The mean square residual variance in log(chl *a*) (i.e. variation among

sites within days) is 0.055, corresponding to a coefficient of variation of 0.54. This is similar to the coefficient of variation in surface chlorophyll among all chemical sites in the lower and middle estuary within spatial surveys, and suggests that the five biological sites are representative of the broader set of chemical sites. If we treat the five sites as replicate samples, differences in mean $\log(\text{chl } a)$ between dates are significant at the 95% level (two-tailed) if the difference exceeds 0.46 (i.e. geometric means differ by a factor of 2.9).

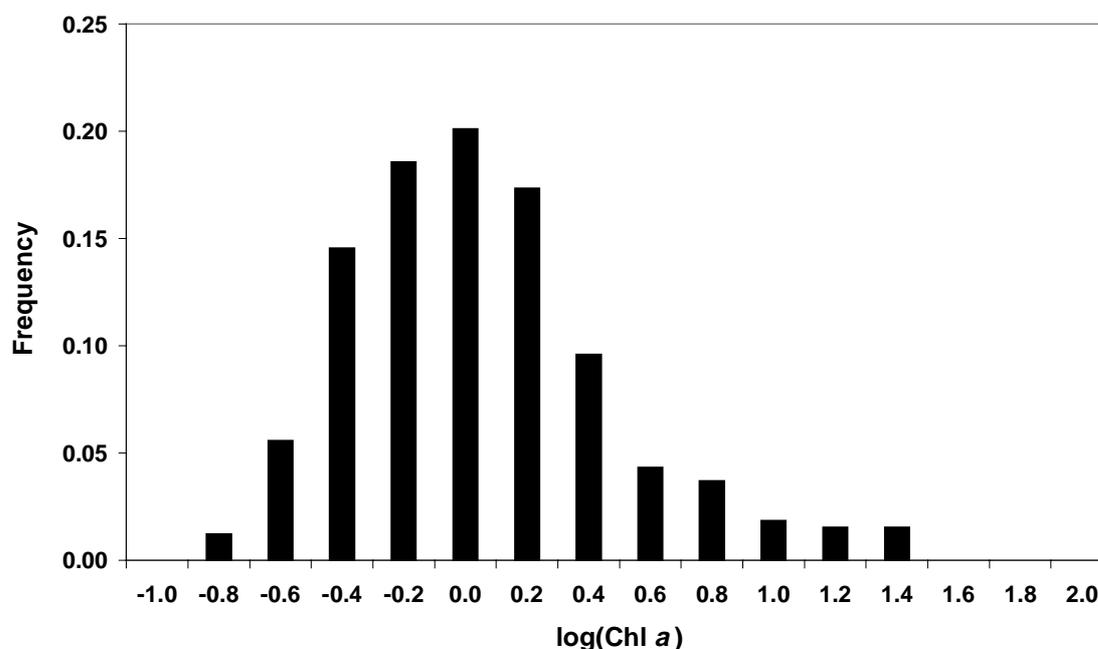


Fig. 5.3 Frequency histogram of $\log(\text{Chl } a)$ for 323 samples from the biological stations.

Table 5.4 Variance over time in depth-integrated samples from the biological stations.

	B1	F1	F3	H3	X3B
Log(chl <i>a</i>)	0.12	0.22	0.23	0.23	0.14
Log(flagellates)	0.027	0.028	0.028	0.043	0.030
Log(dinoflagellates)	0.080	0.15	0.16	0.16	0.12
Log(diatoms)	0.76	0.80	0.66	0.60	0.69

The ANOVA shows that a coherent variation over time is the overwhelmingly dominant signal in the biological site data. While there are no significant differences in mean $\log(\text{chl } a)$ among sites, there are some subtle differences in temporal variation among sites. The variation in $\log(\text{chl } a)$ over time was higher for middle estuary sites than for lower estuary and Port Cygnet sites (Table 5.4). Correlations among sites in the middle estuary and Port Cygnet (F1, F3, H3 and X3B) were high (0.79 to 0.86) compared with correlations between these sites and site B1 in the lower estuary (0.52 to 0.69). This is due to differences in the pattern of high biomass dinoflagellate blooms in the lower and middle estuaries (see section 5.5.2).

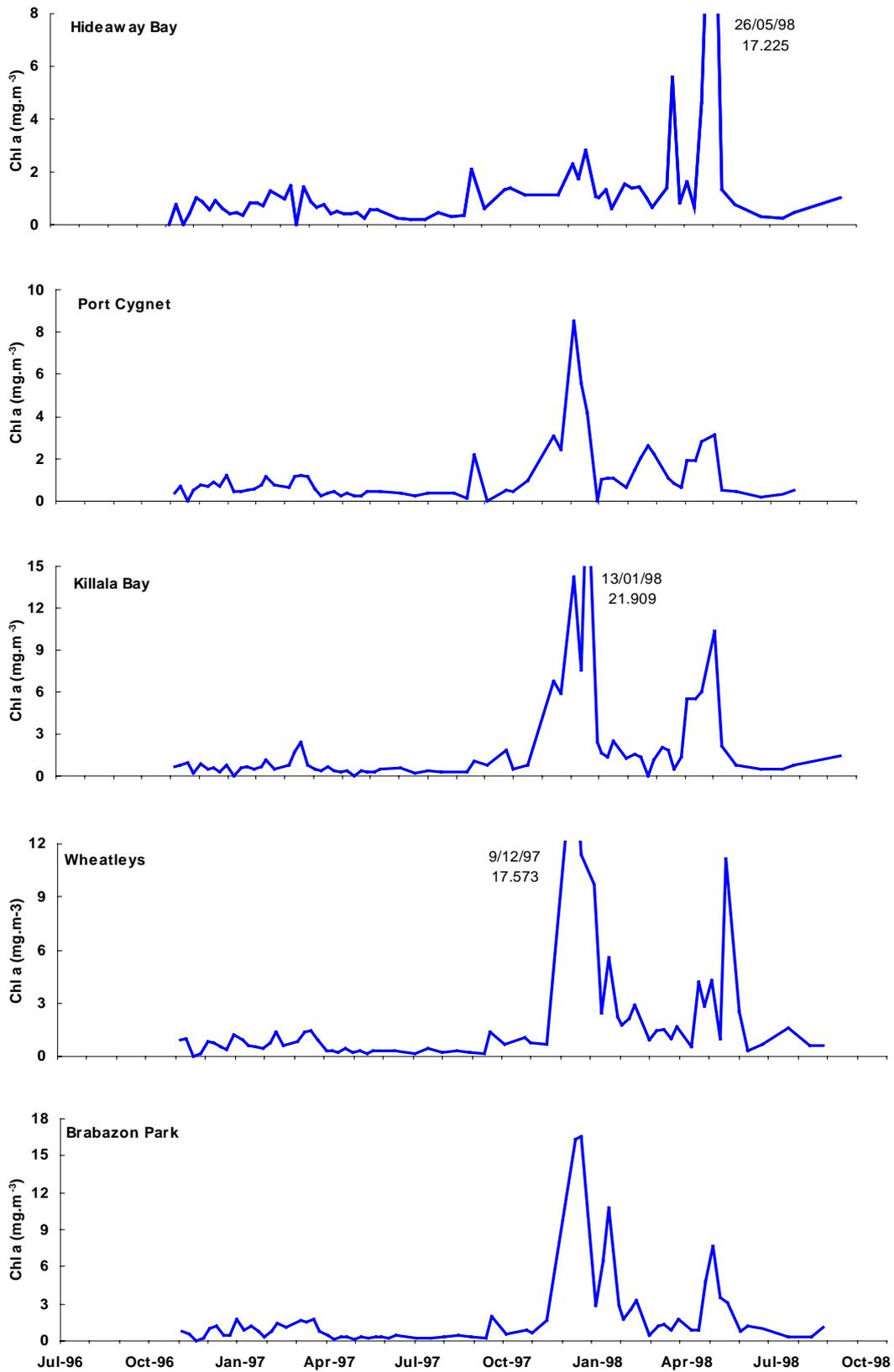


Fig. 5.4 Chlorophyll a from depth-integrated samples at biological stations versus time.

Seasonal variation in chlorophyll *a*

Treating the 5 sites as replicate samples, mean log(chl *a*) ranges over time from -0.69 to 0.84 . We have defined seasons based on the solstice and equinox dates, so that spring runs from 21st September to 21st December, etc. Seasonal means and 95% confidence intervals (based on *t*-tests) are given in Figure 5.5. While there is some evidence of a seasonal pattern (values in winter 97 and winter 98 are consistently lower than spring and summer values in 1996/97 and 1997/98), the dominant signal is interannual. Chlorophyll levels in the spring, summer and autumn period of 1997/98, associated with large dinoflagellate blooms, are substantially higher than values in the corresponding seasons in 1996/97.

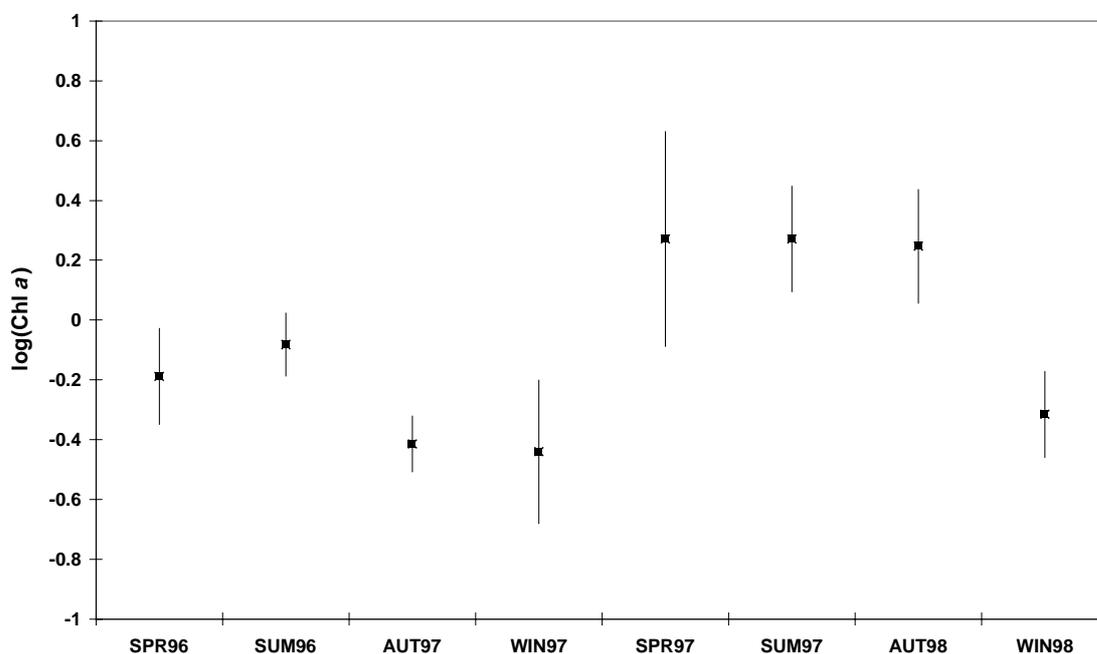


Fig. 5.5 Seasonal means and 95% confidence intervals for log(Chl *a*), based on means of the biological stations.

5.5.2 Bloom composition – cell counts

Cell counts from depth-integrated samples at the biological stations are shown in Figure 5.6. Cell concentrations are provided for small flagellates and four species or species groups: the diatoms, *Chaetoceros* spp. (dominated by *C. socialis*, *C. decipiens* and *C. debilis*) and *Pseudonitzschia* spp. (dominated by *P. pseudodelicatissima* with *P. subpacificica* being the next most abundant); and the dinoflagellates *Ceratium* spp. (*C. furca*, *C. tripos* and *C. fusus*) and *Gymnodinium catenatum*. These species or species groups all formed blooms as defined by high cell numbers elevated above background levels (*Pseudonitzschia* spp. and *Chaetoceros* spp., $>10^5$ cellsL⁻¹; *Ceratium* spp. and *G. catenatum*, $>10^4$ cellsL⁻¹). Diatoms and dinoflagellates dominate the phytoplankton blooms, and these four bloom species/groups dominate cell numbers during blooms. Blooms were observed in spring, summer and autumn of both 1996/97 and 1997/98. Diatom blooms occurred in late spring/early summer and in late summer in both

years at roughly equal intensities. Dinoflagellate blooms occurred only in the summer and autumn of 1997/98.

We conducted a two-way ANOVA for log(cell counts) of small flagellates, total diatoms and total dinoflagellates, comparable to that described for log(chl *a*) above. For small flagellates and dinoflagellates, the ANOVA results are similar to those for chl *a*: the effect of time is highly significant ($P=10^{-54}$ for small flagellates, 10^{-38} for dinoflagellates), but the site effect is not significant. That is, for small flagellates and dinoflagellates, variation over time in mean log(chl *a*) is much larger than variation among sites within times, while there are no significant differences in time-averaged log(chl *a*) among sites. For diatoms, there is a significant site effect ($P = 10^{-7}$), with diatom counts at X3B (Port Cygnet) being on average about 60% higher than at the other sites.

While the variation in log(cell counts) over time is highly significant for all taxa, its magnitude varies considerably among taxa (Table 5.4). Counts of small flagellates show much less relative variation over time (CV = 40%) than dinoflagellate counts (CV = 92%) or diatoms (CV = 190%). Dinoflagellate counts have a similar coefficient of variation to chl *a*, which is not too surprising, as dinoflagellates tend to dominate blooms at times of high chl *a*. The large variation in diatom counts over time reflects their occurrence primarily in ephemeral blooms. Diatom blooms tend to be coincident at all sites, and the diatom time series at different sites are highly correlated, with correlation coefficients from 0.81 to 0.92. For dinoflagellates, correlations are generally lower (0.42 to 0.74), with site B1 (lower estuary) comparatively weakly correlated with other sites, as was the case for chl *a*.

The high coherence in cell counts among the five biological stations is consistent with the earlier observations of spatial coherence in biomass and composition in spatial surveys in the lower and middle estuary and Port Cygnet (see section 5.4). The key exceptions were the early summer diatom bloom of 1996/97, which appeared to be confined to the lower estuary, and the early summer dinoflagellate bloom of 1997/98, which was almost absent at Hideaway Bay. It seems likely that the latter explains the lower correlation of B1 with other sites for both dinoflagellates and chl *a*. The comparatively low relative variation in small flagellate counts over time may partly reflect the fact that this is a broad taxonomic group, but it is also commonly observed that small phytoplankton cells show much less variation over time than larger cells in coastal waters (Harris et al. 1996).

5.5.3 Bloom biomass – cell volume and chlorophyll *a*

Cell counts can be a misleading guide to phytoplankton biomass, as individual phytoplankton cell volumes vary among species by more than three orders of magnitude. We have converted the cell numbers for bloom species to total cell volume, using individual cell volumes estimated from microscope measurements (for the diatoms *Pseudonitzschia* spp. and *Chaetoceros* estimated cell volumes were $200 \mu\text{m}^3$ and $250 \mu\text{m}^3$ respectively, while for *G. catenatum* the cell volume was $1.7 \times 10^4 \mu\text{m}^3$ and for the *Ceratium* spp. it averaged $6 \times 10^4 \mu\text{m}^3$, using Smayda's (1978) methods). The dinoflagellate cells were much larger than the diatoms, so formed a much greater biomass than diatoms at similar cell densities. When dinoflagellates bloom as in 1997/98 they completely dominate the biomass (Fig. 5.7). Consequently, the estimated total biomass is much higher for the dinoflagellate blooms of 1997/98 than in the previous year. Blooms of *Ceratium* spp. and *G. catenatum* alternate in the bloom period,

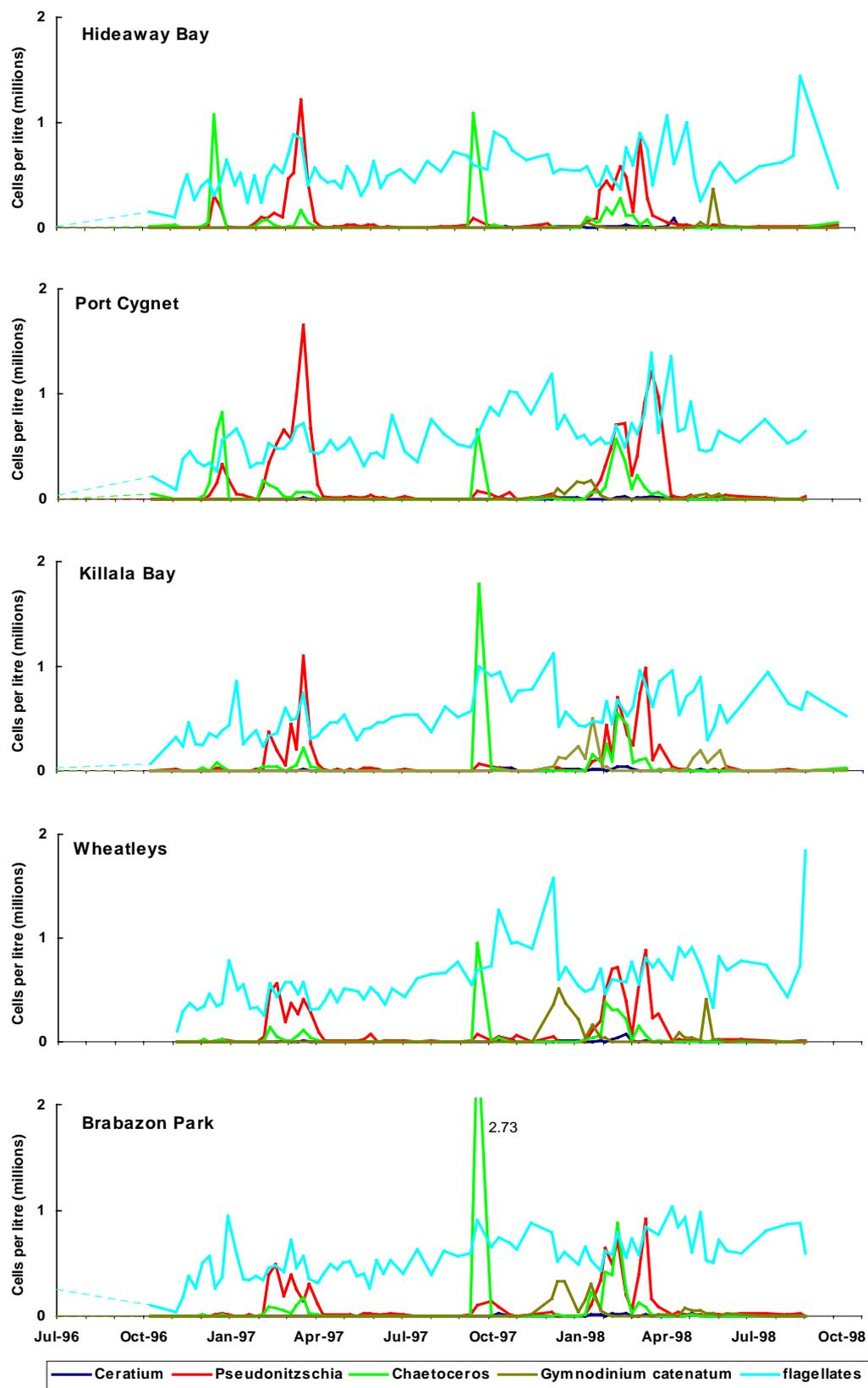


Fig. 5.6 Cell concentrations of bloom species and small flagellates from depth-integrated samples at biological stations versus time.

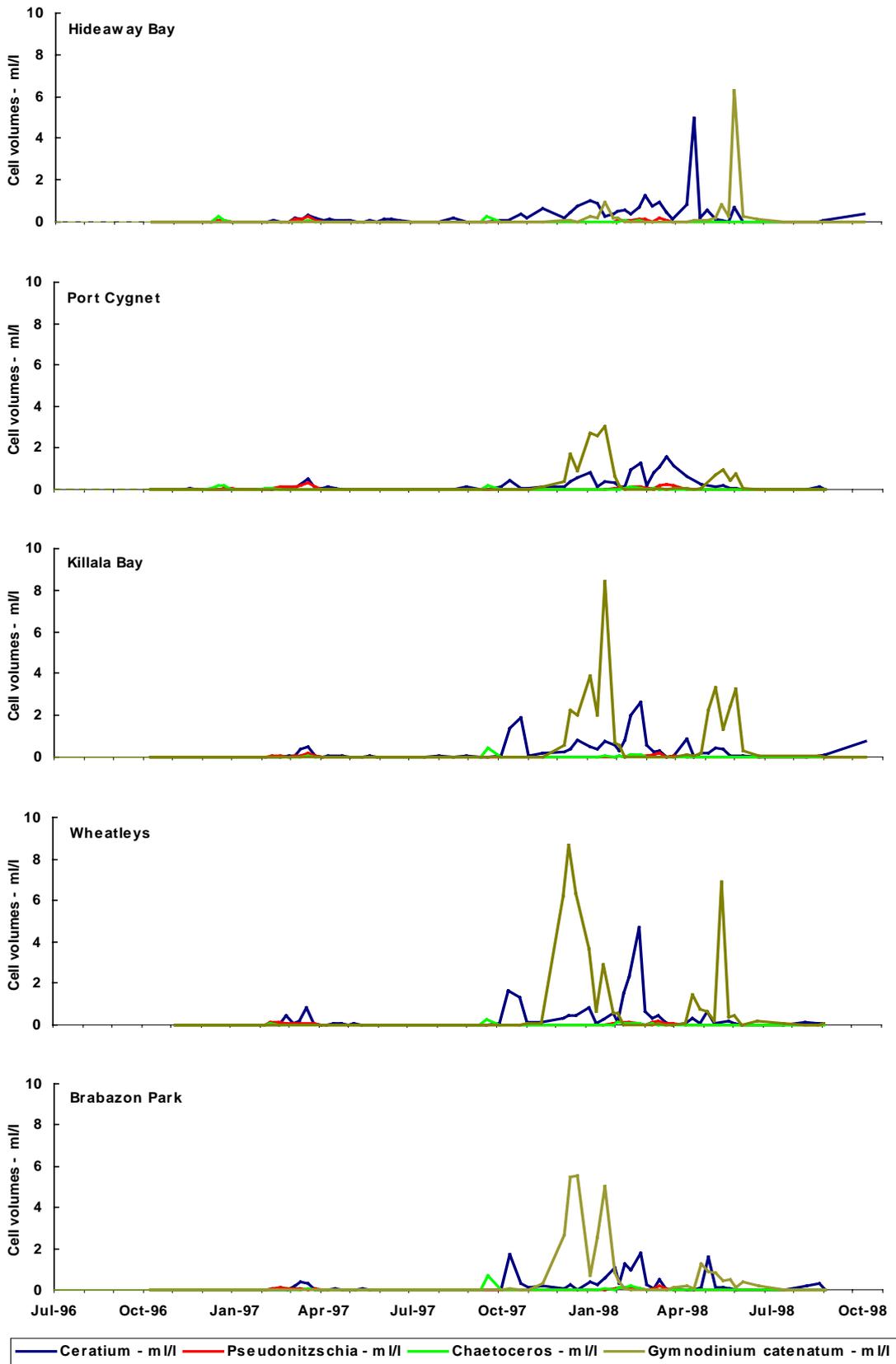


Fig. 5.7 Total cell volume of bloom species in depth-integrated samples at biological stations versus time.

with blooms of *Ceratium* in spring and mid-summer, and blooms of *G. catenatum* in early summer and autumn.

For high concentrations of biomass and chlorophyll *a*, there is a tight relationship between chlorophyll *a* and total cell volume for depth-integrated samples from the biological stations. For total cell volume greater than 3×10^8 ($\mu\text{m}^3 \text{ l}^{-1}$), a regression of $\log(\text{chl } a)$ on $\log(\text{total cell volume})$ has an R^2 of 0.74 and a slope of 0.85. However, for non-bloom low biomass samples, the relationship between chl *a* and total cell volume is poorly defined. The high biomass samples are dominated by one or two bloom-forming species, primarily *G. catenatum*, so one might expect a tight relationship. In contrast, low biomass samples may be dominated by diatom species other than *Pseudonitzschia* and *Chaetoceros* spp. which are not included in cell volume estimates, and/or by small flagellates for which only a crude cell volume estimate is used.

5.5.4 Community composition – marker pigments

The HPLC marker pigments from the biological stations provide a picture of changes in community composition at time scales of weeks to years. Figure 5.8 shows carotenoid composition at Killala Bay versus time. From October 1996 to June 1997 the main marker pigments were fucoxanthin and alloxanthin, indicating dominance by diatoms and cryptophytes. Elevated levels of 19HF (haptophytes) were observed occasionally, including over three weeks in December 1996/January 1997. There was a significant contribution of zeaxanthin (cyanobacteria) at the beginning of the study, from October 1996 to March 1997, with occasional high contributions. However, zeaxanthin then diminished and rarely contributed significantly after March 1997.

The main change from the first to the second year of the study was the appearance of high levels of peridinin, reflecting the dominance of dinoflagellates in the second summer. Peridinin made a large contribution in almost all samples after September 1997, and was the dominant or even sole marker pigment in core dinoflagellate bloom periods from 9 December 1997 to 13 January 1998, and from 28 April 1998 to 3 June 1998. Peridinin first appeared in March 1997, and appeared intermittently, but sometimes as a dominant marker pigment, through the 1997 winter.

5.6 Factors Controlling Phytoplankton Dynamics

Phytoplankton in the Huon Estuary show striking seasonal and interannual changes in biomass and composition. These changes are the outcome of dynamic processes controlling phytoplankton species growth and loss rates. Given the high flushing rates in the Huon Estuary, one would expect loss rates of phytoplankton to be controlled largely by physical advection (Chapter 10). Other losses include grazing by zooplankton or benthic filter-feeders, sedimentation, bacteria and viruses (Zingone 1995), and senescence. As food-web interactions such as grazing and microbial processes were beyond the scope of our study we have little knowledge of these loss processes. However there is evidence from other studies (Lovejoy et al. 1998, J. Skerratt unpublished data) that algicidal bacteria may contribute to phytoplankton losses in the Huon Estuary and in other systems (Kim et al. 1998). As well, several grazers

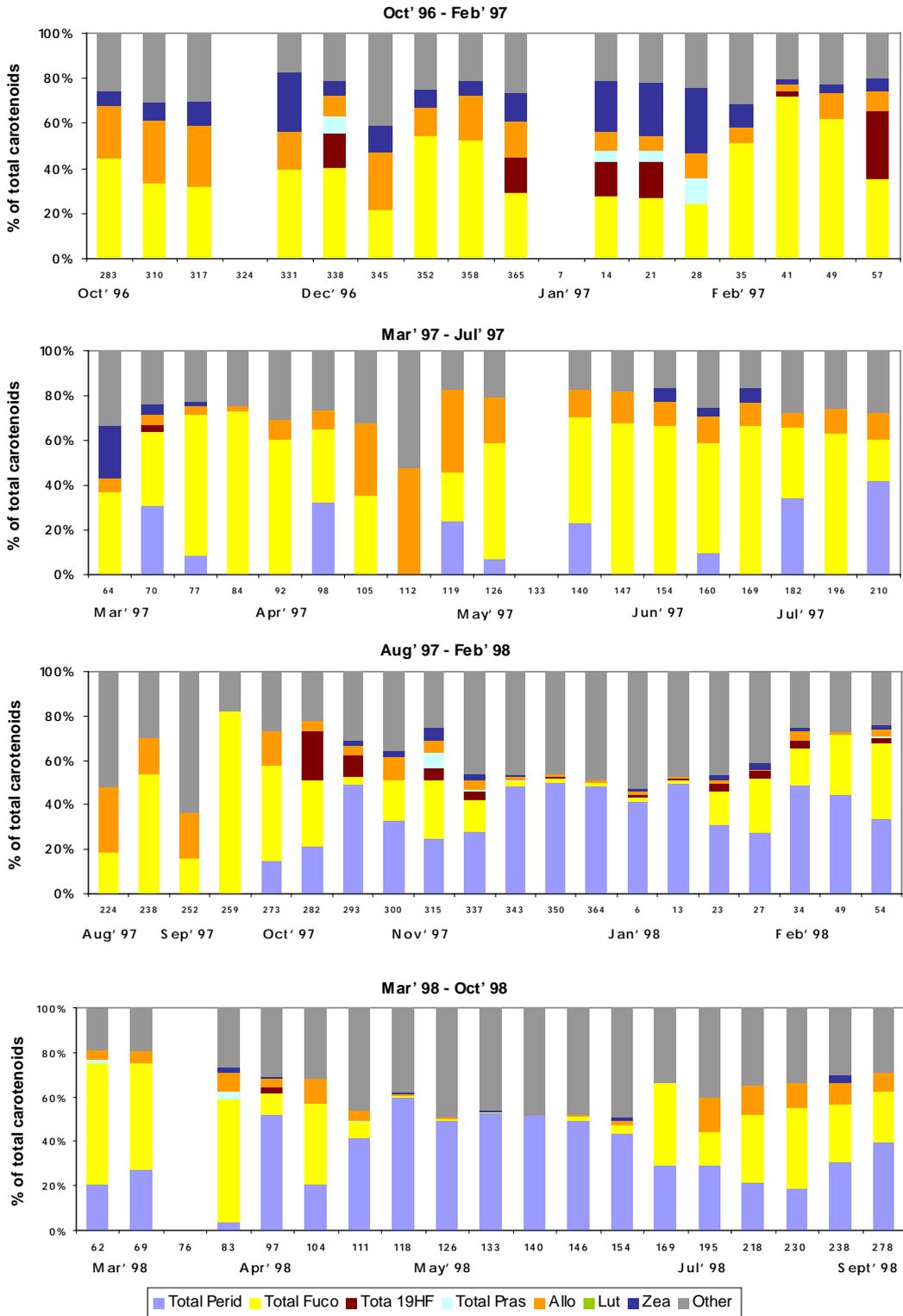


Fig. 5.8 Marker pigment composition for depth-integrated samples from Killala Bay.

were identified in net samples (*Polykrikos schwartzii* feeding on *G. catenatum*; copepods, rotifers and sometimes large numbers of fecal pellets).

Phytoplankton growth rates are typically controlled by light, nutrients (macronutrients or micronutrients) and temperature. We consider the light environment in Chapter 6, and its implications for growth rates in Chapter 10. In this section, we consider the interaction of phytoplankton and macro-nutrient concentrations. Analysis of nutrient ratios suggests N rather than P is likely to be limiting to phytoplankton growth (Chapter 4), so we focus on changes in dissolved inorganic nitrogen.

5.6.1 Nutrients and phytoplankton blooms

Figure 5.9 shows the variations in cell counts of Killala Bay diatoms and dinoflagellates plotted with mid-depth nitrate/nitrite (NO_x), ammonia and silicate concentrations. NO_x concentrations show a consistent seasonal cycle over the two field years. Winter values are elevated, about 4 to 5 μM , while summer values are low (from about October to April). Silicate values are generally around 4 μM , with peaks over 20 μM , probably associated with advection of low-salinity water over the site. Ammonia values were not measured on all occasions, but the observations show high values (around 1 μM) in late summer in both years, and lower values in winter.

There are some interesting associations between phytoplankton blooms and nutrient levels apparent in Figure 5.9. Diatom blooms appear to be associated with the early spring drawdown of NO_x in 1997, and with the autumn reappearance of NO_x in 1997 and 1998. Some diatom blooms appear to be associated with drawdown of silicate. Elevated ammonia values appear to follow summer diatom blooms in 1997 and diatom and dinoflagellate blooms in 1998. *Ceratium* spp. and *G. catenatum* blooms in autumn 1998 are associated with drawdown of ammonia, and ammonia and NO_x , respectively.

Although only data from Killala Bay are plotted here, the results for other biological stations show a very similar pattern. Silicate values are higher further upstream at Brabazon Park and Wheatleys, and lower downstream at Hideaway Bay, reflecting the principal sources of silicate – the Huon and Kermadie Rivers. Winter nitrate values are similar at all stations, reflecting the strong circulation of marine nitrate through the middle and lower estuary in winter (Chapter 10). In 1997/98, peak ammonia values are much lower in Port Cygnet (about 0.2 μM) than at other biological stations (about 1 μM or greater). The reasons for this are not clear. There is some evidence from spatial surveys (HES 6 (Oct '97), HES 7 (Dec '97) and HES 8 (Feb '98)) that ammonia concentration in bottom waters in the lower estuary increase with depth (Chapter 4).

Euphotic zone nutrient concentrations can be uninformative when they are depleted by phytoplankton, and phytoplankton biomass and productivity may respond instead to episodic fluxes of nutrients into this zone. One might expect nutrient concentrations in bottom waters in the estuary to drive nutrient inputs through vertical entrainment (see Chapter 10). *Gymnodinium catenatum* also undergoes large diel vertical migrations (see section 5.7), enabling it to sequester bottom nutrients directly (Doblin et al 2000b).

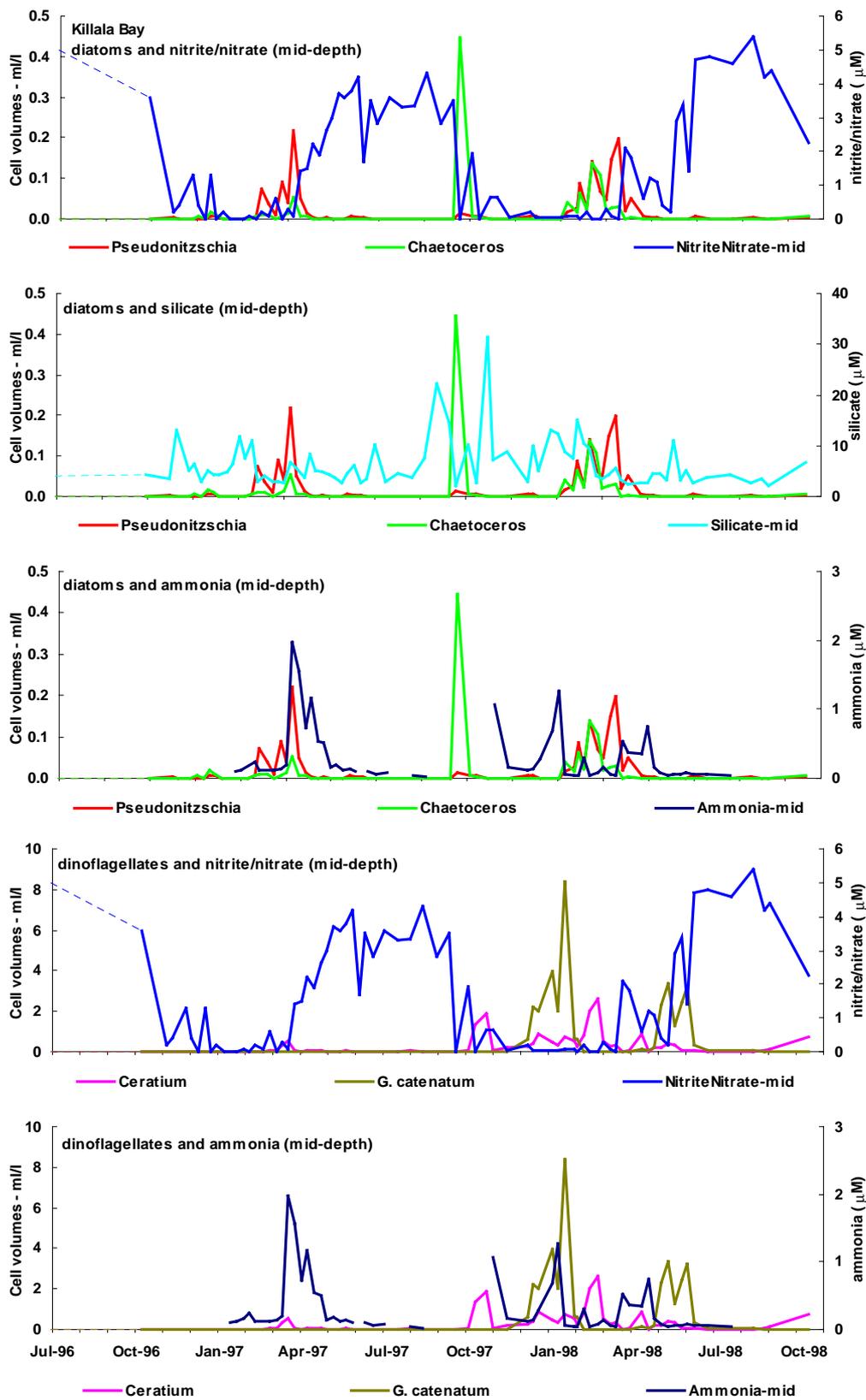


Fig. 5.9 Bloom concentrations of diatoms and dinoflagellates, plotted over mid-depth concentrations of nitrate/nitrite, ammonia and silicate, at Killala Bay.

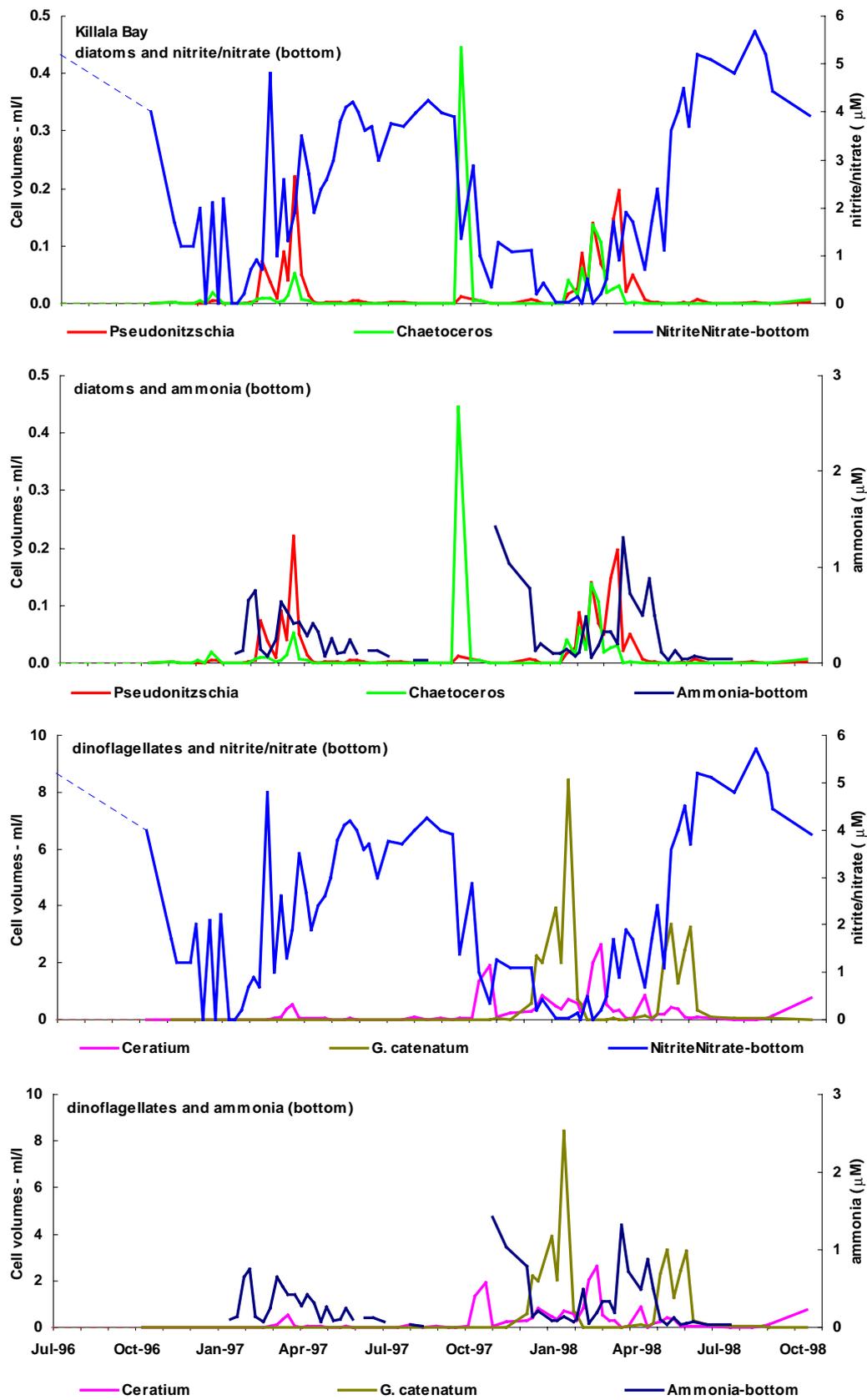


Fig. 5.10a Bloom concentrations of diatoms and dinoflagellates, plotted over bottom depth concentrations of nitrate/nitrite and ammonia, at Killala Bay.

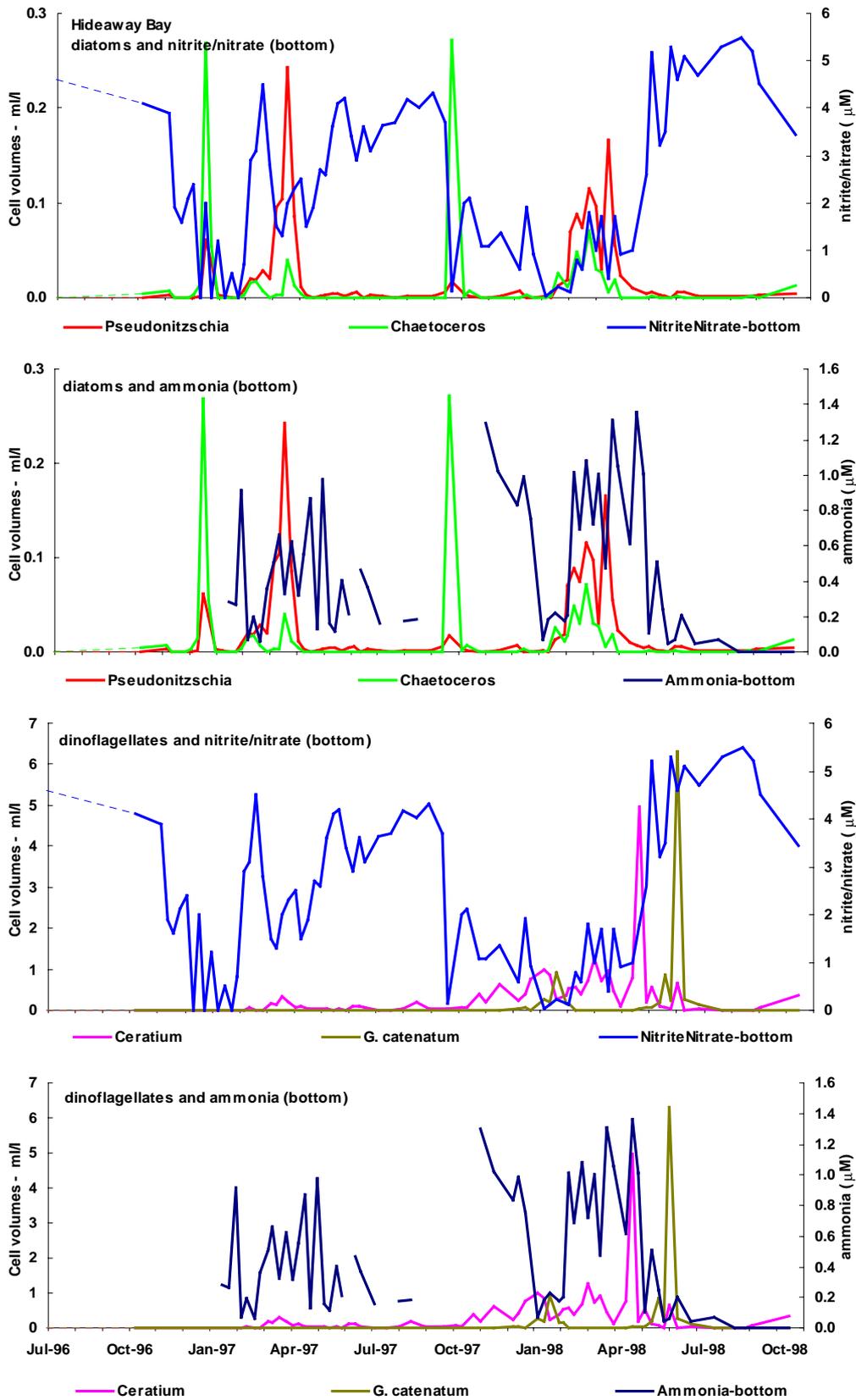


Fig. 5.10b Bloom concentrations of diatoms and dinoflagellates, plotted over bottom depth concentrations of nitrate/nitrite and ammonia, at Hideaway Bay.

Bottom NO_x levels at Killala Bay (Fig. 5.10a) and at Hideaway Bay (Fig. 5.10b) showed a similar seasonal pattern to surface NO_x values, although there is evidence of a strong injection of NO_x in bottom waters at Hideaway Bay in February 1997. In the 1997/98 summer, NO_x values in bottom waters at both Killala and Hideaway Bays fluctuated around 1 to 2 μM . An exceptional period was January 1998 when NO_x values were close to zero. Bottom ammonia values also dropped significantly. This period coincided with the first *G. catenatum* bloom. This same pattern was not seen for *Ceratium* spp. except for April 1998 when there was a decrease in bottom ammonia at Hideaway Bay associated with a bloom. Overall there appears to be much more effective uptake of bottom NO_x and ammonia by *G. catenatum* than *Ceratium* spp., probably because of the greater speed of vertical migration by *G. catenatum* (Levandowsky and Kaneta 1987, Kamykowski et al. 1992).

There is at least circumstantial evidence in these data that *G. catenatum* may show a strong preference for ammonia over nitrate. *G. catenatum* blooms appeared to follow increases in bottom ammonia concentrations, and lead to depletion of bottom ammonia. However, while observed ammonia concentrations were slightly lower in 1996/97 than in 1997/98, it is hard to argue that these differences are sufficient to explain the lack of a dinoflagellate bloom in 1996/97. Bottom NO_x levels were generally higher in 1996/97 than in 1997/98. Observed ammonia concentrations represent a balance between production (from remineralization of organic matter in water column and sediment), uptake by phytoplankton and transport. It is possible that production rates were higher in 1997/98, but masked by higher uptake rates during blooms. However, ammonia concentrations in bottom water between blooms (when uptake rates were presumably low) remained below about 1.5 μM .

5.6.2 Runoff, wind and phytoplankton blooms

The observations of nutrients and phytoplankton in the Huon Estuary are generally consistent with a classical temperate plankton cycle: high surface nutrients in winter (when phytoplankton are light-limited), a spring and late summer diatom bloom, and mid-summer to autumn dinoflagellate blooms. There is evidence of phytoplankton drawdown of nutrients during blooms, and evidence that ammonia may be regenerated after blooms. However, there is little evidence that the dinoflagellate blooms we observed in summer were triggered by elevated nutrients. Indeed, it is difficult to see any difference in nitrogen or silicate availability between the two summers which would explain the occurrence of an intense and prolonged dinoflagellate bloom in 1997/98, but not in 1996/97 (see section 5.8). We therefore considered other factors implicated in dinoflagellate blooms.

Various hypotheses have been proposed to link river runoff to dinoflagellate blooms in estuarine systems. In particular Doblin et al. (1999a) showed that dissolved organic matter from the Huon River could stimulate *G. catenatum* growth by affecting the supply of either macronutrients or micronutrients (including selenium) or both. River flow through October to March in 1996/97 and 1997/98 (Fig. 5.11) showed remarkable similarity. In both periods, bursts of high river flow in November/December were followed by prolonged periods of low river flow in summer, interspersed by some isolated runoff events. Again, it is difficult to identify any interannual differences in runoff patterns that would explain the dinoflagellate blooms in 1997/98.

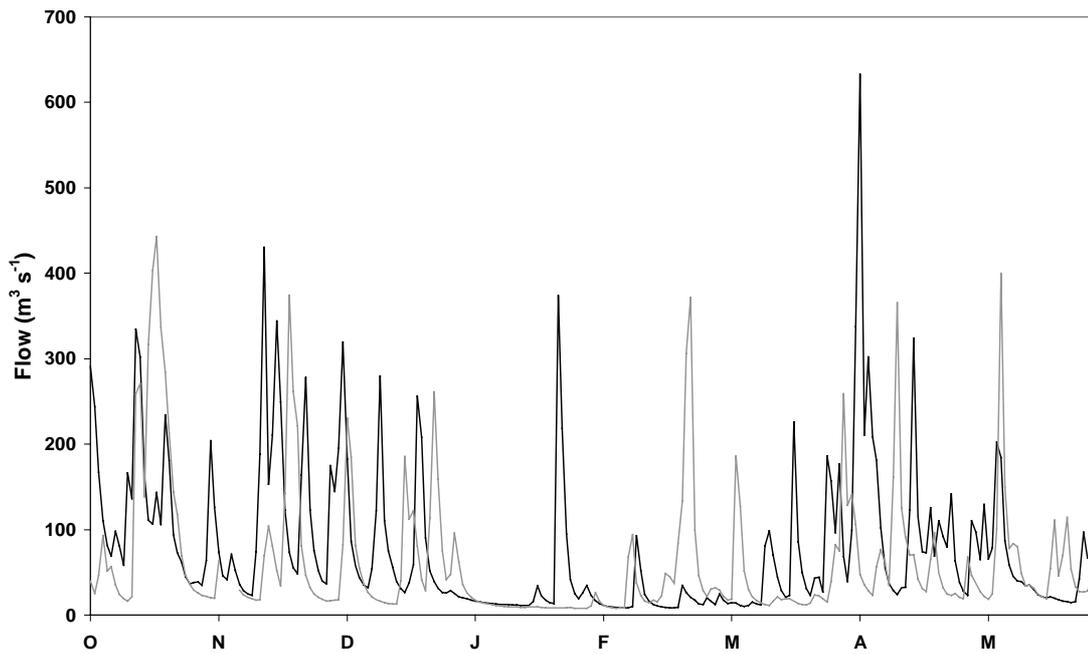


Fig. 5.11 Huon River flow in cumecs versus month for the spring, summer and autumn of 1996/97 (black line) and 1997/98 (grey line).

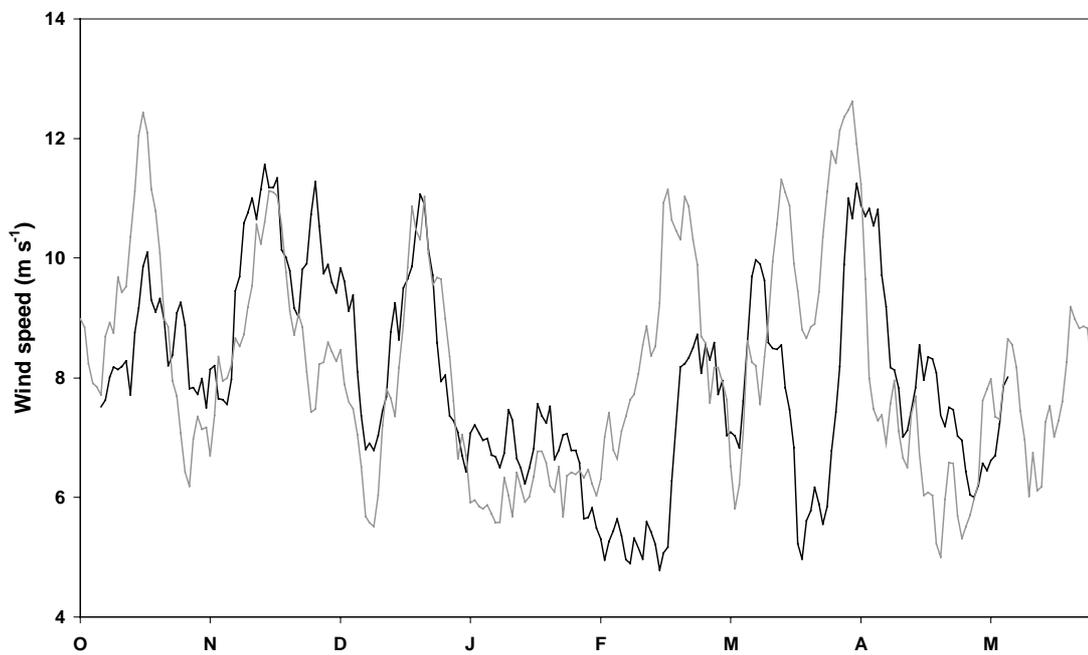


Fig. 5.12 Wind speed at Bruny Island versus month for the spring, summer and autumn period of 1996/97 (black line) and 1997/98 (grey line). Three-hourly wind speed has been smoothed using a 5-day running average.

Dinoflagellate blooms are often associated with stratification and calm conditions, and may be disrupted by wind-induced mixing. Figure 5.12 shows the smoothed wind speed from nearby Cape Bruny (Chapter 3) from October to March 1996/97 and 1997/98. In both periods, wind speeds were high during spring (September – December), and lower in summer (December to March). Indeed average wind speeds are lower in January 1997 than in January 1998, when the dinoflagellate bloom occurred. For a more detailed discussion of the factors influencing *G. catenatum* blooms see section 5.8.4.

5.7 Vertical Structure on Short Time Scales: Results from Automated Profiling Systems

As noted in Chapter 2, previous studies in the Huon and Derwent Estuaries showed that these estuaries display strong vertical gradients and rapid fluctuations in physical, chemical and biological variables. In a pilot study in the Huon, continuous loggers were deployed at fixed depths, but it proved difficult to interpret these results, because changes in vertical distribution of temperature and salinity were not resolved. Time scales of hours to days can be critical for phytoplankton bloom development and runoff events. To address these time scales, and to resolve changes in vertical distribution of key variables, we developed a novel automated profiling system.

5.7.1 The automated profiling system

The profiling system used is described in detail in the supplement. The system had a sensor head, with high-quality temperature, salinity and depth sensors; a compact fluorometer; an oxygen sensor; and a surface data logger with a solar power supply. A winch mounted on a surface platform, lowered and raised the sensor head. During the study, the data was downloaded during weekly/fortnightly sampling trips. Subsequently, a radio-link was added, and data were relayed to shore-based computers in real time.

As discussed in Chapter 2, we originally planned to deploy five systems, one each at the five biological stations, but after reductions in funding at the start of the project, we were able to deploy only two systems – at biological stations next to finfish farms at Killala Bay and Hideaway Bay. The systems were deployed from 1 November 1997.

These were experimental/prototype systems, so not surprisingly there were some initial problems. Most of the problems in the first few months of deployment were related to mechanical difficulties with the winches. Fluorescence data were not available from the Hideaway Bay system in January and February 1998, as the fluorometer had to be removed for recalibration. The most significant instrument failure, however, was the oxygen sensors. The high-quality, rapid-response oxygen sensors we had bought did not survive prolonged immersion in seawater. Further development and testing of profiling oxygen sensors is required.

Both mechanical and instrument failures have led to gaps in the data record. Nonetheless, the system performed well for extended periods, and produced high quality temperature, salinity and fluorescence data, with an unprecedented combination of vertical and temporal resolution. These data are still being analysed. We present here a preliminary analysis of the data set, which we believe demonstrates the promise of this approach as both a monitoring and research tool.

The automated profilers produce large quantities of data. Vertical profiles binned into 0.5 m bins are collected at 1 hour intervals. To capture this temporal and vertical resolution, we have presented most of the data from the profiling systems as time-depth sections, using a colour bar to represent the level of temperature, salinity or fluorescence.

The temperature and salinity data presented here are based on factory calibrations of the sensors. The fluorescence data presented are raw data (volts). These should provide a reasonable relative index of chlorophyll concentration, although there are uncertainties due to variable fluorescence yield and some contribution to fluorescence from CDOM, particularly in surface waters.

5.7.2 Changes in physical structure: runoff and surface-heat fluxes

As expected, the vertical structure at Killala Bay is of a shallow, fresh layer overlying a deep, salty layer (Fig. 5.13). However, the salinity and depth of the surface layer vary rapidly, on time scales of days. Figure 5.13 shows salinity and temperature sections from 1 April 1998 to 31 May 1998. Also plotted is Huon River flow at Frying Pan Creek. Although salinity and river flow vary on similar time scales, there is no consistent relationship between the magnitude of run-off events and surface salinity depression at Killala Bay. This could be partly due to freshwater inputs from catchments downstream of Frying Pan Creek. It has also been noted (Chapter 3) that the freshwater plume in the middle and lower estuary tends to hug the north-eastern shore, on the opposite side to Killala Bay. It is possible that some of the observed changes in salinity at Killala Bay are driven by changes in plume location rather than river runoff.

The temperature section in Figure 5.13 also shows evidence of surface heating and cooling. April is a transitional period between summer surface-heating and winter surface-cooling. There is a period around April 10 when there is a diel oscillation in temperature gradients, so that the surface layer becomes warmer than bottom waters during the day, and cooler than bottom waters over night. By May, and throughout winter, temperatures in surface waters are persistently less than or equal to temperatures in bottom waters.

Surface heating (or cooling) results in much larger temperature gradients during periods with strong salinity stratification, when vertical mixing is reduced. There are episodes in the April-May period where salinity stratification is low and it appears that convective cooling results in overturn of the entire water column.

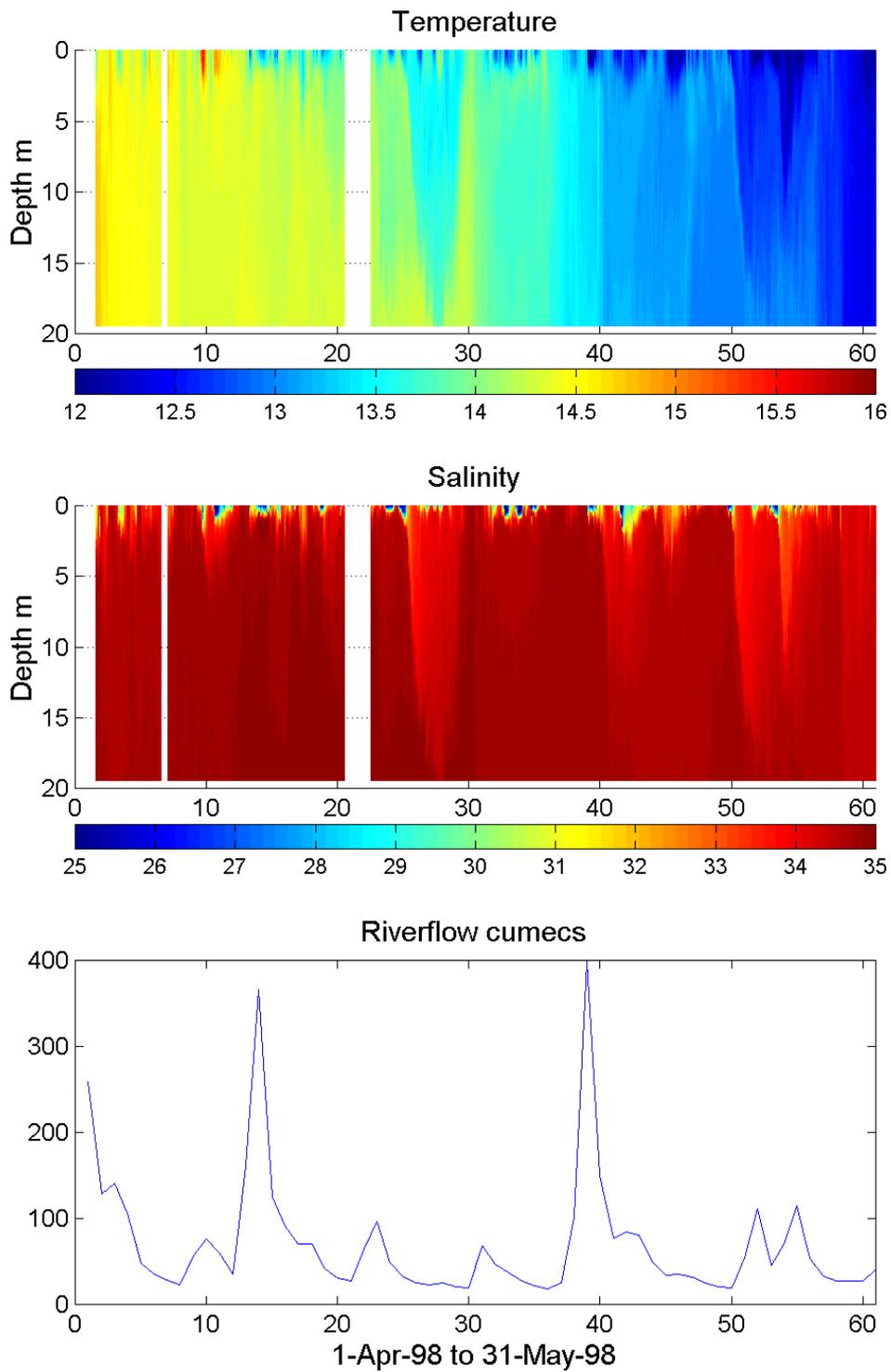


Fig. 5.13 Time-depth sections of temperature and salinity at Killala Bay, and Huon River flow gauged at Frying Pan Creek, from 01 April to 31 May, 1998.

5.7.3 Vertical structure and phytoplankton

We present and discuss a series of snapshots of changes in the vertical distribution of temperature, salinity and phytoplankton biomass (chlorophyll fluorescence) over periods of hours to days, corresponding to different seasons and phytoplankton community structure.

Spring diatom blooms

At Killala Bay in early November 1997, increased levels of fluorescence were associated with increased stratification and surface warming on days 5 and 6 (5-6 Nov) and days 9 to 12 (9-12 Nov) (Fig. 5.14). This elevated fluorescence was not confined to the surface layer, but distributed throughout most of the water column, and coincided with the start of blooms of the diatoms *Chaetoceros* spp. and *Pseudonitzschia* spp. (Fig. 5.6).

Early summer dinoflagellate bloom

By mid-December 1997, a bloom of the toxic dinoflagellate *Gymnodinium catenatum* was firmly established in the Huon Estuary (see section 5.8). There is a striking signature of this bloom in the vertical fluorescence distribution, illustrated in Fig. 5.15 for the period January 1 to 5, 1998. The data show a strong subsurface fluorescence maximum, whose depth changes in a regular diurnal pattern from 2 to 3 m in daylight to near-bottom (about 18 m) at night. During this period there is also a strong diurnal oscillation in temperature and surface salinity which is likely driven by tide and/or wind. However, the amplitude of the depth excursions in fluorescence far exceed those in temperature and salinity, and it is clear that the fluorescence reflects strong vertical migration of *G. catenatum*. Doblin et al.'s (2000b) studies of vertical migration of Huon Estuary strains of *G. catenatum* in the laboratory demonstrated they migrate across salinity barriers to access deep nutrients.

The 1997 early summer bloom of *G. catenatum* ended in mid-January '98. Unfortunately, there is not a continuous data record from Killala Bay throughout the bloom period or from the fluorometer at Hideaway Bay. However, we do have some intriguing data from Killala Bay in late November 1997, which may reflect bloom initiation or advection of *G. catenatum* to the bay (Fig. 5.16). A weak deep chlorophyll maximum is evident over the first 6 days of this record. This maximum gradually shallows from day 4 to 6, and increases substantially in magnitude on day 7 during a period of increased stratification. Increased biomass persists over the next 3 days, and shows evidence on the last day of the organized diel migration pattern characteristic of *G. catenatum*. This record appears to document the early development of a *G. catenatum* bloom. If so, the increase in fluorescence in bottom waters on day 3 could indicate a contribution of *G. catenatum* resting cysts to bloom initiation, either through local re-suspension or advection across the estuary from Brabazon Park / Wheatleys (see section 5.8.4).

We do not have a complete record from either Killala Bay or Hideaway Bay coinciding with the end of the summer dinoflagellate bloom. Surface records of temperature and salinity from the Killala Bay profiler at this time show a rapid decrease in temperature and increase in salinity (evidence of deep mixing) coinciding with strong winds (measured at Cape Bruny) from 15 January (see section 5.8.4). The temperature and salinity record from Hideaway Bay from 25 January to 5 February 1998 (Fig. 5.17) also show evidence of destratification on day 9 (2 February).

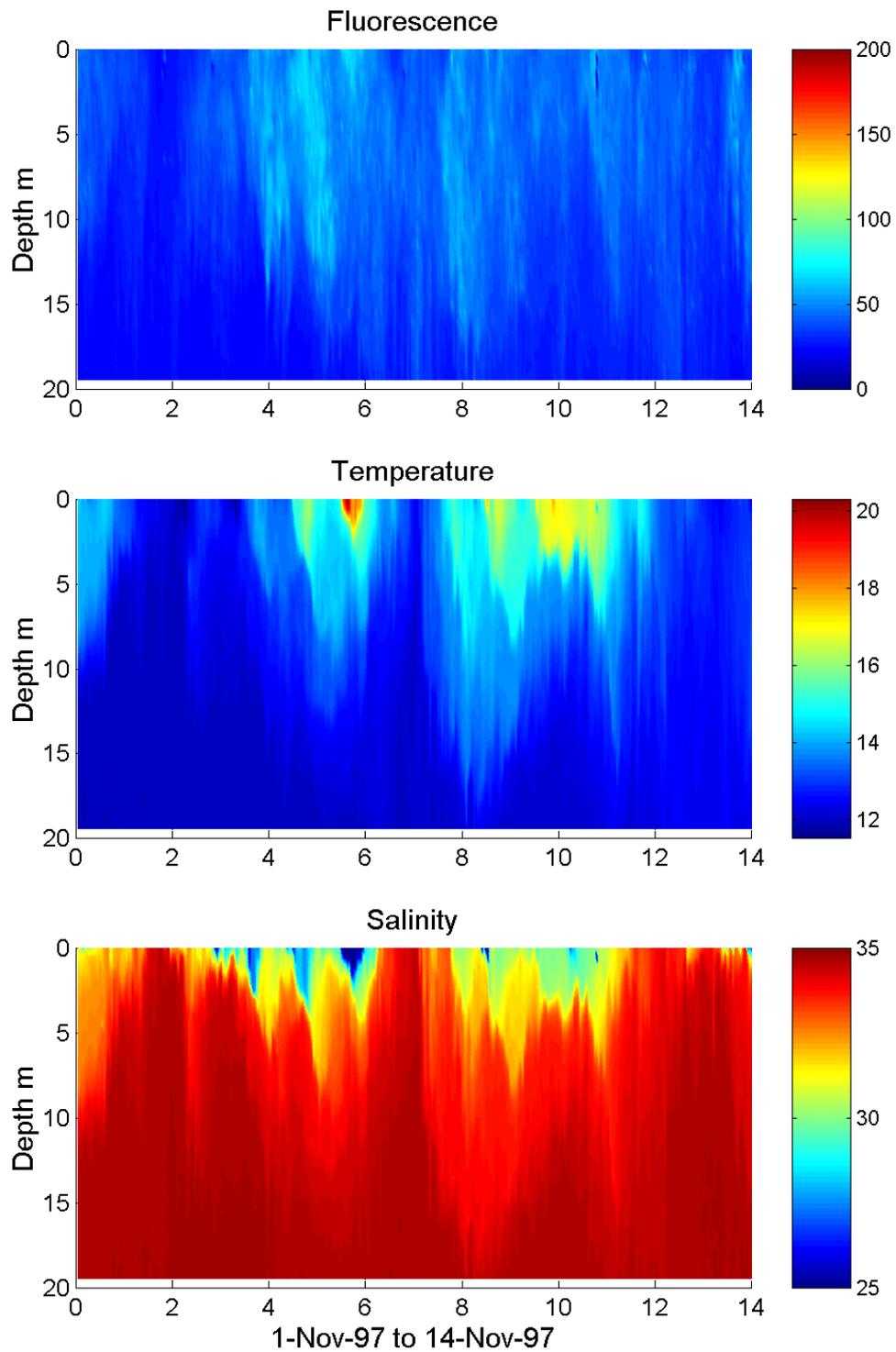


Fig. 5.14 Time-depth sections of fluorescence, temperature and salinity at Killala Bay, 01 to 14 November, 1997.

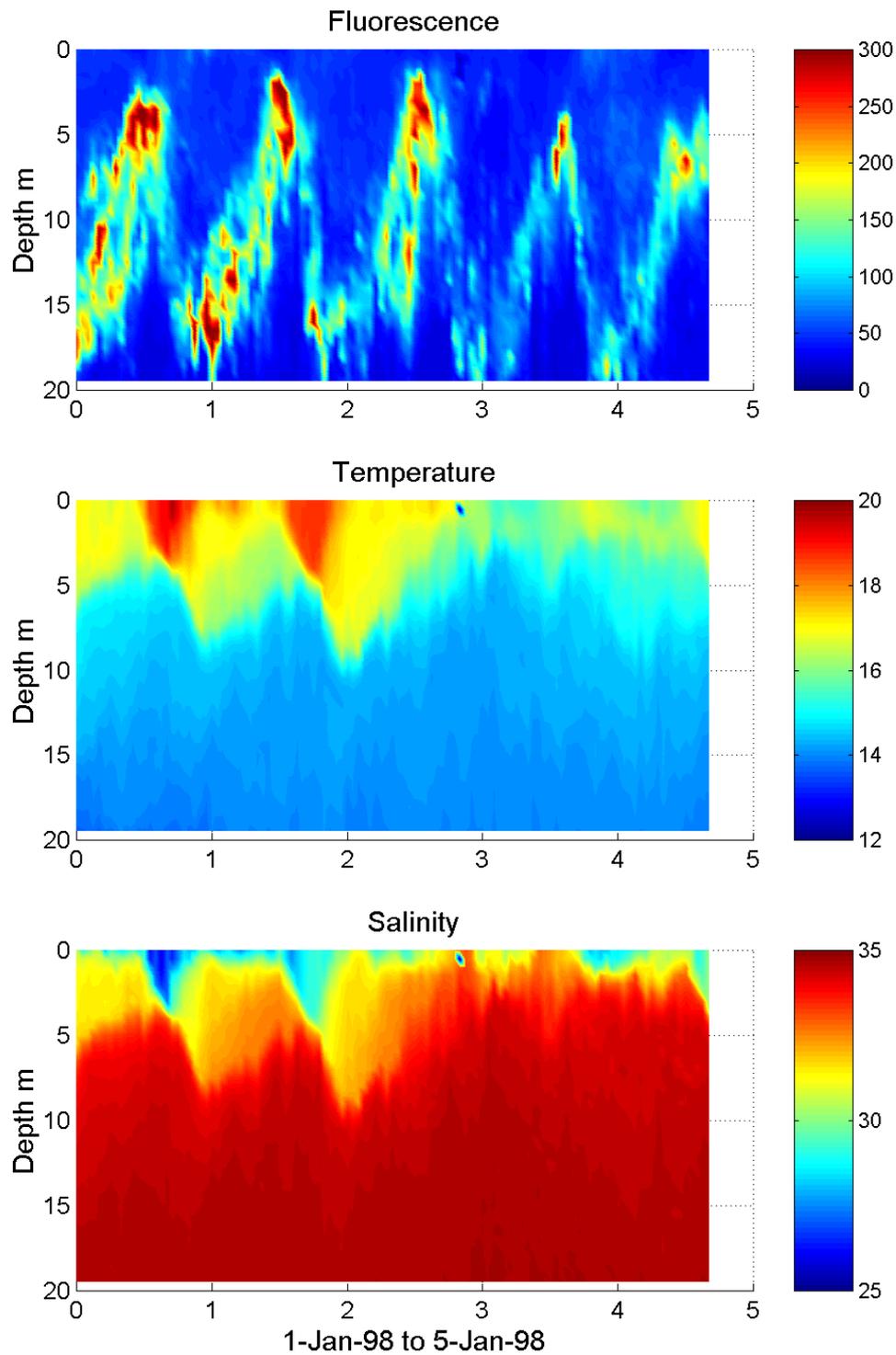


Fig. 5.15 Time-depth sections of fluorescence, temperature and salinity at Killala Bay, 01 to 05 January, 1998.

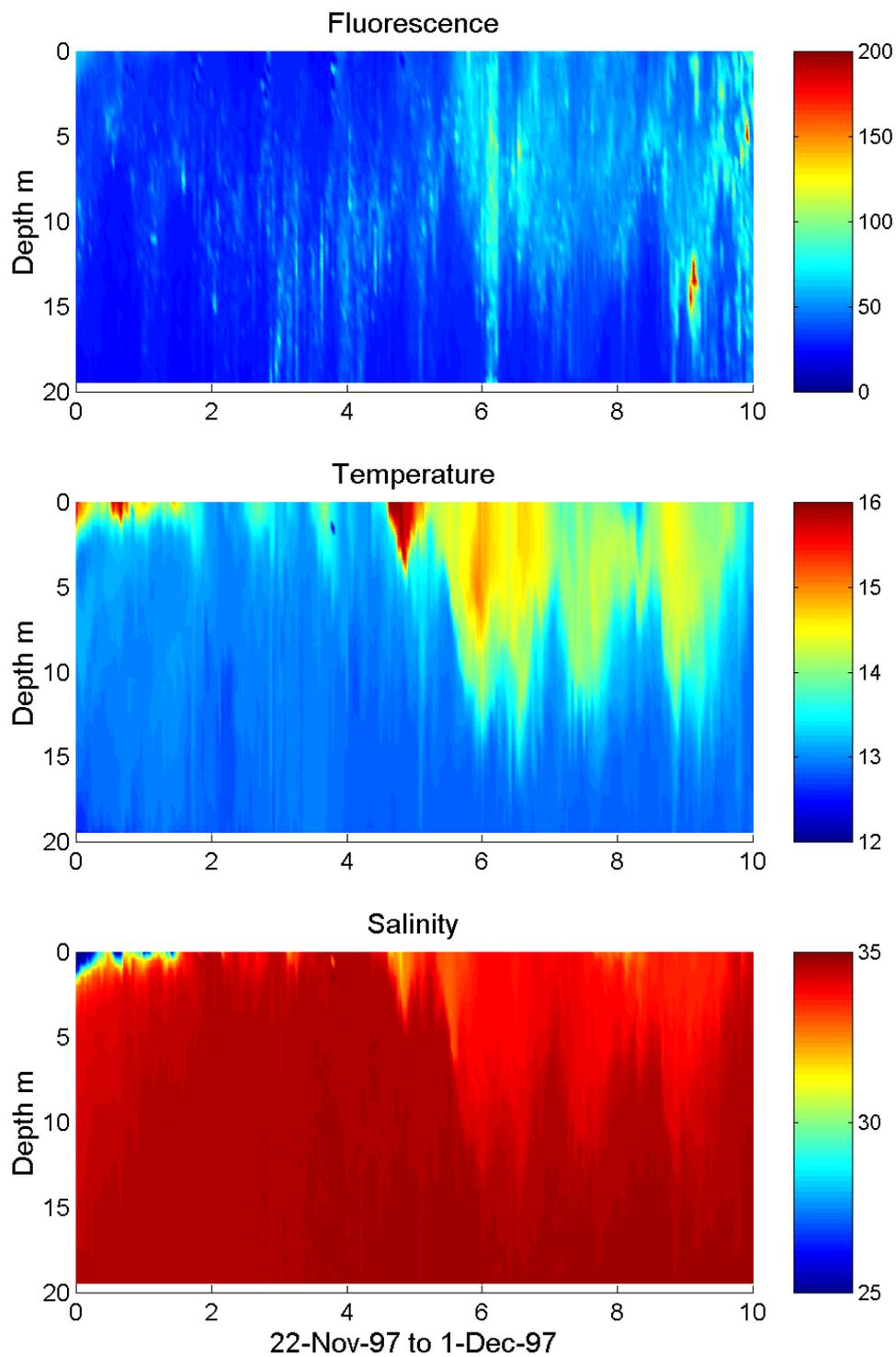


Fig. 5.16 Time-depth sections of fluorescence, temperature and salinity at Killala Bay, 22 November to 01 December, 1997.

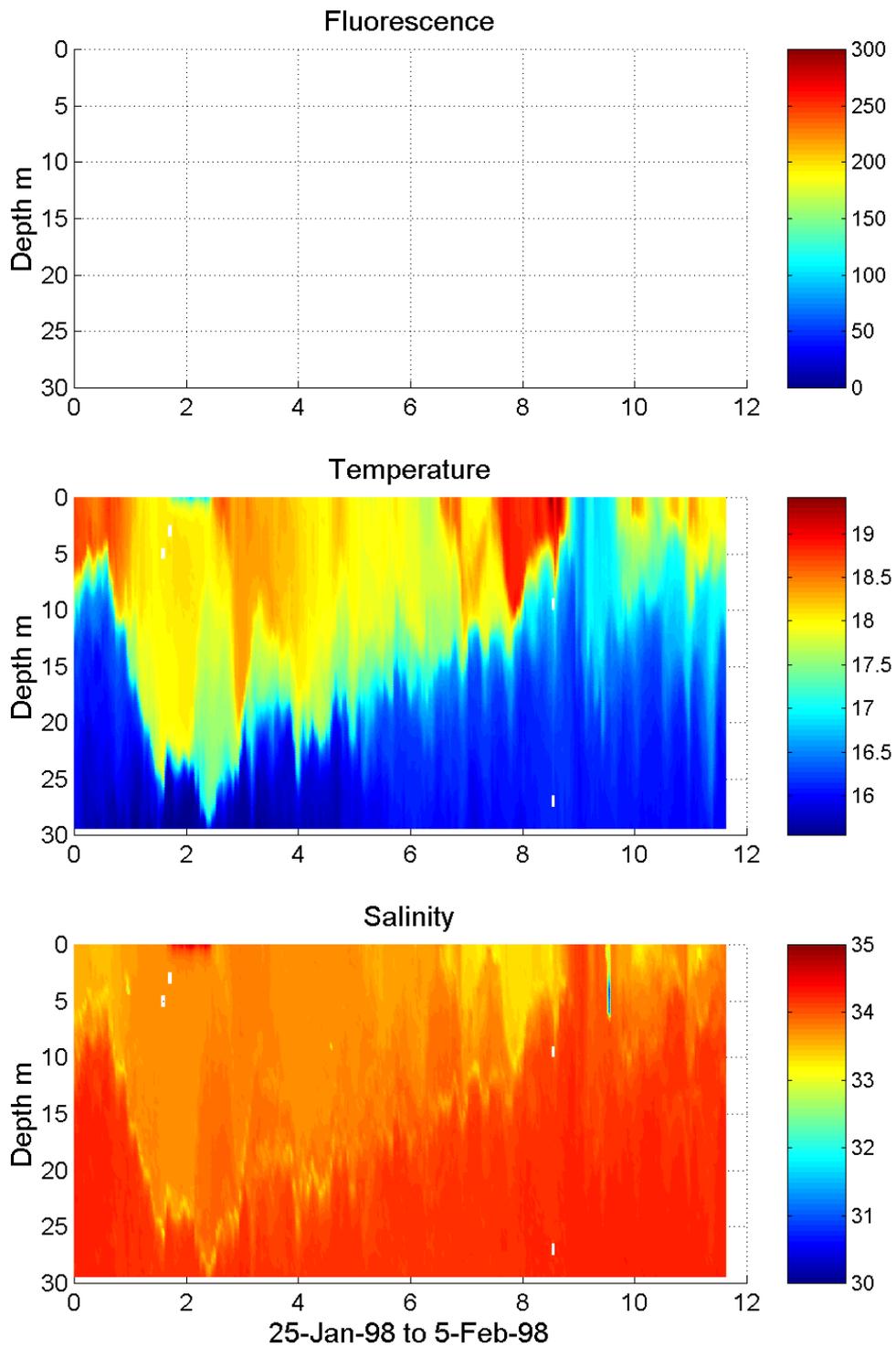


Fig. 5.17 Time-depth sections of temperature and salinity at Hideaway Bay, 25 January to 05 February, 1998. (The fluorometer sensor was disabled.).

Summer dinoflagellate and diatom blooms

The decline of the *G. catenatum* bloom in mid-January was followed by a second period when *Ceratium* spp. dominated the phytoplankton community, preceding or concurrent with (depending on the location) the diatoms *Chaetoceros* spp. followed by *Pseudonitzschia* spp. (see Fig.5.6). From 7 to 16 February, the records from both Killala Bay (Fig. 5.18) and Hideaway Bay (Fig. 5.19) show episodes of destratification with weak vertical temperature and salinity gradients. There is evidence of a weak, deep fluorescence maximum at Hideaway Bay which is located below 10 m throughout the first 7 days of the record, but moves to the surface layer following destratification of temperature and salinity on day 8. There is also a peculiar feature in the Killala Bay fluorescence record: very intense chlorophyll patches or layers through the water column. This is during a mixing event that also coincides with a period of rapid change in the *G. catenatum* biomass (cell volumes of 8.5 mL L^{-1} on 7 February, compared with $<1 \text{ mL L}^{-1}$ on 14 February). The intense chlorophyll patches could represent “clouds” or “flocs” of *G. catenatum* throughout the water column, possibly advected from elsewhere in the estuary (Tassal Ltd., unpublished observations).

The record from Hideaway Bay for 24 March to 6 April (Fig. 5.20) also shows evidence of intermittent destratification of temperature and fluorescence. There is a pronounced cooling event on 3 April, evident throughout the water column. Note that at Hideaway Bay during both February and March/April, the subsurface fluorescence maximum shows a tendency to follow temperature and salinity contours.

Autumn dinoflagellate bloom

The *G. catenatum* bloom was re-established in the estuary in April / May 1998; this is clearly evident in the Killala Bay fluorescence record. The contiguous record for a 20-day period in May shows a remarkably persistent vertical migration throughout the water column (Fig. 5.21). Surface nitrate concentrations are high during May, so that it appears unlikely *G. catenatum* is migrating to bottom waters at night in search of nitrogen. It could be migrating to bottom waters seeking ammonia, but ammonia concentrations in bottom waters are low in May (Fig. 5.10a). Alternatively, it could indicate limitation by some micronutrient whose concentration is elevated in bottom waters.

Winter conditions

There is persistent stratification and surface cooling in July/August'98 at Hideaway Bay (Fig. 5.22) and evidence of some bloom development associated with a runoff event in early August. Again, the fluorescence distribution shows strong associations with temperature and salinity contours. Fluorescence values at Killala Bay in July are very low, with little vertical structure, showing phytoplankton growth is limited, probably by both low temperature and light.

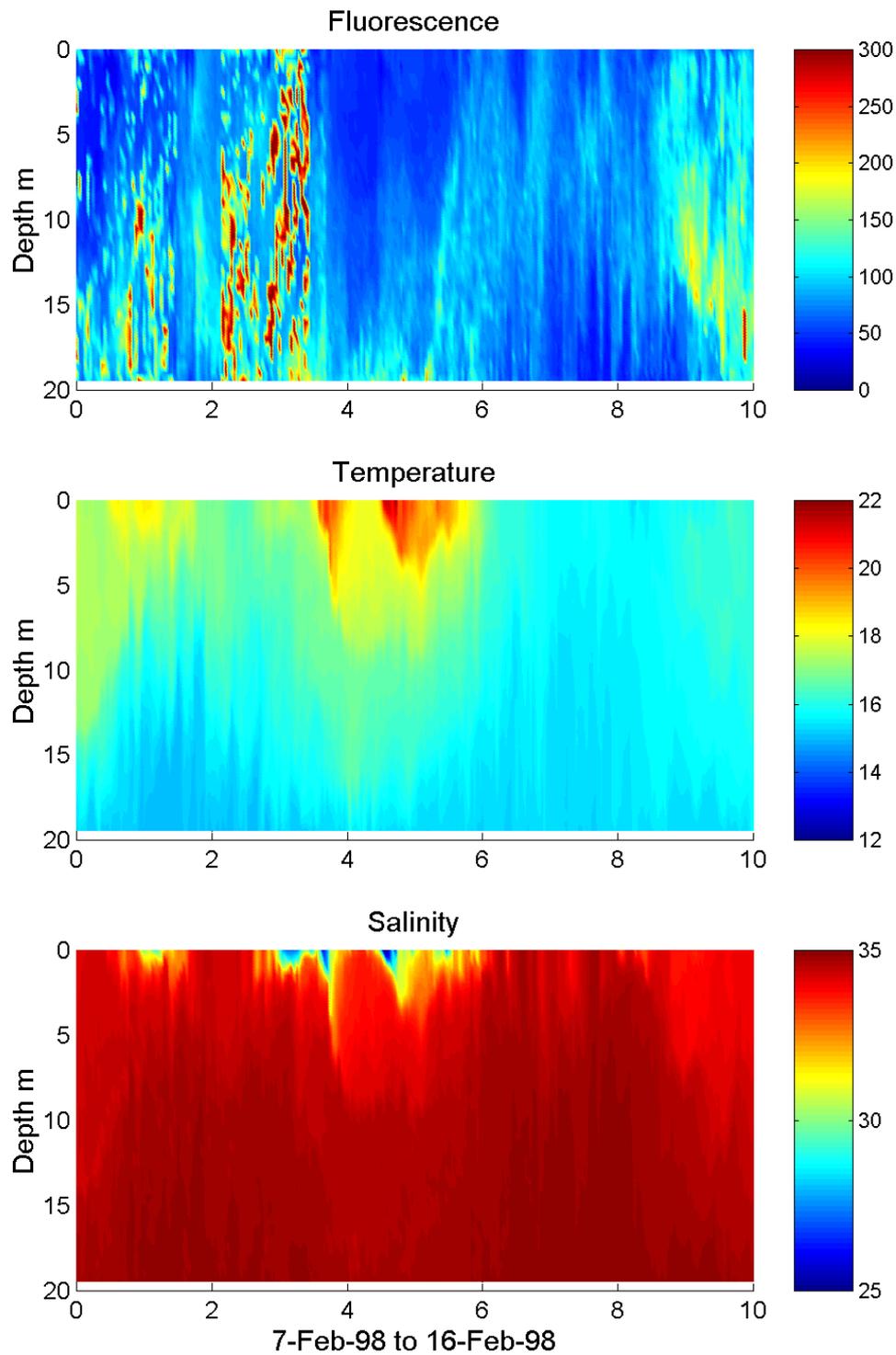


Fig. 5.18 Time-depth sections of fluorescence, temperature and salinity at Killala Bay, 07 to 16 February, 1998.

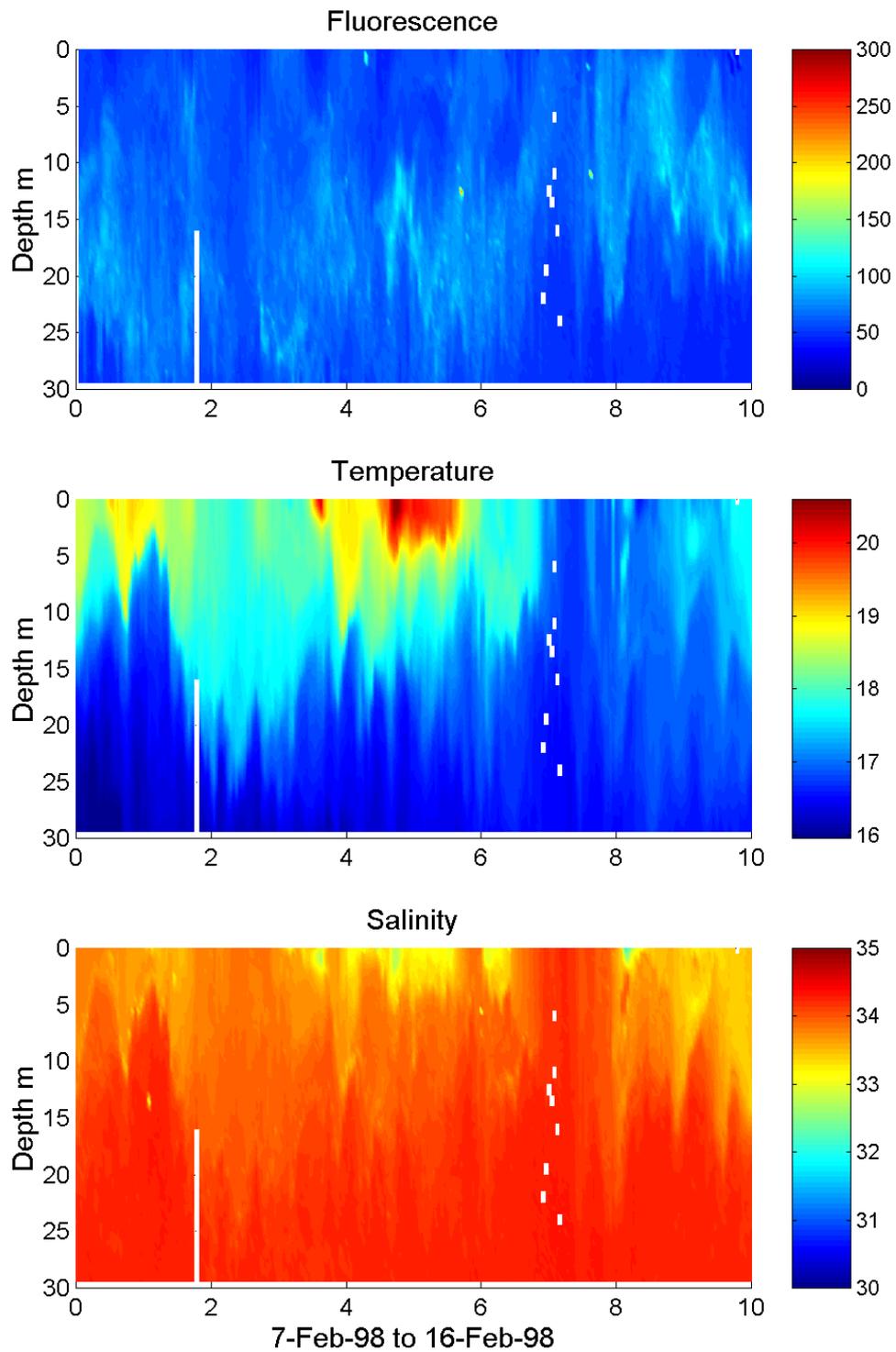


Fig. 5.19 Time-depth sections of fluorescence, temperature and salinity at Hideaway Bay, 07 to 16 February, 1998.

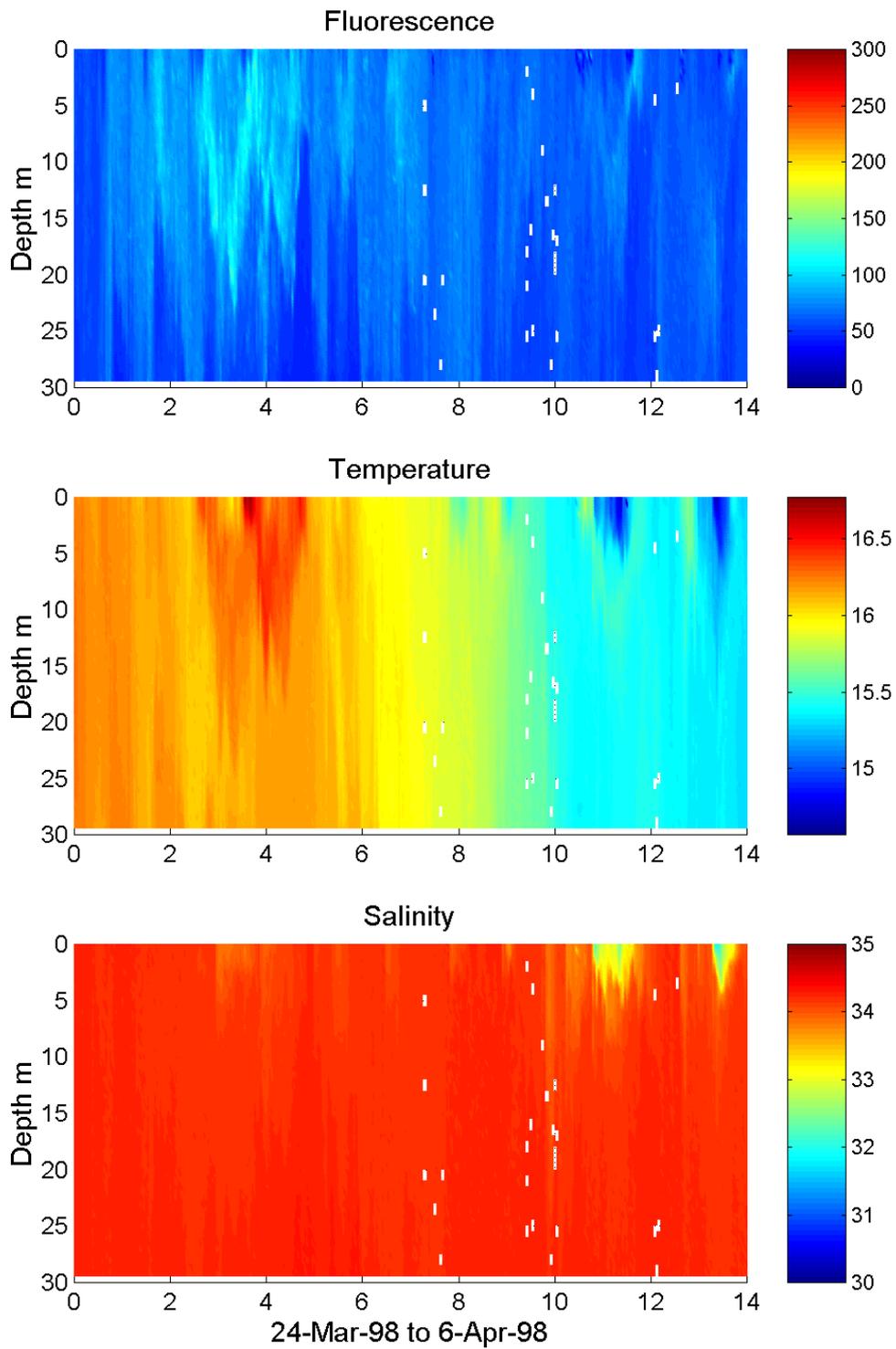


Fig. 5.20 Time-depth sections of fluorescence, temperature and salinity at Hideaway Bay, 24 March to 06 April, 1998.

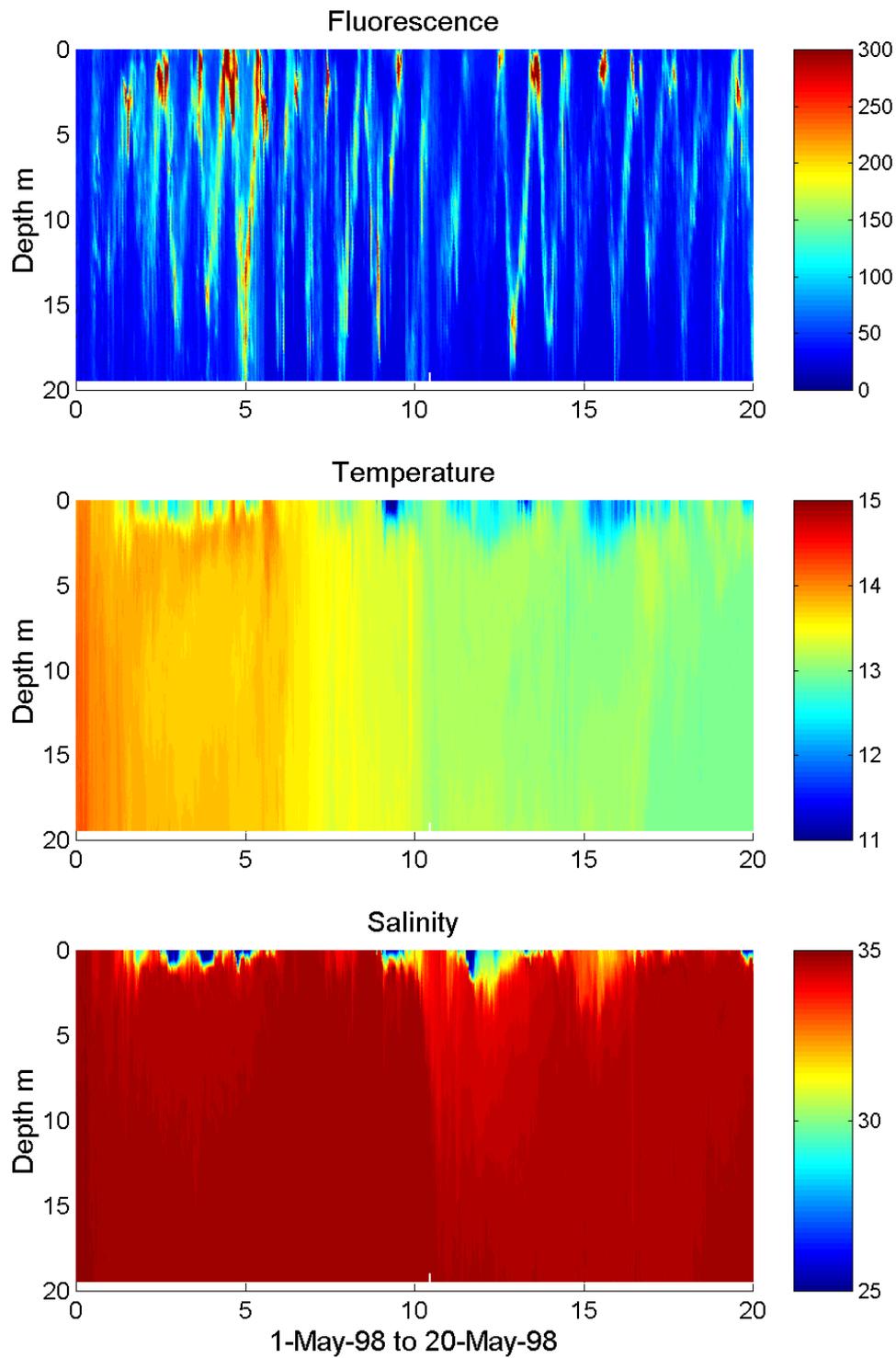


Fig. 5.21 Time-depth sections of fluorescence, temperature and salinity at Killala Bay, 01 to 20 May, 1998.

5.8 Bloom Dynamics of the Toxic Dinoflagellate *Gymnodinium catenatum*

The toxic dinoflagellate *Gymnodinium catenatum* was first recorded from south-east Tasmania in 1985 (Hallegraeff and Sumner 1986) but is believed to have been introduced into the region in the 1970s (McMinn et al. 1997). Since then it has formed toxic blooms in many years causing significant economic losses to the shellfish industry in the Huon Estuary and adjacent waters. A bloom of *G. catenatum* is defined as greater than 10,000 cells/L which is the level at which shellfish tend to become toxic (Hallegraeff et al. 1995). *Gymnodinium catenatum* produces the neurotoxin saxitoxin and its derivatives (PSTs) as part of its normal metabolism. When shellfish eat *G. catenatum*, these toxins accumulate in the tissues of the animal, reaching levels that are harmful to humans and can result in paralytic shellfish poisoning. Since 1986 the Tasmanian Government has operated a biotoxin monitoring program according to US FDA guidelines. These guidelines stipulated closure of shellfish farms to harvesting when levels of toxin exceed 80 µg saxitoxin equivalents per 100 g mouse tissue. This has resulted in closures of shellfish farms in the Huon Estuary for periods of up to 9 months.

Because of the importance of *G. catenatum* blooms in the ecology and economy of the Huon Estuary, the blooms during this study were closely examined. In addition to cell densities and pigment concentrations at the five biological stations, the life history was investigated to investigate the role of resting cysts in bloom dynamics. These characteristics were related to the physical and chemical parameters associated with these blooms in order to gain a greater understanding of the factors regulating *G. catenatum* blooms in the estuary.

5.8.1 Description of the blooms

The last major *G. catenatum* bloom before this study was in 1993, followed by a small bloom in 1994 (Hallegraeff et al. 1995). There were no blooms or records of toxicity after this for three years until late November 1997. In our study, no *G. catenatum* cells were observed in the water column between July 1996 and March 1997 (see Fig. 5.23). In April 1997 *G. catenatum* cells were recorded in the water column in low numbers. Cell numbers increased during October and November 1997, and by early December 1997 cell densities were up to 5.5×10^5 cells L⁻¹ (Fig. 5.23), with mussel toxicity levels in Port Cygnet of 140 saxitoxin equivalents per 100 g mouse tissue (Fig. 5.24; toxin data courtesy of R. Brown, Tasmanian Government). Shellfish farms in Port Cygnet were closed at the end of November 1997 and re-opened in mid February 1998. This summer bloom declined in mid January 1998, but was followed by a second bloom in the autumn that began in mid-April and continued through until late June 1998. Cell numbers were not as high as in the summer bloom although the maximum was similar (4.0×10^5 cells L⁻¹). However, toxicity levels in mussels in Deep Bay, Port Cygnet were higher than in the summer bloom (Fig. 5.24). Monitoring after this study (N. Parker and J. Skerratt, unpublished data) recorded two more blooms, which both had lower cell densities than the 1997/98 blooms; the first was in the summer of 1998/99 and the second in the autumn of 1999.

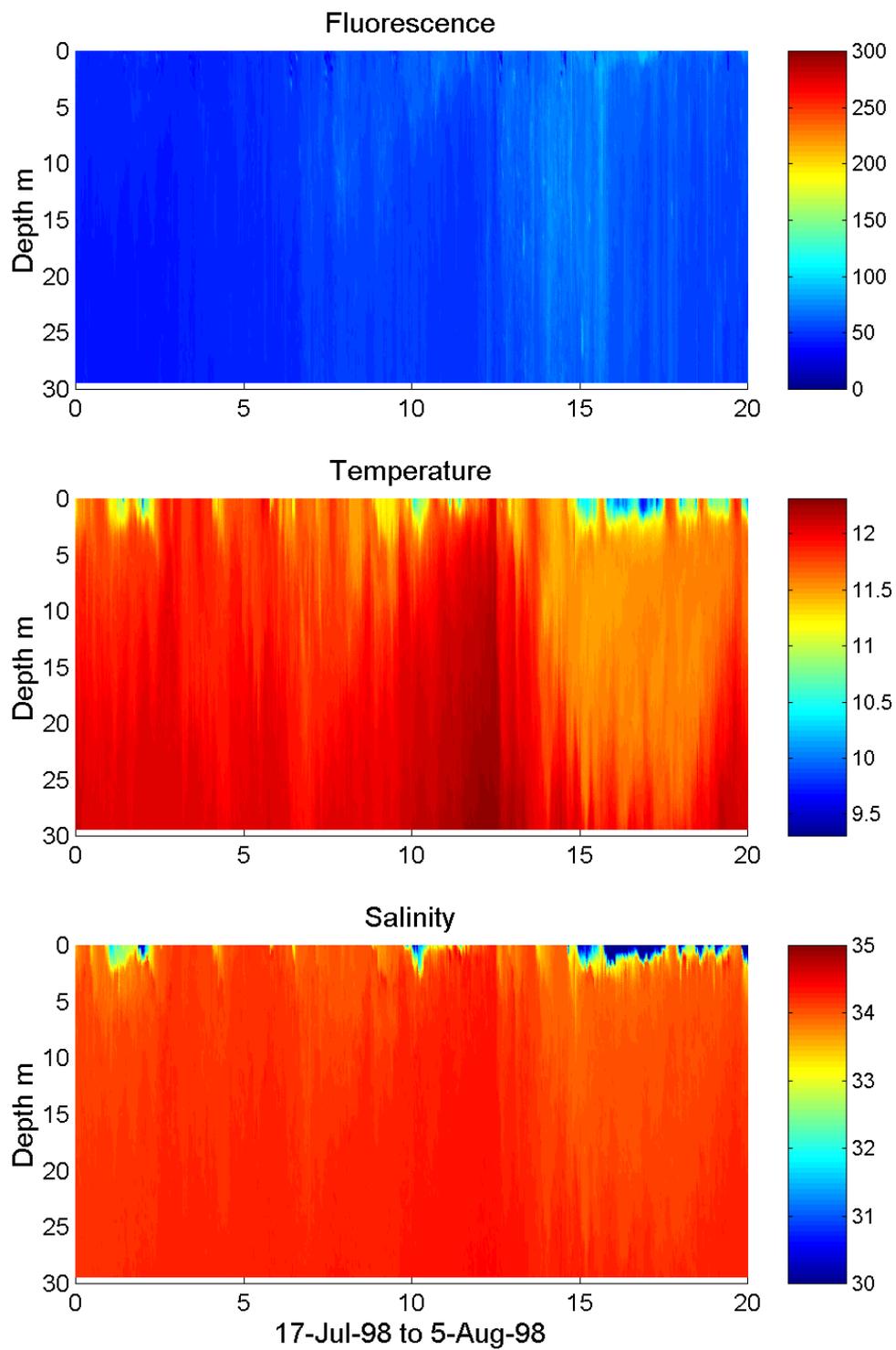


Fig. 5.22 Time-depth sections of fluorescence, temperature and salinity at Hideaway Bay, 17 July to 05 August, 1998.

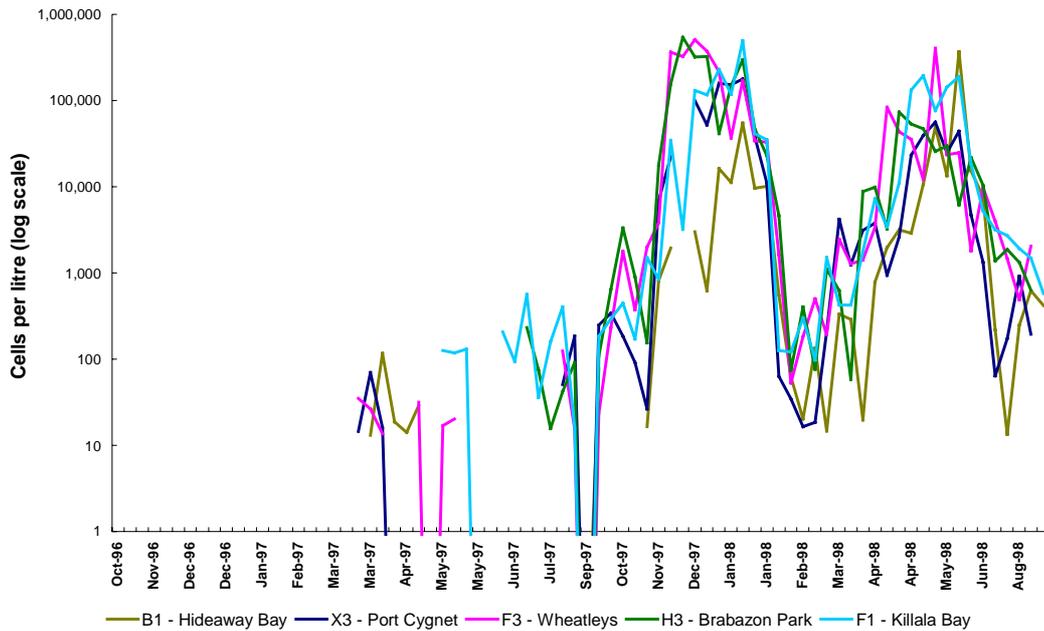


Fig. 5.23 Cell counts of *G. catenatum* during the Huon Estuary Study at the 5 biological stations (log scale).

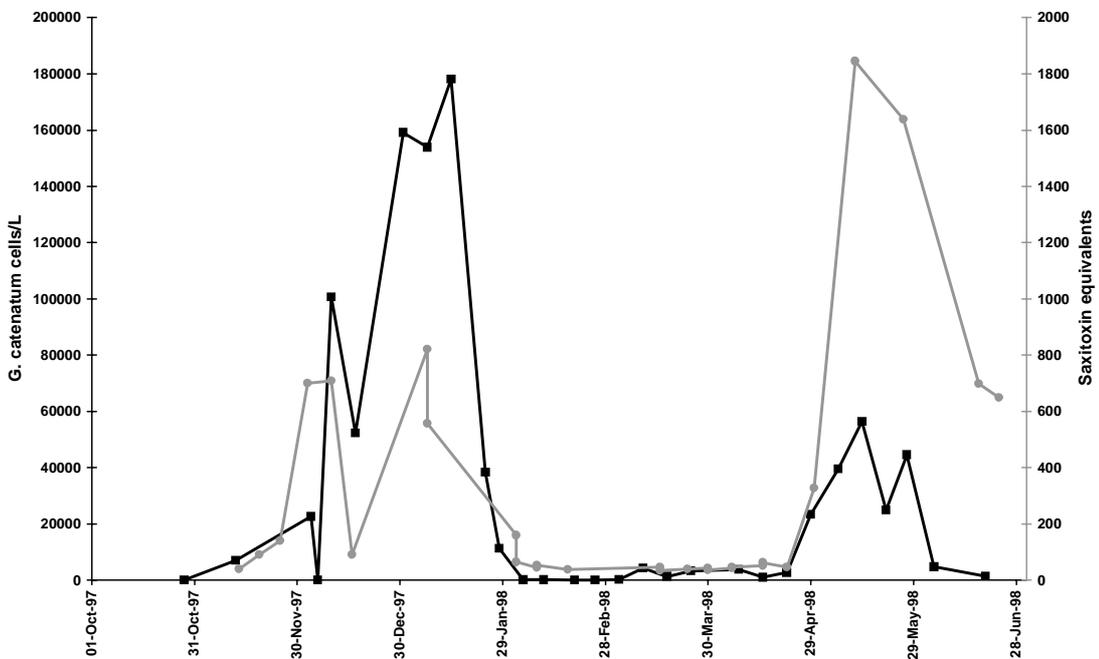


Fig. 5.24 Cell counts of *G. catenatum* (black line) at Deep Bay, Port Cygnet (X3B) compared with shellfish toxin concentrations (grey line) from the same area (toxin data courtesy of R. Brown; Dept. of Health and Human services).

5.8.2 Physical and chemical characteristics of the Huon Estuary during blooms

Figure 5.25 shows fluorescence, salinity and temperature midday profiles from the Killala Bay automatic profiler (see Section 5.7). Figure 5.25a indicates that, at the time of initiation of the first bloom in 1997 the water column at Killala Bay was weakly stratified, with a surface temperature of around 14°C and surface salinity of 33.5 increasing to 35 at depth. This pattern is consistent with the elevated river runoff in October and November (Fig. 5.11). The fluorescence trace indicates an increase in algal biomass with bloom development. There were no extended periods of low winds recorded at Cape Bruny during bloom initiation (Fig. 5.12). At the peak of the bloom in late December 1997 (Fig. 5.25b), the water column was strongly stratified, with surface water temperatures up to 18°C and a low surface salinity of 28. The fluorescence trace shows the subsurface biomass maximum (consisting almost entirely of *G. catenatum* based on microscopic counts and cell volume, Fig. 5.6). The decline of this bloom in mid-January was associated with complete destratification of the water column and surface cooling (Fig. 5.25c), which coincided with strong winds that may have caused localised upwelling (Fig. 5.12). By late January a well-mixed water column was present, with low fluorescence levels throughout (Fig. 5.25d). These data indicate that a partially stratified water column was present during bloom formation and that temperatures of greater than 14°C supported this. The bloom persisted through a variety of salinity and temperature conditions, although the dramatic change in water-column conditions, that occurred in mid January, coincided with bloom collapse.

Conditions during initiation of the autumn bloom were very different to those in the summer bloom. There was some runoff due to rainfall events (Fig. 5.11), but these had little effect on the water column, which was weakly stratified (Figs. 5.25g and h). Water temperatures at the peak of the bloom were just over 12 °C (Fig. 5.25i), having decreased from 14 °C at the time of bloom initiation. Winds were relatively high during this period (ranging from 5-9 m s⁻¹). Bloom decline was not associated with any significant changes in the water column that remained well mixed (Fig. 5.25j), but was probably influenced by decreasing water temperatures coinciding with decreased light availability.

Nutrient conditions in the estuary during the two blooms also differed. Surface macronutrients during the summer bloom were relatively low at the time of bloom initiation with surface nitrate and ammonia of less than 1 µM. Deep nitrate was also low (1-2 µM) although bottom ammonia values were high (about 1µM). Bottom nutrients became increasingly depleted during the bloom, presumably due to vertical migration of *G. catenatum* (Fig. 5.15) allowing it to access these deep nutrients. Because of the dynamic nature of the estuary, nitrogen is replenished both from sediments and from the marine end. Effective uptake of this renewable nutrient supply would result in the low values we recorded. The autumn bloom was also initiated at relatively low surface-nutrient concentrations but higher than the summer bloom (approximately 2 µM) and there was no complete stripping of nitrate from the water column; elevated nutrients were always present at depth, relative to the surface values.

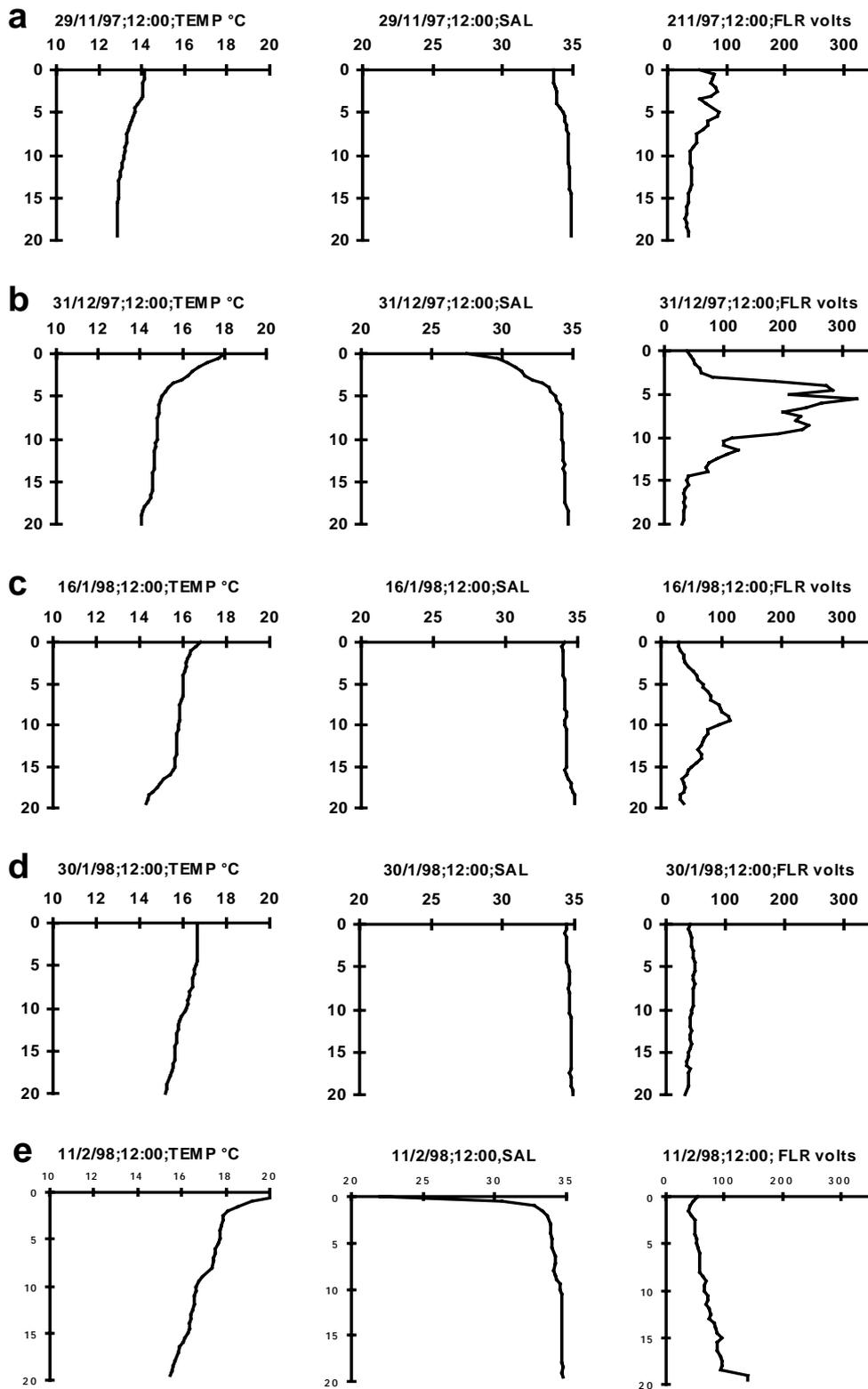


Fig. 5.25a-e Midday water column characteristics at Killala Bay during 1997/98. TEMP = temperature in °C; SAL = salinity; FLR = fluorescence in volts. Y-axis is depth in metres. a – commencement of Summer 97/98 *G. catenatum* bloom; b – peak of Summer 97/98 bloom; c – decline of Summer 97/98 bloom; d – end of Summer 97/98 bloom; e – diatoms and *Ceratium* spp. dominant, February 1998.

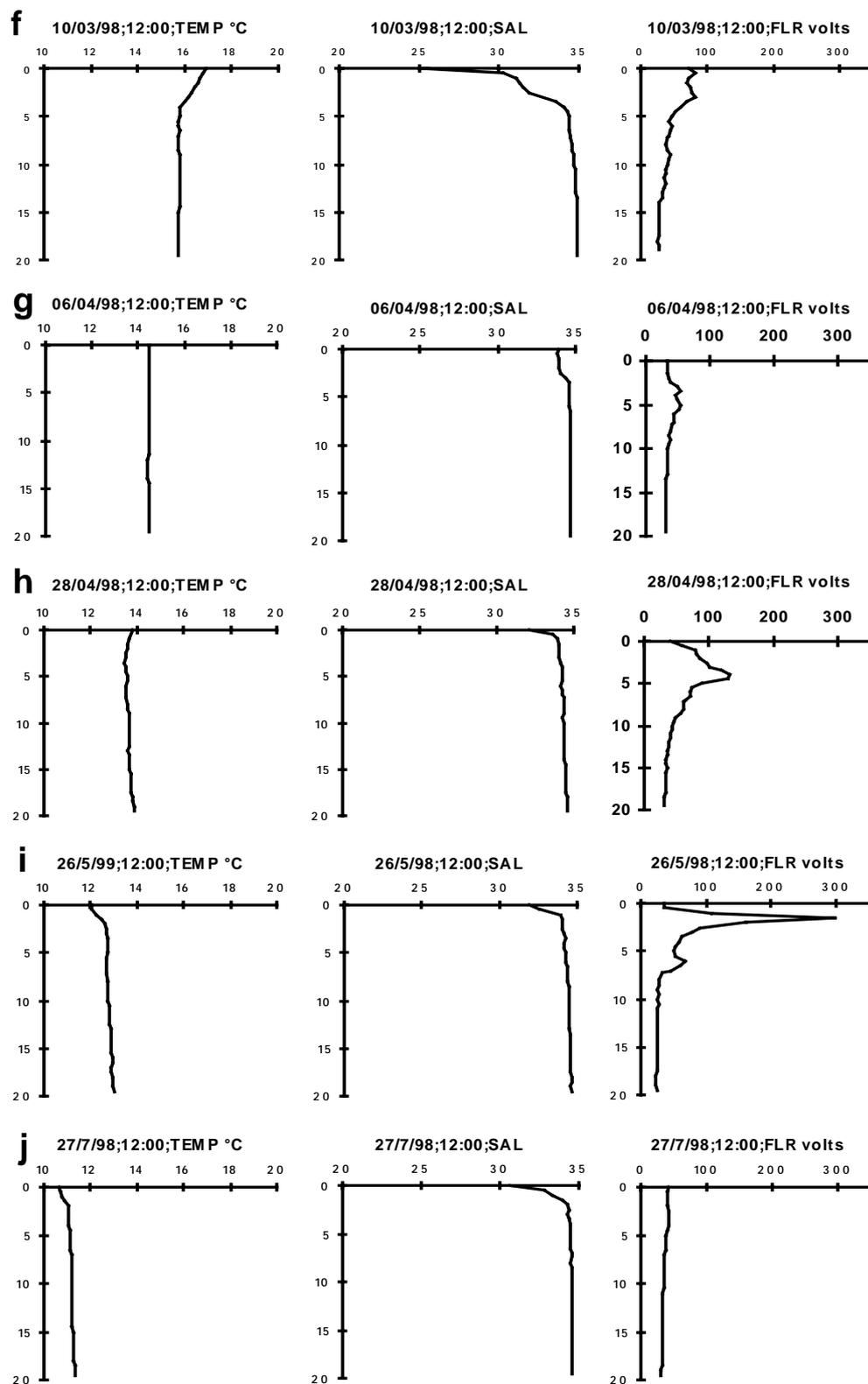


Fig. 5.25f-j Midday water column characteristics at Killala Bay during 1997/98. TEMP = temperature in °C; SAL = salinity; FLR = fluorescence in volts. Y axis is depth in metres. f – diatoms dominant; g – mixed dinoflagellates with increasing *G. catenatum* abundance; h – mixed dinoflagellates and initiation of Autumn *G. catenatum* bloom; i – Peak of Autumn bloom; j – low biomass, mixture of dinoflagellates and diatoms.

5.8.3 *Gymnodinium catenatum* resting cyst dynamics

Gymnodinium catenatum has a very characteristic appearance with its chains of up to 64 cells long (Blackburn et al. 1989). Like most microalgae, *G. catenatum* divides by vegetative cell division (i.e. one cell dividing into two, two into four etc.). This can give an exponential increase in cell numbers when environmental conditions are appropriate for growth. However, *G. catenatum* can also reproduce sexually: gametes fuse to produce a large motile zygote stage or planozygote which in turn forms a non-motile, resistant resting cyst or hypnozygote. On germination this produces another large, motile zygote stage or planomeiocyte which undergoes a reduction nuclear division (meiosis) to produce vegetative cells (for further details see Blackburn et al. 1989).

The resting stage is important, as it is a dormant stage and can survive conditions unsuitable for the vegetative population. For some dinoflagellates, the importance of resting cysts as 'seeds' for the development of blooms has been demonstrated. They are also potential agents for dispersal of populations into new regions, direct sources of toxicity, and potentially play a role in bloom decline by removing cells from the water column (Wall 1971; Anderson 1984). We examined sexual stages in the context of the bloom dynamics for the summer and autumn blooms of 1997/98 as well as the summer and autumn blooms of 1998/99 (N. Parker, unpublished Ph.D. studies).

The four blooms (summer and autumn 1997/98, summer and autumn 1998/99) showed dramatic differences in resting cyst dynamics. During the summer 1997/98 bloom, samples from Killala Bay, Wheatleys, Port Cygnet and Brabazon Park contained sexual stages (including resting cysts, planozygotes and fusing gametes) which demonstrated sexual reproduction in the water column from the time of first detection of the bloom in late November. The concentration of these sexual stages fluctuated throughout the bloom (Fig. 5.26a). It was not possible to discriminate planozygotes from planomeiocytes and thus we are unable to determine whether some of the large 'sexual cells' seen in samples were the result of germination of resting cysts from the sediments (planomeiocytes) rather than resting cyst production in the water column (planozygotes). However, we think the former is unlikely because the sampling strategy was inappropriate for detecting planomeiocytes (which would emerge near the sediment/water interface) and also because the large sexual cells co-occurred with fusing gametes and young resting cysts. Constant resting cyst formation throughout bloom development and decline has not been previously reported for *G. catenatum*. A similar pattern was seen in the summer bloom of 1998/99 with low densities of sexual stages observed early in the bloom and continuing to form throughout the bloom period (Fig. 5.26c). In contrast to the summer blooms, no sexual stages were seen at any of the sites in the autumn 1998 bloom in either integrated water column samples or net tows (April/May 1998, Fig. 5.26b). This was also the case in the autumn bloom of 1999.

Dinoflagellate resting cysts have been shown to form in response to nutrient depletion in culture suggesting that adverse conditions – notably nutrient depletion – stimulate sexual reproduction. However, there is increasing evidence to suggest that resting cysts form in response to a variety of factors (Anderson et al. 1983, Ellegaard et al. 1998, Kremp and Heiskanen 1999, Sgrosso and Montresor 1999) rather than a simple response to nutrient depletion. This is supported by our observations of *G. catenatum* in the Huon Estuary, where macronutrient concentrations do not appear to initiate sexuality and resting cyst formation.

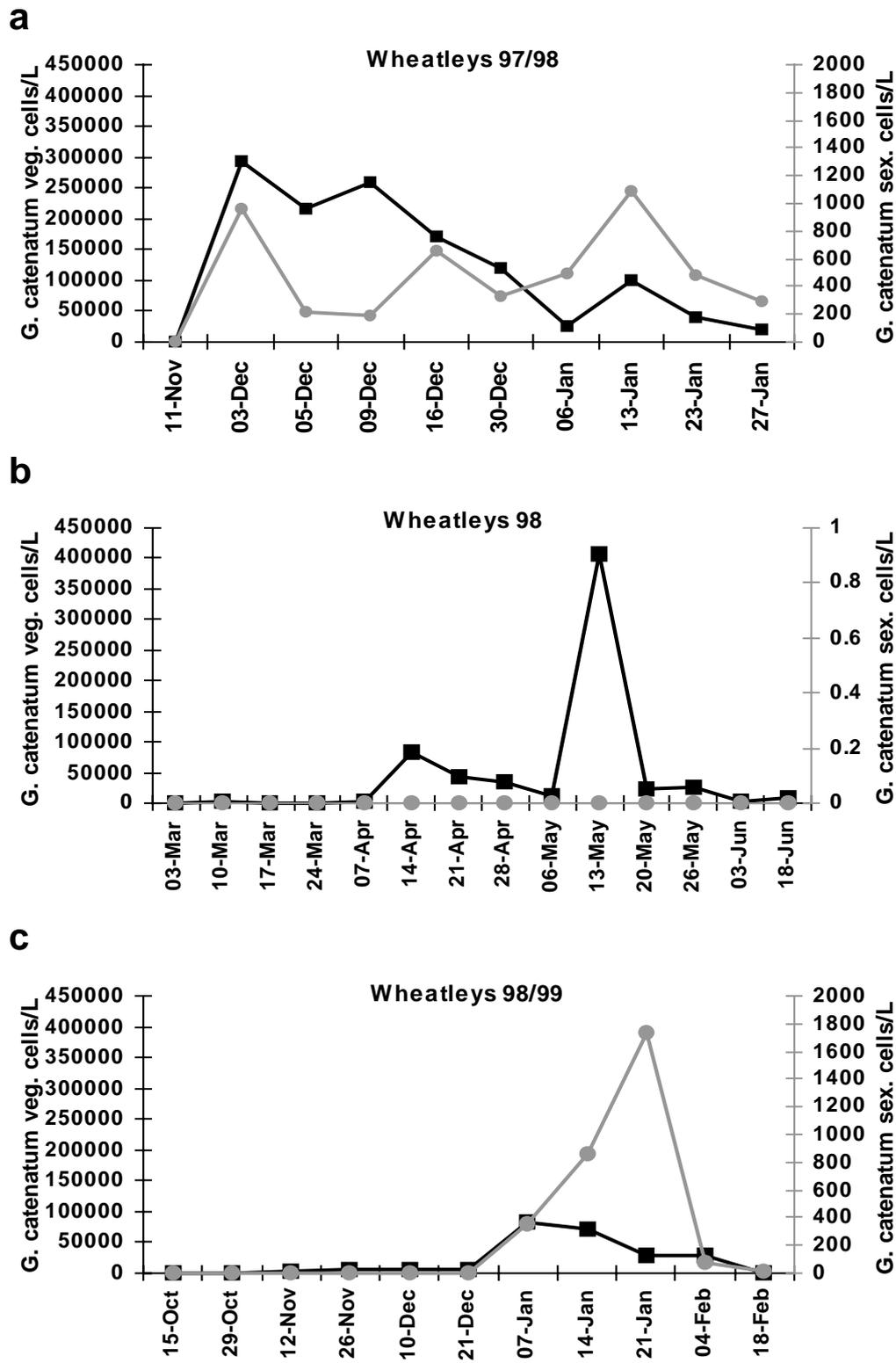


Fig. 5.26 Cell counts of vegetative cells (black line) and sexual stages (grey line) of *G. catenatum* at Wheatleys during 3 blooms a – summer 1997/98; b – autumn 1998; c – summer 1998/99.

Resting cysts form throughout blooms; no significant correlations were found between concentration of sexual stages and all macronutrients analysed (N. Parker, unpublished data). Laboratory experiments have also failed to show a clear encystment response to nutrient depletion (N. Parker unpublished data).

The difference in the resting cyst dynamics between the summer and autumn blooms in the two years could not be explained by key physical parameters. We have shown in laboratory experiments (N. Parker, unpublished data) that the autumn populations were capable of sexual reproduction at the observed cell densities as crosses of strains isolated from both summer and autumn populations had similar levels of sexual compatibility and similarly complex mating systems. We have also found that the lower water temperatures recorded in autumn would not prohibit but may reduce resting cyst formation (N. Parker, unpublished data). This has also been observed in another similar dinoflagellate *Gymnodinium nolleri* (Ellegaard et al. 1998). Future studies will investigate day length as another potential factor as this has been shown to be important in regulating encystment in other dinoflagellates (Sgrosso and Montresor 1999).

5.8.4 Why do *Gymnodinium catenatum* blooms form in the Huon Estuary?

To us, one of the most intriguing findings of the study was the absence of dinoflagellate blooms from spring 1996 through to autumn 1997, whereas dinoflagellate blooms dominated summer/autumn 1997/98. Not only was there an absence of *G. catenatum* blooms, but also of other dinoflagellates (see section 5.5.3 and 5.6, and Figs. 5.6, and 5.7).

The *G. catenatum* blooms investigated during this study occurred under a variety of conditions. During the summer 1997/98 bloom, the water column was more stratified and warmer than in the autumn 1998 bloom. The winds were more variable, with some calm periods of a few days interspersed with strong wind events; during the autumn bloom, the winds were relatively moderate and more persistent. There was river runoff during both blooms, with larger pulses of fresh water during the summer bloom. Nutrient levels tended to be low and well utilised in both blooms.

Hallegraeff et al. (1995) identified key environmental variables that regulate *G. catenatum* blooms and associated shellfish toxicity in southern Tasmanian waters from 1986 to 1994. They examined historical data of toxicity and hydrological and meteorological data in order to ascertain what stimulates *G. catenatum* bloom formation. Their hypothesis is that *G. catenatum* blooms can only develop within environmental constraints, which include a seasonal temperature window from January to June, with major blooms (as shown by high toxicity) only developing when water temperatures are greater than 14°C; a threshold runoff from the Huon River in the weeks preceding a bloom (greater than 100,000 megalitres flow over three weeks measured in the upper Huon River at Frying Pan Creek - this is related to, but not directly correlated with, rainfall); and a calm stable water column for sustained bloom development as indicated by wind speeds of <5 m s⁻¹ (measured at Cornelian Bay, Derwent Estuary) for five days or more. They also found no correlation between macronutrients and bloom initiation.

The *G. catenatum* blooms observed in the study period did indeed develop at water temperatures of approximately 14°C. However, the autumn blooms were most dense at temperatures a little above 12°C. This is similar to blooms in 1987 and 1990 that Hallegraeff et al. (1995) described as exceptions. It is likely that decreasing growth rates associated with decreasing water temperatures contributed to the decline of these blooms (Blackburn et al. 1989).

Our wind data is from Cape Bruny, a site Hallegraeff et al. (1995) found was not well correlated with *G. catenatum* bloom development in the Huon Estuary. Instead they used wind data from Cornelian Bay, Derwent Estuary as a proxy for water column stability. Our detailed data on water column characteristics show that blooms can develop when water column stratification is weak, while strong stratification appears to enhance bloom intensity. Our data also show that major destratification caused by wind and oceanic events may contribute to bloom collapse.

River runoff not only affects water column stability through stratification, but is also a source of dissolved organic matter (DOM), which stimulates growth of dinoflagellates (Doblin et al. 1999a). The presence of a surface layer of humic rich water during much of the bloom periods may be important for both the dissolved organic matter (DOM) it contains as well as the water stratification it endows. Our data from the automated monitoring systems (section 5.7) probably under-emphasize this humic rich layer, as the freshwater flow would have a greater impact on the bloom further up the estuary from Killala Bay. Also the river flow tends to have a stronger impact on the north-eastern side of the estuary on the opposite side from Killala Bay and Hideaway Bay (personal communication, section 5.7.2). The relationship between the growth of *G. catenatum* and DOM is complicated. Doblin et al. (1999a, b, 2000a) demonstrated the interplay between DOM and macronutrient and micronutrient supply, including the important micronutrient selenium. In addition they showed that nutrient uptake was associated with vertical migration in laboratory cultures (Doblin et al. 2000b). While macronutrients do not appear to play a major role in the regulation of blooms (see Figs. 5.9 and 5.10), the ability of *G. catenatum* to fully utilise the macronutrients may be affected by river runoff.

Our study generally supports Hallegraeff et al.'s (1995) key environmental variables for *G. catenatum* bloom development, but we consider that low temperature autumn blooms may not be exceptions, and the January to June window is clearly not exclusive. We also consider the relationship between wind data as used by Hallegraeff et al. (1995) and water column stability needs to be better defined, and their relative importance better elucidated. Water column characteristics obviously give a direct and accurate picture of water column stability. The relationship between wind data and water column stability leading to bloom initiation is complex: wind direction, duration and fetch must all be considered. Strong winds may also resuspend sediments from shallower regions of the estuary, enhancing resting cyst germination by increasing the oxygen levels they are exposed to (anaerobic conditions have been found to prohibit germination of resting cysts; Anderson et al. 1987). We consider detailed analysis of wind data collected at key points within the estuary is necessary to elucidate the role of winds in bloom dynamics.

We can identify the environmental conditions that promote bloom initiation and development – temperatures of 12°C and over; weak to strong water column stratification caused by a humic-rich, low-salinity surface layer and, in some cases, low wind stress. However, neither our

findings, nor those of Hallegraeff et al. (1995) are able to determine what factors or conditions prohibit *G. catenatum* bloom development as occurred in 1995, 1996 and 1997. The capacity for strong vertical migration is probably one key to the success of dinoflagellates in the Huon Estuary. In section 10.4.3, adding vertical migration into the biomass model allows significant increase in the biomass of the estuary. In particular, the relatively high swimming velocity of *G. catenatum* compared with other dinoflagellates should be beneficial to this species. Swimming velocities in dinoflagellates range between 0.03 - 6.5 m h⁻¹ with most velocities below 1.3 m h⁻¹ (Levandowsky and Kaneta 1987, Kamykowski et al. 1992). Our estimates for *G. catenatum* based on profiler data are between 1.5 and 6.0 m h⁻¹. This behaviour may play a significant role in avoiding flushing out of the estuary as the cells move below the rapid outflow of the surface layer. This in turn may support the development and persistence of monospecific *G. catenatum* blooms. As a potentially introduced species (Hallegraeff 1993, Hallegraeff and Fraga 1998, McMinn et al. 1997), deep vertical migration may help *G. catenatum* to outcompete other phytoplankton e.g. *Ceratium* spp. which have recorded swimming velocities of only 0.2 - 0.9 m h⁻¹ (Levandowsky and Kaneta, 1987) and non motile species. The peak chlorophyll concentrations reached in these dinoflagellate blooms imply a nitrogen concentration in phytoplankton which is more than 5 times the peak observed inorganic nitrogen concentrations in the water column. This implies a mechanism for accumulating biomass and/or nitrogen in the estuary. One possible mechanism is the interaction of dinoflagellate vertical migration with the two-layer estuarine circulation, which allows cells to accumulate in the estuary, and scavenge nitrogen from water circulating through the estuary. Another possible mechanism is accumulation of organic nitrogen in bottom sediments, and its release during summer and autumn.

During our study, the seed source for the *G. catenatum* blooms was probably from both resting cysts and over-wintering vegetative cells (except the 1997/98 summer bloom when no over-wintering cells were detectable). Although our sampling strategy precluded sediment/water interface studies for resting cyst germination, the presence of sexual stages during the bloom suggests a genetically heterogeneous origin which would be expected from either resting cyst germination or a diverse over-wintering population (Bolch et al. 1999). In the 1997/98 summer bloom, there were sufficient cells in the water column by early November to initiate a bloom purely by vegetative cell division if a doubling time of 3.3 days is used (based on culture data - Blackburn et al. 1989). However, the complete lack of vegetative cells in the water column prior to April 1997 suggests that resting cyst germination contributed to the population prior to bloom development.

Supply of germinating resting cysts and conditions for germination may play a role in the so far unexplained differences in the dinoflagellate dynamics between 1996/97 and 1997/98. While earlier work showed Port Cygnet to be one source of resting cysts (Bolch and Hallegraeff 1990), in our study blooms were detected first in the brackish water end of the estuary suggesting another potential source. This area has not been sampled for resting cysts in previous sediment studies. To fully establish the potential importance of resting cysts in bloom initiation investigations beyond the scope of our study are required. However, overall we consider life history events, and in particular resting cysts, are important in *G. catenatum* blooms in the Huon Estuary.

5.9 Summary and Conclusions

Phytoplankton in the Huon Estuary respond to a number of key physical and chemical drivers. The estuary receives persistent runoff, maintaining a strong two layer estuarine circulation which results in short flushing times (Chapter 3). The Huon River drains a relatively pristine catchment, and delivers low nutrient loads, but is very highly coloured by natural humics or CDOM. Consequently, the estuary is strongly light limited (see Chapter 6). Because of the estuary's temperate location it is subject to strong seasonal variation in surface irradiance.

Nutrient concentrations are high in bottom waters at the mouth of the estuary, and the two layer circulation results in a strong influx of nutrients from the marine boundary (Chapters 4, 10). There is a strong seasonal cycle in nutrient concentrations, with drawdown of dissolved inorganic nitrogen in surface waters in summer.

In many respects, the physical and chemical properties of the estuary are more characteristic of northern hemisphere estuaries than other Australian estuaries, and this is also true of phytoplankton, which have a fairly classic temperate seasonal cycle. Phytoplankton biomass in winter is uniformly low (about $0.3 \text{ mg chl } a \text{ m}^{-3}$), despite elevated nutrient concentrations, and presumably limited by light and flushing. Phytoplankton blooms occur in spring, summer and autumn, against a constant background of small flagellates. Overall, phytoplankton levels in the estuary are moderate, with (geometric) mean and median chlorophyll *a* values of about $1 \text{ mg chl } a \text{ m}^{-3}$. Chlorophyll *a* exceeds $4 \text{ mg chl } a \text{ m}^{-3}$ about 10% of the time. On the basis of chlorophyll (and nutrient) concentrations, the estuary would be classified as mesotrophic.

Phytoplankton community composition is relatively spatially homogeneous throughout the middle and lower estuary, reflecting the strong circulation and horizontal mixing. This means that monitoring at the five biological stations provides a good picture of temporal changes in community composition in the lower and middle estuary and Port Cygnet.

The small flagellate community was not identified in detail microscopically, but HPLC marker pigments indicated a mixture of haptophytes and prasinophytes, with some cyanobacteria. There were ecologically important blooms consisting of key species or species groups: *Chaetoceros* spp. and *Pseudonitzschia* spp. for the diatoms; and *Ceratium* spp. and the toxic species *Gymnodinium catenatum* for the dinoflagellates. While diatom and dinoflagellate blooms tended to alternate, in summer they sometimes co-existed. While cell numbers of dinoflagellates and diatoms were comparable, the phytoplankton biomass in the estuary was much greater during dinoflagellate blooms than during diatom blooms due to the larger cell size of the dinoflagellates.

There were strong interannual variations in the pattern of blooms. *Gymnodinium catenatum* was not present in the phytoplankton in 1996 and early 1997, but developed extensive blooms in the summer and again in the autumn of 1997/98. This difference in the bloom dynamics is not readily explained by physical and chemical conditions in the estuary over the two years. However key environmental variables that support or enhance blooms are: weak to strong water column stratification caused by a humic-rich, low-salinity surface layer and, in some cases, low wind stress; and temperatures of 12°C and over. In addition the life history of *G. catenatum*, in particular its resting cyst dynamics, may be important in bloom initiation and development.

Macronutrients supply does not appear to play a major triggering role. However, the capacity of *G. catenatum* to access both surface and deep nutrients because of strong vertical migration, particularly ammonia, clearly supports blooms.

During the *Gymnodinium catenatum* blooms in the summer and autumn of 1997/98, chlorophyll *a* levels reached values as high as 50 mg Chl *a* m⁻³, which would normally be classified as eutrophic. The equivalent nitrogen content would be about 30 μM, more than 5 times the maximum observed concentrations of dissolved inorganic nitrogen. This implies a mechanism whereby *G. catenatum* blooms concentrate nitrogen. One possible mechanism is that detrital organic nitrogen accumulates in sediments and is released into bottom waters over short periods in summer and autumn. A second possibility is that, by vertically migrating between out-flowing surface and in-flowing bottom layers, *G. catenatum* is able to maintain its horizontal position in the estuary, and accumulate nitrogen from the water circulating through the estuary. The model results in Chapter 10 suggest that vertical migration plays a critical role in enabling *G. catenatum* to reach the high observed biomass concentrations.

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6 OPTICAL ABSORPTION CHARACTERISTICS OF THE HUON ESTUARY

6.1 Introduction

The objectives of the optical component of the Huon Estuary Study were as follows:

1. To characterise the light environment of the Huon Estuary.
2. To understand the contribution of different components — phytoplankton, detritus and chromophoric dissolved organic matter (CDOM) — to the total light absorption of the water in the Huon Estuary.
3. To assess the potential for using optical characteristics to distinguish algal blooms of different phytoplankton species.
4. To assess the potential of optical measurements for in-water monitoring or remote sensing of water quality.

To answer these questions, we measured the absorption properties of the optical components, along with the pigment concentration and composition of samples collected throughout the estuary over a period of two years.

The total absorption $a(\lambda)$ of any water body can be expressed by:

$$a(\lambda) = a_{\text{ph}}(\lambda) + a_{\text{CDOM}}(\lambda) + a_{\text{d}}(\lambda) + a_{\text{w}}(\lambda)$$

where a_{ph} , a_{CDOM} , a_{d} and a_{w} represent absorption due to phytoplankton, CDOM, detritus (including inorganic suspended matter) and water respectively. In oceanic waters (case 1) the total absorption is dominated by the phytoplankton component, a_{ph} , which correlates significantly with the chl *a* concentration (Morel and Prieur 1977). In estuarine and coastal waters (case 2) the simple relationship found in case 1 waters does not hold, and the contribution of the detrital and CDOM components to the total absorption is often significantly greater than the contribution of the phytoplankton.

Phytoplankton require light for growth. The dominance of the detrital and CDOM components can significantly alter the availability of light, especially in the lower wavelengths, thus affecting the phytoplankton community's composition and the overall productivity of the estuary. To gauge the relative contribution of the components, we compare the absorption coefficients at 440 nm, where the phytoplankton absorption coefficient will be at its maximum value.

The Huon River drains a watershed that includes areas of pristine wilderness as well as agriculture. The freshwater input is strongly coloured by dissolved humic material, but is not considered turbid (see chapter 4, subsection 4.3.4), while clearer waters of oceanic origin intrude into the lower to middle estuary. The mixing of these waters, together with the

occurrence of algal blooms, including blooms of the toxic dinoflagellate *Gymnodinium catenatum* (see chapter 5), make the Huon Estuary optically complex.

The following list of studies does not constitute a comprehensive literature review, but provides examples of the type of optical studies that have been made previously. Bowling et al. (1986) described the spectral distribution and attenuation of underwater irradiance for 50 lakes within Tasmania and Kirk (1976, 1980) surveyed inland water bodies in southern NSW for the contribution of CDOM and particulate fractions to light absorption. Optical studies of estuaries outside Australia include the description of the light attenuation in 9 estuaries in New Zealand (Vant 1990), the optical properties of the San Francisco Bay Estuary (Di Toro 1978) and the measurement of the underwater irradiance in the Delaware Estuary (Biggs et al. 1983). Airborne remote sensing of estuaries and inland waters has also been studied (Dekker et al. 1992, Jupp et al. 1994, Weiss et al. 1997). To date this present study is the most detailed spatial and temporal study of the characteristics of light absorption in an estuarine system within Australia.

6.1.1 Notation

λ	wavelength, nm
a	absorption coefficient, m^{-1}
a_{ph}	absorption coefficient of phytoplankton, m^{-1}
a_{CDOM}	absorption coefficient of chromophoric dissolved organic matter, m^{-1}
a_d	absorption coefficient of detritus, m^{-1}
a_{ph}^N	absorption coefficient of phytoplankton normalised to 675 nm, (dimensionless)
S	spectral slope, nm^{-1}

6.2 Methods

Samples were collected at 30 sites throughout the estuary during spatial surveys and at weekly or fortnightly intervals at 5 sites in the middle to lower estuary (B1, F1, F3, H3, X3B). Samples were stored in the cool and the dark until return to the laboratory, where they were filtered under subdued lighting and stored in liquid nitrogen until analysis.

Optical density (OD) spectra for total particulate and detrital matter were obtained using a spectrophotometer equipped with an integrating sphere. Total particulate material was collected on a glass-fibre filter (GF/F) and scanned from 200 – 900 nm. The pigment component of the particulate matter was extracted using the method of Kishino et al. (1985) and the OD of the detrital component measured. The OD spectra of the phytoplankton pigment was obtained as the difference between the OD of the total particulate and detrital components. Absorption coefficients for the total particulate, phytoplankton and detrital components were determined using the quadratic equations developed by Mitchell (1990).

Samples for CDOM analysis were filtered first through a glass-fibre filter (GF/F) and then through a 0.2 μm polycarbonate filter immediately prior to analysis. The CDOM absorbance was measured in a 10 cm path length quartz cell, from 200–900 nm, using the normal cell compartment of the spectrophotometer, with Milli-q water as a reference.

Samples for pigment analysis were filtered through a glass-fibre filter and stored in liquid nitrogen until analysis by HPLC with photo-diode array detection.

A more detailed description of the sampling and analytical methods can be found in the supplements.

6.3 Results

6.3.1 CDOM

The Huon River carries extremely high concentrations of CDOM, which behave in a near-conservative manner in the estuary, exhibiting a linear mixing plot with salinity (Fig. 6.1). The maximum $a_{\text{CDOM}}(440)$ measured during spatial surveys reached 14 m^{-1} .

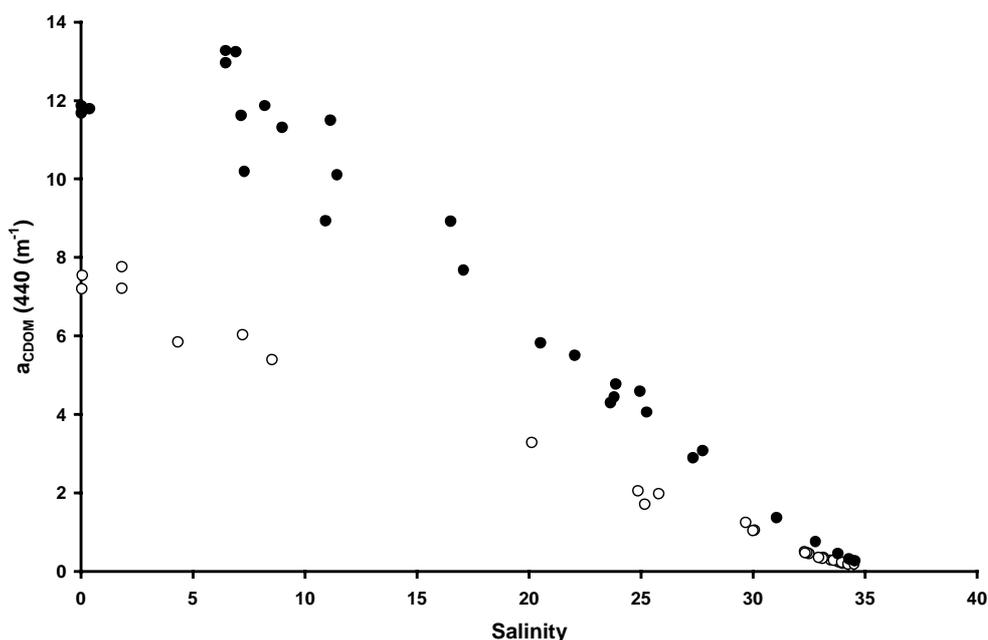


Fig. 6.1 The absorption coefficient for CDOM plotted against salinity. Open circles – HES 4 (Feb '97), minimum freshwater values; closed circles – HES 10 (Aug '98), maximum freshwater values.

However variation among surveys was found in the freshwater $a_{\text{CDOM}}(440)$ values (Fig. 6.1) with the maximum value ranging from 7–14 m^{-1} . In all surveys, $a_{\text{CDOM}}(440)$ values peaked at salinities of 5–10 and then were slightly lower in the freshwater end of the estuary. The additional source of CDOM to the surface waters in this salinity range is possibly due to runoff from the peat layers within the salt marshes around Egg Islands (see Chapter 2).

In a one-off study of five sites in the Huon Estuary, Burgess et al. 1993 recorded $a_{\text{CDOM}}(440)$ values ranging from 12.78 to 0.58 m^{-1} . Of the 50 Tasmanian lakes surveyed by Bowling et al. (1986), only 2 in the south west (Lakes Peddar and Gordon) had $a_{\text{CDOM}}(440)$ values of 7 – 8 m^{-1} and 3 small lakes on the west coast had $a_{\text{CDOM}}(440)$ values of >10 m^{-1} . In a study of lakes in southern NSW, Kirk (1976) recorded a maximum $a_{\text{CDOM}}(440)$ value of 2.9 m^{-1} , which he found dominated the optical characteristics of the lake.

At the Brabazon Park site in the Huon Estuary (H3, middle estuary), considerable variation was also noted in the weekly measurements of $a_{\text{CDOM}}(440)$ in the surface water (Fig. 6.2). A similar variation was found in the surface water at Hideaway Bay (B1) at the mouth of the estuary. The temporal variation in CDOM at both stations is driven primarily by changes in freshwater content (salinity) due to variation in river flow and wind forcing. The occurrence of high CDOM (low salinity) at Hideaway Bay was relatively uncommon: 5 records during the study.

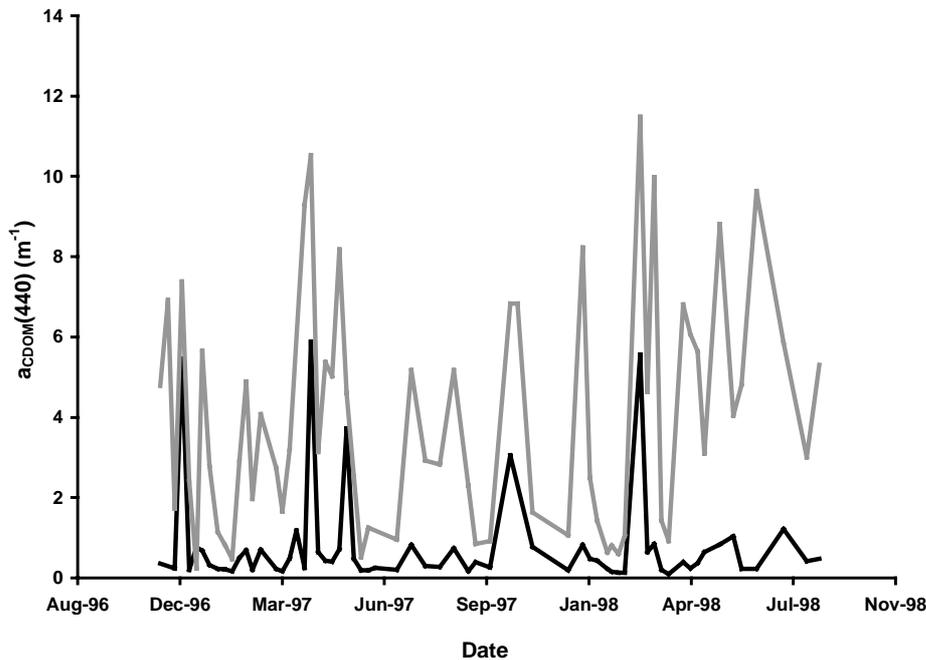


Fig. 6.2 The absorption coefficient of CDOM plotted against time at Brabazon Park (grey line) and Hideaway Bay (black line).

The absorption spectra of CDOM show strongest absorption in the lower wavelengths (blue region) and can be fitted well by an exponential curve:

$$a_{\text{CDOM}}(\lambda) = a_{\text{CDOM}}(440) \times \exp(-S \cdot (\lambda - 440)),$$

(Fig. 6.3). Exponential curves were fitted to all CDOM spectra, and the estimates of spectral slope S were quite constant. The mean and standard deviation of S over all surveys was 0.0154 ± 0.0009 . Other authors have used absorption at other wavelengths (eg. 355 nm) as a measure of CDOM concentration. Because the spectral slope in the Huon Estuary is constant, these can be reliably interconverted: e.g. $a_{\text{CDOM}}(355) = 3.7 \times a_{\text{CDOM}}(440)$.

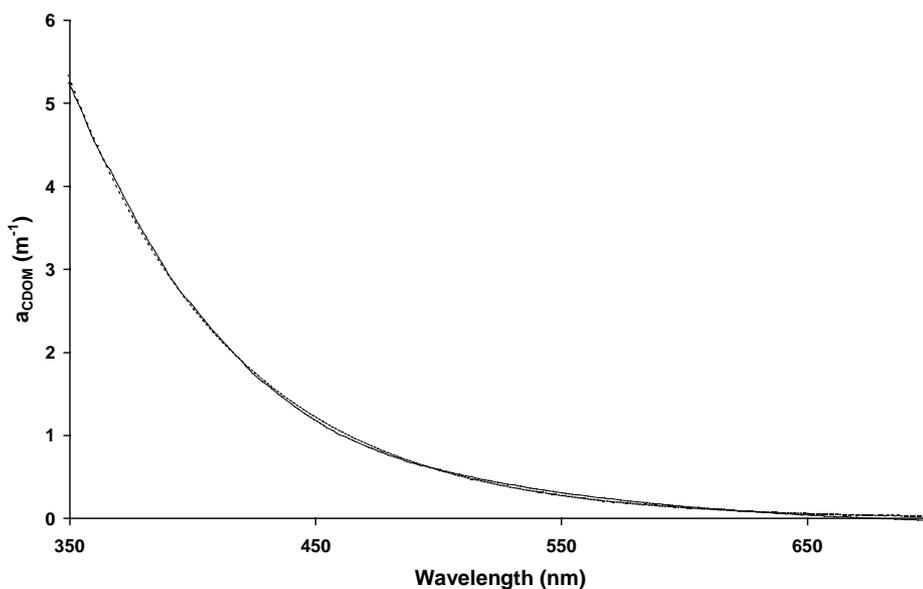


Fig. 6.3 Absorption spectra of CDOM; field sample (solid line), predicted spectra (dotted line).

6.3.2 Detritus

Detrital absorption coefficients throughout the study were high, especially considering the low suspended particulate load carried by the Huon River (chapter 4, section 4.3.4). In the middle to lower estuary, the $a_d(440)$ values show an inverse correlation with salinity (Fig. 6.4), similar to that shown by $a_{\text{CDOM}}(440)$ and salinity, although there is more scatter. (One would not expect non-algal particulate material to behave conservatively in the estuary, as it is subject to sedimentation and resuspension.)

There is a strong departure from conservative mixing behaviour in the upper estuary, with maximum detrital absorption coefficients occurring around salinities of 5 to 10, and values about three times lower occurring in freshwater (Fig. 6.4). A similar trend is seen in the relationship between suspended particulate matter (SPM) and salinity (see Fig. 4.6). The increase in $a_d(440)$ and SPM at around a salinity of 5 represents a source of absorbing particulate matter. This could be due to an external load, either from local runoff or resuspension, especially over the sediments around Egg Island. There could also be a contribution due to coagulation and precipitation of dissolved humic substances (a major fraction of CDOM), caused by the initial increase in ionic strength at salinities between 0 and 5 (Burton 1976) increasing the detrital absorption coefficient. This process appears to be complete by a salinity of about 10, at which stage dilution by seawater becomes the dominant process and

detrital absorption behaves more conservatively. The high particulate absorption values and the relatively low values of SPM supports the idea that adsorption of CDOM onto fine particulate material may be important.

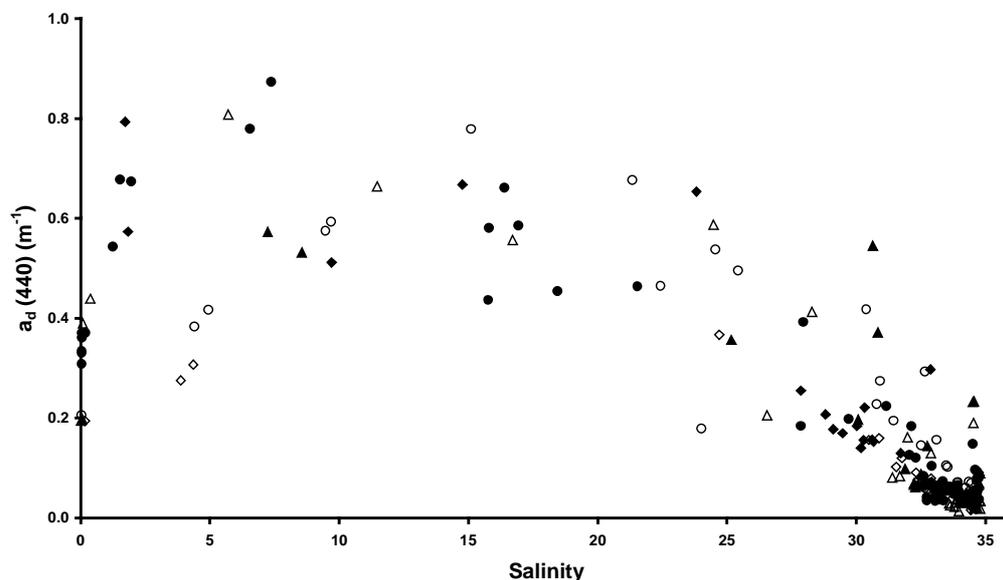


Fig. 6.4 The absorption coefficient for detritus plotted against salinity. (closed circles – HES 2 (Jul '96); closed diamonds – HES 3 (Oct '96); closed triangles – HES 4 (Feb '97); open circles – HES 5 (Jun '97); open diamonds – HES 6 (Oct '97); open triangles – HES 7 (Dec '97).

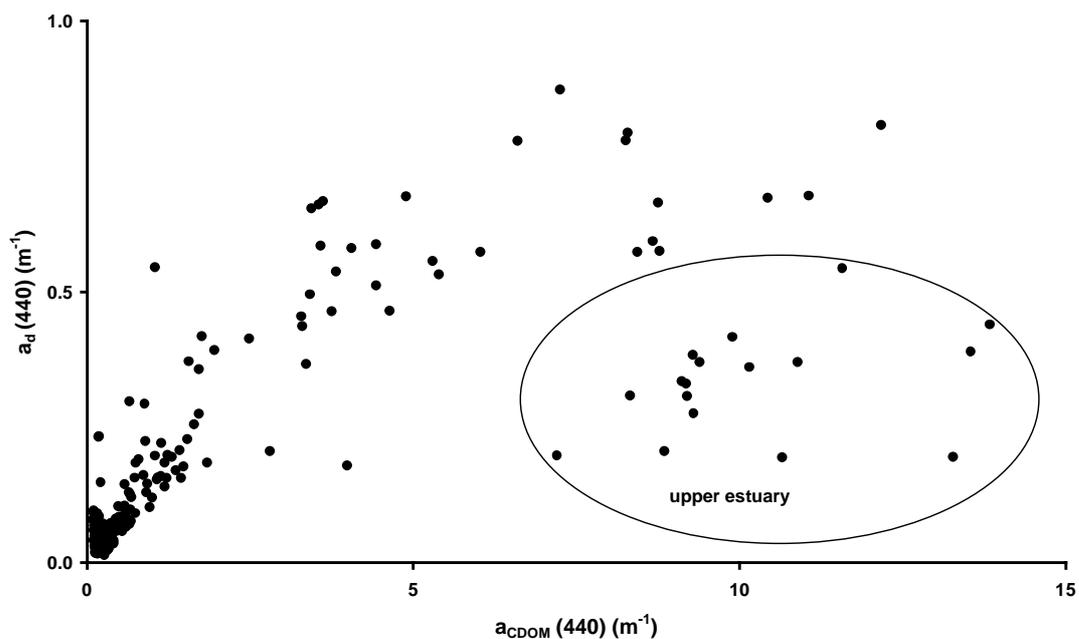


Fig. 6.5 The relationship between the absorption coefficients of CDOM and detritus.

However maximum CDOM absorption coefficients also occur at salinities of 5 to 10, suggesting that if coagulation is occurring then loss in CDOM is balanced by local inputs of CDOM (see section 6.3.1). Additional measurements also showed a strong contribution to light absorption from particulate material in the 0.2 to 0.8 μm range, which may also be due to colloidal or adsorbed CDOM. Excluding data from the upper estuary, $a_d(440)$ values co-vary with $a_{\text{CDOM}}(440)$ values (Fig. 6.5).

6.3.3 Phytoplankton

The pigment composition of any algal group is reflected in the shape of its absorption spectra. As several pigments are diagnostic markers for specific algal groups [Jeffrey and Vesik 1997], simple interpretation of phytoplankton composition can be made. The marker pigments detected in samples from the Huon Estuary and their corresponding algal group are listed in table 5.3 (Chapter 5). The combination of different pigments can produce distinctive spectral shapes in the absorption spectra of phytoplankton assemblages. However, distinguishing algal groups by absorption alone is not straightforward, as the absorption spectra of many of the marker pigment carotenoids are quite similar.

The magnitude of the absorption spectra obviously depends on the phytoplankton concentration. In order to compare spectral shape, we normalised phytoplankton absorption spectra to 1 at 675 nm. To illustrate the variation in spectral shape and magnitude, normalised absorption spectra corresponding to individual samples in which the pigment composition showed one dominant marker pigment, or marker pigments equal in dominance, are shown in

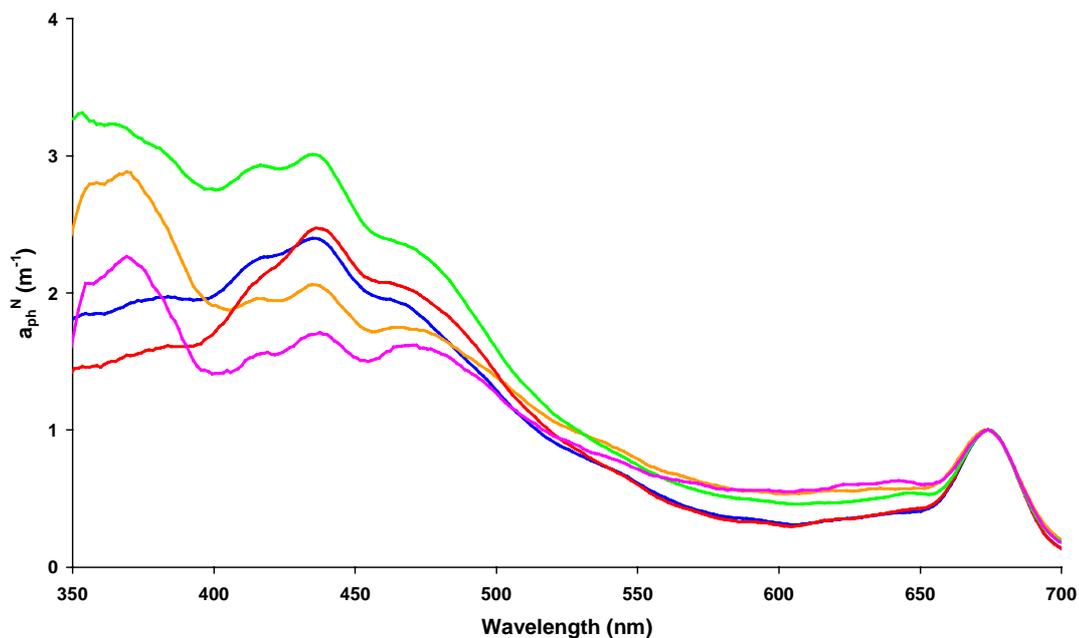


Fig. 6.6 Absorption spectra normalised at 675 nm for 19 HF (blue line); fucoxanthin (green line); peridinin (pink line); fucoxanthin and peridinin (red line) and peridinin, zeaxanthin and prasinoxanthin (orange line).

figure 6.6. A similar plot of the mean phytoplankton absorption spectra for six of the spatial surveys also shows variation in both spectral shape and magnitude (Fig. 6.7). However, with the exception of the spectral region below 400 nm (discussed further below), the mean absorption spectra from the spatial surveys are quite similar in shape. The standard deviations around the mean for each spatial survey show substantial overlap; it would not, therefore, be possible to reliably allocate an individual sample to a particular survey based on spectral shape. (Pigment analysis suggests that there are substantial and spatially consistent differences in phytoplankton composition amongst surveys.)

In early December 1997 the toxic dinoflagellate *Gymnodinium catenatum* bloomed throughout the estuary (see chapter 5). At times the bloom was almost monospecific, as illustrated by the significant linear relationship between water column chl *a* and cell abundance of *G. catenatum* ($r^2 = 0.77$). Towards the end of this dinoflagellate bloom, diatoms became dominant.

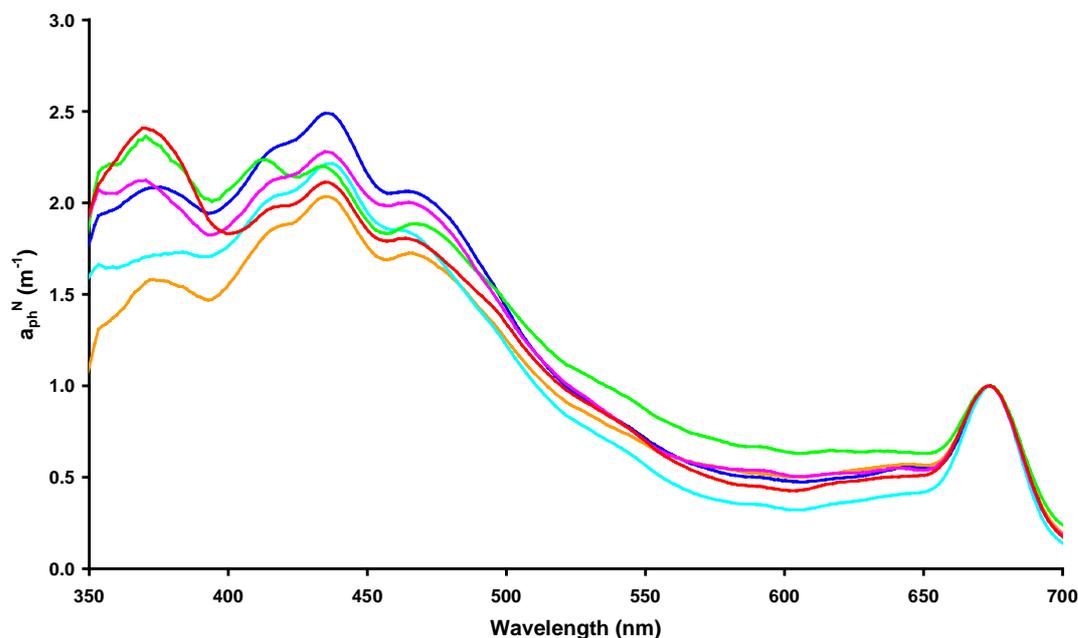


Fig. 6.7 Mean absorption spectra, normalised at 675 nm for spatial surveys; HES 2 (Jul '96 - blue line); HES 3 (Oct '96 - orange line); HES 4 (Feb '97 - aqua line); HES 5 (Jun '97 - green line); HES 6 (Oct '97 - pink line) and HES 7 (Dec '97 - red line).

The absorption spectra of both laboratory cultures and field samples were examined to determine whether different algal classes such as diatoms and dinoflagellates could be distinguished bio-optically. In figure 6.8a, absorption spectra of laboratory cultures of *Chaetoceros socialis* (a common diatom found in the Huon Estuary) and *G. catenatum* are compared, and in figure 6.8b, the mean absorption coefficient of a set of diatom-dominated field samples ($n=32$) is compared with the mean absorption coefficient of a set of dinoflagellate-dominated field samples ($n=23$). The samples were classified as dominated by either diatoms or dinoflagellates from the phytoplankton cell count data and the pigment composition data. Samples that were not clearly dominated by one class or the other were not included in the analysis. The mean absorption spectra are normalised to 1 at 675 nm and the dotted lines indicate ± 1 standard error associated with each mean spectrum.

Comparison of the cultures show the absorbance of *C. socialis* to be significantly lower than that of *G. catenatum* in the 450–620 nm region (Fig. 6.8a). *G. catenatum* also has a more pronounced shoulder at 460 nm than *C. socialis*, possibly a result of the percentage of chl *c* in *G. catenatum* being more than twice the percentage of chl *c* in *C. socialis*.

The difference in the absorption spectra of the field samples is less than that shown by the cultures. The absorption spectra of the dinoflagellate-dominated samples and the *G. catenatum* culture are closer than those of the diatom-dominated samples and the *C. socialis*. For both diatoms and dinoflagellates the ratio of absorbance at 435 nm to that at 675 nm was lower in the field samples than in the cultures, which may be explained by measured differences in pigment composition. Most remote sensing and *in situ* optical sensors have traditionally sampled in the visible light range (440–700 nm). In this range, the differences between diatom and dinoflagellate spectra are minor. Although the standard errors show statistically significant differences between diatom and dinoflagellate mean spectra, the standard deviations of individual samples around the means are much larger than the standard errors, so it would not be possible to reliably distinguish spectra from individual diatom- and *G. catenatum*-dominated field samples, using observations in the 400–700 nm range.

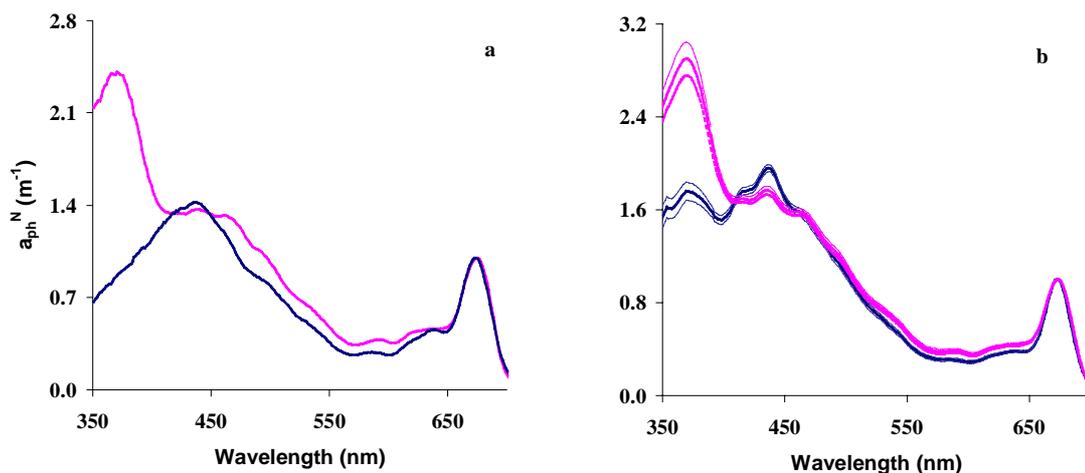


Fig. 6.8. (a) *in vivo* absorption spectra of laboratory cultures of *G. catenatum* (pink line) and *C. socialis* (blue line) normalised at 675 nm. (b) mean absorption spectra of dinoflagellate dominated samples (solid pink line) and diatom dominated samples (solid blue line). The dotted lines are the standard errors associated with the mean absorption spectra.

However, figure 6.8 shows significant and dramatic differences in spectral shape and in magnitude of both the cultures and the diatom- and dinoflagellate-dominated samples, in the ultraviolet region (300–400 nm). We believe these differences are due to the presence of mycosporine-like amino acids and could possibly be used by in-water optical monitoring instruments to indicate the presence of different algal groups.

6.3.4 Remote sensing

Comparing the contributions of the phytoplankton, CDOM and detritus to the total absorption of the water in the Huon Estuary, we find that CDOM is the dominant component at 440 nm (Fig. 6.9). Even under bloom conditions in surface waters' the CDOM absorption at 440 nm was generally greater than the absorption of *G. catenatum* throughout the estuary (Fig. 6.10). In cases such as this, where the total absorption is dominated by the CDOM component and the relative contribution of the phytoplankton component is low, then remote sensing would be unlikely to pick up small changes in the phytoplankton absorption coefficient. B1 at the mouth of the estuary generally has a CDOM absorption lower than any of the other sites, but the chlorophyll concentration there is also lower, and the relative contribution of CDOM remains high (Fig. 6.10). In the middle estuary around Brabazon Park (H3) and Wheatleys (F3), the CDOM absorption is always high and will always dominate the total absorption spectra unless the chl *a* concentration exceeds 40 mg.m^{-3} .

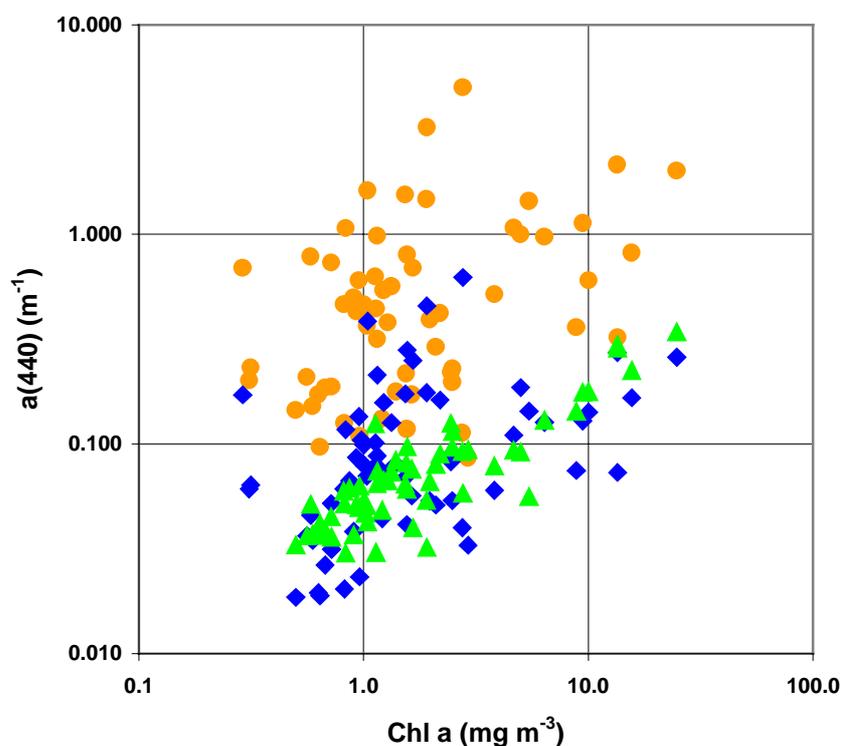


Fig. 6.9 Plot of absorption coefficients for phytoplankton pigment (green triangles), detritus (blue diamonds) and CDOM (orange circles) versus chl *a* concentration for spatial surveys 2 –7 (Jul '96 – Dec '97).

The contribution of the detritus component to the total absorption is also significant, often being around the same order of magnitude as the phytoplankton signal (Fig. 6.9).

Remote sensing of phytoplankton and other optical constituents is based on measurement of upwelling radiance spectra by visible / NIR sensors located on satellites or aircraft. Atmospheric correction is applied to the sensor radiance to estimate the water-leaving radiance or reflectance.

Radiative transfer models show that this reflectance is roughly proportional to the ratio of backscatter to absorption.

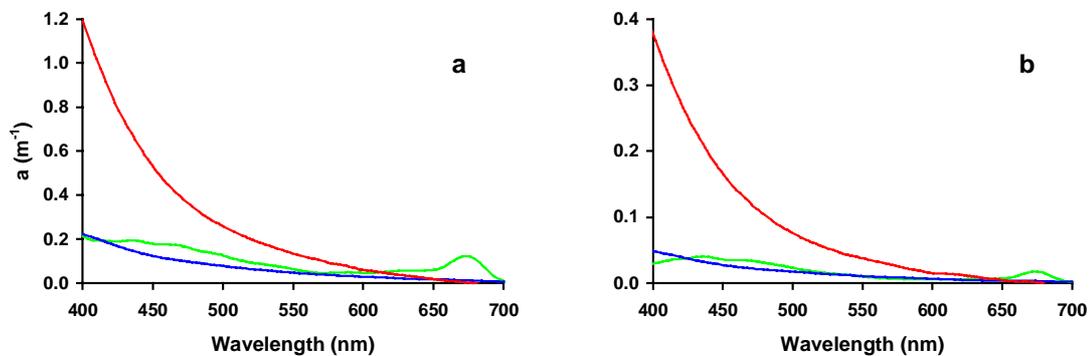


Fig. 6.10 Absorption coefficients for phytoplankton pigment (green line), detritus (blue line) and CDOM (red line) during bloom conditions at (a) Brabazon Park (H3) and (b) Hideaway Bay (B1).

In this study, we have only measured absorption properties of the key optical constituents, and cannot draw conclusions on the extent to which differences in backscatter spectra might allow discrimination of blooms or bloom type (Roesler and McLeroy-Etheridge 1998).

Ocean colour algorithms are generally of two types; case 1 algorithms for determining chl a in oceanic environments and case 2 for determining chl a in coastal/estuarine environments. Case 1 algorithms rely on simple ratios of upwelling radiance or reflectance in the blue and green wavelengths to estimate chl a whilst Case 2 algorithms use multi-spectral or hyperspectral methods to discriminate among optical components such as CDOM and chl a . At all sites in the Huon Estuary, the absorption due to CDOM is so high that water-leaving radiance in the blue and green will be very low and will result in significant overestimates in chl a when case 1 algorithms are used (Parslow et al. 1998). Conventional case 2 algorithms are also unlikely to give reliable estimates of phytoplankton biomass, except under bloom conditions. Scanners with high spectral resolution in the red and NIR, including the chlorophyll fluorescence peak, would be better suited for remote sensing of chlorophyll concentration in estuaries such as the Huon. Because CDOM is a good proxy for salinity, remote sensing could be used to synoptically map surface salinity distributions in the estuary.

6.3.5 Light environment

The high CDOM in the waters of the Huon Estuary significantly affect the underwater light quality of the river. As total absorption in the blue wavelengths is high, light in these wavelengths is rapidly attenuated, with 90% or more of blue light being removed in the top 1 metre (Fig. 6.11a). When an algal bloom is present, total absorption is also high at the red end of the spectrum, and so the underwater light field rapidly becomes impoverished in both red and blue light (Fig. 6.11b). The resulting light field will significantly affect algal growth, as not only

are light levels low, but most of the light is in the wavelengths that are inefficiently harvested by the majority of algal pigments; phycobilliproteins being the exception. Profile data collected hourly in the Huon Estuary showed that, under bloom conditions, *G. catenatum* exhibits a diel cycle in vertical migration (see chapter 5). Such migration allows the *G. catenatum* to avoid the effects of the CDOM and to fully utilise the available light by moving to the surface waters during the daylight hours.

In figures 6.11 a and b we have plotted light penetration to a depth of 2.5 m only, as CDOM concentration is generally highest in the brackish water layer which is concentrated in the top 2 – 3 metres of the water column. CDOM and light attenuation below this depth would both be reduced.

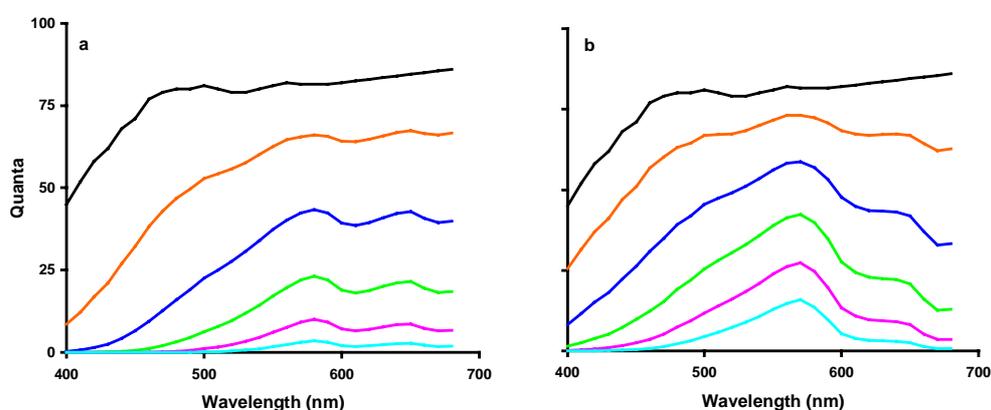


Fig. 6.11 Change in spectral distribution of light with depth – 0m (black line); 0.5m (orange line); 1m (dk blue line); 1.5m (green line); 2m (pink line) and 2.5m (light blue line). (a) high CDOM, low phytoplankton pigment absorption, and (b) low CDOM, high phytoplankton pigment absorption.

6.4 Conclusions

The Huon Estuary is optically very complex and is an extreme example of a case 2 water body. The total absorption, at 440 nm, of the water throughout the entire estuary is dominated by the CDOM component. The values of $a_{\text{CDOM}}(440)$ in the upper estuary (freshwater) ranged from 7 – 14 m^{-1} which are among the highest values recorded in the literature for Australian water bodies. Even under bloom conditions, the CDOM component was found to be dominant unless the chl *a* concentration (indicator of phytoplankton biomass) exceeded 40 $\text{mg}\cdot\text{m}^{-3}$.

The detritus component also contributes significantly to the total absorption and is often of the same order of magnitude as the phytoplankton component.

Due to the significant contribution of the CDOM and detritus components, it would not be possible to use either the optical measurements, such as those used in this study, or conventional remote sensing techniques in the visible wavelengths (400 – 700 nm) to reliably distinguish differences in phytoplankton biomass caused by different algal groups. However, it appears to be possible to distinguish between algal groups based on differences in absorption

spectra in the ultraviolet region (300 – 400 nm). This result requires more research, but it has the potential for routine *in situ* monitoring of absorption spectra being used to indicate the algal group or species causing a bloom event.

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7 ORGANIC CARBON AND NITROGEN IN SEDIMENTS

7.1 Introduction

Surface sediments provide an integrated picture of organic inputs to an environment over time frames of a few years to decades, depending on sedimentation rates. Surveys of surface sediments can thus indicate any localized inputs measured against an estuary-wide baseline and as well provide a better view of long-term average inputs. These surveys rely critically on a good characterisation of the sediment types and their distributions. A key issue in the Huon estuary is the reported high content of organic matter in many of the sediments. High values of organic carbon in the range 16-23.5% (of sediment dry weight) were reported for a fish farm site at Wheatley's Bay (near site F3) (Woodward et al. 1992), but these were determined as loss on ignition after removal of carbonates, which has been shown here (see section 7.3.2) and elsewhere (e.g. Leong and Tanner 1999) to overestimate the organic matter content.

Fish farms are potentially a major source of organic matter from both uneaten food and fish faeces. As much as 20% of the food may be lost directly to the environment, while 26% of the food eaten may end up as faeces (Gowen et al. 1988). Improvements in feeding strategies and feed formulation have lowered these figures, such that Woodward et al. (1992) estimated an 11.4% direct wastage of food from cages containing rainbow trout. Cho and Bureau (1997) report that modern low-pollution or highly digestible, nutrient-dense (HND) diet formulations yield outputs of less than 150 kg solid waste and 3 kg phosphorus per metric ton of salmonid fish produced. Nonetheless the contributions of organic matter from fish farms is significant, and when remineralised it can be a significant source of nutrients (McCaig et al. 1999). In the farms themselves, the build-up of organic matter in the sediments requires that the cages be moved periodically to fallow the site (see Chapter 8), but the extent to which fish-farm-derived organic matter is redistributed to areas outside the cages was not known at the start of our project. It was thus important that we determine the likely origins of organic matter in the estuary, quantify their relative proportions and determine the susceptibility of each type of organic matter to degradation (i.e. determine the proportion of labile to refractory organic matter).

The organic matter content of a sediment is determined by the sum of the various sources and the rates at which these are degraded by biological and chemical processes. In coastal and shelf marine sediments, adsorption of organic matter onto particles (sorptive protection) is thought to help preserve organic matter. The amount adsorbed often approximates to a monolayer coating of organic compounds onto the clastic material (Mayer 1994a,b; Keil et al. 1994; Hedges and Keil 1995). This sequestering of organic matter in the pores of rough surfaces is thought to make it less accessible to microbial attack (Mayer 1994a,b). Organic matter loadings thus often show strong inverse correlations with grain size (Mayer 1994a,b), with low concentrations associated with sands and high concentrations associated with fine muds. For this reason, we determined the grain size distribution of the sediments that were analysed for organic matter content.

Labile organic matter (i.e. matter that is easily broken down by chemical and microbiological pathways), is potentially a major source of nutrients returned to the water column. For example,

Woodward et al. (1992) estimated a nutrient flux of $130 \text{ mmol}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$ for ammonia and $50 \text{ mmol}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$ for filterable reactive phosphorus associated with high organic matter loadings under a fish cage in the Huon. Sediment studies can thus provide information about the role of sediments as a reservoir and a source of nutrients (C, N, P, etc) recycled back into the water column through demineralisation and sediment-water exchange processes. Resuspension is widely seen as facilitating the reinjection of solutes into the water column and hence a factor in the degradation of organic matter. However, this process still remains little studied and poorly quantified.

Many factors affect the degradability of organic matter in a sediment. As discussed above, relatively labile organic matter can escape degradation if adsorbed within sediment pores. Also, some labile organic compounds can be incorporated into refractory macromolecules or humic substances (the so-called condensation pathway). The type of compounds present is also critical, with simple organics such as acetate being rapidly degraded (utilised). As such organic substances in sediments are a major energy source for benthic organisms, they can lead to the build-up of microbial mats under the fish cages (Ritz et al. 1989). However, in an estuary like the Huon, terrestrially derived organic matter consists of refractory humic-like substances that resist degradation. It is perhaps not surprising that rates of organic matter oxidation measured in a variety of sediments and depositional environments range over three orders of magnitude, from $20 \text{ mmol C}\cdot\text{m}^{-2}\cdot\text{year}^{-1}$ to $60 \text{ mol C}\cdot\text{m}^{-2}\cdot\text{year}^{-1}$ (Henrichs 1992).

We expected that much of the organic matter would be derived from freshwater inputs of terrestrial higher-plant material, with additional inputs from autochthonous phytoplankton and localised inputs from salmon farms, sewage treatment plants and stormwater drains. The organic matter from these diverse sources can have very different compositions and degrade at different rates. Much of the terrestrial matter consists of high molecular weight tannin-like compounds, which seem to degrade very slowly, so its remineralisation probably does not contribute greatly to nutrient loads. Another source of terrestrial organic matter in the Huon is the substantial loadings of wood waste from forestry operations and dumping of pulp fibre from the pulp mill at Hospital Bay (Newell 1969). Such cellulosic material can have a high oxygen demand ($4\text{-}10 \text{ g O}_2\cdot\text{m}^{-2}\cdot\text{day}^{-1}$ at 20°C whereas a typical estuarine sediment could use $1.5 \text{ g O}_2\cdot\text{m}^{-2}\cdot\text{day}^{-1}$ at 20°C ; Thomann 1972).

Under finfish cages, and sometimes to a limited extent outside of them, there are high concentrations of organic matter from uneaten fish food and faeces; these are fairly quickly broken down and result in poorer water quality underneath the cages (e.g. FAO 1996). It is of practical importance to measure these breakdown rates so that cages can be moved to allow the sediments to recover. As part of a fallowing study established by staff from Huon Aquaculture and DPIF (now DPIWE), we conducted organic analyses of sediments collected at 3, 6 and 11 month intervals from the site where a fish cage had been removed. Results of this work are described in Chapter 8.

The amount of organic matter in a sediment is also affected by physical processes such as resuspension and deposition. In environments where there is extensive sediment recycling, there is often a close correlation between organic matter content and grain size. Compositional features such as stable carbon isotope ratio and distribution of lipid biomarkers provide a useful basis for comparison.

7.1.1 Methods of identifying sources of organic matter (isotopes and lipid biomarkers)

We adopted two approaches to determine the sources of organic matter in the estuary: stable carbon and nitrogen isotopes, and lipid biomarkers.

Stable isotopes, which are widely used to determine the sources of organic matter in aquatic food webs (Fry and Sherr 1984), had previously been shown to be useful in southern Tasmania (Ye et al. 1991; Fenton 1996). This approach makes use of the fact that different biochemical pathways, and the degree to which carbon (and nitrogen) has been reworked, will lead to different ratios of ^{13}C and ^{12}C (and ^{15}N and ^{14}N). When the resulting stable isotope signatures of the different sources (so-called “end members”) are sufficiently different, it is possible to apportion the sources of organic matter. Isotope values are normally expressed relative to a standard (Vienna Pee Dee Belemnite) and are given in units of parts per thousand (‰). The measurement of stable isotopes was not proposed in the objectives of our original study, but we decided to include them when it was realised that such data could throw light on the origins of organic matter in the estuary complementary to the independently determined biomarker data.

Lipid biomarkers are naturally-occurring compounds that have a distinctive structure that can be related to a particular source such as phytoplankton, bacteria, marine animals or higher-plants. Some of these compounds are quite stable and can persist for thousands of years in sediments. Others, such as phospholipids and carotenoid pigments, are rapidly degraded and so their presence in a sediment can be used as an indicator of living biomass. In this study we have chosen to study fatty acids, fatty alcohols and sterols.

7.2 Sampling Design

As part of the Huon Estuary Study we made a broad survey of the contents of organic matter in surface waters and sediments collected throughout the estuary. The number of samples was dictated by the need to cover the spatial extent of the estuary, as well as provide sufficient samples across-river to discern any compositional gradients. Maps of water depths and sediment types were used to decide sample spacing. Limitations in boat-time reduced the sample set to a practical number. Dissolved organic matter concentrations (DOM: mg/L) and the $\delta^{13}\text{C}$ value of DOM in surface waters were determined from two surveys in July and October 1996. The results are shown in Fig. 7.1. Dissolved organic matter was extracted from the water samples by reverse-phase C18 solid-phase extraction (Amador et al. 1990). Either 2 L or 500 mL of sample was acidified to pH 2 with concentrated HCl and passed through BondElute C18 solid-phase extraction cartridges (Varian) by attaching a peristaltic pump to the exit of the cartridge to pull the sample through the cartridge.

Only one set of surface sediment samples was collected. Given that sediments average inputs over many years, a single sampling can provide an overall picture of organic matter distributions although physical events such as high water flow can redistribute sediments, potentially giving different results for the same site when sampled at different times. Samples were collected at many of the chemical sites as well as selected additional sites in the estuary, making a total of 40 in all (see map in Chapter 2) over the two days 13 and 14 May, 1997. Surface sediments were obtained using a grab sampler and the top 3 cm removed for analysis.

Organic carbon and nitrogen contents and stable isotope values in sediments were measured with a Fisons NA 1500 NCS Elemental Analyser coupled to a Finnigan-MAT Delta S isotope ratio mass spectrometer. Sediments for TOC and ^{13}C analysis were decarbonated with HCl before analysis. Total lipids were extracted by a modified Bligh-Dyer method (Leeming and Nichols 1998) and analysed by capillary gas chromatography with flame ionisation detection.

7.3 Results and Discussion

7.3.1 Organic matter in surface waters

Dissolved organic carbon is a major constituent of the carbon pool in marine and estuarine systems (McCarthy et al. 1996). In estuarine systems, organic matter is derived from both terrestrial sources and *in situ* primary production by algae, and in some cases seagrass, macrophytes and photosynthetic bacteria. Organic matter is an energy source for microorganisms which channel carbon to higher trophic levels. In addition, organic matter can be remineralised by microorganisms, providing nutrients for autotrophic production in the marine environment. The headwaters of the Huon river are in a high rainfall zone with extensive buttongrass (*Gymnoschoenus sphaerocephalus*) moors and wet scrub. Downstream areas are characterised by montane grassy forest interspersed with wet forest dominated by *Eucalyptus obliqua*. We would thus expect that much of the DOM would be derived from terrestrial vegetation.

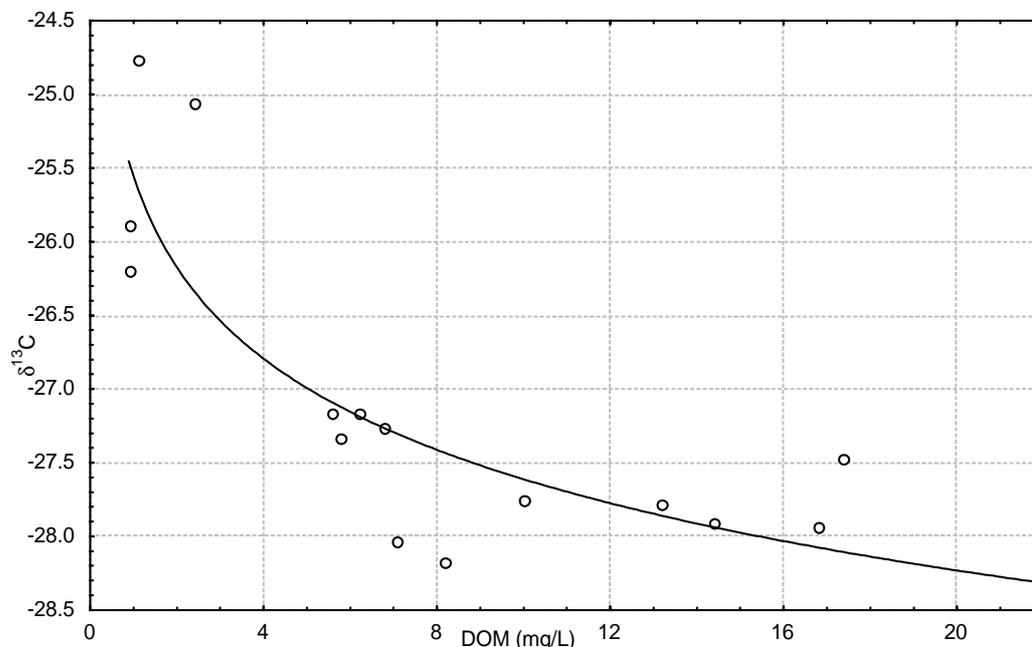


Fig. 7.1 Relationship between $\delta^{13}\text{C}$ values of dissolved organic matter (DOM) and its concentrations in surface waters of the Huon estuary.

DOM concentrations in October 1996 were highest in the upper reaches of the Huon (>18 mg/L). These have ^{13}C -depleted (i.e. light) $\delta^{13}\text{C}$ values around -27‰ characteristic of higher-plant material (Fig. 7.1). Concentrations drop to 5–9 mg/L in the middle estuary and still further to about 2 mg/L near the mouth. This is accompanied by a trend to ^{13}C -enriched (i.e. heavier) $\delta^{13}\text{C}$ values in the range -25 to -26‰ (Fig. 7.1). The amount of organic matter isolated from each sample varied with salinity: marine waters contained much lower amounts of DOM than did fresh water. Therefore, it is clear that most of the DOM in the estuary has a terrestrial origin. It is only at the far marine-end of the estuary that the DOM shows a significant contribution from marine sources (the lightest $\delta^{13}\text{C}$ value measured was -24.8‰ which is consistent with an approximately 60:40 mixture of marine and terrestrial DOM).

A plot of DOM concentrations versus salinity (Fig. 7.2) shows that the mixing of DOM with seawater is non-conservative, with an apparent loss of DOM in the mid-salinity (5–25) range. Some DOM is probably lost during transit of the estuary and mixing with seawater due to colloid formation. It may be that a fraction is remineralised, although this should be a minor effect since transit times are so short. The data are not consistent with a major contribution from fish-farming operations to the DOM measured in the estuary.

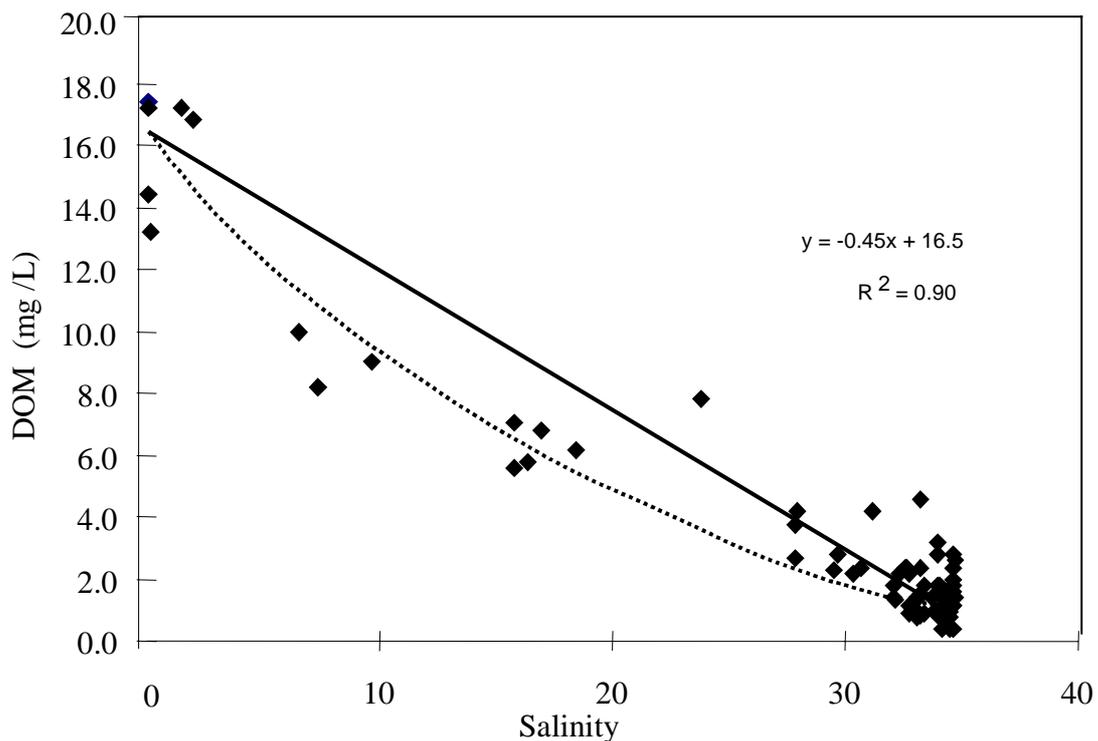


Fig. 7.2 Relationship between DOM concentrations and salinity. The line of best fit is shown ($r^2 = 0.9$), together with a hand-fitted curve suggesting some loss of freshwater-derived DOM in more saline waters.

Table 7.1 Percent sediment organic matter determined by loss-on-ignition (LOI), organic carbon and nitrogen contents in sediments as % dry wt. (%C_{org} and %N_{org}), stable isotope values $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$, carbon:nitrogen ratios in organic matter (C:N) plus estimates of terrestrial and marine organic carbon and nitrogen determined from the stable isotope data.

Site	%C _{org}	LOI	%N _{org}	$\delta^{13}\text{C}$	$\delta^{15}\text{N}$	C:N	LOI/C _{org}	% C Terr	% N Terr	% C Mar	% N Mar
A1	0.4	1.3	0.02	-26.1	6.6	20.0	3.3	63	15	37	85
A3	3.4	13.6	0.46	-23.9	7.1	7.4	4.0	29	7	71	93
A5	3.3	15.2	0.50	-24.4	7.2	6.6	4.6	37	5	63	95
A7	0.4	2.6	0.12	-24.7	6.9	2.9	7.4	42	10	58	90
S1	3.1	17.5	0.10	-23.6	6.9	31.0	5.6	25	10	75	90
S2	3.1	17.5	0.60	-23.7	7.0	5.1	5.7	26	8	74	92
B1	4.1	17.0	0.51	-23.6	7.2	8.1	4.1	25	5	75	95
B2	3.5	nd	0.50	-24.7	7.1	7.0	-	42	7	58	93
B5	3.4	13.7	0.50	-24.3	6.9	6.8	4.0	35	10	65	90
C1	0.6	4.2	0.05	-25.4	6.4	12.8	6.6	52	18	48	82
C2	3.3	18.1	0.49	-24.9	7.0	6.8	5.4	45	8	55	92
S5	4.2	17.9	0.65	-24.9	6.7	6.4	4.3	45	13	55	87
E1	3.0	16.1	0.51	-24.0	6.7	5.8	5.4	31	13	69	87
E5	4.6	17.9	0.65	-25.2	6.0	7.1	3.9	49	25	51	75
F1	4.6	17.9	0.59	-24.7	6.1	7.7	3.9	42	23	58	77
F2	4.5	17.2	0.59	-25.3	5.5	7.6	3.8	51	33	49	67
F3	4.5	17.8	0.74	-25.4	6.5	6.1	4.0	52	17	48	83
S7	5.1	18.9	0.66	-26.8	4.9	7.7	3.7	74	43	26	57
S8	6.6	18.3	0.64	-26.5	4.6	10.3	2.8	69	48	31	52
S9	7.4	19.9	0.68	-26.5	4.3	10.8	2.7	69	53	31	47
H1	7.1	19.2	0.59	-27.3	3.9	12.1	2.7	82	60	18	40
H3	9.6	20.5	0.62	-26.8	3.0	15.5	2.1	74	75	26	25
S10	8.5	23.8	0.74	-27.5	1.7	11.5	2.8	85	97	15	3
S11	7.1	19.4	0.67	-27.1	3.1	10.6	2.7	78	73	22	27
S12	7.8	21.3	0.65	-27.2	3.3	12.0	2.7	80	70	20	30
I1	7.0	21.8	0.64	-26.1	1.8	10.9	3.1	63	95	37	5
I3	3.0	8.1	0.12	-27.6	1.8	24.6	2.7	86	95	14	5
J1	5.0	18.4	0.32	-28.3	2.0	15.6	3.7	97	92	3	8
L1	0.2	1.0	0.11	-27.6	2.7	1.9	4.8	86	80	14	20
N1	6.9	23.9	0.61	-28.6	1.1	11.3	3.5	102	107	-2	-7
N2	0.3	1.5	0.01	-28.3	1.8	26.0	5.8	97	95	3	5
V1	2.6	11.5	0.46	-24.3	6.6	5.7	4.4	35	15	65	85
V2	3.0	17.7	0.51	-24.5	6.5	5.8	6.0	38	17	62	83
V3	5.4	17.5	0.58	-23.4	6.4	9.3	3.2	22	18	78	82
S14	4.7	17.9	0.64	-25.0	6.3	7.3	3.8	46	20	54	80
S15	3.7	13.4	0.55	-24.1	6.1	6.7	3.6	32	23	68	77
X1	4.3	14.4	0.55	-23.7	6.2	7.8	3.3	26	22	74	78
X3	2.3	10.9	0.46	-24.6	6.2	4.9	4.8	40	22	60	78
Y1	5.8	11.3	0.47	-24.9	5.6	12.3	1.9	45	32	55	68
S16	5.2	6.3	0.37	-25.3	4.2	14.1	1.2	51	55	49	45

The percentage of terrestrial and aquatic organic matter was determined from the $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values assuming a terrestrial plant end-member having a $\delta^{13}\text{C}$ value of -28.5‰ and $\delta^{15}\text{N}$ value of 1.5 (as found at the upriver sediment sites) and a marine sediment end-member having a $\delta^{13}\text{C}$ value of -22.0‰ and $\delta^{15}\text{N}$ value of 7.5‰ (see text).

7.3.2 The measurement of organic carbon in sediments

At first sight the measurement of the organic matter content of a sediment might seem to be a straightforward task. In many previous studies, including some conducted in the Huon (e.g. Woodward et al. 1992), the organic matter content was determined simply as the weight loss of a dried sediment on high temperature combustion. This is termed “weight loss on ignition” or simply LOI. We determined LOI values, but in addition we also measured the organic carbon content (C_{org}) directly with an elemental analyser because it is a better measure of organic carbon. As can be seen from Table 7.1, there is a large, but not always systematic, difference between the two sets of values. LOI values will be greater than C_{org} because the former purports to measure organic matter while the latter measures organic carbon, but nonetheless that LOI values are still higher than expected.

Craft et al. (1991) derived an equation linking LOI values and organic carbon content for estuarine marsh soils of: $C_{org} = 0.40 \times LOI + 0.0025 \times LOI^2$. We derived an equation from our data in the hope that it might be possible to convert previously published LOI data to organic carbon. However, using our data the derived relationship was quite different: $C_{org} = 0.17 \times LOI + 0.0062 \times LOI^2$, and the r^2 value was only 0.63 due to high degree of scatter. A linear function of $C_{org} = 0.28 \times LOI$ was fitted to the data, but the r^2 value was only 0.61.

It has now become clear that the LOI method can seriously overestimate the content of organic matter (Leong and Tanner 1999). For example, Mook and Hoskin (1982) showed that up to 20% of the weight of an organic-free sediment dried at 50°C can be lost when ignited in a muffle furnace at 550°C. Much of this loss is attributable to the presence of “bound” or “structural” water (Billen 1978). This bound water is not completely eliminated when a sediment is dried at 100°C, and is thus measured as “organic matter” when the dried sediment is combusted. Conversely, carbon values obtained using an elemental analyser can be inflated if carbonate (shells etc) is present, but this was avoided in our work by treating the sediments with acid before analysis. We thus feel that the C_{org} values offer a more reliable means for comparing organic matter contents in sediments, so the following discussion is based on these values. However, we have included the LOI values to facilitate comparison with previous work.

7.3.3 Relationship between organic carbon (C_{org}) and nitrogen (N_{org}) with grain size

C_{org} values in the estuary are highly variable, ranging from < 1% of sediment dry weight near the mouth of the estuary to values exceeding 8% upstream of the bend (Fig. 7.3). N_{org} values ranged from barely detectable in sandy sediments to a maximum of 0.75% (Fig. 7.3). This high variability in C_{org} and N_{org} contents in the sediments is partly due to a multiplicity of possible sources, but much of the variability can be accounted for by the varying proportion of fine clays and silts (the so-called mud fraction with grain sizes less than 63 μm). The sediment grain size throughout the estuary is highly variable and reflects the changing bottom topography and current strengths in the estuary. Sands predominate in the upper reaches (Fig. 7.4) and at the mouth. Muds are more common in Port Cygnet and in the mid-estuary (Fig. 7.4).

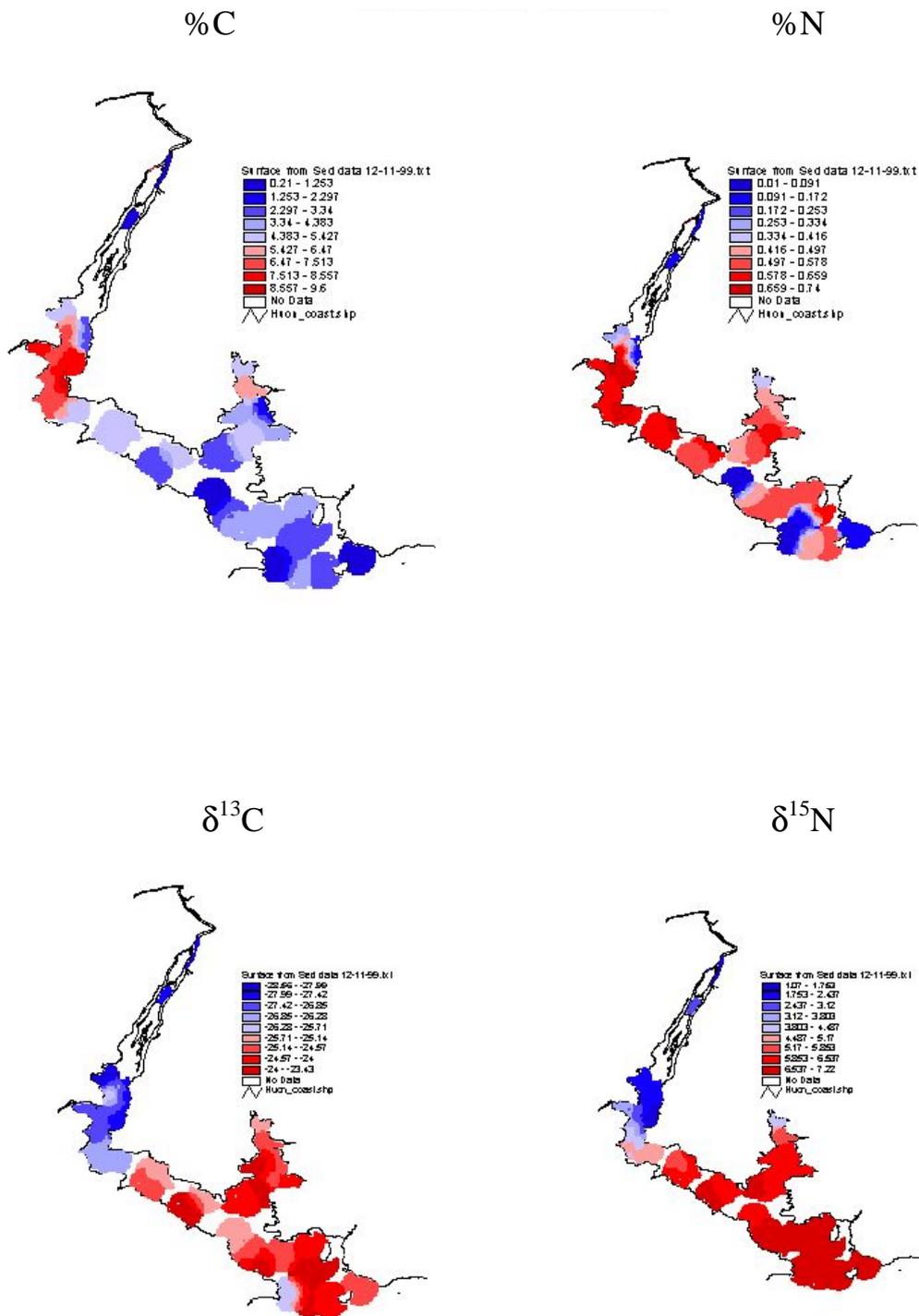


Fig. 7.3 Plots of %C_{org}, %N, $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of organic matter in surface sediments. Note the high proportion of terrestrial-derived organic matter in the upper estuary.

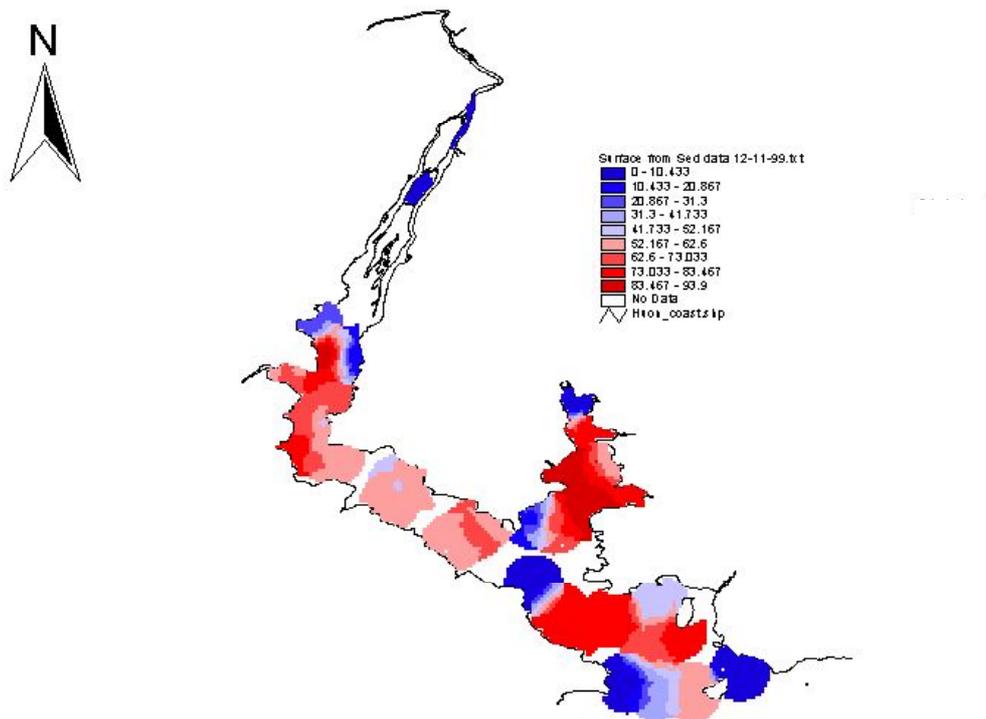


Fig. 7.4 Distribution of mud fraction in sediments of the Huon estuary.

In many estuaries there is a strong correlation between the organic matter content in the sediments and the proportion of mud. A plot of %C_{org} against the proportion of mud reveals a reasonable correlation (Fig. 7.5), with higher contents of organic carbon found in muddier sediments. Interestingly, the samples with greater than expected carbon contents were all associated with high contents of terrestrial-derived organic matter. The correlation of N_{org} with the proportion of mud in the sediment was even stronger ($r^2 = 0.75$, with only a few outliers). A high correlation is typically found in cases where the organic matter is extensively recycled and adsorbed onto particles before being finally incorporated into the sediment. We examined the %C_{org}-grain size relationship for different ranges of organic carbon and found an excellent correlation in the 0-1% C_{org} range ($r^2 = 0.96$) where mud contents were low, irrespective of whether the organic matter was of terrestrial or aquatic origin (whether from phytoplankton, sewage or fish-farm operations) as judged from the $\delta^{13}\text{C}$ values (see next section). However, at higher %C_{org} values the data show a lot of scatter, with many sediments containing more organic matter than predicted from their mud content. The isotope and biomarker data can be used to explain some of these “anomalies”.

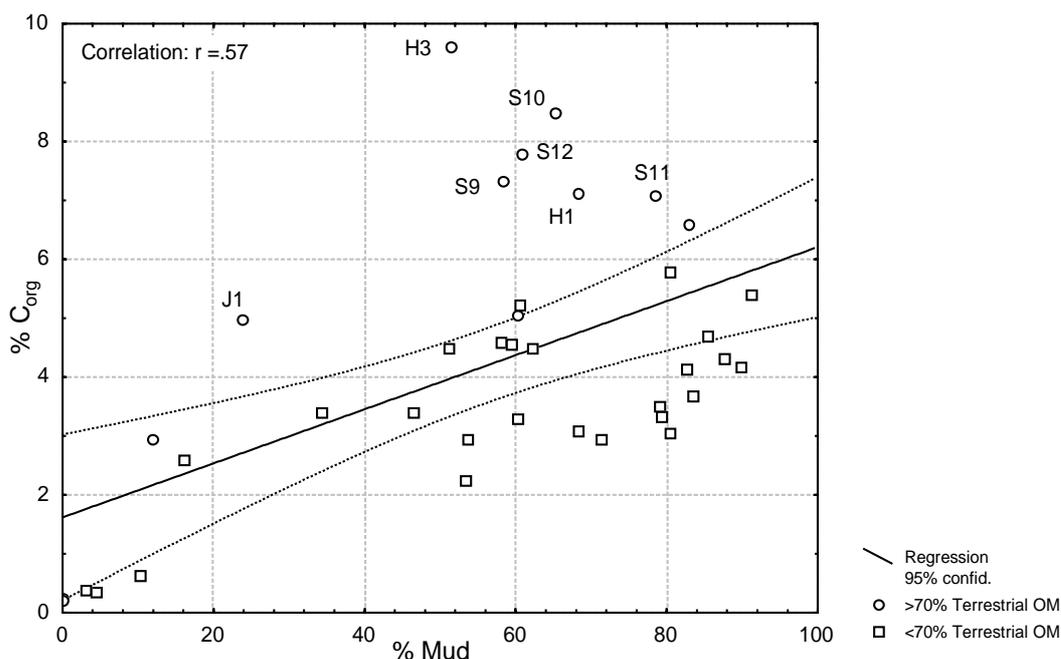


Fig. 7.5 Relationship between organic carbon content in sediments and the proportion of mud particles (i.e. <63 μm grain-size fraction) in Huon estuary sediments. The regression line is calculated from all points, but a more appropriate line would be translated down by about 1.6% C_{org} since sandy sediments have very low organic carbon contents. Dotted lines represent the 95% confidence interval. Outlier points are discussed in the text.

Higher-than-expected organic matter contents can be due to a recent input of organic matter such as from a phytoplankton bloom, or due to the association of organic matter with non-clay particles such as wood fibre. For example, sites S9, S10 and S12 all show 2–3 times more organic carbon than would be predicted from the % C_{org} vs mud correlation, and S11 also shows a greater-than-expected value. The $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values all indicate a high proportion of terrestrial-derived organic matter (>80%) and all have high sitosterol contents. It is possible that the elevated organic matter is due to the presence of wood fibres from the former sawmill at Hospital Bay. The largest anomaly occurs at site H3, which has 3–4 times the amount of organic carbon predicted from its mud content. This site is adjacent to the fish farm at Brabazon Park, but it would be wrong to conclude that the organic matter enrichment is mainly due to the fish farm, since the $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values suggest that at least 75% of the organic matter is from higher plants (Table 7.1).

7.3.4 Terrestrial sources of organic matter

One might expect in a river-dominated system such as the Huon estuary, with its tannin-laden brown waters, that much of the organic matter in the sediments would be of terrestrial origin. Several lines of data support this contention, but only for the upper portions of the estuary. It is clear from the DOM data that the main source here is from terrestrial plants. Concentrations of DOM in the upper estuary are about 18 mg/L dropping to less than 2 mg/L at the mouth of the

estuary. Most of the dissolved organic matter is rapidly transported through the estuary and thus it is not a major contributor to the organic matter deposited in the sediments.

C:N ratios can be useful indicators of the sources of organic matter in an estuary, although the effects of degradation can significantly change these ratios (e.g. Thornton and McManus 1994). The Redfield C:N ratio for actively growing marine phytoplankton is 5.7 (by weight), and values in the range 6–9 are typically reported for autochthonous marine-derived organic matter, whereas values greater than 12 are usually found in terrestrially derived organic matter (Bordovskiy 1965). In the Huon estuary sediment set, C:N values greater than 12 are found at many of the upper estuary sites (Table 7.1); indeed a few such as I3, N2 and S1 have values greater than 20, which is typical of degraded wood from higher plants. A1 has an anomalous value of 20, but this may be a measurement error due to the very small amounts of organic matter in this sediment. Values in the range 12–20 are found at site C1 near Police Point; H1, H3, J1, N1 and S12 in the upper estuary; and S16 and Y1 in Port Cygnet.

The $\delta^{13}\text{C}$ values of organic matter in the sediments (Fig. 7.3) show a range of values from terrestrial (-28.5‰) in the upper estuary to mixed aquatic–terrestrial (-24‰) in the lower estuary. Sediments containing mostly marine (phytoplankton-dominated) organic matter typically give $\delta^{13}\text{C}$ values of -19 to -21‰ . (Fry and Sherr 1984). We have estimated the proportion of aquatic (= phytoplankton) and terrestrial organic carbon from a simple two component mixing model, using as end-members a terrestrial value of -28.5‰ and an aquatic value of -22‰ (Table 7.1). This calculation is necessarily inexact since it does not include multiple sources or allow for changes in the end-member values according to source or seasonal effects. The value for the terrestrial end-member is reasonably well defined by the $\delta^{13}\text{C}$ values of the terrestrially-dominated sediments in the upper estuary (e.g. on transect N), but the marine value is less precise and more prone to variations due to varying CO_2 concentrations, species differences etc. A range of phytoplankton values have been reported in the literature (e.g. -13 to -29‰ ; Descolas-Gros and Fontugne 1990), but a value of -21 to -22‰ seems reasonable for phytoplankton in the Huon, based on values for the marine sediments along transect A (which still contain some terrestrially derived organic matter) and unpublished data for south-eastern Australian marine sediments. Despite these uncertainties, a consistent picture emerges of organic carbon in the upper estuary being almost entirely derived from terrestrial plant material, while there are substantial inputs of non-terrestrial “marine” sources at the middle and lower estuary sites.

The $\delta^{15}\text{N}$ values can be used in the same way to determine contributions from marine and terrestrial sources to the organic nitrogen and a similar picture emerges. In the Huon estuary, the sediment data are consistent with a contribution of terrestrial plant matter having a $\delta^{15}\text{N}$ value of about 1.5‰ and a marine end-member value of about 7.5‰ . In almost all samples, the organic N is dominated by marine sources (exceptions being S10, I1, I3, J1, N1 and N2). This is true even when the carbon shows a strong terrestrial signature, simply because the C:N ratio of terrestrial organic matter is so much higher than in marine organic matter (mean 21 cf. 11 in our data set).

The plot of $\delta^{13}\text{C}$ values vs $\delta^{15}\text{N}$ values shows a very good correlation (Fig. 7.6; $r^2 = 0.89$) with very few outliers which gives us confidence that the organic matter sources can be reasonably modelled by varying the proportion of the marine and terrestrial end-members.

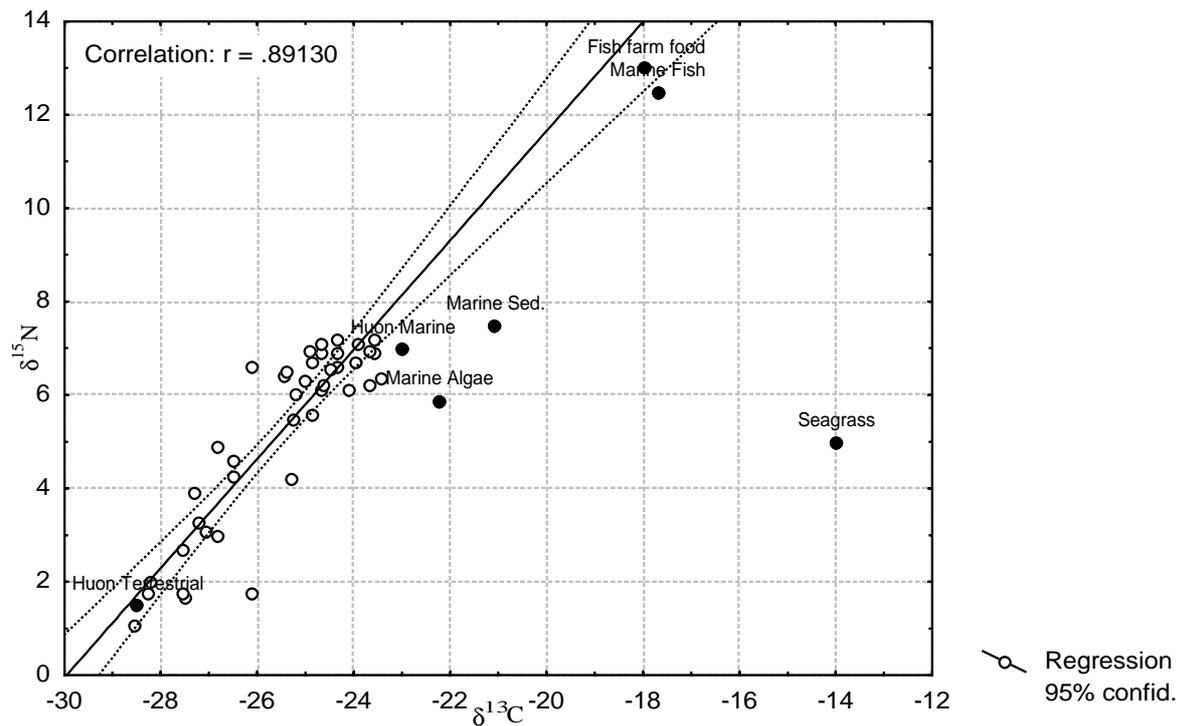


Fig. 7.6 Plot of $\delta^{13}\text{C}$ vs $\delta^{15}\text{N}$. The good correlation supports the model that organic matter in sediments is largely controlled by mixing of marine and terrestrial end-members. Typical values are shown for some marine organisms, but it should be recognised that values can vary with species, environmental conditions and season.

Additional evidence of terrestrial sources can be obtained from the sterol distributions (Fig. 7.7). Sitosterol is the main sterol produced by terrestrial plants (along with smaller amounts of stigmasterol and campesterol). Small amounts are produced by some phytoplankton species, but in general it is a good marker for terrestrial plants (Volkman 1986). Sitosterol comprises over 40% of the total sterols in the upper estuary and still represents 15–25% of the sterols in the lower estuary where aquatic sources become more significant. A plot of sitosterol concentrations versus the percentage of terrestrial-derived organic matter determined from $\delta^{13}\text{C}$ values (Fig. 7.8) shows a very good relationship ($r^2 = 0.81$), indicating that in this environment sitosterol content is a good predictor of terrestrial-derived organic matter in sediments. Other indications of a substantial terrestrial input are high abundances of long-chain fatty acids, hydrocarbons and alcohols (data not shown).

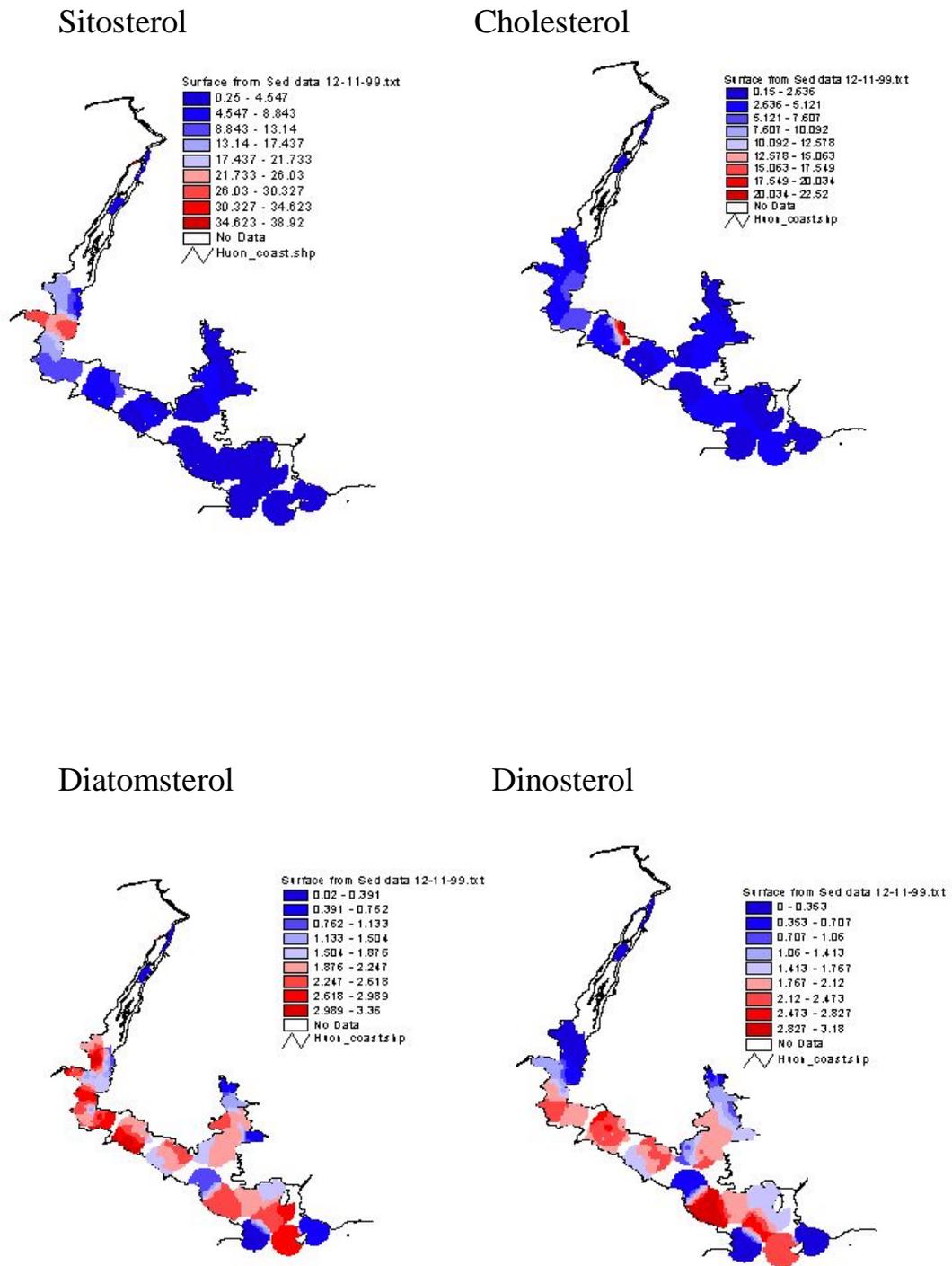


Fig. 7.7 Distribution of sterol biomarkers ($\mu\text{g/g}$ dry sediment) in surface sediments from the Huon estuary (sitosterol: higher plants; cholesterol: animals; diatomsterol: diatoms; dinosterol: dinoflagellates).

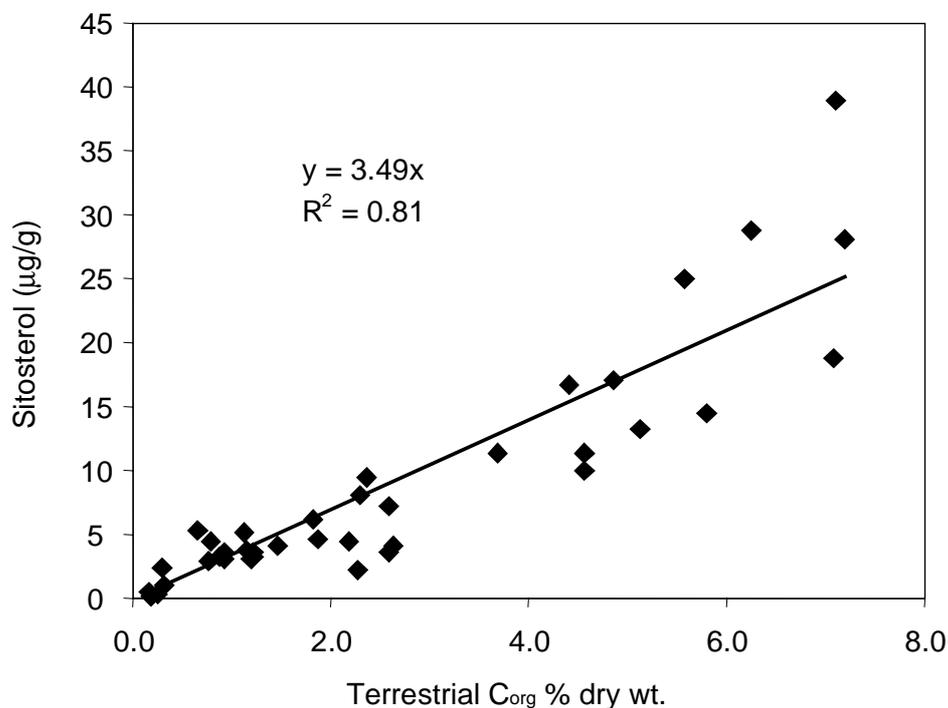


Fig. 7.8 Plot of sitosterol contents ($\mu\text{g/g}$ dry sediment) in sediments versus the content of terrestrial organic carbon (% dry weight) in the sediments estimated from $\delta^{13}\text{C}$ values.

7.3.5 Phytoplankton as a source of organic matter

The $\delta^{13}\text{C}$ values clearly show a trend for organic matter in the upper reaches of the estuary to be mainly terrestrially-derived, while in the middle and lower estuary it is a more mixed terrestrial-aquatic signature. Indeed, depending on the choice of the value of $\delta^{13}\text{C}$ for the aquatic (phytoplankton) end-member, the percentage of aquatic organic carbon at these middle and lower estuary sites is at least 55%, and some are as high as 75%. C:N values for the sediments also show a high proportion of samples with values typical of those found in marine phytoplankton (Table 7.1). Sites A7 and L1 gave anomalous values (i.e. greater than expected N content) presumably reflecting errors in measuring such low organic matter contents.

The sterol distributions provide additional information about the aquatic/phytoplankton-derived organic matter. Sterol distributions in microalgae vary considerably (Volkman 1986), so in favourable cases it is possible to distinguish different algal sources by the sterols found in the sediment. A common sterol in diatoms is 24-methylcholesta-5,22E-dien-3 β -ol (commonly called diatomsterol). This is an abundant sterol in most of the mid-estuary sediments (Fig. 7.7), which is consistent with the observation that diatoms dominated the phytoplankton in the year before the sediments were sampled. Dinoflagellates synthesise a variety of 4-methylsterols,

including dinosterol, but interestingly the latter was not particularly abundant in most of the sediments and below detection in a few (Fig. 7.7). It may be that if the sediments were sampled today, the values might be higher due to the strong dinoflagellate blooms that have occurred in the past two years. Both sterols are patchily distributed in the sediments, suggesting that they are more common in certain areas. The toxic dinoflagellate *Gymnodinium catenatum*, which blooms in the Huon estuary (see Chapter 5), has a distinctive distribution of sterols (Hallegraeff et al. 1991), but this signature was not apparent in the sediment sterol profiles, possibly because the microalga had not bloomed in the three years before the sediment sampling.

Phytol was detected in most sediments at concentrations similar to those of the major sterols. Its ubiquity in the estuary attests to the importance of aquatic sources of organic matter, even in the upper freshwater reaches of the estuary. There was no consistent trend between phytol levels and the proportion of aquatic organic matter as determined by $\delta^{13}\text{C}$ values (data not shown), suggesting that phytol (which is chemically and biologically quite labile) is due to the presence of reasonably fresh, labile organic matter of aquatic origin. At shallow water sites such as N1, much of the phytol may be due to chlorophyll in benthic microalgae (microphytobenthos). Note that at this site the C:N value of 11.3 is only slightly greater than the range 6–9 found in microalgae suggesting the presence of nitrogen-rich phytoplankton-derived organic matter. It might be possible to use the content of phytol as an index of labile organic matter once representative values of phytol to organic carbon in phytoplankton from the Huon estuary are determined.

7.3.6 Are fish farms a significant source of organic matter in the estuary?

Based on overseas research, the daily deposition of organic matter from a cage of 4,000 fish is about 10 kilograms of solid waste. It is quite clear from our study, and observations on the fish farm, that organic matter composed of uneaten food and fish faeces builds up under salmon cages. However, there is little evidence to suggest that it is physically transported to any great extent outside of the cage boundaries under normal river flow. There is of course the potential for considerable transport and redistribution of sediment and its associated organic matter during flood events, but there were none during the study period. Organic carbon contents in the sediments show considerable variability (Fig. 7.3), but much of this can be accounted for by variations in the mud content of the sediment.

The $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of the feed and fish faeces are similar to those found in open-marine samples and thus they provide a reasonably sensitive method for distinguishing fish-farm waste in a background dominated by natural, terrestrially derived, organic matter such as that found in the upper estuary (Fig. 7.3). Unfortunately, the upper estuary farms are near the elbow where the $\delta^{13}\text{C}$ and sediment characteristics change dramatically (Figs 7.3 and 7.4), so it is difficult to discern any influence from the fish farms from the $\delta^{13}\text{C}$ data alone. However, when ^{15}N data are included (Fig. 7.3), it is clear that the isotope signatures of all the sediments studied are very different from the values obtained for fish feed or fish faeces.

The biomarker data provides additional information. Both the fish feed and fish faeces are rich in the animal sterol cholesterol. This is not a unique marker: cholesterol occurs in almost all animals and it has a variety of natural aquatic sources. In the sediments, ratios of cholesterol to phytol (from chlorophyll *a*) almost all fall into a narrow range of 0.5 – 1.2 (data not shown), implying that natural aquatic sources are the main source of cholesterol at almost all sites in the estuary. The one significant exception is site

F3, which has a cholesterol/phytol ratio of 7.3, over seven times the mean of the other sediments. This site is adjacent to the fish farm at Wheatley's Bay and clearly there has been some contribution of organic fish-farm-derived organic matter off the farm. However, the ratio is not elevated at other sites on the F transect, or at sites immediately downstream, so the organic matter enrichment is fairly localised. It is likely that this enrichment is due to unintentional siting of fish farm outside of the lease area in previous years (TSGA, personal communication 2000).

Some of the sites near to fish farms show higher contents of organic matter than would be predicted from the proportion of mud in the sediment (see section above) but, as already discussed, only very few instances of this enrichment seem to be due to fish farm operations. Some sites – e.g. E1 near Desolation Bay and S14 and S15 in Deep Bay (all near shellfish sites) and B1 near Hideaway Bay – do not seem to have elevated organic carbon contents.

7.3.7 Is sewage pollution a problem in the Huon estuary?

The DPIF undertook two surveys of faecal contamination in January and June of 1997 in which *Escherichia coli* was measured in water samples from 49 sites in streams entering the Huon estuary (Bobbi 1998). These snapshots of faecal contamination indicated 11 of the 49 sites had *E. coli* counts above 600 cfu 100 mL⁻¹ during the summer, while 3 sites had elevated levels during the winter. While these concentrations constitute significant faecal contamination (see water quality guidelines, ANZECC 1992), the data indicated the 'hotspots' were localised, with neither the upstream nor downstream sites having the same high concentrations. In some cases the high coliform counts could be linked to visible activities (e.g. flocks of birds, cattle, proximity to domestic effluent treatment), but not always. Examination of sediments in the estuary provided an opportunity to assess the sources and contribution of faecal-derived organic matter to the Huon.

Over the past decade, staff at CSIRO Marine Research have developed specialist methods to trace sewage-derived organic matter in the marine environment (e.g. Leeming et al. 1996). One method measures the abundance of certain steroidal compounds (5 β - and 5 α -stanols and their Δ^5 -sterol precursors) found in the faeces of humans and other animals. The abundance and ratios of these compounds can indicate the extent and likely source of the faecal contamination. The principal human faecal sterol is coprostanol (5 β (H)-cholestan-3 β -ol), which constitutes \approx 60% of the total sterols found in human faeces (Leeming et al. 1996). The C₂₉ homologue of coprostanol is 24-ethylcoprostanol (24-ethyl-5 β (H)-cholestan-3 β -ol) which is the principal faecal sterol excreted by herbivores.

It is important to note that small amounts of 5 β -stanols relative to other sterols and stanols can be produced in sediments not contaminated by faecal pollution (Nishimura 1982), if the sediments are strongly anoxic. A measure of this *in situ* background level is the ratio of coprostanol to 5 α -cholestanol (5 α (H)-cholestan-3 β -ol; Leeming and Nichols 1998). Ratios greater than 0.5 indicate 5 β -stanols have been preferentially reduced from Δ^5 -sterols by gut microbiota, whereas ratios less than 0.3, usually indicate non-preferential reduction by communities of anaerobic bacteria in sediments, and thus may not be of faecal origin (Fig. 7.9). 5 β -Stanols such as coprostanol do not occur naturally in fresh or marine waters or in fully oxic sediments, because only anaerobic bacteria appear capable of hydrogenating Δ^5 -sterols to 5 β -stanols.

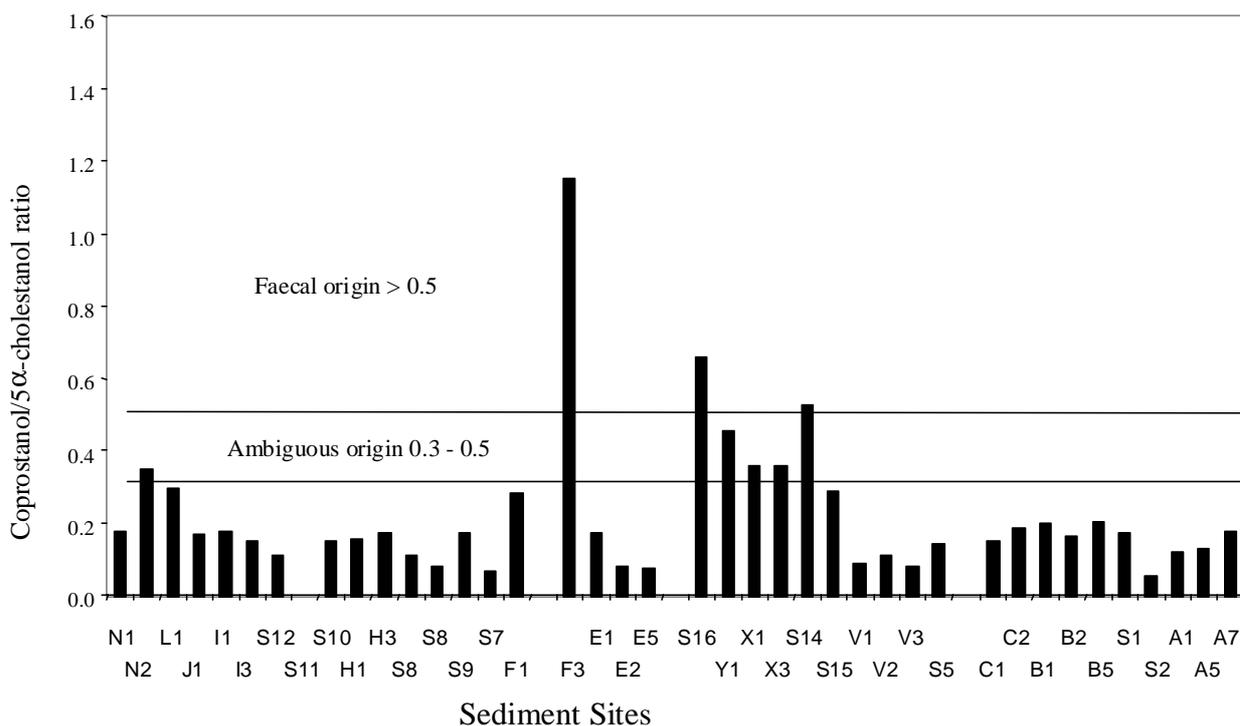


Fig. 7.9 Ratio of the faecal sterol marker coprostanol to 5 α -cholestanol in Huon estuary surface sediments. Faecal contamination is usually present when the ratio exceeds 0.5, and may be present in the range 0.3-0.5. Most sediments gave values much less than 0.3 indicating a lack of measurable faecal contamination.

Small amounts of 5 β -stanols were found in all of the Huon estuary sediments. However, except for sites close to the township of Cygnet, human sewage is not the source. Rather small relative amounts of these compounds are produced in the sediments from reduction of the corresponding sterols. The herbivore faecal biomarker 24-ethylcoprostanol was also found in nearly all sediments, but generally only as a trace component of the sterol fraction and one whose origin, at least in sediments, was ambiguous.

The ratio of coprostanol / 5 α -cholestanol at most sites was below 0.2, clearly indicating an *in situ* origin. Only three sediment samples stood out as possibly containing some organic matter derived from human sewage (i.e. ratios > 0.5, Fig. 7.9). Two of these were sites S16 and S14, both in the northern half of the Cygnet arm of the estuary close to the Cygnet sewage treatment plant (STP) outlet. Indeed, all sites north of the V transect had a 5 β /5 α stanol ratio consistently above 0.3. The sediments in the northern half of the Cygnet arm, therefore, reflect some inputs of human sewage from the township of Cygnet, but the inputs are highly diluted. Sediments at S16 and Y1 contain about twice the amount of organic carbon than expected based on their mud content (Fig. 7.5), but the amounts are only slightly higher at S14 and S15 (Fig. 7.5). No evidence of human faecal contamination from the township of Huonville could be detected, probably because particulate matter from the Huonville STP is transported further and dispersed more widely before deposition.

Usually the absolute concentrations of coprostanol can be compared between locations to give a broad comparison of contamination levels. However, because of the *in situ* production of 5 β -

stanols in all the Huon sediments, this approach was not meaningful. The remaining measure by which the load of sewage-derived organic matter to the Cygnet sediments can be gauged is a comparison of the $5\beta/5\alpha$ stanol ratio. The ratio $5\beta/5\alpha$ stanols in human faecal matter is greater than 15 (Leeming et al. 1998), but after faecal matter material is diluted in receiving waters this ratio is generally much reduced. However, previous surveys of sediments have shown that sites with significant human faecal contamination (i.e. where receiving waters do not meet the primary contact limit of 150 cfu 100 mL⁻¹ of thermotolerant coliforms) have $5\beta/5\alpha$ stanol ratios greater than 1-2 (Leeming and Nichols 1998). On this basis, with $5\beta/5\alpha$ stanol ratios less than 1, the load of human sewage to the Cygnet arm is only a very small proportion of the overall organic matter load.

The other site at which the C_{27} $5\beta/5\alpha$ stanol ratio was above 0.5 was F3. However, this site is unlikely to be contaminated by human sewage. Firstly, there is no clear trend of faecal contamination at any of the adjacent sites. Site F3 is very close to a fish farm and the sediment at this site appears, from the high cholesterol levels found there, to be affected by fish farm-derived organic matter. Coprostanol is not present in fish feed and present at only trace levels in the fish faeces. The ratio of 5α -cholestanol to cholesterol in the fish faeces was 0.14 to 0.28 and the ratio of coprostanol to 5α -cholestanol was 0.05 to 0.14. These ratios are less than those found in uncontaminated sediments. In terms only of the ratio of $5\beta/5\alpha$ stanols, the addition of fish food and/or faeces to adjacent sediments would decrease the ratio. However, in this case, we suggest that the levels of coprostanol at this site are higher than expected due to changes in the benthic microbial community structure caused by the increased load of organic matter from the nearby fish farm. Evidence for this contention comes from the presence of the full suite of possible C_{27} and C_{29} stanols. For the C_{27} stanols, epicoprostanol ($5\beta(H)$ -cholestan- 3α -ol) was present in equal amounts to coprostanol and the rarely identified $5\alpha(H)$ -cholestan- 3α -ol was also present in significant quantities. This stanol profile is not indicative of human faecal contamination. Rather, the presence of all four of the possible stanol isomers (from C_{27} and C_{29} Δ^5 -sterol precursors respectively) is characteristic of permanently anoxic sediments (Volkman unpublished data).

In summary, even though the generally high levels of organic matter present in Huon sediments reduced the sensitivity of the biomarker approach [in sediments] to some extent, it is clear that human faecal matter inputs contribute an extremely small proportion to the overall organic matter inputs, even near the Cygnet STP. Herbivore faecal inputs could not be unambiguously differentiated, but they must be an exceedingly small component of the organic matter input.

7.4 Summary and Recommendations

Sediments in the Huon estuary can have very high organic matter contents, although perhaps not as high as indicated by previous studies using the loss-on-ignition method. Organic carbon values are much higher than found in typical oligotrophic estuaries and are consistent with the classification of the estuary as mesotrophic. There is considerable variability in amounts, which is strongly influenced by the content of mud in the sediments. From a combination of stable carbon and nitrogen isotope signatures and lipid biomarker data, it has been possible to identify the main sources of organic matter in the estuary. For example, the plant sterol sitosterol was used as an indicator of terrestrially derived organic matter. Much of the organic matter in the upper estuary sediments is of terrestrial plant origin. At least in the Hospital Bay area, and

perhaps elsewhere, a significant proportion may be derived from wood waste from the former wood mill. In the middle and lower estuary, most of the organic matter is derived from phytoplankton production in the water column. Fish farms are a significant source of organic matter, but at most sites this is confined within the boundaries of the farm. We did find some evidence of fish farm-derived organic matter in sediments outside the farm at Wheatley's Bay, although even here the effect was quite local. Sewage is not a significant source of organic matter in most parts of the estuary, apart from a relatively minor input at Port Cygnet.

Some further work would be valuable to more fully characterise the sources of organic matter and to examine the susceptibility of different fractions to biological and chemical degradation. Biomarkers for unambiguously identifying fish feed, fish faeces and wood waste would be valuable. The use of stable isotopes shows great promise, and we recommend that the approach be extended to include stable nitrogen isotopes and that the signatures of a wider range of potential sources be determined. The sediment sampling was designed to provide an overall picture of the estuary, rather than to investigate specific sources in detail. There is obvious scope for further, more narrowly targeted, sampling of small sections of the estuary to investigate specific sources such as sewage and off-farm transport of organic matter. Given the importance of grain-size as a determinant of organic matter content in sediments, we recommend that any further sediment studies should include this measurement and that reference sites be carefully chosen so that their sedimentological characteristics match those of sediments in the study area. Loss-on-ignition has been shown to yield unreliable data for high-mud sediments, so we suggest that organic carbon contents should be determined directly in any further work.

7.5 References

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8 FOLLOWING OF MARINE FARMS

8.1 Introduction

Sedimentary organic matter will accumulate under fish cages if biochemical degradation and physical processes remove less than the farming activities put in. The accumulation of highly labile organic matter can modify the sediment's characteristics and processes, leading to sediment anoxia and generation of hydrogen sulphide (H_2S) and methane (CH_4), which are highly detrimental to farmed fish (Weston 1990; Ye et al. 1991; Findlay et al. 1995; Wu 1995). Labile organic matter can also induce a substantial change to the benthic communities directly under and near to fish cages (Ritz et al. 1989; Weston 1990; Ye et al. 1991; Wu 1995). Remineralisation of deposited organic matter also generates dissolved nutrients which, if present in high concentrations and other environmental conditions are appropriate, can lead to enhanced phytoplankton abundance and the possibility of detrimental algal blooms (Caffrey 1995; Harris 1997).

The effects of fallowing on sediments underlying fish cages used to rear sea bream and bass were examined by Karakassis et al. (1999). They found that after 11 months of fallowing, the geochemical conditions (TOC, TON, Eh, chlorophyll, phaeopigments) 10 m from the centre of the original fish cage had returned to values typical of a reference site. However at the centre of the site, the geochemical conditions were highly variable over the 23 months of the study. At this time a high proportion of the benthic fauna consisted of opportunistic species indicating that the site had still not returned to baseline. A seasonal release of nutrients from the sediments was noted resulting in a benthic algal bloom at 19 m water depth despite the low light levels at that depth.

In Tasmania, farmed salmon are fed specially pelletised food which is provided to the fish either by hand feeding or by computerised feeding systems. Recent improvements in feed composition and feeding regimes have reduced wastage, but still a substantial amount of organic carbon (and nitrogen) is deposited to the sediment either as uneaten food or as fish faeces. In the Huon Estuary, the normal practice is to move fish cages periodically to another site so that the sediments can be fallowed. In a study of sediment respiration rates, Woodward et al. (1992) showed that they returned to baseline in 185 days at a site at Wheatley's farm, and in 544 days at a fully marine site at Nubeena. However, when the Huon Estuary project began the dynamics of fallowing had not been extensively studied, and the optimal time required for fallowing had not been determined. Accordingly, we carried out a pilot study to investigate the distribution of accumulated fish waste and the rate at which organic deposits degrade in sediment under and near to fish cages. We hoped to estimate the time needed for the sediment to recover so that fish could be restocked, but this stage was not reached in the 12 months allocated to the study.

To trace the sources and fate of organic matter, we used chemical indicators of protein, lipid and organic matter components. For example %N and $\delta^{15}N$ are indicators of the protein component, fatty acid and sterol profiles are indicators of the lipid component, and %C and $\delta^{13}C$ are both bulk parameters of total organic matter. Measurements of total organic matter alone were not adequate given the high concentrations of naturally-occurring organic matter in the Huon estuary so we needed to develop techniques that would allow us to monitor organic matter

associated with the fish farming itself. The work presented in this Chapter resulted from an opportunity provided to CSIRO by Huon Aquaculture Pty Ltd and the Marine Resources Division of the Tasmanian Department of Primary Industry & Fisheries (Marine Environment, and Research and Assessment Branches; now DPIWE) to collaborate on their fallowing study. These results have been prepared for publication (McGhie et al. 2000)

8.2 Study design

The trial was held at the Huon Aquaculture Pty Ltd lease at Hideaway Bay. Transects were established at two cage sites. Site 1 was located at a depth of 15 m and the transect extended for 30 m westward (up river) with sampling stations at 0, 10, 20, and 30 m intervals from the centre of the cage. At site 2 the water depth was 24 m and the transect extended from the centre of the cage eastwards (down river) for 32 m with stations located at 0, 12, 22, and 32 m intervals. The sediment types differ between the two sites: sediment at Site 1 is coarse sand (1.2% silt/clay <63 μm grain size) whereas at Site 2 it is fine mud (79% silt/clay). The water currents (measured within the farm) have a recorded mean of $0.007 \text{ m}\cdot\text{s}^{-1}$ at 4 m off the bottom, and rarely exceed $0.1 \text{ m}\cdot\text{s}^{-1}$. Water current velocities were measured by an ADCP (Acoustic Doppler Current Profiler) between 13 January and 3 March, 1998 producing a velocity profile from 7 m to 23 m in 36 m of water every 10 minutes.

Two reference sites with sediment characteristics similar to the experimental sites were selected. Reference Site #1 contained 7.7% silt/clay and was 1.4 km from Site 1 and 1.1 km from Site 2. Reference Site #2 contained 91.6% silt/clay and was 0.7 km from Site 1 and 0.5 km from Site 2. Both controls sites were considered to be unaffected by fish farm operations. Results for the control sites are included to provide a comparison with results from fish-farm affected sites. However, as the sediment characteristics between experimental and control sites were not identical, the results should be interpreted with caution, as discussed later.

8.2.1 Sampling procedures

Sediment samples were collected when the fish cages were removed from each site and again 6 and 11 months later. Samples were taken in October 1996, April 1997 and in October 1997 (i.e. at 0, 6 and 11 months), after fallowing began. Three 50 mm diameter cores per station were taken by SCUBA-equipped divers. The redox profile (1 cm intervals) was recorded by inserting a pH/redox probe into the sediment to a depth of 5 cm. Portions of sediment from the top 3 cm were analysed for organic matter, %C, $\delta^{13}\text{C}$, %N, $\delta^{15}\text{N}$, lipids and particle-size. Sediment samples were stored frozen until analysed.

Sediment dry weight was determined by drying sediment overnight at 65°C . The percentage of organic matter was calculated as the loss of weight on ignition (LOI; Greiser and Faubel 1988). Subsequent work revealed that this commonly used technique can overestimate the amount of organic matter present in sediments (Leong and Tanner 1999; see Chapter 7), so comparisons based on LOI data may not always be valid, especially if sediment types vary greatly. Organic carbon and nitrogen contents and stable isotope values in sediments were measured with a Fisons NA 1500 NCS Elemental Analyser coupled to a Finnigan-MAT Delta S isotope ratio mass spectrometer. Sediments for TOC and ^{13}C analysis were decarbonated with HCl before analysis. Total lipids were extracted by a modified Bligh-Dyer method using chloroform-methanol and analysed by capillary gas chromatography with flame ionisation detection.

Component concentrations were calculated from peak areas referenced to those of internal standards: 5 α -cholestane for the sterols and 19:0 fatty acid methyl ester (FAME) for the fatty acids.

8.3 Results and Discussion

8.3.1 Lipid biomarkers

To identify lipid biomarkers that might be useful for following the fate of fish-farm-derived organic matter we determined the major lipids in fish feed and faecal matter from salmon and trout (Table 8.1). The major sterol in fish food is cholesterol, with minor amounts of 5 α -cholestanol, sitosterol, and 5 α -sitostanol. Cholesterol was also the major sterol in fish faeces, but at much lower concentrations than in fish food. Two of the microbial degradation products of cholesterol (i.e. coprostanol and epicoprostanol) were present in faeces at low concentrations, whereas they were not present in the fish food. Fish faeces contained all of the major fatty acids present in the food, but the absolute concentrations were substantially reduced and the relative concentrations differed between food and faeces (Table 8.1). Presumably the differences between food and faeces result from differential adsorption and excretion of fatty acids during passage of the food through the fish gut.

Table 8.1 Fatty acid and sterol composition of fish food and faeces.

	Food (5 mm) ¹		Food (mixed sizes)		Trout faeces ²		Salmon faeces	
Fatty acids	mg/g dry wt				mg/g dry wt			
14:0	7.1	(36) ⁴	6.7	(37)	0.18	(27)	0.37	(24)
16:1 ω 7	8.5	(44)	8.0	(44)	0.026	(4)	0.07	(4)
16:0	19.4	(100)	18.2	(100)	0.68	(100)	1.57	(100)
18:1 ω 9	11.7	(60)	11.3	(62)	0.068	(10)	0.18	(11)
18:0	4.5	(23)	4.3	(24)	0.220	(32)	0.51	(32)
20:5 ω 3	16.4	(84)	15.8	(87)	0.019	(3)	0.06	(3)
22:6 ω 3	13.8	(71)	12.8	(70)	0.028	(4)	0.12	(8)
Sterols								
coprostanol	n.d. ³		n.d.		0.001	(1.4)	0.002	(1.9)
epicoprostanol	n.d.		n.d.		0.001	(1.0)	0.001	(1.1)
cholesterol	12.7	(100) ⁵	15.5	(100)	0.091	(100)	0.084	(100)
5 α -cholestanol	0.49	(3.9)	0.63	(4.0)	0.026	(29)	0.012	(14)
sitosterol	0.20	(1.6)	0.26	(1.6)	0.026	(29)	0.017	(20)
5 α -sitostanol	0.04	(0.3)	0.05	(0.3)	0.053	(5.8)	0.004	(4.6)

1 Food samples provided by Huon Aquaculture. Two samples were provided. One had 5 mm diam. particles, whereas the other contained mixed sizes.

2 Fish faeces were collected by Huon Aquaculture and represent a composite sample of 5 fish.

3 n.d. = not detected.

4 The values in brackets are concentrations normalised to the 16:0 concentration.

5 The values in brackets are concentrations normalised to the cholesterol concentration.

6 Fatty acids are designated x:y ω n where x is the number of carbon atoms, y is the number of double bonds and n is the position of the closest methylene-interrupted double bond to the methyl carbon atom.

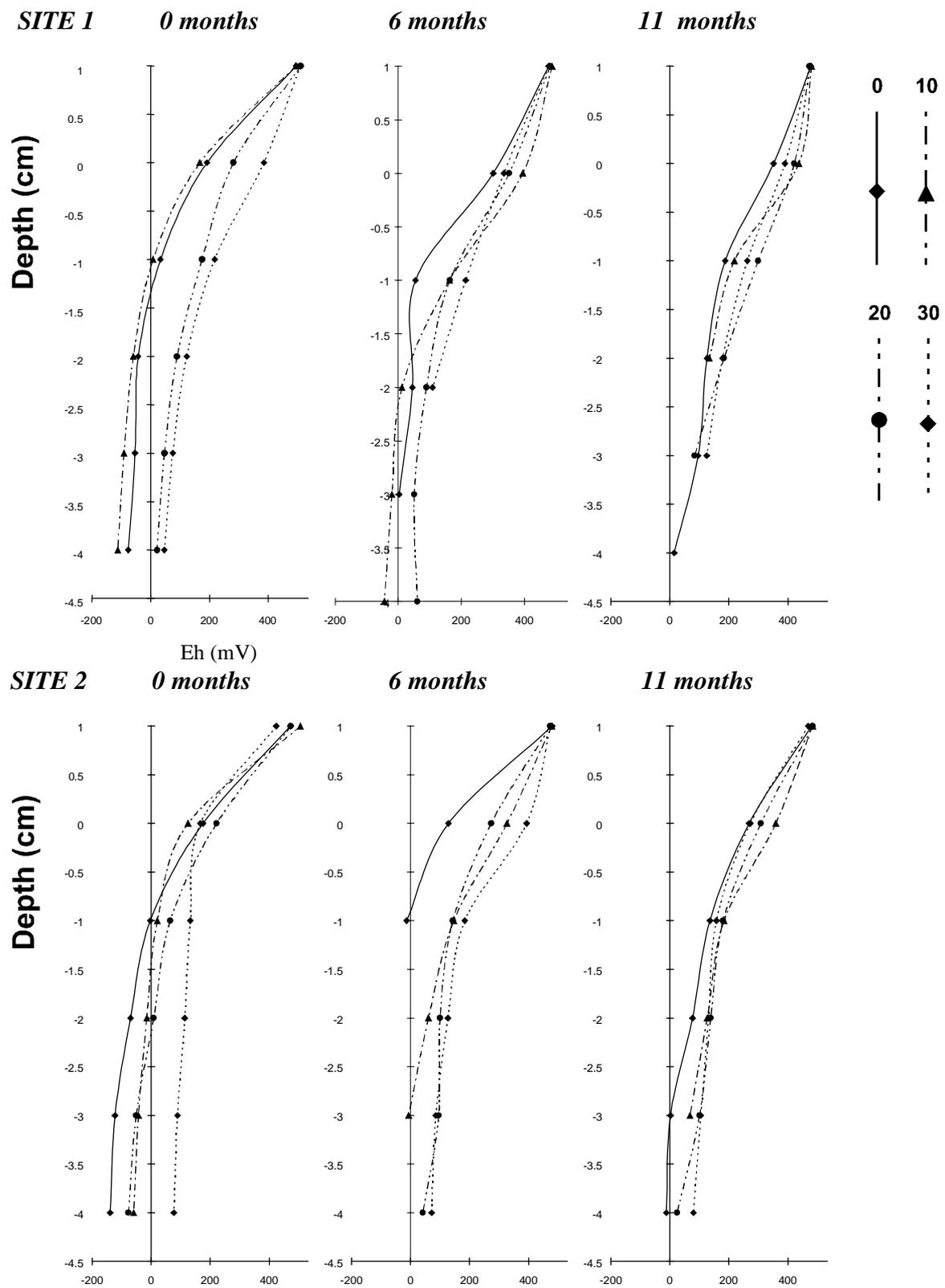


Fig. 8.1 Plots of redox (Eh in millivolts) profiles for each site after 0, 6, and 11-months following. Three cores were collected at each sampling position by SCUBA divers and the redox potential measured in the sediment profile. Profiles refer to 0, 10, 20 and 30 m distances from the centre of the cage.

8.3.2 Organic matter and redox potentials in sediments

Redox profiles (Eh in millivolts) for the sediments are presented in Fig. 8.1. At the start of fallowing, the sediment underneath the fish cages was depleted in oxygen, compared with sediment outside the area of the cages. After 11 months of fallowing, the difference between these sediments had lessened, but was nonetheless still obvious. All stations at Site 1 had similar redox potentials, whereas surface sediment at the 0 m station at Site 2 was still depleted in oxygen.

Organic matter content (determined by loss-on-ignition rather than by direct measurement) were very different at the two sites (Table 8.2). Sediment at Site 1 had less organic matter (LOI range 3.5 - 5.6%; c.f. reference site 4.2%), consistent with its higher sand content, compared with sediment at Site 2 (range 11.5 - 21.6%, c.f. reference site 18.1%). The high organic matter content of sediment at Site 2 was similar to much of the sediment found throughout the Huon Estuary.

The organic matter content of the sediment generally declined with distance from the centre of the fish cage. The difference between the values observed under the cage (0 and 10 or 12 m) and at 30 m indicated that fish farming operations were contributing approximately 1.5 - 3% organic matter to the sediment. Sediment at Site 2 directly under the centre of the cage (0 m) contained fish food, as a result of an accidental spillage, so the organic matter content was artificially high at the start of fallowing. Organic matter concentrations at the centre of the cages declined over 11 months, but were still higher than the sediment at 30 m from the cage centre. Similarly, the $\delta^{13}\text{C}$ values were generally higher at the centre of the cage and declined away from the cage, consistent with carbon enrichment under the cage (Table 8.2).

The more positive $\delta^{13}\text{C}$ values of sediment under the cages indicate enrichment with carbon from a marine source, in this case fish food. The feed used contains a large component of marine fish meal and has a $\delta^{13}\text{C}$ value more positive than sediments in the area of the fish farm (Table 8.2). Percent N was also higher at the centre of the cages than in sediments 30 m from the cage centre (Table 8.2). There is a reduction of %N at both sites during the 11 months of fallowing. The $\delta^{15}\text{N}$ values for sediments with elevated %N concentrations were higher than values for sediments 30 m from the cage centre, which indicates that the additional N in these sediment samples was derived from a higher trophic level – in this case the fish meal used in the fish feed (Peterson and Fry 1987).

8.3.3 Lipid biomarkers in sediments

Lipid concentrations in sediments were greatest directly underneath the fish cages at both sites and decreased away from the cages, indicating the limited spread of fish farm waste to surrounding sediment. At approximately 30 m from the cages, concentrations were still higher than at the reference sites. Similar distributions of organic matter were found at another fish farm site in Tasmania (Ye et al. 1991) and elsewhere (Findlay et al. 1995; Henderson et al. 1997). The relatively small amount of organic matter deposited outside the foot-print of the cage is most likely due to the relatively slow water currents in the Huon Estuary during the 11-month study period.

The results of the sterol and total fatty acid analysis of the sediment from the two sampling sites are presented in Table 8.3. Compounds related to fish food and faeces are listed together with terrestrial markers including the fatty acid 24:0 and long-chain alcohols 24:0 and 26:0. Both the non-saponified neutral lipids (sterol) and acidic fractions (fatty acids) contained minor compounds, which are not reported here.

Concentrations of major fatty acids present in fish food and faeces (16:1 ω 7, 18:1 ω 9, 20:5 ω 3, and 22:6 ω 3) were greatest in sediment at the centre of the cage and declined with distance from the cage at both sites. Previous studies (Johnsen et al. 1993; Henderson et al. 1997) also reported these specific fatty acids in sediments under fish cages which they took to indicative of fish farm waste.

The sedimentary fatty acid contents at Site 1 did not decline until the edge of the cage (10 m), suggesting that organic matter is deposited on the sediment over the entire area occupied by the cage. The concentrations of these four fatty acids declined over the 11-month fallowing period, but the rate of decline varied between sites and between sampling stations within sites. The highest rate of decline occurred at Site 2 directly under the centre of the cage. Here, the fatty acid contents were initially very high, consistent with the accidental spillage of fish food, but they declined so rapidly so that after 11 months of fallowing they were similar to those found at 0 m at Site 1. This rapid decline is to be expected, since the food contains a high proportion of readily degraded organic compounds. In contrast, at 30 m from the centre of the cages, the fatty acid components of the fish food showed little reduction over the 11-month fallowing period and remained higher than those at the reference sites.

The contents of fatty acids indicative of bacterial biomass (*iso*15:0 and *anteiso*15:0) were quite similar at all stations except slightly enhanced values at Site 1 (10 m) at the commencement of fallowing and Site 2 (0 m station) indicating that a greater bacterial biomass was present. Bacterial biomass (as indicated by *iso*15:0 and *anteiso*15:0) in sediment at the centre of the cage at Site 2 showed little change over 6 months, but then declined to reference values after 11 months. The concentrations of the 24:0 fatty acid, a marker of terrestrial plants, were generally higher at site 2 and showed little change with time apart from an anomalously high value at 0 m at site 2 which persisted for 6 months.

The main sterol in all sediments was cholesterol, with concentrations generally highest at the centre of the cages and declining with distance from the cage. Cholesterol concentrations declined with time, although more slowly than did food-derived fatty acids. The microbial degradation products of cholesterol (coprostanol, epicoprostanol and 5 α -cholestanol) were detected in the sediment and their abundance appeared to be related to cholesterol contents. The long-chain *n*-alcohols 24:0 and 26:0, like the fatty acid 24:0, are considered to be markers of terrestrial plant input. Their contents were relatively constant, although these alcohols, like the 24:0 fatty acid, were more abundant at Site 2 at the centre of the cage. Sterol biomarkers indicative of algae were detected at low concentrations indicating that the input of organic matter from microalgae (either phytoplankton or phytobenthos) is relatively small compared with input from fish food and faeces in these sediments.

Table 8.2 Organic matter, nitrogen and carbon contents of sediment from the two experimental sites.

	Distance from cage centre	Organic matter (LOI)	Nitrogen		Carbon	
	m	%	%	$\delta^{15}\text{N}$	%	$\delta^{13}\text{C}$
<i>Site 1</i>						
0 months	0	5.6	0.45	9.6	1.37	-21.72
	10	5.6	0.65	10.1	1.62	-20.04
	20	4.2	0.33	8.8	0.88	-24.53
	30	3.6	0.25	7.2	0.71	-24.95
6 months	0	5.8	0.38	9.5	1.07	-20.56
	10	5.0	0.44	9.8	1.46	-20.26
	20	4.1	0.29	8.4	0.93	-23.62
	30	3.6	0.28	8.2	0.65	-24.37
11 months	0	4.0	0.31	9.7	0.86	-21.13
	10	4.4	0.20	9.5	1.18	-22.46
	20	3.6	0.15	8.9	0.68	-23.34
	30	3.8	0.10	6.5	0.56	-23.57
<i>Site 2</i>						
0 months	0	21.6	1.31	10.5	5.56	-22.30
	12	14.5	0.74	8.2	4.24	-23.17
	22	12.9	0.78	8.5	4.03	-23.13
	32	11.8	0.69	8.1	3.80	-23.09
6 months	0	14.3	1.42	9.9	5.63	-22.03
	12	11.9	0.79	9.1	4.24	-21.86
	22	12.7	0.99	9.0	4.91	-22.30
	32	11.4	0.67	7.7	5.98	-23.47
11 months	0	14.8	0.82	9.1	4.0	-22.47
	12	9.9	0.64	9.5	3.45	-21.90
	22	11.9	0.55	8.5	3.41	-22.59
	32	11.8	0.44	7.6	3.26	-22.67
Control #1		4.2	0.05	6.41	0.65	-25.43
Control #2		18.1	0.49	6.96	4.05	-
Faeces #1			4.9	12.0	40.3	-17.4
Faeces #2			3.9	9.6	31.8	-18.1
Fish food			8.2	13.2	57.4	-18.5
Fish food			8.0	12.3	57.4	-18.2

8.3.4 Faecal and food input to the sediment

The fish-farm derived organic matter accumulating underneath fish cages will be fish faeces and uneaten fish food. Differences in the sterol and fatty acid compositions of fish food, and of salmon and trout faeces, raised the possibility that the fatty acid composition could be used to distinguish fish food from faeces (Table 8.1). The relatively high levels of 20:5 ω 3 (EPA) and 22:6 ω 3 (DHA) in fish food samples are consistent with mackerel as the oil source. Compared with food, the fish faeces contained a different relative composition of fatty acids (Table 8.1). Relative to 16:0, the concentration of the fatty acids 16:1 ω 7, 18:1 ω 9, 20:5 ω 3 and 22:6 ω 3 were all less in fish faeces than in fish food probably because they are preferentially absorbed by salmonids (Sigurgisladdottir et al. 1992). Our results suggest that differences in the ratios of the fatty acids between food and faeces can be used for estimating the amount of food and faeces input to sediment under fish cages.

At Site 1 (0 m), the similar ratios of fatty acids found in the faeces and sediment (Fig. 8.2) demonstrates that most of the fish-farm derived organic matter in the sediment is from fish faeces, although the relatively high amount of 18:1 ω 9 suggests there is also an input from fish food. In contrast at Site 2, the relative amounts of fatty acids indicate there had been a much larger contribution from fish food than from faeces, which is consistent with the accidental spillage that occurred here.

In contrast to fatty acids, the sterol profiles were not useful for discriminating between fish feed and faeces. The microbial transformation products of cholesterol (coprostanol and epicoprostanol) were not present in the food, but were present in small quantities in faeces. However, both products are produced in anoxic sediment by microbial degradation of cholesterol and so they are not always reliable indicators of faecal input in anoxic sediment samples (Nishimura 1982; see Chapter 7). The $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ values (Table 8.2) supported the inference that the sediments were contaminated with either fish food or fish faeces, but could not distinguish between the two.

8.3.5 Degradation of organic matter in sediments

The bulk parameters of organic matter indicated that both total organic matter (%C, $\delta^{13}\text{C}$) and protein contents (%N, $\delta^{15}\text{N}$) reduced over the 11-month study, but at neither site did the organic matter return to background levels, either in terms of total amounts or composition. The $\delta^{15}\text{N}$ and $\delta^{13}\text{C}$ values remained higher in the sediment under the cages than at 30 m distant. This result is consistent with total carbon fluxes calculated by Hall et al. (1990), which showed that greater than 80% of the organic carbon deposited on the sediment under fish cages is buried and about 20% of the carbon is returned to the water column.

Analysis of the fatty acid concentrations also showed that a portion of the organic matter remains after 11 months following (Fig. 8.2). Fatty acids degrade more quickly than bulk organic matter in marine sediments, but the rate varies according to degree of unsaturation (Haddad et al. 1992; Sun and Wakeham 1994; Canuel and Martens 1996; Sun et al. 1997). Polyunsaturated fatty acids degrade more rapidly than saturated fatty acids. Previous studies on fatty acid degradation were based on natural environments where supply of organic matter to the sediment is relatively constant. In our study, organic matter was deposited at both sites at greater than natural levels and then abruptly halted. The data indicate that the total

concentrations of fatty acids in the sediments declined over the 11 months (Table 8.3, Fig. 8.2), at rates that varied both between sites and between sampling stations.

After 11 months, at site 1 the fatty acid concentrations of 20:5 ω 3 and 22:6 ω 3 had reduced to 86% and 60% of initial concentrations. For the saturated 14:0, 16:0 and 18:0 fatty acids the corresponding values were 20%, 24% and 32%. This is at odds with the findings of previous studies which showed that unsaturated fatty acids degrade more rapidly than saturated fatty acids in anoxic sediments (Haddad et al. 1992; Canuel and Martens 1996; Sun et al. 1997). In contrast at Site 2, the initially high content of food-derived fatty acids 20:5 ω 3 and 22:6 ω 3 decreased to just 4% of initial values. The corresponding values for 14:0, 16:0 and 18:0 fatty acids were 6, 10, and 16% respectively, after 11 months. Since the starting concentrations were much greater at Site 2, the total amount of fatty acids degraded is far greater.

The rates of organic matter accumulation in sediment depends on how much material is deposited from the water column and how rapidly it is removed by chemical degradation or by physical processes, such as resuspension. Judging from the sediment organic matter contents outside the cages, transported material does not significantly augment the natural organic matter loadings. Note that at site 2 (Fig. 8.2), there is a small increase in food-related fatty acids at 20 m from the cage centre after 6 months, which suggests that some part of the deposited feed is mobile and can be transported for short distances.

The remineralisation of organic matter to inorganic nutrients (Rowe et al. 1975) is microbially mediated and the rate is dependent on temperature, concentration of organic matter, concentration of electron acceptors, and physical and chemical properties of the organic matter (Canuel and Martens 1996; Holmer and Kristensen 1996). For example, high concentrations of organic matter may stimulate the rate of microbially-mediated remineralisation in the sediment.

In this study, the physical conditions of the two sites (temperature, salinity, and oxygen concentrations) were similar, but the sediment characteristics were not. Also, the source of the fatty acids under the centre of the cages differed. As the rate of fatty acid reduction was greater at site 2, it appears that food-derived fatty acids degrade more rapidly than fatty acids derived from faeces. Faecal-derived polyunsaturated fatty acids may be more protected from degradation than those from fish food. The sources of fatty acids and the 'packaging' of the organic matter are believed to affect rates of degradation (Haddad et al. 1992; Sun and Wakeham 1994). Background total organic matter concentrations differed between the two sites and the microbial population may also differ. However, if it is assumed that the microbial populations responsible for fatty acid degradation were similar at these two sites (since they were in close proximity), the apparent difference in the rates and total amount of fatty acid degradation may be due to differences in the 'packaging' or the 'presentation' of fatty acids to the microbial decay processes occurring in the sediments.

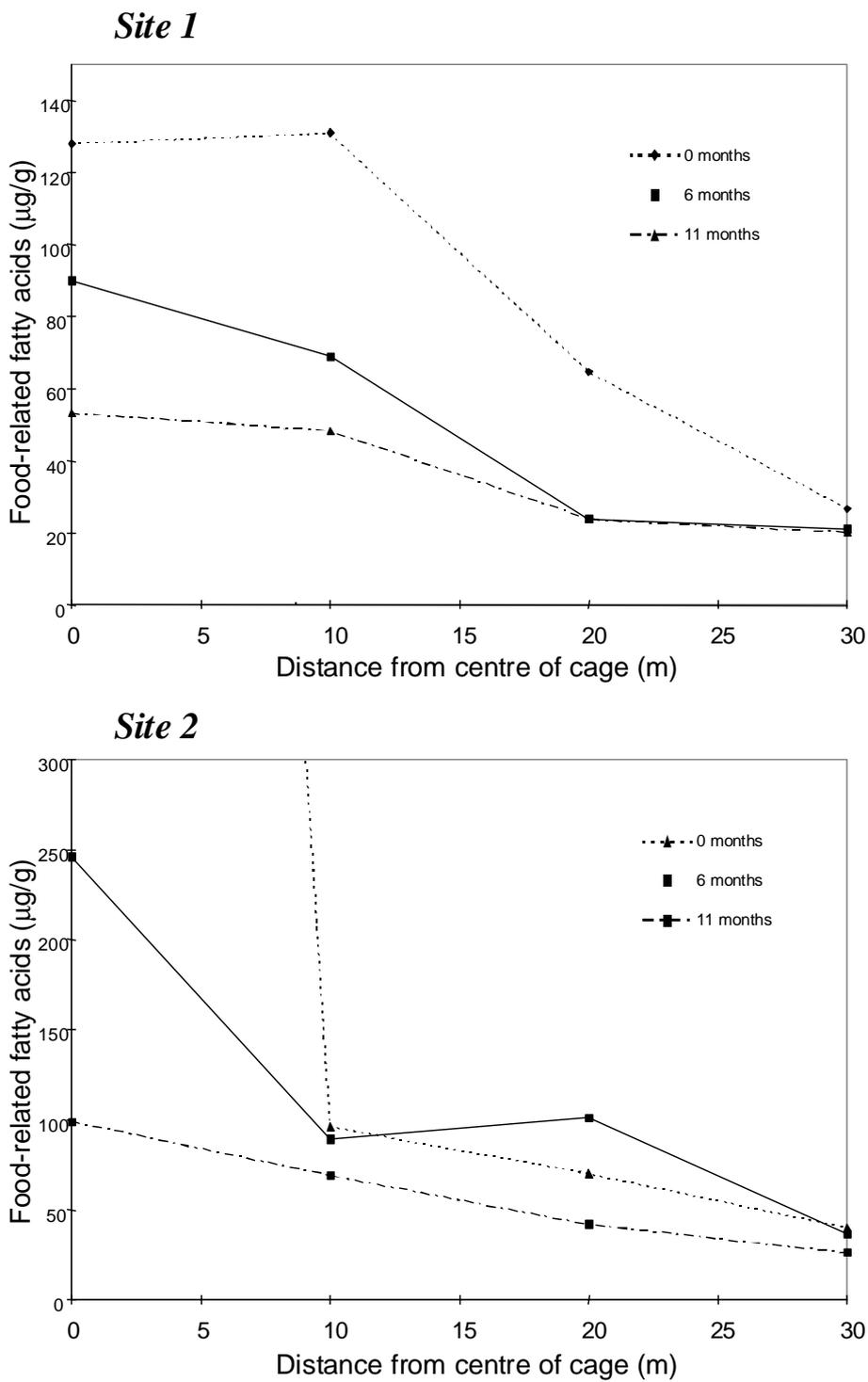


Fig. 8.2 The reduction in the concentration of food-derived fatty acids over the 11-month following period for Sites 1 and 2. Food-derived fatty acids included 16:1 ω 7, 18:1 ω 9, 20:5 ω 3, 22:6 ω 3 (note that these can also be derived from aquatic phytoplankton).

Table 8.3 Fatty acid, sterol, and long-chain alcohol concentrations in sediments.

Distance (m)	Lipid Components ($\mu\text{g/g}$ dry weight)																									
	Site 1												Site 2								Control					
	0 months				6 months				11 months				0 months				6 months				11 months				#1	#2
	0	10	20	30	0	10	20	30	0	10	20	30	0	12	22	32	0	12	22	32	0	12	22	32		
Fatty acids																										
14:0	43	40	17	4.6	17	12	4.6	4.0	8.5	8.0	3.4	2.7	328	21	16	9.6	59	20	25	9.6	21	15	9.6	6.3	0.8	3.1
iso 15:0	6.7	12	4.0	2.8	4.8	4.8	3.2	2.9	3.1	3.9	2.6	2.2	14	4.2	5.8	6.2	12	4.3	8.3	5.3	5.9	4.0	3.7	4.0	0.9	3.7
anteiso 15:0	4.2	11	5.4	4.8	4.9	5.7	5.6	5.2	3.1	4.7	4.0	4.0	6.5	4.7	7.1	8.4	10	4.6	9.3	7.5	5.5	3.9	4.3	6.0	1.1	4.3
16:1 ω 7	36	58	24	15	31	27	15	15	17	21	13	14	422	21	25	17	57	20	37	18	24	18	14	12	2.0	6.2
16:0	17	15	73	24	70	52	22	18	43	36	17	12	948	84	75	41	224	80	93	39	94	64	38	27	3.2	11
18:1 ω 9	64	49	23	6.3	25	19	5.0	3.5	16	13	5.2	2.6	680	36	22	12	76	28	28	9.1	29	22	12	7.0	0.8	2.8
18:0	57	50	26	6.9	27	22	7.4	4.4	18	17	7.0	2.9	230	31	28	14	79	30	35	13	37	26	17	11	0.8	3.0
20:5 ω 3	14	13	9.6	2.9	19	13	2.2	1.7	12	8.6	3.6	2.8	695	22	13	5.5	63	23	21	4.9	25	16	9.3	3.9	0.4	1.0
22:6 ω 3	14	11	8.1	2.4	15	9.9	1.6	0.9	8.2	5.8	2.0	0.7	488	17	10	5.4	50	18	15	4.3	21	13	6.6	3.2	0.5	1.3
24:0	3.9	4.4	3.1	1.6	3.5	2.7	1.9	1.5	2.9	2.8	2.2	1.5	9.0	7.7	8.3	7.4	20	8.5	11	8.1	14	7.8	7.0	8.0	1.4	6.5
bacterial FA ^a	11	23	9.4	7.6	9.7	11	8.8	8.1	6.2	8.6	6.6	6.2	21	8.9	13	15	22	8.9	18	13	11	7.9	8	10	2.0	8.0
food FA ^b	12	13	65	27	90	69	24	21	53	48	24	20	2285	96	70	40	246	89	10	36	99	69	42	26	3.7	11
Sterols																										
coprostanol	3.7	5.6	2.8	1.1	4.2	4.0	1.7	0.7	2.4	3.0	2.4	0.9	9.1	4.4	8.1	3.6	18	5.0	9.2	2.5	17	5.6	4.8	2.2	0.3	1.4
epicoprostanol	5.5	7.6	2.8	1.1	4.9	5.7	2.3	1.1	4.2	5.0	2.5	0.8	17	4.9	5.6	2.3	27	8.3	9.3	2.5	19	6.8	4.0	1.7	0.5	1.6
cholesterol	70	92	39	13	73	66	20	8.7	59	55	24	6.9	241	57	57	26	217	77	88	25	127	83	53	21	1.1	2.7
5 α -sitosterol	5.5	8.2	4.2	1.7	5.9	6.0	3.1	1.8	4.6	5.6	3.4	1.5	19	5.9	7.6	5.6	24	6.0	12	5.7	16	7.0	5.9	4.1	0.3	1.0
sitosterol	6.0	7.7	4.7	2.2	6.5	5.7	3.4	2.5	4.8	4.9	3.2	1.8	19	7.5	9.2	7.0	26	7.6	14	7.3	18	8.4	6.7	4.8	0.2	1.3
sitostanol	2.2	2.7	1.4	0.7	2.5	2.1	1.4	0.8	1.9	1.9	1.3	0.8	5.8	2.6	3.8	2.8	11	3.1	5.8	2.8	8.4	3.4	3.1	2.3	1.1	4.1
stigmasterol	1.2	2.2	1.3	0.4	1.8	1.4	1.4	0.7	1.1	1.2	0.9	0.4	2.9	1.7	2.2	1.7	7.3	2.9	4.1	2.0	4.4	2.1	1.1	1.3	0.1	0.4
24:0 alcohol	1.0	0.8	0.9	0.7	0.7	0.8	0.7	0.7	0.7	1.0	0.8	0.8	7.9	5.3	4.3	4.2	5.6	4.6	5.1	5.1	4.4	3.6	4.3	4.1		
26:0 alcohol	1.0	1.5	0.8	0.5	0.8	0.7	0.6	0.5	0.6	0.7	0.5	0.5	5.0	2.6	2.2	1.9	4.9	2.8	3.7	2.9	2.8	1.9	1.8	1.7		

^a markers for bacterial biomass - *iso* 15:0 and *anteiso* 15:0; ^b markers for fish food - 16:1 ω 7, 18:1 ω 9, 20:5 ω 3, 22:6 ω 3.

8.3.6 Is a one-year fallowing period sufficient?

At the start of the fallowing, the sediment directly under cages was contaminated with organic components originating from fish faeces (Site 1) and unconsumed food (Site 2). Redox measurements (Fig. 8.1) indicated that the sediments were anoxic; i.e. the supply of oxygen from the overlying water was less than the *in situ* consumption of oxygen in the sediment. At the end of 11-month's fallowing, the oxic conditions appear to have been re-established at the centre of both cages even though about 40% of the food-related fatty acids remained at Site 1. Furthermore, at Site 2, the concentration of food-related fatty acids at the end of fallowing was comparable with that found at Site 1 at the start of fallowing. Thus, it appears that the concentration of food-derived fatty acids in sediments was poorly related to sediment redox condition because fatty acids are but one component of the labile organic matter.

Our results for organic nitrogen compounds show that they are not readily degraded. Values of %N directly at the centre of the cages did not show appreciable reduction over the 11-month fallowing period. %N values were still higher at the centre of the cages than at 30 m distance even after 11 months. The $\delta^{15}\text{N}$ value indicated that the additional N remaining in the sediment was derived from fish-farming. The rate of oxidative degradation of organic matter from fish farming operations appears to decline during fallowing perhaps because the form of the organic matter is altered by the diagenetic processes, reducing its bioavailability.

From these data, it appears that a one year fallowing period should allow for surface sediment to return to oxic conditions. However, a more sensitive measure of sediment degradation is the direct measurement of components of fish food and faeces in the sediment. Such a measure shows that after a 11-month fallowing period, substantial organic matter remains undegraded in the surface sediments. A possible consequence of this residual organic matter is that the addition of fresh organic matter as a result of cage restocking might cause a rapid return to anoxic conditions in the sediment with concomitant environmental harm. An important issue for future research is to determine the rate of residual organic matter accumulation during repetitive stocking/fallowing and to determine the relationship between such organic matter accumulation and the ecologically sustainable use of fish-farming sites in the Huon Estuary.

8.4 Conclusions

In summary, this study has shown that waste from fish cages is discernible in the immediate area of fish cages in the Huon estuary, although small amounts of waste may reach beyond this. Furthermore, with efficient farming practices, most waste reaching the sediment is fish faeces rather than unconsumed food. After an 11-month fallowing period, the initial anoxic conditions of the surface sediment directly under cages returned to oxic conditions, even though a significant portion of the organic waste remained in the sediment. If the cage is restocked, there is a risk that the sediment may degenerate more rapidly than sediment that had not previously been subjected to waste from a fish cage.

This pilot study has highlighted many of the questions that need to be answered to successfully monitor the effects of fallowing. Clearly the choice of sample and reference sites needs careful consideration in an environment where sediment grain size (and hence organic matter content) varies greatly as it does in the Huon. For example, the second reference site had a much higher

organic loading than most of the sediments at Site 2. Redox measurements can provide a useful guide to sediment oxidant status, but they do not provide a clear indication of when the sediment has returned to a baseline state. Stable isotope measurements would help identify sources of organic matter, but the close similarity of $\delta^{13}\text{C}$ values for the feed and faeces with typical "marine" values requires careful interpretation of the results. Lipid biomarkers have the potential to provide much more detail about the origins of the organic matter in a sediment, particularly if combined with isotope data for individual compounds. However, even this approach has its limitations since compounds specific to the feed and faeces have yet to be identified. Since these chemical measurements alone do not provide an adequate assessment of the return to baseline conditions, there is value in combining them with monitoring of changes in the benthic fauna.

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9 CONTAMINANTS

9.1 Introduction

Contaminants are a threat to the natural ecosystem, as well as the users, of an estuary. We are referring to the trace inorganic or organic chemicals that are either present at levels far in excess of natural ultra-trace concentrations, or that are not naturally found in the estuarine environment (xenobiotics). Many of them are toxic or have insidious effects on living organisms.

We did not originally propose to survey contaminants in the Huon Estuary, but realised early that the lack of knowledge of their levels and distribution needed to be rectified. The logistics of our survey program enabled us to develop a valid sampling procedure, and the diversity of our environmental investigations favoured our interpretation of the chemical analyses. Much of this work was done in collaboration with other agencies, or with postgraduate students working on associated projects. Other aspects of water quality of the Huon Estuary, separate from chemical contaminants, have been presented in Chapter 4.

Our approach to planning a specialised survey was to identify contaminants of concern from previous reports of high concentrations in the estuary — its waters, sediments or biota. We also noted prolonged or heavy use of chemical substances in the Huon catchment. Our limited survey of waters and sediments was to look for any indications of the expected contaminants, but also to use analytical techniques that could provide information on other unexpected chemical contaminants.

9.1.1 Contaminants guidelines for estuaries

Important concepts of estuarine chemistry and environmental quality guidelines have been covered in section 4.1. They are also fundamental to the substance of this chapter.

The current status of Australian water-quality guidelines has been presented in Subsection 4.1.2. Here, we will focus on the information relevant to chemical contaminants in both water and sediments. The latter are being developed for the first time in the draft ANZECC / ARMCANZ (1998) guidelines.

Table 9.1 lists the existing, but obsolescent, ANZECC (1992) guidelines for metals and metalloids under the ‘protection of aquatic ecosystems’ criterion and contrasts these with the draft trigger levels for aquatic systems (ANZECC / ARMCANZ 1998). A similar compilation for selected organic pesticides is made in Table 9.2.

Table 9.1 Guidelines for concentrations of metals and metalloids under the current protection of aquatic ecosystems criterion (ANZECC 1992), and the draft guideline trigger levels for aquatic ecosystems (Table 3.4.1, ANZECC / ARMCANZ 1998)

TRACE METAL / METALLOID	ANZECC 1992 guidelines ($\mu\text{g L}^{-1}$)		ANZECC/ARMCANZ 1998 guideline trigger levels ($\mu\text{g L}^{-1}$)	
	Freshwater	Marine	Freshwater	Seawater
Aluminium	< 5 or < 100 ^a		1.2	1.2
Arsenic	50.0	50.0		
As(III)			1.6	1.6
As(V)			2.4	2.4
Cadmium	0.2–5.0 ^b	2	0.013 ^b	1.7
Chromium	10.0	50.0		
Cr(III)			9.0 ^b	9.0
Cr(VI)			1.1	3.1
Cobalt	—	—	0.24	0.9
Copper	2.0–5.0 ^b	5.0	0.33 ^b	0.32
Lead	1–5 ^b	5.0	1.2 ^b	0.80
Manganese	—	—	47	47
Mercury	0.1	0.1	0.013	0.030
Nickel	15.0–150.0 ^b	15.0	0.68 ^b	32.6
Selenium	5.0	70.0		0.30 ^c
Se(IV)			1.4	1.6
Se(VI)			2.3	2.4
Uranium	—	—	3.5	ID ^d
Vanadium	—	—	6.0	10.0
Zinc	5.0–50.0 ^b	50.0	2.4 ^b	2.7

a < 5 $\mu\text{g L}^{-1}$ for pH \leq 6.5; < 100.0 $\mu\text{g L}^{-1}$ for pH > 6.5

b Hardness-dependent

c Value is for total selenium

d Insufficient data

Table 9.2 Guidelines for concentrations of selected organic pesticides in water under the current Protection of Aquatic Ecosystems criterion (ANZECC 1992), and the draft guideline trigger levels for Aquatic Ecosystems (Table 3.4.1, ANZECC / ARMCANZ 1998)

PESTICIDE	ANZECC 1992 guidelines maximum concentrations ($\mu\text{g L}^{-1}$)		ANZECC/ARMCANZ '98 guideline trigger levels ($\mu\text{g L}^{-1}$)	
	Freshwater	Salt water	Freshwater	Marine
ORGANOCHLORINE				
Aldrin	0.010	0.010	0.001	0.003
DDE	0.014	0.014	—	—
DDT	0.001	0.001	0.0005	0.003
Dieldrin	0.002	0.002	0.006	0.006
Endosulfan	0.010	0.010	0.001	0.0003
Endosulfan alpha	—	—	—	—
Endosulfan beta	—	—	—	—
Endrin	0.003	0.003	0.001	0.0005
Heptachlor	0.010	0.010	0.005	0.0003
Lindane	0.003	0.003	0.02	0.007
ORGANOPHOSPHORUS				
Chlorpyrifos	0.001	0.001	0.001	0.0007
Diazinon	—	—	0.0001	0.0001
Dimethoate	—	—	0.02	0.02
Parathion	0.004	0.004	0.0004	0.0004
HERBICIDES & FUNGICIDES				
<i>Phenoxyacetic acid herbicides</i>				
2,4-D	—	—	14	15
<i>Triazine herbicides</i>				
Atrazine	—	—	0.5	0.5
Hexazinone	—	—	—	—
Simazine	—	—	0.9	0.9
<i>Miscellaneous herbicides</i>				
Glyphosate	—	—	30	30

Recommended sediment-quality guidelines are presented in Table 9.3. Two interim sediment-quality guideline (ISQG) levels are indicated — low and high. These correspond to the “effects range-low” and “-median” used by NOAA (Long et al. 1995).

Table 9.3 Recommended sediment-quality guidelines for concentrations of selected metals, metalloids and organic pesticides for aquatic ecosystems (Table 3.5.2, ANZECC / ARMCANZ 1998)

CONTAMINANT	ISQG-Low ^a	ISQG-HIGH ^a
METALS / METALLOIDS (mg kg ⁻¹ dry wt.)		
Antimony	2	25
Arsenic	20	70
Cadmium	1.5	10
Chromium	80	370
Copper	65	270
Lead	50	220
Mercury	0.15	1
Nickel ^b	21	52
Silver	1.0	3.7
Zinc	200	410
ORGANICS (µg kg ⁻¹ dry wt.) ^c		
Total DDT	1.6	46
p,p'-DDE	2.2	27
o,p'- + p,p'-DDD	2	20
Chlordane	0.5	6
Dieldrin	0.02	8
Endrin	0.02	8
Lindane	0.32	1.0
Total PCBs	23	—

a ISQG — interim sediment quality guideline

b Influenced by water hardness and salinity

c Normalised to 1% organic carbon

9.1.2 Background on contaminants in the Huon Catchment and Estuary

Past measurements of contaminants in the Huon Estuary have been very sparse indeed. Gallagher (1996) has summarised the available information. State agencies have done spot measurements in the catchment streams. In the estuary, localised data have come from point source surveys, such as that of Hospital Bay, when Port Huon pulp mill was operating there. Other historical contaminants' sources have been horticulture, forestry, long-established townships, tip sites and a waste transfer facility (see Fig. 2.1) A few locations in the estuary have also been surveyed as part of broader regional or state-wide studies.

Contaminants are split into two groups for further discussion — metals and organics.

Metals

Occasional trace-metal determinations have been made as part of water-quality surveys in the Huon River and Estuary, or as part of examinations of degraded sites, such as Hospital Bay and the head of Port Cygnet. However, little or no, information is available on sample collection or the means of analysis, which makes it difficult to evaluate the integrity of the data.

Gallagher (1996 – Appendix 4 therein) reported some data on iron and manganese along the Huon River from its headwaters to the estuary (nine sites). Iron increased downstream from 245 to 410 $\mu\text{g L}^{-1}$ (Scotts Peak to upstream of Mountain River confluence at Pitts Pump), but decreased sharply to 10 $\mu\text{g L}^{-1}$ in presumably estuarine water at Huonville. Manganese was steady around 10 $\mu\text{g L}^{-1}$ in the river, and in the one estuarine sample it was 30 $\mu\text{g L}^{-1}$. Analyses of the Huon River at Judbury by Rivers and Water Supply Commission Tasmania (1983) produced results for iron (220–1500 $\mu\text{g L}^{-1}$; median: 340 $\mu\text{g L}^{-1}$), manganese (3–13 $\mu\text{g L}^{-1}$; median: 6 $\mu\text{g L}^{-1}$), and mercury (< 0.1–0.6 $\mu\text{g L}^{-1}$; median: < 0.1 $\mu\text{g L}^{-1}$). Measurements of the water column from six sites at “surface, middle and bottom” depths in the “Mid Huon” Estuary (Hospital Bay) on 4 March 1987 (Department of Environment and Land Management, *unsourced*) gave the following results.

METAL/METALLOID	RANGE ($\mu\text{g L}^{-1}$)	MEDIAN
Cadmium	0.4–22	1.5
Copper	< 0.5–23	2.9
Manganese	< 10–76	< 10
Mercury	< 0.4	—
Nickel	< 0.1–5	2.5
Zinc	< 0.5–29	3.8
Arsenic	< 0.5–3	< 0.5
Selenium	< 1	—

Chromium and lead have also been determined in waters of the Huon region, but the measurements are too few to comment on.

Cooper et al. (1982) determined metals in mussels and oysters at two locations in the lower Huon Estuary (Police Point and Garden Island Bay) as reference sites for a study of shellfish in the contaminated Derwent Estuary. Zinc (20–30 $\mu\text{g g}^{-1}$, wet weight), cadmium (< 0.2 $\mu\text{g g}^{-1}$) and lead (< 0.2 $\mu\text{g g}^{-1}$) were considered to be at baseline values, typical of uncontaminated sites.

The only data set we found on metal concentrations in sediments was associated with Hospital Bay and the decommissioning of the pulp mill (Chesterman 1995). The samples were collected about the APM wharf (the main discharge point for mill effluents), although results were also reported for a “background” site at Castle Forbes Bay, a small bay 3 km upstream of Port Huon on the western shore. Analytical determinations were made by inductively coupled plasma – mass spectrometry for 70 elements, using a semi-quantitative mode (the analytical laboratory

suggested an accuracy of about $\pm 50\%$). Results are presented in Table 9.4. Higher concentrations were found in the surface sample at site 13 alongside the wharf.

Table 9.4 Concentrations of trace elements from Huon Estuary sediments in a survey of Hospital Bay for AMCOR Paper Group in July 1994 (Chesterman 1995).

COMPONENT mg kg ⁻¹ dmb ^a	SITE 8 ^b	SITE 12T ^b	SITE 12M ^b	SITE 12B ^b	SITE 13 ^b (surface)
Titanium	137	132	160	148	274
Vanadium	59	72	60	59	112
Chromium	34	40	34	33	53
Manganese	103	116	60	43	98
Cadmium	0.2	0.4	0.1	< 0.1	0.3
Iron	23000	24000	18000	15000	41000
Cobalt	9	11	10	9	20
Nickel	24	27	21	17	36
Copper	17	80	20	9	252
Zinc	51	116	57	33	1002
Arsenic	20	18	18	14	22
Molybdenum	4	16	8	1	7
Lead	10	19	11	3	88
Thorium	2	2	2	2	3
Uranium	0	7	3	< 0.1	3

a dmb – dry matter basis; samples were oven dried and digested in nitric / hydrochloric acid mixture

b Sites: 8 – Castle Forbes Bay, 12 – 10 m off eastern side of APM wharf (T, M, B are top, middle and bottom of 0.38-m core, respectively), 13 – sediment surface alongside APM wharf, 15 m from shore

Moderately high levels were recorded for some metals in the northern part of D'Entrecasteaux Channel (Bloom 1975). For example, metal concentrations in a sediment sample from Oyster Cove were 36 mg Pb kg⁻¹, 8.3 mg Cu kg⁻¹ and 172 mg Zn kg⁻¹.

Metals such as copper, lead and zinc, and also arsenic, attracted some additional attention, because they were used in pesticide formulations in the orchards of the catchment for much of this century. The grubbing out of a significant portion of the Huon's orchards in the 1970s has meant that these contaminants could have been mobilised by replacement crops or alternative land management. Fortunately, investigation of the region's soils, and their accumulation of copper, lead and arsenic, reveals that copper and lead are strongly retained (Merry et al. 1983). Arsenic might be lost by leaching from acid soils, or possibly volatilised as arsine after microbial transformation.

Organics

Concerns about synthetic organic chemicals in the Huon region have centred on pesticides. For over a century, the horticulture industry — mostly located along the banks of the Huon River, its tributaries and the estuary — has probably been using pesticides (or more correctly insecticides, miticides, fungicides and herbicides; Wotherspoon et al. (1994) provides detail on annual usage rate). In the last fifty years, organic chemicals have commonly displaced the traditional mineral-based treatments (e.g. lead arsenate, Bordeaux mixture and copper oxychloride). More recently, other industries (e.g. forestry) have used organic chemicals to control competing or pest species.

Organochlorine pesticides — DDT and its breakdown product DDE, and DDD / γ BHC — were detected in a muddy sediment and biota (zooplankton, turrnellid gastropods, southern cardinal fish and a marine conger eel) of the Huon Estuary in 1969 (Steen 1969). This was six to seven years after the last wide-scale application of DDT and DDD in the Huon Valley. Steen (1969) used thin-layer chromatography as a semi-quantitative technique to detect the pesticides at ppm concentrations ($\equiv \text{mg kg}^{-1}$), and was unable to separate the two chlorinated compounds DDD and γ BHC (Lindane). Listed below are his results for pesticide concentrations in a mud sample and estuarine fauna taken off Shipwrights Point (just outside Hospital Bay). The mud sample expressed as ppm of organic matter (dry weight) could be an underestimate, because organic matter was determined by loss on ignition. The portunid crabs had the highest levels of any animal analysed in this work (DDT and DDE could not be resolved for these samples, and are reported as DDT).

SAMPLE	DDT	DDE	DDD / γ BHC
Muddy sediment	0.006–0.022	—	0.008–0.042
Eggs – southern cardinal fish	0.46	1.4–13.4	1.56
Portunid crabs	19.8–24.5	—	0.87–1.1

Steen (1969) also found that the organochlorine pesticide residues were routinely higher in Mountain River roach, trout and eels, relative to the same species in the Russell River (e.g. in eels, DDE up to 11 ppm vs. 3.65 ppm, and DDT 10.1 ppm vs. 1.3 ppm). The Mountain River had a large expanse of orchards along its course, while the Russell River valley has never had orchards.

Organochlorine pesticides have been phased out, or in some cases banned, for use in agriculture and other land-use practices. They have been superseded by chemical compounds that are more specific in their action and less persistent in the environment (Smith et al. 1988). An example is the herbicide atrazine, although it is not itself uncontroversial (Davies et al. 1994).

Wotherspoon et al. (1994) describe how it has been used in the Southern Forests of Tasmania. The last application in the Huon Valley was on newly established plantations in the catchments of the Arve and Russell Rivers in 1993. Wotherspoon et al. (1994) cited Forestry Commission sources that “creeks draining areas of plantation in 1992 and 1993 registered levels generally between 1 ppb (part per billion [$\equiv \mu\text{g L}^{-1}$]) and 4 ppb immediately after spraying, however, two readings of 40 and 70 ppb were detected”. Gallagher (1996) obtained other Forestry Commission data from 1992/93 in the Southern Forests (see Fig. 2.1 for locations of catchments) – Arve River (range: 2–380 $\mu\text{g L}^{-1}$; median: 3 $\mu\text{g L}^{-1}$), Wobbly Creek – tributary of the Esperance River (range: < 1–20 $\mu\text{g L}^{-1}$; median: 8 $\mu\text{g L}^{-1}$), and Hot Springs Creek – tributary

of the Lune River (range: 3–1150 $\mu\text{g L}^{-1}$; median: 30 $\mu\text{g L}^{-1}$). The results are broadly consistent, apart from some very high concentrations after spraying. The issue of persistence of atrazine in Tasmanian waterways and its effects upon aquatic organisms has been reported by Davies et al. (1994). They observed low levels of atrazine in forestry plantation streams after more than a year (e.g. median concentrations of 2.0 $\mu\text{g L}^{-1}$ after rainfall at 15–16 months). Physiological effects in salmonids were noted at atrazine concentrations above 10 $\mu\text{g L}^{-1}$.

Although agricultural groups and integrated catchment strategies aim to decrease usage of pesticides in the Huon Valley (Sustainable Development Advisory Council 1996), the distribution of most currently used pesticides has not been monitored. In the winter of 1995, a survey of drinking water off-takes in the Huon region was made under the auspices of the Tasmanian Government's Agricultural and Silvicultural Chemicals (ASCHEM) Task Force. Samples were collected from Mountain River, Judbury, Geeveston and Cygnet. The suite of pesticides determined were the same as those listed in Table 9.6. None was detected in any of the Huon samples (M. G. Johnstone, DPIWE — *pers. comm.*, November 1999).

9.2 Survey of Contaminants in the Huon Estuary (September 1998)

9.2.1 Sampling and analytical methods

The designated contaminants' survey was planned to follow a regular whole-of-estuary (spatial) survey. On the assumption that the status of the estuary would be established in the first week. In the second week, the contaminants' survey would both build on the first survey by sampling for contaminants, and also repeat some of the standard measurements to assess the week-to-week variability.

In Chapter 2, we stressed the importance of covering the temporal and spatial variability in estuaries. From a single survey this is not possible. However, we improved our scope for contaminants by collecting both water and sediment samples over the full extent of the estuary, as well as two freshwater inputs (Huon and Kermadie Rivers).

Waters

The basis for the selection of sites for water samples was first to use the existing spatial survey sites as a framework (Fig. 2.6). A subset of sites was then selected with reference to known former sites of pollution and other estuarine features, while at the same time giving a spread of sampling sites across the full estuarine gradient. The constraint was that only about 20 water samples would be collected.

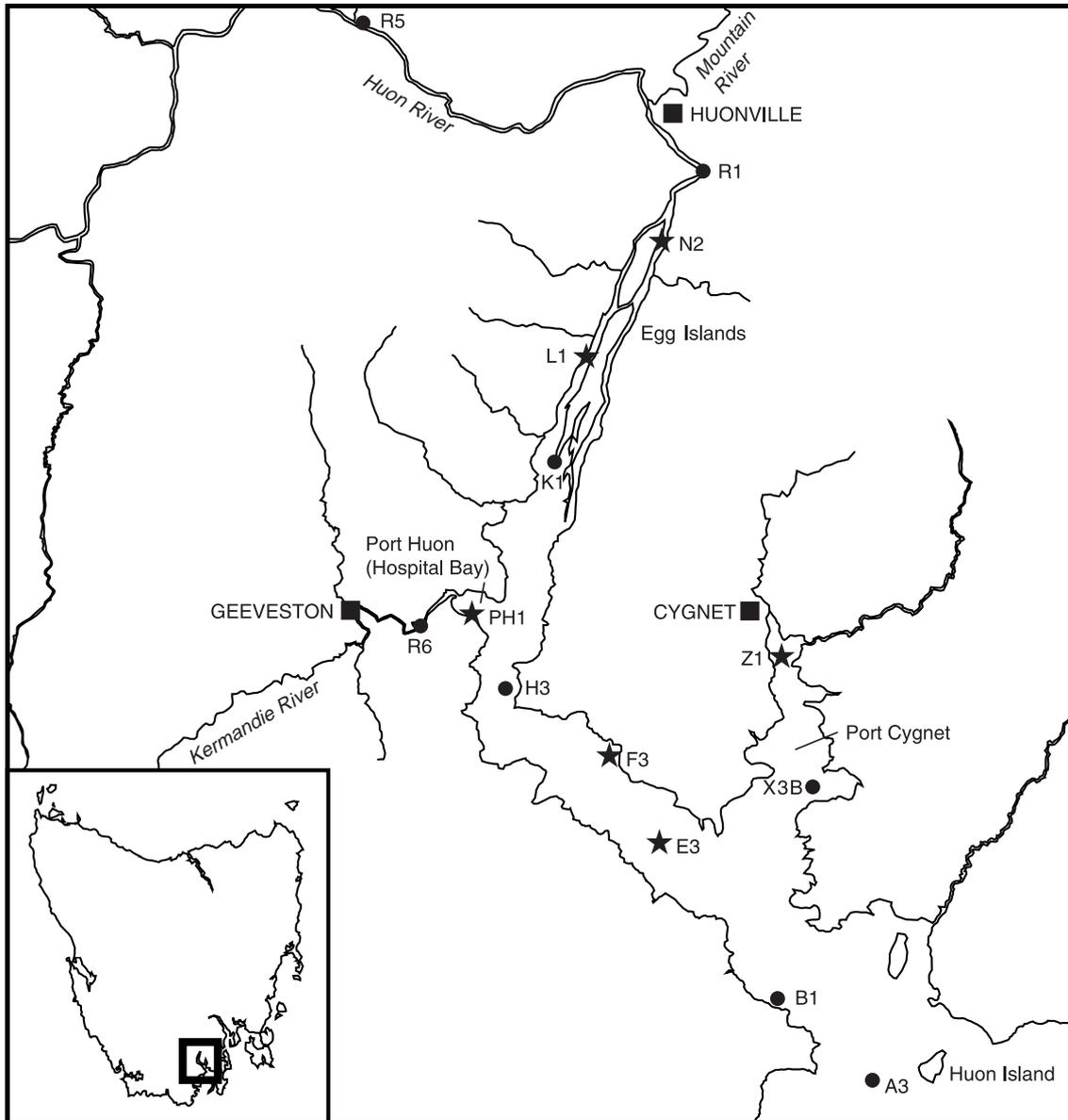


Fig. 9.1 The sampling sites for the survey HES 10 A (Aug / Sep '98) — (★) both water column and sediment, (●) water column only.

The following list gives the sites (river – sea) in the contaminants’ survey. The letters in parenthesis ‘s’ or ‘b’ indicate surface or bottom sample taken, respectively. The comments in square brackets give common site name and other information used in selecting a particular site. **S = number** indicates a site at which we obtained a water sample with salinity *number*. This was a factor in choosing some sites — so we might span the estuarine mixing gradient without large gaps. The only new site was PH1, because the seasonal spatial surveys did not include Hospital Bay (Fig. 9.1). Water samples for pesticides determination were taken at sites R5, N2, L1, R6, PH1, F3 and E3 only.

- R5 (s) [Judbury – riverine end-member (**S = 0**)]
- R1 (s,b) [below Huonville and Ranelagh STP outfall]
- N2 (b) [top of Egg Is., anaerobic zone]
- L1 (s) [below Franklin and Prices Creek (**S = 5–10**)]
- K1 (s) [bottom of Egg Is. (**S = 15–20**)]
- R6 (s) [Kermandie R., below Geeveston STP outfall]
- PH1 (b) [in Hospital Bay, off and inshore of Whale Point wharf]
- H3 (s,b) [Brabazon Park (**S = 10–15**)]
- F3 (s,b) [Wheatleys Bay]
- E3 (s,b) [mid-stream above Port Cygnet arm (**S = 30–34**)]
- Z1 (s) [immediately below Cygnet, and Cygnet STP outfall]
- X3B (s,b) [Deep Bay, off mussel farm (**S = 25–30**)]
- B1 (s,b) [Hideaway Bay]
- A3 (s,b) [Huon Pt – Huon Is. transect, marine end-member (**S = 34–35**)]

After measuring a salinity of 24 with a portable salinometer at site F3 during the survey, we selected it to provide a salinity between 20 and 25. Precision salinity measurement back in the laboratory on a surface sample collected by Niskin bottle gave a value of 19.2. The patchiness of surface waters in the middle estuary, seen in underway measurements of this region (Fig. 3.11), was again illustrated by this observation.

Sampling procedures and analytical instruments, equipment and methods are fully detailed in the supplements to this report. ‘Clean’ (non-contaminating) techniques were used for collecting water samples, and specially selected and prepared containers (appropriate to analytes) were used for all subsamples (e.g. O’Sullivan et al. 1996). Metals were determined directly or after preconcentration (solvent extraction of metal dithiocarbamates and acid back-extraction) by graphite-furnace atomic absorption spectroscopy (GFAAS) at CSIRO Marine Laboratories (Mackey et al. 1997; analyst: Jeanette O’Sullivan), and also by inductively coupled plasma – mass spectrometry (ICP-MS) at the Central Sciences Laboratory, University of Tasmania (analyst: Ashley Townsend). Arsenic in waters was determined by hydride generation – atomic fluorescence spectroscopy (HGAFS) at the School of Chemistry, University of Tasmania (analyst: Alison Featherstone). Mercury in waters was determined by cold-vapour atomic absorption spectroscopy (CVAAS — Plaschke et al 1997; analyst: Kate Berry).

The GFAAS methods for trace metals (Cd, Co, Cu, Fe, Mn, Ni and Zn) and the HGAFS method for arsenic have been thoroughly validated for analyses of marine and estuarine water samples, by routine measurement of certified reference materials (CRMs) and comparisons with measurements made by other independent techniques (Mackey et al. 1997, Featherstone et al. 1998). Results for the above suite of trace metals and arsenic were reported from these methods

as a first choice. The ICP-MS determinations were made either directly on samples diluted with high-purity deionised water, or on preconcentrated samples by way of solvent extraction of metal chelates and back extraction into dilute nitric acid. CRMs were analysed in parallel by these methods. In many instances, ICP-MS results were very similar for the 'standard suite' of trace metals (Cd, Co, Cu, Fe, Mn, and Zn) to those obtained by GFAAS (typically within $\pm 20\%$ to $\pm 45\%$ for Cd, Co, Cu and Fe). For metals not measured by GFAAS (i.e. Pb, U, V and on this occasion also Ni), we present the results as indicative concentrations. Uranium and vanadium were measured directly after dilution of the water sample. Lead and nickel were measured in dilute acid back-extracts (analytes preconcentrated by solvent extraction of metal dithiocarbamates). Their extraction as dithiocarbamate chelates is optimal at the pH of 4.5 used for the standard suite of trace metals (Kinrade and Van Loon 1974, Sekine and Hasegawa 1977).

The suite of organic pesticides was determined by the Division of Environment and Planning Laboratory (DPIWE), using standard methods, as part of a collaborative investigation. Semivolatile organics in water (organochlorine, organophosphorus and other pesticides) were determined by test methods 1501–water (gas chromatography – mass spectrometry, GC-MS) and 1502–water (solid-phase extraction followed by GC-MS). Herbicides in water were determined by test method 1504–water (high-performance liquid chromatography with diode-array detection, HPLC–DAD), and glyphosate by 1508–water (equivalent to USEPA method 547).

Sediments

Sediment samples were taken at six sites in the estuary where water samples had also been taken for contaminants' determination (Fig. 9.1). Hence, any obvious signs of sediment–water interaction should be registered. Chosen sites were in the upper estuary L1 and N2 (anaerobic), middle estuary F3 (at fish farm) and lower estuary E3. Sites previously reported as contaminated — see above and Section 4.2 — were also sampled in Port Huon (PH1) and Port Cygnet (Z1). Sediments were collected with a box corer (225 × 225 mm). From each sediment grab, cores were subsampled from the box by 60-mL plastic syringes with the ends cut off. Ten syringes were inserted into the box full of sediment and then withdrawn randomly. The top 2.5 cm of sediment, from two to three syringes, was extruded into acid-washed polyethylene (metals) or glass (pesticides) jars, and stored on ice in the dark.

Bulk constituents of sediments were determined on dried sediments, after refluxing a subsample for 4 h in concentrated nitric acid, by Inductively Coupled Plasma – Optical Emission Spectroscopy (ICP-OES). Trace elements were determined with ICP-MS after the same treatment. Both sets of measurements were made at the Central Science Laboratory, University of Tasmania (analyst: Ashley Townsend). Sediments were also analysed for pesticides at the Division of Environment and Planning Laboratory (DPIWE): semivolatile organics by test method 2501–soil (GC-MS), herbicides by 2502–soil (GC-MS), and phenoxy acid herbicides by 2503–soil (GC-MS).

9.2.2 Trace metals in Huon Estuary water and sediments

The results from the HES 10A survey (31 August–2 September 1998) of Huon waters for a suite of trace metals and the metalloid arsenic are presented in Table 9.5.

Table 9.5 Concentrations of metals and arsenic in waters of the Huon and Kermantie Rivers and the Huon Estuary (UF – unfiltered; F – filtered (0.45 µm)) from HES 10A (September 1998); values in brackets have not been independently corroborated.

TRACE METALS / METALLOID (µg kg ⁻¹) ^a	Huon River (R5)	Kermantie River (R6)	Estuary (12 sites)
Cadmium (UF)	0.0018	0.0066	0.0018–0.0057
Cadmium (F)	0.0009	0.0020	0.0010–0.0049
Cobalt (UF)	0.076	0.327	< 0.013–0.142
Cobalt (F)	0.027	0.126	< 0.013–0.116
Copper (UF)	0.226	1.640	0.109–0.241
Copper (F)	0.172	0.836	0.085–0.188
Iron (UF) (µg L ⁻¹)	300	1108	9.0–262
Iron (F) (µg L ⁻¹)	225	586	0.9–196
Lead (UF) ^b	[0.17]	[0.84]	[0.019–0.082]
Manganese (UF) (µg L ⁻¹)	3.0	21.8	0.3–16.1
Manganese (F) (µg L ⁻¹)	2.0	14.8	0.3–15.2
Mercury (UF) ^c (µg L ⁻¹)	< 0.002	< 0.002	< 0.002
Nickel (UF) ^b	[0.42]	[1.13]	[0.22–0.33]
Uranium (UF) ^d	[0.030]	[0.028]	[0.79–3.47]
Zinc (UF) (µg L ⁻¹)	0.25	2.85	0.30–2.20
Zinc (F) (µg L ⁻¹)	0.17	1.47	0.21–2.13
Vanadium (UF) ^d	[0.47]	[2.54]	[1.06–1.80]
Arsenic (F) (µg L ⁻¹) ^c	0.038	0.072	0.087–1.71

a Trace metals determined directly by GFAAS (Fe, Mn & Zn), or following solvent extraction of dithiocarbamate chelates (Cd, Co & Cu), and units are µg kg⁻¹, except where indicated otherwise.

b Indicative concentrations for both Lead and Nickel were determined by ICP–MS analysis of chelate extracts, at E3, F3, L1, PH1, N2, R5, R6 and Z1 only

c Acid-labile mercury by CVAAS; Inorganic arsenic (As^V + As^{III}) by HGAFS

d Indicative concentrations for both Uranium and Vanadium were determined by ICP–MS directly, at E3, F3, L1, PH1, N2, R5, R6 and Z1 only.

All of the metals and arsenic were below the latest ANZECC / ARMCANZ (1998) trigger levels in the Huon River and estuary. The metal concentrations in the Huon Estuary were comparable to concentrations of copper and cadmium in the waters of Bathurst Harbour and Port Davey (Mackey et al. 1996). In some instances, they were as low as iron, manganese, nickel and cobalt in this pristine waterway, but were mostly higher for these four metals. It would seem that the Huon Estuary has a higher concentration of particulate iron, and possibly colloidal iron and manganese, with which nickel and cobalt are associated.

The Huon River had marginally higher concentrations of copper, nickel and zinc, a similar level of cadmium, and lower concentrations of iron and manganese than the Old River (main tributary to Bathurst Harbour) (Mackey et al. 1996). It is perhaps not surprising, because these rivers are draining back-to-back catchments. The Huon River has some logging in a corridor in its central catchment; the catchment of the Old River is in the South West National Park, and so has no forestry activity.

In contrast, the Kermadie River had cobalt, copper, nickel and zinc above their respective trigger concentrations. This river carries a substantial load of suspended solids (see Subsections 4.3.4 and 4.3.9). It is likely that much of its trace metal load is associated with this material. Indeed, cobalt and zinc exceeded the trigger level when unfiltered concentrations were considered, but fell below it when instead filtered concentrations were used.

Copper was still two to three times the trigger level in the filtered Kermadie River sample. The reason or reasons for the raised concentrations for copper were not obvious. In the recent State of the Environment Tasmania report (Sustainable Development Advisory Council 1996), it was reported that copper and iron were “naturally at higher concentrations in some of the streams entering the estuary”. Use of copper in the horticulture industry and the effluent from the Geeveston sewage treatment plant are other possible sources.

We have no data for nickel in a filtered water sample from site R6. However, the nickel concentration in the unfiltered sample was higher than any others observed in Tasmanian estuaries (Macquarie Harbour, Bathurst Harbour and Port Davey — Mackey et al. 1996, and the Derwent Estuary — CSIRO, *unpublished data*), apart from a single measurement in the Upper Derwent Estuary ($1.63 \mu\text{g kg}^{-1}$). This anomalous measurement in the Derwent sample was also associated with a high concentration of suspended particulate matter. We reiterate that the nickel determinations made for HES 10A samples are indicative. They were made with a different analytical technique than the earlier studies; the results remain to be corroborated.

It is likely that the higher concentrations of trace metals associated with the Kermadie River plume decline sharply as it enters the estuary. Much of the trace metals load associated with suspended particulate matter is probably deposited onto the sediments of Hospital Bay. If copper (and perhaps nickel) were still thought to be an issue, the hierarchical sequence that is the framework of the ANZECC / ARMICANZ (1998) guidelines recommends that direct toxicity tests should follow. These would be either or both *in situ* field or laboratory ecotoxicological tests on Kermadie River water. The objective would be to establish concentrations that are unlikely to cause biological harm.

The companion results for trace metals in sediments from the HES 10A survey, albeit at fewer sites, are given in Table 9.6. All the metals reported on had measured concentrations in excess of 60% of the certified value for two sediments supplied as certified reference materials (NIST SRM 2704 Buffalo River sediment and National Research Council of Canada (NRCC) BCSS-1 a near-shore marine sediment from the Gulf of St Lawrence). They were not expected to attain the CRM values, because the certified value approximates a total measurement. We were following the recommendations of the ANZECC / ARMICANZ (1998) guidelines by determining metals after acid digestion. This procedure will leave a refractory sediment residue that will contain a portion of most analytes.

Aside from nickel, all metals reported in Table 9.6 were below ISQG-Low concentrations (ANZECC / ARMCANZ 1998). In such circumstances, the expectation is that there is a low risk of any biological effects, because no report has been made of biological effects below this concentration. No further management action is recommended.

Table 9.6 Labile metal concentrations in sediments (acid-leaching treatment – conc. HNO₃, 100 °C for 4 h) from six sites in the Huon Estuary, HES 10A (September 1998)

TRACE METALS (mg kg ⁻¹ dry wt.) ^a	N2	L1	PH1	F3	E3	Z1
Cadmium	0.02	0.08	0.26	[0.1] ^c	0.07	0.08
Cobalt	3.7	4.7	12.5	8.7	7.8	3.9
Copper	3.6	4.2	47.0	22.6	19.7	12.9
Iron ^b	6200	7500	36500	43400	41900	13600
Lead	3.6	4.7	27.7	29.2	25.1	15.7
Manganese	35	42	87	78	75	27
Nickel	6.2	5.6	19.1	21.4	21.9	7.4
Zinc	28	14	92	61	62	45

a Dried at 105 C

b Iron determined by ICP-OES

c Mean value uncertain because of considerable difference between two isotopes ¹¹¹Cd and ¹¹²Cd

Table 9.7 Labile metal concentrations in sediments from Table 9.6 normalised to iron (HES 10A, September 1998)

TRACE METALS (mg kg ⁻¹ dry wt.) ^a	N2	L1	PH1	F3	E3	Z1
Cadmium	0.15	0.45	0.31	~0.1	0.08	0.25
Cobalt	25.9	27.2	14.9	8.7	8.1	12.4
Copper	25.2	24.3	55.9	22.6	20.4	41.2
Iron ^b	43400	43400	43400	43400	43400	43400
Lead	25.2	27.2	32.9	29.2	26.0	50.1
Manganese	245	243	103	78	78	86
Nickel	43.4	32.4	22.7	21.4	22.7	23.6
Zinc	196	81	109	61	64	144

a Normalised relative to iron, with the reference site being F3

Nickel concentrations in Port Huon and lower estuary (sites E3 and F3) sediments were around the threshold concentration (21 mg kg⁻¹ dry wt.) for the ISQG-Low criterion. This seems unlikely to be an issue of contamination, but perhaps naturally higher concentrations of nickel associated with very fine-grained sediments (muds — see Chapter 7). Although the Kermantie

River outflow might have elevated nickel concentrations, as mentioned above, its annual riverine discharge to the estuary is relatively very minor (2% — see Chapters 4 and 10), and consequently its nickel input is negligible. When sediments are normalised to the iron concentration observed in the lower estuary (reference site: F3 — Table 9.7), nickel concentrations are uniform throughout the lower estuary and Ports Huon and Cygnet at about 23 mg kg^{-1} .

Normalising trace metal concentrations against iron (or possibly aluminium, scandium or lithium) is a recommended procedure for interpreting contaminants data in sediments (Loring and Rantala 1992). When trace metals, in addition to nickel, were treated in this manner, it was found (Table 9.7) that copper, cadmium and lead, and possibly zinc, were slightly raised (by a factor of 2–3) in sediments of Hospital Bay (Port Huon) and at the upper end of Port Cygnet, relative to other marine sediments in the estuary. Cadmium was also higher downstream of Franklin (site L1). These observations for the two ports are not unexpected given their past history, and the continued influence of effluents from sewage treatment plants.

Some metals (Mn, Zn and Co) in Table 9.7 had quite different concentrations in the fluvial zone of the estuary than in the marine zone. This might be expected on account of the different depositional characteristics of the two zones, and the different sediment chemistry. The upper estuary sites (L1 and N2) have reducing sediments, which were highlighted by the significant input of dissolved manganese into bottom waters at N2 and the similar site R1 (presumably as Mn^{2+} after reduction of Mn(IV) oxyhydroxides — Fig. 9.2). Cobalt and cadmium (results not shown) also had smaller inputs at these sites.

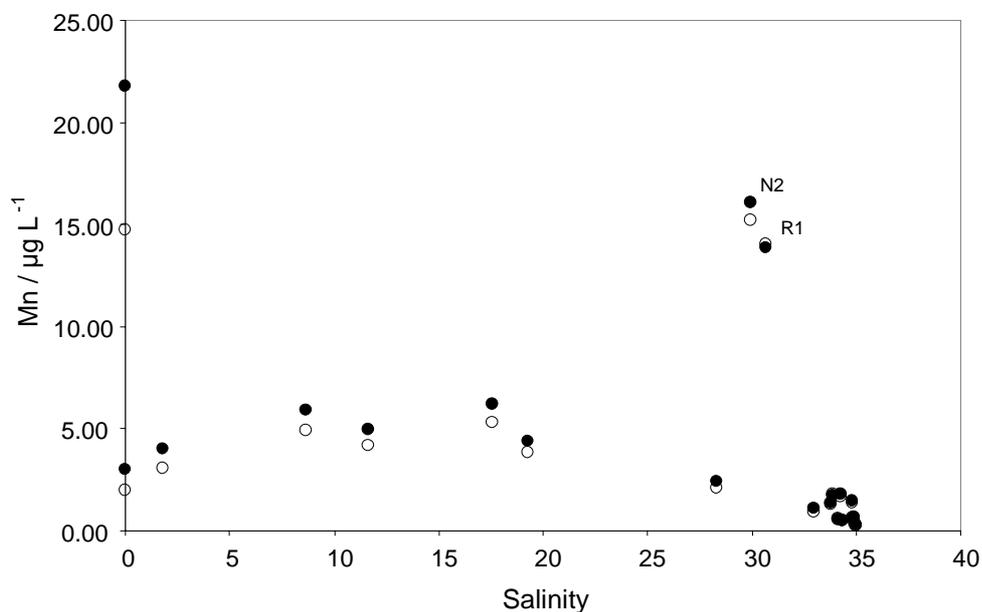


Fig. 9.2 The relation of manganese with salinity for HES 10A (August / September 1998 — (●) unfiltered, (o) unfiltered), showing input of manganese at sites N2 and R1 in bottom waters in the upper estuary. The high values of manganese on the y-axis are from site R6 (see Table 9.5). The precision of the analyses is typically $\pm 8.2\%$ at $12.2 \mu\text{g L}^{-1}$.

Results were too few in this scouting survey of trace metals in sediments to allow a rigorous application of statistics. Nevertheless, as an exploratory exercise we standardised the results in Table 9.6 ((value – mean) / standard deviation) and analysed them by Multidimensional Scaling (not shown). The six sites distributed into three distinct zones of the two-dimensional ‘scatterplot’: PH1 (Hospital Bay) was on its own, E3 and F3 in the lower estuary formed a group, and L1, N2 and Z1 (with the last slightly apart) formed what appeared to be a cluster influenced by terrestrial run-off.

9.2.3 Pesticides in Huon Estuary water and sediments

The suite of pesticides measured in the water and sediment samples collected during HES 10A are shown in Table 9.8. The analytical limit of detection for organochlorine and organophosphorus pesticides in waters by GC-MS was $0.1 \mu\text{g L}^{-1}$; and for the other pesticides, determined by HPLC-DAD (glyphosate apart), it was $0.05 \mu\text{g L}^{-1}$. None of the pesticides could be detected at any of the six estuarine sites.

Table 9.8 Pesticides analysed in waters and sediments collected during survey HES 10A

ORGANOCHLORINE	ORGANOPHOSPHORUS	OTHER PESTICIDE ^a
Aldrin	Chlorpyrifos	Atrazine (triazine)
α -BHC	Diazinon	Hexazinone (triazine)
β -BHC	Dimethoate	Simazine (triazine)
γ -BHC (Lindane)	Disulfoton	2,4-D (chlorophenoxy)
δ -BHC	(Ethyl) parathion	Picloram (pyridine)
Dieldrin	Famphur	Triclopyr (pyridine)
p,p'-DDD	Methyl parathion	Glyphosate (P-glycine) ^c
p,p'-DDE	Phorate	
o,p'-DDT	Sulfotep	
Endosulfan I (α)	Thionazin	
Endosulfan II (β)		
Endrin		
Heptachlor		
Endosulfan sulphate ^b		
Endrin aldehyde ^b		
Heptachlor epoxide ^b		

a Class of herbicide shown in parenthesis

b Breakdown products or metabolites

c Phosphanoglycine — not measured in sediments

For sediment samples, the limit of detection for all organochlorine and organophosphorus pesticides was 0.01 mg kg^{-1} . It was 0.05 mg kg^{-1} for the triazine herbicides, and 0.5 mg kg^{-1} for picloram and triclopyr. Once more, none of the pesticides could be measured at or above their

limits of detection at any of the sites, with one important exception. DDT and DDD were present in the Hospital Bay sediment sample at $180 \mu\text{g kg}^{-1}$ and $140 \mu\text{g kg}^{-1}$, respectively.

The detection limits of the analytical methods used in this scouting exercise for pesticides in waters do not reach the levels of sensitivity indicated for some pesticides in the earlier ANZECC (1992) guidelines (Table 9.2). Moreover, they are typically two or three orders of magnitude above the trigger levels suggested in the draft ANZECC / ARMCANZ (1998) guidelines. They are generally inadequate for the sediments as well, with only the ISQG–High level discernible for some pesticides (Table 9.3).

The above comments aside, the pesticides results do indicate that there is not gross contamination from these contaminants in the Huon Estuary. One disturbing aspect, nevertheless, is the high concentrations of DDT and DDD in the sediment sample from Hospital Bay. They are both above the ISQG–High level. The implication is that a high probability exists that there will be toxic effects — presumably localised to proximate benthic organisms. However, a cautionary note in the ANZECC / ARMCANZ (1998) guidelines observes that because the NOAA values (the basis for the ISQG levels) for DDT pesticides were not based on toxicity to benthic organisms, “[they] may tend to over-predict the likelihood of effects”.

9.3 Summary and Conclusions

The contaminants’ survey covered a limited number of sites in the Huon Estuary. On the basis of our characterisation of the estuary in the preceding chapters, we believe that it is, nonetheless, representative of the range of conditions exhibited in the water body. Results from our survey that demand further investigation, and possible omissions from our survey seen in retrospect, will be dealt with here.

Concentrations of trace metals in the Huon Estuary were almost without exception below the trigger levels for an aquatic system that is substantially natural, as specified in the draft ANZECC / ARMCANZ (1998) guidelines. These observations applied equally to water quality and sediment quality. We have no reason to think that these guidelines are inappropriate for the Huon Estuary. Indeed, the water quality of the Huon River was similar to that of the Old River, part of the pristine Bathurst Harbour Estuary in Tasmania’s South West Wilderness. The Huon Estuary, when compared with Bathurst Harbour, had similar concentrations of some metals (Cd and Cu) and slightly raised concentrations of others (Fe, Mn, Co and Ni).

The lower Kermadie River (site R6) was different. It had concentrations of copper, cobalt, nickel and zinc in unfiltered waters above the interim trigger levels. Much of the load of these metals was in the suspended particulate matter; only copper had concentrations in excess of the guidelines in a filtered sample.

Our sediment survey did not reveal any metals exceeding the interim sediment quality guidelines, although nickel was at the ISQG–Low level. Uniform values for nickel, after normalising its concentrations to iron, suggest natural sources. We observed marginal increases of some metals (Cd, Cu, Pb and Zn) in Port Huon (Hospital Bay) and Port Cygnet samples, when normalised to iron. Much higher concentrations of copper, lead and zinc have been observed recently in Hospital Bay near the APM wharf (Table 9.4 — Chesterman 1995).

These results taken together suggest that Hospital Bay might have patchy metal contamination of sediments.

We could find no evidence elsewhere in the Huon Estuary of contamination from past use of inorganic pesticides (e.g. lead arsenate, copper salts). This supports Merry and co-workers' (1983) contention that the inorganic pesticides are sequestered in agricultural soils (or partially volatilised to the atmosphere in the case of arsenic). It would be useful to determine arsenic and mercury in sediments throughout the estuary. We did not have an analytical method for the former, and not one sufficiently sensitive for the new sediment guidelines for the latter. We have no reason to believe that either arsenic or mercury would be at raised levels in estuarine sediments. Chesterman (1995) recorded that the operation of the pulp mill on Hospital Bay did not make use of mercury biocides, as has happened elsewhere with forest industries.

Although we did not measure all of the so-called priority pollutant metals (omissions include Ag, Be, Cr, Sb, Se and Tl), it is difficult to imagine that one of these elements would behave as a maverick. Usually a *group* of metals and metalloids will be observed at higher concentrations as a result of discharge in waste, extraction of minerals, or other human activity.

The organochlorine pesticides DDT and DDD were found to be at high concentrations (above ISQG-High levels) in a Hospital Bay sediment collected during HES 10A. These are especially persistent pesticides (Smith et al. 1988). Our results are broadly consistent with work done by Steen (1969) thirty years earlier, although his results were a little lower than ours were. His sediments were collected in the estuary just outside Hospital Bay, and he acknowledged that his semi-quantitative measurements might also have been an underestimate. It is fruitless to make anything more of the comparison for two single samples separated by so many years. Nevertheless, these results do suggest that a more extensive survey of pesticides in sediments is needed in and around Hospital Bay.

Over the decades, the waste from sawmilling and then pulping at Port Huon has led to the formation of organic-enriched sediments on the floor of Hospital Bay, and to some extent into the main arm of the estuary (see Subsections 7.3.3 and 7.3.4). It is quite conceivable that these sediments could behave as a sponge for both organic and inorganic contaminants.

As discussed above, the survey of pesticides in this study was a scouting exercise. Apart from Hospital Bay, it did not reveal significant contamination by pesticides in the estuarine system. It should be followed up by a more comprehensive survey. Any further investigation will need to include a range of pesticides known to have been used in the Huon catchment in the past, and those chemicals used currently. For example, the pome fruit industry in the Huon Valley now makes limited use of some selective herbicides and fungicides (Amitrol and other triazoles, and benzimidazoles — W. Boucher, Grove Research Station (DPIWE), *pers. comm.*, August 1998). Analytical methods used in any subsequent pesticide survey will also need to be sufficiently sensitive to accommodate the recommended interim sediment quality guidelines (see Tables 9.2 and 9.3).

We did not analyse sediments or waters for other organic contaminants, such as petroleum hydrocarbons, PAHs or PCBs. They would be expected only as localised trace contamination in parts of the estuary like Hospital Bay, immediately around stormwater drains, or possibly in tip leachate entering streams. Limited information on petroleum hydrocarbons might be obtained from re-processing of some of the chromatographic data (gas chromatography or GC-MS) we obtained during the sediment survey.

9.4 References

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10 NUTRIENT BUDGETS AND WATER-QUALITY MODELS

10.1 Introduction

The preceding chapters have described the physical, chemical and biological state of the Huon Estuary during the 1996 - 98 field study. In this chapter, we present a quantitative analysis and interpretation of nutrient cycling in the estuary, based on application of models to these data sets. Aside from its scientific interest, this analysis can provide valuable information to support management of aquaculture in the estuary and nutrient loads from the surrounding catchment.

The study shows the estuary is characterised at times by elevated nutrient concentrations and phytoplankton blooms, including blooms of a toxic dinoflagellate. It is unclear *a priori* to what extent nutrient loads from catchment, aquaculture and marine sources are responsible. As the Huon estuary is a key site for salmon cage culture, it is important for both industry and regulators that the impact of current nutrient loads, and the capacity of the estuary to withstand additional loads, be understood.

Estuarine models vary widely in aims, in structure and complexity, and correspondingly in data needs. We can broadly distinguish inverse models, which try to infer fluxes from observations, and prognostic (process) models, which try to simulate the dynamical processes underlying observations. Physical circulation models predict circulation and transport of tracers, and the evolution of salinity and temperature fields. Biogeochemical models simulate local processes controlling the cycling of nutrients in estuaries, and predict the effects of loads and transport on water and sediment quality and phytoplankton blooms. Models of all kinds are increasingly regarded as essential tools for scientific synthesis and management.

Models can be developed with very different spatial resolution, from 1-box flushing models to sophisticated 3-D circulation models. Models can be very simple: for example, plotting concentrations of tracers versus salinity and looking for deviations from a conservative mixing line implicitly uses a transport model.

The data needs of models vary widely. Inverse models are usually applied to spatial surveys of physical and possibly biogeochemical variables. Application to salinity fields can, given knowledge of freshwater input, allow estimates of effective physical exchanges among model cells, assuming the system is at steady-state. Once the exchanges are estimated, inverse models can be applied to biogeochemical tracers such as nutrients to estimate horizontal and vertical fluxes, and local sources and sinks.

Biogeochemical process models incorporate scientific understanding of the processes determining local sources, sinks and transformations of biogeochemical variables. These models require forcing data, including physical forcing, loads, and marine boundary conditions. They usually include a number of rate parameters, which may be weakly constrained *a priori*.

Biogeochemical process models have several advantages over inverse models. They allow us to test hypotheses about the importance and nature of key processes against observations in a quantitative sense. Because they can “intelligently” interpolate or extrapolate based on process

understanding, they do not necessarily require data sets with the same intensive spatial and temporal resolution required by inverse models. Perhaps most importantly, they allow us to predict the response of the estuary under different management scenarios.

We use both inverse and prognostic process models to analyse nutrient cycles in the Huon Estuary. The inverse modelling (section 10.3) combines the inverse estimates of physical exchanges in the estuary derived in Chapter 3.4 with nutrient distributions observed on spatial surveys, and estimates horizontal fluxes and local sources and sinks of nutrients. This approach represents a more sophisticated form of the biogeochemical modelling approach developed for the international program Land-Ocean Interactions in the Coastal Zone (Gordon et al. 1996), previously applied to the Derwent and other Australian estuaries (Parslow 1999).

There is a long and ongoing debate about the appropriate level of complexity in ecological process models. More complex models are not necessarily more accurate, and it is generally agreed that an intermediate level of complexity yields the best results. The “optimal” level depends not only on the natural system, but also on the level of understanding and data available, and on the modelling objectives.

There is now a substantial history of successful applications of models in estuarine and coastal systems. More complex models, such as the Port Phillip Bay Integrated Model (Murray and Parslow 1997), represent a range of nutrients, a range of trophic functional components, and some diversity within components, such as different algal groups. Simpler models may represent the response of a generic phytoplankton component to nutrient loads.

We have adopted a relatively simple modelling approach for the Huon Estuary. This is partly for pragmatic reasons: the Huon Estuary Study was not designed to include a large modelling component. Because the estuary is flushed relatively rapidly, the fate and impacts of nutrient loads are dominated more by physical transport than by internal processes (in contrast, for example, with Port Phillip Bay – Harris et al. 1996) and so a simpler representation of biogeochemical processes is more likely to prove adequate. The data set acquired during this study is certainly capable of supporting a more detailed and intensive modelling effort. We discuss the potential benefits of developing these models further at the end of this chapter.

The model used here is based on a simple model of nutrient cycling and phytoplankton biomass in estuaries, developed by CSIRO for the National River Health Program (Parslow et al. 1999). This model (SEEM) represents the cycling of nitrogen through dissolved inorganic, phytoplankton and detrital pools. It is based on the physical geometry and transports calculated in Chapter 3, and takes account of the effects of flushing, light and nutrient limitation on phytoplankton growth, grazing, and recycling of nitrogen in the water column and sediment. The model has been shown to be capable of representing the response of a range of estuaries, including the Derwent Estuary, to both diffuse and point source nutrient loads, in terms of phytoplankton blooms and nutrient concentrations. It has the advantage that it involves a relatively small number of parameters, but has only a simplified sediment component, and is not capable of representing epibenthic responses.

We use here the conventions introduced in Chapter 3 to describe the geometry underlying both inverse and prognostic models. The full length of the estuary is divided into 15 columns, numbered 0 to 14 (Fig. 3.1), and vertically into two layers. The smallest unit, defined by a particular column and layer, is a cell.

We also refer at times to the upper, middle and lower estuary. These roughly correspond to columns 0 to 6, 7 to 10, and 11 to 14, respectively (Fig. 3.1).

Nutrient loads into the estuary are needed both to interpret the budgets, and to drive the prognostic model. In section 10.2, we summarise the estimates of nutrient loads entering the estuary during the period of the Huon field program. We present inverse model results in section 10.3, and prognostic model results in 10.4. The implications for estuarine function and management are discussed in section 10.5.

10.2 Nutrient Loads

Nutrients were measured on 9 spatial surveys about every three months (Chapter 2). These provide some resolution of seasonal variation and cover a wide range of river flows. Loads are calculated here for periods corresponding to each spatial survey.

10.2.1 Huon River loads

River flow was estimated by scaling up data from Frying Pan Creek by a factor of 1.5 to allow for flows entering further downstream. River flow can vary significantly from day to day: this variation is evident in Table 3.2, which shows the effect of applying filters with time constants from 0 to 14 days to the river flow at the time of spatial surveys. The inverse model estimates of transport correlated best with river flow on the day of the survey (subsection 3.2.1, Table 3.1). This is consistent with the very short time flushing times, of order 1 day, estimated for the surface layer of the estuary.

Total loads from the river were calculated by multiplying river flow ($\text{m}^3 \text{s}^{-1}$) by concentrations measured upstream (mmol m^{-3}). For consistency with the transport estimates, load estimates were also based on river flow on the day of the survey. DIN was estimated as the sum of NO_x and NH_4 , DON as $\text{TDN} - \text{DIN}$, and particulate N as $\text{TN} - (\text{DON} + \text{NO}_x + \text{NH}_4)$. Ammonia was not measured on surveys HES 2 (Jul '96), HES 3 (Oct '96) and HES 6 (Oct '97), but levels were generally very low upstream, so a nominal value of 0.1 mmol m^{-3} was assumed for those surveys.

10.2.2 Mountain River

Mountain River enters in inverse model column 2, but its flow is included in the 1.5 multiplier for Huon River flows. As it drains some agricultural land, it might be expected that Mountain River carries higher N concentrations than the Huon; however, there is no significant difference in measured surface nutrient concentrations among model columns 0, 1, 2 or 3, upstream and downstream of the confluence with the Huon. (Concentrations are elevated in column 4, possibly due to supply of nutrients from sediments.) The Mountain River loads are therefore assumed to be included in the Huon River loads calculated in 10.2.1.

10.2.3 Kermandie loads

The Kermandie River carries much higher concentrations of nutrients than the Huon River. However, its average discharge is only $1.8 \text{ m}^3 \text{ s}^{-1}$ (Bobbi 1998), or about 2% of the Huon River's average of $90 \text{ m}^3 \text{ s}^{-1}$ (3.1.6). The Kermandie was not gauged continuously during the study, so flow estimates for the Kermandie were calculated as 2% of the Huon discharge at the same time. Differences in rainfall patterns and catchment characteristics mean this is only a rough approximation; there may be significant errors in individual surveys. Kermandie loads at the time of surveys were estimated by multiplying estimated discharge by measured concentrations. Kermandie concentrations were not measured on survey HES 2 (Jul '96), and concentrations of $10 \text{ mmol m}^{-3} \text{ NO}_x$, $10 \text{ mmol m}^{-3} \text{ NH}_4$, $20 \text{ mmol m}^{-3} \text{ DON}$ and 5 mmol m^{-3} part N, typical of other surveys, were assumed. A value of $10 \text{ mmol m}^{-3} \text{ NH}_4$ was also assumed for HES 3 (Oct '96) and 6 (Oct '97), as ammonia was not measured on those surveys.

10.2.4 STP loads

Data provided by the Huon Valley Council show typical DIN concentrations in STP discharge at Ranelagh as about 0.5 mol m^{-3} , mostly NH_4 . Total N concentrations are not given. The Ranelagh lagoon flow averages about $1200 \text{ m}^3 \text{ d}^{-1}$, which would give a DIN load of about 600 mol N d^{-1} . Wotherspoon et al. (1994) suggest that the Ranelagh Sewage Treatment Plant receives inputs from about 2000 people, which should produce a mean load of 0.7 mol N d^{-1} per person, or $1400 \text{ mol N d}^{-1}$. The DIN estimate accounts for half of this, suggesting there could be an organic N load similar in magnitude to the DIN load. These each represent loads of about 7 mmol N s^{-1} into column 1.

10.2.5 Finfish-farm loads

Based on figures provided by industry, the estimated total annual feed N in 1997 was 191 t N, with 123 t N or 64% input into the estuary, and 36% removed as harvested fish. Of the load to the estuary, we estimated that 13% is particulate and 87% is dissolved. The dissolved fraction is treated as DIN for model purposes. Although these discharges will vary seasonally, they are treated here, as a first approximation, as constant throughout the year, and correspond to $241 \text{ mmol DIN s}^{-1}$, and $36 \text{ mmol Part N s}^{-1}$. While these loads, used in model simulations, are based on 1997 calendar figures, we have also calculated annual budgets for September 1996 to August 1997, and September 1997 to August 1998, corresponding to the two field years of our study. These are presented in section 10.3.6 and show a significant increase in finfish-farm inputs between the two years. The prognostic model deals only in nitrogen units, but in section 10.3.6 we also discuss phosphorous loads from finfish farms.

The underlying data provided by industry allowed us to calculate N loads by farm site. These data allow identification of feed by company, and could have commercial significance, so we agreed not to publish these figures. However, in implementation of the SEEM model, N loads have been assigned by site.

10.3 Nutrient Budgets

10.3.1 Flux-estimation methods

The inverse model described in Chapter 3 is based on a simplified two-layer one-dimensional estuarine geometry, with the estuary divided into 15 columns between the freshwater head and the mouth (Fig. 3.1). The depth of the upper layer increases from 1.4 m upstream to 8.6 m at the mouth. The same upper layer depth distribution was adopted for all surveys (Chapter 3).

Horizontal and vertical advective and diffusive exchanges between adjacent cells were estimated by applying an inverse model to the salinities measured in each survey (section 3.4). Nutrients were measured at stations generally chosen to provide transects across as well as along the estuary (Chapter 2, 4). Most model cells, therefore, contained more than one station. For the purposes of budget calculations, measured nutrient concentrations were averaged within cells. In those cases where cells were not sampled, the missing values were obtained by linearly interpolating data from upstream and downstream cells.

Horizontal and vertical fluxes of nutrients were calculated by multiplying volume exchanges by nutrient concentrations at interfaces. Nutrient concentrations at interfaces were estimated by averaging concentrations of adjacent cells (ie central differencing). Fluxes were calculated for NO_x , NH_4 , TDN, TN, PO_4 , TDP, TP, Chl a. Other fluxes (eg DIN, DON, particulate N (PN), etc) were obtained by summing or differencing.

10.3.2 Sources, sinks and errors

It is possible, in principle, to calculate a number of derived fluxes, sources and sinks from these budgets. The flow in the Huon estuary is strongly stratified, and horizontal fluxes of salinity are dominated by downstream advection in the upper layer, and upstream advection in the lower layer. Both the downstream flux of nutrients in the upper layer, and the upstream flux in the bottom layer, can be calculated, and their difference gives the net downstream flux through vertical sections across the estuary. By comparing horizontal fluxes through adjacent vertical sections, one can estimate the net source or sink of nutrient in each column. It is also possible to calculate the net vertical flux from the bottom to the top layer in each column, and consequently the net source or sink of nutrient in each cell.

Net sources or sinks of nutrients in each cell can be computed, but in practice these estimates are often not reliable, due to a number of potential sources of error. The volume exchanges are based on measurements of salinity and river flow, and depend on the assumption that the salinity distribution is approximately in steady-state with the observed river flow. In practice, the river flow fluctuates. Salinity also fluctuates over time due to tide and wind forcing. In general, tidal excursions are small (about ± 150 m, except in mid-estuary, where they increase to ± 600 m; see section 3.1.4). Tidal fluctuations in salinity should also be small, except in mid-estuary where both tidal fluctuations and along-estuary salinity gradients are large. Wind forcing can cause local fluctuations in salinity by forcing the fresher surface plume from one side of the estuary to the other. The inverse calculations are based on salinity values which are averaged vertically and cross-estuary within cells. This should reduce the effect of wind-

induced fluctuations in the plume position, although cross-estuary gradients may be undersampled in some cells.

Spatial and temporal variation also introduce errors into estimates of mean cell concentrations of nutrients. Nutrient measurements (especially organic forms) are also less precise than salinity, and may contain analytical or sampling errors. Finally, errors arise from ignoring cross-estuary variations in flow, salinity and nutrient concentrations.

These errors are essentially unavoidable, although minimised by the intensive spatial sampling program we adopted. One can gain some indication of the resulting errors in flux estimates by plotting fluxes vs position along the estuary. The net horizontal fluxes in surface and bottom layers vary quite smoothly downstream, with fluctuations of about 5 to 10%. However, estimates of sources or sinks within individual cells fluctuate widely. This is not surprising, as they are obtained as the difference between large fluxes of similar magnitude. Estimates of net downstream flux generally vary smoothly, but at times fluctuate sharply at the mouth.

Net downstream flux represents the difference between downstream flux in the surface layer, and upstream flux in the bottom layer. The net downstream volume flux must equal the river flow. In many cases, nutrient concentrations contrast sharply in bottom and surface layers, resulting in substantial differences in magnitude between surface and bottom fluxes. Under these conditions, the net downstream fluxes are quite reliable. However, in some cases, nutrient concentrations are similar in surface and bottom layers near the mouth. This often occurs when stratification is weak, and salinities are also similar in top and bottom layers. Estimated volume exchanges are then very large, and calculated horizontal fluxes in surface and bottom layers are very large and almost equal. In such circumstances, estimates of net downstream flux are unreliable.

Because of the uncertainties in local divergences, we focus here on net downstream fluxes at column 4 (near Huonville) and at the mouth, and their implications for nutrient budgets for the estuary in between. These budgets provide an overall picture of nutrient sources and sinks and transformations within the estuary as a whole. Cases where net fluxes at the mouth are uncertain are noted in the following discussion, and alternative estimates considered. In general, net fluxes of nitrate and TN are uncertain in winter, because of large opposing surface and bottom fluxes. In spring, autumn and summer, there are usually substantial differences in surface and bottom concentrations, which translate into significant net fluxes.

For purposes of discussion, we have divided the spatial surveys into three groups: "winter" - HES 2 (Jul '96), HES 5 (Jun '97), and HES 10 (Aug '98), "summer" - HES 4 (Feb '97), HES 6 (Oct '97), HES 7 (Dec '97), HES 8 (Feb '98) and "spring/autumn" - HES 3 (Oct '96), HES 9 (May '98). These groups are based on the nutrient regime, rather than strictly on time of year. Winter surveys show similar high marine nitrate concentrations in both surface and bottom waters. Summer surveys show strong depletion of nitrate in surface waters. Spring and autumn surveys represent transitional regimes, with partial depletion of nitrate in surface waters. Thus HES 3 and HES 6 both took place in October, but HES 6 (Oct '97) shows "summer" characteristics, with depletion of surface nitrate.

10.3.3 Nutrient budgets in winter

In the winter surveys HES 2 (Jul '96), HES 5 (Jun '97), and HES 10 (Aug '98), nitrate and phosphate concentrations are high in both bottom and surface layers at the mouth, and river flows are moderate to high (108, 64 and 59 m³ s⁻¹ respectively). As a result, the surface and bottom fluxes of DIN and DIP at the mouth are very large (about 22000, 7700 and 5900 mmol s⁻¹ NO_x, and 2200, 900 and 700 mmol s⁻¹ PO₄, in HES 2, 5 and 10 respectively – Figure 10.1). However, because there is little difference between surface and bottom concentrations, the net fluxes at the mouth are small, and hardly distinguishable from zero. In other words, while there is a large flux of marine nitrate and phosphate circulating through the estuary, there is little evidence of biological uptake within the estuary.

Ammonia concentrations and fluxes are typically low compared with those of NO_x. Surface and bottom fluxes of NH₄ in HES 5 (Jun '97) and 10 (Aug '98) are about 300 mmol s⁻¹, with net fluxes less than 100 mmol s⁻¹ (Fig. 10.1).

Catchment inputs of DIN and DIP in winter are negligible compared with the marine throughput (Fig. 10.1). Downstream net fluxes of NO_x at column 4 for HES 2 (Jul '96), 5 (Jun '97) and 10 (Aug '98) are 40, 7 and –80 mmol s⁻¹ (i.e. upstream flux) respectively. Net downstream fluxes of NH₄ for HES 5 (Jun '97) are 10 mmol s⁻¹, and for HES 10 (Aug '98), 14 mmol s⁻¹. Downstream DIP fluxes at column 4 are also negligible. The estimated Kermantie inputs are also low: 43, 26 and 27 mmol s⁻¹ DIN for HES 2, 5 and 10 respectively.

Estimated finfish-farm loads of DIN (260 mmol s⁻¹) are also small compared with the very large winter fluxes of nitrate through the estuary.

The catchment provides a significant organic N load, principally as DON. Net downstream fluxes of DON past column 4 in HES 2, 5 and 10 are about 1600, 800 and 600 mmol s⁻¹ respectively (Fig. 10.1). Loads of particulate N are typically an order of magnitude lower. By comparison, the Kermantie and finfish-farm loads carry negligible amounts of organic N. The catchment is a minor source of organic P (< 10 mmol s⁻¹).

At the mouth, there are also large surface and bottom fluxes of DON and PN: 20000, 5000 and 3000 mmol s⁻¹ DON, and 5000, 1000 and 2000 mmol s⁻¹ PN, for HES 2 (Jul '96), 5 (Jun '97) and 10 (Aug '98) respectively (Fig. 10.1). These fluxes are due to high concentrations of background DON (possibly marine) and PN (possibly due to resuspension). There is typically a net flux of DON out of the estuary: this flux is highly uncertain in HES 2, but in HES 5 and 10, it is 900 and 440 mmol s⁻¹ respectively, and similar in size to the catchment input of DON. (While there is a substantial difference in estimated surface and bottom DON fluxes in HES 2, both these fluxes are highly uncertain, so that the difference is also uncertain.) The data are therefore consistent with the hypothesis that little DON is produced or consumed in the estuary in winter, and that the catchment DON load is refractory, and exported more or less unchanged.

The net flux of PN at the mouth is also uncertain in HES 2 (Jul '96), but there is a net export of PN in HES 5 (Jun '97) and HES 10 (Aug '98) of 1000 and 2000 mmol s⁻¹ respectively. The estimated net fluxes of Chl *a* out of the estuary are close to zero in HES 5 and 350 mg Chl *a* s⁻¹ in HES 10. At typical C:Chl *a* ratios of 50:1 and Redfield C:N ratios of 5.7:1 (by weight), 350 mg Chl *a* s⁻¹ would correspond to about 250 mmol s⁻¹ PN – much less than the observed PN export. This suggests that the PN exported in winter may be mostly resuspended detritus.

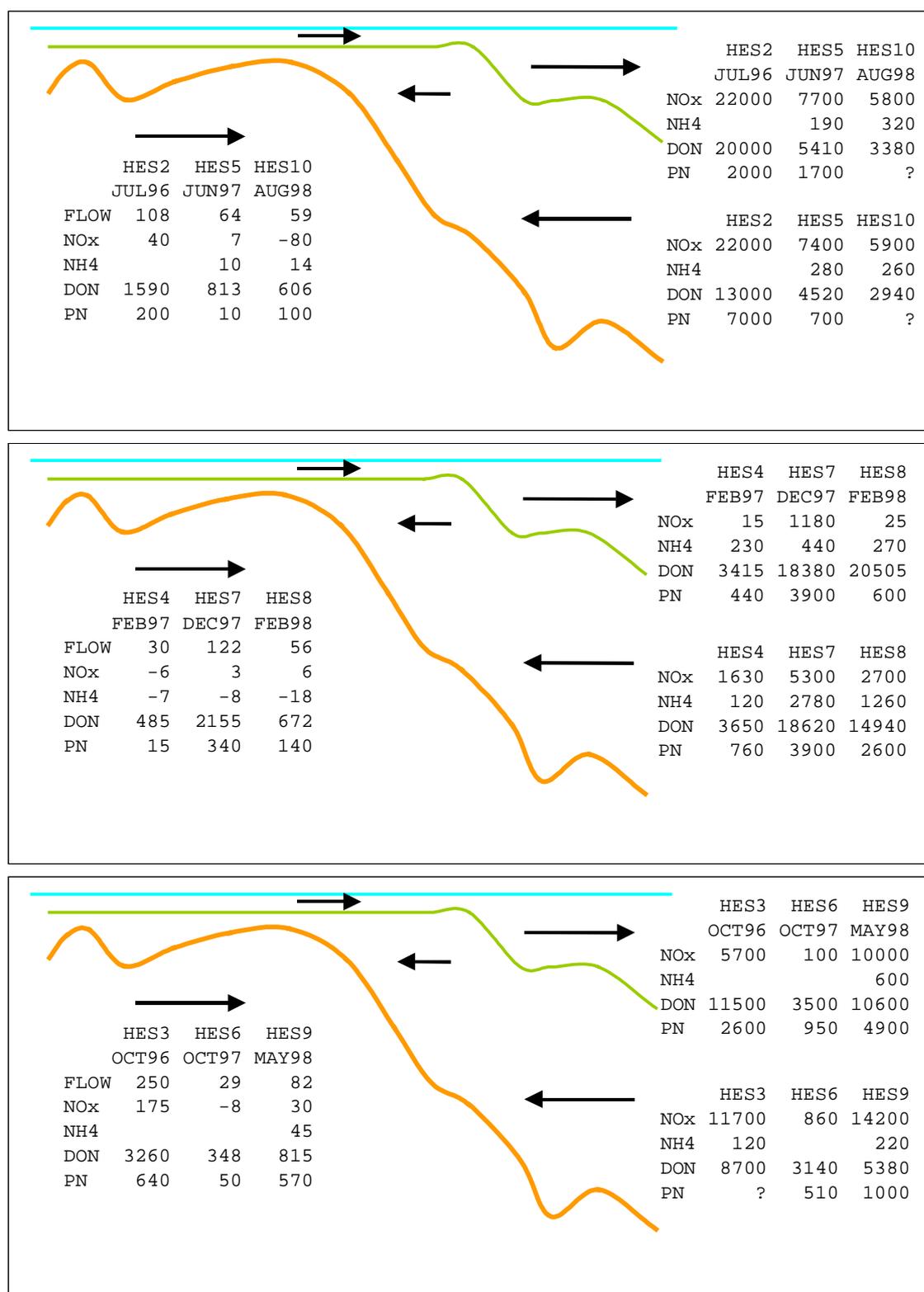


Fig. 10.1 Nitrogen fluxes for Huon Estuary based on inverse model for winter surveys (HES 2,5,10), summer surveys (HES 4,7,8) and spring/summer/autumn surveys (HES 3,6,9). Tables show net downstream fluxes at column 4, and 2-way fluxes at the mouth. All fluxes are in mmol N s^{-1} , except for river flow in $\text{m}^3 \text{s}^{-1}$.

The TN budget for the estuary in winter is uncertain because of the large and almost balanced fluxes of marine DIN and DIP. In HES 5 (Jun '97), allowing for all known loads, the estuary appears to be a net source of approximately 1000 mmol s⁻¹ TN, and 200 mmol s⁻¹ TP. In HES 10 (Aug '98), it appears to be a source of about 1500 mmol s⁻¹ TN, but 0 TP. If accurate, these figures probably represent primarily export of detrital particulate N.

10.3.4 Nutrient budgets in summer

In surveys HES 4 (Feb '97), HES 6 (Oct '97), HES 7 (Dec '97), and HES 8 (Feb '98), nitrate and phosphate are depleted to near zero in the surface layer near the mouth. Concentrations in the bottom layer are also lower than in winter, but still sufficiently high to provide significant fluxes of marine DIN and DIP into the estuary in the bottom layer (Fig. 10.1). Net upstream fluxes of NO_x and DIP at the mouth in HES 4, 6, 7 and 8 respectively, are 1600, 800, 4100, 2700 mmol s⁻¹ NO_x, and 180, 120, 530 and 630 mmol s⁻¹ PO₄.

By comparison, inputs of DIN and DIP from catchments upstream of column 4 (Fig. 10.1), or from Kermantie River, are negligible: less than 100 mmol s⁻¹ DIN and less than 20 mmol s⁻¹ PO₄. Estimated inputs of DIN from finfish farms are also small, but not negligible. Catchment N loads are again dominated by the input of DON from upstream (490, 350, 2150, and 670 mmol s⁻¹ in HES 4 (Feb '97), HES 6 (Oct '97), HES 7 (Dec '97), and HES 8 (Feb '98) respectively). PN loads are typically about 20% of DON loads.

There are large surface and volume fluxes of DON at the mouth in summer (Fig. 10.1), with fluxes out in the surface layer of 3400, 3500, 18400 and 20500 mmol s⁻¹ in HES 4 (Feb '97), HES 6 (Oct '97), HES 7 (Dec '97), and HES 8 (Feb '98) respectively. There is a net export of DON at the mouth in Oct '97 (360 mmol s⁻¹) and Feb '98 (5600 mmol s⁻¹), but a net influx in Feb '97 and Dec '97 (both 240 mmol s⁻¹). With the exception of HES 8 (Feb '98), the net fluxes are small compared with two-way fluxes, and possibly not significantly different from zero. Unlike the winter surveys, in summer there is no simple relationship between the net flux of DON at the mouth and catchment inputs. In HES 4 (Feb '97) and HES 7 (Dec '97), the estuary appears to be a net sink of DON. In HES 6 (Oct '97), the export of DON at the mouth is very close to the river load. In HES 8 (Feb '98), the estuary is a large source of DON. The downstream flux in the surface layer is still much larger than the river load in all cases, so the data are not incompatible with the hypothesis that river DON is largely refractory and is exported unchanged.

The two-way exchanges of PN at the mouth are substantially smaller than DON in all surveys. There is a net PN influx at the mouth of 320 mmol s⁻¹ in Feb '97, and 2000 mmol s⁻¹ in Feb '98, an efflux of 440 mmol s⁻¹ in Oct '97, and zero net exchange in Dec '97. Interestingly, there is a net influx of Chl *a* at the mouth in Oct '97, Dec '97 and Feb '98: 260, 2900 and 1800 mg s⁻¹ respectively, but negligible net flux for Feb '97. The net influx of Chl *a* reflects a bias of chlorophyll towards the bottom layer at the mouth. This has the effect of retaining phytoplankton in the estuary (see modelling below). The large net influx of Chl *a* contributes to a large net influx of PN in Feb '98, but is balanced by an efflux of detrital PN in Dec '97.

When finfish-farm and Kermantie loads are included, the estuary is acting as a net sink of TN in HES 4, 6, 7 and 8 of about 2900, 700, 9600 and 1250 mmol s⁻¹ respectively, and as a net sink of TP of about 140, 66, 740 and 500 mmol s⁻¹, respectively. With the exception of HES 8 (Feb

'98), where the TP and PO₄ influx is anomalously high, the TN and TP sinks are roughly in Redfield balance (16:1), suggesting that N and P are being removed and buried as organic matter.

The overall picture in these summer surveys is of a biologically active estuary that is importing marine DIN and DIP and converting them into organic matter. It is also importing substantial amounts of marine particulate N, including phytoplankton. Only in HES 8 (Feb '98) is the estuary exporting large amounts of organic matter in the form of DON, and this is partly offset by the influx of marine PN. In the other summer surveys, the estuary appears to be sequestering most of the organic matter produced and additional organic matter imported, presumably in sediments.

The fate of the organic matter involved in this internal sink is unclear. Some may be remineralized in the sediment, with the N lost to denitrification, and the PO₄ bound to inorganic sediments. In HES 8 (Feb '98), where the TN and TP sinks are not in balance, it is unlikely that the P sink could be explained by burial of organic matter. The data suggest that P is adsorbed to sediments, while N is exported as DON.

The magnitude of fluxes and production in summer is strongly controlled by river flow. The low river flows (30 and 29 m³ s⁻¹) in HES 4 (Feb '97) and HES 6 (Oct '97) respectively correspond to low exchanges and fluxes. The high river flows (122 and 56 m³ s⁻¹) in HES 7 (Dec '97) and HES 8 (Feb '98) correspond to high exchanges and fluxes. It appears that, over this range of flows, production in the estuary in summer is driven primarily by marine nutrient loads, which are controlled in turn by estuarine circulation.

10.3.5 Nutrient budgets in spring and autumn

The surveys HES 3 (Oct '96) and HES 9 (May '98) do not fit the summer or winter pattern well. (Note that HES 6 (Oct '97), has been plotted in Figure 10.1 with HES 3 and 9, but really fits the summer pattern, with surface nutrient depletion.) Both HES 3 and HES 9 have high DIN and DIP concentrations in bottom waters, while surface concentrations are reduced but not depleted. As a result, there are large net fluxes of marine DIN and DIP into the estuary: 6000 and 4200 mmol s⁻¹ DIN, and 440 and 50 mmol s⁻¹ DIP, in HES 3 (Oct '96) and HES 9 (May '98) respectively. While HES 3 had the highest river flow (250 m³ s⁻¹) of any survey, and river flow in HES 9 was moderate (82 m³ s⁻¹), combined catchment inputs of DIN and DIP in HES 3 and 9 were still very small compared to net marine inputs (~ 300 and 60 mmol s⁻¹ DIN, and 15 and 0 mmol s⁻¹ DIP, in HES 3 and 9 respectively).

The very high river flow in HES 3 (Oct '96) is associated with large net inputs of DON (3300 mmol s⁻¹) and PN (640 mmol s⁻¹). Catchment inputs in HES 9 (May '98) are more modest: 820 mmol s⁻¹ DON and 570 mmol s⁻¹ PN). In Oct '96, there is a strong net export of both DON (2800 mmol s⁻¹) and PN (4200 mmol s⁻¹). The DON export is similar to the catchment DON load, and a substantial fraction of the PN export can be accounted for by phytoplankton export (1950 mg Chl *a* s⁻¹). In May '98, the export of DON (5200 mmol s⁻¹) is much larger than catchment loads, and the export of PN is also large (3900 mmol s⁻¹). The estimated export of Chl *a* is very large (14200 mg s⁻¹) and difficult to reconcile with the PN export.

In Oct '96, the estuary is a net sink for TN (about 2900 mmol s⁻¹) and TP (about 320 mmol s⁻¹). In this survey, the spring bloom appears to be underway, reducing surface nitrate and phosphate in the estuary. Despite the high river flow and rapid flushing, the estuary is biologically active, stripping 6000 mmol s⁻¹ of marine DIN out of the marine throughflow. It appears that about half of this is sequestered in the estuary, while the other half is exported, mostly as phytoplankton biomass.

In May '98, the estuary is a net source for TN (3400 mmol s⁻¹), and TP (260 mmol s⁻¹). The estuary is also biologically active (this survey occurs during a late phase of the dinoflagellate bloom, with very high chlorophyll levels). The estuary strips about 4200 mmol s⁻¹ of marine DIN out of the throughflow, but exports a much larger amount of organic N as both DON and PN. The estuary is acting as a large net source of organic N, presumably derived from sediments.

10.3.6 Annual Budgets of TN, TP

Annual budgets of TN and TP were estimated for the periods September 1996 to August 1997, and September 1997 to August 1998. These periods span the field program of our study, but also contain the spring, summer and autumn bloom periods.

Loads were calculated more or less as in section 10.2, with the following modifications. Catchment inputs were divided into upper catchment (above Judbury) and lower catchment (below Judbury). Additional information was drawn from Wotherspoon et al. (1994), Gallagher (1996) and the Huon Valley Council. Lower catchment inputs were augmented to include estimates of loads from agriculture and grazing. The latter were based on estimated areas of improved pastures and orchards, cropland and urban development, and yields per unit area for these land use types, and totalled 182 t N p.a. and 8.4 t P p.a., more than half of the lower catchment loads, and about 20% (N) and 33% (P) of total catchment loads. (There is considerable uncertainty in estimates of this type: by comparison, loads from finfish farms are very well known.) Estimates of STP loads were augmented to include other smaller STPs (Geeveston and Cygnet) and diffuse loads from non-sewered properties. The result represents an upper bound for inputs due to human waste, as it assumes all loads from non-sewered properties find their way to the estuary. This load is still small compared to catchment and marine inputs.

Estimates of N and P loads from atmospheric deposition and tip leachate are uncertain, but probably small. Annual loads from finfish farms were based on the same assumptions as in section 10.2.3, but the numbers differ as loads there were calculated for calendar 1997. There was an increase of about 30% in estimated loads from 1996/97 to 1997/98.

The exchanges with the marine environment were obtained by averaging the exchanges computed for spatial surveys within each year. These estimates are undersampled with respect to river flow, and consequently the annual exchanges are subject to some uncertainty. However, they do serve to illustrate the very large fluxes of marine nitrogen and phosphorous through the estuary. Interestingly, despite an estimated 30% increase in the separate surface and bottom fluxes of N at the marine boundary from 1996/97 to 1997/98, the net annual flux of N across the

marine boundary is about 100 t N directed into the estuary in both years. (Note this is quite separate from the 30% increase in finfish-farm loads across the two years: both finfish-farm loads and net marine loads have increased in 1997/98.) The estimated net flux of P is about 70 t P inward in 1996/97, and 179 t P inward in 1997/98.

The net fluxes of TN and TP at the marine boundary vary strongly in both magnitude and sign among surveys, so that these annual average net fluxes are particularly subject to error. Taken at face value, they indicate that the estuary is a substantial sink for N and P: 1270 t N and 133 t P in 1996/97, and 1240 t N and 270 t P in 1997/98. The N:P ratio by moles of sequestration is about 22 in 1996/97, and 10.5 in 1997/98. These ratios straddle the Redfield ratio of 16:1 by moles for N:P uptake by phytoplankton.

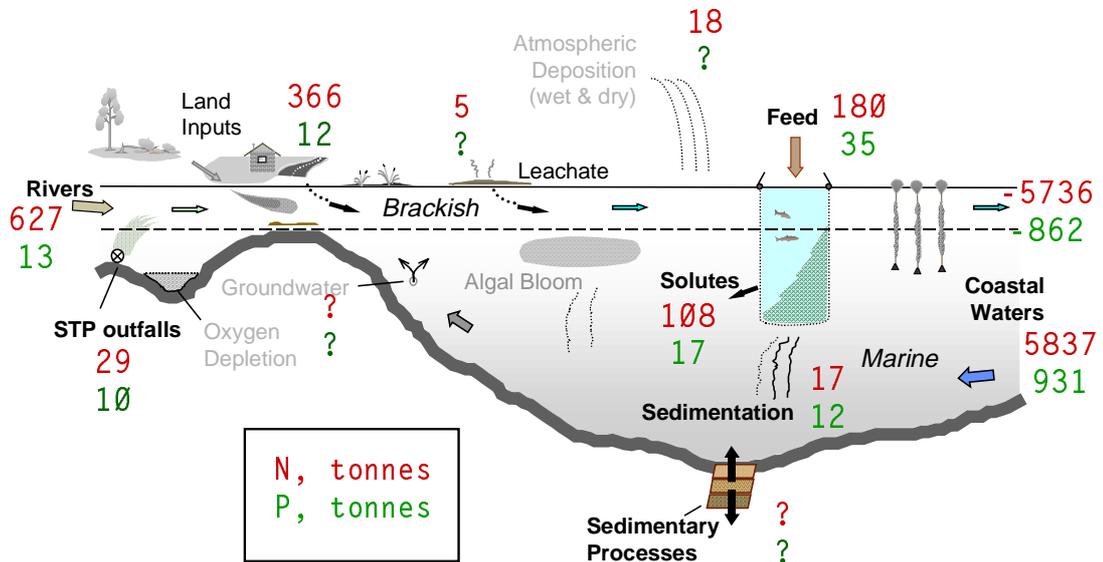
The possible fates of this N and P include burial as organic matter in sediments, burial as inorganic adsorbed species in sediments, or (in the case of nitrogen) denitrification to N_2 gas. The TP sink is most likely dominated by adsorption of inorganic P to sediments. It is difficult without additional information to apportion the TN sink between denitrification and burial. The individual survey budgets suggest that particulate N accumulates in sediments in summer, and some is resuspended and exported in winter. The particulate N pool in the sediments is sufficiently large not to preclude annual burial of substantial amounts of organic N. The high bottom water ammonia concentrations in the upper estuary, and also in the lower estuary in summer following algal blooms, suggest high rates of ammonia release from sediments (see also section 10.4).

One must be careful with TN and TP budgets such as Figure 10.2 in linking net sources to net sinks. The catchment is a large source of TN, but this is mainly in the form of DON, which the analysis of individual surveys suggests is probably exported from the estuary unchanged. (The TN contribution from agricultural land-use in the lower catchment would be expected to include more labile N.) Refractory PON inputs from the catchment could well be buried in the sediments, but these represent less than 20% of the catchment TN load (Fig. 10.1).

The N inputs from finfish farms and sewage are almost all in the form of DIN or labile DON or PON. These would participate in nitrogen cycling within the estuary and could be either buried or lost to denitrification. As discussed, about 40% of the net bottom layer TN influx at the marine boundary consists of nitrate. Part of this is converted to organic N by phytoplankton and may be either denitrified or buried.

If the DON load from the undisturbed catchment is mostly exported unchanged, then the dominant catchment contributions to the internal sink of TN (1240 t N p.a. in 1997/98) are agricultural land-use (up to 190 t N p.a.) and finfish farms (159 t N p.a.), with minor contributions from sewage, leachate and atmospheric deposition. The remainder (about 820 t N p.a.) is supplied at the marine boundary in the form of nitrate. According to this analysis, about two-thirds of the internal sink is supplied by marine nitrate, with the remainder split between agricultural and finfish-farm contributions. It is important to note that the impact of finfish-farm loads will depend on the extent to which they contribute to phytoplankton production and nitrogen cycling within the estuary, before being exported, buried or denitrified. We attempt to assess this using a process model in 10.5.2.

Nutrient Fluxes in the Huon Estuary, 1996-97



Nutrient Fluxes in the Huon Estuary, 1997-98

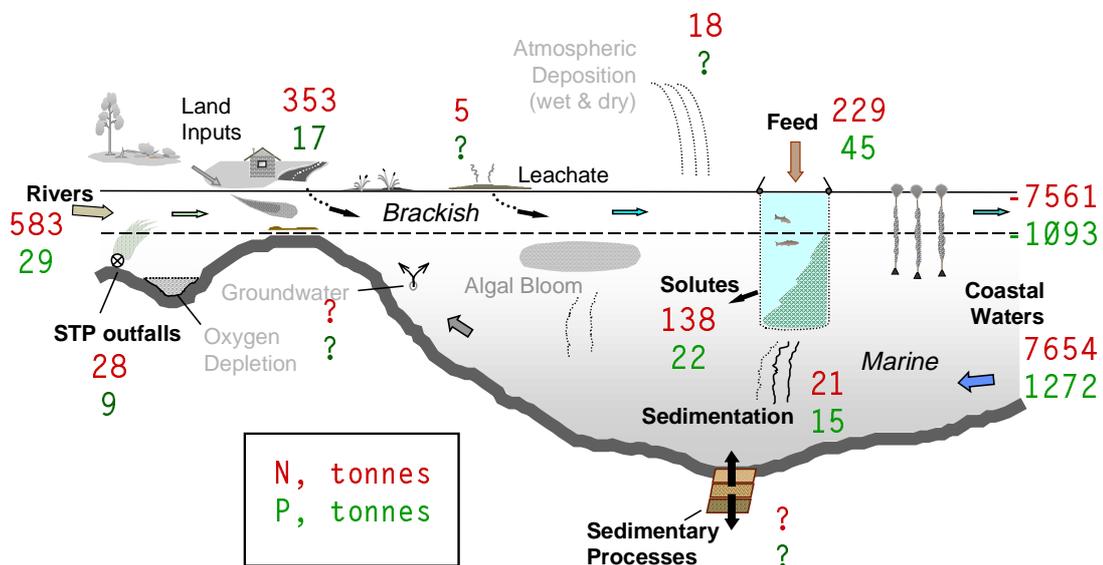


Fig. 10.2 Annual budgets of TN and TP for the Huon Estuary for the periods September 1996 to August 1997, and September 1997 to August 1998. See text for further details.

10.4 Water Quality Model

10.4.1 Model description

We have applied a simple water-quality model (SEEM) to interpret data from the Huon Estuary. This model represents the cycling of nitrogen through four pools: DIN, phytoplankton (PHY), labile detritus (DET) and refractory organic matter (REF). A full description of this model is given in Parslow et al. (1999).

The dynamical equations describing the local rate of change of these pools are:

$$dDIN/dt = -\mu \cdot PHY + (1-f_{DET}) \cdot m \cdot PHY + (1-E_{DEN}) \cdot R_{SED} + R_w$$

$$dPHY/dt = \mu \cdot PHY - m \cdot PHY$$

$$dDET/dt = f_{DET} \cdot m \cdot PHY - r_{DET} \cdot DET$$

$$dREF/dt = f_{REF} \cdot r_{DET} \cdot DET - r_{REF} \cdot REF$$

Here, μ is the phytoplankton growth rate, m the phytoplankton loss rate, f_{DET} the proportion of phytoplankton loss assigned to DET, R_{SED} the sediment respiration rate, E_{DEN} the denitrification efficiency, and R_w the water column respiration rate. The parameters r_{DET} and r_{REF} are the breakdown rates of labile and refractory detritus, and f_{REF} is the proportion of labile detritus converted to refractory detritus.

The phytoplankton growth rate is the product of a maximum growth rate, and light- and nutrient-limitation terms. The light-limitation term depends on the ratio of available light to a light saturation parameter Ik . Nutrient limitation depends on available DIN and a half-saturation constant for growth, K_N .

The available light is calculated using a light attenuation coefficient K given by:

$$K = K_o + k_{PHY}^* \cdot PHY + k_{DET}^* \cdot DET + k_{REF}^* \cdot REF,$$

where the specific light-attenuation coefficients are based on literature values. Light attenuation is very high in the Huon, due to high levels of coloured dissolved organic matter (CDOM) in river input (Chapter 6). The CDOM is diluted as river water mixes with marine waters. In order to obtain the correct distribution of light attenuation, we have used the model variable REF as a tracer for CDOM. The river value is set equal to observed freshwater DON, while the marine boundary value is set to zero. REF breaks down very slowly, and effectively behaves conservatively (as does CDOM) in the estuary. We have assigned a high value to k_{REF}^* to match the observed light attenuation due to CDOM.

The phytoplankton loss term m depends on PHY, and is formulated in such a way as to approximate the functional and numerical response of grazers to increasing phytoplankton density. Grazers have maximum impact at intermediate phytoplankton concentrations, but become satiated at high concentrations (Parslow et al, 1999).

The model represents a sediment layer implicitly, by assuming that a certain fraction σ of each particulate constituent is suspended, and the rest sedimented. This suspended fraction depends on sinking rate, a prescribed resuspension rate, and bottom area. Thus, the remineralisation of organic matter is partitioned between water column and sediment. A fraction E_{DEN} of DIN produced through sediment respiration is lost to denitrification; this fraction decreases with increasing sediment respiration rate, according to an empirical relationship derived in a study of Port Phillip Bay (Murray and Parslow 1999).

This biogeochemical model is implemented within a physical transport model based on the inverse model described in Chapter 3. The estuary is divided vertically into two layers and horizontally into 15 columns, labelled 0 to 14 (Fig. 3.1). Volume exchanges between cells are based on river flow, using the horizontal and vertical exchanges calculated from salinity distributions by inverse methods in Chapter 3. When the transport model was run with these exchanges, it reproduced observed salinity distributions reasonably well (Section 3.4.6).

10.4.2 Model loads and forcing

Although the model is implemented as a time-series model, it has been applied here as a steady-state model, under constant loads and forcing conditions corresponding to each of the spatial surveys. This is consistent with the steady-state assumptions used by the inverse model to estimate the physical exchanges. Given the rapid flushing times characteristic of the estuary even when flow is low, the assumption that water-column concentrations are in quasi-equilibrium with seasonal flows and loads may be reasonable. However, the steady-state assumption, and the simple model representation of sediments, do not allow for seasonal accumulation or drawdown of organic matter in sediments. The implications are discussed further below.

Loads of DIN, detrital N and DON corresponding to each spatial survey were calculated as described in section 10.2. Riverine loads into column 0 were based on river flow and local concentrations. Additional loads from Kermadie River and Ranelagh Sewage Treatment Plant were assigned to the appropriate model cells. Finfish-farm loads of DIN and DET were assigned to cells corresponding to the farm sites. As noted earlier, these loads are assumed to be constant throughout the year, and to be discharged into the bottom layer.

Marine boundary conditions for each survey are imposed by specifying the concentrations of all variables in top and bottom layers in column 14 to be equal to the average observed concentrations in those cells for the survey in question.

Surface-light levels are based on estimated clear sky irradiance, reduced by typical cloud factors for this region.

10.4.3 Simulation of spatial surveys

Although the model contains 19 parameters, many of these have relatively fixed values and/or the model results are insensitive to their values. We have found previously that the model was able to reproduce observations for a variety of estuaries and conditions with relatively few changes in parameters (Parslow et al. 1999). In the runs described here, we started with a

standard set of parameter values, and modified only 3 parameters: maximum phytoplankton growth rate μ_{\max} , half-saturation constant K_N , and phytoplankton sinking rate W_p .

In analysing the model behaviour, we have again divided the spatial surveys into three groups: “winter” - HES 2 (Jul '96), HES 5 (Jun '97), HES 10 (Aug '98), “summer” - HES 4 (Feb '97), HES 6 (Oct '97), HES 7 (Dec '97), HES 8 (Feb '98) and “spring/autumn” - HES 3 (Oct '96), HES 9 (May '98). Note again that we have classified HES 6 (Oct 97) as a summer survey, because of strong surface nutrient depletion.

Winter surveys

The winter surveys have high DIN values (about 4 mmol m^{-3}) in both surface and bottom layers at the marine boundary. DIN concentrations are saturating to phytoplankton growth throughout most of the estuary, but phytoplankton growth is limited by low surface light and high light attenuation. River flows are moderate to high, and the rapid flushing of the surface layer, combined with low growth rates, makes it quite difficult to maintain even the low chlorophyll concentrations observed in winter.

During high flow periods, phytoplankton are flushed out of the surface layer very rapidly. In standard runs, phytoplankton are assumed to be neutrally buoyant, and advected with the estuarine circulation. Predicted phytoplankton biomass is increased if a sinking rate of 0.2 to 2 m d^{-1} is prescribed. This increases retention in the estuary by allowing sufficient phytoplankton to sink out of the upper layer, which is flowing rapidly downstream, into the deep bottom layer, which flows slowly upstream. Conditions for growth in the lower layer are poor, because of the lack of light, but sufficient cells are advected upstream and entrained into the upper layer to support an elevated biomass overall.

During HES 2 (Jul '96), with a river flow of $108 \text{ m}^3 \text{ s}^{-1}$, introducing a phytoplankton sinking rate improves the fit to observed column-integrated biomass, although the vertical distribution is wrong, with the maximum predicted biomass in the lower layer (Fig. 10.3). The best fit to observed phytoplankton distribution is obtained with a relatively high maximum growth rate of 1.5 d^{-1} . In HES 5 (Jun '97), with moderate flow ($\sim 60 \text{ m}^3 \text{ s}^{-1}$), the model best reproduces observed biomass with zero sinking rate and a maximum growth rate of 1.0 d^{-1} (Fig. 10.6). In HES 10 (Aug '98), with moderate flow ($60 \text{ m}^3 \text{ s}^{-1}$), the best fit is obtained for zero sinking rate and a maximum growth rate of 0.5 d^{-1} (Fig. 10.12).

Because phytoplankton uptake of DIN is low in winter, the predicted DIN distribution is determined primarily by circulation and mixing of high DIN marine water and low DIN river water. Thus, high DIN values penetrate further upstream in high-salinity bottom water. The model cannot however reproduce the high DIN concentrations observed in bottom waters near the head of the estuary. The observed concentrations can only be explained by an efflux of DIN from bottom sediments, or an unknown load into bottom waters. The model predicts very little flux of DIN from bottom sediments in winter, as there are no major sources of labile organic N. The efflux of DIN from sediments can be increased by setting denitrification to zero. However, this also has little impact on predicted DIN concentrations, as the predicted sediment respiration is very low.

Possibly the observed DIN is due to breakdown of organic N sequestered in sediments in the previous summer. The simple model used here is unable to represent these effects. There is also

some evidence (see below) that the vertical mixing of surface and bottom waters calculated by the inverse model may be too high near the head of the estuary.

Summer Surveys

In summer, phytoplankton growth is still limited by light availability, especially in the bottom layer. DIN concentrations in the bottom layer at the marine boundary remain high, but DIN is depleted in surface waters throughout the estuary, and often depleted in bottom waters in mid-estuary. As a result, phytoplankton growth is nutrient-limited in surface waters, and possibly in bottom waters in mid-estuary.

The model is generally able to reproduce the typical observed biomass levels in summer, but has difficulty in reproducing its vertical and horizontal distribution exactly. To reproduce the observed drawdown of DIN in summer, the half-saturation constant for nutrient-limited growth, K_{N_s} , needs to be set to about 0.35 mmol m^{-3} . This is relatively low for coastal waters, and corresponds to the value adopted for small flagellates in the Port Phillip Bay model (Murray and Parslow 1997).

In HES 4 (Feb '97), the model best reproduces the observed DIN distribution with zero sinking rate, and a maximum growth rate of 1 d^{-1} (Fig. 10.5). However, the predicted phytoplankton biomass is high throughout the estuary, while the observed biomass is lower through most of the estuary, but with a large isolated peak at 10 km from the head. In HES 6 (Oct '97), the same parameter set results in quite a good fit to observed biomass and DIN in the middle and lower estuary (Fig. 10.7), but the model underestimates DIN in bottom waters, and overestimates DIN in surface waters, at the head of the estuary. This may result from too much vertical mixing.

River flow is low (about $30 \text{ m}^3 \text{ s}^{-1}$) in both HES 4 (Feb '97) and 6 (Oct '97), but reaches $122 \text{ m}^3 \text{ s}^{-1}$ in HES 7 (Dec '97). The model can reproduce the observed high biomass only with a high maximum growth rate (1.5 d^{-1}). The model overestimates surface chlorophyll and DIN upstream.

In HES 8 (Feb '98), with moderate flow ($56 \text{ m}^3 \text{ s}^{-1}$), the model reproduces observed biomass and DIN quite well throughout the middle and lower estuary with zero phytoplankton sinking rate, and a high maximum growth rate of 1.5 d^{-1} (Fig. 10.9). The model again underestimates DIN in the bottom layer at the head of the estuary.

The model can broadly reproduce the summer pattern of increased phytoplankton biomass and DIN drawdown. In the model, this results primarily from increased light availability, and secondarily from reduced flushing rates for some (but not all) surveys. The best fits to summer surveys are obtained using maximum growth rates that are higher on average than those used for winter surveys, but the two sets overlap, and there is not a good correlation between season or temperature and fitted maximum growth rate. Observed variation in species composition (small flagellates, diatoms and dinoflagellates) could account for this, as maximum growth rates across these groups could easily vary by a factor of 2 or more.

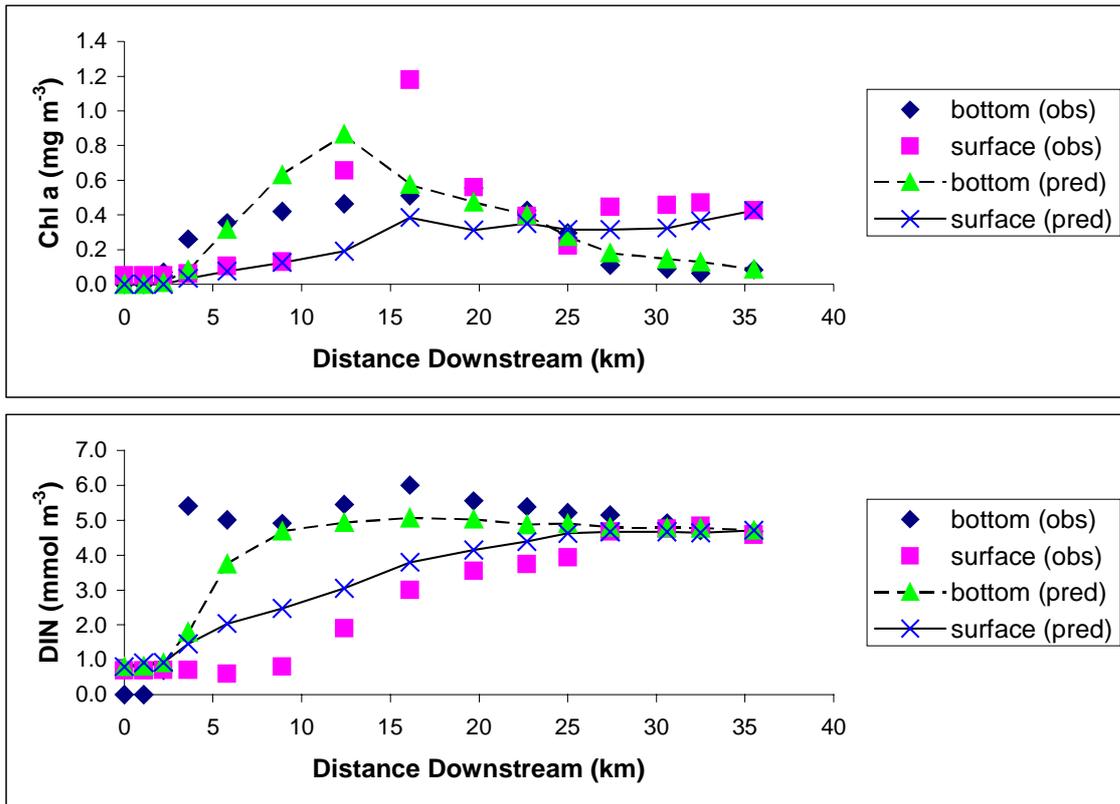


Fig. 10.3 Predicted and observed chlorophyll and DIN concentrations for HES 2 (Jul '96).

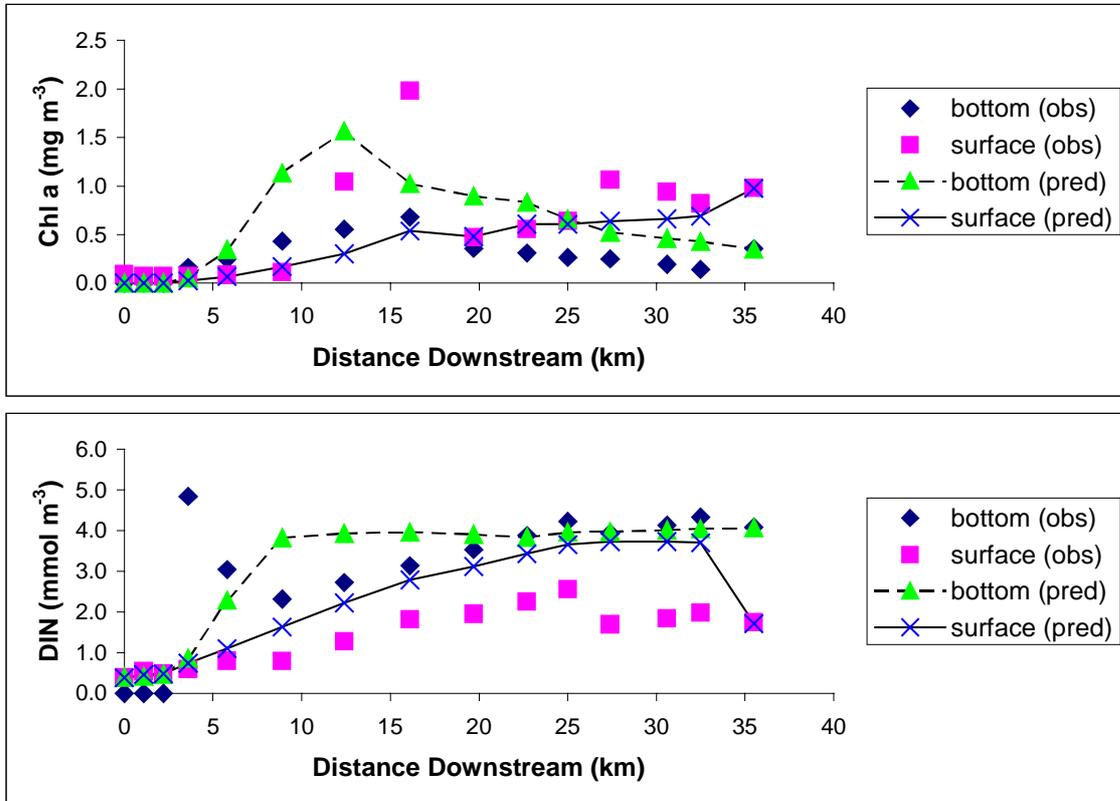


Fig. 10.4 Predicted and observed chlorophyll and DIN concentrations for HES 3 (Oct '96).

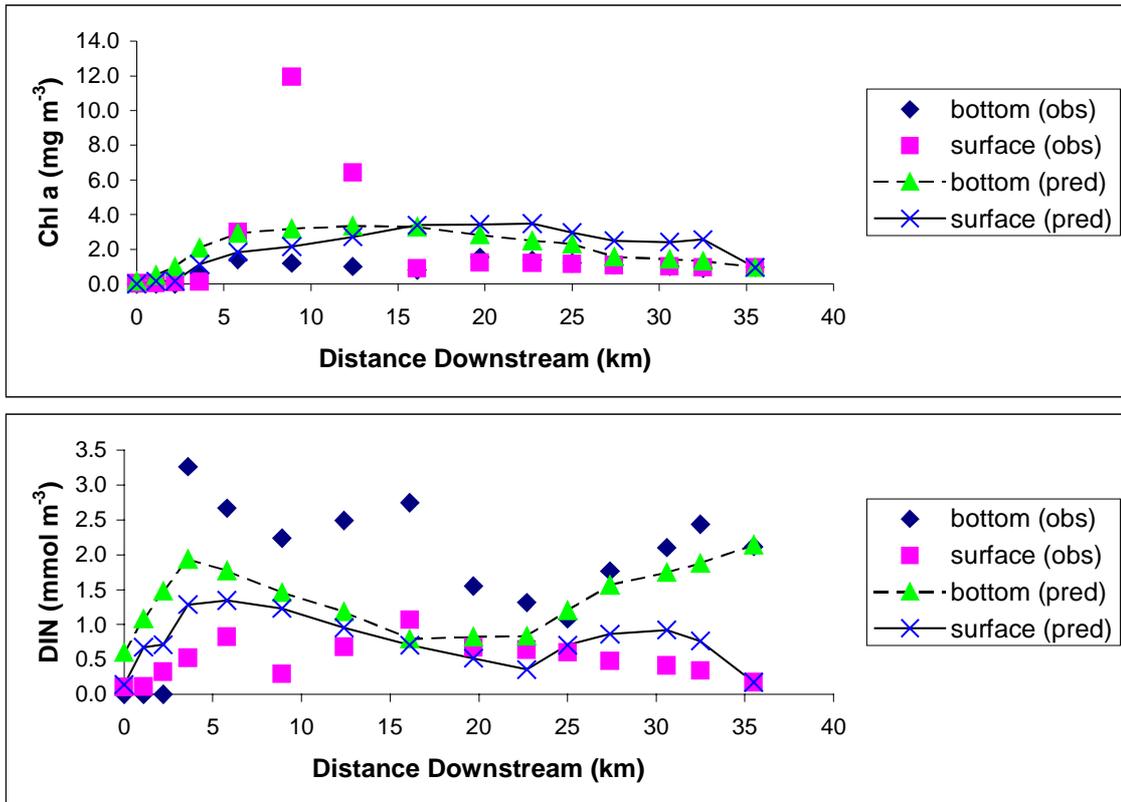


Fig. 10.5 Predicted and observed chlorophyll and DIN concentrations for HES 4 (Feb '97).

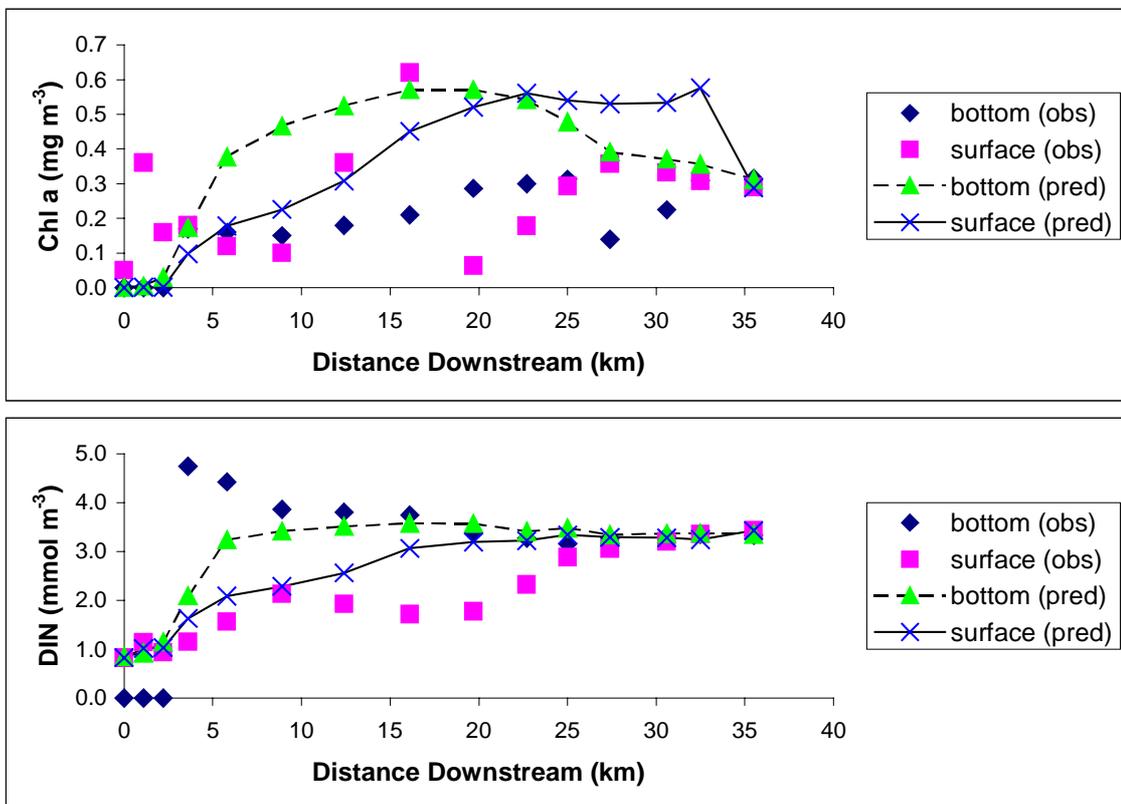


Fig. 10.6 Predicted and observed chlorophyll and DIN concentrations for HES 5 (Jun '97).

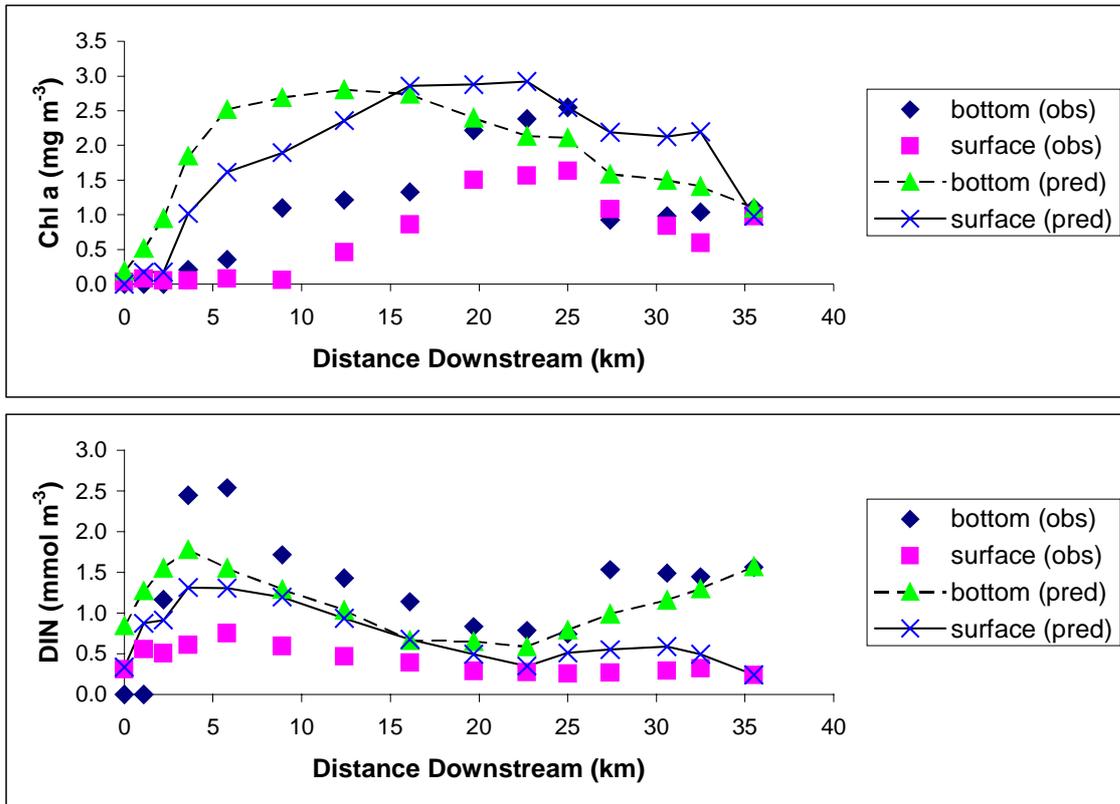


Fig. 10.7 Predicted and observed chlorophyll and DIN concentrations for HES 6 (Oct '97).

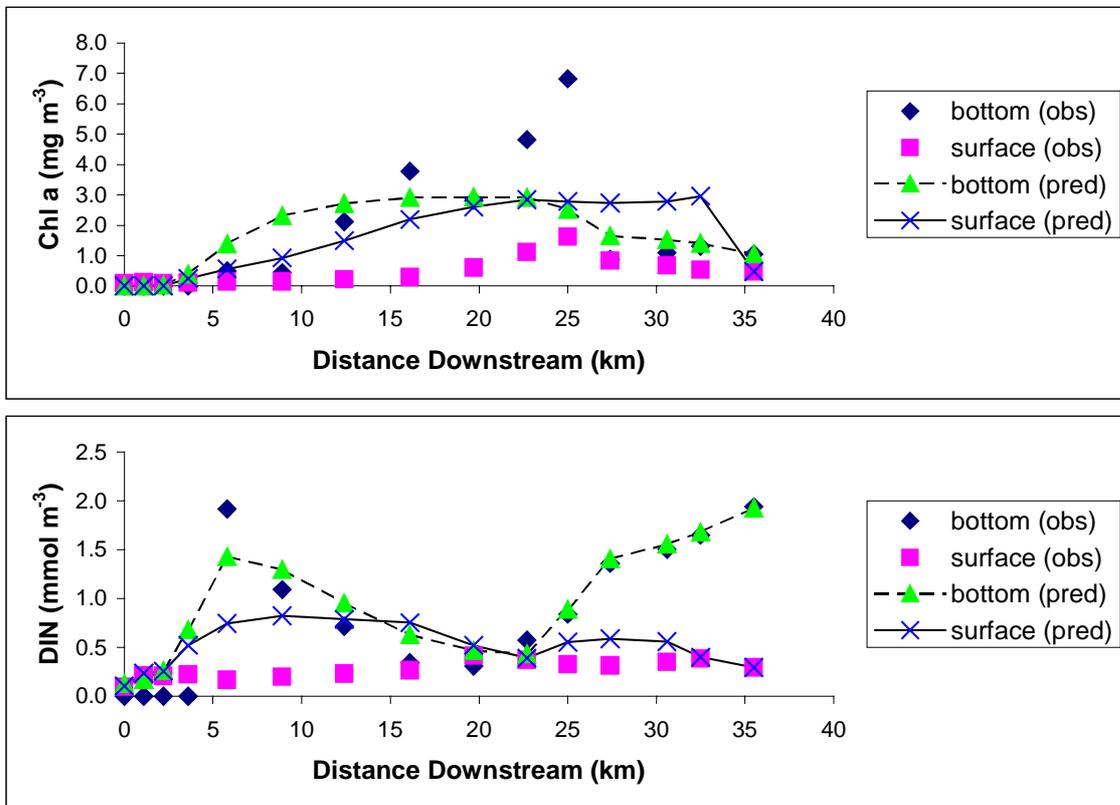


Fig. 10.8 Predicted and observed chlorophyll and DIN concentrations for HES 7 (Dec '97).

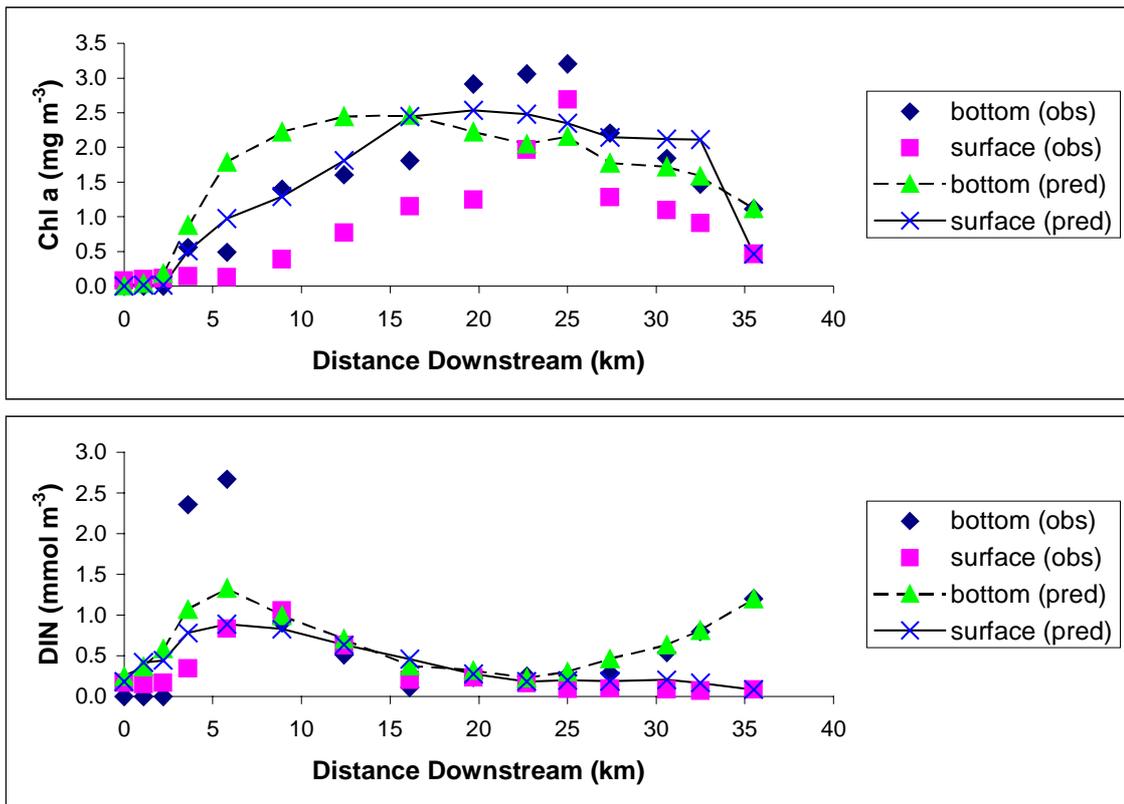


Fig. 10.9 Predicted and observed chlorophyll and DIN concentrations for HES 8 (Feb '98).

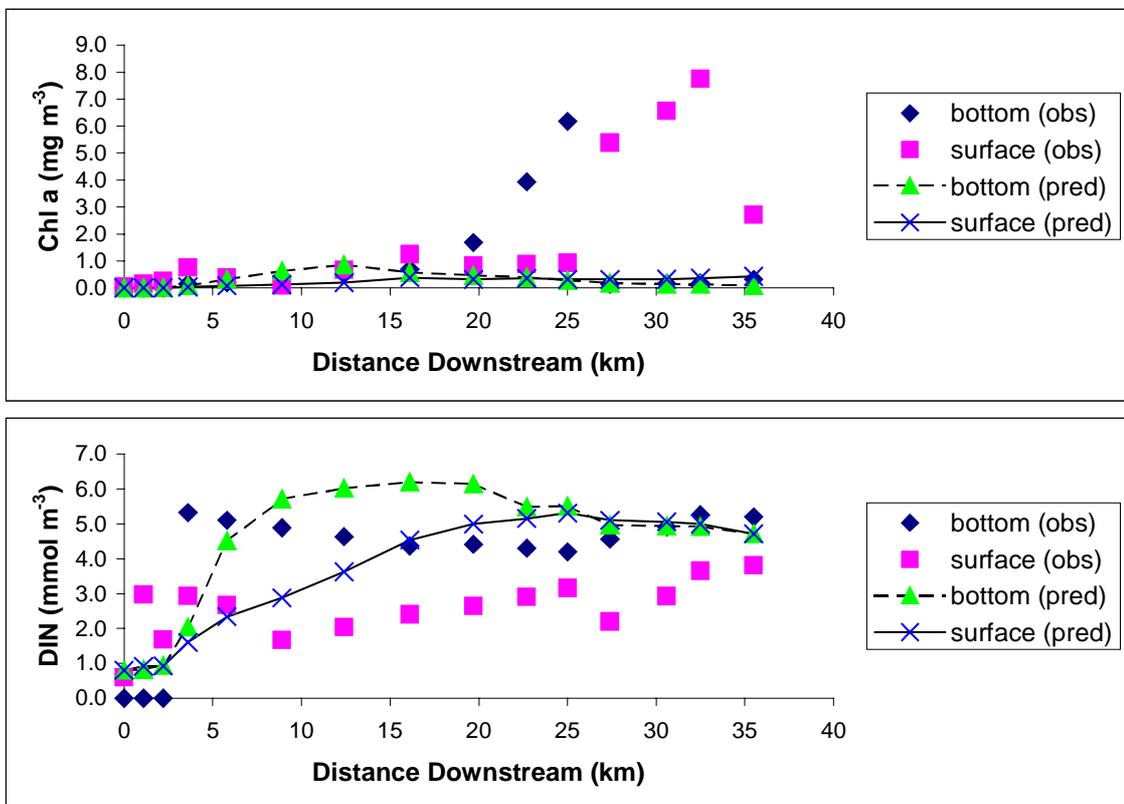


Fig. 10.10 Predicted and observed chlorophyll and DIN concentrations for HES 9 (May '98).

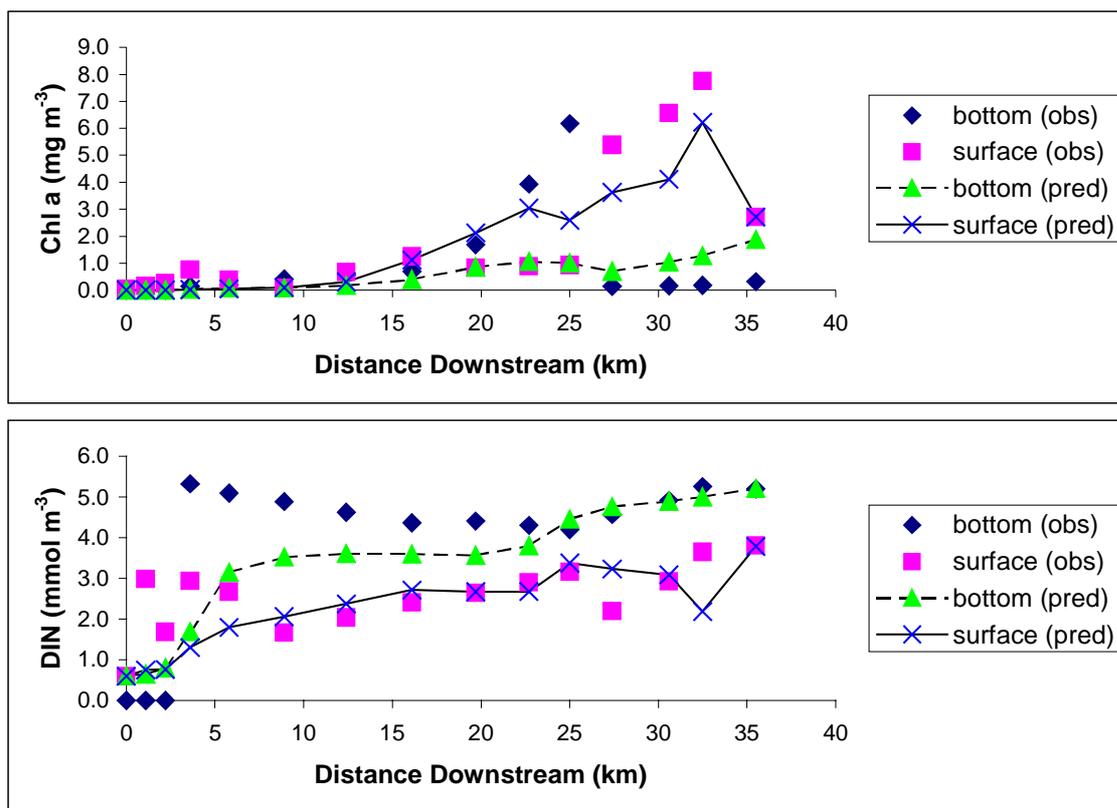


Fig. 10.11 Predicted and observed chlorophyll and DIN for HES 9 (May '98), with vertical migration included.

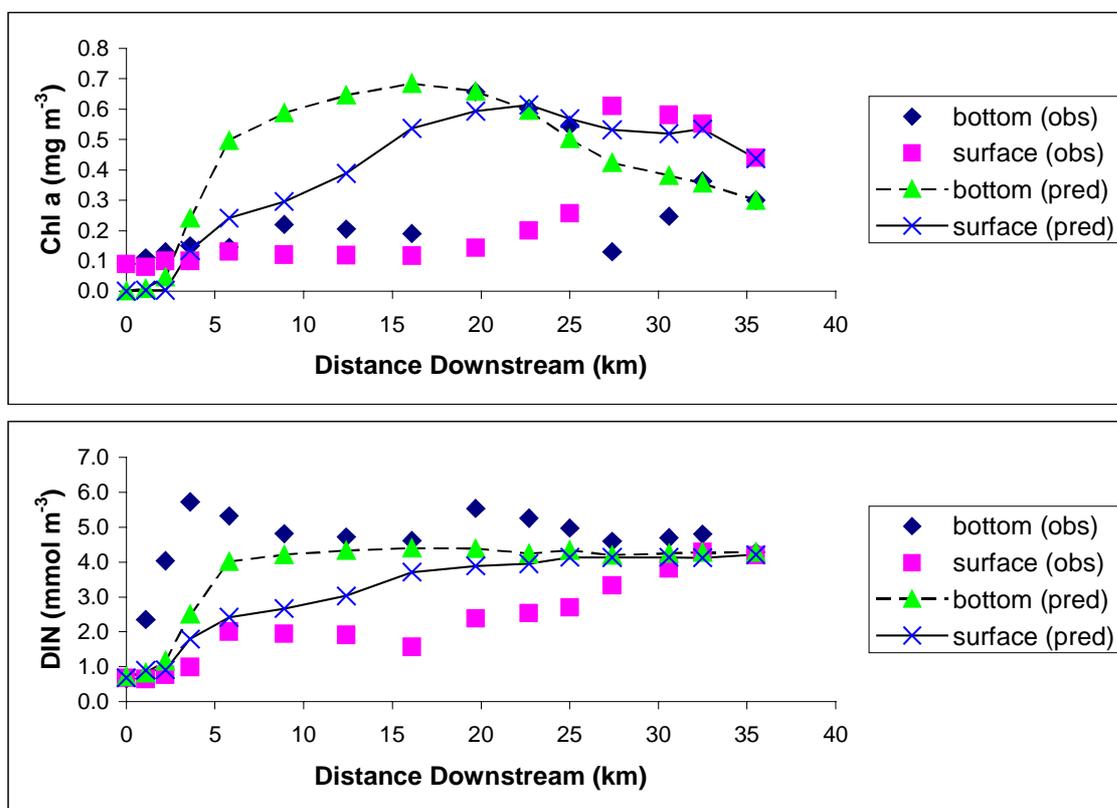


Fig. 10.12 Predicted and observed chlorophyll and DIN concentrations for HES 10 (Aug '98).

It is also possible that the changes in fitted maximum growth rate are compensating for other changes which are not captured by the model. Because of the high flushing rate, the phytoplankton observations strongly constrain the net phytoplankton growth rate (growth minus losses). The DIN observations help to constrain the pattern of variation in phytoplankton growth rate along estuary. However, phytoplankton loss rates in the estuary are not well characterized. Zooplankton grazing rates may vary substantially over time, and with species composition. As we discuss further below, the interaction of sinking or vertical migration with estuarine circulation may also substantially alter the effective phytoplankton loss rates.

Spring and autumn surveys.

The spring survey HES3 (Oct '96) has very high river flow, low to moderate biomass, and partial DIN drawdown in surface waters. Phytoplankton are light-limited but almost nutrient saturated in the lower and middle estuary. With a moderate maximum growth rate (1 d^{-1}) and sinking rate of 0.2 m d^{-1} , the model is able to reproduce the magnitude of the observed biomass in surface and lower layers (Fig. 10.4), although the vertical distribution is incorrect, with the predicted mid-estuary peak in the bottom rather than surface layer.

The model badly over-predicts DIN in the surface layer in the lower estuary. However, this appears to be a physical rather than biological problem: the vertical mixing is too high in the lower estuary, so that DIN inputs in the lower layer at the marine boundary are immediately mixed up into the surface layer in the next column.

The autumn survey HES 9 (May '98) has moderate flow ($82 \text{ m}^3 \text{ s}^{-1}$), high DIN in the bottom layer, partial drawdown of DIN in the surface layer, and very high biomass levels in the lower estuary, but low biomass in the upper estuary. The standard model is unable to reproduce this distribution of DIN and biomass. With a maximum growth rate of 1.5 d^{-1} , and zero sinking rate, the model predicts low biomass (around $1 \text{ mg Chl } a \text{ m}^{-3}$) in the surface layer and bottom layers (Fig. 10.10). With a sinking rate of 0.2 m d^{-1} , the model predicts a mid-estuary peak of $\sim 2 \text{ mg Chl } a \text{ m}^{-3}$ in the bottom layer of the middle estuary, but this is too far upstream, and still much lower than the peak observed biomass.

The model cannot maintain a high biomass of neutrally buoyant or sinking particles in surface waters in the lower estuary, as these waters are flushed much too rapidly. HES 9 (May '98) sampled the final stages of a large dinoflagellate bloom in the summer and autumn of 1997/98 (Chapter 5). Observations from automated profiling monitors show regular diel vertical migrations by dinoflagellates (Chapter 5). This has two potential benefits. The obvious physiological advantage is that dinoflagellates experience high DIN concentrations in bottom waters at night and high light intensities in the daytime. Vertical migration may also help to maintain horizontal position in the estuary in a two-layer flow.

We have attempted to represent vertical migration in the model by allowing phytoplankton to rise for 12 hours per day, and sink for 12 hours per day. However, the phytoplankton growth model does not resolve the diel cycle in light and photosynthesis, and is unable to reproduce dinoflagellate uptake of nutrients in the bottom layer. We have simulated the dinoflagellate ability to capture bottom nutrients by assigning a low saturation light intensity for growth.

With these changes in the model, phytoplankton are still unable to maintain high biomass in the lower estuary because a very low (observed) chlorophyll concentration was assigned as a

marine boundary condition in the bottom layer. The problem is that the observations are made in daylight, when dinoflagellates are near the surface. One would expect dinoflagellates to migrate to bottom waters at night, but the boundary concentrations are fixed in the model. We decided it would be more realistic to increase the phytoplankton biomass in the bottom layer at the marine boundary.

When this is done, the model produces a phytoplankton distribution that matches observations more closely (Fig. 10.11). Biomass is high in the surface just upstream of the boundary and quite low in the upper estuary. While the observed changes in vertical distribution upstream are not captured well, the results suggest that a more detailed model of vertical migration could explain the development and maintenance of dense dinoflagellate blooms.

10.5 Summary and Conclusions

10.5.1 Nitrogen cycling and phytoplankton growth in the Huon Estuary

The Huon Estuary is unusual among Australian estuaries in a number of respects. Physically, it is strongly stratified, with a two-layer overturning circulation driven by relatively high freshwater discharge year-round. NO_x and phosphate concentrations in deep water at the mouth of the estuary (in D'Entrecasteaux Channel) are also high year-round. This, combined with the 2-layer circulation, results in a large influx of NO_x and phosphate into the estuary in bottom waters in winter and summer.

The Derwent Estuary has a similar estuarine circulation, and experiences a large influx of marine nitrate in winter, while NO_x concentrations in bottom waters at its mouth are depleted in summer (Parslow 1999; Parslow et al. 1999). A principal difference between the Derwent and Huon estuaries is that the Derwent estuary is subject to nutrient loads from sewage treatment plants distributed along its length. Their combined DIN input is about 4 times the estimated finfish-farm DIN load into the Huon. In the Derwent, mean ammonia concentrations in surface waters in mid-estuary in summer are around 2 mmol m^{-3} and chlorophyll concentrations there are around 6 mg m^{-3} (Coughanowr, 1997; Parslow et al., 1999). These values are comparable to those predicted for the Huon at 4 times current finfish-farm loads (Fig. 10.14).

The persistence of high NO_x levels in bottom waters in D'Entrecasteaux Channel in summer presumably reflects the channel's proximity to shelf waters south of Tasmania, but is still somewhat surprising, given that past studies have shown nitrate depletion in surface mixed layers on the shelf in summer (Parslow et al, 1996). The results suggest that water from below the seasonal mixed layer on the shelf may find its way along the bottom of D'Entrecasteaux Channel to the mouth of the Huon Estuary in summer. This is consistent with the bathymetry, which shows the deep drowned river valley extending south along D'Entrecasteaux onto the shelf (Chapter 2). It is even possible that the two-layer estuarine circulation extends through southern D'Entrecasteaux Channel, although we have no direct observations to support this.

There is some doubt about the source of NO_x in bottom waters at the mouth of the estuary in summer. As discussed in Chapter 4, the high nitrite : nitrate ratios in bottom water in summer suggest that a substantial fraction of this nitrate may be due to recycling and nitrification in water column and sediments, in both the lower Huon Estuary and D'Entrecasteaux Channel. If

true, this has significant implications for the impact of finfish-farm loads on the estuary (see below).

The Huon Estuary is also somewhat unusual in draining a large catchment, much of which is undisturbed wilderness. Although part of the catchment is used intensively for horticulture and grazing, the budgets presented here show that loads of DIN and DIP from the catchment are much smaller than inputs from the marine end of the estuary. The catchment does provide a significant load of DON, but the observations are consistent with the hypothesis that this load is largely refractory, and passes through the estuary without participating in biogeochemical cycles to any significant extent.

The Huon River carries very high concentrations of CDOM, and as a result, light is strongly attenuated and phytoplankton growth is strongly light-limited, especially in the upper estuary. In winter, this is exacerbated by low surface-light intensities and moderate to high flushing rates. Observations and modelling suggest that the estuary is relatively inactive in a biogeochemical sense in winter, with large fluxes of marine DIN and DIP circulating through the estuary unutilised, due to a combination of high flushing rates, high light attenuation and low temperatures.

Surface irradiance increases in summer, and phytoplankton biomass accumulates sufficiently to draw down DIN in surface waters throughout the estuary, and in bottom waters in mid-estuary (Fig. 10.5, 10.7, 10.8, 10.9). Elevated DIN is maintained in bottom waters at the marine boundary in summer, and the upstream flux of DIN in bottom water at the mouth is still the dominant DIN load into the estuary (Fig. 10.1).

There is evidence from nitrogen and phosphorous budgets for large seasonal exchanges between the water column and sediments in the estuary. In spring and summer, the estuary acts as a net sink for TN and TP, presumably sequestering organic matter formed within the estuary, plus organic matter imported at the marine boundary, within sediments. In autumn and winter, the estuary acts as a net source of TN and TP, apparently exporting organic matter sequestered during the previous summer. As discussed above, and in Chapter 4, there is evidence that some part of the DIN in bottom waters at the mouth of the estuary in summer is due to recycling of nitrogen in bottom sediments at the mouth of the estuary and in D'Entrecasteaux Channel.

Over the study period, it appears that the estuary acted as a substantial net sink for TN (about 1250 t N p.a.) and TP (about 200 t P p.a.). If we assume the DON load from undisturbed catchments is largely refractory, then uptake and sequestration of marine nitrate within the estuary accounts for about two-thirds of the TN sink, with the remainder split roughly equally between agriculture and finfish farms. Marine PO_4 also accounts for most of the TP sink. The relative contributions of burial and denitrification to the TN sink are not known.

The model is generally able to reproduce the magnitude and broad pattern of chlorophyll and DIN distributions observed on the nine spatial surveys, with limited tuning of three parameters: the maximum phytoplankton growth rate, half-saturation constant for phytoplankton growth on DIN, and phytoplankton sinking rate. During high flow periods, the model has difficulty maintaining observed chlorophyll levels in the face of high flushing rates. Retention of phytoplankton within the estuary is increased considerably in the model if phytoplankton sink at low to moderate rates. However, in most cases sinking results in an unrealistic chlorophyll

distribution, with predicted maximum chlorophyll in the bottom layer instead of the surface layer, and too far upstream.

Observations suggest that the dinoflagellate *Gymnodinium catenatum*, which formed dense blooms in the estuary in 1997/98, makes extensive diel vertical migrations between surface and bottom waters. Modelling suggests that this has several important benefits for *G. catenatum*. It can take up DIN at high concentrations in bottom waters at night, and experience high light intensities in surface waters in the day-time. Perhaps equally importantly, by moving between surface and bottom layers, *G. catenatum* can avoid being rapidly flushed out of the estuary in surface waters.

A number of simplifications in the model may contribute to errors in predicted chlorophyll distribution. In the two-layer formulation, flow is uniformly out in the surface layer and in at depth. In the estuary, velocities will vary continuously with depth, and there may be a region around the pycnocline where average velocities are close to zero. At times, phytoplankton were observed to be concentrated in subsurface chlorophyll maxima at the pycnocline (Chapter 5). These phytoplankton may experience much lower flushing rates than predicted by the two-layer model.

The averaging of tracers across the estuary may also lead to artificially high flushing rates. Side embayments, especially Port Cygnet, may be subject to much weaker flushing rates than predicted by the model, allowing phytoplankton biomass to accumulate. Note that the inverse model, based on salinity, estimates exchanges that average over this spatial variation in a way that is appropriate for a neutrally buoyant, conservative tracer injected at the head of the estuary. The approximations inherent in this averaging process may not be appropriate for non-conservative tracers with sources or sinks along the estuary.

For almost all surveys, the model has difficulty in reproducing the high DIN concentrations observed in bottom waters near the head of the estuary. This appears to result from at least two weaknesses: the model underestimates sediment fluxes of DIN, particularly in winter, and the inverse model overestimates vertical mixing between surface and bottom layers at the head of the estuary. This mixing rate is not well constrained by the inverse model, as vertical salinity gradients there are weak. It is also possible that the bottom samples with high DIN concentrations were collected in depressions in the bed of the estuary, which exchange weakly with the main water column.

In summer, the predicted bottom DIN concentrations in the upper estuary in the model increase substantially when denitrification is switched off, resulting in better agreement with observations. This suggests that denitrification efficiency is low, at least in the upper estuary. However, given the highly simplified nature of the sediment model, and in the absence of corroborating *in situ* flux measurements, these results should be taken as suggestive only.

Because the Huon estuary is rapidly flushed and light-limited, it is not particularly sensitive to changes in denitrification efficiency. Switching off denitrification results in increases in peak chlorophyll and DIN in summer of about 20 to 50%, with almost negligible increases in winter. In contrast, in the Port Phillip Bay model, with a flushing time of about one year, switching off denitrification results in massive increases in DIN and chlorophyll, and a switch from mesotrophic to highly eutrophic status (Murray and Parslow 1999).

10.5.2 Effects of changes in finfish-farm loads

The inverse and prognostic analysis described here provide a basis for improved understanding and prediction of the capacity of estuaries such as the Huon to support aquaculture sustainably. The Huon Estuary is a major site of finfish-farm culture in Tasmania. Environmental regulations to date have dealt primarily with the local impact of farms on sediments within and adjacent to leases. However, it is the impact of finfish-farm nutrient loads on estuary water and sediment quality at the scale of the entire estuary that is likely to determine the sustainable carrying capacity.

The simple water-quality model described above allows us to examine the impacts of changes in finfish-farm nitrogen loads on DIN and chlorophyll levels. Current (1997) farm loads were estimated to be (on average) 240 mmol s⁻¹ DIN and 36 mmol s⁻¹ detrital N. We have rerun simulations for each spatial survey with farm loads set at 0, 2, 4 and 10 times these loads. (The relative spatial distribution of loads within the estuary was unchanged.) **These multipliers were chosen arbitrarily to explore the model estuary's response, and are not based on scenarios provided by industry or managers.**

Not surprisingly, the predicted effects of changes in farm loads depend on season. In winter, phytoplankton growth is already nutrient-saturated, so increases in finfish-farm loads produce increases in DIN, but not in chlorophyll (Fig. 10.13). Moreover, DIN concentrations are already high, and flushing rates are moderate to high, so farm loads have a smaller relative effect on DIN. Removing farm loads typically decreases bottom DIN levels by less than 10% in winter. Farm loads increase bottom DIN concentrations by 40% at four times current loads, and double DIN concentrations at ten times current loads

Phytoplankton growth and biomass are currently N-limited in summer, so it is not surprising that farm loads have a greater impact then. Current farm loads are still small compared with DIN inputs at the marine boundary in summer: removing farm loads decreases chlorophyll and DIN by about 25% from around 3 mg Chl *a* m⁻³ and 0.7 mmol m⁻³ respectively (Fig. 10.14). Similarly, doubling farm loads increases chlorophyll by about 25 to 30%, and increases DIN by about 50% to around 1 to 1.5 mmol N m⁻³. At four times current loads, DIN doubles again to around 2 mmol N m⁻³, and chlorophyll increases to values around 5 mg Chl *a* m⁻³, almost twice current levels. At these levels, DIN is only a weak constraint on phytoplankton growth. At ten times current loads, DIN increases to values up to 6 mmol m⁻³, sufficient to saturate phytoplankton growth. Predicted phytoplankton biomass reaches a maximum of about 8 mg Chl *a* m⁻³, due to a combination of self-shading and flushing (Fig. 10.14).

According to the model, doubling 1997 finfish-farm loads would carry some risk of increased frequency or density of summer blooms, while quadrupling loads would put the system on the brink of N saturation, and would substantially increase the risk of prolonged blooms. Increasing loads by a factor of 10 would completely change the nature of the system, producing elevated DIN and large blooms throughout summer.

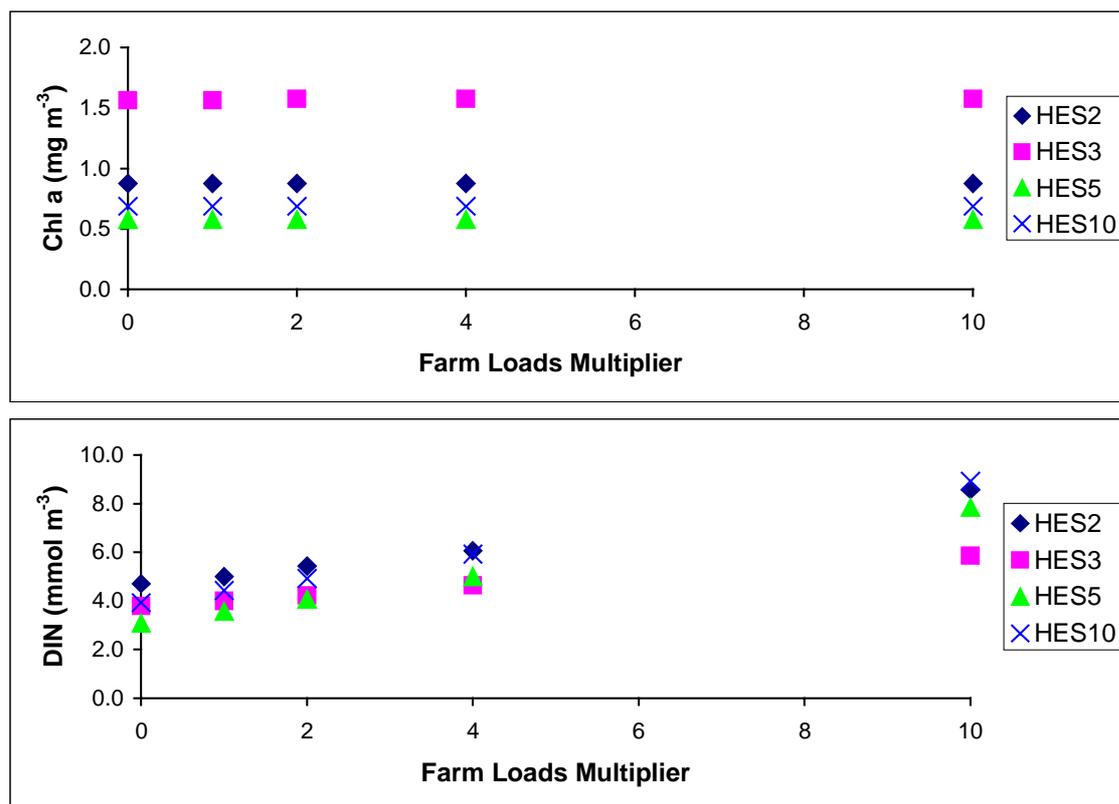


Fig. 10.13 Peak chlorophyll, and DIN in mid-estuary bottom layer, vs farm N load multiplier, for winter/spring surveys HES2 (Jul '96), HES3 (Oct '96), HES5 (Jun '97), HES10 (Aug '98).

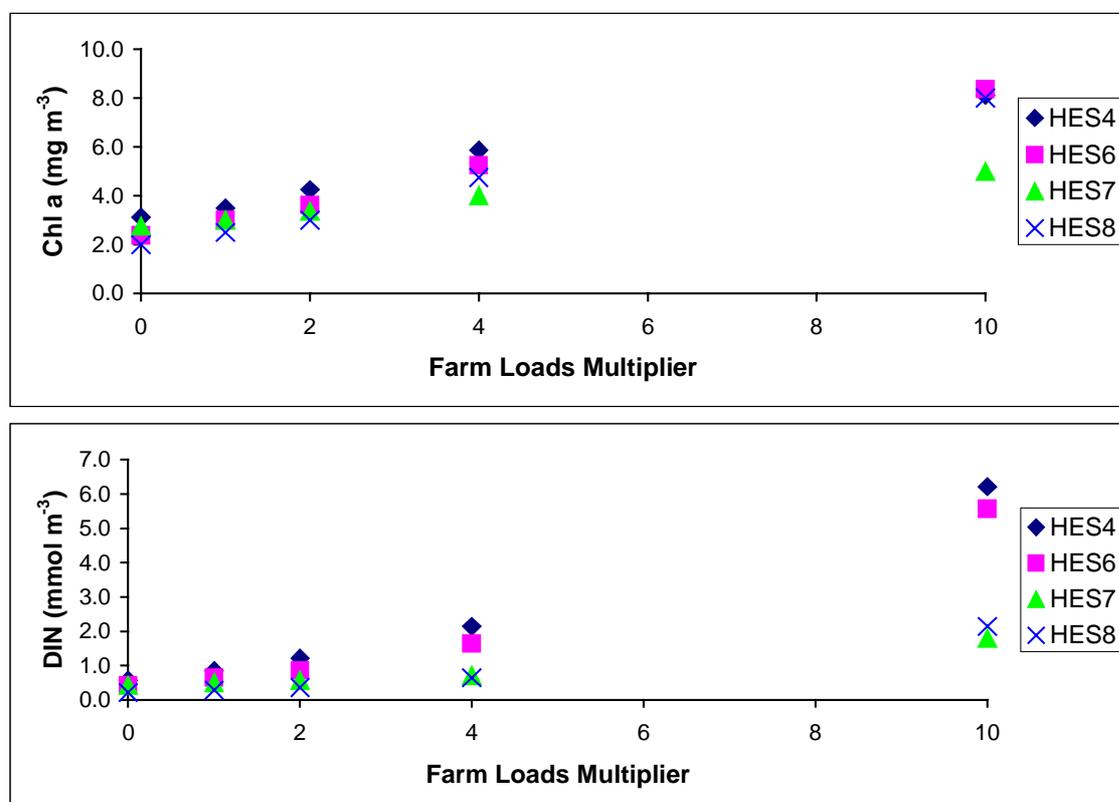


Fig. 10.14 Peak chlorophyll, and DIN in mid-estuary bottom layer, vs farm N load multiplier, for summer surveys HES4 (Feb '97), HES6 (Oct '97), HES7 (Dec '97), HES8 (Feb '98).

There are a number of sources of uncertainty in these model predictions, which would generally lead to underestimating the impact of increased finfish-farm loads. Because the model tends to flush phytoplankton too effectively, the biomass predicted by the model under high loads is probably underestimated. Perhaps more importantly, the scenarios have been run with a fixed marine boundary concentration of DIN. If DIN in bottom waters at the mouth in summer is produced primarily or substantially by regeneration and nitrification of organic matter produced in the estuary, then present finfish-farm loads may be indirectly responsible for some fraction of the current marine loads. These bottom DIN concentrations and loads would then be expected to rise with increased farm loads. It is difficult to quantify this effect without extending the model and observations to include D'Entrecasteaux Channel.

Increases in farm loads could also have effects on phytoplankton composition that are not addressed by the model. The model treats ammonia and nitrate as one pool, DIN. Currently, nitrogen loads in the Huon estuary are dominated by the marine input of nitrate. Finfish farms are likely to provide DIN as NH_4 . Changes in $\text{NO}_3:\text{NH}_4$ ratios might be expected to shift phytoplankton composition. Finfish farms also provide significant inputs of simple organics such as urea and amino acids, which may also favour particular species (Arzul et al. 1999).

As noted in 10.4.3, we have little data or knowledge on zooplankton grazing, and changes in assumed mortality rates would affect model predictions under current N-limited summer conditions. However, it is generally rare for grazers to prevent blooms under nutrient-replete conditions, so that scenarios involving high, saturating concentrations of DIN are likely to result in harmful blooms, regardless of grazer responses.

It should be noted that, because of the two-layer circulation in the estuary, the effects of nutrient loads from finfish farms will depend on their distribution between surface and bottom layers. To the extent that excretion of nutrients occurs in the surface layer, the impacts estimated here will be reduced.

10.5.3 Further development of the model

There are a number of obvious avenues for further development and improvement of the model. The inverse physical model provides only a crude horizontal and vertical resolution of tracer distribution and transport. CSIRO has a hydrodynamic modelling package (MECO) that has been successfully implemented to provide a highly resolved 3-D representation of transport in the Derwent Estuary and in Port Phillip Bay (Walker 1997). If implemented in the Huon Estuary, it would better represent phytoplankton and nutrient distribution and allow questions of flushing in local bays and Port Cygnet to be addressed. Given the concerns we have raised about interactions between nutrient cycling in the Huon Estuary and D'Entrecasteaux Channel, it would be highly desirable to include D'Entrecasteaux Channel in such a model.

The SEEM model provides only a crude representation of sediment cycling. Given the apparent importance of sediment fluxes in both the upper and lower estuary, a more sophisticated treatment of sediments, allowing seasonal and interannual accumulation of organic matter, is desirable. Again, models of this kind are available (Murray and Parslow 1999).

There is an opportunity to build on the knowledge and understanding acquired through this study, through developing and implementing a sophisticated estuarine management support

system for the Huon Estuary and D'Entrecasteaux. This system would assimilate data acquired through a cost-effective monitoring system into a 3-D physical / water-quality model, and provide both operational and strategic management support for the aquaculture industry and environmental managers. This opportunity is discussed further in the next chapter.

10.6 References

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11 KEY FINDINGS AND RECOMMENDATIONS

11.1 Introduction

The main conclusions from this project are considered under two broad headings.

The first is the key findings, where we describe the condition and the functioning of the Huon Estuary. New understanding concerning the water column and sediments is presented: hydrodynamics (layering, mixing and currents in the estuarine waters), current environmental quality, the microalgal community, and other attributes that constitute the natural ecosystem. We also consider features relevant to human use of the estuary, including potential threats to the wellbeing of the natural system. The waterway is a dynamic entity; so we look at processes (physical, chemical and biological), both as individual elements and as part of an integrated system. After this section, we make an appraisal of the core components for effective environmental monitoring of an estuary like the Huon.

The second major heading is pointers for management of the Huon Estuary and its catchment. We outline actions, or changes in practice, that we think should be considered for the environmental benefit of the estuary and its future prudent use.

11.2 Key Findings — Estuarine Condition and Functioning

11.2.1. Run-off and flushing

The Huon Estuary at 40 km in length and about 1.4 km³ in volume is, in global terms, a small estuarine system. Unlike most (mainland) Australian estuaries, it has a regular and strong riverine input throughout the year. This is assured by its western catchment, which falls within the high-rainfall zone of southwestern Tasmania.

The Huon Estuary is also a strongly stratified waterway — with a shallow, fresher layer flowing seaward over a deeper, marine bottom layer that has a more sluggish landward flow. The upper fluvial region of the estuary is so strongly vertically demarcated into two layers that it is classified as a salt wedge. After Port Huon and around the elbow at Brabazon Point, the estuary deepens to take on the characteristics of a fjord-like estuary.

The Huon Estuary has many physical features in common with the neighbouring Derwent Estuary, including a common origin as drowned river valleys, their size, and the manner in which they function. Chapter 3 explored their common physical attributes. Agencies responsible for these waterways can reasonably draw on knowledge gained in studies of one or the other system for the benefit of both.

How does the estuary work?

In a nutshell, the Huon is a fast-flushing estuary with small tides and a very small tidal excursion. Flushing is dominated by a two-layer overturning circulation driven by river flow. Residence times vary with run-off, but range from hours to days for the surface layer, and from days up to one week for the whole water column. Material entering the estuary has a short interval to be processed before it is disgorged to coastal waters. Microalgae, too, have limited opportunity to grow in the estuary before they are swept out of its confines.

Since the tides and tidal currents are small, the estuary does not have the same extent of active sediment resuspension found in estuaries with larger tidal ranges. The suspended particulate matter coming in from the waters of the Huon River is also at very low concentrations. Although the estuary has low turbidity, light penetration is instead severely diminished by high levels of coloured dissolved organic matter (CDOM — commonly termed humic substances) in the river water.

The two-layer overturning circulation in the estuary has the surface outflow entraining deeper marine water, with the result that the surface-layer salinity gradually increases seaward. A compensating landward flow brings in subsurface marine waters. This estuarine circulation at times extends out into the southern part of D'Entrecasteaux Channel. The canyon through this part of the channel could be the conduit for replenishing bottom waters directly from coastal waters off southeast Tasmania.

During our surveys, run-off events produced transient reductions in surface salinity, which typically lasted only a few days, consistent with high flushing rates. The freshwater plume tends to hug the eastern shores, and enter Port Cygnet, its route determined by Earth's rotation ('Coriolis' force) and prevailing westerly winds. However, winds may at times force the plume cross-estuary, weakening the observed correlation between surface salinity on the western shores and run-off. High flows may actually heighten the contrast west to east in salinity. Outcropping of high salinity water ($S > 30$) in the small bays and inlets of the west and southwest shores from Crowthers Bay to Surges Bay was commonly observed. A more detailed understanding of the three-dimensional (3-D) circulation in response to run-off, wind, tide and bathymetry would require the development of a high-resolution 3-D hydrodynamical model. An improved understanding of 3-D circulation would not only help to interpret field observations, but could also have important practical implications for exchange of water between the main body of the estuary and side embayments such as Hospital Bay and Port Cygnet. A 3-D hydrodynamical model should also render better than the simpler models applied so far the interaction between the vertical distribution of phytoplankton (often concentrated in thin layers) and the vertical current shear associated with the 3-D circulation.

The effects of surface heating and cooling are exaggerated in the shallow freshwater plume, which is much warmer than bottom waters in summer, and cooler than bottom waters in winter. These effects are intensified during higher river flows, with melt-waters entering the estuary in winter and early spring. In summer, the high concentrations of dark humic material in the freshwater plume aid heat absorption.

Automated profiling systems installed at Killala Bay and Hideaway Bay also showed rapid fluctuations in vertical structure and stratification associated with run-off events and wind mixing (see Chapter 5).

11.2.2. Environmental quality of the Huon Estuary

The current condition of the Huon Estuary is a melding of its geomorphology, rainfall patterns, the nature of its catchment (soils, vegetation, etc.), and past and current human activities in the region. Based on these features and the few previous studies of the estuary, we made an initial evaluation that the estuary is a *substantially natural* waterway. “Substantially natural” is the term favoured by Environment Australia to describe ecosystems that are intermediate between *high conservation / ecological value* systems and *highly modified* ecosystems. As these classifications are also used in the draft of the revised ‘Australian Water Quality Guidelines for Fresh and Marine Waters’ (ANZECC / ARMCANZ 1998), we evaluated the Huon Estuary under the guidelines for a substantially natural water body. The main results and implications of this evaluation are discussed below.

Does water quality measure up?

Since there is no reference set of environmental information for the Huon Estuary (a cool temperate system in southeastern Australia), we used the ‘interim trigger levels’ from the draft ANZECC / ARMCANZ (1998) guidelines.

In general, the water quality of the Huon Estuary is good. That of its two principal source waters — the Huon River and coastal waters in D’Entrecasteaux Channel — is very high indeed. The Huon River at Judbury is on a par with waters in the Old River (Bathurst Harbour tributary in the wilderness area of the South West National Park) for a range of water quality measures.

In the estuary, several water-quality parameters exceed the interim trigger levels, but these are for valid natural reasons. Naturally high nutrient levels in winter are associated with inputs of marine nitrate and phosphate from continental shelf waters. These marine nutrients also drive phytoplankton blooms in spring, summer and autumn. The low interim trigger levels in the guidelines reflect the fact that the marine boundary for most Australian estuaries is oligotrophic, sub-tropical ocean water. Higher trigger levels are appropriate for Tasmanian estuaries (such as the Huon), which are connected to mid-latitude oceans with a temperate seasonal cycle in nutrients and phytoplankton.

Setting these aside, we identified three issues of concern. The first was dissolved oxygen. Throughout most of the estuary, the waters range from marginally below the trigger level of 90% saturation to close to saturation. However, there were exceptions. What seems to be natural depletion of dissolved oxygen (DO) is evident in the warmer months of the year around Egg Islands, in the upper estuary. DO depletion was possibly a phenomenon of many stratified Tasmanian estuaries before European settlement; the consumption of oxygen is naturally high in the deposition zone of such estuaries with a high input of terrigenous organic matter. An instance of oxygen-depleted conditions has been observed in the isolated and pristine Bathurst Harbour (Edgar and Cresswell 1991).

Intermittent depletion of DO is also observed in the lower Huon estuary, but it is difficult to be categorical about its causes. The very lowest values were recorded near fish farms during weekly / fortnightly monitoring. Unfortunately, we cannot report more fully on either the extent or frequency of these episodes, because the DO sensors failed on the two continuous profilers. In summer, when the water column is strongly stratified, and with the sediments always

organic-rich, the bottom waters of the Huon Estuary are vulnerable to oxygen depletion. Increased loading of organic matter with high biochemical oxygen demand can trigger such events.

The second matter for attention is the influence of land-based activities on the environmental quality of the estuary. Fortunately, they appear to be causing only localised effects at this time. The condition of the lower Kermadie River is one example. Its degradation is seemingly the combined result of STP effluent plus land-use practices in the catchment around Geeveston. The symptoms include high levels of suspended solids, higher concentrations of some forms of nutrients (nitrogen, phosphorus and silicon), and probably some metals (copper, nickel, cobalt and zinc) as well.

Elsewhere in the estuary, sporadic instances of nutrients exceeding guideline trigger levels appear to be traceable to human activities (STP effluents, aquaculture, and diffuse sources, such as septic tank drainage and agricultural run-off). We noted very occasional signs of these inputs in localised higher concentrations of some chemical forms of nitrogen (ammonia, total N) and phosphorus (phosphate and total P). The full extent of these inputs might not be discerned because of the opportunistic nature of microalgae in promptly assimilating excess nutrients.

The brief survey of contaminants (trace metals and pesticides) in survey HES 10A produced reassuring results in that no serious issues for water quality were evident. Nevertheless, this survey was limited in coverage — with respect to both the range of contaminants, and the regions of the estuary and catchment, surveyed. More sensitive and targeted monitoring is needed for the current crop of agricultural chemicals used in the Huon catchment.

A third issue of concern is the appearance of raised levels of ammonia and nitrite in bottom waters in the lower estuary after large microalgal blooms. These levels are still only modest (about 1 μM), but they suggest that there is a significant flux of dissolved inorganic nitrogen from sediments into bottom waters. Recent studies of coastal systems (e.g. Harris et al. 1996) have revealed a positive feedback loop, in which increasing nutrient loads and associated organic matter production lead to oxygen stress in sediments and bottom waters, which increases the efficiency with which nutrients are recycled and released from bottom sediments, thereby magnifying the impact of external loads. The Huon estuary is less vulnerable to these processes because of its relatively short flushing time. However, the observations of sporadic oxygen depletion in bottom waters, together with elevated bottom ammonia levels, suggest we should be cautious about imposing large increases in nutrient loads to the estuary (see further discussion below and in Chapter 10).

Harmful algal blooms, such as those of toxin-producing *Gymnodinium catenatum*, are an issue for water quality; we discuss them separately in the next subsection.

What do the sediments record?

Sediments in the Huon Estuary have high contents of organic matter. In the upper estuary most of this derives from land plants, but in the middle and lower estuaries there is a strong contribution from *in situ* phytoplankton production. Finfish farms and STP effluents were found to be relatively minor contributors of organic matter to the estuary as a whole, although some localised inputs were recognisable.

Apart from small regions of sand, muds (i.e. sediment particles < 63 µm grain size) comprised almost the entire estuary floor in its middle and lower reaches. The greater the proportion of muds, the higher was the organic matter content in the sediments.

Stable carbon and nitrogen isotope signatures and lipid biomarker data were shown to be good indicators of organic matter sources in the estuary. In common with other studies we found that loss-on-ignition (LOI) yielded unreliable estimates of organic matter content in muddy sediments.

Biomarker (sitosterol) and stable isotope data confirm that most of the organic matter in sediments from Hospital Bay and into the middle and upper estuary is of terrestrial plant origin. Studies by Chesterman (1995) and others (see Chapter 9) have indicated abundant wood fibre in these areas, which would appear to be a legacy of the sawmilling and operation of the pulp mill at Whale Point. In the neighbouring Derwent Estuary, eucalypt wood-fibre sludges contain appreciable levels of contaminants. The sludge rafts on top of the sediments in that estuary have absorbed contaminants from heavily polluted waters for half a century. This may not be such an issue for the Huon Estuary because, unlike the Derwent, it has not been exposed to high loads of industrial contaminants. However, the higher concentrations of DDT and DDD in Hospital Bay could be preferentially retained in such sediments. A broader survey of the wood-fibre-contaminated sediments, and their composition, is advisable.

No other pesticides were detected in the sediment samples collected in survey HES 10A (Sep '98). Nonetheless, as mentioned in the water quality section above, it is important that this explorative work be followed up with more sensitive measurements to cover the pesticides currently used in horticulture, other agriculture activities and forestry.

The concentrations of a variety of trace metals in Huon Estuary sediments were below any of the interim sediment quality guidelines advocated in the draft ANZECC / ARMCANZ guidelines (1998), and therefore, require no further management consideration. In the sediments of Hospital Bay and Port Cygnet, we could see a slight imprint of human inputs in the raised levels of some trace metals (copper, cadmium, lead, and possibly zinc).

11.2.3. Algal blooms

In the Huon Estuary, diatom and dinoflagellate blooms alternate from spring to autumn in most years. A third group, small flagellates (a mixture of haptophytes and prasinophytes, with some cyanobacteria), is present throughout the year at relatively constant biomass, and tends to dominate in late autumn and winter when diatoms and dinoflagellates are low or absent. Blooms of the toxin-producing dinoflagellate *Gymnodinium catenatum* (PSP type) are responsible for closure of shellfish sites in the Huon Estuary and neighbouring waters.

We found the wintertime biomass of microalgae was uniformly low (~ 0.3 mg Chl *a* m⁻³) and dispersed throughout the water column of the estuary. Overall, median chlorophyll levels in the lower and middle estuary were about 1 mg Chl *a* m⁻³, and chlorophyll exceeded 4 mg Chl *a* m⁻³ 10% of the time. These moderate biomass levels are consistent with the moderate nutrient levels observed, and with a classification of the estuary overall as mesotrophic.

In both years of our field program (1996/97 and 1997/98), diatom blooms were observed in spring and late summer. They produced a subsurface chlorophyll maximum, but reached only

moderate biomass (1–2 mg Chl *a* m⁻³). The striking difference between the two years was the dense blooms of the dinoflagellates *Gymnodinium catenatum* and, to a lesser extent, *Ceratium* spp., which developed early in the summer and autumn of 1997/98, but not in 1996/97. These blooms reached very high densities, typically 20 mg Chl *a* m⁻³, but rarely greater than 50 mg Chl *a* m⁻³.

The dinoflagellate *G. catenatum* is an actively swimming organism. In microalgal blooms dominated by *G. catenatum*, we observed regular diel vertical migration — toward the surface during daylight, and towards the bottom by night. The automated profiling instruments tracked these movements as a daily oscillation of the subsurface chlorophyll maximum.

Phytoplankton community composition, as indicated by both cell counts and HPLC marker pigments, was relatively uniform throughout the middle and lower estuaries, consistent with rapid physical exchange in the deeper marine end of the estuary. Phytoplankton communities in the brackish surface layer in the upper estuary were quite distinct from those in the middle and lower estuaries. In this shallow fluvial zone, the surface layer is rapidly flushed and so phytoplankton biomass here is generally low.

Port Cygnet, although a side-arm of the Huon Estuary, sometimes needs to be considered as a water body distinct from the main estuary, but the results from investigations of nutrient biogeochemistry and microalgal dynamics do not provide a uniform view of the differences between the two (see Chapters 4 and 5). This study did not have the resources to treat Port Cygnet as a separate entity in the modelling, so we do not have detailed knowledge of its hydrodynamics. It appears to be a marine-dominated system with occasional small patches of fresher surface water, thought to originate in the fresher surface plume that flows down the eastern shore of the estuary. It is usually the site of early depletion of nutrients, and possibly where they are most depleted. However, it shows no signs of having a different microalgal composition, and rarely does it have the peak chlorophyll *a* concentrations. This disparity between chemical and biological observations might arise because physical conditions make Port Cygnet a separate microenvironment. It might also be that, although the microalgae are the same as elsewhere, they function differently in a small water body where only shellfish are cultured. Filtration by shellfish could account for Port Cygnet not having the peak microalgal biomass as measured by chlorophyll *a*.

Which estuarine conditions govern microalgal blooms?

Some environmental factors, such as light and nutrients, are a universal requirement for growth of microalgae. Other circumstances favour individual species or a group of phytoplankton over other microalgae. In the Huon Estuary, we observed a close link between physical conditions (e.g. light, temperature, stratification of the water column and winds) and populations of microalgae.

Light penetration in the estuarine waters is suppressed by the high concentrations of coloured dissolved organic matter in Huon River outflow. In winter, light limitation, cold temperatures and increased estuarine flushing restrict both the growth and biomass of microalgae. Microalgae in bottom waters are likely to be light-limited all year round.

With the arrival of spring, diatoms prosper because they can grow rapidly when light and nutrients are sufficient. Their growth is linked with the initial decline in nutrient concentrations

from their wintertime levels. By the end of spring, the microalgae have often depleted available nitrogen in surface waters, and also diminished levels in bottom waters in the middle estuary. In summer and autumn, when the water column is usually more stable and stratified, dinoflagellates have an advantage because they can migrate vertically. This aptitude allows dinoflagellates not only to sequester nutrients (particularly ammonia) in bottom waters, but also enables cells to optimise their position with respect to light and other conditions, and to avoid being flushed out of the system in the fast-flowing surface waters.

Dinoflagellate blooms in the summer and autumn of 1997/98 reach biomass levels an order of magnitude higher than any observed diatom blooms. The peak chlorophyll concentrations reached in dinoflagellate blooms imply a nitrogen concentration in phytoplankton that is more than 5 times the peak observed for inorganic nitrogen concentrations in the water column. This implies a mechanism for either accumulating biomass or nitrogen in the estuary, or both together. One possible mechanism is accumulation of organic nitrogen in bottom sediments, and its release during summer and autumn. Another is the interaction of dinoflagellate vertical migration with the two-layer estuarine circulation, which allows cells to accumulate in the estuary, and scavenge nitrogen from water circulating through the estuary. Model results (Chapter 10) suggest the latter is a key mechanism for supporting the peak biomass levels observed in 1997/98.

Can blooms of the toxic dinoflagellate be predicted?

In some years, as in 1996/97, dinoflagellate numbers in the Huon Estuary are low, with no blooms developing. Predicting when *G. catenatum* will bloom is still not possible. For although we know what conditions favour growth, we do not know what prevents blooms forming in some apparently suitable years.

We have identified the following conditions as supporting blooms of the dinoflagellate: temperatures above 12°C; a stable water column; adequate nutrient supply, including bottom-water ammonia (recycled nitrogen), and a complexity of factors associated with river run-off (Doblin et al. 1999a). These conditions generally corroborate requirements for bloom development proposed by Hallegraeff et al. (1995).

For several months before March 1997, no vegetative cells of *G. catenatum* were observed in water-column samples, but it is likely that they were present as resting cysts in the sediments. The population was probably restored by cyst germination to initiate the large blooms later that summer and autumn of 1997/98. Our data do not suggest a *mass* germination event as a trigger for any of the blooms observed during the study period. However, it is conceivable that the extent of germination was not adequately resolved by our sampling procedures. For studies of the role of cysts, it would be advisable to collect water samples (or deploy traps) at the sediment-water interface, or just above it, rather than a metre or two off the bottom as in our field program.

Quantitative data on phytoplankton grazers were not collected in this study. However, microscopic observations have revealed several zooplanktonic grazers of *G. catenatum*. The dinoflagellate might also be attacked by other, smaller life forms. At least six marine bacteria, endemic to the Huon Estuary, appear to be able to kill this microalga (Lovejoy et al. 1998, J. Skerratt — *pers. comm.*, December 1999). One might expect the loss rates of most phytoplankton in the Huon to be dominated by the high physical flushing rates rather than

grazing (zooplankton and algicidal bacteria). However, if *G. catenatum* avoids flushing through vertical migration, grazing losses could assume greater importance in principle for this species. The relatively low maximum growth rate, and the accumulation and persistence of a high biomass of *G. catenatum* for extended periods, suggest that actual grazing loss rates were low in 1997/98. While we cannot rule out differences in grazing communities between years, it seems unlikely that grazing could account for the complete absence of *G. catenatum* before March 1997.

It should be possible to make a short-term risk assessment of the probability and extent of a *G. catenatum* bloom in a particular season, using current knowledge. However, to predict when the toxic dinoflagellate will bloom, for how long, and with what intensity, needs more knowledge of the organism's biology and its interaction with the estuarine environment and ecosystem (including the sediment-water interface).

11.2.4. Nutrient and carbon cycling

The key environmental issues in the Huon Estuary are associated with the effects and fate of nutrient and organic matter loads from the catchment, from the marine boundary, and from activities in the estuary, especially salmon farming. In particular, we are concerned about phytoplankton blooms, and signs of oxygen depletion in bottom waters. Previous experience in other coastal systems has shown that, to deal with these issues, one must treat the estuary as an integrated system: a system in which nutrients and carbon are influenced by both physical transport and mixing, and also by biogeochemical processes in both the water column and sediments. Additional complexity is introduced by the sources and sinks for nutrients and carbon varying with respect to both time and location. The Huon Estuary, with its strong two-layer physical circulation and short mixing times, must be treated as an integrated system if we are to understand the distribution and behaviour of nutrients and carbon.

In this study, we combined inverse techniques and budgets, and simple process models, to understand nutrient cycling in the Huon Estuary. It appears that nitrogen is the limiting macronutrient in the middle and lower estuary. Both annual and seasonal nitrogen budgets suggest that inputs of available nitrogen into the estuary are dominated by the influx of dissolved inorganic nitrogen (DIN — nitrate, nitrite and ammonia) in bottom waters at the marine boundary, driven by the two-layer estuarine circulation. If our hypothesis is correct, the estuarine circulation might extend also into the southern D'Entrecasteaux Channel, along the canyon that was the ancient river valley. It then provides the mechanism for supplying nutrient-rich, subsurface coastal waters to the estuary. In winter, much of this nitrogen influx passes through the estuary unutilised. In spring, summer and autumn, phytoplankton convert DIN into organic matter, at times reaching bloom proportions.

The upper (relatively pristine) catchment is a very small source of DIN, but delivers a large load of dissolved organic nitrogen (DON). There is also a large inflow of DON in bottom waters at the marine boundary. Much of this DON may be refractory, and play little part in nitrogen cycling or microalgal production in the estuary. A substantial part of the lower catchment is used for agriculture, and preliminary estimates suggest that this activity delivers a moderate load of total nitrogen (TN), much of which may be labile or available. This load, about 200 t N p.a., is comparable with the load of DIN and labile organic N estimated to be delivered by finfish farms.

We are not able to quantify precisely the role of sediments in nitrogen cycling in the estuary. However, seasonal budgets indicate that the sediments act as a seasonal reservoir, accumulating organic nitrogen in the summer, and releasing it in the winter. On an annual basis, the estuary is a substantial sink for TN (about 1250 t N p.a.), which must be either buried in sediments or lost through denitrification. Measurements in the sediment suggest that the organic nitrogen there is primarily of marine origin, produced by phytoplankton from inorganic forms of the element. Thus, the nitrogen sink in the estuary is predominantly supplied by the influx of marine nitrate.

These analyses suggest that nitrogen cycling and algal production in the Huon Estuary are currently supported mainly by inputs of marine DIN, so in that sense the algal blooms observed there should be regarded as “natural”. However, there is some cause for concern in the observed composition of DIN in bottom waters at the marine boundary in summer and autumn. Relatively high contributions from ammonium and nitrite suggest that much of this DIN is regenerated in water column and sediments, either locally or in adjacent waters (D’Entrecasteaux Channel). The model results reported in Chapter 10 assume that the DIN in bottom waters at the marine boundary in summer is of marine origin, and that its concentration is set externally, independent of loads to the estuary. If this DIN results from recycling of organic matter generated in the Huon Estuary, then this concentration would increase with nutrient loads, thereby increasing the predicted sensitivity of the estuary to increases in anthropogenic nutrient loads. To understand the sources of this DIN more fully, the study domain would need to be extended into D’Entrecasteaux Channel.

Biological activity dominates the processing of nutrients in the waters of the Huon Estuary. In the classification scheme discussed in Chapter 4, it is an extreme example of a “biochemical reactor” with microalgae toward the marine end of the estuary mediating the concentration and distribution of nutrient elements. Almost no evidence of a “geochemical reactor” (reactions on, or adsorption to, particles) was observed in its expected location of the brackish upper estuary. Concentrations of suspended particulate matter in the estuary are very low because it is a microtidal system, with minimal sediment resuspension. Most of the streams discharging to the estuary also have very low concentrations of suspended solids. Geochemical processes are, of course, still very important, but it is the processes occurring in sediments that need to be considered. There is also evidence that some terrestrially derived DOM is removed as it passes through the estuary, but it is not clear whether this might be related to colloid formation or to microbial uptake.

11.2.5. Finfish farming in the Huon Estuary

The sustainability of finfish culture needs to be examined at a range of scales. If one considers the scale of lease site first, on-farm practices can induce local environmental conditions that are not tenable in the long-term. Excessive feed input or overstocking, with inadequate fallowing, can lead to organic matter being deposited on underlying sediments at such a rate that the natural processes of organic matter decomposition in the sediments cannot keep pace. The result is disappearance of much of the sediment biota, anoxia (and concomitant release from the sediments of toxic substances), and often more disease outbreaks (see Chapter 1). Usually this degradation is very obvious, and it directly affects the economics of farm operation, with alarming increases in deaths of finfish stock. Industry practices and government agency regulations in Tasmania, as outlined in Chapter 1, are both designed to prevent such outcomes.

Fallowing of lease sites is a key management action for sustainable finfish farming, and is standard procedure in the salmon industry in Tasmania. Our pilot study of fish-farm fallowing showed that waste from fish cages was largely restricted to the area under the cages, although small amounts could reach to 20 m beyond this. After a 11-month fallowing period, the initial anoxic conditions of the surface sediment directly under cages had returned to oxic conditions, but a significant portion of the organic waste remained in the sediment. A fallowing regimen might also have some advantages in disease control, especially if pathogens or vectors of disease are associated with the sediments beneath cages. However, the time needed for these agents to be neutralised would have to be established in relation to that of benthic recovery of the sediments.

Further work is needed to assess the level of environmental risk from reintroducing a cage to a fallowed area, since sediment conditions may degenerate more rapidly than at a pristine site. The choice of sample and reference sites for monitoring changes needs careful consideration in an environment where sediment grain size (and hence organic matter content) varies as greatly as it does in the Huon estuary. This research should include assessment of the chemical, microbiological and faunal status of the sediments over three years, preferably with monthly sampling over the first 6 months and at increasingly longer periods thereafter.

In the long-term, fish-farming operations with repeated cycles of stocking and fallowing may lead to subtler biogeochemical or ecological effects on sediments and benthic communities close to lease sites. To date, Tasmanian regulators have established operational and monitoring requirements for salmonid farms designed to prevent unacceptable environmental impacts on benthic communities and sediment conditions more than 35 m from lease boundaries (see also Chapters 1 and 8).

Marine farm operations can also impair local water quality near lease sites. Finfish farms are a substantial point source of nutrients both through excretion and remineralisation in sediments. Respiration of the organic matter accumulated below cages constitutes a substantial local sink for dissolved oxygen, which may result in local oxygen depletion. Owing to advection, these water-column effects are much more likely to be detectable beyond lease boundaries. Our observations suggest that bottom-water oxygen depletion may be detectable at times beyond lease boundaries, even though mixing and dilution will reduce these effects moving away from the farm site. More attention may need to be given to developing and applying a mixing zone concept for impacts of finfish farms on water column parameters. The local advection and dispersion of nutrients and dissolved oxygen near leases can be addressed by quantitative numerical modelling. Where many lease sites are established in an estuary or embayment, one must be concerned about the impact of the combined nutrient loads on the system as a whole. The assimilation capacity of the estuary for nutrients will essentially determine its carrying capacity for finfish farms. This is obviously a key issue for both farmers and regulators in managing expansion of the industry. This level — the system scale — has been the principal focus of the Huon Estuary Study.

According to the budget and model analysis in Chapter 10, current loads of DIN from salmon-farms are small compared with marine sources. In a simple biogeochemical model, removal of salmon-farm loads results in reductions in DIN by about 10% in winter, and reductions in DIN and chlorophyll by about 25% in summer.

In the model, doubling farm loads in summer increases chlorophyll by about 25 to 30%, and increases DIN by about 50%. This would result in some risk of increased frequency and density of summer blooms. At four times current loads, DIN doubles again to around 2 mmol N m^{-3} , and chlorophyll increases to values around $4 \text{ mg Chl a m}^{-3}$, or about twice current levels. At these levels, DIN is only a weak constraint on phytoplankton growth, and there is a substantial risk of prolonged blooms. At ten times current loads, DIN increases to values around 6 mmol m^{-3} , sufficient to saturate phytoplankton growth. Predicted phytoplankton biomass reaches a maximum of about $7 \text{ mg Chl a m}^{-3}$, owing to a combination of self-shading and flushing (Fig. 10.14). At this point, the nature of the system has changed completely. Note that, by avoiding flushing and light-limitation, *G. catenatum* would be able to attain much higher biomass than predicted by the model in these experiments.

These predictions by the model may underestimate, and in one aspect overestimate, the impact of increases in nutrient loads, because the model was simplified (see Chapter 10). More sophisticated models would better resolve spatial and temporal variability in the estuary, and better represent the role of sediments. These models could be used in conjunction with a well-targeted monitoring program to support tactical or operational management of finfish farms, and also tackle strategic issues such as the estuary's carrying capacity and catchment management.

The spatial domain appropriate for considering system-scale effects may expand or contract, depending on the issues and key processes. We have already noted that, for the purposes of examining the effects of nutrients from salmon farming in the Huon Estuary, it would seem necessary to expand the domain to include D'Entrecasteaux Channel. For other issues, it may be necessary to consider an even larger spatial domain, and to consider human practices, and environmental and ecological conditions of the past. Some graphic examples are provided by exotic pest species that have not been constrained by the bay, inlet or estuary to which they were introduced. For salmon farming in the Huon Estuary and other coastal waters to be managed as a sustainable enterprise, all of these environmental scales ought to be heeded. It must be said emphatically that, as the scale enlarges, changes often become more subtle; and mariculture (both finfish and shellfish) is but one of many human activities putting pressure on coastal ecosystems. It should not be the focus of undue or unfair attention. Nonetheless, it is in the long-term interests of both industry and environmental managers to take account of these large-scale, system-wide effects and feedbacks.

Finfish farmers can also address many of the issues of their farms' environmental effects on receiving waters through operational solutions. In the Huon Estuary, the salmon industry has installed automated feeders that have cut wastage of feed significantly. Another option is to move toward "bag systems" in place of net pens (such a system is presently being tested in another coastal water body in Southeast Tasmania). These devices allow solid waste to be collected and disposed of; they also open the possibility of treating water circulated in the bag, before returning it to the coastal water body. Optimisation of feed, both in its nutritional formulation and in the stability and resilience of the product when in salt water, provide further opportunity to minimise wastage. Feed manufacturers are seeking to improve digestibility of pellets, and to tailor the macronutrient ratios both for the advantage of the farmed fish, and for the receiving environment (e.g. Talbot and Hole 1994, Hillestad et al. 1999).

11.3 Environmental Monitoring in the Huon Estuary

As with many estuarine systems, the Huon Estuary is subject to strong spatial and temporal gradients. Our multi-tiered approach to sampling (see Chapter 2) has worked well in capturing these scales. Here we review some of the key phenomena and their associated space- and time-scales in the estuary, with a view to the implications for monitoring.

The estuary is strongly stratified much of the time, with a surface layer separated from bottom waters by a sharp pycnocline. Although density stratification within the bottom layer is weaker, significant vertical gradients of nutrients and chlorophyll were observed at times within the bottom layer. Locally, vertical stratification varies strongly on short time scales of hours to days, as a result of variation in river run-off, wind-driven mixing and advection of the surface plume, surface heating and cooling, and tidal motion.

The microalgal biomass (observed as *in situ* chlorophyll fluorescence) was frequently concentrated in subsurface layers. During dinoflagellate blooms, the motile microalgae were observed as chlorophyll-rich bands making extensive and regular vertical migrations. During diatom blooms, these bands were not as mobile, yet they often shifted with vertical excursions of the pycnocline. This makes sampling concentrations of microalgae by conventional bottle methods quite unreliable. Either vertical integrating methods (pump or tube) or continuous profiling methods are recommended.

Nutrient concentrations generally varied monotonically with depth, and a reasonable picture could be obtained from surface and bottom samples. However, our results suggest that more attention needs to be paid to vertical gradients of nutrients (and probably DO) within the bottom layer. Again, consideration should be given to vertical-profiling techniques.

Along-estuary gradients in salinity and nutrients result from the mixing of marine and freshwaters. Cross-estuary gradients typically reflect the tendency of the surface plume to hug the northeastern shore. With traditional sampling techniques, it takes several days to sample the entire estuary, and tracer distributions are potentially misrepresented (e.g. aliasing) by tides and variation in wind and river run-off. Even rapid underway mapping of surface waters takes several hours to traverse the estuary, and is subject to some distortion of data. Given the variability evident in data from local automated profiling samplers, the local details of any particular quasi-synoptic or synoptic (simultaneous observation over all of the estuary, e.g. remote sensing — see below) distribution are probably ephemeral in any case. Unless there is a specific purpose (e.g. to look at dispersion away from a point source), there seems little point in trying to map horizontal distributions with high spatial resolution routinely. It is more important to capture the broad-scale variation along the estuary, and into major side embayments such as Port Cygnet and Hospital Bay.

Nutrient concentrations show a regular seasonal cycle driven by marine boundary concentrations and microalgal uptake, with some short-term variation driven by mixing or advection events, and regeneration after algal blooms. The dinoflagellate blooms, which last for several weeks, are resolved fairly well by weekly monitoring. Some of the diatom blooms are quite short-lived, and may not be adequately sampled by weekly monitoring. In general, phytoplankton composition within the lower estuary is fairly homogeneous, and could be followed adequately at a limited number of sites. Microalgal biomass shows some horizontal

variation, but given its strong temporal and vertical variation, may be better monitored by few sites monitored frequently, than by many sites sampled rarely.

Remote sensing has the potential to monitor phytoplankton and other optical constituents synoptically. Planned satellite sensors will have high spatial resolution (30 m) and high spectral resolution, allowing separation of chlorophyll, CDOM and suspended solids. However, the Huon Estuary is so dominated by CDOM that it may be quite difficult to apply remote-sensing algorithms, except by exploiting red wavelengths (solar-stimulated fluorescence) under bloom conditions. Bio-optical sensors *in situ* could provide information on phytoplankton biomass and composition (for direct use, as well as for 'ground-truthing' remotely sensed data). The spectral signatures identified here below 400 nm (see Chapter 6) deserve further attention.

Bearing in mind the likely tactical and strategic needs, as well as the natural spatial and temporal scales of variation, it seems appropriate to design a monitoring strategy for water-quality variables (T, S, DO, algae, nutrients) around high-frequency sampling at a few sites chosen to resolve the large-scale horizontal gradients. A set of three to five sites along-estuary, with some additional sites to resolve cross-estuary variation, should be adequate. Some sites should be next to finfish farms, to meet both the operational needs of industry and strategic environmental objectives.

The prototype automated-profiling systems tested in this study provide high temporal and vertical resolution at relatively low operational cost. We have shown that, using radio transmission, these sensors can be linked to make a real-time, operational information system. Existing technologies allow monitoring of temperature, salinity, chlorophyll fluorescence, turbidity and dissolved oxygen in this way, although the choice of DO sensors needs further investigation. Automated sensors for nutrients and algal composition should become available in the future.

For some purposes (e.g. monitoring of harmful algal blooms and nutrient status), there is a need to back these automated measurements with *in situ* sampling. This could be done on a regular (weekly) basis. It might be possible to increase the routine sampling interval without loss of information by using an adaptive sampling program, with additional *in situ* sampling triggered by observations from the automated sensors.

How might an environmental monitoring network operate?

An environmental monitoring network would consist of an array of autonomous field stations reporting to a base station where logged environmental data are processed and stored. The purpose of such a network would be to provide information to the mariculture industry (finfish and shellfish) for effective operation of their marine farms, while at the same time monitoring the environmental health of the estuary. The two are compatible activities.

A prerequisite for such a network is that fundamental baseline information has been gathered, so that the physical and ecological functioning of the estuary is known in outline, and its current status is recorded. The study reported here has gathered such information for the Huon Estuary.

Since the natural processes and human activities in the catchment can affect the health of the estuary — the natural ecosystem, as well as aquaculture operations — it is necessary to look at the catchment as the geographic unit.

The monitoring needs to be considered in two parts: the estuary itself, and the catchment and its streams.

The elements of the network

Salinity, temperature and dissolved oxygen will need to be measured in the estuary at all stations. As discussed above, all three variables should be monitored as a continuous vertical profile, rather than at a subset of discrete depths. These measurements are the basis for following estuarine dynamics, and are as relevant for estuary-wide management as for operation of marine farms. If, as in the Huon Estuary, microalgal blooms are an issue, *in situ* chlorophyll fluorescence measurements throughout the water column are valuable as an estimate of biomass. They are needed at the core set of estuarine stations (we have suggested previously three to five of these), and a fluorometer might also be included on a subset of other aquatic sensor systems (see also discussion on estuarine indicators in Ward et al. 1998).

A second tier of measurements might also be considered. Although we think they are less critical on the whole, this would depend on circumstances and location. They could be implemented intermittently by temporarily upgrading an existing field station. Meteorological measurements (especially wind velocity and direction) are valuable for their influence on the hydrodynamics of water bodies, but only a subset of stations would need to be instrumented.

Direct measurement of currents by current meters are useful to establish such matters as the dispersal of wastes from sea cages holding finfish, and they can be used to validate a model's estimates of currents. Nevertheless, a conventional current meter records at just one point. An acoustic Doppler current profiler (ADCP) measures over a depth range, but implementation of a hydrodynamic model of estuarine circulation is a better tool for environmental management. Given routine access to an array of salinity and temperature data, it provides estuary-wide coverage. A variety of applications can be considered, such as simulation of observed conditions, or prediction of the effects of atypical or modified conditions.

Similarly, nutrient measurements are useful when tracing sources, estimating loads, or investigating their biogeochemical cycling. However, from a management perspective, ephemeral injections of nutrients (and patchiness in their distribution) in a water body as large as the Huon Estuary might be very difficult to observe reliably from a small number of monitoring stations in the estuary. Excess nutrients, especially nitrogen in marine systems, are readily assimilated by microalgae. Dilution away from the source in the wider estuary can also challenge methods of nutrient analysis. If possible, the place to measure inputs of nutrients is in the input discharge, rather than in the estuary near the point of entry. For sources within the estuary, such as salmon farms, this is not possible. An alternative approach to gain this information for salmon farms is to use closed experimental or operational systems — for instance, impervious enclosures (“bag” systems) in place of sea cages, or in-ground ponds (e.g. hatchery facilities), where all inputs and outputs are closely monitored.

Microalgal biomass is the more amenable environmental indicator of aquatic eutrophication for seasonal to longer intervals. Nevertheless, nutrient monitoring is useful and necessary to follow

broad-scale changes in nutrient concentrations. It can be done less frequently (weekly) at a selected number of sites, as suggested above, or intermittently at automated field stations by upgrading instrumentation (e.g. continuous monitoring for one day a month).

The frequency of measurements for autonomous field instruments should be hourly, for the reasons given in the previous section in relation to operational monitoring for finfish farms. We envisage automated continuous profilers, as described above, in the middle and lower estuary, and a third in Port Cygnet (because it appears to be functionally different to the main arm of the estuary) as a minimum. The value of measurements from the full water column has been discussed previously. The continuous profilers can also serve as a reference station for associated sensor strings. The sensor strings provide measurements at a series of discrete depths to extend the coverage of estuarine data. They are relatively inexpensive and robust, and thus suited to operational deployment in and around marine-farm leases. Data from the sensor strings would be telemetered to the network base station via the continuous profiler platforms.

The estuarine components of the monitoring network are depicted schematically in Fig. 11.1, which also shows the basic elements of the catchment part of the network. Here we are looking to monitor the effect of run-off on estuarine conditions. The two priority measurements are stream flow and turbidity. The first determines the volume of water discharged from a tributary over a set period — essential for load calculations. Sharply increasing flows, in themselves, are often an early warning of deteriorating water quality. Turbidity is a useful record because it is related to the suspended solids concentration in the stream, but it has enhanced value because turbidity can be a surrogate for nutrients (N and P) concentration and also for the extent of faecal contamination (Bobbi 1998). Measurement of rainfall in the catchment is discretionary for estuarine management, and finfish and shellfish farming. Its advantage is that it can provide earlier warning of increased discharge from a catchment or subcatchment.

In the Huon Estuary, monitoring stations are needed on the main tributaries — the Huon and Kermadie Rivers. Turbidity and stream flow (at different locations) are already gauged on the Huon Estuary. This instance illustrates the advantages of incorporating existing infrastructure into monitoring networks if at all practicable. The small and transient nature of the flow of other minor tributaries make them difficult to instrument. Rainfall measurement in some of these subcatchments, particularly those of Port Cygnet, could provide some workable information on local peaks in discharge.

Derived data products

An environmental monitoring network provides operational data that can be used directly as supplied (e.g. dissolved oxygen for finfish-farm operations). However, the data can be “value-added” by processing it further. *Derived data products* can be the combining of several types of data reported by sensors on field stations, or some other synthesis from the raw data, or they could be extended to use or develop mathematical models. Its use in environmental models has already been touched upon above. Some other examples of derived data products are the following:

- (i) Toxic algal bloom forecasting in a susceptible estuary — synthesis of the probability of microalgal blooms from water temperature, immediate history of rainfall, wind velocity and direction, and

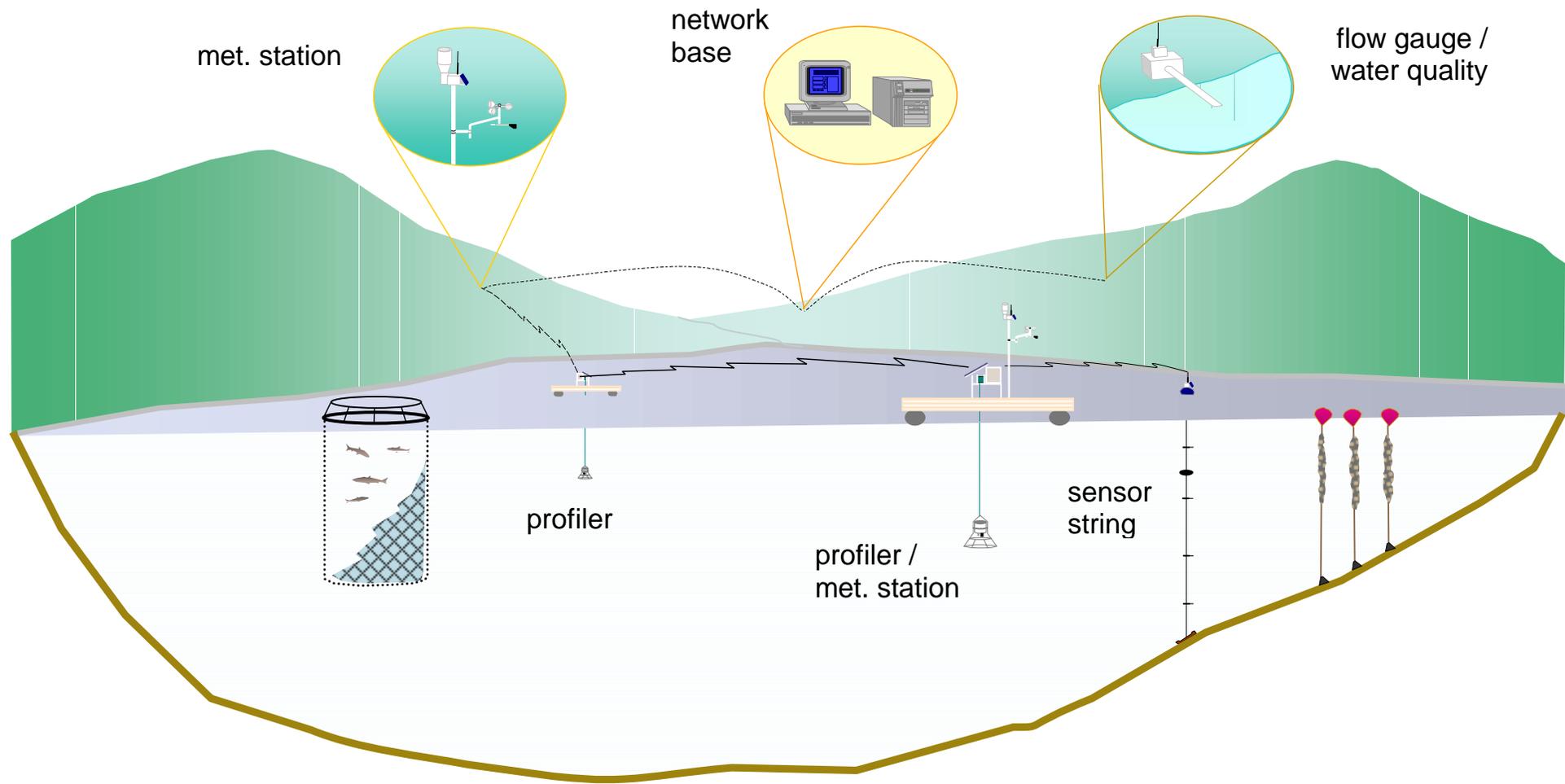


Fig. 11.1 Conceptual diagram of an automated environmental monitoring system for the Huon Estuary. The array of monitoring elements is only indicative of the system.

in situ fluorescence – as an indicator of initial microalgal biomass. (The triggers for the toxic dinoflagellate *G. catenatum* have been discussed above in Subsection 11.2.3).

- (ii) Environmental-health monitoring for shellfish culture in inshore waters — coupling rainfall distribution over catchment, resulting freshwater inflow and turbidity with the known record of waterborne concentrations of pathogenic organisms (*E. coli* and others) to provide early warning of unsafe water quality for harvesting of shellfish.
- (iii) Small-scale dissolved oxygen and toxicant (e.g. free ammonia, hydrogen sulfide) prediction in and around finfish farms at intervals of hours to days — using a combination of monitored dissolved oxygen, temperature, salinity, turbidity (and possibly *in situ* fluorescence) along with a nested hydrodynamic model, fish excretion algorithms and baseline information on sediment characteristics. This is a concept of a tool for fish farms to estimate acute and chronic hazards to fish stock.

The opportunities for environmental monitoring networks will grow rapidly with experience of their application in the field. Their value to the community is not limited to mariculture, nor is it constrained to the estuary, but can be extended to integrated catchment management (e.g. reduced use of pesticides through optimal application). Other obvious uses in the catchment are improved flood-warning services, microclimate forecasting — and for the broader community, an environmental database into which other position-referenced information can be entered.

11.4 Pointers for Management of the Huon Estuary and its Catchment

11.4.1. Catchment management

The quantity and quality of water delivered by the Huon River (above Judbury) is critical to the current ‘substantially natural’ condition of the Huon Estuary, and for preserving that status into the future. Most of the upper catchment remains in its natural state, apart from forestry in selected coops in a central corridor. Nevertheless, it must be borne in mind that any action, in what is otherwise a very healthy catchment, can have broader ramifications. For example, excessive use of fertiliser (urea is often a component) on regrowth forest can modify the nutrient status of upland streams. ⇒ *We advise that the current, near-pristine state of the upper Huon catchment should be carefully maintained because it is the main source of freshwater to the estuary, and that thorough measures are instituted to prevent localised degradation from human activities.*

From limited investigations in this project and earlier studies, it is evident that some of the subcatchments with streams emptying directly into the Huon Estuary (i.e. the lower catchment) are environmentally degraded — by reference to national guidelines, and by comparison with neighbouring streams in the catchment. From our own study, one such example is the Kermadie River and tributaries. Others studies have flagged more subcatchments: Port Cygnet streams (Agnes and Nicholls Rivulet systems), Prices Creek, and possibly part of the Mountain River system (see Gallagher 1996 and Bobbi 1998). These streams constitute a small fraction of the flow into the Huon Estuary. Their continued deterioration might initially pose only localised environmental problems for the estuary, but eventually the increased loads of nutrients and contaminants will cause more widespread trouble. The Huon Catchment – Healthy Rivers Project has recently framed a catchment management plan to tackle such issues (HC–HRP

1997). ⇒ *We advise that management plans designed to prevent further deterioration of the lower subcatchments should be implemented, and steps taken to improve the quality of run-off from the degraded subcatchments.*

11.4.2. Estuary management

The interim nutrient budgets that have been presented in this study draw on numerous sources to calculate loads to the Huon Estuary. They are interim because, although they use the best information available, there are many assumptions in their derivation. It would be useful to maintain an up-to-date record of such information as STP discharge data, use of fertilisers in the catchment, amount of finfish feed used in mariculture, and other similar information, so that nutrient loadings could be calculated as accurately as possible. Coupled with information gained as suggested in the subsection on knowledge gaps (11.4.9), this would ensure that nutrient budgets would be a valuable management tool. ⇒ *We advise that information pertaining to nutrient budget estimates in the Huon Estuary is kept up to date, and is made as comprehensive as possible (this could form part of a broader catchment audit process).*

The historical use of Whale Point by timber processing industries has affected the environmental condition of the site and its surroundings. The land-based issues aside, it has left a legacy of sediments in Hospital Bay and nearby in the main estuary that are enriched with wood fibres (see Chapter 9). The distribution and composition of these altered sediments are not fully described. ⇒ *We advise that any development planned for Hospital Bay (or nearby in the main estuary) that could disturb the underlying sediments should require a survey of the sediment quality and characteristics.*

11.4.3. Carrying capacity for finfish farming

Preliminary modelling of the Huon Estuary suggests that if nutrient loads from finfish farms increased by a factor of 4 (from 1997 loads), the result would be a trebling of DIN and doubling of chlorophyll, and an increased risk of algal blooms. These predictions are subject to some uncertainty, arising from limitations in the models and underlying knowledge (including the historical context of present conditions in the estuary). It is desirable that this uncertainty be reduced (see Subsection 11.4.9).

The implications for management decisions regarding further expansion of finfish farming in the Huon Estuary depend on a number of factors. These include environmental objectives for the Huon Estuary, issues of sustainability of both finfish and other (shellfish) aquaculture in the estuary, and the degree of uncertainty or risk that both industry and managers are prepared to accept. It is increasingly common practice in making environmental decisions to include these factors, along with scientific understanding derived from this and other studies, in a formal risk assessment process. It was not the purpose of this study to undertake such a process, nor to prescribe how it might be done. ⇒ *We advise that a formal risk assessment of the system's carrying capacity should be carried out to underpin any further expansion of finfish farming in the Huon Estuary.*

11.4.4. Operational practices on finfish farms

Our study of fallowing as an environmental management practice for fish farms was a pilot exercise. It revealed that, at the Hideaway Bay site, the sediments within the lease boundary had not recovered fully to reference site conditions after an arbitrary six-months lying fallow. We used chemical biomarker compounds to make this assessment. A similar conclusion was reached by a parallel project at the Tasmanian Aquaculture and Fisheries Institute using biological measures. ⇒ *We advise that the salmonid-farming industry work with government and research agencies to establish a more detailed understanding of the environmental effects of different fallowing practices with the objective of developing generic guidelines.*

11.4.5. Near-field environmental effects of finfish farming

Environmental regulation of finfish farms in Tasmania currently focusses on monitoring and managing near-field changes to surface sediments. Research to date has suggested that, provided industry best-practice for stocking and fallowing is adopted, benthic degradation is likely to be restricted to within tens of metres of cages. In contrast, near-field effects on water quality (nutrients and dissolved oxygen) have received little attention, but they are likely to extend up to hundreds of metres beyond lease sites, as a result of advection. These broader effects are not explicitly considered in current regulations or in compliance monitoring. This raises two kinds of potential problems. Environmental change which managers see as unacceptable may be occurring unnoticed. Conversely, current regulations (and possibly national guidelines) might not provide a realistic basis for managing water column degradation (the concept of a mixing zone may be more appropriate), leaving industry exposed to compliance requirements that are unachievable and unintended. ⇒ *We advise that near-field effects of marine farming on water quality should be reconsidered, to ensure that regulations and compliance measures are appropriate.*

11.4.6. Monitoring

Our study has demonstrated the distinct advantages of the use of a range of technologies and approaches to estuarine monitoring. We have also provided in this chapter (Section 11.3) a framework for developing a monitoring system for the Huon Estuary that satisfies both the operational needs of the aquaculture industry and the expectations of environmental managers. We recognise that implementation and maintenance of a monitoring system can be expensive, and that pragmatic trade-offs might be required initially, taking into account the priorities of both managers and industry. ⇒ *We urge that all users and other stakeholders of the Huon Estuary work in concert to develop a monitoring strategy, and foster the design of monitoring systems, to assist with informed decision-making about the use of the estuary and developments along its shores.*

11.4.7. Harmful algal bloom management

We have identified the environmental conditions that promote *Gymnodinium catenatum* bloom initiation and development. By monitoring water column conditions on appropriate time-scales (daily to weekly), it should be possible to make short-term predictions of the risk of harmful algal blooms (occurrence and extent). We are unable yet to make predictions on longer time

scales. However, based on our observations, we believe that careful monitoring of temperature, salinity, and wind speed and direction may facilitate longer-term predictions. ⇒ *We advise that any monitoring program seeking to identify causative factors for harmful algal blooms include measurements of parameters, at weekly intervals (or even daily at critical periods), that are indicators of environmental conditions known to support bloom events, along with integrated phytoplankton analysis.*

11.4.8. A decision-support system for managing the Huon Estuary.

To be effective, a monitoring program must be tied into management. The management issues for the Huon Estuary that need to be considered include tactical management of aquaculture operations, carrying capacity for aquaculture, management of catchment loads, impacts of catchment loads on environmental health and aquaculture, and interactions of climate and ocean forcing with catchment loads and aquaculture. It is difficult for any one sector or agency to support the monitoring needed for all, or even most, of these issues. Moreover, there are strong interactions among sectors, and large overlaps in information needs. By building on the system understanding developed through the Huon Estuary Study, and by integrating monitoring with system models designed to support management needs, it is possible to reduce the overall cost of monitoring, and to deliver information and analyses that answer the needs of both tactical and strategic management across sectors. ⇒ *Given the importance of the Huon Estuary as a site for salmonid culture, and its potential role as a model system both nationally and internationally, we suggest that the development of an integrated decision-support system for the Huon Estuary should be considered.*

11.4.9. Knowledge Gaps

We believe that this Study represents a substantial advance in our knowledge of the state and function of the Huon Estuary as a dynamical system. At the same time, we find that this advance in knowledge has uncovered new questions and knowledge gaps. This is a common if not clichéd phenomenon, and a source of frustration and even scepticism on the part of managers. However, few would argue that we can learn all there is to know about complex natural environmental systems in one short-term study. It is increasingly recognised that a significant investment is required to establish a sound knowledge base for a given system or system type, and that there are strong strategic benefits in building on this knowledge base through follow-up studies, rather than moving to a new system to start afresh.

Furthermore, it is a scientific custom at the finish of a research project to identify the subsequent research steps for other scientists that should provide advances in knowledge. The investigators are given a perspective of the system (in our case, the Huon Estuary) at the end of their work, which is not afforded those that are starting fresh in its study.

With this in mind, we have prepared a list of key knowledge gaps and research issues which we believe deserve further study in the Huon Estuary. We should emphasise that we are not proposing that management decisions be postponed until this work is carried out. We recognise that management decisions must be taken in their own time-frame, based on the best available current knowledge. We believe that the current study provides a greatly improved knowledge base for decisions in the Huon Estuary. At the same time, we would argue that all of the

research topics listed below have the potential to affect management decisions, and to lead to improved management decisions in the future.

The strongly stratified two-layer circulation in the Huon estuary results in rapid flushing, which reduces its vulnerability to eutrophication. However, stratification increases the potential for DO depletion in bottom waters. This is of concern both for operational management of finfish farms, and for broader environmental management. Although our results indicate that oxygen depletion does indeed occur, they are not adequate to identify the spatial and temporal extent. *⇒ We advise strongly that more detailed investigation be made of the depletion of dissolved oxygen in bottom waters of the estuary to characterise the extent of the problem, to isolate causes, and to develop means of predicting high-risk locations and times of the year.*

The Huon Estuary Study did not include the development of a highly resolved 3-D hydrodynamic model. Our budgeting and modelling results have substantial limitations owing to the coarse horizontal and vertical resolution of the inverse models used. The study has provided a data set suitable for developing and calibrating a high-resolution hydrodynamic model and water-quality model. Such a model could be used to support operational management of fish farms, location and design of new finfish or shellfish farms, and provide a better understanding of environmental impacts. This model could also be used to underpin more sophisticated biogeochemical and ecological models, which deal more realistically with vertical migration and sediment biogeochemistry. *⇒ We recommend the development of a three-dimensional hydrodynamic model, and more sophisticated biogeochemical and ecological models, of the Huon Estuary as management support tools.*

In Chapter 10, we presented interim budgets for nitrogen in the Huon Estuary. While marine sources clearly dominate nitrogen inputs over a year, there is uncertainty remaining as to the source of nitrogen loads in bottom waters in summer. This has potentially major implications for the assimilation capacity for fish-farm and catchment loads. To address this, *⇒ we advise that specific process studies focussed on nutrient regeneration in bottom waters and sediments of the lower Huon Estuary and adjacent D'Entrecasteaux Channel should be undertaken.*

It appears that D'Entrecasteaux Channel may play a significant role, as part of an extended Huon estuarine circulation, as a conduit for marine nitrate, and as a site for nitrogen cycling and regeneration. It is also a region for marine-farm development in its own right. *⇒ We advise that consideration should be given to a follow-up study of nitrogen cycling and algal blooms, building on the Huon Estuary Study, but expanded to include D'Entrecasteaux Channel. It should not be presumed that conditions and processes in the Channel would necessarily mirror those in the Huon Estuary.*

We advise that further studies should be undertaken to measure seasonal variation in organic matter contents in sediments, rates of chemical and biological breakdown, and further refinement of the models to include measures of the sediments' contributions of nutrients to bottom waters. There is obvious scope for further, clearly targeted sampling designs over small sections of the estuary to investigate specific sources such as sewage and transport of organic matter off fish-farm leases. Given the importance of grain size as a determinant of organic matter content in sediments, we recommend that any further sediment studies should include this measurement and that reference sites be carefully chosen so that their sedimentological characteristics match those of sediments in the study area.

It would also be valuable to derive a budget for carbon, in the same manner as for nitrogen and phosphorus. It is the carbon — or more specifically the labile or decomposable organic carbon — which is responsible for much of the consumption of dissolved oxygen in bottom waters of the estuary. ⇒ *We advise that a carbon budget should be developed for the Huon Estuary (including fractionation of the total into labile and refractory forms) so that it might assist in the evaluation of factors contributing to depletion of dissolved oxygen in bottom waters of the estuary.*

To fully understand the bloom dynamics of *G. catenatum* in the Huon Estuary, it is essential to identify the role in bloom initiation of resting cysts in the sediments. ⇒ *We recommend an investigation of the distribution of resting cysts in the sediments of the estuary and the neighbouring D'Entrecasteaux Channel. We also recommend that the sediment-water interface be studied on monthly time scales to identify any seasonality in resting cyst germination.*

There are many other research questions which could follow on from the research described in this report. Examples include the role of grazers (including algicidal bacteria) in regulation of microalgal bloom dynamics, and the possible relevance of 'marine snow' (buoyant particulate matter) as a site for denitrification. They have been omitted from the formal recommendations above because they are not in our view as high priority as the tasks listed there, at least with respect to environmental management of the Huon Estuary and catchment.

In addition, there are other ways to reduce the environmental effects of human activities (marine farming *and* land-based enterprises) on the waterway that can be best dealt with by tactical research in the respective industry, or in the sectors that support the industry. These matters were not in the scope of our project, but we have referred to some areas of possible improvement for finfish farming at the end of Subsection 11.2.5 (see also Butler and Macleod 1999).

11.5 References

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Glossary

abiotic	non-living (<i>e.g.</i> abiotic particles)
absorbance	a measure of the amount of light absorbed by a substance compared to that which passes through it (strictly the logarithmic ratio of the intensity of transmitted light to the intensity of incident light)
ADCP	acoustic Doppler current profiler (an instrument measuring current over the full water column)
advection	<i>oceanography</i> — the movement of water horizontally, or horizontal mixing of bodies of water, on a large scale (<i>c.f.</i> convection)
aerobic	requiring or utilising free oxygen in the air, or dissolved oxygen, for metabolic purposes (<i>c.f.</i> anaerobic — living in the absence of free oxygen)
algal bloom	a sudden appearance of a high concentration of phytoplankton resulting from increased reproduction as a response to favourable conditions
algicidal bacteria	bacteria which produce compounds that kill algae
aliasing	a phenomenon produced by insufficient sampling of a continuous function to depict it as a different function with coincident values at the same points of observation (<i>e.g.</i> “undersampling” of a wave function could incorrectly portray a signal with a lower frequency)
analyte	a component of a sample that is determined by chemical analysis
autotrophic	self nourishing; concerning an organism able to manufacture food from inorganic sources (<i>c.f.</i> heterotrophic)
backscatter	light scattered by particles at angles greater than 90 degrees to the incident beam
benthos	flora and fauna at or near the bottom of the sea, often living in association with aquatic sediments (benthic — <i>adj.</i>)
bioaccumulate	relating to substances that are taken up and concentrated in living organisms (<i>e.g.</i> DDT bioaccumulates in birds of prey); bioconcentrate and biomagnify are synonyms
bioavailable	a substance or a fraction of a substance that can be taken up by living organisms
biogenic	of biological origin, or synthesised by living organisms.
biogeochemistry	a branch of chemistry that deals with chemical elements and their interaction with rocks, sediments, soils, and life forms; and more generally, the cycling and distribution of chemical elements in the environment
biomarker	<i>organic geochemistry</i> — an organic chemical compound that serves as an indicator of an individual organism, a group of organisms, or a biochemical process

biomass	the number of individual organisms (in some area or volume or region) multiplied by the average weight of the individuals (can be applied to a group of organisms, <i>e.g.</i> plankton, or as a quantitative estimate of the entire assemblage of living organisms)
carotenoid	one of a group of biological pigments that are either derived from carotene or are structurally similar (<i>i.e.</i> their backbone is a conjugated polyene chain of eight isoprene units); often “accessory” pigments (yellow – red) to chlorophylls in microalgae (<i>c.f.</i> chlorophyll)
CDOM	chromophoric (or coloured) dissolved organic matter (in estuaries this often comes from soil- or vegetation-derived humic substances); <i>also</i> gilvin (<i>q.v.</i>)
chlorophyll	the green pigment (based on a porphyrin ring, with magnesium as the chelated metal centre) common to all plants, and fundamental to the process of photosynthesis; known in a variety of forms (<i>e.g.</i> chl <i>a</i> — chlorophyll <i>a</i>)
chromophoric	colour-bearing or -producing; relating to a functional group or groups in a chemical compound that produces colour
clastic	<i>geology</i> — consisting of fragments of older rock
coliform	a bacterium related to, resembling, or being <i>E. coli</i> , typically found in the intestinal tracts of humans and other animals
conservative	<i>oceanography</i> — a property that is neither created or destroyed, unchanging; <i>marine chemistry</i> — a property linearly related to salinity (neither added to nor removed during mixing of waters of different salinity)
convection	<i>oceanography</i> — thermally produced upward or downward movement of water (<i>c.f.</i> advection)
coprostanol	a sterol (derived from cholesterol) that is a faecal biomarker (<i>q.v.</i>) for many animals (coprostanol is ~60% of total sterols in human faeces)
C _{org}	amount of carbon present as organic matter, expressed as % dry weight
CRM	certified reference material (for quality assurance of chemical analysis)
CTD	conductivity–temperature–depth — a submersible probe or instrument for measuring these variables in profiling a water column
cyanobacteria	autotrophic photosynthetic bacteria (formerly known as blue-green algae) — single-celled or colonial — containing chlorophyll (<i>q.v.</i>), but not in chloroplasts typical of true plants
denitrification	biochemical reduction of nitrite or nitrate to gaseous nitrogen (N ₂) or nitrous oxide (N ₂ O) carried out by a diverse group of bacteria under oxygen-depleted (but not anoxic) conditions (usually coupled with nitrification: oxidation of ammonia to nitrate)
detritus	<i>geology</i> — consisting of particles or debris worn away from some solid body by detrition, disintegration, or decomposition; <i>biology</i> — organic matter produced by the decay or disintegration of a substance or tissue (detrital — <i>adj.</i>)
diagenesis	the first stages of alteration — chemical, physical and biological — of deposited sediments, but before metamorphism and consolidation (diagenetic — <i>adj.</i>)

diatom	a member of a diverse class of (micro-)algae, Bacillariophyceæ, having siliceous cell walls (mostly unicellular; chloroplasts contain the brown pigment fucoxanthin along with chlorophylls)
diel	involving a 24-hour period (usually includes a day and the adjoining night; <i>also</i> circadian, but not diurnal, which can mean (i) recurring daily, or (ii) occurring in the day time)
diffusion	The net movement, on the molecular scale, of units of a substance from areas of higher concentration to areas of lower concentration of that substance
DIN	dissolved inorganic nitrogen (i.e. nitrate, nitrite and ammonia as a group)
dinoflagellate	a member of the class of single-celled planktonic organisms Dinophyceæ, having two flagella (includes autotrophic, heterotrophic and mixotrophic [able to switch between autotrophic and heterotrophic] species; most species in this group contain the pigment peridinin along with chlorophylls)
discretisation	the act of dividing a continuous variable (often time and space) into discrete components, for purposes of analysis or numerical computation
DO	dissolved oxygen
DOC	dissolved organic carbon
DOM	dissolved organic matter
DON	dissolved organic nitrogen
DOP	dissolved organic phosphorus
ecosystem	an ecological unit composed of the abiotic (<i>q.v.</i>) environment together with one or more communities of organisms, and the interactions among them all — biotic and abiotic
E_h	redox potential (measured in millivolts)
entrainment	<i>fluid dynamics</i> — the transferring and mixing, by friction, of a fluid on one side of an interface by the opposing flow of the fluid on the other side
epibenthos	organisms living at, or just above, the sediment surface (epibenthic — <i>adj.</i>)
euphotic zone	the upper (photic) layer of a water body into which sufficient sunlight penetrates for photosynthesis and abundant plant growth to occur; the maximum depth of which is usually defined at the 1% PAR (<i>q.v.</i>) level
eutrophic	water bodies or habitats having high concentrations of (macro)nutrients, and commonly associated with high production of algae and other plants therein (<i>c.f.</i> oligotrophic)
F	filtered
fallowing	the process of allowing a substrate (soil or sediments) to lie dormant and unused for commercial biological production for a time, so that it might recover from the disruption; <i>aquaculture</i> — the removal of an aquatic farming operation from over, or on, the floor of a water body, so that the surface sediments might return (partially or fully) to their natural state
FAME	fatty acids measured as methyl ester derivatives

flushing time	The time required to replace all the water, or the volume equating to a layer (a slight volume adjustment is needed to account for mixing in a strict mathematical treatment), in an estuary, harbour, or other confined water body by action of river discharge, current and tide
gilvin	dissolved, coloured organic substance (typically yellow to brown) in natural waters, arising from the dissolution of plant or soil matter and usually resistant to chemical and biological attack; <i>also</i> Gelbstoff or yellow substance (gilvin is more commonly applied to freshwaters, and Gelbstoff to marine waters); CDOM (<i>q.v.</i>)
halocline	a salinity gradient; a zone of rapidly changing salinity with depth — usually a boundary between two water masses of different salinity (hence, subhalocline — waters beneath the halocline)
HES	Huon Estuary Study
heterotrophic	concerning an organism unable to synthesise organic compounds essential for its wellbeing; it must acquire them by consuming other organisms (<i>c.f.</i> autotrophic)
HPLC	high performance liquid chromatography — an analytical technique for separating and determining chemical compounds (earlier the letter P in the acronym had stood for pressure)
humic	pertaining to or of humus; formed or derived from plant remains (in <i>aquatic science</i> often referring to humic acid, a complex organic compound that is a coloured solute in natural waters)
hydrodynamics	the branch of physics, which studies the forces acting upon or exerted by liquids — either flowing or static (hydrodynamic(al) — <i>adj.</i>)
infaunal	pertaining to animals living within a substrate; <i>marine biology</i> — living within a soft sediment and being large enough to displace sedimentary grains
kts	knots (nautical miles per hour; 1 kt \equiv 1.15 statute miles per hour or 1.85 kilometres per hour)
leachate	the solution resulting from the percolation or infiltration by water into a substrate to remove substances (often impurities) that can be dissolved or leached (<i>e.g.</i> tip leachate)
LOI	loss-on-ignition — the content of organic matter and carbonate (unless it is removed in a preliminary step) estimated by measuring weight loss in sediment core subsamples after combusting at specified high temperatures.
macronutrient	any chemical element that is required in relatively large amounts for the growth and development of an organism, including such elements as carbon, hydrogen, oxygen and nitrogen (but can vary depending upon the organism — <i>e.g.</i> diatoms (<i>q.v.</i>) require silicon as a macronutrient); <i>c.f.</i> micronutrient
macrophyte	any (usually aquatic) plant large enough to be seen by the naked eye
meiofauna	animals whose size makes them neither small enough to be microfauna (microscopic), nor large enough to be macrofauna (visible to naked eye) — one reference suggests that the shortest dimension is less than 0.5 mm but greater than or equal to 0.1 mm

mesotrophic	the stage in condition of a natural water body or habitat between oligotrophic (<i>q.v.</i>) and eutrophic (<i>q.v.</i>); having intermediate concentrations of nutrients and therefore moderately productive
MFDP	Marine Farming Development Plan (in Tasmania)
microalgae	single-celled, photosynthetic aquatic plants
μM	micromolar (\equiv micromoles per litre) — see mol
μmho	micro-mho (\equiv microsiemens, μS) — unit of electric conductance
micronutrient	any chemical element that is required in trace amounts for the growth and development of an organism, including such elements as iron, zinc, iodine and selenium (but often varies from species to species); <i>c.f.</i> macronutrient
microphytobenthos	microscopic aquatic plants associated with the sediment surface
microtidal	a small tidal range, usually less than two metres
mol	mole (SI chemical unit) — a unit of concentration, <i>e.g.</i> mmol = millimole
monospecific	of one species only
nM	nanomolar (\equiv nanomoles per litre) — see mol
nanoplankton	planktonic organisms 2–20 μm in size; hence, nanoflagellate
NIR	near infra-red
N_{org}	amount of nitrogen present in organic matter, expressed as % dry weight
NO_x	nitrogen oxyanions (in aquatic chemistry: nitrate + nitrite)
oligotrophic	water bodies or habitats having low concentrations of (macro)nutrients, and commonly associated with low production of algae and other plants therein — water column usually remains well oxygenated throughout (<i>c.f.</i> eutrophic)
oxic	oxygenated; containing detectable concentrations of dissolved oxygen (<i>c.f.</i> anoxic — devoid of detectable dissolved oxygen)
PAR	photosynthetically active radiation (wavelengths of between about 400 nm and 700 nm that are used by plants in photosynthesis)
phaeopigments	compounds formed by degradation of chlorophyll and related pigments
phytoplankton	microalgae (<i>e.g.</i> diatoms, dinoflagellates <i>q.v.</i>) that live in the water column and are incapable of swimming against a current
PN	particulate nitrogen
POC	particulate organic carbon
PON	particulate organic nitrogen
porewaters	interstitial waters (water in the spaces between sediment grains)
ppb	parts per billion (w/w unit of concentration, <i>e.g.</i> nanogram (10^{-9}) per gram)
ppm	parts per million (w/w unit of concentration, <i>e.g.</i> microgram (10^{-6}) per gram)

primary production	the production of organic (living) matter by photosynthesising organisms or by chemosynthesising organisms from inorganic materials. Usually expressed as grams of carbon per square metre per year (over the full water column), or per cubic metre per year (volume)
pycnocline	a density gradient; a zone of rapidly changing density with depth — usually a boundary between two water masses of different density (density being influenced by both salinity and temperature)
redox	an abridgement of reduction–oxidation — referring to simultaneous electrochemical reactions of oxidation and reduction (<i>also</i> redox potential — the measured electrochemical cell potential for such a reaction; used to define oxidising and reducing conditions in the aquatic environment)
remineralisation	<i>geochemistry</i> — conversion or degradation of organic matter back to its mineral (inorganic) constituents
resting cyst	a life stage formed by dinoflagellates or diatoms, which remains dormant for some period of time before reforming an active planktonic cell
speciation	<i>chemistry</i> — the chemical form of an element, as determined by analytical means (<i>e.g.</i> nitrate — an oxyanionic form of nitrogen, DOC — a class of carbon compounds, $(\text{CH}_3)_2\text{Hg}$ — an organic form of mercury)
SPM	suspended particulate matter (<i>also</i> suspended solids)
STP	sewage treatment plant
supersaturation	pertaining to waters that are above the saturation concentration of a dissolved gas (saturation is when the solubility of a gas in water has reached its maximum at equilibrium with the atmosphere — the actual concentration varies as a function of temperature and salinity)
TDN	total dissolved nitrogen (inorganic and organic)
TDP	total dissolved phosphorus (inorganic and organic)
TN	total nitrogen (dissolved and particulate)
TP	total phosphorus (dissolved and particulate)
trophic	concerned with nutrition or its regulation; <i>ecology</i> — of or pertaining to the feeding habits of, and the food relationship between, different types of organisms in the food-cycle
turbidity	an operationally determined measurement that is related to the “murkiness” of water (it is quantified by light either scattered from, or absorbed by, suspended particles and colloidal material, with perhaps minor contributions also from CDOM <i>q.v.</i>)
UF	unfiltered
undersaturated	pertaining to waters that are below the saturation concentration of a dissolved gas (usually dissolved oxygen — respiration depletes dissolved oxygen leading to undersaturation); <i>c.f.</i> supersaturation
xenobiotics	synthetic chemical compounds that are not naturally found in the environment (<i>e.g.</i> many of the organochlorine pesticides)

zonal latitudinal or an east–west orientation (*e.g.* zonal flow — the east to west component of wind flow along a global latitude; *c.f.* meridional — longitudinal)

This glossary was developed in part by reference to several World Wide Web on-line resources, among others: The Macquarie Dictionary, Oxford English Dictionary, Academic Press Dictionary of Science and Technology, Glossary of Marine Biology (from *Marine Biology: Function, Biodiversity, Ecology* by Jeffrey Levinton) and Glossary of Oceanography and the Related Geosciences with References (by Stephan K Baum)

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