New Zealand Aquatic
Environment and Biodiversity
Report No. 7
2006
ISSN 1176-9440

Impacts of terrestrial runoff on the biodiversity of rocky reefs

A. Schwarz
R. Taylor
J. Hewitt
N. Phillips
J. Shima
R. Cole
R. Budd

Impacts of terrestrial runoff on the biodiversity of rocky reefs

A. Schwarz[1]
R. Taylor[2]
J. Hewitt[1]
N. Phillips[3]
J. Shima[3]
R. Cole[4]
R. Budd[1]

[1]NIWA
PO Box 11115
Hamilton 3216

[2]Leigh Marine Laboratory
P O Box 349
Warkworth

[3]Victoria University of Wellington
P O Box 600
Wellington

[4]NIWA
P O Box 893
Nelson

New Zealand Aquatic Environment and Biodiversity Report No. 7
2006

Published by Ministry of Fisheries
Wellington
2006

ISSN 1176-9440

Citation:
Schwarz, A.; Taylor, R.; Hewitt, J.; Phillips, N.; Shima, J.; Cole, R.; Budd, R. (2006).
Impacts of terrestrial runoff on the biodiversity of rocky reefs.
New Zealand Aquatic Environment and Biodiversity Report No 7. 109 p.

This series continues the
Marine Biodiversity Biosecurity Report series
which ceased with No. 7 in February 2005.

CONTENTS

EXECUTIVE SUMMARY

Schwarz, A.; Taylor, R.; Hewitt, J.; Phillips, N.; Shima, J.; Cole, R.; Budd, R. (2006). Impacts of terrestrial runoff on the biodiversity of rocky reefs.

***New Zealand Aquatic Environment and Biodiversity Report No. 7.* 109 p.**

This project investigated effects of terrestrially derived sediment on rocky reef biodiversity. The focus was on how terrestrial sediment inputs affect filter feeding invertebrates, planktonic larvae of paua and kina, productivity and relative abundance of kelp, understorey algae and associated epifauna, in order to establish predictive relationships between characteristics of key benthic habitat structuring organisms and their responses to terrestrial sediment inputs. Experiments were conducted in the laboratory as well as *in situ*. Field studies were conducted at four sites on a gradient between the Whitianga Harbour mouth and Hahei and, for larval studies, in Wellington Harbour. The studies showed terrestrial sediment had both indirect (via water clarity) and direct (via sediment effects on filter feeders) effects.

Initially we established that rocky reefs near both the Whitianga Harbour and the Hutt River mouth were influenced by terrestrially-derived suspended sediments over the period of the study (March 03 – July 04). During times of turbid water inflows from the Whitianga Harbour mouth, there was a gradient of increasing water clarity towards Hahei (from site 1 to site 4). Although there were times when water clarity was the same at all four sites, site 1 was generally subject to lower water clarity, and also to reduced clarity over longer time-scales, than the other sites. Periods of low water clarity tended to be coincident with, or to follow, periods of high rainfall.

Indirect effects

The lowest density, biomass and productivity of epifauna on *Eklonia radiata* were found at site 1. Epifauna on *E. radiata* graze on epiphytes on the surface of the macroalga. Reduced epifaunal productivity suggests that anthropogenically increased suspended sediment concentrations on New Zealand’s rocky reefs may have serious consequences at the ecosystem level, since epifauna are responsible for about 80% of the flow of energy and materials through rocky reef animal communities. Although the functional role of seaweed epifauna is poorly understood at present, it is likely that density reductions will have knock-on effects throughout the rocky reef foodweb, both downwards through reduced epifaunal grazing on seaweeds and algal epiphytes, and upwards through reduced availability of food for small fishes.

E. radiata at site 1 exhibited the greatest degree of acclimation to a low light environment for a given water depth. There was also evidence for a lower density of *E. radiata* at this site, however, individual plants were able to maintain mature plant biomass that was not significantly different to equivalent depths at the other three sites. Nevertheless, the reduced photosynthetic potential of *E. radiata* observed suggests reduced production of all primary producers, including epiphytes on *E. radiata* is likely.

Direct effects on filter feeding organisms

The effects of suspended sediments on filter feeders collected from the Hahei site were largely sub-lethal within the range of sediment concentrations experienced in this location. However, there was evidence of increased energy requirements and synergistic effects of duration and exposure. In the laboratory, sponges, green-lipped mussels, and rock oysters all showed signs of physiological stress through altered feeding rate responses to elevated suspended sediment concentration. Although there was no mortality, sponges, oysters, and mussels exhibited significant decreases in condition at suspended sediment concentrations of >15, >15, and >26 mg l^{-1} respectively. Condition and feeding rates of the filter-feeding amphipod *Aora typica* could not be measured, but mean survival of *A.typica* decreased logarithmically with increasing suspended sediment concentrations, and increasing duration of exposure led to increasing sensitivity.

Experiments on larval paua and kina, conducted in Wellington, found that effects of sediments were likely to be lethal. The two test organisms in this study, paua and kina, have differences in life-history strategies, morphologies, and larval durations, and, not unexpectedly, patterns of mortality differed between species. Paua showed greater sensitivity to suspended sediments. The same range of concentrations increased cumulative mortality at larval competency by up to 49% for paua, and up to 27% for kina. For paua, early acute exposure was highly deleterious, resulting in greatly increased larval mortality that was manifested in subsequent stages. Kina exposed to sediments early in development showed some capacity to recover, and those exposed later in development had cumulative mortality more similar to larvae that were chronically exposed to sediments in high concentrations. Developmental abnormalities attributable to suspended sediment regimes were not found in the laboratory, for either species. In contrast, in harbour waters there was a high incidence of developmental abnormalities, suggesting the importance of other environmental factors associated with terrestrial runoff.

The various strands of this study have shown both chronic and toxic effects of elevated suspended sediment concentrations on a range of organisms. We tentatively suggest a series of thresholds (minimum concentrations for effect) for effects based on the findings from this study, although duration of exposure to these concentrations has not been allowed for except as described in the relevant chapters. All thresholds fall well below the levels that some rocky reef environments are exposed to at both Whitianga and Hutt River. The approach that we have taken has suggested putative causative relationships between terrestrially derived sediments inputs to the rocky reef environment, however, it is essential that these predictions are tested at multiple New Zealand locations and under a wider range of physical and ecological conditions.

1. INTRODUCTION

Elements of terrestrial runoff that are likely to affect rocky reef ecosystems are sediments, nutrients, and a range of other pollutants. Elevated sediment loads to the New Zealand coastline have been clearly identified as potentially affecting coastal biodiversity and are the focus of this study.

Catchment-based activities such as agriculture, forestry, and urban development can have significant effects on New Zealand's coastline through increased export of terrestrial sediments and the subsequent reworking by coastal waves and currents. An extreme example of such runoff is the estimated 16 million cubic metres of mud washed into Hawke Bay during cyclone Bola in 1988 (Glasby 1991). The effects of such acute and catastrophic events on biodiversity may be severe and long-lasting (Foster & Carter 1997). Less well known are the chronic effects of sediment loads that pass through New Zealand's estuaries or reach the open coast directly via rivers, at lower concentrations but over longer frames. There is, however, evidence to suggest that differences in water clarity, productivity, and associated variables as a result of increased sediment loading may have profound effects on biota (Schiel & Foster 1986, Estes et al. 1989, Cole & Babcock 1996, Airoldi & Virgilio 1998). Chronic and acute sediment loads are likely to have different underlying mechanisms for structuring biodiversity. Although catastrophic events can be devastating, such events are more difficult to predict and manage than the long-term chronic loadings which are associated with day-to-day catchment management.

Studies of biodiversity, and changes in diversity, should have both a structural and a functional component, since this allows observations to be placed in an ecological context (Lamont 1995, Gray 1997, Anon 2000; The New Zealand Biodiversity Strategy). While it is important to compare biodiversity among locations and times, understanding of both structural and functional components strengthens our ability to predict how the biota might respond to environmental change.

This project has three objectives which address different aspects of the structural and functional components of biota likely to be affected by terrestrial sediment loadings.

Objective 1. Conduct field and laboratory experiments to determine relationships between sediment loading, epifaunal assemblages, and mortality of filter feeding invertebrates.

Objective 2. Conduct field and laboratory experiments to identify the influence of sediment on early life stages of key grazers.

Objective 3. Determine photosynthetic characteristics and survival of large brown seaweeds and understorey algal species in relation to a sediment gradient.

In rocky reef ecosystems small mobile invertebrates ("epifauna"; crustaceans, molluscs, and polychaetes) are an important trophic link between benthic primary producers and predatory fishes (e.g., Jones 1988), nearly all of which feed on epifauna as juveniles. Kelp is the primary habitat for small mobile invertebrates on rocky reefs (Taylor 1998). In addition to indirect effects on kelp habitat, increased sediment loads are likely to have substantial direct negative consequences for epifauna, with most species being either suspension feeders (and thus vulnerable to clogging of their feeding apparatus) or grazers of microscopic epiphytic algae (and thus potentially affected by smothering of the epiphytes). These effects were examined through field studies and dose-response laboratory experiments. (Objective 1). (Sections 3 and 4).

Invertebrates may also be subject to direct and indirect effects of elevated sediment loads. Non-geniculate corallines have been shown to have a role in the metamorphosis and settlement behaviours of various molluscs (e.g., paua) and other key invertebrates. Thus, the direct effects of sediments on non-geniculate corallines can indirectly affect the recruitment success of key algal grazers that might further reshape patterns of biodiversity. Similarly, development and survival (and thus recruitment) of early-life stages of these key grazing species are likely to be affected directly by terrestrially sourced sediments in coastal waters. If recruitment of such species fails, via either direct or indirect mechanisms, then reef biodiversity is dramatically altered. Dominant grazers can have a profound influence on algal recruitment, and may themselves be affected strongly by suspended sediments during their early developmental stages. In this project we examined sediment effects on grazer (paua and kina) larvae and recruitment success (Objective 2). (Section 5).

Algal structure on rocky reefs is generally recognised as a strong overall determinant of rocky reef biodiversity, through its provision of food supplies and modification of three dimensional habitat structure. Therefore, increased sediment loads on rocky reef algae are likely to have major effects on reef biodiversity in general. Large brown kelps, as well as crustose and low-stature algal species, may be affected by sediments in many ways. This may be direct, via burial (including dramatically reduced growth and survival of gametophytes), or shading (which reduces energy supply), or indirectly via modified interactions with other species (such as grazers) that influence abundance and distribution. Objective 3 of this project, focused on the direct effects of sediments on large brown kelp and macroalgae, and invertebrate taxa growing beneath the kelp canopy. (Sections 2 and 6).

A mechanistic understanding of the effects of terrestrial inputs on rocky reef biodiversity ultimately requires a detailed understanding of physical processes acting at appropriate scales. The magnitude of the effects of chronic sediment loads depends largely on water currents, wave action, and sediment type. Thus any field analysis of the effects of sediment needs to be grounded in sufficient understanding of the physical environment at the relevant scales. It was beyond the scope of this project to adopt a multi-disciplinary model catchment, gradient approach (e.g., Keddy 1991), or to model the physical environment. Rather the initial focus in this study was on establishing predictive relationships between some ecological features under a limited range of physical conditions related to terrestrially derived sediment inputs.

This study comprised field and laboratory studies conducted between February 2003 and July 2004. The field work for Objective 2 was carried out in Wellington Harbour at the Hutt River mouth, and Objectives 1 and 3 were focused on sites within and adjacent to the Te Whanganui-A-Hei Marine Reserve. All findings from the Wellington region are described under Objective 2. The remainder of the report refers to the Whitianga / Hahei sites.

In this report we first describe the water clarity characteristics of the Whitianga/Hahei sites and our rationale for site selection as it forms the basis for field experiments and collections for laboratory experiments in all objectives. We then present findings related to rocky reef biodiversity. Finally the findings from each piece of work are summarised, commonalities identified, and hypotheses about sediment effects on some components of New Zealand’s rocky reefs are proposed. (Sections 3 – 8).

2. WATER CLARITY AND STUDY SITE SELECTION

2.1 Existing water clarity data

2.1.1 Methods

Before starting the field work for this study, data from light loggers deployed at Tower Rock (seaward from site 1, this study (see Figure 5)) between 31 May 1997 and 13 February 2001 were made available for analysis by Dr Chris Hickey (NIWA and Hahei Marine Reserve Committee). Over this period two Hobo light intensity dataloggers (Onset Computer Corporation) had been deployed, 2 m apart at 10 and 12 m, in a total water depth of 20 m. In addition, two loggers were recording concurrently within the marine reserve (Kingfish Rock, site 2, this study) over two periods between 31 May 1997 and 22 March 1998.

Hobo loggers, housed in transparent waterproof housings, provide a useful record of the relative amount of irradiance reaching them, and comparisons can be made with other Hobo loggers. However, without some modification of the irradiance collector and detailed calibration against quantum sensors, they do not provide highly accurate measures of the absolute amount of photosynthetically available radiation (PAR) in units of μmol photons m^{-2} s^{-1}. Thus the existing data have been analysed within the constraints of the loggers, to describe the seasonal pattern of irradiance and to determine the attenuation of light between the two sensors moored at the same site but different depths. Attenuation of light, closely linked to water clarity, is one of the key effects of suspended materials in the water column.

The attenuation coefficient (Box 1) for downwelling irradiance (K_d) was determined using the equation:

$K_d = \ln(E_1/E_2)/(z_2-z_1)$ (Equation 1)

The existing underwater light data had been logged by the Hobo loggers at 2 hourly intervals over the course of each day. For calculations of K_d, only values for midday on each day were extracted from the dataset. When the sun is overhead the Hobo loggers exhibit the optimum angle of acceptance; therefore the use of midday values minimises error associated with the acceptance angle. Although the absolute amount of irradiance incident on the sensors will be affected by cloud cover, this does not affect the calculation of K_d from sensors placed at two depths. That value is dependent on water clarity rather than absolute irradiance.

Box 1: Measurement of underwater irradiance and contributions to attenuation

Downwelling irradiance is the value of the irradiance on the upper face of a horizontal surface (Kirk 1994) and as such is generally regarded as the value of most relevance to attached macroalgae. Water clarity and the penetration of irradiance to given depths are commonly described using either secchi depth (Z_{SD}) (the maximum visual range of a white or black and white disc viewed vertically) measurements or the rate of attenuation of downwelling irradiance (Davies-Colley et al. 1993). The attenuation of downwelling irradiance with depth is expressed as the rate of change of the logarithm of the value with depth, or the attenuation coefficient (K_d). The secchi disc has the advantage of being inexpensive and simple to use and there are relationships reported for the relationship between K_d and secchi depth (Geisen et al. 1990). However, the relationship varies considerably and whenever possible, particularly when photosynthetic characteristics of marine plants are being considered, direct measurements of underwater photosynthetically available radiation (PAR) are preferred. In general, PAR, which enters water is attenuated exponentially with increasing depth due to scattering and absorption. The rate at which this happens is dictated by a wide range of biotic and abiotic compounds. Particulate constituents (e.g., suspended solids, phytoplankton), dissolved constituents (yellow substance), and water itself all attenuate light in the water column (Davies-Colley et al. 1993).

The transfer of suspended solids from rivers and estuaries to the sea is important in determining the colour and clarity of the receiving waters. Yellow substance, which is derived from organic matter, is measured as the absorption of membrane filtered water at 400 nm (g440). Terrestrially and freshwater derived yellow colour is attenuated by seawater and near-shore coastal concentrations therefore tend to be lower than those of nearby inland waters. Concentrations of 0.1–0.6 m^{-1} have been measured in northern North Island estuaries (Vant 1990) and values tend to show a decline with distance from source (e.g., Vincent et al. 1989). Offshore values for g440 are typically 0.01 to 1 m^{-1} (e.g., Davies-Colley 1992) and yellow substance is only a minor cause of absorption in estuarine and coastal waters (Gallegos et al. 1990, Vant 1990). Phytoplankton and suspended solids therefore play a much greater role.

2.1.2 Underwater light data set : Whitianga site, 31 May 1997 to 13 February 2001

The expected annual patterns in underwater irradiance are very clear with a marked reduction during winter (Figure 1). Within this annual pattern, a clear daily pattern was seen as shown for 1 and 2 January (Figure 2).

Water clarity as estimated by K_d ranged from about 0.03 to 1.2 for the three years over which complete datasets are available from the Whitianga site (Figure 3). The clearest value (0.03) suggests unexpectedly clear waters for this area, and is probably indicative of the known inaccuracies associated with the Hobo loggers. Nevertheless, this analysis shows that water clarity varies considerably within a year and that general patterns can be ascertained. As an example, for 50% of the time K_d was 0.23 or less, meaning that incident irradiance at 20 m or greater was only 1% of that at the surface. In contrast, a K_d of 1.2 represents fairly turbid water and is equivalent to 1% of incident irradiance penetrating to only 3 to 4 m deep.

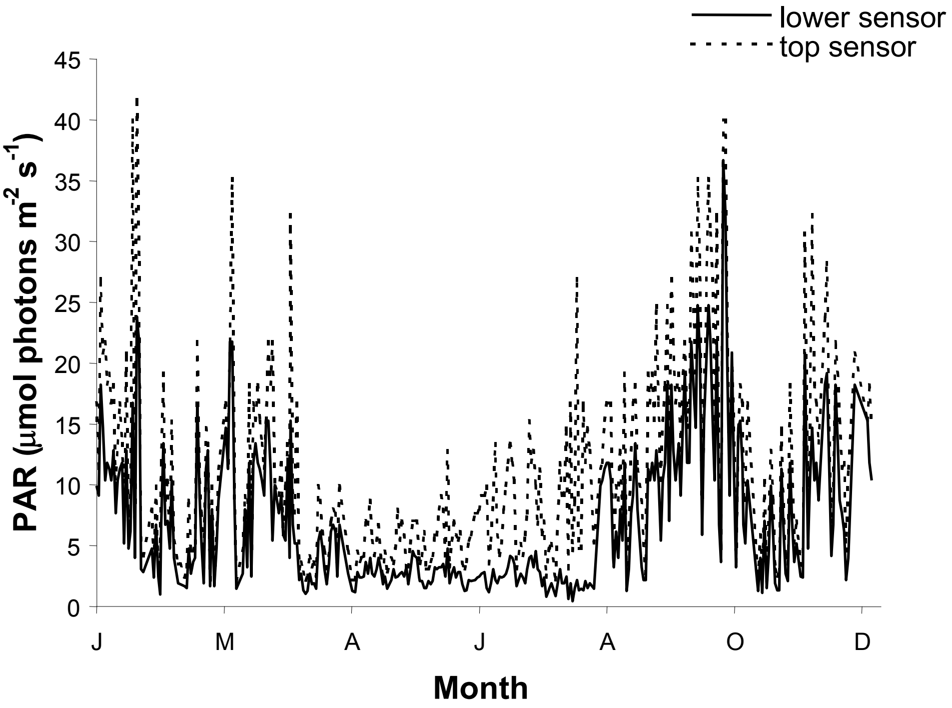


Figure 1: Midday irradiance approximated to PAR (µmol photons m^{-2} s^{-1}) at the upper and lower sensors from 1 January to 31 December 1999 at the Whitianga (Tower Rock) location.

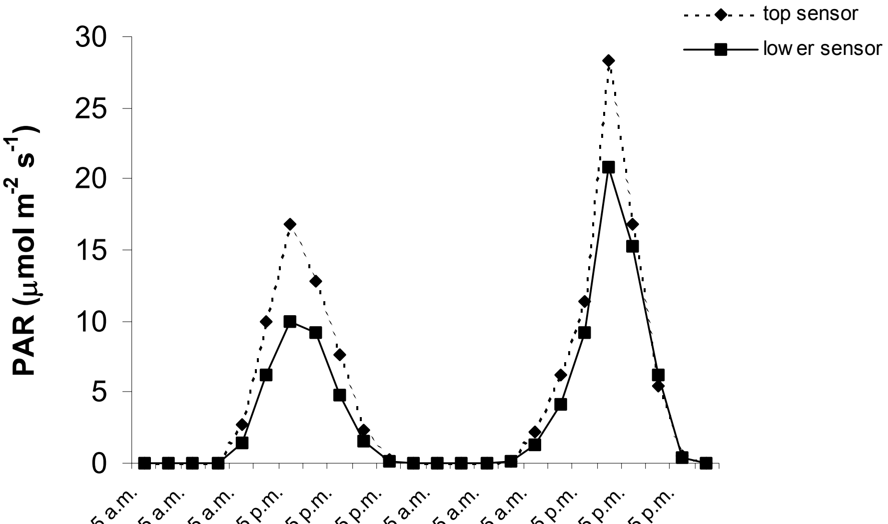


Figure 2: Detail of diurnal underwater irradiance (approximated to µmol photons m^{-2} s^{-1}) on 1 and 2 January 1999 at Whitianga (Tower Rock) location.

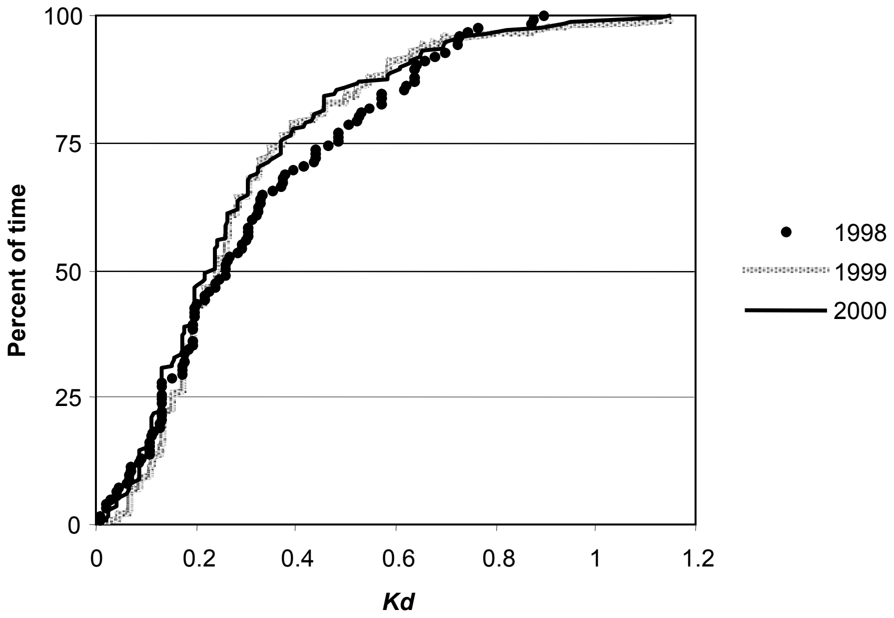


Figure 3: The percent of time that water clarity was equal to or clearer than given value of K_d during three different years at the Whitianga (Tower Rock) location.

There were indications that water clarity varied differently between the Hahei site and the Whitianga site. In the three months between December 1997 and March 1998, K_d ranged from 0.04 to 1.2 m^{-1} at Hahei (Kingfish Rock) and from 0.04 to 1.8 m^{-1} at Whitianga (Tower Rock).

2.1.3 Relationship to rainfall and incident irradiance

We expected that influxes of terrestrially derived sediments would be related to rainfall events, and hence river sediment loads. Flow data are not monitored on rivers flowing into Whitianga Harbour, and so the best indicator was rainfall data obtained from a NIWA monitoring station at Coroglen (Station B65961, 36° 55' S, 175° 41'E), to the west of the study area at the head of the Whitianga Harbour catchment. Patterns in water clarity were examined for evidence of direct correlations with rainfall events. In the existing water clarity dataset there was no consistent relationship between rainfall records at Coroglen and water clarity at the Tower Rock location, although there was some evidence of a low water clarity event associated with high rainfall at the end of January 2000 (Figure 4). We know from observations that the turbid plume from Whitianga Harbour can be pushed either north or south, depending on prevailing winds and currents and therefore may or may not intercept the Tower Rock logger site.

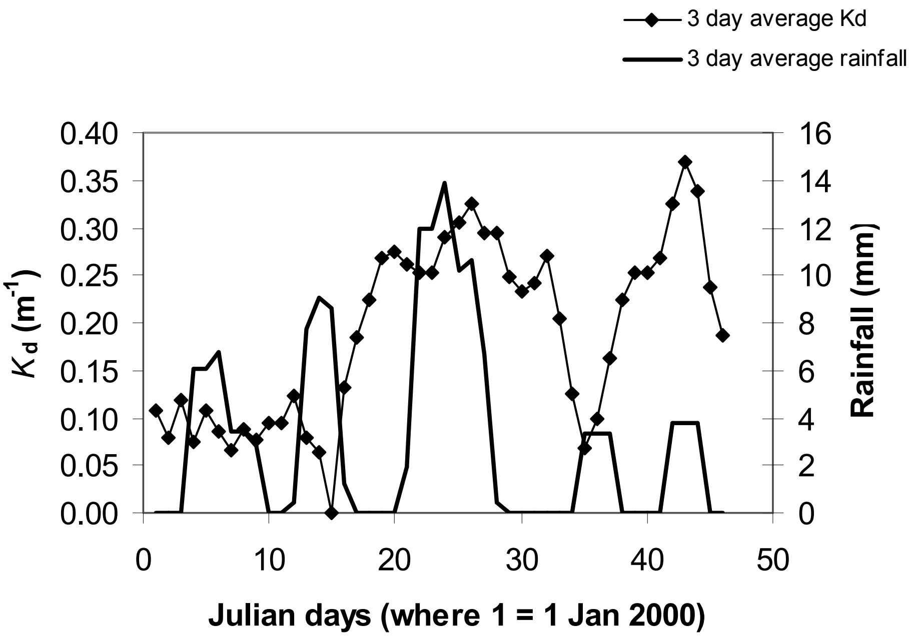


Figure 4: Rainfall and attenuation coefficient K_d at Whitianga (both averaged over three days) between 30 Dec 1999 and 20 February 2000.

2.2 Water clarity information and site selection

2.2.1 Methods

Over the period of the 2003-04 study, water clarity was measured as the attenuation of PAR over a depth profile. On eight occasions between March 2003 and July 2004 (Table 1) a depth profile of PAR was made at the four monitoring sites (Table 2) using a PUV500 profiler (Biospherical Instruments Inc.). From the PUV500 data, the attenuation coefficient for downwelling irradiance (K_d) for PAR was calculated using the same principle as Equation 1, but using log-linear regression analysis for $n > 2$ datapoints (Kirk 1994).

In addition, on five occasions (see Table 1), a water sample was collected from just below the water surface. This was returned to the lab and filtered for analysis of total suspended solids (TSS), phytoplankton chlorophyll *a* (Chl*a*), and dissolved yellow colour. In March 2003, August 2003, and July 2004 a larger number of sites (Figure 5), selected to cover a range of water clarity conditions, were sampled for characteristics shown in Table 1.

Table 1: Dates on which water clarity, (as the attenuation of PAR, K_d) and contributions to attenuation (TSS, YC, and Chl*a*) were measured at each of the four monitoring sites described in Table 3, and the dates on which optical characterisation was carried out at a wider range of sites (n= 7–13).

	K_d	TSS	Yellow colour	Chl*a*	Optics survey
March 03	⁙	⁙	⁙	x	⁙
April 03	⁙				
June 03	⁙	⁙	⁙	⁙	
August 03	⁙	⁙	⁙	⁙	⁙
October 03	⁙				
December 03	⁙				
April 04	⁙	⁙	⁙	⁙	
July 04	⁙	⁙	⁙	⁙	⁙

2.2.2 Results: Optical survey, March 2003, and site selection

The first of the optical surveys on 5 March 2003 was combined with a visit to establish the study sites for objectives 1 and 3. This visit followed a period of very high rainfall in the catchment and a noticeable sediment plume extended from the Whitianga Harbour mouth. These conditions enabled a range of water clarity conditions to be characterised.

The attenuation coefficient at 11 sites surveyed on 5 March (Figure 5) is given in Table 2. K_d ranged from a low (high water clarity) of 0.11 at site Hahei 7 in what appeared to be blue water, to 1.31 at site Hahei 11, within the visible plume coming from the Whitianga Harbour mouth. The magnitude of this difference means that 1% of incident irradiance (euphotic zone) penetrated to only 3.5 m at site Hahei 11 compared to 40 m at site Hahei 7. In August 2003, a gradient in water clarity (decreasing K_d = increasing clarity) was evident seaward from Whitianga Harbour, e.g., K_d site Hahei 1> Hahei 20 > Hahei 21 and site Hahei 22> Hahei 2> Hahei 12> Hahei 11, as well as a less marked gradient down the coast; Hahei 3> Hahei 4> Hahei 5 > Hahei 6 (Figure 5).

On the basis of the March 2003 site visit, Scuba dives by the researchers, water clarity data, and examination of habitat maps of the area held by Waikato University, four sites were chosen to represent a similar topography and exposure, and to provide access to a common depth stratum (10–13 m). Overlaid on this was the objective of covering a gradient of expected sediment loadings. We therefore identified sites along a gradient of increasing water clarity from the Whitianga Harbour mouth (Figure 5). Site details are given in Table 3.

Table 2: Attenuation coefficient K_d at 11 locations on 5th March 2003. The locations of each site are shown on Figures 5A and B.

Site	K_d
Hahei 1	0.29
Hahei 2	0.29
Hahei 3	0.28
Hahei 4	0.27
Hahei 5	0.26
Hahei 6	0.25
Hahei 7	0.11
Hahei 8	0.18
Hahei 9	0.24
Hahei 10	0.31
Hahei 11	1.31

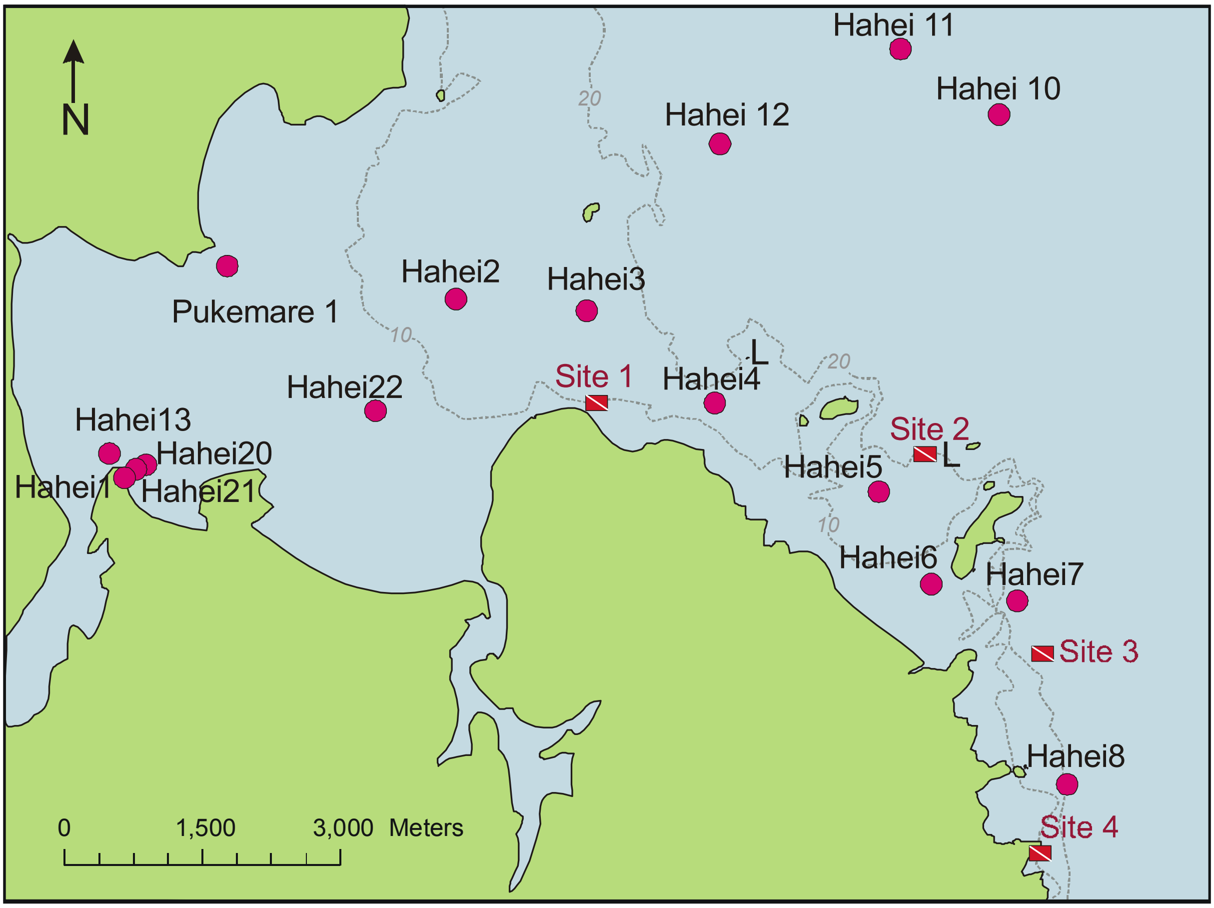


Figure 5: Location of original light logger deployments (1997 – 2001, C. Hickey) (L); optics sites (circles) and experimental sites (squares). Site 1 (Cook Bluff) is in the centre of the map and the other three study sites (Site 2 Kingfish Rock, Site 3 Hapuku Rock and Site 4 Te Pupuha Point were located progressively to the south-east.

Table 3: Location details for Sites 1 to 4.

Site #	Site name	Latitude and longitude	Depth of tagged plants (m)
1	Cook Bluff (Cook)	36° 48.83'S 175° 46.26'E	11
2	Kingfish Rock (Kingfish)	36° 49.40'S 175° 48.70'E	12
3	Hapuku Rock (Hapuku)	36° 50.47'S 175° 49.58'E	13
4	Te Pupuha Point (Pupuha)	36° 51.00'S 175° 49.76'E	11

On average, over the whole study, water clarity was lowest at site 1 (Cook) than any of the other sites. There was evidence for a gradient in average water clarity from site 1 to site 4 (Figure 6).

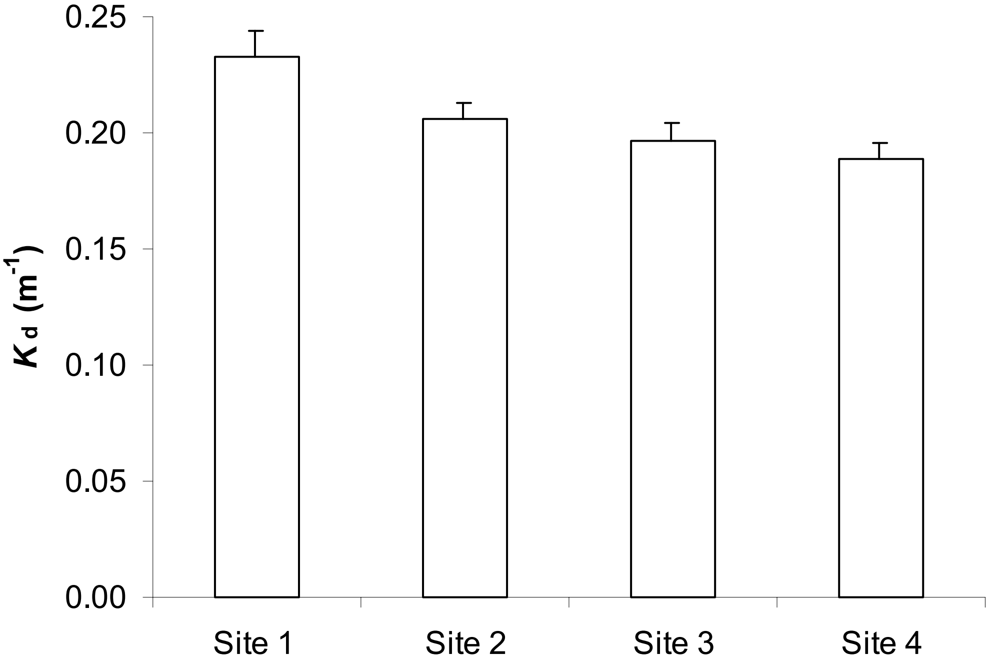


Figure 6: Average K_d (n=7) at the four study sites between March 2003 and July 2004.

Water clarity varied inconsistently between sites over time (Figure 7). For example in August 2003 water was very turbid (K_d = 0.37) at site 1 (Cook) but was similar (range 0.24 to 0.27) at the other three sites (Figure 7). In contrast in October Cook (1) was as clear as Pupuha (4) with a K_d over all four sites ranging from 0.14 to 0.2 (Figure 7). The variability meant that the euphotic depth (1% of incident irradiance) ranged from a minimum of 12 m at Cook (1) in August 2003 to a maximum of 43 m at Pupuha in June (Table 4). Pupuha in June 2003 therefore, had a water clarity within the range measured for oceanic waters of the eastern coast of New Zealand by Kirk et al. (1998).

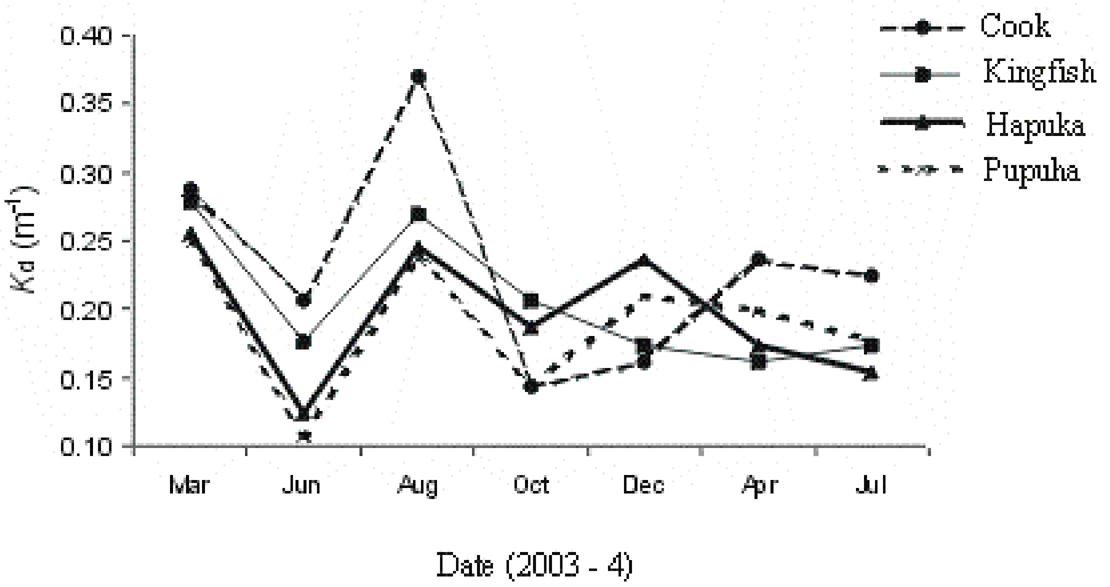


Figure 7: Attenuation coefficient (*K*$_d$) at four study sites between March 2003 and July 2004. (The higher the value for *K*$_d$ the lower the water clarity).

Table 4: Water clarity at each of the four study sites expressed as the attenuation coefficient for downwelling irradiance (*K*$_d$) and as the % of surface irradiance (SI) at the depth of the study sites. Total suspended solids (mg l^{-1}) is also shown. nd = no data.

Date	Cook			Kingfish			Hapuku			Pupuha		
	K_d	% SI 11 m	TSS	K_d	% SI 12 m	TSS	K_d	% SI 13 m	TSS	K_d	% SI 11 m	TSS
March 2003	0.288	4	4.8	0.278	4	4.7	0.257	4	n.d.	0.249	6	n.d.
June 2003	0.206	10	1.0	0.1765	12	0.7	0.1241	20	0.7	0.108	30	0.9
August 2003	0.370	2	16	0.27	4	7.4	0.2451	4	7.3	0.237	7	7.4
October 2003	0.143	21	nd	0.207	8	nd	0.187	9	nd	0.144	21	nd
December 2003	0.161	17	nd	0.174	12	nd	0.237	5	nd	0.209	10	nd
April 2004	0.237	7	1.3	0.161	14	0.6	0.173	11	0.8	0.199	11	1
July 2004	0.225	8	0.7	0.174	12	0.2	0.154	14	0.4	0.176	14	0.4
Average	**0.233**	**10**	**4.8**	**0.206**	**10**	**2.7**	**0.196**	**9**	**2.3**	**0.189**	**14**	**2.4**

2.2.3 Water clarity: synthesis

For all dates listed in Table 1, TSS concentration ranged from 0.133 to 84 mg l^{-1} over all surveyed sites. On average, at the four study sites TSS ranged from 2.4 mg l^{-1} at site 4 (Pupuha) to 4.8 mg l^{-1} at site 1 (Cook). Chl*a* ranged from 1.5 to 4 ug l^{-1} and g440 (for method see Box 1) from 0.015 to 0.255 m^{-1}.

The relationships between TSS, Chl*a,* and K_d show that TSS was the primary component affecting water clarity at our sites. The relationship between TSS and K_d (Figure 8) suggests that above concentrations of 8 mg l^{-1} TSS had a marked effect on water clarity. Below a TSS concentration of about 8 mg l^{-1}, and when water was relatively clear (K_d < 0.2), attenuation of PAR was probably affected by other constituents of the water column (e.g., phytoplankton, yellow colour).

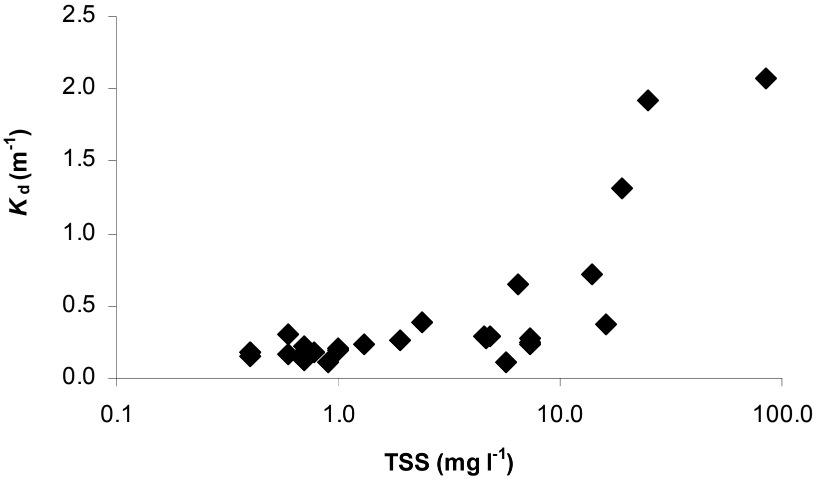


Figure 8: K_d plotted against TSS for all data collected from all sites between March 2003 and July 2004.

There was no consistent relationship between water clarity and Chl*a* (Figure 9). However, there was a log linear relationship between Chl*a* and TSS (Figure 10) suggesting there was an input of Chl*a* from the same source as the TSS. Nevertheless for a 100-fold increase in TSS concentration, the increase in Chl*a* is less than four-fold. Thus while some phytoplankton Chl*a* may also be sourced from river inputs, this is likely to have a negligible effect on water clarity compared to TSS. This does not preclude high Chl*a* concentrations, as a result of coastal phytoplankton blooms from markedly affecting water clarity at times. Indeed, we did observe a bloom of phytoplankton at the study sites in mid summer 2003–2004 that lasted for several weeks, and occurred independently of riverine inputs. This bloom was not quantified.

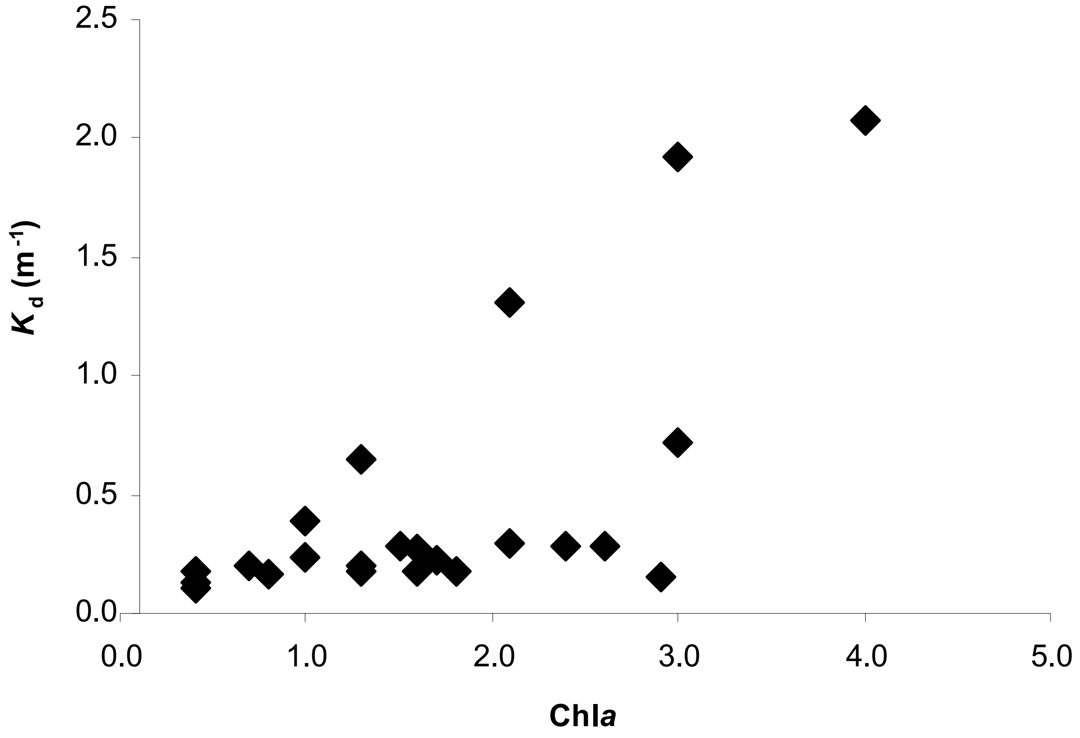


Figure 9: K_d plotted against Chl*a* for all data collected from all sites between March 2003 and July 2004.

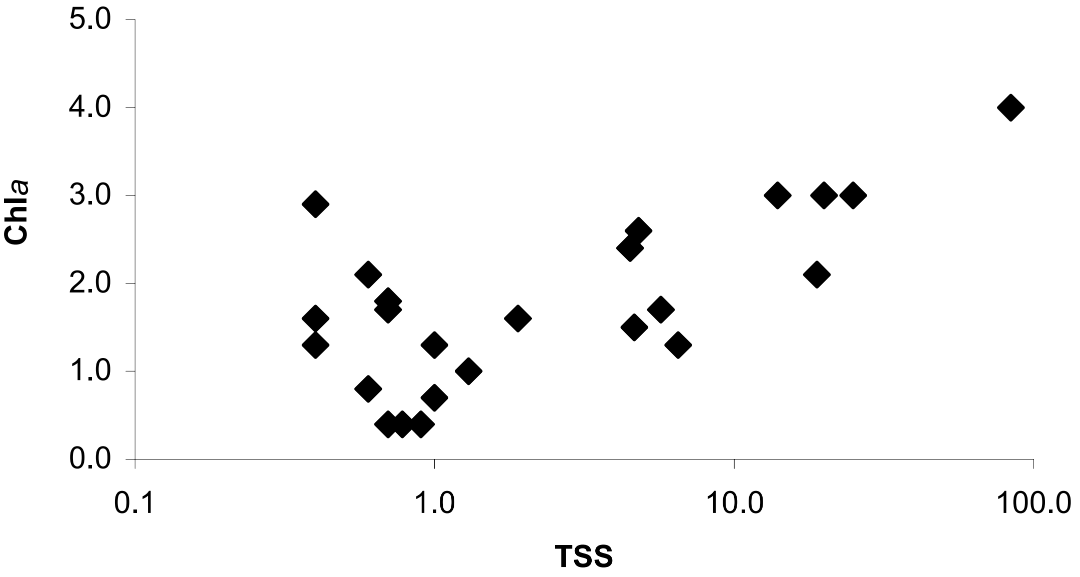


Figure 10: Chl*a* (μg l^{-1}) in relationship to log TSS (mg l^{-1}) for all data collected from all sites between March 2003 and July 2004.

During the March survey the highest concentrations of TSS and g440 were measured at the harbour entrance (Hahei 1 and 13, Figure 5). Concentrations decreased rapidly on mixing with seawater and with distance from the harbour mouth (Figure 11). Chl*a* concentrations were spatially variable. The high concentrations near the mouth indicate that Whitianga Harbour is a major source of suspended sediments to the sites.

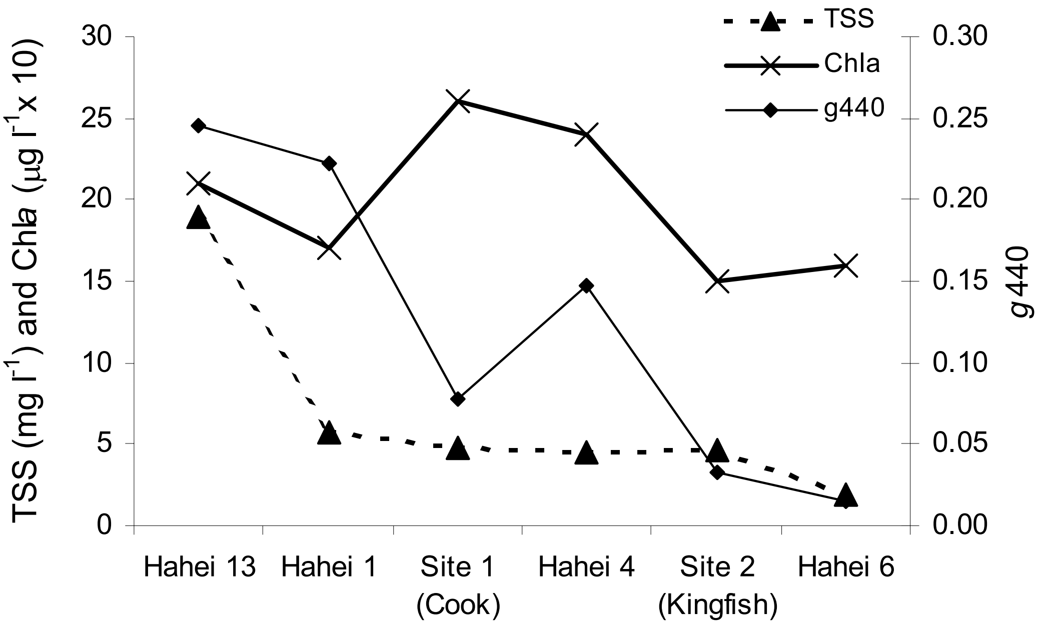


Figure 11: Concentrations of TSS, Chl*a* and g440 at six sites from the Harbour entrance toward site 3 (Hapuku) (for locations see Figure 5) in March 2003.

2.4 Sediment traps

2.4.1 Methods

To further characterise sedimentation at the four study sites, between August 2003 and April 2004 we deployed sediment traps within the 10–13 m depth strata (see Table 3). At each site three tubes 45 cm long with a 7 cm diameter mouth (H/D ratio 6.25) were deployed attached to a steel frame on a lead weight. Traps were emptied at two-month intervals. The entire contents of each trap were dried, ashed to remove organic matter, and weighed. A subsample was removed from the core for later grain size analysis. Sediment grain size, determined on a Galai particle analyser (Galai Cis - 100; Galai Productions Ltd., Midgal Haemek, Israel), was then used to calculate percent volumes for the sand (63 µm–2 mm), silt (4–63 µm) and clay (under 4 µm) fractions.

2.4.2 Results

On each occasion the sediment traps at site 1 (Cook) collected a greater total amount of sediment (Figure 12). Sand formed the greatest proportion of sediment in all the traps with a median grain size ranging from 137 to 180 µm among the sites (Figure 13).

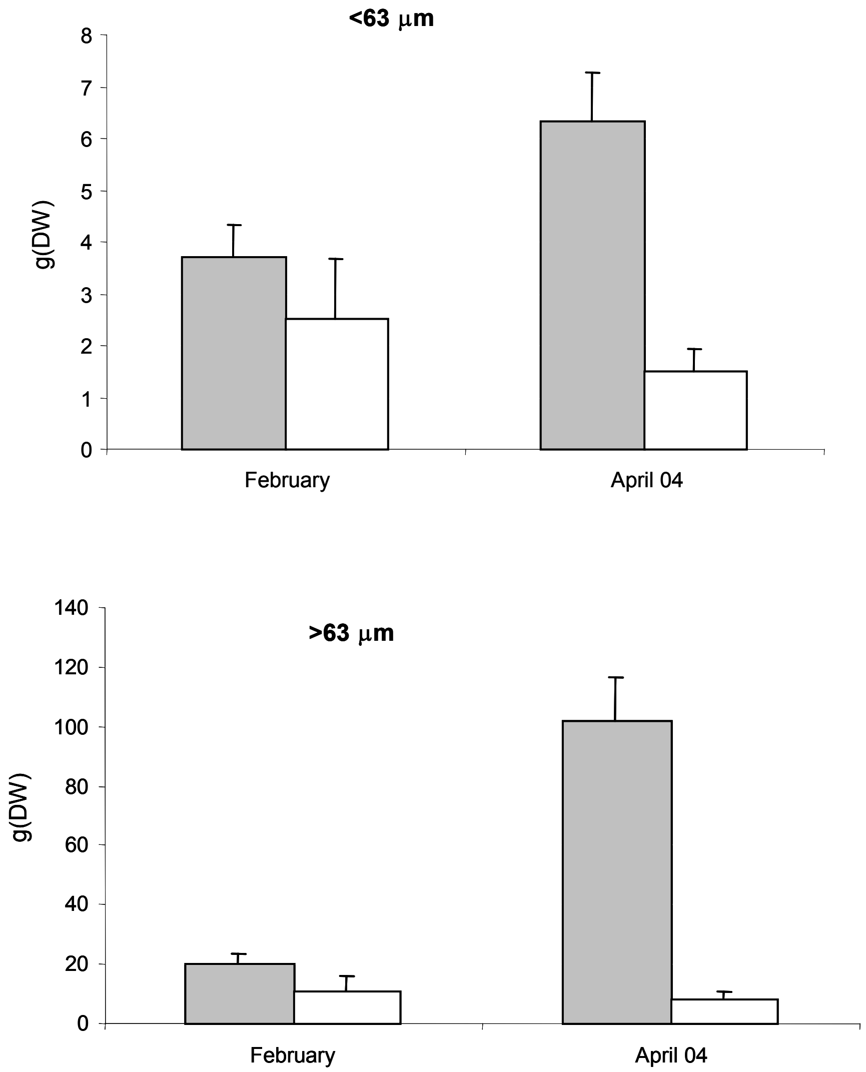


Figure 12: An example comparison of the amount of sediment trap material (g DW) comprising sand (> 63 µm) and silt / clay (< 63 µm). Site 1 (Cook) are shaded blocks, site 2 (Kingfish) are white blocks.

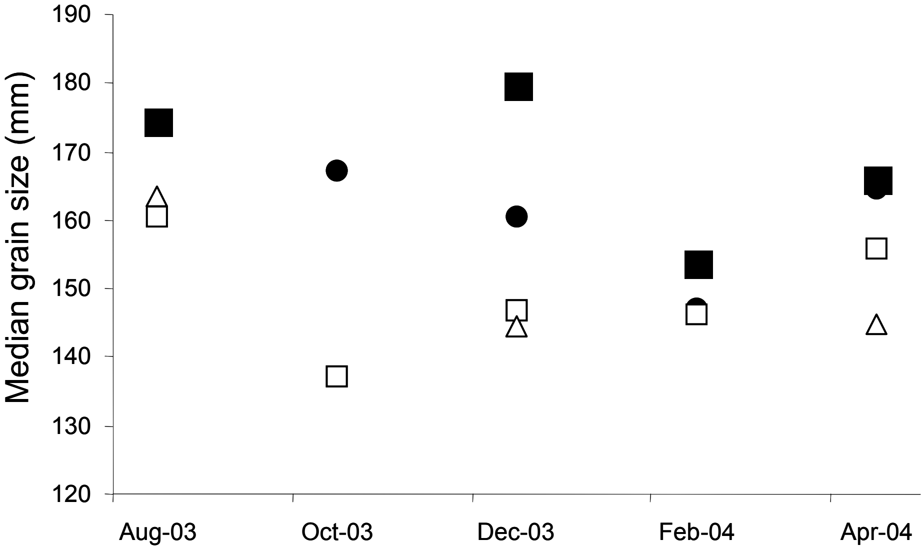


Figure 13: Median grain size at each of the four sites on five sampling occasions between July 2003 and April 2004. Site 1 (●), Site 2 (□), Site 3 (Δ), and Site 4 (■).

3. FIELD AND LABORATORY EXPERIMENTS TO DETERMINE RELATIONSHIPS BETWEEN SEDIMENT LOADING, EPIFAUNAL ASSEMBLAGES AND MORTALITY OF FILTER FEEDING INVERTEBRATES: ARTIFICIAL SUBSTRATE EXPERIMENTS TO DETERMINE EPIFAUNAL ABUNDANCE.

3.1 Introduction

Increased sediment loads are likely to have substantial direct negative consequences for the small mobile invertebrates inhabiting rocky reefs ("epifauna"; crustaceans, molluscs and polychaetes), with most species being either suspension feeders (and thus vulnerable to clogging of their feeding apparatus) or grazers of microscopic epiphytic algae (and thus potentially affected by smothering of the epiphytes). Effects on such epifauna are likely to translate into effects on predatory fishes, nearly all of which feed on epifauna as juveniles. We examined effects of sedimentation on mobile epifauna by measuring their abundance, biomass and productivity on both the naturally occurring kelp *Ecklonia radiata* and on standard artificial habitats, along a gradient of sedimentation. Standard rope-fibre "seaweeds" (hereafter "artificial plants") are rapidly colonised by epifauna, and are potentially useful for evaluating changes in epifaunal assemblages along gradients because they eliminate the confounding influence of changes in seaweed morphology (Edgar 1991).

3.2 Methods

In winter 2003 and summer 2004 epifauna were sampled from four rocky reef sites (Table 3) located along a gradient of sedimentation beginning near the mouth of the turbid Whitianga Harbour (Coromandel) and extending southeast into cleaner waters. Sampling dates are given in Table 5. At about 10 m depth at each site, six individuals of the kelp *E. radiata* (Figure 14a) were collected, and six artificial plants were deployed. Each artificial plant consisted of a 1.5-m length of 'mussel rope' (Kinnears Ropes, Auckland), which was doubled up twice and attached to 2 kg of lead with cable ties (Figure 14b). They were retrieved (n = 0–6 per site) from the seafloor after 2 months, which is sufficient time for epifaunal productivity to stabilise (Edgar 1993). For sampling, each replicate *E. radiata* or artificial plant was enclosed in a plastic bag by a diver, with the *E. radiata* cut off about 1cm above the holdfast, and the artificial plants removed from their lead weight before bagging. Epifauna were later removed in the lab by vigorously shaking the seaweed or artificial plant in fresh water. This water was then poured through a 0.5-mm mesh sieve to trap dislodged animals, and the procedure repeated until no further animals were dislodged (twice for *E. radiata*, 4–6 times for the artificial plants). The efficiency of this method has been confirmed for a number of seaweed species (unpublished results R. Taylor & Steinberg). The stipe length of each *E. radiata* was measured (± 1 cm), and the head and stipe were weighed (±1 g) separately after shaking to remove excess water. The abundance, biomass, and productivity of epifaunal individuals over 0.5mm were estimated following Edgar (1990). Edgar's method makes it possible to estimate assemblage-level biomass and productivity by washing samples through a stack of sieves with geometrically decreasing mesh sizes (8, 5.6, 4, 2.8, 2, 1.4, 1, 0.71, and 0.5 mm), and counting the number of individuals trapped on each sieve (individuals trapped on the 8 mm sieve were not included in subsequent calculations as the body size of animals retained on this sieve had no upper bound). Edgar (1990) gave equations that predict body mass of individuals on each sieve, and productivity as a function of body mass and water temperature. The latter equation is based on a compilation of literature-derived data for individual species and is surprisingly precise (log body weight and log water temperature account for 94% of variation in log productivity). The error involved in predicting the productivity of individual species using such methods can be high, but tends to cancel out in assemblage-level estimates (Edgar 1990). Data were analysed and presented for both the entire epifaunal assemblage and for just crustaceans over 1-mm sieve size, as the latter are the favoured prey of most small predatory reef fishes (Edgar & Shaw 1995).

Table 5: Sampling dates for deployment of artificial plants and collection of *E. radiata.*

Substratum	Season	Deployed	Collected
Ecklonia radiata	Winter		11 Jun. 2003
	Summer		20 Jan. 2004
Artificial plants	Winter	1 Apr. 2003	4 & 5 Jun. 2003
	Summer	9 & 10 Dec. 2003	9 & 10 Feb. 2004

3.2.1 Statistical analyses

Among-site differences in density, biomass, and productivity were tested using one-way analysis of variance (ANOVA) with post hoc Tukey's HSD. Data were square-root transformed before these analyses to homogenise variances. Among-site differences in the taxonomic composition of the epifaunal assemblages were visualised using non-metric multidimensional scaling (MDS). MDS creates low-dimensional maps of relationships among sites, where the distance between two points is proportional to their biological dissimilarity, as determined using a dissimilarity coefficient. Each MDS was run on a Bray-Curtis dissimilarity matrix derived from fourth-root transformed density data for all epifaunal taxa. The fourth-root transformation serves to downplay the influence of extremely abundant species (so they don't dominate the analysis) while maintaining the rare species (Clarke & Warwick 1994).

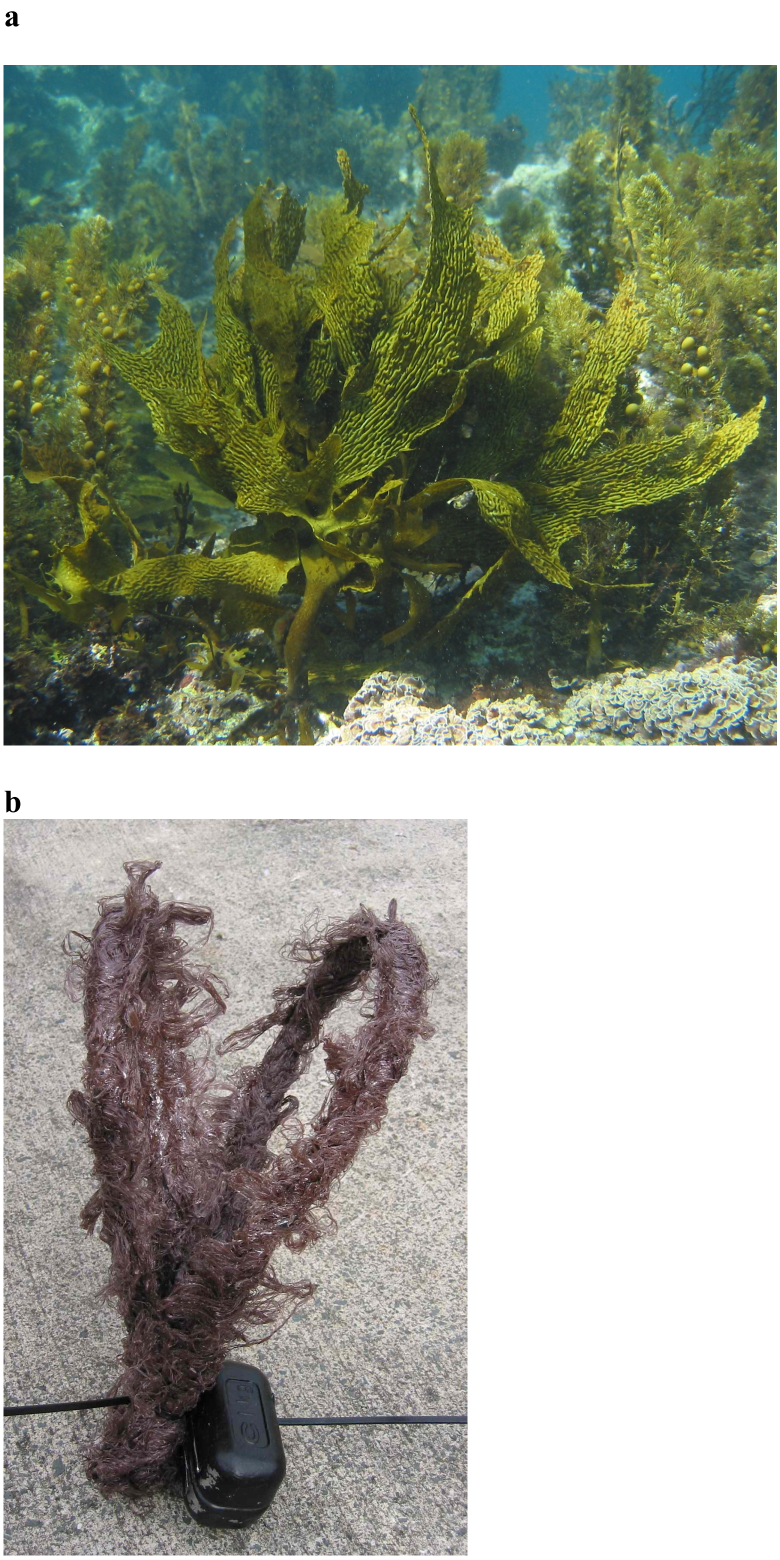


Figure 14: a: *Ecklonia radiata* and b: Artificial plant (photographs by R. Taylor).

3.3 Results

3.3.1 *Ecklonia radiata*

In both winter 2003 and summer 2004 differences between average densities, biomasses, and productivities of mobile epifauna on *E. radiata* were statistically insignificant among sites 2–4, but were about 50% lower at site 1 (Figure 15), which was the site closest to the Whitianga Harbour mouth (see Figure 5) and thus putatively subjected to the most turbid water and the highest rate of sedimentation. Crustaceans over 1-mm sieve size, the component of the epifaunal assemblage that is the primary food source for small fishes, conformed to the same general pattern shown by the epifaunal assemblage as a whole (Figure 16). Harpacticoid copepods and/or gammarid amphipods numerically dominated the epifaunal assemblages at all sites, together making up 78–96% of total individuals (Figure 17). The multivariate MDS revealed some differences in epifaunal community structure among sites (Figure 18), although these results need to be interpreted cautiously as the fit is not good (stress values > 0.2, Clarke and Warwick, 1994). Replicates mostly clustered by site (with some overlap), but the clusters were not ordered with respect to the putative sediment gradient, and the relationships changed somewhat from winter to summer (e.g., the relative positions of sites 2 and 3 switched between seasons). The crude morphology of *E. radiata* did not vary systematically along the sediment gradient (Figure 19), indicating that the seaweed represented a reasonably uniform habitat for epifauna.

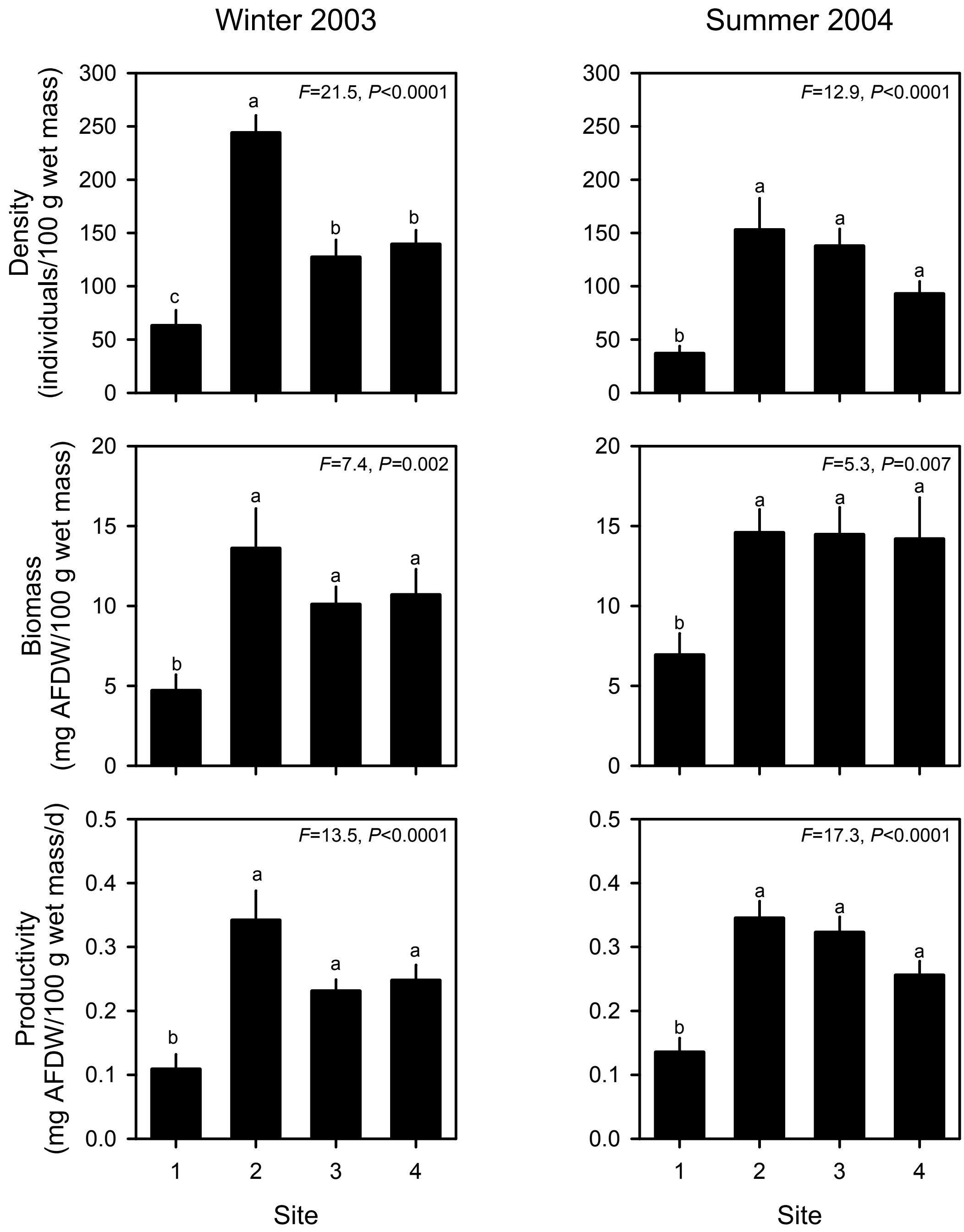


Figure 15: Abundance, biomass, and productivity of total animals on the kelp *Ecklonia radiata* along a gradient of sedimentation at Hahei in winter 2003 and summer 2004. AFDW = ash free dry weight. Higher site numbers are further from a sediment source, the Whitianga Harbour mouth. Bars represent 1 SE. n = 6. Statistics are from one-way ANOVAs on square root-transformed data. Bars sharing same lower case letter do not differ significantly (*P*>0.05) according to Tukey's HSD.

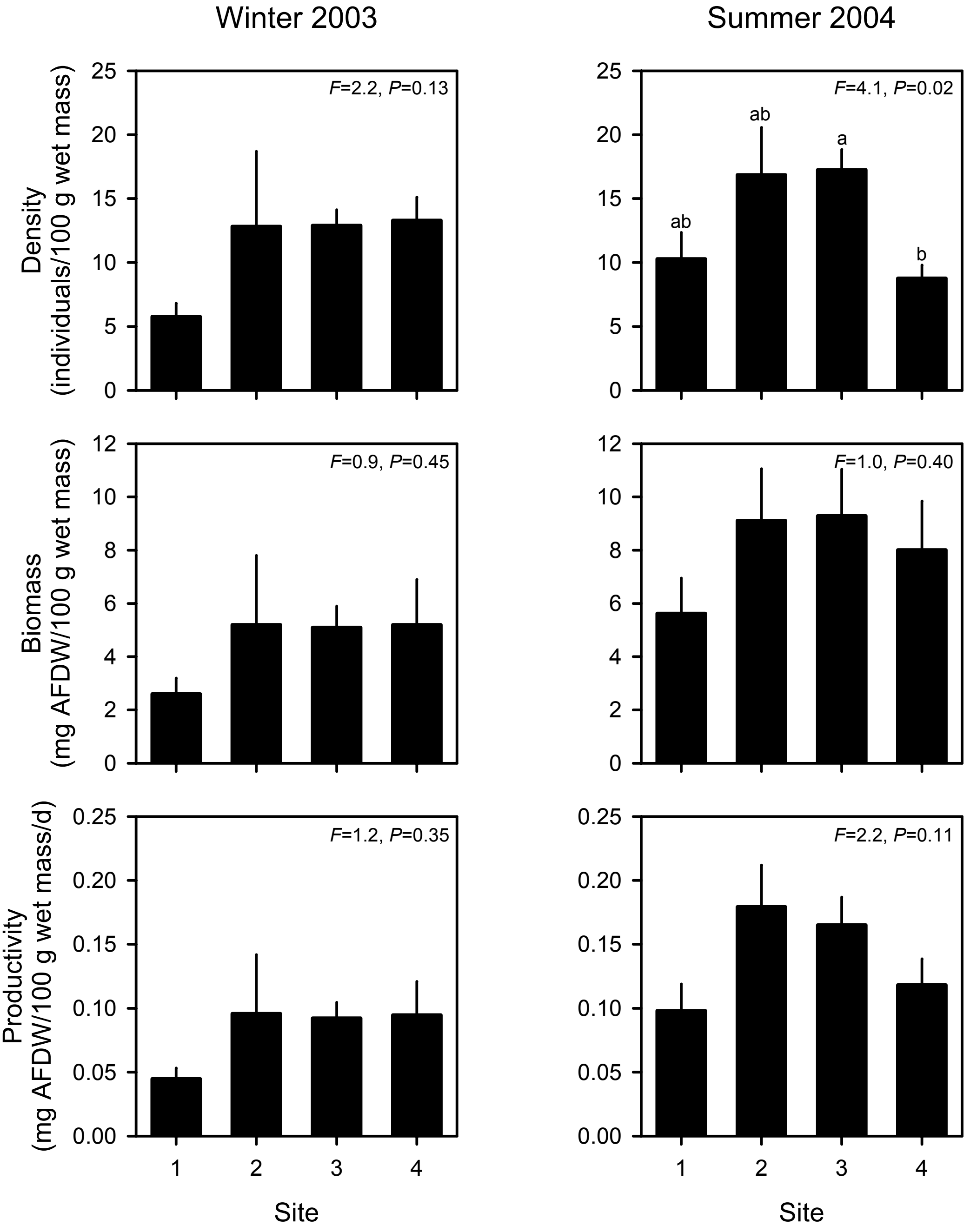


Figure 16: Abundance, biomass, and productivity of crustaceans >1-mm sieve size on the kelp *Ecklonia radiata* along a gradient of sedimentation at Hahei in winter 2003 and summer 2004. Higher site numbers are further from a sediment source, the Whitianga Harbour mouth. Bars represent 1 SE. *n* = 6. Statistics are from one-way ANOVAs on square root-transformed data. Bars sharing same lower case letter do not differ significantly (*P*>0.05) according to Tukey's HSD.

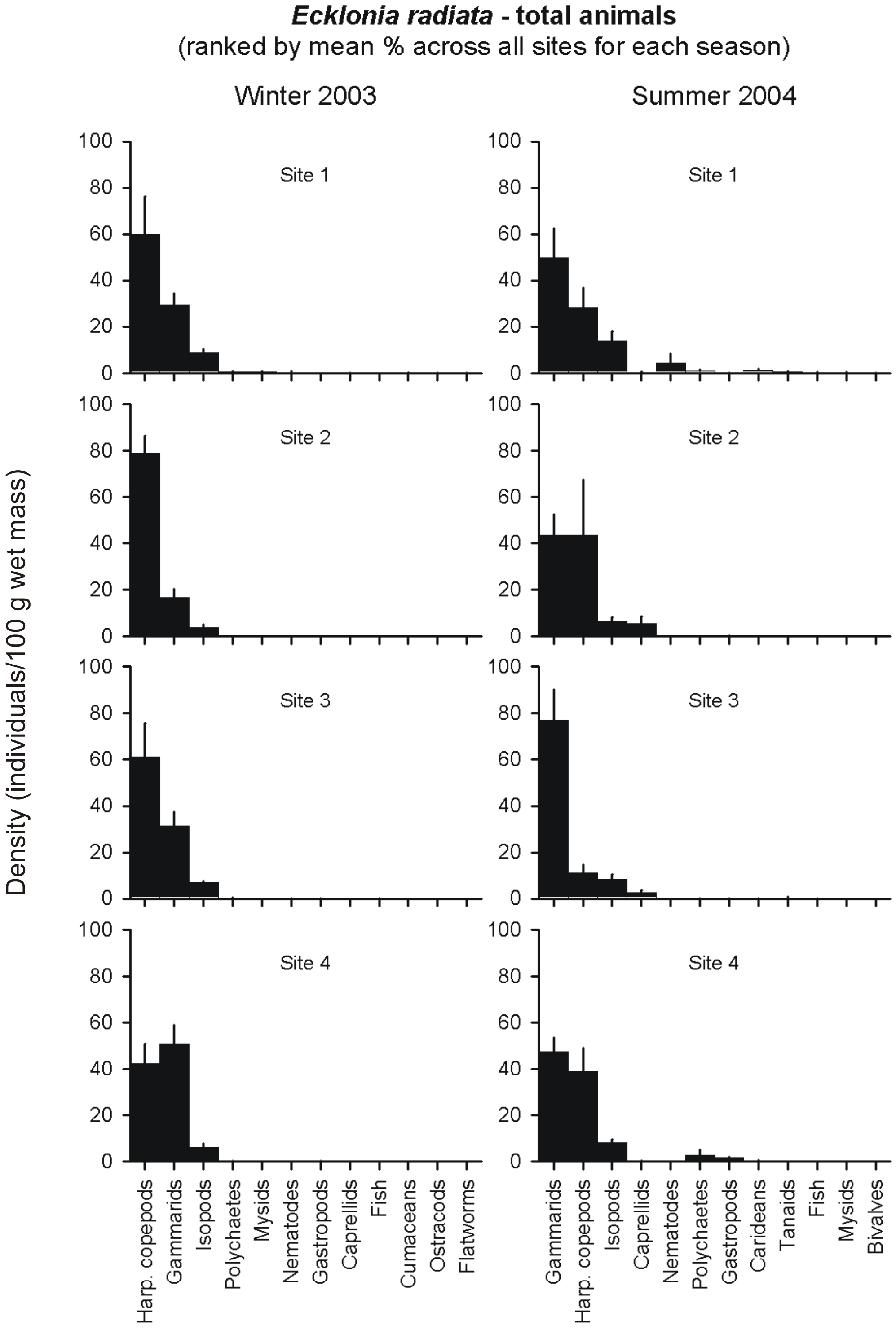


Figure 17: Rank abundances of total animals on the kelp *Ecklonia radiata* along a gradient of sedimentation at Hahei in winter 2003 and summer 2004. Higher site numbers are further from a sediment source, Whitianga Harbour. Bars represent 1 SE. *n* = 6.

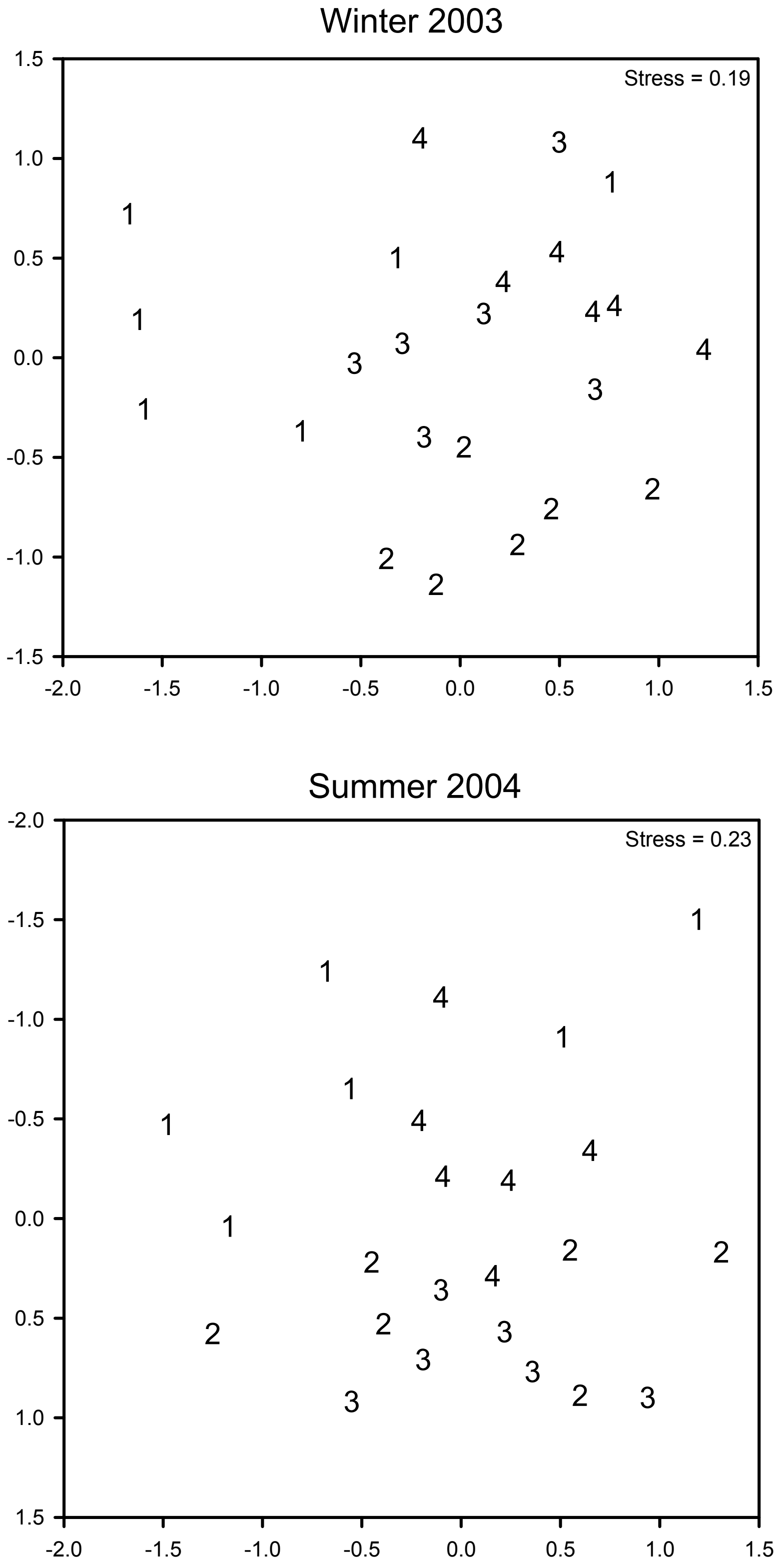


Figure 18: Results of non-metric multidimensional scaling (MDS) analyses of total animals on the kelp *Ecklonia radiata* along a gradient of sedimentation at Hahei in winter 2003 and summer 2004. Higher site numbers are further from a sediment source, Whitianga Harbour. The stress value given for each plot is a measure of the degree to which the MDS was able to accurately represent distances among replicates (values < 0.2 are normally considered acceptable).

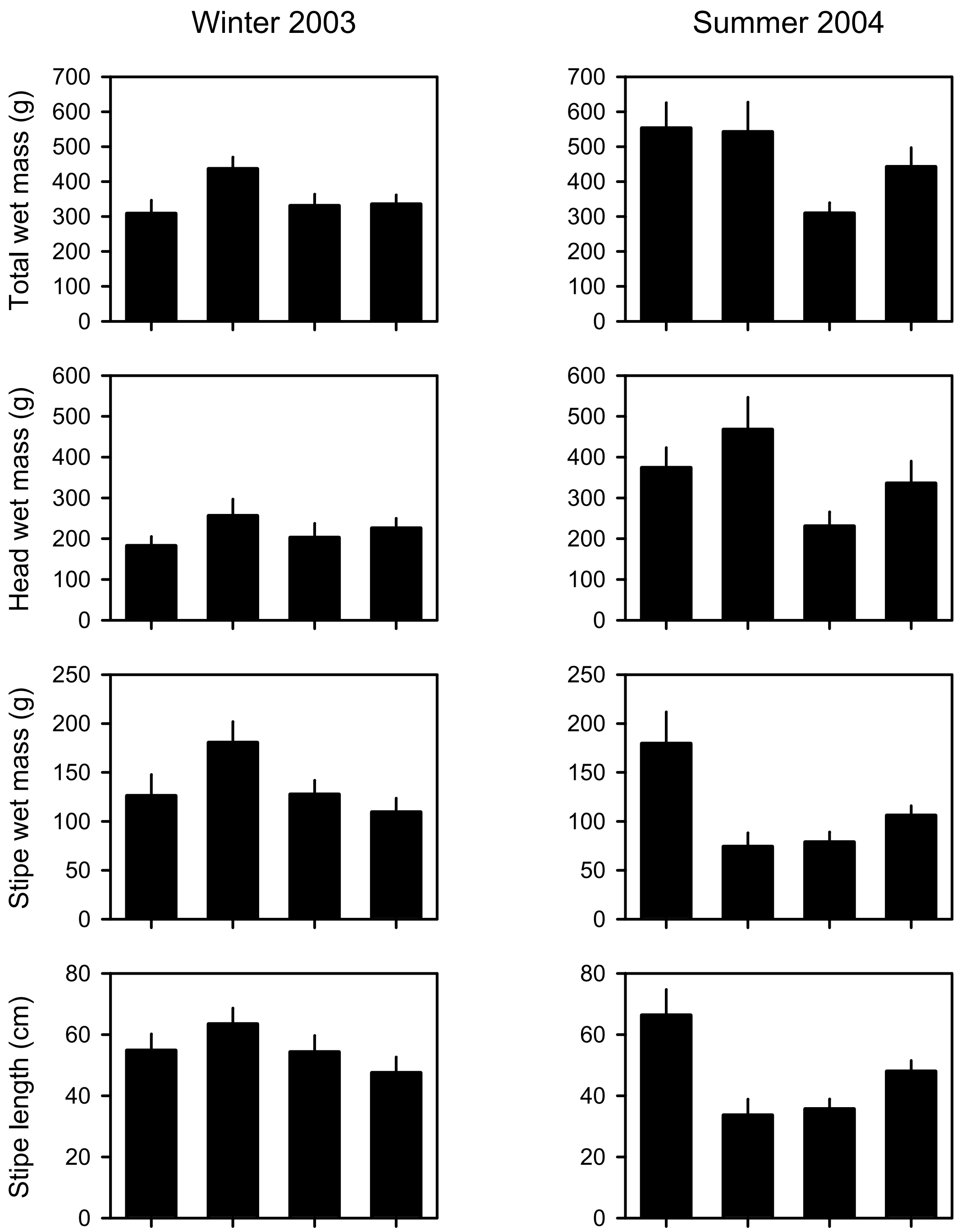


Figure 19: Morphology of the kelp *Ecklonia radiata* along a gradient of sedimentation at Hahei in winter 2003 and summer 2004. Higher site numbers are further from a sediment source, Whitianga Harbour. Bars represent 1 SE. *n* = 6. Total mass = stipe plus head (not including holdfast).

3.3.2 Artificial plants

On the artificial plants, average densities, biomasses, and productivities of total animals did not show consistent trends along the sedimentation gradient between years (Figure 20). In winter 2003, epifaunal biomasses and productivities were highest at site 1 (the opposite pattern to the *E. radiata* epifauna), but in summer 2004 all epifaunal parameters were higher at site 4 than site 1 (artificial plants could not be retrieved from sites 2 and 3 because recreational divers removed all the artificial plants from one site, and the marker buoy was lost from the other site). Crustaceans over 1-mm sieve size also yielded inconsistent patterns – the trend for highest average densities, biomasses, and productivities at site 1 in winter 2003 was not apparent in the summer 2004 data (Figure 21). The artificial plants were colonised by a fauna (Figure 22) that differed in several ways from that on *E. radiata*. The artificial plant fauna included many taxa characteristic of soft sediment habitats, such as ostracods, polychaetes, and bivalves. Harpacticoid copepods, which co-dominated the *E. radiata* epifauna, were very scarce on the artificial plants. Unlike the *E. radiata* epifaunal assemblage, which was dominated by just two taxa, the epifaunal assemblage on the artificial plants was more evenly distributed. Gammarid amphipods, followed by polychaetes, were the most consistently abundant taxa, but their combined density exceeded 50% of total individuals only once. The MDS revealed very clear differences in epifaunal community structure among sites, with no overlapping clusters (Figure 23). Notably site 1 (Cook) is different from sites 2, 3, and 4 consistent with differences in water clarity (Section 2).

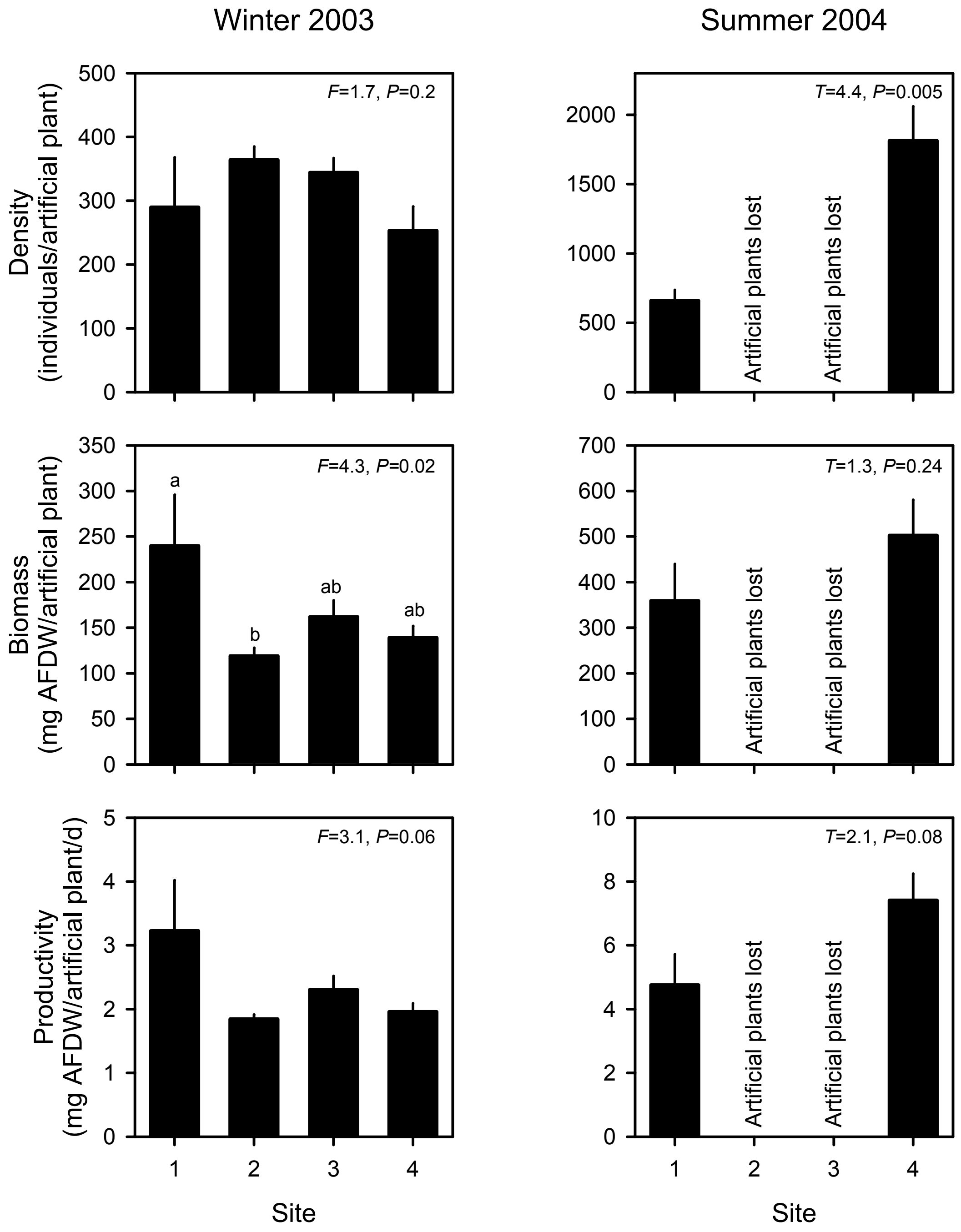


Figure 20: Abundance, biomass, and productivity of total animals on artificial plants deployed along a gradient of sedimentation at Hahei in winter 2003 and summer 2004. Higher site numbers are further from a sediment source, Whitianga Harbour. Bars represent 1 SE. *n* = 3–6. Statistics are from one-way ANOVAs on square root-transformed data. Bars sharing same lower case letter do not differ significantly (*P*>0.05) according to Tukey's HSD.

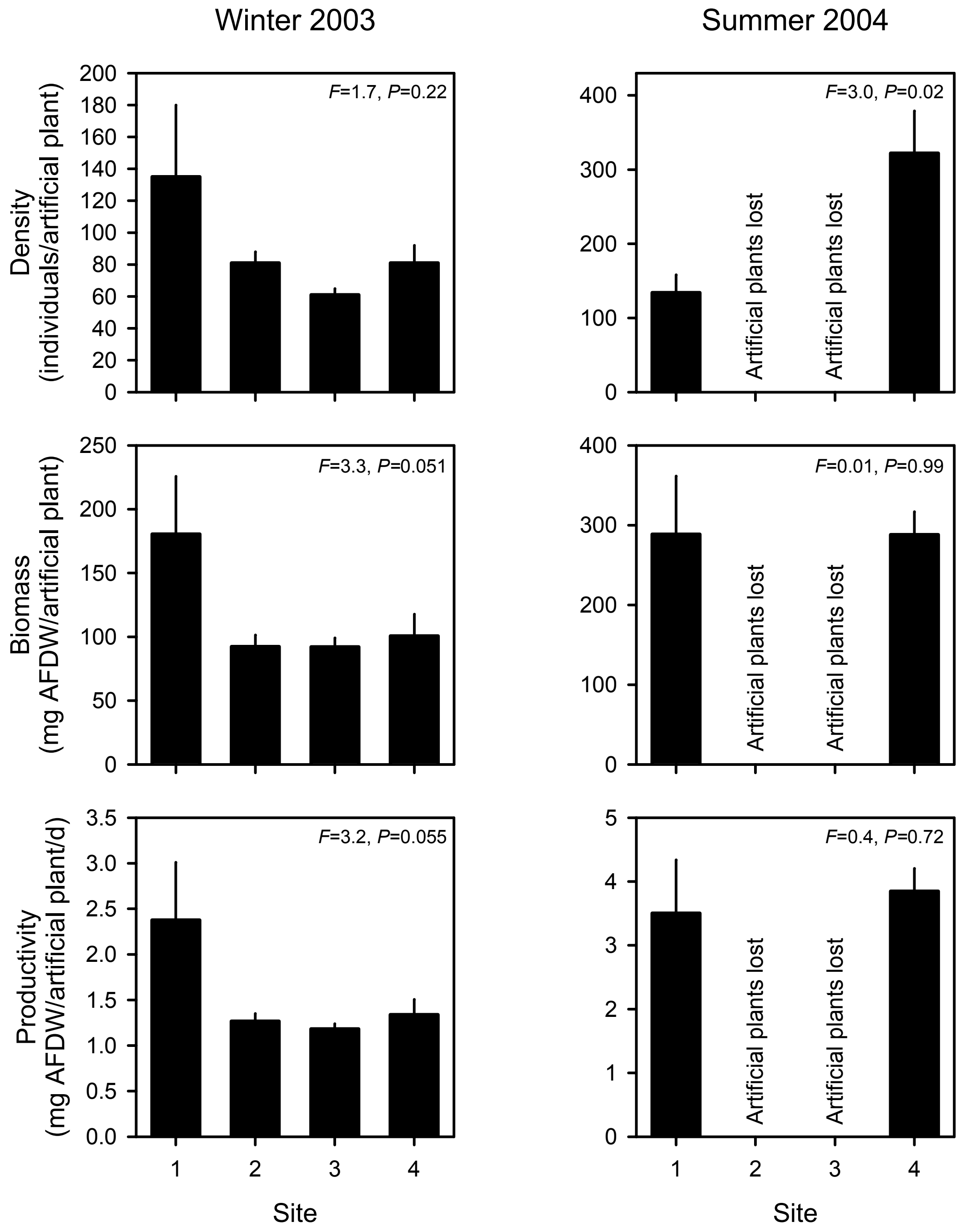


Figure 21: Abundance, biomass, and productivity of crustaceans >1-mm sieve size on artificial plants deployed along a gradient of sedimentation at Hahei in winter 2003 and summer 2004. Higher site numbers are further from a sediment source, Whitianga Harbour. Bars represent 1 SE. n = 3–6. Statistics are from one-way ANOVAs on square root-transformed data. Bars sharing same lower case letter do not differ significantly (*P*>0.05) according to Tukey's HSD.

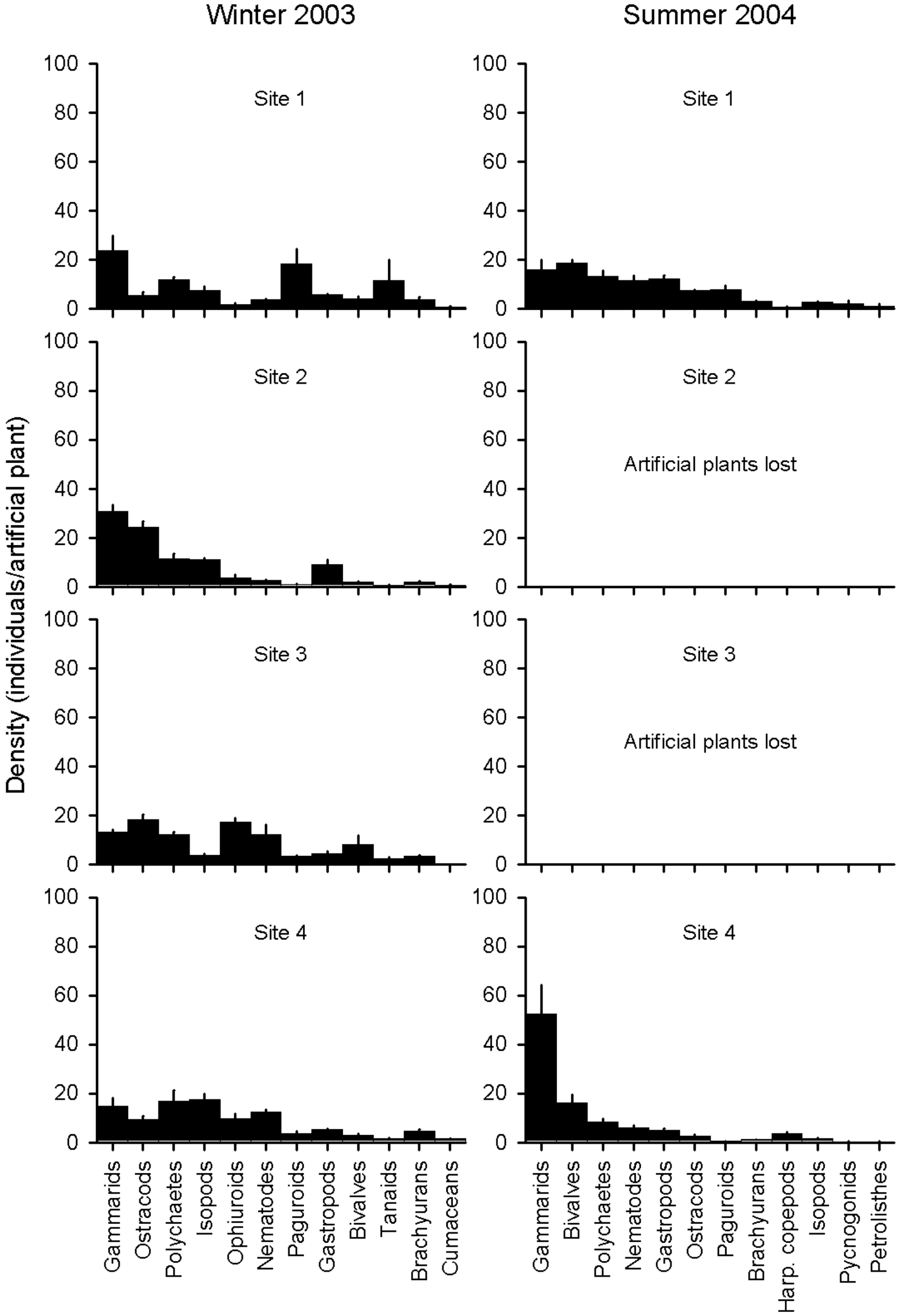
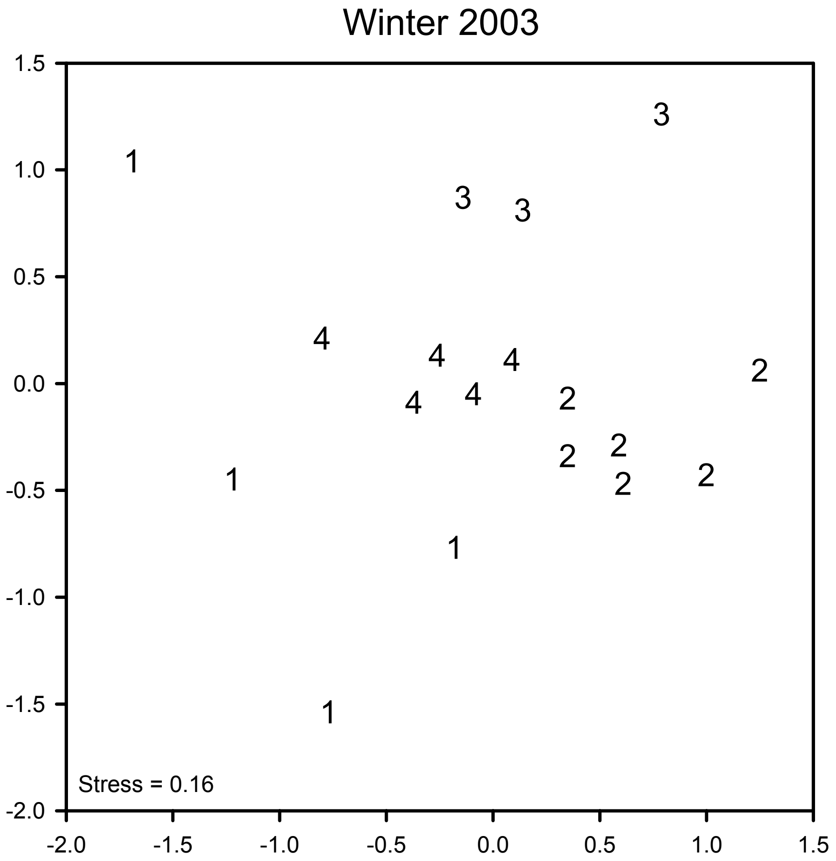


Figure 22: Rank abundances of total animals on artificial plants deployed along a gradient of sedimentation at Hahei in winter 2003 and summer 2004. Higher site numbers are further from a sediment source, Whitianga Harbour. Bars represent 1 SE. *n* = 3–6.



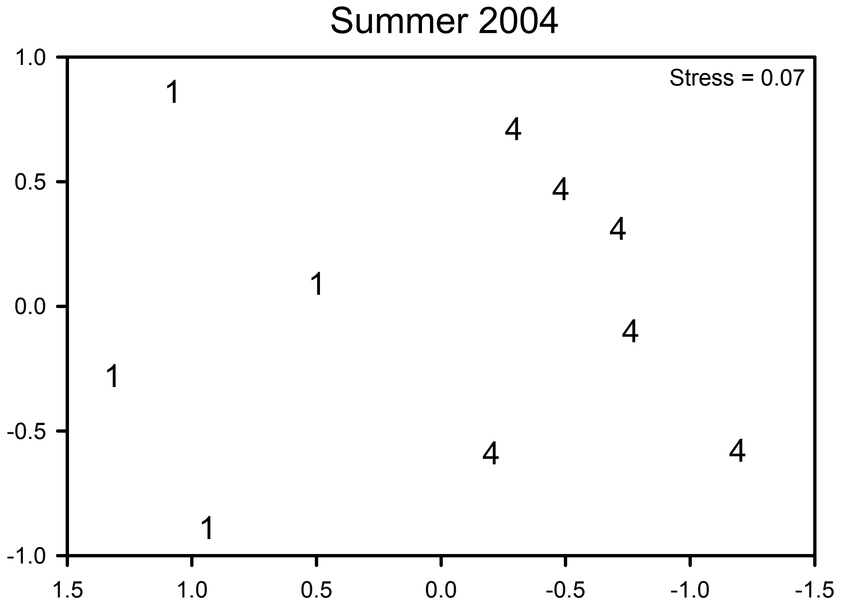


Figure 23: Results of non-metric multidimensional scaling (MDS) analyses of total animals on artificial plants deployed along a gradient of sedimentation at Hahei in winter 2003 and summer 2004. Artificial plants were not recovered from sites 2 and 3 in summer 2004. Higher site numbers are further from a sediment source, Whitianga Harbour. The stress value given for each plot is a measure of the degree to which the MDS was able to accurately represent distances among replicates (values < 0.2 are normally considered acceptable).

3.4 Discussion

Our data are consistent with a negative effect of high levels of sediment (suspended or deposited) on the *Ecklonia radiata* epifauna at site 1, where epifaunal abundance, biomass, and productivity were about 50% lower than at the "cleaner" sites. Determining the mechanism(s) behind this reduction is beyond the scope of the present study, but it is likely that sediment particles clog the feeding apparatus of suspension-feeding epifauna and diminish the productivity of microscopic epiphytic algal food sources by shading and smothering. The reduction in epifaunal productivity has potentially serious consequences at the ecosystem level, since epifauna are responsible for about 80% of the flow of energy and materials through rocky reef animal communities (Taylor 1998). Although the functional role of seaweed epifauna is poorly understood at present, it is likely that density reductions will have effects throughout the rocky reef foodweb, both downwards through reduced epifaunal grazing on seaweeds and algal epiphytes (Duffy 1990), and upwards through reduced availability of food for small fishes (Jones 1988).

The artificial plants were colonised by a fauna that differed substantially from that occurring naturally on adjacent *E. radiata*. The artificial plant fauna included many taxa characteristic of soft sediment habitats, such as ostracods, polychaetes, and bivalves, which were probably responding to the large quantities of fine and coarse sediments that accumulated on the artificial plants (R. Taylor, pers. obs.). Harpacticoid copepods, which co-dominated the *E. radiata* epifauna, were very scarce on the artificial plants. In addition, the animal assemblages on the artificial plants were much more even than those on *E. radiata* (i.e., individuals were more evenly distributed among taxa). The artificial plants comprised a standard habitat that did not confound changes in habitat structure with changes in sedimentation along the gradient, but this advantage was offset by their failure to attract a fauna representative of the adjacent seaweed. Since *E. radiata* dominates the seaweed flora at each reef along the sediment gradient, and did not vary systematically in morphology among sites, it can be considered the most suitable substratum from which to draw conclusions about the effects of sediments on epifauna. A further advantage of *E. radiata* is that the seaweed does not have to be deployed before sampling, and is not vulnerable to removal by humans.

4. LABORATORY EXPERIMENTS TO QUANTIFY THE DIRECT EFFECTS OF SEDIMENTS ON SELECTED SESSILE FILTER FEEDING INVERTEBRATES AND EPIFAUNA

4.1 Introduction

In the first six months of this study a number of fauna were identified in the Whitianga-Hahei rocky reef habitats that we expected could be affected by elevated suspended sediment concentrations. The encrusting suspension-feeding fauna included sponges, bivalves (green-lipped mussels, rock oysters), barnacles and anemones. Anemones and barnacles are difficult to remove from the reefs without damage. Also, we were aware that the effects of sediment on anemones were being investigated as part of a separate Department of Conservation project, so we focused in the first instance on organisms that are known to be amenable to laboratory manipulations (e.g., oysters and mussels). Although sponges can be difficult to keep in laboratories, we conducted preliminary investigations and established that we were able to provide suitable conditions to successfully sample and maintain material from the Whitianga-Hahei reefs for 2 weeks, sufficient time in which to conduct dose-response experiments.

In addition we conducted a dose-response experiment on the amphipod *Aora typica* Kroyer, 1845. *Aora* is a corophid, a super-family whose members generally build small nests or tubes on a range of substrates and harvest suspended particles or material from fine sediment films for food (G. Fenwick, NIWA, pers. comm.). It is a relatively large amphipod and is likely to be a food source for predatory fishes. Individuals were collected from subtidal wrack (predominantly *Xiphophora chondrophylla*) from rocks near Hahei Beach on 19 April 2004 using mask and snorkel. *Aora* was the only amphipod occurring in high densities in the kelp when the experiment was conducted. It is a large amphipod and is likely to be a food source for predatory fish.

Suspended sediment concentrations in surface waters at sites in the Whitianga-Hahei rocky reef habitat were described in Section 2. These set the lower limits of our experimental suspended sediment (SS) concentration range. Only limited suspended sediment data have so far been collected near the seabed around New Zealand. Suspended sediment concentrations in estuaries can increase significantly with sediment runoff during storm events (Dyer 1986, Hicks & Griffiths 1992) and consequently sediment loads can be orders of magnitude higher than normal for short periods (hours to several days). Concentrations of 40–70 mg l^{-1} have been recorded after low rainfall events in Kawau Bay (authors' unpublished data), and in the Marlborough Sounds Fahey & Coker (1992) recorded that ss in surface waters can increase from background levels of 10-20 mg l^{-1} to 1000 mg l^{-1} while flood waters were discharging into the sea.

Concentrations near the seafloor are likely to be higher than those in surface waters; SS concentrations for the laboratory experiments were selected to cover a range from 1-266 mg l^{-1}. Anticipating that *Aora* may be less susceptible to SS (as corophids are found in a variety of sediment types), we added a very high SS concentration (1000 mg l^{-1}) for the set of *Aora* experiments.

4.2 Methods: encrusting suspension feeders

4.2.1 Field collection

On 28 August 2003, sponges (*Tethya burtoni*), green-lipped mussels (*Perna canaliculus*), and rock oysters (*Saccostrea glomerata*) were collected and transferred to the laboratory in Hamilton in dark, cold, bubbled seawater. Twenty-one sponges were collected from Pupuha Rock in about 12 m water depth, and carried to the surface in ziplock bags. Thirty undamaged mussels and about 75 oysters were collected from mooring lines at the eastern end of Hahei Beach and near Kingfish Rock respectively.

4.2.2 Experimental design

The experiment was run in a 16 ^{o}C constant temperature room, with an 11:13 hour light:dark cycle, in 4 l aquaria filled with seawater filtered at 2 μm. Each taxon was exposed to six suspended sediment treatment levels and three replicate aquaria were established for each treatment level. The target SS concentrations were achieved by adding a precalculated amount of a thick sediment slurry to sea-water. Sediment was collected from a roadworks site near Whitianga. The treatments were kept in suspension by bubbling (Ellis et al. 2002).

In the laboratory, animals were cleaned of sediment, measured, and placed in containers representing different size classes (small, medium, and large) of each species. For oysters, clumps of the smaller animals could not be broken up without damaging the animals, so the term “large” could represent a single animal or a clump of smaller individuals. Individuals from each size class were then randomly allocated to a treatment level and replicate block. Treatments were randomly allocated to a position within a block (designated by position within the constant temperature room).

Algae (*Isochrysis* sp., Tahitian strain), a food source for the animals, were added to the aquaria every 12 h and suspended sediment concentrations were maintained at the same time. Water was changed in the aquaria every third day. Separate aquaria containing no animals were set up for each treatment level and used to monitor the amount of suspended sediment naturally settling out of the water. The animals were kept 24 h for acclimatisation before the experiments started.

4.2.3 Experimental measurements

Measurements of feeding rates were made on five occasions over the 13 days of the experiment (days 3, 6, 9, 11, and 13), although here we present data only from days 3, 9, and 13On each occasion turbidity was measured just after the water had been changed and the algae/sediment mix added, and then measured again 1 h later. Suspended sediment concentrations were also measured, and their relationship to turbidity established. Clearance rate (CR) was calculated as the amount of turbidity removed over 1 h, adjusted by the amount of material that naturally dropped out and the dry tissue weight of the animal(s) in the aquarium. The amount of water filtered in 1 h (FR) was calculated as:

$$\text{FR (ml min}^{-1}\text{)} = \text{CR V / Turb}_1 \, (T_2 - T_1) \quad (1)$$

where V is volume of water in aquaria (ml), Turb_1 is the initial turbidity at T_1, and $T_2 - T_1$ is time over which the feeding measurements were made (in this case 60 mins). Again this measurement was adjusted by the dry tissue weight of the animal(s) in the aquarium.

The amount of total particulate matter cleared (CR) is a measure of removal of not just algae but food associated with the sediment and sediment. Thus the measurement is a total of what the animals ingest as well as what they remove from the water column and bind up as pseudofaeces while handling sediment in search of food. Thus CR is not just an estimate of the food removed but of the potential energetic cost. By looking at how both CR and FR change with increased suspended sediment concentration, and over the duration of the experiment, we can determine the potential for elevated SS concentrations to affect the species in longer-term ways such as growth and condition.

4.2.4 Animal condition

Measurements of animal dry tissue were made on three to six individuals at the start and end of the experiment by drying at 60 ^{o}C until constant weight was achieved (about 48 h). These measurements were changed into estimates of condition (or health) by dividing by an estimate of animal size. Diameter was used as a measure of sponge size, and for mussels and oysters the longest internal shell dimension was used. Some suspension feeders (particularly bivalves) will ingest sediment which can cause problems for condition measurements. Rather than dissect out the gut and intestines to remove sediment, we used organic dry weight of the animal instead of the total dry weight in the calculation of condition for the bivalves. Organic dry weight was determined by drying at 60 ^{o}C until a constant weight had been reached, and then ashing in a blast furnace at 400 ^{o}C for 5.5 h. The difference in weights is the organic matter.

4.2.5 Feeding results

Results from the feeding experiments show mussel CRs increased substantially with increasing SS concentration. This increase is seen throughout the 13 days, although decreases were observed over the duration of the experiment (Figure 24a). The size of the decrease from day 3 becomes greater with increasing SS, particularly for SS over 100 mg l^{-1}, suggesting that the mussels are having difficulty with the amount of sediment they need to handle to find food particles at these levels. This supposition is backed by measurements of FR. On day 3, the highest FR was found, as expected, in the controls (Figure 24b). However, after an initial decrease, FR showed a variable response to increasing SS levels.

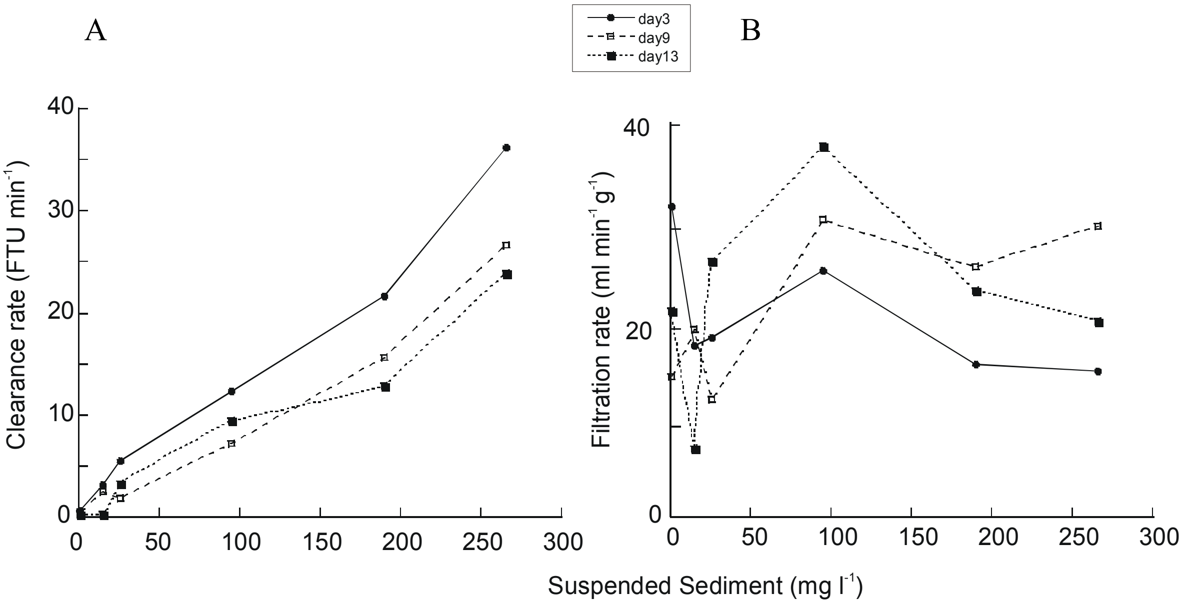


Figure 24: Clearance (A) and filtration rates (B) of the mussels observed in response to different SS levels on days 3, 9, and 13. FTU = standard unit of turbidity.

The oysters had more variable responses in clearance rate than the mussels. CR did not always increase with increasing suspended sediment concentration (Figure 25a), and the rates observed for SS over 100 mg l^{-1} were variable over time. The constant clearance rates observed over time for the controls, also observed on days 9 and 13 by the oysters in the 15 mg l^{-1} treatment, together with the pronounced decrease in FR at 15 mg l^{-1} (Figure 25b) suggests that a clearance rate between 1.5 and 1.8 turbidity units per hour is easily achieved by oysters, and may be optimal for them. Variability of the response of both CR and FR to SS levels over 100 mg l^{-1} suggests that such levels may have long-term effects on oysters.

On day 3, CR of the sponges was higher at each successively higher SS concentration until over 200 mg l^{-1} (Figure 26a). Conversely, there was a general trend of decreasing FR with increasing SS level (Figure 26b). By day 13, CRs were stable only until SS levels of 15 mg l^{-1}, after which they increased (Figure 26a), suggesting that above this level, sponges are having to filter more water to maintain feeding. Although FRs are variable at high SS, on days 9 and 13 a marked decrease can be seen at SS levels over 100 mg l^{-1} (Figure 26b). The drop in FR at this SS level coincides with an increase, by day 13, in FR at SS levels of 266 mg l^{-1}, a strong indication of animal stress at high SS concentration.

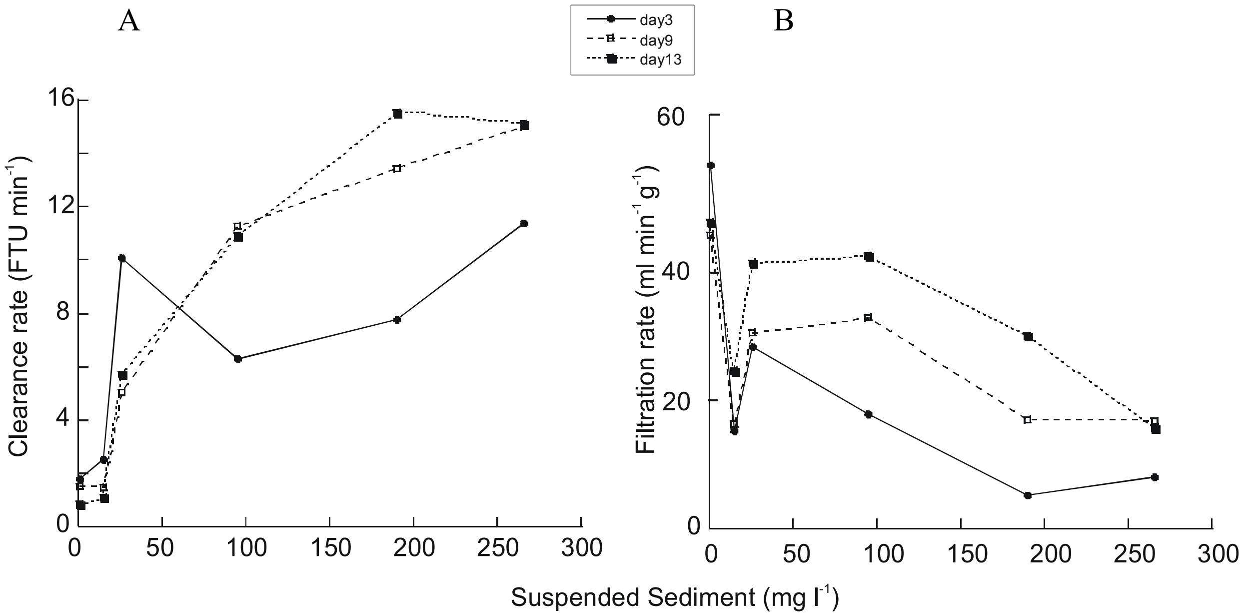


Figure 25: Clearance (A) and filtration (B) rates of the oysters observed in response to different SS levels on days 3, 9, and 13. FTU = standard unit of turbidity.

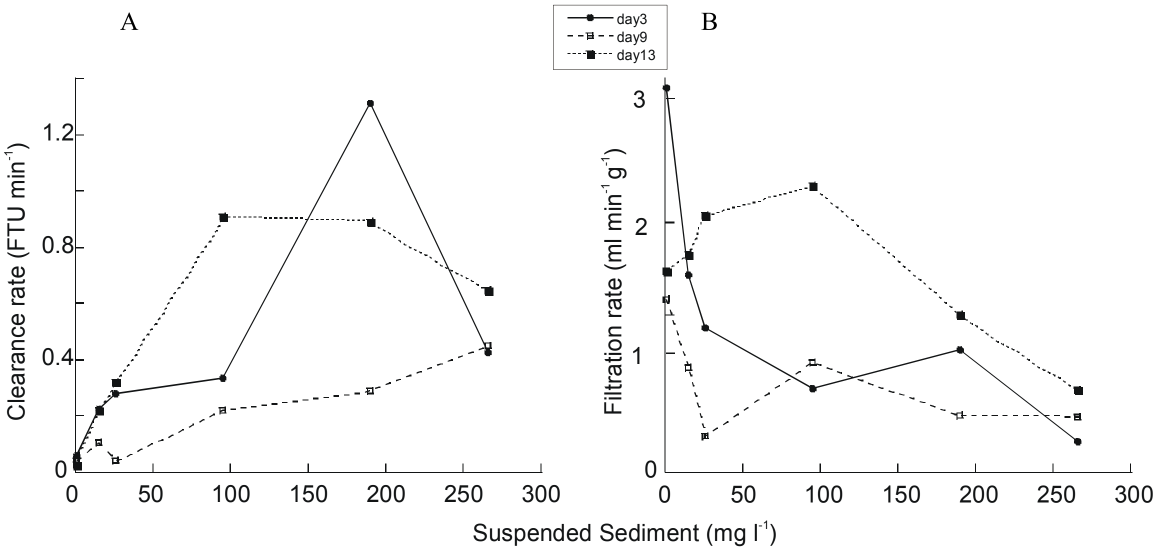


Figure 26: Clearance (A) and filtration (B) rates of the sponges observed in response to different SS levels on days 3, 9, and 13. FTU = standard unit of turbidity.

4.2.6 Condition results

The suggestion that mussels are showing signs of stress when exposed to SS over 100 mg l^{-1} is strongly backed by the mussel condition data (Figure 27a). The mussels exposed to SS over 100 mg l^{-1} for two weeks showed a significant drop in condition from those exposed to SS around 26 mg l^{-1}. This result is expected, as the mussels continue to increase their CR and maintained a high FR with increasing SS.

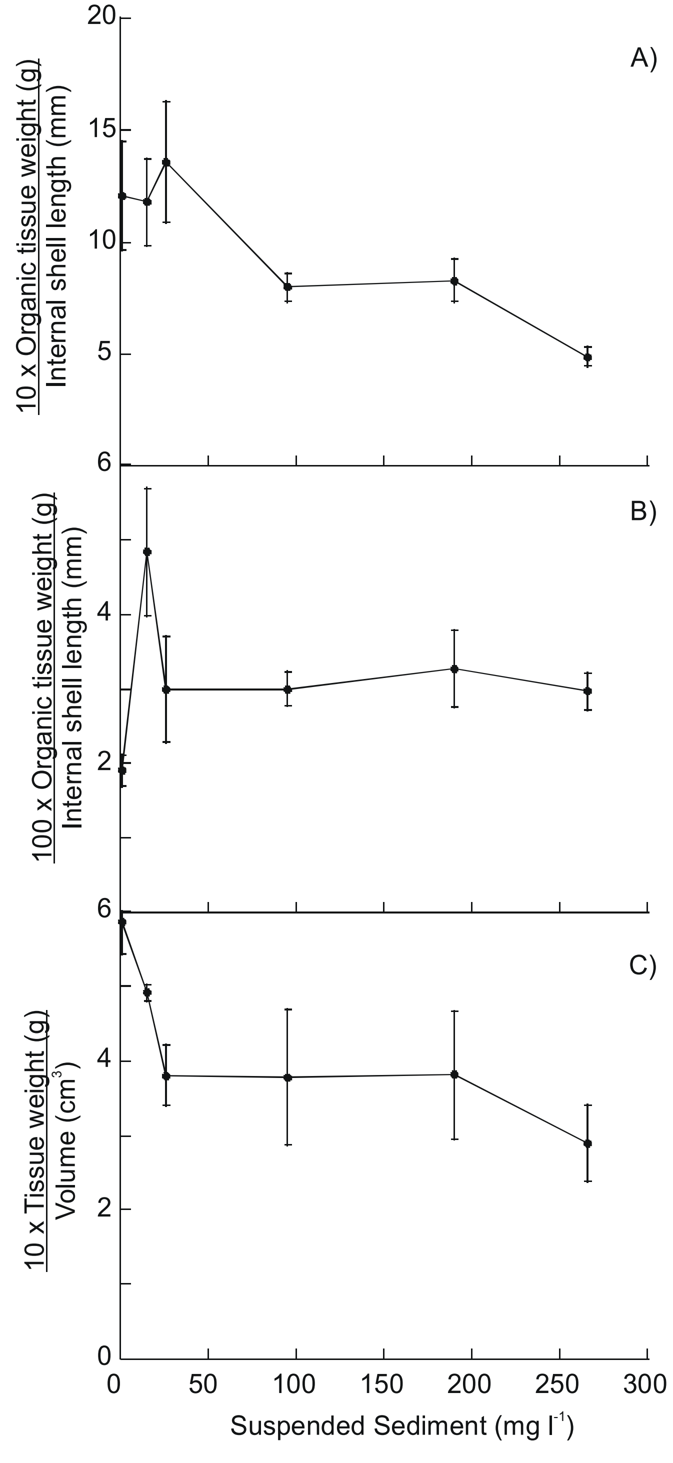


Figure 27: Condition of A) mussels, B) oysters and C) sponges after 13 days in response to different SS levels.

The optimum oyster feeding concentration of 15 mg l^{-1} implied by the feeding experiment coincides with highest or best oyster condition (Figure 27b). Lowest condition was observed in the control oysters.

Sponge condition was also highest at low SS concentrations (Figure 27c), but the highest condition was found for the control treatment.

For all three species, greatest condition was maintained in animals exposed to the lower SS concentration. However, all species responded to increased suspended sediment concentrations differently; with sponges, not unexpectedly, being the most sensitive.

4.3 Methods: Amphipods

4.3.1 Field collection

Individuals of the amphipod *Aora typical* (Figure 27a), were collected from subtidal wrack (predominantly *Xiphophora chondrophylla*) at Hahei Beach (north-facing rocks below the pa site) on 19 April 2004. To minimise handling stress, animals were transported to the lab in chilled, well aerated seawater, together with wrack from the collection site. Total time between field collection and arrival at the lab was 4 h. Animals were acclimated in a temperature-controlled room (16 ± 1 °C) on a 12:12 h light:dark cycle for one day before starting the experiments.

4.3.2 Experimental design

The experiment consisted of a 7-day acute survival test, following general test procedures by ASTM (1993) and Hall & Golding (1998), with some modifications to layout and duration. Turbidity (measured as nephelometric turbidity units, NTU) was used as a substitute measurement of suspended solids concentration. In addition to seawater controls (seawater from Manu Bay, Raglan, filtered through 70 µm mesh), six turbidity levels (nominal turbidity of 3, 9, 15, 35, 60, and 500 NTU) were used. These turbidity levels corresponded to average suspended sediment levels of 1, 15, 95, 190, 280 and 1000 mg.L^{-1} respectively and were prepared by adding different amounts of material from a stock suspension containing 10 g of sediment and 1000 ml filtered seawater. Sediment came from the same roadworks site used in the previous laboratory experiments. Animals were exposed to different turbidity treatments in 200 ml polyethylene jars (10 animals per jar). Each turbidity level was replicated four times; controls were replicated six times. Nominal turbidity levels were maintained by static renewal every 1–2. To assure adequate oxygen supply to animals and to retard settling of suspended clay, treatments were gently aerated with fine (“whisker”) tubing. To minimise aggressive interactions between animals, and to provide an attachment substrate, each jar received a small frond of *Xiphophora chondrophylla*, weighing between 0.5 and 0.8 g, and a 40 x 40 mm piece of plastic mesh. Animals were fed every 2 days with 1 ml of *Isochrysis galbana* culture per jar.

Physicochemical water parameters (turbidity, temperature, salinity, dissolved oxygen, pH, and ammonia) were measured at the start and end of the experiment, as well as before and after each water renewal. Amphipod survival (number of live specimens recovered vs. number added) was determined after 1, 2, 3, 5, and 7 days. For this, amphipods were rinsed from wrack fronds, counted, and transferred to jars containing fresh sediment suspensions. Wrack fragments were subsequently transferred to new suspensions as well.

4.3.3 Results: Amphipods

The survival of controls over the exposure period was acceptable (85–97%), and physicochemical parameters remained within acceptable limits in all treatments throughout the exposure period. All amphipods recovered at the end of the experiment appeared to be healthy and feeding, as evidenced by the large number of faecal pellets on the bottom of treatment jars. Faecal pellets in ss-free control jars consisted primarily of *Isochrysis* detritus, but pellets in high turbidity treatments (Figure 27b) contained a large proportion of sediment and were more numerous, indicative of intensive ingestion of clay.

Average survival was over 65% for all treatments on all sampling occasions. Although survival in all treatments, including controls, dropped over the duration of the experiment, decrease in survival was fastest (and largest) in high suspended sediment treatments (Figure 28). Highest variability in response was observed at 1000 mg.l^{-1}.

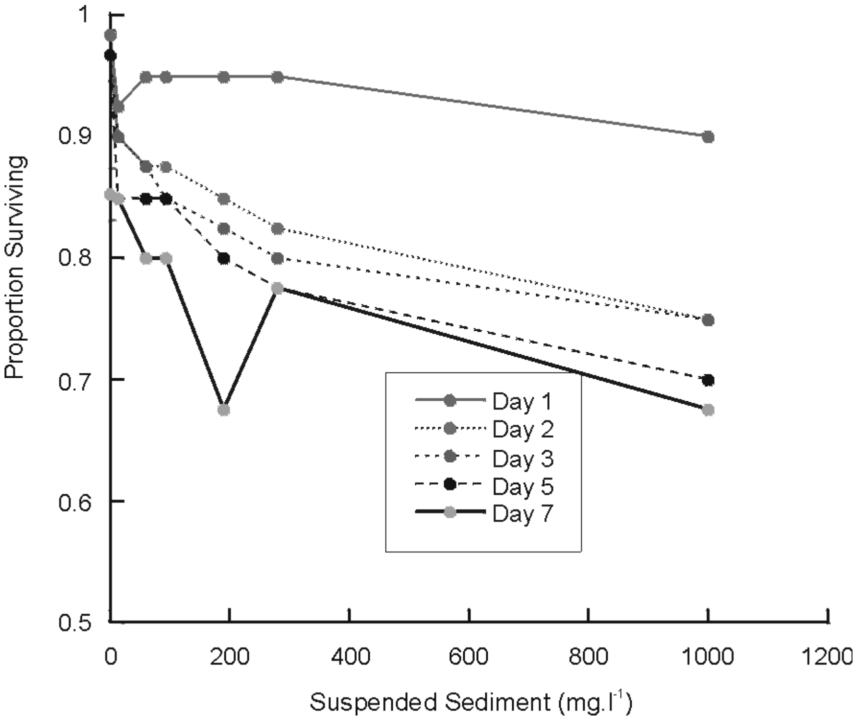


Figure 28: Mean survival of *Aora typica* with increasing suspended sediment concentrations. (Standard errors have been omitted for clarity and are given in Appendix 1).

Mean survival of *Aora* logarithmically decreased with increasing suspended sediment concentrations, and increasing exposure led to increasing sensitivity (Figure 28). An analysis of the relationship between mean survival, ss concentrations (log transformed) and day (by analysis of covariance), revealed that there was a significant relationship between mean survival and ss concentration on each day (Figure 29, Table 6). Survival rate decreased with increased ss concentration as the duration of the experiment increased. While there was also a decrease in survival in the controls that affected this relationship, this did not occur until day 7.

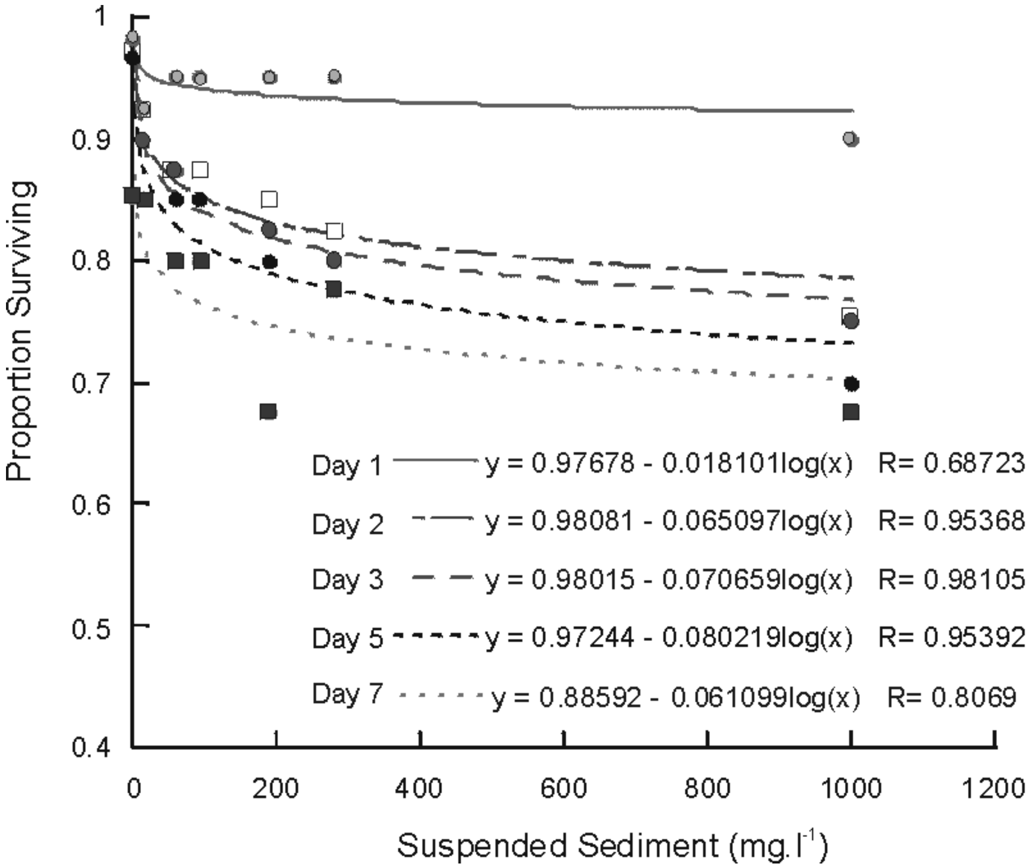


Figure 29: Mean survival of *Aora typica* over time, predicted by suspended sediment concentrations.

Table 6: Results of analysis of covariance on the relationship between mean survival and log transformed suspended sediment concentrations (logSS) on different days.

Source	DF	Mean square	F Value	Pr > F
Model	9	0.0241	29.20	<.0001
Error	25	0.0008		
LogSS	1	0.0989	120.05	<.0001
Day	4	0.0027	3.26	0.0280
logSS*Day	4	0.0034	4.15	0.0103

4.4 Discussion

Sponges, mussels, and oysters all showed signs of physiological stress through altered FR response to elevated suspended sediment concentration.

Mussels showed continued increases in CR with increasing SS, even at the highest SS investigated, thus requiring the use of energy that would otherwise be kept as tissue weight. This response consequently caused a significant drop in mussel condition in SS over 26 mg l-1.

Both oysters and sponges lowered their FR in response to increased SS, resulting in lower nutritional gain and a loss in animal condition from the optimum SS concentration of 15 mg l-1. Oysters showed a functional response to SS over 100 mg l-1 varying their CR and FR.

The amphipod, *Aora typica* showed mortality over the experiment, with decrease in survival being quickest and largest) in high suspended sediment treatments.

5. FIELD AND LABORATORY EXPERIMENTS TO IDENTIFY THE INFLUENCE OF SEDIMENT ON EARLY LIFE STAGES OF KEY GRAZERS.

5.1 Introduction

Terrestrially sourced sediments can have complex direct and indirect effects on rocky reef biodiversity. For example, sediments may directly affect non-geniculate coralline algae which in turn results in reduced recruitment success of key algal grazers. Similarly, development and survival (and thus recruitment) of early-life stages of these key grazer species are likely to be affected directly by terrestrially sourced sediments in coastal waters. If recruitment of such species fails, reef biodiversity is dramatically altered. Because dominant grazers can have a profound influence on algal recruitment, and may themselves be affected strongly by suspended sediments during their early developmental stages, we examined sediment effects on grazer larvae and recruitment success. We conducted a series of laboratory and field experiments involving early life-stages (larvae and juveniles) of paua (*Haliotis iris*) and kina (*Evechinus chloroticus*), and the effects of exposure to suspended sediments (acute and chronic) on the potential recruitment of these key algal grazers.

5.2 Methods

5.2.1 Baseline sediment loads and larval culturing

We quantified ambient suspended sediment loads from 20 l samples of surface water collected at the mouth of the Hutt River, Wellington, New Zealand (41^o 14.9’S 174^o 54.0’E). Samples were collected after rain on 5 May 2003 (2 d after 3 d heavy rain; first large storm event of winter), 13 June 2003 (3 d after 2 d heavy rain), 14 August 2003 (1.5 d after 2.5 d moderate rain) and 22 August 2003 (immediately following 2.5d heavy rain) to characterise average peak sediment loads to Wellington harbour. Samples were pre-filtered through 500 μm mesh to remove large particles and debris, and remaining sediments were allowed to settle undisturbed for 14–20 d before about 19.5 l of water were decanted (subsamples of supernatant seawater were filtered and analysed to ensure no remaining suspended sediments). Samples from 5–30 20 l buckets were then aggregated, rinsed repeatedly to remove salt, dried to a constant weight at 60 ^{o}C, and weighed to calculate ambient sediment per litre seawater.

We estimated an ambient suspended sediment concentration flowing at the mouth of the Hutt River on 5 May 2003 as 0.03677g/l. Based on this, we constructed the following treatments of suspended sediment concentrations for use in our experiments: “½ Ambient” sediment treatment was 0.01839g/l, “1X Ambient” treatment was 0.03677g/L, and “2X Ambient” treatment was 0.07354g/L. Subsequent sampling of suspended sediments in the Hutt River (1.025g/l on 13 June 2003, 0.00688g/l on 14 August 2003, and 0.1468g/l on 28 August 2003) suggest our experimental treatments were well within the range of concentrations near the Hutt River mouth after large rain events.

Sediment treatments used in our experiments were constructed from suspended sediments collected the mouth of the Hutt River on 5 May 2003, subsequently dehydrated, autoclaved (to remove biological contaminants), and proportionately rehydrated using filtered seawater (0.2μm). For each of the following experiments, larval paua were raised at Island Bay Marine Lab. using a system of continuously stirred replicate jars (described Strathmann (1987) and Phillips (2002) maintained at a mean temperature of 12.8 ºC (range: 12.47–13.31 ºC).

5.2.2 Experiment 1. Effects of chronic exposure to suspended sediments: evaluating paua larval development, morphology, and condition

Paua larvae were cultured in densities of 300 l, using three replicate jars for each of the four suspended sediment load treatments (described above). Partial water changes were made at 2–3 d intervals, although sediment concentrations were maintained constant over the entire period of larval development. At each water change interval, we recorded numbers of surviving larvae to estimate effects of sediment treatments on patterns of mortality. To explore the effect of suspended sediments on patterns of larval growth and development, we estimated: maximum shell length; maximum shell width, and aperture diameter, of larval paua subsampled 5–6 d post-fertilisation. All morphometric measurements were analysed separately, using factorial 2-way ANOVA (main effects: “sediment regime” and “jar effect”; Proc GLM, SASv8.02). Response variables (i.e., measurements of individual larvae) were log10 transformed as necessary; all variables met ANOVA assumptions after transformation.

5.2.3 Experiment 2. Chronic exposure to suspended sediments: effects on metamorphosis, and post-settlement paua larvae survival

After experiment 1, competent paua larvae were transferred to sterilised 150 ml glass dishes (Figure 30) containing small cobbles covered with non-geniculate coralline algae (*Lithothamnion* spp.) as a settlement cue (25 larvae/dish). Multiple cobbles were randomly allocated to each dish, as were the larval/juvenile paua treatments. The sediment treatments applied to pre-competent paua larvae were “crossed” with identical treatments applied to the dishes containing paua stages competent to settle and metamorphose into benthic juveniles to test for interactions between chronic exposure to sediments across life history stages. A completely factorial design was not possible due to low rates of larval survival from several of the treatments. Larval treatments of 0X Ambient sediment were crossed with juvenile treatments of 0X, ½X, 1X, and 2X Ambient. The larval treatment of 1X Ambient was crossed with juvenile treatments of 0X and 2X Ambient. We examined the separate and joint effects of chronic exposure to sediments pre- and post-competency on proportion of metamorphic success, and post-settlement survival to 6 days.

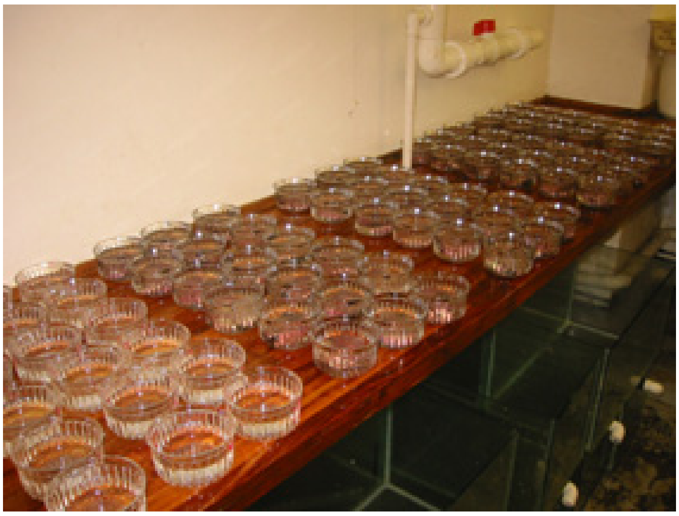


Figure 30: Glass dishes containing *Lithothamnion*-covered cobbles to evaluate effects of sediment treatments on paua settlement and post-settlement performance

5.2.4 Experiment 3. Effects of acute exposure to suspended sediments: the role of timing

Under natural conditions, some larvae may encounter suspended sediment for only a portion of their development. Consequently, we examined the effects of acute exposure to 2X Ambient sediment concentrations administered "Early" in development (1–3 d post-fertilisation) or "late" in development (3–6 d post fertilisation). We randomly allocated larval paua into one of three groups ("early", "late", or a "control" group with no exposure), and reared them in replicate jars (n=3 per treatment). Upon reaching competency, larvae were counted and subsampled to measure morphometric condition indices (maximum larval shell length, maximum larval shell height, larval shell aperture width). Larval treatments were crossed with post-competent/juvenile exposure to sediments, in dishes containing cobbles with non-geniculate coralline algae, and analysed as in Experiment 2 (except here a fully factorial design was possible).

5.2.5 Experiment 4. Exposure to suspended sediments under field conditions

Because the lab experiments described above isolate the effects of Hutt River sediments from other environmental features (e.g., salinity gradients, temperature gradients) that may co-vary with sediment loads (potentially ameliorating or exacerbating the realised effects of sediments on larval development and recruitment), we transplanted cultured paua larvae to field enclosures. We constructed enclosures of 15 cm diameter PVC pipe, cut into 11 cm-lengths, capped at both ends with 150 µm nylon mesh to allow free flow of water and suspended sediments while retaining larvae in densities of about 300 larvae/l. At each of six sites, we deployed three replicate enclosures to steel Y-posts anchored in about 4 m water depth, and about 1.5m off the substrate. Because our first deployment (of 7 d duration) (precluding meaningful estimates of variation in mortality rates), we repeated the experiment at only three sites and for a shorter duration (2 d). This resulted in sufficient survivors to allow estimates of variation in larval mortality and developmental abnormalities as a function of distance from the Hutt River mouth. Numbers were not sufficient to explore patterns of metamorphosis and/or post-settlement success.

5.2.6 Experiment 5. Effects of acute and chronic exposure to suspended sediments: larval development, condition, and survival in the longer-lived feeding larvae of kina

Because kina produce large clutches of larvae that must feed throughout their protracted development, we conducted a final experiment to explore the effects of acute and chronic exposure to sediments on larval development/condition and survival to metamorphic age. We spawned kina larvae from a single spawning pair of adults, (induced to spawn on 13 January 2004 by injecting adults with 1–4 ml 0.5M KCl). Larvae were reared in experimental sediment treatments and with 8000 cells/ml *Dunaliella* added as food at each water change (2–3 d intervals). We used the same set of "chronic" and "acute" treatments as detailed above; "early" acute treatments were administered 2-8 d post-fertilisation. "Late" acute treatments were administered 14–20 d post fertilisation. At each water change interval, we recorded numbers of surviving larvae to estimate effects of sediment treatments on patterns of mortality. To explore the effect of suspended sediments on patterns of larval growth and development for larvae at the 6-armed and 8-armed stage we estimated mid-body length (ML), post-oral arm length (PO), posterio-dorsal arm length (PD), relative PO length (PO:ML), relative PD length (PD:ML), percent bilateral asymmetry PO, and PD of larval kina subsampled. Results were analysed as for Experiment 1.

5.3 Results

5.3.1 Effects of chronic exposure to suspended sediments: evaluating paua larvae development, morphology, and condition

Chronic exposure to suspended sediments, in all concentrations used, had strong negative effects on larval paua survival (Figure 31), and, cumulatively, resulted in a statistically significant doubling of mortality (ANOVA model: arcsine-sqrt transformed cumulative mortality = sediment treatment; $F_{3,11}=8.24$, $p=0.0079$; Proc GLM, SAS v8.02). Chronic exposure to suspended sediments did not have any significant effect on maximum larval shell length, maximum larval shell height, or larval shell aperture width recorded from individuals surviving to 5–6 d post-fertilisation (Table 7A).

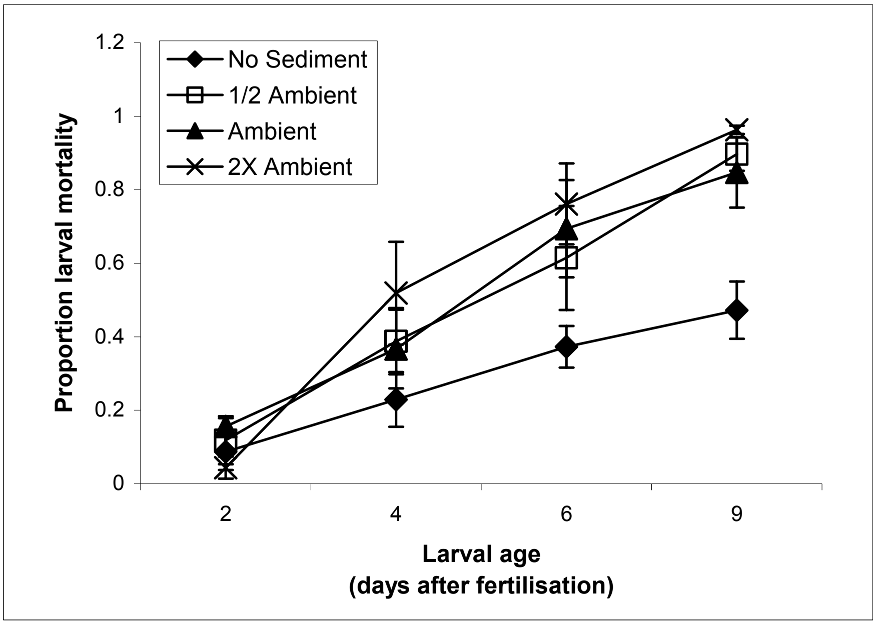


Figure 31: Proportional mortality (mean ± standard error) of larval paua reared under different chronic exposures to suspended sediments. Ambient sediment concentration was 0.03677g/l.

Table 7: Effects of suspended sediment regime (concentration and timing of exposure) on growth and morphology of paua (*Haliotis iris*) 5-6 d post-fertilization, and kina (*Evechinus chloroticus*) larvae at 2 stages of development. Given are pooled results from Experiments 1, 3, and 5, of main effects ("sediment regime", "jar effect") and interactions. Species evaluated in separate ANOVA models. Significant effects in bold.

Response variable	Sediment regime F (p-value)	Jar effect F (p-value)	Interaction F (p-value)
A. Effects on larval paua			
Shell length	2.13 (0.69)	0.10 (0.81)	0.58 (0.07)
Shell width	1.86 (0.11)	0.37 (0.69)	**3.04 (0.002)**
Aperture diameter	60 (0.06)	0.22 (0.90)	1.77 (0.82)
B. Effects on larval kina			
6-armed stage			
Mid-body length (ML)	1.32 (0.27)	0.03 (0.97)	0.85 (0.55)
Mean post-oral arm length (PO)	7.33 (<0.0001)	0.79 (0.46)	1.58 (0.16)
Mean posterio-dorsal arm length (PD)	1.46 (0.22)	0.73 (0.49)	0.48 (0.85)
Relative PO length (PO:ML)	**3.08 (0.02)**	0.55 (0.58)	0.89 (0.52)
Relative PD length (PD:ML)	1.68 (0.17)	0.31 (0.73)	0.89 (0.76)
% Bilateral asymmetry PO	0.57 (0.68)	0.13 (0.88)	1.06 (0.40)
% Bilateral asymmetry PD	0.26 (0.90)	2.99 (0.06)	1.72 (0.13)
8-armed stage			
Mid-body length (ML)	2.53 (0.051)	1.94 (0.15)	0.89 (0.52)
Mean post-oral arm length (PO)	1.76 (0.15)	1.09 (0.34)	1.24 (0.30)
Mean posterio-dorsal arm length (PD)	2.01 (0.11)	0.59 (0.56)	1.94 (0.08)
Relative PO length (PO:ML)	0.67 (0.62)	0.07 (0.93)	0.87 (0.54)
Relative PD length (PD:ML)	2.52 (0.052)	0.22 (0.81)	**2.97 (0.01)**
% Bilateral asymmetry PO	0.72 (0.58)	1.02 (0.37)	0.21 (0.98)
% Bilateral asymmetry PD	1.83 (0.14)	0.22 (0.81)	0.98 (0.47)

Notes: Where required, response variables have been log10 transformed to meet ANOVA assumptions. F-ratios are calculated from Type III SS.

5.3.2 Chronic exposure to suspended sediments: effects on metamorphosis, and post-settlement survival of paua larvae

Individuals chronically exposed to ambient concentrations of suspended sediments as larvae showed a non-significant trend towards elevated post-larval mortality (pooled across juvenile sediment regimes, juvenile mortality rates increased by 4%, Figure 32a). Overall trends (Figure 32, A) suggest that patterns of post-larval mortality of paua may be non-linear functions of sediment concentrations experienced in the juvenile stage, and that these functions may be dependent upon larval experience (i.e., patterns of variation suggest possible interactions between juvenile and larval sediment regimes). However, our experiments do not have sufficient power to discern these potential effects. We present a combined analysis of Experiments 2 and 3 in Table 8. Results of this two-way ANOVA suggest a significant effect of larval experience on post-larval mortality (i.e., a “carry-over” effect). This pattern appears to be driven largely by consistently high rates of post-larval mortality among individuals exposed to sediments early in larval development (i.e., “early acute”), and low rates of post-larval mortality among individuals exposed to sediments late in larval development (“late acute”).

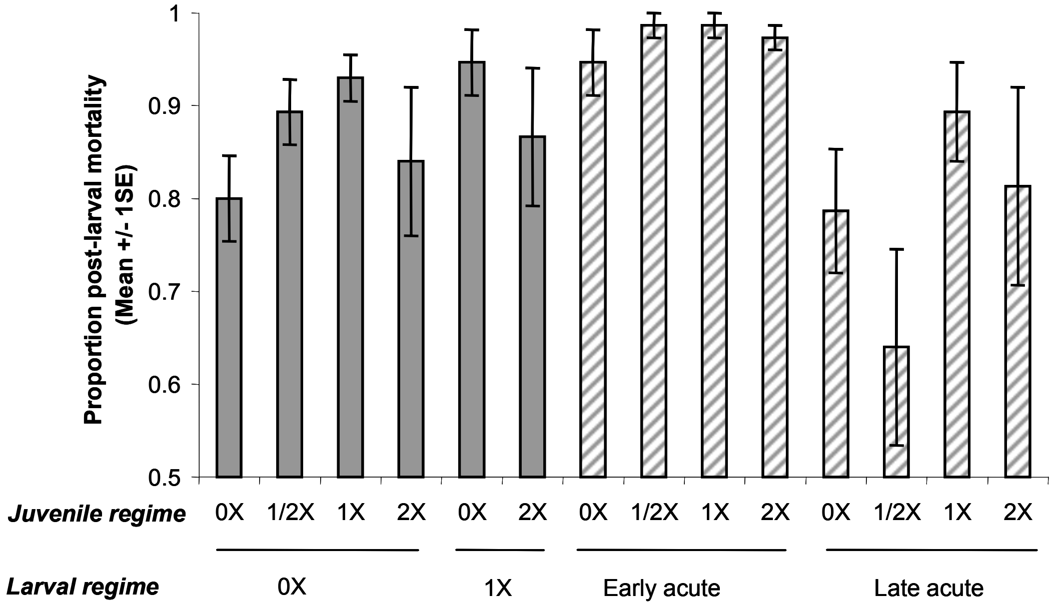


Figure 32: Proportional post-larval mortality of paua reared as larvae under chronic exposure to sediment at 0X or 1X concentrations (A) or acute exposure to 2X concentrations early or late in larval development (B). Larval sediment regimes were crossed with chronic exposure to sediments at 0X, 1/2X, 1X, or 2X concentrations in the post-larval stage to examine carry-over effects of larval experience on juvenile performance, and possible interactions between exposures to sediments across different life-stages. ANOVA statistics given in Table 8.

Table 8: Two-way ANOVA exploring the effects of suspended sediments on the proportion of larvae successfully settled (square-root arc sine transformed). Larval stages were exposed to 0, ½ Ambient, 1X Ambient, or 2X Ambient sediment concentrations in pre-competent stages and post-competent stages (i.e., at an age capable of settlement and metamorphosis) in a factorial experimental design. Pre-competent 2X Ambient treatments consisted of either chronic or acute exposures (see Methods).

Source	DF	Sum of squares	Mean square	F-value	Pr > F
Model	13	0.76574537	0.05890349	2.17	**0.0434**
Error	27	0.73292564	0.02714539		
Corrected total	40	1.49867101			

Source	DF	Type III SS	Mean square	F-value	Pr > F
Pre-competency sediments	3	0.37140588	0.12380196	4.56	**0.0104**
Post-competency sediments	3	0.05730003	0.01910001	0.7	0.5582
Pre- X post-competency sediments	7	0.31960857	0.04565837	1.68	0.1558

5.3.3 Effects of acute exposure of paua larvae to suspended sediments: the role of timing

Paua showed a strong delayed response to sediments administered early in development. Exposure 1–3 days post-fertilisation resulted in elevated mortality 4–6 days post-fertilisation (Figure 33). Acute exposure to sediments late in larval development produced more immediate, but less dramatic, effects. Cumulative patterns of mortality (i.e., assessed 9 d post-fertilisation) suggest that larvae exposed to suspended sediments early in development suffer greater losses than those exposed to sediments later in development (ANOVA model: arcsine-sqrt transformed cumulative mortality = sediment treatment; $F_{2,8}$=3.91, p=0.0818; Proc GLM, SAS v8.02). Acute exposure to suspended sediments had little effect on patterns of larval development ("acute" treatments were included in analyses summarised in Table 7). Consequences for post-larval survival are given in Figure 32b (see also Table 8).

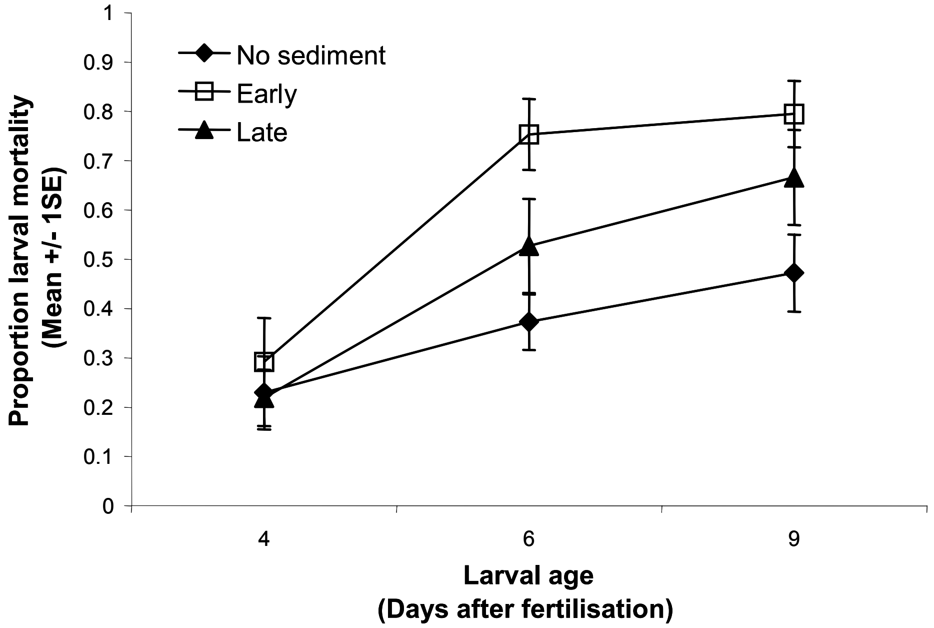


Figure 33: Proportional mortality of larval paua reared under different acute exposures to suspended sediments. Ambient sediment concentration = 0.03677g/l.

5.3.4 Exposure to suspended sediments under field conditions (paua)

Larval paua transplanted to field sites in enclosures for 2 days and subsequently returned to filtered seawater in the lab showed non-significant variation in larval mortality rates (Figure 34A, ANOVA model: arcsine-sqrt transformed cumulative mortality = site; $F_{3,11}$=2.24, p=0.1608; Proc GLM, SAS v8.02) and significant variation in developmental abnormalities (Figure 34B, ANOVA model: arcsine-sqrt transformed proportion abnormalities = site; $F_{3,10}$=5.19, p=0.0337; Proc GLM, SAS v8.02). Mortality rates and developmental abnormalities of larval paua were generally high in the harbour, and mortality rates tended to decrease with distance from the Hutt River mouth. Estimates of suspended sediment concentrations among the three field sites suggest that the presence of a sediment gradient near the river mouth is temporally variable (Figure36).

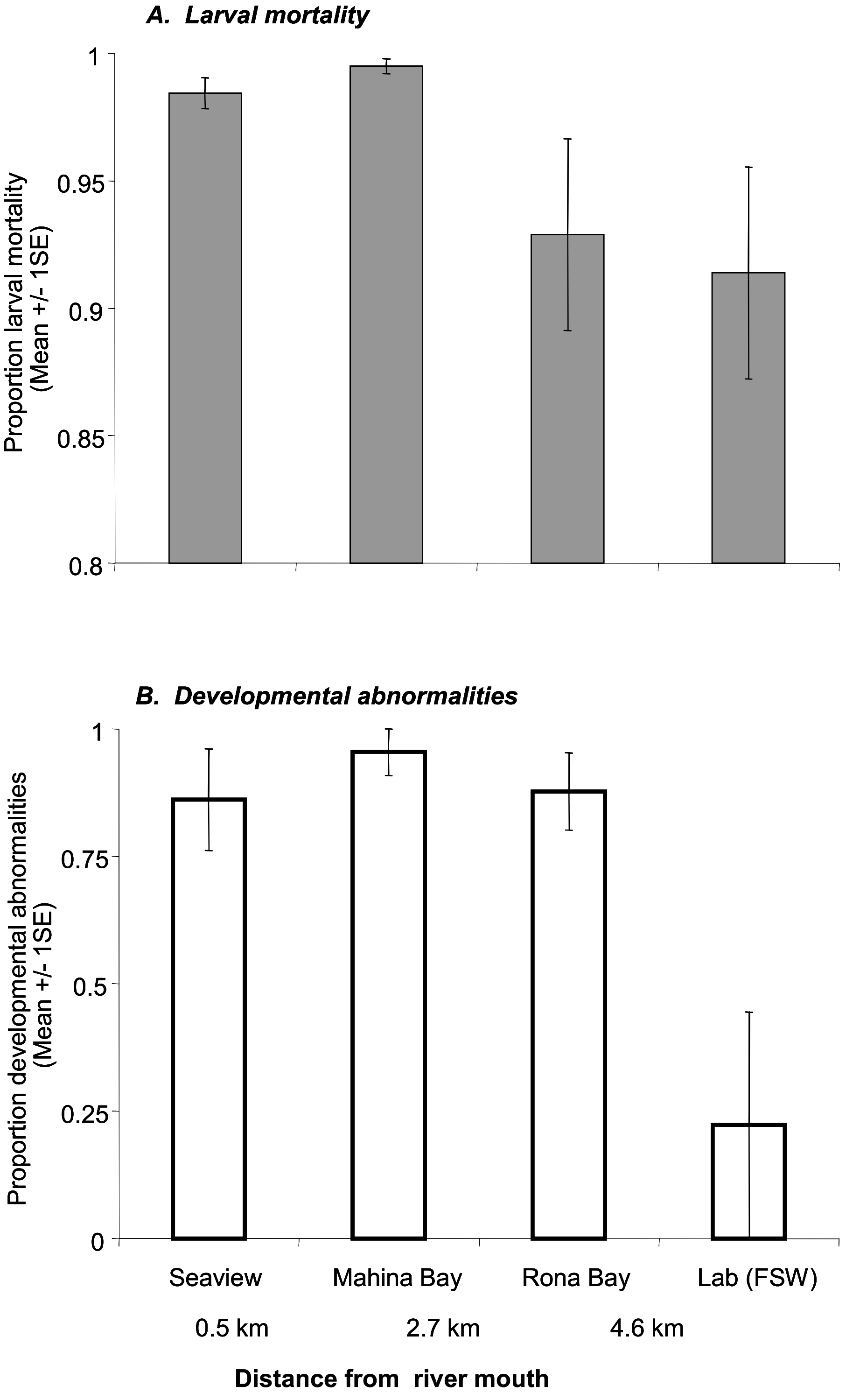


Figure 34: Patterns of mortality and developmental abnormalities of larval paua deployed to three sites at various distances from the Hutt River mouth, and in deployment controls maintained at the Island Bay Marine Lab.

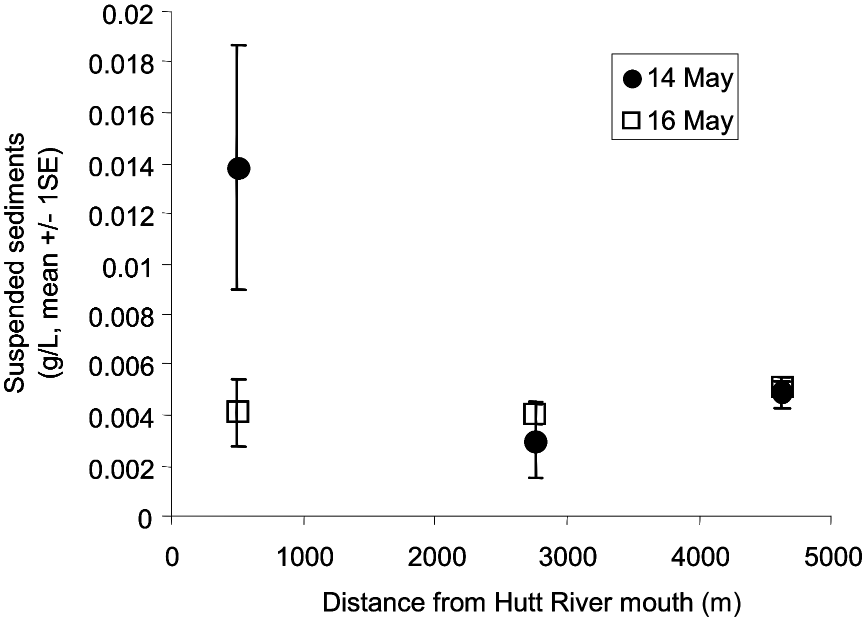


Figure 35: Suspended sediment concentrations at three experimental sites with distance from the Hutt River mouth. Samples were taken at deployment and recovery of larval paua enclosures.

5.3.5 Assessing effects of acute and chronic exposure to suspended sediments: larval development, condition, and survival in the longer-lived feeding larvae of kina

Exposure to suspended sediments had significant negative effects on larval kina survival (Figure 36, ANOVA model: arcsine-sqrt transformed cumulative mortality = sediment treatment; $F_{5,15}$=6.90, p=0.0049; Proc GLM, SAS v8.02). Larval kina that were exposed to sediments early in development showed an immediate but elevated mortality, this effect was not propagated through to older stages (i.e., no evidence for carry-over effects as seen for paua, cumulative mortality was ultimately lower than chronic 2X ambient treatment, *post hoc* Tukey test). Late exposure to sediments resulted in cumulative mortality that did not differ from larvae that were chronically exposed to sediments. Larvae chronically exposed to suspended sediments at 1/2 ambient and ambient concentrations showed responses (mortality rates through development and cumulative mortality) that were intermediate to 2X and 0X treatments. Chronic exposure to suspended sediments also had small but significant effects on two morphological traits at the 6-armed stage, post-oral arm length and relative post-oral arm length, but these effects were transient and did not persist to further stages (see Table 7B).

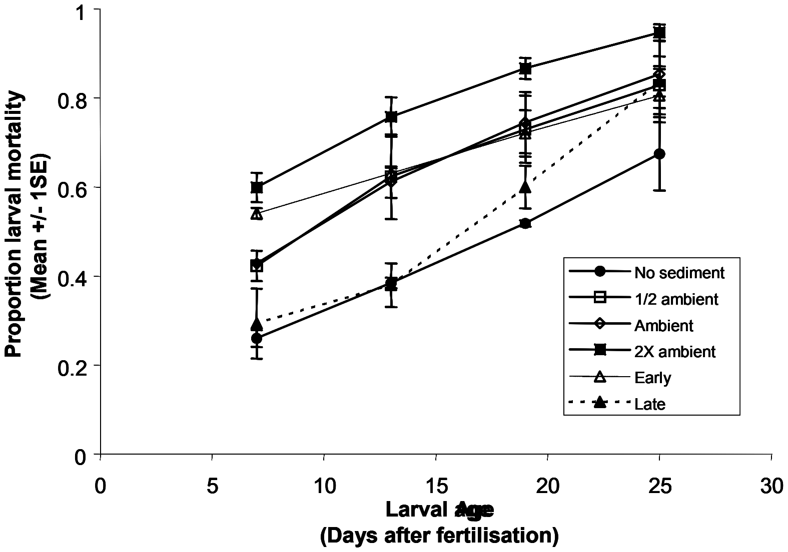


Figure 36: Proportional mortality of larval kina reared under different acute exposures to suspended sediments. Ambient sediment concentration = 0.03677g/l.

5.4 Discussion

Larval paua and kina were both adversely affected by suspended sediments. Despite fundamental differences in life-history strategies, morphologies, and larval durations, sediments increased larval mortality rates of both.

- Paua showed greater sensitivity to suspended sediments. The same range of concentrations increased cumulative mortality at larval competency by up to 49% for paua, and up to 27% for kina. Paua showed similar responses to chronic exposure to suspended sediments across the range of tested concentrations; kina showed a more graded response.

- Effects of timing of exposure differed between species. For paua, early acute exposure was highly deleterious, resulting in greatly increased larval mortality that was manifested in subsequent stages (including some suggestion that consequences may “carry-over” to juvenile stages). Kina exposed to sediments early in development showed some capacity to recover, and those exposed late had cumulative mortality more similar to larvae that were chronically exposed to sediments in high concentrations.

- Laboratory experiments suggested that effects of sediments are more likely to be lethal than non-lethal. We found little evidence for variation in developmental abnormalities attributable to suspended sediment regimes during larval development. In contrast, our field experiment suggested a high incidence of developmental abnormalities in the harbour compared to those larvae held in filtered seawater in the laboratory. This discrepancy may suggest the importance of other environmental factors associated with terrestrial runoff, which, alone or through an interaction with sediments, may produce developmental abnormalities.

6.PHOTOSYNTHETIC CHARACTERISTICS AND SURVIVAL OF LARGE BROWN SEAWEEDS AND UNDERSTOREY ALGAL SPECIES IN RELATION TO A SEDIMENT GRADIENT: MEASUREMENTS OF KELP PHOTOSYNTHESIS AND LIGHT ENVIRONMENT

6.1 Methods

6.1.1 Characterisation of the experimental light environment

Hobo light loggers were installed in April 2003 at each of the four study sites (see Table 3) to quantify the light environment in the 10–13 m depth strata. The aim was to provide a continuous record of irradiance at each site and to detect turbid events over the year when two-monthly site visits were made. Loggers were downloaded and cleaned every two months. There was some damage to loggers during storm events and hence there is not a complete record from every site, particularly the most exposed site 3 (Hapuku Rock). Nevertheless, a reasonably intact record was obtained from June 2003 to July 2004 for Sites 1 (Cook), 2 (King Fish), and 3 (Pupuha). As a backup to the logged data, on each sampling occasion, a profile of underwater irradiance (see Section 2) using the profiling PUV500 was made at each site.

The data collected from the in situ loggers was used in three main ways:

1. Correlation with rainfall
2. Examination of the duration of low clarity periods
3. Enabling predictions of photosynthetic capability of *E. radiata* to be made.

6.1.2 Kelp growth and abundance

Two approaches were taken to assess kelp health and abundance. The first involved transplants of *E. radiata* between sites, the second involved tagging and monitoring growth of undisturbed kelp at each site.

1. After site selection in March 2003, we chose to transplanted whole plants rather than rocks. Concrete blocks were modified to enable entire plants of *E. radiata* to be attached at the holdfast using cable ties. This method proved generally successful, and holdfasts were observed to fasten to the artificial substrata by formation of new haptera over the course of experiments, and no necrosis of the holdfasts were observed. In April 2003, 10 plants were transplanted from site 4 (Pupuha) to each of the other three sites, as well as a control transplant from rock to artificial substrate at site 4. At the time of transfer, plant (stipe and primary) length and photosynthetic characteristics were measured. These investigations were conducted within 20 m x 20 m plots delineated within the 10–13 m depth strata at each of the four sites. Sampling was carried out at two-monthly intervals between April 2003 and July 2004.

2. In April 2003, 20 *E. radiata* plants within the canopy were tagged in situ, within the 20 m x 20 m plot at each site, to enable their growth to be followed over time and for whole plant mortality to be assessed. Growth was measured using the hole punching method of Mann & Kirkman (1981). On every second plant a hole was punched in the primary blade, 5 cm from the meristem. Every two months the distance of the hole from the meristem was re-measured. At the same time, the length of the stipe and the primary lamina was measured for each tagged plant as described by Novaczek (1984). If plants had been lost in storms, new plants were tagged as replacements to maintain n = 20.

In April 2004, a suite of measurements designed to characterise each of the study sites was conducted independently of the two-monthly monitoring. Firstly, a demographic survey was carried out at each of the four sites. Ten mature *E. radiata* plants were selected from a 20 m x 20 m area adjacent to the tagged plants. The whole plant, including the holdfast, was removed from the substrate and returned to the laboratory where they were separated into frond, stipe and holdfast. Cores ($1.54 cm^2$) were subsampled from the lamina and immediately frozen analysis of chlorophylls *a* and *c,* using spectrophotometric techniques after extraction in methanol/dimethyl sulphoxide (Duncan & Harrison 1982). Stipe, length, primary lamina length, and total frond length were measured. The dry weight of holdfast, stipe and lamina were then obtained after drying at 60 °C until they reached constant weight. Secondly, in April 2004, divers placed 5, 1 m x 1 m quadrats at 4 m intervals along one axis of the 20 m x 20 m tagged plot. Within each quadrat, the total number of *E. radiata* plants was counted, including bare holdfasts and stipes without laminae. Entire plants were recorded in one of three size classes according to stipe length; under 5 cm, 5 to 10, 10 to 11, 50, over 50 cm.

6.1.3 Kelp photosynthetic capacity

In August 2003 the in situ tagging was found to be yielding consistent and meaningful results, and this became the focus of the remainder of the study. However, mortality of transplants was high in storms during winter and spring 2003. As there were insufficient replicates to obtain meaningful growth comparisons, we assessed changes of photosynthetic capacity over time of the *E. radiata* transplants at sites 1, 2, and 4 rather than continuing two-monthly growth monitoring. Accordingly we decided to survey understorey taxa in spring and autumn rather than conduct transplants.

It was not always possible to sample all kelp parameters at site 3 (Hapuku). The swell at this site under easterly conditions made conditions unsuitable for intricate in situ measurements by divers, and on two occasions in late 2003 the site could not be located in low visibility after the mooring buoy came free during a storm. Productivity of macroalgae is primarily affected by light, nutrient supply, and water movement. Differences in productivity between sites in close proximity are generally ascribed to differences in irradiance (Kirkman 1989). For this reason changes in water clarity among our study sites were expected to have an effect on photosynthetic capacity of *E. radiata,* and therefore on production and growth. The photosynthetic capacity of tagged and transplanted *E. radiata* was assessed using pulse amplitude modulated fluorometry (Diving-PAM) techniques (Ralph et al. 1998). The Diving-PAM is a self-contained underwater fluorometer used by a diver to make in situ non-destructive measurements of chlorophyll fluorescence on intact plants. On six occasions between April 2003 and April 2004, in situ measurements of photosynthesis-light relationships (rapid light curves, RLC's) were made on five replicate tagged plants and on transplants. Details of the measurement method using PAM fluorometry are given in Box 2.

Algae can modify their photosynthetic apparatus in response to variation in the light environment. Photoacclimation to a lower light intensity usually involves an increase in light harvesting pigments and a decrease in the saturation irradiance intensity (E_k) (e.g., Fairhead & Cheshire 2004). Using a dimensionless relative value of electron transport rate (RETR) as a proxy for photosynthesis, $RETR_{max}$ and E_k were calculated from each rapid light curve (RLC) by fitting a modification of the hyperbolic tangent relationship of Jassby & Platt (1976) to RETR vs PAR data (e.g Figure 37).

Box 2: Measurement of photosynthetic activity using PAM fluorometry

The Diving-PAM fluorometer is fully submersible and allows measurement of variable fluorescence in situ, with centimetre spatial resolution and repeated measurements within a few seconds. Variable fluorescence is a property of photosystem II (PSII), which can be used to estimate photosynthetic activity (Schreiber et al. 1986). The system measures the fluorescence response of PSII to a low intensity, square wave modulated light under ambient irradiance and again during application of a short pulse of saturating white light. When measurements are conducted with fully dark acclimated material, i.e., ambient light is darkness, the amount of fluorescence response before application of the saturating pulse (F_o) resembles the condition when all PSII reaction centres are "open" and available for capture of light for photochemical conversion. The application of the short pulse of saturating irradiance momentarily closes all of the PSII reaction centres, and the second measurement of fluorescence (F_m) represents the light that is spilled to fluorescence under conditions of light saturation (Krause & Weis 1991). The difference in fluorescence ($F_m - F_o$) is called the variable fluorescence (F_v). The ratio of variable fluorescence to maximum fluorescence (F_v/F_m) represents the scope for photosynthesis under ambient conditions (i.e., how far the plant is from light saturation).

F_v/F_m is normally at its highest in non-inhibited material under dark acclimated conditions, when it represents the maximum quantum yield which, for healthy angiosperms, is typically close to 0.8 (Schreiber et al. 1986, Büchel & Wilhelm 1993). As ambient light increases, the proportion of open PSII reaction centres declines, more light is spilled as fluorescence, and the initial steady-state fluorescence (termed F_t when made in illuminated conditions) often increases. High light also often invokes a temporary reduction in F_m to F_m', as mechanisms to dissipate more light energy as heat are mobilised within the plant (dynamic photoinhibition – Krause & Weis 1991). Thus, when measured in the light, the variable fluorescence ratio ((F_m'- F_t)/ F_m' or $\Delta F/F_m$') tends to decline proportionate to the extent to which electron transport chains have become saturated. While F_v/F_m, measured under dark conditions reflects the *maximum* quantum yield, $\Delta F/F_m$' measured at ambient irradiance, reflects the *effective* quantum yield (Y) of the plant under prevailing irradiance condition (Hanelt et al. 1995). There is strong evidence that quantum yields measured using fluorescence and techniques accurately reflect the actual photosynthetic yields of plants (Hanelt et al. 1995, Beer & Björk, 2000, Beer et al. 2000), except perhaps at at very high irradiance (Krause & Weis 1991, Geel et al. 1997, Flameling & Kromkamp 1998)).

The rapid light curve (RLC) function of the Diving-PAM provides information about the photo-acclimated status of a plant. In a RLC, $\Delta F/F_m$' is measured following each of eight, 15 s periods of actinic light, provided by the Diving-PAM in increasing intensities from darkness. The RLC does not achieve steady state conditions during each light step (Schreiber et al. 1986) and so does not necessarily mimic a conventional diel light curve; rather, it provides a snapshot of the current status of light adaptation to the ambient light conditions. From the RLC, a relative electron transport rate (*RETR*) can be calculated, as $\Delta F/F_m$' * *PAR* (Beer & Axelsson 2004), which under most conditions mimics the shape of photosynthesis-irradiance relationships.

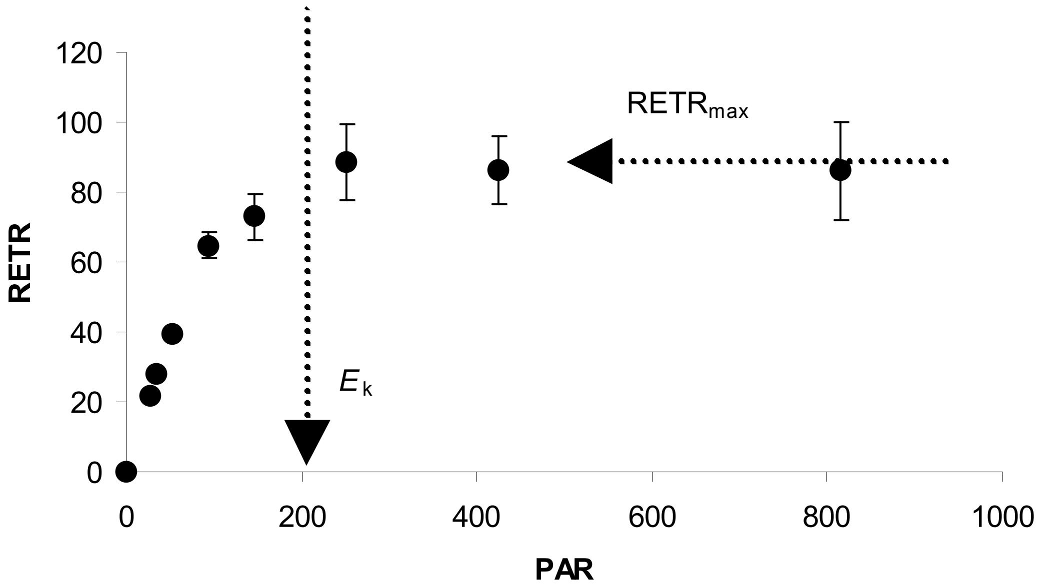


Figure 37: A stylised example of the relationship between RETR (dimensionless) and PAR. The dotted lines illustrate the parts of the relationship that relate to the parameters $\mathrm{RETR}_{\mathrm{max}}$ (maximum relative electron transport rate) and (E_k) the onset of light saturation.

To further elucidate the relationships between ambient irradiance and RETR-irradiance parameters, on 5 and 6 April 2004, the photosynthetic characteristics of *E. radiata* were measured over a depth range from 5 to 20 m (the maximum and minimum depth at which *E. radiata* was found at these sites) at the two clearest (see section 2) of these three sites, site 2 (Kingfish) and site 4 (Pupuha). This sampling was intended to identify the bounds of the photoacclimation potential of the plant

Curves were fitted using Sigmaplot (SPSS Inc, Chicago, IL, USA), fitting the equation:

$\mathrm{RETR} = \mathrm{RETR}_{\mathrm{max}}.\tanh\ (\mathrm{PAR}/E_\mathrm{k})$

Comparisons of derived parameters were made using a repeated measures MANOVA, with depth included as an independent variable, using Statistica for Windows 5.1 (Statsoft Inc., Tulsa, OK).

6.1.4 Kelp productivity

Simple models were constructed to assess the implications of water clarity for the potential productivity of *E. radiata* at the four sites. The objective was to use photosynthesis-irradiance relationships, approximated using RETR-irradiance formulations, coupled with measurements of underwater irradiance, to estimate potential for photosynthesis at 10 m at each site over ecologically meaningful time periods. As discussed above, site 3 (Hapuku) was not always accessible for the in situ photosynthetic measurements. However, the water clarity characteristics of site 3 Hapuku were very similar to those of site 4 (Pupuha, see Table 4), so models were restricted to sites 1, 2, and 4, which adequately represented the range of conditions we aimed to encompass.

In the first instance, PAR incident on *E. radiata* at a depth of 10 m over the course of a year was estimated from incident irradiance obtained from a weather station at Whitianga, from the harbour mouth (NIWA data), and from the range of measured K_d values (see Table 4). The calculated PAR for each hour of the day at the depth of the *E. radiata* was then used to approximate photosynthesis (*P*) as RETR using the RETR-irradiance relationships measured with rapid light curves (RLCs) at each site.

The second approach was to use actual irradiance measured in situ over a 10 day period when water clarity was on average lower at site 1 than at site 2. In both cases, for each hour, RETR was calculated using one of two equations, depending on PAR: 1) if irradiance exceeded E_{k}, i.e., was above saturation (from in situ PAM measurements), then RETR = RETR_{max}; 2) if irradiance was below E_k then RETR = ($\text{RETR}_{max.}/E_k$ x PAR). The value calculated for each hour was then summed over each period.

This simple approach does not account for respiratory losses and RETR has not been normalised to oxygen exchange, therefore it does not enable us to quantify net production. However, it does provide a relative comparison among the sites to establish whether water clarity is likely to have a significant influence on production over short (days to weeks) or long (months to years) time scales.

6.2 Results

6.2.1 Water clarity over time and rainfall

Consistent with the variability in the two-monthly measurements of attenuation coefficient, Section 2, data from the in situ light loggers at all four sites show that water clarity varied markedly over different time scales at all sites (Figure 38). Total rainfall for each day of the study was obtained from the NIWA monitoring station in the Whitianga catchment at Coroglen. Maximum daily rainfall was 180 mm on one day in April 2003 and exceeded 80 mm a day on six occasions during the year (Figure 39).

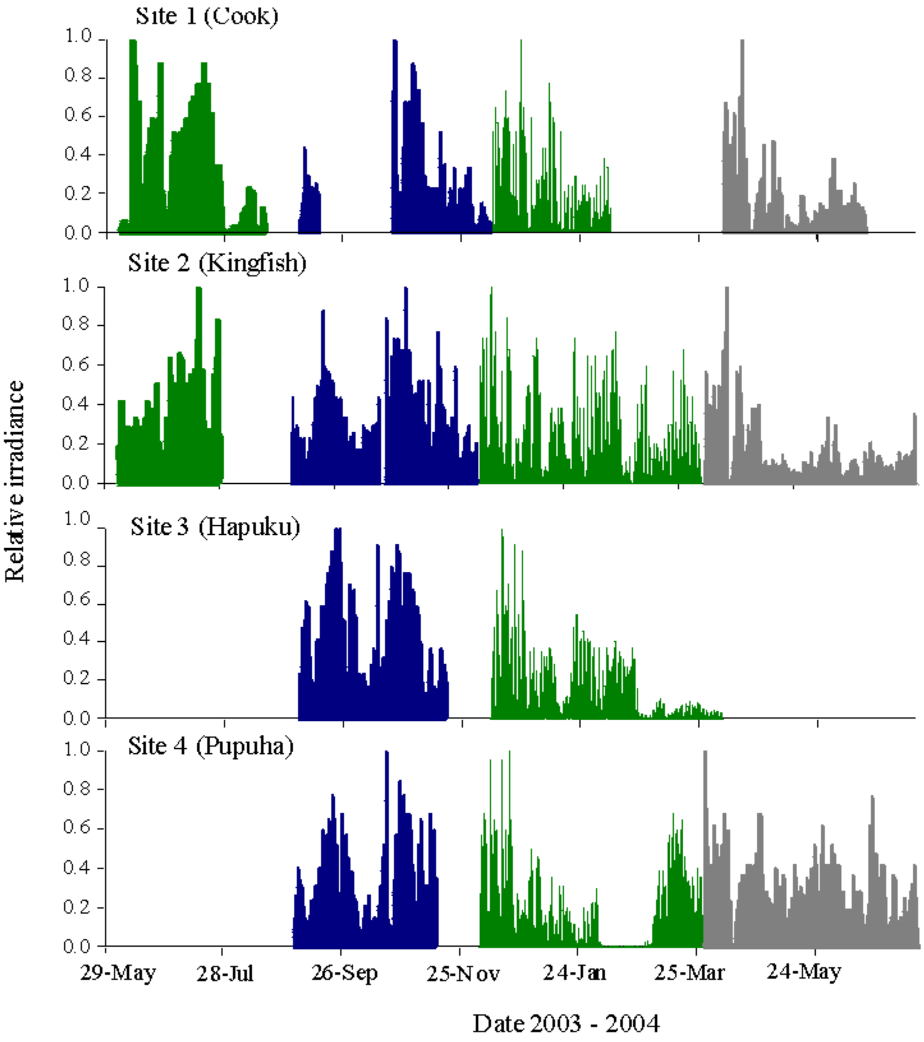


Figure 38: Logged underwater irradiance at the four study sites between June 2003 and May 2004.

Periods of low water clarity showed some relationship to periods of high rainfall particularly at Sites 1, 2 and 3; however this was not always a direct relationship and the influence of wind and waves has not been considered. River flow records would be likely to yield more information about potential relationships, but unfortunately there is no routine monitoring of flow of rivers entering Whitianga Harbour. Nevertheless the amount of irradiance at the depth of the tagged plants was significantly ($P < 0.05$) negatively correlated with rainfall at sites 1 ($r = -0.21$), 2 ($r = -0.22$), and 3 ($r = -0.21$). Irradiance was not significantly correlated with rainfall at site 4 ($r = -0.015$)

.

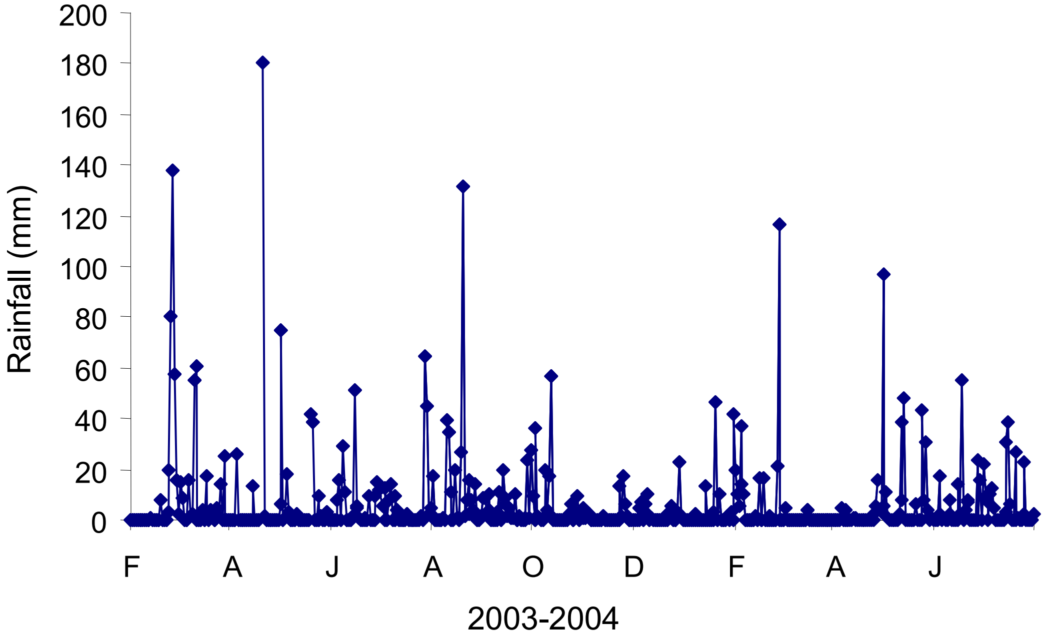


Figure 39: Rainfall over the duration of the study.

Some specific time periods have been extracted from the dataset to illustrate how the duration of relatively low water clarity events can differ between sites. Figure 40 shows data from loggers at site 1 (Cook) and site 4 (Pupuha) over a one month period (3 April to 3 May 2003). This period was one of high rainfall in the Coromandel Peninsula (see Figure 39) and resulted in stream flows of more than 150% of average April flows (http://www.niwa.co.nz/ncc/cu/2003-05/river). Coromandel township experienced flooding on 19 and 20 April 2003.

It is apparent that water clarity was markedly reduced at both sites 1 and 4 between 14 and 21 April (Figure 40). A notable difference between the two sites is that water clarity was reduced a number of days earlier at site 1 than at site 4 (approximately 7 April compared to 10 April).

In July 2003 (Figure 41) a comparison between underwater light at the sampling depths at sites 1 (Cook) and 2 (Kingfish) showed that there are periods when site 1 can remain turbid for a period of at least days after water clarity at Kingfish Rock has subsequently increased again (e.g., 26-28 July, Figure 41). Therefore the higher average TSS data reported from the regular monitoring for site 1 (see Table 4) is likely to be a result of both a higher sediment loading (see section 2 sediment traps) and a longer exposure to elevated suspended sediment concentrations compared to the other study sites.

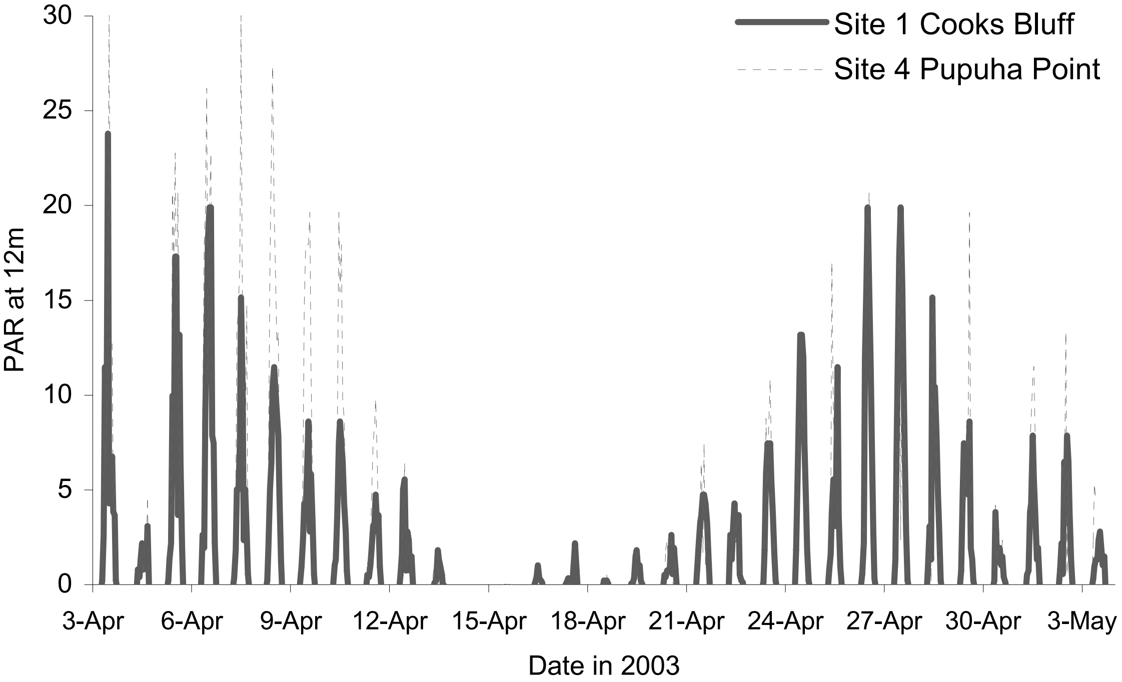


Figure 40: Irradiance as PAR (μmol photons m^{-2} s^{-1}) at a depth of 12 m at sites 1 and site 4 between 3 April and 3 May 2003.

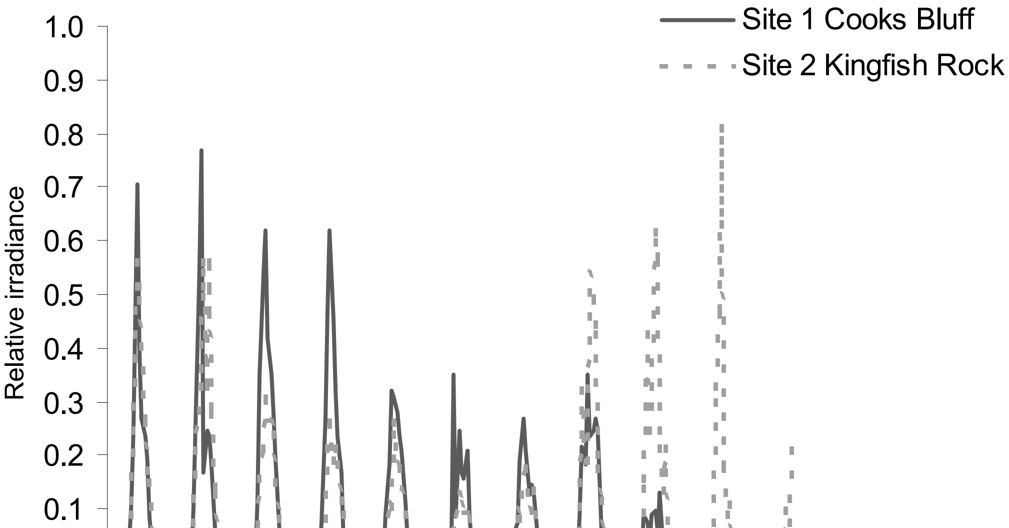


Figure 41: Irradiance relative to the maximum value measured at the site for the inter-site comparison at 8 m at site 1 (Cook) and 12 m at site 2 (Kingfish) between 18 and 28 July 2003.

6.3 Photoacclimation in *Ecklonia radiata*

6.3.1 Photosynthetic characteristics over a depth profile

Ecklonia radiata at site 2 (Kingfish) and site 4 (Pupuha) had different photosynthetic characteristics according to the depth at which it grew. The saturation onset parameter E_k decreased with increasing depth, from 125 at 5 m to 24 at 20 m. RETR showed the same pattern, decreasing from 44 at 5 m to 17 at 20 m. In both cases there was remarkable agreement between the two sites, when E_k and RETR are plotted against the percent of surface irradiance (SI) as a substitute for depth (Figure 42).

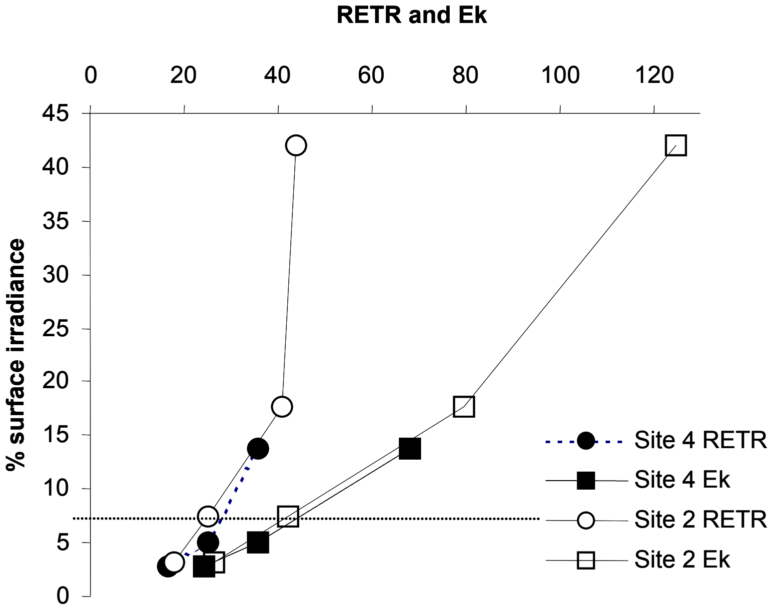


Figure 42: RETR and E_k plotted in relation to the percentage of subsurface irradiance (optical depth) calculated from K_d measured at the time of the photosynthetic measurements. The dotted line indicates the equivalent percentage of surface irradiance at the depth of the study plants (11 m) at site 1 (Cook) on the same day.

The consistency of the data presented in Figure 42, obtained at a range of depths from sites with different water clarity, suggests that the parameters of the RETR vs irradiance curves show consistent relationships with underwater irradiance. Given this, it is reasonable to assume that the RETR-irradiance parameters for a given plant can be inferred from its optical depth. As an example, in April 2004, 7.4% of sub-surface irradiance penetrated to the depth of the study plants at Cook (11 m) (see Table 4). From Figure 42, we can infer an RETR of between 25 and 30 and an E_k of about 40 for *E. radiata* at that depth, on that day. The actual measured values on that day were RETR = 24.5 and E_k = 38.5. The similarity of the predicted and measured values supports the inference of photosynthetic responses of *E. radiata* at depth at the study sites from irradiance incident on the plants. Thus we can use those inferences to make predictions of photosynthetic responses over different time scales if water clarity is known.

6.3.2 Photosynthetic characteristics of *Ecklonia radiata* at the four study sites.

The characteristics of rapid light curves (RLCs) showed some differences between site and time. Winter (June) curves tended to show lower maximum rates (range 30–60) for RETR than summer (December) curves (range 40–90) at all sites (Figure 43). This seasonal difference probably reflects the higher summer irradiance and temperature, while between-site differences are more likely to reflect water clarity in the days leading up to the measurements.

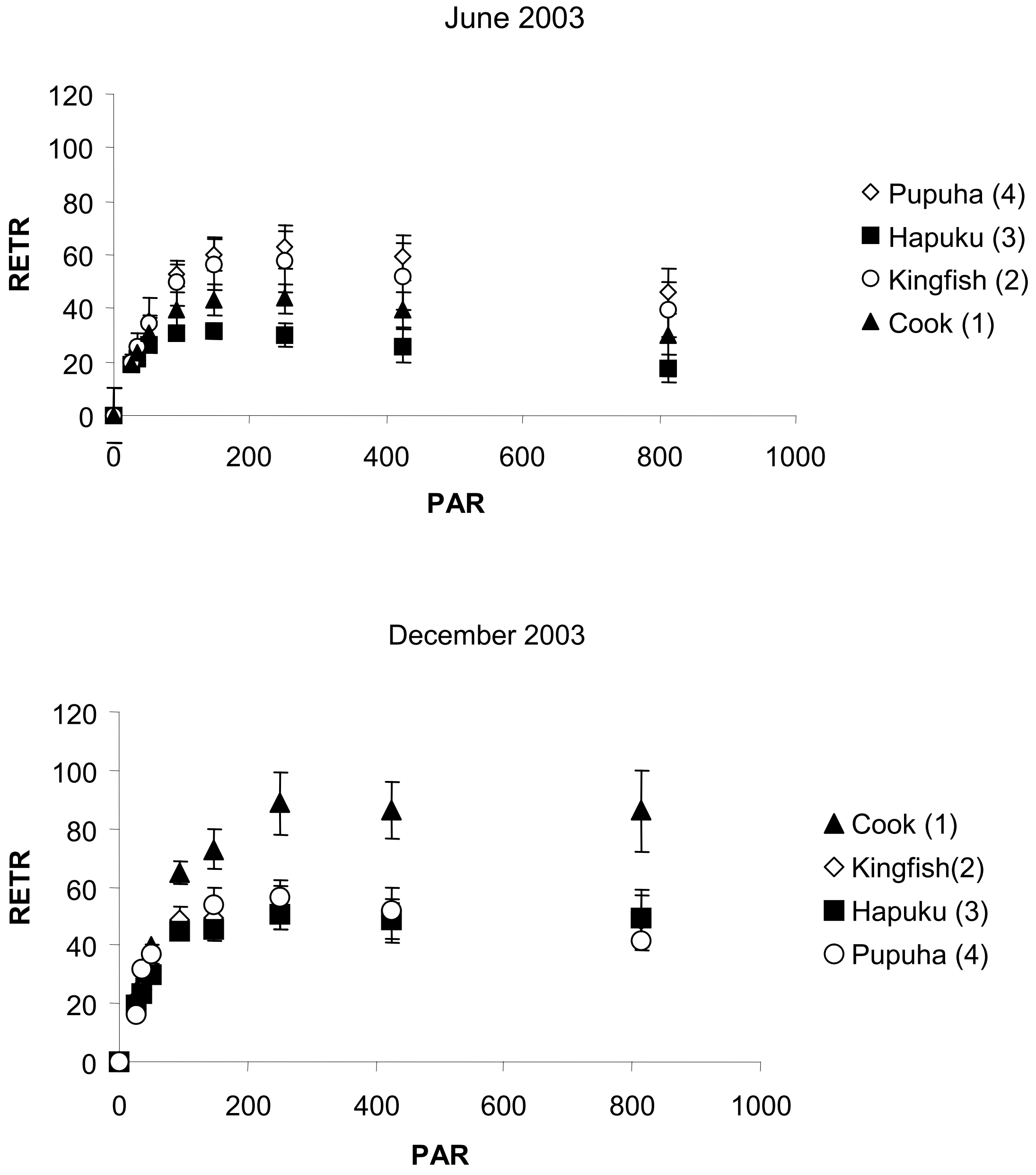


Figure 43: Rapid light curves for *Ecklonia radiata* at each site in winter (June) and summer (December).

Over the whole study, E_k ranged from a minimum of 33 μmol photons m^{-2} s^{-1} at site 2 (Kingfish) in April 2004 to a maximum of 104 μmol photons m^{-2} s^{-1} at site 1 (Cook) in December 2003 (Figure 44).

However, the average of parameters RETR and E_k obtained from replicate RLC's showed no significant difference among sites (ANOVA by date and by site $P > 0.05$) (Figure 44). Overall we can expect that E_k of *E. radiata* will fluctuate from about 30–120 µmol m^{-2} s^{-1} and RETR about 40–90, depending on light conditions before and during the measurements.

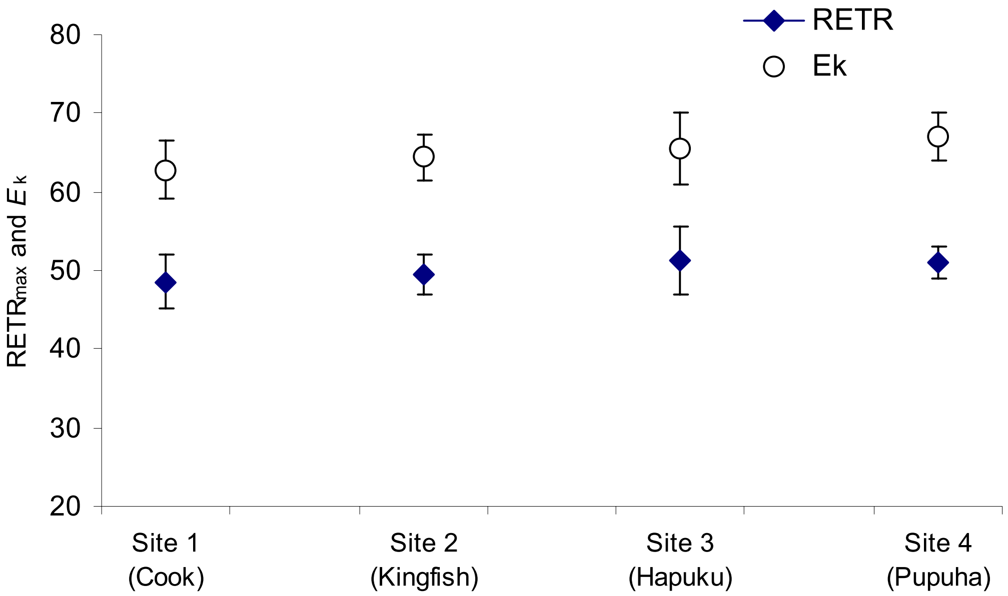


Figure 44: Average values of $RETR_{max}$ and E_k for in situ plants at all sites between April 2003 and April 2004; n = 6.

Transplants exhibited lower values of RETR and E_k than the plants growing in situ. However, there were some site effects as well, transplants at site 1 (Cook) had a significantly ($P < 0.05$) lower average E_k than those at site 4 (Pupuha) and site 2 (Kingfish) transplants, which were similar to each other (Figure 45).

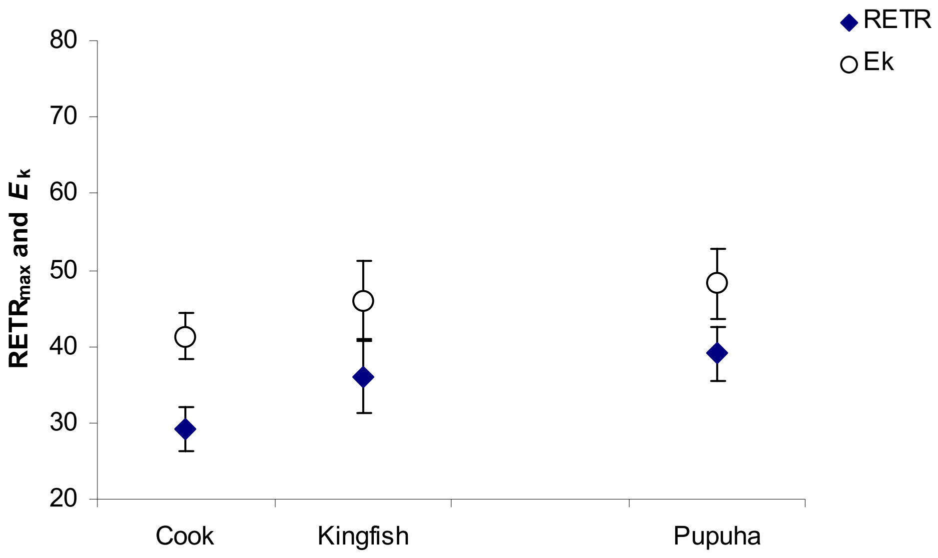


Figure 45: Average values of $\mathrm{RETR_{max}}$ and E_k for transplants at all sites except site 3 (Hapuku) between June 2003 and December 2003; n = 3.

RETR is, however, by definition only a relative measure, linking the electron yield of PSII with incident irradiance. The actual electron transport rate (ETR) and photosynthetic carbon fixation will depend, among other things, on the proportion of incident irradiance absorbed by the thallus. Absorptance depends on thallus thickness and the pigment content (Markager & Sand-Jensen 1992). In April 2004, pigment content of thalli was highest in site 1 (Cook) followed by site 2 (Kingfish) and site 3 (Pupuha), and site 1 (Cook) had the lowest Chl*a*:Chl*c* ratio (Table 9). This suggests a different acclimation to lower average light climate at the site 1, with both a higher concentration of pigments and higher proportion of accessory pigments. These differences were, however, not statistically significant among sites (ANOVA $P > 0.05$) and these data must be considered in context of other photosynthetic characteristics.

Table 9: Mean and standard error Chl*a* and Chl*c* in *E. radiata* (μg cm^{-2}).

	Chl*a*	Chl*c*	Ratio Chl*a*:Chl*c*
Cook (n=5)	23.7 (1.7)	6.5 (0.4)	3.7 (0.14)
Kingfish (n=5)	20.7 (0.7)	5.3 (0.2)	3.9 (0.05)
Pupuha (n=5)	16.4 (1.02)	3.9 (0.2)	4.2 (0.11)

6.3.3 Modelling of photosynthetic characteristics

Calculations of potential photosynthesis over the course of a year were designed to address the question; could water clarity of the ranges experienced at each of the sites on the Hahei gradient affect *E. radiata* production, and if so, by how much? If one assumes that E_k at all sites changed over the course of a year at 10 m in the same way and by the same magnitude, then potential gross production at 10 m at site 1 (Cook) was estimated to be between 50 and 60% of that at site 2 (Kingfish) and site 4 (Pupuha). This was entirely due to a lower total irradiance on the plants at site 1 and does not allow for greater photoacclimation at that site than at the other sites. It was essential to incorporate the potential for acclimation to changing irradiance into these calculations since, if *E. radiata* at Cook was able to acclimate to a reduced water clarity, it may be able to make up that shortfall in production. The capacity for acclimation could be estimated from Figure 42.

As an example, over 10 days between 18 and 28 July (see Figure 41) the potential effect of acclimation was modelled. When E_k was set at the same value (70 μmol photons m^{-1} s^{-1}) for both site 1 (Cook) and site 2 (Kingfish) over these 10 days, then *E. radiata* at site 1 was predicted to achieve 90% of production of site 2, the lower value at site 1 being due to a lower irradiance. If, however, plants at site 1 were able to acclimate to that lower irradiance by reducing E_k to 60 (e.g., see Figure 42) then gross production was predicted to be virtually equal for both sites. Changes of greater magnitude will clearly be possible within the range of E_k values that have been measured over this study (32–104).

The ability of *E. radiata* to acclimate over days or weeks to reduced water clarity by reducing E_k and, in the longer term, by altered pigment content enables more efficient photosynthesis at lower light levels. Essentially such acclimation is equivalent to seasonal changes in relation to light availability (Fairhead & Cheshire 2004) or to the plants growing deeper than 11 m. Depth (light related) differences in production are known for *E. radiata* to be manifest as slower growth rates and reduced population density for deep plants compared to shallow plants (Novaczek 1984, Kirkman 1989). Hence chronic effects of short-medium term reductions in water clarity can be expected to have an energetic cost associated with them.

E. radiata is able to photo-acclimate by changes in photosynthetic parameters and pigment content, enabling it to continue to photosynthesise effectively during times of low water clarity. It is likely that for much of the time such acclimations are sufficient to enable production to be similar at all sites. However, although *E. radiata* has an acknowledged ability to withstand periods of low light (Novaczek 1984), there will be a threshold below which further acclimation is not possible, the compensation depth for growth (Markager & Sand-Jensen 1992). The compensation depth for marine plant growth occurs at a light level where photosynthetic gains equal respiratory losses over ecologically meaningful time scales.

The maximum depth of *E. radiata* at the study sites is limited to some extent by the lack of suitable rocky substrate at depth; however the maximum depth to which they grew at the Whitianga–Hahei study sites had about 3% of incident irradiance. Laminarians are generally considered to have a compensation depth of between 0.2 and 2% of incident irradiance.

6.3.4 *Ecklonia radiata* growth and abundance

Growth rates calculated for the tagged plants have not been normalised to plant size. At the time of tagging the intent was to focus on mature plants, although some smaller individuals were tagged. At the start of the study in April 2003, the total length (stipe plus blade) of tagged *E. radiata* (n=20) at all sites ranged from 25 cm at site 3 (Hapuku) to 113 cm at site 2 (Kingfish) (Figure 46). With the exception of site 3 where 65% of the tagged plants were less than 50 cm long, at least 50% of plants at all other sites were between 50 and 100 cm long.

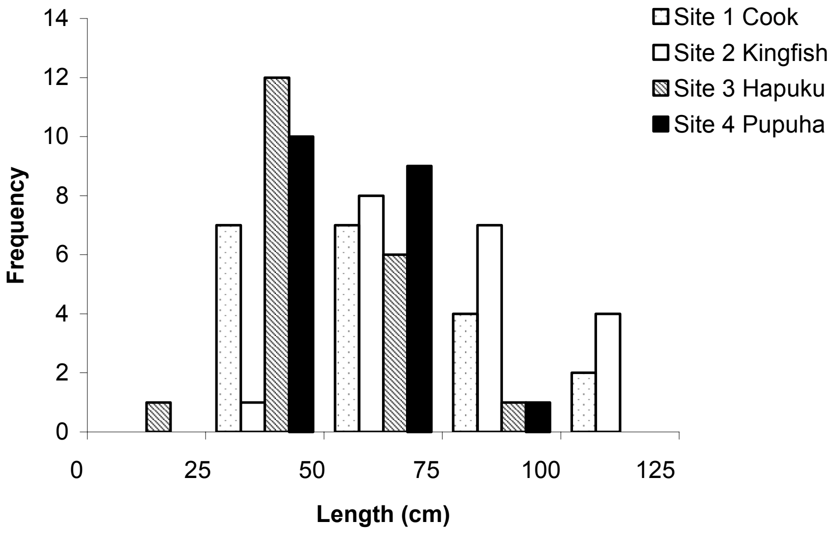
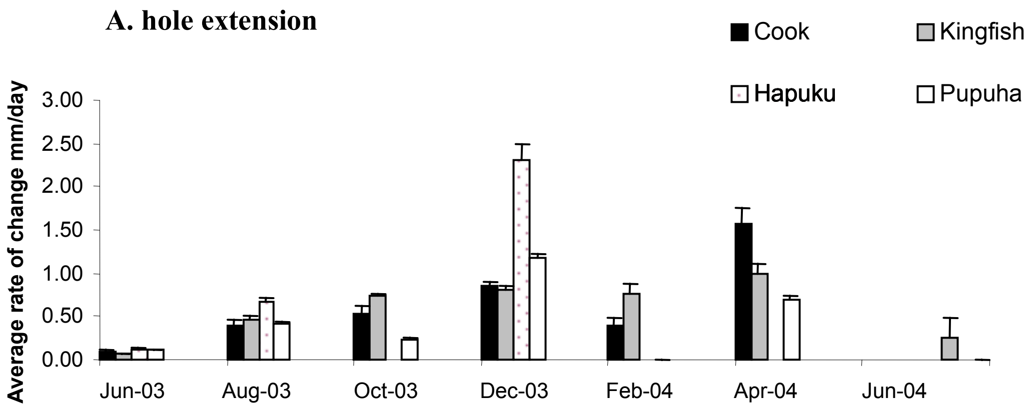
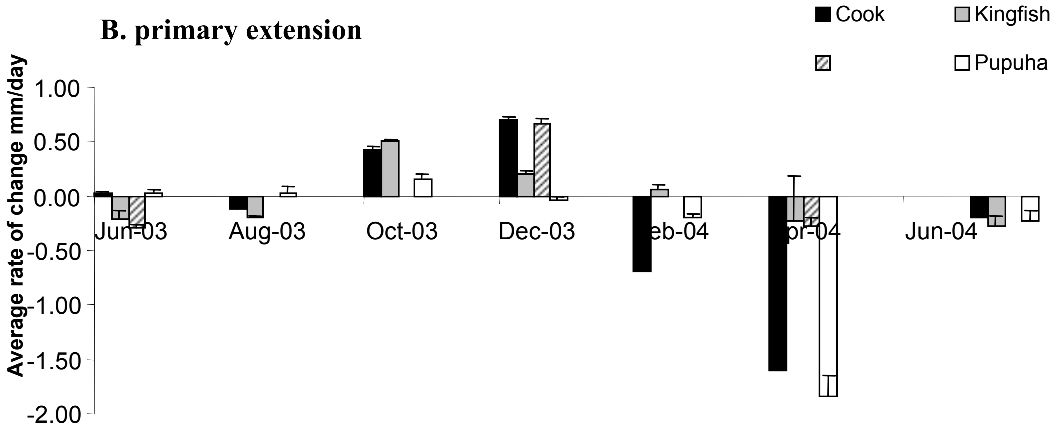


Figure 46: Length frequency of *Ecklonia radiata* at the four sites at the start of the study in April 2003.

Over the study there was a significant positive correlation between rates of stipe and primary lamina elongation ($r^2 = 0.6$, $P < 0.05$). The highest rates of primary lamina extension, measured by the hole punch method, occurred between October and December 2003 and between February and April 2004 (Figure 47A). Average October to December rates for all sites (0.8 mm day^{-1} at site 2 (Kingfish) to 2.31 mm day^{-1} at site 3 (Hapuku)) were significantly higher (ANOVA, post-hoc Tukeys test, $P = 0.007$) than those calculated for April to June 2003 (0.06 mm day^{-1} at Kingfish to 0.12 mm day^{-1} at Hapuku, but were not significantly greater than during any other time intervals. A similar pattern was observed for stipe extension although the only significant difference noted between sampling times was due solely to the December value for site 3 (Hapuku) (Figure 47). With the exception of that one high value, stipe extension never exceeded 0.38 mm day^{-1}.

Very little growth in any plant part was measured at any of the sites between April and June 2003 (less than 0.1 mm day^{-1}), but between June and August the blade elongation rate as measured by hole extension (Figure 47A) increased to between 0.5 and 0.7 mm day^{-1}. Despite this increase in growth rate between April and August 2003, erosion of primary lamina at the distal end meant that there was no overall increase in length at any site. Between August 2003, and December 2003 however, growth exceeded erosion at all sites except Pupuha. Between December 2003 and April 2004, erosion exceeded growth at all sites and, on average, erosion rates were significantly greater in this period than in August–October (P = 0.022) and October–December (P = 0.006).These seasonal patterns are consistent with those observed at Goat Island by Novaczek (1984) in that lamina elongation was greatest in spring / summer, but that deterioration of the lamina was greatest in warm autumn water (February to April).


A. hole extension


B. primary extension

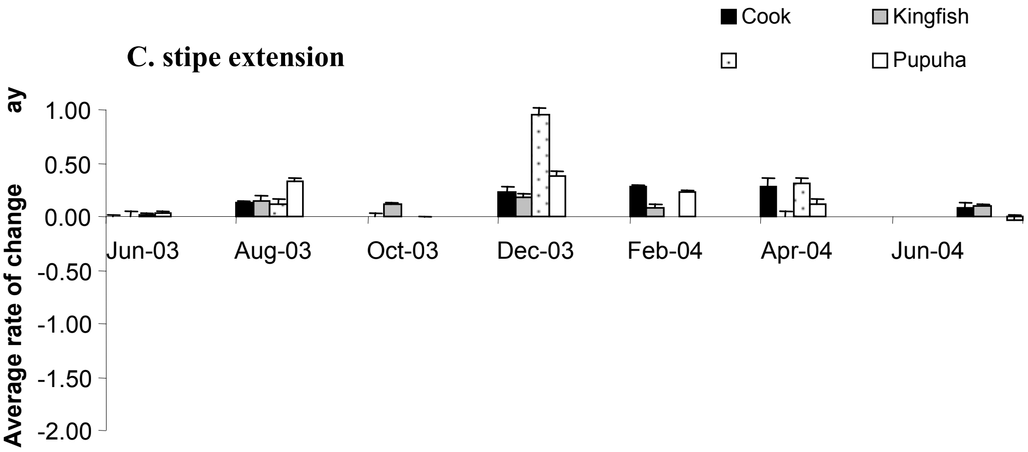


Figure 47: Rate of extension of A, punched hole, B, primary lamina and C, stipe over 7 approximately 2-month intervals between April 2003 and July 2004. There are no data for punched holes in July 2004.

Sites 3 (Hapuku) and 4 (Pupuha) experienced the highest rates of loss or death of plants with about 35% of plants lost on two occasions (Figure 48). Site 1 (Cook) and site 2 (Kingfish) had loss rates of 15% or less except between February and April 2004 when 40% of tagged plants were missing or dead at Cook.

This was consistent with a lower plant density measured at site 1 (Cook) in April 2004. Site 1 had fewer individuals per square metre than any of the other sites (Figure 49) determined largely by fewer individuals more than 10 cm high (Figure 50). Figure 50 also shows that site 1 (Cook) had the highest proportion of bare stipes, characteristic of dieback symptoms described by Cole & Babcock (1996).

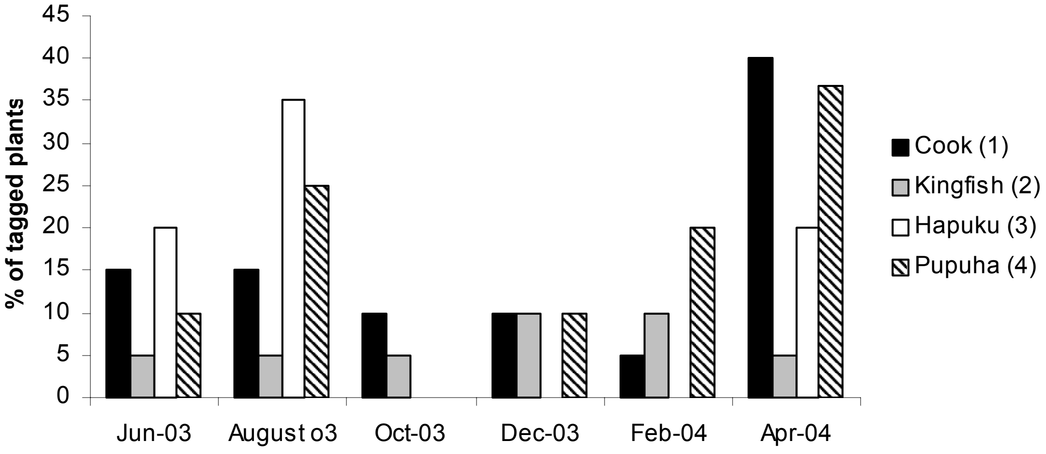


Figure 48: Percentage of tagged plants that had either broken off at the holdfast or of which only a rotting stipe remained at the end of each 2-month period over the study.

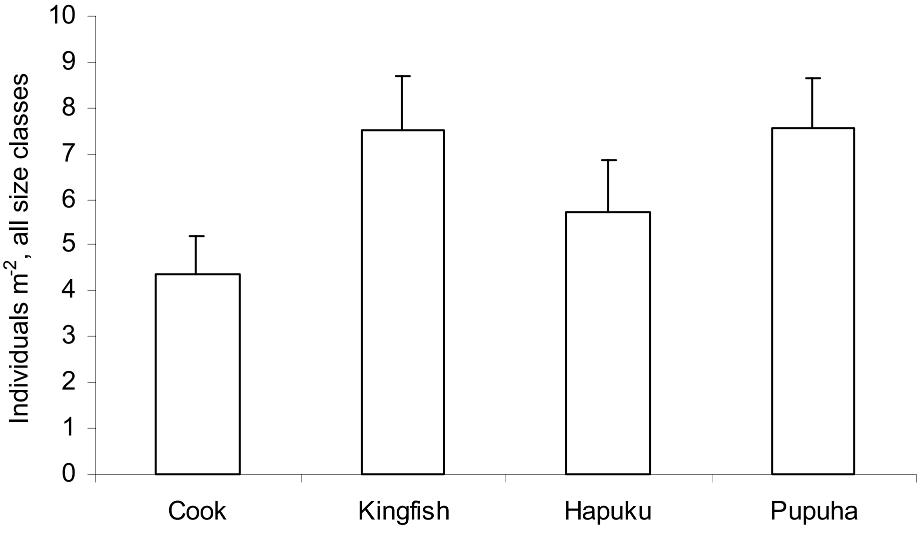


Figure 49: Individuals of all size classes per m^2 in April 2004.

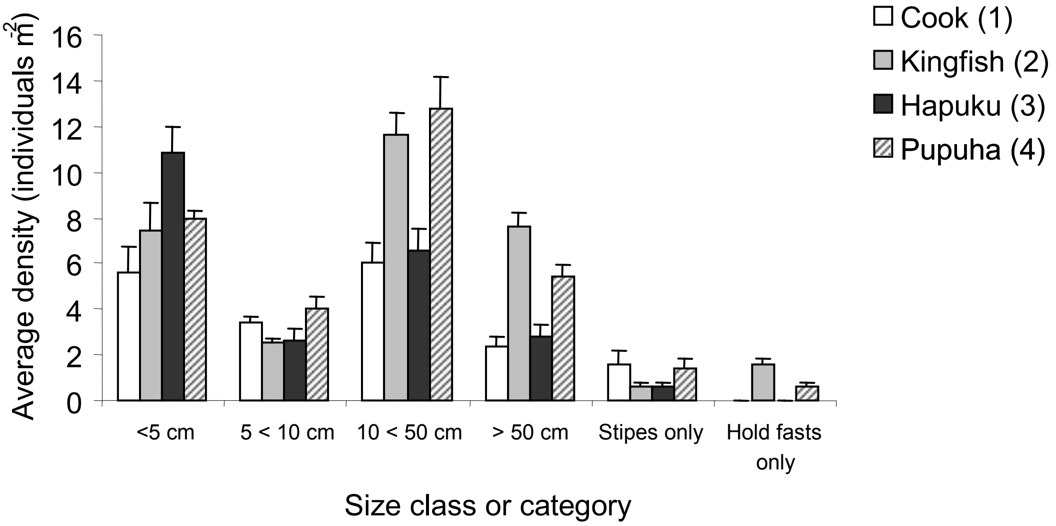


Figure 50: In situ density of *Ecklonia radiata* divided into size classes. Data are average of 1m^2 quadrats (5 per site). The number of bare stipes and holdfasts only are also shown.

Despite a lower density and the potential for reduced production as a result of times of lower water clarity, according to biomass data collected in April 2004 (Figure 51) and in winter 2003 and summer 2004 (see objective 2, see Figure 19), *E. radiata* at site 1 (Cook) still managed to maintain average mature plant biomass at 10 m depth that is not significantly different from the other sites.

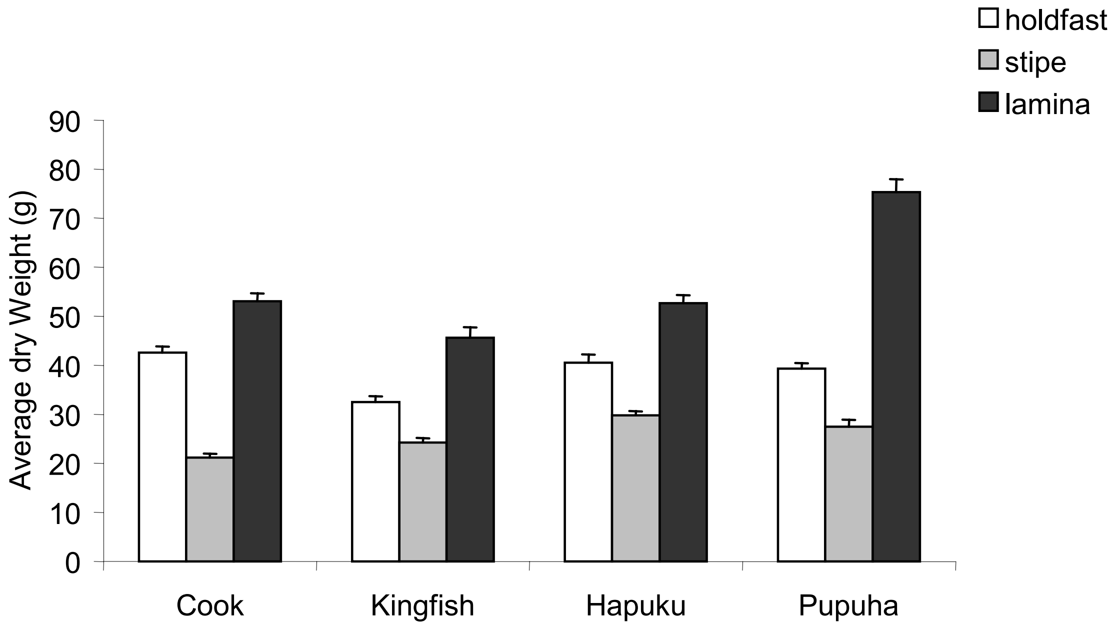


Figure 51: Average and standard error biomass (dry weight) for holdfast, stipe, and laminae of 10 plants from each site in April 2004.

7. UNDERSTOREY TAXA COMMUNITY COMPOSITION AND ABUNDANCE

7.1 Methods

7.1.1 Survey

In September 2003 (spring), five 1 m^2 quadrats were sampled in two depth strata (10–15 m, 5–10 m) at each of the four sites. To provide context, conspicuous macro-invertebrates were counted as well as macroalgae. Point cover estimates were made by recording the proportion of seaweeds and encrusting invertebrates lying under 10 haphazardly allocated points. The points were allocated by a method used by Schiel (1984) and derived from Foster (1982). A string longer than the side of the 1-m^2 quadrat with 10 knots at irregular intervals was placed over the quadrat and haphazardly drawn tight. The identity of organisms underlying the 10 points was recorded. Quadrats were haphazardly distributed within the depth strata, though bottom time limits meant that none were done below 13 m.

Analysis of the survey data from spring 2003 showed no evidence for differences among depth strata (see results below) and so in April 2004, five 0.25 m^2 quadrats were quantitatively sampled at depths between 5 and 10 m only. From each of these quadrats abundances of organisms (count data) and covers of encrusting forms (cover data) were obtained. Photographs of quadrats were taken to aid identification of encrusting organisms.

Some seaweeds and encrusting animals were recorded in broad morphological categories (e.g., brown fuzzy algae) for analysis of principal co-ordinates (see below). These have subsequently been identified by NIWA taxonomists (Appendix 2). Ascidian and sponge species identified from photo quadrats are listed in Appendix 3.

7.1.2 Statistical analysis

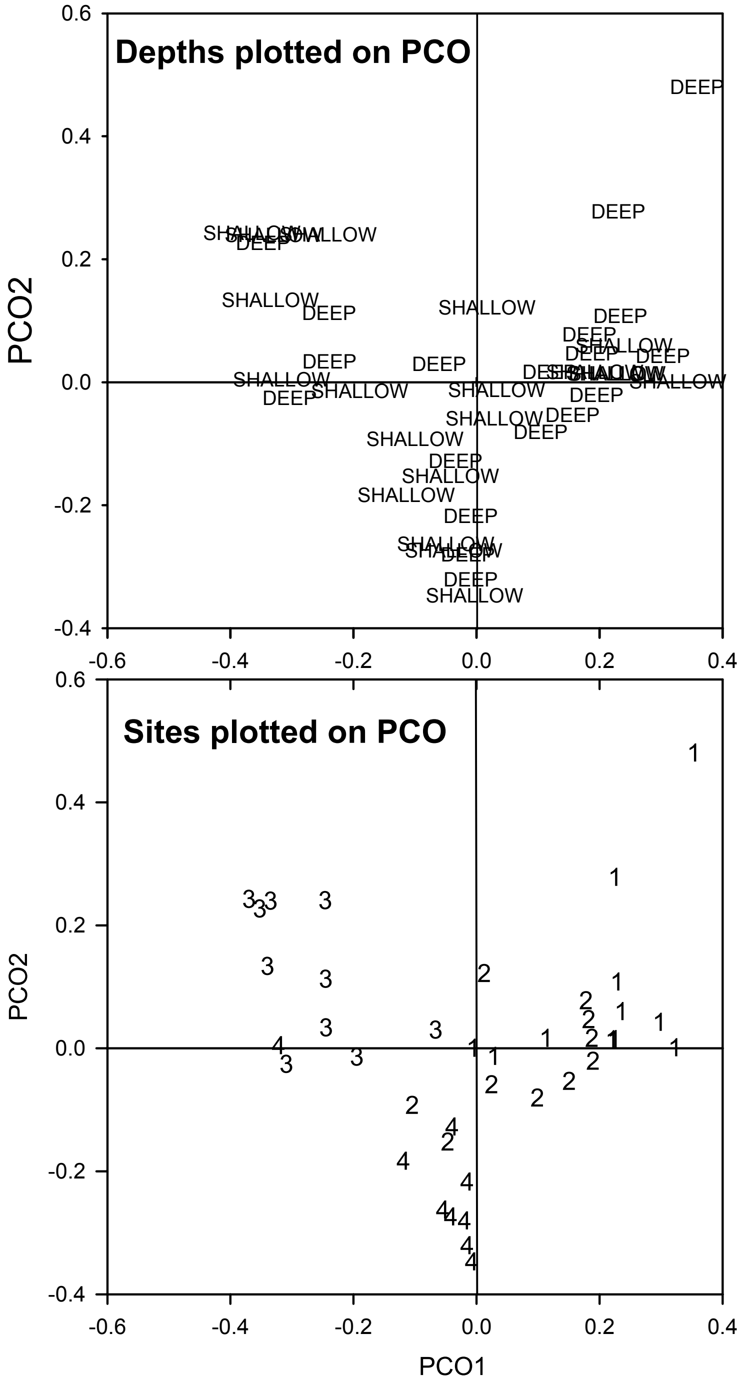
Means and standard errors were used to compare abundances of individual taxa, and of number of species and number of individuals, among sites. Bar charts of unsorted site totals are used to display variation in the fauna among sites graphically. Canonical analysis of principal coordinates (CAP: Anderson & Willis 2003) was used to display statistical differences among sites for both count and cover data. Data were expressed as Euclidean distances for analysis. The software for that analysis provides both unconstrained and constrained analyses; both are presented, since the unconstrained analyses (equivalent to a principal coordinate analysis, PCO) provide an indication of similarities among the entire universe of samples, whereas the constrained analyses (CAP) take the among vs. within site structure of the sampling into account (Anderson & Willis 2003).

7.2 Results

Analysis of the survey data from spring (September) 2003 for understorey taxa, using CAP, showed no evidence for differences among depth strata (Figure 52). There was, however, evidence for a change in understorey fauna and flora along the gradient from site 1 (Cook) to site 4 (Pupuha).

Plots of canonical analysis of the principle co-ordinates of quadrat count data from spring 2003 are shown in Figure 53. The unidentified alga that we refer to in this analysis as ‘brown fuzz’ occurred mainly at sites 1 and 2 (Figure 53). The codes for dominant taxa are listed in Table 10. CCA (crustose coralline algae) was negatively associated with one group of quadrats from sites 1 and 2 on axis 1, and negatively associated with another group of quadrats from those sites on axis 2. *Carpophyllum flexuosum* (CFA and CFR) was positively associated with one of the quadrat clusters from sites 1 and 2 (upper left quadrant), whereas *Ecklonia radiata* (ERA) was negatively associated with sites 1 and 2 on axis 2.

The percentage cover of crustose coralline algae (Figure 54) was calculated for each site with both depth strata combined. There was a strong gradient in percentage cover with the lowest at site 1 and the highest at site 4 (Figure 55).


Depths plotted on PCO

Sites plotted on PCO

Figure 52: Similarity in canopy macroalgae and understorey community structure between depth strata (upper panel) and site (lower panel), spring 2003.

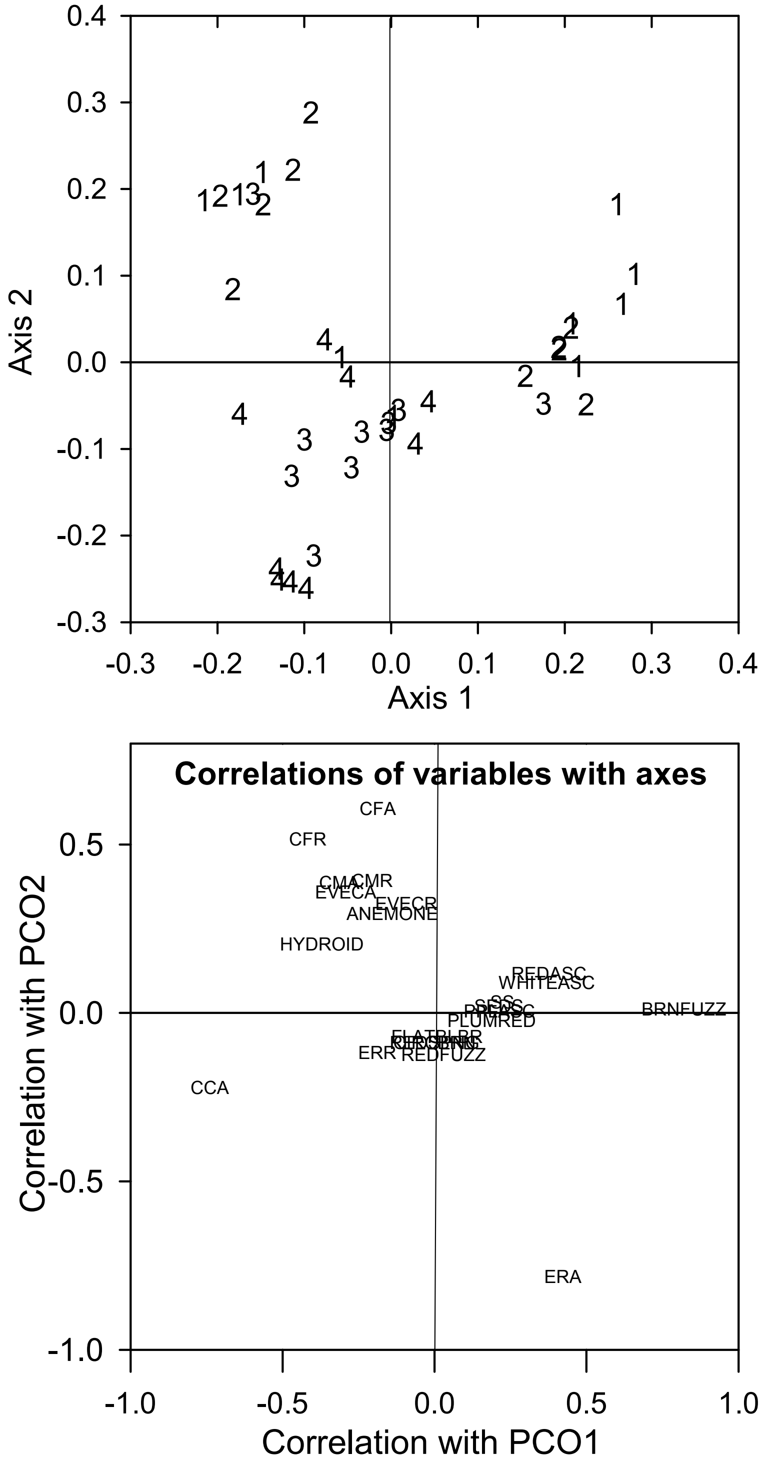


Figure 53: Canonical analysis of principal coordinates of quadrat count data, spring 2003. Upper panel shows sites (scores for 10 quadrats at each site) on the first two coordinate axes, lower panel plots show correlations of biological variables on each of the two PCO axes, plotted against one another. 1, 2, 3, 4 as upper plot represent the sites.

Table 10: Codes used in principal coordinates analysis.

Code	Taxa
SEDS	sediments
ERA	adult *Ecklonia radiata*
ERR	recruit *Ecklonia radiata*
CFA	adult *Carpophyllum flexuosum*
CFR	recruit *Carpophyllum flexuosum*
SS	*Sargassum sinclairii*
CMA	adult *C. maschalocarpum*
CMR	recruit *C. maschalocarpum*
CIRCBRN	*Colpomenia sinuosa*
FLATBLBR	flat-bladed brown
CODIUM	*Codium adhaerens*
REDFUZZ	red fuzz alga
PLUMRED	plumose red alga
EVECA	adult *Evechinus chloroticus*
EVECR	recruit *Evechinus chloroticus*
CCA	crustose coralline algae
BRNFUZZ	brown fuzz alga
REDASC	red ascidian
WHITEASC	white ascidian
PPLASC	*Botrylloides leachii*
HYDROID	indet. hydroid
REDSPNG	red sponge
ANEMONE	*Actinothoe albocincta*
CORTURF	coralline turf
ORNGSPNG	orange sponge
GREYSPNG	grey sponge
DDMNFNGR	?*Didemnum* sp.
C_pun	*Calliostoma punctulata*
T_vir	*Trochus viridis*
C_pur	*Cantharidus purpureus*
LIMPET	*Cellana stellifera*
Zonaria	*Zonaria turneriana*
C_virgata	*Cominella virgata*
C_bicorn	*Cnemidocarpa bicornuata*

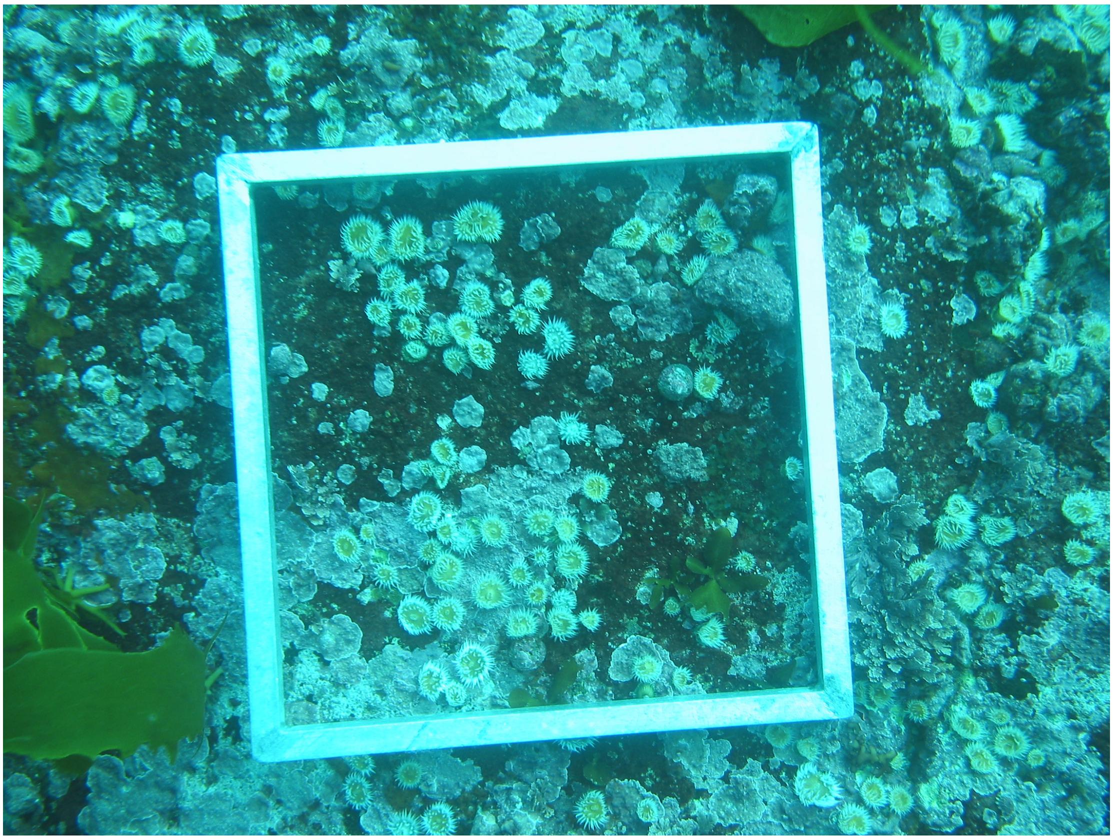


Figure 54: Encrusting (some turfing) coralline algae and anemones (*Actinothoe albocincta*) within 25 cm x 25 cm quadrat at site 3 (Hapuku Rock) (Photo I. Hawes).

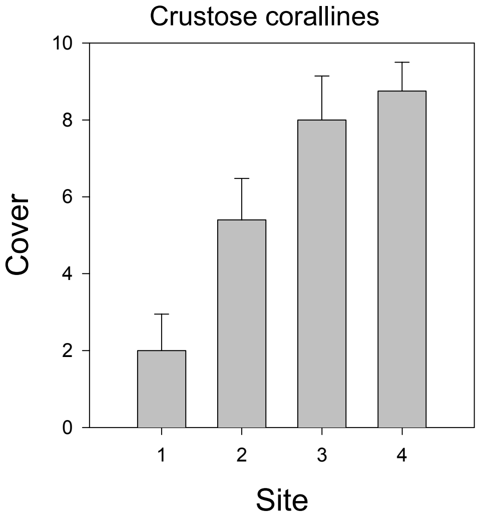


Figure 55: Cover of crustose coralline algae at each of the four sites in spring 2003. Cover was estimated from 10 random points from each of 5 quadrats in 5–10 m and 10–15 m depth strata. Depths are combined in this plot.

In April 2004, species composition varied greatly among sites for count data (Figure 56) and less so for percentage cover data (Figure 58). The fauna at sites 2, 3, and 4 (Kingfish, Hapuku, and Pupuha) (Figure 57a) were as different among themselves as they were from site 1 (Cook) for the count data (see Figure 56). While in April 2003 site 1 had a lower percentage cover of coralline algae than the other sites (see Figure 55), in April 2004, all 4 sites had high covers of crustose coralline algae and small fuzzy red algae (Figure 58). The sum of percentage cover of sediments (SEDS) was, however, clearly highest at Cooks Beach (Figure 58, Figure 57b).

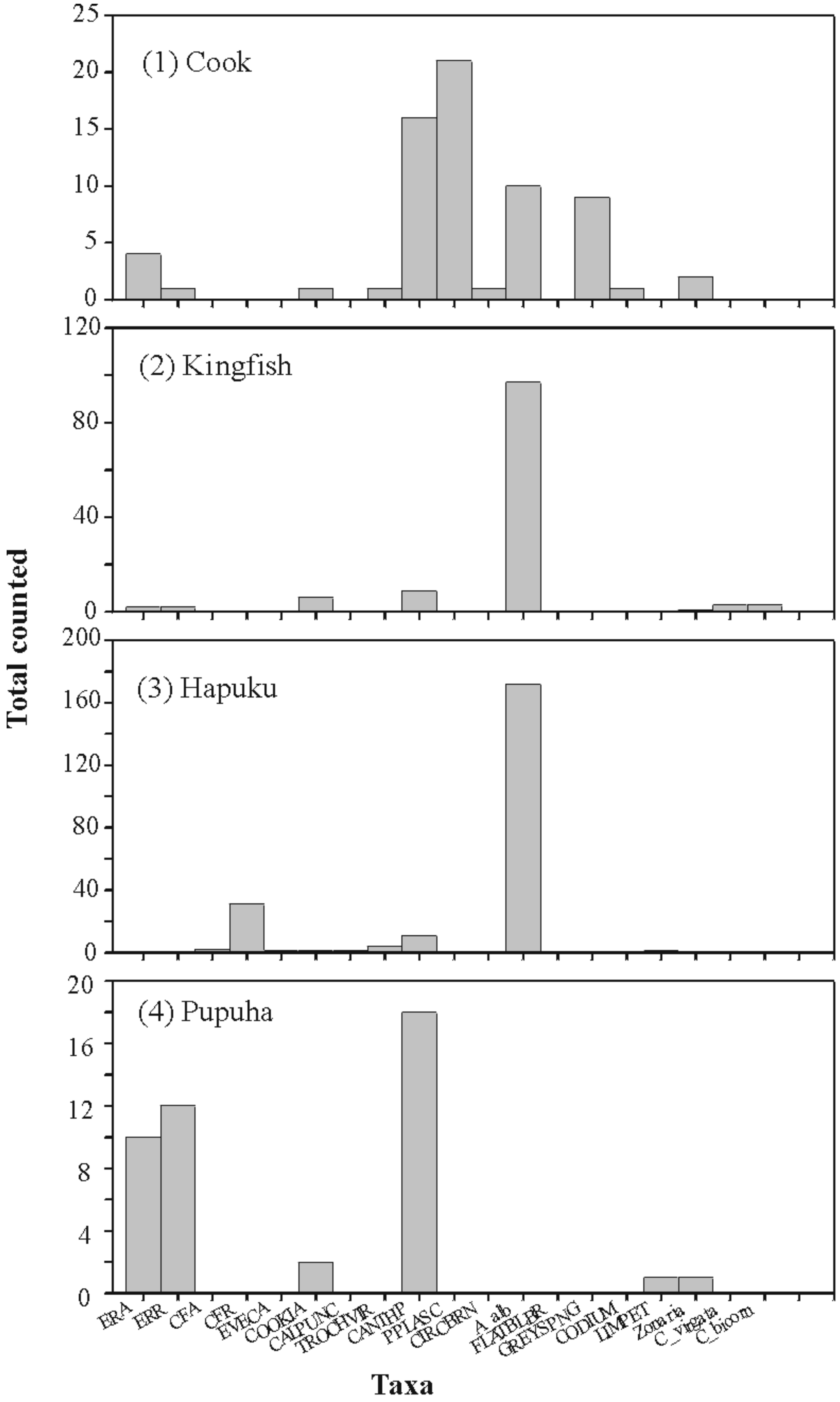


Figure 56: Totals of each taxon counted at each site in April 2004, from site 1 (Cook) closest to sediment source at top, to site 4 (Pupuha), distant from sediment source at bottom. Species codes are given in Appendix 2.

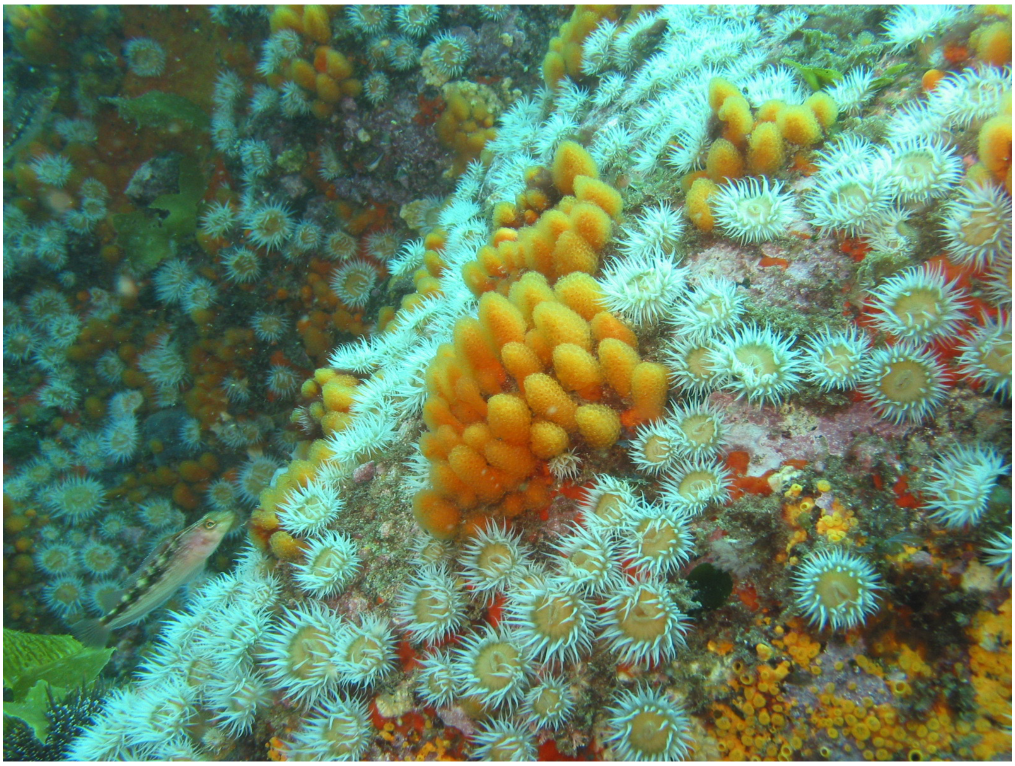


Figure 57a: *Pseudodistoma novaezealandiae* and *Actinothoe albocincta* at site 3 (Hapuku) in April 2004. (Photo R. Wells).

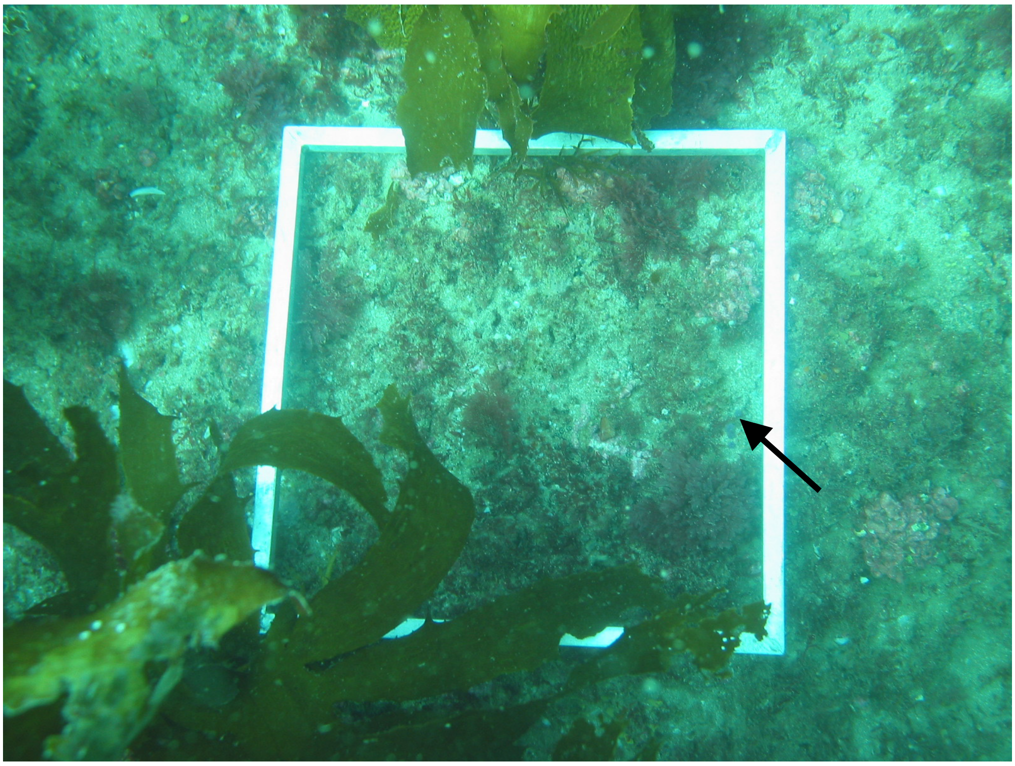


Figure 57b: Brown fuzz, coralline turf, and *Pseudodistoma* sp. (arrow) at site 1 (Cook) in April 2004. (Photo R. Budd).

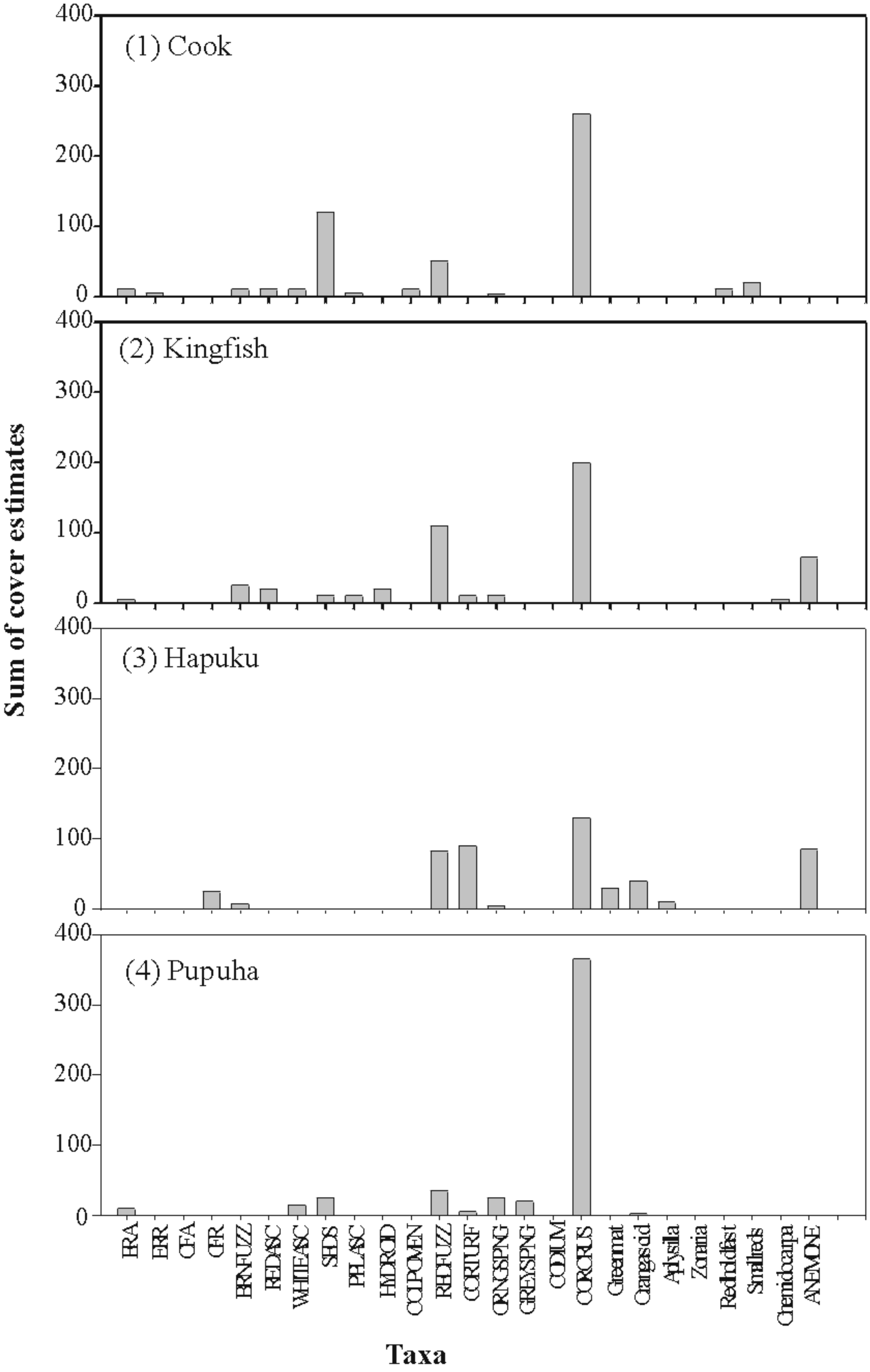


Figure 58: Sums of estimates of cover per site (for species codes see Table 10).

The number of individuals per quadrat and number of species per quadrat were highest at site 3 (Hapuku) (Figure 59). However, trends among the sites differed for numbers of individuals and number of species; the mean number of individuals was lowest at site 4 (Pupuha) and then increased from site 1 (Cook) to site 3 (Hapuku). The number of countable species was highest at site 3, and all three other sites were similar to one another. The pattern in number of species for the percent cover data was similar to that for countable taxa; site 3 was distinct from the other sites (Figure 60).

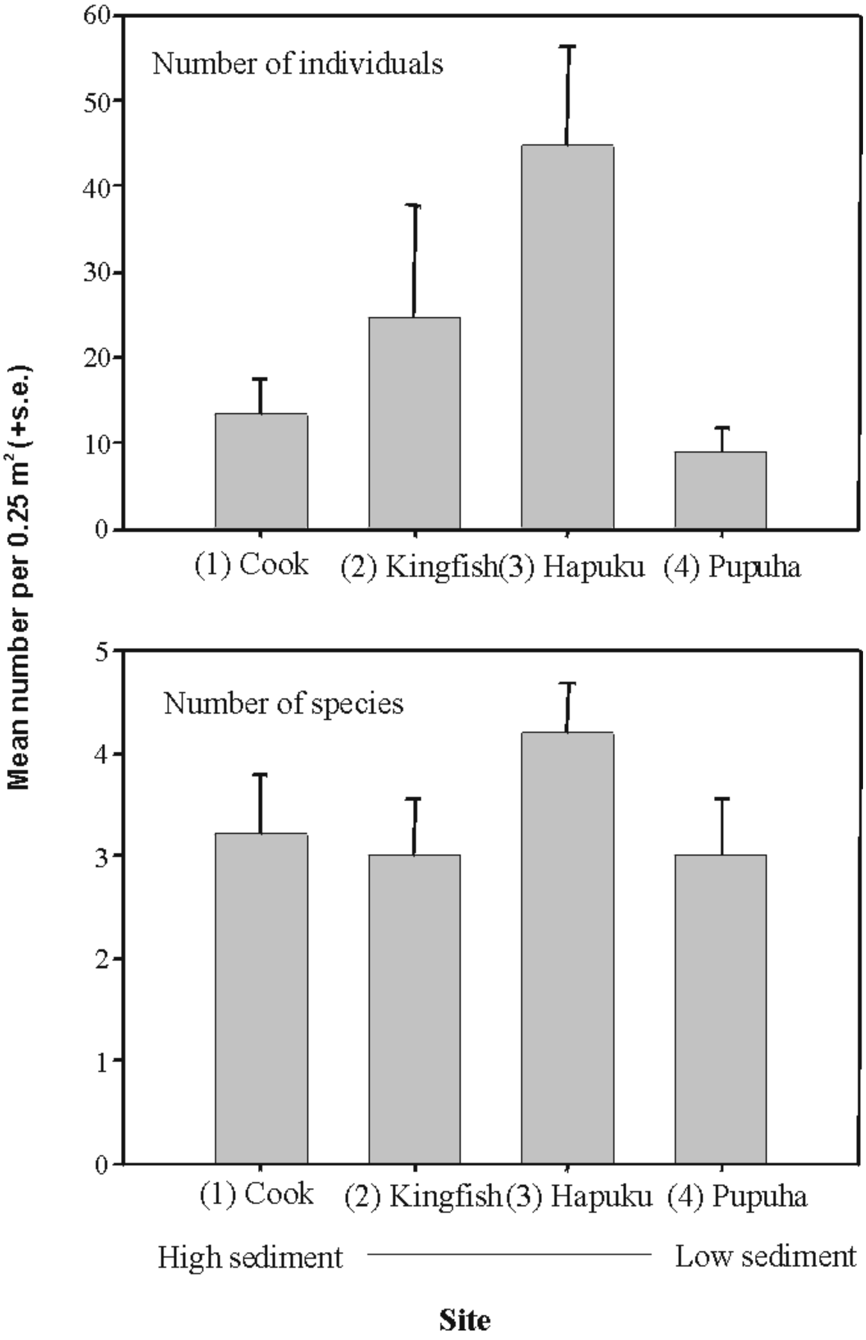


Figure 59: Mean number of individuals (upper) and number of species per site during April 2004 survey.

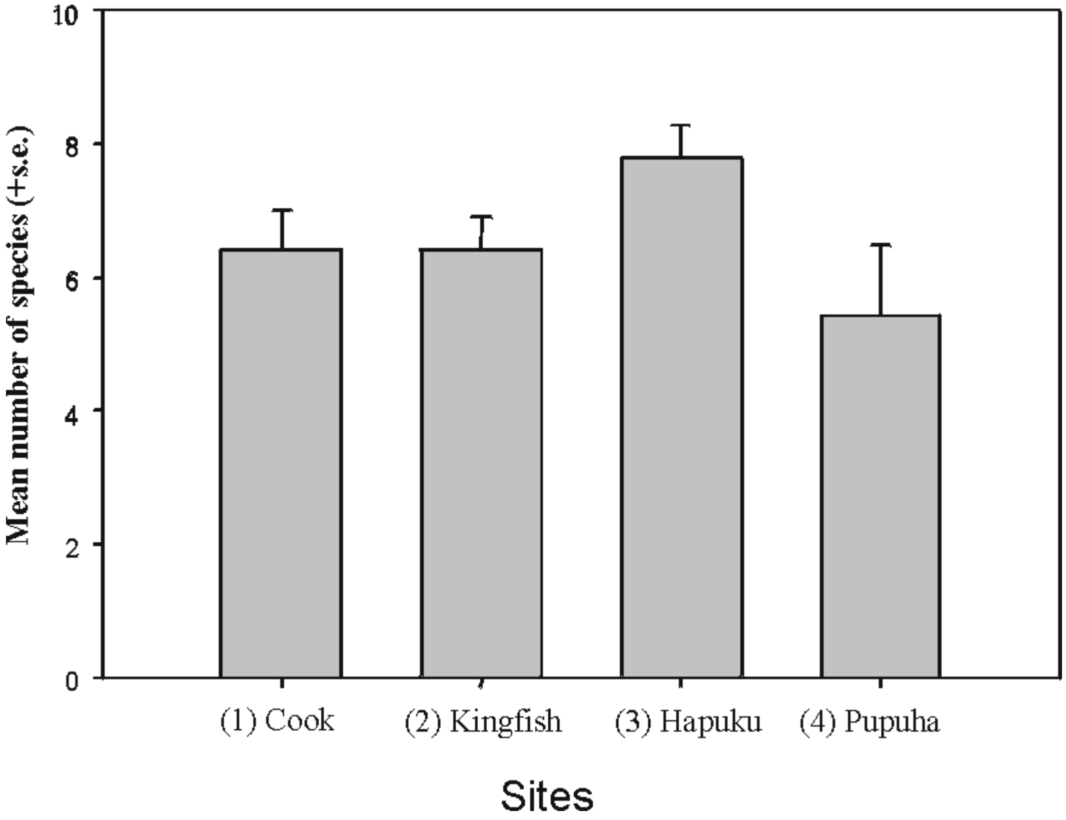


Figure 60: Mean number of species per quadrat at 4 sites.

The first three axes from the constrained CAP analyses of count data accounted for 28, 21, and 14% of variation (Figure 61 A–C). Biplots of quadrat scores on those axes (Figure 62, A–C) emphasised differences between sites 3 and 4. There was high variability among quadrats for sites 1 and 2 on axis 2, whereas quadrats within sites 3 and 4 were each more tightly clustered. The patterns were associated with high abundances of *Colpomenia* and the purple ascidian Botrylloides leachii in three quadrats at site 1, high abundances of *Actinothoe* in four quadrats at site 3 and one at site 2, in contrast to high abundances of *Ecklonia* at site four (Figure 62, D–F).

The first three axes from the constrained CAP analyses of cover data accounted for 36, 21, and 15% of variation (Figure 61, D–F). Biplots of quadrat scores on those axes (Figure 63, A–C) showed site 1 separating from sites 2 and 3 on CAP1, whereas sites 2 and 3 were spread on CAP2. The dominant variable contributing to positive scores on CAP1 was sediments, and scores for quadrats from site 1 were tightly clustered on the positive side of that axis (Figure 63, D–F). Taxa positively associated with sediments included *Zonaria*, red and brown fuzzy algae, and an unidentified small brown alga.

Zonaria was also associated with CAP1, and with negative scores on CAP2, but the association of those variables with CAP scores was undermined by the association between quadrats from sites 1 and 4 occurring with positive scores on CAP2. These appeared to be associated with an orange sponge (Figure 63, F).

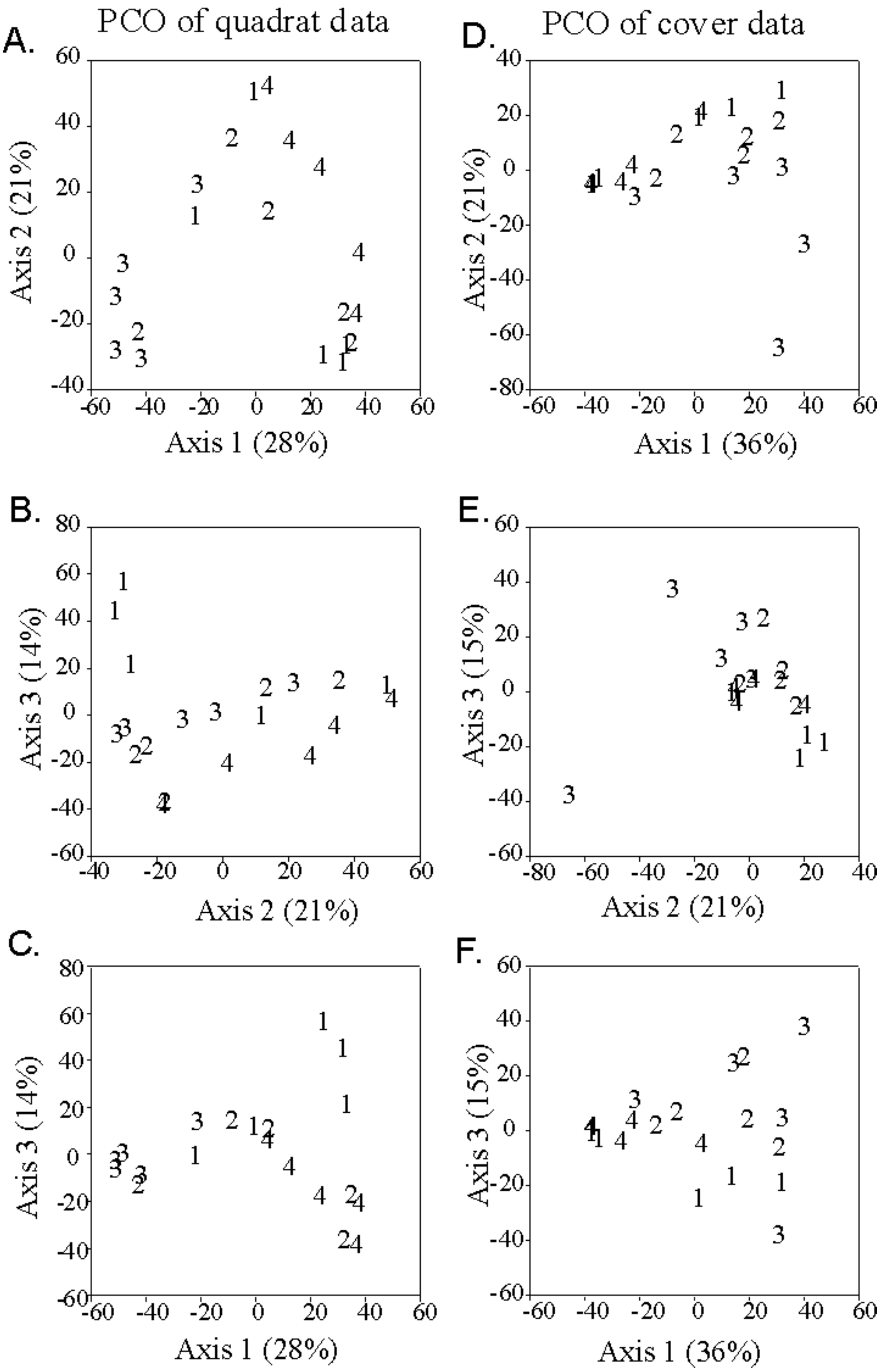


Figure 61: Unconstrained CAP (PCO) analysis of (left column) quantitative (i.e. discrete organisms which could be counted) and (right column) percent cover data for April 2004 survey

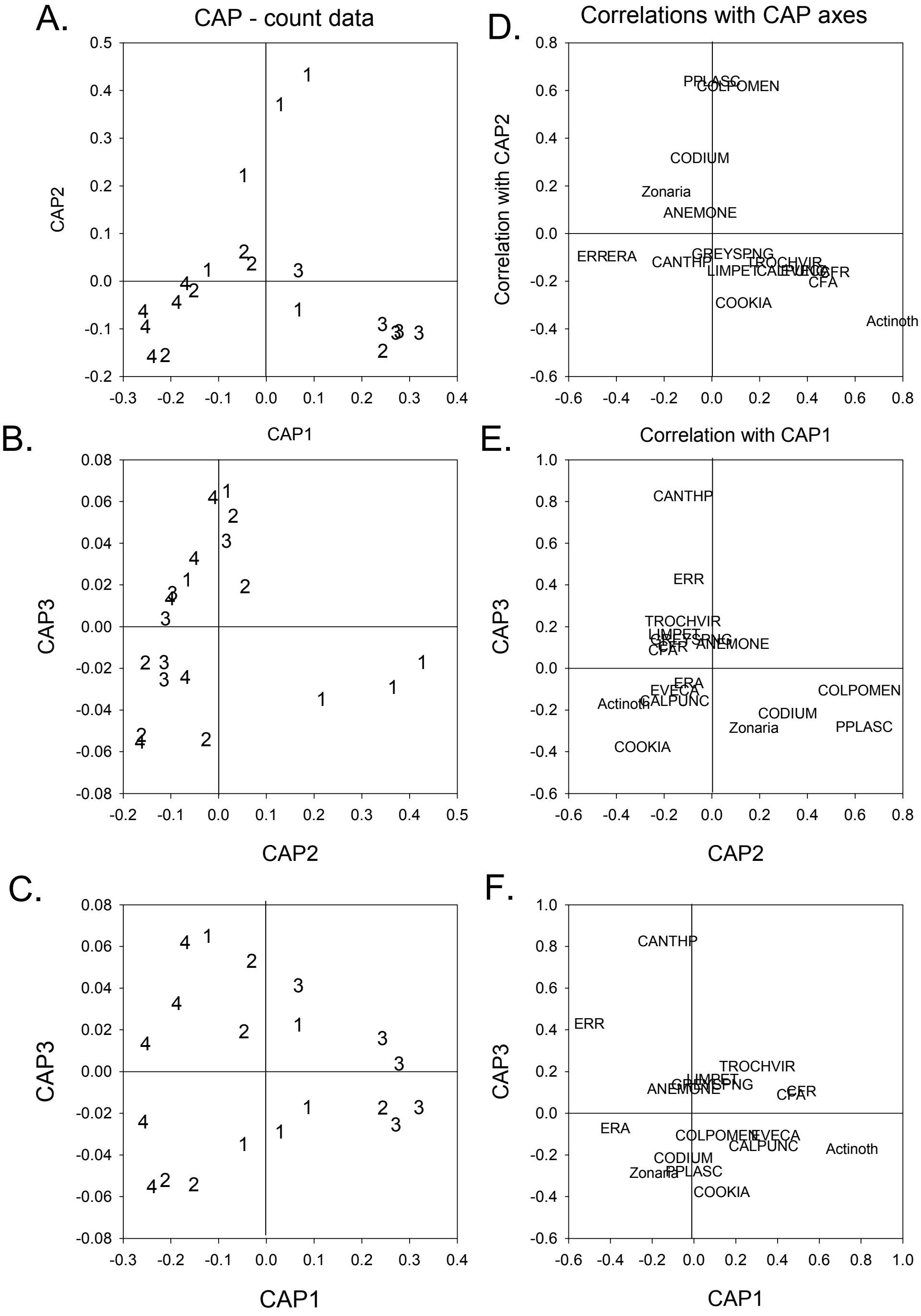


Figure 62: CAP analyses for count data for April 2004. Plots of quadrat scores on CAP axes (A–C), and correlations of count variables with those axes (D-F).

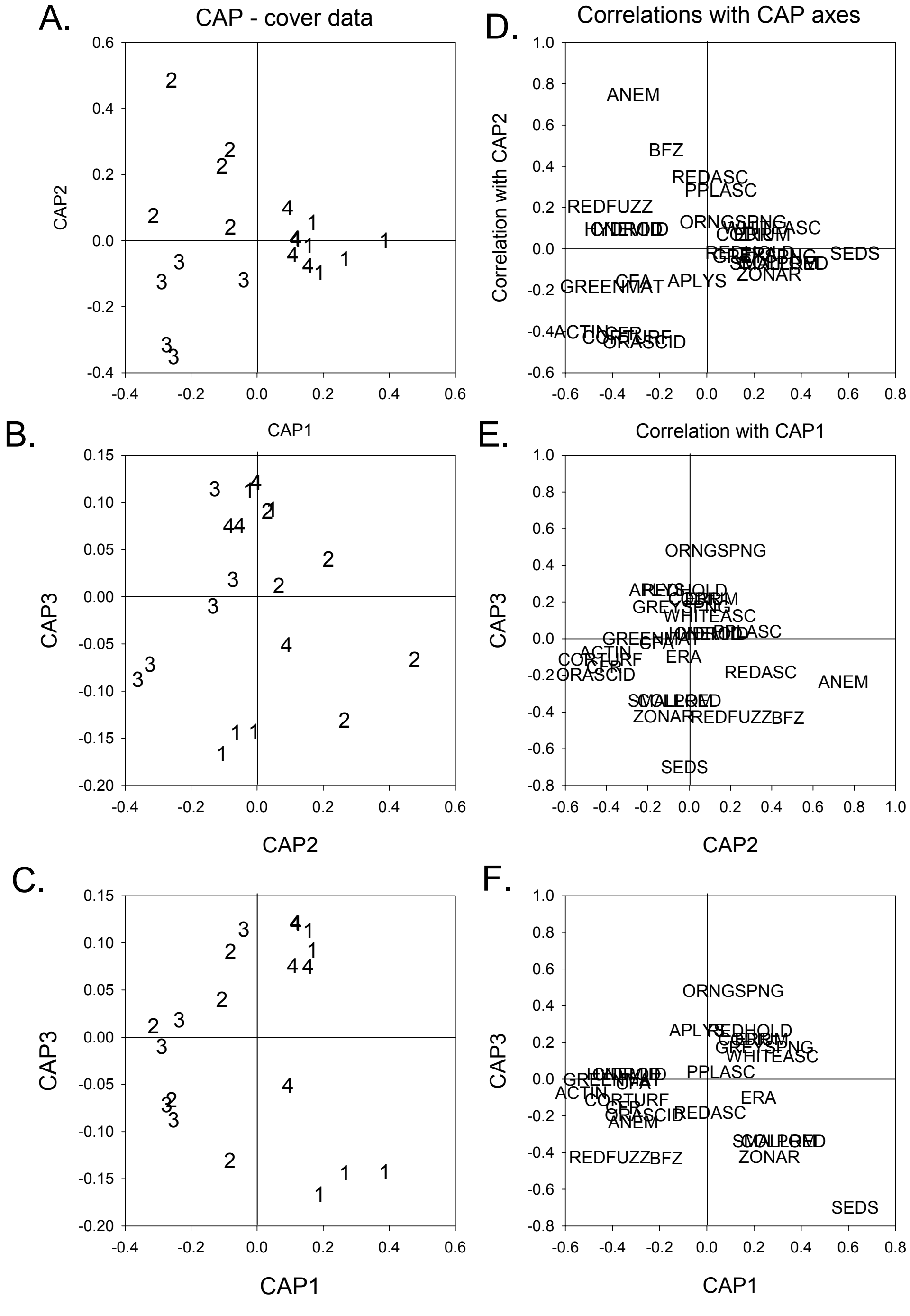


Figure 63: CAP analyses for cover data. Plots of quadrat scores on CAP axes (A–C), and correlations of count variables with those axes (D–F).

7.3 Discussion

The quadrat data did not reveal strong associations of abundances or covers of organisms with the sediment gradient. This might be owing to: (i) limited effect of sediments on the organisms that were quantified; (ii) the impact being most clearly expressed in other depth strata; (iii) the effect being expressed on qualitative attributes of organisms, rather than their abundances; (iv) temporal variation in abundances of organisms obscuring patterns along the spatial gradient of sedimentation; and (v) temporal variation from other disturbances, such as storms, overwhelming or obliterating patterns that might be due to sediments.

These results suggest that at these levels of sedimentation there was no strong effect on the abundance or biodiversity of encrusting organisms. These findings therefore support the results from epifauna and *Ecklonia radiata* in suggesting that these sediment levels reduce productivity in a chronic way rather than exclude taxa or significantly affect abundance. This sort of information enables us to start proposing thresholds at which organisms become “less resilient” and so more susceptible to one-off events such as phytoplankton blooms and acute sediment events (e.g., Cole & Babcock 1996).

8. CONCLUSIONS AND SUMMARY

The objective of this study was to 'assess potential direct and indirect impacts of terrestrial runoff on rocky reef biodiversity by establishing predictive relationships between characteristics of key benthic habitat structuring organisms and their responses to terrestrial sediment inputs from in situ and laboratory experiments.'

From such relationships we aimed to be able to formulate hypotheses about the influence of sediments on grazer-algal interactions.

In the first instance we established that rocky reefs near both the Whitianga Harbour mouth and the Hutt River mouth were influenced by terrestrial-derived suspended sediments over the period of the study (March 2003–July 2004). The terrestrial sediment influence had indirect (via water clarity) and direct (through filter feeders) impacts resulting in reduced productivity for some taxa (*Ecklonia radiata* and associated epifauna) and mortality for others (larval paua and kina).

8.1 Indirect effects

Of the four sites at Hahei, site 1 (Cook) had the lowest water clarity. During times of turbid water inflows from the Whitianga Harbour mouth there was a gradient in water clarity from site 1 to site 4, otherwise sites 2, 3, and 4 had a similar clarity. Although there were times when water clarity was equally good at all four sites, in general, site 1 can be subject to not only lower water clarity than the other sites at any given time, but also to reduced clarity over longer time-scales than the other sites.

Differences between site 1 and the other three sites have become evident in the biological investigations as well. For example the lowest density, biomass, and productivity of epifauna on the naturally occurring *E. radiata* were found at site 1, and the greatest degree of acclimation to a low light environment for a given water depth was evident in *E. radiata* from site 1.

8.2 Direct effects on filter feeding organisms

The effects of suspended sediments on filter feeders collected from the Hahei site were largely sub-lethal within the range of sediment concentrations experienced in this location. However there was evidence of an increased energy requirement in order to be resilient to turbid events and there was evidence of important synergistic effects of duration and exposure. In the laboratory sponges, mussels, and oysters all showed signs of physiological stress through altered feeding rate response to elevated suspended sediment concentration.

Mussels showed continued increases in clearance rate with increasing suspended sediment concentration, even at the highest concentration investigated, thus requiring the use of energy that would otherwise be kept as tissue weight. This response consequently caused significant drop in mussel condition above 26 mg l^{-1}.

Both oysters and sponges lowered their feeding rate in response to increased suspended sediment concentration, resulting in lower nutritional gain and a loss in animal condition at suspended sediment concentration over 15 mg l^{-1}. Oysters showed functional response to suspended sediment concentrations over 100 mg l^{-1} during the experiment by varying their clearance and feeding rates.

The amphipod *Aora typica* showed an average survival rate of over 65% for all treatments up to 1000 mg l^{-1}. Although survival in all treatments dropped over the duration of the experiment, decrease in survival was fastest (and greater) in high suspended sediment treatments. The evidence suggested that mean survival of *Aora* logarithmically decreased with increasing suspended sediment concentrations, and that increasing duration of exposure led to increasing sensitivity.

8.3 Larval stages

In contrast to the adult filter feeding organisms from Hahei, experiments in Wellington found that effects of sediments for larval paua and kina are more likely to be lethal than non-lethal. The two test organisms in this study, paua and kina, have fundamental differences in life-history strategies, morphologies, and larval durations, but sediments increased larval mortality rates of both. Patterns of mortality differed between species.

- Paua showed greater sensitivity to suspended sediments. The same range of concentrations increased cumulative mortality at larval competency by up to 49% for paua, and up to 27% for kina. Furthermore, paua showed similar responses for chronic exposure to suspended sediments across the range of tested concentrations; kina showed a more graded response.

- Effects of timing of exposure differed. For paua, early acute exposure was highly deleterious, resulting in greatly increased larval mortality that was manifested in subsequent stages (including some suggestion that consequences may "carry-over" to juvenile stages). Kina exposed to sediments early in development showed some capacity to recover, and those exposed late had cumulative mortality more similar to larvae that were chronically exposed to sediments in high concentrations.

There was little evidence for variation in developmental abnormalities attributable to suspended sediment regimes during larval development in the laboratory. In contrast, there was a high incidence of developmental abnormalities in harbour waters compared to those larvae held in filtered seawater in the laboratory. This discrepancy suggests the importance of other environmental factors associated with terrestrial runoff, which, alone or through an interaction with sediments, may produce developmental abnormalities.

8.4 Kelp and understory taxa

At the start of this study we predicted that structural complexity of rocky reef communities is determined by the characteristics (diversity, predictability, quantity) of macroalgae, which are in turn determined by the characteristics of the sediment affected light environment. We have not detected major differences in structural complexity of understorey organisms in this sampling over the ranges of suspended sediment that we have encountered. However the reduced photosynthetic potential of *E. radiata* certainly points to reduced production of all primary producers, including epiphytes on *E. radiata*.

Epifauna on *E. radiata* graze on epiphytes on the surface of the macroalga. The noted reduction in epifaunal productivity at site 1 suggests that anthropogenically increased suspended sediment concentrations on New Zealand's rocky reefs generally have the potential to have serious consequences at the ecosystem level, since epifauna are responsible for about 80% of the flow of energy and materials through rocky reef animal communities (Taylor 1998). Although the functional role of seaweed epifauna is poorly understood at present, it is likely that density reductions will have knock-on effects throughout the rocky reef foodweb, downwards through reduced epifaunal grazing on seaweeds and algal epiphytes (Duffy 1990), and upwards through reduced availability of food for small fishes (Jones 1988).

8.5 Thresholds for effects

The ranges of suspended sediment concentrations measured at both sites and used in experiments are illustrated in Figure 64. Concentrations used in experiments for taxa from both Wellington and Whitianga intentionally overlapped to increase our ability to compare and contrast findings. The various strands of this study have shown consistent findings with respect to chronic and toxic effects of a range of suspended sediment concentrations on a range of organisms. We tentatively suggest a series of thresholds (minimum concentrations for effect) for effects based on the findings from this study (Figure 64). This is a rough indication only, as duration of exposure to these concentrations is not allowed for except as described in the relevant chapters. All thresholds fall well below the levels that some rocky reef environments are exposed to at both Whitianga and the Hutt River. As a further comparison, time-averaged measurements of suspended sediment concentration at near-shore coastal sites at New Plymouth, where *E. radiata* also occurs, have been measured at concentrations ranging from ranging from 0.0034 g l^{-1} to 0.1524 g l^{-1} at about 0.5 m (R. Cole, pers. comm.), similar to the range in the Hutt River.

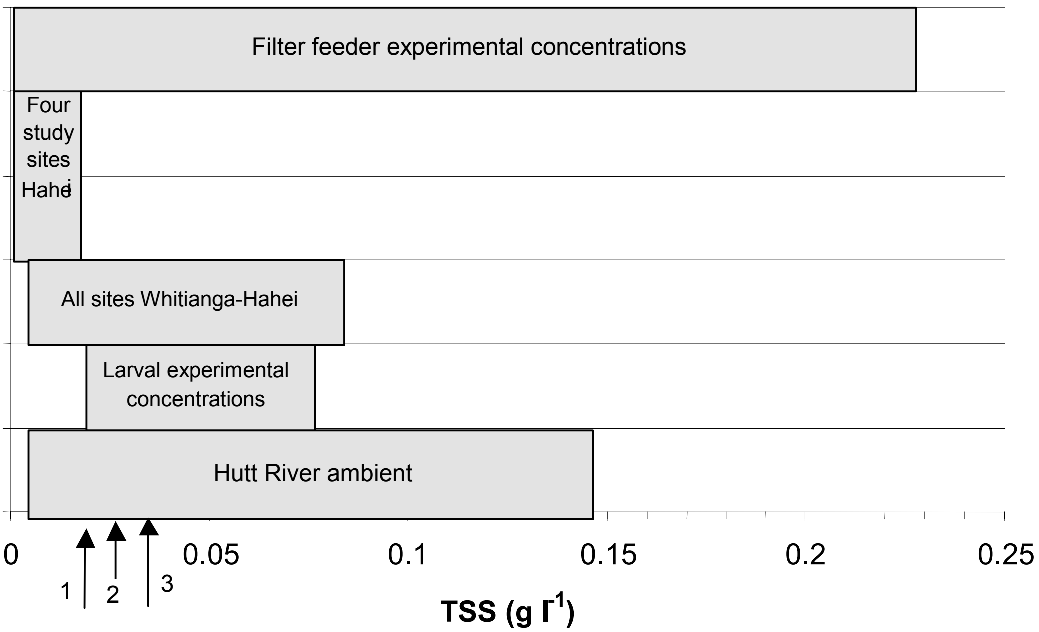


Figure 64: The ranges of total suspended sediment (TSS) concentration measured at the Whitianga and the Hutt River locations, March 2003 to July 2004, and the ranges of suspended sediment concentration used in laboratory experiments

Threshold 1: At the upper end of the concentration range at the four Hahei sites productivity of *E. radiata* and *epifauna* was quantitatively reduced compared to all other sites suggesting a 'reduced productivity' threshold.

Threshold 2: Significant drop in mussel condition

Threshold 3: Paua and kina larval mortality.

The approach that we have taken has suggested putative causative relationships between terrestrially derived sediments inputs to the rocky reef environment. However, it is essential that these predictions are tested at multiple New Zealand locations and under a wider range of physical and ecological conditions.

9. ACKNOWLEDGMENTS

Chris Hickey provided extensive underwater light data that had been collected privately and as part of Chris's role on the Marine Reserve committee. This valuable dataset greatly assisted in the planning of this project. We are grateful to Dirk Immenga (Waikato University) for access to side-scan sonar maps of the study region, Peter Johnson from Ngati Hei, Whitianga, for discussions on historical changes in kelp beds in Mercury Bay, Gilly Adam (Department of Conservation), and Gary Vincent (Whitianga Harbourmaster) all of whom assisted us in establishing this project. We also acknowledge NIWA staff who have assisted in the laboratory and the field, especially Carolyn Lundquist, Iain MacDonald, Stephen Brown, Michael Ahrens, Louise Hunt, and Mike Page. Ian Hawes, Carolyn Lundquist, and Nicole Hancock provided reviews of this report.

This project was funded by the Ministry of Fisheries under Contract ZBD2002/06A.

10. REFERENCES

Airoldi, L; Virgilio, M. (1998). Responses of turf-forming algae to spatial variations in the deposition of sediments. *Marine Ecology Progress Series 165*: 271–282.

ASTM (1993). Standard guide for conducting acute toxicity tests on aqueous effluents with fishes, macroinvertebrates and amphibians E 1192 - 88. *In*: Annual Book of ASTM Standards, pp. 853–865. American Society for Testing and Materials, Philadelphia, Pa. USA.

Anderson, M.J.; Willis, T.J. (2003). Canonical analysis of principal coordinates: a useful method of constrained ordination for ecology. *Ecology 84*: 511–525.

Anon, (2000). The New Zealand Biodiversity Strategy. Our chance to turn the tide. Department of Conservation and Ministry for the Environment. 146 p.

Beer, S.; Axelsson, L. (2004). Limitations in the use of PAM fluorometry for measuring photosynthetic rates of macroalgae at high irradiances. *European Journal of Phycology 39*: 1–7.

Beer, S.; Björk, M. (2000). Measuring rates of photosynthesis of two tropical seagrasses by pulse amplitude modulated (PAM) fluorometry. *Aquatic Botany* 66: 69–76.

Beer, S.; Larsson, C.; Poryan, O.; Axelsson, L. (2000). Photosynthetic rates of *Ulva* (Chlorophyta) measured by pulse amplitude modulated (PAM) fluorometry. *European Journal of Phycology* 35: 69–74.

Büchel, C.; Wilhelm, C. (1993). In vivo analysis of slow chlorophyll fluorescence induction kinetics in algae: progress, problems and perspectives. *Photochemistry and Photobiology* 58: 137–148.

Clarke, K.R.; Warwick, R.M. (1994). Change in marine communities: an approach to statistical analysis and interpretation. Plymouth Marine Laboratory, Plymouth, UK. 144 p.

Cole, R.G.; Babcock, R.C. (1996). Mass mortality of a dominant kelp (Laminariales) at Goat Island, north-eastern New Zealand. *Marine and Freshwater Research 47*: 907–912.

Davies-Colley, R.J.; Vant, W.N.; Smith, D.G. (1993). Colour and clarity of natural waters. Science and management optical water quality. 310 p.

Dayton, P.K.; Robillard, G.A.; Pain, R.T. (1970). Benthic faunal zonation as a result of anchor ice formation in McMurdo Sound, Antarctica. *In*: Holdgate, M.W. (ed.) Antarctic ecology. Academic Press, London. pp. 244–258.

Dromgoole, F.I. (1988). Light fluctuations and the photosynthesis of marine algae. 2. Photosynthetic response to frequency, phase ratio and amplitude. *Functional Ecology 2*: 211–219.

Duffy, J.E. (1990). Amphipods on seaweeds: partners or pests? *Oecologia 83*: 267–276.

Duncan, M.J.; Harrison, P.J. (1982). Comparison of solvents for extracting chlorophylls from marine macrophytes. *Botanica Marina 25*: 445–447.

Dyer, K.R. (1986). Coastal and estuarine sediment dynamics. John Wiley and Sons, Chichester. 342 p.

Edgar, G.J. (1990). The use of the size structure of benthic macrofaunal communities to estimate faunal biomass and secondary production. *Journal of Experimental Marine Biology and Ecology 137*: 195–214.

Edgar, G.J. (1991). Distribution patterns of mobile epifauna associated with rope fibre habitats within the Bathurst Harbour estuary, south-western Tasmania. *Estuarine Coastal and Shelf Science 33*: 589–604.

Edgar, G.J. (1993). Measurement of the carrying capacity of benthic habitats using a metabolic-rate based index. *Oceologia 95*: 115–121.

Edgar, G.J.; Shaw, C. (1995). The production and trophic ecology of shallow-water fish assemblages in southern Australia. II. Diets of fishes and trophic relationships between fishes and benthos at Western Port, Victoria. *Journal of Experimental Marine Biology and Ecology 194*: 83–106.

Ellis, J.; Cummings, V.; Hewitt, J.; Thrush, S.; Norkko, A. (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*: 147–174.

Estes, J.A.; Duggins, D.O.; Rathbun, G.B. (1989). The ecology of extinctions in kelp forest communities. *Conservation Biology 3*: 252–264.

Fahey, B.D.; Coker, R.J. (1992). Sediment production from forest roads in Queen Charlotte forest and potential impact on marine water quality, Marlborough Sounds, New Zealand. *New Zealand Journal of Marine and Freshwater Research*: *26*: 187–195.

Fairhead, V.A.; Cheshire, A.C. (2004). Seasonal and depth related variation in the photosynthesis-irradiance response of *Ecklonia radiata* (Phaeophyta, Laminariales) at West Island, South Australia. *Marine Biology 145*: 415–426.

Flameling, I.A.; Kromkamp, J. (1998). Light dependence of quantum yields for PSII charge separation and oxygen evolution in eucaryotic algae. *Limnology and Oceanography 43*: 284–297.

Foster, M.S. (1982). The regulation of macroalgal associations in kelp forests. *In* Srivastava, L. (ed.), Synthetic and degradative processes in marine macrophytes, pp. 185–2-5W. de Gruyter, Berlin.

Foster, G.; Carter, L. (1997). Mud sedimentation on the continental shelf at an accretionary margin – Poverty Bay, New Zealand. *New Zealand Journal of Geology and Geophysics 40*: 157–173.

Geel, C.; Versluis, W.; Snell, J.F.H. (1997). Estimation of oxygen evolution by marine phytoplankton from measurement of the efficiency of photosystem II electron flow. *Photosynthesis Research 51*: 61–70.

Geisen, W.B.J.T.; van Katwijk, M.M.; den Hartog, C. (1990). Eelgrass condition and turbidity in the Dutch Wadden Sea. *Aquatic Botany 37*: 71–85.

Genty, B.; Briantais, J.M; Baker, N.R. (1989). The relationship between the quantum yield of photosynthetic electron transport and quenching of chlorophyll fluorescence. *Biochimica et Biophysica Acta 990*: 87–92

Glasby, G.P. (1991). Mineralogy, geochemistry, and origin of Pacific red clays: A review. *New Zealand Journal of Geology and Geophysics 34*: 167–176.

Gray, J.S. (1997). Marine biodiversity: patterns, threats and conservation needs. *Biodiversity and Conservation* 6: 153–175.

Hall, J.A.; Golding, L.G. (1998). Standard methods for whole effluent toxicity testing: development and application. NIWA Client Report November 1998. National Institute of Water and Atmospheric Research, Hamilton.

Hanelt, D.; Uhrmacher, S.; Nultsch, W. (1995). The effect of photoinhibition on photosynthetic oxygen production in the brown alga *Dictyota dichotoma*. *Botanica Acta 108*: 99–105.

Hicks, D.M.; Griffiths, G.A. (1992). Sediment load. *In*: Mosley, M.P. (ed.), Waters of New Zealand. Caxton Press, Christchurch, pp. 229–248.

Jassby, A.D.; Platt, T. (1976). Mathematical formulation of the relationship between photosynthesis and light for phytoplankton. *Limnology and Oceanograp*hy *21*: 540–547.

Jones, G.P. (1988). Ecology of rocky reef fish of north-eastern New Zealand: a review. *New Zealand Journal of Marine and Freshwater Research 22*: 445–462.

Keddy, P.A. (1991). Working with heterogeneity: an operator's guide to environmental gradients. *In*: Kolasa, J.; Pickett, S.T.A. (eds). Ecological heterogeneity. Springer-Verlag, Berlin. pp. 181–201.

Kirk, J.T.O. (1994). Light and photosynthesis in aquatic ecosystems. Second edition. Cambridge University Press. 509 p.

Kirk, J.T.O.; Davies-Colley, R. J.; Shooter, D.; Schwarz A. (1998). Optical properties of ocean waters in the vicinity of the Chatham Rise, South Pacific Ocean. *Marine and Freshwater Research 49*: 1–6.

Kirkman, H. (1989). Growth, density and biomass of *Ecklonia radiata* at different depths and growth under artificial shading off Perth, Western Australia. *Australian Journal of Marine and Freshwater Research 40*: 169–177.

Krause, G.H.; Weis, E. (1991). Chlorophyll fluorescence and photosynthesis: the basics. *Annual Revue Plant Physiology. Plant Molecular Biology 42*: 313–349.

Lamont, B.B. (1995). Testing the effect of ecosystem composition/structure on its functioning. *Oikos 74*: 283–295.

Mann, K.H.; Kirkman, H. (1981). Biomass methods for measuring productivity of *Ecklonia radiata*, with the potential for adaptation to other large brown algae. *Australian Journal of Marine and Freshwater Research 32*: 297–304.

Markager, S.; Sand-Jensen, K. (1992). Light requirements and depth zonation of marine macroalgae. *Marine Ecology Progress Series 88*: 83–92.

Myers, A.A.; Moore, P.G. (1975). The New Zealand and south-east Australian species of Aora Kroyer (Amphipoda, Gammaridea). *Records of the Australian Museum 35*: 167–180.

Novaczek, I. (1984). Development and phenology of *Ecklonia radiata* at two depths in Goat Island Bay, New Zealand. *Marine Biology 81*: 189–197.

Phillips, N.E. (2002). Effects of nutrition-mediated larval condition on juvenile performance in a marine mussel. *Ecology 83(9)*: 2562–2574.

Ralph, P.J.; Gademann, R.; Dennison, W.C. (1998). *In situ* seagrass photosynthesis measured using a submersible, pulse-amplitude modulated fluorometer. *Marine Biology 132*: 367–373.

Schiel, D.R. (1984). Poor Knights Islands marine reserve: a biological survey of subtidal reefs. Leigh Marine Laboratory, University of Auckland. 93 p.

Schiel, D.R. ; Foster, M.S. (1986). The structure of subtidal algal stands in temperate waters. *Oceanography and Marine Biology, an Annual Review 24*: 265–307.

Schreiber, U.; Schliwa, U.; Bilger, W. (1986). Continuous recording of photochemical and non-photochemical chlorophyll fluorescence quenching with a new type of modulation fluorometer. *Photosynthesis Research 10*: 51–62.

Strathmann, M. F. (1987). Reproduction and development of the marine invertebrates of the northern Pacific coast, data and methods for the study of eggs, embryos and larvae. University of Washington Press, Seattle. 670 p.

Taylor, R.B. (1998). Density, biomass and productivity of animals in four subtidal rocky reef habitats: the importance of small mobile invertebrates. *Marine Ecology Progress Series 172*: 37–51.

Underwood, A.J.; Denley, E.J.; Moran, M.J. (1983). Experimental analyses of the structure and dynamics of mid-shore rocky intertidal communities in New South Wales. *Oceologia 56*: 202–219.

Appendix 1: Data for amphipod survival shown in Figure 28 for days 1, 2, 3, 5 and 7. SS = Suspended sediment concentration in mg.l^{-1}. Mean + standard errors.

SS	1	2	3	5	7
1	0.98 + 0.02	0.97 + 0.02	0.97 + 0.02	0.97 + 0.02	0.85 + 0.02
15	0.93 + 0.05	0.90 + 0.04	0.90 + 0.04	0.85 + 0.03	0.85 + 0.03
60	0.95 + 0.05	0.88 + 0.06	0.88 + 0.06	0.85 + 0.06	0.80 + 0.06
95	0.95 + 0.03	0.88 + 0.03	0.85 + 0.03	0.85 + 0.03	0.80 + 0.07
190	0.95 + 0.03	0.85 + 0.03	0.83 + 0.03	0.80 + 0.00	0.68 + 0.09
280	0.95 + 0.03	0.83 + 0.05	0.80 + 0.04	0.78 + 0.05	0.78 + 0.05
1000	0.90 +0.00	0.75 + 0.12	0.75 + 0.12	0.70 + 0.14	0.68 + 0.14

Appendix 2: Species list of macroalgae identified from collections in September 2003 and April 2004.

Macroalgal Taxa
Acrosorium venulosum
Antithamnion applicitum
Aphanocladia delicatula
Apoglossum montagneanum
Arthrocardia corymbosa
Arthrocardia corymbosa
Callithamnion sp.
Callophyllis decumbens
Callophyllis depressa
Callophyllis laingiana
Carpomitra costata
Carpophyllum flexuosum
Carpophyllum maschalocarpum
Ceramium sp.
Champia laingii
Cladhymenia coronata
Cladhymenia lyallii
Cladhymenia sp.
Cladophora sp.
Cladophoropsis herpestica
Codium cranwelliae
Corallina officinalis
non-geniculate corallines
Curdiea codioides
Dasya subtilis
Dasyclonium incisum
Delesseriaceae sp.1
Delesseriaceae sp.2
Dictyota ocellata
Dipterosiphonia heteroclada
Distromium skottsbergii
Ecklonia radiata
Erythroglossum sp.
Gigartina sp.
Griffithsia teges
Heterosiphonia tessellata
Hymenena variolosa
Jania sp.
Leathesia sp.
Lomentaria sp.
Perithamnion ceramoides
Peyssonnelia sp.
Platythamnion sp.
Plocamium cirrhosum
Polysiphonia sp.
Pterocladia lucida
Pterosiphonia pennata
Rhodymenia sp.
Sphacelaria sp.
young stages of Fucales
Zonaria aureomarginata
Zonaria turneriana

Appendix 3: List of ascidian and sponge species identified in photographs from April 2004.

Sample #	Site	Phylum	Family	Genus	Species
505-0504	1 Cook	Porifera	Phorbasidae	*Hamigera*	sp.
505-0505	1 Cook	Tunicata	Didemnidae	*Didemnum*	sp.
505-0505	1 Cook	Porifera	Phorbasidae	*Hamigera*	sp.
505-0511	1 Cook	Tunicata	Holoziodae	*Hypsistozoa*	*fasmeriana*
505-0513	1 Cook	Tunicata	Holoziodae	*Hypsistozoa*	*fasmeriana*
505-0513	1 Cook	Tunicata	Pseudodistomiadae	*Pseudodistoma*	sp.
505-0515	1 Cook	Tunicata	Polymastiidae	*Polymastia*	*granulosa*
505-0516	1 Cook	Porifera	Phorbasidae	*Hamigera*	sp.
505-0516	1 Cook	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0517	1 Cook	Porifera	Halichondriidae	*Ciocalypta*	sp.
505-0518	2 Kingfish	Porifera	Tedaniidae	*Tedania*	sp.
505-0519	2 Kingfish	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0519	2 Kingfish	Porifera	Gen. Sp. Indet.		
505-0520	2 Kingfish	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0520	2 Kingfish	Porifera	Gen. Sp. Indet.		
505-0521	2 Kingfish	Porifera	Tedaniidae	*Tedania*	sp.
505-0528	2 Kingfish	Porifera	Dysideidae	*Dysidea*	sp.
505-0539	2 Kingfish	Porifera	Tedaniidae	*Tedania*	sp.
505-0544	2 Kingfish	Porifera	Callyspongiidae	*Callyspongia*	*fistulosa?*
505-0544	2 Kingfish	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0546	2 Kingfish	Porifera	Gen. Sp. Indet.		
505-0555	2 Kingfish	Porifera	Phorbasidae	*Hamigera*	sp.
505-0578	3 Hapuku	Tunicata	Polyclinidae	*gen. Sp. Indet.*	
505-0580	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0581	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0582	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0583	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*

Sample #	Site	Phylum	Family	Genus	Species
505-0583	3 Hapuku	Porifera	Chalinidae	*Haliclona*	sp.
505-0583	3 Hapuku	Tunicata	Styelidae	*Styela*	*plicata*
505-0584	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0584	3 Hapuku	Porifera	Chalinidae	*Haliclona*	sp.
505-0584	3 Hapuku	Tunicata	Styelidae	*Styela*	*plicata*
505-0586	3 Hapuku	Porifera	-	*gen. Sp. Indet.*	
505-0588	3 Hapuku	Porifera	-	*gen. Sp. Indet.*	
505-0589	3 Hapuku	Porifera	Clionidae	*Cliona*	*celata*
505-0593	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0593	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0594	3 Hapuku	Tunicata	Didemnidae	*Didemnum*	sp. *(poss. adamsi)*
505-0595	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
	3 Hapuku	Tunicata	Didemnidae	*Didemnum*	sp. *(poss. adamsi)*
	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	sp.
505-0596	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0596	3 Hapuku	Porifera	Clionidae	*Cliona*	*celata*
505-0597	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0598	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
505-0600	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0603	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0604	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0605	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0606	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0607	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0608	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0609	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0610	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0611	3 Hapuku	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*

Sample #	Site	Phylum	Family	Genus	Species
506-0632	4 Pupuha	Tunicata	Didemnidae	*Leptoclinides*	*marmoreus*
506-0632	4 Pupuha	Porifera	-	*gen. Sp. Indet.*	
506-0633	4 Pupuha				
506-0634	4 Pupuha	Porifera	Mycalidae	*Mycale*	sp.
506-0635	4 Pupuha	Porifera	Tethidae	*Tethya*	sp.
506-0635	4 Pupuha	Tunicata	Didemnidae	*Leptoclinides*	*marmoreus*
506-0636	4 Pupuha	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0641	4 Pupuha	Porifera	Clionidae	*Cliona*	*celata*
506-0642	4 Pupuha	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0643	4 Pupuha	Tunicata	Didemnidae	*Leptoclinides*	*marmoreus*
506-0643	4 Pupuha	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0657	4 Pupuha	Porifera	Mycalidae	*Mycale*	sp. *(orange)*
506-0658	4 Pupuha	Porifera	Mycalidae	*Mycale*	sp. *(orange)*
506-0658	4 Pupuha	Tunicata	Didemnidae	*Leptoclinides*	*marmoreus*
506-0658	4 Pupuha	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0659	4 Pupuha	Tunicata	Pseudodistomiadae	*Pseudodistoma*	*novaezealandiae*
506-0659	4 Pupuha	Porifera	Mycalidae	*Mycale*	sp. *(orange)*