# A REVIEW OF ISSUES RELATED TO DEPLETION OF POPULATIONS OF SELECTED INFAUNAL BIVALVE SPECIES IN THE HAURAKI GULF MARINE PARK A report prepared for the Hauraki Gulf Forum by

**Article**

2 authors:

Some of the authors of this publication are also working on these related projects:

Project    Biotoxins in Abalone View project

# A REVIEW OF ISSUES RELATED TO DEPLETION OF POPULATIONS OF SELECTED INFAUNAL BIVALVE SPECIES IN THE HAURAKI GULF MARINE PARK



**A report prepared for the Hauraki Gulf Forum**

**by**

## Coral M. Grant and Brenda E. Hay

*AquaBio Consultants Ltd*
*P. O. Box 560*
*Shortland St. P. O.*
*Auckland*

**September 2003**

# <u>DISCLAIMER</u>

Cover Photograph: Photograph of the infaunal bivalve species considered in this report (*clockwise from top left*): tuatua (*Paphies subtriangulata*), pipi (*Paphies australis*), cockle (*Austrovenus stutchburyi*), and wedge shell (*Macomona liliana*). (Photograph by Mike McMurtry).

**Contact Details:**

AquaBio Consultants Ltd.
P.O. Box 560
Shortland St P.O.
Auckland 1
New Zealand

Phone/Fax (09) 448- 1505
Email: Hay@AquaBio.co.nz

## ACKNOWLEDGEMENTS

## EXECUTIVE SUMMARY

The Hauraki Gulf Forum is concerned at the perceived depletion of inter-tidal, infaunal bivalve populations in the Hauraki Gulf Marine Park. To this end, AquaBio Consultants Ltd. was asked to undertake the following work:

- To provide a reference bibliography relevant to factors potentially affecting bivalve populations in the Hauraki Gulf Marine Park (HGMP).
- To provide a synthesis of current and expected near-future knowledge of factors potentially affecting bivalve populations in the HGMP.
- To identify gaps in our understanding of factors potentially affecting bivalve populations in the HGMP.
- To identify key research priorities where efforts are likely to be most cost-effective at identifying what is affecting bivalve populations in the HGMP.

The bivalve species of specific interest in this review include the cockle (*Austrovenus stutchburyi*), pipi (*Paphies australis*), tuatua (*Paphies subtriangulata*), and wedge shell (*Macomona liliana*).

Initially, data on the abundance of these shellfish was reviewed to identify the scientific basis for the perception that infaunal bivalve populations in the Hauraki Gulf Marine Park are becoming depleted. This includes a review of data from abundance surveys that have investigated population characteristics on an on-going basis over several years (such as the Ministry of Fisheries "*Inter-tidal Shellfish Monitoring Programme*", Cheltenham Beach Caretakers' *A. stutchburyi* monitoring programme, and various programmes funded by the Auckland Regional Council such as the Mahurangi Estuary Ecological Monitoring Programme). Baseline data provided by one-off studies (e.g. research undertaken for University theses) have also been identified.

The Ministry of Fisheries "*Inter-tidal Shellfish Monitoring Programme*" was initiated in 1992 with the aim of assessing the depletion of inter-tidal shellfish populations through regular surveys. While these surveys represent the most extensive ssurvey activities relating to shellfish depletion, long-term data on infaunal bivalve abundance are limited due to changes in sampling design and methodology across time, changes in survey sites, and the fact that for different years of the survey different combinations of species were included as target species for investigation. In addition, data from inter-tidal surveys may not be representative of the total abundance of *P. australis* and *P. subtriangulata* in an area, since both species may have adult populations that are predominantly subtidal.

Overall, the majority of sites surveyed over the various studies show decreasing trends in inter-tidal infaunal bivalve abundance. However, the small quantity of robust data available makes generalisation to the Hauraki Gulf Marine Park as a whole inappropriate. We note that observation of the depletion of infaunal inter-tidal shellfish stocks in the Hauraki Gulf Marine Park region appears to be supported by a substantial body of anecdotal evidence.

A number of potential stressors of infaunal bivalves in the Hauraki Gulf Marine Park were identified from a literature review, and assessed in detail using a standard risk assessment framework that incorporates hazard identification, dose-response assessment, exposure assessment and risk characterisation.

Potential stressors to infaunal bivalves include:

- anthropogenic contaminants such as organotin compounds and organic booster biocides (such as those associated with marine antifoulants), heavy metals, organochlorines and polyaromatic hydrocarbons;
- human harvesting;
- changes in the marine environment associated with human activity, such as increased sediment loading, nutrient enrichment and climate change;
- natural phenomena of an extraordinary nature such as harmful algal blooms, and diseases/parasite events.

The risk of other potential stressors, including pharmaceutical and personal care products, brominated flame retardants and invasive species, was also briefly reviewed.

In general, paucity of data, particularly about potential impacts on New Zealand species of infaunal bivalves, severely limits robust risk characterisation. However, in broad overview the potential risks to infaunal bivalves in the Hauraki Gulf Marine Park are characterised by:

- A general trend of increasing risk with increasing proximity to metropolitan areas, particularly the large metropolitan area of Auckland. This is matched by a higher level of information about the status of shellfish populations and potential anthropogenic contaminants in areas closer to Auckland.
- A general trend of increasing risk with increasing urbanisation of the coastline.
- Within these trends, there is a general trend of increased risk of anthropogenic contaminants in inter-tidal zones in enclosed estuarine areas as opposed to open coastal environments.
- Risk may vary between bivalve species as a result of physiological and behavioural differences.
- Many potential stressors may impact on several life cycle stages of the shellfish. In instances where data are available, there is a general trend (with some exceptions) of higher sensitivity to potential stressors in larval and juvenile stages than in adults.
- Based on the limited information available, it appears that while many of the potential stressors considered are known to have lethal impacts at high "doses", there are more potential sub-lethal impacts that occur at lower doses. These include impacts that could result in reduced shellfish production (e.g. reduction in reproductive condition). The level of information relating to these long term lower dose effects is minimal.
- While acknowledging that there are significant knowledge gaps, in general it appears that the occurrence of potential stressors at high levels are relatively rare and localised. (An exception to this could be harmful algal blooms, which can have relatively widespread impacts). The occurrence of sub-lethal stressors is likely to be more common.
- Most potential stressors associated with human activity are predicted to have a temporal trend of increasing risk (exceptions to this are the risks associated with organotin compounds, lead, and most organochlorines).
- The immediacy of the risk varies between potential stressors – for example, increased sediment loading as a result of increasing urbanisation of the coastline is likely to present a current risk to infaunal shellfish in a range of areas across the Hauraki Gulf Marine Park, whereas the potential risks associated with climate change have implications in the longer term.

The risk analysis highlights the limits to our knowledge of the way in which potential stressors may impact on infaunal bivalve populations in the Hauraki Gulf Marine Park. Knowledge gaps exist across all levels of the research and risk assessment process. In general, paucity of data,

particularly about potential impacts on New Zealand species of infaunal bivalves, severely limits robust risk characterisation, and this is an issue that needs to be addressed with respect to each potential stressor. However, it is also important to note several other limitations to the risk assessment process used:

- The risk assessment process, which attempts to assess the risk of individual potential stressors, does not address the issue of the potential impact of combinations of stressors. The interaction of combined stressors may have very significant impacts, and warrants research consideration in the future.
- The dose-response data tend to relate to impacts on individuals within a population, and exposure data (where available) relates to stocks within a population. In order to predict the impact of these stressors on infaunal bivalve populations, a good understanding of the population dynamics and population genetics (i.e. the way in which the various stocks or beds within the population are inter-linked) is required.
- Particularly with respect to anthropogenic contaminants, much of the dose response data available are based on obvious and easily measurable impacts on organisms (such as mortality). The significance of subtle and cumulative impacts is only beginning to be recognised by environmental toxicologists. A major issue yet to be addressed by ecotoxicological science (and of potentially critical importance for risk assessment) is the impact on non-target species of stressors eliciting effects (perhaps *via* low but continual concentrations) sufficiently subtle as to go unnoticed in real time – but whose cumulative impacts eventually yield recognizable outcomes having no obvious cause. While it may be difficult to link individual sub-lethal impacts to population declines, the combination of an array of sub-lethal impacts may result in a significant impact at the population level over the long term. For this reason, it may be important to broaden the temporal scale of the approach to the potential problem of shellfish depletion in the Hauraki Gulf Marine Park (i.e. examine data over a longer time period), both with respect to the determination of trends in shellfish abundance, and research associated with identification of the impacts of potential stressors.
- We also note that the volume of data relating to the Coromandel Peninsula region is limited, with areas closer to Auckland having been subject to a greater level of investigation. Because of differences in catchment use between the different regions, it is difficult to generalise risk assessments from areas such as the Auckland metropolitan region to the Coromandel region.

If one can assume that significant, possibly localised depletion of infaunal bivalve populations is occurring within the Hauraki Gulf Marine Park, then consideration of the assessment of the relative risks of potential stressors, including the limitations of that process, does not offer any easy answers as to the probable causes of such depletion. At this stage, the extent to which the majority of individual shellfish stocks are impacted is unknown, and without this information it is difficult to develop any meaningful hypothesis about which potential stressors are likely to be those that are the most significant. We are even unable to robustly identify whether or not shellfish depletion occurs in patterns resembling the broad patterns of risk identified in the risk assessment process. We note that the ubiquity of a potential stressor does not necessarily correlate directly with the significance of its impact.

At present our knowledge of early life cycle stage mortality of all the infaunal bivalve species of interest in this review is minimal, and this limits our ability to detect the impact of potential stressors. Yet it is impacts on these stages of the life cycle that have the greatest potential flow-on effects in terms of future population abundance.

It is apparent that the patterns of depletion of infaunal bivalve populations in the Hauraki Gulf Marine Park are not clearly defined at present, either with respect to geographic extent or severity. However, despite the lack of conclusive studies to indicate that the problem is widespread, there is sufficient evidence to warrant further investigation of this issue.

In addition to many knowledge gaps identified in relation to the impact of specific potential stressors on individual shellfish, several broad issues requiring research were highlighted as a result of our review:

- There is a paucity of robust long-term data required to identify trends in infaunal bivalve populations in the Hauraki Gulf Marine Park. In particular, if meaningful data are to be collected on sub-tidal species such as *P. subtriangulata* (and in part, *P. australis*), surveys should be conducted in appropriate locations. Also important is the geographic spread of data, and collection of data from a variety of sites that differ with respect to hypothesised risk levels associated with potential stressors. In addition it would be useful to monitor infaunal bivalve populations when exposure to potential sterressors changes (such as the closure of shellfish beds to harvesting), so that the relative impact of such stressors can be assessed.
- Even when information is available to allow evaluation of the level of a particular anthropogenic contaminant in the marine environment, the associated knowledge necessary to convert this environmental level into the amount that is actually biologically available to infaunal bivalves is often lacking.
- Little work has been undertaken on the interaction between different potential stressors, particularly stressors that may have sub-lethal impacts. The potential for synergistic effects between different potential stressors needs to be investigated further.
- The paucity of knowledge about factors driving population dynamics in each of the infaunal bivalve species of interest limits the interpretation of risk assessment data with respect to shellfish populations throughout the Hauraki Gulf Marine Park. In other words, even with information on the impacts of potential stressors on shellfish in an individual sense, in most cases we are unable to predict the likely impact on the wider shellfish population. This relates particularly to the way in which shellfish stocks (or beds) are interconnected within a population (e.g. does a localised impact in one stock result in an impact in a distant stock as a result of reduced larval supply?).

Elucidation of the above issues would allow much of the existing environmental data to be interpreted appropriately. The volume of work necessary to definitively identify, and thus effectively manage, the causes of the depletion of infaunal shellfish stocks in the Hauraki Gulf Marine Park is large. The observed trends in shellfish abundance suggest that this issue needs to be resolved in the near future.

The size of the knowledge gaps, and the finite financial resources available to direct toward this issue, suggest that the elucidation of issues related to shellfish depletion in the Hauraki Gulf Marine Park will be difficult without a cohesive strategy to involve wider groups in this process. We suggest that this could be achieved in two ways:

*Facilitation of capacity-building:*

The complex nature of the issues, and the large number of knowledge gaps suggest that significant resources in terms of funding and expertise will be required to fully investigate the potential causes

of shellfish depletion in the Hauraki Gulf Marine Park. Our review of unpublished data, and consequent communication with people and organisations working in the marine environment revealed that there is a wide variety of potential sources of information about issues of relevance to depletion of infaunal shellfish populations in the Hauraki Gulf Marine Park. Much of this information is not utilised to its full potential. It is suggested that significant synergy could be achieved by utilising existing resources to enhance the knowledge base relevant to this issue. Potential contributors to this knowledge base include researchers from research organisations and Universities, community members involved in monitoring beaches, people from organisations that comprise the Hauraki Gulf Forum, shellfish farmers, Area Health Board Health Protection Officers associated with shellfish safety, iwi members associated with resource management issues etc. It is suggested that this process could be facilitated by:

- Development of an internet-based newsgroup to facilitate exchange of information.
- Hosting regular (6-monthly) workshops to discuss the shellfish depletion issue. This will provide an opportunity to disseminate and discuss research results, develop linkages between researchers, and encourage research effort into this area by research providers. Informally published proceedings of the workshops will provide a collation of latest research results.

*Improved use of existing resources:*

We suggest that collection of long-term data on the status of inter-tidal infaunal bivalves could be cost-effectively addressed by improving the use of existing resources, allowing available funding to be directed toward high quality research focused initially on the latter issues. To achieve this we suggest that the benefits of the "Adopt-a-Beach" programme be maximised. This could be facilitated by the Hauraki Gulf Forum through:

- Encouraging appropriately skilled people to become involved, to ensure quality of results
- Funding the establishment of a database of results from the programme, along with a periodic review of the results by an appropriately skilled scientist.

This could potentially broaden the geographic scope of data collected quite significantly.

These data should be complemented by the data collected from monitoring programmes such as the Ministry of Fisheries "Inter-tidal Shellfish Monitoring Programme". It is recommended that the Hauraki Gulf Forum encourage the Ministry of Fisheries to re-focus this programme to ensure that meaningful long term abundance data are collected.

In addition, we suggest that the possibility of undertaking a series of infaunal bivalve population surveys that replicate the location and methods used in historic surveys be explored as a first step to investigating the way in which population abundance has changed over a longer time period. This may assist in identifying subtle long term changes that are nevertheless significant to our understanding of the current situation.

# TABLE OF CONTENTS

## LIST OF FIGURES

## *LIST OF TABLES*

# LIST OF ABBREVIATIONS

| | |
|---|---|
| **A. stutchburyi** | *The New Zealand Cockle – Austrovenus stutchburyi* |
| **AhR** | *Aryl hydrocarbon Receptor* |
| **ANZECC** | *Australian and New Zealand Environment and Conservation Council* |
| **ARC** | *Auckland Regional Council* |
| **AVS** | *Acid Volatile Sulphides* |
| **BCF** | *Bioaccumulation Factor* |
| **BFR** | *Brominated Flame Retardants* |
| **BLM** | *Biotic Ligand Model* |
| **CBC** | *Cheltenham Beach Caretakers* |
| **Chl a** | *Chlorophyll a* |
| **DBT** | *Dibutyltin* |
| **DDD** | *1,1-dichloro-2,2-bis(p-chlorophenyl)ethane* |
| **DDE** | *Dichloro-diphenyl-dichloroethane* |
| **DDT** | *Dichloro-diphenyl-trichloroethanes* |
| **DPhT** | *Diphenyltin* |
| **DS** | *Deposited Sediment* |
| **$EC_{10}$** | *Effect Concentration, the concentration required to produce a particular chronic effect (e.g. decreased growth) to 10% of the test organisms. A time is usually given in brackets to indicate the period over which the experiment was run.* |
| **ERL** | *Effects Range Low – the threshold of possible adverse effects. Concentrations below the ERL value represent a minimal effects range, which should estimate conditions where biological effects would be rarely observed. Concentrations greater than or equal to ERL but lower than ERM value represents a range at which biological effects occur occasionally. In the ANZECC (2000) guidelines Interim Sediment Quality Guidelines-Low (ISQG-L) equals the ERL value obtained from the North American guidelines.* |
| **ERM** | *Effects Range Median – the concentration above which adverse effects would normally frequently occur. In the ANZECC (2000) guidelines Interim Sediment Quality Guidelines-High (ISQG-H) equals the ERM value obtained from the North American guidelines.* |
| **HABs** | *Harmful Algal Blooms* |
| **HBCD** | *Hexabromocyclododecane* |
| **HCB** | *Hexachlorobenzene* |
| **HCHs** | *Hexachlorocyclohexanes* |
| **HGMP** | *Hauraki Gulf Marine Park* |
| **HSDB** | *Hazardous Substances Data Bank* |
| **IPCC** | *Intergovernmental Panel for Climate Change* |
| **ISQG-Low** | *ANZECC Interim Sediment Quality Guidelines – Low, the threshold value of possible adverse effects.* |
| **ISQG-High** | *ANZECC Interim Sediment Quality Guidelines – High, the concentration above which adverse effects would normally frequently occur.* |

| | |
|---|---|
| *LC$_{50}$* | *Lethal Concentration required to kill 50% of the test organism. A time is often given in brackets to indicate the period over which the experiment was run (usually 96 hours for fish or 48 hours for some invertebrates).* |
| *M. liliana* | *The New Zealand wedge shell – Macomona liliana* |
| *MBT* | *Monobutyltin* |
| *MfE* | *Ministry for the Environment* |
| *MFish* | *Ministry of Fisheries* |
| *NPAHs* | *Nitrogen-heterocyclic polycyclic aromatic hydrocarbons* |
| *OCs* | *Organochlorines* |
| *P. australis* | *The New Zealand pipi – Paphies australis* |
| *P. subtriangulata* | *The New Zealand tuatua – Paphies subtriangulata* |
| *PAHs* | *Polycyclic Aromatic Hydrocarbons* |
| *PBBs* | *Poly-Brominated Biphenyls* |
| *PBDEs* | *Poly-Brominated Diphenyl Ethers* |
| *PCBs* | *Poly-Chlorinated Biphneyls* |
| *PCDDs* | *Poly-Chlorinated Dibenzo-p-dioxins* |
| *PCDFs* | *Poly -Chlorinated dibenzofurans* |
| *PCP* | *Pentachlorophenol* |
| *PEC* | *Predicted Environmental Concentration* |
| *PEL* | *Probable Effects Level – the concentration above which adverse effects are predicted to occur frequently.* |
| *PNEC* | *Predicted No Effect Concentration* |
| *PPCPs* | *Pharmaceuticals and Personal Care Products* |
| *PSP* | *Paralytic Shellfish Poisoning* |
| *PSII* | *Photosystem II, found within the chloroplasts of plants* |
| *SPM* | *Suspended Particulate Matter* |
| *SSRIs* | *Selective Serotonin Re-uptake Inhibitors* |
| *TBBPA* | *Tetrabromobisphenol-A* |
| *TBT* | *Tributyltin* |
| *TBTO* | *Tributyltin Oxide* |
| *TCDD* | *Tetrachlorodibenzo-p-dioxin* |
| *TCMS pyridine* | *2,3,5,6-tetrachloro-4-(methylsulphonyl)pyridine, an antifouling booster biocide* |
| *TCMTB* | *2-(thiocyanomethylthio)benzothiazole, an antifouling booster biocide* |
| *TEL* | *Threshold Effects Level – the concentration below which adverse effects are expected to rarely occur.* |
| *TEQ* | *Toxic Equivalents* |
| *TOC* | *Total Organic Carbon* |
| *TPhT* | *Triphenyltin* |
| *US EPA* | *United States Environmental Protection Agency* |
| *WHO* | *World Health Organisation* |

## SECTION 1: INTRODUCTION

The Hauraki Gulf Forum is concerned at the perceived depletion of inter-tidal, soft sediment bivalve populations in the Hauraki Gulf Marine Park (see Figure 1.1 below for a map of the extent of the area encompassed by the Park).
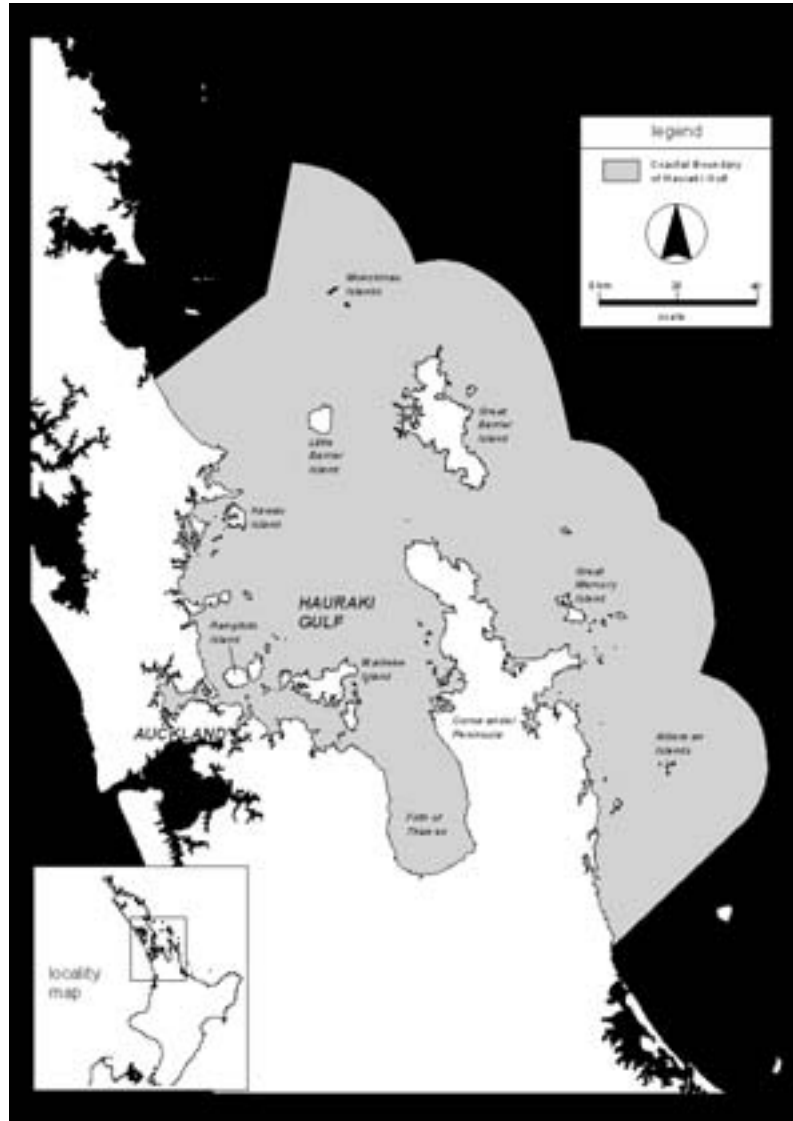


**Figure 1.1: Map showing the spatial extent of the Hauraki Gulf Marine Park along the eastern coast of the North Island of New Zealand.**

The Forum has proposed a programme to address the following issues:

1) What is causing the identified decline in edible bivalve abundance and changes in their distribution and size frequency structure?
2) Are these changes because of human activities over which resource managers may have some control, or are the changes a reflection of natural variability?
3) What can or should be done about it?

The Hauraki Gulf Forum has supported a staged approach to addressing the issue of inter-tidal bivalve depletion in order to produce the best platform from which to make decisions on the most cost-effective options for future work. The initial stages in this process are to identify:

a)      the information needs to properly define the issue, and
b)      any necessary focussed research that needs to be undertaken.

To this end, AquaBio Consultants Ltd. has been asked to undertake the following work:

- To provide a reference bibliography relevant to factors potentially affecting bivalve populations in the Hauraki Gulf Marine Park (HGMP).
- To provide a synthesis of current and expected near-future knowledge of factors potentially affecting bivalve populations in the HGMP.
- To identify gaps in our understanding of factors potentially affecting bivalve populations in the HGMP.
- To identify key research priorities where efforts are likely to be most cost-effective at identifying what is affecting bivalve populations in the HGMP.

The contract brief, which more explicitly details the scope of this knowledge review, is provided in Appendix I.

Briefly, the overall focus of the work is to collate information that might elucidate the observed declines in target bivalve populations within the Hauraki Gulf Marine Park, and make recommendations for future research work. The bivalve species of specific interest include the cockle (*Austrovenus stutchburyi*), pipi (*Paphies australis*), tuatua (*Paphies subtriangulata*), and wedge shell (*Macomona liliana*). Consideration of all life cycle stages of these species has been included in our study. Where relevant, information from other geographic areas, and other bivalve species, has also been included in our review.

As specified in the contract brief, our study encompasses collation and synthesis of information from previous and (where possible) current studies, both from published and unpublished literature. In order to access "grey literature", and information on current and future studies, a mail-out requesting information from research providers and organisations that might have commissioned such research was undertaken. A list of individuals and organisations from which information was requested is provided in Appendix II.

The following report forms one component of the results of our knowledge review. An electronic database of literature references relevant to factors potentially affecting infaunal bivalve populations in the Hauraki Gulf Marine Park has been prepared, and is presented separately on an accompanying CD.

The first sections of this report outline the current situation with respect to inter-tidal, soft sediment, bivalve populations in the Hauraki Gulf Marine Park. They provide background information on the basic biology and ecology of the bivalve species of interest (Section 2) including community level ecological processes such as competition and predation (Section 3), and a review of the results of density and distribution studies of populations of *A. stutchburyi*, *P. australis*, *P. subtriangulata*, and *M. liliana* that have been undertaken in the Hauraki Gulf Marine Park (Section 4). Section 5 reviews factors potentially impacting on these bivalve populations by utilising an ecological risk assessment framework, which is drawn from a commonly used risk analysis model. These

components are synthesised in the subsequent discussion (Section 6), and recommendations are made for prioritisation of future research (Section 7).

## SECTION 2:    *INFAUNAL BIVALVE BIOLOGY/ECOLOGY*

## 2.1    INTRODUCTION

Although the risk assessment presented later considers the potential impact of prospective stressors on individuals, it is the depletion of infaunal bivalves at a population level that is of interest to the Hauraki Gulf Forum.  Thus the biological features that influence population level dynamics are important.  Typically population studies measure where individuals of a species can be found in the environment (population distribution), how many individuals are present (population abundance or if per unit area, population density), the pattern of how individuals are distributed with regard to each other (population dispersion), the size distribution of a population, how fast individuals grow, their reproductive output, birth and death schedules, and the factors that alter these features. Population dynamics is a general title for studies that describe the temporal and spatial changes in the number (or biomass) of individuals in a population, and define the processes that produce these changes (Evans *et al.* 2002).  This section provides an introduction to the biology and ecology of the cockle (*Austrovenus stutchburyi*), pipi (*Paphies australis*), tuatua (*Paphies subtriangulata*) and the wedge shell (*Macomona liliana*), in order to contextualize the risk assessment undertaken later in this document.  The first part of this section provides information about bivalve shellfish in general, following which information specific to the species of interest in this review is presented.

## 2.2    INFAUNAL BIVALVES

Many bivalve shellfish, including those of interest to the Hauraki Gulf Forum in this review, follow a life cycle that incorporates a free-swimming larval stage, and a more sedentary adult stage. Adults are broadcast spawners, i.e. they release eggs and sperm into the sea where fertilisation occurs.  Following fertilisation there are two free-swimming larval stages: a short non-feeding trochophore stage (usually lasting 12-48 hours), followed by a longer veliger stage (2-3 weeks). Veligers feed on phytoplankton.  Although larval stages can swim enough to migrate up and down in the water column, water movement caused by currents and wind basically determines their dispersal.  The existence of a broad scale relationship between current patterns and larval abundance has been identified for marine benthic systems (Roughgarden *et al.* 1987).  Following the veliger stage, larvae undergo metamorphosis into juveniles that resemble small adult shellfish, and "settle" in or on the substrate.

Infaunal bivalve species may have open populations, and consequently stocks at a particular site may be maintained by recruitment of larvae from adults living elsewhere (Underwood & Fairweather 1989).  This complicates consideration of the issue of depletion of particular infaunal bivalve stocks/beds, since stress factors spatially removed from these shellfish stocks may impact on their recruitment.

In addition to dispersal at the larval stage, it is widely recognised that surface sediment movement (i.e. bedload transport) of associated post-settlement infaunal bivalves constitutes an important structuring process influencing their final distribution patterns (Turner *et al.* 1997).  Wind generated wave activity is considered to play the predominant role in driving sediment movement, with tidal advection of sediment also a significant factor (Cummings *et al.* 1995).  This process of passive dispersal is more likely to affect juvenile than adult infaunal bivalves.

In addition to passive distribution due to sediment movement, young individuals of several bivalve species have the ability to undergo active pelagic dispersal, even after a prolonged period of benthic existence (Sigurdsson *et al.* 1976).  In particular, post-settlement dispersal *via* mucous thread

drifting of very small juveniles has been demonstrated for many bivalve species, including *A. stutchburyi, P. australis*, and *M. liliana* (Pridmore *et al.* 1991, Cummings *et al.* 1995, Hooker 1995). Dispersal by drifting may also occur in larger shellfish. For example, Hooker (1995) found that both adult (up to 58 mm shell length) and juvenile (<15 mm shell length) *P. australis* could drift on long mucous threads.

Norkko and co-workers (2001) identified that dispersal of juvenile *A. stutchburyi* and *M. liliana* was decoupled from sediment bedload transport under calm hydrodynamic conditions, illustrating the importance of active dispersal behaviour under these conditions. They suggested that the rate of dispersal of these juvenile bivalves is likely to be markedly influenced by the day-to-day variations in wave conditions at a particular site (Norkko *et al.* 2001).

The degree to which these different dispersal processes impacts on the distribution of different infaunal bivalve species may vary. For example, Turner *et al.* (1997) found that dispersal of juvenile *M. liliana* was weakly correlated to sediment transport (suggesting active dispersal behaviour), while a strong relationship between sediment transport and juvenile *A. stutchburyi* dispersal was identified (suggesting passive dispersal). Recent laboratory work by Pilditch and co-workers (2003) has further investigated the impact of sediment substrate type and flow velocity on bedload transport versus mucous drifting in juvenile *A. stutchburyi* and *M. liliana* (sizes <2mm, 2mm, and >4mm). Results to date suggest that these bivalves respond differently at higher flow rates (e.g. 12-18 cm s$^{-1}$). *A. stutchburyi* were found to be dispersed only via the bedload, while with increasing flow speed *M. liliana* altered from bedload transport to mucous thread drifting. These results emphasis that species-specific interactions between behaviour and flow speed may regulate post-settlement dispersal dynamics (Pilditch *et al.* 2003).

As well as being spatially complex, temporal variability in recruitment may affect the maintenance of adult infaunal bivalve populations. Recruitment into bivalve populations may depend upon an occasional year of high recruitment (Sissenwine *et al.* 1988), while any reduction in the frequency of good recruitment may have strong negative impacts on infaunal shellfish populations. Long term studies of bivalve recruitment show that variation in recruitment can be quite large, with year-to year differences up to 100 or 1000 times being commonplace (see Honkoop & Van Der Meer 1998 for review).

Fluctuations in recruitment can be explained by differences in:

a) Stock size of adults (i.e. proportion of individuals forming the adult population)
b) The average reproductive output by each individual parent (which varies with food availability and water temperature)
c) Survival of gametes, larvae and juveniles.

A review by Gosselin and Qian (1997) concluded that post-settlement mortality of juveniles often exceeded larval mortality, and was a factor in determining benthic invertebrate population structure. Modelling of *Mercenaria mercenaria* population dynamics suggests that reductions in juvenile mortality will have 100 times the impact on the future number of individuals in a population than a proportional reduction in the adult mortality rate (Malinowski & Whitlatch 1988). Since larval and early post-settlement mortality rates are extremely high, very large increases in fecundity are necessary to duplicate the consequence of only slight increases in juvenile survivorship.

## 2.3 NEW ZEALAND COCKLE
### (Tuangi or huangi - *Austrovenus stutchburyi,* previously *Chione stutchburyi*)

The New Zealand cockle, *Austrovenus stutchburyi,* is not a true cockle (family Cardiidae), but actually a venerid clam (family Veneridae) related to the Northern Hemisphere species *Mercenaria mercenaria* and *Protothaca staminia* (Belton 1986). *Austrovenus stutchburyi* is however ecologically similar to the Northern Hemisphere species *Cardium* [= *Cerastoderma*] *edule* (family Cardiidae) (Grace 1973).

*A. stutchburyi* occurs throughout New Zealand, including the North, South, Stewart and Chatham Islands (Powell 1979). They live in lower inter-tidal mud/sand flats of protected estuarine areas (Morton & Miller 1973). Their distribution within harbours is restricted to the soft shore below lowest High Water Neap and may extend down to subtidal depths of 6–8 metres (Larcombe 1971, Stephenson 1981). Following a study of various *A. stutchburyi* populations around New Zealand, Larcombe (1971) suggested that the upper distribution limit of *A. stutchburyi* is likely to be bounded by a minimum submergence time of 3.5 hours per tide. Populations of harvestable *A. stutchburyi* (i.e. 35+ mm) are usually confined to areas covered for eight or more hours per tide (Belton 1986). Although duration of tidal cover and distance from the open sea are the principal determinants of *A. stutchburyi* distribution, substrate type, salinity, wind and tide effects, wave action, and predation are all important factors (Larcombe 1971).

Adult *A. stutchburyi* live near the sediment surface and are suspension-feeders, filtering phytoplankton out of the water column. A study by Gibbs *et al.* (2001) found that *A. stutchburyi* not only filter particulate matter brought into the area in the water column, but also ingest particles resuspended in the vicinity of their habitat, including benthic microphytes. Feeding and pumping water across the gills follows an endogenous circa-tidal rhythm. Most active pumping/feeding (which may be up to 3 litres/hour) occurs during a period of approximately two hours either side of high tide (Beentjes & Williams 1986). The shallow burrowing of this species is reflected in the high proportion of individuals that bear epizoans (i.e. organisms that live on the outside of an animal) (e.g. anemones *Anthopleura aureoradiata*, spionid worms *Boccardia acus*, barnacles *Elminius modestus* and limpets *Notoacmea helmsi*) (Ansell 2001).

The abundance of *A. stutchburyi* may be negatively affected by the percentage of silt in the sediment, and positively affected by the percentage of coarse sand (Clark 1997). Substrate stability is an important factor in *A. stutchburyi* survival, as mature cockles display poor mobility and are prone to burial (Belton 1986). (The relationship between *A. stutchburyi* and sediment is discussed in more detail in Section 4.6 Sediment Loading).

Sexes in *A. stutchburyi* are separate, with individuals over 18 mm producing gonad during the spring and summer, and spawning during summer-autumn (Larcombe 1971). Larvae swim in the water-column for 15-20 days before settling (Stephenson & Chanley 1979). Larcombe (1971) has shown that *A. stutchburyi* can settle and grow in adverse conditions, even to the extent that some populations may never reach spawning condition. However, in a separate study *A. stutchburyi* condition index was found to be ineffective in distinguishing between individuals from polluted and unpolluted sites in the Manukau Harbour (Gardner 1992). PhD candidate Megan Stewart (Auckland University) is currently studying the impacts of urban development on *A. stutchburyi* populations. This research investigates the sensitivity of adult *A. stutchburyi* to changes in habitat quality through transplantation from a relatively clean site (the Whangateau Harbour) to an estuary (Tamaki Estuary) with a known contamination (e.g. heavy metals and PAHs) and sediment

gradient. *Austrovenus stutchburyi* transplanted to the upper reaches of the Tamaki estuary (e.g. where contaminants were highest and sediment grain size was smallest) had high levels of mortality compared to controls and the sites at the mouth of the estuary following the transplant procedure. Interpretation of these results requires further incorporation of the environmental parameters measured, which is currently underway. This research also investigates the impact of adult *A. stutchburyi* density on settlement/recruitment, and the extent to which *A. stutchburyi* settlement/recruitment is affected by sediment type (including contaminated sediment) and hydrodynamics of Okura estuary (Stewart 2002). This work is still in progress.

Larcombe's (1971) studies of various *A. stutchburyi* populations around New Zealand identified that most populations consisted of adults of mean size approaching their maximum lengths (i.e. 40 mm). Growth rate and maximum age is extremely variable and greatly influenced by environmental factors. However, populations 20 years old and reaching shell lengths of greater than 60 mm have been recorded (e.g. Avon-Heathcote Estuary, Christchurch, R. Stephenson, *unpubl. data*). *Austrovenus stutchburyi* populations in the north of the North Island more commonly reach 8-10 years old (Dr. M. Larcombe, Consultant, *pers. comm.*). Growth generally slows considerably once the shell height reaches 40 mm (Kearney 1999). Larcombe (1971) identified that growth rates of *A. stutchburyi* tend to decline with increasing elevation above low water, and also with increasing distance from a harbour entrance. Transplant experiments by Dobbinson *et al.* (1989) showed that high- and mid-shore *A. stutchburyi* transplanted to lower on the shore (i.e. increased immersion time) exhibited increased growth rates. Condition indices of *A. stutchburyi* also showed that tissue volume and weight had a maximum at the low shore level. It is likely that both duration of feeding and food availability are factors affecting growth and condition. Marsden & Pilkington (1995) found that variation in *A. stutchburyi* condition index was related to both salinity and chlorophyll *a* levels in the Avon-Heathcote estuary. Laboratory experiments have shown that exposure of *A. stutchburyi* to low salinity can delay the onset of feeding, and in some cases the normal cycle of feeding activity can cease altogether (Pilkington 1992).

Concern about depletion of *A. stutchburyi* populations has prompted several studies related to the transplantation of *A. stutchburyi* into depleted populations. An analysis of genetic distance in *A. stutchburyi* populations by Lidgard (2001) failed to find evidence of genetic isolation related to distance, suggesting that *A. stutchburyi* populations from around New Zealand are similar. Consequently it was concluded that relocation of individuals from distinct habitats should not greatly affect the genetics of the existing populations (Lidgard 2001). Stewart and Creese (2002) investigated the process of transplanting *A. stutchburyi* into depleted populations, and looked at the success of transplanting juvenile (10-18 mm) versus adult (25-32 mm) *A. stutchburyi*. These pilot trials demonstrated that transplants of adult individuals would be more successful due to their higher retention within the transplant area (Stewart & Creese 2002).

## 2.4   NEW ZEALAND PIPI
## (*Paphies australis*, previously *Amphidesma australe*)

The pipi (*Paphies australis*) belongs to the family Mesodesmatidae (superfamily Mactridae) and is closely related to the tuatua (*P. subtriangulata* and *P. donacina*) and the toheroa (*P. ventricosa*). *Paphies australis* are generally restricted to sandbanks at the mouths of harbours and estuaries (Creese 1988), and occur in the upper 8-10 cm of coarse sediments (Morton & Miller 1973). *Paphies australis* are adapted to moderate shelter and some dilution of salt water with freshwater (Creese 1988).

*Paphies australis* have been generally regarded as an inter-tidal species, but there is evidence to suggest that they occur in sub-tidal channels down to depths of 6-8 metres. These sub-tidal locations may be the main habitat of adult *P. australis* (Venus 1984, Dickie 1986, Creese 1988, Hooker 1995, Cole *et al.* 2000). However, in some areas (e.g. Wenderholm, Pawley *et al.* 1997) inter-tidal populations have been observed to contain adults of a larger mean size than that of sub-tidal populations in the area. Hooker (1995) found that *P. australis* in the Whangateau Harbour had distinct habitat segregation based on size and age. Juvenile *P. australis* occurred high on inter-tidal shores, whereas fully mature adult animals (over 40 mm in shell length) were found at high densities (up to 4400 m$^{-2}$) in sub-tidal beds in the main harbour channels. Intermediate sizes occurred between these habitats.

*Paphies australis* reach a maximum size of approximately 80 mm in length (Creese, 1988). *Paphies australis* greater than 40 mm in length are sexually mature (Hooker & Creese 1995). *Paphies australis* populations on the north-eastern coast of the North Island spawn from early spring through to late summer (Hooker & Creese 1995). Hatchery rearing of *P. australis* larvae suggests that *P. australis* settle 18-22 days after fertilisation (Hooker 1995). In a study of *P. australis* recruitment associated with a sub-tidal adult population in the Whangateau Harbour, Parr (1994) identified that the vertical distribution of associated larval *P. australis* in the water column above the adult bed was essentially random over the tidal cycle, but that certain areas of the harbour displayed higher levels of recruitment due mainly to harbour hydrology. Hooker (1995) found that post-set *P. australis* from the same Whangateau adult population were restricted to a small strip, mid-inter-tidally, in clean coarse sediments.

Dr. R. Creese (Port Stephens Fisheries Centre, NSW, Australia, *Unpubl. data*) found high inter-annual variability in post-set *P. australis* density in the Whangateau Harbour (Dr. Bob Creese, Port Stephens Fisheries Centre, Australia, *pers. comm.*). Studies by both Hull (1996) and Cole *et al.* (2000) concluded that the temporal variation in sub-tidal adult *P. australis* density in Tauranga Harbour (Centre Bank) resulted from variation between annual recruitment events. They also found that post-set mortality and dispersal were important factors in regulating population size structure.

## 2.5   NEW ZEALAND TUATUA
## (*Paphies subtriangulata*, previously *Amphidesma subtriangulatum*)

The colloquial term "tuatua" is applied to two species, which may occur on the same beaches in central New Zealand (Richardson *et al.* 1982). Beu and De Rooij-Schuiling (1982) summarised the distribution patterns of these two species around New Zealand. They identified that *Paphies subtriangulata* is found around the North Island and in the north of the South island. *Paphies donacina* (the deep-water tuatua) occurs around the South Island and the north coast of Stewart Island, and the lower part of the North Island (both east and west coasts).

As outlined previously, the *P. subtriangulata* is closely related to *P. australis* and *P. ventricosa*. However, unlike *P. australis*, *P. subtriangulata* and *P. ventricosa* are surf clams, preferring to live on more open exposed sandy beaches with easily perceptible slope and direct wave access unimpeded by land (Morton & Miller 1973, Redfearn 1987). Other New Zealand surf clams also belonging to the superfamily Mactridae are *Mactra murchisoni* and *Spisula aequilatera*. The yellow clam (*Mesodesma mactroides*), which also belongs to the family Mesodesmatidae, is an overseas species similar to our tuatua, (Marsden 2000).

*Paphies subtriangulata* is the most inshore New Zealand species of surf clam (Cranfield *et al.* 1994). However their distribution is occasionally observed to verge on the inter-tidal zone (Dr. C.

Grant, AquaBio Consultants Ltd., *pers. obs.*).  Tuatua can occur on a remarkable range of beach types, from high energy dissipative beaches to sheltered reflective beaches (Dr. J. Duggan, University of California Santa Barbara, *pers. comm.*, Marsden 1999).  Bivalves within this surf zone are typically relatively mobile, being exposed to a shifting habitat with variable conditions of temperature, wave exposure, moisture levels, food supply and salinity extremes (McLachlan & Erasmus 1983).  Tuatua have short siphons, which restrict their location in the sediment to within 5-10 cm of the surface.  In severe storm conditions, *P. subtriangulata* may become dislodged from the surrounding sediment (Dr. C. Grant, AquaBio Consultants Ltd., *pers. obs.*).

Tuatua greater than 50 mm in shell length are sexually mature (Grant 1994, Marsden 1999).  They tend to have an annual breeding cycle with semi-continuous spawning over the spring/summer months (Grant & Creese 1995 – *P. subtriangulata*, Marsden 1999 – *P. donacina*).  Larval tuatua spend about 20 days in the water column before they are ready to settle (Redfearn 1987).  Tuatua recruitment occurs into the mid inter-tidal zone, with individuals moving to lower tide levels over time (Greenway 1981, Grant 1994, Marsden 2000).  Tuatua beds may show a distinct seasonal migration up and down the beach, occupying deeper water in winter/spring, and migrating back to shallower water by summer (Grace & Grace 1978, Grant 1994).  *P. subtriangulata* reach a maximum size of approximately 75 mm in length.

## 2.6    NEW ZEALAND WEDGE SHELL
## (Hanikura - *Macomona liliana,* previously *Tellina liliana*)

Tellinid bivalves are important members of coastal, harbour, and estuarine benthic communities around the world (Roper *et al.* 1992).  The tellinid bivalve *Macomona liliana* is found in estuaries and sheltered coastal waters throughout New Zealand (Morton and Miller 1973).  A northern hemisphere tellinid, *Macoma balthica,* is morphologically and ecologically similar to our *M. liliana* (Wilcock *et al.* 1994).

Morton and Miller (1973) describe the tidal distribution of *M. liliana* as the lower shore down to Extreme Low Water Spring.  However, other researchers (e.g. Davidson 1989, Roper 1990) have reported individuals in the subtidal benthos.  Taylor (1998) describes their preferred habitat as mid-tide level on large exposed sandflats characterised by well sorted fine sands.  He notes that other (but less favourable) sites include sheltered bays characterised by moderate-poorly sorted sands.  This study found that adult size classes are evenly represented in favourable habitats, whereas in less favourable habitats size classes are dominated by small adults (Taylor 1998).

The presence of high-density *A. stutchburyi* beds may physically obstruct *M. liliana* from feeding due to the difficulties of extending siphons through tightly packed *A. stutchburyi* beds (Morton & Miller, 1973).

Sediment stability also appears to be a factor in determining *M. liliana* distribution patterns.  A study by LeLieveld (2000) found that sediment stability increased as *M. liliana* density decreased.  LeLieveld (2000) surmised that *M. liliana* itself may influence sediment stability through disturbance of the sediment, consequently increasing the movement of sediment by water movement.  It was suggested that this could provide a mechanism to facilitate post-settlement dispersal, and thus control local densities of surface-dwelling organisms.

Adult *M. liliana* live 5-10 cm below the sediment surface and are generally regarded as deposit feeders.  They feed by extending their inhalant siphons out of the sediment to suck in loose detritus and benthic microphytes (including phytoplankton that settle out of the water column) from the

sediment surface (Gibbs *et al.* 2001). However, it has been observed that juvenile *M. liliana* maintained in aquaria extend their siphons straight up into the water column. This suggests that in juveniles, suspension-feeding may be an alternative to deposit-feeding (Pridmore *et al.* 1991). Similar observations have been made with respect to the northern hemisphere Tellinid, *Macoma balthica. Macoma* is capable of switching between feeding modes, depending on water flow parameters (Olafsson 1986). In order to deposit-feed efficiently on benthic microalgae, these Tellinids must live close to the sediment surface, so they can graze a maximal surface area (which is constrained by their siphon length) (de Goeij & Honkoop 2002).

In mid-tide populations, sexual maturity occurs in individuals >22 mm shell length (Taylor, 1998). Spawning may be protracted, beginning in spring and extending through to autumn (Roper *et al.* 1992, Taylor 1998). Recruitment has been observed from December to March (Taylor 1998).

Both Roper *et al.* (1992) and Taylor (1998) observed high recruitment variability of *M. liliana* within sites, between years, in the Manukau Harbour. Roper *et al.* (1992) found that both recruitment years observed showed very high mortality of small individuals, with only a few individuals progressing to the > 5 mm size range. Length frequency plots from the same study indicated that growth in the first year was 2-3 mm (Roper *et al.* 1992).

*M. liliana* may undergo a second settlement phase during which young shellfish use tidal transport to carry them from high-tide flats to low-tide flats and sub-tidal offshore areas (Roper *et al.* 1992). In flowing water conditions in the laboratory, post-settlement *M. liliana* (1-2 mm in length) have been observed to emerge from sediment and float using buoyancy provided by mucous "drifting threads" that were approximately 1.5 times the length of the animal (Cummings *et al.* 1993).

The presence of adult *M. liliana* may promote juvenile recruitment (e.g. Thrush *et al.* 1992). However, it is likely that such adult/juvenile interactions will be density dependent, with low adult densities aiding recruitment and high densities inhibiting it (Zajac & Whitlatch 1985). A more recent study by Thrush *et al.* (2000) showed that windwaves are likely to be an important factor accounting for observed variations in the interaction between adult and juvenile *M. liliana*.

## 2.7   CONCLUDING COMMENTS

In conclusion, even though studies investigating many aspects of the ecology of *A. stutchburyi, P. australis, P. subtriangulata* and *M. liliana* have been undertaken, it is acknowledged by researchers that the population dynamics of these infaunal bivalve species are not well understood. This is no doubt a function of the logistical challenges faced in such studies: Populations may occur over broad spatial areas, and important changes sometimes happen only over long intervals of time. In addition, the early stages of the life cycle of these infaunal bivalves are microscopic and cannot be comprehensively studied directly in their natural habitat (Evans *et al.* 2002).

## 2.8   REFERENCES

**Ansell, A. D.  2001.**  Dynamics of aggregations of a gastropod predator/scavenger on a New Zealand harbour beach. *Journal of Molluscan Studies.* 67: 329-341.

**Belton, R. J.  1986.**  The New Zealand clam, *Chione* (*Austrovenus*) *stutchburyi*: a study of industries based on similar resources in North America and Europe focusing on harvest, handling, resource management, and aquaculture. Winston Churchill Memorial Trust. Wellington, N.Z.  51pp.

**Beentjes, M. P. & B. G. Williams  1986.**  Endogenous circatidal rhythmicity in the New Zealand cockle *Chione stutchburyi* (Bivalvia, Veneridae). *Marine Behavioural Physiology.* 12: 171-180.

**Beu, A. G. & L. A. De Rooij-Schuiling  1982**.  Subgeneric classification of New Zealand and Australian species of *Paphies lesson* (Bivalvia: Mesodesmatidae), and names for the two species of tuatua in New Zealand.  *New Zealand Journal of Zoology*. 9: 211-230.

**Clark, L. J.  1997.**  A study of cockle (*Austrovenus stutchburyi*) populations in the Tamaki Estuary.  Unpubl. M.Sc. thesis.  School of Environmental Science.  University of Auckland.  118pp.

**Cole, R. G., P. J. Hull & T. R. Healy  2000.**  Assemblage structure, spatial patterns, recruitment, and post-settlement mortality of subtidal bivalve molluscs in a large harbour in north-eastern New Zealand.  *New Zealand Journal of Marine and Freshwater Research*.  34: 317-329.

**Cranfield, H. J., K. P. Michael, D. Stotter & I. J. Doonan  1994.**  Distribution, biomass, and yield estimates of surf clams off New Zealand beaches.  New Zealand Fisheries Research Assessment Document, 94/1.  17pp.

**Creese, R. G.  1988.**  Aspects of the ecology of pipi at Whitianga.  Report prepared for Wilkins & Davies Marinas Ltd.  54pp.

**Cummings, V. J., R. D. Pridmore, S. F. Thrush & J. E. Hewitt  1993.**  Emergence and floating behaviours of post-settlement juveniles of *Macomona liliana* (Bivalvia: Tellinacea).  *Marine Behaviour and Physiology*.  24: 25-32.

**Cummings, V. J., R. D. Pridmore, S. F. Thrush, & J. E. Hewitt  1995.**  Post-settlement movement by intertidal benthic invertebrates: do common New Zealand species drift in the water column?  *New Zealand Journal of Marine and Freshwater Research*.  29: 59-67.

**Davidson, R. J.  1989.**  The bottom fauna from three subtidal locations around Banks Peninsula, Canterbury, New Zealand.  *New Zealand Natural Sciences*.  16: 87-95.

**de Goeij, P. & P. J. C. Honkoop  2002.**  The effect of immersion time on burying depth of the bivalve *Macoma balthica* (Tellinidae).  *Journal of Sea Research*.  47(2): 109-119.

**Dickie, B. N.  1986.**  The Whangarei Water Quality Management Plan, Working Report No. 4.  Unpubl. report of the Northland Catchment Commission and Regional Water Board.

**Dobbinson, S. J., M. F. Barker & J. B. Jillett  1989.**  Experimental shore level transplantation of the New Zealand cockle *Chione stutchburyi*.  *Journal of Shellfish Research*.  8(1): 197-212.

**Evans, D. L., C. C. Lautenbacher Jnr. & W. T. Hogarth  2002.**  Molluscan shellfish research and management: Charting a course for the future.  Preliminary proceedings from the Workshop, Charleston, South Carolina, January 2000.  Department of Commerce, USA.  156pp.

**Gardner, M.  1992.**  Use of stress responses of *Austrovenus stutchburyi* for biomonitoring in the Manukau Harbour, New Zealand.  Unpubl. M.Sc. thesis, University of Auckland.  146pp.

**Gibbs, M., S. Thrush, J. Sukias, K. Berkenbusch, D. Thompson & S. Bury  2001.**  Evaluation of terrestrial sediment spatial patterns throughout the Whitford embayment and its benthic food web using stable isotope techniques.  Report prepared for the Auckland Regional Council.  NIWA Client Report ARC01270.

**Gosselin, L. A. & P. –Y. Qian  1997.**  Juvenile mortality in benthic marine invertebrates.  *Marine Ecology Progress Series*.  146: 265-282.

**Grace, R. V.  1973.**  The benthic ecology of the entrance to the Whangateau Harbour, Northland, New Zealand, Ph.D. thesis, Department of Zoology, University of Auckland.

**Grace, R. V. & A. B. Grace  1978.**  Mimiwhangata Marine Monitoring Programme: report on progress 1976-1978.  Volume 1.  Unpublished Report for Lion Breweries Ltd. Mimiwhangata Trust.  22pp.

**Grant, C. M.  1994.**  Demographics and reproduction of the tuatua – *Paphies subtriangulata*.  Unpubl. M.Sc. thesis, Zoology Department, University of Auckland.

**Grant, C. M. & R. G. Creese  1995**.  The reproductive cycle of the tuatua- *Paphies subtriangulata* (Wood, 1828), in New Zealand.  *Journal of Shellfish Research*.  14(2): 287-292.

**Greenway, J. 1981.**  Dargaville tuatua.  Catch '81.  (June).

**Honkoop, P. J. C. & J. Van Der Meer  1998.**  Experimentally induced effects of water temperature and immersion time on reproductive output of bivalves in the Wadden Sea.  *Journal of Experimental Marine Biology and Ecology*.  220(2): 227-246.

**Hooker, S. H. 1995.**  Life history and demography of the pipi – *Paphies australis* (Bivalvia: Mesodesmatidae) in northeastern New Zealand.  Ph.D. thesis, School of Biological Sciences, University of Auckland.  230pp.

**Hooker, S. H. & R. G. Creese  1995.**  The reproductive biology of the pipi, *Paphies australis* (Gmelin, 1790) (Bivalvia: Mesodesmatidae). I. Temporal patterns of the reproductive cycle.  *Journal of Shellfish Research*.  14(1): 7-15.

**Hull, P. J.  1996.**  The ecology of bivalves on Centre Bank, Tauranga Harbour.  Unpubl. M.Sc. thesis.  University of Waikato, Hamilton, New Zealand.

**Kearney, M. B.  1999.**  Ecology and management of *Austrovenus stutchburyi* in the Whangateau Harbour.  Unpubl. M.Sc. thesis.  School of Environmental and Marine Science.  135pp.

**Larcombe, M. F.  1971.**  The ecology, population dynamics, and energetics of some soft shore molluscs.  Unpubl. Ph.D. thesis, University of Auckland.

**LeLieveld, S. D. 2000.** Effects of biota on intertidal sediment stability. Unpubl. M.Sc. thesis. Department of Earth Science. University of Waikato, Hamilton, New Zealand. 95pp.

**Lidgard, C. W. 2001.** Evaluating the population genetic structure of the New Zealand cockle (*Austrovenus stutchburyi*) using allozyme electrophoresis. Unpubl. M.Sc. thesis. Department of Biological Sciences. University of Waikato, Hamilton, New Zealand. 46pp.

**Malinowski, S. & R. B. Whitlatch 1988.** A theoretical evaluation of shellfish resource management. *Journal of Shellfish Research*. 7(1): 95-100.

**Marsden, I. D. 1999.** Reproductive cycles of the surf beach clam *Paphies donacina* (Spengler, 1793) from New Zealand. *Journal of Shellfish Research*. 18(2): 539-546.

**Marsden, I. D. 2000.** Variability in low tide populations of tuatua, *Paphies donacina*, in Pegasus Bay, Canterbury, New Zealand. *New Zealand Journal of Marine and Freshwater Research*. 34: 359-370.

**Marsden, I. D. & R. M. Pilkington 1995.** Spatial and temporal variations in the condition of *Austrovenus stutchburyi* Finlay, 1927 (Bivalvia: Veneridae) from the Avon-Heathcote Estuary, Christchurch. *New Zealand Natural Sciences*. 22:57-67.

**McLachlan, A. & T. Erasmus 1983.** Sandy Beaches as Ecosystems. Dr. W. Junk Publishers, The Hague.

**Morton, J. & M. Miller 1973.** The New Zealand seashore. 2nd Edition. Collins, Auckland. 653pp.

**Norkko, A., V. J. Cummings, S. F. Thrush, J. E. Hewitt & T. Hume 2001.** Local dispersal of juvenile bivalves: Implications for sandflat ecology. *Marine Ecology Progress Series*. 212: 131-144.

**Olafsson, E. B. 1986.** Density dependence in suspension-feeding and deposit-feeding populations of the bivalve *Macoma balthica*: a field experiment. *Journal of Animal Ecology*. 55: 517-526.

**Parr, S. W. 1994.** Bivalve larvae in the Whangateau Harbour. Unpubl. MSc. thesis. Zoology Department. University of Auckland, New Zealand.

**Pawley, M. D. M., G. N. Browne & M. Morrison 1997.** Survey of intertidal shellfish populations in the Auckland metropolitan area (1996/1997) with a summary of previous sampling methodologies and results (1992-1996). New Zealand Fisheries Assessment Research Document Draft for Ministry of Fisheries.

**Pilditch, C. A., C. Lundquist & V. J. Cummings 2003.** Juvenile bivalve dispersal: Do they go with the flow? Oral presentation at New Zealand Marine Sciences Society Annual Conference, Auckland University, New Zealand, Sept 2003.

**Pilkington, R. M. 1992.** Spatial and temporal variation in the growth and condition of *Austrovenus stutchburyi* in the Avon-Heathcote Estuary. Unpubl. M.Sc. thesis, University of Canterbury, Christchurch, New Zealand. 99pp.

**Powell, A. W. B. 1979.** New Zealand mollusca: marine, land and freshwater shells. Collins, Auckland, New Zealand. 500pp.

**Pridmore, R. D., S. F. Thrush, R. J. Wilcock, T. J. Smith, J. E. Hewitt & V. J. Cummings 1991.** Effect of the organochlorine pesticide technical chlordane on the population structure of suspension and deposit feeding bivalves. *Marine Ecology Progress Series*. 76: 261-271.

**Redfearn, P. 1987.** Larval shell development of the northern tuatua, *Paphies subtriangulata* (Bivalvia, Mesodesmatidae). *New Zealand Journal of Marine and Freshwater Research*. 21: 65-70.

**Roper, D. 1990.** Benthos associated with an estuarine outfall, Tauranga Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research*. 24: 487-498.

**Roper, D. S., R. D. Pridmore & S. F. Thrush 1992.** Recruitment to the macrobenthos of *Macomona liliana* (Bivalvia: Tellinidae) in Manukau Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research*. 26: 385-392.

**Roughgarden, J., S. D. Gaines & S. W. Pacala 1987.** Supply side ecology: the role of physical transport processes. *In*: Organisation of communities past and present. P. Giller and J. Gee (Eds.). Blackwell Scientific, Oxford, U.K. Pp. 459-486.

**Richardson, J. R., A. E. Aldridge & P. J. Smith 1982.** Analysis of tuatua populations – *Paphies subtriangulata* and *P. donacina*. *New Zealand Journal of Zoology*. 9: 231-238.

**Sigurdsson, J. B., C. W. Titman & P. A. Davies 1976.** The dispersal of young post-larval bivalve molluscs by byssus threads. Nature, Lond. 262: 386-387.

**Sissenwine, M. P., M. J. Fogarty & W. J. Overholtz 1988.** Some fisheries management implications of recruitment variability. *In*: Fish Population Dynamics, 2nd Edn. J. A. Gulland (Ed.). Wiley, Chichester. Pp. 129-152.

**Stephenson, R. L. 1981.** Aspects of the energetics of the cockle *Chione* (*Austrovenus*) *stutchburyi* in the Avon-Heathcote estuary, Christchurch, New Zealand. Unpubl. Ph.D. thesis, University of Canterbury, Christchurch, New Zealand.

**Stephenson, R. L. & P. E. Chanley 1979.** Larval development of the cockle *Chione stutchburyi* (Bivalvia: Veneridae) reared in the laboratory. *New Zealand Journal of Zoology*. 6: 553-560.

**Stewart, M. 2002.** Impacts of urban development on cockle populations. Interim progress report to the Auckland Regional Council. July 2002. 23pp.

**Stewart, M. J. & R. G. Creese 2002.** Transplants of intertidal shellfish for enhancement of depleted populations: Preliminary trials with the New Zealand little neck clam. *Journal of Shellfish Research*. 21(1): 21-27.

**Taylor, M. D.  1998.**  The influence of environmental factors on reproduction and recruitment of *Macomona liliana* (Class: Bivalvia) in Manukau Harbour, New Zealand.  Unpubl. Ph.D thesis.  School of Biological Sciences.  University of Auckland.  304pp.

**Thrush, S. F., R. D. Pridmore, J. E. Hewitt & V. J. Cummings  1992.**  Adult infauna as facilitators of colonisation on intertidal sandflats.  *Journal of Experimental Marine Biology and Ecology*.  159: 253-265.

**Thrush, S. F., J. E Hewitt, V. J. Cummings, M. O. Green, G. A. Funnell & M. R. Wilkinson  2000.**  The generality of field experiments: Interactions between local and broad-scale processes.  *Ecology*.  81(2): 399-415.

**Turner, S. J., J. Grant, R. D. Pridmore, J. E. Hewitt, M. R. Wilkinson, T. M. Hume & D. J. Morrisey 1997.**  Bedload and water-column transport and colonisation processes by post-settlement benthic macrofauna: Does infaunal density matter?  *Journal of Experimental Marine Biology and Ecology*.  216: 51-75.

**Underwood, A. J. & P. G. Fairweather  1989.**  Supply-side ecology and benthic marine assemblages.  *Trends in Ecology and Evolution*.  4: 16-20.

**Venus, G. C.  1984.**  *Paphies australis* (pipi) in Whangarei Harbour.  Whangarei Harbour Study Technical Report No. 6.  Northland Catchment Commission.

**Wilcock, R. J., R. D. Pridmore, G. L. Northcott, J. E. Hewitt, S. F. Thrush & V. J. Cummings  1994.**  Uptake of chlordane by a deposit-feeding bivalve: Does the depth of sediment contamination make a difference?  *Environmental Toxicology and Chemistry*. 13(9): 1535-1541.

**Zajac, R. N. & R. B. Whitlatch  1985.**  A hierarchical approach to modelling soft-bottom successional dynamics.  *In*: Proceedings of the 19th European Marine Biological Symposium.  P. Gibbs (Ed.).  Cambridge University Press, Cambridge.  Pp. 265-276.

## *SECTION 3: COMMUNITY LEVEL ECOLOGICAL PROCESSES: COMPETITION AND PREDATION*

## 3.1 INTRODUCTION

The ecological processes of competition and predation are community level interactions which produce some of the major forces of natural selection. Although they are naturally occurring processes within benthic communities and infaunal bivalve populations, they are akin to some of the stressors discussed in the following sections in that anthropogenic influences can alter the degree of impact of these interactions at a population level.

## 3.2 COMPETITION

Competition is an interaction which occurs between individuals of the same species (i.e. intraspecific competition) or of different species (i.e. interspecific competition) using the same resource, when the resource is present in <u>limited</u> supply (Curtis & Barnes 1989). Thus, competitive interactions will become more intense or numerous as resource limitation increases. The impact of competition is that the overall fitness/reproductive success of the interacting individuals may be reduced (Curtis & Barnes 1989). Infaunal bivalve species may experience competition for potentially limited resources such as food (phytoplankton), dissolved oxygen (which may vary due to increased eutrophication), and habitat (e.g. through habitat modification/destruction, or competition with invasive species for existing habitat). Physical conditions (such as temperature, salinity etc.) resource availability, and bivalve population densities all very seasonally, and from year to year, and as a consequence the intensity of competition may vary (Curtis & Barnes 1989). Competition for various <u>limited</u> resources may potentially occur at several different bivalve life cycle stages (for example, competition for settlement habitat by larval bivalves). Roper *et al.* (1992) suggested that the presence of other invertebrates within inter-tidal sandflats may physically restrict the burrowing and feeding activities of young bivalves, and may also create competition for food resources. However, due to a paucity of information on their ecology in the natural environment, it is difficult to estimate the level of interspecific competition for larval and post-settlement stages of the infaunal bivalves of interest in this review. However, the relatively similar timing of spawning periods of these infaunal bivalves does suggest that there is potential for larval competition to exist, if food or settlement habitat is a limited resource.

Competition is generally hypothesised to be greatest amongst organisms that have similar requirements and lifestyles (Curtis & Barnes 1989). Levinton (1972) proposed that different modes of feeding (i.e. deposit- versus suspension-feeding) result in different degrees of food limitation and thus competition for food. On a local scale, he suggested that food would be more limiting to deposit-feeders than to suspension-feeders (suggesting that for bivalves in the Hauraki Gulf Marine Park food would be more limiting to *M. liliana* than to *A. stutchburyi*, *P. australis* or *P. subtriangulata*).

Due to known adult niche habitat specialisation (see Section 2), interspecific competition between the four infaunal species of interest in this review can be approximated. It is highly unlikely that interspecific competition for any resources will occur between adult *P. subtriangulata* and *M. liliana* or *A. stutchburyi*, due to vastly different adult habitat preferences. Interspecific competition for limited resources may potentially occur between adult *P. subtriangulata* and *P. australis* at the mouths of estuaries opening onto exposed sandy beaches, where adult populations may co-occur (e.g. Whangateau Harbour/Omaha Beach, Dr. C. Grant, AquaBio Consultants Ltd., *pers. obs.*). In

areas with spatially overlapping populations of *A. stutchburyi*, *P. australis*, and *M. liliana*, interspecific competition may also occur for limited resources. Based on the associations of *A. stutchburyi*, *P. australis* and *M. liliana* at 11 Auckland beaches, Pawley *et al.* (1997) suggested that beaches contained either predominantly populations of *A. stutchburyi* and *M. liliana*, or *P. australis*, (although some beaches did contain associations intermediate between these two) suggesting different habitat requirements for *A. stutchburyi* and *M. liliana* compared to *P. australis*. This observation tends to indicate that the infaunal bivalves *A. stutchburyi* and *M. liliana* may be more likely to compete for any limited resources. It seems unlikely that these bivalves will compete for food resources as they have different feeding modes (e.g. suspension- vs. deposit-feeding). However, competition may occur when space is limited, for example, Morton and Miller (1973) suggested that the presence of high-density *A. stutchburyi* beds may prevent *M. liliana* from feeding due to the difficulties of extending their siphons through tightly packed *A. stutchburyi* beds. However, direct investigation of the impact of interspecific competition between the infaunal bivalve species of interest here requires further investigation.

Dobbinson *et al.* (1989) suggested that it was possible that intraspecific competition for food and space begins in the range between 2000 – 3200 *A. stutchburyi* per square metre. Overseas studies have identified decreasing growth rates with increasing bivalve density for both suspension- and deposit-feeders (Kamermans *et al.* 1992) suggesting intraspecific competition for available food resources. Olafsson (1986) and Kamermans *et al.* (1992) found a clear decline in growth when density of deposit-feeding *Macoma balthica* was experimentally increased. Increased density was found to impact negatively on the growth of the venerid clam *Cerastoderma edule* (Jensen 1992), but only when very high densities (>2700 m$^2$) occurred over an extensive area (i.e. km$^2$). A study by Martin (1984) showed that an experimental decrease in density of *A. stutchburyi* brought about by removal of exploitable size class shellfish resulted in an increased growth rate. His study also suggested that the lower growth rates at ambient densities were due to some biotic factor (such as food availability) rather than competition for space (Martin 1984). An overseas study of interspecific competition of mixed populations of the infaunal bivalves *(M. balthica* and *C. edule*) by Kamermans *et al.* (1992) identified no influence of increased *M. balthica* density on growth and survival of *C. edule*. On the other hand, a negative impact of very high *C. edule* density on *M. balthica* was observed in the form of a slightly decreased condition index.

Competition between infaunal bivalve shellfish for limited resources (at various spatial and temporal scales) is undoubtedly a common natural process occurring within the Hauraki Gulf Marine Park. The level to which infaunal bivalve competition is influenced by anthropogenic processes which may intensify resource limitation within the Hauraki Gulf Marine Park is impossible to estimate with our current knowledge of both potential stressors and competitive interactions between these bivalve species.

## 3.3   PREDATION

Predation is the eating of live organisms (Curtis & Barnes 1989). Predation is an interaction which may occur between individuals of the same species (i.e. intraspecific predation) or of different species (i.e. interspecific predation). Predation may occur at a lethal or sublethal level. When intense, predation can affect cockle populations, especially if predators such as flatfish or crabs feed on juvenile cockles (Dobbinson *et al.* 1989). High levels of predation pressure (by shrimps and crabs) have also been apportioned blame overseas for poor recruitment in the Tellinid bivalve *Macoma balthica* (Flach 2003). It is also pertinent to note that in a theoretical evaluation of shellfish resource management by Malinowski & Whitlatch (1988), it was suggested that predator

control of juvenile clams *Mya arenaria* and *Mercenaria mercenaria* has the greatest potential to significantly increase the maximum sustainable harvest of these species.

Predators of infaunal inter-tidal bivalve populations in New Zealand include: shorebirds (variable oystercatcher - *Haematopus unicolour*; the New Zealand dotterel – *Charadrius obscurus*; black-backed and red-billed gulls – *Larus dominicanus* and *L. scopulinius*), eagle rays (*Myliobatis tenuicaudatus*), paddle crabs (*Ovalipes catharus*), whelks (*Cominella* spp.), starfish (*Cosinasterias calamaria* and *Patiriella regularis*), fish (especially flat fish) and humans. Shorebirds have an important impact on inter-tidal bivalve shellfish because they can focus feeding on high density patches of prey (Sutherland 1982).

Because of its abundance and large size the paddle crab (*Ovalipes catharus*) is possibly the most significant invertebrate predator in the inter-tidal/shallow subtidal region of sandy beaches (Wear & Haddon 1987). Among other components, the paddle crab diet may contain cockles, pipi and tuatua. However studies suggest that *Paphies* species make up the highest proportion of the Molluscan part of the paddle crab diet (Wear & Haddon 1987). *Ovalipes* usually consumes all bivalve species of a very small size, generally < 3-4 mm shell length. This suggests that bivalve recruitment may be influenced. Wear & Hadden (1987) suggest that *O. catharus* probably has little direct effect on established beds of mature bivalve species, as adult *P. australis* and *P. subtriangulata* (> 40 mm shell length) are resistant to crushing, and adult *A. stutchburyi* may be protected by their thick and highly inflated shells.

A study by Ansell (2001) on a muddy inter-tidal New Zealand sandflat with approximately 7 hrs tidal cover suggested that predation levels of *M. liliana* and *A. stutchburyi* by *Cominella glandiformis* were approximately 2-16 % and 3-9 % per year respectively (Ansell 2001). These figures may be an underestimate as, although attractant chemical cues may be less available during high tide periods, predation may also occur during these periods (Ansell 2001). Predation was more commonly observed on intact bivalves rather than those with damaged shells. These figures indicate that *C. glandiformis* is associated with significant rates of natural mortality in both bivalve species. Ansell (2001) suggests that the shallow burrowing of *A. stutchburyi* (in contrast to that of *M. liliana*) may mean they are more readily exhumed during strong tides or storms, leaving them exposed to epifaunal predators.

New Zealand eagle rays feed by pumping water from their mouth to liquefy sediment from which they consume infaunal bivalves (Gregory *et al.* 1979). The intensity of ray predation can be indicated by their sediment disturbance rate of approximately 10 $m^2$ of sandflat per day (Thrush *et al.* 1991). In predator exclusion experiments, Thrush *et al.* (1994) identified a direct negative effect of predators on large (> 8 mm) *Macomona liliana*. Bird exclusion appeared to have the greatest effect when recently recruited bivalves were abundant and rays were absent (over winter). Hines *et al.* (1997) found that eagle ray predation on infaunal bivalves was best described by a nonlinear segmented model. In this model, eagle rays maintained a constant feeding level at low *M. liliana* densities, and rapidly increased their feeding effort above a threshold bivalve density level (Hines *et al.* 1997).

The removal of small planktonic larvae from the water column or the sediment surface may occur as a result of suspension- and deposit-feeding by infaunal bivalves. Whitlatch and co-workers (1997) identified that very high densities (i.e. > 600 $m^2$) of adult *A. stutchburyi* (i.e. 15-30 mm shell length) could cause a reduction in the abundance and size-structure of post-settlement (< 1 mm) *M. liliana*. Whitlatch *et al.* (1997) suggested that this most likely reflected the ingestion of larval *M. liliana* by *A. stutchburyi*.

Sublethal predation is a widespread phenomenon in marine soft-bottom communities (Mouritsen & Poulin 2003). Mouritsen & Poulin (2003) demonstrated that approximately 21% of the *A. stutchburyi* population in Otago Harbour undergo foot-cropping, with approximately 13% of the foot tissue lost to cropping. No difference in the risk of foot-cropping was found with respect to age class (Mouritsen & Poulin 2003). The identity of the foot-cropper is currently unknown but flatfish known for their foot-cropping ability are common on these sandflats. Bioturbation and parasite load are two factors which may cause *A. stutchburyi* to surface more frequently, and thus may influence the level of foot-cropping in *A. stutchburyi* populations (Mouritsen & Poulin 2003). Sublethal predation *via* siphon-nipping by snapper (*Pagrus auratus*) occurs in *P. subtriangulata* and *P. australis* populations (Dr. C. Grant, AquaBio Consultants Ltd, *pers. obs*.) and by flounder (*Rhombosolea* spp.) for *A. stutchburyi* populations (Whitlatch *et al.* 1997). The deeper burial depth of adult *M. liliana* within the sediment, in comparison to the other infaunal bivalve species of interest here, may make them less susceptible to several types of predation.

Nonindigenous species are increasingly being recognised for their impacts on marine and estuarine communities, causing significant changes in abundance and distribution of native species (Walton *et al.* 2002). Walton *et al.* (2002) identified that the relative predation intensity by an introduced crab species (*Carcinus maenas*) on an inter-tidal venerid clam (*Katelysia scalarina*) was higher for juveniles (< 13 mm shell length), and that unlike a native shore crab, *Carcinus maenas* it increased its predation rate with increasing clam density. Walton *et al.* (2002) suggest that this increased *per capita* predation rate could reduce the frequency of good recruitment years.

## 3.4    REFERENCES

**Ansell, A. D.  2001.**  Dynamics of aggregations of a gastropod predator/scavenger on a New Zealand harbour beach. *Journal of Molluscan Studies*.  67: 329-341.

**Curtis, H. & N. S. Barnes  1989.**  Part 3: Biology of Populations.  *In*: Biology, Fifth Edition.  Worth Publishers Inc., New York, U.S.A.  Pp. 1106-1114.

**Dobbinson, S. J., M. F. Barker & J. B. Jillett  1989.**  Experimental shore level transplantation of the New Zealand cockle *Chione stutchburyi*. *Journal of Shellfish Research*.  8(1): 197-212.

**Flach, E. C.  2003.**  The separate and combined effects of epibenthic predation and presence of macro-infauna on the recruitment success of bivalves in shallow soft-bottom areas on the Swedish west coast.  *Journal of Sea Research*.  49: 59-67.

**Gregory, M. R., P. F. Balance, G. W. Gibson & A. M. Ayling  1979.**  On how some rays (Elasmobranchia) excavate feeding depressions by jetting water. *Journal of Sedimentation and Petrology*.  49: 1125-1130.

**Hines, A. H., R. B. Whitlatch, S. F. Thrush, J. E. Hewitt, V. J. Cummings, P. K. Dayton & P. Legendre  1997.**  Nonlinear foraging response of a large marine predator to benthic prey: eagle ray pits and bivalves in a New Zealand sandflat. *Journal of Experimental Marine Biology and Ecology*.  216: 191-210.

**Jensen, K. T.  1992.**  Dynamics and growth of the cockle, *Cerastoderma edule*, on an intertidal mud-flat in the Danish Wadden Sea: effects of submersion time and density. *Netherlands Journal of Sea Research*.  28: 335-345.

**Kamermans, P., H. W. van der Veer, L. Karczmarski & G. W. Doeglas  1992.**  Competition in deposit- and suspension-feeding bivalves: experiments in controlled outdoor environments.  *Journal of Experimental Marine Biology and Ecology*.  162: 113-135.

**Levinton, J.  1972.**  Stability and trophic structure in deposit-feeding and suspension-feeding communities.  *American Naturalist*.  106: 472-478.

**Malinowski, S.  &  R. B. Whitlatch  1988.**  A theoretical evaluation of shellfish resource management.  *Journal of Shellfish Research.* 7: 95-100.

**Martin, N. D.  1984.**  *Chione (Austrovenus) stutchburyi* (Gray): Population responses to exploitation.  Upubl. M.Sc. Thesis, University of Auckland, New Zealand.  75pp.

**Morton, J. & M. Miller  1973.**  The New Zealand seashore.  2nd Edition.  Collins, Auckland.  653pp.

**Mouritsen, K. N. & R. Poulin  2003.**  The risk of being at the top: foot-cropping in the New Zealand cockle *Austrovenus stutchburyi*. *Journal of the Marine Biological Association (U.K.)*.  83: 497-498.

**Olafsson, E. B.  1986.**  Density dependence in suspension-feeding and deposit-feeding populations of the bivalve *Macoma balthica*: a field experiment. *Journal of Animal Ecology*.  55: 517-526.

**Pawley, M. D. M., G. N. Browne & M. Morrison  1997.**  Survey of intertidal shellfish populations in the Auckland metropolitan area (1996/1997) with a summary of previous sampling methodologies and results (1992-1996). New Zealand Fisheries Assessment Research Document Draft for Ministry of Fisheries.

**Roper, D. S., R. D. Pridmore & S. F. Thrush  1992.**  Recruitment to the macrobenthos of *Macomona liliana* (Bivalvia: Tellinidae) in the Manukau Harbour, New Zealand.  *New Zealand Journal of Marine and Freshwater Research*.  26:385-392.

**Sutherland, W. J. 1982.**  Spatial variation in the predation on cockles by oystercatchers at Traeth Malynog, Anglesey. II. The patterns of mortality. *Journal of Animal Ecology*.  51:491-500.

**Thrush, S. F., R. D. Pridmore, J. E. Hewitt & V. J. Cummings  1991.**  Impact of ray feeding disturbances on sandflat macrobenthos: do communities dominated by polychaetes or shellfish respond differently.  *Marine Ecology Progress Series*.  69: 245-252.

**Thrush, S. F., R. D. Pridmore, J. E. Hewitt & V. J. Cummings  1994.**  The importance of predators on a sandflat: interplay between seasonal changes in prey densities and predator effects.  *Marine Ecology Progress Series*. 107: 211-222.

**Walton, W. C., C. MacKinnon, L. F. Rodriguez, C. Proctor & G. M. Ruiz  2002.**  Effect of an invasive crab upon a marine fishery: green crab, *Carcinus maenas*, predation upon a venerid clam *Katelysia scalarina*, in Tasmania (Australia). *Journal of Experimental Marine Biology and Ecology*.  272(2): 171-189.

**Wear, R. G. & M. Haddon  1987.**  Natural diet of the crab *Ovalipes catharus* (Crustacea, Portunidae) around central and northern New Zealand. *Marine Ecology Progress Series*.  35: 39-49.

**Whitlatch, R. B., A. H. Hines, S. F. Thrush, J. E. Hewitt & V. J. Cummings  1997.**  Benthic faunal responses to variations in patch density and patch size of a suspension-feeding bivalve.  *Journal of Experimental Marine Biology and Ecology*.  216: 171-189.

## SECTION 4:   STATUS OF BIVALVE POPULATIONS IN THE HAURAKI GULF MARINE PARK

## 4.1   INTRODUCTION

There is a general perception both amongst members of the Hauraki Gulf Forum and the general public that the abundance of infaunal inter-tidal shellfish in the Hauraki Gulf Marine Park is decreasing.  This section of our report reviews data that have been collected over time regarding the status of these shellfish populations to identify the scientific basis for this perception.

Several beaches in the Auckland region have been closed to shellfish harvesting due to perceived declines in shellfish numbers, or due to changes in their legal status resulting in a no-take marine reserve.  Cheltenham and Eastern Beaches have been closed to all harvesting since 1993 and 1994 respectively.  Long Bay-Okura Marine Reserve was closed to all harvesting in 1995.  Waikawau Beach on the western side of the Coromandel Peninsula was closed to all harvesting in 2000 (as part of the Wilson to Ngarimu Bay closure).  We note that some beaches that have now been closed to shellfish harvesting for 10 years (e.g. Cheltenham and Eastern Beach) still have shellfish densities well below anecdotal historical levels.

The following section outlines the results of abundance surveys of infaunal bivalves undertaken in the Hauraki Gulf Marine Park region, specifically those that provide data on populations of cockles (*Austrovenus stutchburyi*), pipi (*Paphies australis*), tuatua (*Paphies subtriangulata*) and wedge shells (*Macomona liliana*).   The review includes surveys that have investigated population characteristics on an on-going basis over several years (such as the Ministry of Fisheries "Inter-tidal Shellfish Monitoring Programme", Cheltenham Beach Caretakers cockle (*A. stutchburyi*) monitoring programme, and various programmes funded by the Auckland Regional Council (ARC) such as the Mahurangi Estuary Ecological Monitoring Programme etc.).  We have also attempted to identify some of the baseline data provided by one-off studies (e.g. research undertaken for University theses).  We acknowledge that in the latter case there may be further historic data collected as part of wider ecological studies of coastal areas in the region that have not been identified in our literature review process.  We are also aware of data that has been collected that we have not been able to access − for example, although the Ministry of Fisheries runs the "Adopt a Beach programme" there is currently no funding to develop a database for these results (B. Drey, MFish, Auckland, *pers. comm.*).   In addition, data on the distribution of infaunal inter-tidal shellfish, arising from the Auckland Regional Council-funded surveys of inter-tidal seafood resources undertaken by Bioresearches Ltd in the 1990's, have not been specifically included in this review, since these data are embargoed in order to protect the resources.

## 4.2   MINISTRY OF FISHERIES INFAUNAL SHELLFISH SURVEYS (Including data from other surveys at the same sites)

In 1992 the Ministry of Fisheries established an "*Inter-tidal Shellfish Monitoring Programme*" in the greater Auckland Metropolitan Area (i.e. including an area of the East coast within the Hauraki Gulf Marine Park from Wenderholm down to Kawakawa Bay).  The aim of this programme was to assess the depletion of inter-tidal shellfish populations through regular surveys (Akroyd *et al.* 2000).  The inter-tidal area is defined as "*all soft shore habitat above mean low water spring – MLWS*".  Initially both hard and soft shores were sampled, but since 1994 sampling has focused solely on inter-tidal soft shores.  This programme has had no fixed experimental design or assigned treatments (e.g. heavily vs. lightly exploited beaches), and beaches were selected on a fairly *ad hoc*

basis (Pawley *et al.* 1997), although the initial survey by Cook *et al.* (1994) did try to pair unexploited with exploited sites where possible.

In 1999, 2000 and 2001 the geographical area included in the survey was extended to include sites from throughout the Auckland Fisheries Management Area, which includes Northland, Waikato, Coromandel and the Bay of Plenty. These additional sites included shellfish populations considered to be heavily harvested by humans. As only 10 or 12 beaches are generally surveyed each year, it is not clear which combination of these beaches will be surveyed in forthcoming survey years.

Table 4.1 outlines the frequency of sampling for beaches found within the Hauraki Gulf Marine Park that have been sampled in the MFish Inter-tidal Shellfish Monitoring Programme, between 1992 and 2002.

It is clear from Table 4.1 that several of the beaches included in this programme have been sampled less than four times (e.g. Whangateau Harbour and Waikawau Beach (only recently added), Long Bay, Te Haruhi, St Heliers Beach, Bucklands Beach, Omana, Maraetai, Kauri Bay, Kawakawa Bay, Tairua Harbour and Whangamata Harbour). It is thus inappropriate to try and make any clear statements about population trends at many of these beaches, especially for beaches that have not been sampled since 1992/93. However, data from Tairua Harbour and Whangamata Harbour will be discussed further as the level of stock information for infaunal bivalves from the Coromandel area is generally low. However, it should be noted that Waikawau Beach has been closed to human harvesting of shellfish due to local Maori concern at the depletion of shellfish beds.

The following discussion of these MFish surveys only includes sites that are found within the Hauraki Gulf Marine Park that have been sampled 4 or more times (with the exception of Tairua and Whangamata) since 1992, namely:

- Wenderholm, Okoromai Bay, Cheltenham Beach, Eastern Beach, Howick Beach, Beachlands, Umupuia, Tairua Harbour and Whangamata Harbour.

While the overall focus of the survey has remained on estimating the distribution, abundance, and size structure of inter-tidal bivalve populations potentially impacted by human harvesting, several considerations are pertinent to interpretation of long-term data collected in these surveys. Over the period of the survey, the sampling methodology has undergone several transformations. These include changes in quadrat size and/or sieve size used, changes in sampling strategies (e.g. random *versus* stratified sampling), changes in the total area of each individual beach sampled, changes in types of data reported etc. The changes in sampling methodology and changes to data reporting formats, along with errors in transcribing correct data values (both within and between survey reports), make it very difficult to identify meaningful temporal trends in inter-tidal bivalve species abundance within the Hauraki Gulf Marine Park from these data.

**Table 4.1: Table summarising the frequency of sampling for beaches sampled in the MFish Inter-tidal Shellfish Monitoring Programme (1992-2002). Beaches highlighted in red are those currently closed to shellfish harvesting.**

| | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 | Species Present |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Whangateau Harbour | | | | | | | | | | X | | Cockle, pipi |
| Wenderholm | | | X | X | X | X | | | | | | Pipi, wedge, tuatua |
| **Long Bay** | | X | X | X | | | | | | | | Tuatua |
| Okoromai Bay | | X | | | X | X | X | | | | X | Cockle, pipi, wedge |
| Te Haruhi | | | | | | X | | | | X | | Cockle, pipi , wedge |
| **Cheltenham Beach** | | X | X | X | X | X | | | | | | Cockle , pipi, wedge |
| St Heliers Beach | X | | | | | | | | | | | Few cockle, pipi, wedge |
| Bucklands Beach | | | X | X | X | | | | | | | Cockle, pipi, |
| **Eastern Beach** | | X | X | X | X | X | X | | | | X | Cockle, pipi, wedge |
| Howick Beach | | | X | X | X | X | X | | | | | Cockle, pipi, wedge |
| Beachlands | | | | | X | X | X | | | | | Cockle, pipi, wedge |
| Omana | | | | | X | | | | | | | Cockle, pipi |
| Maraetai | | | | | X | X | | | | | | Cockle |
| Umupuia | | | | | | X | X | X | X | X | X | Cockle, pipi, wedge |
| Kauri Bay | X | | | | | | | | | | | Cockle, few pipi, wedge (no public access) |
| Kawakawa Bay | X | X | | | | | | | | | | Cockle, few pipi, wedge |
| **Waikawau Beach** | | | | | | | | | X | X | | Cockle, pipi |
| Tairua Harbour | | | | | | | | | X | | X | Cockle, pipi (subtidal) |
| Whangamata Harbour | | | | | | | | | X | X | X | Cockle, pipi (subtidal) |

Another confounding factor when trying to investigate trends in particular species population dynamics is that for different years of the survey different combinations of species were included as target species for investigation. A summary of the inter-tidal bivalve species specifically identified for inclusion in each fishing year survey of the MFish *Inter-tidal Shellfish Monitoring Programme* is provided in Table 4.2 .

**Table 4.2: Summary of the inter-tidal bivalve species specifically identified for inclusion in each survey of the MFish Inter-tidal Shellfish Monitoring Programme (1992-2002).**

| Species Sampled in MFish Surveys: | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| *A. stutchburyi* | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ |
| *P. australis* | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ |
| *P. subtriangulata* | | | | ☑ | ☑ | ☑ | ☑ | | ☑ | ☑ | |
| *M. liliana* | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | ☑ | | | | |

Table 4.2 indicates that the MFish *Inter-tidal Shellfish Monitoring Programme* provides a consistent data series through time for inter-tidal *A. stutchburyi* and *P. australis* populations. Data for *P. subtriangulata* have been collected for only half the years sampled, and *M. liliana* data have not been collected since 1998.

While *P. australis* have been consistently surveyed, the data collected with respect to size distribution and abundance should be interpreted with caution due to the exclusively inter-tidal location of the surveys. Pawley *et al.* (1997) suggested that some beaches may only ever hold recruits, and that movement of these pipi from surveyed beaches may occur into other areas. As discussed previously (see Section 2 of this report) adult pipi may be found predominantly in sub-tidal locations, so the data obtained from inter-tidal sampling may only be representative of the inter-tidal portion of the population. Walshe & Akroyd (2002) noted this sampling bias and observed that in the 1999 and 2000 MFish surveys most *P. australis* were found in shallow sub-tidal areas rather than in inter-tidal areas. The extent to which data from inter-tidal surveys are representative of overall population abundance trends in *P. australis* populations with potentially substantial subtidal adult stocks is also therefore questionable. If migration of *P. australis* occurs from inter-tidal to adult sub-tidal beds, then this should be quantified as it may influence the population estimates obtained in the inter-tidal surveys.

Similarly, the status of *P. subtriangulata* stocks in the Hauraki Gulf Marine Park is not effectively represented by the location of survey areas chosen in the Ministry of Fisheries inter-tidal shellfish monitoring programme. The majority of beaches chosen for these shellfish surveys are enclosed, moderate wave energy, mud/sand environments. These environments are not those favoured by P. *subtriangulata* populations, which are generally found on open exposed sandy beaches (Morton & Miller 1973). The dearth of *P. subtriangulata* found to date in these surveys (Walshe *et al.* 2002) simply highlights the inability of sites chosen to reflect normal *P. subtriangulata* habitat.

*Paphies subtriangulata* have been recorded at two beaches surveyed in the Ministry of Fisheries surveys in the Hauraki Gulf Marine Park: at Long Bay, where a small population of 40-60 mm adults was recorded from 1993 to 1995, (Pawley *et al.* 1997), (also see below for more recent Long Bay surveys by Walker *et al.* 2000, and Walker & Babcock 2001), and at Wenderholm where a small juvenile population (10-20 mm shell length) was recorded from 1996-1997 (Pawley *et al.* 1997). It is impossible to comment on trends in population abundance for *P. subtriangulata* at Long Bay or Wenderholm as no recent MFish survey data are available and other surveys (e.g. Walker *et al.* 2000) were not designed to collect density information.

We note that some sites may have been less frequently sampled by MFish under the assumption that these sites are being adequately covered by other surveys, for example Cheltenham Beach (Cheltenham Beach Caretakers) and Long Bay (ARC monitoring of Long Bay Marine Reserve).

Based on this series of inter-tidal MFish surveys, it would not be appropriate to draw conclusions regarding the status of *P. subtriangulata* populations in the Hauraki Gulf Marine Park in general. Other areas that might be surveyed subtidally to gain more relevant data regarding trends in *P. subtriangulata* abundance include the exposed coast north of the Auckland Metropolitan Area (e.g. Omaha, Pakiri etc.), along the eastern coast of the Coromandel Peninsula, and the eastern coasts of islands within the Hauraki Gulf.

Due to the fact that *M. liliana* were dropped from the surveys in 1998, it is difficult to evaluate the current status of *M. liliana* stocks in the Hauraki Gulf Marine Park from the MFish surveys (however past results are outlined below).

It is difficult to accurately gauge the historic abundance of *A. stutchburyi*, *P. australis*, *P. subtriangulata* and *M. liliana* at various sites within the Hauraki Gulf Marine Park prior to the commencement of the Ministry of Fisheries surveys in 1992. Some historical density information is available for a few sites from unpublished M.Sc. theses (for example, Cheltenham Beach *A. stutchburyi* data from McQueen (1984) and Howick Beach *A. stutchburyi* and *P. australis* data from Wood (1962)). However, interpreting these data requires caution as the areas surveyed were not the same as the current MFish surveys. Wood's 1962 data are from one transect positioned at the southern end of Howick beach, and although very high densities of up to 1400 *A. stutchburyi* m$^2$ (shell lengths 11-17 mm) were recorded along this transect, it is not possible to generalise these data to obtain a population estimate for the entire beach.

In general, the Ministry of Fisheries surveys have identified that beaches tend to contain either a predominantly *A. stutchburyi*/*M. liliana* association, or *P. australis*, suggesting different habitat requirements for the two groups (Pawley *et al.* 1997).

An historical analysis of MFish survey data for 1993 to 1997 fishing years was undertaken by Pawley *et al.* (1997). This included sites within the Hauraki Gulf Marine Park for which at least four years of data were available, (e.g. Cheltenham Beach, Eastern Beach, Howick and Wenderholm). Although no statistical analysis was provided, Pawley *et al.* (1997) concluded that *A. stutchburyi*, *P. australis* and *M. liliana* had decreased in density (measured as numbers per m$^2$). However, visual interpretation of the information provided (see Fig. 13 in Pawley *et al.* 1997) suggests that while there may have been a declining trend in *A. stutchburyi* abundance overall, *P. australis* and *M. liliana* abundance was highly variable with no obvious overall visual trend apparent.

In the same study, Pawley *et al.* (1997) presented information that indicated that recruitment success of *A. stutchburyi* was very site specific. For example, from 1992 to 1997 very poor sporadic *A. stutchburyi* recruitment occurred at Cheltenham Beach, whereas Eastern Beach received relatively constant recruitment. However, *P. australis* recruitment appeared to occur at most sites in most years (Pawley *et al.* 1997). Like *A. stutchburyi*, *M. liliana* recruitment appeared to be variable within and among beaches and years (Pawley *et al.* 1997).

A change in the sampling methodology from random to stratified sampling occurred in the fishing year 1997/98 (Morrison *et al.* 1999). This makes it difficult to compare annual population abundance estimates prior to this change with those estimated using the new methodology. This is

of particular concern in the case of beaches that have not been recently sampled by MFish, for example, Beachlands, Cheltenham (although see Cheltenham Beach Caretakers' data below) and Howick beaches. It also makes the long term analysis of data from Okoromai Bay, Eastern Beach and Umupuia difficult (e.g. pre 1998 and post 1998 data). However, abundance estimates (based on stratified sampling) are available from Morrison & Brown (1999) for both Beachlands and Howick Beaches (however, no error estimates are provided).
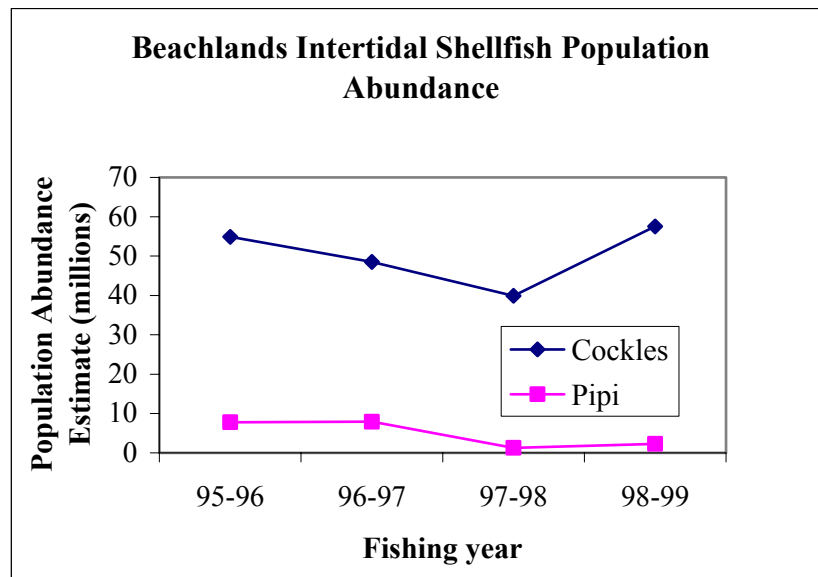


**Figure 4.1: Population abundance estimates for Beachlands inter-tidal bivalve populations, 1996 to 1999 (from Morrison & Brown 1999).**

Figure 4.1 suggests that Beachlands had a large *A. stutchburyi* population between 1995/96 and 1998/99, along with a modest *P. australis* population (*A. stutchburyi* and *P. australis* numbers in 1998/99 estimated as 57.5 million and 2.3 million respectively). The lack of subsequent data means that it is impossible to determine the trends in abundance of these populations since these surveys.

Figure 4.2 (following page) suggests that Howick Beach had modest populations of *A. stutchburyi* and *P. australis* between 1995/96 and 1998/99 (*A. stutchburyi* and *P. australis* numbers in 1998/99 estimated as 0.4 million and 0.96 million respectively). Again, due to lack of subsequent data it is impossible to determine the trends in abundance of these populations since these surveys. However, *A. stutchburyi* densities at Cockle Bay (Howick) are monitored by Howick College, and this population shows declining *A. stutchburyi* population abundance (B. Drey, MFish, Auckland, *pers. comm*.).
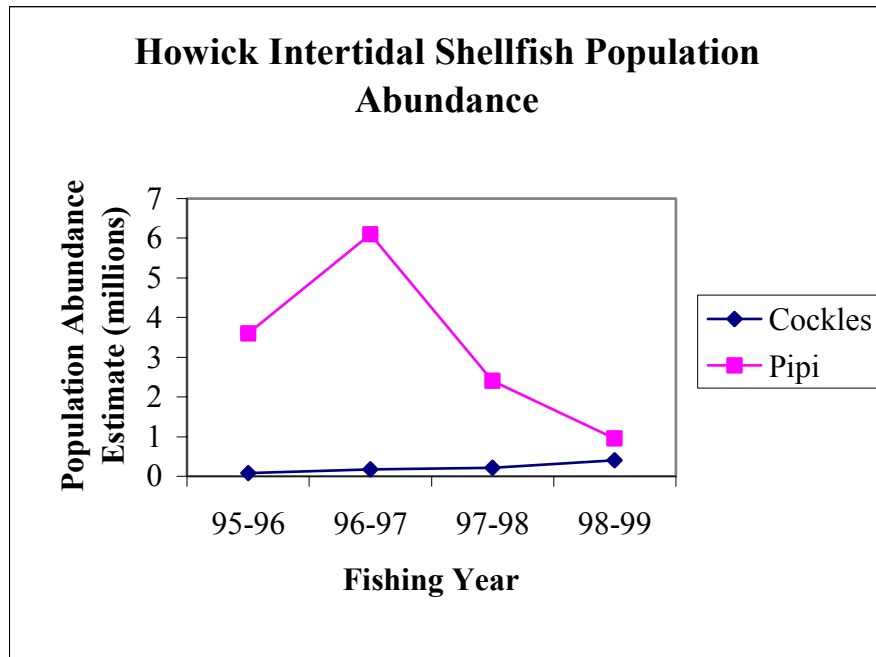
**Figure 4.2: Population abundance estimates for Howick Beach inter-tidal bivalve populations, 1996 to 1999 (from Morrison & Brown 1999).**
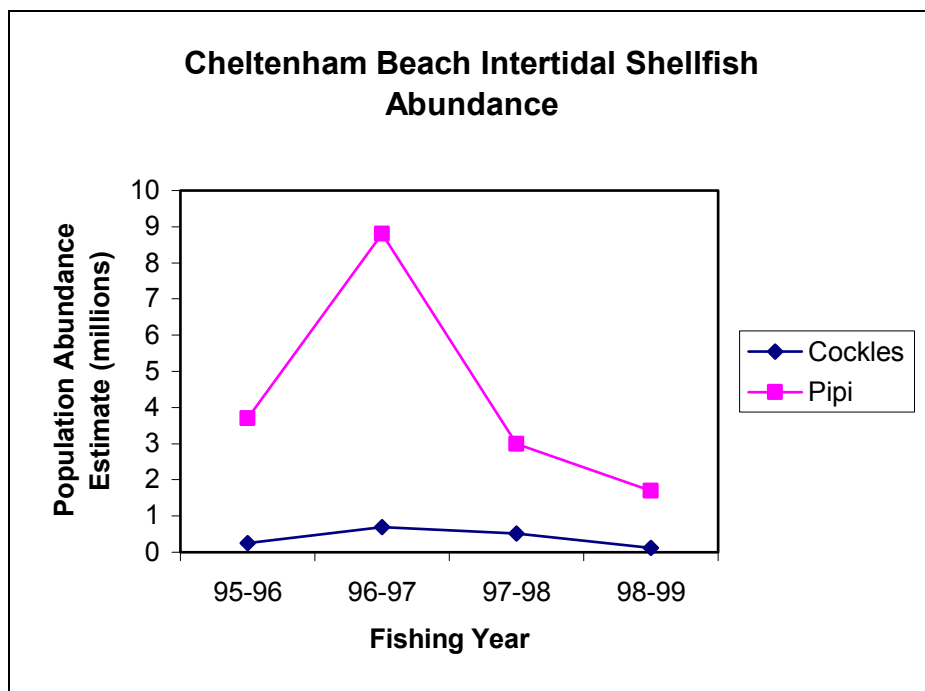


**Figure 4.3: Population abundance estimates for Cheltenham Beach inter-tidal bivalve populations, 1996-1999 (from Morrison & Brown 1999).**

Figure 4.3 suggests that Cheltenham Beach had modest populations of *A. stutchburyi* and *P. australis* between 1995/96 and 1998/99 (*A. stutchburyi* and *P. australis* numbers in 1998/99

estimated as 0.1 million and 1.7 million respectively).   Interestingly, Cheltenham Beach and Howick Beach seemed to have similar declines in *P. australis* numbers over this period  Again, due to lack of subsequent data from recent MFish surveys it is impossible to determine the trends in abundance of these populations with this particular time series.  However, the Cheltenham Beach Caretakers (CBC) have also been monitoring bivalve abundance at this beach, using a systematic grid sampling strategy.  Results of average *A. stutchburyi* and *P. australis* densities over this grid area (which covers the upper third of the inter-tidal zone) are presented in Figure 4.4.
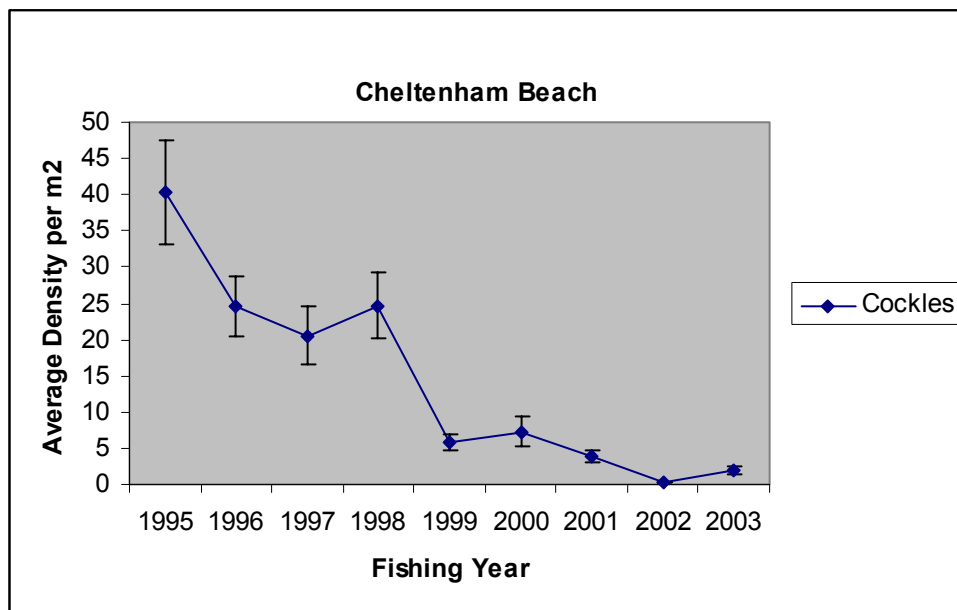


**Figure 4.4: Average density per m$^2$ estimates for the Cheltenham Beach inter-tidal *A. stutchburyi* population, 1995 to 2003.   Data from the Cheltenham Beach Caretakers systematic grid sampling monitoring programme.**

Although the data in Figure 4.4 cannot be compared directly with the previous MFish population abundance data in Figure 4.3 (as differences in sampling methodology and sample area preclude this), the CBC data does suggest that over the grid sampling area *A. stutchburyi* numbers have continued to decline over the period 1995 to 2003 (over this period the beach has been closed to human shellfish harvesting).

Figure 4.5 outlines the *P. australis* data obtained from the same systematic sampling grid at Cheltenham Beach.  The large increase in *P. australis* density in 1998 was due to a heavy settlement of juvenile *P. australis*, which did not survive through to the following year.  It is difficult to identify any pattern in overall population abundance from this graph as in most years heavy *P. australis* settlement occurs at this beach, which would tend to mask the true resident population abundance.
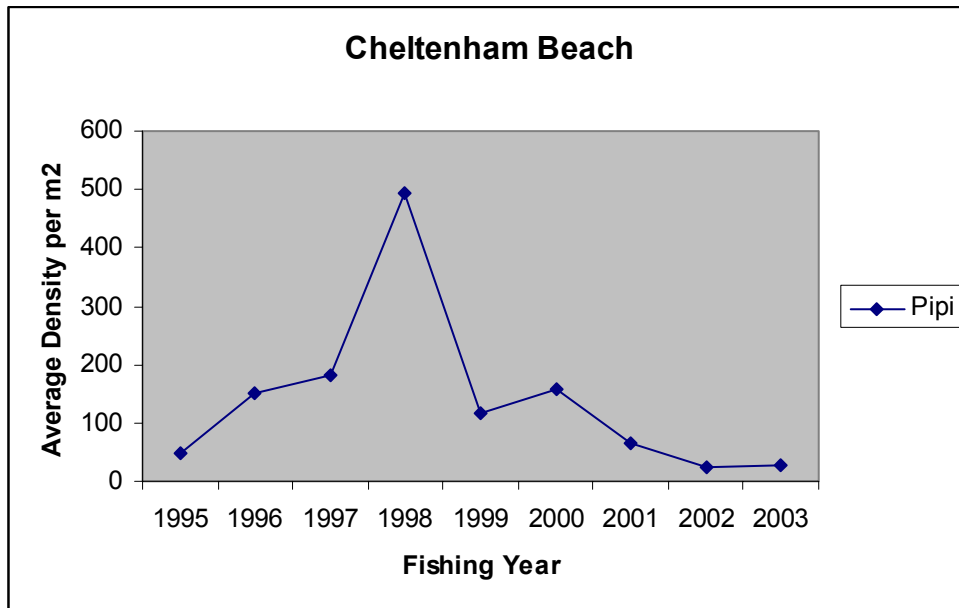
**Figure 4.5: Average density per m$^2$ estimates for the Cheltenham Beach inter-tidal *P. australis* population, 1995 to 2003. Data from the Cheltenham Beach Caretakers systematic grid sampling monitoring programme.**

Several sites have sufficient data to allow comparisons in population abundance after the MFish methodology change to a stratified sampling protocol. These sites are Eastern Beach, Okoromai Bay, Umupuia, Whangamata and Tairua.

Figure 4.6 below outlines the changes in estimated *A. stutchburyi* population abundance at Eastern Beach, Okoromai Bay and Umupuia from 1998-2002. Walshe & Akroyd (2002) in reviewing the changes between the 2000 data and the 2002 data suggest that Eastern Beach and Okoromai Bay showed a significant decline in abundance of *A. stutchburyi* over this period. The low *A. stutchburyi* abundance recorded at Umupuia in 2000 would suggest the population has increased since that survey, but may still be lower than the original abundance estimate from 1998.
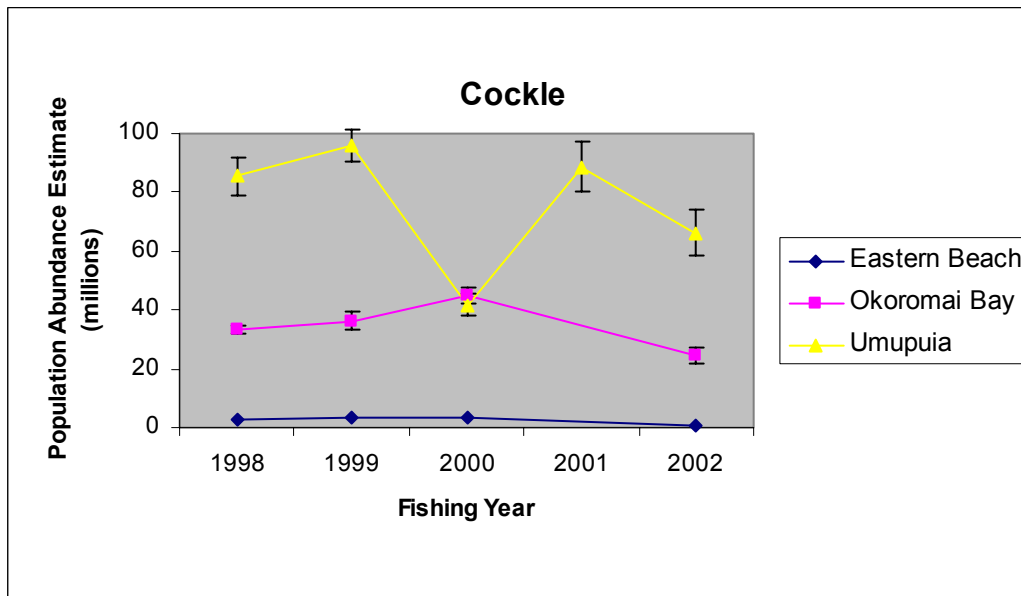


**Figure 4.6: Population abundance estimates for Eastern Beach, Okoromai Bay and Umupuia *A. stutchburyi* populations, 1998-2002 (data compiled from Akroyd *et al.* 2000, Walshe *et al.* 2001 and Walshe & Akroyd 2002).**

Figure 4.7 below outlines the changes in estimated *A. stutchburyi* population abundance in Whangamata and Tairua harbours. Walshe & Akroyd (2002) in reviewing the changes between the 2000 data and the 2002 data suggest that *A. stutchburyi* abundance in Whangamata has significantly increased while *A. stutchburyi* numbers in Tairua Harbour have decreased.
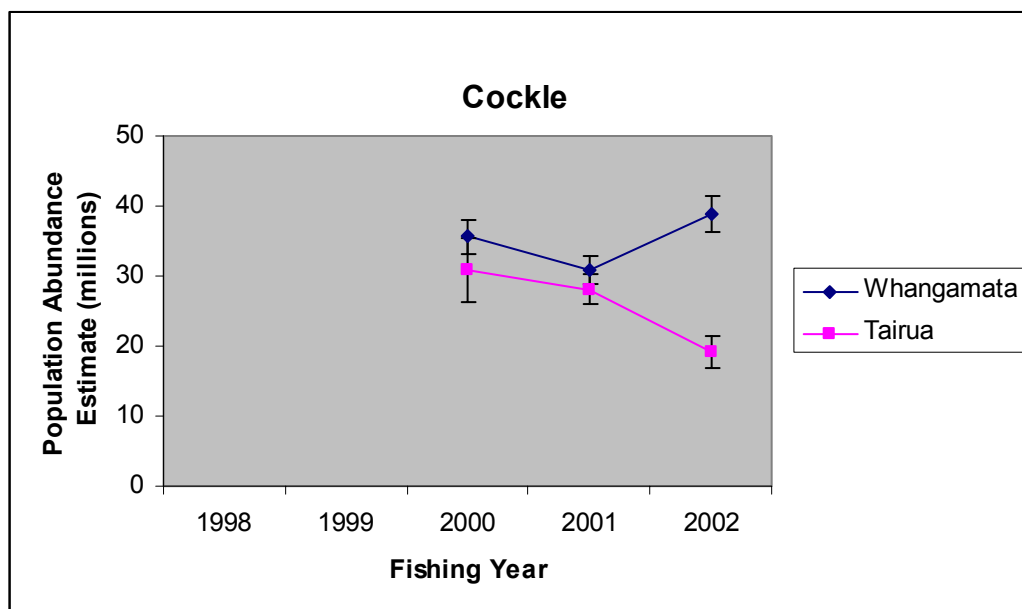


**Figure 4.7: Population abundance estimates for Whangamata and Tairua Harbour *A. stutchburyi* populations, 2000-2002 (data compiled from Akroyd *et al.* 2000, Walshe *et al.* 2001 and Walshe & Akroyd 2002).**

**Figure 4.8: Population abundance estimates for Eastern Beach, Okoromai Bay and Umupuia *P. australis* populations, 1998-2002 (data compiled from Akroyd *et al.* 2000, Walshe *et al.* 2001 and Walshe & Akroyd 2002).**
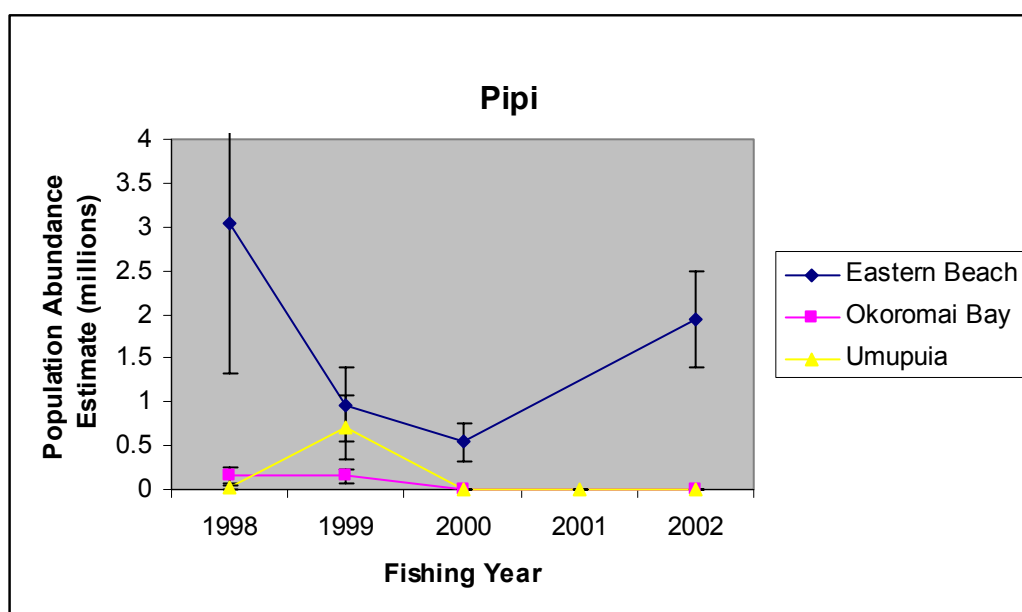
*Paphies australis* population abundance estimates are depicted above for Eastern Beach, Okoromai Bay and Umupuia from 1998 to 2002 (see Figure 4.8).

*Paphies australis* numbers at Okoromai Bay and Umupuia are now so few that population estimates are deemed inappropriate (Walshe & Akroyd 2002). Eastern Beach showed a significant increase in *P. australis* abundance when compared with the 2000 data (Walshe & Akroyd 2002).

Total *P. australis* population abundance showed no significant change at either Whangamata or Tairua Harbour from 2000 to 2002 (Walshe & Akroyd 2002) (Figure 4.9).
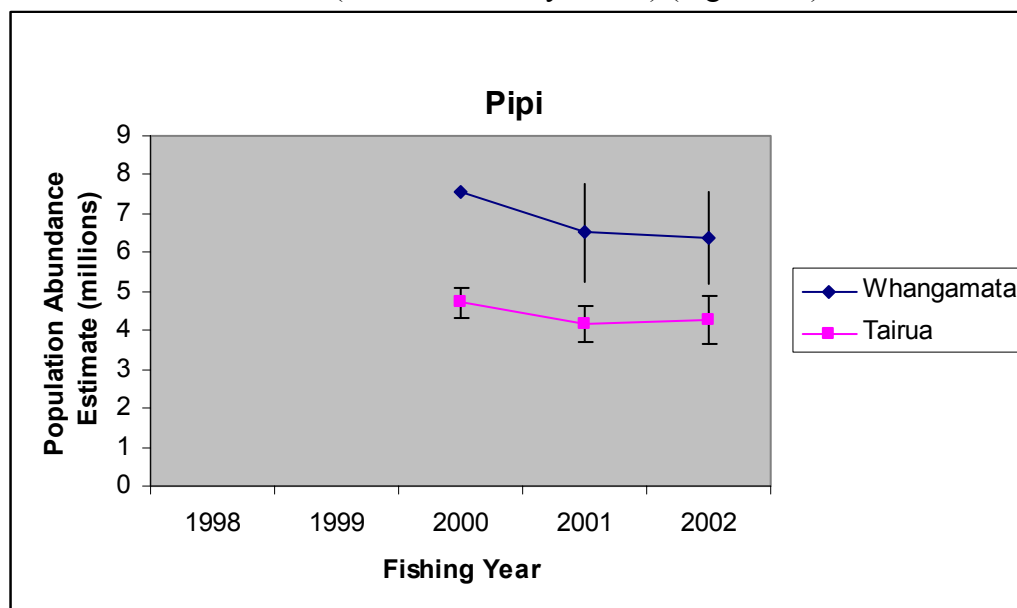


**Figure 4.9: Population abundance estimates for Whangamata and Tairua Harbour *P. australis* populations, 2000-2002 (data compiled from Akroyd *et al.* 2000, Walshe *et al.* 2001 and Walshe & Akroyd 2002).**

In summarising the MFish inter-tidal shellfish survey results we note the following points:

- Long term data for a particular beach are limited due to changes in sampling design and methodology. The longest data sets are for a period of 5 years from 1998 to 2002.
- For the many beaches (approx. 58%) not sampled recently (within the last 2 years) it is difficult to make any comment about trends in shellfish abundance (this includes Wenderholm, Long Bay, Cheltenham Beach (although CBC data indicates continued *A. stutchburyi* decline), St Heliers, Bucklands Beach, Howick Beach, Beachlands, Omana, Maraetai, Kauri Bay and Kawakawa Bay).
- Of those beaches where recent semi-continuous data are available, data comparisons have only been made between 2000 and 2002 data. From this analysis *A. stutchburyi* abundance at Eastern Beach, Okoromai Bay and Tairua Harbour has declined, while *A. stutchburyi* abundance at Umupuia and Whangamata Harbour has increased. *Paphies australis* have virtually disappeared from Okoromai Bay and Umupuia, increased at Eastern Beach and shown no change in Whangamata and Tairua Harbours.
- The information necessary to assess trends in population abundance for *P. subtriangulata* and *M. liliana* is not available.

## 4.3    OTHER RELEVANT STOCK INFORMATION

*Whitianga Estuary* – Creese (1988) investigated the density and distribution of *P. australis* at Buffalo Beach within the Whitianga Estuary.  This inter-tidal/subtidal bed was found to have an average density of 575 *P. australis* per m$^2$.

*Whangapoua Estuary* – *Paphies australis* were also reported from the Whangapoua Estuary by Miller (1987).

*Whangamata Estuary* – Basic distribution/abundance data and size frequency information was collected for *A. stutchburyi*, *P. australis* and *M. liliana* in the Whangamata estuary by Bridgwater & Foster (1984).

*Tairua Estuary* - In 1994, Environment Waikato investigated the biological communities of Tairua Harbour's inter-tidal sediments (White 1996).  The distribution of shellfish throughout the harbour appeared to be largely unchanged from that described by Bridgwater & Foster (1984).  However, the abundance of shellfish was possibly lower in 1994 than in 1984, particularly for edible bivalve species such as *A. stutchburyi* (White 1996).  The maximum *A. stutchburyi* size had also decreased (White 1996).  Limited comparisons for *P. australis* communities were able to be made due to differences in sampling methodology (i.e. in 1984 both subtidal and inter-tidal populations were sampled, while in 1994 only inter-tidal populations were sampled).  Larger *P. australis* were found to live near the harbour mouth, while smaller *P. australis* lived in the sheltered centre of the harbour (White 1996).  It was suggested that densities of *M. liliana* increased from 1984 to 1994 (White 1996).  White's study (1996) also identified very little difference in vegetation around Tairua Harbour between 1984 and 1994.

*Whangateau Harbour* – The Whangateau Harbour has large tidal flats, with a huge *A. stutchburyi* resource in terms of both biomass and spatial extent of the beds (Kearney 1999, Tricklebank & Stewart 2001).  Unpublished information on densities of A. stutchburyi in this harbour is also available from Dr. J. Walsby (Mahurangi Technical Institute) and Dr. C. Pilditch (Waikato University). (Note that Dr Walsby also collects *A. stutchburyi* density data from Snells Beach).

*North Shore Coastline* - Bioresearches (1992) reported finding medium sized *P. australis* (<45 mm) in high densities at some sites (for example Castor Bay) along the Rangitoto Channel shoreline.  Bioresearches (1992) also found *P. subtriangulata* in moderate densities at Mairangi Bay and Milford Beach.  *Paphies subtriangulata* are often scarce or patchily distributed along the beaches of the North Shore of Auckland (Walker *et al.* 2000, based on inter-tidal sampling).  *Paphies subtriangulata* and *P. australis* are found at Long Bay, Torbay, Browns Bay (low *P. australis* numbers), and Mairangi Bay (at mid to low inter-tidal sampling stations, however these surveys were not specifically designed to provide information about densities of these species) (Walker & Babcock 2001).  Sampling undertaken in March 2001 (Walker & Babcock 2001) found that the *P. australis* populations in Torbay and Browns Bay were heavily dominated by juveniles with a mean size of 5 mm shell length (a 1 mm mesh was used to sort samples).  The main population modes for Torbay and Browns Bay *P. australis* were 1-11 & 2-6 mm shell length, respectively.  Very few large adults were recorded.  *Paphies subtriangulata* sampled at Torbay in March 2001 (Walker & Babcock 2001) were again heavily dominated by juvenile size classes with a modal size of 2-9 mm in shell length.  In 1999 Kingett Mitchell conducted a sub-tidal shellfish survey in areas north from the Rangitoto Channel along the North Shore City coastline.  This study found only three edible shellfish species, namely horse mussels, scallops and green-lipped mussels (Kingett Mitchell 1999).

*Mahurangi Estuary* - Infaunal inter-tidal bivalves have been monitored at five sites in the Mahurangi Estuary as part of an ARC ecological monitoring programme that began in July 1994. The inter-tidal sites include Cowans Bay, Hamilton Landing, Jamiesons Bay, Mid-harbour and Te Kapa Inlet. Data from this monitoring programme from July 1994-January 2003 are summarised in a report to the Auckland Regional Council by Cummings *et al.* (2003).

Trends in macrofaunal abundance data from this study were statistically analysed (refer to Cummings *et al.* (2003) for details of these methods). Decreasing trends in *M. liliana* abundance were detected at all sites except Cowans Bay. In addition, numbers of *M. liliana* >16 mm in size have decreased at three sites (Hamilton Landing, Mid-harbour and Te Kapa Inlet). Cummings *et al.* (2003) suggest that lack of individuals of spawning size means recruitment at these sites will rely on immigration of larval and juvenile stages from outside the sites.

Decreasing trends in abundance of *A. stutchburyi* were detected at Hamilton Landing and Te Kapa Inlet. In the last two years *A. stutchburyi* at Hamilton Landing were either absent or in very low numbers (Cummings *et al.* 2003).

Various sediment characteristics in the Mahurangi estuary were also monitored during this study. Over the time of the study there was an increase in amount of fine sand at all sites and a corresponding decrease in the amount of medium sand at inter-tidal sites only. These changes occurred sometime between April 1997 and April 1998, and have persisted. However, there was no obvious response in *A. stutchburyi* or *M. liliana* abundance (or any of the other species noted) to this apparent "pulse" event, which in contrast exhibited gradual declining trends (Cummings *et al.* 2003).

In conclusion, we note that there appears to be strong anecdotal evidence for a decline in infaunal shellfish abundance in many areas within the Hauraki Gulf Marine Park over the long term (i.e. last forty years). There also appears to be widespread concern that there is a general decline in infaunal bivalve abundance across the Hauraki Gulf Marine Park. We would hesitate to attribute this to misconception just because it is not conclusively supported by scientific data. On the contrary, our review of data suggests that it is critical that a more focused research effort to identify long-term trends in abundance of infaunal bivalves in the Hauraki Gulf Marine Park be promoted in order to assist in the identification and management of the potential causes of shellfish depletion.

## 4.4   REFERENCES

**Akroyd, J. M., B. F. J. Manly & K. A. R. Walshe  2000.** Distribution, abundance and size frequency of pipi (*Paphies australis*), cockles (*Austrovenus stutchburyi*) and tuatua *(Paphies subtriangulata*) at eleven beaches during the 1999/2000 fishing year. Report to the Ministry of Fisheries by Akroyd Walshe Ltd., AKI 1999-01. 20pp.

**Bioresearches  1992.** North Shore City Council Hauraki Gulf Outfall Study – Biological Resources. Report prepared for North Shore City Council.

**Bridgwater, G. D. & B. A. Foster  1984.** Marine biota of the Whangamata and Tairua estuarine systems, Coromandel Peninsula. A report prepared for the Hauraki Catchment Board. 59pp.

**Cook, S., P. Doorman, R. Drey, L. Stevens, C. Turbott & M. Cryer  1994.** A pilot study of the intertidal resources of the Auckland metropolitan region 1992. Northern Fisheries Region Internal Report. 18pp.

**Creese, R. G.  1988.** Aspects of the ecology of pipi at Whitianga. Report prepared for Wilkins & Davies Marinas Ltd. 54pp.

**Cummings, V., P. Nicholls & S. Thrush  2003**. Mahurangi Estuary ecological monitoring programme – report on data collected from July 1994 to January 2003. Prepared for Auckland Regional Council. NIWA Client Report HAM2003-066. 77pp.

**Kearney, M. B.  1999.** Ecology and management of *Austrovenus stutchburyi* in the Whangateau Harbour. Unpubl. M.Sc. thesis. School of Environmental and Marine Science. 135pp.

**Kingett Mitchell 1999.** Project Rosedale – Investigations of marine ecology at the existing and possible outfall locations. Report prepared for North Shore City Council. 52pp. + App.

**McQueen, D. A. 1984.** An estimate of the annual production of Cheltenham Beach macrofauna. Unpubl. M.Sc. Thesis, University of Auckland, New Zealand. 109pp.

**Miller, S. T. 1987**. Baseline macrobiota survey of the Whangapoua and Whitianga estuaries, Coromandel Peninsula. Report No. 213 of the Hauraki Catchment Board.

**Morrison, M. A. & G. N. Browne 1999.** Intertidal surveys of shellfish populations in the Auckland region 1998-99 and associated yield estimates. *New Zealand Fisheries Assessment Research Document 99/43*. 21pp.

**Morrison, M. A., M. D. M. Pawley & G. N. Browne 1999.** Intertidal surveys of shellfish populations in the Auckland region 1997-98 and associated yield estimates. *New Zealand Fisheries Assessment Research Document 99/25*. 25 pp.

**Morton, J. & M. Miller 1973.** The New Zealand seashore. 2nd Edition. Collins, Auckland. 653pp.

**Pawley, M. D. M., G. N. Browne and M. Morrison 1997.** Survey of intertidal shellfish populations in the Auckland metropolitan area (1996/1997) with a summary of previous sampling methodologies and results (1992-1996). New Zealand Fisheries Assessment Research Document Draft for Ministry of Fisheries.

**Tricklebank, K. A. & M. J. Stewart 2001.** Parasites in Cockles (Austrovenus stutchburyi): Potential indicators of estuarine health. Unpublished Preliminary Report for the Auckland Regional Council, December 2001.

**Walker, J. W. & R. G. C. Babcock 2001.** The Long Bay monitoring program sampling report July 2000/June 2001. A report prepared by Auckland Uniservices Ltd. for The Auckland Regional Council. 77pp.

**Walker, J. W., R. G. C. Babcock & R. G. Creese 2000.** Long Bay Monitoring Program 2000 Sampling Report. 2000. A report prepared by Auckland Uniservices Ltd. for The Auckland Regional Council. 74pp.

**Walshe, K. A. R. & J. M. Akroyd 2002.** Distribution, abundance, and size frequency of pipi (*Paphies australis*), and cockles (*Austrovenus stutchburyi*) at twelve beaches during the 2001/2002 fishing year. Report for the Ministry of Fisheries, Auckland by Akroyd Walshe Ltd. October 2002. 37pp.

**Walshe, K. A. R., J. M. Akroyd & B. F. J. Manly 2002.** Distribution, abundance, and size frequency of pipi (*Paphies australis*), cockles (*Austrovenus stutchburyi*) and tuatua (*Paphies subtriangulata*) at twelve beaches during the 2000/2001 fishing year. Report for the Ministry of Fisheries, Auckland, by Akroyd Walshe Ltd. January 2002. 41pp.

**White, S. 1996.** Tairua Harbour intertidal benthic ecology 1994. Environment Waikato Internal Report 1996/19. 13pp.

**Wood, D. H. 1962.** An ecological study of a sandy beach (Howick). Unpubl. M.Sc. thesis. University of Auckland, New Zealand. 83pp.

## SECTION 5:     ECOLOGICAL RISK ASSESSMENT

## 5.1     INTRODUCTION

### 5.1.1     An Introduction to the Risk Assessment Process

Risk Analysis is a deliberative approach to lowering risk and the severity of harm by conducting a risk assessment of identified hazards, determining risk management options, and communicating the risk (US EPA 1993). This section of our knowledge review utilises the risk assessment framework as specified in the US EPA Risk Analysis Model.

An ecological risk assessment is the evaluation of the current state of knowledge about a hazard or set of hazards. It makes estimates of the probability that harm will occur after exposure to the hazard. Ecological risk assessments can assist regulators in decision making by providing a systematic evaluation of the state of knowledge related to a set of hazards with respect to a particular species or environment, including the degree of uncertainty. It can include either a qualitative review of published information on the hazards or (in cases where sufficient information is available) a quantitative assessment of risk.

The risk assessment process for each potential hazard includes four steps:

*Hazard Identification*
Identification of the agent of concern (both actual and potential, including physical, biological or chemical possibilities), its adverse effects, and target population (e.g. different life cycle stages of these bivalves, or bivalves with a particular habitat preference, or feeding mode).

*Dose-Response Assessment*
Dose-response assessment provides a determination of the degree of the effects of hazards at different levels/concentrations. Toxicity tests utilised in dose-response assessment should be based on ecologically relevant endpoints such as survival, reproduction, development, and growth.

*Exposure Assessment*
Estimation of the magnitude, duration, and frequency of bivalve exposure to hazards of concern, and the number of bivalves exposed via different pathways (e.g. can it effect different lifecycle stages differently?).

*Risk Characterisation*
Risk characterisation provides an estimate of the risk under various scenarios, drawn from the Hazard Identification, Dose-Response Assessment, and the Exposure Assessment. This step also presents information on the uncertainties of the analysis. The risk characterisation process combines the information compiled in the exposure and effects assessments in order to estimate potential risk. The risk characterisation may simply be deterministic, in which a point estimate of exposure is compared to a point estimate of effects (e.g. the concentration protective of 95 percent of the species). Alternatively, it may be probabilistic, in which the distribution of exposure data is compared to the distribution of effects data, and risk is reported as the percentage species of the population expected to be affected.

The following risk assessment contains a qualitative review of stressors potentially affecting the status of stocks of cockle (*Austrovenus stutchburyi*), pipi (*Paphies australis*), tuatua (*Paphies subtriangulata*), and wedge shells (*Macomona liliana*) in the Hauraki Gulf Marine Park Area. The

risk assessment is structured according to the four steps outlined above. Where information is available, the steps are specific for each bivalve species under investigation, and for each life cycle stage within the species. Potential stressors considered include:

- Anthropogenic contaminants (i.e. contaminants of, or relating to, human beings) such as organotin compounds and organic booster biocides (such as those associated with marine antifoulants), heavy metals, organochlorines and polyaromatic hydrocarbons;
- human harvesting;
- changes in the marine environment associated with human activity, such as increased sediment loading, nutrient enrichment and climate change;
- natural phenomena of an extraordinary nature such as harmful algal blooms, and diseases/parasite events.

Readers will note that of all the potential stressors discussed in the following sections, there are none for which full risk characterisations have been possible. This does not however detract from the process of risk assessment, since it allows gaps in information to be identified in a structured manner.

Figure 5.1 below illustrates the generic process of risk analysis following the US EPA model. It can be seen that a wide variety of research issues need to be addressed prior to completion of risk assessment. It is thus not surprising that knowledge gaps exist with respect to the risk assessment of potential stressors to infaunal bivalve populations in the Hauraki Gulf Marine Park.
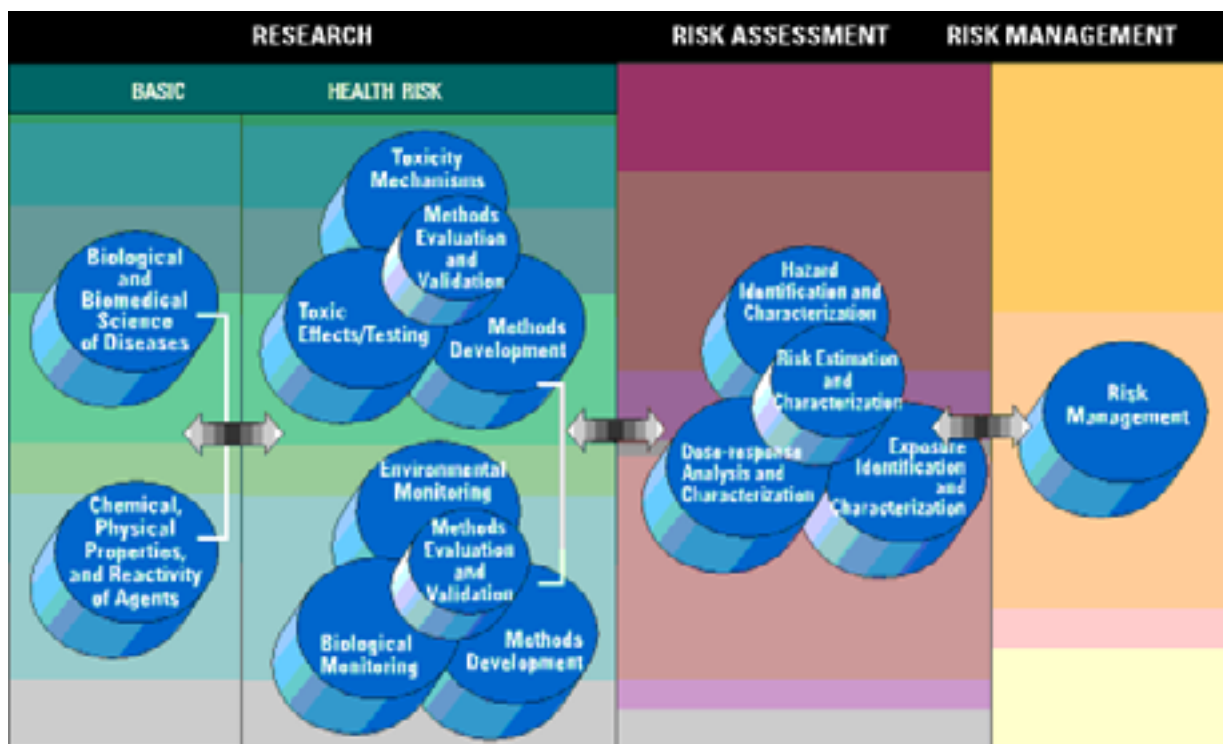


**Figure 5.1: Diagram illustrating the process of risk analysis based on the US EPA model (from Daughton 2000).**

Information for the following risk assessment was gleaned from local and international literature, discussion with New Zealand scientists currently working on projects in this field, and from the

Pesticides Action Network - Pesticides Database on the internet (http://www.pesticideinfo.org). There is a body of information contained in the results of routine monitoring that form part of resource consent requirements within the Hauraki Gulf Marine Park region that has not been reviewed. These data could potentially make a significant contribution to risk assessments also.

It is important to note the limitations of this risk assessment:

- The process of risk analysis encompasses known stressors or hazards only. This process in itself cannot identify new possible stressors.
- Mixture effects (synergism and antagonism) have not been considered in the risk assessment process.
- Of necessity, the depth of this review has been constrained by funding and time constraints.


## 5.1.2    References

**Daughton, C. G. 2000.** Pharmaceuticals and personal care products (PPCPs) as environmental pollutants: Pollution from personal actions, activities and behaviours. US Environmental Protection Agency, Las Vegas, Nevada, a web-site created April 2000, available http://www.epa.gov/nelesd1/chemistry/pharma/index.htm.

**United States Environment Protection Agency (US EPA) 1993.** A guidebook to comparing risks and setting priorities. Washington USEPA Office of Policy, Planning & Evaluation.

## 5.2 STRESSOR: ANTIFOULING PAINTS

### 5.2.1    Hazard Identification

*The potential stressors associated with the antifouling of boats and marine structures include Organotins (e.g. Tributyltin (TBT) and Triphenyltin (TPhT)), Organic Booster Biocides (e.g. Diuron and Irgarol 1051) and Heavy Metals (e.g. copper, zinc).*

***Organotins:***

Organotins are one of the most toxic pollutants of aquatic life known thus far (Fent 2003). Although organotins were completely banned from antifouling paint in NZ in 1993, the long persistence time of tributyltin in sediments may produce a "legacy problem" with respect to its toxic effects on aquatic life. The main organotin compounds in the marine environment are the biocides tributyltin (TBT) (and its associated breakdown products, monobutyltin (MBT) and dibutyltin (DBT)), and triphenyltin (TPhT) (and its associated breakdown product, diphenyltin (DPhT)), derived largely from marine antifouling paints (de Mora *et al.* 2003). Triphenyltins are also used in agrochemicals (e.g. fungicides and insecticides) (Blunden & Evans 1990). TBT may also be introduced into the marine environment from sewage sludge and wastewater (Morcillo *et al.* 1998) and from timber treatment processes (Fent 2003).

Bioavailability of organotins from the water column varies with exposure time, the chemical form of the pollutant, salinity, temperature, pH and the content of organic matter (Coelho *et al.* 2002a). Organotin speciation shows strong pH dependence, with TBT and TPhT present as cations at low pH, and as hydroxides at higher pH (Fent 2003). Cations and hydroxides of these compounds exhibit very different partitioning and sorption behaviour (Fent 2003). Dissolved organic matter such as humic substances cause a reduction in the bioavailability of organotins due to hydrophobic sorption to these substances (Looser *et al.* 2000)

Previous studies of TBT degradation have indicated that this compound has low persistence in the water column (e.g. Unger *et al.* 1988). In natural waters, the TBT leached from antifouling paint is removed relatively quickly from the water column, with a half-life (i.e. the time taken for a substance to decline to 50% (half) of its original concentration) of several days to weeks (Stewart & de Mora 1990). This may occur through photochemical decomposition or microbially induced debutylation (a process which produces the less toxic DBT and MBT species) (de Mora *et al.* 1995). Alternatively and more commonly, TBT (which is particularly surface-active) binds to particulate matter in the water column, with subsequent sedimentation. Thus TBT may become available to benthic biota through a combination of these different pathways (Coelho *et al.* 2002a). In marine sediments organotin compounds are persistent due to their slow degradation rates (Stewart & de Mora 1990). The half-life of TBT in aerobic marine sediments has been estimated to be 2.5-3 years (de Mora *et al.* 1995). A study of TBT degradation in marine sediments (Dowson *et al.* 1996) suggests that TBT either debutylates to DBT or MBT in aerobic sediments or degrades to DBT, which desorbs back into the water column. However, in anaerobic sediment the half-life of TBT was not discernible and appears to be in the order of tens of years (Dowson *et al.* 1996). The ratio of MBT to TBT in sediment can be used to imply the level of recent inputs of TBT into the marine environment (de Mora *et al.* 2003). Consequently, although the use of TBT in marine antifouling paints has been limited by regulation for over a decade in New Zealand, it is still likely to be present in sediments in the Hauraki Gulf Marine Park.

TBT and TPhT have different modes of action (Fent 2003). TBT is known to be a cytochrome P450 inhibitor and is capable of producing complexes by binding with the nitrogen or sulphur atoms of certain amino acids (Alzieu 1996). Despite the fact that bivalves are known to be very sensitive to low concentrations of TBT, only limited information concerning the specific mode of action of organotin compounds is available (Morcillo *et al.* 1998). Morcillo *et al.* (1998) suggest the TBT interacts with androgen metabolism in the clam *Ruditapes decussate*. TBT may cause imposex (i.e. the imposition of male sex organs on females) in neogastropods (whelks) via endocrine disruption (endocrine disrupters can be defined as man-made or naturally occurring substances that can mimic or interfere with the biosynthesis, binding and/or action of hormones, in doing so disrupting physiological processes which are under hormonal control), but its precise mode of action is still incompletely understood even with respect to these organisms (Matthiessen & Gibbs 1998). Currently, both TBT and TPhT can be described as suspected endocrine disrupters.

Tributyltin assimilation by aquatic organisms is generally considered to be dominated by passive diffusion from solution mainly via the gills and other respiratory membranes (Cardwell *et al.* 1999, Coelho *et al.* 2002a). The lipophilic properties of TBT as well as its moderately high octanol-water water partitioning coefficient contribute to its bioaccumulation in marine organisms (Fent 1996). Although TBT in the water column appears to be the predominant source of TBT toxicity to bivalves, exposure may also occur through the diet (Cardwell *et al.* 1999, Coelho *et al.* 2002a,b). Assimilation of organotins from food sources by benthic invertebrates involves different mechanisms to that of uptake from water or sediment, as TBT from dietary sources is taken up cells in the digestive tract and then distributed via the blood (Coelho *et al.* 2002b). Uptake of TBT from seawater by phytoplankton is thought to occur over short time scales (e.g. minutes to hours) (Coelho *et al.* 2002b). TBT uptake by infaunal bivalves may also be related to ingestion and subsequent digestion of TBT bound to sediment particles (Coelho *et al.* 2002b).

A range of effects of exposure of bivalves to TBT is reported in international literature. These include:

- Imposex in neogastropods (Gibbs & Bryan 1996) and oysters (Thain 1986, Roberts *et al.* 1987 cited in Cima *et al.* 1998). Cima *et al.* (1998) observed that oysters in waters of relatively low organotin concentration (i.e. <10 ng L$^{-1}$) show a high incidence of imposex, resulting in a population that cannot reproduce.

- Reduced growth, and shell thickening in oysters (Alzieu 1996).

- Decreased reproductive capacity in oysters (Wolniakowski *et al.* 1987, cited in Elgethum *et al.* 2000).

- Enhanced mortality and malformation of bivalve larvae (Laughlin *et al.* 1988 (*Mercenaria mercenaria*), Lapota *et al.* 1993 (*Mytilus edulis*), Ruiz *et al.* 1995a,b (*Scrobicularia plana*).

  Cima *et al.* (1998) suggest that these morphological impacts may be responsible for a decline in the abundance of various bivalve populations.

- Anderson *et al.* (1996) reported that oysters (*Crassostrea virginica*) have enhanced levels of protozoan infection (*Perkinsus marinus*) when exposed to levels of TBT that are similar to those encountered at contaminated sites in the marine environment.

- Biochemical impacts of TBT contamination have also been observed. Cima *et al.* (1998) suggest that TBT may inhibit the production of reactive oxygen intermediates in the infaunal bivalve *Tapes philippinarum*.

- Awadhesh and co-workers (2000) suggest that TBT at low concentrations could lead to genetic damage (genotoxic effects) in *Mytilus edulis* larvae.

- Organotins at low concentrations may also impact on marine phytoplankton communities (Zhao *et al.* 1990, Alzieu 1996). Based on a review of TBT impacts on phytoplankton species, Alzieu (1996) suggests that a concentration of 0.4 ng TBT-Sn $L^{-1}$ should not be exceeded in order to ensure protection of plankton in the natural environment. The impacts of organotins on phytoplankton stocks may have a flow-on effect for infaunal bivalve species as described above.

The impact of organotins on New Zealand shellfish species has received some research attention. The role of TBT in causing imposex in neogastropods (whelks) has been studied for several New Zealand species (e.g. Stewart *et al.* 1992, Smith 1996, Playfair 1998). Furthermore, shell thickening in *Crassostrea gigas* was correlated with high TBT body burden (King *et al.* 1989). However, there appear to have been no studies undertaken on the impacts of TBT or TPhT specifically on the cockle (*A. stutchburyi*), the pipi (*P. australis*), the tuatua (*P. subtriangulata*) or the wedge shell (*M. liliana*).

***Organic Booster Biocides:***

Copper compounds such as cuprous oxide ($Cu_2O$), copper thiocyanate (CuSCN) or metallic copper are now utilised as the principal biocide in antifouling paints (Voulvoulis *et al.* 2002). The increased use of copper in antifouling paints has raised some concerns about the occurrence of copper in the marine environment, and its impact on aquatic organisms (Voulvoulis *et al.* 1999). (For further discussion on the potential impacts of copper and zinc, please refer to the "Heavy Metals" section later in this report).

Copper exhibits antifouling activity against most organisms, e.g. barnacles, tubeworms and algal fouling species. However, several algal species show marked resilience to copper. In order to produce a complete antifouling paint a number of "*organic booster biocides*" (so named because they help replace the antifouling potency of TBT) are now used in conjunction with copper. Since the phasing out of organotin antifouling paints, organic booster biocides may have become prevalent in the New Zealand coastal marine environment (Dr. C. Stewart, *pers. comm.*).

Organic booster biocides are mainly agrochemicals, and among others include Irgarol 1051, Diuron, Chlorothalonil, Thiram, Ziram, Zineb, Sea-Nine 211, Dichlofluanid, TCMTB (i.e. 2-(thiocyanomethylthio)benzothiazole), Zinc pyrithione and TCMS pyridine (i.e. 2,3,5,6-tetrachloro-4-(methylsulphonyl)pyridine) (Omae 2003). There are currently 46 registered anti-fouling paints on the Agricultural Compounds and Veterinary Medicines Group pesticides database (http://www.nzfsa.govt.nz/acvm/registers-lists/pesticides/index.htm) (This group provides the regulatory control for these chemicals for the NZ Food Safety Authority). Of these approximately 28 % contain Diuron as an active ingredient.

The input of different booster biocides into the marine environment varies depending on their usage in marine antifouling paints, and on the level of runoff contaminated with land-based use of these compounds (they are also used as terrestrial herbicides). The adsorption properties of these

compounds differ, and they thus vary in their tendency to become adsorbed to sediment and settle out of the water column.  This results in differences in environmental persistence (Dr. C. Stewart, *pers. comm.*).  In New Zealand, Diuron and Irgarol 1051 are both quite persistent.   However, while Diuron is mostly water soluble, Irgarol 1051 and Chlorothalonil are more readily adsorbed to sediment and bioaccumulated (Dr. C. Stewart, *pers. comm.*).

Limited data and information are available internationally on the environmental occurrence, fate, toxicity, and persistence of booster biocides in the marine environment (Voulvoulis *et al.* 2002).  It is thus difficult to assess the relative risk of different organic booster biocides (Voulvoulis *et al.* 1999).   Results to date indicate that TCMZ pyridine and TCMTB have environmental characteristics similar to TBT (Voulvoulis *et al.* 2002).  A comparison of TBT with a range of booster biocides by Voulvoulis *et al.* (2002) ranked their environmental performance as follows (highest environmental performance equals the lowest impact to the aquatic environment from their use as antifoulants):

Zinc pyrithione>Zineb >Irgarol >Chlorothalonil >Diuron >Dichlofluanid >TBT

However, in running this comparison the authors noted the significant knowledge gaps relating to the marine toxicity and persistence of many of the booster biocides (Voulvoulis *et al.* 2002).

Overseas studies evaluating the level of booster biocides in the marine environment have identified that Diuron and Irgarol 1051 are most readily analytically detectable using current methods (e.g. Thomas *et al.* 2001).  For this reason, and because they are components of the most commonly used antifoulants (Dr. C. Stewart, *pers. comm.*), most previous studies have focused on Diuron and Irgarol.  There is thus a dearth of information on the presence of many other organic booster biocides in the marine environment, and their potential impacts on marine ecosystems.

No research attention has been focused on the impact of organic booster biocides on New Zealand bivalve species to date (Dr. C. Stewart, *pers. comm.*).  Indeed, internationally there is remarkably little information on the toxicity of organic booster biocides to non-target marine organisms, although data are undoubtedly held in confidence by industry and regulatory authorities (Evans *et al.* 2000, Thomas *et al.* 2001).  However, the existing information for non-target marine organisms suggests that impacts may include deformed growth and teratogenic effects in fish larvae (see Evans *et al.* 2000 for summary).

Different booster biocides have different modes of action, and this is likely to influence their relative toxicity to shellfish.  Both Diuron and Irgarol 1051 are herbicides that act by inhibiting photosystem II (PSII) within the chloroplasts of plants.  This mode of action itself is unlikely to impact on shellfish (Dr. C. Stewart, *pers. comm.*).  However, the possibility that Diuron and Irgarol 1051 may act as endocrine disrupters in marine invertebrates needs further investigation.  In addition, as PSII inhibitors these substances may affect phytoplankton production (Evans *et al.* 2000) and consequently infaunal bivalve food supply.

The synergistic effects of some organic booster biocides (e.g. dithiocarbamates) with copper have also been noted.  In some cases, this results in the formation of lipophilic complexes.  This may in turn enhance the passive uptake of a variety of toxic heavy metals from the water into biota (Phinney & Bruland 1997).

## 5.2.2 Dose-Response Assessment

There appear to be no dose-response data specific to New Zealand shellfish species. However, some dose-response data for similar overseas infaunal bivalve species are available.

*Organotins:*

**Embryo**
- Roberts (1987) found a 48-hr $LC_{50}$ for embryonic *Mercenaria mercenaria* of 1.13 µg TBT-Sn $L^{-1}$ (renewal trial) when exposed to tributyltin chloride.

**Larvae**
- Roberts (1987) found a 48-hr $LC_{50}$ for straight hinged *M. mercenaria* larvae of 1.65 µg TBT-Sn $L^{-1}$ (renewal trial) when exposed to tributyltin chloride.
- Laughlin *et al.* (1988) found that metamorphosis of *M. mercenaria* larvae was completely halted in concentrations 100 ng TBT-Sn $L^{-1}$ or higher (renewal trial over 14 days). These results suggest that such exposure in the field would pose an unacceptable hazard to these fisheries resources. Further research by these authors found a 48-hr $LC_{50}$ for veligers of *M. mercenaria* of between 100-250 ng TBT-Sn $L^{-1}$ (renewal trial) when exposed to tributyltin oxide (TBTO) (Laughlin *et al.* 1989).

**Post-settlement**
- Laughlin *et al.* (1989) found a 25-day $LC_{50}$ for post-settlement larvae of *M. mercenaria* between 7.5-10.0 µg TBT-Sn $L^{-1}$ (renewal trial) when exposed to TBTO.

In addition, Salazar and Salazar (1989) identified that *Protothaca stamina* (the life cycle stage was not recorded) had a 13-day $LC_{50}$ of 110-120 µg TBT-Sn $L^{-1}$ in a static renewal trial of TBTO.

There are no data specific to recruited juveniles or adults. However, Hall & Bushing (1996) (in reviewing a larger set of bivalve toxicity data) identified that in general bivalve adults are very resistant to TBT whereas larval bivalves are very sensitive to it.

Evidence from relevant infaunal inter-tidal bivalve species suggests that TBT is acutely toxic to sensitive life cycle stages (e.g. larvae) of infaunal bivalves at levels as low as 100 ng $L^{-1}$ (i.e. tributyltin oxide). Indirect evidence suggests that exposure to TBT also results in chronic effects on sensitive life cycle stages (i.e. developing bivalve larvae) at concentrations down to $\leq 10\text{-}20$ ng $L^{-1}$ (Cardwell *et al.* 1999).

The dose-response information available for infaunal inter-tidal bivalve species does not include information pertaining to impacts of TBT on adult shellfish. However, some data are available for other bivalve species. In summarizing acute TBT toxicity tests for adult mussels and oysters, Alzieu (1996) found that $LC_{50}$ values ranged between 180-720 µg TBTO $L^{-1}$ when exposure time did not exceed 48-hrs. However, if exposure time exceeded 48-hrs then mortality rates remained high at concentrations under 0.5 µg TBTO $L^{-1}$. Alzieu (1996) further identified that chronic exposure of adult oysters and mussels to TBT concentrations of approx. 1 µg $L^{-1}$ led to significant levels of mortality over a 1-2 month period. A number of condition indices also suggest that long-term exposure to concentrations as low as 16 ng TBT-Sn $L^{-1}$ may affect levels of adult mortality (Henderson 1986).

No clear indication is available for the chronic effects of TBT on the reproductive capacity of infaunal bivalves, although Thain (1986) found some evidence for delayed sexual maturity in oysters exposed to 0.01 µg TBTO L$^{-1}$ for 75 days.

***Organic Booster Biocides:***

There appear to be no data concerning the impact of organic booster biocides on the embryonic, post-settlement, juvenile or adult stages of either New Zealand infaunal inter-tidal species, or similar overseas species.  However, some data regarding larval stages of similar overseas species are available:

- Davis & Hidu (1969) found a 12-day LC$_{50}$ for larval *M. mercenaria* of 5,000 µg L$^{-1}$ (renewal trial) when exposed to the organic booster biocide Diuron.
- The US EPA (2000) found that the organic booster biocide TCMTB intoxicated larval *M. mercenaria* (which made them immobile) at an EC$_{50}$ concentration of 13.9µg L$^{-1}$ in a static 48-hr toxicity test.
- The US EPA (2000) found that the main component (cybutryne) of the organic booster biocide Irgarol 1051 intoxicated larval *Crassostrea virginica* (which made them immobile) at an EC$_{50}$ concentration of 3,200 mg L$^{-1}$ in a static 48-hr toxicity test.

Evidence from relevant infaunal inter-tidal bivalve species thus suggests that Diuron is only moderately toxic to sensitive life cycle stages of infaunal bivalves.  These data also suggest that relatively high levels of TCMTB and Irgarol 1051 in the water column are necessary to cause negative behavioural effects in infaunal inter-tidal bivalve larvae. However, lack of information regarding potential toxicity to adults, potential impacts on growth and reproduction, and possible chronic effects of booster biocides make complete dose-response assessment impossible.

## 5.2.3     Exposure Assessment

Changes in the regulations regarding antifouling substances in recent years are likely to result in a changing profile of risk of exposure of infaunal bivalves to such substances over time.

Evidence of the damaging effects of TBT has prompted action by many countries to regulate or ban their use in antifouling products.  In New Zealand, a partial ban on the use of all organotin antifoulants was introduced in 1983 and a full ban was later introduced in 1993.  Smith (1996) summarised these changes:  *"The Pesticides (Organotin Antifouling Paints) Regulations 1989 specifically targeted pleasure craft, prohibiting the use of all TBT antifoulants on vessels under 25 m in length, but permitted the use of low release copolymer antifoulants on vessels over 25 m in length and/or with an aluminium hull or outdrive. The Pesticides (Organotin Antifouling Paints) Regulations 1993 banned the sale and application of all organotin antifoulants, but had no control over foreign vessels entering the New Zealand Exclusive Economic Zone)"* (Smith 1996).  TBT leached from these overseas vessels will still be infiltrating the coastal marine environment of New Zealand, but it could reasonably be expected that the input of TBT into the marine environment would have been very significantly reduced in recent years.  Conversely it could be expected that the input of organic booster biocides into the marine environment will have increased as alternative antifoulants have replaced those based on organotins.

*Organotins:*

Although the overall level of TBT input to the marine environment has decreased noticeably since the introduction of the organotin antifouling paint regulations, the fact that marine sediments act as a potential store of TBT to be released back into the water column suggests that the impacts of TBT contamination could be on-going.  As previously mentioned, the half-life of TBT in aerobic sediments has been estimated to be 2.5-3 years (de Mora *et al.* 1995), but may be tens of years in anaerobic sediment (Dowson *et al.* 1996).  TBT degradation processes in sediments are likely to be the predominant influence on the overall persistence of TBT in the marine environment (Stewart & de Mora 1990).

Increased concentrations of organotins may be released from sediments through remobilization following dredging or as a result of shipping activity (Smith 1996).  Weather conditions may also affect the level of contaminant impact, as a rough sea state may stir up sediment, thus resuspending contaminants in the sediment back into the water column.

The risk presented to infaunal inter-tidal shellfish populations by antifouling compounds varies spatially across the Hauraki Gulf Marine Park.  Imposex in neogastropods can be used as a sensitive and specific bioindicator of the presence of TBT in seawater (Dr. C. Stewart, *pers. comm.*).  Several studies within the Hauraki Gulf Marine Park may therefore provide an insight as to the spatial distribution of TBT, thus helping to identify the risk to infaunal inter-tidal bivalve populations.  A study of imposex in *Lepsiella scobina* in the Auckland region by Stewart *et al.* (1992) identified a gradient of TBT effects, with the highest levels of imposex occurring in the Waitemata Harbour and decreasing outwards into the Hauraki Gulf.  They related this gradient to levels of boating activity. In this study, imposex effects were measured as far north as Long Bay, while a control site further north at Pakiri Beach showed 0% imposex.  The highest imposex indices were recorded at Milford (where there is a small marina) and Devonport (adjacent to a boat hull washing facility).  High levels of imposex were also found south of the Waitemata Harbour, for example, at Maraetai and on Rakino and Motuihe Islands (Stewart *et al.* 1992).  A more recent study on *Lepsiella scobina* indicated that levels of imposex along the east coast of Auckland were slightly reduced in some populations compared to previous studies, suggesting that ambient TBT levels may be declining (Playfair 1998).  No current data are available.

Antifouling paint residues are present in the Hauraki Gulf Marine Park both in the water column and in coastal sediments.  However, these organotin residues are likely to be found in higher concentrations around wharves, marinas, slipways, dry docks, and other areas with a high level of boating activity (e.g. swing moorings, canal developments etc.) (Krone *et al.* 1989, de Mora *et al.* 1995, Stride 1998 and Evans *et al.* 2000).  Several such areas exist within the Hauraki Gulf Marine Park.  For example, marinas include Gulf Harbour Marina, Westhaven Marina, Half Moon Bay Marina, Pine Harbour Marina, Bayswater Marina, West Park Marina, Whitianga Marina, Coromandel Marina; wharves include: Leigh Wharf, Queen Street Wharf, Princess Wharf, Devonport Naval Base, Devonport Wharf, and Whitianga Wharf etc.

When investigating TBT in sediments of the Waitemata Harbour, De Mora *et al.* (1995) found that the majority of the TBT contamination was confined to marinas.  Stride (1998) identified a gradient of decreasing TBT sediment concentration with increasing distance away from a point source in the Waitemata Harbour (i.e. the Naval dry dock), with low TBT-Sn concentrations (3 ng TBT-Sn g$^{-1}$) occurring as little as 250 m from this point source.  (Note that the New Zealand Navy phased out the use of organotin antifoulants during 1988/89 (Smith 1996).  The Navy has also implemented a wastewater collection and treatment system for the effluent created from the dry dock at the

Devonport Naval Base (de Mora *et al.* 1995)).  TBT can therefore generally be considered as a localized contaminant.  Diffuse contamination may be seen at sites well removed from TBT sources, with decreasing concentrations due to tidal dispersion and *in situ* degradation (Sarradin *et al.* 1991).

Due to the fact that these facilities all require a predominantly sheltered environment in the sub-tidal zone, *P. australis*, *A. stutchburyi*, *M. liliana* and *P. subtriangulata* are not generally found within or in very close proximity to marinas, wharves, commercial slipways or dry docks.  Other areas of high boating activity, including swing moorings, are also sited over areas that remain sub-tidal.  However, infaunal inter-tidal bivalve populations may be found on inter-tidal sand-flats near these sites, for example, *A. stutchburyi* beds within the Tamaki Estuary near Half Moon Bay Marina.  The distance of a particular bivalve population from an individual point source of antifouling paint will affect the level of exposure to that point source.

New input of TBT into the Hauraki Gulf Marine Park is now largely confined to TBT leaching from large commercial ships entering the Auckland Port area.  A large commercial ship leaching TBT at the constant leach rate will contribute more than 200g of TBT to the waters in its immediate vicinity during a three-day port stay (Batley 1996).  Given that there are 8 to 10 ships of this size in the Port of Auckland at any one time, the TBT-Sn contribution to the marine environment is significant (De Mora *et al.* 1995).  Overseas studies have also confirmed that in areas of high shipping activity elevated concentrations of TBT (e.g. 61 ng $L^{-1}$) are still observed (Thomas *et al.* 2001).  However, this input of TBT may decline in the future, as the International Maritime Organisation proposes to prohibit all antifouling using TBT by 2003, and to phase out the presence of TBT on ships' hulls by 2008 (Maguire 2000).  A study of imposex in the whelk *Lepsiella scobina* in the Wellington region by Smith (1996) identified a significant decline in the percentage of imposex in a recreational boating area (Porirua Inlet) after the introduction of the 1989 Pesticide Regulations.  In contrast, in Wellington Harbour (a commercial vessel port) very little decline in imposex was observed.  Smith (1996) suggested that this may relate to the ongoing input of TBT into Wellington Harbour by larger commercial vessels.  (This is difficult to establish definitively from available information as no records are kept of antifoulants on foreign vessels entering New Zealand ports (Smith 1996).  We also note that TBT is also still used in timber treatment processes.)

While TBT levels in the water column in large marinas in Britain during the 1980's sometimes exceeded 1 µg $L^{-1}$ (but were generally around 300-600 ng TBT-Sn $L^{-1}$) (Waite *et al.* 1991), TBT levels in seawater in the Auckland region in 1989 were generally in the range 180-320 ng TBT-Sn $L^{-1}$ at heavily impacted sites (e.g. Westhaven Marina and Tamaki Estuary (Half Moon Bay Marina) (King *et al.* 1989).  Values at the higher end of this New Zealand range exceed the concentrations at which oyster deformities have been observed in overseas studies (e.g. Alzieu *et al.* 1986).  However, out of 19 reported water samples in the King *et al.* (1989) study, only 10 exceeded the limit of quantification suggesting that the remaining 9 samples were <16 ng TBT-Sn $L^{-1}$ (King *et al.* 1989).  It should also be noted that the level of TBT in the water column decreased considerably over very short sampling distances (e.g. from 321 ng $L^{-1}$ at Half Moon Bay Marina to 16 ng $L^{-1}$ approx. 500 m upstream.  This example again illustrates a key point when investigating the impact of TBT on infaunal inter-tidal bivalve populations, namely that TBT can be considered a localised contaminant.

TBT is now more commonly measured in shellfish and sediments than in water.  We were unable to find any recent data for TBT levels in seawater within the Hauraki Gulf Marine Park.  This may be due to the fact that TBT is now recognised as being relatively transient in the water column, and

also that levels measured vary depending on the state of the tide. TBT is also more difficult to measure in seawater (Dr. C. Stewart, *pers. comm.*).

In the 1980's maximum levels of TBT measured in overseas marina sediments were generally between 500-1000 ng g$^{-1}$ (Clark *et al.* 1988). By contrast, a study by King *et al.* (1989) found that the concentration of TBT in sediments were high near Half Moon Bay marina (i.e. 150-240 ng TBT-Sn g$^{-1}$), but only 1-5 ng TBT- Sn g$^{-1}$ at other sites in the Tamaki Estuary. The authors considered that these levels were not excessive in comparison to levels recorded internationally (King *et al.* 1989). TBT levels measured in *Crassostrea gigas*, *Saccostrea glomerata*, *Ostrea angasi* (i.e. the dredge oyster *Tiostrea chilensis*) and *Lepsiella scobina* from the Tamaki Estuary were three to four orders of magnitude higher than levels in the surrounding waters (King *et al.* 1989). However, only *C. gigas* showed evidence of shell thickening.

The distribution of hexane extractable tin (Hex-Sn) (this may include butyl, phenyl and cyclohexyltin) in marine sediments near Auckland was further evaluated by de Mora *et al.* (1995) approximately 1-2 years after the introduction of regulations limiting the use of organotin antifouling paints in New Zealand. Hotspots of contamination were found near washdown facilities (marinas, naval dry dock) with concentrations generally between 300 and 800 ng g$^{-1}$ Hex-Sn. Lower values were found in swing mooring areas (e.g. 4-40 ng g$^{-1}$ Hex-Sn). Commercial wharves had values of 20-70 ng g$^{-1}$ Hex-Sn. A transect across the Tamaki Strait from Rocky Bay (Waiheke Island) to Maraetai Beach had values of <2-12 ng g$^{-1}$ Hex-Sn.

Levels of TBT, DBT, and TPhT were also measured in depositional zone sediment samples in the ARC *Marine Sediment Monitoring Programme* in 1999 (Mills *et al.* 2000). Levels of TPhT were generally below the level of detection of the analytical method used. DBT levels were either below the level of detection, or in the range 2-7 ng Sn g$^{-1}$. TBT levels were either below the level of detection, or in the range 2-11 ng Sn g$^{-1}$ (Mills *et al.* 2000).

In addition to bioavailability, the routes of uptake of organotins are also relevant to assessment of the exposure of infaunal inter-tidal shellfish in the Hauraki Gulf Marine Park. As mentioned earlier, TBT has been assumed to be assimilated by bivalves predominantly from the water soluble fraction through respiratory membranes, but some may also be ingested with food (through contaminated phytoplankton or sediment) (Cardwell *et al.* 1999). The extent to which feeding mode (i.e. deposit-feeding *vs.* suspension-feeding) influences the risk of uptake of organotin does not appear to have been widely investigated. However, Langston & Burt (1991) investigated the importance of water and sediment as pathways for TBT uptake in the benthic infaunal deposit-feeding clam *Scrobicularia plana*. This study identified that TBT body burdens in this deposit-feeding species are dominated by assimilation of sediment-bound TBT (Langston & Burt 1991). A similar study (with water-column TBT at 100 ng Sn L$^{-1}$ and sediment TBT at 0.8 µg Sn g$^{-1}$ (DW) over a 60 day period) using adults of the suspension-feeding infaunal bivalve *Ruditapes decussatus* identified that water-column uptake was the major source for this species, while negligible TBT-uptake was associated with the sediment (Coelho *et al.* 2002a). These studies identify that feeding habit is important in terms of determining which TBT exposure pathway poses the most risk to infaunal bivalves. Therefore, it is possible that the risk of toxicity from sediment-bound TBT is greater with respect to the deposit-feeding *M. liliana,* while water-column TBT may have a higher risk for the suspension-feeding species such as *A. stutchburyi*, *P. australis* and *P. subtriangulata*. Further research is necessary to investigate the comparative level of TBT uptake from water *versus* sediment in the infaunal bivalve species of interest in this study.

Laughlin *et al.* (1988) suggest that it is quite plausible that the primary route of exposure of bivalve larvae to TBT is through consumption of TBT contaminated phytoplankton. However, for adult suspension-feeding bivalves this may not be the case. A study of TBT accumulation in adults of the suspension-feeding infaunal bivalve *Ruditapes decussatus*, by Coelho and co-workers (2002b) estimated that accumulation of TBT from $^{14}$C-TBT labeled phytoplankton (again over a 60 day period) was an order of magnitude lower than from TBT-spiked sediment, and two orders of magnitude lower than direct accumulation from the water-column (see results of Coelho *et al.* 2002a above). With reference to these results Coelho *et al.* (2002b) note that it was possible that the microalgal food supply was limiting in these experiments, and suggest that in nature, algal densities could be significantly higher, and may accordingly contribute significantly more to the TBT load available to suspension-feeding infaunal clams. There have been no studies of TBT contamination of phytoplankton undertaken in New Zealand, so the likely impact on either larval or adult stages of infaunal bivalves currently remains unknown.

### Organic Booster Biocides:

There is currently very little information about the presence of organic booster biocides in the Hauraki Gulf Marine Park, although Chlorothalonil has been found in sediment near Half Moon Bay (Dr. C. Stewart, *pers. comm.*). However, a study of the occurrence of two organic booster biocides (Diuron and Irgarol 1051) in coastal waters with high levels of boating activity, including the Hauraki Gulf Marine Park region, has recently been undertaken by Dr. Carol Stewart (2003) for the Ministry for the Environment. This report is soon to be released. It will provide significant data with respect to the levels and spatial distribution of organic booster biocides in the marine environment. Until that report is released, it is difficult to assess the exposure of infaunal inter-tidal bivalves in the Hauraki Gulf Marine Park to organic booster biocides. Currently it would be appropriate to assume that the level of booster biocides entering the marine environment is likely to have increased in recent years due to their use in antifouling paint formulations, and that it is likely that booster biocides are present within the Hauraki Gulf Marine Park.

Whether or not organic booster biocides can be considered as localised contaminants requires not only further information about their dispersal, but also further investigation of background inputs of each booster biocide into the marine environment from land-based uses (e.g. herbicide use).

As with organotins, increased concentrations of organic booster biocides may be released from the sediments following remobilisation as a result of dredging, shipping activity or rough weather. For example, increased levels of Irgarol 1051 were recorded following dredging in Brighton marina (U.K.) (Bowman *et al.* 2003).

Like organotins, dissolved organic booster biocides in the water fraction are probably assimilated by aquatic organisms via the gills and other respiratory membranes. There appears to be a dearth of information concerning the potential effects of sediment-bound booster biocides, or the possible effects of consumption of phytoplankton contaminated by organic booster biocides by infaunal bivalves. Neither is information available about the comparative risks of intoxication of deposit-feeders (e.g. *M. liliana*) *versus* suspension-feeders (*A. stutchburyi*, *P. australis* and P. *subtriangulata*) arising from organic booster biocides.

## 5.2.4    Risk Characterisation

*Organotins:*

There appear to have been no studies undertaken on the impacts of organotins on *A. stutchburyi*, *P. australis*, *P. subtriangulata* or *M. liliana*) in New Zealand.  However, based on data from overseas species, potential impacts may include decreased reproductive capacity, larval malformation, larval mortality, enhanced susceptibility to disease, and genotoxic effects.  Indirect effects may also occur through the impact of organotins on phytoplankton populations.

Organotins are characterised by relatively low persistence in the water column (days to weeks – Stewart & de Mora 1990).  It appears that the predominant route of uptake of organotins by infaunal bivalves may be via the fraction dissolved in the water column (Cardwell *et al.* 1999).  Following the introduction of regulations limiting the use of organotins in antifoulants, there may be a trend of decreasing risk of the presence of these compounds in the water column (Playfair 1998).  However input from antifoulants on large commercial vessels from overseas has continued (although this will reduce in the near future).  There are no current data on levels of organotins present in the water column in the Hauraki Gulf Marine Park.

Organotins are also characterised by long persistence times in the sediment.  This means that continuing input into the marine environment results in increasing levels, and a persistent legacy of potential toxicity.  Sediment disturbance (for example, due to dredging, shipping activities and rough weather) can cause resuspension and subsequent desorption of organotin compounds from sediment particles, returning organotins back into the water column.  Assuming that there is no continuing input of organotins into the marine environment, continuing deposition of uncontaminated sediment could result in burial of contaminated sediments, thus over time reducing the risk of their resuspension into the water column.  Variation in sample sites, sampling methods and analysis methods make it difficult to determine whether there has been any decrease in sediment contamination levels in those areas that were previously contaminated, but are remote from any continuing input from overseas commercial shipping.  Studies by King *et al.* (1989) and de Mora *et al.* (1995) found 240 ng/g TBT and 166-334 ng/g Hex-Sn respectively in sediments at Half Moon Bay Marina (note however that different analytical extraction methods were used).

Sediment bound organotins are more likely to be accumulated by deposit-feeding infaunal bivalves than by suspension-feeders.  In contrast, suspension-feeders tend to accumulate TBT predominantly from the dissolved water-column phase.

There is evidence to suggest that the distribution of significant levels of organotins in both the water column and sediment is localised to areas with high levels of boating activity.  None of the infaunal bivalves of particular interest to the Hauraki Gulf Forum are likely to be present in high numbers in such areas, although some populations may be located very close to areas of high boating activity.  However, the general absence of direct habitat overlap of infaunal bivalves with the point source of contamination does not preclude the possibility of significant impacts on an inter-tidal infaunal bivalve population arising from organotins in the environment.  The larval stages of bivalves are the most sensitive, and being planktonic, are not confined to one location.  In this phase of their life cycle it is possible that they could be impacted by TBT in the water column through drifting into contaminated areas.  Levels of TBT in the water column in 1989 were in the range 180-320 ng TBT-Sn L$^{-1}$ at heavily impacted sites such as Westhaven Marina and Tamaki Estuary (King *et al.* 1989).  Assuming (in the absence of New Zealand data) that the dose-response data for *M. mercenaria* arising from overseas studies (Laughlin *et al.* 1988, 1989) is broadly applicable to

similar New Zealand species, these levels of TBT would have been sufficient to severely impact on larval survival of our infaunal bivalve species. In the absence of data regarding the presence of organotins in the water column, the extent to which this is still a risk is unknown.

The dearth of information on potential chronic effects (e.g. effects on fecundity, metabolism, growth rate, increased parasite load etc.) of organotins on the juvenile and adult stages of New Zealand inter-tidal infaunal species make definitive evaluation of the magnitude of these impacts impossible. Further information about the response of these species to organotins, along with information about the current levels of organotins in the water column, combined with better understanding of the dynamics impacting on water:sediment partitioning and dispersal of organotins in the Hauraki Gulf Marine Park (including the Coromandel area, for which there are few data), would assist in better assessment of this risk.

### *Organic Booster Biocides:*

Quantitative assessment of the risk presented by organic booster biocides to infaunal inter-tidal shellfish populations in the Hauraki Gulf Marine Park is not possible due to lack of information. However, the following points are relevant to this risk:

- There are minimal data available regarding the effects of organic booster biocides on all non-target marine organisms. Specifically, information on the potential impacts of organic booster biocides (including acute and chronic impacts) on adult stages of infaunal inter-tidal species is required, including potential impacts on growth, and reproductive impairment. Risk may vary with shellfish species, based not only on the physiological responses of the shellfish, but also upon mode of feeding.

- The limited data from overseas species of infaunal bivalves indicate that the organic booster biocides tested are less toxic to larval stages than TBT. Lack of information about levels of organic booster biocides in the water and sediment in the Hauraki Gulf Marine Park, and the lack of data specific to New Zealand infaunal inter-tidal bivalve species, means that it is currently not possible to asses the risk presented by these compounds to shellfish populations in the HGMP.

- Although organic booster biocides are likely to enter the marine environment in areas associated with boating activity there are no current data available on the extent to which these compounds are dispersed from their source. In addition, the extent of input from land-based uses is currently unknown. The spatial distribution of the potential risk presented by organic booster biocides is thus currently unknown.

- With the introduction of organic booster biocides into antifoulants, the input of these compounds into the marine environment is likely to have increased. The rate at which these compounds build up in the marine environment depends on their persistence and input. Whether or not this will result in a trend of increasing risk to infaunal inter-tidal shellfish populations is currently unknown.

There is clearly substantial further research required to determine the risk of impact of organic booster biocides on infaunal inter-tidal shellfish in the Hauraki Gulf Marine Park. The report prepared for the Ministry of the Environment by Dr. C. Stewart will add significantly to the current body of knowledge, and it is recommended that this risk be reassessed when that information becomes available.

# 5.2.5 References

**Awadhesh, H. J., J. A. Hagger & S. J. Hill  2000.**  Tributyltin induces cytogenetic damage in the early life stages of the marine mussel, *Mytilus edulis*. *Environmental and Molecular Mutagenesis*.  35: 343-350.

**Alzieu, C., J. Sanjuan, J. P. Deltriel & M. Borel  1986.**  Tin contamination in Arcachon Bay: effects on oyster shell anomalies. *Marine Pollution Bulletin*.  17: 494-498.

**Alzieu, C.  1996.**  Biological effects of tributyltin on marine organisms.  *In*: Tributyltin: Case study of an environmental contaminant.  S. J. de Mora (Ed.). Cambridge Environmental Chemistry Series Vol. 8.  Cambridge University Press, Great Britain.  Pp. 167-211.

**Anderson, R. S., M. A. Unger & E. M. Burreson  1996**.  Enhancement of *Perkinsus marinus* disease progression in TBT-exposed oysters (*Crassostrea virginica*). *Marine and Environmental Research*.  42:177-180.

**Batley, G.,  1996.**  The distribution and fate of tributyltin in the marine environment.  *In:* Tributyltin: case study of an environmental contaminant. S. J. de Mora (Ed.).  Cambridge Environmental Chemistry Series Vol. 8. Cambridge University Press. Great Britain.  Pp. 139-166.

**Blunden, S. J. & C. J. Evans  1990.**  Organotin compounds.  *In*: The Handbook of Environmental Chemistry.  O. Hutzinger (Ed.).  Springer-Verlag, Berlin.  Pp. 1-44.

**Bowman, J. C., J. W. Readman & J. L. Zhou  2003.**  Seasonal variability in the concentrations of Irgarol 1051 in Brighton marina, UK, including the impact of dredging. *Marine Pollution Bulletin*.  46(4): 444-451.

**Cardwell, R. D., M. S. Brancato, J. Toll, D. DeForest & L. Tear  1999.**  Aquatic ecological risks posed by tributyltin in United States surface waters: Pre-1989 to 1996 data. *Environmental Toxicology and Chemistry*.  18(3): 567-577.

**Cima, F., M. G. Marin, V. Matozzo, L. Da Ros & L. Ballarin  1998.**  Immunotoxic effects of organotin compounds in *Tapes philippinarum*. *Chemosphere*.  37(14-15): 3035-3045.

**Clark, E. A., R. M. Sterritt & J. N. Lester  1988.**  Fate and distribution of TBT in the aquatic environment. *Environmental Science and Technology*.  22: 600-605.

**Coelho, M. R., M. J. Bebianno & W. J. Langston  2002a.**  Routes of TBT uptake in the clam *Ruditapes decussatus*. I. Water and sediments as vectors of TBT uptake. *Marine Environmental Research*.  54: 179-192.

**Coelho, M. R., M. J. Bebianno & W. J. Langston  2002b.**  Routes of TBT uptake in the clam *Ruditapes decussatus*. II. Food as vector of TBT uptake. *Marine Environmental Research*.  54: 193-207.

**Davis, H. C. & H. Hidu  1969.**  Effects of pesticides on embryonic development of clams and oysters and on survival and growth of larvae. *Fisheries Bulletin*.  67(2): 393-404.

**de Mora, S. J., N. C. Stewart & D. Phillips  1995.**  Sources and rate of degradation of tri(n-butyl)tin in marine sediments near Auckland, New Zealand. *Marine Pollution Bulletin*.  30(1): 50-57.

**de Mora, S. J., S. W. Fowler, R. Cassi & I. Tolosa  2003.**  Assessment of organotin contamination in marine sediments and biota from the Gulf and adjacent regions. *Marine Pollution Bulletin*.  46: 401-409.

**Dowson, P. H., J. M. Bubb & J. N. Lester  1996.**  Persistence and degradation pathways of tributyltin in freshwater and estuarine sediments. *Estuarine, Coastal and Shelf Science*.  42: 551-562.

**Elgethum, K., C. Neumann & P. Blake  2000.**  Butyltins in shellfish, finfish, water and sediment from the Coos Bay estuary (Oregon, USA). *Chemosphere*.  41: 953-964.

**Evans, S. M., A. C. Birchenough & M. S. Brancato  2000**.  The TBT Ban: Out of the frying pan into the fire. *Marine Pollution Bulletin*.  40(3): 204-211.

**Fent, K.  1996.**  Ecotoxicology of organotin compounds . *CRC Critical Review of Toxicology*.  26: 1-117.

**Fent, K.  2003.**  Ecotoxicological problems associated with contaminated sites. *Toxicology Letters*.  140-141: 353-365.

**Gibbs, P. E. & G. W. Bryan  1996.**  TBT-induced imposex in neogastropod snails: masculinization to mass extinction.  *In*: Tributyltin: Case study of an environmental contaminant.  S. J. de Mora (Ed.).  Cambridge University Press, Cambridge.  Pp. 212-236.

**Hall, L. W., Jr. & S. J. Bushing  1996.**  A review of acute effects of tributyltin compounds on aquatic biota.  *In*: Organotin. M. A. Champ & P. F. Seligman (Eds.). Chapman & Hall, London. Pp. 157-190.

**Henderson, R. S. 1986**.  Effects of organotin antifouling paint leachates on Pearl Harbour organisms: a site specific flowthrough bioassay.  *In:* Proceedings of the Organotin Symposium, Oceans 86 Conference, Washington, Sept. 23-25, 1986.  4: 1226-1237.

**King, N. G., M. C. Miller, C. J. de Mora  1989.**   Tributyltin levels for seawater, sediment, and selected marine species in coastal Northland and Auckland, New Zealand. *New Zealand Journal of Marine and Freshwater Research*.  23: 287-294.

**Krone, C. A., D. W. Brown, D. G. Burrows, S. L. Chan & U. Varanasi  1989.**  Butyltins in sediment from marinas and waterways in Puget Sound, Washington State, U.S.A. *Marine Pollution Bulletin*.  20: 528-531.

**Langston, W. J. & G. R. Burt  1991.**  Bioavailability and effects of sediment-bound TBT in deposit-feeding clams, *Scrobicularia plana*. *Marine Environmental Research*.  32: 61-77.

**Lapota, D., D. E. Rosenberger, M. F. Platter-Rieger & P. F. Seligman 1993.** Growth and survival of *Mytilus edulis* larvae exposed to low levels of dibutyltin and tributyltin. *Marine Biology*. 115: 413-419.

**Laughlin, R. B. Jr., R. Gustafson, & P. Pendolcy 1988.** Chronic embryo-larval toxicity of tributyltin (TBT) to the hard shell clam *Mercenaria mercenaria*. *Marine Ecology Progress Series*. 48: 29-36.

**Laughlin, R. B. Jr., R. Gustafson, & P. Pendolcy 1989.** Acute toxicity of tributyltin (TBT) to early life history stages of the hard shell clam *Mercenaria mercenaria*. *Bulletin of Environmental Contamination and Toxicology*. 42(3): 352-358.

**Looser, P. W., K. Fent, M. Berg, G. H. Goudsmit & R. P. Schwarzenbach 2000.** Uptake and elimination of triorganotin compounds by larval midge *Chironomus riparius* in the absence and presence of Aldrich humic acid (AHA). *Environmental Science and Technology*. 34: 5165-5171.

**Maguire, R. J. 2000.** Review of the persistence, bioaccumulation and toxicity of tributyltin in aquatic environments in relation to Canada's toxic substances management policy. *Water Quality Research Journal of Canada*. 35(4): 633-679.

**Matthiessen, P. & P. E. Gibbs 1998.** Critical appraisal of the evidence for tributyltin-mediated endocrine disruption in mollusks. *Environmental Toxicology and Chemistry*. 17(1): 37-43.

**Mills, G. N., S. E. Pickmere, G. M. Olsen & R. B. Williamson 2000.** Marine Sediment Monitoring Programme: 1999 Results. NIWA Client Report ARC00253, June 2000. Auckland Regional Council Technical Publication No. 135. October 2000.

**Morcillo, Y., M. J. J. Ronis & C. Porte 1998.** Effects of tributyltin on the Phase I testosterone metabolism and steroid titres of the clam *Ruditapes decussata*. *Aquatic Toxicology*. 42: 1-13.

**Omae, I. 2003.** Organotin antifouling paints and their alternatives. *Applied Organometallic Chemistry*. 17: 81-105.

**Phinney, J. T. & K. W. Bruland 1997.** Trace metal exchange in solution by the fungicides Ziram and Maneb (Dithiocarbamates) and subsequent uptake of lipophilic organic zinc, copper and lead complexes into phytoplankton cells. *Environmental Toxicology and Chemistry*. 16(10): 2046-2053.

**Playfair, J. 1998.** The incidence of imposex in *Lepsiella scobina* (Quoy and Gaimard, 1833) on the Auckland east coast. Unpubl. MSc. thesis. School of Biological Sciences. University of Auckland. Pp. 116.

**Roberts, M. H. J. 1987.** Acute toxicity of tributyltin chloride to embryos and larvae of two bivalve mollusks, *Crassostrea virginica* and *Mercenaria mercenaria*. *Bulletin of Environmental Contamination and Toxicology*. 39: 1012-1019.

**Roberts, M. H., M. E. Bender, P. F. DeLisle, H. C. Sutton & R. L. Williams 1987.** Sex ratio and gamete production in the American oyster exposed to tributyltin in the laboratory. *In*: Oceans '87 Proceedings of the International Organotin Symposium, Vol. 4. Institute of Electrical and Electronic Engineers, New York. Pp. 1471-1476.

**Ruiz, J. M., G. W. Bryan & P. E. Gibbs 1995a.** Effects of tributyltin (TBT) exposure on the veliger larvae development of the bivalve *Scrobicularia plana* (da Costa). *Journal of Experimental Marine Biology and Ecology*. 186(1): 53-63.

**Ruiz, J. M., G. W. Bryan & P. E. Gibbs 1995b.** Acute and chronic toxicity of tributyltin (TBT) to pediveliger larvae of the bivalve *Scrobicularia plana*. *Marine Biology*. 124(1): 119-126.

**Salazar, M. H. & S. M. Salazar 1989.** Acute toxicity of (bis)tributyltin oxide on marine organisms. Technical Report No. 1299, Naval Ocean Systems Centre, San Diego, CA.

**Sarradin, P. M., A. Astruc, V. Desauziers, R. Pinel & M. Astruc 1991.** Butyltin pollution in surface sediments of Arcachon Bay after ten years of restricted use of TBT-based paints. *Environmental Technology*. 12: 537-543.

**Smith, P. J. 1996.** Selective decline in imposex levels in the dogwhelk *Lepsiella scobina* following a ban on the use of TBT antifoulants in New Zealand. *Marine Pollution Bulletin*. 32(4): 362-365.

**Stenalt, E., B. Johansen, S. V. Lillienskjold & B. W. Hansen 1998.** Mesocosm study of *Mytilus edulis* larvae and postlarvae, including settlement phase, exposed to a gradient of tributyltin. *Ecotoxicology and Environmental Safety*. 40: 212-225.

**Stewart, C. 2003.** Antifouling biocides in New Zealand coastal waters. A report prepared for the Ministry for the Environment, Wellington. April 2003.

**Stewart, C. & S. J. de Mora 1990.** A review of the degradation of tri(n-butyl)tin in the marine environment. *Environmental Technology*. 11: 565-570.

**Stewart, C., S. J. de Mora, M. R. L. Jones & M. C. Miller 1992.** Imposex in New Zealand Neogastropods. *Marine Pollution Bulletin*. 24(4): 204-209.

**Stride, A. D. 1998.** An investigation of organotin contaminants in the environs of HMNZ dockyard, Auckland, New Zealand. Unpubl. MSc. thesis. Schools of Environmental Science & Chemistry. 309 pp.

**Thain, J. E. 1986.** Toxicity of TBT to bivalves: effect on reproduction, growth and survival. *In*: Oceans '86 Proceedings of the International Organotin Symposium, Vol. 4. Institute of Electrical and Electronic Engineers, New York. Pp. 1306-1313.

**Thomas, K. V., T. W. Fileman, J. W. Readman & M. J. Waldock 2001.** Antifouling paint booster biocides in the UK coastal environment and potential risks of biological effects. *Marine Pollution Bulletin*. 42(8): 677-688.

**Unger, M. A., W. G. MacIntyre & R. J. Huggett  1988.**  Sorption behaviour of tributyltin on estuarine and freshwater sediments. *Environmental Toxicology Chemistry*.  7: 907-915.

**United States Environmental Protection Agency  2000.**  Environmental Effects Database, Office of Pesticide Programs, Environmental Fate and Effects Division, U.S. EPA, Washington D.C.

**Voulvoulis, N., M. D. Scrimshaw & J. N. Lester  1999.**  Alternative antifouling biocides.  *Applied Organometallic Chemistry*.  13: 135-143.

**Voulvoulis, N., M. D. Scrimshaw & J. N. Lester  2002.**  Comparative environmental assessment of biocides used in antifouling paints. *Chemosphere*.  47: 789-795.

**Waite, M. E., M. J. Waldock, J. E. Thain, D. J. Smith & S. M. Milton  1991.**  Reductions in TBT concentrations in UK estuaries following legislation in 1986 and 1987. *Marine Environment Research*.  32: 89-111.

**Wolinakowski, K. U., M. D. Stephenson & G. S. Ichikowa  1987**.  Tributyltin concentrations and oyster deformation in Coos Bay, Oregon.  *In*: Oceans '87 Proceedings of the International Organotin Symposium, Vol. 4. Institute of Electrical and Electronic Engineers, New York.  Pp. 1438-1443.

**Zhao, L., X. Lu & S. Bingyi  1990.**  Toxic effects of organotin on marine diatoms. *Journal of the Ocean University of Qingdao*.  20(4): 125-131.

## 5.3  STRESSOR:          HEAVY METALS

### 5.3.1      Hazard Identification

Metals are natural constituents of the environment.  However, in recent years, concern has increased internationally over anthropogenic metal pollution in the coastal marine environment (Stecko & Bendell-Young 2000, Keppler & Ringwood 2002).  Elevated heavy metal concentrations can occur throughout the coastal zone due to urbanisation and various anthropogenic activities, with estuaries being amongst the regions most heavily polluted by metals (Forstner & Wittmann 1981).  For some metals, natural and anthropogenic inputs are of the same order (e.g. mercury and cadmium), whereas for others (e.g. copper, lead) inputs due to human activities dwarf natural inputs (Clark 2001).  Recent studies (Beiras & Albentosa 2003) suggest that some heavy metals (e.g. Cu) are already causing deleterious effects on populations of commercial bivalves overseas (e.g. Galician coast, Spain) through inhibition of embryo development.  The aquatic marine environment of New Zealand is also subject to anthropogenic sources of heavy metals.  A survey of the sources of heavy metal inputs into the New Zealand aquatic environment by Smith (1986) identified that for many metals (e.g. copper, nickel, lead and zinc) anthropogenic inputs had exceeded natural inputs by a significant margin, but that it is difficult to separate the two sources.

But what does the term 'heavy metal' really mean?  Although initial definitions of heavy metals were based upon the density of the elemental form of the metal (i.e. metals $> 7g/cm^3$ were included in the heavy metal category), it was soon realised that the density or specific gravity of the metal was not of great significance in relation to predicting the reactivity of a particular metal (Duffus 2001).  In more recent times, the term 'heavy-metals' has been used as a group name for metals and semimetals (metalloids) that have been associated with contamination and potential toxicity or eco-toxicity (Duffus 2001).  Whilst noting the lack of any cohesive chemical rules for metals defined by this term, it is the latter definition that we employ in our review of heavy metals as stressors to infaunal bivalve species.  Specifically, we include the metals cadmium (Cd), chromium (Cr), copper (Cu), nickel (Ni), lead (Pb) and zinc (Zn).

Sources of anthropogenic heavy metal contamination into the Hauraki Gulf Marine Park are outlined below.

- Heavy metals may be derived from industrial discharges.  These discharges may result in high sediment concentrations of heavy metals in adjacent estuaries (higher than those associated with stormwater discharges) (Morrisey *et al.* 2000).  The risk associated with this source of heavy metal input into the marine environment cannot be fully assessed at this time as (being beyond the scope of this project) we have not reviewed the results of associated resource consent monitoring for each potential industrial heavy metal discharge.

- Heavy metals may also be introduced into the marine environment via stormwater runoff.  Metals of concern due to increased concentration in New Zealand urbanised estuaries via stormwater runoff include Cd, Cr, Cu, Ni, Pb, and Zn (Williamson & Morrisey 2000).  Concentrations of Zn, Pb, and Cu found in sediments of most urbanised New Zealand harbours and estuaries that are receiving environments for stormwater discharge are considered high enough to be of ecological concern (Morrisey *et al.* 2000).  In New Zealand urban areas, stormwater is the largest source of toxic contaminants into receiving environments (Le Fevre & Silyn Roberts 2002).  As in many other cities with relatively

steep topography, Auckland's stormwater catchments are superimposed onto the natural stream network, which in turn drains into each sub-estuary (Hume 1983). Copper in stormwater may be derived from vehicle component wear (e.g. brake linings) (Le Fevre & Silyn Roberts 2002), while lead is generally derived from vehicle exhaust emissions (Blasco & Puppo 1999). However leaded fuel was phased out in New Zealand in 1996. Zinc contained in stormwater may be derived from vehicle tire rubber (Glasby *et al.* 1988) and engine additives (Le Fevre & Silyn Roberts 2002). Copper is also introduced into the marine environment through its use in copper-based antifouling paints (Hall & Anderson 1999). Zinc may also be introduced from this source.

- Heavy metals may also enter the marine environment from the following anthropogenic sources: municipal waste (sewage), pesticides, residues, manure, and sludges (Lewis 1995).

As identified above, heavy metals may be potentially available to infaunal bivalves via several anthropogenic sources. These sources of heavy metals may be presented to infaunal bivalves by means of several different exposure pathways, for example,
- from the water column (i.e. either dissolved, colloidal, or sorbed to suspended particulate matter),
- from the deposited sediment (including pore water), or
- via a food source (e.g. phytoplankton).

High proportions of the major contaminants in stormwater are adsorbed onto particulate matter, and reach the estuary in that form (Williamson & Mills 2002). Sheltered estuaries and harbours are therefore susceptible to heavy metal accumulation because particle flocculation promotes the settlement of the particulate matter. Consequently, coastal sediments are now known to act as large, concentrated, storage reservoirs for metals derived from deposition of particles suspended in the water column (Griscom *et al.* 2000, Ciutat & Boudou 2003). The importance of sediments in controlling water quality either as a sink, or as a secondary source, is well recognised, as is the bioavailability of sedimentary contaminants to bottom dwelling organisms (Birch & Taylor 2002).

However, heavy metals <u>in sediments</u> undergo a number of complex geochemical reactions that are dependent on a variety of processes in sediments at different depths (Griscom *et al.* 2000). Metals behave differently in oxic versus anoxic sediments. In anoxic sediments, metal distributions are dominated by sorption to and substitution with $Fe_xS_y$. For example, cadmium is a sulphur seeking metal, and would rapidly coprecipitate in anoxic sediments with $Fe_xS_y$. However, Cr is an oxygen seeking metal, which would associate with iron/manganese oxides (Griscom *et al.* 2000). Further, reactive acid volatile sulphides (AVS) can control pore water concentrations of metals such as Cd, Cu, Ni and Zn (Hansen *et al.* 1996). Increased oxygen fluxes near organisms in anoxic sediments (because of food intake and inhalation processes) may stimulate the mobilisation of some heavy metals (e.g. Cu) and these metals may become potentially available to organisms (Gerringa 1991). Although initial studies of oxic sediments suggested complex geochemistry effects on heavy metal bioavailability, recent studies are more optimistic about the occurrence of consistent geochemical influences on availability (Griscom *et al.* 2000). It has been suggested that total organic carbon (TOC) consistently reduces metal bioavailability (toxicity) in oxic sediments where AVS do not limit elevated pore water metal concentrations (Ankley *et al.* 1996). Sandy sediments generally have a lower TOC content than muddy sediments, suggesting that bioavailability of heavy metals will be lower in muddy sediments than in sandy ones (Timperley & Mathieson 2002). Based on laboratory experiments and field translocation studies, Hummel *et al.* (1998) suggested that the level of oxygenation of the sediments was only indirectly related to, and not directly influencing, the copper content of *Macoma balthica*. Further, no relationship was found between the oxygen

content of the sediment and *Cerastoderma edule* copper content. Heavy metal bioavailability in sediments may also change due to redox cycling, delivery of fresh organic matter from the water column, bioturbation (i.e. sediment mixing activity caused by benthic organisms), and resuspension events (Griscom *et al.* 2000). Bioturbation by infaunal bivalves and other benthic organisms can transfer metals from the sediment to the water column (Ciutat & Boudou 2003). In laboratory studies, bioturbation led to a significant release of Cd or Zn from the sediment, mainly in particulate form (only 0.9% of the Cd was in the dissolved form, while < 2% of the Zn was in the dissolved form) suggesting that the bioavailable (dissolved) fraction remained low (Ciutat & Boudou 2003). The main bioturbator in estuaries around Auckland is the mud crab, *Helice crassa*, with numbers typically reach up to 500 m$^2$ (Morrisey *et al.* 1999)

Griscom *et al.* (2000) identified that the metals (Cd, Co, Se, Zn and Ag) were less available from anoxic than oxic sediments for the tellinid *Macoma balthica*, but that metals in anoxic sediments were still bioavailable. Griscom *et al.* (2000) in investigating the effect of sediment ageing and metal bioavailability suggested that decreased metal bioavailability with time might be explained by a progressive redistribution of the heavy metals into more resistant phases (more complicated binding to sediment particles) in the sediment. In the natural marine environment, metal bioavailability from sediment might be relatively low, in spite of the often very high sediment concentrations (Absil 1993).

A comparison of deposited sediment (DS) versus suspended particulate matter (SPM) in the Fraser River Estuary, Canada, by Stecko & Bendell-Young (2000) identified that SPM contained greater amounts of organic matter, reducible iron (iron oxide) and total metal concentrations (Zn, Cu, Cd and Pb), with a greater proportion of these metals available in reduced forms compared to DS. These differences in potential bioavailability between DS and SPM are important to infaunal bivalves in regard to sediment ingestion and potential toxicity of heavy metals. However, the continuous creation of manganese and iron oxides at the sediment/water interface will precipitate metals such as cadmium in both DS and SPM (Stecko & Bendell-Young 2000).

It has also been clearly established that phytoplankton cells provide an important mode of entry for heavy metals into bivalves (Wikfors & Ukeles 1982, Wikfors *et al.* 1994, Absil *et al.* 1994). Experimental evidence that heavy metals are transferred from metal contaminated microalgae to suspension-feeders is considerable (e.g. Graney *et al.* 1984, LaBreche *et al.* 2002).

The availability and toxicity of metals to organisms <u>in the water</u> depends on the chemical speciation of the metal in that particular aquatic environment. It has been shown that for some metals (i.e. Zn, Cu and Cd) the bioavailable and potentially toxic chemical species are the free ions (Prego & Cobelo-Garcia 2003). Therefore the knowledge of total metal concentration in the water column is, in general, not sufficient to predict its activity/toxicity (Prego & Cobelo-Garcia 2003). Salinity, dissolved organic carbon (e.g. humic and fulvic acids), and pH may influence the speciation and complexation, of metals and thereby affect the influx of dissolved metals into aquatic organisms (Griscom *et al.* 2002). Adsorption of metals to suspended particulate matter usually occurs at low salinities and in the zone of maximum turbidity, while desorption occurs at high salinities due to an increase in major ion concentrations (Prego & Cobelo-Garcia 2003). Humic substances have carboxylic and phenolic groups that can complex metals (Lorenzo *et al.* 2002). At lower oxygen levels in water, carbon-dioxide levels will increase, resulting in a change of the bicarbonate system, giving a shift to more acidic pH. This may lead to more uncomplexed, bioavailable forms of some metals, such as copper (Neuhoff 1983).

The toxicity of different heavy metals to infaunal bivalve species varies. This may be due to the fact that some metals function as important micro-nutrients and co-factors (e.g. Cu, Ni and Zn) in the metabolism of infaunal bivalves, while others with no known metabolic function are always considered toxic (e.g. Cd and Pb) (Gnassia-Barelli & Romeo 1993, Hindarti *et al.* 2000, Keppler and Ringwood 2002). Copper has a high affinity for organic matter while cadmium has a lower tendency to bind to organic ligands (Beiras & Albentosa 2003).

The mode of uptake of heavy metals into infaunal bivalve species has been considered to be largely via direct uptake from dissolved species in the water column. However, metal influx into benthic organisms (including infaunal bivalves) can result from both ingestion of food particles/sediment and from transport across tissues from dissolved sources (Griscom *et al.* 2002). Metals loosely adsorbed to particle surfaces are more bioavailable than those metals trapped in precipitated mineral coatings (Griscom *et al.* 2000). Griscom *et al.* (2002) found that the relative concentrations of bioavailable metals (e.g. Ag, Cd, and Co) in suspension-feeding *Macoma balthica versus* deposit-feeding *Macoma balthica* at the same sites, was dependent on the particular metal in question. The range of bioavailable Ag and Cd concentrations predicted for suspension-feeding tellinids was higher than that predicted for deposit-feeding tellinids. However, for Co the predicted bioavailable concentration range was greater in deposit-feeding tellinids than in suspension-feeding tellinids (Griscom *et al.* 2002). Thus, the bioavailability of sediment-bound metals remains a challenging parameter to quantify (Griscom *et al.* 2002).

The lethal toxicity of heavy metals to embryonic and larval stages of many bivalve species has been reported by a large number of investigators (e.g. Calabrese *et al.* 1973, Martin *et al.* 1981, and LaBreche *et al.* 2002). However, Roper *et al.* (1995) comment that "*Ecological changes in response to sediment contamination by heavy metals are more likely to be as a result of subtle effects on organisms, rather than mortality as a result of direct exposure*". Sublethal effects of contaminants on growth, reproduction, and behavioural responses have long been recognised for their ability to cause adaptive disabilities that may ultimately result in lowered survival in the natural environment (MacInnes & Calabrese 1979). Wikfors *et al.* (1994) identified that post-settlement *Mercenaria mercenaria* fed heavy metal contaminated microalgae had lower growth rates compared to control clams. Matozzo *et al.* (2001) have identified that heavy metal exposure can alter the functional response of haemocytes of the infaunal clam *Tapes philippinarum,* and suggest that the type and concentration of the heavy metal can alter the effects observed. Blasco & Puppo (1999) identified altered aminotransferase activity in adults of the clam *Ruditapes philippinarum* when exposed to different concentrations of Cd, Cu and Pb. While Hamza-Chaffai *et al.* (1998) identified decreased acetylcholinesterase activity in gill tissue of adult *Ruditapes decussatus* when exposed to copper. Altered behavioural responses have also been observed when infaunal bivalves are exposed to heavy metal contamination (e.g. *Macoma balthica*, McGreer 1979; *Protothaca staminea*, Phelps *et al.* 1985; *M. liliana*, Roper *et al.* 1995).

***Cadmium:*** The average concentration of cadmium in the earth's crust is approximately 0.1 $\mu g\ g^{-1}$, while erosion processes have led to a 2-3 fold enrichment of sediment by this metal (Forstner 1980). The average Cd concentration in unpolluted sediments ranges from 0.04 to 0.8 $\mu g\ g^{-1}$, while in contaminated zones Cd may reach values of 30 to 400 $\mu g\ g^{-1}$ (Prego & Cobelo-Garcia 2003). In unpolluted natural waters the concentrations of cadmium are usually $< 1\ \mu g\ L^{-1}$ (Friberg *et al.* 1986). Increased levels of cadmium may be found in coastal waters where cadmium containing sewage sludge is released (Phillips 1991). Bulter & Timperley (1994) note that potential sources of cadmium include sewage, fertiliser (particularly phosphate fertilisers), industrial waste and lithospheric components. Biological studies suggest that Cd transport into organisms is correlated with free $Cd^{2+}$ ions, not $CdCl_2$ concentrations (Sunda *et al.* 1978). The incorporation of cadmium

from its dissolved form takes place by means of passive diffusion, in which the metal is adsorbed on the surface of the gill cells, probably on the mucous, and then passes through the cell membrane, where it links up with high affinity metal ligands (such as metallothioneins) (Viarengo 1989). However, cadmium in the aquatic environment is primarily found bound to bottom sediments and suspended particles, and thus in the water phase its concentration is low (Hindarti *et al.* 2000). Laboratory studies suggest that suspension-feeding bivalves have little capacity to regulate the bioaccumulation of cadmium from the water column (e.g. *Mytilus edulis*, George 1984; *Ruditapes philippinarum*, Blasco & Puppo 1999).

***Chromium:*** Chromites and chromic oxide are two of the main sources of this metal in soils (Prego & Cobelo-Garcia 2003).

***Copper:*** Copper is a micronutrient necessary to virtually all aquatic life, being present in many enzymes involved in metabolic processes that depend on redox reactions, and present in all natural waters and sediments (Lewis & Cave 1982, Scheinberg 1991, Hall & Anderson 1999). Copper is not lipophilic, and shows only a slight tendency for bioaccumulation within organisms (Voulvoulis *et al.* 1999), due largely to the fact that it is also a necessary micronutrient. As a consequence of the role that copper plays in metabolic processes, marine invertebrates possess elaborate homeostatic mechanisms that can maintain internal concentrations of copper within narrow limits while external concentrations may fluctuate over several orders of magnitude (Hall & Anderson 1999). However, if copper is present at relatively high concentrations in the environment, toxicity to aquatic organisms (e.g. phytoplankton and bivalve larvae) can occur (Sunda & Guillard 1976, Hall & Anderson 1999, and His *et al.* 2000). Copper is the most toxic metal, after mercury and silver, to a variety of organisms (Clark 2001). Copper can also have sublethal effects at low concentrations (Flemming & Trevors 1989, Bryan & Langston 1992). Lethal and sublethal effects (such as avoidance behaviour at 67 to 95 µg Cu $g^{-1}$ (dry wt)) of heavy metal contaminated sediments have been observed overseas in the tellinid *Macoma balthica* (McGreer 1979, McGreer 1982).

The chemistry of copper affects its distribution, availability and its potential toxicity to aquatic biota. The most bioavailable (and thus the most toxic) form of ionic, unbound copper is the free hydrated ion $Cu(H_2O)_6^2$ (Hall & Anderson 1999, Voulvoulis *et al.* 1999). In aerobic environments, copper bioavailability is largely controlled by the presence of manganese and iron oxides and dissolved organic material. Copper forms insoluble complexes in seawater which effectively bind the copper and reduce its bioavailability (Hall & Anderson 1999, Voulvoulis *et al.* 1999). Further, studies of copper speciation in coastal waters indicate that more than 99% of the total copper is strongly bound or chelated with organic ligands, leaving the amount of free $Cu^{2+}$ at levels that are non-toxic to most organisms (Bruland 1997). Similarly, Hall *et al.* (1997) found that speciation of copper in Chesapeake Bay seawater meant that $Cu^{2+}$ accounted for only 8% of the total dissolved inorganic fraction, or <0.2% of the total copper in solution. There is also evidence that microorganisms may secrete copper chelators in response to increases in copper concentration in the water column (Bruland 1997). Copper speciation (and thus free $Cu^{2+}$) in seawater is governed by pH, salinity, and the presence of dissolved organic material (de Mora 1996, Hall & Anderson 1999). Ions in an estuary affect the copper chemistry in seawater, transforming copper from a filterable state into particulates which settle to the bottom (Hall & Anderson 1999). Estuarine areas (including harbours and marinas) therefore act as a sink for copper (Martin & Windom 1991). Microalgae may accumulate copper in excess of their metabolic needs, and thus act as micro-reservoirs of copper for feeding infaunal bivalves. A comparison of copper toxicity across a range of estuarine and marine salinities for various marine aquatic species identified that 64% of the time the toxicity of copper was greater at lower salinities (Hall & Anderson 1995), probably due to the dominance of $Cu^{2+}$ at these lower salinities.

Copper toxicity is believed to be due to the nonspecific binding of the reactive metal cation $Cu^{2+}$ to biologically important macromolecules. In infaunal bivalves this is likely to occur on the receptor sites of the gills, as is the case in fish (Hall & Anderson 1999).

*Lead:* Lead in colloidal or particulate form is taken up via epithelial cells (particularly the gills and mantle) of invertebrates, and is incorporated via endocytosis (a process where the cell engulfs the metal particle) (Gnassia-Barelli & Romeo 1993). It would appear that suspension-feeding bivalves have little capacity to regulate the bioaccumulation of lead (e.g. *Ruditapes philippinarum*, Blasco & Puppo 1999). Lead may be highly bioavailable to aquatic organisms from sediment as indicated by overseas studies, which identified that the HCl extractable fraction may be as high as 90% of the total metal concentration (or commonly between 50-75%) (Carballeira *et al.* 2000). Studies in Spain have identified that lead concentrations in the <2 μm fraction of sediments are higher than the 2-63 μm fraction, illustrating a potential relationship between sediment particle size and lead concentration (e.g. Prego *et al.* 1999).

*Nickel:* Ferromanganese minerals and ferrous sulfides are among the natural sources of this element (Prego & Cobelo-Garcia 2003). Nickel concentrations in coastal and estuarine areas generally range from 5 to 85 nM depending on anthropogenic influences (Prego & Cobelo-Garcia 2003).

*Zinc:* Zinc is an abundant element, with an average concentration in the earth's crust of 70 μg g$^{-1}$ Ohnesorge & Wilhelm 1991). Like Cd and Cu, the bioavailable and potentially toxic chemical species of Zn are considered to be the free ions (Prego & Cobelo-Garcia 2003). Due to its high solubility, the dissolved phase may be the primary route of exposure of Zn to infaunal bivalves (Cheggour *et al.* 2001). However in a New Zealand study of stormwater outlfow, Bibby (2001) identified that while Pb was predominantly transported bound to suspended particulate matter (SPM), Zn was transported both bound to SPM and as dissolved components of seawater. In a similar fashion to that identified for Cu, infaunal bivalves may be able to regulate their body burden of Zn, at least within a certain concentration range (Cheggour *et al.* 2001). Toxicity tests have demonstrated that organic complexation reduces the toxicity and bioavailability of zinc, which may suggest why zinc is considerably less toxic than Cu or Cd (Allen *et al.* 1980).

## 5.3.2 Dose-Response Assessment

Some dose-response data specific to the New Zealand shellfish species of interest here are available for Cu, Pb and Zn in sediments (e.g. Roper & Hickey 1994, Roper *et al.* 1995). Supplementary dose-response data for similar overseas infaunal bivalve species are also presented.

### *Cadmium:*
***In phytoplankton:***
**Post-settlement (sublethal toxicity)**
- When fed cadmium contaminated microalgae, post-settlement hard clams *(Mercenaria mercenaria)* exhibited no growth over 12 weeks (although algal cells were taken up by the clams) (Wikfors *et al.* 1994). Control clams displayed normal growth rates over this period. From this study Wikfors *et al.* (1994) suggested that food-borne cadmium interferes directly with food assimilation and/or growth in *M. mercenaria*.

*In the water column:*
**Embryo (lethal toxicity)**
- Calabrese *et al.* (1973) reported a 48-hr LC$_{50}$ for Cd of 3800 µg L$^{-1}$ for embryos of the oyster *Crassostrea virginica*.

**Larvae (sublethal toxicity)**
- Martin *et al.* (1981) identified a 48-hr EC$_{50}$ (i.e. % developed to normal D-shape larvae) for Cd of 611 µg L$^{-1}$ for *Crassostrea gigas* larvae.
- Beiras & Albentosa (2003) identified a 48-hr EC$_{50}$ for larval development of *Ruditapes decussatus* (i.e. % developed to normal D-shape larvae) of 424 µg L$^{-1}$ for Cd.

**Juveniles (sublethal toxicity)**
- Toxicity testing identified a 7-day EC$_{50}$ of 86.7 µg L$^{-1}$ for growth of 1-1.2 mm juvenile *Mercenaria mercenaria* when exposed to dissolved Cd (Keppler & Ringwood 2002).

**Adults (sublethal toxicity)**
- Infaunal adult clams (*Ruditapes philippinarum*) exposed to 200 or 600 µg Cd L$^{-1}$ over 7 days showed inhibited alanine aminotransferase activity compared to control clams (Blasco & Puppo 1999).

*In sediment:*
No specific dose-response data are available, although Stecko & Bendell-Young (2000) have identified that cadmium can be taken up from contaminated sediments by both *Macoma balthica* and *Protothaca staminea*.

## Chromium:
*In the water column:*
**Larvae (sublethal toxicity)**
- Martin *et al.* (1981) identified a 48-hr EC$_{50}$ (i.e. % developed to normal D-shape larvae), of 4538 µg L$^{-1}$ for the larvae of the oyster *Crassostrea gigas*.

## Copper:
It is difficult to interpret the dose-response relationship in studies that deal with sublethal levels of copper due to the complicated speciation measurements (Absil *et al.* 1996). The problems associated with ligand complexation of copper in seawater mean that static bioassays with environmentally relevant copper concentrations are difficult to undertake (Absil *et al.* 1996).

*In phytoplankton:*
**Adults**
- In laboratory feeding experiments using ^{64}Cu-labelled microalgae Absil *et al.* (1994) demonstrated that food-associated copper could contribute a significant amount to the total body burden of adult *Macoma balthica*.

*In the water column:*
**Embryo (lethal toxicity)**
- Calabrese *et al.* (1973) reported a 48-hr LC$_{50}$ for Cu of 103 µg L$^{-1}$ for embryos of the oyster *Crassostrea virginica*.

**Embryo (sublethal toxicity)**
- Coglianese & Martin (1981) reported a 48-hr EC$_{50}$ (abnormal embryonic development) for Cu of 10 µg L$^{-1}$ for embryonic *Crassostrea gigas*.

**Larvae (lethal toxicity)**

- Calabrese *et al.* (1973) reported a 12-day $LC_{50}$ for Cu of 33 µg $L^{-1}$ for larval *Crassostrea virginica*.
- LaBreche *et al.* (2002) estimated a 24-hr $LC_{50}$ of 142 µg $L^{-1}$, and an 8-day $LC_{50}$ of 12 µg $L^{-1}$, for larval *Mercenaria mercenaria* exposed to Cu in solution. In this experiment larval clams that were fed microalgae exposed to copper had a lower survival rate than those not fed microalgae.

**Larvae (sublethal toxicity)**

- Martin *et al.* (1981) identified a 48-hr $EC_{50}$ (i.e. % developed to normal D-shape larvae), of 5.3 µg $L^{-1}$ for larvae of the oyster *Crassostrea gigas*.
- LaBreche *et al.* (2002) found a 24-hour $EC_{50}$ estimate of modified swimming behaviour for larval *M. mercenaria* of 20.2 µg $L^{-1}$ when exposed to Cu.
- Calabrese *et al.* (1977) found a 12-day $EC_{50}$ for growth (static renewal test) of 16.9 µg $L^{-1}$ Cu for larval *M. mercenaria*.

**Juveniles (sublethal toxicity)**

- The US EPA (1995) identified that juveniles of the infaunal surf clam *Mulinia lateralis* had a 48-hr $EC_{50}$ of 14.9-21 µg $L^{-1}$ Cu.

**Adults (sublethal toxicity)**

- Infaunal adult clams (*Ruditapes philippinarum*) exposed to 10 or 20 µg Cu $L^{-1}$ over 7 days exhibited inhibited alanine aminotransferase activity in the 20 µg treatment (only in the gill tissues) compared to control clams (Blasco & Puppo 1999).
- In a 20 day study of exposure of *Ruditapes decussatus* to sublethal concentrations of copper (e.g. 100 µg $L^{-1}$) Sobral & Widdows (1997) observed that at the end of the experiment adult clams had clearance rates 50% lower than the controls. They also exhibited respiration rates 145% higher than respiration rates in controls. It was suggested that these levels were enough to cause sustained impairment of physiological functions.
- When copper (>250 mg Cu $L^{-1}$) was added to water overlying sediments, Stirling (1975) found adverse effects on burrowing rates of *Tellina tenuis*.

*In sediment:*

Very few studies have been conducted to specifically investigate the toxicity of copper in sediment to infaunal bivalves (McPherson & Chapman 2000).

**Juveniles (lethal toxicity)**

- Burgess *et al.* (1994) investigated the effects of clean, copper-spiked sediment on survival of juveniles of the infaunal surf clam *M. lateralis* over a 12 day period. The 96-hr $LC_{50}$ values were 14.6-51.1 µg $g^{-1}$ Cu (dry wt).
- Copper dosed sediments were found to have a pronounced effect on the survival of juvenile *M. liliana* in New Zealand (Roper & Hickey 1994). Mortality occurred at 30 µg Cu $g^{-1}$ (dry wt) (Roper & Hickey 1994).

**Juveniles (sublethal toxicity)**

- Burgess *et al.* (1994) investigated the effects of clean copper spiked sediment on growth of juveniles of the infaunal surf clam *M. lateralis* over a 12 day period. The 96-hr $EC_{25}$ for growth was <5.8-20.8 µg $g^{-1}$ Cu (dry wt). It is important to note that all organic carbon was removed from the sediment by preheating the sediment in a muffle furnace; this process may have had an impact on the bioavailability of copper in this experiment.
- Copper dosed sediments were found to have a pronounced effect on the behaviour of juvenile *M. liliana* in New Zealand (Roper & Hickey 1994). Sediments containing 5 µg Cu $g^{-1}$ (dry wt) were avoided and burial rates were significantly slowed at 15 µg Cu $g^{-1}$ (dry wt). However,

these experiments simulated an accidental spillage or discharge of Cu, and do not take into account bioavailability/chemical speciation issues in aged sediment.

- Further work by Roper *et al.* (1995) with copper-dosed sediments (conducted with sediment collected from several sites) identified that the percentage of juvenile *M. liliana* that buried themselves after 10 min was significantly reduced at 25 µg Cu g$^{-1}$ (dry wt). After a 96-hr exposure, juvenile *M. liliana* crawled away from sediment with 10 µg Cu g$^{-1}$ (dry wt), and if a weak water current was provided they would leave via drifting on mucous threads.

**Adults (sublethal toxicity)**
- Phelps *et al.* (1985) identified that gravel sediment enriched with 4.4 µg Cu g$^{-1}$ significantly increased burrowing and reburrowing of *Protothaca staminea*. Clams with increased burrowing time into copper-enriched sediment had up to 25% mortality after 12-29 days burial in the sediment. The majority of clam mortality was associated with clams that burrowed in the first 48 hours of the experiment (Phelps *et al.* 1985).
- Hamza-Chaffi *et al.* (1998) identified that adult *Ruditapes decussatus* exposed to 75 µg L$^{-1}$ of copper for 5-days had significantly decreased acetylcholinesterase activity (an enzyme essential to the correct transmission of nerve impulses) in gill tissue when compared to control clams.

## *Lead:*

In general the toxicity of lead is extremely variable, depending on the permeability of the organism and its ability to detoxify the metal (Bryan 1984). Risks associated with inorganic lead contamination are considered to be low (Gnassia-Barelli & Romeo 1993). However, it is generally recognised that the acute toxicity of organic lead is much higher than that of inorganic lead (Gnassia-Barelli & Romeo 1993).

### *In phytoplankton:*
We were unable to locate any data relating to the impact on infaunal bivalves of the ingestion of phytoplankton contaminated with lead.

### *In the water column:*
**Embryo (lethal toxicity)**
- Calabrese *et al.* (1973) reported a 48-hr LC$_{50}$ for Pb of 2450 µg L$^{-1}$ for embryos of the oyster *Crassostrea virginica*.

**Larvae (sublethal toxicity)**
- Martin *et al.* (1981) identified a 48-hr EC$_{50}$ (i.e. % developed to normal D-shape larvae) for Pb of 758 µg L$^{-1}$ for *Crassostrea gigas* larvae.

**Adults (sublethal toxicity)**
- Infaunal adult clams (*Ruditapes philippinarum*) exposed to 350 or 700 µg Pb L$^{-1}$ over 7 days showed inhibited alanine aminotransferase activity compared to control clams (Blasco & Puppo 1999).

### *In sediment:*
**Juveniles (sublethal toxicity)**
- Pilot investigations by Roper *et al.* (1995) of lead dosed sediments found no effect on burial or crawling behaviour of juvenile *M. liliana* at Pb concentrations up to 100 µg Pb g (dry wt)$^{-1}$. This supports the hypothesis that lead is less toxic to infaunal bivalves than other heavy metals such as copper and zinc.

## Nickel:

### *In the water column:*
**Embryo (lethal toxicity)**
- Calabrese *et al.* (1973) reported a 48-hr $LC_{50}$ for Ni of 1180 µg $L^{-1}$ for embryos of the oyster *Crassostrea virginica*.

**Larvae (sublethal toxicity)**
- Sublethal - Martin *et al.* (1981) identified a 48-hr $EC_{50}$ (i.e. % developed to normal D-shape larvae) of 349 µg $L^{-1}$ for the larvae of the oyster *Crassostrea gigas*.
- Calabrese *et al.* (1977) found a 12-day $EC_{50}$ for growth (static renewal test) of 1518 µg $L^{-1}$ Ni for larval *Mercenaria mercenaria*.

## Zinc:
In general the toxicity of zinc is extremely variable, depending on the permeability of the organism and its ability to detoxify the metal (Bryan 1984).

### *In the water column:*
**Embryo (lethal toxicity)**
- Calabrese *et al.* (1973) reported a 48-hr $LC_{50}$ for Zn of 310 µg $L^{-1}$ for embryos of the oyster *Crassostrea virginica*.
- **Larvae (sublethal toxicity)**
- Martin *et al.* (1981) identified a 48-hr $EC_{50}$ (i.e. % developed to normal D-shape larvae), of 119 µg $L^{-1}$ for the larvae of the oyster *Crassostrea gigas*.
- Calabrese *et al.* (1977) found a 12-day $EC_{50}$ for growth (static renewal test) of 239.3 µg $L^{-1}$ Zn for larval *Mercenaria mercenaria*.

### *In sediment:*
**Juvenile (sublethal toxicity)**
- Work by Roper *et al.* (1995) of zinc-dosed sediments in New Zealand (conducted with sediment collected from several sites) identified that the percentage of juvenile *M. liliana* burrowing after 10 min was significantly reduced at 80 µg Zn g (dry wt)$^{-1}$. After a 96-hr exposure, juvenile *M. liliana* crawled away from sediment with 40 µg Zn g (dry wt)$^{-1}$, and if a weak current was provided they would leave via drifting on mucous threads.

One aspect that has not been considered in the above dose-response assessments is the combined impact from exposure to several different heavy metals. The toxicity of a heavy metal can be enhanced (synergistic interaction), reduced (antagonism), or unaffected (no interaction) by another (Beiras & Albentosa 2003). If two metals do not affect the toxicity of one another, and both act by the same mechanism (as may be expected for $Cu^{2+}$ and $Zn^{2+}$), the combined effects of their toxicity will simply be additive (Beiras & Albentosa 2003).

## *In summary:*

Heavy metals appear to be more toxic to infaunal bivalve embryonic and larval stages than to adults.

His *et al.* (2000) have comprehensively reviewed the available data on embryo toxicity of water-column metals to bivalves, and concluded that interspecific differences in sensitivity are low compared to method variability. They report average (geometric mean) $EC_{50}$ values of 24 µg $L^{-1}$ for

Cu, 320 µg L$^{-1}$ for Zn, 968 µg L$^{-1}$ for Pb, and 2219 µg L$^{-1}$ for Cd.  A study by Beiras & Albentosa (2003) of toxicity of heavy metals to larval infaunal clams (*Ruditapes decussatus*) identified results consistent with the values mentioned above (e.g. EC$_{50}$ values of 9.1 µg L$^{-1}$ for Cu, 129 µg L$^{-1}$ for Zn, between 156-312 µg L$^{-1}$ for Pb and 424 µg L$^{-1}$ for Cd).  Thus, in terms of the relative ranking of the most toxic to the least toxic heavy metals when delivered via a water-column route to infaunal bivalves, Cu > Zn > Pb > (Ni & Cd) > Cr.  The lack of data pertaining to sediment bioavailability of these heavy metals makes it impossible at this stage to rank the relative toxicity of these metals to the infaunal bivalves in the Hauraki Gulf Marine Park.  In addition, further information is required on the impacts of exposure to combinations of heavy metals in the marine environment.

### 5.3.3    Exposure Assessment

Heavy metal contamination of the near-shore water-column and sediments does occur within the Hauraki Gulf Marine Park.  For example, a survey of heavy metal pollution in the Waitemata Harbour (<20 µm sediment fraction) by Glasby *et al.* (1988) identified an anthropogenic influence of Pb, Zn and Cu in Meola Creek (Pb & Cu), Whau Creek (Cu), and the city (an area including the Devonport Naval Base, Downtown Auckland and part of the North Shore) (Pb & Zn).  However, the risk presented to infaunal bivalve populations by heavy metals varies spatially across the Hauraki Gulf Marine Park for a number of reasons.

- Heavy metals may be associated with stormwater discharge.  Heavy metals associated with stormwater discharge e.g. Zn, Pb and Cu (Morrisey *et al.* 2000), are likely to be higher in sediments of urbanised estuaries.  Because estuaries trap fine particles and the associated heavy metals attached to them, these areas are likely to have the highest heavy metal load derived from stormwater.  Morrisey *et al.* 2000 identified that for Hellyers-Kaipatiki Creek and Pakuranga Creek the heavy metals Cu, Pb, and Zn exceed the "Effects range-low" (ERL) sediment values suggested by Long *et al.* (1995), which theoretically indicates the potential for occasional chronic ecological effects.  Tanked water supply in rural and holiday areas may decrease the level of stormwater associated heavy metals supplied to the marine environment.  However, increased development of New Zealand's coastal environment will increase the level of stormwater associated heavy metals.
- Heavy metals are associated with antifouling paint.  Metals associated with antifouling paints such as Cu and Zn will also be highest in the water column and sediment near marinas, dry docks, slipways, swing moorings and wharves.  These point sources are scattered throughout the Hauraki Gulf Marine Park.  However, an ecological risk assessment of copper in seawater in Europe found that highest concentrations of dissolved copper were recorded in harbours and marinas, with slightly lower values found in estuaries, followed by lower concentrations again on the open coast (Hall & Anderson 1999).  Dissolved copper concentrations were also found to increase in the summer months around marinas and harbours.
- Heavy metals are associated with industry.  Commercial ports and wharf areas have high levels of heavy metals in their associated sediments (e.g. Viaduct basin (Auckland Regional Services Trust, 1996)).
- Heavy metals are associated with municipal waste.  As sewage is discharged along the eastern beaches of the North Shore, this may suggest elevated risk of heavy metals in this area.  Holiday communities may also be supplying heavy metals to the marine environment via septic tanks.
- Heavy metals may also be associated with agricultural runoff from fertilisers and pesticides (e.g. cadmium in fertilisers, Rothbaum *et al.* 1986).

The dispersal of heavy metals entering the marine environment may also differ with the type of source the particular metals are associated with. For example, modelling of contaminant distribution from Auckland's stormwater network by Morrisey *et al.* (2000) assumes *"that 75% of the sediment and associated contaminants entering the upper reaches of an estuary will be deposited in a settling area near the freshwater point source, in an area equivalent to about 4% of the catchment's area"*. The model further assumes *"that fine sediments and associated metal contaminants from small land catchments will be mostly spread over the inter-tidal area between the outfall and the low-tide channel"* (Williamson & Morrisey 2000). Further, *"metals from large catchments will be more widely spread throughout the estuary because of the relatively high rates and volumes of flow"*. However, for heavy metals derived from antifouling paints such as copper residues are likely to be found in higher concentrations around wharves, marinas, slipways, dry docks, and other areas with a high level of boating activity (e.g. swing moorings, canal developments etc.).

It is difficult to evaluate the potential exposure to infaunal bivalves by heavy metals in the less urbanised areas of the Hauraki Gulf Marine Park, as background heavy metal data are extremely limited in these marine environments (for example, on the northern coast above Whangaparoa Peninsula and south of Wairoa estuary and including the Firth of Thames and Coromandel Peninsula). However, it is logical to assume that heavy metals associated with stormwater runoff will be lower in marine environments associated with less urbanised areas, and that marine environments bounded by rural areas will have heavy metals derived from different sources such as fertiliser and pesticides usage. Further, we have an incomplete understanding of the level of heavy metals found in open coastal marine environments, compared to estuaries. This may vary due to the differences in the composition of the sediment.

The size of a bivalve and its associated filtration rate also affects the rate of metal uptake (Lee *et al.* 1998). Lee *et al.* (1998) found that the weight specific metal influx rate ($\mu g\ g^{-1}$ [dry wt] $d^{-1}$) was negatively correlated with the tissue dry weight of the tellinid *Macoma balthica*, but that most rate constants determining physiological turnover of assimilated metals were not affected by clam size. The exception was the rate constant for Cd loss, which resulted in faster turnover in large *M. balthica* than in smaller clams. The rate constant of loss for *M. balthica* was: Zn ($0.012\ d^{-1}$) < Cd ($0.018\ d^{-1}$) < Cr ($0.024\ d^{-1}$).

Although some information exists on the level of heavy metals found in bivalves in the Hauraki Gulf Marine Park region, this largely relates to samples taken near possible point sources of pollution, and it is difficult to interpret how these levels of bioaccumulation relate to actual toxicity in bivalves. Alternatively, the level of exposure of these infaunal bivalves to heavy metals may be gained by investigating the concentrations of heavy metals found in the accompanying sediment. Sediments are preferred to water for chemical monitoring because pollutant concentrations in sediment are much higher and less variable in time and space (Beiras *et al.* 2003). In 1998, the ARC initiated a *"Marine Sediment Monitoring Programme"* with the specific aim of providing baseline data to allow future trend analysis of contaminant build-up in inter-tidal marine sediments (Williamson *et al.* 1999). It should be noted that although comprehensive, the ARC regional contaminant monitoring framework on concentrations of heavy metals in sediment does not provide an effective basis for determining adverse effects on aquatic organisms. Sediment contaminant analysis can document the presence of contaminants but the potential for adverse effects is not readily predictable (Ringwood & Keppler 1998).

Due to their association with stormwater discharge to the marine environment, the *Marine Sediment Monitoring Programme* focuses on the heavy metals Zn, Pb, and Cu. Twenty-seven sites are

sampled in this study, comprising predominantly muddy estuarine sites, with 5 sites situated on the East Coast north of the Waitemata Harbour, 12 sites in the Waitemata Harbour, 3 sites in the Tamaki Estuary and 6 sites in the Manukau Harbour. The exposed East Coast south of the Waitemata Harbour is not sampled. Sampling of marine sediments is undertaken every 2 years for this programme. Data are now available from surveys undertaken in 1998, 1999, and 2001, and the next sampling series is currently in progress (*D. McCarthy, ARC, pers. comm.*).

Within the Hauraki Gulf Marine Park estuarine areas such as the southern shore of the Waitemata Harbour (being an older urbanised area), the Tamaki Estuary and Hobson Bay have the highest stormwater associated heavy metal sediment load (Kelly *et al.* 2003). In relation to stormwater heavy metal contaminants, the concentrations of the metals Cu and Zn are predicted to increase in estuarine sediments in the Auckland region in the future while Pb levels are predicted to decline (due to the removal of lead from petrol in 1996) (Williamson & Mills 2002). In the 2001 survey, levels of Zn in the <63 μm sediment fraction ranged from approximately 45 to 300 μg g$^{-1}$ within the Auckland region (Mathieson *et al.* 2002). If zinc accumulation rates continue in a linear fashion its concentration in sediment will double from 250 to 500 μg g$^{-1}$ in 17 years at some sites in the Tamaki estuary. Recorded accumulation rates in the Whau estuary of the Waitemata Harbour suggest zinc concentrations will double from 300 to 600 μg g$^{-1}$ in 13 years (Kelly *et al.* 2003). In the 2001 survey, levels of Cu in the <63 μm sediment fraction ranged from approximately 4 to 44 μg g$^{-1}$ within the Auckland region (Mathieson *et al.* 2002). Copper concentrations are generally increasing at rates of about one eighth of the rate of zinc concentrations (Timperley & Mathieson 2002). In the 2001 survey levels of Pb in the <63 μm sediment fraction ranged from approximately 5 to 95 μg g$^{-1}$ within the Auckland region (Mathieson *et al.* 2002).

Heavy metal analysis of marine sediments has also been recently undertaken by the North Shore City Council (Le Fevre & Silyn Roberts 2002). This report identified a few instances where the ANZECC Interim Sediment Quality Guidelines (ISQG)-Low value was exceed for both Pb and Zn (e.g. Pb=50 μg g$^{-1}$ and Zn=200 μg g$^{-1}$), suggesting that biological effects could be occurring occasionally in the marine environment. These ANZECC guidelines relate to Total metal concentrations in whole sediment samples, while the North Shore City Council study measured bioavailable metals (via weak acid digestion) in the <63 μm sediment fraction. It is thus difficult to predict the actual level of biological impact from these levels of sediment contamination.

Both the Auckland Regional Council Marine Sediment Monitoring Programme and the survey by the North Shore City Council focus predominantly on examining the input of heavy metals into sediments associated with the receiving environments of stormwater discharges. In addition to a wider geographic coverage, data that consider the impact of other potential sources, (such as copper from antifoulants associated with boating activity) also need to be considered to provide a more comprehensive assessment of the exposure to infaunal bivalves to the risk of heavy metals. Presumably monitoring data collected as part of conditions of relevant resource consents could supplement this, but a review of these data was beyond the scope of the current project.

There appears to be a paucity of information relating to levels of heavy metals in the water column within the Hauraki Gulf Marine Park. In order to establish the status of metal pollution in the dissolved phase, there is a need to establish background concentrations (Prego & Cobelo-Garcia 2003). Dissolved contaminants associated with stormwater are diluted substantially when discharged to estuaries, and are generally not expected to exert toxic effects while in the water column (Williamson & Mills 2002). However, this does not adequately address the question of whether increasing sediment concentrations of some heavy metals (e.g. Cu, Pb and Zn) result in

increased bioavailability of these metals at the sediment/water interface, and thus increased risk to infaunal bivalves.

Bibby (2001) identified that while Pb was predominantly transported bound to suspended particulate matter (SPM), Cu and Zn were transported both bound to SPM and as dissolved components of seawater. Modelling predicted that while toxic effects of Pb and Cu in the water column would be low, zinc is likely to present the greatest risk of toxicity to aquatic life in the Mangemangeroa estuary as metal levels in the catchment increase.

Modes of exposure of infaunal bivalves to potentially toxic heavy metals include the soluble ionic form directly from the water, ingestion of metals bound to sediment, and consumption of contaminated phytoplankton cells.

Our general understanding of the exposure to metals an organism may experience living in a benthic environment is still inadequate (Griscom *et al.* 2002). For the species of interest in this study, there is the potential for a clear difference in exposure route for heavy metals between the deposit-feeder *M. liliana* and the suspension-feeding *A. stutchburyi*, *P. australis* and *P. subtriangulata*. However, our current limited knowledge of the level of bioavailable metals in sediment makes it difficult to assess whether this source will result in elevated risk to deposit-feeding infaunal bivalves. The relationship between ingestion and assimilation from various potential sources is uncertain. However, overseas studies are beginning to address this issue. A study investigating the source of cadmium exposure to two infaunal bivalves, a facultative suspension-feeder *Macoma balthica* and an obligate suspension-feeder *Protothaca staminea* by Stecko & Bendell-Young (2000) identified that the uptake of radio labelled $^{109}$Cd from deposited sediment compared to water accounted for 80% of the accumulated radiotracer. Furthermore, uptake of the $^{109}$Cd from deposited sediment was far greater than that from suspended particulate matter, most notably for *P. staminea* (Stecko & Bendell-Young 2000). For *M. balthica*, the amount of cadmium accumulation was best explained by the degree of isotope desorption from the deposited sediment, whereas for *P. staminea* cadmium accumulation was best described by its feeding behaviour (Stecko & Bendell-Young 2000). To assess the potential impact of heavy metals in sediment on marine infauna, the degree to which sediment-bound metals are assimilated into animal tissue must be quantified (Griscom *et al.* 2000). Digestion processes within the gut of different bivalves may enhance the dissolution of metals and increase assimilation more in some species than in others (Griscom *et al.* 2000). Differing gut reducing potentials may inhibit or enhance remobilization of trace metals into the dissolved phase and facilitate increased uptake (Griscom *et al.* 2000). Our knowledge of these processes still remains very limited.

Absil *et al.* (1996) identified a different response in copper uptake from sandy spiked sediment when comparing *M. balthica* and *Cerastoderma edule*. *Macoma balthica* accumulated Cu from the sandy spiked sediment while *C. edule* did not at all. Absil *et al.* (1996) attributed this difference to the difference in feeding modes i.e. deposit- *versus* suspension-feeders, and suggested that *M. balthica* could accumulate copper directly from the sediment. There appear to be no comparable data for New Zealand infaunal bivalve species.

In addition, the role that phytoplankton play in the transport of heavy metals into infaunal bivalves in New Zealand requires further investigation.

## 5.3.4　　Risk Characterisation

It is apparent from a review of the literature that potential sources of input of heavy metals are present within the Hauraki Gulf Marine Park.  There appears to be a broad trend of increasing risk of exposure to heavy metals with increasing proximity to populated coastal areas.  For a variety of reasons (e.g. input of stormwater, flocculation and sedimentation processes etc.), in general the risk of heavy metal exposure from sediments could be expected to be higher in estuarine areas and areas associated with boating activity than in other coastal environments.

There are limited data available regarding the potential toxicity of heavy metals to the New Zealand bivalve species *A. stutchburyi*, *P. australis*, *P. subtriangulata* and *M. liliana*.  However, based on these data, and data from overseas bivalve species, it is suggested that heavy metals appear to be more toxic to embryonic and larval stages than to adults.  This means that the potential impacts are harder to detect in the field.

Infaunal bivalves may be exposed to potentially toxic heavy metals in several different ways, including soluble ionic forms directly from the water, ingestion of metals bound to sediment, and consumption of contaminated phytoplankton cells.  While some monitoring for heavy metals occurs in the Hauraki Gulf Marine Park, it provides very limited data with respect to the potential for infaunal bivalves to be exposed through these different routes.

In addition, the availability and toxicity of metals to organisms in the water depends on the chemical speciation of the metal in that particular aquatic environment.  Therefore the knowledge of total metal concentration is, in general not sufficient to predict its activity/toxicity (Prego & Cobelo-Garcia 2003).  It can be argued that one of the main shortcomings of risk assessments of the impacts of metals on the marine environment is that they are predominantly based on total or dissolved metal concentrations (Janssen *et al.* 2003).  There is now extensive evidence to suggest that neither total nor dissolved aqueous metal concentrations are good predictors of metal toxicity and bioavailability (Janssen *et al.* 2000).  New biotic ligand models (BLMs) based on the critical metal concentration on the biotic ligand (a ligand where the dissolved chemical species bind) which is associated with, for example, 50% mortality (or another endpoint) is assumed to be independent of the water quality characteristics (Janssen *et al.* 2003).  However, to date most initial BLM efforts have concentrated on predicting short-term metal toxicity to fish, and there do not appear to be any data for infaunal shellfish using this model.

Given the paucity of relevant data, robust characterisation of the risks posed by heavy metals to infaunal bivalves in the Hauraki Gulf Marine Park is extremely difficult.  However, based on the limited data available from the Auckland Regional Council *Marine Sediment Monitoring Programme* (Mathieson *et al.* 2002) we note the following:

- Based on studies by Roper and Hickey (1994) and Roper *et al.* (1995), current zinc levels in the <63 μm sediment fraction (when standardised for the % volume this sediment fraction makes of the total sediment) at Whau (Upper), Whau, Wairau, Motions, Pakuranga (Upper), Tamaki, Meola, Henderson, Whau (Lower), Oakley, Pakuranga (Lower), and Lucas Creeks could be sufficient (i.e. >80 μg g$^{-1}$) to cause impacts in juvenile *M. liliana* (reduced burrowing, an indication of stress). Levels of Zn in sediments at found at Hobson, Cheltenham, Kaipatiki, and Paremoremo Streams exceed those at which avoidance of contaminated sediments have been observed experimentally in juvenile *M. liliana* (i.e. >40 μg g$^{-1}$)

- Copper levels (when standardised for the % volume the <63 μm sediment fraction makes up of the total sediment) were generally below the levels to cause impacts on juvenile *M. liliana* (i.e. reduced burrowing at 25μg g$^{-1}$ and mortality at 30 μg g$^{-1}$, (Roper & Hickey 1994, Roper *et al.* 1995).
- The levels of lead recorded in the sediment during the survey in the Auckland region were lower than those found to have no impact on juvenile *M. liliana* in laboratory-based trials by Roper *et al.* (1995).

Predicted trends of increasing copper and zinc mean that these metals are likely to have an increasing impact in the future. However, the potential impact of lead is likely to decrease.

We caution that the results discussed above are from a limited number of surveyed sites that represent predominantly one type of contamination source (stormwater) and one route of assimilation into the shellfish (via sediment). In addition, interpretation of the risk represented by these data is complicated by uncertainties about the bioavailability of the metals in the sediment, both at the surveyed sites and in the laboratory-based dose-response trials with *M. liliana*. We acknowledge that there are dangers in interpreting data as simplistically as in the bullet points above, and suggest that the appropriate conclusion to be drawn from the presented examples is that they are merely indicative that zinc and copper contamination in the marine environment in some areas in the Hauraki Gulf Marine Park could potentially present a risk to infaunal shellfish populations. We emphasize that at this stage the extent of this risk is largely unknown, and suggest that further work is required to quantify the risk more definitively.

## 5.3.5    References

**Absil, M. C. P. 1993.** Bioavailability of metals for the deposit feeder *Macoma balthica*, with special emphasis on copper. Ph.D. thesis. Agricultural University Wageningen, The Netherlands.

**Absil, M. C. P., J. J. Kroon & H. T. Wolterbeck 1994.** Availability of copper from phytoplankton and water for the bivalve *Macoma balthica*. 2. Uptake and elimination from $^{64}$Cu labeled diatoms and water. *Marine Biology*. 118: 129-135.

**Absil, M. C. P., M. Berntssen & L. J. A. Gerringa 1996.** The influence of sediment, food and organic ligands on the uptake of copper by sediment-dwelling bivalves. *Aquatic Toxicology*. 34: 13-29.

**Allen, H. E., R. H. Hall & T. D. Brisbin 1980.** Metal speciation: effects on aquatic toxicity. *Environmental Science and Technology*. 14: 441-443.

**Ankley, G. T., D. M. Di Toro, D. J. Hansen & W. J. Berry 1996.** Technical basis and proposal for deriving sediment quality criteria for metals. *Environmental Toxicology and Chemistry*. 15(12): 2056-2066.

**Auckland Regional Services Trust 1996.** America's Cup 2000 Facilities. Assessment of Effects on the Environment. Vol. 1. Prepared for the Auckland Regional Council by the Auckland Regional Services Trust. July 1996.

**Beiras, R. & M. Albentosa 2003.** Inhibition of embryo development of the commercial bivalves *Ruditapes decussatus* and *Mytilus galloprovincialis* by trace metals; implications for the implementation of seawater quality criteria. *Aquaculture, article in press*.

**Beiras, R., N. Fernandez, J. Bellas, V. Besada, A. Gonzalez-Quijano & T. Nunes 2003.** Integrative assessment of marine pollution in Galacian estuaries using sediment chemistry, mussel bioaccumulation, and embryo-larval toxicity bioassays. *Chemosphere*. 52: 1209-1224.

**Bibby, R. L. 2001.** Transport mechanisms for Cu, Pb, and Zn in a catchment undergoing development (Mangemangeroa). Unpubl. MSc. thesis. School of Environmental Science. University of Auckland. 134pp.

**Birch, G. & S. Taylor 2002.** Possible biological significance of contaminated sediments in Port Jackson, Sydney, Australia. *Environmental Monitoring and Assessment*. 77: 179-190.

**Blasco, J. & J. Puppo 1999.** Effect of heavy metals (Cu, Cd and Pb) on aspartate and alanine aminotransferase in *Ruditapes philippinarum* (Mollucsa: Bivalvia). *Comparative Biochemistry and Physiology, Part C*. 122: 253-263.

**Bruland, K. W. 1997.** Workshop on research Needs for Coastal Pollution in Urban Areas, National Science Foundation, Wisconsin.

**Bryan, G. W. 1984.** Pollution due to heavy metals and their compounds. In: Marine Ecology, Vol. 5, Part 3. Kinne O. (Ed.). John Wiley and Sons, Chichester. Pp. 1289-1431.

**Bryan, G. W. & W. J. Langston  1992.**  Bioavailability, accumulation and effects of heavy metals in sediments with special reference to United Kingdom estuaries: a review. *Environmental Pollution*.  76: 89-131.

**Burgess, R. M., R. A. Rogers, S. A. Rego, J. M. Corbin & G. E. Morrison  1994.**  Sand spiked with copper as a reference toxicant material for sediment toxicity testing: A preliminary evaluation. *Archives of Environmental Contamination and Toxicology*.  26: 163-168.

**Butler, C. A. & M. H. Timperley  1994.**  An assessment of the mobility and transport of cadmium in the Mahurangi catchment.  ESR Research Progress Report.  June 1994.

**Calabrese, A., R. S. Collier, D. A. Nelson & J. R. MacInnes  1973**.  The toxicity of heavy metals to the American oyster *Crassostrea virginica*. *Marine Biology*.  18: 162-166.

**Calabrese, A., J. R. MacInnes, D. A. Nelson & J. E. Miller  1977.**  Survival and growth of bivalve larvae under metal stress. *Marine Biology*.  41: 179-184.

**Carballeira, A., E. Carral, X. M. Puente & R. Villares  2000.**  Regional scale monitoring of coastal contamination. Nutrients and heavy metals in estuarine sediments and organisms on the coast of Galacia (northwest Spain). *International Journal of Environment and Pollution.*  13: 534-572.

**Cheggour, M., A. Chafik, W. J. Langston, G. R. Burt, S. Benbrahim & H. Texier  2001.**  Metals in sediments and the edible cockle *Cerastoderma edule* from two Moroccan Atlantic lagoons: Moulay Bou Selham and Sidi Moussa. *Environmental Pollution*.  115: 149-160.

**Clark, R. B.  2001.**  Marine Pollution.  Oxford Press, Oxford.

**Coglianese M. P. & M. Martin  1981.**  Individual and interactive effects of environmental stress on the embryonic development of the Pacific oyster *Crassostrea gigas*.  I. The toxicity of copper and silver.  Marine Environmental Research.  5(1): 13-28.

**Ciutat, A. & A. Boudou  2003.**  Bioturbation effects on cadmium and zinc transfers from contaminated sediment and on metal bioavailability to benthic bivalves. *Environmental Toxicology and Chemistry*.  22(7): 1574-1581.

**De Mora, S. J.  1996.**  Tributyltin: case study of an environmental contaminant.  Cambridge Environmental Chemistry Vol. 8., Cambridge University Press, Cambridge, 1996.

**Duffus, J. H.  2001.**  Heavy metals a meaningless term.
http://www.iupac.org/publications/ci/2001/november/heavymetals.html

**Flemming, C. A. & J. T. Trevors  1989.**  Copper toxicity and chemistry in the environment: a review. *Water, Air, and Soil Pollution*.  44: 143-158.

**Forstner, U.  1980.**  Handbook of Environmental Chemistry, Vol. 3.  Springer, Berlin.

**Forstner, U. & G. T. W. Wittmann  1981.**  Metal pollution in the aquatic environment.  Second revised edition. Springer-Verlag, Berlin.  486pp.

**Friberg, L., T. Kjellstrom, G. F. Nordberg & M. Piscator  1986.**  In: Handbook on the Toxicology of Metals, Vol. 1.  Friberg, L. *et al*. (Eds.).  Elsevier, Amsterdam.

**George, S. G.  1984.**  Intracellular control of Cd concentrations in marine mussels. *Marine Environmental Research*.  14: 465-468.

**Gerringa, L. J. A.  1991.**  Mobility of Cu, Cd, Ni, Pb, Zn, Fe and Mn in marine sediment slurries under anaerobic conditions and at 20% air saturation. *Netherlands Journal of Sea Research*.  27: 145-156.

**Glasby, G. P., P. Stoffers, P. Walter, K. R. Davis & R. M. Renner  1988.**  Heavy-metal pollution in the Manukau and Waitemata harbours, New Zealand. *New Zealand Journal of Marine and Freshwater Research*.  22: 595-611.

**Gnassia-Barelli, M. & M. Romeo  1993.**  Some aspects of lead ecotoxicity in the marine environment. *Aquatic Toxicology*.  26: 163-170.

**Graney, R. L. Jr., D. S. Cherry & J. Cairns Jr.  1984.**  The influence of substrate, pH, diet and temperature upon cadmium accumulation in the Asiatic clam (*Corbicula fluminea*) in laboratory artificial streams. *Water Research*.  18: 833-842.

**Griscom, S. B., N. S. Fisher & S. N. Luoma  2000.**  Geochemical influences on assimilation of sediment-bound metals in clams and mussels. *Environmental Science and Technology*.  34: 91-99.

**Griscom, S. B., N. S. Fisher & S. N. Luoma  2002.**  Kinetic modelling of Ag, Cd, and Co bioaccumulation in the clam *Macoma balthica*: quantifying dietary and dissolved sources. *Marine Ecology Progress Series*.  240: 127-141.

**Hall, L. W. Jr., & R. D. Anderson  1995.**  The influence of salinity on the toxicity of various classes of chemicals to aquatic biota. *CRC Critical Reviews in Toxicology*.  25: 281-346.

**Hall, L. W. Jr., R. D. Anderson, R. V. Kilian, & B. L. Lewis  1997**.  Acute and chronic toxicity of copper to the estuarine copepod *Eurytemora affinis*: influence of organic complexation and speciation. *Chemosphere*.  35: 1567-1597.

**Hall, L. W. & R. D. Anderson  1999.**  A deterministic ecological risk assessment for copper in European saltwater environments. *Marine Pollution Bulletin*.  38: 201-218.

**Hansen, D. J., W. J. Berry, J. D. Mahony, W. S. Boothman, D. M. Di Toro, D. L. Robson, G. T. Ankley, D. Ma, Q. Yan & C. E. Pesch  1996.**  Predicting toxicity of metal-contaminated field sediments using interstitial

concentration of metals and acid-volatile sulfide normalizations. *Environmental Toxicology Chemistry*. 15: 2080-2094.

Hamza-Chaffai, A., M. Romeo, M. Gnassia-Barelli & A. El Abed  1998.  Effect of copper and lindane on some biomarkers measured in the clam *Ruditapes decussatus*. *Bulletin of Environmental Contamination and Toxicology*. 61: 397-404.

Hindarti, D., J. T. Spickett and L. H. Evans  2000.  Distribution of cadmium in the cockle (*Katelysia scalarina*) in Princess Royal Harbour, Albany, Western Australia. *Environmental Toxicology*. 15(1): 40-47.

His, E., R. Beiras & M. N. L. Seaman  2000.  The assessment of marine pollution-bioassays with bivalve embryos and larvae. *In*: Advances in Marine Biology, Vol. 37.  Southward, A. I., P. A. Tyler & C. M. Young (Eds.). Academic Press London.  Pp. 1-178.

Hume, T. M. 1983.  Upper Waitemata Harbour sediments and the inferred impacts of future catchment and estuary use change.  Report for the Auckland Regional Authority, Auckland, New Zealand.

Hummel, H., P. Magni, C. Amiard-Triquet, F. Rainglet, R. Modderman, Y. van Duijn, M. Herssevoort, J. de Jong, L. Snitsevorg, M. Ytsma, R Bogaards & L. de Wolf  1998.  Influence of the level of oxygenation in sediment and water on copper bioavailability to marine bivalves: laboratory experiments and translocation experiments in the filed. *Hydrobiologia*. 373/374: 297-310.

Janssen, C. R. K. De Schamphelaere, D. Heijerick, B. Muyssen, K. Lock,  B. Bossuyt *et al.*  2000.  Uncertainties in the environmental risk assessment of metals. *Human Ecological Risk Assessment*. 6: 1003-1018.

Janssen, C. R., D. G. Heijerick, K. A. C. De Schamphelaere & H. E. Allen  2003.  Environmental risk assessment of metals: tools for incorporating bioavailability. *Environment International*. 28: 793-800.

Kelly, S., D. McCarthy, B. Williamson & M. Timperley  2003.  Monitoring confirms rapid changes in Auckland (NZ) estuarine sediment contaminants.  Notes of presentation by Dr. S. Kelly of ARC.

Keppler, C. J. & A. H. Ringwood  2002.  Effects of metal exposure on juvenile clams, *Mercenaria mercenaria*. *Bulletin of Environmental Contamination and Toxicology*. 68: 43-48.

LaBreche, T. M. C., A. M. Dietrich, D. L. Gallagher & N. Shepherd  2002.  Copper toxicity to larval *Mercenaria mercenaria* (hard clam). *Environmental Toxicology and Chemistry*. 21(4): 760-766.

Lee, B. G., W. G. Wallace & S. N. Luoma  1998.  Uptake and loss kinetics of Cd, Cr, and Zn in the bivalves *Potamocorbula amurensis* and *Macoma balthica*: effects of size and salinity. *Marine Ecology Progress Series*. 175: 177-189.

Le Fevre, N. & G. Silyn Roberts  2002.  North Shore City Council Sediment Sampling Programme.  Technical Report, May 2002.  Prepared by URS Ltd. for North Shore City Council. 78pp.

Lewis, A. G.  1995.  Copper in Water and Aquatic Environments.  Report.  International Copper Association.  New York

Lewis, A. G. & W. R. Cave  1982.  The biological importance of copper in oceans and estuaries. *Oceanography & Marine Biology: An Annual Review*. 20: 471-695.

Long, E. R., D. D. MacDonald, S. L. Smith & F. D. Calder  1995.  Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management*. 19: 81-97.

Lorenzo, J. L., O. Nieto, & R. Beiras  2002.  Effect of humic acids on speciation and toxicity of copper to *Paracentrotus lividus* larvae in seawater. *Aquatic Toxicology*. 58: 27-41.

MacInnes, J. R. & A. Calabrese 1979.  Combined effects of salinity, temperature and copper on embryos and early larvae of the American oyster, *Crassostrea virginica*. *Archives of Environmental Contamination and Toxicology*. 8: 553-562.

Mathieson, T. J., G. M. Olsen & J. L. Hawken  2002.  Marine Sediment Monitoring Programme – 2001 Results. NIWA Client Report ARC 02282 to Auckland Regional Council, July 2002. 71pp.

Matozzo, V., L. Ballarin, D. M. Pampanin & M. G. Merin  2001.  Effects of copper and cadmium exposure on functional responses of haemocytes in the clam, *Tapes philippinarum*. *Archives of Environmental Contamination and Toxicology*. 41(2): 163-170.

Martin, J. M. & H. L. Windom  1991.  Present and future roles of ocean margins in regulating marine biogeochemical cycles of trace elements. *In*: Ocean Margin Processes in Global Chang.  Mantoura, R. F. C., J. M. Martin, & R. Wollast (Eds.).  Dahlem Workshop, Wiley, Chichester, U.K.

Martin, M., K. E. Osborn, P. Billig & N. Gliekstein 1981.  Toxicities of ten metals to *Crassostrea gigas* and *Mytilus edulis* embryos and *Cancer magister* larvae. *Marine Pollution Bulletin*. 12: 305-309.

McGreer, E. R.  1979.  Sublethal effects of metal contaminated sediments on the bivalve *Macoma balthica* (L.). *Marine Pollution Bulletin*. 10: 259-262.

McGreer, E. R. 1982.  Factors affecting the distribution of the bivalve *Macoma balthica* (L.) on a mudflat receiving sewage effluent, Fraser River estuary, British Columbia. *Marine Environmental Research*. 7:131-149.

McPherson, C. A. & P. M. Chapman  2000.  Copper effects on potential sediment test organisms: the importance of appropriate sensitivity. *Metal Pollution Bulletin*. 40(8): 656-665.

---

**Morrisey, D. J., T. H. DeWitt, D. S. Roper, & R. W. Williamson  1999**.  Variation in depth and morphology of burrows of the mud crab, *Helice crassa*, among different types of intertidal sediment in New Zealand.  *Marine Ecology Progress Series*.  182: 231-242.

**Morrisey, D. J., R. B. Williamson, L. Van Dam & D. J. Lee  2000.**  Stormwater contamination of urban estuaries. 2. Testing a predictive model of the build-up of heavy metals in sediments.  *Estuaries*.  23(1): 67-79.

**Neuhoff, H. G.  1983.**  Synergistic physiological effects of low copper and various oxygen concentrations on *Macoma balthica*.  *Marine Biology*.  77: 39-48.

**Ohnesorge, F. K. & M. Wilhelm  1991.**  Metals and their compounds in the environment, Part II, Chapter II:36.  VCH, Pp. 1309-1342.

**Phelps, H. L., W. H. Pearson & J. T. Hardy  1985**.  Clam burrowing behaviour and mortality related to sediment copper.  *Marine Pollution Bulletin*.  16(8): 309-313.

**Phillips, D. J. H.  1991.**  *Reviews in Environmental Contamination and Toxicology*.  120:105-129.

**Prego, R., M. J. Belzunce, E. Helios-Rybicka & M. C. Barciela  1999.**  Cadmium, manganese, nickel and lead contents in surface sediments of the lower Ulla River and its estuary (northwest Spain).  *Boeltin Instituto Espanol Oceanografia*.  15(1-4): 495-500.

**Prego, R. & A. Cobelo-Garcia  2003.**  Twentieth century overview of heavy metals in the Galacian Rias (NW Iberian Peninsula).  *Environmental Pollution*.  121: 425-452.

**Ringwood, A. H. & C. J. Keppler  1998.**  Seed clam growth: An alternative sediment bioassay developed during EMAP in the Carolinian Province.  *Environmental Monitoring and Assessment*.  51: 247-257.

**Roper, D. S. & C. W. Hickey  1994.**  Behavioural responses of the marine bivalve *Macomona liliana* exposed to copper- and chlordane-dosed sediments.  *Marine Biology*.  118: 673-680.

**Roper, D. S., M. G. Nipper, C. W. Hickey, M. L. Martin & M. A. Weatherhead  1995.**  Burial, crawling and drifting behaviour of the bivalve *Macomona liliana* in response to common sediment contaminants.  *Marine Pollution Bulletin*.  31: 471-478.

**Rothbaum, H. P., R. L. Goguel, A. E. Johnston & G. E. G. Mattingly  1986.**  Cadmium accumulation in soils from long-continued applications of superphosphate.  *Journal of Soil Science*.  37: 99-107.

**Scheinberg, I. I.  1991.**  Copper.  *In*: Metals and their compounds in the environment.  Merian E. (Ed.).  VCH, Weinheim.  Pp. 893-908.

**Smith, D. G.  1986.**  Heavy metals in the New Zealand aquatic environment; a review.  *Water and Soil miscellaneous publication,* 100.  108pp.

**Sobral, P. & J. Widdows  1997.**  Effects of copper exposure on the scope for growth of the clam *Ruditapes decussatus* from Southern Portugal.  *Marine Pollution Bulletin*.  34(12): 992-1000.

**Stecko, J. R. P. & J. I. Bendell-Young  2000.**  Uptake of $^{109}$Cd from sediments by the bivalves *Macoma balthica* and *Protothaca staminea*.  *Aquatic Toxicology*.  47: 147-159.

**Stirling, E. A.  1975.**  Some effects of pollutants on the behavior of the bivalve *Tellina tenuis*.  *Marine Pollution Bulletin*.  6: 122-124.

**Sunda, W. G. & R. R. L. Guillard  1976.**  The relationship between cupric ion activity and the toxicity of copper to phytoplankton.  *Journal of Marine Research*.  34: 511-529.

**Sunda, E. G., D. W. Engel & R. M. Thuott  1978.**  Effect of chemical speciation on toxicity of cadmium to grass shrimp, *Palaemonetes pugio*: importance of free cadmium ion.  *Environmental Science and Technology*.  12: 409-413.

**Timperley, M. & T. Mathieson  2002.**  Marine sediment monitoring programme 1998-2001: Review of results and procedures.  Auckland Regional Council Technical Publication.  No. 193.  July 2002.  55pp.

**US EPA  1995.**  Ambient water quality criteria – saltwater copper addendum.  (Draft).  US Environmental Protection Agency, Environmental Research Laboratory, Narragansett, RI and Office of Water, Office of Science and Technology, Washington, DC.  14 April 1995.  35pp.

**Viarengo, A.  1989.**  Heavy metals in marine invertebrates; mechanisms of regulation and toxicity at the cellular level.  *CRC Critical Reviews in Aquatic Science*.  1(2): 295-317.

**Voulvoulis, N., M. D. Scrimshaw & J. N. Lester  1999.**  Alternative antifouling biocides.  *Applied Organometallic Chemistry*.  13: 135-143.

**Wikfors, G. H. & R. Ukeles  1982.**  Growth and adaptation of estuarine unicellular algae in media with excess copper, cadmium or zinc, and effects of metal-contaminated algal food on *Crassostrea virginica* larvae.  *Marine Ecology Progress Series*.  7: 191-206.

**Wikfors, G. H., J. W. Twarog, Jr., G. E. Ferris, B. C. Smith & R. Ukeles  1994.**  Survival and growth of post-set oysters and clams on diets of cadmium-contaminated microalgal cultures.  *Marine Environmental Research*.  37: 257-281.

**Williamson, R. B., K. J. Costley, G. N. Mills & S. E. Pickmere  1999.**  Marine Sediment Monitoring Programme Design and 1998 Results.  NIWA Client Report ARC90240, August 1998.  Auckland Regional Council Technical Publication, No. 107.  June 1999.

**Williamson, R. B. & G. N. Mills  2002.**  Sediment quality guidelines for the regional discharges project.  Unpubl. report prepared by Diffuse Sources Ltd. for the Auckland Regional Council.  April 2002.  71pp.

***Williamson, R. B. & D. J. Morrisey  2000.***  Stormwater contamination of urban estuaries.  1. Predicting the build-up of metals in sediments.  *Estuaries*.  23(1): 56-66

## 5.4 STRESSOR: ORGANOCHLORINES (OCs)

### 5.4.1 Hazard Identification

Organochlorines (OCs) are organic (carbon-based) nonpolar compounds that contain chlorine. Although a few of these compounds occur naturally, the vast majority are synthetically produced. Organochlorines are stable and non-ionic, which makes them persistent in the environment and gives them low water solubility (Williamson & Wilcock 1994). This low water solubility means that organochlorines are rapidly removed from the water column and accumulate in sediment. Overseas, organochlorine (e.g. PCB) contaminated marine sediments have resulted in restricted use of natural resources in some areas. For example, New Bedford Harbour (MA, USA) has been closed to finfish and shellfish harvesting since 1979 (Connolly 1992).

Organochlorines include: Dioxins which comprise polychlorinated dibenzo-p-dioxins (PCDDs, 75 possible congeners) and the polychlorinated dibenzofurans (PCDFs, 135 possible congeners); polychlorinated biphenyls (PCBs); organochlorine pesticides including dichloro-diphenyl-trichloroethanes (DDT), aldrin, dieldrin, endrin and chlordane; chlorophenols, in particular pentachlorophenol (PCP); hexachlorocyclohexanes (HCHs, e.g. lindane); and hexachlorobenzene (HCBs). DDT can be converted to the compounds dichloro-diphenyl-dichloroethane (DDE) or 1,1-dichloro-2,2-bis(p-chlorophenyl)ethane (DDD) that have longer half-lives than the original compound (Williamson & Wilcock 1994).

But how do organochlorines enter the marine environment? Dioxins are not produced intentionally but are released to the environment from a variety of industrial discharges, combustion processes, and because of their occurrence as unwanted by-products in various chlorinated chemical formulations (Scobie *et al.* 1999). Other organochlorines originate from agricultural use (i.e. herbicides and pesticides) - these include DDT, HCBs, the dieldrin group, chlordane and lindane. PCBs (e.g. Aroclor) were commercially produced overseas, and comprise a family of 209 chemical congeners with varying degrees of chlorination (Smith *et al.* 1999). Although, PCBs have never been manufactured in New Zealand, they have been imported and used extensively (Scobie *et al.* 1999). PCBs have been used in industry as heat transfer fluids, hydraulic fluids, solvent extenders, flame retardants and dielectric fluids (Waid, 1986). The accidental and deliberate dumping of these contaminants has led to global contamination and bioaccumulation of PCBs within the food chain. In New Zealand, wood preservation procedures use chlordane, PCPs, and associated chlorophenols (Shaw 1990, Williamson & Wilcock 1994). Gifford *et al.* (1995) have reviewed PCP contamination of freshwater systems in New Zealand.

Organochlorines may be presented to infaunal bivalves by means of several different exposure pathways, for example, from the water column (i.e. either dissolved, colloidal, or sorbed to suspended particulate matter), from the deposited sediment (including pore water) or via a food source (e.g. phytoplankton) (Burgess & McKinney 1999). Currently the dissolved phase is perceived to be the primary bioavailable exposure phase, even though nonpolar organic contaminants such as organochlorines are known to readily associate with particulate and colloidal phases far more than to the truly dissolved phase (Burgess *et al.* 1996). In the natural environment, the potential for exposure to each phase is constrained by several physico-chemical parameters, including contaminant solubility, lipophilicity and colloidal and particulate organic carbon (Burgess & McKinney 1999). Rates at which sediments containing organochlorines are ingested, deposited and subsequently reburied have been mainly studied in freshwater invertebrates under both field

and laboratory conditions (Mulsow *et al.* 2002). Most research on organochlorines in marine invertebrates has focused on contaminant bioaccumulation and depuration processes (Mulsow *et al.* 2002). However, Burgess and McKinney (1999) investigated the influence of different exposure pathways (e.g. interstitial water, overlying water or whole sediments) on PCB uptake by the suspension-feeder *Mulinia lateralis* and the deposit-feeder *Yoldia limatula*. PCB concentrations in each exposure type were separated into dissolved, colloidal and particulate phases (however in the overlying water only dissolved and colloidal phases were considered) and then compared to concentrations accumulated by the two bivalve species. Interstitial water concentrations reflected a combination of both dissolved and particulate phase PCBs. In these exposures the deposit-feeder *Y. limatula* accumulated more PCBs than the suspension-feeder *M. lateralis* (Burgess & McKinney 1999), with colloidal PCBs being accumulated more than those in the dissolved phase. Overlying water concentrations of PCBs were similar to those in interstitial waters. In all overlying water experiments, the suspension-feeding *M. lateralis* had greater accumulation of PCBs than *Y. limatula*. Both bivalves accumulated PCBs roughly equally from both the colloidal and dissolved phases (Burgess & McKinney 1999). In the whole sediment exposures, particulate phase PCB distributions were bimodal, one of low molecular weight PCBs, and one of medium molecular weight PCBs. For the deposit-feeding *Y. limatula*, PCBs accumulated were very similar to the particulate phase distributions, while for *M. lateralis* (a suspension-feeder) PCBs accumulated were correlated with the overlying water phase. Further, *Y. limatula* accumulated lower molecular weight PCBs while *M. lateralis* accumulated higher molecular weight PCBs (Burgess & McKinney 1999). These data suggest differences in PCB accumulation based on feeding strategy, especially in the early stages of exposure when *Y. limatula* accumulated far more PCBs than *M. lateralis*, probably as a result of direct sediment ingestion. In summary, Burgess and McKinney (1999) concluded that it was impossible to demonstrate the bioavailability of a single phase (i.e. dissolved, colloidal or whole sediment) from these exposure trials due to the high level of correlation between all phases used. However, the dissolved phase did not appear to play a significant role even in the suspension-feeding bivalve *M. lateralis*. This research suggests that organochlorines from the dissolved phase may not be the primary route of exposure to infaunal bivalves. Interestingly, the presence of sublethal concentrations (5-20 $\mu g \ g^{-1}$) of DDT in laboratory sediments modified sediment mixing and feeding rates in the deposit-feeding polychaete worm *Heteromastus filiformis* (Mulsow *et al.* 2002)

Organochlorines can be <u>very</u> persistent in the environment. For example, in the water column, pentachlorophenol (PCP) has a half-life ranging from 20 to 200 days (Mckay *et al.* 1997). The half-life of DDT in soil is 2 to 15 yr and ~150 yr in aquatic ecosystems (HSDB 1998). Field and laboratory studies in the United Kingdom demonstrated that very little breakdown of DDT occurred in estuarine sediments over the course of 46 days (WHO 1989). Although very little information exists on the half-life of many organochlorines in marine sediments, their half-lives in soil have been approximated as shown in Table 5.4.1 below.

| Organochlorine | Half-life in soil (temperate climate) |
|---|---|
| Aldrin | ~5 yrs |
| Dieldrin | >2 yrs |
| Endrin | Up to 12 yrs |
| Chlordane | ~ 1 yr |
| HCB | > 2.7 yrs |
| PCBs | > 40 days |
| Dioxins (1,2,3,4-TCDD) | 10 yrs |

**Table 5.4.1   Approximation of half-lives of selected organochlorines in soil (from ICCA 1998).**

Thus it may be prudent to assume that these sediment-associated organochlorines are potentially available to benthic organisms even after they are no longer produced (Frankel 1995) or introduced into the aquatic environment.

For the infaunal clam *Mya arenaria* laboratory exposure studies have established that water column dioxin (e.g. (3H)TCDD) is taken up through the gills and mobilised to the gonad which serves as a final reservoir (Rhodes *et al.* 1997).

The cellular response to dioxin toxicity in clams is unknown (Kelley & Van Beneden 2000), although it may be similar to vertebrates in which dioxin binds to the cytoplasmic aryl hydrocarbon receptor (AhR) and causes enhanced cell proliferation and differentiation (i.e. tumour formation). Lindane is a neurotoxin acting on the GABA receptor, which functions as a chlorine channel through the nerve membrane (Walker *et al.* 1996). Further, experimental evidence suggests that PCBs disrupt embryonic development of the nervous system in the clam *Spisula solidissima* (Smith *et al.* 1999).

A range of organochlorine exposure effects on bivalves have been reported in the international literature. These include:

- Sublethal neurotoxic effects on clam embryos (e.g. Smith *et al.* 1999 (*Spisula solidissima*))

- Mortality of larval clams (e.g. Portman 1972 (*Mercenaria mercenaria*), Davis & Hidu 1969 (*Mercenaria mercenaria*))

- Adverse effects on larval development (e.g. Davis & Hidu 1969 (*Mercenaria mercenaria*), Stiles *et al.* 1991 (*Mercenaria mercenaria*), Ryan *et al.* 2001 (*Mercenaria mercenaria*))

- Reduced growth in oysters (e.g. Parrish *et al.* 1976 (*Crassostrea virginica*))

- Reduced filtration rate (e.g. Mane *et al.* 1979 (*Katelysia optima*))

- Mortality in adult clams (e.g. Ikematsu *et al.* 1963, Eisler 1970 (*Mercenaria mercenaria*), Portman 1972 (*Cerastoderma edule*), Rajendran *et al.* 1989, (*Katelysia optima*)).

Dioxins containing 1, 2, or 3 chlorine atoms are thought to be of no toxicological significance (Scobie *et al.* 1999). However, 17 dioxin congeners with chlorine atoms in the 2,3,7,8-positions are thought to pose a risk to environmental health (Scobie *et al.* 1999). The presence of high levels (40%) of 2,4,5,T in the marine environment has been hypothesised to be the cause of high levels of gonadal tumours in *Mya arenaria* in eastern Maine, U.S.A (Brown *et al.* 1996, Rhodes *et al.* 1997). Further research has indicated that *Mya arenaria* may upregulate an E3 ubiquitin-protein ligase when exposed to dioxins ((3H)TCDD) which inhibits the degradation of tumour forming cells (Kelley & Van Beneden 2000). Wintermyer & Cooper (2000) identified that *Crassostrea virginica* gametes containing maternally transferred tetrachlorodibenzo-*p*-dioxin (TCDD) resulted in poor fertilization rates (52-99% unfertilised eggs) and poor survival (3.2%) compared to control embryos which had 80% survival through to the straight hinge stage.

The impact of chlordane on the New Zealand deposit-feeder *M. liliana* and the suspension-feeder *Austrovenus stutchburyi* has received considerable research attention, specifically with respect to the sediment exposure pathway (e.g. Pridmore *et al.* 1991, Pridmore *et al.* 1992, Wilcock *et al.* 1994 and Roper & Hickey 1994) after early sediment surveys indicated that organochlorine

contaminants in sediments of the Manukau Harbour were similar to other industrialized countries (e.g. Fox *et al.* 1988). This research has identified that chlordane dosed sediments cause significant behavioural modifications (e.g. avoidance behaviour) in juvenile infaunal clams while adults remain largely unaffected (Pridmore *et al.* 1991, 1992). However, there is very little comparative information available internationally on the toxicity of chlordane contaminated sediments to infaunal bivalves (Roper & Hickey 1994). In further investigations of chlordane uptake by adult *M. liliana* Wilcock *et al.* (1994) identified that *M. liliana* could accumulate chlordane from depths > 2 cm below the sediment-water interface, and suggest that this may occur *via* sediment ingestion. However, their body burdens were the lowest when chlordane was not present in the upper 2 cm of sediment (Wilcock *et al.* 1994). The total chlordane taken up by *M. liliana* was small in comparison with the quantities lost through dissolution in re-circulating seawater (Wilcock *et al.* 1994). There is very little information relating to the impact or organochlorines on *P. australis* and *P. subtriangulata*.

## 5.4.2    Dose-Response Assessment

### PCBs:

#### In the water column:
**Embryo (sublethal toxicity)**
- Smith *et al.* (1999) identified that *Spisula solidissima* embryos exposed to Aroclor 1254 (a commonly found mixture of PCBs) at 1-500 mg L$^{-1}$, caused a dose-dependent decrease in serotonergic cell number (these cells help shape the developing embryos nervous system) over periods of 24, 48 and 72 hours post-fertilisation.

**Larvae (sublethal toxicity)**
- Ryan *et al.* (2001) identified a 48-hr EC$_{40}$ (i.e. % developed to normal D-shape larvae) for Aroclor 1254 of approx. 9.97 mg L$^{-1}$ for *Mercenaria mercenaria* larvae. They note that the PCB effects demonstrated in this experiment might lead to selection of genetically resistant populations in the wild.

### OC pesticides:

#### In the water column:
**Embryo (sublethal toxicity)**
o   Davis & Hidu (1969) identified a 48-hr EC$_{50}$ for developing *Mercenaria mercenaria* eggs of 10 mg L$^{-1}$ when exposed to aldrin.

**Larvae (lethal toxicity)**
- Portmann (1972) reported a 48-hr LC$_{50}$ for DDT and dieldrin of 10 mg L$^{-1}$ for larvae of *Cerastoderma edule*.
- Davis & Hidu (1969) reported a 12-day LC$_{50}$ for aldrin of 410 µg L$^{-1}$ for larval (2 day old) *M. mercenaria*.

**Larvae (sublethal toxicity)**
o   Davis & Hidu (1969) identified significant impacts on growth of larval (2 day old) *M. mercenaria* at 250 µg L$^{-1}$ of aldrin over a 12-day study.

**Adult (lethal toxicity)**
- Eisler (1970) reported a 96-hr lethal concentration of 10 mg L$^{-1}$ for adult *M. mercenaria* when exposed to DDT, aldrin, dieldrin or endrin in a static toxicity test.
- Portmann (1972) reported a 48-hr LC$_{50}$ for DDT and dieldrin of 10 mg L$^{-1}$ for adult *C. edule*.

- Rajendran *et al.* (1989) reported lethal concentrations of DDT for adults of the estuarine clam *Kateylsia optima*. Lethal concentrations of DDT decreased through time as follows: 24-hr – 16.5 µg L$^{-1}$, 48-hr – 15.8 µg L$^{-1}$, 72-hr – 14.6 µg L$^{-1}$, 96-hr – 13.2 µg L$^{-1}$ and 120-hr – 11.0 µg L$^{-1}$.

**Adults (sublethal toxicity)**
- Parrish *et al.* (1976) established an EC$_{50}$ of 6.2 µg L$^{-1}$ for shell growth in the oyster (*Crassostrea virginica*) exposed to cis/trans-chlordane in solution.
- Mane *et al.* (1979) identified a change in filtration rate for *K. optima* when exposed to 1 mg L$^{-1}$ of endrin for a period of a few hours.

*In sediment:*
**Juvenile (lethal toxicity)**
- Roper and Hickey (1994) found a 10-day LC$_{50}$ of 238 ng g$^{-1}$ (dry wt) for *M. liliana* placed in chlordane dosed sediments.

**Juvenile (sublethal toxicity)**
- Pridmore *et al.* (1991, 1992) examined the impact of technical chlordane on the inhabitants of an inter-tidal sandflat in the Manukau Harbour, New Zealand. Technical chlordane was added to the inter-tidal sediment to give an initial concentration of 37 ng chlordane cm$^{-2}$. After three tidal cycles 38% of this chlordane had moved down into the sediment to depths greater than 2 cm deep (Pridmore *et al.* (1991, 1992). This left a mean near-surface sediment concentration of 2.8 to 8.7 ng chlordane g$^{-1}$ (dry fines). Five to thirteen tides after the addition of technical chlordane, densities of *A. stutchburyi* and *M. liliana* began to decline (the majority of these bivalves at this site were < 2.5 mm). This decline continued for another 31 tides (total densities declined by 30 and 40% of the original density respectively), then reversed as individuals repopulated the site. It was impossible to tell from this experiment if numbers of bivalves declined due to mortality or due to behavioural changes e.g. migration (Pridmore *et al.* 1991, 1992). However, migration may have played a key role as high numbers of juveniles were caught in drift nets surrounding the chlordane dosed site. Pridmore *et al.* (1991, 1992) also suggest that for the juvenile size classes used in this experiment, there is unlikely to be a difference in chlordane uptake due to feeding mode as young *M. liliana* may be suspension-feeders.
- Roper and Hickey (1994) identified that chlordane dosed sediments (4-40 ng chlordane g$^{-1}$ (dry wt)) caused a significant avoidance movement away from the dosed sediment up to 20 ng g$^{-1}$ (dry wt). However, increasing chlordane concentrations above this reduced movement away from dosed sediments (Roper & Hickey 1994). Further Roper & Hickey (1994) found that chlordane had no affect on burial rates of *M. liliana* when compared to control sediments.

**Adult (sublethal toxicity)**
- Pridmore *et al.* (1991) found that adult bivalves were largely unaffected when chlordane was applied to inter-tidal sandflat sediments in the Manukau Harbour for a 3 month period. Large *M. liliana* accumulated 4 times as much chlordane as the similar sized *A. stutchburyi*. This difference may be as a consequence of their different feeding modes or because of differences in chlordane metabolism (Pridmore *et al.* 1991).

## HCHs:

*In the water column:*
**Larvae (lethal toxicity)**
- Portmann (1972) reported a 48-hr LC$_{50}$ for lindane of 10 mg L$^{-1}$ for larvae of *Cerastoderma edule*.

- Davis & Hidu (1969) reported a 12-day $LC_{50}$ for lindane of 10 mg $L^{-1}$, for larval (2 day old) *Mercenaria mercenaria*.

**Adult (lethal toxicity)**

- Rajendran *et al.* (1989) reported lethal concentrations of lindane for adults of the estuarine clam *Kateylsia optima*. Lethal concentrations of lindane decreased through time as follows: 24-hr – 38.5 µg $L^{-1}$, 48-hr – 37.6 µg $L^{-1}$, 72-hr – 23.0 µg $L^{-1}$, 96-hr – 18.3 µg $L^{-1}$ and 120-hr – 15.0 µg $L^{-1}$.

**Adult (sublethal toxicity)**

- Hamza-Chaffi *et al.* (1998) identified that adult *Ruditapes decussatus* exposed to 34.5 µg $L^{-1}$ of lindane for 5-days had significantly decreased acetylcholinesterase activity (an enzyme essential to the correct transmission of nerve impulses) when compared to control clams.

## *In summary:*

Organochlorines appear to display only moderate/low level acute toxicity to larval and adult infaunal bivalves via the water column. However, the impact of chronic low-dose water-column exposure of infaunal bivalves to organochlorines needs further investigation, as does the impact of organochlorines associated with sediment. Very little information is available about the impact of organochlorines on the reproductive capacity of adult infaunal bivalves.

## 5.4.3    Exposure Assessment

Changes in the regulations regarding organochlorine use over the last 40 years are likely to have shaped the level of these contaminants present in the marine environment today.

*"The use of pesticides in New Zealand was not subject to compulsory regulatory control until the Agricultural Chemicals Act 1959 established the Agricultural Chemicals Board. The use of persistent organochlorine pesticides was then progressively restricted by a succession of legislation, so that, by the mid 1970s their use had effectively ceased in agriculture and horticulture. All persistent organochlorine pesticides* [including DDT, lindane, aldrin, dieldrin, chlordane, hexachlorobenzene (HCB), Heptachlor, Endrin and Toxaphene] *except PCP were formally deregistered by the Pesticides Board in 1989, and PCP was deregistered in 1991"* (Scobie *et al.* 1999). This would suggest that many of these organochlorine pesticides (with the exception of DDT and endrin) are no longer entering the marine environment from land-based use (based on the half-lives in soil for these organochlorines provided previously). However, Hamza-Chaffai *et al.* (1998) comment that although the use of lindane in most countries is now prohibited, this organochlorine persists in soils and may reach the marine environment through erosion processes.

With respect to dioxins, PCDDs and PCDFs released into the environment are thought to have historically resulted from manufacture and use of the herbicide 2,4,5-T, the use of PCP in the timber industry and spillages of PCBs (Scobie *et al.* 1999). The manufacture of 2,4,5-T in New Zealand ceased in 1987, although some stocks remained which were likely to have been used after this date (Scobie *et al.* 1999). However, current inputs of dioxins to the environment include: waste incineration, metallurgical industries, industrial and domestic coal and wood combustion, exhaust emissions from diesel and unleaded petrol, controlled and uncontrolled fires and sewage wastes (Scobie *et al.* 1999). The current level of dioxins reaching the marine environment from these sources is difficult to estimate.

PCBs have never been manufactured in New Zealand, but have been imported and used extensively (Scobie *et al.* 1999). *"In March 1986, the New Zealand Customs Department placed a prohibition on importing PCBs, and later that year regulations to control the importation of PCBs were*

*promulgated as an amendment to the Toxic Substances Regulations 1983. In 1988, a further amendment to the Toxic Substances Regulations 1983 prohibited the use and storage of PCBs with effect from 1 January 1994. Following two extensions, this regulation came into effect on 1 August 1995"* (Scobie *et al.* 1999). PCBs may still be entering the marine environment today from these historic sources.

There is a lack of extensive New Zealand survey data on organochlorine contaminants in the marine environment (Scobie *et al.* 1999). Very little information exists concerning the concentrations of the organochlorines of interest in this study in seawater. Limited information is available for organochlorine concentrations in sediment. To gain an understanding of the possible trends of organochlorine concentrations in marine sediments and thus the potential exposure to infaunal bivalves, a review of past organochlorine sediment studies is appropriate. Table 5.4.2 (next page) outlines the results of several studies which have measured various organochlorines in marine sediments in New Zealand.

Initially, Fox *et al.* (1988) measured organochlorine contaminant levels in sediment from the Manukau Harbour. PCB levels in sediment at some sites near industrial land use were found to be high. For example, Mangere Inlet had a total PCB concentration of 14.4 ng g$^{-1}$ (dry wt). This study found that levels of total DDT were fairly uniformly distributed throughout the harbour (1.2-2.3 ng g$^{-1}$ (dry wt)), suggesting a diffuse source (e.g. catchment land use) or prolonged redistribution throughout the harbour by the action of tides, wind and biota (Fox *et al.* 1988). Total dieldrin was found to have a uniform distribution within the Manukau Harbour ranging from 0.3 to 0.5 ng g$^{-1}$ (dry wt) which again suggests a diffuse source possibly from pastoral applications. Dieldrin use on pastures was discontinued in 1966 (Harrison 1973). However, chlordane was found to be approximately twice the concentration of DDT in these sediments (e.g. 0.9 to 5.3 ng g$^{-1}$ (dry wt)) (Fox *et al.* 1988).

Methoxychlor and endosulfan were not detected at any sites, while HCBs were only detected in trace amounts. Small amounts (0.1-0.2 ng g$^{-1}$ (dry wt)) of pentachloroanisole a derivative of PCP were also detected. In summary the highest levels of organochlorines found in the Fox *et al.* (1988) study were for PCBs, DDT, chlordane and lindane. Fox *et al.* (1988) suggest that due to New Zealand's isolation and small population, atmospheric transport of organochlorines into the marine environment was likely to be insignificant.

In late 1989, Holland and co-workers (1993) further investigated the variability of organic contaminants in the inter-tidal sediments of the Manukau Harbour. Results from this study are presented in Table 5.4.2 on the following page.

In 1996 the Ministry for the Environment (MfE) commenced a national *"Organochlorines Programme"* to characterise the extent of contamination of the New Zealand environment by selected organochlorines (including dioxins (e.g. PCDDs and PCDFs), PCBs, HCHs, organochlorine pesticides (DDT, aldrin, dieldrin and chlordane) and chlorophenols (e.g. PCP) (Scobie *et al.* 1999). The only sampling site which is pertinent to the Hauraki Gulf Marine Park area is Hellyers Creek in the Upper Waitemata Harbour (see Table 5.4.2). No samples were taken from the Coromandel Peninsula in this study. However to potentially investigate any general long term trends in organochlorine concentration in inter-tidal sediment through time, a comparison with the data of Fox *et al.* (1988), and Holland *et al.* (1993) for "Big Muddy Creek" from the Manukau Harbour is also provided (see Table 5.4.2).

**Table 5.4.2: Summary of organochlorine concentrations found in inter-tidal sediments around New Zealand. Data values followed by * are not comparable between separate studies due to differences in analytical methodologies used.**

| Organochlorine | Fox et al. (1988) Manukau | | Holland et al. (1993) Manukau (sample date 1989) | | MfE (1999) New Zealand wide study (sample date 1996) | | |
| --- | --- | --- | --- | --- | --- | --- | --- |
| | Min-Max range ng g$^{-1}$ (dry wt) | Big Muddy Creek ng g$^{-1}$ (dry wt) | Min-Max range ng g$^{-1}$ (dry wt) | Big Muddy Creek ng g$^{-1}$ (dry wt) | Min-Max range ng g$^{-1}$ (dry wt) | Big Muddy Creek, Manukau ng g$^{-1}$ (dry wt) | Hellyers Creek, Waitemata ng g$^{-1}$ (dry wt) |
| PCBs | 0.5-14.2* | 2.1* | 0.20-2.08* | 0.21-0.33* | 0.12-8.8* | 0.16* | 2.05-8.8* |
| Total DDT | 1.2-2.3* | 1.2* | 0.07-21.93* | 0.09-.61* | <0.04-5.2* | 0.09* | 1.35-1.75* |
| op- & pp-DDT | 0.1 | 0.1 | 0.13-1.28 | 0.14-0.48 | <0.02-0.48 | <0.03 | 0.24-0.48 |
| pp-DDE | 0.3-0.8 | 0.3 | 0.02-0.45 | 0.02-0.05 | <0.01-3.29 | <0.04 | 0.69-1.18 |
| Lindane | <0.1-1.4 | 1.1 | 0.03-0.50 | 0.03-0.34 | <0.01-0.046 | <0.02 | <0.03-0.046 |
| Total HCH | 0.1-2.0 | 1.5 | Not included | Not included | <0.03-<0.066 | <0.04 | <0.05-<0.066 |
| Aldrin | Not detected | Not detected | Not included | Not included | <0.01 | <0.01 | <0.01 |
| Endrin | Not detected | Not detected | Not included | Not included | Not included | Not included | Not included |
| Dieldrin | 0.3-0.5 | 0.3 | 0.05-0.41 | 0.05-0.14 | <0.05-0.38 | <0.1 | 0.21-0.34 |
| Total Chlordane | 0.9-5.3* | 1.4* | Not included | Not included | <0.06-<0.25* | <0.12* | <0.18-<0.25* |
| α- & γ-Chlordane | 0.5-2.9 | 0.5 | 0.04-0.38 | 0.04-0.10 | <0.04-0.19 | <0.08 | <0.1-0.19 |
| Heptachlor & Heptachlor epoxide | 0.2-0.7 | 0.7 | Not included | Not included | <0.02-<0.08 | <0.04 | <0.058-0.08 |
| Methoxychlor | Not detected | Not detected | Not included | Not included | Not included | Not included | Not included |
| Endosulfan | Not detected | Not detected | Not included | Not included | Not included | Not included | Not included |
| HCBs | Trace <0.1 | Trace <0.1 | Not included | Not included | <0.01-0.83 | <0.01 | 0.036-0.075 |
| PCP | Not included | Not included | Not included | Not included | <0.3-0.4 | <0.3 | <0.3-0.4 |

The MfE study (Scobie et al. 1999) was the first to quantify the level of dixons present in inter-tidal marine sediments, so the identification of trends in concentrations of these contaminants through time in New Zealand is not possible. Dioxins were present in sediment at Hellyers Creek at concentrations between 1.47 - 2.71 ng I-TEQ kg$^{-1}$ (dry wt) (TEQ = toxic equivalents of each congener measured) (Scobie et al. 1999). This sample, along with 11 out of 18 other New Zealand sites where PCDDs and PCDFs were measured, was above the Canadian sediment quality guidelines for the protection of aquatic life (i.e. 0.25 ng I-TEQ kg$^{-1}$ (dry wt)) (Scobie et al. 1999). The specific relevance of this exceedence to infaunal bivalve species can not be evaluated on the currently available information. There are little other published data on concentrations of dioxins in the New Zealand estuarine or coastal environment (Scobie et al. 1999).

PCBs do occur in marine sediments in the Hauraki Gulf Marine Park. However, due to the differences in analytical methods used (i.e. different PCB congeners measured) in the studies outlined in Table 5.4.2 it is impossible to identify any temporal trends in PCB concentrations in New Zealand inter-tidal marine sediments. In 1990 the Auckland Regional Council measured PCB concentrations of between 0.57-16.6 ng g$^{-1}$(dry wt) in sediments of the Tamaki estuary (ARC 1992). Total PCBs concentration measured at the Hellyers Creek site were between 218-471 µg kg$^{-1}$ TOC or 2.05-8.80 ng g$^{-1}$(dry wt) (Scobie et al. 1999). The current spatial and temporal distribution of PCBs in inter-tidal sediments in the Hauraki Gulf Marine Park remains largely unknown.

DDT does occur in marine sediments in the Hauraki Gulf Marine Park. However, due to the differences in analytical methods used (i.e. different DDT breakdown products analysed) in the studies outlined in Table 5.4.2, it is impossible to identify any temporal trends in Total DDT

concentrations in New Zealand inter-tidal marine sediments. Total DDT concentrations measured at the Hellyers Creek site in the MfE study were between 1.35-1.75 ng g$^{-1}$(dry wt) (Scobie *et al.* 1999). Comparison of the parent op- and pp-DDT residues and the pp-DDE breakdown products is possible between these studies, but only for one site (Big Muddy Creek, Manukau Harbour). These data (see Table 5.4.2) tend to suggest a decrease in parent DDT residues at this site. However, the range of values obtained from all Manukau sites e.g. <0.03-<0.1 in the MfE study would suggest no change in parent DDT through time. The breakdown product pp-DDE appears to have remained unchanged since 1989 at the Big Muddy Creek site. How these data relate to levels of DDT in sediments in the Hauraki Gulf Marine Park is difficult to evaluate. Some more recent organochlorine analysis in marine sediments has also been undertaken by the North Shore City Council as part of their sediment monitoring programme (Le Fevre & Silyn Roberts 2002). This survey identified DDT and its derivatives at several sites, with Total DDT being between 0.54-96.7 ng g$^{-1}$ (dry wt). The lowest concentration was recorded from Okura River head, while the highest concentration was recorded in sediment from Esmonde Rd estuary. The top end of this range is far higher than any values previously recorded in Table 5.4.2. Commercial ports and wharf areas may have elevated levels of total DDT in their associated sediments (e.g. Viaduct basin, Total DDT of 31 ng g$^{-1}$ (dry wt) (Kingett Mitchell 1996)).

Lindane does occur in marine sediments in the Hauraki Gulf Marine Park. Lindane was measured in sediment at the Hellyers Creek site at a concentration of 0.046 ng g$^{-1}$(dry wt). This was the highest level recorded in sediment (Scobie *et al.* 1999). Comparison of lindane concentrations in sediment is possible between these studies, but only for one site (Big Muddy Creek, Manukau Harbour). Lindane appears to have declined in sediments at the Big Muddy Creek site. In 1996, this site had a concentration of <0.02 ng g$^{-1}$(dry wt) (Scobie *et al.* 1999) compared to a level of 1.1 ng g$^{-1}$(dry wt) from the Fox *et al.* (1998) study. The same trend is apparent when a comparison of the range is undertaken for all Manukau sites, e.g. <0.02-0.31 ng g$^{-1}$(dry wt) (Scobie *et al.* 1999) compared to <0.1-1.4 ng g$^{-1}$(dry wt) (Fox *et al.* 1988). Lindane may be declining in coastal inter-tidal sediments and presently occurs at very low levels. More recent organochlorine analysis in marine sediments has also been undertaken by the North Shore City Council (Le Fevre & Silyn Roberts 2002) as part of their sediment monitoring programme. Lindane was not detected at any of the North Shore City sediment monitoring sites (Le Fevre & Silyn Roberts 2002).

Aldrin and endrin appear to be present only in trace amounts in New Zealand inter-tidal sediments (see Table 5.4.2). More recent organochlorine analysis in marine sediments has also been undertaken by the North Shore City Council (Le Fevre & Silyn Roberts 2002) as part of their sediment monitoring programme. Aldrin and endrin were not detected at any of the North Shore City sediment monitoring sites (Le Fevre & Silyn Roberts 2002).

Dieldrin does occur in marine sediments in the Hauraki Gulf Marine Park. At Hellyers Creek dieldrin was measured at levels between 0.21-0.34 ng g$^{-1}$ (dry wt) (this was close to the maximum concentration of 0.38 ng g$^{-1}$ (dry wt) measured in the New Zealand-wide study) (Scobie *et al.* 1999). Comparison of dieldrin concentrations in sediment is possible between these studies, but only for one site (Big Muddy Creek, Manukau Harbour). In the Manukau Harbour the Big Muddy Creek site had a concentration of <0.1 ng g$^{-1}$(dry wt) compared with 0.3 ng g$^{-1}$(dry wt) from the Fox *et al.* (1998) study. The same trend is apparent when a comparison of the range is undertaken for all Manukau sites, i.e. <0.1-0.27 ng g$^{-1}$(dry wt) (Scobie *et al.* 1999) compared to 0.3-0.5 ng g$^{-1}$(dry wt) (Fox *et al.* 1988). Dieldrin concentrations may be decreasing in inter-tidal sediments of the Manukau Harbour. How these data relate to levels of dieldrin in sediments in the Hauraki Gulf Marine Park is difficult to evaluate. However, low concentrations (0.07 – 0.15 ng g$^{-1}$) of dieldrin have been reported in marine sediments off East Coast Bays (NIWA 1999). More recent

organochlorine analysis in marine sediments has also been undertaken by the North Shore City Council (Le Fevre & Silyn Roberts 2002) as part of their sediment monitoring programme. Dieldrin was not detected at any of the North Shore City sediment monitoring sites (Le Fevre & Silyn Roberts 2002).

Chlordane does occur in marine sediments in the Hauraki Gulf Marine Park. In the 1996 MfE study concentrations of total chlordane and total heptachlor were only detected in one sediment sample (above the level of detection) which was from upper Hellyers Creek, with a concentration of 0.187 ng g$^{-1}$ (dry wt) chlordane and 0.018 ng g$^{-1}$ (dry wt) heptachlor (Scobie *et al.* 1999). Due to the differences in analytical methods used (i.e. different chlordane breakdown products analysed) in the studies outlined in Table 5.4.2 it is impossible to identify any temporal trends in Total chlordane concentrations in New Zealand inter-tidal marine sediments. Comparison of the sum of $\alpha$ - and $\gamma$ - chlordane, and Heptachlor and Heptachlor epoxide is possible between these studies, but only for one site (Big Muddy Creek, Manukau Harbour). In the Manukau Harbour, the Big Muddy Creek site had a concentration of <0.08 ng g$^{-1}$(dry wt) (Scobie *et al.* 1999) $\alpha$ - and $\gamma$ - chlordane compared with 0.5-2.9 ng g$^{-1}$(dry wt) from the Fox *et al.* (1988) study. The same trend is apparent when a comparison of the range is undertaken for all Manukau sites, e.g. <0.08-0.19 ng g$^{-1}$(dry wt) (Scobie *et al.* 1999) compared to 0.3-0.5 ng g$^{-1}$(dry wt) (Fox *et al.* 1988). In the Manukau Harbour the Big Muddy Creek site had a concentration of <0.04 ng g$^{-1}$(dry wt) (Scobie *et al.* 1999) heptachlor and heptachlor epoxide compared with 0.7 ng g$^{-1}$(dry wt) from the Fox *et al.* (1988) study. The same trend is apparent when a comparison of the range is undertaken for all Manukau sites, e.g. <0.02-<0.07 ng g$^{-1}$(dry wt) (Scobie *et al.* 1999) compared to 0.2-0.7 ng g$^{-1}$(dry wt) (Fox *et al.* 1988). Thus chlordane concentrations appear to be decreasing in inter-tidal sediments of the Manukau Harbour. How these data relate to levels of chlordane in sediments in the Hauraki Gulf Marine Park is difficult to evaluate. More recent organochlorine analysis in marine sediments has also been undertaken by the North Shore City Council (Le Fevre & Silyn Roberts 2002) as part of their sediment monitoring programme. Chlordane was not detected at any of the North Shore City sediment monitoring sites (Le Fevre & Silyn Roberts 2002).

Only very small amounts of HCBs were found in the studies undertaken to date (see Table 5.4.2).

Pentachlorophenol (PCP) does occur in marine sediments in the Hauraki Gulf Marine Park. No trends in this contaminant through time can investigated as it was not analysed in any previous surveys. Pentachlorophenol was measured at Hellyers Creek at just on the limit of detection (i.e. 0.4 ng g$^{-1}$ (dry wt)) for sediment (Scobie *et al.* 1999). Some industrialised areas may have higher levels than this - for example, in 1989-90 the ARC measured total chlorophenol concentrations of 0.8-12.0 ng g$^{-1}$ (dry wt) in sediments of the Tamaki estuary (and identified a contaminant gradient from the inner to the outer estuary (ARC 1992).

Thus in summary, it is difficult to evaluate temporal trends in the concentration of Dioxin, PCBs, DDT and PCPs in inter-tidal marine sediments. However, PCBs were found to be higher in sites near urbanised areas (Scobie *et al.* 1999) Lindane, dieldrin and chlordane may be declining in inter-tidal sediments. Aldrin and endrin appear to be virtually absent. HCBs appear to be present in very low concentrations. Scobie *et al.* (1999) summarise their study of organochlorine pesticides in inter-tidal New Zealand sediments by stating that "*organochlorine pesticide concentrations are low in New Zealand sediments*", but that "*the major organochlorine residues in sediments are those originating from DDT*" (i.e. an order of magnitude higher than other residues).

Another indication of trends in organochlorine contamination of the New Zealand marine environment can be gained from the ARC Manukau Harbour Action Plan – Shellfish Quality

Survey (Evans 2001). Shellfish quality surveys of *Crassostrea gigas* flesh have been undertaken at four sites (i.e. Cornwallis, Granny's Bay, Pahurehure Inlet and Hingaia Bridge) within the Manukau Harbour annually since 1987. Organochlorines included in this survey (normalised for tissue lipid content) include total chlordane, PCB's, total DDT, lindane and dieldrin. Trend analysis of the last 14 years data by Evans (2001) identified declines in lipid normalised oyster tissue concentrations of all organochlorines measured (e.g. total chlordane, total DDT, and dieldrin) at all sites, with the exception of PCB's and lindane. Concentrations of total PCB's in oysters have decreased over the 14 year survey period at the Cornwallis and Granny's Bay sites. However, no trends are apparent at the Hingaia Bridge or Pahurehure Inlet sites. Lindane concentrations have been lower than the detection limit in the majority of samples over the 14 years of the survey, and thus trend analysis is redundant. These results suggest that the environmental levels of these organochlorine contaminants in the Manukau Harbour are declining.

The spatial distribution of these organochlorines is difficult to identify based on the limited survey data available. The widespread use of many of the organochlorine herbicides and pesticides would suggest a diffuse source of these organochlorines via erosion and run-off from land along the entire coastline of the Hauraki Gulf Marine Park. This suggests a diffuse exposure of all infaunal bivalve populations within the Hauraki Gulf Marine Park. Dioxins and PCBs, which originate largely from industrial and urban sources, could be considered to be more abundant in the marine environment adjacent to urbanised/industrialised land usage.

The level of bioavailability of these organochlorines when bound to sediments to infaunal bivalves is currently unknown.

The influence of feeding mode on organochlorine uptake by infaunal bivalves has been investigated with respect to New Zealand and overseas species. Concentrations of organic contaminants in the suspension-feeder *A. stutchburyi* have been reported to be lower than in the deposit-feeder *M. liliana* from the Manukau Harbour (Hickey *et al.* 1995). These differences may be attributable to differences in metabolism and/or exposure via different feeding modes. An overseas study by Burgess and McKinney (1999) suggested differences in PCB accumulation based on feeding strategy, especially in the early stages of exposure when the deposit-feeder *Y. limatula* accumulated far more PCBs than the suspension-feeder *M. lateralis*, probably as a result of direct sediment ingestion.

## 5.4.4    Risk Characterisation

Many organochlorines have been banned from continued use e.g. DDT, aldrin, dieldrin, endrin, chlordane, PCPs, lindane and HCBs) and thus their current concentrations seawater and sediments within the Hauraki Gulf Marine Park are largely derived from their historical land-based usage. However, it is possible input of dioxins and PCBs into the marine environment still continues today. In general, organochlorines have low water solubility, which means that they are rapidly removed from the water column and accumulate in sediment. The long half-lives of many of the organochlorines make them particularly persistent in the environment. However, with the exception of DDT, the current levels of many of the organochlorine pesticides appear to be low and/or decreasing.

Based on limited sediment survey data withing the Hauraki Gulf Marine Park, aldrin and endrin do not appear to be present in marine sediments and are therefore likely to pose no risk to infaunal bivalves.

Organochlorines may be presented to infaunal bivalves by means of several different exposure pathways, for example, from the water column (i.e. either dissolved, colloidal, or sorbed to suspended particulate matter), from the deposited sediment (including pore water) or *via* a food source (e.g. phytoplankton) (Burgess & McKinney 1999).

The percentage of bioavailable organochlorine from sediment pools within the Hauraki Gulf Marine Park is currently unknown.  This adds uncertainty to our understanding of the level of risk.  Uncertainty is also associated with our limited knowledge of the spatial distribution of organochlorine concentrations within the Hauraki Gulf Marine Park, and with the limited amount of data available to identify temporal trends in the concentrations of these compounds.

Dose-response data shows moderate/low acute toxicity to larval infaunal bivalves from the dissolved phase.  When this is coupled with the fact that organochlorines may be largely particle-bound, a low risk of acute toxicity to suspension-feeding infaunal bivalves is suggested.  Chronic low dose impacts are difficult to evaluate as little information exists to help characterise the effects on infaunal bivalves.  While dose-response information exists on the lethal and sublethal toxicity of chlordane in sediments to New Zealand infaunal bivalves, the concentrations currently available to infaunal bivalves in marine sediments are well below these levels, suggesting that there is likely to be minimal impact from this source.

It has been suggested that the mode of feeding could impact on the level of exposure to organochlorines, with deposit-feeders accumulating more contaminants from the sediment (e.g. Gurgess & McKinney 1999).  The impact of this in terms of potential toxicity to infaunal bivalves is largely unquantified.

In summary, while the concentrations of many of the organochlorine pesticides are decreasing in marine sediments due to their banned land-based use, there are several organochlorines which may be or become problematic within the marine environment.  These include DDT, dioxins and PCBs.  The level of impact these organochlorines currently have on infaunal bivalve populations requires further investigation.  It is also worthy of note that as many of these organochlorine pesticides have been banned from use, new non-residual pesticides have been developed as replacements.  Some of these non-residual pesticides may be more toxic than the organochlorine compounds they have replaced (Sollod & Proulx 1998).  No risk assessment has been made of these replacement pesticides in our review.

## 5.4.5    References

**ARC, 1992.**  Tamaki catchment and estuary: Review of water quality.  ARC Environment and Planning Division Technical Publication No. 20, Auckland Regional Council.
**Brown, D. J., G. C. Clark, G. R. Gardner & R. J. Van Beneden 1996.**  Identification of dioxin-specific binding proteins in marine bivalves. *Marine Environmental Research*. 42(1-4): 7-11.
**Burgess, R. M., R. A. McKinney & W. A. Brown 1996.**  Enrichment of marine sediment colloids with polychlorinated biphenyls (PCBs): effects of PCB solubility and chlorination. *Environmental Science and Technology*. 30: 2556-2566.
**Burgess, R. M. & R. A. McKinney 1999.**  Importance of interstitial, overlying water and whole sediment exposures to bioaccumulation by marine bivalves. *Environmental Pollution*. 104: 373-382.
**Connolly, J. P. 1992.**  The relationship between PCBs in biota and in the water and sediment from New Bedford Harbour, a modelling evaluation.  *In*: Persistent pollutants in marine ecosystems.  Walker, C. H. & D. R. Livingston (Eds.).  Pergamon Press, New York.  Pp. 63-80.
**Davis, H. C. & H. Hidu 1969**.  Effects of pesticides on embryonic development of clams and oysters and on survival and growth of the larvae. *Fishery Bulletin*. 67(2): 393-404.
**Eisler, R. 1970.**  Latent effects of insecticide intoxication to marine molluscs. *Hydrobiologia*. 36(3-4): 345-352.

**Evans, B. 2001**. Annual Report, Manukau Harbour Shellfish Quality Survey 2000. Auckland Regional Council Technical Publication, No. 151, August 2001. 51pp.

**Fox, M. E., D. S. Roper & S. F. Thrush 1988.** Organochlorine contaminants in surficial sediments of Manukau Harbour, New Zealand. *Marine Pollution Bulletin*. 19(7): 333-336.

**Frankel, R. 1995.** Ocean environmental management. Prentice-Hall, New York.

**Gifford, J. S., M. C. Judd, P. N. McFarlane and S. M. Anderson 1995.** Pentachlorophenol (PCP) in the New Zealand environment: Assessment near contaminated sites and remote freshwater lakes. *Toxicological & Environmental Chemistry*. 48(1): 69-82.

**Hamza-Chaffai, A., M. Romeo, M. Gnassia-Barelli & A. El Abed 1998.** Effect of copper and lindane on some biomarkers measured in the clam *Ruditapes decussatus*. *Bulletin of Environmental Contamination and Toxicology*. 61: 397-404.

**Harrison, D. L. 1973.** Current status of organochlorine pesticide residues. *Proceedings of the Pollution Research Conference, Wairakei, New Zealand*. Pp. 409-415.

**Hazardous Substances Data Bank (HSDB) 1998.** The Department of Health & Human Services, national Library of Medicine, Division of Specialized Information Services. Toxicology and Environmental Health Information Program (TEHIP), Bethesada, MD.

**Hickey, C. W., D. S. Roper, P. T. Holland & T. M. Trower 1995.** Accumulation of organic contaminants in two sediment-dwelling shellfish with contrasting feeding modes: Deposit- (*Macomona liliana*) and filter-feeding (*Austrovenus stutchburyi*). *Archives of Environmental Contamination and Toxicology*. 29: 221-231.

**Holland, P.T., C. W. Hickey, D. S. Roper & T. M. Trower 1993.** Variability of organic contaminants in intertidal sandflat sediments from Manukau Harbour, New Zealand. *Archives of Environmental Chemistry*. 25: 456-463.

**Ikematsu, W., Y. Tanaka, M. Azeta, S. Ehira, H. Morokuma & S. Yamaguchi 1963.** The toxic effect of pentachlorophenate, a herbicide, on aquatic organisms. *Bulletin of the Seikai National Fisheries Research Institute*. Nagasaki. 28: 1-23.

**International Council of Chemical Associations (ICCA) 1998.** Procedure for identifying further persistent organic pollutant (POP) candidate substances for international action. April 1998.

**Kelley, M. L. & R. J. Van Beneden 2000.** Identification of an E3 ubiquitin-protein ligase in the softshell clam (*Mya arenaria*). *Marine Environmental Research*. 50(1-5): 289-293.

**Kingett Mitchell and Associates 1996.** Effects on the marine environment associated with the development of the proposed America's Cup harbour facility. Appendix F, 53pp.

**Le Fevre, N. & G. Silyn Roberts 2002.** North Shore City Council Sediment Sampling Programme. Technical Report, May 2002. Prepared by URS Ltd. for North Shore City Council. 78pp.

**Mane, U. H., M. S. Kachole & S. S. Pawar 1979.** Effect of pesticides and narcotants on bivalve molluscs. *Malacologia*. 18: 347-360.

**Mckay, D., W.-Y. Shui & K.-C. Ma 1997.** Illustrated Handbook of Physical-Chemical Properties and Environmental Fate of Organic chemicals. Lewis Publishers. Pp. 351 & 374.

**Mulsow, S., P. F. Landrum & J. A. Robbins 2002.** Biological mixing responses to sublethal concentrations of DDT in sediments by *Heteromastus filiformis* using a $^{137}$Cs marker layer technique. *Marine Ecology Progress Series*. 239: 181-191.

**National Institute for Water & Atmospheric Reserch (NIWA) 1999.** Oceanographic and water quality studies for the North Shore City Outfall. Assessment of Environmental Effects. Report prepared for the North Shore City Council by NIWA, May 1999.

**Parrish, P. S., S. C. Schimmel D. J. Hansen, J. M. Patrick & J. Forester 1976.** Chlordane: effect on several estuarine organisms. *Journal of Toxicology and Environmental Health*. 1: 485-494.

**Portmann, J. E. 1972.** Results of acute toxicity tests with marine organisms, using a standard method. *In*: Marine Pollution and Sea Life, FAO, Rome, Italy. Rulvo, M. (Ed.). Fishing News (Books) Ltd., London, England. Pp. 212-217.

**Pridmore, R. D., S. F. Thrush, R. J. Wilcock, T. J. Smith, J. E. Hewitt & V. J. Cummings 1991.** Effect of the organochlorine pesticide technical chlordane on the population structure of suspension and deposit feeding bivalves. *Marine Ecology Progress Series*. 76: 261-271.

**Pridmore, R. D., S. F. Thrush, V. J. Cummings & J. E. Hewitt 1992.** Effect of the organochlorine pesticide technical chlordane on intertidal macrofauna. *Marine Pollution Bulletin*. 24(2): 98-102.

**Rajendran, N., R. Rajendran, O. Matsuda & V. K. Venugopalan 1989.** Acute toxicity of organochlorine pesticides to fishes and shellfishes of a tropical estuary. *Journal of the Faculty of Applied Biological Sciences Hiroshima University*. 28(1/2): 37-48.

**Rhodes, L. D., G. R. Gardner & R. J. Van Beneden 1997.** Short-term tissue distribution, depuration and possible gene expression effects of (3H)TCDD exposure in soft-shell clams (*Mya arenaria*). *Environmental Toxicology and Chemistry*. 16(9): 1888-1894.

**Roper, D. S. & C. W. Hickey  1994**.  Behavioural responses of the marine bivalve, *Macomona* liliana exposed to copper- and chlordane-dosed sediments. *Marine Biology*.  118: 673-680.

**Ryan, R. L., K. L. Lachmayr, J. A. Jay & T. E. Ford  2001.**  Developmental effects of PCBs on the hard clam (*Mercenaria mercenaria*). *Journal of Environmental Science and Health*.  A36(9): 1571-1578.

**Scobie, S., S. J. Buckland, H. K. Ellis & R. T. Salter  1999.**  Organochlorines in New Zealand: Ambient concentrations of selected organochlorines in estuaries.  Organochlorines Programme.  Ministry for the Environment, Wellington, New Zealand.  178pp.

**Shaw, C. P.  1990**.  Timber preservation chemicals in the New Zealand environment.  Report for the Ministry of The Environment, by Cawthron Institute, Nelson, New Zealand.

**Smith, T. J., R. J. Wilcock, R. D. Pridmore, S. F. Thrush, A. F. Langdon, A. L. Wilkins & G. L. Northcott  1992.**  Persistence of chlordane applied to an intertidal sandflat.  *Bulletin of Environmental Contamination and Toxicology.*  49: 129-136.

**Smith, C. R., C. M. Baker, L. F. Baker, K. Jessen-Eller & C. L. Reinisch  1999**.  Polychlorinated Biphenyls (PCBs) selectively disrupt serotonergic cell growth in the developing *Spisula* embryo. *Toxicological Science*.  50: 54-63.

**Sollod,  A.  &  D.  Proulx  1998**.  Global  contamination,  wildlife  health  and  biotechnology.  http://www.biotech.icmb.utexas.edu/pages/wildlife.html.

**Stiles, S., J. Choromanski, D. Nelson, J. Miller, R. Greig & G. Sennefelder  1991**.  Early reproductive success of the hard clam (*Mercenaria mercenaria*) from five sites in Long Island Sound. *Estuaries*.  14(3): 332-342.

**Waid, J. S.  1986.** *PCBs and the environment*, CRC Press, Boca Raton, Florida.

**Walker, C. H., S. P. Hopkin, R. M. Silby & D. B. Peakall  1996**.  Biochemical effects of pollutants. *In*: Principles of ecotoxicology.  Taylor and Francis, London.  Pp. 131-146.

**Wilcock, R. J., R. D. Pridmore, G. L. Northcroft, J. E. Hewitt, S. F. Thrush & V. J. Cummings  1994**.  Uptake of chlordane by a deposit-feeding bivalve: Does the depth of sediment contamination make a difference? *Environmental Toxicology and Chemistry*.  13(9): 1535-1541.

**Williamson, R. B. & R. J. Wilcock  1994.**  The distribution and fate of contaminants in estuarine sediments:  Recommendations for environmental monitoring and assessment.  NIWA Consultancy Report No. ARC109/3.  Auckland Regional Council Technical Publication No. 47.  67pp.

**Wintermyer, M. L. & K. R. Cooper  2000.**  Tissue distribution of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) in the adult Eastern oyster (*Crassostrea virginica*, Gmelin) and its effect on egg and larval survival. *Marine Environmental Research*.  Abstract. Pp. 245-246.

**World Health Organization (WHO)  1989.** Environmental Health Criteria 83, DDT and its Derivatives: Environmental Effects. World Health Organization, Geneva.

## 5.5  STRESSOR:  POLYCYCLIC AROMATIC HYDROCARBONS (PAHs)

### 5.5.1     Hazard Identification:

Polycyclic aromatic hydrocarbons (PAHs, sometimes called polynuclear aromatic hydrocarbons) are organic compounds composed of two or more benzene rings fused together.  This chemical structure provides stability against biological and chemical breakdown, allowing PAHs to persist in the marine environment.

Polycyclic aromatic hydrocarbons are formed by a variety of processes, including indirect and direct biosynthesis, fossil fuel production and distribution (Lyons *et al.* 2002), and incomplete combustion of organic matter (Ahrens *et al.* 2002).  Some PAHs (e.g. fluoranthene) occur naturally in bacteria, fungi, and algae (Ahrens *et al.* 2002).  Once formed, PAHs can be transported into the aquatic environment by a number of pathways including fossil fuel distribution, stormwater runoff (Makepeace *et al.* 1995), and sewage effluent.  Secondary sewage treatment removes some PAHs, but most are released to aquatic environments through Sewage Treatment Plant outfalls (Eisler 1987).  Eisler (1987) suggests that PAHs from fossil fuels overwhelm all others in terms of global inputs.  With the increased use of diesel vehicles in New Zealand (a primary source of PAHs), there is some concern that urban levels of PAHs could increase to unacceptable levels (McIntosh 1999).  This may have flow on impacts for the coastal marine environment.  In an overseas study, Hoffman *et al.* (1984) noted that stormwater runoff from urban areas and highways accounted for 71% of the high molecular weight PAHs and 36% of the total PAH loading to Narrangansett Bay, U.S.A.  The load of PAH in urban runoff was higher from the highway and industrial land use sources than from commercial and residential areas (Hoffman *et al.* 1984).  These environmental contaminants can accumulate in marine and freshwater sediments (Newsted & Giesey 1987).

The 16 PAHs which are recommended as priority pollutants by the World Health Organisation, the European Economic Community, and the United States Environmental Protection Agency (Hellou 1996) are: naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[*a*]anthracene, chrysene, benzo[*b*]fluoranthene, benzo[*k*]fluoranthene, benzo[*a*]pyrene, indeno[*1,2,3-ed*]pyrene, dibenzo[*a,h*]anthracene and benzo[*g,h,i*]perylene.

Low molecular weight PAHs such as naphthalene, are water-soluble, and do not persist in the marine environment (Williamson & Wilcock 1994).  Dissolved PAHs are likely to degrade rapidly through photo-oxidation (U.S. EPA 1980).  They degrade most rapidly at higher concentrations, at elevated temperatures and oxygen levels, and at higher levels of solar irradiation.  Different PAHs vary significantly in their relative sensitivity to photo-oxidation (U.S. EPA 1980).  Roesijadi *et al.* (1978) examined the accumulation of naphthalenes from Prudhoe Bay crude oil contaminated sediments by the infaunal bivalve clam *Macoma inquinata*.  They found that the efficiency of PAH uptake by the clams from sediments was much lower than from water.  Conversely, the majority of PAHs are virtually insoluble in water and will bind preferentially to sediment and suspended matter (Williamson & Wilcock 1994).  Thus, the greatest fraction of PAH is associated with sediments, benthic organisms and pore waters, with concentrations in these compartments being several orders of magnitude greater than those of overlying waters (Meador *et al.* 1995).  The ultimate fate of PAHs that accumulate in sediments is believed to be biotransformation and degradation by benthic organisms (U.S. EPA 1980).  Cerniglia and Heitkamp (1991) addressed microbial degradation of PAHs in aquatic environments.  They note that a wide variety of bacteria, fungi and algae have demonstrated the ability to metabolize PAHs.  Low molecular weight PAHs, such as naphthalene,

degrade rapidly, while higher molecular weight PAHs such as benz(*a*)anthracene and benzo(*a*)pyrene are more resistant to microbial attack (Cerniglia & Heitkamp 1991). They also note that the most rapid biodegradation of PAHs occurs at the water/sediment interface. Deeper sediments usually contain little oxygen, thus inhibiting microbial metabolism (Cerniglia & Heitkamp 1991).

Sediment bound PAHs can be resuspended from bottom sediments into the water column by physical (i.e. water and wind induced) or biological (i.e. bioturbation) processes (Ciarelli *et al.* 1999). In dynamically bioturbated environments, PAH accumulation by benthic organisms from food and from resuspended sediment may be substantial (McElroy *et al.* 1990). Ciarelli *et al.* (1999) identified that bioturbation by an estuarine amphipod (*Corophium volutator*) affected the concentration of suspended solids in the water column, and consequently increased the aqueous concentration of fluoranthene in the water column and increased the uptake by suspension-feeders (*Mytilus edulis*). Neff (1979) attempted to integrate the degradative processes associated with PAH removal from aquatic environments. He concluded that the residence time of PAHs in marine waters is brief. The lower molecular weight aromatics (benzene to phenanthrene) are removed primarily by evaporation and microbial activity. Higher molecular weight PAHs are removed mainly by sedimentation and photo-oxidation (Neff 1979). In nutrient rich, biologically active, aerobic sediments, the degradation of PAH is dramatically increased by healthy bacterial and fungal communities. In anaerobic sediments, the heavier molecular weight PAH (4 through 7 rings) may persist for years (Neff 1979).

Several higher molecular weight PAHs are known to be carcinogenic to bottom dwelling fish (e.g. benz[*a*]pyrene, Baumann 1998), although no reference to carcinogenic effects (e.g. neoplasia or germinomas) in molluscs could be found. Many bioaccumulated PAHs cause additive, narcotic toxicity in exposed organisms (Swartz *et al.* 1995). Increased body burdens of PAHs have also been shown to lower the condition index in oysters (*Crassostrea virginica*, Bender *et al.* 1988) and mussels (*Mytilus edulis*, Granby & Spliid 1995). As nearly all PAHs are hydrophobic and lipophilic (Eisler 1987), there is the potential for these compounds to become associated with stable lipid pools in aquatic organisms. Energy is generally stored as glycogen in bivalves until gametogenesis when the glycogen and lipid stores are converted into eggs and sperm. The eggs contain significant lipid reserves and can become a repository for lipophilic PAHs. For example, Lowe and Pipe (1986) observed that long term exposure (i.e. months) to diesel oil (a source of PAHs) at 30 and 130 mg L$^{-1}$ caused internal degeneration of gametes produced by *M. edulis.*

Generally, PAHs do not show extremely high acute toxicity to aquatic life, with the majority of intermediate to high molecular weight PAHs not exhibiting acute toxicity within their water solubility limits (Lyons *et al.*, 2002). However, a growing body of evidence suggests that PAHs may pose a greater hazard to aquatic organisms due to their potential to become phototoxic (Pelletier *et al.* 1997, Ahrens *et al.* 2002 and Lyons *et al.* 2002). PAHs become phototoxic after the organism contaminated with them is exposed to ultraviolet (UV) radiation (Ahrens *et al.* 2002). Intermediate molecular weight PAHs, such as anthracene, fluoranthene and pyrene have demonstrated photo-induced toxicity (e.g. Newsted & Giesey 1987, Pelletier *et al.* 1997, Spehar *et al.* 1999, Ahrens *et al.* 2002 and Lyons *et al.* 2002). Laboratory studies to date have shown that in the presence of UV-light, the acute toxicity of fluoranthene and pyrene to marine bivalve larvae and embryos is significantly increased (up to 45 times) when compared with tests under fluorescent light (Spehar *et al.* 1999, Lyons *et al.* 2002). Ahrens *et al.* (2002) investigated the sublethal phototoxicity of fluoranthene (exposure was from dissolved fluoranthene in the water column) to juveniles of the New Zealand deposit-feeding bivalve *Macomona liliana*. The ability of the shellfish to re-bury into sediment was used as an indicator of sub-lethal effects. Exposure of

juvenile clams to 1 hr of UV light resulted in fluoranthene toxicity (i.e. reduced ability to rebury with an $EC_{50}$ of 46 µg $L^{-1}$) that was three times higher than without UV light ($EC_{50}$ of 153 µg $L^{-1}$). This suggests that phototoxic PAHs appear to be acutely toxic to juvenile bivalves at concentrations well below their aqueous solubility limits. Phototoxic PAHs are most likely to impact on small transparent organisms such as larval and post settlement infaunal bivalves (Pelletier *et al.* 1997). In the environment fluoranthene and pyrene appear to be the PAHs with the most potential impact on fish through photocytotoxicity (Schrimer *et al.* 1998).

N-heterocyclic PAHs (NPAHs) such as the azaarene family containing one in-ring nitrogen atom have also been shown to have phototoxic effects. Azaarenes do occur in natural systems (i.e. defence toxins of plants and sponges) but more commonly occur as by-products of wood processing and chemical manufacturing plants (Weigman *et al.* 2001). Although NPAHs are present in lower concentrations in the environment (e.g. 1-10% of PAH concentrations, Gissel-Nielsen & Nielsen 1996), they are more water-soluble than PAHs. Some NPAHs also display increased phototoxic effects. Weigman *et al.* (2001) have identified phototoxic effects of azaarene on the marine flagellate *Dunaliella tertiolecta*, demonstrating the effects of NPAHs [and possibly PAHs] on a necessary part of the bivalve food chain.

Polycyclic aromatic hydrocarbon phototoxicity is caused by the absorbance and transfer of UV radiation energy from the excited state of the PAH compound to molecular oxygen, thereby forming superoxide redox anions that cause redox cycling and subsequent cell death (Newsted & Giesey 1987).

In comparison to vertebrates, invertebrates metabolise PAHs relatively slowly, which can result in high concentrations of the parent compounds (Ankley *et al.* 1995). It has been shown that these compounds can be transferred to their offspring. The literature concerning PAH metabolism in molluscs is somewhat unclear (Simpson *et al.* 2002). Consequently, the role that bivalve metabolism plays in altering the composition of PAHs in the marine environment remains largely unknown. However initial work has shown that shellfish metabolism can convert pyrene to the water soluble conjugates of hydroxypyrene and pyrenediol in *Mya arenaria* and *Protothaca staminea* (Simpson *et al.* 2002). In a study of two New Zealand bivalve species (*M. liliana* and *Austrovenus stutchburyi*), Hickey *et al.* (1995) suggested that a generally high correlation in PAH concentrations bioaccumulated from sediment between each species indicated that metabolic detoxification mechanisms were possibly similar. These types of studies are important in determining whether metabolism by molluscs plays a significant role in the bioavailability of these compounds.

## 5.5.2    Dose-Response Assessment

## Toxicity

### *In phytoplankton:*
- Dobroski and Epifanio (1980) found that direct uptake of $^{14}$C-benzo[*a*]pyrene from seawater (at 10 µg $L^{-1}$) by the diatom *Thalassiosira pseudonana* was much greater than the rate of trophic transfer from the diatom to *Mercenaria mercenaria* clam larvae. This was attributed to greater efficiency of direct uptake and to the larger quantity of benzo[*a*]pyrene available in the water. A comparison of direct uptake by bivalves (as reported in the literature) with trophic transfer (measured in the present investigation) indicated that both processes may be equally important in accumulation of benzo[*a*]pyrene in natural populations of bivalves (Dobroski & Epifanio 1980).

*In the water column:*

**Embryo (lethal toxicity)**

- Spehar *et al.* (1999) reported that fluoranthene was not lethal to embryos of the estuarine suspension-feeding bivalve *Mulinia lateralis* at its saltwater solubility limit of 127 μg L$^{-1}$ (but see also comparable phototoxicity data below).

**Embryo (sublethal toxicity)**

- Pelletier *et al.* (1997) reported 48-hr EC$_{50}$s for development of embryonic *M. lateralis* for the PAHs, anthracene, fluoranthene and pyrene, of 4260, 58.8 and >11900 μg L$^{-1}$ respectively. These EC$_{50}$ results suggest that for both anthracene and pyrene there is no toxicity to embryonic *Mulinia lateralis* as the EC$_{50}$ values are above the seawater solubility limits of these PAHs. However, fluoranthene does exhibit toxicity to developing *M. lateralis* embryos in a dissolved state.

**Larvae (sublethal toxicity)**

- Davis & Hidu (1969) identified significant impacts on growth of larval (2 day old) *M. mercenaria* at between 10-50 μg L$^{-1}$ of phenanthrene over a 12-day study.

**Juvenile (lethal toxicity)**

- Pelletier *et al.* (1997) reported 96-hr LC$_{50}$s for juvenile *M. lateralis* for the PAHs, anthracene, fluoranthene and pyrene, of >13300, 3310 and >9454 μg L$^{-1}$ respectively. These LC$_{50}$ results suggest that for all the PAHs investigated there is no toxicity to juvenile *M. lateralis* as the LC$_{50}$ values are above the seawater solubility limits of these PAHs. However, see below for comparable phototoxicity data.

**Juvenile (sublethal toxicity)**

- Pelletier *et al.* (1997) reported 96-hr EC$_{50}$s for growth of juvenile *M. lateralis* for the PAHs, anthracene, fluoranthene and pyrene, of >13300, 900 and >9454 μg L$^{-1}$ respectively. These EC$_{50}$ results suggest that for all the PAHS investigated there is no toxicity to juvenile *M. lateralis* as the EC$_{50}$ values are above the seawater solubility limits of these PAHs. However, see below for comparable phototoxicity data.

- In a New Zealand study, Ahrens *et al.* (2002) reported a 96-hr EC$_{50}$ of 153 μg L$^{-1}$ of fluoranthene for total numbers of juvenile (0.5 -2.0 mm) *M. liliana* reburying. This EC$_{50}$ result suggests that for fluoranthene there is no toxicity to juvenile *M. liliana* as the EC$_{50}$ value is above the seawater solubility limits of this PAH. However, see below for comparable phototoxicity data.

**Adult (sublethal toxicity)**

- Donkin *et al.* (1989) identified a 1.67-hr EC$_{50}$ of 380 μg L$^{-1}$ of acenaphthene on feeding behaviour of adult *Mytilus edulis* in a static toxicity test.

## Phototoxicity

*In phytoplankton:*

- Weigman *et al.* (2001) have observed phototoxic effects of azaarenes (NPAHs) on the marine flagellate *Dunaliella tertiolecta*, with 72-hr EC$_{50}$s ranging between 0.11- 571 μM.

*In the water column:*

**Embryo (lethal phototoxicity)**

- Spehar *et al.* (1999) reported a 48-hr LC$_{50}$ of 2.8 μg L$^{-1}$ fluoranthene for embryos of the bivalve *Mulinia lateralis*.

- Lyons *et al.* (2002) identified that pyrene and benzo[*a*]pyrene completely inhibited the development of any *Crassostrea gigas* embryos to D-shape larvae at a concentration of 5 µg L$^{-1}$ for 48-hr, with UV light.

**Embryo (sublethal toxicity)**
- Pelletier *et al.* (1997) reported 48-hr EC$_{50}$s for development of embryonic *M. lateralis* for the PAHs anthracene, fluoranthene and pyrene, of 6.47, 1.09 and 0.23 µg L$^{-1}$ respectively.

**Juvenile (lethal phototoxicity)**
- Pelletier *et al.* (1997) reported 96-hr LC$_{50}$s for juvenile *M. lateralis* for the PAHs anthracene, fluoranthene and pyrene, of 68.9, 1.8 and 1.68 µg L$^{-1}$ respectively.

**Juvenile (sublethal phototoxicity)**
- Pelletier *et al.* (1997) reported 96-hr EC$_{50}$s for growth of juvenile *M. lateralis* for the PAHs, anthracene, fluoranthene and pyrene, of >82.8, >0.81 and >0.91 µg L$^{-1}$ respectively.
- In a New Zealand study, Ahrens *et al.* (2002) reported a 96-hr EC$_{50}$ of 46 µg L$^{-1}$ of fluoranthene for total numbers of juvenile (0.5 -2.0 mm) *M. liliana* reburying after exposure to 1 hr of UV radiation. UV exposure for more than 2 hrs at 50 µg L$^{-1}$ killed all juvenile *M. liliana*.

*In summary:*

Dissolved PAHs may be incorporated into phytoplankton, which in turn can provide an alternate exposure pathway for PAH accumulation in infaunal bivalves (Dobroski and Epifanio 1980). In general PAHs are not acutely toxic to infaunal bivalves within their seawater solubility limits in the absence of UV light. However, upon the addition of UV light PAHs become acutely toxic to embryonic, larval and post-settlement infaunal bivalves (Pelletier *et al.* 1997, Ahrens *et al.* 2002).

## 5.5.3     Exposure Assessment

PAH contamination of the water-column and sediments does occur within the Hauraki Gulf Marine Park. It is probable that the major source of PAHs entering the marine environment is from fossil fuel combustion products (Williamson & Wilcock 1994). PAHs from this source enter the marine environment via stormwater discharges. The concentration of PAHs from stormwater discharge is predicted to increase in estuarine sediments in the urbanized Auckland region in the future (Williamson & Mills 2002). However, in general most ARC sediment monitoring sites show low PAH accumulation rates (Timperley & Mathieson 2002). It is difficult to evaluate the level of PAHs in coastal sediments in other parts of the Hauraki Gulf Marine Park. However, in a study investigating parasite loads in *A. stutchburyi* as a potential indicator of estuarine health, Tricklebank and Stewart (2002) evaluated the PAH load of sediments from Whangateau Harbour, Okura Estuary, Waitemata Harbour, Tamaki Estuary, Mangemangeroa Estuary and Wairoa Estuary. PAHs were detected in sediments from the Waitemata Harbour, Tamaki Estuary and Mangemangeroa, but were below the detection limit in samples from Whangateau, Okura and Wairoa. This suggests that it is likely that inter-tidal habitats in rural areas will have a far lower PAH sediment load. As the relative persistence of different PAHs in the marine environment is largely a function of their molecular weight, infaunal bivalves are most likely exposed to the medium weight PAHs such as fluoranthene and pyrene (Williamson & Wilcock 1994). The major PAHs found in sediments and bivalves of the NZ coastal environment are fluoranthene and pyrene (Holland *et al.* 1993, Hickey *et al.* 1995).

Once PAHs have become bound to sediment they may be re-suspended via bioturbation. Ciarelli *et al.* (1999) identified that bioturbation by an estuarine amphipod (*Corophium volutator*) affected the concentration of suspended solids in the water column, and consequently increased the aqueous concentration of fluoranthene in the water column, and thus increased the uptake by suspension-feeders (*Mytilus edulis*).

Bivalves can accumulate various chemicals from the surrounding water and sediment, and from their food. Because PAHs bind preferentially to sediment, their toxic effects primarily act on sediment dwelling organisms (Varanasi 1989). Those bivalves which feed on the sediment surface (deposit-feeders) may be more at risk than those species which suspension-feed slightly above this surface. This is supported by a New Zealand study which investigated contrasting contaminant exposure pathways to infaunal adult bivalves. Hickey *et al.* (1995) found that the deposit-feeding bivalve *M. liliana* accumulated general slightly higher concentrations of PAHs than the suspension-feeding *A. stutchburyi*.

In the natural environment the level of phototoxicity experienced by an infaunal bivalve is dependent on the PAH dose (i.e. concentration in the animal) as well as the duration and intensity of the UV exposure (Newsted & Giesey 1987). The level of UV exposure to infaunal bivalves is largely defined by the level of transparency of the bivalve shell. UV exposure is therefore greatest to developing embryos and larvae in the water column, as well as to newly metamorphosed post-settlement bivalves. With increasing bivalve age the shell becomes increasingly calcified and provides a greater barrier to UV exposure. Furthermore, although adult infaunal bivalves may contain high tissue PAH concentrations, their habitat within the sediment protects them significantly from UV light. In contrast, due to their small size, post-settlement infaunal bivalves are restricted to the upper layers of the inter-tidal sediment, and the processes of surface sediment movement may also expose post-settlement bivalves to high levels of UV light. PAH phototoxicity is therefore largely confined to the very early life-cycle stages of the infaunal bivalve species of interest in this study. As all the infaunal bivalve species of interest in this study appear to settle in the inter-tidal zone, they are all likely to be potentially impacted by phototoxic PAHs. However, as *P. subtriangulata* populations generally occur in deeper waters on the more open exposed coast, they are less likely to be exposed to UV radiation. The level of PAHs in sediments in these environments is largely unknown.

Shallow urbanised estuaries may provide the necessary mix of the PAHs and UV radiation required for phototoxic effects to occur in post-settlement infaunal bivalves associated with the sediment surface (Ahrens *et al.* 2002). Most of the phototoxicity work to date has been conducted in the Northern Hemisphere where UV radiation levels are generally lower than in the Southern Hemisphere. Southern Hemisphere bivalve species may be at greater risk of UV-induced phototoxicity unless they have evolved adaptations to cope with these higher UV levels (Ahrens *et al.* 2002).

From laboratory-based toxicity experiments, Ahrens and co-workers (2002) have attempted to estimate the level of exposure of *M. liliana* to PAH phototoxicity in the New Zealand natural environment. They report that historical adult tissue concentrations of total PAHs of 4-40 ng g$^{-1}$ (dry wt) (Manukau Harbour, Hickey *et al.* 1995) are nearly 5000 times lower than the no-observable-effects dose of 150 µg g$^{-1}$ (dry wt) estimated for PAH phototoxicity in juvenile *M. liliana* in their current study. Nevertheless, Ahrens *et al.* (2002) note that because of their high uptake rates, juvenile bivalves may still potentially accumulate a very high body burden in highly polluted PAH sites. Ahrens *et al.* (2002) also stress the greater exposure of juvenile bivalves to UV light due to their tendency to be very near the sediment surface. They further suggest that to understand the potential level of phototoxicity to juvenile infaunal shellfish, the common range of PAH concentrations in tissues of resident juvenile clams should be investigated. The ability to generalise the results of the study by Ahrens *et al.* (2002) on juvenile *M. liliana* to the other infaunal bivalve species of interest here may largely rely on identifying whether juvenile *M. liliana* are deposit-feeders or in fact are initially suspension-feeders. If they are suspension-feeding at this

early stage, then the results of this study may also be broadly applicable to *A. stutchburyi*, *P. australis* and *P. subtriangulata*.

Although infaunal adult bivalves may contain high tissue PAH concentrations, they may live at sediment depths that generally protect them from incident UV light. However, the pelagic larvae of these bivalves may be exposed to sunlight for extended periods (Pelletier *et al.* 2000). Thus, embryos released into the water column with a sufficient maternal PAH body burden may suffer toxic effects following exposure to UV light (Pelletier *et al.* 2000). Pelletier *et al.* (2000) exposed benthic adult bivalves (*Mulinia lateralis*) to water with added fluoranthene (1.25-10 $\mu$g L$^{-1}$), and identified a correlation between increased fluoranthene concentration and adult fluoranthene body burden. Following this exposure, embryos from these adult clams were exposed to UV light. Embryo viability decreased as parent fluoranthene concentration increased (LC$_{50}$ = 6.04 $\mu$g L$^{-1}$). Under fluorescent light there was no significant viability decrease in embryo numbers. Pelletier *et al.* (2000) also reported the same relationship for fluoranthene taken up by adults from the sediment. However, in this study the resulting embryo viability was far more variable. These results suggest that maternal transfer of phototoxic PAHs from infaunal bivalves to their offspring may represent a significant adverse affect to these populations (Pelletier *et al.* 2000). Measurement of adult PAH tissue concentrations may be an acceptable substitute for direct measurement of embryo PAH levels in order to predict the risk to infaunal larvae of phototoxic effects (Pelletier *et al.* 2000). Adult residues above 5,000 $\mu$g/g lipid would be predicted to produce embryos at high risk of photo-enhanced toxicity (Pelletier *et al.* 2000).

The presence of NPAHs is not well documented in the environment (Weigman *et al.* 2001). This is true for the New Zealand marine environment as well as internationally, and makes exposure assessment of these compounds to infaunal selfish currently impossible.

## 5.5.4    Risk Characterisation

Low molecular weight PAHs such as naphthalene, are water-soluble, and do not persist in the marine environment (Williamson & Wilcock 1994). These dissolved PAHs are likely to degrade rapidly through photo-oxidation (U.S. EPA 1980) suggesting that the risk of toxicity to infaunal bivalves from these low molecular weight PAHs will also be low. Conversely, the majority of PAHs are virtually insoluble in water and will bind preferentially to sediment and suspended matter (Williamson & Wilcock 1994). Thus, the greatest fraction of PAH is associated with sediments, benthic organisms and pore waters, with concentrations in these compartments being several orders of magnitude greater than those of overlying waters (Meador *et al.* 1995).

Sediment bound PAHs can be resuspended from bottom sediments into the water column by physical (i.e. water and wind induced) or biological (i.e. bioturbation) processes which may make the PAHs more bioavailable to suspension feeders (Ciarelli *et al.* 1999). Dissolved PAHs may be incorporated into phytoplankton, which in turn can provide an alternate exposure pathway for PAH accumulation in infaunal bivalves (Dobroski and Epifanio 1980).

PAHs are generally not acutely toxic to infaunal bivalves within their water solubility limits. However when bivalves that have bioaccumulated PAHs are exposed to UV light, acute phototoxicity of some PAHs (e.g. anthracene, fluoranthene and pyrene) can occur at much lower PAH concentrations. Bioaccumulated PAHs may cause chronic sublethal impacts such as damage to developing gametes (Lowe and Pipe 1986). PAH phototoxicity is likely to impact embryonic, larval and post-settlement infaunal bivalves due to their position in the water column (increased UV exposure), the transparency of their shells, and their proximity to the sediment surface. N-

heterocyclic PAHs (NPAHs) such as the azaarene family containing one in-ring nitrogen atom have also been shown to have phototoxic effects (Weigman *et al.* 2001).

Bivalves have low rates of PAH metabolism, which can result in high concentrations of the parent compounds (Ankley *et al.* 1995) that can be transferred to their offspring.

There is some evidence to suggest that increased adult body burdens of PAHs can decrease reproductive condition and cause gamete degeneration in bivalves (Lowe & Pipe 1986). No information is currently available to evaluate this impact within the Hauraki Gulf Marine Park.

The concentration of PAHs from stormwater discharge is predicted to increase in estuarine sediments in the urbanized Auckland region in the future (Williamson & Mills 2002). PAH exposure to infaunal bivalves is suggested to be far higher near urbanised land use than in rural less populated areas. The major PAHs found in sediments and bivalves of the New Zealand coastal environment are fluoranthene and pyrene (Holland *et al.* 1993, Hickey *et al.* 1995).

Those bivalves which feed on the sediment surface (deposit-feeders) may be more at risk than those species which suspension-feed slightly above this surface.

At this time, the degree of the effect of phototoxicity on aquatic ecosystems is uncertain due to environmental factors that may enhance or mitigate UV light penetration to the site of toxic action in the organism (Spehar *et al.* 1999). Although many organisms have developed protective mechanisms against UV light, they may not be enough to protect organisms during periods such as low tides (Spehar *et al.* 1999). Furthermore, environmental PAHs are always found as complex mixtures, but most laboratory studies are conducted using a single chemical (Goldstein 2001). Results from these studies may not predict effects when the agent is part of a mixture of PAHs. In general, much more research is required on enhanced phototoxicity in order to determine the risks to infaunal bivalves.

Most sediment quality guidelines have been developed for individual chemicals. Sediment contamination usually involves complex mixtures, especially for related compounds such as PAHs. These mixture effects must be taken into consideration when addressing the level of exposure of bivalve shellfish to PAHs from sediment. Initial research into mixtures of the phototoxic PAHs fluoranthene and pyrene suggests that the interaction provides additive effects (Swartz *et al.* 1997).

In summary *"There is currently insufficient information to assess the likely phototoxicity of Auckland sediments. However, phototoxicity will increase the toxicity of sediments under certain circumstances, although by how much, and under what conditions, are uncertain. If the increase in toxicity is similar to the approximately 10-fold increase observed for fluoranthene in water exposures (US EPA 1993b), then an ISQG-low value of approximately 170 ng/g for high molecular weight PAHs could result. The effect this would have on the potential toxicity of Auckland estuarine sediments is that 70% of the sites could have PAH levels that would be potentially phototoxic"* (Williamson & Mills 2002).

## 5.5.5    References

**Ahrens, M. J., R. Nieuwenhuis & C. W. Hickey  2002.** Sensitivity of juvenile *Macomona liliana* (Bivalvia) to UV-photoactivated fluoranthene toxicity. *Environmental Toxicology*. 17(6): 567-577.
**Ankley, G. T., R. J. Erickson, G. L. Phipps, V. R. Mattson, P. A. Kosian, B. R. Sheedy & J. S. Cox  1995.** Effects of light intensity on the phototoxicity of fluoranthene to a benthic macroinvertebrate. *Environmental Science and Technology*. 29: 2828-2833.

**Baumann, P. C. 1998.** Epizootics of cancer in fish associated with genotoxins in sediment and water. *Mutations Research*. 411: 227-233.

**Bender, M. E., W. J. Hargis Jr., R. J. Huggett & M. H. Roberts Jr. 1988.** Effects of polynuclear aromatic hydrocarbons on fishes and shellfish: An overview of research in Virginia. *Marine Environmental Research*. 24: 237-241.

**Cerniglia, C. E. & M. A. Heitkamp 1991.** Chapter 2: Microbial Degradation of Polycyclic Aromatic Hydrocarbons (PAH) in the Aquatic Environment. *In:* Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment. Varanasi, U. (Ed.). CRC Press, Boca Raton, FL. Pp. 41-68.

**Ciarelli, S., N. M. Van Straalen, V. A. Klap & A P. Van Wezel 1999.** Effects of sediment bioturbation by the estuarine amphipod *Corophium volutator* on fluoranthene resuspension and transfer into the mussel (*Mytilus edulis*). *Environmental Toxicology and Chemistry*. 18(2): 318-328.

**Davis, H. C. & H. Hidu 1969**. Effects of pesticides on embryonic development of clams and oysters and on survival and growth of the larvae. *Fishery Bulletin*. 67(2): 393-404.

**Dobroski, C. J. & C. E. Epifanio 1980.** Accumulation of benzo(*a*)pyrene in a larval bivalve via trophic transfer. *Canadian Journal of Fisheries &Aquatic Science*. 37: 2318 - 2322.

**Donkin, P., J. Widdows, S. V. Evans, C. M. Worrall & M. Carr 1989.** Quantitative structure-activity relationship for the effect of hydrophobic organic chemicals on rate of feeding by mussels (*Mytilus edulis*). *Aquatic Toxicology*. 14(3): 277-294.

**Eisler, R. 1987.** Polycyclic aromatic hydrocarbon hazards to fish, wildlife, and invertebrates: A synoptic review. *U.S. Fish and Wildlife Service Biological Reports*. Report No. 11. 81 pp.

**Gissel-Nielsen, G. & T. Nielsen 1996.** Phototoxicity of acridine an important representative of a group of tar and creosote contaminants, N-PAH compounds. *Polycyclic Aromatic Compounds*. 8: 243-249.

**Granby, K. & N. H. Spliid 1995.** Hydrocarbons and organochlorines in common mussels from the Kattegat and the Belts and their relation to condition indices. *Marine Pollution Bulletin*. 30(1): 74-82.

**Goldstein, L. S. 2001.** To BaP or not to BaP? That is the question. *Environmental Health Perspectives*. 109(8): 356-357.

**Hellou, J. 1996**. Polycyclic aromatic hydrocarbons in mammals, finfish, and molluscs. *In*: Environmental Contaminants in Wildlife. CRC Press. Pp. 229-250.

**Hickey, C. W., D. S. Roper, P. T. Holland & T. M. Trower 1995.** Accumulation of organic contaminants in two sediment-dwelling shellfish with contrasting feeding modes: Deposit- (*Macomona liliana*) and filter-feeding (*Austrovenus stutchburyi*). *Archives of Environmental Contamination and Toxicology*. 29: 221-231.

**Hoffman, E. J., G. L. Mills, J. S. Latimer & J. G. Quinn 1984.** Urban runoff as a source of polycyclic aromatic hydrocarbons to coastal waters. *Environmental Science and Technology*. 18(8): 580-587.

**Holland, P.T., C. W. Hickey, D. S. Roper & T. M. Trower 1993.** Variability of organic contaminants in intertidal sandflat sediments from Manukau Harbour, New Zealand. *Archives of Environmental Chemistry*. 25: 456-463.

**Lowe, D. M. & R. K. Pipe 1986.** Hydrocarbon exposure in mussels: A quantitative study of the responses in the reproductive and nutrient storage cell systems. Aquatic Toxicology. 8(4): 265-272.

**Lyons, B. P., C. K. Pascoe & I. R. B. McFadzen 2002.** Phototoxicity of pyrene and benzo[a]pyrene to embryo-larval stages of the pacific oyster *Crassostrea gigas*. *Marine Environmental Research*. 54: 627-631.

**Makepeace, D. K., D. W. Smith, & S. J. Stanely 1995.** Urban stormwater quality: summary of contaminant data. *Critical Review of Environmental Science and Technology*. 25: 93-139.

**McElroy, A. E., J. W. Farrington & J. M. Teal 1990.** Influence of mode of exposure and the presence of a tubiculous polychaete on the fate of benzo(*a*)anthracene in the benthos. *Environmental Science and Technology*. 24: 1648-1655.

**McIntosh, J. 1999.** Shellfish Quality Assessment. Environmental Report 99/08. Environment Bay of Plenty, April 1999. 25 pp.

**Meador, J. P., J. E. Stein, W. L. Reichert & U. Varanasi 1995.** Bioaccumulation of polycyclic aromatic hydrocarbons by marine organisms. *Review of Environmental Contamination and Toxicology*. 143: 79-165.

**Neff, J. M. 1979.** Polycyclic aromatic hydrocarbons in the aquatic environment: Sources, fates and biological effects. Applied Science, London, U. K.

**Newsted, J. L. & J. P. Giesey 1987.** Predictive models for photoinduced acute toxicity of polycyclic aromatic hydrocarbons to *Daphnia magna*, Strauss (Cladocera, Crustacea). *Environmental Toxicology Chemistry*. 6: 445-461.

**Pelletier, M. C., R. M. Burgess, K. T. Ho, A. Kuhn, R. A. McKinney & S.A. Ryba 1997.** Phototoxicity of individual polycyclic aromatic hydrocarbons and petroleum to marine invertebrate larvae and juveniles. *Environmental Toxicology Chemistry*. 16: 2190-2199.

**Pelletier, M. C., R. M. Burgess, M. G. Cantwell, J. R. Serbst, K. T. Ho & S. A. Ryba 2000.** Importance of maternal transfer of the photoreactive polycyclic aromatic hydrocarbon fluoranthene from benthic adult bivalves to their pelagic larvae. *Environmental Toxicology and Chemistry*. 19(11): 2691-2698.

**Roesijadi, G., D. L. Woodruff & J. W. Anderson 1978.** Bioavailability of naphthalenes from marine sediments artificially contaminated with Prudhoe Bay crude oil. *Environmental Pollution*. 15(3): 223-230.

**Schrimer, K., A. G. J. Chan, B. M. Greenberg, D. G. Dixon & N. C. Bols 1998.** Ability of 16 priority PAHs to be photocytotoxic to a cell line from the rainbow trout gill. *Toxicology*. 127: 143-155.

**Simpson, C. D., W. R. Cullen, T.Y.T. He, M. Ikonomou & K. J. Reimer 2002.** Metabolism of pyrene by two clam species, *Mya arenaria* and *Protothaca staminea*. *Chemosphere*. 49: 315-322.

**Spehar, R. L., S. Poucher, L. T. Brooke, D. J. Hansen, D. Champlin & D. A. Cox 1999.** Comparative toxicity of fluoranthene to freshwater and saltwater species under fluorescent and ultraviolet light. *Archives of Environmental Contamination and Toxicology*. 37: 496-502.

**Swartz, R. C., D.W. Schults, R. J. Ozretich, J. O. Lamberson, F. A. Cole, T. H. DeWitt, M. S. Redmond & S. P. Ferraro 1995.** $\sum$PAH: A model to predict the toxicity of polynuclear aromatic hydrocarbon mixtures in the field-collected sediments. *Environmental Toxicology Chemistry*. 14: 1977-1987.

**Swartz, R. C., S. P. Ferraro, J. O. Lamberson, F. A. Cole, R. O. Ozretich, B. L. Boese, D. W. Schults, M. Behrenfeld & G. T. Ankley 1997.** Photoactivation and toxicity of mixtures of polycyclic aromatic hydrocarbon compounds in marine sediment. *Environmental Toxicology and Chemistry*. 16(10): 2151-2157.

**Timperley, M. & T. Mathieson 2002.** Marine sediment monitoring programme 1998-2001: Review of results and procedures. Auckland Regional Council Technical Publication. No. 193. July 2002. 55pp.

**Tricklebank, K. A. & M. J. Stewart 2001.** Parasites in cockles (*Austrovenus stutchburyi*): Potential indicators of estuarine health. Unpublished Preliminary Report for the Auckland Regional Council, December 2001.

**U.S. Environmental Protection Agency (U.S. EPA), 1980.** Water Quality Criteria Document for Polynuclear Aromatic Hydrocarbons. Prepared by the Office of Research and Development, Environmental Criteria And Assessment Office, Cincinnati, OH for the Office of Water Regulations and Standards, Washington, DC.

**Varanasi, U. 1989.** Metabolism of Polycyclic Aromatic Hydrocarbons in the Aquatic Environment. CRC Press, Boca Raton, Florida.

**Weigman, S., P. L. A. van Vlaardingen, E. A. J. Bleeker, P. de Boogt & M. H. S. Kraak 2001.** Phototoxicity of azaarene isomers to the marine flagellate *Dunaliella tertiolecta*. *Environmental Toxicology and Chemistry*. 20(7): 1544-1550.

**Williamson, R. B. & R. J. Wilcock 1994.** The distribution and fate of contaminants in estuarine sediments: Recommendations for environmental monitoring and assessment. NIWA Consultancy Report No. ARC109/3. Auckland Regional Council Technical Publication No. 47. 67pp.

**Williamson, R. B. & G. N. Mills 2002.** Sediment quality guidelines for the regional discharges project. Unpubl. report prepared by Diffuse Sources Ltd. for the Auckland Regional Council. April 2002. 71pp.

## 5.6 STRESSOR:        SEDIMENT LOADING

### 5.6.1    Hazard Identification

Sediment loading is a generic term that encompasses suspended sediment concentrations (units of mass (or volume) per volume of water), sedimentation rate, and turbidity (Thrush *et al.* 1998). Sedimentation rate is a measure of the net accumulation or erosion of sediment on the seabed per unit time, that is, the difference between the rate at which sediment is settling from suspension and the rate at which sediment is being eroded from the seabed.  Turbidity is an optical property of the water column that is at least partly dependent on suspended sediment concentration (Thrush *et al.* 1998).

Sediment runoff from land use has recently been recognised as a significant threat to marine coastal communities (GESAMP, 1994, quoted in Norkko *et al.* 2002).  Most of the sediment runoff from the land catchment is deposited in harbours and estuaries, since they are natural retention systems. These fine sediments can be resuspended by waves.  It is thus likely that the long-term consequences of increased sediment input are elevation of sedimentation rates, suspended sediment levels and turbidity (Thrush *et al.* 1998).

Sediment input may arise from the continuous process of surface erosion in the land catchment and from episodic events such as landslides, extreme rain events and flooding.  Such episodic events can result in catastrophic deposition of fine terrigenous sediments and elevated turbidity.  This may significantly influence the structure and function of macrobenthic communities (Ellis *et al.* 2000, quoted in Norkko *et al.* 2002).

Catastrophic sedimentation events in estuaries, which may deposit layers of fine terrigenous sediments several centimetres thick in short time periods, modify habitats over broad spatial scales and can affect resource availability and lead to mass mortality of benthic fauna.  Catastrophic sediment deposition affects benthic invertebrates by:

- Physical smothering of the substrate surface, causing suffocation of organisms that cannot climb through the deposited sediment;
- Changing sediment grain size, affecting porosity and sediment stability, and thus biogeochemical fluxes, rates of faunal movement, and bioturbation.  This can also change the suitability of sediment for recruitment of juveniles.
- Changing sediment food quality (Norkko *et al.* 1999; Norkko *et al.* 2002).

The accompanying increase in levels of suspended sediment can:

- Reduce light intensity in the water (through increased turbidity), and thus limit primary production (i.e. the growth of phytoplankton).
- Clog the feeding structures (gills) of suspension feeders.
- Dilute planktonic food quality (especially with respect to sub-surface sediments with a short transition time to the estuary (Norkko *et al.* 1999)).

Bivalves regulate ingestion by producing pseudofaeces above a certain seston concentration (Iglesias 1992) (i.e. if there are too many particles in the water, bivalves will discard them off the gills instead of taking them in through the mouth).  Generally particles in the size range 2-12 microns are utilised by bivalves (Jorgensen, 1990).  It is sediment particles in this size range that are

most relevant with respect to impact on infaunal inter-tidal bivalves, since these particles are accessible to the shellfish and require processing on the gill membranes. As a result of high suspended sediment levels, bivalves must invest energy in removing particles of little nutritive value from their gills, and (for particles ingested with plankton) from their gut (Willows 1992). High turbidity levels arising from increased sediment loading may thus impact negatively on bivalve growth.

There is a link between land use and sediment loading. Sediment core samples taken from tidal creeks show that sedimentation rates change as land use changes (Vant *et al.* 1993): predevelopment sediment yields increase during subdivision and development, then decrease as impervious cover and revegetation protect the land surface from the erosive effects of rainfall.

There is some evidence to suggest that even low human densities in an estuarine catchment can result in changes to estuarine characteristics. A Tasmanian study demonstrated a direct relationship between catchment urbanisation (measured as human population density within the catchment) and the increasing silt/clay content of estuaries and changes in biota (Edgar & Barrett 2000, quoted in Norkko *et al. 2002*).

Patterns of turbidity in estuaries are complex and result from several interacting factors (ARC 1996):
- Freshwater inflow
- Sediment resuspension
- Tidal and wind mixing
- Flocculation of fine particles due to mixing of fresh and salt water
- Presence of sheltered areas which act as silt traps, such as reed beds and mangroves.

Fine sands and cohesive silts and clays can be carried in suspension at a velocity slower than that required to erode them. Consequently they may be carried into an estuary and deposited because of a velocity drop on arrival at the estuary or at slack water, but the requirement of a higher velocity to resuspend them once deposited means they progressively accumulate (Hume 1983, quoted in ARC 1996).

Hydraulic and sediment input factors create distinct erosion, transport and accumulation zones in harbours and estuaries (Williamson & Wilcock, 1994, quoted in ARC 1996). A variety of factors influence the establishment of depositional areas, including (Hume 1983, Williamson & Wilcock 1994, quoted in ARC 1996):
- Chemical processes such as flocculation, causing fine particles to aggregate together, so they become heavier and settle out
- Hydraulic factors such as volumes of freshwater inputs
- The overall shape of the estuary, especially as it affects the slowing of inflowing water velocity on entering the estuary
- The configuration of the inter-tidal and subtidal areas
- Proximity to sources of sediment input
- Shelter from wave action
- Reduced effectiveness of tidal flushing
- Presence of vegetation such as mangroves which tend to slow water velocities, encouraging deposition of sediment nearby.

Most sediment is deposited in the inter-tidal zone (Vant *et al.* 1993). A recent study of estuaries in the Auckland region by Swales *et al.* (2002) found that inter-tidal flats are accumulating sediment on average 0.8 mm/year faster than subtidal flats. This means that in terms of exposure, inter-tidal species are particularly at risk from elevated sedimentation rates in estuaries.

## 5.6.2    Dose-Response Assessment

Norkko *et al.* (2002) note that:

*"Gradients in the intensity of physical and biological process often occur in estuaries. Thus both sources of disturbance and the mechanisms important in driving recovery will vary with location. To improve our understanding of benthic recovery process following disturbance, it is imperative to recognise that different communities and habitats respond differently (e.g. Thrush et al. 1991)".*

Research associated with the potential impacts of increased sediment loading in the marine environment has predominantly focused on three aspects: the impact of sediment deposition, tolerance of changes in sediment size, and the impact of elevated suspended sediment levels. A review of the current knowledge in these areas is presented in the discussion below. However, we note that these aspects of increased sediment loading are interlinked. In the natural environment, the potential stressors associated with increased sediment loading may be applied simultaneously – for example, deposition of terrigenous clay can be accompanied by increased suspended sediment. Thus although the impacts may be considered separately, in the natural environment such impacts may be cumulative.

### *Sediment Deposition*

The maximum depth at which infaunal bivalves live in the sediment is determined by the length of their siphon, through which water containing oxygen and food is drawn. Cockles (*A. stutchburyi*) and pipi (*P. australis*) have short siphons and normally live near the sediment surface, whereas wedge shells (*M. liliana*), which have longer siphons, are commonly founding living deeper than 5cm beneath the sediment surface (Norkko *et al.* 1999). Bivalves subject to additional deposits of sediment may either reposition themselves towards the sediment surface, or if this is not possible, switch to anaerobic metabolic pathways until conditions improve.

Several studies relevant to the risk of burial of New Zealand infaunal bivalve species by catastrophic sedimentation events have been undertaken:

Manipulative laboratory experiments were conducted by Creese (1988) to investigate a variety of factors related to *P. australis* reorientation to the surface following burial. These included
- the impact of different depths of sand applied to the surface
- *P. australis* orientation upon burial
- the effects of continued *P. australis* reburial over a period of 5 days.

Creese (1988) found that *P. australis* could withstand disturbance and inundation to depths of at least 10 cm. Medium sized *P. australis* (30-45 mm shell length) could even cope quite well with burial less than 40 cm of sand or with continued reburial fewer than 15cm of sand a day (for 4 days). An exception to this was when *P. australis* were placed upside down into the surface of the sediment, as they were not able to use their foot to right themselves. (Note that these experiments were undertaken using sand from the same habitat or of a similar grain size to the substrate in which the *P. australis* were found. These experiments therefore provide some indication of the ability of

*P. australis* to withstand burial *per se*, without the additional potential impacts of the smaller sediment grain size that might be associated with catastrophic deposition events arising from floods, e.g. deposition of terrigenous clay).

A series of laboratory experiments were conducted by Norkko *et al.* (1999) to investigate the ability of a variety of organisms from the Okura estuary to surface through different thicknesses of mud and/or survive short-term burial. *Macomona liliana, A. stutchburyi* and *P. australis* were included in the study. The experiments demonstrated that the bivalves were not capable of climbing up through the layers of clay at any of the depths tested (3, 6, and 9 cm layers), although *P. australis* were able to extend their siphons up through the 3-cm layer allowing them to survive. There was a general trend of increasing stress levels (measured by their ability to re-bury themselves in sandy Okura sediments) with increasing depth and duration of smothering (measured after 1, 3, and 6 days). All the species showed signs of sub-lethal stress after only one day of smothering. However, the responses at all depths of clay were species-specific, with mortality levels consistently highest in *M. liliana* and lowest in *P. australis*.

Interestingly, an overseas study by Peterson (1985, quoted in Ellis *et al.* 2002) investigating the macrofaunal changes following an intense rainstorm that caused catastrophic sedimentation showed different patterns of species-specific survival. In this study, deposition of silt and clays substantially increased mortality of suspension feeding bivalves, with higher mortality among infaunal siphonate suspension-feeders than among infaunal siphonate deposit-feeders.

A field experiment by Norkko *et al.* (1999) at two sites in Okura estuary confirmed the findings of their laboratory experiments – within a week after deposition of clay (in layers 3, 6 and 9 cm deep) nearly all of the animals in the underlying sediment had died, including all the bivalves.

A further laboratory-based study of short-term behavioural responses of benthic invertebrates to burial by terrestrial clay was undertaken by Norkko *et al.* (2001). The response of *A. stutchburyi* (mean size 21.5 mm) to the application of layers of clay (0.5, 1 and 1.5 cm in depth) was recorded over 6 days. No mortality was recorded. The researchers' observations of the movement of *A. stutchburyi* into the clay layers suggested that the animals attempted to maintain their position with respect to the sediment-water interface. In the 0.5 cm and 1 cm treatments, an increasing number of *A. stutchburyi* moved out of the sand into the clay layer over the course of the 6-day study. However, in the 1.5 cm treatment, the number of *A. stutchburyi* found in the clay increased from day 1 to day 3, but by day 6 all animals had moved back into the sandy sediment (Norkko *et al.* 2001). It would have been interesting to observe the impact of these differences over a longer time period.

Laboratory experiments by Dobbie (1996) showed that *A. stutchburyi* were able to re-surface through sediment burial depths of up to 10 cm.

Inter-specific differences in response to burial may have a basis in differences in shellfish physiology. For example, a study by Carroll & Wells (1995) found differences between the way in which slowly burrowing bivalves, such as *P. australis* and *A. stutchburyi* support anaerobic energy production compared to rapidly digging surf clams such as *P. subtriangulata*. Their study showed that rapidly digging shellfish perform better than slowly burrowing shellfish under functional anaerobiosis (such as caused by rapid digging activity), but do not fare as well at low oxygen levels.

Critical depth is size/age dependent, and is related to the normal burial depth and the length of siphon of the species (Norkko *et al.* 1999). This suggests that there may be differences in risk related to shellfish size.

Berkenbusch *et al.* (2001) conducted a study in the Whitford estuary to assess the impact of the deposition of thin layers of terrigenous clay on estuarine communities and ecosystem processes over ten days. Results showed that thin clay deposits ranging from 3-7 mm in depth have an impact on benthic communities, with negative effects increasing with the depth of deposited clay. The study was undertaken at two sites with different community composition. *Austrovenus stutchburyi* were the dominant species at one site. The abundance of small-sized *M. liliana* (<5mm) declined significantly at both sites with increasing clay depth, as did the abundance of small *A. stutchburyi* (<5 mm) at the "Cockle" site (Berkenbusch *et al.* 2001). This may suggest that small individuals of these species are more likely to be affected by deposition of terrigenous clay than larger individuals. It also suggests that the deposition of thin layers of terrigenous clay may impact on inter-tidal infaunal bivalve populations.

A study by Gibbs *et al.* (2001) suggests that the impacts of thin layers of terrigenous deposition on infaunal bivalves may differ with mode of feeding.

***Change in sediment particle size***

A review by the Auckland Regional Council (1996) notes "*Estuarine animals may be able to recolonise new deposits without significant changes in the species assemblage present, provided the overlying deposits have similar characteristics to those smothered and unaffected adjacent areas can provide a source of colonising animals*" (ARC 1996).

In a study of the Upper Waitemata Harbour, Cummings *et al.* (2002) found that the bivalves *P. australis* and *M. liliana* were not present at sites containing higher (i.e. > 67 %) levels of mud. *Austrovenus stutchburyi* was found in a wide range of sediment types but was more abundant at less muddy sites. A study by Norkko *et al.* (2001) investigated the sensitivity of macrofaunal species to fine sediments in the Whitford embayment. They identified the optimum range (i.e. the percentage silt/clay where species exhibit their highest abundances) and distribution range (i.e. the total range of occurrence over different silt/clay concentrations) of a range of species, including *P. australis*, *M. liliana* and *A. stutchburyi*. The results indicated that *P. australis* are highly sensitive to increased silt/clay content (optimum range and distribution range both 0-5% silt/clay content). Both *M. liliana* and *A. stutchburyi* were classed as sensitive. The optimum range of *M. liliana* was found to be 0-5%, and their distribution range 0-40%. The optimum range of *A. stutchburyi* was 5-10% with the distribution range 0-60% (Norkko *et al.* 2001). Cummings *et al.* (2002) concluded that all three species could be considered to be sensitive to increases in sediment and mud content.

***Impact of suspended sediment***

Although estuarine fauna and flora are adapted to higher average turbidity than their freshwater or open coastal counterparts, they are nevertheless susceptible to damage from accelerated sediment deposition resulting from human activities (ARC 1996).

High suspended sediment loads decrease feeding efficiency resulting in decreased food uptake and increased energy expended in the production of pseudofaeces. A loss of utilisable energy sources such as protein and sugars results in decreased energy available for reproduction and growth and at certain levels will ultimately result in death (Ellis *et al.* 2002).

In a study at 10 sites in the Mahurangi Harbour by Thrush *et al.* (1998), approximately 95% of the particles collected in sediment traps were within the appropriate size range to be accessible to suspension-feeding bivalves (i.e. 2-12 microns). This is typical of turbid estuaries, where 70-80% of the particles are normally in this size range (Widdows *et al.* 1998).

A study by Hewitt *et al.* (2001) investigated the effects of suspended sediment on *A. stutchburyi*, *P. australis* and *M. liliana* in the Whitford embayment through a series of laboratory and field experiments. Although adult *A. stutchburyi* and *P. australis* both exhibited the ability to continue feeding in high levels of suspended sediment over the short term (up to one week), the physiological condition of adult *P. australis* and *A. stutchburyi* has been shown to be adversely affected by prolonged elevations in suspended sediment concentrations (Hewitt *et al.* 2001). Results from the same study suggest that suspended sediment concentrations >400 mg/l for 14 days will decrease adult *A. stutchburyi* condition. The degree to which condition decreases depends on the length of exposure to high concentrations of suspended sediment. The effect on the condition of *P. australis* was more detrimental, with decreases in condition in all sediment treatments when concentrations >80 mg/l persisted for 14 days. Wood (1962) noted that *P. australis* siphons lack the filtering circlet of tentacles that are present in *A. stutchburyi*, and suggested that this may make *P. australis* more susceptible to the impact of fine suspended sediment.

The same study by Hewitt *et al.* (2001) identified that growth rates of juvenile *A. stutchburyi* and *M. liliana* can be adversely affected by high suspended sediment concentrations. Juvenile *M. liliana* were more sensitive to high suspended sediment concentrations than were *A. stutchburyi*, and unlike *A. stutchburyi*, showed negative impacts to high suspended sediment concentrations occurring as few as five times a month (Hewitt *et al.* 2001). In addition, adverse effects of elevated suspended sediment concentrations on reproductive state were detected for adult *P. australis* and large adult *A. stutchburyi*, but not for small adult *A. stutchburyi* (Hewitt *et al.* 2001). Teaioro (1999) has shown that energy acquisition by *P. australis* was reduced during short-term (8 hr) exposure to elevated suspended silt/clay concentrations.

A study in the Mahurangi estuary, combined with laboratory experiments, identified that the physical and biochemical condition of the horse mussel (*Atrina zelandica*) is correlated with suspended sediment concentration (as indicated by settling flux) around the estuary (Thrush *et al.* 1998; Ellis *et al.* 1999; Ellis *et al.* 2002). The study determined critical levels of suspended sediments that result in loss of biomass and levels that are so high as to limit the distribution of *Atrina* in field conditions (Ellis *et al.* 2002). There has been a significant decline in abundance of *Atrina* at both subtidal sites in the Mahurangi estuary in the period from October 1994-January 2003. Based on previous studies of the impact of suspended sediment on *Atrina* by the same research group (Ellis *et al.* 1999), Cummings *et al.* (2003) concluded that this decline is consistent with changes that might be expected from increased suspended sediment concentrations. It is possible that increased levels of suspended sediment could also be implicated in the decline in abundance of *A. stutchburyi* and *M. liliana* in the same estuary (Cummings *et al.* 2003). However at this stage the relationship is only tentatively inferred and has not been specifically investigated.

The impact of elevated levels of suspended sediment on the larval stages of the infaunal bivalve species of interest does not appear to have been investigated. Consideration of the impacts on the veliger stage, which normally feed on phytoplankton, could be relevant to the potential impacts on the populations as a whole.

### 5.6.3    Exposure Assessment

Changes in land use and modifications of coastlines due to human development have both increased rates of sedimentation and changed the areal extent of depositional environments in estuaries (Edgar & Barrett 2000).

Core samples taken from estuarine sediment in the Waitemata Harbour provide evidence of increasing sedimentation rates over a long time scale.  Average pre-human sedimentation rates in the Waitemata Harbour were 0.03-0.1 mm/year.  Rates increased to an estimated rate of 1 mm/year after Maori settlement of the area as a result of small-scale land clearance and vegetation burn-off (Hume & McGlone 1986).  With European settlement, land clearance associated with logging and agriculture increased sedimentation rates to 2-3 mm/year.  Since the 1950's, intensive agriculture and urban development have resulted in rates of 6-9 mm/year (Auckland Regional Council 1996). Swales *et al.* (2002) note that of particular concern is the fact that sedimentation rates in all estuaries in their study (which included Mahurangi, Puhoi, Okura, Henderson Creek, Waitemata Harbour, Te Atatu, Whitford, Wairoa and Te Matuku), including remote estuaries with no apparent risk factors (such as Te Matuku at Waiheke Island) have increased in the last 20 years (1980-2001).

As previously mentioned, most sediment is deposited in the inter-tidal zone (Vant *et al.* 1993).  This means that inter-tidal species such as *A. stutchburyi*, *M. liliana* and *P. australis* are more likely to be exposed to the risks associated with increased sedimentation and changes in sediment particle size *P. subtriangulata*, which is predominantly found subtidally.

Areas most at risk from increased sediment load are those relatively enclosed areas that are
*   undergoing modification in land use that increases run-off (such as removal of vegetation, and particularly urban sub-division),
*   have a small size relative to catchment area, and
*   have poor flushing (Swales, 1989).

Typically, these tend to be the upper reaches of estuarine and harbour areas.

The impact of catchment sediment runoff varies with the receiving environment.  Hume *et al.* (2002) note that different types of estuaries are affected by catchment sediment runoff in different ways because the relative influence of different physical and biological processes varies between estuary types.  Their conceptual models, describing how different types of estuaries (tidal creek, drowned valley, tidal lagoons, tidal rivers and coastal embayments) deal with catchment sediment loads, are presented in Hume *et al.* (2002).  Swales *et al.* (2002) provide a discussion of the way in which different estuary types in the Auckland area are impacted by catchment sediment run-off.  In their recent study of sediments in Auckland estuaries, Swales *et al.* (2002) found that there is little difference in sedimentation rates between different estuary types.  They also noted that muddy estuaries do not necessarily have the highest sedimentation rates (Swales *et al.* 2002).

It could be assumed that urbanisation of any catchment within the Hauraki Gulf Marine Park might present the risk of increased sediment loading in the marine environment to some degree, but this has been studied in depth in few areas.  Cummings *et al.* (2003) note that sediment loading has probably influenced estuaries and benthic communities in Whitford, Okura and Upper Waitemata.  A recent study comparing current benthic communities to decadal rates of sedimentation indicates that Puhoi and Wairoa estuaries have been subject to high rates of sediment accumulation (Lundquist *et al.* submitted, quoted in Cummings *et al.* 2003).  Cummings *et al.* (2003) also note

that they have observed high levels of turbidity and plumes of sediment-laden water exiting the Matakana estuary after severe storms.

The Mahurangi estuary has also been identified as an estuary at risk of increased sediment loading. Various sediment characteristics in the Mahurangi estuary were monitored at five sites as part of an ecological monitoring programme that began in July 1994 (Cummings *et al.* 2003). From July 1994-January 2003 there was an increase in amount of fine sand at all sites and a corresponding decrease in the amount of medium sand at inter-tidal sites only. These changes occurred sometime between April 1997 and April 1998, and have persisted. However, there was no obvious response in *A. stutchburyi* or *M. liliana* abundance (or any of the other species noted) to this apparent "pulse" event. These species however, exhibited gradual declining trends (Cummings *et al.* 2003).

While estuarine areas are those most likely to suffer the physical impacts of increased sediment input, there may be impacts in other environments also. Sediment trap deposition rates measured by Walker & Babcock (2001) on Long Bay, Campbells Bay, Torbay, Little Manly, Stanmore Bay and Waiwera showed significant differences between areas. Campbells Bay showed the highest total sedimentation, and this was manly composed of coarse resuspended marine sediments (Walker & Babcock 2001). At all sites fine sediments were the minority of sediments present in traps. However, there is concern that the marine environment of Long Bay may be subject to increased sedimentation from land run-off. During heavy rainfall events terrestrial runoff enters the Long Bay marine environment through both the Awaruku and Vaughan Streams as well as through the Okura estuary (Walker & Babcock 2001). Modelling of sediment input from Awaruku and Vaughan Streams, Okura and Weiti Estuaries by Green *et al.* (2000) suggested that sediment deposition would occur within the entire Long Bay area from both streams and the Okura estuary via wind generated long shore currents.

Swales *et al.* (2002) suggest that highly infilled estuaries in the latter stages of aging (such as the Maungamaungaroa and Wairoa estuaries) may pass a considerable portion of their sediment load to the neighbouring coastal environment. Evidence for this was found in the surprisingly high sediment accumulation rates (i.e. 2-8 mm/year) in Whitford and Wairoa subtidal environments, and in sediment cores from Te Matuku (Waiheke Island). They noted that Te Matuku has a small catchment, much of which is covered with native bush. The researchers suggested that the most likely source for the sediment depositied at Te Matuku is mainland estuaries such as the Wairoa (Swales *et al.* 2002).

The risk of catastrophic depositional events is likely to be different to the risk of low load depositional events. Berkenbusch *et al.* (2001) note that it is expected that low load depositional events that produce thin layers of sediment are likely to be more frequent and cover a larger area than larger catastrophic events. They suggest that the frequency of such events would be higher in upper estuary and channel margin habitats. Catchment modelling of a sub-environment in the Okura estuary based on current land use practices predicts that 21 depositional events that could deposit 2-3 mm of clay on an upper estuary sandflat would occur over the 35 year period modelled, while there would be insufficient sediment load to initiate depositional events resulting in a 2 cm thick layer of sediment (Berkenbusch *et al.* 2001, Stroud *et al.* 1999).

## 5.6.4 Risk Characterisation

The potential effects of catchment runoff on marine ecosystems have been studied in a variety of different ways. These include studies of the stratigraphy of subsurface sediments, general field observations (which Hume *et al.* (2002) suggest are generally not specific enough or too short-term

to infer a causal relationship), long term monitoring programmes (such as the Mahurangi Estuary monitoring programme (Cummings *et al.* 2003)), and field and laboratory experiments. (Hume *et al.* (2002) provide a relatively recent review of some of these studies.) Some of these studies have included investigation of the impacts of increased sediment loading on *A. stutchburyi*, *P. australis* and *M. liliana*. Little attention has been paid to the possible impacts on *P. subtriangulata*, possibly because most of the obvious impacts of increased sediment loading occur in relatively sheltered estuarine areas, not habitats where *P. subtriangulata* are generally found. Neither have the potential impacts (particularly of increased levels of suspended sediment) on the larval stages of bivalves been considered.

There is a significant body of data that suggests that urbanisation of catchment areas increases the sediment loading in the marine environment, particularly, but not exclusively, in enclosed estuarine areas. It could thus be assumed that the increasing urbanisation of coastal areas within the Hauraki Gulf Marine Park is resulting in increasing sediment loading into the marine environment. This may also result in decreases in sediment particle size in some areas as a result of input of terrigenous sediments. For example, stepwise trend in decreasing sediment particle size has been observed over eight years of monitoring in the Mahurangi estuary (Cummings *et al.* 2003). The key question is whether or not this increase in sediment loading impacts significantly on infaunal bivalve populations in the region.

Data from field and laboratory experiments suggest that catastrophic deposition events could result in the mortality of infaunal bivalve shellfish. Modelling of a sub-environment in the Okura estuary based on current land use practices suggested that low load deposition events are likely to occur more frequently than catastrophic events. While experiments suggest that there is a lower risk of mortality as a result of burial following low load depositional events, data also suggest that elevated levels of suspended sediment may result in sub-lethal impacts on *P. australis*, *A. stutchburyi* and *M. liliana* (there are no experimental data available for *P. subtriangulata*). These include negative impacts on growth rate and in some cases (shown experimentally in adult *P. australis* and large adult *A. stutchburyi*, but not small adult *A. stutchburyi*), reproductive condition. Patterns of rainfall, along with catchment characteristics etc. on the Coromandel Peninsula may be significantly different from those at Okura. We cannot therefore generalise the situation at Okura to other areas in the Hauraki Gulf Marine Park.

Currently the effect of these potential impacts on infaunal bivalve populations in the Hauraki Gulf Marine Park remains unquantified. However, with a trend of increasing urbanisation of coastal areas within the region, it could be assumed that the risks associated with elevated sediment loading will increase in the future.

## 5.6.5    References

**Auckland Regional Council  1996.**  The environmental impacts of accelerated erosion and sedimentation.  ARC Environment Division Technical Publication No. 69, March 1996.

**Berkenbusch, K., S. Thrush, J. Hewitt, M. Ahrens, M. Gibbs & V. Cummings  2001.**  Impact of thin deposits of terrigenous clay on benthic communities.  Report prepared for the Auckland Regional Council.  NIWA Client Report ARC01269

**Carroll, J. L. & R. M. G. Wells  1995.**  Strategies of anaerobiosis in New Zealand infaunal bivalves: Adaptations to environmental and functional hypoxia.  *New Zealand Journal of Marine & Freshwater Research.* 29: 137-146.

**Creese, R. G.  1988**.  Aspects of the ecology of pipi at Whitianga.  Report prepared for Thames-Coromandel District Council.  56pp.

**Cummings, V., S. Hatton & P. Nicholls  2002.**  Upper Waitemata Harbour (UWH) Benthic Habitat Survey.  Unpubl. report for the Auckland Regional Council.  NIWA Client Report ARC02285.  August 2002.  49pp.

Cummings, V., P. Nicholls & S. Thrush 2003. Mahurangi Estuary ecological monitoring programme – report on data collected from July 1994 to January 2003. Prepared for Auckland Regional Council. NIWA Client Report HAM2003-066. 77pp.

Dobbie, N. J. 1996. The benthic fauna of Pine Harbour Marina. Unpubl. MSc Thesis. University of Waikato, New Zealand.

Edgar, G. J. & N. S. Barrett 2000. Effects of catchment activities on macrofaunal assemblages in Tasmanian estuaries. *Estuarine, Coastal & Shelf Science.* 50: 639-654.

Ellis, J., V. Cummings, J. Hewitt, S. Thrush & A. Norkko 2002. Determining effects of suspended sediment on condition of a suspension feeding bivalve (*Atrina zelandica*): Results of a survey, a laboratory experiment and a field transplant experiment. *Journal of Experimental Marine Biology and Ecology.* 267: 147-174.

Ellis, J. I., A. Norkko & S. F. Thrush 2000. Broad-scale disturbance of intertidal and shallow sublittoral soft-sediment habitats: Effects on the benthic macrofauna. *Journal of Aquatic Ecosystems Stress Recovery.* 7: 57-74.

Ellis, J. I., S. F. Thrush, G. Funnell, J. E. Hewitt, A. M. Norkko, D. Schultz, J. T. Norkko 1999. Developing techniques to link changes in the condition of horse mussels (*Atrina zelandica*) to sediment loading. Report for the Auckland Regional Council. NIWA Client Report ARC 90230. 19pp.

GESAMP (Group of Experts on the Scientific Aspects of Marine Environmental Protection) 1994. Anthropogenic influences on sediment discharge to the coastal zone and environmental consequences. *Rep. Stud. GESAMP* 52. UNESCO-TOC, Paris.

Gibbs, M., S. Thrush, J. Sukias, K. Berkenbusch, D. Thompson & S. Bury 2001. Evaluation of terrestrial sediment spatial patterns throughout the Whitford embayment and its benthic food web using stable isotope techniques. Report prepared for the Auckland Regional Council. NIWA Client Report ARC01270.

Green, M. O. S., M. J. Oldman, J. W. Senior & A. K. Cooper 2000. Long Bay Sedimentation Study. Unpubl. report for North Shore City Council. NIWA Client Report NSC00209. 57pp.

Hewitt, J., S. Hatton, K. Safi & R. Craggs 2001. Effects of suspended sediment levels on suspension feeding shellfish in the Whitford embayment. Prepared for the Auckland Regional Council. NIWA Client Report No. ARC01267. 32pp.

Hume, T. M. 1983. Upper Waitemata Harbour sediments and the inferred impacts of future catchment and estuary use change. Report for the Auckland Regional Authority, Auckland, New Zealand.

Hume, T., K. Bryan, K. Berkenbusch & A. Swales 2002. Evidence for the effects of catchment sediment runoff preserved in estuaries and sediments. Report prepared for the Auckland Regional Council. NIWA Client Report ARC01272. June 2001. Revised February 2002.

Hume, T. M. & M. S. McGlone 1986. Sedimentation patterns and catchment use change recorded in the sediments of a shallow tidal creek, Lucas Creek, Upper Waitemata Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research.* 20(4): 677-687.

Iglesias, J. I. P., E. Navarro, J. P. Alvarez & I. Armentia 1992. Feeding particle size selection in cockles *Cerastoderma edule* (Lamarck) exposed to variable conditions of food concentration and quality. *Journal of Experimental Marine Biology and Ecology.* 162: 177-198.

Jorgensen, C. B. 1990. Bivalve filter feeding: hydrodynamics, bioenergetics, physiology and ecology. Olsen & Olsen, Fredensborg, Denmark. 140 pp.

Lundquist, D, S. F. Thrush, J. W. Oldman & A. K. Senior (submitted). Limited transport and recolonization potential in shallow tidal estuaries.

Norkko, A. J., S. Talman, J. Ellis, P. Nicholls & S. Thrush 2001. Macrofaunal sensitivity to fine sediments in the Whitford embayment. Report prepared for the Auckland Regional Council. NIWA Client Report ARC01266/2.

Norkko, A., S. F. Thrush, J. E. Hewitt, V. J. Cummings, J. Norkko, J. I. Ellis, G. A. Funnell, D. Schultz & I. MacDonald 2002. Smothering of estuarine sandflats by terrigenous clay: The role of wind-wave disturbance and bioturbation in site-dependent macrofaunal recovery. *Marine Ecology Progress Series.* 234: 23-41.

Norkko, A., S. F. Thrush, J. E. Hewitt, J. T. Norkko, V. J. Cummings, J. I. Ellis, G. A. Funnell & D. Schultz 1999. Ecological effects of sediment deposition in Okura Estuary. Report prepared for the Auckland Regional Council. NIWA Client Report ARC90243.

Peterson, C. H. 1985. Patterns of lagoonal bivalve mortality after heavy sedimentation and their paleocological significance. *Paleobiology.* 11(2): 139-153.

Stroud, M. J., A. B. Cooper, A. B. Bottcher, J. G. Hiscock & N. B. Pickering 1999. Sediment runoff from the catchment of Okura estuary. Report prepared for the Auckland Regional Council. NIWA Client Report ARC90241/1. 37pp.

Swales, A. 1989. The effects of urbanisation and consequent sediment generation on the Upper Pakuranga Estuary, Auckland. Unpubl. MSc. thesis, University of Auckland, New Zealand.

**Swales, A., T. M. Hume, M. S. McGlone, R. Pilvio, R. Ovenden, N. Zviguina, S. Hatton, P. Nicholls, R. Budd, J. Hewitt, S. Pickmere & K. Costley  2002.**  Evidence for the physical effects of catchment sediment runoff preserved in estuarine sediments: Phase 2 (field study).  Report prepared for the Auckland Regional Council.  NIWA Client Report ARC01272.  146pp.

**Teaioro, I.  1999.**  The effects of turbidity on suspension feeding bivalves.  Unpubl. M.Sc. thesis.  Department of Biological Sciences.  University of Waikato, Hamilton, New Zealand.  109pp.

**Thrush, S., V. Cummings, A. Norkko, J. Hewitt, R. Budd, A. Swales & G. Funnell  1998.**  Response of horse mussels (*Atrina zelandica*) to sediment loading.  Report prepared for the Auckland Regional Council.  NIWA Client Report ARC 80230.  22pp.

**Thrush, S., R. D. Pridmore, J. E. Hewitt & V. J. Cummings  1991.**  Impact of ray feeding disturbances on sandflat macrobenthos: Do communities dominated by polychaetes or shellfish respond differently?  *Marine Ecology Progress Series*.  69: 245-252.

**Vant, W. N., R. B. Williamson, T. M. Hume & T. J. Dolphin.  1993**.  Effects of future urbanisation in the catchment of Upper Waitemata Harbour.  Report to Auckland Regional Council.  NIWA Consultancy Report No. ARC220R.

**Walker, J. & R. Babcock  2001.**  The Long Bay monitoring programme Sampling Report July 2000/June 2001.  Unpubl. report for Auckland Regional Council prepared by Auckland Uniservices Ltd.  June 2001.  77pp.

**Widdows, J.**, **M. D. Brinsley, N. Bowley & C. Barrett 1998.**  A benthic annular plume for *in situ* measurement of suspension feeding/biodeposition rates and erosion potential of intertidal cohesive sediments.  *Estuarine, Coastal & Shelf Science*  46: 27-38.

**Williamson, R. B. & R. J. Wilcock  1994.**  The distribution and fate of contaminants in estuarine sediments: Recommendations for environmental monitoring and assessment.  NIWA Client Report No. ARC109/3.  Auckland Regional Council Technical Publication No. 47.

**Willows, R. I.  1992.**  Optimal digestive investment: A model for filter feeders experiencing variable diets.  *Limnology and Oceanography*.  37: 829-847.

**Wood, D. H.  1962.**  An ecological study of a sandy beach (Howick).  Unpubl. MSc thesis, University of Auckland, New Zealand.

## 5.7 STRESSOR:         NUTRIENT ENRICHMENT

## 5.7.1    Hazard Identification

Eutrophication is the enrichment of water by nutrients (normally nitrates and phosphates). Increased anthropogenic nutrient addition to coastal waters due to urbanization (e.g. wastewater), agriculture, deforestation, and atmospheric deposition is one of the leading problems altering the environmental quality of coastal waters today (GESAMP 1990, Valiela *et al.* 1997 a,b).  In some areas of Europe (e.g. the Baltic Sea) the total nitrogen load has increased by four times and the total phosphorous load by eight times since the beginning of the 20th century (Larsson *et al.* 1985). Coastal areas have higher natural nutrient values than the open sea, due to inflows from land, more shallow and complex topography and higher water temperatures (Bonsdorff *et al.* 2002).

Sources of nutrient addition to the marine environment can be broadly separated into two categories: readily identifiable point sources (such as sewage effluents) and diffuse sources (such as run-off from agricultural land).  Sewage effluent commonly has high concentrations of nitrogen in the form of ammonia, which is oxidised by microorganisms to nitrate (Wilcock & Stroud 2000). Development of coastal areas may increase nitrogen (N) loading to coastal waters (Weiss *et al.* 2002).

Since nitrogen is the major limiting nutrient in coastal waters, increases in phytoplankton abundance, particulate organic matter, and nitrogen content of seston (i.e. the total animate and inanimate particulate matter in natural waters) are likely to increase as land-derived nitrogen load increases (Ryther & Dunstan 1971, Valiela, 1995).  Estuarine ecosystems may show phosphorus-limitation at their freshwater extreme, grading through to nitrogen-limitation at their seaward end (UK Eutrophication Forum 2002).  However, the common occurrence of suspended sediments within estuaries means that light generally limits algal growth (as has been suggested for the Manukau Harbour by Vant & Safi 1996).  In the UK, the contributions of phosphorus entering surface waters have been estimated as follows: 43% agricultural, 43% human and household waste including detergents, 8% industry and 6% from background sources (Morse *et al.* 1993).  The actual nutrient contributions in any given catchment will depend on the nature of the catchment and the human activities that occur within it.

Increased nutrient inputs may change the quantity and quality of phytoplankton available as food for bivalves.  This occurs because nutrients such as nitrogen and phosphorus are required by phytoplankton, so their increased presence in the water acts as a "fertiliser", promoting phytoplankton growth.  Alteration in food supply through nutrient enrichment may impact on bivalve growth rate (Weiss *et al.* 2002).

The effect of eutrophication at an ecosystem level is dependent on factors such as the abundance and distribution of the zooplankton grazer population, and on the presence of toxicants that might affect these populations.  In some cases, nutrient enrichment of a given ecosystem will have no apparent effect, because phytoplankton production is limited by factors other than nutrient supply. However, in certain circumstances, nutrient enrichment can lead to negative effects, including ecosystem modifications (e.g. changes in species composition due to variable physiological ability to withstand new environmental conditions), algal blooms, and in extreme cases (through decomposition of plant biomass), oxygen depletion.  Increased oxygen consumption in bottom waters occurs when excess organic matter (produced from eutrophic conditions) sinks and is decomposed at or near the bottom (Bonsdorff *et al.* 2002).  This may cause anoxic conditions with

the formation of hydrogen sulphide and nutrient flux from the sediment as a consequence (Larsson *et al.* 1985). Hypoxic and anoxic conditions in bottom waters can destroy whole benthic communities (Bonsdorff *et al.* 2002). Thus eutrophication can have both temporary and long-term effects on aquatic ecosystems. While increased phytoplankton abundance can increase bivalve growth rates, anoxic conditions associated with eutrophied waters may also serve to depress bivalve growth rates (Weiss *et al.* 2002). Indirect effects of eutrophication may also impact on infaunal bivalve populations. For example, Aarnio & Mattila (2000) identified that drifting algal mats caused by intense eutrophication in Baltic coastal waters decreased the predation efficiency of juvenile flounder (*Platichthys flesus*) on the infaunal bivalve *Macoma balthica*. Understanding these changes may be important for monitoring declining bivalve populations (Shriver *et al.* 2002). The adverse ecological effects of eutrophication on saline waters have been evaluated by the UK Eutrophication Forum (2002) as follows:

*Effects associated with increased growth and changing species composition of phytoplankton and bacteria:*
- Changing species composition of marine phytoplankton community, often with a shift away from diatoms to less desirable species;
- Changing plant community structure in estuaries, moving from vascular plants, through macroalgae with eventual dominance of bloom-forming phytoplankton;
- Decay of algal blooms in marine waters causing reduced dissolved oxygen levels, with potential production of methane and hydrogen sulphide, adversely affecting marine zooplankton, benthic fauna (e.g. bivalves) and fish;
- Nuisance algal blooms will become more pronounced;
- Increased occurrence of toxic algal blooms.

*Effects associated with increased growth and changing species composition of microphytobenthos (small-bottom dwelling algae) and macroalgae:*
- Proliferation and associated de-oxygenation of underlying muds in inter-tidal areas, affecting benthic faunal communities,
- Changing species composition (i.e. loss of sensitive slow-growing species).

## 5.7.2    Dose-Response Assessment

Weiss *et al.* (2002) found that chlorophyll *a* (Chl *a* - a proxy for phytoplankton) was higher in estuaries with a higher nitrogen load, and that shellfish (*Mercenaria mercenaria* and *Mya arenaria*) grew fastest in the estuaries where Chl *a* was highest.

Hydrogen sulphide in sediment was found to negatively affect recruitment of *Macoma balthica* (Bonsdorff & Wenne 1989). We have not identified any dose-response data with respect to the impact of hydrogen sulphide on *A. stutchburyi*, *P. australis*, *P. subtriangulata* or *M. liliana* in New Zealand.

Shiver *et al.* (2002) identified that the nitrogen in juvenile bay scallops (*Agropecten irradians*) and in suspended particulate matter (SPM) was specific to the watershed and estuary in which scallops were grown (using stable nitrogen isotopes), and not to food sources carried in from deeper waters by tidal currents. This strongly suggests a close coupling of watershed derived nitrogen into food (SPM) and scallop assimilation of this nitrogen into tissue. Further, Shriver *et al.* (2002) identified that land derived nitrogen load seemed unlikely to directly alter condition, reproductive potential, or mortality of bay scallops. In summary, Shriver *et al.* (2002) suggested that estuaries undergoing

anthropogenic nutrient additions (i.e. eutrophication) may provide food concentrations above the maximum ration assimilable, resulting in high bivalve growth rates.

## 5.7.3    Exposure Assessment

Evaluation of the level of eutrophication occurring in coastal waters and thus the potential exposure of this stressor to infaunal bivalves in the Hauraki Gulf Marine Park must draw from several studies of baseline water quality parameters conducted in this broad geographic area.  The Auckland Regional Council has monitored coastal water quality as part of the *"Long-Term Baseline (LTB) Programme"* since the late 1980's (Wilcock & Stroud 2000).  (Some of these parameters have only been monitored since the early 1990's).  Results from this programme are published yearly, while longer-term trend analysis is conducted every four years.  The most recent trend analysis currently available incorporates data to January 2000 (e.g. Wilcock & Stroud 2000).   This sampling programme includes monthly sampling of sites within the Hauraki Gulf Marine Park from north to south as follows: Goat Island, Ti Point, Kawau Bay (Algies Beach), Mahurangi, Orewa, Browns Bay, Chelsea, Hobsonville, Henderson, and Whau Creek.  There are no sites on the open coast of the Hauraki Gulf Marine Park south of Browns Bay.  For this sampling programme, Goat Island has been included as a reference site because:

- There is little development on nearby land that could adversely affect the water quality,
- Further development of the land is unlikely in the future.

Results from this monitoring programme suggest that ammonia, nitrite and nitrate concentrations are generally low (i.e. long-term median values of between 0.006-0.022, 0.001-0.004 and 0.004-0.019 mg $L^{-1}$ respectively) in seawater from sites within the Hauraki Gulf Marine Park (Wilcock & Stroud 2000).  Trend analysis identified very few trends in data from sites within the Hauraki Gulf Marine Park (Wilcock & Stroud 2000).  Results from the same monitoring programme suggest that dissolved oxygen concentrations for the Waitemata Harbour sites were decreasing over this same period, while all other East Coast sites showed no change or an increasing (e.g. Goat Island) dissolved oxygen concentration (Wilcock & Stroud 2000).  Long–term median dissolved oxygen concentrations ranged between 86-99% saturation over these sites.  Total phosphorus and dissolved reactive phosphorous also showed low concentrations (i.e. long-term median values of between 0.020-0.050 and 0.010-0.020 mg $L^{-1}$ respectively) in seawater for sites within the Hauraki Gulf Marine Park, and again few trends could be identified (Wilcock & Stroud 2000).  Chlorophyll *a* has been monitored at two baselines sites within the Hauraki Gulf Marine Park (i.e. Mahurangi, and Browns Bay) since May 1993.  Two more sites have been added since then (i.e. Goat Island and Ti Point), but the comparative record only covers from January 1998 onwards (Wilcock & Stroud 2000).  In January 2000 these data were considered insufficient for trend analysis (Wilcock & Stroud 2000).  Long–term median chlorophyll *a* concentrations ranged between 0.8 and 1.6 $\mu$g $L^{-1}$ over these sites.

In addition to this long-term baseline monitoring, several special investigations of other sites have been conducted by the Auckland Regional Council.  These sites include the Mahurangi Harbour, the Upper Waitemata Harbour and the Tamaki Estuary for the period 1992-2001 (Wilcock & Kemp 2002).   In summarising results of these investigations, Wilcock & Kemp (2002) found that ammonia, nitrite and nitrate concentrations were generally low (i.e. long-term median values of <0.2 mg $L^{-1}$) in seawater from sites within the Mahurangi Harbour, the Upper Waitemata Harbour and Tamaki Estuary.  However, concentrations of ammonia in some freshwater sites within the Tamaki estuary have at times exceeded values which are known to be harmful to fish (Wilcock & Kemp 2002).   Nevertheless, on the basis of medians and trends, these concentrations were

considered not to pose a threat to aquatic life in the Tamaki Estuary (Wilcock & Kemp 2002). Nitrite concentrations were close to the limit of detection i.e. 0.001 mg L$^{-1}$(Wilcock & Kemp 2002) in all the areas studied. Nitrate concentrations in the Mahurangi Harbour were low at the more saline sites (with a long–term median of <0.01 mg L$^{-1}$ at the most seaward site) and showed a downward trend over time (Wilcock & Kemp 2002). Upper Waitemata Harbour sites had uniformly low nitrate concentrations with a long term median of <0.1 mg L$^{-1}$ (these levels are approximately ten times those found at sites in the lower Waitemata Harbour from the long-term baseline monitoring programme) and no significant trend through time. Tamaki Estuary sites had the highest nitrate concentrations, with a long-term median of 0.013 mg L$^{-1}$ at the most seaward site. Wilcock and Kemp (2002) suggested that nitrate levels at some freshwater sites further up the Tamaki Estuary are likely to stimulate plant growth that may be problematic, and identified little change in this variable through time. Total phosphorus concentrations were very low in the Upper Waitemata Harbour sites (e.g. 0.003-0.006 mg L$^{-1}$) and show little change with time (these median concentrations were 10 times lower than lower Waitemata Harbour sites included in the long-term baseline monitoring programme). Mahurangi Harbour sites had concentrations consistent with those from other sites within the Hauraki Gulf Marine Park included in the long-term baseline monitoring programme. Seawater sites within the Tamaki Estuary again showed concentrations of Total Phosphorous similar to other sites within the Hauraki Gulf Marine Park included in the long-term baseline monitoring programme (i.e. 0.05 mg L$^{-1}$). Median concentrations of dissolved reactive phosphorus are also similar to other sites within the Hauraki Gulf Marine Park included in the long-term baseline monitoring programme (i.e. 0.02-0.03 mg L$^{-1}$). Median concentrations of chlorophyll *a* were approximately 10 times lower in the Mahurangi Harbour, upper Waitemata Harbour and Tamaki Estuary than in other sites within the Hauraki Gulf Marine Park included in the long-term baseline monitoring programme.

A broader description of nitrogen and phosphorous nutrient dynamics within the Hauraki Gulf and the Firth of Thames can be found in a report entitled "*Factors related to the sustainability of shellfish aquaculture operations in the Firth of Thames: a preliminary analysis*" (Broekhuizen *et al.* 2002).

There is the potential to use macroalgal species as indicators of environmental nitrogen loading (e.g. the work by Neil Barr (Leigh Marine Laboratory) with *Ulva* species in New Zealand).

### 5.7.4     Risk Characterisation

Eutrophication is globally one of the leading problems altering the environmental quality of coastal waters today (GESAMP 1990, Valiela *et al.* 1997 a,b). Coastal areas have higher natural nutrient values than the open sea, due to inflows from land, more shallow and complex topography, and higher water temperatures (Bonsdorff *et al.* 2002). Estuarine ecosystems may show phosphorus-limitation at their freshwater extreme, grading through to nitrogen-limitation at their seaward end (UK Eutrophication Forum 2002).

From the currently available water quality information it is suggested that the level of nutrient enrichment (e.g. nitrogen and phosphorus) in the Hauraki Gulf Marine Park is generally low, with potential eutrophication more likely to occur in the upper reaches of estuaries than outer estuarine sites or along the open coast.

Eutrophication is considered to be a problem only when nutrient enrichment causes adverse biological effects or when the risk of such adverse effects arising is judged to be significant. However there are potential difficulties associated with trying to quantify this risk in both

theoretical and practical terms.  Risk assessment methodology is broadly based on a comparison between predicted environmental concentration (PEC) and predicted no effect concentration (PNEC).  However, a PNEC cannot be defined for nutrient enrichment.  The ecosystem reaction to increased nutrient concentrations depends on many factors which vary spatially and temporally.  Phytoplankton within a water body will react to additional nutrient input only if it is the growth-determining factor, which depends on other factors such as temperature, light, low water-flow, concentrations of other nutrients (e.g. iron, silicate oligo-elements) and ecosystem balance (e.g. grazer populations).  Equally however, the ecosystem will not react to addition of nutrients if it is already at concentrations higher than is needed by the phytoplankton, as is often the case.  "No effect" or "effect" of an increase in nutrient loadings to a given part of a water body is thus dependent on a number of other factors: is plant/algal development being limited in the present circumstances by nutrient availability in the water?  Or is it being limited by other nutrients or by other factors?  A "No effect concentration" thus cannot be defined either in theory, or by any form of practical or field "assay" or test.  The results will in each case depend directly on the other parameters set for the test: concentrations of other nutrients, light, temperature, algal and grazer species and density.

Due to the multiplicity of factors that may impact on the potential effects of eutrophication it is currently difficult to assess the risk of nutrient enrichment to infaunal bivalve populations within the Hauraki Gulf Marine Park.  The only negative temporal trend identified in the monitoring programmes is a decreasing dissolved oxygen concentration at some sites in the Waitemata Harbour, but all the concentrations measured were well within the physiological tolerance range of bivalve shellfish (Wilcock & Stroud, 2000).

## 5.7.5     References

**Aarnio, K. & J. Mattila  2000.**  Predation by juvenile *Platichthys flesus* (L.) on shelled prey species in a bare sand and a drift algae habitat. *Hydrobiologia*. 440. 347-355.

**Bonsdorff, E. & R. Wenne  1989.**  A comparison of condition indices of *Macoma balthica* (L.) from the northern and southern Baltic Sea. *Netherlands Journal of Sea Research*. 23(1): 45-55.

**Bonsdorff, E., C. Ronnberg & K. Aarnio  2002.**  Some ecological properties in relation to eutrophication in the Baltic Sea. *Hydrobiologia*. 475/476: 371-377.

**Broekhuizen, N., J. Zeldis, S. A. Stephens, J. W. Oldman, A. H. Ross, J. Ren & M. R. James  2002.**  Factors related to the sustainability of shellfish aquaculture operations in the Firth of Thames: a preliminary analysis. NIWA Client Report EVW02243, prepared for Environment Waikato and the Auckland Regional Council. February 2002.

**GESAMP  1990.**  The state of the marine environment.  Joint Group of Experts on the Scientific Aspects of Marine Pollution.  Reports and Studies, Number 39.  United Nations Environmental Programme.

**Larsson, U., R. Elmgren & F. Wulff  1985**.  Eutrophication and the Baltic Sea: causes and consequences. *Ambio*. 14: 9-14.

**Morse *et al.* 1993.**  The economic & environmental impact of phosphorus removal from wastewater in the European Community.  Selper Publications, London.

**Ryther, J. & W. Dunstan  1971**.  Nitrogen, phosphorus, and eutrophication in the coastal marine environment. *Science*. 171: 1008–1013.

**Shriver, A. C., R. H. Carmichael & I. Valiela  2002.**  Growth, condition, reproductive potential, and mortality of bay scallops, *Argopecten irradians*, in response to eutrophic-driven changes in food resources. *Journal of Experimental Marine Biology and Ecology*. 279: 21-40.

**UK Eutrophication Forum  2002.**  http://www.fwr.org/eutrophi/eutrphuk.htm

**Valiela, I. 1995.**  Marine Ecological Processes, 2nd Edition.  Springer-Verlag, New York.

**Valiela, I., J. McClelland, J. Hauxwell, P. J. Behr, D. Hersh & K. Foreman  1997a.**  Macroalgal blooms in shallow estuaries: controls and ecophysiological and ecosystem consequences. *Limnology and Oceanography*. 42: 1105– 1118.

**Valiela, I., G. Collins, J. Kremer, K. Lajtha, M. Geist, B. Seely, J. Brawley, & C. H. Sham  1997b.**  Nitrogen loading from coastal waters to receiving estuaries: new method and application. *Ecological Applications*. 7: 358–380.

**Vant, W. N. & K. A. Safi  1996.**  Size-fractionated phytoplankton biomass and photosynthesis in Manukau Harbour, New Zealand.  *New Zealand Journal of Marine and Freshwater Research.*  30: 115-125.

**Weiss, E., R. H. Carmichael & I. Valiela  2002.**  The effect of nitrogen loading on the growth rates of quahogs (*Mercenaria mercenaria*) and soft-shell clams (*Mya arenaria*) through changes in food supply.  *Aquaculture.* 211: 275-289.

**Wilcock, R. J., & M. J. Stroud  2000.**  Year 2000 Summary Report.  Baseline water quality stream, lake, and saline waters.  Auckland Regional Council Technical Publication, No. 132.

**Wilcock, R. J. & C. L. S. Kemp  2002.**  Water quality surveys of Mahurangi Harbour, Upper Waitemata Harbour & Tamaki Estuary 1992-2001.  Auckland Regional Council Technical Publication, No. 191, July 2003.

## 5.8 STRESSOR: CLIMATE CHANGE/GLOBAL WARMING

### 5.8.1 Hazard Identification

"*Environmental change is a fact of life on earth, as is the evolution and extinction of species*" (Kennedy *et al.* 2002). At the outset it should be noted that scientists are still debating the true extent of global warming and its impacts (see de Freitas, 2003). However, there is now sufficient evidence to suggest that recent climate change has affected a broad range of organisms with diverse geographical distributions (Walther *et al.* 2002). Significant environmental factors that are expected to vary as part of global climate change, and thus might impact infaunal inter-tidal shellfish populations include:

- temperature,
- sea-level rise,
- the availability of water and associated nutrients from precipitation and runoff from land,
- wind patterns, and
- storminess (Kennedy *et al.* 2002).

Over the past 100 years, the earth's sea-level has risen by an average of 0.1-0.2 metres while the climate has warmed by approximately 0.6 ºC (Chiswell *et al.* 2001, Walther *et al.* 2002). Sea-level projections by the Intergovernmental Panel on Climate Change (IPCC) indicate a further sea-level rise of 0.3 to 0.5 m over this century (Goring & Bell 2001). Models suggest that the Southern Hemisphere will warm more slowly than the globe as a whole, because water sinking near the Antarctic will carry heat away from the surface to the deeper ocean (Mullan *et al.* 2001).

For New Zealand, climate modelling suggests faster warming in the north than in the south, and increasing westerly winds over the entire country (Mullan *et al.* 2001). Stronger westerlies may enhance the influx of warm subtropical waters from the north of New Zealand, but this may be offset by wind-induced upwelling of cooler subsurface waters (Carter 2001). This increase in westerly winds is predicted to increase sediment deposition at beaches on New Zealand's eastern coast (MfE 2001). Droughts may increase on the eastern coast (IPCC modelling, Nodder & Boyd 2001), which may influence the level of stormwater and nutrient runoff to the coastal marine environment. Additional modelling work suggests that sea-surface primary production around New Zealand will remain at levels similar to those observed currently (Nodder & Boyd 2001). Due to variations in climate patterns, and the El-Niño Southern Oscillation (ENSO) index, sea-levels around New Zealand are not expected to rise gradually, but to exhibit stepwise changes which may overshadow sea-level rise due to global warming (MfE 2001). Thus, over decades, marked jumps in sea-level caused by shifts in regional climate patterns are likely to be of greater importance than gradual sea-level rise.

Adverse effects of global climate change to infaunal inter-tidal bivalve populations may include both direct and indirect effects. Direct effects may include outcomes such as
- changes in the geographic distribution of bivalve species (via species specific temperature and precipitation physiological thresholds (Najjar *et al.* 2000, Kennedy *et al.* 2002, Walther *et al.* 2002)),
- changes in wind speed and direction may alter transport of larval bivalves along the coast or within estuaries,

---

- changes in temperature may alter timing of adult spawning (i.e. breeding period) (Penuelas & Filella 2001, MfE 2001) and growth rates (both larval and adult) (Brown and McLachlan 2002),
- changes in the severity and frequency of storms may also impact directly on infaunal bivalve populations via increases in mortality due to storm-related fatigue. Infaunal inter-tidal species may be affected by changes in both air and sea water temperatures.

Indirect effects may include disappearance of the lowest parts of the tidal flats in coastal areas (due to sea-level rise) thus decreasing the total space for inter-tidal bivalves (Honkoop 1996), along with the possible impacts of increased temperature on primary production (e.g. phytoplankton) (Brown & McLachlan 2002). Decreased precipitation and delivery of fresh water may alter species distributions and food webs in estuaries and affect the amount of time required to flush nutrients and contaminants from the system (Najjar *et al.* 2000, Kennedy *et al.* 2002).

Climate change may also impact on the frequency of harmful algal blooms and the relative success of invasive species introductions. Several species of toxic phytoplankton have been noted to enjoy a wider distribution during warmer periods (Tester 1996). Parasitic and predatory relationships involving inter-tidal infaunal bivalves may also be sensitive to changes in temperature (Najjar *et al.* 2000, see Harvell *et al.* 1999 for a review of links between climate change and other marine diseases). Predation pressure in marine ecosystems generally increases from the poles to the tropics (Vermeij 1978), so warming due to climate change could cause an ecological shift to increased predation if it led to greater diversity and numbers among predators (Kennedy *et al.* 2002). Sanford (1999) suggested that small changes in seawater temperature within the normal thermal tolerance range interfere with the controlling effect of a starfish on its mussel prey (however this study did not evaluate the changes in abundance of both predator and prey due to climate change).

The scale of global impacts of climate change on infaunal inter-tidal bivalves may potentially be quite large, with changes in water temperature causing changes to the range of some bivalve species. Unpredictable changes to coastal current patterns may also cause significant impacts to bivalve larval dispersal, although with the present information the magnitude of this effect is hard to gauge. The scale of change in physiological parameters in bivalve shellfish is also difficult to quantify as adaptation and equilibration may make specific impacts of climate change insignificant. Brown & McLachlan (2002) suggest that increased UV radiation is unlikely to have significant effects on species living within the sand (however, increased UV may cause indirect impacts, for example via increased conversion of phototoxic PAHs). In addition, Brown and McLachlan (2002) suggest that the effects of a rise in seawater temperature due to global warming up to 2025 are in general likely to be subtle rather than dramatic. A worst case scenario for the year 2025 appears to be a water temperature increase of between 1 and 1.25 ºC (Brown and McLachlan 2002). Given this time scale, and the relatively small changes in ambient seawater temperature envisaged, they predict that compensatory metabolic adaptations to temperature change are likely for most benthic species. The potential indirect effects of climate change to infaunal inter-tidal bivalves are also complex, and prediction is impossible without a clear understanding of the regional system in which they live.

The direct impact of climate change on NZ shellfish species has received no research attention to date. However, internationally, the impact of climate change on bivalve populations has been the subject of some research. Interestingly, a large proportion of this work appears to be in grey literature which was not easily accessible for this review. Honkoop & Van Der Meer (1998) found that changes in immersion time and winter temperature produced differing effects on reproductive output in different bivalve species, largely depending on the timing of their spawning. Strasser and

co-workers (2003) identified that successful bivalve recruitment may be correlated with severe winters, and that climate change may affect this pattern, producing more mild winters and subsequently less bivalve recruitment.  Brinkman *et al.* (2001) identified that higher environmental temperatures may cause lower larval growth for the Baltic Tellinid *Macoma balthica*.  Based on mesocosm studies Brinkman and co-workers (2001) further suggested that for bivalves, the two major climate change aspects had opposite effects: sea-level rise stimulated biomass and production, whereas increased temperature depressed bivalve production.  Weinberg *et al.* (2002) estimated that an increase in seawater temperature of 2 $^o$C at the southern geographic boundary would extend the period of thermal stress for ocean quahog (*Arctica islandica*) larvae from the present 167 days to 231 days, lowering survival of the larvae and depressing their recruitment to the resident population, which could eventually die out over a period of decades.  Kennedy & Mihursky (1971) identified that the commercially important soft clam (*Mya arenaria*) in Chesapeake Bay lives near its southern distribution limit and may be eliminated from the Bay if bottom water temperatures approach and remain near 32 $^o$C in summer.

## 5.8.2    Dose-Response Assessment

Due to the difficulty of accurately measuring change with respect to the combined effects of environmental factors influenced by climate change and the long time-period over which these changes occur (i.e. many bivalve generations), it is hard to directly develop a dose-response relationship.

There are no specific data to indicate the way in which climate change may impact on infaunal bivalves in the Hauraki Gulf Marine Park.  While the impact on different species is likely to vary due to interspecific differences in physiological tolerances, there may also be interspecific differences in dose-response due to other factors.  For example, Keller and co-workers (1999) demonstrated that increases in winter seawater temperatures could result in increased cropping of phytoplankton by zooplankton, causing a reduced supply of detrital material for benthic organisms.  The relative impact of this compared to reduced phytoplankton supplies to suspension-feeders is currently unknown.

Climate change may alter the geographic distribution of certain species due to an increase in temperature exceeding a known thermal tolerance range for a particular species.  Roy *et al.* (2001) have suggested that with respect to climate change the range limits of large-bodied bivalve species are more unstable than those of small-bodied bivalves.  They also suggested that this trend is not impacted by feeding mode (e.g. suspension- *versus* deposit-feeding) or modes of reproduction, or development.  Since the thermal tolerances of the four infaunal bivalve species of interest in this review are currently not clearly defined, the current geographic spread of these species must be used to evaluate their known thermal tolerance range.  All of these species have geographic distributions to the north of current populations in the Hauraki Gulf Marine Park.  This suggests that these species can tolerate temperatures that are warmer than those encountered within the Hauraki Gulf Marine Park.  It is therefore unlikely that climate change will cause the disappearance of these species within the Hauraki Gulf Marine Park.  Investigation of the tolerance limits of these species would aid in a more accurate evaluation of this impact.

## 5.8.3    Exposure Assessment

As climate change is a global phenomenon, it will by definition be a structuring process influencing ecosystems within the Hauraki Gulf Marine Park, and indeed the rest of New Zealand.  However,

when considering the impact of climate change, regional changes which can be highly spatially heterogeneous are more relevant to ecological responses to climate change (Walther *et al.* 2002).

Much of the emphasis of research on climate change to date has been on modeling the impacts of predicted changes. The following section therefore summarizes relevant information for each of the significant environmental factors that are expected to vary as part of global climate change.

*Temperature:*
Variability in both air and seawater temperature may impact infaunal inter-tidal species. Seawater temperatures in temperate regions may rise to levels that are stressful or lethal to resident organisms (Kennedy *et al.* 2002). There is some confidence about predicting what would happen to organisms and oxygen concentrations as temperature increases, but less confidence with respect to the influence of temperature on interactions among organisms (e.g., predator-prey, parasite-host, competition for resources) (Kennedy *et al.* 2002). Diurnal temperature ranges have decreased because minimum temperatures are increasing at about twice the rate of maximum temperatures (Walther *et al.* 2002). Most aquatic organisms extract the oxygen necessary for their metabolic demands and survival from the water in which they live. Temperature affects the dissolved oxygen concentrations in water. For each degree C that seawater warms, oxygen solubility (the capacity to dissolve oxygen) decreases by about 2% (Najjar *et al.* 2000). Consequently, higher seawater temperature raises the metabolism of cold-blooded aquatic animals (such as bivalves), thereby increasing their metabolic need for oxygen. Changes in oxygen concentration also depend on biotic factors. Results from the Auckland Regional Council's "*Long-Term Baseline (LTB) programme*" for monitoring coastal water quality indicate that saline water temperatures within the Hauraki Gulf Marine Park have increased on average about 0.24 °C per year for the 11 sites from the East Coast and Waitemata Harbour (for the period 1991 to 1999) (Wilcock & Stroud 2000). Results from the same monitoring program suggest that dissolved oxygen concentrations for the Waitemata Harbour sites were decreasing over this same period, while all other East Coast sites showed no change or an increasing (e.g. Goat Island) dissolved oxygen concentration (Wilcock & Stroud 2000). However, increased seawater temperature may not be the only factor impacting on the level of dissolved oxygen in seawater as nutrient enrichment may also impact on this parameter (refer to the nutrient enrichment section). The magnitude of the effect of temperature mediated changes in oxygen solubility in seawater on Hauraki Gulf Marine Park infaunal inter-tidal bivalves is currently not known. The magnitude of the effect of temperature mediated changes in oxygen solubility in Hauraki Gulf Marine Park infaunal inter-tidal bivalves is currently not known.

With global increases in temperature, species are expected to track the shifting climate and shift their distributions poleward in latitude. However, changes in distribution will be subject to dispersal ability and resource availability (Walther *et al.* 2002).

Climate change may also alter species interactions by changing the timing of physiological events (Penuelas and Filella, 2001). For example, altered temperature regimes could advance or retard the timing of reproduction for many species. Changes in temperature may mean that production of young of certain species becomes uncoupled with production of their specific larval food source (Kennedy *et al.* 2002).

*Sea-level Rise:*
There is some confidence about predicting the effects of sea-level rise on shallow continental margins, including flooding of wetlands, shoreline erosion, and enhanced storm surges (Kennedy *et al.* 2002). In regions under ENSO (El Nino/Southern Oscillation) influence, change may happen rapidly during warm periods, with setbacks during cool periods (Walther *et al.* 2002). Step-wise

changes in sea-level may influence the distribution of bivalve populations within the Hauraki Gulf Marine Park.

***Rainfall and nutrient flux from land:***
There is less confidence in the ability to predict the effects of climate change on precipitation, and its effects on nutrient flux from land (Kennedy *et al.* 2002). Precipitation affects runoff into estuaries and therefore influences estuarine circulation, concentrations of nutrients and contaminants, stratification and oxygen deficits, and recruitment of some species (i.e. the addition of new individuals to a species' population).

***Wind patterns and coastal currents:***
There is less confidence in the ability to predict the effects of climate change on wind patterns, and the effects on coastal currents (Kennedy *et al.* 2002). Wind speed and direction influence coastal circulation, including currents that deliver fish and invertebrate larvae from coastal waters into estuaries (Kennedy *et al.* 2002).

***Storminess:***
Storms and their associated winds can have major negative effects on coastal ecosystems and shoreline structures (Kennedy *et al.* 2002). However, predicting global patterns of change resulting from global warming suffer from a considerable measure of uncertainty (e.g. storminess and current patterns) (Brown & McLachlan 2002).

No reliable projections on changes in storm frequency or intensity in New Zealand in response to climate change are currently possible (MfE 2001). Any change in storminess will impact on Hauraki Gulf Marine Park infaunal bivalves. Increased storminess will have greater impact on infaunal bivalve populations on open exposed coasts within the Hauraki Gulf Marine Park, e.g. north of Whangaparoa Peninsula and on the eastern coast of Coromandel Peninsula.

Infaunal bivalve populations in the Hauraki Gulf Marine Park may also be impacted by the predicted increased sediment deposition on the eastern coast due to increasing westerly winds. Changes in wind direction may also alter transport of larval bivalves along the coast. It is difficult to identify the extent of this influence, as our current knowledge of larval transport of these infaunal bivalve populations within the Hauraki Gulf Marine Park is limited.

Droughts may also become more frequent on the eastern coast influencing the level of stormwater and nutrient runoff to the coast. This may impact Hauraki Gulf Marine Park infaunal bivalves directly and also via the indirect effects this will have on associated contaminant loads (e.g. metals).

The level of primary production is predicted to stay at levels similar to those currently observed, so this should not impact on Hauraki Gulf Marine Park infaunal bivalves.

## 5.8.4    Risk Characterisation

Predictions of the various effects of climate change on infaunal bivalves and their ecosystems are associated with varying degrees of confidence. Understanding how species respond to changing climates is essential for predicting the biological consequences of regional climate change (Davis *et al.* 1998).

Considering that the infaunal bivalves of interest in this risk assessment are not near the northern limits of their geographic distributions, it is unlikely that increased temperatures would cause these

species to become geographically extinct within the Hauraki Gulf Marine Park. Paine (1993) argued that immediate marine extinction in response to climate warming is less likely than as a result of direct human disruption of aquatic systems by overexploitation, introduction of exotic species, and coastal pollution. However the possibility that climate changes will also affect how an organism responds to other environmental stressors cannot be ignored.

Information on the possible changes to regional climate patterns and the effects of climate variability on bivalve stocks is still insufficient to allow adequate prediction of possible impacts of long-term climate change. However, it is likely that impacts will vary between species depending on their physiological tolerances, biology and behavioural characteristics. The effects of climate change potentially impact on a complex set of variables within the habitats of infaunal bivalves, and significant further work would be required to be able to predict the impact of such change on infaunal bivalve populations in the Hauraki Gulf Marine Park.

## 5.8.5    References

**Brinkman, A. G., M. Baptist, J. Drent, M. Janssen-Stelder, M. W. W. van der Tol, B. J. Ens, M. Vonk & K. Kersting  2001.** Modelling the impact of climate change on the Wadden Sea ecosystem. Report for the Dutch National Research Programme on Global Air Pollution and Climate Change. 210pp.

**Brown, A. C. & A. McLachlan  2002.** Sandy shore ecosystems and the threats facing them: Some predictions for the year 2025. *Environmental Conservation*. 29: 62-77.

**Carter, L.  2001.** Currents of Change: The ocean flow in a changing world. *Water & Atmosphere*. 9(4): 15-17.

**Chiswell, S., M. Bowen and B. Mullan  2001.** Climate change: New Zealand in a warming world. *Water & Atmosphere*. 9(4): 8-9.

**Davis, A. J., L. S. Jenkinson, J. T. I. Lawton, B. Shorrocks & S. Wood  1998.** Making mistakes when predicting shifts in species range in response to global warming. *Nature*. 391: 783-786.

**De Freitas, C.  2003.** Viewpoint: Jury still out on global warming. *New Zealand Geographic*. 64: 6-8.

**Goring, D. & R. Bell  2001.** Sea level on the move. *Water & Atmosphere*. 9(4): 20-21.

**Harvell, C. D., K. Kim, J. M. Burkholder, R. R. Colwell, P. R. Epstein, D. J. Grimes, E. E. Hofmann, E.K. Lipp, A. D. M. E. Osterhaus, R. M. Overstreet, J. W. Porter, G. W. Smith & G. R. Vasta  1999.** Emerging marine diseases – climate links and anthropogenic factors. *Science*. 285: 1505-1510.

**Honkoop, P. J. C.  1996.** Winter temperature, immersion time and bivalve reproduction. Annual Report Netherlands Institute for Sea Research. Pp. 56-57.

**Honkoop, P. J. C. & J. Van Der Meer  1998.** Experimentally induced effects of water temperature and immersion time on reproductive output of bivalves in the Wadden Sea. *Journal of Experimental Marine Biology and Ecology*. 220(2): 227-246.

**Keller, A. A., C. A. Oviatt, H. A. Walker & J. D. Hawk  1999.** Predicted impacts of elevated temperature on the magnitude of the winter-spring phytoplankton bloom in temperate coastal waters: a mesocosm study. *Limnology and Oceanography*. 44: 344-356.

**Kennedy, V. S. & J. A. Mihursky  1971.** Upper temperature tolerances of some estuarine bivalves. *Chesapeake Science*. 12: 193-204.

**Kennedy, V. S., R. R. Twilley, J. A. Kleypas, J. H. Cowan Jr. & S. R. Hore  2002.** Coastal and marine ecosystems & global climate change. Potential Effects on U.S. Resources. Prepared for the PEW Centre on Global Climate Change. 52pp.

**Ministry for the Environment (MfE)  2001.** Climate change impacts on New Zealand. Report prepared by the Ministry for the Environment, June 2001.

**Mullan, B., M. Bowen & S. Chiswell  2001.** The Crystal Ball: Model predictions of future climate. *Water & Atmosphere*. 9(4): 10-11.

**Najjar, R.G., H. A. Walker, P. J. Anderson, E. J. Barron, R. J. Bord, J. R. Gibson, V. S. Kennedy, C. G. Knight, J. P. Megonigal, R. E. O'Connor, C. D. Polsky, N. P. Psuty, B. A. Richards, L. C. Sorenson, E. M. Steele & R. S. Swanson  2000**. The potential impacts of climate change on the mid-Atlantic coastal region. *Climate Research*. 14: 219-233.

**Nodder, S. & P. Boyd  2001.** What grows up must fall down: The potential impact of climate change on plankton and carbon export. *Water & Atmosphere*. 9(4): 28-29.

**Paine, R. T.  1993.** A salty and salutary perspective on global change. *In:* Biotic Interactions and Global Change. Kareiva, P. M., J. G. Kingsolver & R. B. Huey (Eds.). Sinauer, Sunderland, MA. Pp. 347-355.

**Penuelas, J. & I. Filella  2001.** Responses to a warming world. *Science*. 294: 793-795.

**Roy, K., D. Jablonski & J. W. Valentine  2001.**  Climate change, species range limits and body size in marine bivalves. *Ecology Letters*. 4: 366-370.

**Sanford, E.  1999.**  Regulation of keystone predation by small changes in ocean temperature. *Science*. 283: 2095-2097.

**Strasser, M., R. Dekker, K. Essink, C. Gunther, S. Jaklin, I. Kroncke, P. B. Madsen, H. Michaelis & G. Vedel  2003.**  How predictable is high bivalve recruitment in the Wadden Sea after a severe winter? *Journal of Sea Research*. 49(1): 47-57.

**Tester, P. A.  1996.**  Climate change and marine phytoplankton. *Ecosystem Health*. 2(3): 191-197.

**Vermeij, G. J.  1978.**  Biogeography and Adaptation: Patterns of Marine Life.  Harvard University Press, Cambridge, MA.

**Walther, G-R, E. Post, P. Convey, A. Menzel, C. Parmesan, T. J. C. Beebee, J-M. Fromentin, O. Hoegh-Guldberg & F. Bairlein  2002.**  Ecological responses to recent climate change. *Nature*. 416: 389-395.

**Weinberg, J. R., T. G. Dahlgren & K. M. Halanych  2002.**  Influence of rising sea temperatures on commercial bivalve species off the U.S. Atlantic coast. *In:* Fisheries in a Changing Climate. N. A. McGinn (Ed.). American Fisheries Society Symposium, 32. Bethesda, MD.

**Wilcock, R. J., & M. J. Stroud  2000.**  Year 2000 Summary Report.  Baseline water quality stream, lake, and saline waters.  Auckland Regional Council Technical Publication, No. 132.

## 5.9.1     Hazard Identification

Globally there is concern that coastal systems are currently under heavy human harvest pressure, which may threaten not only the fisheries, but also the very existence of some harvested species (Kennedy *et al.* 2002).  Peterson (2002) found that human over-fishing can impact significantly on recruitment of bivalve shellfish if adult spawning stock biomass is reduced by such intensive harvest that it lowers the recruitment derived from progeny of the over-fished generation of adults. There is no commercial harvesting of cockles (*A. stutchburyi),* pipi (*P australis),* tuatua (*P. subtriangulata*) or wedge shells (*M. liliana*) in the Hauraki Gulf Marine Park. However, it is a common perception that recreational harvesting has been a major contributor to the decline of shellfish abundance at popular beaches in the Auckland region (Gardner 1992, Stewart & Creese 2002, Walshe & Akroyd 2002).  There have been few studies that specifically examine the impact of human harvesting on New Zealand infaunal bivalve species in the Hauraki Gulf Marine Park.

Their abundance in ancient shell middens suggests that *P. australis* and *A. stutchburyi* have a long human harvest history in New Zealand (Best 1986).  A survey of non-commercial harvesters at 13 soft-shore habitats in the Auckland Metropolitan area by the Ministry of Agriculture and Fisheries during the summer of 1991-92 suggested that *A. stutchburyi* and *P. australis* were popular species targeted by non-commercial harvesters, with *P. subtriangulata* collected less frequently (the selection of beaches surveyed may have influenced these data) (Drey & Hartill 1993).  There is no mention of the collection of any wedge shells (*M. liliana*) in this survey.  Assuming that the species collected were correctly identified, this suggests that *M. liliana* may be less frequently targeted by non-commercial shellfish gatherers.

The extent to which human harvesting poses a hazard to infaunal bivalve populations depends on the way in which harvesting activities are managed.  There are a variety of options with respect to shellfish resource management practices, ranging from harvesting strategies that limit minimum size and quantity of adults that may be taken, to re-seeding programmes that supplement local populations with artificially planted juveniles.  Other management strategies include predator control, enhancing the suitability of potential larval settlement areas, and planting adults to enhance the spawning stock.  These options differ since each concentrates on a different portion of the organism's life history (Malinowski & Whitlatch 1988).  The Ministry of Fisheries currently manage the harvesting of shellfish within the Hauraki Gulf Marine Park under the Fisheries Act (1996), which has as its purpose the utilisation of fisheries resources while ensuring sustainability. Currently the harvest of shellfish within the Hauraki Gulf Marine Park is managed through the institution of a daily bag limit.  In addition, in areas where shellfish stocks are depleted, a closure to the harvesting of all shellfish may be imposed.

## 5.9.2     Dose-Response Assessment

The impact of harvesting of any marine species is dependent upon the biology of the exploited species.  Adams (1980) illustrated the theoretical relationship between the harvestability and life history strategies of various fish species.  He found that those species with more "r-selected" traits (e.g. fast growth, early maturity, production of a large number of offspring) could withstand more intense harvesting than more "k-selected"  (e.g. slow growth, delayed maturity) species.  Infaunal bivalves have "r-selected" traits.

In addition to reduction in gamete production, harvest of adult spawners from a bivalve population may result in a number of other impacts. For marine broadcast spawners, one of the main constraints influencing fertilisation success is the rapid dilution of gametes. Fertilisation success decreases rapidly as the distance between adults increases (Metaxas *et al.* 2002). Thus it would be expected that reduction in density of infaunal bivalve beds through harvesting could potentially impact negatively on fertilisation success.

However, it appears that infaunal shellfish populations may be less sensitive to removal of adult shellfish from the population than to proportional impacts on the survivorship of larval and juvenile stages. In studies overseas that included the clams *Mercenaria mercenaria* and *Mya arenaria,* and the oyster *Crassostrea virginica,* Malinowski & Whitlatch (1988) found that there were positive correlations between size and fecundity, and size and survivorship (i.e. fecundity and survivorship increased with increasing shellfish size). Population growth rates were two to three orders of magnitude more sensitive to changes in survivorship in larval and juvenile stages of the life cycles than proportional changes in either survivorship or fecundity in adult size classes (Malinowski & Whitlatch 1988). In all cases, population growth rate was at least four orders of magnitude more sensitive to changes in juvenile survivorship than to deviations in fecundity of adults. Since larval and early post-settlement mortality rates are extremely high, very large increases in fecundity are necessary to duplicate only slight increases in juvenile survivorship (Malinowski & Whitlatch 1988).

There have been few studies that specifically examine the impact of human harvesting on New Zealand infaunal bivalves. Martin (1984) investigated the population responses of *A. stutchburyi* to exploitation by carrying out an experiment in the Whangateau Harbour simulating two levels of exploitation (total removal of *A. stutchburyi,* and removal of exploitable size class *A. stutchburyi*), and by monitoring an *A. stutchburyi* bed that was being exploited (Snake Bank, Whangarei Harbour). Results showed that total removal of *A. stutchburyi* led to lower levels of recruitment when compared to recruitment levels at ambient densities or densities reduced by exploitation. There was limited evidence that removal of exploitable size class *A. stutchburyi* led to recruitment levels that were higher than those observed at ambient densities. *A. stutchburyi* from areas that had all exploitable sized individuals removed showed significantly higher growth rates than correspondingly sized *A. stutchburyi* from areas of ambient density. *A. stutchburyi* from the simulated exploitation areas had significantly lower mortality rates than those in areas of ambient density (Martin 1984). Subsequent studies of the growth rate of cockles on Snake Bank (e.g. Cryer 1997) have found significantly higher growth rates than those observed by Martin (1984), and it has been suggested that this is as a result of the lower densities of *A. stutchburyi* on the bank in recent years. It thus appears that density effects are relevant to the risk posed by human harvesting. If population growth is limited by intraspecific interactions (such as competition for food or space), harvesting can change the demography of the harvested species. In general, sustained harvesting is most easily tolerated by those species that experience increased productivity after individuals have been removed (Malinowski & Whitlatch 1988). This is consistent with the results of the study by Martin (1984), except with respect to levels of harvesting that result in removal of adults from the bed altogether. Martin (1984) suggested that the lack of recruitment in beds from which all *A. stutchburyi* had been removed could be as a result of alteration of the substrate conditions, or elimination of chemical cues required for settlement (i.e. *A. stutchburyi* could be a gregarious species). There appear to have been no similar studies undertaken with respect to *P. australis*, *P. subtriangulata* or *M. liliana*.

In addition to the direct impacts of human harvesting arising from the removal of shellfish, harvesting activity may also have indirect impacts, such as those arising from the disturbance to the

sediment during the process of harvesting. Our literature search did not identify any studies that specifically examine the indirect impacts of non-commercial harvesting on New Zealand infaunal bivalve species.

In consideration of dose-response relationships, it is pertinent to note that although Cheltenham Beach has been closed to shellfish harvest since 1993, data collected by the Cheltenham Beach Caretakers indicate that the abundance of *A. stutchburyi* has continued to decline (see Section 4 *Status of bivalve populations in the Hauraki Gulf Marine Park*). Analysis of their size-frequency data for *P. australis* would further elucidate potential trends in abundance of this species. The Cheltenham Beach Caretakers have also collected long-term data on the abundance of *M. liliana* and other species at Cheltenham Beach, but these data have yet to be analysed with respect to trends in abundance. Eastern Beach was closed to harvesting in 1994, Long Bay in 1995, and Waikawau Beach in 2000. Inconsistencies and limitations in survey methodology mean that it is difficult to comment on the response of infaunal bivalve populations at Eastern Beach, although there does not appear to have been an increase in *A. stutchburyi* numbers from 1998 to 2002 (see Figure 4.6 in Section 4 earlier in this report). At Waikawau Beach numbers of *P. australis* increased from 0.32 $\pm$ 0.07 million in 1999/2000 to 1.29 $\pm$ 0.25 million in 2000/01. In both surveys, the adult mode of the population (comprising the majority of the population) consisted of individuals between 40-60 mm in shell length. In the 2000/01 survey some recruitment of smaller *P. australis* was also observed, consisting of individuals between 10-20 mm in shell length (Akroyd *et al.* 2000, Walshe *et al.* 2002). In the absence of long term data to provide an indication of the normal inter-annual variability of this population, it is difficult to interpret these data in terms of assessing a response to closure to harvesting. There have been no surveys of infaunal bivalve abundance at Long Bay since the closure to harvesting, so the resulting impact on shellfish abundance there is unknown.

### 5.9.3    Exposure Assessment

In 1987, the Ministry of Agriculture and Fisheries undertook a marine recreational fishing survey throughout New Zealand (Sylvester *et al.* 1991). This was undertaken by telephone survey. This showed that the largest estimated number of fishers for any area within New Zealand occurred in two North Island areas: the Hauraki Gulf (108,906 fishers) and Coromandel-Bay of Plenty (102,402 fishers). The next most popular fishing area was east Northland, which had considerably lower numbers of fishers (41,308) (Sylvester *et al.* 1991). The same survey showed that the estimated number of fishers gathering *A. stutchburyi* (approximately 30,000 fishers), *P. australis* (approximately 75,000 fishers) and *P. subtriangulata* (approximately 24,000 fishers) was in each case highest in Quota Management Area 1 (QMA1, an area from North Cape to Cape Runaway) (Sylvester *et al.* 1991). These results suggest that the Hauraki Gulf Marine Park lies within an area that has relatively high non-commercial fishing pressure for infaunal shellfish compared to elsewhere in New Zealand.

A more recent National Marine Recreational Fishing Survey has been undertaken for the Ministry of Fisheries (Fisher & Bradford 1998). This study was based on a national telephone and diary survey. The survey used a preliminary telephone survey to determine the number of households that contain marine non-commercial fishers from a random selection of households with a phone. One randomly selected fisher from each of these households was asked to keep a diary of their recreational (i.e. non-commercial) fishing trips during the year. The figures provided by the diarists were scaled to give estimates for the total marine non-commercial fishing population of New Zealand. The data collected in this survey is likely to be of better quality than that collected in the telephone survey in 1987, which relied on participants to recall their fishing activities some time after the event. (Note however that in both these studies the population surveyed included only

those people that have telephone connections. It is possible that those people who do not have telephones may be more reliant on non-commercially harvested seafood as a source of food). Amongst other data, data relating to the number of cockles (*A. stutchburyi*), pipi (*P. australis*) and tuatua (*P. subtriangulata*) gathered by diarists in various regions were collected in this survey. Data for regions within the Hauraki Gulf Marine Park are presented in Table 5.9.1 below.

| Region | *A. stutchburyi* | *P. australis* | *P. subtriangulata* |
|---|---|---|---|
| **Barrier Islands** | 100 | 0 | 0 |
| **Western Gulf** | 585 | 885 | 0 |
| **Inner Gulf** | 449 | 154 | 510 |
| **Firth of Thames** | 0 | 122 | 50 |
| **Eastern Gulf** | 0 | 430 | 75 |
| **Eastern Coromandel** | 413 | 3,570 | 1,549 |
| **Total** | 1,547 | 5,161 | 2,184 |

**Table 5.9.1: Numbers of shellfish gathered by diarists in each fishing region over one year in a survey by Fisher & Bradford (1998). [Barrier Islands include the Mokohinau Islands, Little Barrier Island and Great Barrier Island; Western Gulf includes Cape Rodney to Piripiri Point; Inner Gulf includes Piripiri Point to Orere Point; Firth of Thames includes Orere Point to Deadmans Point; Eastern Gulf includes Deadmans Point to Cape Colville, including Channel Island; and Eastern Coromandel includes Cape Colville to Waihi Bluffs, excluding Mayor Island].**

Of the three infaunal shellfish species, diarists gathered greatest numbers of pipi (*P. australis*). Fewer cockles (*A. stutchburyi*) were gathered than the other two species. The data suggest that within the Hauraki Gulf Marine Park, the eastern Coromandel region is the most popular area for gathering *P australis* and *P. subtriangulata*, while more *A. stutchburyi* are gathered in the western Hauraki Gulf than other regions. Among other factors, this pattern is likely to be influenced by both the distribution and abundance of shellfish of each species throughout the region. The survey results suggested that the diarists mostly fish near home, but that significant numbers of diarists from the Auckland region fish in the eastern Coromandel region (Fisher & Bradford 1998).

Drey & Hartill (1993) undertook a qualitative survey of inter-tidal harvesting by non-commercial fishers in the Auckland Metropolitan area through interviews of people engaged in gathering shellfish. The objectives of the survey were to characterise the activities of different human population groups in terms of their cultural preferences for harvesting particular species of inter-tidal organisms with regard to the quantities being taken and the areas being patronised. There were significant limitations to their survey relating to several sources of bias in their data, plus information of variable quality (these limitations are discussed in detail in their report). Thus Drey & Hartill (1993) caution against trying to draw quantitative conclusions from their data. The study identified that there was potentially a problem with compliance with the bag limit for *A. stutchburyi* and *P. australis*, but the magnitude of this problem could not be quantified from the data due to bias in the survey method (Drey & Hartill 1993).

A further survey of shellfish harvesting in the Auckland Metropolitan area was undertaken by Hartill & Cryer in 1997-1998 (Hartill & Cryer 1999). The purpose of this survey was "*to estimate the annual harvest of pipi, cockles and tuatua and characterise the overall pattern of harvest from 1 December 1997 to 30th November 1998 at four beaches within the Auckland metropolitan area*" (Hartill & Cryer 1999). For various reasons, two of the beaches initially chosen for the survey were

deemed unsuitable, and there were problems with data collection at one of the replacement sites. The three beaches where the survey was ultimately undertaken included Cornwallis Beach and Mill Bay in the Manukau Harbour, and Okoromai Bay (Whangaparaoa Peninsula). The researchers concluded that the levels of harvesting of *A. stutchburyi* at Okoromai Bay and Cornwallis Beach were similar to calculated estimates of optimal levels of yield, and that therefore harvest levels were sustainable. They noted that the estimated annual harvest at Mill Bay slightly exceeded the optimal levels of yield.

Subsequent to this survey the daily bag limits for infaunal shellfish were revised, and for example, the daily limit for *A. stutchburyi* harvested in this region was reduced from 150 per person to 50 per person. The extent to which this has reduced harvest pressure on infaunal bivalve populations is currently unknown, as there have been no subsequent surveys of shellfish harvesting activity at the beaches where the previous surveys were conducted. However, a survey that includes investigation of shellfish harvesting at Okoromai Bay, Umupuia, Tairua and Whangamata (along with other areas outside the Hauraki Gulf Marine Park) is currently being undertaken for the Ministry of Fisheries by Akroyd Walshe Ltd, and the results of this should be available mid-2004 (K. Walshe, Akroyd Walshe Ltd., *pers. comm.*). The inclusion Okoromai Bay, which was previously subject to a survey of non-commercial harvesting activity (Hartill & Cryer 1999), may provide some data relevant to this issue.

From their survey, Hartill & Cryer (1999) calculated that the total estimated number of cockles harvested from Okoromai Bay during the 12-month survey period was 1,069,625, equating to a total of 17,192 kg. The mean length of cockles harvested ranged from 32.8 mm in winter to 35.9 mm in autumn. The mean lengths taken appear to vary slightly between sites, as the mean lengths at Mill Bay and Cornwallis were in the ranges 27.2 mm (summer) to 29.7 mm (spring), and 31.8 mm (summer) to 26.8 mm (winter) respectively. The researchers suggested that the observed seasonal differences in the length frequency and mean lengths of cockles harvested could be related to the sample sizes taken, since these differences decreased with increasing sample size (Hartill & Cryer 1999).

Hartill & Cryer (1999) also investigated the length frequency distribution of *A. stutchburyi* harvested during the four seasons at Cornwallis, Mill Bay and Okoromai Bay compared to the length frequency distribution of the *A. stutchburyi* populations present at these sites. Statistical analysis indicated that *A. stutchburyi* in the size classes approximately >31 mm in length were harvested in a higher proportion than present in the population at Okoromai Bay. (At Mill Bay and Cornwallis, *A. stutchburyi* approximately greater than 23 mm and 31 mm respectively were harvested in a higher proportion than were present in the populations). Thus it appears that harvesters were actively targeting larger *A. stutchburyi*.

One of the most intensive studies of non-commercial harvesting of infaunal bivalves in the Hauraki Gulf Marine Park was that undertaken by Kearney (1999) to investigate the ecology and harvesting of *A. stutchburyi* from Lew's Bay in the Whangateau Harbour. In addition to a survey of the *A. stutchburyi* population to determine their abundance, the study involved structured interviews of all harvesters on the beach to ascertain the access points to Lew's Bay, knowledge of the legal daily bag limit, frequency of harvesting, purpose of harvest (i.e. recreational or customary use), other species being harvested, number of harvesters in each group, ethnic identity of harvesters and the location of their permanent residency. For each group, the total wet weight of harvested *A. stutchburyi* was measured. Using a predetermined conversion factor, the weight was used to determine compliance with the daily bag limit for each group. Results of the study suggested that the annual harvest of *A. stutchburyi* from Lew's Bay for 1997-1998 was estimated to be

approximately 28 tonne, or if adjusted for the higher level of harvesting observed during the holiday period December 24th-January 4th, approximately 33 tonne (Kearney 1999).  Kearney further notes that this figure suggests that the estimated non-commercial harvest figures for Quota Management Area 1 previously derived from the Ministry of Fisheries telephone and diary surveys (e.g. 55 tonne, Annala *et al.* 1998) grossly underestimated the actual harvest, since Lew's Bay is only one of many harvestable stocks within the area.  Kearney (1999) also observed that the majority of sampled harvesters (52%) were taking in excess of the daily bag limit (which at the time was 150 *A. stutchburyi* per person per day).

*A. stutchburyi* over 18 mm in length are sexually mature (Larcombe 1971).  Kearney (1999) found that in Lew's Bay, the mean length of *A. stutchburyi* harvested was 36 mm, and 99% were in excess of 30 mm.  These results are broadly consistent with those of other researchers (e.g. Belton 1986, Annala *et al.* 1998, Hartill & Cryer 1999, Connew 2000), which suggests that the minimum acceptable size harvested appears to be close to 30 mm.  Kearney also observed that harvesters targeted the largest individuals within an area, and that mean sizes of shellfish collected varied between areas depending on the size frequency distribution in each area (Kearney 1999).

There are no data available regarding the size of *P. australis* gathered by non-commercial harvesters in the Hauraki Gulf Marine Park.  However, limited data were gathered at Mill Bay and Cornwallis by Hartill & Cryer (1999) in their surveys of non-commercial shellfish harvesting in the Auckland metropolitan area.  They found that *P. australis* in the range of 36-46 mm in length were gathered at Cornwallis, with a mean length of 39.6 mm (n=32).  At Mill Bay, the size range was 30-48 mm in length, with a mean length of 37.8 mm (n=40).  The extent to which the results at these two sites can be generalised to *P. australis* populations in the Hauraki Gulf Marine Park is currently uncertain.  *P. australis* greater than 40 mm in length are sexually mature (Hooker & Creese 1995), so the *P. australis* at these sites would have been predominantly below or just at the size of sexually mature shellfish.  If the size frequency of the portion of the *P. australis* population available for harvest in the inter-tidal zone were representative of the population as a whole, then harvest pressure on this size class could result in a significant decrease in the biomass of spawning shellfish.  However, two factors suggest caution in this interpretation of the data.  Firstly, as discussed in Section 2 previously, subtidal locations may be the main habitats of adult *P. australis*.  If the main harvest activity were focussed on the larger sized shellfish in the inter-tidal zone, significant numbers of large adults could remain in the subtidal portion of the population.  Secondly, the observations by Hartill & Cryer (1999) were based on a low level of harvesting of *P. australis*, and the authors suggested that this may have been as a result of the small size of the shellfish available for harvest at the sites studied.

In a study of *A. stutchburyi* in Raglan Harbour, Connew (2000) found that human predation pressure was determined by the ease of access to a particular site.  Kearney (1999) observed that harvest activity targeting *A. stutchburyi* was patchy in distribution within Lew's Bay, Whangateau Harbour, with several sites appearing to be preferred fishing spots and receiving much more attention from harvesters than other areas.  Results indicated that areas with the highest density of large *A. stutchburyi* were not the most heavily harvested sites, with the more remote areas of harvestable *A. stutchburyi* less frequently harvested.  Interviews with harvesters suggested that factors such as site location (e.g. distance required to return catch to land), ease of collection, and preferred traits of *A. stutchburyi* in different areas (e.g. white shells, sandy substrate *vs.* muddy substrate, and proximity to water) greatly influenced the harvesters' choice of a specific site.  These data suggest that there are some portions of the stock of *A. stutchburyi* in the Whangateau Harbour that are less exposed to human harvesting activity than others.  The extent to which this applies to

infaunal shellfish beds elsewhere in the Hauraki Gulf Marine Park is unknown, but such patterns could potentially lessen the severity of the impact of harvesting on infaunal bivalve populations.

Exposure to human harvesting activity may vary between infaunal shellfish species as a result of a variety of factors, including the desirability of the species, availability and accessibility. Kearney (1999) observed that the harvesting of shellfish appears to be largely species specific, with the majority of harvesters targeting particular species. None of the studies of non-commercial harvesting activity have mentioned the harvest of *M. liliana*, even when listing secondary species harvested (e.g. Drey & Hartill 1993, Kearney 1999). Neither has gathering of *M. liliana* been observed in the study of non-commercial harvesting activity currently being undertaken by Akroyd Walshe Ltd. (K. Walshe, Akroyd Walshe Ltd, *pers. comm.*). From the limited data available, it thus appears that *M. liliana* may not be as actively targeted by non-commercial harvesters as are *A. stutchburyi, P. australis,* or *P. subtriangulata*. However, further data are required to robustly assess the exposure of *M. liliana* to non-commercial harvesting.

It appears that dense populations of shellfish may attract more harvest activity, since for example, in the Hartill & Cryer study only low levels of harvest activity were found in areas where infaunal shellfish population levels were low (Hartill & Cryer 1999). Connew (2000) found that harvest patterns of *A. stutchbury* in the Raglan Harbour were density dependent, with little harvesting occurring on low density beds. Differences in accessibility of shellfish may result in different levels of exposure to harvest activity. This includes accessibility to the coastline, and to the shellfish beds themselves. For example, those infaunal bivalves that have a sub-tidal distribution (such as *P. subtriangulata,* and to a lesser extent *P. australis*) are more difficult to collect. Marsden (2000) noted that collection of *P. subtriangulata* is normally restricted to a few days each lunar cycle during the low spring tides. Similarly, Kearney (1999) noted that on extreme spring low tides the number of people harvesting *P. australis* from Lew's Bay (Whangateau Harbour) increased considerably. The data collected by Fisher & Bradford (1998) (see Table 5.9.1) indicate that *P. australis* are a popular species for non-commercial harvesters. However, as the larger sized adults of *P. australis* may be found subtidally (see Section 2 earlier), these shellfish may be subject to less harvest pressure than those shellfish species such as *A. stutchburyi* that are more intertidal and therefore more easily accessible to harvesters.

Research suggests that shellfish gathering varies temporally. The New Zealand-wide survey by Fisher & Bradford (1998) found that January is the most popular month for hand-gathering seafood, and that more trips to hand-gather seafood are undertaken in the summer months than at other times of year (see Table 5.9.2). They also observed that public holidays that make long weekends tend to increase fishing activity (Fisher & Bradford 1998).

|  | Jan | Feb | Mar | Apr | May | June | July | Aug | Sept | Oct | Nov | Dec |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| No. of Trips | 267 | 108 | 66 | 66 | 29 | 24 | 15 | 24 | 47 | 31 | 24 | 67 |

**Table 5.9.2:  Number of trips undertaken by diarists to hand-gather seafood, by month, from a survey undertaken by Fisher & Bradford (1998).**

Similarly, in their survey Hartill & Cryer (1999) observed that the greatest harvest occurred in the summer and the lowest in winter. At Okoromai Bay, they found that cockle harvesting occurred predominantly in the weekends. In his study of non-commercial harvest of *A. stutchburyi* at Lew's Bay in the Whangateau Harbour, Kearney (1999) also observed greatest harvest activity in summer,

with lowest in winter, with higher harvests in weekends than week days, and a substantial increase in harvest during public holidays. Similar results were observed by Connew on cockle beds in the Raglan Harbour (Connew 2000). Particularly high levels of harvest activity were observed in the first two weeks in January (Kearney 1999). Larcombe (1971) observed that *A. stutchburyi* spawn in summer-autumn. If substantial harvest activity were to occur before the shellfish spawned, this could potentially impact on the numbers of gametes produced by the impacted population.

## 5.9.4     Risk Characterisation

It is apparent that the Hauraki Gulf Marine Park lies within an area of high non-commercial harvesting activity relative to other areas in New Zealand. Studies suggest that non-commercial harvesters from the Auckland metropolitan area contribute substantially to levels of harvest activity elsewhere in the Hauraki Gulf Marine Park (Fisher & Bradford 1998, Kearney 1999). Thus it cannot be assumed that harvest pressure is necessarily greatest in shellfish beds closest to high population areas such as Auckland, since a variety of other factors may impact on harvest activity (e.g. perceptions of water quality, economic reliance on shellfish as a food source etc). The majority of studies of shellfish harvesting activity have included sites predominantly within or very close to the Auckland metropolitan area, and thus it is difficult to comment definitively on geographic patterns in the risk of infaunal shellfish depletion arising from human harvesting.

Two of the species of concern, *P. australis* and *P. subtriangulata,* have adult populations of harvestable size that may be predominantly subtidal, and the location of shellfish population abundance surveys and surveys of harvesting activity (with the exception of the diary and telephone surveys by Sylvester *et al.* 1991 and Fisher & Bradford 1998) within the Hauraki Gulf Marine Park have focussed on inter-tidal areas only. None of these areas have included those types of habitat favoured by *P. subtriangulata*. This suggests that data relating to these species may not provide a good indication of either the status of the populations or related harvest activity.

Limited data suggest that non-commercial harvesting activity may vary between species, with *M. liliana* likely to be subject to less harvest activity than *A. stutchburyi, P. australis* or *P. subtriangulata.*

Human harvesting may potentially impact directly and indirectly on infaunal bivalve abundance in a number of ways (e.g. reduction in gamete production through removal of sexually mature adults, reduction in fertilisation efficiency due to reduced adult densities, reduction in juvenile settlement due to density dependent settlement, impacts (including sub-lethal impacts) relating to disturbance through harvesting activity etc). There have been relatively few studies on these interactions with respect to New Zealand species of infaunal bivalves. Gaps in our knowledge of the way in which stocks in one area impact on the abundance of shellfish in other areas also limit our understanding of the impacts of human harvesting activity. In addition, the implications of the relationship between shellfish abundance and patterns in the level of harvesting activity require further investigation. Our understanding of the way in which these dynamics are currently impacting on the structure of infaunal bivalve populations in the Hauraki Gulf Marine Park is constrained by both limitations in dose response data (i.e. how much is likely to cause what level of response), and very limited information about their exposure to harvesting activity.

There has been only one study (i.e. Hartill & Cryer 1999) relevant to the Hauraki Gulf Marine Park region in which an attempt has been made to quantitatively assess the impact of non-commercial harvesting of infaunal bivalves in terms of the sustainability of the stock. This study included only one site within the Hauraki Gulf Marine Park. The researchers concluded that the levels of *A.*

*stutchburyi* harvested at Okoromai Bay (Whangaparaoa Peninsula) were sustainable (Hartill & Cryer 1999). Data from surveys of abundance of *A. stutchburyi* at Okoromai Bay in the subsequent years (1998/99 and 1999/2000) support this finding (see Section 4, Figure 4.6). However, it would be inappropriate to conclude from the data at this one site that non-commercial harvesting of infaunal bivalves occurs at a sustainable level throughout the Hauraki Gulf Marine Park in general.

Although Cheltenham Beach has been closed to shellfish harvest since 1993, data collected by the Cheltenham Beach Caretakers group indicate that the abundance of *A. stutchburyi* has continued to decline (see Section 4, Figure 4.4). The reason for this is currently uncertain, but in-depth analysis of some of the data relating to other shellfish species collected during the same surveys could assist in elucidating this. The dearth of robust long-term population abundance data (see section 4) prevents determination of the response in shellfish abundance at other beaches that have been closed to harvesting (e.g. Eastern Beach, Long Bay and Waikawau Beach).

There are insufficient data to determine whether there are any temporal trends in non-commercial harvesting activity, or how the reduction in the legal daily bag limit from December 1999 has impacted on harvest pressure on shellfish populations. Several studies of harvesting activity have noted poor compliance with legal daily bag limits (Drey & Hartill 1998, Kearney 1999), and this increases the risk of harvesting pressure on infaunal bivalve populations. We also note that bag limits may not necessarily limit harvest pressure on shellfish beds if the number of harvesters increases, so trends in harvesting activity could be important in managing the human harvesting pressure on infaunal shellfish beds.

Management of shellfish harvesting requires knowledge of the abundance of shellfish in the beds to be harvested, along with knowledge of the population biology of the shellfish. In general, stock assessment serves as the primary means for fisheries managers to determine whether harvests will occur, how long harvest will be allowed and what size the harvest will be (McHugh 2001). To set the size of an upcoming harvest, fishery managers combine stock assessment data with some theoretical model or management goal (Ulanowicz *et al.* 1980, Malinowski & Whitlatch 1988). Generally the models require age- and size-specific mortality and fecundity data. Managers have the competing goals of allowing harvest to occur while retaining some portion of the shellfish for future harvests. Setting harvest targets that have a high probability of achieving these competing goals is improved by having a good understanding of the relative importance of the many biological processes (e.g. recruitment, predation, disturbance, critical population sizes) uncovered by studies of population dynamics and genetics (Mann *et al.* 1994; Whitlatch & Osman 1994). Our current knowledge of the status and trends in abundance of infaunal bivalve populations is limited (see Section 4), as is our understanding of the population dynamics of the infaunal bivalve species of concern. These uncertainties increase the risk associated with human harvesting of infaunal shellfish beds.

In conclusion, harvest pressure undoubtedly has the potential to impact significantly on the abundance of infaunal bivalve populations (e.g. Peterson 2002). The extent to which current harvesting activity is causing depletion of infaunal bivalve populations in the Hauraki Gulf Marine Park is difficult to assess based on available information. Populations that are easily accessible are likely to be at greater risk than those that are difficult to access. Data from Cheltenham Beach suggest that harvest pressure may not be the only stressor impacting on infaunal bivalve abundance. Limited data suggest that *M. liliana* may be less at risk than *A. stutchburyi, P. australis* or *P. subtriangulata.* Although different shellfish species may differ in their response to environmental stressors, this potential difference in harvest pressure could assist in isolating the impacts of harvesting from those of other potential stressors in areas where *M. liliana* and *A. stutchburyi* co-

occur. The survey of shellfish stocks and harvesting activity that is currently being undertaken by Akroyd Walshe Ltd. for the Ministry of Fisheries will provide some additional data to improve the assessment of the risk of human harvesting to infaunal bivalve populations in the Hauraki Gulf Marine Park. However, it is apparent that significantly more information is required to provide a robust risk characterisation. The increasing population in Auckland and elsewhere in the region of the Hauraki Gulf Marine Park is likely to increase the harvest pressure on shellfish stocks in the region in the future, and sound data will be important to ensure that this risk is appropriately managed.

## 5.9.5 References

**Adams, P. B. 1980.** Life history patterns in marine fishes and their consequences for fisheries management. *Fisheries Bulletin.* 78: 1-12.

**Akroyd, J. M., B. F. J. Manly & K. A. R. Walshe 2000.** Distribution, abundance and size frequency of pipi (*Paphies australis*), cockles (*Austrovenus stutchburyi*) and tuatua *(Paphies subtriangulata)* at eleven beaches during the 1999/2000 fishing year. Report to the Ministry of Fisheries by Akroyd Walshe Ltd., AKI 1999-01. 20pp.

**Annala, J. H., K. J. Sullivan, C. J. O'Brien & S. D. Iball 1998.** Report from the fishery assessment plenary, May 1998: stock assessments and yield estimates. Ministry of Fisheries.

**Belton, R. J. 1986.** The New Zealand clam, *Chione* (*Austrovenus*) *stutchburyi*: a study of industries based on similar resources in North America and Europe focusing on harvest, handling, resource management, and aquaculture. Winston Churchill Memorial Trust. Wellington, New Zealand. 51pp.

**Best E. 1986.** Fishing methods and devices of the Maori. V. R. Ward, Government Printer, Wellington, NZ.

**Connew, C. L. 2000.** Towards the sustainable management of the cockle population (*Austrovenus stutchburyi*) in Raglan Harbour, New Zealand. Unpubl. M.Sc. thesis. School of Technology (Environmental). University of Waikato, Hamilton, New Zealand. 124pp.

**Cryer, M. 1997.** Assessment of cockles on Snake Bank, Whangarei Harbour, for 1996. NZ Fisheries Assessment Research Document 97/2. 29pp.

**Drey, B. & B. Hartill 1993.** A qualitative survey of intertidal harvesting by amateur fishers in the Auckland Metropolitan area. Northern Fisheries Region Internal Report No. 11, MAF Fisheries North Region, Auckland. 22pp.

**Fisher, D. & E. Bradford 1998.** National marine recreational fishing survey 1996: Catch and effort results by fishing zone. Research report for Ministry of Fisheries Research Project REC9701. National Institute of Water & Atmospheric Research.

**Gardner, M. 1992.** Changes in shellfish abundance on Cheltenham Beach and in human perceptions of its significance. Report to the Royal Forest & Bird Protection Society. 24pp.

**Hartill, B. & M. Cryer 1999.** User survey of shellfish harvesting in the Auckland metropolitan area. Report for Ministry of Fisheries Research Project REC9707. National Institute of Water & Atmospheric Research. September 1999.

**Hooker, S. H. & R. G. Creese 1995.** The reproductive biology of the pipi, *Paphies australis* (Gmelin, 1790) (Bivalvia: Mesodesmatidae). I. Temporal patterns of the reproductive cycle. *Journal of Shellfish Research.* 14(1): 7-15.

**Kearney, M. B. 1999.** Ecology and management of *Austrovenus stutchburyi* in the Whangateau Harbour. Unpubl. M.Sc. thesis. School of Environmental and Marine Science, University of Auckland. 135pp.

**Kennedy, V. S., R. R. Twilley, J. A. Kleypas, J. H. Cowan Jr. & S. R. Hore 2002.** Coastal and marine ecosystems & global climate change. Potential Effects on U.S. Resources. Prepared for the PEW Centre on Global Climate Change. 52pp.

**Larcombe, M. F. 1971.** The ecology, population dynamics, and energetics of some soft shore molluscs. Unpubl. Ph.D. thesis, University of Auckland.

**McHugh, J. L. 2001.** Management of hard clam stocks *Mercenaria mercenaria*. *In:* J. N. Kraeuter & M. Castagna (Eds.) Biology of the hard clam. Developments in Aquaculture and Fisheries Scientce 31. Elsevier, New York. Pp. 631-650.

**Malinowski, S. & R. B. Whitlatch 1988.** A theoretical evaluation of shellfish resource management. *Journal of Shellfish Research.* 7: 95-100.

**Mann, R., J. S. Rainer & R. Morales-Alamo 1994.** Reproductive activity of oysters, *Crassostrea virginica* (Gmelin, 1791) in the James River, Virginia during 1987-1988. *Journal of Shellfish Research.* 13: 157-164.

**Marsden, I. D. 2000.** Variability in low tide populations of tuatua, *Paphies donacina*, in Pegasus Bay, Canterbury, New Zealand. *New Zealand Journal of Marine and Freshwater Research.* 34: 359-370.

**Martin, N. D. 1984.** *Chione (Austrovenus) stutchburyi* (Gray): Population responses to exploitation. Upubl. M.Sc. Thesis, University of Auckland, New Zealand. 75pp.

**Metaxas, A., R. E. Scheibling & C. M. Young 2002.** Estimating fertilization success in marine benthic invertebrates: A case study with the tropical sea star *Oreaster reticulates. Marine Ecology Progress Series.* 226: 87-101.

**Peterson, C. H. 2002.** Recruitment overfishing in a bivalve mollusc fishery: hard clams (*Mercenaria mercenaria*) in North Carolina. *Canadian Journal of Fisheries & Aquatic Sciences.* 59(1): 96-104.

**Stewart, M. J. & R. G. Creese 2002.** Transplants of intertidal shellfish for enhancement of depleted populations: Preliminary trials with the New Zealand little neck clam. *Journal of Shellfish Research.* 21(1): 21-27.

**Sylvester, T., D. Sharples & B. Hartill 1991.** Marine recreational fishing survey (1987). Draft report. MAF Fisheries (North). April 1991.

**Ulanowicz, R. E., W. C. Caplins & E. A. Dunnington 1980.** The forecasting of oyster harvest in central Chesapeake Bay. *Estuarine and Coastal Marine Science.* 11: 101-106.

**Walshe, K. A. R. & J. M. Akroyd 2002.** Distribution, abundance, and size frequency of pipi (*Paphies australis*), and cockles (*Austrovenus stutchburyi*) at twelve beaches during the 2001/2002 fishing year. Report for the Ministry of Fisheries, Auckland by Akroyd Walshe Ltd. October 2002. 37pp.

**Walshe, K. A. R., J. M. Akroyd & B. F. J. Manly 2002.** Distribution, abundance, and size frequency of pipi (*Paphies australis*), cockles (*Austrovenus stutchburyi*) and tuatua (*Paphies subtriangulata*) at twelve beaches during the 2000/2001 fishing year. Report for the Ministry of Fisheries, Auckland, by Akroyd Walshe Ltd. January 2002. 41pp.

**Whitlatch, R. B. & R. W. Osman 1994.** A qualitative approach to managing shellfish populations: Assessing the relative importance of trophic relationships between species. *Journal of Shellfish Research.* 13: 229-242.

## 5.10.1 Hazard Identification

Phytoplankton (sometimes called "microalgae") are tiny plants that float around in the sea, either as single cells or in chains of cells joined together. They normally form the food of suspension-feeding shellfish. A rapid increase of a particular species of phytoplankton to high numbers is commonly called an "algal bloom".

While most phytoplankton blooms are beneficial to suspension-feeding shellfish, some may cause negative impacts on shellfish populations ("harmful algal blooms" or "HAB's"). Phytoplankton blooms harmful to shellfish may be either toxic or noxious – that is, either producing specific toxins, or causing anoxia through the decay process, or in some cases, simply clogging the gills of suspension-feeding shellfish (Shumway 1990).

Among the 5,000 species of marine phytoplankton, some 300 species can at times occur in such high numbers that they obviously discolour the surface of the sea (so-called "red tides"). Sixty or so species are known to have the capacity to produce potent toxins (Hallegraeff 1995, Burkholder 1998). Most of the phytoplankton that are known to produce endotoxins (i.e. toxins retained in algal cells) are dinoflagellates, although certain benthic diatoms, notably *Pseudo-nitzschia* species, (Bacillariophyceae) are also commonly toxin producers. Some members of other taxonomic divisions are also of potential concern because they produce endotoxins or exotoxins, or because they produce copious mucilage that blankets and suffocates finfish and shellfish. These include chrysophytes (e.g. *Chrysochromulina polylepis*, *Phaeocystis* spp., *Prymnesium patelliferum*, *Prymnesium parvum*, *Chattonella antiqua*, *Chattonella marina*, *Heterosigma akashiwo*) and blue-green algae (e.g. *Nodularia spumigena*) (Burkholder 1998).

### *Toxic Algal Blooms*

Many algal toxins are of concern because of their impact on the health of human shellfish consumers, and this has been the subject of much research in recent years. However, there is an increasing realisation that algal toxins may also have significant impacts on marine organisms. Algal toxins of potential human health significance in New Zealand include the following toxins and their derivatives: saxitoxins (from some *Alexandrium* species, and *Gymnodinium catenatum*), brevetoxins (from some *Karenia* species), domoic acid (from some *Pseudo-nitzschia* species), okadaic acid and closely related dinophysistoxins (from *Dinophysis* species and some *Prorocentrum* species), pectenotoxins (from some *Dinophysis* species), and yessotoxin (from *Protoceratium reticulatum*) (Hay *et al.* 2000). Monitoring for marine biotoxins, and associated research has also identified several other potential toxins (Hay *et al.* 2000). These include gymnodimine (produced by *Karenia* species) (Mackenzie *et al.* 1996); palytoxin (produced by *Ostreopsis* species, Briggs *et al.* 1998); a novel polyether compound from *Coolia monotis* (Rhodes & Thomas, 1997); and an as yet uncharacterised toxin currently known as "Wellington Harbour toxin", which is known to be produced by *Karenia brevisulcata* (Seki *et al.* 1995; Chang 1999).

In addition, research by Dr Lesley Rhodes of Cawthron Institute has identified the presence of *Pfiesteria* species in New Zealand. *Pfiesteria* are potentially toxic dinoflagellates found in the sediments of estuaries. *Pfiesteria piscicida*, first identified in USA, exhibits more than 20 life stages and ranges in size from 5-450 microns. *Pfiesteria* is usually benign, with toxicity believed to

be due to "shifts in the environment" (Rhodes 2000). *Pfiesteria* becomes toxic in the presence of fish, and can make fish moribund in 2-3 seconds, killing healthy test fish in less than 5 minutes. When toxic, impacts on shellfish include mortality of pediveligers. However, cysts pass through the gut of adult shellfish (Rhodes 2000). *Pfiesteria shumwayae* Steidinger et Burkholder has now been identified in a number of New Zealand estuaries and has been shown to be ichthyotoxic in bioassays (Rhodes *et al.* 2002). *Pfiesteria piscicida*, a species known overseas to be toxic to shellfish, has been found in Tasman Bay. Both species are likely to be widespread in New Zealand (Dr L. Rhodes, Cawthron Institute, *pers. comm.*).

As well as effects from known toxin producing microalgae and heterotrophic dinoflagellates, the direct effects of some other toxins (e.g. toxins from marine blue-green algae – nodularin from *Nodularia spumigena*) on shellfish populations still remain poorly documented (Burkholder 1998). Toxic freshwater blue-green algae may also occur sporadically in marine coastal waters and embayments (Burkholder 1998).

Unfortunately, particularly with respect to New Zealand species of shellfish, there has been little research on the impacts of toxic phytoplankton on shellfish themselves. Most of the reports of the impacts of harmful algal blooms on inter-tidal soft-sediment shellfish in New Zealand are anecdotal. There are several events in which shellfish deaths have been associated with toxic algal blooms:

- In the summer of 1982-83, presence of the prymnesiophyte species *Prymnesium calathiferum* sp. nov., in association with a bloom of the diatom *Cerataulina pelegica* (Cleve) Hendey, caused fish and shellfish mortalities (undisclosed species) in the inshore waters of Bream Bay, Northland. *Prymnesium* species are known to produce biotoxins. (Chang & Ryan 1985; Taylor *et al.* 1985).

- There are reports of shellfish deaths in northern New Zealand in the summer of 1992-93 (Smith *et al.* 1993), some of which may have been linked to the production of toxins such as brevetoxin or "Wellington Harbour toxin" from *Karenia* species. *Karenia* species were identified in phytoplankton samples taken from the Bay of Plenty, the Hauraki Gulf and the eastern coast of Northland over the time the deaths were observed (Chang 1993; L. Mackenzie, Cawthron Institute, *Unpubl. data*, quoted in Smith *et al.* 1993). Mouse bioassays at the time indicated that a variety of shellfish from the area contained lipid soluble toxins (data from Marine Biotoxin Monitoring Programme, Ministry of Health). Brevetoxin was identified in *A. stutchburyi* from the Bay of Plenty (Ishida *et al.* 1995), and mussels (*Perna canaliculus*) from the Coromandel Peninsula (Morohashi *et al.* 1995; Murata *et al.* 1998). This event was preceded by a phytoplankton bloom that extended discontinuously from Bream Tail to the western coast of Coromandel. This bloom was dominated by raphidophytes, predominantly *Fibrocapsa japonica*, which is known to produce neurotoxins similar to brevetoxins (Khan *et al.* 1996). Some *Heterosigma akashiwo* were isolated from Coromandel samples, and *Gymnodinium* (now known as *Karenia*) species were also present (Rhodes *et al.* 1993).

- In the summer of 1998 there was a large bloom of *Karenia* species, subsequently identified as *Karenia brevisulcata* (Chang 1999). Sporadic kills of marine fauna (including paua) were reported off Wairarapa and Kaikoura, and there were massive fish and marine fauna kills in Wellington Harbour (Chang *et al.*, 1998; Tong, 1998). Staff from the NIWA Mahanga Bay Hatchery noted "dead and dying clams on Petone Beach", and dead fish, paua and starfish at Mahanga Bay. Mortality also occurred in the NIWA hatchery as a result of toxins released into the water during the bloom. Among other organisms, all *A. stutchburyi* larvae in the hatchery

died. However, *A. stutchburyi* spat survived. Adult mussels and oysters held on a raft out in Mahanga Bay also survived the bloom (Tong 1998).

- A bloom of *Karenia* species occurred in the Hauraki Gulf and east of the Coromandel Peninsula from late winter-spring in 2002. This resulted in fish kills in the Whangaparaoa area, and mortality of oysters on farms in the Mahurangi Harbour and Waiheke Island, and paua in a paua farm at Kennedy Bay (Coromandel Peninsula) (authors' *pers. obs.*; Mr C. Hunter, Waipaua Aquafarms Ltd, *pers. comm.*). Anecdotal evidence suggests that scallops outside the Mahurangi Harbour were also impacted. Observations by oyster farmers suggest that subsequent Pacific oyster spat in the Mahurangi Harbour was within the normal range of settlement density, although settlement was observed to be later than usual (Mr W. Berger, Huawai Oysters, *pers. comm.*). The impacts of this bloom on *A. stutchburyi*, *P. australis*, *P. subtriangulata* and *M. liliana* are unknown. Dr. Penny Truman of ESR, in collaboration with AquaBio Consultants Ltd, is currently working to identify the causative agent(s) and toxin(s) in this event. Dr. Hoe Chang (NIWA, Wellington) is also currently working on the identification of a previously unidentified *Karenia* species from this bloom (Dr. H. Chang, NIWA, *pers. comm.*).

There have been very few studies undertaken on the impact of marine biotoxins on New Zealand shellfish, and no studies specifically on the infaunal shellfish of interest to the Hauraki Gulf Forum in this report. A study of the persistence of paralytic shellfish poisoning (PSP) toxins (i.e. saxitoxins and their derivatives) in tuatua in the Bay of Plenty by Mackenzie *et al.* (1996) focussed on the public health implications of toxicity rather than shellfish health. However, the study showed that these toxins persisted in tuatua at low levels for 22 months after the initial contamination event (a bloom of *Alexandrium minutum*), although the toxin profiles changed within this time. This change is consistent with the high capacity for *in vivo* PSP carbamate to decarbamoyl analogue conversion in overseas species of surfclams (Cembella *et al.* 1993). As with butter clams (*Saxidomus giganteus* – a species found in North America that also retains PSP toxins for long periods of time (Quayle 1969, quoted in Mackenzie *et al.* 1996)), most of the toxin was retained in the tuatua siphons. It has been postulated that surfclams have evolved to use the retention of PSP toxins as a chemical defence mechanism against predation, possibly specifically against siphon-nipping fish (Kvitek 1993, Mackenzie *et al.* 1996). It is noted that other shellfish species (mussels, oysters and *A. stutchburyi*) from the Bay of Plenty monitored for PSP toxins over the same time period showed a comparatively rapid decline in toxin levels after the initial contamination event (Mackenzie *et al.* 1996, Hay *et al.* 2000).

In another study by New Zealand researchers, preliminary toxicity assays by Mackenzie *et al.* (1996) using larvae of the dredge oyster (*Tiostrea chilensis*) revealed some debilitating effects when these larvae were exposed to whole cells, culture filtrates and sonicated cell extracts from gymnodimine-producing *Karenia* species. These effects included the shedding of the primary swimming cilia from the velum (within 30 minutes), gradual disintegration of the velum margins, and death at between 7 and 24 hours exposure. It is possible that gymnodimine and/or other toxins produced by *Karenia selliforme* might also be toxic to larvae of other species of New Zealand bivalves, but this has not been investigated.

Mortality of adult shellfish is arguably the most obvious impact of toxic algal blooms, and has also been observed in shellfish overseas (e.g. Kim *et al.* 1995, Odebrecht *et al.* 1995). Such mortality can impact on recruitment in shellfish populations – for example, mortality of adult scallops by a toxic bloom of *Gymnodinium breve* (now called *Karenia brevis*) was identified as the cause of recruitment failure of natural populations of bay scallops (*Argopecten irradians*) in North Carolina (Summerson & Peterson 1990). However, in addition to adult mortality, chronic effects such as

stimulation of gamete resorption in reproductive adults, failure and/or delay of larval settlement and metamorphosis, mortality of early life history stages, and/or reduced growth of juveniles (Tettelbach & Wenczel 1993) may also occur and impact significantly on shellfish populations.

Landsberg (1996) suggests that the frequency of two types of cancer occurring in bivalves (disseminated neoplasia and germinomas (i.e. gonadal tumours) has increased over the previous 25 year period. Disseminated neoplasia affects about 15 species of shellfish worldwide and is common in the USA in soft shell clams (*Mya arenaria*), cockles (*Cerastoderma edule)* and blue mussels (*Mytilus edulis*). It is similar to vertebrate leukemia in that tumour cells rapidly divide, and ultimately invade connective tissue, and in advanced stages kill the host (Miosky *et al.* 1989). Landsberg (1996) also suggests a correlation between the incidence of disseminated neoplasia in some bivalve species and blooms of saxitoxin-producing *Alexandrium* species. However, bivalves that accumulate high concentrations of saxitoxin or neosaxitoxin such as the surf clam *Spisula solidissima* are apparently not affected by disseminated neoplasia or germinomas (Landsberg 1996). (Note that tuatua are similarly known to accumulate high concentrations of saxitoxins − Mackenzie *et al.* 1996, Hay *et al.* 2000). The incidence of these cancers is not correlated with blooms of *Gymnodinium catenatum*, another phytoplankton species that produces saxitoxins (Landsberg 1996).

**Table 5.10.1 below outlines chronic or sublethal impacts of harmful algae on adult (unless otherwise stated) inter-tidal shellfish (adapted from Burkholder 1998).**

| Impact: | Causative Agent: | Presence in NZ: | Affected Bivalve Species: |
|---|---|---|---|
| Narcosis (prolonged lethargy, reduced fright response, and/or reduced shell valve closure, reduced burrowing) | *Alexandrium* spp. | Yes | Various (Bricelji *et al.* 1987; Shumway & Cucci 1987) |
| | *Pfiesteria piscicida* | Yes (Dr L. Rhodes, Cawthron, *Unpubl. data*) | Northern quahog (*Mercenaria mercenaria*), (Burkholder *et al.* 1995) |
| | *Gymnodinium breve* (now called *Karenia brevis*) | Possibly (Dr H. Chang, NIWA, *pers. comm.*) | Surf clams, southern quahog, coquinas |
| Depressed or altered feeding | *Alexandrium* spp. | Yes | Various (Shumway & Cucci 1987) |
| | *Pfiesteria piscicida* | Yes (Dr L. Rhodes, Cawthron, *Unpubl. data*) | Northern quahog (*Mercenaria mercenaria*), (Burkholder *et al.* 1995) |
| | *Aureococcus anophagefferens* | | Various (Anderson 1995) |
| Haemocyte abnormalities | *Pseudo-nitzschia pungens* f. *multiseries* | Yes (now called *P. multiseries*) | Pacific oysters (*Crassostrea gigas*), (Jones *et al.* 1995) |

| | | | |
|---|---|---|---|
| Poor shell growth | *Prorocentrum minimum* | Possible component of 1992-93 bloom at Waiheke (Smith *et al.* 1993) | Eastern oyster (C*rassostrea virginica*), (Luckenbach *et al.* 1993) |
| Depressed general shell growth | *Gyrodinium aureolum* | Yes | Blue mussels (*Mytilus edulis*), (Widdows *et al.* 1979). |
| Inhibition of feeding | *Prorocentrum minimum* | See above | Post-settlement - Hard clams (*Mercenaria mercenaria*), (Wikfors & Smolowitz 1993) |
| Increased mucous production | *Alexandrium* spp. | Yes | Various molluscs (Shumway & Cucci 1987) |
| Disseminated neoplasia, germinomas | *Alexandrium tamarense, Alexandrium minutum, Alexandrium fundyense Alexandrium catenella,* other PSP dinoflagellates | Yes | Softshell clams (*Mya arenaria*), northern quahog (*Mercenaria mercenaria*), cockle (*Cerastoderma edule*), (Landsberg 1996) |
| Mantle and gill lesions | *Gyrodinium aureolum* | Yes | Eastern oyster (*Crassostrea virginica*) |
| Improper settling | *Pfiesteria piscicida* | Yes (Dr L. Rhodes, Cawthron, *Unpubl. data*) | Pediveliger - Eastern oyster (Crassostrea virginica), (Burkholder *et al.* 1995) |

### *Noxious Algal Blooms*

Non-toxic bloom-forming algae may become so densely concentrated that they generate anoxic conditions resulting in indiscriminate kills of both fish and invertebrates. Oxygen depletion can be due to high respiration by the algae (at night or in dim light during the day), but more commonly is caused by bacterial respiration during decay of the bloom (Hallegraeff 1995).

Some algal species can seriously damage the gills of marine organisms. This damage may be mechanical (such as caused by the spikes protruding from some *Chaetocerus* species), or through the production of hemolytic substances (Hallegraeff 1995). Microalgae may also cause an impact by producing mucilage that blankets and suffocates fish and shellfish (e.g. *Phaeocystis pouchetii*, Chang 1983).

## 5.10.2    Dose-Response Assessment

The Marine Biotoxin Monitoring Programme, now administered by the NZ Food Safety Authority, includes weekly monitoring of phytoplankton samples from sites in the Hauraki Gulf Marine Park, and fortnightly or monthly monitoring of biotoxin levels in shellfish from associated sites. The programme is potentially a source of data relevant to assessment of the impact of harmful algal blooms on shellfish populations. However, to date the collection of these data have not been complemented by data on the health of infaunal bivalve populations, even during harmful algal bloom events. Neither have there have been any formal studies undertaken on the impact of harmful algal blooms on infaunal bivalves found in New Zealand. There are thus no detailed dose-response data available. However, the following considerations are relevant to the dose-response assessment of soft sediment bivalves to harmful algal blooms:

Studies on the accumulation and retention of algal biotoxins by a variety of bivalve shellfish indicate that there are significant differences between shellfish species. (Hay *et al.* (2000) reviewed the available data for New Zealand shellfish species). These differences may be caused by a range of factors, including differences in feeding selectivity, toxin retention and metabolism of toxins.

Response to anoxic conditions caused by noxious algal blooms is also likely to vary between shellfish species. For example, a study by Carroll & Wells (1995) found differences between the way in which slowly burrowing bivalves such as *P. australis* and *A. stutchburyi* support anaerobic energy production compared to rapidly digging surf clams such as *P. subtriangulata*. Their study showed that rapidly digging shellfish perform better than slowly burrowing shellfish under functional anaerobiosis (such as caused by rapid digging activity), but do not fare as well at low oxygen levels. This suggests that low oxygen levels caused by algal blooms are more likely to impact on *P. subtriangulata* more severely than *P. australis* or *A. stutchburyi*. (*M. liliana* cannot be compared, as it was not included in this study).

Sensitivity to harmful algal blooms may vary with life cycle stage. For example, Tong (1998) noted that *A. stutchburyi* larvae in the NIWA Mahanga Bay hatchery exposed to toxins from the *Karenia brevisulcata* bloom (up to 33 million cells/litre) in Wellington Harbour died, while *A. stutchburyi* spat in the hatchery survived.

Dose-response to harmful algal blooms may vary with temperature. For example, the risk of mortality due to anoxia caused by phytoplankton blooms is likely to be higher at higher temperatures for a particular bloom concentration, since the dissolved oxygen content of seawater decreases as water temperature increases. For each degree Centigrade that seawater warms, oxygen solubility (the capacity to dissolve oxygen) decreases by about 2% (Najjar *et al.* 2000). In addition, the impact of toxic algal blooms may also vary with temperature due to variations in feeding rates with temperature (Lesser & Shumway, 1993).

The impacts of marine biotoxins on a bivalve species may also depend on the previous history of toxicity in their growing area. For example, a study by Bricelj *et al.* (2000) examined the magnitude and causes of intraspecific variation in sensitivity to PSP neurotoxins, and thus capacity for toxin accumulation, in North American soft shell clams, *Mya arenaria*. Individual sensitivity was measured by inhibition of clam burrowing response after laboratory exposure to a highly toxic strain of *Alexandrium tamarense*, and *in vitro* block of the action potential in isolated nerves exposed to saxitoxin. Their results suggest that *Mya* populations recurrently affected by toxic blooms may experience genetic or epigenetic adaptation to PSP toxins via natural selection of more resistant individuals.

MacQuarrie and Bricelj (2000) showed that blooms of PSP-producing dinoflagellates can cause both lethal and sub-lethal effects on the soft shell clam *Mya arenaria*, but that these effects vary in their expression both within and among populations. Shellfish from two areas, one with a history of recurrent, annual toxin events (Lepreau Basin, New Brunswick), and one with no history of toxin exposure (Lawrencetown River Estuary, Nova Scotia) were compared in their responses to PSP toxins during laboratory exposure to *Alexandrium tamarense*. Population differences in feeding and burrowing during toxin exposure were reflected in their differential ability to accumulate toxins, with clams from the area with regular toxin events accumulating up to ten times the toxin levels of those from the area with no history of toxicity. Clams from Lepreau Basin exhibited >98% survival, while those from Lawrence River Estuary suffered cumulative mortality of 32%, which started after one week of toxin exposure.

There is a possibility that intra-specific variation in dose-response to PSP toxins between shellfish populations could occur within the Hauraki Gulf Marine Park. For example, phytoplankton and shellfish monitoring data from the Ministry of Health Marine Biotoxin Monitoring Programme indicate that soft-sediment shellfish at the southern boundary on the eastern coast of the Coromandel Peninsula (for example, Waihi) have been exposed to several significant PSP-toxin events since 1993. There have been no corresponding events in the Hauraki Gulf itself. Whether the different histories of toxin events impact on dose-responses to PSP toxins of shellfish from the different areas has not been investigated.

## 5.10.3    Exposure Assessment

Globally, there are some indications that blooms of harmful marine phytoplankton may be increasing in frequency and extent (Smayda 1990, Shumway 1995). There are insufficient data to determine whether this is the case in the Hauraki Gulf Marine Park.

In some cases, the geographic expansion of known toxin producing species has been linked to introductions via ballast water exchange along shipping routes (Hallegraeff *et al.* 1990). The bloom of *Gymnodinium catenatum* on the western and south-eastern coasts of the North Island in 2000 provides an example of a harmful algal bloom of a species that had not been previously reported in New Zealand (Mackenzie & Adamson 2000). The introduction of new toxic species into the Hauraki Gulf Marine Park, especially species that produce resting cysts, would result in an increased risk of exposure of shellfish populations to harmful algal blooms in the future.

Although blooms of harmful microalgae are sometimes correlated with coastal development, responses to nutrient enrichment remain poorly understood (Burkholder 1998). Many of the toxin-producing species have complex life cycles that respond to a diverse collection of physical, chemical and trophic conditions (Burkholder 1998). In this context, some normally non-toxic species may become toxic when exposed to nutrient regimes from excessive over-enrichment (e.g. *Pseudo-nitzschia multiseries*, Bates *et al.* 1991). Some algal blooms have also been linked to El Niño weather patterns, which has led to the suggestion that global climate change and warming may also encourage their growth (Epstein *et al.* 1994).

Routine phytoplankton monitoring has occurred in the Hauraki Gulf Marine Park area as part of the Marine Biotoxin Monitoring Programme since 1997 (and at some sites within the Gulf itself since 1995). The data include not only counts of potentially toxic species, but also counts of the ten most abundant species (and at some shellfish farming sites, full phytoplankton counts of all species >10 microns in size). Analysis of these data could provide some valuable information to assist in assessment of the risk of exposure of infaunal bivalves to harmful algal blooms. However, given that patterns of toxic phytoplankton occurrence may be influenced by factors (e.g. lunar tidal modulation) which have a periodicity in excess of 18 years (White 1987), the historic data currently available represent a relatively short time period. Consequently the ecophysiology of toxic species found in New Zealand is not well understood, and there is a lack of detailed environmental data that could be used in prediction of future risk. However, some general considerations are relevant to the assessment of the risk of exposure of infaunal bivalves to harmful algal blooms:

Exposure may vary with geographical location. For example, *Alexandrium* species produce resting cysts that may lie dormant in the sediment for years until the right conditions for germination occur. There is thus a greater risk of an *Alexandrium* species bloom in areas where blooms have previously occurred. Data from the Marine Biotoxin Monitoring Programme indicate that several *Alexandrium* blooms have occurred in the Bay of Plenty, extending into the south-eastern part of the Hauraki

Gulf Marine Park. There have been no major *Alexandrium* blooms in the Hauraki Gulf itself. Thus the risk of exposure to *Alexandrium* blooms appears to be higher in the eastern Coromandel region of the Park compared to other regions. It is pertinent to note that to date there has been no record of *Gymnodinium catenatum* (another saxitoxin-producing species) in the Hauraki Gulf Marine Park. This species bloomed extensively over many months on the western coast of the North Island in 2000. There is a potential risk of introducing this species into the Hauraki Gulf Marine Park through the transfer of cysts in bilge water, ballast water, dredged sediment etc.

Data on the distribution and abundance of some potentially harmful species is poor. For example, only samples from one site (i.e. Whangamata) within the Hauraki Gulf Marine Park have been examined from the presence of *Pfiesteria* species (Dr. L. Rhodes, Cawthron Institute, *pers. comm.*). The sampling sites and methods incorporated in the Marine Biotoxin Monitoring Programme are not appropriate for detection of these species. Although they are likely to be present in estuaries in the Hauraki Gulf Marine Park (Dr. L. Rhodes, Cawthron Institute, *pers. comm.*), their distribution, abundance, and impact on infaunal bivalve populations are thus unknown at present.

Exposure may vary with the ecological habitats of the shellfish species. For example, *Pfiesteria* species flourish in estuaries where there is nutrient enrichment and poor flushing, and can lie dormant as cysts for years until nutrients trigger a bloom (Rhodes 2000). *P. australis* and *P. subtriangulata* are less likely to occur in conditions where *Pfiesteria* species might bloom than are *A. stutchburyi* or *M. liliana*.

Exposure may vary with the feeding modes of different species, (i.e. deposit-feeding *c.f.* suspension-feeding). Some toxic dinoflagellates (such as *Alexandrium* species) produce benthic resting cysts that are also toxic. It is likely that the adult stage of deposit-feeding shellfish could be more exposed to risks associated with these cysts than would suspension-feeders, but further investigation is required to establish this impact.

It is also possible that behavioural adaptations, such as burrowing behaviour or selectivity in feeding, could influence the degree of exposure to biotoxins. This has not been investigated in New Zealand species.

The risk of exposure to harmful algal blooms may vary with season, and this can impact on the shellfish population dynamics if different stages of the life cycle are impacted differently. Phytoplankton productivity is generally highest in spring and autumn. Blooms that impact as a result of high cell densities are less likely to occur in winter than in other seasons. A review of phytoplankton data from the Marine Biotoxin Monitoring Programme from the Hauraki Gulf from January 1995-July 1999 showed no seasonal trends in blooms of *Pseudo-nitzschia*, *Dinophysis* or *Karenia/Gymnodinium* species (Hay *et al.* 2000). There were insufficient occurrences of *Alexandrium* species to draw any conclusions about the seasonality of its occurrence. We note that analysis of the data for the Hauraki Gulf Marine Park area from the Marine Biotoxin Monitoring Programme from its inception to the present could provide greater insight into the risk of exposure of shellfish to marine biotoxins.

## 5.10.4    Risk Characterisation

There appears to have been no research undertaken specifically on the impact of harmful algal blooms on *A. stutchburyi*, *P. australis*, *P. subtriangulata* or *M. liliana* in New Zealand. Consequently it is difficult to assess with any certainty the risk that harmful algal blooms present to

these shellfish populations in the Hauraki Gulf Marine Park. However, the following considerations are pertinent to this risk:

- Blooms of both noxious phytoplankton, and phytoplankton potentially toxic to bivalve shellfish, have been recorded in the Hauraki Gulf Marine Park. This includes at least three blooms with potentially significant impacts over the last ten years.

- Data collected as part of the Marine Biotoxin Monitoring Programme suggest that in the HGMP region the greatest risk from potentially toxic phytoplankton blooms may arise from blooms of *Karenia* species. Blooms of potentially toxic *Alexandrium* species are more likely to occur on the south-eastern side of the Coromandel Peninsula than elsewhere in the Park. Blooms of potentially toxic and/or noxious chrysophytes have also been recorded in the Park.

- The impacts of harmful algal blooms are likely to vary according to shellfish species and life cycle stage. It is possible that early stages in the life cycle may be more sensitive than the adult stage. In the Hauraki Gulf Marine Park, observations of phytoplankton blooms that have impacted on bivalve shellfish have been based on observation of adult mortality. If earlier stages of the life cycle are more sensitive, it is possible that the actual impact of harmful algae on bivalve populations in the area may be underestimated by these reports.

- The previous history of toxic blooms in an area may impact on the sensitivity of the shellfish population to toxicity. It is unknown whether there are any such genetic differences between conspecific shellfish stocks in the Hauraki Gulf Marine Park.

- While harmful algal blooms are a natural phenomenon, human activity could increase the risk of their occurrence in the Hauraki Gulf Marine Park, for example, through nutrient enrichment (particularly of poorly flushed areas), and the transport of cysts of new toxic algal species into the area.

It is apparent from the available information that harmful algal blooms may impact on infaunal bivalves within the Hauraki Gulf Marine Park. However, at this stage the significance of this in terms of the status of infaunal bivalve stocks within the Park is basically unknown. In order to assess the risk of harmful algal blooms to infaunal bivalves in the Hauraki Gulf Marine Park with more certainty, further information is required. This includes determination of dose-response for the potentially toxic and noxious algae at each stage of the life cycle of each shellfish species. A better assessment of the risk of exposure of shellfish populations to harmful algal blooms is also required to assist in quantification of risk. Within the limits of available data, this could be obtained through analysis of phytoplankton data collected as part of the Marine Biotoxin Monitoring Programme.

Until such information is available, harmful algal blooms must be regarded as a potential risk to infaunal bivalve populations. The actual extent of their impact is at this stage unknown.

## 5.10.5   References

**Anderson, D.M. (Ed) 1995.** ECOHAB – The ecology and oceanography of harmful algal blooms, a national research agenda. Publ. by National Science Foundation and National Oceanic and Atmospheric Administration, Washington D.C. USA.

**Bates, S. S., A. S. W. de Freitas, J. E. Milley, R. Pocklington, M. A. Quilliam, J. C. Smith & J. Worms  1991.** Controls on domoic acid production by the diatom *Nitzschia pungens* f. *multiseries* in culture: nutrients and irradiance. *Canadian Journal of Fisheries and Aquatic Sciences*. 48: 1136-1144.

**Bricelji, V. M., J. Epp & R Malouf  1987.**  Intraspecific variation in reproductive and somatic growth cycles of bay scallops *Argopecten irradians*.  *Marine Ecology Progress Series.* 36: 123-137.

**Bricelj, V. M., B. M. Twarog, S. P. MacQuarrie, P. Chang & V. L. Trainer  2000.**  Does the history of toxin exposure influence bivalve population responses to PSP toxins in *Mya arenaria*? I) Burrowing and nerve responses. *Journal of Shellfish Research.* 19(1): 635.

**Briggs, L., L. Rhodes, R. Munday & N. Towers  1998.**  Detection of palytoxin using a haemolysis neutralisation assay. *Proceedings of the Marine Biotoxin Science Workshop No. 10* (20th November 1998). Pp. 91-97.

**Burkholder, J. M.  1998.**  Implications of harmful microalgae and heterotrophic dinoflagellates in management of sustainable marine fisheries. *Ecological Applications.* 8(1): S37-S62.

**Burkholder, J. M., H. B. Glasgow, Jr. & C. W. Hobbs  1995.**  Fish kills linked to a toxic ambush predator dinoflagellate: distribution and environmental conditions. *Marine Ecology Progress Series.* 124: 43-61.

**Carroll, J. L. & R.M. G. Wells  1995.**  Strategies of anaerobiosis in New Zealand infaunal bivalves: Adaptations to environmental and functional hypoxia. *NZ Journal of Marine & Freshwater Research.* 29: 137-146.

**Cembella, A. D., S. E. Shumway & N. I Lewis  1993.**  Anatomical distribution and spatio-temporal variation in paralytic shellfish toxin composition in two bivalve species from the Gulf of Maine.  *Journal of Shellfish Research.* 12: 389-403.

**Chang, F. H.  1983.**  The mucilage-producing *Phaeocystis pouchetii* (Prymnesiophyceae), cultures from the 1981 "Tasman Bay slime". *New Zealand Journal of Marine and Freshwater Research.* 17: 165-168.

**Chang, F. H.  1993.**  Toxic dinoflagellates found in New Zealand during January and February. *Aquaculture Update.* 5: 5.

**Chang, F. H. 1999** *Gymnodinium brevisulcatum* sp. nov. (Gymnodiniales, Dinophyceae), a new species isolated from the 1998 summer toxic bloom in Wellington Harbour, New Zealand. *Phycologia.* 38(5): 377-384.

**Chang, F. H., J. McKoy & M. Uddstrom  1998**. The summer 1998 *Gymnodinium* c.f. *mikimotoi* blooms on the East Coast and in Wellington Harbour of New Zealand. *Proceedings of the Marine Biotoxin Science Workshop No. 9,* 1st May 1998. Pp.37-43.

**Chang, F. H., P. Redfearn, B. Diggles, C. Glasby & S. Allen 1998.**  Some characteristics of the Wellington Harbour *Gymnodinium* toxins: impacts on marine animals and algae*. Proceedings of the Marine Biotoxin Science Workshop No. 10,* (20 November 1998). Pp. 77-83.

**Chang, F. H. & K. G. Ryan  1985.**  *Prymnesium calathiferum* sp. nov. (Prymnesiophyceae), a new species isolated from Northland, New Zealand. *Phycologia.* 24(2): 191-198.

**Epstein, P. R., T. E. Ford & R. R. Colwell  1994.**  Marine ecosystems. *In*: Health and climate change. Lancet, special issue. Pp. 14-17.

**Hallegraeff, G. M.  1995.**  Harmful algal blooms: A global overview. *In:* Manual on harmful marine microalgae. Hallegraeff, G. M., D. M. Anderson, A. D. Cembella, and H. O. Enevoldsen (Eds.)  Intergovernmental Oceanographic Commission, Manuals and Guides No. 33, UNESCO. Pp. 1-22.

**Hallegraeff, G. M., C. J. Bolch, J. Bryan & B. Koerbin  1990.**  Microalgal spores in ships' ballast water: a danger to aquaculture. *In*: Toxic Marine Phytoplankton. E. Granéli, B. Sundstrom, L. Edler and D. M. Anderson (Eds.). Elsevier, New York, U.S.A. Pp. 29-40.

**Hay, B. E., C. M. Grant & D-J. McCoubrey  2000.**  A review of the marine biotoxin monitoring programme for non-commercially harvested shellfish. Part 1: Technical Report. A report prepared for the NZ Ministry of Health by AquaBio Consultants Ltd. NZ Ministry of Health. 224 pp.

**Ishida, H., A. Nozawa, K. Totoribe, N. Muramatsu, H. Nukaya, K. Tsuji, K. Yamaguchi, T. Yasumoto, H.F. Kaspar, N. Berkett, & T. Kosuge 1995.**  Brevetoxin B1, a new polyether marine toxin from the New Zealand shellfish *Austrovenus stutchburyi. Tetrahedron Letters.* 36: 725-728.

**Jones, T. O., J. N. C. Whyte, N. G. Ginther, L. D. Townsend & G. K. Iwama 1995.**  Haemocyte changes in the Pacific oyster, *Crassostrea gigas*, caused by exposure to domoic acid in the diatom *Pseudonitzschia pungens* f. *multiseries. Toxicon.* 33: 347-353.

**Khan, S., O. Arakawa & Y. Onoue  1996.**  Neurotoxin production by a Chloromonad *Fibrocapsa japonica* (Raphidophyceae). *Journal of the World Aquaculture Society.* 27(3): 254-263.

**Kim, H. G., J. S. Park, Y. Fukuyo, K. H. An & P. C. Abreu  1995.**  Noxious dinoflagellate bloom of an undescribed species of *Gyrodinium* in Chungmu coastal waters, Korea. In: *Harmful Marine Algal Blooms.* P. Lassus, G. Arzul, E. Erard, P. Gentien and C. Marcaillou (Eds.). Technique et Documentation – Lavoisier, Intercept Ltd. Pp. 59-61.

**Kvitek, R. G.  1993.**  Paralytic shellfish toxins as a chemical defence in the Butter Clam (*Saxidomus giganteus*). In: *Toxic Phytoplankton Blooms in the Sea*. T.J. Smayda & Y Shimizu (Eds.) Elsevier Science Publishers, B.V. Amsterdam. Pp. 407-412.

**Landsberg, J. H.  1996.**  Neoplasia and biotoxins in bivalves: is there a connection? *Journal of Shellfish Research*. 15(2):203-230.

**Lessor, M. P. & S. E. Shumway  1993.**  Effects of toxic dinoflagellates on clearance rates and survival in juvenile bivalve molluscs. *Journal of Shellfish Research.* 12(2): 377-381.

**Luckenbach, M. W., K. G. Sellner, S. E. Shumway & K. Greene  1993.**  Effects of two bloom forming dinoflagellates, *Prorocentrum minimum* and *Gyrodinium uncatenum*, on the growth and survival of the eastern oyster *Crassostrea virginica* (Gmelin, 1971). *Journal of Shellfish Research.* 12(2): 411-415.

**Mackenzie, L. & J. Adamson  2000.**  The biology and toxicity of *Gymnodinium catenatum*: An overview with observations from the current bloom. *Proceedings of the Marine Biotoxin Science Workshop No. 14 (24th November 2000).*  Pp. 13-22.

**Mackenzie, L., A. Haywood, J. Adamson, P. Truman, D. Till, T. Seki, M. Satake & T. Yasumoto  1996**  Gymnodimine contamination of shellfish in New Zealand. <u>In</u>: *Harmful and toxic algal blooms.*  Yasumoto, T., Oshima, Y. & Fukuyo, Y. (Eds.).  Intergovernmental Oceanographic Commission, UNESCO, Paris, France. Pp. 97-100.

**Mackenzie, L., D. White & J. Adamson  1996.**  Temporal variation and tissue localisation of Paralytic Shellfish Toxins in the New Zealand tuatua (surfclam) *Paphies subtriangulata*. *Journal of Shellfish Research.* 15(3): 735-740.

**MacQuarrie S. P. & V. M. Bricelj  2000.**  Does the history of toxin exposure influence bivalve population responses to PSP toxins in *Mya arenaria*?: II) Feeding, survival and toxin accumulation. *Journal of Shellfish Research.* 19(1): 636-637.

**Miosky, D. L., R. M. Smolowitz & C. L. Reinisch  1989**.  Leukemia cell specific protein of the bivalve mollusc, *Mya arenaria*. *Journal of Invertebrate Pathology*.  53: 32-40.

**Morohashi, A., M. Satake, K. Murata, H. Naoki, H. F. Kaspar & T. Yasumoto  1995.**  Brevetoxin B3, a new brevetoxin analog isolate from the Greenshell mussel *Perna canaliculus* involved in neurotoxic shellfish poisoning in New Zealand. *Tetrahedron Letters.*  36: 8995-8998.

**Murata, K., M. Satake, H. Naoki, H.F. Kaspar, & T. Yasumoto  1998.**  Isolation and structure of a new brevetoxin analog, Brevetoxin B2, from Greenshell mussels from New Zealand. *Tetrahedron Letters*. 54: 735-742.

**Najjar, R.G., H. A. Walker, P. J. Anderson, E. J. Barron, R. J. Bord, J. R. Gibson, V. S. Kennedy, C. G. Knight, J. P. Megonigal, R. E. O'Connor, C. D. Polsky, N. P. Psuty, B. A. Richards, L. C. Sorenson, E. M. Steele & R. S. Swanson  2000**.  The potential impacts of climate change on the mid-Atlantic coastal region. *Climate Research*.  14: 219-233.

**Odebrecht, C., L. Rorig, V. T. Garcia & P. C. Abreu  1995.**  Shellfish mortality and a red tide event in southern Brazil. In: *Harmful Marine Algal Blooms.*  P. Lassus, G. Arzul, E. Erard, P. Gentien and C. Marcaillou (Eds.).  Technique et Documentation – Lavoisier, Intercept Ltd.  Pp. 213-218.

**Quayle, D**.  **1969.**  Paralytic Shellfish Poisoning in British Colombia. *Fisheries Research Board of Canada Bulletin*.  168: 1-68.

**Rhodes, L. L.  2000.**  *Pfiesteria* in New Zealand. *Proceedings of the Marine Biotoxin Science Workshop No. 14,* 24th November 2000, Pp. 87-89.

**Rhodes, L. L., A. J. Haywood, W. J. Ballantine & A. L. Mackenzie  1993**.  Algal blooms and climatic anomalies in north-east New Zealand August-December 1992. *New Zealand Journal of Marine & Freshwater Research.*  27: 419-430.

**Rhodes, L. L. & A. E. Thomas  1997.**  *Coolia monotis* (Dinophyceae): A toxic epiphytic microalgal species found in New Zealand. *New Zealand Journal of Marine & Freshwater Research.* 31: 139-141.

**Rhodes, L. L., J. M. Burkholder, H. B. Glasgow, P. A. Rublee, C. Allen & J. E. Adamson  2002.**  *Pfiesteria shumwayae* (Pfiesteriaceae) in New Zealand. *New Zealand Journal of Marine and Freshwater Research*. 36(3): 621-630.

**Seki, T., M. Satake, L. Mackenzie, H. F. Kaspar & T. Yasumoto  1995.**  Gymnodimine, a new marine toxin of unprecedented structure isolated from New Zealand oysters and the dinoflagellate <u>Gymnodinium</u> sp. *Tetrahedron Letters*.  36: 7093-7096.

**Shumway, S. E.  1990.**  A review of the effects of algal blooms on shellfish and aquaculture. *Journal of the World Aquaculture Society.* 21(2): 65-104.

**Shumway, S. E.  1995.**  Phycotoxin-related shellfish poisoning: bivalve molluscs are not the only vectors. *Reviews in Fisheries Science.*  3: 1-31.

**Shumway, S. E. & T. L. Cucci  1987.**  The effects of the toxic dinoflagellate *Protogonyaulax tamarensis* on the feeding and behaviour of bivalve molluscs. *Aquatic Toxicology.* 10: 9-27.

**Smayda, T. J.  1990.**  Novel and nuisance phytoplankton blooms in the sea: evidence for a global epidemic. In: *Toxic Marine Phytoplankton.*  E. Granéli, B. Sundstrom, L. Edler and D. M. Anderson (Eds.). Elsevier, New York, U.S.A.  Pp. 29-40.

**Smith, P., F.H. Chang & L. Mackenzie 1993.**  Toxic phytoplankton and algal blooms, summer 1992/93. *In*: Marine toxins and New Zealand shellfish. *The Royal Society of New Zealand, Miscellaneous Series 24.*  Pp. 11-17.

**Summerson, H. C. & C. H. Peterson  1990.**  Recruitment failure of the bay scallop, *Argopecten irradians concentricus*, during the first red tide, *Ptychodiscus brevis*, outbreak recorded in North Carolina. *Estuaries.* 13: 322-331.

**Taylor, F. J., N. J. Taylor, & J. R. Walsby  1985.**  A bloom of the planktonic diatom *Ceratium pelagica* off the coast of north-eastern New Zealand in 1983 and its contribution to an associated mortality of fish and benthic fauna. *Internationale Revue der gesamten Hydrobiologie.* 70(6): 773-795.

**Tettelbach, S. T. & P. Wenczel  1993.**  Reseeding efforts and the status of the bay scallop *Argopecten irradians* (Lamarck, 1819) populations in New York following the occurrence of "brown tide' algal blooms. *Journal of Shellfish Research*. 12: 423-431.

**Tong, L.  1998.**  Toxic dinoflagellates: Mahanga Bay experience February-April 1998. *Proceedings of the Marine Biotoxin Science Workshop No. 9,* 1st May 1998.  Pp. 44-48.

**White, A. W. 1987.**  Relationships of environmental factors to toxic dinoflagellate blooms in the Bay of Fundy. *Rapports et Proces-verbaux des Reunions Conseil International pour Exploration de la Mer*.  187: 38-46.

**Widdows, J.**, **M. N. Moore, D. M. Lowe & P. N. Salkeld  1979.**  Some effects of a dinoflagellate bloom (*Gyrodinium aeroulum*) on the mussel (*Mytilus edulis*). *Journal of the Marine Biological Association (U.K.).* 59: 522-524.

**Wikfors, G. H. & R. M. Smolowitz  1993.**  Detrimental effects of a *Prorocentrum* isolate upon hard clams and bay scallops in laboratory feeding studies.  In: *Toxic Phytoplankton Blooms in the Sea*.  T. J. Smayda & Y. Shimizu (Eds.).  Elsevier Science Publishers B. V.  Pp. 447-452.

## 5.11 STRESSOR: DISEASES/PARASITES

### 5.11.1 Hazard Identification

Disease is one of the factors that can regulate the abundance and productivity of animal populations (Kermack & McKendrick 1991a, 1991b, quoted in Hofmann *et al.* 1995).

Molluscan shellfish pathology is a relatively new field of research, and much of the work done has taken place over the last 20 years. This has largely been prompted by problems associated with high mortality in commercial shellfish culture overseas. Most of this information thus relates to commercial shellfish species outside New Zealand.

Although most organisms living in inter-tidal habitats are hosts for parasites (Thomas *et al.* 1997), scientists have paid little attention to the influence of parasites in structuring inter-tidal communities (Thomas *et al.* 1998). The definitive determination of the impacts of parasites and potential disease-causing agents on shellfish can be extremely difficult. While mortality is the most obvious result of infection or infestation, diseases and parasites may also cause sub-lethal effects (such as retarded growth, reduced fecundity, loss of condition etc.) that may be less obvious. Exposure to one pathogen or parasite may also make shellfish more susceptible to others, so causality of impacts is difficult to confirm with respect to individual agents.

In New Zealand comparatively little work has been undertaken on the diseases and parasites that impact on bivalve shellfish. However, since the first observation of the protistan parasite *Bonamia* in flat oysters (*Tiostrea chilensis*) in Foveaux Strait (Dinamani *et al.* 1987, Doonan *et al.* 1994), the commercial shellfish industry has been under passive disease surveillance (i.e. investigation of disease events as and when they occur).

*Bonamia* is arguably the most serious pathogen that has been found in bivalve shellfish in New Zealand, and has caused significant mortality in flat oysters in the Bluff oyster fishery. However, it has only been found in flat oysters located around the South Island and in Wellington Harbour (Hine 2002). To date, *Bonamia* has not been found in any other New Zealand bivalve species (Dr. B. Diggles, NIWA, *pers. comm.*), including flat oysters in the North Island. This may indicate high species specificity, since the impacted flat oyster populations have been shown to be genetically distinct from North Island flat oyster stocks (Faighil *et al.* 1999). This suggests that the risk of Bonamiasis to infaunal bivalve species such as *A. stutchburyi, P. australis, P. subtriangulata* and *M. liliana* could be low.

Other than *Bonamia*, the only potentially serious pathogen that has been identified in bivalve shellfish in New Zealand is a protistan parasite called *Perkinsus olseni*. *Perkinsus olseni* was first observed in *M. liliana* in the Kaipara Harbour in 1999 (Hine & Diggles 2002). Since then it has also been found in *A. stutchburyi* from Okura Estuary, Waitemata Harbour, Tamaki Estuary, Mangemangeroa Estuary and Wairoa Estuary in 2000 (Tricklebank & Stewart 2001). Hine (2002) also found *Perkinsus olseni* in another bivalve, *Barbatia novaezelandiae* (which lives in the low inter-tidal zone attached by byssus threads to rocky substrate) from Rangitoto Island, Waitemata Harbour. In a more extensive survey of *Perkinsus* infection in shellfish in New Zealand in early 2001, Hine & Diggles (2002) found that *Perkinsus olseni* only occurred in clams from the northeast coast of the North Island and the Kaipara Harbour. *P. olseni* was predominantly found in *A. stutchburyi* with only one *P. australis* found infected (from Onerahi, Whangarei Harbour).

---

Overseas, *Perkinsus* species have been found in a variety of infaunal bivalves including *Ruditapes philippinarum* (Choi & Park 1997), *Macoma balthica* (Kleinschuster *et al.* 1994, Coss *et al.* 2001), *Mya arenaria* (McLaughlin & Faisal 1998), and *Mercenaria mercenaria* (Coss *et al.* 1999). In several overseas studies, *Perkinsus* has been considered as the cause of clam mortalities. In Spain, *Perkinsus atlanticus* (now known as *P. olseni*) has been linked to mortality in the shallow sub-tidal Venerid clams, *Tapes decussatus* and *Ruditapes philippinarum* (Navas *et al.* 1992). *Perkinsus* was also suggested as the cause of population declines of *Ruditapes philippinarum* in Japan (Hamaguchi *et al.* 1998). Park & Choi (2001) observed that heavy *Perkinsus* infections in older *R. philippinarum* caused retarded growth, delayed gamete maturation and altered population dynamics and stability. In addition, Choi *et al.* (2002) have suggested that heavy infections of *Perkinsus* in the infaunal clam *Ruditapes philippinarum* may exert potential deleterious effects on growth and reproduction by interfering with the reproductive maturation and filtration activities of the clam. Similarly, Bower (2003) suggests that heavy infections may interfere with respiration, growth and/or survival.

While there are little data available on the impact of various levels of infection of *P. olseni* on New Zealand *A. stutchburyi*, histopathology suggests that (at least in autumn) the parasite causes only chronic inflammatory disease (Hine & Diggles 2002). (Note, however, that disease infection intensity of *P. marinus* in *Crassostrea virginica* in eastern U.S.A. follows a well defined seasonal cycle (Hofmann *et al.* 1995), and that this aspect has not been investigated in New Zealand to date).

There are limited data available on other parasites potentially impacting on infaunal bivalve populations. An ARC-funded study of parasites in *A. stutchburyi* by Tricklebank & Stewart (2001) investigated the parasite load of *A. stutchburyi* at six estuaries along the East Coast of the Auckland region (e.g. Whangateau, Okura, Waitemata, Tamaki, Mangemangeroa, and Wairoa estuaries). Parasites that were identified as occurring regularly in *A. stutchburyi* from these sites included:
- the parasitic copepod *Pseudomyicola spinosis*,
- the protozoan parasite *Perkinsus olseni* (absent from the Whangateau Harbour),
- trematodes most likely *Curtuteria australis* and *Meiogymnophallus* sp.,
- Rickettsia-like organisms in gill tissue, and
- *Trichodina* sp.

A study by Hine (2002) also identified trematode metacercariae (probably *Curtuteria australis*) in *A. stutchburyi* (predominantly in the foot region) from Mangemangeroa Estuary in 2000. Similar parasites have been found in studies undertaken elsewhere in New Zealand. For example, the trematode, *Curtuteria australis,* has also been found in *A. stutchburyi* from Otago Harbour (Mouritsen *et al.* 2003).

The level of risk associated with these bivalve parasites is difficult to evaluate. These parasites are found commonly in/on overseas bivalve species e.g. gill *Trichodina* in *Cerastoderma edule*, *Macoma balthica* and *Tapes philippinarum* (Bower *et al.* 1994); Trematodes in *Mercenaria mercenaria*, *Cerastoderma edule*, *Macoma balthica*, *Tellina tenuis* (Bower & McGladdery 2003); and Rickettsia-like organisms in *Cerastoderma edule*, *Mercenaria mercenaria*, *Tapes philippinarum* and *Tellina tenuis* (Bower 1998). It is therefore highly likely that these species will commonly occur in all of the infaunal bivalve species of interest here. Generally, at low densities these parasites are considered innocuous, however, at higher densities they may cause tissue damage and mortality (e.g. high intensities of the trematode *Cercarai cerastodermae* caused mortalities in *Cerastoderma edule* (Jonsson & Andre 1992)).

## 5.11.2    Dose-Response Assessment

*Perkinsus olseni*

The life history of *Perkinsus* species has been studied in detail by Perkins (1988), who described the structure of each life stage.  Three life stages were identified:  trophozoite (found within the bivalve hemocytes), presoosporangia, and biflagellated zoospores (which can be released into the sea).  Perkins (1988) suggested that the trophozoites are probably the normal agents of transmission, as suggested by the ease with which infections can be transmitted with infected host tissue in which there are no zoospores.  A study by Volety & Chu (1994) suggested that trophozoites of *Perkinsus marinus* are more infective than prezoosporangia, supporting the suggestion that they are possibly the principal agents of disease transmission in the field.  When Perkins exposed *Crassostrea virginica* or *C. virginica* tissue explants to millions of zoospores, only very light infections resulted (Perkins 1988).  However, *Perkinsus* zoospores are released from infected infaunal bivalves into the water column and are then filtered out of the water by healthy infaunal bivalves during normal feeding activity (Auzoux-Bordenave *et al.* 1995).  As a consequence of this filtering process the gills of infaunal bivalves appear to be the target tissues of *Perkinsus* species (Bower, 2003).  Perkins (1988) noted that zoospores appear to establish infections in the gills or mantle by penetrating the gill, labial palp, or mantle epithelium after losing their flagella and encysting within or between the cells.

There are no dose-response data available with respect to *Perkinsus olseni* for New Zealand infaunal bivalves.  The value of extrapolating from overseas data for different shellfish/*Perkinsus* species is questionable, since the virulence of the pathogen and the responses of the host may differ.  For example, Perkins (1988) noted that *"The species of Perkinsus found in Macoma balthica* (subsequently identified as *P. atlanticus*) *can cause mortality but it is not usually as virulent as P. marinus.  The host often encapsulates the parasite with connective tissue fibres and cellular elements, resulting in small isolated groups of parasitic cells in the connective tissue.  This is in contrast to P. marinus, the cells of which are distributed without encapsulation throughout almost all organs and in the hemolymph"*.  However, some data from overseas could provide an indication of the kinds of factors that may impact on dose-response.

The level of infection of *Perkinsus* in bivalves can be affected by environmental parameters such as salinity and temperature.  For example, a rapid range expansion *of Perkinsus marinus* to the north has occurred in oysters (*Crassostrea virginica*) in the U.S.A. in association with warmer winters (Cook *et al.* 1998).

A study by Paynter and Burreson (1991) provided some insight into the physiological effects of *Perkinsus marinus* on *Crassostrea virginica*.  They quantified the retardation of growth induced by infection in whole populations of oysters, showed that disease-related mortalities were low until the second year of infection, and that infection was correlated with reduced condition index.  Reduction in condition was not associated with increased mortality.  They also showed that salinity was inversely correlated with the intensity of disease development and subsequent reduction of growth rate, but not the time to infection or the effect of infection on condition index.  Similarly, in a study by Ragone and Burreson (1993) to investigate the effect of salinity on *Perkinsus marinus* in *Crassostrea virginica,* cumulative oyster mortalities progressively increased with increasing salinity.

The study by Paynter & Burreson (1991) found a positive correlation between size and infection i.e. the larger animals tended to acquire the disease sooner and more intensely.  They concluded that

---

this supported the theory that juvenile bivalves are less likely to become infected due to the limited volume of water they filter compared to large animals (Paynter & Burreson 1991).

### Other Parasites

To properly assess the impact of a parasite on its host population, a detailed knowledge of the parasite and host life cycles and population dynamics is necessary as a starting point (Davey & Gee 1988). Certainly with respect to the potential parasites observed in infaunal bivalves in the Hauraki Gulf Marine Park to date, this information is lacking.

The actual relationship between some of the observed organisms and their hosts is not clear – it varies from symbiotic (in this case defined as an association between two organisms in which the nature of the relationship is unknown (Bower *et al.* 1992)) to parasitic.

Studies by Mouritsen and co-workers (Mouritsen 2002, Mouritsen *et al.* 2003) on parasites in *A. stutchburyi* provide some useful information. The free-swimming cercariae of the parasite *Curtuteria australis* are taken in by *A. stutchburyi* via the feeding current, and encyst in the foot (Mouritsen 2002). Mouritsen (2002) suggests that this pattern of infestation may be reinforced by the fact that the tip of the foot is the most prominent obstacle as the cercariae swim around the mantle cavity. The trematode *Curtuteria australis* has been shown to influence the behaviour of *A. stutchburyi* when present at high densities (Thomas *et al.* 1998). Heavily infected *A. stutchburyi* are unable to bury normally (due to reduced growth of the foot by heavy *Curtuteria* infestations in that region), and often lie entirely exposed at the surface of the mud (Thomas *et al.* 1998). This parasite can be locally very common, with infection intensities of sometimes greater than one thousand cysts per individual (Thomas & Poulin 1998). *Austrovenus stutchburyi* with the foot found to be significantly impacted by *Curtuteria australis* generally contain about 100-1000 metacercariae per specimen (Thomas & Poulin 1998).

While there are not detailed data available regarding the other potential parasites observed in infaunal bivalves in the Hauraki Gulf Marine Park to date, some data from overseas appear to support preliminary observations made here. Following their study of parasites and symbionts in Manila clams (*Tapes philippinarum*), Bower *et al.* (1992) concluded that the wide distribution and benign host relationships of *Trichodina* species and parasitic copepods suggests that these organisms may be enzootic to Manila clams. Bower *et al.* (1992) noted that the relationships between *Trichodina* sp. and Manila clam bordered on symbiosis. However, although they observed no pathology in *Trichodina*-infested Manila clams in British Columbia, the close association between some specimens and the mantle epithelium suggested that the relationship may be parasitic. Also, mortalities attributed to *Trichodina* sp. have been reported in cockles (*Cerastoderma edule*) and oysters (*Crassostrea angulata*) in Europe (Lauckner 1983). It has been suggested that the potential for *Trichodina* sp. to become pathogenic to infaunal bivalves is probably dependent on conditions that will allow considerable increases in *Trichodina* numbers. These conditions have not been defined.

The impact of the copepod, *Pseudomyicola spinosus*, on infaunal bivalve species in the Hauraki Gulf Marine Park remains uncertain due to lack of investigation. A number of similar parasitic copepods occur in bivalves overseas (e.g. *Pseudomyicola ostreae* and *Mytilicola orientalis* both of which occur in a variety of bivalves, including Manila clams (Bower *et al.* 1992), and *Mytilicola intestinalis*, which occurs predominantly in *Mytilis* species, but has also been observed in clams *Cerastoderma edule* and *Tapes decussatus* (Davey & Gee 1988)). Bower *et al.* (1992) concluded that the long association between Manila clams and their copepod parasites without indication of

disease suggests that these parasites are not likely to be a problem in the future. Similarly, in a detailed review of the relationship between *Mytilicola intestinalis* and blue mussels (*Mytilis edulis*), Davey & Gee (1988) concluded that the relationship was more of a commensal association in which the copepod uses the bivalve to concentrate algal food and then feeds on that fraction which is not utilised by the blue mussel. They concluded that possibly *M. intestinalis* could be responsible for introducing more virulent pathogens, or that synergistic effects operate to enhance the response of infested blue mussels to such pathogens or pollutants. While these observations are not directly applicable to New Zealand species, they do indicate that the presence of a supposed parasite may not necessarily imply a significant impact on individual shellfish or on the dynamics of an infaunal bivalve population. The significance of the presence of potentially parasitic organisms in New Zealand infaunal bivalves requires further investigation.

The interaction between stress and parasitism is an area that requires further investigation. In general, it has been shown that exposure of an organism to multiple pollutants can lower its resistance to a variety of parasites. While the parasites do not kill the host, they do reduce its ability to reproduce, which could have a detrimental effect on the population over a period of time (Yevich & Barszcz 1983, quoted in Davey & Gee, 1988). There is concern that environmental stresses may increase the susceptibility of infaunal bivalves to parasites and diseases. In an attempt to elucidate this relationship, Tricklebank & Stewart are investigating the parasite load in *A. stutchburyi* with respect to the contaminant load of the surrounding sediments, sediment particle size, and water quality parameters (Tricklebank & Stewart 2001). The results of this study are currently being analysed and will be available when the final report from the study is completed.

## 5.11.3    Exposure Assessment

*Perkinsus olseni*

Transmission of *Perkinsus* species is generally reported as being from moribund hosts to neighbouring hosts i.e. lateral proximal transmission, although in an experimental situation zoospores of *Perkinsus karlssoni* were observed adhering to D-stage larvae of bay scallops approximately 48 hours post-spawning, suggesting that this could be a method of transmission between generations within a population (McGladdery *et al.* 1993).

Cross-species transmission has been demonstrated experimentally for some species of *Perkinsus*, although host-specificity appears to be the norm in the wild (Goggin *et al.* 1989 Ray 1954, both quoted in McGladdery *et al.* 1993). Note however that *Perkinsus olseni* has been found in several species of infaunal shellfish in New Zealand, so (assuming that the *Perkinsus* species has been correctly identified in each case) this does not appear to be a host-specific disease agent. The risk of cross-infection from one New Zealand infaunal bivalve species to another is currently unknown.

Also unknown is whether differences in feeding mode (suspension-feeding *versus* deposit-feeding) among New Zealand infaunal bivalves impact on the susceptibility to infection by *Perkinsus* species.

With an apparent affinity for warm water, *P. olseni* is likely to have a distribution limited to the north of the North Island of New Zealand (Hine 2002, Hine & Diggles 2002). Within the Hauraki Gulf Marine Park, a survey of *A. stutchburyi* in 2000 detected *Perkinsus olseni* in samples taken from Okura Estuary, Waitemata Harbour, Tamaki Estuary, Mangemangeroa Estuary and Wairoa Estuary in 2000 (Tricklebank & Stewart 2001). In the same study *Perkinsus* was absent from *A. stutchburyi* in the Whangateau Estuary. In a survey of shellfish health in New Zealand in 2000,

Hine (2002) detected *P. olseni* in approximately 80 % of *A. stutchburyi* from Mangemangeroa Estuary, Auckland, with infection occurring in the digestive gland, mantle and gills (Hine 2002). In the same survey, Hine (2002) also found *Perkinsus olseni* in *Barbatia novaezelandiae* (which lives in the low inter-tidal zone attached by byssus threads to rocky substrate) from Rangitoto Island, Waitemata Harbour.

In a more extensive survey of *Perkinsus* infection in shellfish in New Zealand in early 2001, Hine & Diggles (2002) found that *Perkinsus olseni* only occurred in clams from the northeast coast of the North Island and the Kaipara Harbour. *P. olseni* was predominantly found in *A. stutchburyi* with only one *P. australis* being infected. *P. olseni* was absent in clams from the Coromandel Peninsula (Coromandel, Cooks Beach, Kennedy Bay & Mercury Bay) (*A. stutchburyi* (N=200), *P. australis* (N=6) and *M. liliana* (N=21)) and the South Island of New Zealand (*A. stutchburyi* (N=574), *P. australis* (N=65) and *M. liliana* (N=2) (Hine & Diggles 2002). (Note that a sample size of 30 animals gives a 95% chance of detecting one or more infected specimens when the detectable infection rate is 10% or more in a population over 100,000 – Ossiander & Wedemeyer 1973, quoted in Morrison *et al.* 1993).

At the present time our understanding of the spatial distribution of *Perkinsus olseni* in infaunal bivalves in the Hauraki Gulf Marine Park remains incomplete due to the limited investigation undertaken to date. Hine & Diggles (2002) have suggested that *P. olseni* may be enzootic (i.e. it is continually present within a population or geographical area) in New Zealand.

Although *P. olseni* (*P. atlanticus*) may infect many bivalve genera in the tropics, in more temperate countries (e.g. Korea, Japan, and Spain) it infects only Venerid clams (Hine 2002). The NZ cockle, *Austrovenus stutchburyi,* belongs to this family. The low numbers of *P. australis* and *M. liliana* investigated to date make it difficult to identify the true prevalence of *Perkinsus* within these species. The level of occurrence of *Perkinsus* in *P. subtriangulata* is currently unknown, as to date this species has not been included in the surveys undertaken. It is also pertinent to note that potential increases in temperature due to global warming could result in an increase in the virulence of *Perskinsus olseni*, and an increase in the geographical range of species impacted.

### Other Parasites

There appear to be only limited data on the patterns of infestation of infaunal bivalve species in either the Hauraki Gulf Marine Park, or in New Zealand generally. It is expected that the study currently being completed by Tricklebank & Stewart will significantly add to this body of knowledge with respect to *A. stutchburyi*. A limited amount of data is available with respect to *A. stutchburyi*.

Because released *Curtuteria* cercariae swim very close to the surface of bottom sediments (Allison, 1979), Mouritsen (2002) suggested that infaunal bivalves are consequently exposed to the risk of infestation. It appears that infestation by parasites may vary with environmental and ecological factors. In a study of *A. stutchburyi* in Otago Harbour, Mouritsen *et al.* (2003) found individuals from a high-tide site contained a 2-fold higher parasite load than those from a low tide site. They also found an inverse relationship between the density of *A. stutchburyi* beds and the level of infestation by parasites (i.e. the more dense beds had a lower mean infection density per individual *A. stutchburyi* than more sparsely populated beds).

In conclusion, it is pertinent to note that the movement of shellfish (or other marine organisms that might be carriers of pathogens) from one area to another can increase the risk of exposure to disease

and parasites. This has been the cause of significant concern overseas, where there have been several catastrophic disease events in commercial shellfish growing areas linked circumstantially to the introduction of new shellfish species into an area, or the transfer of shellfish stock (Ford 1992). While in the absence of a reported disease outbreak there are no routine controls on the movement of shellfish throughout New Zealand, introduction of marine species from overseas is limited by legislation. This limits the risk of exposure to new diseases and parasites from overseas. There is insufficient information to evaluate the risk of introduction of new diseases and parasites through the discharge of ballast water from overseas ships.

## 5.11.4 Risk Characterisation

There have been no major disease events in infaunal bivalve populations recorded in the Hauraki Gulf Marine Park. However one potentially serious pathogen (*Perkinsus olseni*) has been observed in *A. stutchburyi* populations in the region. There is limited evidence to suggest that *P. olseni* could also infect *M. liliana* and *P. australis*, although the small numbers of these species sampled to date means that the prevalence of *P. olseni* in these species is unclear. In addition, our understanding of the spatial distribution of this pathogen is limited by lack of data. There are no data on the presence of *P. olseni* in *P. subtriangulata*.

A number of other parasitic organisms have also been recorded in infaunal bivalve populations in the Hauraki Gulf Marine Park. It is likely that all these potential pathogens, including *P. olseni*, are enzootic (i.e. continually present in the population). The chronic impacts of low level infections/infestations are very difficult to measure in terms of their overall impact on population dynamics. The impact of these potential pathogens on infaunal bivalve populations in the region is thus currently unclear.

Infestation, and/or the impacts of infestation, may vary with both the physiological state of the shellfish (for example, the stage of the reproductive cycle, seasonal changes in energy partitioning etc.), and environmental factors (such as temperature, salinity, presence of toxic contaminants etc.) (e.g. Hofmann *et al.* 1995). A sub-lethal infection could potentially result in significant mortality in shellfish exposed to other sub-lethal stressors. Thus impacts of an enzootic pathogen may vary over time, and between populations at different geographic locations.

There is a complete dearth of information about diseases or parasites that might impact on the early life cycle stages of infaunal bivalve populations in the Hauraki Gulf Marine Park.

With global warming and an increasing risk of other environmental stressors, it could be argued that disease could potentially play a greater role in contributing to the regulation of abundance and productivity in infaunal bivalves in the future. Clearly, much further investigation of potential pathogens in infaunal bivalve populations, including a good knowledge of their background levels in the population, methods of transmission and population dynamics is required to be able to robustly characterise and appropriately manage their risk to these shellfish in the Hauraki Gulf Marine Park.

## 5.11.5 References

**Allison, F. R. 1979.** Life cycle of *Curtuteria australis* n.sp. (Digenea: Echinostomatidae: Himasthlinae), intestinal parasite of the South Island pied oystercatcher. *New Zealand Journal of Zoology.* 6: 13-20**.**
**Auzoux-Bordenave, S., A. M. Vigario, F. Ruano, I. Domart-Coulon & D. Doumenc 1995.** *In vitro* sporulation of the clam pathogen *Perkinsus atlanticus* (Apicomplexa, Perkinsea) under various environmental conditions. *Journal of Shellfish Research.* 14(2): 469-475.

**Bower, S. M. 1998.** Synopsis of infectious diseases and parasites of commercially exploited shellfish: *Rickettsia*-like and Chlamydia-like organisms of clams and cockles. http://www-sci.pac.dfo-mpo.gc.ca/shelldis/pages/rlclocc_e.htm.

**Bower, S. M. 2003.** Synopsis of infectious diseases and parasites of commercially exploited shellfish: *Perkinsus* of clams and cockles. http://www-sci.pac.dfo-mpo.gc.ca/shelldis/pages/perkincc_e.htm.

**Bower, S. M., J. Blackbourn & G. R. Meyer 1992.** Parasite and symbiont fauna of Japanese Littlenecks, *Tapes philippinarum* (Adams and Reeve, 1850), in British Columbia. *Journal of Shellfish Research.* 11(1): 13-19.

**Bower, S. M. & S. E. McGladdery 2003**. Synopsis of infectious diseases and parasites of commercially exploited shellfish: trematode Metacercariae of clams and cockles. http://www-sci.pac.dfo-mpo.gc.ca/shelldis/pages/tremetcc_e.htm.

**Bower, S. M., S. E. McGladdery & I. M. Price 1994.** Synopsis of infectious diseases and parasites of commercially exploited shellfish: Gill *Trichodina* of clams and cockles. http://www-sci.pac.dfo-mpo.gc.ca/shelldis/pages/gilltrcc_e.htm.

**Choi, K.–S. & K.-I. Park 1997.** Report on the occurrence of *Perkinsus* in the Manila clam, *Ruditapes philippinarum* in Korea. *Journal of Aquaculture.* 10: 227-237.

**Choi, K.-S., K.-I. Park, K.-W. Lee & K. Matsuoka 2002.** Infection intensity, prevalence, and histopathology of *Perkinsus* sp. in the Manila clam, *Ruditapes philippinarum*, in Isahaya Bay, Japan. *Journal of Shellfish Research.* 21(1): 119-125.

**Cook, T., M. Folli, J. Klinck, S. Ford & J. Miller 1998.** The relationship between increasing sea-temperature and the northward spread of *Perkinsus marinus* (Dermo) disease epizootics in oysters. *Estuarine, Coastal Shelf Science.* 46: 587- 597.

**Coss, C. A., J. A. F. Robledo, G. R. Vasta & G. M. Ruiz 1999.** Identification of a new *Perkinsus* species isolated from *Macoma balthica* by characterisation of the ribosomal RNA locus, evidence of its presence, simultaneous with *P. marinus*, in *Crassostrea virginica*, *Macoma mitchelli* and *Mercenaria mercenaria*. *Journal of Shellfish Research.* 18: 318 (Abstract).

**Coss, C. A., J. A. F. Robledo & G. R. Vasta 2001.** Fine structure of clonally propagated *in vitro* life stages of *Perkinsus* sp. isolated from the Baltic clam *Macoma balthica*. *Journal of Eukaryotic Microbiology.* 48(1): 38-51.

**Davey, J. T. & J. M. Gee 1988.** *Mytilicola intestinalis,* a copepod parasite of blue mussels. *In*: Disease processes in marine bivalve molluscs. W. S. Fisher (Ed.). American Fisheries Society Special Publication. 18: 64-73.

**Dinamani P., P.M. Hine & J. B. Jones 1987.** Occurrence and characteristics of the haemocyte parasite *Bonamia* sp. in the New Zealand dredge oyster *Tiostrea lutaria* . *Diseases of Aquatic Organisms.* 3: 37-44.

**Doonan, I. J., H. J. Cranfield & K. Michael 1994.** Catastrophic reduction of the oyster, *Tiostrea chilensis* (Bivalvia, Ostreidae), in Foveaux Strait, New Zealand, due to infestation by the protistan *Bonamia* sp. *New Zealand Journal of Marine and Freshwater Research.* 28: 335-344.

**Faighil, D. O., B. A. Marshall, T. J. Hilbish & M. A. Pina 1999.** Trans-Pacific range extension by rafting is inferred for the flat oyster *Ostrea chilensis*. *Biological Bulletin.* 196: 122-126.

**Ford, S. E. 1992.** Avoiding the transmission of disease in commercial culture of molluscs, with special reference to *Perkinsus marinus* (Dermo) and *Haplosporidium nelsoni* (MSX). *Journal of Shellfish Research.* 11(2): 539-546.

**Goggin, C. L., K. B. Sewell & R. J. G. Lester 1989.** Cross infection experiments with Australian *Perkinsus* species. *Diseases of Aquatic Organisms*. 7: 55-59.

**Hamaguchi, M., N. Suzuki, H. Usuki & H. Ishioka 1998.** *Perkinsus* protozoan infection in short-necked clam *Tapes* (=*Ruditapes*) *philippinarum* in Japan. *Fish Pathology* (Tokyo). 33: 473-480.

**Hine, P. M. 2002.** Results of a survey on shellfish health in New Zealand in 2000. "Surveillance" magazine. MAF BioSecurity. Wellington, New Zealand. 29(1): 3-7.

**Hine, P. M. & B. K. Diggles 2002.** The distribution of *Perkinsus olseni* in New Zealand bivalve molluscs. "Surveillance" magazine. MAF BioSecurity. Wellington, New Zealand. 29(1): 8-11.

**Hofmann, E. E., E. N. Powell, J. M. Klinck & G. Saunders 1995.** Modelling diseased oyster populations I. Modelling *Perkinsus marinus* infections in oysters. *Journal of Shellfish Research.* 14(1): 121-151.

**Jonsson, P. R. & C. Andre 1992.** Mass mortality of the bivalve *Cerastoderma edule* on the Swedish west coast caused by infestation with the digenean trematode *Cercaria cerastodermae* I. *Ophelia.* 36: 151-157.

**Kermack, W. O. & A. G. McKendrick 1991a.** Contributions to the mathematical theory of epidemics – I. *Bulletin of Mathematical Biology.* 53: 33-55.

**Kermack, W. O. & A. G. McKendrick 1991b.** Contributions to the mathematical theory of epidemics – II: The problems of epidemicity. *Bulletin of Mathematical Biology.* 53: 57-87.

**Kleinschuster, S. J., F. O. Perkins, M. J. Dykstra & S. L. Swink 1994.** The *in vitro* cycle of *Perkinsus* species (Apicomplexa, Perkinsidae) isolated from *Macoma balthica* (Linneaus, 1758). *Journal of Shellfish Research.* 13: 461-465.

---

**Lauckner, G. 1983.** Diseases of Mollusca: Bivalvia. *In*: O. Kinne (Ed.), Diseases of marine animals. Volume II: Introduction, Bivalvia to Scaphopoda. Biologische Anstalt Helgoland, Hamburg. Pp. 477-961.

**McGladdery, S.E., B. C. Bradford & D. J. Scarratt 1993.** Investigations into the transmission of parasites of the Bay Scallop, *Argopecten irradians* (Lamarck, 1819), during quarantine introduction to Canadian waters. *Journal of Shellfish Research*. 12(1): 49-58.

**McLaughlin, S. M. & M. Faisal 1998.** Histopathological alterations associated with *Perkinsus* spp. infection in the soft-shell clam *Mya arenaria*. *Parasite*. 5: 263-271.

**Morrison, C. M., A. R. Moore, V. M. Marryatt & D.J. Scarratt 1993.** Disseminated sarcomas of soft-shell clams, *Mya arenaria* Linnaeus 1758, from sites in Nova Scotia and New Brunswick. *Journal of Shellfish Research*. 12(1): 65-69.

**Mouritsen, K. N. 2002.** The parasite-induced surfacing behaviour in the cockle *Austrovenus stutchburyi*: a test of an alternative hypothesis and identification of potential mechanisms. *Parasitology*. 124: 521-528.

**Mouritsen, K. N., S. McKechnie, E. Meenkan, J. L. Toynbee & R. Poulin 2003.** Spatial heterogeneity in parasite loads in the New Zealand cockle: the importance of host condition and density. *Journal of the Marine Biological Association of the UK*. 83: 307-310.

**Navas, J. I., M. C. Castillo, P. Vera & M. Ruiz-Rico 1992.** Principal parasites observed in clams*, Ruditapes decussatus* (L.), *Ruditapes philippinarum* (Adam et Reeve), *Venerupis pullastra* (Montagu) *and Venerupis aureus* (Gmelin), from the Huelva coast (S.W. Spain). *Aquaculture*. 107: 193-199.

**Ossiander, F. J. & C. Wedemeyer 1973.** Computer program for sample sizes required to determine disease incidence in fish populations. *Journal of the Fisheries Research Board Canada*. 30: 1383-1384.

**Park, K. –I. & K. -S. Choi 2001.** Spatial distribution of the protozoan parasite *Perkinsus* sp. found in Manila clams, *Ruditapes philippinarum*, in Korea. *Aquaculture*. 203: 9-22.

**Paynter, K. T. & E. M. Burreson 1991.** Effects of *Perkinsus marinus* infection in the Eastern Oyster, *Crassostrea virginica*: Disease development and impact on growth rate at different salinities. *Journal of Shellfish Research*. 10(2): 425-431.

**Perkins, F. O. 1988.** Parasite morphology, strategy, and evolution: Structure of Protistan parasites found in bivalve molluscs. *In*: Disease processes in marine bivalve molluscs. W. S. Fisher (Ed.). American Fisheries Society Special Publication. 18: 93-111.

**Ragone, L, M. & E. M. Burreson 1993.** Effect of salinity on infection progression and pathogenicity of *Perkinsus marinus* in the Easter Oyster, *Crassostrea virginica* (Gmelin). *Journal of Shellfish Research*. 12(1): 1-7.

**Ray, S. M. 1954.** Biological studies of *Dermocystidium marinum*. Rice Institute Pamphlet No. 41 (Special issue): 1-14.

**Thomas, F. & R. Poulin 1998**. Manipulation of a mollusc by a trophically transmitted parasite: convergent evolution or phylogenetic inheritance? *Parasitology*. 116: 431-436.

**Thomas, F., F. Cezilly, T. de Meeus, A. Crivelli & F. Renaud 1997.** Parasitism and ecology of wetlands: a review. *Estuaries*. 20: 646-654.

**Thomas, F., F. Renaud, T. De Meeus & R. Poulin 1998.** Manipulation of host behaviour by parasites: ecosystem engineering in the intertidal zone? *Proceedings of the Royal Society of London*. Series B. Biological Sciences. 265: 1091-1096.

**Tricklebank, K. A. & M. J. Stewart 2001.** Parasites in cockles (*Austrovenus stutchburyi*): Potential indicators of estuarine health. Unpublished Preliminary Report for the Auckland Regional Council, December 2001.

**Volety, A. K. & F. -L. E. Chu 1994.** Comparison of infectivity and pathogenicity of meront (trophozoite) and prezoosporangiae stages of the oyster pathogen *Perkinsus marinus* in Eastern oysters, *Crassostrea virginica* (Gmelin, 1791). *Journal of Shellfish Research*. 13(2): 521-527.

**Yevich, P. P. & C. A. Barszcz 1983.** Histopathology as a monitor for marine pollution. Results of the histopathologic examination of the animals collected for the U.S. 1976 mussel watch program. *Rapports et Proces-Verbaux des Reunions, Conseil International pour l'Exploration de la Mer*. 182: 96-102.

## 5.12 OTHER POTENTIAL STRESSORS

### 5.12.1 INTRODUCTION

The preceding risk assessment has identified a number of potential stressors to populations of inter-tidal infaunal bivalves in the Hauraki Gulf Marine Park. These are stressors that we consider are likely to have the most potential significance at present. In the process of identifying these potential stressors we also identified some emerging issues or potential stressors that may require consideration in the future. These are briefly discussed in the following section.

### 5.12.2 PHARMACEUTICALS AND PERSONAL CARE PRODUCTS (PPCPs)

A diverse group of bioactive compounds that has received relatively little research attention as potential environmental pollutants includes pharmaceuticals and active ingredients in personal care products, both human and veterinary, including not just prescription drugs and biologics, but also diagnostic agents, nutraceuticals, fragrances, sun-screen agents, and numerous others (Daughton & Ternes 1999). Comprehensive reviews of the current state of knowledge regarding PPCPs as environmental pollutants are provided by Daughton (2001) and Daughton & Ternes (1999). A web-site established by the US Environmental Protection Agency also provides comprehensive documentation of available information (Daughton 2000).

PPCPs and their bioactive metabolites can be continually discharged into the marine environment as complex mixtures. Although it can be assumed that the occurrence of PPCPs in waters is not a new phenomenon, awareness of their presence in aquatic environments has increased over the last decade due to improved methods of detection (Daughton 2001). As more information is able to be gathered, there are increasing concerns about the potential impact of releasing PPCPs into the environment (Daughton 2000).

PPCPs may be released directly into the environment through disposal and wastage from external application, and indirectly through excretion, washing and swimming. They may enter the marine environment *via* treated and untreated sewage effluent or *via* terrestrial runoff carrying agriculturally applied antimicrobials or excreta from medicated stock or pets (Daughton 2001). In addition, the potential use of pharmaceuticals as marine antifouling agents is under investigation (e.g. Rittschof *et al.* 2003).

There is little information available about the levels of PPCPs occurring in the marine environment. In overseas studies, only limited subsets of the large spectrum of commercial PPCPs have been documented in the environment so far, and many therapeutic classes have not yet been subject to survey (Daughton 2001). It is important to note that the spectrum of PPCPs identified in one region or country can differ from those in another as a result of prescribing and usage patterns. We are unaware of any work of a similar nature being undertaken in New Zealand.

The most well-established concerns regarding environmental release of PPCPs relate to the promotion of pathogenic resistance to antimicrobials, the reproductive/developmental effects of hormones, and the ubiquitous persistence of musks (we note that bioaccumulation of musk has been observed in the mussel *Mytilus edulis* (Geyer *et al.,* quoted in Daughton & Ternes 1999)). The potential ecological concerns regarding other impacts of these, and of other classes of PPCPs, are

ill-defined (Daughton 2001). Specifically, there appears to have been little work undertaken to investigate their potential impacts on bivalves.

Many PPCPs are bioactive compounds that act at very low concentrations. For most of these compounds, the impacts on non-target organisms are largely unknown. Non-target organisms can have receptors or receptor tissue distributions that do not exist in target organisms, and therefore unexpected effects can result from unintentional exposure (Daughton & Ternes 1999).

The potential for selective serotonin reuptake inhibitors (SSRIs), which are a major class of widely prescribed antidepressants that include Prozac, Zoloft, Luvox and Paxil, to exert subtle effects on bivalves has been noted (Daughton & Ternes 1999). Serotonin is a biogenic amine common in both vertebrate and invertebrate nervous systems. In bivalves, serotonin regulates reproductive functions including spawning, oocyte maturation, and parturition, along with control of a range of other functions in molluscs such as heart-beat rhythm, feeding, beating of cilia, swimming motor patterns, and induction of larval metamorphosis. SSRIs increase serotonin neurotransmission by inhibiting its reuptake at the synapses through inhibition of the transporter enzymes. It is well known that serotonin at concentrations of $10^{-4}$ M to $10^{-3}$ M (i.e. approximately 0.18-1.8 g/L) induces spawning in bivalves (Daughton & Ternes 1999). However, experiments with zebra mussels by Fong (1998) found that Prozac (fluoxetine) and Luvox (fluvoxamine) were extremely potent inducers, eliciting spawning behaviour at aqueous concentrations many orders of magnitude lower than serotonin. For example, fluvoxamine (the most potent in this trial) elicited significant spawning in male mussels at $10^{-9}$ M (i.e. approx. 0.318 µg/L). Another study by Fong *et al.* (1998) showed that fluvoxamine induced significant parturition in fingernail clams at 1 nM concentration. These studies suggest that if low concentrations of SSRIs were present in the marine environment, there is potential for impacts on the reproduction of bivalves to occur.

Although concentrations of individual compounds in the marine environment may be low, additive effects may result in significant impacts. For therapeutic classes comprising multiple members (SSRIs, calcium channel blockers and anticholinergics being three examples), even if the individual concentrations of each drug were low in the aquatic environment, the combined concentrations of all members sharing a common mode of action could prove significant.

While some PPCPs may have short half-lives in the marine environment, their continuous input into the marine environment can result in life-long multi-generational exposure of aquatic organisms. Continual exposure can add an extra dimension to the exposure dose required for eliciting an effect (by reducing it even further) (Daughton 2001).

Ecological risk assessments of PPCPs are currently difficult because little-to-no aquatic hazard and exposure information exists in the peer-reviewed literature for most of these compounds. The full extent, magnitude and ramifications of the presence of PPCPs in the aquatic environment are thus largely unknown (Daughton 2001). Specifically there appear to be no data on the occurrence of PPCPs in the marine environment in the Hauraki Gulf Marine Park, or of the potential impact on the infaunal bivalves on interest in this study.

## 5.12.3   BROMINATED FLAME RETARDANTS

Brominated flame retardants (BFRs) are substances used in plastics, textiles (soft furnishings), electronic circuitry, and other fire prevention materials (Gustafsson *et al.* 1999, de Wit 2002). Bromine itself is a highly reactive element, a halogen in the same class as chlorine and iodine. The most frequently used brominated flame retardants are tetrabromobisphenol-A (TBBPA),

hexabromocyclododecane (HBCD), poly-brominated biphenyls (PBBs) and polybrominated diphenyl ethers (PBDEs) (de Boer *et al.* 1998). There are theoretically 209 PBDE congeners (de Wit 2002).

Some brominated flame retardants are additives mixed into polymers, and are not chemically bound to the plastic or textiles, and therefore may separate or leach from the surface of their product applications into the environment. Additives include PBBs, PBDEs and HBCDs. Others, such as TBBPA, are reactive and are bound to the material chemically. However, some of the reactive flame retardant may not have polymerised and may be released to the environment (de Wit 2002).

Brominated flame retardants enter the environment during production processes or through leaching from discarded electrical and other products (Joyce 2003). Additive flame retardants (i.e. those that have been added into other materials) can easily leak out of treated materials during the entire life cycle of the product (this includes leaching from landfills), causing diffuse contamination of the surrounding environment (Bergman 1989). BFRs may also be associated with sewage sludge (e.g. Hale *et al.* 2001).

Overseas time trend studies all indicate increasing levels of PBDEs in the environment since the 1970's. However, little data are available outside of northern Europe and Japan (de Wit 2002). There has been no environmental monitoring for brominated flame retardants in the marine environment in the Hauraki Gulf Marine Park.

It has been suggested that BFR compounds behave similarly to PCBs and DDT in the marine environment (de Boer *et al.* 1998). PBDEs have high binding affinity to particles and a tendency to accumulate in sediments (de Wit 2002). De Wit (2002) suggested that the presence of lower brominated PBDE and HBCDs in fish indicates their bioavailability from sediment. Because PBBs and PBDEs are highly lipophilic (fat-loving) and resistant to degradative processes, they are expected to bioaccumulate easily into marine organisms (de Boer *et al.* 1998). The partitioning of BFRs between the water column and sediments in the NZ marine environment is currently unknown.

Internationally there is some concern at the potential impact of BFRs on marine ecosystems. TBBPA and PBDEs and/or their metabolites have been shown to be biologically active (de Wit 2002). To date the impact of BFRs on NZ shellfish species has not been investigated and only limited work has been undertaken overseas. However, BFRs have been identified in shellfish species overseas (e.g. in Norway, Joyce 2003). A study by Gustafsson and co-workers (1999) showed that PBDEs could be bioaccumulated from seawater by blue mussels (*Mytilus edulis*).

Very little is known about the active metabolism of PBDEs in bivalves (Gustafsson *et al.* 1999) and the associated toxicological impacts. In addition, the mode of uptake of BFRs by infaunal inter-tidal bivalves is currently not known. Possible uptake mechanisms include uptake from the water column *via* the gills, and ingestion of contaminated phytoplankton and sediment particles.

Given the paucity of information available, both with regard to levels of BFRs in the NZ marine environment and bivalve ecotoxicology data, it is not currently possible to evaluate the level of risk posed by brominated flame retardants in the marine environment in the HGMP. However, given that these compounds may be increasing in the marine environment, this issue may warrant further investigation.

## 5.12.4 INVASIVE SPECIES

Most marine invasions around the world occur in estuarine environments and sheltered bays (Chiswell *et al*. 2002). Non-indigenous species are increasingly being recognised for their impacts on marine and estuarine communities, causing significant changes in abundance and distribution of native species (Walton *et al*. 2002). One impact of these non-indigenous species may be competition with native infaunal bivalve species for limited resources (e.g. food, space).

Foreign marine bivalves introduced into Hauraki Gulf Marine Park waters in the 1960s and 1970s include *Theora lubrica*, *Limaria orientalis*, *Musculista senhousia* and *Crassostrea gigas* (Willan 1985, Hayward *et al*. 1997, Cranfield *et al*. 1998). Of these introduced species, *Crassostrea gigas* is not likely to compete for resources with any of the infaunal species of interest in this study as it occupies a different habitat niche, living attached to rocky shores/substrates. The Asian mussel *Musculista senhousia* may primarily compete for space with the infaunal bivalve species of interest in this review. *M. senhousia* lives on sandy inter-tidal and shallow subtidal sediments, in mats of byssal threads which accumulate mud. A study by Creese *et al*. (1997) found that densities of other New Zealand infauna, especially bivalves, were on average 8 times more abundant in control patches (no *Musculista*) than within *Musculista* patches. Species most affected were suspension-feeders such as pipi (*Paphies australis*), cockles (*Austrovenus stutchburyi*) and black mussels (*Xenostrobus pulex*). However, the study found that *Musculista* patches persist for only 1-2 years, and it was thus concluded that adverse effects may be short-term and localised (Creese *et al*. 1997).

The introduction of invasive species into the New Zealand marine environment *via* ships' ballast water or hulls is of great concern (Chiswell *et al*. 2002). The main factor controlling the spread of an invasive species along the coastline of NZ is the local coastal currents (Chiswell *et al*. 2002). NIWA has assessed the risk of introduction of invasive species *via* ballast water release based on flow modelling of waters inside the 12-mile Territorial Sea (Chiswell *et al*. 2002). The results of this modelling are presented in Figure 5.12.1.

From Figure 5.12.1 it is clear that the entire Hauraki Gulf Marine Park is situated in an area where the risk associated with invasive species is likely to be high.
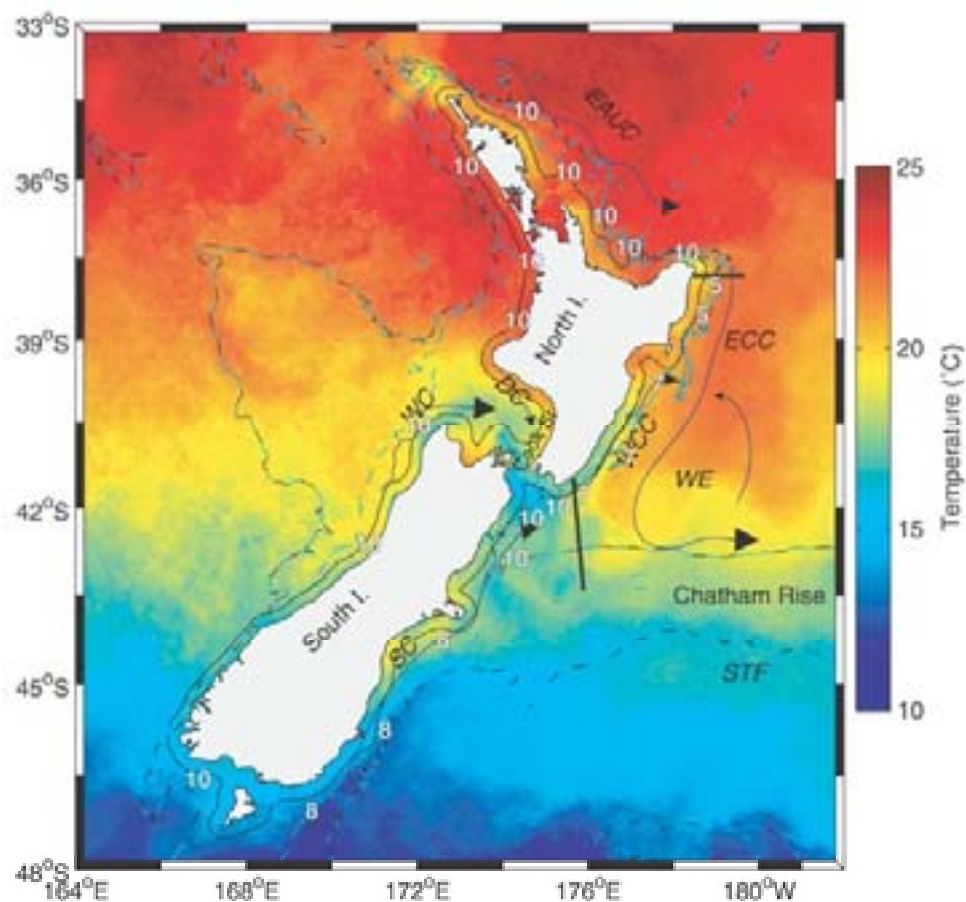
**Figure 5.12.1: Map showing the level of risk of invasive species impacting the NZ coastal environment from the discharge of ballast water. The numbers adjacent to the coastline are a measure of relative risk. 0 is no risk to the environment, and 10 is almost certain environmental impact if the discharge contains invasive species that can adapt to the local environment.**

## 5.12.5    References

**Bergman, A.  1989.**  Brominated flame retardants in a global environmental perspective.  Proceedings, Workshop on Brominated Aromatic Flame Retardants, Skokloaster, Sweden, October 24-26, Pp. 13-23.

**Chiswell, S., G. Inglis & B. Hayden  2002.**  Minimising the impact of ballast water.  Aquatic Biodiversity & Biosecurity.  1: 4.

**Cranfield, H. J., D. P. Gordon, R. C. Willan, B. A Marshall, C. N. Battershill, M. P. Francis, W. A. Nelson, C. J. Glasby & G. B. Read  1998.**  Adventive marine species in New Zealand.  *NIWA Technical Report 34.*  48pp.

**Creese, R., S. Hooker, S. De Luca & Y. Wharton  1997.**  Ecology and environmental impact *of Musculista senhousia* (Mollusca: Bivalvia: Mytilidae) in Tamaki Estuary, Auckland, New Zealand.  *New Zealand Journal of Marine and Freshwater Research.*  31(2): 225-236.

**Daughton, C. G.  2000.**  Pharmaceuticals and personal care products (PPCPs) as environmental pollutants: Pollution from personal actions, activities and behaviours.  US Environmental Protection Agency, Las Vegas, Nevada, a web-site created April 2000, available http://www.epa.gov/nelesd1/chemistry/pharma/index.htm.

**Daughton, C. G.  2001.**  Pharmaceuticals in the environment: Overarching issues and overview.  *In*: Pharmaceuticals and personal care products in the environment: Scientific and regulatory issues.  C. G. Daughton & T. Jones-Lepp (Eds.).  *Symposium Series 791.*  American Chemical Society, Washington, D. C.  Pp. 2-38.

**Daughton, C. G. & T. A. Ternes  1999.**  Pharmaceuticals and personal care products in the environment: Agents of subtle change?  *Environmental Health Perspectives.*  107(Suppl. 6): 907-938.

**de Boer, J., P. G. Wester, H. J. C. Klamer, W. E. Lewis & J. P. Boon  1998.**  Do flame retardants threaten ocean life?  *Nature.*  394: 28-29.

de Wit, C. A.  **2002.**  An overview of brominated flame retardants in the environment.  *Chemosphere*.  46: 583-624.

**Fong, P. P.  1998.**  Zebra mussel spawning is induced in low concentrations of putative serotonin re-uptake inhibitors.  *Biological Bulletin*. 194: 143-149.

**Fong, P. P., P. T.  Huminski, & L. M. d'Urso  1998.**  Induction of potentiation of parturition in fingernail clams (*Sphaerium striatinum*) by selective serotonin re-uptake inhibitors (SSRIs).  *Journal of Experimental Zoology*. 280(3): 260-264.

**Gustafsson, K., M. Bjork, S. Burreau & M. Gilek  1999.**  Bioaccumulation kinetics of brominated flame retardants (Polybrominated Diphenyl Ethers) in blue mussels (*Mytilus edulis)*.  *Environmental Toxicology and Chemistry*. 18(6): 1218-1224.

**Hale, R. C., M. J. La Guardia, E. P. Harvey, M. O. Gaylor, T. M. Mainor & W. H. Duff  2001**.  Persistent pollutants in land-applied sludges. *Nature*.  412: 140-141.

**Hayward, B. W., A. B. Stephenson, M. Morley, J. L. Riley & H. R. Grenfell  1997.**  Faunal changes in the Waitemata Harbour sediments, 1930s-1990s. *Journal of the Royal Society of New Zealand*.  27(1): 1-20.

**Joyce, S.  2003.**  Norwegian government may impose new controls on brominated flame retardants.  *International Environment Reporter*.  26(6): 284-285.  March 2003.

**Rittschof, D., C.-H. Lai, L.-M. Kok & L.-M. Teo Serena  2003.**  Pharmaceuticals as antifoulants: Concept and principles. *Biofouling*.  19: 207-212.

**Walton, W. C., C. MacKinnon, L. F. Rodriguez, C. Proctor & G. M. Ruiz  2002.**  Effect of an invasive crab upon a marine fishery: green crab, *Carcinus maenas*, predation upon a venerid clam *Katelysia scalarina*, in Tasmania (Australia). *Journal of Experimental Marine Biology and Ecology*.  272(2): 171-189.

**Willan, R. C.  1985.**  Successful establishment of the Asian mussel *Musculista senhousia* in New Zealand.  *Records of the Auckland Institute & Museum*.  22: 85-96.

## SECTION 6:     DISCUSSION

In very broad overview the potential risks to infaunal bivalves in the Hauraki Gulf Marine Park are characterised by:

- A general trend of increasing risk with increasing proximity to metropolitan areas, particularly the large metropolitan area of Auckland. This is matched by a higher level of information about the status of shellfish populations and potential anthropogenic contaminants in areas closer to Auckland.
- A general trend of increasing risk with increasing urbanisation of the coastline.
- Within these trends, there is a general trend of increased risk of anthropogenic contaminants in inter-tidal zones in enclosed estuarine areas (i.e. depositional areas) as opposed to open coastal environments.
- Risk may vary between bivalve species as a result of physiological and behavioural differences.
- Many potential stressors may impact on several life cycle stages of the shellfish. In instances where data are available, there is a general trend (with some exceptions) of higher sensitivity to potential stressors in larval and juvenile stages than in adults.
- Based on the limited information available, it appears that while many of the potential stressors considered are known to have lethal impacts at high "doses", there are more potential sub-lethal impacts that occur at lower doses. These include impacts that could result in reduced shellfish production (e.g. reduction in reproductive condition). The level of information relating to these long term lower dose effects is minimal.
- While acknowledging that there are significant knowledge gaps, in general it appears that the occurrence of potential stressors at high levels are relatively rare and localised. (An exception to this could be harmful algal blooms, which can have relatively widespread impacts). The occurrence of sub-lethal stressors is likely to be more common.
- Most potential stressors associated with human activity are predicted to have a temporal trend of increasing risk (exceptions to this are the risks associated with organotin compounds, lead, and most organochlorines). The immediacy of the risk varies between potential stressors.

The preceding risk analysis highlights the limits to our knowledge of the way in which potential stressors may impact on infaunal bivalve populations in the Hauraki Gulf Marine Park. Knowledge gaps exist across all levels of the research and risk assessment process. In general, paucity of data, particularly about potential impacts on New Zealand species of infaunal bivalves, severely limits robust risk characterisation, and this is an issue that needs to be addressed with respect to each potential stressor. However, it is also important to note several other limitations to the risk assessment process used.

Firstly, the risk assessment process, which attempts to assess the risk of individual potential stressors, has not addressed the issue of the potential impact of combinations of stressors. The interaction of combined stressors may have very significant impacts. A dramatic illustration is provided by an example from a study by Kuralec ((1992), quoted in Daughton & Ternes 1999) of the combined impact of two pharmaceutical compounds, verapamil and 2-aminoanthracene, on fish. When added together to polluted waters at concentrations that would not individually elicit any response from fish (1 ppm and 0.53 $\mu M$ respectively), the fish exhibited dramatic avoidance attempts, escalating to the point of frantic escape attempts. There have been several studies undertaken that examine the impact of sediments contaminated with a range of potential stressors on New Zealand infaunal bivalves: De Luca-Abbott (2001) identified that biochemical indices

(adenylate energy charge (AEC), total adenylate nucleotide pool (TANP), glycogen & RNA) were influenced by sublethal stress when *A. stutchburyi* were transplanted into contaminated sediments (contaminants included PAHs, heavy metals and organochlorines). The detection of these stress responses suggests that reproductive potential and growth of *A. stutchburyi* may be reduced in contaminated sediments (De Luca-Abbott 2001). In another study, transplants of *A. stutchburyi* between control/clean (Whangateau Harbour) and contaminated sediments (Tamaki Estuary-contaminated with heavy metals & PAHs) have provided preliminary results which suggest that at some contaminated sediment sites *A. stutchburyi* mortality (over 60 or 90 days) was exceptionally high (Stewart 2002). These preliminary results suggest that contaminated habitat affects *A. stutchburyi* survival in the Tamaki Estuary. Further data from this study may also provide information on contaminant body burden, which may then be related to condition indices of these transplants (M. Stewart, *work in progress*). In another current study, Tricklebank & Stewart are investigating the parasite load in *A. stutchburyi* with respect to the contaminant load of the surrounding sediments, sediment particle size, and water quality parameters (Tricklebank & Stewart 2001). This will provide very valuable data in the future.

Secondly, the dose-response data tend to relate to impacts on individuals within a population, and exposure data (where available) relates to stocks within a population. In order to predict the impact of these stressors on infaunal bivalve populations, a good understanding of the population dynamics and population genetics (i.e. the way in which the various stocks or beds within the population are inter-linked) is required.

In addition, particularly with respect to anthropogenic contaminants, much of the dose response data available are based on obvious and easily measurable impacts on organisms (such as mortality). The significance of subtle and cumulative impacts is only beginning to be recognised by environmental toxicologists. A major issue yet to be addressed by ecotoxicological science (and of potentially critical importance for risk assessment) is the impact on non-target species of stressors eliciting effects (perhaps *via* low but continual concentrations) sufficiently subtle as to go unnoticed in real time – but whose cumulative impacts eventually yield recognizable outcomes having no obvious cause (Daughton & Ternes 1999). Daughton & Ternes (1999) further note that "*Abnormal behaviour can masquerade as seemingly normal deviation within a natural statistical variation. Change can occur so slowly that it appears to result from natural events, with no reason to presume artificial causation. It is difficult to connect the issues of cause and ultimate effect, in part because of the ambiguous and subjective nature of subtle effects, but especially when these effects are confounded as aggregations of numerous, unrelated interactions*". This suggests that it may be difficult to link individual sub-lethal impacts to population declines, but that the combination of an array of sub-lethal impacts may result in a significant impact at the population level over the long term. For this reason, it may be important to broaden the temporal scale of our approach to the potential problem of shellfish depletion in the Hauraki Gulf Marine Park (i.e. examine data over a longer time period), both with respect to the determination of trends in shellfish abundance, and research associated with identification of the impacts of potential stressors.

We also note that the volume of data relating to the Coromandel Peninsula region is limited, with areas closer to Auckland having been subject to a greater level of investigation. Because of differences in catchment use between the different regions, it is difficult to generalise risk assessments from areas such as the Auckland metropolitan region to the Coromandel region.

If one can assume that significant, possibly localised depletion of infaunal bivalve populations is occurring within the Hauraki Gulf Marine Park, then consideration of the assessment of the relative risks of potential stressors, including the limitations of that process, does not offer any easy answers

as to the probable causes of such depletion.  At this stage, the extent to which the majority of individual shellfish stocks are impacted is unknown, and without this information it is difficult to develop any meaningful hypothesis about which potential stressors are likely to be those that are the most significant.  We are even unable to robustly identify whether or not shellfish depletion occurs in patterns resembling the broad patterns of risk identified in the risk assessment process (see bullet points above).  We note that the ubiquity of a potential stressor does not necessarily correlate directly with the significance of its impact – for example, unlike some of the anthropogenic contaminants, pressure from human harvesting is likely to occur throughout the shellfish beds in the Hauraki Gulf Marine Park.  However, despite long-standing closures to harvesting in some areas (such as Cheltenham Beach), infaunal shellfish populations have continued to decline.

At present our knowledge of early life cycle stage mortality of all the infaunal bivalve species of interest in this review is minimal, and this limits our ability to detect the impact of potential stressors.  Yet it is impacts on these stages of the life cycle that have the greatest potential flow-on effects in terms of future population abundance.

It is apparent that the patterns of depletion of infaunal bivalve populations in the Hauraki Gulf Marine Park are not clearly defined at present, either with respect to geographic extent or severity.  However, despite the lack of conclusive studies to indicate that the problem is widespread, there is sufficient local anecdotal evidence to warrant further investigation of this issue.  We note that in the international literature there are suggestions of declining recruitment of some similar infaunal bivalve species, for example, with respect to *Mya arenaria* (Vassiliev *et al.* 1999) and *Mercenaria mercenaria* (Peterson 2002- data set covering 11 years).

In the discussion above, we have suggested that some of the changes that might occur in infaunal bivalve populations in the Hauraki Gulf Marine Park could be slow and subtle.  However, this does not detract from the immediacy of the problem of shellfish depletion.  Due to the measurable decline, and low numbers of shellfish remaining in some stocks, along with anecdotal evidence that suggests significant declines, we suggest that there is a measure of urgency to at least elucidate the extent of the depletion of infaunal bivalve shellfish population in the Hauraki Gulf Marine Park.

## References:

**Daughton, C. G. & T. A. Ternes  1999.**  Pharmaceuticals and personal care products in the environment: Agents of subtle change? *Environmental Health Perspectives*.  107(Suppl. 6): 907-938.

**De Luca-Abbott, S.  2001.**  Biomarkers of sublethal stress in the soft-sediment bivalve *Austrovenus stutchburyi* exposed in-situ to contaminated sediment in an urban New Zealand Harbour.  *Marine Pollution Bulletin*.  42(10): 817-825.

**Kuralec, B.  1992.**  The multixenobiotic resistance mechanism in aquatic organisms. *Critical Reviews in Toxicology*.  22(1): 23-43.

**Peterson, C. H.  2002.**  Recruitment overfishing in a bivalve mollusc fishery: hard clams (*Mercenaria mercenaria*) in North Carolina. *Canadian Journal of Fisheries & Aquatic Sciences*.  59(1): 96-104.

**Stewart, M. J.  2002.**  Impacts of urban development on cockle populations.  Interim progress report to ARC.  Unpubl. report to Auckland Regional Council.  July 2002.  24pp.

**Tricklebank, K. A. & M. J. Stewart  2001.**  Parasites in cockles (*Austrovenus stutchburyi*): Potential indicators of estuarine health.  Unpublished Preliminary Report for the Auckland Regional Council, December 2001.

**Vassiliev, T., W. Congleton, B. Beal & S. Fegley  1999.**  An investigation of *Mya arenaria* (soft-shell clam) recruitment in Maine. *Journal of Shellfish Research*.  18(1): 311-312.

## *SECTION 7:      CONCLUSION AND RECOMMENDATIONS*

It is apparent that there are a range of questions that require answers in order to elucidate the issue of the depletion of infaunal bivalve populations in the Hauraki Gulf Marine Park.  These include:

- *Is shellfish depletion occurring?  How much and where?*

- *Is depletion occurring as part of species succession due to broad scale impacts or are the impacts specific to particular infaunal bivalve species?*

- *What is causing depletion at a local level?*

- *What are the impacts of localised depletion on stocks in a wider area?  i.e. how interlinked are the various stocks?*

- *What is the appropriate response to manage this risk?*

In addition to many knowledge gaps identified in relation to the impact of specific potential stressors on individual shellfish, several broad issues requiring research were highlighted as a result of our review:

- There is a paucity of robust long-term data required to identify trends in infaunal bivalve populations in the Hauraki Gulf Marine Park.  In particular, if meaningful data are to be collected on sub-tidal species such as *P. subtriangulata* (and in part, *P. australis*), surveys should be conducted in appropriate locations.  Also important is the geographic spread of data, and collection of data from a variety of sites that differ with respect to hypothesised risk levels associated with potential stressors.  In addition, it would be useful to monitor infaunal bivalve populations when exposure to potential stressors changes (such as the closure of shellfish beds to harvesting), so that the relative impact of such stressors can be assessed.
- Even when information is available to allow evaluation of the level of a particular anthropogenic contaminant in the marine environment, the associated knowledge necessary to convert this environmental level into the amount that is actually biologically available to infaunal bivalves is often lacking.
- Little work has been undertaken on the interaction between different potential stressors, particularly stressors that may have sub-lethal impacts.  The potential for synergistic effects between different potential stressors needs to be investigated further.
- The paucity of knowledge about factors driving population dynamics in each of the infaunal bivalve species of interest limits the interpretation of risk assessment data with respect to shellfish populations throughout the Hauraki Gulf Marine Park.  In other words, even with information on the impacts of potential stressors on shellfish in an individual sense, in most cases we are unable to predict the likely impact on the wider shellfish population.  This relates particularly to the way in which shellfish stocks (or beds) are interconnected within a population (e.g. does a localised impact in one stock result in an impact in a distant stock as a result of reduced larval supply?).

Elucidation of the above issues would allow much of the existing environmental data to be interpreted appropriately.  The volume of work necessary to definitively identify, and thus effectively manage, the causes of the depletion of infaunal shellfish stocks in the Hauraki Gulf

Marine Park is large. The observed trends in shellfish abundance suggest that this issue needs to be resolved in the near future.

The size of the knowledge gaps, and the finite financial resources available to direct toward this issue, suggest that the elucidation of issues related to shellfish depletion in the Hauraki Gulf Marine Park will be difficult without a cohesive strategy to involve wider groups in this process. We suggest that this could be achieved in two ways:

*Facilitation of capacity-building:*

The complex nature of the issues, and the large number of knowledge gaps suggest that significant resources in terms of funding and expertise will be required to fully investigate the potential causes of shellfish depletion in the Hauraki Gulf Marine Park. Our review of unpublished data, and consequent communication with people and organisations working in the marine environment revealed that there is a wide variety of potential sources of information about issues of relevance to depletion of infaunal shellfish populations in the Hauraki Gulf Marine Park. Much of this information is not utilised to its full potential. It is suggested that significant synergy could be achieved by utilising existing resources to enhance the knowledge base relevant to this issue. Potential contributors to this knowledge base include researchers from research organisations and Universities, community members involved in monitoring beaches, people from organisations that comprise the Hauraki Gulf Forum, shellfish farmers, Area Health Board Health Protection Officers associated with shellfish safety, iwi members associated with resource management issues etc. It is suggested that this process could be facilitated by:

- Development of an internet-based newsgroup to facilitate exchange of information.
- Hosting regular (6-monthly) workshops to discuss the shellfish depletion issue. This will provide an opportunity to disseminate and discuss research results, develop linkages between researchers, and encourage research effort into this area by research providers. Informally published proceedings of the workshops will provide a collation of latest research results. (We note that a similar process has been facilitated by MAF Regulatory Authority in Wellington over the last ten years with respect to the marine biotoxin issue, with great success).

*Improved use of existing resources:*

We suggest that collection of long-term data on the status of inter-tidal infaunal bivalves could be cost-effectively addressed by improving the use of existing resources, allowing available funding to be directed toward high quality research focused initially on the latter issues. To achieve this we suggest that the benefits of the "Adopt-a-Beach" programme be maximised. This could be facilitated by the Hauraki Gulf Forum through:

- Encouraging skilled people to become involved, to ensure quality of results
- Funding the establishment of a database of results from the programme, along with a periodic review of the results by appropriately skilled scientists.

This could potentially broaden the geographic scope of data collected quite significantly.

These data should be complemented by the data collected from monitoring programmes such as the Ministry of Fisheries "Inter-tidal Shellfish Monitoring Programme". It is recommended that the

Hauraki Gulf Forum encourage the Ministry of Fisheries to re-focus this programme to ensure that meaningful long term abundance data are collected.

In addition, we suggest that the possibility of undertaking a series of infaunal bivalve population surveys that replicate the location and methods used in historic surveys be explored as a first step to investigating the way in which population abundance has changed over a longer time period (e.g. 20-30 years). This may assist in identifying subtle long term changes that are nevertheless significant to our understanding of the current situation.

Lastly we note that one of the benefits of sustainable populations of infaunal bivalves in the Hauraki Gulf Marine Park would be their availability for harvest by the public. The maintenance of water quality to ensure that shellfish are safe to eat when harvested is also a pressing issue.

# *APPENDIX I      CONTRACT BRIEF*

### *Hauraki Gulf Localised Shellfish Depletion Issue.*
### *Project: Stage 1 Contract Brief: 'Review Current State of Knowledge'*

Proposed staging of the efforts aimed at addressing the issue of infaunal inter-tidal shellfish depletions first involves scoping work to identify outstanding information needs (stage 1) and their relative priority for action (stage 2), followed by focused research, if necessary, to fill gaps in understanding. This brief covers the first stage (stage 1) of scoping work discussed in the background section above.

## Objectives for Stage 1 Work Covered by this Brief

The objectives of stage 1 work covered by this brief are;
1.      To provide a synthesis of the current and anticipated near future (i.e., next 1-3 years) state of knowledge of factors potentially affecting shellfish populations (primarily *A. stutchburyi*, *P. australis*, *P. subtriangulata* and *M. liliana*) in the Hauraki Gulf Marine Park area.
2.      To identify gaps in our understanding.
3.      To provide a strong platform from which the critical knowledge gaps, and priorities for future investigation and research, can be identified.

## Stage 1 Work Required

The work required by this brief is the Stage 1 review of the current state of knowledge outlined in the background section above. Specific work requirements are further detailed below:

### *Study Area*
The overall focus of the work is to explain the observed declines in target shellfish populations within the Hauraki Gulf Marine Park. However, a synthesis of the current state of knowledge of factors potentially related to the decline is expected to benefit from the results of studies carried out anywhere within the full spatial range in New Zealand of the species of interest, or of related or similar species. Likewise, relevant key findings from overseas work should be considered.

### *Search Information Sources.*

Stage 1 information search should include the following:
- Consider all reasonably identifiable actual or potential physical, chemical, and biological threats to the shellfish populations.
- Consider all life cycle stages of the shellfish.
- Consider the full range of habitats within which the shellfish (primarily *A. stutchburyi*, *P. australis*, *P. subtriangulata* and *M. liliana*) are found, not just the sites where localised depletion has been identified.
- Consider relevant information on the species of interest in locations outside the Hauraki Gulf.
- Expand on the currently available bibliographic report compiled for the Ministry of Fisheries (copies will be made available).

In addition to normal literature searches through libraries etc, identify other related work in areas pertinent to the issue carried out by all relevant agencies.

---

This will require contacting a range of organisations and groups to identify any additional completed work of relevance to the issue, or work in progress, that may not have been published or appear in external literature sources.

This search should include, but not necessarily be limited to, the following agencies;

- Department of Conservation.
- Ministry of Fisheries.
- Ministry for the Environment
- Museums.
- Crown Research Institutes.
- Regional Councils
- Universities
- Polytechs and Technical institutes,
- Non Governmental Organisations

**Deliverable Output: Current State of Knowledge; Bibliography and Report.**

The results of the information search above are to be presented in a report as outlined below. The report must be made available in both electronic and hard copy format.

Produce a written report that provides the following:

- A bibliography of the information sources located in the above search.
- Interpretation of the information identified by the search to produce a synthesis of current (and expected near future) knowledge of the factors potentially affecting relevant shellfish populations.
- Assessment of factors to identify areas that are well understood, and in particular areas where significant knowledge gaps are apparent.
- Discussion of the various factors potentially affecting relevant shellfish populations with particular reference to distinguishing anthropogenic influences from natural influences.

Recommendations, with supporting rationale, of the priority for further areas of research required to address the issue of shellfish depletion in the Hauraki Gulf study area.

## *APPENDIX II – SUMMARY OF MAILOUT PARTICIPANTS AND RESPONSES*

| *First Name* | Last Name | Company/Institution | Response Type / Alternate Respondent Name / Information Received |
|---|---|---|---|
| **CONSULTANTS / CRIs** | | | |
| Jo-Anne | Akroyd | Akroyd Walshe Ltd | Phone/Kim Walshe/Information on current study of shellfish stocks and non-commercial harvesting activity |
| Brian | Coffey | Brian T Coffey & Associates | None |
| Martin | Cryer | NIWA Auckland | ***Joint Email / Dave Roper / suggested FRST research is available as published material, and for ARC work should contact the ARC, suggested searching NIWA Website |
| Vonda | Cummings | NIWA Hamilton | See *** |
| Ben | Diggles | NIWA Wellington | Email / Citation for paper on shellfish disease agents survey. |
| Graham | Don | Bioresearches Ltd | None |
| Graham | Fletcher | NZ Institute for Crop & Food | Email / No relevant information available. |
| Malcolm | Green | NIWA Hamilton | See *** |
| Gail | Greening | ESR Ltd. | *No relevant information available.* |
| Judy | Hewitt | NIWA Hamilton | See *** |
| Chris | Hickey | NIWA Hamilton | See *** |
| Mike | Hine | National Centre for Disease Investigation, Wellington | *See B. Diggles* |
| Terry | Hume | NIWA Hamilton | See *** |
| Paul | Kennedy | Kingett Mitchell Ltd | None |
| Michael | Larcombe | Private Consultant | Phone – Annotated comments |
| Carolyn | Lundquist | NIWA Hamilton | See *** |
| Dan | McClary | Kingett Mitchell Ltd | None |
| Shaw | Mead | ASR Ltd | Email / No relevant information available. |
| Mark | Morrison | NIWA Auckland | See *** |
| Barry | Robertson | Cawthron Institute | None |
| Tara | Ross-Watt | Boffa Miskell Ltd | Email / Shellfish Bibliography and summary of current work. |

| First Name | Last Name | Company/Institution | Response Type / Alternate Respondent Name / Information Received |
|---|---|---|---|
| Environmental | Scientist | Meritec Ltd. | None |
| Paul | Scotti | HortResearch Ltd. | Email / Citation for reference to pipi haemolymph protein in relation to green-lipped mussel haemolymph work.  Citation added to database. |
| Carol | Stewart | Private Consultant | Email / Citation for antifouling biocides report and reference list. |
| Mike | Taylor | Cawthron Institute | Mail/ Paper received re invasive species |
| Simon | Thrush | NIWA Hamilton | See *** |
| Mike | Timperley | NIWA Auckland | See *** |
| Karen | Tricklebank | Private Consultant | Email / Abstract of parasite work in cockles conducted for the ARC. |
| Simon | West | Bioresearches Ltd. | None |
| Ron | Wong | NZ Institute for Crop & Food | *See G. Fletcher response.* |

## *UNIVERSITIES / POLYTECHS*

| First Name | Last Name | Company/Institution | Response Type / Alternate Respondent Name / Information Received |
|---|---|---|---|
| Andrea | Alfaro | Auckland University of Technology | None / John Buckridge / Provided access to Long Bay dissertations in the AUT Faculty Library. |
| Karin | Bryan | University of Waikato | Email / Email address for Ph.D. student (Giles Hilke) working on shellfish.  Follow up email sent.  Giles not working on shellfish. |
| Paul | Decker | Mahurangi Technical Institute | Phone - John Walsby (Information received re benthic sampling in the Whangateau Harbour and Snells Beach) |
| Richard | Ford | Leigh Marine Laboratory, Univ. of Auckland | Email / Citations for reports undertaken for the ARC, also summaries for 2 proposed masters projects. |
| Jonathan | Gardner | Victoria University | None |
| Keith | Gregor | Bay of Plenty Polytechnic | None |
| Bruce | Haywood | University of Auckland | None |
| Terry | Healy | University of Waikato | None |
| Islay | Marsden | University of Canterbury | None |
| Barrie | Peake | Otago University | Email / Sent abstracts from M.Sc students work.  These were not relevant but added to database. |
| Conrad | Pilditch | Waikato University | Email / Sent citations for student work and reference to benthic sampling Whangateau data set**.** |

| First Name | Last Name | Company/Institution | Response Type / Alternate Respondent Name / Information Received |
|---|---|---|---|
| Keith | Probert | Otago University | None |
| Mere | Roberts | University of Auckland | Email / Email address for MSc student (Ou Li) working on shellfish.  Follow up email sent. |
| Mary | Sewell | University of Auckland | None |
| Megan | Stewart | Leigh Marine Laboratory, Univ. of Auckland | Letter / Referred to ARC for copies of progress reports. |
| Marine | Studies | Northland Polytechnic | Email / Kevin McAdams / No relevant information available. |
| Jarrod | Walker | Leigh Marine Laboratory, Univ. of Auckland | *See R. Ford response*. |
| Robert | Wear | Victoria University | Email / Citations to surf clam references in the Wellington region. Citation added to database. |
| Stephen | Wing | Otago University | Email / Craig Irwin / PhD thesis and 2 MSc theses looking at cockles |

## REGIONAL / CITY COUNCILS

| First Name | Last Name | Company/Institution | Response Type / Alternate Respondent Name / Information Received |
|---|---|---|---|
| Andy | Armstrong | Gisborne District Council | Email / Andrew Reid / No research work undertaken, testing for fecal contamination is done, possibly some information in consent applications. |
| Jo | Fagan | Wellington Regional Council | Email / Gary Stephenson / Report – Investigation on the contamination of shellfish by urban stormwater.  Reference to NIWA Pauatahanui Inlet cockle study with Guardians of Pauatahanui. |
| Dan | Govier | Taranaki Regional Council | Email / No relevant information available. |
| Peter | Hamill | Marlborough District Council | Email / Peter Hamill / No relevant information available. |
| Andrea | Harris | Manawatu-Wanganui Regional Council | Email / No relevant information available. |
| James | Hollaway | Southland Regional Council | Email / Contact for work on Toheroa and reduced river flows. |
| Bruce | Howse | Northland Regional Council | Email / Reference to NIWA project underway to assess the sustainability of areas of the Whangarei Harbour for the reseeding of shellfish (cockle and pipi).  Copy of deepwater port monitoring programme received. |

| First Name | Last Name | Company/Institution | Response Type / Alternate Respondent Name / Information Received |
|---|---|---|---|
| Jane | MacGibbon | Hawke's Bay Regional Council | Email / No relevant information available, but are starting baseline coastal surveys in a few months time (to be repeated every 10 yrs). |
| Stephen | Park | Environment Bay of Plenty | None / Baseline monitoring of metal contaminants in shellfish undertaken. |
| Helen | Rouse | West Coast Regional Council | Email / Trevor James / Crown Public health collect virus, faecal and phycotoxin information. |
| Rob | Smith | Tasman District Council | Email / No relevant information available, suggest MFish for commercial cockle info. in area. |
| Ken | Taylor | Environment Canterbury | None / No relevant information available. |
| Marion | Weaver | Otago Regional Council | Email / Geraldine Moore / No relevant information available. |
| Jim | Hodges | North Shore City Council | Phone / CD of Sediment Monitoring Report. |
| **GOVERNMENT AGENCIES** | | | |
| Tony | Beauchamp | Department of Conservation | None |
| Kirsty | Johnston | Ministry for the Environment | Email / Megan Linwood / No relevant information available. Although website helpful for background information. |
| Greg | Martin | Regional Conservator (Waikato), Department of Conservation | Email / John Gumbley / No relevant information available. |
| Rob | McCallum | Regional Conservator (Auckland), Department of Conservation | Letter / Kala Sivaguru / Pollen Island Report. Reference to Long Bay work – inter-tidal rocky shores. |
| **OTHER** | | | |
| Bill | Brownell | EcoQuest NZ | None |
| Mollusc | Curator | Te Papa | None |
| Mollusc | Curator | Auckland War Memorial Museum | None |
| **MAORI** | | | |
| Honey | Renata | | None |
| Betty | Williams | | None |
| Paul | Majurey | Hauraki Maori Trust Board | Letter received from William Peters, Ngati Maru Runanga – no relevant scientific studies undertaken to date – concern expressed |

| *First Name* | Last Name | Company/Institution | Response Type / Alternate Respondent Name / Information Received |
|---|---|---|---|
| | | | at declining stocks |
| Laly | Haddon | Ngatiwai Trust Board | None |
| Hariata | Gordon | | None |
| Te Warena | Taua | | None |