

Effects of mangrove removal on benthic communities and sediment characteristics at Mangawhai Harbour, northern New Zealand

Andrea C. Alfaro

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The spread of mangroves at many locations in temperate northern New Zealand provides a stark contrast to the well-documented trend in mangrove forest decline recorded through the tropics and subtropics. To explore this difference, improved understanding is needed of New Zealand's mangrove ecosystems and how they respond to anthropogenic disturbance. The effect of mangrove removal on the community ecology of mangrove stands and adjacent habitats was investigated within Mangawhai Estuary, northern New Zealand, between March 2004 and September 2006. The vegetation, benthic macrofauna, and sediments were sampled within habitats (marshgrass, mangrove stands, pneumatophore zones, sandflats, and channels) at a treatment site (mangroves removed) and two undisturbed sites, before and after mangrove-removal activities. Mature mangrove habitats had less total abundance and fewer taxa than all the other habitats sampled and were dominated by pulmonate snails (*Amphibola crenata*) and mud crabs (*Helice crassa*). Whereas faunal composition varied seasonally as a result of life-history dynamics, temporal changes could be attributed to mangrove-removal activities. Mangrove eradication was followed by immediate changes in the sediment from a muddy to sandier environment, which favoured an overall increase in the abundance of crabs, snails, and bivalves. However, unexpected topographic catchment reconfigurations in late 2005 may have caused a subsequent increase in the delivery of silt and organic content to the study area and an overall decrease in faunal density in March and September 2006. The study provides direct evidence of the effect of mangroves on sediment and benthic faunal characteristics and the importance of catchment-derived imports to estuarine ecosystems.

Keywords: *Avicennia marina australasica*, benthic fauna, biodiversity, estuarine ecology, mangrove removal, sediment characteristics.

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A. C. Alfaro: Faculty of Health and Environmental Sciences, School of Applied Sciences, Auckland University of Technology, Private Bag 92006, Auckland 1020, New Zealand. Correspondence to A. C. Alfaro: tel: +64 9 9219999 ext. 8197; fax: +64 9 9219627; e-mail: andrea.alfaro@aut.ac.nz.

Introduction

Vascular plants have a strong influence on coastal community structure, function, and successional patterns (Bertness, 1991, 1992; Snelgrove *et al.*, 2000; Bortolus *et al.*, 2002; Levin and Talley, 2002). Earlier studies on plant–animal interactions focused on the effect of plants as (i) sediment modifiers (e.g. light, temperature, and chemistry regulators of benthic habitats; Bertness and Hacker, 1994; Alongi *et al.*, 2000; Levin and Talley, 2002), (ii) a food source (e.g. fresh and detrital organic matter; Peterson *et al.*, 1985; Kieckbusch *et al.*, 2004; Alfaro *et al.*, 2006; Levin *et al.*, 2006), and (iii) structural support (e.g. nursery habitats, coastal stabilization, and run-off filtration; Gleason *et al.*, 1979; Warren and Neiring, 1993; Valiela *et al.*, 2001). However, the specific mechanism by which vascular plants shape these ecosystem-level processes is still not fully understood. A major difficulty in advancing knowledge in this field is the rarity of manipulative experiments owing to the often protected status of wetlands. A few manipulative studies have been possible alongside recovery and restoration programmes within saltmarshes (Levin and Talley, 2002; Gratton and Denno, 2005; Pagliosa and Lana,

2005; Whitcraft and Levin, 2007) and mangrove habitats (Botero and Salzwedel, 1999; Sherman *et al.*, 2000; Macintosh *et al.*, 2002; Gladstone and Schreider, 2003). However, studies often are limited by constrained experimental designs, low replication, and a lack of suitable controls.

Research on tropical and subtropical mangrove-dominated estuaries have resulted in a wealth of information regarding the diversity and ecological value of these habitats (Lindgarth and Hoskin, 2001; Valiela *et al.*, 2001; Ashton and Macintosh, 2002; Bouillon *et al.*, 2002; Macintosh *et al.*, 2002; Bosire *et al.*, 2004). Therefore, mangroves in subtropical and tropical regions are regarded as biologically important areas that provide food and shelter for a diversity of organisms and result in rich ecosystems (Laegdsgaard and Johnston, 2001; Valiela *et al.*, 2001; Diop, 2003; Duke *et al.*, 2007). Nevertheless, some areas around the world have been cleared of mangroves to give way to aquaculture farms and coastal development or to provide firewood and other mangrove-derived products for local residents (Stonich, 1992; Valiela *et al.*, 2001; Diop, 2003). It is estimated that one-third of the world's mangrove forests have been lost in the past 50 years

(Alongi, 2002). In places, the destruction and degradation of mangrove habitats has prompted conservation and rehabilitation efforts. Mangrove-management projects have been undertaken and documented in many parts of the world (Spurgeon, 1998; Tri *et al.*, 1998; Botero and Salzwedel, 1999; Franks and Falconer, 1999; Barbier, 2000). These projects often involve extensive quantification of benthic fauna, mangrove vegetation, and physical parameters of the water and sediment (Kelahe *et al.*, 1998; Ashton *et al.*, 2003). Some of these studies also included comparisons with adjacent estuarine habitats (Kitheka, 1997; Davis *et al.*, 2001; Dittmar *et al.*, 2001; Lindegarth and Hoskin, 2001) and seasonal contrasts (Gordon *et al.*, 1995; Crowe, 1999). In general, tropical rehabilitated areas have shown an increase in species abundance and biodiversity associated with mangrove plantations (Botero and Salzwedel, 1999; Sherman *et al.*, 2000; Macintosh *et al.*, 2002). For example, Macintosh *et al.* (2002) recorded generally greater macrofaunal diversity, abundance, and biomass in conserved and replanted mangrove sites compared with sites cleared of mangroves in the Ranong Province, Thailand.

New Zealand's temperate mangrove ecosystems may differ significantly from their tropical counterparts, in that their biodiversity and ecological value may not be as high as previously thought (Alfaro, 2006). In the 1970s, a strong movement to preserve mangroves developed on the assumption that New Zealand's and subtropical and tropical mangrove forests had comparably high ecological value because they shared similar physical and biological properties (Chapman, 1976). To date, there is insufficient scientific information to clearly evaluate the ecological importance of New Zealand's mangroves. Nevertheless, a conservation movement still exists that advocates protection of mangrove habitats. On the other hand, many community groups and environmental managers have raised concerns regarding the unrestricted spread of mangroves over other habitats (e.g. seagrass beds, sandflats), which they argue has detrimental ecological and socio-economic effects (Green *et al.*, 2003; Schwarz, 2003).

Based on aerial photographs and maps, rapid mangrove spread has been documented in many estuaries throughout northern New Zealand, such as Whangape Harbour, Puhoi Estuary, Kaipara, Manukau, Waitemata, and Tauranga Harbours (references in Morrisey *et al.*, 2007). Other sites have experienced little or no increase in mangrove area, mostly because of the small size of the estuary or elevation limits on their landward sides (Morrisey *et al.*, 2007). Nationwide, a total net gain of ~3200 ha of mangroves has been estimated for the 20-year period between 1984 and 1996/1997 (Morrisey *et al.*, 2007). Mangroves also are seen as an unwanted species in Hawaii, where introduction of *Rhizophora mangle* in the early 1900s initiated a dramatic change in the native habitat of endangered Hawaiian waterbirds (Allen, 1998; Cox and Allen, 1999; Rauzon and Drigot, 2002). Subsequent mangrove-removal activities within the wetlands of Nu'upia Ponds, Mokapu Peninsula, Hawaii, resulted in successful restoration and recolonization by native birds (Rauzon and Drigot, 2002). However, it is important to note that contrary to mangroves in Hawaii, *Avicennia marina australasica* is endemic to New Zealand and may require a different management system.

In New Zealand, studies of mangrove habitats have focused largely on the effects of sedimentation, or infilling of estuaries owing to mangrove spread (Woodroffe, 1982, 1985; Young and Harvey, 1996), and on the production of mangrove leaf litter as a source of nutrients to the ecosystem (Woodroffe, 1982, 1985; May, 1999). More recently, investigations on the role of associated

flora and fauna within these habitats have increased in number (Morrisey *et al.*, 2002; Alfaro, 2006; Alfaro *et al.*, 2006). Morrisey *et al.* (2002) conducted a comparative study of benthic fauna within young (3–12-year old) and old (>60 years) mangrove stands in Manukau Harbour, New Zealand. Their findings suggest that, as mangrove stands mature, the abundance and biodiversity of the associated benthic fauna tend to decrease. This decrease in benthic biota was proposed to be a response to increased compaction of sediments around older mangroves (Morrisey *et al.*, 2002). Although studies of benthic associations in different types of mangrove stands may aid management decisions, the ecological importance of mangrove habitats, relative to adjacent estuarine habitats, also needs to be addressed for New Zealand mangroves. Alfaro (2006) investigated the community composition of benthic fauna within mangrove, seagrass, sandflat, and channel habitats at Matapouri Estuary, northern New Zealand. Mangrove habitats had significantly less benthic abundance and biodiversity than adjacent seagrass beds and subtidal channels, but pneumatophore areas at the fringe of the mangrove stands contained a high diversity of organisms (Alfaro, 2006). At that study site, the dominant benthic fauna within the ecosystem derive nutrients from a variety of sources, such as bacteria and brown algae, but mangrove-derived nutrients have only a localized effect on the foodweb, with little export of organic matter to adjacent habitats, such as sandflats (Alfaro *et al.*, 2006).

Within Mangawhai Harbour, northern New Zealand, an environmental permit (CON20031099401) was granted in August 2003 by the Northland Regional Council to remove a 0.26-ha fringe of mangrove trees for water access. This permit provided the unusual opportunity to evaluate the ecological effects of mangrove removal on the estuarine ecosystem. The aims of this study were to quantify the effect of mangrove-removal activities on benthic abiotic and biotic components. Disturbed and undisturbed mangrove areas, adjacent pneumatophore zones, marshgrass, sandflats, and subtidal channels were monitored over time to test the hypotheses that (i) mangrove removal would alter sediment characteristics, (ii) mangrove removal would change the benthic faunal composition, and (iii) variations in sediment characteristics and faunal composition would differ among habitats (i.e. mangrove areas, pneumatophore zones, marshgrass, sandflats, and subtidal channels) and over time. To quantify the long-term effects of this anthropogenic activity, ecological data were collected before mangrove removal and subsequently at 6-month intervals over a period of 3 years.

Material and methods

The study site (36°7'0"S 174°34'0"E) is located in central Mangawhai Harbour, northern New Zealand, at the northeastern corner of the Molesworth Causeway, ~2 km from Mangawhai Heads and ~4 km from Mangawhai Village (Figure 1). This part of the estuary consists of a wide channel (~10 m near the bridge), with mangrove trees (*A. marina australasica*) and sand/mudflats on both sides of the channel. The 0.26-ha area of mangrove removal is located on the east side of the causeway, just north of the bridge (Figure 1).

The ecological study was conducted between March 2004 and September 2006, with sampling once before and five times after the April 2004 mangrove tree-removal activities. Characterization of vegetation, macrofauna (epifauna and infauna), and sediment was undertaken within the mangrove-removal area and adjacent habitats (treatment site; Figure 1, Site 1). In addition, two control

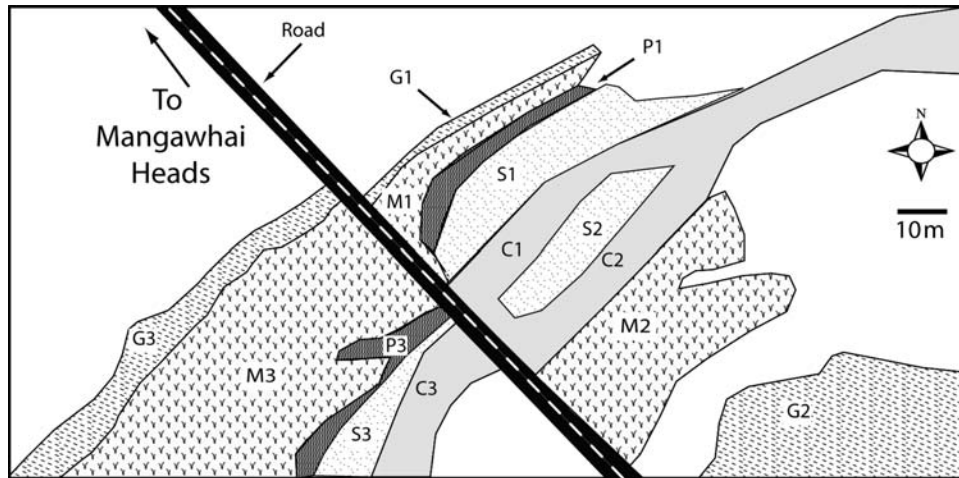


Figure 1. Map of Mangawhai Estuary (Middle Harbour). Sampling sites include a mangrove-removal site (G1, M1, P1, S1, and C1), a control site across the channel (G2, M2, S2, and C2), and another control site on the west side of the causeway (G3, M3, P3, S3, and C3). Habitats within sites include marshgrass (G1, G2, and G3), mangroves (M1, M2, and M3), pneumatophores (P1 and P3), sandflats (S1, S2, and S3), and channel (C1, C2, and C3). G, marshgrass; M, mangrove; P, pneumatophore; S, sandflat; and C, channel. Numbers 1, 2, and 3 refer to mangrove-removal site (1) and two control sites (2 and 3). Note that there was no pneumatophore habitat in Site 2.

sites were identified and sampled. The first control site is located just south of the treatment site, across the channel (Figure 1, Site 2), and the second west of the treatment site, on the other side of the causeway (Figure 1, Site 3). It is recognized that Sites 2 and 3 are not true control sites (not randomly chosen relative to the treatment site), but are similar enough to the treatment site to provide comparative undisturbed sites. Although the potential exists for exchange effects of sediment and organisms among sites, this is likely to be minimal owing to the strong physical barriers (tidally dominated channel and causeway) between sites. Within each sampling site, several unique habitats were identified between the edge of the catchment and the subtidal channel. The habitats, or zones, were identified according to their unique vegetation, inundation time, and general faunal characteristics and previously reported ecological differences (cf. Alfaro, 2006). Slight variations exist among equivalent habitats (e.g. different pneumatophore lengths among sites), but the results from earlier studies suggest comparability of habitats (cf. Alfaro, 2006). The habitats within each site included marshgrass, mangrove, pneumatophore (aerial roots), sand/mudflat, and channel, except for Site 2 which did not have a pneumatophore habitat.

The marsh habitat is dominated by jointed rush, *Leptocarpus similis*, with other small terrestrial grasses and plants among the rushes. Pneumatophores from nearby mangroves also extend into the marshgrass habitat. The mangrove habitat is densely covered with mature mangrove trees (*A. marina australasica*) 4–5 m high. The top 5 cm of sediment is characterized as a silty mud layer, and there is a dense mat of fibrous mangrove roots (>20 cm deep) below this layer. This habitat was removed from Site 1 in April 2004. The pneumatophore habitat is a zone of dense mangrove pneumatophores next to the fringing mangrove trees at all three sites. However, the pneumatophore zone at Site 2 was too narrow (<1 m) to sample. Sandflat habitats were found within all three sites, although their sediment composition varied slightly. Subtidal channel habitats were dominant features of the study location. The main channel constricts as it approaches the bridge to the east (Site 3), then divides into two branches ~50 m to the east, just past the bridge. All samples within these

habitats were taken during low tide, at a water depth of ~20–50 cm.

Sampling of the vegetation

A survey of the vegetation was conducted within all mangrove habitats for the three sites. The survey included quantification of trees (mangroves taller than 0.5 m and >2.5 cm in stem diameter) and saplings (mangroves taller than 0.5 m, but <2.5 cm in stem diameter) within five quadrats of 25 m². The mean height and diameter at breast height (dbh) of ten random mangrove trees, anywhere in the habitat, also were recorded. The density of mangrove seedlings (mangroves shorter than 0.5 m), propagules, and pneumatophores in mangrove and pneumatophore habitats was determined using five replicate quadrats of area 0.25 m². The percentage cover of mangrove leaf litter and marshgrass inside each of five quadrats of 0.25 m² within mangrove, pneumatophore, and marshgrass habitats also were recorded. All measurements were standardized to an area of 1 m² to compare among parameters.

Sampling of the macrofauna

Macrofauna (infauna and epifauna) densities were determined within each habitat and site. Five randomly placed sediment cores (25 cm × 25 cm, 5 cm depth) were used to collect macrofauna within each habitat during low tides. Each replicate sediment sample was sieved through a 2-cm and then a 0.5-mm mesh. The 0.5-mm mesh sieve was used to determine the density and distribution of juvenile species (e.g. bivalves) and worms. Preliminary samples at the study sites indicated that most infauna (~98% of all individuals) were within the top 5 cm of the sediment within mangrove and marshgrass habitats. Hence, a sampling depth of 5 cm provided an accurate representation of the density of infauna in these habitats and minimized the arduous task of sieving through dense root systems. This 5 cm depth was used for all cores within all habitats to standardize the samples. All live animals found in all samples were enumerated, recorded, and standardized to a volume of 1 m³. Samples

were kept refrigerated until sieving could be completed, within 3 d of each sampling event.

Sampling of the sediment

Three replicate sediment samples (15 cm × 15 cm, 5 cm depth) were taken from each habitat at each site, a total of 45 samples per sampling event. All sediment samples were first refrigerated, then analysed within 3 d of collection. They were used to determine grain-size and organic content. Grain-size characterization was done by sieving each dried sample through several sieves of different mesh size (mud ≤ 150 μm, fine sand = 150–300 μm, medium sand = 300–600 μm, coarse sand ≥ 600 μm), and calculating the proportions. Total organic content was obtained through weight differences before and after total combustion at 450°C after acidification to a pH of 2 (Parrish, 1998). Sediment compaction was determined by measuring the depth of penetration of a sharpened steel rod (50 cm long, 0.7 cm diameter, 150 g weight) when dropped from a distance of 1.2 m above the sediment surface (Morrisey *et al.*, 2002). Compaction relates to the porosity and permeability of the sediment. Variations in these properties may create microhabitats with different soil chemistry and biological activity.

Statistical analyses

Multiple factor ANOVAs were used to test vegetation differences among sites, habitats, and dates, and individual three-way ANOVAs with Tukey tests were performed to test benthic faunal differences among sites, habitats, and dates. Multiple dimension scaling (MDS) plots were constructed, based on similarities of correlation matrices, to detect groupings of habitat type for each sampling date. Principal component analysis (PCA) was used to detect habitat differences, based on sediment characteristics for each sampling date. Data that did not meet the requirements for parametric analyses were transformed with an arcsine or square root $x + 0.5$ transformation to meet these requirements. Statistica 6.0 and Primer 6 software were used for statistical analyses.

Results

The three mangrove habitats had similar vegetation parameters (i.e. mean densities of mangrove trees, saplings, seedlings, propagules, pneumatophores, and tree height and dbh) before mangrove

removal (two-way ANOVA; habitat, $p > 0.001$; vegetation, $p < 0.001$), and these parameters continued to be consistent through the sampling period in the remaining two mangrove stands (three-way ANOVA; habitat, $p > 0.001$; vegetation, $p < 0.001$; date, $p > 0.001$; Table 1). However, sapling, seedling, and pneumatophore densities varied considerably between March 2004 and September 2006 in the mangrove habitat (M1) after tree removal (two-way ANOVA; vegetation, $p < 0.001$; date, $p > 0.001$; Figure 2). Sapling and seedling densities increased throughout the sampling period, although saplings were only present in March 2004 and in March and September 2006. Only a few propagules were present in this habitat in March 2005 and 2006. Pneumatophore density increased sharply from March to September 2004, then decreased steadily until the end of the sampling period (Figure 2). Marshgrass habitats were similar in vegetation composition among sites, except for Site 1, which was slightly more elevated than the other two sites and had a greater density of marshgrass reed cover throughout the sampling period (ANOVA; $p < 0.001$; Tukey test, Site 1 differs from Sites 2 and 3; Table 1). The pneumatophore habitats had different densities of pneumatophores; Site 3 had higher mean densities than Site 1 throughout the sampling period (Table 1). As expected, the mean percentage of mangrove leaf litter inside random quadrats was higher in the mangrove habitats (Table 1).

Overall, mangrove habitats had less total abundance and fewer total numbers of taxa than any of the other habitats sampled within all sites (Figure 3; Appendix). Individual three-way ANOVAs and Tukey tests, with date, site, and habitat (excluding pneumatophore habitats) as fixed factors, were performed for total numbers of individuals and for total number of taxa (Table 2). Significant differences were observed among date, site, and habitat factors, signalling the high spatial and temporal variability of the area. Significant interactions among main factors suggest that different ecological processes (e.g. differential seasonal effects, variations in habitat preferences) are responsible for the faunal distribution and abundance at the study site. Hence, detailed variations among dates, sites, and habitats were necessary (Tukey tests, in Table 2). These analyses specifically highlight the significant temporal variability for some taxa. The MDS analyses revealed clear separation between the sandflat/channel habitats and the rest of the habitats for all sites and sampling dates (Figure 4). Generally, the pneumatophore habitats clustered

Table 1. Mean (\pm s.e.) values for plant parameters recorded from marshgrass (G), mangrove (M), and pneumatophore (P) habitats within three estuarine sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) in Mangawhai Estuary, New Zealand, for the sampling period March 2004–September 2006.

| Plant characteristic | Site 1 | | | Site 2 | | Site 3 | | |
|--|-------------|------------------------|-------------|--------------|-------------|--------------|-------------|--------------|
| | G | M | P | G | M | G | M | P |
| Marshgrass (% cover m ⁻²) | 348.0 ± 9.6 | – | – | 164.4 ± 18.4 | – | 222.0 ± 21.2 | – | – |
| Mangrove trees (number m ⁻²) | – | 0.5 ± 0.0 ^a | – | – | 0.6 ± 0.0 | – | 0.6 ± 0.0 | – |
| Mangrove tree height (m) | – | 4.2 ± 0.4 ^a | – | – | 4.6 ± 0.8 | – | 4.9 ± 0.9 | – |
| Mangrove tree dbh (cm) | – | 5.7 ± 0.4 ^a | – | – | 5.8 ± 0.7 | – | 5.0 ± 0.6 | – |
| Saplings (number m ⁻²) | – | 0.0 ± 0.0 | – | – | 0.0 ± 0.0 | – | 0.1 ± 0.0 | – |
| Seedlings (number m ⁻²) | 0.0 ± 0.0 | 31.6 ± 5.6 | 2.8 ± 2.0 | 0.4 ± 0.0 | 0.0 ± 0.0 | 0.0 ± 0.0 | 1.2 ± 1.2 | 14.0 ± 4.0 |
| Propagules (number m ⁻²) | 0.0 ± 0.0 | 0.4 ± 0.4 | 0.8 ± 0.4 | 0.0 ± 0.0 | 0.0 ± 0.0 | 0.0 ± 0.0 | 0.8 ± 0.8 | 1.2 ± 0.4 |
| Pneumatophores (number m ⁻²) | 0.4 ± 0.4 | 102.0 ± 8.8 | 75.2 ± 10.8 | 5.2 ± 2.8 | 68.4 ± 10.4 | 39.6 ± 16.4 | 66.0 ± 12.0 | 152.4 ± 28.0 |
| Leaf litter (% cover m ⁻²) | 0.0 ± 0.0 | 4.8 ± 1.6 | 0.0 ± 0.0 | 1.2 ± 0.4 | 22.0 ± 12.4 | 0.4 ± 0.0 | 24.0 ± 12.4 | 0.4 ± 0.0 |

^aSampling during March 2004 only (before mangrove removal).

Note that there was no pneumatophore habitat at Site 2.

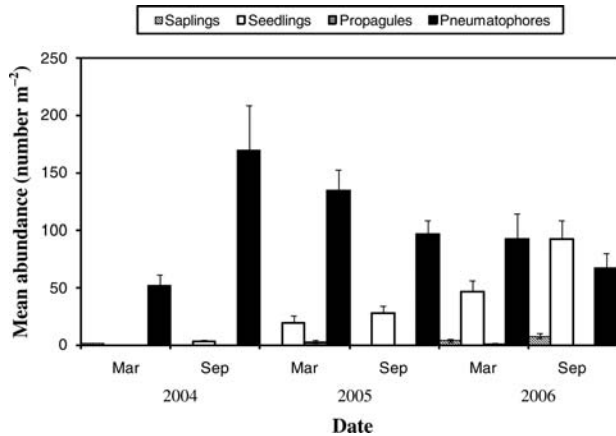


Figure 2. Mean abundance (\pm s.e.) of saplings, seedlings, propagules, and pneumatophores in the mangrove habitat (M1; mangrove-removal area) from March 2004 to September 2006.

between the sandflat/channel habitats and the marshgrass/mangrove habitats. All mangrove habitats clustered before mangrove removal (March 2004), but M1 separated from M2 and M3 on all other sampling dates, except for March 2005 (Figure 4). The marshgrass habitat in Site 1 separated from those in Sites 2 and 3 during all sampling dates. The range of stress values for all MDS plots was low (0.02–0.05), strongly supporting the groupings.

The abundance of the dominant macrofauna in each habitat was plotted for each site between March 2004 and September 2006 to identify spatial and temporal variations and to elucidate ecological differences between mangrove-removal and control sites (Figures 5–8). The most abundant organism, *Potamopyrgus antipodarum*, had an overall mean density (\pm s.e.) of $1.33 \pm 0.01 \text{ m}^{-3}$, but was present only in the marshgrass and mangrove habitats (Figure 6). Polychaete and oligochaete worms were the next most abundant organisms, with overall mean densities (\pm s.e.) of 0.04 ± 0.00 and $0.23 \pm 0.01 \text{ m}^{-3}$, respectively (Figure 8). The snail *Zeacumantus lutulentus* was found in all habitats, with an overall density of $0.38 \pm 0.00 \text{ m}^{-3}$, but was most common in the pneumatophore, sandflat, and channel habitats (Figure 5). The mud crab *Helice crassa* was generally present in all habitats, except for the channel, whereas the mud snail *Amphibola crenata* was found mostly in the marshgrass and mangrove habitats (Figures 5 and 6). The overall mean densities (\pm s.e.) for *H. crassa* and *A. crenata* were 0.05 ± 0.00 and $0.30 \pm 0.02 \text{ m}^{-3}$, respectively. The cockle *Austrovenus stutchburyi* was common within mangrove, pneumatophore, sandflat, and channel habitats, and the pipi *Paphies australis* was found in sandflat and channel habitats only (Figure 7). The overall mean densities (\pm s.e.) for *A. stutchburyi* and *P. australis* were 0.11 ± 0.01 and $0.14 \pm 0.01 \text{ m}^{-3}$, respectively. Individual three-way ANOVAs and Tukey tests, with date, site, and habitat (excluding pneumatophore habitats) as fixed factors, were performed for all major macrofauna (Table 2). Significant effects and interactions were observed for most fauna, but *post hoc* tests revealed a range of significant date pairs, indicating high temporal variability (Table 2). In general, mangrove removal appeared to result in an immediate increase and subsequent decrease in crab, snail, and bivalve abundance in surrounding areas, with polychaetes following the inverse pattern.

There were consistent differences in sediment characterization among habitats. The mean depth of the penetrometer was consistent among sites and over time, although there were distinct differences among habitats. The mean penetration depth (\pm s.e.) was lower in the marshgrass ($5.6 \pm 1.3 \text{ mm}$), sandflat ($14.6 \pm 1.0 \text{ mm}$), and channel ($4.7 \pm 0.6 \text{ mm}$) habitats than in the mangrove ($15.6 \pm 0.9 \text{ mm}$) and pneumatophore ($15.9 \pm 1.1 \text{ mm}$) habitats (ANOVA; $p > 0.001$). Grain-size analyses of sediments within all habitats and sites revealed similar characteristics across sites, fine and medium sandy sediments being the most abundant sediments in most habitats (Figure 9). Comparisons among habitats generally revealed marsh habitats to display a greater proportion of coarse sand sediments, mangrove and pneumatophore habitats to exhibit more muddy sediments, and sandflat and channel habitats to have greater fine and medium sediments (Figure 9). The temporal patterns in grain-size composition indicate that Site 1 experienced a general decrease in fine sand from March to September 2004, especially for the mangrove, pneumatophore, and sand habitats (but not the channel in Site 1), and an increase in finer sediments from September 2005 to September 2006 (Figure 9). For Sites 2 and 3, an increase in both fine and coarser sediments was observed during the later part of the study period.

The PCA ordination of sediment characteristics revealed clear habitat clustering for mangrove, marshgrass, and pneumatophore habitats at the beginning of the study (Figure 10; March 2004). Therefore, sediment conditions within these habitats were similar among sites at the start of the study. A distinct separation of mangrove habitat at Site 1 (mangrove-removal site) was found at and after September 2004, which agrees with the increase in sand composition there immediately after mangrove removal (Figure 9). Marshgrass habitats generally clustered throughout the study period, whereas greater separation and mobility was observed in the rest of the habitats, especially in the sandflats and channel (Figure 9). The variable loadings for principal components 1–3 accounted for 100% of the variation for all sampling dates. In general, component 1 had a high (positive or negative) loading for fine and coarse sand, whereas component 3 was mostly associated with mud. Component 2 was highly variable in its loading composition. Component 1 explained ~ 40 – 60% of the variation, component 2 ~ 20 – 40% , and component 3 almost 20% of the total variation.

There was generally greater organic content for mangrove and pneumatophore habitats than for marshgrass, sandflat, and channel habitats (Figure 11). Distinctive spatial and temporal differences were observed across habitats and sites. At Site 1, the organic content in all habitats, but especially in the mangroves and pneumatophores, increased slightly immediately after mangrove removal (Figure 11). This trend was followed by a general decrease in organic content in sediment samples, and a sharp increase between March 2005 and September 2006. At Sites 2 and 3, the temporal changes were less pronounced. A three-way ANOVA resulted in significant date and habitat factors ($p < 0.001$) and non-significant site factor and all interactions ($p > 0.001$).

Discussion

Mangrove ecosystems

Although many ecological studies of subtropical and tropical mangroves have revealed high species diversity and abundances

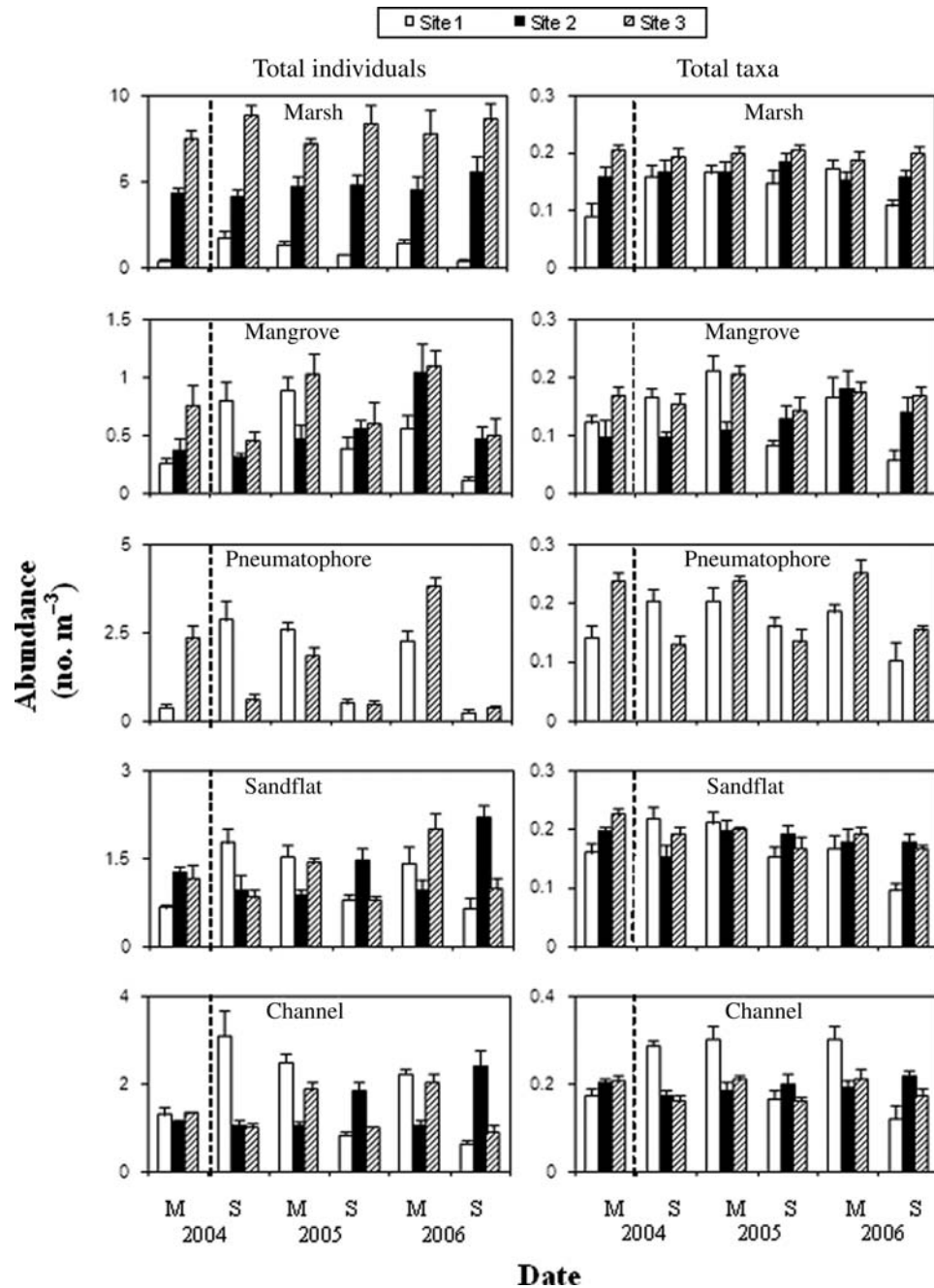


Figure 3. Mean abundance (\pm s.e.) of total individuals and total taxa within different habitats at three sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) from March 2004 to September 2006 (mangrove removal illustrated by vertical dashed line). Note that there was no pneumatophore habitat at Site 2 and that the y-axes have different scales.

(Dittmann, 2000; Ashton and Macintosh, 2002; Macintosh *et al.*, 2002), the mangrove habitats found in temperate New Zealand may differ in various aspects, including sedimentation rate and water characteristics (i.e. temperature, salinity, inundation time, rate of nutrient recycling), arboreal architecture, and species composition of associated flora and fauna. Although mangrove densities in many tropical areas are declining (Laegdsgaard and Johnston, 2001; Valiela *et al.*, 2001; Diop, 2003), New Zealand mangroves are spreading dramatically in some locations, mostly a consequence of accelerating sedimentation rates from anthropogenic catchment modifications (Hume and Dahm, 1992; Ellis

et al., 2004). In addition, cooler temperatures and shorter periods of tidal inundation within New Zealand coastal areas appear to decrease the decomposition rate, which dampens the reincorporation of organic matter into the foodweb compared with tropical and subtropical mangrove ecosystems (Alfaro, 2006). A lack of intertidal plant competitors also may facilitate mangrove spread in New Zealand. In Hawaii, where mangroves (*R. mangle*) also are spreading rapidly, a high propagule production rate (Cox and Allen, 1999) and a lack of propagule predators (Steel *et al.*, 1999) have been suggested as two potential reasons for the success of mangroves there (Chimner *et al.*,

Table 2. Statistical analyses (three-way ANOVAs) for total number of individuals and taxa, and major macrofauna for six dates between March 2004 and September 2006 (M4, S4, M5, S5, M6, and S6), three sites (Site 1, mangrove-removal area; Sites 2 and 3, controls), and four habitats (G, marshgrass; M, mangrove; S, sandflat; C, channel).

| Source | d.f. | Mean square | F-value | p-value | d.f. | Mean square | F-value | p-value |
|--|------|-------------|---------|--------------|--|-------------|----------|--------------|
| Total individuals ^a | | | | | Total taxa ^b | | | |
| Date (D) | 5 | 1 818.48 | 3.61 | 0.003 | 5 | 19.02 | 14.42 | 0.001 |
| Site (S) | 2 | 86 011.32 | 170.95 | 0.001 | 2 | 13.04 | 9.88 | 0.001 |
| Habitat (H) | 3 | 275 953.01 | 548.45 | 0.001 | 3 | 55.25 | 41.87 | 0.001 |
| D × S | 10 | 4 229.31 | 8.40 | 0.001 | 10 | 18.81 | 14.25 | 0.001 |
| D × H | 15 | 584.10 | 1.16 | 0.303 | 15 | 3.09 | 2.34 | 0.003 |
| S × H | 6 | 91 116.54 | 181.09 | 0.001 | 6 | 13.67 | 10.36 | 0.001 |
| D × S × H | 30 | 868.21 | 1.73 | 0.013 | 30 | 2.19 | 1.66 | 0.019 |
| Error | 288 | 503.32 | | | 288 | 1.32 | | |
| <i>Helice crassa</i> ^c | | | | | <i>Zeacumantus lutulentus</i> ^d | | | |
| Date (D) | 5 | 3.13 | 1.55 | 0.173 | 5 | 10.05 | 4.98 | 0.001 |
| Site (S) | 2 | 25.43 | 12.62 | 0.001 | 2 | 72.31 | 35.83 | 0.001 |
| Habitat (H) | 3 | 46.06 | 22.85 | 0.001 | 3 | 123.61 | 61.25 | 0.001 |
| D × S | 10 | 1.84 | 0.91 | 0.523 | 10 | 2.72 | 1.35 | 0.203 |
| D × H | 15 | 1.09 | 0.54 | 0.915 | 15 | 5.07 | 2.51 | 0.002 |
| S × H | 6 | 20.14 | 9.99 | 0.001 | 6 | 34.16 | 16.93 | 0.001 |
| D × S × H | 30 | 1.04 | 0.52 | 0.984 | 30 | 3.35 | 1.66 | 0.019 |
| Error | 288 | 2.01 | | | 288 | 2.01 | | |
| <i>Potamopyrgus antipodarum</i> ^e | | | | | <i>Amphibola crenata</i> ^f | | | |
| Date (D) | 5 | 84.00 | 0.26 | 0.936 | 5 | 59.24 | 3.32 | 0.006 |
| Site (S) | 2 | 45 025.11 | 137.78 | 0.001 | 2 | 543.34 | 30.42 | 0.001 |
| Habitat (H) | 3 | 165 631.60 | 506.86 | 0.001 | 3 | 3 900.91 | 218.40 | 0.001 |
| D × S | 10 | 199.22 | 0.61 | 0.805 | 10 | 24.83 | 1.39 | 0.184 |
| D × H | 15 | 91.90 | 0.28 | 1.000 | 15 | 33.00 | 1.85 | 0.028 |
| S × H | 6 | 43 639.21 | 133.54 | 0.001 | 6 | 382.34 | 21.41 | 0.001 |
| D × S × H | 30 | 202.17 | 0.62 | 0.943 | 30 | 20.13 | 1.13 | 0.301 |
| Error | 288 | 326.80 | | | 288 | 17.86 | | |
| <i>Austrovenus stutchburyi</i> ^g | | | | | <i>Paphies australis</i> ^h | | | |
| Date (D) | 5 | 54.76 | 8.71 | 0.001 | 5 | 20.363 | 2.7070 | 0.021 |
| Site (S) | 2 | 8.53 | 1.36 | 0.259 | 2 | 154.269 | 20.5085 | 0.001 |
| Habitat (H) | 3 | 959.75 | 152.61 | 0.001 | 3 | 606.669 | 80.6503 | 0.001 |
| D × S | 10 | 10.19 | 1.62 | 0.100 | 10 | 11.559 | 1.5367 | 0.126 |
| D × H | 15 | 18.35 | 2.92 | 0.001 | 15 | 7.652 | 1.0172 | 0.437 |
| S × H | 6 | 16.57 | 2.63 | 0.017 | 6 | 51.514 | 6.8482 | 0.001 |
| D × S × H | 30 | 14.96 | 2.38 | 0.001 | 30 | 4.493 | 0.5973 | 0.955 |
| Error | 288 | 6.29 | | | 288 | 7.522 | | |
| Polychaetes ⁱ | | | | | Oligochaetes ^j | | | |
| Date (D) | 5 | 9.68 | 2.66 | 0.023 | 5 | 77.35 | 3.5029 | 0.004 |
| Site (S) | 2 | 7.00 | 1.92 | 0.148 | 2 | 2.63 | 0.1193 | 0.888 |
| Habitat (H) | 3 | 94.88 | 26.07 | 0.001 | 3 | 2 630.25 | 119.1209 | 0.001 |
| D × S | 10 | 4.54 | 1.25 | 0.260 | 10 | 11.31 | 0.5124 | 0.881 |
| D × H | 15 | 3.50 | 0.96 | 0.495 | 15 | 55.15 | 2.4978 | 0.002 |
| S × H | 6 | 5.90 | 1.62 | 0.141 | 6 | 97.55 | 4.4180 | 0.001 |
| D × S × H | 30 | 1.64 | 0.45 | 0.994 | 30 | 10.17 | 0.4606 | 0.993 |
| Error | 288 | 3.64 | | | 288 | 22.08 | | |

Tukey tests for appropriate comparisons are shown. Significant p-values are emboldened.

^aNon-significant Tukey tests: M4 × S4, M4 × S6, S4 × M5, S4 × S5, S4 × M6, S4 × S6, M5 × S5, M5 × M6, M5 × S6, S5 × M6, S5 × S6, M6 × S6.

^bNon-significant Tukey tests: M4 × S4, M4 × S5, M4 × S6, S4 × S5, S4 × M6, M5 × M6, S5 × S6.

^cNon-significant Tukey tests: All date pairs are non-significant; Sites: 1 × 2; habitats: G × S.

^dNon-significant Tukey tests: M4 × S4, M4 × M5, M4 × S5, M4 × M6, M4 × S6, S4 × M5, S4 × S5, S4 × M6, M5 × S5, M5 × M6, S5 × M6, M6 × S6; Sites: 1 × 2; habitats: G × M.

^eNon-significant Tukey tests: All date pairs are non-significant; habitats: M × S, M × C, S × C.

^fNon-significant Tukey tests: M4 × S4, M4 × M5, M4 × S5, M4 × M6, M4 × S6, S4 × M5, S4 × S5, S4 × M6, S4 × S6, M5 × S5, M5 × M6, S5 × M6, M6 × S6; Sites: 2 × 3; habitats: S × C.

^gNon-significant Tukey tests: M4 × M5, S4 × M5, S4 × S5, S4 × M6, S4 × M6, S4 × S6, M5 × S5, M5 × M6, S5 × M6, S5 × S6, M6 × S6; Sites: 1 × 2, 1 × 3, 2 × 3; habitats: G × M.

^hNon-significant Tukey tests: M4 × S4, M4 × M5, M4 × S5, M4 × M6, M4 × S6, S4 × M5, S4 × S5, S4 × M6, S4 × S6, M5 × S5, M5 × M6, M5 × S6, S5 × M6, M6 × S6; Sites: 1 × 3; habitats: G × M, S × C1.

ⁱNon-significant Tukey tests: M4 × S4, M4 × M5, M4 × S5, M4 × S6, S4 × M5, S4 × S5, S4 × M6, S4 × S6, M5 × S5, M5 × M6, M5 × S6, S5 × M6, S5 × S6, M6 × S6; Sites: 1 × 2, 1 × 3, 2 × 3; habitats: G × M, S × C.

^jNon-significant Tukey tests: M4 × S4, M4 × M5, M4 × M6, S4 × M5, S4 × S5, S4 × M6, S4 × S6, M5 × S5, M5 × M6, M5 × S6, S5 × M6, S5 × S6, M6 × S6; Sites: 1 × 2, 1 × 3, 2 × 3; habitats: G × M, S × C.

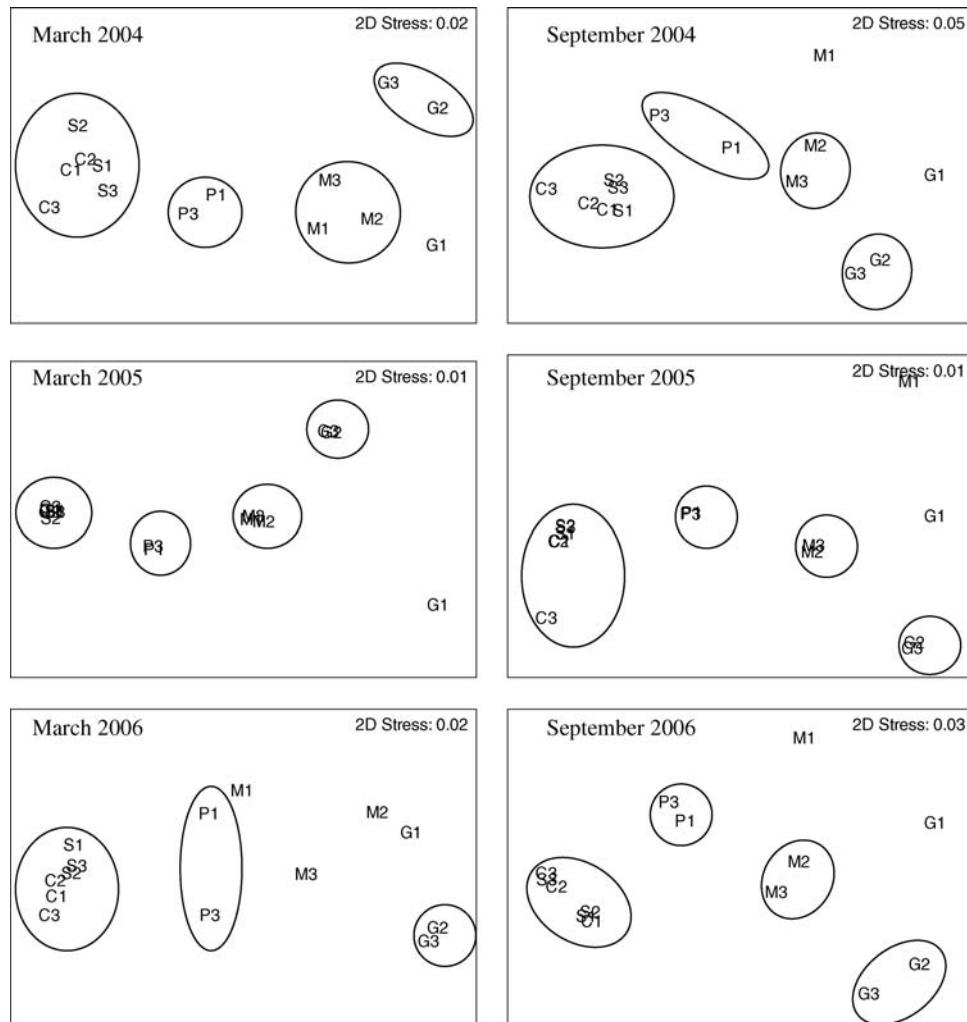


Figure 4. Non-metric MDS plots of habitats based on faunal assemblages at Mangawhai Harbour. Habitats include G, marshgrass; M, mangroves; P, pneumatophores; S, sandflats; and C, channel at three sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) from March 2004 to September 2006. Note that there was no pneumatophore habitat at Site 2.

2006). New Zealand's *A. marina* also produces a large number of resilient propagules, which appear to have few predators. The viviparous nature of these propagules allows for continuous development and growth throughout the dispersal stages, with no physiological control to inhibit or delay embryonic development (Fountain and Outred, 1991). In addition, few grazers on mangrove propagules or leaves have been reported in New Zealand. In fact, Alfaro (2006) suggested that the low abundance of mangrove grazers (i.e. crabs) in New Zealand, compared with tropical mangrove habitats, may result in a minimal transfer of nutrients from mangroves to the rest of the estuarine ecosystem, so locking high vegetative biomass within this specific habitat. Based on these ecological differences, and whatever the cause, New Zealand (and Hawaiian) mangrove habitats are being subjected to management and control of their spread, rather than undergoing the replanting and protection that is underway for some tropical mangrove areas.

Faunal distribution

The overall abundance patterns of benthic fauna among habitats and sites indicates that mature mangrove habitats have less total

abundance and fewer total numbers of taxa than any of the other habitats studied (although not during all sampling events). These differences may be related to mature trees tending to form dense mats of fibrous roots, which may not provide a suitable habitat for many macroinvertebrates. Indeed, Morrisey *et al.* (2002) found that although mature stands of mangroves in the Manakau Harbour had higher concentrations of organic matter, they had fewer associated taxa, and many species were less abundant, than in younger mangrove stands nearby. Oxidic respiration in mangrove habitats is increased by tidal influence, crab bioturbation, and physiological activities of the roots; nonetheless, mangrove respiration rates are lower than in other coastal areas, such as saltmarshes (Alongi *et al.*, 2000). This intrinsic characteristic of older mangrove stands may make it difficult for benthic fauna to inhabit these areas, as suggested by Alfaro (2006). Greater quantities of leaf litter within mangrove habitats also may cause a decrease in benthic fauna, because tannins from leaf litter have been identified as a cause of low species diversity in mangrove habitats in Australia (Alongi, 1987).

The species composition within each habitat in the Mangawhai study tended to be similar for all sites, and dominant species

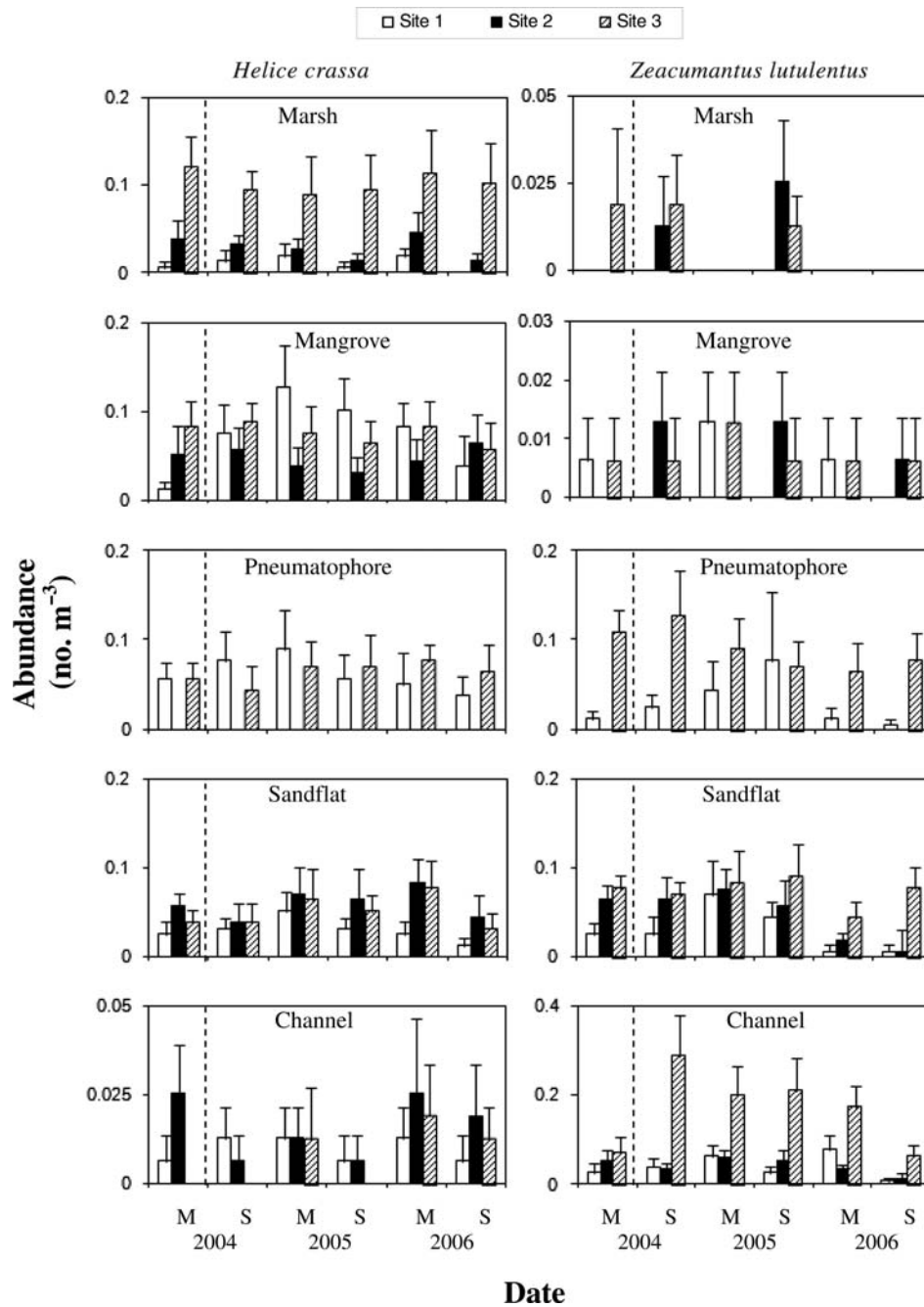


Figure 5. Mean abundance (\pm s.e.) of *H. crassa* and *Z. lutulentus* within different habitats where these species were found at three sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) from March 2004 to September 2006 (mangrove removal illustrated by vertical dashed line). Note that there was no pneumatophore habitat at Site 2 and that the y-axes have different scales.

generally had clearly defined ranges. Among all species encountered during the study period, the snail *P. antipodarum* was the most abundant, and it clearly dominated marsh habitats. However, its abundance was low or non-existent in the mangrove habitats. This trend is consistent with the benthic abundance and distribution studies of Alfaro (2006), and Alfaro *et al.* (2006) in Matapouri Estuary, northern New Zealand. However, Morrisey *et al.* (2002) found *P. antipodarum* in greater numbers in older mangrove stands in the Manakau Estuary, Auckland, New Zealand. At Mangawhai, the pulmonate snail *A. crenata* was found in greatest numbers along the transition between

marshgrass and mangrove habitats. That snail has been reported to live in the muddy substrata of mangrove habitats (May, 1999; Morrisey *et al.*, 2002; Ellis *et al.*, 2004). However, in the current study, both juveniles and adults of this species were found in greater numbers in the sandy substrata of the marshgrass habitats. At Mangawhai, the scavenger snail *Z. lutulentus* was found predominantly in the sandflat, channel, and pneumatophore habitats, although its distribution tends to be patchy and dependent on food sources. Other than polychaete and oligochaete worms, which were found throughout the study sites, the mud crab *H. crassa* had the widest distribution range. This small crab, of

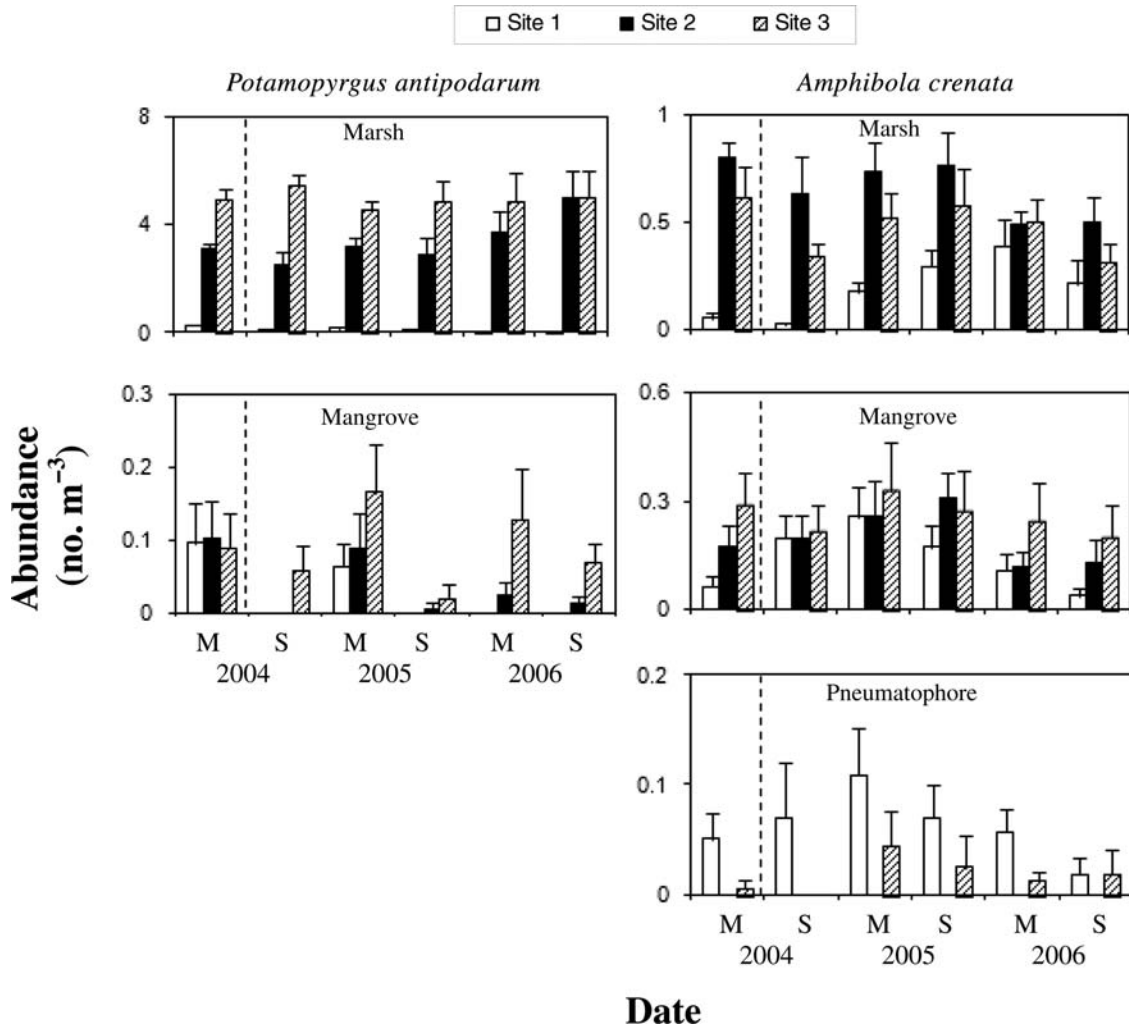


Figure 6. Mean abundance (\pm s.e.) of *P. antipodarum* and *A. crenata* within different habitats where these species were found at three sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) from March 2004 to September 2006 (mangrove removal illustrated by vertical dashed line). Note that there was no pneumatophore habitat at Site 2 and that the y-axes have different scales.

~ 10 – 20 cm in carapace width, was found in all habitats, but predominantly in mangroves. It has been reported to feed on mangrove leaves (May, 1999), but it is more likely that it is a scavenger on different types of organic matter, including other dead invertebrates (Alfaro *et al.*, 2006).

Two bivalves were dominant in this study. The cockle *A. stutchburyi* was abundant in sandflats and subtidal channels. Its tolerance to muddy sediments is reflected by its extended distribution, in lesser numbers, into the mangrove habitats. Although little is known about where it recruits in New Zealand estuaries (Stewart and Creese, 2002; Marsden, 2004), juveniles often were found in the sandflat and channel habitats. The pipi *P. australis* was recorded almost exclusively in channel and sandflat habitats, although its numbers were generally low.

Temporal changes

A significant increase in overall faunal composition was observed 6 months after the removal of mangrove trees at Site 1, especially within mangrove and pneumatophore habitats. This increase in population density was reflected by a higher mean number of individuals, number of taxa, and dominant species (i.e. crabs, snails,

bivalves) within that site. These biological changes were accompanied by changes in sediment composition, from high silt to coarser sediment immediately after removal, and an increase in total organic content in the sediment. Although all mangrove trees were removed from the site, the results from the September 2004 sampling event showed a greater number of pneumatophores within the mangrove and pneumatophore habitats, probably a consequence of the removal of the uppermost sediments by water motion, which could have uncovered shorter pneumatophores. Another possible explanation for this pattern is that pneumatophores continued to grow for a short time, although the trees were removed, because many pneumatophores have photosynthetic tissue.

Subsequent faunal and sediment samples from March 2005 to September 2006 indicate similar temporal changes among sites, although many were more pronounced at Site 1. The general biological pattern was of an increase in species abundance just after mangrove removal, followed by a sharp decrease in species abundance in March and September 2006. These biological changes appear to be related to an increase in silt and organic content, which was most noticeable at Site 1, but was apparent too in

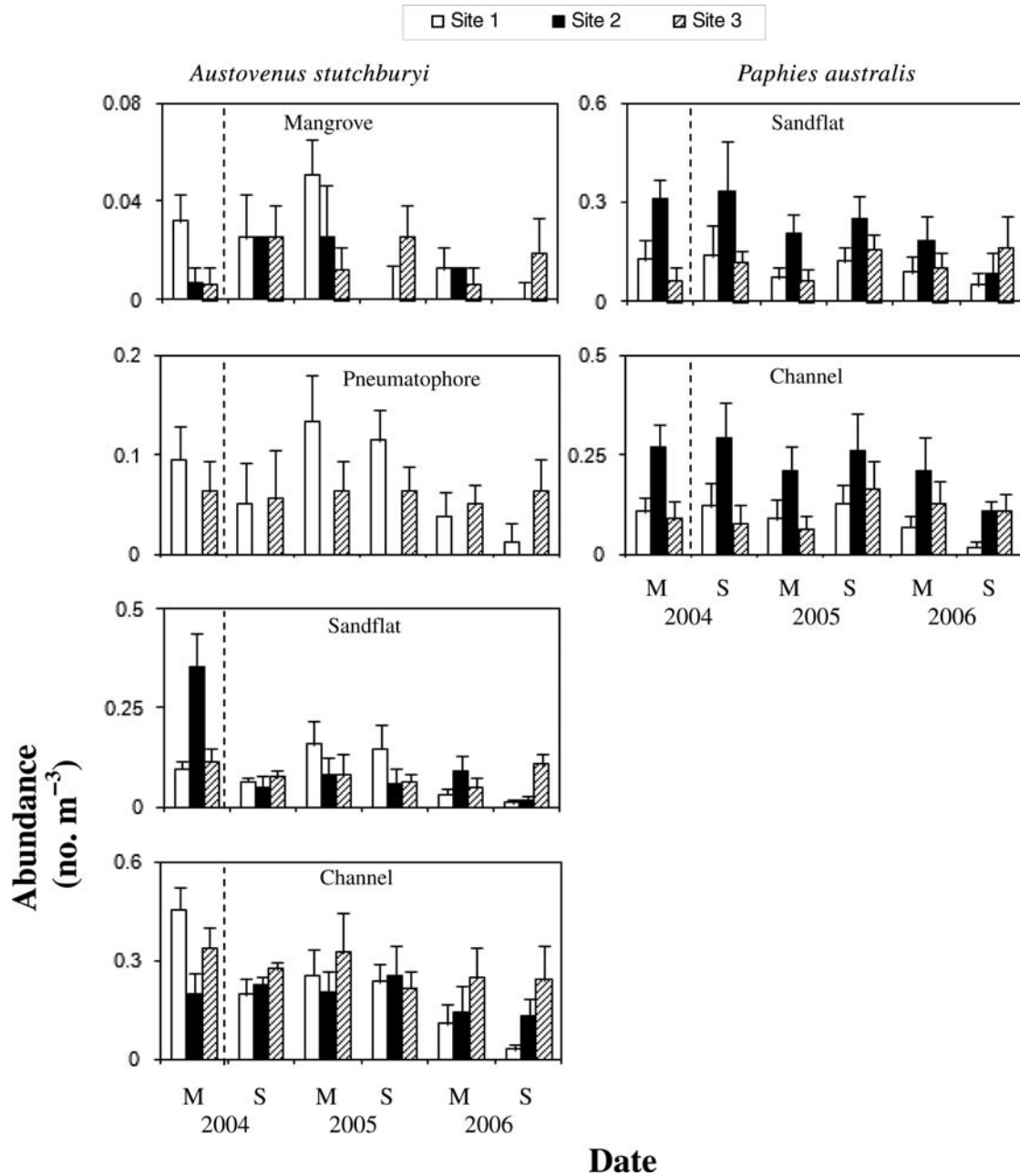


Figure 7. Mean abundance (\pm s.e.) of *A. stutchburyi* and *P. australis* within different habitats where these species were found at three sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) from March 2004 to September 2006 (mangrove removal illustrated by vertical dashed line). Note that there was no pneumatophore habitat at Site 2 and that the y-axes have different scales.

pneumatophore, sandflat, and channel habitats at Sites 2 and 3. It is likely that these physical changes were the result of intensive catchment activity just above Site 1, where a new building development is underway. Disturbance of buried peat within catchment soils, as a result of topographic reconfiguration by bulldozing, may have resulted in the increased organic content recorded in adjacent coastal sediments (J. Lockwood, pers. comm.). Although these changes in sediment structure may have been the major contributors to the faunal changes observed over time, reproductive cycles and seasonal migration patterns are likely to have been superimposed on the ecological trends. For example, at Matapouri Estuary in northern New Zealand, Alfaro (2006)

found distinctive abundance and biodiversity patterns within specific estuarine habitats, which often were related to recruitment and migration patterns.

Highly mobile species, such as the crab *H. crassa*, dramatically increased in abundance following mangrove removal at Site 1, then slowly decreased in subsequent sampling dates. The pattern suggests that crabs may have migrated to the site to take advantage of the increase in organic content following the disturbance. Although crabs may exhibit intraspecific competition for food, it does show gregarious behaviour within favourable muddy habitats (Morrisey *et al.*, 1999; Sivaguru, 2000). This aggregation behaviour also may be responsible for its high densities in the marshgrass and

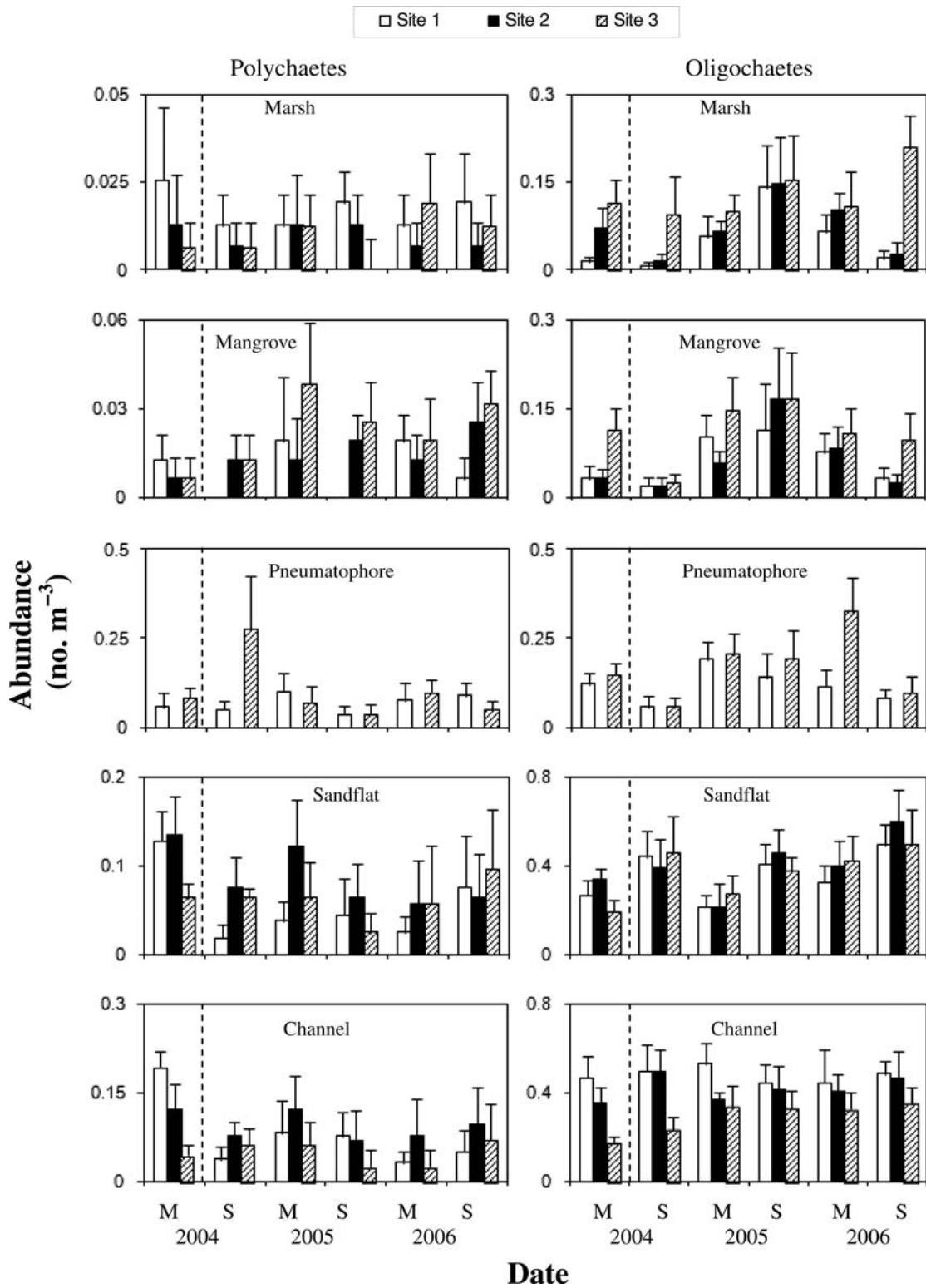


Figure 8. Mean abundance (\pm s.e.) of polychaetes and oligochaetes within different habitats at three sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) from March 2004 to September 2006 (mangrove removal illustrated by vertical dashed line). Note that there was no pneumatophore habitat at Site 2 and that the y-axes have different scales.

mangrove habitats. For other species, such as the mud snail *A. crenata*, the affinity to marsh and muddy substrata in the upper intertidal clearly define the distribution patterns

(Pilkington and Pilkington, 1982). Although abundances were similar within Sites 2 and 3 throughout the sampling period, mangrove clearance at Site 1 resulted in a sharp increase in that snail's

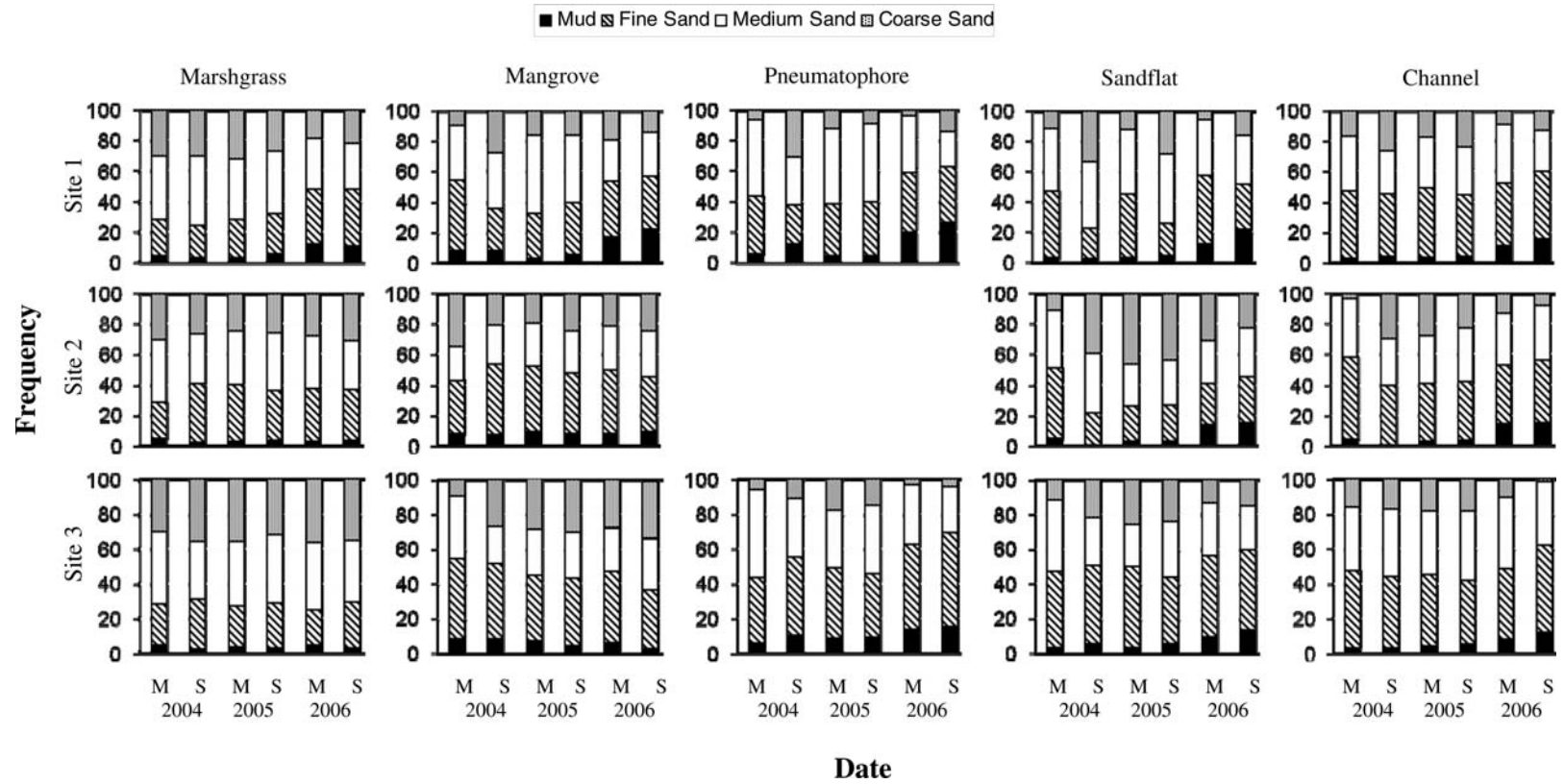


Figure 9. Frequency of grain-size distribution (mud, $< 150 \mu\text{m}$; fine sand, $150\text{--}300 \mu\text{m}$; medium sand, $300\text{--}600 \mu\text{m}$; coarse sand, $> 600 \mu\text{m}$), at three sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) for marshgrass, mangrove, pneumatophore, sandflat, and channel habitats between March 2004 and September 2006 (mangrove removal was between March and September 2004). Note that there was no pneumatophore habitat at Site 2.

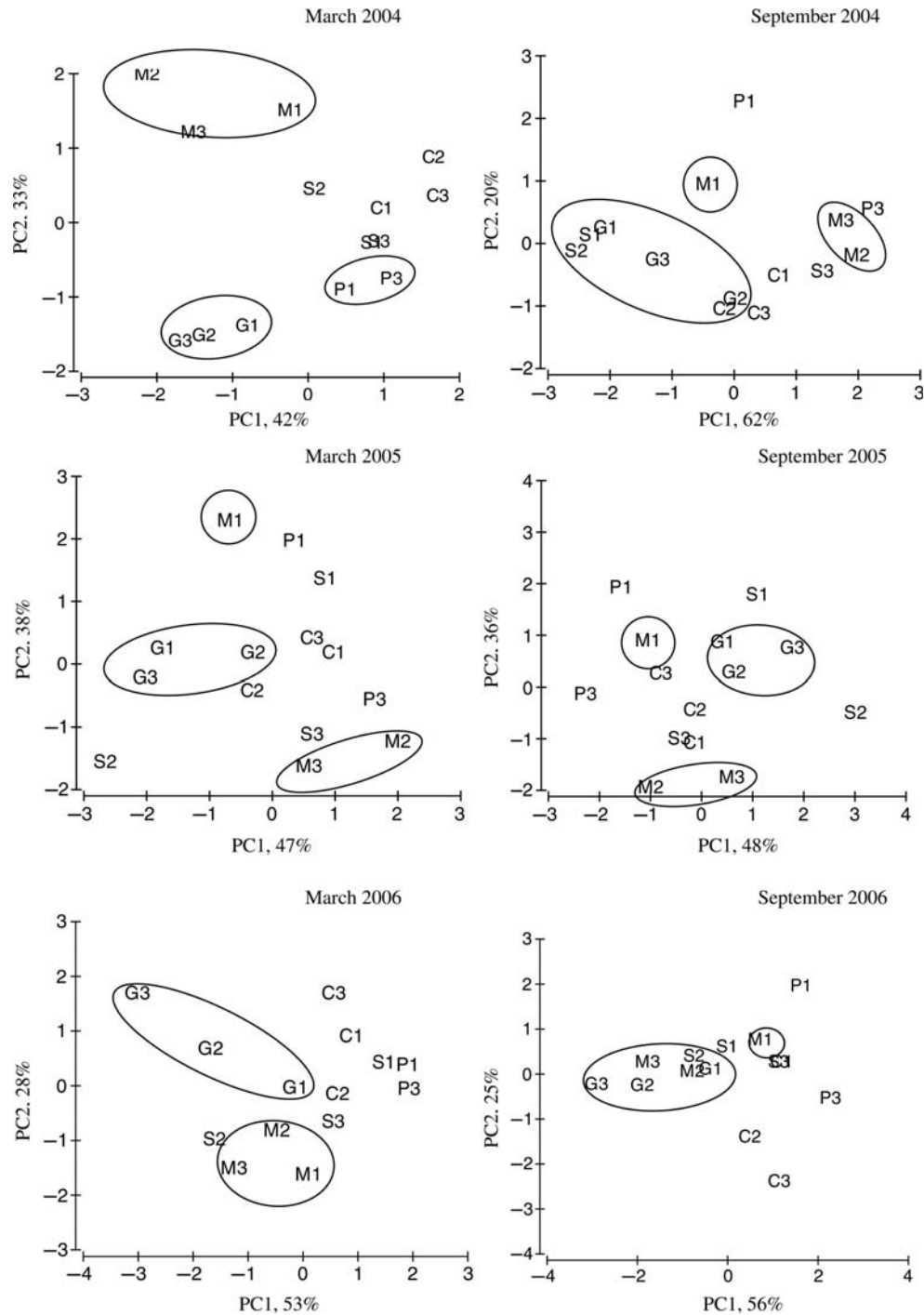


Figure 10. PCA plots of sediment (percentage mud and fine, medium, and coarse sand) across habitats (G, marshgrass; M, mangroves; P, pneumatophores; S, sandflats; C, channel) and sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) between March 2004 and September 2006 (mangrove removal was between March and September 2004).

abundance within marshgrass and mangrove habitats. These results suggest that the mangrove trees themselves may not enhance snail population densities, but that the muddy sediment and high organic content may facilitate growth and survival. Another pulmonate snail (*P. antipodarum*) exhibited stable, high densities in marshgrass habitats, but its density decreased in the mangrove habitat at Site 1 over time. The scavenger *Z. lutulentus* was most abundant in sandflat and channel habitats at Site 3,

where it often feeds on cockles and other dying invertebrates. A slight increase followed by a decrease in population density over time for this species was apparent for Sites 1 and 2, probably a consequence of their normally patchy distribution around food sources, which vary in time and space.

The dominant bivalves, *A. stutchburyi* and *P. australis*, both had generally constant densities throughout the sampling period, with a distinct decrease in population density in September 2006,

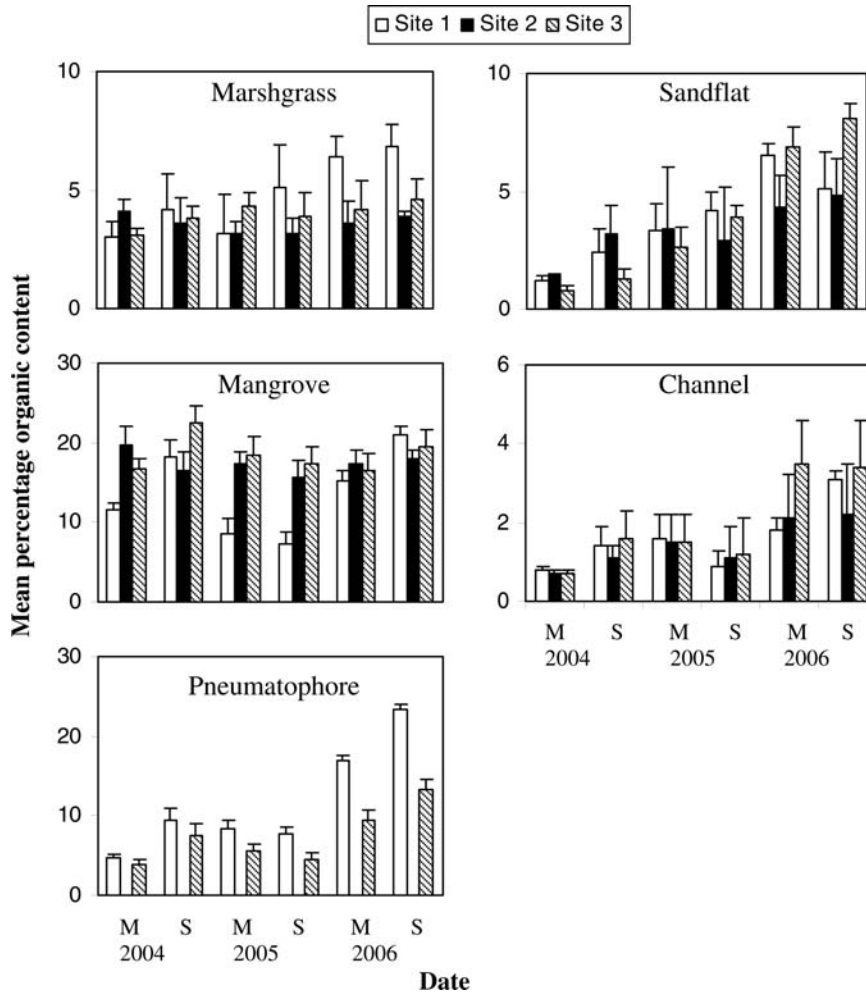


Figure 11. Mean percentage organic content (\pm s.e.) in sediment samples from five habitats within three sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) between March 2004 and September 2006 (mangrove removal was between March and September 2004). Note that the y-axes have different scales.

compared with previous sampling events. This decrease in density coincided with the increase in fine sediments and organic content observed, which can be attributed to catchment reworking. In addition, the abundance and distribution patterns of these species coincide with seasonal reproductive patterns (Hooker, 1995; Stewart and Creese, 2002). Finally, polychaete and oligochaete worms were most abundant in sandflat, channel, and pneumatophore habitats for all sites throughout the sampling period. Generally, polychaetes decreased in density immediately following mangrove removal at Site 1, and similar changes also were observed for oligochaetes. Although the densities of both groups of worm varied over time, no dramatic temporal changes were observed at Sites 2 and 3 (except for some instances at Site 3), indicating their success within muddy/sandy habitats.

Habitat modifications

It has been suggested that natural and anthropogenic changes to the structure of mangrove stands have a direct impact on the physical processes operating within the habitat and their associated fauna (Wolanski *et al.*, 1992). For example, the creation of canopy gaps may cause changes in soil characteristics (Clarke and Kerrigan, 2000) and promote the growth of saplings (Sherman *et al.*, 2000).

Moreover, significant changes in the abundance and composition of associated taxa take place after disturbances of mangrove architecture (Schrijvers *et al.*, 1995; Skilleter, 1996; Kelaher *et al.*, 1998; Skilleter and Warren, 2000; Bosire *et al.*, 2004). The results of this study show that removal of mangroves altered the sediment characteristics and abundance of macrofauna within the habitat, suggesting that the transformation from mangrove to mudflat habitat immediately increased species abundance and diversity. However, these faunal characteristics continued to change over the next 2.5 years and could not be fully separated from further anthropogenic catchment activities at the study site. In western Kincumber Broadwater, New South Wales, Australia, dramatic changes in benthic faunal composition and increased community structure variability took place after mangrove canopy pruning (from 5 to 1 m height; Gladstone and Schreider, 2003). However, that study evaluated faunal effects only once, 5 years after the event, and the authors interpreted the ecological changes based on comparisons with nearby undisturbed stands. In this study, faunal and sediment characteristics were recorded before and after the clearance event, monitored for 3 years, and compared with undisturbed sites. In addition, ecological effects on adjacent habitats (i.e. sandflats, subtidal channels) also were investigated.

In summary, this study provides a comprehensive evaluation of the effects of mangrove removal on faunal and sediment characteristics in a mangrove-dominated estuary. The results also suggest that ecological changes may extend beyond the immediate area (~100 m) and into other habitats and that the effects may persist for 3 or more years after the event. The results may provide valuable ecological information for environmental managers seeking to control the spread of mangroves in the estuaries of northern New Zealand.

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Appendix

List of the taxa found inside and outside cores within each of five different habitats (G, marshgrass; M, mangroves; P, pneumatophores; S, sandflats; and C, channel) within three estuarine sites (Site 1, mangrove-removal area; Sites 2 and 3, control sites) in Mangawhai Estuary, New Zealand, between March 2004 and September 2006. Note that there was no pneumatophore habitat at Site 2.

| Taxon | Site 1 | | | | | Site 2 | | | | Site 3 | | | | |
|-------------------------------------|--------|---|---|---|---|--------|---|---|---|--------|---|---|---|---|
| | G | M | P | S | C | G | M | S | C | G | M | P | S | C |
| Polychaeta | | ✓ | ✓ | ✓ | ✓ | | ✓ | ✓ | ✓ | ✓ | | ✓ | ✓ | ✓ |
| Oligochaeta | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ |
| Sipuncula | | | | | ✓ | | | ✓ | ✓ | | | | | ✓ |
| Polyplacophora | | | | | | | | | | | | | | |
| <i>Chiton glaucus</i> | | | | ✓ | ✓ | | | | ✓ | | | | ✓ | |
| <i>Sypharochiton pelliserpentis</i> | | | ✓ | | ✓ | | | | | | | | ✓ | |
| Gastropoda | | | | | | | | | | | | | | |
| <i>Amphibola crenata</i> | ✓ | ✓ | ✓ | ✓ | | ✓ | ✓ | | | ✓ | ✓ | ✓ | | |
| <i>Bulla quoyii</i> | | ✓ | ✓ | | | | | | | | | | | |
| <i>Cominella glandiformis</i> | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ |
| <i>Cominella maculosa</i> | | ✓ | | | | | | | | | | ✓ | | |
| <i>Diloma subrostrata</i> | | ✓ | ✓ | ✓ | | | | ✓ | ✓ | | | ✓ | ✓ | |
| <i>Lepsiella scobina</i> | | | ✓ | | | | ✓ | | | | ✓ | | | |
| <i>Melagraphia aethiops</i> | | ✓ | ✓ | | | | | | | | ✓ | | | |
| <i>Nerita atramentosa</i> | | | | | | | ✓ | | | | ✓ | | | |
| <i>Ophicardelus costellaris</i> | ✓ | ✓ | ✓ | | | ✓ | | | | ✓ | ✓ | | | |
| <i>Potamopyrgus antipodarum</i> | ✓ | ✓ | ✓ | | | ✓ | | | | ✓ | ✓ | ✓ | | |
| <i>Turbo smaragdus</i> | | | ✓ | | | | ✓ | | | | ✓ | | | |
| <i>Zeacumantus lutulentus</i> | | ✓ | ✓ | ✓ | ✓ | | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ |
| <i>Zeacumantus subcarinatus</i> | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ |
| Bivalvia | | | | | | | | | | | | | | |
| <i>Austrovenus stutchburyi</i> | | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | | ✓ | ✓ | ✓ | ✓ |
| <i>Felaniella zelandica</i> | | | | ✓ | | | | | | | | | | |
| <i>Macomona liliana</i> | | ✓ | | ✓ | | | | ✓ | | | | ✓ | ✓ | |
| <i>Nucula hartvigiana</i> | | | ✓ | ✓ | ✓ | | | ✓ | | | | ✓ | ✓ | |
| <i>Paphies australis</i> | | | | ✓ | ✓ | | | ✓ | ✓ | | | ✓ | ✓ | ✓ |
| <i>Saccostrea cucullata</i> | | | | | | | ✓ | | | | ✓ | ✓ | | |
| <i>Venericardia purpurata</i> | | | | ✓ | ✓ | | | ✓ | ✓ | | | | | ✓ |
| <i>Xenostrobus pulex</i> | | | | | | | ✓ | | | | ✓ | ✓ | | |
| Cirripedia | | | | | | | | | | | | | | |
| <i>Chamaesipho brunnea</i> | | ✓ | ✓ | ✓ | | | ✓ | | | | ✓ | ✓ | ✓ | ✓ |
| <i>Chamaesipho columna</i> | | ✓ | ✓ | | ✓ | | ✓ | | ✓ | | ✓ | ✓ | ✓ | |
| Amphipoda | | | | | | | | | | | | | | |
| <i>Corophium acutum</i> | | | | ✓ | ✓ | | | ✓ | ✓ | | | | ✓ | ✓ |
| Unidentified | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ |
| Isopoda | | | | | | | | | | | | | | |
| <i>Isocladus armatus</i> | | | | ✓ | ✓ | | | ✓ | | | | | ✓ | ✓ |
| Unidentified | | ✓ | ✓ | | | | | ✓ | ✓ | | | | ✓ | |
| Decapoda | | | | | | | | | | | | | | |
| <i>Elamena producta</i> | | | | | | | | ✓ | | | | ✓ | | |
| <i>Helice crassa</i> | ✓ | ✓ | ✓ | ✓ | | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | |
| <i>Palaemon affinis</i> | | | | | ✓ | | | ✓ | | | | | | ✓ |
| Cirripoda | | | | | | | | | | | | | | |
| <i>Chamaesipho brunnea</i> | | ✓ | ✓ | | | | ✓ | | | | ✓ | ✓ | | |
| <i>Chamaesipho columna</i> | | ✓ | ✓ | | | | ✓ | | | | ✓ | ✓ | | |
| <i>Helmsi scapha</i> | | | ✓ | ✓ | ✓ | | | ✓ | ✓ | | | ✓ | ✓ | |
| Osteichthyes | | | | | | | | | | | | | | |
| <i>Anguilla dieffenbachii</i> | | | | | ✓ | | | | | | | | | |