Monitoring Environmental Impacts of Recreational Boat Anchoring on Eelgrass (*Zostera marina* L.) and Benthic Invertebrates in the Gulf Islands National Park Reserve of Canada

by
Kate Elizabeth Leatherbarrow
B.Sc., University of Calgary, 2003

A Thesis Submitted in Partial Fulfillment of the
Requirements for the Degree of
MASTER OF SCIENCE
in the Department of Geography

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ABSTRACT

The goal of this study was to characterize the ecology and recreational boating activity at two popular anchoring sites located in the waters of the Gulf Islands National Park Reserve of Canada (Sidney Spit and Tumbo Island). The three components of the study were to characterize the distribution of eelgrass (*Zostera marina* L.), build an inventory of anchoring/mooring activity, and characterize the benthic infauna at each site. These observations were used to evaluate the impact of anchoring on the eelgrass and invertebrate communities. No visible loss of eelgrass was documented, but the results at one of the two sites support the hypothesis that benthic communities in high anchoring intensity areas are in poorer health than those in low anchoring intensity or mooring areas, a characteristic of communities residing in disturbed and fragmented eelgrass beds. Recommendations for site management and long-term monitoring are proposed based on these results.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

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CHAPTER ONE: OBJECTIVES

1.1 - Overview

Marine environments currently face several threats, including overfishing and collapse of marine food webs (Jackson *et al.* 2001), coastal development (Weintstein and Reed 2005), pollution from spilling/dumping of toxic substances (Peterson *et al.* 2003), and changes in sea surface temperature related to global climate change (McWilliams *et al.* 2005). Marine protected areas (MPAs) are a common management tool, intended to mitigate damage to marine environments (Agardy *et al.* 2003). Activities managed or forbidden in protected areas can include fisheries, aquaculture, industrial/commercial development, recreation and tourism, extraction of minerals and petroleum products, and pollution (Salm *et al.* 2000). MPAs are a flexible management tool that can be adapted to the specific threats and uses of each individual protected area.

Parks Canada has an active presence in the coastal waters of British Columbia with jurisdiction over the marine components of two coastal national park reserves legislated under the National Parks Act. The primary mandate of the National Parks Act is to preserve ecological integrity, "*a condition that is determined to be characteristic of its natural region and likely to persist, including abiotic components and the composition and abundance of native species and biological communities, rates of changes and supporting processes*" (Canada National Parks Act 2000). Therefore, it is important that any management plan start with a good understanding of the park's ecology. A secondary mandate of the National Parks Act is to provide a space for outdoor recreation and foster environmental stewardship, so it is also important to manage these activities such that ecological integrity of the park is not compromised.

Parks Canada also has management opportunities on the West Coast through the new National Marine Conservation Areas (NMCA) Act, passed in 2002. Though the program is still new at the time of writing, eventually all five marine regions on the Pacific coast (as established in Canada's National Marine Conservation Areas Plan) will be represented by NMCAs (Mercier and Mondor 1995). The mandate of the NMCA Act is different from that of the National Parks Act in that it promotes *sustainable use* of the conservation areas. Therefore, non-sustainable extractive activities such as oil and gas exploration will not be permitted, whereas fisheries, aquaculture or shipping may still be considered acceptable in certain parts of the NMCA on a case-by-case basis.

One management approach that is appropriate for maintaining ecological integrity and controlling sustainable use is the zoning approach, where "particular uses [are] confined to particular zones within the MPA where they are appropriate or where their uses do not conflict with other uses" (Salm *et al.* 2000, pp 44). Zoning allows selective control with varying levels of use and strict protection as well as spatial separation of conflicting user groups (Salm *et al.* 2000). Salm recommends that each marine protected area be composed of a one or more core conservation areas separated from human use areas by buffer zones. Effective zoning requires in depth knowledge of the human use and ecology of a region, including locations of commercial and recreational activities, First Nations' activities, sensitive habitats, breeding areas, feeding areas, migratory routes and trophic linkages (Salm *et al.* 2000).

The Gulf Islands National Park Reserve of Canada (GINPR) (Figure 1.1) , representing the Strait of Georgia lowlands natural region, was established in May 2003. This is a coastal park with a very significant marine component, composed of 15 islands as well as several islets and reefs. In fact, the park is comprised of almost equal terrestrial and marine areas, with 35 km^2 of land and intertidal area and 26 km^2 of marine areas (Parks Canada 2005). Park jurisdiction extends at least 200 m from the low tide line in all areas, with extended boundaries at certain locations. There is also an NMCA proposed for this region to complement the existing national park, which would give Parks Canada even more authority over marine waters in the southern Strait of Georgia. The area currently under consideration in the feasibility study for this NMCA is indicated in Figure 1.1. Given the heavy commercial and recreational use of this region, having jurisdiction over these waters will allow Parks Canada to explore implementing a zoning strategy to resolve conflicting uses and help maintain sustainable use of the region as outlined in the NMCA act.

The southern Gulf Islands are a very popular region for recreational boating (McDade 2000) with user densities rivaling the most popular recreational boating areas in the world. However, recreational boating has been shown to cause pollution, increase turbidity, disperse invasive species and disturb benthic habitats (Backhurst and Cole 2000). Research on the specific environmental effects of recreational boating in this region is required to make informed management decisions that minimize the damage caused by this recreational activity and maintain ecological integrity. Parks Canada has expressed a particular interest in evaluating the impacts of physical disturbance on benthic habitats caused by recreational boat anchoring, and this study was designed in response to that interest.

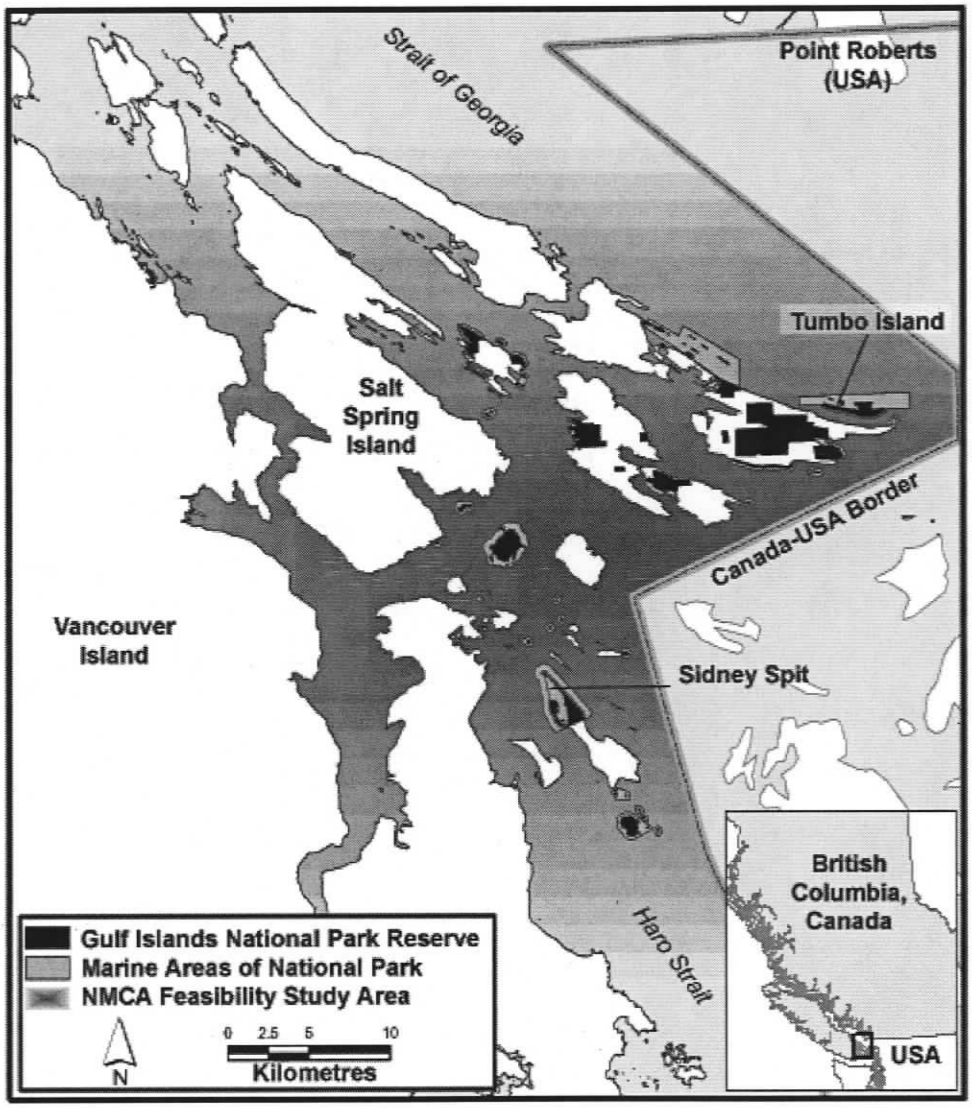

Figure 1.1 - Gulf Islands National Park Reserve of Canada – terrestrial and marine components (February 2005). Also includes the area being considered in the National Marine Conservation Area feasibility study.

1.2 - Thesis goals and design

The goal of this study was to characterize the ecology and boating activity at two popular anchoring sites (Sidney Spit and Tumbo Island) located in the waters of the GINPR (indicated on Figure 1.1). Three specific objectives of this study were to:

1) Characterize the distribution of habitats at each site (Chapter 3);
2) Build an inventory of anchoring activity at each site (Chapter 4); and
3) Characterize the benthic infauna at each site and evaluate the impact of anchoring pressure on the nature of these invertebrate communities (Chapter 5).

The major objective of the first two components was to collect baseline data about the habitats and human activity at the selected sites and to provide a protocol for ongoing monitoring of these habitats and activities. In the habitat description component, a remotely operated underwater video camera was used to record the location of eelgrass under each anchoring site. In the human activity component, facility operators recorded the locations and attributes of boats using the anchoring sites. Together, these data were compiled in a geodatabase and used to test the null hypothesis that anchoring activity does not cause a change in eelgrass bed configuration.

The objective of the third component of the study was to test whether the physical disturbance caused by boat anchors affects the invertebrate communities under the selected anchorages. Samples of benthic infauna were collected, preserved, sorted, and identified to test the null hypothesis that invertebrate community composition is not affected by anchoring pressure. The protocol established in this part of the study also has the potential to be implemented as a long term monitoring program.

The results of this thesis will be presented with the following structure. A literature review explaining the global significance of eelgrass and the human threats to eelgrass (including recreational boating) will introduce the research (Chapter 2). The three research objectives stated earlier will be discussed in order, with the description of the ecology and human activity at each site (in Chapters 3 and 4) leading into a discussion of the possible impacts of anchoring pressure on invertebrate communities at those sites (Chapter 5). Finally the thesis will conclude with a summary of findings and recommendations as to the feasibility of continuing this study as a long term monitoring program (Chapter 6).

This study will provide baseline data to park managers regarding the habitats, organisms and human use at these sites as well as a protocol for ongoing monitoring that could be expanded to other anchoring sites in the park. This information will be useful for park managers to make informed decisions regarding the implementation of zoning regulations that effectively minimize user conflicts and maximize ecological integrity. The baseline data and long term monitoring program would also be useful in evaluating the effectiveness of the management strategies implemented inside the protected area (Edgar 2004).

1.3 - Study site description

The GINPR currently has three locations with mooring buoys: Sidney Spit, Tumbo Island (Cabbage Island) and Beaumont at South Pender Island. Sidney Spit and Tumbo Island were selected as the two sites for this study. Sidney Spit (Figure 1.2), with nineteen mooring buoys, is a relatively exposed location just east of the Saanich peninsula. It is a very popular destination for residents of the Victoria area due to its close proximity to the city. Tumbo Island (Figure 1.3) is a very sheltered anchorage at the easternmost edge of the Gulf Islands. There are ten mooring buoys at this location, and the site is less easily accessible from Victoria. Both sites have onshore picnic and washroom facilities, with nightly moorage fees of ten dollars per buoy. Anchoring is permitted at both locations free of charge, though anchoring is not allowed inside the lagoon at Sidney Spit.

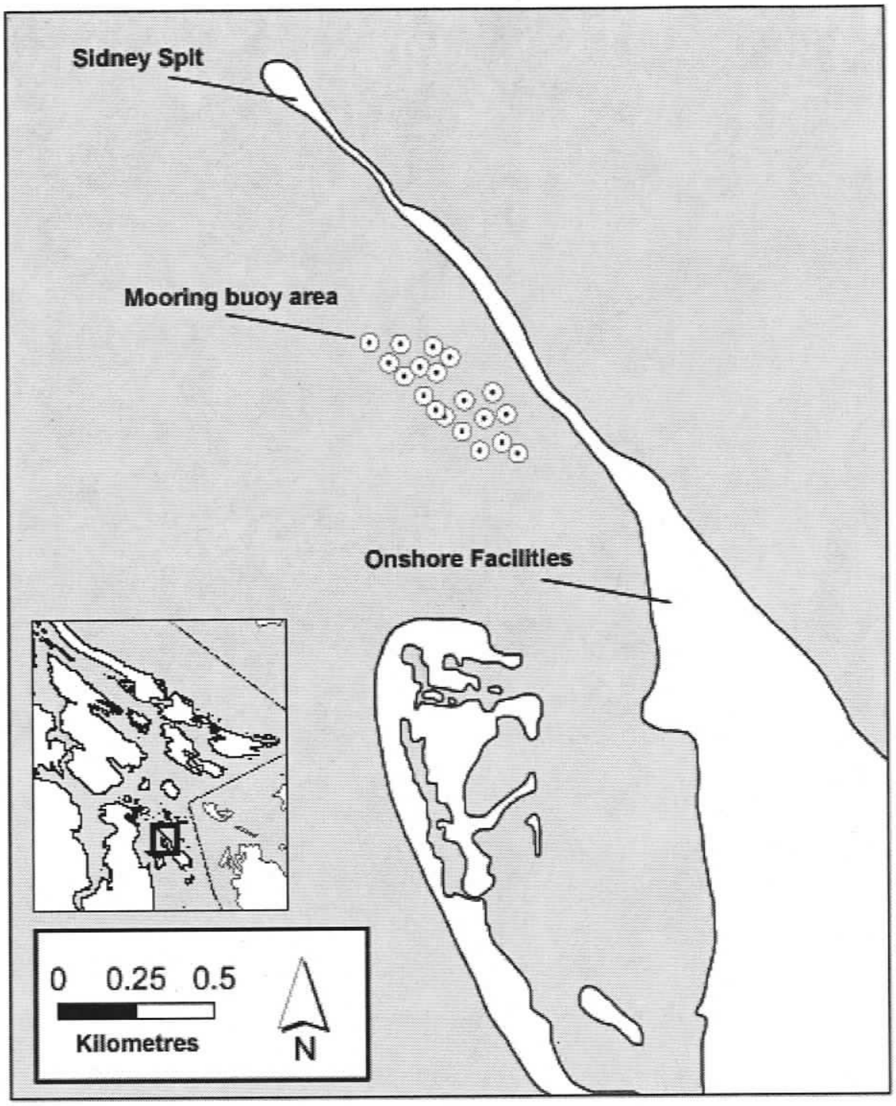

Figure 1.2 - Sidney Spit study area

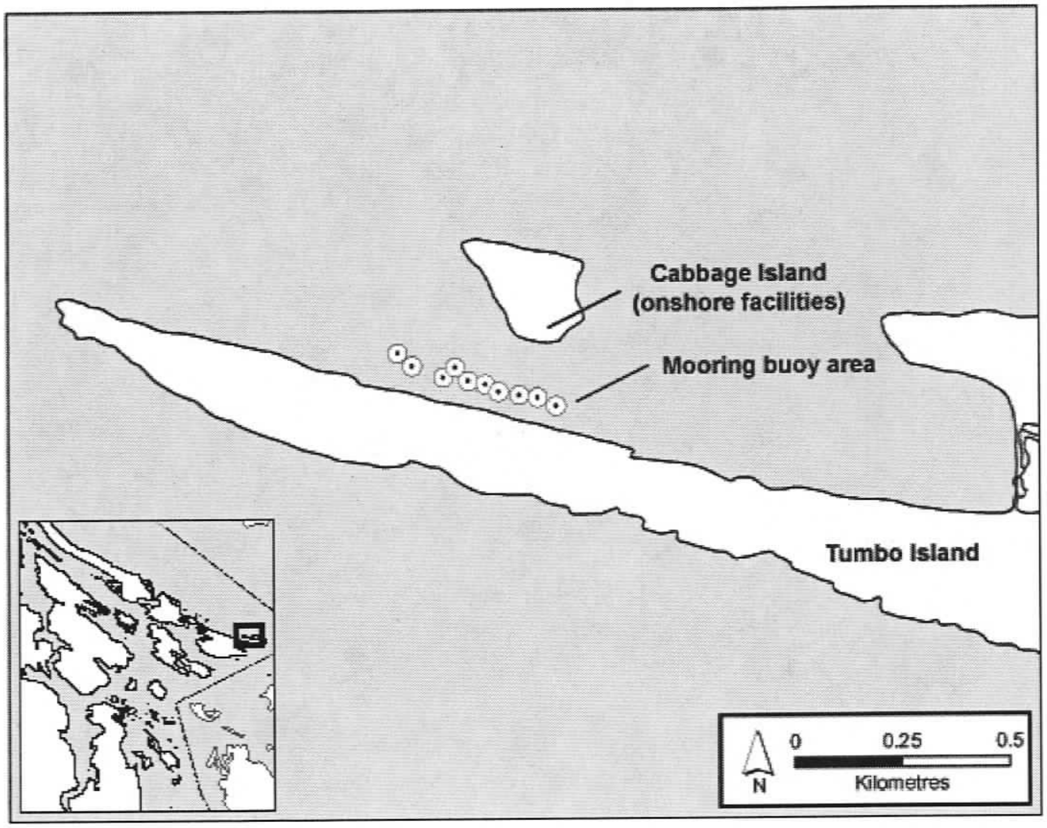

Figure 1.3 - Tumbo Island study area

CHAPTER TWO: LITERATURE REVIEW

2.1 - Physical oceanography of the region

The waters surrounding the southern Gulf Islands, located between Vancouver and Vancouver Island, are representative of the southern Strait of Georgia marine natural region (Mercier and Mondor 1995). Within the Strait, tidal range is on the order of 4m and the average tidal current is 1m/s (reaching 5m/s in narrow and shallow passages) (Leblond 1983). Runoff from the Fraser River provides 73% of the freshwater input to the Strait, creating a stratified plume and estuarine conditions in the central Strait that can reach all the way to the Gulf Islands (Leblond 1983, Johannessen *et al.* 2003). However, the waters between the islands are well mixed due to extensive tidal mixing in the passages between the islands (Leblond 1983), and the overall turnaround time in the Strait is approximately one year (i.e., time elapsed between water entry into and exit from the strait) (Mackas and Harrison 1997).

The Fraser River also provides 63% of the suspended particle input to the Strait. This particle input is almost completely trapped in the Strait by sedimentation before the water flows through the Strait of Juan de Fuca (Johannessen *et al.* 2003). This substratum is composed of generally coarse sediments (sand and gravel) eroded from Pleistocene deposits by strong tidal currents (Luternauer *et al.* 1983). There are high inputs of nitrogen and dissolved/particulate organic carbon through natural processes; anthropogenic inputs of these nutrients are not substantial compared to ambient levels and only urbanized shorelines and inlets with low flushing rates are considered to be vulnerable to eutrophication (Johannesson *et al.* 2003, Mackas and Harrison 1997).

2.2 - Significance of eelgrass in coastal ecosystems

In the southern Strait of Georgia, there are nearshore areas of soft sediments and low wave energy of that provide excellent conditions for the growth of eelgrass (*Zostera marina* L.), a key component of coastal ecosystems. Eelgrass is a marine angiosperm, growing mainly in subtidal but sometimes in intertidal zones (Phillips and Menez 1988), from +1.8m down to -6.6m (Phillips 1984) Maximum depth of eelgrass is usually limited by light availability and therefore varies with water clarity (Dennison 1987). Eelgrass is a perennial that normally reproduces asexually in clonal fashion, but will occasionally reproduce sexually through flowering (Phillips and Menez 1988).

Eelgrass provides many important ecosystem services in nearshore waters related to carbon and nutrient cycling. It is an important primary producer, fixing energy from sunlight into organic carbon compounds (Duarte 2002). Eelgrass is also subjected to relatively low grazing pressure, and therefore is a significant carbon sink and a rich source of organic carbon in coastal food webs (through the detrital pathway). Seagrasses as a whole store about 15% of the ocean's carbon and export 24.3% of their carbon to adjacent ecosystems (Duarte 2002).

Duarte's 2002 review suggests several more key services provided by seagrass ecosystems, considered by Constanza *et al.* (1997) to be one of the most valuable ecosystems in the biosphere. The roots of eelgrass beds stabilize sediments, preventing erosion from waves and tidal currents and buffering shoreline change. The eelgrass canopy dampens wave oscillations in the water inside eelgrass beds, providing habitat for fish and invertebrates that prefer a lower wave energy environment. Eelgrass blades also provide physical habitat for epiphytes and invertebrates such as encrusting bryozoans and small gastropods. The abundance of fish and invertebrates found in eelgrass beds attracts many foraging seabirds such as Marbled Murrelets and Brandt geese. These characteristics of eelgrass beds have led some researchers to put forward the disputed "nursery hypothesis," stating that eelgrass beds provide critical habitat for juveniles of many commercially important fish and invertebrates (Jackson *et al.* 2001a). Whether or not eelgrass beds truly are nurseries (Heck *et al.* 2003), it is clear that many organisms utilize the habitat created by eelgrass beds and that healthy eelgrass habitat is an important component of coastal food webs.

Mercier and Mondor (1995) also point out that the southern Strait of Georgia is one of the most heavily populated marine regions in the country, with shoreline development, overfishing, pollution and heavy marine traffic contributing to stress on the coastal ecosystem. The next section will explore how these human impacts have a negative impact on eelgrass bed health and interfere with all the important ecosystem functions provided by eelgrass.

2.3 - Human activities and eelgrass bed health

Perhaps the most obvious human impact on eelgrass health is the destruction of eelgrass beds as a result of shoreline development. For example, dredging and seawall construction physically removes eelgrass shoots from the substratum, resulting in the loss of entire beds (Duarte 2002). Construction of permanent physical infrastructure, such as docks, shades the bottom and prevents eelgrass beds from obtaining enough light for normal photosynthesis (Burdick and Short 1999). These structures cast a shadow on the benthos that, depending on the latitude of the site, can remain quite constant throughout the year. Burdick and Short (1999) found that eelgrass under and adjacent to docks in coastal Massachusetts had reduced light levels, resulting in depressed shoot density and canopy structure.

Because eelgrass often colonizes estuarine areas, it is also vulnerable to development that occurs upstream in nearby watersheds (Duarte 2002). Logging or land clearing for construction leads to erosion and increased sediment loads in the watershed. This sediment settles out in the estuary due to the abrupt change in water velocity and can smother eelgrass beds located at the point of deposition. Even if the sediment load isn’t high enough to cause smothering, it can still decrease water clarity and therefore photosynthetic potential of the eelgrass beds. Industrial and agricultural pollution are also potential concerns: organic and inorganic waste from these sources promote eutrophication of coastal waters and interfere with light availability to eelgrass (although the naturally high nutrient loads in the Strait mean that these may not be the primary concern in the GINPR).

Heavy marine traffic also has negative impacts on eelgrass beds. In shallow waters, turbulence from propellers stirs up benthic sediments, increasing suspended sediments in the water and decreasing light penetration to the substratum, interfering with primary productivity (Beachler and Hill 2003). Physical turbulence also releases nutrients stored in benthic sediments, changing the water's nutrient load and potentially promoting phytoplankton blooms which could suppress macrophytes (Anthony and Downing 2003). Marinas and wharves are known to be a source of fecal coliforms in marine waters; shellfish harvesting is prohibited within 125 metres of these areas (Harding 1994). Increased levels of petroleum pollutants, especially diesel fuels, are associated with marinas (Piehler *et al.* 2002). Biocidal compounds included in paints for boat hulls to act as anti-fouling agents have also been shown to leach and accumulate in sediments under marinas (Albanis *et al.* 2002). Erikksen *et al.* (2004) found that macroalgal communities near marinas were rich in species that thrive in nutrient rich, turbulent habitats and depauperate in species sensitive to light availability.

Finally, anchoring by recreational boaters can cause physical damage to benthic habitats. This is especially obvious in seagrass beds; swaths of seagrass are removed by scouring from anchors and anchor chains, resulting in gap formation (Walker *et al.* 1989). Given the importance of eelgrass in the nearshore environment, the consequences associated with eelgrass bed fragmentation will have repercussions on other organisms and coastal processes.

2.4 - Physical disturbance, the intermediate disturbance hypothesis and fragmentation

Connell (1978) proposed the intermediate disturbance hypothesis based on his observations of coral reef and tropical rainforest biodiversity. This theory is based on the premise that communities are rarely in a state of equilibrium, and that continually changing species composition is required to maintain diversity in a community. Diversity is expected to be highest when disturbance is at intermediate frequency and intensity. If disturbance is infrequent or of low intensity, then diversity will remain low, because new colonizers are competitively excluded by existing climax species. Conversely, if the disturbance is too intense or too frequent, then many species will be damaged or killed and diversity will also remain low.

Because marine ecosystems have a high level of redundancy, the effects of a reduction in diversity are often hard to identify immediately; often other organisms remain that can step in to fill the niche of a removed species (Jackson *et al.* 2001b). However, when enough species are removed, eventually there won't be any organisms left to fill available niches, leading to ecosystem collapse (Jackson *et al.* 2001b). Paul and Anne Ehrlich have proposed the analogy of rivet removal from an airplane wing: "A dozen rivets, or a dozen species, might never be missed. On the other hand, a thirteenth rivet popped from a wing flap, or the extinction of a key species involved in the cycling of nitrogen, could lead to a serious accident. (Ehrlich and Ehrlich 1981, pp xiii)" Although the specific consequences of the loss of one species in an ecosystem are impossible to predict, this is not a justification for allowing loss of biodiversity to occur.

Of course an extreme example of disturbance is the case where habitat patches are completely eliminated, resulting in habitat fragmentation. Concern about fragmentation was raised in the late seventies and early 1980s, when a recurring topic in the conservation biology literature was the question of whether a single large reserve or several small reserves would be more effective in preventing extinctions (referred to by the acronym SLOSS – single large or several small). This debate was based on MacArthur and Wilson's (1967) theory of island biogeography, considering discrete habitat patches as "islands". Simberloff and Abele (1982) considered this question in the context of static habitats that had been historically fragmented and found that the probability of extinction is not necessarily reduced by having several small reserves. However, Wilcox and Murphy (1985) point out that this analysis could be misleading because the more realistic situation of habitat fragmentation is slightly different; in this case, swaths of material are cut from an intact habitat. Not only is the habitat divided into smaller patches (insularization) but the total area of available habitat is reduced and the edge:interior habitat ratio is increased.

As a result, the question of fragmentation and reserve geometry has shifted from a strict focus on island biogeography towards metapopulation theory (Hanski and Simberloff 1997), considering the interactions of a single species within a network of patches in the absence of a large "mainland patch" (a critical component of island biogeography). An extension, metacommunity theory, accounts for assemblages of interacting species in similar network of patches (Harrison and Bruna 1999). Fragmentation issues have also been considered in source-sink models and percolation theory, respectively addressing movement between habitats of different quality and the role of landscape configuration in dispersal (Pulliam *et al.* 1992, Dias 1996, Boswell *et al.* 1998). Though it is now generally accepted that spatial strategies (such as maintaining corridors between patches) can remedy the effects of habitat fragmentation to some degree, species-specific conservation strategies are still very difficult to define (Harrison and Bruna 1999).

Insularization and habitat loss are both known to lead to a decline in biodiversity (Wilcox 1980). The influence of edge effects also becomes more pronounced in fragmented habitats; Saunders *et al.* (1991) point out that wind, precipitation, solar irradiance and nutrient fluxes are altered at habitat edges when they are surrounded by developed land. In the marine realm, edges created by gaps formation in seagrass meadows are subject to stronger currents and higher sedimentation rates than meadow interiors (Uhrin and Holmquist 2003). Habitat edges are also known to be an area of increased species interactions such as predation and competition (Fagan *et al.* 1999); an increase in edge:interior ratio will therefore change dynamics of the resident community by increasing the number of interactions between individuals. Finally, an increase in edge:interior ratio has an especially deleterious effect on interior specialists; the decline of interior species in fragmented habitats is greater than predicted by habitat loss alone (Bender *et al.* 1998). In contrast, decline of edge specialists is less than predicted by habitat loss alone (Bender *et al.* 1998). This disproportional loss of interior specialists changes the dynamics of the community associated with this fragmented habitat and has a negative impact on the ecological integrity of the community.

2.5 - Physical disturbance of seagrass beds and soft sediments

The main type of physical disturbance studied in seagrass meadows is fragmentation caused by gap creation in meadows. Anchors are not the only source of scars in seagrass meadows – other sources of damage include boat propellers (Bell *et al.* 2002), sedimentation (Preen *et al.* 1995), disease (Wilson 1949) and herbivory (Thayer *et al.* 1984). Unfortunately, the impact of gaps and edge effects on seagrass meadow community structure remains poorly understood and research focuses not on overall diversity but instead on responses of specific species. Uhrin and Holmquist (2003) found that crabs and mollusks were significantly affected by scar formation in *Thalassia testudinium*, with decreased densities within 5m of scar edges. However, shrimp and fish densities did not differ between interior and edge zones. Bell *et al.* (2002) also report no significant difference in shrimp density or pinfish density between scarred and intact *Thalassia testudinium* meadows, but report a small increase in pipefish in scarred meadows. Bell *et al.* (2001) found reduced densities of polychaete worms in the outer metre of *Halodule wrightii* and Irlandi *et al.* (1999) found that juvenile bay scallops in beds of mixed *Zostera marina* and *Halodule wrightii* had higher survivorship in patch interiors than in patch edges. These studies are not very informative when considered in isolation, and it seems that responses are very specific to each type of seagrass community and invertebrate species.

With respect to eelgrass beds, Bostrom and Bosendorffer (1997) observed higher abundance and diversity of invertebrates in eelgrass stands as compared to adjacent bare sediment. Bowden *et al.* (2001) report that patch edges of *Zostera marina* have higher abundance of opportunistic taxa (those associated with disturbed habitats) than patch interiors, while Frost *et al.* (1999) found that fragmented *Zostera marina* meadows have a higher abundance of benthic infauna normally associated with unvegetated sediments. These studies were the only ones that addressed the response of the entire seagrass community to fragmentation. Frost *et al.* (1999) found no significant difference in abundance, number of taxa or diversity between fragmented and unfragmented beds. Bowden *et al.* (2001) found no significant difference in abundance, number of taxa, or diversity between meadow interiors and edges. However, the fragmented beds analyzed in these studies were not fragmented as a result of recent disturbance – the fragmentation in these meadows had been created historically. The intermediate disturbance hypothesis predicts that the effect of a disturbance will decrease with time, so it is not surprising that no size or edge effects were observed in these studies. However, their results highlight the need for looking past reductionist measures of community change like abundance or diversity and analyzing the specific species that make up each community. Otherwise, important differences like the observed shift towards opportunistic taxa could be missed.

Although the gap formation resulting from physical disturbance to seagrass meadows is easily visible, it is less obvious how physical disturbance affects communities living in the soft sediment under the eelgrass beds and in the newly formed gaps. The majority of research on soft bottom habitats has been conducted on the effects of trawling, which is a much larger scale of disturbance than the physical disturbance caused by recreational boat anchors. However, the same general principles still apply. Patches can be partially or fully defaunated as a result of physical damage (Dernie *et al.* 2003) and species displaced by disturbance to a less favourable habitat might be vulnerable to increased predation (Ramsay and Kaiser 1998). Disturbance affects the stability and roughness of the sediment bed (Peterson *et al.* 1987), and changes to sediment topography can change the local hydrodynamic regime, affecting organic matter deposition and larval settling (Thrush *et al.* 1992). From this evidence, it is clear that soft sediment communities also suffer damage from physical disturbance despite the absence of immediately visible physical structure.

Some advocates of trawling have suggested that the disturbance of benthic communities by trawling gear represents an intermediate level of disturbance and therefore maintains biodiversity as predicted by the intermediate disturbance hypothesis (Thrush and Dayton 2002). However, little evidence exists to indicate that benthic communities are in fact limited in diversity by resource competition (Wilson 1991), and theoretical consideration of multitrophic level communities (representative of most marine ecosystems) suggests that disturbance does not have an effect on the coexistence of competitors (Wootton 1998). Therefore, the intermediate disturbance hypothesis might not be applicable to soft bottom benthic communities and any amount of human-induced disturbance may be too intense to allow persistence of natural communities (Thrush and Dayton 2002). Indeed, Currie and Parry (1996) found that the abundance of six species decreased by 20-30%

while the abundance of only one species increased in their study on the environmental impacts of scallop dredging.

Indeed, this hypothesis is supported by the work of Veale *et al.* (2000) and Thrush *et al.* (2001), who report decreased species diversity in response to homogenization of bottom sediments by trawling results. A review of the effects of trawling on benthic communities found that habitat-structuring species as well as large and long-lived organisms are lost as a result of physical disturbance (Thrush and Dayton 2002). Work by Dernie *et al.* (2003) shows that chronic disturbance by towed nets and dredges in the Irish sea has caused a shift from communities made up of sessile epibenthic fauna of high biomass to communities made up of smaller infauna that are resilient to physical damage. Combined with the evidence that the intermediate disturbance hypothesis may not apply to soft-bottom communities, there is every reason to believe that even the small-scale physical disturbance caused by recreational boat anchors will have a negative impact on the benthic communities at anchoring sites.

2.6 - Consequences of disturbance to benthic communities

Though unglamourous, marine benthic macrofauna represent the most diverse assemblage on earth at the phylum level, constituting the largest ecosystem on earth in terms of areal coverage and playing key roles in sediment structuring, nutrient cycling, and secondary production (Snelgrove 1998). In his 1998 review of benthic macrofaunal diversity, Snelgrove states that filter feeders improve water clarity by removing particles from suspension, and deposit feeders modify and oxygenate the sediment structure through bioturbation and selective particle uptake, repackaging, and egestion in fecal pellets. As a result, the activities of macrofauna directly influence sediment stability and transport as well as metabolized pollutant burial in coastal areas. Snelgrove also points out that oxygenation and organic matter produced by benthic macrofauna closely link them to microbial food webs and therefore make them influential in nitrogen, sulphur and carbon cycling at a global scale. A final key service provided by benthic macrofauna that Snelgrove discusses is its source of nutrition to higher trophic levels, through direct secondary production as well as the recycling of detritus back into the food web. It is clear that benthic macrofaunal communities provide several critical ecosystem services, and that it is important to maintain functional and genetic diversity within them in order that they continue providing these services.

2.7 - Physical disturbance of eelgrass by recreational boat anchoring

Very few studies have specifically studied the environmental impacts of recreational boat anchoring. Seagrass loss associated with boat anchoring has been documented at Rottnest Island in Western Australia (Walker *et al.* 1989; Hastings *et al.* 1995) and at marine parks in the Northwestern Mediterranean Sea (Francour *et al.* 1999; Milazzo *et al.* 2004), but these studies focused on quantifying the amount and rate of seagrass loss rather than the faunal communities associated with the seagrass beds, considering seagrass purely as a physical structure for habitat. Hastings *et al.* (1995) found that seagrass loss was positively correlated with heavy mooring intensity, but that the amount of loss was increased when the patch edge was exposed to prevailing swell or current. This indicates that environmental parameters should be carefully considered when assessing the degree of damage caused by recreational boat anchoring.

Milazzo *et al.* (2004) found that damage varied with the type of anchor being used and the fashion in which the anchor was deployed. They found that chains did not cause more damaged shoots than ropes, and that the traditional "Navy" or "Yachtsman" type anchor (referred to as a Hall anchor in the Milazzo study) caused the lowest number of damaged shoots. They also found that the weighing of the anchor was the most critical phase of anchoring to cause damage to seagrass shoots.

Creed and Amado Filho (1999) evaluated the damage caused by anchoring in a Brazilian seagrass bed, and found that gaps were usually colonized only with pioneer species of macroalgae even after one year, corresponding with the work of Walker *et al.* (1989). These results suggest that in seagrass beds, even if damage is temporally constrained to a few months out of the year, the gaps created in seagrass beds cannot recover to their normal state between boating seasons.

Only one study was found specifically addressing the impacts of anchoring on benthic communities, focusing on activity around Kawau Island in Northeastern New Zealand (Backhurst and Cole 2000). They compared sites with light, medium and heavy anchoring intensity and found that organism abundance was not significantly different among sites. Another component of their study, examining the rate of community recovery after experimentally induced anchor damage, led them to the conclusion that communities recover fast enough that recreational boat anchoring does not need to be regulated in their study region. However, the authors recognize that their sampling design likely underestimates the amount of anchoring taking place and the degree of damage caused by this activity. The simulated anchoring events used to create experimental damage in this study were also not representative of true anchoring events since the anchor was not "set" and the boat did not swing at anchor, therefore excluding the impacts of any potential chain-dragging on the benthos. Based on these criticisms, despite the fact that Backhurst and Cole's (2000) investigation was conducted in a temperate marine ecosystem not unlike the Southern Strait of Georgia, the results of their investigation do not provide an adequate basis for decision making in the Gulf Island National Park Reserve and further research must be conducted.

CHAPTER THREE: MAPPING OF EELGRASS WITH UNDERWATER VIDEO

3.1 - Introduction

3.1.1 - Background

Mapping is a useful tool in the management and conservation of seagrass beds (Kelly *et al.* 2001). The first component of this study required a technique to identify the extent of the eelgrass beds at both anchoring sites in the study area. Several approaches to eelgrass mapping in the shallow subtidal have been described in the literature, varying with respect to spatial scale, cost efficiency and time intensity.

Mapping underwater features is challenging due to the inaccessibility of the submarine environment. Diving is the only direct method to observe benthic features such as eelgrass, but is not very practical due to the limited mobility of divers, time constraints on the length of dives and the cost of training and equipment. Direct observation may also be invasive to the eelgrass bed and there is the possibility inexperienced divers could inadvertently cause damage by disturbing vegetation. Although diving is an excellent approach to some aspects of underwater data collection, remote sensing provides higher quality data and more extensive areal coverage for mapping applications.

3.1.2 - Remote sensing approaches to underwater mapping

Aerial photography and satellite imagery provide a time-efficient solution to obtaining accurately georeferenced data sets covering large stretches of coastline, and are both used to map seagrass beds (Robbins 1997; Ward *et al.* 2003). With the ever-growing archive of such images, there is also a possibility for comparing present eelgrass bed size and location with different historical time points. However, interpretation is limited by the scale and resolution of the image as well as the conditions during image capture. Eelgrass beds are most clearly visible when the tide is low, the water is clear and calm, and the sun is not directly overhead (U.S. NOAA Coastal Services Centre 2001). Unfortunately, a special flight is often required to fulfill these conditions, and archival imagery originally captured for other purposes is frequently unfit for mapping purposes; as a result, data acquisition is often expensive and comparison with archival data is not always an option. Data obtained from remotely sensed imagery must also be ground truthed in order to validate the image interpretation, so some fieldwork must be budgeted into the project. This approach has successfully documented loss of eelgrass cover in the San Juan Islands (Wyllie-Echeverria *et al.* 2005).

Another remote sensing technique is using acoustic methods to map underwater habitats, based on the premise that different benthic surfaces have different acoustic signatures (Pasqualini *et al.* 1998). The data, obtained in swaths from adjacent boat transects, is costly to obtain but provides precise, fine scale georeferenced data that can create a very detailed map with minimal interpolation. The quality of the map depends on the interpretation of the acoustic signatures and again requires ground truthing to ensure accuracy.

Video imagery is a third remote sensing option for seagrass mapping (Norris *et al.* 1997). In this case, a camera is towed behind a boat to record transects of the benthos and georeferenced from an on-board GPS unit. This is the most direct type of observation behind diving, and provides information not only on the distribution of the eelgrass but also the quality of the eelgrass (e.g., epiphyte load, density) and the fauna inhabiting the bed. It can be labour intensive as the boat speed must be kept very slow and a fairly dense grid of transects should be recorded in order to maximize the resolution of the dataset. The quality of the imagery captured is also dependent on water clarity and tidal currents. However, this is a cost-efficient method of capturing imagery for eelgrass mapping. Underwater video is most useful for mapping small, shallow, sheltered areas, and it is the approach used for this study (Norris *et al.* 1997).

3.1.3 - Objective and hypothesis

The objective of this component of the study is to test if there is a difference in eelgrass bed size and patchiness between pre-boating and post-boating observations. The null hypothesis is that there is no significant difference in eelgrass bed size or patchiness between sampling periods.

3.2 - Methods

3.2.1 - Field work

A grid of underwater video transects was recorded with a custom built videocamera, towed at 1 knot from the *Pacific Legacy* (a 33 foot Titan 3000 pilot with twin outboard 225hp engines). Feed from the camera was monitored on a 13-inch television/VCR combo in order to maintain a relatively constant height above the bottom (approximately 1m). Overlaid on the screen was a stamp of the date, time, coordinates (from a handheld DGPS input), and depth/temperature (from sensors mounted in the underwater camera housing). The image was captured on VHS tape (from the built in VCR) or digital video tape. The flowchart of this setup is indicated in Figure 3.1. Initial transects were recorded at each site in May 2004, and saved GPS tracks were repeated in September 2004, with the intention of comparing the spatial data from before (May) and after (September) the main boating season.

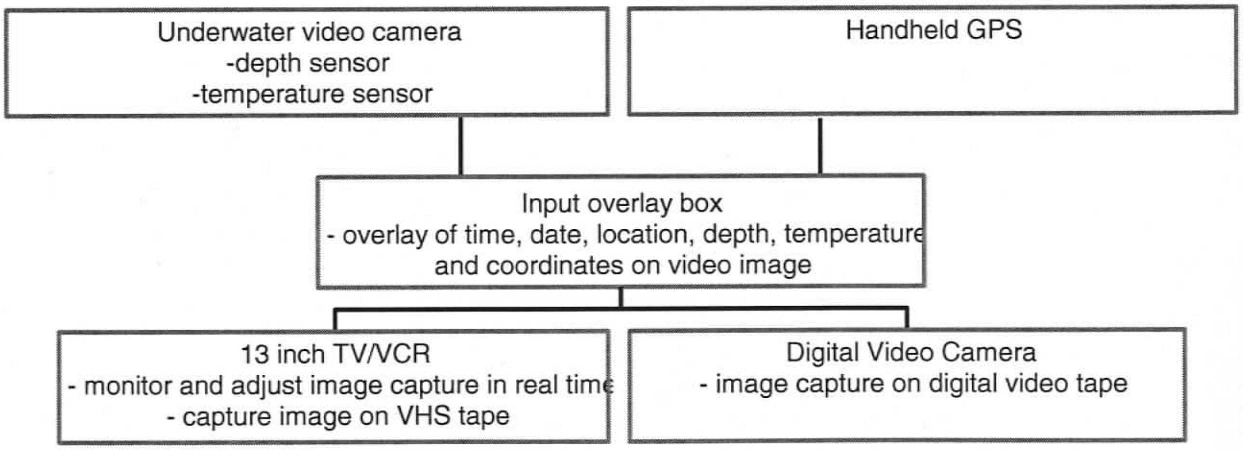

Figure 3.1 - Flow chart of image capture setup

3.2.2 - Image interpretation

Video was manually coded by freezing the tape at 20 second (approximately 10m) intervals and recording information from the overlay into an Excel spreadsheet as well as presence or absence of eelgrass. Depths were corrected to account for tidal height using the online hydrographic survey charts (CHS 2005). Coordinates were imported into ArcGIS as XY point files, and other fields were added in attribute tables. All layers were saved in a personal geodatabase using the NAD27 datum and the UTM coordinate system.

3.2.3 - Spatial analysis

Since the transect data points were not evenly distributed through the study site (Figures 3.2 and 3.3), interpolation of the eelgrass presence/absence points presented a challenge. Indicator kriging is an interpolation technique designed for binary data (such as eelgrass presence/absence), creating a surface predicting the probability of eelgrass occurrence at any given point (Myers 1997). The kriging algorithm incorporates data from twenty closest neighbouring points along axes parallel and perpendicular to the transect direction (weighting closer points more heavily) when calculating the probability of eelgrass occurrence for any given point on the prediction surface. The algorithm can also account for anisotropy in a dataset, and is therefore appropriate for use with the linear data distribution obtained from transects used in this study. The algorithm calculates an error prediction for every point on the probability surface as well as a global root mean square error, which allows the researcher to judge the validity of the generated probability surface. Therefore, the boundary of the eelgrass polygon generated by this interpolation technique is more meaningful and reliable than the boundary indicated by the Thiessen polygon method.

The original intention of this study was to overlay the eelgrass layers from before and after the boating season to test whether anchoring is causing gap formation as observed by Walker *et al.* (1989) and Hastings *et al.* (1995). However, the density of the data points was not high enough to draw meaningful conclusions about changes in the eelgrass beds over the summer, an issue that will be addressed in the discussion of this chapter. According to Fuller *et al.* (2003), 90% accuracy is required in each layer in order to make comparisons with 75% certainty. Since the interpolation of the transect points does not achieve 90% accuracy in either layer, it is impossible to do a statistical comparison between pre- and post-boating season eelgrass beds. Therefore, the raw data points were used for simple visual comparison of eelgrass presence/absence between time points, and all May and September data points were used together in the interpolation to get an "average" eelgrass distribution over the study period.

3.3 - Results

3.3.1 - Raw data points

Figures 3.2 and 3.3 indicate the points where eelgrass was observed to be present and absent during each of the sampling periods. Eelgrass absence was defined as the complete absence of any eelgrass shoots visible in the paused frame. A simple visual inspection of differences in eelgrass distribution between months reveals only minor changes from May to September. These differences are most likely within the acceptable range of sampling error and will be addressed in the discussion.

3.3.2 - Interpolation of eelgrass polygon

The indicator kriging interpolation is displayed in figures 3.4 and 3.5. Figure 3.4 shows the 90%, 75%, and 50% eelgrass probability contours overlaid with the presence/absence data points at each site. From these figures, the 50% probability contour was selected as the best fit for the point data. Figure 3.5 displays the 50% eelgrass contour with the bathymetry at each site. The boundary of the mapped area at each site was determined by prediction error contours from the interpolation (<35% at Sidney Spit and <20% at Tumbo Island). Although there were more points in the interpolation at Sidney Spit than Tumbo Island, there was a lower prediction error at Tumbo Island than at Sidney Spit which can be attributed to the higher data point density at Tumbo Island (Table 3.1). The mean standard error, root mean square error, and standardized root mean square error of the kriging interpolation were all less than 1m at both sites.

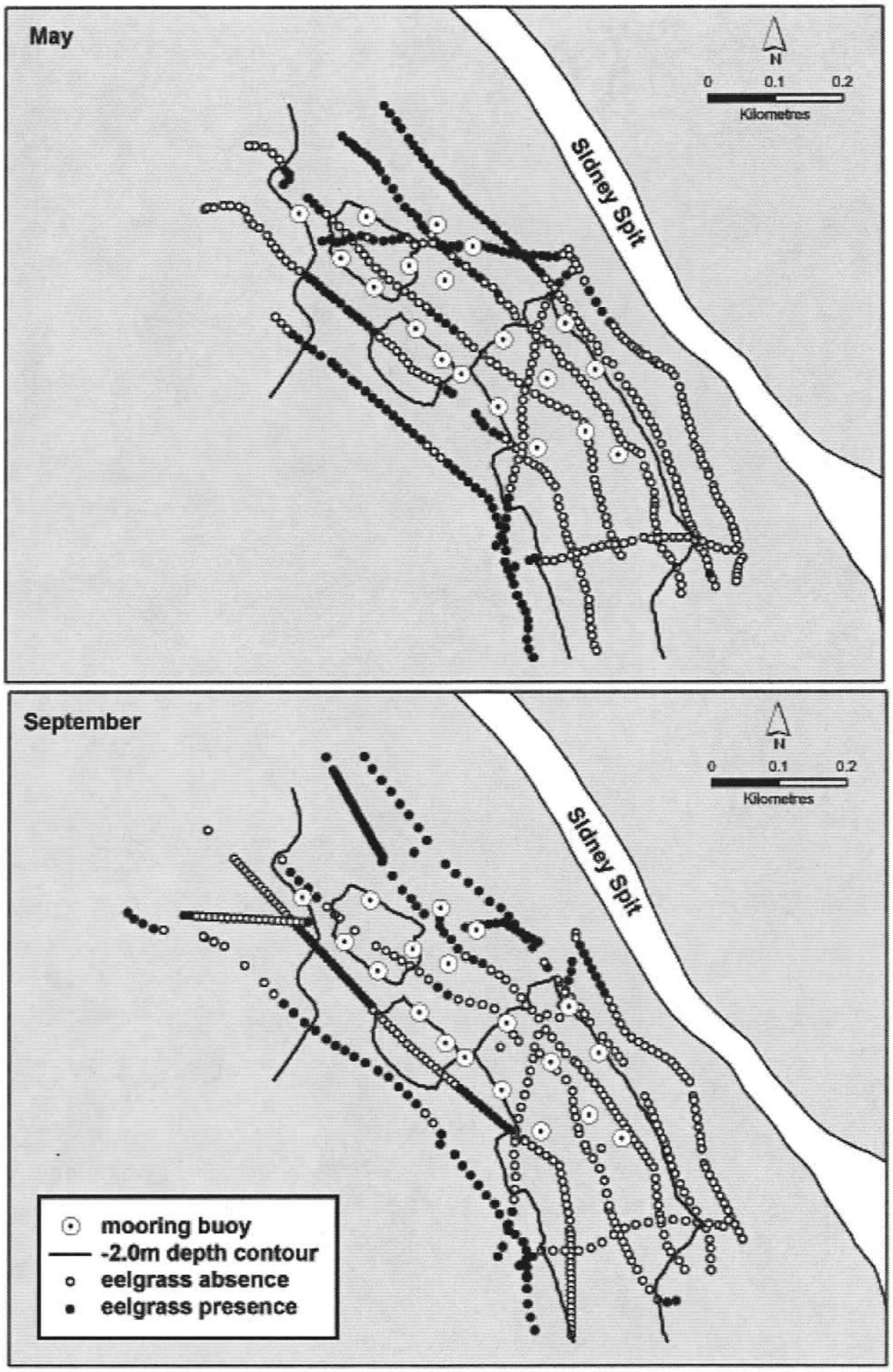

Figure 3.2 - Raw transect data points indicating presence and absence of eelgrass at Sidney Spit in May and September, 2004.

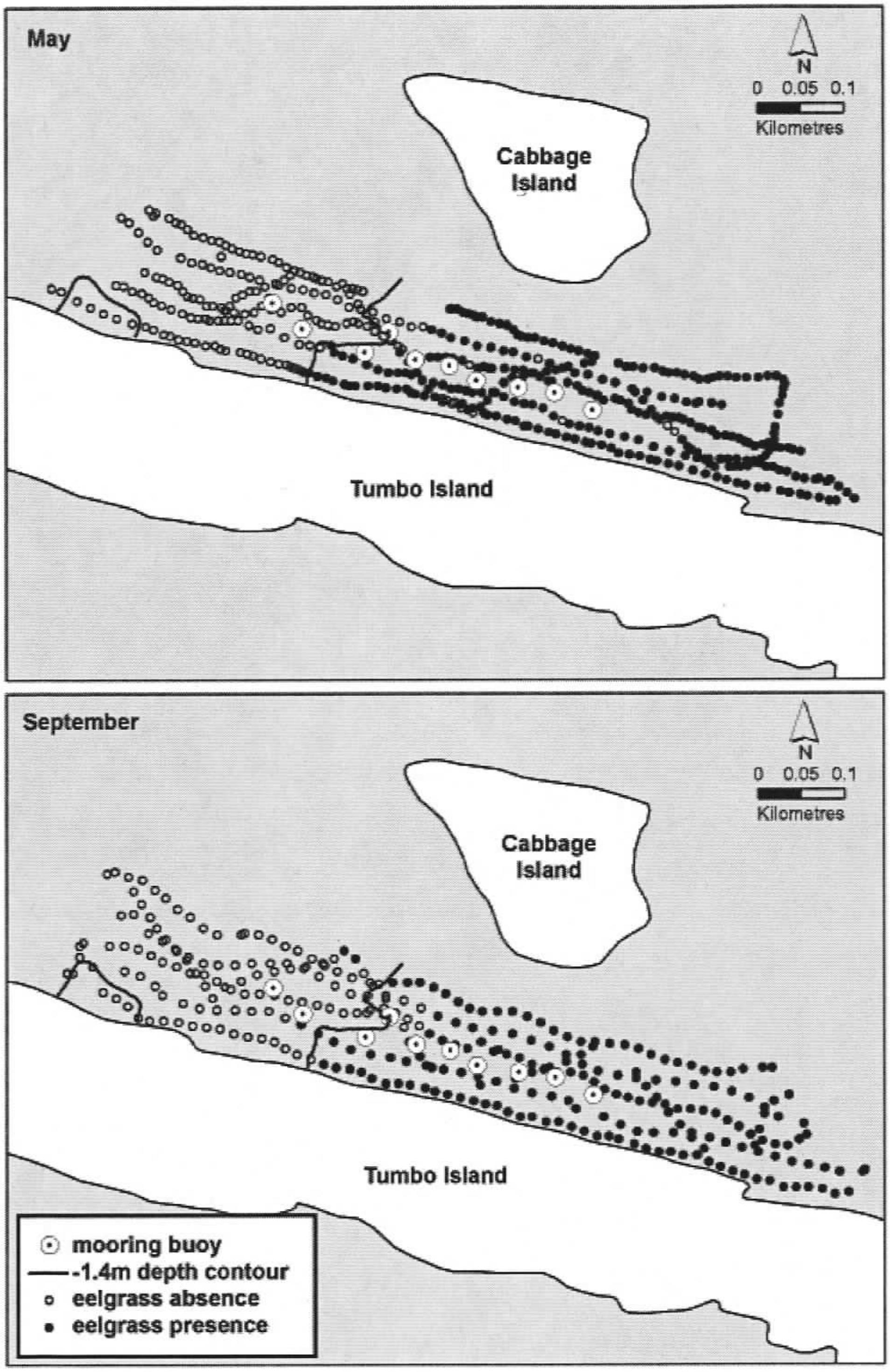

Figure 3.3 - Raw transect data points indicating presence and absence of eelgrass at Tumbo Island in May and September, 2004.

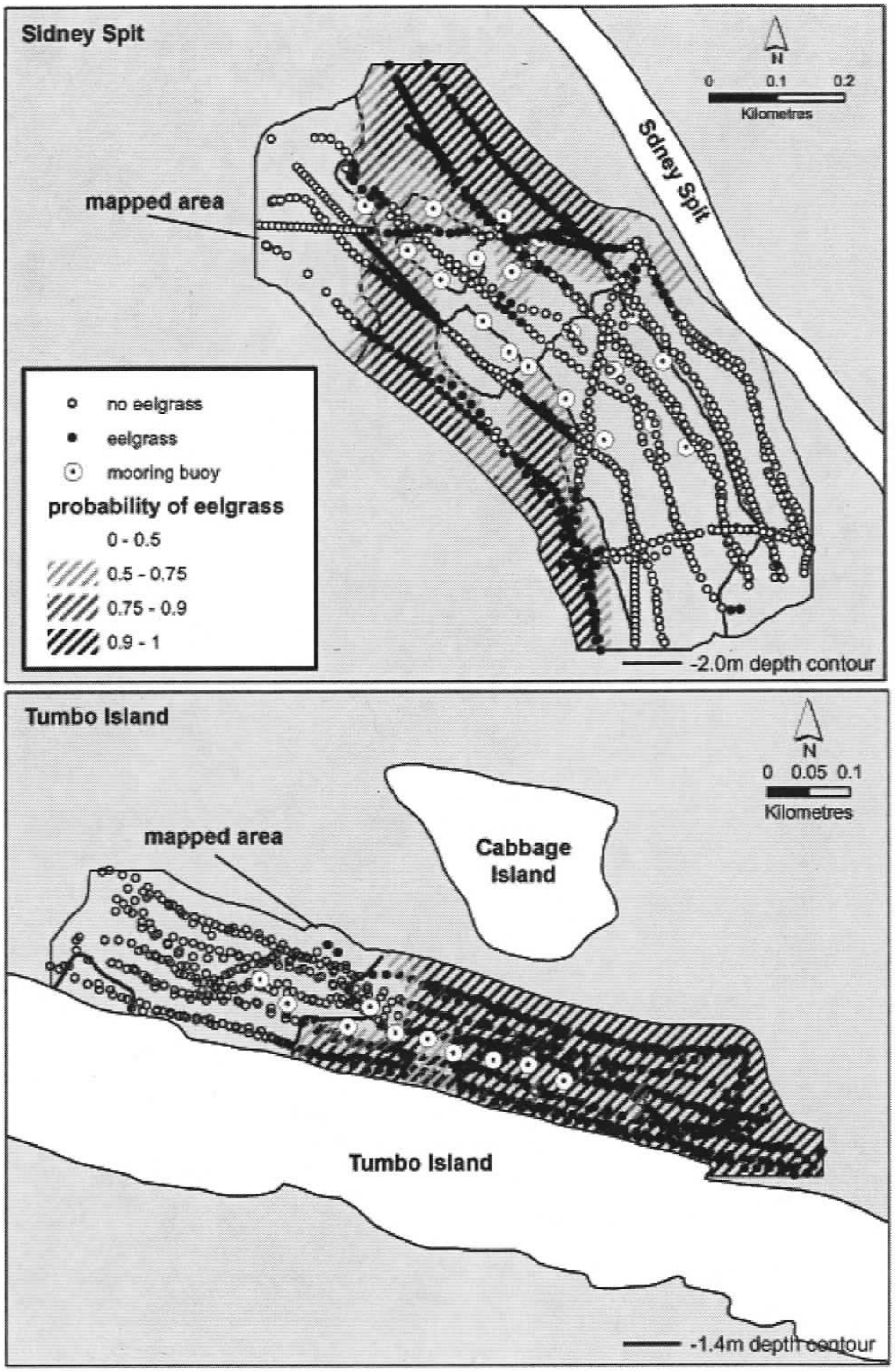

Figure 3.4 - Overlay of eelgrass presence/absence points on contours of eelgrass probability generated from indicator kriging (using points from both sampling months)

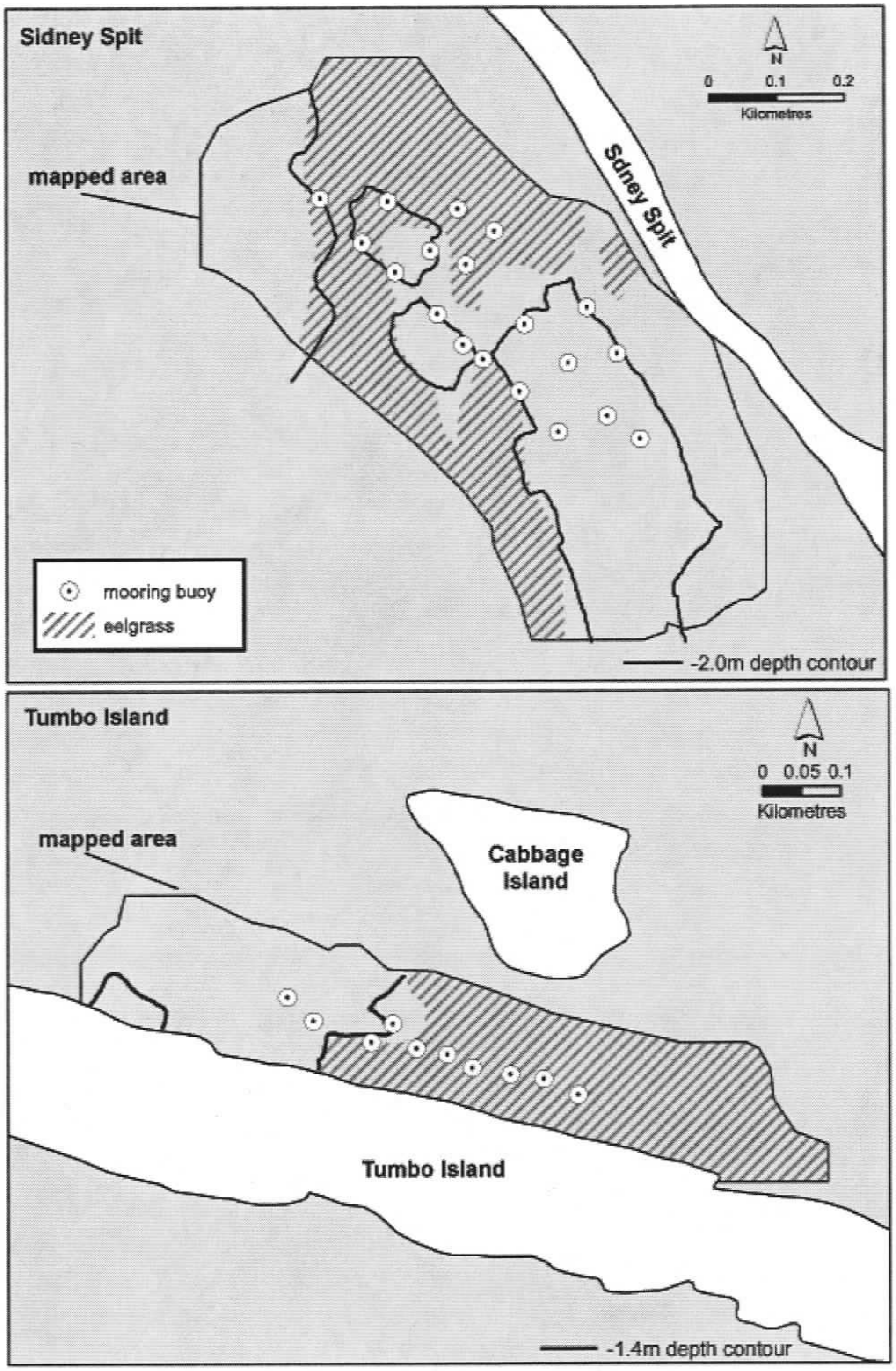

Figure 3.5 - Final eelgrass polygons (50% probability contour) at Sidney Spit and Tumbo Island.

Table 3.1 - Details of the kriging interpolation analysis

	Sidney Spit	**Tumbo Island**
Number of points	984	759
Prediction error	<35%	<20%
Average Standard Error	0.3954m	0.2231m
RMS	0.2423m	0.1679m
RMS Standardized	0.6117m	0.752m
Total area mapped	0.403km^2	0.158km^2

3.4 - Discussion

Visual comparison shows very little difference between the distribution of eelgrass before and after the boating season. The small qualitative differences in absence/presence of eelgrass can likely be attributed to sampling error and small shifts in the boundaries of the bed over the summer growth season. One point to note is that eelgrass distribution is confined to depths shallower than approximately 2.0m below chart datum at Sidney Spit and 1.4m below chart datum at Tumbo Island. This suggests that light availability, limited by water depth and clarity, is a strong factor in determining eelgrass distribution at both sites, which is consistent with the literature (Dennison 1987). The fact that eelgrass is restricted to slightly shallower depths at Tumbo Island (by 0.6m) is possibly due to the closer proximity of this site to the Fraser River delta. Input of nutrients and sediments from the Fraser watershed would be diluted somewhat by the time they reached Sidney Spit, resulting in clearer water and deeper penetration of light. Further research on water quality in the region (nutrient analysis and Secchi disk) should be conducted to verify this speculation.

Also, it is important to note that the edge of the mapped area is not the limit of the eelgrass bed at either site. At Tumbo Island, the eelgrass bed was observed to extend into the shallow area directly east of the mooring buoys. At Sidney Spit, the bathymetry was more variable, but again, eelgrass was observed to continue south of the mooring area into the lagoon.

Although the maps generated from the underwater video data were sufficient to establish general locations of eelgrass for invertebrate sampling, further quantitative assessment of the eelgrass density or epiphyte load was difficult due to the nature of the image capture. The camera, housed in a rectangular stainless steel casing, was not designed to be towed behind a boat and was not hydrodynamic - it was difficult to keep a consistent camera angle and distance above the bottom. There was no scale of reference to define the size of the field of view, the angle of view varied with the transect and current directions, and water clarity was poor at times. Improving the equipment would allow more data to be extracted from the video imagery. A more hydrodynamic camera with a stabilizing fin would allow the operator to keep the field of view more consistent. A scale to define the size of the field of view (such as a pair of laser pointers mounted at a known width) would allow for shoot density measurements (given the angle of view stays relatively consistent). Commercial underwater videography systems manufactured for sport fishing are quite appropriate to this application: the cameras are very hydrodynamic and easy to manipulate, the camera, cable and screen can be self-contained in a waterproof case, the system can run off a 12 volt DC battery, and the GPS overlay can still be stamped on to the recorded image.

In order to collect data that could quantitatively detect changes in eelgrass patchiness at a scale smaller than 10m caused by recreational boat anchoring, a much finer grid of data points would be required, such as that in the study by Norris *et al.* (1997) that had a grid of 13 by 15 transects covering 0.041 square kilometers. The authors found that a grid of this density can identify changes in eelgrass cover as small as 5%. The two sites in this study covered approximately 0.55 square kilometers, with six transects at each site taking approximately one hour each to record. Since the boat speed cannot exceed one knot without losing image quality, increasing the transect density would be a time intensive endeavour and was not feasible within the scope of this study (where the priority was to establish a baseline of eelgrass locations at each anchoring site). However, the benefits of increased transect density are demonstrated in the results of this study: the Tumbo Island site was less than half the size of the Sidney Spit site but had a similar number of data points identified from the transects. As a result, the data points are closer together and the prediction accuracy was 80% at Tumbo Island in the kriging interpolation as opposed to only 65% at Sidney Spit.

Another source of error in this mapping technique was the offset of the camera towed behind the boat from the DGPS located onboard the boat, which varied depending on the depth at which the camera was towed. It would be most accurate to have the georeferencing device mounted directly on the camera. Unfortunately, the band that GPS operates on does not propagate underwater; an underwater pinger mounted on the camera in communication with an array of surface buoys that are in turn located by standard GPS is a reasonable alternative (Sgorbini *et al.* 2002).

Qualitatively, the eelgrass at both sites was observed to be discontinuous with small patches 1-2 m wide), a high diatomaceous epiphyte load and many algal macrophytes growing among the eelgrass shoots compared to beds in Clayoquot Sound and Gwaii Hanaas (pers. comm., Cliff Robinson), regions with much lower human population. High epiphyte loads shade the bottom of the eelgrass bed, limiting light availability and resulting in decreased shoot density and overall bed area (Hauxwell *et al.* 2003). This loss is thought to be related to decreased shoot recruitment and increased shoot mortality, and has been linked to substantial (80-96%) to complete eelgrass loss in the Waquoit bay of coastal Massachusetts (Hauxwell *et al.* 2003). Increased turbidity and sediment suspension would also decrease light availability, leading to the same result (Duarte 2002). Cardoso et al. (1998) agree that the sustained effects of eutrophication (causing high epiphyte/macrophyte load) will lead to complete eelgrass loss and replacement by coarser sediment, green macroalgae, and opportunistic invertebrates.

Eelgrass is known to thrive in nutrient poor waters (Duarte 2002), and nutrient levels in the Southern Strait of Georgia are known to be naturally high (anthropogenic sources of nitrogen are not considered to be significant in comparison with natural inputs (Mackas and Harrison 1997)). Therefore, the strong human presence in the Southern Strait of Georgia cannot necessarily be held responsible for the high epiphyte load observed at these two sites.

3.5 - Summary

Boundaries of eelgrass beds are in constant flux, balancing seasonal growth with loss from sedimentation or physical disturbance. The underwater video imagery captured for this study served the purpose of coarse mapping for identification of general eelgrass location at the two study sites and selecting sampling locations for other components of the study. However, sufficient error was introduced in data capture, image interpretation and interpolation that quantitative measures of eelgrass patchiness at the 10m scale were not feasible in this study, and the null hypothesis that there was no significant difference in eelgrass bed boundaries before and after the main boating season was not disproved. Improving the image capture equipment quality and the transect density are two critical steps to improve the study and properly test the quantitative hypothesis. On a qualitative level, eelgrass beds at both sites were observed to exhibit characteristics of poor health, including small-scale patchiness and high epiphyte/macrophyte load.

CHAPTER FOUR: FREQUENCY AND LOCATION OF ANCHORING AND MOORING ACTIVITY

4.1 - Introduction

Uprooting of seagrass shoots by recreational boat anchors has been directly observed by Milazzo *et al.* (2004) and Francour *et al.* (1999), as well as indirectly by (Walker *et al.* 1989; Hastings *et al.* 1995). Based on the findings of Francour *et al.* (1999), anchoring in seagrass beds results in fragmentation and loss in shoot density, with areas of weak rhizome mat being at particular risk. Reduction of anchoring pressure in seagrass beds is also reported to allow recovery from this damage (Francour *et al.* 1999). Salm *et al.* (2000) suggests that in order to protect benthic communities, site management plans could include a combination of boater education programs, installation of mooring buoys, restoration of damaged sites, and periodic closures of heavily used areas.

Mooring buoys are a key strategy used in the Great Barrier Reef Marine Park to prevent damage to coral reefs from recreational boat anchors (Great Barrier Reef Marine Park Authority 2003). The GINPR has installed mooring buoys at three sites (Sidney Spit, Tumbo Island (also known as Cabbage Island), and South Pender Island) which are available at a nightly fee of ten dollars (there is no charge for anchoring).

Having a better understanding of how boats are using those anchoring/mooring sites will open up management opportunities for park staff to reduce anchoring in eelgrass beds and the resulting loss of eelgrass density. The original intention of this component of the study was to test the correlation between anchoring intensity and fragmentation observed in the mapping component of the study. However, since the resolution of the data did not allow a quantitative analysis of eelgrass fragmentation, the revised goal of this chapter is to describe the types of boats making overnight use at the study sites as well how the boaters are using the sites (both the mooring buoys and the anchoring areas). This is mainly a descriptive endeavour; results will be used to identify possible management strategies to reduce anchoring in eelgrass beds in the GINPR.

4.2 - Methods

Datasheets (See Appendix One) were distributed to facility operators (contract staff employed by Parks Canada to collect user fees – different facility operators worked at each site). Staff were instructed to inspect sites in the evening (7-9PM, a time by which most boaters would be in for the evening) and mark the location and type of each anchored and moored boat on a map, as well as the observation time and the weather conditions. Boat classes were established as follows: small sailboats (<30ft), medium sailboats (30-40ft), large sailboats (>40 ft), small powerboats (<20ft), medium powerboats (20-30ft) and large powerboats (>30ft) (Pers. Comm., Phil Dearden). However, no instructions were given regarding the frequency of data collection; observations were recorded daily during the month of July, 2005, at Sidney Spit (thirty one completed datasheets) and randomly throughout the months of July and August, 2005, at Tumbo Island (thirty completed datasheets) (Table 4.1). Maps were scanned and imported to ArcGIS. Each map was warped to match the geodatabase maps with the

ArcMap georeferencing extension, with control points selected from the mooring buoys and coastline. Boat locations were then digitized and accompanying data for each point were entered into an attribute table. Anchoring density was mapped by drawing a 50x50m grid over the site and recording the number of anchored boats observed in each cell. Statistical and graphical analysis of attribute data was carried out in SPSS 13.0.

Table 4.1 - Distribution of data collection days through the recording period

	Days of data recording		
Site	**Weekday**	**Weekend**	**Total**
Sidney Spit	17	14	31
Tumbo Island	12	18	30

4.3 - Results

A total of 673 boats were observed at Sidney Spit (465 moored, 208 anchored) (Table 4.2a), and 336 boats were observed at Tumbo Island (218 moored, 117 anchored) (Table 4.2b). The highest mooring buoy usages and anchoring densities were observed at Sidney Spit. The maximum buoy usage and boat anchoring density at Sidney Spit were 29 uses/month and 14 boats/0.25km^2/month, respectively, in comparison to only 25 uses/month and 8 boats/0.25km^2/month at Tumbo Island (Figure 4.1). The maximum number of boats (43 at Sidney Spit, 22 at Tumbo Island) was observed on a Saturday at both locations (Tables 4.2a and 4.2b).

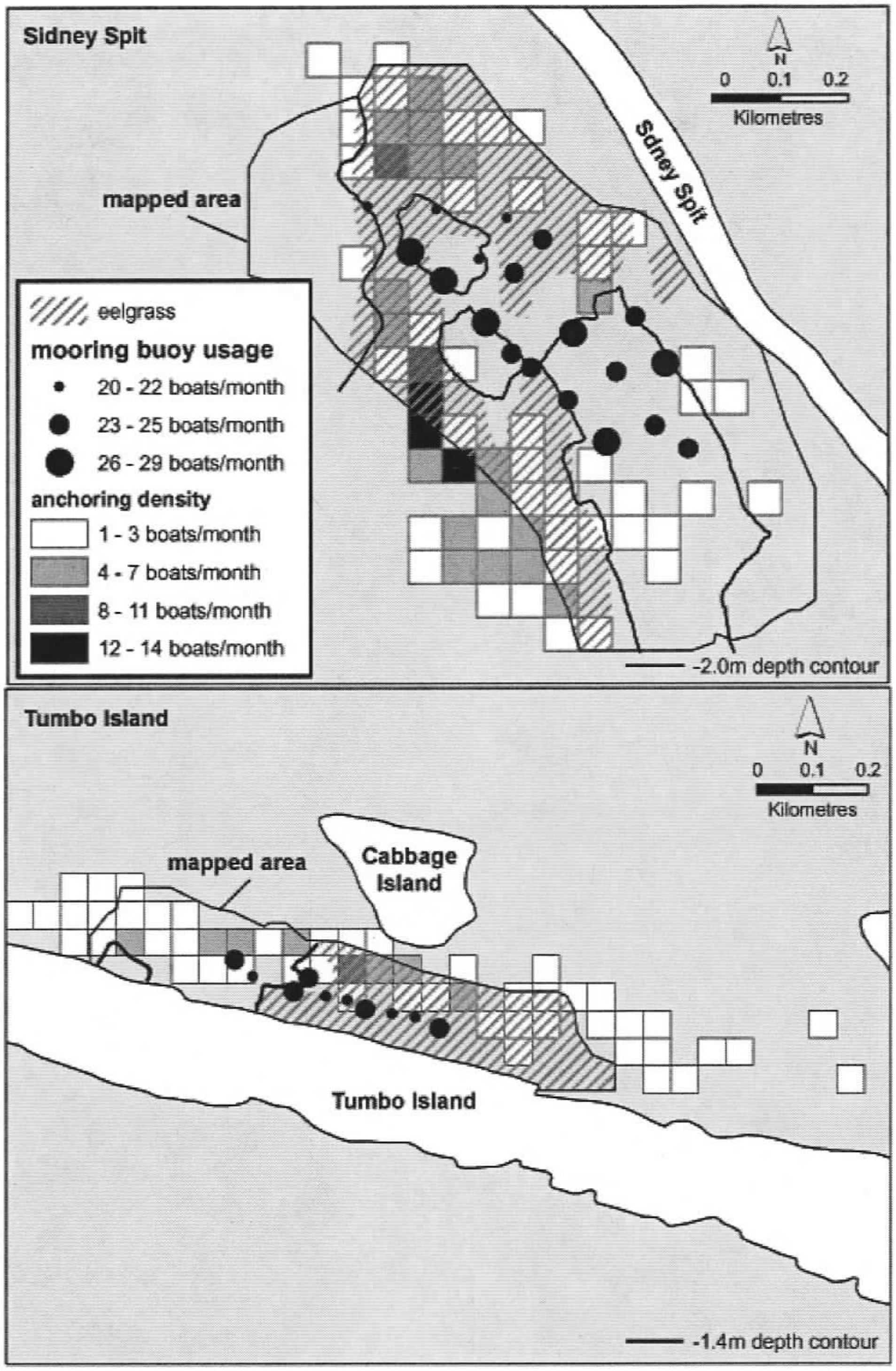

Figure 4.1 - Map of boat usage at Sidney Spit and Tumbo Island, showing mooring buoy usage and anchoring density

Table 4.2a - Dates of boat anchoring data collection at Sidney Spit. * denotes the highest recorded number of boats.

Date	Day	Total Boats	Moored	Anchored	Available Buoys
01-Jul-04	Thursday	28	16	12	3
02-Jul-04	Friday	11	11	0	8
03-Jul-04	Saturday	32	19	13	0
04-Jul-04	Sunday	21	13	8	6
05-Jul-04	Monday	24	19	5	0
06-Jul-04	Tuesday	7	7	0	12
07-Jul-04	Wednesday	10	8	2	11
08-Jul-04	Thursday	9	5	4	14
09-Jul-04	Friday	15	12	3	7
10-Jul-04	Saturday	24	18	6	1
11-Jul-04	Sunday	19	16	3	3
12-Jul-04	Monday	24	19	5	0
13-Jul-04	Tuesday	20	18	2	1
14-Jul-04	Wednesday	23	15	8	4
15-Jul-04	Thursday	13	6	7	13
16-Jul-04	Friday	31	19	12	0
17-Jul-04	Saturday	19	16	3	3
18-Jul-04	Sunday	16	12	4	7
19-Jul-04	Monday	23	18	5	1
20-Jul-04	Tuesday	19	16	3	3
21-Jul-04	Wednesday	20	13	7	6
22-Jul-04	Thursday	25	18	7	1
23-Jul-04	Friday	30	19	11	0
24-Jul-04	Saturday	42*	19	23	0
25-Jul-04	Sunday	24	18	6	1
26-Jul-04	Monday	17	13	4	6
27-Jul-04	Tuesday	16	12	4	7
28-Jul-04	Wednesday	27	17	10	2
29-Jul-04	Thursday	32	19	13	0
30-Jul-04	Friday	26	17	9	2
31-Jul-04	Saturday	26	17	9	2
	TOTAL	**673**	**465**	**208**	

Table 4.2b - Dates of boat anchoring data collection at Tumbo Island. * denotes the highest recorded number of boats.

Date	Day	Total Boats	Moored	Anchored	Available Buoys
02-Jul-04	Friday	10	6	4	4
03-Jul-04	Saturday	14	8	6	2
08-Jul-04	Thursday	8	7	1	3
09-Jul-04	Friday	14	9	5	1
10-Jul-04	Saturday	10	7	3	3
12-Jul-04	Monday	6	6	0	4
16-Jul-04	Friday	15	10	5	0
17-Jul-04	Saturday	22*	10	12	0
18-Jul-04	Sunday	8	4	4	6
20-Jul-04	Tuesday	12	9	3	1
24-Jul-04	Saturday	21	10	11	0
27-Jul-04	Tuesday	9	8	1	2
30-Jul-04	Friday	16	10	5	0
31-Jul-04	Saturday	15	9	6	1
01-Aug-04	Sunday	14	8	6	2
03-Aug-04	Tuesday	5	5	0	5
04-Aug-04	Wednesday	11	9	2	1
07-Aug-04	Saturday	9	7	2	3
08-Aug-04	Sunday	10	9	1	1
10-Aug-04	Tuesday	13	10	3	0
14-Aug-04	Saturday	17	9	8	1
15-Aug-04	Sunday	15	8	7	2
16-Aug-04	Monday	16	10	6	0
19-Aug-04	Thursday	9	7	2	3
20-Aug-04	Friday	9	4	5	6
23-Aug-04	Monday	4	3	1	7
26-Aug-04	Thursday	4	0	4	0
27-Aug-04	Friday	11	10	1	0
29-Aug-04	Sunday	2	1	1	9
30-Aug-04	Monday	7	5	2	5
	TOTAL	**336**	**218**	**117**	

At Tumbo Island, the highest anchoring density was observed directly north of the mooring buoy area, directly beside the onshore facilities (Figure 4.1). It was interesting to observe that at Tumbo Island, there were boats anchoring quite far east of the mooring area in the very shallow part of the bay. Although this portion of the site was not mapped via underwater video, the shallow depth and protection from waves provide ideal eelgrass habitat. Indeed, it was noted during field work that the eelgrass bed was continuous in this area.

At Sidney Spit, the mooring buoy area coincided with a channel that is slightly deeper than the surrounding area; accordingly, very little eelgrass was observed under the mooring buoy area and most eelgrass was observed in areas available for anchoring (Figure 4.1). In contrast, a larger proportion of the available anchoring area at Tumbo Island was free of eelgrass. As a result, 85% percents of boats anchoring inside the mapped area were observed to anchor over eelgrass beds, compared to only 47% at Tumbo Island (Table 4.4)

Table 4.3 - Percentage of anchoring occurring in mapped eelgrass beds

	Sidney Spit	**Tumbo Island**
Cells with observed anchoring	65	55
Number of cells with observed anchoring within mapped area	42	30
Number of boats observed anchoring in mapped area	123	79
Number of boats observed anchoring in mapped eelgrass	105	35
Percentage of anchoring in mapped area occurring in eelgrass	85%	44%

It was not unexpected to observe a large difference in the type and size of boats using the two different anchoring sites considering the differences in accessibility from the Victoria area. At Sidney Spit, the largest user group was small sailboaters (likely people taking a short trip from the Victoria area), whereas at Tumbo Island the largest user group was large powerboaters (likely people stopping over as part of a longer trip) (Figure 4.2). Also, it was observed that boaters were still choosing to anchor even when there were still available mooring buoys (Figure 4.3). Although, on average, there were enough buoys to meet the demand of all site users on weeknights (approximately five anchored boats and five available buoys at Sidney Spit, and two anchored boats and two available buoys at Tumbo Island), there were insufficient buoys to meet user demand on weekend nights at both sites (approximately eight anchored boats and three available buoys at Sidney Spit, and five anchored boats and two available buoys at Tumbo Island) (Figure 4.3).

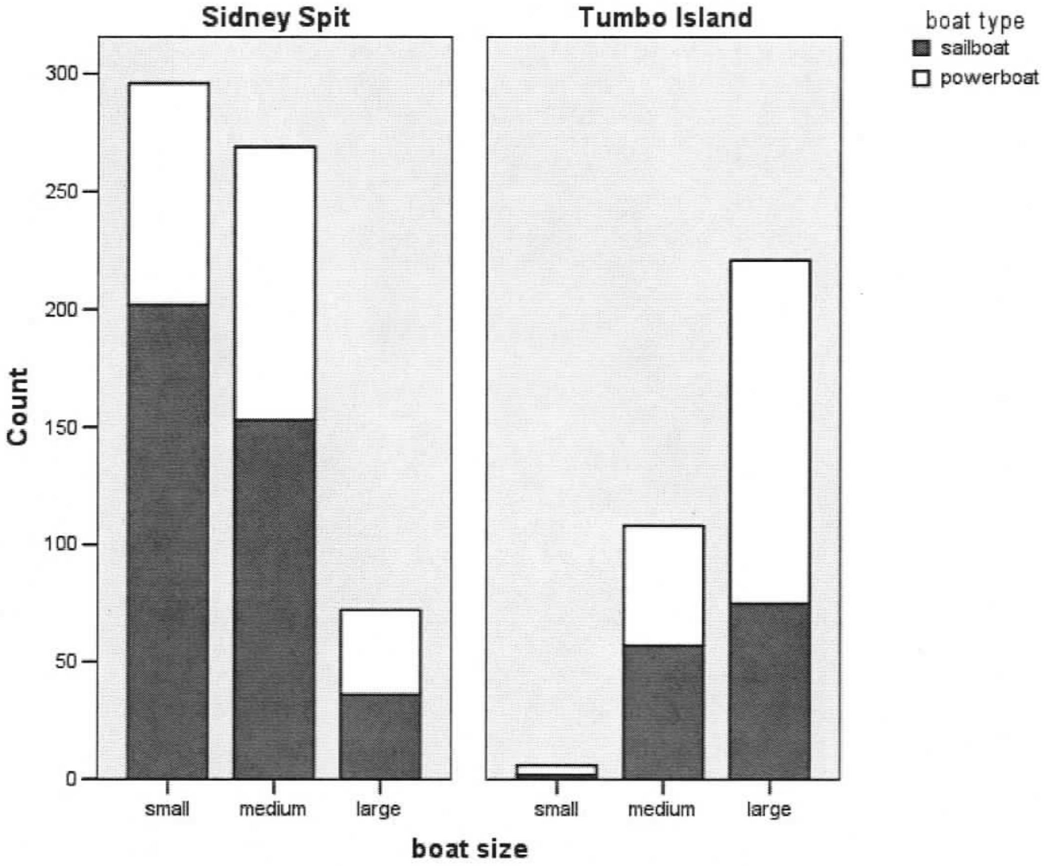

Figure 4.2 - Number of boats observed at Sidney Spit and Tumbo Island, by size and type of boat

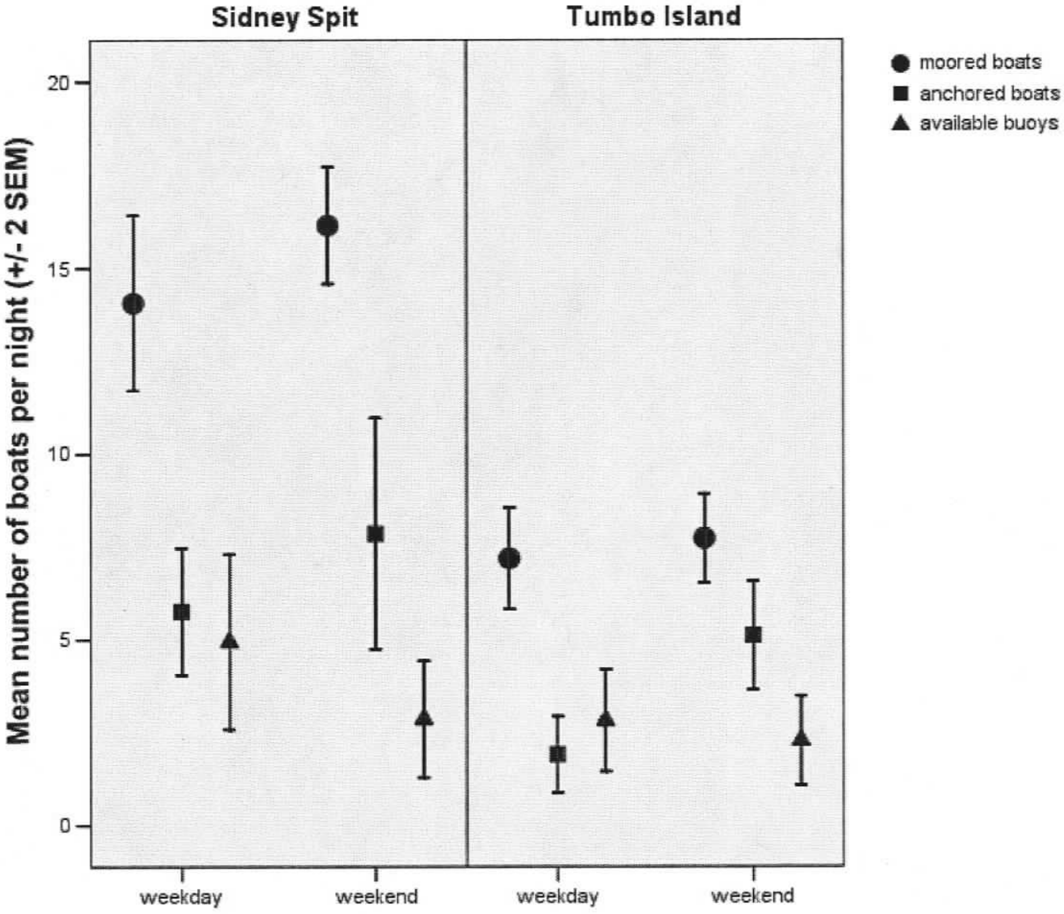

Figure 4.3 - Mean number of anchored boats, moored boats and available mooring buoys on weeknights and weekend nights at Sidney Spit and Tumbo Island

4.4 - Discussion

4.4.1 - General observations

Many boats are anchoring directly over eelgrass at both sites. At Tumbo Island, it was not expected that many boats would be anchoring east of the mooring buoy area due to the shallow water depth and distance from the onshore services. At Sidney Spit, it appears that boaters unable to find an available buoy or choosing not to use a mooring buoy do not have many eelgrass-free areas to anchor in. It was also interesting to see that boaters are still anchoring even when mooring buoys are still available. Clearly there are opportunities to change boater behaviour in order to encourage use of mooring buoys or anchoring in areas away from eelgrass beds.

Reasons why boaters may not be using mooring buoys include not wanting to pay the ten dollar per night mooring fee or not trusting that the mooring buoys are properly maintained by the Park (believing that setting their own anchor will be more secure than a mooring buoy). Boaters are also generally in “holiday mode” and aren’t necessarily interested in being lectured about the importance of eelgrass or told they should move their boat by parks staff after having set anchor for the evening. Therefore it is important to approach any attempt to educate boaters or change their behaviour with a certain degree of caution.

4.4.2 - Voluntary anchoring programs

The Jefferson County Marine Resources Committee has introduced a Voluntary Anchor Protection Zone for the eelgrass beds in Port Townsend Bay in Puget Sound, a self-regulatory approach as described by Antonini and Sidman (1994). Eelgrass bed locations were established through a 2001 shoreline inventory, and marker buoys were installed around the perimeter of the eelgrass beds with signage encouraging boaters to anchor in slightly deeper water outside the shore-fringing eelgrass beds (Jefferson County Marine Resources Committee 2005). The Port Townsend community has strong participation in the project, and has onshore education tools including brochures, dock signage, a demonstration eelgrass bed built at an old pier, and displays at community events detailing the importance of eelgrass beds in the region and the importance of anchoring outside the perimeter of the beds. Initial phases of the project have shown success, with a reduction of anchoring inside eelgrass beds from 20% in 2003 to only 1.4% in 2004 (Pearson *et al.* 2005). A more comprehensive monitoring program is in place that will hopefully show further success in years to come.

The capital expenditure on buoys and signage is a one-time expense (possibly with minor maintenance) and staff time would not be required to make contact with boaters or enforce the policy since it is a voluntary program. Not only are boaters encouraged to be environmentally responsible by choosing a different anchoring location, but are also motivated to anchor elsewhere since anchoring in vegetation such as eelgrass is less secure than bare sediment and can result in dragging. Marker buoys and signage installed at Sidney Spit and Tumbo Island could direct boaters to use anchoring areas south and west of the mooring buoy area, respectively. Signage could also include statements about services funded by mooring fees and mooring buoy quality/maintenance that would alleviate boater concerns about the cost and safety of using mooring buoys.

4.4.3 - Boat and anchor-specific considerations

The type of anchor used by a boat also makes a difference in the amount of physical damage caused to an eelgrass meadow. Milazzo *et al.* (2004) found that Hall type anchors (also known as Navy/Yachtsman anchors) are preferable to Danforth and Folding grapnel anchors, causing significantly less damage to *Posidonia oceanica* shoots. Their results showed that the most critical damage-causing phase was the weighing of the anchor. In contrast, Francour *et al.* (1999) found that the "locking-in" phase of the anchoring cycle uprooted the most shoots. This could be related to the larger boat and heavier anchor tested in the Francour study, which would sink deeper into the seagrass matte and change the dynamics of the anchoring cycle. In any case, encouraging boaters to switch to Hall anchors, or to take care in locking in/weighing their anchors would be another strategy to decrease damage to eelgrass beds.

The fact that Tumbo Island has a different dominant type of user than Sidney Spit is also of note. The larger boats that dominate Tumbo Island have heavier anchors and chains than the small craft that dominate Sidney Spit. Francour *et al.* (1999) found that a 12 kg anchor uprooted 33.5 ± 5.8 shoots per cycle, in comparison to 1.8 ± 0.2 to 5.5 ± 3.5 uprooted shoots per cycle observed in by Milazzo *et al.* (2004) with 4 kg anchors. As a result, the eelgrass beds at Tumbo Island may be subject to higher risk of anchoring damage due to the larger boats frequenting this site. Milazzo *et al.* (2004) suggest that the self-regulatory approach may be feasible with small craft, but that larger craft should be subject to more formal anchoring restrictions, due to their greater potential to cause damage.

4.5 - Summary

This study shows that anchoring currently takes place in eelgrass beds located at Sidney Spit and Tumbo Island. Several management opportunities exist to decrease anchoring pressure on eelgrass beds. These include the following:

1) addressing boater concerns about mooring buoy safety and fees;
2) encouraging boaters to use all available mooring buoys before setting anchor;
3) encouraging boaters to use traditional "Navy"/"Yachtsman" type anchors that are least damaging to seagrasses;
4) educating boaters on the ecological and safety hazards of anchorage inside eelgrass beds; and
5) clearly identifying eelgrass beds at each site and encouraging boaters to anchor outside the margins of these areas.

CHAPTER FIVE: INFLUENCE OF BOAT ANCHORING ON INVERTEBRATE COMMUNITIES

5.1 - Introduction

5.1.1 - Benthic invertebrates as indicators of ecological health

Beyond the simple physical uprooting of eelgrass shoots by boat anchoring, it is important to investigate the consequences of this decrease in structural complexity and sediment disturbance on the communities inhabiting eelgrass beds. As described in section 2.5, benthic invertebrates are a key component of marine ecosystem functioning – therefore, the loss of ecosystem services resulting from poor benthic invertebrate community health can have repercussions at higher trophic levels. Benthic invertebrates are also good indicators of disturbance because:

1) they are relatively sessile and therefore cannot avoid conditions at a given location;
2) they have reasonably long lifespans and integrate the effects of disturbance over time; and
3) they can be classified into functional groups with predictable responses to stress (Dauer 1993).

Analysis of changes in benthic invertebrate communities is often used to detect environmental consequences of heavy metal contamination, petroleum pollutants, organic enrichment, and physical disturbance (Warwick 1993). Biological indicators are particularly useful in a study like this one where it is difficult to directly quantify anchoring disturbance (as opposed to directly quantifying pollutant concentrations). In well-studied areas, an index of benthic biotic integrity can be developed as an environmental monitoring tool to indicate degraded areas of poor sediment and/or water quality in a watershed and assess overall regional health, such as in Chesapeake Bay (Weisberg *et al.* 1997)

Although it does not appear that benthic invertebrates have been used before as an indicator of physical damage from boat anchoring, they are an appropriate choice for this study because they have been previously used as an indicator of physical damage from dredging (Dernie *et al.* 2003)), an indicator of eutrophication of eelgrass beds (Cardoso *et al.* 1998) and an as an indicator of small-scale fragmentation effects (patches of 6-9m^2) in eelgrass beds on a similar spatial scale (<1km) (Frost *et al.* 1999).

5.1.2 - Environmental variables affecting the distribution of benthic invertebrates

However, disturbance is not the only factor structuring invertebrate communities. Distribution of organisms is also influenced by sediment properties, organic content, nutrient availability, and depth, which can confound the analysis of environmental impact studies (Warwick 1993). Water depth (related to light availability and water column pressure) affects organism distribution at large scales (e.g., shallow subtidal vs. deep sea). In this study, all samples were obtained in the shallow subtidal within a 2m range, and therefore depth is not expected to be an important factor influencing community composition. However, sediment properties have been shown to influence larval settling and benthic invertebrate distribution (Snelgrove and Butman 1994) and are of greater interest than depth in this study. These differences are a function of the variation in physical habitat provided by sediment grains of different size, as well as nutrient availability that varies with pore size (a function of sediment size profile) (Erftenmeijer and Koch 2001). Decreased porewater exchange in silt/clay sediments can result in concentration of nutrients and toxins compared to sandy sediments (Kenworthy *et al.* 1982).

5.1.3 - Measures of invertebrate community health and indicator organisms

Healthy communities are generally considered to be those that are stable and undisturbed, with high abundance/biomass, richness (number of taxa represented), and diversity (measure of the number of taxa represented and how evenly they are distributed) (Pearson and Rosenberg 1978). In contrast, disturbed communities are characterized by reduced community biomass, species richness and equilibrium species, as well as increased biomass/abundance of opportunistic species (Dauer 1993). These opportunistic species are usually short lived (nemerteans, nematodes, and polychaetes such as glycerids, capitellids, spionids, and nereids) in comparison to long-lived equilibrium species such as bivalve molluscs, ophiuroids and maldanid polychaetes (Pearson and Rosenberg 1978). The opportunistic species previously listed are also known to be specific indicators of disturbance in eelgrass beds (Frost *et al.* 1999, Cardoso *et al.* 1998). Communities under stress also exhibit increased variability (i.e., high variance about a mean abundance or diversity measure) (Warwick and Clarke 1993) and decreased trophic diversity (Gaston *et al.* 1998). Healthy communities exhibit high ratios of equilibrium species to opportunist species, and high ratios of crustaceans to polychaetes and molluscs (UNESCO 2000). It is also expected that there will be a shift from large bodied to small bodied organisms in response to disturbance, because large-bodied (larger size class) organisms are more susceptible to physical damage (Warwick and Clarke 1994).

Although interpretation of changes in community structure requires training and experience in benthic biology, protocols are now available to analyze community change in a valid, objective way (Clarke 1993). It is also recommended to use a combination of approaches to build support for a hypothesis and approach the problem from different directions rather than relying on a single analysis (Dauer 1993).

5.1.5 - Objective and hypothesis

The objective of this component of the study was to test the impact of boat anchoring on invertebrate communities located at Sidney Spit and Tumbo Island. Unfortunately, finding an appropriate control site to serve as a comparison for an "undisturbed" community was problematic. Logically, at a given anchoring site, all areas of appropriate depth will most likely be used for anchoring. Glasby and Underwood (1998) showed that controls for studying impacts of marinas should be located at least 1.5km away from marinas – Sidney Spit and Tumbo Island are both small anchorages and islands; observing this distance requirement would oblige the selection of control sites at completely different islands in the park. However, to use a completely separate site not affected by anchoring introduces possible differences in external oceanographic conditions that also influence community composition (such as wave exposure, sediment profile, nutrient availability) which would in turn confound the "control" results obtained at this site, making them inappropriate for comparison with the anchoring site.

Therefore, a gradient of anchoring impacts were selected for comparison within each site: low anchoring intensity, high anchoring intensity, and mooring areas, with the mooring areas serving as a proxy reference sites. The temporal sampling comparison was designed so that both sampling periods would be within the "summer" invertebrate community, but would compare community differences from before and after the main summer boating season. If the community composition/health indices at these mooring areas (not subject to the physical impacts of anchoring) remain constant between the pre-/post-anchoring timepoints, then it is reasonable to consider them as references.

The null hypothesis is that there are no significant differences between anchoring intensities or sampling periods. The alternative hypothesis is that the high anchoring intensity communities will be significantly different than the low anchoring intensity and mooring area communities, and that post-boating samples in anchoring areas will be significantly different than pre-boating samples in anchoring areas. The high anchoring intensity/post-boating samples are expected to exhibit the characteristics of poor community health outlined earlier in the introduction.

5.2 - Methods

5.2.1 - Field work

Invertebrate and sediment samples were collected in June and September, 2004, at the locations indicated in Figure 5.1. Sampling station selection was made before the anchoring intensity data were collected, so Parks staff familiar with the study sites were consulted to recommend areas that were likely to have the highest and lowest anchoring intensities. Upon visual inspection of the overlaid data layers, the locations selected for sampling do appear to represent areas of lower and higher anchoring intensity at each site. Now that baseline data for anchoring intensity exist, future sampling could be targeted to known areas of highest and lowest anchoring intensity.

Two different boats were used for sampling: the *Pacific Legacy* (a 33 foot Titan 3000 Pilot) and the *Gulf Islands' Spirit* (a 26 foot Landing Craft). A total of fifteen sample stations were randomly selected during each sampling period, with five samples each in areas of high anchoring intensity, low anchoring intensity, and mooring buoys, as outlined in table 5.1. Samples were obtained with a Petite Ponar grab (surface area = $0.025m^2$). The coordinates, depth and time of each sample were recorded using the onboard GPS and depth sounder (Raystar 120 GPS receiver on both vessels, Raytheon Fishfinder L750 on the *Pacific Legacy* and and Raymarine ST60 Autohelm Series on the *Gulf Islands' Spirit*) - depth was kept as consistent as possible between all samples, though the uneven bottom did not always make this possible. Samples were stored in coolers while on the boat and were processed immediately upon return to the dock.

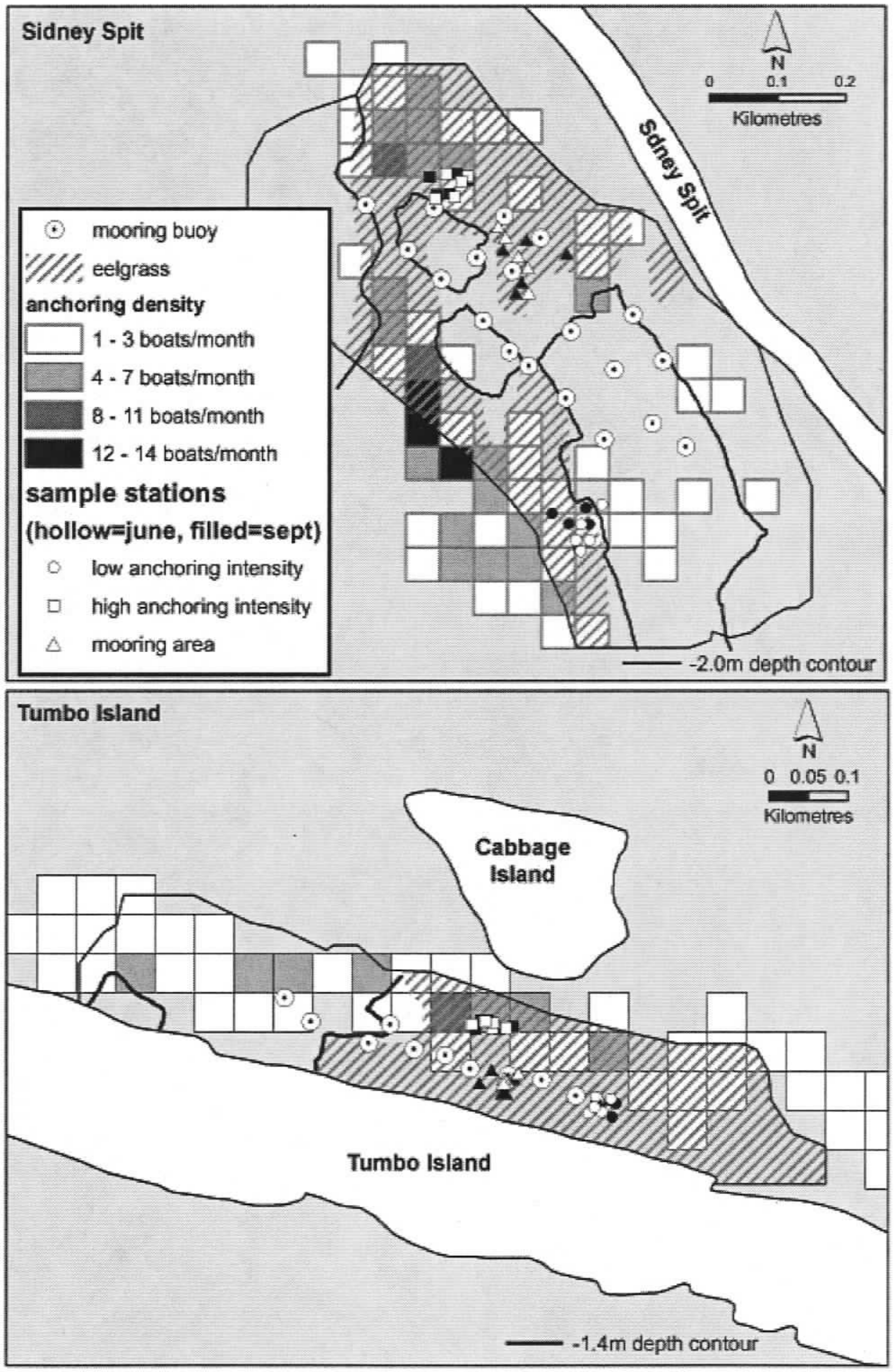

Figure 5.1 - Location of sampling stations at Sidney Spit and Tumbo Island

Table 5.1 - Sampling station layout

	Sidney Spit		**Tumbo Island**	
	June	**September**	**June**	**September**
Low anchoring intensity	5 samples	5 samples	5 samples	5 samples
Mooring area	5 samples	5 samples	5 samples	5 samples
High anchoring intensity	5 samples	5 samples	5 samples	5 samples

Two grabs were obtained at each sample station, one for sediment analysis and one for organism analysis. The grabs intended for sediment analysis were frozen and stored at -4°C. The grabs intended for organism analysis were gently washed through nested 1mm and 0.5mm sieves, creating two size class groups from each sample (>1 mm and 0.5-1 mm). Organisms retained on the screens were washed off with a wash bottle and fixed in 7% buffered formalin with Rose Bengal. Formalin was buffered with 1/4 teaspoon of borax per litre (Nagorsen and Peterson 1980) and Rose Bengal concentration was 4g/L formaldehyde (Eleftheriou and Holme 1984). Rose Bengal stains organic tissue and facilitates sorting of organisms from other debris retained in the sample (Eleftheriou and Holme 1984). After fixation for at least 48 hours, samples were transferred to 70% ethanol for permanent storage.

5.2.2 - Sediment analysis

After thawing, sediment samples were dried overnight at 60 degrees C. 10g subsamples were burned in the muffle furnace at 800 degrees C for 12h in order to determine total organic carbon. The mass difference before and after combustion represents the mass of organic material present in the original sample, allowing the calculation of percent organic content by mass. Particle size distribution was determined with the hydrometer method published by AIMS (1997) (modified from Bouyocous, 1962), based on differential settling speeds of particles suspended in a water column. Settling speed is inversely proportional to particle size (sand (0.063-2mm) settles out of suspension first, then silt (4-63um), and finally clay (<4um)). Sand particles settle out immediately: a hydrometer reading after twenty seconds of settling gives the mass of silt and clay still in suspension. A second reading after two hours give the mass of clay left in suspension after the silt has settled out. The resulting data give the percent by mass of sand, silt, and clay in each sample. This method was selected because the standard dry sieving technique is not suitable for the fine particles making up these sediment samples, and because it facilitated processing a large number of samples quickly.

5.2.3 - Invertebrate identification

Organism samples were sorted and invertebrates identified to family level where possible with the aid of a dissecting microscope. Work by Somerfield and Clarke (1995) shows that it is not necessary to identify all the way to species level because patterns are preserved at higher taxonomic levels, and with higher certainty. Identification resources included Banse and Hobson (1974), Hobson and Banse (1981), Harbo (1997), Kozloff (1996), Coan *et al.* (2000) and personal communication with Mr. Kelly Sendall, manager of the invertebrate collection at the Royal BC Museum. The following is a list of taxa identified in the course of this study:

- **Nematoda**
- **Nemertea**
- **Polychaeta**: Pectinariidae, Neridae, Glyceroidea, Spionidae, Cirratulidae, Aphrodititoidea, Syllidae, Opheliidae, Spiroribidae, Maldanidae, Capitellidae
- **Bivalvia**: Mytilidae, Cardiidae, Veneridae, Tellinidae, Lucinidae, Lacunidae, Littorinidae, Trochidae
- **Gastropoda**: Doridoidea, Columbellidae, Naticidae, Acteonidae, Lottiidae, Cerithiidae, Pyramidellidae
- **Amphipoda**: Gammeridea, Caprelliadea
- **Ostracoda**
- **Copepoda**
- **Other Arthropoda**: Cumacea, Mysidacea, Isopoda, Tanaideae, Crabs, Cirripedia
- **Echinodermata**: Ophiuroidea, Strongylocentrotus, Dendraster

5.2.4 - Statistical analysis

Raw sample depths were corrected to depth below chart datum, using the known sampling time and tidal charts as described in chapter 3. Statistical analyses were performed and charts were generated with the Plymouth Routines In Ecological Multivariate Ecological Research package (PRIMER 5.0) and SPSS 13.0.

The simplest analysis of community differences between anchoring intensities involves comparing the abundance/biomass of individuals and families observed between intensities. Abundance (simple numerical count) and biomass (cumulative mass of individuals in each taxon) are considered to be interchangeable because they generally show the same overall result (Warwick 1993) – in this study, only abundance was measured.

Other measures of community diversity include Pielou's evenness (how evenly different taxa are represented within those individuals) and Shannon diversity (a measure of the number of taxa (richness) as well as the taxon distribution within a group of individuals – a very diverse community will be rich in taxa and will not be dominated by a single taxon). A third measure of diversity is the phylogenetic relatedness of the taxa in a given sample, which is measured by taxonomic distinctness, calculating the average taxonomic distance between each pair of species in a sample (Clarke and Warwick 1998). Larger values indicate that species are less closely related. Variation in taxonomic distinctness is the variance of the pairwise distances, an indication of how evenly the taxa in the sample are distributed (Clarke and Warwick 2001)

These indices can be used to test whether or not the phylogenetic diversity of a given site falls within an expected range of phylogenetic diversities for the sample pool or regional species pool (Clarke and Warwick 1998, 2001). The output consists of a "funnel" showing the expected average taxonomic distinctness and variance for a given number of species in a given pool. Each site is plotted on the graph, and if the site falls outside of the funnel, it is considered to be an outlier. Further attention is warranted to examine the specific species composition at that site and identify what makes it an outlier. A similar funnel plot is also generated comparing the variation in taxonomic distinctness at each site to the average variation in taxonomic distinctness at all sites, which is interpreted the same way. The taxonomic distinctness test is robust to relaxations in sampling design and is not dependent on sampling effort (Clarke and Warwick 1998, 2001). The aggregation table used for the analysis is included in Appendix Three.

Work by Pearson and Rosenberg (1978) and UNESCO (2000) suggests that indicator taxa for disturbed habitats include opportunists such as carnivorous polychaetes, nemerteans and nematodes. In contrast, indicator taxa of stable, undisturbed habitats include tube-dwelling polychates, bivalves, and crustaceans. Indeed, the work of Frost et al. (1999) and Bowden et al. (2001) supports these choices of indicator taxa in eelgrass beds. Therefore, indicator ratios of (neutral + equilibrium):(opportunist) species and (molluscs + crustaceans):(polychaetes) will be used to as a measure of disturbance in this study. Higher ratios indicate less disturbed communities.

The multivariate approach requires the calculation of a matrix comparing the similarity in invertebrate community composition of each size class at all sample stations. All multivariate analyses were performed at the family level (the list of families observed is included in Appendix Three). The Bray-Curtis similarity measure was selected for this analysis, and is commonly used in ecological studies (Bray and Curtis 1957). The Analysis of Similarity (ANOSIM) test (Clarke 1993), a non-metric analog of the Analysis of Variance test comparing rank similarities between factor groups, was used to test the significance of differences in community assemblages between anchoring intensities.

Interactive effects of site, season, and organism size class on differences in community assemblages between anchoring intensities were tested with the nested ANOSIM (Clarke 1993). In the absence of interactive effects, sample data could be pooled or averaged among factors to improve power in testing the differences between anchoring intensities. Mulitdimensional scaling (MDS) is a non-metric tool that allows visualization of inter-site similarities by plotting between-site "distances" (rank-order similarities from the Bray-Curtis similarity matrix) (Clarke 1993). Sites that are most similar are plotted closest together and sites that are least similar are plotted furthest away from each other, allowing the identification of clusters of similar sites. Of course, there is not always perfect agreement between rank orders, so significance of the clustering pattern is described with a Kruskal-Wallis stress value, comparing the plotted inter-site distances with the known correct inter-site distances. Stress values of less than 0.1 are considered to indicate clustering patterns that are significantly different from random (Quinn and Keough 2002). These MDS plots can be compared to more traditional cluster diagram in order to validate the observed clustering patterns.

In order to identify the taxa that contribute most to the community assemblage differences between anchoring intensities, the similarity percentages (SIMPER) analysis performs pairwise comparisons between each factor level (low, mooring and high), testing similarity in taxon type and abundance (Clarke and Warwick 2001). In order to identify the external factors that may account for patterns in community composition between samples, the BIOENV analysis overlays similarity matrices of external variables (sample depth, sediment properties) on the community assemblage similarity matrix and identifies the variable(s) most strongly correlated with differences in community composition (Clarke and Ainsworth 1993). The RELATE analysis is similar, but is a pairwise test of correlation between any two similarity matrices.

5.3 - Results

5.3.1 - Environmental factors

The depth of samples ranged from 0.5m to 4.3m below chart datum at Sidney Spit and 0.8m to 2.0m below chart datum at Tumbo Island (Fig 5.2). This was due to the fact that the bottom depth at Sidney Spit was much more variable than at Tumbo Island, and also because Sidney Spit was more wave/wind exposed, making it harder to hold the boat steady over an intended sampling site. However, since all samples lie within the shallow subtidal zone, this variation in sample depth is considered acceptable.

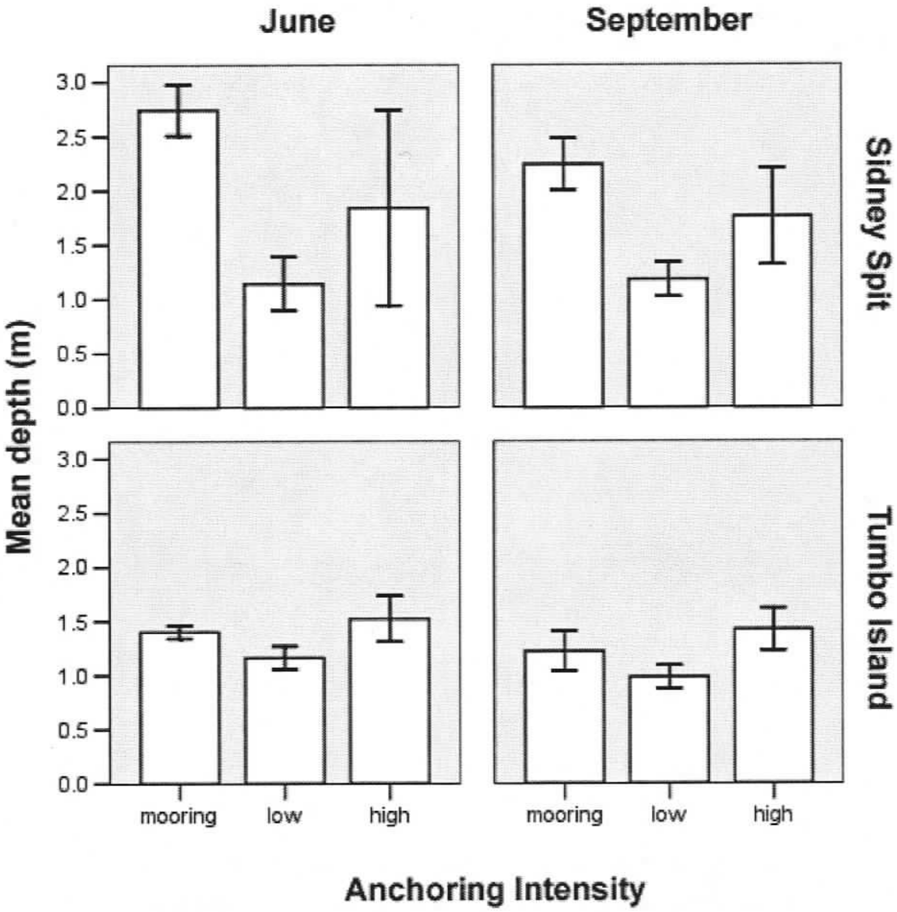

Figure 5.2 - Mean depth below chart datum of all sample stations sorted by site and season. Error bars represent two standard errors of mean.

Sand represented the dominant sediment size at both sites, comprising approximately 70% of the Sidney Spit samples and 90% of the Tumbo Island samples (Figure 5.3). The remainder of the Sidney Spit samples were distributed evenly between silt and clay particles, whereas no silt particles were observed in the Tumbo Island samples (clay particles made up the remainder of the samples). The correlation between depth and particle size is significant at 1.7% (sample statistic = 0.115) when using the RELATE function to compare the depth similarity matrix to the sand/silt/clay similarity matrix.

Average organic content (percent by mass) was higher at Sidney Spit (3.8%) than Tumbo Island (2.3%) (Figure 5.4). Organic content is significantly negatively correlated with particle size (sand/silt/clay content) at 0.1% (sample statistic = 0.231) using the RELATE function (finer sediments have higher organic content), but is not significantly correlated with depth at 18.1% (sample statistic = 0.052). There were no statistically significant differences in organic content or sediment profiles between sampling stations within sites, or between months within sites (p>5%, Mann-Whitney U-test) except in the Tumbo Island high anchoring intensity area. In September, 4% of the sand content was replaced by silt (p=3%, Mann-Whitney U-test). This could be related to the fact that is the location of a high physical disturbance regime, indicating that the anchoring disturbance is causing increased settling of fine particles over pitted areas (Dernie *et al.* 2003). For the most part, though, the physical characteristics of the sediment are remaining consistent through the early summer (June) and late summer (September) sampling periods.

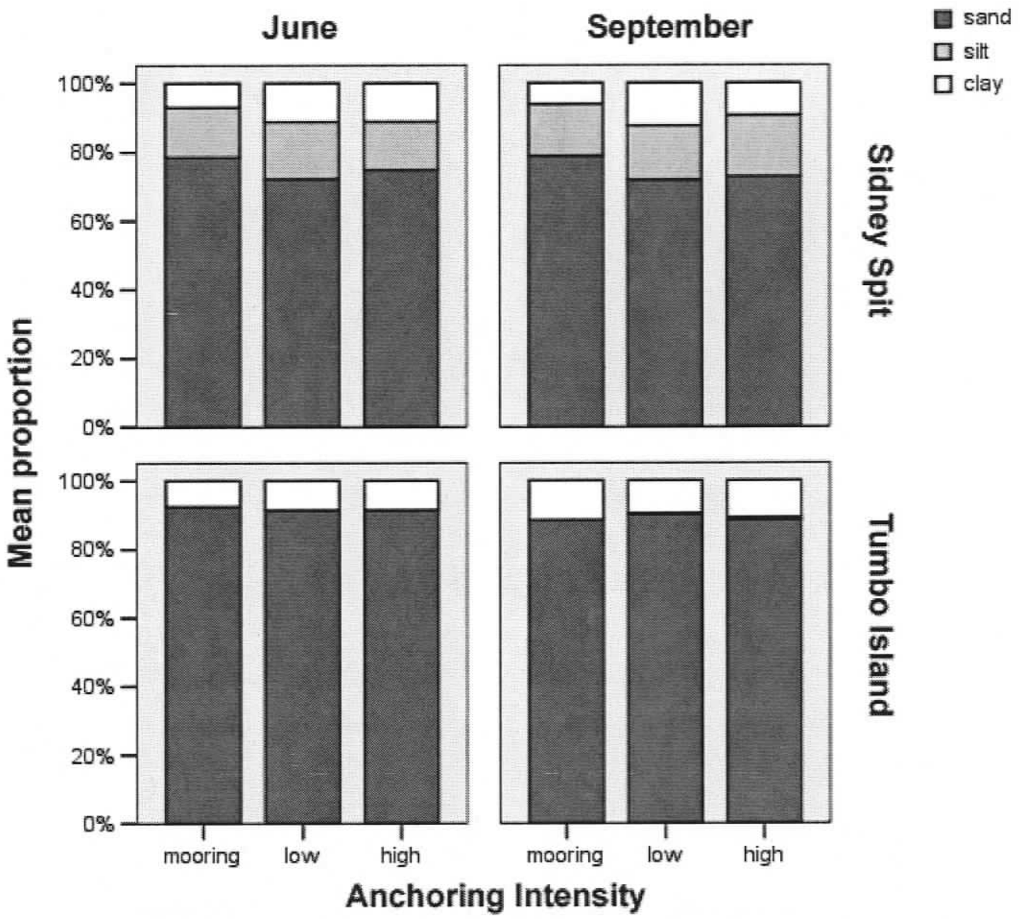

Figure 5.3 - Distribution of particle sizes at sample stations, sorted by site and season.

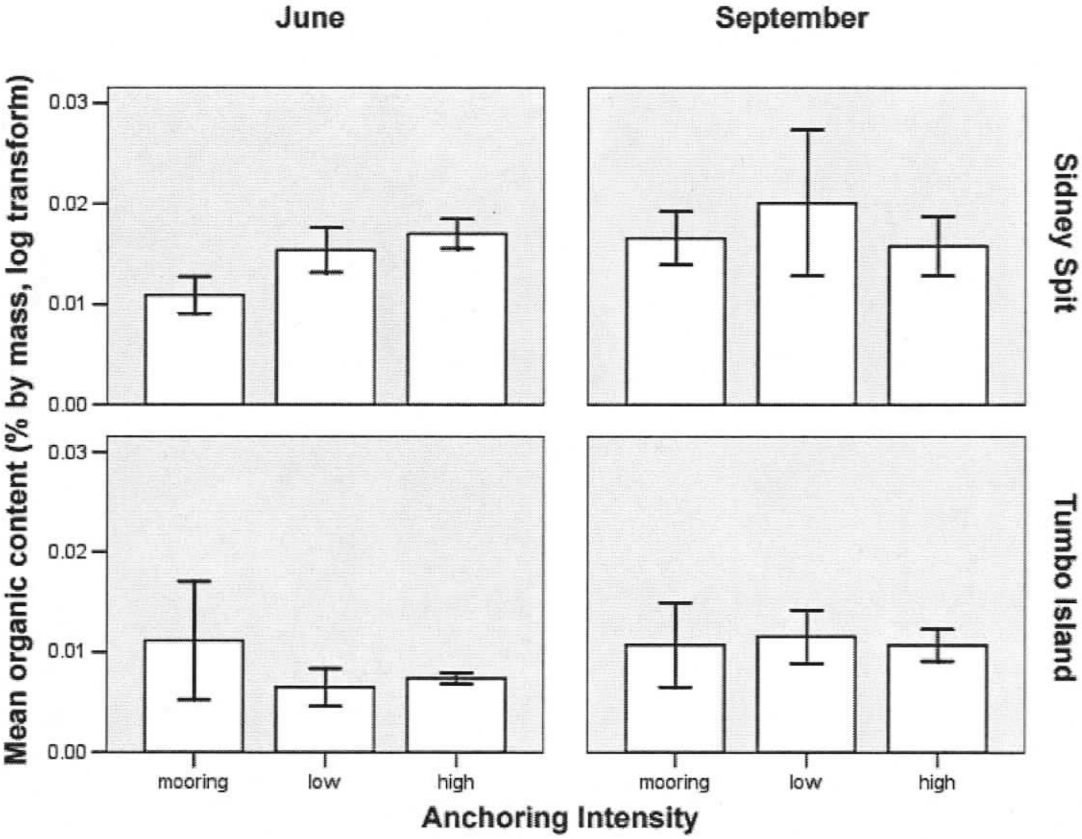

Figure 5.4 - Mean organic content (percent by mass, log transform) of all sample stations sorted by site and season. Error bars represent two standard errors of mean.

5.3.2 - Interactive effects

The global differences in community structure were significant between anchoring intensity groups at the 0.1% level, with significant differences between each pair of factors (low, high, and mooring) (Table 5.2). However, when site, season, and organism size class were tested for interactive effects with the nested ANSOSIM, all three factors were significant at the 10% level (Table 5.3). Although higher than the usual significance threshold of 5%, this still indicates that there is only a 1/10 chance that differences between sites, seasons, or size classes within intensity groups are due to chance. In this case, it is recommended to take the conservative approach and not discount the possibility of interactive effects of site, season, and size class (Warwick 1993). Therefore, data was not pooled or aggregated at higher levels to improve test power.

Table 5.2 - ANOSIM testing differences in community similarity between low, mooring and high anchoring intensity groups (all sites, seasons, and size classes). Overall differences are significant (Global R = 0.076, significance = 0.1%). * denotes significance at p<5%.

Groups	**R Statistic**	**Significance Level %**
Low, Mooring	0.095	0.2*
Low, High	0.052	1.3*
Mooring, High	0.085	0.5*

Table 5.3 - Nested ANOSIM testing interactive effects of organism size class, site, and season on differences observed in community structure between anchoring intensity groups. ** denotes significance at p<5%. * denotes significance at p<10%.

	Significance of differences between anchoring intensity groups (averaged across nested factor)		**Significance of differences between nested factor groups (using anchoring intensity groups as samples)**	
Nested factor	**Global R**	**Significance**	**Global R**	**Significance**
Organism size class	0.086	0.1%**	1	10%*
Site	0.926	0.1%**	1	10%*
Season	0.222	0.1%**	1	10%*

5.3.3 - Univariate measures of community health

Figures 5.5 through 5.8 indicate differences in number of organisms, number of families, evenness, and diversity between anchoring intensity, sorted by site, season, and size class. P-values of significant differences in these comparisons are indicated in tables 5.4 and 5.5 (non-parametric (rank order) analyses were selected because the data did not meet the criteria for parametric analyses). Generally, few statistically significant differences were observed; because these indices are sample size dependent (and the sample size of this study was small), perhaps these differences would become clearer with increased sampling. Also, mooring sites rarely differed between pre- and post-boating observations, indicating that the use of mooring area as a proxy for reference sites is not unreasonable.

At Sidney Spit, higher number of families and individuals were observed in anchoring areas than in mooring (reference) areas, which does not support the hypothesis that anchoring causes decreased organism and family abundance (Figs. 5.5 and 5.6). However, lower evenness and diversity were observed in the anchoring areas than the mooring areas (Figs. 5.7 and 5.8), though not at significant levels.

The results at Tumbo Island better support the alternative hypothesis, with reduced family abundance, individual abundance, and diversity at high anchoring intensity areas than at mooring areas (Figs. 5.5, 5.6 and 5.8). Low anchoring intensity areas, however, were observed to have higher family abundance, individual abundance, and diversity than mooring areas (Figs. 5.5, 5.6 and 5.8). Evenness did not vary between anchoring intensities. Also, no significant pre-/post- boating season differences were observed at this Tumbo Island.

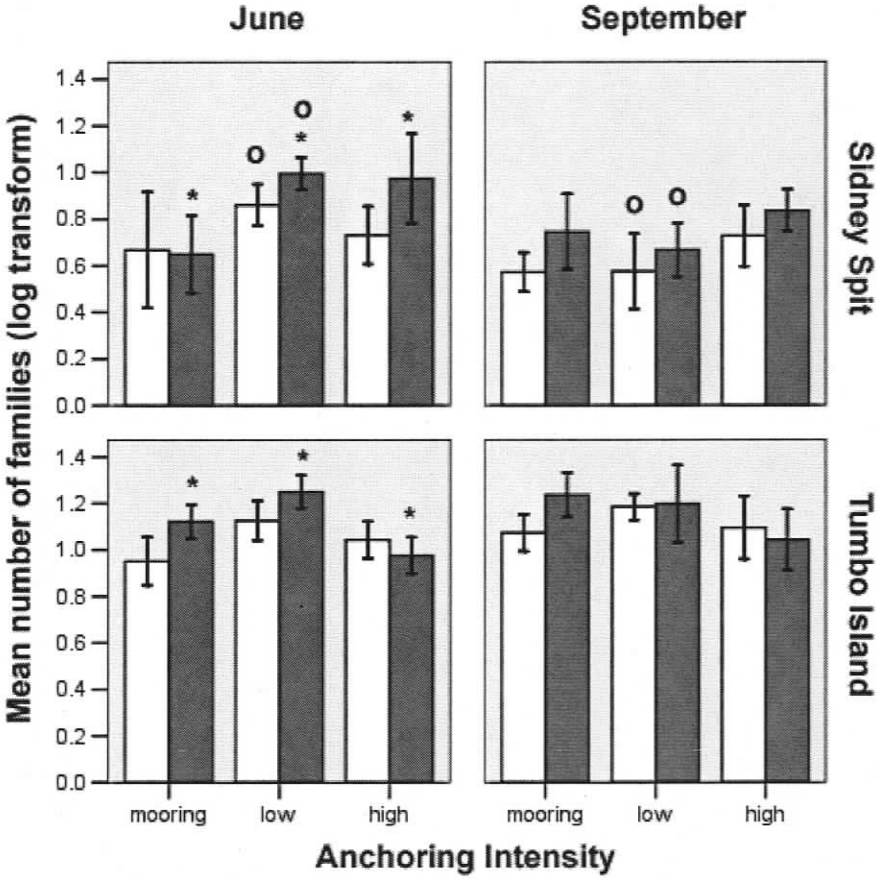

Figure 5.5 - Comparison of mean number of families by anchoring intensity, sorted by site, season, and size class. White bars represent 0.5mm size class and shaded bars represent 1.0mm size class. Error bars represent two standard errors of mean. * represents significant (p<5%) differences between anchoring intensities within a given site/size class/sampling period. ◦ represents significant (p<5%) differences between sampling periods within a site/size class/anchoring intensity. Statistics are summarized in tables 5.4 and 5.5.

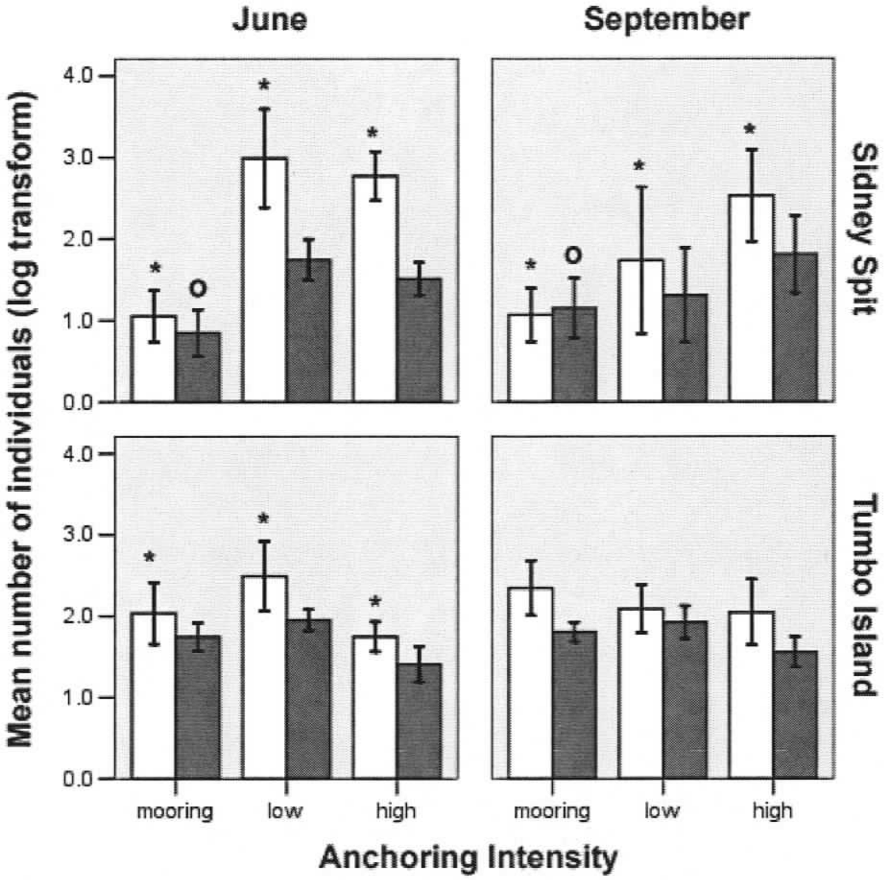

Figure 5.6 - Comparison of mean number of individuals by anchoring intensity, sorted by site, season, and size class. White bars represent 0.5mm size class and shaded bars represent 1.0mm size class. Error bars represent two standard errors of mean. * represents significant (p<5%) differences between anchoring intensities within a given site/size class/sampling period. ◦ represents significant (p<5%) differences between sampling periods within a site/size class/anchoring intensity. Statistics are summarized in tables 5.4 and 5.5.

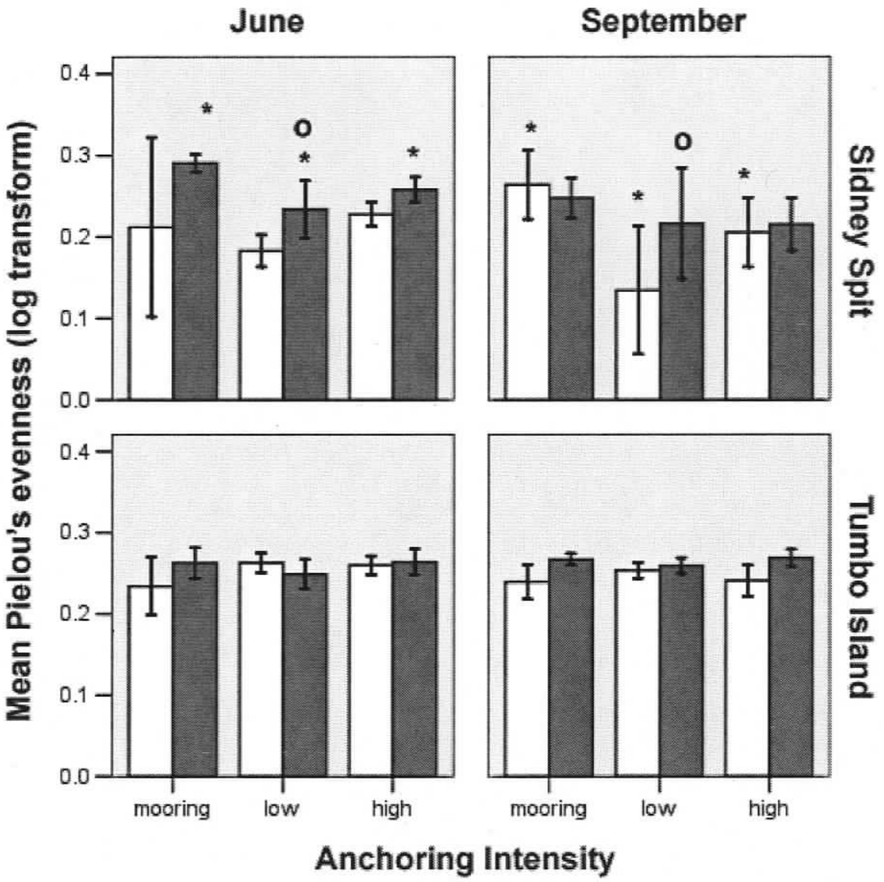

Figure 5.7 - Comparison of log transformed Pielou's evenness by anchoring intensity, sorted by site, season, and size class. White bars represent 0.5mm size class and shaded bars represent 1.0mm size class. Error bars represent two standard errors of mean. * represents significant (p<5%) differences between anchoring intensities within a given site/size class/sampling period. ∘ represents significant (p<5%) differences between sampling periods within a site/size class/anchoring intensity. Statistics are summarized in tables 5.4 and 5.5.

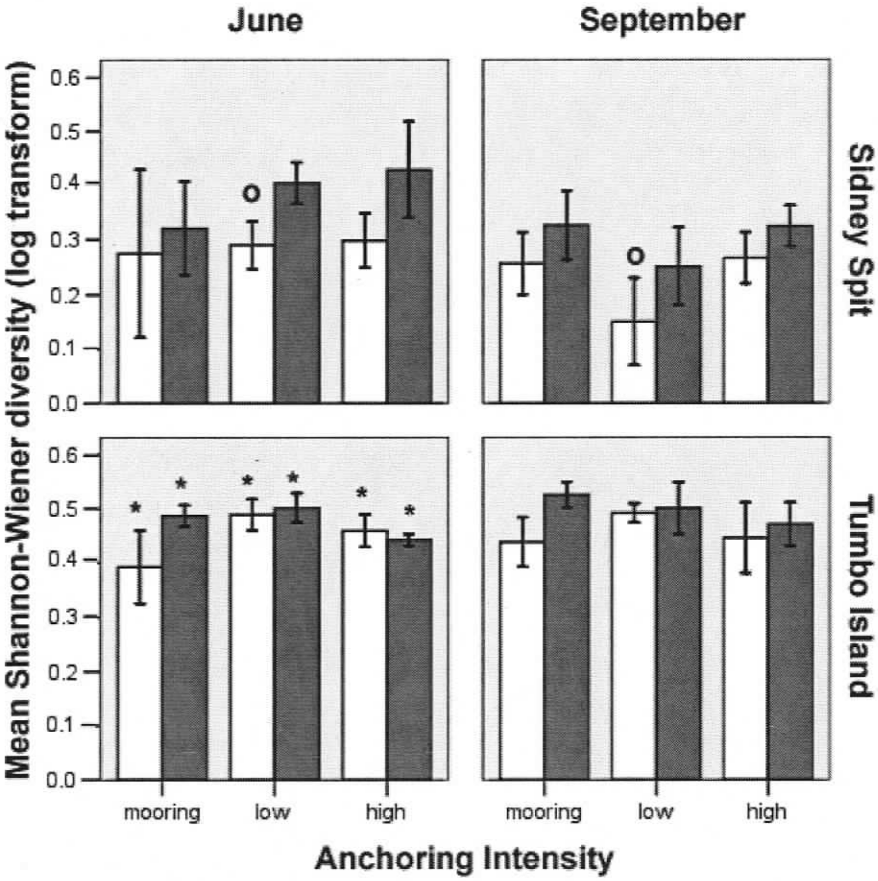

Figure 5.8 - Comparison of log transformed Shannon diversity by anchoring intensity, sorted by site, season, and size class. White bars represent 0.5mm size class and shaded bars represent 1.0mm size class. Error bars represent two standard errors of mean. * represents significant (p<5%) differences between anchoring intensities within a given site/size class/sampling period. ◦ represents significant (p<5%) differences between sampling periods within a site/size class/anchoring intensity. Statistics are summarized in tables 5.4 and 5.5.

Table 5.4 - Kruskal-Wallis test - significance of ranked differences in univariate community measures between anchoring intensities within a given site/size class/sampling period. * denotes significance at p<5%

Sample Group		**total number of families**	**total number individuals**	**Pielou's evenness**	**Shannon diversity**
SS	**June 0.5mm**	0.325	*0.008	0.065	0.878
	June 1.0mm	*0.02	*0.01	*0.017	0.065
	Sept 0.5mm	0.228	*0.029	*0.038	0.048
	Sept 1.0mm	0.17	0.074	0.364	0.253
TI	**June 0.5mm**	0.088	*0.046	0.264	*0.046
	June 1.0mm	*0.005	*0.011	0.395	*0.015
	Sept 0.5mm	0.15	0.3	0.595	0.289
	Sept 1.0mm	0.189	0.148	0.33	0.208

Table 5.5 - Mann-Whitney U-test - significance of ranked differences in univariate community measures between sampling periods within a site/size class/anchoring intensity * denotes significance at p<5%

Sample Group		**total number of families**	**total number individuals**	**Pielou's evenness**	**Shannon diversity**
SS	**mooring 0.5mm**	0.523	1.000	0.465	0.465
	mooring 1.0mm	0.456	0.173	*0.047	0.753
	low 0.5mm	*0.017	*0.028	0.347	*0.016
	low 1.0mm	*0.009	0.347	0.754	*0.016
	high 0.5mm	1.000	0.465	0.602	0.465
	high 1.0mm	0.117	0.117	0.076	0.076
TI	**mooring 0.5mm**	0.089	0.251	0.754	0.347
	mooring 1.0mm	0.074	0.465	0.754	0.076
	low 0.5mm	0.338	0.207	0.251	0.754
	low 1.0mm	0.673	0.754	0.465	0.917
	high 0.5mm	0.395	0.175	0.117	0.917
	high 1.0mm	0.334	0.251	0.602	0.251

5.3.4 - Multivariate analyses of community assemblages

Aggregating organisms into groups of organisms typical of opportunistic, equilibrium, and neutral communities (according to Appendix Two, based on Pearson and Rosenberg (1978)), samples from Sidney Spit and Tumbo Island formed two distinct clusters (Figure 5.9, 3D stress = 0.08) that were significantly different (ANOSIM, global R = 0.459, significance = 0.1%). Most polychates, nematodes and nemerteans were considered to be opportunists, whereas tube-dwelling polychaetes, bivalves and crustaceans were considered to be representative of equilibrium communities experiencing less frequent physical disturbance. All other organisms were considered to be neutral. The SIMPER routine showed that 89% of the similarity in the Sidney Spit samples was attributed to opportunistic species, while 97% of the similarity in the Tumbo Island samples was attributed to equilibrium and neutral species; the differences between these two sites can be attributed to these differences in dominant organism types (Table 5.6) and highlights that Sidney Spit communities are characterized by opportunists (indicative of disturbed habitat) and Tumbo Island communities are characterized by equilibrium organisms (indicative of more stable habitat). These dissimilarities highlight again that there are different factors influencing the communities at each site and therefore results must be treated separately.

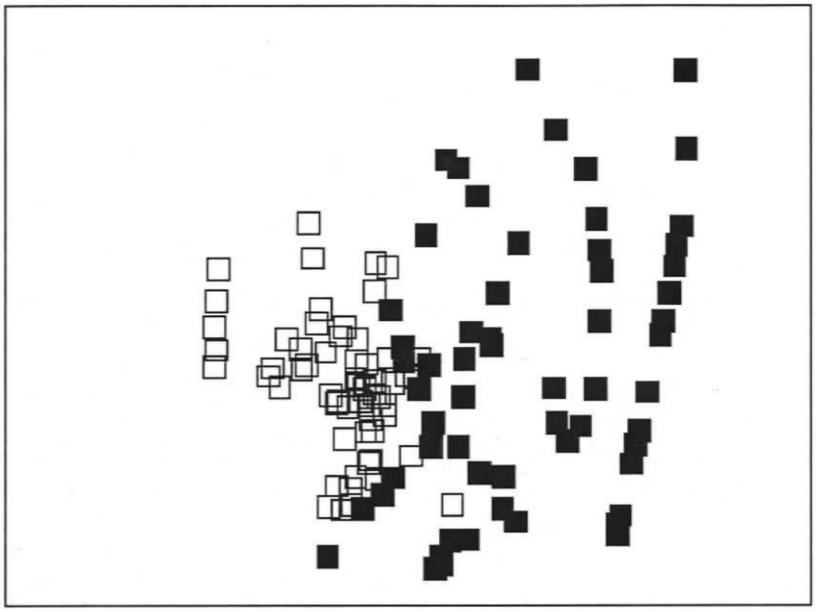

Figure 5.9 - MDS plot comparing similarity of sites based on organism grouping into opportunistic, equilibrium, and neutral groups (representative of disturbed and stable communities). Solid squares = Sidney Spit, open squares = Tumbo Island. 2D stress = 0.11, 3D stress = 0.08.

Table 5.6 - Result of SIMPER comparing similarity of organisms between sites, based on organism representative of disturbed and stable communities (overall dissimilarity between sites = 77.5)

Group	Mean abundance Sidney Spit	Mean abundance Tumbo Island	Mean dissimilarity	Percent contrib.
Opportunists	305.55	48.9	37.51	48.4
Neutral	42.88	65.4	24.23	31.26
Equilibrium	3.83	26.87	15.76	20.34

At both sites, significant differences in community assemblage (based on aggregation into groups of equilibrium, opportunist, and neutral organisms) were most often observed between pre-/post- boating observations at low and high anchoring intensities, with significant differences rarely observed between low and high intensity anchoring areas (ANOSIM, Table 5.7). These results again support the use of mooring areas as "reference sites" since the communities did not often seem to vary temporally through the course of this study.

Although there were significant differences observed in community structure, none of these differences could be specifically attributed to external environmental or temporal factors (BIOENV, table 5.8). No significant correlations were observed, and no single factor stood out from the others.

Table 5.7 - Results of ANOSIM testing significance of differences between community assemblages related to sampling periods, sorted by site, anchoring intensity and size class. Community similarity is based on aggregation of organisms into equilibrium, opportunist and neutral groups. SS = Sidney Spit, TI = Tumbo Island, L = low anchoring intensity, M = Mooring area, H = High anchoring intensity. * denotes significance of p<5%.

Sample group		**Mooring**		**Low**		**High**	
		R	**Sig.**	**R**	**Sig.**	**R**	**Sig.**
SS	**0.5mm**	0.034	31.0%	0.476	*1.6%	0.204	10.3%
	1.0mm	0.1	19.8%	0.376	*0.8%	0.348	*4.8%
TI	**0.5mm**	-0.056	59.5%	0.168	12.7%	0.056	23.0%
	1.0mm	0.416	*1.6%	0.532	*2.4%	0.6	*1.6%

Table 5.8 - Results of BIOENV analysis indicating external variables most strongly correlated with differences observed in community assemblages between sampling periods, sorted by site, anchoring intensity and size class. External variables tested were depth, sampling period, sediment size structure and organic content SS = Sidney Spit, TI = Tumbo Island

Sample group		**Best correlated factor(s)**					
		Mooring		**Low**		**High**	
		Factor(s)	**Corr.**	**Factor(s)**	**Corr.**	**Factor(s)**	**Corr.**
SS	**0.5mm**	-	-	all	-0.124	-	-
	1.0mm	-	-	all	0.122	all	0.364
TI	**0.5mm**	-	-	-	-	-	-
	1.0mm	all	0.04	all	0.19	all	0.349

At Sidney Spit, significant differences in community assemblage (based on aggregation into groups of equilibrium, opportunist, and neutral organisms) were most often observed between mooring/low intensity and low/high intensity groups, with no significant differences observed between low and high intensity anchoring areas. (ANOSIM, Table 5.9). In contrast, at Tumbo Island, significant differences in community assemblage were most often observed between mooring/high intensity and low/high intensity groups, with no significant differences between mooring/low intensity groups.

Table 5.10 shows the environmental variables that are most strongly correlated with the patterns in community assemblages observed. Although there are differences in community assemblage between anchoring intensities at Sidney Spit, the factor that is most often correlated with community assemblage at this site is depth of sample, not anchoring intensity. As shown in section 5.3.1, depth is strongly correlated with particle size distribution, so these differences in community assemblage are most likely a function of sediment size between sites rather than the anchoring intensity itself. This correlation is also only apparent at the 0.5mm size class of organisms: there are only weak correlations in the 1.0mm size class at Sidney Spit. In contrast, at Tumbo Island, there are strong correlations between anchoring intensity and community structure in both months and size classes.

Table 5.9 - Results of ANOSIM testing significance of differences between community assemblages related to anchoring intensity, sorted by site, season and size class. Community similarity is based on aggregation of organisms into equilibrium, opportunist and neutral groups. SS = Sidney Spit, TI = Tumbo Island, L = low anchoring intensity, M = Mooring area, H = High anchoring intensity. * denotes significance of < 5%.

Sample group		**Global test**		**Pairwise tests**					
				M/L		M/H		L/H	
		R	**Sig.**	**R**	**Sig.**	**R**	**Sig.**	**R**	**Sig.**
SS	**June 0.5mm**	0.54	*0.1%	0.824	*0.8%	0.808	*0.8%	0.108	21.4%
	June 1.0mm	0.37	*0.5%	0.776	*0.8%	0.432	*4.0%	-0.028	54.8%
	Sept 0.5mm	0.262	2.7%	0.24	5.6%	0.604	*1.6%	-0.026	51.6%
	Sept 1.0mm	0.013	36.5%	-0.082	59.5%	0.126	15.9%	0.034	23.8%
TI	**June 0.5mm**	0.266	*3.2%	0.08	23.8%	0.136	11.9%	0.584	*2.4%
	June 1.0mm	0.472	*0.1%	-0.016	46.8%	0.652	*0.8%	0.864	*0.8%
	Sept 0.5mm	0.065	23.5%	-0.008	47.6%	0.164	16.7%	0.06	33.3%
	Sept 1.0mm	0.256	*0.8%	0.056	24.6%	0.368	*4.8%	0.468	*2.4%

Table 5.10 - Results of BIOENV analysis indicating external variables most strongly correlated with differences observed in community assemblages between anchoring intensities, sorted by site, sampling period and and size class. External variables tested were depth, sampling period, sediment size structure and organic content SS = Sidney Spit, TI = Tumbo Island * Indicates correlation of greater than 0.4.

Sample group		**Best correlated factor(s)**	
		Factor(s)	**Corr.**
SS	**June 0.5mm**	depth	0.477*
	June 1.0mm	depth, clay content	0.404*
	Sept 0.5mm	-	-
	Sept 1.0mm	-	-
TI	**June 0.5mm**	anchoring intensity, organic content	0.539*
	June 1.0mm	anchoring intensity	0.614*
	Sept 0.5mm	-	-
	Sept 1.0mm	anchoring intensity, silt content	0.480*

MDS plots and cluster diagrams showed significant distinction between samples from anchoring areas and mooring areas in the 0.5mm size class at Sidney Spit (Figure 5.10). Although stress values for MDS plots of the 1.0mm size class were significant (<0.1), no clear pattern could be discerned from the data. At Tumbo Island, there were no obvious clustering patterns in the 0.5mm size class samples, but low intensity/mooring sites tended to cluster away from the high anchoring intensity sites in the 1.0mm size class in both June and September (Fig. 5.11).

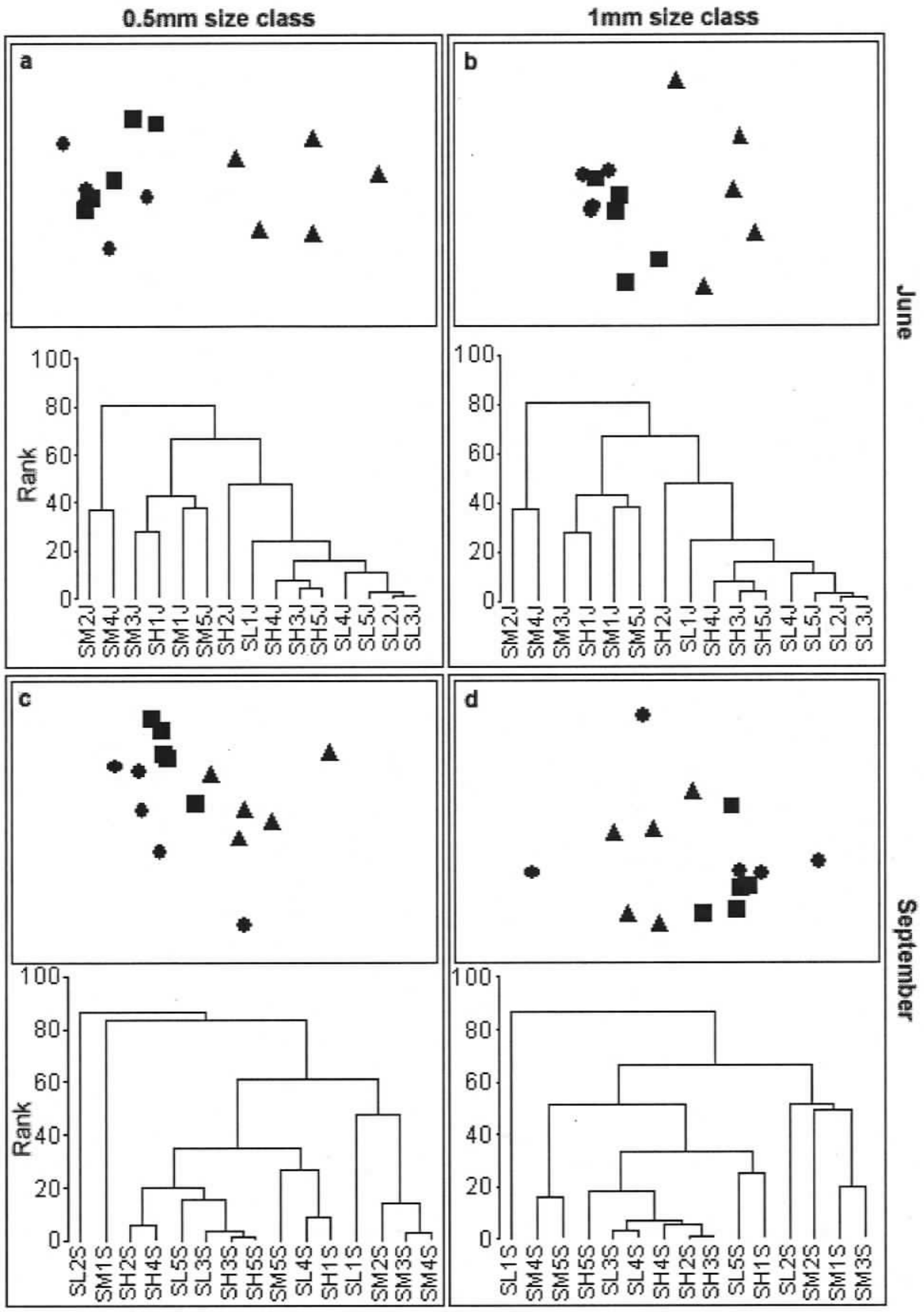

Figure 5.10 - MDS plots and cluster diagrams indicating similarities in community assemblages between anchoring intensity samples in each season and size class at Sidney Spit. ● = low anchoring intensity, ■ = high anchoring intensity, ▲ = mooring area. 2-D and 3-D stress values for MDS plots are, respectively, (a) 0.06 and 0.04, (b) 0.11 and 0.06, (c) 0.07 and 0.04, and (d) 0.12 and 0.08.

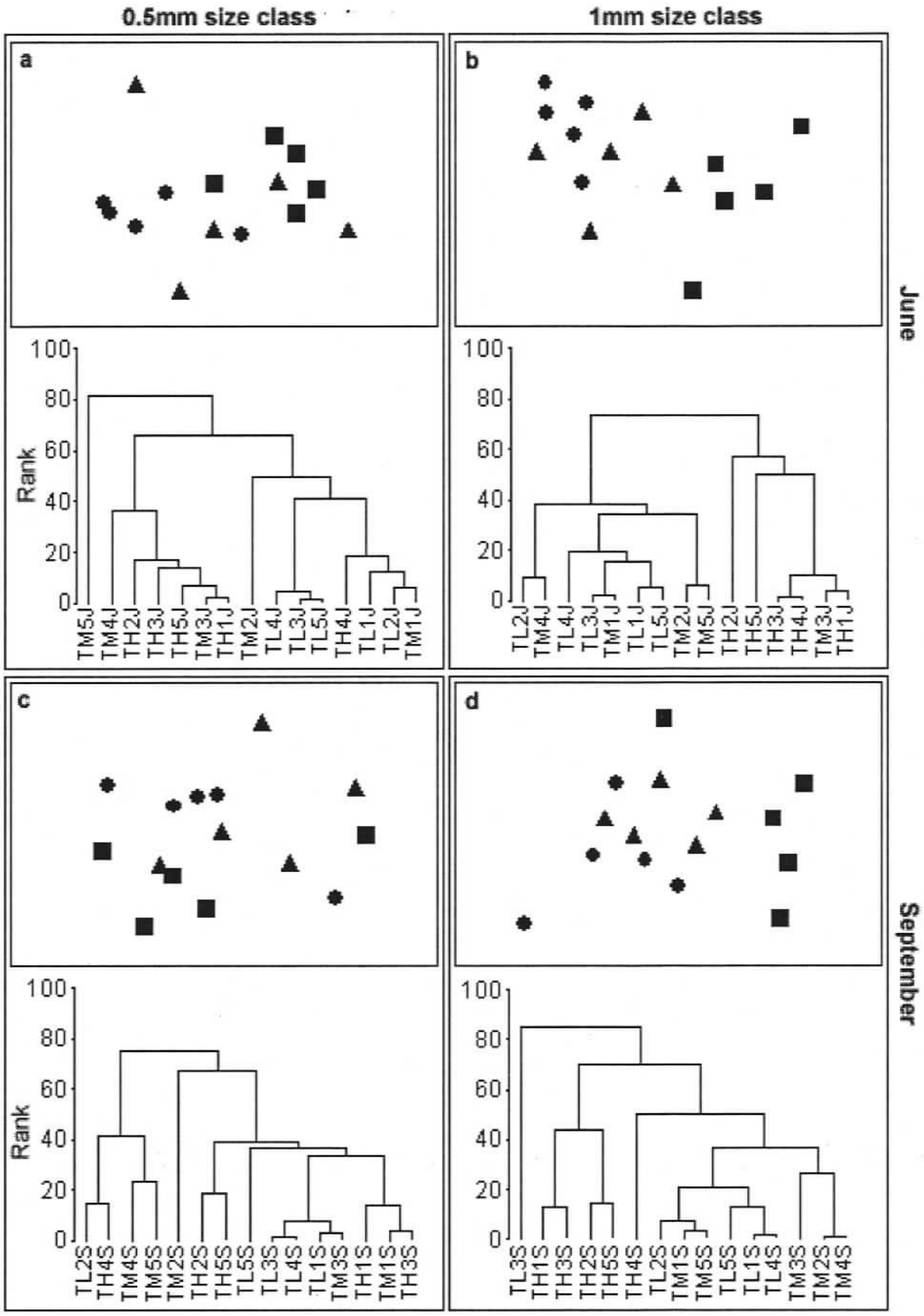

Figure 5.11 - MDS plots and cluster diagrams indicating similarities in community assemblages between anchoring intensity samples in each season and size class at Tumbo Island. ● = low anchoring intensity, ■ = high anchoring intensity, ▲ = mooring area. 2-D and 3-D stress values for MDS plots are, respectively, (a) 0.11 and 0.06, (b) 0.10 and 0.06, (c) 0.15 and 0.09, and (d) 0.13 and 0.08.

As already established from Table 5.6, the Sidney Spit communities are dominated by opportunistic organisms (nemerteans, nematodes, and polychaetes), both numerically (Figure 5.12) and proportionally (Figure 5.13). At Tumbo Island, there was a much broader representation of organisms from neutral and equilibrium communities (Figures 5.12 and 5.13). Especially in September, overall abundance of individuals decreased with anchoring intensity, but polychaete abundance remained constant, resulting in higher proportional abundance of this taxon in the high anchoring intensity samples (i.e., a higher proportional representation of opportunists than in mooring or low intensity samples).

Community composition ratios are explored further in Figures 5.14 and 5.15. Summaries of p-values for significance of differences between anchoring intensities and sampling periods are included in Tables 5.11 and 5.12. Again, non-parametric (rank order) tests were used since the data did not meet the criteria for parametric analyses. For both indices, high ratios are considered to be indicative of healthy invertebrate communities, and low ratios are considered to be indicative of invertebrate communities in poor health (UNESCO 2000). In general, no differences were observed in these indices between sampling periods within mooring areas, again supporting the uses of these sampling areas as references. At Sidney Spit, the ratios were quite variable, as reflected in the large standard errors, whereas at Tumbo Islands there was less variance in the ratios. In low and high anchoring intensity areas, both indices decreased significantly between June and September. Areas of high anchoring intensity also generally tended to have lower ratios than in low anchoring intensity areas.

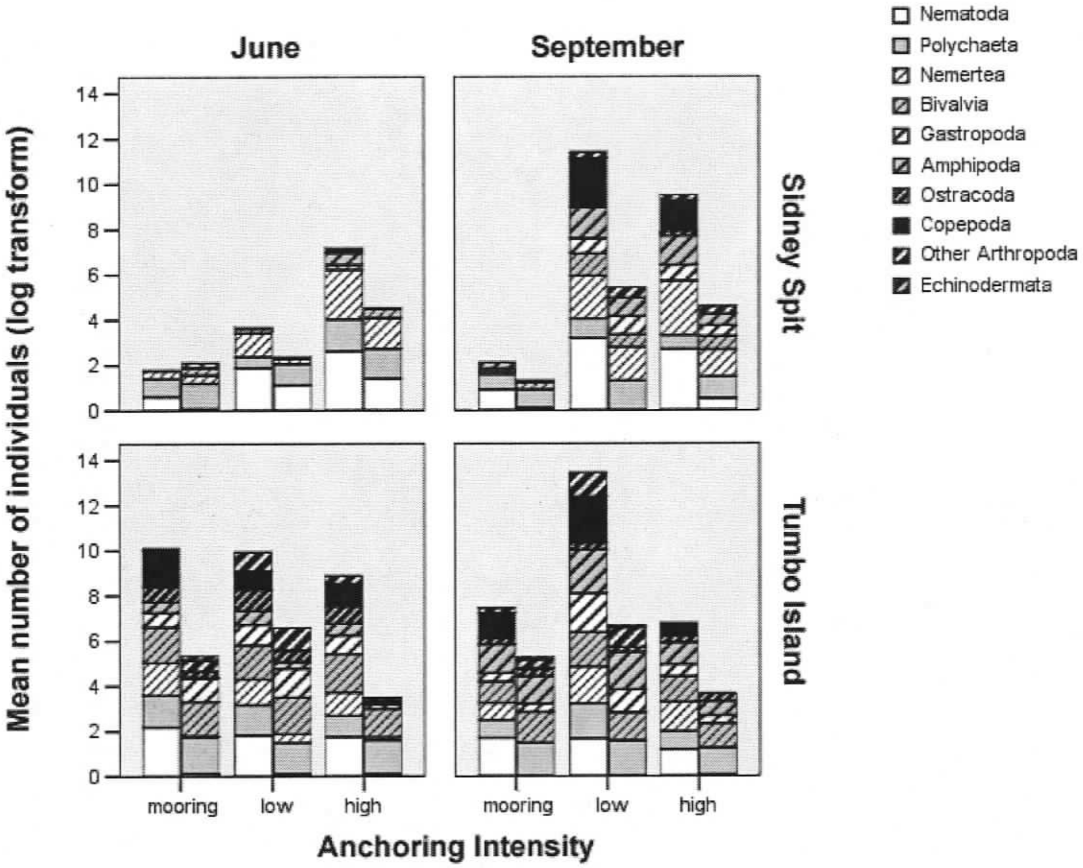

Figure 5.12 - Number of individuals of each taxon (log transform) observed by anchoring intensity, sorted by site, season, and size class. The left-hand bar at each intensity represents the 0.5mm size class, and the right-hand bar represents the 1.0mm size class. The groupings indicated in this figure are for the sake of clarity and do not reflect any groupings used for multivariate analysis.

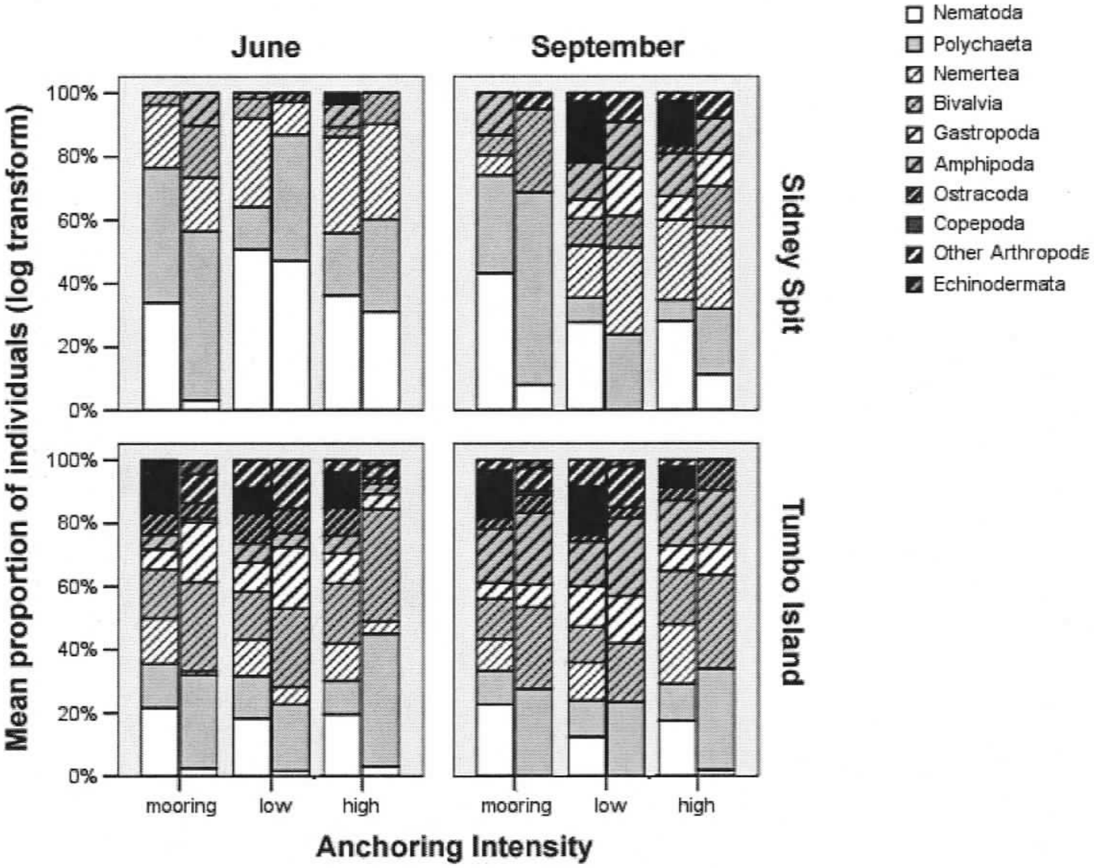

Figure 5.13 - Proportion of individuals of each taxon (log transform) observed by anchoring intensity, sorted by site, season, and size class. The left-hand bar at each intensity represents the 0.5mm size class, and the right-hand bar represents the 1.0mm size class. The groupings indicated in this figure are for the sake of clarity and do not reflect any groupings used for multivariate analysis.

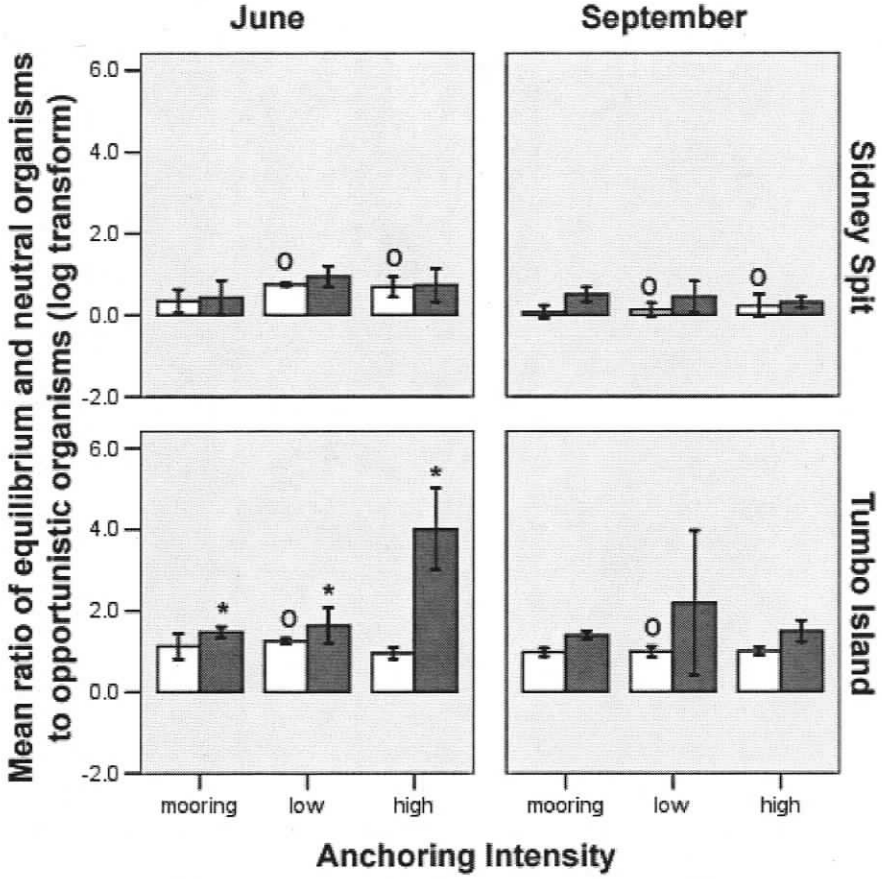

Figure 5.14 - Comparison of mean ratio of equilibrium and neutral organisms to opportunistic organisms by anchoring intensity, sorted by site, season, and size class. Higher ratios indicate healthier communities. White bars represent 0.5mm size class and shaded bars represent 1.0mm size class. Error bars represent two standard errors of mean. * represents significant ($p<5\%$) differences between anchoring intensities within a given site/size class/sampling period. ◦ represents significant ($p<5\%$) differences between sampling periods within a site/size class/anchoring intensity. Statistics are summarized in tables 5.4 and 5.5.

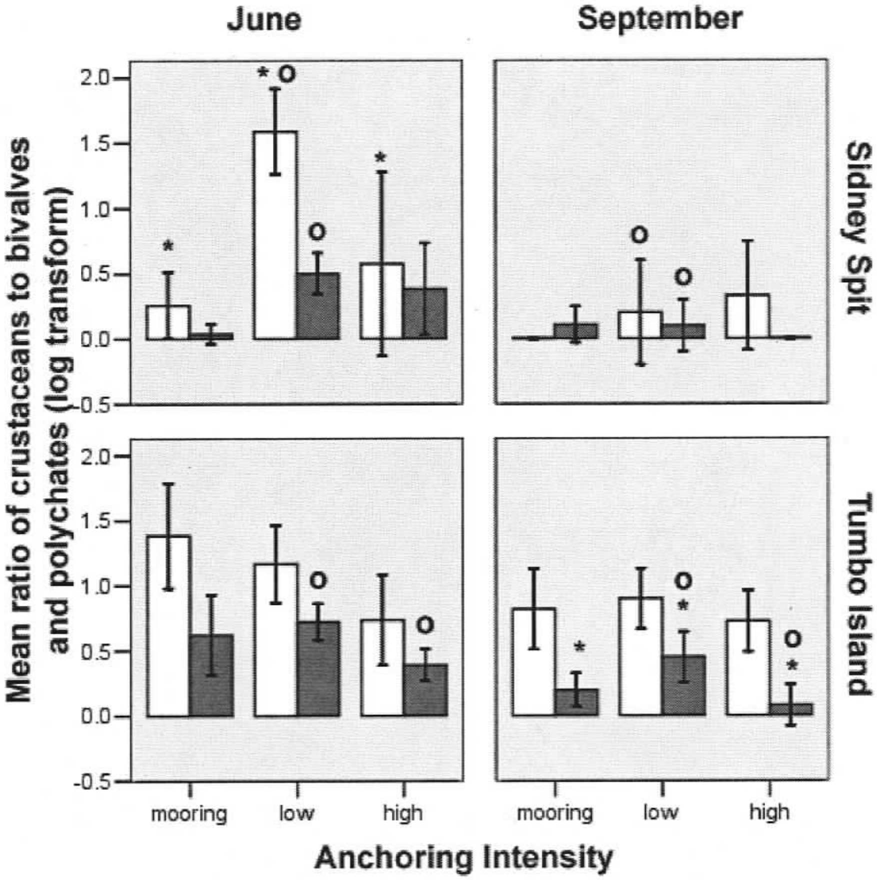

Figure 5.15 - Comparison of mean ratio of crustaceans to bivalves and polychaetes by anchoring intensity, sorted by site, season, and size class. Higher ratios indicate healthier communities. White bars represent 0.5mm size class and shaded bars represent 1.0mm size class. Error bars represent two standard errors of mean. * represents significant (p<5%) differences between anchoring intensities within a given site/size class/sampling period. ◦ represents significant (p<5%) differences between sampling periods within a site/size class/anchoring intensity. Statistics are summarized in tables 5.4 and 5.5.

Table 5.11 - Kruskal-Wallis test - significance of ranked differences in community composition measures between anchoring intensities within a given site/size class/sampling period. * denotes significance at p<5%

Sample Group		**ratio of crustaceans to polychaetes and molluscs**	**ratio of equilibrium and neutral organisms to opportunists**
SS	**June 0.5mm**	*0.03	0.878
	June 1.0mm	0.045	0.262
	Sept 0.5mm	0.339	0.896
	Sept 1.0mm	0.362	0.169
TI	**June 0.5mm**	0.039	0.075
	June 1.0mm	0.093	*0.026
	Sept 0.5mm	0.566	0.691
	Sept 1.0mm	*0.039	0.852

Table 5.12 - Mann-Whitney U-test - significance of ranked differences in community composition measures between sampling periods within a site/size class/anchoring intensity * denotes significance at p<5%

Sample Group		**ratio of crustaceans to polychaetes and molluscs**	**ratio of equilibrium and neutral organisms to opportunists**
SS	**mooring 0.5mm**	0.054	0.196
	mooring 1.0mm	0.368	0.917
	low 0.5mm	*0.007	*0.008
	low 1.0mm	*0.024	0.175
	high 0.5mm	0.638	*0.045
	high 1.0mm	0.054	0.117
TI	**mooring 0.5mm**	0.076	0.754
	mooring 1.0mm	0.076	0.465
	low 0.5mm	0.347	*0.009
	low 1.0mm	*0.047	0.347
	high 0.5mm	0.917	0.602
	high 1.0mm	*0.024	0.009

Finally, plots of average taxonomic distinctness (Figure 5.16) and variation in taxonomic distinctness (Figure 5.17) do not indicate that there are many outliers in this group of samples. The majority of the samples fall within the expected range of variation, and most of outliers are samples from high anchoring intensity stations, having lower average taxonomic distinctness and lower variation in taxonomic distinctness than normal (indications of environmental degradation (Warwick and Clarke 1997)). Although variability in taxonomic distinctness/variation in taxonomic distinctness is common, those samples that fall below the 95% confidence interval are thought be occupying habitats that have been degraded in some way (Warwick and Clarke 1997). A group of sites that stands out especially are the high anchoring intensity samples from the Tumbo Island 1.0mm size class in June. These sites were characterized by low abundance of organisms in this larger size class (expected to be more vulnerable to disturbance (Warwick and Clarke 1994)) and were dominated by molluscs and annelids (many annelids being deposit feeders indicative of disturbance (Cardoso *et al.* 1998)). They also had poor representation of arthropods in comparison to the sample pool, an important taxon for nutrition at higher trophic levels (Gaston *et al.* 1998). These results support the alternative hypothesis that physical disturbance from boat anchoring is taking place at some level, and that these impacts may be held over from the previous season.

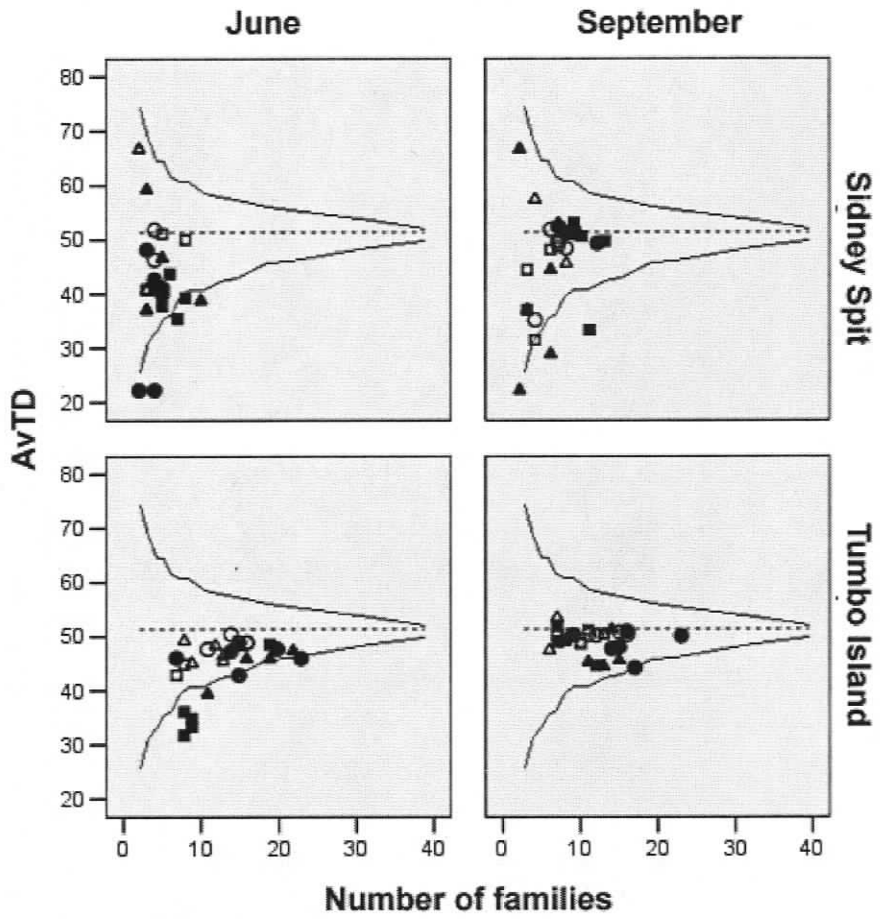

Figure 5.16 - Average taxonomic distinctness as a function of number of families observed. Dotted line indicates mean expected taxonomic distinctness for the family pool sampled in this study and solid lines indicate 95% confidence interval. Open symbols indicate 0.5mm size class, solid symbols indicate 1.0mm size class. ● = low anchoring intensity, ■ = high anchoring intensity, ▲ = mooring area.

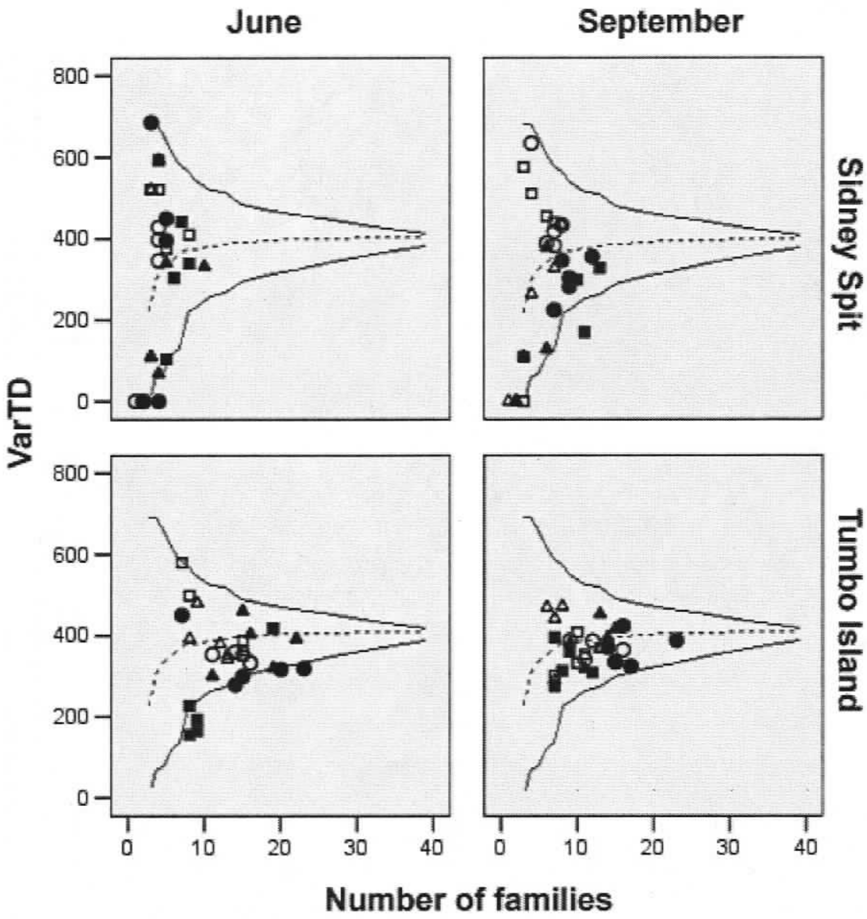

Figure 5.17 - Average taxonomic distinctness as a function of number of families observed. Dotted line indicates mean expected variation in taxonomic distinctness for the family pool sampled in this study and solid lines indicate 95% confidence interval. Open symbols indicate 0.5mm size class, solid symbols indicate 1.0mm size class. ● = low anchoring intensity, ■ = high anchoring intensity, ▲ = mooring area.

5.4 - Discussion

5.4.1 - Sidney Spit

The invertebrate communities at Sidney Spit were characterized by large proportion representation of opportunistic organisms, and lower overall abundance and diversity than Tumbo Island. Few clear patterns in univariate measures or community assemblages were observed with respect to anchoring intensity or pre-post boating observations – instead, measures did not show consistent patterns and had high variances (an indication of stressed communities) (Warwick and Clarke 1993) and community differences were more strongly influenced by environmental variables than anchoring intensity.

These results indicate that although the invertebrate communities at Sidney Spit may be in poor health, recreational boat anchoring is not the only factor influencing these communities. Also, since the invertebrate sampling areas had to be selected before the anchoring intensity data were collected, the fact that the lowest and highest intensity anchoring areas were not targeted may have affected the ability to draw conclusions in this study.

5.4.2 - Tumbo Island

The results from Tumbo Island better supported the alternative hypothesis that communities experiencing higher anchoring disturbance are in poorer health than less disturbed communities. Tumbo Island is likely a better site to test this hypothesis since it is more isolated and less subject to influence from nearby human population. Also, although there was less overall use of the site, the boats were bigger and would therefore cause a larger disturbance than the smaller boats most often observed at Sidney Spit.

Support for the alternative hypothesis at Tumbo Island came from several components of the study:

1) Abundance of individuals and families as well as diversity were negatively correlated with high anchoring intensity and post-boating observations.
2) High anchoring intensity communities were observed to be significantly different from low anchoring intensity and mooring area communities. The differences in community assemblage were more strongly correlated with anchoring intensity differences than with environmental factors.
3) Post-boating season communities were observed to be significantly different than pre-boating season communities in both anchoring intensity areas but rarely in mooring areas. However, these community assemblage differences were not correlated with any environmental or time-difference factors.
4) High anchoring intensity communities clustered separately from low intensity/mooring communities in the MDS plots.
5) High anchoring intensity and post-boating samples had highest proportional abundance of opportunists while low intensity/mooring communities had the highest proportional abundance of crustaceans, the preferred prey species for fish (Gaston *et al.* 1998)

6) Some high anchoring intensity samples were observed to have lower than average taxonomic distinctness and variation in taxonomic distinctness compared to the whole sample pool, which was attributed to the lack of arthropods in these sampling areas.

Although each piece of evidence by itself is not conclusive, the fact that all aspects of the analysis support the same alternate hypothesis is good evidence that the null hypothesis can be rejected. The consistency of the results among analyses indicates a certain degree of robustness to the conclusion (Dauer 1993). It is also important to remember that these indicators of poor community health translate to direct effects in the marine ecosystem in terms of lost ecosystem services and decreased food availability to higher trophic levels (Snelgrove 1998).

It is interesting to note that at Tumbo Island, low anchoring intensity communities were more similar to the mooring (reference) communities than to the high anchoring intensity communities, and in some cases had the highest indices of community health. Although Thrush and Dayton (2002) suggest that the intermediate disturbance hypothesis does not apply to soft bottom communities, these results show that small disturbance (low anchoring intensity) does not have a significant negative impact on community health. This would concur with Connell's 1978 hypothesis that intermediate disturbance maintains diversity by allowing organisms from early and late successional stages to co-exist (Connell 1978). In the case of Tumbo Island, it is possible that the low anchoring intensity area sampled in this study falls within the intermediate range of disturbance for this community and is therefore still quite healthy, whereas the high anchoring intensity sampling area falls in the unacceptably high level of disturbance resulting in poor community health at this location.

Eelgrass provides structural complexity, shelter from predators, dampening of wave energy and fine sediment particles, supporting a higher abundance and diversity of benthic invertebrates compared to adjacent sandy areas (Bostrom and Bonsdorff 1997, Jackson *et al.* 2001a, Duarte 2002, Heck *et al.* 2003). Although no physical fragmentation was quantified in the course of our study in the GINPR, the invertebrate communities in high anchoring intensity areas exhibited some similar characteristics to communities located in disturbed and fragmented eelgrass beds – namely an abundance of opportunist polychaetes and fewer crustaceans. The results observed in this study concur with the work done by Bowden *et al.* (2001) and Frost *et al.* (1999), who found that fragmented eelgrass beds and eelgrass bed edges had a higher abundance of opportunistic organisms than unfragmented beds and bed interiors. Cardoso *et al.* (2004) also found a dominance of deposit feeding polychates in the most disturbed areas of eelgrass beds. This translates directly to an impact at higher trophic levels, because fish prefer mobile epifauna (such as small crustanceans) to subsurface detritivores (such as polychaetes) (Gaston *et al.* 1998). Therefore, the absence of documented eelgrass loss in this study should not be taken at face value; there may still be community level impacts of anchoring activity at these locations, especially if the physical disturbance is happening at the <10m scale not detected in this study.

It is also interesting to note that physical processes seem to hold equal importance with biological interactions in structuring soft-bottom benthic communities in eelgrass beds. Wilsion (1991) suggests that resource competition is not a factor in structuring these communities, while Orth *et al.* (1984) stress the importance of eelgrass shoots and rhizome mat as a refuge from predation. Physical structural complexity, sediment size, and nutrient availability are most often stressed in the literature as driving mechanisms behind the structuring of benthic communities in eelgrass beds (Snelgrove and Butman 1994, Heck *et al.* 2003). However, Orth *et al.* (1984) do point out that unvegetated patches within eelgrass beds are most vulnerable to predation. This would be consistent with physical disturbance and eelgrass uprooting being associated with decreased overall organism abundance/biomass, with invertebrates in newly created patches being more susceptible to predation.

5.4.4 - Study design improvements

One weakness of this study was that it only looked at temporal variation of communities within a single season, and could not test if the same variations were observed year after year. There is no way to know if this snapshot of communities in the Gulf Islands is representative of long term trends. This limitation highlights the importance of establishing long term monitoring programs with fixed sampling programs in order to test long term human impacts and effectiveness of management strategies, a key component of adaptive management in marine protected areas (Salm *et al.* 2000).

Also, conclusions regarding the possible negative impacts of anchoring are most powerful if the same trend is observed at several sites (Clarke 1993). Unfortunately, since the alternative hypothesis was only supported at one of the two sites investigated in this study, conclusive proof of damage from anchoring cannot be established. Expanding the sampling program to include other anchorages such as Winter Cove would allow stronger conclusions to be drawn.

This issue is related to the statistical power of this study, i.e., the ability of the test to not make a type II error (β), where the null hypothesis is incorrectly accepted (Quinn and Keough 2002). Power is defined as 1- β, so the higher the power of the test, the higher the probability that a significant treatment effect was not incorrectly rejected. Power is a function of sample size, variance, effect size (how big is the change) and significance level. Now that the variance in this dataset is known, a required sample size for a given variance, power, significance and effect size could be calculated. Perhaps the small sample size of this study impeded the ability of this study to draw firm conclusions as to the effects of physical disturbance in eelgrass beds in the GINPR and a larger sample size would allow a more confident acceptance or rejection of the null hypothesis.

Finally, other environmental factors that should be included in future studies are nutrient and dissolved oxygen concentrations, which are known to be stressful to benthic invertebrates (Boesch and Rosenberg 1981, Cardoso *et al.* 2004). Although sediment profile and organic content are two important environmental factors influencing benthic community assemblages, discounting additional (possibly confounding) environmental factors would improve the robustness of conclusions related to community changes caused by anchoring disturbance.

5.5 - Summary

This component of the study provides evidence supporting the alternative hypothesis that high anchoring intensity is correlated with poor community health at only one of the study sites. However, this support is quite robust, with all aspects of the analysis supporting the same conclusion. The inconclusive results at the second site are likely related to the high human activity in the area, confounding the ability of this study design to discern the impacts of recreational boat anchoring from other human disturbances. Including more study sites, better targeting sampling locations (to areas of highest and lowest anchoring activity) and increasing the sample size would form a stronger basis for drawing conclusions in further research.

CHAPTER SIX: SUMMARY

6.1 - Conclusions

6.1.1 - Anchoring and eelgrass disturbance

Although the aim of this study was to correlate anchoring activity with changes in eelgrass bed configuration in the GINPR, no significant seasonal differences in eelgrass were observed at the 10m scale in the course of this study. If anchoring is causing patchiness at a smaller scale in these eelgrass beds, it could not be quantified due to an insufficient density of data points and inconsistent image quality that limited the interpretation of the video footage - the results of this study should not be considered as a strong support of the null hypothesis, but rather as insufficient evidence to reject the null hypothesis. However, the results do indicate that anchoring is most certainly occurring in eelgrass beds at both study sites, and that the highest anchoring densities observed at both sites also coincide with eelgrass beds.

6.1.2 - Anchoring and benthic community impact

The results of this study support the hypothesis that benthic communities in high anchoring intensity areas are in poorer health than those in low anchoring intensity or mooring areas at Tumbo Island but not at Sidney Spit. This likely stems from the differences in use at each site – although higher overall anchoring usage was observed at

Sidney Spit, larger boats were observed at Tumbo Island. Therefore, the greater physical impact of larger anchors and stronger forces exerted by heavier boats on the benthos could account for the stronger correlation between anchoring intensity and community health at Tumbo Island.

Overall, Tumbo Island communities were generally healthier than Sidney Spit at all anchoring intensities, and Sidney Spit communities were most strongly correlated with environmental factors rather than anchoring intensity. The results of the Sidney Spit site are likely confounded by the high overall human usage in this area, in contrast to Tumbo Island where the main human activity is boat anchoring.

6.1.3 - Anchoring management opportunities

Although the results of this study were not conclusive, there is a growing body of literature supporting the negative impacts of recreational boat anchoring on seagrass meadows. If a precautionary approach were to be taken in the GINPR, there are several management opportunities for the park to decrease the amount of anchoring in eelgrass beds. They include the following:

1) alleviate concerns regarding mooring buoy use by addressing mooring buoy safety and user fee issues;
2) encourage boaters not to set anchor if there are still available mooring buoys;

3) educate boaters on the ecological and safety hazards of anchorage inside eelgrass beds through shore signage;
4) inform boaters that if anchoring inside an eelgrass bed is necessary, traditional "Navy"/"Yachtsman" type anchors are least damaging to seagrasses;
5) clearly identify the perimeter of eelgrass beds at each site with buoys and encourage boaters to anchor outside these margins, through signage on shore and on the buoys; and
6) target anchorage sites frequented by larger boats since they cause greater disturbance.

6.2 - Long term monitoring and further directions

Part of the intention of this study was also to establish a protocol for long-term monitoring of anchoring activity impacts in the GINPR. These final remarks will address the feasibility of implementing different monitoring approaches employed in this study.

6.2.1 - Underwater video and mapping

Interpretation of underwater video for mapping was limited by the equipment that was available for this study. However, with some equipment upgrades and additional transects as described in chapter 2, higher quality video footage would yield a wealth of information. Although fragmentation or patchiness were not observed at the 10m scale, reducing the distance between observation points would allow the detection of eelgrass bed disruption at a finer scale. Mapping could also be targeted to known areas of high anchoring intensity now that the general eelgrass and anchoring distributions are known, in order to look at fine-scale effects of anchoring. Adding a frame of reference would also allow quantification of eelgrass density and epiphyte load; alternatively, a vertically dropped camera mounted on a quadrat could be used for these fine-scale measurements, providing a consistent perspective on the eelgrass bed for comparison between samples.

Using a large boat was necessary in order to facilitate site access and supply AC power for the video monitor. However, this vessel consumed a lot of fuel, and the slow speed required to maintain good image capture quality was taxing on the engines. Using a self contained image capture system running off a 12 volt battery would allow data to be captured from a smaller vessel, which would be easier to maneuver for fine mapping, allow better speed control at low velocity, and be less costly to fuel.

Although the process of obtaining and reviewing footage can be time consuming, it is important to note that it does not take any specialized academic training to record or interpret underwater footage – anyone given a clear set of instructions can code the video. After footage has been captured, digital video can be reviewed and coded on any computer, and a non-specialist can easily follow instructions to document the presence/absence of eelgrass, shoot density, and epiphyte load in each frame. Training in biology would only be required if invertebrate epifauna were to be documented.

6.2.2 - Site usage

The methods used to collect and interpret data on boat anchoring density were straightforward and satisfactory. One possible improvement would be to eliminate inconsistencies in recording dates between facility operators by providing clearer instructions to the data recorders. Also, printing datasheets directly from the GIS database used for interpretation would eliminate the minimal error in digitizing introduced by warping the scanned datasheets

Expanding this portion of the study to include interviews with boat operators would also improve our understanding of motives behind selection of anchoring location and choosing to moor vs. anchor. This study focused on the biological consequences of anchoring; increasing the scope of the study to include this social science aspect would be useful to refine site management and deter anchoring in eelgrass beds.

6.2.3 - Benthic invertebrate community health

From sample collection, processing and identification to statistical analysis and interpretation, this component of the study was most time consuming and the most difficult to interpret. Although the hypothesis was supported in one size class of organisms at one site, this is not the basis for firm conclusions. In order to improve the power of the interpretation, it would be necessary to increase the sample size and environmental variables in the analysis and show a consistent negative impact of anchoring at a number of sites. Given the level of specialization required for sample analysis and statistical interpretation as well as the time consuming nature of the analysis, it would be costly to expand this component of the study on a long-term basis.

6.2.4 - Recommendations for long term monitoring

Given limited field time, personnel, and financial resources, it would be wise to focus efforts in order to maximize the results of future research. After considering the strengths and weaknesses of each component of the study, I believe it would be most advantageous to focus on the underwater video and anchoring activity components of the study.

To some degree, the data quality of the underwater video and invertebrate sampling suffered for trying to fit everything into a few limited field days. Focusing field resources on obtaining a large amount of high quality video imagery would be the most likely approach to provide conclusive evidence documenting eelgrass bed fragmentation. The video can still be very data-rich and has the advantage of not requiring specialized equipment or training for interpretation. As long as a clear set of instructions were provided, interns or co-op students could easily take on projects analyzing this type of data. In contrast, improving the invertebrate community component of the study to the point where evidence would be reasonably conclusive would likely be the scope of a PhD dissertation.

Continuing to document use of mooring buoys and preferred anchoring areas as well as speaking with boaters about how they are using anchoring sites will allow the park to better anticipate boater demands at anchoring areas and plan accordingly to minimize eelgrass damage. This data is not costly to collect or process, and again does not require specialized training.

In conclusion, this study has provided some evidence of eelgrass community damage as a result of recreational boat anchoring in the GINPR. Lessons learned from this study were useful in determining the feasibility of different types of analyses, as well as planning strategies to maximize the returns on future sampling efforts. Finally, this study also proposes several management strategies specific to anchoring sites in the GINPR to encourage use of mooring buoys and decrease anchoring in eelgrass beds.

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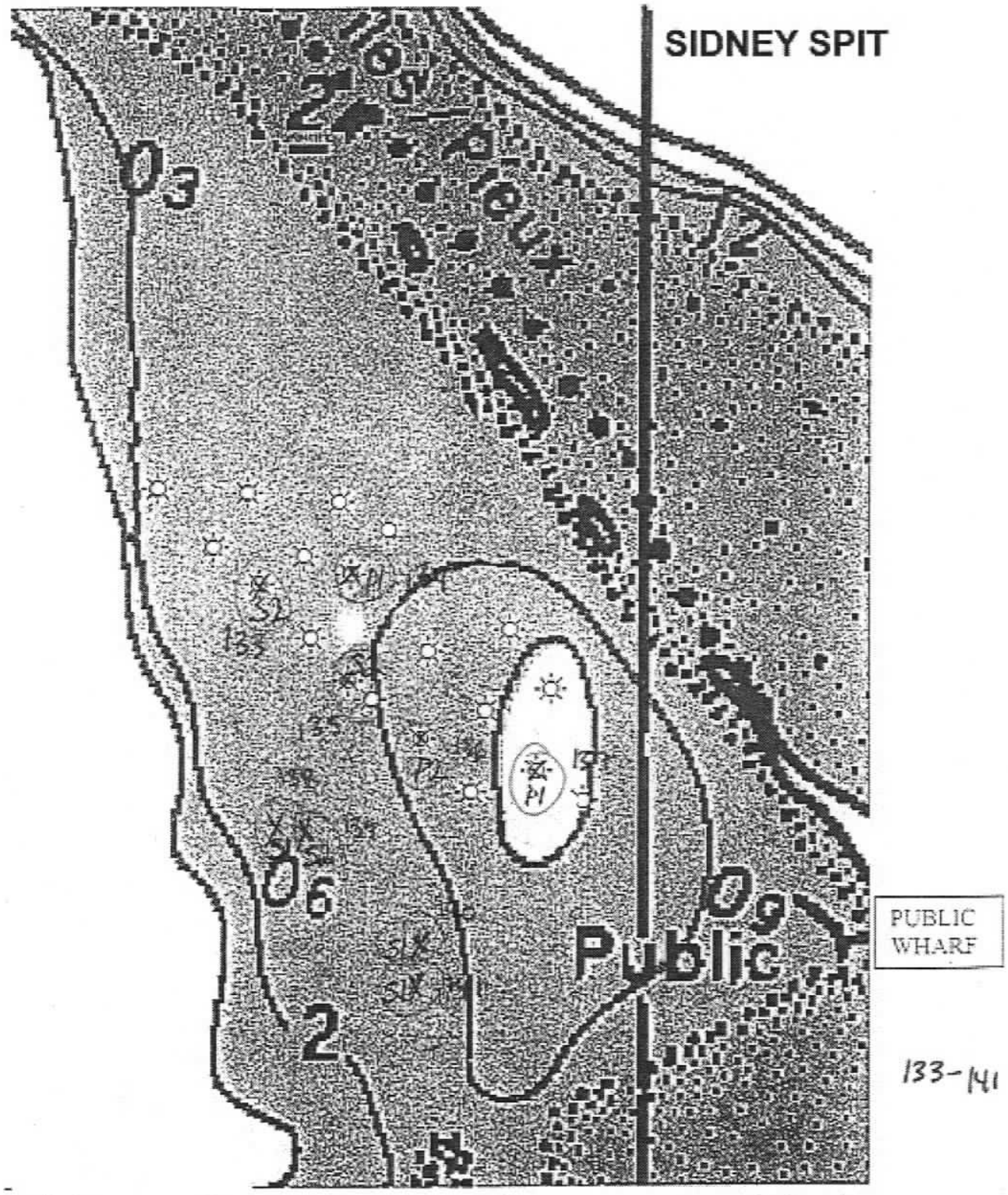
APPENDICES

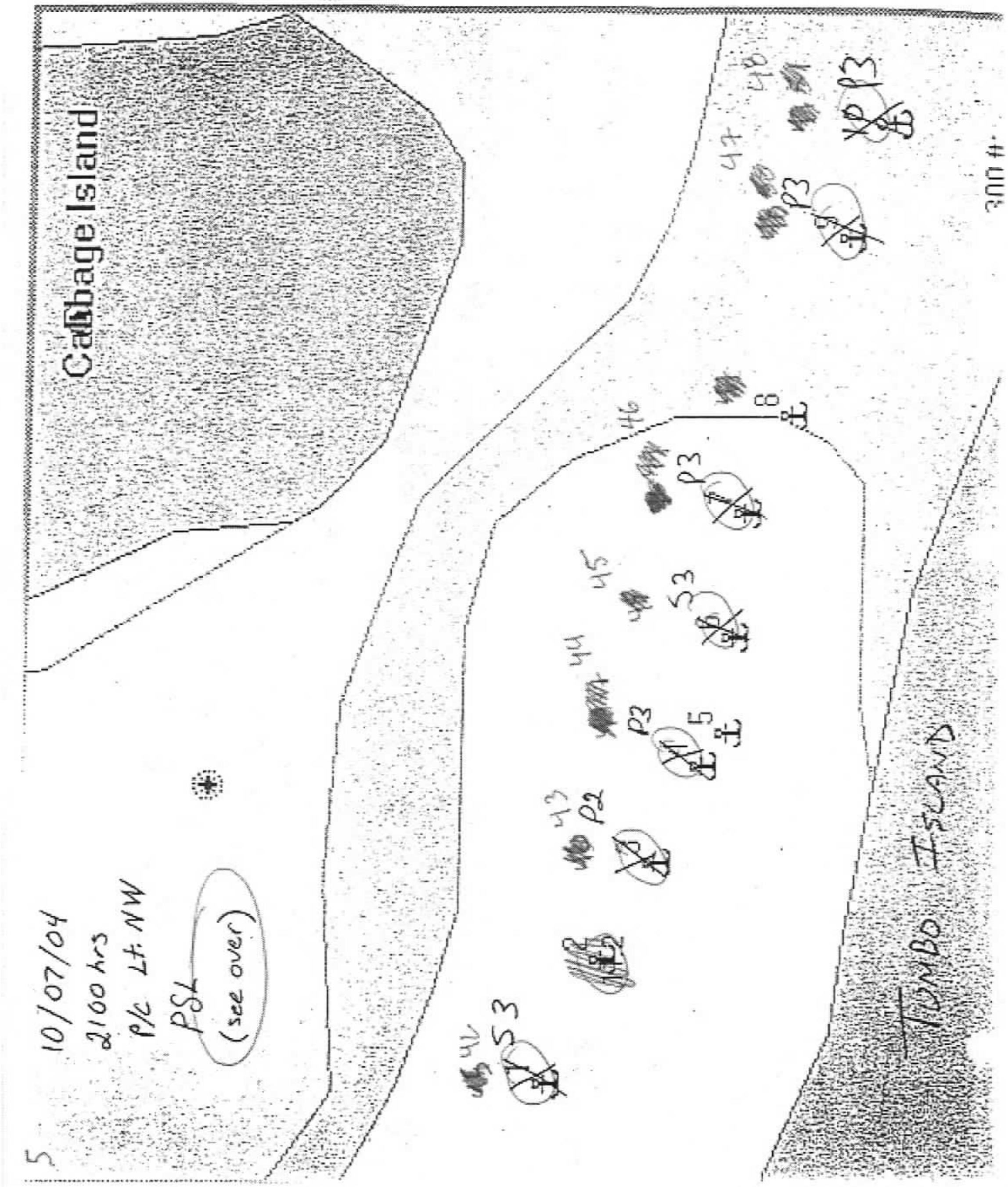
Appendix One: Sample data sheets showing boat usage at each study site returned by facility operators



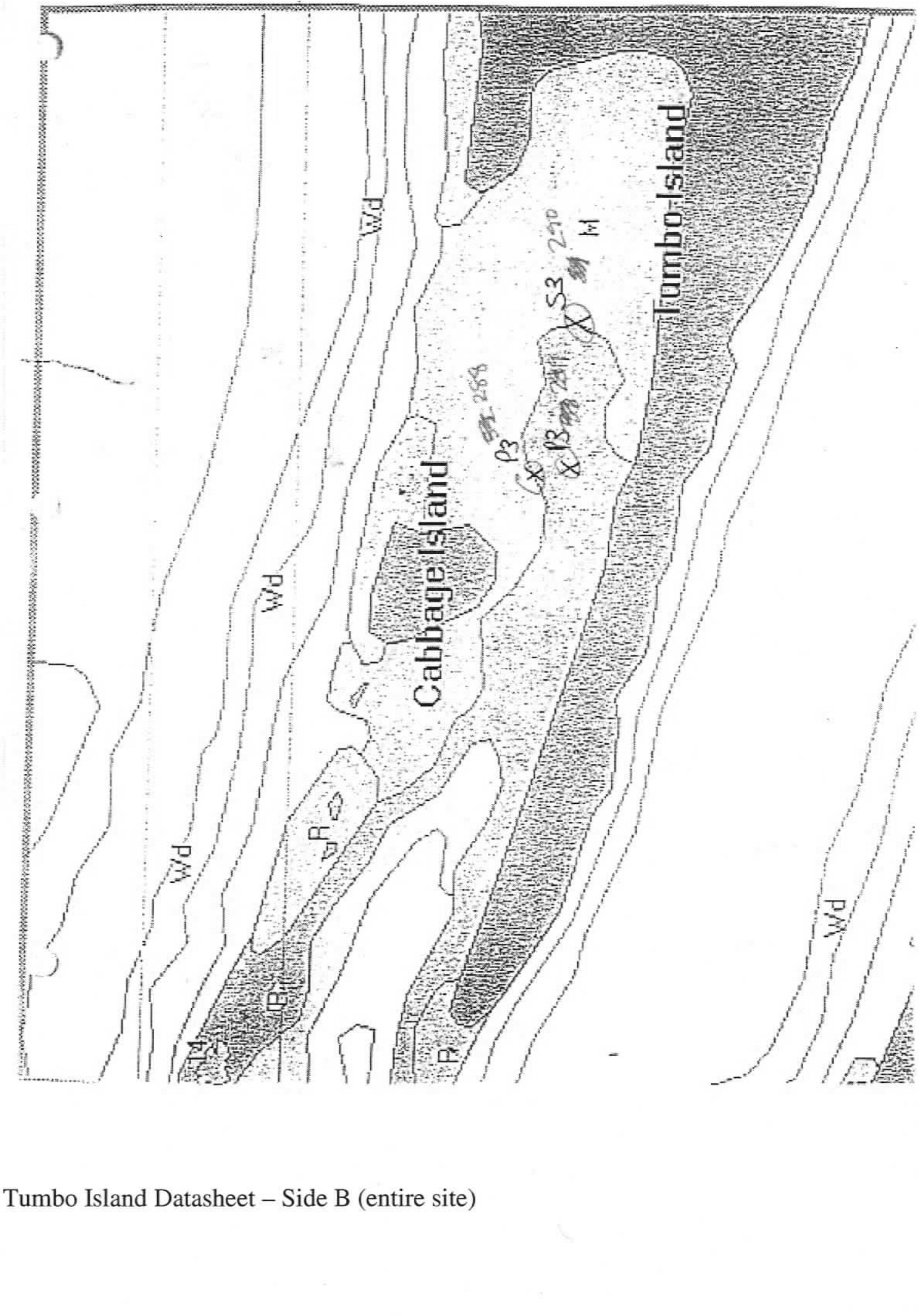
Date July 8 Time 9 00 General weather overcast Observer MIKE W

Please mark the location of all anchored or moored vessels on the map with an X, using the mooring buoys as reference points. Beside the X, Please mark the type and size of vessel according to the codes: Sailboats: S1 <30ft: S2 30-40ft: S3 >40ft Power boats: P1: <20; P2 20-30ft; P3: > 30ft

Sidney Spit Datasheet



Tumbo Island Datasheet – Side A (mooring area)



Tumbo Island Datasheet – Side B (entire site)

Appendix Two: Organisms representative of opportunistic, equilibrium, and neutral communities (after Pearson and Rosenberg, 1978)

Opportunists:	Nematoda Nereidae Glyceroidea Spionidae Nemerteans Capitellidae
Equilibrium:	Maldanidae Mytilidae Cardiidae Veneridae Tellinidae Lucinidae Ophiuroidea
Neutral:	Pectinariidae Cirratulidae Aphrodititoidea Syllidae Opheliidae Spiroribidae Lacunidae Littorinidae Trochidae Doridoidea Columbellidae Naticidae Acteonidae Lottiidae Cerithiidae Pyramidellidae Gammeridea Caprellidae Ostracoda Cumacea Mysidacea Isopoda Tanaideae Crab Cirripedia Copepoda Strongylocentrotus Dendraster

Appendix Three: Aggregation table for all observed taxa

(from Brands 1989-2006)

Family	Order	Class	Phylum	Superphylum
Nematoda			Nematoda	Aschelminthes
Pectinariidae	Terebellida	Polychaeta	Annelida	Eutrochozoa
Neridae	Phyllodocida	Polychaeta	Annelida	Eutrochozoa
Glyceroidea	Phyllodocida	Polychaeta	Annelida	Eutrochozoa
Spionidae	Spionida	Polychaeta	Annelida	Eutrochozoa
Cirratulidae	Terebellida	Polychaeta	Annelida	Eutrochozoa
Aphrodititoidea	Phyllodocida	Polychaeta	Annelida	Eutrochozoa
Syllidae	Phyllodocida	Polychaeta	Annelida	Eutrochozoa
Opheliidae	Scolecida	Polychaeta	Annelida	Eutrochozoa
Spiroribidae	Sabellida	Polychaeta	Annelida	Eutrochozoa
Maldanidae	Scolecida	Polychaeta	Annelida	Eutrochozoa
Nemerteans			Nemertea	Eutrochozoa
Capitellidae	Scolecida	Oligochaeta	Annelida	Eutrochozoa
Mytilidae	Mytiloidea	Bivalvia	Mollusca	Eutrochozoa
Cardiidae	Veneroida	Bivalvia	Mollusca	Eutrochozoa
Veneridae	Veneroida	Bivalvia	Mollusca	Eutrochozoa
Tellinidae	Veneroida	Bivalvia	Mollusca	Eutrochozoa
Lucinidae	Veneroida	Bivalvia	Mollusca	Eutrochozoa
Lacunidae	Mesogastropoda	Gastropoda	Mollusca	Eutrochozoa
Littorinidae	Mesogastropoda	Gastropoda	Mollusca	Eutrochozoa
Trochidae	Archaeogastropoda	Gastropoda	Mollusca	Eutrochozoa
Doridoidea	Nudibranchia	Gastropoda	Mollusca	Eutrochozoa
Columbellidae	Neogastropoda	Gastropoda	Mollusca	Eutrochozoa
Naticidae	Mesogastropoda	Gastropoda	Mollusca	Eutrochozoa
Acteonidae	Cephalispidea	Gastropoda	Mollusca	Eutrochozoa
Lottiidae	Archaeogastropoda	Gastropoda	Mollusca	Eutrochozoa
Cerithiidae	Mesogastropoda	Gastropoda	Mollusca	Eutrochozoa
Pyramidellidae	Mesogastropoda	Gastropoda	Mollusca	Eutrochozoa
Gammeridea	Amphipoda	Malacostraca	Arthropoda	Panarthropoda
Caprelliadea	Amphipoda	Malacostraca	Arthropoda	Panarthropoda
Ostracoda			Ostracoda	Panarthropoda
Cumacea	Cumacea	Malacostraca	Arthropoda	Panarthropoda
Mysidacea	Mysida	Malacostraca	Arthropoda	Panarthropoda
Isopoda	Isopoda	Malacostraca	Arthropoda	Panarthropoda
Tanaideae	Tanaidacea	Malacostraca	Arthropoda	Panarthropoda
Crab	Decapoda	Malacostraca	Arthropoda	Panarthropoda
Cirripedia		Maxillopoda	Arthropoda	Panarthropoda
Copepoda		Maxillopoda	Arthropoda	Panarthropoda
Ophiuroidea		Stelleroidea	Echinodermata	Coelomopora
Strongylocentrotus	Echinoida	Echinoidea	Echinodermata	Coelomopora
Dendraster	Clypeastroidea	Echinoidea	Echinodermata	Coelomopora

Family	Infrakingdom	Branch	Subkingdom	Kingdom
Nematoda	Ecdysozoa	Protostomia	Bilateria	Animalia
Pectinariidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Neridae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Glyceroidea	Lophotrochozoa	Protostomia	Bilateria	Animalia
Spionidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Cirratulidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Aphrodititoidea	Lophotrochozoa	Protostomia	Bilateria	Animalia
Syllidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Opheliidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Spiroribidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Maldanidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Nemerteans	Lophotrochozoa	Protostomia	Bilateria	Animalia
Capitellidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Mytilidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Cardiidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Veneridae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Tellinidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Lucinidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Lacunidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Littorinidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Trochidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Doridoidea	Lophotrochozoa	Protostomia	Bilateria	Animalia
Columbellidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Naticidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Acteonidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Lottiidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Cerithiidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Pyramidellidae	Lophotrochozoa	Protostomia	Bilateria	Animalia
Gammeridea	Ecdysozoa	Protostomia	Bilateria	Animalia
Caprelliadea	Ecdysozoa	Protostomia	Bilateria	Animalia
Ostracoda	Ecdysozoa	Protostomia	Bilateria	Animalia
Cumacea	Ecdysozoa	Protostomia	Bilateria	Animalia
Mysidacea	Ecdysozoa	Protostomia	Bilateria	Animalia
Isopoda	Ecdysozoa	Protostomia	Bilateria	Animalia
Tanaideae	Ecdysozoa	Protostomia	Bilateria	Animalia
Crab	Ecdysozoa	Protostomia	Bilateria	Animalia
Cirripedia	Ecdysozoa	Protostomia	Bilateria	Animalia
Copepoda	Ecdysozoa	Protostomia	Bilateria	Animalia
Ophiuroidea		Deuterostomia	Bilateria	Animalia
Strongylocentrotus		Deuterostomia	Bilateria	Animalia
Dendraster		Deuterostomia	Bilateria	Animalia