Factors affecting the habitat usage of estuarine juvenile fish in northern New Zealand

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Abstract

New Zealand's estuaries are coming under increasing pressure from anthropogenic change and degradation. Few, if any remain in a pristine state. Such environments play an important role as nursery areas for juvenile fish. However the processes underlying the patterns of juvenile fish habitat use are poorly understood. This thesis examined abiotic and biotic factors, influencing the distribution and abundance of juvenile fishes over multiple spatial and temporal scales within 9 northern New Zealand estuaries and one offshore island, and the impacts of changing environmental conditions on the foraging success and health of juvenile fishes.

Small fish assemblages, abiotic variables (e.g. sediment grain size, suspended sediments, current velocity) and biotic variables (e.g. benthic invertebrates and predatory fishes) were quantified along a 20km environmental gradient in the Manukau Harbour over two seasons. Fish assemblages showed strong spatial and temporal variability along the gradient, with densities declining from the sheltered, muddy upper Pahurehure Inlet to the clearer, deeper, faster-flowing sandy areas of the lower harbour. BIOENV and CCA analysis revealed that depth, current velocity and turbidity were the most strongly correlated with the fish assemblages. Other contributors retained in the CCA forward selection included mysid abundance, quantity of biogenic habitat and percentage of fine sediments (<64μm). Dietary composition shifted seasonally in response to changes in food source availability, with a significant increase in consumption of mysids in spring (contributing 66% of total dietary biomass). During spring, the density of new recruits were positively correlated with mysid abundance. Diversity of juvenile fish and dietary breadth were highest at mid harbour sites, which also contained the most biogenic structure.

The impact of turbidity on juvenile fish was assessed using snapper (50-90mm FL) as a model species. In experimental aquaria, increasing suspended sediment (TSS) levels resulted in decreased foraging success. Longer-term (one month) exposure to increased TSS resulted in higher weight losses, higher mortality rates and significant increases in gill deformation (epithelial hyperplasia, and fusion of the lamellae) which impaired respiratory function. Other sublethal effects included increased coughing and gulping at the surface, higher respiration rates and decreased activity, which are consistent with the effects of anoxia. A field survey of seven northern estuaries revealed that juvenile snapper had significantly lower condition indices in the more impacted estuaries characterized by increased sedimentation and catchment urbanisation. Higher levels of gill deformation and parasite loads were also recorded, particularly for Manukau and Mahurangi Harbours. Dietary analysis revealed a change in foraging tactics with increasing turbidity from active (probably visual) selection of pelagic prey (zooplankton; 0.5-0.71mm), to larger, slower moving benthic prey (≥ 0.71 mm). Increasing TSS can thus potentially restrain juvenile snapper condition and growth by reducing the overall food supply available and perhaps relative nutritional values by reducing their ability to select optimal prey sizes, Results suggest that physiological stress in fishes in response to increased TSS can decrease immunological competence and growth.

Fish and macro invertebrates in seagrass, mangroves, sand and mudflats were sampled from seven northern estuaries over late summer to assess the relative value of specific habitats for juvenile fish. Different habitats supported different species assemblages of fish and invertebrates which varied with latitude, geographical setting (east/west) and between and within estuaries (tidal position). A small number of species such as yellow-eyed mullet and exquisite goby had more ubiquitous distributions. Benthic faunal diversity, abundance biomass and productivity tended to be highest in seagrass habitat (particularly subtidal seagrass), followed by sand, mangroves and mud habitats. Results support the paradigm that seagrass meadows are a valuable nursery habitat in northern New Zealand, particularly for species such as snapper and trevally.

Ontogenetic dietary shifts were evident for the majority of fish species, with meiofaunal crustaceans (0.5-1mm) predominating. Newly recruited fish exhibited an obligatory planktivorous stage, with a gradual transition to larger crustaceans such as mysids, gammaridean amphipods and caridean shrimps and crabs. Habitat-related differences in diet were also evident, reflecting benthic prey availability and diversity.

Overall, these findings indicate that ongoing large-scale environmental changes within estuaries are affecting the functioning of fish nurseries both directly (by reducing the fitness of individual fish) and indirectly (by reducing the area of biogenic habitats such as subtidal seagrass beds). This highlights the need for resource management to include linkages to catchment level effects on estuarine habitats and the juvenile and small fish assemblages that are associated with them.

Dedicated to my mother –
Delce Alison Lowe

Who taught me to appreciate the small
wonders of nature.

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And now it is finally done.

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Introduction


Photo: R. Grace

‘Yellow-eyed mullet in Whangateau Harbour’.

Chapter 1: General Introduction

Estuaries are highly productive systems, supporting elevated abundances of both fish and invertebrate species (Beck et al., 2001; Blaber et al., 2000; Edgar et al., 2000; Caddy, 2007). Estuaries are now widely recognised as 'nursery areas' for many juvenile fish species, due to their benign, shallow, sheltered environs, elevated food supplies (e.g. Orth et al., 1984; Nagelkerken et al., 2008) and protection from predation (Potter et al., 1990; Paterson & Whitfield, 2000; Beck et al., 2001; 2003.

Estuaries are comprised of a complex mosaic of habitats (mudflats, sandflats and biogenic structures such as marshes, mangroves, seagrass meadows and mussel/oyster reefs), which are interconnected through fish movement (Heck et al., 2003). The mobility of fish and flexible foraging strategies across multiple habitats during their life cycle gives them a crucial role in coupling of habitats and processes (Sheaves, 2005). Nevertheless, the use of multiple habitats makes fishes especially vulnerable to the effects of habitat modification across these 'movement corridors' between estuaries and offshore habitats (Gillanders, 2003). Degradation of one habitat can result in bottlenecks for recruitment across the whole mosaic (Gillanders, 2003; Ray, 2005; Gratwicke et al., 2006; Jaureguizar et al., 2006). However, given the historical pre-eminence of estuaries as preferred settlement sites for humans (60% of the world's population is located within 60 km of the coast), the intensity of human perturbation continues to accelerate, interrupting these connectivities (Ray, 2005). Estuaries are now thought to be amongst the most anthropogenically degraded ecosystems on earth, with few considered 'pristine' (Edgar et al., 2000; McLusky & Elliot; 2004; Lotze et al., 2006; Airoldi & Beck, 2007).

1.1 Coastal ecosystems: threats and stressors

Human activities within estuaries, including construction of canals and marinas, dredging and reclamation for port facilities, damming and diverting tributaries in conjunction with urban and industrial development in coastal watersheds have extensively modified their physical and hydrological characteristics, while catchments and shorelines have largely been cleared of their natural vegetation (Edgar et al., 2000; Kennish, 2002; McLusky & Elliot, 2004; Vasconcelos et al., 2007).

As the ultimate receiving environment for land-based activities, estuarine waters and sediments have accumulated increasing quantities of heavy metals, organic materials and nutrients from urban, industrial and intensive agricultural practices (Moss, 2008; Courrat et al., 2009). As a consequence, the ecological function of estuaries is being increasingly challenged by a multitude of environmental issues, including eutrophication, pollution, anoxia and increasing sedimentation, all of which have had deleterious impacts on estuarine fauna (Kennish, 2002; Goto & Wallace, 2011).

The magnitude of historical change within temperate near shore habitats over the past 150-300 years has been profound (Airoldi & Beck, 2007; Lotze et al., 2006; 2011) with >90% of formerly important species depleted, 65% of seagrass and wetland habitats destroyed in conjunction with degraded water quality and accelerated species invasions. Centuries of overexploitation have reduced once diverse and productive estuarine and coastal areas to impoverished versions of their pristine counterparts, with a resultant loss of species biomass, biodiversity and ecological resilience (Turner et al., 1999a; Jackson, 2001; Jackson et al., 2001; Thrush et al., 2004; 2006; Le Pape et al., 2007), and the eventual loss of important juvenile fish habitats. These subtle, long-term changes, often veiled from general public notice, has resulted in reduced expectations (i.e. ‘sliding baselines’) of what constitutes a ‘normal’ functioning ecosystem (Pauly, 1995; Tegner & Dayton, 1999).

1.2. Requirements for effective management

Despite the extent of degradation (Turner et al., 1999a; Edgar & Barret, 2000; Heck et al., 2003; Kennish et al., 2008) and increased vulnerability to the effects of climate change (Waycott et al., 2011; Willis et al., 2007; Gillanders et al., 2011), estuarine and marine conservation lags far behind terrestrial conservation (Edgar et al., 2000). Nonetheless, there has been an increasing emphasis on ecosystem protection, conservation and surveillance. Fisheries management is beginning to shift from traditional single species management towards ecosystem-based protection, i.e., preserving the structure and functions of habitats and ecosystems (Blaber, 2000; Beck et al., 2003; Jaureguizer et al., 2006). Conservation of the ‘sequential chain’ of habitats required by fish throughout their lives is a key tenet (Fluharty, 2000). Recent legislative framework for fisheries management reflects this conceptual shift (e.g. New Zealand: Fisheries Act, 1996; Ministry of Fisheries Strategy Document 2030 (MoF, 1997, 2012); USA Sustainable Fisheries Act, 1996; European Marine Strategy Framework Directive 2008/56/EC (Fluharty, 2000; Strickland & Grosse, 2000; Le Heron et al., 2008; Borja et al., 2010). However, despite the widespread acknowledgement of the need for ecosystem based management, relatively little is still known of the basic ecology of most estuarine fish species, including their basic habitat and food requirements (Johnson, et al., 2012).

1.2.1 *Identifying 'essential fish habitat'*

One of the foremost issues facing management and conservation of coastal waters is to prioritize 'essential fish habitats' (EFH), i.e. 'those waters and substrates necessary to fishes for the purpose of spawning, breeding, feeding, or growth to maturity' (Laegdsgaard & Johnson, 1995; Beck et al., 2001; Fluharty, 2000, Stål et al., 2007). For species with complex life cycles (e.g. egg, larval, juvenile & adult stages), priority is usually given to those habitats that can be identified as nurseries (Kraus & Secor, 2005). Nursery habitat is defined as an area that contributes disproportionately to the size and numbers of adults relative to other juvenile habitats either on a unit area (Beck et al., 2001) or absolute basis (Dahlgren et al., 2006).

With many coastal adult fish populations utilising habitats and areas that are spatially discrete from those of their juvenile populations, an understanding of the connectivity between juvenile and adult habitats is fundamental (Beck et al., 2001; Gillanders, et al., 2003). Quantifying what species live where, and identifying the key environmental factors that drive their distributions (for both juveniles and adults), over varying regional/geographic scales is an essential prerequisite for addressing more complicated themes such as habitat connectivity (Edgar et al., 1999; Edgar & Barret, 2000; Ley, 2005; Boström et al., 2006; Jaureguizar et al., 2006). To date, most studies have focused on quantifying density, biomass and length frequencies of single species in one or two habitats, have been limited in spatial scale, and have not considered biotic factors such as food availability within varying habitats concurrently (Heck et al., 2003; Stål et al., 2007), but see (Sanchez-Jerez et al., 2002; Kaiser et al., 2004; Hinz et al., 2005).

1.3 New Zealand

Within New Zealand, estuarine fish research prior to the 1990's was primarily descriptive and limited to a small number of estuaries (e.g. Webb, 1973; Colman, 1974a,b; Eldon & Kelly, 1985; Kilner & Ackroyd, 1978; Davenport, 1979; Roper & Jillet, 1981; Park, 1984; Jellyman et al., 1997). Since the late 1990's, studies have begun to address questions surrounding fish habitat associations within estuaries (e.g. Saunders 1999, Francis et al., 2005; 2011, Hartill et al., 2003, Morrison et al., 2002, 2007, unpubl. data and paper in prep.; Schwarz et al., 2006; Thrush et al., 2002; Capone, 2008; Usmar, 2009). Current research suggests biogenic habitats (e.g. seagrass meadows, green-lipped mussels, oyster reefs and horse mussels) are important nurseries for juvenile fish (Francis et al., 2005; 2011; Usmar, 2009; Morrison et al., 2012; in review). International research suggests increasing habitat complexity can benefit juvenile fish distribution by providing shelter from waves and currents (Foncesca & Koehl, 2006; Widdows et al., 2008), and refuge from predators (e.g. Marshall & Elliot, 1998; Hindell et al., 2000 a,b) in addition to providing increased surface area for elevated densities of prey, particularly for habitats such as seagrass (see reviews by Bell & Pollard, 1989; Gillanders, 2006).

However, historical data on changes in the distribution and abundance of important biogenic habitats over time is limited (e.g. seagrass meadows: Park, 1999; Turner & Schwarz, 2004; Matheson et al., 2010; Morrison et al., 2009, 2012). Fundamental information gaps still extend across much of New Zealand's marine environment, particularly for juvenile stages in terms of what species associate with different biogenic habitats, and where and how extensively these habitats occur.[1] Additionally the relative roles of factors such as food and shelter in determining the attractiveness of various biogenic habitats to juvenile fish are mostly unknown.

New Zealand's terrain is predominantly mountainous and hilly with 50% of the land mass at slopes of $>28^{\circ}$, and many areas are composed of highly erodible soft siltstones/mudstones, particularly in north eastern New Zealand (Hicks et al., 2000; Morrison et al., 2009). This, coupled with high conversion rates of native forest to pasture for intensive agriculture, has substantially increased the susceptibility of slopes to landslides (Glade, 2003, Fig. 1.1). Inputs of sediments to the coastal zone are now especially high by world standards, approaching almost 1% of total world sediment yields (Robertson & Stevens, 2006; Morrison et al., 2009).

With the majority of sediments entering New Zealand estuaries during flood events (Hicks, 1984; Hicks et al., 2000; Oldman et al., 2009), the predicted increase in frequency and intensity of storms associated with climate change (Willis et al., 2007) is likely to increase levels of sedimentation within both freshwater and marine environments. Increased sedimentation can produce a wide range of effects, both from deposition of fine sediments on the seafloor, and as suspended sediments in the water column. Ongoing re-suspension and deposition events (e.g. by storms and fishing gear) may shift sediments between these two states. Elevated suspended sediments can alter many of the physical attributes of the environment, including increased turbidity, reduced visibility and depth of the photic zone and subsequent photosynthetic activity, food availability and plant biomass (Moore, 1977; Bruton, 1985). Direct effects on fish species

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[1] Biogenic habitats, defined as, three dimensional emergent habitats formed by plants and animals separated from surrounding seafloor e.g. seagrass meadows, horse mussels; sponges).

include reductions in visibility of pelagic food and clogging of gills, with associated acute and/or chronic impacts, e.g. immediate physiological stress and reduced growth rates and reproductive fitness. Indirect effects include the loss of important nursery habitats such as biogenic habitat formers, e.g. horse mussels (*Atrina zealandica*) and seagrass meadows (*Zostera muelleri*), (Inglis, 2003; Ellis et al., 2004; Cummings et al., 2005; see review of Morrison et al., 2009). This occurs via abrading, clogging and smothering organisms and reductions in prey assemblages (Norkko et al., 2002; Thrush et al., 2004; Lohrer et al., 2006a, Morrison et al., 2009).

Conversely, increasing deposition of fine muds has seen New Zealand's mangrove forests (*Avicennia marina*) rapidly expanding in the upper North Island (Table 1.1), (Schwarz, 2003), with the Firth of Thames mangrove forest expanding by an average of 20 m/yr over the last 50 years (Swales et al., 2007; Morrisey et al., 2007). This has led to intense and sometimes acrimonious societal debate as to the ecological value of these forests relative to the ecological functions and social amenities that have diminished (e.g. shellfish beds; sandy beaches; access to waterways). Until recently, there has been virtually no quantitative data on what small fish assemblage's mangrove forests may support (Morrisey et al., 2007; 2010).

Most of our current knowledge on the effects of suspended sediments on fish is based on freshwater species (see reviews of Newcombe & Jensen, 1996; Bash, 2001; Wilber & Clarke, 2001) with a focus on laboratory experiments, measuring the impacts of acute exposure. Little empirical information is available on chronic responses to high concentrations for extended periods, especially for estuarine and marine species (Au et al., 2004), or under natural field conditions. Elevated suspended sediments have been shown to increase mortality and cause sublethal responses including reductions in feeding rates, changes in type of prey consumed, reduced growth, delayed hatching, avoidance of suspended sediments, along with physiological changes including changes in blood physiology and gill structure, increased respiration and cough responses and increased susceptibility to diseases (see reviews of Newcombe & MacDonald, 1991; Kerr, 1995; Morrison et al., 2009).

However, virtually no information exists on the direct effects of sedimentation on estuarine and marine fish in New Zealand. With the continuing decline in the health of estuarine waters from elevated rates of sedimentation and increasing eutrophication, research on the impacts of these stressors on juvenile fish and their estuarine habitats is critical.

1.4 Study Sites

Sampling for this study was undertaken across nine northern New Zealand estuaries encompassing both east (Rangaunu, Mahurangi, Whangateau, Waitemata, Tamaki, Tairua) and west (Kaipara, Manukau, Kawhia) coasts, as well as a coastal island subtidal seagrass site: Urupukapuka Island, Bay of Islands (BOI), (Fig. 1.2). They spanned 551 km in latitude (from north to south) and differed considerably with respect to their morphology, size, hydrology, sediments and degree of exposure to wave action. A summary of the key environmental characteristics of each estuary is given in Table 1.1.

1.5 Aims

The primary aims of this thesis were to determine what factors govern spatial variation in the nursery value of estuarine juvenile fish habitats within northern New Zealand estuaries. With most research focusing on individual species and habitats within single estuaries, this research is the first in New Zealand to consider all habitats encompassing (a) an environmental gradient within a single estuary (Chapter Two), and (b) multiple habitats at a regional scale linking the distribution of juvenile fish assemblages to prey availability within multiple habitats (Chapters Two & Four). It is also the first local study to evaluate the effects of changing environmental conditions (i.e. increasing suspended sediments/turbidity) on the health of juvenile fishes.

Given the ongoing degradation of estuarine habitats and increasing intensity of human perturbation, knowledge of how juvenile fish assemblages are structured, and information on the effects of changing environmental conditions, will provide valuable baseline information against which to better predict the future effects (e.g. seagrass loss) of cumulative human-induced impacts on the extent and quality of the estuarine habitats.

Do environmental gradients within estuaries matter?

Chapter Two: quantifies spatial and temporal changes in the distribution and abundance of juvenile fishes (<125 mm FL) along a 20 km estuarine gradient within the Manukau Harbour. Potential causal relationships between fish assemblage variation and environmental factors such as current speed, suspended sediments, depth, habitat, position within estuary and biotic factors including prey availability and presence of predators are identified. Prey species consumed were quantified for the entire juvenile fish assemblage relative to fish species, age/size and habitat type and related to estimated secondary productivity of different habitats along the turbidity gradient.

How does habitat and environmental change influence the foraging success and health of juvenile fishes?

Chapter Three examines how habitat and environmental change may influence the feeding and health of juvenile estuarine fishes. Laboratory experiments, using snapper as a model species, are undertaken to assess the effects of increasing turbidity/total suspended sediments (TSS) on their ability to feed on live prey over different substrates in short term (30 minute) trials. Effects of longer-term (30 day) exposure to elevated turbidity/TSS on health and growth are also assessed. To link laboratory results to field conditions, health and condition indices of juvenile snapper in seven northern New Zealand estuaries are investigated across a turbidity/TSS gradient.

How important are biogenic habitats to juvenile fish within New Zealand estuaries?

Chapter Four: compares the feeding ecology of juvenile fish assemblages within and between habitats and estuaries across six northern estuaries and one offshore island, and how this varies ontogenetically. Relationships between fish assemblages and environmental parameters such as habitat type (i.e. sand flat, seagrass, mangrove and intertidal mudflat) and invertebrate prey availability are assessed. Prey species consumed were quantified for the entire juvenile fish assemblage relative to fish species, age/size and habitat type and related to estimated secondary productivity of the four habitats.

A

Photo: Gisborne Council

B

NIWA file photo

Figure 1.1 Extensive soil-slip erosion on hill country pasture at Gisborne post cyclone Bola (A); Example of sediment plume from a northern estuary (B).

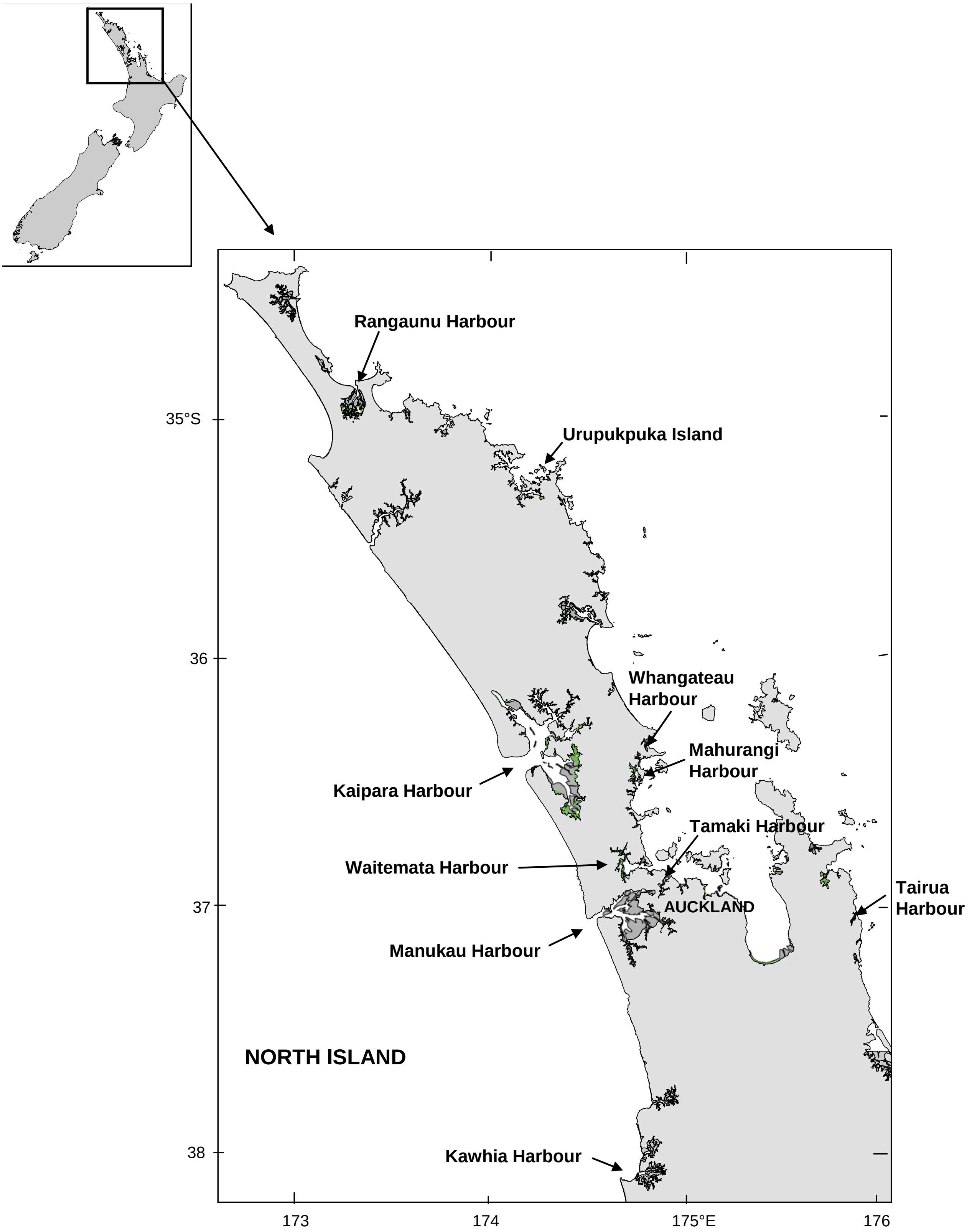


Figure 1.2 Map of northern New Zealand showing the location of the ten estuaries sampled.

Table 1.1 Physical characteristics and habitat composition of the ten harbours surveyed for this study (A, adapted from T. Hume, NIWA Estuary Environment Classification database); Tree architecture, sediment and water column properties for the four harbours sampled for mangroves, (B), (adapted from Morrisey et al., 2007).

						% of HW area						Land Cover (% catchment)						
Harbour	**Coast**	**Type**	**Area (km²)**	**Catchment Area (km²)**	**Depth**	**Intertidal Area** (% of HW)	**Sand Area** (% of HW)	**Mud Area** (% of HW)	**Mangroves Area** (% of HW)	**Mangrove Area (km²)**	**Seagrass Area (km²)**	**Natural**	**Pastoral**	**Exotic forest**	**Urban**	**Misc**	**Mangrove % increase**	**Period**
Rangaunu	East	TL	101.7	552	2	78	51	50	25.8	30	20	30.0	67.8	1.9	0.5	0.3	33%	1944-1981[1]
Mahurangi	East	DV	24.6	122	3	51.6	32	23	19.6	4.8	0.01	21.9	64.6	9.8	3.1	0.7	9.8%	1960-2004[2]
Whangateau	East	DV	7.5	42	2	85	78	77	7	0.5	0.3[e]	18.6	34.5	4.2	42.6	0.1		
Waitemata	East	DV	79.9	427	4	36	24	23	12	4.9[a]	0.3[f]	21.9	64.6	9.8	3.1	0.7	6.3%	1950-1996[3]
Tamaki	East	DV	16.9	109	3	40	32	32	8	---	---	2.0	24.7	0.2	73.1	0.0		
Tairua	East	DV	6.0	282	1	51	51	51	---	3.5	~1.29[b]	70.4	15.9	12.4	1.2	0.0	194%	1983-2008[4]
Kawhia Harbour	West	DV	67.6	499	2	74	74	73	0	0.01	7.93	46.1	53.8	0.1	0.0	0.0		
Kaipara	West	DV	743.1	6266	5	41.9	34	29	8.3	61.7	51.28[c]	17.9	70.5	11.3	0.2	0.1	18%	1953-1996[5]
Manukau	West	DV	365.1	1023	6	61.8	60	60	1.7	6.1	---	20.4	67.9	0.9	10.5	0.2	160%	1939-1996[6]
Urupukapuka Is	East		2.65	---	5	---	---	---	0	0	0.047[d]							1967-2006

A.

DV= drowned valley; TL= tidal lagoon; HW=high water

[a] Gao et al., (2004)

[b] WRC (2005)

[c] ARC (2001)

[d] Matheson et al., (2010)

[e] Hartill et al., (2000)

[f] S. Hailes (pers. comm.)

[1] Shaw et al., (1990); NCC (1984); May (1999)

[2] ARC (unpublished)

[3] Lucas Creek, Morrisey et al., (2003)

[4] NCC (1984), EW (2009), (cited in Morrisey, 2010)

[5] Mullet Creek, Morrisey (1999)

[6] Puhinui Creek, Morrisey et al., (1999

B.

Harbour	Seedlings	Saplings	Trees	Branches	Trunk	Height	P. Zone	Channel	TSS	TOC	Grain	Clarity (cm)
Rangaunu	20.5 (11.6)	3.1 (2.1)	30.6 (8.3)	68.5 (8.4)	44.1 (4.9)	4.0 (0.3)	15.1 (1.4)	181.6 (43.7)	14.6 (5.1)	3.5 (0.6)	119.4 (14.7)	114.1 (10.4)
Mahurangi	17.0 (4.8)	1.0 (0.8)	21.2 (5.6)	52.2 (9.5)	47.7 (3.1)	3.6 (0.5)	11.8 (3.2)	51.7 (17.0)	27.1 (7.2)	2.7 (0.2)	43.5 (8.3)	51.3 (4.9)
Kaipara	5.1 (2.5)	0.8 (0.2)	25.8 (7.8)	99.2 (10.6)	56.0 (9.4)	4.5 (0.4)	11.2 (1.0)	80.8 (30.9)	61.7 (20.8)	3.2 (0.3)	46.5 (15.1)	46.7 (6.6)
Manukau	10.7 (3.4)	0.3 (0.2)	41.7 (17.6)	79.8 (12.6)	31.2 (3.7)	3.4 (0.2)	6.2 (1.7)	144.5 (54.0)	41.7 (15.8)	2.6 (0.3)	24.7 (4.0)	70.2 (11.2)

Data for tree architecture was gathered from within a 10x10m plot immediately behind the set fyke net. Seedlings/saplings were counted within three random 1m^2 quadrats. Branches= height of first branching (cm); Trunk=diameter (cm); P. zone= width of pnuematophore zone (m); Channel=distance to nearest channel (m); TSS= Total suspended sediments (g/m^3); TOC=Total organic carbon; Grain=Grain size.

Chapter Two

Variation in the assemblage of small fishes along an environmental gradient in the Manukau Harbour

‘Pahurehure Inlet, Manukau Harbour’.

Chapter 2: Variation in the assemblage of small fishes along an environmental gradient in the Manukau Harbour

2.1 Introduction

Estuaries are transition areas, between land and freshwater and are arguably more complex than other ecosystems, with highly interrelated physical, chemical and biological processes that are subject to extreme variability (Elliot et al., 2002; Islam & Tanaka, 2006; Martinho, 2009). In addition to filtering pollutants, recycling nutrients and stabilizing shorelines, estuaries contribute to coastal food webs via their high primary and secondary productivity (Beck et al., 2001; Kennish, 2002). One of their most important roles is providing nursery habitats for a suite of fish and invertebrates with complex life cycles, including many commercially important species (e.g. Lenanton & Potter, 1987; Costa et al., 2002; Caddy, 2007; Selleslagh & Amara, 2008; Payne & Gillanders, 2009).

While few species spawn within estuaries (estuarine residents), many marine species utilise estuaries during juvenile development (marine migrants), with larvae hatching in adjacent coastal waters, entering estuaries as post larvae or early juveniles over spring/summer taking advantage of suitable conditions for growth, namely high food availability, water temperature and low biotic stress (Blaber & Blaber, 1980; Vasconcelos et al., 2010). Other species utilize estuaries only occasionally (freshwater and marine straggler), or transitionally as a migratory pathway (i.e. anadromous/catadromous species). For the purposes of this study the functional groupings proposed by Elliot et al., (2007) standardizing fish guild definitions based on estuarine use, feeding and/or reproductive mode have been utilized and adapted for the New Zealand situation (Fig. 2.1).

Many mechanisms may influence the distribution of fish within estuaries, including changes in local environmental conditions (Day et al., 1989; Kennish, 1990; Akin et al., 2005), and large-scale seasonal migrations (Blaber & Blaber, 1980; Potter et al., 1988; Akin et al., 2003). Considerable research has been undertaken on the relationship between environmental factors and the distribution of fish within estuaries. Numerous abiotic factors have been shown to influence the utilization of estuaries by fish (e.g. Marshall & Elliot, 1998; Whitfield, 1999; Cabral et al., 2001; Akin et al., 2005; 2003 and references therein; Martinho et al., 2007; Selleslagh et al., 2009; Nicolas et al., 2010). Salinity, depth and turbidity on a spatial scale, and temperature on a temporal scale, have been regarded as the main parameters governing community structure (Loneragon et al., 1986; Martin et al., 1995; Thiel et al., 1995; Cyrus &

Blaber, 1987a; 1992; McLusky & Elliot, 2002; Akin et al., 2003; Martinho & Able, 2003; Maes et al., 2004; Jauraguizar et al., 2006, 2004, 2003; Selleslagh & Amara, 2008a, 2008b; Selleslagh et al., 2009). Biotic processes, such as food availability, competition and predation have also been suggested as influential in driving spatio temporal patterns of fish distribution (Holbrook & Schmitt, 1989; Edgar & Aoki, 1993; Lankford & Targett, 1994). Until recently, few studies have simultaneously analysed the effects of both biotic and abiotic factors on estuarine fish assemblages, either internationally (e.g. Akin et al., 2005; Selleslagh & Amara, 2008b; Selleslagh et al., 2009; 2011) or within New Zealand (Saunders, 1999; Morrison et al., in prep, Morrisey et al., 2007; 2010; Usmar, 2009). Consequently, large information gaps exist with regard to biotic interactions, particularly predator-prey interactions and prey availability (Stål et al., 2007).

Given the increasing focus on ecosystem based management, with protection of the sequential chain of habitats required by juvenile and adult fish a key tenet, knowledge of the relative importance of both biological and environmental drivers upon fish assemblages, particularly juvenile marine migrants within estuaries is pertinent for resource managers.

The Manukau Harbour has gained a high profile environmentally, due to water quality concerns associated with the impacts of increasing urbanization, land reclamation and industrial/sewage discharges, particularly after the opening of the Waste Purification Works at Mangere in 1960 (Menzies & Duder, 1987; Matthews et al., 2005). As a consequence, long term monitoring programs have been undertaken by the Auckland Regional Council (see reviews of Hewitt & Hailes, 2007; Hailes & Hewitt, 2012; Kelly, 2007). In addition, numerous studies on the effects of pollution (Aggett & Simpson, 1986; Pridmore et al., 1990; Holland et al., 1993; Matthew et al., 2005), phytoplankton dynamics (Vant & Budd, 1993; Vant & Safi, 1996; Vant et al., 1998; Cahoon et al., 1999; Cahoon & Safi, 2003), chemistry, sediment transport and hydrology (e.g. Henriques, 1977; Dolphin et al., 1995; Green & Bell, 1995; Green et al., 1997; Bell et al., 1998; Pritchard et al., 2008) have been published.

There have also been extensive studies of benthic community processes of the intertidal sand flats, describing macrobenthos spatio temporal patterns in the Manukau Harbour (e.g. Grange, 1977; Pridmore et al., 1990; Thrush, 1991; Hewitt et al., 1997; Thrush et al., 1997; Taylor, M., 1998; Hewitt & Hailes 2007) and to identify processes responsible for these patterns at the population (Cummings et al., 1993, 1996; Turner et al., 1997; Thrush et al., 1997; Hewitt et al., 1996, 1997) and community level (Thrush et al., 1992). Limited work on the subtidal soft sediment communities has been previously described by Powell (1937), Grange (1979) and Henriques (1980).

By comparison, limited research has been undertaken on the juvenile fish assemblages within the harbour. Morrison et al., (2002) quantified the small fishes associated with a tidal flat within the Pahurehure Inlet to assess spatial patterns of abundance over tidal and diurnal cycles, in addition to a seven year monitoring program of small fish distribution from two sites (Morrison, unpubl. data; Francis et al., 2005). Two further studies have documented yellow-belly flounder (*Rhombosolea leporina*) diet, distribution and abundance (Pearks, 1985; Mutoro, 2001).

The Manukau Harbour's cultural significance to the Maori people is very strong and the harbour is extensively utilized as a recreational fishery (pelagic and shellfish) in addition to supporting local commercial fisheries, especially for flounder and mullet (Morrison et al., 2002). However, there is a perception that estuarine fish stocks are declining due to overfishing and environmental degradation (Waitangi Tribunal, 1989).

2.2 Aims

The aims of this chapter were to (a) describe and quantify changes in the inshore juvenile and small fish community (<125mm FL) of the Manukau Harbour along a 20 km estuarine environmental gradient over two seasons, and (b) attempt to explain changes by reference to biotic (prey community, predators) and abiotic factors (e.g. currents, depth, suspended sediments, substrate type). Ordination techniques used in this study allow the simultaneous species-specific analysis of the fish community and associated environmental data to clarify potential control mechanisms in structuring these assemblages. Determining the relative magnitude of the contributions made by the various abiotic and biotic factors can facilitate the ongoing management and conservation of this harbour.

2.3 Methods

2.3.1 Study Location

The Manukau Harbour is New Zealand's second largest estuary (340km^2), and is adjacent to New Zealand's largest city, Auckland (Fig. 2.2). It opens into the Tasman Sea, a high energy exposed coast, via a narrow (2.2km) 30m deep channel with a bar 5km offshore (Menzies & Duder, 1987). The inner harbour has four main channels, subdivided by extensive shallow banks, of which 40% (145km²) are exposed mudflat at low spring tides. Tidal channels range in depth from 5m in the upper harbour to 40m at the mouth.

The catchment area (850km²) is only about twice that of the harbour itself, so freshwater runoff is negligible (Vant & Williams, 1992). Consequently, harbour waters are typically well mixed, with salinity only varying by 3% between the heads and innermost shore. Temperatures change little horizontally or vertically, with values in the range of 10-21° C (Heath et al., 1977; Vant & Smith, 1992; Bell et al., 1997; Green et al., 1997).

Tides are semi-diurnal and mesotidal (Green et al., 1997). Tidal range in the harbour is amongst the highest in New Zealand, exceeding 4m during spring tides with a neap range of less than 2m (Hume et al., 1992; Heath et al., 1977). The large tidal range generates current speeds that can exceed 2m s^{-1} (1m above the bed) in parts of the channels. Peak spring tide currents on the mid–upper intertidal flats are < 0.25 m s^{-1} (Heath et al., 1977; Bell et al., 1997; Swales et al., 2004). Estimated residence times for water in the upper reaches of the three main channel systems (Papakura Channel, Waiuku Channel, and the Wairopa/Purakau system), under mean annual hydrological conditions, are 12, 13 and 26 days respectively (Vant & Williams, 1992).

Strong winds blowing over large fetches (up to 25km at high tide) can generate waves higher than 2m in places (Dolphin & Green, 1997). Sediment resuspension on the intertidal flats is largely controlled by waves resulting in formation of a rippled seabed over much of the harbour (Green et al., 1977; Swales et al., 2004). Movement of predominantly fine sand seabed sediments is primarily by tidal currents in the harbour channels and over much of the banks (Menzies & Duder, 1987). As a result of the strong currents driven by tidal flows and wind mixing, water clarity in the Manukau Harbour is usually low, especially in the upper reaches (Vant, 1991; Green & Bell, 1995; Dolphin et al., 1995; Bell et al., 1997; Cahoon & Safi, 2002).

Sediments of the Manukau Harbour include extensive areas dominated by sand (64%), muddy sand (19%), shell hash-sand (6%), sandy mud (5%), and mud (0.5%), with additional small proportions of mixtures of these types (Gregory et al., 1994). Typically, sandy sediments predominate in the middle of the harbour, with higher current flow and wave activity, with higher frequencies of shell hash and coarse sediments in the deeper channels. Sandy muds characterize the intertidal flats (Grange, 1977; Dolphin et al., 1995).

Sampling for this survey was undertaken within the Pahurehure Inlet (Fig. 2.2), extending 20km down the Papakura Channel to the lower harbour area at Karore Bank. Pahurehure Inlet is bordered by a mixture of residential development and industry on its northern side. Auckland’s southern motorway crosses its eastern boundary and rural land lies to the south. Environmental pressures on this inlet are increasing with expanding urban development planned for the southern side.

Wind fetch and associated wave action are limited due to the enclosed nature of the inlet compared to the open expanses of the central harbour. The main channel is 5-10m deep, with strong current flows of 0.9m s^{-1} at neap (Hume, 1979) and is bounded by extensive tidal banks, composed predominantly of soft deep mud. Mangroves and salt marsh are limited in extent, occupying a small proportion of the inlet (Morrison et al., 2000).

Pahurehure Inlet opens out into the Papakura Channel which is characterized by increasing depth, higher current speeds (Bell et al., 1998) and wave activity. Higher frequencies of sand, shell hash and coarse sediments occur in the deeper channel areas (Grange,1977; Dolphin et al., 1995) along with increasing biogenic structure such as horse mussels and sponges (pers. obs.).

2.3.2 Sampling Methods

2.3.2.1 Beam trawl

A small beam trawl, based on the design of Hamer et al., (1998), and Morrison & Carbines (2006) was used to sample small benthic associated fishes. This consisted of a 4 m beam, from which was suspended a trawl net composed of 9mm mesh, 3.0m wide, with a 6m deep cod-end.

Sampling was undertaken within a 2.5 hour window either side of high tide rather than low tide to ensure full stomach contents. Six sampling locations (1 km² blocks) covering 20km were spread down the Papakura Channel (Fig. 2.2). This encompassed an environmental gradient running from the upper harbour through to a mid to lower harbour area. At each location, 4 beam trawl shots were placed across the inter-tidal flats, and adjacent subtidal channel bank, and the channel floor. Sampling was undertaken during autumn (March-April) and repeated in spring (October–November) 2003. Hereafter, these two sampling periods will be nominally referred to as ‘seasons’. Whilst it is acknowledged that demonstrating true seasonality requires two to three years of sampling to show consistent patterns, this was not logistically possible in this survey.

Within each location, beam trawl tows were made along the depth contours. Tows were assigned adaptively in the field, as little or no bathymetric/habitat information was available to pre-assign tow allocations. For each shot, start and end point coordinates, and associated water depths were recorded. Tows were undertaken for four minutes at a speed of 1.5-2 knots, measured from the time when the warp came up hard on the trawl to when hauling commenced. Average distance towed was 295m, although the true distance towed across the seafloor will have been slightly greater due to the lag between commencing hauling, and the trawl lifting clear of the seafloor. A 5:1 warp-to-depth ratio was used in water depths of less than 10m, reducing to 4:1 in deeper waters (maximum depth 25m).

Catch was sorted, identified to species level, and fork length was measured (±1mm). Fish were immediately placed in an ice slurry, then preserved in 10% buffered formalin. The diets of all fish collected by trawl were examined except when there were more than 20 individuals of a species collected at a site on one sampling date. In these cases, a subset of 20 animals ranging in size from the smallest to the largest was sub sampled for gut content analysis (Edgar & Shaw, 1995a). If there were two clear age cohorts then 20 individuals were subsampled from each cohort. Fish utilized for gut analysis were injected with 10% buffered formalin to preserve prey items in the field.

In the laboratory, foreguts were removed and the contents identified to species level where possible under a dissecting microscope. In order to estimate biomass and directly compare the size-distribution of ingested prey with invertebrate size-classes recorded during the benthos sampling program (see section 2.3.2.3 below) animals were allocated to sieve size-classes by eye using a graticule in the microscope and a reference collection consisting of a mixture of species retained by different sized sieves as per Edgar et al., (1994). The percent occurrence of detritus, macroalgae and sessile animals (sponges; bryozoans) was estimated by volume from the cover of these dietary components across the bottom of a Petri dish.

2.3.2.2 Physical and biological parameters

At each station, a number of physical and biological variables were recorded (Table 2.1). To quantify changes in benthic community structure down the gradient, distribution and abundance of macrofauna at each of the six locations were determined from 12 spatially matched Smith-MacIntyre grab samples, covering a surface area of $0.11m^2$ to a maximum depth of 20cm. The sample was sieved on a 1mm mesh, and the retained fauna was preserved in 10% buffered formalin solution.

In the laboratory, the samples were transferred to 70% isopropyl alcohol and stained with 6.2% rose-bengal. Samples were washed through a log series of sieves (1.0, 1.4, 2, 2.8, 4, 5.6, 8, 11.2, 16, 22 mm mesh sizes) using the methods described by Edgar (1990a), and the abundance of each size class of each taxon was recorded.

2.3.2.3 Estimation of faunal biomass and productivity

The biomass of benthic invertebrates between 1 and 5.6mm (0.5-5.6mm for gut content) was estimated for each sample by multiplying the number of animals retained by sieves of different size by the mean ash-free dry weight (AFDW) of animals for each sieve size-class, as calculated from the regression equations listed in Table 2 of Edgar (1990a). A separate equation was used for caprellids as recommended by Edgar (1990a). For gut analysis, AFDW values for plankton were calculated from Newcombe (2009).

Productivity of the benthic fauna was estimated using the biomass estimates for each animal and the equation $P = 0.0049 * B^{0.80}\ T^{0.89}$ (Edgar, 1990a), which relates daily macrobenthic productivity P (µg d^{-1}) to ash-free dry weight B (µg) and water temperature T (ºC) (Edgar, 1990a). Estimation of the mean daily rates of production of animals provides an index of contribution to the flux of energy and materials (Edgar, 1990a).

Due to the ineffectiveness of the grab sampler for capturing larger crustaceans, the abundance of the mysid shrimp (an important prey item for estuarine fish) captured during beam trawling was used as an index of their abundance and taken into account as a biotic factor in the environmental matrix for spring. In addition, the presence of biogenic structure such as pinnid horse mussels (*Atrina zelandica*), sponges, green lipped mussels (*Perna canaliculus)* and hydroids was also ranked (i.e. absent, present, common, very abundant) from the beam trawl by-catch for the spring sampling as a qualitative estimate of habitat heterogeneity.

To estimate the density of large (≥100mm) piscivores, at each location, three multi-panel gillnets consisting of four joined mesh panels of 12m each (mesh size 1.5, 2.5, 3.5 and 4.5 inches), with a drop of ~2-2.5m, were set in the channel just before darkness, and left to fish for 3-3.5 hours. On retrieval, all fish were removed, placed in slurry ice bins, and measured back at the laboratory.

Biomass was estimated for those species considered potential predators of small fish from length weight regressions and included in the environmental matrix. Fish length-wet weight equations were taken from Taylor & Willis (1998); Hartill & Walsh (2005) and Fish base (2008).

Surficial core sediment samples were collected from the top 5cm of each grab sample for sediment grain size analysis. Samples were predigested with 5% hydrogen peroxide and agitated for 24 hours to remove organic matter. Sediments were disaggregated by ultrasonic dispersion for 10 minutes prior to analysis. To ensure mixing, samples were then aerated via a tube from the bottom of the sample with an Air Cadet vacuum pressure aerator, and flow rate was adjusted according to the particle size in the sample. A sub sample was drawn off in a pipette and placed into a Malvern Mastersizer 2000 Particle Analyser to determine volumetric particle size distributions. Mean and maximum current velocities for each location were derived from the DHI MIKE3 FM HD and MT model of the region (Pritchard et al., 2008).

To quantify total suspended sediment (TSS) load, water samples (1000ml) were collected over a full tidal cycle using a Van Dorn sampler at each of the 6 locations during the spring sampling. Within each location two samples were collected from the channel/bank area, 1m above the sea bed and at 1m from the surface respectively, with one depth sample being collected from the intertidal flats 1m above the sea bed. Intertidal sampling position was adjusted according to the tide, following the low tide boundary. In the laboratory water samples were filtered through acid-washed, dried and pre-weighed polycarbonate membranes (0.45 µm) using plastic, acid-washed, vacuum filtration equipment. After filtration, the membrane was re-dried at 60ºC until a constant weight was reached, and the membrane re-weighed to give the weight of total suspended solids in the volume filtered. Water clarity was measured concurrently at the channel/bank and intertidal sites using a 25 cm black-and-white secchi disc.

2.4 Data Analysis

2.4.1 Fish

In order to standardize the area sampled between beam trawls, fish abundances were expressed as number of individuals per 100m². Spatial and seasonal variations in total abundance and individual species abundance were fourth-root transformed to prevent the analysis being dominated by highly abundant species, prior to the analysis in PRIMER 6.0 (Clarke & Warwick, 2001). As the primary interest of this study was assessing potential environmental factors affecting the distribution of juvenile (0+ year class) and small fish, individuals measuring over 125mm FL were excluded prior to analysis. Analyses of similarities (ANOSIM) was employed to test whether differences in fish assemblages across season, stations and habitats were significant, with Global R values obtained using 999 permutations. Similarity percentages (SIMPER) using the Bray-Curtis similarity matrix, were used to determine which species typified the fish assemblages for each station/habitat/season and made the greatest contribution to any dissimilarities between them. Multi-dimensional scaling (MDS) provided an ordination for visual assessment of differences in fish assemblage composition across the 6 locations by season.

2.4.2 Dietary analysis

To ascertain resource utilisation of the benthos, prey items consumed by the fish assemblages were grouped into 9 general categories (Table 2.2) for analysis. Resource utilisation was estimated by comparing the prey consumed by the fish to the potential food resources present in the habitats. Prey species accounting for ~95% of total prey biomass was used as the criteria to select a subset of potential food resources available in the benthic habitats to include in the Principal Component Analysis (PCA). Biomass estimates of potential predators caught in the gill nets were also estimated from length weight regressions to include in the environmental matrix.

PCA was used to assess differences in environmental variables across sites and times, including prey/predator biomass (Table 2.1). Initial analyses included the entire environmental data set. PCA models were then applied using the best subset of environmental variables that explained the most variation. Two analyses have been presented: two seasons with identical variables, two seasons combined; an additional analysis for spring where a greater range of explanatory variables were collected in the field. These included suspended sediments, secchi depth, presence of biogenic structure and mysid abundance. Variables were examined for normality and were log transformed where appropriate.

The relationship between fish assemblages, individual species, and environmental variables was displayed by superimposing the environmental data as symbols scaled to size, to the data value on the biotic MDS and by the BEST procedure (Clarke & Warwick, 2001) which finds the best match between the multivariate among-sample patterns of the biotic assemblage and the environmental variables. In addition, relationships between fish assemblages, individual species and environmental variables, were assessed using canonical correspondence analysis (CCA) implemented using CANOCO (ter Braak & Smilauer, 2002). Contributions of environmental explanatory variables were assessed using automatic forward selection. Significance was determined at the $P<0.05$ level using a Monte Carlo permutation test set at 499 permutations.

2.5 Results

2.5.1 Environmental Parameters

2.5.1.1 Physical

Most of the environmental parameters measured varied as a gradient along the longitudinal gradient from the inner (Pahurehure Inlet) to the outer reaches of the estuary (Papakura Channel/Karore Bank). Depths ranged from 1.5 to 25m along the gradient. Current speeds generally increased down the gradient for all habitats with peak velocities occurring in the main channel of 0.53 m s^{-1} (Fig. 2.3 & 2.4). Overall, intertidal and channel sites recorded the greatest variability between average and peak tidal current speeds.

There was strong temporal and spatial variability in the concentrations of suspended sediment in the water column (Fig. 2.5) Suspended sediment was highest and most variable at the inner, intertidal sites M1-M3, reaching a maximum of 188.4 g m^{-3} over the ebb tide associated with a turbidity fringe. Suspended sediments were lowest and least variable over flood tide at outer channel sites M4-M6, averaging 14.96 g m^{-3} (SE+/- 4.98). Suspended sediment concentrations showed little variation in the top 1m of the water column both temporally and spatially.

Conversely, water clarity generally increased going down the gradient (Fig. 2.6), especially over flood tide with secchi depth ranging from an average of 51 to 120cm at sites M1 to M6. Water clarity was the most variable at the inner site M1 over ebb tide between intertidal (17cm) and channel (100cm) locations.

Profiles of surficial sediments showed a trend of increasing particle size towards the mouth of the estuary, grading from fine silt/clay (<0.64µm) to fine/coarse sands (0.64-500µm), with visual observations confirming increasing amounts of shell/hash in the channel sites (Fig. 2.7). Sediments in the enclosed inner sites (M1-M2), particularly the intertidal areas were dominated by soft muds (58%) with some fine sand (33.8%) in addition to abundant wood and leaf fragments. There was a rapid transition from fine muds to fine-coarse sands (~70%) at the mid harbour site M3, with increasing proportions of coarse sand (39%) at the lower sites particularly in the channels associated with increasing current speed.

2.5.1.2 Biotic

Benthos

A total of 34,945 individuals belonging to 235 invertebrate taxa were collected from the benthic sampling. Total abundance of benthic organisms varied seasonally, with the spring total almost 50% higher than that of autumn (Fig. 2.8). Highest densities were recorded at mid harbour (M4-M5), with a peak of 19,576 indiv. per m^2, primarily comprised of the small bivalve *Nucula hartvigiana*. Annually, the most abundant benthic taxa were bivalves (60%), polychaetes (22%), gastropods (10%) and crustaceans (2.6%). Highest species richness occurred at the lower harbour sites with increasing biogenic structure (M4-M6), particularly the bank and channel habitats, with site M5 recording a maximum of 105 species from one grab sample (Fig. 2.9). Estimated biomass and productivity were higher in the mid harbour bank/channel habitats (M3-M5), particularly for spring averaging 167.58 ± 81.55 ($\mu g\ m^2\ d^{-1}$) and 22.87 ± 14.95 ($g\ m^2\ d^{-1}$) respectively (Figs. 2.10 & 2.11). These sites were dominated by the gastropods, *Maoricolpus roseus* and *Zethalia zelandica* and the bivalve *N. hartvigiana*. In contrast, the inner sites (M1-M2) possessed low faunal biomass and productivity averaging 0.94 ± 0.11 ($g\ m^2\ d^{-1}$) and 13.30 ± 1.73 ($\mu g\ m^2\ d^{-1}$) for both seasons. Polychaete species such as *Heteromastus filliforms, Timarete anchylochaeta* and Nereid sp., along with the mud crab *Helice crassa* and the penaid shrimp *Pontophilus australis* characterized the inner turbid sites.

Macrocrustaceans

During the present study 3 species of shrimp belonging to the family Mysidae were sampled over spring: *Tenagomysis novaezelandiae, T. macropsis* and *T. chiltoni.* High abundances were recorded at the inner sites (M1, M2), although catch rates were highly variable (Fig. 2.12). Densities reached a peak of 12,682 individuals per $100m^2$ at site M2. Mysids were in low abundance or absent in the lower estuary. Modest numbers of *Pontophilus australis* were recorded at sites M1 and M3, particularly in the bank/channel habitats. Low numbers of *Palaemon affinis* were sampled throughout the estuary.

Variation in predator guild composition and abundance

A total of 28 species of fish were caught in the gill nets (≥100mm FL), totalling 1,177 individuals (Table 2.3). Eight species were identified from prior research (Williams, 2009; Morrison pers. comm.) as potential predators of juvenile fish, hereafter referred to in this chapter as the predator guild. These include, in decreasing order of abundance, kahawai *Arripis trutta,* rig *Mustelus lenticulatus,* snapper *Pagrus auratus,* trevally *Psuedocaranx dentex,* school shark *Galeorhinus galeus,* gurnard *Chelionichthys kumu*, barracouta *Thyrsites atun*, and the hammerhead shark *Sphyrna zygaena.* Shark species were almost exclusively (99%) comprised of juvenile ‘pups’, while the remaining species collected (i.e. snapper; kahawai; gurnard; barracouta) were predominately adults.

Predator guild density increased going down the gradient, with 72% of the total catch collected from the lower harbour (M4-M6), and was associated with higher water clarity, depths and currents. Similarly, 63% of the estimated total predator guild biomass was sampled from the lower estuary (Fig. 2.13). While total biomass varied little between seasons, distribution within the estuary differed, with spring biomass reaching a peak of 10,083g per gillnet at the outer harbour site M6, whereas autumn biomass peaked mid harbour (M3) at 5,567g per gill net. Composition of the predator guild also shifted, with higher densities of kahawai and sharks recorded during autumn, with snapper, trevally and gurnard predominating in the spring (Fig. 2.14). Rig was the only shark in autumn to be found in higher numbers at the inner site (M2) reflecting its preference for benthic feeding in the soft sediments.

Biogenic structure

Biogenic structure was largely confined to the lower harbour sites (M4-M6), particularly for the bank habitats. Site M4 recorded the highest amount of biogenic bycatch for both bank and channel habitats with horse mussels, hydroids and sponges predominating (Table 2.4). Green lipped mussels were recorded only at site M5.

2.5.1.3 Environmental PCA

Patterns of station relationships revealed by PCA (environmental matrix subset) for both seasons with identical variables, show a habitat gradient going down the harbour which differed little between the seasons and was driven largely by physical factors (i.e. currents, depth and substrate type) (Fig. 2.15; Table 2.5 A,B). Predator and individual invertebrate prey biomass categories were deleted after the initial analyses, as these factors did not add any explanatory power to the variance.

The first two principal components from the combined season environmental data explained 81% of the variance (PC1 explained 54.4% and PC2 26.7%; Fig. 2.16A, Table 2.5 C). Results of the first component indicated a positive correlation with fine sands and higher current speeds associated with the lower harbour sites (M4-M6), while a positive correlation with clay/silt characterised the inner harbour sites (M1-M3). Within site habitat variation also showed some separation along the second PC axis, with most of the channel habitats associated with depth and coarse sands, while the majority of intertidal and bank habitats were characterised by fine sand, silt and currents.

The first two components of the spring PCA analysis, including additional variables (suspended sediments; secchi; mysid abundance; summed invertebrate prey biomass; biogenic structure) explained 50.5% of the variation (Fig. 2.16B; Table 2.6). The first component was positively associated with current speeds, fine sands and water clarity, and negatively correlated with suspended sediments, substrates <64µm and mysid abundance. The second component was positively correlated with variation in water clarity (standard error), and negatively correlated with invertebrate prey biomass and fine sand. There was a clear separation of sites along the first PC axis with upper harbour sites (M1-M3) associated with mysid abundance and suspended sediments, while lower harbour sites were associated with increased water clarity, currents and fine sands. Sites also showed some separation along the second PC axis, with higher invertebrate prey biomass associated with the mid harbour sites M3-M5, particularly for site M4 which also had high levels of associated biogenic structure.

2.5.2 Fish Assemblage: abundance and composition

Overall, 8,592 fish from 27 species including one species of squid (*Sepioloidea pacifica*) were collected over the autumn and spring sampling (Table 2.7). The three most abundant families, based on total abundance were Gobiidae (42.6%), Pleuronectidae (27.3%) and Clupeidae (15.3%) accounting for 85 % of the total catch. Ninety three percent of all individuals were juveniles or adults of small sized demersal species with seven species comprising 98% of the total catch. These included in order of respective importance, exquisite goby *Favonigobius exquisitus*; yellow-belly flounder *Rhombosolea leporina*; anchovy *Engraulis australis*; sand flounder *Rhombosolea plebeia*; mottled triplefin *Grahimina capito*; speckled sole *Peltorhamphus latus* and snapper *Pagrus auratus* (Figs. 2.17 & 2.19). Individuals of the remaining 15 species were captured occasionally in modest numbers.

Ecological guilds, were dominated by marine migrants, comprising estuarine opportunist (MMO) species 45%, followed by marine estuarine dependent (MMD) 17%, and marine stragglers (17%). Resident species made up 14% (Elliot et al., 2007; Fig. 2.1). Smelt was the only anadromous species.

Spatio temporal distribution

Fish abundance showed strong temporal and spatial variability along the estuarine gradient (Fig. 2.19A). Density declined from the upper to lower harbour, ranging from 56.2 to 1.7 individ. per 100m² (M1-M6) for both season and habitat. Abundance peaked in the spring, with 88% of the total catch collected at the sheltered inner sites M1-M3 with associated lower current speeds and higher turbidity. M1 recorded the highest and most variable density (30 ± 1.5 to 52 ± 13.7 individ. per 100 m²) for the intertidal and bank locations respectively. Similarly, species diversity declined gradually down the estuary, averaging 5.5 ± 3 (M1-M6) for both seasons (Fig. 2.19B). A similar trend was evident in spring with the exception of the mid harbour site M4, which recorded higher diversity for all habitats peaking at 8.75 ± 0.85 on the bank.

Temporal and spatial variations in abundance and fish length of dominant species

A gradual transition was observed for the dominant twelve species down the gradient. Fish assemblages in the enclosed upper sites (M1-M2), particularly the intertidal/bank areas were dominated by exquisite goby, anchovy and yellow-belly flounder, along with small numbers of kahawai and spotted stargazer (Fig. 2.20). These species exhibited a strong recruitment pulse (0-50mm FL) in the spring. In contrast, a second assemblage of species consisting of mottled triplefin, and marine migrants such as sand flounder, speckled sole, snapper, gurnard and jack mackerel were more abundant in the channel/bank habitats over autumn, with a more widespread distribution down the gradient. In the spring, higher densities of snapper, gurnard and jack mackerel were found in the lower estuary (M4-M6), with related higher water clarity and depth, while sand flounder and speckled sole remained in low numbers throughout the estuary. Pink cod was almost exclusively confined to the channels and was only collected in the spring. Other marine opportunists primarily caught in the spring sampling included the snake eel (*Ophisurus serpens),* long-finned worm eel (*Muraenichthys breviceps)*; leather jacket (*Parika scaber),* *Sprattus spp.* and the bumblebee squid (*Sepioloidea pacifica).*

There was evidence for limited seasonal movement within sites from the channel to bank habitats for species such as the exquisite goby, mottled triplefin and yellow-belly flounder (Fig. 2.21). However, there was no evidence for seasonal movement of fish going down the gradient (Fig. 2.22).

2.5.3 Prey Utilisation

Ninety four prey taxa were identified from the stomach contents, with the majority of fish feeding primarily on benthic epifauna rather than infaunal or planktonic prey (Fig. 2.23). A more detailed analysis of individual species diet, including ontogenic changes, is covered in Chapter Four. Prey assemblages were dominated by epifaunal crustaceans, with pericarids (mysids cumaceans, gammaridean amphipods) and decapods being the most abundant prey items for 82% of the fish species collected (<125 mm FL). Zooplankton feeders included anchovy, jack mackerel, sprat and yellow-eyed mullet, with the latter two species being represented by only one or two individuals. Calanoid and harpacticoid copepods (*Paracalanus indicus; Euterpina acutifrons*) were the most important food items for these species, especially in autumn.

No exclusively piscivorous fish species were collected, however estuarine and spotted stargazers, and the snake eel had significant proportions of exquisite goby in their diet. Few species fed predominantly on infaunal animals such as polychaetes and bivalves, apart from mottled triplefin and spotty, the latter of which only comprised ten individuals. Bivalves and bivalve siphons were taken in significant numbers by the two flounder species (yellow-belly & sand flounder), while speckled sole consumed high numbers of cumaceans. Snapper utilized the widest prey spectrum, feeding on bivalves and polychaetes, in addition to gastropods, the ophiuroid *Amphiura rosea* and crustaceans, both epifaunal and pelagic.

The fish assemblage showed a significant seasonal dietary shift in spring, with mysids strongly dominating the diets (contributing 66 % total dietary biomass) of fish species with high recruitment into the inner harbour (sites M1-M2), especially for the intertidal and bank habitats. Mysids dominated (>50%) the diet of anchovy, yellow-belly flounder, spotted stargazer, exquisite goby, trevally, kahawai, pink cod and sprats. While the proportion of other epibenthic crustaceans dropped from ~50% in autumn to only 17.2% biomass in spring. Conversely, fish species found in the deeper bank/channel habitats from the mid to lower harbour (snapper, speckled sole and sand flounder) did not have highly specialized diets.

2.5.4 Fish assemblages and environmental influence

MDS ordination plots on the overall fish assemblage data shows the inner sites (M1, M2) clustering together on the left hand side of the first axis, while those from the lower sites tended to cluster on the right side (Fig. 2.24A). ANOSIM tests revealed significant differences between sites (R=0.239, P<0.01), with the exception of inner sites M1-M2 (Table 2.8). Fish assemblages were also significantly different between seasons (R=0.125, P< 0.01) and habitats (R=0.126, P<0.01; Table 2.9, Fig. 2.24B). SIMPER analysis indicated that the same seven to eight species (>5% contribution) collectively contributed ~70% towards the dissimilarity between the assemblages going down the gradient (Table 2.10). Overall, contributions of individual species to the dissimilarities were generally small (<15%). These included, (in order of influence) exquisite goby, anchovy, sand flounder, yellow-belly flounder, speckled sole, mottled triplefin and snapper, while species such as jack mackerel, gurnard and pink cod contributed ~5% towards the dissimilarity for the mid to lower harbour sites only. Similarly, the same suite of species contributed to within site habitat differences (Table 2.11) and seasonal variability (Table 2.12), with the relative abundance of each species varying, rather than community structure from a larger species pool.

Depth and current velocity were identified by the BIOENV procedure as the primary environmental variables influencing fish assemblages, having the highest correlation for autumn (R=0.49; p<0.01); spring (R=0.40; p<0.01) and combined seasons (R=0.41; p<0.01). Additional variables for the spring analysis having a significant influence in declining order, included combinations of current velocity, depth and water clarity (R=0.39; p<0.01), in addition to suspended sediment loads (R=0.39; p<0.01; Table 2.13).

The CCA ordination based on species abundances, and the environmental data matrix indicated that the first two axes significantly explained ~72% of the fish assemblages for autumn and ~55% for spring respectively (Monte Carlo permutation tests, n = 499; P= 0.002; P=0.002, Table 2.14). Figure 2.25 indicates the relative environmental preference of fish species for autumn and spring. These are largely consistent with the previous analysis, with depth, currents and substrate composition being identified as significant habitat variables for both seasons. The spring analysis (additional variables) also identified, in order of importance mysids, biogenic structure, suspended sediments (minimum) and water clarity, as significantly affecting fish distribution (Table 2.14). These factors were also retained in the forward selection procedure of CCA as significant contributors to the ordination (p< 0.05).

Both autumn and spring vectors indicated species such as yellow-belly flounder, spotted stargazer, anchovy, exquisite goby and kahawai were correlated with muddy substrates, lower suspended sediments loads and higher mysid abundance. While those species clustered on the right side of the autumn ordination plot, such as jack mackerel, snapper and speckled sole were weakly associated with increasing depth, current speed and coarser substrates. The spring ordination indicated a positive shift in the association of these species with increasing water clarity, biogenic structure and depth. Sand flounder showed no positive correlations, being found in a wide range of conditions for both seasons. Of the remaining species, most were associated with increasing depth and coarse substrates. However, too few individuals were sampled to infer robust interpretation.

2.6 Discussion

2.6.1 Fish assemblage composition

Despite its proximity to Auckland city, the study area supported relatively high abundances of demersal and semi-pelagic inshore fish compared to other northern estuaries (Francis et al., 2005). Gobiidae, Pleuronectidae and Clupeidae were the most important families. At the species level, the fish fauna was largely dominated by the exquisite goby, followed by yellow-belly flounder, anchovy, sand flounder, mottled triplefin, speckled sole and snapper. Dominance by a few species is a pattern generally observed in most temperate shallow water communities (New Zealand: May, 1979; Knox, 1983; Jellyman et al., 1997; Saunders, 1999; Morrison et al., 2002; Francis et al., 2005; Australia: Bell, et al., 1984; Potter et al., 1990; Blaber et al., 1995; Europe: Selleslagh & Amara, 2008a, b; Selleslagh et al., 2009; USA: Thayer et al., 1987).

The low representation of mugilids (e.g. yellow-eyed mullet) in this study, a ubiquitous estuarine species in other New Zealand estuaries (Jellyman et al., 1997; Morrison et al., 2002; Francis et al., 2005) is thought to be an artefact of net avoidance due to slow towing speeds of the beam trawl, also reported by other researchers (Martinho et al., 2007). In addition, yellow-eyed mullet are found in higher abundances on the very shallow intertidal flats during daylight hours (Morrison et al., 2002), which precludes beam trawling

2.6.2 Estuary fish use

The Manukau fish community showed strong temporal and spatial variability along the estuarine gradient with many species having discernible habitat affinities. Overall numbers declined from the sheltered, muddy, turbid areas in the upper Pahurehure Inlet to the clearer, sandy areas of the lower harbour. Seasonal variations of the community structure mainly reflected differences in the times of recruitment of the more abundant species (e.g. yellow-belly flounder; sand flounder; exquisite goby). There was a strong recruitment pulse in the spring, with 88% of the total catch being collected within Pahurehure Inlet during that seasons sampling, particularly within the sheltered, intertidal/bank areas, showing its high importance as a nursery area for some species. This is consistent with other New Zealand studies (Webb, 1973; Colman, 1974a, b; Davenport, 1979; Roper & Jillett, 1981; Forester, 1983; Pearks, 1985; Roper 1986; Saunders, 1999; Morrison et al., 2002) and is thought to be related to annual production cycles. It has also been suggested that some species utilize the highly productive inner harbour areas during the spring, with numbers tending to decline over winter due to predation and emigration of larger fish to deeper waters during cooler months (Colman, 1978; Paul, 2000; Selleslagh & Amara, 2008b).

There was little evidence of seasonal ontogenetic habitat shifts for juveniles (as evidenced by size) going down the gradient in the Manukau (Fig. 2.22). However, the larger size cohorts (>125mm FL) for species such as yellow-belly flounder, sand flounder and speckled sole (which were effectively caught in the beam trawl), were found predominantly in the deeper channel areas in autumn suggesting an ontogenetic shift depth-wise at the location scale. Lower numbers were recorded in spring. This suggests emigration out of the estuary for spawning offshore over winter as observed in other New Zealand studies (Colman, 1973; 1974a, b; 1978). Although seasonal changes noted in this study spanned only one year, longer term annual monitoring down this same environmental gradient (2001–2007) has found these general fish assemblage patterns to be largely stable over the time period measured (Morrison unpubl. data).

The high proportion of juveniles in the system (93% of fish captured were juveniles or adults of small sized demersal species) is similar to that typically found in other temperate estuaries of the northern hemisphere (e.g. Cyrus & Blaber 1992; see review Elliot et al., 2007; and within New Zealand, e.g. Webb, 1973; Kilner & Akroyd, 1978; Roper & Jillett, 1981; Saunders 1999; Morrison et al., 2002; Francis et al., 2005). Marine migrants were the dominant ecological guild, comprising 45% estuarine opportunist, 17%, marine estuarine dependent and 17% marine stragglers. These results are reflective of New Zealand estuaries having no true estuarine species (McDowell, 1976; Jellyman et al., 1997). These results also confirm the important role played by estuaries for those marine species which utilize this habitat as a nursery and/or as a migratory pathway. This may reflect opportunistic utilisation of this near shore environment as a refuge from predators and/or increased feeding possibilities (McLusky, 1989; Blaber, 1997; Lazzarie et al., 1999; Selleslagh & Amara, 2008a; Selleslagh et al., 2009).

2.6.3 Linking environmental variables to fish communities

Changes in the nearshore fish assemblage structure in the Manukau was primarily correlated with physical environmental variables. Depth and current velocity were identified by BIOENV and CCA analyses as the dominant environmental gradient for both seasons followed by water clarity and suspended sediments loads for the spring analysis (additional variables). Overall, this agrees well with information gained from the PCA of habitat and environmental variables (Figs. 2.15, 2.16). The only exception was the spring PCA (additional variables), where depth had lower explanatory power. This may be a result of high correlation between some of the added variables, making it difficult or impossible to distinguish between their effects using this method (Clarke & Ainsworth, 1993).

Results are consistent with other surveys both in freshwater (Baltz et al., 1987; Yu & Le, 2005) and marine estuarine communities (Jaureguizar et al., 2003; 2006; Martino & Able, 2003; Courrat et al., 2009) whereby depth was a dominant variable, co-varying with temperature and salinity. As water depths only ranged from ~1-25m across the locations sampled in the Manukau, depth per se was unlikely to be a direct driver of fish distribution and abundances; but was likely to co vary with other factors such as species interactions associated with changes in depth, such as availability of shelter, protection from large predators in shallow water, and increases in food availability for pelagic feeders in deep water (Kupschus et al., 2001). The effects of the high current speeds generated by tidal flow in estuaries with a large tidal range, as in the Manukau (4m) has been documented as being a significant factor in determining juvenile fish communities in other studies with juvenile fish preferring calm areas for settlement and growth (Ross & Epperly, 1985; Blaber et al., 1994; see review Blaber, 1997). The Pahurehure Inlet, with its enclosed nature, shallow waters and inherent hydrographic stability make it a very suitable settlement area for juvenile fish species.

The importance of turbidity as a major factor influencing juvenile fish distribution has been well documented in Australia (Blaber & Blaber, 1980; Blaber et al., 1990; Cyrus & Blaber 1992), and in south east African estuaries for both field and laboratory situations (Cyrus & Blaber, 1987a, b, c; Marais, 1988; Whitfield, 1994). For example, significantly higher numbers of juvenile fish were recorded from 14 turbid and semi turbid estuaries along the southeast coast of South Africa by Marais (1988) than for estuaries with higher water clarity.

The reason (s) as to why turbidity is so significant to juvenile fish is somewhat equivocal. It has been suggested that the protective isolation created by turbidity provides a ‘refuge’ from visual predators such as teleosts and birds (Blaber & Blaber, 1980; Cyrus & Blaber, 1992; Gregory, 1993; Abrahams & Kattenfield, 1997), in addition to increasing feeding success in suspenoid rich waters (Boehlert & Morgan, 1985; Whitfield, 1999). Indeed, the existence of turbidity gradients from the sea to estuarine environments has been suggested as a navigation aid for fry in locating nursery grounds (Cyrus & Blaber, 1987c; Blaber & Blaber, 1980; Trnski, 2002). However, excessively high turbidity levels have been shown to have deleterious effects on fish egg survival, hatching success, feeding efficiency, health/growth rate and population size (see review Wilber & Clark, 2001; see Chapter Three ; Morrison et al., 2009).

As revealed by CCA and BIOENV, suspended sediments and water clarity (i.e. turbidity) were the second most important factors structuring the fish assemblages in the Manukau in spring. These results are consistent, at a broad scale with a 25 harbour survey of New Zealand estuaries by Francis et al., (2005). Elevated turbidity levels are a common feature of the water clarity in the Manukau, a result of the tide and wind generated currents in combination with locally generated wind waves suspending and advecting fine sediments from the exposed intertidal banks (Bell et al., 1988; Vant, 1991). With substrate type and depth closely linked with turbidity, highest suspended sediment loads were recorded over the shallow, muddy substrates (<64μm) within the Pahurehure Inlet in conjunction with the highest densities of spring recruits.

Turbidity preferences of fish are species-specific (Cyrus & Blaber, 1987a,b) and vary with ontogeny (Blaber & Cyrus, 1983). For example, within the Manukau, newly recruited yellow-belly flounder were found in high numbers within the turbid inner harbour, associated with soft sediments relating to their burying behaviour to escape predators (Pearks, 1985) and distribution of suitable prey (Park, 1984; Saunders, 1999; Selleslagh et al., 2009). Older 1+ individuals are thought to migrate out of such estuaries to deeper, clearer waters to different feeding grounds and to spawn offshore (Colemn,1974a,b). Other species showing positive correlations with turbidity included exquisite goby, stargazers, anchovy and kahawai. Conversely, lowest densities of all species were found at the lower harbour sites (M5, M6), possibly due to lower food availability (associated with coarse substrates), greater current speeds, less shelter (particularly for M6) and higher water clarity (with greater expected exposure to predators).

One of the fundamental paradigms in estuarine nursery-ground ecology is the concept of refuge provided for vulnerable nekton in shallow water habitats from predation (Blaber & Blaber, 1980; Boesch & Turner, 1984; Paterson & Whitfield, 2000). Results from this study showed that larger piscivorous fish such as kahawai, snapper, trevally and small shark species (as captured in gillnets), were largely restricted to deeper waters and is consistent with other surveys (Whitfield & Blaber, 1978; Blaber & Blaber, 1980; Boesch & Turner, 1984; Blaber et al., 1985; Whitfield, 1999; Paterson & Whitfield, 2000). However, no significant correlations were detected in the Manukau between predator and juvenile fish distribution. Results may have been confounded by a range of potential sampling biases arising from predator behaviour and sampling gears. For instance, ambush predators such as stargazers, are sedentary by nature and are likely to be underrepresented in catches using stationary gear such as gill nets (Sheaves, 2001). Conversely, active schooling predators such as kahawai and trevally need only visit an area for a short time to impose significant mortality on prey fish (Hindell et al., 2000a). This coupled with high spatial patchiness of new recruits in estuarine shallow waters (Rozas & Minello, 1997; Minello & Rozas, 2002) suggests that a substantial level of sampling maybe required to provide adequate estimations of both predator and prey abundance (Sheaves, 2001; Baker & Sheaves, 2009). Recent research indicates that the role of small and occasional piscivores in the shallow estuarine habitats has been considerably underestimated (Sheaves, 2001; Baker & Sheaves, 2005; 2007; 2009) and has the potential to be a major structuring force through the ingestion of new recruits. Results of this study are supportive of this concept with dietary analysis (see Chapter Four), revealing that of the piscivores captured by beam trawling, ~54% were less than 100mm (FL).

It has been suggested that in highly turbid estuarine areas avoidance of visual predators is less important than the opportunistic exploitation of large amounts of food (Healey, 1971). In the present study, no significant correlations were found between the availability of macrobenthic prey biomass collected in the grab samples and fish distribution. This suggests that the availability of potential prey was probably not an overall limiting factor driving juvenile fish utilization of the estuary. However, fish distribution generally reflected the spatial patterns of their predominant prey. For example, higher numbers of sand flounder and speckled sole were recorded mid harbour (site M3) which was characterised by very high densities of bivalves, an important dietary component (see Chapter Four). While species such as snapper and jack mackerel consumed a wider prey spectrum including gamarridean amphipods and brittlestars, which were found in greater numbers in the lower harbour areas (M4,M5) in addition to decapod and isopod species.

Spatial and trophic niche overlap was observed for many of the fish species with epibenthic crustaceans (pericarids/decapods) supplying most of the dietary intake for the small fish community at all sites (Fig. 2.23). This agrees with other surveys (Day, 1981; Edgar & Shaw, 1995b; Horinouchi & Sano, 2000; Martinho et al., 2007) and suggests that competition was probably diminished by the abundant food supply, particularly for spring. However, this study covered only two sampling periods and the ecological relationships between fish and their prey is a complex one due to ontogenetic changes in morphology which can lead to variation in foraging abilities (Montgomery, 1977; Boubeé & Ward, 1997a; Gillanders 1997, Sudo & Azeta, 2001). This may be very significant in determining distributions of different size/age classes of fish (Morrison, 1990; Szedlmayer & Lee, 2004).

As a means to cope with their dynamic environment, estuarine fishes often exhibit flexible spatial and temporal feeding strategies (adaptive foraging). This often takes the form of feeding guilds centered on seasonally abundant food resources such as mysid shrimps (Burke, 1995; Hostens & Mees, 1999; Freyer et al., 2003; Akin & Winemiller, 2006). Results from this study support these findings with mysid abundance retained by CCA forward stepwise analysis as significantly contributing to the ordination.

The autumn stomach contents revealed mysids to be an important dietary component of many fish species (Fig. 2.23). However, mysid capture rates in the grab samples were low. An epibenthic sledge was ruled out for the spring sampling due to the potential difficulties with biogenic structure and high currents at some of the sampling locations. In addition, the extreme patchiness associated with mysid schooling, and their effective evasive behaviours, often results in only 1-10% sampling efficiency from sledges (Jumars, 2006; Carlton & Hamner, 1987; Eleftheriou & Holme, 1984). Abundance was therefore estimated from the beam trawl captures as a broadscale measure of availability.

Dietary composition within the Manukau shifted seasonally in response to changes in food resource availability, with a significant increase in consumption of mysids (contributing 66% total dietary biomass) particularly for the spring recruits (Fig. 2.23). This can be attributed to greater availability of this resource over spring with densities reaching 12,682 individuals per beam trawl within the upper harbour intertidal/bank sites (M1-M2; Fig. 2.12). These sites were characterized by large amounts of debris, suspended sediments, soft substratum (<64μm) and low current velocity. Although the abundance of zooplankton was not measured in this study, other research suggests that these turbid regions are characterized by higher concentrations of zooplankton, particularly copepods (calanoid & harpacticoid) which are favoured prey of both mysids and postlarval fish (Blaber, 1977; Swenson, 1978; Carter, 1986; Maes et al., 1998; Islam et al., 2006; Jumars, 2006). Mysids often replace copepods progressively in the diet of juvenile fish (Mauchline, 1980; Sorbe, 1981, cited in Hostens & Mees, 1999; May, 1981; Sumpton & Greenwood, 1990). Therefore, migrating into these prey rich areas would confer favourable conditions for optimal growth along with a smooth transition from a pelagic to demersal diet/habitat (Tanaka et al., 1987). It has also been suggested that mysid density gradients may influence the movement of flounder to nursery grounds (Burke, 1995).

The results of this study emphasize the ecological importance of mysids as a significant component of the estuarine food web within the Manukau. Mysids played a key role in energy transfer as intermediate prey, serving as a link between the benthic and pelagic systems, particularly over spring. This pattern is consistent with findings from other estuaries (Mees et al., 1994; Roast et al., 1998; Vilas et al., 2007). Considering their significance to coastal fish production further study into the distribution and population dynamics of mysids is warranted (Susuki et al., 2009).

Sediment characteristics, substratum heterogeneity (e.g. horse mussels; green lipped mussels; hydroids; sponges) and vegetation (e.g. seagrass meadows) can also affect juvenile fish distribution through influencing prey availability (Heck et al.,1995; Edgar et al., 1994; Edgar, 1990b; see Chapter Four), by providing shelter (attenuating current flow: Green et al., 1998; Foncesca & Koehl, 2006; Hendriks et al., 2006; Widdows et al., 2008) and refuge from predators (Blaber & Blaber, 1980; Parrish 1989; Marshall & Elliot, 1998; Nagelkerken et al., 200; Thrush et al., 2002). Other studies have reported higher invertebrate and fish abundance, species diversity and biomass with increasing habitat complexity both within New Zealand (Henriques, 1980; Usmar, 2009; Morrison & Carbines, 2006; Schwarz et al., 2006; McLeod, 2009; Morrison unpubl. data), Australia (Edgar, 1992; Edgar & Shaw, 1995b; Martin et al., 1995; Jenkins & Wheatley, 1998; Travers & Potter, 2002; Bloomfield & Gillanders, 2005) and the United States (Orth et al.,1984; Heck et al., 1995; Minello, 1999; Humphries et al., 2011).

Biogenic structure was identified by CCA forward selection as being a significant environmental factor for juvenile fish distribution in spring for the lower estuarine sites (M4-M6), particularly for species such as snapper, gurnard and jackmackerel (Fig. 2.25). Results support recent research undertaken in seven northern, New Zealand estuaries by Morrison et al., (in reveiw), and in the Mahurangi estuary by Usmar (2009), and the Kaipara Harbour (author unpubl. data) showing juvenile snapper densities significantly correlated with bottom structure elements including horse mussels and sponges.

Juvenile fish diversity also peaked mid harbour at site M4 in spring, which recorded the highest amount of biogenic bycatch. Invertebrate prey diversity (from grab samples) was also positively correlated with habitat complexity (Fig. 2.9), with highest species richness occurring at the lower harbour sites (M4-6). This was reflected in the increasingly varied prey items consumed by species such as snapper and gurnard at these sites (Fig. 2.23). However, overall abundance of juvenile fish within the Manukau declined going down the gradient due to the sequential recruitment events of the different species into the inner harbour.

Historical evidence suggests that the Manukau harbour has undergone significant change in terms of habitat degradation from urbanization/deforestation of catchments with resultant increased sedimentation, turbidity (Vant, 1990; Swales et al., 2002) and concomitant loss of extensive seagrass meadows since the 1960's (Heinriques, 1977; Veitch, 1978; Turner et al., 1999a). These meadows were once described as "*splendid Zostera fields up to a mile across*" by Morton & Miller (1973). Early research in the Manukau by Powell (1937) and anecdotal evidence (Waitangi Tribunal, 1989) suggests a decline in the abundances of commercially important species (e.g. flounder, mullet, snapper) in the Manukau in conjunction with declining water clarity. Waitangi Tribunal witnesses recalled "how the murky waters were once crystal clear and the thick mud once firm white sand" (Waitangi Tribunal, 1989). With recent research in northern New Zealand (Schwarz et al., 2006; Morrison et al., 2007; see Chapter Four) identifying subtidal seagrass meadows as important nursery grounds for species such as snapper and trevally, depletion of harbour fishing may be partially linked to the loss of this significant biogenic habitat. Declines of 40% in commercial fish catches have also been recorded in Western Port Bay, Australia after parallel reductions (70%) in seagrass cover (Jenkins et al., 1993). Unfortunately, these habitats are especially vulnerable to anthropogenic impacts such as increasing sedimentation and turbidity from land-based activities (Morrison et al., 2009).

2.6.4 Comparisons with similar studies

In the present study, both abiotic and biotic variables were used to explain the spatio temporal fish assemblages. This contrasts with many previous studies which have utilised abiotic factors only (Selleslagh & Amara, 2008a,b; Selleslagh et al., 2009 and references therein). The first two axes of the CCA accounted for 72% and 54.8% of the variance due to fish abundance-environmental relations for autumn and spring respectively. The cumulative variance of species data explained by the first four axes of CCA was 15.1% (autumn), 33.2% (spring- additional variables) and 49% for BIOENV analysis (Tables 2.13, 2.14). These results are comparable to recent research in the United Kingdom (Marshall & Elliot, 1998; Selleslagh & Amara, 2008; Selleslagh et al., 2009), Portugal (Martinho et al., 2007); Turkey (Akin et al., 2005) and the USA (Akin et al., 2003; Martino & Able, 2003), where the proportion of explained variation of species data was generally low (~18-47%). These results reinforce the generally difficult task of predicting the outcomes of multiple processes within estuaries, whereby a complex set of factors are interacting simultaneously in a non-linear fashion (Perillo et al., 2009), including fish species (and size classes within species) experiencing and responding to the environment differently according to their own unique suite of physiological tolerances and spatio temporal scales (Pittman et al., 2004).

This suggests that other physical and biological factors, not measured in this study could account for some of the unexplained variation. These include temperature (Thiel, 1995; Whitfield, 1999; Akin et al., 2005; Selleslagh & Amara, 2008a,b), salinity (Thiel, 1995; Maes et al., 1998, 2004; Marshall & Elliot, 1998; Jaureguizar et al., 2003; Selleslagh & Amara, 2009), water quality (pollution, eutrophication site contamination), disease (Meng & Powell, 1999); biological interrelationships such as group behaviour, competition and recruitment space (Holbrook & Schmitt, 1989; Menge & Olsen, 1990; Edgar & Aoki, 1993; Martino & Able, 2003; Gratwicke et al., 2006), slow recruitment after environmental disturbance/habitat degradation (Moore, 1977; Bruton, 1985; Jenkins et al., 1993; Edgar & Barret, 2000; Stevens et al., 2006), estuary mouth status/size (Marais, 1988; Whitfield, 1999; Vorwerk et al., 2003) and ontogenetic changes in physiology (Montgomery, 1977; Gillanders,1995; Boubee & Ward, 1997) and habitat requirements (Morrison, 1990).

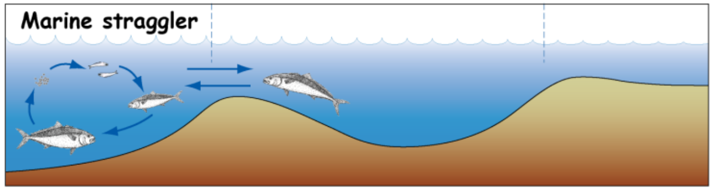
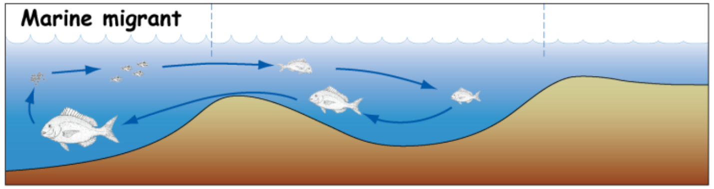
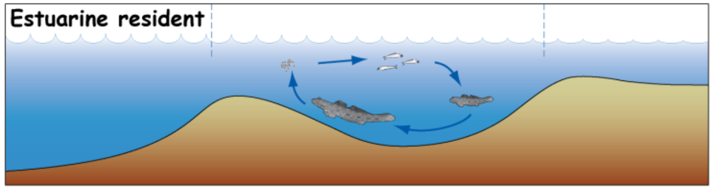
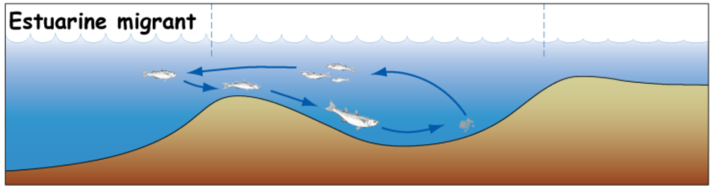
Although temperature and salinity were not measured in this study, prior research indicates that the harbour waters are typically well mixed, with salinity only varying by 3% between the heads and innermost shore with little freshwater inflow (Vant & Williams, 1992), and differences would be minimal in this study area. In addition, water temperatures change little horizontally or vertically, with values in the range of 19-21° C (Heath et al., 1977; Vant & Williams, 1992; Bell et al., 1997; Green et al., 1997). Any temperature variation that occurred during this study would co vary with depth and distance down the harbour.

Observations in this estuary suggest that broad-scale (10-20km) patterns are primarily related to the individual species responses to their physical environment, possibly set initially during seasonal migrations of larvae into the estuary (Blaber & Blaber. 1980; Potter et al., 1986; Akin et al., 2003). Finer spatial patterns (1km) appear to be the result of habitat associations that are most likely driven by habitat selection, foraging, competition, and/or predator avoidance. Results are consistent with theoretical views on community structure whereby dominant abiotic variables are thought to act like a ‘physiological sieve’ structuring the community framework, while biotic interactions refine distribution patterns within this structure (Menge & Olsen, 1990; Kupschus & Tremain, 2001; Martino & Able 2003; Akin et al., 2005). Results concur with Menge & Sutherland’s (1987) theory whereby physically controlled, dynamic systems tend to have low diversity and more dynamic assemblages (as observed in many estuaries), while conversely in physically stable systems, environmental factors play a minor role and biotic factors predominate.

In sum, the overall picture within the Manukau Harbour is of a moderately diverse assemblage of juvenile and small fish species in a physical environment characterized more by gradients than sharp changes, with relative abundance of each species varying, rather than community structure from a larger species pool. Among the potential explanatory variables examined, depth, currents and turbidity (i.e. suspended sediments and water clarity) were the most important determinants in shaping the assemblages. Other contributors included mysid distribution, biogenic habitat and substrate type (<64μm). As a collective these variables are strongly correlated. However, it must be emphasized that any habitat correlations with relative fish abundance are not necessarily causative. Further manipulative investigation at different spatio temporal scales would be required to clarify these processes.

The high numbers of juveniles present, including several commercially important species (flounders; mullets), particularly within the Pahurehure Inlet confirms the importance of the Manukau Harbour as a nursery ground for those marine migrants utilizing this estuary as a ‘transition zone’ (Gillanders et al., 2003) to offshore recruitment into the coastal adult population. Identifying and quantifying the links between small fish assemblages and their environmental determinants as undertaken in this study can be an important and useful tool for management and protection of essential habitats and for maintaining sustainable fisheries. Knowledge of the response of estuarine fishes to changes in environmental conditions will not only enhance our biological understanding of estuarine fish, but will also contribute to our understanding of the potential effects of ongoing or new anthropogenic stressors e.g. suspended sediments (Selleslagh & Amara, 2008) as well as the potential outcomes of habitat restoration. This study helps to set a management baseline for the Manukau Harbour and similar estuarine systems (e.g. the Kaipara and Hokianga Harbours) in the New Zealand context.

However, while intensive in detail, this study was conducted in only one ‘impacted’ estuary (albeit the second largest in New Zealand), and as such represents a small sampling fraction of species overall environmental use patterns, and of their tolerance to different environmental conditions. Given that species appear to vary in the degree to which they utilize different estuaries, both within and among areas (Able & Grothues, 2007; Morrison et al., 2007; Morrison, unpubl. data), and the natural and human induced variability within individual estuaries, future research will need to include multiple estuary, regional scale approaches; encompassing both ‘degraded’ and ‘pristine’ reference sites thereby mitigating the ‘shifting baseline’ concept (Edgar et al., 1999; Edgar & Barret, 2000; Vasconcelos et al., 2010). This would allow identification of more subtle impacts affecting estuarine processes throughout larger regions and subsequently allow the responses of fish assemblages to natural and anthropogenic stressors to be identified and quantified.

Marine Environment Estuarine Environment Freshwater Environment	**Description**	**Species**	**Specific example**
Marine straggler 	Species that spawn at sea, usually enter estuaries in only low numbers, and use only the lower-most marine parts of the estuary. Usually such species are most abundant in coastal areas adjacent to the estuaries.	Kingfish, red cod, barracouta, blue warehou, bastard red cod, leatherjackets, ahuru, mackerel, red gurnard, red mullet (goatfish) and even occasional hoki in more southern waters. Historically also included species such as hapuka and ling.	Kingfish - this warmer water species can seasonally be found in northern estuaries as adults hunting small bait fish, usually as lone fish. Most adults are however found out on the open coast, while small juveniles are found offshore in deeper waters associated with drifting kelp and other forms of debns on the surface of the sea. Historically however, there are accounts of seasonal 'runs' into estuaries, e.g. of large schools following kahawai schools up into the Rangirere Arm of the Manukau Harbour until the 1970s. Such runs were eliminated by heavy fishing pressure, and have not been reported in more recent decades.
Marine migrant 	Species that spawn at sea, and whose larvae and/or juveniles then often enter estuaries in large numbers. Can be further divided into a) marine estuarine opportunists, that may occur in large numbers, but may also use nearshore coastal waters as an alternative habitat, and b) marine estuarine-dependent, that require sheltered estuarine habitats as juveniles, but that live along coasts where are no such equivalent habitats available out on the coast, forcing them to be dependent on estuarine habitats only.	Snapper, trevally, kahawai, parore, sand flounder, stout and slender sprats, anchovy, spotty, spotted and estuarine stargazers, specked sole.	Snapper - a dominant coastal fisheries species in northern New Zealand, which uses nursery habitats that have higher levels of structure in them e.g. subtidal seagrass meadows, horse mussel beds, sponge gardens, the edge of kelp forests, and even man-made structures such as mooring blocks and wharves. Where these nursery grounds occur depends on the region looked at. For example, on the west coast of the North Island they are largely confined to estuaries, especially the large estuaries such as the Kaipara and Manukau Harbours, as the open coast is so exposed that such habitats do not occur there (making these populations "marine estuarine-dependent"); while in contrast on the east coast, such nursery habitats are also found in large shallow coastal bays, and around coastal islands, making these populations "marine estuarine opportunists". Juvenile snapper are eaten by kahawai, John Dory, and even adult snapper (cannabilism), which is why they prefer habitats with structure in which to hide from these and other predators.
Estuarine resident 	Species capable of completing their entire life cycle within estuaries.	Sand and exquisite gobies, estuarine triplefin, bridled goby, Asian goby.	Sand gobies - this small species is restricted to northern New Zealand, and occurs in high abundances in both intertidal and subtidal bare sediment habitats. They prefer areas of sandier sediments (hence the name), and clearer waters. While we still have much to learn about the spawning and larval parts of the life history of this and other small estuarine species, it is likely that the entire life history (adults → eggs → larvae → juveniles → adults) can be completed inside estuarine environments. Sand gobies are eaten by species including spotted and estuarine stargazers, speckled sole, red gurnard, and two species of marine eels.
Estuarine migrant 	Species that have larval stages of their life cycle completed outside the estuary or are also represented by discrete coastal or freshwater populations.	Seahorses (brood their young in a pouch), clingfish, yelloweyed mullet, piper (garfish), pipefish species, grey mullet .	Yellow-eyed mullet - this species is found throughout New Zealand, but dominates beach seine catches in northern estuaries. Almost all of these catches are of juvenile fish, covering several different age classes (0, 1 and 2 year olds), while large adult fish (20 -50 cm) are rare. Spawning probably takes place in the coastal zone, and small larval yellow-eyed mullet can be found miles off the coast, sheltering under drift algae that has become detached from coastal reefs by storms. These larvae make their way into estuaries, and then start their juvenile phase.

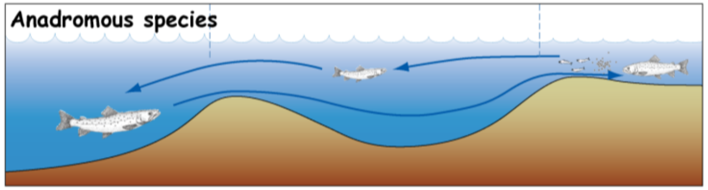
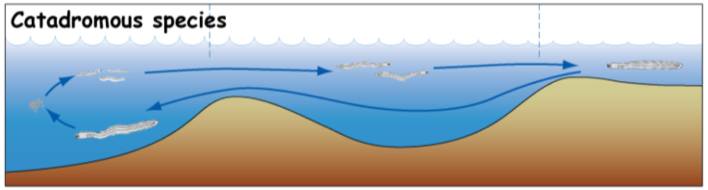
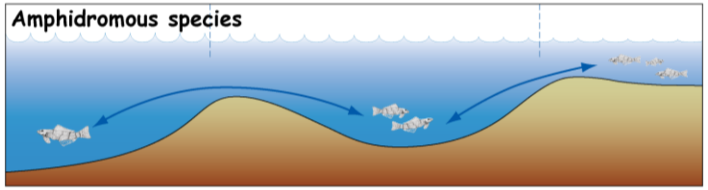
	Species that undergo their greatest growth at sea and which prior to attaining sexual maturity, migrate into rivers where spawning subsequently occurs.	Smelt, lamprey, brown trout, Chinook salmon.	Brown trout - an introduced species, occasionally caught by beach seining in South Island estuaries, where the water temperatures are cool enough for this freshwater dominated species to use such habitats. Both riverine and sea trout occur, being variants of the same species. Here we are referring to sea run trout only.
	Species that spend much of their trophic (feeding) life in freshwater and which subsequently migrate out to sea to spawn.	Shortfin and longfin eels, black flounder, inanga.	Longfin eel - this species is only found in New Zealand, that can grow up to two metres long and to 50 kilograms in weight. At around 25 years of age (but sometimes up to sixty) they migrate down rivers and out into the Pacific Ocean, where they spawn and then die. After some time feeding in the ocean, small elvers (young eels) make their way back to New Zealand, and migrate up rivers and into other freshwater systems such as inland lakes, where they settle into their adult lives. Longfin eels are considered under threat due to fishing pressures and loss of habitats.
	Species which migrate between the sea and freshwater and in which the migration in either direction is not related to spawning.	Koaro, torrentfish, banded kokopu, shortjaw kokopu, giant kokopu and freshwater bullies (bluegill, redfin, common, and giant).	Torrentfish - this species is specially adopted to living in high flow, 'white-water' freshwater habitats. We know very little about reproduction, but small juveniles are occassionally caught in estuaries as they migrate into rivers from the sea. For example they have been sampled in Lake Ferry (east of Wellington) in low numbers in the summer.

Figure 2.1 Ecological guilds: Life-cycles of fish species utilizing estuaries (adapted from Elliot et al., 2007).

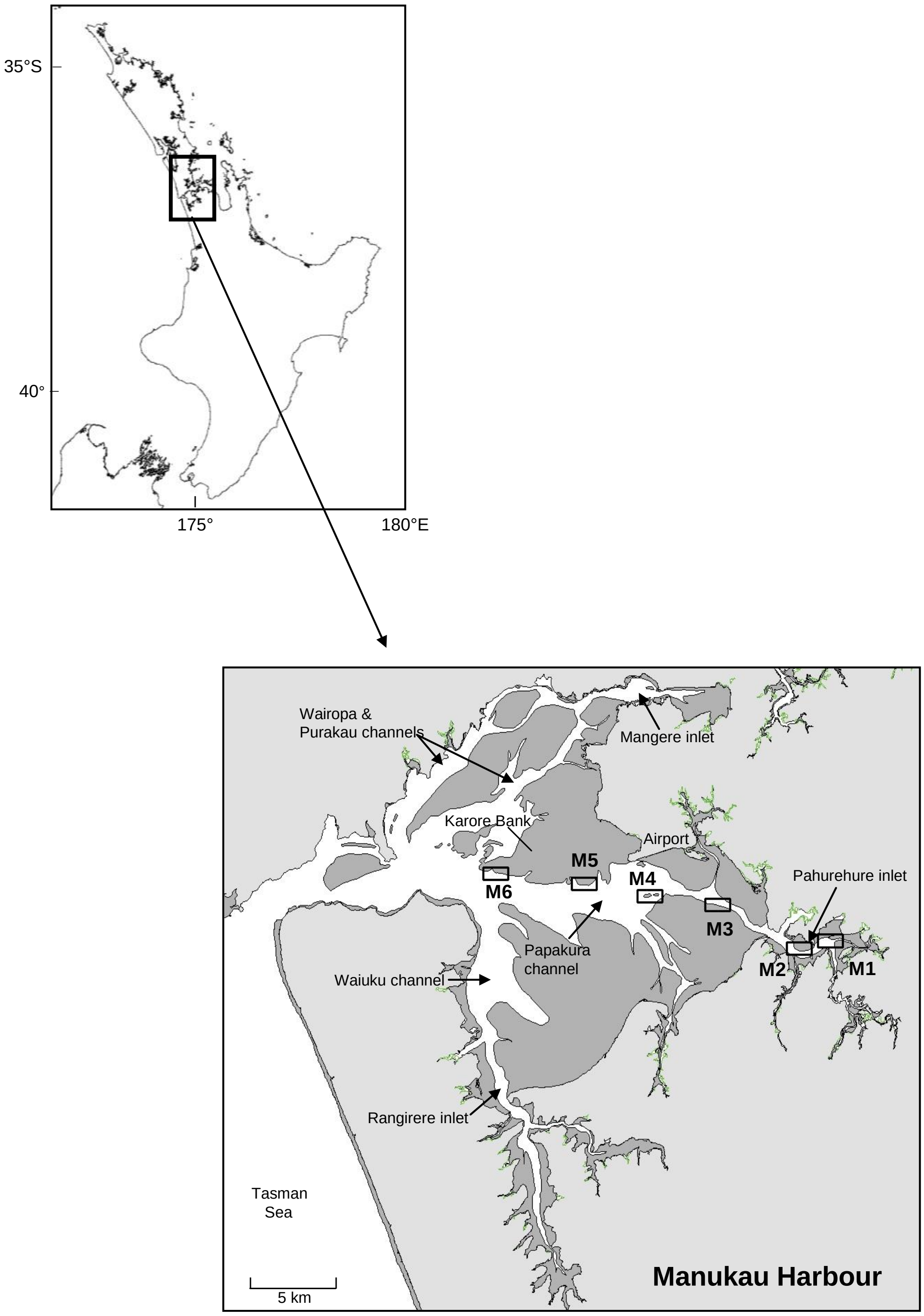


Figure 2.2 Map of study location showing sampling sites.■ Intertidal flats; ☐ Subtidal channels; ▨ Land

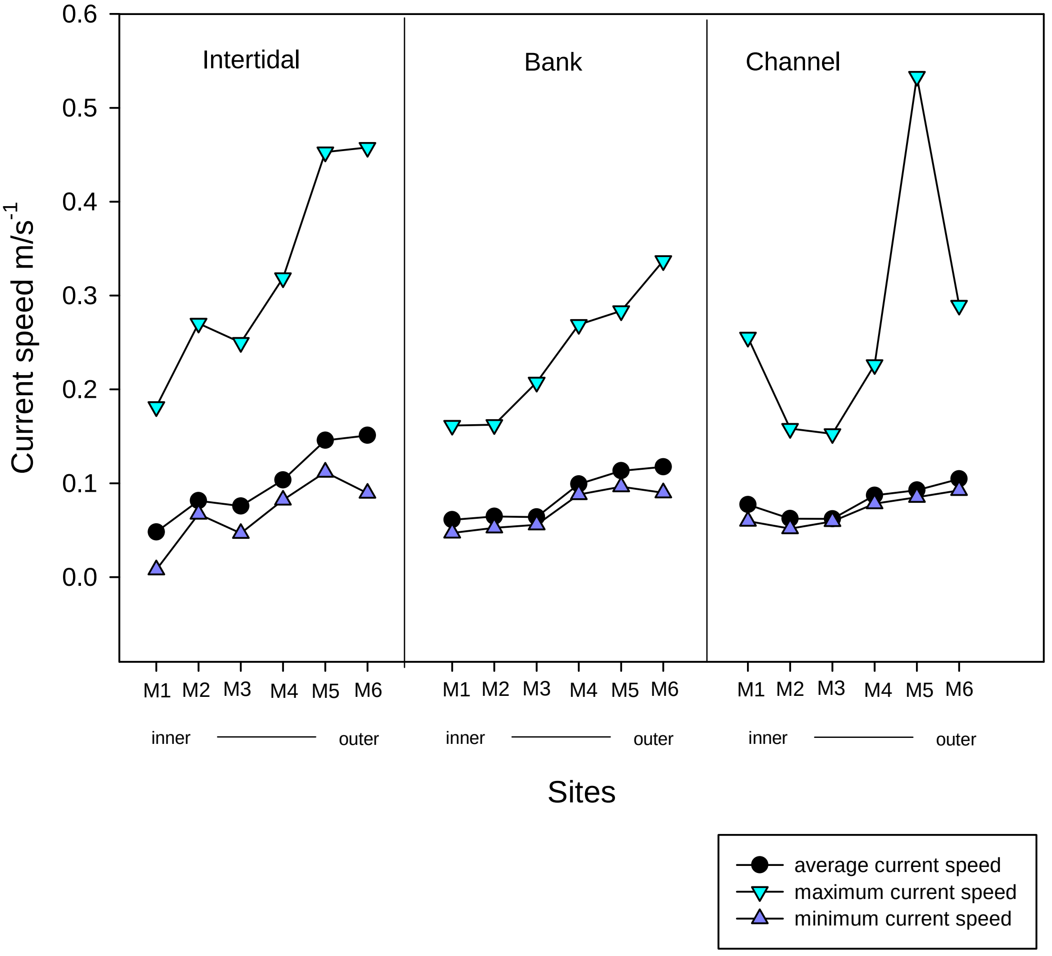


Figure 2.3 Mean, maximum and minimum current velocities modelled for intertidal/bank/channel habitats at six sites. Derived from the DHI MIKE# FM HD and MT model of the region (Pritchard et al., 2008).

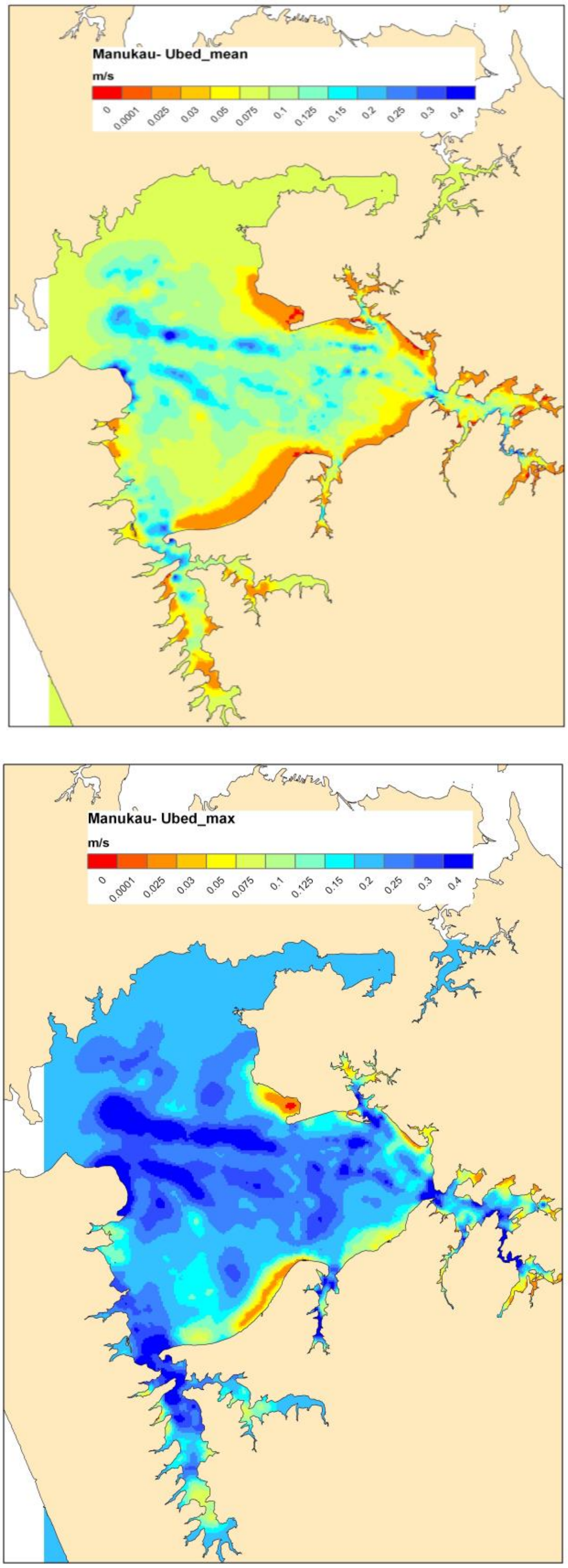


Figure 2.4 Mean and maximum current velocities for the Manukau Harbour, derived from the DHI MIKE3 FM HD and MT model of the region (Pritchard et al., 2008).

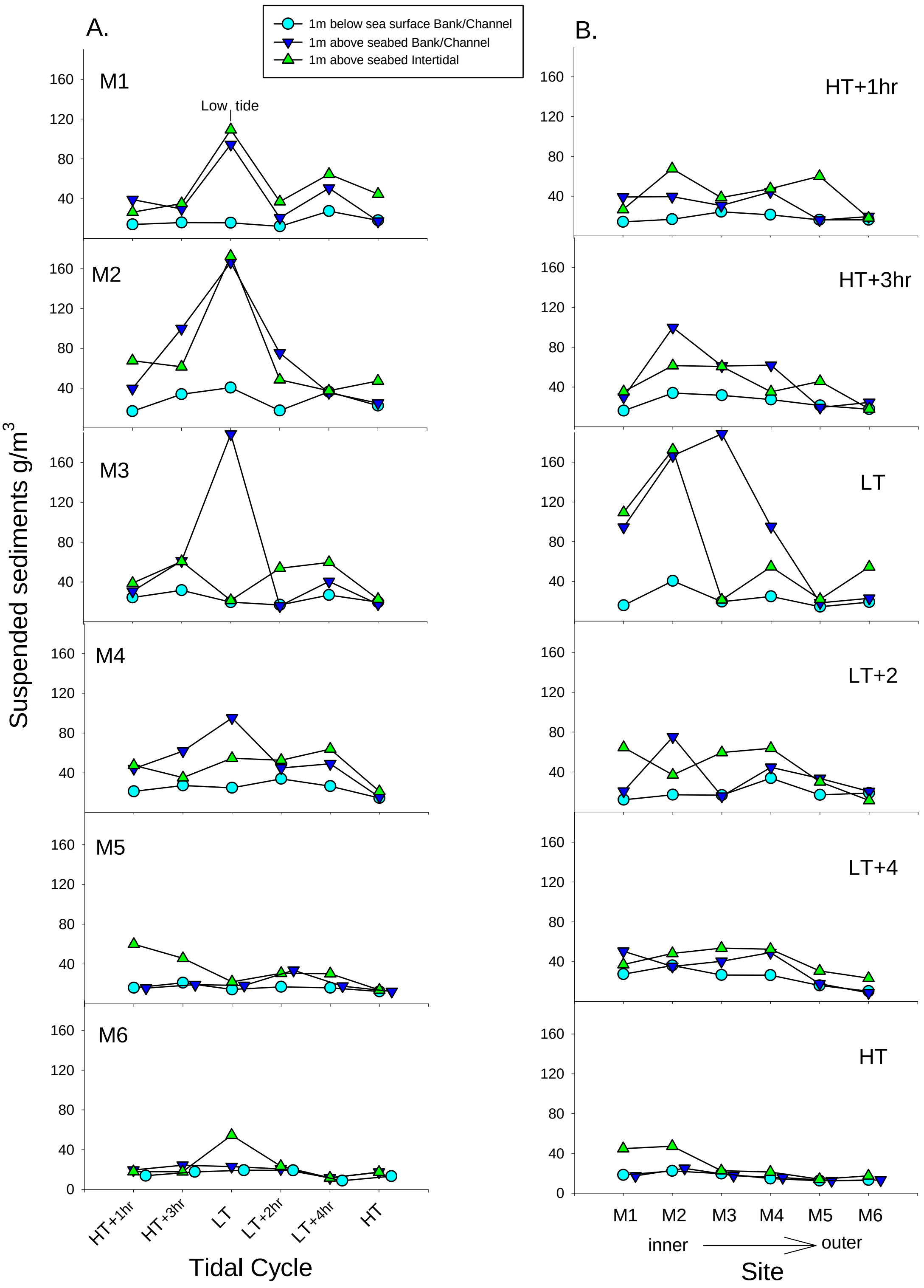


Figure 2.5 Suspended sediments collected over one complete tidal cycle, plotted spatially by site (A), and temporally for all sites (B). Two samples were collected from the bank/channel habitat: 1m above the seabed and 1m below the sea surface. One sample was collected from the intertidal habitat 1m above the seabed.

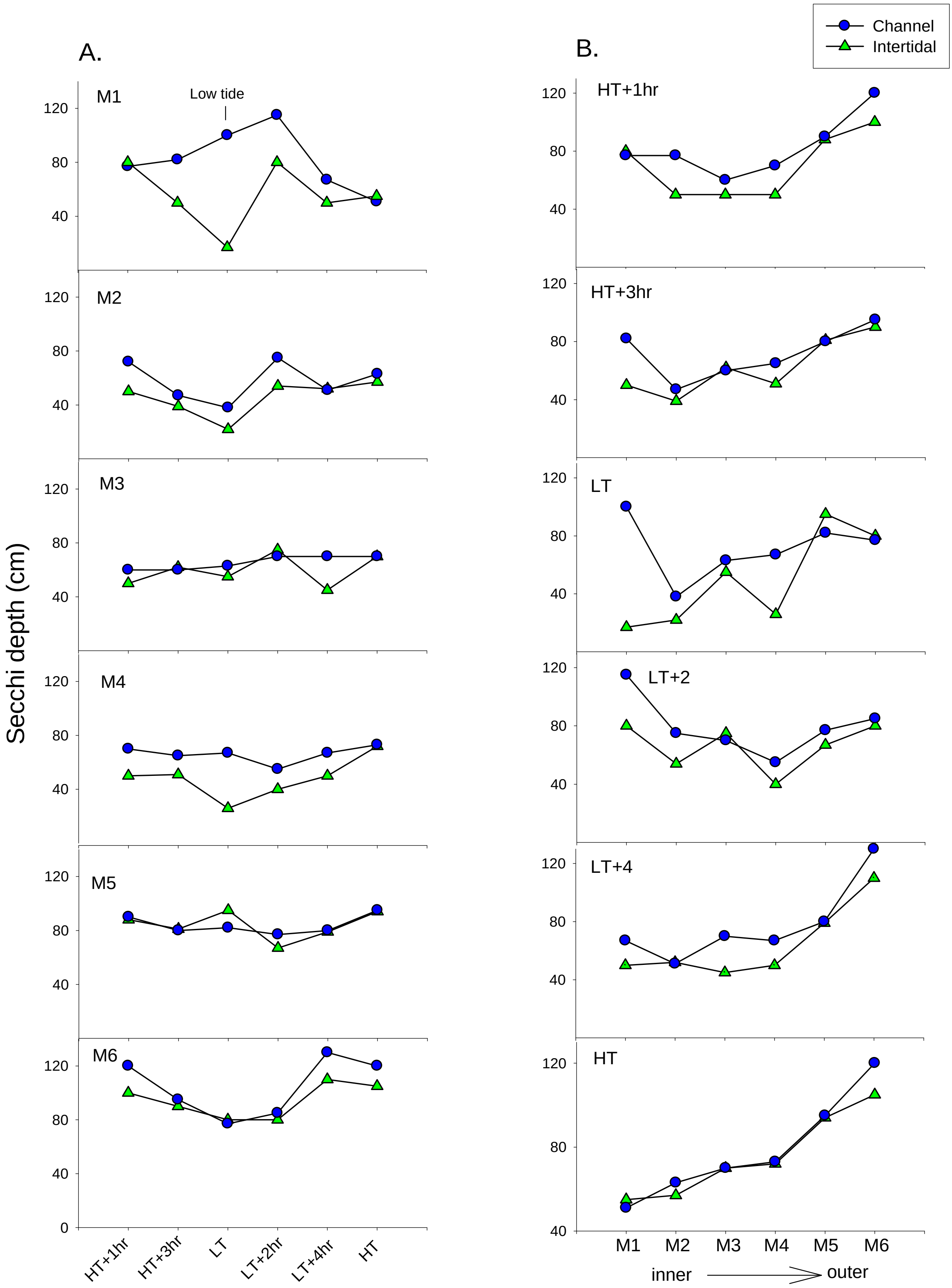


Figure 2.6 Secchi depth (cm) plotted over one complete tidal cycle, spatially by site (A) and temporally for all sites (B).

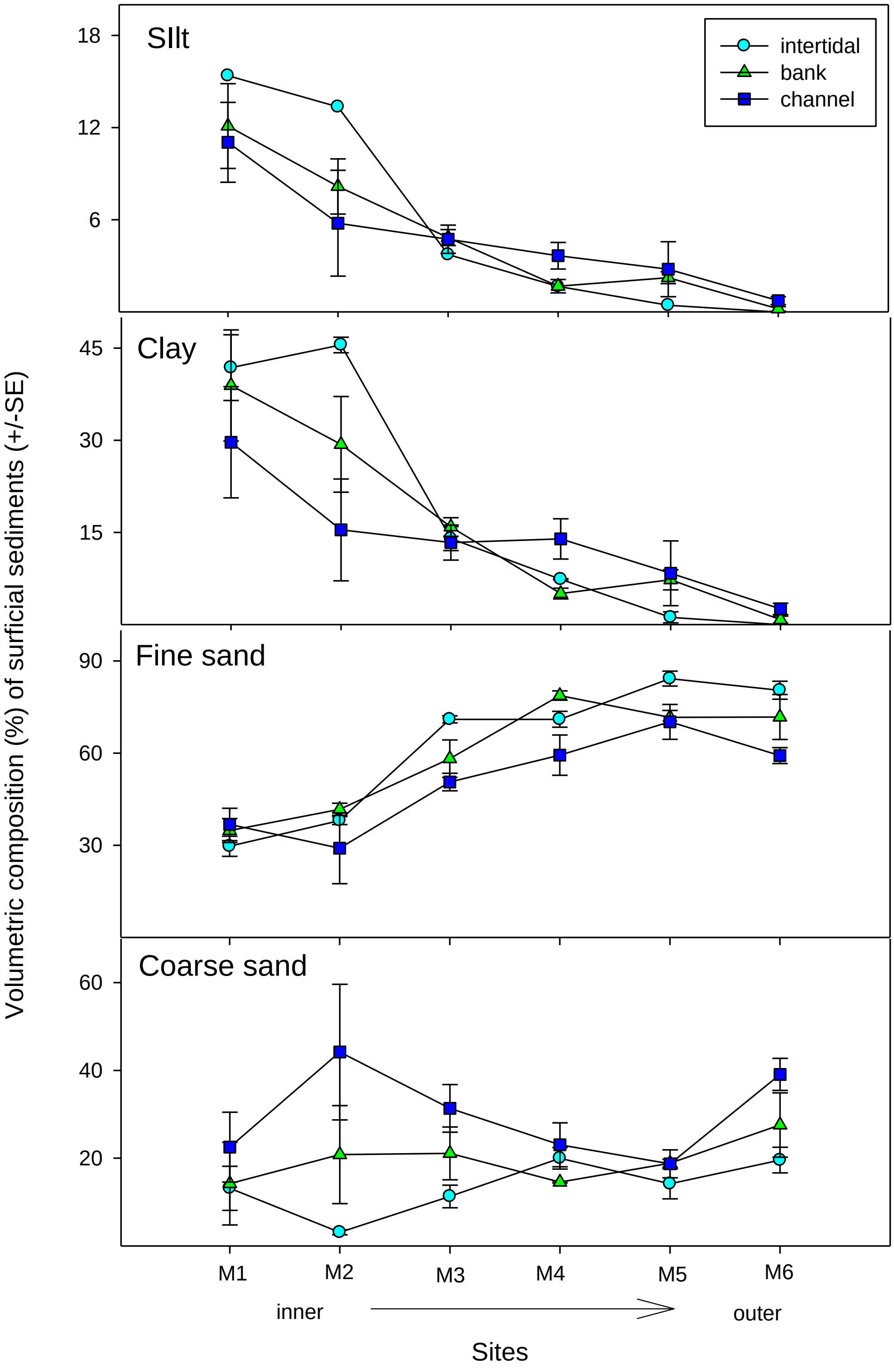


Figure 2.7 Percentage volumetric composition of surficial sediments (±SE) collected from the intertidal/bank/channel habitats at six sites. Note the Y values are different for each graph.

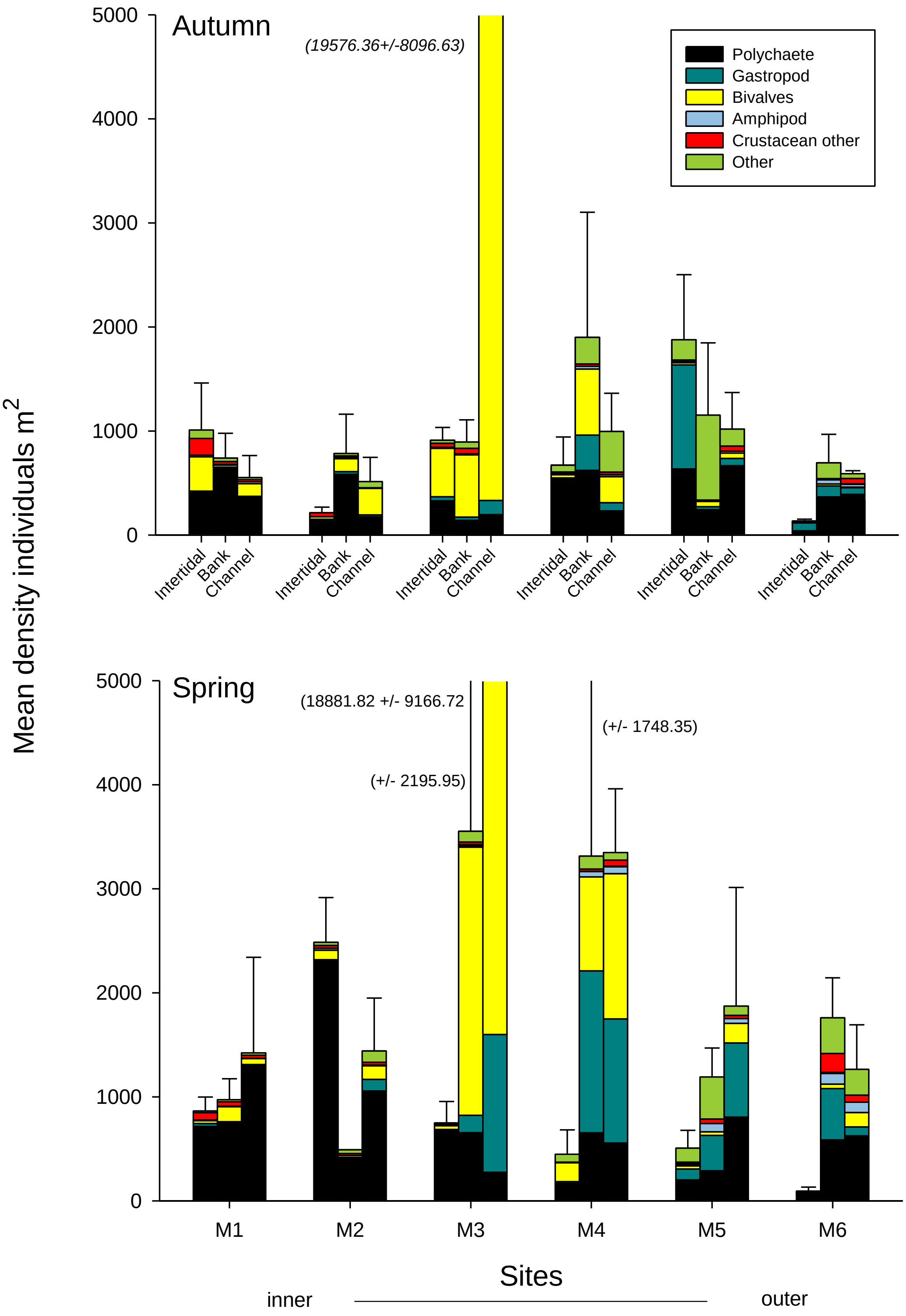


Figure 2.8 Mean abundance (±SE) of epifaunal and infaunal animals (1-5.6 mm sieve size) collected from the intertidal/bank/channel habitats at 6 sites over two seasons. Numbers in brackets denote values above cut-off.

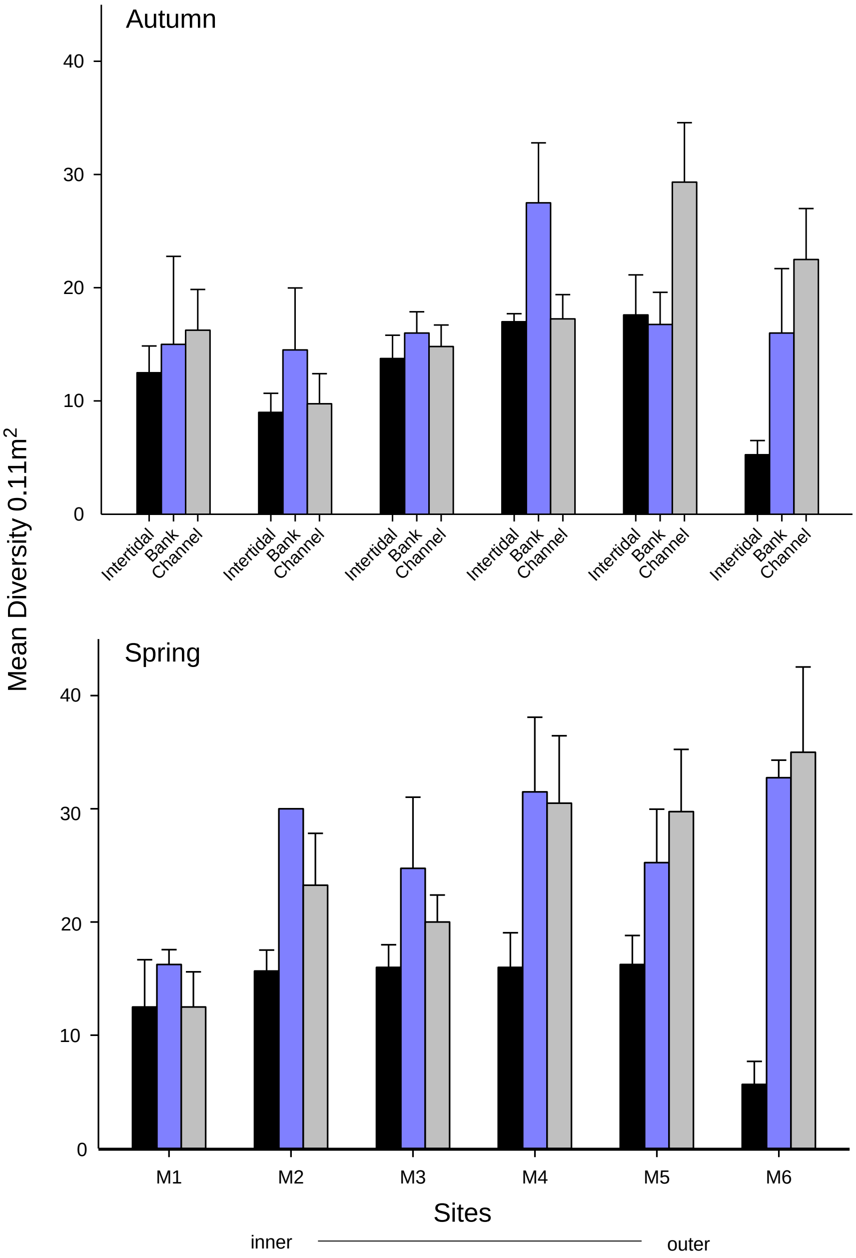


Figure 2.9 Mean total number (±SE) of benthic species collected per grab sample ($0.11m^2$) from the intertidal/bank/channel habitats at six sites over two seasons.

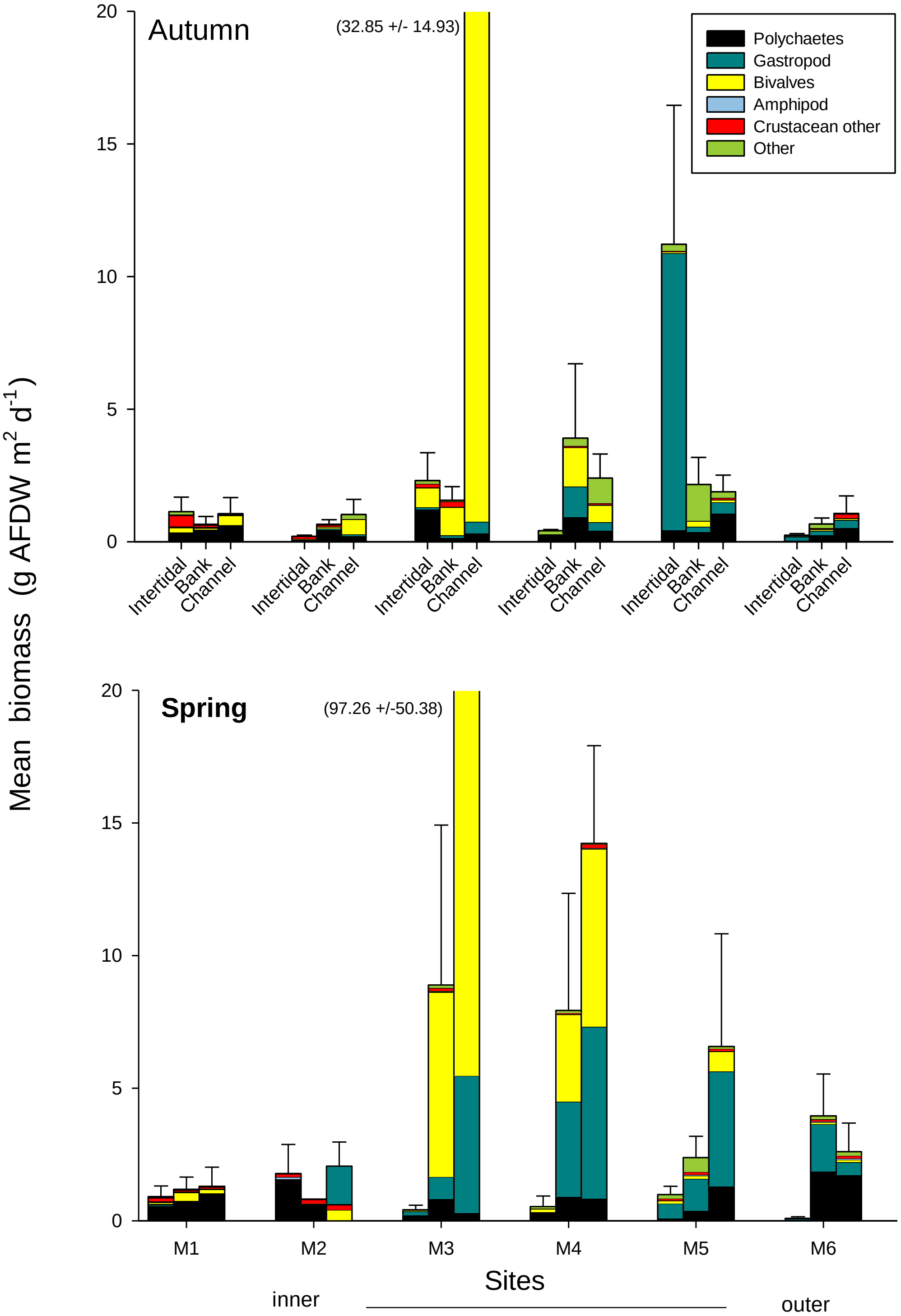


Figure 2.10 Mean biomass (±SE) of epifaunal and infaunal animals (1-5.6 mm sieve size) collected from the intertidal/bank/channel habitats at 6 sites over two seasons. Numbers in brackets denote values above cut-off.

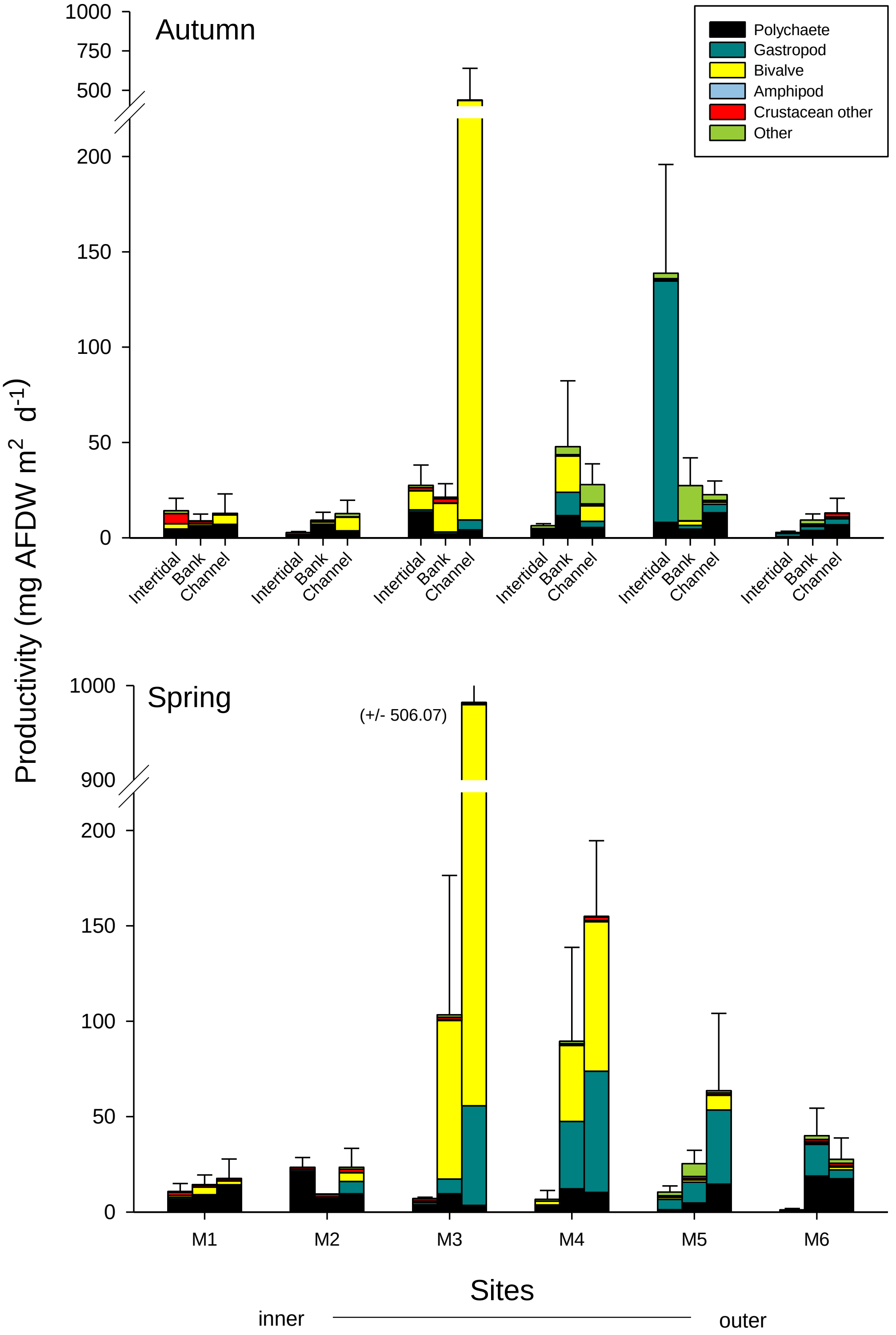


Figure 2.11 Mean estimated productivity (±SE) of total benthos (1-5.6 mm sieve size) collected from the intertidal/bank/channel habitats at 6 sites over two seasons.

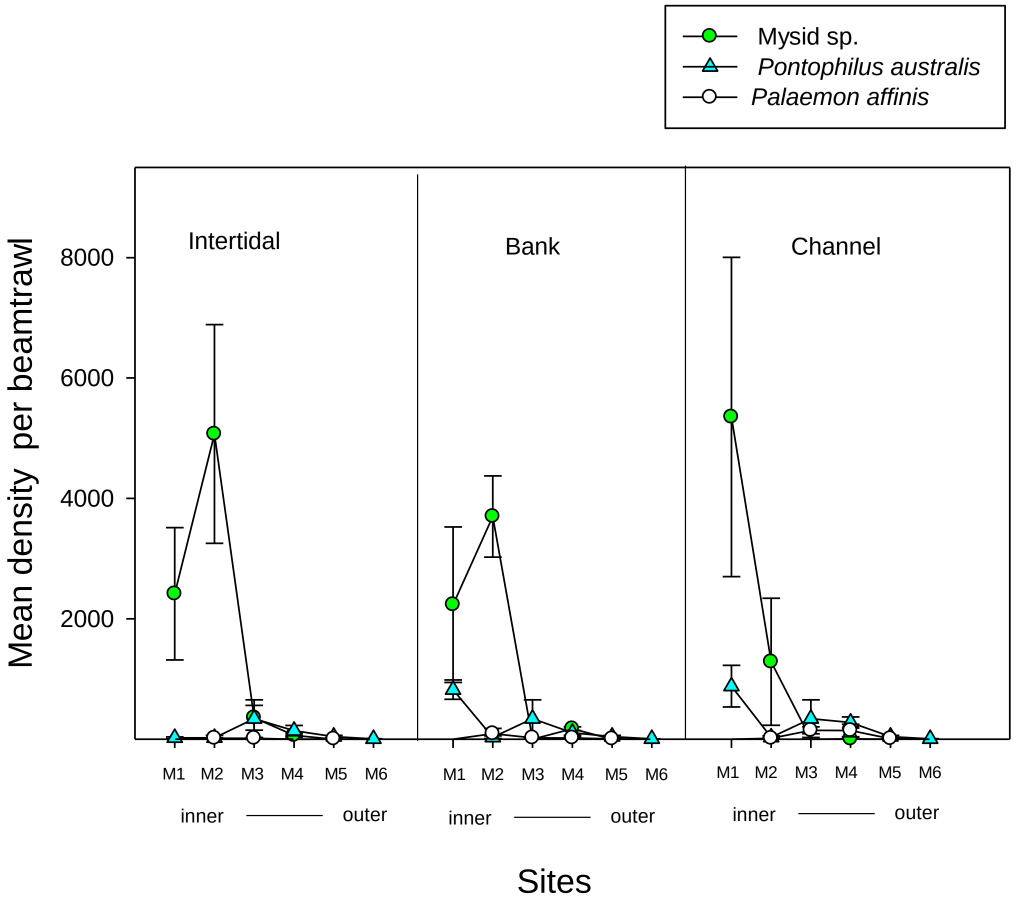


Figure 2.12 Mean density (± SE) of macrocrustaceans collected by beam trawl for the Spring sampling from the intertidal/bank/channel habitats at 6 sites.

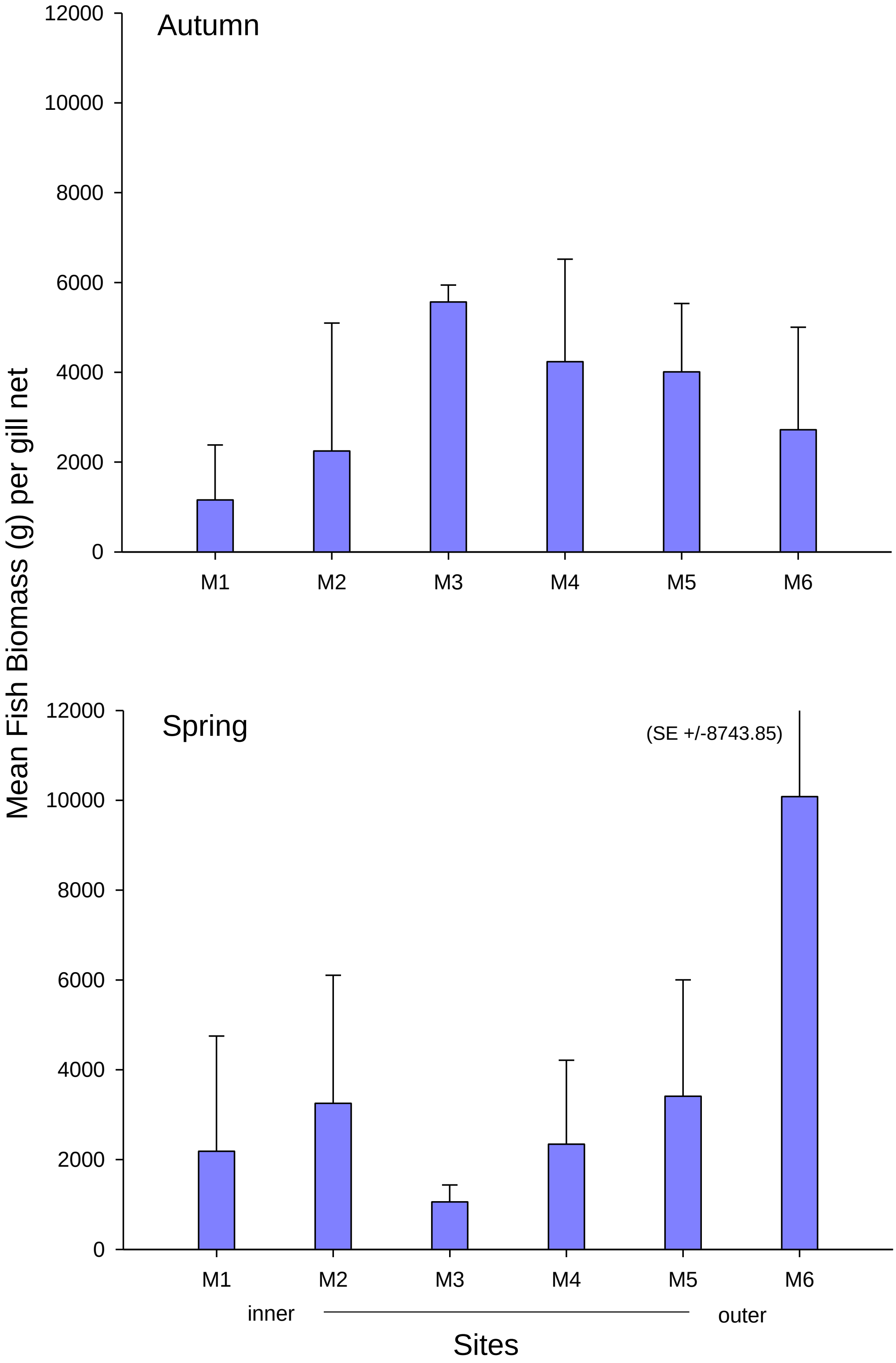


Figure 2.13 Mean biomass (±SE) of predator guild fishes per gillnet from bank/channel habitats at 6 sites over two seasons.

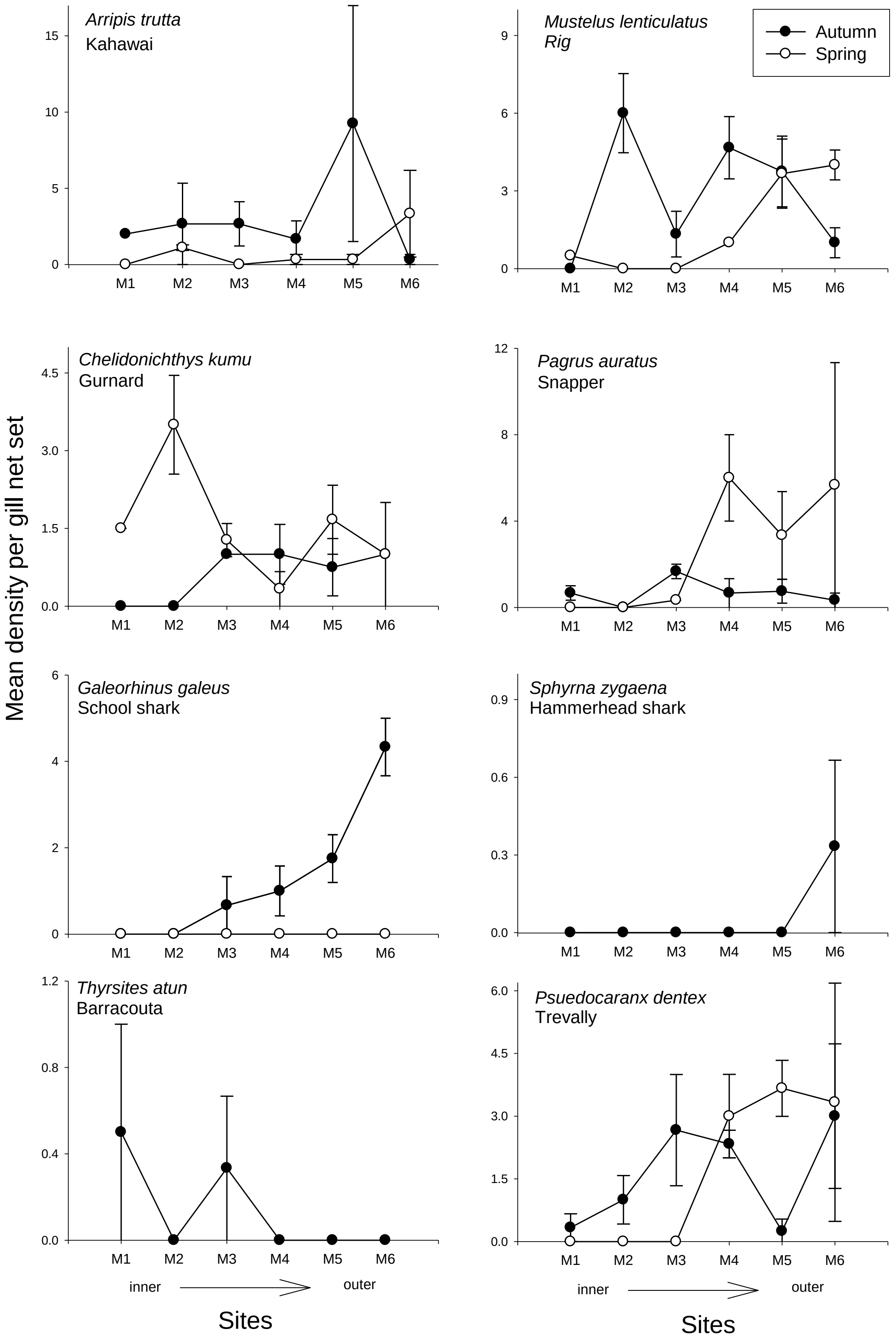


Figure 2.14 Mean abundance (±SE) of predator guild fishes collected in gillnets from bank/channel habitats at 6 sites over two seasons.

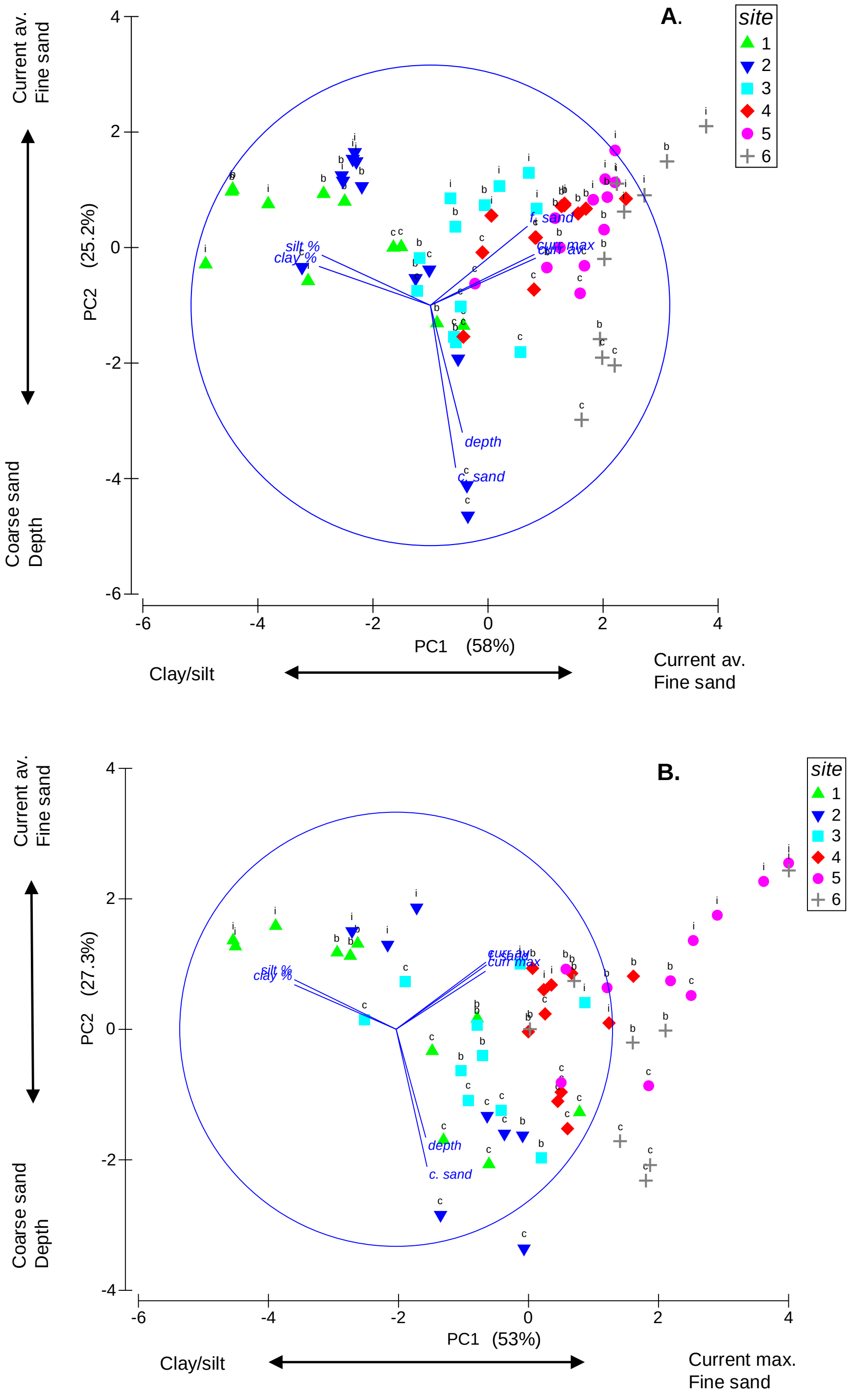
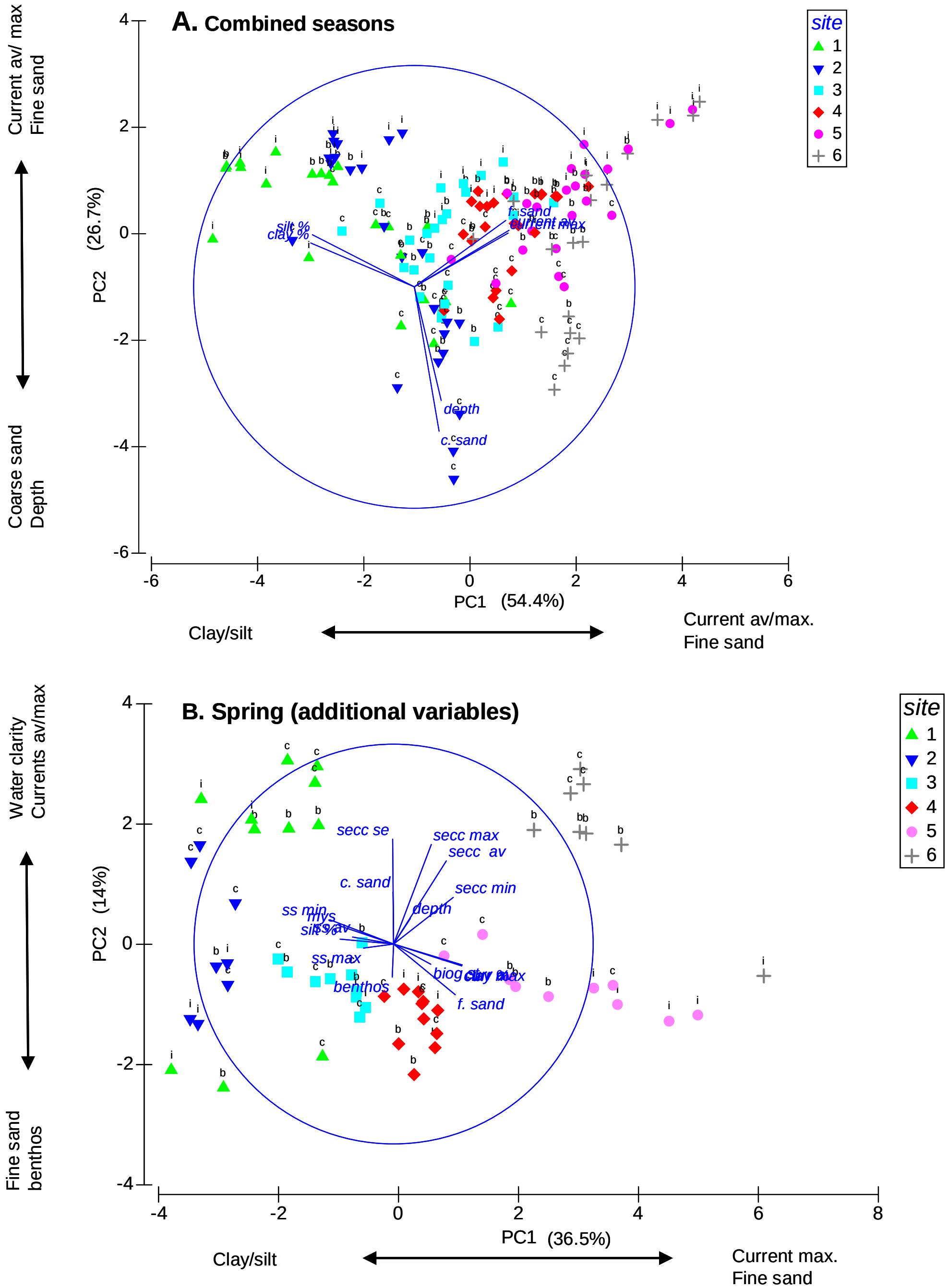


Figure 2.15 PCA score plots illustrating the associations of site and habitat relationships with the environmental matrix subset for Autumn (A) and Spring (B) with identical variables.

i= intertidal; b=bank; c=channel; c.sand=coarse sand; f.sand=fine sand; current av=average current speed; Current max=maximum current speed. Note: overlaid variables represent fine sand; current av & current max.



c. sand= coarse sand
f. sand= fine sand
silt = % silt
Clay=%clay
i=intertidal

curr av=average current speed
curr max=maximum current speed
benthos=prey biomass
biog st=biogenic structure
b=bank

sec av=secchi average
sec min=secchi minimum
sec max=secchi maximum
secc se=secchi standard error
c=channel

ss min=suspended sediments minimum
ss av=suspended sediments average
ss max=suspended sediments maximum
biog st=biogenic structure
mys=mysids

Figure 2.16 PCA score plots illustrating the association of site and habitat relationships with the environmental matrix subset for combined seasons with identical variables (A) and Spring with additional variables (B). Note: overlaid variables represent fine sand; current avg. & current max.

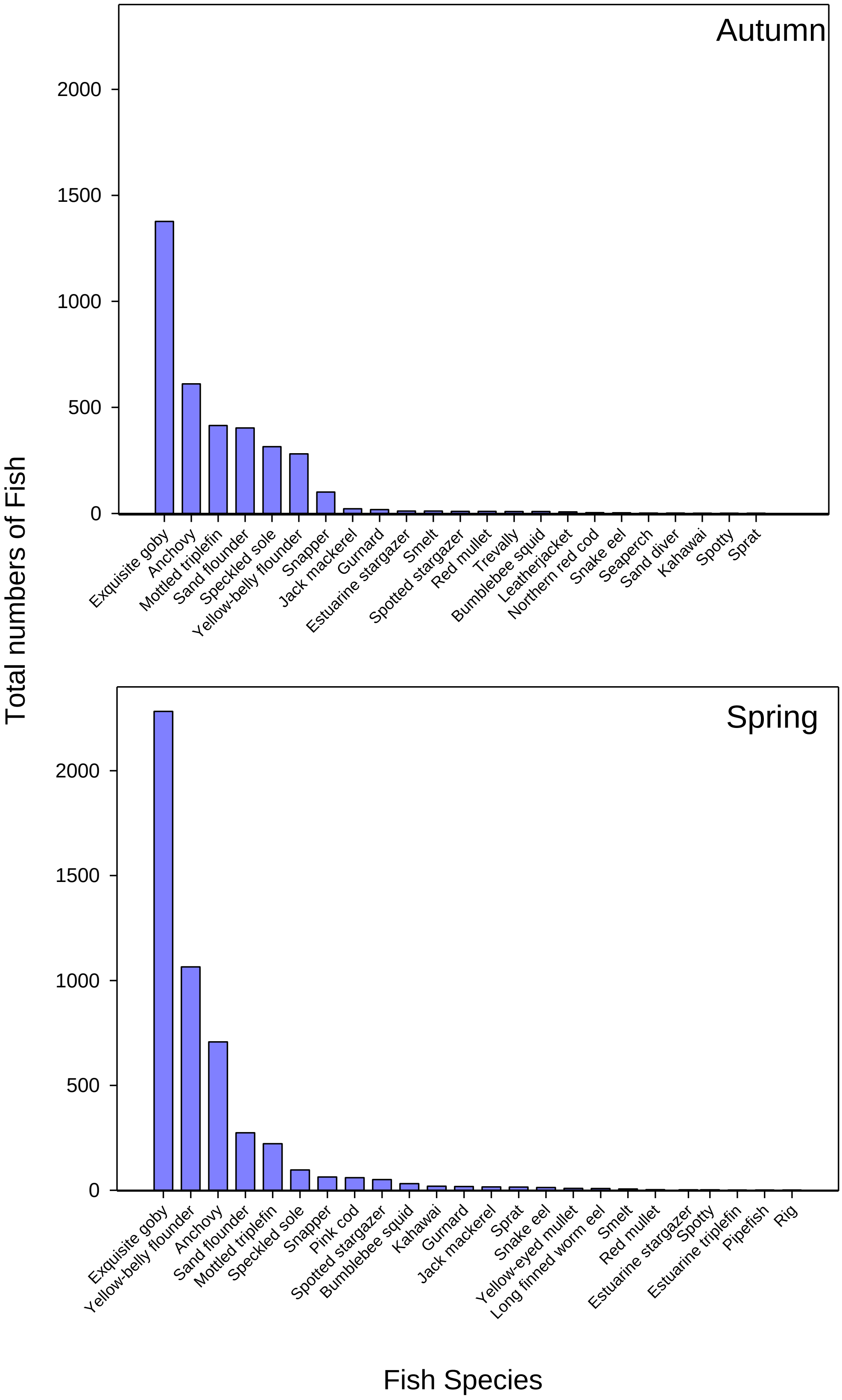


Figure 2.17 Total abundance of fish species caught by beam trawl at the 6 sites over two seasons.

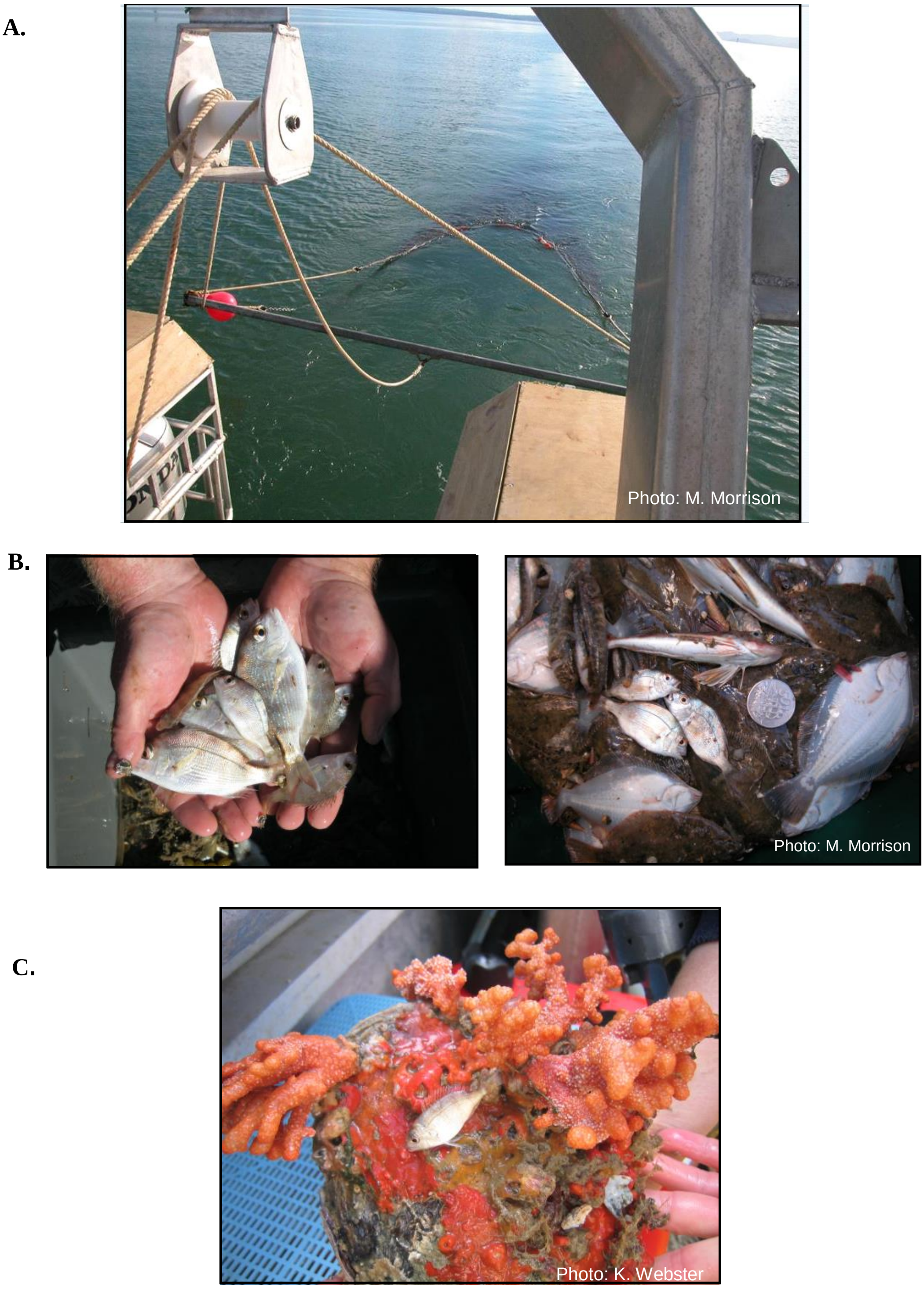


Photo: M. Morrison

Photo: M. Morrison

Photo: K. Webster

Figure 2.18 Beam trawl net deployed (A); examples of fish species (juvenile snapper, gurnard, flounder) caught within Pahurehure Inlet (B); biogenic structure (horse mussels; *Atrina zealandica*) utilized by juvenile fish as refuge (C).

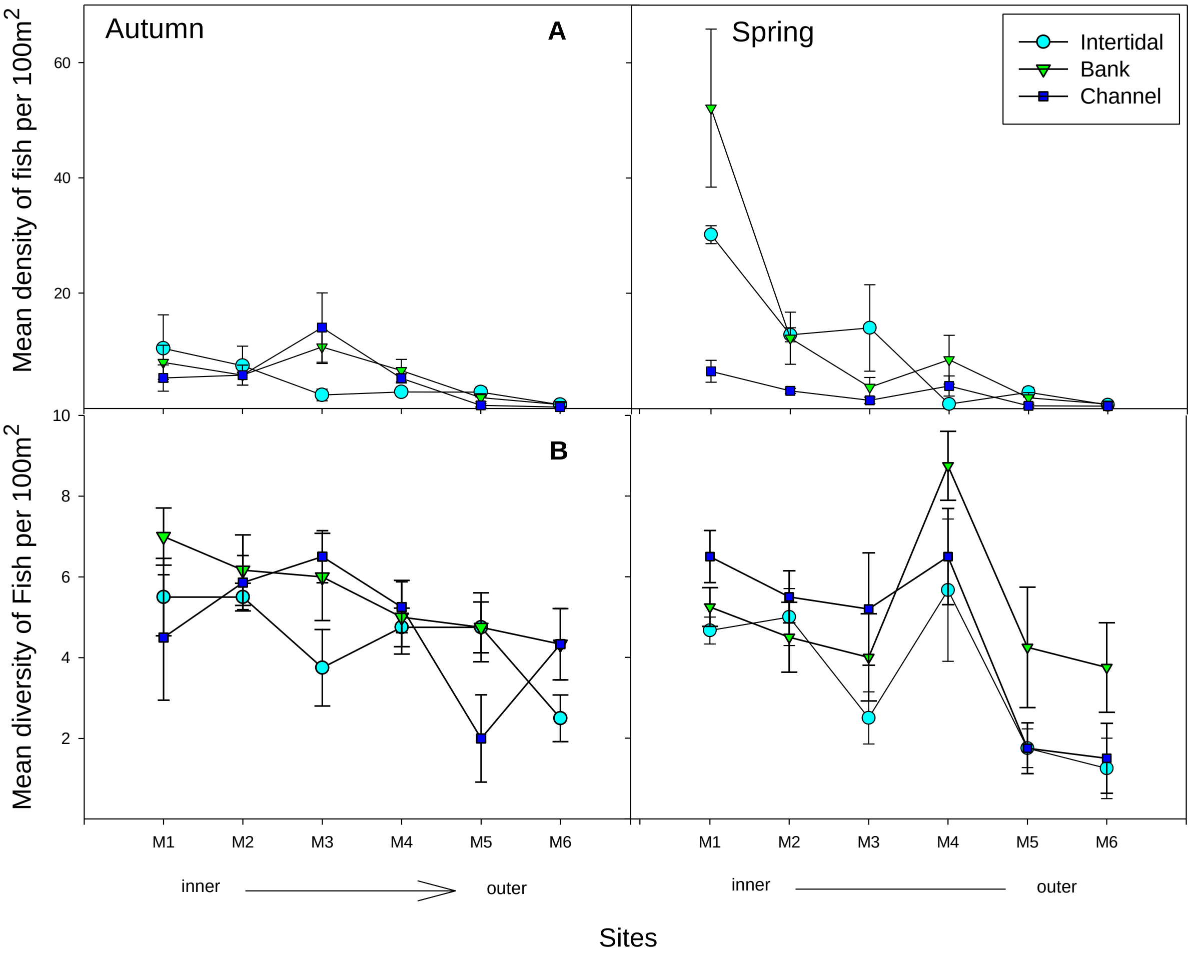


Figure 2.19 Mean density (±SE) (A), and diversity (B) of fish species caught by beam trawl from intertidal/bank/channel habitats at 6 sites over two seasons.

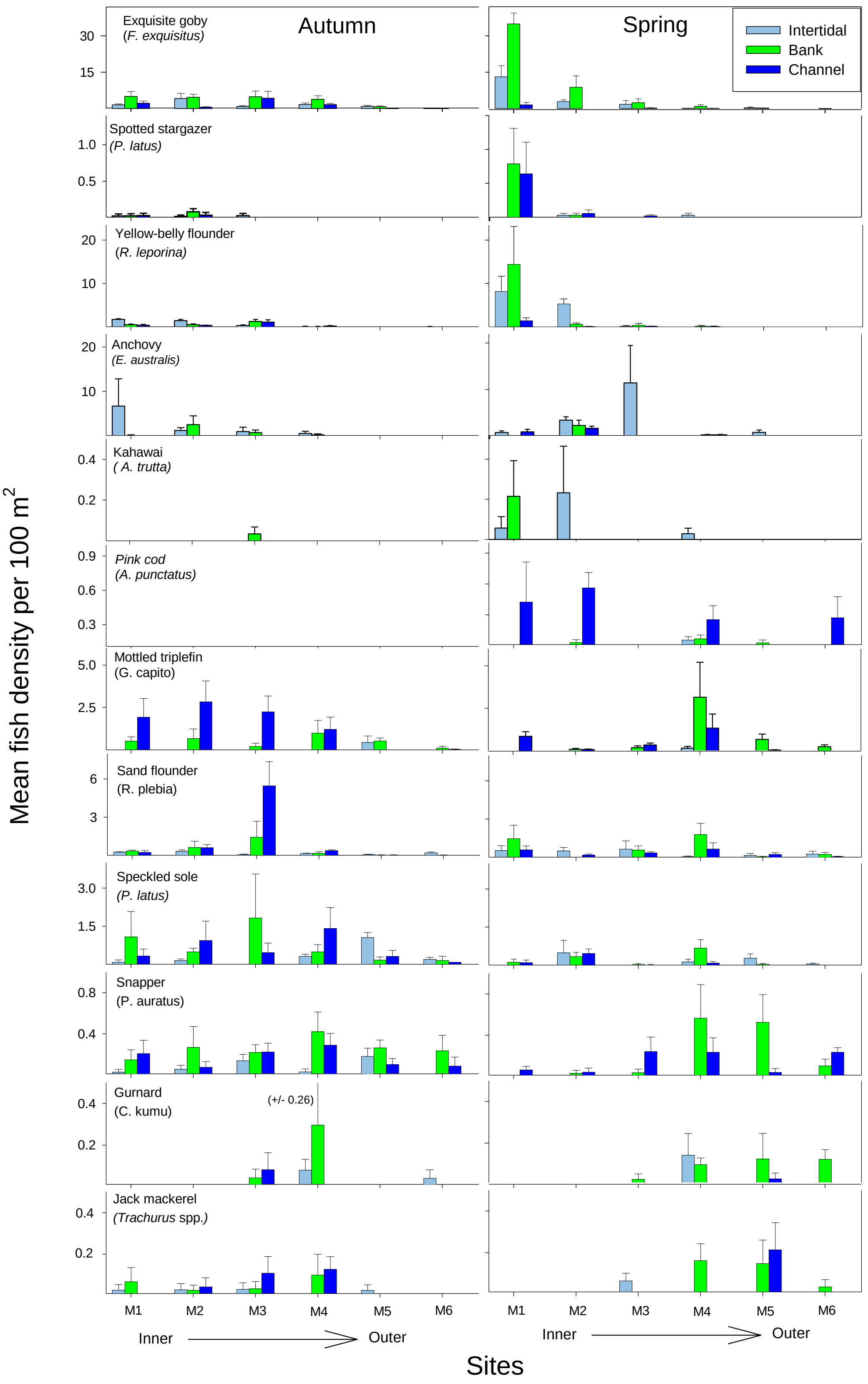


Figure 2.20 Mean abundance (±SE) of the twelve most common fish species collected by beam trawl from the intertidal/bank/channel habitats at 6 sites over two seasons.

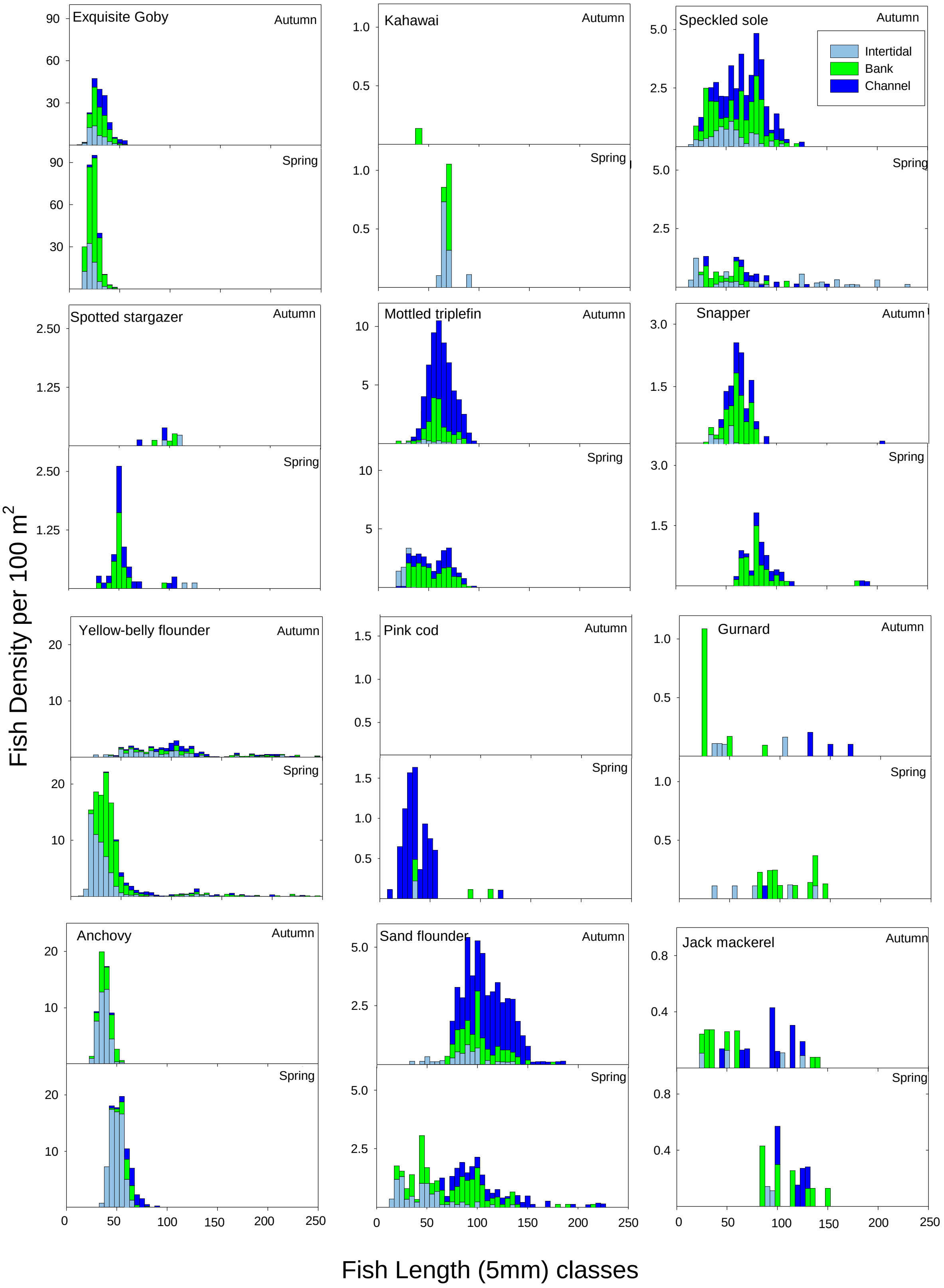


Figure 2.21 Length frequency of the twelve most common fish species collected by beam trawl from the intertidal/bank/channel habitats at 6 sites over two seasons.

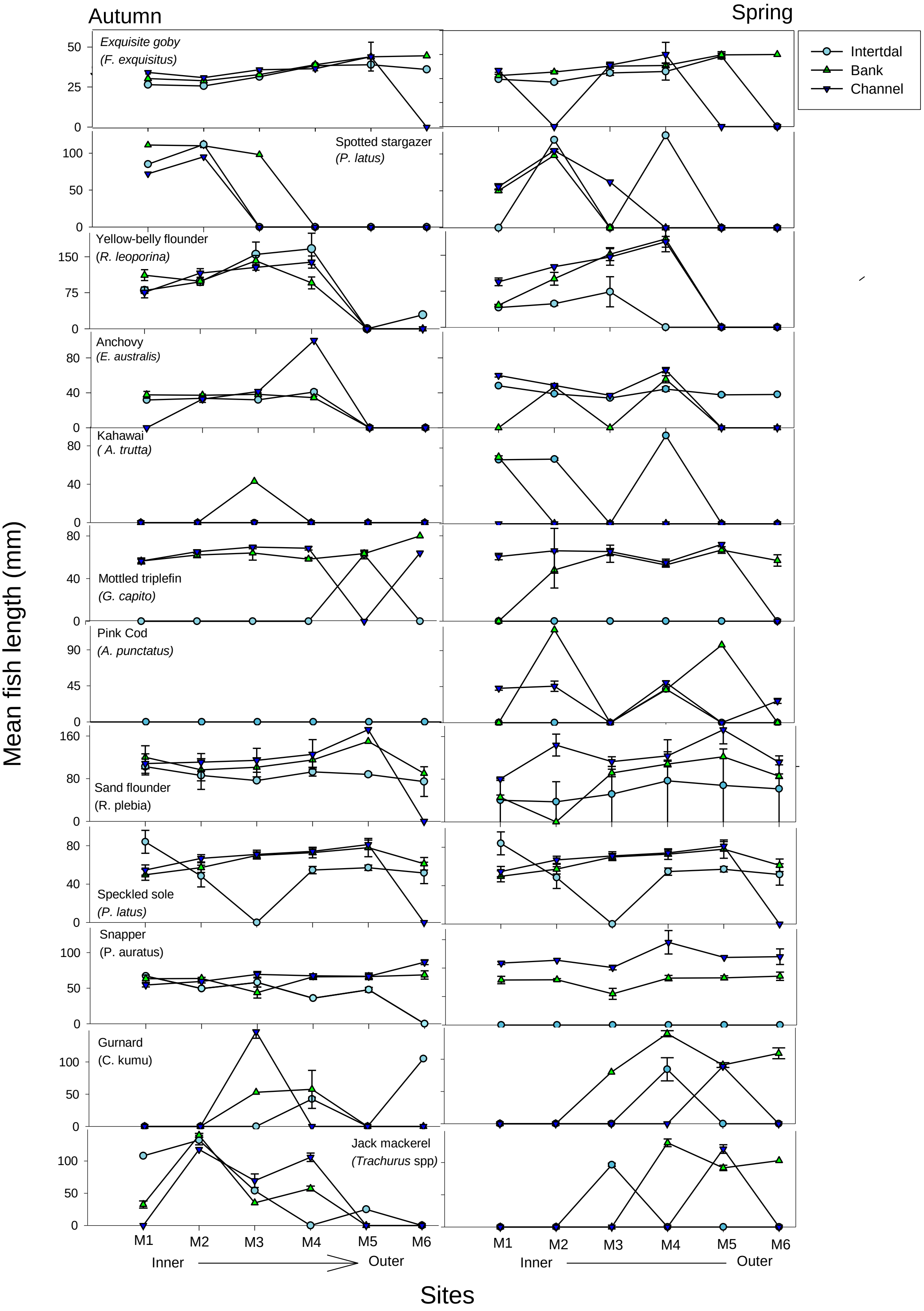


Figure 2.22 Mean Length frequency (±SE) of the twelve most common fish species collected by beam trawl from the intertidal/bank/channel habitats at 6 sites over two seasons.

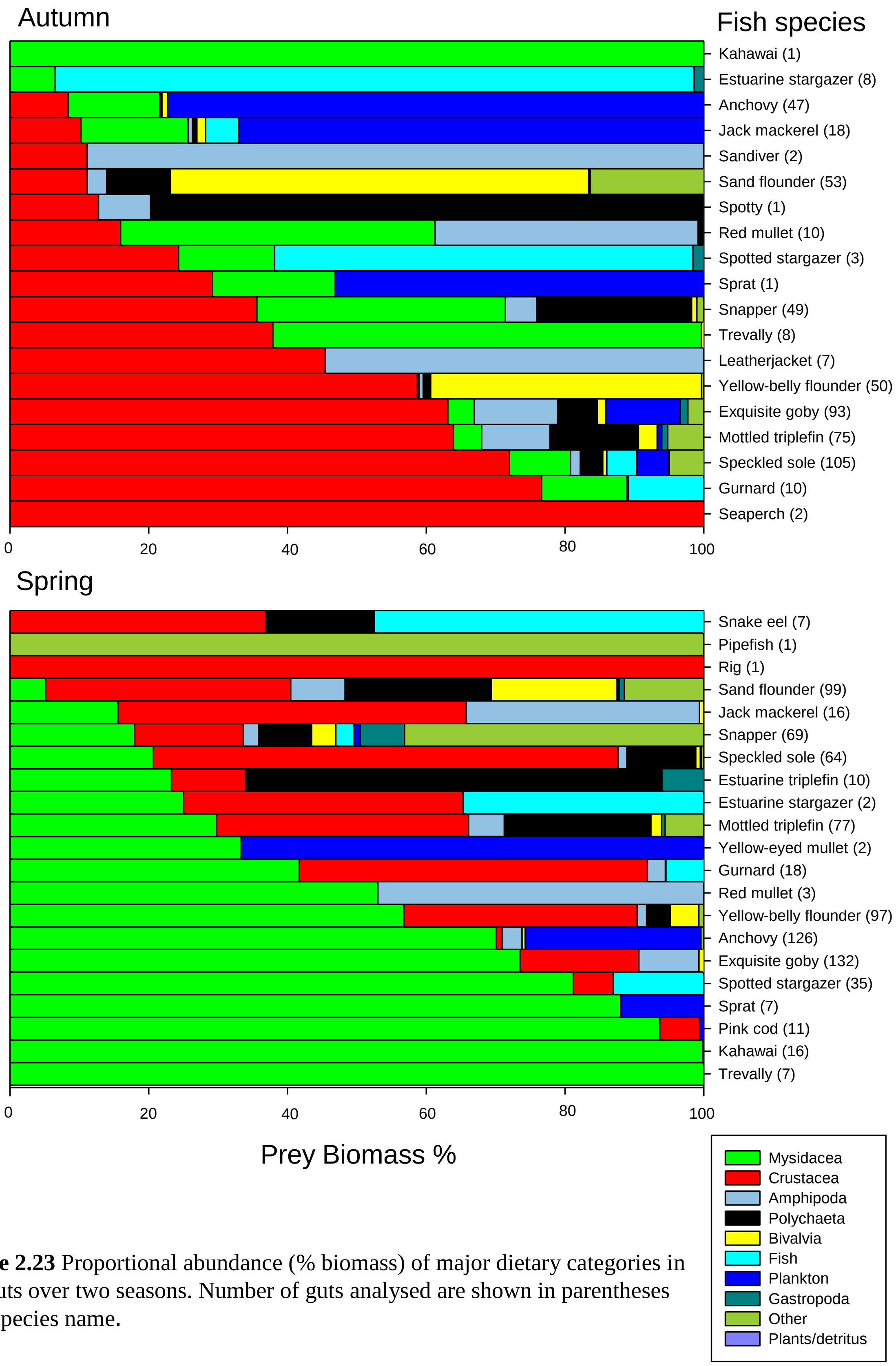


Figure 2.23 Proportional abundance (% biomass) of major dietary categories in fish guts over two seasons. Number of guts analysed are shown in parentheses after species name.

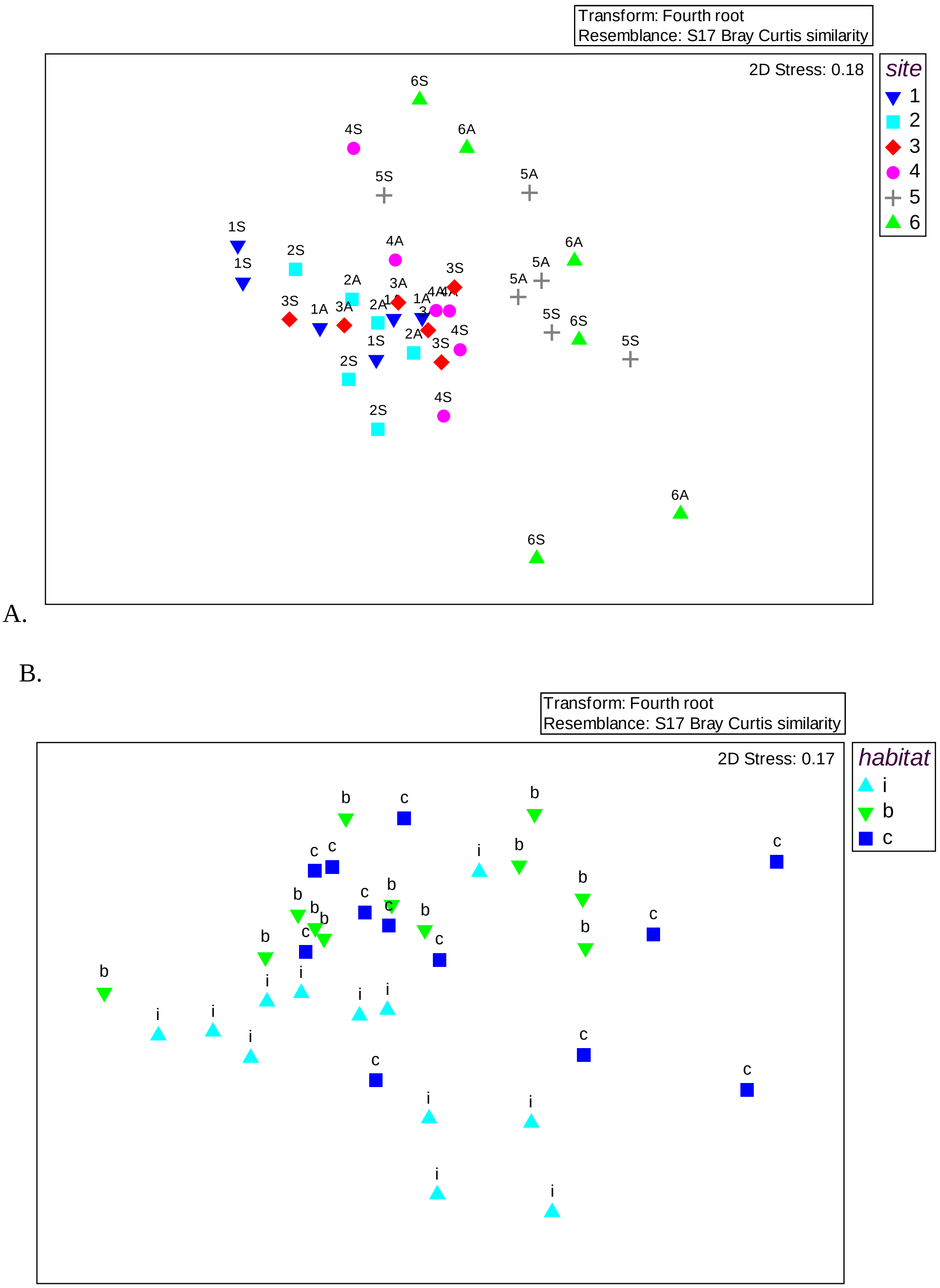
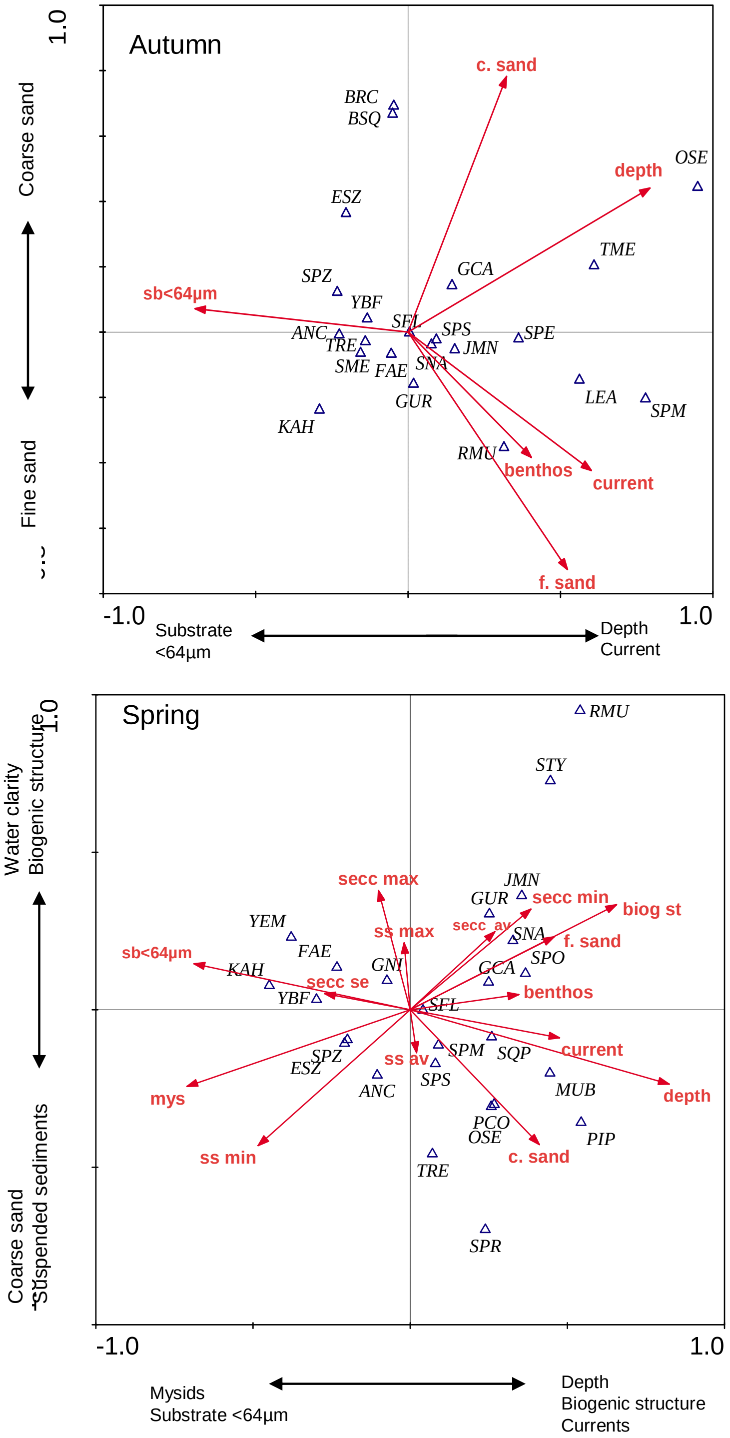


Figure 2.24 MDS ordination of fish assemblage data, plotted by site and season* (A), and habitat (B). *seasonal identifiers A=Autumn; S=Spring.



ANC Anchovy
ESZ Estuarine stargazer
FAE Exquisite goby
GCA Mottled triplefin
GNI

GUR Gurnard
JMN Jack mackerel
MUB
OSE
PCO Pink cod

PIP Pipefish
RMU
SNA Snapper
SPS Speckled sole
SFL Sand flounder

STY Spotty
SPR Sprat
SPO Dogfish
SPZ Spotted stargazer
SQP Bumblebee squid

(sb<64µm=% substrate silt/clay; biog st= biogenic structure; secc max=secchi maximum; secc av=secchi average; secc min=secchi minimum; secc se,=secchi standard error; mys=mysids; benthos=prey biomass; c. sand=coarse sand; f. sand=fine sand; ss max=maximum suspended sediments; ss av=average suspended sediments; ss min=minimum suspended sediments).

Figure 2.25 CCA ordination diagram based on species abundances, with biotic and abiotic environmental factors represented by vectors.

Table 2.1 Physical and biological variables sampled at each site. The levels of each variable are shown in brackets.

Variable type	**Variable name**	**Description**
Season	*Season*	Samples collected over two seasons a. Autumn b. Spring
Site location	*Station*	Position in Harbour down 20km gradient (M1 to M6)
	Habitat zone	Habitat zone within each station a. intertidal (i), b. bank (b) c. channel (c)
Site physical	*Depth*	Average depth per tow (m)
	*Clarity**	Water clarity at each site measured by Secchi disc (m) a. average b. maximum c. minimum
	Current speed	Current speed (m s^{-1}) a. average for normal condition b. maximum speed Estimated from Pritchard et al., (2008) model
	*Suspended sediments**	Suspended sediment concentration (g/m^3) a. intertidal/ subtidal bank (1m above seabed) b. channel (1m above seabed; 1m from sea surface)
	Substratum	Volumetric composition (%) of surficial sediments (±SE) clay, silt, fine sand, coarse sand
Site biological	*Biogenic habitat**	Presence of biogenic structure (horse mussels, sponges, mussels) a. none b. present c. common d. very abundant
	Benthic prey	a. Estimated biomass of aggregated prey categories (g AFDW m^{-2}) from benthic grab samples (amphipods; bivalves; crustacean; gastropods; mysids[1]; polychaetes; other) b. Mysids*, mean density per beam trawl (±SE)
	Predators	Estimated mean biomass (g) of predator guild fishes per gill net (±SE)

* only sampled in spring

[1] mysid biomass was excluded from the prey categories owing to extremely low numbers (n=3) being collected in the autumn grab samples due to gear avoidance/patchiness. Mysid abundance was subsequently estimated as bycatch from beam trawls for spring sampling and included in the analysis.

Table 2.2 Prey categories used in trophic analyses

Category	**Description**
Amphipods	Predominantly gammarids
Bivalves	All bivalves including siphons
Crustacea	All crustaceans excluding copepods, decapod zoeae, mysids, amphipods
Gastropods	
Mysids	All mysid shrimps
Plankton	Calanoid/harpacticoid/cyclopoid copepods; cladocerans & barnacle cyprids; decapod zoeae
Polychaeta	
Other	Nematodes, oligochaetes, ophiuroids, insects & eggs
Fishes	All fishes and fish remains, including larvae
Plants/detritus	

Table 2.3 Mean abundance (±SE) of predator guild fishes collected in gillnets from bank/channel habitats at 6 sites over two seasons.

Stations		**M1**		**M2**		**M3**		**M4**		**M5**		**M6**		**Total**
Species		**AUT**	**SPR**	**AUT**	**SPR**	**AUT**	**SPR**	**AUT**	**SPR**	**AUT**	**SPR**	**AUT**	**SPR**	
Kahawai	av	2.00	–	2.67	1.11	2.67	–	1.67	0.33	9.25	0.33	0.33	3.33	23.69
Arripus trutta	se	(0)	–	(2.67)	(0.18)	(1.45)	–	(1.20)	(0.33)	(7.74)	(0.33)	(0.33)	(2.85)	
Rig		–	0.50	6.00	–	1.33	–	4.67	1.00	3.75	3.67	1.00	4.00	25.92
Mustelus lenticulatus		–	(0)	(1.53)	–	(0.88)	–	(1.20)	(0)	(1.36)	(1.33)	(0.58)	(0.58)	
Gurnard		–	1	–	2	1	3	1	4	0.75	5	1	6	24.75
Chelidonichthys kumu		–	(1.50)	(1.50)	(1.50)	(0)	(1.27)	(0.58)	(0.33)	(0.55)	(1.67)	(0)	(1.00)	
Snapper		0.67	–	–	–	1.67	0.33	0.67	6.00	0.75	3.33	0.33	5.67	19.42
Pagrus auratus		(0.33)	–	–	–	(0.33)	(0.00)	(0.67)	(2.00)	(0.55)	(2.03)	(0.33)	(5.67)	
School shark		–	–	–	–	0.67	–	1.00	–	1.75	–	4.33	–	7.75
Galeorhinus galeus		–	–	–	–	(0.67)	–	(0.58)	–	(0.55)	–	(0.67)	–	
Hammerhead shark		–	–	–	–	–	–	–	–	–	–	0.33	–	0.33
Galeorhinus galeus		–	–	–	–	–	–	–	–	–	–	(0.33)	–	
Baracouta		–	0.50	–	–	–	0.33	–	–	–	–	–	–	0.83
Thyrsites atun		–	(0.50)	–	–	–	(0.33)	–	–	–	–	–	–	
Trevally		0.33	–	1.00	–	2.67	–	2.33	3	0.25	3.67	3.00	3.33	19.58
Pseudocaranx dentex		(0.33)	–	(0.58)	–	(1.33)	–	(0.33)	(1.00)	(0.29)	(0.67)	(1.73)	(2.85)	
Total		3.00	2.00	9.67	3.11	10.00	3.67	11.33	14.33	16.50	16.00	10.33	22.33	122.27

Table 2.4 Biogenic structure (measured volumetrically as bycatch) from beam trawls

Habitat	**Site**					
	M1	M2	M3	M4	M5	M6
Intertidal	nil	nil	nil	nil	nil	nil
Bank	nil	nil	nil	HY *** HM*** SP***	SP*** HM*** GM*	SP** HM**
Channel	nil	nil	nil	HM*** SP**	nil	SP* HM*

SP=Sponges; HM=Horse mussels; HY=Hydroids; GM=Green lipped mussels
Absent: nil, Present:*, Common:**(25- 49L), Very abundant:*** (50-75L)

Table 2.5 Eigenvectors for variables used in Principal Component Analysis (PCA) of site data for subset of environmental variables, indicating strength of the correlations between variables and principal components. (Coefficients in bold indicate that the variable makes an important contribution to the canonical axis).

Analysis 1:

Autumn **A.**

Variable	**PC1**	**PC2**
Depth	0.134	**-0.530**
Current maximum	**0.435**	0.212
Current average	**0.440**	0.197
% Clay	**-0.466**	0.162
%Silt	**-0.454**	0.209
% Fine sand	**0.406**	**0.329**
% Coarse sand	0.106	**-0.676**

Spring **B.**

Variable	**PC1**	**PC2**
Depth	0.137	**-0.499**
Current maximum	**0.413**	0.268
Current average	**0.416**	**0.309**
% Clay	**-0.470**	0.205
%Silt	**-0.471**	0.229
% Fine sand	**0.418**	**0.297**
% Coarse sand	0.143	**-0.634**

Combined seasons **C.**

Variable	**PC1**	**PC2**
Depth	0.122	**-0.514**
Current maximum	**0.426**	**0.243**
Current average	**0.430**	**0.256**
% Clay	**-0.470**	0.198
%Silt	**-0.462**	0.236
% Fine sand	**0.415**	**0.300**
% Coarse sand	0.112	**-0.653**

Table 2.6 Eigenvectors for variables used in Principal Component Analysis (PCA) of site data for Spring (additional) environmental variables, indicating strength of the correlations between variables and principal components.

Analysis 2:

Spring B

Variable	**PC1**	**PC2**
Mysid abundance	**-0.293**	0.179
Prey biomass	-0.003	**-0.236**
Depth	0.112	-0.077
Current speed average	**0.344**	-0.104
Current speed maximum	**0.322**	-0.053
Suspended sediments maximum	-0.149	-0.133
Suspended sediments minimum	**-0.321**	0.082
Suspended sediments average	-0.198	-0.101
Secchi maximum	0.186	**0.538**
Secchi minimum	**0.300**	0.232
Secchi average	0.270	0.385
Secchi standard error	0.001	**0.496**
% Silt	**-0.307**	0.165
% Clay	**-0.305**	0.168
% Fine sand	**0.317**	**-0.238**
% Coarse sand	0.041	0.051
Biogenic structure	0.193	-0.125

Table 2.7 Mean abundance per 100m2(±SE) of all fish species collected in the Manukau by beam trawl at six sites over two seasons. *int=intertidal; Bk=Bank; Ch= Channel.

Stations	**M1**						**M2**						**M3**					
Species	**Int**		**Bk**		**Ch**		**Int**		**Bk**		**Ch**		**Int**		**Bk**		**Ch**	
Exquisite goby	1.50	(0.43)	4.98	(2.05)	2.15	(0.97)	4.13	(2.10)	4.66	(1.27)	0.55	(0.22)	0.80	(0.44)	4.87	(2.39)	4.25	(2.90)
Favonigobius exquisitus	13.08	4.6731	34.92	4.5366	1.4765	1.0857	2.816	0.8292	8.786	4.727	–	–	1.713	1.649	2.412	1.6105	0.1943	0.1943
Yellow-belly flounder	1.70	(0.14)	0.45	(0.18)	0.35	(0.20)	1.41	(0.27)	0.49	(0.16)	0.29	(0.08)	0.29	(0.14)	1.07	(0.52)	1.33	(0.58)
Rhombosolea leporina	8.16	(3.48)	14.45	(8.73)	1.39	(0.66)	5.30	(1.10)	0.60	(0.28)	0.04	(0.04)	0.15	(0.15)	0.39	(0.35)	0.11	(0.05)
Anchovy	6.73	(6.13)	0.16	(0.09)	–	–	1.21	(0.67)	2.55	(1.98)	0.06	(0.06)	0.98	(0.98)	0.74	(0.59)	0.04	(0.04)
Engraulis australis	0.73	(0.36)	–	–	0.88	(0.57)	3.38	(0.74)	2.25	(1.14)	1.64	(0.48)	11.43	(8.05)	–	–	0.03	(0.03)
Sand flounder	0.26	(0.03)	0.35	(0.07)	0.24	(0.14)	0.33	(0.12)	0.62	(0.49)	0.61	(0.25)	0.07	(0.04)	1.43	(1.26)	5.47	(1.91)
Rhombosolea plebeia	0.54	(0.39)	1.46	(1.07)	0.59	(0.31)	0.50	(0.28)	–	–	0.17	(0.09)	0.65	(0.65)	0.57	(0.33)	0.35	(0.09)
Mottled triplefin	–	–	0.51	(0.26)	1.92	(1.11)	–	–	0.67	(0.57)	2.84	(1.23)	–	–	0.19	(0.19)	2.24	(0.95)
Grahimina capito	–	–	–	–	0.85	(0.27)	–	–	0.07	(0.07)	0.06	(0.04)	–	–	0.17	(0.10)	0.33	(0.12)
Speckled sole	0.09	(0.09)	1.08	(1.00)	0.33	(0.27)	0.16	(0.07)	0.50	(0.14)	0.94	(0.76)	–	–	1.83	(1.72)	0.47	(0.37)
Peltorhamphus latus	–	–	0.12	(0.12)	0.10	(0.10)	0.49	(0.49)	0.34	(0.17)	0.46	(0.18)	–	–	0.04	(0.04)	0.02	(0.02)
Snapper	0.03	(0.03)	0.15	(0.10)	0.21	(0.13)	0.05	(0.04)	0.27	(0.20)	0.07	(0.06)	0.14	(0.06)	0.22	(0.07)	0.22	(0.09)
Pagrus auratus	–	–	–	–	0.06	(0.04)	–	–	0.03	(0.03)	0.04	(0.04)	–	–	0.04	(0.04)	0.24	(0.14)
Spotted stargazer	0.03	(0.03)	0.03	(0.03)	0.03	(0.03)	0.02	(0.02)	0.08	(0.04)	0.04	(0.04)	0.03	(0.03)	–	–	–	–
Genyagnus monoterygius	–	–	0.79	(0.53)	0.64	(0.47)	0.03	(0.03)	0.03	(0.03)	0.06	(0.06)	–	–	–	–	0.02	(0.02)
Pink cod	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Auchenoceros punctatus	–	–	–	–	0.43	(0.39)	–	–	0.03	(0.03)	0.56	(0.15)	–	–	–	–	–	–
Bumblee squid	–	–	–	–	–	–	–	–	0.05	(0.04)	0.17	(0.08)	–	–	–	–	–	–
Sepioloidea pacifica	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.05	(0.03)	0.02	(0.02)
Jack mackerel	0.03	(0.03)	0.07	(0.07)	–	–	0.03	(0.03)	0.03	(0.03)	0.04	(0.04)	0.03	(0.03)	0.03	(0.03)	0.11	(0.08)
Trachurus spp	–	–	–	–	–	–	–	–	–	–	–	–	0.06	(0.04)	–	–	–	–
Gurnard	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.04	(0.04)	0.08	(0.08)
Chelidonichthys kumu	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.03	(0.03)	–	–
Kahawai	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.03	(0.03)	–	–
Arripus trutta	0.06	(0.06)	0.21	(0.18)	–	–	0.23	(0.23)	–	–	–	–	–	–	–	–	–	–
Sprats	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Sprattus spp.	–	–	–	–	–	–	0.24	(0.24)	–	–	0.06	(0.06)	–	–	–	–	–	–
Snake eel	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Ophisurus serpens	–	–	–	–	0.03	(0.03)	–	–	0.09	(0.09)	0.03	(0.03)	–	–	–	–	0.03	(0.03)
Trevally	0.06	(0.03)	–	–	–	–	–	–	0.02	(0.02)	–	–	–	–	0.06	(0.04)	0.07	(0.06)
Pseudocaranx dentex	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.09	(0.05)
Red mullet	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Upeneichthys lineatus	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Estuarine stargazer	–	–	0.14	(0.10)	0.06	(0.06)	0.06	(0.04)	–	–	0.04	(0.03)	–	–	–	–	–	–
Leptoscopus macropygus	–	–	0.03	(0.03)	–	–	–	–	–	–	0.03	(0.03)	–	–	–	–	–	–
Smelt	–	–	0.09	(0.09)	–	–	0.02	(0.02)	0.07	(0.04)	–	–	–	–	–	–	–	–
Retropina retropina	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Yellow-eyed mullet	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Aldrichetta forsteri	0.06	(0.06)	0.14	(0.07)	0.04	(0.04)	–	–	–	–	–	–	–	–	–	–	–	–
Long-finned worm eel	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Muraenichthys breviceps	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Leatherjacket	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Parika scaber	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Bastard red cod	–	–	–	–	–	–	–	–	0.02	(0.02)	0.12	(0.08)	–	–	–	–	–	–
Pseudophycis breviuscula	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Spotty	–	–	–	–	–	–	–	–	–	–	0.02	(0.02)	–	–	–	–	–	–
Notolabrus celidotus	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Sand diver	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Tewara cranwellae	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Seaperch	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Helicolenus percoides	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Rig	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Mustelus lenticulatus	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Pipefish sp.	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
						–												
Estuarine triplefin	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Grahamina nigripenne	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.03	(0.03)	–	–
Autumn Total																		
Spring total																		
Total av. per 100m²																		

Table 2.7 continued…

Stations	**M4**						**M5**						**M6**						
Species	**Int**		**Bk**		**Ch**		**Int**		**Bk**		**Ch**		**Int**		**Bk**		**Ch**		**Total**
Exquisite goby	1.65	(0.67)	3.79	(1.51)	1.61	(0.58)	0.86	(0.49)	0.87	(0.40)	0.06	(0.06)	0.04	(0.04)	0.05	(0.05)	–	–	1377
Favonigobius exquisitus	0.079	0.0491	0.898	0.6827	0.0825	0.0825	0.33	0.3305	0.1897	0.1514	–	–	–	–	0.035	0.0352	–	–	2283
Yellow-belly flounder	0.05	(0.05)	0.05	(0.05)	0.16	(0.12)	–	–	–	–	–	–	0.03	(0.03)	–	–	–	–	281
Rhombosolea leporina	–	–	0.16	(0.16)	0.08	(0.08)	–	–	–	–	–	–	–	–	–	–	–	–	1065
Anchovy	0.55	(0.49)	0.24	(0.24)	0.05	(0.05)	–	–	–	–	–	–	–	–	–	–	–	–	281
Engraulis australis	0.11	(0.00)	0.23	(0.11)	0.23	(0.11)	0.76	(0.49)	–	–	–	–	0.03	(0.03)	–	–	–	–	707
Sand flounder	0.15	(0.04)	0.17	(0.13)	0.38	(0.07)	0.06	(0.04)	0.04	(0.04)	0.03	(0.03)	0.21	(0.08)	0.02	(0.02)	–	–	403
Rhombosolea plebeia	0.06	(0.06)	1.79	(0.88)	0.65	(0.49)	0.15	(0.15)	0.04	(0.04)	0.17	(0.13)	0.26	(0.22)	0.23	(0.16)	0.04	(0.04)	274
Mottled triplefin	–	–	0.99	(0.75)	1.21	(0.73)	0.43	(0.39)	0.63	(0.17)	–	–	–	–	0.11	(0.11)	0.03	(0.03)	415
Grahimina capito	–	–	3.14	(2.06)	1.33	(0.83)	–	–	0.66	(0.31)	0.03	(0.03)	–	–	0.23	(0.11)	–	–	222
Speckled sole	0.32	(0.09)	0.50	(0.29)	1.41	(0.82)	1.06	(0.20)	0.21	(0.15)	0.32	(0.24)	0.21	(0.08)	0.16	(0.16)	–	–	315
Peltorhamphus latus	0.14	(0.11)	0.68	(0.34)	0.09	(0.06)	0.29	(0.17)	0.04	(0.04)	–	–	0.06	(0.03)	–	–	–	–	97
Snapper	0.03	(0.03)	0.42	(0.19)	0.29	(0.11)	0.18	(0.08)	0.26	(0.09)	0.10	(0.06)	0.00	(0.00)	0.23	(0.15)	0.09	(0.09)	101
Pagrus auratus	–	–	0.57	(0.33)	0.23	(0.14)	–	–	0.53	(0.27)	0.04	(0.04)	–	–	0.10	(0.07)	0.04	(0.04)	63
Spotted stargazer	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	10
Genyagnus monoterygius	0.03	(0.03)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	51
Pink cod	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
Auchenoceros punctatus	0.06	(0.03)	0.07	(0.04)	0.25	(0.14)	–	–	0.03	(0.03)	–	–	–	–	–	–	0.27	(0.20)	60
Bumblee squid			–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	9
Sepioloidea pacifica	0.09	(0.05)	0.48	(0.25)	0.30	(0.13)	–	–	–	–	–	–	–	–	–	–	0.02	(0.02)	32
Jack mackerel	–	–	0.10	(0.10)	0.13	(0.06)	0.03	(0.03)	–	–	–	–	–	–	–	–	–	–	22
Trachurus spp	–	–	0.16	(0.08)	–	–	–	–	0.15	(0.11)	0.21	(0.13)	–	–	0.04	(0.04)	–	–	16
Gurnard	0.08	(0.05)	0.30	(0.27)	–	–	–	–	–	–	–	–	0.04	(0.04)	–	–	–	–	18
Chelidonichthys kumu	0.14	(0.11)	0.10	(0.03)	–	–	–	–	0.12	(0.12)	0.03	(0.03)	–	–	0.12	(0.05)	–	–	18
Kahawai	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1
Arripus trutta	0.03	(0.03)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	19
Sprats	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1
Sprattus spp.	0.03	(0.03)	0.13	(0.09)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	15
Snake eel	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	3
Ophisurus serpens	–	–	–	–	0.14	(0.07)	–	–	–	–	–	–	–	–	–	–	0.04	(0.04)	13
Trevally	0.03	(0.03)	–	–	–	–	0.03	(0.03)	–	–	–	–	–	–	–	–	–	–	9
Pseudocaranx dentex	–	–	–	–	–	–	0.06	(0.06)	–	–	–	–	–	–	–	–	–	–	6
Red mullet	–	–	–	–	–	–	0.16	(0.12)	0.08	(0.08)	–	–	–	–	0.08	(0.08)	–	–	10
Upeneichthys lineatus	–	–	–	–	–	–	–	–	0.09	(0.05)	–	–	–	–	–	–	–	–	3
Estuarine stargazer	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	11
Leptoscopus macropygus	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2
Smelt	–	–	–	–	–	–	–	–	–	–	–	–	0.13	(0.13)	–	–	–	–	11
Retropina retropina	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
Yellow-eyed mullet	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
Aldrichetta forsteri	0.03	(0.03)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	9
Long-finned worm eel	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
Muraenichthys breviceps	–	–	0.06	(0.06)	0.18	(0.10)	–	–	–	–	–	–	–	–	–	–	–	–	8
Leatherjacket	–	–	0.03	(0.03)	–	–	0.03	(0.03)	0.08	(0.05)	–	–	–	–	0.08	(0.05)	0.04	(0.04)	8
Parika scaber	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
Bastard red cod	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	4
Pseudophycis breviuscula	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
Spotty	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1
Notolabrus celidotus	–	–	–	–	–	–	–	–	0.04	(0.04)	–	–	–	–	0.04	(0.04)	–	–	2
Sand diver	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.04	(0.04)	0.04	(0.04)	2
Tewara cranwellae	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
Seaperch	–	–	–	–	–	–	–	–	0.04	(0.04)	–	–	–	–	0.04	(0.04)	–	–	2
Helicolenus percoides	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
Rig	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
Mustelus lenticulatus	–	–	0.03	(0.03)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1
Pipefish sp.	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
	–	–	–	–	0.03	(0.03)	–	–	–	–	–	–	–	–	–	–	–	–	1
Estuarine triplefin	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0
Grahamina nigripenne	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1
Autumn Total																			**3624**
Spring total																			**4968**
Total av. per 100m ²																			**8592**

Table 2.8 R-values and significance levels (p) of ANOSIM comparisons between fish assemblages of the 6 sites sampled. Non-significant pair-wise comparisons are given in bold.

Site	M1	M2	M3	M4	M5	M6
M1						
M2	**0.017** **0.266**					
M3	0.1 0.003	0.129 0.003				
M4	0.279 0.001	0.199 0.001	0.114 0.001			
M5	0.429 0.001	0.415 0.001	0.205 0.001	0.135 0.002		
M6	0.47 0.001	0.528 0.001	0.265 0.001	0.222 0.002	0.062 0.047	

Table 2.9 R-values and significance levels (p) of ANOSIM comparisons between fish assemblages of the 3 habitats sampled.

Habitat	**R**	**p**
intertidal, bank	0.107	0.001
intertidal, channel	0.183	0.001
bank, channel	0.086	0.001

Table 2.10 Results of SIMPER analysis for the average abundance of important discriminating species and their percent contribution to the average dissimilarity between sites.

	Average abundance by site					Average abundance by site			
Species	**M1**	**M2**	**Contrib%**	**Cum.%**	**Species**	**M4**	**M5**	**Contrib%**	**Cum.%**
Exquisite goby	1.46	1	15.14	15.14	Exquisite goby	0.77	0.45	13.43	13.43
Anchovy	0.57	0.78	12.13	27.28	Mottled triplefin	0.46	0.43	11.01	24.44
Yellow-belly flounder	1.18	0.77	11.5	38.77	Speckled sole	0.59	0.47	10.39	34.83
Mottled triplefin	0.47	0.4	9.96	48.74	Anchovy	0.42	0.1	9.69	44.52
Speckled sole	0.31	0.57	9.19	57.93	Sand flounder	0.59	0.25	9.63	54.15
Sand flounder	0.72	0.51	8.31	66.24	Snapper	0.4	0.44	8.75	62.91
Spotted stargazer	0.32	0.16	6.23	72.47	Gurnard	0.25	0.06	5.43	68.33
Snapper	0.24	0.2	5.54	78.01	Jack mackerel	0.19	0.16	5.28	73.61
Average dissimilarity = 51.44					Average dissimilarity = 70.42				
	M1	**M3**				**M1**	**M5**		
Exquisite goby	1.46	0.81	17.17	17.17	Yellow-belly flounder	1.18	0	18.3	18.3
Yellow-belly flounder	1.18	0.62	13.06	30.23	Exquisite goby	1.46	0.45	18.01	36.32
Anchovy	0.57	0.45	12.51	42.74	Anchovy	0.57	0.1	9.57	45.89
Sand flounder	0.72	0.7	10.01	52.75	Sand flounder	0.72	0.25	8.98	54.87
Mottled triplefin	0.47	0.4	9.6	62.35	Mottled triplefin	0.47	0.43	8.26	63.12
Speckled sole	0.31	0.27	6.9	69.25	Speckled sole	0.31	0.47	7.59	70.72
Snapper	0.24	0.38	6.58	75.83	Snapper	0.24	0.44	6.32	77.04
Spotted stargazer	0.32	0.05	5.74	81.57					
Average dissimilarity = 57.50					Average dissimilarity = 75.05				
	M2	**M3**				**M1**	**M6**		
Anchovy	0.78	0.45	14.54	14.54	Exquisite goby	1.46	0.1	22.13	22.13
Exquisite goby	1	0.81	14.17	28.72	Yellow-belly flounder	1.18	0.03	18.13	40.26
Sand flounder	0.51	0.7	10.58	39.29	Anchovy	0.57	0.03	9.62	49.88
Yellow-belly flounder	0.77	0.62	10.29	49.58	Sand flounder	0.72	0.37	8.11	57.99
Speckled sole	0.57	0.27	9.77	59.35	Mottled triplefin	0.47	0.21	7.93	65.92
Mottled triplefin	0.4	0.4	9.46	68.81	Speckled sole	0.31	0.22	5.93	71.85
Snapper	0.2	0.38	6.69	75.5	Snapper	0.24	0.23	5.3	77.15
Average dissimilarity = 61.25					Average dissimilarity = 82.28				
	M1	**M4**				**M2**	**M6**		
Yellow-belly flounder	1.18	0.18	15.85	15.85	Exquisite goby	1	0.1	16.7	16.7
Exquisite goby	1.46	0.77	14.08	29.92	Anchovy	0.78	0.03	14.12	30.82
Mottled triplefin	0.47	0.46	9.14	39.06	Yellow-belly flounder	0.77	0.03	13.61	44.44
Anchovy	0.57	0.42	8.43	47.49	Speckled sole	0.57	0.22	9.01	53.45
Speckled sole	0.31	0.59	8.02	55.51	Sand flounder	0.51	0.37	8.32	61.77
Sand flounder	0.72	0.59	6.88	62.4	Mottled triplefin	0.4	0.21	8.19	69.95
Snapper	0.24	0.4	6.13	68.53	Snapper	0.2	0.23	5.6	75.56
Average dissimilarity = 62.85					Average dissimilarity = 82.24				
	M2	**M4**				**M3**	**M6**		
Exquisite goby	1	0.77	11.97	11.97	Exquisite goby	0.81	0.1	17.3	17.3
Yellow-belly flounder	0.77	0.18	11.68	23.65	Sand flounder	0.7	0.37	12.49	29.79
Anchovy	0.78	0.42	10.94	34.59	Yellow-belly flounder	0.62	0.03	12.16	41.95
Mottled triplefin	0.4	0.46	9.53	44.11	Anchovy	0.45	0.03	10.94	52.89
Speckled sole	0.57	0.59	8.03	52.15	Mottled triplefin	0.4	0.21	8.77	61.66
Sand flounder	0.51	0.59	7.97	60.12	Snapper	0.38	0.23	8.36	70.02
Snapper	0.2	0.4	6.59	66.71	Speckled sole	0.27	0.22	7.15	77.17
Average dissimilarity = 62.74					Average dissimilarity = 81.60				
	M3	**M4**				**M4**	**M6**		
Exquisite goby	0.81	0.77	12.78	12.78	Exquisite goby	0.77	0.1	15.25	15.25
Anchovy	0.45	0.42	11.12	23.9	Speckled sole	0.59	0.22	10.6	25.84
Sand flounder	0.7	0.59	10.37	34.27	Sand flounder	0.59	0.37	9.61	35.45
Speckled sole	0.27	0.59	9.83	44.09	Mottled triplefin	0.46	0.21	9.61	45.06
Mottled triplefin	0.4	0.46	9.69	53.79	Anchovy	0.42	0.03	9.37	54.43
Yellow-belly flounder	0.62	0.18	9.67	63.46	Snapper	0.4	0.23	8.34	62.76
Snapper	0.38	0.4	7.53	70.99	Gurnard	0.25	0.13	5.91	68.68
					Pink cod	0.21	0.09	5.82	74.5
Average dissimilarity = 66.18					Average dissimilarity = 78.42				
	M3	**M5**				**M5**	**M6**		
Exquisite goby	0.81	0.45	15.2	15.2	Speckled sole	0.47	0.22	14.36	14.36
Sand flounder	0.7	0.25	11.95	27.16	Mottled triplefin	0.43	0.21	12.47	26.83
Yellow-belly flounder	0.62	0	11.67	38.83	Snapper	0.44	0.23	12.09	38.92
Anchovy	0.45	0.1	11.14	49.97	Sand flounder	0.25	0.37	11.43	50.35
Speckled sole	0.27	0.47	10.38	60.36	Exquisite goby	0.45	0.1	11.12	61.47
Mottled triplefin	0.4	0.43	10.14	70.49	Jack mackerel	0.16	0.03	5.23	66.69
Snapper	0.38	0.44	8.47	78.96	Gurnard	0.06	0.13	5.06	71.75
Jack mackerel	0.16	0.16	5.25	84.22	Anchovy	0.1	0.03	5.03	76.77
Average dissimilarity = 74.71					Average dissimilarity = 79.24				
	M2	**M5**							
Exquisite goby	1	0.45	14.3	14.3					
Yellow-belly flounder	0.77	0	13.68	27.98					
Anchovy	0.78	0.1	13.56	41.54					
Mottled triplefin	0.4	0.43	9.19	50.73					
Speckled sole	0.57	0.47	8.9	59.62					
Sand flounder	0.51	0.25	8.33	67.95					
Snapper	0.2	0.44	7.14	75.09					
Average dissimilarity = 70.04									

Table 2.11 Results of SIMPER analysis for the average abundance of important discriminating species and their percent contribution to the average dissimilarity for within site habitat differences.

	Average abundance by habitat			
Species	**intertidal**	**bank**	**Contrib%**	**Cum.%**
Exquisite goby	0.83	1.09	15.44	15.44
Anchovy	0.68	0.32	12.87	28.31
Yellow-belly flounder	0.56	0.5	11.6	39.91
Mottled triplefin	0.04	0.49	9.65	49.56
Sand flounder	0.49	0.49	9.11	58.67
Speckled sole	0.38	0.48	9.07	67.74
Snapper	0.14	0.41	7.92	75.65
Gurnard	0.07	0.15	3.75	79.41
Jack mackerel	0.08	0.14	3.11	82.51
Spotted stargazer	0.06	0.12	2.58	85.09
Trevally	0.05	0.05	1.62	86.72
Bumblebee squid	0.03	0.07	1.6	88.32
Red cod	0.03	0.04	1.35	89.67
Pink cod	0.03	0.05	1.26	90.93

Average dissimilarity = 67.13

Species	**intertidal**	**channel**	**Contrib%**	**Cum.%**
Exquisite goby	0.83	0.44	13.82	13.82
Anchovy	0.68	0.28	12.64	26.46
Mottled triplefin	0.04	0.66	11.03	37.49
Yellow-belly flounder	0.56	0.41	10.44	47.93
Sand flounder	0.49	0.61	9.59	57.52
Speckled sole	0.38	0.39	9.17	66.69
Snapper	0.14	0.38	6.98	73.68
Pink cod	0.03	0.2	4.45	78.13
Jack mackerel	0.08	0.12	3.36	81.49
Spotted stargazer	0.06	0.11	2.52	84.01
Snake eel	0	0.12	2.33	86.35
Gurnard	0.07	0.03	1.84	88.19
Bumblebee squid	0.03	0.07	1.61	89.8
Trevally	0.05	0.04	1.6	91.4

Average dissimilarity = 68.60

Species	**bank**	**channel**	**Contrib%**	**Cum.%**
Exquisite goby	1.09	0.44	16.2	16.2
Anchovy	0.49	0.66	10.65	26.84
Sand flounder	0.49	0.61	9.52	36.36
Yellow-belly flounder	0.5	0.41	9.16	45.51
Speckled sole	0.48	0.39	8.7	54.21
Snapper	0.41	0.38	7.75	61.96
Anchovy	0.32	0.28	7.36	69.32
Pink cod	0.05	0.2	4.27	73.58
Jack mackerel	0.14	0.12	3.6	77.18
Gurnard	0.15	0.03	3.11	80.3
SQP	0.11	0.10	3.00	83.3
Spotted stargazer	0.12	0.11	2.96	86.26
Snake eel	0.02	0.12	2.36	88.62
Trevally	0.05	0.04	1.42	90.04

Average dissimilarity = 68.60

Table 2.12 Results of SIMPER analysis for the average abundance of important discriminating species and their percent contribution to the average dissimilarity between seasons.

	Average abundance by season			
Species	**Autumn**	**Spring**	**Contrib%**	**Cum.%**
Exquisite goby	0.91	0.67	15.19	15.19
Anchovy	0.32	0.55	11.06	26.25
Yellow-belly flounder	0.5	0.47	10.53	36.78
Mottled triplefin	0.45	0.34	9.39	46.18
Speckled sole	0.54	0.28	9.33	55.51
Sand flounder	0.54	0.52	9.32	64.82
Snapper	0.41	0.21	7.59	72.41
Pink cod	0	0.2	3.71	76.11
Jack mackerel	0.12	0.1	3.32	79.43
Gurnard	0.06	0.11	2.87	82.3
Spotted stargazer	0.07	0.13	2.7	85
Bumblebee squid	0	0.13	1.97	86.97
Snake eel	0.02	0.08	1.62	88.59
Trevally	0.06	0.03	1.5	90.09

Average dissimilarity = 70.47

Table 2.13. Summary of BIOENV results based on Spearman rank correlations between fish assemblage structure and environmental variables.

Season	Best set	correlation ρ	P
Combined	de*curr av.	0.41	<0.01
Autumn	de*curr av.	0.49	<0.01
Spring*	de*curr av.	0.40	<0.01
	de*curr av.*sec min.	0.396	<0.01
	de*curr av.*sus sed max.	0.392	<0.01

* Additional variables sampled: suspended sediments/secchi.

de = depth, curr av. = average current speed, sec min. = secchi minimum,
sus sed max. = suspended sediments maximum

Table 2.14 Results of canonical correspondence analysis relating individual fish species abundance data to environmental variables for Autumn and Spring.

Autumn **A.**

		Canonical coefficient		
Environmental variable		Axis 1.		Axis 2.
Depth[1]		**0.5797**		0.3193
Current[1]		**0.4393**		-0.3072
% Substrate <64 μm*[1]		**-0.5105**		0.0522
%Fine sand		0.3819		**-0.5268**
%Coarse sand[1]		0.2358		**0.5666**
Benthos (prey biomass)		0.2951		-0.278
Axes	**1**	**2**	**3**	**4**
Eigenvalues	0.149	0.095	0.039	0.025
Cumulative percentage variance				
of species data explained	7.3	12	13.9	15.1
of species-environment relation explained	44.1	72.1	83.6	91.1

Spring
(additional variables) **B.**

		Canonical coefficient		
Environmental variable		Axis 1.		Axis 2.
Depth[1]		**0.7737**		-0.2023
Current[1]		**0.4470**		-0.0762
Mysids[1]		**-0.6670**		-0.2084
Suspended sediments (max)		-0.0182		0.1822
Suspended sediments (min)[1]		-0.4542		**-0.3694**
Suspended sediments (av)		0.0197		-0.1168
Secchi (max)[1]		-0.0944		**0.3248**
Secchi (min)		0.3609		0.2746
Secchi (av)		0.2533		0.2126
Secchi (std err)		-0.2558		0.0443
% Substrate <64 μm*[1]		**-0.6461**		0.1257
% Fine sand		0.43		0.1987
% Coarse sand[1]		0.3857		**-0.3668**
Biogenic structure[1]		**0.6167**		**0.2865**
Benthos (prey biomass)		0.3242		0.0417
Axes	**1**	**2**	**3**	**4**
Eigenvalues	0.381	0.229	0.146	0.088
Cumulative percentage variance				
of species data explained	15	24	29.7	33.2
of species-environment relation explained	34.2	54.8	67.9	75.8

*percentage of substrate comprised of silt/clay; [1] Factors retained by forward stepwise analysis as significantly contributing to the ordination ($p<0.05$)

Chapter Three

Effects of turbidity on juvenile snapper


Photo: C. Middleton

‘Newly recruited snapper Whangapoua Harbour, Coromandel’.

Chapter 3: Effects of turbidity on juvenile snapper

3.1 Introduction

Estuaries are important conduits between terrestrial watersheds and coastal waters, and are recognized as ‘transitional’ ecotones, linking terrestrial and marine habitats (Ray, 2005). Estuaries comprise the first and often ultimate receiving environment for contaminants generated on the land and transported by rivers and streams (Ellis et al., 2004; Scarsbrook, 2008), and thus are particularly vulnerable to anthropogenic land-based effects. While high suspended sediment loads are a ubiquitous natural feature of many estuarine waters, the rate and extent of terrestrial sediment delivery has accelerated in recent decades. Suspended sediments and associated decreases in water clarity from silt and clay erosion is now considered one of the most pervasive water pollution problems worldwide (Gray, 1997; Sorrenson et al., 1997; GESAMP 2001). Indeed, suspended sediment is a significant parameter in water quality monitoring, pollution control and environmental impact assessment (e.g. European Water Framework Directive (WFD, 2000/60/EC); Marine Strategy Framework Directive, (MSFD, 2008/56/EC); United States Clean Water Act, (2000), (cited in Borja et al., 2010).

3.1.1 Mechanisms of impact

Increases in sedimentation into the coastal zone can elicit a wide range of effects, both from deposition from fines onto the seafloor, and as suspended sediments in the water column. Ongoing re-suspension and deposition events (e.g. by storms and fishing gear) may shift sediments between these two states. Suspended sediments can affect many of the physical attributes of the environment, including increased turbidity, reduced visibility and depth of the photic zone and subsequent photosynthetic activity, food availability and plant biomass (Moore, 1977; Bruton, 1985; Fig. 3.1). Direct effects on fish species include reductions in visibility of pelagic food and clogging of gills, with associated acute and/or chronic impacts, e.g. immediate physiological stress and reduced growth rates and reproductive fitness. Indirect effects include the loss of important nursery habitats such as biogenic habitat formers via abrading, clogging and smothering organisms and reductions in prey assemblages (Thrush et al., 2004; see review Morrison et al., 2009). These effects do not act in isolation and may have synergistic outcomes.

Most of our current knowledge on the effects of suspended sediments on fish is based on freshwater (in particular salmonid) species. These studies have shown that elevated concentrations of suspended sediments can increase mortality and cause non-lethal adverse effects which include reduction in feeding rates, changes in type and/or ratio of prey consumed, changes in blood physiology and gill structure, avoidance of suspended sediments, increased respiration and cough responses, and increased susceptibility to diseases (see reviews of Bruton, 1985; Lloyd, 1987; Newcombe & MacDonald, 1991; Kerr, 1995; Newcombe & Jensen, 1996; Moore, 1997; Bash et al., 2001; Wilber & Clarke, 2001). Existing information on the effects of suspended sediment is largely based on acute exposure experiments, with little empirical information available on chronic responses to high concentrations for extended periods, or under natural field conditions. This is especially true especially for marine species (Au et al, 2004). Research on behavioural responses of estuarine fish to suspended sediment plumes is also largely missing. Studies referenced in this chapter have quantified suspended solids using a mixture of nephelometric turbidity units (NTU) and actual suspended sediment concentrations (mg/L); the two are not directly comparable through conversion to a common metric. Therefore, studies have been reported using the same units as the authors used. Explanations of terminology used in this chapter are included in Appendix 3.1.

3.1.2 New Zealand case studies

Within New Zealand, estuarine urbanization has been relatively recent, with many rural catchments having been entirely urbanized only in the last 30 years or less (Swales et al., 2002). The development of intensive agriculture, forestry and mining, coupled with steep catchments and easily erodible soils has resulted in sedimentation rates ten times higher than before humans arrived (Griffiths & Glasby, 1985; Hicks et al., 2000; Parkyn et al., 2002; Swales et al., 2003). Sedimentation is arguably now the most important land-based stressor in New Zealand. This includes both suspended sediment and deposition effects and associated decreases in water clarity (which may also be driven by eutrophication) (Morrison et al., 2009).

New Zealand freshwater

In New Zealand, evidence is emerging of sub-lethal concentrations of suspended sediments affecting the upstream migrations of native freshwater fish species, of which 70% are diadromous (McDowell 1990). Feeding experiments on migrant banded kokopu (*Galaxis fasciatus*) showed reduced feeding rates above 25 NTU, increased avoidance reactions, and decreased migration rate in natural streams (Boubeé et al., 1997; Rowe & Dean, 1998). Field studies suggest that turbidity occurs over this level during the whitebait migration season (August–December), for more than 10% of the time, which could reduce the upstream migration of banded kokopu, ultimately resulting in reduced recruitment of juveniles in turbid rivers (Richardson et al., 2001). Other field studies have noted reduced densities of juvenile kaoaro, inanga, and banded kokopu in highly turbid waters following floods (McDowall & Eldon, 1980), and within catchments containing high production pasture compared with forested streams (Hanchett, 1990; Minns, 1990; Rowe, 1999; Rowe et al., 1992). Research on 38 East Cape streams by Richardson & Jowett (2002) revealed fish abundance and diversity reduced from nine to two species as sediment loads increased. Streams with higher suspended sediments were shallower with swifter stream habitat, finer substrate and less fish cover (Richardson & Jowett, 2002).

3.1.3 Habitat degradation

Increased turbidity is now a characteristic of lowland reaches of many New Zealand rivers with suspended sediment concentrations primarily related to flow rate. This can be temporarily increased (over periods of months to years) by changes in land use such as conversion of forest to pasture, or by landslides following rainstorms (Hicks & Griffiths, 1992). Logging of forested catchments may result in changes to stream light levels, water temperatures, flow patterns, stream bank stability, bed characteristics, and also increase the size and frequency of floods (Morgan & Graynoth 1978). With New Zealand having one of the highest conversion rates of forestry to pasture in the world (over 60%), impacts on native fish species may be profound (McDowell, 2000). Reports of prodigious whitebait catches declining drastically over the past 100 years has been documented by Phillips (1924a cited in McDowell, 2000) with reported "cartloads" of whitebait coming from the Hutt River in the 1880's. Catches such as this were common from all over New Zealand.

Ultimately, these impacted waterways discharge into the coastal marine environment with concomitant broad scale changes to estuarine systems. Increasing deposition of fines (silt/clay) has been documented in many northern New Zealand estuaries with concurrent alterations of soft sediment macrobenthic community structure (Lohrer et al., 2004; 2006a; Norkko et al., 2002; Thrush et al., 2004) and increasing spread of mangroves (*Avicennia marina*) (Morrisey et al.,2007, 2010).

To date there is virtually no information on the direct effects of stressors such as sedimentation on fish in the New Zealand marine context, particularly for larval and juvenile estuarine fish which are thought to be the most susceptible of all life history stages to elevated levels of TSS (Sigler et al., 1984; Wilber & Clarke, 2001). This is an important area of research as events occurring in the first few months of life can strongly influence patterns of post-settlement survivorship (Sutherland & Meyer, 2007). Rapid growth is critically important for early juvenile fishes, which can suffer high mortality rates from predation which typically varies inversely with body size (Houde, 1987; Sogard, 1992; Francis, 1994; Manderson et al., 2002).

3.1.4 Aims

The aim of this chapter was to examine how habitat/environmental change may influence juvenile fish feeding and health of juvenile fishes. This study investigates the effects of increased turbidity/TSS on the ability of juvenile snapper to feed on live prey in laboratory tanks for short term (30 minute) trials. Similarly, impacts of longer term (30 day) exposure to elevated turbidity/TSS on snapper health and growth are assessed in the laboratory. The effect of changing substrate type on short term (30 minutes) foraging rates on live prey was also assessed. To compliment and link these laboratory results to field conditions, health and condition indices of juvenile snapper in seven northern New Zealand estuaries were investigated across a turbidity/TSS gradient.

Snapper (*Pagrus auratus*) were chosen as a model species as they are one of the most abundant inshore demersal finfish species in northern New Zealand, and comprise an important recreational and commercial fishery (Francis, 1994; Ministry of Fisheries Plenary Report, 2010). Due to its wide distribution, snapper are considered to be an adaptive species, yet recent research suggests that the current fishery is much reduced from historical conditions (Parsons et al., 2009). For some stocks (e.g. West Coast, SNA 8), catch rates are reliant on a limited number of strong year classes (Morrison et al, in review). Therefore, the assessment of juvenile condition in estuarine nurseries and the concomitant abiotic factors affecting juvenile health (such as suspended sediments) is an essential step towards sustainable management of these coastal fisheries.

3.2 Methods

Tank-based experiments were undertaken between May and September 2004 at the Leigh Marine Laboratory. Field based collections were carried out between May to July 2006 at seven northern North Island harbours (Fig. 3.2).

3.2.1 Laboratory experiments

Experimental animals

Juvenile snapper (*P. auratus*) between 50-90mm FL were mostly obtained from Rangaunu Harbour using beach seine for capture. However, technical difficulties with the oxygenation equipment during the five hour transportation to the Leigh Marine Laboratory resulted in the loss of some animals. Due to prohibitive transport costs, the remaining experimental animals required for short term trials were captured by potting using opera traps (see Fig. 3.3) within nearby Mahurangi Harbour. Fish were placed into flow through holding tanks (500 l) at the Leigh Marine Laboratory and were fed daily with diced mussels and shrimp pellets. After two weeks, fish were transferred to the experimental tanks and fed mysids daily to acclimatize.

Mysids (*Tenagomysis* sp.) were utilized for the experiments as they are a common prey item for juvenile snapper. Mysids were collected from a nearby mangrove creek in the Whangateau Harbour by means of a small plankton net (355µm mesh) during the incoming tide. Samples were collected from dusk onwards, predominantly at night. Mysids were held in a flow-through glass aquarium (60 x 35 x 30 cm) and were fed daily with frozen brine shrimp pellets and detritus off shells collected from the harbour.

3.2.2 Feeding Experiments

Experimental Protocol

Juvenile snapper (50-90 mm FL) were acclimatized to the experimental tanks for two hours and starved for 24 hours prior to each trial. At the start of the experiments, sixty mysids were released into the opposing end of the each tank which was separated from the fish by a perspex divider for 30 minutes to acclimatize, after which the divider was removed and fish left to feed. After 30 minutes of feeding, fish were removed, the tank was completely drained through a 250 µm sieve, and the remaining mysids counted. All mysids missing from the tanks were considered predated. Preliminary trials revealed that all shrimps were successfully recovered from tanks containing no fish.

Experiments were conducted in 15 rectangular plastic tanks (50 x 26 x 28 cm) located in a building with overhead ambient light which allowed the use of natural photoperiods. Tanks were covered with removable nets (10 mm mesh), which allowed the natural ambient light to penetrate while preventing egress of snapper from the tanks. All experiments were undertaken during late afternoon (1500-1700 h), as most shrimps and visual fish predator species have diel activity patterns (e.g. Minello, 1987; Macia et al., 2003). In all experiments, there were three replicate tanks, each tank containing three fish for each treatment. Each tank was a closed system with a submersible pump (Hi-tech 3500; 3.6L/min) and aerator. The pump was switched off two hours prior to the experiments.

Experiment 1: Effects of turbidity on foraging rates:

Predation rates on free swimming mysids were compared between five turbidity levels ≤10 (control), 20, 40, 80 and 160 nephelometric turbidity units (NTU), measured with a Hach 2100 Portable Turbidity sensor and are representative of the turbidity range experienced by juvenile snapper in their natural habitat in nearby Mahurangi Harbour (Oldman & Swales, 1999). Due to natural variation of incoming ambient seawater, the clear water control tank was labelled as ≤ 10 NTU as per Cyrus & Blaber (1987a, b).

Turbidity was produced by means of fine surficial estuarine sediments (<64μm) collected from the Mahurangi Estuary. Replicate samples of sediment were collected to encompass potential spatial variability in distributions of grain sizes. Sediment, along with surface water was allowed to settle. The container was then stirred and left for six minutes before the top 10cm of water and sediment was siphoned off and poured through a 64μm sieve. This was repeated until enough stock solution was obtained. Sediment was collected twice a week and kept in a well aerated container. Turbidity levels were adjusted in the tanks by the addition of varying amounts of stock solution to the ambient incoming seawater until the intended turbidity levels (i.e. 20, 40, 80 160 NTU) were achieved. To maintain water depth, a corresponding volume of water was removed after the addition of stock solution. Turbidity levels were measured at the conclusion of the feeding trials ~ 3 hours later. There was little discernible difference for the lower turbidity treatments (20, 40, 80 NTU). However, the 160 NTU treatments were up to 10% lower for some trials. To circumvent any substrate effects for the short and long-term turbidity experiments, tank bottoms were left clear without any sediment cover.

Experiment 2: Effects of substrates on foraging

To determine the effect of substrate type on predation rates of free swimming mysids, three substrates were chosen to be representative of local benthic conditions. These included mud, sand and sandy-mud mixture with no substrate as a control. Sediments were collected from the Mahurangi Harbour and processed using the same protocol as for Experiment 1. Sand was passed through a 250μm sieve to remove larger sized particles and debris, and was kept aerated before use. Substrate types were added to the tanks 24 hours prior to the trials to allow for settling. Turbidity levels were measured prior to the fish being added to the tank, during feeding and at the conclusion of feeding.

Experiment 3: Effects of turbidity on growth

Longer term tank experiments were conducted over one month utilizing the same experimental protocol (turbidity levels & replicates) as for the short term trials. Weight and length of each fish was measured at the start of the experiment and again at the conclusion.

Following preliminary trials, sediments were kept suspended by a closed recirculation system for each tank with three forward tipping cups (200ml) positioned over the top of each tank which provided the greatest surface area of disturbance on the bottom of the tank (adapted from Barr, 2007; Fig. 3.3A). Each cup pivoted independently which kept the overall level of disturbance to a minimum for the fish. However, trials revealed that there was a small area along the rear wall of the tank where the turbulence did not reach. To counteract this, a jug of seawater was poured to re-suspend this sediment daily. Turbidity levels were measured each day from random locations within the tank and additional sediment from the stock solution was added to maintain turbidity levels within 20% of the target level. Approximately 25% of the tank water was changed daily to allow addition of stock solution and to minimize nitrogen waste levels. To minimize water quality problems, water and sediment were completely replaced every seven days. Fish were removed and placed into holding tanks of ambient seawater for ~10 minutes while tanks, pumps and connecting pipes were cleaned.

Initially, fish were placed into the experimental tanks for 48 hours with ambient seawater. The cups were then turned on and off over a period of 48 hours to allow the fish to acclimatize to the apparatus and ensure they were feeding satisfactorily. Sediment was added gradually over 48 hours for the higher NTU treatments. Cups were switched off daily (between 1500 and 1700 h) for 20 minutes and fish were left to feed undisturbed on ~120 live mysids per tank. Data on daily ambient seawater temperatures were taken from the Leigh Marine Laboratory’s Climate Data Archives (unpubl. data).

Respiration rates

During the final week of the experiment, the ventilation rate of the fish was assessed and expressed as rate of opening of gill operculum per 15 seconds. The short time frame was necessary to accommodate the reduced visibility of fish in the higher turbidity treatments moving away from the forward wall of the tank. Any mucous discharge from the gills was noted.

Gill Histology

On termination of the trial, the first gill arch on the right side of each fish was removed and immediately fixed in Bouin's fluid for 24 hours, then transferred to 70% ethanol. Samples were then dehydrated in graded ethanol concentrations and embedded in paraffin wax. Sagittal sections (4-7µm thick) were cut and mounted on glass slides. Sections were deparaffinized in xylene, hydratated in ethanol and stained with hematoxylin-eosin.

Changes to the epithelial surfaces of gills were examined for evidence of any pathological changes that compromised respiratory function. This included epithelial hyperplasia of the pillar system (increased proliferation of cells, particularly at the base and tips of the lamellae), fusion of the secondary lamellae and the presence of parasites. Changes in the gill lamellae were quantified along three randomly chosen entire gill filaments. Where fusion of lamellae had occurred, the number of individual lamellae which were joined together was counted as per Tricklebank (1997).

3.2.3 Field Study

To appraise the tank experiments in a field situation, I attempted to collect ten juvenile snapper (50-100mm FL) from each of seven northern North Island estuaries, covering a spectrum of perceived environmental degradation (sedimentation and associated water turbidity; Fig. 3.2). Potting with opera traps was undertaken between 1400–1800 h on an incoming tide (Fig. 3.3B). Higher catch rates generally occurred after 1500h. Pots were set over similar substrates (sandy/mud) in each estuary. Upon capture, fish were humanely killed and the first right gill arch was removed using the same protocol as for the laboratory based experiments. Fish were then placed into an ice slurry for transportation back to the nearest laboratory.

Physical parameters

To quantify total suspended sediment (TSS) load, water samples (1000ml) were collected at each of the seven estuaries. Samples were processed using the same protocols as outlined in Chapter Two (see section 2.2). Water clarity was measured concurrently using a 25 cm black-and-white secchi disc. Water parameters including turbidity (NTU), temperature, salinity, pH and dissolved oxygen were also collected using a Horiba U10 multi probe at each site.

Biological parameters

To assess any regional differences in the growth and nutritional condition of the juvenile snapper, a Relative Condition Index (CI) was calculated. In the laboratory, snapper were measured to the nearest mm (FL) and total wet weight of each fish was determined. The liver and digestive system (i.e. stomach and intestine) were removed and weighed separately. Carcass weight was determined by subtracting liver and digestive system weight form the total wet weight. A relative condition index (CI) was calculated as per Francis (1997):

Relative Condition Index = Carcass weight/Expected carcass weight

The Expected carcass weight ($=a\ L^{b}$) is the weight of an 'average' snapper of the same length. Parameters a and b were estimated from linear regressions of $\log^{1o}$ carcass weight versus $\log^{1o}$ L applied to pooled snapper samples. Samples were pooled after initial testing to ensure they had homogenous regression slopes as per Francis (1997).

Fore-guts were then preserved in 10% Formalin and the contents later identified to species level where possible under a dissecting microscope. To estimate biomass, samples were processed utilizing the same protocols as for Chapter Two (see section 2.2) Animals were allocated to sieve size-classes by eye using a graticule in the microscope and a reference collection consisting of a mixture of species retained by different sized sieves as per Edgar et al., (1994). Prey were classified as either 'Benthic' or 'Pelagic' and biomass was summed across these two categories. Pelagic prey included all zooplankton and gastropod veligers, whilst benthic prey comprised amphipods, mysids, shrimps, isopods, decapods, bivalves and polychaetes. For the purposes of this study, mysids were considered benthic in the more turbid harbours due to their propensity to school within clear/shallow waters as an antipredator behaviour versus more relaxed independent movement in more turbid environments close to the seafloor (Mauchline, 1980).

3.2.4 Statistical analysis

Experimental and field data was analysed using regressions plotted with Sigma Plot and one way analysis of variance (ANOVA) using the software R. R analysis was run on tank means. If a significant ($p \leq 0.05$) F value resulted from the ANOVAs, a Tukey's multiple comparison test was used to determine which means differed significantly. Fish that were lost through mortality were excluded from the statistical analysis. Multi-dimensional scaling (MDS) was used to assess differences in environmental variables across the estuaries sampled in the field study using PRIMER 6.0 (Clarke & Warwick, 2001).

3.3 Results

3.3.1 Laboratory Experiments

Experiment 1: Effects of turbidity on foraging

Juvenile snapper showed a consistent positive response to the presence of mysids within the tanks with rapid movements towards the mysids prior to the divider being removed. Regression analysis revealed an overall significant decrease in foraging success with increasing turbidity levels with only ~8% of mysids being consumed in the tanks with turbidities equating to storm conditions (160 NTU), while over 77% were consumed for the control (≤ 10NTU), (Fig. 3.4A, Table 3.1A). There was no significant difference between mysid consumption for the ≤10 and 20 NTU treatments and similarly for the 40 and 80 NTU treatments.

Experiment 2: Effects of substrates on foraging

ANOVA results show there was a significant effect of substrate type on foraging success (Fig. 3.4B; Table 3.1A). Significantly higher numbers of mysids were consumed for the bare-bottom control (97%) followed by the sandy/mud (95%) sediments. Lowest foraging success was recorded for sandy sediments at 77%. Concurrent turbidity measurements taken prior, during, and at the conclusion of feeding revealed sandy/mud sediments ranged between 20-34 NTU, whilst muddy sediments varied between 25-52 NTU (Fig. 3.4C). Sandy sediments, recorded the lowest turbidity variation of 12-17 NTU, which was comparable to the control (i.e. 9-10 NTU).

Experiment 3: Effects of turbidity on growth

Sublethal effects

Regression analysis showed a significant negative relationship between fish weight loss over 30 days and increasing turbidity (Fig. 3.5A). An asymptote was reached at the 40 NTU treatment above which weight loss levelled off. This suggests the maximum weight loss for the 30 day trial was reached at the 40 NTU treatment. Weight loss for the ≤ 10, 20 NTU treatments averaged ~7%, doubling to ~14% for the higher treatments (40, 80, 160 NTU). There were no statistically significant changes between individual treatments due to high individual variance of the fish (Table 3.1B).

Behavioural

Quantitative behavioural observations of fish at the higher turbidity levels (80, 160 NTU) was difficult due to poor visibility. However, behaviour of the fish that could be observed revealed increased gill flaring, coughing and gulping at the surface along with decreased activity levels, particularly for the 80, 160 NTU treatments. Fish in these treatments tended to settle on the bottom with little interaction or aggressive encounters evident. Significant increases in respiration rates were recorded for the higher turbidity levels (Fig. 3.5D; Table 3.1B). These ranged from 20 per 15 sec. (SE ± 1.48) for ≤ 10 NTU, to 27.5 (SE ± 0.34) for 160 NTU treatment.

Gills

Normal gill morphology was categorized as equally spaced secondary lamellae, intact cellular layers with no signs of fusion between the lamellae (Tricklebank, 1997; Figure 3.7). Statistically significant histopathological changes in gill epithelia were observed with increasing turbidity levels (Fig. 3.5; Table 3.1B). The rate of epithelial hyperplasia (causing dilation of the lamellae), increased significantly across treatments from 20 (± 1.30) per filament in the control (≤10 NTU) to a peak of 59.8 (± 3.7) for 80 (NTU; Fig 3.4B). Similarly, lamellar fusion increased significantly across turbidity treatments particularly for the 80 and 160 NTU treatments (Fig. 3.5C). However, there was no evidence for mechanical abrasion or lodging of sediments into gill epithelia of the lamellae.

Bacteria

The presence of epitheliocystis, a bacterial condition affecting the gills of fish, was significantly greater in the higher turbidity treatments (Fig. 3.6; 3.7; Table 3.1B). Eighty eight percent of the fish in the 160 NTU tanks and 67% of fish in the 80 NTU tanks had lesions present. This declined to 0.25% in the 20 NTU treatment. No infection was present in the control (≤10 NTU, Table 3.3). Similarly, rates of infection per gill filament ranged from a peak of 3.0 (± 1.18) per fish for 160 NTU to 0.25 (± 0.16) for the 20 NTU treatments.

Lethal effects

Overall mortalities were low. Two fish died in the lower turbidity treatments (≤10, 20 NTU) associated with aggressive encounters (e.g. tail nipping and chasing) within the first week of the trial. Three fish in the higher turbidity treatments (80, 160 NTU) died in the final days of the experiment. These fish were extremely thin and lethargic, having lost ~25% body weight, and were very pale in coloration with evidence of fin rot from fungal infections.

3.3.2 Field study

Physical parameters

Multivariate analyses of environmental data were done on the basis of Euclidean distances on normalized data. Multidimensional scaling (MDS) separated the estuaries into three groups, reflecting an overall latitudinal gradient from the more pristine northern harbours (Rangaunu, Whangateau, Mahurangi) to the southern harbours (Tamaki and Waitemata), and between these the two west coast harbours (Kaipara and Manukau) (Fig. 3.8B). Environmental parameters including turbidity (NTU), secchi (clarity) and suspended sediments (TSS) provided strong evidence of reduced water quality in some harbours (Fig. 3.8A). Highest water clarities were recorded at the northern harbour sites (e.g. Rangaunu, Whangateau, Mahurangi), whilst the Waitemata Harbour had the lowest water clarity and highest TSS.

Biological parameters

Average snapper length was not significantly related to the order of estuaries along the TSS environmental gradient, so avoiding any potential confounding between the two (Fig 3.9B; Table 3.2) and the response variables. Slightly higher lengths recorded from Kaipara & Manukau may be the result of faster growth rates on the west coast (N. Davies; M. Morrison, pers. comm.). Capture rates varied between harbours, with the lowest number of fish (3) collected in the highly sedimented Waitemata Harbour.

A significant negative relationship was found between the condition indices and increasing suspended sediment load. Rangaunu Harbour, the most pristine location, had the highest average condition indices, whilst the Waitemata Harbour, the least pristine, recorded the lowest average condition indices (Fig. 3.9A; Table 3.2). Results from the remaining five intermediate harbours showed some differences, but these were more variable along the suspended sediment gradient.

Gill Condition

Overall, the rate of epithelial hyperplasia increased significantly with increasing suspended sediment loads (Fig. 3.10A; Table 3.2). This ranged from a low of 11.86 (± 2.80) per filament in Rangaunu Harbour to a peak of 44.9 (± 2.69) in the Mahurangi Harbour. The higher rate recorded for Mahurangi is an exception given it is a more northern harbour. Similarly, the rate of lamellar fusion per filament was significantly higher with increasing suspended sediments (Fig. 3.10C; Table 3.2; 3.4). Rangaunu Harbour recorded the lowest rate of fusion at 1.4 (± 0.86) per filament, whilst Waitemata Harbour averaged 23.7 (± 4.18).

Other histopathological lesions included hypertrophy (swelling) and shortening of the lamellae (Fig. 3.7; Table 3.5). The occurrence of lamellae affected was proportionately greater in the southern harbours, particularly for Mahurangi (85%) and Manukau (80%). Conversely, the condition was largely absent from the northernmost harbours Rangaunu, (0%) and Whangateau, (2%). Between these, Kaipara recorded 20% and Waitemata and Tamaki recorded 30% respectively.

Overall, the prevalence of the bacterial condition epitheliocystis, increased significantly with higher suspended sediments (Fig. 3.10C; Table 3.2; 3.4). However, there were no significant individual pairwise comparisons, due to the variation in sample sizes collected and large variability in occurrence between individual fish at each site. Epitheliocystis was only recorded at four of the southern harbours. This included the Manukau, which recorded the highest proportion of fish affected (67%), followed by Mahurangi (37.5%), Waitemata (33%) and Tamaki (14%).

Prey

Analysis of the stomach content data found crustaceans; comprising mysid shrimps and copepods were the major dietary component in all seven estuaries. The summed biomass across the two prey categories, i.e. ‘Benthic’ and ‘Pelagic’ revealed their relative contribution varied significantly, relative to suspended sediment levels (and associated secchi measures) (Figs. 3.11; 3.12; Table 3.2). In higher water clarity estuaries (e.g. Rangaunu, Mahurangi, Whangateau, Tamaki), pelagic prey dominated the diet (98%), especially calanoid copepods such as *Paracalanus indicus* and the cladoceran *Penilia avirostris*, while in the more turbid estuaries (Manukau, Kaipara, Waitemata) diet was dominated exclusively by benthic prey comprising mysid and Caridean shrimps including *Tenagomysis* sp. and *Palaemon affinis* juveniles. Modest numbers of benthic copepods (*Hemicyclops* sp.*)* and amphipod species were also consumed. In addition, the average size of prey items increased with elevated turbidity levels, with 91% of the pelagic prey dominated by smaller size classes (0.5 - 0.71mm). Conversely, ~70% of benthic prey items consumed were greater than or equal to 0.71mm.

3.4 Discussion

3.4.1 Short term exposure

High suspended sediment loads can alter fish foraging patterns and success by reducing visual acuity due to either shading by particulate matter or scattering of light by suspended particles (Benfield & Minello, 1996). Abrupt increases in turbidity, caused by sudden downpours and short retention times of flood waters in catchments are becoming increasingly common due to climate change (Hicks et al., 2000; Wheatcroft, 2000 Willis et al., 2007; Thrush et al., 2003). Yet surprisingly little is known about behavioural effects of short term intermittent exposure for estuarine fish (e.g. Berg & Northcote, 1985).

The effects of short term pulses of sediments (~1hour) have been shown to disrupt feeding behaviour of salmonids at turbidity levels as low as 20 NTU (Berg, 1982), by reducing the reactive distance for visual feeding fish species, (Vinyard & O'Brien, 1976; Confer et al., 1978; Gardner, 1981; Berg & Northcote, 1985; Barret et al., 1992), or by reducing fish feeding rates (Sigler et al., 1984; Redding et al., 1987; Gregory, 1993; Gregory & Northcote, 1993; Benfield & Minello, 1996; Macia et al., 2003). For example feeding rates of juvenile coho salmon (*Oncorhynchus kisutch*) dropped by 45% at a turbidity of 100 NTU (Reid, 1998). While Vinyard & O'Brien (1976) found that the distance at which Bluegill (*Lepomis macrochiru),* reacted to their prey decreased from 27cm at 6.25 NTU, to 5cm at 30 NTU. Results from this study support these findings. There was a significant clear decline in foraging success of juvenile snapper evident with increasing suspended sediment levels, ranging from 77% in the controls (≤ 10 NTU) through to ~8% success rate for the treatments equating to storm conditions (160 NTU) (Fig. 3.4; Table 3.1A).

It is of note that foraging rates declined significantly once 20 NTU was exceeded (i.e. 40-160 NTU treatments). This correlates with the natural background of suspended matter (~20 NTU) thought to be present in most estuarine systems (Dyer, 1972; cited in Bruton, 1985). Other studies have shown that moderate levels of TSS may enhance the visual contrast of prey items, increasing overall feeding rates as reported for the larval Pacific herring (*Clupea pallasi*) (Boehlert & Morgan, 1985). Similarly, Utne (1997) found that optimum reaction distances of the goby (*Gobiussculus flavescens*) to planktonic prey were measured at turbidity levels of 10-20 (Jackson Turbidity Units; approximately equivalent to NTU). Other studies have correspondingly found prey selectivity affected and significantly lower foraging rates beyond a threshold of 40 NTU for both freshwater (Rowe & Dean, 1998; De Robertis et al., 2003;

Robertson et al., 2007; Shoup & Wahl, 2009;) and marine species (Macia et al., 2003; Nurmien & Horppila, 2006). However, as with any laboratory study, enclosure effects were unavoidable. Results therefore reflect the relative changes in feeding rate in response to changes in suspended sediment levels rather than actual foraging rates in the field (Bash et al., 2001; Meager et al., 2005).

In summary, results suggest that short term pulses of suspended sediment can reduce foraging success and if prolonged, may ultimately impair fitness of juvenile snapper populations and subsequent recruitment and survival. Within New Zealand, predicted increases in sediment runoff from land and subsequent rapid sedimentation events into estuarine and coastal regions are forecast with climatic variability (Hicks, 2000; Willis et al., 2007). This, coupled with the high proportion of catchment-derived sediment remaining within estuaries following floods (Oldman, 2009 and references therein), increases the likelihood of ongoing re suspension and longer term exposure to higher turbidities However little or no data is available on longer term chronic exposure on estuarine fish within New Zealand.

3.4.2 Substrate effects

Substrate type significantly influenced the rate of predation on mysids by juvenile snapper. However, substratum effects were less pronounced than the effects of turbidity (experiment 1). There were no significant differences between the control (clear) and sandy/mud sediments, with over 96% of mysids consumed, while 83% and 77% of mysids were consumed in the mud and sand sediments respectively (Fig. 3.4B; Table 3.1A). Concurrent turbidity measurements taken during and at the conclusion of feeding, revealed water over sandy/mud sediments ranged between 20-34 NTU, whilst muddy sediments varied between 25-52 NTU. Interestingly, sandy sediments scored the lowest predation rate, although recording a low turbidity of ~14 NTU, which was comparable to the control (i.e. ~10 NTU).

Turbidity effects can vary, dependent upon the behaviour of prey species (Macia et al., 2003). Characteristics such as size, pigmentation and motion also influence the ability of visually foraging fish to detect their prey (Confer et al., 1978; Pekan-Hekim, 2007). Littoral mysids are generally more active at lower light levels (Mauchline, 1980; Hecht & van der Lingen, 1992) and will bury themselves or hide in vegetation if there is clear/shallow water during the day or school (Jumars, 2006). The lower turbidities recorded over the sandy sediments along with the lowest predation rates (i.e. 20% less than sandy/mud), suggest that mysids may have sought refuge and camouflage on the sand and gained some protection from foraging snapper with lowered activity levels coupled with hiding/burial. Similarly, behaviour of mysids observed in the control tanks with the lowest turbidities, revealed mysids predominantly immobile on the tank floor. However, with no substrate for camouflage, juvenile snapper foraging was highly successful (i.e. 97% of mysids consumed).

Conversely, the higher turbidity levels recorded for the muddy sediments (25-52 NTU) afforded protection for the mysids due to the reduced visual acuity of the juvenile snapper in turbidities exceeding 40 NTU. This agrees with short-term experimental results. In conjunction with higher activity levels of mysids at lower light levels, turbidity may have affected the ability of mysids to see and evade the juvenile snapper more than the ability of snapper to see mysids, as the fish present a larger visual target (Giske, 1994; cited in Meager et al., 2005). Mysid vision is fundamentally different from fish therefore turbidity may not affect them in the same way (Meager et al., 2005). Although juvenile snapper may not have been able to see the mysids perfectly at the intermediate turbidity levels recorded for sandy/mud sediments (i.e. 20 - 40 NTU), the encounter rates and subsequent predation rates may have been higher due to the increased activity levels of the mysids (Macia et al., 2003).

Thus, sediment type (and associated turbidity levels) can and does influence predation rates on mysids by juvenile snapper. Mysids should prefer areas which offer maximum protection (i.e. higher turbidities). Greater numbers of mysids are indeed found in higher turbidity zones (and associated muddy sediments) of estuaries (e.g. see reviews of Mauchline, 1980; Roast et al., 1998; Jumars, 2006). Whilst, the high foraging success of juvenile snapper in sandy/mud sediments are in accordance with juvenile snapper distribution found in the wild (Francis, 1995; M. Morrison pers. comm.)

3.4.3 Long-term exposure

Past, experimental work on suspended sediments has largely focused on concentration levels, while exposure durations remained fixed (see review by Wilber & Clark, 2001). Newcombe and MacDonald (1991) suggested that the use of concentration of suspended sediments alone is a poor indicator of physiological and behavioural effects. The authors developed a ‘stress index’ for salmonids using both concentration and duration of exposure which indicated that ‘sublethal’ effects from longer term exposure (4-42 days) can have deleterious effects. Subsequent predictive models utilizing sediment concentration and duration of exposure (Newcombe & Jensen, 1996; (BCMELP, 1998; CCME, 1999 cited in Birtwell, 1999) reveal significant trends of increasing harm to fish with increasing exposure duration. There is still, however a paucity of relevant empirical data for estuarine fish for longer term chronic and the cumulative effects of 'short term pulses' of suspended sediments (Amara et al., 2007).

Results from the 30 day exposure trials in this study (Fig. 3.5; Table 3.1A) are generally consistent with prior research showing reduced feeding/growth rates with chronic exposure to increasing suspended sediments. This has been documented for both freshwater (e.g. Buck, 1956; Herbert & Merkens 1961; Sigler et al., 1984; McLeay et al., 1987; Shaw & Richardson, 2001), estuarine species (e.g. Sherk et al., 1974, 1975; see reviews of Kerr, 1995; Newcombe & Jensen, 1996; Wilber & Clarke, 2001; Au et al., 2004; Amara et al., 2007), and planktivorous coral reef fish (Partridge & Michael, 2010), (Refer Appendix 3.2).

In this study, juvenile snapper showed significant weight loss with increasing turbidity levels. Weight loss reached a maximum (~14%) at 40 NTU, subsequently levelling off for the 80 and 160 NTU treatments. However, no statistically significant changes between individual treatments were evident due to high individual variance of the fish. Similar patterns of threshold responses, with a rapid escalation of ill effects as duration of sediment exposure increased from 1-6 days at concentrations above 55 mg/L (particularly for larvae and some adult estuarine fishes) has been documented (see review Newcombe & Jensen, 1996). Similar results have been recorded for freshwater fish (e.g. Vinyard & O'Brien, 1976; Confer et al., 1978; Gregory et al., 1993).

The nominal weight losses recorded for the control group ($\leq$ 10 NTU) could have been a result of stress from confinement and handling and/or ambient sea surface temperatures being the lowest on record for 60 years during this trial (Autumn/Winter) (Leigh Marine Laboratory's Climate Data Archives, unpubl. data). Growth rates for 0+ juvenile snapper are known to be correlated with temperature, and slow dramatically over the first winter (Francis, 1994; pers. comm. S. Pether, Senior Research Technician, Bream Bay; NIWA).

3.4.4 Sublethal effects

Gills

Gill pathological symptoms observed in the juvenile snapper included epithelial hyperplasia (thickening) of the pillar system and fusion of the secondary lamellae (Fig. 3.7). These changes represent sublethal rather than acute effects and can be considered a defence mechanism by increasing the distance across which irritants must diffuse to reach the bloodstream (Mallat, 1985). However, cell proliferation of epithelial tissue, and eventual loss of surface by clubbing and fusing of lamellae can impair respiration and ammonia excretion and lead to respiratory stress and ammonia intoxication (Au et al., 2004; Goldes et al., 1988; Bergstedt & Bergersen, 1997; Kerr, 1995). Bacterial infection (i.e. epitheliocystis) of the gills was also recorded. Epitheliocystis infect the epithelial and chloride cells of gills and cause hypertrophy (swelling of individual cells) which can result in respiratory distress (Meijer et al., 2006). Histologically, the disease is characterized by the presence of lesions (cysts) within a hypertrophic epithelial cell filled with bacteria described as ‘chlamydia like’ (L. Tubbs, Pathobiologist, Guelph University; pers. comm.; Meijer et al., 2006).

In this study, juvenile snapper showed a progressive increase in the incidence of damage to gill lamellae with increasing concentration of suspended sediments. Hyperplasia, became significantly different from the control at 40 NTU. The occurrence of epitheliocystis and lamellar fusion increased significantly in a more curvilinear fashion with higher suspended sediment loads (Fig. 3.5C; 3.6; Table 3.1B, 3.3), with high variability recorded between individual fish within treatments. Other authors have noted that, under any given set of exposure conditions, each kind of gill lesion can vary widely in intensity within a single fish (Herbert & Merkens, 1961; Mallat, 1985 and references therein). Findings are in agreement with other estuarine studies showing damage to gill structure, including hyperplasia in the pillar system, fusion and reduction of epithelial volume strongly correlated to suspended sediment concentration for both estuarine (Sherk et al., 1974; O’Conner et al., 1977; Au et al., 2004) and freshwater species (see reviews Kerr, 1995; Newcombe & Jensen, 1996; Wilber & Clarke, 2001; Berry, 2003; Morrison et al., 2009). However, contrary to previous research, there was no evidence for intensified mucus production for any of the treatments in this study. This may be a factor of turbidity levels being comparatively lower in this study from prior research. In addition, characteristics of sediments used in the experiments, particularly their abrasiveness (angularity) may have varied from other studies causing differential effects. Sharp angular sediment has been shown to cause excessive mucous discharge (Sutherland & Meyer, 2007).

Evidence suggests increased ventilation rates under highly turbid conditions are a means to compensate for reduced respiratory efficiency, thereby maintaining a constant oxygen uptake (Horkel & Pearson, 1976). This has been recorded for numerous freshwater (Horkel & Pearson, 1976; Berg & Northcote, 1985; Newcombe & MacDonald, 1991; Servizi & Martens, 1992) and estuarine species (Sherk et al., 1974; O’Conner et al., 1977; Neuman et al., 1982; Au et al., 2004). Results from this study confirm these trends with a significant rise in respiration rates associated with higher turbidities (80, 160 NTU) (Fig. 3.5D, Table 3.1A). Other qualitative behavioural observations, including increased gill flaring, coughing and gulping at the surface, in addition to reduced activity levels in the higher turbidity treatments, are all indicative of anoxia which is consistent with published information (Berg, 1982; Berg & Northcote, 1985; McLeay et al., 1987; Servizi & Martens, 1992). Paler coloration, fin rot and early mortality observed in 17% of fish from the 80 and 160 NTU treatments (and higher incidence of epitheliocystis) concurs with prior research (Herbert & Merkings, 1961; Ritchie, 1972; McLeay et al., 1984; Appleby & Scarrett, 1989; Redding et al., 1987; see reviews Moore, 1977; Newcombe & Jensen, 1996). Collectively, these studies suggest that exposure to elevated sediment levels decreased tolerance rates to disease and time to death.

Although not measured in this study (due to technical difficulties associated with the small size of the fish), hematological compensation for lost respiratory efficiency associated with longer term chronic exposure has been documented. Changes in blood physiology such as elevated levels of blood sugars (Servizi & Martens, 1992), plasma glucose (Servizi & Martens, 1987), microhematrocrit (packed red cell volume), haemoglobin concentrations, red cell counts (Sherk et al., 1974, 1975; O’Connor et al., 1977; Redding et al., 1987; Appleby & Scarret, 1989) and cortisol levels (Schreck, 1981) have been recorded with increasing suspended sediment concentrations (see Appendix 3.2).

However, with laboratory based studies, experimental artefacts were unavoidable, with factors such as prey availability, intra and interspecific encounters being controlled (Bash et al., 2001). In addition, spatial and temporal factors such as distribution, abundance, or availability of suitable habitat, time of year, frequency, duration and magnitude of storm events (with associated increase in current velocities) may be more deleterious in the field (Bruton, 1985). In these experiments fish were exposed to sediment unlikely to be contaminated. However, in coastal environments, suspended sediments frequently serve as a sink for contaminants (Hack et al., 2007). These chemicals can cause sublethal stress and in combination with suspended sediments, may have important interactive effects on marine fishes (Au et al., 2004). Therefore results may be conservative in their estimation of suspended sediment impacts under field conditions.

In summary, these experiments demonstrate that although turbidity levels in this range (10-160 NTU) are unlikely to cause high mortality in juvenile snapper, prolonged exposure to turbidities greater than or equal to the 40 NTU can result in adverse growth and developmental effects from (a) reduced prey capture success due to impaired vision and/or (b) increased metabolic costs from physiological stress (e.g. respiratory distress/disease). Hence, if exposure continues in the long-term, reduced energy acquisition in turn could be reflected in lower condition indices of fish from turbid estuaries relative to those with higher water clarity and/or by reduced fish abundance in highly turbid estuaries (Hecht & van der Lingen, 1992; Au et al., 2004).

3.4.5 Field study

Juvenile marine migrants generally spend a short time period in estuaries (Elliot et al., 2007), usually restricted to the first months or year of life where growth rates are rapid, leading to reduced life stage duration and enhanced survival from predation (Houde, 1987; Sogard, 1992; Manderson 2002; Vasconcelos et al., 2009). Small variations in growth and mortality during this critical period are known to effect recruitment levels to offshore populations (Houde, 1987; Vasconcelos et al., 2009). Condition indices are effective proxies of growth rates and nutritional status, providing information on a fish’s response to quality of the habitat. This can be indicative of differences in the nursery role of estuaries or habitats within them (Vasconcelos et al., 2009).

In this survey there were significant differences among the biological variables measured on juvenile snapper collected from estuaries with varying degrees of anthropogenic disturbance. Juvenile snapper had significantly lower condition indices in the more impacted estuaries characterized by increasing sedimentation, concomitant with lower water clarities and increasing urbanization (e.g. Waitemata, Manukau) (Fig. 3.9A; Table 3.2), whilst Rangaunu Harbour (Northland), the most ‘pristine’, had the highest average condition indices. These results agree with other surveys which have demonstrated inverse relationships between condition indices of fish and water clarity in the field (in conjunction with increasing urbanization) (Craig & Babuluk, 1998 cited in Grecay & Targett 1996; Amara et al., 2007; Courrat et al.,2009; Zingle & Paaver, 2010). Larval fish have also been shown to be negatively impacted with increasing turbidity up to 16 NTU (e.g. Utne-Palm, 2002; Salonen et al., 2009; Partridge & Michael, 2010).

However, it must be noted that field results were ‘point’ samples collected from one area within each estuary. For very large systems such as the Kaipara Harbour, TSS levels can vary across different regions (author unpubl. data). Samples were collected during settled weather and therefore do not account for spatial and temporal factors such as frequency, duration and magnitude of prior storm events, the distribution, abundance, or availability of suitable habitat, time of year or the cumulative or synergistic effects of the multiple chemical and physical stressors that fish are exposed to, particularly those estuaries close to urban centers.

Declines in condition indices suggest lower energy reserves as a result of altered feeding rates and/or increased metabolic rates resulting from stress (Bergstedt & Bergersen, 1977; Barton & Schreck, 1987). This may ultimately affect growth and influence survival, year class strength, and recruitment. It has been suggested that sediment related increase in stress response and reduction in growth rates may be in part due to increased gill damage, which may operate via increased respiratory impairment (Schreck, 1981).

Gills

Overall, the field study showed a strong inverse relationship between the lamellar changes observed and condition indices of juvenile snapper with higher rates of gill damage recorded from the southern estuaries with higher suspended sediment loads. Lesions observed in the gills included epithelial hyperplasia, fusion, shortening and hypertrophy (swelling of epithelial cells) of secondary lamellae and the presence of epitheliocystis (Fig.3.10; Tables 3.1B; 3.4; 3.5). Moreover, the occurrence of histopathological alterations such as hypertrophy and shortening of the lamellae show close similarity to lesions brought about by elevated levels of other environmental pollutants such as e.g. zinc**, (**Bhagwant & Elahee, 2002), nickel (Al-Attar, 2007); nitrogen (Schlacher 2007) and phosphate (Omoregie et al., 2009).

Therefore, the markedly elevated histopathological alterations (hypertrophy /shortening) recorded for both the Manukau (80%) and Mahurangi (85%) harbours may well be a result of other environmental pollutants acting synergistically with TSS. The Manukau Harbour, located adjacent to New Zealand’s largest city, has historically had high levels of copper (Cu), lead (Pb), zinc (Zn) and polycyclic aromatic hydrocarbons (PAHs) in the estuarine sediments (Hack et al., 2007). A review of water quality in the Auckland region (Scarsbrook, 2008) also noted elevated levels of nitrates and phosphates for both Weymouth (Manukau) and the Mahurangi Harbour. Indeed, Mahurangi recorded the highest concentrations of enterococci and faecal coliforms in the region. Sources include partially treated and untreated sewage and runoff from agriculture. It is of note that the Mahurangi Harbour is also a highly popular boating destination over holiday periods.

However, determining cause and effect relationships between specific chemicals and their effects on juvenile fish is particularly difficult given the myriad of contaminants and their potential synergisms (Tricklebank, 1997 and references therein). Recent studies indicate that these contaminants also have detrimental effects on the benthic fauna with reductions in abundance and changes in growth and reproductive rates (Ministry for the Environment, 1997). This may also indirectly affect fish health via decreased prey availability. In addition, higher levels of both TSS and pollutants have been shown to predispose fish to opportunistic infections (Redding et al., 1987; Goldes et al., 1988). Nowak and LaPatra (2006) found more frequent and severe epitheliocystis in fish exposed to sewage. This agrees with findings in this study, with highest rates of infection being recorded for the Manukau and Mahurangi Harbours respectively. In summary, results suggest that physiological stress in fishes in response to increased TSS can decrease immunological competence and growth.

Whether the gills of juvenile snapper can recover from these histopathological changes (noted for both the laboratory and field survey) is unknown. However, studies on freshwater species (e.g. Fukuda, 1983; Goldes et al.,1988; Karan et al.,1998) have shown virtual complete recovery from severe reactive hyperplasia in less than a month when the stimulus was removed and adequate water quality was available. However, for estuarine fish, exposure to elevated levels of TSS can be elevated for extended periods and the effects could well be cumulative with ongoing sediment pulses.

Prey selection

Dietary analysis of stomach contents found crustaceans (mysid/Caridean shrimps & copepods) to comprise the major dietary component in all seven estuaries. However, the relative contribution of the summed biomass of the two prey categories, i.e. ‘Benthic’ and ‘Pelagic’ varied significantly relative to TSS levels (and associated secchi measures) (Fig. 3.12, Table 3.2). Pelagic prey, particularly calanoid copepods and cladocerans dominated (98%) the diets of those fish collected in the higher water clarity estuaries (e.g. Rangaunu, Mahurangi, Whangateau, Tamaki), while in the more turbid estuaries (Manukau, Kaipara, Waitemata), benthic prey such as mysids predominated (100%). In more turbid environments mysids tend to relax anitpredator behaviours i.e. schooling, and remain close to the seafloor, moving independently), (Jumars, 2006; Mauchline, 1980). Concurrently, the average size of prey items varied with turbidity/TSS level. Ninety percent of ‘pelagic’ prey consumed in the higher water clarity estuaries was dominated by smaller size classes (i.e. 0.5 - 0.71mm). Conversely, ~70% of prey consumed in the turbid estuaries were greater than or equal to 0.71mm. These findings suggest that increasing turbidity levels cause a change in feeding strategy from active (probably visual) selection of pelagic prey (zooplankton), to larger, slower-moving benthic prey. This type of behavioural shift in foraging tactics has been documented in other studies, often with a ‘turbidity’ threshold of around 40 NTU, above which declining visual acuity results in more opportunistic, ambush type predation (Moore & Moore, 1976; Marais, 1984; Hecht & van der Lingen, 1992; Grecay & Targett, 1996; Løkkeborg, 1998; Macia et al., 2003 and references therein; De Robertis 2003; Shoup & Wahl, 2009).

Other authors have suggested that feeding in turbid environments can result in higher energetic costs associated with increased foraging time on evasive prey along with increased exposure to predation, therefore it is more cost effective to switch strategies (Whitfield, 1994; Meager et al.,2005, 2007; Engström-Öst & Mattila, 2008). Prior research has shown no significant differences in the condition and abundance of fish altering foraging strategies as a result of changing turbidities for both marine (Moore & Moore, 1976; Cyrus & Blaber, 1988; Hecht & van der Lingen, 1992) and freshwater species (Hayes & Rutledge, 1991; Hayes et al.,1992). Conversely, turbidity can reduce the costly anti-predator behaviour of fish and increase the energy gain for growth as observed for juvenile Chinook salmon (Gregory & Northcote, 1993). Alternatively, others suggest it may lead to ‘a false security effect’ (Lehtiniemi et al., 2005), resulting in an easy meal for the predator (Salonen & Engström-Öst, 2010).

However, in contrast to these previous studies, juveniles as opposed to adult fish were surveyed in this study. It appears that lifestage and behaviour can also mediate the influences of TSS with larval stages and eggs being particularly vulnerable (Newcombe & MacDonald, 1991; Kerr, 1995; Wilber & Clarke, 2001; Morrison et al., 2009; Salonen & Engström-Öst, 2010). For example Partridge & Michael (2010) found significant declines in the foraging success of larval snapper, with tolerance to suspended solids negatively correlated with concentration and exposure time (3-18 h). Once larvae began feeding, exposure to suspended solids of 157 mg l^{-1} for 12 hours resulted in a 50% mortality rate. Young fish may be restricted in their foraging abilities from eating larger prey by their small gape size and the assimilation of larger prey, due to their poorly differentiated guts (Confer & Lake, 1987; Boubeé & Ward, 1997; Morrison, 1990; Gillanders, 1997, Sudo & Azeta, 2001). Newly settled snapper (20-30mm) are known to feed predominantly on copepods (see Chapter Four; Usmar, 2009), particularly calanoids, which are present in water layers close to the substratum (Tanaka et al., 1987; Hostens & Mees, 1999). Larger juveniles subsequently progress to larger sized crustaceans such as amphipods/mysids (>0.75mm) (see Chapter Four).

Optimal diet theory predicts that predators should forage selectively on prey that maximize their net energy intake rate (Lankford & Targett, 1997). Recent research has shown that juvenile fish will actively select smaller, easily digestible prey because it assimilates quickly and translates into greater somatic growth (Mills et al., 1984; Lankford & Targett, 1997; Gning et al., 2009). For example, growth of yellow perch (*Perca flavescens*) was found to vary by 50-300% dependent upon zooplankton species and size consumed (Confer & Lake, 1987). Similarly, juvenile weakfish (*Cynoscion regalis*) were found to preferentially select the smaller sized mysids (*Neomysis americana*) over the larger shrimp (*Crangon septemspinosa*), even where *Crangon* biomass was ~20 times greater than *Neomysis*. *Neomysis* were digested and evacuated from the stomach 1.8 times faster than *Crangon.* Therefore, increase in prey size does not automatically confer optimal growth for juvenile fish. Rather, post consumptive processes can potentially constrain long-term energy intake rate and growth (Lankford & Targett, 1997).

Thus, increasing TSS can potentially restrain juvenile snapper condition and growth by reducing the overall food supply available and perhaps relative nutritional values, by reducing their ability to visually pick zooplankton, and/or a reduction in the actual zooplankton assemblages available due to changing environmental conditions. However, the extent to which the difference in diets are due to differences in prey abundance or to turbidity levels in the estuaries is not known as prey densities were not sampled concurrently. Simultaneous spatiotemporal assessments of prey abundance merits further study. Nevertheless, changes in TSS can affect the abundance, nutritional value and composition of zooplankton and other prey species.

3.4.6 Concomitant effects on prey

Elevated levels of TSS have been shown to reduce the diversity and abundance of both pelagic and benthic invertebrate prey for both freshwater and estuarine systems (Quinn et al., 1992; Harding et al., 2000; see reviews: Thrush et al., 2004; Gibbs & Hewitt 2004; Lloyd, 1987; Newcombe & MacDonald, 1991; Bash et al., 2001; Berry et al., 2003; Morrison et al., 2009) by abrading, clogging and smothering organisms; reducing interstitial spaces and reducing food supply and quality through decreased light attenuation and hence aquatic algae and plant productivity (Fig. 3.1).

An 18 year study in France revealed suspended matter controlled the long-term temporal trends and nutritional values for copepods and two mysid species. Lower nutritional values were associated with higher turbidities (David et al., 2005). Similarly, reduced feeding, lower growth rates and higher mortality of planktonic copepods and cladocerans has been recorded with exposure to higher turbidities (Paffenhoffer, 1972; Sherk et al., 1976; Hart, 1986, 1988; Koenings et al., 1990; Berry et al., 2003 references therein). At the Cape Rodney to Okakari Point Marine Reserve, the abundance of zooplankton declined from ~100,000 individuals per m³ to ~50 per m³ for a period of seven days following a storm (pers. obs.). Suspended sediments have also been found to be acutely toxic to young-of-the-year amphipods (an important component of larger (50-90mm FL) juvenile snapper diets, see Chapter Four) (Forbes et al., 1981), while Schwarz (2006) found decreased survival rates for the gammarid amphipod *Aora* sp. at high suspended sediment concentrations.

Accordingly, both the magnitude and timing of sediment pulses may have profound and long-term effects on survival and recruitment of larval/juvenile fish particularly where diet is restricted due to ontogeny (Moore, 1977; Newcombe & Macdonald, 1991; Gregory, 1993; Campana, 1996; Bash et al., 2001; Murphy et al., 2012; Partridge & Michael., 2010). Research in New Zealand suggests that reduced growth for juvenile snapper may translate into smaller sized adults with concomitant declines in fecundity at the population level (Gilbert, 2006; N. Davies; C. Walsh pers. comm.). Variation in fish population size as a result of change in individual growth during early life stages has been demonstrated for other fish species (Houde, 1987; Campana, 1996; Taborsky, 2006). Thus, further research on the effects of TSS on younger life stages such as larvae and juveniles less than 50 (mm FL) and tracking development of fish from a known range of sedimented estuaries utilizing otolith chemistry to estimate either daily/annual growth as fish recruit to offshore fisheries would enhance understanding of these processes.

3.4.7 Turbidity thresholds

A review of the literature indicates there are significant thresholds of susceptibility to suspended sediment in ultra sensitive species and life stages (Lloyd, 1987; Wilber & Clarke, 2001; Sigler et al., 1984). Results from this study revealed more gradual incremental responses to increased TSS Juvenile snapper showed significantly higher rates of sublethal stress. (i.e. decreased foraging, increased weight loss gill deformation & disease) above 20 and 40 NTU for both short (30 minutes) and long term (30 days) laboratory experiments. Similarly, field results revealed significant declines in condition indices, altered foraging strategies and gill deformation with increasing TSS, particularly above 35 mg/L. However, to date there are no guidelines/standards for protecting key species of estuarine fish (i.e. tolerance ranges) from turbid conditions in New Zealand (Richardson et al, 2001).

3.4.8 Turbidity standards

With declining estuarine water quality becoming an increasingly significant coastal management issue within New Zealand (Parkyn et al., 2002; Quinn & Stroud, 2002; Morrison et al, 2009; ARC, 2010; Land and Water Forum, 2012), developing region specific water quality guidelines such as those developed by NW United States and British Colombia Canada (Bash et al., 2001; Borja, et al., 2010; Foley et al., 2010), would enhance future reporting on state and trends in New Zealand’s harbour’s and estuaries. This is particularly pertinent given the paucity of ‘pristine’ reference sites (Scarsbrook, 2008). As an example, Lloyd’s (1987) review on acceptable turbidity standards for Alaskan salmonids indicated that water quality criterion allowing increases of 25 NTU and ~7 NTU (0-25mg/L) above ambient turbidity in clear, running coldwater habitats provided moderate to high levels of protection for this species. However, there are difficulties in setting rigid standards for TSS.

Concentration of TSS in natural waters can be influenced by factors such as topography, geology, soil condition, intensity and duration of rainfall, amount of vegetation in the drainage basin and past and current human activities (Bash et al., 2001; Sorrenson et al., 1997). This can result in variations on a daily and yearly basis. In addition, the relationships between current surrogate measurement tools (i.e. water clarity; turbidity (NTU); TSS) although correlated can vary widely between watersheds. Therefore site specific criteria would need to be determined for levels of turbidity (Birtwell, 1999; Bash et al., 2001).

Future perspectives - Fish as Bio indicators

These results, in conjunction with other studies provide evidence that turbidity deserves recognition as an important environmental stressor that can cause significant degradation of estuarine ecosystems with concomitant secondary flow on effects to fisheries (Moore, 1997; Au et al., 2004; Morrison et al., 2009).

Although a comprehensive water quality monitoring network exists across 27 estuary and nearshore sites in the Auckland region (Scarsbrook et al., 2008), there is a paucity of relevant empirical data as to the extent to which of these physico-chemical conditions effect the health of estuarine fishes. A subset of ‘biological indicator’ fish species (e.g. snapper; flounder), and histological indicators used in this study could be included in future routine long-term environmental monitoring of both pristine (e.g. Rangaunu) and impacted sites (e.g. Manukau). Indeed, there is a growing interest in the use of fish (particularly marine juvenile migrants; Courrat et al., 2009) as ‘biological indicators’ of environmental change (i.e. water quality evaluation) in transitional waters (Whitfield & Elliot, 2002; McLusky & Elliot, 2007) (see Appendix 3.3). This has been recently highlighted by the adoption of the European Marine Strategy Framework Directive (MSFD; 2008/56/EC) which aims to achieve ‘Good Ecological Status’ in all water bodies by 2015 (Mee et al., 2008; Martinho, 2009; Borja et al., 2010).

Population and community dynamics of fishes are strongly driven by fish health, and as such must be adequate to fulfil key functions including reproduction, recruitment, and growth (Schalcher, 2007). Results from this study collectively suggest that increased suspended sediment levels in the northern New Zealand marine environment can have negative effects on individual fish and their fitness. Lower growth and nutritional status of juvenile snapper may potentially lead to increased vulnerability to predation, physiological stress and disease in addition to lower overwinter survival and subsequent recruitment to commercially exploitable stocks (Francis, 1994; Adams, 2002; Amara et al., 2007). Findings from this study supports the level of concern being shown over escalating point source inputs of both suspended sediments and concentrations of nutrients (phosphates/nitrates) from dairying/forestry into rivers and estuaries within northern New Zealand (Parkyn, 2002; ARC, 2010; Land and Water Forum, 2012), and highlights the need for management to encompass both marine and terrestrial catchments for the effective protection and sustainability of juvenile fish nurseries.

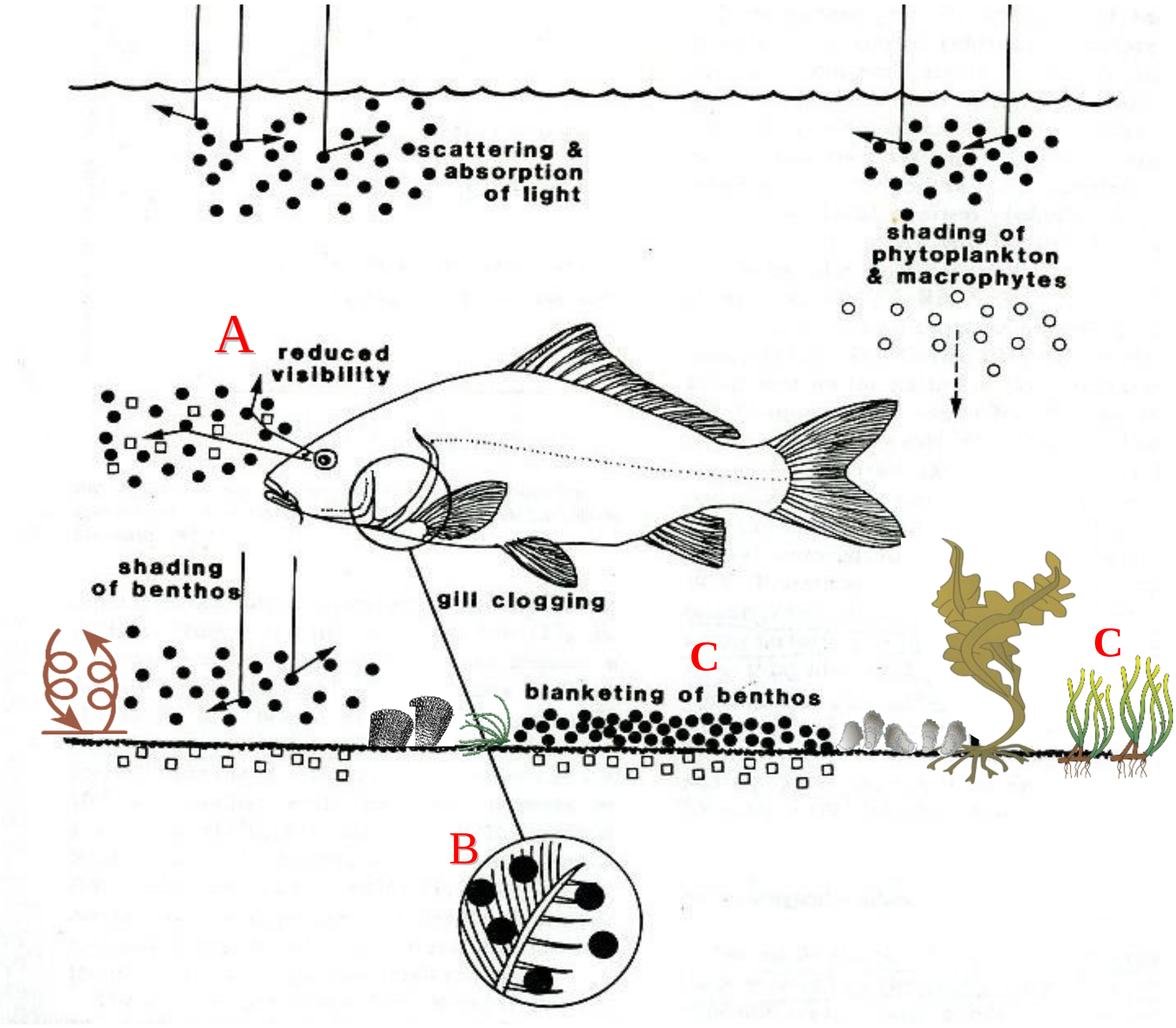


Figure 3.1 Conceptual diagram of the effects of suspended sediments on fish. Direct: A. reduction in foraging efficiencies - reduced visual acuity; B. clogging of gills with fine sediments; Indirect: C. smothering of benthos and reduced growth of biogenic habitats due to declining water clarity.

Arrows show the extent to which suspended sediments cause light to be scattered or absorbed. Adapted from Bruton (1985).

●= suspended sediments; ○=phytoplankton; ☐ = zooplankton and benthic invertebrates

= ongoing deposition and resuspension of sediments.

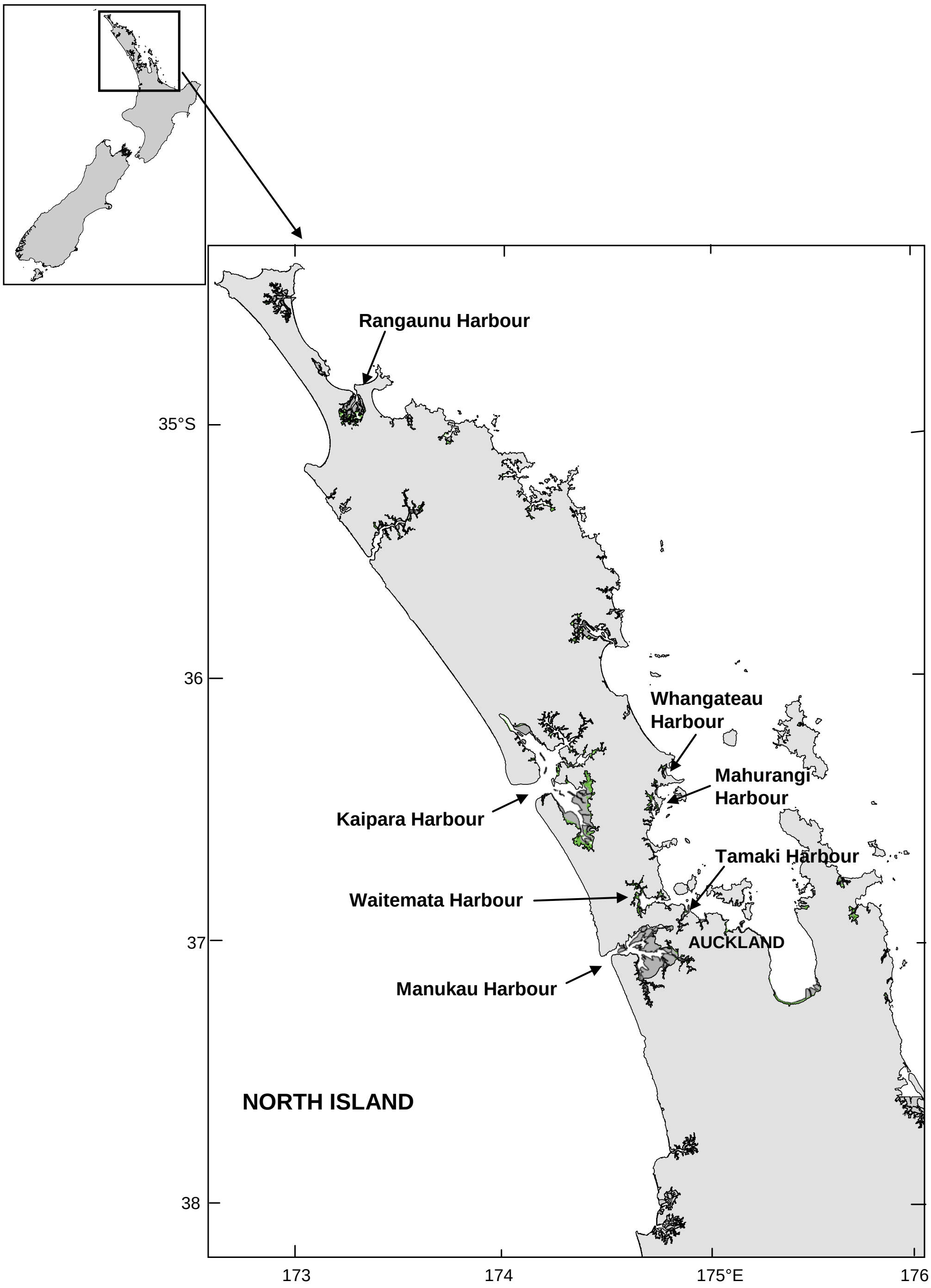


Figure 3.2 Location of the seven sites sampled for juvenile snapper in northern New Zealand.

Figure 3.3 Experimental tanks used for the 30 day exposure trials of juvenile snapper to five levels of turbidity (A); Opera traps utilized for field collection of juvenile snapper from the seven northern New Zealand estuaries (B).

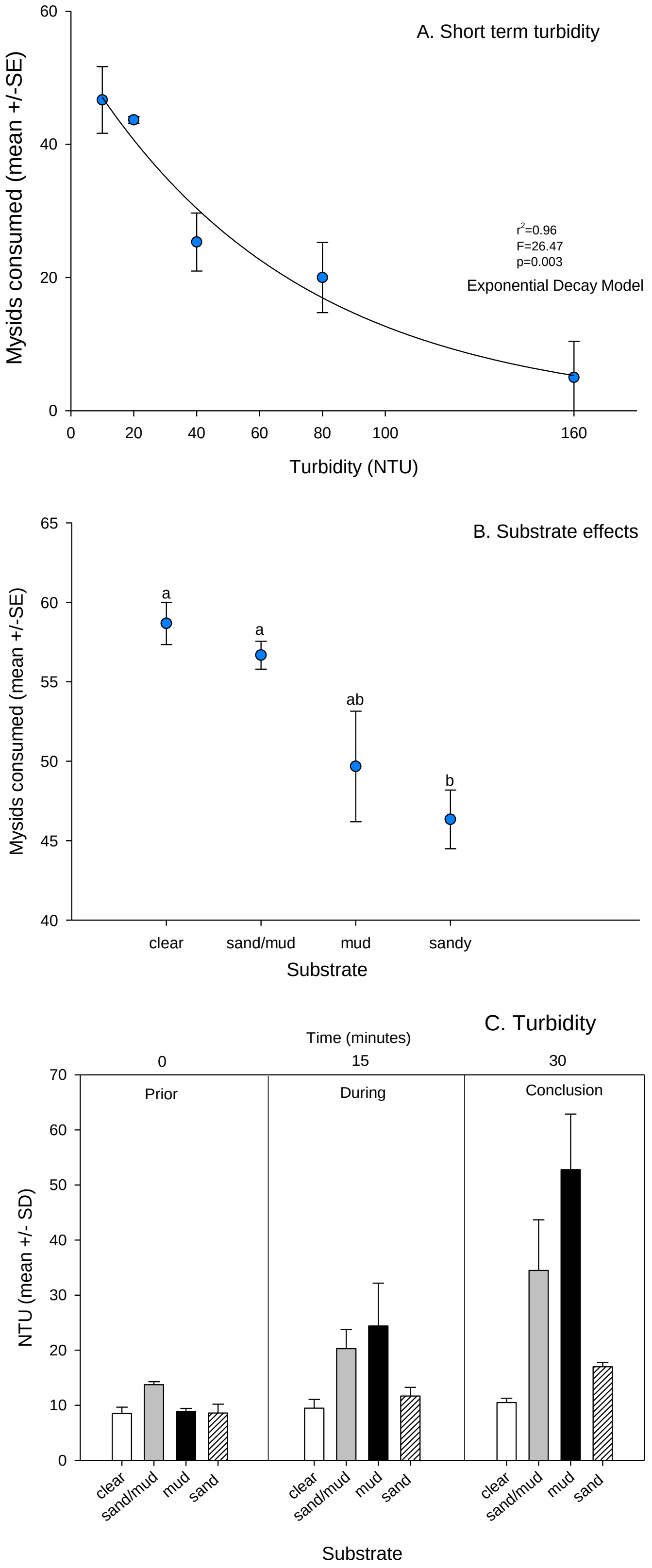


Figure 3.4 Mean number of mysids consumed (±SE) over 30 minutes versus: (a) turbidity (NTU) concentrations for tank experiments; solid lines indicate fitted linear regression (b) substrate type (c) turbidity levels prior to, while feeding and at conclusion of the feeding trial for substrate types. Values which are not significantly different (*P* > 0.05) share common superscripts.

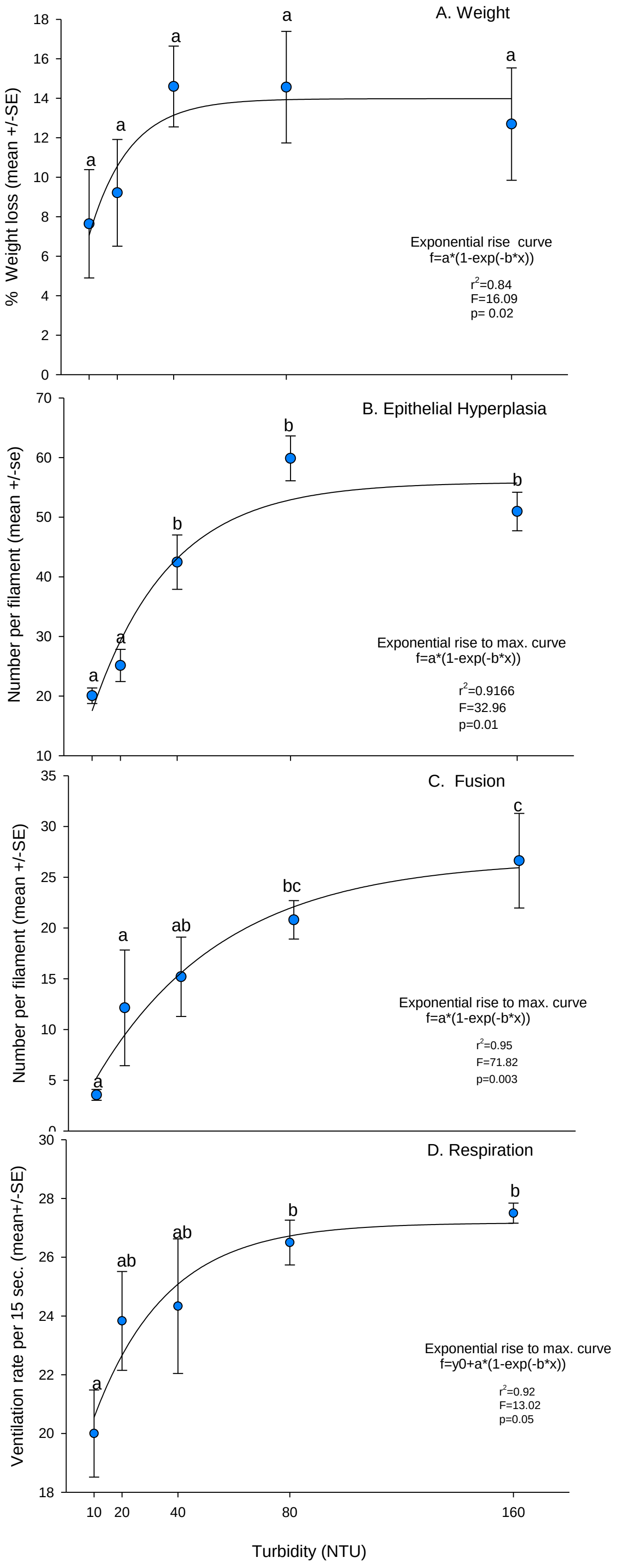


Figure 3.5 Snapper responses in 30 day experiment on exposure to five turbidity treatments: mean percentage weight loss (±SE) (A); number of lamellae per filament in the gills with epithelial hyperplasia (B); lamellar fusion (C); and respiration rate per 15 second observation (D). Values which are not significantly different (*P* > 0.05) share common superscripts. Solid lines indicate fitted linear regression.

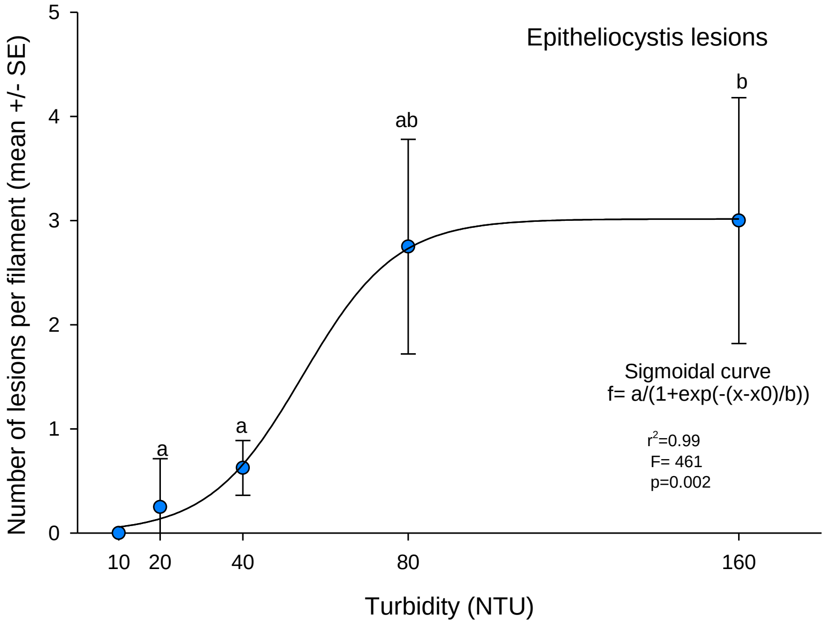


Figure 3.6 Mean number (±SE) of epitheliocystis lesions per filament in the gills of snapper for each of the five turbidity treatments after 30 days. Values that are not significantly different (*P* > 0.05) share common superscripts. Solid lines indicate fitted linear regression.

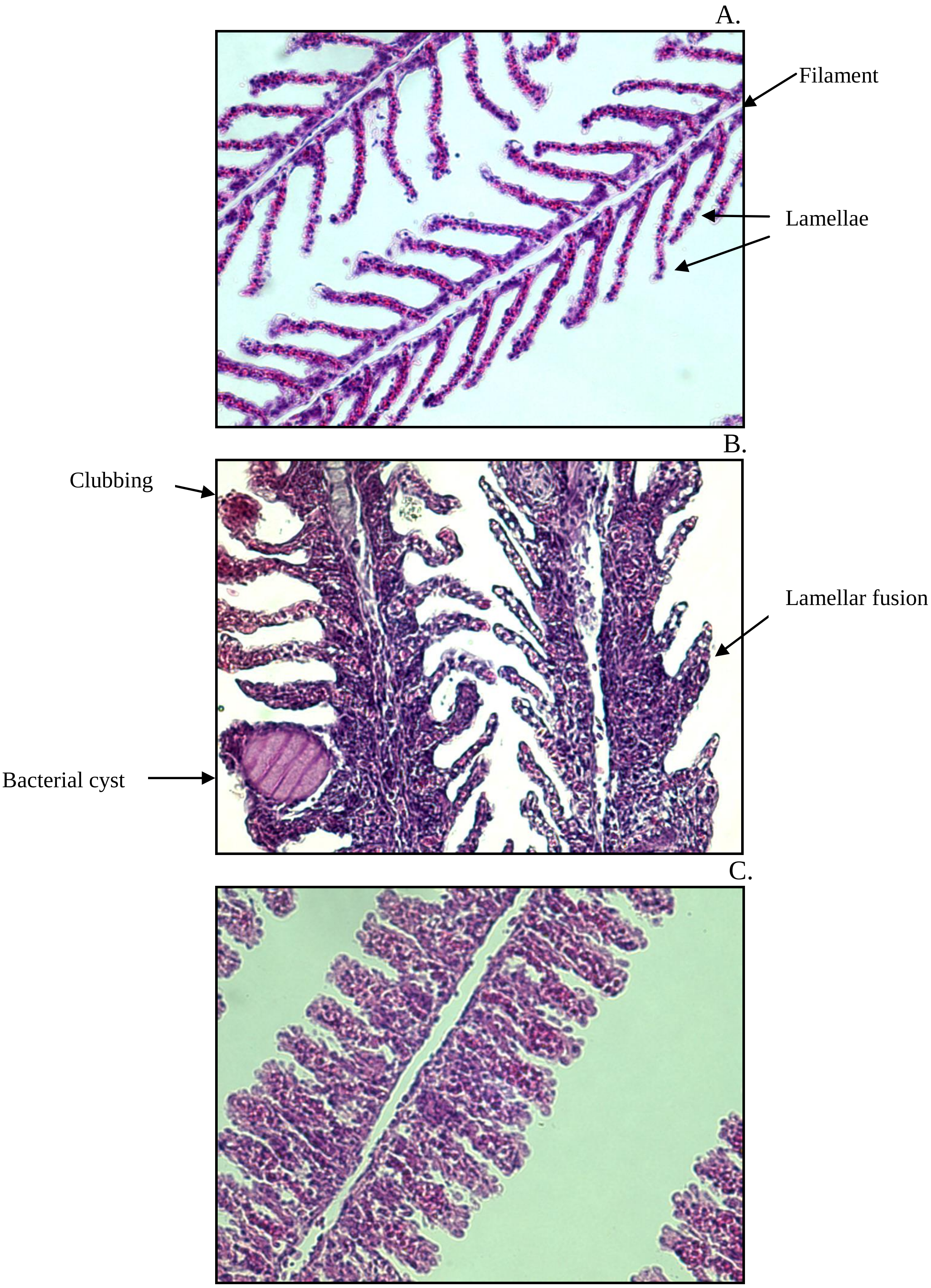


Figure 3.7 Changes in juvenile snapper gill structure with increasing suspended sediment loads. Normal gill filament from a Rangaunu Harbour juvenile snapper (A), an example of hyperplasia (cell proliferation) of gill filaments (B), especially at the base and tips (clubbing) of the lamellae, from a juvenile snapper in the 160 NTU treatment level of the tank-based experiments. Note also the presence of a bacterium (lower left corner); thought to be associated with higher levels of stress in fish (and associated reduced resilience to infection), an example of hypertrophy (thickening) and shortening of a gill filament from a Manukau Harbour juvenile snapper (C).

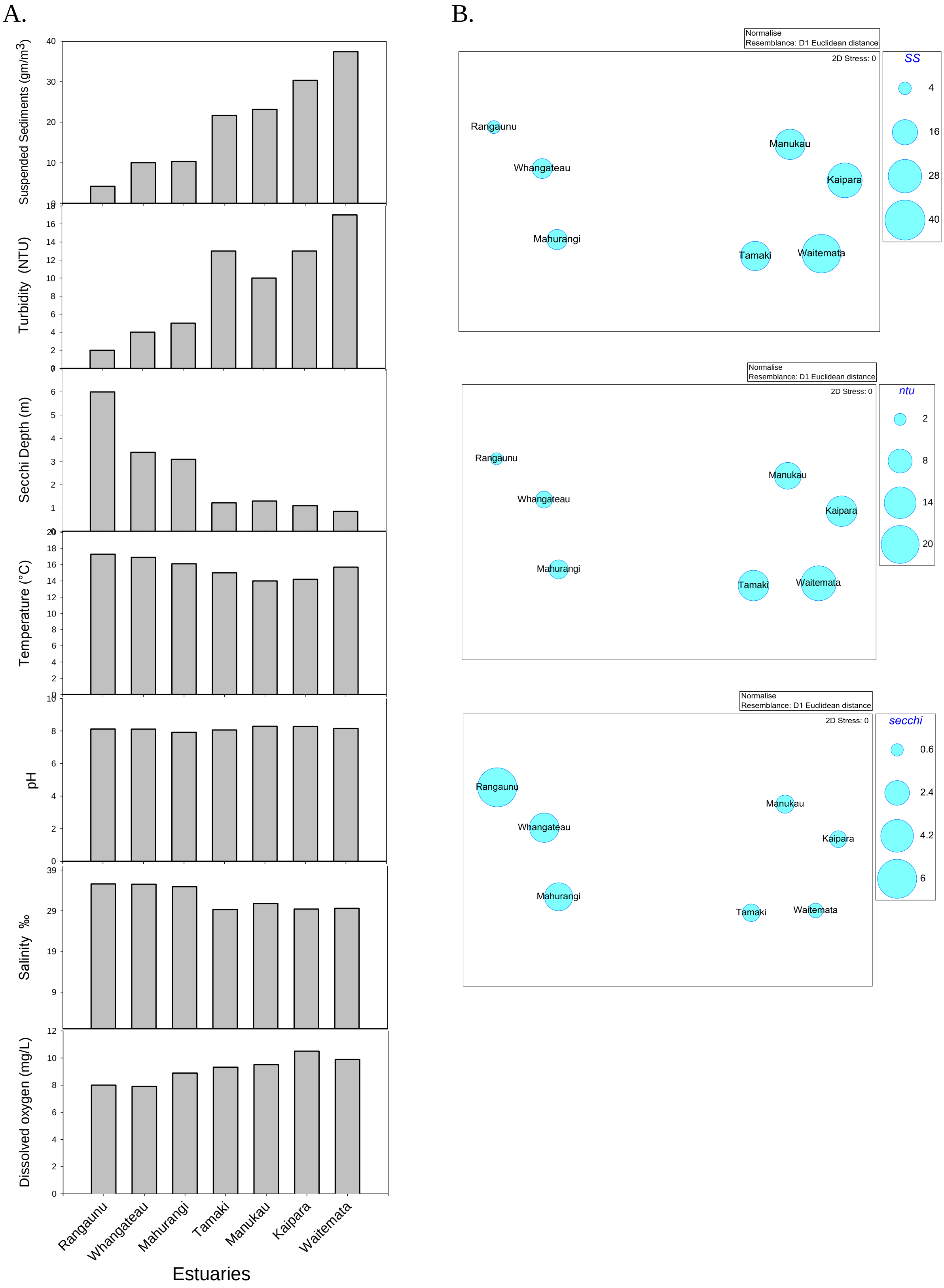


Figure 3.8 Environmental variables plotted by the seven harbours sampled (A). MDS ordination of combined environmental variables plotted by harbour and superimposed bubble plots with values for three environmental variables showing most variation (B). SS: suspended sediments g/m^3; NTU: Nephelometric turbidity units; Secchi: Secchi depth (m).

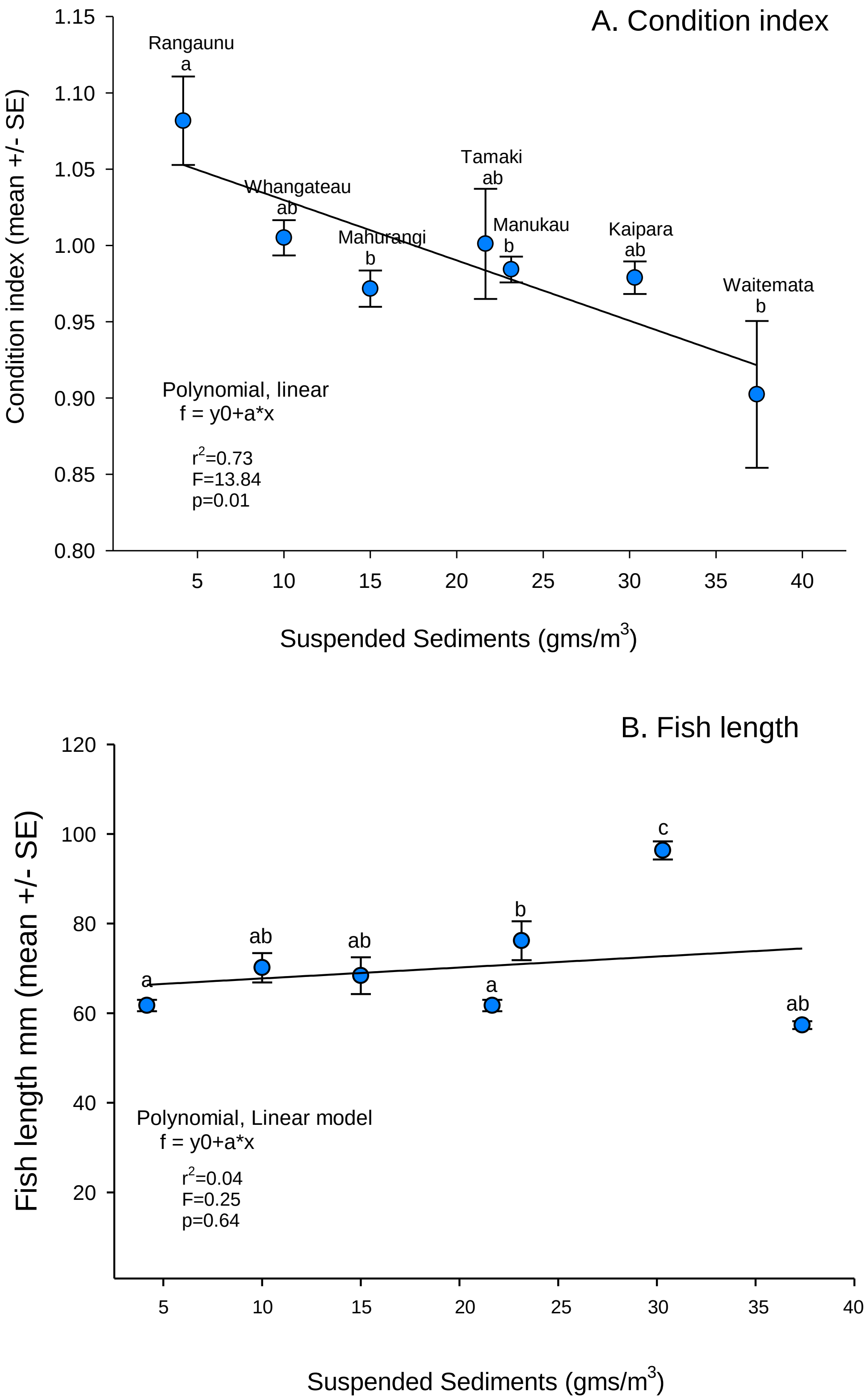


Figure 3.9 Variation in relative condition index (A) and average length for juvenile snapper (B) collected from seven northern New Zealand estuaries. Values which are not significantly different (*P* >0.05) share common superscripts. Solid lines indicate fitted linear regression.

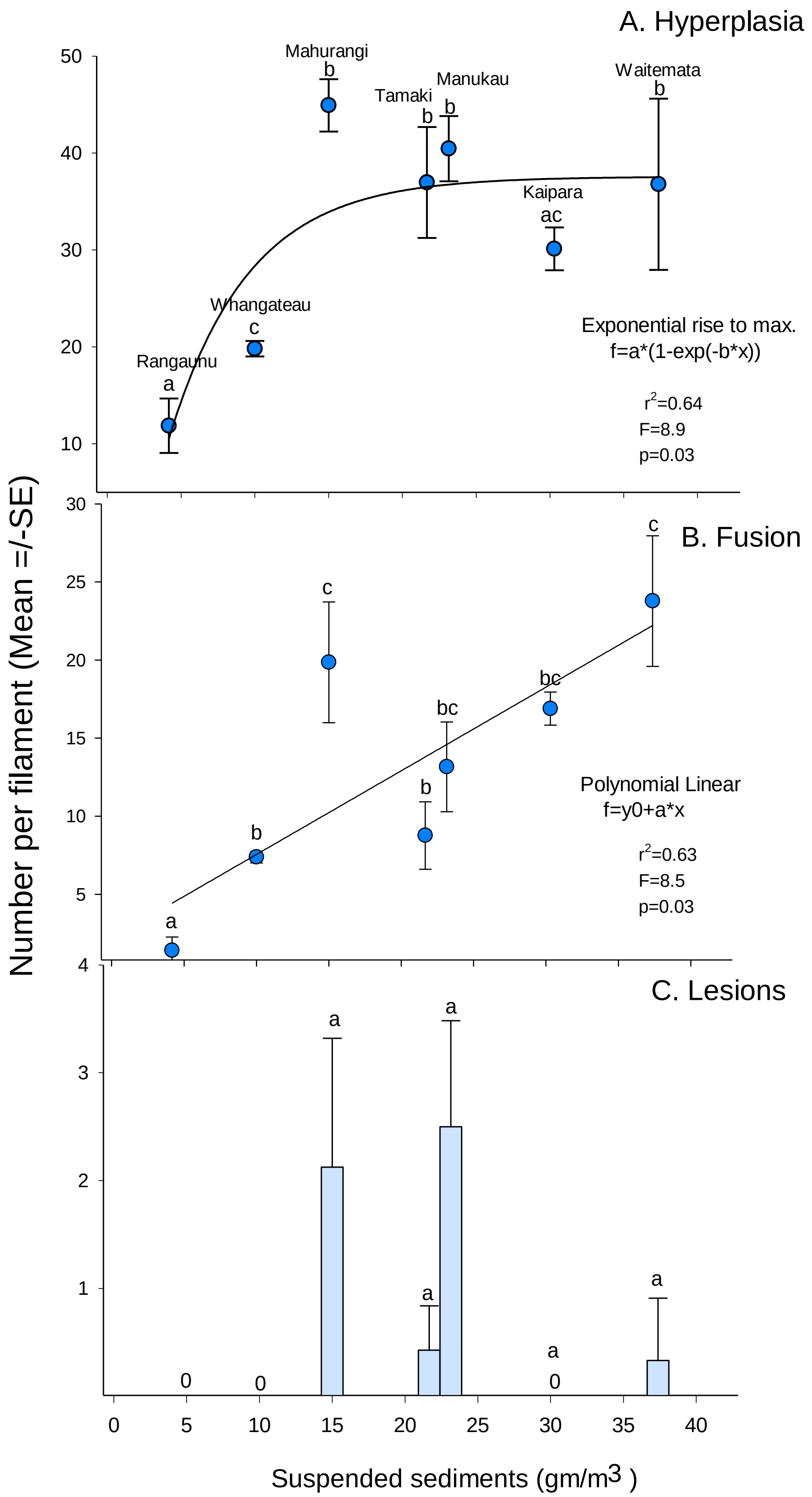


Figure 3.10 Mean number of lamellae per filament (±SE) with epithelial hyperplasia (A), lamellar fusion (B) and epitheliocystis lesions (C) in the gills of juvenile snapper from each of the seven harbours. Values which are not significantly different ($P > 0.05$) share common superscripts. Solid lines indicate fitted linear regression.

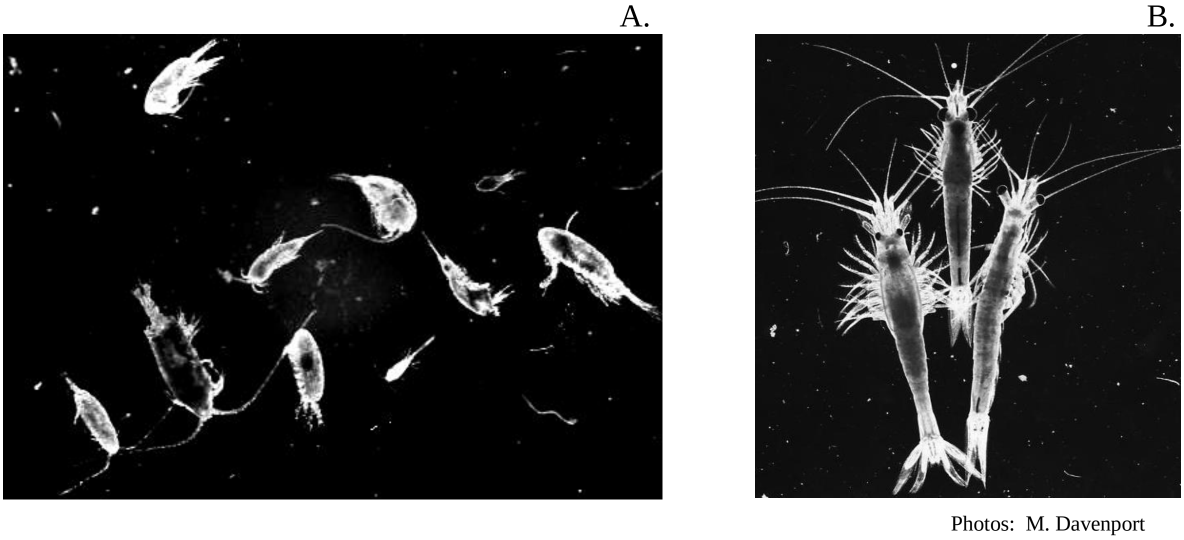


Figure 3.11 Examples of 'pelagic' and 'benthic' prey categories consumed by juvenile snapper calanoid copepods (A) and mysids (*Tenagomysis* sp.) (B).

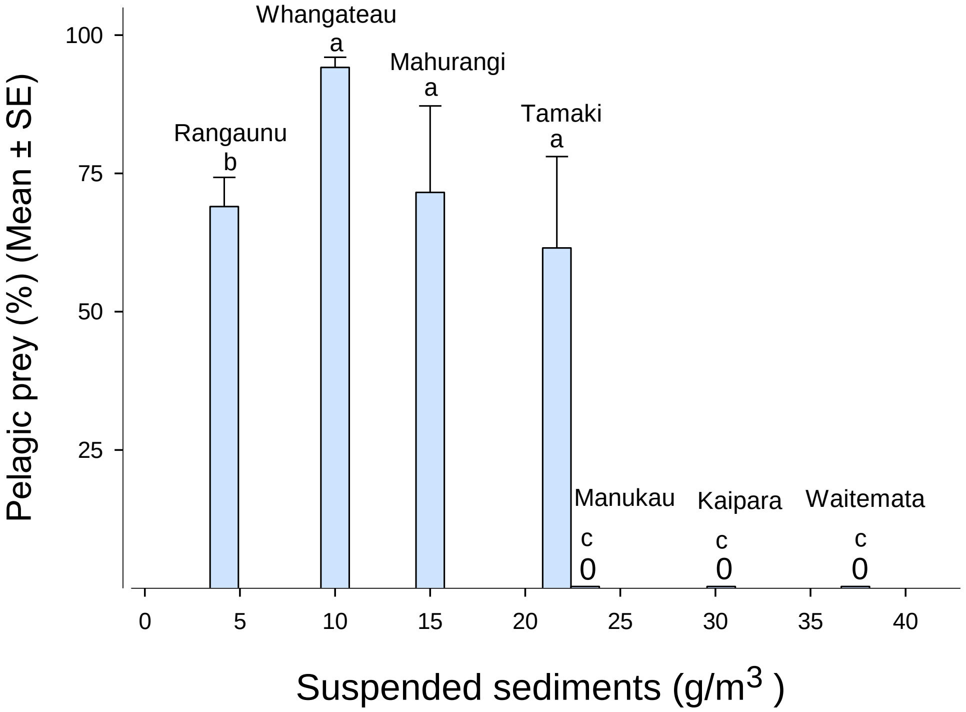


Figure 3.12 Relative proportions of pelagic prey biomass consumed by juvenile snapper with increasing suspended sediment levels across each of the seven harbours. Values which are not significantly different ($P > 0.05$) share common superscripts.

Table 3.1 (A) ANOVA results for mean numbers of mysids consumed by juvenile snapper at five turbidity levels and four substrate types over 30 minutes.

Source	df	*F*	*P*	Sign.
Turbidity	4	128.21	2.20x10^{-6}	***
Substrate	3	7.436	0.001	***

Table 3.1 (B) ANOVA results for effects of long term (30 day) exposure of juvenile snapper to five turbidity levels on weight, histopathological changes in gills[1] [2] [3], and respiration rate.

Source	df	*F*	*P*	Sign.
Weight variation (%)	1	3.98	0.053	
Epithelial hyperplasia[1]	4	10.249	2.09x10^{-5}	***
Lamellar fusion[2]	4	5.4949	0.0021	**
Bacterial lesions[3]	4	4.2591	0.0063	**
Respiration rate	4	3.8392	0.0145	*

Table 3.2 ANOVA results for mean change in condition indices, fish lengths, histopathological changes in gills[1] [2] [3], and proportion of pelagic prey consumed for the seven harbours sampled.

Source	df	*F*	*P*	Sign.
Condition indice	6	4.5225	0.0012	**
Fish length	6	7.5114	1.66x10^{-5}	***
Epithelial hyperplasia[1]	6	12.294	1.45x10^{-7}	***
Lamellar fusion[2]	6	14.808	1.52x10^{-8}	***
Bacterial lesions[3]	6	2.7001	0.0273	*
Pelagic prey (%)	6	13.979	1.15x10^{-8}	***

NS, not significant; *P<0.05; **P<0.01; ***P<0.001

Table 3.3 Percentage of juvenile snapper with epitheliocystis lesions for long term (30 day) exposure to five turbidity levels.

Turbidity level*	Percentage of snapper with lesions
≤ 10 NTU	0
20 NTU	25
40 NTU	50
80 NTU	66
160 NTU	85

*Nephelometric turbidity levels

Table 3.4 Percentage of juvenile snapper with epitheliocystis lesions in each of the seven harbour's. Harbours are ranked on TSS loads.

Harbour	Percentage of snapper with lesions
Rangaunu	0
Whangateau	0
Mahurangi	37.5
Tamaki	14
Manukau	67
Kaipara	0
Waitemata	33

Table 3.5 Percentage of juvenile snapper with hypertrophy (shortening of lamellae) in each of the seven harbours.

Harbour	Percentage of snapper with hypertophy
Rangaunu	0
Whangateau	2
Mahurangi	20
Tamaki	85
Manukau	80
Kaipara	30
Waitemata	30

Appendix 3.1 Three water quality tests related to sedimentation in streams and estuaries.

Turbidity is an optical property of water where suspended and dissolved materials which include silt, clay, finely divided organic and inorganic matter, chemicals, plankton and other microscopic organisms, cause light to be scattered rather than transmitted in straight lines. Not always correlated with total suspended solids due to the effects of size and shape of particles. Turbidity is normally measured in Nephelometric Turbidity Units (NTU). Prior to this, turbidity was measured using Jackson turbidity units (JTU) which are approximately equivalent to NTU's at higher turbidities

Total Suspended Sediments (TSS) is the actual measure of mineral and organic particles transported in the water column. It consists primarily of silt and clay sized particles which may also carry nutrients, metals, industrial and agricultural chemicals. Measured as dry weight per volume (mg/L)

Water Clarity describes the distance that an organism can see underwater and is affected by suspended and dissolved materials. Correlations between water clarity and turbidity or TSS may vary between watersheds. Water clarity is measured with a secchi disc in cm (adapted from Bash et al., 2001).

Appendix 3.2 Examples of freshwater and estuarine fish responses to suspended sediment concentration and exposure duration combinations. Adapted from Newcombe & Jensen (1996): Wilber & Clarke (2001).

Species	Lifestage	Concentration	Duration	Effect	Source
Freshwater					
Coho salmon *Oncorhynchus kisutch*	J**	240 mg/L (30 NTU)	24 h	Coughing rate increased eightfold	Servizi & Martens (1992)
Coho salmon	J**	60 NTU	3 d	Coughing rate increased significantly; fish moved to substrate. Feeding rate reduced by 33%	Berg (1982)
Coho salmon	J*	30 - 60 NTU	12 h	Feeding rate reduced by 50% Dominance hierarchies broke down (re established at 0–20 NTU). Increased gill flaring	Berg & Northcote (1985)
Coho salmon Cutthroat trout *Oncorhynchus clarkii*	J**	100 NTU 0 - 400 NTU	–	Feeding rate reduced by 70% Feeding rate reduced to 0%.	Gregory & Northcote (1985)
Coho salmon	A**	100 NTU	–	Feeding rate reduced by 40%	Reid (1998)
Coho salmon; Steelhead salmon *Salmo gairdneri*	A**	25-50 NTU	14 d	Reduction in growth rate	Sigler et al., (1984)
Coho salmon	A*	100 NTU 200 NTU	7 d	-Feeding rate reduced -Reduced resistance to bacterial pathogen; increased plasma cortisol	Redding et al., (1987)
Coho salmon Cutthroat trout *Oncorhynchus clarkii*	J**	100 NTU 0-400 NTU	10-15 min	Feeding rate reduced by 70% Feeding rate reduced to 0%	Harvey & White (2008)
Chinook salmon	J**	180 mg/L (35 NTU)	14 d	Reduction in growth rate	Sigler et al., (1984)
Atlantic salmon	A*	>35 NTU 22-42 NTU	2.5 h	Feeding rate reduced Rapid decline in territorial behaviour	Robertson et al., (2007)
Bluegills *Lepomis macrochirus*	A**	190 NTU	3 min	Feeding rate reduced by 50% at 190 NTU	Gardner (1981)
Bluegills	A**	15mg/L	1 h	Reduced capacity to locate prey	Vinyard & O’Brien (1976)
Largemouth Bass *Micropterus salmonoides*	J*	5 NTU 40 NTU	19 h 55 h	Reduced feeding rate Prey switching	Shoup & Wahl (2009)
Largemouth Bass	L**	16 NTU	1 h	Reduced feeding rate Increasing size of prey	Reid et al., (1999)
Northern pike	A*	30 NTU	–	Condition indice decreased	Salonen et al., (2009)
Perch *Perca fluviatilis*	A	15-30 NTU	2 h	Feeding rate reduced to 0%	Nurmien & Hoppila (2006)
Green sunfish	J**	9600 mg/L	1 h	Increased gill vent rates	Horkel & Pearson (1976)
Rainbow trout *Oncorhynchus mykiss*	J*	160 NTU	30 min	Increased epibenthic feeding	Rowe et al., (2003) NZ
Rainbow trout	A*	15 NTU 30 NTU	40 d	Reduction in growth rate	Sykora et al., (1972)
Rainbow trout	A*	5 mg/L to 20-40 mg/L	2 h	45% reduction in reactive distance 80% reduction in reactive distance	Barret et al., (1992)
Goby *Gobiomorphus cotidianus*/Smelt *Retropinna retropinna*	A*	5g/m^3 – 20-40g/^{3}	–	Prey switching from Mysids (turbid) to Chironomidae larve (clear)	Hayes & Rutledge (1992) NZ
Banded kokopu *Galaxis fasciatus*	A**	17–25 NTU	30 min.	Feeding rate reduced by 36%	Rowe & Dean (1998) NZ
Banded kokopu	J**	>25 NTU	20 min.	50% avoidance response	Boubée et al., (1997) NZ
Banded kokopu	J*	20–25 NTU	–	Significant decrease in migration upstream	Richardson et al., (2001)
Banded kokopu	A*	120 mg/L	–	Avoidance: Mean numbers reduced by 89.5% upstream in rivers	Rowe et al., (2000) NZ
Smelt *Retropinna retropinna*	A*	100 mg/L	30 min	Feeding rate reduced	Rowe & Dean (1998) NZ
-Smelt	A**	1700-3000 NTU	24 h	50% mortality rate	Rowe et al., (2004)

-Banded kokopu		17,500-21,000 NTU			
Estuarine					
Artic grayling *Thymallus arcticus*	J**	100 mg/L	1 h	Reduction in feeding rate	McLeay et al., (1984)
Artic grayling	A**	1000 mg/L	42 d	33% reduction in growth rate	McLeay et al., (1987)
Cape Silverside *Atherina breviceps*	A**	3-91 NTU	5 min	Reactive distance declined from 81 mm to 37 mm	Hecht & van der Lingen (1992)
Skipjack *Elops machnata*	L*	2000 mg/L	4 seasons	Prey switching from pelagic in clear estuary (3–8 NTU) to benthic prey (68–91 NTU)	Hecht & van der Lingen (1992)
Pacific herring *Clupea harengus pallasi*	L**	0 –2000 mg/L 1000 mg/L	2 h	Feeding rate reduced Damage to epidermis	Boehlert & Morgan (1985)
Green grouper *Epinephelus coioides*	J**	0-2000 mg/L	6 weeks	Damage to gill structure: hyperplasia; epithelium lining	Au et al., (2004)
Striped Bass *Morone saxatilis*	L**	200- 500 mg/L	42 min	Feeding rate on copepods reduced by 40%	Breitberg (1988)
Striped Bass	L**	485 mg/L	1 d	50% mortality rate	Morgan et al (1973)
Bluefish	J**	800 mg/L	1 d	100% mortality	Sherk et al (1974)
Atlantic maidenhead	J**	800 mg/L	1 d	100% mortality	Sherk et al (1974)
White perch	J**	750 mg/L	1 d	100% mortality	Sherk et al (1974)
White perch	A**	650 mg/L	5 d	Hematocrit increased Damage to gill structure	Sherk et al (1975)
White perch	A**	305 mg/L	2 d	10% mortality rate	Sherk et al (1975)
White perch	A**	985 mg/L	2 d	50% mortality rate	Sherk et al (1975)
Hogchoker	A**	1,200 mg/L	5 d	Hematocrit increased	Sherk et al (1975)
Striped killifish	A**	960 mg/L	5 d	Hematocrit increased	Sherk et al (1975)
Bass striped	A**	1,500 mg/L	14 d	Hematocrit increased	Sherk et al (1975)
Snapper *Pagrus auratus*	L**	10-10,000 mg/L	4-24 h	50% mortality after 12h @156 mg/L for larvae (mouth open); Feeding rate on copepod nauplii significantly reduced (larvae 15 days post hatch)	Partridge & Michael (2010)
Coral Reef					
Damselfish (*Acanthochromis polyacanthus*)	J**	0-180 mg/L (~30NTU)	6 weeks	50% mortality in 180 mg/L; 50% reduction in growth rates compared to control;	Wenger et al (2012)

* Field study ** Laboratory Study A=Adult, J=Juvenile, L=Larvae
100 mg/L ~ 23 NTU

Appendix 3.3 - Future Perspectives

Assessing the ecological status of transitional waters

The use of fish (particularly marine juvenile migrants; Courrat et al., 2009) as ‘biological indicators’ of environmental change in transitional waters has been recently highlighted by the adoption of the European Marine Strategy Framework Directive (MSFD; 2008/56/EC) which aims to achieve ‘Good Ecological Status’ in all water bodies by 2015 (cited in Mee et al., 2008; Martinho, 2009). Fish are thought to be useful indicators of estuarine health based on their sensitivity to changes in water quality, habitat availability and quality and the intensity of fishery exploitation (Whitfield & Elliot, 2002; Courrat et al., 2009; Martinho, 2009). Key advantages of utilizing fish include (a) measurement of physical and chemical attributes of the water column may not be an adequate surrogate measure for ecological responses (b) fish are comparatively long-lived and therefore provide a longer-term record of environmental stress. In contrast, measurements on invertebrates and small plants and physico-chemical variables are subject to greater temporal variance; (c) fish encompass a variety of trophic guilds and thus likely to cover all components affected by anthropogenic disturbance. Finally fish have a high public awareness such that the general public is more likely to respond to environmental initiatives based on fish communities than data on invertebrates or aquatic plants (see reviews Whitfield & Elliot, 2002; Schlacher et al., 2007; Martinho, 2009). This is particularly relevant for northern New Zealand estuaries, given the strong recreational/commercial snapper fishery, and increasing pressures from coastal development expanding northwards from Auckland city.

Chapter Four

The value of different estuarine habitat as fish nurseries

‘Juvenile jack mackerel over seagrass meadow, Urupukapuka Island (BOI)’.

Chapter 4: The value of different estuarine habitats as fish nurseries

4.1 Introduction

Estuarine fishes encounter a complex mosaic of habitats, including biogenic habitats such as mangroves, seagrass meadows, mussel and oyster beds along with non vegetated areas such as intertidal mudflats and sand, each with varying levels of resources, predators, competitors and physico-chemical suitability (Sogard 1994, Grecay & Targett 1996; Craig & Crowder, 2000; Nemerson & Able, 2004; see Chapter Two). Determining the relative values of these habitats in such open, interconnected systems is challenging given the varying requirements of juvenile fish, which alter with age (e.g. Colman, 1978; Morrison, 1990; Jenkins & Wheatley, 1998; Nemerson & Abe, 2004). However, given the increasing levels of anthropogenic stressors and associated habitat degradation, fragmentation and loss (e.g. Edgar et al., 2000b; Kennish, 2002; Caddy, 2007; Morrison et al., 2009; Grech et al., 2011), the identification of nursery habitats and knowledge of the trophic ecology and habitat use of juvenile fishes within these habitats is fundamental for sustainable management and conservation (Nunn et al., 2011).

To date, research on estuarine coastal fish assemblages has primarily focused on quantifying differences in abundance, biomass and/or size distribution of fishes to evaluate the importance of one or two habitats, e.g. mangroves (Bell et al., 1984; Robertson & Duke 1987; Thayer et al., 1987; Blaber et al., 1989; Morton, 1990; see review of Faunce & Serafy, 2006), seagrass (e.g. Bell & Pollard, 1989; Blaber et al., 1992; Connolly, 1994; Jenkins et al., 1997a) and mudflats (Meng & Powell, 1999; Hindell & Jenkins, 2005; Cabral et al., 2007). However, most fish species occur across multiple habitats, often with movement linkages between habitats at a range of spatial and temporal scales.

Despite this, international research comparing the relative abundance of fish and invertebrates concurrently, over multiple habitats such as mangroves, seagrass, and non vegetated habitats are few (but see Laegdsgaard & Johnson, 1995; Edgar & Shaw, 1995b; Rozas & Minello, 1998; Minello, 1999; Nagelkerken et al., 2000b, 2001; Castellanos & Rozas, 2001; Bloomfield & Gillanders, 2005; Hosack et al., 2006; Lugendo et al., 2006). This is even more pronounced for concurrent dietary analyses linking prey availability with stomach contents (but see Sanchez-Jerez et al., 2002; Kaiser et al., 2004; Hinz et al., 2005 and references therein; Stål et al., 2007). Research to date indicates that habitats with high structural complexity (e.g. seagrass meadows) support high diversity of small fish assemblages and invertebrates (Heck et al., 2003; Jung &

Swearer, 2011) by providing greater protection from predation and increased food availability (Hindell & Jenkins, 2004).

Research on New Zealand estuarine fishes has similarly focused on fish density and length frequency distributions both within (e.g. Jellyman et al., 1997; Hartill et al., 2003; Morrison et al., 2002; 2007; in prep; Morrison & Carbines, 2006; Schwarz et al., 2006) and between estuaries (Francis et al., 2005; 2011). Whilst benthic community composition studies have encompassed different habitats types, including sand flats (e.g. Hewitt et al., 1997; Turner et al., 1997), intertidal mudflats (e.g. Grange, 1979; Heinrique, 1980; Pridmore et al., 1990), mangroves (Mason & Ritchie, 1979; Morrisey et al., 2003; 2007; Ellis et al., 2004) and seagrass (e.g. Turner et al., 1999b; van Houte-Howes et al., 2004; Schwarz et al., 2006; Mills & Berkenbusch, 2009) these have generally been limited spatially and confined to one or two habitats (but see Alfaro, 2006; Alfaro et al., 2006; Battley et al., 2011). While these studies provide initial evidence for the differential contributions of these habitats towards biodiversity, they do not directly evaluate and predict the ecological value for fish through trophic dynamics and therefore potential ramifications of their loss.

In comparison to Australian temperate studies (e.g. Edgar et al., 1995a, c) fish dietary research within New Zealand has generally been limited in spatial extent, confined to either single species or groups of species in the same habitat (e.g. flounder: Park, 1984; Pearkes, 1985; Grogan, 1982; Livingston, 1987; Saunders, 1999; Mutoro, 2001; Capone, 2008; blue cod: Jiang & Carbines, 2002; snapper: Kingett & Choat, 1981), (but see Usmar, 2009; Williams, 2009). Subsequently, large information gaps exist, particularly for larvae and 0+ juveniles with regard to food web dynamics, particularly prey availability within different habitats at varying spatial scales (Sanchez-Jerez et al., 2002; Stål et al., 2007).

4.2 Aims

In the present study, inshore juvenile fish communities and benthic invertebrates were sampled concurrently at seven localities across northern New Zealand to determine any consistent relationships between fish assemblages and environmental parameters such as habitat type (i.e. sand flat, seagrass, mangrove and intertidal mud) and invertebrate prey. The objectives were to describe and compare the feeding ecology of juvenile fish (<125mm) assemblages and how this varied ontogenetically, both within and between habitats and estuaries across large spatial scales. Knowledge of specific prey types consumed at different sites can elucidate how habitat selection may operate, given that fishes are known to grow faster when consuming a preferred prey as opposed to an equal quantity of an alternate prey (Lankford & Targett, 1997; Nemerson & Abe, 2004; Chapter Three).

This study forms part of a larger New Zealand wide research program that aims to quantify the importance of seagrass meadows, and mangroves within northern New Zealand waters to coastal fish.

4.3 Methods

4.3.1 Study Locations

Sampling was undertaken in seven northern New Zealand estuaries encompassing both east (Rangaunu, Urupukapuka Island, Bay of Islands (BOI), Mahurangi, Tairua) and west (Kaipara, Manukau, Kawhia) coasts along a 551km latitudinal gradient (from north to south) (Fig. 4.1). The seven estuaries differed considerably with respect to their morphology, size, hydrology, sediments and degree of exposure. A summary of the key environmental characteristics of each estuary is given in Chapter One (Table 1.1).

4.3.2 Habitats

4.3.2.1 Seagrass

Seagrass flora within New Zealand is represented by a single species (*Zostera muelleri*) which occurs predominantly intertidally, but also extends into the shallow subtidal of sheltered estuaries, and permanently submerged meadows of a small number of offshore islands where water clarity is greatest (max. depth recorded is 7m), (Turner & Schwarz, 2006). It forms extensive monospecific beds, or mosaics, of discrete patches surrounded by unvegetated sediments, which are particularly vulnerable to water quality degradation (i.e. sedimentation/eutrophication). For this reason, seagrass meadows are often referred to as ‘coastal canaries’ i.e. barometers of marine health (McKenzie & Yoshida, 2009). Significant declines have been associated with increasing turbidity within New Zealand (see review of Morrison et al., 2009). With only 44 km^2 of seagrass remaining within New Zealand, it is a relatively uncommon habitat.

Rangaunu Harbour seagrass meadows, the most extensive in this study, were characterized by sheltered sites, high water clarity and sands with low mud content, with seagrass directly abutting the mangrove forest edge (Fig. 4.2A). Such co occurrence of seagrass and mangroves only occurs in the more pristine northern New Zealand estuaries (Morrison et al., 2012). Other east coast sites, including the offshore Urupukapuka Island (Bay of Islands) and Tairua were similarly dominated with sands of varying grain sizes, low turbidity levels with medium densities of seagrass. In contrast the more exposed west coast sites, particularly the extensive Kaipara and Manukau estuaries (including Kawhia) were characterized by higher turbidities, sediment loads (see Chapter 1, Table 1.1 A&B), current speeds and wave action (Gibbs et al., 2012). Seagrass sites tended to be sparse and patchy in distribution (Fig. 4.2B).

4.3.2.2 Mangroves

New Zealand's inter-tidal mangrove forests are mono-specific (*Avicennia marina*) and confined to the upper half of the North Island, forming often dense stands along the littoral margins of most major estuaries (Morrisey et al., 2007) Mangroves have rapidly expanded in spatial extent over the past 100 years, largely in response to elevated levels of sedimentation from increasing anthropogenic activities in surrounding catchments (Swales et al., 2007, 2008). Mangroves on the west coast are generally characterized by larger, more complex trees (in height, trunk diameter, and first height of branching), higher suspended sediment loads, lower water clarities, high total organic carbon in the sediments, and smaller mean sediment grain sizes. Whilst most east coast mangrove sites comprised smaller, less complex trees, higher water clarities and larger mean sediment grain sizes (Morrisey et al., 2007; see Chapter One, Table 1.1).

Sampling was undertaken between 2003 and 2006, during late summer (March - April). This is a time of year when juvenile fish abundance and species diversity are highest in New Zealand estuaries (M. Morrison pers. comm.). Samples were collected from 27 sites comprising four habitat types (a) seagrass (intertidal/subtidal), (b) sand (intertidal/subtidal), (c) mangroves and (d) intertidal muds which varied with harbour (Table 4.1). Not all habitats were able to be sampled due to weather conditions/gear difficulties and availability of sand sites adjacent (within 100m) to seagrass meadows. Several different capture methods were utilized as no one technique is suitable for all types of habitat (Gillanders, 2006).

4.3.3 Sampling methods

4.3.3.1 Fish samples

Seagrass beds consisting of monospecific stands of *Zostera muelleri* were sampled in March and April 2006 over 2.5 hours either side of low tide for both intertidal and subtidal meadows. Fishes were collected from both unvegetated sand and seagrass habitats using four replicate beach seine hauls across an area of ~50 m x 11 m² into the shore. Nets were 11m wide and 2.3m high, with 9mm mesh and a 4m long cod end. Five meter bridles and 15 m sweeps were attached to each end of the net. When fishing, a sweep/bridle angle of ~25-35º produced a net mouth width of ~ 9m, considered to be most effective for fish retention as per Francis et al., (2005) and Morrison et al., (2002). Samples collected from the deeper subtidal sites (~4m) at Urupukapuka Island (BOI) required the deployment of the beach seine from a boat. The net was set parallel to the shoreline and then hauled straight to shore. Samples sites were selected randomly and were placed at least 100m apart. However, not all habitats were represented in all locations due to technical gear problems and weather constraints for the beach seining (Table 4.1). In addition, bare sites for the Kaipara and upper Rangaunu Harbours were omitted due to the paucity of sufficient bare sediments nearby (i.e. within 100-200m). Additional sampling was undertaken within Rangaunu Harbour to include comparative sampling of the extensive seagrass meadows from the upper and lower harbour. Small fish assemblages from intertidal mudflat sites were sampled utilizing the same methodology, with four replicate beach seine hauls in March 2003, from two sites located in the upper Manukau Harbour.

Mangrove stands *(Avicennia marina)* were sampled between March and April, 2004 with passive fine mesh fyke nets (9mm). These were set parallel to the forest edge just prior to high tide and left to fish until the tide cleared the cod end entrance. Nets consisted of a 5m fyke cod-end, flanked by wings 6m long and 1.5m high to conduct fish to the trap along with surface float and bottom lead-line (Fig. 4.3). This gave an overall sample of fish leaving per 14.5m of mangrove forest. Four estuarine systems were selected for sampling, spanning a perceived cline of environmental degradation from relatively pristine (Rangaunu) to heavily impacted (Manukau). A summary of the characteristics of each is given in Chapter One; Table 1.1, including decadal scale changes in mangrove cover, where available. Within each estuary, three sampling stations were placed along a channel axis, from the start of the mangrove forest in the lower harbour, to the uppermost navigable zone. All fish caught were sorted, identified to species level, and fork length was measured down to the nearest mm. Fish were immediately placed in an ice slurry, then preserved in 10% buffered formalin.

The diets of all fish collected by seine or fyke net were examined except when more than 12 individuals of a species were collected, whereby a subset of 10 animals was utilized ranging in size from smallest to the largest. Fish utilized for gut analysis were injected with 10% buffered formalin to preserve prey items in the field. Stomach contents for dietary analysis were processed using methodology detailed in Chapter Two 2 (2.2.3). In brief, gut contents were identified to species or genus level under a dissecting microscope, and partitioned visually by sieve size class as per Edgar et al., (1994). This enabled the size distributions of ingested prey to be compared with samples from the benthos. Data on the mean ash-free dry weight of animals retained by different sieve sizes, as calculated from the regression equations listed in Table 2 of Edgar (1990a) were used to estimate the biomass of individual prey items consumed. This allowed the proportional biomass of each prey species within the stomach to be calculated.

4.3.3.2 Benthos

At each seagrass site, 100m long transects, perpendicular to the shore, were laid within each seagrass bed and above ground density/coverage visually estimated from an area covering 250m by 150m (Fig. 4.4). Four replicate core samples (13cm diameter, 15cm deep) were placed at random intervals along each transect at least 50m inside the bed where practicable to avoid edge effects where seagrass occurred. Sites were selected with low shore gradients to ensure similar inundation periods. Subtidal cores were collected by divers, with intact cores returned to the surface for processing. The maximum lengths of five randomly selected plants per core were measured. Sand sites were positioned at least 100m away from seagrass sites.

Benthic invertebrates within mangroves were collected by randomly dropping the corer (four replicates) within a ten by ten metre plot located directly behind the fyke net inside the forest edge. Benthos from intertidal mud habitats were sampled (at high tide) with four replicate 0.11m^2 Smith & MacIntyre grabs at each of the two sites. All samples were sieved on a 1mm mesh, and remaining fauna was preserved in 10% buffered formalin solution. In the laboratory, the samples were degassed and preserved in 70% isopropyl alcohol and stained with 6.2% rose-bengal. Samples were washed through a log series of sieves (1.0, 1.4, 2, 2.8, 4, 5.6, 8, 11.2, 16, 22 mm mesh sizes) using the methods described by Edgar (1990a), and the abundance of each size class of each species recorded. Invertebrate production was then estimated using animal sizes and relationships relating production to individual biomass and water temperature as per protocols outlined in Chapter Two (see 2.2.3), (Edgar et al., 1994a; Edgar & Shaw, 1995bc).

4.3.4. Data Analysis

4.3.4.1 Fish

Fish species measuring over 125mm FL (apart from garfish and short-finned eels), were excluded prior to analysis as the primary focus of this study was assessing foraging ecology and potential key environmental factors (habitat type; prey availability) affecting juveniles (0+ year class) fish. To standardize captures between beach seine hauls, fish abundances for seagrass, sand and mudflat habitats were expressed as number of individuals per 100m². Spatial variations in total abundance and individual species density were square-root transformed to improve data normality prior to the analysis in PRIMER 6.0 (Clarke & Warwick, 2001) for both within and between habitat and harbour comparisons. For comparisons between all habitats, which included fyke-net (passive method) and beach seine data (active method), all abundance estimates were transformed into presence/absence data as abundances could not be scaled to a common unit area. Multi-dimensional scaling (MDS) provided an ordination for visual assessment of differences in fish assemblage composition across the 7 locations by habitat type and tidal position.

Analyses of similarities (ANOSIM) was employed to test whether differences in fish assemblages across sites and habitats were significant, with Global R values obtained using 999 permutations. Similarity percentages (SIMPER) using the Bray-Curtis similarity matrix were used to determine which species typified the fish assemblages for each site, habitat and harbour and which made the greatest contributions to any dissimilarities between them.

4.3.4.2 Dietary analysis

To determine resource utilisation of the benthos by fish assemblages, prey items were grouped into 12 general categories (Table 4.2) for analysis. As for Chapter Two, categories were selected to reflect different feeding modes (i.e. benthic/pelagic), and dominance as a prey item, and are not taxonomically equivalent. For example, amphipods, mysids and ‘other pericarids’, although all are crustaceans, were given individual categories due to dominance as a prey item. Similarly, plankton was separated to denote the pelagic component. Statistical analysis was limited to 16 of the 32 taxa examined due to inadequate sample sizes for the others across all sites/estuaries ($n < 20$).

The mean percentage volumetric contributions (biomass) to the different dietary categories for each dietary sample were then calculated and square root transformed, as is appropriate for percentage data (Platell & Potter, 2001). Analysis of similarities (ANOSIM) was utilized to test whether dietary samples of each fish species varied significantly within and between habitats (a) within a single estuary and (b) between the same habitat, across multiple estuaries where applicable. *R*-statistic values for pairwise comparisons provided by ANOSIM were used to clarify the degree to which diets varied within species. Principal Components Analysis was used to determine size-related ontogenetic differences in diet for individual fish species utilizing length frequency data.

4.4 Results

4.4.1 Fish community structure

A total of 27,926 fish comprising 44 species and one species of squid (*Sepioloidea pacifica)* were collected from three surveys (four habitats) at 27 sites, covering seven northern North Island harbours. These included several species of commercial and/or recreational importance. (Table 4.3) The four most abundant families, based on total abundance were Gobiidae (25.6%), Mugilidae (25.1%), Sparidae (15%) and Pleuronectiformes (10.5%). Species in order of decreasing overall abundance were yellow-eyed mullet, snapper, sand goby, exquisite goby, grey mullet, garfish, yellow-belly flounder, mottled triplefin, spotty and parore. Ecological guilds were dominated by marine migrants, comprising estuarine opportunist species (38%), followed by marine estuarine dependent (26%), estuarine migrants (12%) and estuarine residents (20%). Smelt, the only anadromous species caught, and short-finned eels, the single catadromous species captured comprised the remaining 4%.

4.4.2 Spatial variation in fish abundance

Seagrass/Sand

Overall, fish abundance showed strong spatial variability along the latitudinal gradient (north to south), between the east and west coast harbours, and between habitat types within estuaries (Fig. 4.5). Highest densities were recorded at Rangaunu, the northernmost harbour, particularly for inner subtidal seagrass sites where densities reached 256.8±134.8 individ. per $100m^2$. The only exception was for Urupukapuka Island (BOI), the only offshore island seagrass site, which recorded the lowest density at 12.7±4.6 individ. per $100m^2$. Sand habitats generally returned lower catch rates compared to both intertidal/subtidal seagrass sites, ranging from 5±0.47 to 14.7±3.8 individ. per $100m^2$ for Rangaunu and Kaipara respectively. Conversely, Kawhia recorded a slightly higher catch rate for sand than for seagrass (85.5 ± 58 vs. 66 ± 4 individ. per $100m^2$). No latitudinal or coastal trends were evident for species diversity. However, seagrass habitats pooled over all sites had double the number of species counts than for sand habitats (8.5 vs. 4.2).

Mud

Densities in the intertidal mud habitat were comparable to Rangaunu seagrass sites averaging 213±66.5 individ. per $100m^2$, whilst diversity ranged from 8.5 to 11.5 taxa per site (Fig. 4.5).

Mangrove

Fish densities within mangroves showed no consistent latitudinal, coastal or within estuary variability (Fig. 4.5). Rangaunu Harbour recorded the lowest average catch rate (108±13.1 individ. per fyke net), whilst abundance peaked in the Manukau Harbour at 904±605. Elevated numbers and large variability recorded for Manukau were the result of one large catch of grey mullet at a mid harbour site. Similarly, one large catch of pilchards was recorded for Mahurangi. Species diversity ranged from 5.6 to 7.6 for Kaipara and Mahurangi Harbours respectively.

4.4.3 Spatial variations in composition and fish length of dominant species

Abundances varied among habitats with some species showing strong habitat associations (Fig. 4.6, 4.7). Other species such as exquisite gobies, triplefins and yellow-eyed mullet were more cosmopolitan in their distribution, being caught in all habitats and equipment types.

Northeastern harbours, particularly subtidal seagrass meadows were dominated by high numbers of snapper with moderate numbers of garfish, trevally, parore and spotty in addition to estuarine resident species such as gobies (exquisite/sand) and mottled triplefin (Fig. 4.7, 4.8). In contrast, intertidal seagrass was characterized by gobies in northeastern estuaries, with a large catch of yellow-eyed mullet collected in Kawhia, along with modest numbers of speckled sole and sand flounder. Similarly, intertidal sandflat assemblages in northeastern harbours were dominated by low numbers of gobies and by pelagic schooling species such as yellow-eyed mullet, and anchovy for the subtidal sandflats, particularly on the west coast.

Demersal flounder species (yellow-belly, sand and speckled sole) comprised 65% of the intertidal mudflat assemblage. Other common species in order of decreasing abundance were exquisite goby, yellow-eyed mullet, sprat and mottled triple-fin. Mangrove assemblages were dominated (86%) by small semi-pelagic species which included mullets, pilchards, anchovies and smelt (Figs. 4.6, 4.7). Demersal species such as the estuarine triplefin, short-finned eel and flounder comprised the remaining assemblage. Estuarine residents including exquisite goby and mottled triplefin were caught in modest numbers.

Size-frequency distribution of the sixteen dominant species, (Figs. 4.9-4.12) show that all habitats were dominated by juveniles (<125mm FL; excluding garfish and short-finned eels) or adults of small sized species (seagrass/sand, 98.5%; mud, 97.5%; mangroves, 98%). Overall there was little evidence for latitudinal or coastal variation in length frequencies. However, higher abundances of fish species greater than 125mm FL (e.g. 52% of snapper) were caught at Urupukapuka Island (BOI). Snapper showed some spatial ontogenetic variation within Rangaunu Harbour, with higher numbers of juveniles (15-25mm FL) collected from the lower harbour sites for both intertidal and subtidal seagrass. Upper subtidal seagrass snapper were slightly larger, averaging 40-60mm FL (Fig. 4.9).

4.4.4 Seagrass characteristics

Overall, blade length varied with tidal position (Table 4.4). This was consistent across both coasts. Intertidal seagrass averaged 9.6 (±0.12) cm, while blade lengths for subtidal meadows were significantly longer at 22.8 (±2.4) cm. However, density and coverage varied geographically. Subtidal meadows, extending to ~4m depth in lower Rangaunu Harbour were the most verdant and dense of all the harbours sampled, followed by BOI (Fig. 4.4). Medium densities were recorded for the remaining northeastern harbours (both intertidal & subtidal sites), with either ‘continuous’ or ‘patchy’ coverage observed. In contrast, seagrass coverage at the more exposed west coast meadows of Kaipara and Kawhia was sparser.

4.4.5 Benthos

Species abundance, diversity and composition

A total of 212 invertebrate taxa were collected from the benthic sampling. Total abundance of benthic invertebrates varied considerably between habitats (Fig. 4.13). Highest densities were recorded from east coast seagrass sites, particularly the subtidal, with associated longer blade lengths and higher water clarities, with densities reaching a peak of 16,800 ±2472 indiv. per m^2 from the lower Rangaunu Harbour. By comparison, densities for sandy substrata were appreciably lower for all sites averaging just 2528±1202 indiv. per m^2. Conversely, mangrove habitats from the west coast, recorded higher overall numbers than those from the east, averaging 3536±269; 1471±326 respectively. Muddy substrates recorded the lowest density at site M2 with 213±54 indiv. per m^2. Corresponding trends were evident for both species richness (Fig. 4.13B), biomass and productivity (Fig. 4.14), with higher values consistently associated with vegetated seagrass habitats, particularly subtidal, followed by intertidal, sand, mangroves and lastly mud habitats.

Seagrass/Sand

Sandy habitats were characterized by higher numbers of infaunal species (~40%), including spionid polychaetes (viz. *Magelona dakini* & *S. benhami)*, in addition to bivalves (24%) such as *Austrovenus stutchburyi* and *Nucula hartvigiana*. Bivalves dominated intertidal seagrass meadows, particularly for the lower Rangaunu Harbour comprising 68% of total numbers. Higher densities of cockles, *A. stutchburyi* >5.6mm were also collected from these sites (see Appendix 4.1), in addition to *N. hartvigiana* and the wedge shell *Macamona liliana*. Modest numbers of the invasive date mussel (*Musculista senhousia*) were collected in the Kaipara Harbour. Capitellids (e.g. *Notomastus tenuis*) and spionids such as *Aquilaspio aucklandica* dominated polychaete densities. Overall, modest numbers of gammarid amphipods averaging 367±117 indiv. per m^2 were collected from the intertidal seagrass sites with the exception of Tairua, which recorded the highest densities of juvenile *Aora typica* (10,792.4 ±4134 indiv. per m^2) in this study.

Subtidal seagrass, particularly the northeastern sites of Rangaunu and BOI were distinguished by the highest species diversity, averaging 42 and 32 taxa per core respectively (Fig. 4.13). A diverse range of gastropods were recorded, especially for BOI (24 sp.). These included high densities of the small rissoid snails (*Estea zosterophilia* & *Eationella sp*.), in addition to *Maoricolpus* sp., *Marginella* sp. and bubble shells Haminoidiae, Olividae (olive shells). BOI polychaetes included tubeworms Sabellidae, Serpulilidae and Onuphidae. Other mobile epifauna included modest numbers of amphipods (395.5±112 indiv. per m^2), hermit crabs, *Pagurus* sp. and shrimps (*P. affinis*).

In contrast, tubicolous polychaetes including *Chaetopteris* sp., owenids and spionids dominated Rangaunu subtidal seagrass assemblages comprising 45% of total numbers. Bivalves (viz. *N. hartvigiana*) comprised a further 25% of infauna. Mobile epifauna (category 'Other') included brittlestars (*Amphiura* sp.) and actinarid anemones in addition to crabs and caridean shrimps, which contributed disproportionately higher biomass/productivity estimates due to their larger size. Amphipod (*Paradexamine* sp.) densities were considerably higher at 1,883±904 indiv. per m^2 although only contributing 11.5% of total numbers. By comparison, gammarid amphipods dominated subtidal seagrass sites at Kaipara (51%) recording 2,467±370 indiv. per m^2. Polychaetes including the Sabellid *Owenia fusiformis*, Spionid sp., and *Euclymene* sp. contributed 24% to overall densities. Low numbers of crabs (e.g. *H. crassa*, *Halicarcinus* sp.) were also present.

Mangroves

Polychaetes dominated infaunal abundance at almost all sites, with *Heteromastus filiformis, Scoloplos cylindrifer and Aricidea* sp. being the most common species. Amphipods were predominantly infaunal (e.g. *Corophium* sp., *Phoxocephallid* sp.), with highest densities at Rangaunu. High abundances of the small gastropod *Potammopyrgus anitipodarum* were collected from the Manukau and Kaipara Harbours, whilst crustaceans (viz *H. crassa*) were present in modest numbers particularly for the west coast sites, and dominated the biomass and productivity estimates (Fig. 4.13A,B). Interestingly, diversity was roughly constant across all harbours.

Mud

Decapods, predominantly *H. crassa* and the shrimp, *Pontophilus australis* comprised ~50% of the epifauna for both sites (Fig. 4.13A, B). A more diverse community of polychaetes (total of 15 taxa across all samples) dominated infaunal abundances. Spionidae (e.g. *Aquilaspio aucklandica*; *Aonides oxycephala* & *Scolecolpides* sp.) were the most abundant Family, followed by Capitellidae (*Heteromastus filliformis*) and *Nereid* sp., along with moderate numbers of bivalves (i.e. *Austrovenus stutchburyi; Nucula hartvigiana*).

4.4.6 Multivariate community analysis

Harbour

MDS ordination plots on the overall fish assemblage data (presence/absence) shows mangrove sites clustering together on the right hand side of the first axis while those from mudflats tended to cluster on the upper right hand side. Seagrass, particularly those on the east coast (Rangaunu, BOI, Tairua) were largely found on the lower left hand side. Sand and west coast seagrass sites were found interspersed with mud sites (Fig. 4.15). ANOSIM tests revealed fish assemblages were significantly different from all four habitats sampled (Global R=0.467, p<0.001) (Table 4. 5).

Seagrass

MDS ordination of seagrass and sand habitats (density data) showed a similar trend with east coast seagrass sites clustering together in the lower half of the second axis, and west coast sites generally occupying the upper half. Kaipara was the only exception, with subtidal seagrass grouped with eastern sites (Fig. 4.16). ANOSIM comparisons revealed significant differences between fish assemblages of all five harbours sampled (Table 4.6). SIMPER analysis indicated that the same seven to ten species (>5% contribution) collectively contributed ~80% towards dissimilarity between harbours (Table 4.7). Overall, contributions of individual species to the dissimilarities was moderate (~15-30%). These included in order of importance sand goby, snapper, exquisite goby, garfish, spotty, mottled triplefin for the northeastern harbours, with additional species such as trevally, speckled sole, parore and anchovy contributing ~6% towards dissimilarity for the more southern harbours.

Mangroves

Spatial structure of mangrove fish communities (density data) also varied significantly with harbour (Global R=0.682, $p<0.001$). Position within the harbour was not significant (Global R= 0.15, $p<0.08$). MDS ordination shows east coast sites (Rangaunu & Mahurangi) lying above those from the west coast (Manukau & Kaipara) (Fig. 4.17). However, due to the limited number of sample replications, pairwise comparisons were significant only at the 10% level (Table 4.8). SIMPER analysis revealed grey mullet contributed the most towards species dissimilarity (~50%) for west coast harbours (Manukau & Kaipara), while yellow-eyed mullet, estuarine triplefin and pilchards contributed between 15-20% for east coast harbours (Table 4.9).

4.4.7 Within-estuary variation

A. Rangaunu Harbour

Fish assemblages (presence/absence data) within Rangaunu Harbour were significantly different between habitats i.e. mangrove, sand and seagrass (Global R=0.99; $p<0.001$), with the MDS plot showing a clear separation of the three assemblages (Fig. 4.18A). However, pair-wise comparisons between mangrove and sand assemblages (Table 4.10A), although showing complete separation (R=1, $p<0.1$) were not statistically significant due to the low number of permutations (10). Nevertheless, due to the strong Global R, Clarke & Gorley (2006) recommend R values as the most useful to interpret in these instances.

Within habitats, fish assemblage density varied significantly between intertidal and subtidal seagrass, although position within the harbour (upper/lower) was not significant (Table 4.10B). This was reflected in the lack of overlap between the intertidal/subtidal samples on the ordination plot (Fig. 4.18B). SIMPER analysis revealed overall contributions of individual species to dissimilarities between habitats was moderate (<15%) (Table 4.11A).

The same suite of species contributed to within site habitat differences and for tidal position of seagrass sites. These included in order of importance snapper, gobies (sand & exquisite), along with garfish and spotty contributing from between 19.69 to 10.02% respectively for seagrass and sand. Conversely, sand goby, yellow-eyed mullet and snapper were the three most important discriminating species between intertidal and subtidal seagrass contributing 27.47% and 12.45% respectively.

B. Kaipara

Composition of fish fauna (presence/absence data) within the Kaipara harbour varied significantly between habitats (Global R=0.526, p<0.001) with no overlap in the MDS ordination (Fig. 4.19A). Pairwise comparisons (Table 4.12A) of assemblage composition between habitats, although significant were not particularly pronounced for seagrass and sand habitats (R=0.309, *p*<0.048). As for Rangaunu Harbour, mangrove and sand communities exhibited strong separation on the MDS although this was not statistically significant due to the low number of permutations (R=1, *p*<0.29). Fish density pairwise comparisons also followed a similar trend (Table 4.12B), but with significant separation of intertidal and subtidal seagrass communities (R=1, *p*<0.02), reflected in the MDS ordination (Fig. 4.19B).

SIMPER analysis revealed a different suite of species than those for Rangaunu, (Table 4.13A), with generally low overall individual contributions to dissimilarities of between 6-7%. The few qualifying as discriminatory species (~12-14%) between habitats included anchovy, grey mullet, short-finned eels, snapper and yellow-eyed mullet. For seagrass and sand sites, anchovy and exquisite goby were the important discriminating species between seagrass and sand sites. Similarly exquisite goby and yellow-eyed mullet typified differences between intertidal and subtidal seagrass assemblages (Table 4.13C).

Mud

Fish assemblages (presence/absence data) within the Manukau Harbour were significantly different between mangrove and intertidal mud habitats (Global R=0.832, *p*<0.006), with the MDS plot showing a clear separation of assemblages (Fig. 4.20). Important discriminating species for mangroves included grey mullet, short finned eels and smooth pipefish, contributing between ~10 -7% respectively (Table 4.15). Speckled sole, sprat and exquisite goby characterized intertidal mud habitats contributing 7-8.6%.

4.4.8 Prey utilization

Broad dietary composition

Diets of the 22 most abundant fish species were investigated. Stomach contents of 966 fishes were examined of which 60 were empty. Ninety five prey taxa were identified, with the overwhelming majority (88%) of fish feeding on small epibenthic crustaceans (Fig. 4.21). Overall, prey assemblage biomass was dominated by gammaridean amphipods (44%), mysids (15%), zooplankton (13%) and decapods (13%). Species with atypical diets included bridled goby and grey mullet which fed predominantly on detritus/fine algae; yellow-eyed mullet which consumed red/green algae (*Polysiphonia* sp. & *Ulva* sp.) along with detritus, and garfish which was the only species to consume substantial quantities of seagrass in addition to zooplankton.

Dietary variation with habitat

There were strong habitat related differences in diet. Fish collected within mangroves primarily fed on zooplankton. These included schooling species such as anchovy, smelt and yellow-eyed mullet. Harpacticoid and cyclopoid copepods (*Euterpina acutifrons; Corycaeus aucklandicus*) were the most important food items, numerically dominating (87%) the diets of fish collectively within mangroves. Short-finned eels largely fed on larger decapods such as *Helice crassa* and *Palaemon affinis.* In contrast, major food items for intertidal mudflat fish (e.g. sand & yellow-belly flounder and sole) included sessile benthic prey such as polychaetes, nematodes bivalves/siphons, cumaceans, and infaunal amphipods such as *Torridoharpinia hurleyi.* Diets of those species highly associated with seagrass (e.g. snapper, trevally, parore and spotty) were dominated by gammaridean amphipods (35-73% of the total gut biomass), followed by mysids (19%), decapods (12%) and plankton (7%), whilst individuals collected over sand consumed larger proportions of infaunal species. However, in terms of numerical abundance, copepods dominated (~75%) the diets of fish collected from intertidal seagrass for both Rangaunu (viz. *Paracalanus indicus*) and Kaipara Harbours (viz. *Euterpina acutifrons)*, while gammarid amphipods dominated prey consumed over subtidal seagrass sites for both harbours (particularly for the Kaipara 89%). Size of the most dominant prey also varied, with ~50% of amphipods eaten by fish within subtidal seagrass ≥ 1mm, whilst amphipods consumed from intertidal seagrass meadows were generally smaller (~87% ≤ 1mm).

Prey diversity was generally higher for fish collected from seagrass habitats ranging from 51 to 10 species for snapper and garfish respectively. Fish collected primarily from intertidal muds (e.g. flounders, speckled sole), ranged from 16 to 26 species. Whilst, dietary breadth for mangroves was generally lower, between 9-11 taxa, excluding short-finned eels (24). Lowest prey diversity was recorded from sand habitats (5) for sand goby.

4.4.9 Dietary changes with fish size and habitat

The diets of 13 of the most abundant species characterizing each of four habitats of fish in northern estuaries are summarized for each of the 7 harbours (see Figs 4.22 - 4.34).

Mangrove- associated species

Grey mullet (Mugil cephalus)

Grey mullet diet was almost totally composed of fine algal and detrital material, with the exception of Kaipara where early juveniles (0-40mm) consumed plankton (*Euterpina acutifrons*) and one individual consumed decapod larvae (Fig. 4.22). The first two axes of the PCA analysis explained 98.8% of the variation, with no significant differences in diet between mangrove harbour sites (Global R= 0.016 p<0.053).

Yellow-eyed mullet (Aldrichetta forsteri)

Similarly, yellow-eyed mullet were highly dependent on copepods (*P. indicus*) during post-settlement (30-50 mm), with the highest mean number of prey (1200 indiv. per gut) recorded for this study at Urupukapuka Island (BOI) (Fig. 4.23). Increasing quantities of algae/detritus (mangrove sites) and *Polysiphonia* sp./*Ulva* sp. (seagrass sites) were consumed with increasing length (>50mm FL), in addition to modest numbers of mysids, amphipods, and insect larvae (Chironomidae) within mangroves. PCA analysis showed a clear ontogenetic change with size, with the first two axes explaining 84.6% of the variation (Fig. 4.23E). No significant differences were detected between habitats (Global R=0.037, p<0.127). Diet across seagrass sites only was distinguished by higher numbers of plankton ingested, while sand sites had more benthic prey items (i.e. ‘other’ nematodes; plant/detritus), (R=0.319, p<0.003).

Anchovy (Engraulis australis)

Equally, anchovy fed almost exclusively on harpacticoid copepods in addition to moderate numbers of gastropod veligers for all size classes, with smaller numbers of amphipods and polychaetes being consumed by 20-59mm (Fig. 4.24). PCA analysis showed 100% of variance was explained by these three dietary categories. There were barely discernable differences between prey consumed between harbours (R=0.168, p<0.01) and habitat (R=0.168, p<0.013).

Short-finned eel (Anguilla australis)

Diet of short-finned eels was dominated by larger benthic crustaceans, particularly the crab *Helice crassa* across all size classes (168-655mm) (Fig. 4.25). Larger numbers of caridean shrimps, particularly *P. affinis* were consumed with increasing size along with exquisite gobies at some sites. Early juvenile (150-200mm) diet so included amphipods, isopods, bivalves and polychaetes PCA analysis showed a clear ontogenetic change with size, with 93.8% of the variance explained by the first two axes (Fig. 4.25E). No significant differences in diet were detected between mangroves in the four harbours sampled (Global R= 0.119, p>0.958).

Muddy substrate associated species

Yellow-belly flounder (Rhombosolea leporina)

Mysids were the major prey item for early juvenile (20-39mm TL) yellow-belly flounder (Fig. 4.26 C,D) The grapsid crab, *H. crassa* was eaten by all size classes. Numbers declined with increasing size class. Conversely, bivalves/siphons (viz. *Paphies australis*) became increasingly important above 60-79mm (Fig. 4.26 B,C) and were largely eaten over the intertidal mudflats. Polychaetes (e.g. *Neanthes* sp.) were consumed between 60-99 mm. PCA analysis of the major prey categories revealed 76.5% of variability was explained by the first two axes (Fig.4.26 D), revealing a clear ontogenetic dietary shift. ANOSIM analysis found no significant differences in diet between mud and mangrove habitats (Global R=0.063, p<0.864) and between harbours (Global R=0.101, p<0.750).

Sand flounder (Rhombosolea plebeia)

Similarly, newly settled sand flounder (<25mm) consumed greater numbers of mysids (although sample size was small 6 indiv.) (Fig.4.27 D,E). Dominance of nematodes (category ‘Other’) increased with size class. Modest numbers of amphipods (*Corophium* sp.), zooplankton and cumaceans were consumed between 0-99 mm, while juvenile crabs became more important with increasing size. Differences in prey composition between habitats were not particularly pronounced, although significant (Global R=0.362, p<0.001). Pairwise comparisons revealed that diet was most different between sand and mangrove habitats (R=0.706, *p*<0.001), followed by seagrass and mangrove habitats (R=0.515, p<0.001) (Table 4.16). Seagrass had the highest diversity of prey items (i.e. Kawhia) and was the only habitat to have mussels as a dietary item. Similarly, muddy substrate had a wider prey diversity than sand habitats, which was mostly comprised of nematodes and cumaceans.

Speckled sole (Peltorhampus latus)

Speckled sole preyed primarily on cumaceans (viz. *C. lemuran*), constituting 43% of total biomass, particularly for the larger size-classes (20-99 mm), followed by bivalves (22%), (Fig. 4.28 D,E). PCA analysis explained 80.2% (first two axes) of the variance with the smaller size classes showing association with zooplankton and mysid eigen-vectors along the first axis. As for sand flounder, differences in diet between habitats was not marked (Global R=0.201 $p<0.001$), with mud and sand barely separable (Table 4.17). Mussels distinguished the prey over sandy habitats (Fig. 4.28 A,B), while the consumption of exquisite goby and zooplankton differentiated prey from mud habitats.

Exquisite goby (Favonigobius exquisitus)

Clear ontogenetic shifts in diet are shown in the PCA with 97% of total variation explained by the first two axes, showing an obvious progression of increasing lengths from left to right (Fig. 4.29, E). Biomass was dominated by amphipods (36%) and zooplankton (18%) respectively, which was also reflected in the diet of smaller size classes (20-40mm), in addition to modest numbers of cumaceans and mysids (Fig. 4.29 D). Crab species (viz. *Halicarcinus* sp.) became increasingly important with size. Although significant (Global R=0.173, $p<0.001$), pairwise comparisons revealed there was little discernable difference in diet between habitats (Table 4.18A). Increased ingestion of infaunal species (e.g. cumaceans) characterized prey consumed for both sand and intertidal muds. However, there was an overall increase in decapod consumption for west coast sites (Fig. 4.29A-C). Similarly, dietary differences between habitats within Rangaunu Harbour were barely separable (Table 4.18b).

Seagrass associated species

Snapper (Pagrus auratus)

Ontogenetic shifts in snapper diet were clearly evident, with 83.7% of variation explained by the first two axes of the PCA (Fig.4.30E). Newly settled recruits (20-29 mm) preyed primarily on calanoid copepods (*Paracalanus indicus*) and were the numerically dominant prey item, constituting 44% total biomass, with consumption gradually declining to 1% after juveniles reached 70mm (Fig. 4.30D). Although plankton was consumed at all sites, its importance declined from east to west coast sites. Lower biomass is not apparent in Kaipara/Kawhia (Fig 4.30D) due to the dominance of larger sized crabs and shrimps. Gammarid amphipods (e.g. *Paracalliope novaezealandiae*) and mysids similarly declined with increasing length, averaging 35% and 25% respectively of total biomass for 20-80mm size classes. Conversely, there was an increase in the contribution of larger and more mobile decapods, namely *Halicarcinus* sp., juvenile crabs and the shrimps *Palaemon. affinis* and *Pontophilus australis* for juveniles between 50-100 mm. Modest numbers of polychaetes (*Neanthes* sp.; *Eunicid* sp.) and cumaceans were also eaten above 40mm.

Differences in prey composition between habitats were not especially marked, although significant (Global R=0.275, p<0.001; Fig.4.30A-C). Pairwise comparisons showed that diet was most different between seagrass and mud habitats (R=0.305, *p*<0.008), followed by seagrass and sand (Table 4.19). Prey ingested from sand and mud habitats showed an increased infaunal component (i.e. polychaetes; bivalves). Zooplankton distinguished seagrass sites (Rangaunu), particularly the lower intertidal, whilst Kaipara snapper also consumed small amounts of plant material and cumaceans. Prey diversity reflected this trend, with 22 prey species recorded for seagrass verses 5 for mud sites. Similarly, diets from sandy sites (Kaipara only) had an increased infaunal component (viz. cumaceans; polychaetes), in addition to bivalve siphons. No significant differences in diet were detected between upper/lower (R=0.034, *p*<0.326) and intertidal/subtidal (R=0.129, *p*<0.987) seagrass sites within Rangaunu Harbour, or within Kaipara Harbour (sand & seagrass sites) (R=0.128, *p*<0.07). However, there was a significant geographic effect, albeit small, between Rangaunu and Kaipara subtidal seagrass sites (R=0.201, *p*<0.001), similarly between BOI and Kaipara (R=0.361, *p*<0.001). Kaipara snapper had the highest consumption of amphipods (e.g. Aora sp & *P. novaezealandieae*), both for biomass (60%) and abundance (83%), with plankton constituting only 3% (total numbers). Whilst mysids and zooplankton dominated prey eaten in Rangaunu (44-48% total biomass). However, results may be confounded by virtue of Rangaunu sites having higher numbers of snapper between 20-40mm FL.

Garfish (Hyporhamphus ihi)

Garfish had the narrowest diet range (10 taxa) of seagrass associated species and were predominantly herbivorous, consuming 74% total biomass of seagrass fragments followed by zooplankton (12%) and *Hymenoptera* sp. (8%). Early juveniles (80-119mm) fed primarily on calanoid copepods, along with *Hymenoptera* sp. (category ‘Other’) gathered from the water surface (Fig. 4.31D-E), along with smaller numbers of cumaceans. Consumption of plant material increased with growing size. The same three dietary categories explained 92.4% of dietary variation for the first two axes of the PCA analysis (Fig. 4.31E). There was no significant dietary difference between harbours (R=0.142, *p*<0.998) or between habitats, i.e. sand and seagrass (R=0.121, *p*<0.994). However, within Rangaunu harbour, prey varied significantly between upper and lower sites (R=0.443, *p*<0.001) with calanoid copepods dominating the lower sites, while seagrass and *Hymenoptera* sp. were the main prey for upper seagrass sites.

Trevally (Psuedocaranx dentex)

Gammarid amphipods (viz. *Paracalliope novaezealandiae; Aora* sp.) and mysids were consumed by all size classes, dominating overall biomass, 55% and 27% respectively. Larger size classes (80-119mm) also included cumaceans, fish scales and zooplankton in their diet (Fig. 4.32A-D). No significant differences were detected between sand/seagrass habitats (R=0.147, *p*<0.095). Dietary variation between some harbours (seagrass only) could not be analyzed due to the small sample size. However, Kaipara differed from Rangaunu and Kawhia by virtue of trevally almost exclusively eating gammarid amphipods (~81% biomass), while mysids characterized the latter two harbours.

Spotty (Notolabrus celidotus)

Prey biomass was dominated by amphipods (58.8%), decapods (11.66%) and mysids (13%). PCA analysis showed a clear ontogenetic shift, with 87.8% of dietary variance explained by the first two axes (Fig. 4.33D-E). New recruits consumed mysids until 60-79mm. Amphipods (7 species; both epifaunal and infaunal) dominated the diet, particularly for size classes 40-79mm, thereafter declining substantially with a corresponding increase in the contribution of crabs, viz. *H. whitei*. Modest numbers of bivalves (0.5mm), zooplankton and fish scales were also eaten. ANOSIM revealed only minor differences in diet between seagrass meadows within the four harbours (Gobal R=0.265, *p*<0.005), (Table 4.20). BOI was characterized by the addition of zooplankton and cumaceans to the diet. Within Rangaunu Harbour, higher numbers of juvenile bivalves were consumed from the upper harbour sites, which only marginally differed from the lower seagrass sites where amphipods predominated (R= 0.276, *p*<0.006).

Mottled triplefin (Grahamina capito)

Amphipods were the principal prey, accounting for 45% of the total biomass, followed by decapods (25%) and mysids (12.5%). Plankton comprised the diet of the smallest size class (20mm; only one individual), followed by a sharp transition to amphipods and mysids at 30mm (Fig. 4.34). Numbers of amphipods gradually declined with increasing size, whilst decapods increased in importance. Ontogenetic change was evident in the PCA with a clear progression from left to right, of increasing size classes (Fig. 4.34E). Although significant, variation between habitats was barely discernable (Global R=0.191, *p*<0.001) (Table 4.21). Subtidal seagrass sites were distinguished by elevated numbers of amphipods, while increased proportions of infaunal prey (cumaceans), along with *Chironomidae* larvae were consumed within mangroves (Fig. 4.34 A-C). Mud habitats also had a higher infaunal component (viz. polychaetes, cumaceans & bivalves).

4.5 Discussion

4.5.1 Broad-scale Patterns

4.5.1.1 Fish assemblage composition

The seven northern, North Island estuaries supported relatively high abundances of demersal and semi-pelagic inshore fish compared to other inshore coastal habitats (Francis et al., 2011). Gobiidae, Mugilidae, Sparidae and Plueronectiformes were the most important families. Species in decreasing order of importance included yellow-eyed mullet, snapper, sand goby, exquisite goby, grey mullet, garfish, yellow-belly flounder, mottled triplefin, spotty and parore. Although most estuarine species occur across a range of environmental gradients, this study found many had discernible habitat affinities (e.g. snapper and trevally with subtidal seagrass), with different habitats supporting different species assemblages (Figs. 4.6-4.8; see section 4.5.3). Whilst other species were more cosmopolitan in their distribution (e.g. gobies, triplefins and yellow-eyed mullet), being caught in all habitats and equipment types (Fig.4.7). Several species showed coastal variation, with grey mullet and trevally associated with west coast harbours and parore collected only on east coast sites.

Spatial variation in abundance and length of dominant species

There was little evidence for latitudinal or coastal variation in length frequencies, with all habitats dominated (~98%) by juveniles (<125mm FL) or adults of small sized species (Figs. 4.9-4.12). However, higher abundances of fish species greater than 125mm FL (e.g. 52% of snapper) were caught at Urupukapuka Island (BOI), the only offshore seagrass site. Additionally, snapper, garfish, trevally and spotties showed some spatial ontogenetic variation within Rangaunu Harbour seagrass meadows i.e. increasing length frequencies from intertidal to subtidal (see Fig. 4.9). Snapper also recorded a slight trend for increased length frequencies within the Kaipara Harbour. Size distribution did not differ substantially between different habitats. There was a slight trend for smaller fishes i.e. ~30mmFL (gobies) to be present in sand habitats, in contrast with larger flounder species (~70-80 mm FL) caught within mangrove and mud habitats.

4.5.2 *Fish diet*

Dietary data is largely in accordance with prior research, with species displaying omnivory and broad dietary overlap (particularly for seagrass associated fish), characteristics typical of estuarine fishes (Layman & Silliman, 2002; Sanchez–Jerez et al., 2002, Nunn et al., 2011). Epibenthic crustaceans were the dominant prey of the majority (88%) of the 22 species investigated, with poor utilisation of infauna and plants, consistent with other temperate studies (e.g. Pollard, 1984; Bell & Pollard, 1989; Edgar & Shaw 1995a; Hyndes et al., 1997; Yamada et al., 2010; Hindell et al., 2011; see review of Nunn et al., 2011). Gammaridean amphipods dominated prey biomass (44%) followed by mysids (15%), zooplankton (13%) and decapods (13%). Only three species consumed detritus/fine algae (bridled goby, grey and yellow-eyed mullet), while garfish was the only species to consume substantial quantities of seagrass (Fig.4.21).

4.5.2.1 *Ontogenetic changes in diet*

Ontogenetic dietary shifts were evident for the majority of the 22 species surveyed (Figs. 4.22-4.34) with most fish preying on meiofaunal crustaceans 0.5-1mm in length. Zooplankton (viz. *P. indicus; E. acutifrons),* dominated the diets of new recruits (20-40mm), particularly for seagrass associated species. Consumption of mysids and gammaridean amphipods increased progressively with growth to be subsequently replaced with the ingestion of larger crustaceans such as caridean shrimps and crabs (viz. *Halicarcinus* sp*.; Helice crassa*). Whilst mullet species and bridled goby changed from plankton to fine algae/detritus and garfish switched from plankton to seagrass material. In contrast, diet for flounder species shifted from mysids (20-30mm) to include infaunal invertebrates including cumaceans, bivalves/siphons and polychaetes before progressing to crabs (Figs. 4.26-4.28). These findings concur with other surveys (e.g. Day, 1981; Pollard, 1984; Tanaka et al., 1987; Holbrook & Schmitt, 1988; Edgar & Shaw, 1995b; Horinouchi & Sano, 2000; Platell & Potter, 2001; Kanou et al., 2002; Gning et al., 2009; see reviews by Hemminga & Duarte, 2000; Nunn et al., 2011).

In sum, meiofaunal crustaceans, particularly harpacticoid and calanoid copepods and gammaridean amphipods and mysids were overwhelmingly more important than molluscs/polychaetes in linking primary production to fishes (Jenkins et al., 2011). Consumption of harpacticoid (known to aggregate near the seafloor), and calanoid plankters by newly settled juvenile fish is advantageous given their higher caloric values (i.e. 35% higher than for amphipods), and high protein content (Volk et al., 1984), facilitating rapid growth to escape size-dependent predation (Gning et al., 2009). Similarly, mysids are known to be an important prey for small demersal fish due to their behaviour of remaining stationary on the surface of seagrasses and tendency to ‘settle and bury’ on sand, in addition to schooling in clear, shallow water during the day (Mauchline, 1980; Takahashi et al., 1999; Jumars, 2006; Yamada et al., 2010; see Chapter Three).

4.5.2.2 *Habitat related changes in diet*

Stomach content (prey biomass) also varied with habitat. Overall, zooplankton and herbivorous feeding (detritus/fine algae) dominated the diets of fishes (e.g. anchovy, smelt and yellow-eyed mullet) primarily collected within mangroves, whilst endobenthic prey such as polychaetes, bivalves/siphons, cumaceans and infaunal amphipods characterized prey for fish collected mainly over intertidal mudflats/sand (Fig. 4.21). In contrast, diets of those species highly associated with seagrass (e.g. snapper, trevally, parore and spotty) were dominated by mobile epibenthic prey such as gammaridean amphipods, mysids, decapods, and to a lesser extent plankton. These results are consistent with other studies (e.g. Robertson, 1980; Pollard, 1984; Hostens & Mees, 1999; Linke et al., 2001; Platell & Potter, 2001; Travers & Potter, 2002). Benthic infauna, although abundant within seagrass was largely under-utilized as a food source in accordance with prior studies (Pollard, 1984). Dietary trends tended to reflect the overall relative abundance of those prey in the environment (see detailed section below) with the exception of infauna (e.g. polychaetes/bivalves) and gastropods. This agrees with dietary research on labrids (Australia: Lek et al., 2011 and references therein; Sanchez-Jerez et al., 2002) and flounder (Ireland: De Raedemaecker et al., 2011).

Although significant, there was little discernable dietary difference (i.e. low *R*-statistic values) across multiple habitats for the more cosmopolitan species (e.g. sand/exquisite goby, mottled triplefins). Rather, trends were more reflective of prey availability in the benthos, suggesting opportunistic and/or flexible feeding strategies (Day, 1981). For example exquisite goby and mottled triplefins consumed more infaunal prey when caught over sand/mud/mangroves than for seagrass where more epifaunal amphipods, mysids and zooplankton were consumed. This concurs with Edgar’s (1999) research on two goby species in Western Port Australia. However, schooling species feeding predominantly on zooplankton (e.g. early juvenile yellow-eyed mullet) showed no marked dietary differences between habitats (e.g. mangroves, seagrass), which would be expected given their pelagic feeding strategy (Bloomfield & Gillanders, 2005).

The high degree of dietary overlap recorded in this study suggests that food was abundant, particularly for seagrass associated species over summer (viz. gammarid amphipods). This corresponds with the seasonal peak of abundance recorded for these crustaceans in New Zealand (Choat & Kingett, 1982). Seasonality of predation, particularly on amphipods by seagrass associated fish has been shown in prior studies (e.g. Heck & Orth, 1980a; Heck & Thoman, 1981; Motta et al., 1995 and references therein). Similarly, Baker-Dittus (1978) and Motta et al., (1995) found greatest dietary overlap when foods became abundant.

Prey diversity

Overall, dietary breadth reflected benthic biodiversity (Fig. 4.13). Higher prey diversities were recorded from those fish species occupying habitats with more structurally complex biogenic structure, viz. subtidal seagrass (e.g. snapper, 51 taxa). This was particularly evident for the pristine northeastern harbours with longer blade lengths, providing greater surface area for foraging invertebrates and/or refuge (e.g. Rangaunu; BOI), followed by intertidal seagrass. Fish collected primarily from intertidal mudflats ranged from 16 to 26 species for flounders/sole. Whilst, dietary breadth for mangroves was generally lower, between 9-11 taxa. Lowest prey diversity was recorded from sand habitats (5) for sand goby. These results concur with Jiang & Carbines (2002) survey in Foveaux Strait, where biodiversity of the epibenthos over complex three dimensional biogenic habitats was positively correlated to diversity in the blue cod *(Parapercis colias)* diet. Although not measured in this study, stomach fullness was generally greater in the more complex seagrass habitats (pers. obs.). Increased food consumption may be a response to relaxed predator avoidance behaviours, evident in more open habitats (Allen-Ankins et al., 2012), and/or a reflection of increased food availability.

4.5.3 Habitat level patterns

4.5.3.1 Mangroves

Whilst the role of mangroves as important and/or crucial juvenile nursery areas has been well documented in tropical and subtropical mangrove systems, particularly those of USA and Australia (see reviews by Faunce & Serafy, 2006; Morrisey et al., 2010), with highly diverse and abundant fish and decapod assemblages being recorded (e.g. Laegdsgard & Johnson, 1995; Vance et al., 1996, Nagelkerken et al., 2000a, b; 2001; Gillanders, 2006), research on fisheries within temperate systems is scant (i.e.111 tropical vs. 11 temperate studies), (Faunce & Serafy, 2006; Morrisey et al., 2007; 2010). Notwithstanding this bias, these findings were until recently extrapolated uncritically to temperate mangroves (e.g. Chapman, 1976b, cited in Morrisey et al., 2007). However, recent research in Australia (e.g. Bell et al., 1984; Clynick & Chapman, 2002;

Hindell & Jenkins, 2004; 2005; Blomfield & Gillanders, 2005; Smith & Hindell, 2005; Payne & Gillanders, 2009) has shown fish diversity consistently lower in temperate mangroves than in subtropical and tropical mangroves, with few, generally small bodied species of little or no commercial importance predominating, often equally abundant in alternative habitats.

Results from this survey show mangrove assemblages dominated (86% of total numbers) by small semi-pelagic schooling species, which included yellow-eyed mullet, grey mullet, pilchards, anchovies and smelt (Fig. 4.6-4.8), (see also Morrison et al., in prep, for 7 harbour survey results; Morrisey et al., 2007). Their propensity for aggregating however, make them difficult to sample precisely (Morrison et al., 2002). This was evident for grey mullet in the Manukau Harbour and pilchards in the Mahurangi Harbour (Fig. 4.8). Other demersal species such as the estuarine triplefin, short-finned eel and flounder comprised the remaining assemblage. Sand and yellow-belly flounder were caught in modest numbers, in contrast to the abundant densities recorded over intertidal mudflats in this study (Fig. 4.8) and prior New Zealand surveys (Morrison et al., 2002; Francis et al., 2005; 2011). Estuarine residents including exquisite goby and mottled triplefin were caught in modest numbers. With the exception of grey mullet (West coast only) and short-finned eels, no commercial species were particularly common in mangrove habitats.

The value of mangroves for fish species is usually explained in terms of refugia from predation amongst prop roots/pneumatophores (Nagelkerken et al., 2000a; Laegsgaard & Johnson, 2001; see review of Faunce & Serafy, 2006) and elevated foraging opportunities via high prey abundances. Indeed, some tropical studies have reported higher benthic biomass than adjacent seagrass meadows (e.g. Kolehmainen & Hildner, 1975, cited in Alfaro, 2006; Sheridan, 1997). By contrast, temperate studies of benthos within Australia and New Zealand mangroves have reported more modest density and species diversity (Ellis et al., 2004; Morrisey et al., 2003, 2007, 2010; Alfaro, 2006; Stokes, 2009; Cowles et al., 2009), (Table 4.22A). Findings from this survey concur, with comparable species abundance and benthic diversity recorded for all sites (Fig 4.13).

Subsurface deposit feeders dominated benthic fauna, primarily polychaetes (*Scoloplos cylindrifer*, *Heteromastus filiformis* and other capitellids) whilst the grapsid crab *Helice crassa* dominated biomass and productivity of both Kaipara and Manukau Harbours, with the exception of high numbers of the surface dwelling gastropod *Potamopyrgus antipodarum* at the inner site of both Kaipara and Manukau Harbours. Overall, molluscs and crustaceans were rare.

Interestingly, Rangaunu Harbour, considered the most pristine (high water clarity) of the four harbours, recorded similar species diversity and abundance to that of the other harbours, including the more environmentally impacted Manukau Harbour.

Low benthic density and diversity recorded in this survey may be in part due to increased tannin levels from mangrove detritus/mud (Alongi et al., 2000; Ellice et al., 2004); fewer crab species present as bioturbators in comparison to diverse tropical mangrove assemblages (Alfaro, 2006); cooler temperatures and shorter tidal inundation in New Zealand decreasing the decomposition rates (Alfaro, 2006, 2010), and/or accelerated sedimentation levels resulting in smothering and negative functional effects (Ellis et al., 2004; see review of Morrisey et al., 2010).

Dietary data, revealed that zooplankton (viz. *E. acutifrons*, *C. aucklandicus)* dominated the diets both numerically (87%) and constituted ~50% of prey biomass of fish primarily collected within mangroves (i.e. anchovy, smelt; yellow-eyed mullet) (Fig. 4.21). This agrees with studies in Queensland, Australia (e.g. Blaber, 1980; Robertson et al., 1988; Manson et al., 2005), although consumption of brachyuran zoea was modest in comparison to other surveys (e.g. Robertson & Duke, 1987; Robertson et al., 1988; Rönnbäck et al., 1999). Other trophic groups included algal and detrital feeders i.e. grey mullet, and yellow-eyed mullet (>90mm FL). The remainder (flounder, triplefins & exquisite goby) consumed small benthic epifauna (isopods, crabs) and both epifaunal and infaunal amphipods, while short-finned eels fed on larger decapods such as *Helice crassa* and peracarid shrimps. The extremely low densities of amphipods present in the benthos, suggests that fish may be feeding on the intertidal flats while migrating with the incoming tide (Morrison et al., 2003). Similarly, short-finned eels have been occasionally observed feeding at night over shallow upper harbour seagrass meadows in Whangapoua estuary (Schwarz et al., 2006). Interestingly, this was the only species found to have a positive association with seafloor structure (seedlings/saplings/tree density) which suggests possible utilization of mangroves as a daytime refugia (Morrison et al., in prep.; Morrisey et al., 2007).

In sum, the benthic invertebrate fauna of New Zealand's, mangroves appear to be modest in both abundance, species diversity and productivity when compared to other estuarine habitats (e.g. seagrass this study; Table 4.22A,B), (Alfaro, 2006; Schwarz et al., 2006; Morrison, unpubl. data, and that of tropical mangroves (Table 4.23), (Sheridan, 1997; Schrijvers et al., 1998), thus offering only limited foraging opportunities for small fish. Additionally, fish species abundant within mangroves, fed on food resources ubiquitous throughout the estuary (i.e. zooplankton), or targeted specific dietary components common in mangrove forests and associated seafloor (i.e. fine algae/detritus). Although the possibility of increased availability of planktonic prey within mangroves was not addressed in this study, prior research indicates that seagrass habitat generally has equal or greater total zooplankton and copepod densities than for mangroves and/or bare substrates (e.g. Robertson et al., 1988; Renkawitz et al., 2011). Additionally, provision of refugia for small fish within New Zealand mangroves may also be limited due to *Avicennia marina*'s structurally less complex aerial roots (i.e. short and sparse), (Alfaro, 2006), when compared with the complex buttress roots and multiple growth forms of tropical species. Further, lower tidal inundations make this a temporarily available habitat, which contrasts to the continuous access afforded some tropical mangrove systems which are permanently inundated (e.g. Curacao, Nagelkerken et al., 2001; Florida, Ley et al., 1999). Therefore any advantages accruing from utilization of mangroves must be considered within the overall interconnected mosaic of habitats (Sheaves, 2005). Overall, mangrove forests within New Zealand do not appear to provide enhanced foraging opportunities compared to alternative habitats, nor refugia on a continuous basis. These may be strong contributing factors to the relatively low value of mangrove forests to most species in northern New Zealand (Morrisey et al., 2007; 2010; Morrison et al., in prep.).

4.5.3.2 Mudflat Habitats

Historically, shallow intertidal mudflats have been economically and ecologically undervalued and have been particularly vulnerable to anthropogenic modifications such as dredging and reclamation (Meng & Powell, 1999; Saunders, 1999; Kanou et al., 2002). However, mudflats are now recognized as important juvenile fish feeding areas, particularly for benthic species such as flounder given their propensity for burial in soft sediments to escape predators and substratum-mediated food habits (Davenport, 1979*;* Park, 1984; Pearks, 1985*;* Elliot et al., 2002 and references therein; Nicolas et al., 2007). Results from this study concur, revealing surprisingly high densities and diversity of fish species within the Manukau Harbour, comparable to subtidal seagrass meadows in Rangaunu (Fig. 4.5). Fish assemblages were dominated (65%) by demersal flounder species (yellow-belly, sand and speckled sole), followed by the ubiquitous estuarine residents exquisite goby, mottled triplefin, and to a lesser extent semi pelagic species such as yellow-eyed mullet, sprats, jack mackerel and anchovy. This is consistent with other temperate surveys (e.g. Colman, 1974b; May, 1979; Grogan, 1982; Knox, 1983; Meng & Powell, 1999; Saunders, 1999; Francis et al., 2005; Morrison et al., 2002; Cabral et al., 2007; see Chapter Two).

Major food items for intertidal mudflat fish (viz. flounder and sole) included sessile/buried endobenthic prey such as polychaetes/tentacles, nematodes bivalves/siphons, cumaceans, and infaunal amphipods such as *Torridoharpinia hurleyi*. Moderate numbers of mysids were also eaten. In contrast to prior studies (Hicks, 1984; Park, 1984; Livingston, 1987; Saunders, 1999), consumption of harpacticoid copepods for juvenile flounder was limited and confined to species such as exquisite goby, yellow-eyed mullet and anchovy. Similarly, newly settled flounder recruits sampled from the same sites in spring fed almost exclusively on high densities of the seasonally abundant mysids present (see Chapter Two; Fig. 2.23), reflecting their opportunistic feeding strategy, as reported by other researchers (Pearks, 1985, Saunders,1999; Kanou et al., 2002; Cabral et al., 2007; Yamamoto & Tominaga, 2007).

Benthos was dominated by infauna, comprising a diverse community of polychaetes (15 taxa) (viz. Spionids and Capitellids) along with moderate numbers of bivalves (i.e. *Austrovenus stutchburyi; Nucula hartvigiana*). While decapods predominantly *H. crassa* and the shrimp, *Pontophilus australis* comprised ~50% of the epifauna, for both sites. Given fish densities, the abundance, biomass and productivity of the benthos was surprisingly low (although densities increased in spring, see Chapter Two, Fig. 2.8), with results comparable to some mangrove and intertidal sand sites (Fig.4.13–4.14). However, sampling biases were evident with some prey items not being well represented in the grab samples such as cumaceans, which are known to inhabit soft mud/sand, in addition to mysids. Additionally, fish may be feeding in the deep low tide channels/banks prior to migrating (up to 1000m; Morrison et al., 2002) to the shoreward margins of these extensive tidal flats.

In sum, the extensive intertidal mudflats of Pahurehure Inlet support a diverse and abundant assemblage of juvenile demersal fish species. However, as for mangrove habitats, it is only temporarily available to small fish assemblages due to diurnal tidal constraints.

4.5.3.3 Seagrass/Sand Habitats

Estuary fish use

The question of whether seagrass meadows are valuable as nursery areas for juvenile fish has received extensive coverage and are widely regarded as important habitats for both fish and crustaceans, including juveniles of commercially important species (e.g. Heck & Orth, 1980b; Bell & Pollard, 1989; Sogard & Able, 1991; Gray et al., 1996; Jenkins & Wheatley 1998;

Connolly, et al., 1999; Mattila et al., 1999; see reviews by Hemminga & Duarte, 2000; Travers & Potter, 2002). However some of these accepted paradigms are being challenged (see Edgar & Shaw, 1995c; Gillanders & Kingsford, 1996; Heck et al., 2003).

The high structural complexity of seagrass meadows as distinct from unvegetated habitat has been shown to increase food availability (Robertson et al., 1988; Bell & Pollard, 1989; Edgar 1990b; Connolly, 1994, 1997), provide refuge from predation (Nelson & Bonsdorff, 1990; Mattila et al., 1999, Hindell et al., 2000a, 2001; Orth, 1992; Attrill et al., 2000), attenuate the effects of currents and waves (Eckman, 1987; Foncesca & Koehl, 2006; Foncesca & Cahalan, 1992; Heiss et al., 2000; Peterson et al., 2004; Widdows et al., 2008) facilitating increased deposition of larvae (Connolly & Hindell, 2006).

Fish abundance and composition

Fish abundance in this survey showed strong spatial variability along the latitudinal gradient, generally declining from north to south, between east and west coast harbours, habitat types (i.e. seagrass vs. sand) and within estuary (subtidal/intertidal seagrass), although position within the estuary (upper/lower; Rangaunu) was not significant (Fig. 4.5; Table 4.10). The latter result may have been confounded by a large storm system present for two days prior to sampling subtidal seagrass in the lower Rangaunu harbour which resulted in sizeable amounts of detached seagrass debris in the water column. This may have adversely affected distribution of newly settled recruits via mortality/migration from physical disturbance (waves), (Jenkins et al., 1997a; Moran et al., 2004) and alteration of prey availability due to declines in epifauna, viz. amphipods (Edgar, 1990b; Schwarz et al., 2006), and/or less zooplankton due to higher turbidity levels and inability to capture prey (see Chapter Three).

Rangaunu Harbour, the northern most pristine sheltered estuary, with continuous, extensive meadows and high water clarities recorded the highest densities of the survey, peaking at 661.3 individ. per 100m^2 for an inner subtidal seagrass site (Fig. 4.5). Whilst densities declined eight fold in the more exposed west coast Kaipara Harbour, with associated sparse/patchy seagrass coverage, and higher turbidities (Table 4.4). Urupukapuka Island (BOI), the only offshore subtidal seagrass site was an exception, recording the lowest density of juvenile fish despite higher water clarity and the longest blade lengths (Table 4.4). BOI was characterized by higher numbers of fish >125mm (e.g. snapper 52%) than new recruits. This may be in part due to the sites exposure and depth (>4m) resulting in closer proximity and greater densities of larger predators, minimal unidirectional flows over the meadows (bringing less planktonic food) in addition to longer blade lengths hindering fish foraging efficiency (Heck & Thoman, 1981; Stoner, 1982 cited in Jenkins & Hamer, 2001; Motta et al., 1995 and references therein). These results concur with research from other offshore seagrass sites both within New Zealand (e.g. Slipper Island, Bay of Plenty; Schwarz et al., 2006) and Tanzania (Kimirei et al., 2011) which also recorded lower juvenile densities. This suggests that offshore seagrass beds may provide less fitness benefits for smaller predation prone individuals than more shallow, estuarine locations affording refuge from larger predators (Kimirei et al., 2011; Kimirei, 2012*)*.

Non vegetated habitats are generally characterized by comparatively low species diversity and abundance of fish species (e.g. Australia: Bell & Pollard, 1989; Conolly, 1994; Gray et al., 1996; Bloomfield & Gillanders, 2005; New Zealand: Francis et al., 2005; 2011). In this survey, sand habitats returned significantly lower densities (~20 fold decline) compared to both intertidal/subtidal seagrass sites excluding Kawhia, which recorded a slightly higher catch rate for sand than for seagrass. However, catches for both habitats within Kawhia were dominated with high abundances of yellow-eyed mullet a more cosmopolitan, schooling species prone to large variations in abundance (Fig. 4.7). No latitudinal or coastal trends were evident for species diversity. However, seagrass habitats pooled over all sites had double the number of species counts than for sand habitats (8.5 vs. 4.2), (Fig. 4.5).

These results support prior research within New Zealand (Francis et al., 2005; 2011; Schwarz et al., 2006; Miller, 2011; Morrison et al., 2007, 2012; unpubl. data) and Australia (e.g. Orth & Heck, 1980; Orth et al., 1984; Bell & Pollard, 1989; Murphey & Foncesca, 1995; Edgar et al., 1995c; Jenkins et al., 1996), with seagrass, particularly subtidal meadows with higher densities/ continuous cover, longer blade lengths and associated higher water clarities (e.g. Rangaunu) recording higher overall density, and biomass (pers. obs.) for juvenile fish than for unvegetated sites (Fig. 4.5; Table 4.4). Such variations may be due to spatio temporal environmental differences between beds (e.g. intertidal sites comprising a temporarily available habitat, Heck & Orth, 1980; Bell and Westoby 1986 a,b; Bell et al., 1987; Worthington et al., 1992), but may also relate to depth distribution of pre settlement larvae (Murphy et al., 2011); in addition to greater food availability and refuge relative to intertidal seagrass/sand habitats (Bell & Pollard, 1989; Gray et al., 1996).

Fish assemblage composition

Juveniles of many species had discernible habitat affinities, with snapper for example, almost exclusively found in high densities within subtidal meadows (Figs. 4.6-4.9). Densities are the highest recorded for a variety of habitats (e.g. coralline turf; horse mussels, sponges, sand/mud) within New Zealand (Table 4.23), suggesting a significant conservation value as a nursery habitat. Other discriminating species for northeastern subtidal meadows included garfish, gobies, parore, spotties, and triplefins (Fig. 4.8, Table 4.11). Whilst additional species such as trevally, and speckled sole, were recorded for west coast assemblages.

In contrast, both sand and intertidal seagrass habitats were dominated by small cryptic species such as gobies and/or schooling species including yellow-eyed mullet and anchovies along with modest numbers of speckled sole and sand flounder in the more silty/turbid west coast estuaries (i.e. Kaipara and Kawhia), (Figs. 4.6-4.7). Given the lack of cover (sand) and/or sparse, patchy distribution of intertidal seagrass, camouflage and schooling provide refuge to avoid predation (Jenkins et al., 1997b). These species were more cosmopolitan in their distribution, also being found in seagrass, mangroves and mud habitats.

Benthos

Numerous international studies have documented the high abundance, diversity, biomass and productivity of macro invertebrate assemblages within seagrass compared to adjacent unvegetated habitats (e.g. Orth, 1973; Edgar et al., 1994; Heck et al., 1995; Bostrom & Bonsdorff, 1997; Edgar & Barrett, 2002; Polte et al., 2005; see reviews by Orth et al., 1984, Hemminga & Duarte, 2000; Gillanders, 2006). Faunal abundance and diversity has also been shown to positively co-vary with seagrass biomass (i.e. density and blade length), (e.g. Stoner, 1980; Summerson & Peterson, 1984; Sogard et al., 1987; Lubbers et al., 1990; Edgar et al., 1994a; Edgar & Shaw, 1995c; Heck et al., 1995; Boström & Bonsdorff, 1997; Connolly, 1997; Mattila et al., 1999; Edgar & Barret 2002). This has been attributed to increased resource availability (Edgar, 1990b; Connolly, 1997), reduced competition and refuge from hydrodynamic forces (Murphey & Fonseca, 1995; Bostrom & Mattila, 1999), and predation (Heck & Thoman, 1981; Orth et al., 1984; Stunz & Minello, 2001). However, prior emphasis on the role of vegetation protecting macrofauna from predation is unclear due to the ability of many highly mobile macrofaunal species (e.g. amphipods) to actively select more dense/complex seagrass habitat (Stoner, 1980; Leber, 1985; Bell & Westoby, 1986a; Howard et al., 1989; Edgar, 1990b).

In addition, if both prey and predator densities co-vary with increasing habitat complexity then each should counteract the other. Recent experimental work by Scheinin et al (2011) revealed when animal densities were allowed to co-vary with vegetation density; vegetation had no impact on prey survival. They concluded that "increased food resources and reduced competition within vegetation may promote prey and thereby also predator abundance to a greater extent than previously thought".

Results from this survey are consistent, with generally higher densities of macrofauna recorded from subtidal seagrass, particularly east coast sites with concurrent longer blade lengths, increased density and continuous coverage (e.g. Rangaunu). In contrast, densities declined 3.5 fold for west coast subtidal sites (Kaipara Harbour, Fig. 4.13). This may be a reflection of the more sparse and 'patchy' (20-50m patch size), seagrass coverage associated with wave exposure within this harbour in addition to higher levels of sedimentation and associated turbidities due to catchment disturbance (Haggitt et al., 2008; Gibbs et al., 2012; M. Morrison pers. comm.). Wave depth, exposure and current speeds have all been shown to be negatively correlated with percentage seagrass cover (Foncesca & Bell, 1998; Turner et al., 1999b, 1999b; Polte et al., 2005; Turner & Schwarz, 2006a).

Overall, densities for sandy substrata were appreciably lower than for subtidal seagrass (five fold decline), (Fig. 4.13). Whilst, intertidal seagrass sites generally recorded lower densities than for subtidal (two fold decline). However, some west coast sites (i.e. Kaipara, Kawhia); with lower seagrass biomass (sparse/patchy coverage) had comparable densities to sandy habitats, particularly when considering all size groups (i.e. 1-22mm; Appendix 4.1). This included elevated numbers of larger bivalves and concurs with van Houte-Hewes et al., (2004) comparative study of macrofauna in the Coromandel. An exception was Tairua which recorded very high numbers of the juvenile amphipod *Aora typica.* Diversity showed similar trends averaging 31.3, 14.5 and 11 species per core for subtidal, intertidal seagrass and sand respectively. However, division between intertidal vegetated and sand sites was more pronounced for biomass and productivity estimates (Fig. 4.14).

Lower faunal density/biomass/productivity observed for intertidal relative to subtidal seagrass supports prior research conducted in northern New Zealand (Ellis et al., 2004; van Houte-Howes et al., 2004; Alfaro; 2006; Schwarz et al., 2006; Mills & Berkenbusch, 2009; Morrison unpubl. data; Table 4.22). Analogous results were found in 48 Tasmanian estuaries, whereby density/diversity declined over three and five fold ranges down the shore from high water mark to the shallow sub littoral (Edgar & Barret, 2002). This may be a result of the large fluctuations in environmental conditions (i.e. periodic desiccation and fluctuating temperatures), experienced by intertidal habitats, resulting in stunted growth (shorter blade lengths), and lower overall diversity and productivity (Jackson, 1972 cited in Edgar 1990b; Schwarz et al., 2006; Polte & Asmus 2006; Hemminga & Duarte, 2000; but see Edgar 1990b). In contrast, subtidal habitats are more environmentally benign and stable (i.e. reduce effects of currents/waves; provide shelter from predation; support larval settlement), and are characterized by more complex structure, with higher density and longer stems (Edgar et al., 1994; Schwarz et al., 2006; Hindell et al., 2000a), providing up to 20 times more surface area for epifaunal animals to graze (Pollard, 1984).

Benthic macrofauna composition:

Sandflat sites were dominated by a higher percentage (~40%) of infaunal species which included Spionid polychaetes, in addition to bivalves (viz. *Austrovenus stutchburyi* & *Nucula hartvigiana*). Bivalves also dominated intertidal seagrass meadows, particularly for the lower Rangaunu sites (Fig. 4.13). Crustaceans, viz gammarid amphipods, and to a lesser extent, caridean shrimps and crabs (*Halicarcinus* sp.; *H. crassa*) dominated the epifaunal assemblages. Overall, modest numbers of amphipods were collected (with the exception of Tairua) from intertidal sites. By contrast, subtidal seagrass, particularly northeastern sites (Rangaunu; BOI) were characterized by a highly diverse range of gastropods (24 sp.) while Rangaunu sites were dominated by tubiculous polychaetes and bivalves (*viz. N. hartvigiana*). Amphipod densities were considerably higher than for intertidal seagrass (with the exception of Tairua).

In the present study, seagrass supported higher epifaunal densities, particularly of amphipods. Given the dominance of amphipods in the diets of seagrass associated fish (with the exception of Tairua), densities were modest, (Fig. 4.13). This may be due to seasonal availability of prey, with amphipod densities generally declining in late summer/early autumn (e.g. Choat & Kingett, 1982; Alfaro, 2006; Edgar, 1990 b,c; Edgar & Aoki, 1993; Duffy & Hay, 2000; Akin & Winemiller, 2006). Epifaunal assemblages may also be concurrently shaped by the influx of high numbers of juvenile fish and subsequent predation rates altering density and size frequency distributions of the epifaunal community (seagrass: Edgar & Aoki, 1993; Edgar & Shaw, 1995a; Motta et al., 1995 and references therein; macroalgae: Duffy & Hay, 2000). However, evidence is equivocal due to highly mobile epifauna (see Kingett & Choat, 1981; Bennet & Branch, 1990).

A survey by Edgar & Shaw (1995c) in Western Port, Australia estimated that most of the production of macrobenthic crustaceans (>1mm) were cropped by fish, which suggests that competition for resources may be intense (Sanchez-Jerez et al., 2002). Declining numbers and emigration of juvenile fish recorded from seagrass meadows from late autumn was linked to declining prey availability and concomitant decrease in seagrass biomass (Edgar & Shaw, 1995a). Similarly, winter minima for seagrass biomass have also been recorded in New Zealand (Turner & Schwarz, 2006 a,b) along with emigration of species such as snapper in early winter to more offshore habitats (M. Morrison pers. obs.). In contrast, results from this survey show the majority of epibenthic crustaceans consumed (viz. amphipods) by juvenile fish were between 0.5-0.75mm (63%), with only 37% greater than 1mm. This corresponds with results obtained for coralline turf flats at Leigh, with fish only consuming 15-38% of crustacean productivity >1mm (Taylor, R. 1998). Nonetheless, observations of 0+ snapper defending feeding territories against each other in coralline turf habitats suggests juvenile fish can and do competitively exclude con specifics from prime sites (Kingett & Choat, 1981; Yamaoka et al., 1991, cited in Francis, 1995).

Epifaunal size also varied with habitat, with a higher proportion of amphipods >1.4mm collected from subtidal seagrass (~ 41%) than for intertidal (17%) or sand (19%) (Table 4.24). This trend was also reflected in the dietary prey size (see diet section below). Similarly, infauna (viz. polychaetes), although recording comparable densities across most habitats showed elevated biomass from subtidal seagrass sites indicating larger size classes present than for intertidal/sand habitats Fig. 4.14). Higher size dominance values would be expected given infauna is virtually inaccessible, particularly under dense seagrass to fish and decapod predators (Edgar, 1994).

Diet

Grazing amphipods are ubiquitous in marine benthic communities and are often the dominant primary consumers within seagrass meadows (Duffy & Hay, 2000, Cowles et al., 2009). They have been recognized (along with other epifaunal crustaceans e.g. mysids) as critical players in near shore trophic transfer due to their small size, high abundance, short generation times and high rates of secondary production (Edgar & Aoki, 1993; Motta et al., 1995; Taylor, R. 1998; Duffy & Hay, 2000; Jenkins et al., 2011). Amphipods are the major dietary component of a number of inshore fish species documented within New Zealand (Choat & Kingett, 1982; Russell, 1983; Taylor, R. 1998), Australia (Edgar, 1990b and references therein) and France (Selleslagh et al., 2011).

Results of this survey from seagrass meadows revealed epifaunal crustaceans (viz gammaridean amphipods) were the major food source for fish at all sites (Fig. 4.21), followed by mysids, decapods and plankton, and concurs with prior studies (e.g. Pollard, 1984; Edgar & Shaw, 1995a,c; Horonouchi & Sano, 2000; Yamada et al., 2010; see reviews by Bell & Pollard, 1989; Gillanders, 2006; Jenkins et al., 2011; Nunn et al., 2011). Infaunal benthos was of much lesser importance, with species such as spotty, sand flounder and triplefin consuming modest numbers of bivalves/siphons over intertidal seagrass. Additionally, polychaetes and cumaceans were consumed by gobies, triplefins and flounder over sandy habitats. No species were exclusively piscivorous as recorded in Port Phillip Bay, Australia (Hindell et al., 2000b). However, sites were not located near reefs (apart from BOI) where larger predators are more common (Bell & Pollard, 1989). Seagrass itself was hardly utilized, being consumed by only one species (garfish). Diets broadly overlapped at the level of prey species, with common species of amphipods (e.g. *Aora* sp., *Paradexamine* sp.) consumed by numerous fish species. This is consistent with findings from Edgar & Shaw's (1995a) survey at Western Port, Victoria.

Variation of diet within seagrass meadows

Prey items varied in composition/size with tidal position within seagrass meadows. Zooplankton (viz. *E. acutifrons*) numerically dominated the diets of fish caught from intertidal sites for both Kaipara and Rangaunu Harbours, whilst amphipods were the preferred prey from Kaipara (89%) and Rangaunu (63%) subtidal meadows. Amphipods consumed within subtidal seagrass were larger (i.e. 50% >1mm) than those ingested over intertidal sites ~12% >1mm) and reflected benthic size frequencies. This was also reflected in sizes of fish caught within these habitats. For example, small bodied species such as exquisite goby and sand goby (25-30mm FL) dominated intertidal seagrass assemblages. In contrast, larger snapper (~50-70mm FL) predominated at subtidal seagrass sites, whilst newly settled snapper (i.e. 20mm) primarily consumed plankton from the lower Rangaunu Harbour (near entrance) intertidal seagrass sites. The dominance of planktonic prey suggests that post larval snapper are initially using seagrass primarily as a refuge. Indeed snapper were observed emerging from the seagrass, holding position in the currents and foraging on plankton entrained in the passing water (pers. obs.).

In sum, northern New Zealand seagrass, particularly subtidal meadows, supported a relatively diverse and abundant juvenile fish assemblage, including high numbers of several species that are commercially important (e.g. snapper, trevally) and is supportive of the paradigm that seagrasses provide an enhanced nursery habitat for this region. Results suggest a close association between the abundance of fish and productivity of macro invertebrates, particularly crustaceans which comprised the major dietary item which also co varied with seagrass biomass (blade length/density), (Stoner, 1980). These results support associated studies both in New Zealand (Schwarz et al., 2006; Morrison et al., in review), and Australia (Marais, 1984; Connolly, 1994; Edgar & Shaw, 1995b,c; Edgar, 1999; Bloomfield & Gillanders, 2005).

Geographical setting: are all seagrass meadows equal?

With seagrass landscape attributes such as bed fragmentation, continuity of cover, size and shape along with structure of the plants themselves (i.e. biomass, density, blade length) displaying strong relationships to the physical setting of an area (Turner et al., 1999b and references therein; Connolly & Hindell, 2006; Jelbart et al., 2007; Mills & Berkenbusch, 2009; Morrison et al., 2012 in review), the overall coastal differences in abundance, evident in this study may reflect climatic and geological differences at the landscape level. West coast estuaries are more exposed to wind and waves, and soils are generally softer and more erodible, resulting in higher silt/clay loadings and concomitant elevated turbidities within estuaries, leading to less optimal growing conditions for seagrass (Vant, pers. comm.; Gibbs et al., 2012; Morrison et al., in prep.). Conversely, east coast estuaries tend to be more sheltered. Geology tends to be comprised more of volcanic rock, less susceptible to erosion with resultant lower turbidities and fine sands predominating (Vant, pers. comm.; Morrison et al., in prep.), providing more benign conditions for seagrass growth (particularly at depth) and fish recruitment. However, given the paucity of sheltered areas along the exposed west coast, the value of subtidal seagrass would be disproportionately greater for species such as snapper and trevally. For example, Kaipara harbour, with 432 km² of subtidal area may provide the majority (~80%) of recruits for the coastal snapper stock (Morrison et al., in revision). Thus, deterioration in estuaries within this region (i.e. increased turbidity/sedimentation) may have far greater impacts on levels of snapper recruitment in to coastal fisheries.

A New Zealand wide survey of seagrass meadows (69 estuaries) revealed latitudinal variation, with overall declining densities of fish from the North to South Island (Morrison et al., 2007; in review; unpub data). Snapper, trevally, grey mullet and parore were largely absent as juveniles from seagrass meadows south of Cook Strait, whilst spotties were still recorded in high abundances south to Farewell Spit. These species were subsequently supplanted by leatherjackets and pipe fish species extending into southland. These results collectively demonstrate that the accepted paradigm of seagrass meadows providing important juvenile finfish nurseries varies for individual species across latitudinal and coastal scales and with tidal position within estuaries for New Zealand. Similarly, overseas reviews note that given the unique characteristics of each estuary, universal generalizations are difficult (Hemminga & Duarte, 2000; Heck et al., 2003; Heck & Orth, 2006; Horonouchi, 2007).

Exclusivity of seagrass as nursery areas

Whilst the ‘nursery function’ of seagrass is a widely accepted paradigm. It is largely derived from extensive studies detailing higher densities of juvenile fish and invertebrates compared to adjacent unvegetated habitat (see reviews by Heck et al., 2003; Hemminga & Duarte, 2000; Gillanders, 2006). Few studies have compared the potential range of habitats available or possible linkages through ontogenetic movement (but see Parrish, 1989; Gillanders & Kingsford, 1996; Bloomfield & Gillanders, 2005). A review by Heck et al., (2003) challenged the exclusivity of seagrass beds as nursery areas finding that although density, growth and survival were greater in seagrass than non-vegetated habitats, there were few significant differences when seagrass was compared to other structured habitats (oyster & cobble reefs, macroalgal beds, mangroves) and that structure per se, rather than type of structure may determine nursery habitat value.

The review found densities of fish in northern hemisphere seagrass (viz North America) were greater for 75% of surveys, compared to 36% for southern hemisphere (viz Australia). However, this data set covered a wide range of different seagrass species, with many (45) of the comparisons coming from a single paper reporting visual fish censuses from tropical mangrove/seagrass systems of the Caribbean (Nagelkerken et al., 2000b) which are notably different to temperate New Zealand environs. Additionally, visual counts are inappropriate for identifying small juvenile fish (Bloomfield & Gillanders, 2005) and also contrasts to standard sampling techniques which include seining or trawling. The use of varying sampling techniques reflects the inherent difficulties of utilizing appropriate sampling methods between multiple habitats, making generality at large biogeographic scales difficult (Gillanders, 2006). Similarly, in this survey, fyke nets utilized for fish collection within mangroves do not provide estimates of density per unit area to allow direct comparison with beach seine trawls utilized over mud/sand and seagrass habitats.

4.5.4 Conclusion

In sum, while overall fish abundances in this survey were relatively high compared to other shallow coastal habitats, species diversity was comparatively modest reflecting New Zealand’s isolation from continental systems and temperate climate (Francis et al., 2011). Densities were comparable between subtidal seagrass and intertidal mudflats, followed by intertidal seagrass and sand habitats.

Whilst numbers within mangrove forests were lower, considering the fyke nets fished for 6 hours (apart from two high catches of yellow-eyed and grey mullet from Mahurangi & Manukau Harbours). This survey found different habitats supported different species assemblages of fish and invertebrates which varied with latitude, geographical setting (east/west), between and within estuaries (tidal position). A small number of species (e.g. yellow-eyed mullet; exquisite goby) had more ubiquitous distributions. Foraging strategies and the dominant prey items of fish assemblages varied with habitat. For example, fish assemblages within seagrass predominantly consumed hyperbenthic epifaunal crustaceans, whilst zooplankton dominated the diets of fish within mangroves and sessile, benthic prey items were primarily consumed over intertidal mudflats. In contrast to tropical and subtropical studies (see review of Beck et al., 2003), benthic faunal richness and abundance were most often greatest in seagrass habitat (particularly subtidal), as was macro invertebrate biomass and estimated production followed by sand, mangroves and mud habitats within northern, North Island estuaries. Results support increasing evidence that usage of different habitats by juvenile fish is dependent upon environmental context, and that gross physical attributes of habitats may not always be of predictive value in fisheries ecology (Jenkins et al., 2011 and references therein).

However, while the spatial extent of this study was broad, encompassing latitudinal, coastal and estuarine differences, sampling was only undertaken on one occasion. Due to the large scale geographic nature of the study and field work logistics, data was collected during different time periods, albeit over the same season (February-April) during high juvenile fish densities. It therefore represents a preliminary ‘snapshot’ of the dietary preferences of the fish species and distribution over late summer. Nonetheless, earlier temperate studies have indicated that assemblage structure of both fish and benthos is mainly governed by seasonal dynamics, while inter annual variation is low (Stål et al., 2007; Hailes & Hewitt et al., 2012; Morrison, unpubl. data). Thus, given the strong seasonal changes in diet documented for temperate ecosystems (Layman & Silliman, 2002; Akin & Winemiller, 2006; see Chapter Two; Fig. 2.23) and associated benthos (Taylor, R. 1998; Choat & Kingett, 1982) future research needs to include temporal variation in food web analyses. It is acknowledged that factors influencing prey selectivity (i.e. capture efficiency, handling time, digestion rate of prey items), were not accounted for. Additionally, core sampling revealed low capture rates for mysids, a dominant prey item. Nonetheless, results offer a preliminary insight into feeding strategies of different fish species, keystone prey species and their distribution within the four habitats.

Ultimately, understanding of the relative contribution of the different habitats to recruitment offshore will require information on spatio temporal variability in ontogenetic habitat use including not only density estimates, but growth and survival rates during juvenile habitat utilization (Beck et al., 2003; Fodrie et al., 2009; Nunn et al., 2011) and subsequent emigration. Further research utilizing stable isotope analysis, in combination with traditional dietary analyses and estimates of prey availability (as undertaken in this survey) along with nutritional condition indices (using RNA-DNA ratio analysis; Nunn et al., 2011) and otolith microchemistry (e.g. Gillanders, 2003; Morrison et al., 2012 in review), may allow better identification of key habitats of all life stages/species. As noted by Nunn et al., (2011), given that recruitment into adult fish stocks are, directly, or indirectly, limited by the quality and quantity of habitat and food available to larval and juvenile fishes, such information is a fundamental pre requisite for fisheries management and ecosystem based management (Hinz et al., 2005; Nunn et al., 2011; De Raedemaeker et al., 2011).

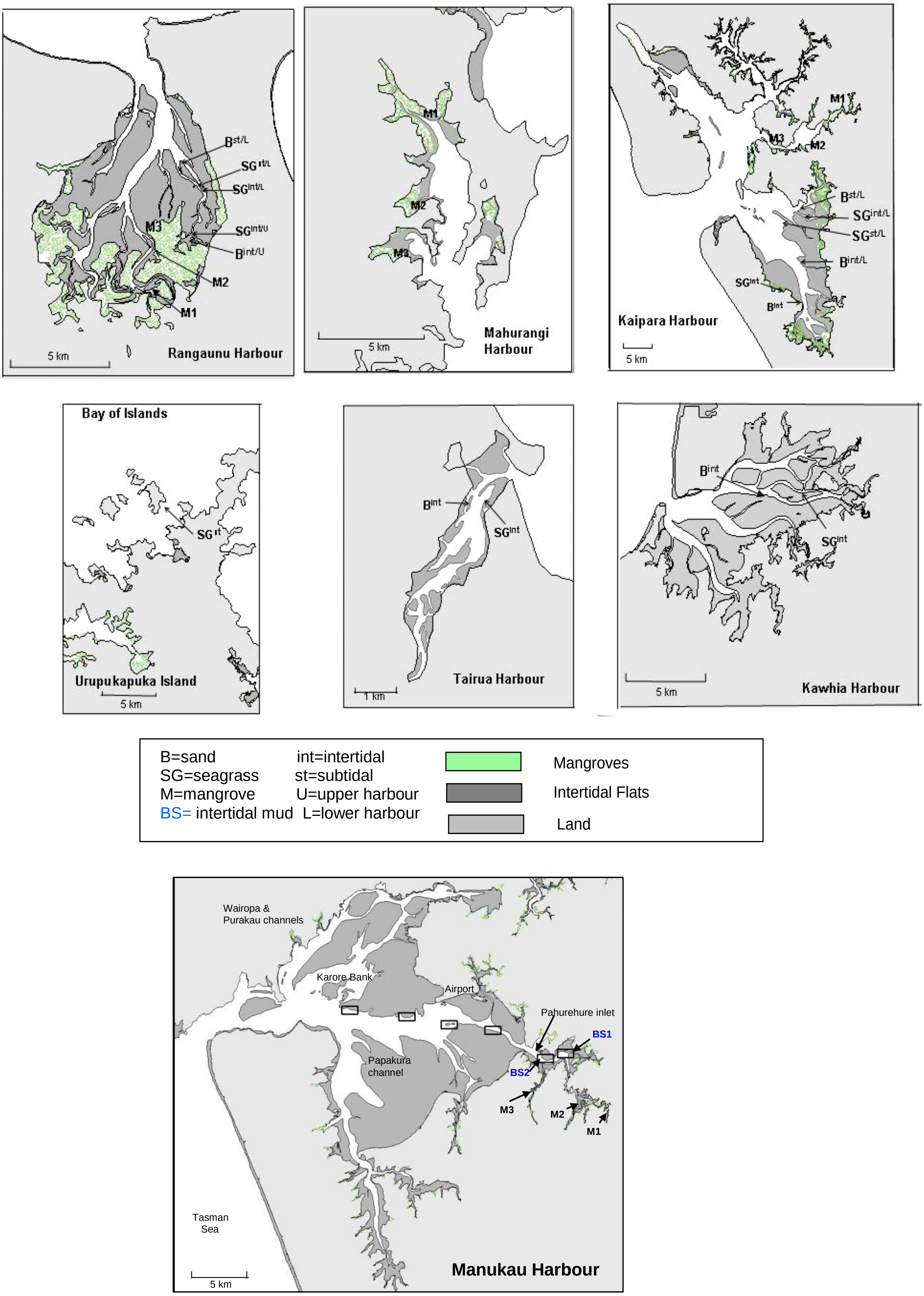


Figure 4.1 Map of study area showing the 7 sampling sites and habitat types surveyed in northern New Zealand.

Photo: C. Middleton

Photo: Biomarine

Figure 4.2 Photo showing the extensive intertidal seagrass meadow abutting mangrove forest within Rangaunu Harbour, with beach seine net in foreground (A), large scale mosaic of subtidal seagrass patches (each 20-50m in diameter) in the southern Kaipara Harbour (B).

Figure 4.3 Fyke net set flush against mangrove forest within Rangaunu Harbour (A), exposed fyke net at low tide (B), and an example of beach seine catch from seagrass habitat.

A. Subtidal seagrass continuous cover - lush

B. Medium density seagrass - sparse

C. Intertidal seagrass – medium density short

Subtidal seagrass meadow (darker shaded area) Urupukapuka. (Bay of Islands)

Figure 4.4 Examples of seagrass meadow characteristics used in the survey and the offshore island site sampled for subtidal seagrass Urupukapuka Island (Bay of Islands). Estimated percentage seagrass cover (A= >75%; B = 26%-50%; C = 51%-75%), as per cover scale in Appendix 1 of Schwarz et al (2006).

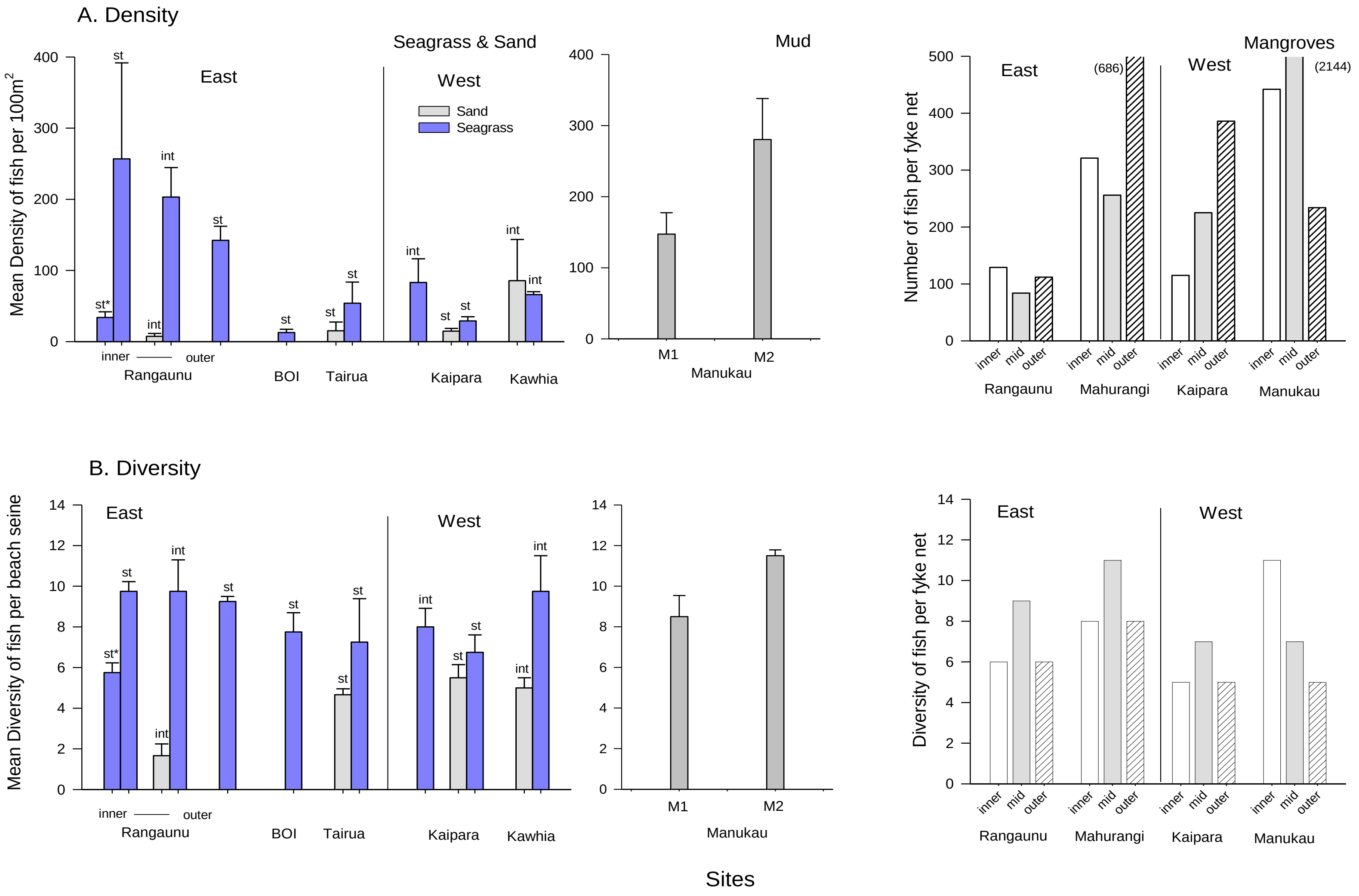


Figure 4.5 Mean density (±SE) (A), and diversity (B) of fish species caught by beach seine and fyke net from seagrass, sand, intertidal mud and mangrove habitats from 7 northern Harbours.

East=East coast; West=West coast; st*=subtidal seagrass site that comprised 40% sand & 60% seagrass; st=subtidal; int= intertidal; inner= inner harbour; mid=mid harbour; outer=outer harbour

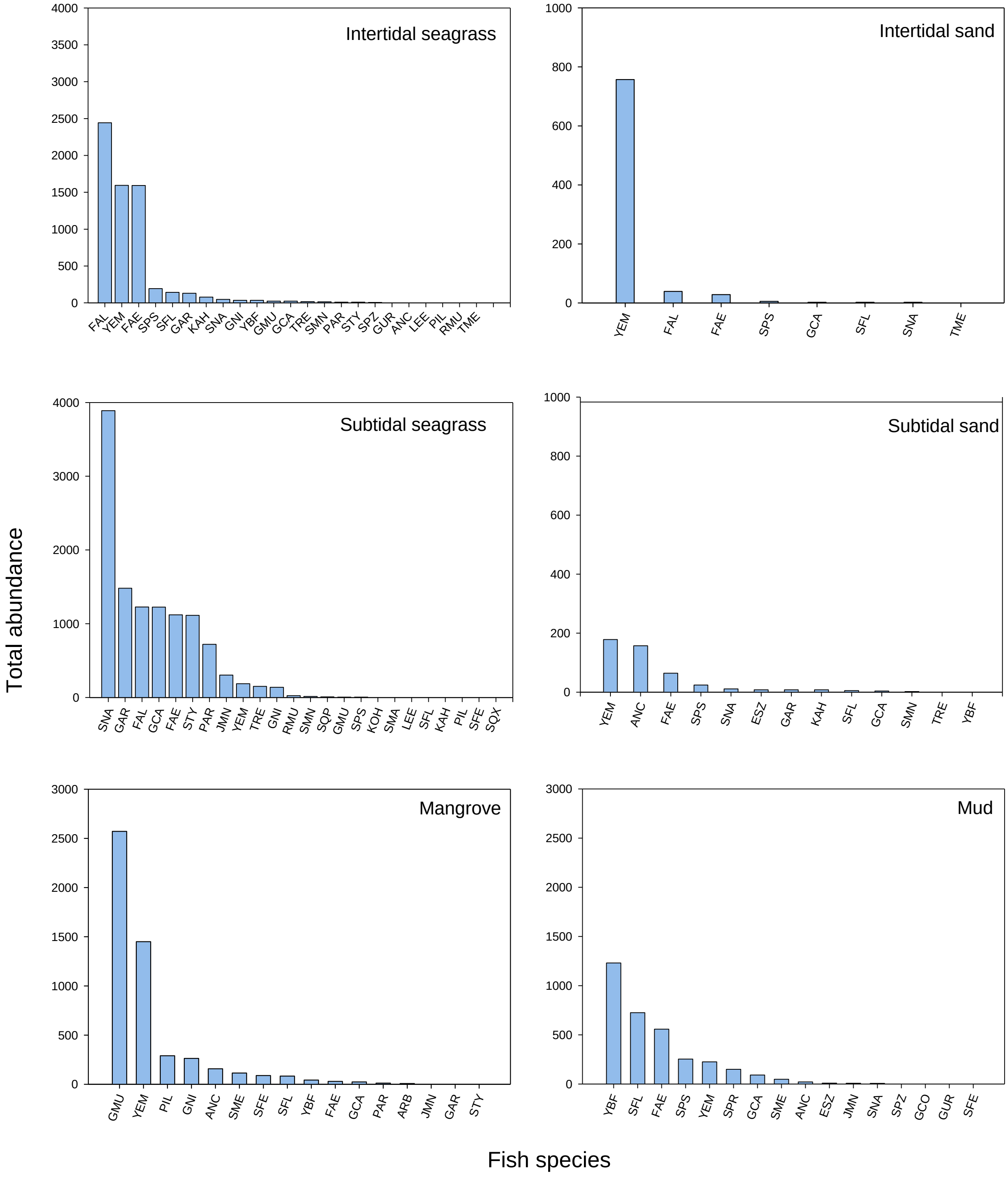


Figure 4.6 Total abundance of fish species caught by beach seine and fyke net from seagrass, sand, intertidal mud and mangrove habitats at 27 sites from 7 northern Harbours.

ANC: Anchovy
ARB: Bridled goby
ESZ: Estuarine stargazer
FAE: Exquisite goby
FAL: Sand goby
GAR: Garfish
GCA: Mottled triplefin
GCO: Common bully
GMU: Grey mullet
GNI: Estuarine triplefin
GUR: Gurnard
JMN: Jack mackerel
KAH: Kahawai
KOH: Koheru
LEE: Speckled pipefish
PAR: Parore
PIL: Pilchard
RMU: Red mullet
SFE: Short-finned eel
SFL: Sand flounder
SMA: Smooth pipefish
SME: Smelt
SMN: Black pipefish
SNA: Snapper
SPR: Sprats
SPS: Speckled sole
SPZ: Spotted stargazer
SQP: Sepiolid squid
SQX: Squid
STY: Spotty
TME: Striped clingfish
TRE: Trevally
YBF: Yellow-belly flounder
YEM: Yellow-eyed mullet

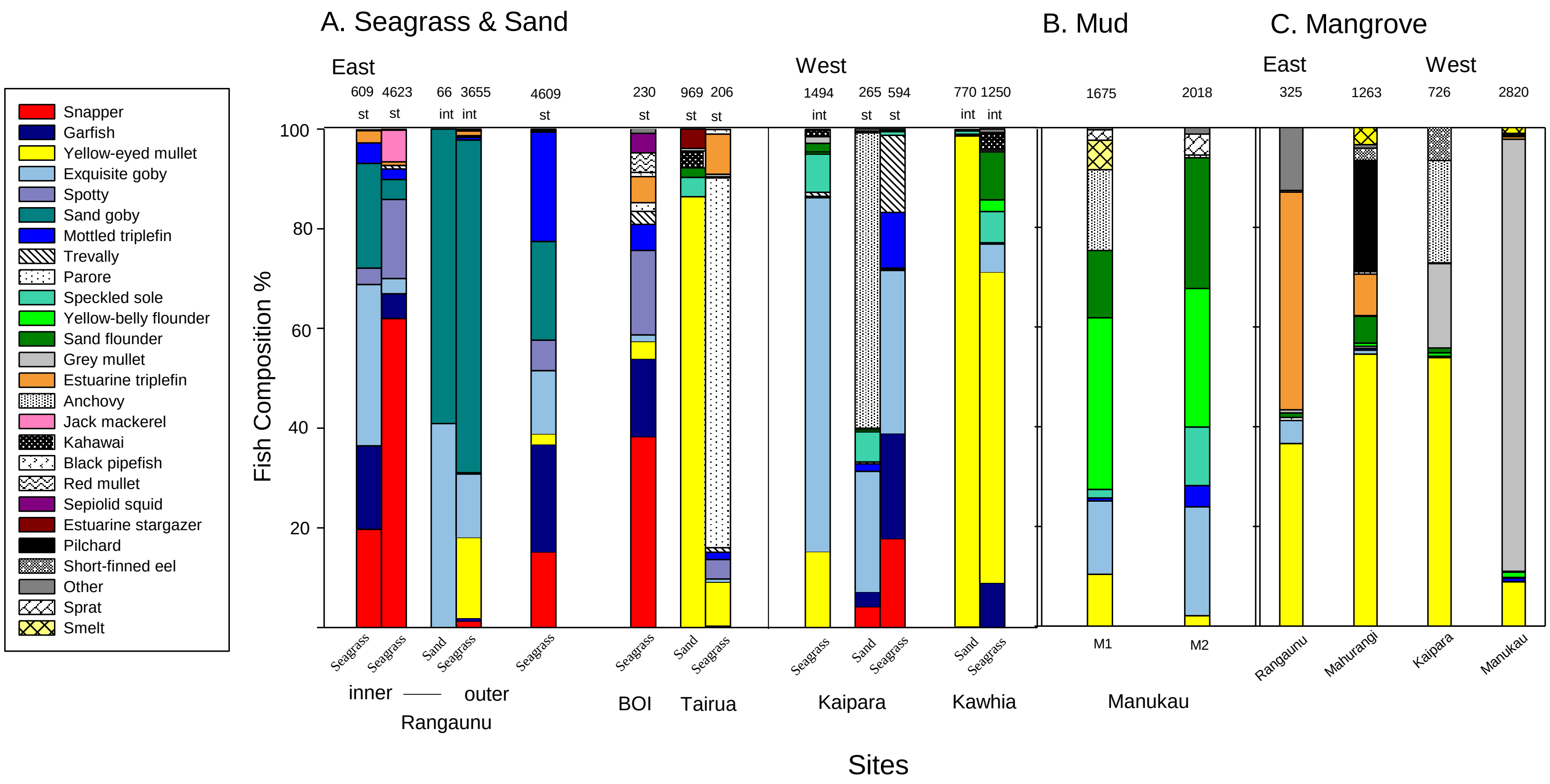


Figure 4.7 Composition and proportional abundance of the fish assemblage at each of the four habitats from 27 sites at seven northern Harbours. Site identifiers as per Fig. 4.5.

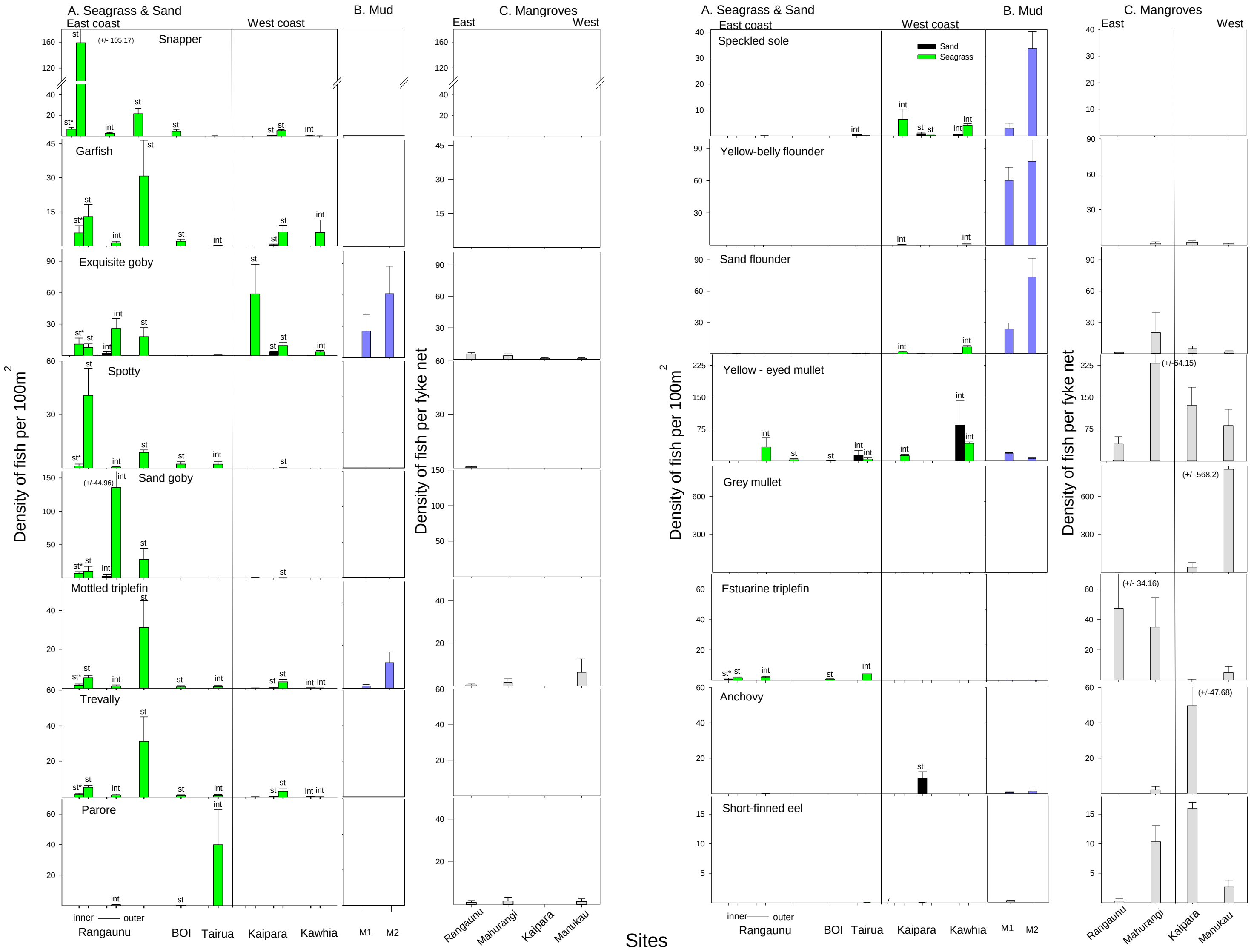


Figure 4.8 Mean density (±SE) of the sixteen most common fish species collected by beach seine and fyke net from sand, intertidal mud, seagrass and mangrove habitats from 7 northern harbours.

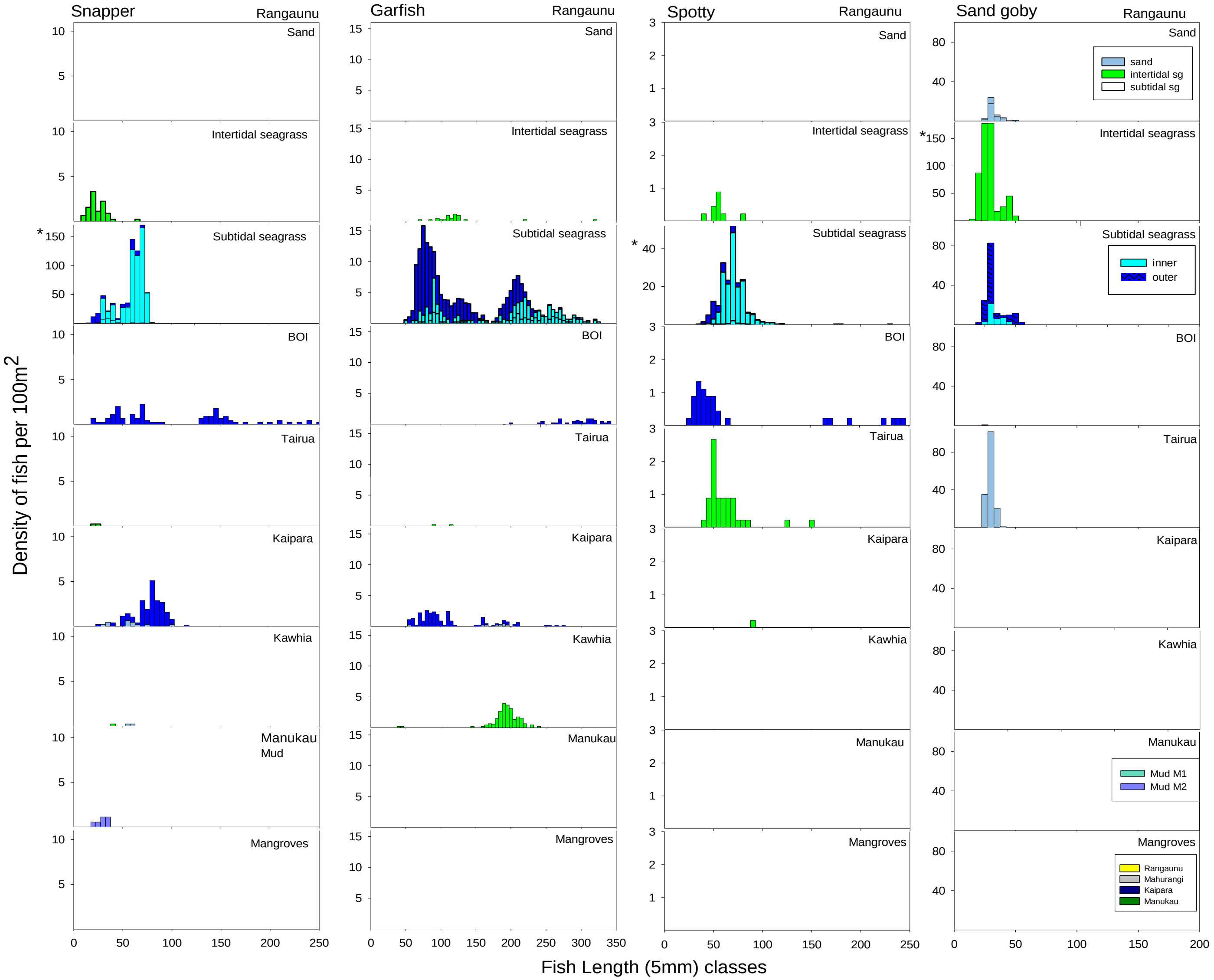


Figure 4.9 Length frequency of the four common fish species collected by beach seine and fyke net from sand, seagrass, intertidal mud, and mangrove habitats from 7 northern harbours.

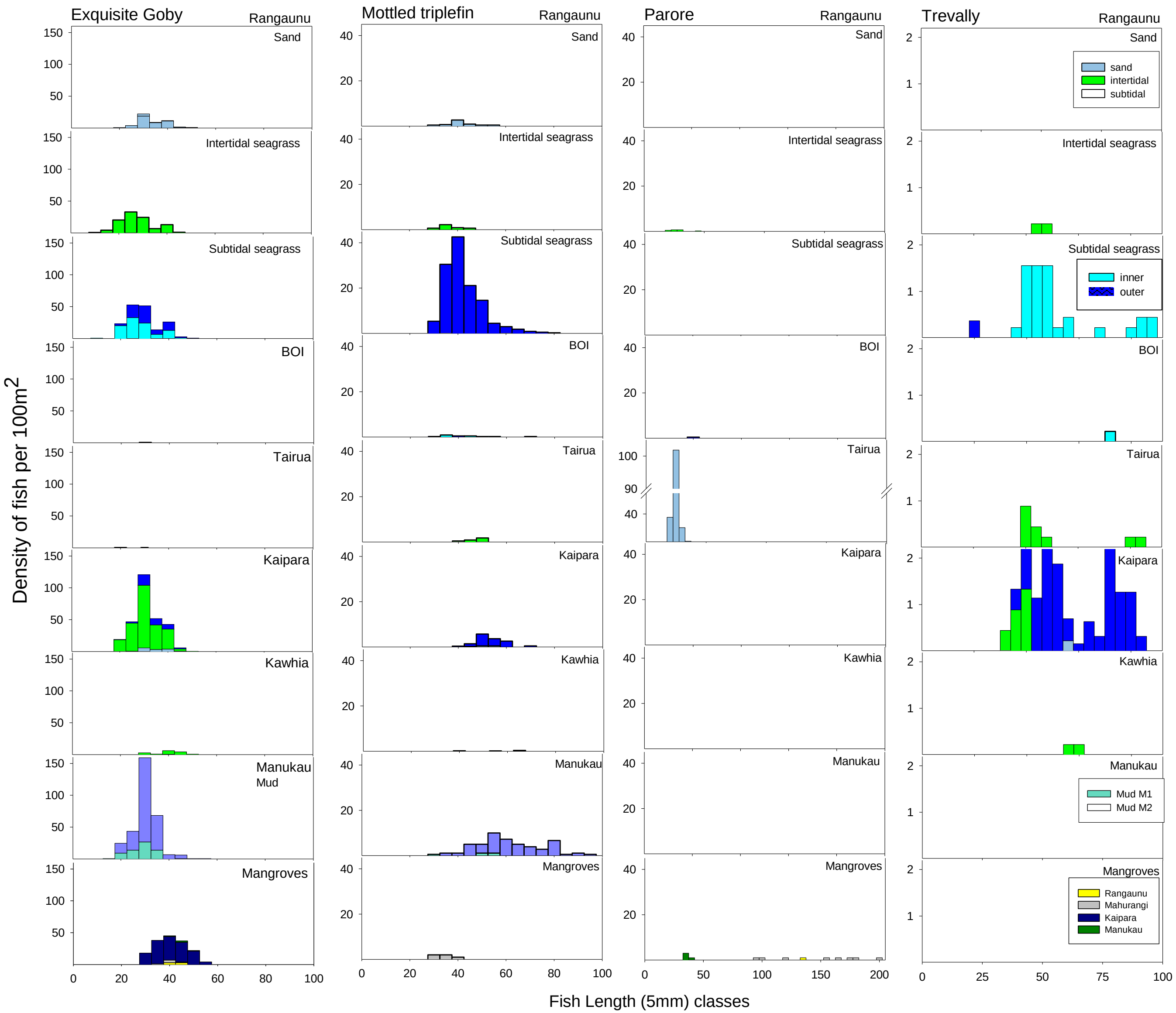


Figure 4.10 Length frequency of the four common fish species collected by beach seine and fyke net from sand, seagrass, intertidal mud and mangrove habitats from 7 northern harbours.

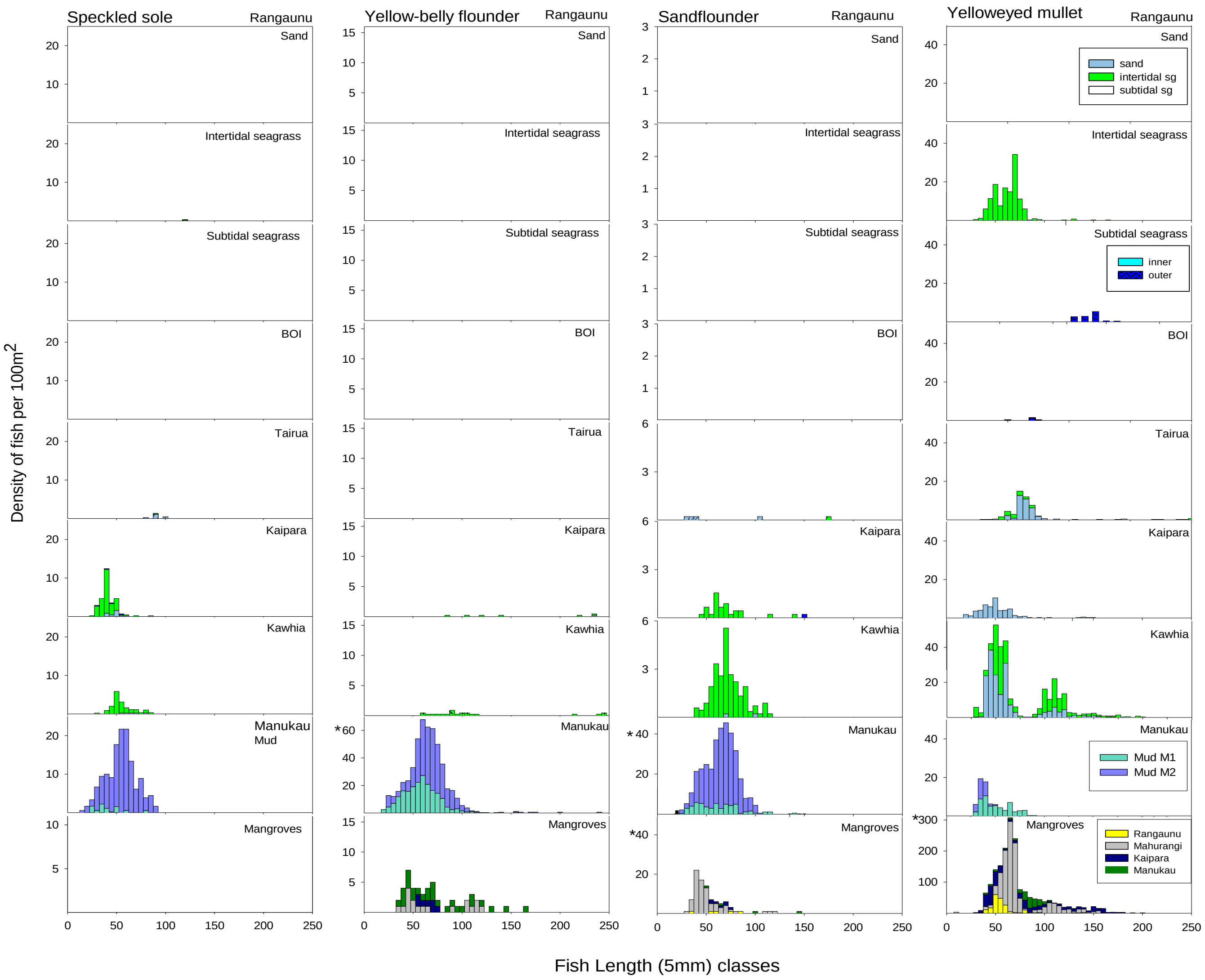


Figure 4.11 Length frequency of the four common fish species collected by beach seine and fyke net from sand, seagrass, intertidal mud and mangrove habitats from 7 northern harbours.

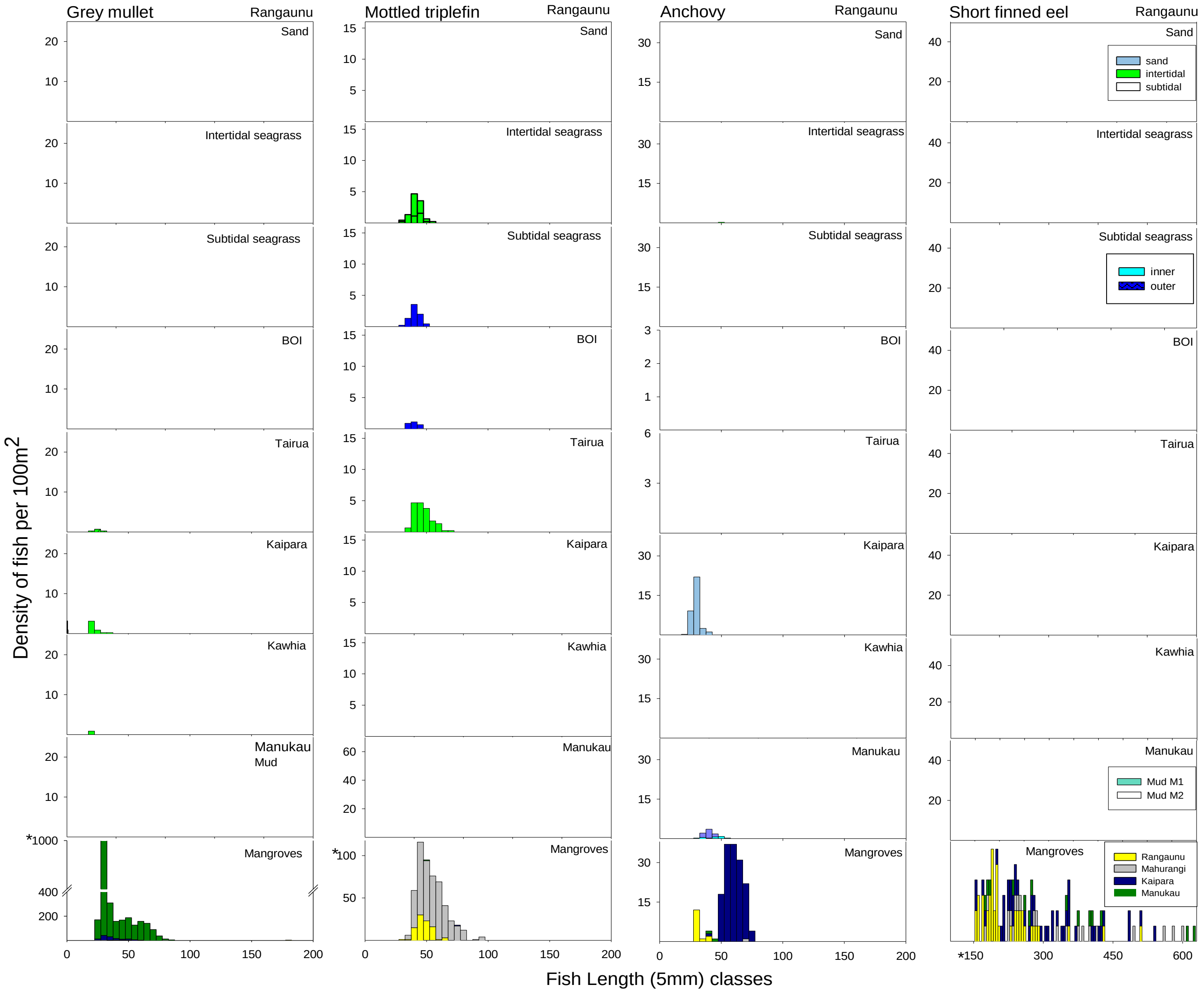


Figure 4.12 Length frequency of the four common fish species collected by beach seine and fyke net from sand, seagrass, intertidal mud and mangrove habitats from 7 northern harbours.

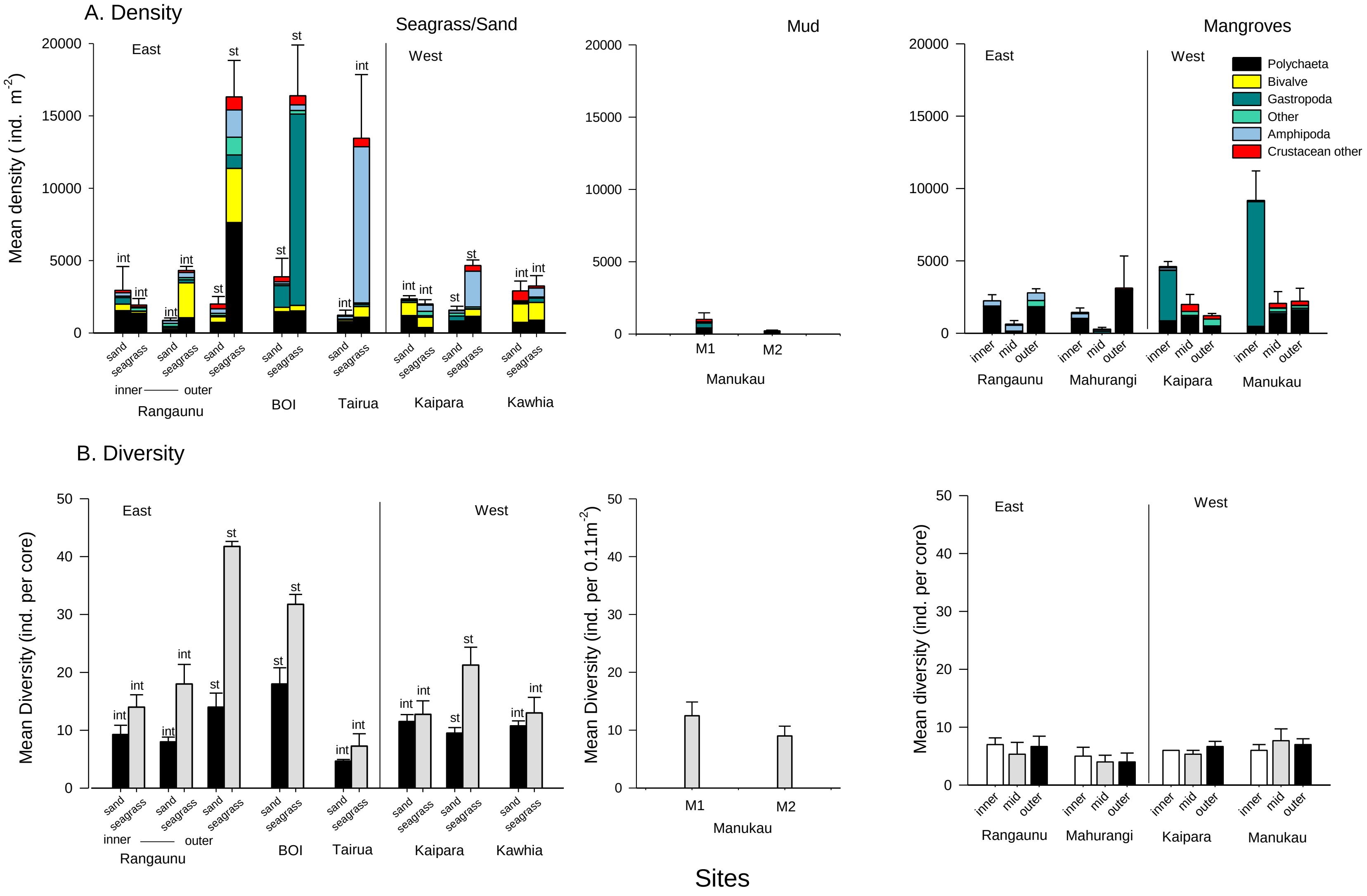


Figure 4.13 Mean density (±SE) (A), and diversity (B) of benthic invertebrates (1-5.6mm sieve size) collected from seagrass, sand, intertidal mud and mangrove habitats at 27 sites from 7 northern harbours. Site identifiers as per Figure 4.5.

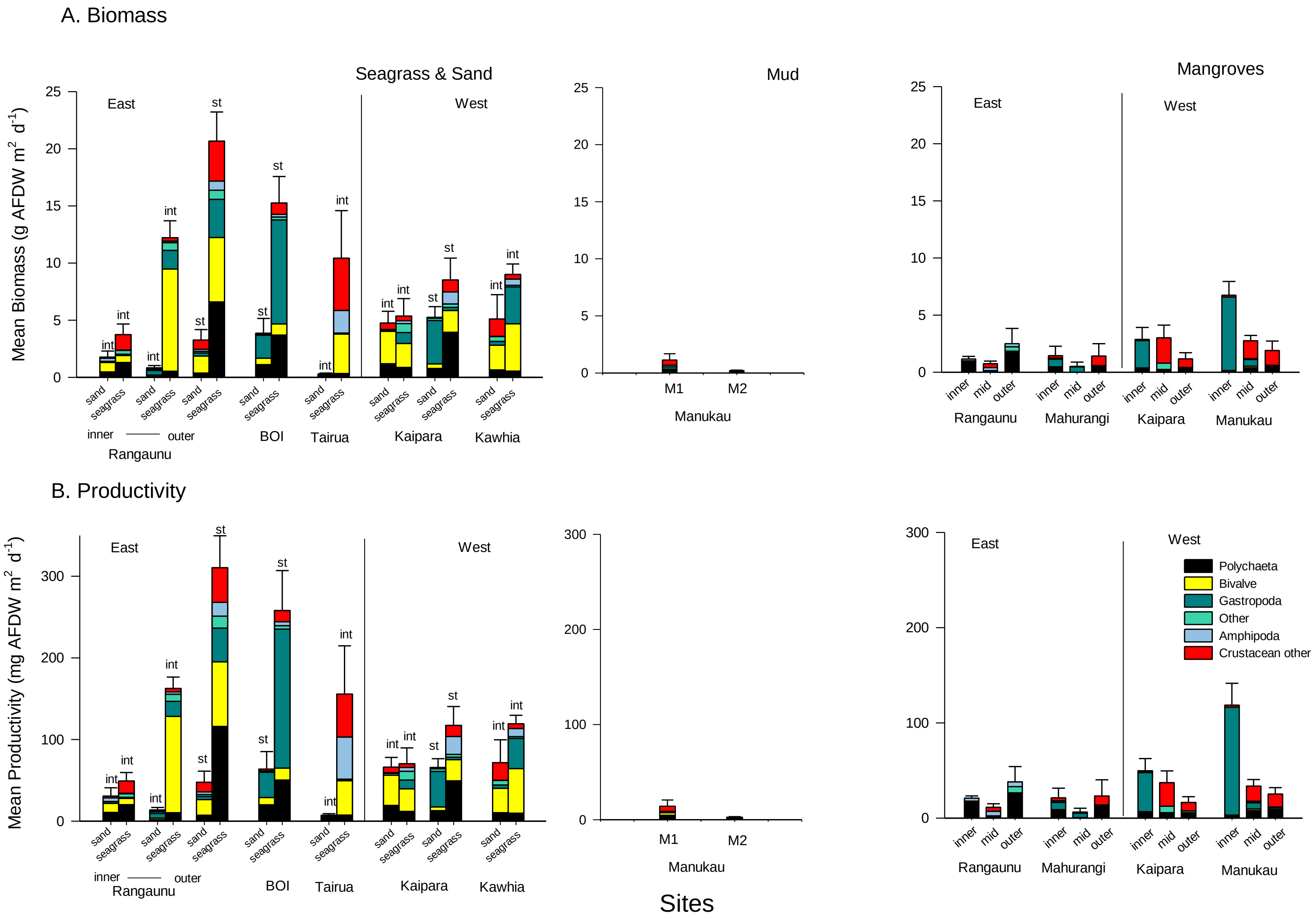


Figure 4.14 Mean biomass (±SE) (A), and estimated productivity (B) of benthic invertebrates (1-5.6mm sieve size) collected from seagrass, sand, intertidal mud and mangrove habitats at 27 sites from 7 northern harbours. Site identifiers as per Figure 4.5.

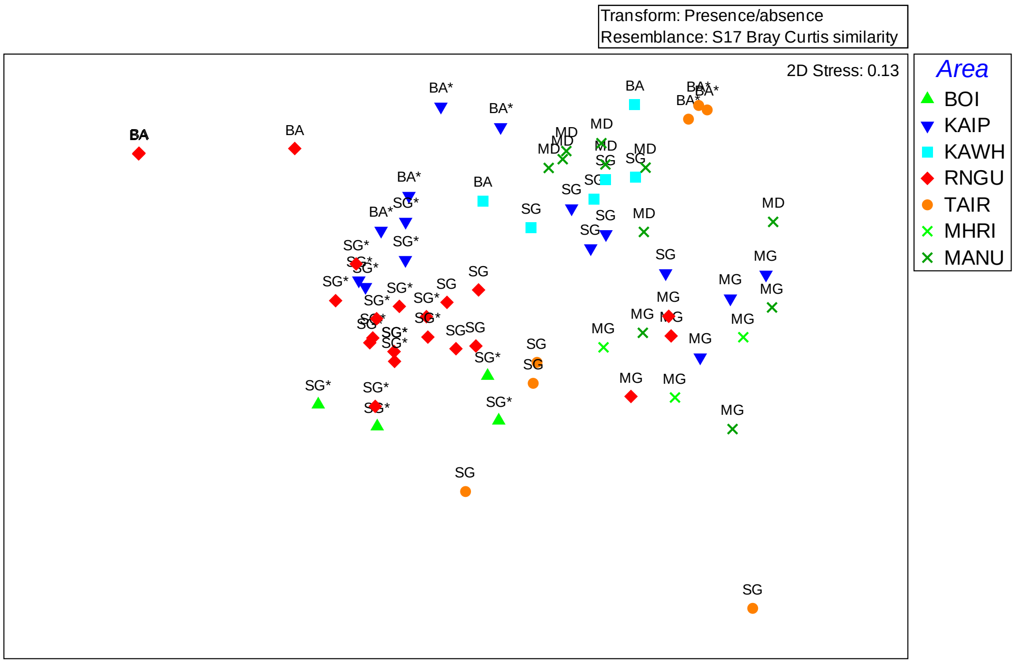


Figure 4.15 MDS ordination of fish assemblage data showing pattern of similarities between all harbours and habitats sampled. Data were presence/absence transformed.

Site identifiers: BOI=Urupukapuka Island; KAIP=Kaipara; KAWH=Kawhia; RNGU= Rangaunu; TAIR=Tairua; MHRI=Mahurangi; MANU=Manukau.
Habitat identifiers: BA=sand; SG=seagrass; MG=mangrove; MD=mud; *=subtidal sites; otherwise intertidal

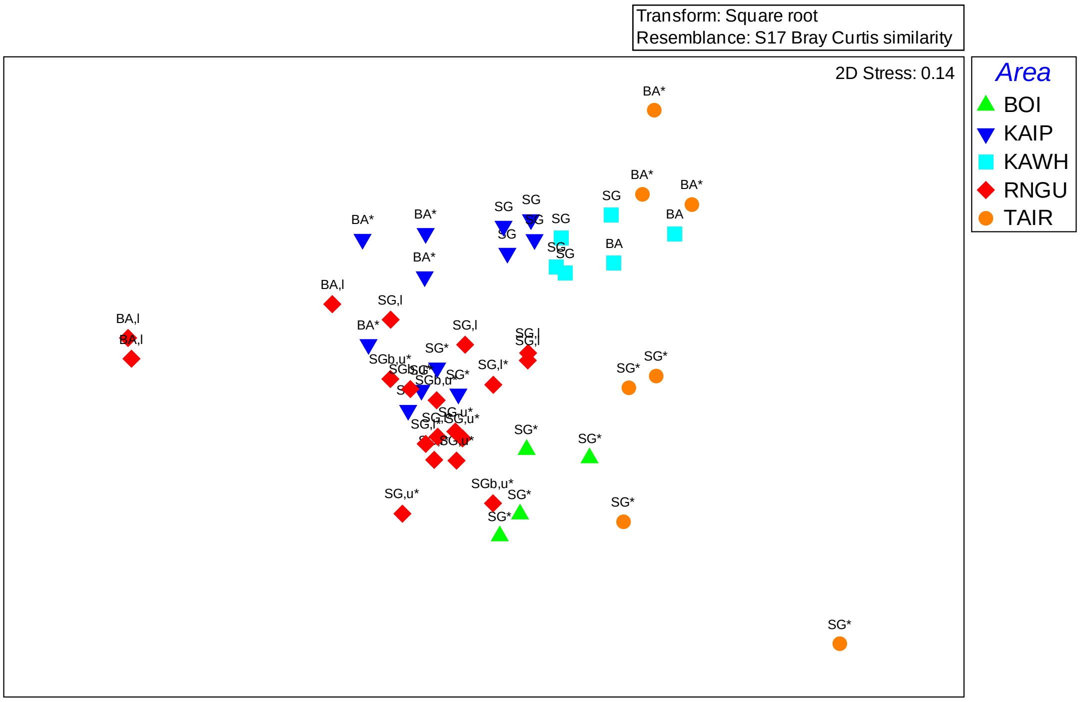


Figure 4.16 MDS ordination of seagrass and sand fish assemblage data showing pattern of similarities between harbour and habitat.

Site identifiers: BOI=Urupukapuka Island; KAIP=Kaipara; KAWH=Kawhia; RNGU= Rangaunu; TAIR=Tairua;
Habitat identifiers: BA=sand; SG=seagrass; MG=mangrove; MD=mud; *=subtidal sites; otherwise intertidal site; u=upper harbour, l=lower harbour.

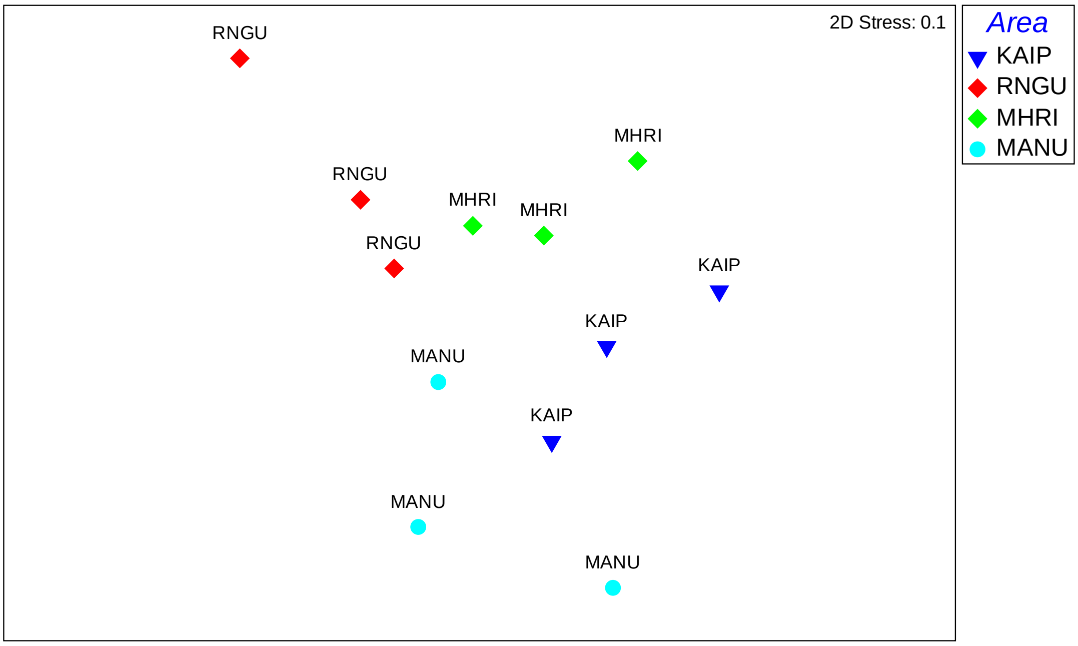


Figure 4.17 MDS ordination of mangrove fish assemblage data showing pattern of similarities between harbour and habitat.

Site identifiers: KAIP=Kaipara; RNGU= Rangaunu; MHRI= Mahurangi; MANU= Manukau

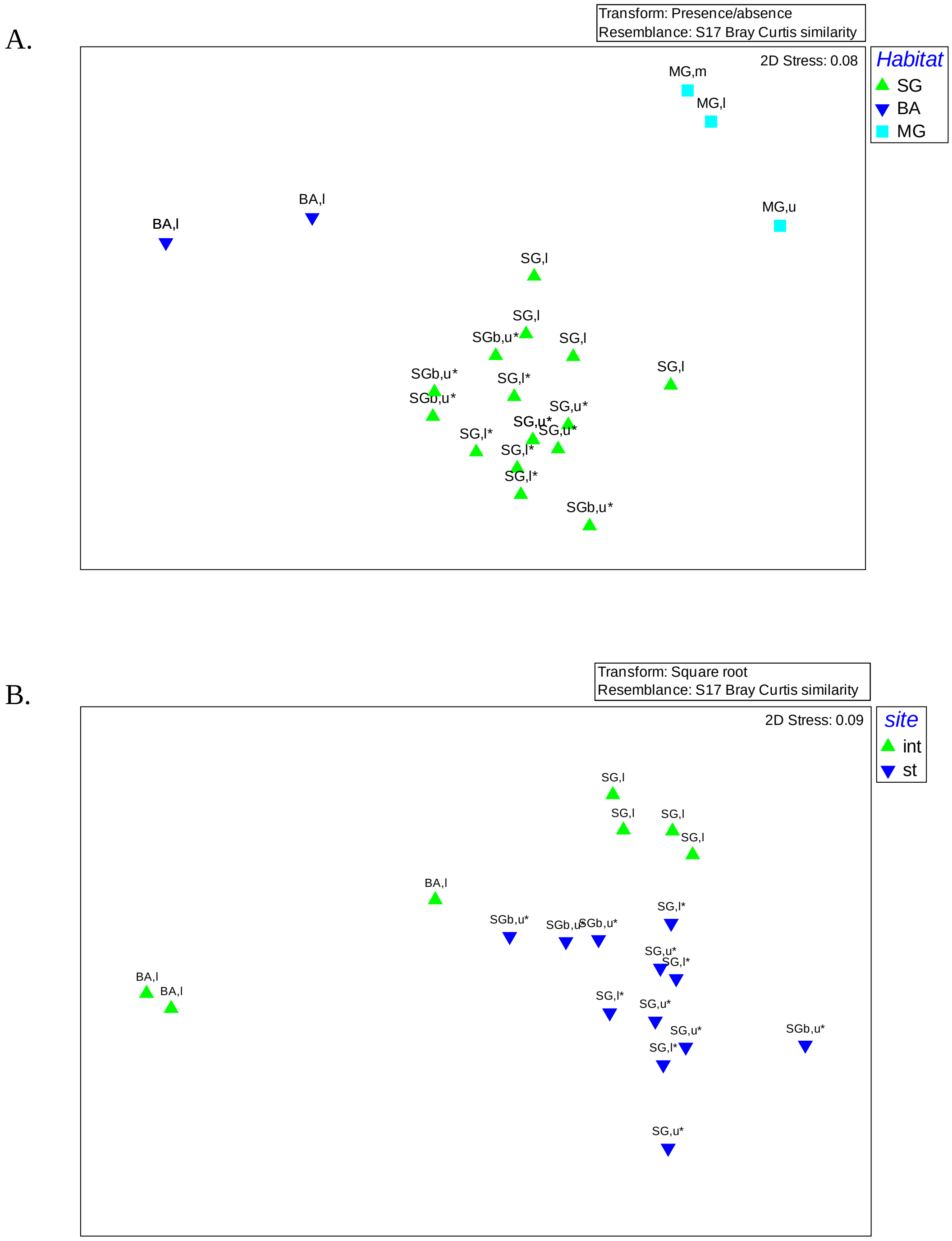
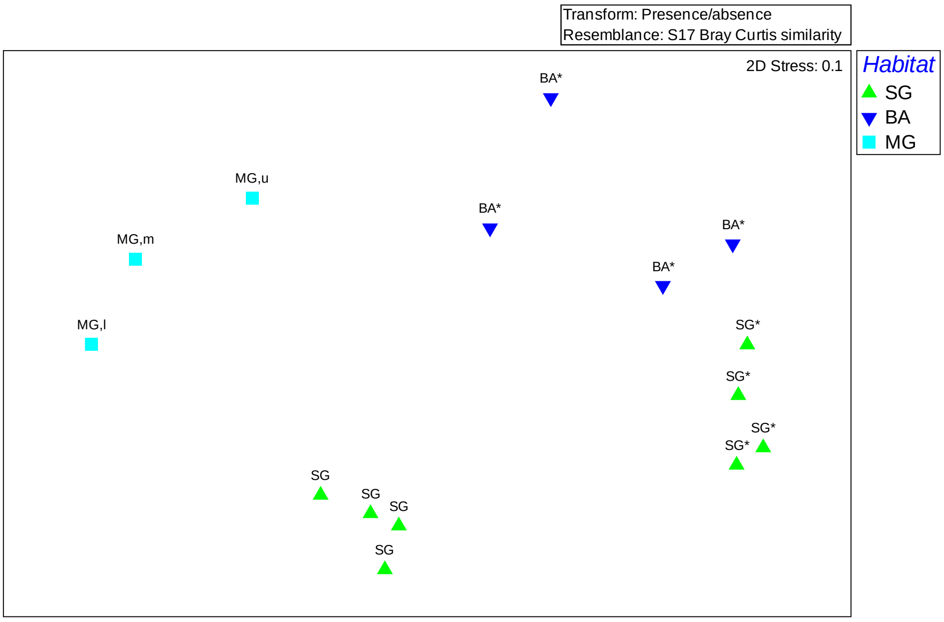


Figure 4.18 MDS ordination of fish assemblage data for Rangaunu Harbour showing pattern of similarities between all habitats sampled (A). Data were presence/absence transformed; Seagrass and sand habitats (B). Data were square root transformed. Habitat identifiers as per Figure 4.16.

.

A.



B.

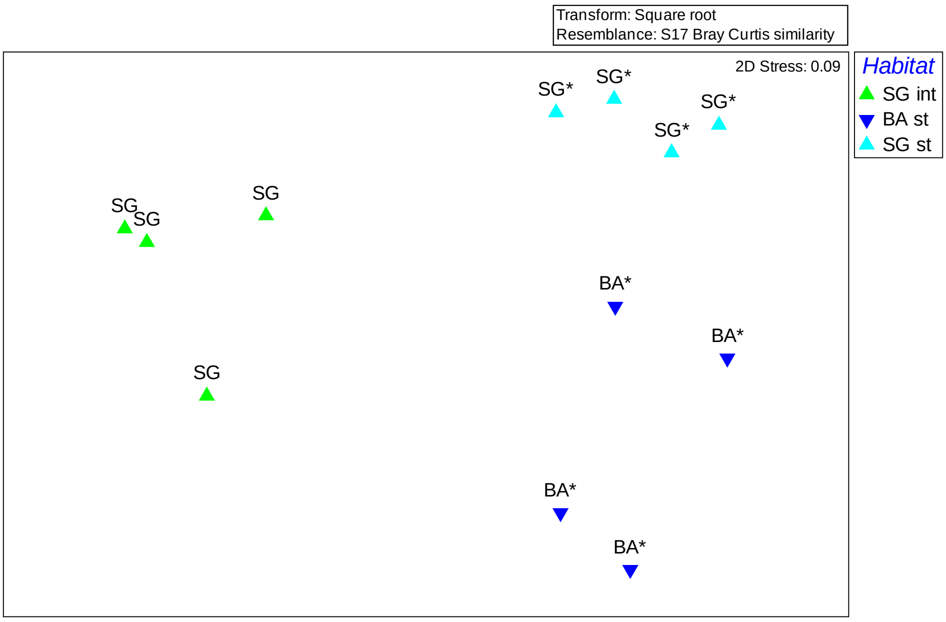


Figure 4.19 MDS ordination of fish assemblage data for Kaipara Harbour showing pattern of similarities between all habitats sampled (A). Data were presence/absence transformed; seagrass and sand habitats (B). Data were square root transformed. Site identifiers as per Figure 4.16.

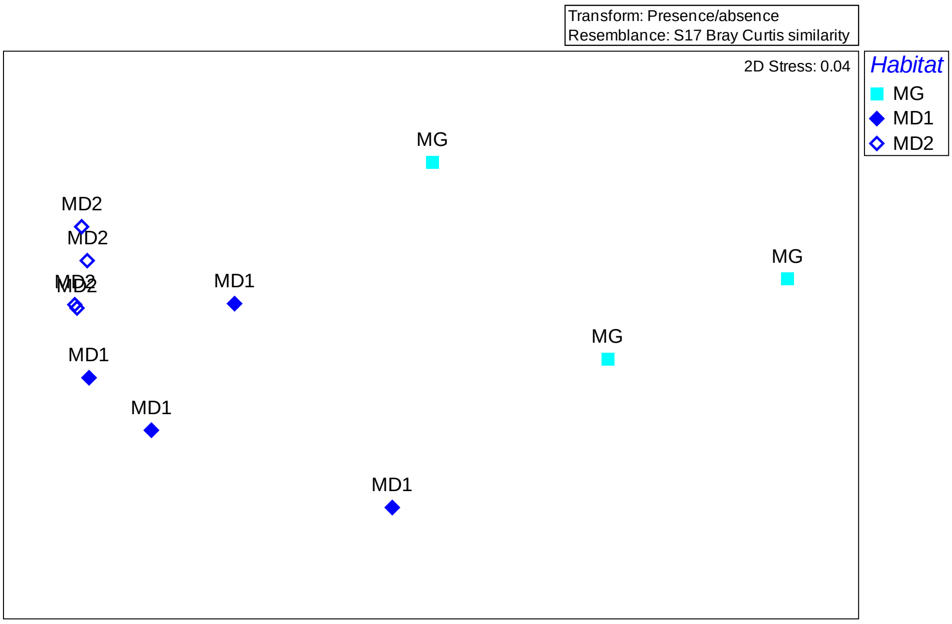


Figure 4.20 MDS ordination of fish assemblage data for Manukau Harbour showing pattern of similarities between both habitats sampled. Data were presence/absence transformed. MG=mangrove; MD=mud; numerals denote site number.

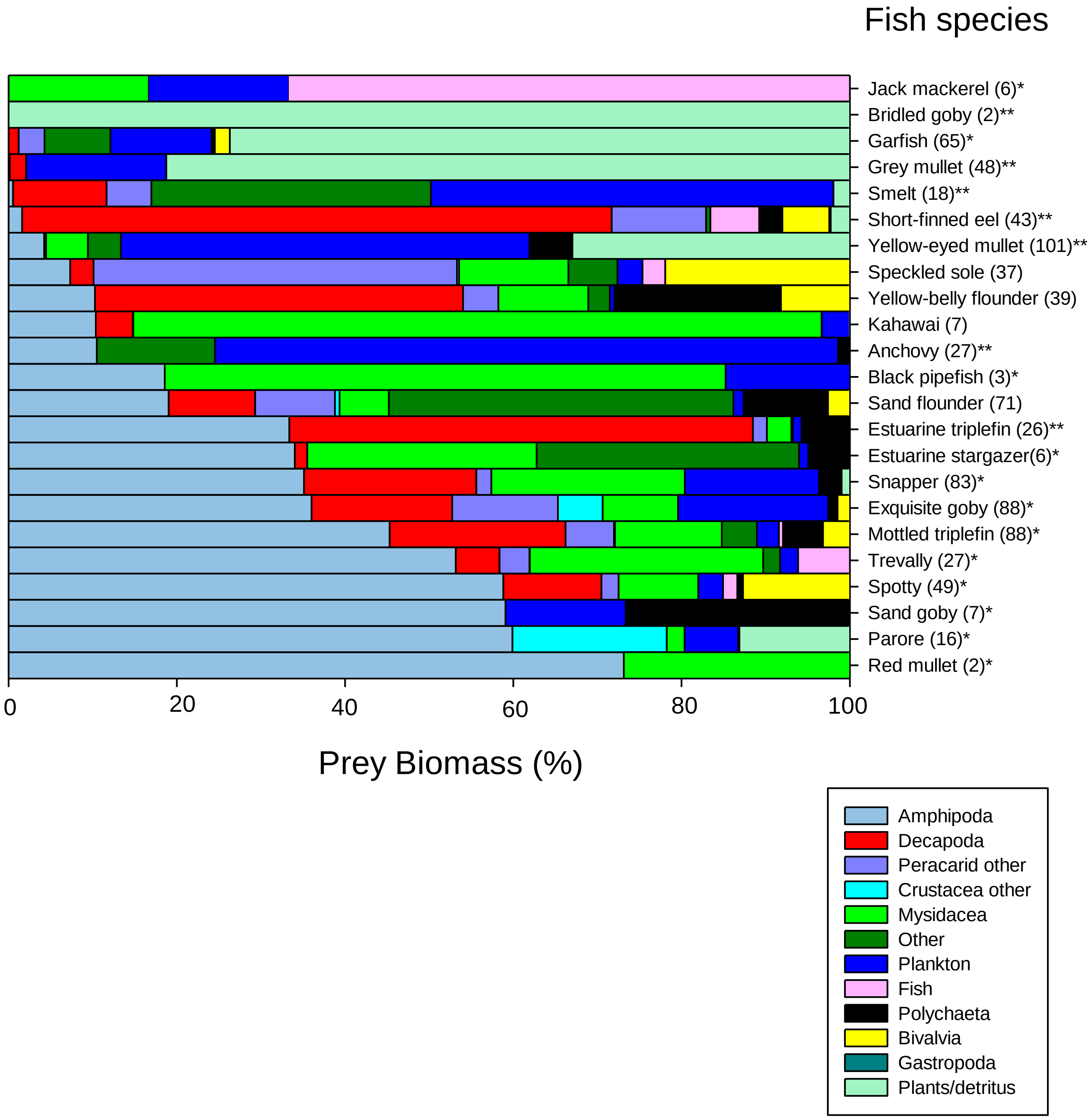


Figure 4.21 Proportional abundance of the major dietary categories in fish diets consumed across all habitats and harbours and fish sizes. Sample sizes are given in parentheses after species name. Superscripts denote which habitat (s) fish species were predominantly caught in. (NB. For jack mackerel; spotty and trevally Fish= fish scales only).

Key: ** Mangrove associated species
* Seagrass associated species
† Species found across all habitats
Mud associated species

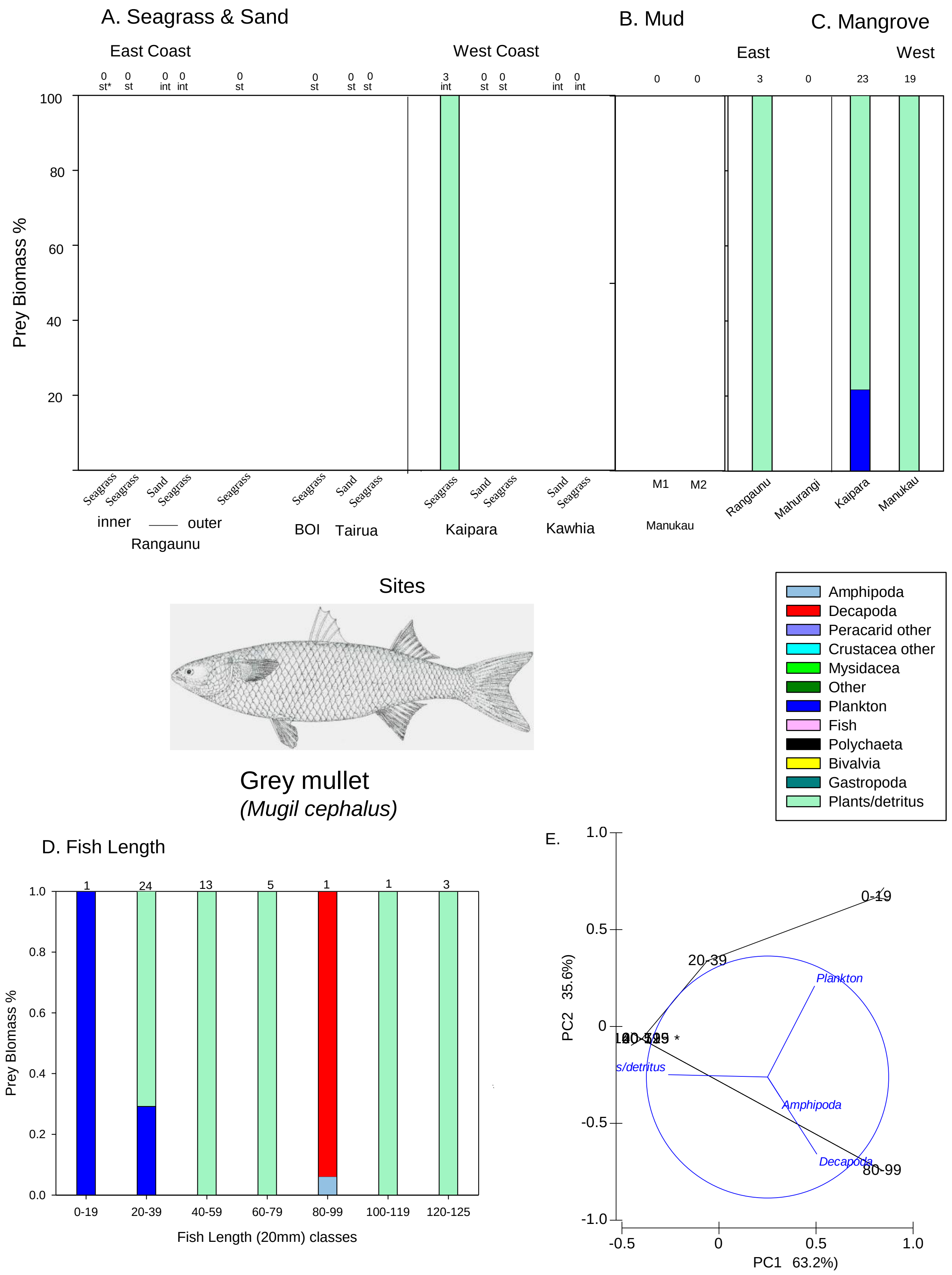


Figure 4.22 Proportional abundance of major dietary categories in grey mullet fish guts consumed across all habitats and harbours (A-C); by (20mm) length class (D), number of guts analyzed are shown above each histogram; PCA trajectory score plots of major dietary categories consumed by (20mm) length class (E); *Overlap of two size classes 40-59; 60-79 mm.

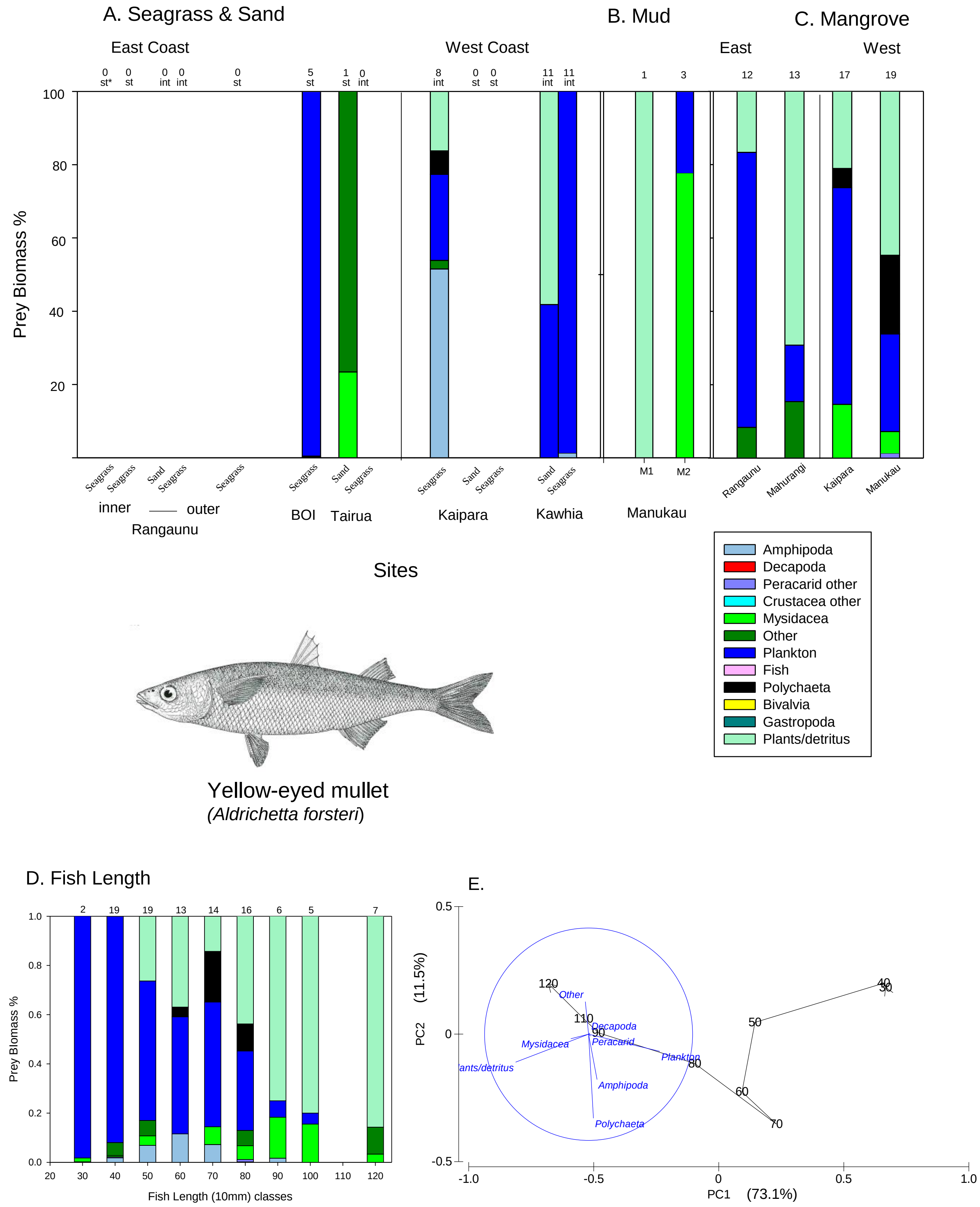


Figure 4.23 Proportional abundance of major dietary categories in yellow-eyed mullet fish guts consumed across all habitats and harbours (A-C); by (10mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (10mm) length class (E); *Peracarid=Peracarid other. Number of guts analyzed are shown above each histogram.

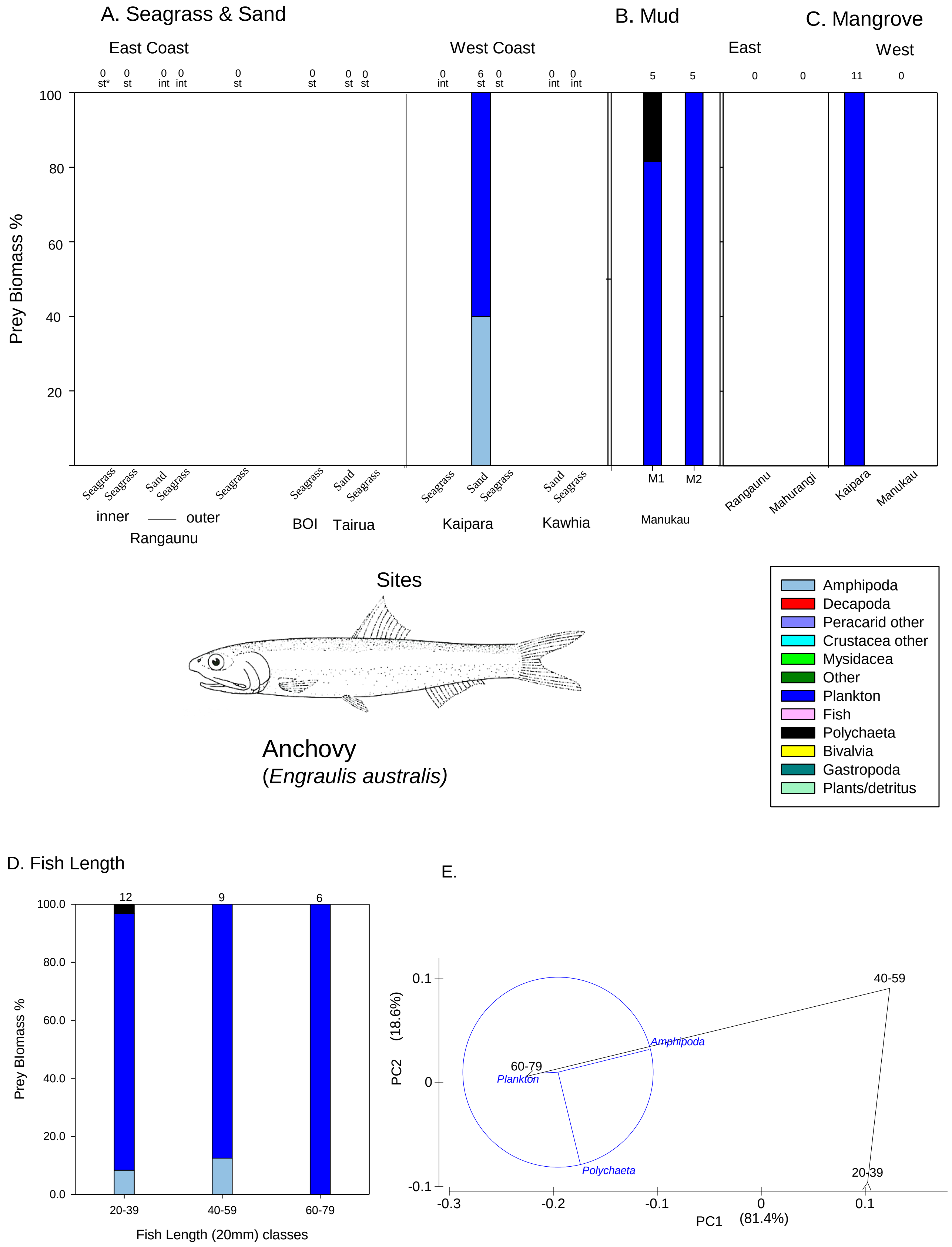


Figure 4.24 Proportional abundance of major dietary categories in anchovy fish guts consumed across all habitats and harbours (A-C); by (10mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (10mm) length class (E). Number of guts analyzed are shown above each histogram.

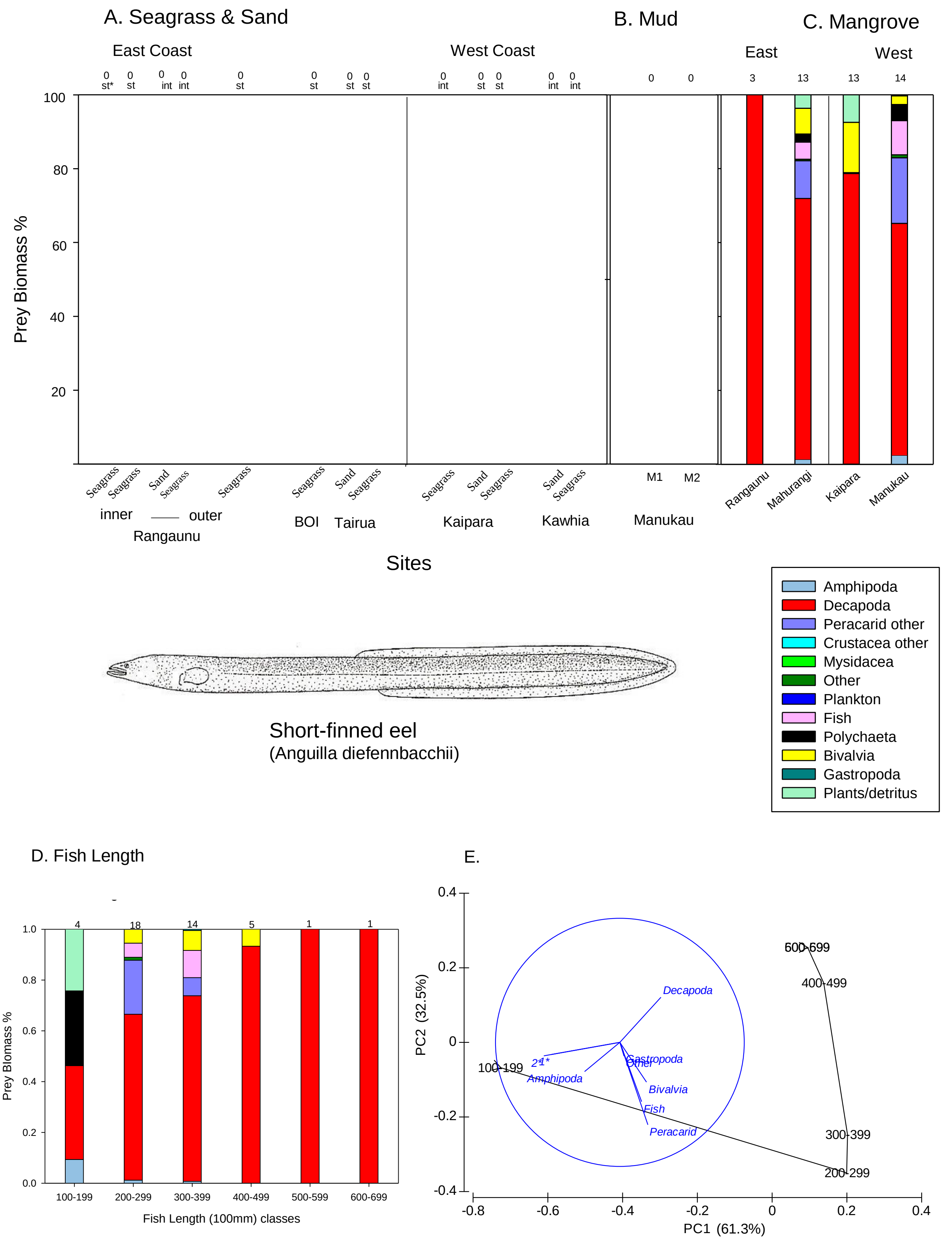


Figure 4.25 Proportional abundance of major dietary categories in short-finned eel fish guts consumed across all habitats and harbours (A-C); by (100mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (100mm) length class (E); Number of guts analyzed are shown above each histogram.

KEY: 1= Plant/detritus 2= Polychaeta
Peracarid=Peracarid other

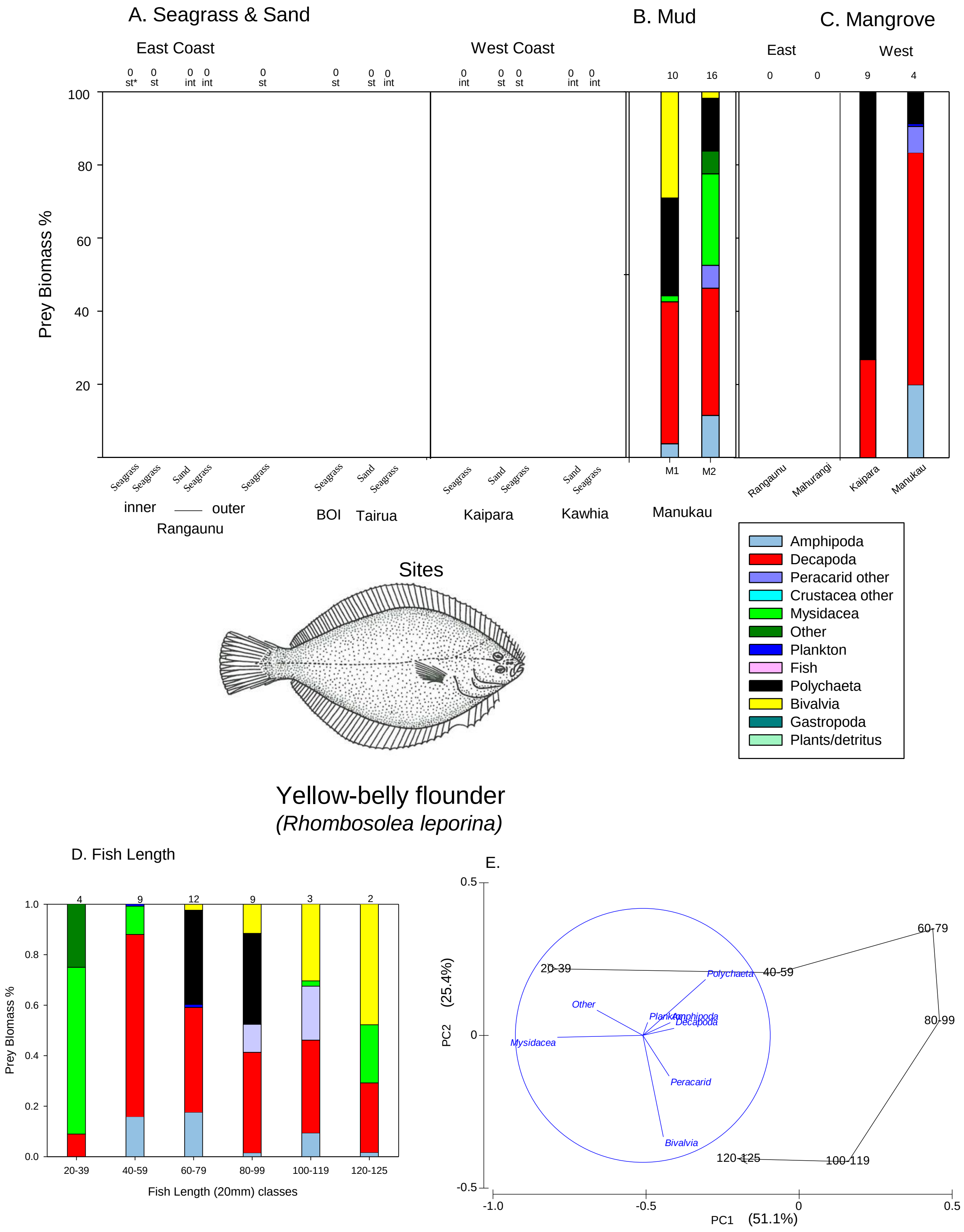


Figure 4.26 Proportional abundance of major dietary categories in yellow=-belly flounder fish guts consumed across all habitats and harbours (A-C); by (100mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (100mm) length class (E); *Peracarid=Peracarid other. Number of guts analyzed are shown above each histogram.

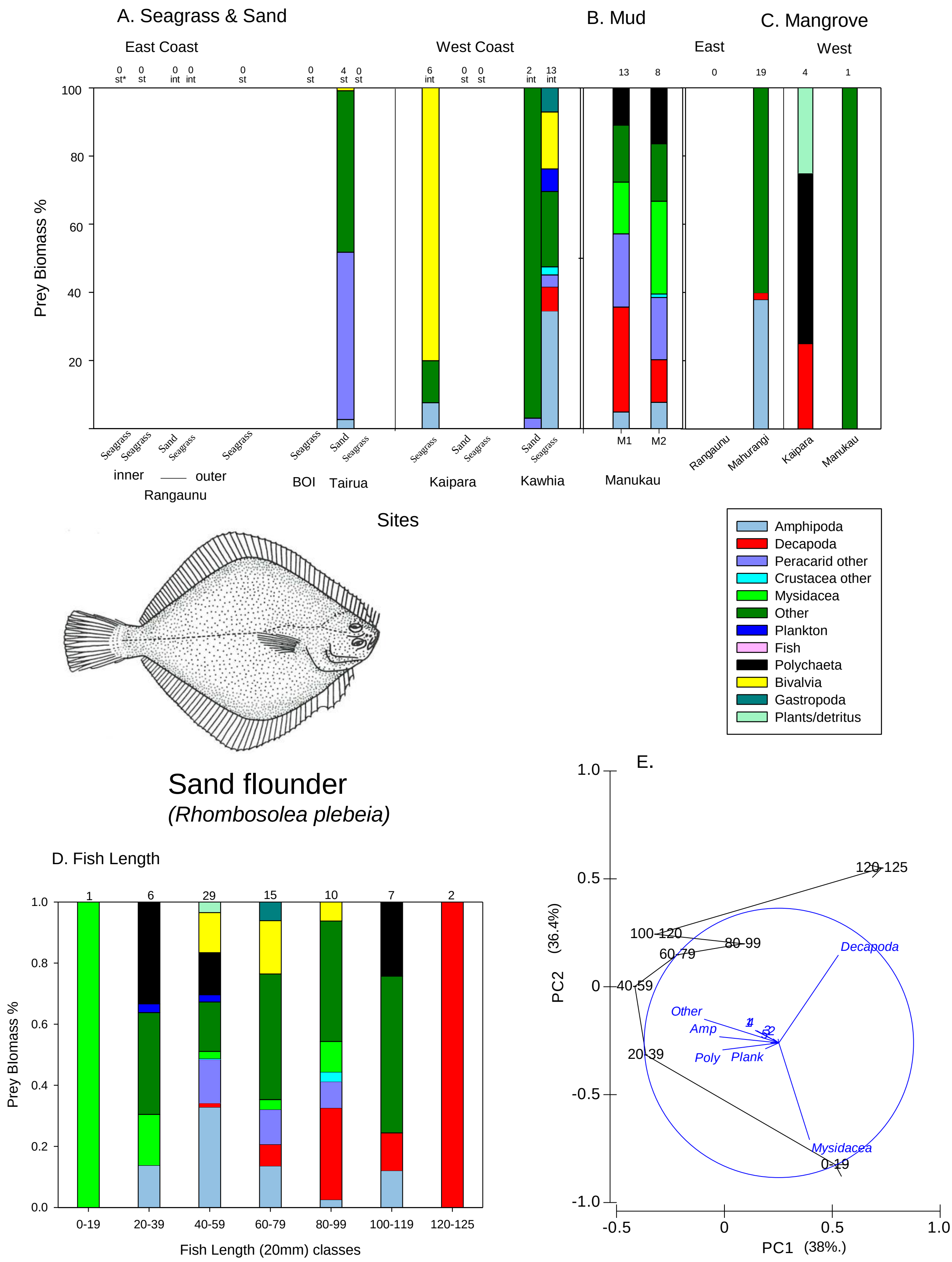


Figure 4.27 Proportional abundance of major dietary categories in sand flounder fish guts consumed across all habitats and harbours (A-C); by (20mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (20mm) length class (E). Number of guts analyzed are shown above each histogram.

KEY: Amp=Amphipods Poly=Polychaetea Plank=Plankton
2= Crustacea other 3= Gastropoda 5= Plant/detritus
1, 4 overlap= Bivalvia; Peracarid other

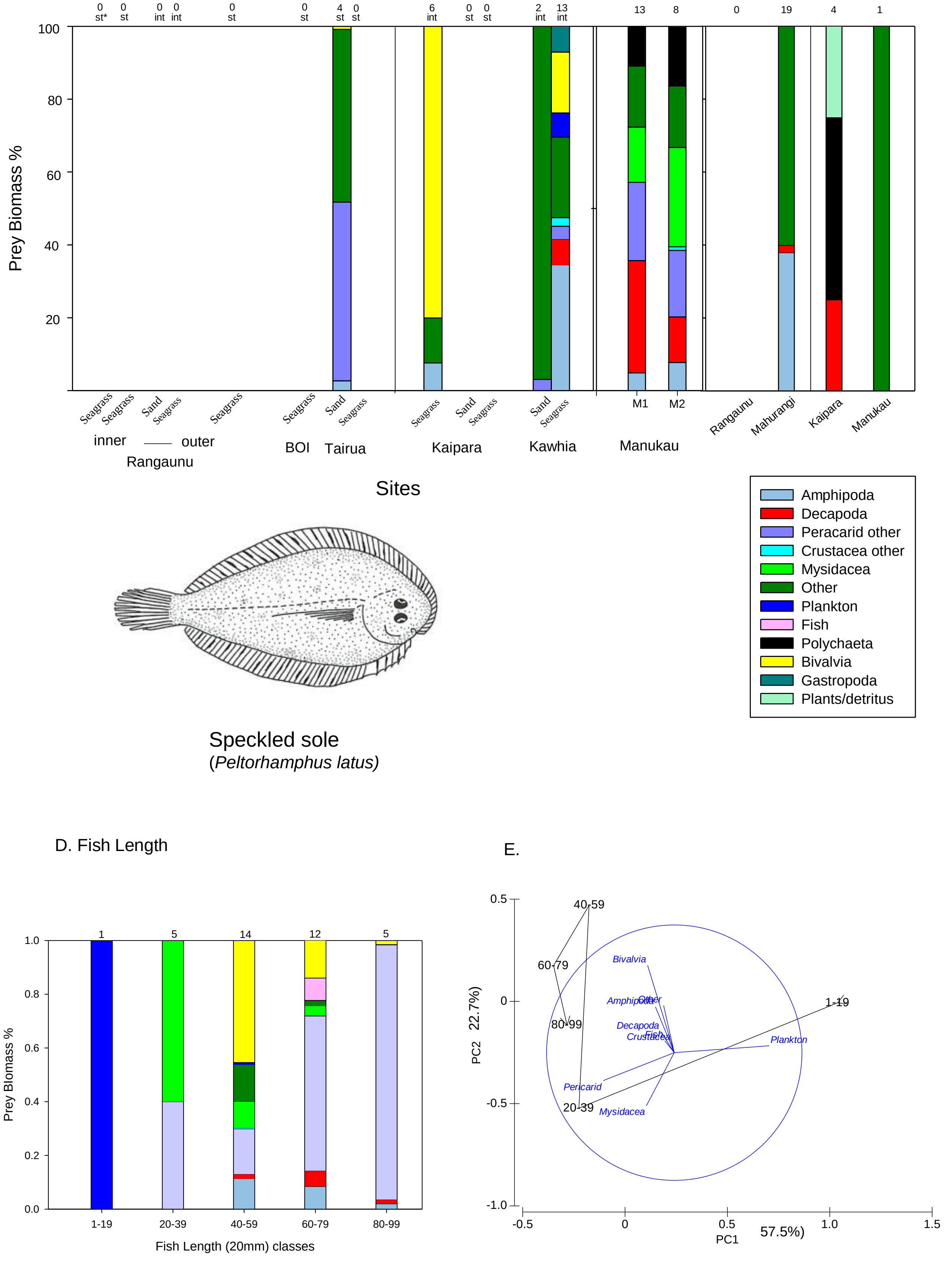


Figure 4.28 Proportional abundance of major dietary categories in speckled sole fish guts consumed across all habitats and harbours (A-C); by (20mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (20mm) length class (E); *Peracarid=Peracarid other. Number of guts analyzed are shown above each histogram. *Overlap of categories Amphipod/Other; Crustacea/Fish.

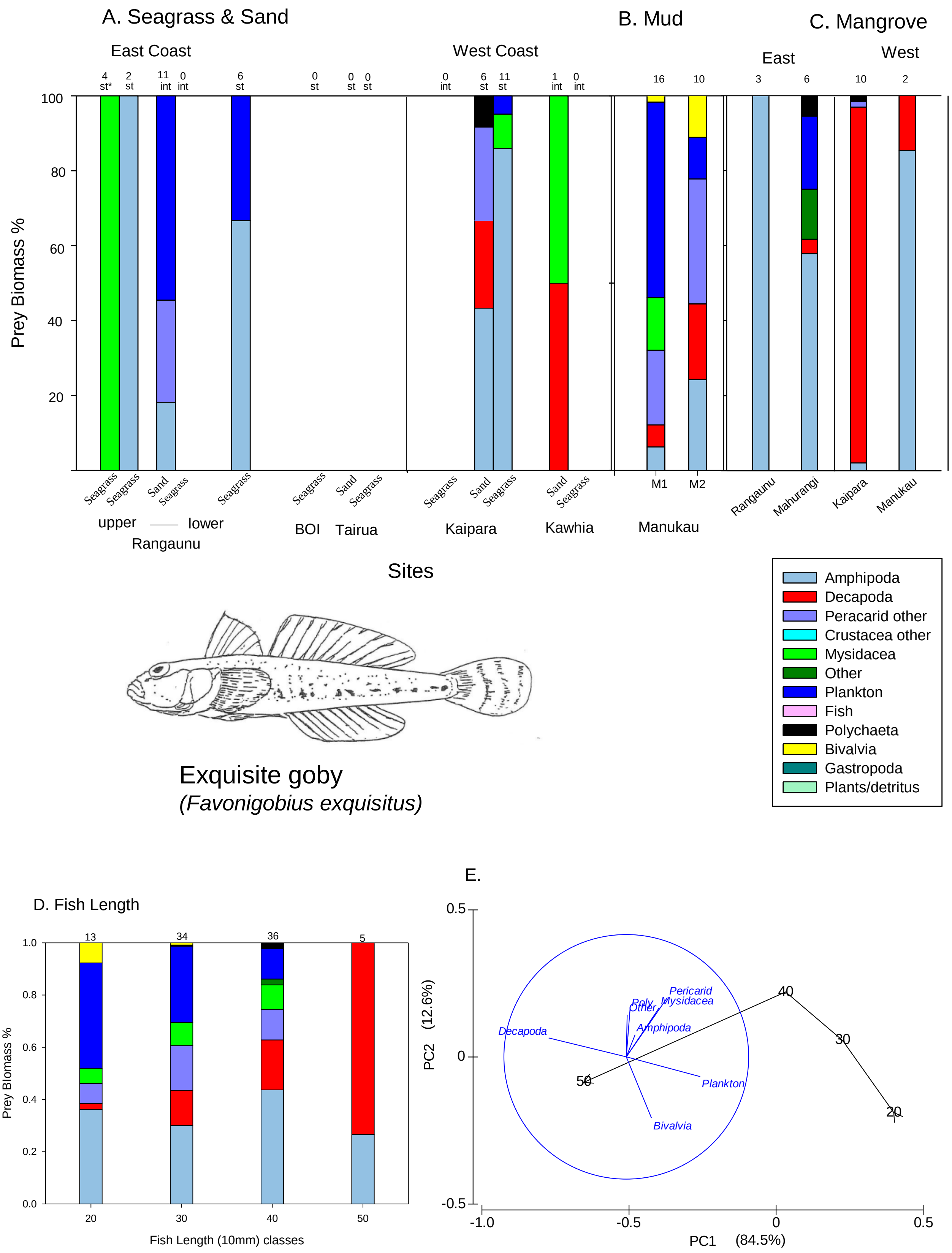


Figure 4.29 Proportional abundance of major dietary categories in exquisite goby fish guts consumed across all habitats and harbours (A-C); by (20mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (20mm) length class (E); *Poly=Polychaeta; Peracarid=Peracarid other. Number of guts analyzed are shown above each histogram.

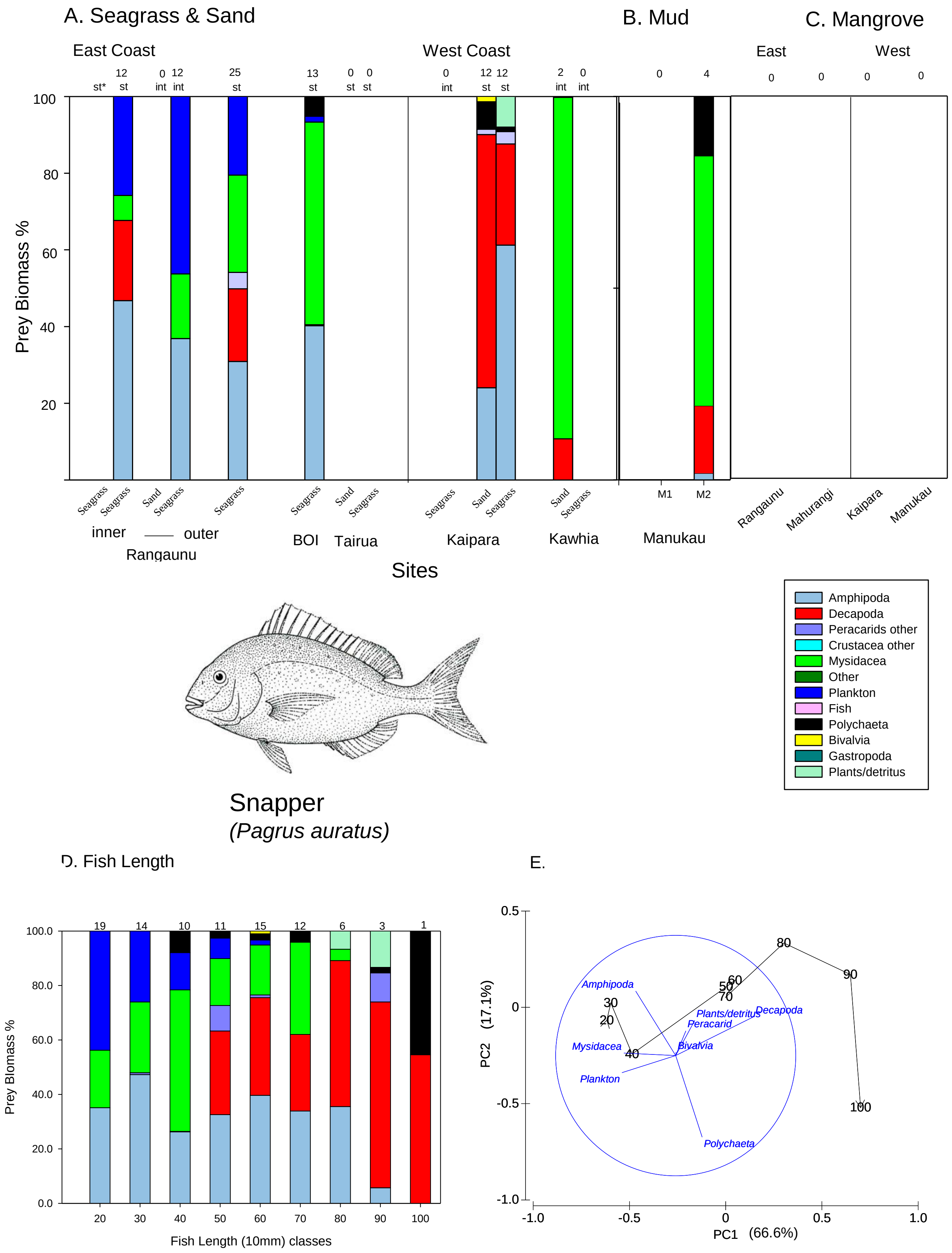


Figure 4.30 Proportional abundance of major dietary categories in snapper fish guts consumed across all habitats and harbours (A-C); by (10mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (10mm) length class (E); *Peracarid=Peracarid other. Number of guts analyzed are shown above each histogram.

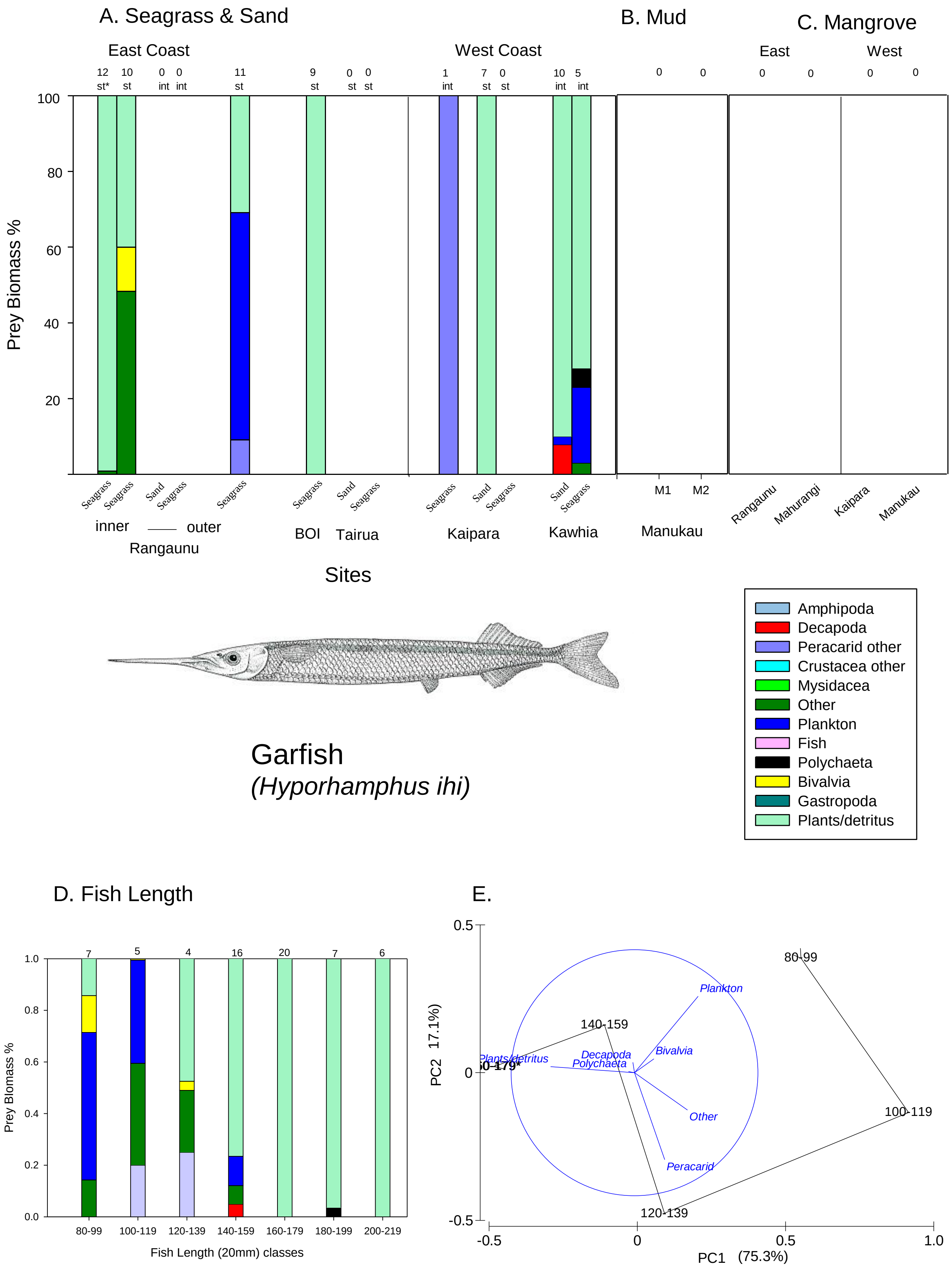


Figure 4.31 Proportional abundance of major dietary categories in garfish fish guts consumed across all habitats and harbours (A-C); by (20mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (20mm) length class (E); *Overlap for length class 169-219mm; Peracarid=Peracarid other. Number of guts analyzed are shown above each histogram.

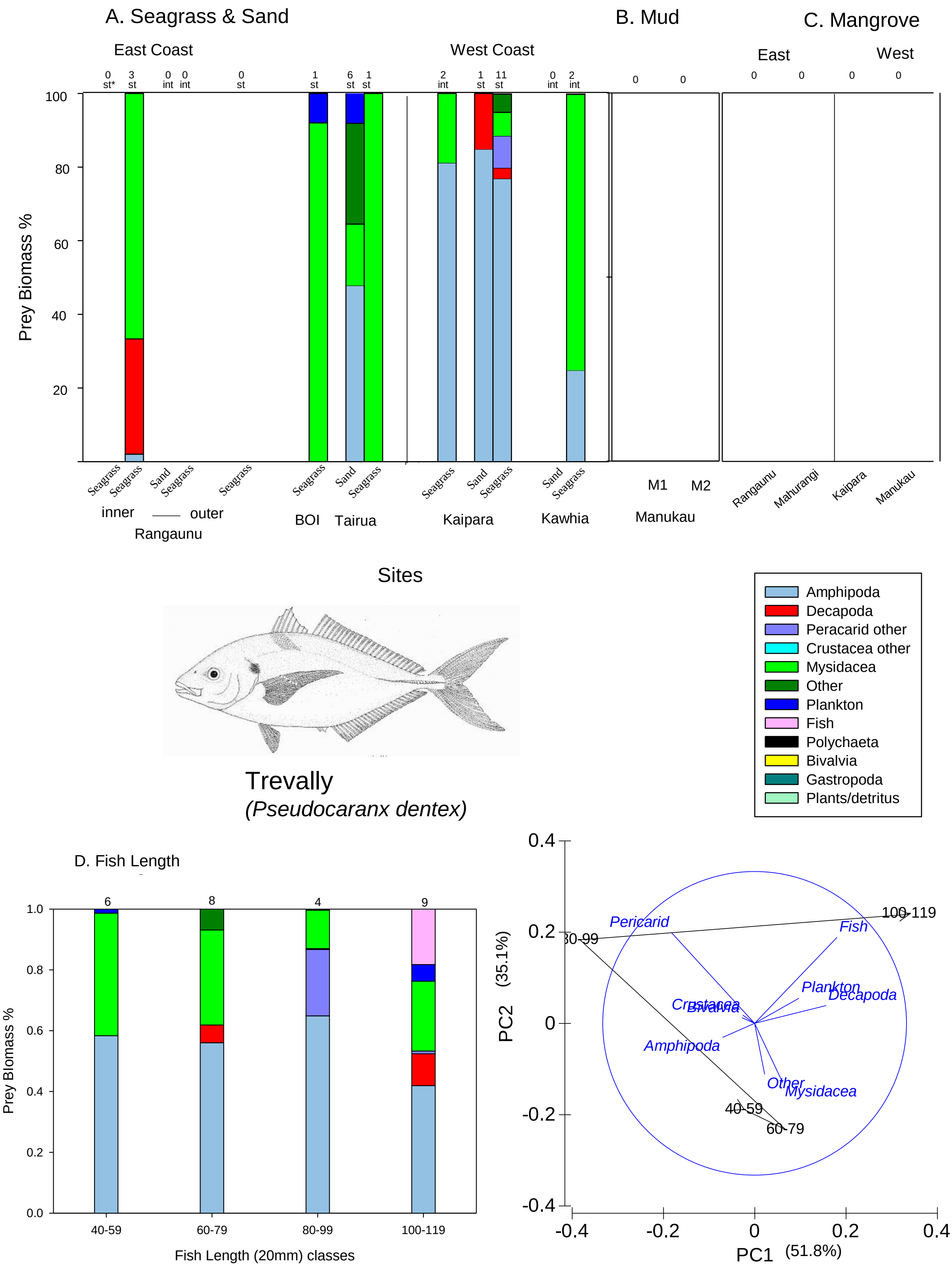


Figure 4.32 Proportional abundance of major dietary categories in trevally fish guts consumed across all habitats and harbours (A-C); by (20mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (20mm) length class (E); *Peracarid=Peracarid other. Number of guts analyzed are shown above each histogram.

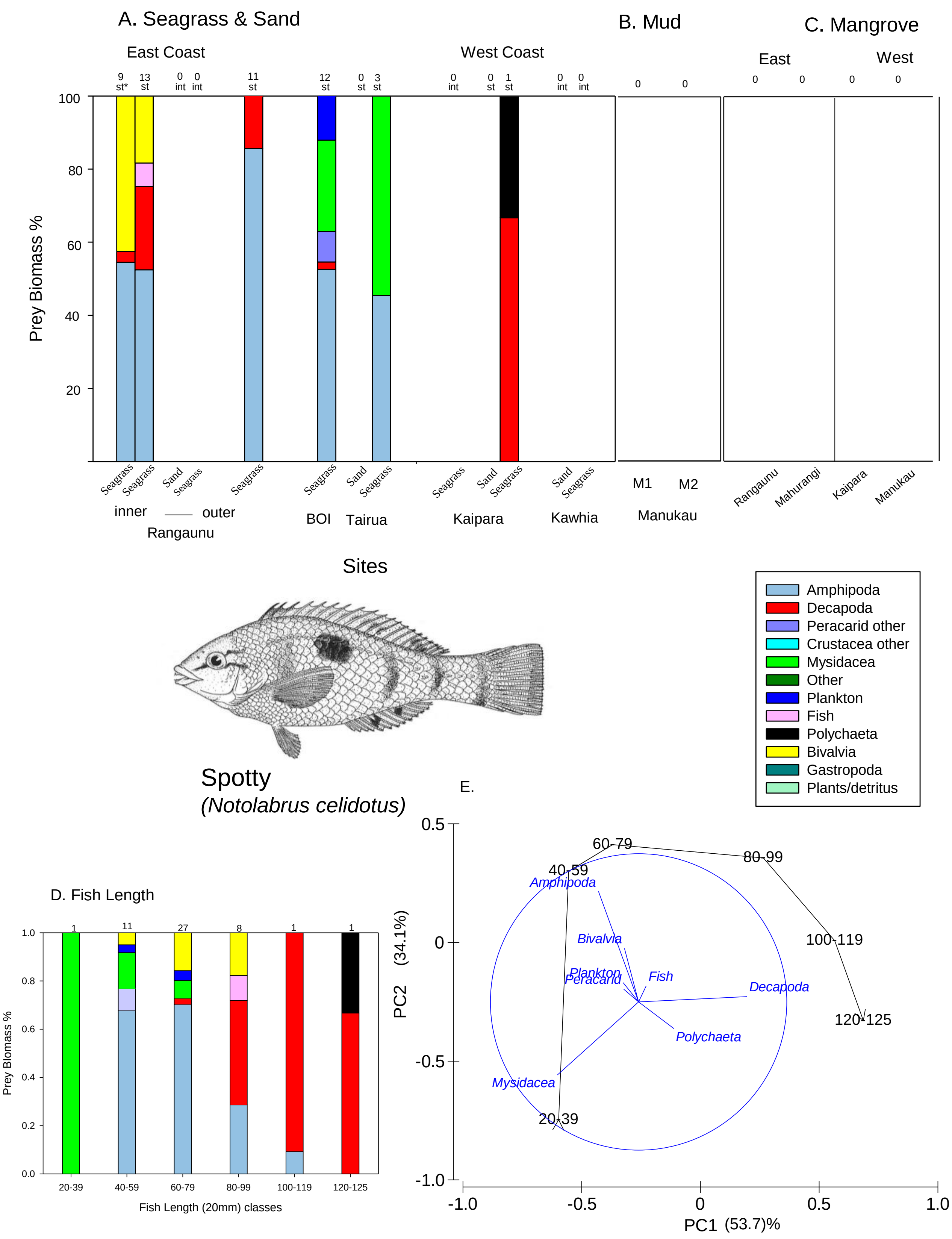


Figure 4.33 Proportional abundance of major dietary categories in spotty fish guts consumed across all habitats and harbours (A-C); by (20mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (20mm) length class (E); *Peracarid=Peracarid other. Number of guts analyzed are shown above each histogram.

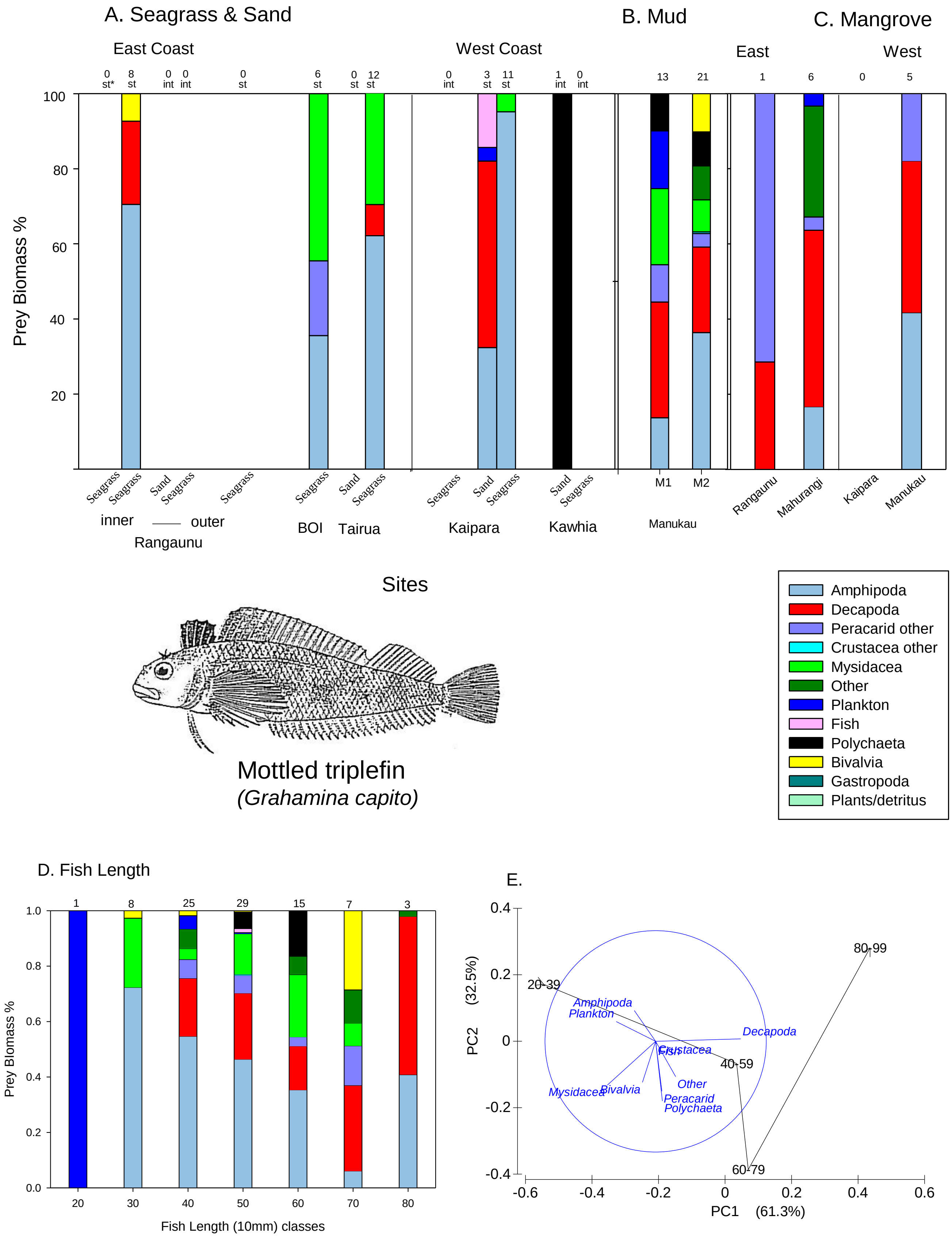
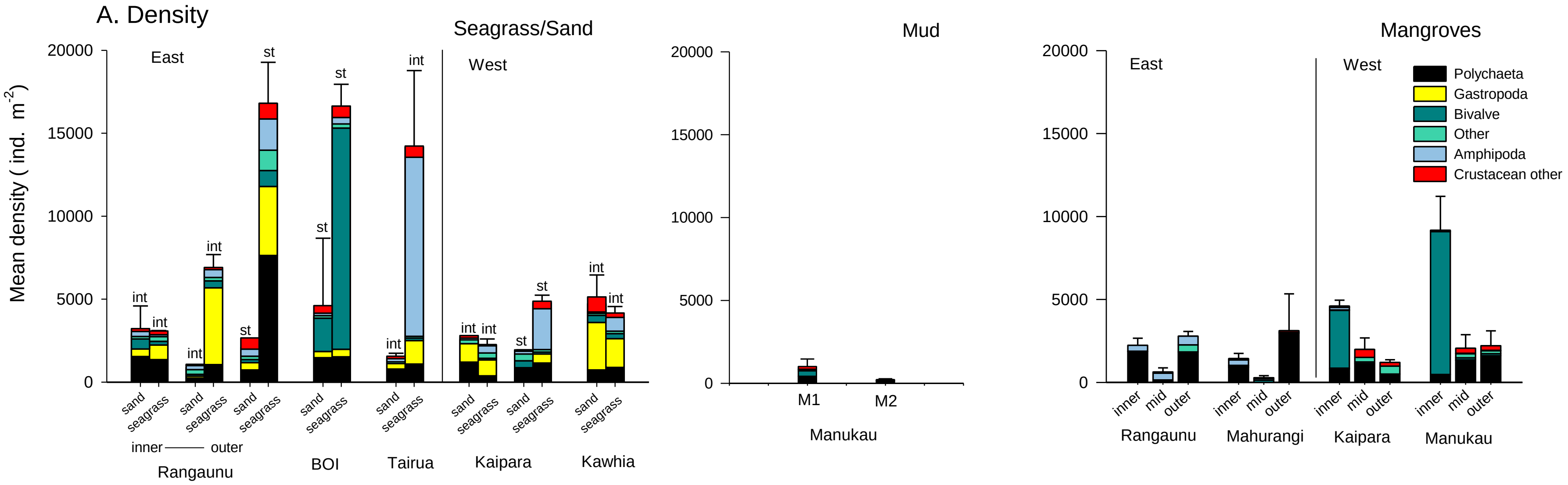


Figure 4.34 Proportional abundance of major dietary categories in mottled triplefin fish guts consumed across all habitats and harbours (A-C); by (10mm) length class (D); PCA trajectory score plots of major dietary categories consumed by (10mm) length class (E). *Overlap Crustacea/Fish; Peracarid=Peracarid other. Number of guts analyzed are shown above each histogram.

Appendix 4.1 Mean density (±SE) of benthic invertebrates (1-22 mm sieve size) collected from seagrass, sand, intertidal mud and mangrove habitats at 27 sites from 7 northern harbours.



East=East coast; West=West coast; st=subtidal; int= intertidal; inner= inner harbour; mid=mid harbour; outer=outer harbour

Table 4.1 Summary of harbours and habitats sampled in the survey.

		Rangaunu		BOI	Tairua	Kaipara	Kawhia	Mahurangi	Manukau
		upper	lower	Urupukapuka Is.					
Seagrass $^{\text{intertidal}}$		†	fish			fish	fish		
		benthos	benthos		benthos	benthos	benthos		
Seagrass $^{\text{subtidal}}$		fish*[1]	fish	fish	fish	fish			
			benthos	benthos		benthos			
Sand $^{\text{intertidal}}$		fish[2]	fish			†	fish		
		benthos	benthos		benthos	benthos	benthos		
Sand $^{\text{subtidal}}$		†	†	†	fish	fish			
		benthos	benthos	benthos		benthos			
Mangrove		fish				fish		fish	fish
		benthos				benthos		benthos	benthos
Mud									fish
									benthos

- †Not all habitats were able to be sampled due to weather conditions/gear difficulties and availability of sand sites adjacent (within 100m) to seagrass meadows.
- Fish*[1] reclassified from sand to subtidal seagrass as beachseining comprised 65% seagrass; 35% sand.
- Tairua fish samples were reclassified as subtidal due to narrow (1-2m) fringing subtidal habitat comprising part of the beach seining.

Table 4.2 Prey categories used in trophic analyses.

Category	**Description**
Amphipods	Predominantly gammarids
Decapods	
Peracarids (other)	Predominantly cumaceans, some isopods & tanaids
Crustacea (other)	All crustaceans excluding copepods, decapods, pericarids, mysids
Mysids	All mysid shrimps
Plankton	Calanoid/harpacticoid/cyclopoid copepods; cladocerans, barnacle cyprids; decapod zoeae
Fishes	All fishes, including larvae
Polychaetes	
Other	Nematodes, oligochaetes, ophiuroids, insects & eggs
Bivalves	All bivalves including siphons
Gastropods	
Plants/detritus	

Table 4.3 (A) Mean density (±SE) of all fish species collected from sand and seagrass habitats by beach seine at 14 sites from 5 harbours. Species are ranked in decreasing order of mean density.

Seagrass fish

		Rangaunu										**BOI**		**Tairua**				**Kaipara**						**Kawhia**				
Sites		**Upper**				**Lower**												**Lower**										
		Subt		**Subt**		**Inter**		**Inter**		**Subt**		**Subt**		**Subt**				**Inter**		**Subt**		**Subt**		**Int**		**Int**		**Total**
Species		**SG***		**SG**		**Sand**		**SG**		**SG**		**SG**		**Sand**		**SG**		**SG**		**Sand**		**SG**		**Sand**		**SG**		
Snapper	*Pagrus auratus*	6.67	(1.83)	159.14	(105.18)	–	–	2.56	(0.95)	21.57	(5.17)	4.89	(1.63)	–	–	0.11	(0.11)	–	–	0.61	(0.19)	5.18	(1.00)	0.22	(0.22)	0.06	(0.06)	201.00
Sand goby	*Favonigobius lentiginosus*	7.11	(2.47)	10.28	(7.30)	2.89	(2.56)	135.55	(44.96)	28.15	(16.40)	–	–	–	–	–	–	0.17	(0.11)	–	–	0.08	(0.05)	–	–			184.22
Yellow-eyed mullet	*Aldrichetta forsteri*	–	–	–	–	–	–	32.83	(21.51)	2.96	(2.96)	0.44	(0.27)	13.20	-8.70	4.67	(2.70)	12.67	(3.18)	–	–	–	–	84.11	(58.11)	41.20	(3.67)	192.09
Exquisite goby	*Favonigobius exquisitus*	10.94	(5.75)	7.89	(3.24)	2.00	(2.00)	25.83	(9.44)	18.06	(8.62)	0.17	(0.11)	–	–	0.33	(0.26)	58.93	(28.24)	3.56	(0.59)	9.52	(3.29)	0.11	(0.11)	3.70	(1.16)	141.03
Garfish	*Hyporhamphus ihi*	5.67	(3.20)	12.83	(5.31)	–	–	1.33	(0.70)	30.71	(15.73)	2.00	(0.96)	–	–	0.11	(0.11)	–	–	0.44	(0.31)	6.14	(2.91)	–	–	5.86	(5.53)	65.11
Spotty	*Notolabrus celidotus*	1.11	(1.04)	40.72	(15.15)	–	–	0.50	(0.32)	8.70	(1.40)	2.17	(1.26)	–	–	2.11	(1.33)	–	–	–	–	0.06	(0.06)	–	–	–	–	55.37
Mottled triplefin	*Grahimina capito*	1.39	(0.70)	5.44	(1.14)	–	–	1.11	(0.52)	31.23	(13.77)	0.67	(0.59)	–	–	0.78	(0.78)	0.06	(0.06)	0.22	(0.22)	3.25	(1.30)	0.22	(0.00)	0.07	(0.07)	44.45
Parore	*Girella tricuspidata*	–	–	–	–	–	–	0.50	(0.29)	–	–	0.22	(0.09)	–	–	39.87	(23.17)	–	–	–	–	–	–	–	–	–	–	40.59
Kahawai	*Arripus trutta*	–	–	–	–	–	–	0.06	(0.06)	–	–	–	–	0.40	(0.21)	–	–	1.06	(0.98)	0.06	(0.06)	0.06	(0.06)	–	–	2.22	(1.52)	3.84
Jack mackerel	*Trachurus* spp	–	–	16.33	(16.33)	–	–	–	–	0.34	(0.30)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	16.67
Speckled sole	*Peltorhamphus latus*	–	–	–	–	–	–	0.06	(0.06)	–	–	–	–	–	–	0.06	(0.06)	6.39	(3.91)	0.89	(0.51)	0.22	(0.13)	0.56	(0.11)	4.13	(0.60)	12.30
Estuarine triplefin	*Grahamina nigripenne*	0.83	(0.58)	1.89	(0.41)	–	–	1.94	(0.69)	–	–	0.67	(0.40)	0.59	(0.20)	4.33	(2.44)	–	–	–	–	–	–	–	–	–	–	10.26
Anchovy	*Engraulis australis*	–	–	–	–	–	–	0.06	(0.06)	–	–	–	–	–	–	–	–	–	–	8.72	(3.72)	–	–	–	–	–	–	8.78
Sand flounder	*Rhombosolea plebeia*	–	–	0.06	(0.06)	–	–	–	–	–	–	–	–	–	–	0.06	(0.06)	1.44	(0.56)	0.06	(0.06)	–	–	0.22	(0.22)	6.35	(1.29)	8.19
Trevally	*Pseudocaranx dentex*	–	–	1.72	(0.98)	–	–	0.11	(0.11)	0.12	(0.05)	0.33	(0.21)	–	–	0.50	(0.25)	0.67	(0.18)	0.06	(0.06)	4.49	(1.69)	–	–	0.11	(0.11)	8.12
Yellow-belly flounder	*Rhombosolea leporina*	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.33	(0.21)	0.06	(0.06)	–	–	–	–	1.56	(0.55)	1.94
Grey mullet	*Mugil cephalus*	–	–	–	–	–	–	–	–	–	–	–	–	0.07	(0.07)	0.28	(0.28)	1.11	(0.53)	–	–	–	–	–	–	0.22	(0.22)	1.69
Black pipefish	*Stigmatopora nigra*	–	–	–	–	–	–	0.56	(0.29)	0.09	(0.06)	0.11	(0.11)	–	–	0.50	(0.29)	0.17	(0.11)	–	–	–	–	–	–	0.09	(0.05)	1.52
Red mullet	*Upeneichthys lineatus*	0.06	(0.06)	0.39	(0.11)	–	–	–	–	0.25	(0.10)	0.50	(0.29)	–	–	0.06	(0.06)	–	–	–	–	–	–	–	–	0.03	(0.03)	1.28
Bumblee squid	*Sepioloidea pacifica*	–	–	–	–	–	–	–	–			0.50	(0.32)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.50
Smooth pipefish	*Stigmatopora macropterygia*	0.06	(0.06)	–	–	–	–	–	–	–	–	0.11	(0.11)	–	–	0.06	(0.06)	–	–	0.06	(0.06)	–	–	–	–	–	–	0.28
Spotted stargazer	*Genyagnus monoterygius*	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.28	(0.14)	0.28
Gurnard	*Chelidonichthys kumu*	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.06	(0.06)	–	–	–	–	–	–	0.15	(0.09)	0.20
Koheru	*Decapterus koheru*	–	–	0.17	(0.17)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.17
Striped clingfish	*Trachelochismus melobesia*	–	–	–	–	–	–	0.06	(0.06)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.11	(0.11)	–	–	0.17
Speckled pipefish	*Leptonotus elevatus*	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.10	(0.06)	–	–	0.03	(0.03)	0.13
Pilchard	*Sardinops neopilchardus*	–	–	–	–	–	–	0.06	(0.06)	0.03	(0.03)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.09
Short-finned eel	*Anguilla australis*	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.06	(0.06)	–	–	–	–	–	–	–	–	–	–	0.06
Squid		–	–	–	–	–	–	–	–	0.03	(0.03)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.03
Estuarine stargazer	*Leptoscopus macropygus*	–	–	–	–	–	–	–	–	–	–	–	–	0.59	0.37	–	–	–	–	–	–	–	–	–	–	–	–	0.59
Total av. per 100 m2		33.83		256.86		4.89		203.11		142.26		12.78		14.85		53.87		83.04		14.72		29.10		85.56		66.07		1000.93

Subt=subtidal; Inter=intertidal; SG=Seagrass SG=Site with 65% seagrass and 35% sand

Table 4.3 (B) Mean density (±SE) of all fish species collected from intertidal mud habitat by beach seine at two sites from the Manukau Harbour. Species are ranked in decreasing order of mean density.

Sites		**M1**		**M2**		**Total**
Species						
Yellow-belly flounder	*Rhombosolea leporina*	60.19	(12.25)	77.92	(19.88)	138.11
Sand flounder	*Rhombosolea plebeia*	23.64	(5.52)	73.47	(17.89)	97.11
Exquisite goby	*Favonigobius exquisitus*	25.72	(15.73)	61.25	(26.20)	86.97
Speckled sole	*Peltorhamphus latus*	3.00	(1.77)	32.92	(6.29)	35.92
Yellow-eyed mullet	*Aldrichetta forsteri*	18.17	(1.51)	5.83	(2.18)	24.00
Sprat	*Sprattus* spp.	3.67	(3.45)	11.81	(2.86)	15.47
Mottled triplefin	*Grahamina capito*	1.03	(0.66)	11.94	(5.01)	12.97
Smelt	*Retropinna retropinna*	10.33	(4.97)	0.00	(0.00)	10.33
Anchovy	*Engraulis australis*	0.64	(0.47)	1.53	(1.00)	2.17
Estuarine stargazer	*Leptoscopus macropygus*	0.39	(0.26)	0.97	(0.35)	1.36
Jack mackerel	*Trachurus* spp	0.00	(0.00)	1.11	(0.32)	1.11
Snapper	*Pagrus auratus*	0.00	(0.00)	0.83	(0.36)	0.83
Spotted stargazer	*Genyagnus monoterygius*	0.00	(0.00)	0.42	(0.14)	0.42
Estuarine triplefin	*Grahamina nigripenne*	0.22	(0.22)	0.00	(0.00)	0.22
Short-finned eel	*Anguilla australis*	0.17	(0.11)	0.00	(0.00)	0.17
Common bully	*Gobiomorphus cotidianus*	0.00	(0.00)	0.14	(0.14)	0.14
Gurnard	*Chelidonichthys kumu*	0.00	(0.00)	0.14	(0.14)	0.14
Total av. per 100m^2		147.17		280.28		427.24

Table 4.3 (C) Mean density (±SE) of all fish species collected from mangrove habitats by fyke net at 12 sites from 4 harbours. Species are ranked in decreasing order of density.

Sites			**Rangaunu**		**Mahurangi**		**Kaipara**		**Manukau**		**Total**
Species		**Group**									
Grey mullet	*Mugil cephalus*	Pel	0.7	(0.3)	0.7	(0.3)	41.0	(36.5)	814.7	(568.0)	857.0
Yellow-eyed mullet	*Aldrichetta forsteri*	Pel	39.7	(17.4)	229.7	(64.2)	130.3	(43.0)	83.3	(37.9)	483.0
Pilchard	*Sardinops neopilchardus*	Pel	–	–	94.0	(94.0)	0.0	(0.0)	2.7	(2.7)	96.7
Estuarine triplefin	*Grahamina nigripenne*	B	47.3	(34.0)	35.0	(19.5)	0.3	(0.3)	5.0	(4.0)	87.7
Anchovy	*Engraulis australis*	Pel	–	–	2.0	(2.0)	49.7	(47.7)	0.7	(0.7)	52.3
Smelt	*Retropinna retropinna*	Pel	12.7	(11.7)	14.3	(2.4)	0.0	(0.0)	11.0	(8.5)	38.0
Short-finned eel	*Anguilla*	B	0.3	(0.3)	10.3	(2.7)	16.0	(1.0)	2.7	(1.2)	29.3
Sand flounder	*Rhombosolea plebeia*	B	1.0	(0.0)	22.7	(18.4)	2.3	(1.5)	1.7	(0.9)	27.7
Yellow-bellied flounder	*Rhombosolea leporina*	B	–	–	2.7	(1.3)	1.7	(0.9)	10.0	(9.5)	14.3
Exquisite goby	*Favonigobius exquisitus*	B	5.0	(1.2)	3.3	(1.8)	0.7	(0.7)	0.7	(0.7)	9.7
Mottled triplefin	*Grahamina capito*	B	–	–	1.7	(1.7)	–	–	6.3	(6.3)	8.0
Parore	*Girella tricuspidata*	B	0.7	(0.7)	1.7	(1.7)	–	–	1.3	(1.3)	3.7
Bridled goby	*Arenigobious bifrenatus*	B	–	–	2.7	(1.8)	–	–	–	–	2.7
Jack mackerel	*Trachurus sp.*	Pel	0.7	(0.3)	–	–	–	–	–	–	0.7
Garfish	*Hyporhamphus ihi*	Pel	–	–	0.3	(0.3)	–	–	–	–	0.3
Spotty	*Notolabrus celidotus*	B	0.3	(0.3)	–	–	–	–	–	–	0.3
Total av. per fyke net			108.3		421.0		242.0		940.0		1711.3

Table 4.4 Physical characteristics of the seagrass meadows and environment of the five harbours surveyed.

Harbour	Depth (m)	Blade length (cm) Mean (±SE)	Density/ Coverage	Exposure	Coast	Water clarity (NTU)	Substrate
Rangaunu $^{\text{Upper}}$	1.0 $^{\text{Intertidal}}$	9.4 (±1.9)	Med/Continuous	Sheltered	East	1	Fine sand
Rangaunu $^{\text{Upper}}$	>1.5 $^{\text{Subtidal}}$	17.3 (±6.5)	Med/Continuous	Sheltered	East	<1	Fine sand
Rangaunu $^{\text{Lower}}$	1.2 $^{\text{Intertidal}}$	9.9 (±1.4)	Med/semi continuous	Sheltered	East	1	Fine sand
Rangaunu $^{\text{Lower}}$	> 2 $^{\text{Subtidal}}$	21.54 (±2.2)	Lush/Thick[†]	Sheltered	East	<1	Fine sand
Urupukapuka Is. BOI	3.5 $^{\text{Subtidal}}$	29.3 (±2.1)	Med/Patchy[ac]	Open coast	East	0.39	Sandy[c]
Tairua	0.2 $^{\text{Subtidal}}$	10.0(±3.6)	Med/Continuous	Semi sheltered	East	8[d]	Coarse sand
Kaipara $^{\text{Lower}}$	< 2 $^{\text{Intertidal}}$	9.3 (±9.64)	Sparse/Patchy[a]	Exposed	West	~16[e]	Fine sand
Kaipara $^{\text{Lower}}$	> 2 $^{\text{Subtidal}}$	23.1 (±2.5)	Sparse/Patchy[a]	Exposed	West	~13[e]	Coarse sand
Kawhia	0.3 $^{\text{Intertidal}}$	9.3 (±1.4)	Sparse/Patchy[b]	Exposed	West	~20[d]	Fine iron sand/
							muddy clay[d]

[†] =subtidal meadow lush and verdant thinning with depth
[a]= patch size 20-50 meters
[b]=patch size meters

[c]Matheson et al (2010)
[d]Vant (pers. comm.)
[e]Author unpubl. data

Table 4.5 R- and p-values of ANOSIM comparisons between fish assemblages of the four habitats sampled. Non significant pair-wise comparisons are given in bold.

Habitat	R-value	Significance level (p)
Seagrass, Sand	0.253	0.004
Seagrass, Mangrove	0.589	0.001
Seagrass, Mud	0.446	0.001
Sand, Mud	0.393	0.003
Sand, Mangrove	0.704	0.001
Mangrove, Mud	0.829	0.001

Table 4.6 R- values and significance levels (p) of ANOSIM comparisons between fish assemblages of the five Harbours sampled. Non significant pair-wise comparisons are given in bold.

Harbour	Rangaunu	BOI	Tairua	Kaipara	Kawhia
Rangaunu					
BOI	0.36				
	0.002				
Tairua	0.714	0.464			
	0.001	0.033			
Kaipara	0.368	0.529	0.668		
	0.001	0.001	0.001		
Kawhia	0.642	1	0.398	0.408	
	0.001	0.005	0.004	0.005	

Table 4.7 Results of SIMPER analysis for all seagrass/sand sites showing the important discriminating fish species and the percent contribution to the average dissimilarity between the five harbours.

Average abundance by harbour

Species	BOI	RNGU	Contrib%	Cum.%
Sand goby	0	4.54	20.34	20.34
Snapper	2.13	4.09	13.79	34.13
Exquisite goby	0.28	3.08	13.36	47.49
Garfish	1.17	2.36	9.84	57.33
Spotty	1.28	2.12	9.41	66.74
Mottled triplefin	0.51	1.92	7.62	74.35
Yellow-eyed mullet	0.47	1.14	5.88	80.24
	RNGU	**TAIR**		
Sand goby	4.54	0	15.85	17.38
Snapper	4.09	0.1	11.43	29.91
Exquisite goby	3.08	0.22	10.26	41.16
Yellow-eyed mullet	1.14	2.08	9.24	51.29
Parore	0.11	2.89	8.92	61.07
Garfish	2.36	0.1	7.47	69.26
Spotty	2.12	0.67	6.22	76.08
Mottled triplefin	1.92	0.25	5.88	82.53
	KAIP	**RNGU**		
Sand goby	0.16	4.54	17.39	17.39
Snapper	1	4.09	12.83	30.22
Exquisite goby	3.83	3.08	12.76	42.98
Garfish	0.9	2.36	9.19	52.18
Spotty	0.04	2.12	7.21	59.38
Yellow-eyed mullet	1.16	1.14	7.16	66.54
Mottled triplefin	0.68	1.92	7.15	73.69
Anchovy	0.87	0.02	5.35	79.04
	KAWH	**RNGU**		
Yellow-eyed mullet	7.1	1.14	24.47	24.47
Sand goby	0	4.54	13.81	38.29
Snapper	0.19	4.09	10.65	48.94
Exquisite goby	1.29	3.08	7.55	56.49
Garfish	1.02	2.36	7.45	63.94
Sand flounder	1.77	0.02	5.84	69.78
Spotty	0	2.12	5.65	75.43
Speckled sole	1.59	0.02	5.55	80.98
Mottled triplefin	0.24	1.92	5.25	86.24
	BOI	**TAIR**		
Snapper	2.13	0.1	16.39	16.39
Parore	0.4	2.89	13.72	30.11
Yellow-eyed mullet	0.47	2.08	12.41	42.52
Garfish	1.17	0.1	8.76	51.28
Spotty	1.28	0.67	8.35	59.62
Estuarine triplefin	0.58	0.9	6.25	65.87
Red mullet	0.57	0.07	5	70.87

Average abundance by harbour

	BOI	KAIP	Contrib%	Cum.%
Exquisite goby	0.28	3.83	20.43	20.43
Snapper	2.13	1	8.86	29.29
Spotty	1.28	0.04	7.84	37.13
Garfish	1.17	0.9	7.47	44.6
Yellow-eyed mullet	0.47	1.16	7.36	51.97
Anchovy	0	0.87	7.17	59.14
Trevally	0.4	0.91	5.68	64.83
Speckled sole	0	1.01	5.65	70.48
Mottled triplefin	0.51	0.68	5.11	75.59
	BOI	**KAWH**		
Yellow-eyed mullet	0.47	7.1	32.83	32.83
Snapper	2.13	0.19	9.09	41.92
Sand flounder	0	1.77	7.58	49.51
Speckled sole	0	1.59	7.31	56.82
Garfish	1.17	1.02	6.77	63.59
Spotty	1.28	0	5.87	69.46
	KAIP	**KAWH**		
Yellow-eyed mullet	1.16	7.1	33	33
Exquisite goby	3.83	1.29	12.72	45.73
Sand flounder	0.42	1.77	7.56	53.29
Garfish	0.9	1.02	7	60.29
Speckled sole	1.01	1.59	6.34	66.63
Anchovy	0.87	0	5.39	72.02
Snapper	1	0.19	5.02	77.04
	KAIP	**TAIR**		
Exquisite goby	3.83	0.22	20.13	20.13
Yellow-eyed mullet	1.16	2.08	11.87	32
Parore	0	2.89	11.78	43.78
Anchovy	0.87	0	7.1	50.88
Snapper	1	0.1	6.66	57.54
Garfish	0.9	0.1	5.82	63.36
Trevally	0.91	0.33	5.43	68.79
Speckled sole	1.01	0.39	5.33	74.12
	KAWH	**TAIR**		
Yellow-eyed mullet	7.1	2.08	32.61	32.61
Parore	0	2.89	11.69	44.3
Sand flounder	1.77	0.3	8.2	52.5
Speckled sole	1.59	0.39	6.7	59.2
Exquisite goby	1.29	0.22	6.52	65.72
Garfish	1.02	0.1	5	70.72

RNGU=Rangaunu, MHRI=Mahurangi MANU=Manukau TAI=Tairua
BOI= Urupukapuka Island KAIP=Kaipara KAW=Kawhia

Table 4.8 R- and p-values of ANOSIM comparisons between fish assemblages of the four harbours sampled for mangroves. Non significant pair-wise comparisons are given in bold.

Harbour				
All Harbours 0.682 (Global R) p < 0.001				
	Rangaunu	Mahurangi	Kaipara	Manukau
Rangaunu				
Mahurangi	0.481 0.1			
Kaipara	0.926 0.1	0.667 0.1		
Manukau	0.704 0.1	0.889 0.1	0.556 0.1	

Table 4.9 Results of SIMPER analysis for mangroves, showing the average abundance of important discriminating fish species and their percent contribution to the average dissimilarity between harbours.

Average abundance by harbour

Species	**RNGU**	**MHRI**	**Contrib%**	**Cum.%**
Yellow-eyed m	5.82	14.86	26.05	26.05
Pilchard	0	5.6	14.44	40.49
Estuarine triple	5.95	5.49	9.77	50.26
Sand flounder	1	3.56	9.76	60.02
Smelt	2.47	3.76	8.22	68.24
Short-finned e	0.33	3.15	8.2	76.44
	RGNU	**KAIP**	**Contrib%**	**Cum.%**
Estuarine triple	5.95	0.33	18.95	18.95
Yellow-eyed m	5.82	11.13	17.04	35.99
Anchovy	0	4.68	13.37	49.35
Grey mullet	0.67	4.95	13.26	62.62
Short-finned e	0.33	4	12.07	74.68
Smelt	2.47	0	8.08	82.77
Exquisite goby	2.2	0.47	5.97	88.74
	MHRI	**KAIP**	**Contrib%**	**Cum.%**
Estuarine triple	5.49	0.33	14.56	14.56
Pilchard	5.6	0	13.69	28.25
Yellow-eyed m	14.86	11.13	12.18	40.43
Anchovy	0.82	4.68	11.44	51.87
Grey mullet	0.67	4.95	11.34	63.21
Smelt	3.76	0	10.4	73.61
Sand flounder	3.56	1.22	8.84	82.45
	RGNU	**MANU**	**Contrib%**	**Cum.%**
Grey mullet	0.67	24.88	51.09	51.09
Estuarine triple	5.95	1.67	10.39	61.49
Yellow-eyed m	5.82	8.62	8.48	69.96
Smelt	2.47	2.76	6.47	76.43
	MHRI	**MANU**		
Grey mullet	0.67	24.88	41.06	41.06
Yellow-eyed m	14.86	8.62	12.45	53.51
Pilchard	5.6	0.94	10.11	63.62
Estuarine triple	5.49	1.67	6.91	70.54
Sand flounder	3.56	1.05	6.28	76.81
	KAIP	**MANU**	**Contrib%**	**Cum.%**
Grey mullet	4.95	24.88	44.32	44.32
Anchovy	4.68	0.47	10.67	54.99
Yellow-eyed m	11.13	8.62	9.85	64.84
Smelt	0	2.76	7.83	72.67
Short-finned e	4	1.55	6.58	79.24

RNGU=Rangaunu	MHRI=Mahurangi
KAIP=Kaipara	MANU=Manukau.

Table 4.10 R- and p-values of ANOSIM comparisons between fish assemblages for three habitats sampled within Rangaunu Harbour (A), seagrass/sand only (B).

A. Rangaunu: all habitats

Habitat	R-value	Significance level (p)
Seagrass/Sand	0.987	0.001
Seagrass/Mangrove	0.991	0.001
Mangrove/Sand	1	0.1

B. Rangaunu: Seagrass, Sand only

Site	R-value	Significance level (p)
Habitat (seagrass/sand)	0.846	0.001
Intertidal/subtidal (seagrass only)	0.227	0.001
Upper/Lower harbour (seagrass only)	0.018	0.43

Table 4.11 Results of SIMPER analysis for the average abundance of important discriminating fish species for Rangaunu Harbour and the percent contribution to the average dissimilarity between all habitats (A), seagrass/sand only (B); tidal position - seagrass only (C).

A. Rangaunu all habitats

Average abundance by habitat

Species	**SAND**	**SEAGRASS**	**Contrib%**	**Cum.%**
Snapper	0	1	14.27	14.27
Garfish	0	0.94	13.37	27.63
Mottled triplefin	0	0.88	12.39	40.02
Spotty	0	0.75	10.01	50.03
Exquisite goby	0.33	0.94	9.4	59.43
Estuarine triplefin	0	0.63	8.42	67.85
Red mullet	0	0.5	6.57	74.42
Trevally	0	0.5	5.97	80.4
	SAND	**MANGROVE**		
Sand goby	1	0	13.37	13.37
Estuarine triplefin	0	1	13.37	26.74
Sand flounder	0	1	13.37	40.12
Yellow-eyed mullet	0	1	13.37	53.49
Grey mullet	0	0.67	9.88	63.36
Exquisite goby	0.33	1	9.27	72.64
Jack mackerel	0	0.67	8.43	81.07
Smooth pipefish	0	0.67	8.43	89.51
	SEAGRASS	**MANGROVE**		
Snapper	1	0	9.33	9.33
Sand flounder	0.06	1	8.81	18.14
Garfish	0.94	0	8.74	26.88
Sand goby	0.94	0	8.64	35.52
Mottled triplefin	0.88	0	8.16	43.69
Grey mullet	0	0.67	6.61	50.3
Yellow-eyed mullet	0.31	1	6.59	56.89
Smooth pipefish	0	0.67	6.03	62.92
Jack mackerel	0.19	0.67	5.56	68.48
Spotty	0.75	0.33	5.5	73.98

B. Habitat: Seagrass/Sand only

Average abundance by habitat

Species	**SAND**	**SG**	**Contrib%**	**Cum.%**
Snapper	0	4.86	19.69	19.69
Sand goby	1.32	5.15	18.43	38.11
Exquisite goby	0.82	3.5	13.65	51.77
Garfish	0	2.81	13.11	64.88
Spotty	0	2.52	10.02	74.9
Mottled triplefin	0	2.28	9.62	84.52
Yellow-eyed mullet	0	1.35	5.05	89.56

*SG=Seagrass

C. Intertidal/Subtidal: Seagrass only

Average abundance by tidal position

Species	**intertidal**	**subtidal**	**Contrib%**	**Cum.%**
Sand goby	11.06	3.18	27.47	27.47
Yellow-eyed mullet	4.54	0.29	14.77	42.24
Snapper	**1.53**	5.97	12.45	54.7
Spotty	0.49	3.19	8.58	63.27
Exquisite goby	4.77	3.08	8.38	71.66
Garfish	0.94	3.43	8.28	79.94
Mottled triplefin	0.88	2.74	6.52	86.46

Table 4.12 R- and p-values of ANOSIM comparisons between fish assemblages for three habitats sampled within Kaipara Harbour (A), seagrass/sand only (B).

A. Kaipara: all habitats

Habitat	**R-value**	**Significance level (p)**
Seagrass/Sand	0.309	0.048
Seagrass/Mangrove	0.571	0.012
Mangrove/Sand	**1**	**0.29**

B. Kaipara: Seagrass, Sand only

Site	**R-value**	**Significance level (p)**
Habitat (seagrass/sand)	0.397	0.016
Intertidal/subtidal (seagrass only)	1	0.02

Table 4.13 Results of SIMPER analysis for the average abundance of important discriminating fish species for Kaipara Harbour and the percent contribution to the average dissimilarity between all habitats (A), seagrass/sand only (B); tidal position - seagrass only(C).

Average abundance by habitat

Species	SAND	SEAGRASS	Contrib%	Cum.%
Anchovy	0	1	14.03	14.03
Trevally	0.88	0.25	9.56	23.59
Mottled triplefin	0.63	0.25	7.87	31.46
Speckled sole	0.63	0.5	7.1	38.56
Garfish	0.5	0.5	7.02	45.58
Sand goby	0.5	0	7.01	52.59
Sand flounder	0.5	0.25	6.8	59.39
Snapper	0.5	1	6.67	66.06
Yellow-eyed mullet	0.5	0	6.67	72.73
Kahawai	0.38	0.25	5.87	78.6
Yellow-belly flounder	0.25	0.25	5.28	83.88
Grey mullet	0.38	0	5.14	89.01
	SAND	**MANGROVE**		
Grey mullet	0	1	12.01	12.01
Short-finned eel	0	1	12.01	24.03
Snapper	1	0	12.01	36.04
Yellow-eyed mullet	0	1	12.01	48.06
Exquisite goby	1	0.33	7.78	55.84
Sand flounder	0.25	0.67	6.99	62.83
Yellow-belly flounder	0.25	0.67	6.78	69.61
Garfish	0.5	0	5.96	75.57
Speckled sole	0.5	0	5.46	81.03
	SEAGRASS	**MANGROVE**		
Short-finned eel	0	1	10.54	10.54
Trevally	0.88	0	8.95	19.49
Anchovy	0	0.67	6.85	26.34
Exquisite goby	1	0.33	6.85	33.2
Grey mullet	0.38	1	6.68	39.87
Mottled triplefin	0.63	0	6.6	46.48
Speckled sole	0.63	0	6.3	52.78
Yellow-belly flounder	0.25	0.67	6.17	58.95
Garfish	0.5	0	5.53	64.48
Snapper	0.5	0	5.53	70
Yellow-eyed mullet	0.5	1	5.53	75.53
Sand flounder	0.5	0.67	5.34	80.87
Sand goby	0.5	0	5.27	86.14

B. Habitat: Seagrass/Sand only

Average abundance by habitat

Species	SAND	SG	Contrib%	Cum.%
Anchovy	0	2.6	17.83	17.83
Exquisite goby	4.81	1.86	17.32	35.15
Yellow-eyed mullet	1.74	0	10.62	45.76
Trevally	1.3	0.12	9.04	54.81
Snapper	1.12	0.75	8.13	62.94
Garfish	1.12	0.46	7.86	70.8
Speckled sole	1.18	0.67	7.2	78
Mottled triplefin	0.91	0.24	6.41	84.41

C. Intertidal/subtidal: Seagrass only

Average abundance by habitat

Species	INT	SUBT	Contrib%	Cum.%
Exquisite goby	6.71	2.39	20.89	20.89
Yellow-eyed mullet	3.47	0	18.03	38.92
Speckled sole	2.02	0.5	8.23	47.15
Snapper	0	1.49	7.68	54.83
Anchovy	0	1.3	7.51	62.34
Garfish	0	1.35	6.7	69.04
Sand flounder	1.15	0.06	5.62	74.65
Trevally	0.79	0.97	5.53	80.18
Grey mullet	0.87	0	5.42	85.6

*SG=seagrass; INT=intertidal seagrass; SUBT=subtidal seagrass

Table 4.14 R- and p-values of ANOSIM comparisons between fish assemblages for mangrove and mud habitats sampled.

A. Manukau: both habitats

Habitat	R-value	Significance level (p)
Mangrove/Mud	0.832	0.006

Table 4.15 Results of SIMPER analysis for the average abundance of important discriminating fish species for Manukau Harbour and the percent contribution to the average dissimilarity between mud and mangrove habitats.

A. Manukau: all habitats

	Average abundance by habitat			
Species	**Mangrove**	**Mud**	**Contrib%**	**Cum.%**
Grey mullet	1	0	9.93	9.93
Smooth pipefish	1	0	9.93	19.86
Sprat	0	0.88	8.6	28.45
Speckled sole	0	0.88	8.26	36.71
Short-finned eel	1	0.25	7.08	43.8
Exquisite goby	0.33	0.88	6.42	50.22
Estuarine triplefin	0.67	0.13	6.18	56.4
Mottled triplefin	0.33	0.75	5.74	62.14
Estuarine stargazer	0	0.63	5.72	67.86
Anchovy	0.33	0.63	5.43	73.29

Table 4.16 R- and p-values of ANOSIM comparisons between prey assemblages of the four habitats sampled for sand flounder. *

Global R=0.362, p<0.001

Habitat	R-value	Significance level (p)
Seagrass, Sand	0.174	0.034
Seagrass, Mangrove	0.512	0.001
Seagrass, Mud	0.204	0.001
Sand, Mud	**-0.087**	**0.814**
Sand, Mangrove	0.706	0.001
Mangrove, Mud	0.329	0.001

Table 4.17 R- and p-values of ANOSIM comparisons between prey assemblages of the three habitats where speckled sole was collected. *

Global R=0.201, p<0.001

Habitat	R-value	Significance level (p)
Seagrass, Sand	0.50	0.05
Seagrass, Mud	**0.138**	**0.45**
Sand, Mud	0.191	0.001

Table 4.18 (A) R- and p-values of ANOSIM comparisons between prey assemblages of the four habitats where exquisite goby was collected. *

Global R=0.173, p<0.001

Habitat	R-value	Significance level (p)
Seagrass, Sand	0.219	0.001
Seagrass, Mangrove	0.194	0.001
Seagrass, Mud	0.237	0.001
Sand, Mud	**0.001**	**0.413**
Sand, Mangrove	0.167	0.001
Mangrove, Mud	0.182	0.001

Table 4.18(B) R- and p-values of ANOSIM comparisons between prey assemblages of the three habitats where exquisite goby was collected within Rangaunu Harbour. *

Global R=0.129, p<0.028

Habitat	R-value	Significance level (p)
Seagrass, Sand	0.188	0.01
Sand, Mangrove	**0.197**	**0.07**
Seagrass, Mangrove	**-0.167**	**1.00**

*Non significant pair-wise comparisons are given in bold.

Table 4.19 R- and p-values of ANOSIM comparisons between prey assemblages of the three habitats where snapper was collected.*

Global R=0.275, p<0.001

Habitat	R-value	Significance level (p)
Seagrass, Sand	0.265	0.001
Sand, Mud	0.249	0.03
Seagrass, Mud	0.305	0.008

Table 4.20 R- and p-values of ANOSIM comparisons between prey assemblages of the four harbours where spotty was collected.*

Global R=0.265, p<0.005

Harbour	Rangaunu	BOI	Tairua	Kaipara
Rangaunu				
BOI	0.222			
	0.016			
Tairua	**0.326**	**-0.075**		
	0.052	**0.585**		
Kaipara	**0.426**	0.359	**0.667**	
	0.154	0.002	**0.250**	

Table 4.21 R- and p-values of ANOSIM comparisons between prey assemblages of the four harbours where mottled triplefin was collected.*

Global R=0.191, p<0.001

Habitat	R-value	Significance level (p)
Seagrass, Sand	0.444	0.015
Seagrass, Mangroves	0.405	0.001
Seagrass, Mud	0.172	0.001
Sand, Mangroves	**0.021**	**0.355**
Sand, Mud	**-0.058**	**0.668**
Mangrove, Mud	**-0.026**	**0.664**

*Non significant pair-wise comparisons are given in bold.

Table 4.22 (A) Comparative studies of macroinvertebrate density and diversity.

Habitat	**Sieve size**	**Locality**	**Density (indiv. per m^{2})**	**Diversity (per core)**	**Source**
Mangrove	>0.5mm	Florida USA$^{\text{Tropical}}$	37,752	–	Sheridan (1997)
Mangrove	>0.5mm	Southern Kenya$^{\text{Tropical}}$	23,000	–	Schrijvers et al (1998)
Mangrove	>0.5mm	Brazil$^{\text{Temperate}}$	≤7250	17	Netto & Gallucci (2003)
Mangrove	>0.5mm	Manukau NZ	~6000	2.2 - 8	Morrisey (2002)
Mangrove	>0.5mm	Matapouri NZ	2215	~2^{c}	Alfaro (2006)
Mangrove	>0.5mm	Whitford NZ	~1600	8.13	Ellis et al (2004)
Mangrove	1-5.6mm	Northern NZ	2641	6	Present study
Mangrove	1-5.6mm	Manukau NZ	4478	6.8	Present study
Mudflats	1-5.6mm	Manukau NZ	≤ 947	9	Present study
Mudflats	>0.5mm	Northern NZ	~1000	n/a	Cowles et al (2009)
Sand	>0.5mm	Matapouri NZ	8382	~5^{c}	Alfaro (2006)
Sand	>1.0 mm	Western Port, Australia	4023	–	Edgar et al (1994)
Sand	1-5.6mm	Northern NZ	2529	10.7	Present study
Seagrass $^{\text{int}}$	>1.0 mm	Whangapoua NZ$^{\text{upper}}$	3500	8**	van Houte-Howes et al (2004)†
Seagrass $^{\text{int}}$	>1.0 mm	Whangapoua NZ$^{\text{lower}}$	5600	10**	van Houte-Howes et al (2004)†
Seagrass $^{\text{int}}$	>1.0 mm	Wharekawa	8000	11	van Houte-Howes et al (2004)†
Seagrass $^{\text{int}}$	>0.5mm	Matapouri NZ	13906	~20^{c}	Alfaro (2006)
Seagrass $^{\text{int}}$	1-5.6mm	Northern NZ $^{\text{all Harbours}}$	5899	13	Present study
Seagrass $^{\text{sub}}$	1-5.6mm	Northern NZ $^{\text{all Harbours}}$	12739	31.6	Present study
Seagrass $^{\text{sub}}$	1-5.6mm	Northern NZ $^{\text{East}}$	16716	36.7	Present study
Seagrass $^{\text{sub}}$	1-5.6mm	Northern NZ $^{\text{West}}$	4784	21.2	Present study
Seagrass $^{\text{sub}}$	1-5.6mm	Urupukapuka Is NZ	16386	31.5	Present study
Seagrass $^{\text{sub}}$	>1.0 mm	Slipper Island NZ^{a}	29000	33*	Schwarz et al (2006)†
Seagrass $^{\text{sub}}$	>1.0 mm	Mercury Island NZ^{b}	7300	17*	Schwarz et al (2006)†

*=10cm cores; **=13cm cores; ^{a}=blade length 29cm, ^{b}= blade length 7.8cm, ^{c}=15 x 15cm quadrat
† Schwarz et al (2006); van Houte-Howes et al (2004) data are for within seagrass beds only & averages of all samples

Table 4.22 (B) Comparative studies of biomass and estimated productivity of invertebrates >0.5mm sieve size for shallow coastal habitats. Adapted from Taylor (1998).

Habitat	Sieve size	Locality	Depth	Production (g AFDW m^2 yr^{-1})	Source
Turf coralline flats	0.5 -8mm	Northern NZ	Subtidal	100	Taylor (1998)
Coralline turf	0.5 -8mm	Northern NZ	Subtidal	~146	Cowles et al (2009)
Seagrass[1]	>1.4mm	Cape Cod. USA	Intertidal	23 - 139.1	Heck et al (1995)
Seagrass[3]	>0.5mm	Portugal	Intertidal	58	Sprung (1994)[a]
Seagrass[3]	>0.5mm	Portugal	Intertidal	58	Sprung (1994)[a]
Seagrass sediments[2]	0.5 -8mm	Northern NZ	Intertidal	3.65	Cowles et al (2009)
Seagrass blades[2]	0.5 -8mm	Northern NZ	Intertidal	1.82	Cowles et al (2009)
Seagrass[2]	1-5.6mm	Northern NZ	Intertidal	25.7 -59.3	Present study
Seagrass[2]	1-5.6mm	Northern NZ	Subtidal	42.8 - 113.3	Present study
Sand	1-5.6mm	Northern NZ	Intertidal	2.6 - 26.1	Present study
Sand	1-5.6mm	Northern NZ	Subtidal	17.0 - 24.0	Present study
Sand	>0.5mm	Portugal	Intertidal	34	Sprung (1994)[a]
Sand (fine)	>0.5mm	Wales	Subtidal	26	Warwick et al (1978)[a]
Sand	>1.4mm	Cape Cod. USA	Intertidal	5.5 - 9.1	Heck et al (1995)
Mud (Autumn)	1-5.6mm	Northwest NZ	Intertidal	1 - 1.9	Present study
Mud (Spring)	1-5.6mm	Northwest NZ	Intertidal	3.9 - 8.5	Present study
Mud	>1.4mm	Cape Cod. USA	Intertidal	6.5 - 10.6	Heck et al (1995)
Mud	>0.5mm	England	Intertidal	13	Warwick & Price (1975)[a]
Mud	>0.5mm	Portugal	Intertidal	72	Sprung (1994)[a]
Mud	>0.5mm	England	Subtidal	2	Buchanan & Warwick (1974)[a]
Mud	0.5 -8mm	Northern NZ	Intertidal	~7.3	Cowles et al (2009)
Mangroves	1-5.6mm	Northern NZ East	Intertidal	2.2 - 18.1	Present study
Mangroves	1-5.6mm	Northern NZ West	Intertidal	6 - 43.2†	Present study
Mangrove	0.5 -8mm	Northern NZ	Intertidal	~7.3	Cowles et al (2009)

Seagrass[1]= *Zostera marina*
Seagrass[2]= *Zostera muelleri*
† 43g value from one site in Manukau dominated by gastropod potomygyrus; otherwise > value=18.1
[a] references cited in Taylor (1998)

Table 4.23 Comparative studies on density (per 100m^2) of juvenile snapper collected in shallow coastal habitats of northern New Zealand.

Habitat	Depth	Species	Locality	Density per 100m^2	Source
Turf		Coralline algae/& sedimentary flats next to rocky reefs/algae	Leigh	~27	Kingett & Choat (1981)
				50	Choat & Kingett (1982)
Sponge Gardens		e.g. *Polymastia granulosa, Aaptos aaptos Raspailia topsenti, Axinella nsp, Cinachyra nsp*	Leigh	4.6	Battershill (1987)
Reef/Sand interface			Leigh	5.6 - 10.4	Ross et al (2007)
Seagrass $^{\text{subtidal}}$		*Zostera muelleri*	Slipper Island, Coromandel	40	Schwarz et al (2006)
Seagrass $^{\text{subtidal}}$		*Zostera muelleri*	Rangaunu $^{\text{upper}}$	160.00 473.37†	Present study
Seagrass$^{\text{intertidal}}$		*Zostera muelleri*	Rangaunu $^{\text{lower}}$	2.5	Present study
Sand			Kaipara; Kawhia*	0.4	Present study
Seagrass $^{\text{subtidal}}$		*Zostera muelleri*	Whangapoua	4.99	Morrison unpub data
Seagrass $^{\text{subtidal}}$		Artificial seagrass units	Whangapoua	670-1650	Morrison unpub data
Mussells		Artificial horse mussels *Atrina zelandica*	Mahurangi	40-120	Usmar (2009)

†= peak numbers from one haul, Rangaunu Upper subtidal
*=snapper only caught on sand in these two estuaries

Table 4.24 Proportional abundance (%) of different sieve size classes of amphipods collected in benthic samples for sand, intertidal and subtidal seagrass habitats from 5 harbours.

Habitat	Core sieve size (mm)					
	1	1.4	2	2.8	4	5.6
Subtidal seagrass	29.83	40.82	27.78	1.29	0.28	0
Intertidal seagrass	63.26	17.62	12.16	5.65	1.31	0
Sand	33.63	32.41	26.93	7.03	0	0

Chapter Five

General Discussion

‘Snapper inside Goat Island Marine Reserve’.

Chapter 5: General Discussion

The magnitude and rate of historical change within temperate nearshore habitats has dramatically accelerated over the past 150-300 years, with a resultant loss of species biomass, biodiversity and ecological resilience (Turner et al., 1999; Jackson, 2001; Jackson et al., 2001; Thrush et al., 2004; Lotze et al., 2006; 2011). Impacts from anthropogenic stressors continue to compromise water quality within estuaries, particularly within urban catchments, by the overloading of nutrients, increased sedimentation, pathogens and the accumulation of chemical contaminants (Kennish, 2002; Airoldi & Beck, 2007).

Despite a relatively short history of human settlement, New Zealand's estuaries have not escaped such analogous impacts (see reviews by Morrison et al., 2009; 2012; Parsons et al., 2009). During early settlement, terrigenous sedimentation was increased due mainly to deforestation, followed by strong increases in nitrogen and phosphate loadings in the 1950's due to intensification of agriculture (see review of Morrison et al., 2009). Available records point to substantial losses of important biogenic habitats such as seagrass meadows and mussel beds, especially in systems with large land catchments (e.g. Morton & Miller, 1973; Inglis 2003; Matheson et al., 2010).

Given the parlous state of coastal systems and the intricate linkages between estuarine and coastal environments, there has been increasing recognition of the need to move away from single-species fisheries management to ecosystem-based management, whereby the identification and protection of essential juvenile fish habitats is prioritized (França et al., 2012; Martinho et al., 2012). However, information pertaining to the relative importance of abiotic, and particularly biotic habitat variables at different spatial scales is limited for demersal fish species (Beck et al., 2001; 2003; Stål et al., 2007; Johnson et al., 2012). This is particularly evident for many demersal fish species within New Zealand (Morrison et al., 2012). Given the paucity of information, the objectives of this thesis were to assess what abiotic and biotic factors affect the distribution and abundance patterns of estuarine juvenile fish assemblages at different spatial and temporal scales, to identify important 'nursery' areas (Chapters Two & Four), and to determine the effect of environmental degradation (e.g. increasing sedimentation levels) on the foraging success and health of juvenile fish (Chapter Three).

What factors govern spatial variation in nursery value of estuarine juvenile fish habitats within New Zealand estuaries?

The distribution of juvenile fishes within estuaries and their use of particular areas results from responses of individuals to multiple environmental variables, which can be spatiotemporally dynamic (e.g. salinity, temperature, depth, currents) or more enduring (e.g. sediment type, biogenic habitat). To date, the majority (72%) of research on estuarine use patterns has focused on abiotic variables, at large spatial scales (>100km), with salinity, depth, and turbidity on a spatial scale, and temperature on a temporal scale regarded as the most important parameters governing species distribution (McLusky & Elliot, 2004; Akin et al., 2005 and references within; Francis et al., 2005, 2011; Selleslagh et al., 2009; Johnson et al., 2012). However, biotic processes, including food availability, predation and competition, have been also shown to influence spatiotemporal patterns of fish distribution (Ross & Epperly, 1985; Holbrook & Schmitt, 1989; Lankford & Targett, 1994) yet only comprise 28% of research to date and typically focus at smaller spatial scales (<100km) (Johnson et al., 2012). It is now acknowledged that the best predictors of habitat suitability are models combining abiotic variables with biotic ones, such as prey density. With most research focusing on individual habitats within single estuaries, this research is the first in New Zealand to consider all habitats encompassing (a) an environmental gradient within a single estuary, and (b) multiple habitats at a regional scale linking juvenile fish distribution to prey availability within multiple habitats.

Environmental gradients within estuaries

Abiotic (e.g. currents, suspended sediments; substrate type) and biotic (prey availability; predation) variables were examined concurrently for the entire fish assemblage along a 20km environmental gradient within the Manukau Harbour. Habitats ranged from sheltered, turbid intertidal muds to clearer, sandy areas with increasing biogenic structure. BIOENV and CCA analysis revealed that depth, current velocity and turbidity (i.e. suspended sediments and water clarity) were most strongly correlated with the fish assemblages. Other contributors retained in the CCA forward selection included mysid distribution, biogenic habitat and substrate type (proportion of sediment <64μm).

CCA analysis accounted for 72% and 55% of the variation in fish assemblage structure during autumn and spring respectively (first two axes), which compares favourably with recent research in the UK (e.g. Marshall & Elliot, 1998; Selleslagh et al., 2009;), Portugal (Martinho et al., 2007) and the USA (Akin et al., 2003; Martino & Able, 2003), where the proportion of explained variation of species data was generally lower (~18-47%). The unexplained variance in this survey reflects the complexity of the system, with factors interacting in a non-linear fashion (Perillo et al., 2009).

The juvenile fish assemblage within the Manukau Harbour showed strong temporal and spatial variability along the 20km estuarine gradient, with overall numbers declining from the sheltered, muddy, areas in the upper Pahurehure Inlet to the clearer, sandy areas of the lower harbour. Seasonal variations reflected differences in the times of recruitment of the more abundant species. There was a strong recruitment pulse in the spring, with 88% of the total catch collected within Pahurehure Inlet, particularly within the sheltered, shallow intertidal/bank areas, showing its high importance as a nursery area for some species (e.g. flounder).

A gradual transition was recorded for the dominant 12 species down the environmental gradient. Enclosed upper areas were dominated by exquisite goby, mottled triplefin, anchovy and yellow-belly flounders. A different assemblage consisting of sand flounder, speckled sole, snapper, gurnard and jack mackerel dominated bank and channel habitats of the mid to lower estuary, associated with clearer water and greater depths. Larger predatory fish (e.g. kahawai, snapper, trevally and shark species) were largely restricted to deeper waters at the seaward end of the environmental gradient, and were similarly associated with greater biogenic structure and higher water clarity. Dietary composition shifted seasonally in response to changes in food source availability, with a significant increase in consumption of mysids (contributing 66% of total dietary biomass) in spring, with their spatial patterns matching those of the spring recruits. Diversity of juvenile fish and dietary breadth were highest at mid-harbour sites, which also recorded the largest amounts of biogenic bycatch. Invertebrate prey diversity (from benthic grab samples) was also positively correlated with increasing habitat complexity.

Observations in this estuary suggest that broad scale (10-20km) patterns were primarily related to individual species responses to their physical environment and presumably large scale seasonal migrations of larvae into the estuary, whilst smaller scale patterns (1km) appear to be the result of habitat selection, foraging, and/or predator avoidance. These results are consistent with theoretical views on community structure whereby dominant abiotic variables are thought to structure the community like a ‘physiological sieve’ in more dynamic systems such as the Manukau, with biotic interactions refining distributions within this structure (Menge & Olsen, 1990; Kupschus & Tremain, 2001; Martino & Able, 2003; Akin et al., 2005). Conversely, in more benign systems, biotic factors are thought to predominate (e.g. Rangaunu Harbour, Chapter Three), (Menge & Sutherlands, 1987).

How important are biogenic habitats to juvenile fish within New Zealand estuaries?

The important role of biogenic habitat for juvenile fishes has been well documented internationally with habitats such as seagrass meadows (Heck et al., 1997, 2003; Beck et al., 2003; Gillanders, 2006), and mangroves (see review of Faunce & Serafy, 2006) considered crucial juvenile nurseries, particularly in tropical and subtropical regions.

Assemblages of small fishes were quantified across 7 northern estuaries, within which four habitat types were sampled: sand, intertidal muds, seagrass meadows and mangrove forests along with concurrent benthic sampling to assess prey availability. Fish diversity was comparatively modest, as expected from New Zealand's geographic isolation from diverse continents and temperate climate zone (Francis et al., 2011). Overall, densities were comparable between subtidal seagrass and intertidal mudflats, followed by intertidal seagrass and sand. Densities within mangrove forests, although not directly comparable due to the different sampling method used there, appeared to be lower. Different habitats supported different species assemblages of fish and invertebrates, which varied with latitude, geographical setting (east/west), and between and within estuaries (position within estuary). A small number of species such as yellow-eyed mullet and exquisite goby had more ubiquitous distributions. In contrast to tropical and subtropical studies (Beck et al., 2003), benthic faunal diversity and abundance were most often greatest in seagrass habitat (particularly subtidal), as was macro invertebrate biomass and estimated production, followed by sand, mangroves and mud habitats.

Seagrass meadows, particularly subtidal ones, supported a more diverse and abundant juvenile fish assemblage than the other habitats, including high numbers of several species that are commercially important (e.g. snapper and trevally). This supports the paradigm that seagrasses are an important nursery habitat in this region. There was a close association between the abundance of fish and the productivity of macroinvertebrates, particularly epifaunal crustaceans (viz gammarid amphipods), and their main prey species. In contrast, sand and intertidal seagrass habitats supported much lower densities (e.g. 20-fold decline from subtidal seagrass to sand) and diversity of juvenile fish with small cryptic species such as gobies and/or schooling species including yellow-eyed mullet and anchovies predominating. Given the lack of habitat cover (sand) and/or sparse, shorter blade lengths, concomitant with lower benthic faunal density, biomass and productivity, and temporal tidal availability, these habitats could be considered of lesser value to juvenile fish.

The substantial expansion of New Zealand's mangrove forests over the past 50 years in response to increasing sedimentation has led to intense societal debate as to their ecological value (e.g. Mahurangi and Whangamata proposed removals). Until recently, research from tropical and subtropical systems was uncritically and inappropriately transferred directly across to temperate mangrove systems, without any empirical surveys (Morrisey et al., 2007, 2010; Morrison unpubl. data). Value of mangroves for fish species is usually explained in terms of refugia from predation amongst prop roots/pneumatophores (Laegsgaard & Johnson, 2001; Nagelkerken et al., 2000a) and increased foraging opportunities (e.g. Sheridan, 1997). Contrary to tropical and subtropical systems (see review by Faunce & Serafy, 2006), this survey found benthic invertebrate fauna (predominantly infaunal) within mangroves to be modest in terms of abundance, species diversity, biomass and productivity, providing only limited foraging opportunities for small fish when compared to other estuarine habitats (e.g. seagrass). This result is consistent with other recent research in Australian temperate mangroves (Clynick & Chapman, 2002, Hindell & Jenkins, 2005, Smith & Hindell, 2005). Fish species captured in high abundances within mangroves were predominantly small, semi-pelagic schooling species (e.g. grey mullet, yellow-eyed mullet, anchovy), which either fed on food resources present throughout the estuary (i.e. zooplankton) or targeted food types common in mangrove forests and the associated seafloor (i.e. fine algae/detritus). Moreover, provision of refugia for small fish within New Zealand mangrove forests may be limited due to *Avicennia marina*'s structurally less complex aerial roots (compared to complex buttress roots of tropical species) and exposure to air at low tide. These may be strong contributing factors to the relatively low value of mangrove forests to most fish species in northern New Zealand (Morrisey et al, 2007, 2010).

Historically, intertidal mudflats have been ecologically undervalued within New Zealand, and have been particularly vulnerable to anthropogenic modifications such as dredging and reclamation (Meng & Powell, 1999; Saunders, 1999; Kanou et al., 2002). The extensive intertidal mudflats of Pahurehure Inlet supported a diverse and abundant assemblage of demersal fish species dominated by demersal species such as flounder. Given fish densities, the abundance, biomass and productivity of the benthos were surprisingly low (although densities increased in spring, viz. mysid species, their main prey item; see Chapter Two), with results comparable to some mangrove and intertidal sand sites. However, sampling biases were evident with some prey items not being well represented in the grab samples (e.g. cumaceans). Additionally, fish may be feeding in the deep low tide channels/banks prior to migrating (up to 1000m; Morrison et al., 2002) to the shoreward margins of these extensive tidal flats.

Landscape and environmental context

Results support increasing evidence that usage of different habitats by juvenile fish is dependent upon habitat quality and environmental context, and that gross physical attributes of habitats (e.g. structure *per se*) may not always be of predictive value in fisheries ecology (Skilleter et al., 2005; Jenkins et al., 2011). For example, while Urupukapuka Island had the longest seagrass blade lengths and highest water clarities, it returned low numbers of juvenile fishes. Closer proximity to reefs and greater densities of larger predators with depth may mean that offshore beds confer less benefits for predation-prone new recruits (Kimerei et al., 2011; Kimirei, 2012). A New Zealand-wide survey revealed clear latitudinal changes in fish species abundance and composition within seagrass meadows from the North to the South Island (Morrison et al., 2007; 2012).

Seagrass landscape attributes such as bed fragmentation, continuity of cover, size and shape, along with the structure of the plants themselves (i.e. biomass, density, blade length) display strong relationships to the physical setting of an area (Mills & Berkenbusch, 2009; Connolly & Hindell, 2006; Turner et al., 1999a).

Environmental conditions varied strongly between coasts, and this was reflected in declining densities for both fish and macro invertebrate biomass and productivity from east to west coasts for seagrass meadows. Conversely, this trend, although less pronounced was reversed for mangrove-associated fish and benthic fauna. West coast sites were characterized by higher suspended sediments and associated turbidities, higher silt/clay loadings and a greater degree of exposure to wind and waves, leading to less optimal growing conditions for seagrass (Gibbs et al., 2012; Morrison et al., in prep; Vant pers. comm.). Conversely, east coast sites were generally more sheltered, with fine sands predominating and greater water clarities, more conducive to seagrass growth at depth, with seagrass meadows directly abutting the mangrove forests within Rangaunu Harbour. Interestingly, species associations within mangrove forests were also correlated with degree of water clarity for species such as parore on the east coast, and grey mullet (associated with higher turbidities) on the west coast only (Morrisey et al., 2007; 2010). Moreover, subtidal seagrass beds on the exposed, west coast (e.g. Kaipara) would be of disproportionately higher value to species such as snapper and trevally, given the paucity of alternative sheltered recruitment areas along this high-energy coastline. Since ~80% of the west coast snapper stock is estimated to originate from within the Kaipara Harbour (Morrison et al., in review), and juveniles of this species' are strongly associated with subtidal seagrass/horse mussels within this harbour (Chapter Four; author unpubl. data), these biogenic habitats are highly valuable, and deteriorating water quality within this harbour is likely to have deleterious consequences for recruitment to the fishery.

Diet

Knowledge of the foraging ecology of juvenile fishes and factors affecting the acquisition and assimilation of prey is a fundamental prerequisite for sound management and conservation of fish species and their habitats. Yet information pertaining to juvenile estuarine (and marine) species within New Zealand is extremely limited in both spatial extent and the proportion of the assemblage considered (e.g. Capone, 2008; Mutoro, 2001; Jiang & Carbines, 2002; Choat & Kingett, 1981; Saunders, 1999; Usmar, 2009).

Ontogenetic dietary shifts occurred in the majority of species surveyed in this study, with the majority (63%) of fishes preying on meiofaunal crustaceans 0.5-1mm in length. Most marine fish are planktivorous at the onset of exogenous feeding, with limited abilities to detect, capture and digest prey (Nunn et al., 2011). Results in this study concur, with zooplankton dominating the diets of new recruits (20-30mm FL). This was particularly so for seagrass-associated species, which later transition to larger crustaceans such as mysids, gammaridean amphipods and caridean shrimps and crabs (Chapter Two & Four). Dominance of key prey items varied seasonally (viz mysids in spring; amphipods in late summer/early autumn), reflecting opportunistic and flexible temporal foraging strategies enabling juvenile fish to cope with their dynamic environment (Burke, 1995; Hosten & Mees, 1999; Freyer et al., 2003; Akin & Winemiller, 2006). Results emphasize the importance of both mysids and amphipods in the estuarine food web, playing a key role in energy transfer between the benthic and pelagic systems. Further, given the dependence of larval and newly-settled juvenile fish upon zooplankton, further research is warranted into the demography and dynamics of these three key trophic groups, particularly given their sensitivity to changing environmental conditions (Chapter Three).

Overall, stomach content (prey biomass) varied with habitat and tended to reflect the overall relative abundance of those prey in the environment. For example, endobenthic prey such as polychaetes, bivalves/siphons, cumaceans and infaunal amphipods characterized prey for fish collected mainly over intertidal mudflats/sand in late summer. In contrast, diets of those species highly associated with seagrass (e.g. snapper, trevally) were dominated by mobile epibenthic prey such as gammaridean amphipods, mysids and decapods and is consistent with prior research (e.g. Edgar & Shaw, 1995a,c; Horonouchi & Sano, 2000; Jenkins et al., 2011; Nunn et al., 2011). The more cosmopolitan species (e.g. yellow-eyed mullet) showed no marked dietary differences between habitats, which would be expected given their pelagic feeding strategy (Bloomfield & Gillanders, 2005).

Dietary breadth reflected benthic biodiversity, with higher prey diversities recorded from those fish species occupying more structurally complex biogenic structure, viz subtidal seagrass (e.g. snapper, 51 taxa) versus sand (sand goby, 5 taxa). This was especially pronounced for the pristine northeastern harbours with high water clarity and longer blade lengths, providing greater surface area for foraging invertebrates and/or refuge (e.g. Rangaunu Harbour) and is supportive of prior research in New Zealand (Jiang & Carbines, 2002). These results support the concept that structurally complex habitats enhance diversity, size ranges and abundance of prey. This is thought to enable segregation of species and minimise the potential for competition/ predation (Nunn et al., 2011). However, given the high degree of dietary overlap and extremely high densities of fish present, particularly in subtidal seagrass (viz. 0+ snapper), combined with seasonal declines of both amphipod densities and seagrass biomass in late summer (Choat & Kingett, 1982; Turner & Schwarz 2006a,b), most of the production of juvenile epifaunal crustaceans (63% of epibenthic crustaceans consumed measured 0.5-0.71mm in this survey) may in fact be cropped by fishes, leading to competition for food as documented in Western Port, Australia (Edgar & Shaw, 1995c). Indeed, observations of 0+ snapper defending territories against conspecifics in coralline turf suggests juvenile fish can and do competitively exclude conspecifics from prime sites (Kingett & Choat, 1981; Yamaoka et al., 1991, cited in Francis, 1995). Results support the contention that feeding success plays a key role in determining habitat utilization by new recruits (Jenkins et al., 2011).

Implications for future research

While the spatial extent of this multi habitat survey was broad, encompassing latitudinal (551km), coastal and estuarine differences, sampling was only undertaken on two occasions

(Chapters Two & Four) and therefore represents an initial ‘snapshot’ of the diet of the fish species present in late summer/spring. Nonetheless, the results offer an important insight into diets of different fish species, and the distribution of their prey within the four habitats (see Table 5.1). Identifying and understanding the relative contributions of putative nursery habitats to recruitment offshore ultimately requires information on spatio temporal variability in ontogenetic habitat use including not only density estimates, but growth and survival rates (Beck et al., 2003; Fodrie et al., 2009; Nunn et al., 2011). Ideally, this would encompass multiple habitats (including other biogenic habitats such as horse mussel beds/oyster reefs not surveyed in this study) utilized as ‘transition corridors’ by the species under investigation (Gillanders, 2006; Beck et al., 2003). Stable isotope analysis, in conjunction with traditional dietary analyses and estimates of prey availability (as undertaken in this study) along with nutritional condition indices (using RNA-DNA ratio analysis; Nunn et al., 2011) and otolith microchemistry (e.g. Gillanders & Kingsford, 2000; Gillanders, 2005; Morrison et al., in review) to quantify movement between juvenile and adult habitat could better identify key habitats of all life stages/species.

Impacts of changing environmental conditions

While high suspended sediments loads are a natural feature of many estuarine waters, the rate and extent of terrestrial sediment delivery has accelerated in recent decades (Thrush et al., 2004). Average sedimentation rates within New Zealand today are typically 10 times higher than before humans arrived (Swales et al., 2002, 2003). Elevated sedimentation and turbidity is now a characteristic of many New Zealand estuaries and is arguably the most important land-based stressor in New Zealand (Thrush et al., 2004; see review of Morrison et al., 2009). This includes both suspended sediment and deposition effects and associated decreases in water clarity. However, current knowledge on the effects of suspended sediments on fish is primarily based on freshwater (in particular salmon) species and largely based on acute exposure experiments (see reviews by Kerr, 1995; Moore, 1997; Bash et al., 2001; Wilber & Clarke, 2001). This is the first study within the New Zealand estuarine/marine context to assess the direct effects of stressors, particularly for juvenile estuarine fish, which are thought to be the most vulnerable to elevated levels of suspended sediments (Sigler et al., 1984; Wilber & Clarke, 2001; Nunn et al., 2011).

The impacts of changing environmental conditions (i.e. sedimentation; water clarity) on juvenile fish health was assessed using snapper (50-90mm FL) as a model species. Short-term tank experiments (Chapter Four) revealed a clear decrease in foraging success (i.e. number of mysids consumed) with increasing suspended sediment levels (TSS), which ranged from 10-160 NTU. Longer-term exposure (over one month) resulted in higher weight losses and mortality, and significant increases in gill deformation (epithelial hyperplasia, and fusion of the lamellae) associated with impaired respiratory function. Other sublethal effects included increased coughing and gulping at the surface, higher respiration rates and decreased activity, which are consistent with the effects of anoxia (Berg, 1982; Berg & Northcote, 1985; McLeay et al., 1987; Servizi & Martens, 1992).

Associated field survey results (7 northern estuaries) revealed significant differences among the biological variables measured for juvenile snapper (50-100mm FL), with varying degrees of anthropogenic disturbance. Juvenile snapper had significantly lower condition indices (i.e. an effective proxy for growth rates and nutritional status), in the more impacted estuaries characterized by increasing sedimentation, concomitant with lower water clarities and increasing urbanisation (e.g. Waitemata, Manukau). Rangaunu Harbour, the most ‘pristine’, had the highest average condition index. Higher levels of gill deformation (hyperplasia/fusion) and parasite loads were also recorded, particularly for the Manukau and Mahurangi Harbours. Results suggest that physiological stress in fishes in response to increased TSS can decrease immunological competence and growth.

Dietary analysis revealed a change in foraging tactics with increasing turbidity. Pelagic prey, particularly copepods (0.5-0.71mm), dominated the diet in higher water clarity estuaries, while in the more turbid estuaries (Manukau, Kaipara, Waitemata) diet was dominated by larger (≥ 0.71 mm) benthic prey (mysids/shrimps/amphipods). This suggests a change in feeding strategy from active (probably visual) selection of pelagic prey (zooplankton), to larger, slower-moving benthic prey. This type of behavioural shift in foraging tactics has been documented in other studies, often with a ‘turbidity’ threshold of around 40 NTU, above which declining visual acuity results in more opportunistic, ambush-type predation (e.g. Hecht & van der Lingen, 1992; Macia et al., 2003; De Robertis, 2004; Shoup & Wahl, 2009).

New fish recruits are largely restricted in their foraging abilities for larger prey by their smaller gape size, and limited assimilation abilities for larger prey due to their poorly differentiated guts. (Confer & Lake, 1987; Boubeé & Ward, 1997; Gillanders, 1997; Sudo & Azeta, 2001). Juvenile fish are known to actively select smaller, easily digestible prey (Mills et al., 1984; Lankford & Targett, 1997; Gning et al., 2009), such as planktonic calanoid and harpacticoid copepods due to their high caloric and protein content (35% higher than for amphipods) thus facilitating rapid somatic growth to escape size-dependent predation (Volk et al., 1984). Increasing TSS can thus potentially reduce the condition and growth of juvenile snapper by reducing the overall food supply available and perhaps relative nutritional values by reducing their ability to select optimal prey sizes, e.g. visually pick zooplankton, and/or a reduction in the actual zooplankton assemblages available due to changing environmental conditions. Reduced growth for juvenile snapper may translate into smaller-sized adults with attendant declines in fecundity at the population level (Gilbert, 2006; N. Davies; C. Walsh pers. comm.).

Higher suspended sediment concentrations have also been shown to control long-term temporal trends and lower the nutritional values for copepods and two mysid species in an 18-year study in France (David et al., 2005). Reduced feeding, lower growth rates and higher mortality of planktonic copepods have also been recorded with exposure to higher turbidity (Paffenhoffer, 1972; Hart, 1988; Koenings et al., 1990; Berry et al., 2003) and after storms (pers. obs.; Lowe, 1983). Similarly, suspended sediments have also been found to be acutely toxic to young-of-the-year amphipods (Forbes et al., 1981) while Schwarz et al., (2006) found decreased survival rates for the gammarid amphipod *Aora* sp. at high TSS – a key prey item for seagrass associated fish.

Loss of biogenic habitat

Deposition of suspended sediments can concurrently alter substrate composition through increased clay and silt content, deleteriously affecting the distribution of infaunal and epibenthic species (e.g. Edgar & Barrett, 2000; Norkko et al., 2002; Thrush et al., 2004; see review of Morrison et al., 2009). Available records point to substantial declines of important biogenic structure with increasing sedimentation such as horse mussels (Ellis et al., 2002; Norkko et al., 2002; Hewitt & Pilditch 2004; Lohrer et al., 2004, 2006a), sponges (Lohrer et al., 2006b), green-lipped mussels/oysters and seagrass meadows (Park, 1999; Inglis, 2003; Morrison et al., 2009, 2012; Matheson et al., 2010). Tauranga Harbour, for example, has lost an estimated 90% of its subtidal seagrass beds between 1959 and 1996 (Park, 1999). Seagrass meadows are regarded as sensitive indicators of water quality, being particularly vulnerable to the effects of increasing TSS and eutrophication, yet to date, little protection has been afforded the 44 km^2 of seagrass meadows remaining within New Zealand, with only two of 16 marine reserves containing seagrass. There is also a paucity of information concerning current spatial extent and lack of long-term baseline data required to distinguish natural variability from anthropogenic activities (Turner & Schwarz, 2004).

It has been argued that indirect effects of sedimentation, including the loss of these important biogenic habitats (which provide important transitional larval settlement areas, in addition to enhanced feeding and refuge for juvenile fish; see Chapter Four), may in some cases far outweigh the direct physiological effects such as those catalogued in this study and others for both marine (e.g. seagrass declines: Vanderklift & Jacoby, 2003: Pihl et al., 2006; Baden et al., 2012; see review of Morrison et al., 2012) and freshwater systems (Reynolds, 1988; Newcombe & Jensen, 1996; Rowe & Dean, 1998). With juvenile snapper known to defend small feeding territories from similar sized cohorts (Kingett & Choat, 1981; Yamaoka 1991 cited in Francis 1997), reductions in the availability of preferred habitat may cause crowding into limited areas, and/or result in occupation of marginal habitats in terms of prey productivity. This may result in additional stress (e.g. susceptibility to disease) from increased intraspecific and interspecific competition (Coutant, 1985; cited in Adams, 2003).

Results from this study collectively suggest that increased suspended sediment levels in the northern New Zealand marine environment can have negative effects on individual fish and their fitness. Lower growth and nutritional status of juvenile snapper may potentially lead to increased vulnerability to predation, physiological stress and disease in addition to lower overwinter survival and subsequent recruitment to commercially exploitable stocks (Francis, 1994; Adams, 2003; Amara, 2007).

Collectively, these findings support the level of concern being shown over escalating point source inputs of both suspended sediments and concentrations of nutrients (phosphates/nitrates) from dairying/forestry into rivers and estuaries within northern New Zealand (Parkyn, 2002; ARC, 2010; Gibbs 2012), and highlight the need for holistic management to encompass both marine and terrestrial environments at regional scales (Edgar & Barrett, 2000) for effective protection and sustainability of juvenile fish habitats.

Gaps: long-term studies

Most temperate regions lack pristine reference sites that have not already undergone significant anthropogenic change (Edgar & Barrett, 2000; Jackson et al., 2001b; Lotze et al., 2006; Morrison et al., 2012). Additionally, there is a paucity of long-term, integrated monitoring at ecosystem levels to provide baseline historical data, against which to assess the cumulative impacts (often masked) of human-induced change (e.g. for seagrass meadows; Kennish, 2002). Whilst this study documented fish densities and habitat associations over a three year period, existing patterns of distribution may not necessarily inform us of the most preferred and/or optimal habitats for juvenile fish due to habitat fragmentation and loss (Beck et al., 2001; Levin & Stunz, 2005).

Within New Zealand, long-term studies on estuarine fauna are limited to benthic fauna within the Manukau Harbour (25 years, Hailes & Hewitt, 2012) and the Mahurangi Harbour (11.5 years, Cummings et al., 2005). Juvenile fish assemblages have fared worse, with only one study documenting changes over a 7 year period within the Manukau and Mahurangi Harbours (Morrison unpubl. data). Additionally, only limited baseline data exists for most marine reserves prior to establishment in New Zealand (Jennings et al., 2011, cited in Taylor et al., 2011). Nonetheless, recent research has highlighted the value of local ecological knowledge for providing information on decadal scale changes within marine systems (i.e. Poor Knights Islands, (Taylor et al., 2011); snapper distribution (Parsens et al., 2009); Kaipara Harbour, (author unpubl. data). Inclusion of longer time scales can thus provide information on the overall status and recovery of impacted systems and enhance the ability of predicting future change of the species or system under study (Johnson et al., 2012; Smale et al., 2012).

There is now an increasing focus on the use of fish (particularly marine migrants) as ‘biological indicators’ of environmental change (i.e. water quality evaluation) in estuarine waters (Mee et al., 2008; Courrat, 2009; Borja et al., 2010; Valesini et al., 2010b). While a comprehensive water quality monitoring network exists across 27 sites within the Auckland region, there is a dearth of information as to the effects of these physico-chemical conditions on fish health (Scarsbrook et al., 2008). Future long-term monitoring of a subset of ‘biological indicator’ fish species (e.g. snapper; flounder), and histological indicators utilized in this study (see Chapter Three) could be included for future monitoring of both pristine (Rangaunu) and impacted sites (e.g. Manukau) , to better link individual fish health and associated population outcomes to human-driven environmental changes.

Implications for conservation, restoration and management

Until now, the conservation tool of choice for protecting estuarine and marine habitats within New Zealand has been the designation of ‘marine reserves’. However, as noted by Agardy (2005), there is a mismatch of scale, with many of the marine reserves too small to be effective, and do not address the crucial links between land/freshwater and coastal systems. The problems facing estuarine management are far more persuasive and complex, involving multiple stressors that include habitat loss and an array of pollutants. Human impacts do not act in isolation, rather they are synergistic, cumulative and interconnected over large spatial scales (e.g. Estes et al., 1998; Babcock et al., 1999; Peterson et al., 2000; Baden et al., 2012).

While restoration of important habitats is an established method in terrestrial and freshwater systems, examples within marine systems are few (Turner et al., 1999a and references therein; e.g. Whangarei Harbour seagrass restoration, Matheson et al., 2009). Overseas results have also been equivocal as to the return of full ecological functioning of these habitats, even 3 years on from restoration (e.g. seagrass; Meyer et al., 1993; cited in Turner et al., 1999a). This emphasizes the need to protect and conserve the remaining habitats of significance.

Given the predicted increase in the magnitude and frequency of extreme weather events, including heavy rainfall, with concomitant flow-on effects on water quality and habitat degradation, juvenile fish in estuaries will face a raft of new challenges (Martinho, 2009). Results from this study highlight the inherent difficulties of determining essential juvenile fish-habitat associations, given the dynamic nature of estuaries, the complexity of factors that can create site specific variation, the range of scales (Edgar, 1995c) over which they vary, and the ecological requirements of the individual species in question, which themselves vary with ontogeny and ecological context (Johnson et al., 2012). However, defining these fish-habitat associations is a necessary prerequisite to sustainable ecosystem-based management and conservation of demersal fish stocks.

Ultimately, to maintain the ecological function of the estuarine ecosystems, the applicability and success of marine reserves and ecosystem-based management will be based on the social, political and biophysical environment of the ecosystem in question (Blaber, 2002). However, as sagely noted by Pomeroy (1995), the main focus of management should be people not fish per se.

Table 5.1 Summary of fish habitat associations, and diets, of estuarine New Zealand fish.

Family	Species	Common name	Life cycle	Habitat usage in northern estuaries	General distribution and estuarine habitat usage	Commercially valuable?	Dietary patterns
Mugilidae	*Aldrichetta forsteri*	Yellow–eyed mullet Aua	EM	MG**** SG/SD/I*	Common in shallow bays and estuarine waters around New Zealand. Large schools often enter river mouths and harbours to feed. They disperse along coastal beaches during cooler months. Coastal spawning occurs late winter to summer. Juveniles and sub adults (<15 cm) extremely abundant across most estuarine habitats excepting clear, deeper water areas close to estuary entrance.	Limited	Juvenile (30–90 mm) diet is strongly driven by zooplankton, especially the harpacticoid copepod *Euterpina acutifrons* and calanoid copepod *Corycaeus aucklandicus*, in addition to barnacle cyprids and mysid shrimps. Also feeds on plant material and detritus. Larger individuals (90–130 mm) feed predominantly on detritus and algae gulped up with mud or sand.
Mugilidae	*Mugil cephalus*	Grey mullet Kanae	MMD	MG *** SG/I* (west coast only)	Common in sheltered bays and estuaries in northern New Zealand, distributed as far south as Tasman Bay. School & migrate between fresh/estuarine water and the sea. Spawning occurs in mid summer out in coastal waters. High juvenile (20–40 mm) association with west coast mangroves. Lower abundances of same size class also found in association with inter-tidal sea-grass meadows – west coast only. Larger juveniles (50–150 mm) commonly sampled in shallow sub-tidal areas away from mangroves. Juvenile abundances largely coincide with estuarine mangrove's distributional extent. Adults are wide ranging, but are commonly caught around mangroves, and have been observed inside mangrove forests. Strong use of estuarine systems, also found in coastal surf zones. Known to move between west coast estuaries as adults (tagging).	High. Fully exploited stocks. Most of commercial fishery is west coast associated	Mainly detrital feeders. Some inter-harbour diet variability, with juveniles (25–30 mm) in the Kaipara Harbour consuming high numbers of the copepod *Euterpina acutifrons.*
Mugilidae	*Upeneichthys lineatus*	Goatfish Red mullet Ahuruhuru	EM	SG/ST* HM *	Found in warmer waters north of Cook Strait. Rare south of Hawke Bay and Farewell Spit. Juveniles are sometimes found in modest abundances in estuaries, in association with sub-tidal sea-grass and horse mussel beds. Adults are rare in estuaries. Structure associated.	Limited	Benthic. Feeds on amphipods (*Methalimendon* sp.; Parakalliope) and mysids (*Tenagomysis* sp.)
Tripterygiidae	*Grahimina capito*	Mottled triplefin	ER	SG *** MG * MD** †	Widespread in estuarine systems, (particularly on the west coast) intertidal pools and sheltered harbours. Usually associated with seafloor habitat structure. Also often found in high abundances in sub-tidal sea-grass areas. Patchily abundant in some mangrove forests, and associated shallow channels	No	Feeds mainly on small amphipods (e.g. *Parakalliope novaezelandiae*) and crabs (*Helice crassa, Halicarcinus whitei).* In mangrove forests, diet also includes isopods and midge larvae.

Tripterygiidae	*Grahamina nigripenne*	Estuarine triplefin	ER	MG ** SG/I *	Widespread in estuarine systems, especially on the east coast. Usually associated with seafloor habitat structure. Occasionally found in mangroves and intertidal seagrass.	No	Primarily benthic. Consumes amphipods such as *Parakalliope novaezelandiae*, *Corophium* sp and crabs (e.g. *Helice crassa*)
Gobiidae	*Favonigobius exquisitus*	Exquisite goby	ER	MD*** MG * SG/ST** †	Very abundant on open inter-tidal and sub-tidal flats, down to a depth of at least 19 metres. Modest abundances in mangroves.	No	Benthic and planktonic diet. Consumes amphipods along with the copepod *Euterpina acutifrons* progressing to crab species (*Helice crassa*) and other crustaceans including cumaceans and
Gobiidae	*Favonigobius lentiginosus*	Sand goby	ER	SG/I` **	Occurs on the west and east coasts of northern New Zealand. Moderate numbers found in Rangaunu Harbour seagrass.	No	Benthic feeder, consuming amphipods such as *Parakalliope* and *Methalimendon* sp.
Gobiidae	*Arenigobius bifrenatus*	Australian bridled goby	ER	MG *	Invasive. Mangrove associated – agrees with its habitat usage in Australia (endemic species). Found in relatively high abundance in Mahurangi Harbour mangroves as large adults. This species has been sampled from this harbour across multiple years, suggesting an established breeding population.	No	Detrital feeder. One juvenile specimen examined for gut contents.
Pleuronectidae	*Rhombosolea plebeia*	Sand flounder Patiki	MMD	MD**** SG/I*	Widespread, with very high juvenile abundances in estuarine systems, especially in shallow water mud-habitats, and sand to a lesser extent. Adults are relatively uncommon in estuarine systems, but common in shallow coastal areas. Tagging studies in the 1960s showed strong links between estuarine nursery grounds and coastal fisheries (Christchurch region – no mangroves). Modest numbers of juveniles and sub-adults in mangroves, along with occasional adults.	Yes, most common flounder caught in the South Island.	Consumes bivalve species such as mussels *(Perna canaliculus)* and pipi (*Paphies* sp.) in addition to crabs and a variety of other small crustaceans including cumaceans, mysids and amphipods. Larger individuals (>90 mm) also fed on brittle–stars (*Amphiuera rosea*) along with mud and detritus.
Pleuronectidae	*Rhombosolea leporina*	Yellow-bellied flounder	MMD	MD **** MG*	Widespread distribution, more common in the North Island. This species is strongly estuarine associated, and was also historically fished along open coast beaches. Juveniles are most abundant in upper harbour, turbid muddy habitats, and absent from sand-flats. Adults are more widely distributed than juveniles, including sand habitats, migrating offshore in winter. Especially common in west coast systems, which support relatively large fisheries. Found as both juveniles and sub-adults in mangroves, along with occasional adults, but abundances modest compared to other habitats	Yes, short-lived species (~ 3+ years), strong natural fluctuations in abundance.	Feeds mainly on mysids, nematodes and crab sp. (*Helice crassa; Halicarcinus sp.*) .Also consumes bivalves (*Paphies australis*), bivalve siphons, and amphipods.
Pleuronectidae	*Rhombosolea retiaria*	Black flounder	CA	Occasional in river	Strongly associated with riverine estuarine systems with high fresh-water inputs. Not yet sampled from	No	No stomach analysis.

				estuaries	northern estuarine systems.		
Pleuronectidae	*Peltorhamphus latus*	Speckled sole Patiki rore	MMD	MD ***	Widely distributed, occurring in bays and sheltered waters out to 100 m. Spawning occurs from late winter to spring inshore. Juveniles do not move more than half-way up tidal flats. A dominant species in many tidal-flat fish assemblages, especially in more muddy habitats. Adults are uncommon in estuaries.	No	Primarily benthic. Feeds mainly on small crustaceans including mysids and cumaceans. (*Colurostylis lemuran*). Also feeds on mussels, ostracods and the harpacticoid copepod, *Euterpina acutifrons.*
Clupeiidae	*Sardinops neopilchardus*	Pilchard	MMO	MG*	Widespread in inshore waters around New Zealand. Often encountered in open water habitats of west coast. Pelagic, often in surface schools, most common in spring and summer. One large catch in the mangrove study. Probably only extends into mangroves on the fringes of its main distribution in more open estuarine waters.	Limited, targeted as bait fishery (adult) in coastal waters	No stomach analysis.
Clupeiidae	*Engraulis australis*	Anchovy	MMO	MD ** MG *	Found in surface waters around the North Island and west of the South Island. Juveniles very common on estuarine tidal mud-flats and near the water surface in sub-tidal areas. Probably moves to coastal waters as approaches adult size. One large catch in mangrove study. Probably extends into mangroves on the fringes of its main distribution in more open estuarine waters.	A few bait fisheries	Planktivorous. Feeds mostly on calanoid copepods (e.g. *Paracalanus indicus*) and the harpacticoid copepod *Euterpina acutifrons*.
Clupeiidae	*Sprattus* spp.	Sprat	MMO	MD**	Common around the South Island and scattered around the North Island to Auckland. Form densely packed schools at depths of 50 m. Sampled as occasional juveniles (2 closely related species). Juveniles are sometimes relatively common on tidal mud-flats, and near the water surface in sub-tidal areas. Probably migrates to coastal waters as approaches adult size.	No	Mainly planktivorous (calanoid copepods) with some mysids consumed (only specimens from Manukau Harbour examined).
Retropinnidae	*Retropina retropina*	Smelt	AN	MG **	Common in freshwater systems, and in very high numbers in upper estuarine habitats (and estuaries) dominated by high freshwater inputs. Includes more southern estuaries without mangroves. Consistently found in mangrove habitats, in modest abundances. Adults return to freshwater to spawn.	Small fishery in the Waikato, taken with whitebait. Traditionally important species	Planktivorous. Predominantly consumes the harpacticoid copepod *Euterpina acutifrons.* Also feeds on midges (Chironomidae sp.).
Carangidae	*Trachurus* spp	Jack mackerels	MMO	MG * SG-ST ***	Widespread, spawning in groups in Spring/Summer. Forms schools midwater and spend sometime near bottom. Found across a range of estuarine habitats in relatively low numbers – strong association with sub-tidal sea-grass meadows in some estuaries.	Yes, large coastal trawl and purse seine fisheries	Diet strongly driven by zooplankton (calanoid copepods). Feeds mainly on mysids in seagrass meadows. Adults also consume small pelagic fish.

					Sampled as occasional juveniles in mangroves.		
Hemiramphidae	*Hyporhamphus ihi*	Garfish Piper/Half-beak Ihe	EM	SG *** MG *	Widespread and abundant in estuarine habitats with clearer waters, schooling just below the surface. Spawning occurs inshore during spring and summer. The demersal eggs sink to the bottom and adhere to seaweed/seagrass until hatching. Juveniles common within sea-grass meadows. Sampled as occasional adults in mangroves, not common in the turbid water conditions often associated with mangroves. Important as prey for larger fishes and seabirds.	Limited	Juvenile diet is predominantly planktonic copepods within seagrass meadows. Adults consume seagrass fragments in addition to algae and insects that land on the water surface.
Syngnathidae	*Stigmatopora macropterygia*	Smooth - pipefish	MMD	SG**	Generally found in association with sub-tidal seagrass, and around seaweed, in clearer water conditions with sandier substrates. Structure associated.	No	Predominantly planktonic. Consumes high numbers of the copepod *Paracalanus indicus*. Feeds on some benthic crustaceans such as mysids and amphipods.
Syngnathidae	*Stigmatopora nigra*	Black pipefish	MMD	SG **	Found in harbours, rock pools, reefs and open seabed usually associated with seaweed. High affinity with seagrass. Use their tails to anchor themselves to seagrass from which they ambush prey.	No	Planktonic diet. Feeds predominantly on calanoid copepods such as *Temora turbinata* with some amphipods and mysids.
Sparidae	*Pagrus auratus* (previously *Chrysophrys auratus)*	Snapper	MMO	SG-ST**** HM **	Abundant north of Cook Strait. Spawn in spring/early summer in large bays with nearby estuaries. Juveniles occupy sheltered habitats during their first summer with adults moving to deeper coastal waters in winter. Very high juvenile abundances associated with estuarine sub-tidal sea-grass meadows, horse mussels, sponge assemblages – strongly structure associated. Low juvenile abundances sampled in Rangaunu Harbour mangroves – at sites with very clear water, and adjacent seagrass meadows – may utilise mangroves to a spatially limited extent in relatively (rare) pristine environments. Historical records of large snapper actively feeding in large numbers in mangrove forests.	Yes, fully exploited fisheries	Newly recruited juveniles within seagrass meadows (15–40 mm) are predominantly planktivorous, feeding on calanoid copepods with greater numbers of mysids consumed with age. Larger individuals (50–100 mm) feed on crustaceans such as crabs, amphipods (*Aora typica*), shrimps (*Palaemon affinis*) and bivalves. Mysid spp. dominant in the diets of all juveniles associated with seafloor structure (horse mussels/sponges) in the Manukau Harbour. Adults consume a wide range of invertebrates including crabs, polychaetes, shellfish & small fish.
Labridae	*Notolabrus celidotus*	Spotties Paketi/Pakirkiri	MMD	SG/ST *** MG*	Abundant in shallow sheltered waters of harbours. Juveniles rare on the west coast, very common in east coast estuaries in association with structure, especially sub-tidal sea-grass meadows. Adults less common, thought to move off to other habitats. Very occasional juveniles in mangrove forests, east coast only. As with juvenile snapper, some evidence from Rangaunu Harbour that juvenile's of this species use mangroves to a limited extent where seagrass is adjacent (more pristine systems).	No	Primarily benthic. Feeds on amphipods, mysids and juvenile bivalves within seagrass meadows. Can feed on plankton over reefs.

Arripidae	*Arripus trutta*	Kahawai	MMO	SD	Present throughout NZ inshore coastal waters, more abundant north of Cook Strait. Pelagic, living from mid water to the surface. Juveniles are largely associated with exposed sloping beaches with coarser sediments, and clearer water conditions, both inside and outside estuaries. Juveniles are very rare in turbid, muddy upper harbour areas. Very occasional juveniles in mangrove forests. Adults have been caught along the edges of mangrove forests over tidal mud-flats, feeding on juvenile flounders.	Yes	Benthic. Juveniles (30–80mm) mainly feed on small mysid shrimps (*Tenagomysis* sp.), especially in the Manukau Harbour. Some calanoid copepods such as *Temora turbinata* consumed in clearer waters. Some of the larger juveniles (>60 mm) fed on the exquisite goby (*Favononigobius exquisitus*). Adult diet is largely planktonic crustaceans and small schooling fishes. Crabs, polychaetes and shellfish are also consumed.
Carangidae	*Pseudocaranx dentex (previously Caranx lutescens or C. georgianus)*	Trevally	MMO	SG-ST ** CH*	Common around North Island and northern half of the South Island to depths of 80 m. Adults school in large numbers on the bottom as well as in mid water. Juveniles especially common in west coast harbours, in association with sub-tidal sea-grass, and along the edges of channels with strong currents and clearer waters. Also found in association with reef-associated sponge assemblages, and in the Waikato River mouth.	Yes	Juveniles feed mainly on amphipods (Peradexamine sp.; *Parakalliope novaezelandiae*) and mysid shrimps. Consumes some zooplankton in sea-grass meadows. Adult diet ranges from planktonic crustaceans to benthic items such as crabs, brittlestars, heart urchins and bivalves.
Monocanthidae	*Parika scaber*	Leatherjacket	MMO	SG/ST** HM **	Widespread and common in rocky/weedy areas near shore to 30 m. Spawning is from winter to spring. Juveniles settle into weed beds during summer. Juveniles occasionally found in modest abundance in association with sub-tidal sea-grass and horse mussel beds, larger animals very rare. Structure associated.	Yes	Juveniles (30–66 mm) feed on benthic crustaceans such as amphipods and isopods. Adults graze on encrusting animals such as sponges and ascidians.
Trigilidae	*Chelidonichthys kumu*	Red gurnard Kumukumu	MMO	MF/ST *	Distributed throughout New Zealand to 180 m. Coastal spawners. Feelers used to scare prey into the open water. Juveniles sampled in low abundances from west coast estuaries, generally over muddier bottoms. Adults are seasonally common in some estuaries such as the Manukau and Kaipara harbours.	Yes	Juveniles (28–80 mm) feed mainly on mysids and cumaceans. Larger juveniles (>80 mm) also feed on crabs and small fish e.g., the exquisite goby (*Favononigobius exquisitus*).
Kyphosidae	*Girella tricuspidata*	Parore	MMD	SG/ST *** MG **	Largely a north-eastern New Zealand species, main adult distribution spatially coincides with mangrove estuary distributional extent. Adults are common across both soft sediment and rocky reef systems and utilise mangrove forests, in small schools.	Very limited	Predominantly a benthic feeder. Diet includes amphipod species such as *Parakalliope novaezelandiae*, and barnacle cirri in mangrove forests. Smaller individuals (20–30 mm) also feed on zooplankton species such as the copepods

					On the east coast, high juvenile abundances are often associated with sub-tidal sea-grass, some Neptune's Necklace (*Hormosira banksii*) beds, brown kelp (*Carophyllum*) forests and man-made structures e.g., floating pontoons. Abundance in mangrove forests is lower, but the large extent of mangroves, relative to other habitats, makes it likely to be an important juvenile nursery. Given their preference for higher water clarities, juveniles were probably concentrated in the pneumatophore zone/forest edge zone. This species uses a series of inter-connected habitats as it grows in size/age.		*Paracalanus indicus*, and *Temora turbinata).* Adults are predominantly herbivorous consuming algae.
Anguillidae	*Anguilla australis*	Short-finned eel	CA	MG **	Juveniles and sub-adults common in more structurally complex mangrove habitats. Only encountered during night sampling. Rarely sampled in other estuarine habitats, and usually in highly turbid, muddy areas. Also seen at night foraging for crabs over shallow sea-grass meadows.	Yes, fully exploited fisheries. Some fishing of mangrove habitats.	Feeds mainly on crustaceans such as the crab *Helice crassa*, and shrimp *Palaemon affinis*, along with some amphipod species. Larger eels (>350 mm) also feed on small fish e.g., exquisite goby (*Favonigobius exquisitus*).
Ophichthidae	*Muraenichthys breviceps*	Long finned - worm eel	MMO	CH/SG *	Found in North Island coastal waters in seafloor burrows. Associated with northern estuarine seagrass beds.	No	Feeds on bivalve siphons and small fish. Only sampled from the Manukau.
Ophichthidae	*Ophisurus serpens*	Snake eel	MS	CH	Present along the north east coast. Adults occur near rough bottom in about 100 m. Juveniles are found in shallow estuarine areas.	No	Feeds on shrimps *(Pontophilus australis*), crabs and small fish.
Gobiescoidae		Clingfish sp.	EM	SG/I *	Benthic living. Found from the intertidal zone to depths of 100 m. Usually found beneath rocks and algae. Also found in seagrass beds and in association with shells	No	No stomach analysis.
Scorpaenidae	*Helicolenus percoides*	Scorpion fish Sea perch Pohuiakaroa	MMO	B/ST *	Found throughout New Zealand but most common south of Cook Strait. Found on reefs or nearby open bottom. Ovoviviparous: gives birth to small larvae within a jelly mass which are planktonic.	No	Only sampled in the Manukau. Juveniles feed predominantly on crabs (e.g. *Hemigrapsis crenulatus)* and isopods.
Moridae	*Auchenoceros punctatus*	Pink cod	MMO	CH/ST	Occurs in scattered localities around New Zealand. Quite abundant in some areas, notably the Firth of Thames. Only sampled in the Manukau in this study, in subtidal channels with higher water clarity.	No	Feeds almost exclusively on mysids with small amounts of plankton.
Moridae	*Pseudophycis breviuscula*	Bastard red cod	MS		Occurs around the northern North Island.	Limited by-catch	
Triakidae	*Mustelus lenticulatus*	Rig Spotted dogfish	MMD	MD	Found throughout coastal waters, more common in the north. Abundant in estuaries during spring and summer when females come in to give birth to their pups. Migrate to outer shelf during autumn. Favours muddy areas for feeding.	Yes	Feed on a variety of small fish and invertebrates (crabs, polychaetes) when juveniles and progressively larger animals as they grow.

Sepiariidae	*Sepioloidea pacifica*	Bumblebee squid	ER	S/ST **	Present throughout New Zealand. An ambush predator, it lies in wait for small invertebrates in sandy habitats.		Feeds on amphipods, mysids and other small crustaceans.
Uranoscopidae	*Leptoscopus macropygus*	Estuarine stargazer	MMO	SG* SD** Mud*	Found throughout New Zealand shallow coastal waters.		Juvenile diet strongly driven by mysids and small fish in the Manukau. Diet in other northern estuaries includes amphipods and the cumacean *C. lemuran*.
Uranoscopidae	*Genyagnus monoterygius*	Spotted stargazer Kourepoua	MMO	SD* MD*	Widespread throughout the north island and east coast of the South Island (0-200 m). Conceal themselves in the sand/mud and ambush passing prey. Spawn in spring and early summer.		Juveniles fed almost exclusively on mysid shrimps in the Manukau along with some Caridean shrimps, i.e., *Pontophilus australis.* Larger juveniles (>80 mm) also fed on small fish such as the exquisite goby (*Favonigobius exquisites).* Adults eat a variety of small fish and crabs.
Creediidae	*Tewara cranwellae*	Sand diver		SD MD	Usually found burrowing in sand or mud.	No	No stomach analysis.
Other species found in northern estuaries – not sampled for dietary analysis in this study							
	Galaxias maculatus and other spp.)	Inanga White-bait	SC		Sampled in high abundances as small adults in freshwater dominated (river) estuaries, including southern systems without mangroves.	Yes	
	Zeus faber	John dory	MS		Seasonally move into estuaries and shallow water areas (colder months), important predator of small fishes, including juvenile snapper.	Yes	
	Galeorhinus australis	School shark	Semi MMD		Uses estuaries for breeding and pupping grounds (spring and summer). Associated with higher water clarity in the lower estuary for the Manukau.	Yes	Feeds on small fish such as yellow-eyed mullet, sand flounder and grey mullet.
	Sphyrna zygaena	Hammerhead sharks	Semi MMD		Juveniles relatively common in warmer months, probably pup in harbour systems and shallow coastal embayments (e.g. Tamaki Strait, Firth of Thames). Favours clear water and sandy habaitats.	No	Feeds on demersal fish such as sand flounder, yellow-eyed mullet and grey mullet.
	Thrysites atun	Barracouta	MS		Occasionally sampled as large adults in channel areas (gill-nets).	Yes	
	Seriola lalandii	Kingfish	MS		Known to have once displayed seasonal ‘runs’ into some estuaries following kahawai (e.g. Rangirere sub-estuary, Manukau Harbour), but ‘fished out’ in past decades. Adults still caught in estuarine systems	Yes	
	Myliobatus tenuicaudatus	Eagle ray	MMO		Seasonally common in some estuaries, including large numbers of both juveniles and adults.	No	
	Scorpis lineolatus	Sweep	MS			No	
	Seriolla brama	Common warehou	MS			Yes	
	Lotella rhacinus	Rock cod	MS			By-catch only	

Habitat:
MG = Mangrove
MD = mudflats
SG = Seagrass
SD = sand
HM=Horse mussels

Position:
I=Intertidal
ST=Subtidal
CH=Channels

Abundance:
Absent
* Rare
** Common
*** Very common
**** Highly abundant

Guild:
ER= Estuarine resident
EM= Estuarine migrant
CA= Catadromous
AN= Anadromous
MS= Marine straggler
MMO= Marine estuarine opportunist
MMD= Marine estuarine dependent

NB. Refer to Fig. 2.1 for guild classification (Elliot et al., 2007). Guild classification is somewhat fluid depending on the location within New Zealand. For example, Leather jackets become more abundant in the seagrass meadows in the South Island.

General references utilized: Crisp et al., (1990); Francis (2001); Francis et al., (2005; 2011); Jellyman et al., 1997; Kilner & Akroyd 1978; Morrison (1990); Morrison et al., (2002; 2007; in prep); Morrison & Carbines 2006; Paul (2000; 2003); Paulin (1998); Paulin et al., (2001); Morrison pers. com.; Author pers. obs.

Additional References for individual species:
Flounder: Colman (1974a; 1974b; 1978); Grogan (1982); Livingston (1987); Mutoro (2001); Park (1984); Pearks (1985); Roper & Jillet (1981); Saunders (1999)
Sole: Park (1984); Garfish: Russell (1983); Spotty: Newcoombe (2009); Yellow-eyed mullet: Taylor & Paul (1998); Short-finned eel: Ryan (1978); http://collections.tepapa.govt.nz; Smelt : McDowall, (2000).

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